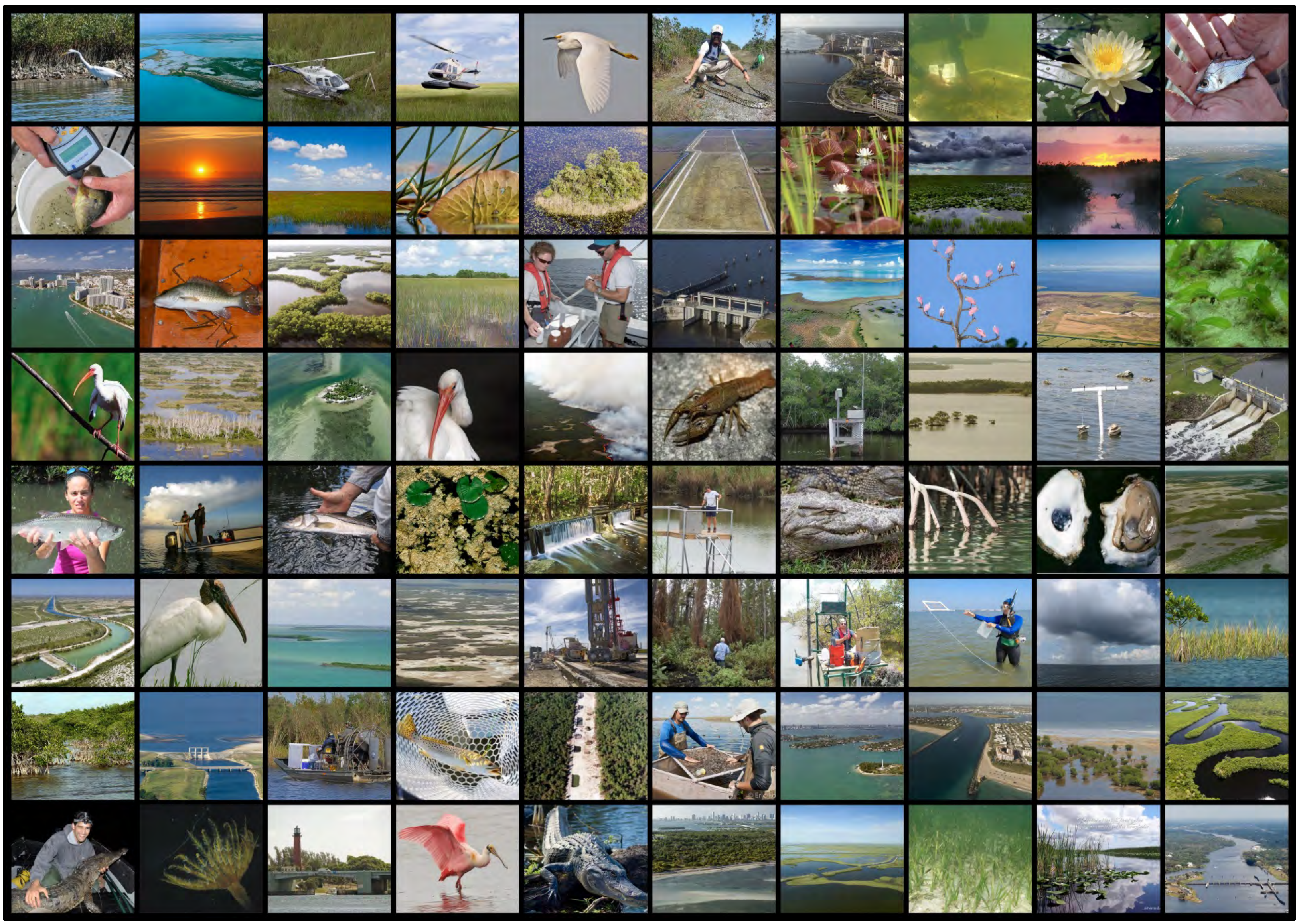
COMPREHENSIVE EVERGLADES RESTORATION PLAN

2014 System Status Report

AUGUST 2014

RESTORATION COORDINATION AND VERIFICATION



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EXECUTIVE SUMMARY

BACKGROUND

The 2014 System Status Report (SSR) evaluates current monitoring data to determine if the goals and objectives of the Comprehensive Everglades Restoration Plan (CERP) are being met. The report incorporates data collected by the Restoration Coordination and Verification (RECOVER) Monitoring and Assessment Plan (MAP) program for CERP, data from CERP projects and data provided by RECOVER partners. The information provided builds on previous reports produced in 2006, 2007, 2009 and 2012.

The goal of the MAP program is to document status and trends of the essential and defining attributes of the south Florida ecosystem. These RECOVER monitoring data are used to assess the status and trends in hydrology affected by restoration project implementation and system water management operations, as well as in ecological parameters (e.g., wading birds) that respond to changes in the quantity, quality, timing and distribution of water. This information is measured against pre-CERP reference conditions and is used to help determine if the goals and objectives of CERP are being met.

This comprehensive understanding of the system enables the successful use of adaptive management principles to track and guide restoration activities. Although CERP implementation has been slower than originally envisioned, progress in project construction and operation has begun in some areas, especially in the Southern Coastal Systems region.

INTENDED USE AND ORGANIZATION

A robust systemwide monitoring and assessment program like the MAP is also a key component of the CERP Adaptive Management Program. Adaptive management is a structured management approach that links science to decision making in order to improve the probability of restoration success. Scientific information collected by the MAP and reported in the SSR is fed into the decision-making process, allowing managers and decision makers to use the best available science during CERP implementation. Information about the application of adaptive management to CERP is detailed in the CERP Adaptive Management Integration Guide.

Key Findings from the report will be used to assist decision-makers on the timing of planning and implementation of certain CERP features. These data also inform the scientific community in south Florida and provides a basis for such planning efforts as the Central Everglades Planning Project. This project was one of a number of accelerated planning efforts led by the United States Army Corps of Engineers. This 18-month process relied heavily on the performance measures and ecological tools that were developed over the past several years under RECOVER's MAP program.

The 2014 SSR also provides input into the 2015 Report to Congress, required by the Water Resources Development Act of 2000. Produced every five years, the intent of the Report to Congress is to inform the highest levels of the United States government on the progress made toward the goals and objectives of CERP.

The SSR is divided into four geographical regions: Lake Okeechobee, Northern Estuaries, Greater Everglades, and Southern Coastal Systems. This organization helps facilitate the monitoring and analysis, but in no way is meant to imply that the Everglades ecosystem is a series of discrete, unconnected habitats. On the contrary, it is a very complex, vast and inter-connected system of lakes, estuaries, freshwater marshes and forests that needs to be considered as a whole.

Systemwide analysis continues to be a challenge, but progress has been made to this end. Examples of this can be found in the treatment of data in the southern Everglades and estuaries.

The 2009 report introduced the concept of a “webument.” This web-based document framework was the basis for the 2014 report. The format allows for the display of information at multiple levels of detail and complexity. The reader is free to navigate using user-friendly buttons and pictures to explore the information. The “Key Findings” will have videos, pictures and high level summaries, whereas more detailed data reporting can be found for any given ecosystem restoration attribute or region.

2014 SYSTEM STATUS REPORT HIGHLIGHTS

Systemwide Hydrology Context

South Florida was mostly drier than normal from Water Year (WY) 2009 to WY 2013 due to below-average rainfall years (WY 2009, WY 2011 and WY 2012) compared to the historical average (1900–1995). This drier climate had dramatic effects on the hydrology in the currently managed system because it descreased flows into and out of each basin and reduced water levels. These hydrologic patterns had different effects in different regions of the system:

- Reduced inflows to Lake Okeechobee and below average rainfall, coupled with the Lake Okeechobee Regulation Schedule, helped reduce high stage events and kept water levels within desired stage ranges for ecology. This also meant less discharge to the Northern Estuaries from the lake, which benefited some parts of estuarine ecology.
- However, back-to-back dry years decreased basin flow to the Northern Estuaries resulting in high salinities for long periods of time that impacted submerged aquatic vegetation and oysters in the Caloosahatchee River and St. Lucie estuaries.
- South of Lake Okeechobee, reduced flows and rainfall decreased marsh water levels (surface and groundwater) in the Water Conservation Areas (WCAs) and caused severe dry downs. This resulted in shorter hydroperiods, and higher salinities in estuaries, which negatively affected ecosystem indicators in the Greater Everglades (too dry in northern parts of WCAs and Everglades National Park, and too wet in southern parts of WCAs) and Southern Coastal Systems (salinities too high overall).

The following bulleted sections summarize the effects of hydrology on ecosystem indicators in each region.

Lake Okeechobee

- Ecology improved between 2009–2012, compared to 2004–2008 due to less rain and flow entering Lake Okeechobee and changes to the Lake Okeechobee Regulation Schedule to reduce high stages.
  - Nutrient concentrations reduced 20% or more.
  - Nearshore submerged aquatic vegetation increased by 12,200 acres.
  - Increased healthy periphyton (algae) communities.
- **Recommendation:** Storage projects, such as dispersed water management envisioned in the Lake Okeechobee Watershed Project, completion of the aquifer storage and recovery technical report, Kissimmee River Restoration (construction/operations), and optimization of the lake regulation schedule (operations) will all help to meet CERP water quantity, timing, and distribution goals during wet years and realize ecological restoration for the lake and Northern Estuaries ecological restoration.

Northern Estuaries

- Although high flow discharge events from Lake Okeechobee have contributed to the estuarine habitat damage in the Northern Estuaries in WY 2004 and WY 2005, new data indicate that supplemental low flows to the St. Lucie Estuary during extremely dry years, especially when they occur back-to-back, may be needed to maintain healthy oyster populations in the middle estuary.
- **Recommendations:** Indian River Lagoon - South (under construction) development of operational plan development should incorporate low flow requirements to support estuary restoration. The C-43 West Basin Storage Reservoir Project (awaiting authorization) will greatly improve low flows to the Caloosahatchee River Estuary once constructed and operational.

Greater Everglades

- Long-term trends of fish biomass have declined over the past three decades in portions of WCA 3A, WCA 3B, and Everglades National Park.
- Periphyton communities continue to indicate altered hydrology and nutrient conditions in WCA 3A and the eastern edge of Everglades National Park.
- Greater than 50% loss of tree islands in Shark River Slough of Everglades National Park between 1954 and 2004, similar to WCA 3 trends.

- **Recommendation:** Approval, authorization, and implementation of the Central Everglades Planning Project will supply additional water to dry areas of the system (e.g. Northern WCA 3A, WCA 3B, Shark River Slough and Florida Bay), reducing severe fires and improving hydrology. This will lead to improvements in periphyton, vegetation, wildlife populations including small fish and invertebrates, wading birds and alligators, and ridge and slough landscapes.

Southern Coastal Systems

- Florida Bay salinity conditions moved further from restoration targets over the past 4 years compared to the 2009 SSR.
- Large patches of dead seagrass were observed in western and central Florida Bay.
- **Recommendation:** Central Everglades Planning Project implementation and operations that consider the need for water flow into the dry season will improve salinity and ecological (submerged aquatic vegetation, invertebrates, and fish) conditions in Florida Bay and avoid harm in Biscayne Bay.

Restoration Improvements

There were demonstrated restoration improvements in certain ecosystem indicators as a result of restoration projects being constructed and operational over the past 4 to 5 years:

- **Biscayne Bay Coastal Wetlands Project:** Hydrology improved in the operational Deering Estate Flow-way portion of the Biscayne Bay Coastal Wetlands expedited project.
- **Picayune Strand Restoration Project:** Picayune Strand showed higher water levels near the filled Prairie Canal (1 to 2 feet higher) and vegetation is starting to show signs of improvement, moving closer to reference conditions.
- **Central and Southern Florida Project Operations:** Roseate spoonbill nesting improved, most likely due to favorable climatic conditions and better real-time environmental coordination with water management operational decisions.
- **C-111 South Dade:** Hydroperiods were 50 days longer (on an annual average basis) along the eastern edge of Everglades National Park as a result of the C-111 South Dade Project and extended rainfall.
- **Cape Sable Canal Plugging:** Crocodile nesting and population trends increased due to Cape Sable plug restoration projects (over past two decades).

In summary, continued trends (2009–2013) in altered hydrology and degraded ecology across the system necessitates the need for CERP restoration projects to move forward, while continuing to provide flood protection and water supply for more than 7.9 million people in south Florida. Initial restoration successes from implementing CERP and non-CERP projects (C-111 South Dade, Biscayne Bay Coastal Wetlands Projects, Picayune Strand Restoration, and Central and Southern Florida Project Operations) demonstrate the value of incremental restoration actions. Authorizing, constructing, and operating more CERP restoration projects (Indian River Lagoon - South, Lake Okeechobee watershed storage, C-43 West Basin Storage Reservoir, aquifer storage and recovery, Picayune Strand Restortaion Project, Broward County Water Preserve Areas, and Central Everglades Planning Project) will allow future SSRs to demonstrate incremental achievement of systemwide restoration goals and objectives.

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TABLE OF CONTENTS

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ACRONYMS, ABBREVIATIONS AND UNITS OF MEASUREMENT

°C	degrees Celsius
µg/L	microgram per liter
µm	micrometer
1mil	1 million acre-feet of storage north of Lake Okeechobee condition
2mil	2 million acre-feet of storage north of Lake Okeechobee condition
2008 LORS	2008 Lake Okeechobee Regulation Schedule
500k	500,000 acre-feet of storage north of Lake Okeechobee condition
ac-ft	acre-feet
AMBI	AZTI's Marine Biotic Index
ANOVA	analysis of variance
AWD	average water depth
AWT	average water temperature
BBCA scale	Braun-Blanquette cover/abundance scale
BBCW Project	Biscayne Bay Coastal Wetlands Project
BCNP	Big Cypress National Preserve
BCWPA	Broward County Water Preserve Areas
C&SF Project	Central and Southern Florida Project
C-111 SCWP	C-111 Spreader Canal Western Project
CEM	conceptual ecological model
CEPP	Central Everglades Planning Project
CERP	Comprehensive Everglades Restoration Plan
cfs	cubic feet per second
CGM	CERP Guidance Memorandum
Chla	chlorophyll *a*
CISMA	cooperative invasive species management area
cm	centimeter
c/n	chicks per nest
CPTWP	construction phasing, transfer and warranty plan
CRE	Caloosahatchee River Estuary

CY	Calendar Year
DASM	digital aerial sketch mapping
DBHYDRO	SFWMD's hydrometerologic, water quality and hydrogeologic data retrieval system
DIN	dissolved inorganic nitrogen
DPM	Decomp Physical Model
ECB	2008 LORS baseline condition used to compare storage options north of Lake Okeechobee
ECISMA	Everglades Cooperature Invasive Species Management Area
EDEN	Everglades Depth Estimation Network
EDDMaps	Early Detection and Distribution Mapping System
EDRR	early detection rapid response
EIRAMP	Everglades Invasive Reptile and Amphibian Monitoring Program
EIS	environmental impact statement
ELVeS	Everglades Landscape Vegetation Succession Model
ENP	Everglades National Park
ERTP	Everglades Restoration Transition Plan
ET	evapotranspiration
EWD	Experimental Water Deliveries
FDEP	Florida Department of Environmental Protection
FHAP	Florida Habitat Assessment Program
FIAN	Fish and Invertebrate Assessment Network
FLEPPC	Florida Exotic Plant Pest Council
FWC	Florida Fish and Wildlife Conservation Commission
FY	Fiscal Year (begins October 1 and ends September 30 of the following year)
g/m^2	grams per square meter
GE	Greater Everglades (region)
GIS	geographic information system
GPS	global positioning system
ha	hectare

IBBEAM	Integrated Biscayne Bay Ecological Assessment and Monitoring Project
IOP	Interim Operational Plan
ISOP	Interim Structural and Operational Plan
kg	kilogram
km	kilometer
km^2	square kilometer
LIDAR	light detection and ranging
LO	Lake Okeechobee (region)
LRD	Loxahatchee River District
LRE	Loxahatchee River Estuary
LSU	landscape sampling units
LWL	Lake Worth Lagoon
m	meter
m^2	square meter
MAP	Monitoring and Assessment Plan
MFL	mimimum flows and levels
mg/L	milligram per liter
Miami-Dade DERM	Miami-Dade County Department of Environmental Resources Management
mm	millimeters
MMN	Marine Monitoring Network
N	nitrogen
NAVD88	North American Vertical Datum of 1988
NE	Northern Estuaries (region)
NGGE	Northern Golden Gate Estates
NGVD	National Geodetic Vertical Datum of 1929
NGVD29	National Geodetic Vertical Datum of 1929
NNC	numeric nutrient criteria
NOAA	National Oceanic and Atmospheric Administration
NPS	National Park Service

NSM	Natural Systems Model
P	phosphorus
PIR	project implementation report
POR	period of record
ppb	parts per billion
ppt	parts per thousand
PSRP	Picayune Strand Restoration Project
psu	practical salinity unit (equivalent to parts per thousand [ppt])
PSU	Primary Sampling Unit
RBNERR	Rookery Bay National Estuarine Research Reserve
RECOVER	Restoration Coordination and Verification
RESOPS model	Reservoir Sizing and Operations Screening model
S_y	specific yield difference
SAV	submerged aquatic vegetation
SCA	Stochastic Cellular Automata model
SCS	Southern Coastal Systems (region)
SD	standard deviation
SE	standard error
SEACOM	Florida Bay Seagrass Community Model
SFWMD	South Florida Water Management District
SFWMM	South Florida Water Management Model
SGGE	Southern Golden Gate Estates
shoots/m^2	shoots per square meter
SIRL	Southern Indian River Lagoon
SLE	St. Lucie Estuary
SRP	soluable reactive phosphorus
SRS	Shark River Slough
SRSI	salinity regime suitability index
SSR	System Status Report
TTI	Ten Thousand Islands

TN	total nitrogen
TP	total phosphorus
USACE	United States Army Corps of Engineers
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WAAS	wide area augmentation system
WADEM	Wader Distribution Evaluation Modeling
WAI	wetland affinity index
WCA	Water Conservation Area
WMA	wildlife management area
WSE	Water Supply and Environmental Lake Okeechobee Operations Schedule
WY	Water Year (Begins May 1 and ends April 30 of the following year)

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CHAPTER 1

INTRODUCTION

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TABLE OF CONTENTS

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CHAPTER 1 INTRODUCTION

EVERGLADES RESTORATION BACKGROUND

A natural treasure is in trouble. The Everglades of today (**Figure 1-1**) are not the same place that Mrs. Marjory Stoneman Douglas wrote about in her 1947 book, River of Grass. The Florida Everglades were once a vibrant, free-flowing river of grass that provided clean water from Lake Okeechobee all the way to Florida Bay. It was a vital haven for storks, alligators, panthers, and other wildlife. Today this extraordinary ecosystem–unlike any other in the world–is dying.

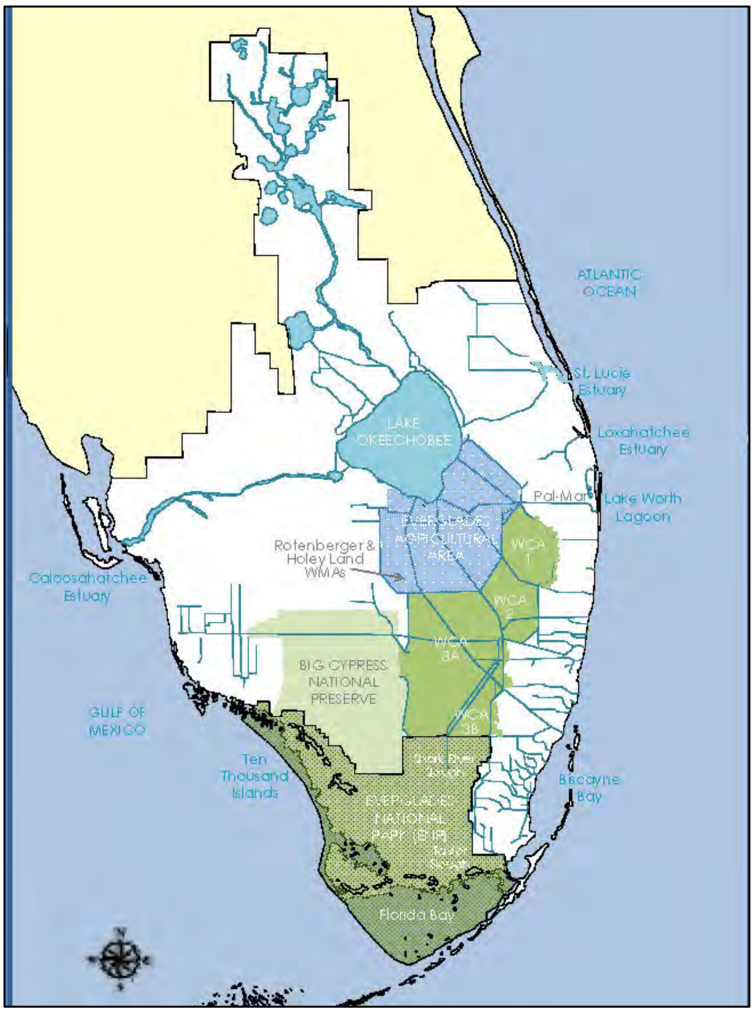

Figure 1-1. The south Florida Everglades ecosystem.

The south Florida Everglades ecosystem stretches south from Orlando through the Chain of Lakes, the Kissimmee Valley, Lake Okeechobee, the remaining Everglades, and the waters of Florida Bay and the coral reefs. This south Florida ecosystem is much larger than what most people see when they visit the "Everglades", which is usually just Everglades National Park.

Over the past sixty years, people have encroached upon the ecosystem that once was the domain of panthers, alligators, and flocks of birds so vast that they would darken the sky. With the arrival of people came the desire to manage the water, to tame the free flowing River of Grass. The Central and Southern Florida Flood Control Project was authorized 50 years ago to provide flood protection and fresh water to south Florida, an area covering 18,000 square miles. This project accomplished its intended purpose and allowed people to more easily live on the land by guarding communities against flooding and ensuring an adequate water supply for drinking and irrigation. However, it did so at a tremendous ecological cost to the Everglades by diverting approximately 70 percent of the natural water flows that are the essence of the natural system. While the population of people has risen from 500,000 in the 1950s to more than 7.9 million today, the numbers of native birds and other wildlife have dwindled and some have vanished. The size of the Everglades has been reduced by half, and the surviving remnants suffer from a severe shortage of clean, reliable water.

Water is the lifeblood of the Everglades ecosystem. Compared to the historic Everglades, water no longer follows the timing and duration of the natural ecosystem, nor can it move freely throughout the entire system with canals and pump stations controlling the water. Lake Okeechobee, the second largest freshwater lake in the United States, is where much of the surface water in south Florida is held. It is home to many fish and wildlife species, and is an important recreational area for hiking, biking, fishing, hunting, and birding. Its health is also declining and severely threatened. Clean water is becoming increasingly unavailable to the estuaries and bays that are critical nurseries and homes to many fish and wildlife. This trend is the same for water available to people, with water shortages and restrictions now a way of life in some parts of south Florida.

RESTORING A DYING ECOSYSTEM

In an historic partnership, the United States Army Corps of Engineers (USACE) and the South Florida Water Management District worked from 1993 to 1999 with more than 100 ecologists, hydrologists, engineers, and other professionals from more than 30 federal, state, tribal, and local agencies to take a systemwide look at water. A Comprehensive Everglades Restoration Plan (CERP) (http://www.evergladesplan.org/about/about_cerp_brief.aspx) was developed to restore and preserve south Florida's natural ecosystem while enhancing water supplies and maintaining flood protection. CERP called for a series of water system improvements over more than 35 years, with a now estimated total cost of $13.5 billion. This plan would build off several South Florida Everglades Ecosystem Restoration Foundation projects that were authorized prior to CERP in the 1990s. CERP was authorized by the United States Congress, as part of the 2000 Water Resources Development Act (http://www.evergladesplan.org/wrda2000/wrda.aspx) with its fundamental goal to capture most of the fresh water that currently flows unused to the ocean and gulf and deliver it when and where it is needed most throughout south Florida. Eighty percent of this "new" (captured) water is to be devoted to environmental restoration. The remaining 20 percent will benefit cities and agriculture, enhancing water supplies and supporting a strong, sustainable economy for south Florida well into the twenty-first century. In short, CERP provides the necessary road map for improving the quantity, quality, timing, and distribution of the water that is vital to the health of the Everglades and the people of south Florida.

CERP GOALS AND OBJECTIVES

CERP will enhance ecological values by increasing the total spatial extent of natural areas, improving habitat and functional quality, and improving native plant and animal species abundance and diversity. CERP will promote economic and social goals by increasing availability of fresh water (agricultural, municipal, and industrial), reducing flood damages (agricultural and urban), providing recreational and navigation opportunities, and protecting cultural and archeological resources and values. CERP projects were envisioned to work with pre-CERP foundation projects. The initial list of projects and benefits developed for CERP fall into several categories. The specifics of each category from the original CERP are included below with an understanding that much of what was originally envisioned has changed 14 years later:

Specifically, CERP will do the following:

- **Improve the health of over 2.4 million acres of the south Florida ecosystem, including Everglades National Park**
- **Improve the health of Lake Okeechobee**
- **Significantly reduce damaging freshwater releases to the estuaries**
- **Improve water deliveries to Florida and Biscayne bays**
- **Improve water quality**
- **Enhance water supply and maintain flood protection**

- **Surface Water Storage Reservoirs –** 15 projects would store 1.5 million acre-feet of water to prevent it being sent to sea and be used for natural system and water supply purposes.

- **Water Preserve Areas (19 areas) –** 36,000 acres to treat urban runoff, store water, reduce seepage, and improve existing wetland areas.

- **Manage Lake Okeechobee as an Ecological Resource (Operations Plan Change) –** lake regulation schedule would be changed to reduce extreme high and low levels that harm ecology.

- **Improve Water Deliveries to Estuaries (Operations Plan Change)** – reduce high flows to estuaries during excess rainwater periods, and during times of low rainfall, stored water can be retrieved to sustain the estuaries.

- **Underground Water Storage (Aquifer Storage and Recovery) –** 300 five-million gallons per day underground storage that can be used during dry times for multiple objectives.

- **Stormwater Treatment Areas –** Up to 30,000 acres of stormwater treatment areas to treat urban and agricultural runoff before discharge into natural areas, in addition to 40,000 acres to treat developed by the state to treat Everglades Agricultural Area water.

- **Improve Timing and Delivery of Water to Everglades –** adjust rainfall-driven operational plan to improve the timing of water sent to Water Conservation Areas and Everglades National Park (ENP).

- **Removing Barriers to Sheetflow –** Removal of up to 500 miles of project canals and levees to reestablish natural sheetflow of water through the Everglades.

- **Storing Water in Quarries –** Limestone quarries would be used to store water for Florida Bay, the Everglades, and Miami-Dade County residents.

- **Wastewater Reuse –** Two advanced wastewater treatment plans would be developed to provide 220 million gallons of water per day to Miami-Dade county, Biscayne Bay wetlands, and northeastern Shark River Slough in the Everglades.

- **Improve Water Deliveries to Biscayne Bay –** Overall plan would protect and restore Biscayne Bay coastal wetlands and treat stormwater runoff before it enters the bay.

- **Improve Freshwater Flows to Florida Bay –** Overall Plan would improve water deliveries to Shark River Slough, Taylor Slough, and wetlands east of ENP to send to Florida Bay.

Figure 1-2 shows the general location of these projects.

Figure 1-2. Map showing the general location of types of CERP projects.

CERP projects build off of the following foundation projects, many of which have been constructed partially or in whole:

- **Kissimmee River Restoration –** restores 40 square miles of river-floodplain ecosystem.

- **C-51 Stormwater Treatment Area –** Clean and send water into Water Conservation Area (WCA) 1, which is within the Arthur R. Marshall Loxahatchee National Wildlife Refuge.

- **C-111 South Dade –** Detention areas to reduce seepage out of ENP to improve natural hydrologic conditions in Taylor Slough and eastern portions of ENP.

- **Modified Water Deliveries –** One-mile bridge has been completed to restore more natural flows to ENP.

- **Seminole Big Cypress –** to provide benefits to Seminole Tribe of Florida lands.

RECOVER MONITORING AND ASSESSMENT PROGRAM AND ADAPTIVE MANAGEMENT

Congress recognized that uncertainty existed in implementing CERP and achieving the plan's goals and objectives. CERP was authorized under the expectation that adaptive management principles would be applied to update its implementation based on knowledge gained throughout the process. Adaptive management is a structured management approach that links science to decision making in order to improve the probability of restorations success. Restoration Coordination and Verification (RECOVER) (http://www.evergladesplan.org/docs/fs_recover_jan_2012.pdf), a systemwide interdisciplinary science team made up of representatives from 12 agencies and tribes, developed the Monitoring and Assessment Plan (MAP) (http://www.evergladesplan.org/pm/recover/recover_map.aspx) to provide the framework for what monitoring was needed and how it would be assessed and reported to support adaptive management implementation. The goal of the MAP program is to document status and trends of the essential and defining attributes of the south Florida ecosystem. These RECOVER monitoring data are used to assess the status and trends in hydrology affected by restoration project implementation and system water management operations, as well as in ecological parameters (e.g., wading birds) that respond to changes in the quantity, quality, timing, and distribution of water. This information is measured against pre-CERP reference conditions and is used to help determine whether the goals and objectives of CERP are being met. A robust systemwide monitoring and assessment program like the MAP is also a key component of the CERP Adaptive Management Program (http://www.evergladesplan.org/pm/recover/recover_map.aspx).

2014 SYSTEM STATUS REPORT PURPOSE AND BACKGROUND

Key findings from this 2014 System Status Report (SSR) will be used to assist decision-makers on the timing of planning and implementation of CERP features. These data also inform the scientific community in south Florida and provide a basis for such planning efforts as the Central Everglades Planning Project (CEPP) (http://www.evergladesplan.org/pm/projects/proj_51_cepp.aspx). This project was one of a number of accelerated pilot planning efforts led by the USACE. This 18-month process relied heavily on the performance measures and ecological tools that were developed over the past several years under RECOVER's MAP program. The 2014 SSR also provides input into the 2015 Report to Congress, required by the Water Resources Development Act of 2000. Produced every five years, the intent of the Report to Congress is to inform the highest levels of the United States government on the progress made toward the goals and objectives of CERP.

The SSR is divided into four geographical regions: Lake Okeechobee, Northern Estuaries, Greater Everglades, and Southern Coastal Systems (**Figure 1-3**). This organization helps facilitate the monitoring and analysis, but in no way is meant to imply that the south Florida ecosystem is a series of discrete, unconnected habitats. On the contrary, it is a very complex, vast and inter-connected system of lakes, estuaries, freshwater marshes, and forests that needs to be considered as a whole. In addition, status of ecosystem indicators are linked to current and future restoration project and operations that is likely affecting or will likely affect change in status, whenever possible. Where projects have been constructed and are operational, restoration results are provided in the SSR. Systemwide analysis continues to be a key responsibility of RECOVER but is often a challenge to conduct when data are limited. Examples of systemwide data synthesis can be found in the southern Everglades and coastal systems sections.

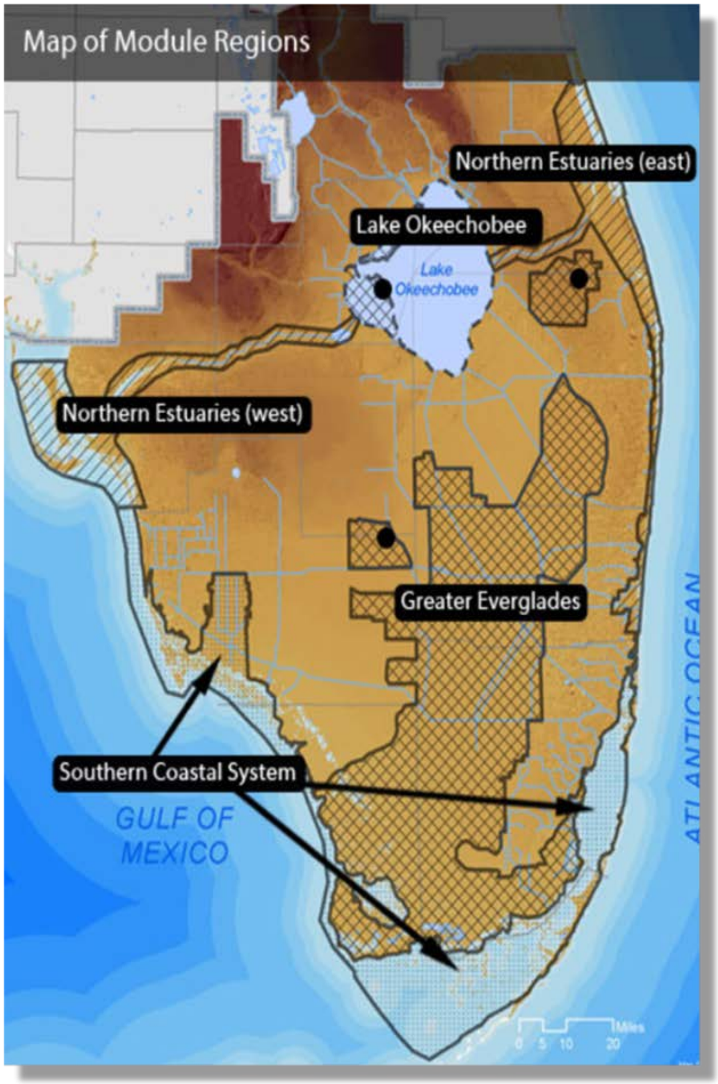

Figure 1-3. Module regions used in the MAP and SSR documents.

The 2009 SSR introduced the concept of a "webument." This web-based document framework was the basis for the 2014 SSR. The format allows for the display of information at multiple levels of detail and complexity. The reader is free to navigate online with user-friendly buttons and pictures to explore the information. The Key Findings web page has videos, pictures, and high-level summaries, whereas the Systemwide Science and each geographical region pages has more detailed data reporting and conclusions.

CHAPTER 2

KEY FINDINGS

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TABLE OF CONTENTS

CHAPTER 2 KEY FINDINGS

INTRODUCTION

The *2014 System Status Report* (SSR) provides an in-depth assessment of the monitoring data provided by the Restoration Coordination and Verification (RECOVER) Monitoring and Assessment Plan (MAP) (RECOVER 2009) in conjunction with historical data and data from non-MAP sources. The monitoring data on the status and trends of physical (e.g., hydrology), chemical (e.g., nutrients), and biological (e.g., wading birds) parameters are assessed in this report to establish pre-Comprehensive Everglades Restoration Plan (CERP) reference conditions and ultimately to determine whether the CERP goals and objectives of improved water quantity, quality, timing, and distribution are being met. In addition, monitoring is also assessed for progress towards CERP goals and objectives related to ecological flora, fauna, and landscape responses to hydrology. This is the fifth such report required by CERP programmatic regulations to support adaptive management feedback to improve CERP planning, design, and implementation (RECOVER 2012). These key findings are intended to convey key scientific information to water managers, budget directors, decision-makers, and the public about the status of the Everglades ecosystem to support restoration and water management decisions. Key findings are designed to highlight significant status and trend changes, demonstrate restoration project or operations effects, and to highlight new knowledge gained and its importance to support and/or update restoration planning and implementation efforts. The key findings are arranged in both a systemwide section that deals with some of the key big picture drivers and stressors affecting the Everglades ecosystem, as well as in region-specific sections that explain specific ecological indicator trends. The key findings in this chapter summarize in-depth science that can be found in the main SSR document, annual monitoring reports, and other published reports.

Setting the Stage

The following paragraphs summarize hydrology and other natural drivers of change in the south Florida Everglades ecosystem. South Florida was mostly drier than normal from Water Year[1] (WY) 2009 to WY 2013 based on below average rainfall years (WY 2009, WY 2011 and WY 2012) compared to the historical 95-year average (1900–1995). The drier climate has had dramatic effects on the hydrology in the currently managed system. Water flows into and out of each basin were reduced in the majority of cases compared to the historical average (1972–2013) (**Table 2-1**) with flows to Water Conservation Area (WCA) 2 and Everglades National Park (ENP) being the most notable exceptions.

[1] A water year begins on May 1 and ends on April 30 of the following year. For example WY 2013 began on May 1, 2012 and ended on April 30, 2013, and does not include high rainfall and flow events that occurred in summer and fall 2013 (WY 2014).

Table 2-1. South Florida Water Management District area basin average annual inflows and outflows for WY 2009–WY 2013 (current) compared to historical flows (1972–2013). WCA-2 and ENP inflows are higher due to improvements in the ability to flow water south through the stormwater treatment areas and C-111 South Dade projects.

Water Body	WY 2009 to WY 2013 Annual Average Flows (acre-feet)	Historical Annual Average Flows (1972–2013) (acre-feet)	Percent Difference
Lake Kissimmee Outflows	704,045	704,329	-0.04
Lake Istokpoga Outflows	220,135	214,868	2.45
Lake Okeechobee Inflows	1,852,004	2,073,208	-10.67
Lake Okeechobee Outflows	1,009,482	1,422,072	-29.01
St. Lucie Canal Inflows (S-308)	136,946	252,139	-45.69
St. Lucie Canal Outflows (S-80)	143,367	298,688	-52.00
Caloosahatchee Inflows (S-77)	368,500	511,963	-28.02
Caloosahatchee Outflows (S-79)	996,286	1,202,746	-17.17
WCA 1 Inflows	266,654	477,534	-44.16
WCA 1 Outflows	314,339	443,383	-29.10
WCA 2 Inflows	826,410	631,274	30.91
WCA 2 Outflows	671,586	637,225	5.39
WCA 3 Inflows	1,125,389	1,172,139	-3.99
WCA 3 Outflows	1,063,156	1,004,772	5.81
ENP Inflows	1,184,953	977,031	21.28

Reduced inflows to Lake Okeechobee and below average rainfall overall coupled with the Lake Okeechobee Regulation Schedule implemented in 2008 (2008 LORS) helped reduce high stage events and kept water levels mostly within desired stage ranges for ecology. This also meant less discharge to the Northern Estuaries from Lake Okeechobee, which benefited estuarine ecology during the wet season but reduced flows to undesirable levels for the Caloosahatchee River Estuary (CRE) in the dry season. However, moving south through the system to the WCAs, lower flows and rainfall decrease marsh water levels (surface and groundwater) (see **Figure 2-1**, comparing WY 2009–WY 2013 and WY 2000–WY 2008) and caused severe dry downs in the system. This resulted in shorter hydroperiods, and higher salinities in estuaries, which negatively affected ecosystem indicators in the Greater Everglades and Southern Coastal Systems. In fact, water levels went below ground for a much longer portion of time from mid-April to June during WY 2009–WY 2013 in WCA 2A, northern and central WCA 3A, and ENP. Intense dry downs such as these place parts of the system at risk to severe fires and soil oxidation that can damage vegetation, burn peat, and decrease landscape structure heterogeneity (ridge and slough and tree islands). Landscape areas impacted by fires and oxidation can release nutrients from the soil during subsequent wetter periods, which can make their way to downstream wetlands and coastal waters.

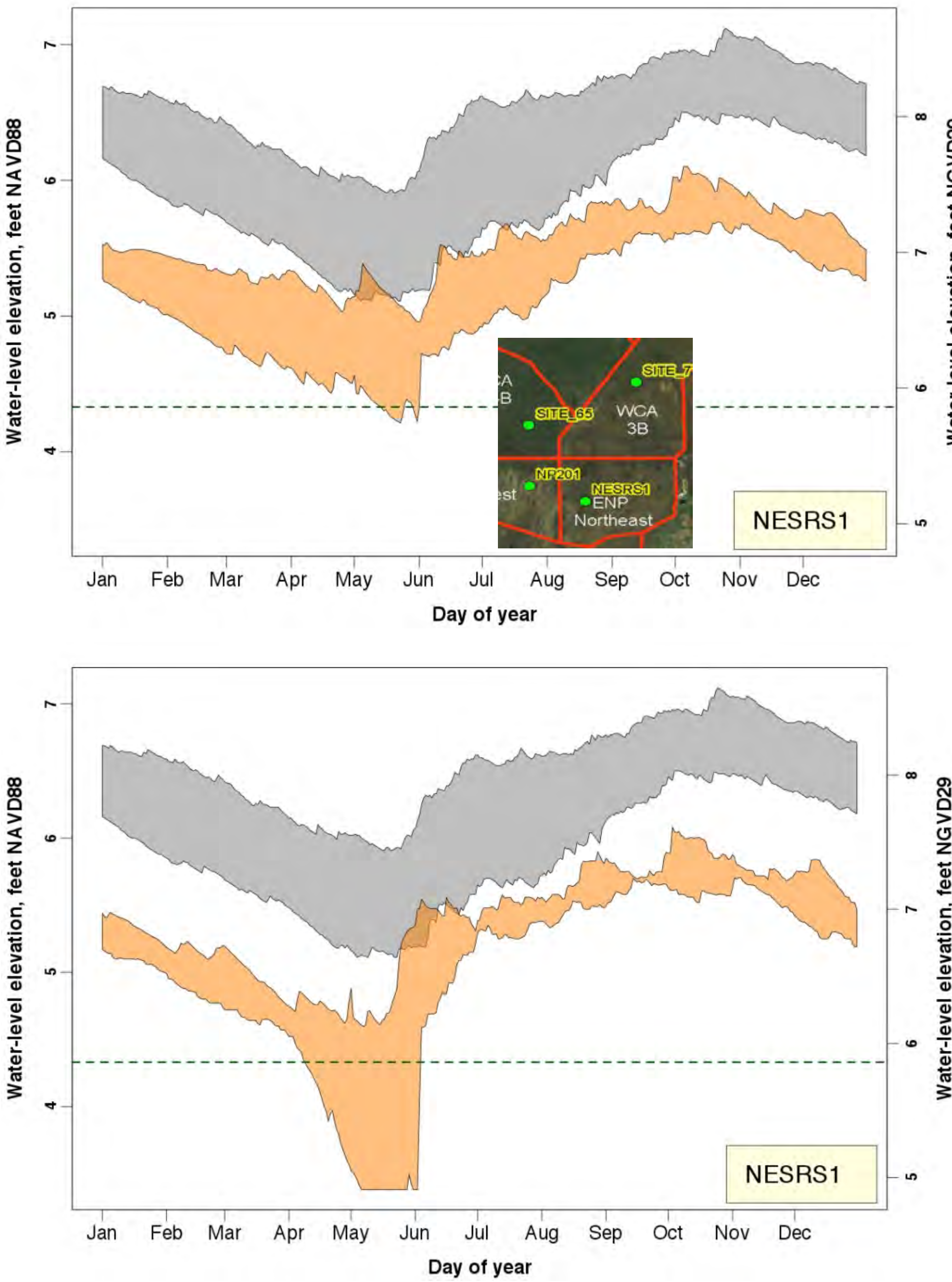


Figure 2-1. 25th to 75th percentile water levels for WY 2000–WY 2008 (top panel) and WY 2009–WY 2013 (bottom panel) at gaging station NESRS1 located within northeastern ENP. Measured water levels (light orange ribbon) are compared to pre-drainage water levels estimated by the Natural System Model (NSM) 4.6.2 (grey ribbon). Dark orange represents overlap area of observed data and the NSM benchmark and shows that, while water levels at the gage sometimes reach the lower estimates of historic levels, more water is needed in this area to achieve hydrologic restoration. Green dashed line represents the average ground elevation at the gage. Left Y-axis represents feet of water level elevation geodetically referenced to the National American Vertical Datum of 1988 (NAVD88). Right Y-axis represents water levels geodetically referenced to the National Geodetic Vertical Datum of 1929 (NGVD29).

Water levels in the southern Everglades are a major factor determining salinity in northern Florida Bay. In Florida Bay, salinity increased during WY 2009–WY 2013 and moved further away from the restoration target compared to WY 2000–WY 2008. Wet season salinity performance was below the target at almost all sites for all three performance metrics in those zones closest to the Everglades that are expected to be most affected by CERP (north, east, east-central and central zones). Salinity conditions in southern Biscayne Bay also showed unfavorable conditions compared to the target.

SYSTEMWIDE KEY FINDINGS

New Science – Disturbance Events

Natural, and sometimes extreme, variations in Florida's climate have long presented challenges for water managers and restoration decision makers. The unpredictability of the events can make them difficult to capture in monitoring, and therefore hard to incorporate in the knowledge base and tools used for decision making. **The monitoring for this SSR captured the January 2010 cold event that resulted in below freezing temperatures through portions of the Everglades and had profound effects throughout the ecosystem. Other disturbance events detected were a large algal bloom in the Southern Coastal Systems and the effects of altered estuarine currents from roadway changes. The information gathered from these disturbance events can be incorporated into tools to support restoration decision making.** The temperatures during the freeze greatly affected ecological parameter trends (e.g., depression of shoal grass in Biscayne Bay, and reduction in epifauna and mangrove fish species richness). Significant mortality in both native (e.g., American crocodiles, manatees, sea turtles, wading birds, and sportfish) and invasive nonnative species (e.g., Mayan cichlids, red-rimmed melania and armored catfish) was observed following the cold event (**Figure 2-2**). Other events documented in the Southern Coastal Systems that affected ecological indicators since the last SSR include the macroalgal bloom in the inshore waters of north-central Biscayne Bay that displaced seagrasses (mostly turtle grass) and a non-CERP project (the U.S. Highway 1 Road Expansion finished in 2008) that increased exchanges between northeastern Florida Bay and southernmost Biscayne Bay and affected submerged aquatic vegetation (SAV) and fish densities. Future efforts to assess restoration responses will enable the consideration of these rare event-related effects on the status and trends of CERP ecosystem restoration indicators, and ongoing restoration efforts will benefit from the data and knowledge gained regarding climatic events and anthropogenic changes to inform water management and ecological decisions.

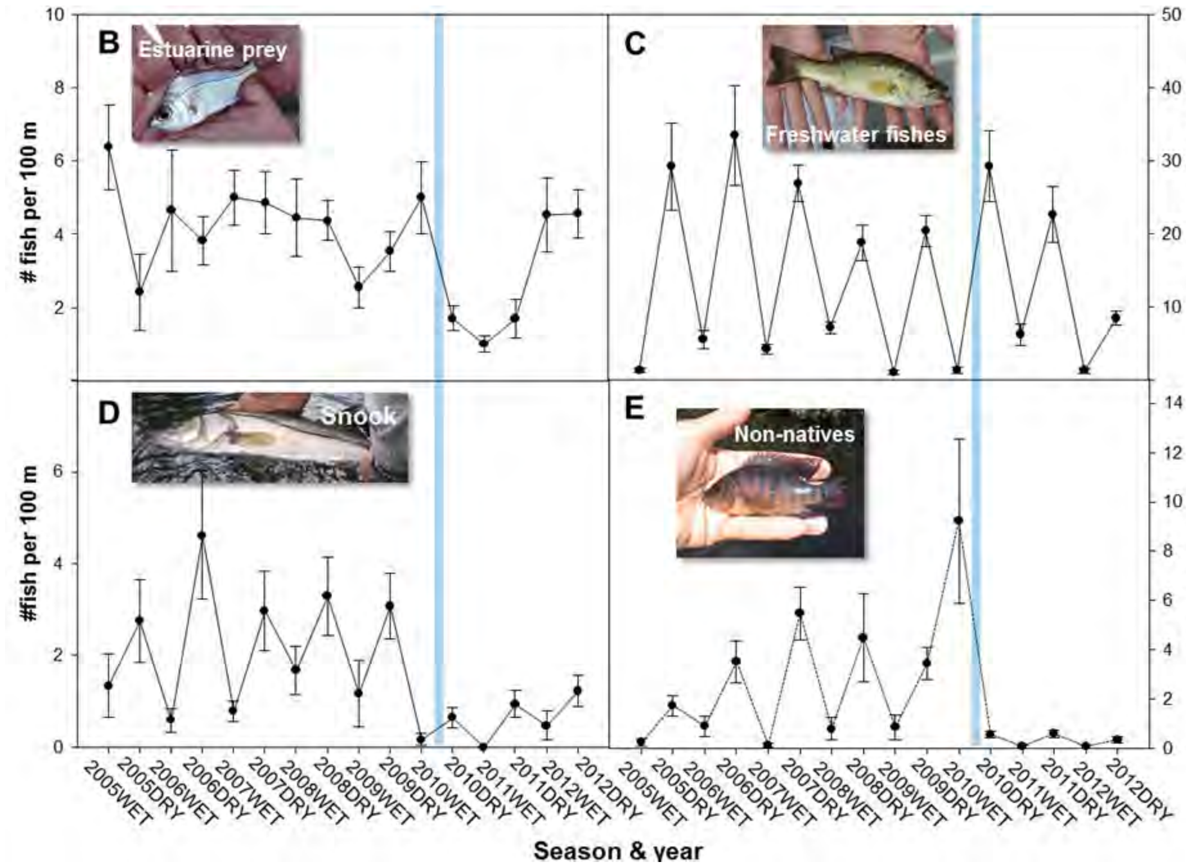


Figure 2-2. Seasonal and yearly abundance of (B) estuarine prey, (C) freshwater marsh fishes (both larger predators and smaller prey), (D) snook, and (E) nonnative fishes across long-term ecotonal mangrove creek sites in the upper Shark River, ENP. Data are from boat electrofishing and reflect the number (#) of fish caught per 100 meters (m) of mangrove creek shoreline at 10 fixed sites. Wet season samples correspond to November–December, while dry season samples aggregate early (February–March) and late (April–May) samples. The timing of the 2010 cold event is shown by the blue line.

New Science – Fire

ENP and Big Cypress National Preserve (BCNP) assessments of aboveground brush fire frequency concluded that insufficient aboveground brush fire (over the entire period of record [1946–2012]) is the most common condition occurring in fire adapted habitats in both ENP and BCNP (See Figure 2-3). Speaking broadly, there are two kinds of fire in the Everglades: aboveground brush fires that are a natural component of Everglades ecology, and damaging peat fires that have increased with Everglades drainage and cause dramatic losses of soil and elevation. Aboveground brush fires help these ecosystems transition from degraded to restored conditions. The desired outcome for fire adapted areas where significant shifts in hydrology are expected is an ecosystem that is well hydrated, experiences fewer days where extremely dry conditions prevail, and experiences frequent aboveground brush fires that are less severe than soil consuming peat fires. The more frequent but lower intensity fires are expected to produce a mosaic of vegetation types within a diverse landscape, which will be an ecological improvement from the current degraded conditions (such as the eastern edge of ENP that was added in 1989). **Recommendation: Incorporate the WCAs, Picayune Strand, C-111 Spreader Canal, and Biscayne Bay Coastal Wetland Project areas in the systemwide fire assessments of fire frequency and develop performance measure criteria to inform restoration success and the need for improvements to water management and landscape-level vegetation management.**

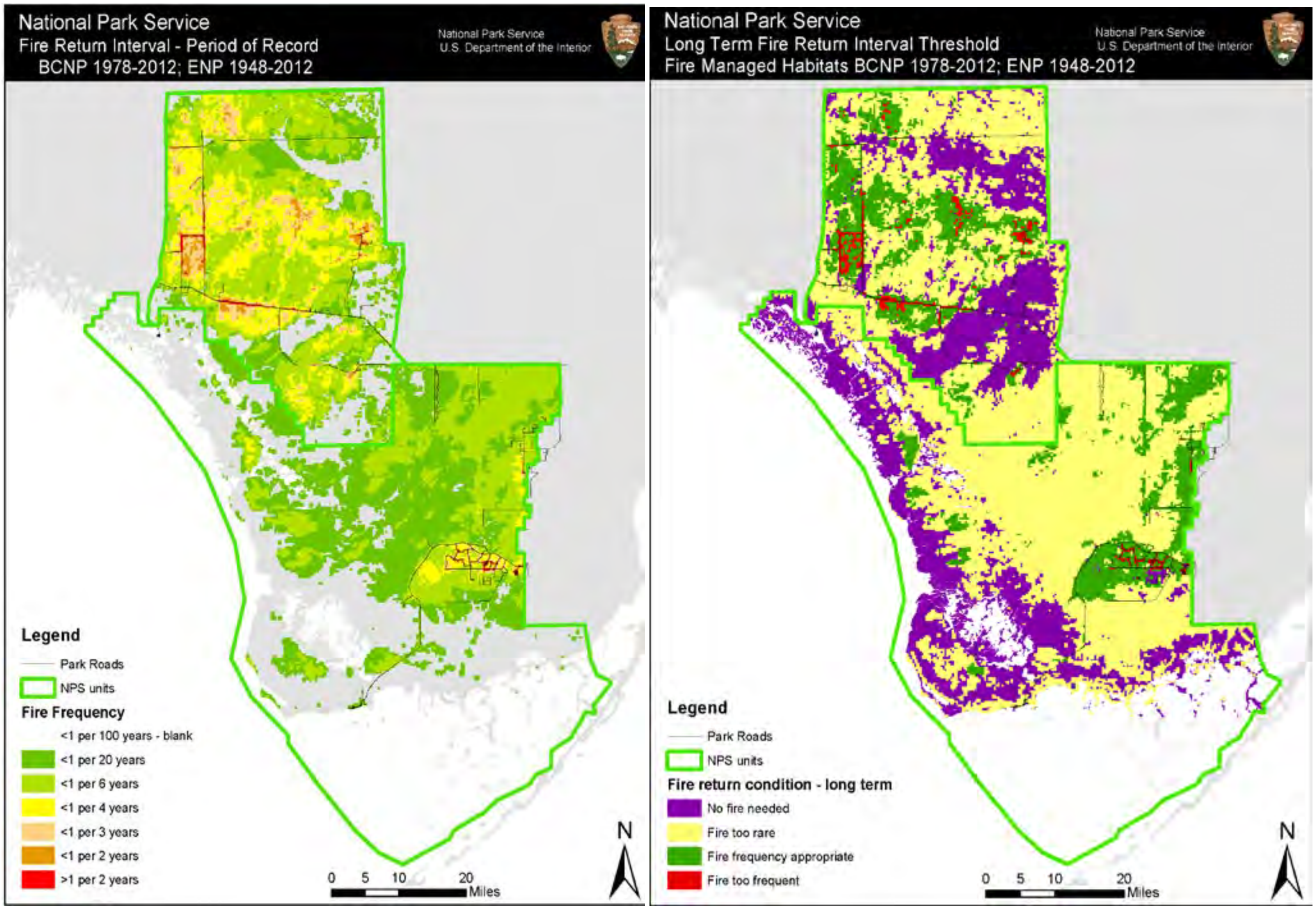


Figure 2-3. Depiction of long-term fire return times (left panel) and fire return conditions relative to desired return time (right panel) observed in ENP and BCNP.

Status and Trends – Systemwide Ecological Indicators

The systemwide ecological indicators reported by the South Florida Ecosystem Restoration Task Force's Science Coordination Group in 2012 (Brandt et al. 2012), reflect the continued patterns of severely altered hydrology throughout the ecosystem (Table 2-2). These findings are consistent with an assessment reported by the National Research Council in 2012 (NRC 2012) and reinforced with additional indicators reported in the 2014 SSR.

Table 2-2. Systemwide ecological indicators stoplight color status and trends. Red color – Substantial deviations from restoration targets creating severe negative condition that merits action. Yellow color – Current situation does not meet restoration targets and may require additional restoration action. Green color – Situation is within the range expected for a healthy ecosystem within the natural variability of rainfall. Black color – No data or inadequate data.

	WY 2008	WY 2009	WY 2010	WY 2011	WY 2012
Lake Okeechobee					
Invasive Exotic Plants Species					
Lake Okeechobee Nearshore Zone Submersed Aquatic Vegetation					
Northern Estuaries					
Invasive Exotic Plant Species					
Eastern Oysters					
Greater Everglades					
Crocodilians					
Fish and Macroinvertebrates (WCA 3 and ENP only)					
Invasive Exotic Plants					
Periphyton and Epiphyton					see footnote a
Wading Birds (White Ibis and Wood Stork)					
Southern Coastal Systems					
Crocodilians					
Southern Estuaries Algal Blooms[b]					
Florida Bay Submersed Aquatic Vegetation					
Invasive Exotic Plants					
Juvenile Pink Shrimp[c]		see footnote d			
Wading Birds (Roseate Spoonbill)					see footnote e
Wading Birds (White Ibis and Wood Stork)					

a. Species composition data is not available.

b. Algal bloom indicator values are for calendar years 2007 through 2011, roughly corresponding to the water years shown.

c. The status for juvenile pink shrimp (*Farfantepenaeus duorarum*) contains information for data collected for September–October.

d. Data from WY 2008–WY 2010 were used as a baseline.

e. Prey community data have not yet been processed.

Status and Trends – Systemwide Summary

Hydrology remains altered across the system (too dry in northern WCAs and in ENP, and too wet in southern WCAs) and continues to impact ecosystem indicators of restoration success.

- Periphyton communities in WCA 3A and along the eastern edge of ENP continue to indicate altered hydrology and nutrient conditions.
- Fish populations illustrate long-term decreases in number and biomass in WCA 3A, WCA 3B, portions of Shark River Slough, and Taylor Slough.
- Tree islands greater than one hectare in size decreased by 50% between 1954 and 2004 in Shark River Slough (RECOVER 2013).
- Florida Bay salinity conditions were further from restoration targets over the past four years compared to the 2009 SSR (RECOVER 2011)
- Large patches of dead seagrass were also observed on western and central Florida Bay banks.

Certain ecosystem indicators illustrate improvements due to restoration projects being constructed and operational.

- Hydrology improved due to the operational part of the Deering Estate Biscayne Bay Coastal Wetlands expedited project.
- Picayune Strand showed higher water levels near the filled Prairie Canal (1 to 2 feet higher) and vegetation is starting to show signs of improvement and moving closer to reference conditions.
- Hydroperiods were 50 days longer (on an annual average basis) along the central-eastern edge of ENP as a result of the C-111 South Dade Project.
- Roseate spoonbill nesting improved, most likely due to favorable climatic conditions and better real-time environmental coordination with water management operational decisions.
- Crocodile nesting and population trends increased along the southwestern Florida coast due to the Cape Sable plug restoration projects (over the past two decades).

This continued trend in altered hydrology and degraded ecology coupled with demonstrations of small restoration successes shows the need for and value of authorizing, constructing, and operating more CERP restoration projects to achieve systemwide hydrological (water quantity, quality, timing, and distribution) and ecological (flora, fauna, and landscape) goals and objectives.

Projects and Operations – Applied Science for Central Everglades Planning Project

The Central Everglades Planning Project (CEPP) used approved RECOVER performance measures and new ecological planning tools developed from RECOVER MAP data to inform the planning process. CEPP used RECOVER performance measures in their evaluations of project performance and development of habitat unit calculations because they had already been vetted through the RECOVER review process and did not require additional model review by the United States Army Corps of Engineers other than approval by the National Ecosystem Planning Center of Expertise. Additional ecological tools were used to understand effects on oysters, SAV, cyanobacteria, nutrients, alligators, wading birds (great egrets, wood storks, and white ibis), Everglades landscape vegetation, marl prairie habitats, prey fish and large fish (bass), spotted seatrout, juvenile crocodiles, and pink shrimp.

NORTHERN ESTUARIES

New Science – Salinity and Storage

New data indicates that supplemental freshwater inflows to the SLE during extremely dry years, especially when they occur back to back, may be needed to maintain healthy oyster populations in the middle estuary. Climate (temperature and rainfall) patterns during WY 2009–WY 2013 led to new findings in the distribution of flow requirements for oysters in the SLE. In previous years, the groundwater and tributary contribution eliminated the need for additional inflows. In fall 2008, oysters experienced a die-off in the central area of SLE from storm events that decreased salinity. Oysters recovered in WY 2009 and WY 2010 during dry and average rainfall years, respectively (green dots in **Figure 2-4**). However, in the next two back-to-back dry years (WY 2011 and WY 2012), salinity measurements exceeded the optimal range of 12 to 20 at U.S. 1 Roosevelt Bridge due to decreased freshwater supply and salinities were above the optimal range. High salinities (> 20) reduced the health and survivorship of oysters by increasing *Perkinsus marinus* (dermo) disease (note the red dots in **Figure 2-5**) and predation rates (not shown). **Recommendation: Utilize these findings that reflect changes in groundwater and tributary flows during extended dry periods with salinities exceeding 20 for future CERP planning of Indian River Lagoon - South Project phases, including operations.**

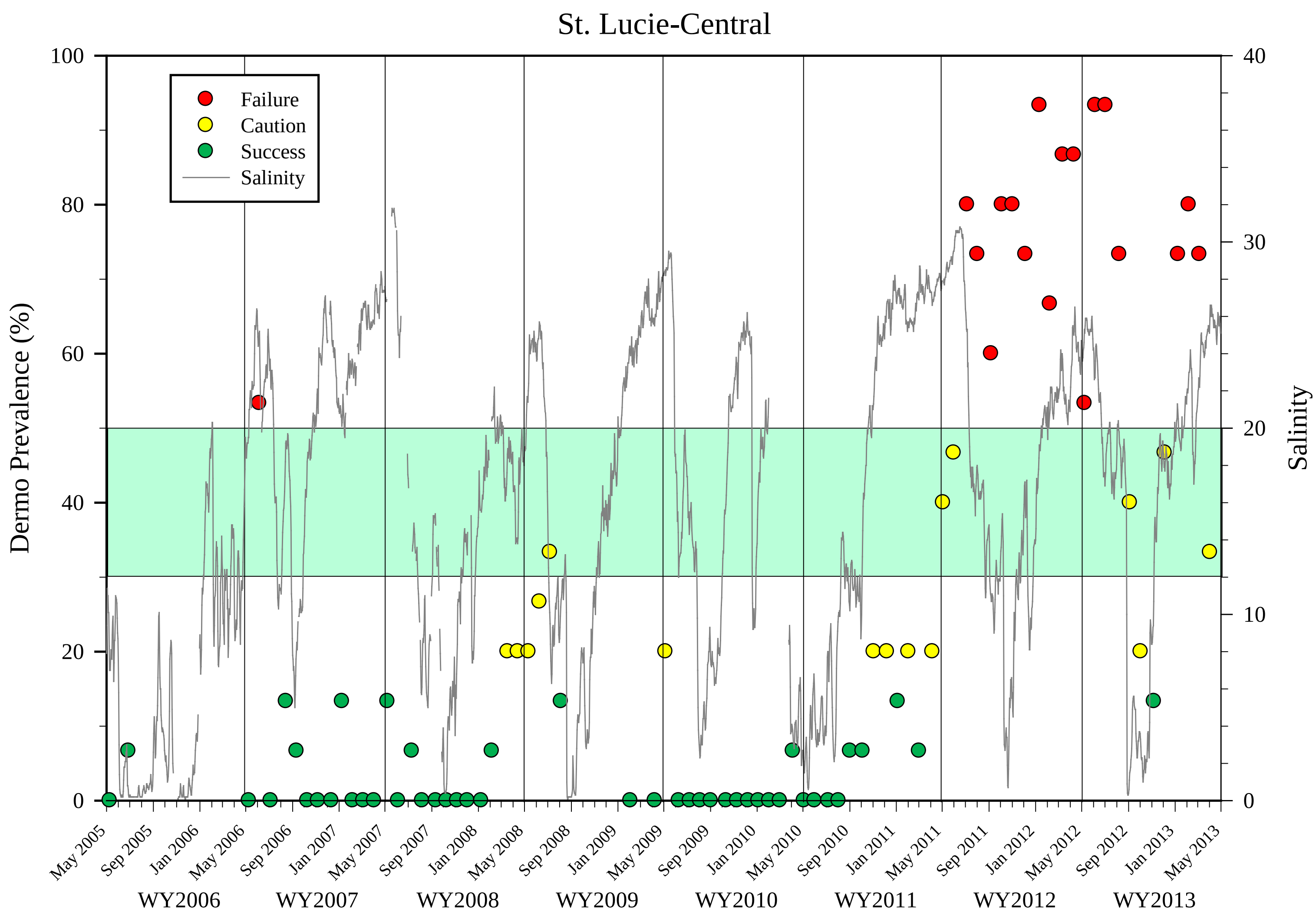


Figure 2-4. Percent of oysters infected with dermo disease in the SLE middle estuary, daily salinity from the surface at the U.S. 1 Roosevelt Bridge, in Stuart, Florida, and the favorable salinity range (green band) at same location.

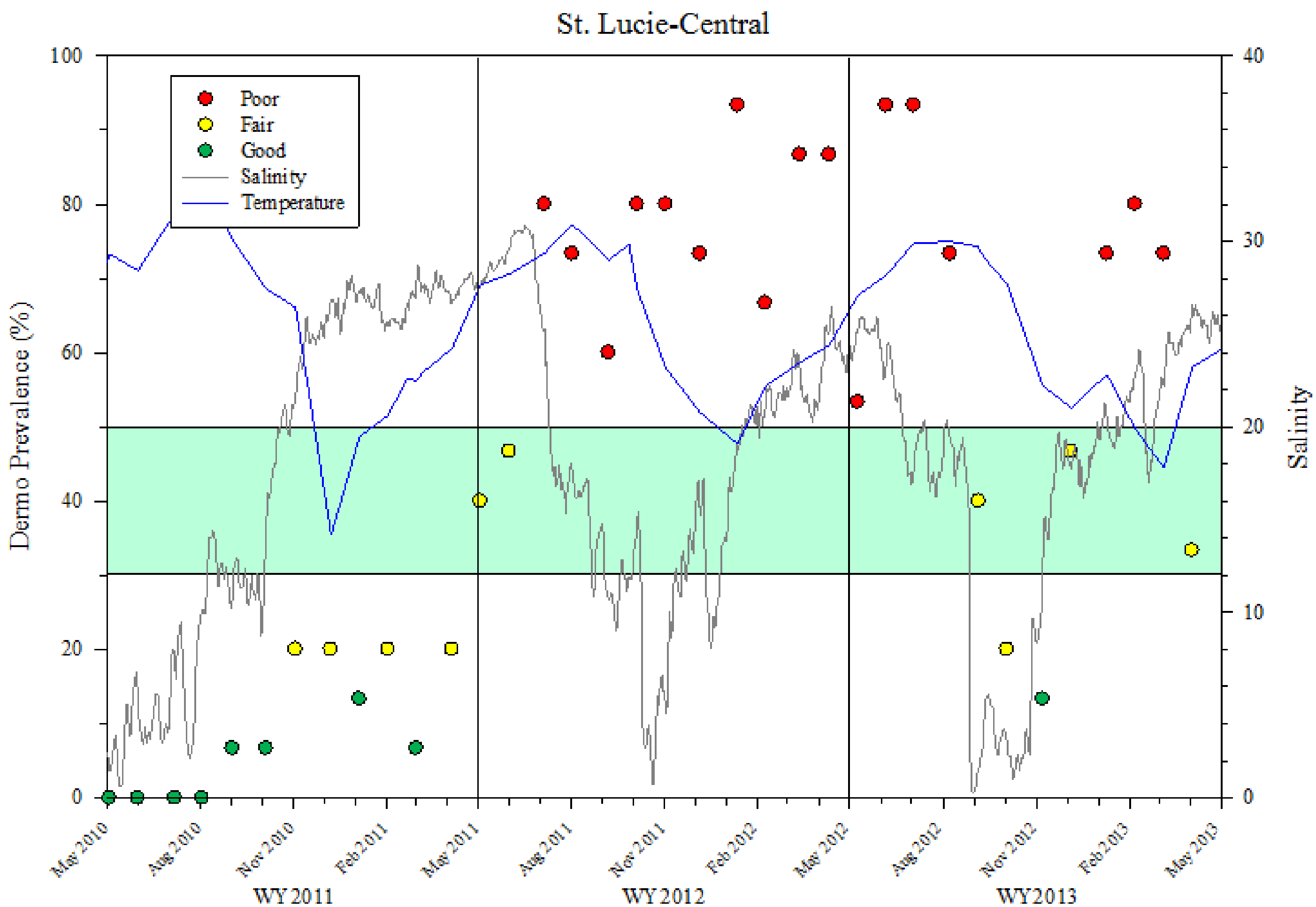


Figure 2-5. Monthly prevalence (red, yellow, and green circles) of oysters infected with *Perkinsus marinus* (dermo) during monthly collections from the SLE middle estuary and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. Daily mean salinity exceeded the optimum range of 12–20 at the U.S. 1 Roosevelt Bridge from October 30, 2010–July 10, 2011 (254 days) leading to significant increases in prevalence of dermo among oysters in the estuary as water temperatures increased in spring and summer 2011. The magnitude and duration of this dry period far exceeds any other dry period recorded during this study and provides first time evidence suggesting higher than optimal salinities can be detrimental to oysters in the SLE.

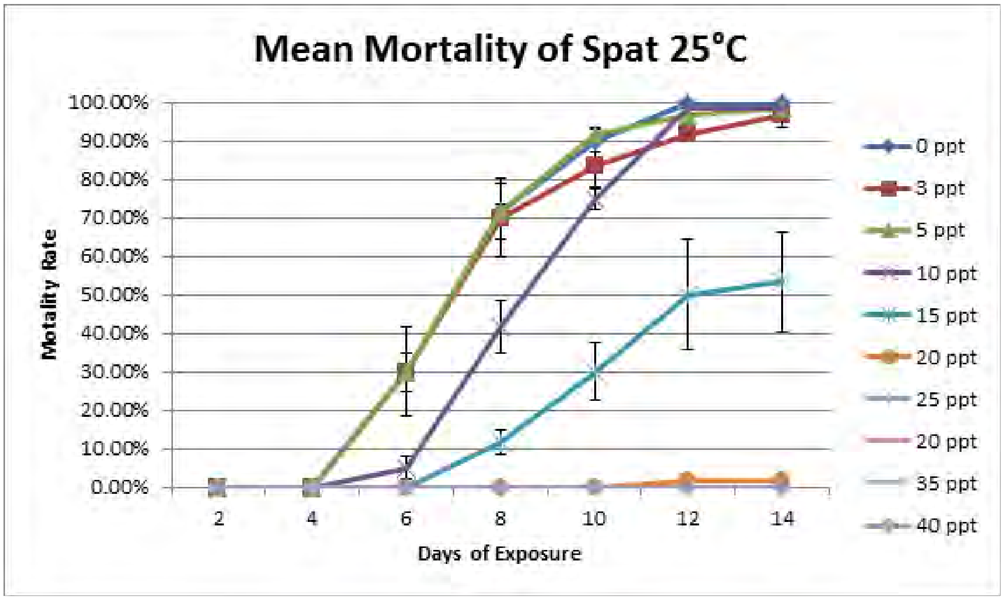
New Science – Submerged Aquatic Vegetation Recovery Times

Rates of recovery of SAV from an extreme low salinity event associated with Hurricane Wilma ranged from several months to several years depending on the particular species and specific estuary. When salinity at the US 1 Roosevelt Bridge in Stuart, Florida is within the established envelope for oyster health (12–20), salinities further downstream where seagrass beds exist are within seagrass tolerable ranges. The data also suggest that rapid drops (1 to 2 per day) in salinity may be detrimental to seagrass even if these are within a tolerable range and recovery can take months to years. Controlled mesocosm experiments are recommended to confirm this hypothesis. Seagrass recovery from an adverse impact caused by an extreme high flow/low salinity event (2005) in the Southern Indian River Lagoon (SIRL) took 8 to 17 months for Johnson's seagrass, 8 to 10 months for shoal grass, and approximately 4 years for manatee grass. In response to the same event in the CRE, shoal grass in San Carlos Bay recovered within 1 year while turtle grass required about 3 years. In contrast, tape grass in the upper estuary of CRE takes greater than 3 years to recover from a high salinity event. Factors other than salinity and freshwater inflow, such as nutrients, turbidity, and light penetration, are additional controlling factors on seagrass growth and diversity. Collection of additional data would help to determine the causes of current patterns in species abundance and diversity.

New Science – New Oyster Physiology Information Gained from Mesocosm Experiments

Data from the CRE indicate that management activities that decrease peak freshwater discharges (high flow events) and allow a more gradual release of water into the estuary are beneficial. Oysters typically spawn in the CRE from May through November, with a peak in July–August. The majority of spawning occurs during the rainy season when flows are usually the highest and salinities the lowest. In an effort to assist water managers, provide operational strategies, and predict the effects of flow and salinity, experiments (mesocosms) were performed on early oyster life stages. While it was known that adult oysters can tolerate salinities of less than 5 for 1 to 2 weeks it is important to establish ranges for more vulnerable life stages. It was determined that in the CRE, salinities of less than 15 are stressful to larval and juvenile stages, especially when salinity change is rapid. When salinity change is gradual, juvenile oysters can tolerate salinities less than 15 for 1 to 2 weeks depending on the temperature. This information confirms that previous planning hypotheses related to management activities that decrease peak discharges and allow a more gradual release of water into the estuary would be beneficial.

New evidence from field and mesocosm experiments also demonstrates that high temperatures result in additional stress to oysters by increasing predation rates and lowering salinity tolerance. Mesocosm experiments demonstrated that regardless of salinity, high temperatures (e.g., 30 degrees Celsius or 86 degrees Farenheit) are stressful to juvenile oysters and affect their survival (**Figure 2-6**), and the combination of high temperatures and low salinities are particularly stressful. Minimizing high flows that depress salinities to less than 10 during the summer, when temperatures are high, will aid in the survival and growth of oysters. Field studies show that at higher salinities and temperatures, predation is a significant factor affecting oysters and should be considered in modeling or predictive studies. This new evidence when combined with oyster models provides better information related to hydrologic planning models. **<u>Recommendation:</u> Incorporate temperature effects into existing oyster models to better predict restoration outcomes.**



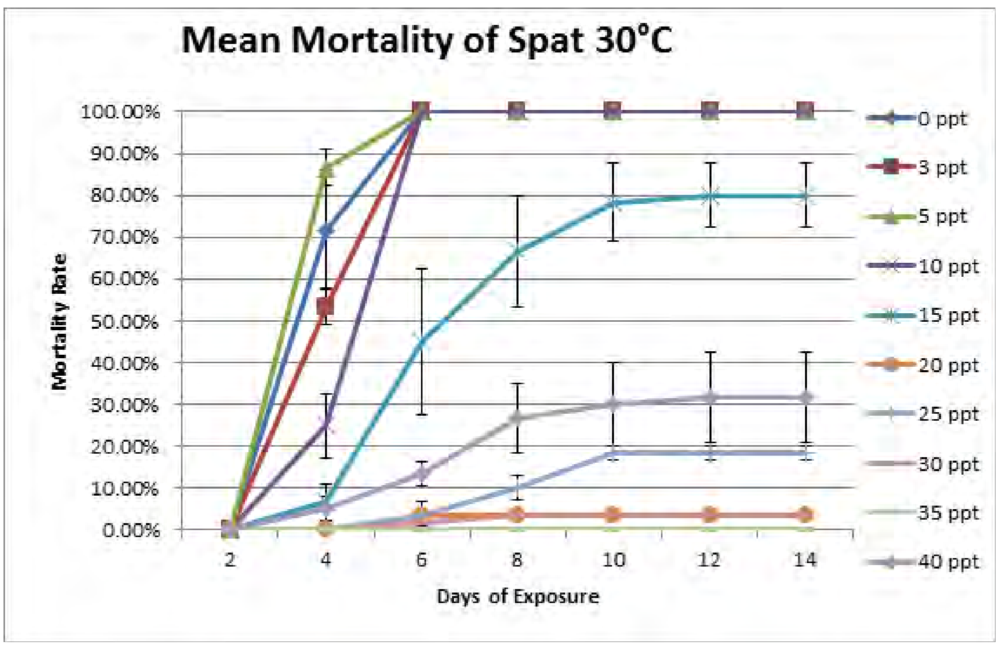


Figure 2-6. Mean mortality of juvenile oysters (“spat”) exposed to a range of salinities. Juvenile oysters at 25 psu were subjected to acute salinity change to mimic high volumes of freshwater releases into the estuary and mortality observed. Results suggest that while salinities less than 15 may be stressful to juvenile oysters, summer water temperature of 30 degrees Celsius is also stressful increasing mortality rates at all salinities. (Note: ppt – parts per thousand, which is an outdated salinity unit. Salinity is now considered an unitless measure.)

Projects and Operations – Oysters – Artificial Substrate and Timing

For oysters to form reefs that provide habitat to other invertebrates and fish, they require the correct salinity as well as hard substrate on which to build. The addition of artificial hard substrate is necessary for oyster restoration since most has been lost due to dredging, large-scale oyster mortality, and by being covered by siltation. In 2010, Martin County and the Loxahatchee River District, with funding from the American Recovery and Reinvestment Act through National Oceanic and Atmospheric Administration (NOAA), added 5.8 acres of hard artificial substrate to the Northwest Fork of the Loxahatchee River. This successful restoration of oyster reefs provided habitat for 5,000 pounds of nonoyster animal biomass (small fish, crabs and shrimp) 20 months after construction. **Naturally occurring oyster populations provide a sufficient supply of larvae to occupy suitable substrate.** Similarly, Martin County constructed 22 acres of artificial oyster reefs in the St. Lucie Estuary. Results indicated that the best time to deploy substrate is just prior to spawning and when salinity is in the tolerable range for oyster growth and survival. Recruitment onto the artificial reef was successful indicating that there were sufficient numbers of naturally occurring oysters available to provide larvae for settlement on that substrate. **Recommendation: Document the availability of hard substrate in all estuaries and in areas where restoration of flows from projects like Indian River Lagoon – South and Caloosahatchee River (C-43) West Basin Storage Reservoir will create suitable salinity ranges for oysters, and undertake substrate enhancement.**

Projects and Operations – Bottom Substrate Effects on Health of Benthic Macrofauna

Improvements in salinity conditions as well as the removal of muck deposits will be necessary for full restoration of bottom ecological communities in the SLE. Salinity and sediment composition are the major determinants of benthic macrofauna (e.g., worms, clams, and burrowing shrimp) species diversity in the SIRL and SLE. Species diversity is greatest at the sites in the SIRL and the lower SLE, that have a more stable, higher salinity compared with SLE North and South forks. Coarser or sandy sediments found in SIRL maintain more diverse communities of benthic macrofauna. Mucky sediments found in the North and South forks of the SLE support relatively low species diversity. The long-term data set that includes both wet and dry cycles has been helpful in establishing a strong relationship between salinity and species diversity at other sites in the SIRL and lower SLE (**Figure 2-7**). However, this relationship is not observed at the sites with mucky sediments (**Figure 2-8**), suggesting that improvements in salinity alone might not lead to an increase in species diversity. Enhancing species diversity in the central SLE and the North and South forks will require removal of muck for full restoration. **<u>Recommendation:</u> Add muck removal to the CERP Indian River Lagoon - South Project implementation schedule sooner than currently planned to realize earlier benefits to the SLE. This should be done in an incremental fashion in the vicinity of reservoirs and stormwater treatment areas as they become operational.**

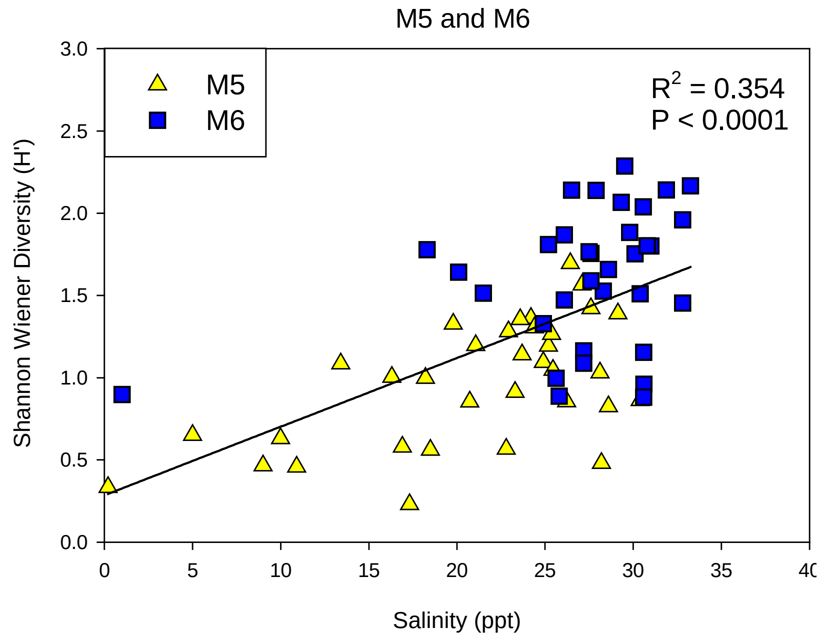


Figure 2-7. Relationship between bottom salinity and diversity of fauna living in the sediment at two stations (M5 and M6) in the SLE near the US Roosevelt 1 Bridge. These sediments are relatively sandy.

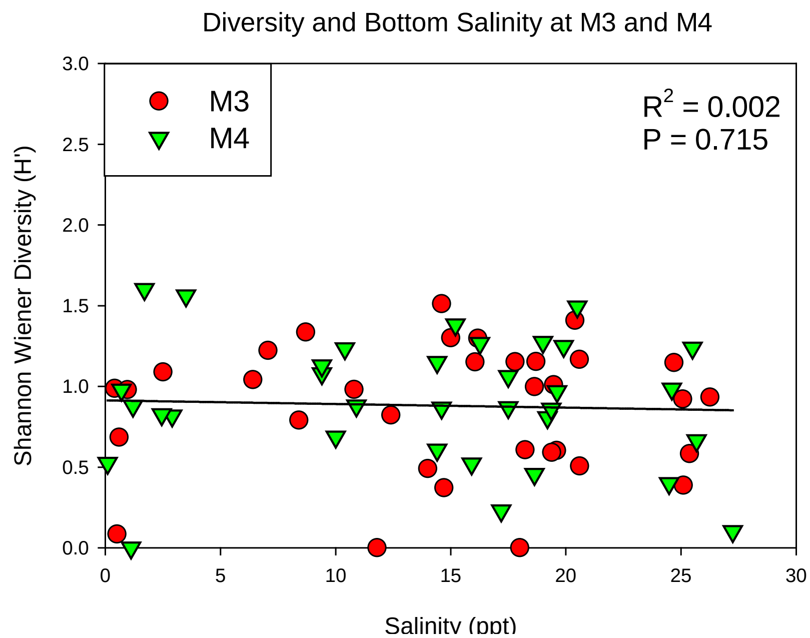


Figure 2-8. Relationship between bottom salinity and species diversity is illustrated for sites M3 and M4 located in the North and South forks of the SLE, respectively. Salinity is highly variable at these sites, but there is no relationship between salinity and diversity at these locations based on analysis by linear regression. These sediments have a high content of muck.

LAKE OKEECHOBEE

New Science – Performance Indicators Scenario Modeling

Statistically significant correlations ($p < 0.05$) were found between lake stage and (1) SAV, (2) panfish (bluegill and red-ear sunfish), and (3) reduced problematic algal blooms (cyanobacteria-dominated) and were used to develop additional performance metrics scoring to evaluate storage scenarios to improve Lake Okeechobee ecology. Modeling confirmed that increasing the amount of storage north of Lake Okeechobee incrementally improved lake conditions (over the existing 41-year baseline) by increasing the amount of time the lake was within the "stage envelope" (i.e., ecologically preferable range) (**Figure 2-9**). Three storage scenarios of an additional five hundred thousand, one million, and two million acre-feet were modeled north of Lake Okeechobee in an attempt to evaluate the effects on the lake's biota. All modeled storage scenarios provided ecological benefits to these three ecological indicators although the amount of improvements between the difference storage volumes appeared to be incrementally smaller as reservoir size increased. Refining the lake's regulation schedule operational protocols can greatly affect ecological benefits. **Recommendation: Additional storage is needed to improve and maintain ecological benefits presented in this SSR. Future storage planning efforts should consider dispersed water storage, Kissimmee River restoration, aquifer storage and recovery, and optimization of future Lake Okeechobee regulation schedule operations to further investigate the scope of these additional ecological benefits per modeled scenarios.**

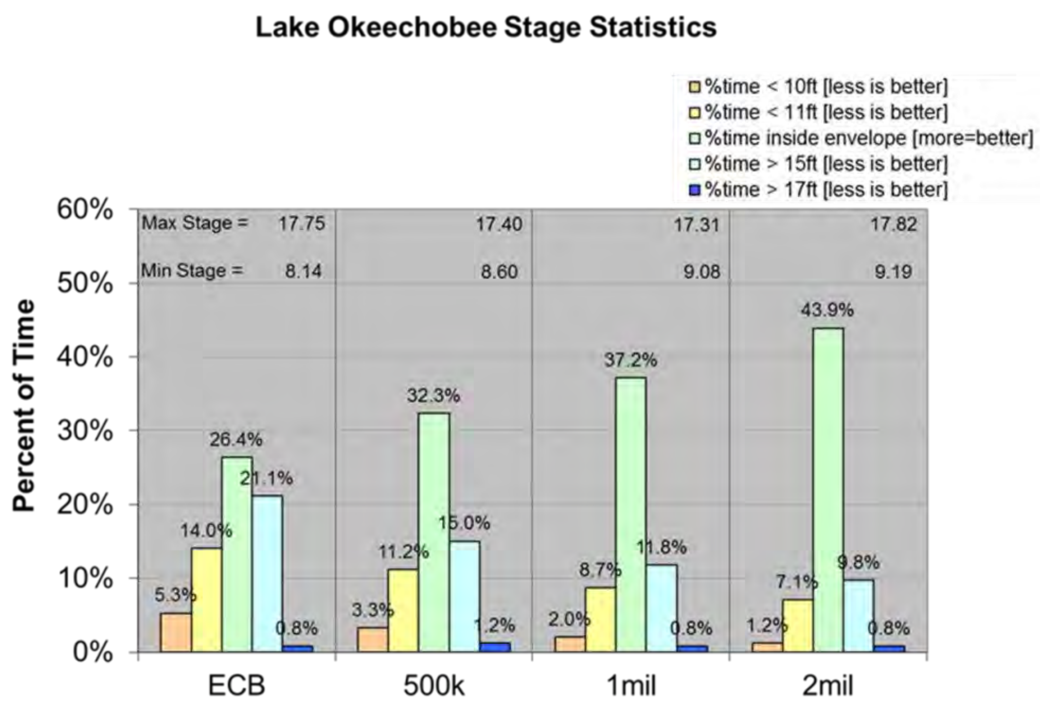


Figure 2-9. Percent of time modeling scenarios—existing conditions base (ECB), storage of 500,000 (500k), 1 million (1mil), 2 million (2mil) acre-feet—indicate lake stage would have been in the ecologically preferred stage envelope (green bar – 12 to 15.5 feet).

Status and Trends – Ecology

Lake Okeechobee ecology improved over the WY 2009–WY 2013 period compared to the WY 2004–WY 2008 (back-to-back years of hurricane impacts directly affecting the lake in 2004 and 2005) period in part due to favorable climatic conditions and 2008 LORS changes. While the primary driver in developing the 2008 LORS intermediate schedule was protecting the Herbert Hoover Dike, this schedule maintains the lake about a foot lower than the previous schedule, thereby helping to keep the lake in the ecologically desirable stage envelope more frequently.

Compared to 2009 five-year rolling averages, current mean total phosphorus (TP) load and nutrient concentrations in the water column were reduced by 19 and 33 percent respectively, summer nearshore SAV coverage increased by 12,200 acres, periphyton were more abundant, and algal bloom frequency slightly increased by 2.6 percent. Ecological conditions likely improved when Lake Okeechobee water levels stayed within the desired stage envelope (ecologically preferred range of 12 to 15.5 feet) due to beneficial climatic conditions and the 2008 LORS reducing high lake stages (no high stage exceedences). These results also indicate that Lake Okeechobee ecology will improve once more storage outside of Lake Okeechobee is available to reduce the frequency of above average inflows during wet years and the ecological desired stage envelope is incorporated as part of Lake Okeechobee operations.

Projects and Operations – Lake Okeechobee Storage Benefits to Estuaries

All modeled storage scenarios north of Lake Okeechobee show benefits to the St. Lucie Estuary (SLE) and CRE via reductions in frequency of large freshwater releases (> 3,000 cubic feet per section [cfs]) from Lake Okeechobee, as well as increases in frequency of optimal flows (350–2,000 cfs).

GREATER EVERGLADES

New Science – Predator-Prey – Fish Dynamics at the Everglades Marsh-Mangrove Ecotone

We have a better understanding of the relationship between natural climatic variability and water management practices that affect the dynamics of both freshwater and estuarine fishes in the mangrove zone. Freshwater marsh fishes move downstream into mangrove creeks during the dry season, and estuarine predators move between upstream and downstream regions of the Shark River estuary (**Figure 2-10**). This hydrological-driven displacement of marsh fishes provides foraging opportunities for important recreational estuarine fisheries like snook. Mangrove creeks act as critical dry down refuges for marsh fishes, similar to the role of alligator holes elsewhere in the ecosystem, therefore, seasonal drydowns in marshes can spill over to influence population dynamics of fishes in the coastal mangrove zone. Pulsing of marsh prey into the estuary, where they are consumed by estuarine as well as freshwater consumers such as largemouth bass, may represent a shift in energy flow such that prey produced in the marsh that would historically be available for wading bird foraging are now displaced into the estuary as a result of too frequent seasonal marsh dry downs. **Recommendation: Supply additional water into the estuary via Shark River Slough to reduce the frequency and duration of seasonal marsh dry downs through CERP restoration projects like CEPP to increase the availability of forage fish to wading birds in the marsh and send marsh fish into the upper estuary over a longer**

part of the year. Additional flow to the marsh should also support an increased number of large freshwater fish (e.g., largemouth bass) in the freshwater marsh and the associated decrease in salinity in the upper estuaries should increase their use of coastal creeks.

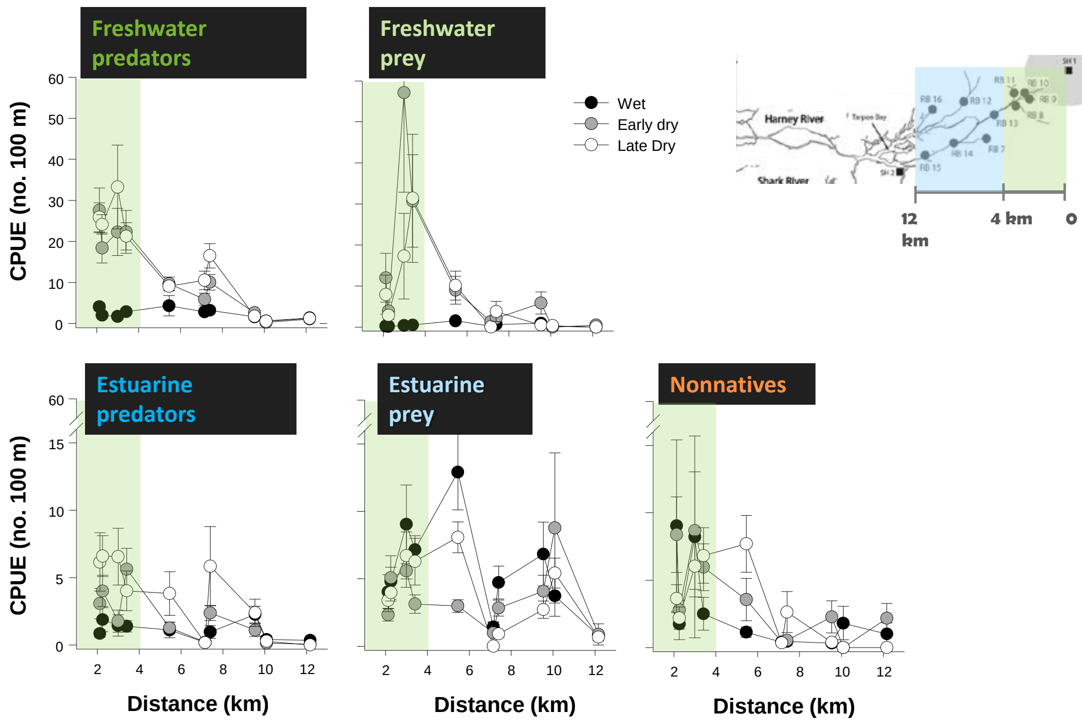


Figure 2-10. Abundance (number of fish per 100 meters of mangrove creek shoreline) of the five main functional groups of fishes found in the ecotone (mean ± standard error): freshwater or marsh predators, freshwater or marsh prey, estuarine predators, estuarine prey, and nonnative taxa as function of distance to marshes for three seasonal samples. Green shading indicates sites located in the upper 4 kilometers of the estuary and closest to freshwater marsh habitats.

New Science – Predator-Prey – Dry season prey

Droughts decrease the density of larger (> 2 centimeter [cm] total length) fishes that are the preferred prey for great egrets and wood storks, but they increase the density of crayfish, the primary food for white ibis. Linking prey performance to wading bird performance can help improve predictive tools for restoration and operational planning. Whereas fish make up the bulk of wading bird diets, particularly for large birds like the wood stork and great egret, some species can switch from fish to crayfish and may be less affected by a drought. Crayfish are the primary food of the white ibis, so droughts generally have a strong positive effect on this species. This observation departs from previous work, which concentrated on the longer hydroperiod effect on fishes and illustrates that long-term management needs to include hydrological variation. Droughts appear to cue the system to create pulses of crayfish prey for ibises, while wet years appear to create robust populations of fishes for herons, egrets, and storks (**Figure 2-11**). It seems likely that this mix of conditions favors different avian strategies and was typical of the much larger pre-drainage Everglades. This information has important ramifications for interpreting restoration success trends for different wading bird species. **<u>Recommendation:</u> To recover historic nesting populations of wading birds, water management seasonal operations need to mimic natural cycles of rainfall and interannual variation both in space and time.**

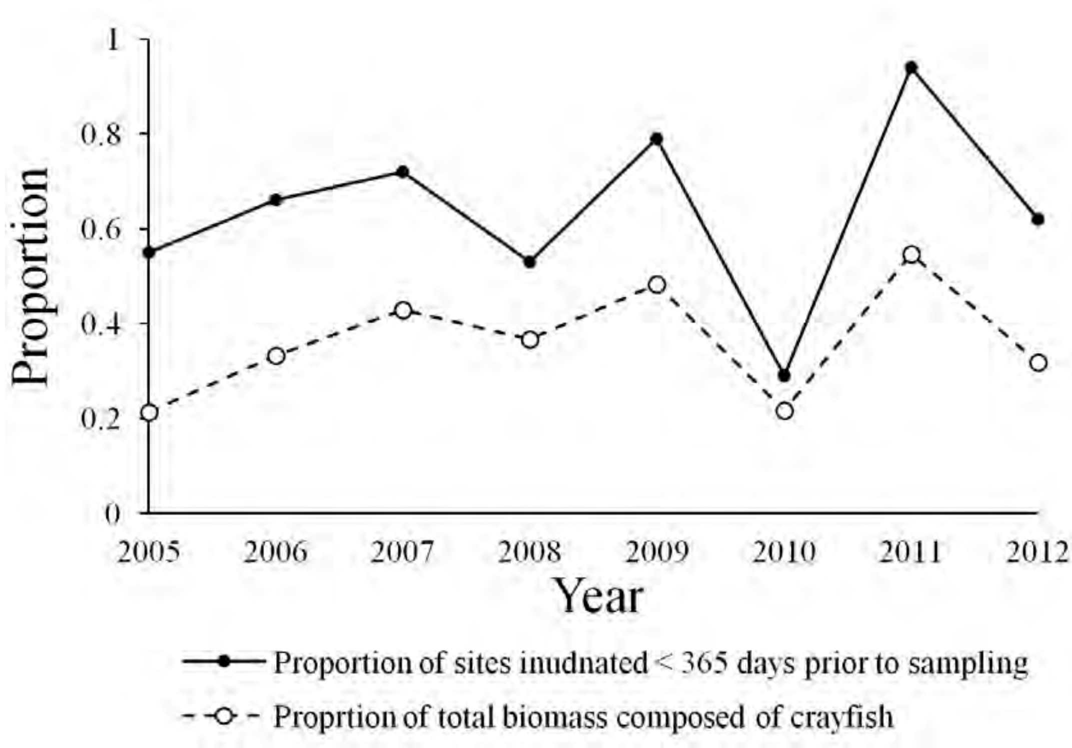


Figure 2-11. Proportion of sites where the marsh surface dried (water depth < 5 cm in data from the Everglades Depth Estimation Network [EDEN]) in < 365 days prior to sampling and proportion of total wading bird prey biomass composed of crayfish from 2005 to 2012 across all sites sampled in the Everglades.

New Science – Predator-Prey – Wading birds

The same hydrologic patterns can produce different population trends (positive or negative) in different wading bird species (white ibis and great egret), which is an important realization for interpreting monitoring results in relation to restoration performance. Monitoring from 1986 to 2013 shows that white ibises were more sensitive to hydrologic reversals than great egrets. They were more selective of foraging sites (particularly after hydrological reversals), their clutch size declined, and their fledged chicks (those leaving the nest) were in poorer physical condition when compared to great egrets during suboptimal habitat conditions. Poor foraging conditions are produced by dry seasons with interrupted drying patterns that followed wet seasons with poor food production (e.g., dry wet seasons). Such conditions will likely produce earlier, and larger negative responses (e.g., foraging site abandonment, increased stress, and nest failure) in white ibis than great egrets. Ibises seem likely to respond more rapidly to changing hydrological conditions, suggesting that they will be one of the first responders to hydrological restoration (**Figure 2-12**). Because great egrets responded less acutely to changing hydrological conditions during this study, restoring hydrological conditions across the Everglades should not be expected to produce as large or as quick a response in nesting patterns for egrets as for ibises; present hydrologic patterns in the Everglades appear to suit the great egret well.

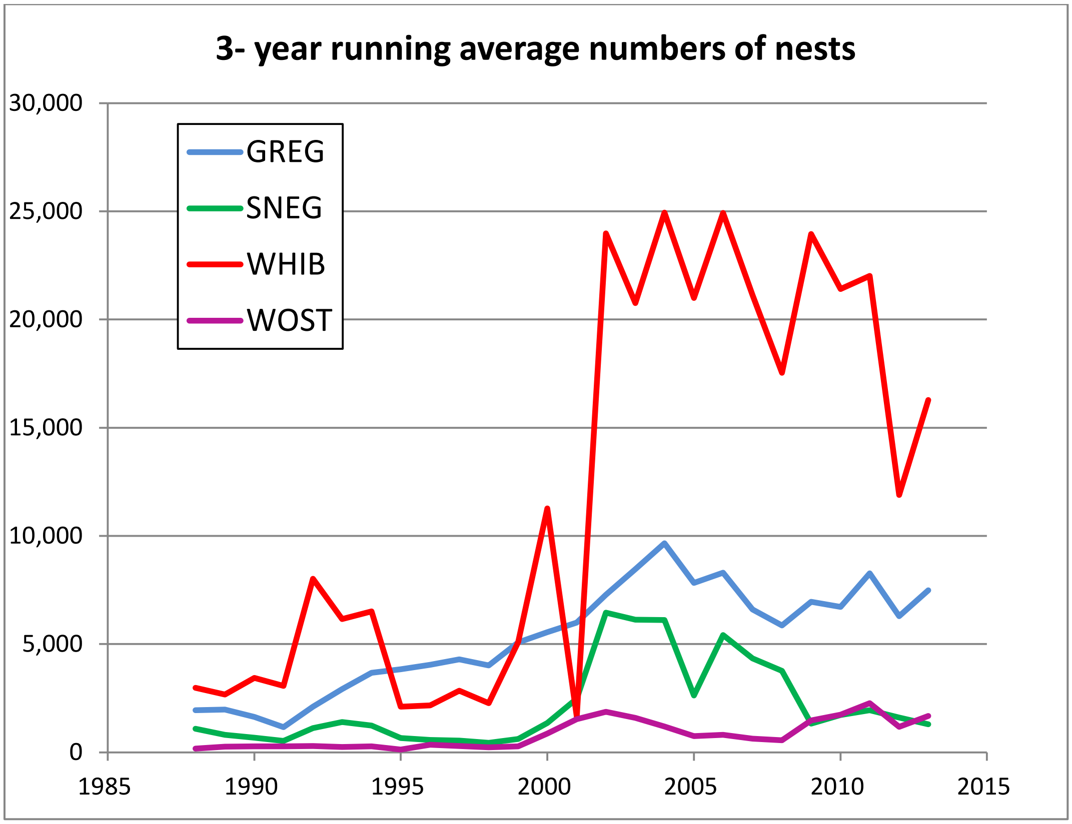


Figure 2-12. Total numbers of nests in the Greater Everglades by four indicator species 1986–2013. Great Egrets = GREG, Snowy Egrets = SNEG, White Ibis = WHIB, and Wood Storks = WOST.

New Science – Predator-Prey – Trends in American Alligator Relative Density.

Areas that experience dry downs that last longer than two months (60 days) or repeated dry downs at intervals shorter than once every three to five years are not likely to support populations of alligators that are at or approaching restoration targets. Survey data, which encompass various hydrologic conditions, provide a comprehensive look at patterns of change and defines appropriate hydrologic targets for the alligator. The multi-year data indicate that multi-year hydroperiods are important in maintaining alligator populations and data support the hypothesis that dry downs (water levels 15 cm above ground) occurring an average of once every three to five years would be optimal. Repeated and intense dry downs affect the ability of alligators to reproduce if they occur during April/May and the survival of hatchling and juvenile alligators regardless of when they occur. Over the past ten years of alligator monitoring (2003–2012), the average number of alligators per kilometer in northern WCA 3A (Tower), northern Shark River Slough (Frog City), and also in WCA 2A (WCA2A) have decreased corresponding to undesirable hydrological conditions for alligators (**Figure 2-13**). Population trends were steady in central (HD) and southern (N41) WCA 3A and WCA 1, where the hydrology was more desirable. **<u>Recommendation</u>: Consider alligators' need for multi-year hydroperiods and dry downs to 15 cm occurring no more frequently than once every three to five years in future CERP project and operations planning efforts.**

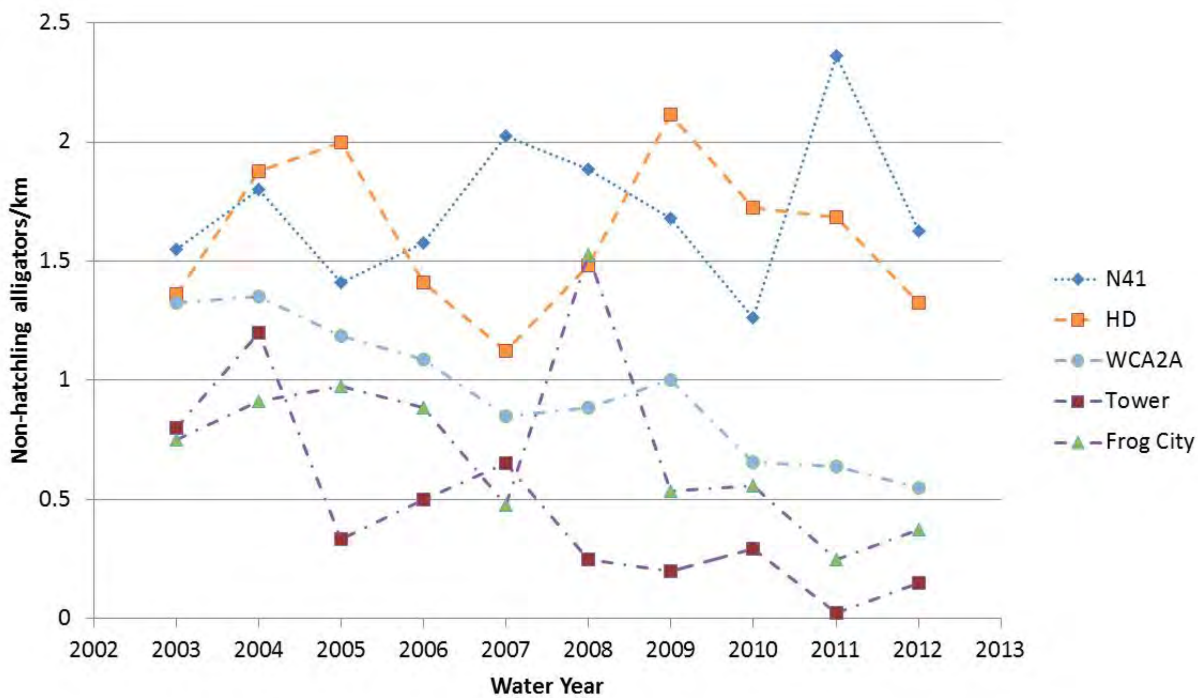


Figure 2-13. Average alligators per kilometer of two spring and two fall surveys by water year for five areas where alligators are monitored. WCA 1 is not included on the graph because densities are much higher (ranging from yearly averages of 4.0 to 8.2 alligators per kilometer).

New Science – Landscape – Ridge and Slough Patterning in Interior Everglades Peatlands

Good ridge and slough vegetation and landscapes were found at mean annual slough water depths of 20 to 50 cm, although a large proportion of sites near the drier end of this range are conspicuously degraded. **<u>Recommendation:</u> Maintain a spatially-averaged long-term mean annual water depth of 35 to 50 cm (measured from slough bottom) for areas in which ridge and slough structure is to be restored and/or conserved.** CEPP and CERP is expected to improve hydrologic conditions to maintain ridge and slough in central WCA 3A, improve ridge and slough in southern Shark River Slough, and start the long process to regain the ridge and slough landscape in northwestern WCA 3A, WCA 3B, and northeastern Shark River Slough.

New Science – Landscape – Surface-Groundwater Interactions in Everglades Tree Islands

Three main findings illustrate the mechanisms that lead to the degradation of tree islands and give insight into how tree islands might be restored. These findings are that healthy tree islands (1) exhibit spatial and temporal variability in plant water uptake that is focused in the shallow soil profile; (2) exhibit evapotranspiration that increases from wetter communities to drier communities within the tree island; and (3) actively accumulate ions in soil water of the drier, “high head” plant community. **<u>Recommendation:</u> Identify specific hydrologic criteria for plant performance, mineral retention, and organic matter accumulation across the region considering natural seasonal variation that can be used to update models for project and operations planning.**

New Science – Landscape – Hydrologic Driven Short-Term Vegetation Dynamics

Hydrology is a determining factor with respect to changes in both marshes and tree islands at the population, community, and ecosystem levels, and these responses are rapid (decadal). Fluctuations in the hydrologic regime from 1999 to 2012, resulted in below average water levels and shorter hydroperiods in Shark River Slough, promoted an increase in spikerush and sawgrass cover at the expense of open water sloughs and slough vegetation in the marshes, and the expansion of woody plants across the full suite of communities comprising the tree island gradient, i.e., bayhead forest, bayhead swamp, and sawgrass tail. In prolonged dry conditions, it is the progression towards sawgrass, and the establishment and growth of trees in the peat environment that drives successional processes towards the expansion, growth, and maturation of tree islands in the ridge and slough landscape. Reestablishment of historical hydrologic regimes is the primary goal of the ongoing restoration efforts under CERP. Changes in water management associated with restoration will likely result in changes in the balance and boundaries between herbaceous species and woody communities throughout the marshes within ridge-slough landscape, while in tree islands, the proportion of flood-tolerant and flood-intolerant woody species will change, resulting in a shift in species assemblages and tree island function.

Status and Trends – Nutrients – Periphyton Assessments (Total Phosphorus and Species)

A multi-metric (TP and periphyton community type) approach for WY 2011 showed periphyton alterations in central and southern WCA 3A, and indicated a community shift towards higher TP-tolerant periphyton communities along the eastern boundary of ENP. Single-metric (TP only) assessments for WY 2012 show similar trends in WCA 3A, and show increasing alteration in WCA 3B, but did not detect periphyton community changes along the ENP boundary (**Figure 2-14**). This data set demonstrates that 70% of the periphyton multi-metric condition downstream of inflow structures from WY 2007–WY 2011 can be explained by TP concentrations at the inflows, and suggests that legacy TP and local biogeochemical processes can account for the remaining variability. The strong relationship between periphyton metrics downstream of inflow structures and TP concentrations in inflows from WY 2007–WY 2011 points to the continuing relevance of nutrient inputs to Everglades ecology. However, the multi-metric periphyton assessment is sometimes inconsistent with flows and loads across basins, with very low and stable ambient levels in WCA 3A and WCA 3B, and with declining concentrations being seen widely in ENP. Full interpretation of the periphyton metric for marsh impairment must consider inflow and legacy TP, local biogeochemical processes, and other factors influencing periphyton ecology.

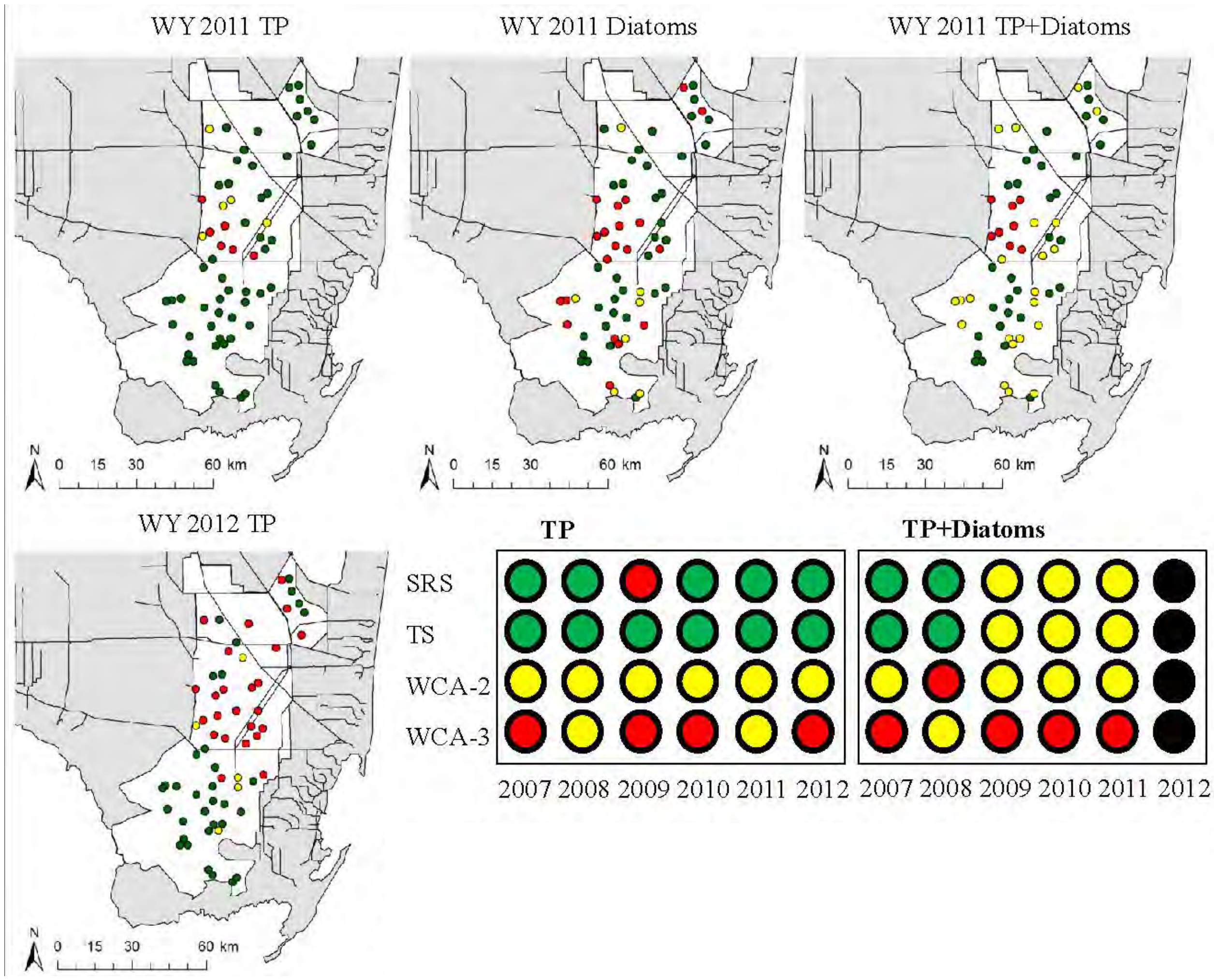


Figure 2-14. Baseline (green), cautionary (yellow), and altered (red) condition at each sampling unit during wet season sampling using the periphyton TP species and TP + diatom metric in WY 2011 and TP only in WY 2012. SRS = Shark River Slough; TS = Taylor Slough.

Status and Trends – Predator-Prey – Long-term Trends of Fish and Crayfish Biomass.

Long-term analysis of prey fish in WCA 3A, WCA 3B (16-year period of record), and ENP (31-year period of record) indicates the increased frequency of drying events led to decreased biomass of fish in these areas. This finding indicates that while the abundance of potential wading bird prey is necessary prior to making prey available through marsh dry downs that concentrate fish, it alone is not sufficient to assure abundant prey needed to sustain nesting. It is possible that the increased frequency of drying in most study sites makes prey more available in the short term (concentrated in shallow marsh pools to make wading bird feeding easier), while simultaneously depleting the store of prey regionwide and in subsequent years. Under these circumstances, wading bird consumption of prey resources could outpace the rate of prey replenishment. Wet season prey biomass contributes to, but does not alone explain, the formation of high quality prey patches. However, low prey abundance may hamper the formation of high quality dry season prey patches. Multi-decadal, slow declines in prey biomass is hypothesized to adversely affect dry season prey availability. **Recommendation: To recover historical nesting populations of wading birds, water management seasonal operations need to mimic natural cycles of rainfall and interannual variation both in space and time (i.e., rain-driven operations).**

Projects and Operations – Predator-Prey – Water Management and Wading Birds

The optimal water level recession rate for several wading bird species is between 5 millimeters (mm) per day and 7 mm per day and is part of Everglades Restoration Transition Plan operational rules. There is an interaction between the effects of wet season prey production and recession rates on prey availability. Recession rates have a greater effect on wading bird foraging when prey availability is low. Water level recession rate is critically important for wading bird nesting, and it is one of the most directly controlled hydrologic parameters that can be translated to water management actions. This finding has now been robustly documented in field studies and validated through more recent modeling. **Recommendation: Strive to meet optimal recession rates (5 to 7 mm per day), during the dry season without causing hydrologic reversals due to water management actions.**

Projects and Operations – Model Support for Routine Operations

Wading bird data were used to improve real-time assessments of foraging conditions in the Everglades and are used in weekly water management operational decisions by the South Florida Water Management District and United States Army Corps of Engineers in coordination with stakeholders. This is an example of effective environmental scientist coordination with water managers that has been beneficial to Everglades flora and fauna. In 1999, the South Florida Water Management District began to use a wading bird habitat suitability model to provide real-time assessments of the foraging conditions for wading birds in the Everglades. The model was updated with new information from MAP and other studies in 2008 to support water management operations decisions (**Figure 2-15**).

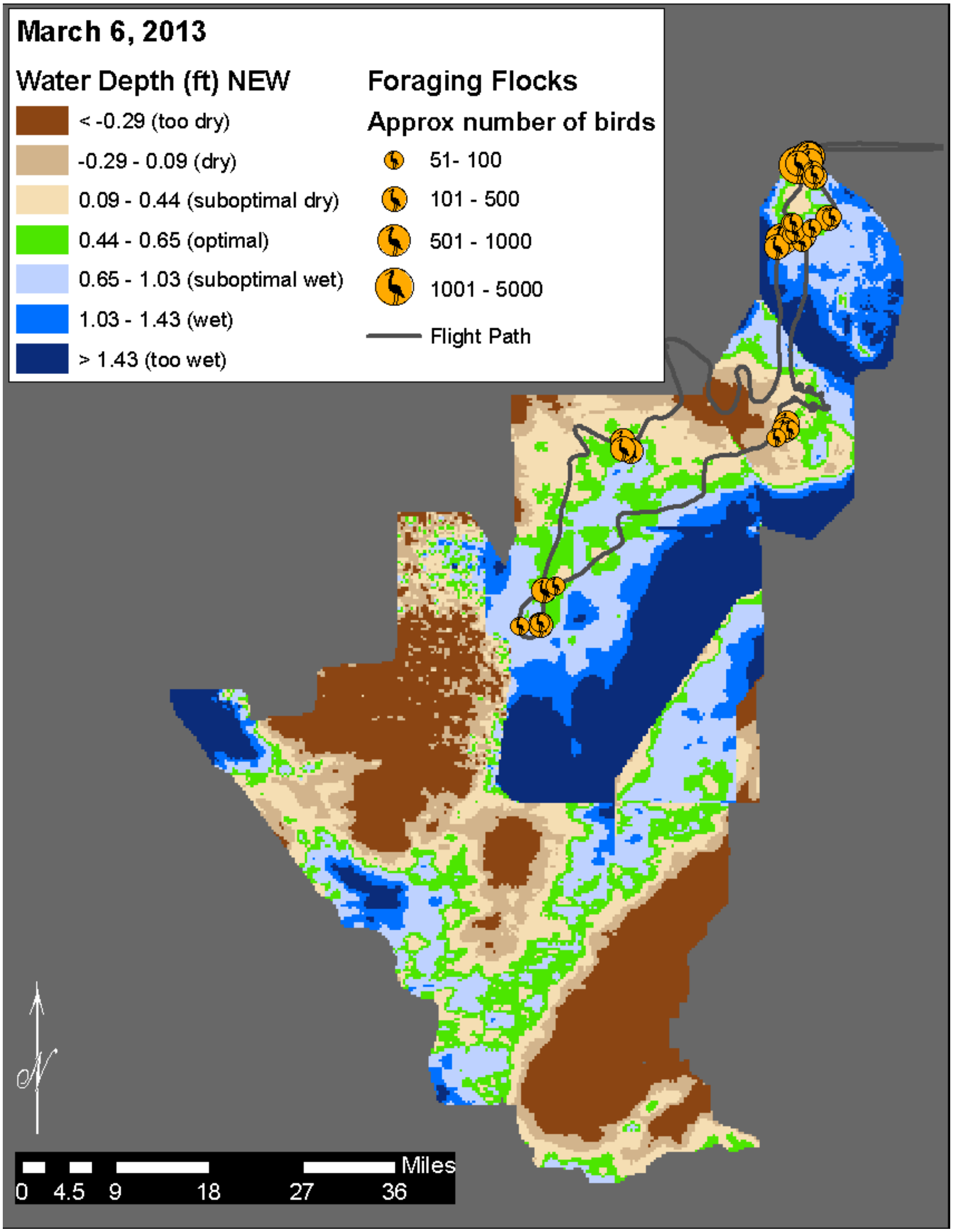


Figure 2-15. The wading bird model is used to illustrate where foraging conditions are good (green and light blue/brown) or poor (dark blue to dark brown) in the system to inform water management operations to help facilitate wading bird nesting success.

SOUTHERN COASTAL SYSTEMS

New Science – Seagrass and Spotted Seatrout

A strong positive linear relationship was found between the amount of seagrass habitat cover and the frequency of occurrence and abundance of juvenile sportfish (spotted seatrout and gray snapper). It was also found that spotted seatrout prefer moderate estuarine salinity (15 to 30). These findings suggest that restoring Florida Bay's estuarine salinity conditions, with the expectation that this will improve seagrass habitat, will substantially benefit sportfish populations (particularly spotted seatrout) and thus benefit recreational fishing in ENP.

New Science – Modeling

Recent refinements in paleoecological evaluations and related hydrologic modeling indicate that obtaining pre-drainage salinities in Florida Bay would require water levels in Shark River Slough to be 0.5 to 0.8 feet higher than the current observed levels (Figure 2-16). This would require about two times more flow than is currently being discharged (on average) under Tamiami Trail into Shark River Slough. Evaluations also indicate that flow of fresh water into Taylor Slough may be 3 to 4 times less than under pre-drainage conditions. These estimates will be used to develop the stage and flow performance measure that integrates Greater Everglades and Southern Coastal Systems hydrology.

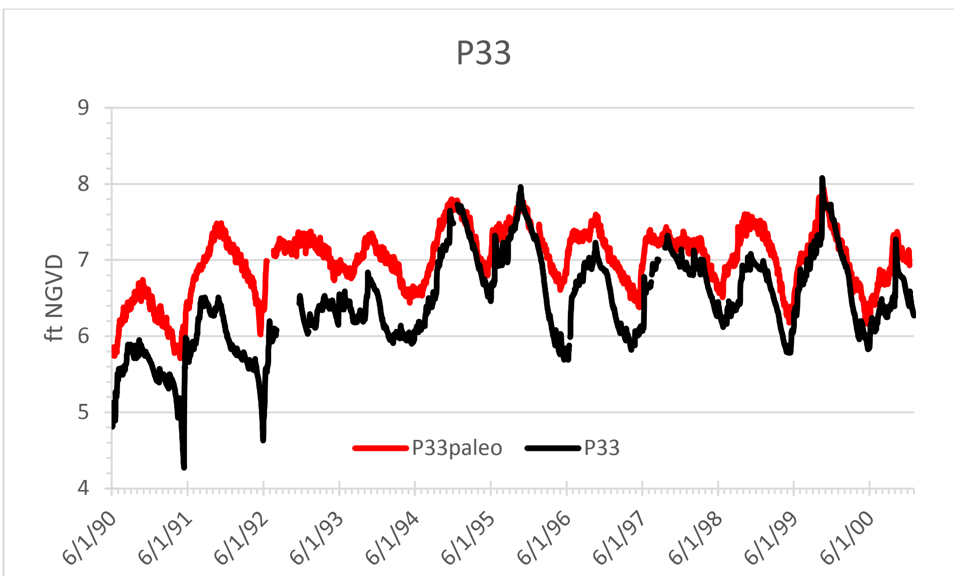


Figure 2-16. Comparison of paleoecological estimated pre-drainage stages at the P33 gage in the middle of Shark River Slough with current stage conditions.

Status and Trends – Salinity Conditions in Florida Bay and Biscayne Bay

Salinity increased in Florida Bay. Conditions were further away from the restoration target since the last SSR (WY 2009–WY 2012 compared to WY 2000–WY 2008) likely due to reduced rainfall that resulted in reduced flows from the water management system. **Operation of the expedited component of the BBCW Project yielded hydrologic improvements in the project area wetlands. In contrast, systemwide monitoring of downstream salinity conditions in southern Biscayne Bay have not shown any project-related effects; unfavorable conditions, compared to the target, persisted.** Without increased freshwater flow to maintain mesohaline (5 to 18) salinity, for at least part of the year, the desired shift to more productive estuarine flora and fauna will not be realized. **Recommendation: Incorporate downstream estuarine area responses in future CERP systemwide and project operations planning. In addition, development and refinement of salinity targets for Biscayne Bay and the lower southwestern coast is important to ensure all areas of the Southern Coastal Systems can be evaluated during future restoration and operations planning efforts.**

Status and Trends – Florida Bay Seagrass

Seagrass data indicate a declining trend in species diversity over the last decade. Turtle grass has increased in density; however, shoal grass has decreased baywide. While manatee grass increased on the western edge, it decreased in the interior of Florida Bay. Current patterns of seagrass distribution and abundance are similar to 1984 patterns that preceded a mass seagrass die-off in 1987. Large patches of dead seagrass were observed in May 2012 on western and central Florida Bay banks. This trend of decline in seagrass species diversity and increased vulnerability is expected to continue. A positive shift in seagrass community composition (increase in shoal grass and decrease in turtle grass presence) was observed in Madeira Bay, south of the Taylor Slough drainage area, and may see additional improvement as a result of the C-111 Spreader Canal Western project implementation. Factors other than salinity and freshwater inflow, such as nutrients, turbidity, and light penetration, are additional controlling factors on seagrass growth and diversity. Collection of additional data associated with seagrass growth and diversity controlling factors would help to determine the causes of current patterns in species abundance and diversity, and whether the pattern of seagrass diversity reduction is a normal recurring successional pattern.

Project and Operations – Florida Bay Roseate Spoonbills

Florida Bay roseate spoonbills have successfully nested in seven of the last eight years. Successful nesting is defined as at least 1 chick per nest. The increase in total nest numbers is due to favorable climate conditions and better environmental coordination with water managers. Roseate spoonbills are a CERP indicator species because their reproduction is dependent on aquatic food that abnormally fluctuates due to altered hydrologic and water quality conditions in the Greater Everglades wetlands. The population successfully nested in 7 out of the last 8 years, including the last 2 years (**Figure 2-17**) indicating an increasing population since 2011 (see the Southern Coastal Systems chapter). However, a two-year upswing in population should be viewed with caution. The recent positive trend is tied to two key factors: (1) generally favorable rainfall patterns, and (2) improved coordination between water managers and field biologists dating back to 2005, which has resulted in better hydrologic conditions. Perhaps just as important was the surprising recent finding that beginning in 2009–2010, spoonbills began nesting at a colony on the mainland near Taylor River that had been inactive since the early 1980s. In 2011–2012, the colony was accessed and surveyed twice, resulting in an estimate of 164 nests (nearly doubling the total nest count for Florida Bay) with a high degree of success (> 1 chick per nest). The 2012–2013 surveys indicate that the number of nests was at least as large as the 2011–2012 effort and nesting effort was successful.

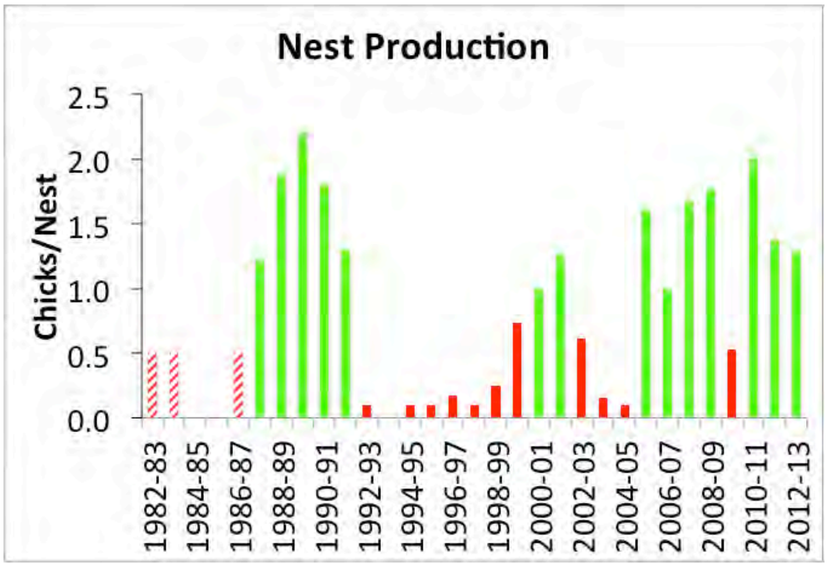


Figure 2-17. Spoonbill nest production (average number of chicks per nest) in northeastern Florida Bay since 1982–1983. No surveys were performed in 1984–1985, 1985–1986, and 1993–1994. Red indicates a failed nesting attempt (< 1 chick per nest), green indicates successful nest attempt. Cross-hatching indicates that the nesting attempt was recorded as failed; however, no numerical value was available.

Status and Trends – Florida Bay American Crocodile

American crocodiles continue to show a positive response to a non-CERP hydrologic restoration project in ENP. The plugging of canals near Cape Sable likely moved salinity regimes toward restoration targets, resulting in a concurrent increase in the crocodile encounter rate and crocodile nesting effort in that area (**Figure 2-18**). This finding is consistent with the CERP hypothesis that restoration of salinity regimes will benefit crocodiles, and it highlights a clear positive response of a CERP ecological indicator to hydrologic restoration efforts.

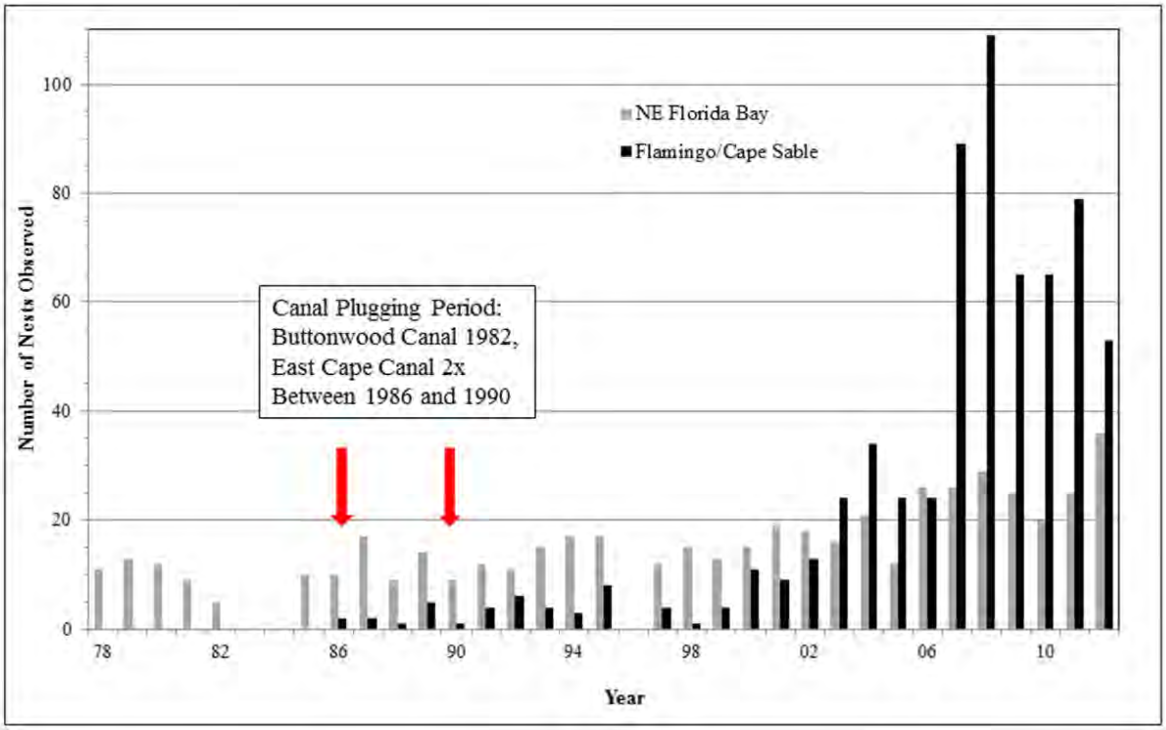


Figure 2-18. Number of American crocodile nests per year found in the Flamingo/Cape Sable area (Buttonwood and East Cape canals) and northeastern Florida Bay (Little Madeira and Joe bays) between 1978 and 2012. Crocodile nests were first discovered in the Flamingo/Cape Sable area after Buttonwood and East Cape Canals were plugged in the early 1980s. Since then, the number of nests per year has increased more rapidly in the Flamingo/Cape Sable area compared to northeastern Florida Bay. The lag in increase since canal plugging is due to time required for hatchlings from late 1980s cohorts to reach sexual maturity.

Status and Trends – Ten Thousand Islands Nekton and Salinity

Analysis of the nekton communities in Pumpkin and Faka Union bays compared to the reference site (Fakahatchee Bay) indicated the restoration target for nekton (Fakahatchee Bay conditions) may need updating. Nekton are free swimming organisms and generally move independent of currents (e.g., fish). Because historical nekton data were not available prior to watershed alteration, scientists used best professional judgment to set a target during Picayune Strand Restoration Project (PSRP) planning. Analysis of nekton community composition (2000–2009) revealed that current conditions in Pumpkin and Faka Union bays are 75% similar to those in Fakahatchee Bay. This indicates the current restoration target in the PSRP project implementation report (PIR)/environmental impact statement (EIS) (USACE and SFWMD 2004) may need further investigation. **Recommendation: Qualitative hydrologic and ecologic restoration targets in the PSRP final PIR/EIS should be reevaluated based on new information gained from RECOVER and other partner agency funded monitoring and research.**

RECOVER and the PSRP monitor changes to salinity and oysters in the main rivers associated with Faka Union, Pumpkin, and Fakahatchee bays to determine restoration success from the PSRP. However, **the current salinity monitoring network does not have the appropriate spatial and temporal resolution to document low salinity conditions in Faka Union Bay and high salinity conditions in Pumpkin Bay in comparison to the target salinity conditions in Fakahatchee Bay**. Without this information, it will be difficult, to determine restoration improvements in the salinity regimes and subsequently develop quantitative ecological restoration targets. **<u>Recommendation</u>: In order to assess post construction improvements, additional monitoring of salinity in Fakahatcheee, Faka Union, and Pumpkin bays is needed to assess PSRP restoration success in the estuary.**

Status and Trends – Water Quality

Although there has not been a significant trend in overall water quality since the 2009 SSR, water quality along the Southwest Florida Shelf and in southeastern Everglades has degraded. Along the Southwest Florida Shelf there has been a significant downward trend over the past five years with a noticeable decline in WY 2012. There were also marked increases in chlorophyll *a* concentrations in waters adjacent to the southeastern Everglades (northeastern Florida Bay; and Barnes Sound/Manatee Bay/Blackwater Sound), as well as in southern Florida Bay, in WY 2011 through WY 2013. These changes may be related to climatic patterns, with unusually high dry season rainfall in recent years.

Projects and Operations – Biscayne Bay Coastal Wetlands Project, Expedited Portion

Deering Estate Flow-way Component: **Monitoring results demonstrate a clear improvement of hydrologic conditions in response to the S-700 pump station operations and flow-way in the Deering Estate.** With pumping at 100 cfs, the wetland inundation is at 95% of the target (**Figure 2-19**). Vegetation in Cutler Slough already appears to be responding to the changed hydrology.

L-31E Flow-way Component: **Water diversions are well short of the target and downstream hydrologic and ecologic responses have not been detected.** For WY 2012 and WY 2013 combined, approximately 10,000 acre-feet is estimated to have been diverted into the wetlands east of the L-31E Canal, which is short of the flow target of 46,000 acre-feet (L-31E culverts combined). This diverted water previously would have been released at canal mouths as damaging point source discharge to the bay. **<u>Recommendation:</u> Provide additional freshwater (3 months or more) to limit the occurrence of high salinity conditions during the dry season and support the establishment of mesohaline (5 to 18 psu) salinity conditions with the onset of the wet season. Modeling is needed to inform these operational changes consistent with project and regional operational plans.**

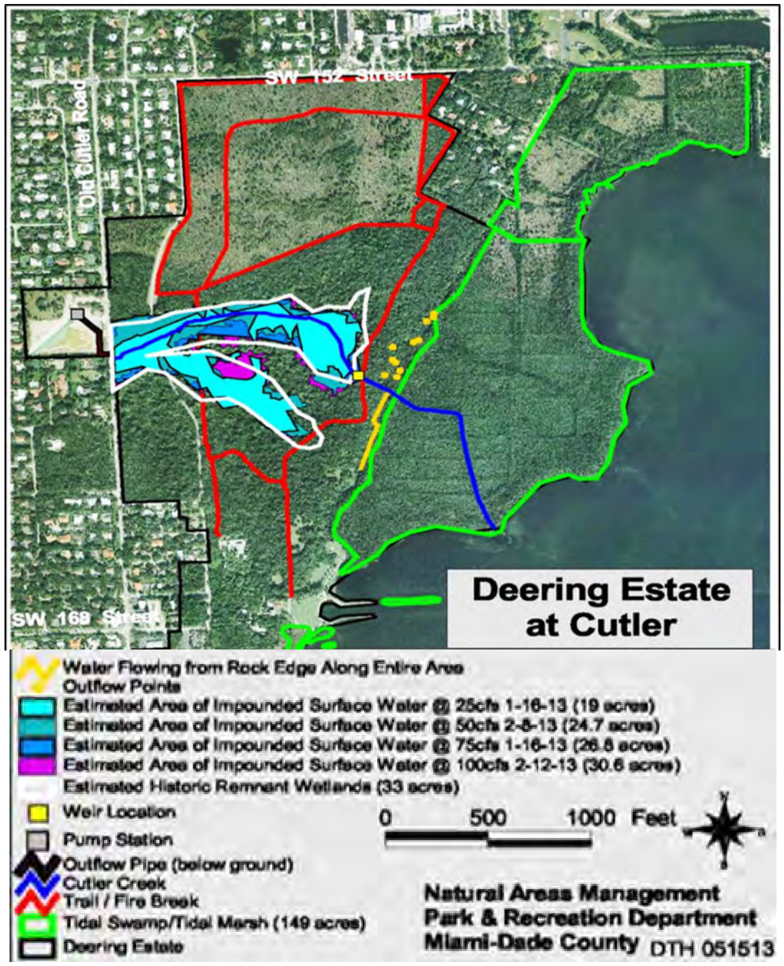


Figure 2-19. Delineation of the historical freshwater wetland slough in Deering Estate (white polygon) and areas of inundation at different pump rates.

Projects and Operations – Picayune Strand Restoration Project

PSRP construction has progressed significantly. Positive ecological responses are being realized with changes in the hydrology and vegetation of the areas near the back-filled Prairie canal. Water levels in wells near the filled Prairie Canal were consistently higher than those in the well near the unfilled Merritt Canal (usually by about 2 feet during the dry season and by 1 to 2 feet in the wet season). Vegetation had a corresponding positive response to hydrologic improvements. Vegetation index values in restoration areas (cypress swamp sampling plots) are converging on target values (reference plots in Fakahatchee Strand). **<u>Recommendation</u>: Implement the plugging of the Miller, Merritt, and Faka Union canals and implement the project's operational plan to improve water levels and freshwater ecology in the Picayune Strand area. These upstream improvements will help restore the estuarine waters south of Picayune Strand, where the canals presently cause salinities in some downstream estuaries to be high and others to be low in relation to expected natural conditions.**

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CHAPTER 3

SYSTEMWIDE SCIENCE

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TABLE OF CONTENTS

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CHAPTER 3 SYSTEMWIDE SCIENCE

INTRODUCTION

This chapter covers many of the drivers and stressors that affect changes in the south Florida ecosystem, which includes the Northern Estuaries (NE), Lake Okeechobee (LO), Greater Everglades (GE), and Southern Coastal Systems (SCS) and described in the Total System Conceptual Ecological Model (CEM) (Ogden et al. 2005). The following sections are included in this chapter:

- **Systemwide Hydrology –** Hydrology trends ("stressors") can change ecological effects and attribute status monitored by the Restoration Coordination and Verification Program (RECOVER) Monitoring and Assessment Plan (MAP). Rainfall averages and flows into and out of each south Florida basin are summarized for 2009–2013 compared to historical averages to provide a context for hydrologic trends across the whole system.

- **Climate Change –** A summary of climate change effects as a major driver of ecosystem change in the context of Everglades restoration is provided. Climate change effects have a potential to greatly affect all MAP ecosystem restoration indicators and Comprehensive Everglades Restoration Plan (CERP) restoration success. Although the MAP does not directly fund climate change monitoring, climate change science for the Everglades ecosystem has been synthesized through several efforts and summarized in this document.

- **Invasive Species –** An overview of invasive species monitoring and management in the context of Everglades restoration is presented. Invasive species are an emerging driver of change that has significant implications for MAP ecsoystem restoration indicators. Science funded by various south Florida agencies is summarized in **Appendix 3-1** to better understand how invasive species are being addressed.

- **Fire –** An assessment of fire history in Everglades National Park (ENP) and Big Cypress National Preserve (BCNP) is included to cover the approach to assessing the fire stressor in one portion of the GE region. Fire has important implications for the GE and SCS landscapes and vegetation changes monitored by the MAP, as well as achieving CERP restoration success.

- **Systemwide Ecological Indicators –** The ecosystem indicator (ecological effects and attributes) response to hydrology and other drivers/stressors as explained in **Figure 3-1** on the next page is covered by each region in more detail but is summarized overall in this section by referencing the South Florida Ecosystem Restoration Task Force's Science Coordination Group's *System-wide Ecological Indicators for Everglades Restoration 2012* report (Brandt et al. 2012). The report includes many of the CERP MAP (RECOVER 2009) restoration indicators and several others not considered by the MAP for Water Year (WY) 2010 to WY 2012. (A water year begins on May 1 and ends on April 30 of the following year, i.e., WY 2013 began on May 1, 2012 and ended on April 30, 2013).

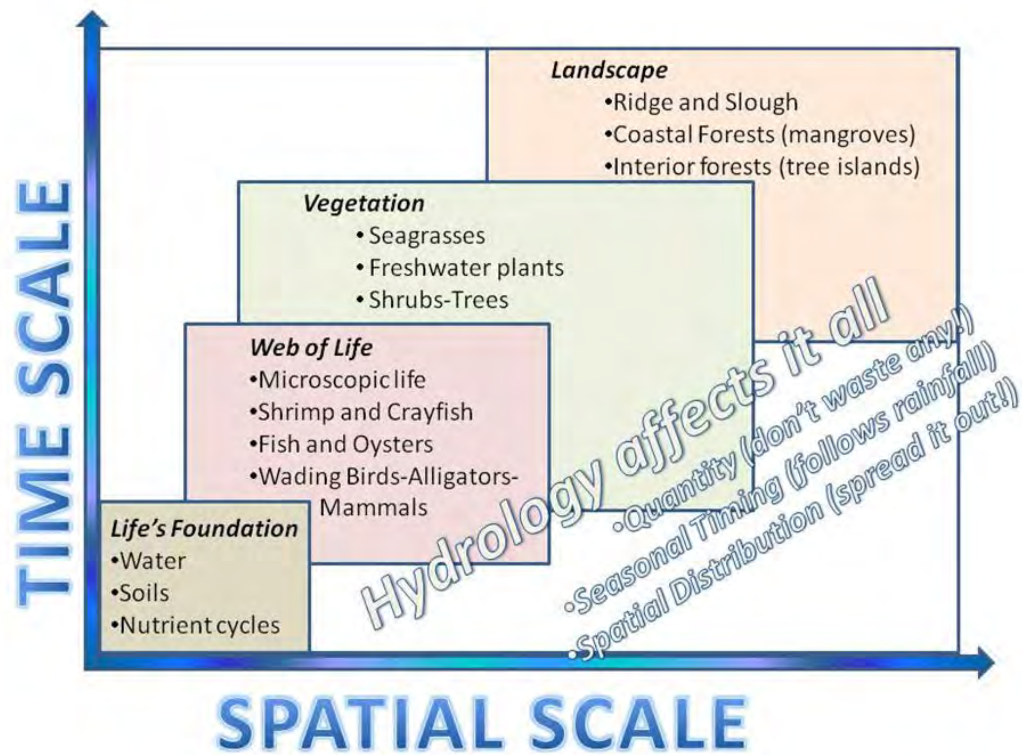


Figure 3-1. Conceptual relationship between hydrology and ecosystem attributes.

- **Wading Birds –** A summary on wading birds' status is provided for the whole system based on the larger 2013 *South Florida Wading Bird Report* developed by the South Florida Water Management District (SFWMD) (Cook 2013). Wading birds are good indicators of overall system health. More detailed wading bird information for the LO and GE regions are contained in Chapter 7: Greater Everglades.

- **Adaptive Management –** This section focuses on the most recent guidance on how to implement adaptive management for CERP, and project and program-level adaptive management plans that have been developed since the *2009 System Status Report* (SSR) (RECOVER 2011a). It identifies how the MAP monitoring and SSR reporting is necessary to understand restoration performance, address priority uncertainties about achieving restoration goals and objectives, and inform managers and stakeholders on restoration success and performance issues that require adjustments to improve CERP implementation. This section also identifies the current restoration projects that depend on the MAP monitoring and reporting to help answer questions specific to project restoration effects.

- **Interim Goals Review –** This section contains a review of information related to interim goals to help evaluate restoration success, progress in implementing adaptive management, and the sustainability of the MAP monitoring program. Interim goals were developed in 2007 as a means by which CERP restoration success could be evaluated at specific intervals to inform agency managers. The summary of interim goals in this SSR reviews the current MAP's ability to monitor and assess them and also documents the development of new ecological models to help inform interim goals.

- **MAP Sustainability -** The last section contains an update on the current sustainability of the MAP in achieving its original goals for CERP. It explains the value of the MAP monitoring that remains and the impacts of what was lost after funding cuts in Fiscal Year (FY) 2012 (October 1, 2011–September 30, 2012).

SYSTEMWIDE HYDROLOGY

South Florida was mostly drier than normal (below historic rainfall average) from WY 2009 to WY 2013. WY 2009, WY 2011, and WY 2012 were dry years with below average rainfall over the SFWMD area. Average annual rainfall over the SFWMD area is 52.8 inches.

Table 3-1 depicts water year rainfall on each SFWMD rainfall area and the ENP as reported in Chapter 2 of the *South Florida Environmental Report* since 2010 (Abtew et al. 2010, 2011, 2012, 2013, 2014). Generally, surface water flows reflect the water year rainfall condition. But, water management decisions such as releasing or holding water in storage and other factors may not show rainfall and flow correspondence in some basins. Also, stage at the end of the previous water year influences stage for the following water year (**Table 3-2** and **Figure 3-2**).

Table 3-2 depicts water year flows for major hydrologic units from WY 2009 to WY 2013 as reported in the respective *South Florida Environmental Reports* (Abtew et al. 2010, 2011, 2012, 2013, 2014). **Table 3-3** depicts average flows from WY 2009 to WY 2013 and historical average flows (1972–2013). **Figure 3-3** depicts average flows from WY 2009 to WY 2013 for the major hydrologic units with arrow width relative to flow amount. Outflows include regulatory releases, water supply and other purpose releases where applicable.

In summary, the past five water years (WY 2009–WY 2013) have been drier (see **Table 3-1** above) overall, which has helped reduced high stages in LO and lessened high flow events in NE but also continued to negatively impact the GE and SCS regional hydrology. In the north, there have been improvements that likely indicate how the ecosystem will respond in the future once CERP restoration projects are in place to maintain ecosystem resilience in all climatic periods (sequence of wet years or dry years). In the south, the drought has continued to impact the GE wetlands hydrology, ecosystem characteristics, flora, fauna, and landscape types, as well as salinity trends and ecology in the SCS region. The details of south Florida ecosystem trends and restoration project improvements are captured in the regional chapters of this SSR.

Table 3-1. Water year rainfall on SFWMD rainfall areas and ENP. [a, b]

Area	WY 2009	WY 2010	WY 2011	WY 2012	WY 2013	WY 2009–WY 2013 Average	Historical Average (1900-1995) [c]
Upper Kissimmee	40.06	70.02	40.63	46.7	47.14	48.91	50.09
Lower Kissimmee	44.57	52.96	38.43	44.29	52.33	46.52	44.45
Lake Okeechobee	37.84	50.27	33.84	36.55	47.22	41.14	45.97
East Everglades Agricultural Area	44.41	61.26	39.6	45.24	51.63	48.43	53.48
West Everglades Agricultural Area	44.44	70.34	38.91	47.13	53.34	50.83	54.95
Water Conservation Areas 1 and 2	47.4	65.63	43.83	53.87	62.8	54.71	51.96
Water Conservation Area 3	44.29	60.6	40.91	55.09	56.15	51.41	51.37
Martin/St. Lucie	45.08	56.18	36.27	48.94	51.84	47.66	54.14
Palm Beach	47.3	66.39	40.26	51.76	62.6	53.66	61.54
Broward	46.87	74.7	46.66	60.92	61.23	58.08	58.13
Miami-Dade	47.54	65.09	50.51	61.44	66.36	58.19	57.11
East Caloosahatchee	46	61.61	38.45	47.39	48.37	48.36	50.68
Big Cypress Preserve	49.45	62.05	40.87	53.02	54.03	51.88	54.12
Southwest Coast	50.9	63.69	43.24	52.46	50.82	52.22	54.12
ENP	42.6	60.45	49.32	53.82	51.36	51.51	55.22
Totals	45.24	61.43	40.36	49.1	53.17	49.86	52.75

a. Source: Abtew et al. (2010, 2011, 2012, 2013 and 2014).

b. Color coding: dark red ≤ -10%; orange -10% < to ≤ -5%; no color -5% < to < +5%; light blue ≤+5% to < +10%; dark blue ≥ +10%.

c. Source: Ali and Abtew (1999).

Table 3-2. Water year flows through major hydrologic units. [a]

Water Body	Flow (acre-feet)					
	WY 2009	WY 2010	WY 2011	WY 2012	WY 2013	WY 2009–WY 2013 Average
Lake Kissimmee Outflows	494,638	1,307,625	464,320	813,987	439,653	704,045
Lake Istokpoga Outflows	289,438	180,353	122,298	228,042	280,544	220,135
Lake Okeechobee Inflows	2,090,775	2,400,337	847,538	1,821,336	2,100,036	1,852,004
Lake Okeechobee Outflows	1,141,084	554,299	1,563,628	746,499	1,041,902	1,009,482
St. Lucie Canal Inflows (S-308)	172,602	75,928	285,379	47,201	103,622	136,946
St. Lucie Canal Outflows (S-80)	164,506	130,693	268,794	119	152,722	143,367
Caloosahatchee Inflows (S-77)	375,722	198,633	586,309	180,461	501,374	368,500
Caloosahatchee Outflows (S-79)	1,016,333	1,087,299	1,141,054	598,840	1,137,904	996,286
Water Conservation Area (WCA) 1 Inflows	336,293	310,183	152,641	170,256	363,897	266,654
WCA 1 Outflows	334,724	521,037	217,410	14,812	483,713	314,339
WCA 2 Inflows	905,864	1,299,071	466,619	386,176	1,074,320	826,410
WCA 2 Outflows	737,421	884,433	419,808	378,071	938,199	671,586
WCA 3 Inflows	1,212,107	1,470,963	722,267	899,567	1,322,042	1,125,389
WCA 3 Outflows	1,556,182	1,137,001	826,206	571,304	1,225,088	1,063,156
ENP Inflows	1,392,932	1,355,548	935,389	744,176	1,496,719	1,184,953

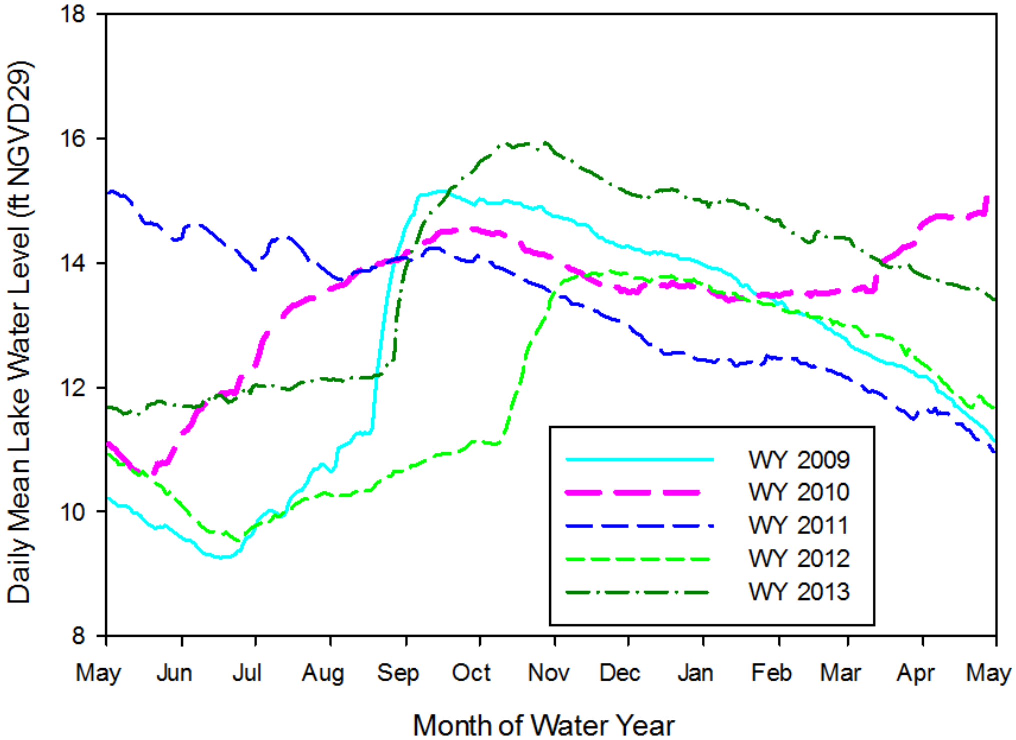
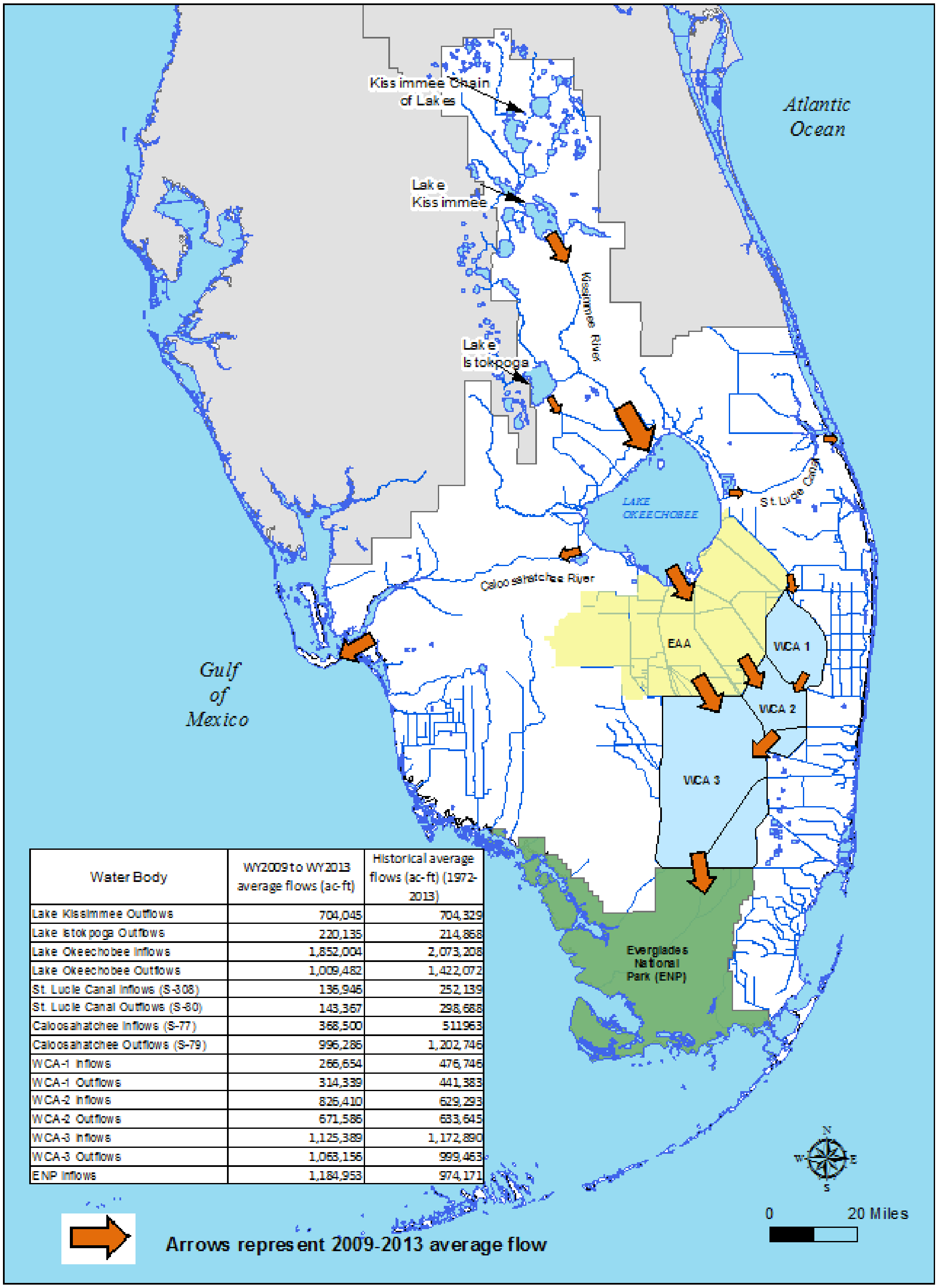


Figure 3-2. Daily mean water level for Lake Okeechobee (WY 2009–WY 2013). (Note: ft NGVD29 – feet National Geodetic Vertical Datum of 1929.)

Table 3-3. Average flows through major hydrologic units (WY 2009–WY 2013) and historical average flows (1972–2013).

Water Body	WY 2009 to WY 2013 Average Flows (acre-feet)	Historical Average Flows (1972–2013) (acre-feet)
Lake Kissimmee Outflows	704,045	704,329
Lake Istokpoga Outflows	220,135	214,868
Lake Okeechobee Inflows	1,852,004	2,073,208
Lake Okeechobee Outflows	1,009,482	1,422,072
St. Lucie Canal Inflows (S-308)	136,946	252,139
St. Lucie Canal Outflows (S-80)	143,367	298,688
Caloosahatchee Inflows (S-77)	368,500	511,963
Caloosahatchee Outflows (S-79)	996,286	1,202,746
Water Conservation Area (WCA) 1 Inflows	266,654	477,534
WCA 1 Outflows	314,339	443,383
WCA 2 Inflows	826,410	631,274
WCA 2 Outflows	671,586	637,225
WCA 3 Inflows	1,125,389	1,172,139
WCA 3 Outflows	1,063,156	1,004,772
ENP Inflows	1,184,953	977,031



Water Body	WY2009 to WY2013 average flows (ac-ft)	Historical average flows (ac-ft) (1972-2013)
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St. Lucie Canal Outflows (S-80)	143,367	298,688
Caloosahatchee Inflows (S-77)	368,500	511963
Caloosahatchee Outflows (S-79)	996,286	1,202,746
WCA-1 Inflows	266,654	476,746
WCA-1 Outflows	314,339	441,383
WCA-2 Inflows	826,410	629,293
WCA-2 Outflows	671,586	633,645
WCA-3 Inflows	1,125,389	1,172,890
WCA-3 Outflows	1,063,156	999,463
ENP Inflows	1,184,953	974,171

Figure 3-3. Average flows from WY 2009 to WY 2013 and historical average flow (1972–2013). (Note: ac-ft – acre-feet; EAA – Everglades Agricultural Area; WCA – Water Conservation Area; arrow size is commensurate with flow amount.)

CLIMATE CHANGE

Climate Change Overview

Change in the global climate system, i.e., climate change, is included in the 2014 SSR to recognize it as an emerging major driver of ecosystem change. Climate change must be addressed to accomplish CERP goals of restoring a healthy Everglades ecosystem and sustaining its unique plant and wildlife diversity for the benefit of future generations. Current understandings about global and regional climate trends relevant to south Florida are identified in this report. A range of possible future scenarios are also identified as a framework for continuing discussions on advanced planning, performance monitoring and adaptive management actions that can help increase the resiliency of Everglades restoration efforts now and in the future.

Global Climate Trends

There is growing recognition and acceptance that Earth's global climate is warming, that the rate of warming is likely to accelerate, and increasingly warmer global conditions are likely to persist well beyond 2100. A widely recognized international information source, the Intergovernmental Panel on Climate Change, periodically synthesizes climate change related peer reviewed research. The Intergovernmental Panel on Climate Change's fifth assessment report, *Climate Change 2013*, will be published in four volumes, starting with *The Physical Science Basis*, which was released on September 27, 2013 (IPCC 2013; http://www.ipcc.ch/). Key findings from the *The Physical Science Basis* are listed below:

- **Observed Changes in the Climate System –** Warming of the climate system is unequivocal, and since the 1950s, many of the observed changes are unprecedented over decades to millennia. The atmosphere and oceans have warmed, the amounts of snow and ice have diminished, sea level has risen, and the concentrations of greenhouse gases have increased, resulting in ocean acidification.

- **Quantification of Climate System Responses –** Observational and model studies of temperature change, climate feedbacks and changes in the Earth's energy budget together provide confidence in the magnitude of global warming in response to past and future forcing. Models have improved to reproduce observed continental-scale surface temperature patterns and trends over many decades, including the more rapid warming since the mid-twentieth century and the cooling immediately following large volcanic eruptions (very high confidence). It is extremely likely that the human influence has been the dominant cause of the observed warming since the mid-twentieth century.

- **Future Global and Regional Climate Change –** Continued emissions of greenhouse gases will cause further warming and changes in all components of the climate system. Global surface temperature change for the end of the twenty-first century is likely to exceed +1.50 degrees Celsius (°C) relative to 1850–1900. The contrast in precipitation between wet and dry regions and between wet and dry seasons will increase, although there may be regional exceptions. The global ocean will continue to warm during the twenty-first century, affecting ocean circulation. Global mean sea level will continue to rise during the twenty-first century and will very likely exceed that observed during 1971–2010 due to increased ocean warming and increased loss of mass from glaciers and ice sheets. Climate change will affect carbon cycle processes in a way that will exacerbate the increase of carbon dioxide in the atmosphere (high confidence). Most aspects of climate change will persist for many centuries even if the carbon dioxide emissions are stopped.

Observed Regional Climate Trends for the Southeastern United States

The United States Global Change Research Program was authorized in the Global Change Research Act of 1990 to "assist the nation and the world to understand, assess, predict, and respond to human-induced and natural processes of global change." The program promotes strategic coordination of federal global change research by thirteen federal agencies and submits the *National Climate Assessment* report every four years to the United States President and Congress. The current *Draft National Climate Assessment* report (U.S. Global Climate Change Research Program 2013; http://www.globalchange.gov/what-we-do/assessment/draft-report-information.html) analyzed data drawn primarily from the National Weather Service's Cooperative Observer Network, which has been in operation since 1895. Regional trends anticipated for the southeastern United States indicate that the region has globally not exhibited an overall warming trend in surface temperature over the twentieth century. However, temperatures in recent years (since the 1970s) have steadily increased across the region, with the most recent decade (2001–2010) being the warmest on record. For the southeastern region as a whole, long-term trends in precipitation show a statistically significant upward trend in fall and a slight downward trend in summer. The number of extreme hot days has tended to decrease or remain the same, while the number of warm summer nights has increased. The number of extreme cold days has decreased across the region. Year-to-year variability in precipitation has increased over the last several decades across much of the region, with more exceptionally wet and dry summers. The frequency of extreme precipitation events has been increasing across the region, particularly in the past two decades.

Future Climate Scenarios for the Southeastern United States and South Florida

Some experts believe large grid-size global and regional climate models used for early National Climate Assessment studies may not have sufficient detail to adequately simulate the complex weather patterns over peninsular Florida, particularly south Florida, which is bounded on three sides by the Gulf of Mexico and Atlantic Ocean. In the absence of such refined climate models, SFWMD has completed a preliminary screening level analysis using multiple runs of the South Florida Water Management Model (SFWMM) using a range of future conditions plausible for 2060 in south Florida. These include +18 inches of sea level rise, +1.5° C temperature (for increased evapotranspiration), and a range of changes in historic hydrologic patterns (rainfall change = 0%, +10% and -10%). A February 14–15, 2013 Climate Change in South Florida technical workshop sponsored by Florida Atlantic University, United States Geological Survey (USGS) and National Oceanic and Atmospheric Administration (NOAA) through the Florida Sea Grant program discussed the model outputs for these scenarios, along with CERP performance indicators. The workshop presentations are available online at the Florida Center for Environmental Studies web pages at http://www.ces.fau.edu/climate_change/ecology-february-2013/Sessions-Presentations.php. A summary of conference information indicates the following potential effects with this particular scenario: 18 inches of sea level rise, 1.5° C temperature increase and 10% decreased rainfall with the following water management infrastructure and operation changes based on a 2010 baseline:

- **Lake Okeechobee –** Two meter decrease in average high and low water levels over multiple years would make marsh lands dry out and be prone to fires, vegetation die off, and would dramatically shape lake ecology.

- **Freshwater Wetlands –** Major impacts will occur to soils, vegetation, fish, wildlife, and invasive species. In addition there will be reduced peat production, and increased rate of peat loss and fire risk. Aquatic fish production would likely decrease affecting wading birds and alligators. Plant species and community responses would likely continue to decline due to changes in the range of water level fluctuations over a wet-dry cycle. Soil biogeochemical process would likely result in more soil oxidation, release of mercury and sulfate, and an increase of methyl mercury that could have sublethal effects on some faunal species.

- **Coastal and Marine Ecosystems –** Florida Bay salinity will become more like the ocean with 1.5 feet of sea level rise. Higher summer temperatures may negatively affect seagrass habitat, fish, and wildlife, coral will continue to be impacted, mangroves will continue to migrate inland, and peat loss in freshwater wetlands due to saltwater intrusion may result in additional nutrient impairments to Florida Bay.

The results of the above investigations are being published in a special issue of *Environmental Management* expected to be released in 2014. Further work will be needed to improve the scenario projections and the impact analyses.

Future Sea Level Change Scenarios for South Florida

United States Army Corps of Engineers (USACE) guidance requires consideration of scenarios for three possible future conditions representing a range of future rates of sea level change from a continuation of the historic global rate of approximately 0.2 meter (m) by 2100 up to a high rate of 1.5 m by 2100. The USACE does recognize 2.0 m as the credible upper limit for sea level change by 2100, and if requested by local interests, will include that in scenario analyses for long-term investments that are sensitive to sea level change (USACE 2011). USACE and NOAA guidance (NOAA 2012) requires consideration of local relative sea level change, which combines global sea level change plus local vertical land movement (uplift or subsidence), which is assumed to continue indefinitely at a constant rate. **Table 3-4** is a list of NOAA tide stations in south Florida that have periods of record long enough to be useful for developing sea level change scenarios per USACE and NOAA guidance. Key West has the longest continuous period of record for tide stations in Florida. The rate of relative sea level change at Key West is close to the rates at other south Florida tide stations, so Key West will be used as a proxy for all CERP sea level change scenarios. **Figure 3-4** provides a visual comparison of USACE and NOAA sea level change scenarios for Key West, Florida. **Table 3-5** includes the calculated USACE and NOAA sea level change values for Key West, Florida.

Table 3-4. Comparison of relative sea level change at sites in Florida. [a]

Tide Station – Number and Name	Analysis Period of Record	Sea Level Rise [b] (millimeter per year)
8721120 Daytona Beach (Inactive)	1925 to 1983	2.32
8723170 Miami Beach (Inactive)	1931 to 1981	2.39
8723970 Vaca Key (< 40 years)	1971 to 2006	2.90
8724580 Key West (1913 to present)	1913 to 2006	2.20
8724110 Naples	1965 to 2007	1.97
8725520 Fort Myers	1965 to 2007	2.32

a. From NOAA 2013.

b. Relative sea level change = estimated global sea level trend (1.7 millimeter per year) + local vertical land motion.

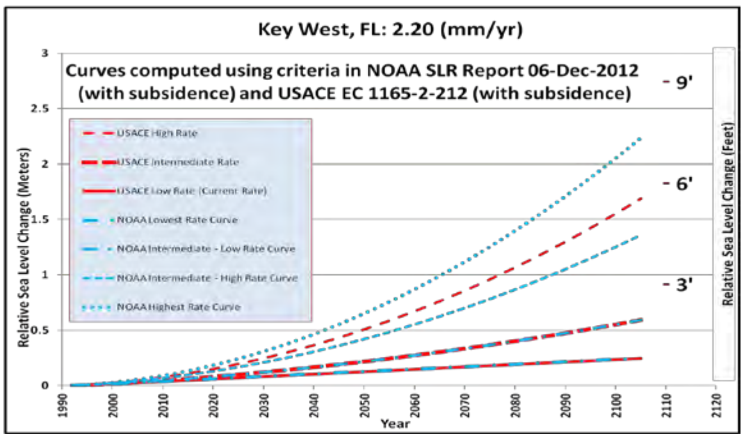


Figure 3-4. Sea level change scenarios approved by USACE (2011) and NOAA (2012). (Note: mm/yr – millimeters per year.)

Table 3-5. Comparison of USACE and NOAA sea level increases for different sea level change scenarios.

Year	USACE NOAA Low	USACE Intermediate NOAA Intermediate Low	NOAA Intermediate High	USACE High	NOAA High
Scenario	**Local Historic Relative Sea Level Rise**	**Global Sea Level Rise +0.5 m by 2100**	**Global Sea Level Rise +1.2 m by 2100**	**Global Sea Level Rise +1.5 m by 2100**	**Global Sea Level Rise +2.0 m by 2100**
1992	0.0	0.0	0.0	0.0	0.0
2010	0.1	0.2	0.2	0.3	0.3
2060	0.5	0.9	1.8	2.2	2.9
2100	0.8	1.8	4.1	5.1	6.7
2110	0.9	2.1	4.8	6.0	8.0
2120	0.9	2.4	5.6	7.0	9.3

Notes: USACE projections are for historic, modified National Research Council (NRC) Curve I and modified NRC Curve III rates of sea level change developed for Key West, Florida per USACE (2011). The existing condition is based on guidance in NRC (1987). The projections are developed using the local historic rate of sea level rise at Key West as reported by NOAA (2012)—2.20 millimeters per year. NOAA projections use the same existing condition equations modified for different global sea level rise scenarios. The NRC, USACE and NOAA guidance documents do not address dates beyond 2100. All projections start from 1992 control for the national survey datum.

How Does Climate Change Affect Restoration Efforts and Indicators?

Ongoing change in the global climate system including natural variability is a key driver impacting various stressors—precipitation, temperature, evapotranspiration, sea level rise, tropical storms and fire—that effect both natural and human processes and ecosystems in south Florida. A key concept is that "stationarity is dead", meaning the era of climate conditions where natural systems fluctuate within an unchanging envelope of variability is over. Restoration of conditions similar to pre-development Everglades ecosystem conditions (circa 1900 or earlier) in the remaining Everglades natural areas is still achievable, but gradual long-term shifts in the location of some Everglades ecosystem habitats and functions is unavoidable in order to sustain them for future generations. Historic climatic conditions are no longer a reliable indicator of future climatic conditions, and changing conditions create stresses that drive adaptation. This concept has implications for measuring restoration success, as well as informing project planning, design, and operations to achieve CERP goals and objectives.

Everglades monitoring data plus related records from SFWMD and others might provide more detailed local information on south Florida weather patterns in recent years. That additional data since the 2009 SSR will be too short to establish an unquestioned trend, but it may be possible to review recent data in conjunction with existing longer data sets to identify changes such as more frequent extreme rainfall events that are consistent with anticipated climate change trends. Continuing efforts to understand the skills of both global and regional models for south Florida and their projections are needed. A cost-effective way to do this would be for RECOVER to work with collaborative efforts such as the Florida Climate Institute to consult on climate change and sea level rise projections. RECOVER will also need to consider how components of the Everglades ecosystem may respond to the new stresses, and perhaps to identify refinements in the monitoring and assessment program to better understand how to help improve conditions for successful and sustainable adaptation.

Climate change scenarios should be considered in future efforts by RECOVER, in coordination with the South Florida Ecosystem Restoration Task Force, to update CERP interim goals and interim targets documents (USDOA and State of Florida 2007, USDOA et al. 2007, RECOVER 2005; http://www.evergladesplan.org/pm/recover/igit_subteam.aspx) and performance measures (http://www.evergladesplan.org/pm/recover/eval_team_perf_measures.aspx). Restoration success metrics that incorporate climate change considerations serve as a framework to help develop long-term climate adaptation strategies as part of CERP restoration efforts. Presentations from the February 14–15, 2013 South Florida Climate Change Workshop and subsequent papers include information that will also be useful in informing interim goals and interim targets and appropriate performance measure updates that are beyond the scope of the 2014 SSR.

USACE Civil Works projects, like the Central Everglades Planning Project (CEPP), are required to analyze the effects of sea level change on achieving project benefits. The CEPP sea level change analysis concludes that relative to the without project condition, there is a loss of freshwater wetland benefits up to 12% in the southern part of ENP-south and ENP-southeast zones as increased salinity causes peat soil collapse and a shift from freshwater vegetation to saline tolerant vegetation (see http://www.evergladesplan.org/pm/projects/project_docs/pdp_51_cepp/dpir/082813_cepp_dpir_anne x_i_seal_level_rise.pdf for more information). However, the newly created saltwater wetland habitat will result in the expansion of the Florida Bay north, west, central, east-central, and east estuarine zones. The habitat associated with the increase in area subject to saline conditions is not factored into the analysis because there would likely be a transition period from a mangrove/graminoid marsh community to a nearshore estuarine environment that may take 20 years or more post-saltwater inundation to be realized. With CEPP in place and delivering higher volumes of freshwater flows, habitat function is likely to be much higher when compared to the future without CEPP under the sea level change scenarios considered in the analysis, indicating that CERP projects such as CEPP could help mitigate some of the impacts of sea level change.

Climate Related Concerns and Technical Uncertainties

The following broad questions are provided to illustrate emerging climate change-related concerns that need to be considered in the context of both restoration and water management for flood control and water supply:

- ***Sea Level Change Since Design in the 1950s Plus Monthly/Seasonal Variations in Tide Elevations* – Impacts on Central and Southern Florida (C&SF) Project System Drainage Performance.** The C&SF Project, as originally designed and built, included a number of gravity drainage structures that discharge to the ocean or other tidally influenced water bodies. The structure design standard was often to achieve full discharge capacity with the upstream canal stage either +6 inches or +9 inches above the anticipated tailwater elevation based on local mean high water elevations in Biscayne Bay and other coastal water bodies per tidal data in "Tide Tables and Tidal Bench Marks, Part I (East Coast), 1950" published by the United States Coast and Geodetic Survey. Tide elevations also vary seasonally by location for a few months and can temporarily increase long-term mean sea level or mean high water elevations by up to +6 inches as shown by the NOAA graph in **Figure 3-5**. Note that this seasonal increase in tidal water elevations comes in September and October, which often have greater needs for flood drainage capacity due to heavy rainfall, winds and storm surge events from passing tropical storms and hurricanes. In addition, high astronomical tides occur for a few days in the spring and fall when the orbits of the sun and moon line up so they have a combined influence on tides. High astronomical tides for Biscayne Bay can be more than +1 foot above normal mean high water, which further increases negative impacts on the stormwater gravity drainage systems. These negative impacts can extend far inland when coastal areas are unable to drain effectively due to high tide conditions at the canal outlets. This negatively impacts water management flexibility to address human and natural system needs.

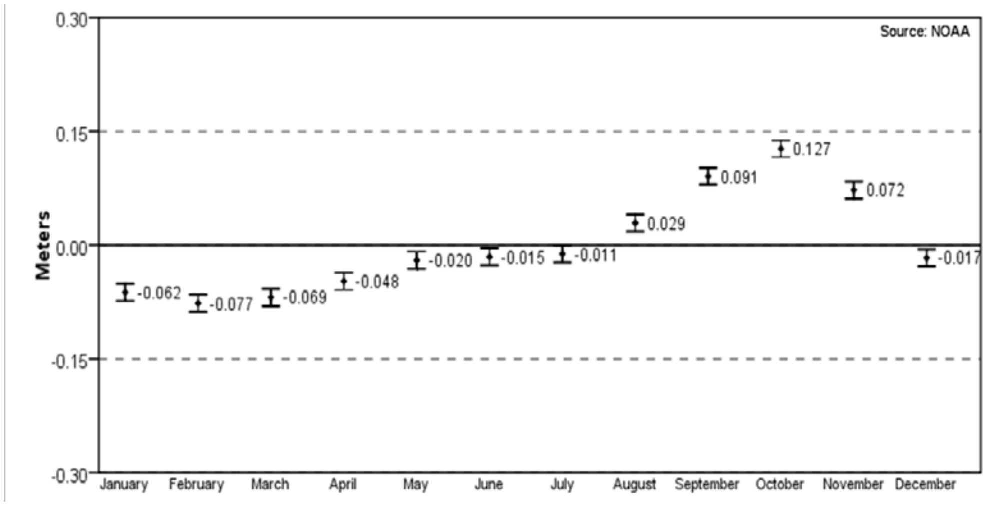


Figure 3-5. Average seasonal cycle at Key West, Florida (NOAA) Tide Station 87724580.

- ***Monthly/Seasonal Variations in Tide Elevations* – Impacts on Coastal Performance Measures.** Salinity targets at specific locations may need to consider the seasonal depth of tidal waters and the seasonal variations in target salinities in a specific location.

- ***Tidal Range Variation*.** Small tidal range and flat topography mean large horizontal shifts in tidal zones for even small sea level rise increases. The tidal range (from mean lower low water to mean higher high water) in south Florida at many locations, such as along Florida Bay and Shark River Slough (SRS) is small (around 2 feet) so 1 foot of sea level rise will force relocation of 50% of the established tidal zone. **Figure 3-5** shows the average seasonal cycle in tide elevation for Key West. Seasonal tides for Key West are pertinent for potential impacts on Florida Bay and tidal elevations in ENP.

- ***Potential Rapid Peat Loss Due to Saltwater Intrusion*.** A recent vegetation mapping report from the University of Miami indicates mangroves have advanced roughly 10 miles upstream in the SRS area over the past 10 years. This suggests that saltwater may already be much further upstream in the SRS area than previously understood. Saltwater can act as a catalyst for rapid decomposition of freshwater peat soils like those common in SRS (Willard and Bernhardt 2011, Hackney and Williams 2012). **Figure 3-6** to **Figure 3-9** show SRS with and without peat soils for sea level rise of +1-foot and +2-foot increments. Rapid peat loss could help turn SRS into a beginning replacement for Florida Bay, a potential significant landscape change, with only 1 to 2 feet of sea level rise based on USACE guidance discussed above. Multiple field monitoring stations with sensors at multiple depths are needed to help determine if salt water is moving upstream in the peat soils, perhaps moving below lower density freshwater flows.

- ***Potential Water Shortages Due to Increased Temperature and Precipitation Changes***. One of the concerns raised by the SFWMD modeling of potential future scenarios was the increased risk of water use restrictions and severe water shortages.

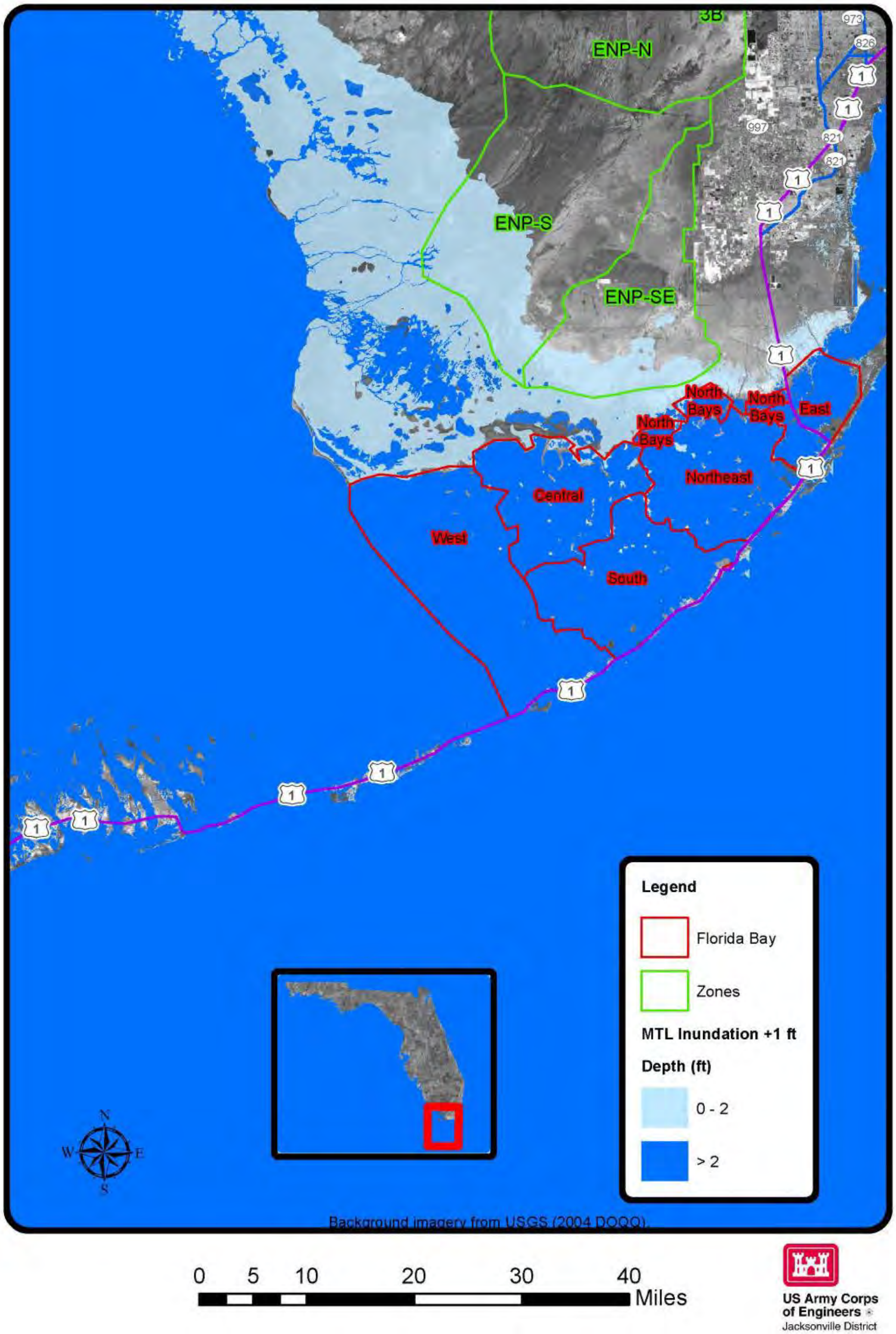


Figure 3-6. Southern Everglades existing topography with inundation without peat loss for +1 foot of sea level rise. (Note: ft – feet and MTL – mean tide level.)

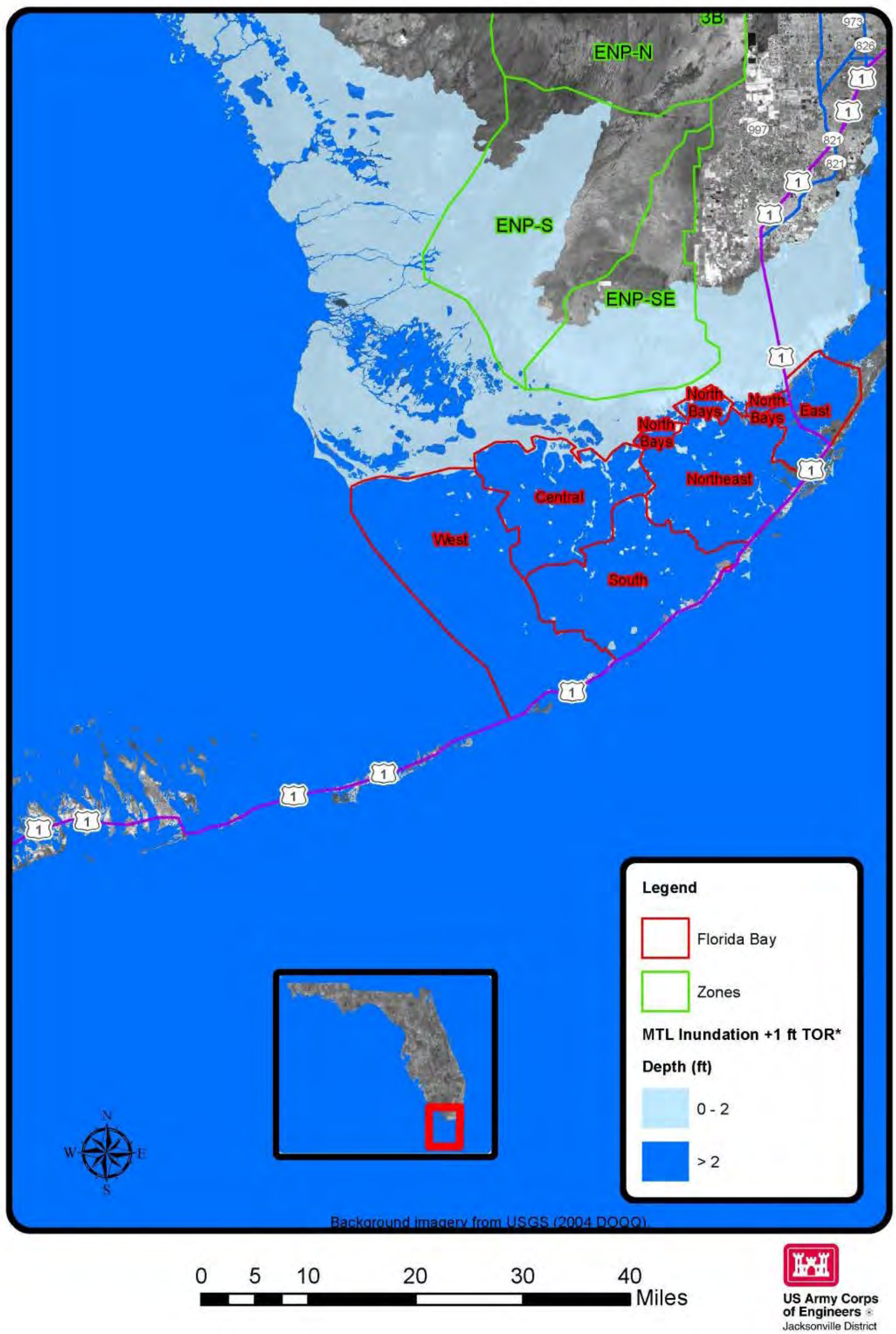


Figure 3-7. Southern Everglades existing topography with inundation with peat loss for +1 foot of sea level rise. (Note: ft – feet; MTL – mean tide level; and TOR – top of rock.)

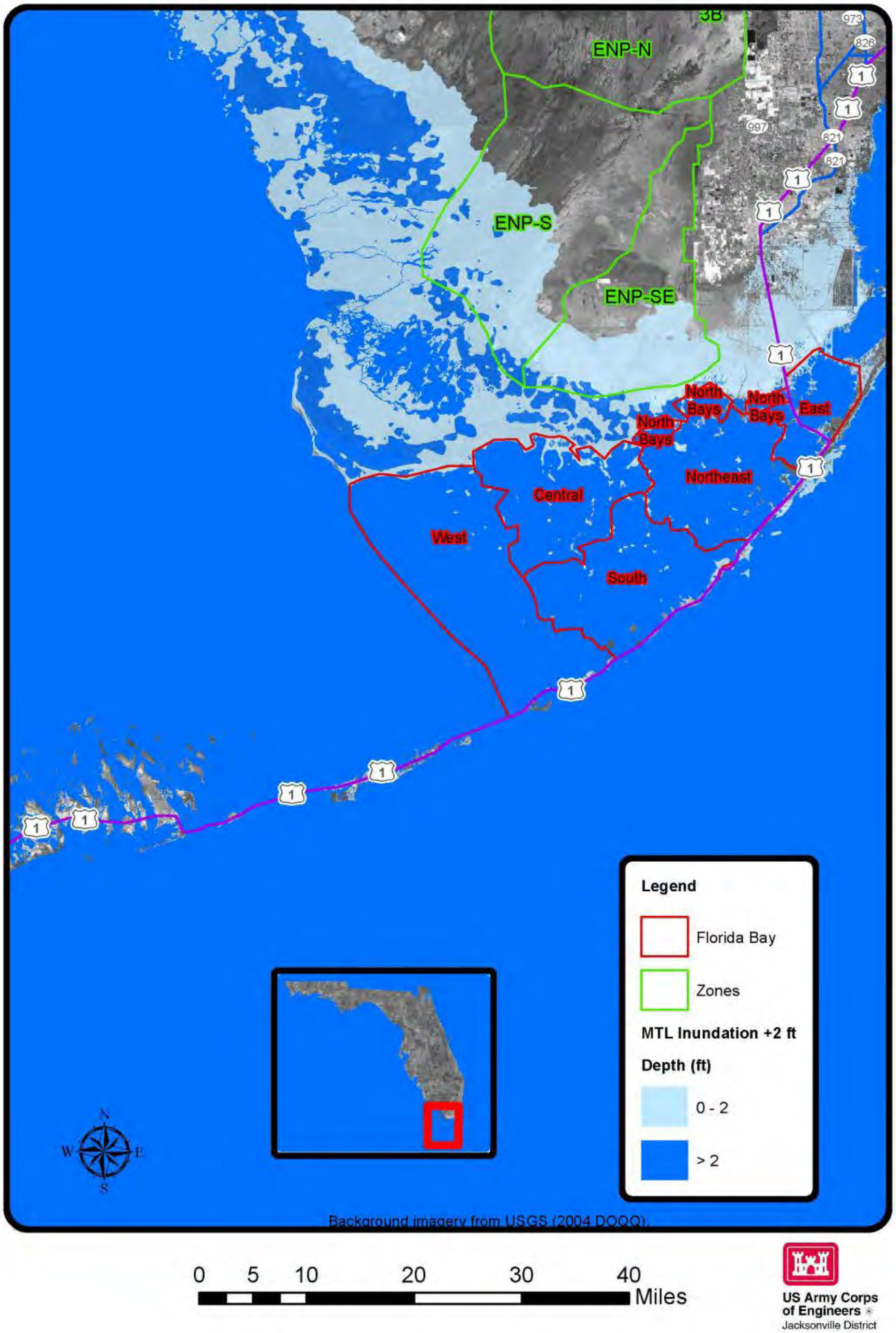


Figure 3-8. Southern Everglades existing topography without peat loss for + 2 feet of sea level rise. (Note: ft – feet and MTL – mean tide level.)

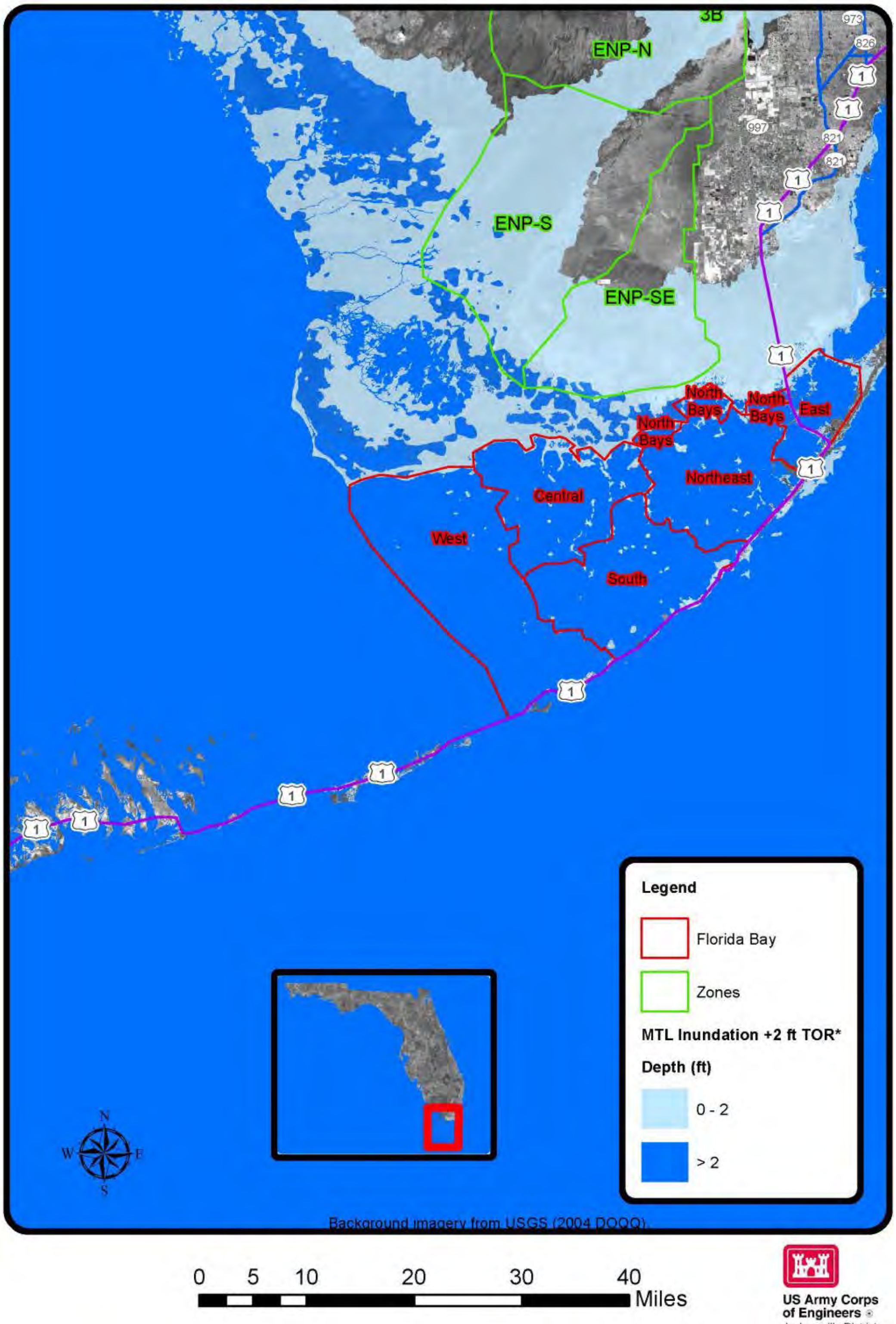


Figure 3-9. Southern Everglades existing topography with peat loss for +2 feet of sea level rise. (Note: ft – feet, MTL – mean tide level, and TOR – top of rock.)

INVASIVE SPECIES

Introduction

Executive Order 13112, entitled Invasive Species, signed February 3, 1999, states an "invasive species means an alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health.” Invasive species are directly implicated in losses of native species, biodiversity, ecosystem functions, ecosystem services, and livelihoods worldwide (U.S. Congress, Office of Technology, 1993). Increasingly, technology and globalization are reducing the barriers that once allowed unique species and ecosystems to evolve without continuous disturbances from biological invasions. As a result, rates of biotic exchange are increasing on all continents and the trend is expected to continue despite heightened international awareness of the impacts of biological invasions (Millennium Ecosystem Assessment, 2005). Florida is particularly vulnerable to the introduction, invasion and naturalization of nonnative species. This is due to several factors including a subtropical climate, dense human population centers, major ports of entry and the pet, aquarium and ornamental plant industries. Major disturbance to the landscape has also increased Florida’s vulnerability for establishment of invasive species. Invasive species are another factor that needs to be addressed in order to achieve Everglades restoration success.

Restoration Context

Successful restoration of south Florida ecosystems hinges on the ability to reverse the environmental degradation primarily caused by human activities over the last 100-plus years. While the CERP and RECOVER efforts involve numerous factors, the potential impact of invasive species has emerged as a high priority for CERP. Invasion of south Florida’s natural habitats by nonnative plant and animal species has significantly altered the ecosystem, particularly by displacing native species. Without successful control of invasive species, the benefits of restoration efforts will be reduced through the following outcomes: decreased biodiversity; displaced native plant and animal communities; decreased wildlife habitat and forage opportunities; altered soil erosion and accretion rates, hydrology, and fire patterns; altered predator/prey relationships; degraded environmental quality; and continued spread of diseases to native plants, animals and other organisms. In addition to environmental impacts, invasive species also impact human health, reduce agricultural production and property values, degrade aesthetic quality, decrease recreational opportunities, and threaten the integrity of human infrastructure such as waterways/navigation channels, locks, levees, dams and water control structures.

As both drivers and stressors of ecosystems, invasive species can alter ecosystem patterns and processes on both small and large scales and may result in unexpected successional trajectories as Everglades restoration proceeds (Ogden et al. 2005, Doren et al. 2009b). Therefore, the presence of invasive, nonnative species may greatly reduce certainty in the ability to predict restoration outcomes, particularly with regard to CERP performance measures (available online at http://www.evergladesplan.org/pm/recover/eval_team_perf_measures.aspx), which are used to help predict potential and measure actual restoration success. Invasive species have numerous direct and indirect effects to CERP performance measures identified by RECOVER, e.g., recovery of native vegetation mosaics and increased native fish recruitment in Lake Okeechobee, and ridge and slough/tree island landscape sustainability and recovery of native fauna (aquatic prey, wading birds, and alligators) in the GE region. A summary of CERP performance measures and invasive species effects is presented in **Appendix 3-1**.

Invasive Species Management

Scientists and land managers face enormous challenges when attempting to address biological invasions across a landscape as vast as the Florida Everglades. Effective management requires an integration of effective control tools, monitoring, research, early detection, rapid assessment and response strategies, preventative regulations and measures, and close interagency coordination (Masters and Sheley, 2001), all of which are described in **Appendix 3-1**. Nonnative species are rarely eradicated from the natural areas they invade (MacDonald et al. 1989, Bomford and O’Brien 1995). As such, the realistic expectation of “successful management” is limiting the impacts of biological invasions, not complete eradication. Ecosystems resulting from restoration will contain new species assemblages and biotic interactions relative to their predisturbance state, and these new biotic components and interactions are certain to include nonnative species (Norton 2009).

In the context of Everglades restoration, nonnative species invasions will continue to exert pressure on native species and ecosystem function in the restored condition. Nonnative species are expected to respond differently to restoration and, in some cases, may continue to act as drivers of ecosystem change as restoration proceeds. Improved hydroperiods and water quality may reduce the competitiveness of aggressive plant species, such as Brazilian pepper (*Schinus terebinthifolius*) and melaleuca (*Melaleuca quinquenervia*), but such plant species are likely to persist in the restored condition. Other species, particularly many invasive animals, will continue to find suitable niches after restoration. Project features will likely aid in the spread of invasive species into areas previously not inhabited. Thus, without a long-term commitment to invasive species management, the goals of Everglades restoration are unlikely to be achieved.

Regional Invasive Species Status and Trends

The RECOVER MAP does not have specific efforts focused on invasive species monitoring. As mentioned above, invasives species are recognized as drivers/stressors to the south Florida ecosystem and some MAP funded efforts happen to provide information on invasive species presence. However, most of the information collected on invasive species by agency invasive species-specific funded efforts was synthesized in **Appendix 3-1**. The agencies funding these efforts include the SFWMD, National Park Service (NPS), Florida Department of Agriculture and Consumer Services, Florida Fish and Wildlife Conservation Commission, United States Department of Agriculture, USACE, United States Fish and Wildlife Service, and USGS. **Table 3-6** identifies priority nonnative animal species. **Table 3-7** identifies Florida Exotic Plant Pest Council Category 1 plant species (invasive plant species that alter native plant communities by displacement, changed community structure or ecological function, or hybridization with natives).

Table 3-6. Priority nonnative animal species by CERP RECOVER region.

Species		Region			
Amphibians		**SCS**	**GE**	**NE**	**LO**
Rhinella marina	giant toad		x		x
Osteopilus septentrionalis	Cuban treefrog	x	x		x
Reptiles		**SCS**	**GE**	**NE**	**LO**
Anolis equestris equestris	knight anole		x		
Anolis sagrei	brown anole		x		x
Boa constrictor	common boa		x		
Caiman crocodilus	spectacled caiman	x	x		
Eleutherodactylus planirostris	greenhouse frog	x	x		x
Hemidactylus mabouia	tropical house gecko	x	x		
Iguana iguana	green iguana		x		x
Python molurus bivittatus	Burmese python	x	x		
Python sebae	northern African python		x		
Tupinambis teguixin	gold tegu		x		
Tupinambis merianae	Argentine black & white tegu		x		
Varanus niloticus	Nile monitor		x		
Birds		**SCS**	**GE**	**NE**	**LO**
Alopochen aegyptiacus	Egyptian goose	x	x	x	x
Cairina moschata	muscovy duck	x	x	x	x
Porphyrio porphyrio	purple swamphen	x	x		x
Streptopelia decaocta	Eurasian collared-dove		x		x
Sturnus vulgaris	European starling		x		x
Threskiornis aethiopicus	sacred ibis		x		
Mammals		**SCS**	**GE**	**NE**	**LO**
Rattus rattus	black rat		x		x
Sus scrofa	feral pig	x	x		x
Fishes		**SCS**	**GE**	**NE**	**LO**
Astronotus ocellatus	oscar	x	x		x
Belonesox belizanus	pike killifish	x	x		
Channa marulius	bullseye snakehead	x	x		
Cichla ocellaris	butterfly peacock cichlid	x	x		
Cichlasoma bimaculatum	black acara	x	x	x	x
Cichlasoma managuense	jaguar guapote		x		
Cichlasoma urophthalmus	Mayan cichlid	x	x		x
Clarias batrachus	walking catfish	x	x		x
Hemichromis letourneuxi	African jewelfish		x		
Hoplosternum littorale	brown hoplo		x	x	x
Macrognathus siamensis	spot-finned spiny eel		x		
Monopterus albus	Asian swamp eel		x		

Table 3-6. Continued.

Species		Region			
Fishes (Continued)		**SCS**	**GE**	**NE**	**LO**
Oreochromis aureus	blue tilapia	x	x	x	x
Oreochromis mossambicus	Mozambique tilapia		x	x	
Pterois volitans	lionfish	x		x	
Pterygoplichthys disjunctivus	vermiculated sailfin catfish				x
Pterygoplichthys multiradiatus	Orinoco sailfin catfish		x		x
Tilapia mariae	spotted tilapia	x	x		
Invertebrates		**SCS**	**GE**	**NE**	**LO**
Balanus reticulatus	barnacle	x			
Balanus trigonus	barnacle	x		x	
Callinectes bocourti	Bocourt's swimming crab	x			
Charybdis hellerii	Indian Ocean portunid crab	x		x	
Cittarium pica	West Indian top shell				
Corbicula fluminea	Asian clam	x	x	x	x
Cuthona perca	Lake Merritt cuthona	x			
Daphnia lumholtzi	water flea		x		x
Glossodoris sedan	marine nudibranch	x			
Haliplanella luciae (= H. lineata)	sea anemone	x			
Litopenaeus stylirostris	Western blue shrimp	x			
Litopenaeus vannamei	Pacific white shrimp	x			
Lyrodus mediolobatus	Indo-Pacific shipworm			x	
Melanoides tuberculatus	red-rim melania	x			
Metamasius callizona	Mexican bromeliad weevil		x		
Mytella charruana	charru mussel			x	
Parapristina verticillata	fig wasp (of F. microcarpa)		x		
Paratachardina lobata	lobate lac scale		x		
Perna viridis	green mussel			x	
Phyllorhiza punctata	spotted jellyfish			x	
Pinctada margaritifera	black-lipped pearl oyster			x	
Pomacea insularum (P. maculata)	island applesnail	x	x		x
Solenopsis invicta	imported fire ant	x	x		x
Sphaeroma terebrans	wood-boring isopod	x		x	
Sphaeroma walkeri	fouling isopod	x		x	
Styela plicata	sea squirt			x	
Sundanella sibogae	bryozoan			x	
Tridacna crocea	giant clam			x	
Tridacna maxima	giant clam			x	
Victorella pavida	bryozoan			x	
Watersipora subovoidea	bryozoan			x	
Xyleborus glabratus	redbay ambrosia beetle		x		
Zachrysia provisoria	Cuban garden snail		x		x

Table 3-7. Florida Exotic Plant Pest Council Category 1 plant species by CERP RECOVER region.

Florida Exotic Plant Pest Council Category I Invasive Plants		SCS	GE	NE	LO
Abrus precatorius	rosary pea	x	x		x
Acacia auriculiformis	earleaf acacia	x	x		
Ardisia elliptica	shoebutton ardisia		x		
Bauhinia variegata	orchid tree	x	x		
Bischofia javanica	bishopwood		x		
Calophyllum antillanum	Santa Maria	x	x	x	
Casuarina equisetifolia	Australian pine	x	x		x
Colocasia esculenta	wild taro		x		x
Colubrina asiatica	lather leaf	x	x	x	
Cupaniopsis anacardioides	carrotwood		x		
Dioscorea bulbifera	air-potato	x	x		x
Eichhornia crassipes	water-hyacinth	x	x		x
Eugenia uniflora	Surinam cherry	x	x		
Ficus microcarpa	laurel fig	x	x		x
Hydrilla verticillata	hydrilla	x	x	x	x
Hygrophila polysperma	green hygro		x		
Hymenachne amplexicaulis	West Indian marsh grass	x	x		x
Imperata cylindrica	cogon grass	x	x		x
Jasminum fluminense	Brazilian jasmine		x		
Lantana camara	lantana	x	x		x
Ludwigia peruviana	Peruvian primrosewillow	x	x		x
Lumnitzera racemosa	Indo-Pacific black mangrove	x			
Luziola subintegra	tropical watergrass		x	x	x
Lygodium microphyllum	Old World climbing fern	x	x		x
Lygodium japonicum	Japonese climbing fern	x			
Manilkara zapota	sapodilla		x		
Melaleuca quinquenervia	melaleuca, paper bark	x	x		x
Melinis repens	Natal grass	x	x		x
Nephrolepis brownii (N. multiflora)	Asian sword fern		x		
Nephrolepis cordifolia	tuberous sword fern	x	x		x
Neyraudia reynaudiana	burmareed	x	x		x
Panicum repens	torpedo grass	x	x		x
Pennisetum purpureum	Napier grass	x	x		x
Pistia stratiotes	waterlettuce		x		x
Psidium guajava	guava	x	x		x
Rhodomyrtus tomentosa	downy rosemyrtle	x			
Ruellia tweediana (R. simplex)	Mexican petunia		x		x
Salvinia minima	water spangles	x	x		x
Scaevola taccada	beach naupaka	x	x	x	

Table 3-7. Continued.

Florida Exotic Plant Pest Council Category I Invasive Plants (Continued)		SCS	GE	NE	LO
Schinus terebinthifolius	Brazilian pepper	x	x		x
Schefflera actinophylla	Australian umbrella tree	x	x		
Scleria lacustris	Wright's nutrush		x		x
Solanum viarum	tropical soda apple	x			x
Syngonium podophyllum	arrowhead vine	x	x		
Syzygium cumini	Java plum	x	x		x
Tectaria incisa	incised halberd fern		x		
Thespesia populnea	seaside mahoe	x	x	x	
Urena lobata	caesarweed	x	x		x
Urochloa mutica	Para grass	x	x		x

Status and trends are presented in **Appendix 3-1** for the following species: Australian pine (*Casuarina equisetifolia*), Brazilian pepper, melaleuca, Old World climbing fern (*Lygodium microphyllum*), red bay ambrosia beetle (*Xyleborus glabratus*) (and associated laurel wilt disease), Nile monitor (*Varanus niloticus*), Burmese python (*Python molurus bivittatus*), northern African python (*Python sebae*), and nonindigenous fish species. Progress in managing Australian pine, Brazilian pepper, and melaleuca is clearly documented in **Figure 3-10**, while also revealing challenges in containing expansion of Old World climbing fern.

Requirement for CERP Projects

CERP Guidance Memorandum 062.00: Invasive and Native Nuisance Species Management (SFWMD and USACE 2012b) requires CERP projects to include invasive and native nuisance species management in the project life cycle. In addition, it provides guidance to project delivery teams for comprehensively assessing invasive and native nuisance species during all phases of a project and requires each project to develop and implement an invasive and nuisance species management plan. The plan is a living document that is updated as needed throughout the life of a project. Plans have been completed for the C-111 Spreader Canal Western Project (C-111 SCWP), Biscayne Bay Coastal Wetlands (BBCW) Project, and CEPP; however, only CEPP includes management for both plants and animals.

Prior to implementation of the guidance memorandum, CERP Projects were not required to complete an invasive and nuisance species management plan. The first vegetation management plan for a CERP project was completed by the Picayune Strand Restoration Project (PSRP) Project Delivery Team and was included in the project construction phasing transfer and warranty plan. PSRP is the first CERP Project to actively manage invasive plants during construction. In addition, vegetation management plans have been completed for the following projects: Indian River Lagoon-South, Site 1, Broward County Water Preserve Area, and Decomp Physical Model. Site 1 was the first project to include language in the contract specifications to prevent the transfer of invasive species during construction activities.

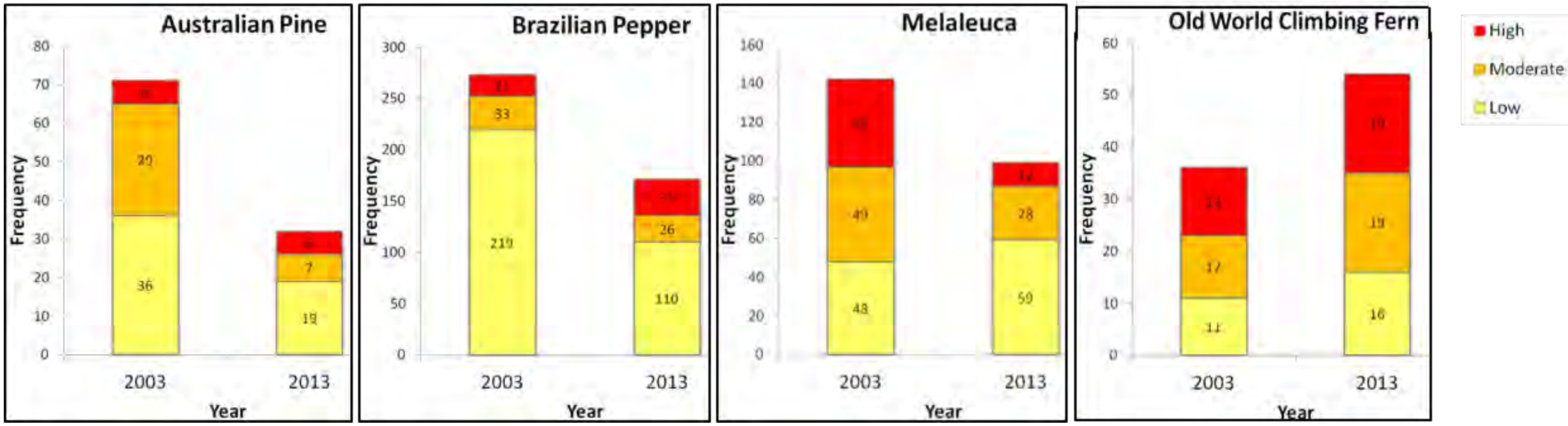


Figure 3-10. Distribution and relative abundance of Australian pine, Brazilian pepper, melaleuca, and Old World climbing fern in the GE region 2003–2013.

Conclusion

The species and results discussed in the full report represent only a fraction of the nonnative and invasive species that occur within the GE region, and should be considered "case studies." Over time, focal species and efforts to address them should be expected to change in response to changing perceptions of threat to ecosystem function, as well as identification of new and significant threats.

Natural systems within the GE region are under increasing threat of invasion by nonnative species, including nonnative fish, amphibians, reptiles, and birds. Melaleuca, Old World climbing fern, Brazilian pepper, Burmese pythons and Cuban tree frogs (*Osteopilus septentrionalis*) are already established in some of the remote Everglades natural areas and Nile monitors, yellow anacondas (*Eunectes notaeus*), tegu lizards (*Tupinambis teguixin*), and spectacled caiman (*Caiman crocodilus*) have all been found on or near the Everglades. Although fishes and reptiles have been of primary concern, amphibians (Cuban tree frogs), birds (sacred ibis [*Threskiornis aethiopicus*], purple swamp hens [*Porphyrio porphyrio*]) and macroinvertebrates (island apple snails [*Pomacea insularum*]) all potentially threaten the Everglades and ecosystem restoration. Methods to intercept, eradicate, or contain these invaders have not kept pace with the increasing risk.

Sustained invasive plant management efforts have resulted in substantial declines in some priority invasive plant species (e.g., melaleuca, Australian pine). Integration of herbicidal control, biological controls, and prescribed fire has contributed to the success of the melaleuca regional control effort. Old World climbing fern, an aggressive invader of Everglades tree islands continues to increase in abundance throughout the region. Existing management programs are insufficiently funded and control tools remain limited to nonselective herbicides and two biological control agents. Agencies should continue with a restrained program of aerial spraying (to limit damage to nontarget native species), combined with proactive ground treatment of selected infestations to avoid losses of wetland tree species. Critical research and monitoring needs include improvements to herbicide translocation to rhizomes (Hutchinson et al. 2010), improved herbicide efficacy in standing water, continued research towards development of effective biological control agents, ground-based monitoring for incipient populations, and development of restoration techniques for heavily impacted tree islands.

Conversely, intensive efforts to control Burmese pythons have shown limited success, despite ongoing efforts and funding expended to develop control methods. This indicates that effort alone may not always ensure success, and some control efforts may inherently be more difficult and costly than others, and the possibility remains that meaningful control may not always be feasible.

Preventing introduction and establishment of invasive species is the first line of defense against invasions and is a must for ensuring the survival of native species in native habitats. Early detection and rapid response efforts increase the likelihood that invasions will be successfully contained or eradicated while populations are still localized (ECISMA 2009). Once populations are widely established, options for management become limited and expensive, and are often ineffective. Populations of invasive species that are established and widespread will require long-term management programs for control, containment, and protection of vulnerable resources.

In most cases we do not have necessary information for detailed, specific evaluations of CERP projects for the need for management activities to control invasive species. However, partial removal of canals and levees could encourage spread or provide sites for colonization of tegus, monitors, pythons, and Cuban tree frogs. In this case the most effective and lowest cost management option is early detection and rapid removal of invasive species during and post project (Westbrooks 2004).

FIRE

Introduction

The Everglades ecosystem contains several plant communities that consist of mostly “fire adapted species,” which are defined as plant species that have traits that allow them to survive fires. These species are expected to thrive in areas where low intensity fires occur frequently. Pinelands, short hydroperiod marl prairies, sawgrass marshes, and coastal prairies are the four major vegetation types that occur throughout the Everglades Protection Area, and which are targeted for fire management by both BCNP and ENP fire management programs (NPS 2010, NPS 2013). Both of these NPS units (ENP and BCNP) set goals for burning all fire adapted communities by defining a desired range of fire return times, and both NPS units have Natural Resource Management programs that encourage the application of fires where fire severity is matched to the context of an area in support of the goal of producing a healthy mosaic of vegetated habitats. BCNP has set a goal of burning their fire adapted communities every 3 to 5 years, while ENP has set a goal of burning pinelands every 3 to 7 years, coastal prairies every 2 to 10 years, and sawgrass marsh or marl prairie every 3 to 12 years (NPS 2010, NPS 2013)[1].

Fire management is a complex and challenging enterprise. An overarching goal for fire management is to maintain safe conditions for visitors to the NPS unit and in lands adjacent to the NPS unit. Fire managers perform several different types of activities to ensure public safety and resource protection, including suppressing wildfires that occur when fire risks are unacceptably high; monitoring and/or containing wildfires during naturally occurring fire seasons; setting prescribed fires in areas that have not experienced fires for periods longer than the desired return time; and setting prescribed fires to reduce fuel loading in areas where the risk of future wildfires is too high. Fire managers have an emergency management budget process that allows them to spend additional funds when extreme fire risk conditions occur—in particular during extended droughts.

Developing and implementing a fire management program in south Florida is made even more challenging as a result of several issues: presence of invasive exotic plant species that are widely distributed across the landscape, history of land uses that has disturbed tens of thousands of acres of vegetation under management, and continuing challenges associated with human altered hydrologic patterns. All three of these factors may increase the likelihood that unnaturally intense fires or the

[1] Fire is an important driver of change across the Greater Everglades regional areas. The fire assessment area mapped was limited to the data available at the time for Big Cypress National Preserve and Everglades National Park. Future assessment reports hope to incorporate fire assessments across the system (i.e., including the Water Conservation Areas) by both incorporating data from state agencies and agreeing to a common protocol to assess and report fire frequency.

absence of fires negatively impact important natural resources like tree islands, marsh soils, and wetland vegetation, which are defining characteristics of the Everglades ecosystem. The application of prescribed fires on the landscape can mitigate these challenges to some extent.

The existing regional water management infrastructure and current water management practices may influence fire management in at least three ways: (1) altering the fire pattern; (2) altering the vegetation pattern; and (3) increasing oxidation of peat soils. Altered fire pattern is identified in the Ridge and Slough Conceptual Ecological Model (CEM) (Ogden 2005), the Southern Marl Prairies CEM (Davis et al. 2005), and the Big Cypress Regional Ecosystem CEM (Duever 2005). In the Ridge and Slough CEM, the shortened hydroperiods that result from reduced water storage capacity are linked to altered fire patterns and the destruction of tree islands (Ogden 2005). The Southern Marl Prairies CEM identifies the linkage between water management practices and both shortened hydroperiods and increased drought severity in driving high intensity dry season fires. Burning practices exacerbated by compartmentalization are also connected to the occurrence of high intensity dry season fires in the Southern Marl Prairies CEM (Davis et al. 2005).

There are several ways that vegetation patterns can be altered by regional water management infrastructure and operations. In BCNP, the spread of cattail (*Typha* spp.) and primrose willow (*Ludwigia peruviana*) is identified as a direct result of the presence of nutrient enriched inflows through the canal system (Duever 2005). Cattail and primrose willow are less likely to be combusted by low intensity fires and can lead to biomass accumulation that ultimately triggers high intensity fires. Another possible effect of regional water management infrastructure results from lowered water tables and shortened hydroperiods that drive a shift in the relative abundance of plant species across the landscape from wetland plant associations to terrestrial plant communities that are fire adapted (i.e., grasses and less flood tolerant woody plants) in all three CEMs (Davis et al. 2005, Duever 2005, Ogden 2005). Large-scale shifts in vegetation from wetland to more drought tolerant species has been documented by shifts in pollen abundance in peat soil cores from Water Conservation Area (WCA) 1 in 1950–1960, and during the 1930s in WCA 2, WCA 3, and Taylor Slough (Willard et al. 2001). These shifts coincided with the development of flood control infrastructure around Lake Okeechobee and in the Everglades Protection Area (Willard et al. 2001).

Regional water management infrastructure and operations can increase oxidation of peat soils (Duever 2005, Ogden 2005) through two different processes. Rapid consumption of peat soils during intense fire events can destroy centuries of accumulated peat in a single event. Gradual loss of peat soils through oxidation can occur when hydroperiods are shortened, and the rate of decay of peat accelerates due to the presence of more oxygen in soils that are exposed to the atmosphere. This process has been described as “microbial” oxidation of soils, which can result in the loss of a decade or more of accumulated peat soil in a single year; 0.6 inches of peat loss per year is the observed long-term average in the Everglades Agricultural Area (Shih et al. 1998), while maximum peat accumulation rates in the Everglades are between 0.04 and 0.1 inch per year (Larsen et al. 2007).

All three of these consequences increase the burdens on fire management programs. Large areas of vegetation containing fire adapted species require increased fire management. Periods of increased drought conditions create a greater need for prescribed fires that reduce hazardous fuels and wildfire suppression. Both of these activities are focused on protecting organic soils and plant communities (i.e., tree islands, cypress prairies, and cypress domes) that are not adapted to frequent fires, but potentially combustible when drought conditions prevail (Zaffke 1983, Ruiz et al. 2013). The overarching consequence of the existing regional water management system is ongoing ecosystem decline. Degradation occurs across the landscape in a complex pattern that results from a combination of fast and slow processes (Philippi 2007, Ryan et al. 2007). Conceptualizing how degradation occurs across a landscape can be challenging. Direct observation of degradation processes across hundreds of square kilometers requires considerable intellectual and physical resources. Managing a vast area that is experiencing these kinds of disruptions is complex.

Goals for fire management in ENP and BCNP are set in this very challenging context. Vegetation patterns, rates of biomass accumulation (often called “fuel accumulation” in fire management literature), hydrology, water quality, wildlife, special status species, critical habitats of special status species, cultural resources, wilderness characteristics, historical land use, park operations, visitor uses and experience, urban-wildland interface challenges, and air quality issues are all evaluated carefully as a part of the development of a fire management program.

An adaptive management framework for identifying successes and challenges associated with fire management is being used in both BCNP and ENP (NPS 2010, NPS 2013). The operational definition of adaptive management is defined for the Department of Interior in Williams et al. (2009), and this paper is developed to support the “double loop” learning process. The first loop occurs annually and is focused on the use of monitoring that identifies the location where fires have occurred to determine whether multi-year burn/suppression/fire monitoring plans need to be altered as a result of changing environmental conditions (i.e., should strategies and techniques change). The second learning loop occurs on longer time increments (5 to 10 year basis, or when viewed as necessary by stakeholders) and is focused on clearly describing the existing challenges to managing natural resources and revisiting the assumptions (i.e., problem formulation and identification of objectives) that are used to formulate multi-year strategies.

This summary takes only a few small steps in using monitoring information to evaluate the status of fire management in BCNP and ENP. A comprehensive geo-database containing historical fire records for BCNP and ENP is the basis of this analysis. This database was developed, carefully inspected, and delivered to the NPS by the USGS in 2010 and subsequently updated to include fires through 2012 (Smith et al. in review). Only a few aspects of this database are filtered and summarized here.

Methods

ArcGIS version 10.1 (ESRI, Redlands, California) was used to calculate a summary of the long-term fire return time for the terrestrial portions of ENP and BCNP from a shape file containing all documented fires since the first few years after the establishment of ENP in 1947 and BCNP in 1974. A 400 m-by-400 m grid (based from the grid used to summarize water depths across the Everglades Protection Area, which is the Everglades Depth Estimation Network or “EDEN”) was also developed into a shape file. A count of all the fires that occurred since the inception of each NPS unit was developed for each of the 52,189 grid cells that cover the terrestrial portions of both NPS units. The data set contains information about 2,813 fires that occurred in BCNP (1978–2012) and 2,050 fires that occurred in ENP (1948–2012) that cover approximately 6,993 square kilometers of NPS administered lands. The fire count observed in each cell is then divided by the number of years that span the period of record of each NPS unit. This calculation yields an “average” number of fires per year, and the reciprocal of this number (number of years per fire) represented one of seven “fire return time” categories in **Figure 3-11**.

Once a visual representation of the fire return time is developed, a second question emerges, namely, “Where do fire return times agree or disagree with our resource management goals?” Since fire management goals are predominately expressed in terms of goals for specific habitats, it is necessary to relate the fire return times observed in **Figure 3-12** to specific habitat types. The location of habitats types in ENP and BCNP were primarily derived from University of Georgia remote sensing-based vegetation classification maps (Welch et al. 1999). For the purpose of this brief report, the presence of fire adapted communities in all 52,189 grid cells is determined by applying a spatial join of the attributes of the University of Georgia map to each cell in the 400 m-by-400 m grid. Often more than one instance of a habitat occurred in a cell, so multiple instances were reduced to a single record of each type using a dissolve process. One consequence of this step is that the actual acreage of fire adapted communities does not directly match the acreages implied on the summary maps, since many 400 m-by-400 m cells include both fire adapted and non-fire adapted habitat types. Each vegetation type is associated with a specific desired return time—3 to 5 years for BCNP pinelands, prairies, and sawgrass marshes; 3 to 7 years for ENP pinelands; 3 to 10 years for ENP coastal prairies; and 3 to 12 years for ENP sawgrass marsh or marl prairie. The observed fire return time for each cell is compared to the minimum and maximum desired return time, and the cell is registered in one of four categories: (1) fire not needed (i.e., no fire adapted communities present); (2) fire too infrequent; (3) fire frequency appropriate; or (4) fire too frequent.

The final step is to quantify the degree of performance. This step is achieved in two parts. First, the NPS management units are differentiated into landscape types based on the physical geography (or physiography) of the NPS units. Physiographic regions are designated based on the combination of elevation, soils, hydrology, and vegetation communities observed in discreet areas (**Figure 3-12**). BCNP is divided into 10 physiographic regions (modified by authors from Duever et al. 1986), while ENP is divided into 12 discreet physiographic regions (Schomer and Drew 1982, Davis 1943, White 1970, Puri and Vernon 1964). Two of these regions are shared across the common boundary of ENP and BCNP, so the result is a set of 20 physiographic regions that are available for summary. These geographic distinctions are used to help identify location-specific differences in fire performance. Once each of the grid cells is identified with the region that it falls within, a summary of fire return time and fire performance by region is calculated. Fire return time is summarized as a histogram of the grid cells for each of the 20 regions. Fire performance is summarized by counting number of grid cells in each region that occur in all four categories of performance.

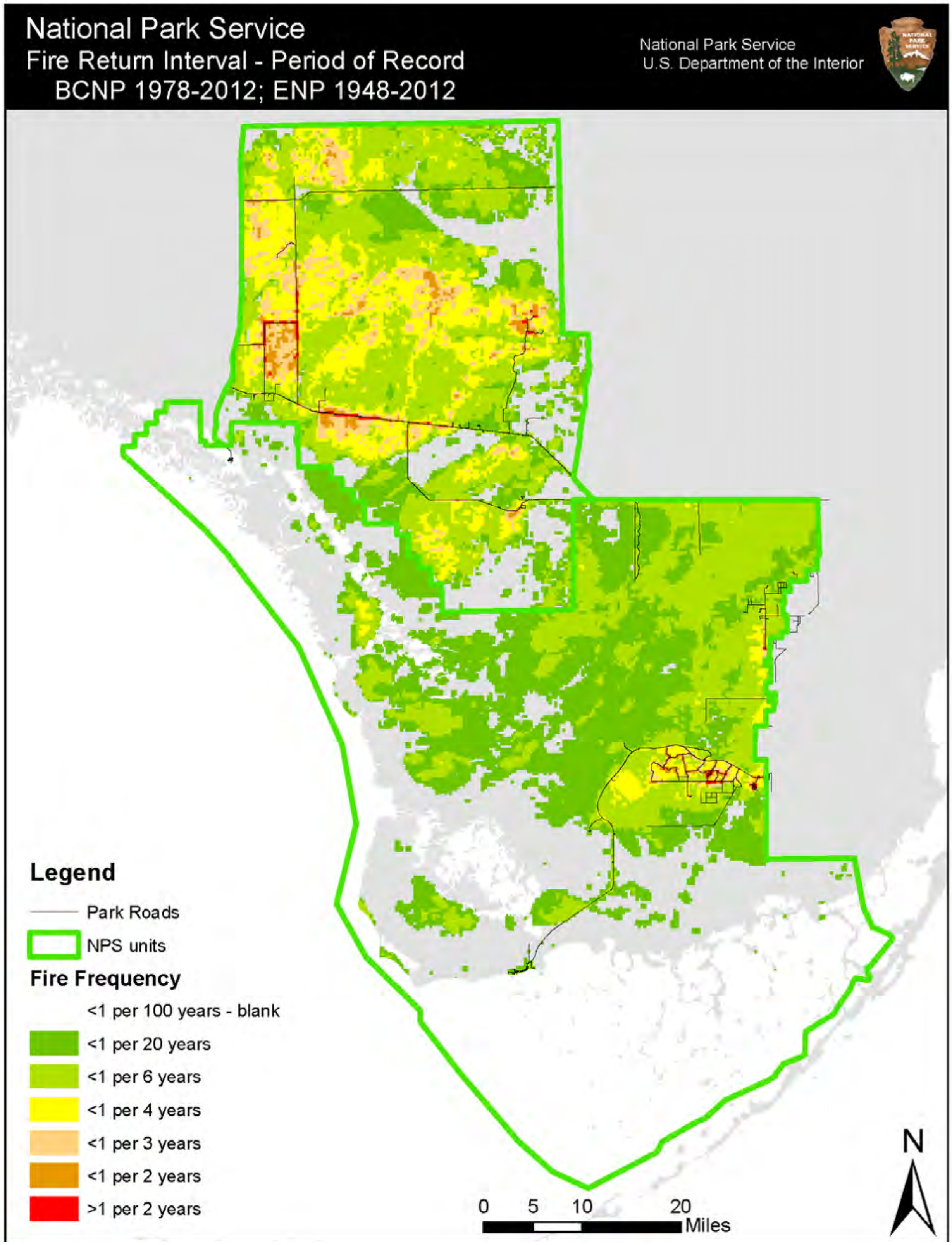


Figure 3-11. Long-term fire return times observed in BCNP and ENP.

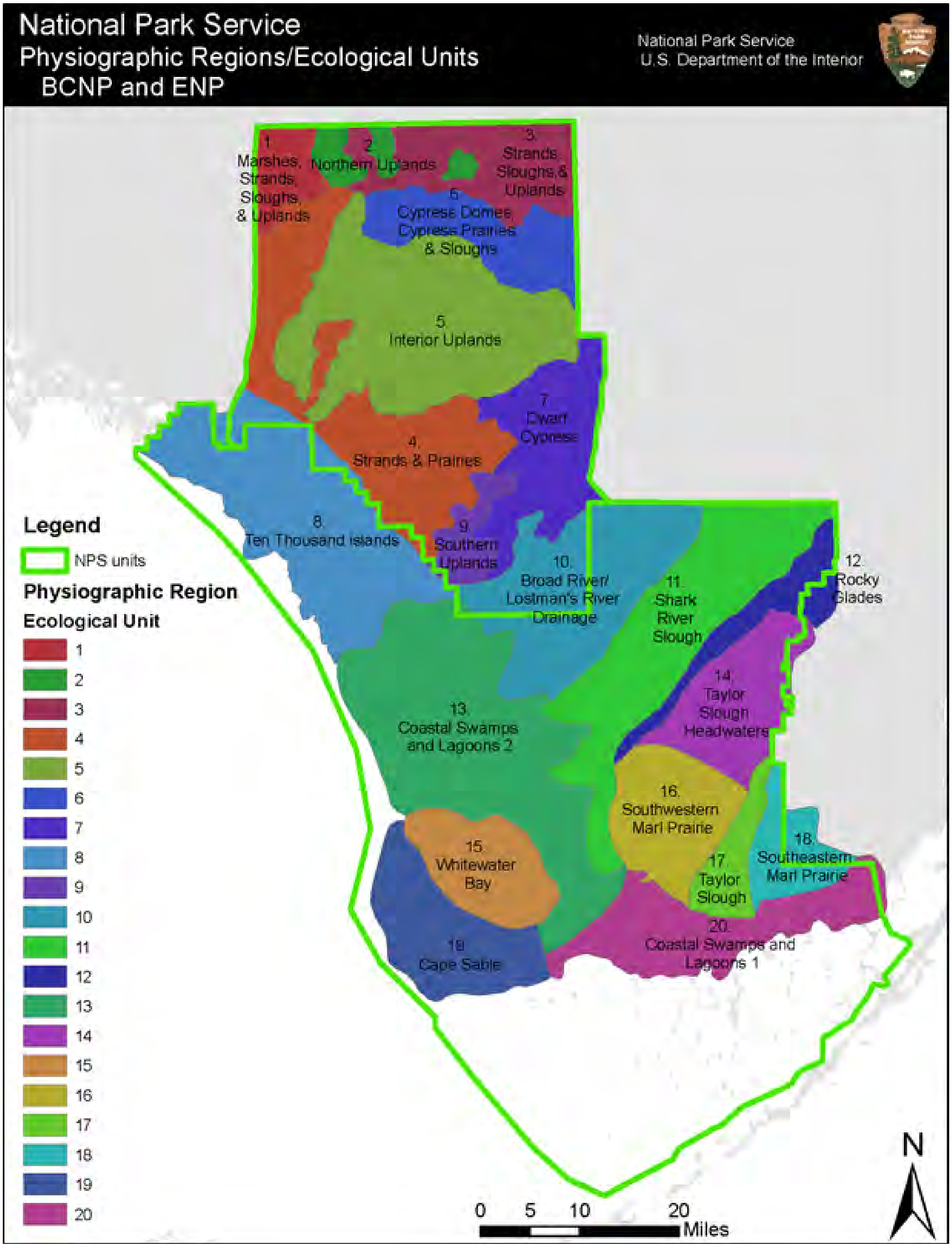


Figure 3-12. The ecological regions of BCNP and ENP are designated based on similarity in the vegetation assemblages, hydrology, and elevation patterns within each region. The combination of these three factors is identified as physiography (or physical geography).

Results

Before results are summarized, it is essential to discuss the caveats and limitations associated with the databases that were used in this analysis.

Fire History Geodatabases

The fire history geodatabases are a collection of data detailing the location and attributes of fires that have occurred in ENP and BCNP. These databases were created from fire records from 1948 through 2012 for ENP and 1978–2012 for BCNP. All geodatabases were built according to the "Guidelines for Building Fire History Spatial Data Layers" and the "Proposed NPS Spatial Data Standards for Fire History". Hard copies of paper fire records recorded on DI-1201, DI-1202, and Wildland Fire Report forms were obtained from both parks. Information was interpreted from the forms to assign the attributes associated with the fires. The design of the form changed over time and access to the form instructions for earlier versions was not available. Hence, the consistency of the data changed as our ability to interpret the forms changed with the versions of the forms.

Spatial data about the fires were derived from various sources, including hand drawn maps in the paper fire records, hand drawn mylar maps, and digital data layers. Fire perimeter polygons were constructed using the maps associated with the fire reporting forms. In some cases digital perimeters were available. When this occurred, these perimeters were compared to the paper maps associated with the fire reporting forms and checked for accuracy. Where conflicts in information were apparent, it was assumed that the map in the fire report was the most correct information source and the final information was taken from the map. When a fire did not have a map indicating a perimeter, standard shapes were used as perimeter proxies. Circles were used when the "area burned" acreage estimate from the fire reporting form was ≤ 1.0 acre. Rectangles were used for estimated burned acreages larger than 1.0 acre and were sized to equal the acreage reported. These circle and rectangle fire "perimeters" were positioned geographically as best as possible based on the fire narrative. The caveats and limitations described here were provided by Ann Foster, USGS.

Vegetation Map

There are also important limitations associated with the use of the University of Georgia vegetation map. First, the grid that is used to summarize fires is less detailed than the vegetation map in many areas. The maps for fire goals are made based on a fire adapted community occurring within the 400 m-by-400 m grid cell on which the fire is summarized. This process is useful for the purpose of evaluating widespread vegetation types (i.e., pinelands and prairies), but would need to be adapted to effectively evaluate less widespread communities (i.e., hardwood hammocks). A second important caveat associated with the University of Georgia map is a result of how the map was made and the resulting degree of accuracy of the map. The University of Georgia map was created using aerial imagery to assign a three-tiered hierarchical vegetation classification (Everglades Vegetation Classification System; Jones and Remillard 1997, Madden et al. 1999) for the entire mapped area. The accuracy of the map was tested by using field visits to 254 randomly selected locations, and it revealed that the most detailed classification of vegetation was incorrect more than 50% of the time, and major vegetation classes (the second level in the hierarchy) were incorrectly classified 22% to 38.5% of the time (Bradley and Woodmansee 2008). This means that the results presented in **Figure 3-13** need to be viewed with significant caution.

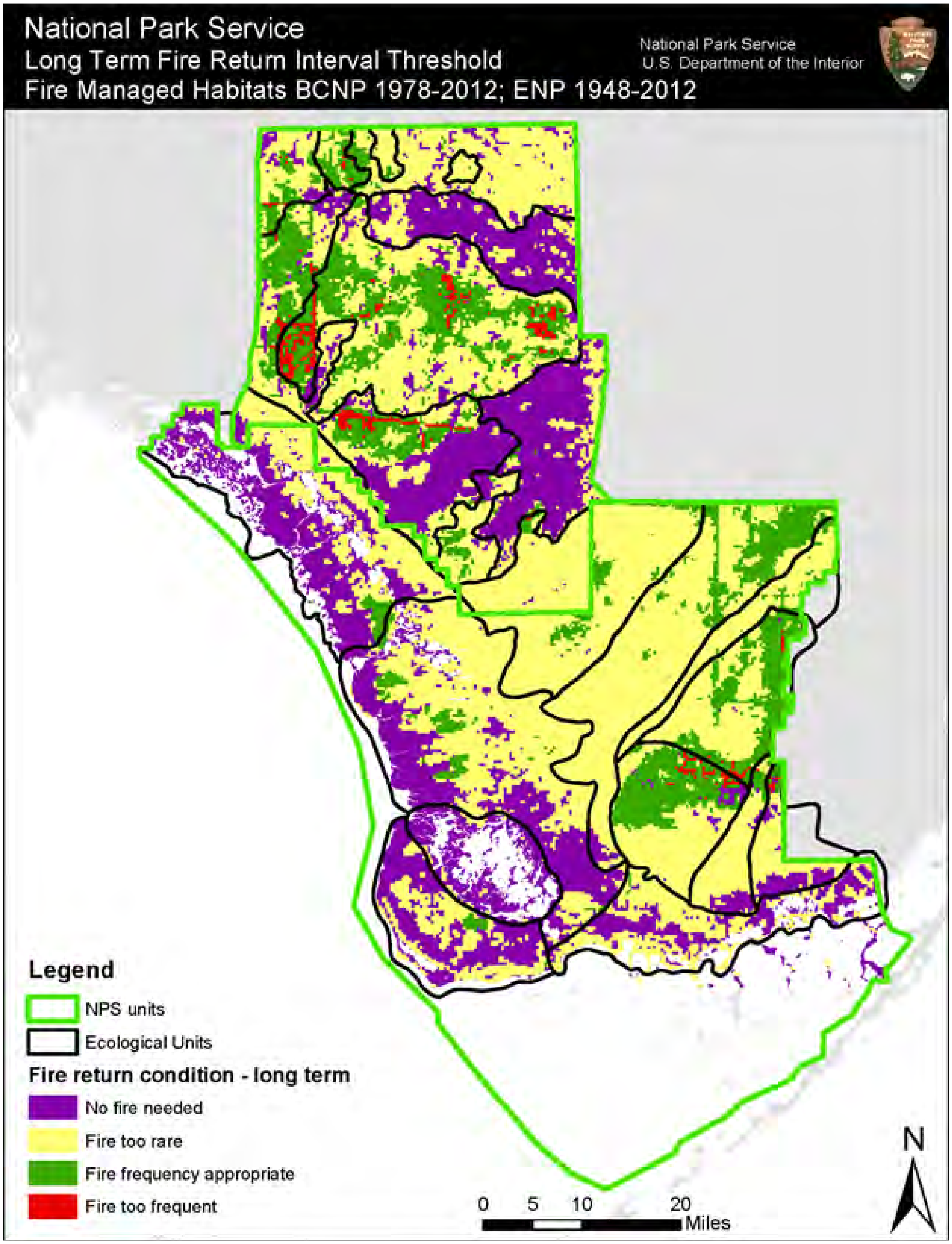


Figure 3-13. A map depicting fire return conditions (relative to a desired return time) within the twenty ecological regions of BCNP and ENP. Note, the predominance of no fire needed areas in a few regions.

A decision to include the University of Georgia vegetation map in this analysis is made in order to demonstrate the process of summarizing historical fire information using a vegetation map. Even though concerns about the accuracy of the vegetation map exist, it is expected that this analysis will be updated in the near future through the RECOVER vegetation mapping project to produce an improved vegetation map that has known accuracy above the 80% threshold required for maps produced by the NPS. The key issue to note is that the adaptive management process is designed to use information with known challenges to evaluate management decisions. The information synthesis and analysis that supports adaptive management should repeat at regular intervals, and information quality is expected to improve over time. An improved vegetation map for ENP and BCNP that is accuracy assessed will address one of the main uncertainties with the results presented here.

To summarize briefly, these are the main caveats/limitations associated with this analysis:

- Although fire perimeters are represented, the patchiness of burns within perimeters are not represented in this analysis.
- A few fire perimeters (probably less than ten out of almost 5,000 fires) are not accurately represented.
- Some fires may have occurred that are not present in the database. The sense is that this is a small number of fires covering a small area (relative to the size of the two NPS units).
- The grid size that was used to summarize fire and vegetation information delivers a representation of fire patterns that needs to be interpreted carefully.
- The vegetation map used to identify locations where fire management is focused has not been comprehensively accuracy assessed. The limited assessment of accuracy that did occur indicated that habitats were misidentified 22% to 38.5% of the time. This is a significant concern, and as a result, the quantification of performance of fire return time should be treated very cautiously.
- Areas that are not targeted for fire management are listed as "no fire needed." Fire perimeters may surround these areas, but the vegetation type cannot be confirmed, so these areas were simply removed from consideration relative to the target fire return time.

Fire Return Time

Fire return times calculated over the entire period of record indicate that fires are occurring at regular intervals over large portions of both ENP and BCNP. Fire return times are shortest near roads and least frequent (or not recorded) throughout the coastal areas and in the long hydroperiod wetlands in the interior of both NPS units. Fire return times appear to be generally shorter in BCNP, where nearly 35% of the preserve has a long-term fire return time of ≤ 8 years, compared to ENP where < 10% of the park occurs in this same fire return time category (**Figure 3-11** and **Figure 3-14**). When these two NPS units are differentiated into regions based on their physical geography, fire return times are clearly different between physiographic regions. Fire return times are longest in areas along the coast:

Whitewater Bay, Coastal Swamps and Lagoons 1, Ten Thousand Islands, Coastal Swamps and Lagoons 2, and Cape Sable; and fire return times are shortest in interior uplands and prairie units: Interior Uplands; Northern Uplands; Southern Uplands; Strands and Pinelands; Marshes, Strands, Sloughs and Uplands; Southwestern Marl Prairie; and the Taylor Slough Headwaters (includes the Long Pine Key region in ENP) (**Figure 3-14**).

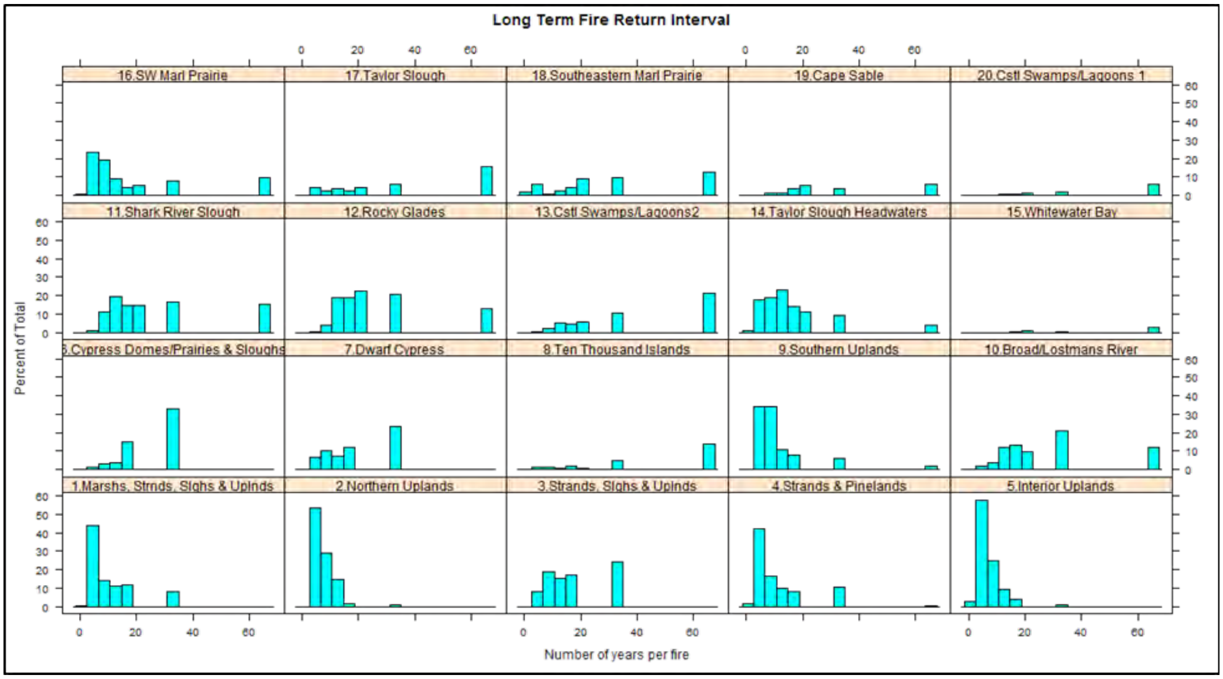


Figure 3-14. A histogram is used to summarize the relative frequency of long-term fire return times observed in discreet physiographic regions of ENP and BCNP. The proportion of 400 m-by-400 m grid cells occurring in four-year time classes is graphed for each region. Regions are identified with the same names that were presented in Figure 3-12.

<u>*Long Time Return Times Relative to Goals*</u>

The next step is to consider how fire return times are performing relative to the goals set by fire managers. Not enough fire (over the entire period of record) is the most common category occurring in both BCNP and ENP (**Figure 3-15**). At least 28% of lands in ENP contain vegetation types that do not require fire management, while just over 20% of BCNP contains vegetation types that do not require fire management. Almost 25% of BCNP lands and ~12% of ENP lands exhibit long-term fire return times that are consistent with current fire management goals; and areas that burn too frequently are quite uncommon (~ 3% of BCNP and < 1% of ENP; **Figure 3-15**).

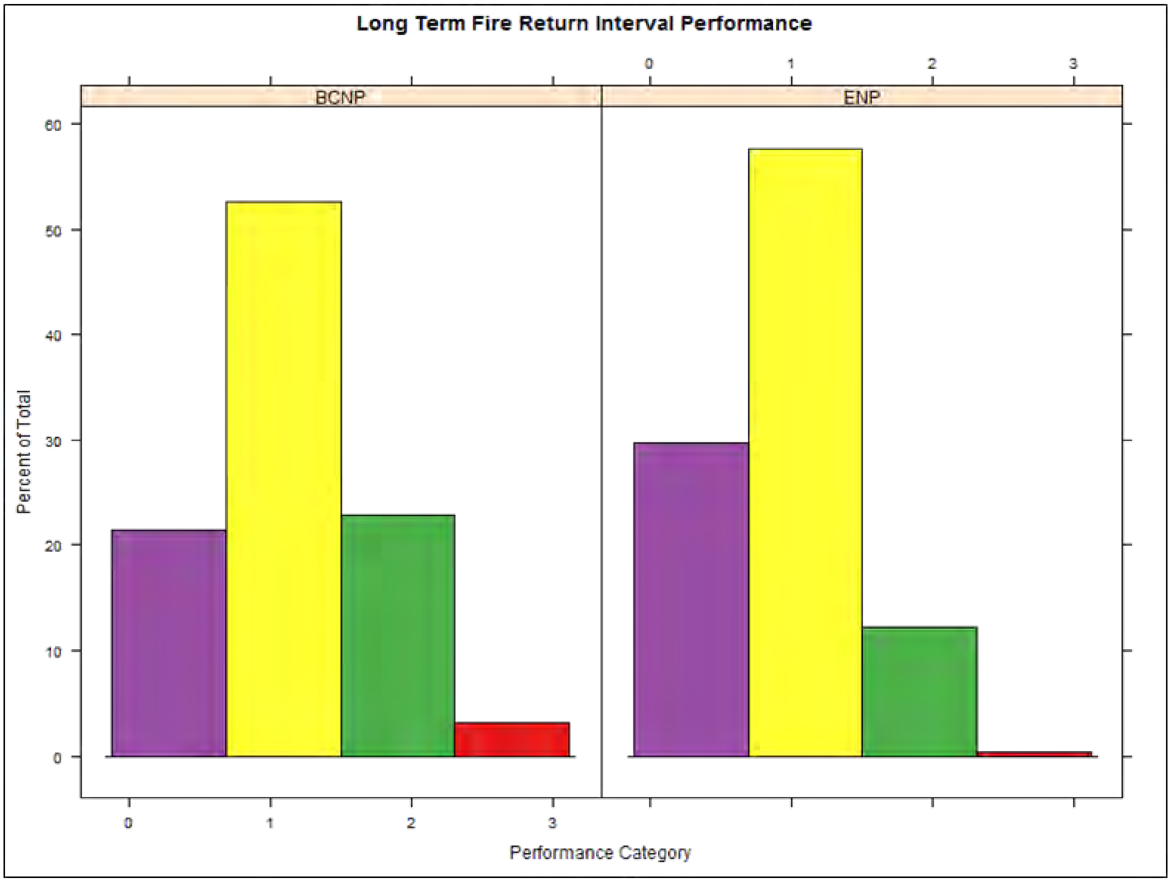


Figure 3-15. A histogram is used to summarize the relative frequency of different long-term fire return performance categories for ENP and BCNP. The proportion of 400 m-by-400 m grid cells occurring in each category is graphed. Categories are the same as depicted in Figure 3-13: 0 = "No fire needed" (purple); 1 = "Fire too rare" (yellow); 2 = "Fire frequency appropriate" (green); 3 = "Fire too frequent" (red).

The absence of a need for fire is clearly demonstrated in many of the coastal areas and in the very wet areas in the interior. These areas have large values for Performance Category zero - "No fire needed" (**Figure 3-16**). For those areas where fire is necessary, the pattern that is observed across the ENP and BCNP holds true within nearly every management unit, the "Fire too rare" category is the most frequent occurrence.

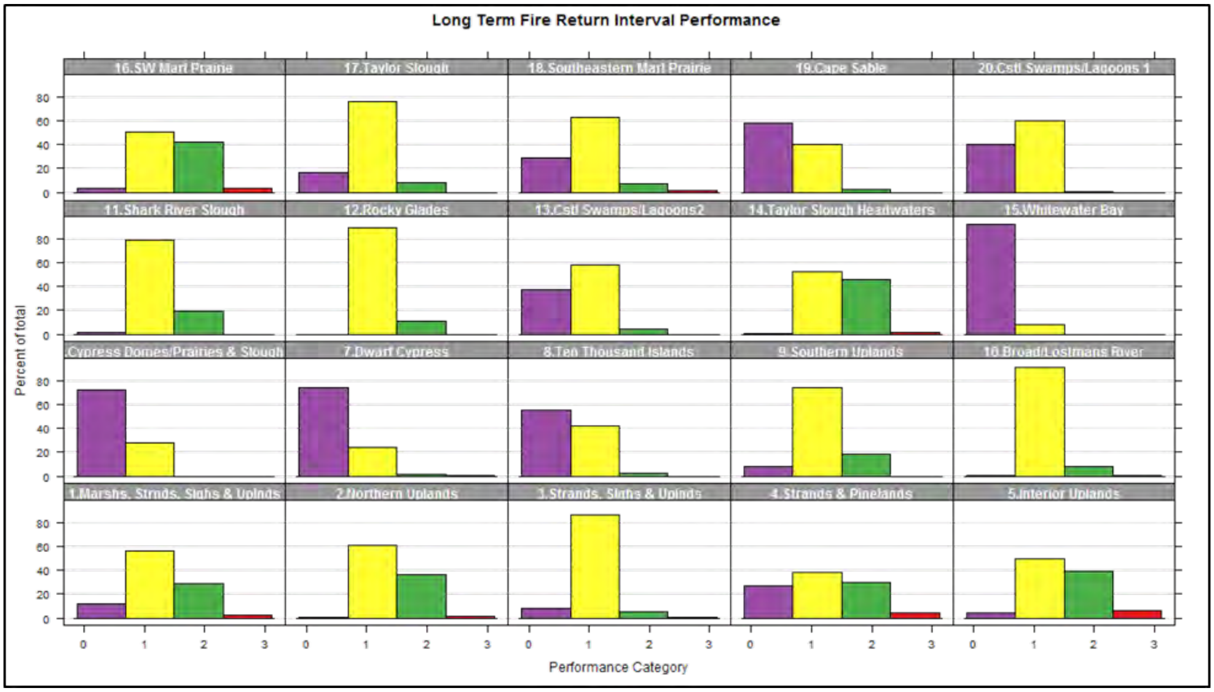


Figure 3-16. A histogram is used to summarize the relative frequency of different long-term fire return performance categories in discreet physiographic regions of ENP and BCNP. The proportion of 400 m-by-400 m grid cells occurring in each category is graphed for each region. Categories are the same as depicted in Figure 3-13: 0 = "No fire needed" (purple); 1 = "Fire too rare" (yellow); 2 = "Fire frequency appropriate" (green); 3 = "Fire too frequent" (red).

Discussion

This brief document represents one of the first attempts to reconcile the historical data set of fire history in ENP and BCNP with the current management. Because it is a first step, the results that are presented should be interpreted cautiously, and used to gain insight into the status of fire management in these areas rather than a grade or "scorecard" for fire management. Perhaps the most important result is the evidence that these types of maps and region-specific summaries can be processed and presented for review in a few weeks using ArcGIS supported by MSAccess **(**Microsoft, Redmond, Washington) and R-Studio software (R-Studio, Boston, Massachusetts).

Opportunities to address the key uncertainties/limitations associated with the analysis presented here include the following:

- Comparing the proportion of fire adapted communities that are identified in updated/improved vegetation mapping efforts.
- Developing additional summary metrics for fire that allow us to evaluate other important aspects of fires that are considered as a part of the fire management program: hydrology, water quality, wildlife, special status species, critical habitats, cultural resources, wilderness

characteristics, historical land use, park operations, urban-wildland interface issues, visitor uses and experience, and air quality issues are all subjects where development of one or more evaluation metrics may be appropriate.

- Focus on more recent information, and development of metrics that can be used to schedule management priorities and actions in the near future (Hiers et al. 2003).

- Integrate the evaluation of fire into the evaluation of other management programs, such as exotic plant species management, wildlife protection program, etc.

The first step in pursuit of these four opportunities for improvement involves a discussion among fire managers, natural resource managers, and technical staff to ensure that the different perspectives brought forward by these groups can be effectively quantified at a large spatial scale. Subsequent analysis should be based on priorities established in this initial discussion. Taking these steps is a normal part of the early stages in development of an adaptive management process.

Many of the linkages between changes to the regional hydrologic management system and the history of fire occurrence are not effectively evaluated in this document, although they are considered as part of the practice of fire management. Key issues like changes in fire seasonality, fire severity, or the number of severe fires should be explored with quantitative analysis. What has been evaluated in this report is the proportion of land area within each park that contains habitats that are targeted for fire management and identifying the locations where most of the fire managed lands occur. This is a good first step for evaluating fire management programs. A very interesting next step for these parks could be to use vegetation succession models (like Everglades Landscape Vegetation Succession Model [ELVeS]; Supernaw et al. 2011) to identify predicted shifts in the proportion of fire adapted communities that result from changes in water availability from CERP.

Taking more steps in conducting quantitative evaluations of fire history and potential effects of watershed-scale restoration actions can help identify locations where changes to fire management are likely to occur. It also helps natural resource managers effectively communicate how restoration actions can reduce certain types of management burdens. In the case of fire management, one can reasonably expect that the introduction of more water into a national park is likely to alter the proportionality of vegetation types within the park, and should reduce the fraction of the park that is dominated by species that are fire adapted. This would occur because the plant communities observed in areas that are rehydrated should shift to wetland community types (like sloughs) that do not require frequent fire management. Identifying when management burdens might be reduced and where the reductions are likely to occur may be important for estimating the costs and benefits that result from watershed-scale restoration.

It is essential to note that the need for fire management is certain to continue, and that the occurrence of fire is an important aspect of driving transitions from degraded to restored conditions. The desired outcome for areas where significant shifts in hydrology are expected is an ecosystem that is well hydrated, experiences fewer days where extremely dry conditions prevail, and experiences frequent fires that are less severe than what has been observed in the past six decades. This type of fire condition is expected to produce a mosaic of vegetation types within a landscape that exhibits a greater degree of heterogeneity than is present in areas that are considered degraded today (like the East Everglades Addition Lands).

WADING BIRDS – SYSTEMWIDE

The 2013 *South Florida Wading Bird Report* (Cook 2013) lists status and trends for wading birds for the entire SFWMD area including GE, LO, portions of NE, SCS, and the Kissimmee River Basin (**Figure 3-17**). In summary, the 2013 breeding season was an average year for wading birds in south Florida in terms of total number of nests. The estimated nest count was 48,291, which is a 57% improvement on the average for the last three years and is similar to the average for the past decade.

There were marked differences in nesting effort among species. The improved nesting effort in 2013 was largely due to increases by white ibises (*Eudocimus albus*), wood storks (*Mycteria americana*), and great egrets (*Ardea alba*), while the smaller herons and egrets (snowy egret [*Egretta thula*], tricolored heron [*Egretta tricolor*], and little blue heron [*Egretta caerulea*]) experienced another poor year. Of particular note was the limited nesting by little blue herons and tricolored herons, which continues a steep and steady decline in nesting activity for these species (down 50% and 21%, respectively, relative to the eight-year average).

Also of concern is the failure by wood storks to nest at Corkscrew Swamp Sanctuary, a traditional and important nesting area for this species. Storks have nested there only once (2009) during the past seven years.

The unusual inland nesting exhibited by roseate spoonbills (*Ajaja ajaja*) in WCA 3A during 2011 and 2012 did not occur this year but a comparable nesting effort was reported in central ENP. In Florida Bay, spoonbill nesting effort and success was similar to last year, which continues a relative improvement on previous years.

Fledging success was generally poor for all species in 2013, largely due to a series of rain-induced water level reversals. The exception was the wood stork, which was largely successful in the WCAs and northern ENP, but not in coastal ENP.

This multi-agency annual report continues to be an essential resource for guiding Everglades restoration strategies and weekly operational decisions, and is an effective tool for communicating important SFWMD information to the general public. Wading birds are good indicators of overall south Florida ecosystem health. More detailed wading bird status and trends for LO and GE are included in Chapter 7: Greater Everglades of this SSR.

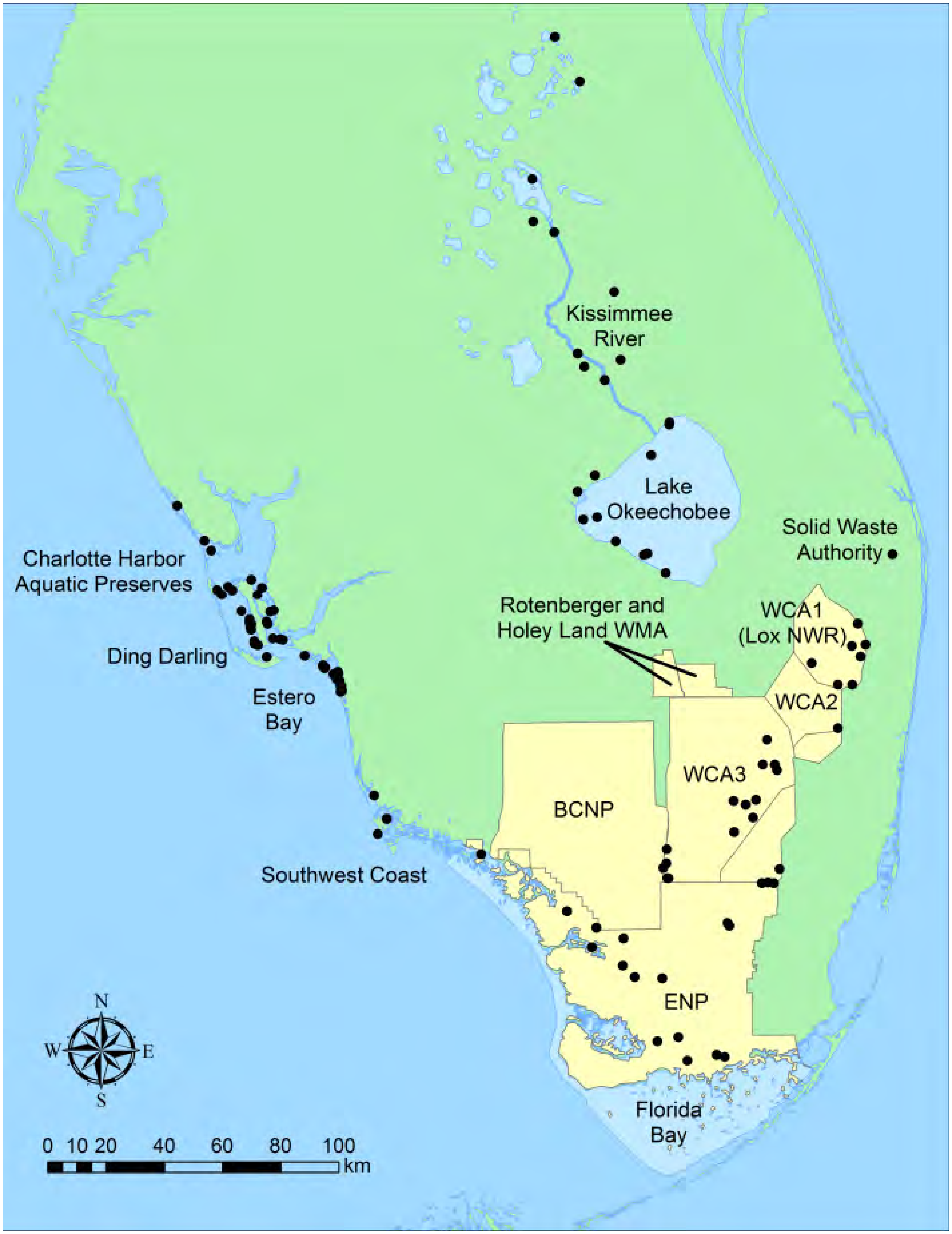


Figure 3-17. Locations of wading bird colonies with ≥ 50 nests in south Florida in 2013.

SYSTEMWIDE ECOLOGICAL INDICATORS

The *Systemwide Ecological Indicators for Everglades Restoration* report provides information for the South Florida Ecosystem Restoration Task Force on the status of key components of the ecosystem and how they are responding to restoration and management activities of CERP and non-CERP restoration projects (Brandt et al. 2012). It is a broad view of the ecosystem and is focused around 11 ecological indicators that were carefully selected to focus the ability to assess the success of Everglades restoration from a systemwide perspective. All but one of the ecological indicators, invasive exotic plants, are interim goals and part of the RECOVER MAP (RECOVER 2009). The report uses a red, yellow, green traffic light format to convey status and trends of each indicator in a quick, understandable way. Stoplight colors are assigned based on criteria explained in a special issue of the scientific journal *Ecological Indicators* (Doren et al. 2009). The full report (Brandt et al. 2012) with more detail for each indicator is available at http://www.sfrestore.org/documents/2012_system_wide_ecological_indicator_report.pdf.

Table 3-8 provides a snapshot of the status of each indicator by geographic region (listed from north to south) for the last five years. Results shown here are consistent with an assessment done by the National Research Council (2012), reflecting the continued patterns of severely altered hydrology throughout the ecosystem. An exception is WY 2011 in Lake Okeechobee where the nearshore zone submersed aquatic vegetation exceeded the target level because of successive years where the lake was near or below the lower end of the ecologically desired stage envelope with concomitant improved light penetration. Other chapters within this SSR provide more in depth status and trends analyses that help back up the colors in this table.

Table 3-8. Indicators at a glance.

Stoplight Legend

Red Substantial deviations from restoration targets creating severe negative condition that merits action.

Yellow Current situation does not meet restoration targets and may require additional restoration action.

Green Situation is within the range expected for a healthy ecosystem within the natural variability of rainfall. Continuation of management and monitoring effort is essential to maintain and be able to assess "green" status.

Black No data, or inadequate amount of data, were collected due to lack of funding.

	WY 2008	WY 2009	WY 2010	WY 2011	WY 2012
Lake Okeechobee					
Invasive Exotic Plants Species	Yellow	Yellow	Yellow	Yellow	Yellow
Lake Okeechobee Nearshore Zone Submersed Aquatic Vegetation	Red	Red	Yellow	Green	Red
Northern Estuaries					
Invasive Exotic Plant Species	Yellow	Yellow	Yellow	Yellow	Yellow
Eastern Oysters	Yellow	Yellow	Yellow	Yellow	Yellow
Greater Everglades					
Crocodilians	Red	Red	Red	Red	Black
Fish and Macroinvertebrates (WCA 3 and ENP only)	Yellow	Yellow	Yellow	Yellow	Red
Invasive Exotic Plants	Yellow	Red	Red	Red	Red
Periphyton and Epiphyton	Yellow	Yellow	Yellow	Yellow	see footnote a
Wading Birds (White Ibis and Wood Stork)	Red	Red	Red	Red	Red
Southern Coastal Systems					
Crocodilians	Red	Red	Red	Red	Red
Southern Estuaries Algal Blooms[b]	Yellow	Green	Green	Yellow	Yellow
Florida Bay Submersed Aquatic Vegetation	Yellow	Yellow	Yellow	Yellow	Yellow
Invasive Exotic Plants	Yellow	Yellow	Yellow	Yellow	Yellow
Juvenile Pink Shrimp[c]	see footnote d			Yellow	Yellow
Wading Birds (Roseate Spoonbill)	Red	Yellow	Red	Red	Red (see footnote e)
Wading Birds (White Ibis and Wood Stork)	Red	Red	Red	Red	Red

a. Species composition data is not available.

b. Algal bloom indicator values are for calendar years 2007 through 2011, roughly corresponding to the water years shown.

c. The status for juvenile pink shrimp (*Farfantepenaeus duorarum*) contains information for data collected for September–October.

d. Data from WY 2008–WY 2010 were used as a baseline.

e. Prey community data have not yet been processed.

ADAPTIVE MANAGEMENT

The CERP Adaptive Management Program and the role of the MAP in supporting the program have been further defined with recent program-level guidance and project adaptive management plans. The *2011 CERP Adaptive Management Integration Guide* (RECOVER 2011b) identifies nine activities to integrate adaptive management into the CERP program and project-level implementation process of planning, design, construction, and operations (**Figure 3-18**). These activities were formalized into *CERP Guidance Memorandum 56: Guidance for Integration of Adaptive Management into CERP Program and Project Management* (USACE and SFWMD 2011b).

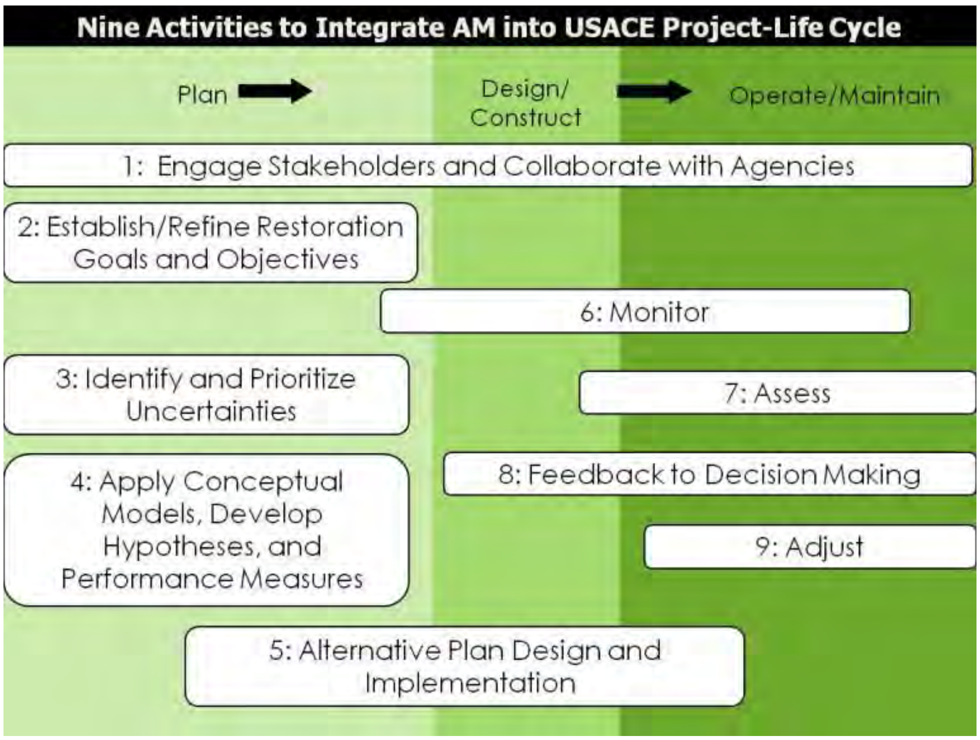


Figure 3-18. Nine activities to integrate adaptive management (AM) into USACE project-life cycle process (planning, design, construction, operations and maintenance).

A CERP program-level adaptive management plan has been partially drafted and is planned for completion in 2014. Most components of the program-level adaptive management plan have already been developed and referenced in the draft plan (**Figure 3-19**).

The remaining tasks are to (1) consolidate a list of priority programmatic uncertainties and strategies to address them by CERP projects, operations, and RECOVER science, and (2) complete the programmatic management option matrices for all regions in CERP (see example from the 2009 SSR (RECOVER 2011a): http://www.evergladesplan.org/pm/ssr_2009/ssr_figure_pdfs/fig_am_matrix.pdf).

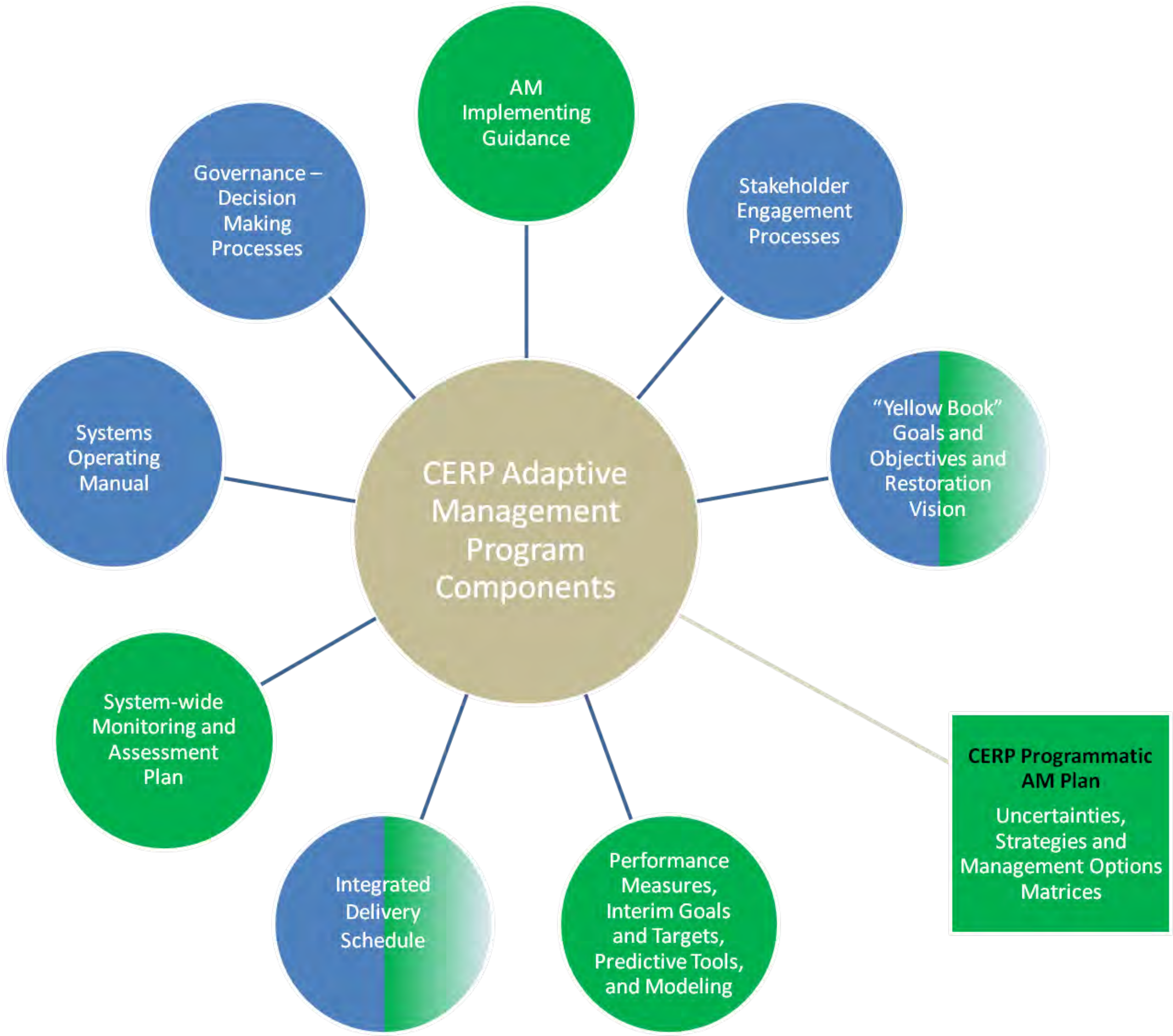


Figure 3-19. Diagram of CERP Adaptive Management (AM) Program Components. Green – RECOVER product or responsibility; blue – programmatic product or responsibility; blue/green – shared responsibility; circle – existing; square – new.

The RECOVER MAP and SSRs are important components of the CERP program-level adaptive management approaches. As stated in the 2009 SSR, the SSR (Activity 7; **Figure 3-18**) accomplishes the following: (1) presents the data generated by the MAP (Activity 6; **Figure 3-18**) to establish pre-CERP reference conditions of ecological stressors, effects, and attributes; (2) reports ecosystem status and trends and interim goals and interim targets; and (3) addresses key questions about why natural system response to restoration has changed in relation to the implementation of restoration projects. The MAP also supports the applied science strategy (RECOVER 2004) that contributes to the following: (1) helps identify and address uncertainties (Activity 3; **Figure 3-18**); (2) improves understanding of goals and objectives (Activity 2; **Figure 3-18**); (3) updates the CEMs and working hypotheses about how the system will respond to restoration projects and operations (Activity 4; **Figure 3-18**); and (4) provides the data needed to develop predictive models and performance measures used to evaluate project plans (Activity 4; **Figure 3-18**). These models are used to help develop incremental (interim goals and targets) and long-term restoration targets to either corroborate expected restoration progress, both systemwide and project-level, or to report lack of restoration progress and the need for adaptive management actions to improve restoration performance. MAP monitoring is also used by multiple CERP projects as part of their project-level adaptive management plans:

- Broward County Water Preserve Areas (USACE and SFWMD 2012a): http://www.evergladesplan.org/pm/projects/project_docs/pdp_45_bc_wpa/final_pir_2012/032212_vol_3_bcwpa_pir_annex_e.pdf
- Biscayne Bay Coastal Wetlands (USACE and SFWMD 2011a): http://www.evergladesplan.org/pm/projects/project_docs/pdp_28_biscayne/010612_fpir/010612_vol_3_annex_e.pdf
- Decomp Physical Model for Decompartmentalization of WCA 3 (USACE and SFWMD 2010): http://www.evergladesplan.org/pm/projects/project_docs/pdp_12_decomp/060410_decomp_ea_final/april_2010_decomp_ea_app_e_bk.pdf
- CEPP (USACE and SFWMD 2013): http://www.evergladesplan.org/pm/projects/project_docs/pdp_51_cepp/dpir/082813_cepp_dpir_annex_d_adaptive_mgmt.pdf

The following discussion provides an example of how MAP monitoring will be used to support CERP adaptive management in the Caloosahatchee River Estuary (CRE) and LO. CERP restoration projects will restore more natural flows through the estuary to reestablish more natural salinity ranges and promote the return of healthy submerged aquatic vegetation (SAV), oyster, and fish communities. As explained in the CEM for the Caloosahatchee Estuary (Barnes 2005) and RECOVER's Northern Estuaries salinity performance measure (RECOVER 2007a), current issues relate to lack of flows (< 450 cubic feet per second [cfs]) during the dry season, which increases high salinity conditions that move up the Caloosahatchee River and impact upstream SAV resources that prefer lower salinities (see **Figure 3-20** for an example of degraded SAV habitat). In addition, the other issue is that too much flow during the wet season of very wet years, which impacts oysters and other SAV species (**Figure 3-21**).

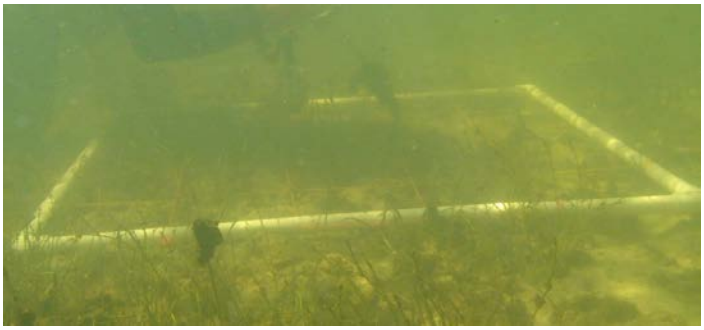
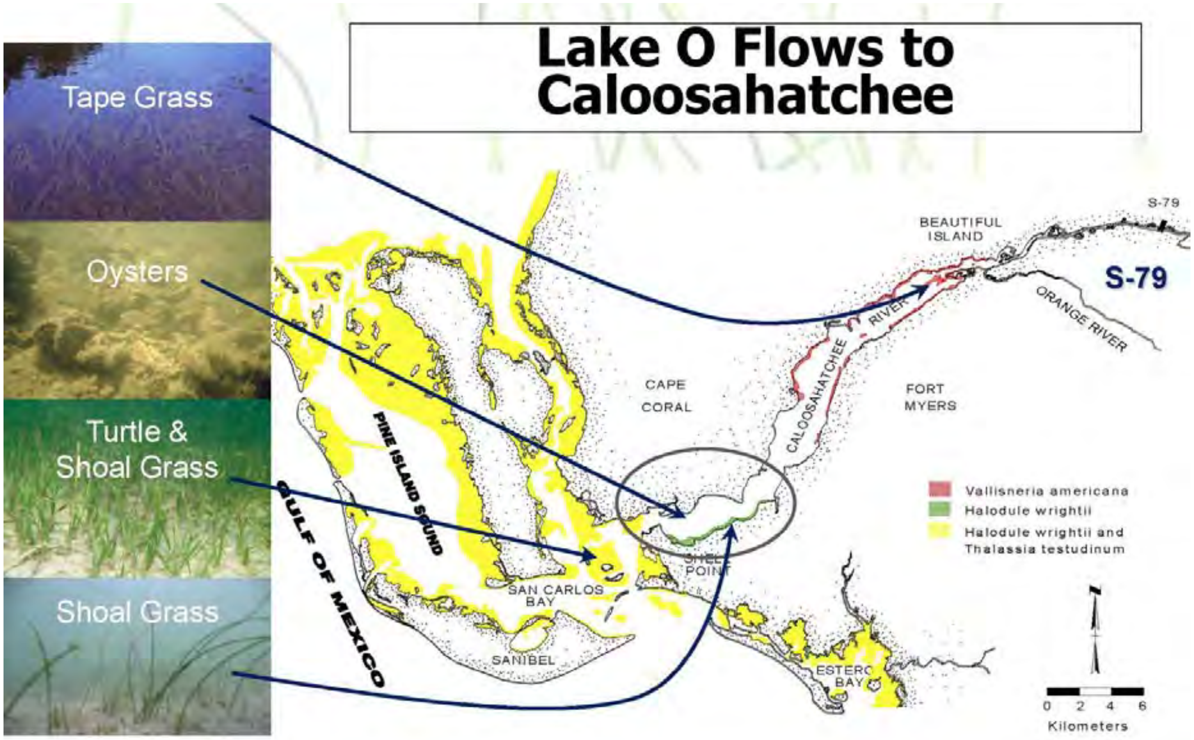


Figure 3-20. Photograph of degraded tape grass (*Vallisneria americana*).



Tape Grass		Shoal Grass		Oysters		Turtle & Shoal Grass	
<10 psu	**>450 cfs**	>6 psu	**<2800 cfs**	>3-5 psu	<4000 cfs	>20 psu	**<4500 cfs**

Figure 3-21. Photographs of SAV and oyster resources in the CRE and corresponding salinity threshold requirements.

(Note: cfs – cubic feet per second; Lake O – Lake Okeechobee; psu – practical salinity units)

In the present, low flow (monthly average < 450 cfs) and high flow (monthly average > 2,800 cfs) exceedences have occurred 24% and 19% of the time based on SFWMM and Regional Simulation Basins Model modeling (41-year period of record), respectively (**Table 3-9**).

This trend is not expected to improve in the future without CERP projects. CERP projects like the Caloosahatchee River (C-43) West Basin Storage Reservoir are expected to reduce low flow and high flow exceedences to 5% and 16% of the time. MAP monitoring and assessments, through the SSR, will be able to validate (or not) improvements in SAV species that prefer lower salinities in the upper estuary (e.g., tape grass [*Vallisneria americana*]) (**Figure 3-22**).

Table 3-9. Predicted performance of CERP projects compared to target, existing conditions (current), and future without restoration.

Performance Measure	Target	Current	Future Without	C-43	CEPP	CERP
< 450-cfs Low Flow	0%	24%	24%	5%	5%	5%
> 2,800-cfs High Flow	0%	19%	19%	16%	14%	5%


Figure 3-22. Principal investigator sampling seagrass (left photo) and picture of healthy tape grass (right photo).

CEPP is expected to provide additional reductions in high flow exceedences (down to 14% of the time). MAP monitoring will be able to show any improvements of turtle grass (*Thalassia testudinum*) and shoal grass (*Halodule wrightti*) and Eastern oysters (*Crassostrea virginica*) in the lower estuary.

Recent assessments reveal improvements to SAV and oysters in the CRE. This is likely due to lower rainfall patterns and the Lake Okeechobee regulation schedule negating the need to release water from Lake Okeechobee. However, historically and most recently in WY 2013 and early indications from WY 2014 reveal that the current Lake Okeechobee and water management system storage is not able to handle high rainfall years and water managers need to make high flow releases to the estuaries in order to reduce the risk of Herbert Hoover Dike failure (flood risk management). Only until the full storage envisioned by CERP is completed, will there be a reduction in high flows (down to 5% of the time) that would increase confidence in achieving sustainable restoration improvements to oyster and seagrass resources in all areas of the CRE, such that they become more resilient to rare high and low flow events.

MAP also supports the CERP Adaptive Management Program in addressing key uncertainties about why performance has or has not changed. In the CRE case, if there is no measurable response in tape grass in the upper estuary after the Caloosahatchee River (C-43) West Basin Storage Reservoir is completed, or an improvement in oyster response (decreased mortality, increased growth, and less disease) in the lower estuary after CEPP is implemented, then it will be necessary to review the list of additional stressors that maybe preventing realization of expected restoration benefits. RECOVER monitoring of nutrients and sediments or lack of adequate light penetration may reveal that water quality has not improved sufficiently to observe the expected tape grass and oyster response. Alternatively, monitoring results may indicate improvements and that further reduction in high flows is needed or the low flow base threshold should be higher than 450 cfs.

INTERIM GOALS REVIEW

Background on Interim Goals

As defined in §385.3 of the Programmatic Regulations (DOD 2003), interim goals are to be used for two major purposes for the CERP. First, the interim goals are to be used in CERP planning as a guide for project design, as a criterion for development of CERP project scheduling, and to assist in comprehensive plan updates and modification using adaptive management. Second, they are used as benchmarks for the comparison with field information during the implementation and operation of CERP projects in order to assess whether ecosystem performance is moving towards CERP restoration goals. If CERP projects are not moving towards interim goals, then an adaptive management strategy should be undertaken. In this context, interim goals are expected to play a significant role in informing the adaptive management program for CERP, and are used to report to congress and others on restoration progress.

Interim goals are “a basis…for periodically evaluating the accuracy of predictions of system responses to the effects of the [Comprehensive Everglades Restoration] Plan” (§385.3), and “shall reflect the incremental accomplishment of the expected performance level of the Plan” (§385.38(c)).

Guidance from the Programmatic Regulations states the following:

- “Purpose. Interim goals are a means by which the restoration success of the Plan may be evaluated at specific intervals by agency managers.” and “The [United States Army] Corps of Engineers and the South Florida Water Management District shall sequence and schedule projects as appropriate to achieve the interim goals and the interim targets established pursuant to §385.39 to the extent practical given funding, technical and other constraints.” §385.38(b)
- “The [United States Army Corps of Engineers] and …sponsors shall implement the Plan in a manner to continuously improve the expected performance level of the Plan…” §385.8(d)

- "Endorsement of the Plan as a restoration framework is not intended as a constraint on innovation during implementation….The adaptive management process provides a means…for improving the performance of the Plan." §385.9(c)

The *Recover Team's Recommendations for Interim Goals and Interim Targets for the Comprehensive Everglades Restoration Plan* (RECOVER 2005) was submitted to the United States Departments of the Army and Interior and the State of Florida (through the SFWMD) to be used as a reference in the development of the *Intergovernmental Agreement Among the United States Department of the Army, The United States Department of the Interior, and the State of Florida Establishing Interim Restoration Goals for the Comprehensive Everglades Restoration Plan* (USDOA et al. 2007), which is available online at http://www.evergladesplan.org/pm/pm_docs/prog_regulations/081607_int_goals.pdf. The agreement was signed in 2007 and the signatories supported, as a high priority, the continued development and refinement of the recommended indicators and interim goals contained within the RECOVER report. It was expected that RECOVER would continue to refine its recommendations for indicators and interim goals, develop predictive measures for interim goals for certain indicator conditions that lacked predictive measures, and develop incremental performance predictions for the interim goals. RECOVER was also to continue to develop and refine a strategy, which would allow RECOVER to be able to assess progress towards achieving the interim goals established by the agreement. The strategy is described in the *Monitoring and Assessment Plan (MAP), Part 2 2006 Assessment Strategy for the MAP, Final Draft* (RECOVER 2006) and the 2009 *CERP Monitoring and Assessment Plan* (RECOVER 2009).

This review of the interim goals will do the following:

- Highlight work that has been accomplished, through various agency and RECOVER efforts, towards development of ecological models that directly pertain to the current interim goal indicators.

- Examine whether the goals themselves, as stated in the intergovernmental agreement, are consistent with the current status of the MAP, as to make assessments of progress towards meeting interim goals defensible. It must be possible to measure the status of the interim goal indicators using data collected from the field.

Ecological Modeling Tools

Ecological models are tools that link ecological effects to hydrology in restoration planning. The National Research Council of the National Academy of Sciences, which conducts biennial independent scientific review of CERP, concluded in their 2010 report that "Improved species models… are urgently needed to provide more rigorous scientific support for water management decisions" (NRC 2010). Ten of the 29 interim goal indicators are primarily hydrological, and use either hydrologic field data or hydrological models for assessing restoration progress; they are not discussed here. Two of the indicators have to do with water quality (Lake Okeechobee and Everglades phosphorus) and are also not discussed in this section, as work continues outside of RECOVER to develop these types of tools. There have been 10 ecological models related to the interim goals developed to date and are summarized below. Interim goal indicators are underlined.

Worthy of note is that the entire suite of models summarized below (with the exception of Lake Okeechobee) was used extensively in the planning for CEPP. RECOVER and project delivery team members used the models to shape and refine project alternatives and perform systemwide evaluations of the alternatives. Ecological effects forecast by the models were documented in the project implementation report (USACE and SFWMD 2013), and played a significant role in the development of the biological assessment (USACE 2013) and biological opinion (USFWS in review).

Northern Estuaries Region

American Oysters in Northern Estuaries

- Oyster simulation models for the Caloosahatchee and St. Lucie estuaries are currently simplified versions of a framework derived to evaluate potential effects of increased area of oyster habitat on St. Lucie Estuary (SLE) water quality (Buzzelli et al. 2012a). The model uses an idealized oyster-salinity relationship, variable temperature, and a constant suspended solid concentration to predict oyster density.

Submerged Aquatic Vegetation in Northern Estuaries

- Shoal grass (CRE) and manatee grass (*Syringodium filiforme*) (SLE) are simplified models derived to quantify effects of variable freshwater discharge and salinity on seagrass shoot density (Buzzelli et al. 2012b).

Lake Okeechobee Region

Lake Okeechobee Aquatic Vegetation

- The SFWMD Lake and River Ecosystems Section has recently started using Lake Okeechobee stage correlations with both SAV and summer cyanobacterial abundances, then applied the scoring to examine if storage north of the lake might be ecologically beneficial. For more information, see Chapter 6: Lake Okeechobee in this SSR.

Greater Everglades Region

Aquatic Fauna Regional Populations in Everglades Wetlands

- Densities of small fishes (standard length < 8 centimeter; number per square m) are highly dependent on variation in hydroperiod. Based on 10-year time series data (1996–2006), a logistic model was parameterized to predict small fish densities based on the time between drying events. The model estimates the average density of small fish at each primary sampling unit in the Florida International University Trexler lab's CERP MAP data set (Catano and Trexler 2013).

- Catch-per-unit-effort for large fishes has been developed using 15-year time series data (1997–2012). A generalized logistic model was parameterized to predict largemouth bass (*Micropterus salmoides*) catch-per-unit-effort based on the average length of drying events in days for each study site. Large fishes in the Everglades are particularly sensitive to hydrological disturbances such as drought or drying events because they require deeper water than smaller fishes (Catano and Trexler 2013).

American Alligator

- An American alligator (*Alligator mississippiensis*) production index has been developed (SFNRC 2010a), which is an update of the 2004 Spatially Explicit Species Index Model. The model integrates five probability functions that address principal factors of alligator biology and their annual production potential. The index is the geometric mean of (1) habitat availability, (2) breeding potential, (3) courtship and mating, (4) nest building, and (5) nest flooding.

Systemwide Wading Bird Nesting Pattern

- A wood stork (*Mycteria americana*) foraging potential model (SFNRC 2010b) has been developed to predict the relative suitability of foraging conditions for wood storks within Everglades freshwater marshes during the breeding season (December 1–July 15). This model calculates wood stork foraging probabilities in Everglades freshwater marshes based on estimated water depth and recession rates. Foraging probability can either be scored in all cells in the model domain, or in cells within a specific radius (e.g., 23.4 kilometer) surrounding a known wood stork colony.

- The Wader Distribution Evaluation Modeling (WADEM) (Beerens and Noonburg 2013) is a series of models related to temporal foraging conditions and spatial foraging conditions that are joined with additional parameters to evaluate nesting success. The spatial foraging condition model was used to evaluate wading birds in the technical evaluation of CEPP for great egret (*Ardea alba*), white ibis (*Eudocimus albus*) and wood storks. This model focuses on hydrologic characteristics (model cell depth, recession rate, days since last dry down, recession reversals, spatial position, and hydroperiod) to determine the frequency of model cells used over time and provides a spatial depiction of landscape productivity (i.e., patch abundance), via annual frequency of cell use and total cell abundance.

- A wading bird nesting model developed in the Gawlik lab at Florida Atlantic University predicts the number of nests for the great egret, white ibis and wood stork (Gawlik et al. 2013). Variables considered in the model are hydrological (e.g., stage and seasonal amplitude) but represent ecological processes affected by hydrologic patterns. Three hypotheses representing these processes are that nesting is controlled by (1) spatial extent of foraging habitat, (2) production of prey in the wet season, or (3) the progressive dry down of the marsh.

Snail Kite

- A Florida apple snail (*Pomacea paludosa*) population model (Romanach 2013) has recently been used as a proxy for the Everglade snail kite (*Rostrhamus sociabilis plumbeus*); Florida apple snails being virtually the exclusive food source for the kite. The persistence of apple snails depends largely on hydrologic regime and temperature. A size-structured population model has been developed to simulate the response of apple snails to a range of water conditions that include timing, frequency and duration, water depths and temperature.

Southern Coastal Systems

Submerged Aquatic Vegetation in Southern Estuaries

- SEACOM, developed at the SFWMD, is a seagrass community ecological simulation model that assesses the impact of management strategies on the Florida Bay SAV community. The model predicts biomass and species composition of *Thalassia, Halodule* and *Ruppia*. Abundance of each species at each site modeled (e.g., Taylor River, Trout Cove, and Whipray Basin) is presented as aboveground standing crop in grams carbon per meter squared versus time during the model simulation period (Madden 2013).

Juvenile Shrimp Densities in Florida Bay and Biscayne Bay

- A shrimp production index simulates growth, survival, and potential harvests from a specified monthly cohort as a function of salinity and temperature. Model simulations are repeated each year of the model period of record (1965–2005) to produce a time series of growth and survival for a cohort of shrimp entering the bay in a given month.

American Crocodile

- An American crocodile (*Crocodylus acutus*) hatchling growth and survival salinity index (Brandt 2013) is calculated for August through December, the period following hatching when hatchlings are most vulnerable to high salinities. If hatchlings survive to December, they are generally large enough to tolerate higher salinities, and other factors such as food become more important. Based on the integration of field and laboratory studies, salinity < 20 practical salinity units (psu) was assigned the highest index score of 1; salinity between 20 to < 30 psu, a score of 0.6; 30 to < 40 psu, a score of 0.3; and ≥ 40 psu, a score of 0. Salinity data provided for the Marine Monitoring Network at Joe Bay, Trout Cove, and Madeira Bay (the stations closest to where the highest densities of crocodile nests are), and Long Sound, Little Blackwater Sound, Terrapin Bay and Garfield Bight (generally closer to shoreline stations in areas where crocodiles could occur) are used in calculating the index.

Other Ecological Models Not Directly Related to Interim Goals

There have been other ecological models developed that do not directly relate to the current interim goal indicators. These models (and their indicators) may be taken into consideration when RECOVER revises its interim goals technical report in the coming years. These models are as follows:

- **ELVeS –** The Everglades Landscape Vegetation Succession model, developed by the South Florida Natural Resource Center (2011a), is a spatially-explicit simulation of vegetation community response to environmental changes.
- **Marl Prairie Indicator –** This indicator uses the return frequency of hydrologic metrics to compare existing and target conditions in the marl prairies to conditions under hydrologic restoration alternatives (SFNRC 2011b).
- **Juvenile Spotted Seatrout in Florida Bay –** This habitat suitability index model (Kelble et al. 2011) uses a logistic regression to assess how the frequency of occurrence of juvenile spotted seatrout (*Cynoscion nebulosus*) varies in response to environmental parameters (turbidity, temperature, salinity, and spatial coverage and density of three species of seagrass).

Consistency of Interim Goal Statements with the Monitoring and Assessment Plan

As stated above, to make assessments of progress towards meeting interim goals defensible, it must be possible to measure the status of the interim goal indicators using data collected from the field. A summary of interim goals and their links to MAP hypothesis clusters were presented in the 2009 SSR (RECOVER 2011a). In this 2014 SSR, the review of the interim goal statements from the 2007 intergovernmental agreement was undertaken further to understand their consistency with the monitoring parameters that are funded in the current MAP program. Six of the 17 ecological interim goals are either inconsistent with the current MAP program (e.g., American oyster) or RECOVER no longer has funding to collect these data.[2] Interim goal ecological indicators/MAP monitoring parameters found to be inconsistent are discussed in the following sections, as are other factors related to current MAP status.

[2] The review focused on the ecological interim goal indicators. Assessments of the 10 hydrologic interim goal indicators may be accomplished through data collection (gages, stations) by SFWMD, ENP and others. The MAP partially funds EDEN, which assists with real-time water level monitoring, ground elevation modeling and water surface modeling. EDEN enables the assessment of biotic responses to hydrologic change.

Northern Estuaries Region

American Oysters in Northern Estuaries

- The goal statement is to increase areal coverage of oysters.
- American oysters are not mapped on a regular basis in the NE region (i.e., areal extent). Oyster ecology monitored at selected sites includes oyster density, reproduction and recruitment, juvenile growth and survival, physiological condition, and distribution and frequency of oyster disease.
- This information is important to understand why oysters may respond or not to CERP restoration projects, but does not address the actual acreage of oysters produced.
- The goal statement for oysters is inconsistent with the current MAP parameters.

Lake Okeechobee Region

Water quality phosphorus concentrations, phytoplankton, emergent vegetation and SAV are all monitored by the SFWMD and not funded by the MAP. See Chapter 6: Lake Okeechobee for a more detailed discussion.

Greater Everglades Region

Everglades Total Phosphorus

- The MAP does not fund water quality monitoring in the Everglades. Regulatory monitoring is carried out by the SFWMD and others.

Periphyton Mat Cover, Structure and Composition

- Periphyton presence, biomass and nutrient concentration are funded by the MAP. However, due to reductions in budget, periphyton composition is no longer analyzed.

American Alligator

- No MAP data was collected post-2012. Due to reductions in budget, MAP no longer funds alligator monitoring.

Snail Kite

- Endangered species monitoring is not funded by the MAP. Endangered species monitoring is generally carried out by other restoration project funding related to Endangered Species Act compliance or as part of the Department of Interior's species recovery programs.

Southern Coastal Systems

Juvenile Shrimp Densities in Florida Bay and Biscayne Bay

- Due to reductions in budget, shrimp density monitoring in Florida Bay is not funded. Limited nearshore monitoring remains in Biscayne Bay.

American Crocodile

- No MAP data was collected post-2012. Due to reductions in budget, MAP no longer funds crocodile monitoring.

Florida Bay Algal Blooms

- The MAP does not fund monitoring for algal blooms in the bay; however, there is water quality monitoring performed by SFWMD and others.

Conclusions

In a long-term restoration program such as CERP, it is important that goals are set with a means of tracking the goals over time as restoration projects are implemented. RECOVER issued its technical report in 2005 to facilitate creating the intergovernmental agreement in 2007 and also developed and began implementation of the MAP to track restoration progress. This section provides a review of new ecological modeling tools that will aid in future RECOVER efforts to update its technical report. An analysis of MAP methodologies and other factors is provided in the context of the current set of interim goal indicators.

The goal of Everglades restoration is to recover an Everglades-type of ecosystem, recapturing its unique defining characteristics. Implementation of the CERP–and now CEPP, which is a subset of CERP projects–requires a significant public investment over a sustained period of time. In order to ascertain that the program is indeed being successful in moving the system towards its restoration goals, a scientifically rigorous assessment plan must also be maintained.

MONITORING AND ASSESSMENT PLAN SUSTAINABILITY

RECOVER was tasked by the Design Coordination Team, comprised of USACE and SFWMD funding agencies, in November 2010 to optimize CERP-wide monitoring and prioritize the MAP monitoring components (or monitoring projects) due to a reduction in funds available in FY 2012 for MAP monitoring.

This effort is summarized in the 2012 SSR (RECOVER 2012), which is available at http://www.evergladesplan.org/pm/ssr_2012/ssr_main_2012.aspx. The process built upon previous efforts to coordinate, optimize, and streamline monitoring projects. A varity of criteria were applied as a part of this optimization effort. The results of the process were documented for use during future MAP prioritization in the *RECOVER MAP Regional and System-wide Prioritization Process* document dated August 2011 (RECOVER 2011c).

As a part of the prioritization process, all MAP contracts were reduced to the greatest extent practicable while maintaining scientific integrity to meet the needs of the MAP. Then, all MAP contracts were ranked first as Tier 1, then as Tier 2 and Tier 3; Tier 1 being of highest priority. All Tier 1 MAP contracts were deemed a priority and had to be funded; therefore all Tier 1 MAP contracts were given a Priority of 1. Tier 2 and Tier 3 MAP contracts were also given a priority for funding; however, all Tier 2 or Tier 3 contracts did not receive a Priority of 2 or a 3. These contracts were ranked beginning with 2 up to 15. A detailed MAP prioritization spreadsheet was developed that documents the subsequent changes in monitoring including the rank assigned to each contract. As a result of the budget reductions, the MAP monitoring components were either reduced in scope or not funded for FY 2012, FY 2013 and FY 2014. **Table 3-10** documents changes in monitoring and the ability of the FY 2012 MAP monitoring components to assess the objectives and performance of projects and CERP as a whole.

Along with **Table 3-10**, revised monitoring maps have been developed that reflect the changes in sampling locations resulting from the reduction in available funds for MAP monitoring projects. These maps are available in **Appendix 3-2**. There are numerous maps that show monitoring locations across the CERP area in south Florida. Several years of monitoring results and associated assessment will be required to determine if the reduction in MAP monitoring projects underway in FY 2012, FY 2013 and FY 2014 will meet the intent of the plan outlined in the 2009 MAP (RECOVER 2009) to assess the ability of the CERP program to meet the ecosystem restoration goals established in the development of CERP.

In addition, an effort was initiated in FY 2013 to review the documented FY 2011 MAP prioritization process. The goal of the review was to determine if potential improvements to the process exist that will aid in prioritizing MAP monitoring components in future years and to make any appropriate changes to the process. That effort was put on hold pending new information that will be gained when the results of FY 2012 and FY 2013 monitoring projects are available for review and assessment by RECOVER.

Table 3-10. Changes in monitoring and the ability of the FY 2012 MAP monitoring components to assess the objectives and performance of projects and the overall CERP. Gray cells indicate monitoring has not been funded since FY 2012.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
West Coast Oyster Monitoring (CRE)	Oysters serve as an excellent indicator species because salinity conditions suitable for oysters produce optimal conditions for a suite of other desirable estuarine organisms; and given their sedentary nature, it is easy to determine cause-and-effect relationships between the water quality and health of these organisms. Five aspects of oyster ecology are being monitored in the CRE: (1) density of adult oysters, (2) reproduction and recruitment, (3) juvenile oyster growth and survival, (4) physiological condition as measured by condition index, and (5) distribution and frequency patterns of the oyster diseases (dermo).	1	NE	25.8%	• Reduction in the number of sampling stations (reduce stations from 5 to 4 per month with 3 repetitions) will decrease statistical ability to detect change as CERP projects are implemented. • Reduction in the number of parameters (including elimination of condition index) monitored per station may decrease the ability to understand changes in oyster reefs over time and link these changes to implementation of CERP projects. Doing a detailed analysis of the existing 10-year data set will allow us to make reductions to the program that will minimize detrimental effects.	• The remaining data will be critical for use in evaluating restoration success, discerning project effects, informing operations of the C&SF Project, and for adaptive management. Data will also help with stoplight indicator development.
East Coast Oyster Monitoring (SLE, Lake Worth Lagoon [LWL] and Loxahatchee River Estuary [LRE])	Oysters serve as an excellent indicator species because salinity conditions suitable for oysters produce optimal conditions for a suite of other desirable estuarine organisms; and given their sedentary nature, it is easy to determine cause-and-effect relationships between the water quality and health of these organisms. Five aspects of oyster ecology are being monitored in the SLE, LRE, and LWL: (1) density of oysters, (2) reproduction and recruitment, (3) juvenile oyster growth and survival, (4) physiological condition as measured by condition index, and (5) distribution and frequency patterns of the oyster diseases *Perkinsus marinus* (dermo) and *Haplosporidium nelsoni* (MSX).	1	NE	20.0%	• Reduction in the number of parameters (eliminate condition index, MSX, dissolved oxygen, and pH) monitored per station may decrease the ability to understand changes in oyster reefs over time and link these changes to implementation of CERP projects. Doing a detailed analysis of the existing data set will allow us to make reductions to the program that will minimize detrimental effects.	• The remaining data will be critical for use in evaluating restoration success, discerning project effects, informing operations of the C&SF Project, and for adaptive management. Data will also help with stoplight indicator development.
Wading Birds & Aquatic Fauna Monitoring (includes Biscayne Bay)	Assessing the demersal prey base fish community in reference to salinity and water levels allows for the inference of the relative value of the wetlands to higher trophic levels (e.g., wading birds). This project studies aquatic fauna in the mangrove zones of Florida and Biscayne bays, as well as wading bird colony location, size, and timing in Florida Bay and roseate spoonbill nesting success in Florida Bay.	1	SCS	36.0%	• Loss of 60% of prey fish monitoring sites to include all control sites in Cape Sable significantly reduces the ability to detect cause-and-effect changes in the roseate spoonbill population resulting from C-111 SCWP, WCA 3 Decompartmentalization and Sheet Flow Enhancement (Decomp), non-CERP projects such as Tamiami Trail Bridge Project and Modified Water Deliveries to ENP (Modwaters), and changes to operations (Combined Operations Plan). Spatial coverage severely reduced. Control site, SRS sampling, hydrologic sampling, and target site are lost. • Loss of the ability to make upstream to downstream comparisons in the area most likely to be affected by the C-111 SCWP. • Loss of ability to tie hydrology to prey fish distribution and abundance in response to CERP projects (e.g., C-111 SCWP).	• Spoonbill nest surveys and focal colony success (how many chicks actually fledge) will continue to be monitored. Six hydrology and prey fish stations will be retained, along with several long-term stations (data has been collected since 1990). The focus will be on the C-111 SCWP and Modwaters areas of influence.
Coastal Gradients - Monitoring of Flow, Salinity, and Nutrients in the GE and SCS	RECOVER MAP funds 10 monitoring stations that, together with the existing coastal monitoring network, creates a network of 40 sites that can be analyzed for coastal gradients of flow, salinity and nutrients. The purpose of this network is to collect real-time data in the coastal zone of ENP and report on the interactions between the Everglades mangrove transition zone and the freshwater wetlands. This network supplies critical hydrologic information where none previously existed and establishes a baseline data set of hydrologic conditions prior to any CERP watershed modifications.	1	SCS	56.9%	• Significantly reduced ability to detect salinity, flow, water level, and nutrient changes from the C-111 SCWP, CERP/non-CERP projects that affect lower SRS (e.g., Tamiami Trail Bridge) and the effects of operations to ENP. • Loss of 40% of hydrologic stations in Florida Bay and the southwest Florida Coast (18 total stations), significantly impairs regional hydrology assessment, including detection of saltwater intrusion. • Loss of hydrologic data that is critical for existing hydrologic, water quality and ecological model development.	• Monitoring stations (stage, flow and salinity) focused on the C-111 SCWP and Modwaters areas of influence along the coastal mangrove fringe in Florida Bay and the southwest Florida Coast will be retained. Will be able to detect changes in stage and flow and subsequently salinity resulting from C-111 SCWP and Modwaters with less certainty.

Table 3-10. Continued.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
Florida Bay Juvenile Sportfish Monitoring	The spotted seatrout spends its entire life history within Florida Bay, and distributions vary in response to salinity. Thus data collected on their population dynamics will help to delineate changes due to water management versus natural variability. Relationships to salinity can be used in the evaluation process to predict the impact of a CERP project(s) on sportfish populations in Florida Bay.	1	SCS	0.0%	• Shift of monitoring to eastern Florida Bay (to capture change from C-111 SCWP), may eliminate sampling in western Florida Bay (most productive basin for juvenile sportfish). • Inability to estimate the regional effects of CERP/non-CERP projects and operations to economic benefits for the region.	• Ability to detect change to sportfish populations in central and eastern Florida Bay resulting from C-111 SCWP and Modwaters effects on the estuaries.
South Florida Fish Habitat Assessment Program (FHAP)	Seagrasses and macroalgae (also known as SAV) are sensitive indicators of changing water quality conditions (e.g., salinity levels, light availability and nutrients). FHAP documents the status and trends of SAV distribution, abundance, and species composition in Florida Bay and the southwestern Everglades, as well as providing data on turtle grass reproduction, shoot characteristics, and epiphyte loads. It also provides process-oriented data such as photosynthetic efficiency. Resource managers can use these data to address ecosystem response issues on a near real-time basis and to weigh alternative restoration options. FHAP provides data on SAV in Florida Bay and Lostman's River to southern Biscayne Bay. This program documents the status and trends of seagrass distribution, abundance and reproduction, and epiphyte loads. It also provides process-oriented data such as photosynthetic efficiency. Resource managers can use these data to address ecosystem response issues on a near real-time basis and to weigh alternative restoration options.	1	SCS	35.8%	• Loss of early indicator of water quality changes and nutrient availability. Without these indicators, impending algal blooms may not be detected, which results in a decline in seagrass health. • Early indicators of water quality and nutrients availability being lost are (1) pulse-amplitude modulated fluorescence, which is an early indicator of changes in water quality, and (2) epiphyte abundance, which is an early indicator of water column nutrient increase.	• Ability to detect change to SAV communities in central and eastern Florida Bay resulting from C-111 SCWP and Modwaters effects on the estuaries.
Biscayne Bay Salinity Monitoring	The Biscayne Bay Salinity Monitoring Network's primary purpose is to provide the quality and quantity of data sufficient for determining the effects of CERP on salinity regimes in Biscayne Bay and for other investigators to use in their analysis of effects on organisms. This project provides descriptive, spatial, and temporal analysis of salinity patterns in Biscayne Bay, and data from this project has been used to develop freshwater inflow needs for Biscayne National Park.	1	SCS	42.6%	• Loss of 21 salinity monitoring sites (two-thirds of the sites) throughout Biscayne Bay, Card Sound, and Manatee Bay resulting in less certainty in determining salinity (distribution and patterns) both in the immediate vicinity of the BBCW Project and at the larger, regional scale. • Loss of sites impairs ability to assess RECOVER's Biscayne Bay salinity performance measure. • Loss of salinity monitoring in central and northern Biscayne Bay inhibits the ability to detect regional changes. • Biscayne Bay monitoring has been combined to increase sampling efficiency. Principal investigators provided a combined scope (for Biscayne Bay salinity monitoring, Biscayne Bay epifauna, Biscayne Bay SAV, and Biscayne Bay mangrove fish) that decreases costs 44.5% by sharing field technicians.	• Monitoring stations (hi-resolution salinity) focused on the BBCW Project and C-111 SCWP areas of influence will be retained. Will be able to detect changes in salinity resulting from C-111 SCWP and BBCW Project.
Hydrology & Salinity Monitoring in Ten Thousand Islands (TTI)	There is little information regarding the present day quantity, timing and quality of freshwater flow to the TTI. This study complements an ongoing USGS effort designed to measure the discharge and salinity of rivers flowing from Tamiami Trail towards the TTI in order to help with identifying desirable surface water flow rates and assessing restoration success. Specifically, this project operates and maintains 4 flow monitoring stations and 1 boundary conditions monitoring station in the TTI and describes the hydrodynamic characteristics and the temporal and spatial salinity variability of creeks and estuaries within the TTI area in relation to prominent physical boundaries, which are generally composed of mud flats, ridges and oyster beds.	1	SCS	18.4%	• Loss of flow data at East River control site. Stage and flow are not closely correlated here. Cannot secondarily calculate flow and will not be able to compare flow from remaining sites against a control. • No flow at a newly established station in Palm River; this station captures additional flows to the western estuaries from PSRP, which were not originally anticipated.	• 5 monitoring stations to include the 3 basins referenced in the *Picayune Strand Restoration Project Final Integrated Project Implementation Report and Environmental Impact Statement* (USACE and SFWMD 2004) (Pumpkin Bay, Faka Union Bay, and Fakahatchee Bay). 3 of the 5 stations in the area anticipated to reflect the greatest changes in flow from the project will retain flow, stage, and salinity measurements. One new station in Palm River will reestablish a station where USGS collected 8 months (October 2003–April 2004) of stage and salinity measurements.

Table 3-10. Continued.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
Everglades Depth Estimation Network (EDEN)	EDEN is an integrated network of roughly 260 gauging stations (25 of which are maintained by RECOVER) that assist with real-time water level monitoring, ground-elevation modeling, and water-surface modeling that provides scientists and managers with current (1999–present), on-line water depth information for the GE region. This enables the assessment of biotic responses to hydrologic change.	1	GE	30.0%	• About 12–15 of the RECOVER MAP funded water level gages (RECOVER has funded 25 gages which are about 10% of the network) were discontinued in March 2012. • A less dense network of gages will result in a corresponding loss of accuracy in the daily water level surfaces for the Everglades. • Because gages are used to estimate missing data at nearby gages, a less dense network reduces the ability to fill data gaps (common in remote locations like the Everglades) and produce consistent model results.	• The EDEN daily water level surfaces and other EDEN hydrologic data sets for the Everglades will continue to be produced on a real-time basis for EDEN users and available on a publically accessible website. • Users include scientists, water management operators, and water resource managers who use the EDEN data for correlation of biological and ecological data with hydrology, development of performance measures, decisions for water management operations, and input as hydrology for biological and ecological models.
Wet Season Trophic Sampling	An overarching hypothesis for Everglades restoration is that hydrology controls the production and concentration of aquatic prey organisms, which in turn determine the magnitude and success of wading bird nesting during any given year. A set of hypothesized mechanistic relationships between hydrology, productivity of the prey base, and the behavioral patterns of several representative wading bird species are being tested in order to determine how large populations of wading birds that were one of the defining characteristics of the predrainage Everglades ecosystem can most effectively be restored. The wet season is the period of production of aquatic prey populations, which is directly related to hydroperiod.	1	GE	43.0%	• Reduced ability to use prey fishes to make systemwide inferences resulting from CERP projects and C&SF Project operational changes (e.g., Lake Okeechobee Regulation Schedule Study) due to loss of dry season sampling. • Complete loss of periphyton and fish sampling/information in Lake Okeechobee and Pal Mar/Corbett that describes the water quality and prey-based fishes affected by Lake Okeechobee operations and within the Indian River Lagoon watershed. • No species identification of periphyton (an indicator of water quality and depth), which provides a linkage between hydrologic and water quality impacts of operations on wading bird production in the Everglades. Periphyton is both a food source and habitat for wading bird prey (e.g., fish). • Loss of information that contributes to interim goals, stoplight reporting and the *Five-Year Report to Congress*.	• In order to nest successfully, wading birds rely on having enough prey (high densities) and thus the status and the trends of prey fish densities need to be known. • This project will continue to deliver population data on key aquatic fauna and periphyton as an index pattern of production in time and space. The goal is to help understand this with respect to ecosystem restoration.
Dry Season Aquatic Fauna Sampling	An overarching hypothesis for Everglades restoration is that hydrology controls the production and concentration of aquatic prey organisms, which in turn determine the magnitude and success of wading bird nesting during any given year. A set of hypothesized mechanistic relationships between hydrology, productivity of the prey base, and the behavioral patterns of several representative wading bird species are being tested in order to determine how large populations of wading birds that were one of the defining characteristics of the pre-drainage Everglades ecosystem can most effectively be restored. The dry season is a period of concentration of aquatic prey populations, which is controlled by rates of water level recession.	1	GE	40.0%	• Loss of information as to how operations and climatic variation affect wading bird prey and wading bird nesting success due to loss of wet season sampling. • Loss of field technician, which results in loss of vegetation transects. • No MAP contribution to annual *South Florida Wading Bird Report*. • Loss of information that contributes to interim goals, stoplight reporting and the *Five-Year Report to Congress*.	• Ability to analyze dry season wading bird prey density and nesting success.
Wading Birds Monitoring	This project continues to build on an existing database of wading bird reproductive success and productivity information extending back to the 1960s for wood storks and to the 1930s for roseate spoonbills. The area of study covers WCAs 1, 2, 3, ENP, and BCNP. This data set has already served as an early warning of the collapse of ecosystem function, the widespread contamination of the wetland biota with mercury, and the critical functions provided by droughts.	1	GE	10.0%	• Loss of information as to how operations and climatic variation affect wading birds; situation exacerbated by potential loss of Modwaters funding for wading bird monitoring (i.e., would no longer be a systemwide wading bird survey). • Loss of vegetation transects. • Loss of information that contributes to interim goals, stoplight reporting and the *Five-Year Report to Congress*. • Loss of permit-required mercury monitoring by SFWMD (MAP pays for transportation). If it is necessary to continue mercury monitoring, SFWMD would need to pick up transportation costs.	• Wading birds are key species in determining the health of the Everglades as a whole. Provided Modwaters funding is continued, this study helps to retain systemwide information on the distribution and nesting success of wading birds.

Table 3-10. Continued.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
Monitoring Tree Island Condition in Southern Everglades	Tree islands are important centers of biodiversity embedded within the marsh mosaic. Changes in water management associated with hydrologic restoration will result in changes in the internal water economy of tree islands, as well as in the risk of fire, which in turn will lead to changes in plant function and species composition. Data collected by this project will enable the linkage of marsh hydrology predicted by models or attained in the real world with ecological responses of tree islands to determine the success of restoration activities. The areas of focus are ENP and SRS.	1	GE	40.0%	• Loss of ability to assess tree island productivity (tree island canopy structure), which is an indication of Everglades "health" due to loss of litter dynamic sampling and water level sampling. • Loss of one field technician. • Loss of information that contributes to interim goals.	• Ability to assess water stress on the species composition of tree islands and contribute information to real-time operations.
Landscape Pattern - Ridge, Slough and Tree Island Monitoring	This project focuses on three components: (1) mapping vegetation features from aerial photographs, (2) aerial surveys for classification of tree island type, and (3) ground surveys of water depth and plant community structure. Data on these components will be used to quantify aspects of the hydrologic regime, determine relationships between vegetation and water depth, quantify the distribution and spatial structure of peat elevations, and groundtruth broader-scale maps based on remote sensing and aerial surveys. This will maximize the likelihood of change detection during CERP implementation.	1	GE	56.0%	• Complete loss of sampling in WCA 1 and the marl prairies in ENP, which supports understanding of Modwaters and C-111 SCWP restoration performance. Ridges and sloughs will only be sampled in WCA 2, WCA 3 and ENP. • Loss of ground sampling at three tree islands in SRS. • Loss of mapping the “health” of ridge and slough patterning (one of the defining characteristics of the Everglades). • Loss of information that contributes to interim goals.	• Continued ability to characterize ridge and slough landscape, including elevation and vegetation patterns, albeit within a reduced spatial extent
Tree Island Monitoring	This project expands upon an existing network of monitoring wells and has installed new wells on tree islands in order to quantify groundwater discharge and tree island-marsh hydrologic connectivity across a disturbance gradient. The areas of focus are WCA 3A and WCA 3B. Monitoring well, isotopic, and transpiration data can then be used as a diagnostic tool to assess tree island functioning and health to inform CERP implementation.	1	GE	10.0%	• Loss of information used to better develop tree island performance measures and hydrologic connectivity between tree islands and the marsh. This information is used for system operations due to decreased sampling frequency. • Loss of information that contributes to interim goals.	• Only work being performed on tree islands compares belowground processes (productivity) on islands that range from “pristine” to almost wholly degraded. This information will allow an understanding of why some of the tree islands are dying.
Monitoring of Marl Prairie Slough Gradients	In the southern Everglades, marl prairie habitats are present on either side of SRS. Vegetation structure and composition gradually change along an elevation and depth gradient, thus transects have been established to monitor changes in vegetation along that gradient. Changes in vegetation can then be related to changes in hydrology resulting from restoration activities.	1	GE	20.0%	• Loss of plant community monitoring in the western marl prairies (at 5-meters fine scale), which provides information about systemwide hydrology and is an indicator of hydrologic change and the effects of restoration due to loss of the western marl prairie 5-m transect. • Loss of understanding about gradients from marl prairie to slough that cause changes in vegetation, in response to changes in hydrology. • Loss of information that contributes to interim goals.	• Can inform system operations for timing and distribution of water releases to northeastern SRS in support of ridge and slough habitat.
Lake Okeechobee Wading Bird Monitoring	This study focuses on populations of wading birds in Lake Okeechobee, and collaborates with other studies in Florida Bay, the Everglades, and the Big Cypress region. It builds on an existing database of nesting effort, reproductive success, and productivity information extending back to the 1960s, or further in case of the wood stork. This data is critical in that it addresses the hypothesis that restored hydrology will generate more dense populations of fish and macroinvertebrates, enhance foraging opportunities for wading birds, and increase breeding and nesting in the coastal areas.	1	LO	14.0%	• Loss ability to correctly identify the start, peak, and/or end of wading bird nesting season due to reduced sampling from 5 months to 4 months. • Reduced sampling may lead to underestimates of nest failure rates (i.e., if they initiated and then failed between sampling visits). • Next year's data may not be directly comparable to previous years if sample times/size has changed.	• Retains some ability to assess nesting success of wading birds in Lake Okeechobee.

Table 3-10. Continued.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
Monitoring of Oysters in the Ten Thousand Islands	Freshwater diversion from natural flow-ways in the TTI by basin development and road construction has adversely impacted downstream estuaries by changing their salinity regimes and water quality. The PSRP will alter the existing salinity regime; however, the timing and quality of freshwater flow to the TTI and its effects on the nearshore benthic community is not yet well defined. This project will provide a comprehensive baseline of oyster population and health in the estuaries downstream of the project.	1	SCS	Not available	• This oysters monitoring was added to fill a gap in monitoring for the PSRP on the downstream estuaries of the TTI. No consistent and continuous ecological monitoring under the MAP exists to detect the effects of hydrologic changes to the TTI estuaries.	• Addition of only ecological parameter being monitored by CERP to measure the effects of the PSRP on the downstream estuarine health.
Submerged Aquatic Vegetation (Indian River Lagoon, SLE, LWL, and CRE)	Historically, natural freshwater discharges facilitated the presence of healthy floral and faunal communities, including SAV. As development increased, however, management practices resulted in coastal areas with frequent high and low salinity extremes and degraded ecology. This monitoring aims to collect baseline data, quantify relationships between freshwater discharges and subsequent salinity and water quality patterns, and quantify how salinity and water quality patterns in turn impact SAV distribution, community structure, and variability. SAV is a key indicator of restoration success.	2	NE	83.0%	• A reduction in monitoring (loss of 15 stations) will focus only on specific areas of the estuary (upstream in the CRE and closest to the SLE in the Indian River Lagoon) and therefore will not provide the larger picture of other “non-CERP” induced changes.	• By using in-house SFWMD resources, a program, although limited in scope, will be kept in place in key areas of the NE region (SLE and CRE) where the first effects of CERP projects being implemented are expected. These would include C-44, C-43 and any changes to Lake Okeechobee operations.
Monitoring of Ridge and Slough Maintenance and Degradation	The core mechanism for the maintenance of the ridge and slough landscape is peat accretion: high production in ridges creates a stable vegetative configuration at higher soil elevations, and low production in sloughs creates another at lower soil elevations. Loss of the patterning would be catastrophic. This project aims to better understand how the ridge and slough landscape is maintained and identify the critical thresholds above and/or below which those mechanisms are altered and landscape change ensues. The area of focus is the hydrologic gradient between WCA 3A north through the best conserved areas of WCA 3A south, the stabilized areas of WCA 3B, and the impounded areas at the southern end of WCA 3A.	3	GE	61.4%	• Partial loss of ability to detect elevation and nutrient/phosphorus changes in ridges and sloughs. Diminishes ability to detect phosphorus patterns in soils (e.g., ridges generally have higher soil phosphorus concentrations than sloughs) due to loss of ground sampling at 3 tree islands in south SRS. • Partial loss of understanding of mechanism necessary to restore ridge and slough. • Loss of analytic modeling and expertise in chemical analysis. • Loss of information supporting interim goals.	• Maintain reduced ability to detect change in ridge and slough pattern, which is a defining characteristic of the Everglades.
Biscayne Bay Mangrove Fish Monitoring	It is hypothesized that within mainland mangrove habitats, CERP-related impacts will likely be the strongest and most easily discerned from other effects. This study is the longest running fish monitoring study ever conducted in Biscayne Bay and adjacent waters. Data is collected on seasonal and spatial variation in fish composition and diversity, as well as frequency of occurrence, density and size structure. Emphasis is placed on evaluating relationships between the shoreline fish community and variation in salinity/freshwater flow to evaluate restoration success.	4	SCS	24.1%	• Loss of sampling sites in northern Biscayne Bay (specifically loss of dry season sampling). Also loss of only long-term fish monitoring in northern Biscayne Bay and Card Sound. This decreases the ability to detect operational changes on the central and northern portions of Biscayne Bay. •Inability to assess seasonal variability of fish abundance and distribution and compromises performance measure development (specifically, use of goldspotted killifish [*Floridichthys carpio*] as a performance measure). • Discontinuation of laboratory-based habitat suitability index validation, which will result in the ecological tool not being available for use for evaluations and assessments for CERP projects. • Limited ability to document changes to nearshore SAV habitat and associated fauna. • Biscayne Bay monitoring has been combined to increase sampling efficiency. Principal investigators provided a combined scope (for Biscayne Bay salinity monitoring, Biscayne Bay epifauna, Biscayne Bay SAV, and Biscayne Bay mangrove fish) that decreases costs 44.5% by sharing field technicians.	• Ability to detect change to mangrove fish populations in Manatee Bay, Card Sound, and Biscayne Bay resulting from C-111 SCWP and BBCW Project effects on the estuaries. This project comprises the only remaining ecological indicator for Biscayne Bay.

Table 3-10. Continued.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
Benthic Infaunal Monitoring in the SLE and Indian River Lagoon	Benthic infaunal communities (worms and mollusks that live in the soft sediment on the estuary bottom) are primarily stationary, and are therefore continuously exposed to changes in the environment. This is one of the main reasons why benthic infaunal monitoring is commonly regarded as one of the best tools for evaluating the health and long-term changes within the marine environment. The main objectives of this project are to evaluate the present health status of the SLE and Indian River Lagoon, determine the cause of long-term changes, pinpoint and evaluate anthropogenic disturbances, and calculate a health index for each monitored site in order to monitor change over time.	5	NE	41.3%	• Reduction in the number of stations monitored from 15 to 9 will decrease our ability to detect changes as CERP projects are implemented and will give us a less complete coverage of all of the areas of the estuary that may be impacted. • The reduction will decrease our ability to distinguish between natural changes and those brought about by CERP implementation.	• Stations in key areas will allow us to continue on a limited basis to assess the current and future health of the benthic community in the SLE.
Monitoring of Marsh Mangrove Fishes	This project was developed to improve understanding of the role of the marsh-mangrove ecotone in the southern Everglades as habitat for freshwater fishes. Historically, nesting (and likely foraging) by wading birds occurred in the highest numbers in this area, yet very little is known about what drives prey abundance, distributions, and concentrations in this part of the ecosystem. The goal is to understand how the fish community will respond to restoration conditions, which are expected to increase the pooling of fresh water at the ecotone, resulting in a wider and seasonally-extended oligohaline zone.	6	GE	43.1%	• Loss of ability to detect change between saltwater (estuarine) and freshwater (marsh) habitat for fishes with loss in sampling (loss of drop net, North River site, and minnow traps). Changes at this interface are an early indicator of CERP effects as projects are constructed. • Loss of information that contributes to interim goals and stoplight reporting.	• Compares seasonal marine fish population in creeks and their diet, testing the hypothesis that larger marine predators are eating bird food fish due to lack of freshwater discharge. • Only testing done at the saltwater-freshwater interface ecotone biota.
Vegetation Mapping in Everglades National Park	In order to assess restoration of the Everglades and document changes in species composition and distribution, production of a spatially and thematically accurate vegetation map for ENP and BCNP is necessary. This mapping project involves field collection, remote sensing procedures, and vegetation map interpretation.	7	GE	2.4%	• Delay of photo-interpretation of aerial imagery flown in 2009 (delay = undesirable lapse in time between aerial imagery and interpretation).	• Pre-CERP baseline mapping for ENP is needed in order to discern change from Modwaters and C-111 SCWP. • Provides an entire photo-interpreted pre-CERP baseline map in ENP and will be finished in 4 years, using 6-year old photos.
Biscayne Bay Nearshore Submerged Aquatic Vegetation Monitoring	This study focuses on nearshore benthic habitats (< 500 meters from shore) of southern Biscayne Bay to evaluate spatial patterns of abundance of SAV in relationship to (1) distance to shore and (2) water management canals that discharge fresh water from upland sources. The initial findings of this effort have already indicated distinct seasonal and species-specific patterns of abundance and spatial distribution of seagrasses and macroalgae that are directly influenced by the inflow of fresh water into nearshore Biscayne Bay. These areas are critical nursery habitats for pink shrimp and economically-valuable fishes such as gray snapper (*Lutjanus griseus*), hogfish (*Lachnolaimus maximus*), spotted seatrout and pinfish (*Lagodon rhomboides*).	8	SCS	52.1%	• Loss of dry season sampling (loss of seasonal variability component). • Loss of spatial coverage, including control areas and only area of overlap with other regional SAV project (Manatee Bay). • Sampling only between Matheson Hammock and north of Barnes Sound. • Macroalgae completely lost, including development of macroalgal indicators. • Cannot develop a performance measure for SAV in Biscayne Bay, Card Sound and Barnes Sound. Performance measures are a primary tool that the RECOVER program uses to assess CERP effects and evaluate CERP alternatives. • Biscayne Bay monitoring has been combined to increase sampling efficiency. Principal investigators provided a combined scope (for Biscayne Bay salinity monitoring, Biscayne Bay epifauna, Biscayne Bay SAV, and Biscayne Bay mangrove fish) that decreases costs 44.5% by sharing field technicians.	• Habitat information connects the changes in hydrology via the BBCW Project to fauna. • Several of these habitat and faunal indicators have the ability to provide economic valuation to restoration benefits. • Ability to directly link SAV to water quality

Table 3-10. Continued.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
Monitoring of Biscayne Bay Epifaunal Communities	The epifaunal community is the principal source of prey for recreationally and commercially important fish that inhabit the SCS region during all or a part of their lifetime. This project emphasizes community-based performance, directly addressing the restoration objective of reestablishing an estuarine fish and invertebrate community in nearshore southern Biscayne Bay. It is a critical element in an integrated sampling regime that is examining the ecology of the seagrass systems and mangrove systems in relation to each other and spatial and temporal salinity patterns.	9	SCS	55.6%	• Loss of dry season sampling (lose seasonal variability). • Up to 50% reduction in spatial sampling coverage (results in loss of statistical power). • Loss of caridean shrimp identification and abundance studies. • Limited development of predictive models and habitat suitability indices to be used in performance measures. Performance measures are a primary tool that RECOVER uses to assess CERP effects and evaluate CERP alternatives. • Biscayne Bay monitoring has been combined to increase sampling efficiency. Principal investigators provided a combined scope (for Biscayne Bay salinity monitoring, Biscayne Bay epifauna, Biscayne Bay SAV, and Biscayne Bay mangrove fish) that decreases costs 44.5% by sharing field technicians. • Loss of analysis to categorize species based on their relative abundance and the salinity of their distributions.	• Estimate of effects of the BBCW Project and regional operations on epifaunal communities (pink shrimp and other invertebrates) and their ties to economic benefits for Biscayne Bay, Card Sound, and Barnes Sound.
Alligator and Crocodile Monitoring	Responses of crocodilians are directly related to suitability of environmental conditions, including hydropattern. CERP hypotheses related to alligators and crocodiles state that restoration success or failure can be evaluated by comparing recent and future trends and status of crocodilian populations with historical population data and model predictions. Importantly, these data can be used in an analysis designed to distinguish between effects of CERP and those of non-CERP events such as hurricanes or droughts.	10	GE	100.0%	• Loss of information for keystone (American alligator) and endangered species (American crocodile). • Loss of crocodile information in support of C-111 SCWP and BBCW Project. • Loss of information that contributes to interim goals and stoplight reporting.	• No monitoring remains in WCA 2 and WCA 3. FY 2013 sampling for crocodiles and alligators in ENP, WCA 1, and BCNP was completed with partial funding from NPS, USGS and USACE. Continued funding for sampling in all areas is uncertain.
Monitoring Aquatic Fauna in Big Cypress	Previous quantitative data on aquatic fauna communities is nearly nonexistent for the BCNP; however, these forested wetlands formerly functioned as critical feeding and nesting sites for wading birds. This project aims to collect quantitative baseline data on the constituent aquatic communities and their ecology in order to detect changes in natural and artificial habitats resulting from restoration activities.	11	GE	100.0%	• Loss of fish and invertebrate information in eastern BCNP. This was the only monitoring ongoing in BCNP.	No monitoring remains.
Sediment Elevation Tables	In 1998, the USGS began establishing surface elevation tables at the hydrology stations on the Lostmans and Shark rivers. Monitoring is focused on examining impacts of altered freshwater inflow regimes on coastal wetland hydrology and salinity regimes, effects on primary productivity, and on the dynamics of sediment elevation.	12	GE	100.0%	• Loss of measurement of sediment accumulation in the mangrove zone. • Loss of long-term study to measure sea level rise. Should be funded through the Climate Change Program.	No monitoring remains.
Fish and Invertebrate Assessment Network (FIAN) - USGS Component	Studies quantifying seagrass-associated fish and invertebrate community composition and abundance using a 1-square m throw-trap have been ongoing in south Florida since 1983. FIAN aims to quantify seagrass-associated fish and invertebrate (i.e., shrimp and crabs) populations and communities and their associations with prevailing environmental conditions (e.g., salinity and temperature) and seagrass/algae habitat (SAV). FIAN samples pink shrimp and concurrent habitat and environmental conditions in all three major southern coastal ecosystems—Florida Bay, Biscayne Bay, and the southwest coast mangrove systems, including Whitewater Bay.	13	SCS	100.0%	• Loss of ability to estimate effects of CERP (C-111 SCWP and BBCW Project), non-CERP (Modwaters), and regional operations on pink shrimp populations and invertebrates, the connectivity between changes in hydrology to associated estuarine habitat and sportfish, and the tie to economic benefits. • No monitoring of pink shrimp, which is an indicator species and used as a stoplight indicator and interim goal. • Loss of development of predictive models and habitat suitability indices for potential indicator species, which could be used in performance measures. Performance measures are a primary tool that RECOVER uses to assess CERP effects and evaluate CERP alternatives.	No monitoring remains.

Table 3-10. Continued.

Contract Name	Description of Monitoring	Priority	MAP Region	% Reduction	What is Lost - Impact of Reduced Funding on Monitoring	Value/Use of Remaining Monitoring Data
Fish and Invertebrate Network (FIAN) - NOAA Component	Studies quantifying seagrass-associated fish and invertebrate community composition and abundance using a 1-square meter throw-trap have been ongoing in south Florida since 1983. FIAN aims to quantify seagrass-associated fish and invertebrate (i.e., shrimp and crabs) populations and communities and their associations with prevailing environmental conditions (e.g., salinity and temperature) and seagrass/algae habitat (SAV). FIAN samples pink shrimp and concurrent habitat and environmental conditions in all three major southern coastal ecosystems—Florida Bay, Biscayne Bay, and the southwest coast mangrove systems, including Whitewater Bay.	14	SCS	100.0%	• Loss of ability to estimate effects of CERP (C-111 SCWP and BBCW Project), non-CERP (Modwaters), and regional operations on pink shrimp and invertebrates, the connectivity between changes in hydrology to associated estuarine habitat and sportfish, and the tie to economic benefits (pink shrimp). • No monitoring of pink shrimp, which is an indicator species and used as a stoplight indicator and interim goal. • Loss of development of predictive models and habitat suitability indices for potential indicator species, which could be used in performance measures. Performance measures are a primary tool that the RECOVER program uses to assess CERP effects and evaluate CERP alternatives.	No monitoring remains.
Water Quality, Salinity & Circulation Monitoring	The goals of this project are to monitor and understand water quality variability and circulation and transport variability in the south Florida coastal region surrounding and including Florida Bay and the Florida Keys National Marine Sanctuary on tidal to interannual time scales; to monitor and understand the role of the Loop Current and Florida Current in influencing water quality along the Southwest Florida Shelf and Florida Keys including the Dry Tortugas; and to monitor and understand the advection and dispersion of nutrient loads discharging on the Southwest Florida Shelf and directly into Florida Bay.	15	SCS	0%	• This project was not funded in FY 2011 nor FY 2012. • Lose ability to see salinity and other water quality conditions across the entire estuary (Florida Bay) over time. • Loss of supplemental data to augment the continuous salinity monitoring information generated by the NPS and SFWMD.	No monitoring remains.

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APPENDIX 3-1: INVASIVE AND NUISANCE SPECIES

Systemwide Updates

Invasive Species Overview

Executive Order 13112, entitled Invasive Species, signed February 3, 1999, states an "invasive species means an alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health." Alien species means, with respect to a particular ecosystem, any species, including its seeds, eggs, spores, or other biological material capable of propagating that species, that is not native to that ecosystem. Invasive species are broadly defined and can be a plant, animal, fungus, plant disease, livestock disease or other organism. The terms 'alien' and 'exotic' also refer to nonnative or nonindigenous species.

Invasive species are directly implicated in losses of native species, biodiversity, ecosystem functions, ecosystem services, and livelihoods worldwide (U.S. Congress, Office of Technology 1993). Increasingly, technology and globalization are reducing the barriers that once allowed unique species and ecosystems to evolve without continuous disturbances from biological invasions. As a result, rates of biotic exchange are increasing on all continents and the trend is expected to continue despite heightened international awareness of the impacts of biological invasions (Millennium Ecosystem Assessment, 2005).

Nationally, more than 50,000 species of introduced plants, animals, and microbes cause more than $120 billion in economic damages and control costs each year (Pimentel et al. 2005). The large majority of plants and animals that enter the continental United States enter through the port of Miami (Center et al. 1998). Approximately 10 to 15% of introduced species will become established and 10% of the established species may become invasive (U.S. Congress, Office of Technology 1993). Invasive species are the second largest threat to biodiversity following only habitat destruction (Wilcove et al. 1998); invasive species are second in destructive nature only to human development. In the United States, invasive species directly contributed to the decline of 49% of threatened and endangered species (Wilcove et al. 1998).

Florida is particularly vulnerable to the introduction, invasion and naturalization of nonnative species. This is due to several factors including a subtropical climate, dense human population centers, major ports of entry and the pet, aquarium and ornamental plant industries. Major disturbance to the landscape has also increased Florida's vulnerability for establishment of invasive species. Alteration of the landscape for urban development, flood control and agricultural uses has exacerbated nonnative plant and animal invasions. Stein et al. (2000) estimated that over 32,000 exotic species (25,000 plants and 7,000 animals) have been introduced into Florida, compared to the approximately 4,000–5,000 native species of plants and animals present in the state. Not all of these species are invasive, but by sheer number alone, there is great potential for invasion of natural areas by these nonnatives.

Invasive Species and Restoration

Successful restoration of south Florida ecosystems hinges on the ability to reverse the environmental degradation primarily caused by human activities over the last 100-plus years. While the Comprehensive Everglades Restortion Plan (CERP) and Restoration Coordination and Verification Program (RECOVER) efforts involve numerous factors, the potential impact of invasive species has emerged as a high priority for CERP. Invasion of South Florida's natural habitats by nonnative plant and animal species has significantly altered the ecosystem, particularly by displacing native species. Without successful control of invasive species, the benefits of restoration efforts will be reduced.

Without effective control and management, invasive nonnative species will continue to decrease biodiversity; displace native plant and animal communities; reduce wildlife habitat and forage opportunities; alter the rates of soil erosion and accretion, fire regimes and hydrology; upset predator/prey relationships; degrade environmental quality; and spread diseases to native plants, animals and other organisms. In addition to environmental impacts, invasive species also impact human health, reduce agricultural production and property values, degrade aesthetic quality, decrease recreational opportunities, and threaten the integrity of human infrastructure such as waterways/navigation channels, locks, levees, dams and water control structures.

Potential Interactions with CERP Performance Measures

As both drivers and stressors of ecosystems, invasive species can alter ecosystem patterns and processes on both small and large scales and may result in unexpected successional trajectories as Everglades restoration proceeds (Ogden et al. 2005, Doren et al. 2009). Therefore, the presence of invasive, nonnative species may greatly reduce certainty in the ability to predict restoration outcomes, particularly with regard to CERP performance measures. For example, the aggressive spread of two highly invasive grasses, torpedograss (*Panicum repens*) and tropical American watergrass (*Luziola subintegra*), on the Lake Okeechobee (LO) marsh will likely not be reversed through improved hydroperiods and reduced nutrient loading alone. Left unchecked, these species would certainly impact LO region performance measures (e.g., recovery of the native vegetation mosaic or increased native fish recruitment), but the extent of damage is not predictable. Other invasive plant species such as Old World climbing fern (*Lygodium microphyllum*) and Brazilian pepper (*Schinus terebinthifolius*) are expected to significantly affect Greater Everglades (GE) region performance measures (e.g., ridge and slough community sustainability) if their continued spread across the landscape is not reversed.

While less understood and more difficult to quantify, the impacts associated with the establishment and spread of invasive, nonnative animals could also alter restoration outcomes and directly or indirectly impact CERP performance measures. The biotic interactions occurring among nonnative and native animal species and what these interactions mean for native populations and ecological function is hard to predict since little information regarding these interactions exists. For example, although the Cuban treefrog (*Osteopilus septentrionalis*) became established in the Florida Keys nearly 90 years ago and has now spread throughout most of the state, only rudimentary information on ecosystem impact is available. Ongoing research does indicate, however, the Cuban treefrog aggressively competes with and preys upon native frog species (Johnson 2007, Waddle et al. 2010), but the overall effect on population dynamics and trophic relationships at the ecosystem scale is unknown. Another example is Burmese python (*Python molurus*) predation on the American alligator (*Alligator mississippiensis*), waterbirds, and many other native fauna that are intended to benefit from ecosystem restoration, as well as compete for similar prey in the GE region (Snow et al. 2007). The impact of these interactions has not been quantified, but the potential for Burmese pythons to alter American alligator abundance and distribution, a CERP performance measure, is a significant concern and warrants further study (Mazzotti et al. 2009). The little research that has focused on the interactions between and among invaders and native species suggests that biological invasions often have direct and indirect effects on ecosystem processes. Documented examples of such interactions include disruption of plant-animal reproductive mutualisms (Traveset and Richardson 2006), reduced fitness through habitat alteration (Pearson 2009), and changes to abiotic factors such as fire (Roberts 1997).

Scientists and land managers face enormous challenges when attempting to address biological invasions across a landscape as vast as the Florida Everglades. Effective management requires an integration of effective control tools, monitoring, research, preventative regulations, and close interagency coordination (Masters and Sheley 2001). The realistic expectation of “successful management” is limiting the impacts of biological invasions, not complete eradication. Nonnative species are rarely eradicated from the natural areas they invade (MacDonald et. al 1989, Bomford and O’Brien 1995). Ecosystems resulting from restoration will contain new species assemblages and biotic interactions relative to their pre-disturbance state, and these new biotic components and interactions are certain to include nonnative species (Norton 2009). In the context of Everglades restoration, agencies must accept that invasions will continue to exert pressure on native species and ecosystem function in the restored condition.

Invasive Species Response to Restoration

Nonnative species are expected to respond differently to restoration and, in some cases, may continue to act as drivers of ecosystem change as restoration proceeds. Improved hydroperiods and water quality may reduce the competitiveness of aggressive plant species, such as Brazilian pepper and melaleuca (*Melaleuca quinquenervia*), but such plant species are likely to persist in the restored condition. Other species, particularly many invasive animals, will continue to find suitable niches after restoration. Project features will likely aid in the spread of invasive species into areas previously not inhabited. Thus, without a long-term commitment to invasive species management, the goals of Everglades restoration are unlikely to be achieved. Additional examples of invasive plant species’ anticipated responses to restoration are provided in **Table A3-1-1**.

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Table A3-1-1. Anticipated invasive species impact and restoration responses.

Species	Impact / Response	Management Impacts
Australian Pine (*Casuarina* spp.)	Minimal Decrease	Not tolerant of prolonged flooding; will continue to invade canal banks, spoil piles, and other elevated soils
Brazilian Pepper (*Schinus terebinthifolius*)	Minimal Decrease	Mature plants may survive wetter conditions, but growth rates and seedling survival would likely decrease
Bishopwood (*Bischofia javanica*)	Decrease	Cannot tolerate flooding; likely to persist in disturbed areas with shorter hydroperiods.
Burma Reed (Neyraudia reynaudiana)	Decrease	Tolerates a wide range of conditions; quickly invades open, dry, and sunny sites.
Lead Tree (*Leucaena leucocephala*)	Decrease	Cannot tolerate flooding; likely to invade disturbed, upland areas near construction footprints.
Melaleuca (*Melaleuca quinquenervia*)	None Expected	Will continue to be a serious threat, regardless of changes in hydrology.
Napier Grass (*Pennisetum purpureum*)	Increase	Will continue to invade canal banks, levees, spoil areas and other disturbed sites.
Primrose Willow (*Ludwigia peruviana*)	Increase	Tolerant of sustained flooding and very competitive in nutrient-rich soils.

Overview of Selected Invasive Taxa

This section is intended to provide a brief overview of selected invasive species widely established in south Florida. The species discussed here are only a small representation of priority invasive species and were selected to illustrate the diversity of invasive taxa and varied challenges they bring to Everglades restoration. Rodgers et al. 2013 provides a more comprehensive overview of the numerous priority invasive species impacting south Florida's natural areas.

Melaleuca

Melaleuca is an evergreen tree reaching up to 33 meters, with a narrow crown and distinctive whitish peeling bark (Langeland et al. 2008). A native of Australia, melaleuca was introduced into South Florida and the Everglades in the early 1900s (Meskimen 1962). Melaleuca successfully invades Everglades marsh and pineland habitats, where it can form dense single species stands. Established melaleuca can significantly change plant species composition and structure (Rayamajhi et al. 2006, 2009, Hofstetter 1991) and reduce the carrying capacity of some wildlife species (Mazzotti et al. 1981). Prior to initiation of an organized state and federal melaleuca control project in 1990, melaleuca was widely distributed throughout the Everglades ecosystem (Laroche and Ferriter 1992). Agency efforts to control melaleuca through a combination of chemical, physical and biological control have been successful in containing melaleuca and reducing its spread (Rodgers et al. 2011). Despite these efforts, melaleuca remains widely distributed on private and some public lands throughout south Florida. Areas that have been treated that are adjacent to these private and public lands will require continued control and surveillance efforts.

Old World Climbing Fern

Old World climbing fern is a twining, evergreen fern of Asian origin. The fronds exhibit indeterminate growth which can reach up to 30 meters (Langeland et al. 2008) in height and can quickly smother surrounding vegetation. Old world climbing fern was introduced into Florida in the 1950s as an ornamental (Beckner 1968) and has since spread throughout much of south and central Florida (Ferriter and Pernas 2006). It is one of the greatest threats to south Florida's mesic upland and wetland ecosystems. This fire adapted fern grows high into tree canopies smothering entire bayhead forest tree islands (Pemberton and Ferriter 1998). Old World Climbing fern forms rachis mats up to 100 centimeters (cm) thick along the ground, drastically reducing native plant species composition and abundance (Brandt and Black 2001). In fire-adapted Everglades plant communities, high fire intensities resulting from these thick mats reportedly cause the mortality of otherwise fire-adapted native species (Hutchinson et al. 2006).

Burmese Python

The Burmese python is widely established in the southern Everglades (Snow et al. 2007). This large constrictor is a top predator known to prey upon more than 20 native Florida animal species and is implicated in substantial declines of mammal populations in Everglades National Park (ENP) (Dorcas et al. 2011). Control of this species is a top priority among agencies and policy makers. Record cold temperatures during January 2010 caused widespread mortality of Burmese pythons in south Florida (Mazzotti et al. 2011). A 52 percent reduction in the number of Burmese pythons removed (Florida Fish and Wildlife Conservation Commission [FWC] data) was noted in 2011 compared to the previous year. However, Burmese pythons of all age classes continue to be removed from the Everglades. The Burmese python is found throughout the southern Everglades, particularly in ENP and adjacent lands (e.g., East Coast Buffer Lands; north ENP boundary along Tamiami Trail) (**Figure A3-1-1**). Effective and feasible tools are currently very limited to manage large constrictor snake species if they become introduced into natural areas. Control options for this species are very limited, especially over large landscapes. Reed and Rodda (2009) reviewed control tools and their applicability to large constrictors in Florida. Potential controls include visual searching, traps, detection dogs, "Judas snakes," pheromone attractants, and toxicants. Many tools have the potential to benefit from additional research, but none is ready for landscape-level control or eradication of giant constrictor snake populations (Reed and Rodda 2009).

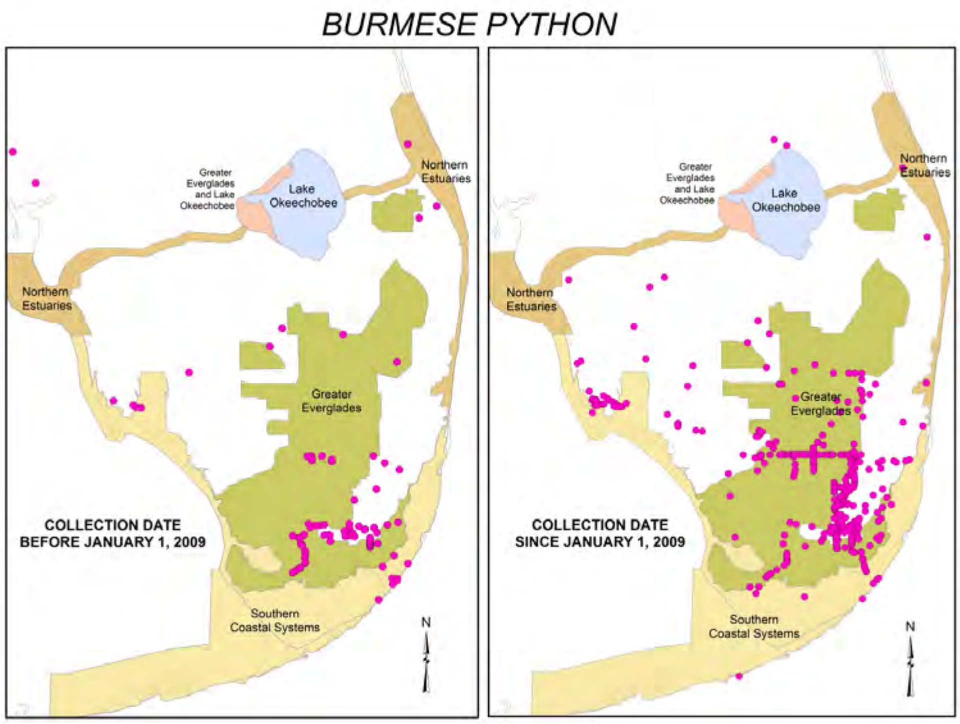

Figure A3-1-1. Locations of Burmese pythons removed from south Florida from 1999 through 2008 (left panel) and from 2009 to present (right panel)

Lionfish

The lionfish (*Pterois volitans*) is a nonlethal, venomous marine fish native to the Indian and Pacific oceans. Introduced to the region in the mid-1990s via aquarium releases in the United States (Hamner et al. 2007, Freshwater et al. 2009), the fish is now spreading throughout Caribbean and United States coastal waters at an alarming rate, including the Gulf of Mexico (Schofield et al. 2009). This predatory fish poses a significant threat to coral reef and mangrove ecosystems by significantly decreasing survival of fauna through predation and competition (Albins and Hixon 2008, Barbour et al. 2010, Albins 2013). Such reductions of herbivorous species could lead to overgrowth of seaweeds and subsequent coral reef decline (Morris et al. 2009).

Currently, within the western Atlantic, the lionfish is distributed from the southeastern Caribbean to North Carolina (Whitfield et al. 2007), which is believed to be the species' northern limit for overwintering survival. This invasive fish is now commonly observed along South Florida's Atlantic and Gulf Coasts. Pelagic eggs and larvae spread on ocean currents. Population densities reported in the Atlantic are much higher than in their native Indo-Pacific range (Morris and Whitfield 2009). Lionfish frequency of occurrence, abundance, and biomass were found to increase three- to six-fold between 2010 and 2011 in the Florida Keys in a variety of reef and nonreef habitats (Ruttenberg et al. 2013).

Control options are limited for this species. National Oceanic and Atmospheric Administration (NOAA) researchers are developing trapping techniques, which may have utility in deep or remote habitats. Fishery management strategies to recover predator populations (e.g., over-fished grouper) may help, but more information about potential predators is needed. Currently, the most feasible control strategy may be localized and intensive lionfish control programs targeting sensitive marine environments (e.g., fish nursery habitats) (Green et al. 2012).

Laurel Wilt Disease

Laurel wilt is a lethal disease of redbay (*Persea borbonia*) and other members of the Laurel family (Lauraceae). The disease is caused by a fungus (*Raffaelea lauricola*) that is introduced into trees by the wood-boring redbay ambrosia beetle (*Xyleborus glabratus*) (FDACS 2011). A native of Asia, the beetle was likely introduced into the United States via infested wood used for shipping crates (Harrington et al. 2011). Once infected, susceptible trees rapidly succumb to the pathogen and die. It also impacts other native and nonnative members of the Lauraceae (Hanula et al. 2009) including swamp bay (*P. palustris*), an important species of many Everglades plant communities.

Laurel wilt disease is now found throughout Florida. Since detection of the redbay ambrosia beetle in 2010 in Miami-Dade County, laurel wilt has spread across 133,740 hectares (ha) of the central Everglades and is also present in Water Conservation Area (WCA) 1. Laurel wilt is also widespread throughout the South Florida Water Management District’s (SFWMD's) East Coast land management region and the Kissimmee River Basin. There is currently no feasible method for controlling this pest or associated disease in natural areas. A systemic fungicide (propiconazole) can protect individual trees for up to one year, but widespread utilization in natural areas is impractical (Mayfield et al. 2008). Biological control and development of laurel wilt resistant strains of swamp bay are proposed for potential research focus. State and federal agencies are monitoring the spread of laurel wilt disease and the red bay ambrosia beetle through the Cooperative Agricultural Pest Survey program. There is little to no research underway to assess the ecological impacts of laurel wilt disease. Critical research areas include evaluating feasibility of identifying and propagating laurel wilt resistant strains of swamp bay, *Persea* seed/genetic conservation efforts, potential chemical or biological control tools, assessing impacts to vulnerable plant communities such as Everglades tree islands, and impacts on the Palamedes swallowtail butterfly (*Papilio palamedes*) and other host-specific commensals.

Overview of Invasive Species Management

A successful invasive species management program includes prevention, monitoring, early detection/rapid response, control, research, education, outreach and public awareness, and interagency coordination. As with most management activities, there are a number of risks and uncertainties associated with invasive species management. As restoration proceeds, invasive species may establish and spread as a direct result or independently of restoration activities. Invasive species management programs should be implemented throughout each phase of the restoration process to assist with achieving restoration goals.

Prevention

Prevention is the first line of defense and the most efficient and cost-effective approach to reduce the threat of invasive nonnative species. Successful prevention would reduce the rate of introduction and establishment and thereby reduce invasive species impacts. Critical elements to prevention include identifying high risk pathways, implementing actions to impede introductions, and utilizing tools to reduce intentional and unintentional introductions.

Monitoring

Monitoring is the collection and analysis of population measurements in order to determine changes in status and progress towards meeting a management objective (Elzinga et al. 1998). Monitoring is usually intended to detect relatively small changes in populations over time. Invasive species surveys and inventories detect populations and describe their spatial distributions over large landscapes, and are imperative to early detection of new populations.

Early Detection and Rapid Response

Early Detection and Rapid Response (EDRR) is the most cost-effective strategy to locate, contain, and eradicate invasive species early in the invasion process and will minimize ecological and economic impacts of nonnative species (Rejmanek and Pitcairn 2002). Early detection is a comprehensive and integrated system of active or passive surveys to locate, identify and report new species in order to implement management and control procedures while it is feasible and less costly (NISC 2008).

Control and Management

Integrated pest management is the coordinated use of the most appropriate strategies to prevent or reduce unacceptable levels of invasive species and their damage by utilizing the most economical means, and with the least possible hazard to people, property and the environment. Objectives of management can include complete eradication, population suppression, limiting spread, and reducing effects of invasive species. Often, invasive species are firmly established so that preventing further spread and reducing impacts by implementing control measures is the most feasible management measure. This concept is known as maintenance control. Maintenance control is defined as controlling an invasive species in order to maintain the population at the lowest feasible level.

Research

Research provides scientific information such as species biology and life history, control tools and methods for regulators, managers and the public to address invasive species. This is a critical element of successful management since for the majority of the introduced species there is minimal existing information.

Education, Outreach and Public Awareness

A successful strategy to address invasive species issues will require the public's understanding, cooperation and assistance. In order to acquire assistance from the public, a wide variety of education, outreach and training programs will be needed to raise awareness and gain public support. Targeting user groups such as gardeners, boaters, fisherman, pet owners, and other stakeholders will help to ensure a successful education, outreach and public awareness program is implemented. These user groups along with community organizations and industry can assist with prevention, monitoring and EDRR efforts.

Interagency Coordination

A regional management strategy and integration of a broad spectrum of control measures across multiple jurisdictions are required to successfully manage invasive species. As such, numerous groups and agencies are involved with nonnative invasive species management in south Florida and Everglades restoration. These include organizations such as the Florida Exotic Pest Plant Council (FLEPPC), Cooperative Invasive Species Management Areas (CISMAs), and the South Florida Ecosystem Restoration Task Force. CISMAs are a formal partnerships of federal, state, and local government agencies; tribes; individuals; or interested groups that manage invasive species and is defined by a geographic boundary. These partnerships provide an essential framework for communication and coordination for natural resource managers across jurisdictional boundaries. To date, there are 18 CISMAs in Florida, seven of which occur either wholly or partially within the CERP footprint (Rodgers et al. 2014).

Risk and Uncertainties

A major cause of uncertainty, in most cases, is that information for detailed, specific pre-project evaluations are needed for management activities to control particular invasive species. With the exception of a few well established and well studied species such as melaleuca, there is an information deficit on the status, potential impacts, and effective control techniques for priority species. This is particularly true for nonnative animals. Current knowledge on invasion mechanisms suggests that some restoration activities may facilitate the spread of certain priority species in the Everglades. For example, partial removal of canals and levees could encourage spread of, or provide sites for colonization by, numerous invasive species, including Brazilian pepper, Old World climbing fern, tegus, Nile monitors (*Varanus niloticus*), pythons, and Cuban treefrogs. However, there remains considerable uncertainty regarding the degree to which different species will respond, if at all, to restoration activities and how these responses will impact restoration performance measures.

Biological Control of Invasive Plant Species

Most nonindigenous species in Florida have limited or no predators, parasites, or pathogens. With few natural enemies in their new range, some nonindigenous species are able to grow larger, produce more offspring, spread quickly, and dramatically degrade Florida’s sensitive habitats. Classical biological control involves identifying host-specific natural enemies from the nonindigenous species’ native range and introducing them into Florida to reestablish equilibrium in the growth of the nonindigenous pest population.

Biological control research and implementation has yielded great successes in Florida but it is not a panacea. Detailed and lengthy studies are required to ensure that potential biological control agents will only attack the targeted invasive species and not native or agronomically important species. Biological control agents that are determined to be safe must pass through a lengthy review by state and federal regulatory agencies before they can be introduced. Despite these hurdles, biological control research and implementation has led to important advances in invasive plant management.

Melaleuca

The melaleuca weevil (*Oxyops vitiosa*) was introduced in 1997 and is now established on melaleuca throughout the region. Feeding by the weevil reduces the tree's reproductive potential as much as 90% (Tipping et al. 2008). Trees that do reproduce have smaller flowers producing fewer seeds (Pratt et al. 2005, Rayamajhi et al. 2008). The melaleuca psyllid (*Boreioglycaspis melaleucae*) was released in 2002. Data indicate that feeding by psyllids induces leaf drop, eventually resulting in tree defoliation. United States Department of Agriculture (USDA) entomologists have determined that psyllid feeding on melaleuca seedlings results in 60% mortality in less than a year (Franks et al. 2006). The combined effect of feeding by the weevil and the psyllid has led to more than 80% stem mortality in some stands (**Figure A3-1-2**) as well as decreases in melaleuca canopy cover over a 10-year period (1997–2007), resulting in a fourfold increase in other plant species diversity following the introduction of biological control agents (Rayamajhi et al. 2009). The melaleuca midge (*Lophodiplosis trifida*) is the most recent biological control agent for melaleuca. The larvae feed on the internal structures of the stem, which damages the flow of nutrients to melaleuca buds and leaves. Feeding by the insect also causes the stems to produce galls that dramatically alter the morphology of melaleuca stems. Feeding damage by larvae can kill small individuals and, in concert with the other melaleuca biological control agents, provides increased control of this invasive tree.

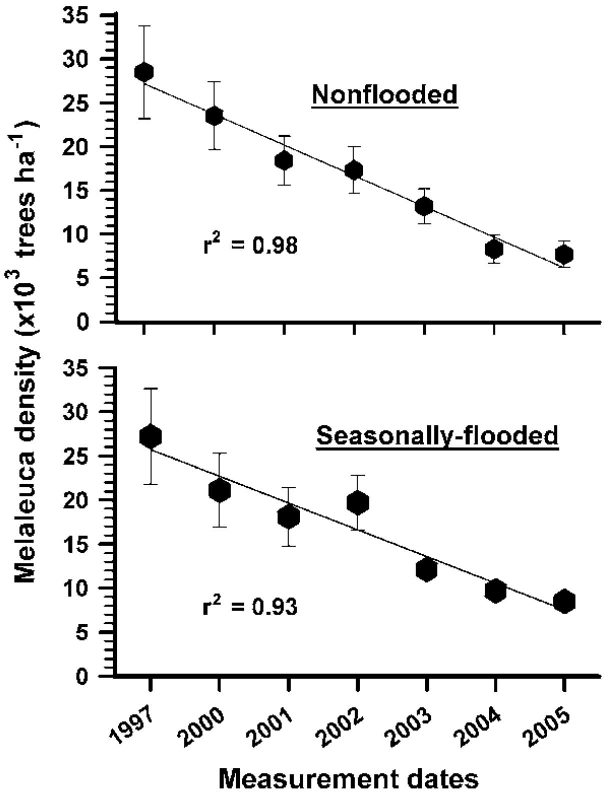

Figure A3-1-2. Density changes in two melaleuca populations in south Florida between 1997 and 2005. (Source: Rayamajhi et al. 2009. Note: x10^3 trees ha^{-1} – 1,000 trees per hectare.)

Old World Climbing Fern

The white lygodium moth (*Austromusotima camptozonale*) was the first agent to be released against Old World climbing fern in Florida. Releases of this insect began in 2004 and continued through 2007, with more than 40,000 individuals being mass reared and released, but no establishment was obtained. During 2011–2012, a second colonization effort with the moth was initiated using insects from a new lab colony. Approximately 18,000 larvae were distributed in several open releases, but aside from sporadic recoveries of relatively low numbers of progeny, there was no evidence to indicate that populations were establishing in the field.

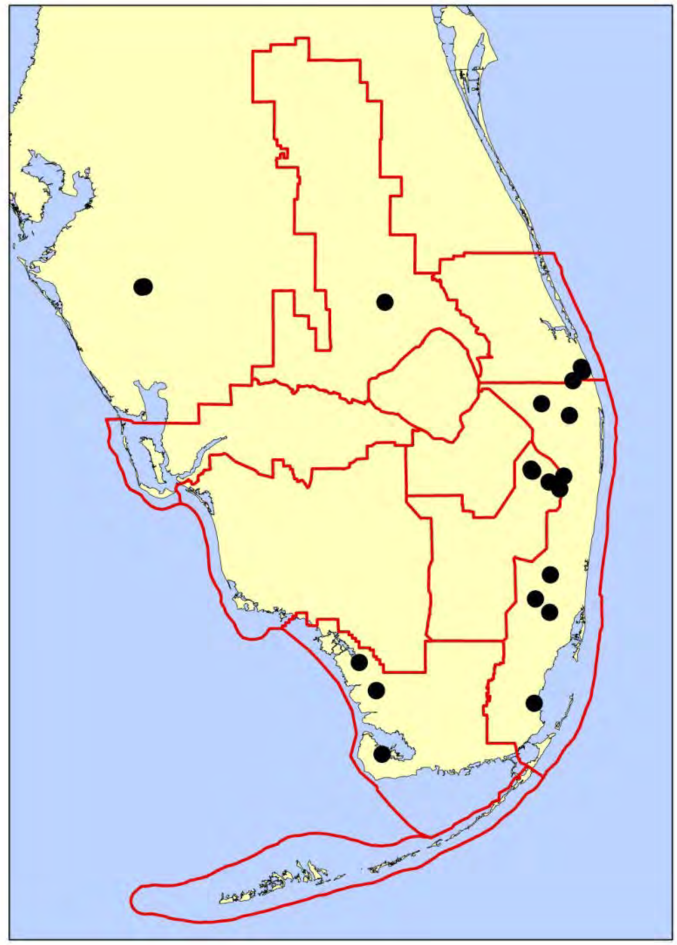

Figure A3-1-3. Release locations of the brown lygodium moth since 2008.

The brown lygodium moth (*Neomusotima conspurcatalis*) was released in Florida in 2008 and rapidly established large field populations at release sites (Boughton and Pemberton 2009) (**Figure A3-1-3**). Moth populations in Martin County have successfully survived four winter seasons without additional insect releases. Subsequent surveys revealed that moths are established in all sites into which they were released with the exception of ENP. An additional release of 13,500 larvae was made in May 2013.

The lygodium gall mite (*Floracarus perrepae*) induces leaf roll galls on the leaves of Old World climbing fern. The gall mite was released in 60 plots at five sites in south Florida during 2008 and 2009. Within release sites, the mite marginally established and continues to be present at low numbers and successful gall induction on field plants were much lower than anticipated. However, the mite has shown the ability to undergo long distance dispersal and colonize sites far from the release sites. Recently, a verified lygodium gall mite population was found in ENP and in Martin County, approximately 230 kilometers (km) and 20 km, respectively, from the release sites in Jonathan Dickenson State Park. It is too early to evaluate effects of these two biological control agents now established in Florida.

Important Initiatives and Policy/Regulatory Changes since CERP Authorization

Between the pet trade and horticulture industry Florida is a mecca for nonnative plants and animals. Plant nurseries and pet trades are multi-million dollar industries. Many of these nonnative species are invasive, posing ecologic and economic harm. In turn, federal, state and local governments spend millions of dollars each year attempting to control invasive species.

When the *Central and Southern Florida Comprehensive Review Study Final Integrated Feasibility Report and Programmatic Environmental Impact Statement* (also known as the "Yellow Book"; USACE and SFWMD 1999) was being developed, little attention was given to potential impacts of invasive species on restoration benefits. The focus was on "getting the water right." Invasive species cause harm to native flora and fauna. Ecosystem restoration planning and management need to consider impacts to restoration benefits and success due to the presence of invasive species, as well as the potential for a project to make things worse in certain areas by increasing abundance and expanding the spatial extent of invasive species.

Since CERP was initiated, there have been several legislative and policy actions taken by the federal and Florida State governments to prevent and reduce the introduction and spread of invasive species. These actions are important tools for preventing adverse impacts to Everglades restoration due to invasive species. **Table A3-1-2** includes a list of significant actions and initiatives that have occurred since CERP's inception.

Table A3-1-2. Important initiatives and policy/regulatory changes since 1999.

<table>
<tr><th>Year</th><th>Event</th><th>Brief Description</th></tr>
<tr><td rowspan="2">1999</td><td>Executive Order 13112, Invasive Species</td><td>On February 3, 1999, President Bill Clinton signed Executive Order 13112. It established a National Invasive Species Council and requires all federal agencies to evaluate the effects of federal actions with respect to invasive species. The order requires that federal actions that cause or promote the introduction or spread of invasive species not be carried out unless it has been determined that the benefits of the action clearly outweigh the potential harm caused by the invasive species.</td></tr>
<tr><td colspan="2">Central and Southern Florida Comprehensive Review Study Final Integrated Feasibility Report and Programmatic Environmental Impact Statement (USACE and SFWMD 1999)</td></tr>
<tr><td>2000</td><td>Water Resources Development Act of 2000</td><td>This act authorized the plan in the Central and Southern Florida Comprehensive Review Study Final Integrated Feasibility Report and Programmatic Environmental Impact Statement (USACE and SFWMD 1999)</td></tr>
<tr><td>2005</td><td>Early Detection and Distribution Mapping System (EDDMapS) Launched</td><td>EDDMapS was developed and launched in 2005 by the Center for Invasive Species and Ecosystem Health at the University of Georgia. EDDMapS is a web-based mapping system for documenting distribution of invasive species. EDDMapS is a national database network that includes data from other databases/organizations and volunteer observations. The data in EDDMapS is available to anyone. In 2011–2012, EDDMapS released applications for iPhone and Android smart phones, making it very easy and convenient for an individual to report a sighting immediately and the associated data and pictures.</td></tr>
</table>

Table A3-1-2. Continued.

Year	Event	Brief Description
2008	Everglades CISMA (ECISMA) Formed	The Florida Invasive Species Partnership is comprised of federal, state, and local government agencies, nongovernmental organizations, and universities. It coordinates the development of regional CISMAs, facilitates communication between CISMAs, and provides training and on-line access to existing resources and efforts. The CISMAs are local organizations that provide a means for sharing invasive plant and animal management information and resources across jurisdictional boundaries. There are currently 18 CISMAs covering the state of Florida. Eight of the 17 CISMAs partially or wholly cover the CERP footprint: Central Florida CISMA, ECISMA, Florida Keys Invasive Species Task Force, Heartland CISMA, Lake Okeechobee Aquatic Plant Management Interagency Task Force, Osceola County CISMA, Southwest Florida CISMA, and Treasure Coast CISMA. ECISMA, which encompasses the "heart" of the Everglades, was formalized in 2008 by a Memorandum of Understanding between the United Army Corps of Engineers (USACE), United States Fish and Wildlife Service (USFWS), SFWMD, FWC, and National Park Service (NPS).
2009	USACE Invasive Species Policy	The USACE implemented nationwide policy (CECW-ZA Memorandum, USACE Invasive Species Policy, dated June 2, 2009) that applies to all civil works projects and programs. The policy requires measures to prevent and reduce the establishment of invasive and nonnative species as a component of project implementation.
2010	CERP Memorandum – Requirements for Project Implementation Reports and Other Implementation Documents	During discussions between SFWMD and USACE regarding the CERP Master Agreement, it was agreed additional policy guidance should be provided regarding the management of invasive and native nuisance species. As a result, USACE issued a policy memorandum [CECW-SAD Memorandum, CERP – Requirements for Project Implementation Reports (PIRs) and Other Implementation Documents, dated May 27, 2010], which provides guidance for projects to assess impacts to restoration benefits due to invasive species and the need for future invasive species management.
2010	State of Florida – Conditional and Prohibited Species	As of July 2010, the State of Florida does not allow conditional and prohibited species from being acquired or kept as personal pets. Chapter 68-5 (Rules Relating to Nonnative Species) of the Florida Administrative Code, effective August 23, 2010, lists the conditional and prohibited species. The state implemented a hunting regulation that allows licensed hunters to remove pythons and other conditional reptiles (snakes and lizards) during established hunting seasons on certain wildlife management areas (WMAs), including Everglades and Francis S. Taylor WMA, Rotenberger WMA, Holey Land WMA, and Big Cypress WMA. The conditional reptiles include the following: • Indian or Burmese python (*Python molurus*) • Reticulated python (*Python reticulatus*) • Northern African python (*Python sebae*) • Southern African python (*Python natalensis*) • Amethystine python (*Morelia amethistinus*) • Scrub python (*Morelia kinghorni*) • Green anaconda (*Eunectes murinus*) • Nile monitor (*Varanus niloticus*)

Table A3-1-2. Continued.

Year	Event	Brief Description
2011	USACE Specification Language for Construction Activities (Prevention of Invasive and Nuisance Species Transfer)	In 2011, the Jacksonville District, USACE, developed and implemented specification language to prevent the spread of invasive species due to construction activities. The specification language requires proper cleaning of construction equipment and personal protective equipment to prevent the spread of invasive species and requires the contractor to have an Invasive and Nuisance Species Transfer Prevention Plan.
	USDA, Animal and Plant Health Inspection Service – Not Authorized Pending Pest Risk Analysis	The USDA, Animal and Plant Health Inspection Service regulates the import of horticultural and nursery plants. A regulation known as Quarantine 37 (Q-37) applies to plants or parts of plants intended for cultivation, planting, or propagation. The Animal and Plant Health Inspection Service has the authority to block the import of devastating pests and diseases that can hitchhike on imported plants. It completed a regulatory change of 7 Code of Federal Regulations 319, which created a special category of plant imports that could not be imported until their risks were assessed and deemed acceptable. Such risk determinations, or "screening," for plants is known as "not authorized pending pest risk analysis". Previously, Q-37 regulations classified imported nursery plants as either prohibited or restricted (allowed with conditions) and no risk analysis was required. The rule revision relies on scientific evidence to determine if a plant species is a potential pest or pest host. The rule became effective June 2011.
2012	Lacey Act Amendment	Effective March 2012, four nonnative constrictor snake species were listed as injurious under the Lacey Act, and thus, prohibited for importation and interstate movement in the United States. The four species include Burmese python, northern African python, southern African python, and yellow anaconda. Although the Burmese and northern African pythons are established in south Florida, the Lacey Act amendment will help prevent the introduction of new individual species into the United States, and south Florida. With the prevention of new individual species introduced, and the current on-going efforts to capture/kill the pythons already established in south Florida, it is hopeful the adverse impacts from these snakes to the Everglades ecosystem will be curtailed.
	CERP Guidance Memorandum 062.00: Invasive and Native Nuisance Species Management	Prior to July 2012, CERP projects were inadequately and inconsistently addressing invasive and native nuisance species. In July 2012, *CERP Guidance Memorandum 062.00: Invasive and Native Nuisance Species Management* (USACE and SFWMD 2012) was approved and signed into place. It requires CERP projects to incorporate invasive and native nuisance species management into the lifecycle of CERP projects. The memorandum provides guidance to project delivery teams for comprehensively assessing invasive and native nuisance species during the planning, design, construction, and operations, maintenance, repair, replacement, and rehabilitation phases. Items to consider, address, and assess during each phase are described in the memorandum. The memorandum also requires each project develop and implement an invasive and nuisance species management plan, which is intended to be a living document, updated throughout the life of a project.

Table A3-1-2. Continued.

Year	Event	Brief Description
2013	Python Challenge	FWC hosted a Python Challenge January 12–February 10, 2013 to raise public awareness about the Burmese python and why this python is a threat to the Everglades ecosystem. 68 Burmese pythons were harvested during the challenge. Scientific data was gathered from each python, which will be used to help manage for and assess impacts to the natural Everglades system.
	2014 System Status Report – Invasive Species Section	For the first time, RECOVER's System Status Report includes a section on invasive species.

CERP Projects That Address Invasive Species Issues

Prior to 2010, there was not a requirement for CERP project implementation reports (PIRs) and other implementation documents to address invasive and native nuisance invasive species issues. CERP PIRs completed prior to 2010 (C-43 Western Basin Reservoir, Indian River Lagoon – South, Picayune Strand Restoration Project (PSRP), and Site 1 Impoundment) did not address invasive species issues; however, invasive vegetation management was addressed in the construction phasing, transfer, and warranty plan (CPTWP). In order to effectively address invasive species as per the *2010 CERP Memorandum – Requirements for Project Implementation Plans and Other Implementation Documents*, the *CERP Guidance Memorandum 062.00: Invasive and Native Nuisance Species Management* (USACE and SFWMD 2012b) was developed and implemented. Some CERP projects have incorporated invasive species considerations into their PIRs. The section below discusses the projects that have an invasive and nuisance species management plans, per the guidance in the memoraundum, or another type of invasive species management plan.

Melaleuca Eradication and Other Exotic Plants Project

The Melaleuca Eradication and Other Exotic Plants Project is the only project within CERP that is dedicated to addressing invasive plant issues. This project is a collaborative effort of USACE, USDA and SFWMD.

Project construction began on the research annex in July 2011 in Davie, Florida. Federal funding for the project was provided through the American Recovery and Reinvestment Act of 2009. The physical construction of the research annex was completed in August 2013. The completion of this facility marks the first completed CERP project. After construction was completed, the project was transferred to the SFWMD, which is the local sponsor.

The new facility is part of a long-term plan to use biological controls to supplement existing management efforts to control and reduce the most aggressive, widespread and problematic invasive nonnative plants in south Florida. The focus of this systemwide approach includes biological controls to combat melaleuca, Brazilian pepper, Australian pine (*Casuarina equisetifolia*) and Old World climbing fern.

The Melaleuca Eradication and Other Exotic Plants Project encompasses approximately 18,000 square miles from Orlando to the Florida Reef Tract. The Kissimmee River, Lake Okeechobee and the Everglades are the dominant watersheds that connect a mosaic of wetlands, uplands, coastal areas, and marine areas. The study area includeds all or part of the following counties: Monroe, Miami-Dade, Broward, Collier, Palm Beach, Hendry, Martin, St. Lucie, Glades, Lee, Charlotte, Highlands, Okeechobee, Osceola, Orange and Polk.

The operational phase of the project will include rearing, releasing and monitoring the insects for aforementioned invasive plant species. Rearing will include the cultivating of insects to reduce or stop the reproductive capacities of melaleuca, Brazilian pepper, Old World climbing fern and Australian pine. A release strategy will be developed to ensure insects are more widely distributed. Monitoring will include field level monitoring of the approved biological controls and their effects on nonnative plants species. This will aid in determining success of the project.

C-111 Spreader Canal Western Project

The C-111 Spreader Canal Western Project (C-111 SCWP) completed an initial Nuisance and Exotic Vegetation Control Plan in 2009. The plan was incorporated into the January 2011 final PIR, updated in September 2013 for inclusion in the CPTWP, and renamed the Invasive and Nuisance Species Management Plan to more closely follow CERP Guidance Memorandum (CGM) 062.00. The invasive species management plan for the project focuses on vegetation and does not include and address invasive animal species.

The C-111 SCWP PIR/environmental impact statement (EIS) has been finalized (USACE and SFWMD 2011b) but the project has not been authorized for appropriations by congress to receive construction funds. However, the SFWMD has moved forward with some components of the project, as indicated in **Table A4-1-3**. Control and surveillance of nuisance and nonnative invasive plant species within the construction footprints was completed by SFWMD. Substrate removal was completed within the Frog Pond Detention Area. Species treated included species listed in the latest version of the FLEPPC invasive plant list and the Florida Department of Agriculture and Consumer Services prohibited plant list. Once initial treatment and/or removal were completed, SFWMD initiated surveillance to ensure regrowth or new invasions did not occur.

Burmese pythons, Argentine black and white tegus (*Tupinambis merianae*), and nonindigenous fish species are established in the C-111 SCWP project footprint. Impacts to restoration benefits and success for the C-111 SCWP project due to these species have not been assessed.

Table A3-1-3. Completed features of the C-111 SCWP.

Component	Items within Component
Frog Pond Detention Area	• Frog Pond Detention Area (Cells 1, 2 and 3) • Cement Lined Channel, C-200 • Unlined Header Channel, C-200 • Fixed Crest Weir, S-202A • Fixed Crest Weir, S-202B • Fixed Crest Weir, S-202C • Emergency Spillway, S-203A • Emergency Spillway, S-203B • Emergency Spillway, S-203C • Box Culvert, S-201
Pump Stations/Intake Canals	• Intake Canal, S-199 • Pump Station, S199 • Intake Canal, S-200 • Pump Station, S-200
Aerojet, C-110 and L-31E Canal Modifications	• Cement Lined Channel, C-199 • Unlined Channel, C-199A • Concrete Weir, AJ-1 • Box Culvert • Dredging at Aerojet Canal (2,500 linear feet) • Earth Plug at Aerojet Canal, AJ-2 • (10) Earth Plugs at C-110 Canal • (1) Earth Plug at L-31E Canal
Spillway Structure, S-198	• Not completed at this time.

Biscayne Bay Coastal Wetlands

The Biscanyne Bay Coastal Wetlands (BBCW) Project completed an initial Nuisance and Exotic Vegetation Control Plan in 2009. The plan was incorporated into the July 2011 Final PIR (USACE and SFWMD 2011a), updated September 2013 for inclusion in the CPTWP, and renamed the Invasive and Nuisance Species Management Plan to more closely follow CGM 062.00.

The BBCW PIR/EIS has been finalized, but the project has not been authorized for appropriations by congress to receive construction funds. However, the SFWMD is moving forward with some components of the BBCW Project plan. The SFWMD has completed the following project features:

Deering Estate Flow-way

- C-100A Spur Canal Extension, intake canal, 580 linear feet
- S-700 Pump Station (100 cubic feet per second)
- 60-inch Concrete Pipe (560 linear feet)
- Culvert Crossing at Old Cutler Road (75 linear feet)
- Outflow Structure

- Constructed Wetland (2.5 acres)

L-31E/L-31E Flow-way

- Four culverts with flap gates and manatee barriers (50 linear feet)

The invasive species management plan for the BBCW Project only focuses on vegetation and does not include and address invasive animal species. During construction of the L-31E features, a total of 291 acres of invasive vegetation was treated/removed (**Table A3-1-4**). Invasive species were not treated/removed in the Deering Estate area. It has been reported that nonnative fish species have spread into new areas due to the culverts installed in the L-31E Flow-way area. These nonnative fish species currently are not being monitored to evaluate potential impacts to project benefits.

Table A3-1-4. Invasive vegetation management completed within the L-31 Flow-way area.

Date	Total Acres Treated/Removed of Invasive Plant Species	Dominant Target Invasive Plant Species
2011	27	*Ardisia elliptica* (Shoebutton ardisia) *Casuarina equisetifolia* (Australian pine) *Lygodium microphyllum* (Old World climbing fern) *Cestrum diurnum* (Day-blooming jasmine) *Schinus terebinthifolius* (Brazilian pepper)
2012	239	
2013	25	
Total	**291**	

Picayune Strand Restoration Project

The PSRP PIR/EIS was completed September 2004 (USACE and SFWMD 2004b). Invasive species issues were not considered during the planning phase of the project. A vegetation management plan was completed in 2009 and was included in the CPTWP. This was the first CERP project to develop a management plan that addresses invasive plants and the first project to conduct management efforts during construction. However, funding is limited. The PIR/EIS did not include costs for invasive species management. As such, the appropriations received for construction did not include funds for invasive species management. Available funding for invasive species management during construction will continually be a challenge.

The PSRP encompasses approximately 58,000 acres. The historic installation of drainage canals and roads, as well as logging operations, caused severe alterations to the project landscape, which allowed for invasion of plant and animal species. Due to the size of the project, magnitude of the invasion, and limited funding, only invasive plant species present within the construction footprint (canal backfill, road degradation, and logging tram removal areas) will be managed. The lack of managing invasive species within the entire PSRP footprint will likely adversely impact restoration success and prevent anticipated restoration benefits from being achieved.

Construction of the PSRP is occurring in phases. The construction contracts have not been including specification language ("Prevention of Invasive and Nuisance Species Transfer") to prevent the spread of invasive and nuisance species due to construction activities. This simple management measure would help prevent spreading invasive species at the project and potentially reduce additional invasive species management costs.

Site 1 Impoundment/Fran Reich Preserve

The Site 1 Impoundment (now called Fran Reich Preserve) PIR/environmental assessment was completed in August 2006 (USACE and SFWMD 2006). The vegetation management plan was completed in March 2010, during the development of the CPTWP. Site 1 Impoundment was the first CERP project to include the USACE, Jacksonville District, specification language for construction activities, "Prevention of Invasive and Nuisance Species Transfer."

The 1,870-acre project footprint is unique since a majority of the area is covered by invasive species, dominated by Australian pine, melaleuca, and Brazilian pepper. Historically, the project lands were used for plant nurseries and pastures. In 2009, the project was split into two construction phases in order to take advantage of American Recovery and Reinvestment Act funds. Phase 1 construction activities encompass about 750 of the 1,800 acres. The invasive vegetation is being removed from Phase 1 lands; however, Phase 2 lands are not being managed and it is currently uncertain if Phase 2 will be completed. Thus, the species from the Phase 2 lands will spread to the Phase 1 lands, requiring the Phase 1 lands to be in constant treatment/management to control the invasive species on Phase 1 lands, especially at the constructed 6-acre wetland area, if ecological benefits are to be maintained. Construction phasing and the potential for a phase not to occur were not considered during development of the vegetation management plan. If Phase 2 does not occur, it is uncertain how the project lands will be managed and, thus, any restoration benefits achieved.

Indian River Lagoon - South and C-44 Reservoir and Storm Water Treatment Area

The Indian River Lagoon - South PIR/EIS was completed in 2004 (USACE and SFWMD 2004a). The C-44 Reservoir and Stormwater Treatment Area components developed a vegetation management plan in May 2010. Because invasive species management was not included until after the project received authorization for appropriations, funding for management is limited. The features currently under construction included the "Prevention of Invasive and Nuisance Species Transfer" specification language in contract.

Brazilian pepper, Australian pine, para grass (*Urochloa mutica*), and West Indian marsh grass (*Hymenachne amplexicaulis*) are dominant invasive species of concern. Agricultural canals exist in the project footprint, with abundant invasive aquatic vegetation. New canals and reservoirs will be constructed. Spread of invasive aquatic vegetative species will be a concern in the new canals, reservoirs, and the downstream estuaries.

Decomp Physical Model

The Decomp Physical Model (DPM) vegetation management plan was developed in February 2011. The DPM project included the USACE, Jacksonville District specification language for construction activities, "Prevention of Invasive and Nuisance Species Transfer." DPM is the first CERP project to complete surveillance and actively manage invasive and native nuisance species during the installation and operational periods.

Once operational, the project will introduce new flows in WCA 3B. These new flows could contribute to the spread and establishment of invasive and nuisance species. Cattail (*Typha* spp.) is a native nuisance species of concern for healthy marshes in the Everglades and its spread is a major concern for the WCA 3B marshes. The project completed a baseline vegetation survey via an unmanned aerial system prior to installation activities. Approximately 3,500 acres were flown (745 acres in the "pocket" between L-67A and L-67C, and 2,755 acres in WCA 3B, south of the L-67C) with the unmanned aerial system. The imagery obtained was used to determine exact areas and acreage with invasive species and cattails. Once the project is operational, subsequent surveys will be conducted to determine if any invasive species or cattail have spread as a consequence of the DPM project. If there has been any spread attributable to the DPM project, the plants will be managed. This is also the first CERP project that the Florida Department of Environmental Protection included invasive and native nuisance species management as a condition for a permit issued under the CERP Regulation Act.

Broward County Water Preserve Areas

The Broward County Water Preserve Areas (BCWPA) completed an invasive and nuisance species management plan in 2012. The BCWPA PIR (USACE and SFWMD 2012a) received a signed Chief's Report in May 2012 and record of decision in October 2012 (see http://www.evergladesplan.org/pm/projects/proj_45_broward_wpa.aspx). The project has not received authorization/appropriations to begin construction activities. A significant portion of the project lands are currently inhabited by invasive vegetation. Melaleuca is the dominant invasive plant species in the BCWPA project area. SFWMD actively manages invasive vegetation near the BCWPA project lands. However, management on BCWPA project lands will not occur until the project receives authorization/appropriations.

Central Everglades Planning Project

The Central Everglades Planning Project (CEPP) completed its initial invasive and nuisance species management plan in 2013. CEPP is the first CERP project to address invasive species during the planning phase. CEPP is also the first CERP project to address invasive vegetation and invasive animal species. The project released its planning document for public review in September 2013 (USACE and SFWMD 2013). CEPP has taken an in-depth approach to addressing invasive species issues. CEPP features present numerous new pathways for the spread and establishment of invasive species that will adversely affect project benefits. The invasive and nuisance species management plan thoroughly addresses potential vectors and methods for preventing and managing invasive species.

Priority Invasive Species by RECOVER Region

Table A3-1-5 includes a list of nonnative species that occur within each of the four RECOVER regions—Northern Estuaries (NE), Lake Okeechobee (LO), Greater Everglades (GE), and Southern Coastal Systems (SCS)—and are thought to be sufficiently invasive to potentially adversely affect natural habitats. The purpose is to provide a perspective on the magnitude of the invasive problem and to identify the primary invasive species involved. The table does not include nonnative species that are found within natural areas but are restricted to roadsides, trails, recently farmed plots, or other man-made areas. Plants are limited to FLEPPC Category I species (FLEPPC 2011). Subjective decisions had to be made in selection of species—different people would make different lists—but they could all serve the purpose. Some useful resources include the following:

- Everglades CISMA (ECISMA), Invasive Species Distribution Maps http://www.evergladescisma.org/distribution/
- Early Detection and Distribution Mapping System (EDDMaps) Species Distribution Maps (for nonnative species) http://www.eddmaps.org/distribution/
- FLEPPC 2011 Invasive Plant Lists. http://www.fleppc.org/list/list.htm
- FWC http://www.myfwc.com/nonnatives
- Institute for Regional Conservation database of plant species lists for conservation areas in South Florida http://regionalconservation.org/ircs/database/database.asp
- United States Geological Survey Nonindigenous Aquatic Species http://nas.er.usgs.gov/

Table A3-1-5. Summary of important south Florida invasive nonindigenous animal species and Category 1 invasive plant species within RECOVER regions.

Species					
Amphibians		**SCS**	**GE**	**NE**	**LO**
Rhinella marina	Giant Toad		x		x
Osteopilus septentrionalis	Cuban Treefrog	x	x		x
Reptiles		**SCS**	**GE**	**NE**	**LO**
Anolis equestris equestris	Knight Anole		x		
Anolis sagrei	Brown Anole		x		x
Boa constrictor	Common Boa		x		
Caiman crocodilus	Spectacled Caiman	x	x		
Eleutherodactylus planirostris	Greenhouse Frog	x	x		x
Hemidactylus mabouia	Tropical House Gecko	x	x		
Iguana iguana	Green Iguana		x		x
Python molurus bivittatus	Burmese Python	x	x		
Python sebae	Northern African Python		x		
Tupinambis teguixin	Gold Tegu		x		
Tupinambis merianae	Argentine Black & White Tegu		x		
Varanus niloticus	Nile Monitor		x		

Table A3-1-5. Continued.

Species					
Birds		**SCS**	**GE**	**NE**	**LO**
Alopochen aegyptiacus	Egyptian Goose	x	x	x	x
Cairina moschata	Muscovy Duck	x	x	x	x
Porphyrio porphyrio	Purple Swamphen	x	x		x
Streptopelia decaocta	Eurasian Collared-Dove		x		x
Sturnus vulgaris	European Starling		x		x
Threskiornis aethiopicus	Sacred Ibis		x		
Mammals		**SCS**	**GE**	**NE**	**LO**
Rattus rattus	Black Rat		x		x
Sus scrofa	Feral Pig	x	x		x
Fishes		**SCS**	**GE**	**NE**	**LO**
Astronotus ocellatus	Oscar	x	x		x
Belonesox belizanus	Pike Killifish	x	x		
Channa marulius	Bullseye Snakehead	x	x		
Cichla ocellaris	Butterfly Peacock Cichlid	x	x		
Cichlasoma bimaculatum	Black Acara	x	x	x	x
Cichlasoma managuense	Jaguar Guapote		x		
Cichlasoma urophthalmus	Mayan Cichlid	x	x		x
Clarias batrachus	Walking Catfish	x	x		x
Hemichromis letourneuxi	African Jewelfish		x		
Hoplosternum littorale	Brown Hoplo		x	x	x
Macrognathus siamensis	Spot-Finned Spiny Eel		x		
Monopterus albus	Asian Swamp Eel		x		
Oreochromis aureus	Blue Tilapia	x	x	x	x
Oreochromis mossambicus	Mozambique Tilapia		x	x	
Pterois volitans	Lionfish	x		x	
Pterygoplichthys disjunctivus	Vermiculated Sailfin Catfish				x
Pterygoplichthys multiradiatus	Orinoco Sailfin Catfish		x		x
Tilapia mariae	Spotted Tilapia	x	x		
Invertebrates		**SCS**	**GE**	**NE**	**LO**
Balanus reticulatus	Barnacle	x			
Balanus trigonus	Barnacle	x		x	
Callinectes bocourti	Bocourt's Swimming Crab	x			
Charybdis hellerii	Indian Ocean Portunid Crab	x		x	
Cittarium pica	West Indian Top Shell				
Corbicula fluminea	Asian Clam	x	x	x	x
Cuthona perca	Lake Merritt Cuthona	x			
Daphnia lumholtzi	Water Flea		x		x
Glossodoris sedan	Marine Nudibranch	x			
Haliplanella luciae (= H. lineata)	Sea Anemone	x			
Litopenaeus stylirostris	Pacific White Shrimp	x			
Litopenaeus vannamei	Pacific White Shrimp	x			
Lyrodus mediolobatus	Indo-Pacific Shipworm			x	
Melanoides tuberculatus	Red-Rim Melania	x			

Table A3-1-5. Continued.

Species					
Invertebrates (continued)		**SCS**	**GE**	**NE**	**LO**
Metamasius callizona	Mexican Bromeliad Weevil		x		
Mytella charruana	Charru Mussel			x	
Parapristina verticillata	Fig Wasp		x		
Paratachardina lobata	Lobate Lac Scale		x		
Perna viridis	Green Mussel			x	
Phyllorhiza punctata	Spotted Jellyfish			x	
Pinctada margaritifera	Black-Lipped Pearl Oyster			x	
Pomacea insularum (P. maculata)	Island Applesnail	x	x		x
Solenopsis invicta	Imported Fire Ant	x	x		x
Sphaeroma terebrans	Wood-Boring Isopod	x		x	
Sphaeroma walkeri	Fouling Isopod	x		x	
Styela plicata	Sea Squirt			x	
Sundanella sibogae	Bryozoan			x	
Tridacna crocea	Giant Clam			x	
Tridacna maxima	Giant Clam			x	
Victorella pavida	Bryozoan			x	
Watersipora subovoidea	Bryozoan			x	
Xyleborus glabratus	Redbay Ambrosia Beetle		x		
Zachrysia provisoria	Cuban Garden Snail		x		x
Category I Invasive Pest Plants		**SCS**	**GE**	**NE**	**LO**
Abrus precatorius	Rosary Pea	x	x		x
Acacia auriculiformis	Earleaf Acacia	x	x		
Ardisia elliptica	Shoebutton Ardisia		x		
Bauhinia variegata	Orchid Tree	x	x		
Bischofia javanica	Bishopwood		x		
Calophyllum antillanum	Santa Maria	x	x	x	
Casuarina equisetifolia	Australian Pine	x	x		x
Colocasia esculenta	Wild Taro		x		x
Colubrina asiatica	Lather Leaf	x	x	x	
Cupaniopsis anacardioides	Carrotwood		x		
Dioscorea bulbifera	Air-Potato	x	x		x
Eichhornia crassipes	Water-Hyacinth	x	x		x
Eugenia uniflora	Surinam Cherry	x	x		
Ficus microcarpa	Laurel Fig	x	x		x
Hydrilla verticillata	Hydrilla	x	x	x	x
Hygrophila polysperma	Green Hygro		x		
Hymenachne amplexicaulis	West Indian Marsh Grass	x	x		x
Imperata cylindrica	Cogon Grass	x	x		x
Jasminum fluminense	Brazilian Jasmine		x		
Lantana camara	Lantana	x	x		x
Ludwigia peruviana	Peruvian Primrosewillow	x	x		x
Lumnitzera racemosa	Indo-Pacific Black Mangrove	x			

Table A3-1-5. Continued.

Species					
Category I Invasive Pest Plants (continued)		**SCS**	**GE**	**NE**	**LO**
Luziola subintegra	Tropical Watergrass		x	x	x
Lygodium microphyllum	Old World Climbing Fern	x	x		x
Lygodium japonicum	Japanese Climbing Fern	x			
Manilkara zapota	Sapodilla		x		
Melaleuca quinquenervia	Melaleuca, Paper Bark	x	x		x
Melinis repens	Natal Grass	x	x		x
Nephrolepis brownii (N. multiflora)	Asian Sword Fern		x		
Nephrolepis cordifolia	Tuberous Sword Fern	x	x		x
Neyraudia reynaudiana	Burmareed	x	x		x
Panicum repens	Torpedo Grass	x	x		x
Pennisetum purpureum	Napier Grass	x	x		x
Pistia stratiotes	Waterlettuce		x		x
Psidium guajava	Guava	x	x		x
Rhodomyrtus tomentosa	Downy Rosemyrtle	x			
Ruellia tweediana (R. simplex)	Mexican Petunia		x		x
Salvinia minima	Water Spangles	x	x		x
Scaevola taccada	Beach Naupaka	x	x	x	
Schinus terebinthifolius	Brazilian Pepper	x	x		x
Schefflera actinophylla	Australian Umbrella Tree	x	x		
Scleria lacustris	Wright's Nutrush		x		x
Solanum viarum	Tropical Soda Apple	x			x
Syngonium podophyllum	Arrowhead Vine	x	x		
Syzygium cumini	Java Plum	x	x		x
Tectaria incisa	Incised Halberd Fern		x		
Thespesia populnea	Seaside Mahoe	x	x	x	
Urena lobata	Caesarweed	x	x		x
Urochloa mutica	Para Grass	x	x		x

Greater Everglades Region

Introduction

Thirty-three species of invasive, nonindigenous plants and at least 70 nonindigenous species of animals are present in the GE region (**Table A3-1-6**). Well established species such as melaleuca, Old World climbing fern, and the Burmese python continue to be systemwide priorities for long-term management. Eradication of these established invasive species is considered unlikely, so management objectives focus on population suppression to minimize ecological impacts. The Interagency Melaleuca Management Program is an example of a successful long-term control effort. Melaleuca has been systematically cleared from large portions of the GE region, and is now considered under maintenance control in many areas. Biological control of melaleuca is showing promising results, with documented reductions of melaleuca. Despite the significant progress in controlling melaleuca, the plant continues to reestablish in previous infestation areas. This fact underscores the need for long-term monitoring and control of this species throughout the region.

A growing list of newly detected or geographically isolated nonindigenous species is the focus of containment and possible eradication within or immediately adjacent to the GE region. For example, the Argentine black and white tegu is a relatively recent introduction to Florida and is now the object of an interagency rapid response control effort in southeastern Florida near ENP and the Southern Glades Wildlife Management Area. **Table A3-1-6** includes a list of plant and animal species currently targeted for rapid response efforts in or near the GE region. This species list only includes priority species that agencies are actively managing for eradication (removal of self-sustaining populations in south Florida) or regional containment (preventing introduction from other regions or limiting spread within the region).

Table A3-1-6. Invasive, nonindigenous species currently targeted for rapid response efforts in and near the GE region.

Classification	Common Name	Scientific Name	Location(s)	Response Goal
Plant	Tropical American Watergrass	*Luziola subintegra*	Western Miami-Dade County	Regional Containment
	Feathered Mosquito Fern	*Azolla pinnata*	WCA 1 WCA 2A	Regional Containment
	Mile-a-minute	*Mikania micrantha*	Western Miami-Dade County	Eradication
	West Indian Marsh Grass	*Hymenachne amplexicaulis*	WCA 2A	Regional Containment
Reptile	Northern African Python	*Python sebae*	Bird Drive Basin C-4 Impoundment Pennsuco Wetlands	Eradication
	Argentine Black and White Tegu	*Tupinambis merianae*	C-111 Basin Southern Glades Wild Management Area	Regional Containment
	Chameleons	*Chamaeleo calyptratus, Furcifer oustaleti*	C-111 Basin Western Miami-Dade County	Eradication
	Spectacled Caiman	*Caiman crocodilus*	Western Miami-Dade County	Eradication
	Nile Monitor	*Varanus niloticus*	Central Palm Beach County Miami-Dade County	Eradication
Bird	Sacred Ibis	*Threskiornis aethiopicus*	Throughout region	Eradication*
Mammal	Gambian Pouched Rat	*Cricetomys gambianus*	Florida Keys	Eradication

*Possibly eradicated

As previously discussed, certain invasive species are expected to alter Everglades restoration outcomes if they are allowed to proliferate. Highly invasive plant species can alter native plant communities in the Everglades by displacing native species and detrimentally affecting ecological functions (Mazzotti et al. 1981, Gordon 1998, Ewel 1986, Rayamajhi et al. 2009, Brandt and Black 2001, Hutchinson et al. 2006). Invasive animal species in the Everglades can displace and compete with native wildlife species (Waddle et al. 2010, Rice et al. 2011), alter food web dynamics (Dorcas et al. 2011), prey on or compete with endangered species (Greene et al. 2007, Dove et al. 2011), or act as foreign disease vectors (Kendra et al. 2013). **Table A3-1-7** summarizes the potential impacts of priority invasive species or guilds in reference to the CERP RECOVER performance measures, which can be viewed at http://www.evergladesplan.org/pm/recover/eval_team_perf_measures.aspx. In most cases, there is limited information on individual species to develop highly accurate predictions on the magnitude and scale of these potential impacts to restoration.

Table A3-1-7. Potential invasive species threats to RECOVER performance measures for the GE region.

Performance Measure	Invasive Species/Guild	Potential Threat or Risk
American Alligator Distribution, Size, Nesting and Condition	Argentine Black and white tegu Nile monitor Burmese python	Reduced reproduction due to egg and hatchling predation Direct competition for food resources Direct predation by pythons
American Crocodile – Juvenile Growth and Survival	Argentine Black and white tegu Nile monitor Burmese python	Reduced reproduction due to egg and hatchling predation Direct competition for food resources Direct predation by pythons
Marl Prairie Cape Sable Seaside Sparrow Habitat	Melaleuca Australian pine	Degradation of nesting habitat due to changes in plant community structure and fire regimes
Prey-Based Freshwater Fish Density Performance Measure	Nonindigenous freshwater fish	Reduced native small fish density due to predation or competitive interactions
Ridge and Slough Community Sustainability	Melaleuca Australian pine Old World climbing fern Brazilian pepper	Alteration of plant community structure, microtopography, and fire regimes
Wet Prairie	Melaleuca Australian pine	Displacement of native plant community Alteration of fire regimes Loss of wildlife habitat

Distribution and Abundance of Four Priority Invasive Plant Species in the Greater Everglades

Four invasive, nonindigenous plant species—Australian pine, Brazilian pepper, melaleuca, and Old World climbing fern—are well established in the GE region and are considered high priorities for control due to documented ecological impacts. Interagency biologists have conducted regionwide aerial surveillance and mapping for these four species since 1993. The aerial reconnaissance program has two primary objectives: (1) determine the regional distribution and relative abundance of invasive plants targeted for management, and (2) provide rapid and cost-efficient spatial data to land managers to direct control efforts.

Methods

Data presented in this section were collected in 2003 and 2012–2013 as part of a biennial regional aerial reconnaissance program led by SFWMD and the National Park Service (NPS). Biologists in low-flying aircraft (~150 meter altitude) made visual estimates of invasive plant locations and abundance along fixed east-west transects. A detailed description of survey methods is provided by Pernas and Ferriter (2008) and Rodgers et al. (2011). Zonal analysis of these data using 1-km and 4-km grid systems was utilized to determine the current status of the four species as well as landscape-level changes in distribution and abundance between 2003 and 2013. Estimates of infestation area and canopy area (infestation polygon area x percent cover of target species) were calculated for the 2012–2013 data set.

Results and Discussion

Australian Pine

Australian pine is the least abundant of the targeted species in the survey area with a total infestation area of 2,765 ha as of 2013 (**Table A3-1-8**). Percent cover of Australian pine ranged from 0.003% to 48% within 1-km cells (mean = 2.3%) (**Figure A3-1-4**). This species is now at maintenance control levels in most areas of the region, meaning that continuous low intensity management will keep this species at a low infestation level. The remaining large infestations within the region occur in the South Dade Wetlands and Model Lands Basin where it forms dense stands to widely scattered patches in remote mangrove swamps and sawgrass marsh. A comparison between the 2003 and 2012–2013 regional distribution data indicates a decrease in relative abundance of Australian pine over the ten-year period, particularly in the eastern Everglades where active herbicide control programs are in place (**Figure A3-1-5**).

Table A3-1-8. Infested area and canopy area of four priority invasive plant species within the GE region in 2013. Canopy area is calculated as the product of infestation polygon area and percent cover of target species in the polygon.

Species	Infested Area (ha)	Canopy Area (ha)
Brazilian Pepper	14,442	3,499
Melaleuca	7,326	935
Old World Climbing Fern	9,046	2,111
Australian Pine	2,765	423

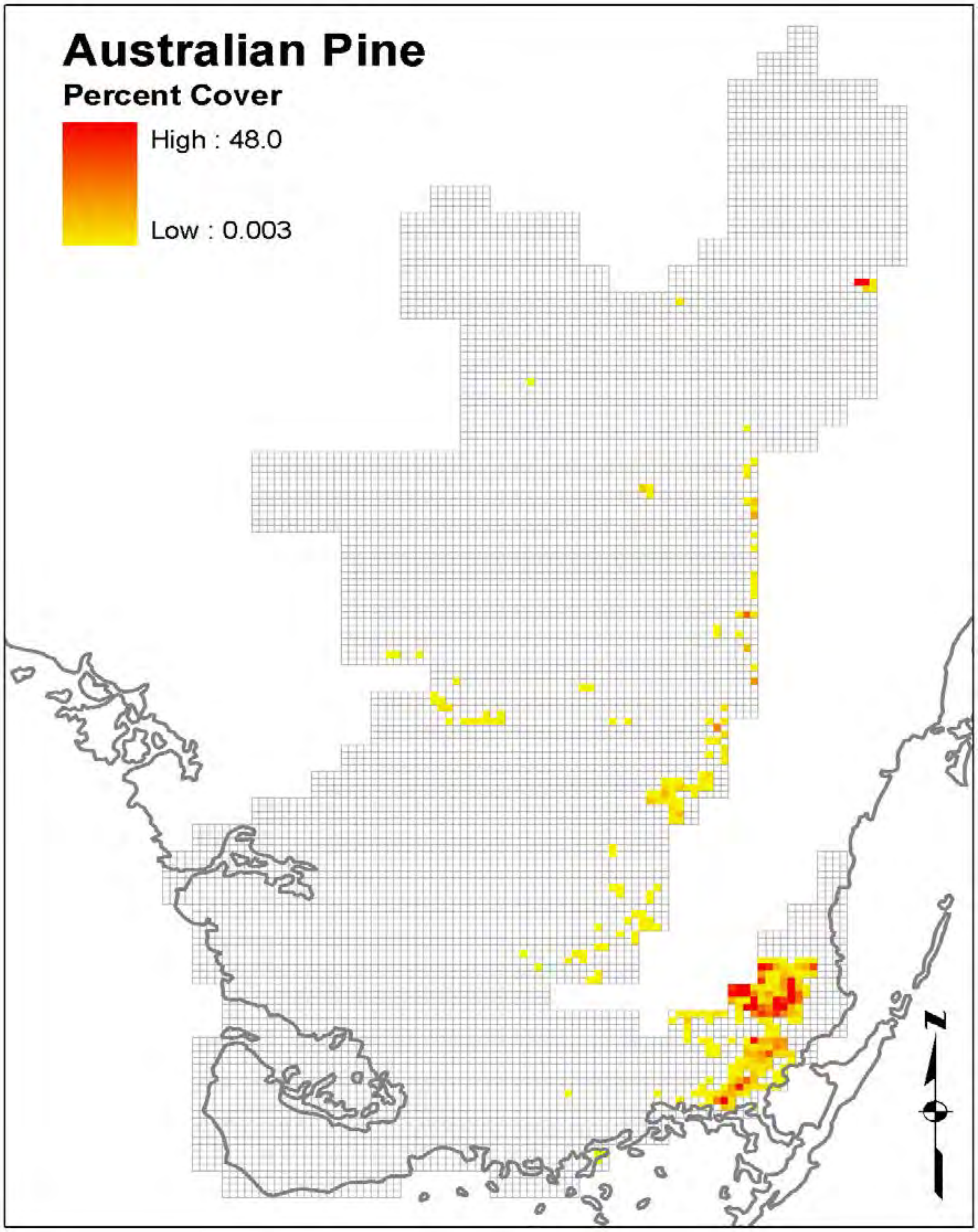

Figure A3-1-4. Distribution and relative abundance of Australian pine in the GE region in 2013. Zonal analysis (1-km grid) utilizes digital aerial sketch mapping data collected in 2012 and 2013. Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. Values represent percent cover within 1-km grid cells (RANGE = 0.003 - 48.0%).

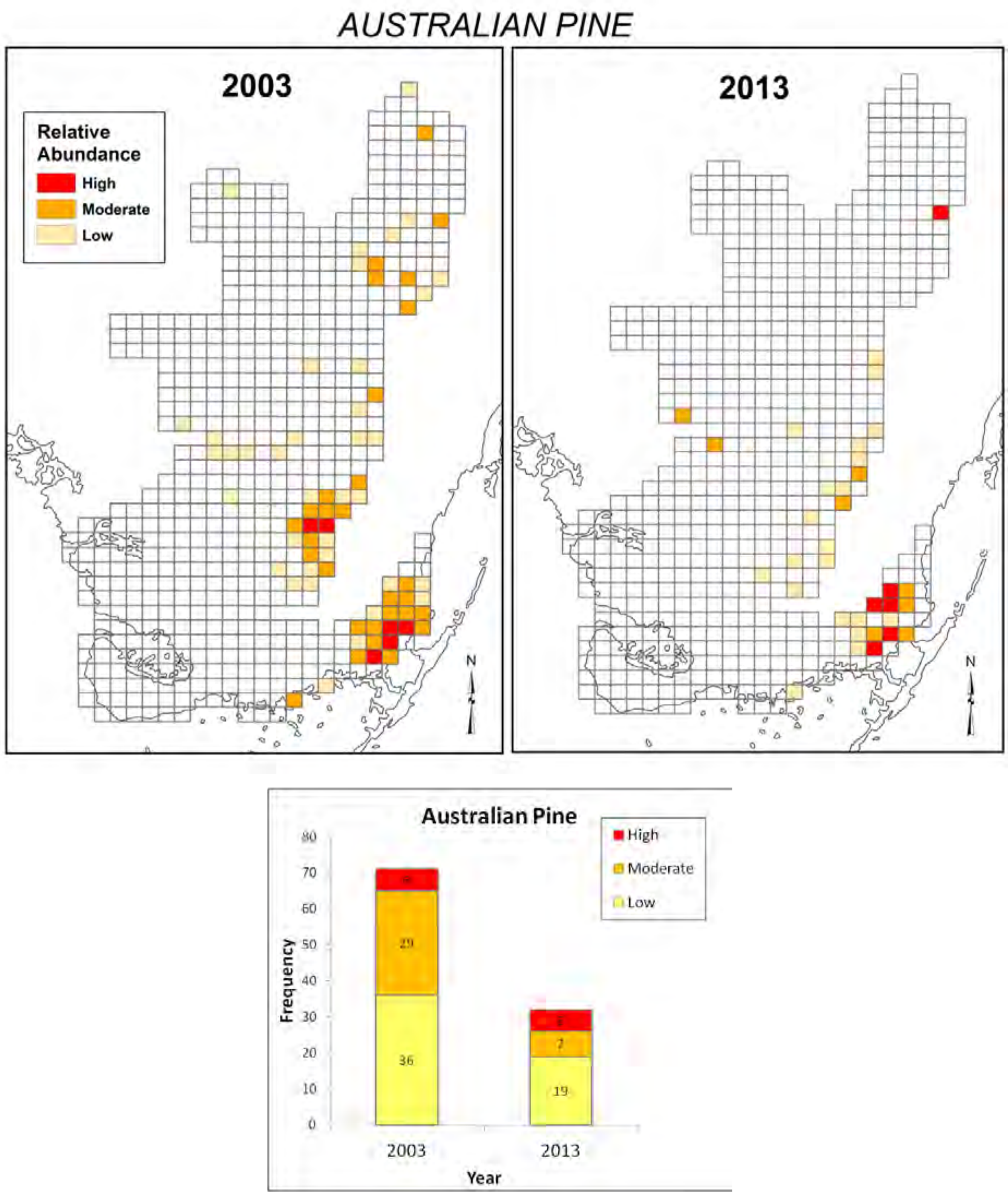

Figure A3-1-5. Distribution and relative abundance of Australian pine in the GE region 2003–2013. Zonal analysis (4-km grid) utilizes systematic reconnaissance flight data (2003) and digital aerial sketch mapping data (2012–2013). Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. The stacked bar chart shows the frequencies of 4-km grids with high, moderate, and low infestation levels in 2003 and 2013.

Brazilian Pepper

Brazilian pepper is widely distributed throughout the survey area with an estimated infestation area of 14,442 ha (**Table A3-1-8**). Percent cover of Brazilian pepper ranged from 0.001% to 87.5% within 1-km cells (mean = 3.0%) (**Figure A3-1-6**). It is a major component of buttonwood (*Conocarpus erectus*) swamps and graminoid marshes along the fringes of southwestern mangrove swamps of the ENP. The most severe infestations extend from the Ten Thousand Islands Area to Cape Sable, representing roughly 60% of the total infestation area within the survey area.

This invasive plant was also detected on tree islands throughout the central Everglades region. In some cases, this species is dominant or co-dominant in the canopy. Ground-based observations of tree islands infested with Brazilian pepper revealed that little to no understory native vegetation remains beneath the canopy. A comparison between the 2003 and 2012–2013 regional distribution data indicates an overall decrease in the relative abundance of Brazilian pepper over the ten-year period (**Figure A3-1-7**). This reduction is likely due to ongoing regional herbicide control efforts in WCA 2, WCA 3, and Big Cypress National Preserve (BCNP). However, the frequency of grid cells classified with high abundance increased 67% during the ten-year period. This increase of high abundance patches occurs primarily in the southwestern portions of ENP in and around Cape Sable.

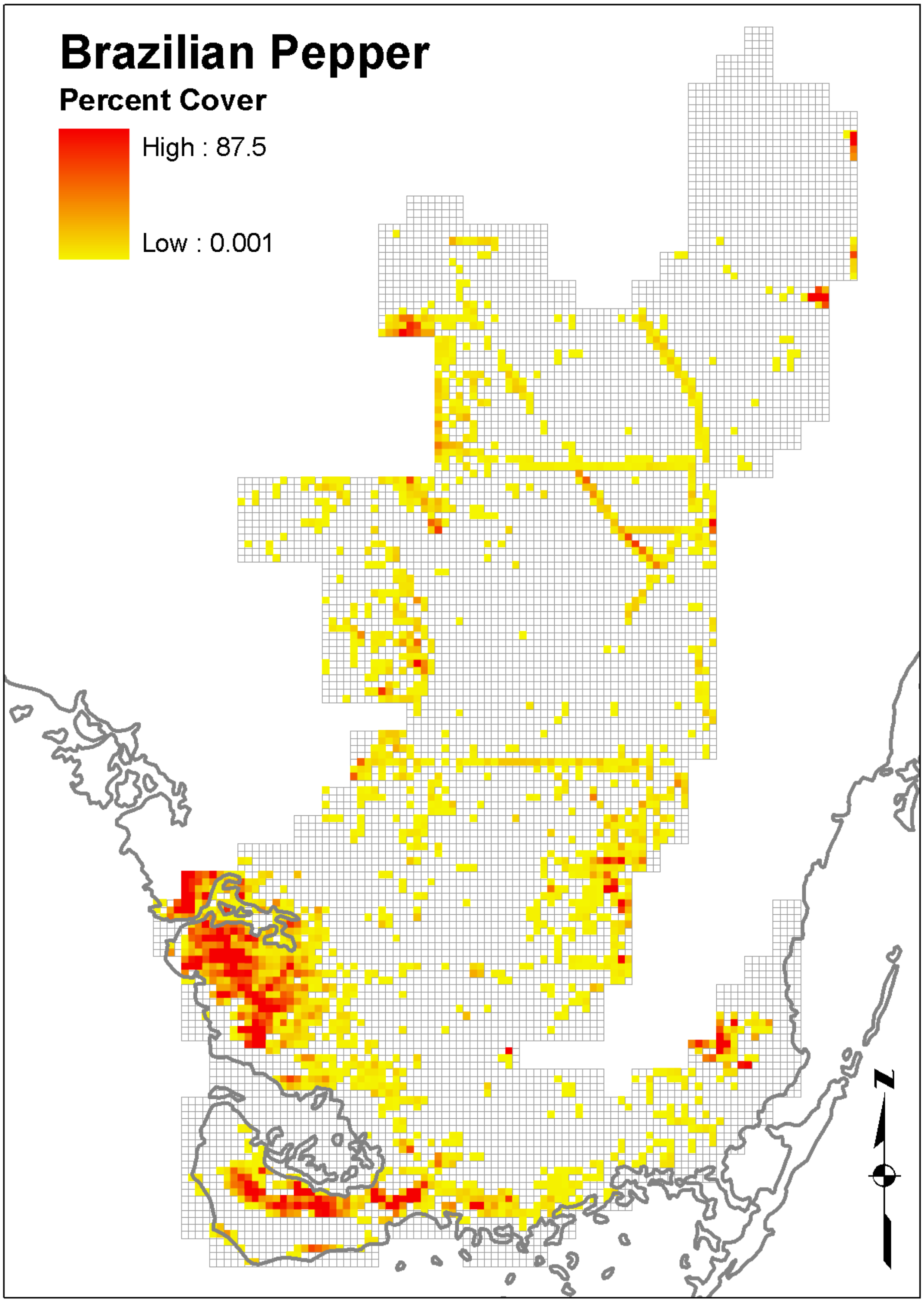

Figure A3-1-6. Distribution and relative abundance of Brazilian pepper in the GE region in 2013. Zonal analysis (1-km grid) utilizes digital aerial sketch mapping data collected in 2012 and 2013. Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. Values represent percent cover within 1-km grid cells (RANGE = 0.001 - 47.5%).

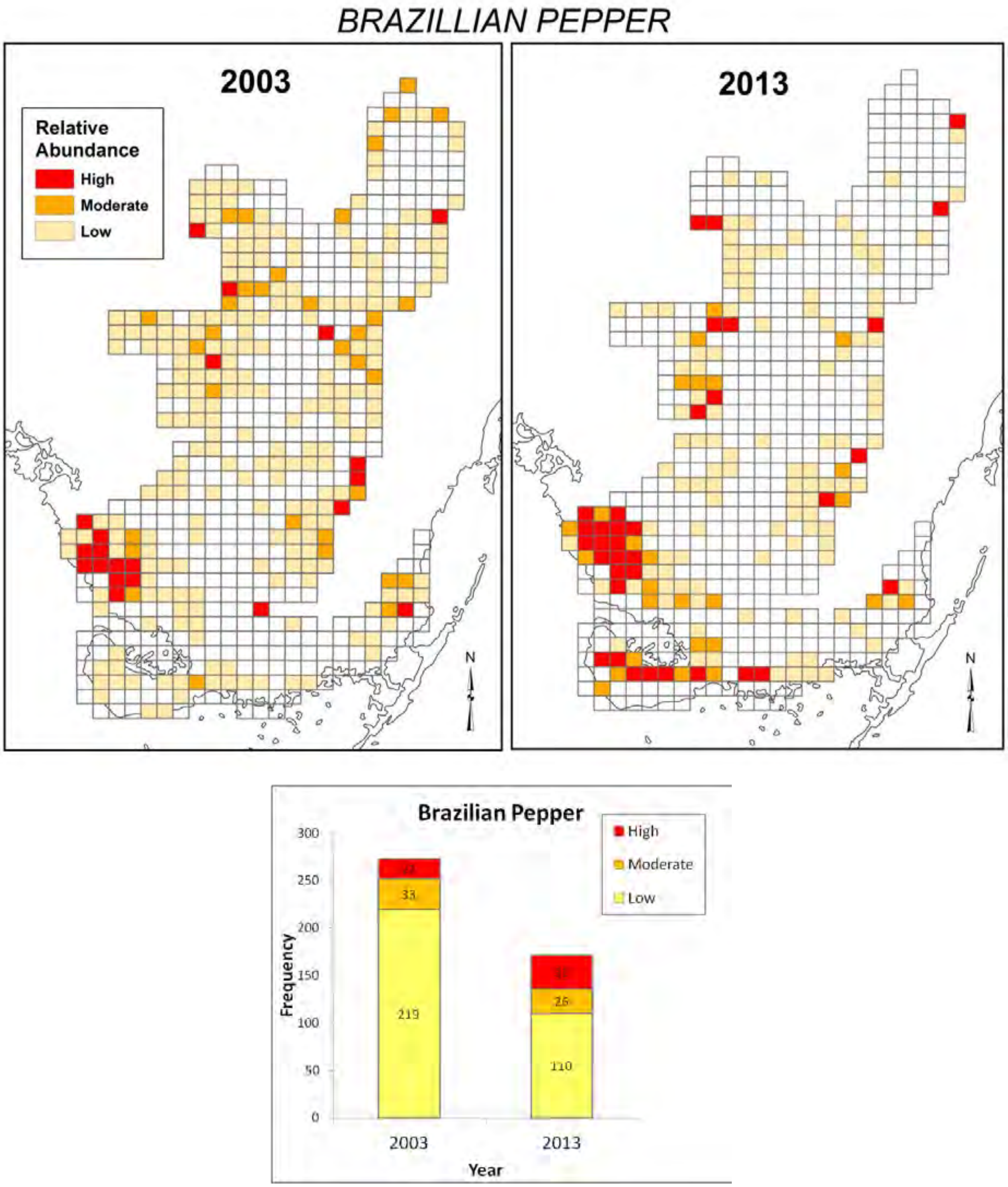

Figure A3-1-7. Distribution and relative abundance of Brazilian pepper in the GE region 2003–2013. Zonal analysis (4-km grid) utilizes systematic reconnaissance flight data (2003) and digital aerial sketch mapping data (2012–2013). Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. The stacked bar chart shows the frequencies of 4-km grids with high, moderate, and low infestation levels in 2003 and 2013.

Melaleuca

Melaleuca occupies an estimated 7,326 ha within the GE region (**Table A3-1-8**). Percent cover of melaleuca ranged from 0.002% to 63.3% within 1-km cells (mean = 2.1%) (**Figure A3-1-8**). The most significant infestations occur in SFWMD-owned project lands in the East Coast Buffer Area and the northern portions of WCA 1.

While the general distribution in the region is similar between 2003 and 2013, the relative abundance of melaleuca decreased substantially (**Figure A3-1-9**). Grid cells classified as high abundance decreased 73% during the ten-year period. Intensive herbicide control efforts throughout the region, combined with the successful establishment of biological control agents, have greatly reduced the intensity of melaleuca infestations in the region.

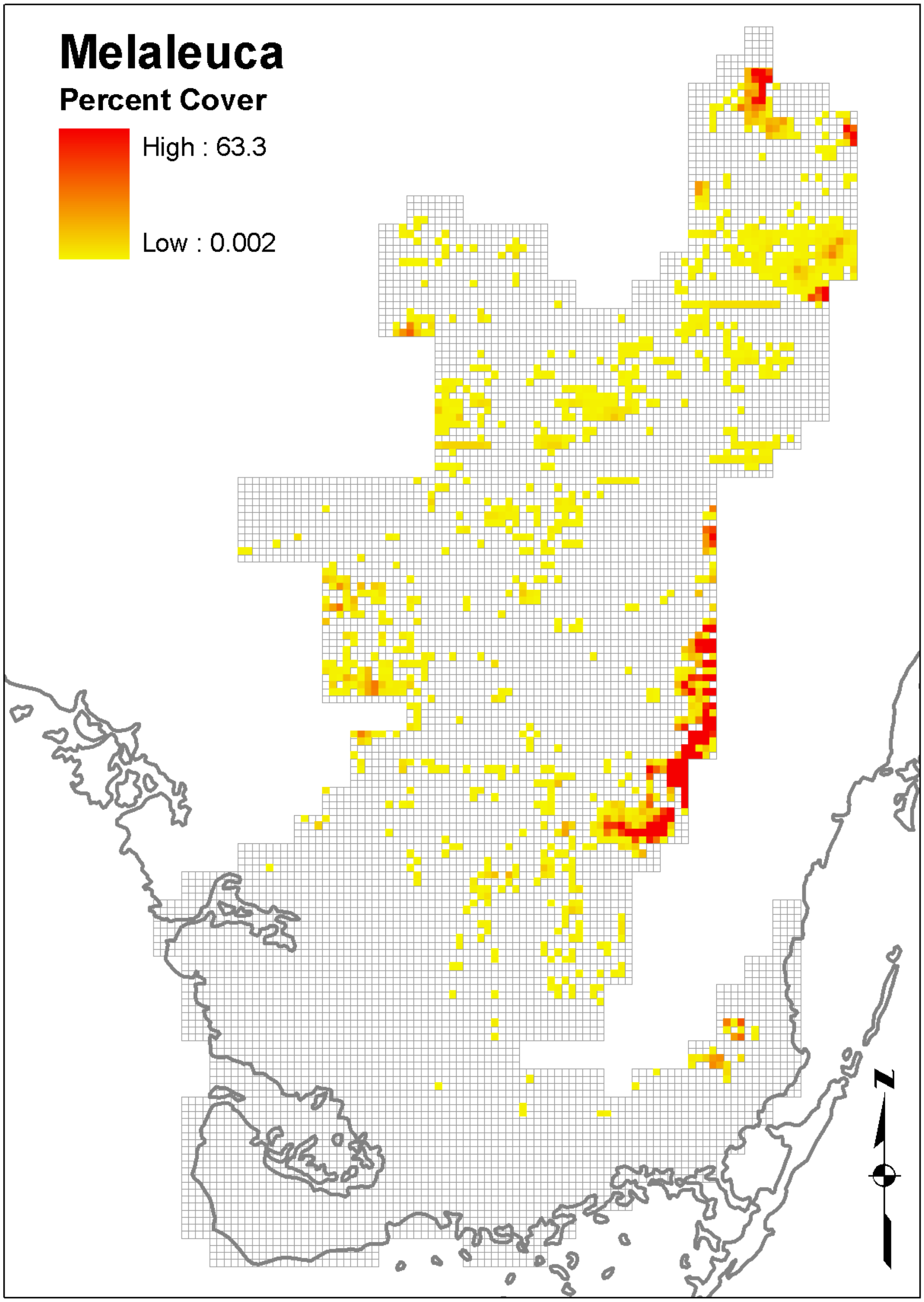

Figure A3-1-8. Distribution and relative abundance of melaleuca in the GE region in 2013. Zonal analysis (1-km grid) utilizes digital aerial sketch mapping data collected in 2012 and 2013. Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. Values represent percent cover within 1-km grid cells (RANGE = 0.002 - 63.3%).

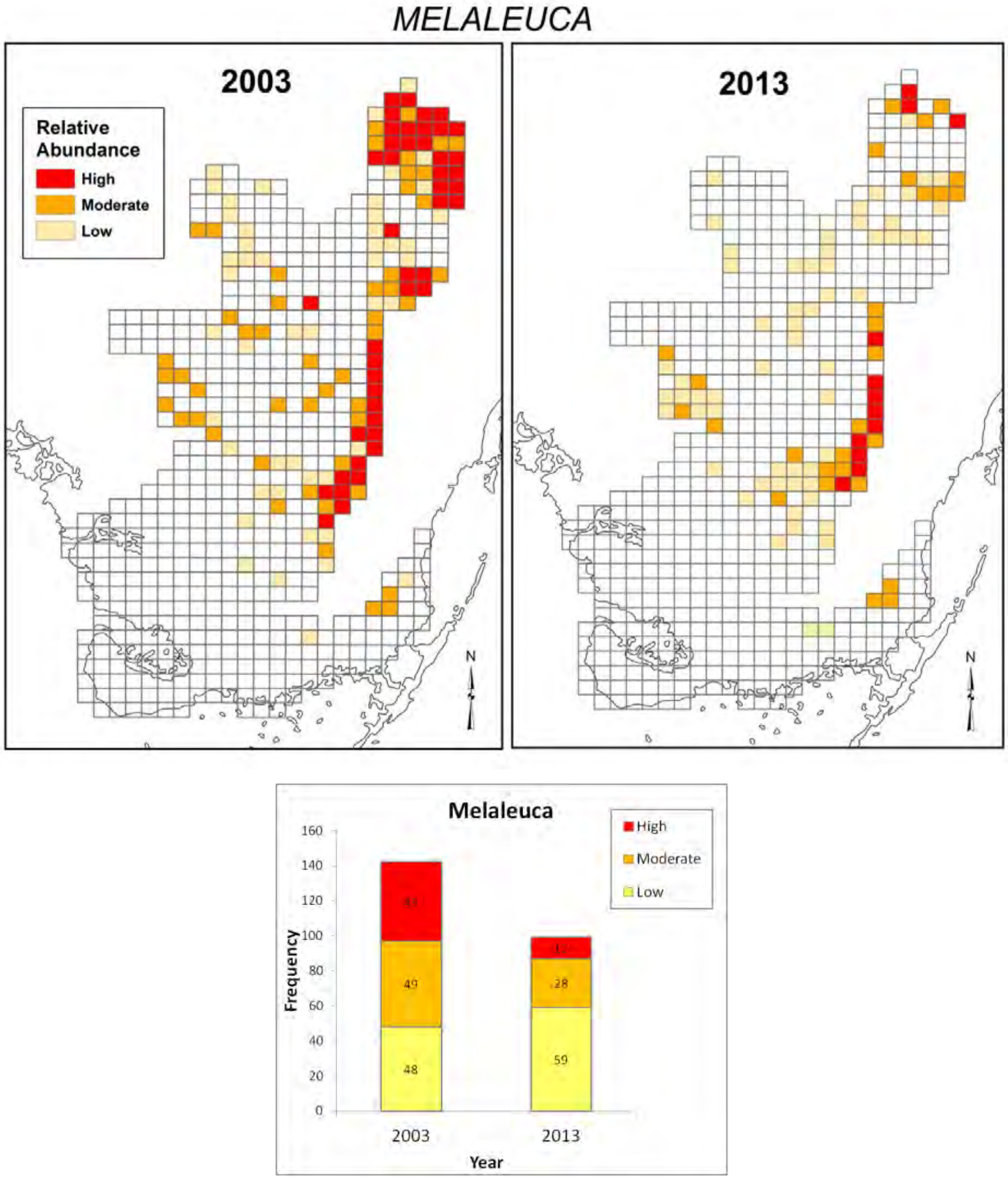

Figure A3-1-9. Distribution and relative abundance of melaleuca in the GE region 2003–2013. Zonal analysis (4-km grid) utilizes systematic reconnaissance flight data (2003) and digital aerial sketch mapping data (2012–2013). Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. The stacked bar chart shows the frequencies of 4-km grids with high, moderate, and low infestation levels in 2003 and 2013.

Old World Climbing Fern

Old World climbing fern is estimated to occupy 10,367 ha within the survey area (**Table A3-1-7**). Percent cover of Old World climbing fern ranged from 0.003% to 37.5% within 1-km cells (mean = 3.2%) (**Figure A3-1-10**). The majority of Old World climbing fern (~75%) occurs within WCA 1, where it aggressively forms dense mats over tree island canopies.

Distribution and abundance estimates for this invasive vine increased in the graminoid marshes of southwestern ENP between 2003 and 2013. Grid cells with high or moderate abundance increased 200% within the Ten Thousand Islands and Cape Sable regions of the region (**Figure A3-1-11**). Old World climbing fern was infrequently detected in eastern sections of the Everglades using digital aerial sketch mapping (DASM). However, ground-based observations of subcanopy infestations in WCA 3A and WCA 3B confirm that this invasive plant is widely scattered at low densities in these areas. Diligent monitoring and herbicide treatments within the WCA 3 tree islands are critical to avoid dense Old World climbing fern infestations similar to those in WCA 1.

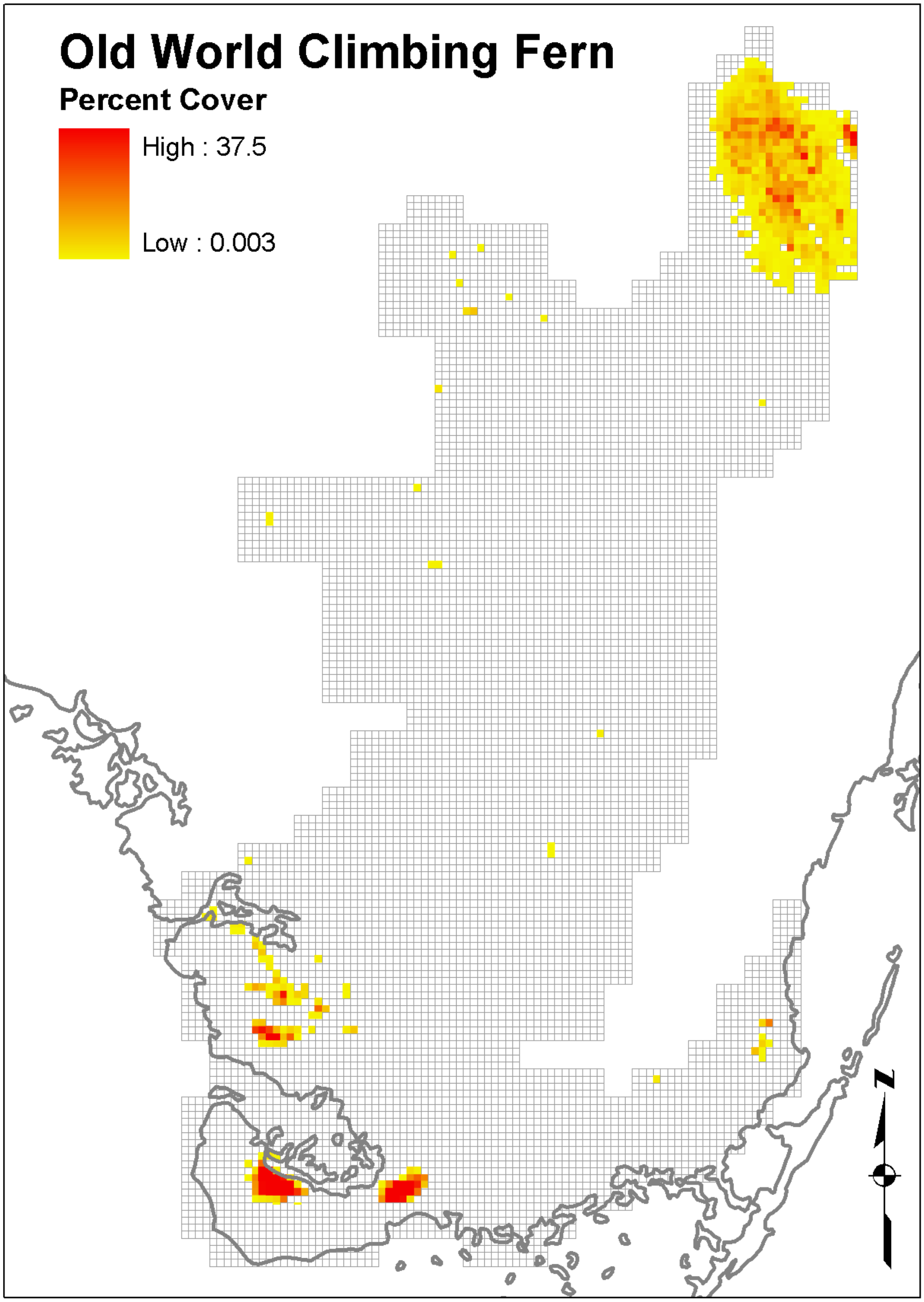

Figure A3-1-10. Distribution and relative abundance of Old World climbing fern in the GE region in 2013. Zonal analysis (1-km grid) utilizes digital aerial sketch mapping data collected in 2012 and 2013. Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. Values represent percent cover within 1-km grid cells (RANGE = 0.003 - 37.5%).

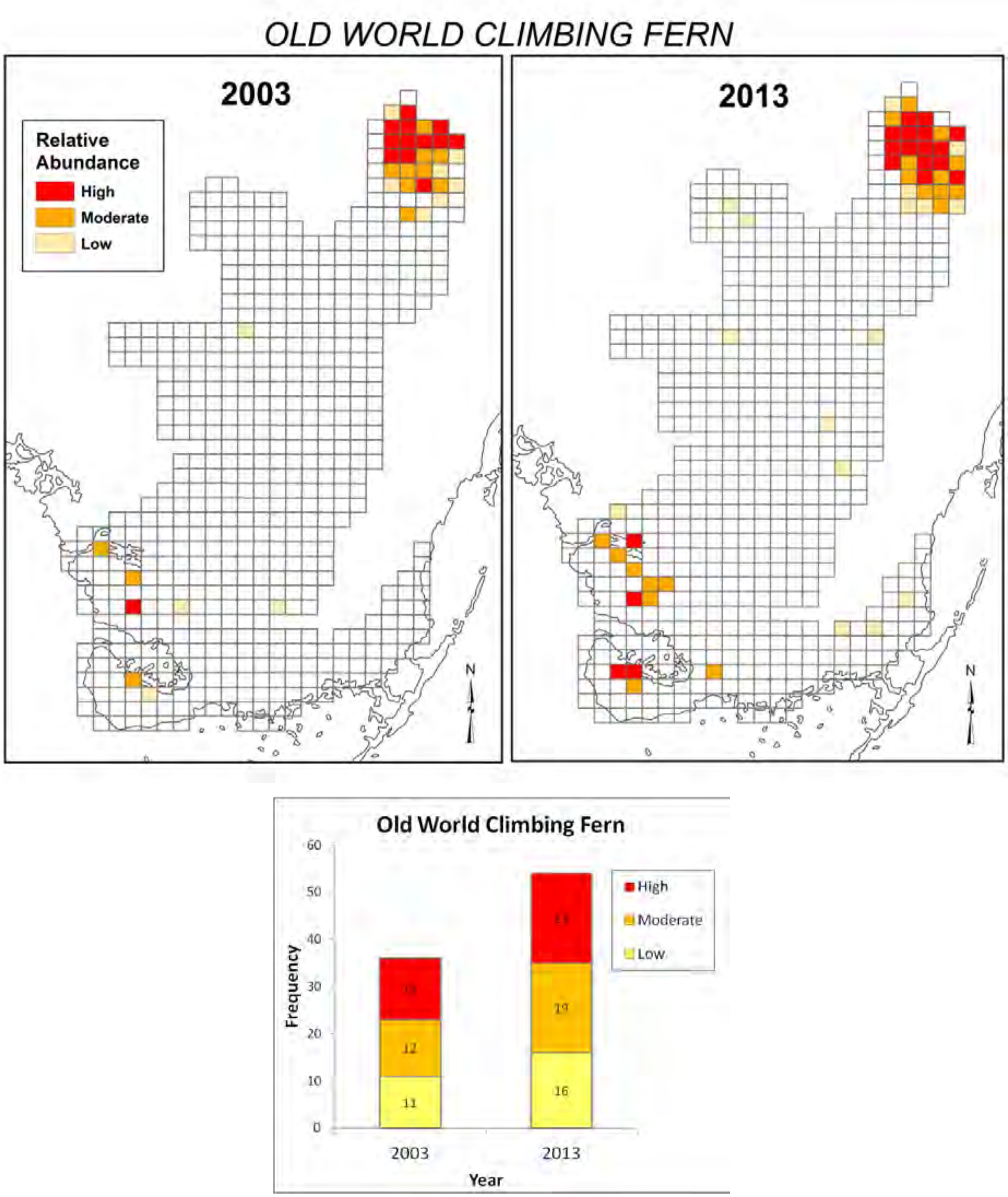

Figure A3-1-11. Distribution and relative abundance of Old World climbing fern in the GE region in 2003 and 2013. Zonal analysis (4-km grid) utilizes systematic reconnaissance flight data (2003) and digital aerial sketch mapping data (2013). Relative abundance classes are based on three equal intervals from visual abundance estimates during aerial surveys. The stacked bar chart shows the frequencies of 4-km grids with high, moderate, and low infestation levels in 2003 and 2013.

Distribution and Abundance of Priority Invasive Nonindigenous Animals

An Integrated Early Detection, Rapid Response, Management, and Monitoring Program for Everglades Invasive Reptiles and Amphibians

The Everglades Invasive Reptile and Amphibian Monitoring Program (EIRAMP) was initiated in 2010, to address concerns defined by the ECISMA EDRR Strategic Plan (ECISMA 2009). This inventory and monitoring program, designed to detect invasive animal species before they become established, helps to provide a foundation to meet the state and Department of Interior science needs for invasive wildlife management. It provides natural area managers with life history and location information to contribute to development of effective control methods for nonnative reptiles and amphibians that threaten ecosystem health. This program also involves surveying for native reptiles, amphibians, and mammals concurrently with surveys for other invasive species, which provides baseline data to determine impacts of exotic species on native fauna and ecosystems within state lands and other regional conservation lands.

Determining ecosystem impacts of invasive species is a major need, since it allows natural resource managers to determine priorities in a funding-limited situation. However, quantitative data on the impacts of invasive species is often lacking. Obstacles to determining impacts include a lack of suitable data on pre-invasion conditions and a lack of appropriate statistical models for estimating species co-occurrence and detection probabilities.

The objective of this project was to develop an EDRR, management, and monitoring program for invasive reptiles and amphibians and their impacts within the Everglades CISMA. The major objectives of this project were as follows:

- Determine status and spread of existing populations, and occurrence of new populations of invasive reptiles and amphibians
- Provide early detection, rapid response, control, and containment capability for removal of invasive reptiles and amphibians
- Perform body and data management for removed specimens
- Evaluate status and trends of populations of native reptiles, amphibians, and mammals
- Synthesize results in an adaptive framework to enhance removal of invasive species, and to determine impacts of invasive species on native wildlife assemblages
- Provide input into CERP and CEPP project planning and reports

Results from the first four tasks are summarized below for a five-month period (December 1, 2012 to April 30, 2013).

Methods

Standard surveys for invasive reptiles and amphibians were conducted throughout the Everglades region from December 1, 2012 to April 30, 2013. Standardized routes followed fixed transects along roads, levees, canals, and natural areas (**Figure A3-1-12**), and were constrained by distance or time. Tracks of all surveys were recorded using a global positioning system (GPS) device. Locations of invasive species were recorded using a GPS device and reported using the EDDMapS system. Native species of reptiles, amphibians, and mammals were sampled using the same techniques as for invasive species, except native species were not removed. A database was created for invasive reptiles and amphibians in south Florida.

EIRAMP provided a rapid response capability to sightings of priority nonnative wildlife species by ECISMA personnel and their cooperators. Trained and equipped staff responded rapidly and thoroughly to newly detected nonnative species or new locations for old species as needed. EIRAMP used live capture (e.g., tongs, hooks, or nets), live traps (e.g., box or noose), or lethal measures (e.g., firearms) as deemed appropriate. All lethal methods were in compliance with the 2007 American Veterinary Medical Association Guidelines on euthanasia.

EIRAMP provided systematic removal services for established species of invasive wildlife such as Argentine black and white tegus and Burmese pythons. Trained and equipped staff removed nonnative species in core and peripheral areas. The same methods as described under rapid response were used.

Results

EIRAMP participants encountered 483 amphibians, 1,137 reptiles, and 274 mammals during the surveys. Of these 152 were nonnative amphibians, 636 nonnative reptiles, and 52 nonnative mammals. EIRAMP also found four bullseye snakehead (*Channa marulius*), which are nonnative fish. A total of 76 species with three being nonnative amphibians, 11 nonnative reptiles, and seven nonnative mammals (**Table A3-1-9**) were encountered. The leopard frog (*Lithobates sphenocephalus*) was the most abundant native amphibian with a total of 101 observations, the squirrel treefrog (*Hyla squirrella*) was second most abundant native amphibian species with a total of 99 observations, and the green treefrog (*Hyla cinerea*) was third most abundant at 61 observations. Of the nonnative amphibians encountered, the Cuban treefrog was the most abundant with 117 observations, the greenhouse frog (*Eleutherodactylus planirostris*) was second with 24 observations, and the cane toad (*Rhinella marina*) was third most detected at 11 observations.

The Florida water snake (*Nerodia fasciata*) was the most abundant native reptile with 124 observations, the peninsular ribbon snake (*Thamnophis sauritus*) was second with 85 observations, and the Florida green water snake (*Nerodia floridana*) was third with 38 observations. The tropical house gecko (*Hemidactylus mabouia*) was the most abundant nonnative reptile with 308 observations, the brown anole (*Anolis sagrei*) was second with 232 observations and the Argentine black and white tegu was third with 36 observations.

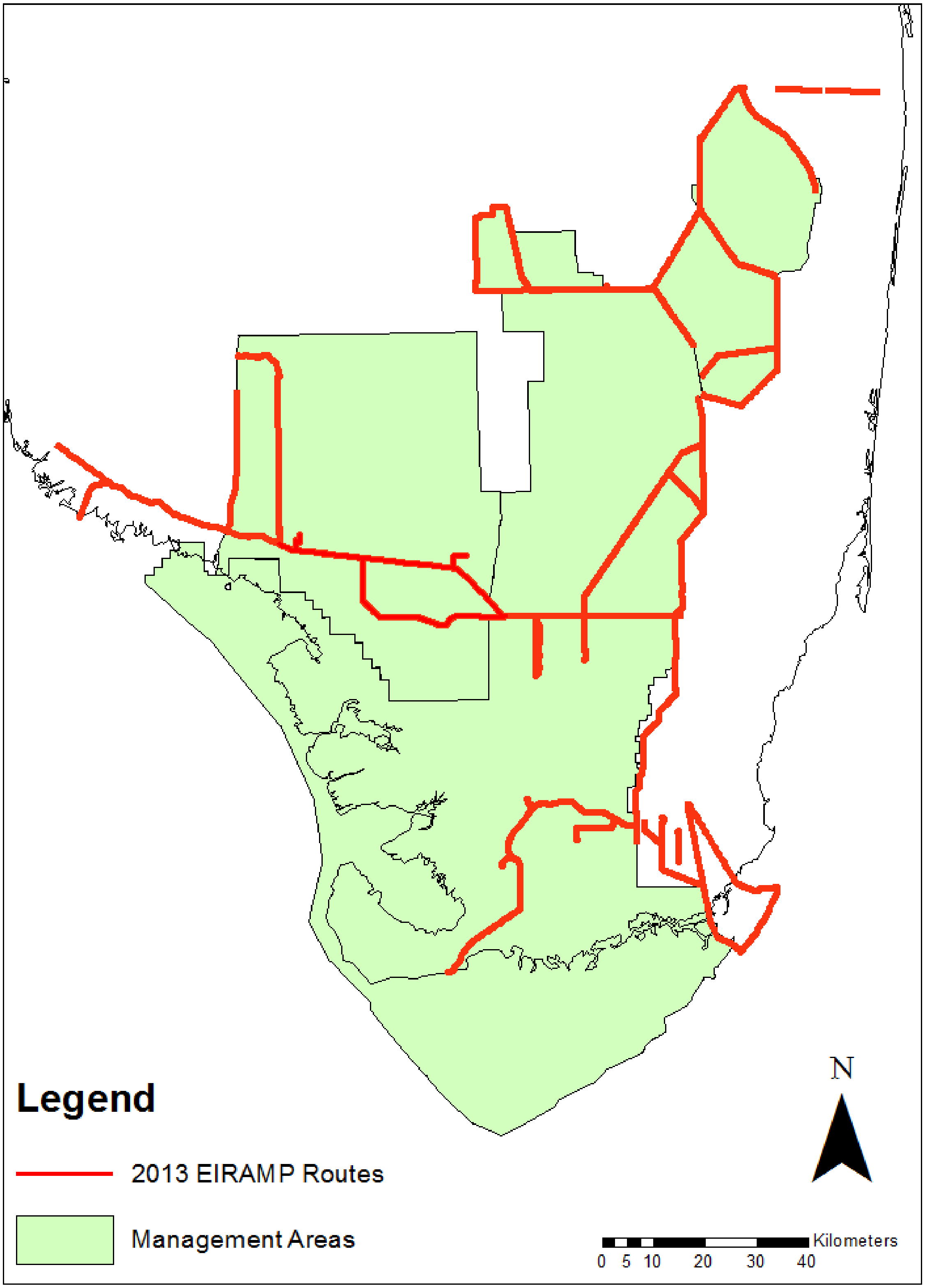

Figure A3-1-12. Map of locations of EIRAMP routes in south Florida.

Table A3-1-9. List of species detected in alphabetical order by common name (*introduced species).

Classification	Common Name	Scientific Name
Amphibian	Cricket Frog	*Acris gryllus*
	Cuban Treefrog	*Osteopilus septentrionalis**
	Green Treefrog	*Hyla cinerea*
	Greenhouse Frog	*Eleutherodactylus planirostris**
	Leopard Frog	*Lithobates sphenocephalus*
	Little Grass Frog	*Pseudacris ocularis*
	Marine/Cane Toad	*Rhinella marina**
	Oak Toad	*Anaxyrus quercicus*
	Pig Frog	*Lithobates grylio*
	Pinewoods Tree Frog	*Hyla femoralis*
	Eastern Narrowmouth Toad	*Gastrophryne carolinesis*
	Southern Toad	*Anaxyrus terrestris*
	Squirrel Treefrog	*Hyla squirella*
	Two Toed Amphiuma	*Amphiuma means*
Reptile	Argentine Black and White Tegu	*Tupinambis merianae**
	Basilisk Lizard	*Basiliscus vittatus**
	Black Spinytail Iguana	*Ctenosaura similis**
	Brahmini Blind Snake	*Ramphotyphlops braminus**
	Brown Anole	*Anolis sagrei**
	Brown Water Snake	*Nerodia taxispilota*
	Burmese Python	*Python* molurus bivittatus*
	Common House Gecko	*Hemidactylus frenatus**
	Corn Snake	*Pantherophis guttatus*
	Cuban Knight Anole	*Anolis equestris**
	Dusky Pigmy Rattlesnake	*Sistrurus miliarius*
	Eastern Diamondback Rattlesnake	*Crotalus adamanteus*
	Eastern Garter Snake	*Thamnophis sirtalis*
	Eastern Mud Snake	*Farancia abacura*
	Everglades Racer	*Coluber constrictor*
	Everglades Rat Snake	*Pantherophis alleghaniensis*
	Florida Box Turtle	*Terrapene carolina*
	Florida Brown Snake	*Storeria dekayi*
	Florida Cottonmouth	*Agkistrodon piscivorus*
	Florida Green Water Snake	*Nerodia floridana*
	Florida Kingsnake	*Lampropeltis getula*
	Florida Mud Turtle	*Kinosternon subrubrum*
	Florida Redbelly Turtle	*Pseudemys nelsoni*
	Florida Scarlet Snake	*Cemophora coccinea*
	Florida Softshell Turtle	*Apalone ferox*
	Florida Water Snake	*Nerodia fasciata*
	Green Anole	*Anolis carolinensis*
	Green Iguana	*Iguana iguana**
	House Gecko	*Hemidactylus mabouia**
	Indo-Pacific Gecko	*Hemidactylus garnotii**
	Mangrove Salt Marsh Snake	*Nerodia clarkii*
	Mediterranean Gecko	*Hemidactylus turcicus**
	Northern African Python	*Python sebae**
	Northern Curly-Tailed Lizard	*Leiocephalus carinatus**
	Peninsula Cooter	*Pseudemys floridana*
	Peninsula Ribbon Snake	*Thamnophis sauritus*
	Red-eared Slider	*Trachemys scripta*
	Rough Green Snake	*Opheodrys aestivus*

Table A3-1-9. Continued.

Classification	Common Name	Scientific Name
Reptile (continued)	Scarlet Kingsnake	*Lampropeltis triangulum*
	Southern Ringneck	*Diadophis punctatus*
	Spectacled Caiman	*Caiman crocodilus**
	Striped Crayfish Snake	*Regina alleni*
	Striped Mud Turtle	*Kinosternon baurii*
	Tokay Gecko	*Gekko gecko*
	Veiled Chameleon	*Chamaeleo calyptratus**
Mammal	Black Rat	*Rattus rattus**
	Bobcat	*Lynx rufus*
	Coyote	*Canis latrans*
	Domestic Cat	*Felis domesticus*
	Domestic Dog	*Canis familiaris*
	Florida Panther	*Puma concolor*
	Grey Fox	*Urocyon cinereoargenteus*
	Grey Squirrel	*Sciurus carolinensis*
	Hispid Cotton Rat	*Sigmodon hispidus*
	Horse	*Equus ferus*
	Marsh rabbit	*Sylvilagus palustris*
	Marsh Rice Rat	*Oryzomys palustris*
	Nine-banded Armadillo	*Dasypus novemcinctus**
	Otter	*Lontra canadensis*
	Raccoon	*Procyon lotor*
	Virginia Opposum	*Didelphis virginiana*
	White Tailed Deer	*Odocoileus virginianus*
	Wild Hog	*Sus scrofa**

The Virginia opossum (*Didelphis virginiana*) was the most observed native mammal with 60 sightings followed by the raccoon (*Procyon lotor*) with 52 sightings, and the white-tailed deer (*Odocoileus virginianus*) with 36 sightings. The domestic cat (*Felis catus*) was the most observed nonnative mammal with 20 sightings followed by the domestic dog (*Canis lupus familiaris*) with 14 and the nine-banded armadillo (*Dasypus novemcinctus*) with 3 sightings.

Rapid responses produced one northern African python (*Python sebae*), following passage of a cold front in early March. A new population of veiled chameleons (*Chamaeleo calyptratus*) was detected in Miami-Dade County near the entrance to ENP. Another population for which only one individual had previously been documented was confirmed nearby in a rural neighborhood and the two populations do not seem to be connected to each other. At least one more population near the L-31E is almost certainly extant; however, this has not been confirmed. Rumors of panther chameleon (*Chameleo pardalis*) and caiman lizard (*Dracaena guianensis*) have been investigated but thus far have not been confirmed.

EIRAMP participants removed 35 spectacled caiman (*Caiman crocodilus*) during rapid responses. Most came from the footprint of the BBCW Project area. The EIRAMP team responded to three rapid response calls from the 1-800-IVE-GOT1 hotline, and spent 46 person hours searching during those calls. This effort resulted in the capture of one spectacled caiman in the vicinity of Cutler Bay and one black-throated monitor (*Varanus albigularis*) in Lake Worth. The third response was in Florida City in regard to a Nile crocodile (*Crocodylus niloticus*) that had been in that area. However, the animal was not found.

Other species detected during opportunistic surveys were three Cuban knight anoles (*Anolis equestris*), four black spiny-tailed iguanas (*Ctenosaura similis*), six tokay geckos (*Gekko gecko*), eight green iguanas (*Iguana iguana*), one Florida kingsnake (*Lampropeltis getula floridana*), two Florida water snakes, two Florida green water snakes, one Cuban treefrog, three eastern ratsnakes (*Pantherophis alleghaniensis*), one raccoon, one eastern garter snake (*Thamnophis sirtalis*), and one red-eared slider (*Trachemys scripta elegans*).

A total of eight Nile monitors were observed and six were removed during four surveys of the C-51 Canal in Palm Beach County. Eight Burmese pythons were removed during systematic surveys of the C-110 Canal and the eastern boundary of ENP in southern Miami-Dade County. During trapping efforts in March and April, 12 Argentine black and white tegus were removed from Florida City. A total of 89 Oustalet's chameleons (*Furcifer oustaleti*) were removed out of 91 observed from December through April during systematic surveys in Florida City. December was the most successful month with 29 animals found. One adult male veiled chameleon was removed during one survey. Since that time, this population has been discovered, the location where these animals occur has been under heavy collection pressure from amateur reptile enthusiasts, thus making it difficult to find animals. Ten additional veiled chameleons were observed on opportunistic surveys but could not be removed due to their presence on various pieces of private property.

Discussion and Recommendations

This study represents the first comprehensive effort in south Florida to actively detect amphibian, mammal, and reptile invasive species in and near natural areas. A total of 21 nonnative species (three amphibians, 16 reptiles, and three mammals) were detected (**Table A3-1-9**). Established surveys were conducted 179 times over 21 sites. Oustalet's chameleon surveys were conducted five times from December through April in agricultural fields near Homestead, Florida and 89 of 91 individuals observed on these surveys were removed. Argentine black and white tegu trapping produced 305 trap days. Trapping and rapid response projects removed 12 Argentine black and white tegus around Homestead, Florida. Nile crocodile surveys produced an effort of 32 person hours during our reporting period. A total of eight Burmese pythons were removed from along the C-110 Canal and eastern ENP. Nile monitor surveys along the C-51 Canal removed six of eight observed individuals.

The tropical house gecko was the most observed nonnative species with 308 observations and was found in 14 of our 21 sites. Brown anoles were the second most observed species with 232 observations and were found in 16 of our 21 sites. Cuban treefrogs were observed 117 times and were found in 12 of our 21 sites. Argentine black and white tegus were observed 36 times; they were only observed along the C-110 Canal. Greenhouse frogs were observed 24 times at seven of 21 sites. The feral cat was observed 20 times at seven of 21 sites. Cane toads were observed 11 times at six of 21 sites. Domestic dogs were observed 14 times in three of 21 sites. Nine-banded armadillos were sighted three times in three of 21 sites.

Still, in most cases there is not the necessary information for detailed, specific evaluations of CERP and CEPP projects needed for management activities to control invasive species. However, partial removal of canals and levees could encourage spread or provide sites for colonization of tegus, monitors, pythons, and Cuban treefrogs. In this case, the most effective and lowest cost management option is early detection and rapid removal of invasive species during and post-project. To accomplish this, the University of Florida, FWC and SFWMD have begun collaboration on the EIRAMP described above. The purpose of the project is to develop a monitoring and management program for priority invasive reptiles and amphibians, and determine their impacts to south Florida. Relevant to CEPP, EIRAMP seeks to determine status and spread of existing populations and the occurrence of new populations of invasive reptiles and amphibians, and to provide additional early detection and rapid response capability for removal of invasive reptiles and amphibians.

Status of the Red Bay Ambrosia Beetle and Laurel Wilt disease in the Greater Everglades

Laurel wilt is a lethal disease of redbay and other members of the Laurel family, including swamp bay, an important species of Everglades tree island plant communities (Sklar and van der Valk 2002). The disease is caused by a fungus that is introduced into trees by the wood-boring redbay ambrosia beetle (FDACS 2011). The redbay ambrosia beetle is native to Asia, and was likely introduced into the United States via infested wood used for shipping crates (Harrington et al. 2011). Once infected, susceptible trees rapidly succumb to the pathogen and perish. Laurel wilt is causing up to 100% mortality of red bay in canopies of mixed forests in northern Florida (Shields et al. 2011). Since its arrival in 2002, the red bay ambrosia beetle has spread quickly throughout the southeastern United States. In March 2010, the redbay ambrosia beetle was found in Miami-Dade County in the Bird Drive Basin, less than five km from WCA 3B. Laurel wilt disease was subsequently confirmed on swamp bay trees in February 2011. Prior to this, Martin County was the southernmost Florida County where the disease was documented.

Methods

In March 2011, SFWMD, NPS, and Florida Department of Agriculture and Consumer Services utilized DASM to determine the spatial extent and abundance of laurel wilt in the eastern Everglades. The SFWMD and NPS conducted a second laurel wilt DASM survey in the central portion of the Everglades (WCA 3A, WCA 3B, tribal lands, BCNP, and ENP) between May and June 2013.

Results/Discussion

Figure A3-1-13 shows the estimated distribution and abundance of diseased swamp bay trees in the central region of GE in 2011 and 2013. The 2011 survey identified 105 symptomatic swamp bay trees scattered throughout the Bird Drive Basin, northward into the Pensucco Wetland area, and westward into ENP (**Figure A3-1-13**). Subsequent confirmation of laurel wilt in mapped trees confirmed that the disease was spreading into the Everglades. Laurel wilt was later found in the central portion of WCA 1 at four locations during the 2012 DASM invasive plant survey (data not shown). As of June 2013, the area of occupancy of laurel wilt within the central Everglades is 133,740 ha. Within this area, 332 tree islands contained symptomatic trees. The estimated percentage of affected canopy ranged from 0.25% to 50% (mean = 7.46%). The large majority of tree islands showed very low levels of tree canopy loss (<5% of tree island canopy affected), though a number of large tree islands experienced substantial loss of tree canopy (**Figure A3-1-14**).

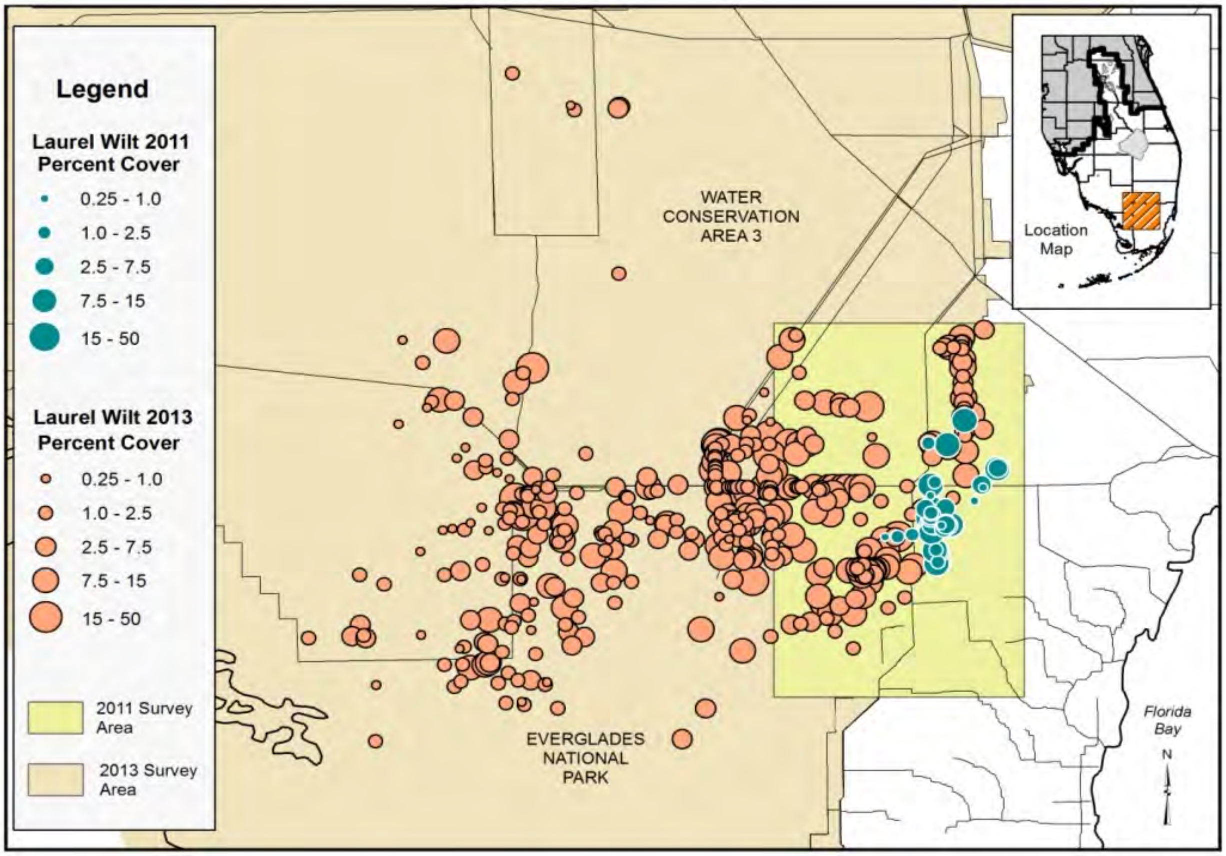

Figure A3-1-13. Distribution and abundance (as percent cover) of laurel wilt-infected swamp bays in the central Everglades in 2011 and 2013.

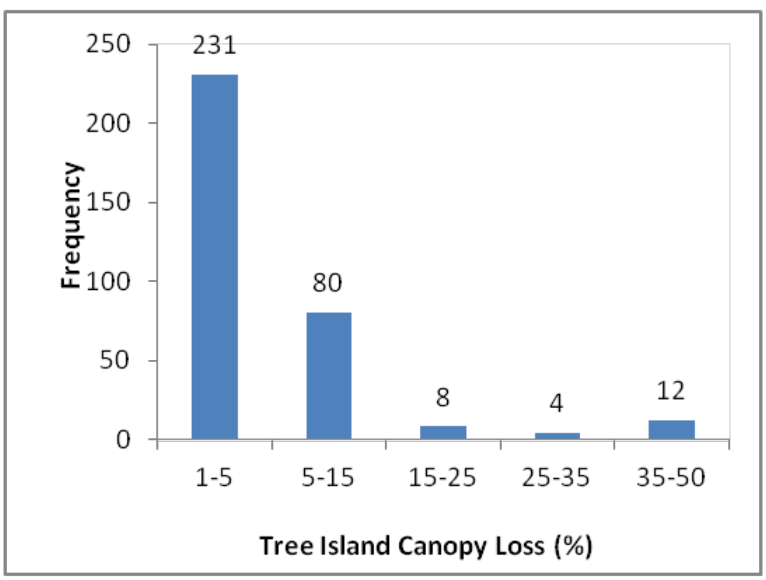

Figure A3-1-14. Frequency of tree island canopy loss estimates as of July 2013.

Tree island vegetation data in WCA 3A, WCA 3B and ENP indicates that swamp bay is relatively common in tree islands to the north and south of the current laurel wilt distribution (Engel et al. 2009). Additionally, ground truthing of tree islands with laurel wilt-infected trees confirmed that many swamp bays remain nonsymptomatic. These observations suggest that the spatial extent of laurel wilt will continue to expand in the Everglades and swamp bay loss within the current area of occupancy is likely to continue. The short- and long-term impacts of canopy loss in tree islands due to laurel wilt are not known. Tree islands with moderate to high canopy loss may become more vulnerable to invasion by Old World climbing fern and Brazilian pepper. SFWMD scientists and land managers are developing a management strategy for the most severely impacted tree islands. The strategy includes aerial- and ground-based reconnaissance to detect early expansions of these invasive plants. This information will be used to direct rapid response efforts by invasive plant control teams.

Priority Animals for Containment or Eradication

Invasive reptiles and amphibians have increasingly become a management issue in and around natural areas of the Everglades footprint. Due to lack of dedicated resources and successful control tools, land managers have largely had to rely on volunteer efforts and coordinating limited resources through ECISMA to deal with newly discovered populations on and adjacent to the Everglades. In recent years, funding has become available for EDRR monitoring programs, such as EIRAMP, and FWC has created more personnel positions in south Florida. These new resources have allowed managers to obtain a better assessment of the distribution and abundance of several priority species, some of which may be capable of being eradicated. Additional resources will be needed to continue pressure on priority populations and allow for successful eradication or containment.

Because of the urgency of the situation and limited resources, the overall strategy has been to remove nonnative species as they are encountered while conducting preliminary assessments to determine overall population establishment and geographic extent. Removal is either done during surveys or by live trapping. Specimens are collected for diet analysis, reproductive status, and in some cases released and tracked using radio telemetry, in order to assess the possible impacts on the Everglades ecosystem. In addition to conducting regular and opportunistic surveys, agencies have developed reporting mechanisms for the public and other agency personnel. Reports are regularly monitored to direct rapid removal efforts and possibly learn about new populations. The public can report nonnative species via the 1-888-IVE-GOT1 hotline and the EDDMapS online reporting website or application. Providing effective outreach is an important part of EDRR and containment that is still being conducted on a largely volunteer basis.

Nile Monitor

Nile monitors are large African lizards capable of exceeding a total length of seven feet. They eat a variety of invertebrates, vertebrates, and eggs depending on what they are able to find while hunting and scavenging. Due to their generalist diet and adaptability to canal banks, and both fresh and saltwater marshes, Nile monitors could negatively impact a variety of listed native species in the Everglades either through competition or predation (Engeman et al. 2011).

Nile monitors have been widely popular in the pet trade but are now classified as Conditional Species in Florida, which prohibits their being obtained as personal pets. At least three populations of Nile monitors exist in Florida and a fourth possible population is being investigated. Nile monitors have been reported in Miami-Dade County near the Reserve Base Homestead Air Reserve Base for more than 20 years. USDA Wildlife Services staff working on the base control monitors opportunistically, but the animals are very wary of humans. A population of Nile monitors located in Palm Beach County was reported to FWC in 2010 and subsequent surveys confirmed monitors along the C-51 Canal from Interstate 95 to within a few miles of Stormwater Treatment Area 1 East. In cooperation with SFWMD, FWC began a trapping and surveying program. Trapping has been ineffective but surveying and removing with firearms has been more successful. A University of Florida EIRAMP route was established in the most heavily populated area to both monitor and control the population. A number of confirmed monitor sightings in Broward County has prompted monitoring but it is not yet known whether these sightings represent a population or continued sightings of a few individuals. **Table A3-1-10** shows verified Nile monitor reports per year for Miami-Dade and Palm Beach counties.

Table A3-1-10. Verified Nile monitor reports per year by county.

Year	Miami-Dade County	Palm Beach County
2010	11	0
2011	3	22
2012	6	26
2013	3	7

Spectacled Caiman

Spectacled caiman are found over a wide area of Central and South America and have been reported in Broward and Miami-Dade counties since the 1960s. While the spectacled caiman is smaller than Florida's native crocodilians, caimans may compete with them for food and habitats (http://www.myfwc.com/wildlifehabitats/nonnatives/reptiles/spectacled-caiman/). Univeristy of Florida and FWC have been investigating multiple isolated reports of caiman populations in 2013 and opportunistically removing them. While spectacled caiman sightings have occurred within the GE region, sightings have not been consistent and it is not known if caimans are currently reproducing. A large number of hatchlings were removed but sightings are inconsistent. University of Florida will continue to conduct opportunistic surveys for this species in Miami-Dade and Broward counties.

Northern African Python

The northern African python is a large constrictor snake native to sub-Saharan Africa reported to reach a maximum total length of 20 feet. In their native range, they are found in a variety of habitat types and have a generalist diet, taking prey opportunistically (Reed and Rodda 2009). Their adaptability and large adult size make the species a high priority for management.

Northern African pythons were not particularly popular in the pet trade but there is evidence of a breeding population in an area of western Miami-Dade County bordering ENP and several parcels of state-owned Everglades habitat (Bird Drive Basin). ECISMA cooperators have been conducting surveys in the area since 2010 to remove individuals from the population and determine the range. Surveys are organized on a monthly basis using volunteers from a number of agencies and entities to cover as many established routes as possible. Surveys are conducted most frequently during winter and spring months when pythons are most active during the day but some night routes are also surveyed during warmer months. Northern African pythons are listed as Conditional Species in Florida, meaning they can no longer be acquired for personal use. In 2012, this python species was listed as an Injurious Species under the Lacey Act prohibiting their importation into the United States and insular territories, and movement between states. **Table A-3-11** shows the number of north African pythons removed and observed by year.

Table A3-1-11. Number of northern African pythons removed and observed by year.

Year	Removed	Observed
2010	14	4
2011	3	0
2012	1	0
2013	5	1

(Source: FWC Exotic Species Database, accessed September 18, 2013)

Chameleons

Breeding populations of at least two chameleon species are known in Miami-Dade County immediately adjacent to the GE region (Gillette et al. 2010). Oustalet’s chameleons are the second largest chameleon species in the world and are native to Madagascar. The native range of veiled chameleons extends across an area of the Middle East and they also grow relatively large. Both species primarily eat insects but large adults are capable of eating small vertebrates.

ECISMA cooperators began conducting monthly surveys for Oustalet’s chameleons in 2011 and wildlife biologists from the University of Florida began conducting surveys for veiled chameleons in 2013, discovering the first Miami-Dade County breeding population. Chameleon breeding populations do not seem to be able to expand rapidly on their own but instead are rumored to be spread by humans. There is potential for known populations to be eradicated but by all indications it will take considerable effort.

Status of Nonindigenous Fish Species in the Greater Everglades

It is becoming widely recognized that the full scope of benefits envisioned by CERP may not be attained without management of nonindigenous species. Restoration actions may promote the spread of nonindigenous species if project designs do not take their management into consideration. As a result, CGM 062.00 “establishes the processes and procedures, and provides guidance… on how to consider and incorporate invasive and native nuisance species management into CERP projects.” Identifying ways to deliver water without delivering nonindigenous fish species to natural areas is among the biggest invasive species management challenges for Everglades restoration, along with successful control of these species.

Nonindigenous fishes were first reported in the waters of south Florida in the 1950s (Courtenay et al. 1974, Shafland 1986). By 2007, 34 nonindigenous fish species maintained reproducing populations (Shafland et al. 2008), although roughly three times as many species have been recorded from Florida waters (110 as of 2012; USGS 2012). Seven nonindigenous fishes had been reported commonly throughout the GE region (Trexler et al. 2000) and nine species had been documented from ENP by 2000 (Loftus 2000). Since 2000, eight additional nonindigenous species have been observed in ENP (Kline et al. 2013).

The widespread prevalence of canals on the south Florida landscape has been considered a primary factor for the survival and spread of nonindigenous fishes in south Florida. The unnaturally deep waters of canals provide tropical nonindigenous species a refuge from occasional cold winter temperatures, permanent waters during dry seasons, and corridors for dispersal throughout south Florida that has been implicated in species range expansions (Courtenay and Robins 1975, Shafland and Pestrak 1982, Loftus 1988, Trexler et al. 2000, Harvey et al. 2010, Gandy et al. 2012). Various CERP projects that propose to alter water deliveries from canals to hydrate Everglades marshes may also pose a risk of spreading nonindigenous fishes.

Relative abundance and changes in the number of nonindigenous fishes are indicators used to assess ecological condition of Everglades marshes. The *System-wide Ecological Indicators for Everglades Restoration* (Brandt et al. 2012) reports the relative abundance of nonindigenous fishes from long-term quantitative monitoring efforts in Everglades slough regions. A relative abundance of 0% nonindigenous fishes is considered ideal, and a desired target of the performance measure is a fish community composed of less than 2% nonindigenous species by density. The management directives for protected natural areas (e.g., ENP), emphasize native species and natural conditions and have a goal to minimize the risk of spreading invasive species (Executive Order 13112). The status of nonindigenous fishes throughout the GE region is discussed here by summarizing results from monitoring efforts and recent studies.

Methods

Aquatic fauna have been sampled annually across the GE region as part of the CERP Monitoring and Assessment Plan (MAP) from 2005 to 2012. Three replicate throw trap (1 square meter [m^2]) samples collected small fish (standard length < 8 cm) from wet prairie habitats dominated by *Eleocharis* spp. or other plant species where vegetation density does not limit sampler use. Samples are collected during the late wet season (September–December). Approximately 150 sites were sampled annually over the eight years of the study. The mean density (number per m^2) and relative abundance of native and nonindigenous fishes was estimated over the period of the study. In addition to the CERP MAP study, Kline et al. (2013) documented the presence and relative abundance of nonindigenous fishes observed in ENP from multiple studies from 1997 to 2012. They compared their results to data collected earlier and reported by Trexler et al. (2000). The results of these studies and other recent studies are discussed.

Results and Discussion

Nonindigenous fishes were collected across the entire GE region by the CERP MAP project; however, the density was typically much lower than that of native species. The density of native fishes was generally higher in the WCAs (typically between 10 and 40 fish per m^2) than in ENP (typically between 0.01 and 20 fish per m^2; **Figure A3-1-15**). Within ENP, native fish density was highest through the center of Shark River Slough (**Figure A3-1-15**). At all but two sites, nonindigenous fish density averaged < 0.75 fish per m^2 (**Figure A3-1-16**) and was approximately 0.5% of the relative abundance of all fish caught throughout the GE region (**Table A3-1-12**). In areas of southern ENP and southeastern BCNP, nonindigenous fishes were > 6% of the total number of fish caught at 12 sites, and > 15% at four of the study sites (**Figure A3-1-17**), but represented approximately 1.7% of all fish collected within the ENP and southern BCNP region, the region with the highest relative abundance of nonindigenous fishes (**Table A3-1-12**).

While the relative abundance of nonindigenous species was less than 2% in all regions sampled by the CERP MAP project for the entire period of study, recent studies suggest nonindigenous fishes may pose a risk of impact in areas where they attain a high relative abundance. Areas of the Rocky Glades, the short hydroperiod marshes east of Shark River Slough, and the mangrove fringe of southern ENP support a higher relative abundance of nonindigenous species than do the wet prairie and slough habitats (Trexler et al. 2000, Kline et al. 2013). Within the Rocky Glades, a high frequency of dry downs makes the area a refuge for native species (Kobza et al. 2004, Rehage et al. 2013) and nonindigenous species can dominate the dry season refuge habitats at the expense of native species. When Mayan cichlids (*Cichlasoma urophthalmus*) reach high abundance in the mangrove fringe of southern ENP, the native fish assemblage composition differed and there was an inverse correlation between Mayan cichlid abundance and the abundance of some small native fish species (Harrison et al. 2013). These are among the few studies designed to directly test for potential impacts of nonindigenous species and more studies are needed to determine if or how these results translate throughout the Everglades food web.

Table A3-1-12. Number of throw trap (1-m^2) samples, total catch, catches of nonindigenous fishes, and relative abundance of nonindigenous fishes (%) reported by region sampled during the CERP MAP aquatic fauna project 2005–2012. Column headings are for African jewelfish (AJ), black acara (BA), brown hoplo (BH), blue tilapia (BT), mayan cichlid (MC), pike killifish (PK), spotted tilapia (ST), walking catfish (WC), and unidentified cichlid juvenile (UC).

Region	N	Study Period	Total Fishes	Nonnative Species									Nonindigenous (%)
				AJ	BA	BH	BT	MC	PK	ST	WC	UC	
Lake Okeechobee	72	2005–2010	4,707	0	0	0	0	0	0	0	0	0	0
Pal-Mar	144	2005–2011	1,288	0	0	1	0	0	0	0	0	0	0.1
WCA 1	360	2005–2012	7,911	0	1	1	0	3	0	1	0	0	0.1
WCA 2A	192	2005–2012	2,710	0	1	0	0	5	0	0	0	0	0.2
WCA 2B	45	2005–2012	1,196	0	0	0	0	1	0	0	0	0	0.1
Holey Land Wildlife Management Area	12	2005–2007	83	0	0	1	0	0	0	0	0	0	1.2
WCA 3A	1,008	2005–2012	17,754	2	4	2	0	29	1	1	1	0	0.2
WCA 3B	261	2005–2012	3,811	0	1	0	0	5	0	0	0	0	0.2
Pennsuco Wetlands	15	2005–2007	154	0	0	0	0	0	0	0	0	0	0
ENP and southeastern BCNP	1,294	2005–2012	9,583	47	8	1	1	68	30	3	0	7	1.7
All regions sampled	**3,403**	**2005–2012**	**49,197**	**49**	**15**	**6**	**1**	**111**	**31**	**5**	**1**	**7**	**0.5**

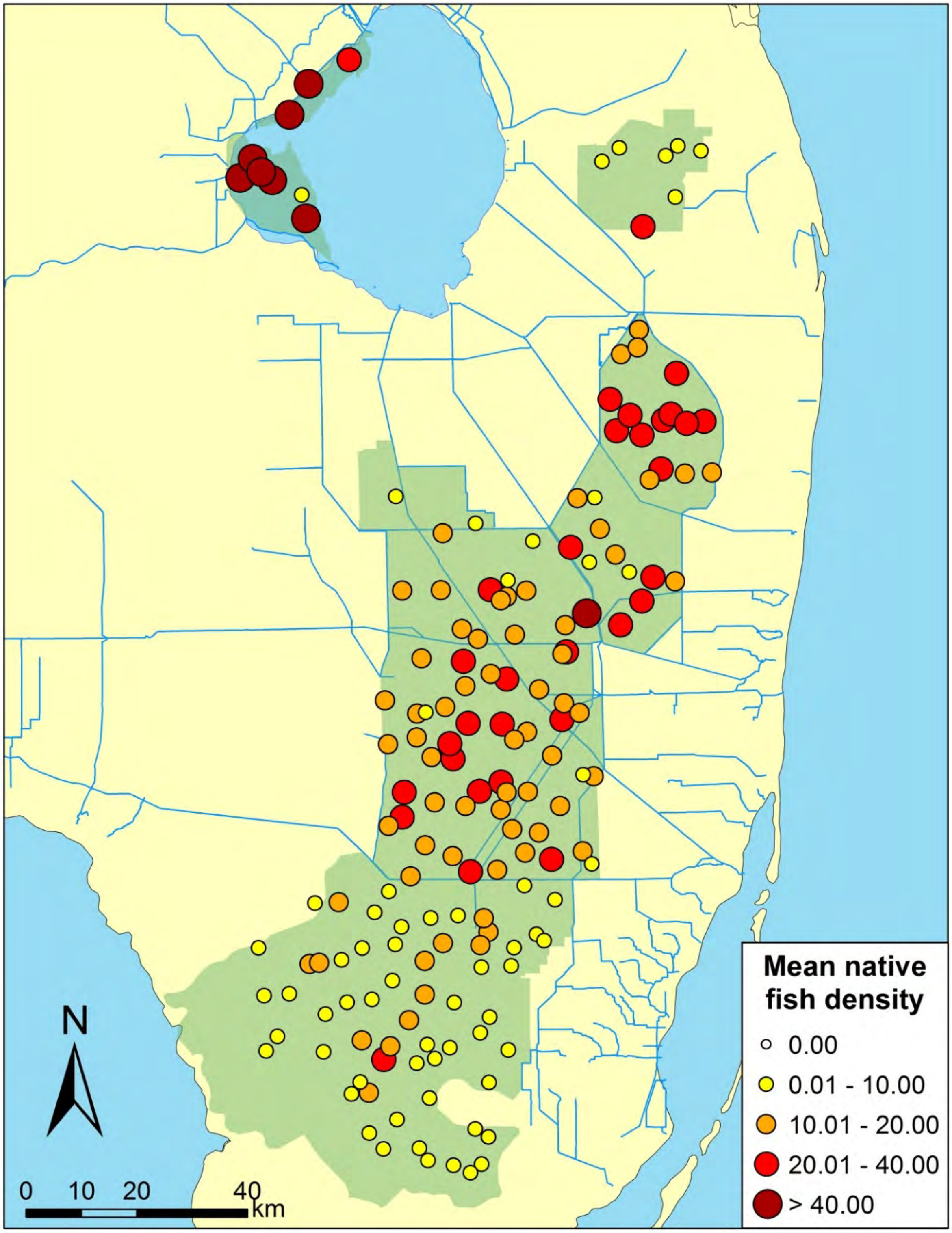

Figure A3-1-15. Mean density (fishes per m^2) of native fishes captured in throw traps as part of CERP MAP monitoring during the late wet season 2005–2012.

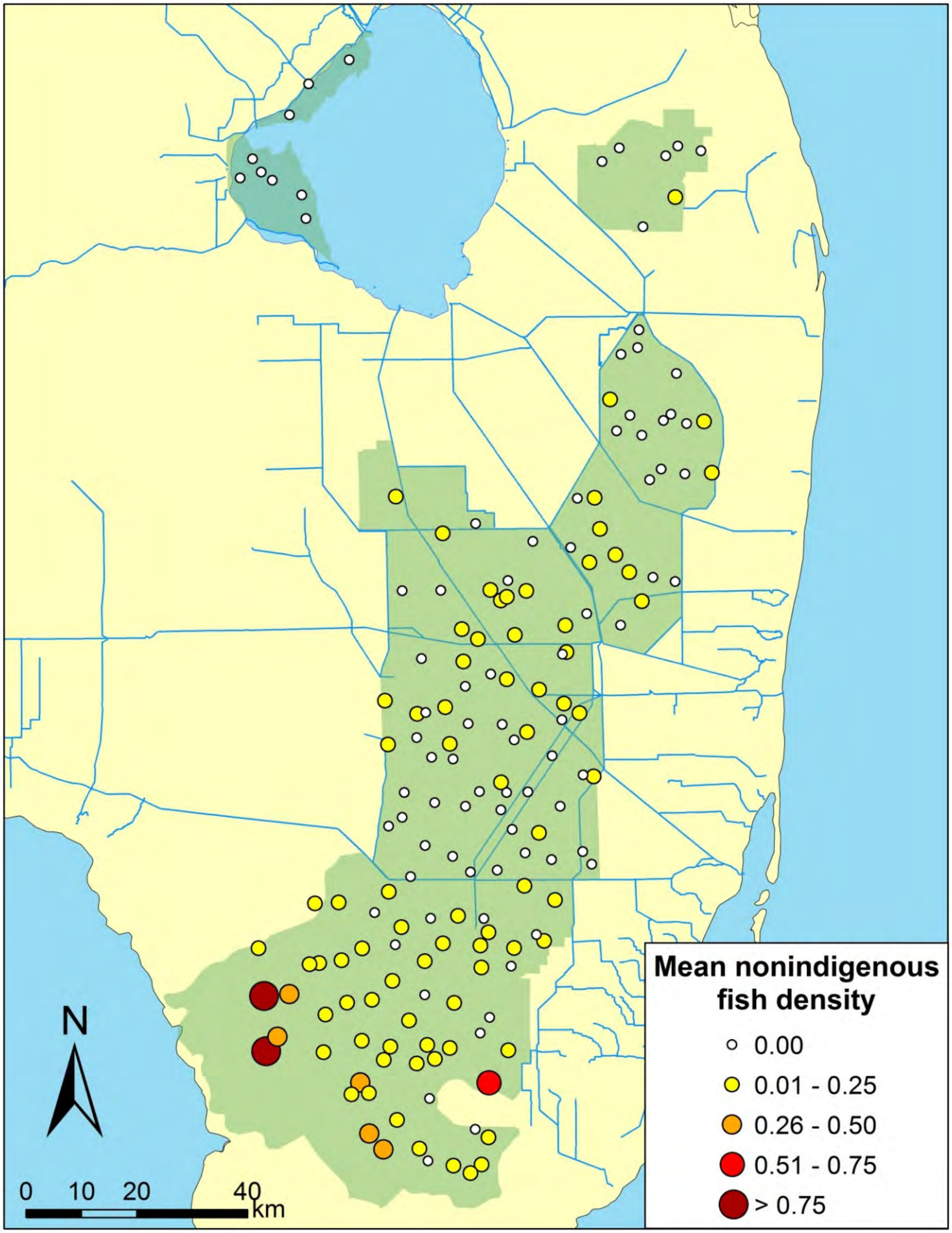

Figure A3-1-16. Mean density (fish per m^2) of nonindigenous fishes captured in throw traps as part of CERP MAP monitoring during the late wet season 2005–2012.

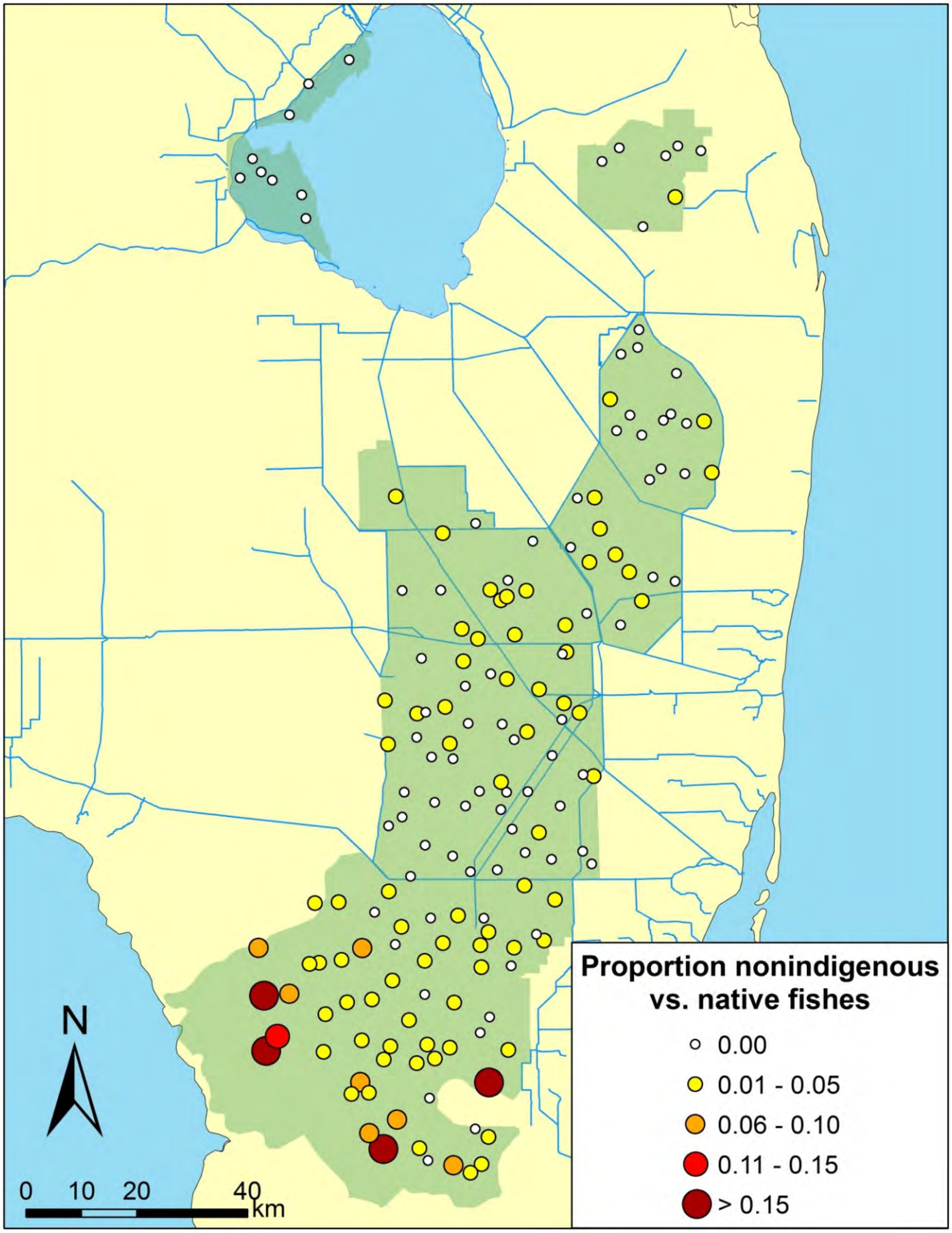

Figure A3-1-17. Proportion of nonindigenous fishes versus native fishes captured in throw traps as part of CERP MAP monitoring during the late wet season 2005–2012.

Despite the low relative abundance of nonindigenous species averaged across the GE region, there has been an increase in the number of species observed. Eight nonindigenous species were throughout the GE region by the CERP MAP project (**Table A3-1-12**). ENP/BCNP and WCA 3A had the highest number of nonindigenous species collected (7 species each) of all the regions (**Table A3-1-12**). The Mayan cichlid, black acara (*Cichlasoma bimaculatum*), and brown hoplo (*Hoplosternum littorale*) were species collected in the most regions (six, five, and five regions, respectively), and Mayan cichlid, African jewelfish (*Hemichromis letourneuxi*), and pike killifish (*Belonesox belizanus*) were caught in the highest numbers with most individuals caught in the ENP and southeastern BCNP region (**Table A3-1-12**). In ENP, the increase of eight nonindigenous fishes after 2000 brought the total number observed to 17 species (Kline et al. 2013). Over the history of ENP, increases in the number of nonindigenous species appeared to correspond with both the increase in nonindigenous fishes reproducing statewide in Florida and with water management changes that altered flows from canals to the marshes (Kline et al. 2013). Although not all of the new species have become widespread or attained a high relative abundance, several species have been collected over a large area, are expanding their range, or have attained a high relative abundance in some areas of ENP (Kline et al. 2013).

At present, there is at least twice the number of nonindigenous species reproducing within the south Florida canal system than have been found in the marshes of the GE region. Unlike invasive plant management activities, where established removal protocols exist for certain species, management of nonindigenous fishes once established in natural marshes is problematic. With no way to predict the impacts of new species, preventing future introductions may be the key to management of nonindigenous fishes. Options and technologies that have the potential to reduce the risk of spreading nonindigenous aquatic species (GLMRIS 2012) should be explored for applicability to Everglades restoration. Management actions that focus on the vulnerability of tropical nonindigenous fishes to periodic cold winter temperatures could be a particularly feasible approach. Removing thermal refuges by modifying or restoring unnatural deep water habitats such as canals, borrow ponds, and ditches to allow natural winter cooling has been suggested (Schofield et al. 2009, Kline et al. 2013). Actions designed to limit the spread of or control nonindigenous fishes prior to their dispersal into natural marshes may be the most effective way to maintain a low relative abundance and limited number of nonindigenous fish species in protected Everglades wetlands.

Key Findings and Recommendations

The species and results discussed above still represent only a fraction of the nonnative and invasive species that occur within the GE region, and these should be considered “case studies.” Over time, focal species and efforts to address them should be expected to change in response to changing perceptions of threat to ecosystem function, as well as identification of new and significant threats. The key findings and recommendations are as follows:

- Melaleuca and Australian pine have decreased in abundance across most of the GE region in the last decade. At the regional level, Brazilian pepper has decreased in abundance in the last decade, but has increases in abundance at local scales (e.g., Cape Sable mangrove fringe). Old World climbing fern has increased in abundance and distribution over the same time period. Heavy infestations now occur in tree islands of WCA 1 and cordgrass marshes of southwestern ENP.

- Regionwide surveys in 2012–2013 detected 21 nonindigenous animal species (3 amphibian, 11 reptile, and 7 mammal) in and near natural areas of the GE region.
- Since 2010, 78 Nile monitors have been removed from Miami-Dade and Palm Beach counties, although no monitors have been removed from Everglades natural areas to date.
- Numbers of Argentine black and white tegus removed from natural areas within and adjacent to the GE region continues to rise each year.
- Laurel wilt disease occupies an area of roughly 133,000 ha in the region. Swamp bay mortality is highly variable in tree islands resulting in 0.25% to 50% canopy loss on affected tree islands. Most tree islands currently exhibit less than 5% canopy loss, although tree demography data suggests potential increases in swamp bay mortality with time.
- Density of nonindigenous fishes is typically much lower than that of native fishes throughout the region. While the relative abundance of nonindigenous species was < 2 % in all regions sampled by the CERP MAP project of the entire period of study, recent studies suggest nonindigenous fishes may pose a risk of impact in areas where they attain a high relative abundance.

2013 Stoplight Indicator

- Maintenance control achieved for melaleuca, Australian pine and Brazilian pepper in some portions of the region.
- Recent control efforts in WCA 1 achieving significant reductions of melaleuca.
- Systematic monitoring program in place.
- Aggressive spread of Old World climbing fern and Brazilian pepper threatening integrity of Everglades tree islands and other habitats.
- Still several other species present (e.g., shoebutton Ardisia [*Ardisia elliptica*]) with little or no control effort or efficacy.

Lake Okeechobee Region

Introduction

More than 80 nonnative plant and 100 nonnative animal species have been identified within the LO region (Ferriter et al. 2008). The *Lake Okeechobee Protection Program Exotic Species Plan* (SFWMD et al. 2003) identifies the primary species that threaten native plants, animals and the ecological health of the lake. Since the development of the plan, new plant and animal species have invaded Lake Okeechobee and are serious threats to the lake's ecology and therefore are included in **Table A3-1-13**. The plant species listed in the table are managed in accordance with concurrent funding by the responsible agencies. Management for the majority of the animal species is limited to monitoring. Feral pig (*Sus scrofa*) removal is targeted when there are impacts to man-made structures; however management of animal species is limited due to lack of funding and information regarding impacts and effective control measures.

Table A3-1-13. Primary nonnative species and potential impacts in the LO region.

Plant Species	Potential Impacts
melaleuca (*Melaleuca quinquenervia*)	Displaces native species, reduces wildlife habitat value, alters hydrology, modifies soil resources and changes fire regimes.
torpedograss (*Panicum repens*)	Displaces native plant species and eliminates productive habitat for fish and other wildlife.
Brazilian pepper (*Schinus terebinthifolius*)	Quickly colonizes disturbed sites, displaces native vegetation, suppresses native plant growth (allelopathic agents) and impacts human health.
hydrilla (*Hydrilla verticillata*)	Displaces submersed native plants, alters fish populations, shifts zooplankton communities and affects water chemistry.
water hyacinth (*Eichornia crassipes*)	Displaces and uproots native plant communities, interferes with water movement and degrades water quality.
water lettuce (*Pistia stratiotes*)	Impacts submersed plant communities and interferes with water movement.
Animal Species	**Potential Impacts**
island apple snail *(Pomacea maculata*)	Destroys native aquatic vegetation, competes with native aquatic fauna, alters water quality and hybridizes with other species.
feral pig (*Sus scrofa*)	Forages on ground-nesting animals, consumes eggs of birds and reptiles, creates disturbance and destroys native plant communities through rooting/trampling.
blue tilapia (*Oreochromis aureus*)	Competes/impedes successful native fish spawns, destroys submersed vegetation through feeding/burrowing and alters plant communities.
spiny water flea (*Daphnia lumholtzii*)	Potential to impact larval fish and water quality.
Asiatic clam (*Corbicula fluminea*)	Outcompetes/displaces native mollusks and potential to change substrate and eliminate burrowing insect populations.
sailfin catfish (*Pterygoplichtys ultiradiatus*)	Competes/impedes successful native fish spawns, destroys submersed vegetation through feeding/burrowing and alters plant communities.

Distribution and Abundance of Melaleuca 1993–2012

By the early 1990s, roughly one century after its initial introduction to Florida, melaleuca had spread over hundreds of thousands of hectares in southern Florida. The western marsh of Lake Okeechobee proved to be excellent habitat for melaleuca and large portions of the marsh were converted to dense monotypic melaleuca forests (Turner et al. 1998). What was considered to be an insurmountable invasive species problem, melaleuca is now successfully managed on Lake Okeechobee through sustained interagency commitment and integrative management strategies (Laroche 1998). Regional monitoring was an important component of the melaleuca management effort, since it facilitated a regional containment strategy actively restricting the continued spread of the plant while systematically controlling existing populations. This section provides a brief synopsis of the results of melaleuca monitoring on Lake Okeechobee between 1993 and 2012.

Methods

Data presented in this section were collected between 1993 and 2012 as part of a regional aerial reconnaissance program led by SFWMD and NPS. Biologists in low-flying aircraft (~150-meter altitude) made visual estimates of melaleuca abundance along fixed east-west transects in 1993, 1997, and 2001. The relative density of melaleuca was estimated at each sampling location along the transects. A follow-up aerial survey was conducted in 2012 using digital aerial sketch mapping techniques. This method also utilized transects and visual estimates of melaleuca abundance. Detailed descriptions of survey methods are provided by Pernas and Ferriter (2008) and Rodgers et al. (2011). Zonal analysis of these data was conducted on a 2-km grid system. This approach facilitates landscape-level assessments of the distribution and relative abundance of melaleuca over time. The intensity of infestation within each cell was calculated based on the density-weighted sum of melaleuca observations within each cell.

Results and Discussion

Results of the aerial survey program show a dramatic decrease in melaleuca on the Lake Okeechobee marsh between 1993 and 2012 (**Figure A3-1-18**). The frequency of melaleuca occupancy in grid cells decreased from 31 to 0.1 percent over this nine-year period. The changes in distribution over time reflect the SFWMD's sustained control efforts using a containment strategy at an estimated cost of $11 million. This involved systematic herbicide and manual control efforts from the leading edge of the infestations (north and south) toward the larger infestations to the west. The intensity of infestation for each monitoring event is depicted in the color-coded stacked bar graph in **Figure A3-1-19**. In 1993, roughly 25% of the infested areas were classified as high intensity infestations (> 50% cover). By 2001, nearly all dense infestations were removed and by 2012 only low level infestations were found in two grid cells.

SFWMD and partner agencies continue to monitor and control melaleuca on the Lake Okeechobee marsh and anecdotal evidence from land managers suggests melaleuca reestablishment is occurring at a slower rate in the marsh (Francois Laroche, personal communication). This is expected given the reduction in propagule pressure over time from reproductive plants. But land managers also note that colonizing plants consistently show evidence of moderate to heavy leaf and stem damage from biological control agents. The restoration of diverse, native herbaceous plant communities in former melaleuca forests is evident throughout the western marsh. This recovery has occurred through natural recruitment. Unfortunately, much of the recovering marsh habitat is heavily invaded by invasive graminoid species such as torpedograss and tropical American watergrass.

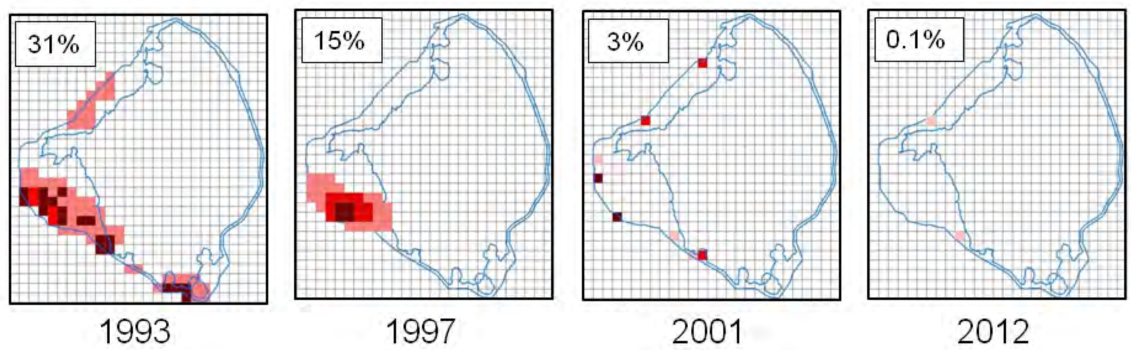

Figure A3-1-18. Distribution and abundance of melaleuca on the Lake Okeechobee western marsh between 1993 and 2012. Darker red colors indicate higher densities of melaleuca within 2-km grid cells. Percentage of marsh habitat inhabited by melaleuca indicated in the top right corner for each year.

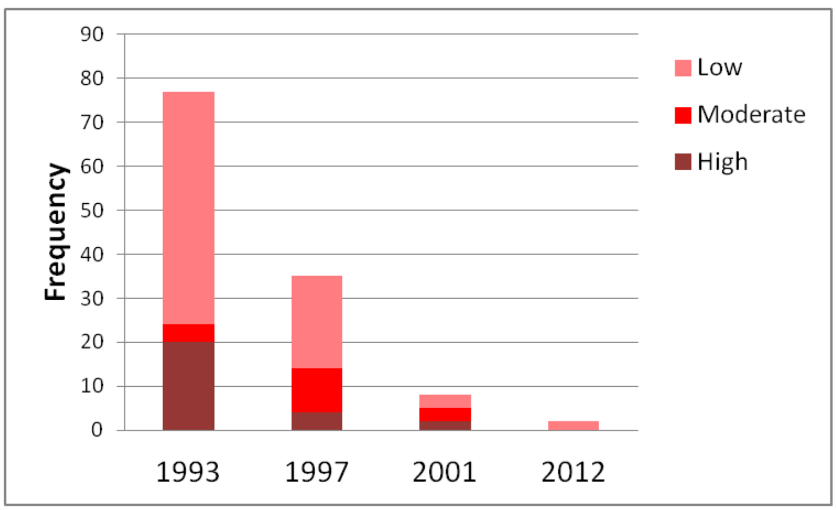

Figure A3-1-19. Frequency of 2-km grid cells containing melaleuca by infestation level on the Lake Okeechobee marsh 1993–2012.

Status of Floating Aquatic Plants in Lake Okeechobee

Water hyacinth (*Eichornia crassipes*) and water lettuce (*Pistia stratiotes*) pose a significant risk to the ecology of the Lake Okeechobee marsh community (Rodgers et al. 2012). The exact date when these plants were first introduced is unknown. Anecdotal accounts indicate water hyacinth was present on Lake Okeechobee in the early 1900s (SFWMD 1997). Water hyacinth and water lettuce are floating aquatic plants that are native to the Amazon River basin of South America. They both reproduce vegetatively and sexually (Langeland and Burks 1998). Both plants reproduce rapidly, however the explosive growth rate of water hyacinth exceeds any other tested vascular plant (Wolverton and McDonald 1979). One acre of water hyacinth can weigh more than 200 tons and can double its size in a little as 6 to 18 days (Mitchell 1976).

Floating aquatic plants impact operations and maintenance activities, as well as the ecology of the lake. These plants form large mats that block waterways, federal navigation channels, and locks. The plants can make fishing and other water related activities impossible. Large floating mats of water hyacinth greatly diminish water flow and can slow down, and even prevent, flow through water and flood control structures. Floating aquatic plants inhibit the air-water interface, resulting in reduced oxygen levels in the water. Reduced oxygen levels in the water adversely impact fish and other aquatic wildlife. In addition, these plants can eliminate submersed native plant species by blocking sunlight and uprooting emergent plant species.

Since these plants are free floating they inhabit many areas of the lake. USACE has implemented a management program for floating invasive plants on Lake Okeechobee since the 1920s. Since eradication is not possible, the goal of the management program is to achieve maintenance control. The Lake Okeechobee Aquatic Plant Management Interagency Task Force works with USACE to implement the management program. The task force provides technical expertise, makes recommendations for management measures, monitors plant populations and conducts surveillance to determine efficacy of treatments.

The acreage of floating plant populations ranges from year to year due to several factors including the management program, seasonal and hydrological conditions, available funding, nutrient loading and extreme weather events such as tropical storms and hurricanes. On average 6,000 acres of water hyacinth and water lettuce are treated on the lake each year at an average cost of approximately $1,000,000. Recently, the acreage treated per year has increased significantly. Acreages treated increased per calendar year (CY): CY2009 ~ 8,100 acres, CY2010 ~15,000 acres, and CY2013 ~ 8,100 as of August 2013.

It is difficult to determine the one factor that prompted the increased growth of the floating plant populations since many factors influence growth and often occur simultaneously. It is speculated that lake elevation influences the increase and decrease of plant populations. Aerial estimates of treatable floating plant acreages observed from 1989 to 2013 range from 0 to over 3,000 acres. In **Figure A3-1-20** lake elevations and aerial estimates of treatable floating plant acreages are provided from 1989 through 2013. During this period of time, less treatable floating plants were present on the lake under higher (15.5 and 17.5 feet mean sea level) and extreme low lake elevations (below 10 feet mean sea level). This could be attributed to several factors. Under higher lake elevations, floating plants were more accessible and occurred less in native plant communities, which allow herbicide treatments to be conducted more efficiently. Extreme low lake elevations reduced the inhabitable areas for floating plants since many areas were dry, however moist soils in some areas allowed floating plants to persist until the lake level increased. In many cases, following extreme low lake elevations due to drought, there was a significant increase in treatable floating plant acreages due to seed germination and persistence of the plants in slightly wet to moist soil areas. Since 2008, there has been more extreme variation in the amount of treatable floating plants on the lake, which could be associated with lower lake elevations.

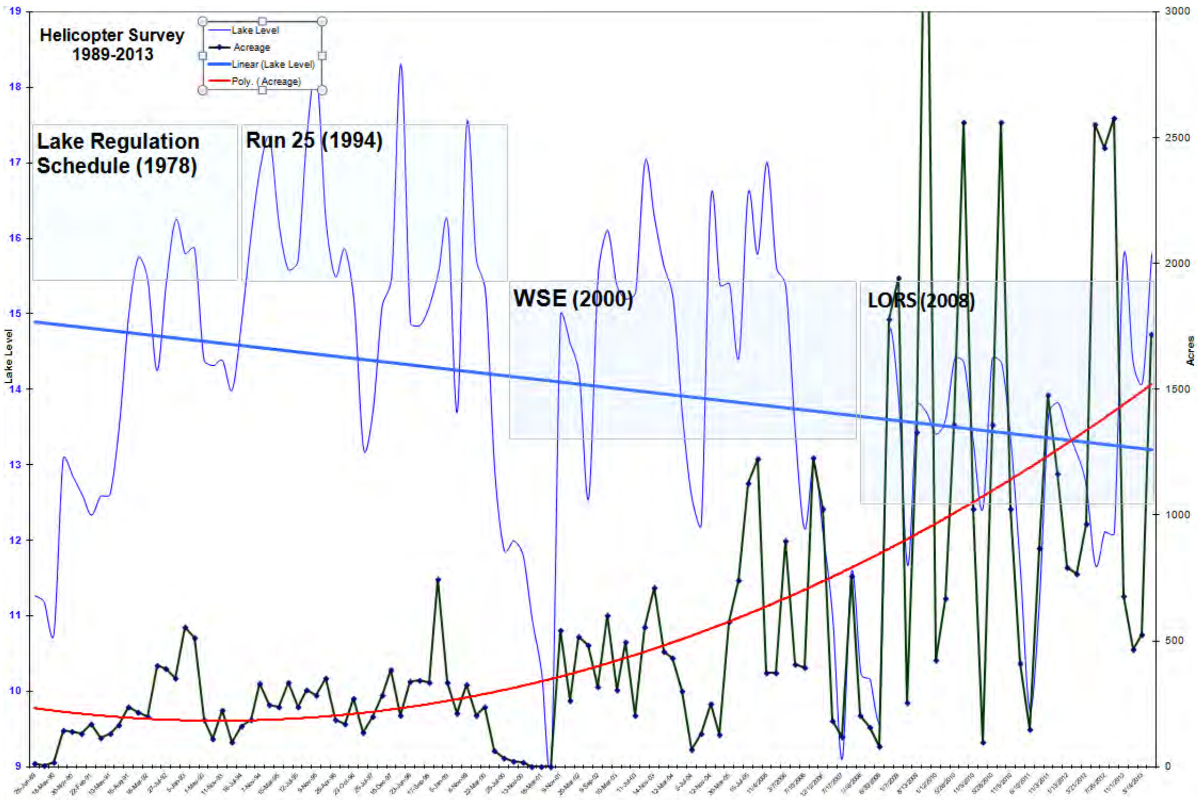

Figure A3-1-20. Lake Okeechobee stage and floating aquatic plant abundance 1989–2013.

Recently, another factor that has contributed to increased floating plant acreage is the presence of Everglade snail kite (*Rostrhamus sociabilis plumbeus*) nests in the northwestern marshes. In order to avoid impacting snail kite nests, treatments are curtailed within 1,640 feet of an active nest site, which is the FWC recommended distance. The number of snail kite nests is high, so acreage within the set-back is large. Curtailing treatments in these areas allows small plant populations to persist for several months without being treated and contributes to the increased amount of treatable floating plants on the lake.

Invasive Grasses in the Western Marsh

<u>Tropical American Water Grass</u>

Tropical American water grass (*Luziola subintegra*) is native to Mexico, Central and South America (extends south to Argentina) and the Caribbean basin. It grows in both aquatic and terrestrial habitats and reproduces vegetatively and by seed. The first record of occurrence of tropical American watergrass in the United States was at Lake Okeechobee in 2007 (Kunzer and Bodle 2008). Two large mats (approximately 5 and 198 acres each) were found near Harney Pond Canal in Fisheating Bay at Lake Okeechobee. From the initial population, this plant quickly spread and by July 2009 more than 2,000 acres of the plant were treated in the lake by SFWMD. The plant was also found at the mouth of Fisheating Creek in both emergent and terrestrial forms. Since Fisheating Creek is the only unregulated flow into Lake Okeechobee, it is thought this area was the point of introduction. Since the initial sighting of tropical American watergrass in December 2007, other populations have been found in the Cody's Cove-Eagle Bay area, near Observation Shoal and inside Lake Okeechobee near the S-77 structure and downstream in the Caloosahatchee River (C-43 Canal). The majority of populations have occurred in areas that receive water flow from Fisheating Bay.

Torpedograss

Torpedograss is an invasive nonnative perennial grass present in most regions of south Florida, but most dominant in disturbed wetlands such as those in the LO region. It is native to Europe, Asia and Africa and is a prominent invasive grass in many agricultural crops (Hossian et al. 1999). It invades terrestrial, wetland and aquatic natural areas and is one of the most invasive perennial grasses in tropical and subtropical regions of the world (Sutton 1996).

In 1996, approximately 16,581 acres of torpedograss were present in LO's marsh (Hanlon and Brady 2005). This one invasive grass species displaced approximately 17% of the lake's marsh that previously consisted of native plant communities and shallow areas of open water. The SFWMD implemented a torpedograss management program in 2000 with an objective to reduce the distribution and limit further expansion into new areas of the lake (Hanlon and Brady 2005).

A study conducted by Hanlon and Brady (2005) consisted of (1) mapping the distribution of torpedograss and quantifying its areal coverage to establish a baseline; (2) evaluating the rate of expansion in relation to different hydrologic conditions; (3) determining the efficacy of herbicide treatments; and (4) documenting plant community succession following treatments. The baseline coverage maps were generated for the central and northwestern marsh in LO by utilizing color infrared images that were created in 1994 and 1996. Three study sites were selected and images collected in 1999 were used to determine the rate of torpedograss expansion in these areas. The average rate of expansion in the three study sites from 1994 to 1999 was 21% (west site 26%, middle site 32%, east site 13%). This study did not determine the primary reason for the expansion, however, it did identify that expansion occurred in areas where flooding depth was less than 0.5 meter for 18 months, less than 1 meter for 48 months and exceeded 1.0 meter for 10 months. The study indicates that torpedograss invades sites that are inundated for extended periods of time and the grass is able to tolerate relatively deep water. Treatment efficacy was rated as moderate for the west site (64%) and middle site (80%) and excellent at the east site (96%). After treatment, native plants such as spikerush (*Eleocharis cellulosa*) and fragrant water lily (*Nymphaea odorata*) established in the former torpedograss areas.

Approximately 10,000 acres of torpedograss were treated from 2004 to 2006 and more than 20,000 acres were treated from 2007 to 2009 (SFWMD et al. 2011). Some treatments provided several years of control while others were less effective. Evaluations continue in order to reduce this variability. Treatments have significantly reduced torpedograss coverage to an estimated 9,000 acres on the lake at this time. However, funding for treatments in 2012 was severely curtailed and the plant population is once again expanding in the LO marshes (Rodgers et al. 2014).

Herbicide treatments for invasive plants on LO are coordinated through the Lake Okeechobee Aquatic Plant Management Interagency Task Force and are performed by SFWMD with funding from the FWC Invasive Plant Management Control Trust Fund. Development of selective biological control of torpedograss is not likely to be successful because of the broad similarities of grass species. Numerous herbicides have recently received approval from the United States Environmental Protection Agency for use in aquatic sites. Some of these herbicides are expected to be effective on grasses. Testing of the herbicides to evaluate effectiveness on torpedograss is planned within the near future.

Island Apple Snail

The island apple snail (*Pomacea maculata*) is considered one of the 100 world's worst invasive alien species (Lowe et al. 2000). It was first identified in Florida in 1987 in canals around Lake Okeechobee (Rawlings et al. 2007). Typically, island apple snails are globular or spherical in shape; they may have bands, brown, black, or yellowish-tan with color patterns being variable. Island apple snails inhabit shallow freshwater habitats and have temperature tolerances from 15 to 36 degrees Celsius (Ramakrishnan 2007). The island apple snail has a voracious appetite for vegetation and in other countries has converted lush ecosystems into barren areas. Potential impacts to Florida flora and fauna include destruction of native aquatic vegetation, competition with native aquatic fauna, alteration of water quality, hybridization with other species (Rodgers et al. 2013), and introduction of nonnative parasites (Karatayev et al. 2009). Economic impacts could include damage to rice crops, which has been documented in Asia, and increased maintenance costs for levees due to constant burrowing, which has been documented in Texas (Burlakova et al 2009). The island apple snail is an omnivorous feeder; the preferred diet is aquatic plants however they will also consume periphyton, detritus, and fish and snail eggs (EFSA 2012). It was identified by Burlakova et al. (2009) that apple snails will feed on native plant species such as coontail (*Ceratophyllum demersum*), spider-lily (*Hymenocallis latifolia*), widgeongrass (*Ruppia maritime*), and lanceleaf arrowhead (*Sagittaria lancifolia*). In addition, smooth cordgrass (*Spartina alterniflora*), California bulrush (*Schoenoplectus californicus*), water cana (*Canna flaccida*), and common cattail, may be consumed but only when shoot dryness is low. Island apple snails are also known to feed on invasive plants such as alligator weed (*Alternanthera philoxeroides*) and water hyacinth. Due to their broad diet, it is thought these snails could remain at high densities once they have consumed the available aquatic plants within a given area. The island apple snail has a high fecundity and reproduces over a long period of time (March to early November) (Burlakova et al. 2010). In 2006, a federal regulation was implemented by USDA to require a permit requiring for importation or interstate shipment of all marine and freshwater snails. No permits are issued for the genus *Pomacea* except for *P. diffusa*, the spike-topped apple snail (Rawlings et al. 2007).

At this time, little research exists that identifies impacts the island apple snail could have on the native apple snail (*Pomacea paludosa*), the endangered Everglade snail kite and LO ecology. In 2010, the Lake Okeechobee Apple Snail Monitoring Program was initiated in order to conduct a detailed monitoring and survey program for the Florida apple snail within portions of the LO western littoral zone. The program was implemented in accordance with an agreement between the United States Fish and Wildlife Service and USACE. LG2 Environmental Solutions, Inc. was awarded a contract to implement the monitoring program. The overall objective was to determine the response of apple snails to long-term changes in water regulation and shifts in vegetative structure across the littoral zone. Another objective was to determine the rate at which snail populations recover from extreme flood or drought in the littoral zone. The monitoring program is being implemented for five years (2010–2015) in the western littoral zone, from the Herbert Hoover Dike to the waterward edge of the littoral zone. While the focus of the monitoring program was to monitor the native apple snail, data was also collected for the island apple snail. Three years of the monitoring project have been completed.

Data gathered in 2010 and 2011 indicate the island apple snail has become well established in the northwestern littoral zone of the lake and was spreading from the initial point of entry, which is thought to be the Kissimmee River. Data gathered in 2012 indicates that not only is the island apple snail well established, but it is also spreading southward. In addition, in locations where the island apple snail was captured, there was typically substantially higher numbers of captures (101) of this species compared to the native apple snail (29) in other areas. The 2012 samplings were the first samplings to capture both native and invasive snails in the same sampling site. Not only were they captured within the same sampling site, but on six occasions, within the same throw trap.

Monitoring Period – 2010

- 22 sampling locations, within six polygons of potential apple snail habitat
- Six polygons sampled, four contained apple snails
- Two of the four polygons sampled, native apple snails only
- Two of the four polygons sampled, island apple snails only
- No areas sampled resulted in the capture of both the native and island apple snails

Monitoring Period – 2011

- 20 sampling locations in eight polygons (down gradient along transects from 2010)
- Eight polygons sampled, six contained apple snails
- Three of eight polygons sampled, native apple snails only
- Three of eight polygons sampled, island apple snails only
- No areas sampled resulted in the capture of both the native and island apple snails

Monitoring Period – 2012

- 33 sampling locations in eight polygons (between sampling locations 2010–2011)
- Eight polygons sampled, six contained apple snails
- One of eight polygons contained the native apple snails only
- Three of eight polygons contained the island apple snails only
- Two of eight polygons contained both the native and island apple snails
- Two of eight polygons no snails were captured

Overview of Impacts to Lake Okeechobee Region Performance Measures

In the LO region, at least one potential threat from an invasive species (and in some cases influenced by many species) was identified for each performance measure. In addition, one performance measure has the potential to influence both the reduction and expansion of invasive nonnative plants. **Table A3-1-14** provides examples of threats that could impact the success of each performance measure. It is expected that a better understanding of nonnative animal impacts over time will reveal more potential threats and risks. Examples for potential threats for each performance measure are identified below in **Table A3-1-14**.

Table A3-1-14. Performance measures and potential threats or risks of invasive species for the LO region.

Performance Measure	Potential Threat or Risk
Native Vegetation Mosaic	Reduction of spatial extent of native plant populations and elimination of native plant communities.
Native Fish Recruitment	Reduction of suitable foraging habitat, decreased water quality and increased planktonic algae.
Macroinvertebrates	Organic and nutrient loading, reduction of spatial extent for expansion of submersed native plant communities and elimination of native plant communities.
Water Quality Mosaic	Organic and nutrient loading, macrophyte reduction and increased phytoplankton biomass.
Diatom/Cyanobacteria Ratio	Increased concentrations of nutrients and planktonic algae.
Lake Stage	Increased invasive plant populations and increased management of those species.

The vegetation mosaic performance measure could be impacted by several invasive plant species such as torpedograss, tropical American water grass and hydrilla (*Hydrilla verticillata*) through competition and expansion of existing plant populations. Expansion of these species would reduce the spatial extent for native plant population expansion. In addition, it is not known how the island apple snail will effect native plant communities, however this species has the potential to eliminate large expanses of native plants and to leave these areas largely devoid of vegetation. Moderate snail densities are likely to reduce or eliminate preferred plant species and the potential for large expanses of native plant communities to be eradicated, which would have a direct effect on plant biodiversity (Scheffer et al. 1993).

The native fish recruitment performance measure could be impacted by invasive plant species such as torpedograss, tropical American watergrass, hydrilla and water hyacinth. These invasive nonnative plant species reduce spatial extent of native plant communities and thereby reduce suitable habitat for cover and foraging. While at low densities these plant communities may provide some environmental benefits, they are capable of rapid expansion and do not provide optimal conditions conducive for native fish recruitment. It is not known how the nonnative island apple snail will affect native plant communities. There is a risk the snail could eliminate large expanses of native plants and this would decrease suitable cover and foraging habitat for juvenile fish. In addition, since aquatic plants serve as food or as a substrate for periphyton, which is consumed by macroinvertebrates (James et al. 2000), fish and waterfowl (Lodge et al. 1998), the island apple snail could potentially impact the macroinvertebrate performance measure. Other effects of plant communities being eliminated by the apple snail could include degraded water quality, increases in nutrient concentrations, subsequent increased floating plant growth, and phytoplankton biomass, which could also impact native fish recruitment, the water quality mosaic, and the diatom/cyanobateria ratio performance measures.

The lake stage performance measure has potential to both increase and decrease the rate of expansion of invasive plant populations. For example, it appears the distribution of torpedograss is strongly influenced by hydrologic conditions (Smith et al. 2004). Torpedograss has been identified to be restricted to the higher elevation regions of the LO marsh that had an average hydroperiod of 81% (Richardson et al. 1995). During the 2001 drought, torpedograss expanded into Moonshine Bay which demonstrated its ability to invade lower elevation regions of the marsh (Hanlon and Brady 2005). This raises a concern regarding lake stage and the potential for lower lake levels to permit torpedograss to further invade areas currently devoid of the invasive grass species. In addition, it appears there is a strong correlation between the lake stage and the floating plant populations on Lake Okeechobee (**Table A3-1-13**). It is possible that lower lake elevations could allow expansion of nonnative terrestrial vegetation (Lockhart 1995).

Key Findings and Recommendations

A current comprehensive plan for managing priority nonnative species on LO could not be located. The initial Lake Okeechobee Protection Program Exotic Species Plan was completed in 2002 and updated in 2003 (SFWMD et al. 2003). Since that time, many new species have been identified to be potential threats to the lake's ecology. The plan should be updated to include all aspects of management strategies including prevention, monitoring, EDRR procedures, research needs, and control and management measures as well as education and outreach.

The lack of monitoring of invasive species is a critical factor. There is not a comprehensive monitoring program to collect information regarding the current distribution and rate of expansion of invasive species. Data collected is often result of monitoring other species. It is recommended that a program be implemented to monitor invasive species and their impacts.

Due to potential impacts, it is recommended that a monitoring program for island apple snails be implemented within the LO region. Based on the monitoring data collected for native apples snails (and identified in the monitoring report) the following is recommended: (1) monitor the spread of the island apple snail and their impacts on native plant and animal communities, water quality, nutrient concentrations, etc.; (2) further investigate control measures for the island apple snail; (3) study conditions that affect island apple snail reproduction and dispersal; (4) study the occurrence of parasites in apple snails; and (5) study the effects of the island apple snail on native apple snails.

Funding is critical to managing invasive species. Many species are not addressed due to the lack of funding. In some cases, invasive species that were successfully managed aggressively, such as torpedograss, are now apparently gaining foothold due to the lack of funding to continue aggressive treatments. It is recommended that species such as torpedograss, tropical American water grass, water hyacinth, and water lettuce be aggressively managed, within the extent possible, to minimize impacts to the ecology of the lake. Control measures for species that are hard to control such as torpedograss and tropical American watergrass should be further tested and developed. In addition it is recommended that additional funding be acquired to address the species that are not currently being managed.

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APPENDIX 3-2: MONITORING MAPS

The following revised monitoring maps have been developed to reflect the changes in sampling locations resulting from the reduction in available funds for MAP monitoring projects that were listed in **Table 3-10** in the Monitoring and Assessment Plan Sustainability section within the Systemwide Science chapter. **Table A3-2-1** below provides a cross-reference for each map and the contract name provided in **Table 3-10**.

Table A3-2-1. Cross-reference of contract names listed in Table 3-10 and figures within this appendix.

Contract Name	Figure Number
West Coast Oyster Monitoring	**Figure A3-2-1**
East Coast Oyster Monitoring	**Figure A3-2-2**
Wading Birds & Aquatic Fauna Monitoring	**Figure A3-2-3**
Coastal Gradients	**Figure A3-2-4**
Florida Bay Juvenile Sportfish Monitoring	**Figure A3-2-5**
South Florida Fish Habitat Assessment Program (FHAP)	Not yet available
Biscayne Bay Salinity Monitoring	Not yet available
Hydrology & Salinity Monitoring in Ten Thousand Islands	**Figure A3-2-6**
Everglades Depth Estimation Network (EDEN)	**Figure A3-2-7**
Wet Season Trophic Sampling	**Figure A3-2-8**
Dry Season Aquatic Fauna Sampling	**Figure A3-2-9**
Wading Birds Monitoring	Not yet available
Monitoring Tree Island Condition in Southern Everglades	**Figure A3-2-10**
Landscape Pattern - Ridge, Slough and Tree Island Monitoring	**Figure A3-2-11**
Tree Island Monitoring	**Figure A3-2-12**
Monitoring of Marl Prairie Slough Gradients	**Figure A3-2-13**
Lake Okeechobee Wading Bird Monitoring	**Figure A3-2-14**
Monitoring of Oysters in the Ten Thousand Islands	**Figure A3-2-15**
Submerged Aquatic Vegetation	**Figure A3-2-16**, **Figure A3-2-17**, **Figure A3-2-18**, **Figure A3-2-19**
Monitoring of Ridge and Slough Maintenance and Degradation	**Figure A3-2-11**
Biscayne Bay Mangrove Fish Monitoring	**Figure A3-2-20**
Benthic Infaunal Monitoring in the St Lucie Estuary and Indian River Lagoon	Not yet available
Monitoring of Marsh Mangrove Fishes	**Figure A3-2-21**
Vegetation Mapping in Everglades National Park	Not yet available
Biscayne Bay Nearshore Submerged Aquatic Vegetation Monitoring	Not yet available
Monitoring of Biscayne Bay Epifaunal Communities	Not yet available
Alligator and Crocodile Monitoring	**Figure A3-2-22**, **Figure A3-2-23**, **Figure A3-2-24**
Monitoring Aquatic Fauna in Big Cypress	**Figure A3-2-25**
Sediment Elevation Tables	**Figure A3-2-26**
Fish and Invertebrate Network (FIAN) - USGS Component	**Figure A3-2-27**
Fish and Invertebrate Network (FIAN) - NOAA Component	**Figure A3-2-27**
Water Quality, Salinity & Circulation Monitoring	**Figure A3-2-28**

Note: NOAA – National Oceanic and Atmospheric Administration; USGS – United States Geological Survey

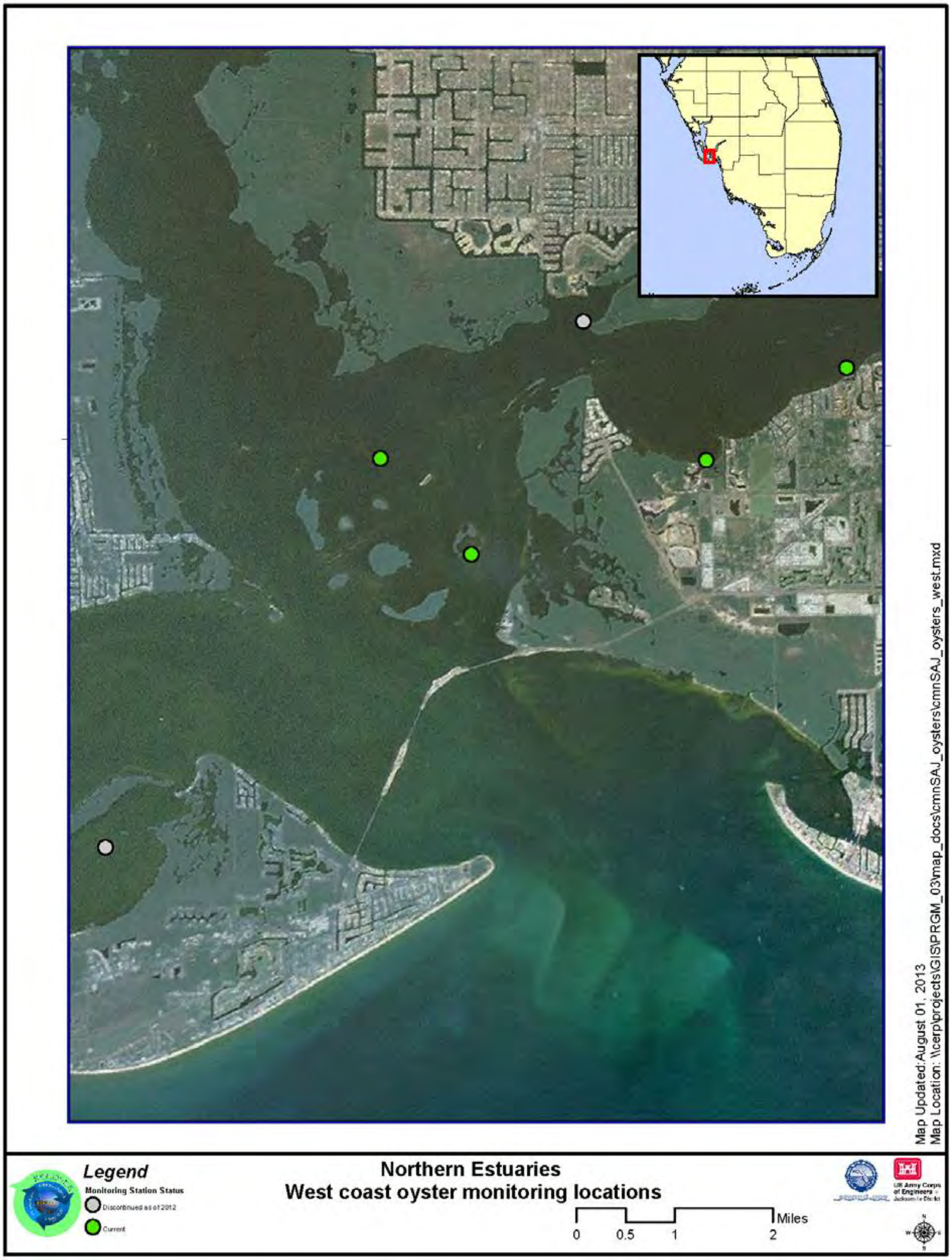

Figure A3-2-1. Monitoring map for West Coast Oyster Monitoring.

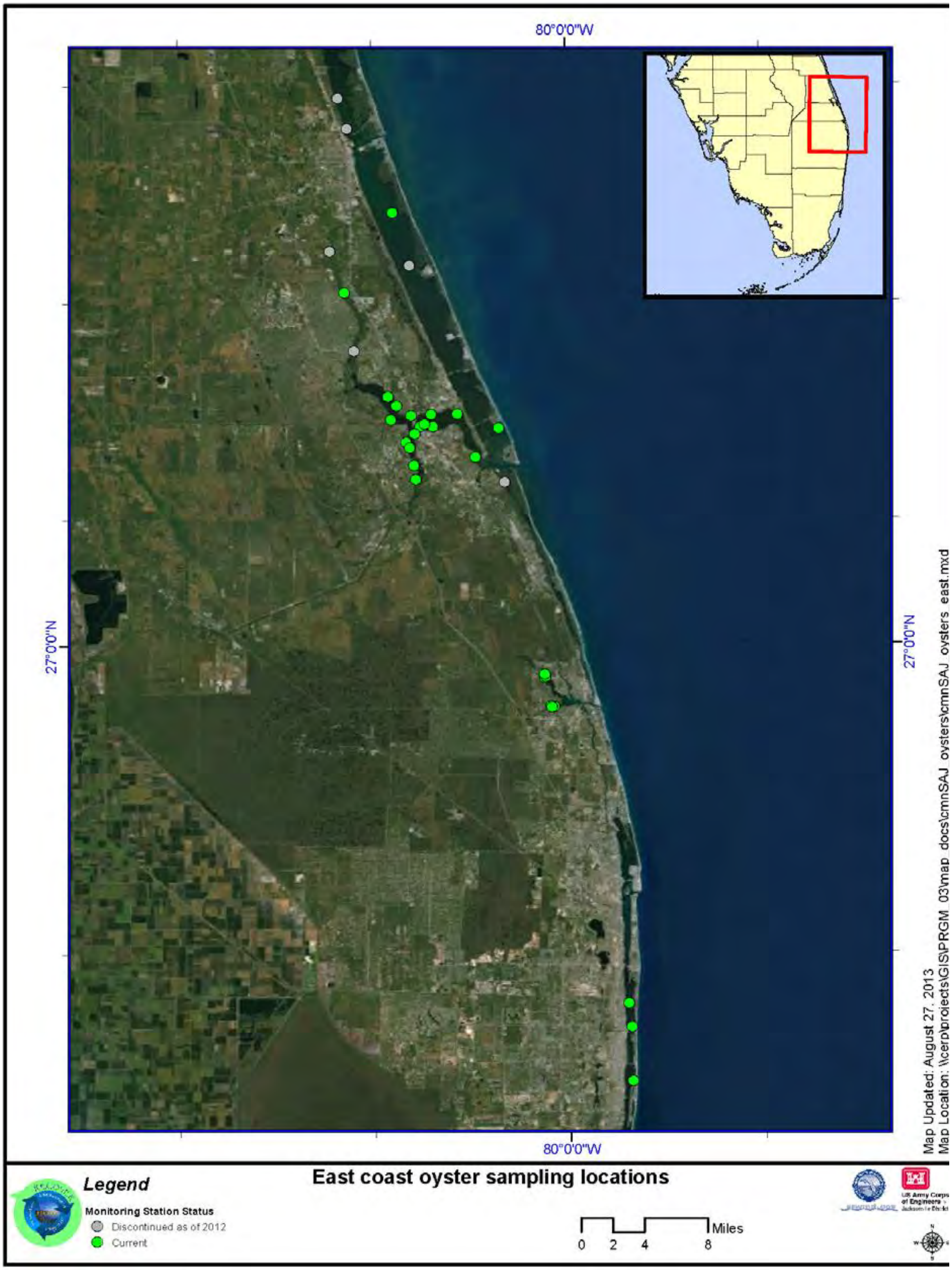

Figure A3-2-2. Monitoring map for East Coast Oyster Monitoring.

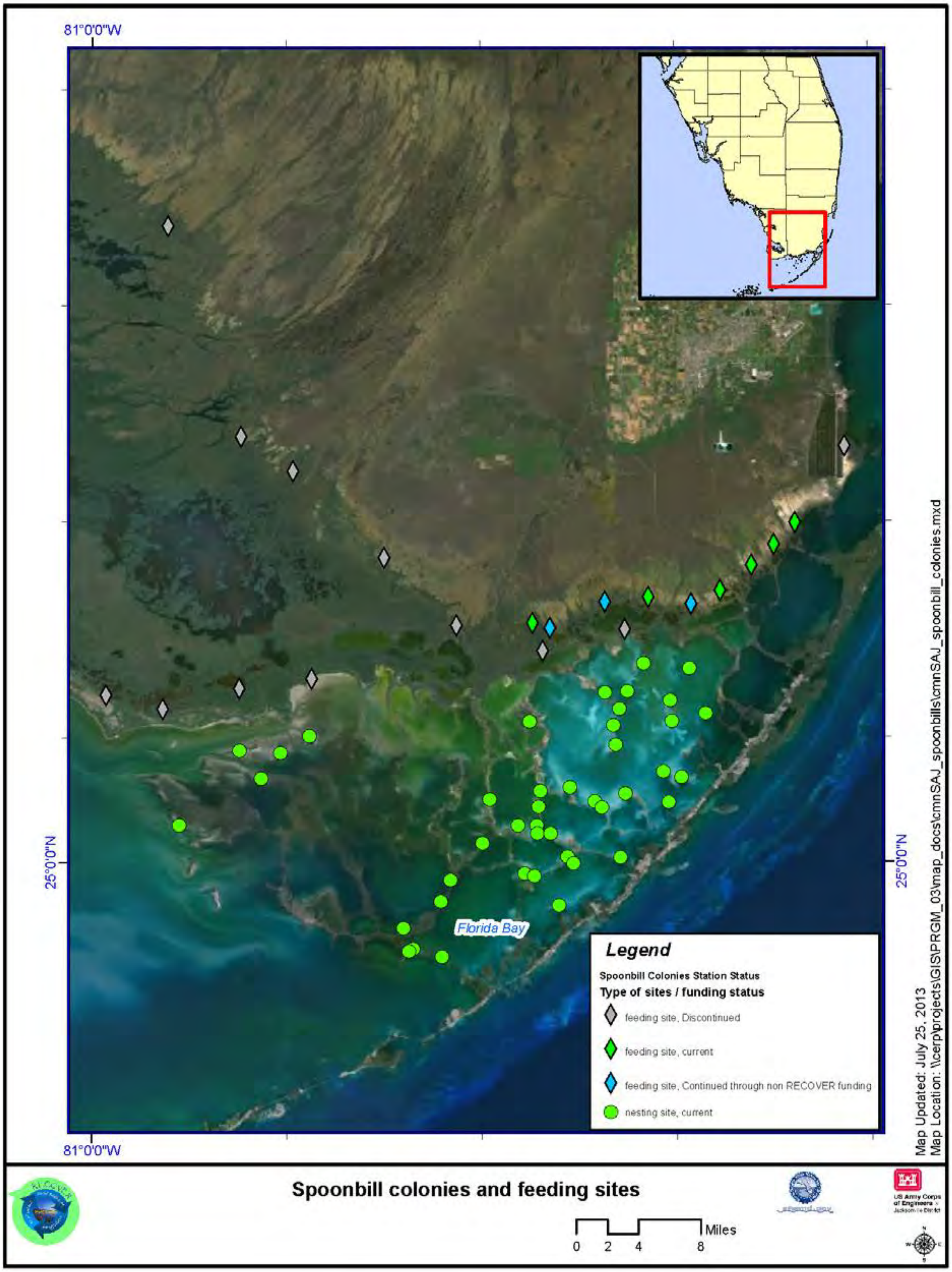

Figure A3-2-3. Monitoring map for Wading Birds & Aquatic Fauna Monitoring.

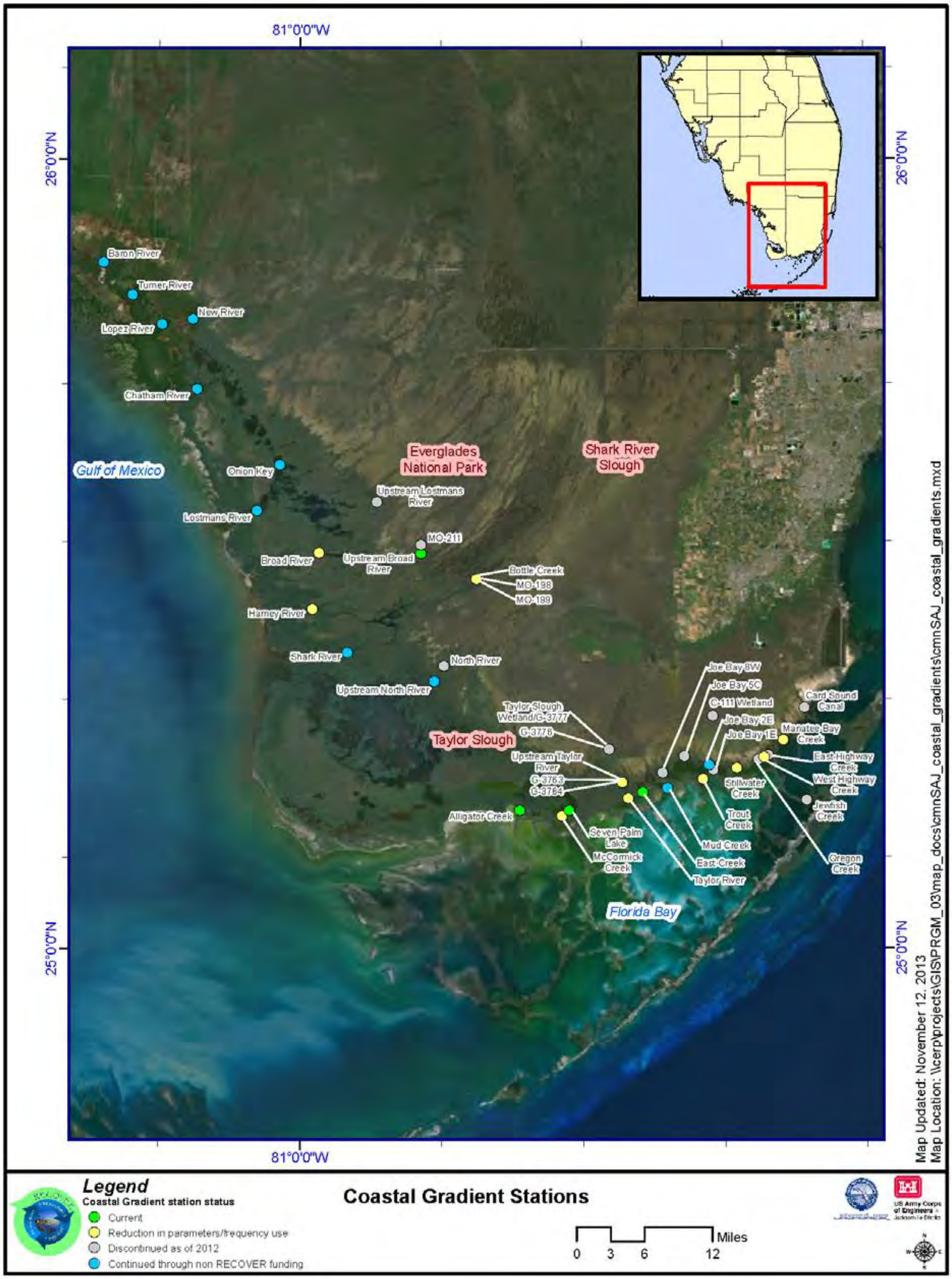

Figure A3-2-4. Monitoring map for Coastal Gradients.

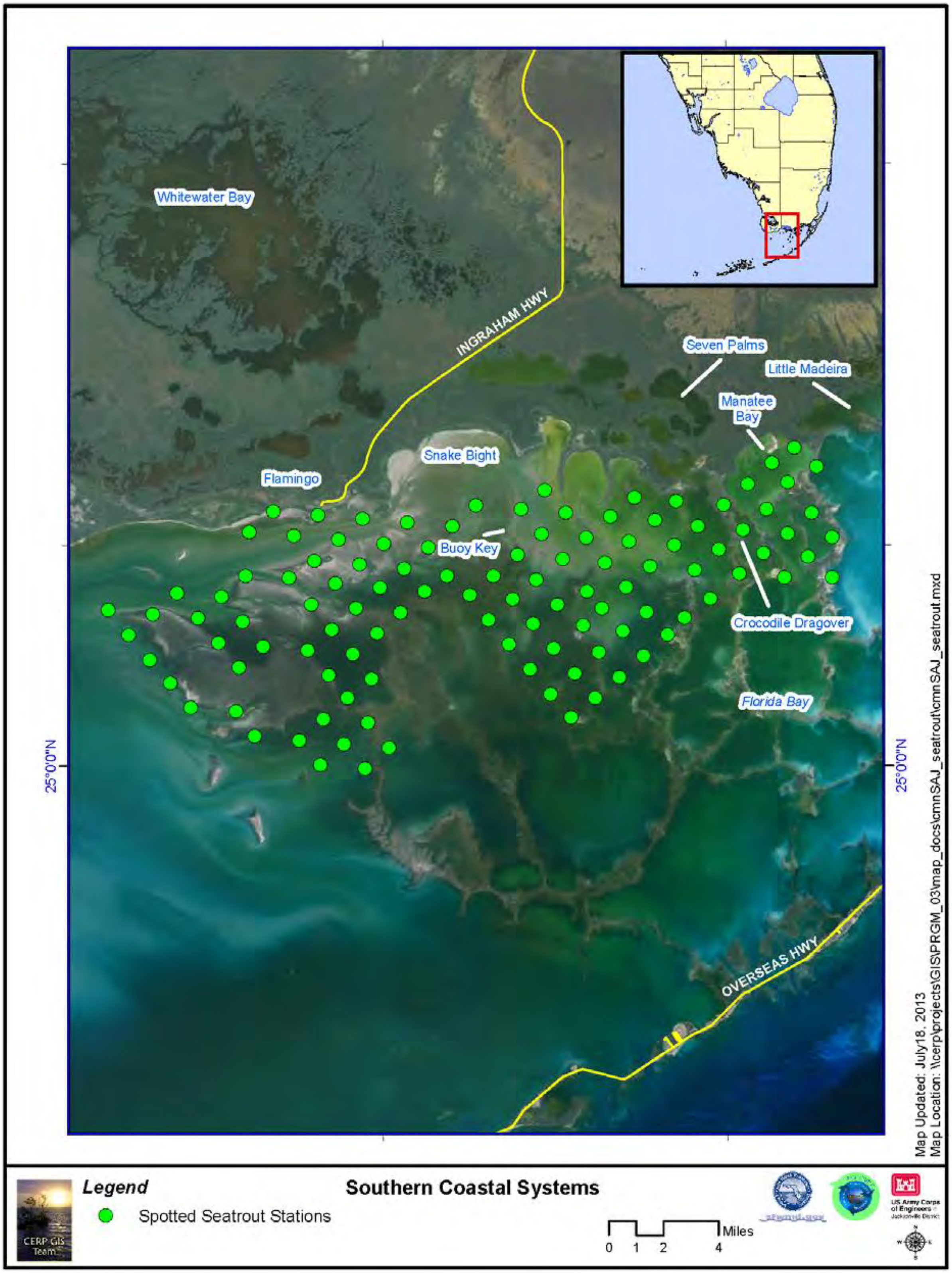

Figure A3-2-5. Monitoring map for Florida Bay Juvenile Sportfish Monitoring.

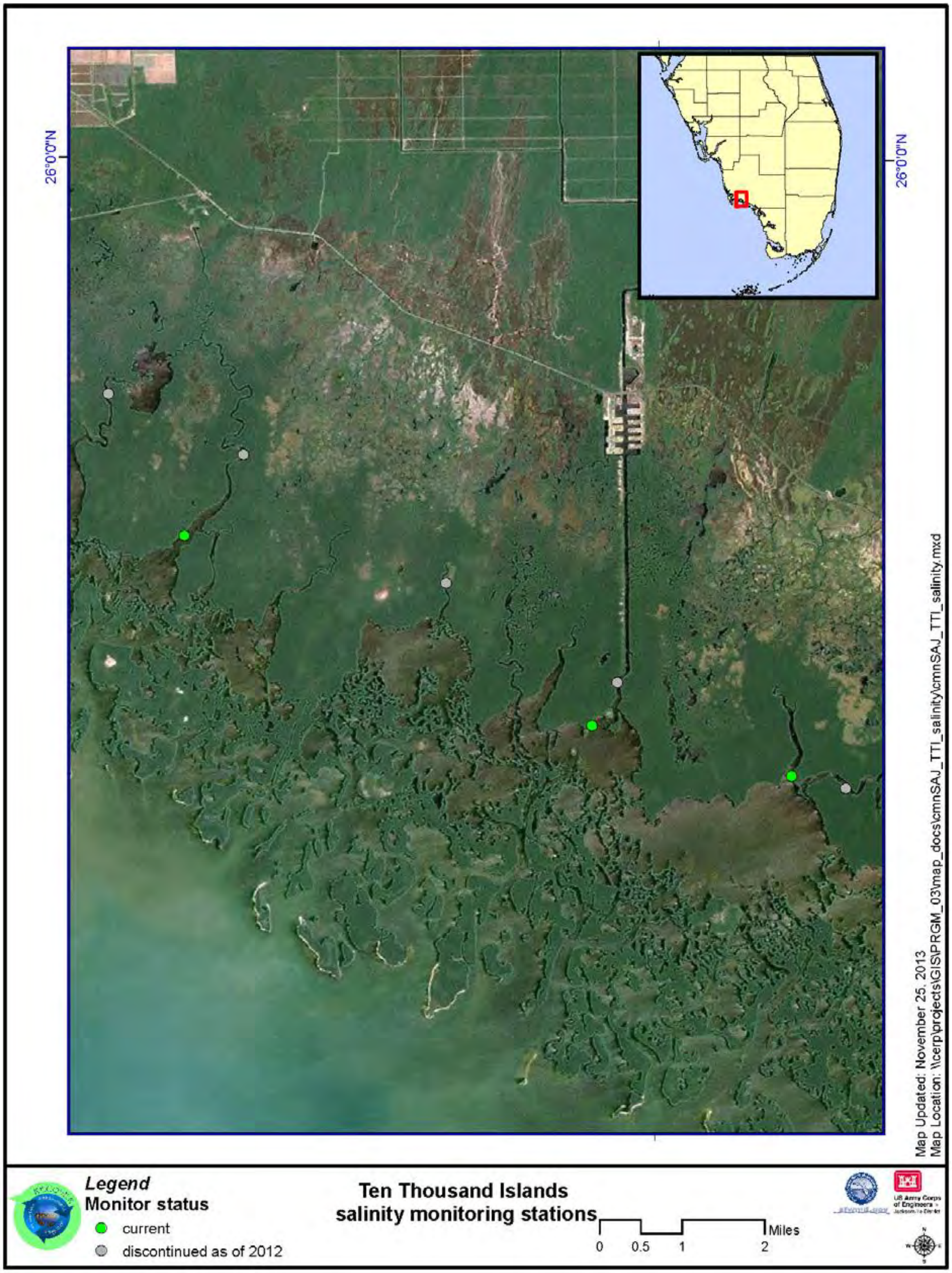

Figure A3-2-6. Monitoring map for Hydrology & Salinity Monitoring in Ten Thousand Islands.

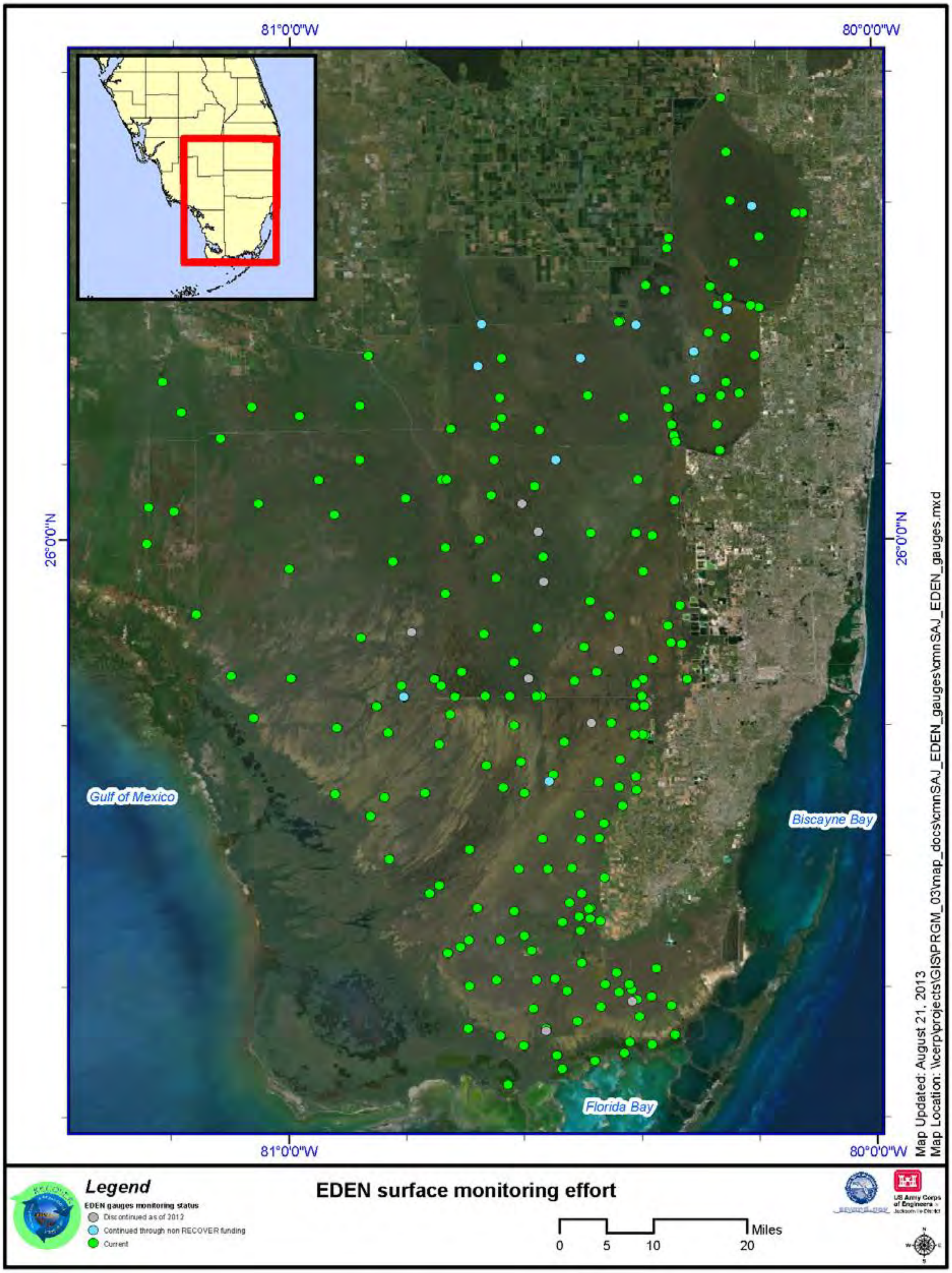

Figure A3-2-7. Monitoring map for EDEN.

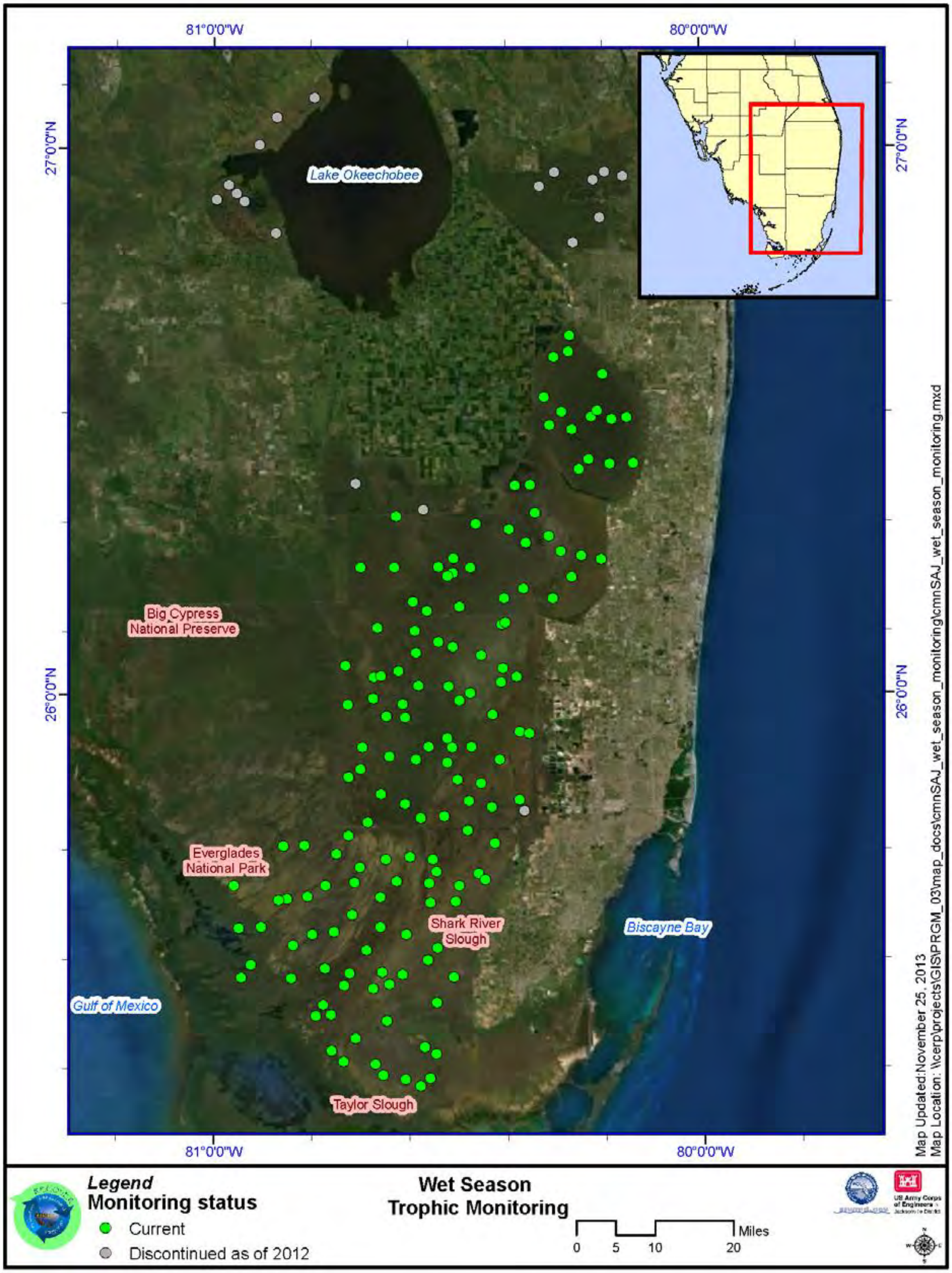

Figure A3-2-8. Monitoring map for Wet Season Trophic Monitoring.

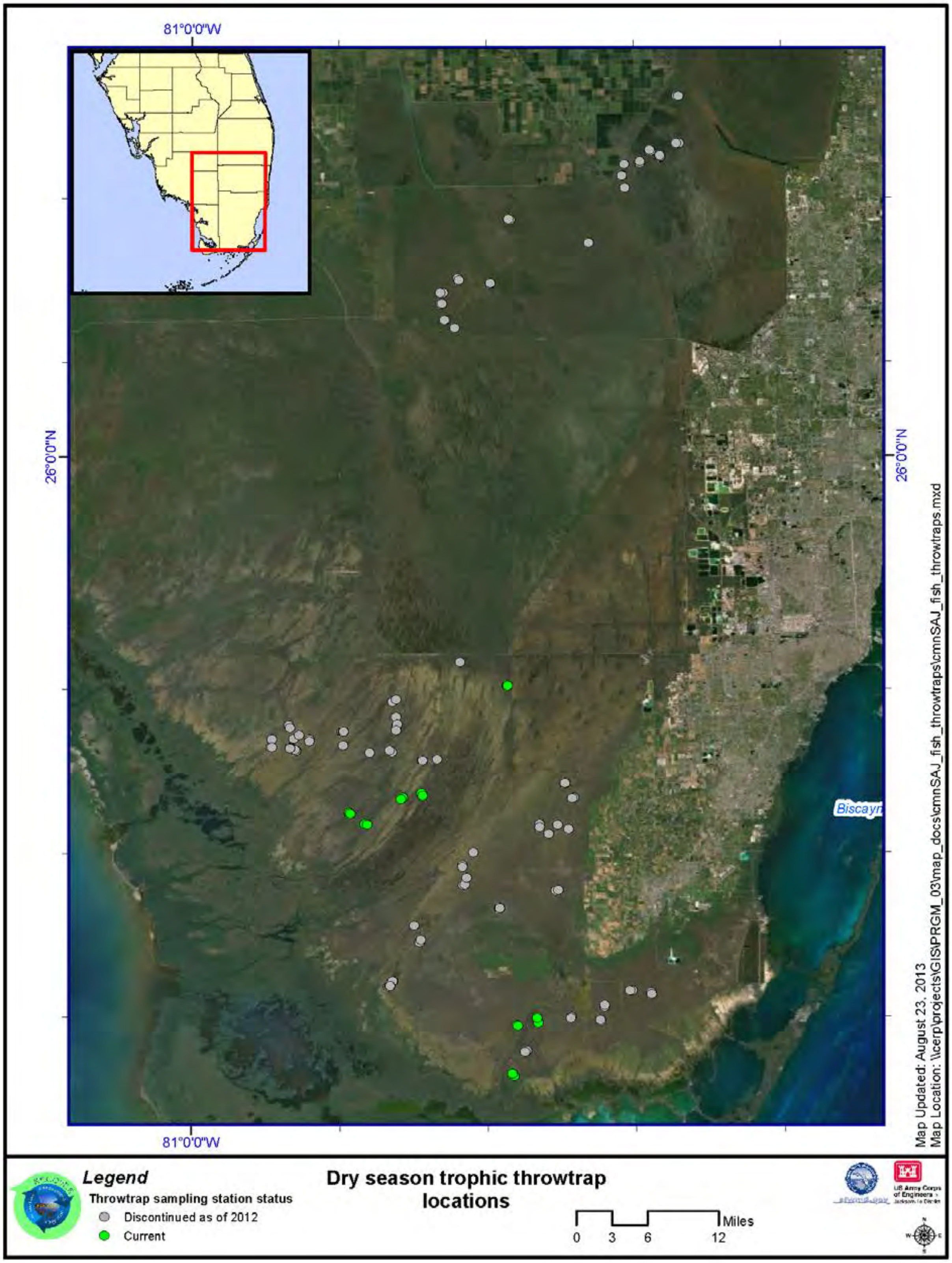

Figure A3-2-9. Monitoring map for Dry Season Aquatic Fauna Sampling.

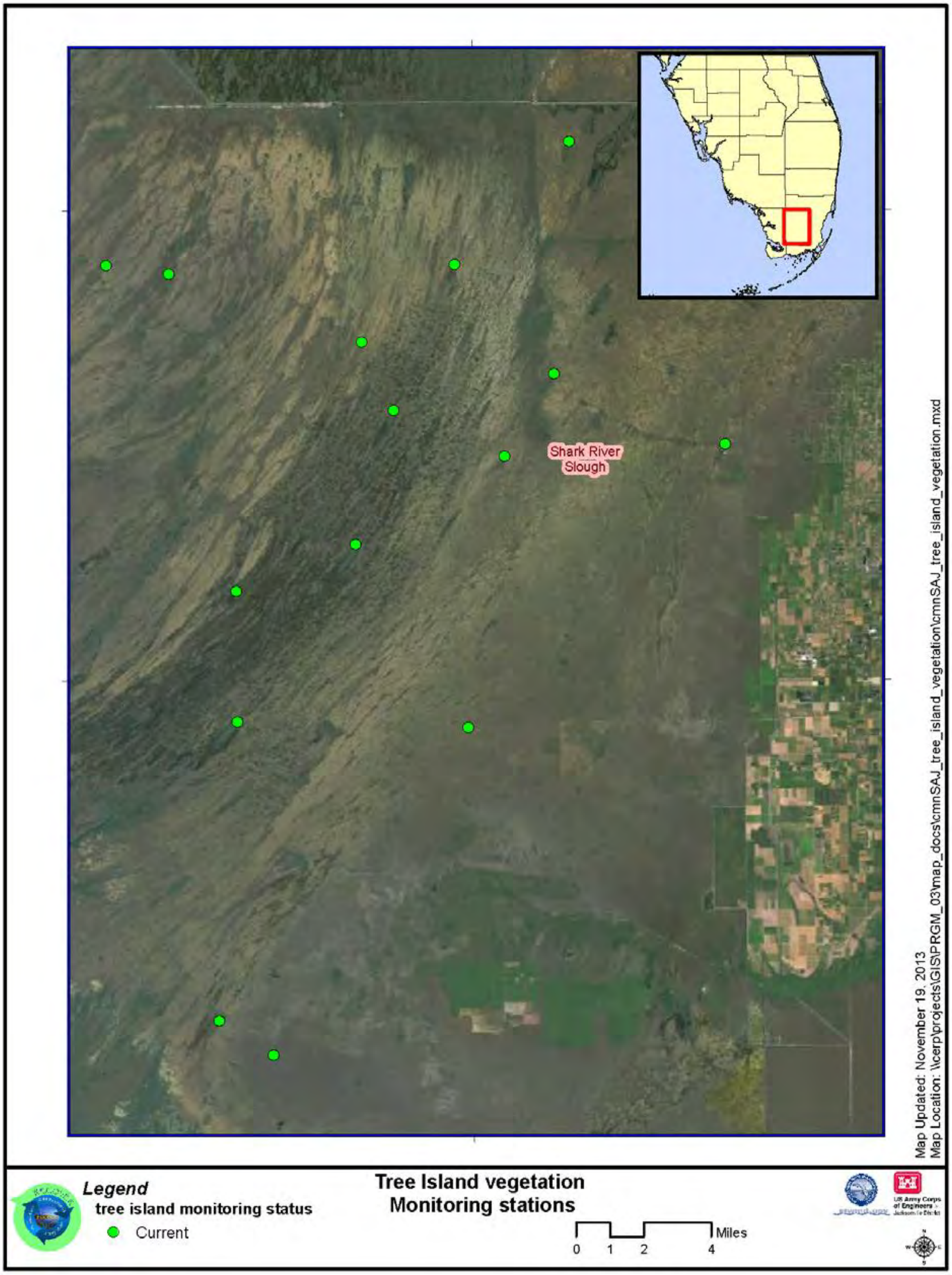

Figure A3-2-10. Map for Monitoring Tree Island Condition in Southern Everglades.

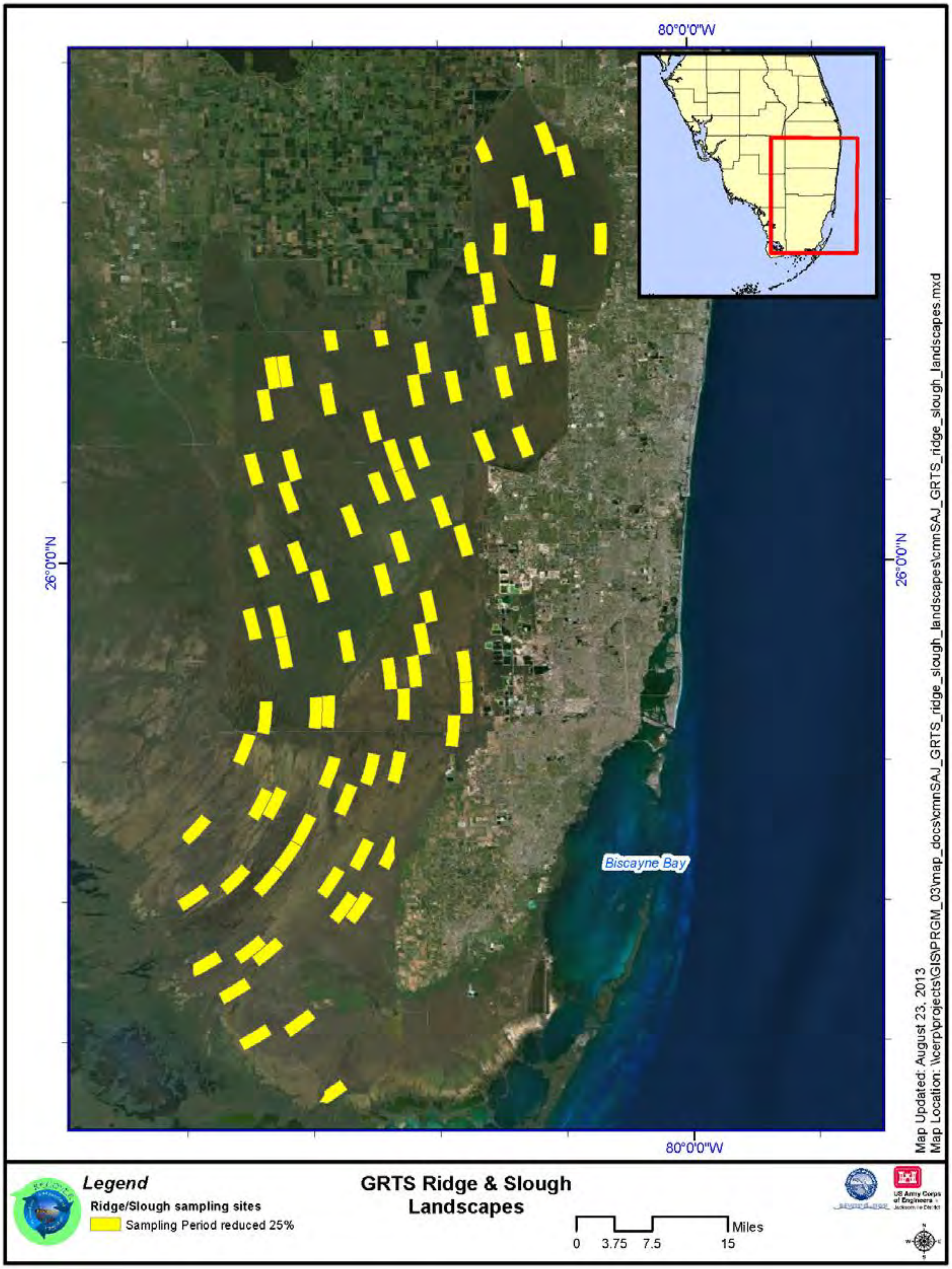

Figure A3-2-11. Map for Landscape Pattern - Ridge, Slough and Tree Island Monitoring and Monitoring of Ridge and Slough Maintenance and Degradation.

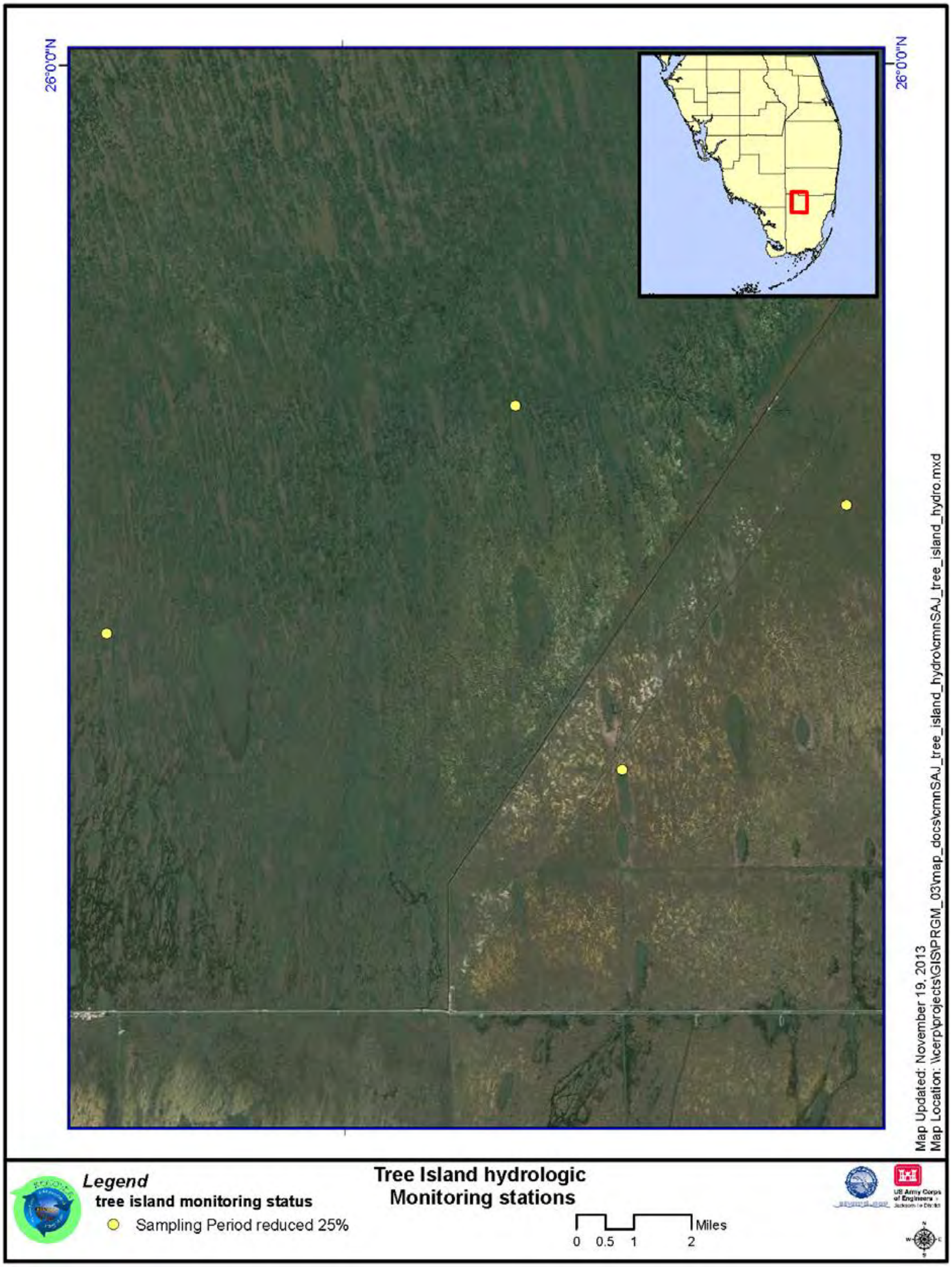

Figure A3-2-12. Map for Tree Island Monitoring.

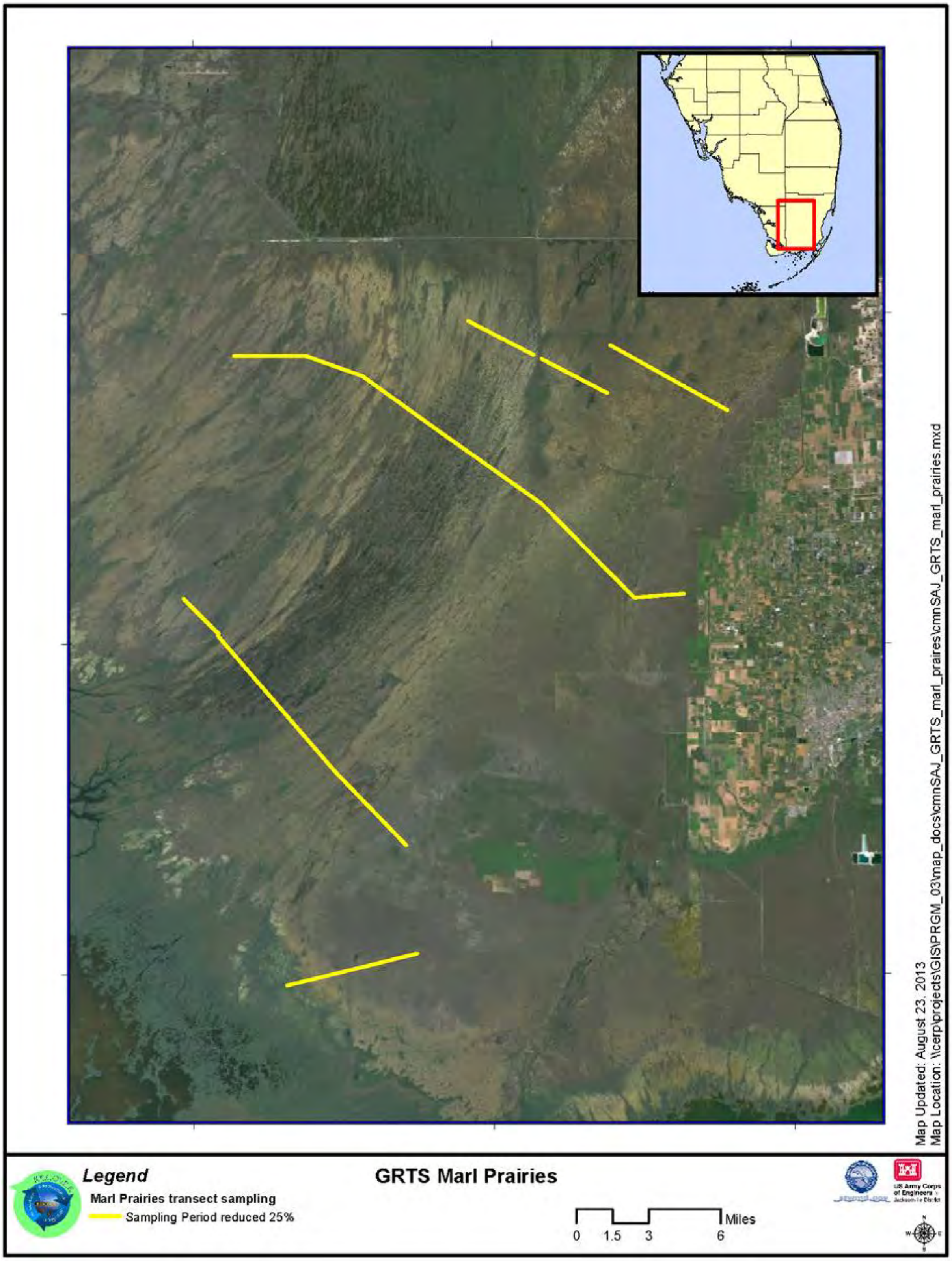

Figure A3-2-13. Map for Monitoring of Marl Prairie Slough Gradients.

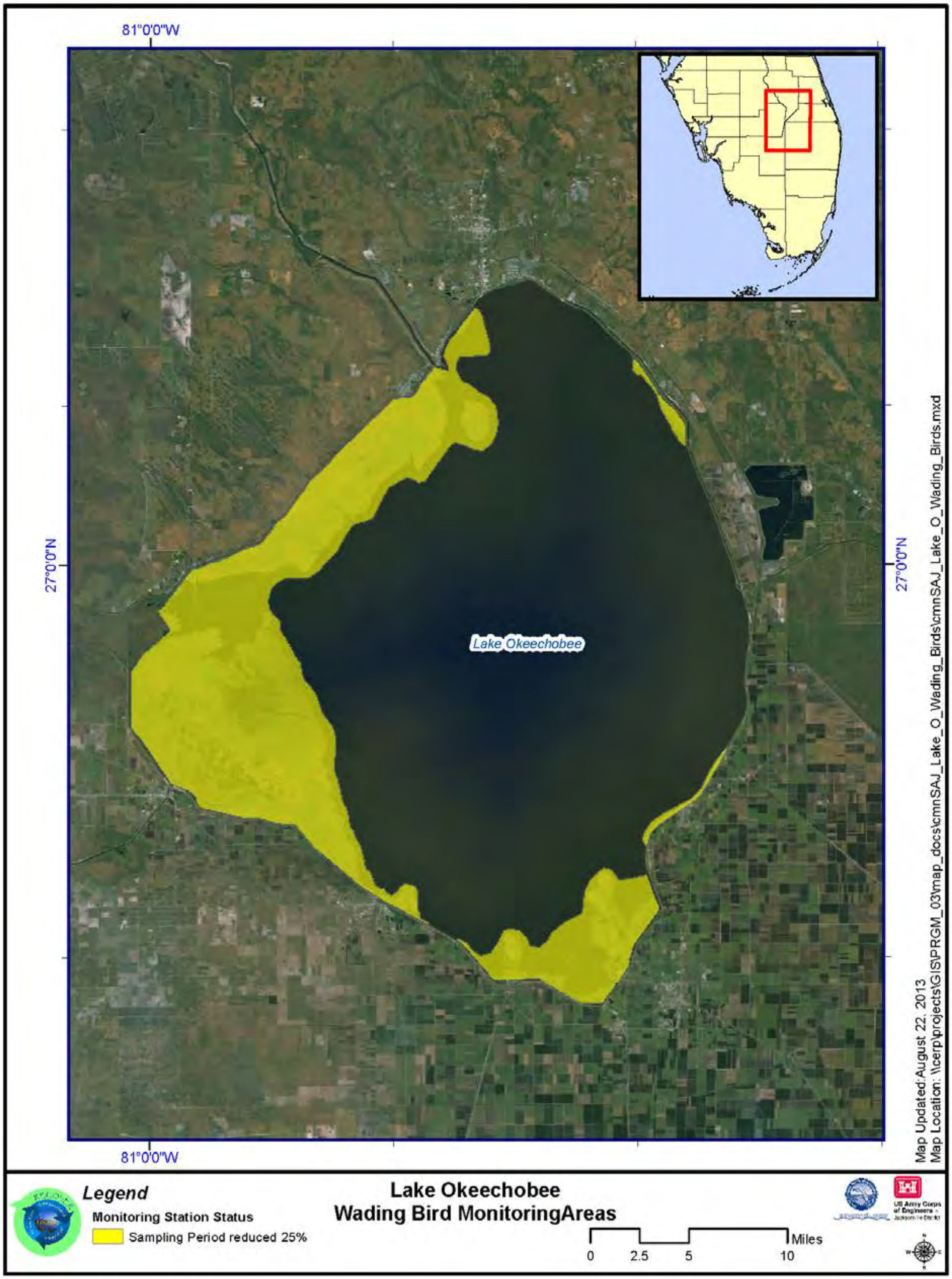

Figure A3-2-14. Map of Lake Okeechobee Wading Bird Monitoring.

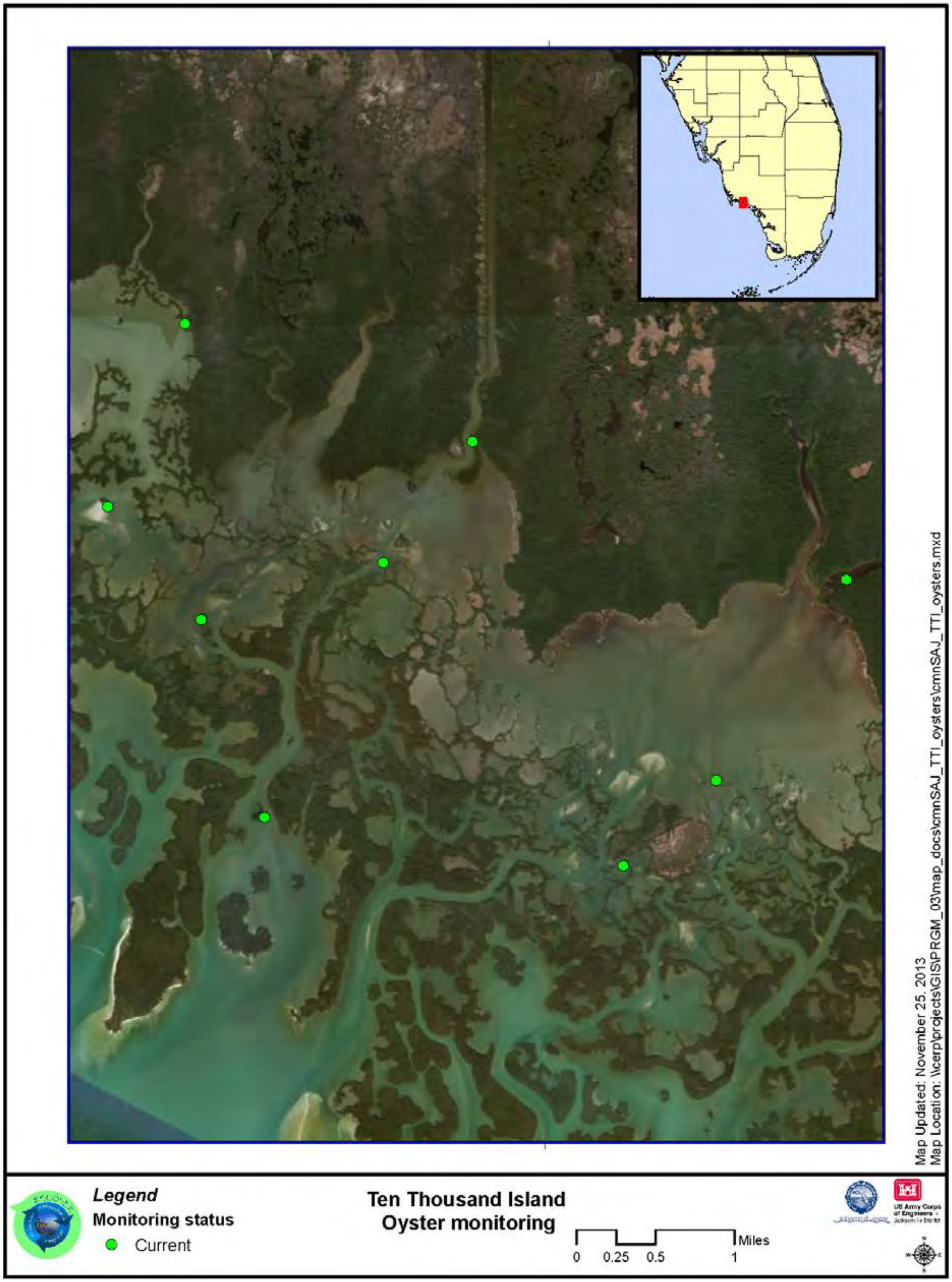

Figure A3-2-15. Map for Monitoring of Oysters in the Ten Thousand Islands.

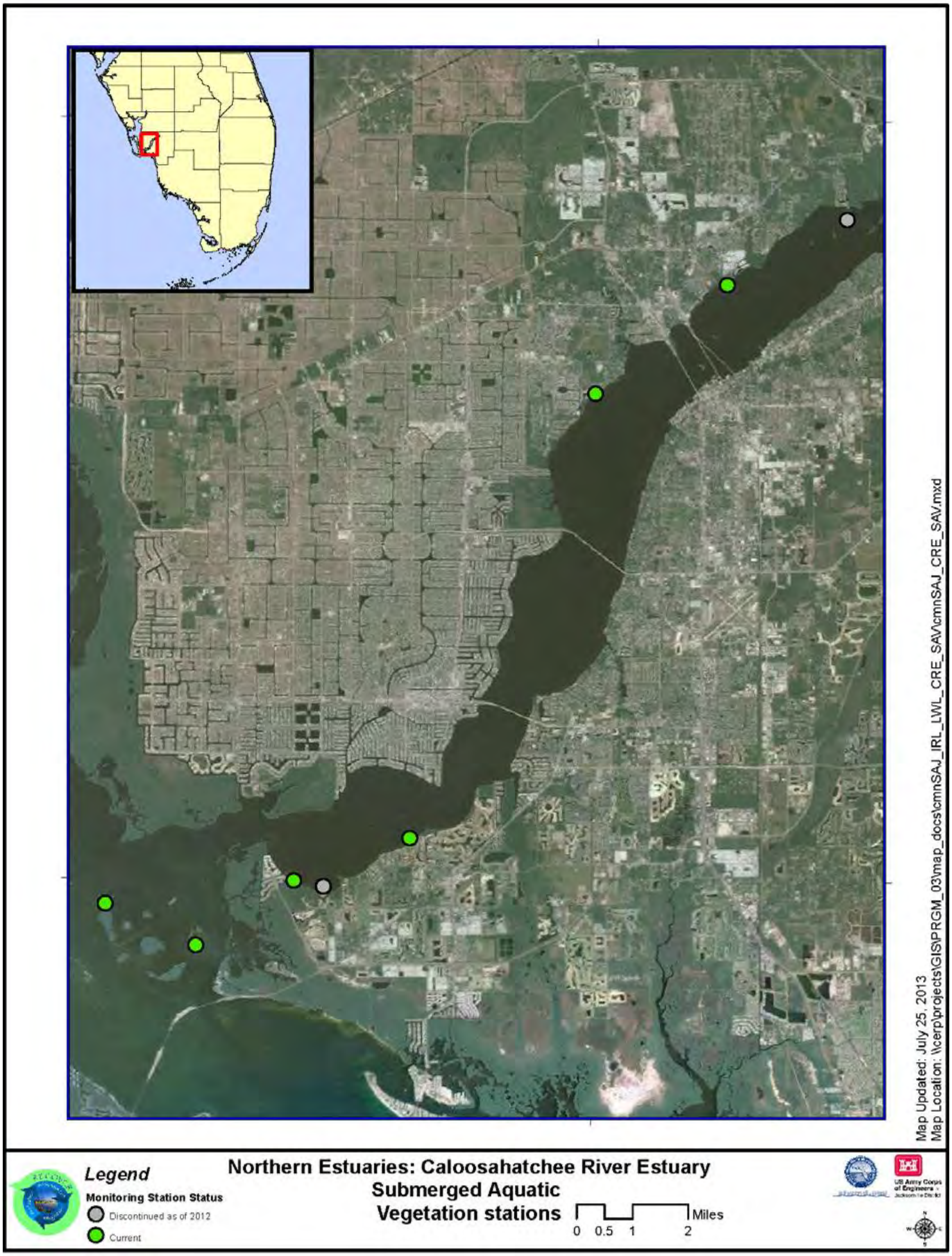

Figure A3-2-16. One of four monitoring maps for Submerged Aquatic Vegetation.

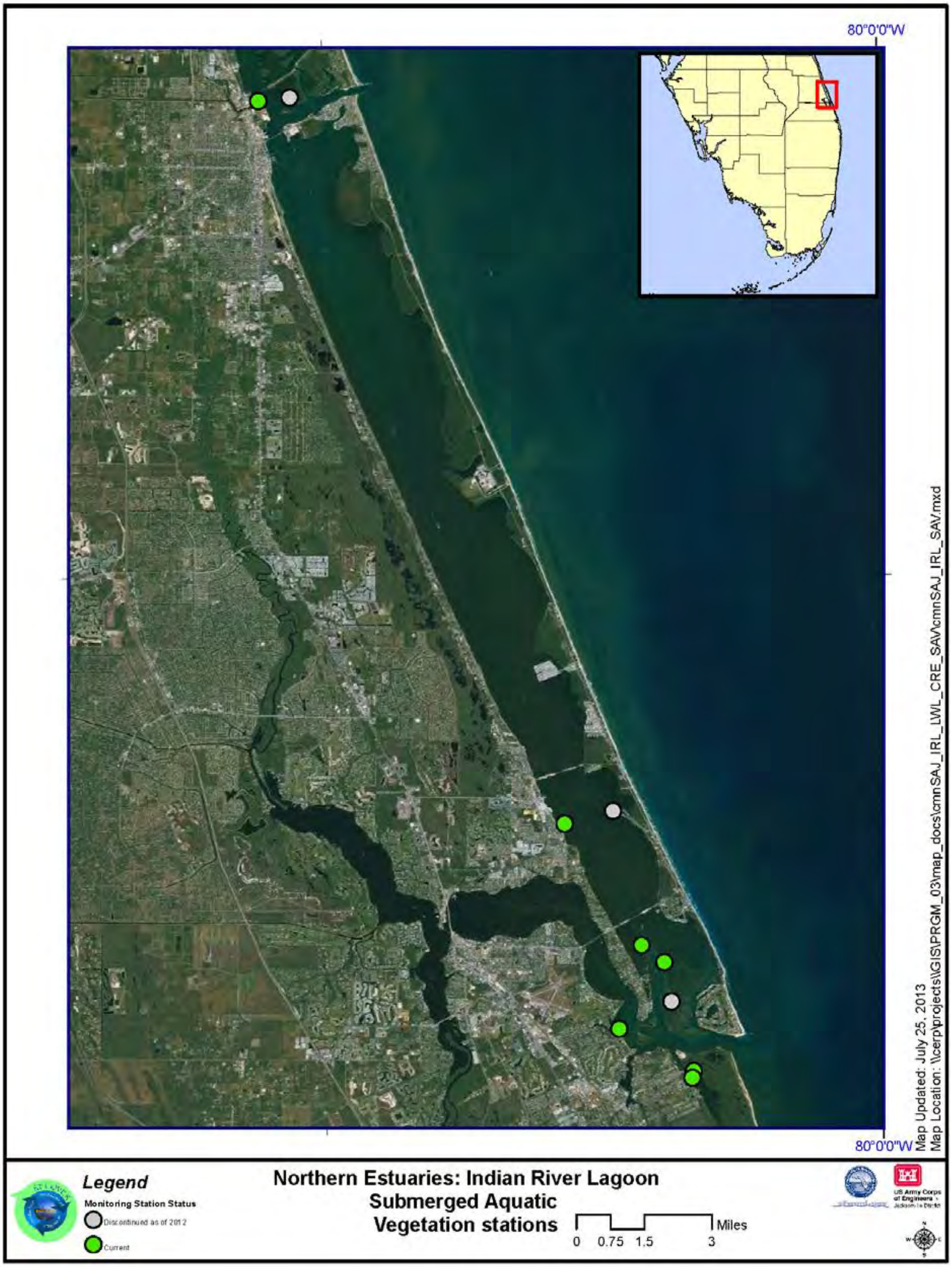

Figure A3-2-17. Two of four monitoring maps for Submerged Aquatic Vegetation.

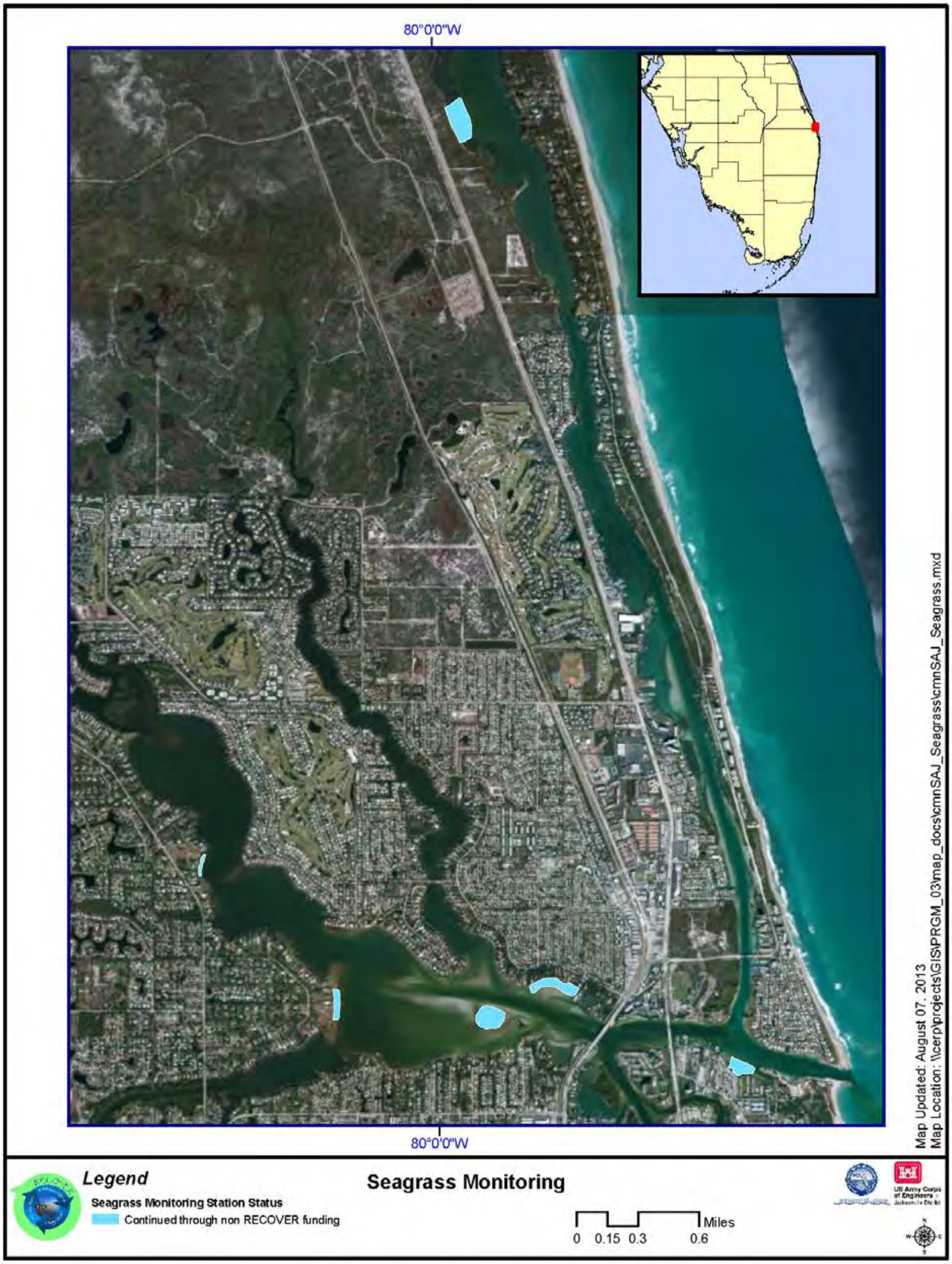

Figure A3-2-18. Three of four monitoring maps for Submerged Aquatic Vegetation. This map shows monitoring sites in the Loxahatchee River Estuary.

Figure A3-2-19. Four of four monitoring maps for Submerged Aquatic Vegetation.

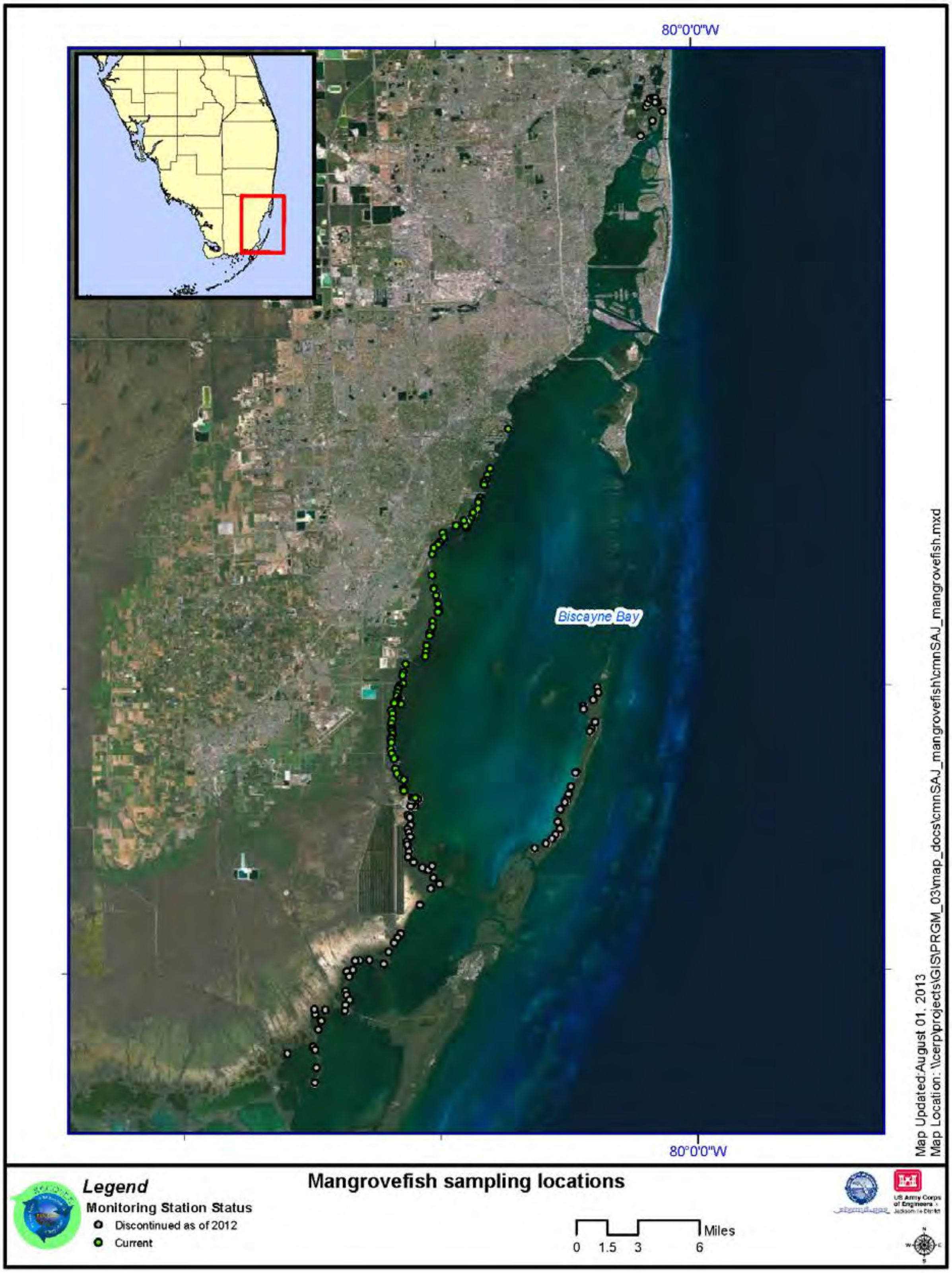

Figure A3-2-20. Map of Biscayne Bay Mangrove Fish Monitoring.

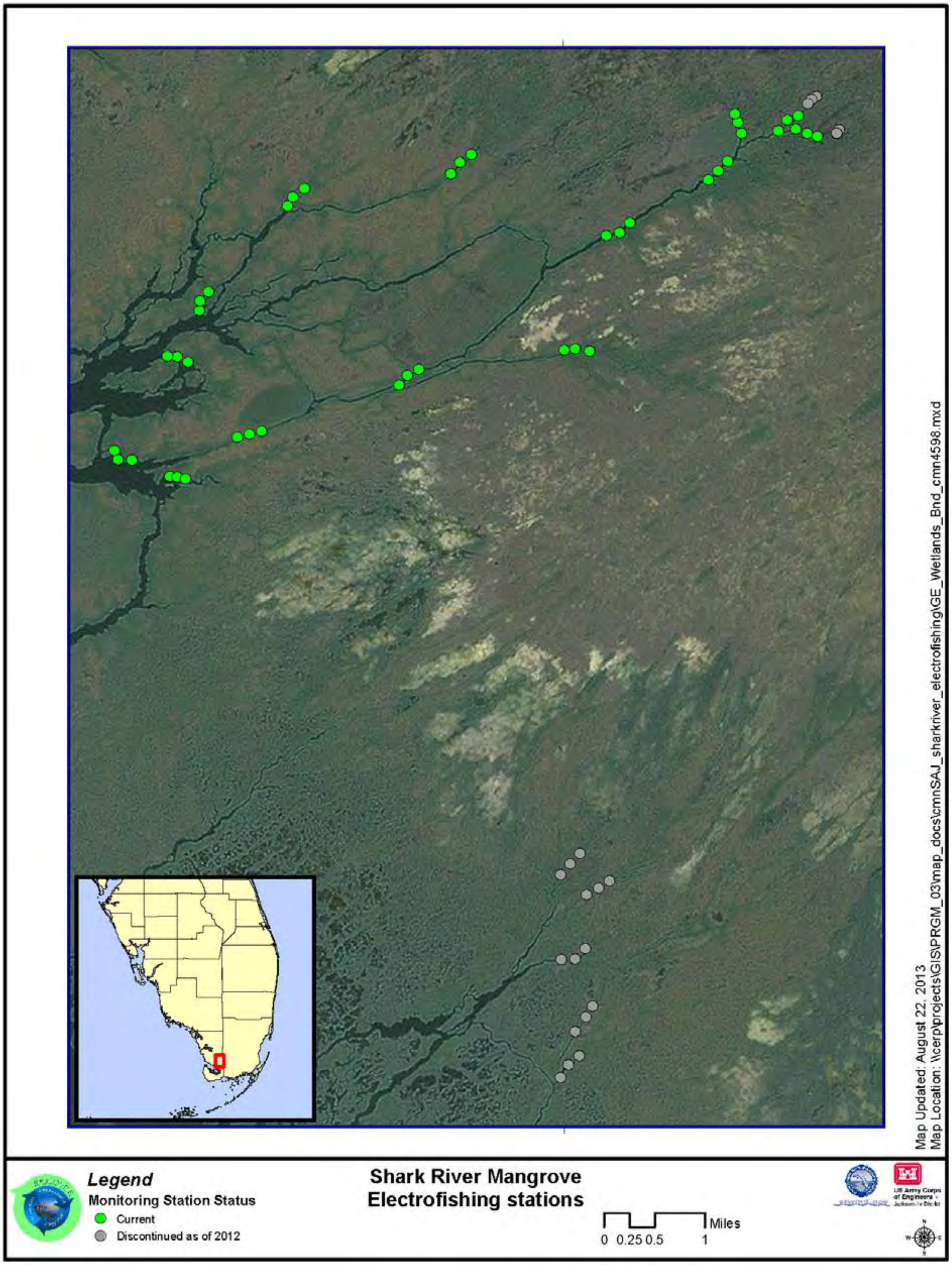

Figure A3-2-21. Map of Monitoring of Marsh Mangrove Fishes.

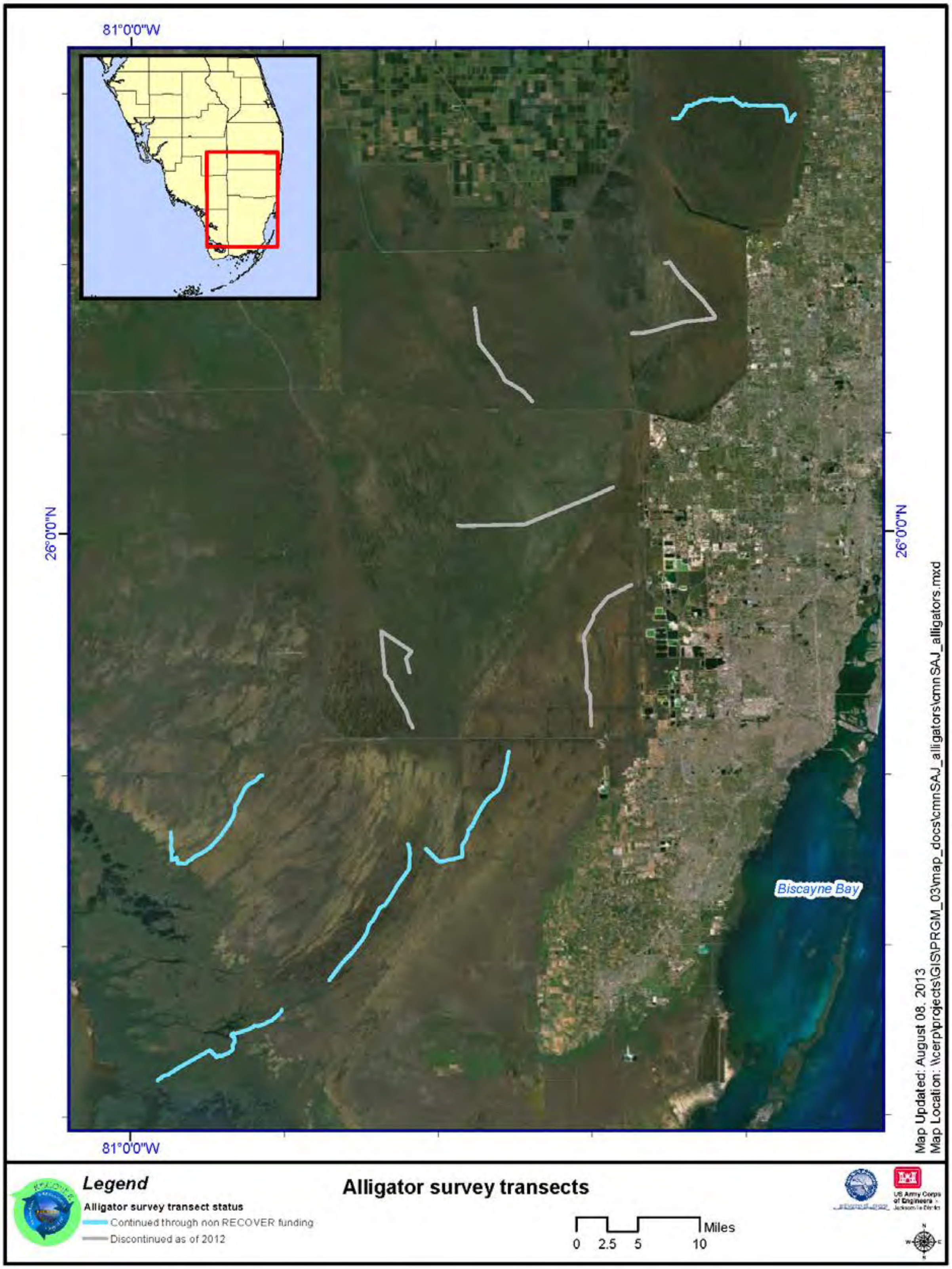

Figure A3-2-22. One of three maps for Alligator and Crocodile Monitoring.

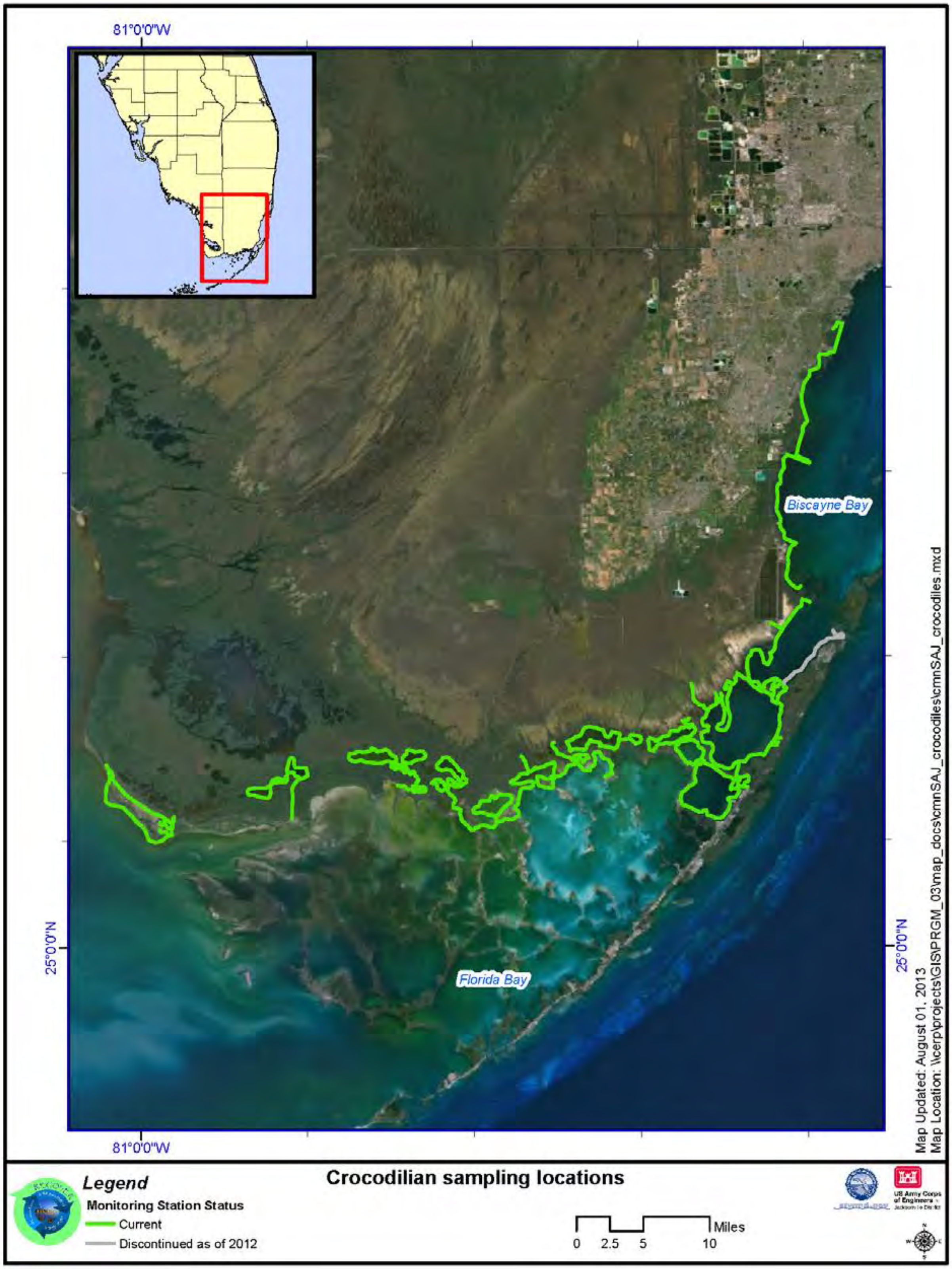

Figure A3-2-23. Two of three maps for Alligator and Crocodile Monitoring.

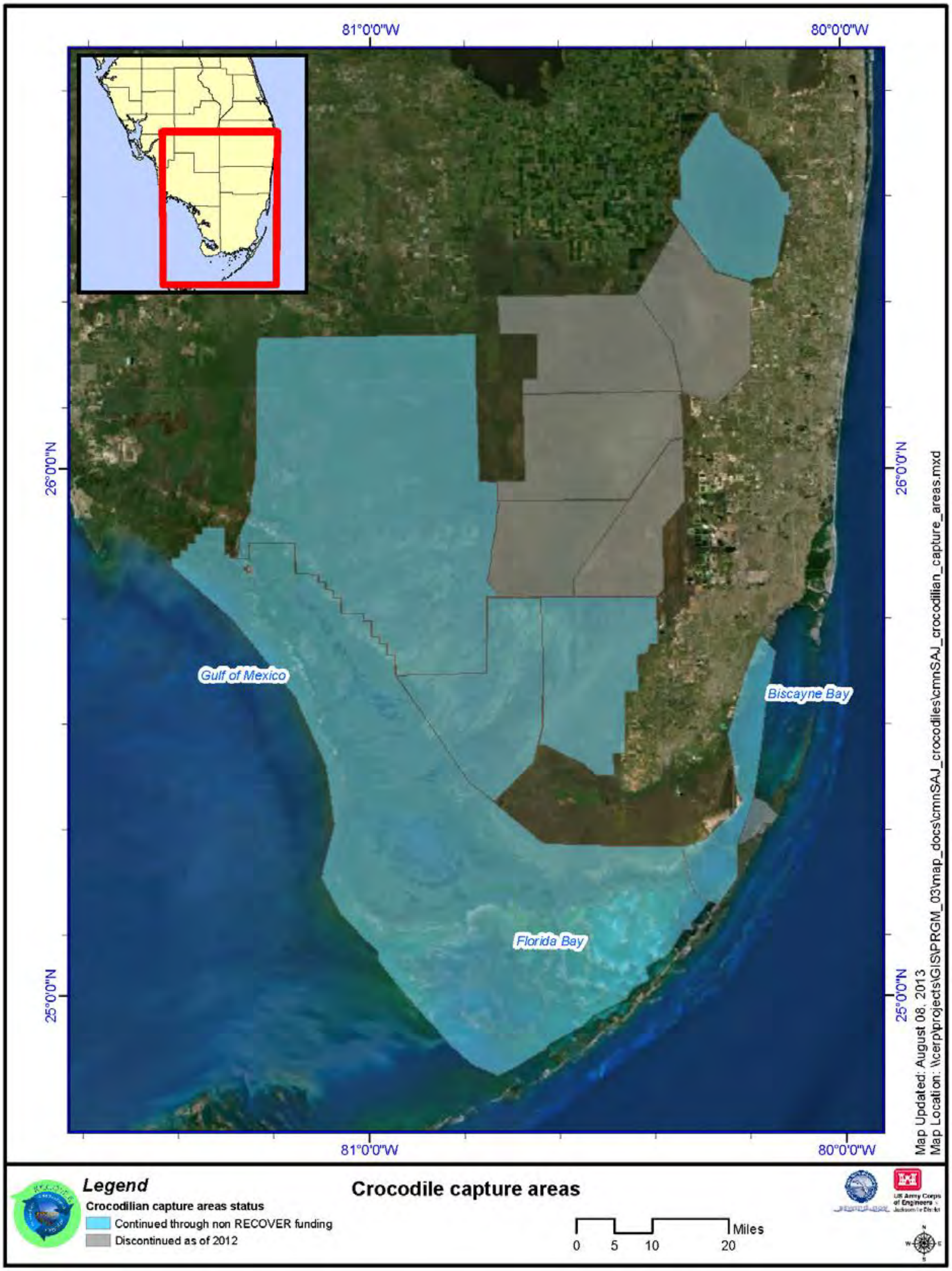

Figure A3-2-24. Three of three maps for Alligator and Crocodile Monitoring.

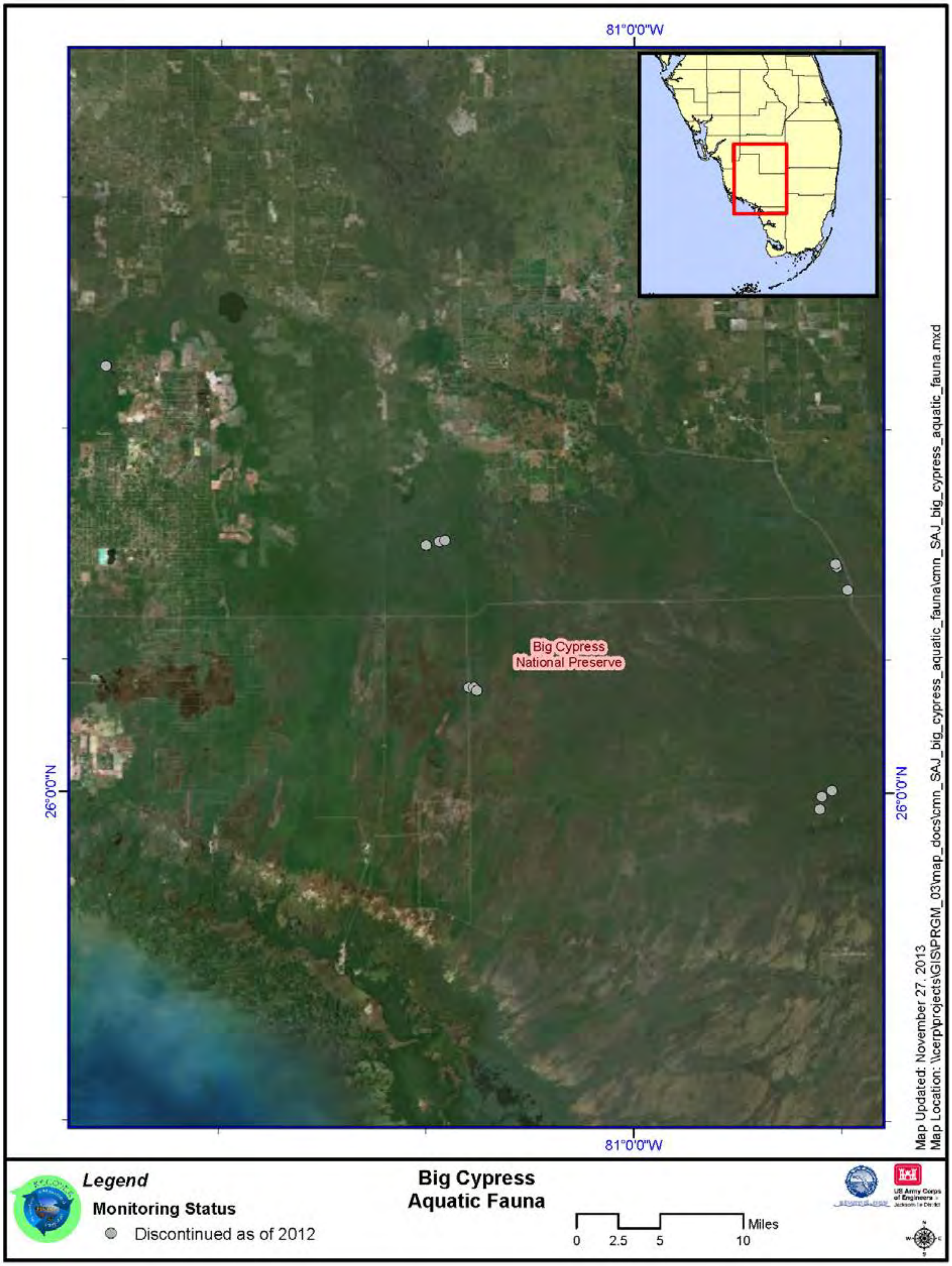

Figure A3-2-25. Map for Monitoring Aquatic Fauna in Big Cypress.

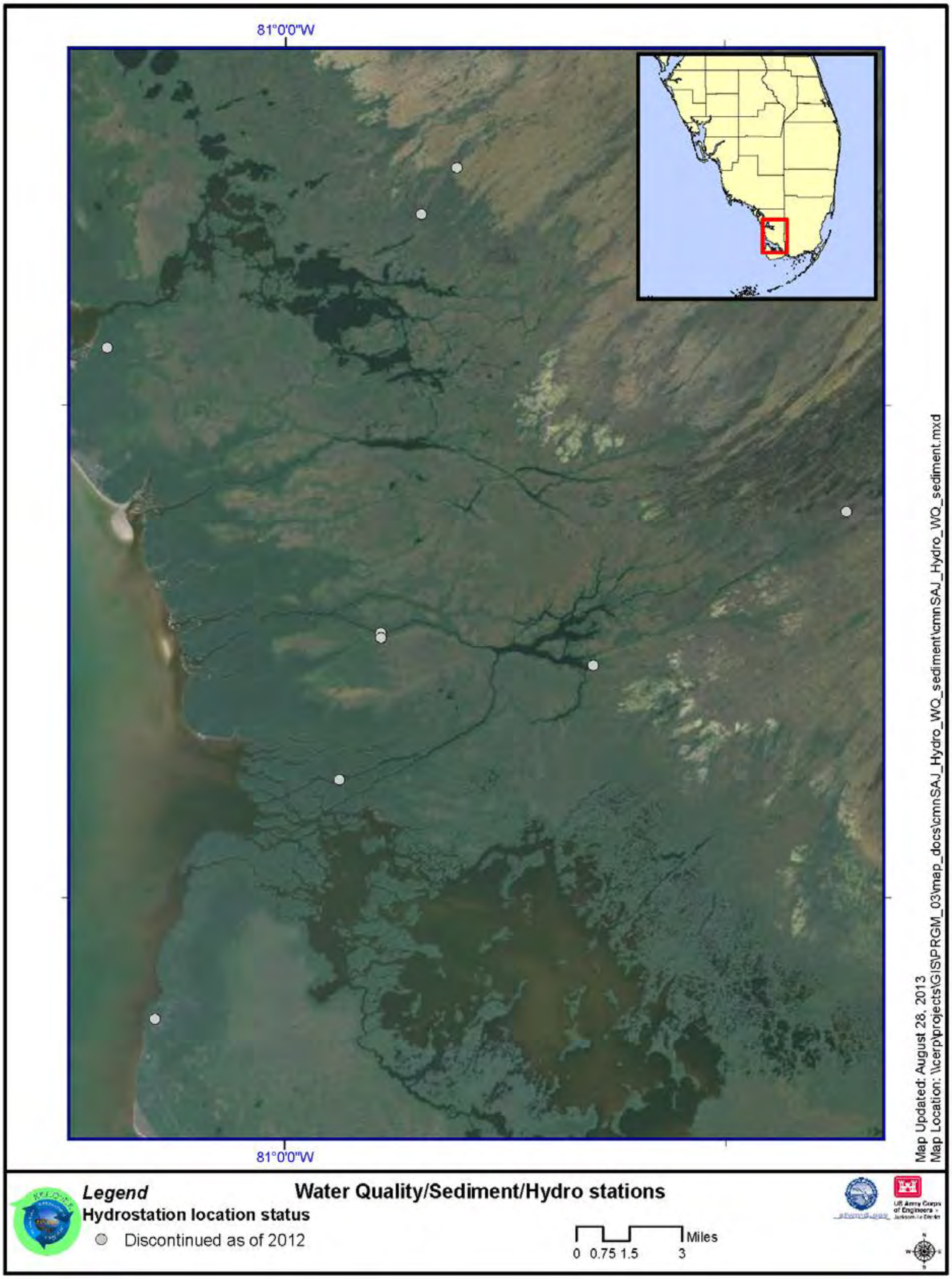

Figure A3-2-26. Monitoring maps for Sediment Elevation Tables.

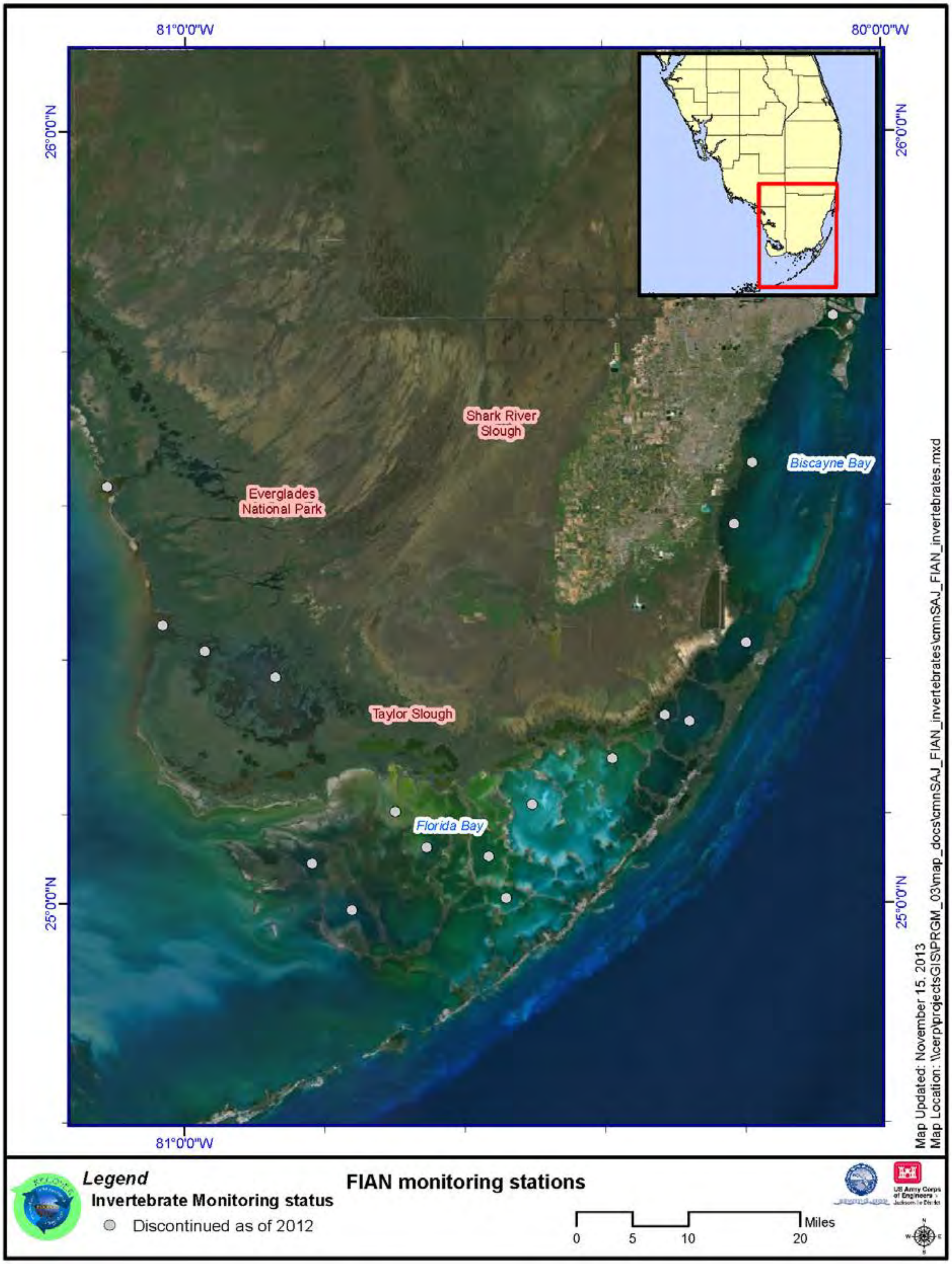

Figure A3-2-27. Monitoring map for FIAN.

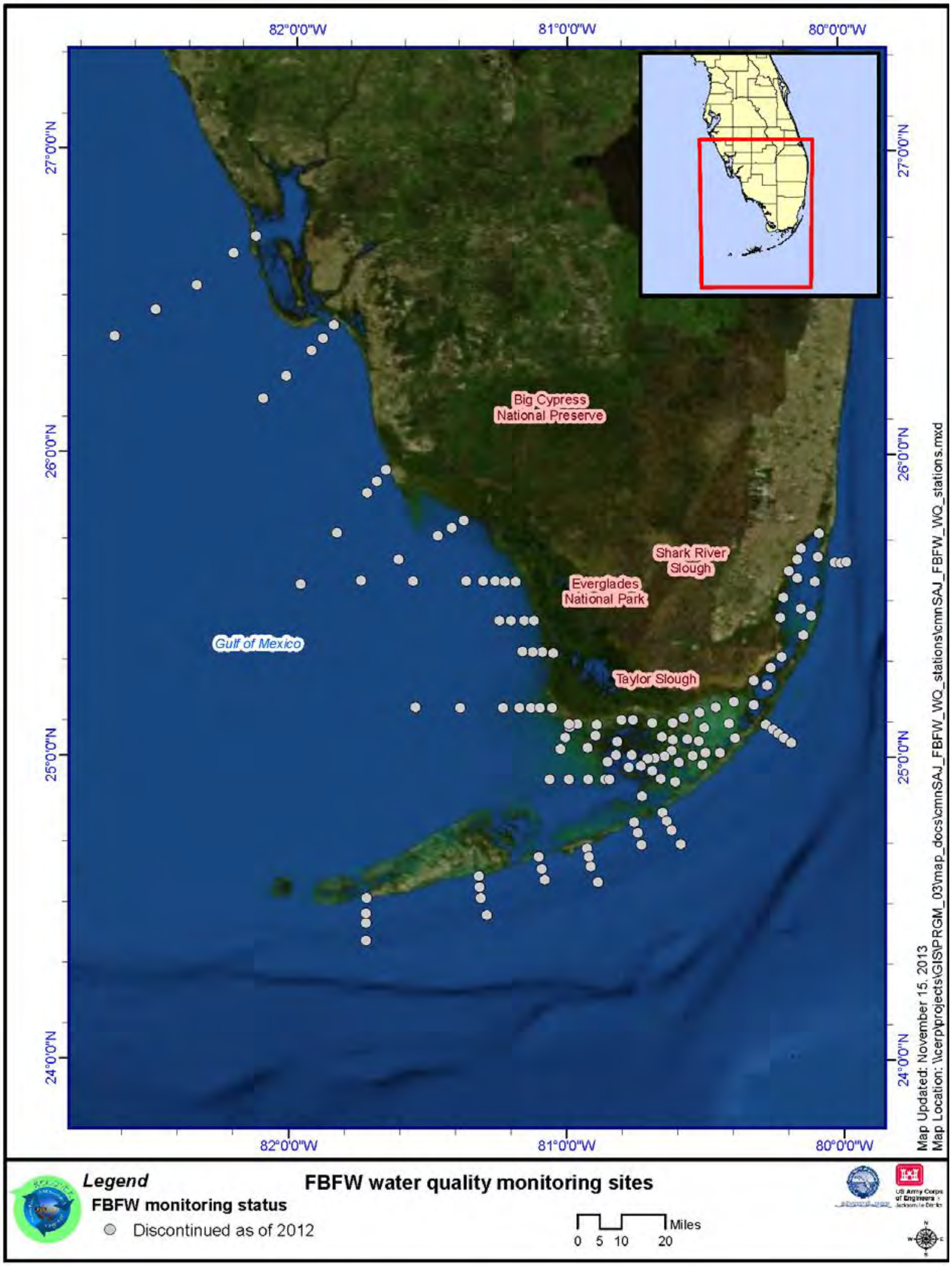

Figure A3-2-28. Map for Water Quality, Salinity & Circulation Monitoring.

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CHAPTER 4

NORTHERN ESTUARIES

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TABLE OF CONTENTS

CHAPTER 4 NORTHERN ESTUARIES

INTRODUCTION

The Northern Estuaries include the Caloosahatchee River Estuary (CRE) on the west coast of Florida and the St. Lucie Estuary (SLE), Southern Indian River Lagoon (SIRL), Loxahatchee River Estuary (LRE), and Lake Worth Lagoon (LWL) on the east coast (see RECOVER [2007] for descriptions). Historically, natural freshwater discharges into these water bodies sustained an appropriate range of salinity conditions, which supported healthy plant and animal communities. However, freshwater inflows to all the Northern Estuaries have been altered from the pre-development state. The particular details of landscape changes in the contributing watersheds may be different, but the major results have been fairly consistent: in general, flows are more variable and extreme, with greater high volume flows and lesser low volume flows. In estuarine receiving waters, this higher variability in freshwater inflow results in a concomitant increase in salinity variability and extremes that degrade and damage existing plant and animal communities.

Because the general effect of altered freshwater inflows on salinity in estuarine receiving waters is similar among the Northern Estuaries, so is the major hypothesized outcome of Comprehensive Everglades Restoration Plan (CERP) implementation. The construction and operation of CERP projects will help regulate freshwater inflows that will in-turn provide salinity envelopes that avoid damaging high and low salinity extremes (RECOVER 2007).

The Restoration Coordination and Verification Program (RECOVER) has monitored the responses of estuarine ecological indicators under a wide range of climatic conditions and freshwater inflows over the past ten water years (Water Year [WY] 2004–WY 2013; a water year begins on May 1 and ends on April 30 of the following year). For example WY 2006 was very wet and included Hurricane Wilma in October 2005. By contrast WY 2007 was extremely dry. This rich and comprehensive set of observations allows assessment of resilience to high freshwater inflows and recovery from the damage that they cause. The high salinity events associated with droughts are also extremely informative and have elicited unanticipated responses in ecological indicators. Lastly, through the responses of ecological indicators, the efficacy of target salinity envelopes for the Northern Estuaries can now be evaluated as detailed in this chapter.

In the Northern Estuaries, monitoring focuses on oysters and submerged aquatic vegetation (SAV) (primarily marine seagrasses but includes some freshwater species as well) because these are prominent features of the estuarine landscape in south Florida and provide critical structural habitat for other estuarine organisms. Being stationary, they integrate conditions in the overlying water over time and make excellent indicators of ecological conditions. In the SLE and adjoining SIRL benthic infauna (e.g., clams and worms) are also sampled. This group of organisms has also proven to be a valuable indicator of environmental change. Monitoring by RECOVER provides a continuous record of ecological conditions in the Northern Estuaries. This data helps resource managers and other interested parties track the health of these systems and their responses to both natural and anthropogenic perturbations as well as to specific water management actions.

CALOOSAHATCHEE RIVER ESTUARY

Water quality in the CRE is dependent on surface water inputs, especially from the S-79 water control structure, which supplies water from the eastern watershed and Lake Okeechobee and acts as a salinity barrier. Since the historic watershed area was smaller and the river was not connected to Lake Okeechobee and its watershed (Flaig and Capece 1998), the maximum flow was significantly smaller. Now, flows at S-79 reach over 20,000 cubic feet per second (cfs). At 5,000 cfs, the estuary is completely fresh down to the mouth of the river. When flows from S-79 are below 300 cfs, salt water extends up river and the water is clearer principally due to reductions in colored dissolved organic matter. Changes in aquatic resources in the CRE were in response to reduced rainfall in WY 2011 and early WY 2012. The total amount of fresh water entering the estuary was less than the long-term average and particularly reduced in WY 2012, when there were no discharges from Lake Okeechobee. In turn, reduced freshwater input drove salinity increases during the WY 2011 dry season and the beginning of WY 2012. In response to increased salinity, numbers of eastern oysters (*Crassostrea virginica*) increased as did salt-tolerant seagrasses such as shoal grass (*Halodule wrightii*) and manatee grass (*Syringodium filiforme*). Total nitrogen (TN) and total phosphorus (TP) loadings to the CRE were less than the long-term averages, respectively, thereby reflecting reduced basin runoff and inflows from Lake Okeechobee. However, a significant algal bloom occurred upstream of S-79 in the CRE in May–June 2011 (WY 2012). This bloom was associated with a period of greatly reduced freshwater inflow and high seasonal temperatures. Additional water quality information can be found in Chapter 10 of the *2013 South Florida Environmental Report – Volume I*, which can be viewed at www.sfwmd.gov/sfer.

Caloosahatchee River Estuary Oysters

Introduction and Background

The eastern oyster is ecologically important because it improves water quality by filtering particles from the water; an individual oyster can filter 4 to 34 liters of water per hour, removing phytoplankton, particulate organic carbon, sediments, pollutants and microorganisms from the water column. Oyster bars provide habitat for numerous organisms and several studies have demonstrated the species richness of oyster bars (Wells 1961, Bahr and Lanier 1981, Grabowski and Peterson 2007). Oysters serve as an excellent indicator species because salinity conditions suitable for oysters also produce optimal conditions for a suite of other desirable estuarine organisms. In addition, given their sedentary nature, it is easy to determine cause-and-effect relationships between the water quality and health of these organisms, hence their use in the International Mussel Watch Program and the National Oceanic and Atmospheric Administration's Status and Trends Program. Due to limited funding for monitoring, the oyster makes an ideal candidate for a performance measure that can be used in all phases of adaptive management. The oyster has been used by CERP project planners to help select the best plan alternative and it provides an interim goal by which restoration progress can be predicated and then monitored (RECOVER 2005). The monitoring and assessment process can then feed back into operational decisions. The RECOVER Northern Estuaries oyster habitat performance measure can be found at www.evergladesplan.org/pm/recover/recover_docs/et/ne_pm_oysterhabitat.pdf.

CERP implementation and other restoration projects should restore more natural freshwater inflows by retention in reservoirs and stormwater treatment areas, wetland rehydration, and changing delivery patterns; removal of muck; and introduction of artificial substrate into south Florida estuaries. Implementation of these should provide beneficial salinity and habitat conditions that will promote the reestablishment of healthy oyster beds.

Figure 4-1 is a conceptual model of stressors that impact oysters and thus oyster reef and secondary habitat. The working hypotheses for oysters are discussed in detail in the *Monitoring and Assessment Plan (MAP), Part 2 2006 Assessment Strategy for the MAP* (RECOVER 2006) and the *2007 System Status Report (*SSR) (RECOVER 2007b). These documents can be accessed at www.evergladesplan.org/pm/recover/assess_team.aspx, In **Figure 4-1**, boxes with dashed lines (sediment, substrate and food) are not currently being measured. However, depending on the need and the model output, these factors may be included in future monitoring.

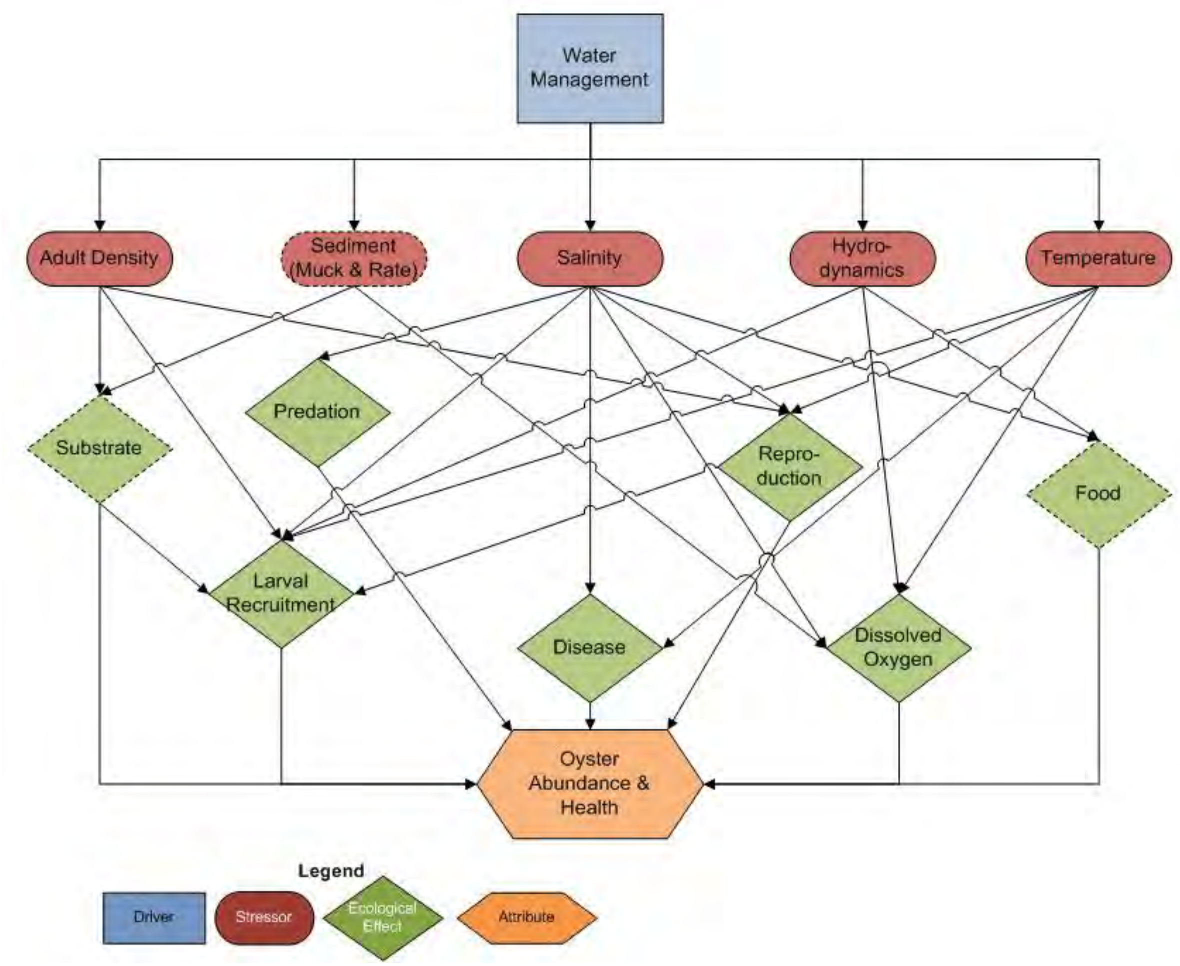


Figure 4-1. Conceptual model for oysters in the Northern Estuaries.

Study Area

Oysters are monitored in the CRE on the west coast of Florida at the sites shown in (**Figure 4-2**).

Monitoring

Five aspects of oyster ecology are being monitored: density of adult oysters, reproduction and recruitment, juvenile oyster growth and survival, physiological condition as measured by condition index, and the distribution and frequency patterns of the oyster diseases *Perkinsus marinus* (dermo). Monthly water quality (physio-chemical parameters such as dissolved oxygen, specific conductance, salinity, pH and temperature) sampling is conducted in conjunction with field sampling at each study site. For details on monitoring techniques please see the *CERP Monitoring and Assessment Plan: Part 1 Monitoring and Supporting Research* (RECOVER 2004) available at www.evergladesplan.org/pm/recover/recover_map_2004.aspx and the 2007 SSR (RECOVER 2007b) available at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

Methodology

Since oyster response data were available since WY 2001, all available data (2001–2013) was used to determine how freshwater inflows and seasonal changes influenced oyster responses described below. Methodology of various sampling protocols is detailed in the 2009 SSR (RECOVER 2011).

Due to Fiscal Year 2012 (October 1, 2011–September 30, 2012) RECOVER monitoring budget reductions, funding in the CRE was reduced. Based on data mining of the first ten years of oyster monitoring, the program was modified to best achieve the objectives. For example, it was determined that Tarpon Bay, one of the current sampling stations, would be eliminated since data from it and the closest station, Kitchel Key, were providing similar results. In addition, given the location of Tarpon Bay inside a bay with a narrow opening and its own freshwater source, it may not reflect a distance-gradient from the S-79 structure in the CRE. For that reason, oyster responses from 2011 onwards only include those from Pepper Tree Point (spat recruitment only), Iona Cove, Bird Island and Kitchel Key. Due to funding constraints, sampling at Cattle Dock location was discontinued in 2010.

Analyses of freshwater inflows into the CRE at the S-79 structure and salinities at various oyster sampling locations in the CRE suggest that the stations near S-79 tend to be more controlled by flow from the S-79 structure (**Figure 4-3**). Other factors such as watershed runoff from the tidal basin and tidal influence tend to be important at sites further downstream of the structure.

Mean daily flow, mean 30-day moving average of daily flow, and mean salinity for wet and dry seasons at the S-79 structure were calculated.

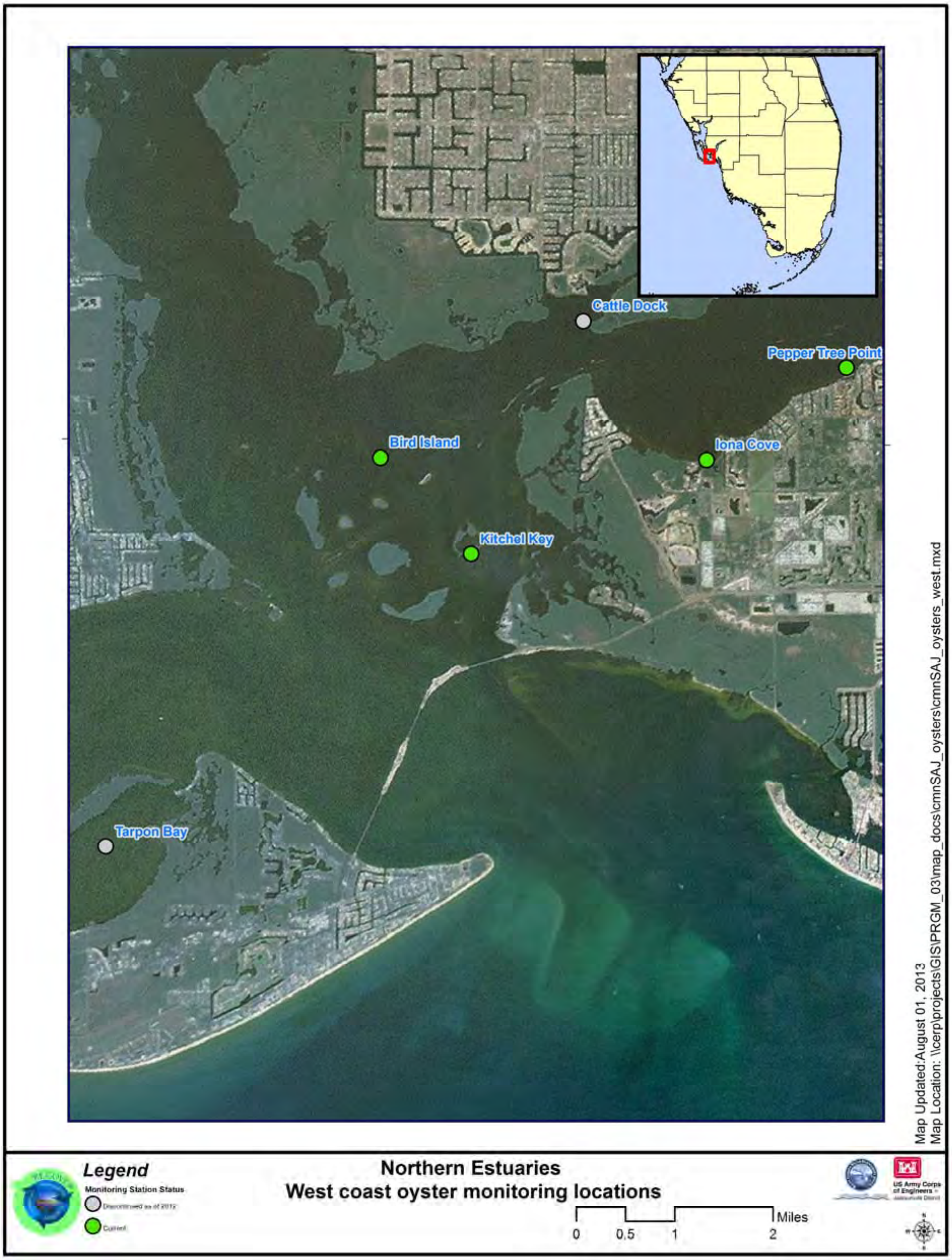


Figure 4-2. Oyster monitoring sites in CRE. Green symbols denote current oyster monitoring locations. White symbols denote discontinued oyster sampling locations.

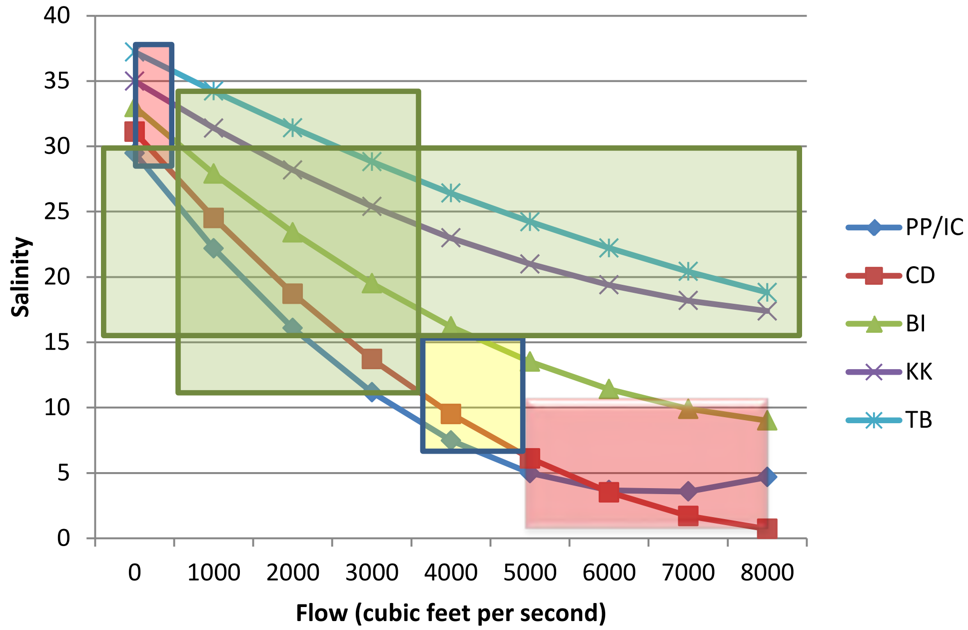


Figure 4-3. Relationship between monthly mean of the 30-day moving average freshwater inflows at the S-79 structure and salinity at various oyster sampling locations taken during monthly oyster sampling in the CRE. Green color indicates flows that would yield salinities that are favorable to juvenile and adult oysters. Yellow denotes "caution" or potential harm. Red denotes severe mortality at these flows/salinities. Salinities between 15 and 30 are favorable for oyster growth and survival. Intersection of the green boxes represents ideal salinity values for oysters in the Caloosahatchee Estuary. (Note: PP/IC – Pepper Tree Point/Iona Cove, CD – Cattle Dock, BI – Bird Island, KK – Kitchel Key, and TB – Tarpon Bay.)

It was further suggested that analyses using field observational data be augmented with data from controlled mesocosm experiments to determine how salinity affects survival and growth of various early life stages of oysters. In a controlled experiment, a factor, such as salinity, can be varied while holding other factors fixed. This leads to stronger cause-and-effect relationships and better estimates of optimal conditions. This also helps with variable selection in the models and leads to improved management decisions. Laboratory mesocosm experiments included exposure-response curves describing the effect of salinity on reproduction and development of early life stages of oysters, and effect of acute and gradual salinity changes on larval and juvenile oyster survival. Laboratory mesocosm studies were conducted to define exposure curves describing effects of flow and salinity, controlling for other factors. These studies are useful for corroboration of the results of the field observations and contribute to stronger causal models.

Results

Dry, Moderate, and Wet Years

Water years are classified as either wet, dry or normal based on salinities at Cape Coral Bridge in the CRE. Salinity values of <16, ≥ 16 to ≤ 28, and > 28 provided criteria for classifying water years as wet, normal, or dry years, respectively. In addition, salinity values during oyster sampling were also considered while determining wet, dry and normal years. Based on this criterion, water years 2003, 2004, 2005 and 2006 were wet, water years 2002, 2007, 2010, 2011 and 2013 were normal, and water years 2008, 2009 and 2012 were dry years (**Table 4-1**).

Table 4-1. Summary of salinity at three locations in the CRE from WY 2003 to WY 2013. Locations were Cape Coral, Shell Point, and the Sanibel Bridge. Salinity values of < 16, ≥ 16 to ≤ 28, and > 28 come from the performance measures for the CRE.

Cape Coral			Days Salinity < 16		Days Salinity ≥ 16 and ≤ 28		Days Salinity > 28	
Water Year	Average Salinity for All Observations	Total Number of Daily Observations	Number	%	Number	%	Number	%
2003	3.8	101	101	100.0	0	0.0	0	0.0
2004	6.5	272	240	88.2	32	11.8	0	0.0
2005	10.7	364	242	66.5	122	33.5	0	0.0
2006	6.6	365	330	90.4	35	9.6	0	0.0
2007	17.6	323	117	36.2	195	60.4	11	3.4
2008	25.7	322	9	2.8	206	64.0	107	33.2
2009	17.6	365	121	33.2	185	50.7	59	16.2
2010	13.2	365	196	53.7	169	46.3	0	0.0
2011	13.4	315	150	47.6	163	51.7	2	0.6
2012	17.7	343	132	38.5	174	50.7	37	10.8
2013	13.9	345	198	57.4	146	42.3	1	0.3
Total	**13.3**	**316**	**167**	**55.9**	**130**	**38.3**	**20**	**5.9**

Salinity

Based on previous studies and available information, the following salinity targets were suggested for the CRE to maintain and enhance valued ecosystem components such as oysters. Mean monthly flows at S-79 should be maintained between 450 and 2,800 cfs, with approximately 75% of flows from S-79 between 450 and 800 with most of the remainder in the 800 to 2,800 cfs range. The CERP systemwide performance measure for Northern Estuaries salinity envelopes does not specify a salinity envelope at a particular location in the CRE (RECOVER 2007). Rather, the document refers to generalized beneficial salinity conditions and ranges. For the area around Shell Point and San Carlos Bay, a flow of 500–2,000 cfs results in salinities of ~16–28 at all stations, conditions that are favorable to sustain and enhance CRE oyster populations. These results are confirmed by laboratory experiments. In general, flows between 500 and 3,500 cfs will yield salinities between 24 and ~12. The lower end of this range is tolerable to adult oysters, while stressful to juvenile oysters.

Condition Index

The oyster condition index measures the relative amount of flesh in an oyster as the ratio of the weight of the meat to the weight of the shell. It is used as an overall indicator of the health and vitality of the oyster. Oyster condition index typically increases with increasing salinity downstream. Condition index also varies with season, especially reproduction, which is highly controlled by seasonal temperature. There was a significant interaction ($F_{8,\ 405}$=2.537, P < 0.025) between type of year and station, driven by differences between upstream and downstream stations during wet years.

Condition index increased at the upstream locations (Iona Cove, Cattle Dock) as well as at the downstream location (Tarpon Bay) during wet years. There was no significant difference at the midstream locations (Bird Island and Kitchel Key) between year types (**Figure 4-4**). While there were significant differences in overall condition index between sites ($F_{4,\ 8.57}$=10.721, P=0.001), the increase in condition index during wet years compared with dry and normal years was only marginally significant

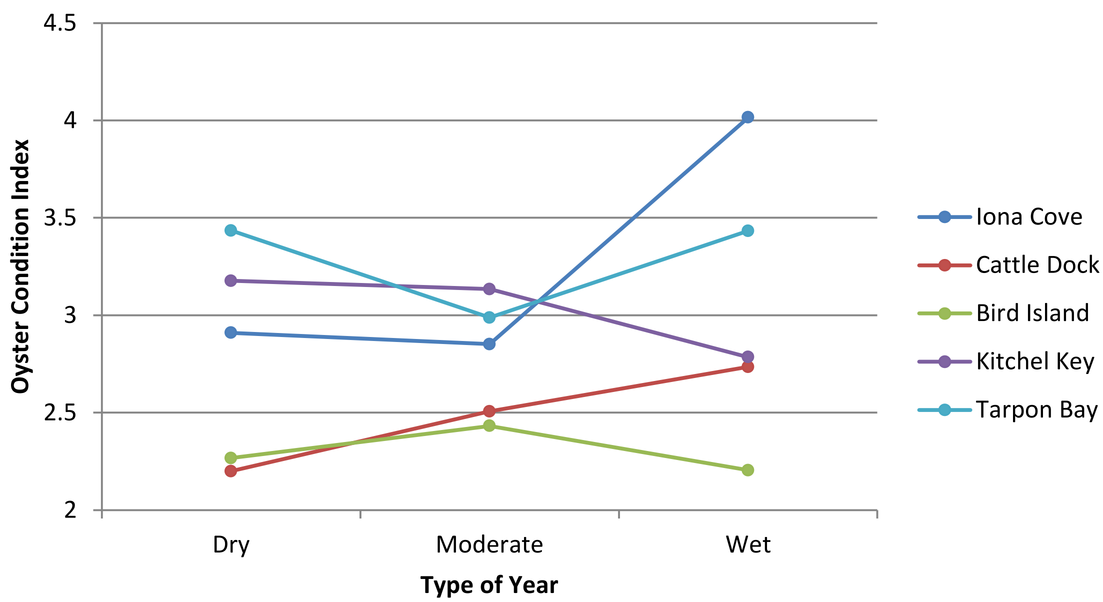


Figure 4-4. Mean condition index of oysters from all sampling locations in the CRE during the study period. Condition index increased at upstream (Iona Cove, Cattle Dock) and downstream (Tarpon Bay) locations, but showed little difference at mid-stream locations (Bird Island and Kitchel Key). At upstream stations (Iona Cove and Cattle Dock), mortality of oysters during the wet seasons of the wettest years resulted in incomplete data for those stations. The condition index values are higher because they are calculated using only the data from the remaining months of the wet years.

($F_{2,\ 0.763}$= 1.91, P=0.161), probably due the small sample sizes for wet and dry years (**Figure 4-5**). On the other hand, the apparent increase in condition during wet years at upstream sites is due to a sampling artifact. No living oysters were found at Iona Cove during the wet seasons of the wettest years, so condition index values for those years are based on samples taken during the rest of the year. This result is included only to explain the significant interaction and the reason the upstream sites should be excluded from this particular analysis.

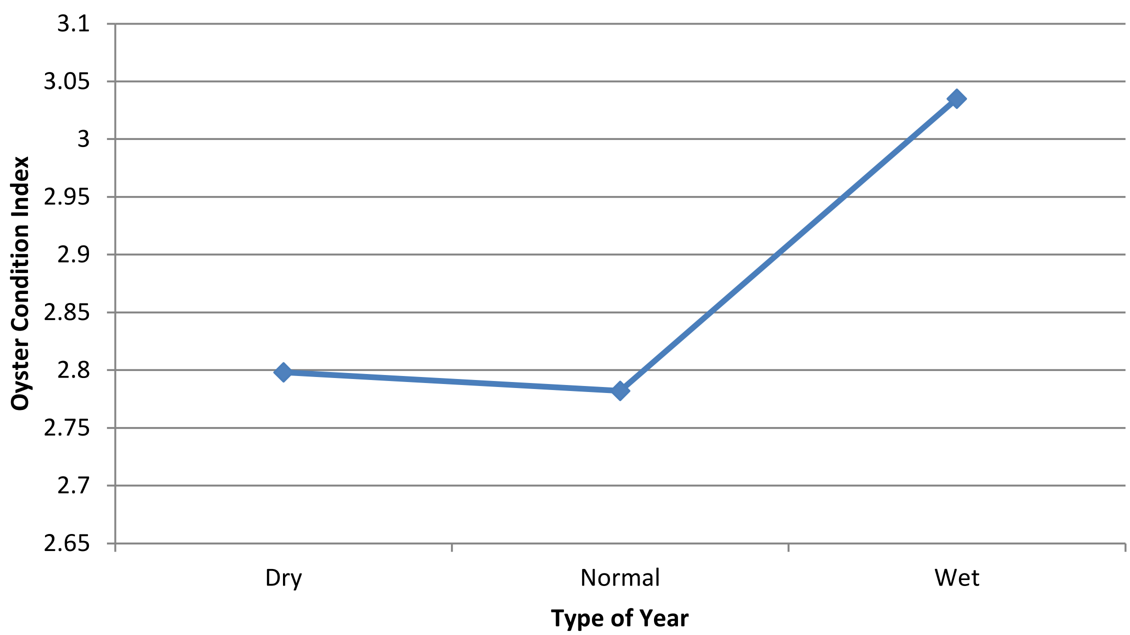


Figure 4-5. Mean condition index of all oysters from CRE during dry, normal (moderate), and wet years. Condition index slightly increased during wet years.

Disease

Disease prevalence and intensity from the parasitic protozoan *Perkinsus marinus* varied widely at all sampling locations during the study period ranging between 0 and 100%. While there were no site-related differences, type of year had a significant effect on disease prevalence ($F_{2,\ 4576.63}$=6.458, P=0.004), and there was no significant interaction between them. Prevalence was significantly higher in dry years than in moderate or wet years (**Figure 4-6**). Disease prevalance was higher during dry years at all stations in the CRE (**Figure 4-7**). While disease prevalence ranged between 0 and 100%, the mean infection intensity was relatively low with values < 1.5 (on a scale of 0 to 5) (**Figure 4-8**). A combination of freshwater releases and seasonal rainfall during warm summer months enabled the disease levels to be low. Similar to prevalence of *P. marinus*, infection intensity was significantly higher during dry years ($F_{2,1.21}$=0.67, P=0.035) compared to normal or wet years. Increased freshwater input into the estuary due to regulatory releases, as well as watershed runoff, contributed to low salinities and resulted in lower intensity and prevalence of *P. marinus* in oysters during wet years (**Figure 4-9**).

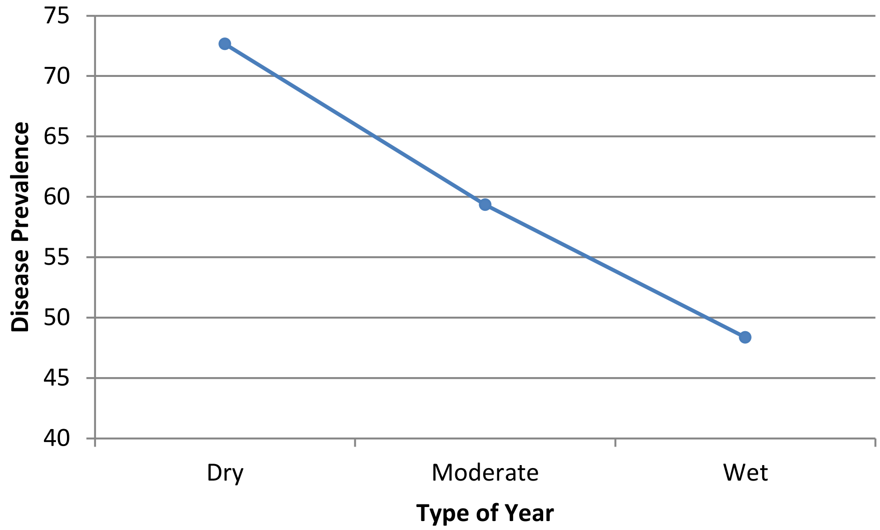


Figure 4-6. Mean prevalence of *Perkinsus marinus* infection in oysters from all sampling locations in the CRE. Disease prevalence is significantly higher during dry and moderate years compared to wet years when salinities tend to be low.

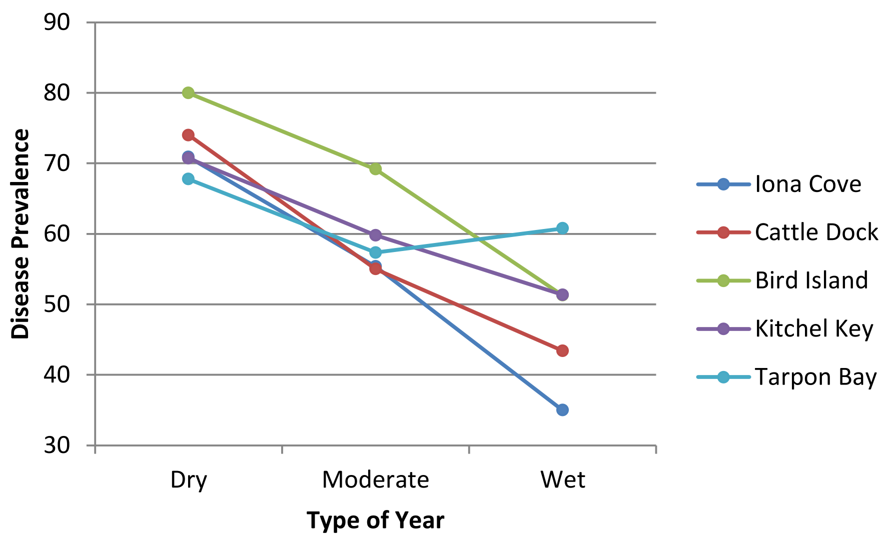


Figure 4-7. Mean prevalence of *Perkinsus marinus* in oysters from all sampling locations in the CRE during the study period. With the exception of Tarpon Bay, where salinities are always higher, oysters at all stations showed a decrease in *P. marinus* prevalence during wet years compared to dry and normal years.

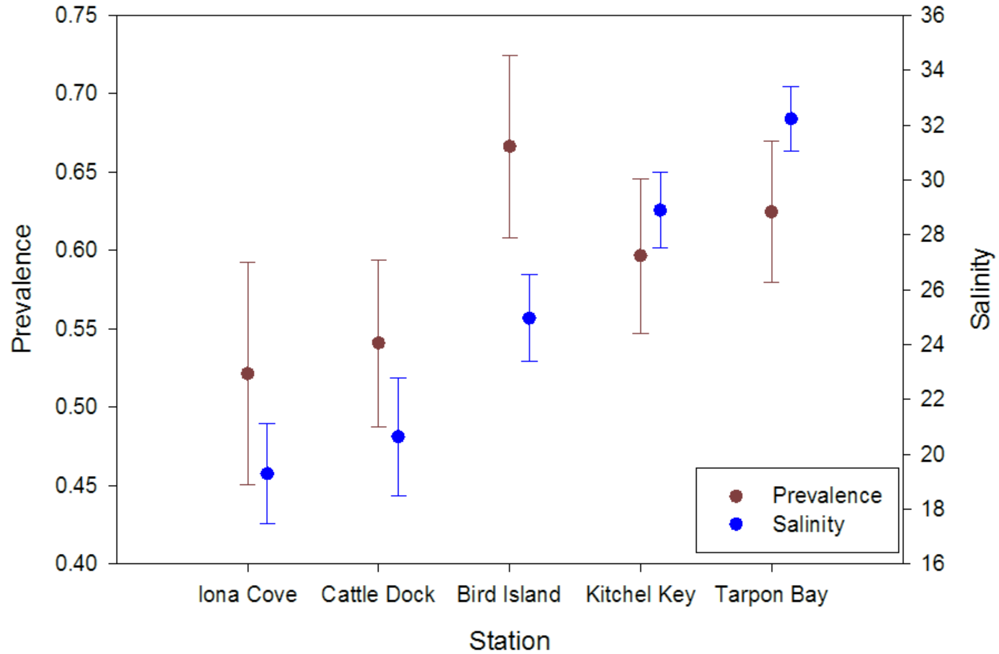


Figure 4-8. Mean disease prevalence of *Perkinsus marinus* in oysters during the study period from various sampling locations in the CRE. Stations Iona Cove, Cattle Dock, Bird Island, Kitchel Key and Tarpon Bay are from upstream to downstream. It appears that higher salinities result in higher disease prevalence in oysters.

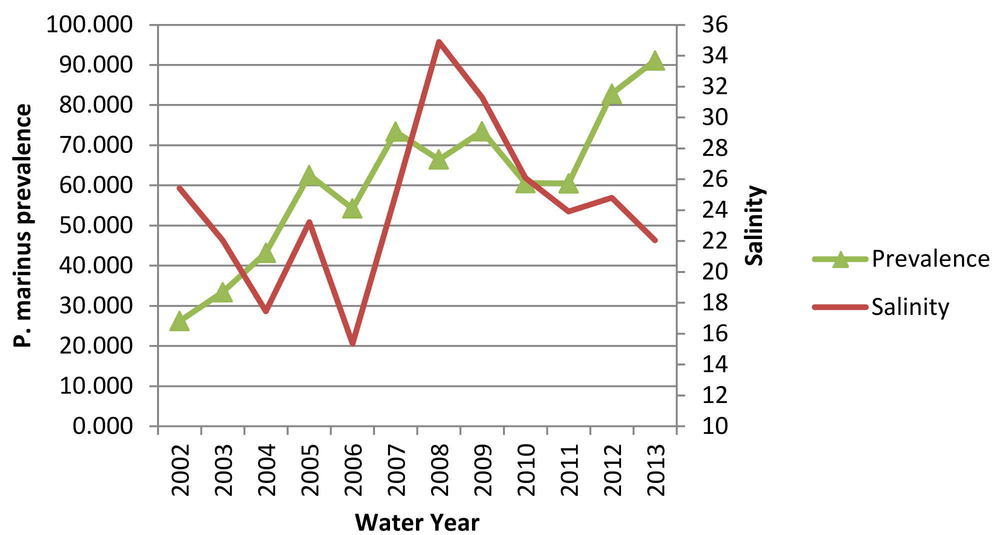


Figure 4-9. Mean prevalence of *Perkinsus marinus* infection in oysters from all sampling locations in the CRE. Disease prevalence is significantly higher during dry and moderate years compared to wet years when salinities tend to be low. Annual averages are also influenced by the timing of rainfall and releases during the year.

While P. marinus intensity was not significantly different between sites at the 95% confidence interval level, it is significant at the 90% confidence level ($F_{4,1.65}$=2.49, P = 0.058). Upstream locations (Iona Cove and Cattle Dock) have lower intensity of disease compared to downstream locations, suggesting that low salinities affect disease levels in oysters (**Figures 4-10** and **4-11**). Slightly higher prevalence and intensity at Bird Island may be due to the subtidal nature of oysters at this location in contrast to intertidal locations in other areas of the estuary.

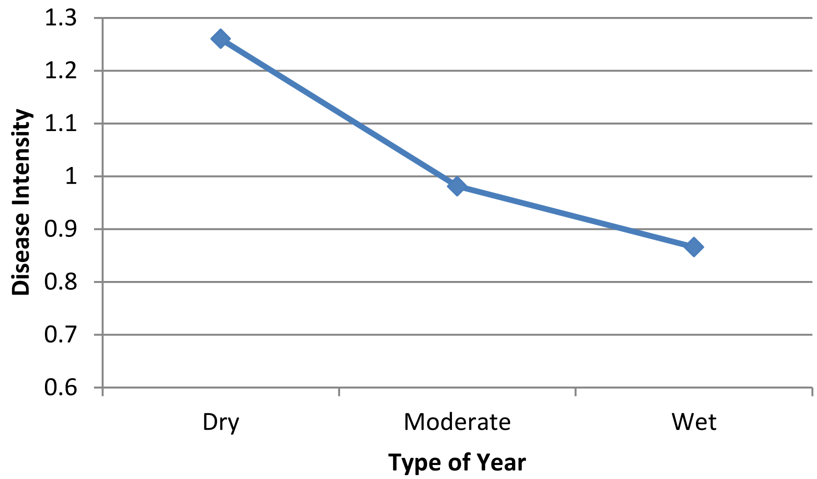


Figure 4-10. Mean disease intensity of *Perkinsus marinus* in oysters during the study period from all sampling locations in the CRE.

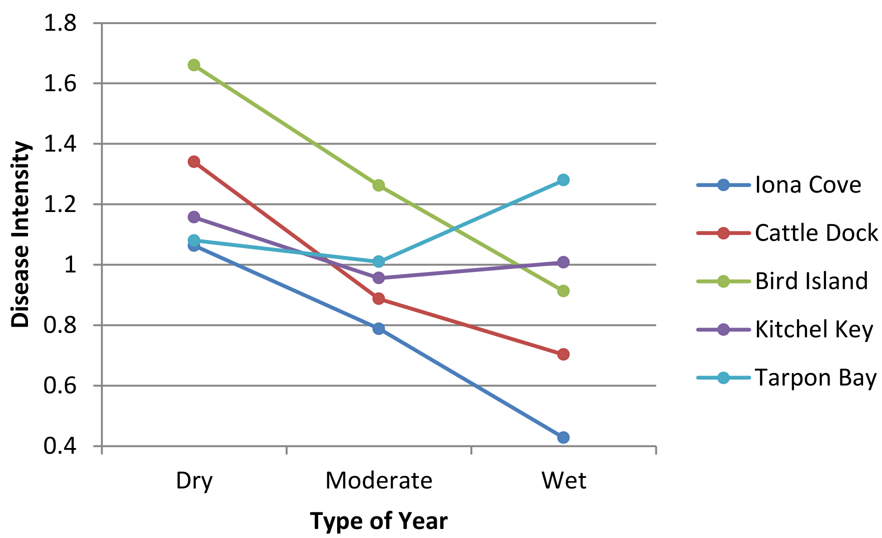


Figure 4-11. Mean intensity of *Perkinsus marinus* in oysters from all sampling locations in the CRE during the study period. With the exception of Tarpon Bay, where salinities are always higher, oysters at all stations showed a decrease in *P. marinus* intensity during wet years compared to dry and normal years.

Recruitment

Oyster spat recruitment varied significantly between stations ($F_{5,669}$=5.098, P < 0.001). There was an increasing trend of higher spat recruitment during wet years compared to normal and dry years, however, this trend was not significant ($_{F2,96}$=1.83, P=0.17) (**Figure 4-12**). Mean spat recruitment increased with increasing distance downstream (**Figure 4-13**). While there was no difference in spat recruitment at the upstream locations (Pepper Tree Point, Iona Cove, and Cattle Dock) between wet, normal and dry years due to low recruitment, recruitment at the mid-stream to downstream locations increased during wet years compared to normal and dry years due to physical flushing (transport) of larvae to downstream locations and/or mortality of larvae and juveniles at upstream locations due to low salinities. This trend is especially true during wet years, where recruitment at downstream locations (Bird Island, Kitchel Key and Tarpon Bay) significantly increased during wet years when mean recruitment was 5 to 10 fold higher compared to dry or normal years. Low recruitment at downstream locations during dry years may be due to the direct effect of high salinities on fecundity and/or mortality due to high salinities and predation of spat.

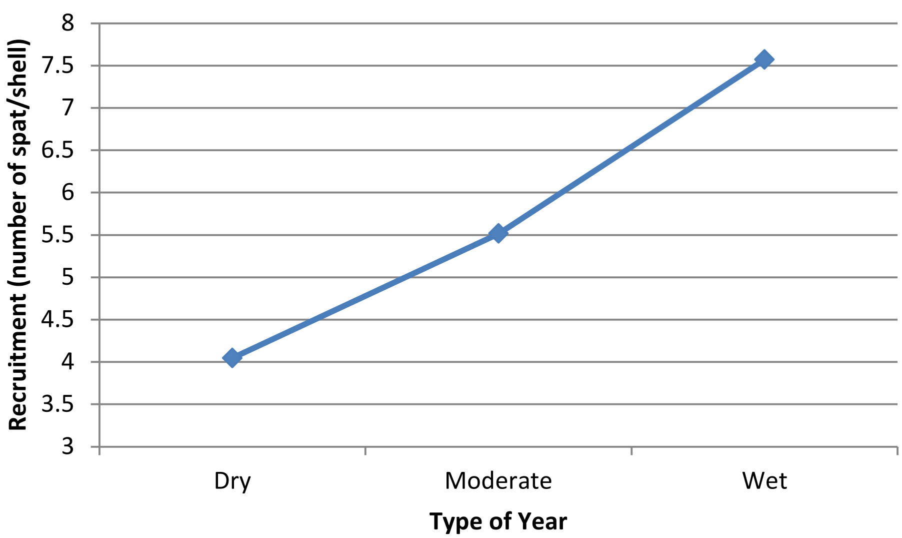


Figure 4-12. Mean spat recruitment of oysters from all sampling locations in the CRE. Spat recruitment, although not significantly different, is higher during wet years compared to dry and moderate years.

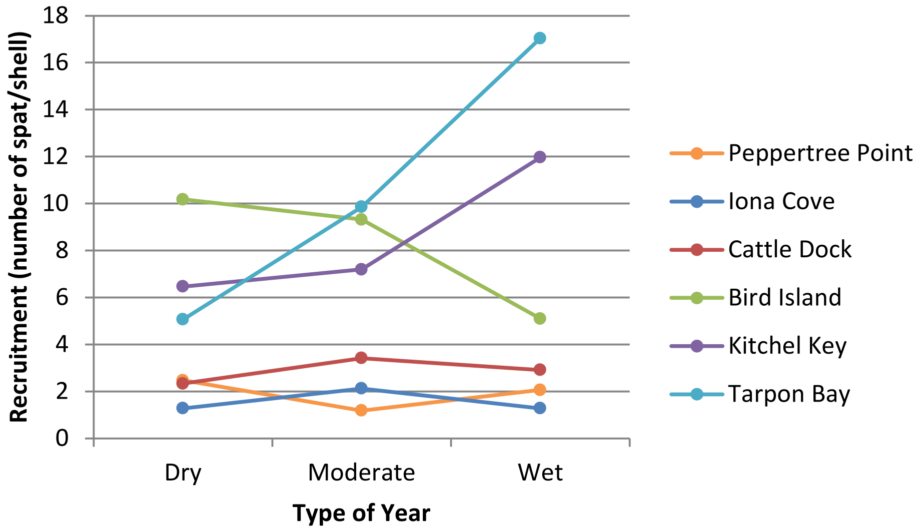


Figure 4-13. Mean recruitment of oyster spat at all sampling locations during the study period. Mean recruitment at upstream stations (Peppertree Point, Iona Cove, and Cattle Dock) decreased during wet years probably as a result of low salinities and/or physical flushing of oyster larvae to downstream locations, while recruitment was higher at downstream locations (Kitchel Key and Tarpon Bay) during wet years due to favorable salinities and/or larval flushing to these locations.

Density

Live oyster density is high in the CRE with values consistently exceeding 800 oysters per square mile at all sampling locations (**Figure 4-14**), with the exception of a period following a very wet year (e.g., WY 2005). While the live density increased at downstream locations during wet years compared to dry years, upstream location (e.g., Iona Cove) density decreased during wet years and increased during dry years (**Figure 4-15**). Decrease in live density at upstream locations may be due to low recruitment and survival of larvae due to low salinities as well as the effect of prolonged low salinities on adult oysters themselves. Increase in live density during dry years is due to low flushing and retention of larvae in the estuary as well as favorable salinities. Lower densities of oysters at downstream locations during normal and dry years may be due to a combination of disease pressure as well as predation pressure because of higher salinities. A combination of favorable salinities and higher spat recruitment at downstream locations results in higher live densities during wet years; however, when dry or normal conditions return during winter periods or normal/dry years, salinities become high resulting in decreased growth and/or increased predation.

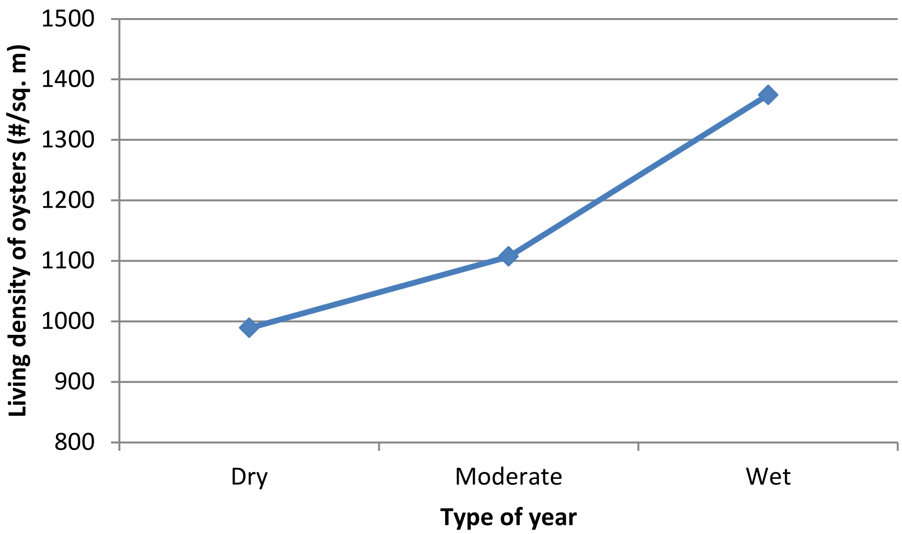


Figure 4-14. Mean living density of oysters from all sampling locations in the CRE. Living density, although not significantly different, is slightly higher during wet years compared to dry and moderate years.

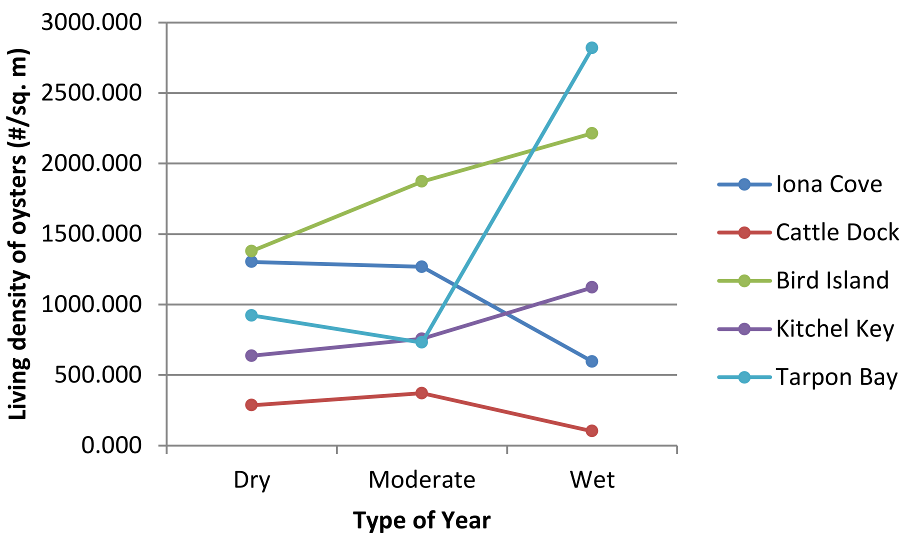


Figure 4-15. Oyster density by station and type of water year. Density is lower at upstream sites and increased at downstream sites during wet years, which is further evidence of increased mortality upstream and flushing of larvae during wet years.

Gonadal Index

Gonadal index increases from 1, when an oyster is not reproducing and gonads are empty of gametes, to 5, when the oyster is fully reproductive and gonads are full of gametes. It then decreases to 1 as the gametes are shed and the gonads become depleted. Thus, mean gonadal condition index is higher when larger proportions of the population are reproductive at the same time or if the duration of the reproductive season increases. In either case, increased gonadal condition index should lead to increased reproductive output.

Gonadal Index of oysters was significantly affected by site and type of year. Gonadal index of oysters was significantly higher at the upstream location (Iona Cove) compared to all other locations in the mid- and downstream locations ($F_{4,4.95}$=21.33, P=0.0001). Overall, gonadal index of oysters was significantly higher during wet years compared to dry and normal years ($F_{2,2.72}$=23.48, P=0.0001) (**Figure 4-16**). There was also a significant interaction between type of year and sampling location ($F_{8,1.58}$=3.42, P=0.004). **Figure 4-17** indicates that the overall pattern (wet year > dry and normal years) was observed at all sampling sites except Bird Island.

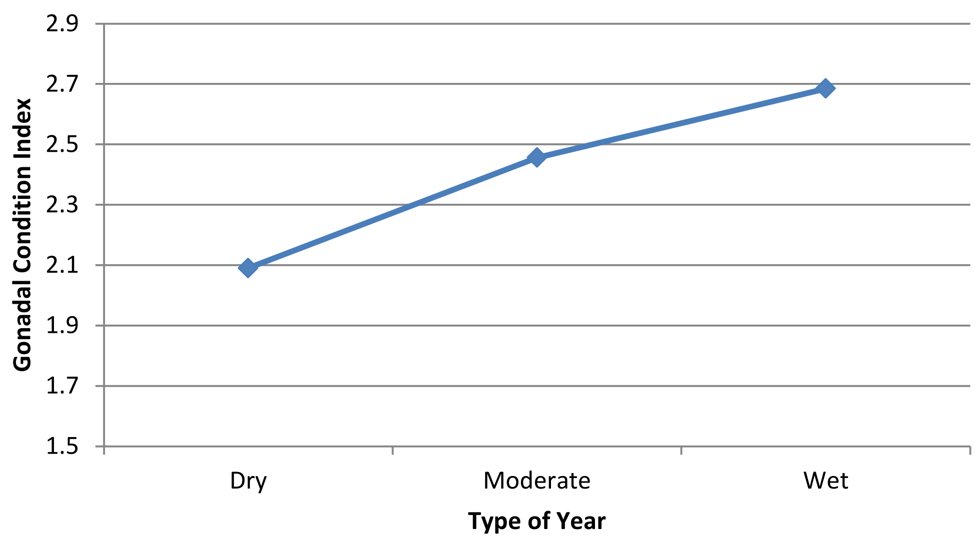


Figure 4-16. Mean gonadal index of oysters from all sampling locations in the CRE. Gonadal index of oysters is significantly higher during normal years compared to dry years and higher during wet years compared to normal and dry years.

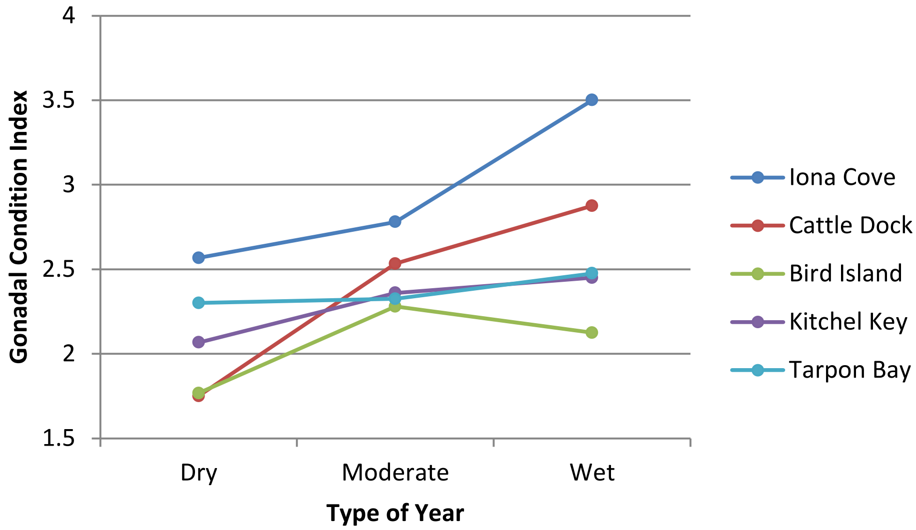


Figure 4-17. Mean gonadal condition of oysters at all sampling locations during the study period. Gonadal condition of oysters increased during wet years compared to moderate (normal) and dry years except at Bird Island.

Mortality

Mortality was monitored from 2008 to 2013 in mesh bags that were closed to predators and mesh bags that were open to predators. Monitoring at the Cattle Dock station was discontinued in 2011, and was discontinued at Tarpon Bay in 2013. Mean annual mortality is calculated as the geometric mean of monthly mortality rates. Mortality in open bags was significantly higher than in closed bags (**Figure 4-18**), which is attributed to predation. The lack of difference at Cattle Dock is probably due to the small sample size, and may be influenced by the extremely wet years that comprise two of the three observations at that station.

Variance was higher in opened bags (**Figure 4-19**) due to greater variance among stations that is probably driven by predator response to changes in salinity upriver. Lower salinities upriver may exclude marine predators during wet and moderate years, but not during dry years, when salinities upriver increase.

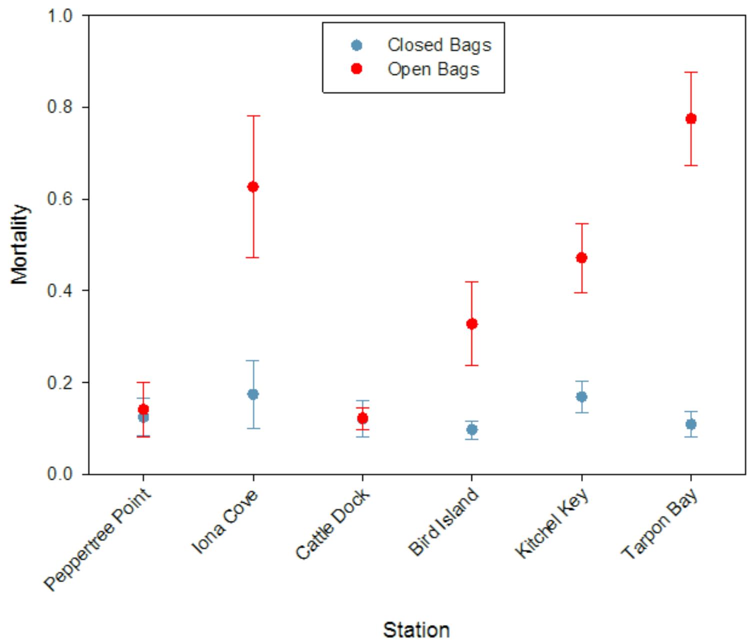


Figure 4-18. Comparison of mortality in closed and opened bags at each monitoring station. The stations are ordered from farthest upriver at the left to closest to the Gulf of Mexico on the right. Symbols represent mean ± standard error.

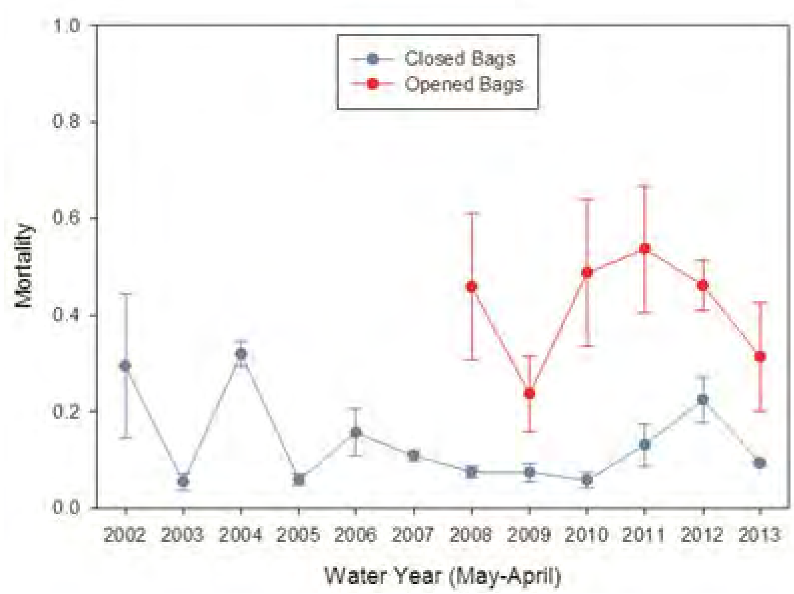


Figure 4-19. Comparison of mean mortality in closed and opened bags by year. Symbols represent mean ± standard error, based on estimated annual means at each station.

Mesocosm Studies

Freshwater inflows and salinity can dramatically impact survival and growth of early life stages of oysters. Effects of salinity on the gamete, embryo, veliger and late spat (8–10 millimeter) stages of oysters were examined under controlled conditions using mesocosms. Gametes, embryos and larvae obtained from oysters acclimated to a salinity of 30 (typical of late spring salinities) were exposed to a range of salinities (0, 3, 5, 10, 15, 20, 25, 30, 35, and 40) for 4 days at 25 degrees Celsius (°C). Fertilization success, development to D-larvae and survival were observed in gametes, while developmental abnormalities and survival were examined in embryos and larvae exposed to a range of salinities typically observed in the CRE. Older spat (10–15 millimeters) were exposed to similar salinity ranges (0–40) for 2 weeks but at 25° C and 30° C, with the latter being the temperature observed during peak spawning season in the CRE (**Figure 4-20**).

Fertilization success decreased, and developmental abnormality and mortality increased at salinities below 15 or above 35 for gametes and embryos (results not shown). Larval mortality and developmental abnormality were high below salinities of 10. Older spat appeared to tolerate salinities ≤ 5 for up to 4 days at 25°C, but encountered significant mortalities thereafter. However, at 30°C, juvenile oysters encountered mortality within 2 days at salinities < 15. Salinities above 20 were favorable for survival of juvenile oysters. These results suggest that in addition to salinity, temperature is a stressor and should be included in predictive models. Also, while each environmental factor is detrimental, in combination they exert a synergistic effect with oysters dying more quickly at lower salinities and higher temperatures (within 2 days at 30° C and within 6 days for 25° C at 10 and below) than accounted for by the separate effects of these factors.

When salinities were gradually decreased to simulate a prolonged exposure to low salinities resulting from extended releases and/or watershed runoff due to seasonal rainfall, juvenile oysters at 25° C were able to tolerate salinities ~ 5 for up to 12 days, but at 30° C encountered significant mortality within 8 days at salinities ~10 (**Figure 4-21**). Similar to acute salinity change experiments, juvenile oysters at 30° C encountered significantly higher mortality compared to 25°C.

Limiting the frequency and quantity of freshwater releases during warm summer months may enhance survival of juvenile oysters. These results can inform resource managers in making regulatory freshwater releases and help aid in adaptive management of freshwater releases into southwest Florida estuaries. Information about the optimal timing and amount of freshwater releases that affect the survival of oysters (including juvenile oysters) for southwest Florida estuaries is lacking. Since the relationship between freshwater inflows and salinities in the CRE is known, this study provides resource managers with target freshwater inflows to minimize salinity-related mortality of the early life stages of oysters. For example, minimizing high volume regulatory freshwater releases to flows less than 3,500 cfs into the estuary for less than 1 week may enhance oyster recruitment, growth and survival of oysters in the CRE.

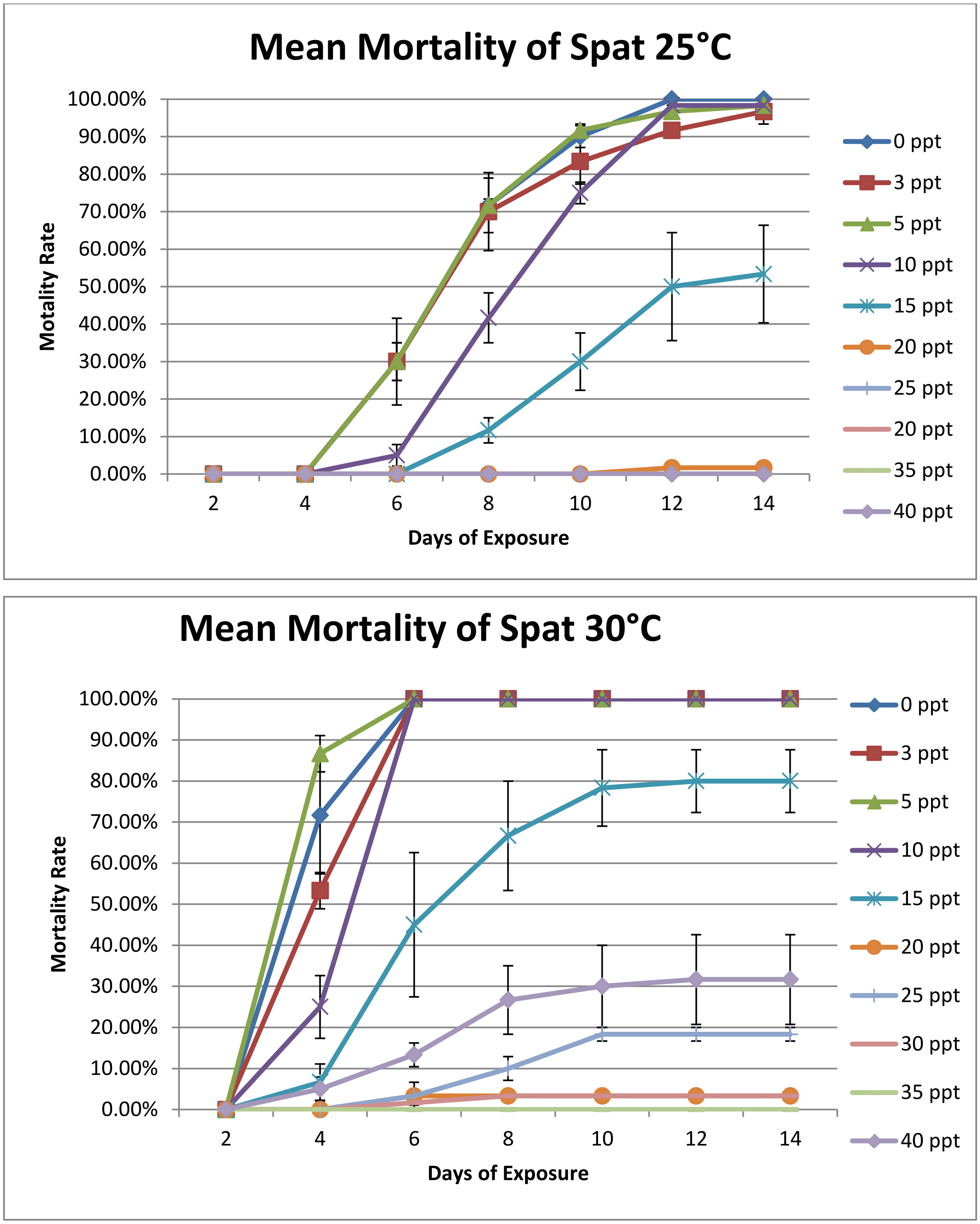
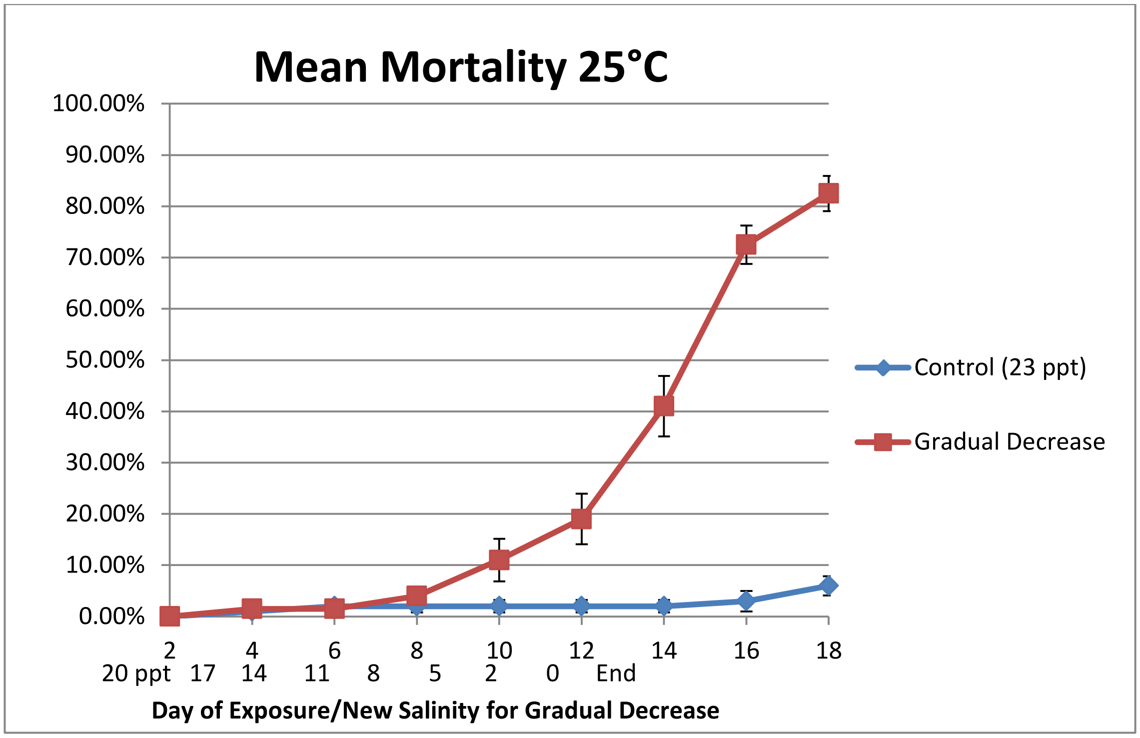


Figure 4-20. Mean mortality of juvenile oysters (10 to 15 millimeter) exposed to a range of salinities. Juvenile oysters at a salinity of 25 were subjected to acute salinity change to mimic high volumes of freshwater releases into the estuary and mortality observed. Results suggest that while salinities < 15 may be stressful to juvenile oysters, summer water temperature of 30°C is also stressful. [Note: ppt – parts per thousand, which is an outdated unit for salinity. Salinity is now considered a unitless measurement.]



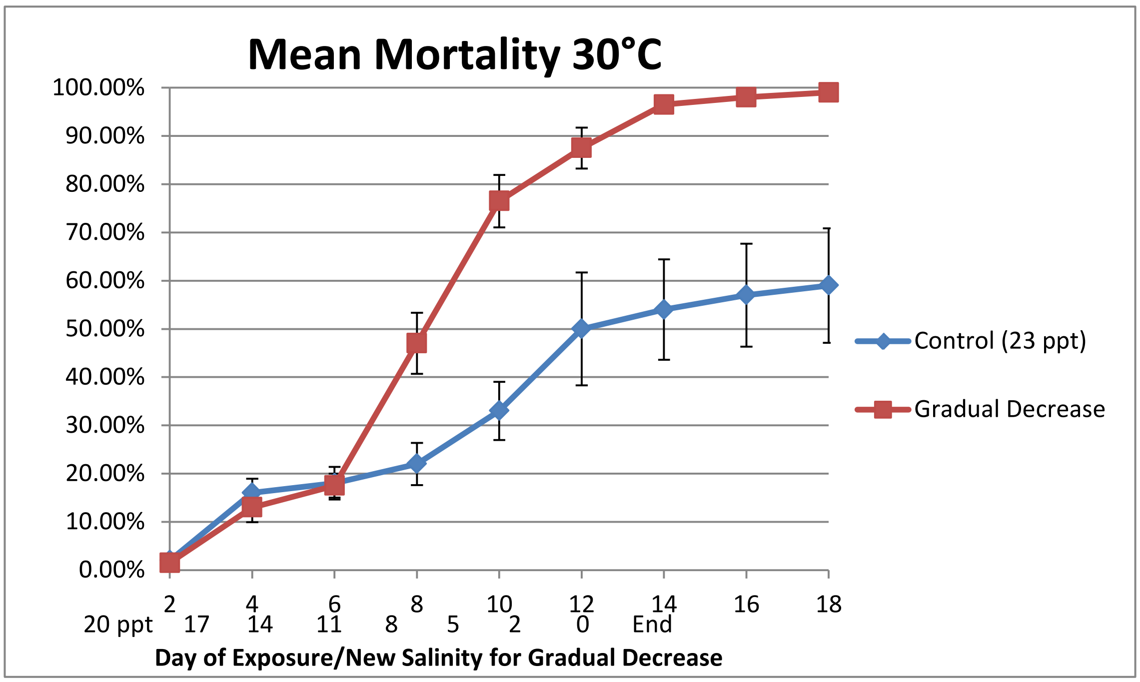


Figure 4-21. Mean mortality of juvenile oysters (10 to 15 millimeters) exposed to various salinity regimes via gradual (2-3 salinity change. Oysters at a salinity of 25 were subjected to a salinity change of 2 to 3 per day and salinity decreased over period of 2 weeks to simulate prolonged freshwater discharge into the estuary. Top line on the X-axis is the duration of exposure in days, while the bottom line is the salinity in the treatments on that day. [Note: ppt – parts per thousand, which is an outdated unit for salinity. Salinity is now considered a unitless measurement.]

Overview Discussion

A significant relationship exists between freshwater inflows and salinities at various points in the CRE. Flows below 3,500 cfs into the estuary from the S-79 structure will result in a salinity regime that enables oysters to survive and grow. Disease prevalence was lower at upstream locations and increased with distance downstream, suggesting that higher salinities result in increased disease incidence. In addition, disease prevalence and intensity decreased during wet years compared to dry years suggesting that freshwater releases help alleviate disease pressure. Limited freshwater releases for durations of less than two weeks will result in lower prevalence and intensity of disease in oysters and higher oyster survival. Oysters in the CRE appear to spawn actively between May and October, a period that coincides with freshwater releases and watershed runoff. While downstream locations attract higher spat recruitment due to higher substrate availability and estuarine conditions during high flow summer and fall months, growth and survival of juveniles is poor. Limiting freshwater releases to less than 3,500 cfs during these months will limit flushing of oyster larvae to downstream locations and create a favorable salinity regime for spat recruitment and survival. Low disease incidence, high condition index, sufficient spat recruitment and high growth rate at the upstream locations (e.g., Iona Cove) suggest that with the provision of suitable substrate and limiting freshwater flows during the spawning season, oyster reefs will survive and grow in the upstream locations. With CERP implementation and subsequent reduction in freshwater flows, it is anticipated that oyster reef development will be shifted upstream compared to current locations.

Some refinements in the monitoring program, development of predictive tools, and further knowledge of all factors effecting the reestablishment, health and long-term survival of the oyster communities are needed. At this time, the oyster hypotheses do not need to be refined, but some gaps in knowledge, such as the effect of contaminants on oysters and how varying salinities impact oysters need to be studied. The existing sampling design and sampling frequency can adequately assess the direction and magnitude of change in oyster metrics. The limited data set over the recent drought shows that predation may be a substantial stressor as salinities increase. Given that predation pressure is significant in some locations, such information is necessary and a longer data set will enhance the oyster habitat suitability index by strengthening the predictability of potential suitable habitat.

Combining salinity tolerance targets and larval supply with a particle transport model will offer resource managers locations where oyster larvae are expected at a given flow rate. Since substrate is critical for settlement of oyster larvae and subsequent development of reefs, combination of field monitoring and particle transport models can inform resource managers about managing freshwater inflows that are favorable to oysters (and other biota) as well as ensuring adequate substrate is present for larval settlement at the right places for development of reefs.

Caloosahatchee River Estuary Submerged Aquatic Vegetation

Introduction

SAV, which includes both seagrasses and freshwater aquatic plants, forms productive and species-rich habitats throughout the world (Hemminga and Duarte 2000). SAV contributes to human well-being by stabilizing sediments, improving water quality, and enhancing commercial and recreational fisheries resources (Worm et al. 2006). Unfortunately, SAV has declined dramatically in many regions (Orth et al. 2006). Effects of human activities on habitat quality have been implicated in most of these declines; increased sediment and nutrient inputs lead to high turbidity and eutrophic overgrowth of SAV by epiphytes (Kemp et al. 2004, Orth et al. 2006), and anthropogenic changes in climate and hydrography can exceed SAV species' physiological tolerances (Doering and Chamberlain 2000, Doering et al. 2001, Moore and Jarvis 2008). Changes in hydrography, such as altered water flow and salinity, have been identified as particularly serious threats to SAV in southwest Florida estuaries (Doering et al. 2002, Mazzotti et al. 2007a,b).

CRE has historically provided an ideal setting for SAV (Barnes 2005). Broad, shallow shoals and narrow tidal range provide extensive soft-bottom habitat that remains submersed but within the well-lit photic zone. Barrier islands and mangrove-lined shorelines protect these shallows from damaging waves, yet inlets and channels allow for healthy tidal flow. Perhaps most importantly, the more than 40 kilometer (km) linear extent of the CRE is associated with a gradual progression from marine to freshwater salinities; an environmental gradient that permits the adjacent coexistence of SAV species adapted to different ranges of salinity (Whittaker 1967, Zieman and Zieman 1989).

The highest salinity conditions are found in the polyhaline lower estuary (San Carlos Bay). The predominant SAV species in this zone are the seagrasses *Thalassia testudinum* (turtle grass) and *Halodule wrightii* (shoal grass) (Mazzotti et al. 2007a). Though both *T. testudinum* and *H. wrightii* require salinity > 20 for optimal growth, *T. testudinum* has zero growth below 17 (Zieman and Zieman 1989), whereas *H. wrightii* may exhibit some growth in salinity as little as 12, and can survive even lower salinities for short periods of time (Doering et al. 2002). Further, *T. testudinum* is less tolerant than *H. wrightii* of the low light, high nutrient conditions that often accompany freshwater flow (Barnes 2005, van Tussenbroek et al. 2006). Together, these factors restrict *T. testudinum* to the lower estuary, whereas *H. wrightii* is also found (albeit at lesser densities) in the mesohaline middle estuary from Shell Point northeast towards Fort Myers (Mazzotti et al. 2007a). At the fresher end of the middle estuary zone, *H. wrightii* is replaced by *Ruppia maritima* (widgeon grass), a euryhaline SAV species capable of growing at both high and low salinities (Murphy et al. 2003). *R. maritima* could presumably occupy any point along the salinity gradient in the CRE, but it tends to be sparse and ephemeral where it occurs and it generally does not form dense beds like the estuary's other SAV species. However, *R. maritima* is often found intermixed with *H. wrightii* and may be distributed more broadly than reported due to the fact they resemble one another.The oligohaline upper estuary (from Fort Myers to Beautiful Island and east to the S-79 lock and dam) harbors sparse *R. maritima* and sometimes dense beds of *V. americana* (tape grass) (Mazzotti et al. 2007b). *Vallisneria americana* is a freshwater plant that cannot tolerate prolonged exposure to salinities > 10 (Doering et al. 2001). Although each salinity zone in the CRE is suited to at least one SAV species, it is the most saline and the least saline zones that have historically harbored the densest beds of SAV (Barnes 2005).

Whereas the 40-km spatial gradient in salinity along the CRE contributes to the estuary's overall SAV diversity, temporal fluctuations in salinity at any particular point in the estuary are a potential threat to SAV health. SAV can tolerate moderate and brief salinity fluctuations, like those associated with typical tidal and seasonal cycles, but more extreme fluctuations result in partial or total mortality of SAV beds (Mazzotti et al. 2007a,b). In addition to the magnitude and duration of salinity fluctuations, the frequency of these disturbances is an important factor determining the harm done to SAV. Rare natural events like droughts and hurricanes can seriously damage SAV, but these occur infrequently enough that SAV beds sometimes have time to recover between disturbances.

In contrast, anthropogenic changes in the hydrology of the CRE watershed, including water management practices at the S-79 lock and dam, have contributed to a high frequency of severe salinity fluctuations, which have caused SAV mortality and inhibited SAV recovery in parts of the CRE (Barnes 2005). For this reason, the South Florida Water Management District (SFWMD) has recommended that freshwater flow at the S-79 lock and dam be maintained between daily mean values of 450 cfs and 2,800 cfs (Doering et al. 2002). The relationship between water management, salinity fluctuations, and SAV health in the CRE was investigated, with a focus on evaluating the effectiveness of the 450–2,800 cfs flow envelope for SAV conservation.

Methods

SAV abundance and species composition were monitored monthly or bimonthly at eight shallow, nearshore sites in the CRE from January 1998 until the present (**Figure 4-22** and **Table 4-2**). Until 2009, water quality characteristics, including Secchi depth, chlorophyll *a* concentration, pH, dissolved oxygen, conductivity, salinity, and photosynthetic active radiation, were also monitored at each SAV site. Water quality monitoring at SAV sites ceased in 2009, but water quality data from nearby permanent monitoring stations (CES04-CES10, **Figure 4-22**) have remained available throughout the period of record. The focus of this report is primarily on the influence of just one aspect of water quality—salinity.

SAV monitoring sites were arranged along the estuary from the most river-influenced (Site 1, Beautiful Island), to the most marine (Sites 7 and 8, San Carlos Bay). Monitoring initially focused only on the upper estuary (Sites 1 to 4) where *V. americana* occurred. Site 3 was eliminated in 2002. Monitoring in the middle estuary (Sites 5 and 6) and lower estuary (Sites 7 and 8) began in January 2004. Site 6 was moved slightly in April 2012 and the new site was designated Site 6B.

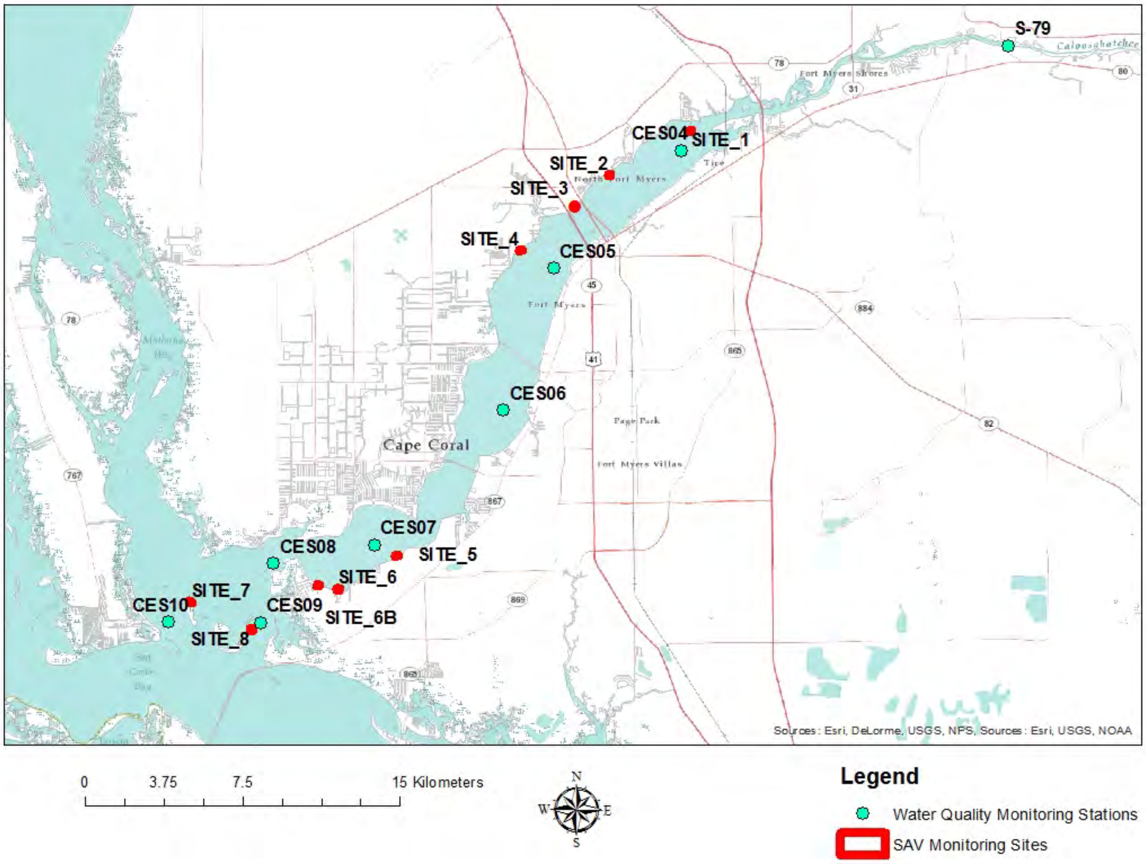


Figure 4-22. SAV and water quality monitoring sites in the CRE 1998–2013.

Table 4-2. A summary of SAV site locations, period of record, and sampling scheme.

Site Number	Estuarine Zone	Predominant SAV Species	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
1	Upper	*V. americana* *R. maritima*	T	T	T	T	T	T	T	T	T	T	T	T	P	P		QZ
2			T	T	T	T	T	T	T	T	T	T	T	T	P	P	P	P
3			T	T	T	T												
4			T	T	T	T	T	T	T	T	T	T	T	T	P	P	P	P
5	Middle	*H. wrightii* *R. maritima*							T	T	T	T	T	T	P	P	P	P
6									T	T	T	T	T	T	P	P		
6B																	P	P
7	Lower	*T. testudinum* *H. wrightii*							T	T	T	T	T	T	P	P	P	P
8									T	T	T	T	T	T	P	P	P	P

Key: T – sampled along transects; P – sampled within polygons; QZ – sampled using "quadzilla"

At each SAV site, on each monitoring date, 20 to 30 points were surveyed with 1-square meter (m^2) quadrats. Prior to 2009, points for quadrat placement were arranged along two 100-meter (m) transects, one parallel to shore and the other perpendicular to shore, intersecting at the 50-m mark to form a "+" shape. A stratified random number generation scheme was used to select five points along each of the four arms of the of the crossed transects. After 2009, the crossed transect method was replaced by a method that distributed 30 points within a rectangular area called a "polygon." The polygon for each site was drawn in geographic information system (GIS) software such that it covered approximately the same one-hectare area that had been spanned by the crossed transects at that site. Polygons remained in fixed locations, but the 30 sampling points within each polygon were randomly redistributed each month SAV was assessed. Wide area augmentation system (WAAS)-enabled handheld global positioning satellite (GPS) units allowed researchers in the field to navigate accurately to each of the designated points. In 2012 the polygon for site 6 was adjusted slightly and redesignated as site 6.1. Also in 2012, the polygon for Site 1 was removed, but it was reinstated in 2013 with broader boundaries and a unique coarse-scale monitoring method with 9-m^2 "quadzilla" quadrats measured at each point. The purpose of this change was to enhance the power to detect sparse and patchy *V. americana* in the area of Beautiful Island.

With the exception of the 9-m^2 quadrat surveys at Beautiful Island, SAV assessment at all sample points was achieved using 1-m^2 quadrat frames divided by string into 20 x 20 centimeter (cm) grid cells (25 cells) or 10 x 10 cm grid cells (100 cells). However, during the period of record some changes were made to the types of SAV data collected with each 1-m^2 quadrat (**Table 4-3**). From 1998 to 2007, shoots were counted from a subset of 10 x 10 cm cells within the quadrat, and these data were extrapolated to shoots per m^2 (shoots/m^2), which was the primary measure of SAV abundance during that period. In 2008 and 2009, presence/absence of shoots was recorded within each 10 x 10 cm and 20 x 20 cm cell of the 1-m^2 quadrat. These "grid counts" ranged from 0 to 100 for 10 x 10 cm cells, and 0 to 25 for 20 x 20 cm cells. The higher resolution 0–100 grid counts were dropped in 2010, but 0–25 grid counts have been maintained. In 2012, a visual percent cover estimate for the whole 1-m^2 quadrat was added as a compliment to the grid counts. Unbroken, dense cover of SAV was considered to be 100% cover, and field researchers were trained to estimate cover using this as their benchmark.

Table 4-3. Summary of principal measures of SAV abundance throughout the period of record.

Main Types of SAV Data Collected	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
Shoots/m^2	X	X	X	X	X	X	X	X	X	X						
Grid counts (1–100)											X	X				
Grid counts (1–25)											X	X	X	X	X	X
Percent cover estimate															X	X
Canopy height	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X

Key: X – data type was collected during this period

Because the primary data type for SAV abundance changed twice during the period of record, it was necessary for time series analyses to convert all data to one, common type—percent cover. Percent cover was chosen because it conveys information about both the presence and the density of SAV, and because percent cover could be relatively easily inferred from other monitoring data for the years in which it was not recorded directly. Conversions from shoots/m^2 to percent cover were developed for 1998–2007, and conversions from 0–25 grid count to percent cover were developed for 2008–2011. Conversion from shoots/m^2 to percent cover was accomplished using prior data describing the mean number of shoots/m^2 for a normal density (100% cover) bed of each SAV species: 600 shoots/m^2 for *V. americana* (Hauxwell 2007), 3,000 shoots/m^2 for *R. maritima* and *H. wrightii* (Dunton 1990), and 400 shoots/m^2 for *T. testudinum* (Lapointe et al. 1994). Conversion of 0–25 grid count to percent cover was based on linear regression analyses of the 2012–2013 SAV data in which grid count and visually estimated percent cover were recorded concurrently. Initial regressions were poor fits, but R^2 values of 58–70% were eventually achieved by incorporating both canopy height and grid count into the following linear model: percent cover = c*(canopy height*grid count), where c is the regression coefficient (**Figure 4-23**). The value for c varied according to SAV species, but was near 0.1 for all species. A conversion specific to *V. americana* could not be developed, due to a paucity of *V. americana* in 2012–2013, so the conversion factor for *T. testudinum* was used. While there is some residual variability in the conversion models at the scale of a single 1-m^2 quadrat, estimates of monthly mean percent cover, per site are insulated from this variability because each is the average of 20 to 30 individual quadrat estimates.

In addition to the SAV data collected, S-79 flow measurements and CRE salinity data were compiled for the period of record. Daily mean stream flow data from S-79 were downloaded from the United States Geological Survey (USGS) National Water Information System: Web Interface (waterdata.usgs.gov/nwis). Monthly means were calculated from the S-79 streamflow data and were plotted as a time series (**Figure 4-24**). The primary measures of salinity in the CRE were approximately biweekly measurements taken from the monitoring stations CES04, CES07, and CES09 in the upper, middle, and lower estuary, respectively (**Figure 4-22**). These salinity data were downloaded from the SFWMD's DBHYDRO online database. Though salinity at CES04, CES07, and CES09 was recorded at multiple depths, only data from the 0.5-m water depth was used, which is mostly likely to represent the salinity conditions experienced by shallow water SAV. As with the S-79 flow data, salinity data was converted to to monthly mean values to generate a time series plot of salinity at CES04 (**Figure 4-25**).

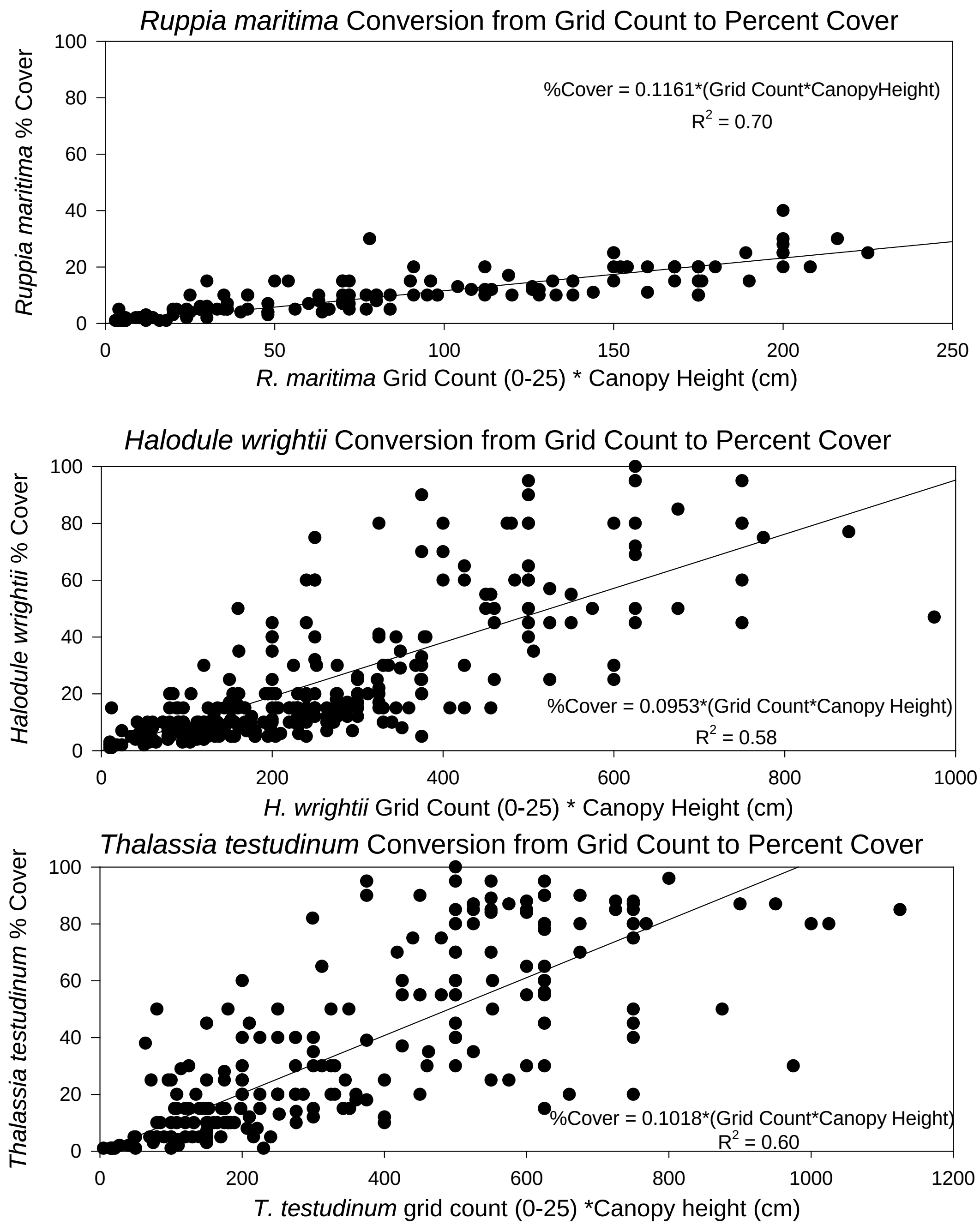


Figure 4-23. Regression-based conversion between multiple methods for census of *R. maritima* (top), *H. wrightii* (middle), and *T. testudinum* (bottom) abundance. Grid count is the count of 20 x 20 cm cells in a 1-m^2 quadrat that are occupied by the species. Canopy height is the height of the canopy in the quadrat, excluding the tallest 20% of shoots. Percent cover is a visual estimate of the proportion of the 1-m^2 quadrat that is vegetated by the species, normalized to benchmarks for 100% of the species cover. The x-axis is a derived statistic; the product of grid count and canopy height, which is used as an intermediate step in the conversion.

Results and Discussion

<u>*Flow Rates*</u>

Mean monthly flow rate through the S-79 canal varied from 0 cfs to more than 10,000 cfs during the period of record, which included prolonged episodes of flow below the recommended minimum of 450 cfs, and flow above the recommended maximum of 2,800 cfs (**Figure 4-24**).

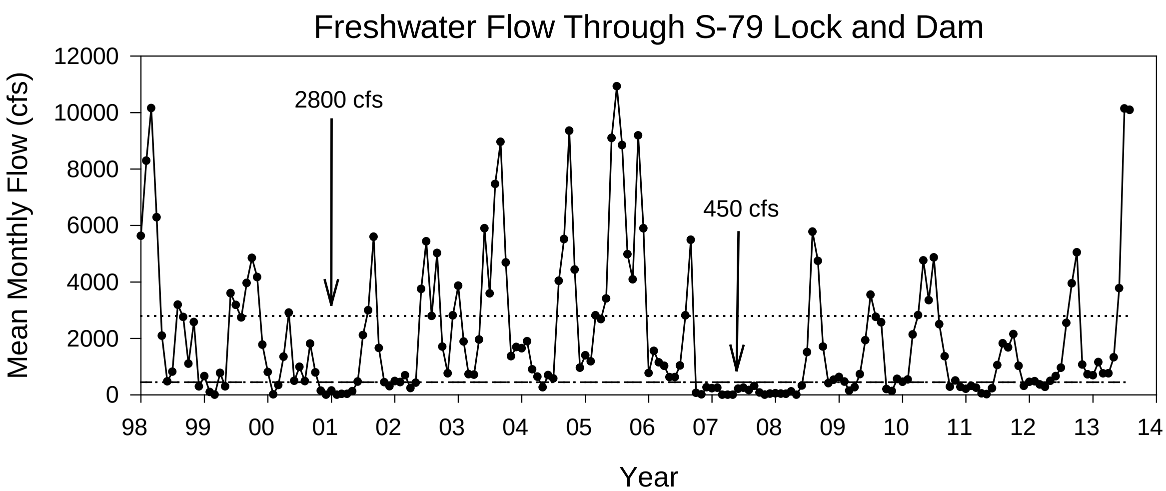


Figure 4-24. Mean monthly flow through S-79 lock and dam, 1998–2013. Horizontal lines indicate lower and upper boundaries of recommended flow envelope (450 cfs and 2,800 cfs, respectively).

<u>*Salinity*</u>

Salinity was highly variable in all regions of the CRE, but salinity variation was most extreme in the upper estuary (monitoring station CES04), where it ranged from 0 to 26 during the period of record (**Figure 4-25**).

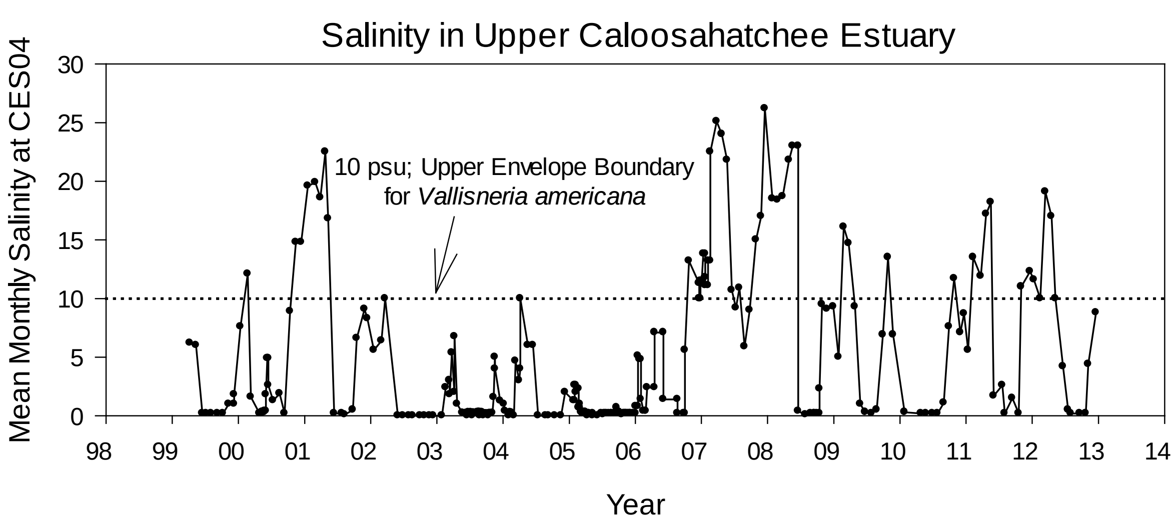


Figure 4-25. Mean monthly salinity at the CES04 water quality monitoring station in the upper CRE.

Relationships between Flow and Salinity

S-79 flow strongly influenced salinity in all regions of the estuary; salinity clearly decreased as S-79 flow increased (**Figures 4-26** through **4-28**). However, there was some residual variation in CRE salinity that was not perfectly explained by S-79 flow. This is likely attributable to freshwater flows from portions of the CRE watershed other than the S-79 lock and dam, along with other factors such as mixing by winds and tides in **Figures 4-26** through **4-28**. The degree to which variation in CRE salinity could be predicted by S-79 flow is evident in the R^2 values for the regression models, which were 0.76, 0.72, and 0.45 for the upper, middle, and lower estuary, respectively (**Figures 4-26** through **4-28**). The relatively low R^2 value for the model of lower estuary salinity may reflect the greater influence of coastal oceanographic processes such as tidal flushing in the lower estuary as compared with the upper estuary.

The relationship between S-79 flow and salinity in the upper estuary (**Figure 4-26**) supports the 450-cfs envelope boundary as an effective minimum flow for keeping salinities below the 10 tolerance of *V. americana*. Salinities in the upper estuary rarely exceeded 10 when flow was above 450 cfs.

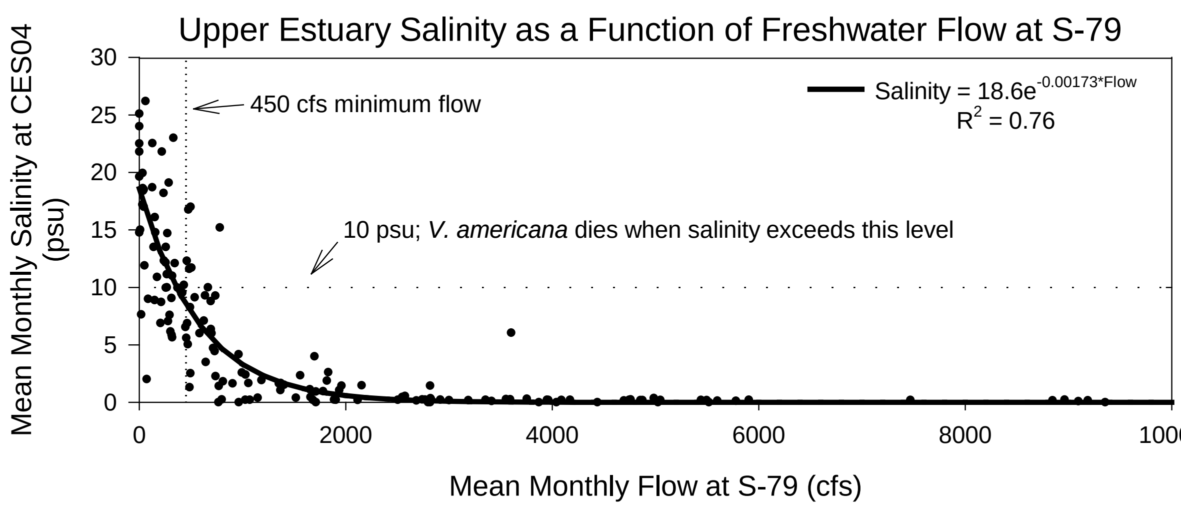


Figure 4-26. Salinity at station CES04 in the upper CRE plotted against mean monthly freshwater flow through the S-79 structure. The dotted vertical line indicates the lower bound of the recommended flow envelope for S-79. Dashed horizontal line indicates the maximum salinity tolerance threshold for the freshwater aquatic plant *V. americana*, which was once abundant in this region of the estuary. The solid black line represents salinity levels predicted by a simple exponential decay function of S-79 flow, which is indicated by the equation in the upper right hand part of the panel. [psu – practical salinity unit, which is an outdated unit for salinity. Salinity is now considered a unitless measure.]

The salinity versus flow relationship in the middle estuary (**Figure 4-27**) showed that upper envelope flows of 2,800 cfs corresponded with salinities of approximately 10 at CES07. This is above the lethal lower limit of 6 for *H. wrightii*, but below the approximately 15 required for appreciable *H. wrightii* growth in this system (Doering et al. 2002). This suggests that prolonged flow near 2,800 cfs would result in decline of *H. wrightii* in the middle estuary due to lack of growth needed to compensate for losses due to grazing, etc. Therefore, the persistence of *H. wrightii* in this part of the estuary must depend on periods of more optimal salinity conditions (~20) associated with flows near the lower envelope boundary of 450 cfs. In summary, adhering to the 2,800-cfs flow maximum will protect middle-estuary *H. wrightii* from acutely lethal low salinities, but to allow positive growth of *H. wrightii* will also require a flow regime that includes periods of flow near the lower envelope boundary of 450 cfs. The middle estuary is likely to remain the most challenging CRE zone for SAV, but the coverage and density of *H. wrightii* should increase relative to current levels once a more stable flow regime is restored by CERP projects.

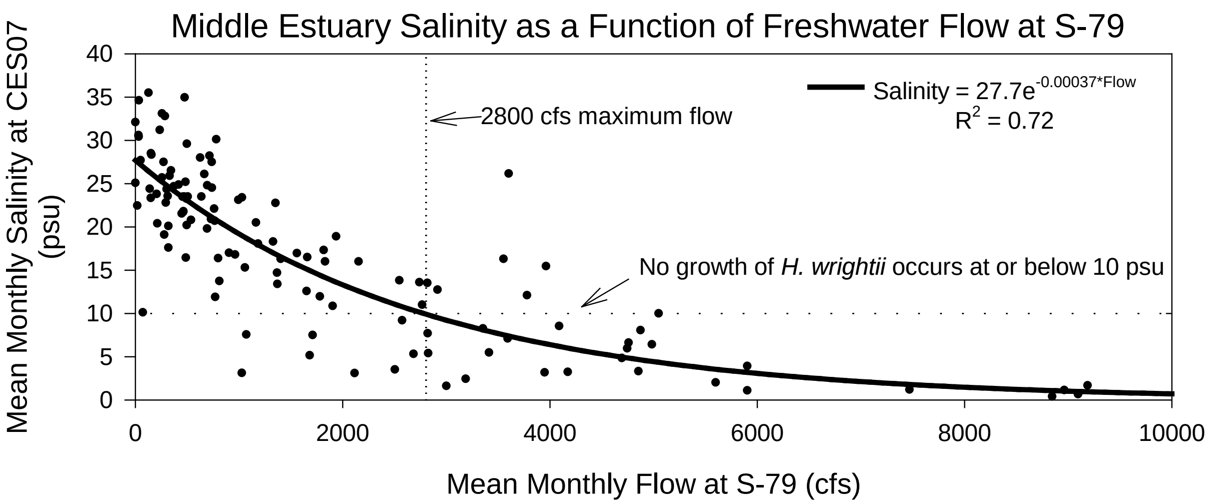


Figure 4-27. Salinity at station CES07 in the middle CRE is plotted against mean monthly freshwater flow through the S-79 structure. The dotted vertical line indicates the upper bound of the recommended flow envelope for S-79. The dashed horizontal line indicates the salinity level required for dense growth of the seagrass *H. wrightii*. The solid black line represents salinity levels predicted by a simple exponential decay function of S-79 flow, which is indicated by the equation in the upper right hand part of the panel. [psu – practical salinity unit, which is an outdated unit for salinity. Salinity is now considered a unitless measure.]

In the lower estuary, salinity was generally > 20 when S-79 flow was < 2,800 cfs, implying that the flow envelope was effective in maintaining desirable salinity conditions for growth of both *H. wrightii* and *T. testudinum*. When flow at S-79 was above 5,000 cfs, however, salinity in the lower estuary was invariably below 20 (**Figure 4-28**). This helps explain previous reports of reduced seagrass coverage and density in the lower estuary in association with S-79 flows > 4,500 cfs (Mazzotti et al. 2007a).

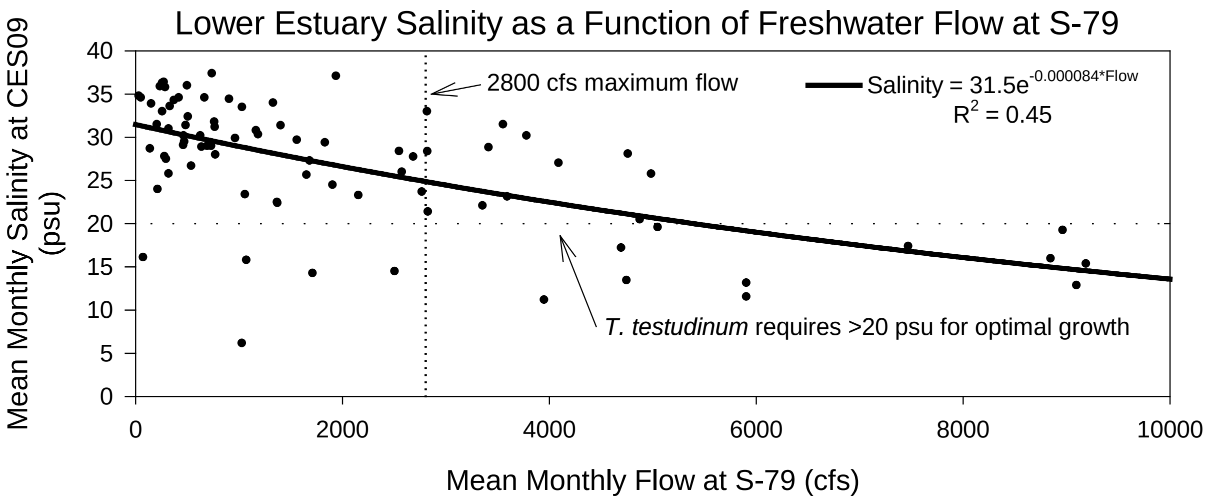


Figure 4-28. Salinity at station CES09 in the lower CRE (San Carlos Bay) plotted against mean monthly freshwater flow through the S-79 structure. The dotted vertical line indicates the upper bound of the recommended flow envelope for S-79. The dashed horizontal line indicates the salinity level required for optimal growth of the seagrasses *T. testudinum* and *H. wrightii*. The solid black line represents salinity levels predicted by a simple exponential decay function of S-79 flow, which is indicated by the equation in the upper right hand part of the panel. [psu – practical salinity unit, which is an outdated unit for salinity. Salinity is now considered a unitless measure.]

Impacts of Flow and Salinity on SAV

The freshwater SAV species *V. americana* was abundant in the CRE when monitoring began in 1998 (**Figure 4-29**). It formed lush beds in the upper estuary, from Beautiful Island to Fort Myers. High salinity events in 1999, 2000, and 2001 (**Figure 4-25**) decimated *V. americana*, eliminating it from the areas where it was formerly observed. These high salinity events were related to low flows through the S-79 structure, which were far below the lower limit of the flow envelope (**Figures 4-24** through **4-26**). Though flows through S-79 were maintained at higher levels from 2002 to 2006, *V. americana* remained nearly absent until a partial recovery was notable by 2004. In late 2006, *V. americana* was again decimated by a high salinity event. *V. americana* remained absent in the CRE from 2007 to 2009 with a minimal reoccurrence in 2010, which was eliminated by high salinities in 2011.

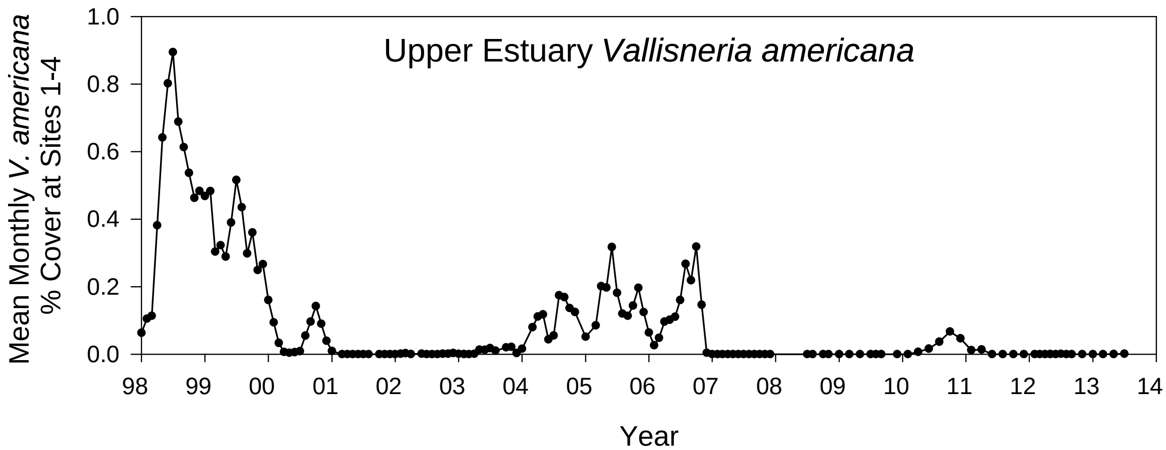


Figure 4-29. *V. americana* percent cover (1.0 = 100%) in the upper CRE from 1998 to 2013. These are mean values from three to four SAV monitoring sites in the upper estuary region.

The time series of *V. americana* abundance, in conjunction with the salinity and flow data (**Figures 4-24** through **4-26**), support 450 cfs as the lower limit for maintaining *V. americana* habitat in the CRE. Further, they show that *V. americana* recovery after a near-total die-off is a slow process. Assisted restoration (planting) of *V. americana* could hasten the recovery process, but only if a suitable salinity regime for recovery is established by CERP projects allowing consistent within-envelope flows at S-79.

While unusually high salinity harms *V. americana* in the upper CRE (**Figures 4-24** through **4-26**), unusually low salinity harms seagrasses in the middle and lower parts of the estuary. In the 15-year data set, episodes of low salinity in the middle and lower estuary correspond closely with flows in excess of 2,800 cfs from the S-79 canal (**Figures 4-27** and **4-28**). The seagrass most vulnerable to stress from low salinity is the *H. wrightii* that grows in the Iona Cove region of the middle estuary (**Figure 4-30**). Salinities are extremely variable in this region (0–36, **Figure 4-27**), and therefore stressful for *H. wrightii*, which requires salinities > 20 for optimal growth (Rudolph 1998). *H. wrightii* was nearly absent from the middle estuary from 2004 to 2007; a period of low salinity related to frequent > 2,800 cfs flows from S-79 (**Figures 4-24** and **4-30**). When salinities increased in 2007, due to within- or below-envelope flows from S-79, *H. wrightii* began to recover. Though *H. wrightii* in the middle estuary remains sparse and seasonally ephemeral, it has a good chance of firmer establishment if S-79 flows can be kept below 2,800 cfs for most of the year.

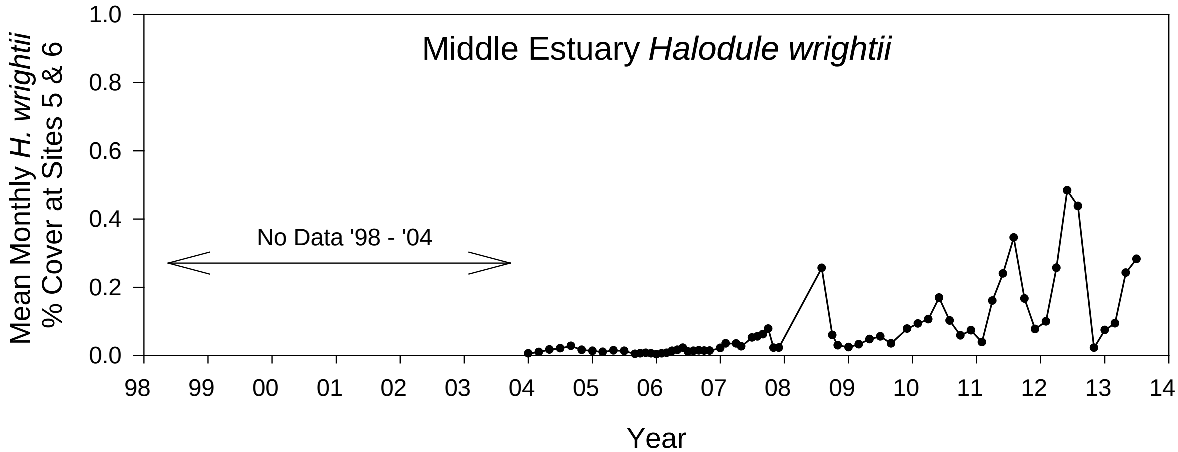


Figure 4-30. ***H. wrightii*** **percent cover (1.0 = 100%) in the middle CRE from 2004 to 2013. These are mean values from two SAV monitoring sites in the middle estuary region.**

West of Shell Point, in the lower estuary, are the thickest and most diverse seagrass beds surviving in the CRE area. They include a mixture of *T. testudinium* and *H. wrightii* (**Figure 4-31**). In comparison with the obvious, strong effects of salinity changes on *V. americana* in the upper CRE, and on *H. wrightii* in the middle CRE, the effects of salinity changes in the lower estuary are more subtle (**Figure 4-31**). Natural seasonal cycles of seagrass growth and senescence, and the variable effects of grazers and nutrients, are among a number of factors besides salinity that can strongly influence estuarine seagrass communities (Herzka and Dunton 1997, Douglass et al. 2010). Nevertheless, comparison of S-79 flows (**Figure 4-24**) and lower estuary SAV (**Figure 4-31**) suggest that both *H. wrightii* and *T. testudinum* in the lower estuary are negatively impacted by suboptimal salinities associated with high S-79 flows. In particular, the seagrass declines seen in late 2005 and early 2006 are associated with high discharges related to Hurricane Wilma (October 2005). It is apparent that *H. wrightii* recovered from this disturbance in one year, while *T. testudinum* required about three years to become the dominant seagrass species once more. Though both *H. wrightii* and *T. testudinum* perform best in salinities > 20, *H. wrightii* is more resistant to lower salinities and poor water quality in general, and it tends to recover faster following disturbances (Lapointe et al. 1994). It has been hypothesized previously that high flows from S-79 could shift the competitive balance between *T. testudinum* and *H. wrightii* in favor of the latter (Barnes 2005, Mazzotti et al. 2007a), and our current analysis supports this hypothesis (**Figure 4-31**).

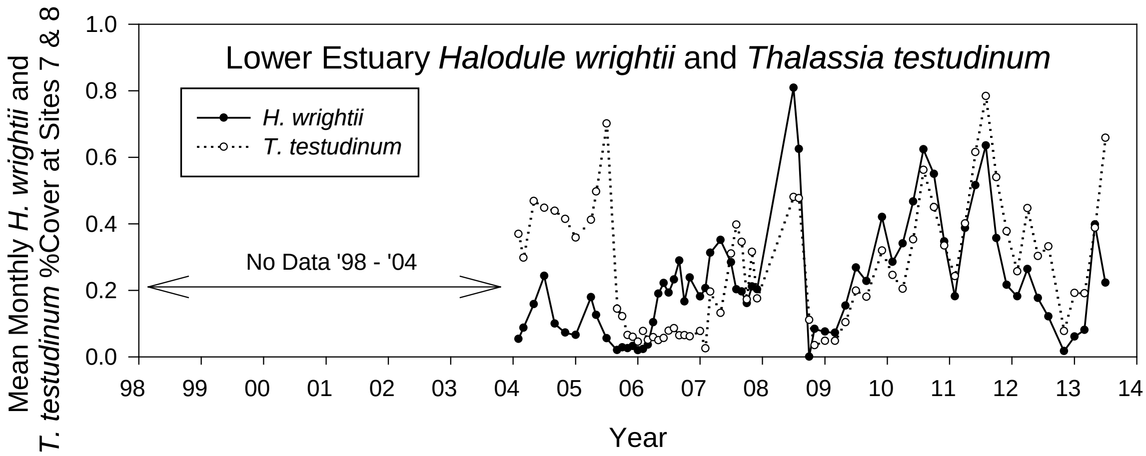


Figure 4-31. *H. wrightii* and *T. testudinum* percent cover (1.0 = 100%) in the lower CRE (San Carlos Bay) from 2004 to 2013. These are mean values from two SAV monitoring sites in the lower estuary region.

Conclusion

Monitoring over the past 15 years has shown that SAV in all regions of the CRE responds to variations in salinity related to S-79 flows. Although each SAV species occurring in the CRE has its own optimal level of salinity, data indicate that all species respond positively to reduced temporal variation in salinity, which is achieved when S-79 flows are maintained within the recommended 450–2,800 cfs envelope. CERP projects allowing consistent within-envelope flows at S-79 should therefore reduce temporal variation in salinity and increase SAV coverage and density in all regions of the estuary.

ST. LUCIE ESTUARY AND SOUTHERN INDIAN RIVER LAGOON

Freshwater flows into the SIRL affect turbidity, color and nutrients, which in turn affect light available to SAV (Crean et al. 2007, RECOVER 2007b). Of these parameters, color and turbidity appear to be the key parameters controlling light attenuation (Crean et al. 2007). Additionally, freshwater discharges reduce the normally high lagoon salinities. Changes in aquatic resources in the SLE were ultimately a response to reduced rainfall in WY 2011 and early WY 2012. The total amount of fresh water entering the two estuaries was less than the long-term average and particularly reduced in WY 2012, when there were no discharges from Lake Okeechobee. In turn, reduced freshwater input drove salinity increases during the WY 2011 dry season and the beginning of WY 2012. In response to increased salinity, numbers of the eastern oyster increased as did salt-tolerant seagrasses such as *Halodule wrightii* (shoal grass) and *Syringodium filiforme* (manatee grass). TN and TP loadings to the SLE were less than the long-term averages, respectively, thereby reflecting reduced basin runoff and inflows from Lake Okeechobee. Additional water quality information can be found in Chapter 10 of the *2013 South Florida Environmental Report – Volume I* available at www.sfwmd.gov/sfer.

St. Lucie Estuary Oyster Monitoring

Introduction

Oysters are monitored in the SLE by the Florida Fish and Wildlife Conservation Commission's (FWC's) Fish and Wildlife Research Institute at locations shown in (**Figure 4-32**). Freshwater inflows into the estuary have been altered from a natural state to one in which inflows are more variable and extreme thus adversely affecting local oyster populations. The estuary has experienced impacts as a result of the altered salinity regime: exposure to high freshwater inflows during the wet season, which leads to a rapid decline in oyster health and abundance, and too little freshwater inflow during the dry season or drought periods, which leads to a gradual increase in predation, disease, and mortality. In the SLE, salinities in the North and South forks of the SLE are too low and salinities in the middle estuary often fall within the optimal salinity range. However, during dry years, like 2011 and 2012, salinities in the middle estuary often exceeded the optimal range.

Results

The established salinity envelope for the SLE is a range of 12 to 20 as measured at the U.S. 1 Roosevelt Bridge, which is located at the junction of the North Fork, South Fork, and middle estuary (**Figure 4-32**). The goal of this salinity envelope is to reestablish a salinity range most favorable for the survival and health of juvenile marine fish, oysters, and SAV. Observed salinities at Fish and Wildlife Research Institute monitoring stations in the SLE closely correlated with surface salinities at the U.S. 1 Roosevelt Bridge (R^2=0.934) but, as expected, salinities at the middle estuary stations were slightly higher and salinities in the forks were lower, up to 2.5 fresher at the most upstream stations, than those measured at the bridge. Preliminary analysis reveals that when 7-day averages of the summed flows from the three major SLE canals (C-23, C-24, and C-44) are approximately 100 cfs or less, salinities exceed the optimal range deemed most favorable for survival and health of juvenile marine fish, oysters, and SAV (**Figure 4-33**). At those low flow rates, most of the variation can likely be attributed to tidal fluctuations and local runoff. As the summed flows increase, salinity at the U.S. 1 Roosevelt Bridge drops precipitously, falling below the desired minimum of 12 at flow rates of 500 to 700 cfs, and reaching lethal salinities of < 5 at flow rates as low as 1,100 cfs. Those values closely resemble those outlined in the 2011 draft revision of the salinity performance measure, which suggested minimum inflows of 175 cfs and maximum inflows of 1,300 cfs.

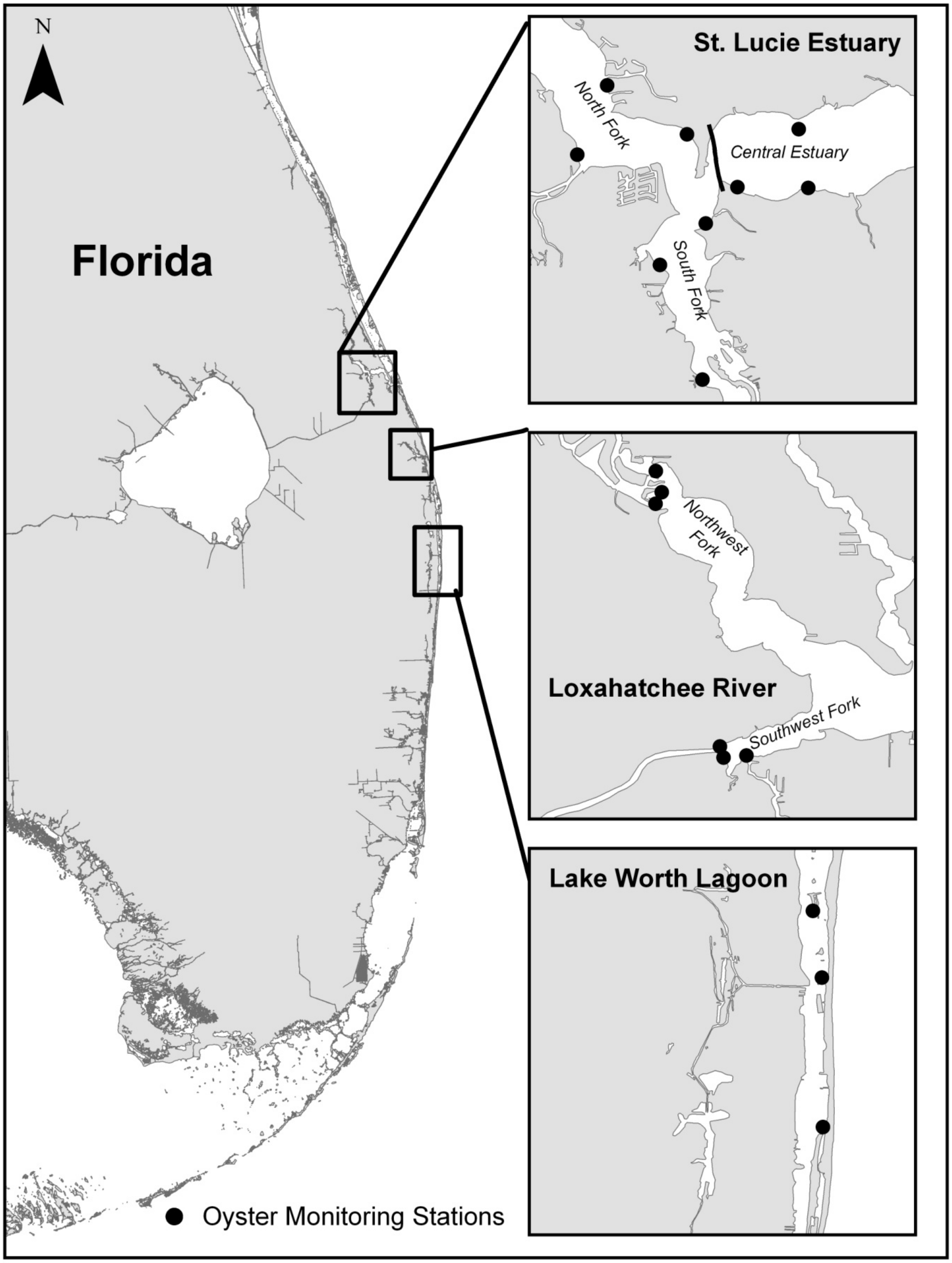


Figure 4-32. Fish and Wildlife Research Institute oyster monitoring study sites and stations on the east coast of Florida.

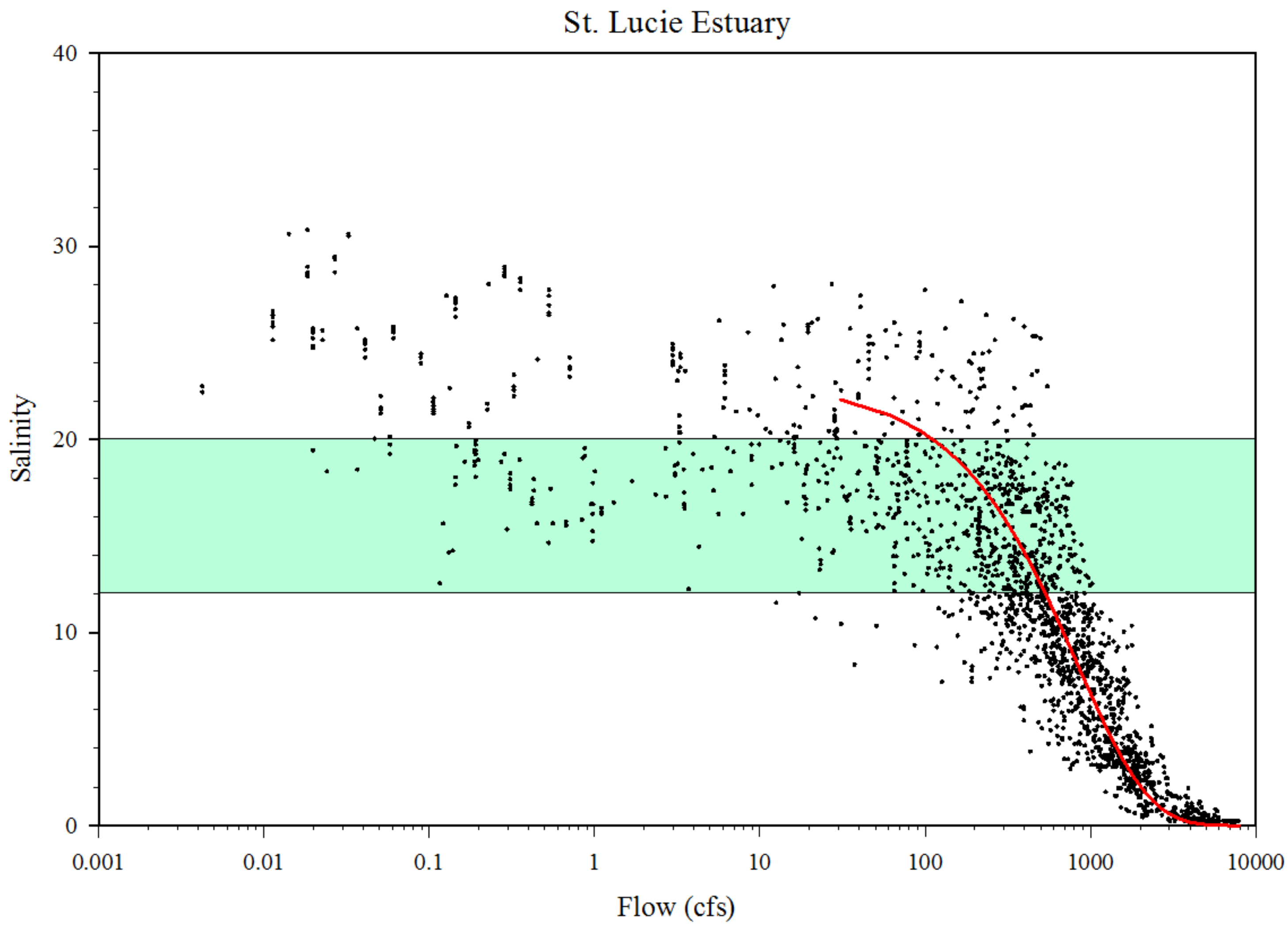


Figure 4-33. Mean daily salinity recorded from the surface at the U.S. 1 Roosevelt Bridge in the SLE (waterdata.usgs.gov/nwis) versus the 7-day averages of the summed flows from the three major SLE canals. The red line shows the best fit regression line for the plotted data. The green band represents the salinity range deemed most favorable for survival and health of juvenile marine fish, oysters, and SAV.

An analysis of salinities recorded at the U.S. 1 Roosevelt Bridge from WY 2006 to WY 2013 shows that mean annual salinity was below the optimal range in WY 2006, above the optimal range in WY 2007, and within the optimal range for the remaining water years. (A water year begins on May 1 and ends on April 30 of the following year.) However, more detailed examination reveals that there were frequent excursions below the optimal range in WY 2009 (July–October), WY 2010 (August–September), WY 2011 (April-July) and WY 2013 (September–October) and protracted periods when salinities exceeded the optimal range in WY 2009–WY 2013 (**Figure 4-34**). In many cases, an extreme low salinity event and a prolonged high salinity event occurred within the same water year in the SLE. The biological responses of oysters to these salinity fluctuations vary depending on the parameter, and the magnitude and duration of the excursion. One to two months below and only a few months above the salinity envelope are enough to evoke an acute, measurable response suggesting that oysters are impacted on shorter time scales than a water year. In an effort to relate the biological responses of oysters to changes in salinity, each measured biological parameter has been categorized into thresholds representing good (green), fair (yellow), or poor (red) for oysters in the SLE (**Table 4-4**) (Volety et al. 2009).

Table 4-4. Thresholds for measured biological parameters of oysters in the three SLE study sites: the North Fork, South Fork, and middle estuary. Please note that thresholds for density differ for the middle estuary (Volety et al. 2009).

Parameter	Poor (Red)	Fair (Yellow)	Good (Green)
Density North Fork and South Fork	0-20 oysters per square meter (/m^2)	21-100 oysters/m^2	> 100 oysters/m^2
Density Middle Estuary	0-100 oysters/m^2	101-500 oysters/m^2	> 500 oysters/m^2
Recruitment Rate (May–October)	0-1 spat per shell per month	> 1-5 spat per shell per month	> 5 spat per shell per month
Dermo Infection Prevalence	> 50%	21–50%	0-20%
Dermo Infection Intensity	Heavy	Moderate	Uninfected to Light

The density of live oysters in the SLE is typically an order of magnitude higher in the middle estuary than in the North and South forks of the estuary. In the forks, live densities rarely meet the good (green) threshold and most often fall into the fair (yellow) or poor (red) category (**Table 4-4** and **Figures 4-35** and **4-36**). In the middle estuary, the density thresholds are set higher than those of the North and South forks and densities of live oysters most commonly fall into the caution category (**Table 4-4** and **Figure 4-37**). Due to the semi-annual frequency of density sampling, analyses based on water years may be appropriate as long as short-term low salinity events are considered. For example, mean salinity in WY 2013 fell within the optimal range but the low salinity event that occurred from September to October 2012 adversely impacted oyster densities in the North and South forks where salinities were < 5 during that period.

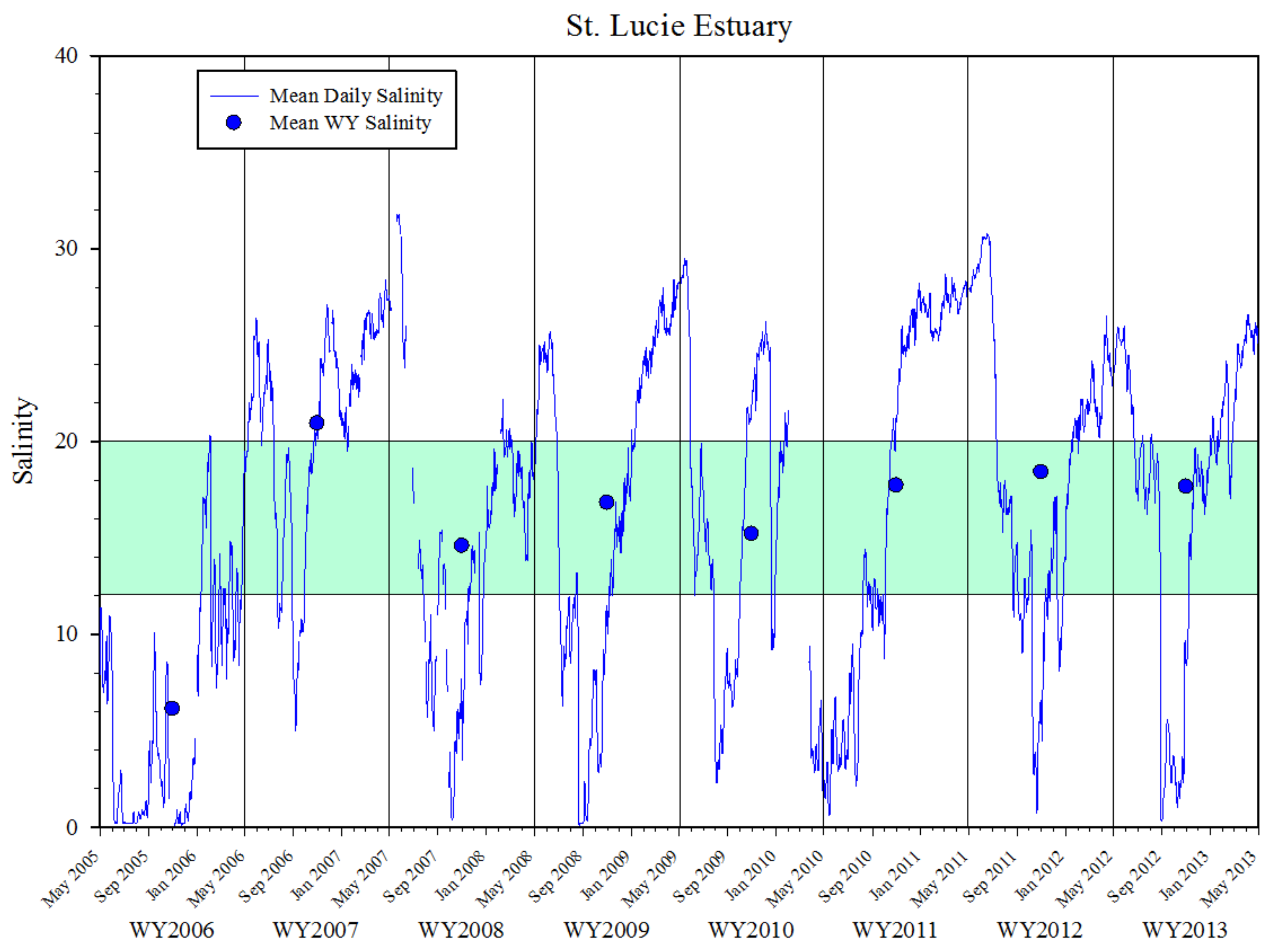


Figure 4-34. Mean daily salinity and mean water year salinity recorded from the surface at the U.S. 1 Roosevelt Bridge in the SLE (waterdata.usgs.gov/nwis). The green band represents the salinity range deemed most favorable for survival and health of juvenile marine fish, oysters, and SAV.

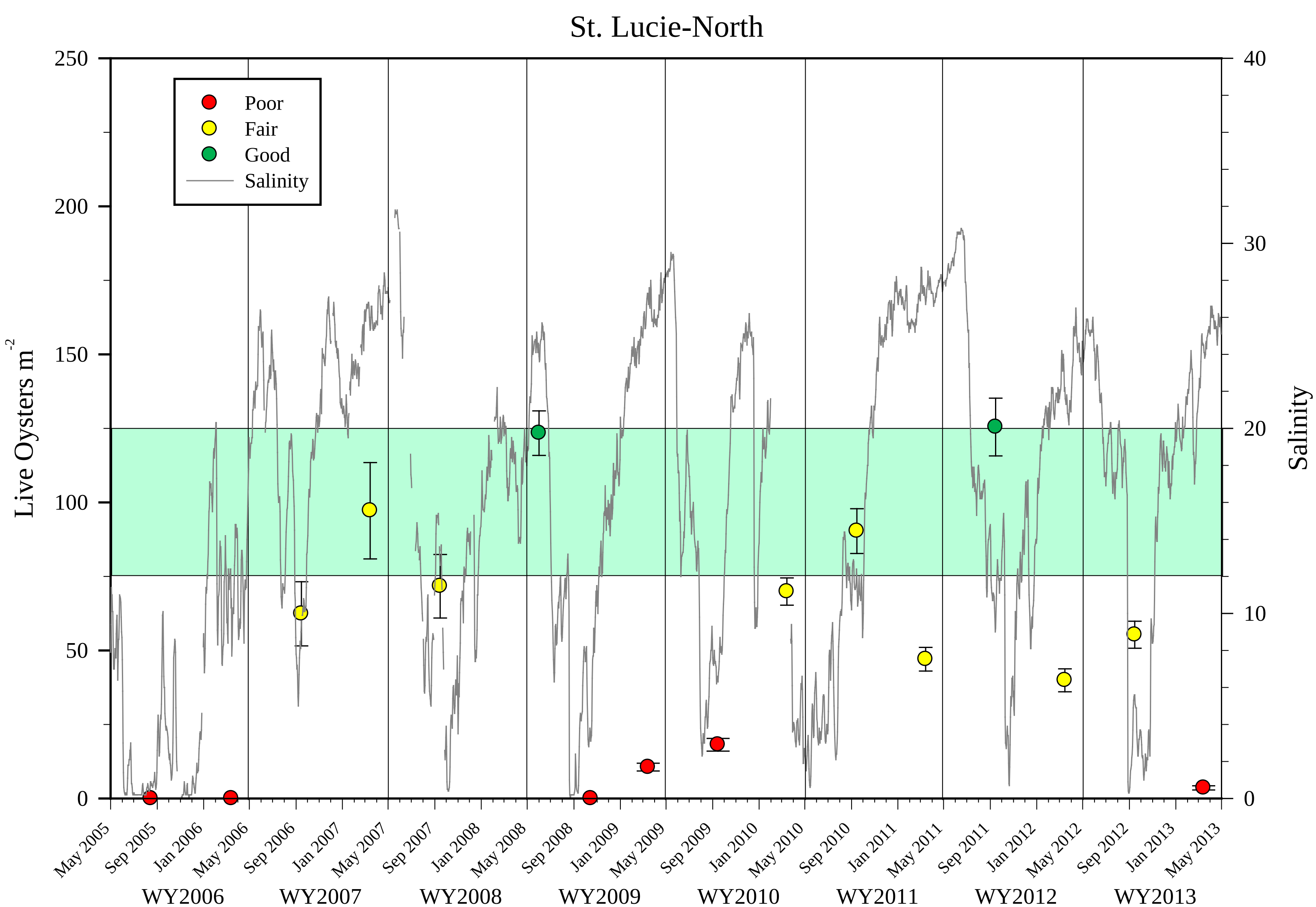


Figure 4-35. Mean number (± standard deviation) of live oysters (red, yellow, and green circles) in the North Fork of the SLE during semi-annual surveys and daily salinity from the surface at the US1 Roosevelt Bridge. The green band represents the salinity range at the U.S. 1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

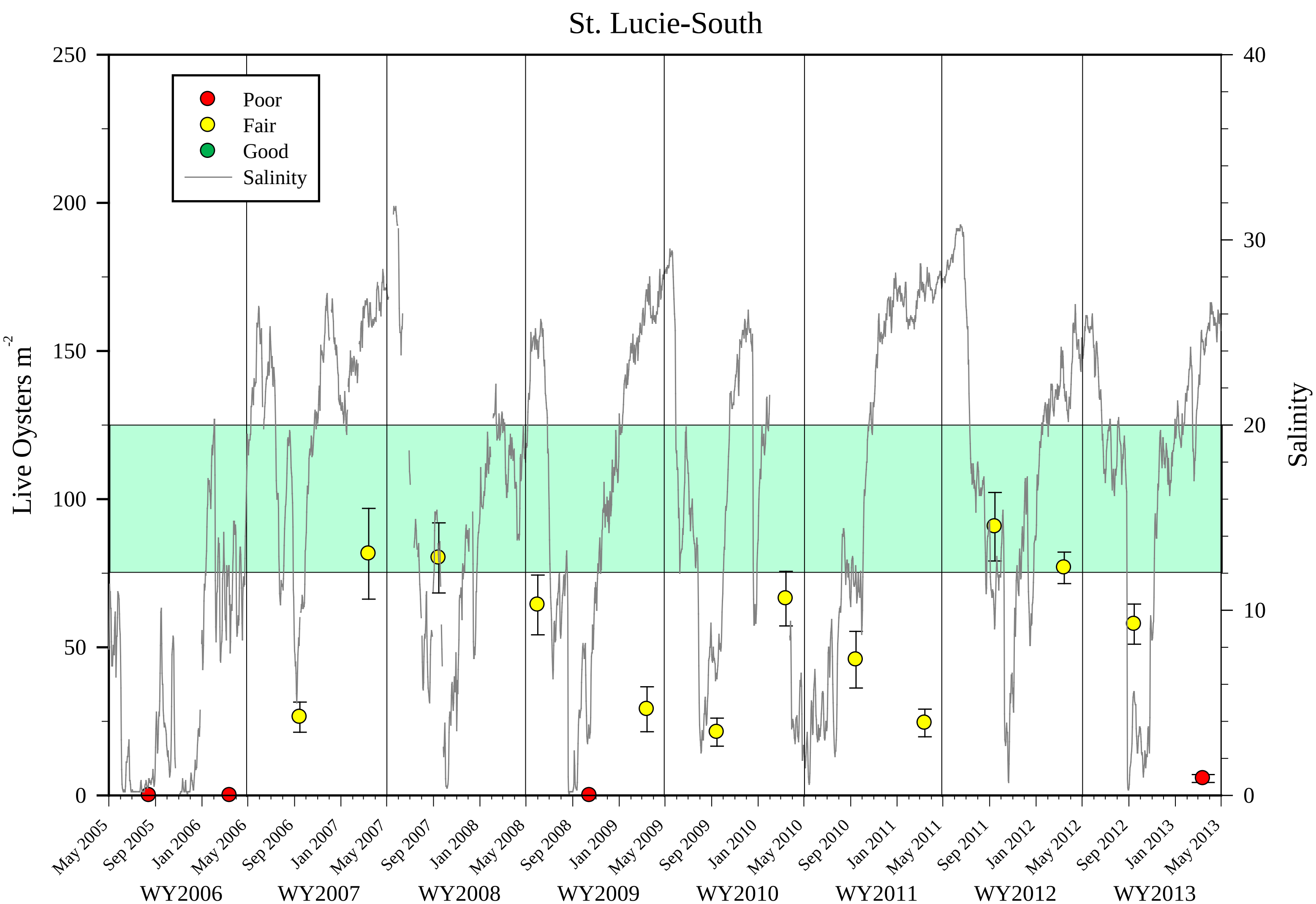


Figure 4-36. Mean number (± standard deviation) of live oysters (red, yellow, and green circles) in the South Fork of the SLE during semi-annual surveys and daily salinity from the surface at the US1 Roosevelt Bridge. The green band represents the salinity range at the U.S. 1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

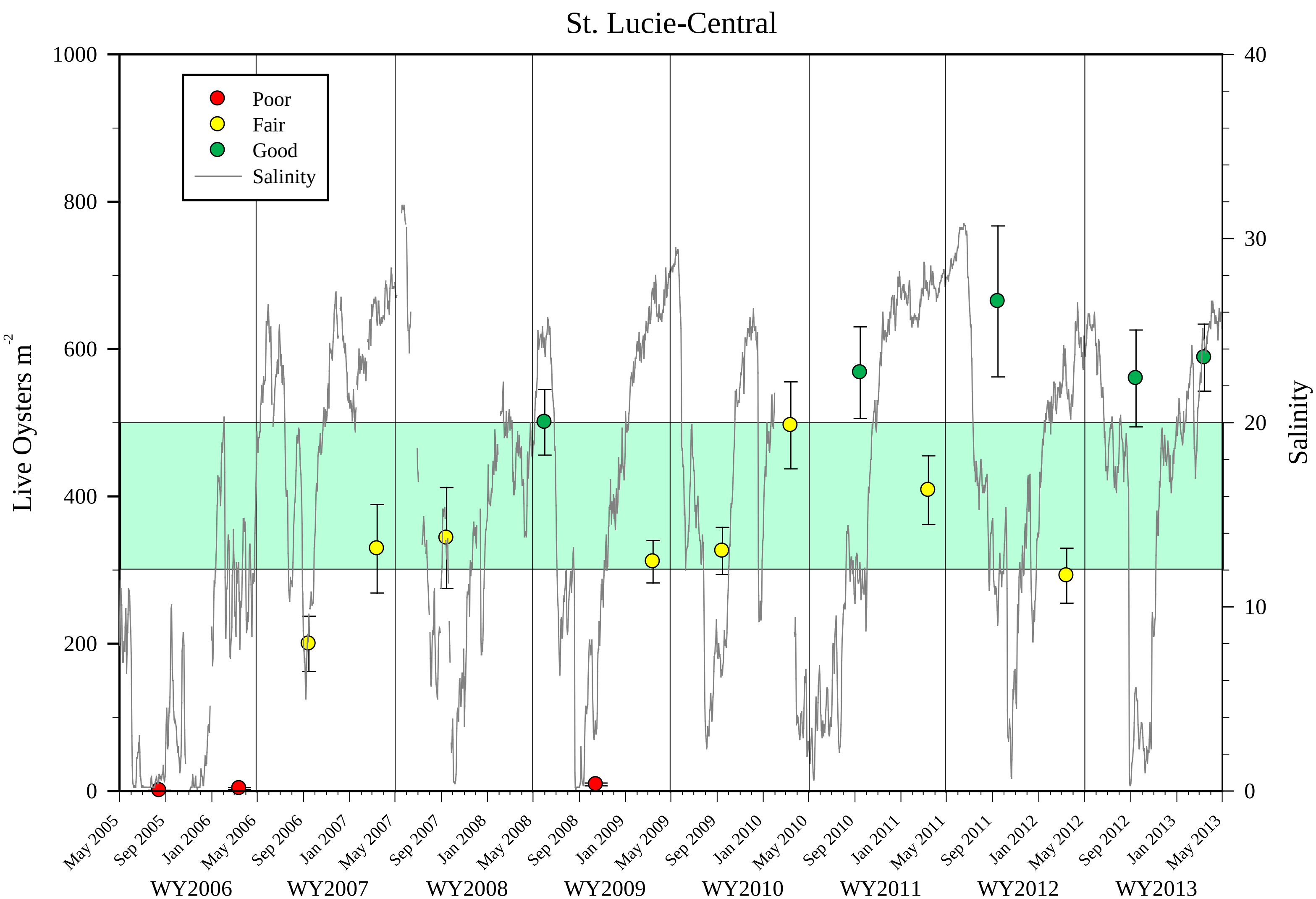


Figure 4-37. Mean number (± standard deviation) of live oysters (red, yellow, and green circles) in the SLE middle estuary during semi-annual surveys and daily salinity from the surface at the US1 Roosevelt Bridge. The green band represents the salinity range at the U.S. 1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

The timing of reproductive development and larval recruitment in the SLE is similar among oysters in the forks and the middle estuary. Peak reproductive activity typically occurs between April and September and is usually higher in the months during or after a period with higher salinities. For example, the percentage of oysters in the gonadal development stage was higher in WY 2010 (October–February), WY 2011/WY 2012 (November–July) and in the spring of calendar year 2012 (**Figures 4-38** through **4-40**). Larger percentages of developing oysters generally coincided with higher recruitment rates, especially in the North Fork during the latter parts of WY 2011 (February–March 2011) and WY 2012 (February–May 2012), but recruitment rates are generally low in the SLE and most commonly classified in the poor category (**Table 4-4** and **Figures 4-41** through **4-43**). In WY 2010, despite several months with a high percentage of developing oysters preparing to spawn, recruitment rates remained low suggesting that the adult oysters spawned but that the larvae did not survive in the low salinity environment and/or they were physically flushed out of the estuary. The greater adult densities measured in the North Fork and the middle estuary following that period of low salinity in WY 2010 support the belief that adult oysters are more tolerant to low salinities (5-10) than larval or juvenile oysters.

The impacts from infection by the parasitic protozoan *Perkinsus marinus* (dermo) were typically higher in the middle estuary than the North and South forks of the SLE. During the first several years of the study (WY 2006–WY 2011), disease prevalence (green; < 20% of oysters infected) and intensity (green; mean of < 1 on the Mackin Scale of infection intensity) rarely exceeded acceptable limits (**Table 4-4** and **Figures 4-44** through **4-50**). Beginning in fall 2010, prolonged periods of higher salinity resulted in ratings of fair and poor in the middle estuary and the two forks (**Figure 4-47**). This indicates that there are detrimental impacts when salinities exceed the optimal range for extended periods and that there may be times when supplemental freshwater inflows would benefit oyster populations. A periodic flush of fresh water can reduce disease prevalence and intensity for one to three months. For example, in late 2012, salinity fell below 5 for several weeks at the U.S. 1 Roosevelt Bridge resulting in a substantial decrease in prevalence over the next three months.

When estuarine conditions allowed for oyster survival, growth rates of tagged oysters planted in the SLE were typically higher in the North and South forks than in the middle estuary. Oysters of similar size (~34 millimeter [mm] shell height) planted in February 2011 grew at a rate of < 1 millimeter (mm) per month over the next 12 months in the middle estuary, but at a rate of 1.5 to 2 mm per month in the North and South forks. In the following year, oysters grew more rapidly in all areas but mean growth rates remained higher in the forks. This indicates that growth rates were lower in regions during periods with higher salinities. Mortality of planted oysters was generally higher in open cages than in closed cages suggesting that there is macrofaunal predation. However, in the SLE, rapid decreases in salinity commonly killed tagged oysters in both closed and open cages making an accurate determination of predation pressure impossible to assess.

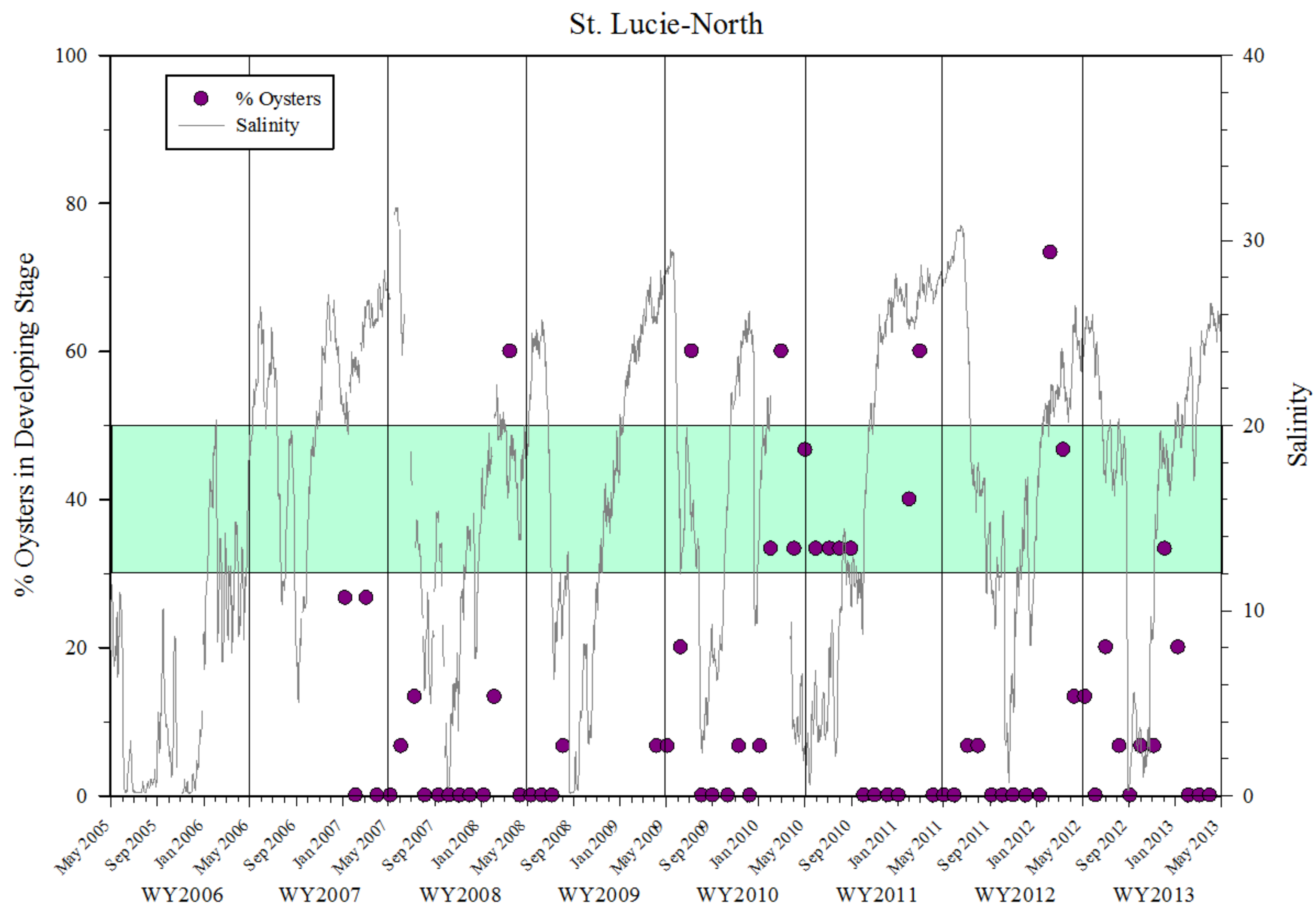


Figure 4-38. Reproductive development, represented as the percentage of oysters in the gonadal development stage, of oysters collected from the North Fork of the SLE each month and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

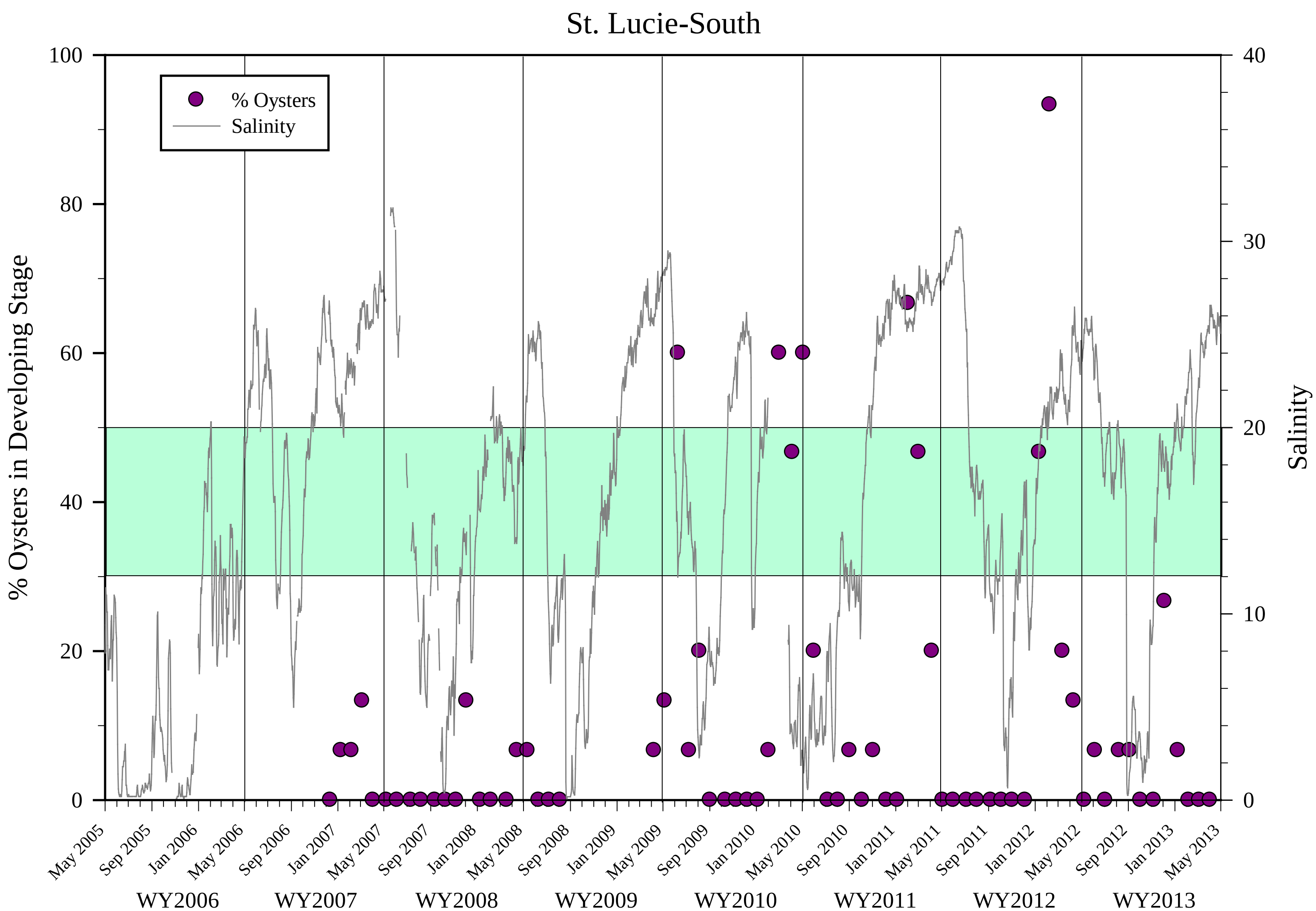


Figure 4-39. Reproductive development, represented as the percentage of oysters in the gonadal development stage, of oysters collected from the South Fork of the SLE each month and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

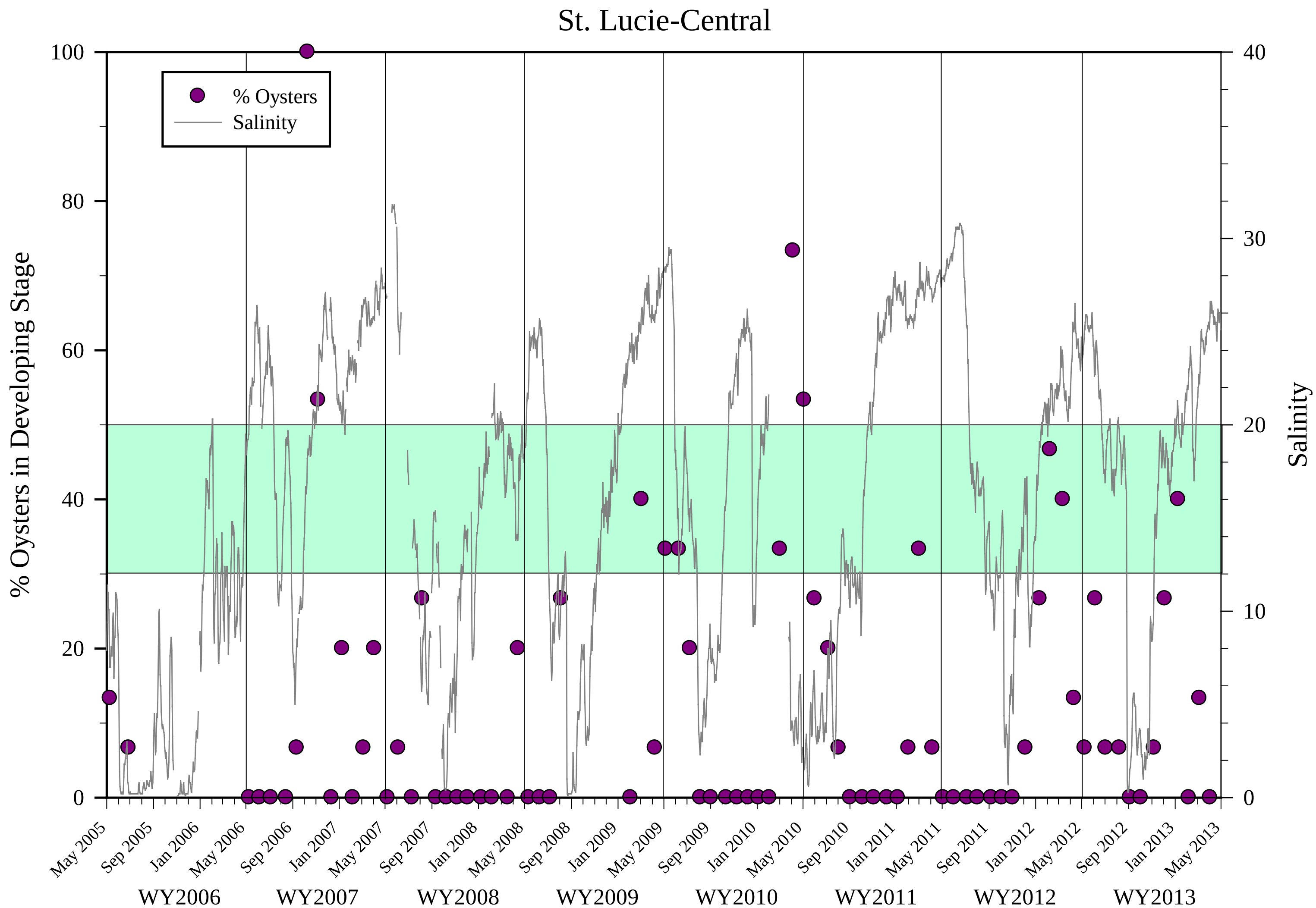


Figure 4-40. Reproductive development, represented as the percentage of oysters in the gonadal development stage, of oysters collected from the SLE middle estuary each month and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

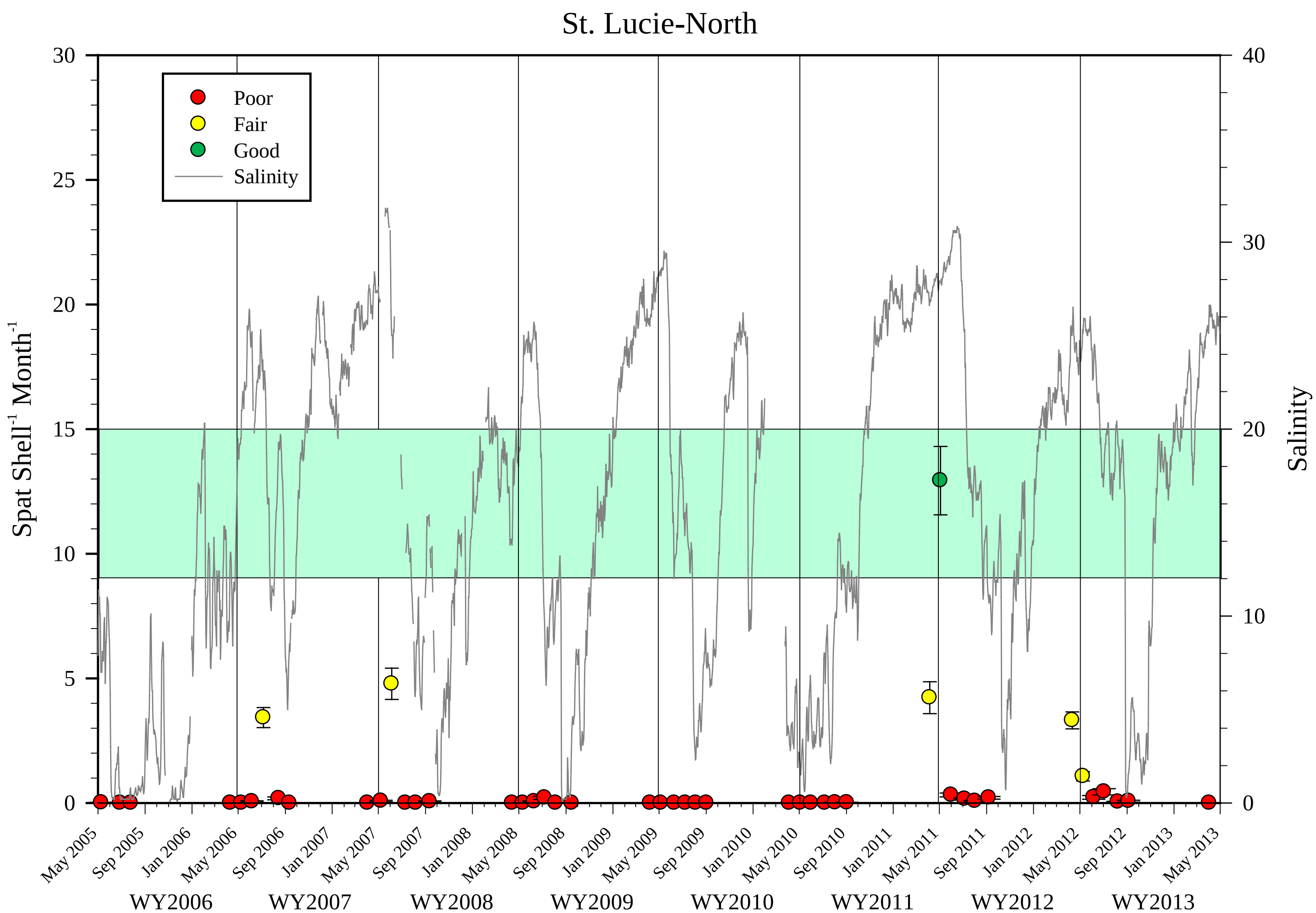


Figure 4-41. Mean number (± standard deviation) of oyster recruits per shell (red, yellow, and green circles) during monthly collections from April through September of each calendar year in the North Fork of the SLE and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

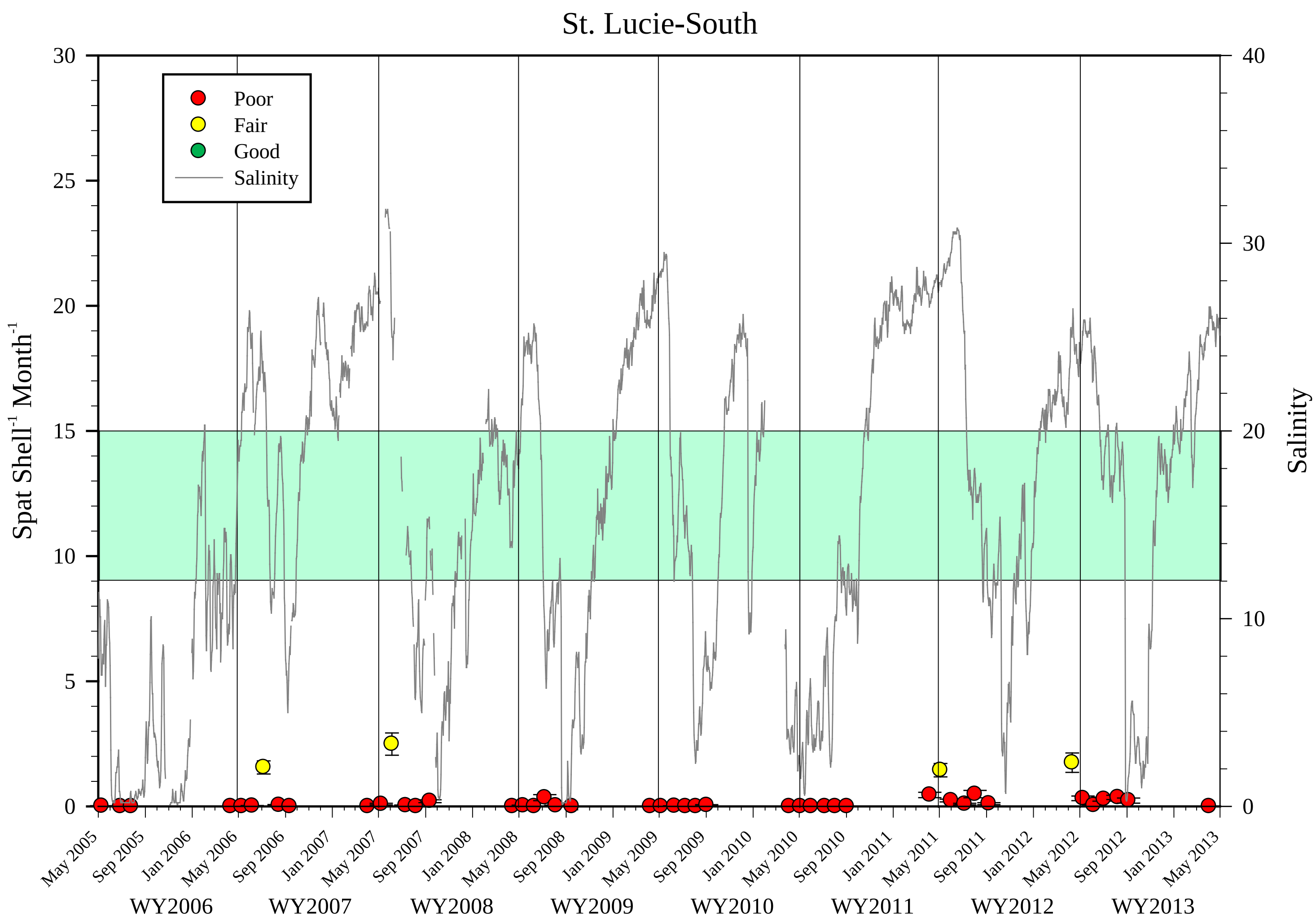


Figure 4-42. Mean number (± standard deviation) of oyster recruits per shell (red, yellow, and green circles) during monthly collections from April through September of each calendar year in the South Fork of the SLE and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

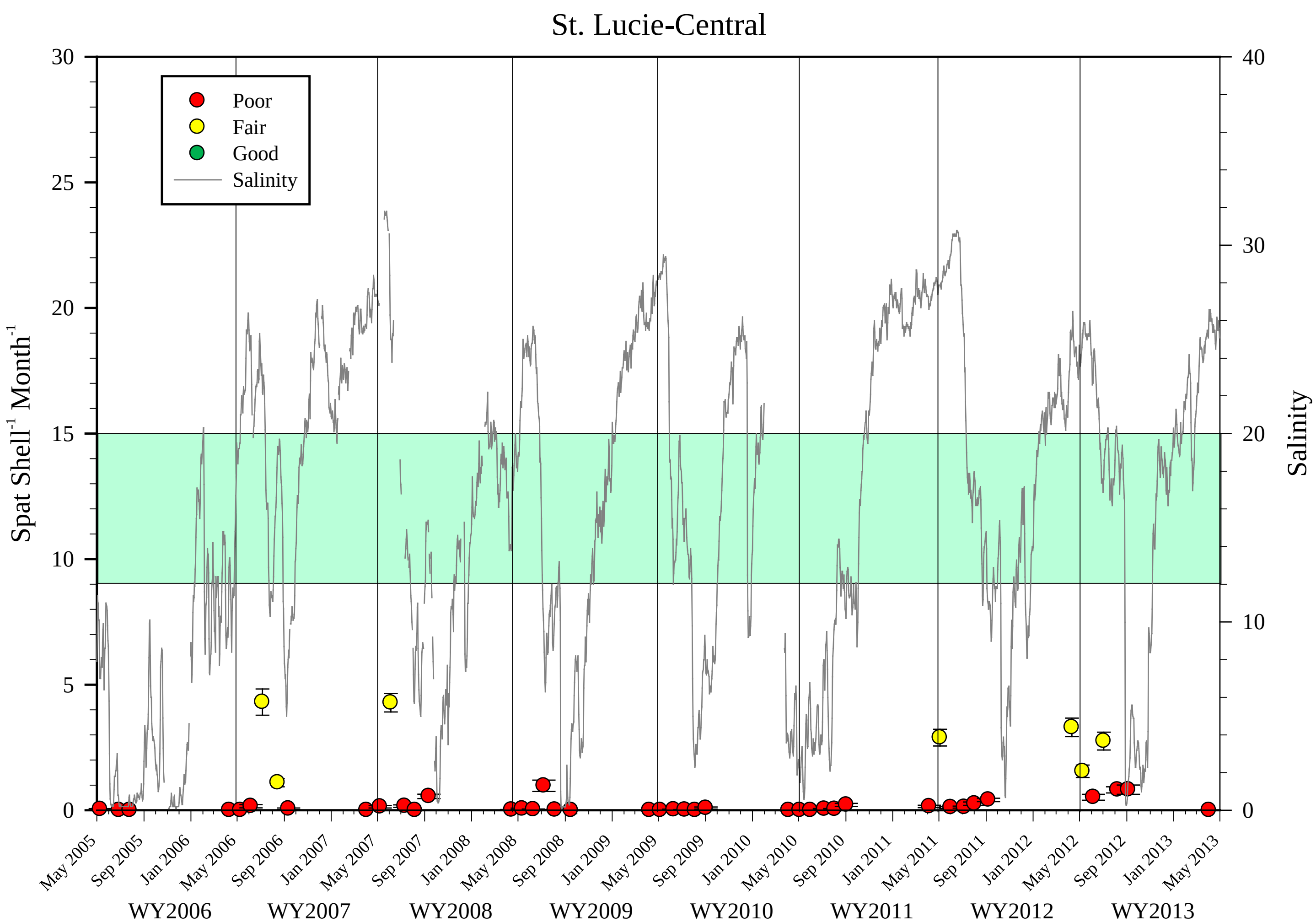


Figure 4-43. Mean number (± standard deviation) of oyster recruits per shell (red, yellow, and green circles) during monthly collections from April through September of each calendar year in the SLE middle estuary and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

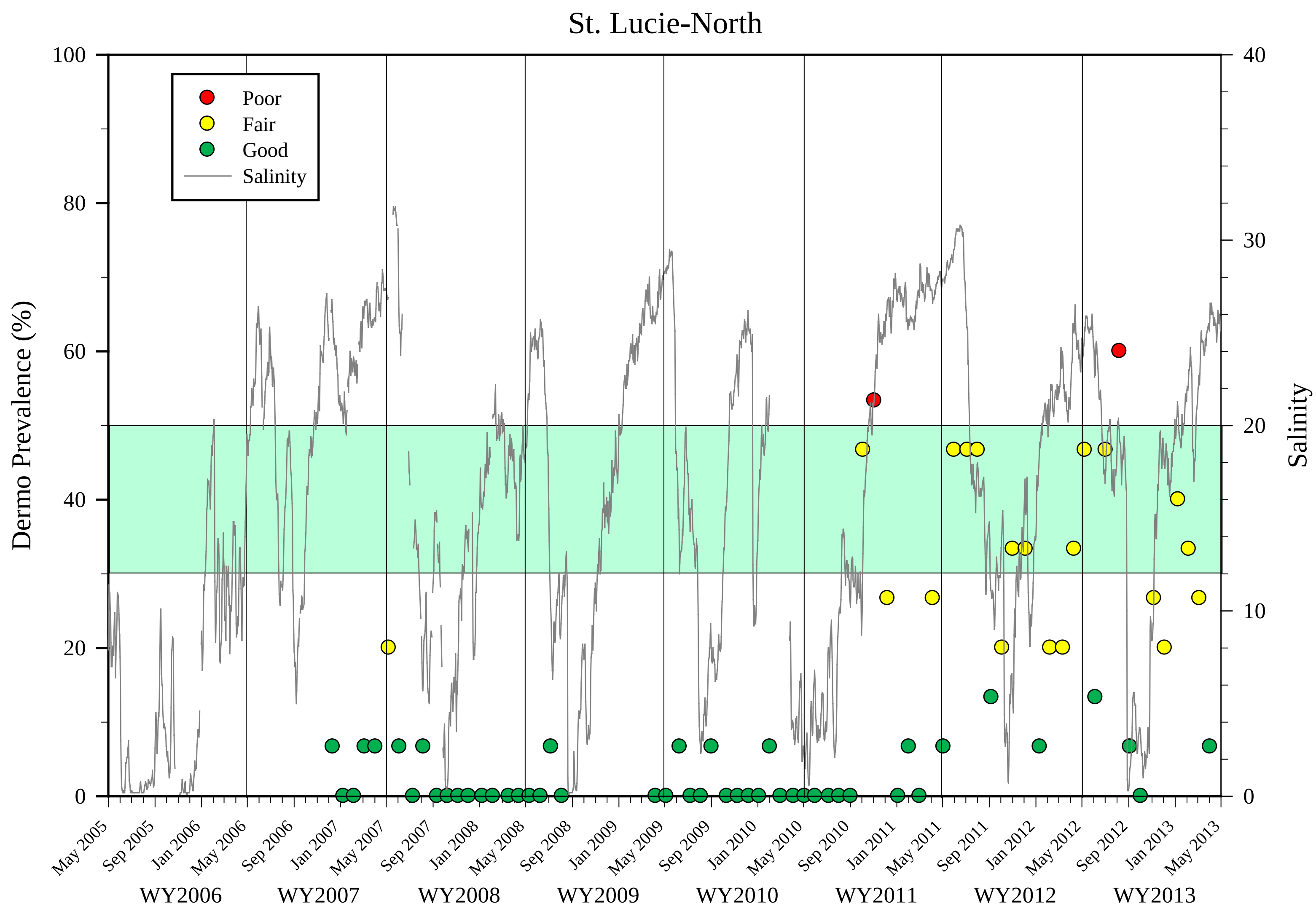


Figure 4-44. Monthly prevalence (%; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the North Fork of the SLE and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

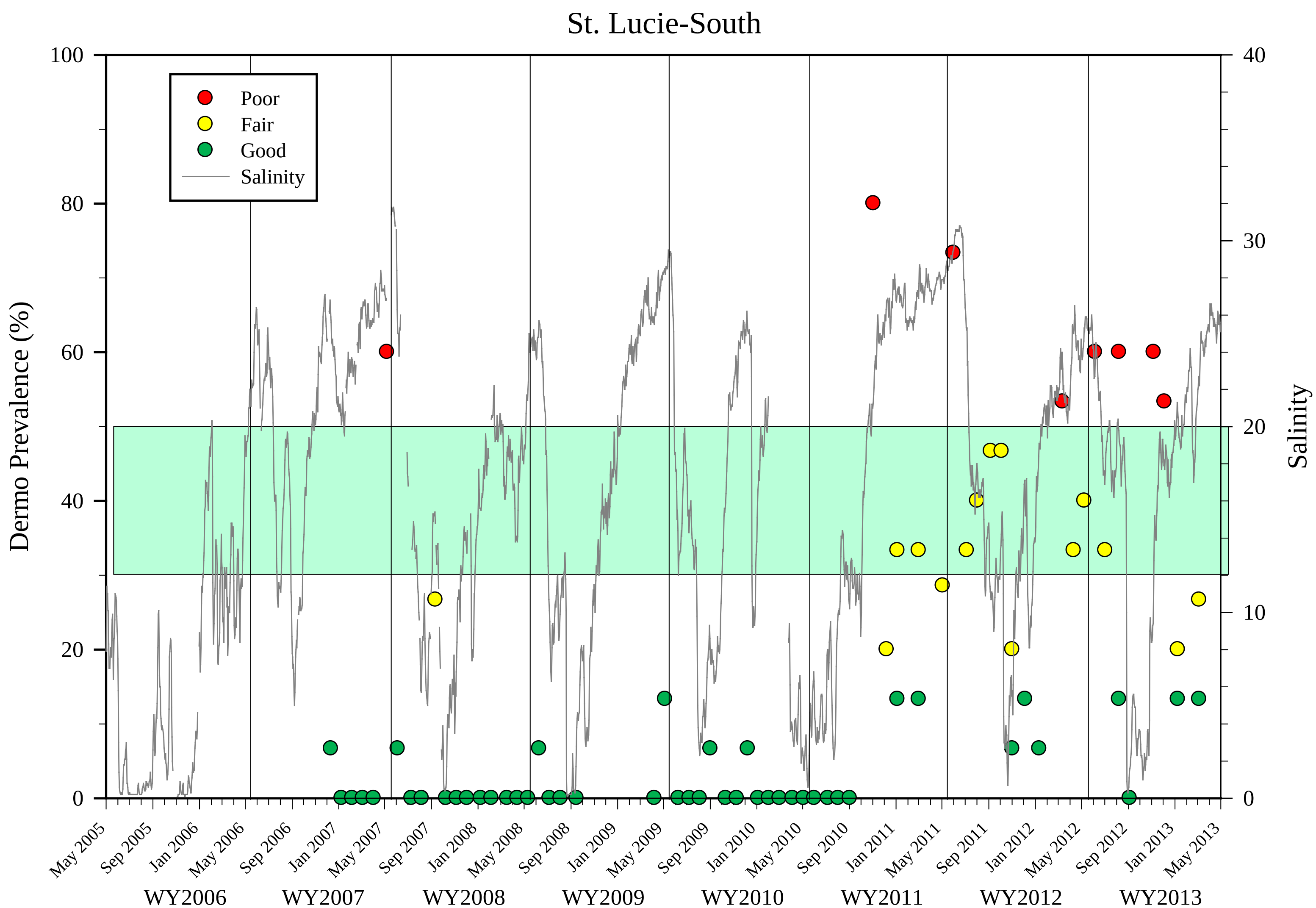


Figure 4-45. Monthly prevalence (%; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the South Fork of the SLE and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

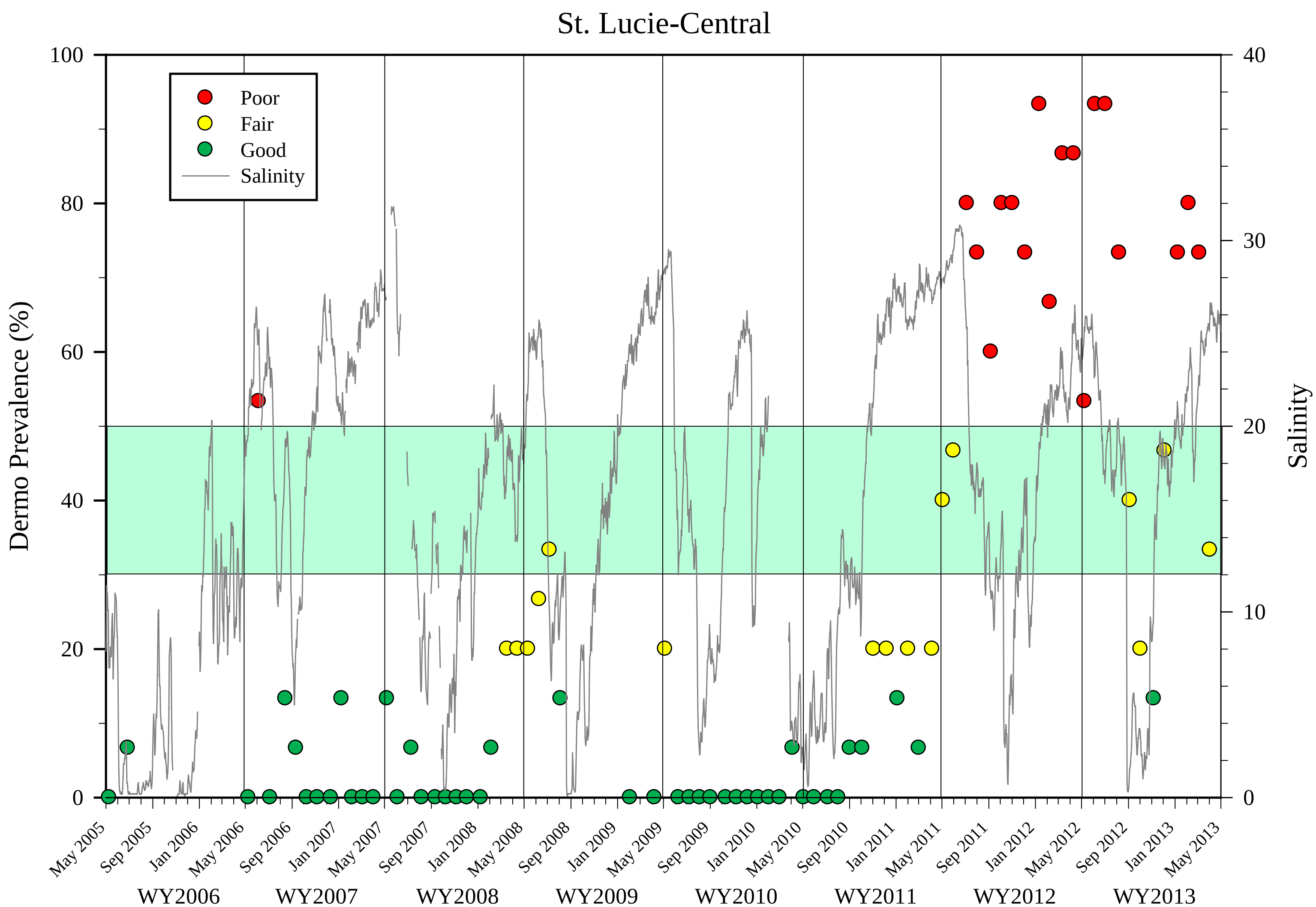


Figure 4-46. Monthly prevalence (%; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the SLE middle estuary and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

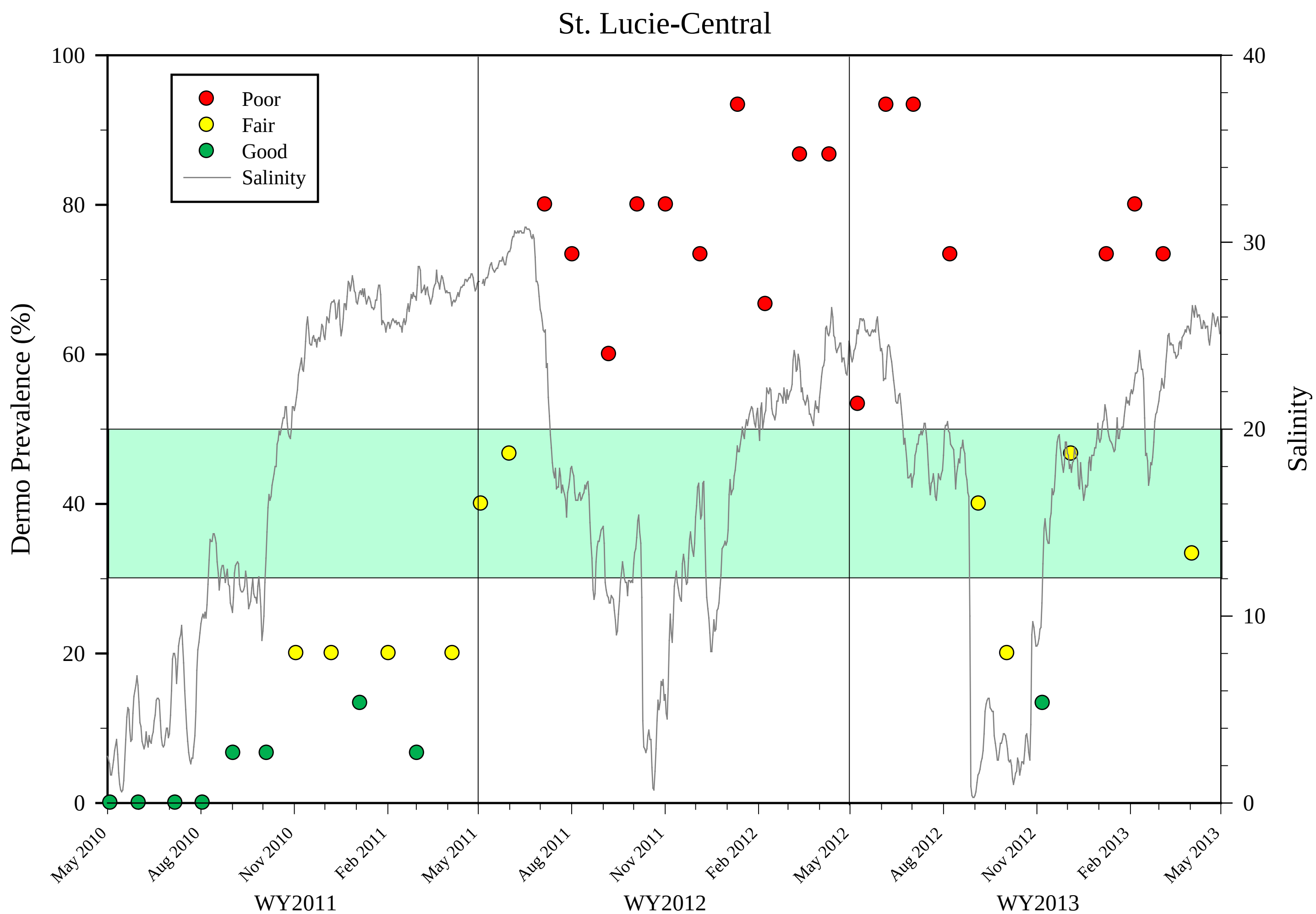


Figure 4-47. Monthly prevalence (%; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the SLE middle estuary and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. Daily mean salinity exceeded the optimum range of 12–20 at the US1 Roosevelt Bridge from October 30, 2010–July 10, 2011 (254 days) leading to significant increases in *P. marinus* prevalence among oysters in the estuary as temperatures increased in spring and summer 2011. The magnitude and duration of this dry period far exceeds any other dry period recorded during this study and provides first time evidence suggesting higher than optimal salinities can be detrimental to oysters in the SLE.

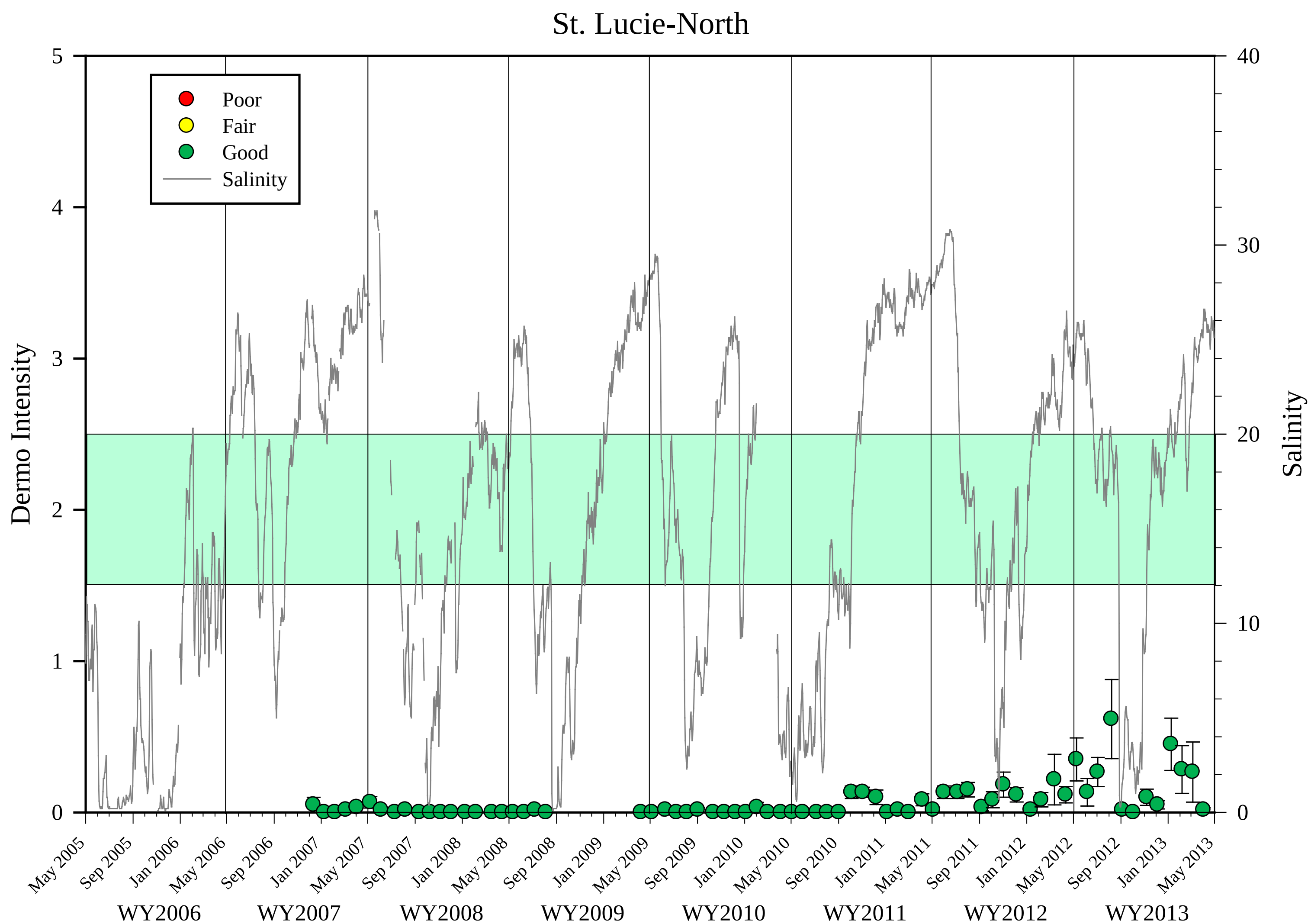


Figure 4-48. Monthly mean infection intensity (± standard deviation; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the North Fork of the SLE and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

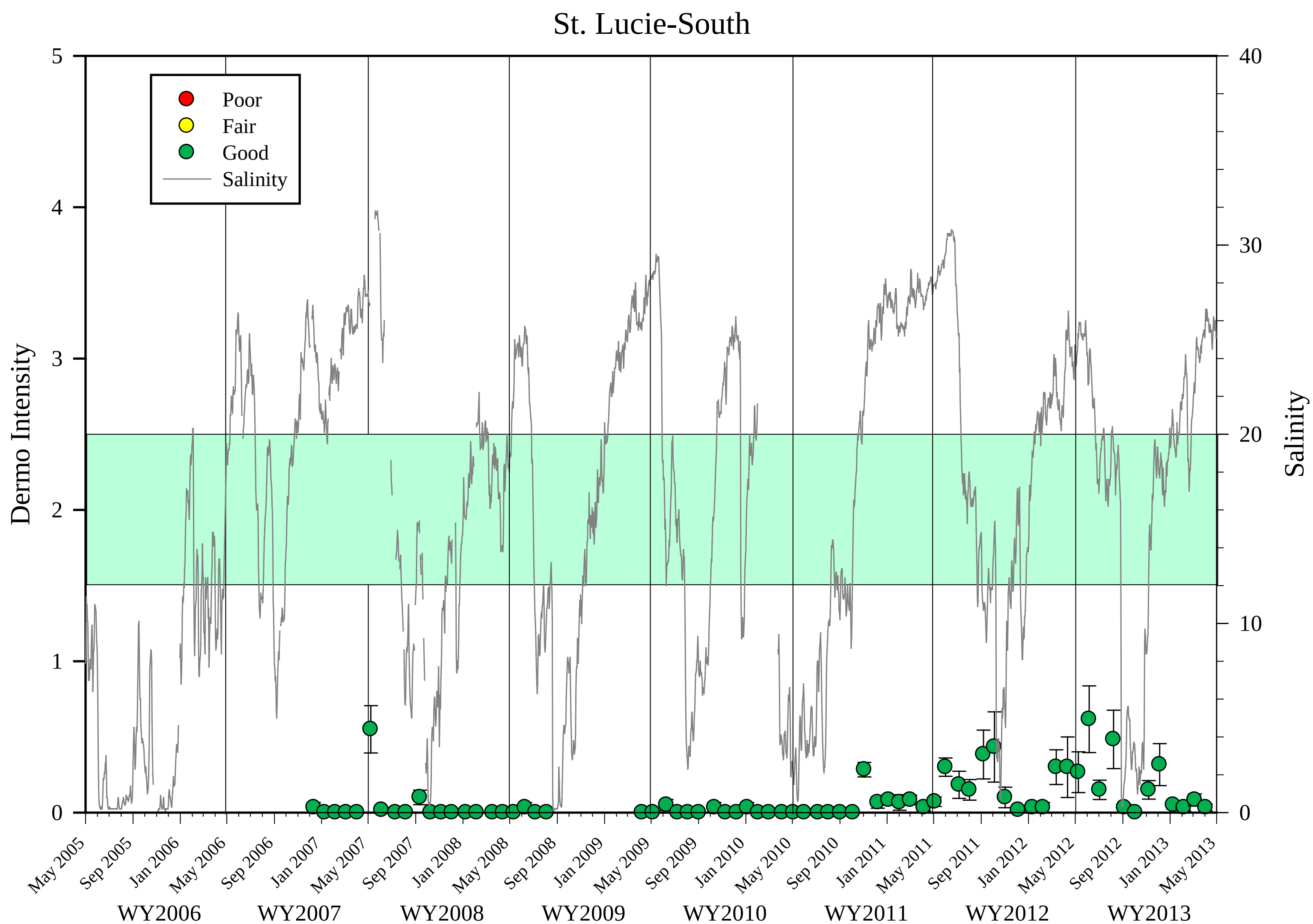


Figure 4-49. Monthly mean infection intensity (± standard deviation; red, yellow, and green circles) of oysters infected with P*erkinsus marinus* during monthly collections from the South fork of the SLE and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

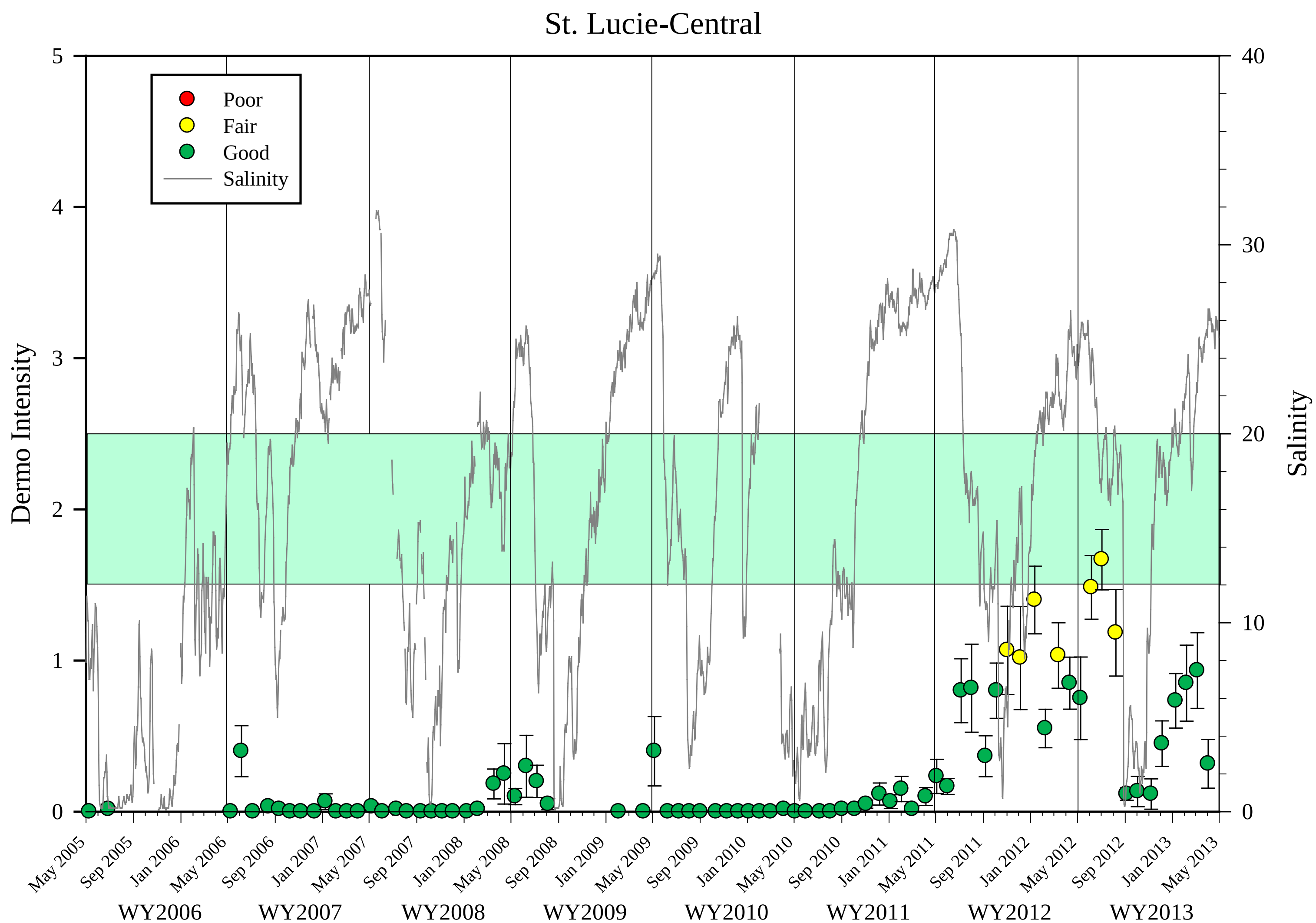


Figure 4-50. Monthly mean infection intensity (± standard deviation; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the SLE middle estuary and daily salinity from the surface at the U.S. 1 Roosevelt Bridge. The green band represents the salinity range at the US1 Roosevelt Bridge deemed most favorable for oyster survival and health in the estuary.

Conclusion

Oyster populations in the SLE have been negatively impacted by the highly variable freshwater inflows that are a result of the altered local hydrology. Periods of extremely high flow result in acute damage to oyster populations. Extended periods of reduced flow result in gradual increases in disease and predation rates that result in compromised oyster health and survivorship. The variability in and of itself can compound the problem because rapid shifts between dry and wet regimes reduce the opportunity for acclimatization by the oyster and other estuarine inhabitants. In the SLE, low salinity events have had the most devastating impact on oysters but in recent years prolonged high salinity events have also occurred.

Several steps can be taken to strengthen understanding of the relationship between flow and salinity in the estuary. For example, estuarine conditions are commonly summarized by water year, but oysters are often impacted on shorter time scales. In many cases, an extreme low salinity event and a prolonged high salinity event occurred within the same water year; therefore, a more detailed examination will likely yield a better understanding of how changes in salinity impact oyster biology over varying time scales. The estuary also has multiple potential sources of freshwater input and these should be considered in sum when interpreting freshwater inflow impacts on water quality, oysters and other estuarine fauna. Finally, collection of frequent and continuous water quality data (salinity, temperature) directly over oyster beds within the estuary, as attained by the data logger on the U.S. 1 Roosevelt Bridge in the SLE, would allow for development and/or refinement of minimum/maximum flow rates to better prevent extreme salinity changes that lead to conditions that are detrimental to oysters.

St. Lucie Estuary Seagrass

Background

In 2007 and 2008, ten seagrass monitoring stations were established in the SLE and SIRL by the RECOVER program (**Table 4-5** and **Figure 4-51**). The monitoring was designed to document current seagrass conditions in areas where CERP project construction is expected to benefit seagrass resources. Eight sites are in the vicinity of the SLE and the remaining two near the Taylor Creek (C-25) discharge. Sites are 1 to 2 acres in size with average depths ranging from 36 to 110 cm.

Table 4-5. Seagrass monitoring site characteristics.

Site Location	Site Name	Acres	Hectares	Mean Depth (cm)	Seagrass Found at Site
St. Lucie Inlet Northeast	SLI_NE	2	0.8	36	*Syringodium filiforme* (manatee grass) *Halodule wrightii* (shoal grass) *Thalassia testudinum* (turtle grass) *Halophila johnsonii* (Johnson's seagrass)
St. Lucie Inlet Southeast	SLI_SE	1	0.3	40	*Halodule wrightii* (shoal grass) *Halophila johnsonii* (Johnson's seagrass)
Fort Pierce Inlet Northwest	FP_NW	2	0.7	54	*Halodule wrightii* (shoal grass) *Syringodium filiforme* (manatee grass) *Thalassia testudinum* (turtle grass) *Halophila engelmanni* (star grass) *Halophila decipiens* (paddle grass)
Southwest of St. Lucie Inlet Southeast	SITE_3	1	0.3	56	*Syringodium filiforme* (manatee grass) *Halodule wrightii* (shoal grass) *Halophila johnsonii* (Johnson's seagrass)
Fort Pierce Inlet Northeast	FP_NE	2	0.8	61	*Syringodium filiforme* (manatee grass) *Halodule wrightii* (shoal grass) *Thalassia testudinum* (turtle grass)
Northwest of Boy Scout Island	SITE_1	2	0.8	72	*Syringodium filiforme* (manatee grass) *Halodule wrightii* (shoal grass) *Halophila johnsonii* (Johnson's seagrass)
Boy Scout Island	BSI	2	0.8	77	*Syringodium filiforme* (manatee grass) *Halodule wrightii* (shoal grass) *Halophila johnsonii* (Johnson's seagrass)
Willoughby Creek	WILL_CK	2	0.8	77	*Halodule wrightii* (shoal grass) *Halophila johnsonii* (Johnson's seagrass)
Ocean Breeze Park	OC_BR_PK	2	0.8	82	*Syringodium filiforme* (manatee grass) *Halodule wrightii* (shoal grass) *Halophila johnsonii* (Johnson's seagrass)
Joes Point	JOES_PT	2	0.7	110	*Halodule wrightii* (shoal grass) *Syringodium filiforme* (manatee grass) *Thalassia testudinum* (turtle grass) *Halophila engelmanni* (star grass) *Halophila decipiens* (paddle grass)

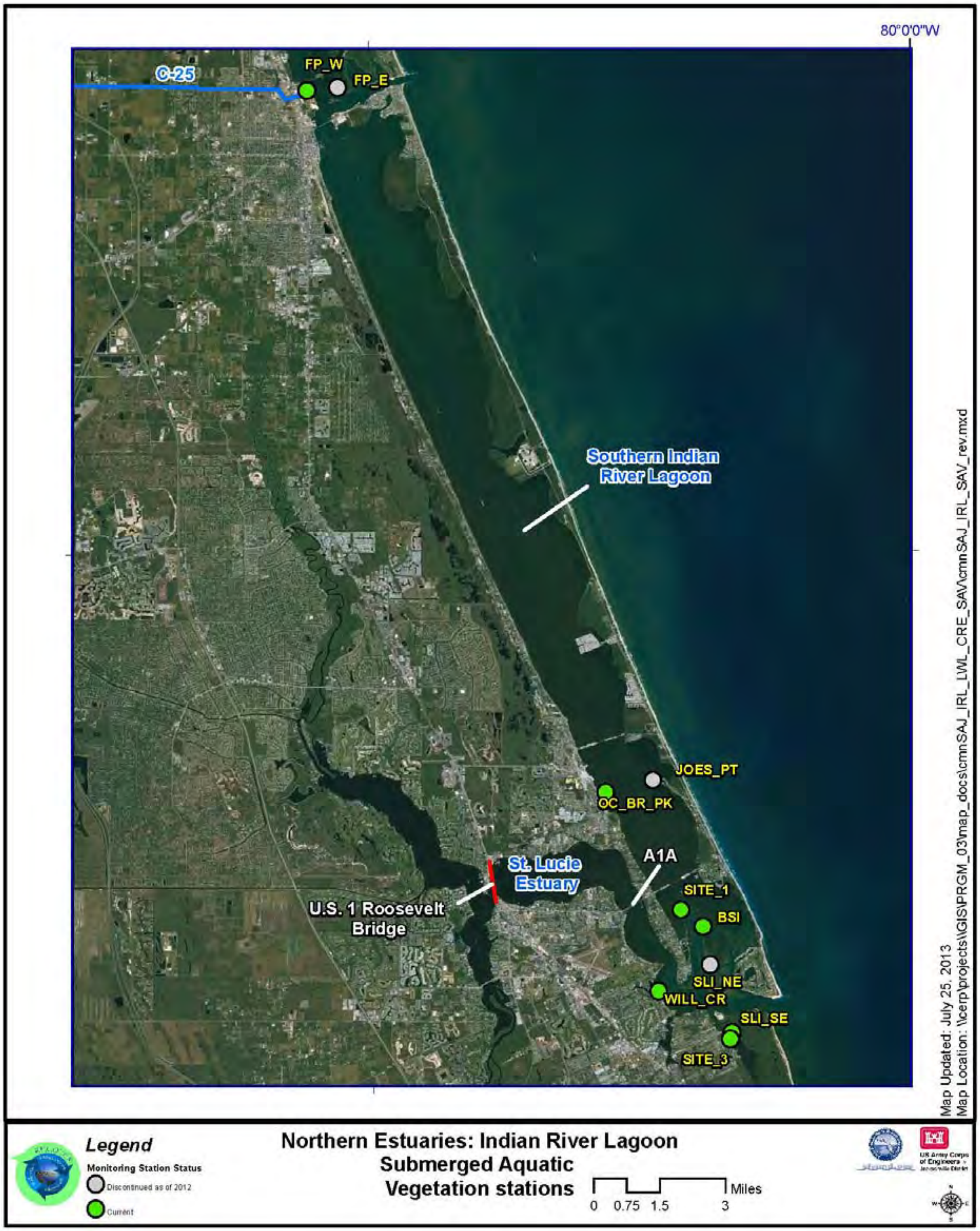


Figure 4-51. Location of seagrass monitoring sites in SLE and SIRL.

Initial monitoring occurred at various times in 2007 and 2008. Beginning in December 2008, monitoring was done bimonthly at most sites, with monthly monitoring at the three sites closest to the influence of the SLE (BSI, WILL_CR, and SLI_SE). In April 2012, the monitoring frequency was changed as reflected in **Table 4-6** and monitoring was discontinued at three sites (FP_NE, JOES_PT, and SLI_NE).

Table 4-6. Revised monitoring schedule initiated in April 2012.

Site Name	January	February	March	April	May	June	July	August	September	October	November	December
SIRL Water Quality	■	■		■		■	■	■		■		
FP_NW				•		•		•		•		
OBP				•		•		•		•		
SITE_1				•		•		•		•		
BSI	•			•	•	•	•	•	•	•		
WILL_CR [a]	•			•	•	•	•	•	•	•		
SLI_SE	•			•	•	•	•	•	•	•		
SITE_3				•		•		•		•		

a. Water quality associated with this location is collected monthly (SE O1)

The monitoring protocol was developed by a team of RECOVER scientists. At each site, thirty points distributed across the site are monitored using 1- m^2 quadrats subdivided into 25 equal quadrants. The number of quadrants containing seagrass are counted and recorded. Seagrass percent occurrence per quadrat is determined by dividing the number of quadrants occupied by seagrass by the total possible quadrants (25) then multiplying by 100. To determine the percent occurrence for the entire site, the quadrat percent occurrences (N = 30) are averaged for each monitoring event. Canopy height is also measured within each quadrat and a site average calculated (N = 30).

Evaluation

RECOVER seagrass data (2007/2008–April 30, 2013) and associated salinity data were used to evaluate the appropriateness of SLE salinity targets for protecting seagrass. Questions addressed included the following:

- Does seagrass in the SLE/SIRL respond as predicted when CERP targets are met?
- Has our understanding of how seagrass responds to salinity changed?
- Are the SLE salinity targets appropriate for seagrass?
- How long does it take a seagrass species to recover from an adverse impact caused by an extreme salinity event?

Targets

Salinity targets were set for the SLE based on salinity ranges favorable to juvenile marine fish, oysters, and seagrass. Current targets call for maintaining salinity between 12 and 20, which is referred to as a "salinity envelope" at the U.S. 1 Roosevelt Bridge. Seagrass distribution in the SLE is limited and currently no beds occur near the U.S. 1 Roosevelt Bridge. The most upstream, persistent seagrass bed in the estuary is approximately six miles downstream of the U.S. 1 Roosevelt Bridge near Willoughby Creek. One of the RECOVER seagrass monitoring sites is located in this bed (WILL_CR).

Salinity

The salinity targets are set at the U.S. 1 Roosevelt Bridge; however, the A1A Bridge salinity data is more representative of salinities experienced by WILL_CR seagrass. **Figure 4-52** shows salinity targets and salinity at both locations for the study period. Salinity patterns were similar at the two locations with salinity values higher at the A1A Bridge. From July 1, 2007 through April 30, 2013, salinity targets were only met 33% of the time. Additionally, consecutive days within the salinity envelope rarely exceeded 30 days (the most frequent seagrass monitoring occurred at monthly intervals). Typically, when salinity was within the envelope, it rapidly increased or decreased (**Figure 4-52**). When the salinity envelope was met, salinity at the A1A Bridge was above 20 96% of the time and salinity never fell below 16 (**Figure 4-53**).

Salinity data is collected near the RECOVER seagrass sites at the frequency shown in **Table 4-6**. While not as frequent as the daily average data available from the U.S. 1 Roosevelt Bridge and A1A Bridge sites, data from these stations showed similar patterns to the A1A and U.S. 1 Roosevelt bridges sites but tended to have higher and less variable salinity (**Figure 4-54**).

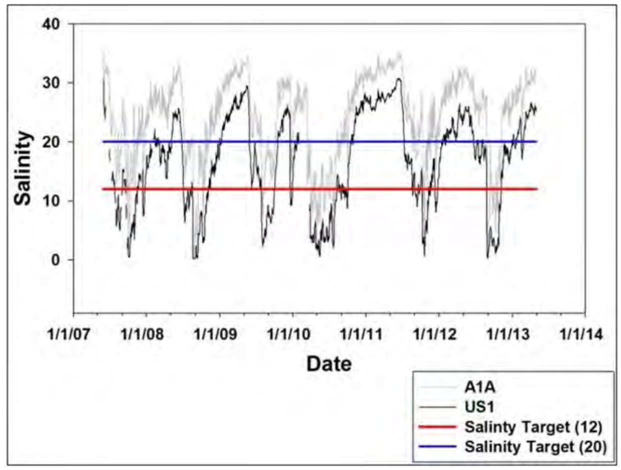


Figure 4-52. Salinity at U.S. 1 Roosevelt and A1A bridges compared to SLE salinity targets for July 1, 2008–April 30, 2013.

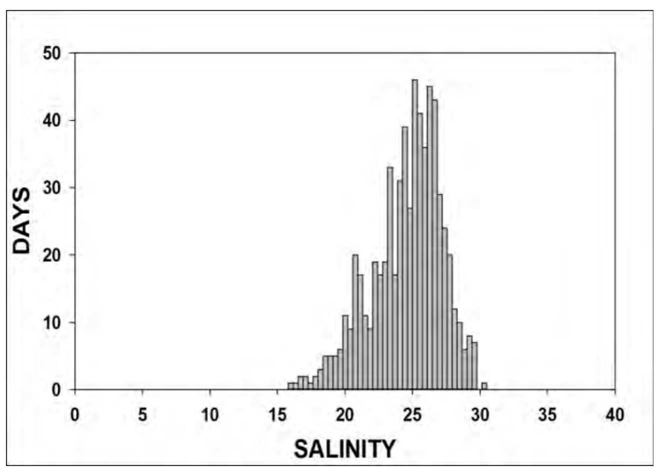


Figure 4-53. Salinity frequency distribution at the A1A Bridge when salinity targets were met at the US1 Roosevelt Bridge for July 1, 2008–April 30, 2013.

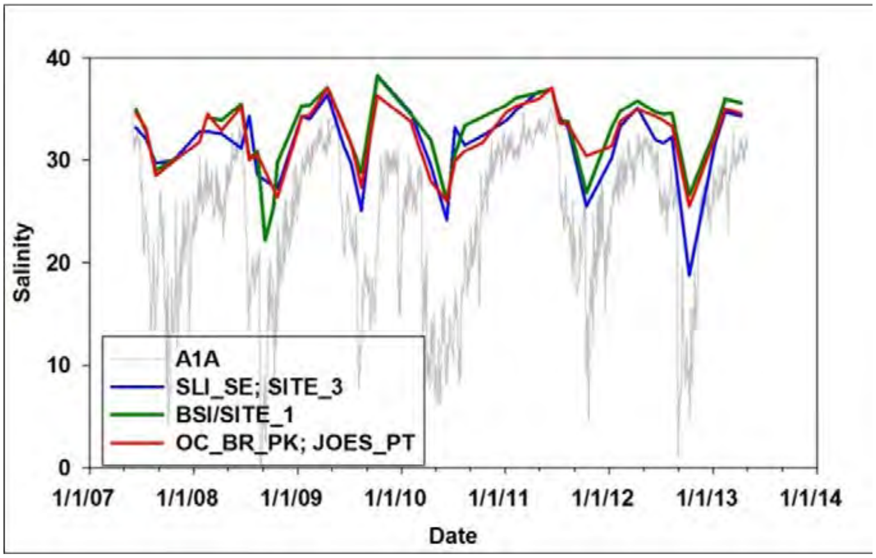


Figure 4-54. Salinity at the A1A Bridge site and RECOVER seagrass monitoring sites.

Seagrass Response

The Willoughby Creek (WILL_CR) site is the closest seagrass site to the influence of SLE freshwater discharges. Two seagrass species occur at this site: *Halodule wrightii* (shoal grass) and *Halophila johnsonii* (Johnson's seagrass). Percent occurrence of both species appeared to respond to changes in salinity (**Figure 4-55**).

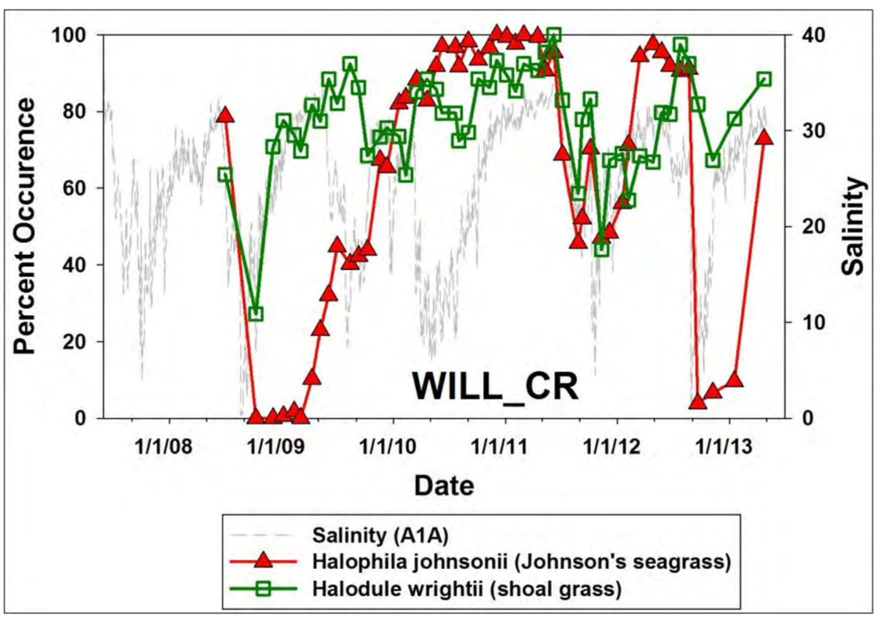


Figure 4-55. Percent occurrence of seagrass at Willoughby Creek and salinity at A1A Bridge.

"In Envelope" Seagrass Response. The salinity envelope was rarely achieved during this study. During the Willoughby Creek seagrass monitoring period (July 1, 2008 – April 30, 2013), there were only four "in envelope" events lasting longer than 30 consecutive days (**Figure 4-56**); no event lasted longer than sixty days. Consequently, there is very little seagrass data to evaluate for "in envelope" dates. However, during these four relatively short events, seagrass percent occurrence either increased or did not change (**Table 4-7**).

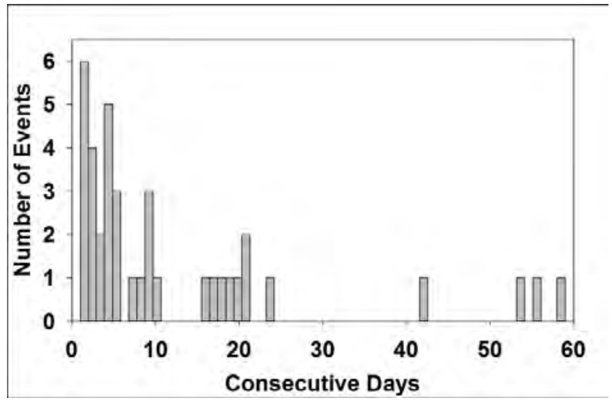


Figure 4-56. Frequency distribution showing consecutive days the salinity envelope was achieved between July 1, 2008 and April 30, 2013.

Table 4-7. "In envelope" salinity events lasting longer than 30 days and Willoughby Creek seagrass percent occurrence data collected closest to the start and end of the "in envelope" event.

Date "Salinity Envelope" Targets Were Met	Consecutive Days "in Envelope"	Percent Occurrence			
		Halodule wrightii		*Halophila johnsonii*	
		Start	End	Start	End
November 6, 2008	56	27	71	0	0
May 30, 2009	59	88	92	32	40
July 11, 2011	42	82	72	68	46
November 7, 2012	54	37	78	7	9

Salinity and Seagrass Percent Occurrence Changes. Dates when seagrass percent occurrence declined or increased at least 20% at Willoughby Creek were identified (**Tables 4-8** and **4-9**). Then salinity data from the U.S. 1 Roosevelt Bridge site for those dates was categorized into one of three salinity groups: below target (< 12), within target (≥ 12 and ≤ 20), or above target (> 20).

Table 4-8. Seagrass percent occurrence declines at Willoughby Creek.

Seagrass Monitoring Date at Start of Decline	Consecutive Days between Seagrass Monitoring	Percent Occurrence			
		Halodule wrightii		*Halophila johnsonii*	
		Start	End	Start	End
July 1, 2008	100	63	27	78	0
August 12, 2009	57	91	68	40	44
June 8, 2011	79	100	59	95	46
October 5, 2011	36	83	44	70	47
July 24,2012	106	97	67	90	7

Table 4-9. Seagrass percent occurrence increases at Willoughby Creek.

Seagrass Monitoring Date at Start of Decline	Consecutive Days between Seagrass Monitoring	Percent Occurrence			
		Halodule wrightii		*Halophila johnsonii*	
		Start	End	Start	End
October 9, 2008	307	27	92	0	40
August 2, 2010	310	72	100	92	95
August 26, 2011	40	59	83	46	70
November 10, 2011	257	44	97	47	91
November 7, 2012	167	67	88	7	73

Seagrass percent occurrence declines were most often associated with U.S. 1 Roosevelt Bridge salinities below 12 (**Figure 4-57**). For example, between July and October of 2008, salinity dropped from 25 to 9 over 100 days at the A1A Bridge; salinity was below 12 at the U.S. 1 Roosevelt Bridge virtually the entire time. Percent occurrence dropped to zero for *H. johnsonii* and to 27% for *H. wrightii* during that event.

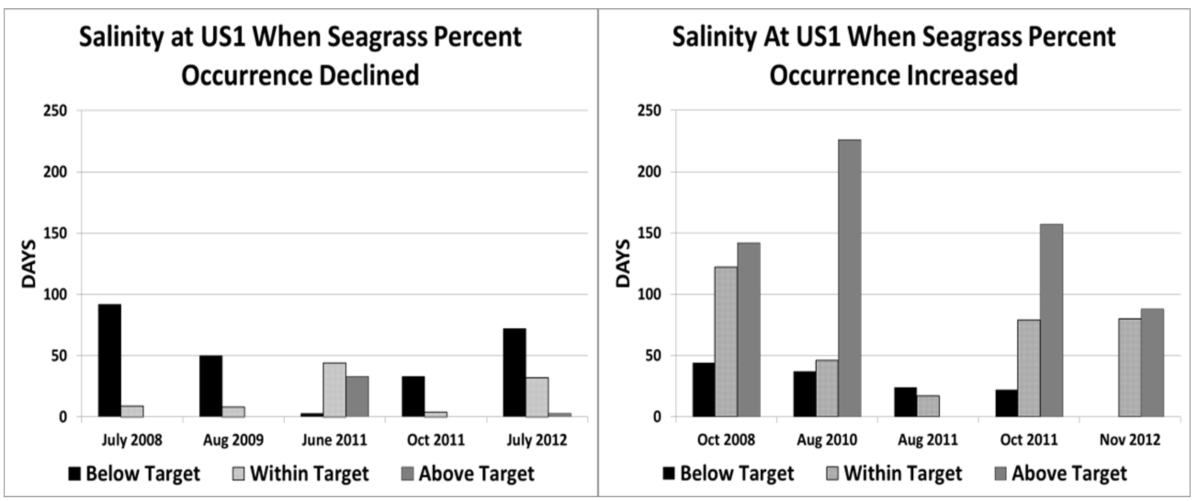


Figure 4-57. Salinity at the U.S. 1 Roosevelt Bridge site associated with changes in percent occurrence in seagrass at the Willoughby Creek site. Below target = salinity < 12; within target = salinity ≥ 12 and ≤ 20; above target = salinity > 20.

Similar results occurred for other events, except June 2011. Percent occurrence declined during the growing season when salinity was in or above the salinity envelope. Prior to this decline in percent occurrence, salinity was above 20 at the US1 Bridge for approximately 8.5 months, with salinity at the A1A Bridge reaching 35. Following this prolonged period of high salinity, salinity fell from 35 to 18 at the A1A Bridge, and seagrass percent occurrence declined.

Percent occurrence increases were typically associated with salinity above 12 at U.S. 1 Roosevelt Bridge. However, during a short-term event in August 2011, percent occurrence increased even though salinity was low.

At other sites, seagrass percent occurrence tended to be less variable than at Willoughby Creek but percent occurrence declines were often observed at times similar to those documented for Willoughby Creek (**Figure 4-58**). These percent occurrence declines occurred following drops in salinity. However, as shown on **Figure 4-58**, salinity remained high at all sites even at the lowest point in the salinity drop.

At Boy Scout Island and Site 1, *Syringodium filiforme* (manatee grass) percent occurrence steadily increased through 2010, with only minor declines following low salinity events. This steady increase was part of recovery from 2004/2005 hurricane impacts. After the hurricanes, *S. filiforme* was largely absent from both sites. As *S. filiforme* recovery continued, *H. wrightii* and *H. johnsonii* were out-competed at both Boy Scout Island and Site 1 (**Figure 4-58**). Manatee grass percent occurrence continued to increase, reaching 100% coverage again in 2010 at Boy Scout Island. *S. filiforme* was also largely eliminated at Site 3 following the hurricanes. Much of the *S. filiforme* at this site was buried with muck during the hurricanes and has not recovered.

In general, canopy heights tended to follow seasonal patterns (**Figure 4-59**). However, at Boy Scout Island and Site 1, initial increased height over time resulted from a shift in canopy species (*H. wrightii* to *S. filiforme*). Response to salinity changes was less pronounced than with percent occurrence data.

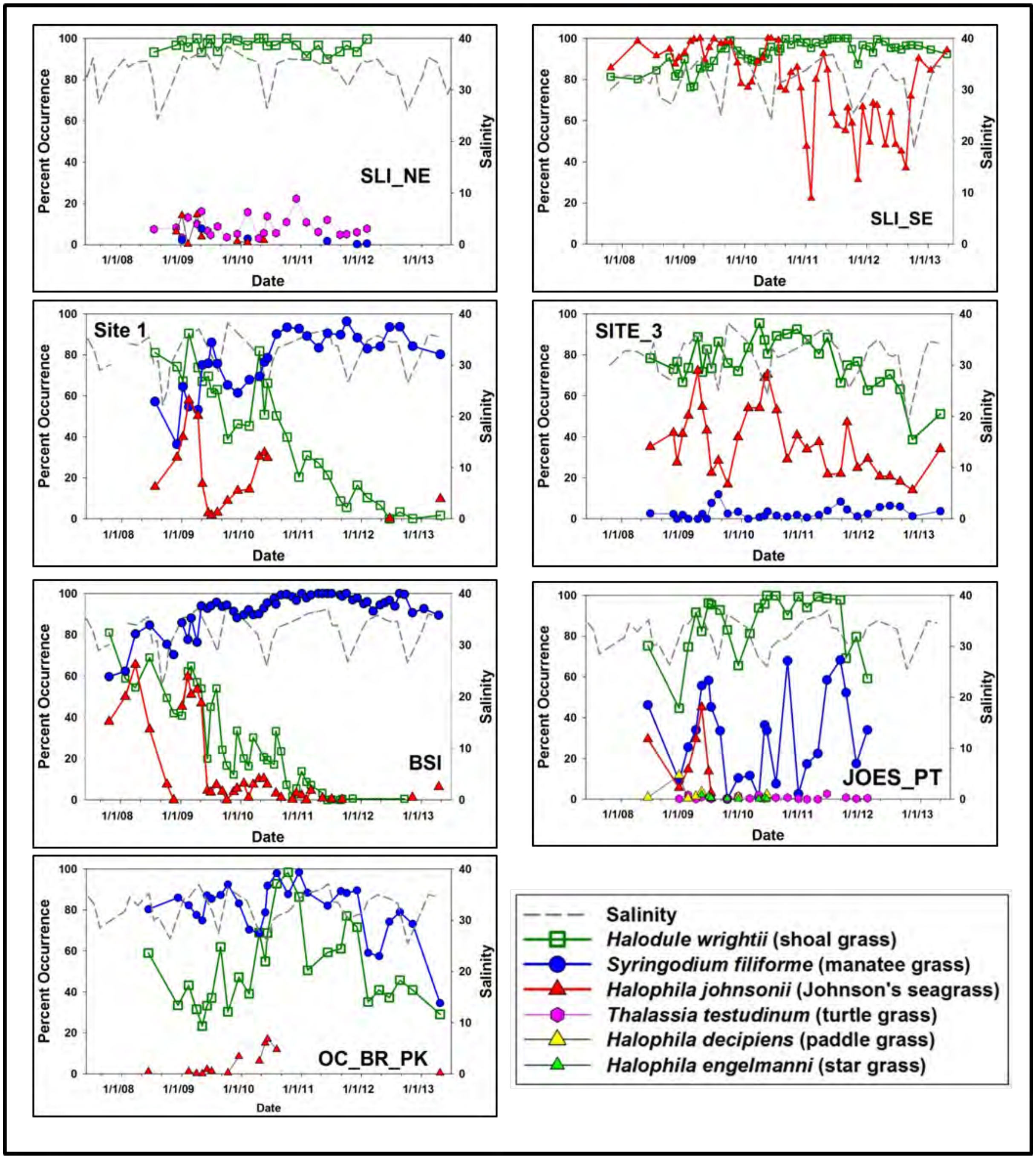


Figure 4-58. Seagrass percent occurrence and salinity at seagrass sites near the influence of the SLE.

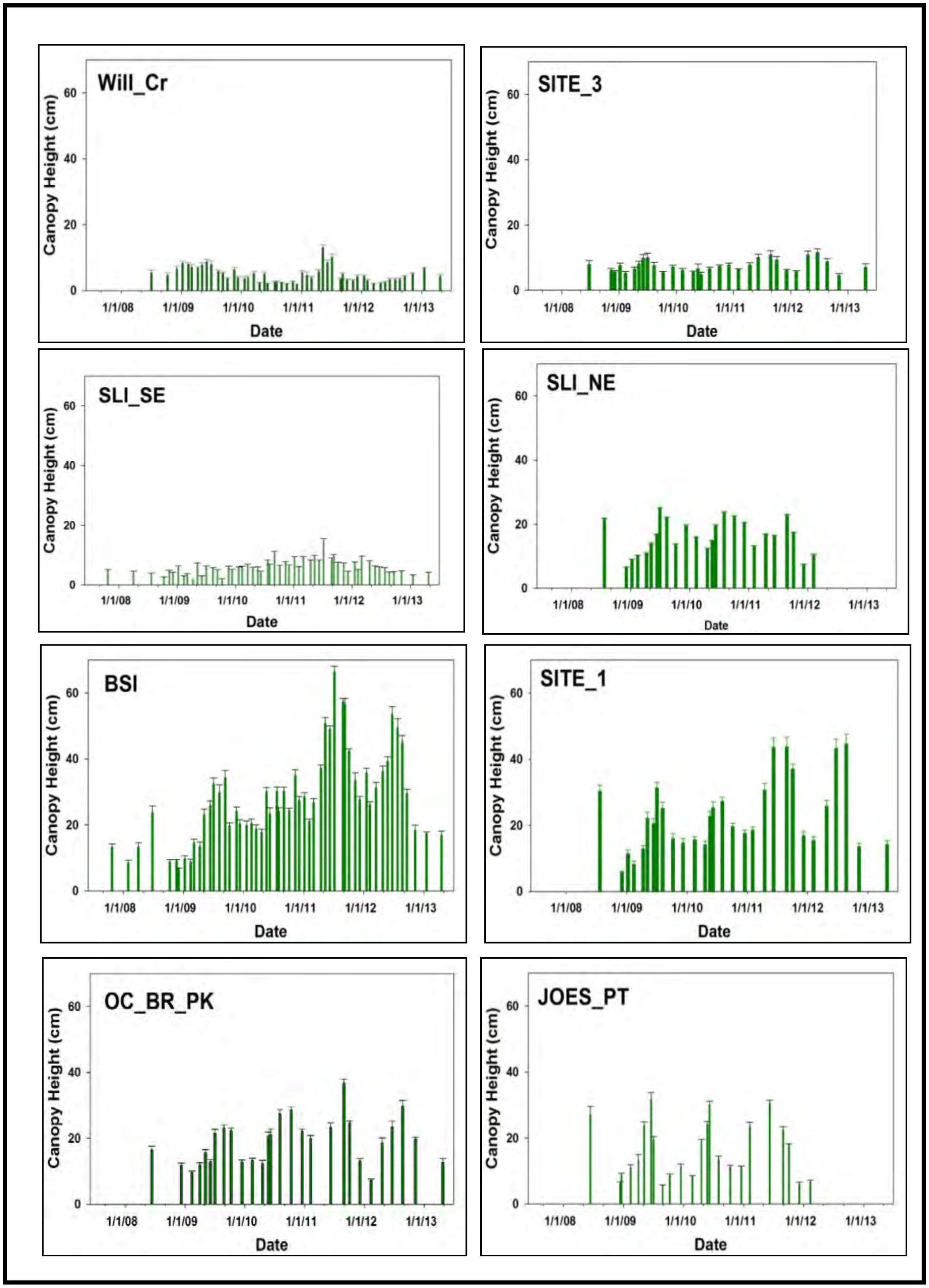


Figure 4-59. Seagrass canopy height at monitoring sites near the SLE.

C-25/Taylor Creek Sites. While not part of the SLE salinity envelope evaluation, the Fort Pierce site data is provided for informational purposes. The eastern site is a *S. filiforme* dominated site; percent occurrence of *S. filiforme* remained high throughout monitoring (**Figure 4-60**). The western site, adjacent to the C-25 discharge, is dominated by *H. wrightii* (**Figure 4-60**). Percent occurrence of the seagrass appeared to respond to salinity changes. As with other sites, canopy height typically reflected seasonal growth (**Figure 4-61**).

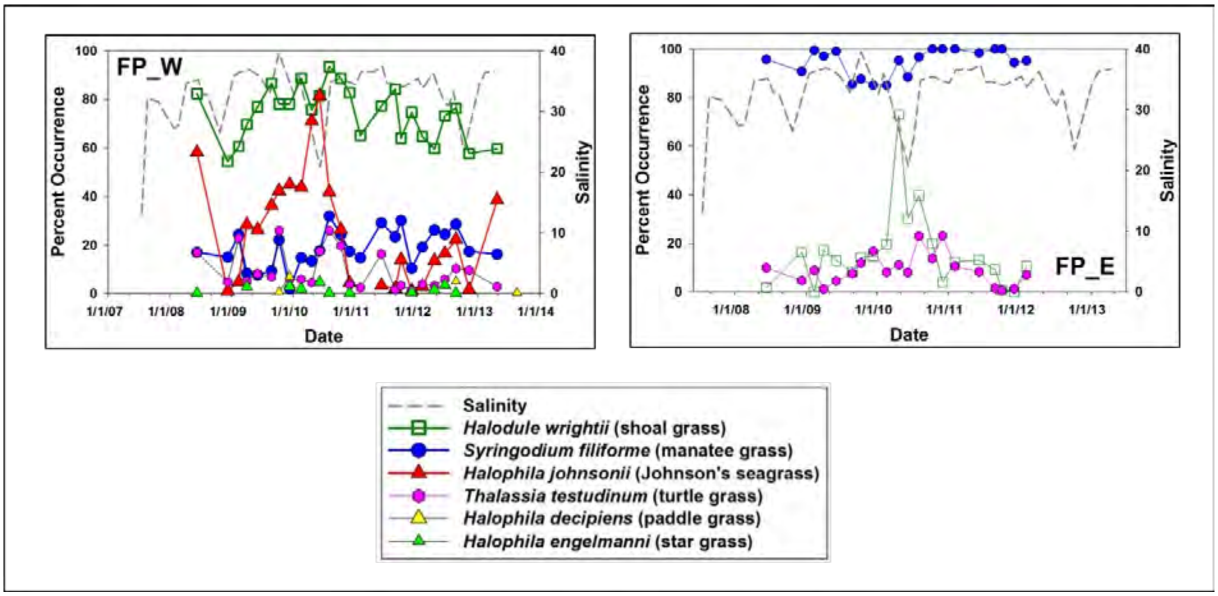


Figure 4-60. Seagrass percent occurrence at monitoring sites near the C-25 discharge.

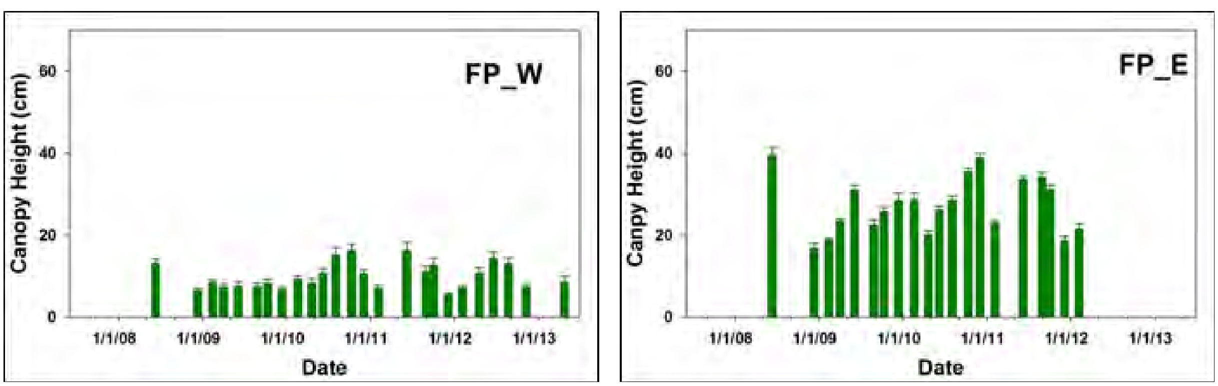


Figure 4-61. Canopy height at monitoring sites near the C-25 discharge.

Discussion

The questions posed under the Evaluation section above are answered is this section:

- **Does seagrass in the SLE/SIRL respond as predicted when CERP targets are met?**

  Unfortunately CERP targets were rarely met during this study. However, salinity below the lower target (12) resulted in seagrass percent occurrence declines. Recovery typically occurred when salinities were higher than 12 at the U.S. 1 Roosevelt Bridge site.

- **Has our understanding of how seagrass responds to salinity changed?**

  The percent occurrence data patterns presented above suggest that even when salinity is relatively high (> 20) and within generally accepted seagrass salinity ranges, rapid/steep drops in salinity may lead to declines in seagrass percent occurrence (in this study, mean daily salinity at the A1A Bridge dropped from 35 to 18 over 60 days). Even the sites farthest from the SLE, Ocean Break Park and Joe's Point, had percent occurrence declines following such drops in salinity. These are the two deepest sites monitored, so light attenuation associated with salinity drops may amplify impacts to the seagrass.

- **Are the SLE salinity targets appropriate for seagrass?**

  The lower salinity target of 12 at the U.S. 1 Roosevelt Bridge site appears to be appropriate for maintaining seagrass beds near the mouth of the SLE and in the adjacent SIRL. During this study, seagrass percent occurrence declined when salinity fell below 12 at the U.S. 1 Roosevelt Bridge site and recovery occurred at higher salinities.

  Analysis of salinity data revealed when the salinity envelope was met, salinity rarely fell below 20 and never below 16 at the A1A Bridge, with a maximum salinity of 31 (**Figure 4-57**). These salinities fall within the tolerance range for all seagrass species found in the SIRL (Irlandi 2006), suggesting that the salinity envelope is appropriate for existing seagrass beds.

  Steep drops in salinity during this study appeared to result in percent occurrence declines, even when salinity was within accepted seagrass tolerance ranges. Having an upper bound of 20 at the U.S. 1 Roosevelt Bridge site will result in less salinity variability than currently occurs and is expected to provide appropriate conditions for seagrass growth.

- **How long does it take a seagrass species to recover from an adverse impact caused by an extreme salinity event?**
  - ***H. johnsonii***: At Willoughby Creek, recovered from near 0% occurrence to greater than 80% over periods of 8 to 17 months.
  - ***H. wrightii:*** At Willoughby Creek, never declined to zero, but sharp declines did occur in 2008 and again in 2011. Recovery (percent occurrence near 100) took from 8 to 10 months.

- **S. filiforme**: Most was eliminated from the Boy Scout Island site in 2006. Recovery (percent occurrence near 100) took approximately four years.

St. Lucie Estuary and Southern Indian River Lagoon Benthic Macroinvertebrates

Introduction

The SIRL is one of the most biologically diverse estuaries in the continental United States (Fernald 1982, FDER 1989, Swain et al. 1994, www.sms.si.edu/irlspec/). This estuary is a particularly valuable coastal resource to southeast Florida, and is also a crucial nursery ground for economically and ecologically important fish species that occur along the entire eastern seaboard. However, rapid and expansive population growth along the SIRL and the SLE, particularly since 1970, and Lake Okeechobee regulatory releases have resulted in watershed alteration, harmful algal blooms, declining water quality, and increased sediment loading. Consequently, this fragile ecosystem is currently under threat. Critical to protecting and conserving this invaluable natural resource is heightened awareness of SIRL biodiversity through enhanced research and environmental and conservation education.

Benthic infaunal communities are excellent indicators of water quality and are used in many environmental monitoring programs as targets for investigations of environmental disturbance (Borja and Dauer 2008, Borja and Tunberg 2011). The organisms that comprise these communities are primarily invertebrates that are sessile or of limited mobility. Most are secondary consumers, and they provide linkages between primary producers and higher level consumers. Many organisms are filter feeders, detritivores or omnivores, and a few are carnivores. Some representatives are commercially harvested (e.g., crabs, shrimps). They are constant features of the bottom sediments, and vary predictably in association with the physical habitat and in response to man-made and natural changes (Santos and Simon 1980). Furthermore, unlike fluctuating populations of planktonic organisms or many pelagic fish species, adults of most benthic invertebrate species are sessile, and thus they are extremely good indicators of locally-induced environmental changes (Borja and Tunberg 2011).

Benthic infaunal animals have been monitored quarterly at 9 sites (**Figure 4-62**) in the SLE and SIRL since February 2005. This report includes analyses of 34 sampling dates between February 2005 and April 2013 for these 9 sites. Samples from an additional 6 sites have been collected but not sorted and identified after July 2011. Measurements of environmental variables, including sediment type, organic content, surface water turbidity, Secchi depth, surface and bottom water temperature, salinity, oxygen, and pH have also been taken quarterly at the same sites. In this report, analyses of species richness and diversity in response to these environmental factors are presented and discussed. A particular emphasis is placed on effects of salinity and sediment types on species diversity and other features of these benthic assemblages.

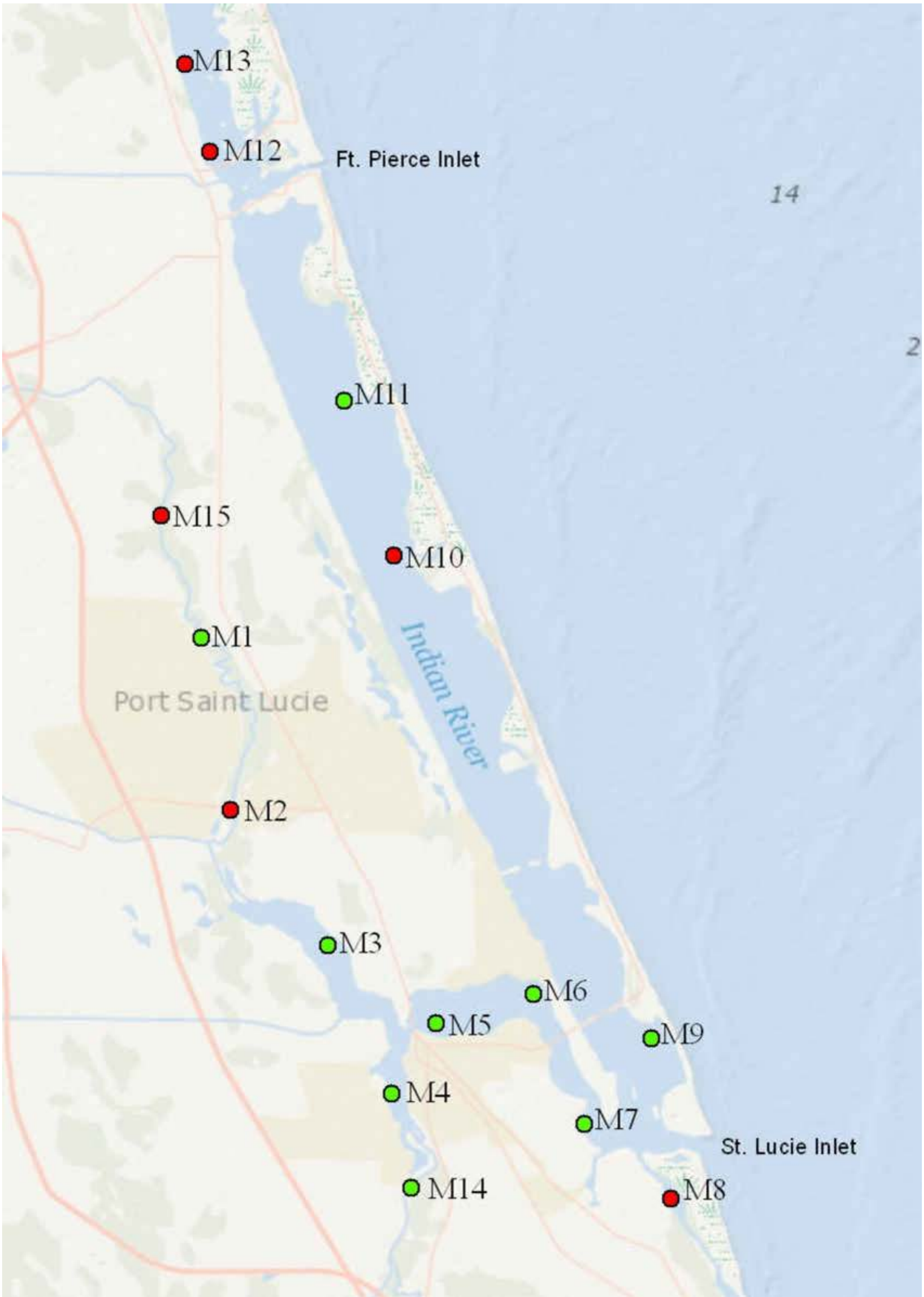


Figure 4-62. Sites for quarterly benthic monitoring program in the SLE and IRL. Sites in green are currently sampled and analyzed each quarter; these sites are included in the data analyses in this report. Sites in red have been sampled but not processed after July 2011.

Methods and Data Analyses

Field Samples

Quantitative benthic infaunal sampling was performed at 15 fixed sites (**Figure 4-62**). All sites are located within Martin and St. Lucie counties. Sites M14 and M15 were added in July 2007. Only sites M1, M3, M4, M5, M6, M7, M9, M11 and M14 are currently included for sample processing. All other sites are being collected quarterly and stored for future processing when time and funding become available.

Three quantitative infaunal samples were collected quarterly at each sampling site. Two bottom substrate samples were collected quarterly at each site for sediment analyses. The sediment from these cores is further divided into subsamples from two substrate depths (0–2 cm and 2–5 cm). Surface and bottom water temperature, dissolved oxygen, salinity, conductivity, pH, and turbidity were measured at each sampling site on each sampling occasion. The time of day and weather conditions were also recorded at each sampling site on each sampling occasion. Surface and bottom conductivity was also measured starting in April 2008.

Laboratory Procedures

Faunal samples were identified to the lowest possible taxon. The number of each separate taxon was counted during this process. Water content and loss on ignition (organic content) in the sediment were calculated.

Data Management and Analysis

The arithmetic means and the corresponding variances and standard error values are calculated for abundance of the separate taxa (and totally) from each site. The diversity of the fauna is calculated by using both the total species richness and the Shannon-Wiener Diversity index. Species richness is expressed as the total number of taxa found in all samples combined at each station (3 replicates). The Shannon-Weiner index was developed from information theory and is based on measuring uncertainty. The degree of uncertainty of predicting the species of a random sample is related to the diversity of a community. If a community is dominated by one species (low diversity), the uncertainty of prediction is low because a randomly-sampled species is most likely going to be the dominant species. However, if diversity is high, uncertainty is high.

An additional benthic macroinvertebrate index that has been previously applied to the SLE and SIRL for an evaluation of the environmental "health" is the M-AMBI index (Borja and Tunberg 2011). AZTI's Marine Biotic Index (AMBI) software was developed by AZTI-Tecnalia for assessment of the quality of benthic macroinvertebrate assemblages (Borja et al. 2000). AMBI calculates an index based on the proportion of species assigned to one of five levels of sensitivity to increasing levels of disturbance, from very sensitive to opportunistic species. M-AMBI includes the addition of multivariate species richness and Shannon diversity components to AMBI (Borja et al. 2007, 2008, 2009).

Results and Discussion

<u>*Salinity and Flow*</u>

Periodic pulses of freshwater flow from Lake Okeechobee occurred during the project sampling period. Additionally, total flow into the SLE, measured as flow rates at structures S-80 + S-48 + S-49, includes flow from the C-44, C-23, and C-24 canals into the watershed. These pulses of fresh water can be seen during years with high rainfall, and periods of exceptionally high total flow were observed in 2005–2006 and periodically over the years (**Figure 4-63**). Pulses of freshwater flow were also observed in 2008, 2009–2010 and 2012, usually during the late summer time period. Flow from Lake Okeechobee, flow from other sources such as the C-23 and C24 canals, and drainage from the rest of the watershed contribute to these high total flows. The rate of freshwater flow is directly related to salinity in the SLE (**Figure 4-64**). Both flow from Lake Okeechobee as well as non-lake flow show strong linear relationships to salinity measured at the U.S. 1 Roosevelt Bridge in Stuart (**Figures 4-65** through **4-67**). For this report various characteristics of the benthic communities are related to salinity, and this direct relationship between freshwater flow into the SLE and salinity should be recognized as the driver for salinity fluctuations.

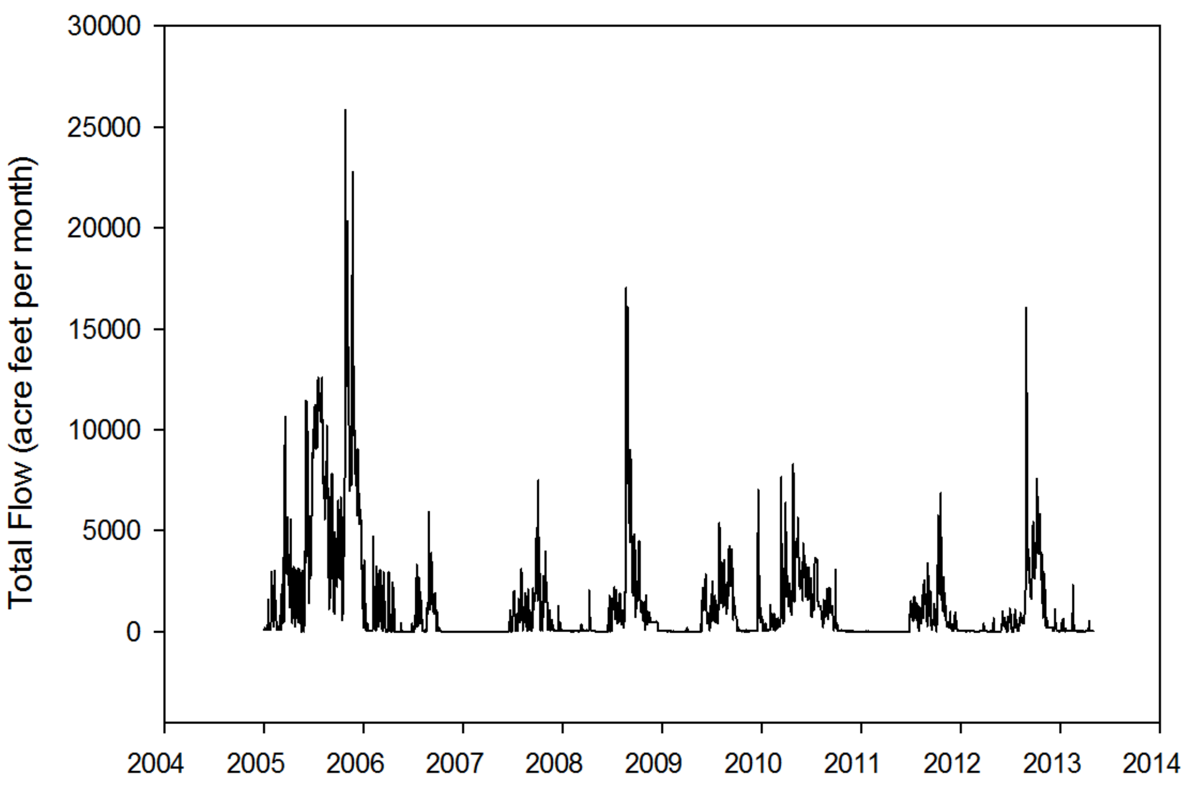


Figure 4-63. Total freshwater flow rates measured at structures S-80 + S48 + S49. Periods of heavy flow during wet years are interspersed with low or no flow during drier seasons and years.

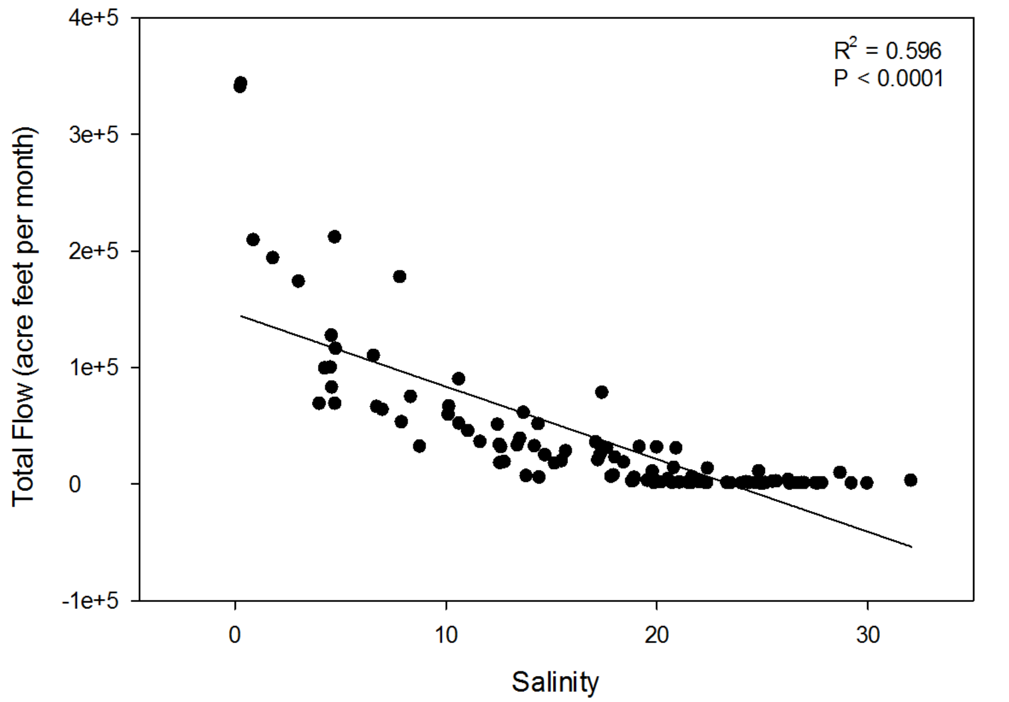


Figure 4-64. Total flow to the SLE (Figure 4-63) includes Lake Okeechobee flow and non-lake flow into the SLE. The graph illustrates the strong negative relationship between flow rates and salinity levels measured at the U.S. Roosevelt Bridge in Stuart.

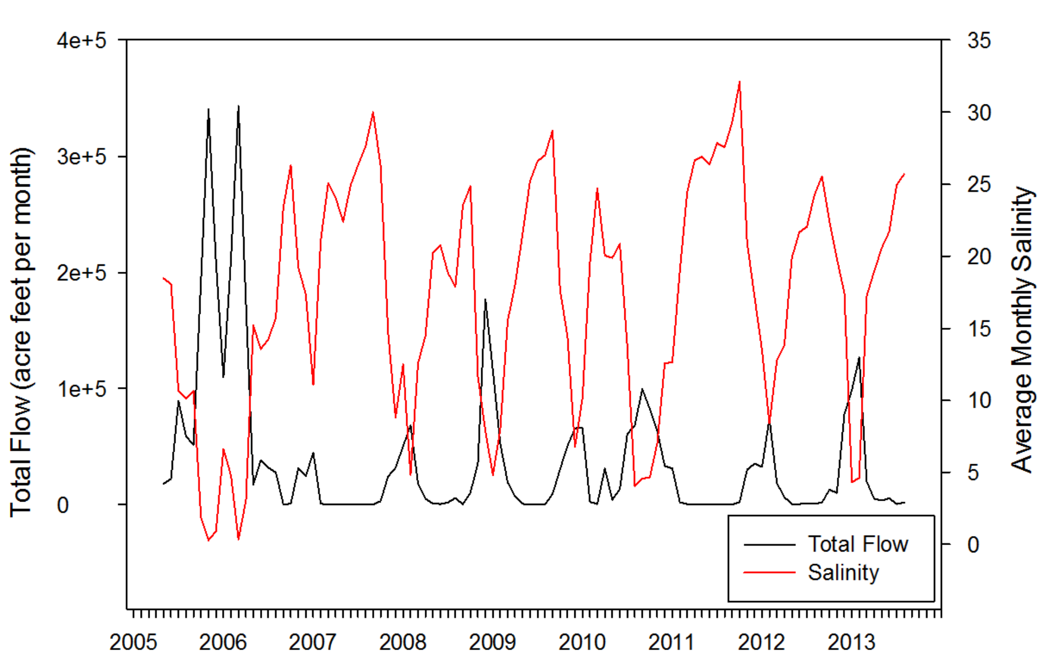


Figure 4-65. Total flow to the SLE includes Lake Okeechobee flow and non-lake flow into the SLE. The graph illustrates the strong relationship between flow rates and salinity levels measured at the US1 Roosevelt Bridge in Stuart. Large drops in salinity are only observed during periods of elevated total flow.

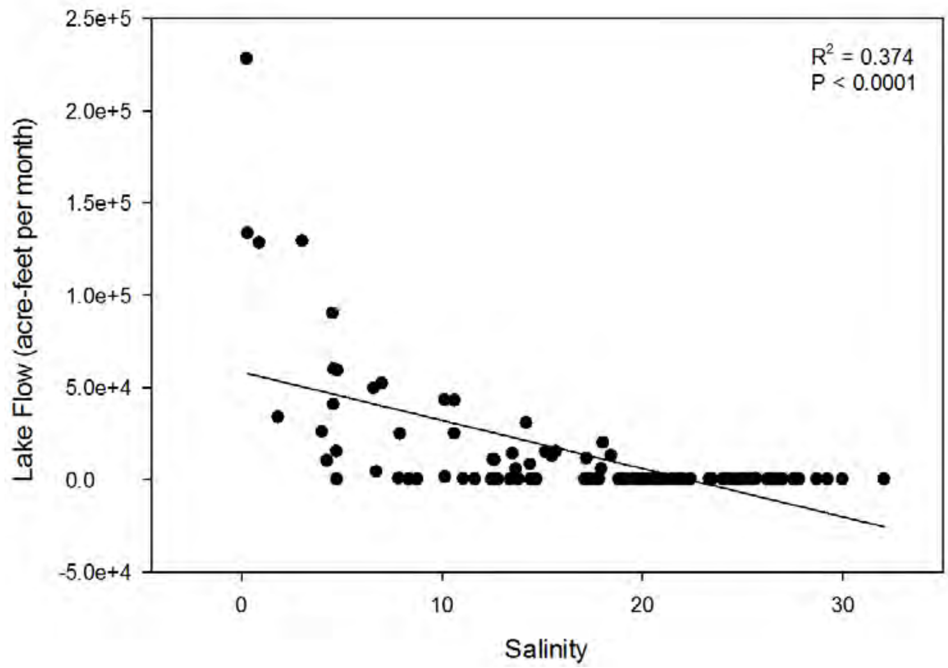


Figure 4-66. Relationship between flow from Lake Okeechobee and salinity levels measured at the US1 Roosevelt Bridge in Stuart. There is a significant relationship with salinity dropping during periods of high flow from Lake Okeechobee releases.

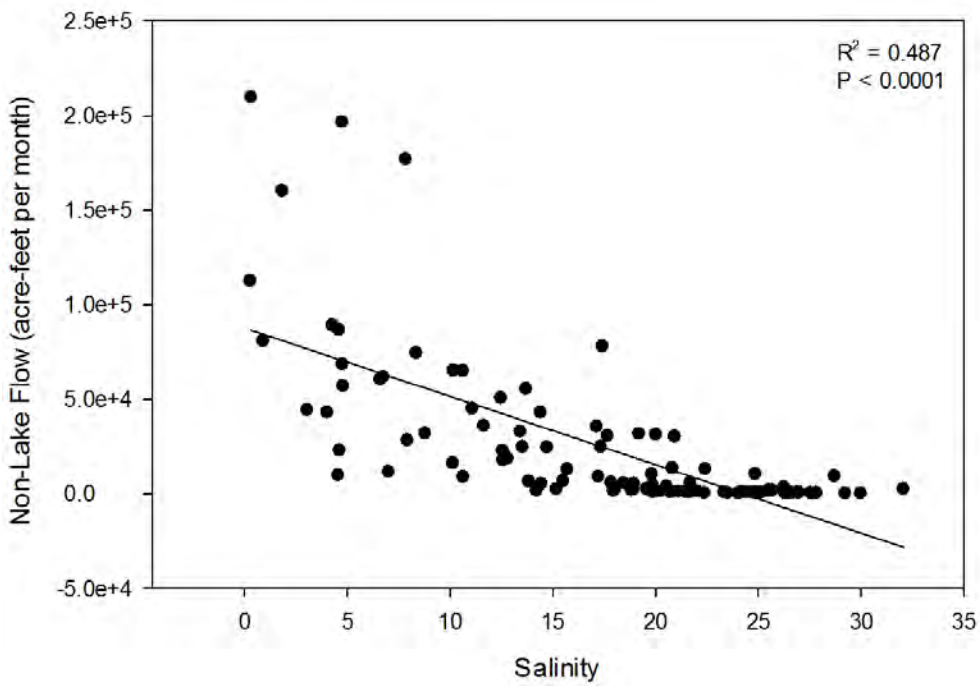


Figure 4-67. Relationship between flow from non-lake sources and salinity levels measured at the US1 Roosevelt Bridge in Stuart. There is a significant relationship with salinity dropping during periods of high non-lake flow into the SLE.

<u>M-AMBI Index</u>

Overall M-AMBI values at each site show expected patterns based on environmental conditions at each location (**Figure 4-68**). The sites with highest M-AMBI are located at the mouth of the SLE (M7) or in the SIRL (M9 and M11); these sites have consistently higher salinity, less impact from freshwater flows, and higher species diversity (**Figure 4-69**). Environmental conditions are more stable at these sites. M-AMBI values at the other sites fall into the bad or poor categories during most years. These results are consistent with those reported by Borja and Tunberg (2011) who were the first to apply M-AMBI data to the SLE and SIRL system. The sites with the lowest M-AMBI values were associated with the most degraded sites (M1, M3, M4, M5, and sometimes M6). This index incorporates information about the community structure, including species richness, diversity, and proportion of opportunistic species that are pollution tolerant, and therefore can provide an overall assessment of the infaunal community composition. It has also been suggested that the index is more sensitive to human pressures than to natural variability (Borja and Tunberg 2011), which makes it useful as a management tool.

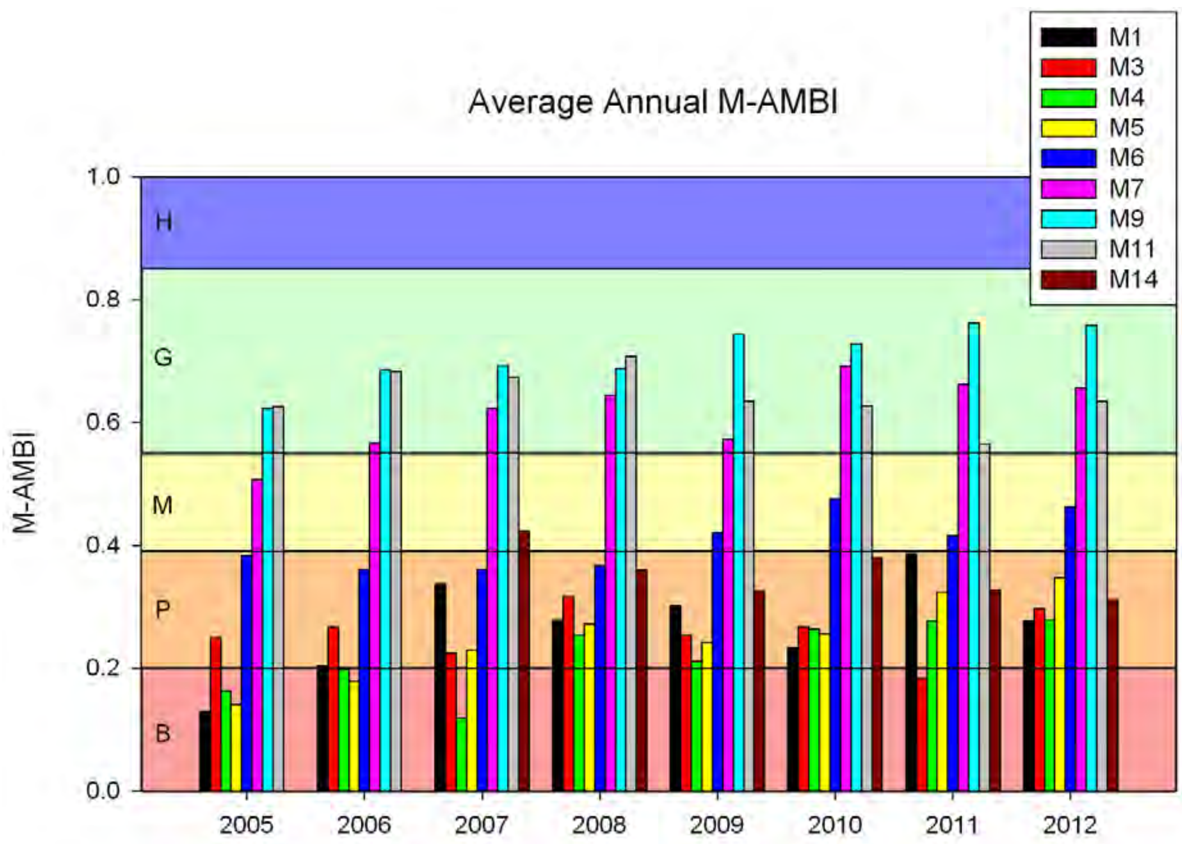


Figure 4-68. Average annual M-AMBI for each monitoring site in the SLE and SIRL. The relative categories for the M- AMBI of B (bad), P (poor), M (medium), G (good), and H (high) are color coded as a visual guide to these categories and to aid in determining differences among the sites.

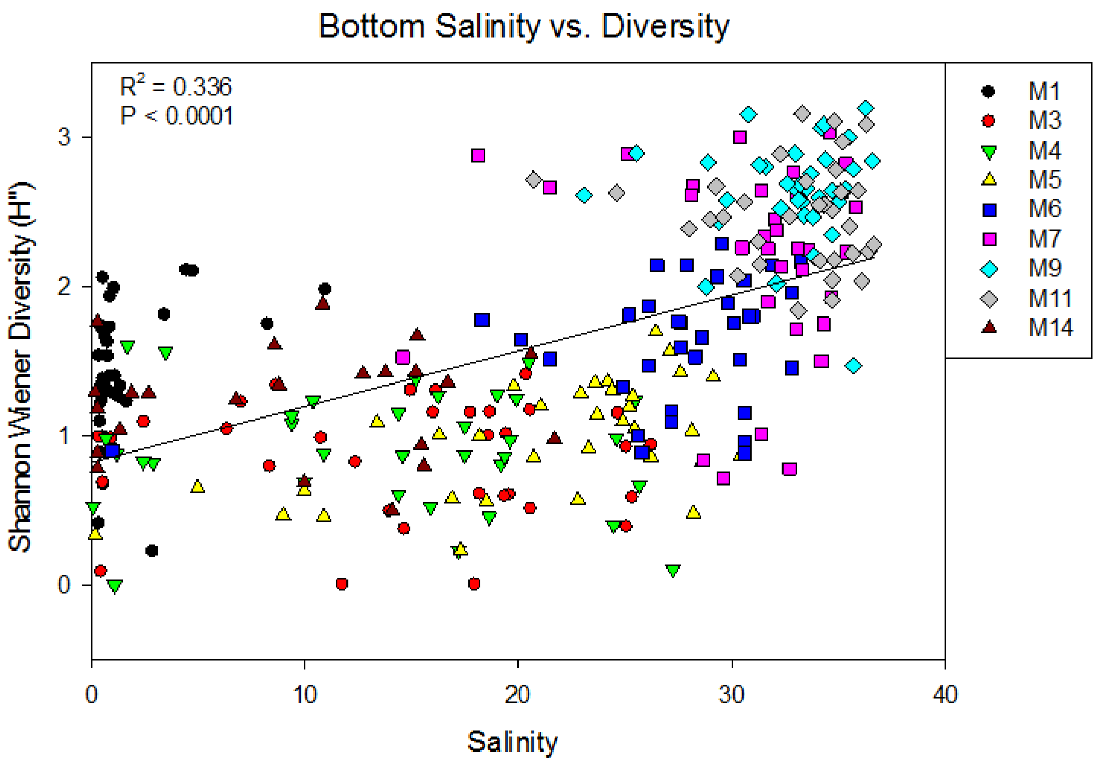


Figure 4-69. Relationship between salinity at the bottom and species diversity is illustrated for all nine sites. Each point represents one quarterly sample and these are color coded for each sampling location. Linear regression shows significant positive relationship between salinity and diversity.

Species Diversity and Salinity Relationships

Species diversity based on the Shannon Wiener Diversity index is strongly positively correlated with salinity (**Figure 4-69**). All of the highest diversity sites are located at the mouth of the SLE (M7) or in the SIRL (M9 and M11); these sites have consistently higher salinity and less impact from freshwater flows. Only a few sites in the SLE (M3, M5, M4, and M14) have salinity values that frequently fall within the targeted range of 12–20. Site M5, near the U.S. Roosevelt Bridge, is subjected to variable freshwater inputs and has variable salinity measurements, and shows this same positive relationship between diversity and salinity (**Figures 4-70** and **4-71**). Site M6, slightly downstream of M5 with higher salinity measurements overall, also shows the same positive relationship (**Figure 4-70**). This close relationship between salinity and species composition can be observed when number of taxa (species) is plotted along with salinity over time at site M5 (**Figure 4-71**). The total number of species drops to just a few species during periods of low salinity. One of the species that is tolerant of low salinity is the tube-dwelling polychaete worm *Streblospio benedicti*, which at times can be quite abundant under both high and low salinity conditions (**Figure 4-72**). This species can tolerate variable salinity and other environmental stresses and can effectively exploit newly disturbed areas (Thistle 1981, Grassle and Grassle 1974).

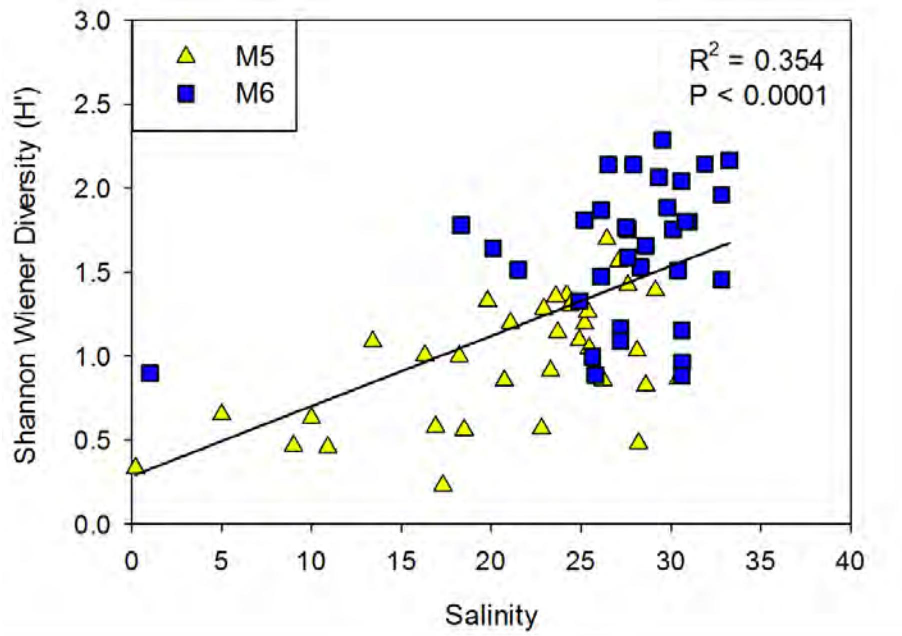


Figure 4-70. Relationship between salinity at the bottom and species diversity at site M5 near the US1 Roosevelt Bridge site and the adjacent site M6. Each point represents one quarterly sample. Linear regression shows significant positive relationship between salinity and diversity.

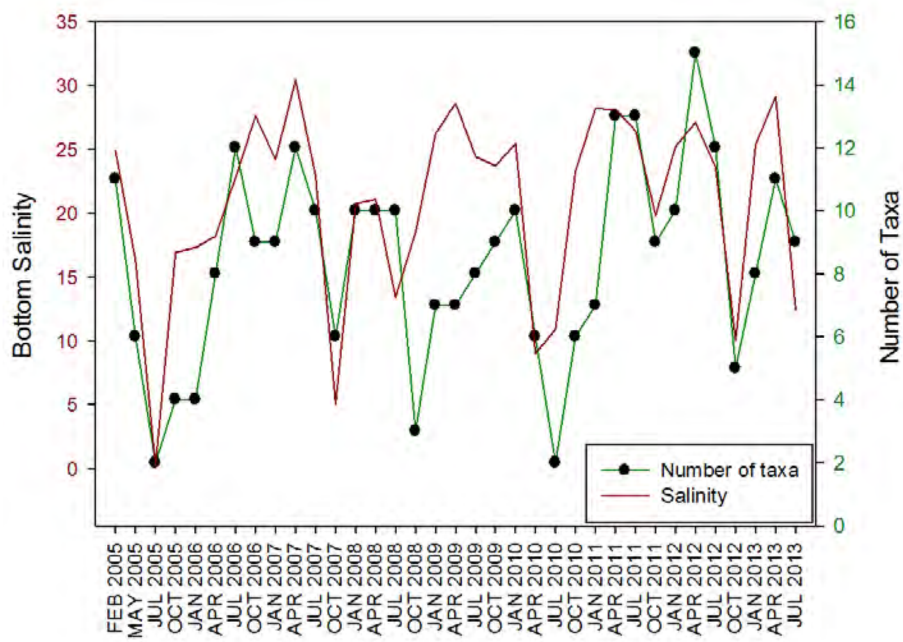


Figure 4-71. Salinity at the bottom is shown on the same graph with number of taxa (species) over time for each quarterly sample. The number of taxa closely tracks bottom salinity.

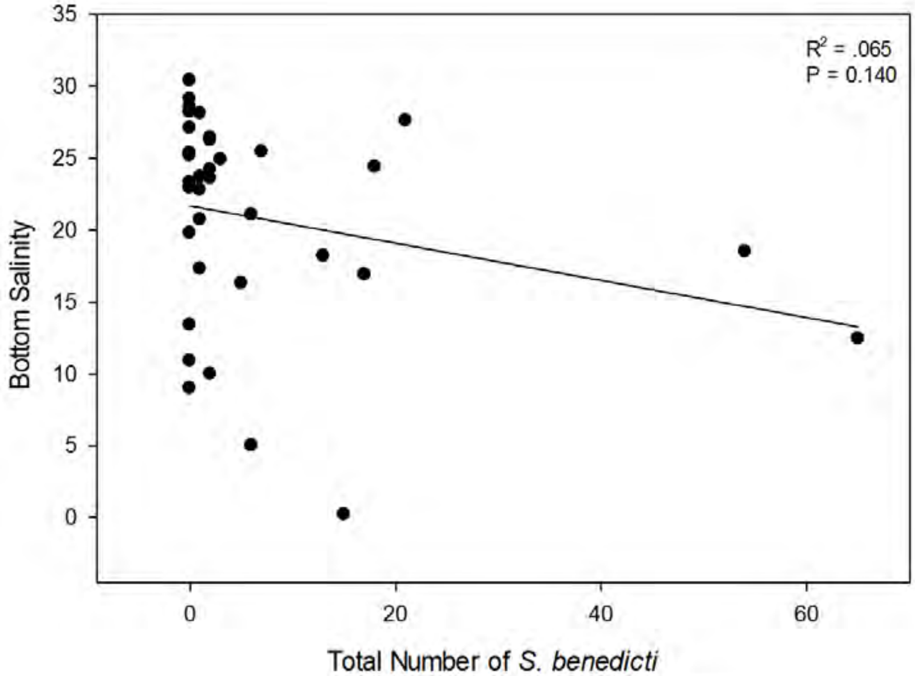


Figure 4-72. Number of individuals of the polychaete worm *Streblospio benedicti* in relationship to bottom salinity at site M5.

A positive relationship between the M-AMBI index and bottom salinity is also observed (**Figure 4-73**), similar to the patterns found between Shannon Weiner diversity and salinity (**Figure 4-69**). The sites with the highest diversity are located at the mouth of the SLE (M7) or in the SIRL (M9 and M11); these sites have consistently higher salinity and less impact from freshwater flows.

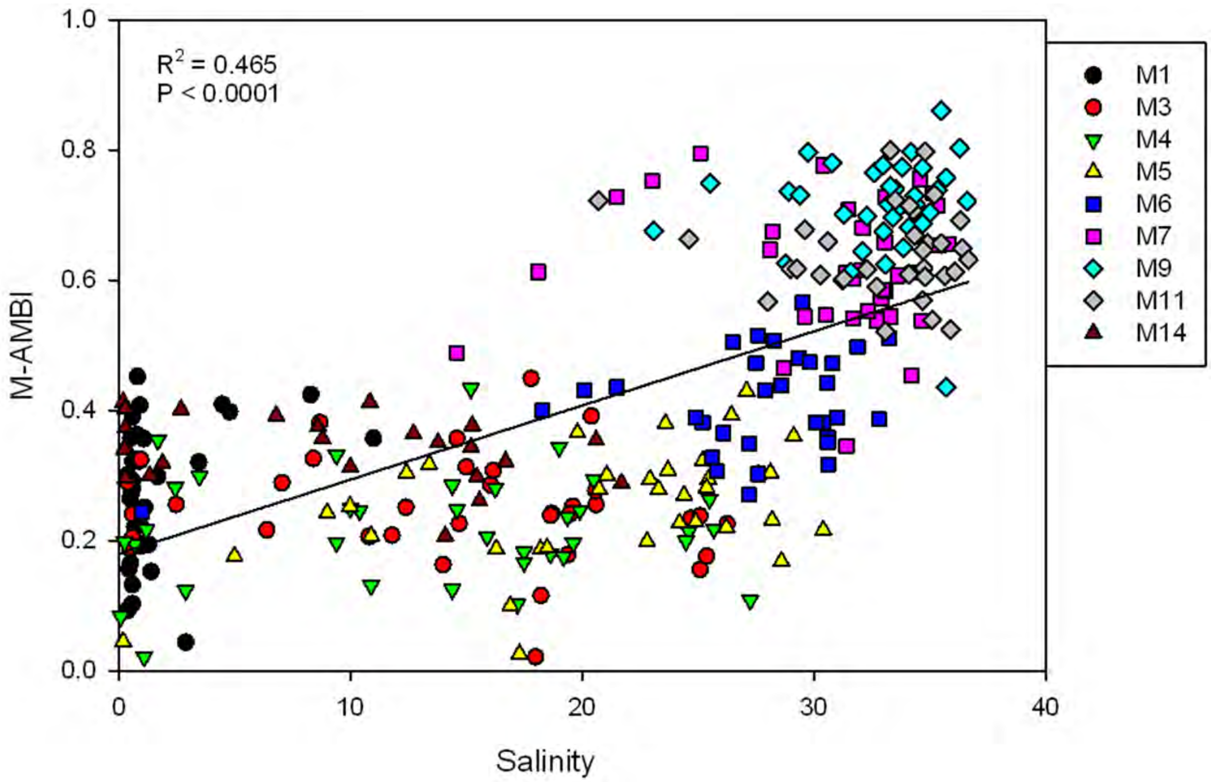


Figure 4-73. Relationship between salinity at the bottom and M-AMBI is illustrated for all nine sites. Each point represents one quarterly sample and these are color coded for each sampling location (Figure 4-62). Linear regression shows significant positive relationship between salinity and M-AMBI.

Interestingly, some of the SLE sites do not show the same clear relationship between species diversity and salinity. For example, M3 and M4, which are located in the North Fork and South Fork of the St. Lucie River, respectively, do not show a correlation between diversity and salinity (**Figure 4-74**). These sites have variable salinity and variable but consistently lower diversity than sites that are further down river. These observations suggest that factors other than salinity might be influencing diversity at these locations that are subject to frequent physical perturbations.

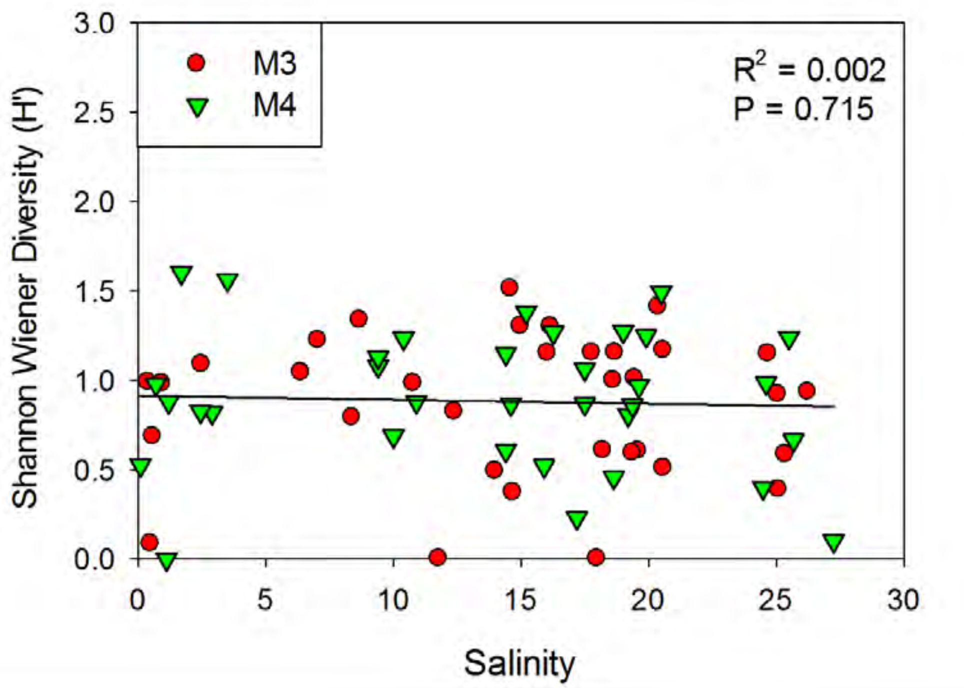


Figure 4-74. Relationship between salinity at the bottom and species diversity is illustrated for sites M3 and M4 in the St. Lucie River. Salinity is highly variable at these sites, but there is no relationship between salinity and diversity at these locations based on analysis by linear regression.

Species Diversity and Dissolved Oxygen Relationships

One abiotic variable that is known to affect species diversity and abundance of macroinfaunal invertebrates is dissolved oxygen content of the water (Santos and Simon 1980). While a significant positive relationship was observed between species diversity and bottom dissolved oxygen concentrations, the R2 value is low, and it is likely that dissolved oxygen concentrations are more important at some sites than others (**Figure 4-75**). Only a few sites, mostly those in the North and South forks of the SLE, show low values for dissolved oxygen concentrations below 4 milligrams per liter (mg/L) at the bottom of the estuary.

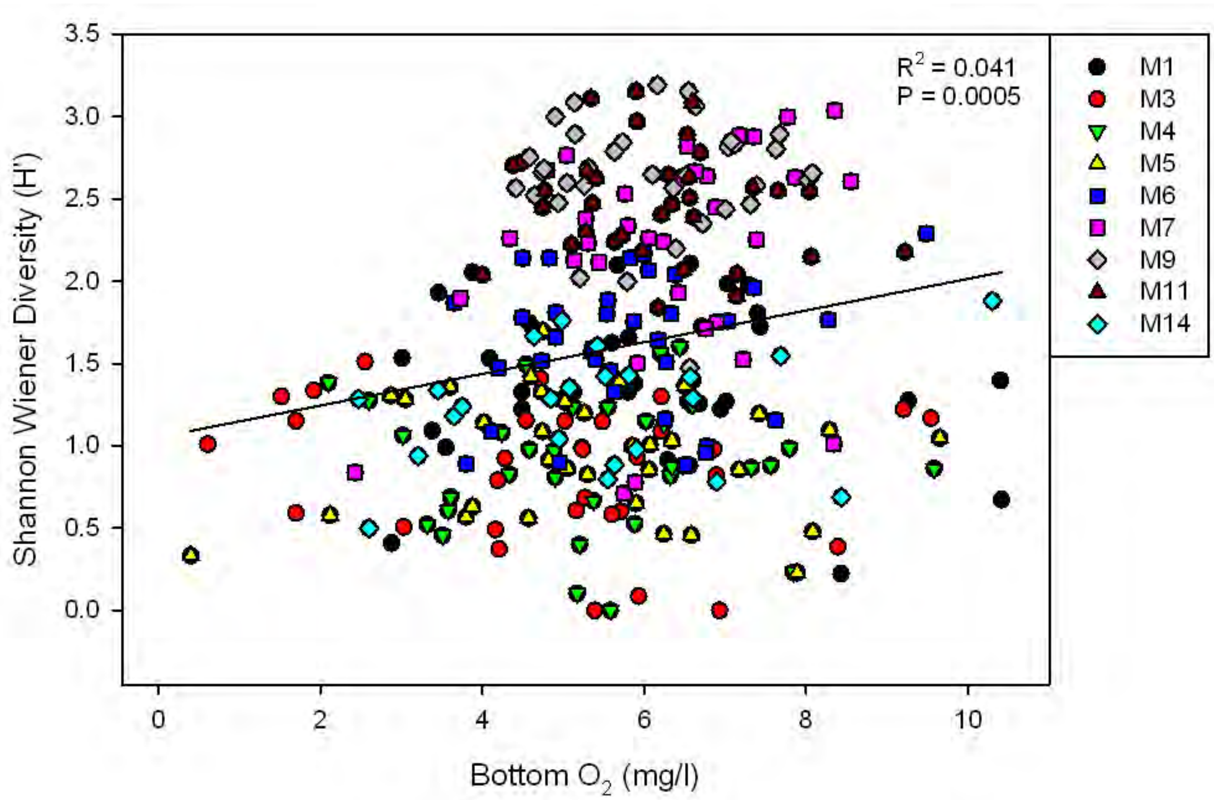


Figure 4-75. Relationship between dissolved oxygen (O_2) at the bottom and species diversity is illustrated for all nine sites. Each point represents one quarterly sample and these are color coded for each sampling location (Figure 4-62). Linear regression shows significant positive relationship between bottom oxygen concentration and diversity.

Biodiversity and Sediment Type

A significant relationship exists between sediment type and species diversity (**Figure 4-76**) and M-AMBI (**Figure 4-77**). Coarser grain sizes support an infaunal macroinvertebrate community with higher diversity. Sites with mucky, organic-rich sediments, especially those in the SLE such as sites M3, M4, and M5, have lower diversity. The presence of fine grain sediments, often termed muck, appears to be directly related to low diversity and low M-AMBI measurements in the SLE.

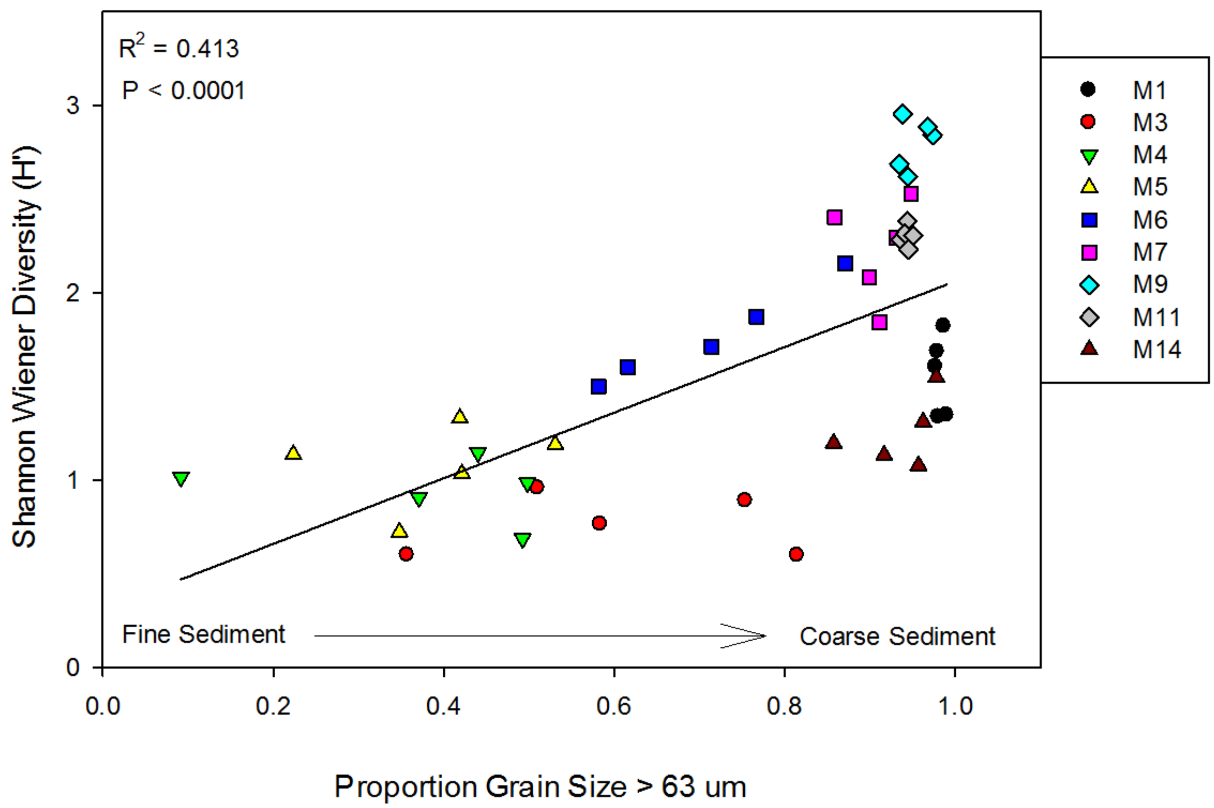


Figure 4-76. Relationship between sediment grain size and species diversity is illustrated for all nine sites. Each point represents one quarterly sample and these are color coded for each sampling location (Figure 4-62). Linear regression shows significant positive relationship between bottom oxygen concentration and diversity.

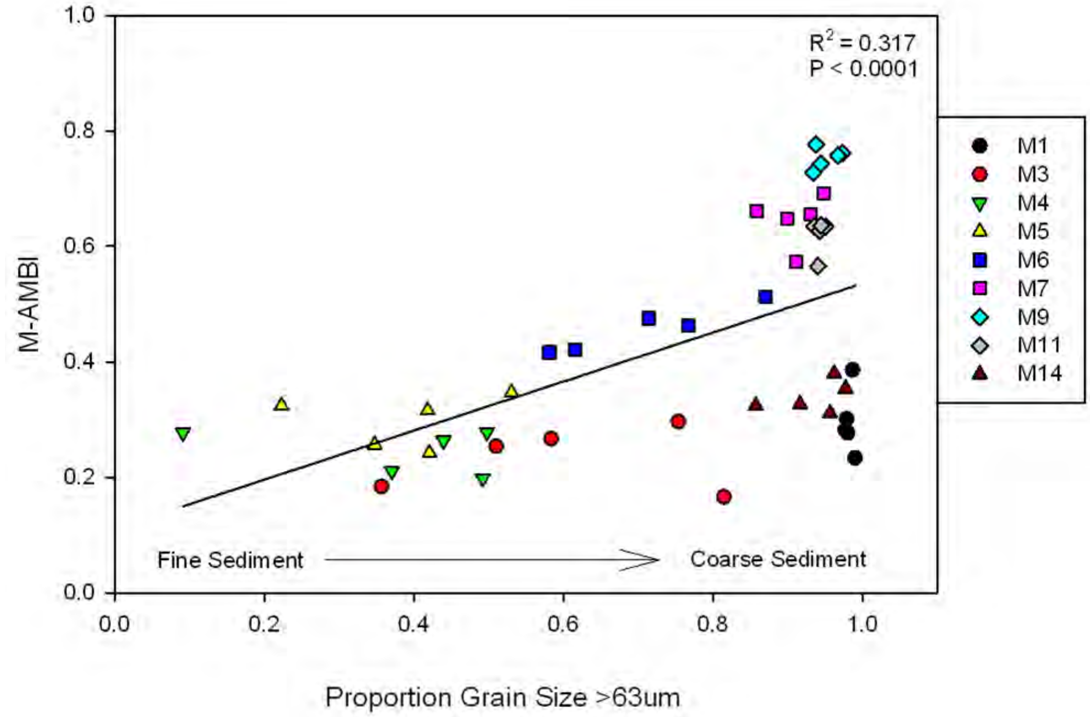


Figure 4-77. Average annual M-AMBI at each site in relationship to proportion sediment grain size > 63 micro meter (µm). Linear regression shows that there is a significant positive relationship, with higher M-AMBI found in coarser, sandier sediments with larger grain sizes.

Conclusions

Variation in salinity explains much of the variation in species diversity and M-AMBI values at these study sites in the SLE and SIRL. Species diversity is greatest at the sites in the SIRL and the lower SLE, which have more stable and higher salinity. To maintain optimal conditions for infaunal invertebrates the salinities should be above 20. Decreases in salinity conditions can cause rapid responses in these infaunal organisms, with defaunation events occurring at sites impacted by heavy freshwater flow. Only a few tolerant, opportunistic species, such as the polychaete worm *S. benedicti*, survive and become dominant under these low salinity conditions.

Another abiotic factor that profoundly impacts species diversity and M-AMBI is the sediment type and grain size. Coarser sediments, which have a high proportion of the sediment with grain size above 63 micrometers (µm), maintain more diverse communities of infaunal invertebrates. Mucky sediments found in the SLE at sites, such as M3 and M4, support relatively low species diversity. Interestingly, the strong relationship seen between salinity and species diversity at other sites in the SIRL and lower SLE is not observed at these sites, suggesting that improvements in salinity alone might not lead to an increase in species diversity or M-AMBI values at these sites. It is likely that the fine, mucky sediments that dominate these sites restrict the species that can live there to only a few opportunistic species.

The value of the benthic infaunal data set that has now developed as a result of nine years of monitoring cannot be underestimated. Few long-term data sets exist for this region, despite the many years of research on the SIRL by a number of research institutions and scientists. Now containing data for 36 quarterly sampling periods, the data can be utilized to look at relationships between biotic and abiotic factors with meaningful statistical analyses. Additional analyses are planned over the next year to further utilize this valuable data set for understanding these benthic infaunal invertebrate communities.

LOXAHATCHEE RIVER ESTUARY

Multiple agencies, including the SFWMD, Loxahatchee River District (LRD), Florida Department of Environmental Protection (FDEP), Florida Park Service, and others, have been working to protect and restore the Loxahatchee River, the first of only two federally designated Wild & Scenic Rivers in Florida. The primary threats to the Loxahatchee River are (1) diminished freshwater flows during the dry season, which leads to saltwater intrusion that harms the freshwater cypress habitat; and (2) flood control releases of fresh water into the estuary during the wet season. Ongoing restoration efforts seek to provide short- and long-term solutions that increase base flows into the Northwest Fork during the dry season, 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 (SFWMD 2006). Conversely, effectively managing (whenever feasible) flood control releases into the estuary during the wet season helps to ensure protection of estuary, oysters and seagrasses.

Since 1971, LRD has been working toward its mission to preserve and protect the Loxahatchee River through an innovative wastewater treatment and reuse program and an active river research, monitoring, and restoration program. The river monitoring work, conducted by LRD's WildPine Laboratory, includes the following:

- The Datasonde project that uses autonomous instrumentation to collect near-continuous (15 or 30 minute intervals) data on water temperature, salinity, and pH at 7 sites throughout the Loxahatchee River
- Oyster monitoring that includes assessing oyster recruitment, density, size, and survival throughout the estuary's extant oyster beds as well as completed oyster restoration projects
- Seagrass monitoring and mapping that includes bi-monthly assessment of seagrasses at 5 sites throughout the river, and large-scale mapping projects completed in 2007 and 2010
- The RiverKeeper water quality project to assess the water quality of nearly 30 parameters including TN, TP, chlorophyll *a*, fecal coliform bacteria, etc., at approximately 40 sites throughout the watershed

LRD has conducted this work to document the condition and ecological health of the river and to identify and prioritize suitable locations for restoration efforts. Over the past 35 years, LRD has contributed significantly to the understanding of the ecology of the Loxahatchee River (see http://www.loxahatcheeriver.org/reports.php). While numerous reports have been written regarding the Loxahatchee River, perhaps none are as comprehensive as the *Restoration Plan for the Northwest Fork of the Loxahatchee River* (SFWMD 2006), and the subsequent addendum completed in 2012 (SFWMD et al. 2012). These documents characterize the watershed, discuss various restoration alternatives, and identify the preferred restoration flow scenario.

The purpose of this report is to provide a concise characterization of the conditions and findings in the Loxahatchee River Watershed during the evaluation period of January 2005 through April 2013.

Loxahatchee River Estuary Study Area

The Loxahatchee River Estuary encompasses approximately 400 hectares and drains a watershed of approximately 700 square kilometers located in northeastern Palm Beach County and southeastern Martin County, Florida (**Figure 4-78**). Fresh water discharges into the estuary from the North Fork, the Northwest Fork, and the Southwest Fork of the Loxahatchee River. The hydrology of the basin has been substantially altered by flood control efforts since the 1950s. Historically (pre-1950), most surface water runoff reaching the estuary originated in Loxahatchee and Hungryland sloughs and flowed gradually to the Northwest Fork. In the 1930s, the Lainhart Dam, a small fixed-weir dam, was constructed in the Northwest Fork at river mile 14.5 to reduce “over drainage” of upstream reaches of the Northwest Fork during the dry season. Since 1947, the Jupiter Inlet, the eastern link to the Atlantic Ocean, was expanded through ongoing dredging projects. Furthermore, in 1958, a major canal (C-18) and flood control structure (S-46) were constructed to divert flows from the Northwest Fork to the Southwest Fork, which increased the intensity and decreased the duration of storm-related discharge to the estuary. These hydraulic modifications promoted increased saltwater flows into the previously freshwater portions of the Northwest Fork.

Loxahatchee River Estuary Hydrologic Analysis

Dry Season – Low Flow

The primary objective of the *Restoration Plan for the Northwest Fork of the Loxahatchee River* and its 2012 addendum is to ensure adequate freshwater flows to provide protection to the freshwater cypress swamp community from saltwater intrusion, particularly during the dry season (SFWMD 2006, SFWMD et al. 2012). SFWMD developed and adopted a minimum flows and levels (MFL) rule in 2003 (Chapter 40E, Florida Administrative Code). The intent of the MFL criteria is to protect the remaining floodplain swamp community from “significant harm”:

> A MFL violation occurs within the Northwest Fork of the Loxahatchee River when an exceedance of the minimum flow criteria occurs more than once every six years. An “exceedance” is defined as when Lainhart Dam flows to the Northwest Fork of the river decline below 35 cubic feet per second (cfs) for more than 20 consecutive days within any given calendar year or when the 20-day moving average salinity measured at River Mile 9.2 exceeds 2.

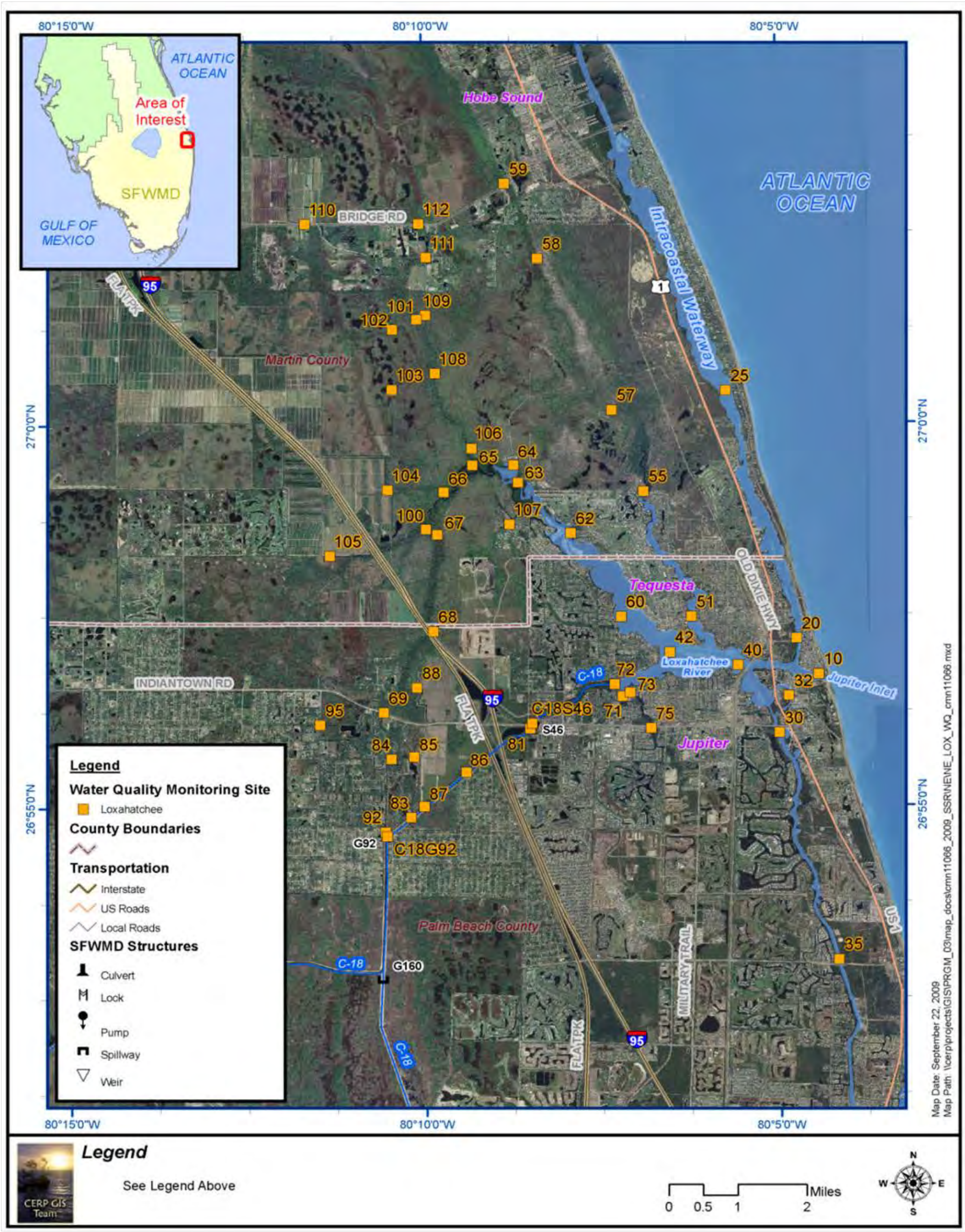


Figure 4-78. Map of Loxahatchee River Watershed.

Among the most noteworthy observations during this assessment period is the improvement in the numbers of days meeting the MFL. **Table 4-10** shows the number of days that flow at Lainhart Dam was less than 35 cfs. Beneficial rains and hydraulic conditions resulted in fewer days of low flow in 2008 and 2010. In spring 2011, SFWMD utilized temporary and permanent infrastructure to test the conveyance of supplemental freshwater flows from the L-8 Reservoir, through Grassy Waters Preserve then into the Loxahatchee River. The water comes to the Loxahatchee River from Grassy Water Preserve via the C-18 Canal through the G-161 water control structure. The SFWMD conducted a 50-day test of supplemental flow from March 1 to April 19, 2011. The supplemental flows provide the additional water needed to maintain a minimum of 35 cfs at Lainhart Dam through much of the dry season. Following the supplemental flow test, an unusually late start to the wet season in spring 2011 accounted for the majority of the 80 days of low flow. In spring 2012 and 2013, meaningful supplemental flows from Grassy Waters Preserve, indicated by the days of flow through G-161, helped to ensure the MFL criteria were met through the majority of the dry season. The "wet years" and supplemental flow tests clearly demonstrated that the MFL criteria can be achieved with nominal additions of fresh water, and that the MFL targets provide the intended protection to the freshwater habitat, which is further described below.

Table 4-10. Counts of respective flow conditions by year from January 1, 2000 through April 30, 2013, Loxahatchee River, Florida. Data from SFWMD's hydrometeorological database, DBHYDRO.

Criteria	Year													
	2000	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013[1]**
Low Flow														
Number of days MFL not met (< 35 cfs at Lainhart Dam)	118	162	70	98	151	69	171	134	41	70	2	80	2	13[2]
Number of days dry season supplemental flows from Grassy Waters Preserve (measured at G-161)[3]								0	3	24	39	49	70	85
Flood Control														
Number of days flood control releases at S-46 greater than 600 cfs	2	15	2	0	17	8	0	1	1	0	1	0	17	

1. January 1, 2000 through April 30, 2013.
2. USGS Station 02277600.
3. Started October 2007.

Wet Season – Flood Control Releases

During the wet season, the objective for water managers is to minimize the intensity and frequency of flood control releases. Large freshwater releases reduce salinities in the estuary to levels that harm estuarine and marine flora and fauna with oyster and seagrasses being among the most notable. Salinity analysis conducted by LRD in 2008 demonstrated that harmful salinity variability within the estuary was significantly reduced when flows at the primary flood control structure (S-46) were less than 300 cfs. The salinity analysis suggests that seagrasses are likely stressed by salinities less than 15, which occurs when flow through the S-46 is greater than 600 cfs. These findings suggest that, within the Loxahatchee River system, a longer duration lower flow release is preferred to heavy flows for a shorter period (i.e., pulsed flows). LRD communicated these findings to the SFWMD Operations staff and it appears they are managing the system to maintain lower flows whenever feasible. The health and extent of oysters and seagrass throughout the estuary indicate a functional system.

The recent exception to moderated flood control releases was the 17 days of flow greater than 600 cfs following passage of Tropical Storm Isaac in August 2012 (**Table 4-10**). SFWMD water managers had to release huge volumes of water to deal with flooding in some communities. **Figure 4-79** illustrates the relative magnitude of Tropical Storm Isaac flood control releases relative to other significant storm events. Subsequent assessment of oyster communities indicated severe impacts to oyster reefs in the Southwest Fork and moderate impacts to oysters in upstream reaches of the Northwest Fork. Additional discussion is provided in the oyster section below.

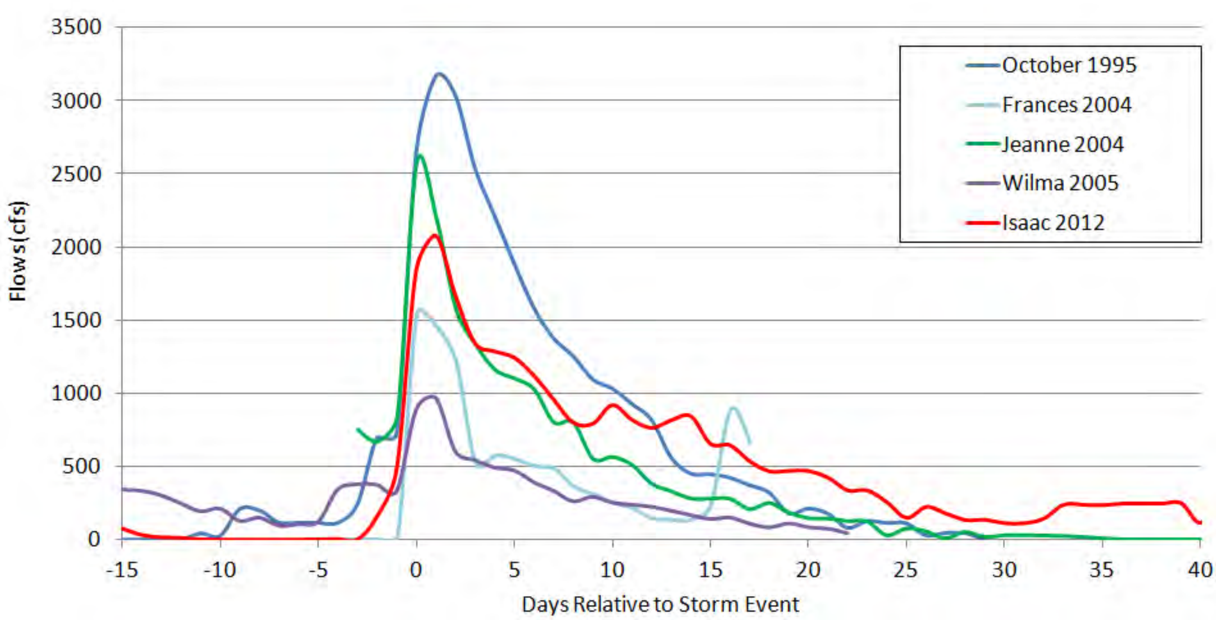


Figure 4-79. Comparison of storm-related flood control releases from S-46.

Loxahatchee Estuary Salinity

High frequency (15-minute interval) datasonde instrument data provides detailed salinity data that improves understanding of how the estuary functions. From these data the daily minimum and maximum salinity can be extracted. Cumulative distribution plots of the daily minimum and maximum values are created to form a "salinity envelope" that illustrates the total distribution of salinity at a selected location. An example is provided in **Figure 4-80** for the "Oyster Island" site in the middle estuary. These data show that 50% of the time, salinity at Oyster Reef site were greater than 15 and less than 25 (i.e., organisms living at this site typically experienced a salinity envelope of 15 to 25). Organisms at this site experienced salinities less than 10 less than 20% of the time. Such salinity envelopes will further understanding of organismal responses to salinity conditions in estuaries, which can be highly dynamic.

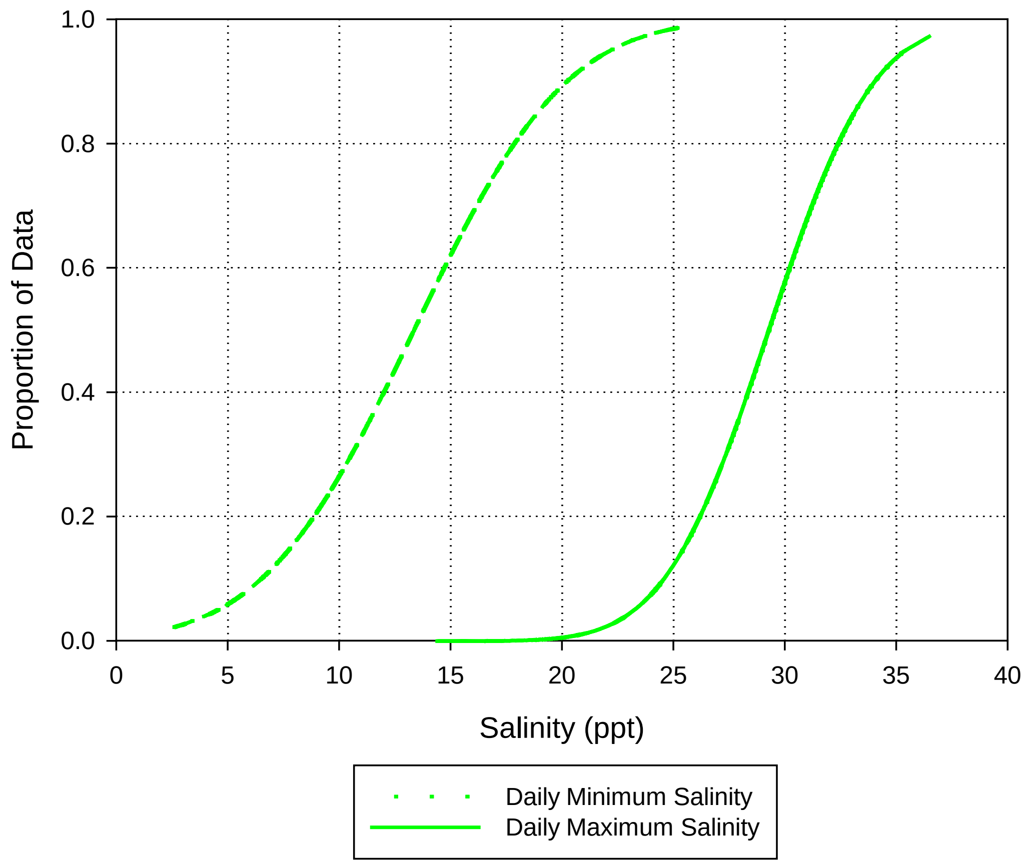


Figure 4-80. Typical salinity envelope for the oyster reef site at River Mile 4.1 in the Northwest Fork of the Loxahatchee River. Dashed and solids lines are the lower and upper bounds of the envelope shown as the cumulative distribution plot of the daily minimum and maximum salinity recorded by a datasonde at 15-minute intervals between November 2008 and October 2009, an uneventful year for rainfall and river flows.

Salinity envelopes enable comparison between salinity distributions for typical versus atypical conditions at the various sampling sites. For example, during the dry season, the effects of drought and low freshwater inflows on salinity in freshwater swamp reaches can be compared. **Figure 4-81** illustrates a "typical" dry season salinity envelope relative to the unusually dry, dry season and decreased freshwater flows at the Kitching Creek (surface) station. The figure clearly shows the dry season salinity envelope is skewed towards much higher salinities for the "unusually dry" period relative to the "typical dry" period. For example, daily salinities at this site exceeded 10 during 2% of a typical dry season days whereas salinities exceeded 10 during 40% of unusually dry periods.

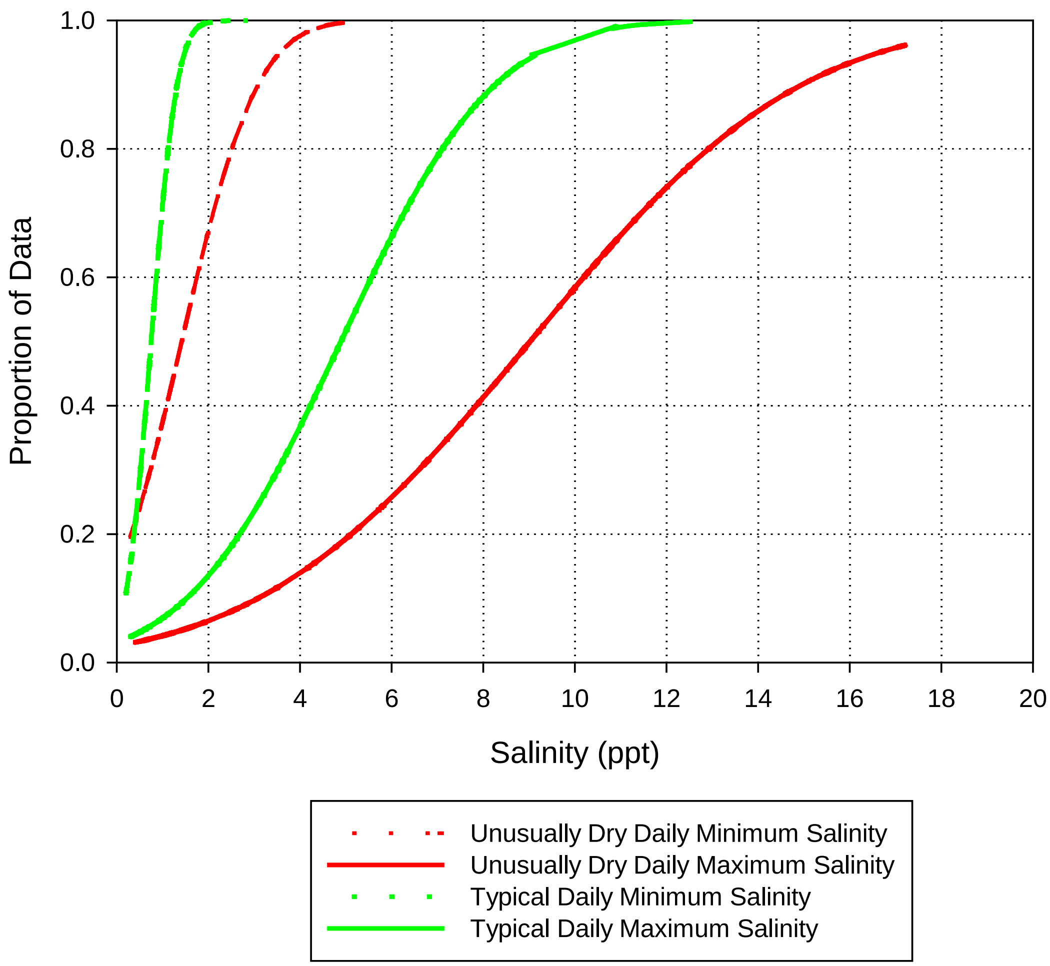


Figure 4-81. Comparison of dry season (November–April) salinity envelopes at the upstream Kitching Creek site for "typical" and "unusually dry" years in the Loxahatchee River, Florida. Green lines illustrate the lower and upper bounds of the salinity envelope for a typical dry season from November 2008–April 2009. Red lines illustrate the lower and upper bounds of the salinity envelope for the unusually dry, dry season in November 2006–April 2007 showing the increased salinities resulting from the decreased freshwater flows.

Conversely, during the wet season, concern shifts to assessing the effects of flood control releases on seagrasses and associated fauna in the lower estuary. **Figure 4-82** illustrates a typical wet season salinity envelope at the North Bay site versus an unusually wet, wet season that clearly reduced the salinity distribution. These figures illustrate the magnitude of salinity variation in the estuary, and then monitoring of seagrass and oysters track the subsequent effects to these resources. Generally, the estuary experiences limited periods of harmful low salinity conditions. The system seems remarkably resilient to such 'normal' perturbations—decimated oysters recover in 12 months and some seagrass species appear to recover despite loss of coverage. However, infrequent, high magnitude disturbances such as observed following Tropical Storm Isaac appear to have disproportionate damaging effects on seagrasses (see the Submerged Aquatic Vegetation section below for a more thorough discussion). These salinity envelope plots help specifically identify the apparent stressful conditions causing seagrass mortality. Fortunately, the system seems to be managed well during typical conditions given the extent and health of the resources (oysters and seagrass). Unfortunately, there are no simple solutions for managing the system and protecting the resources during these extreme events.

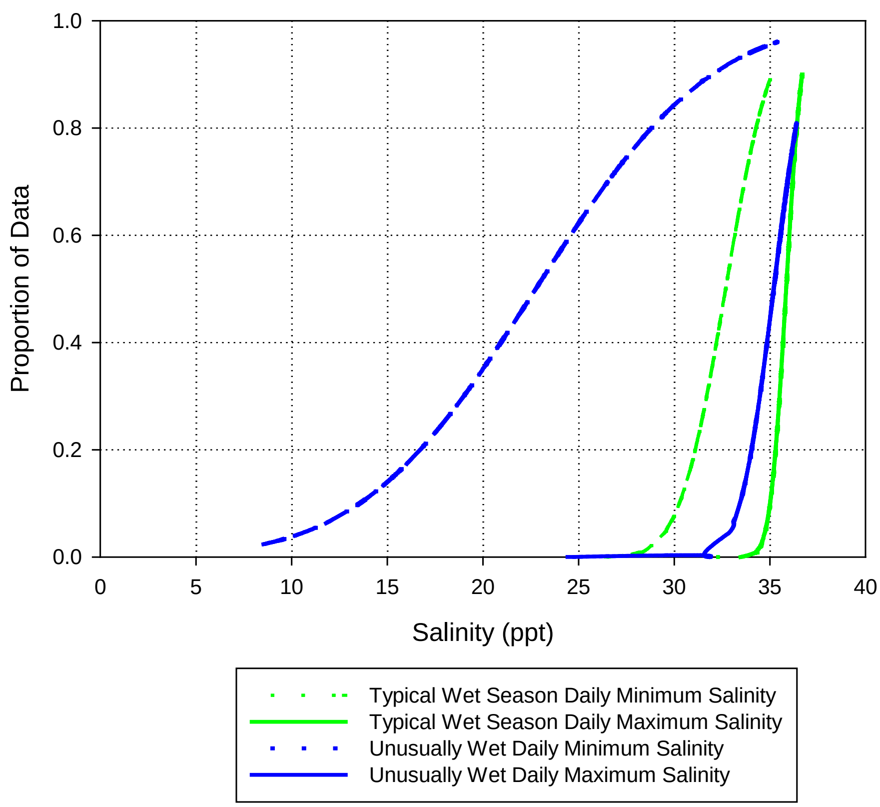


Figure 4-82. Comparison of wet season (May–October) salinity envelopes at the downstream North Bay site for "typical" and "unusually wet" years in the Loxahatchee River. Green lines illustrate the lower and upper bounds of the salinity envelope for a typical wet season from May 2009–October 2009. Blue lines illustrate the lower and upper bounds of the salinity envelope for the unusually wet, wet season in May 2007–October 2007 showing the decreased salinities resulting from the increased freshwater flows and flood control releases.

Loxahatchee River Estuary Oysters

Oysters within the LRE are monitored by two different agencies. The LRD has a regular monitoring program at key locations to track natural reefs in the estuary. Also, during the past few years they performed specialized monitoring of restoration sites in several areas of the lagoon. These sites were monitored by the LRD twice a year for recruitment and density counts of live oysters. In addition, as part of the larger east coast oyster monitoring program, FWC measures several key indicators of oyster survival and health in the LRE at different locations than those monitored by the LRD. Both types of monitoring and an assessment of the data collected are provided in this report.

Loxahatchee River District Oyster Program

Since 2007, LRD has monitored oyster recruitment activity on National Oceanic and Atmospheric Administration (NOAA) restoration sites in the Northwest and Southwest forks. From this data, a bimodal recruitment pattern was observed with the first peak of settlement occurring in spring and a second peak occurring in fall. This monitoring also enables evaluation of the magnitude of spawning events. In the Northwest Fork (**Figure 4-83,** top panel) average spat counts are routinely greater than 50 spat per array of 10 shells (i.e., 5 spat per shell). In the Southwest Fork, settlement rates are comparatively lower but there was an episode of intense settlement in spring and fall 2012 (**Figure 4-83,** bottom panel).

During summer 2010, the LRD partnered with Martin County to obtain American Recovery and Reinvestment Act funding from NOAA to restore 5.84 acres of oyster reef. This project was an effort to augment extant natural oyster reefs and provide additional habitat. An added benefit was the pre-mitigation for possible losses of oyster resulting from increased base flows as predicted by the *Restoration Plan for the Northwest Fork of the Loxahatchee River* (SFWMD 2006). Ongoing monitoring of the restoration reefs each summer and winter indicates oyster densities ranging from 300 to 800 oysters per m^2 and typical shell heights between 25 and 43 millimeters (mm) (**Figure 4-84**). As anticipated, densities are generally decreasing (compared to post construction) and average oyster size is increasing. The oyster restoration sites are flourishing in terms of both oysters (LRD) and macrofauna (Florida International University) (reports and publications are available online at www.loxahatcheeriver.org/reports.php.)

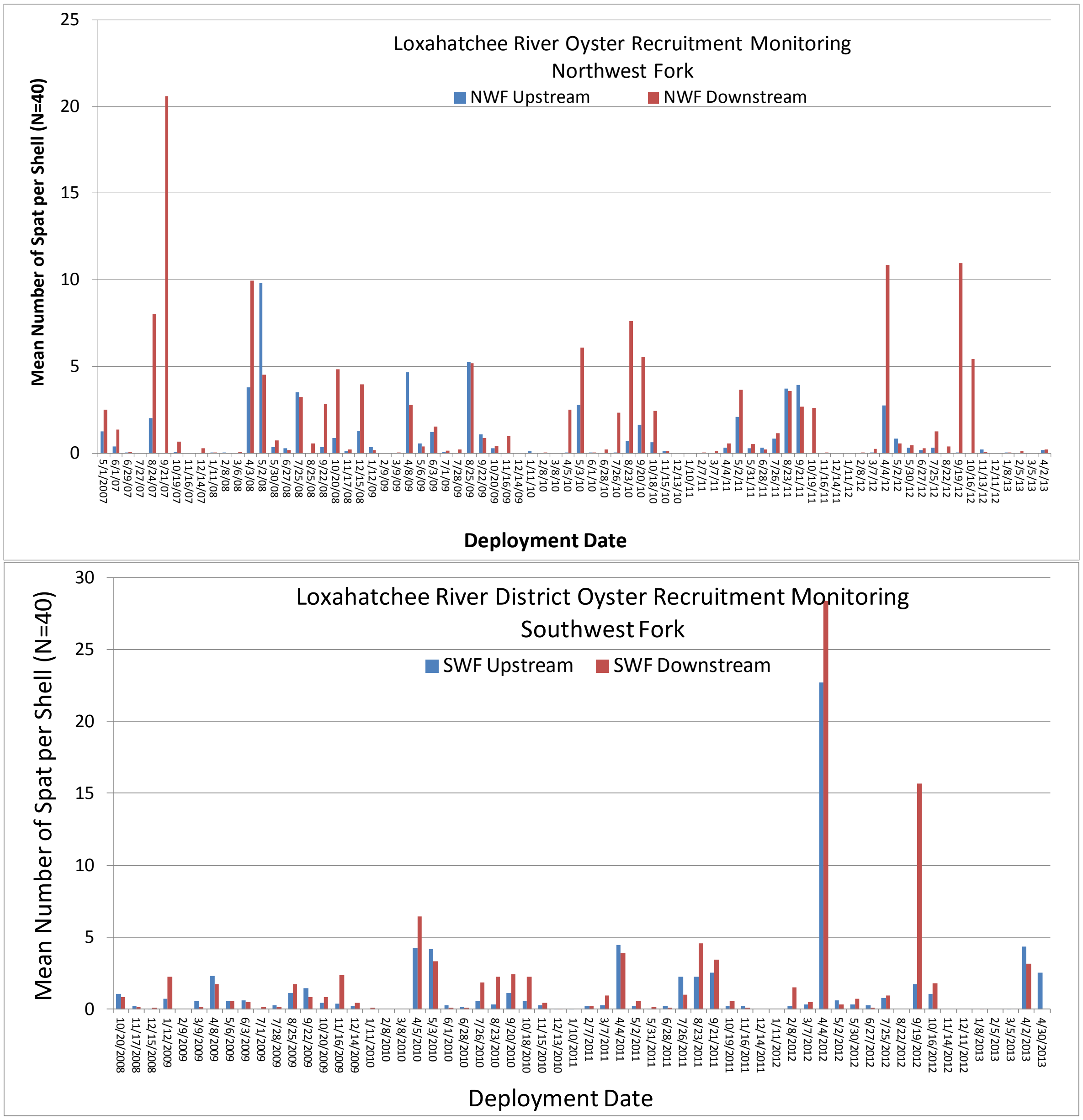


Figure 4-83. Oyster recruitment monitoring results for the Northwest Fork (NWF) (top panel) and the Southwest Fork (SWF) (bottom panel) of the Loxahatchee River.

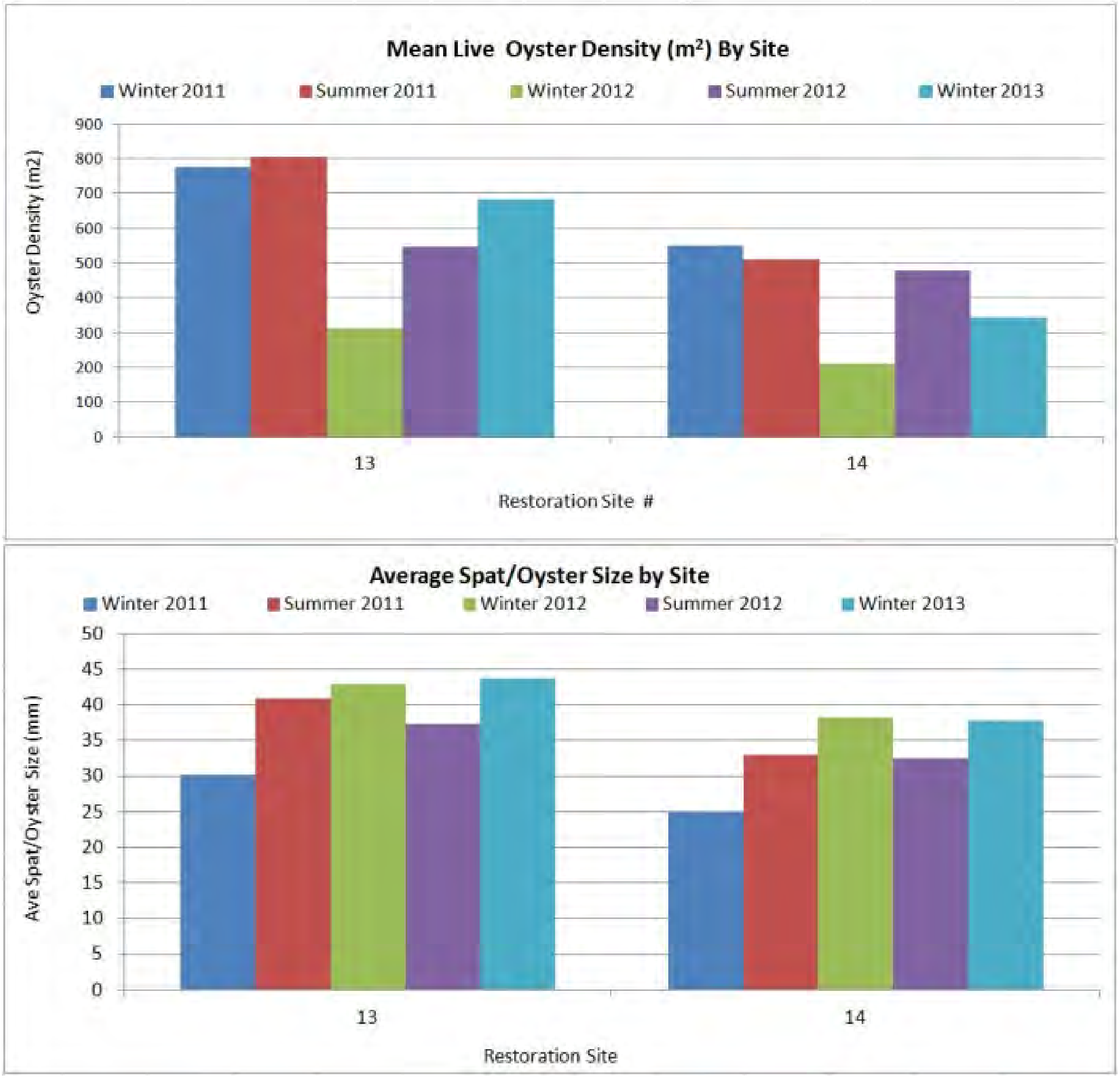


Figure 4-84. Oyster density (top panel) and size (bottom panel) by sampling event at the NOAA oyster restoration sites 13 and 14, Loxahatchee River, Florida

LRD mapped and assessed all oyster beds in the Northwest and Southwest forks in 2008. **Figure 4-85** shows the location and areal coverage of oyster beds in the Northwest Fork (left panel) and the Southwest Fork (right panel). The approximate total acreage of oysters in the estuary is 15.1 acres, with 13.9 acres (92%) in the Northwest Fork, and 1.2 acres (8%) in the Southwest Fork.

Figure 4-85. Locations of oyster beds from the 2008 oyster mapping project for the Northwest Fork (left) and Southwest Fork (right).

In summer 2013, LRD reassessed the oyster reefs by sampling oyster density, vitality, and size at the exact sample points assessed in 2008. **Figure 4-86** shows the vitality (percent live) at each bed for the Northwest Fork (left panel) and the Southwest Fork (right panel). The largest and healthiest oyster beds in the estuary are in the vicinity of the newly constructed oyster restoration reefs shown by the white fill (left panel). In 2013, fewer than 50% of oysters sampled at the upstream oyster beds were alive. Oyster beds in the Southwest Fork demonstrated good survival rates despite negative impacts following the flood control releases due to Tropical Storm Isaac in August 2012.

Figure 4-86. Oyster beds symbolized by percent live during 2013 for the Northwest Fork (left) and Southwest Fork (right).

To compare the observations of 2008 to 2013, the relative percent difference in live oysters was computed for each oyster bed. The most notable difference was the reduction in the number of live oysters in the upstream reaches of the Northwest Fork (**Figure 4-87**). These results, albeit a one-time comparison in a highly variable organism, suggest relative decrease in live oysters upstream of Island Way Bridge. This decrease in oysters was a predicted outcome of increased flows in the 2006 restoration plan. Fortunately, the large-scale oyster reef restoration project provided more than double the acreage of oyster beds than are present north of the Island Way Bridge, and many of those oysters north of the Island Way Bridge are still viable with 50 to 75 percent live oysters.

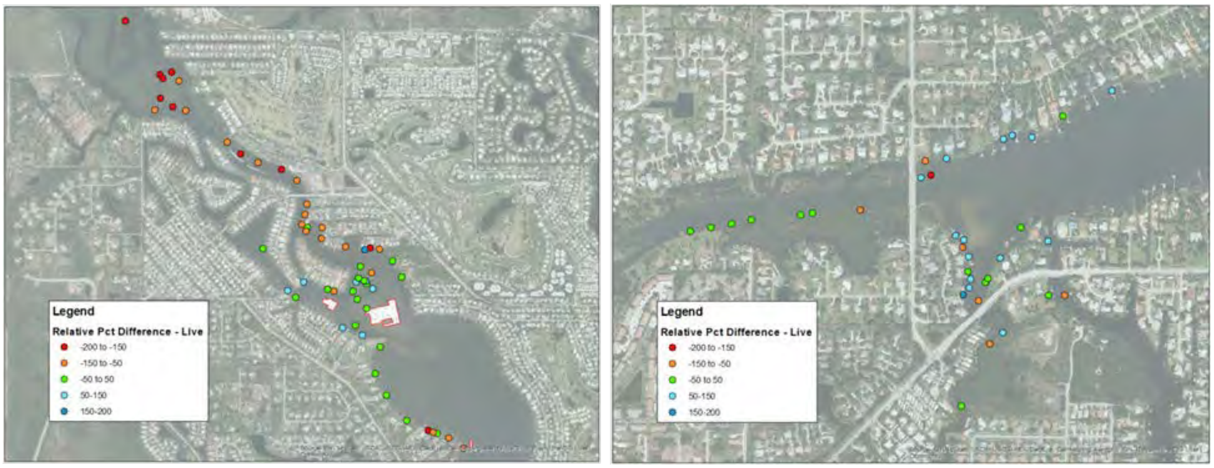


Figure 4-87. Relative percent difference of the number of live oysters at each oyster bed in 2008 versus 2013. Green indicates minimal change from 2008 while warm colors (towards red) are decreases in live oysters and cool colors (towards blue) are increases in live oysters between 2008 and 2013.

Another way of visualizing these data is **Figure 4-88** where the oyster beds are spatially grouped moving from downstream (left) to upstream (right). There is a decrease in the relative percent difference in both the total count of oysters and counts of live oysters in the downstream oyster beds (left) then increased towards the vicinity of the oyster restoration reefs (middle), then decreased once more as sampling moved upstream from the area of restoration reefs. The lines show the precipitous decrease in the percent of live oyster in the upstream reaches.

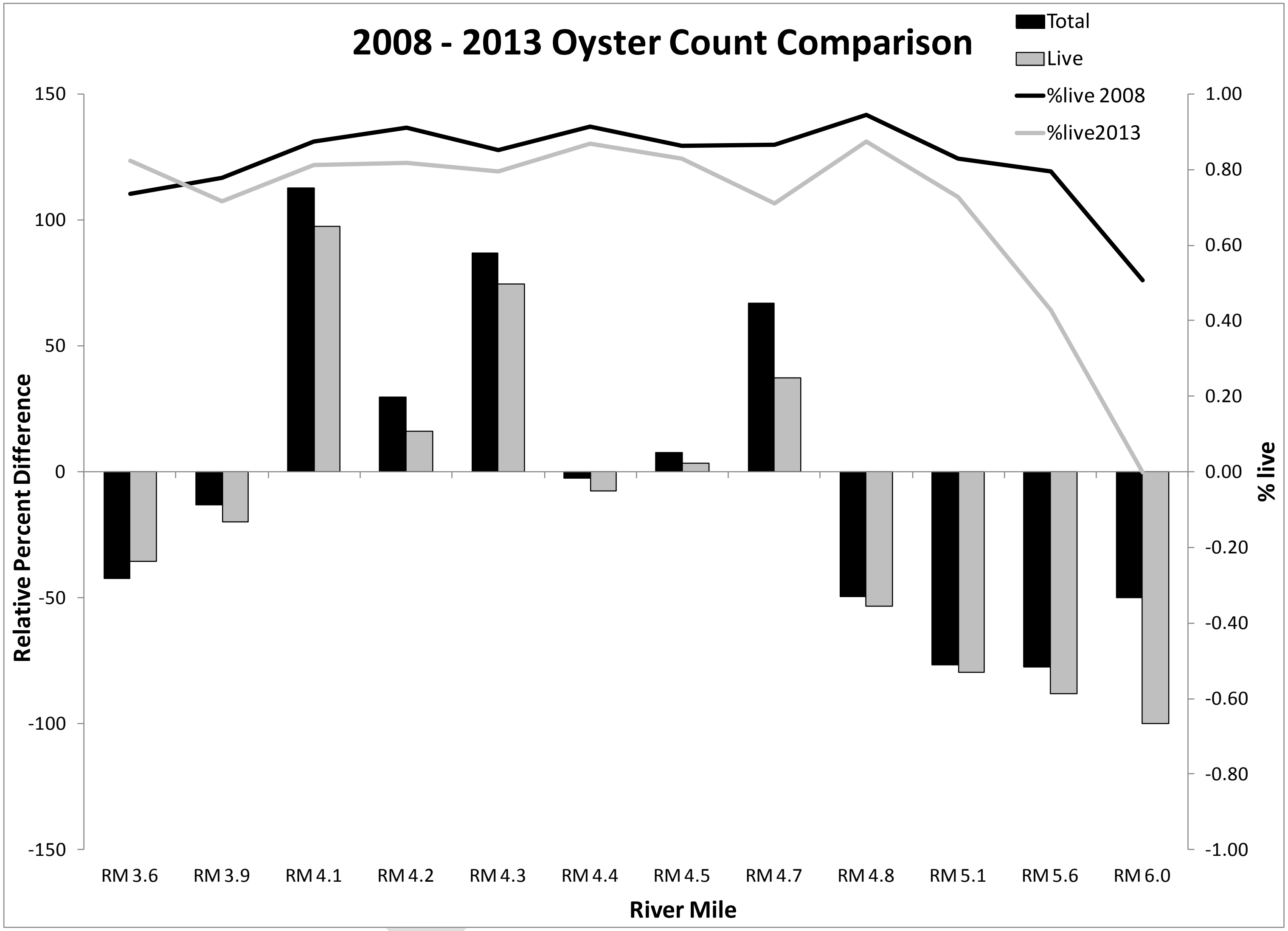


Figure 4-88. Spatially compartmentalized oyster count comparisons of relative percent difference of total and live oyster counts (bars) and percent live for 2008 to 2013 (lines).

Florida Fish and Wildlife Conservation Commission Monitoring

Oysters are also monitored in LRE by the FWC at locations shown in **Figure 4-32**. Freshwater inflows into the estuary have been altered from a natural state to one in which inflows are more variable and extreme thus adversely affecting local oyster populations. The estuary has experienced impacts as a result of the altered salinity regime: exposure to high freshwater inflows during the wet season, which leads to a rapid decline in oyster health and abundance, and too little freshwater inflow during the dry season or drought periods, which leads to a gradual increase in predation, disease, and mortality. In the LRE, salinities typically exceed the optimal salinity range and only fall within the optimal range intermittently during the wet season.

The goals of the established salinity performance measure for the LRE are to reduce saltwater intrusion in the Northwest Fork and to reduce peak freshwater discharges through S-46 to the Southwest Fork. Specifically, the goal is to have variable dry season flows between 50 and 110 cfs, with a mean monthly flow of 69 cfs over Lainhart Dam, while providing an additional 30 cfs from the downstream tributaries. Conditions should not permit salinity to fall below 15 at River Mile 1.74 for six days or more. At present, salinity goals in the LRE are optimized for seagrass (specifically *Syringodium*) and upstream flora but not oysters. Based on preliminary analysis, mean flows from S-46 and Lainhart Dam that exceed 400 cfs for 7 days can reduce salinities below tolerable limits for oysters in the prime habitat (~river mile 4.5 in the Northwest Fork; Florida Wildlife Research Institute monitoring sites in Southwest Fork). The currently recommended minimum flow as described above (~100 cfs summed flow) will likely be sufficient to maintain salinities below or near upper optimal limits at these sites.

Based on an analysis by water year, salinity conditions were generally within the optimal range of 12–20 in the Northwest Fork (**Figure 4-89**), but consistently exceeded those limits in the Southwest Fork (**Figure 4-90**). Monthly data revealed that there were protracted periods in the Southwest Fork when salinity exceeded 20 (December 2005–August 2006, October 2006–June 2007, December 2007–June 2008, November 2008–May 2009, July 2009–May 2010, November 2010–August 2011, and November 2012–April 2013) and also brief but intense periods when salinity fell below 5 (June 2005, October 2007, October 2008, September 2010, November 2011, and September 2012). In the Northwest fork, salinities fell below 12 for longer periods in WY 2006 (September 2005–December 2005), WY 2008 (June 2007–October 2007), WY 2010 (August 2009–September 2009), and WY 2013 (Jul 2012–October 2012). In an effort to relate the biological responses of oysters to changes in salinity, each measured biological parameter has been categorized into thresholds representing good (green), fair (yellow), or poor (red) for oysters in the LRE (**Table 11**).

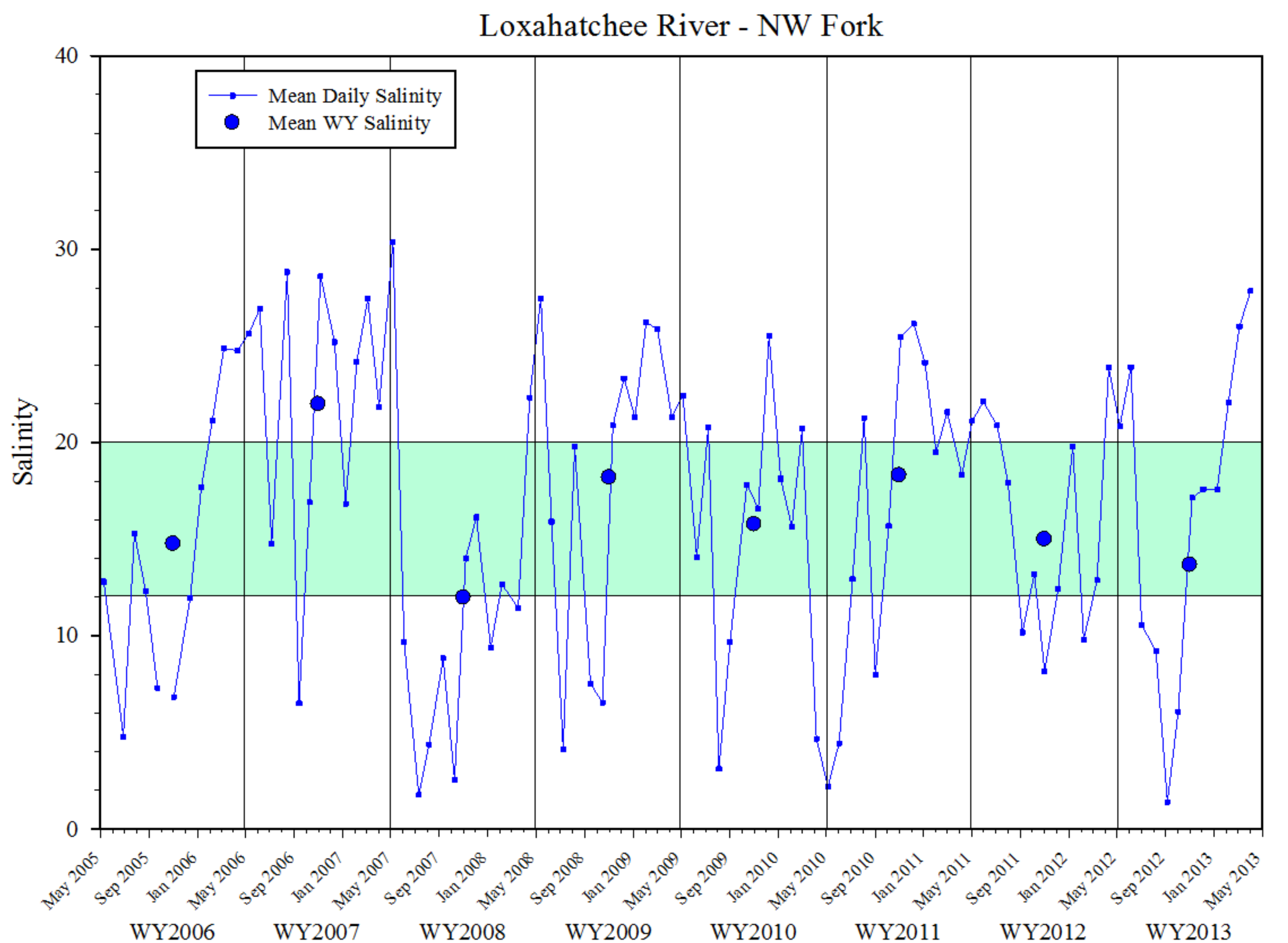


Figure 4-89. Mean monthly salinity and mean water year (May 1–April 30) salinity from the Northwest Fork of the LRE. The green band represents the salinity range deemed most favorable for survival and health of juvenile marine fish, oysters, and SAV.

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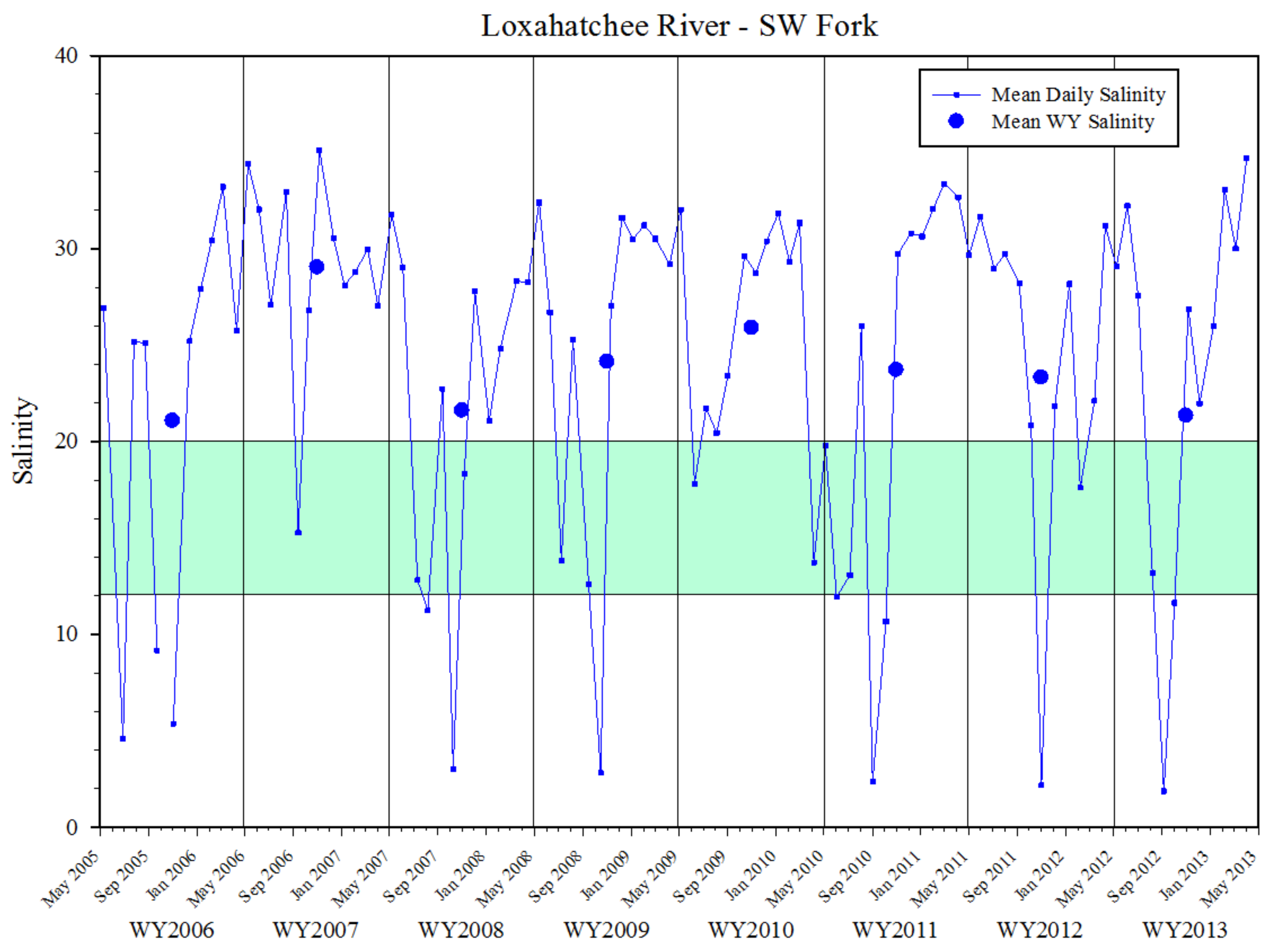

Figure 4-90. Mean monthly salinity and mean water year (May 1–April 30) salinity from the Southwest Fork of the LRE. The green band represents the salinity range deemed most favorable for survival and health of juvenile marine fish, oysters, and SAV.

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Table 4-11. Thresholds for measured biological parameters of oysters in the two LRE study sites.

Parameter	Poor (Red)	Fair (Yellow)	Good (Green)
Density	0–100 oysters/m^2	101–500 oysters/m^2	> 500 oysters/m^2
Recruitment Rate (May–October)	0–1 spat per shell per month	> 1–5 spat per shell per month	> 5 spat per shell per month
Dermo Infection Prevalence	> 50%	21–50%	0–20%
Dermo Infection Intensity	Heavy	Moderate	Uninfected to Light

The density of live oysters is typically higher in the Northwest Fork than in the Southwest Fork of the LRE; however, in both forks, live densities rarely meet the good (green) threshold and most often fall into the fair category (**Figures 4-91** and **4-92**). During WY 2006 and WY 2007, live densities fell into the poor category (red; < 100 oysters/m^2) for three of the four surveys conducted in the Southwest Fork during that time. Conversely, live densities during WY 2009, WY 2010, and WY 2011 reached the good threshold in the Northwest Fork. With the exception of WY 2006, low salinity events do not appear to be a major factor limiting oyster populations in the LRE.

When compared to reproductive development in SLE oysters, LRE oysters generally have more months of the year when at least some developing stage oysters are present and also higher percentages of developing oysters (**Figures 4-93** and **4-94**). As a result, there appears to be more frequent and sustained, low-level recruitment, which may add resiliency to the population. During the peak recruitment season (April–September), recruitment rates in the LRE are moderate and typically classified in the fair category (**Figures 4-95** and **4-96**). Because salinity generally exceeded the optimal range in the Southwest Fork, some of the highest recruitment peaks followed or were associated with small inflows of fresh water (September 2005, September 2006, September 2008, and May–September 2010). Similar positive associations between freshwater inflow and recruitment occurred in the Northwest Fork (September 2006, May–September 2008, and September 2010).

The impacts from dermo infection were slightly higher in the Southwest Fork than in the Northwest Fork of the LRE. During the first year of the study (WY 2006), infection prevalence and intensity in both forks were similar and rarely exceeded acceptable limits (**Figures 4-97** through **4-100**). This coincided with a period of lower salinities in both the Northwest and Southwest forks. However, since that time, infection prevalence has risen steadily to fair and poor levels. This indicates that freshwater inflows into the estuary have not been of sufficient magnitude or duration to provide relief from disease pressure.

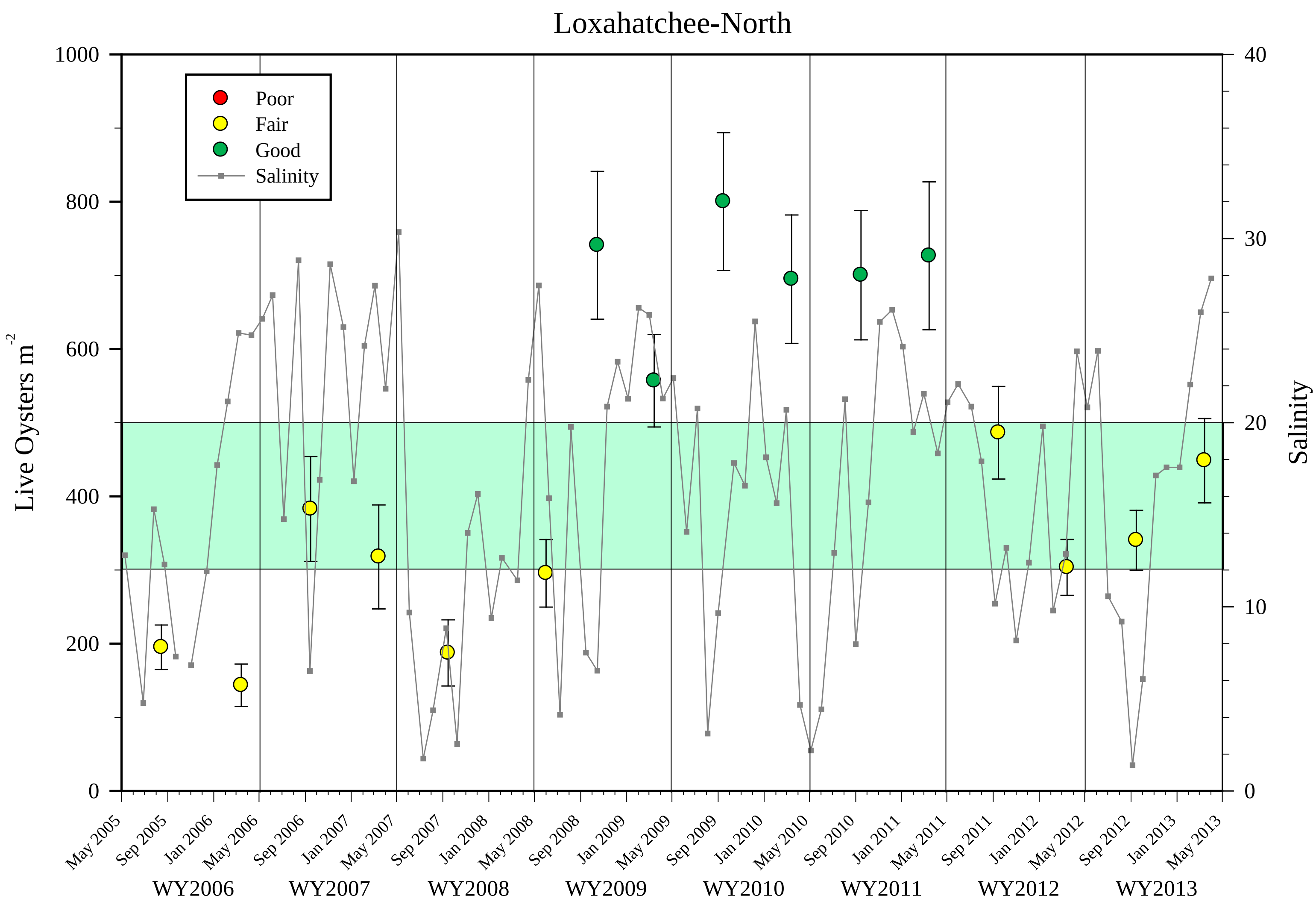

Figure 4-91. Mean number (± standard deviation) of live oysters (red, yellow, and green circles) in the Northwest Fork of the LRE during semi-annual surveys and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

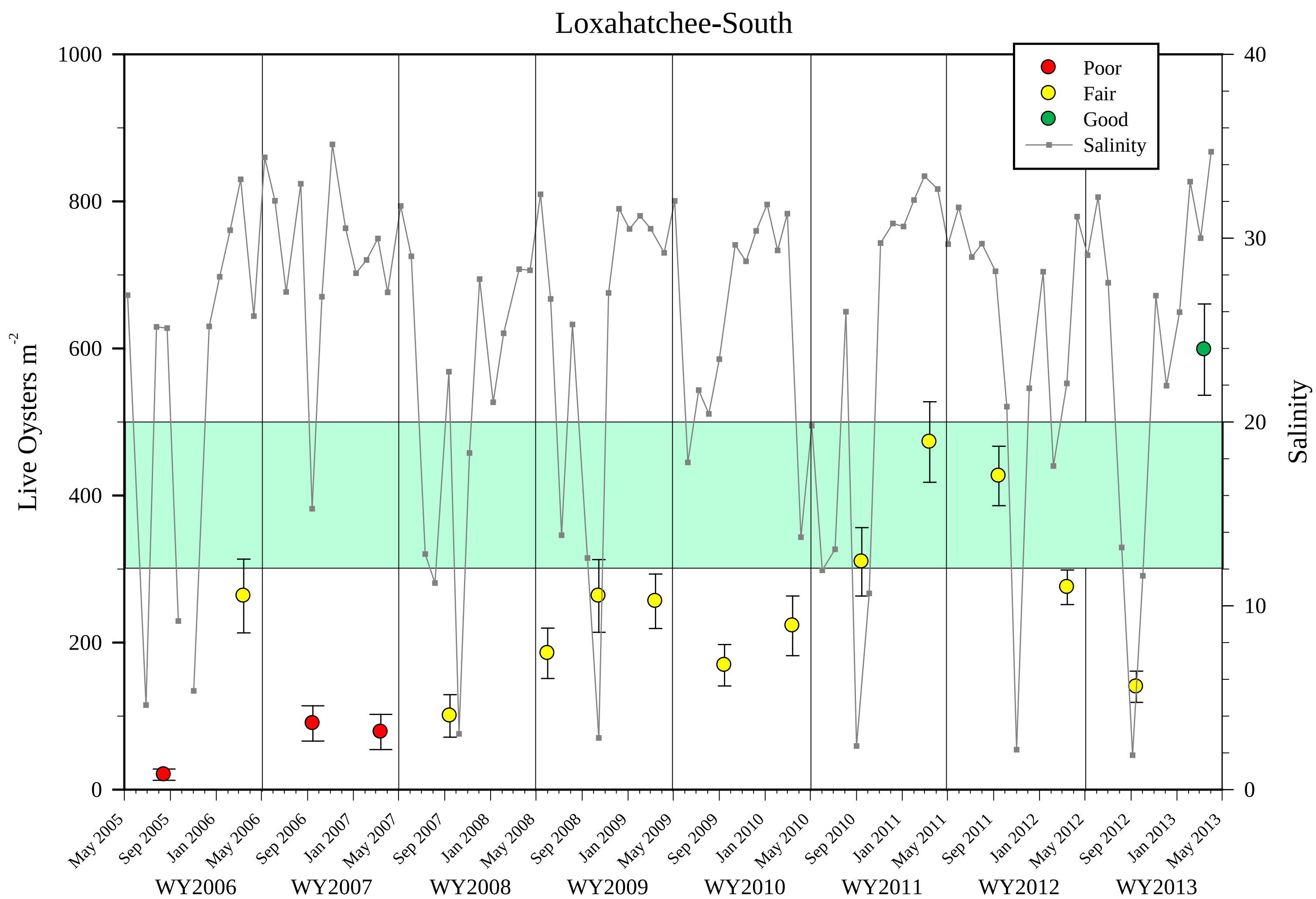

Figure 4-92. Mean number (± standard deviation) of live oysters (red, yellow, and green circles) in the Southwest Fork of the LRE during semi-annual surveys and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

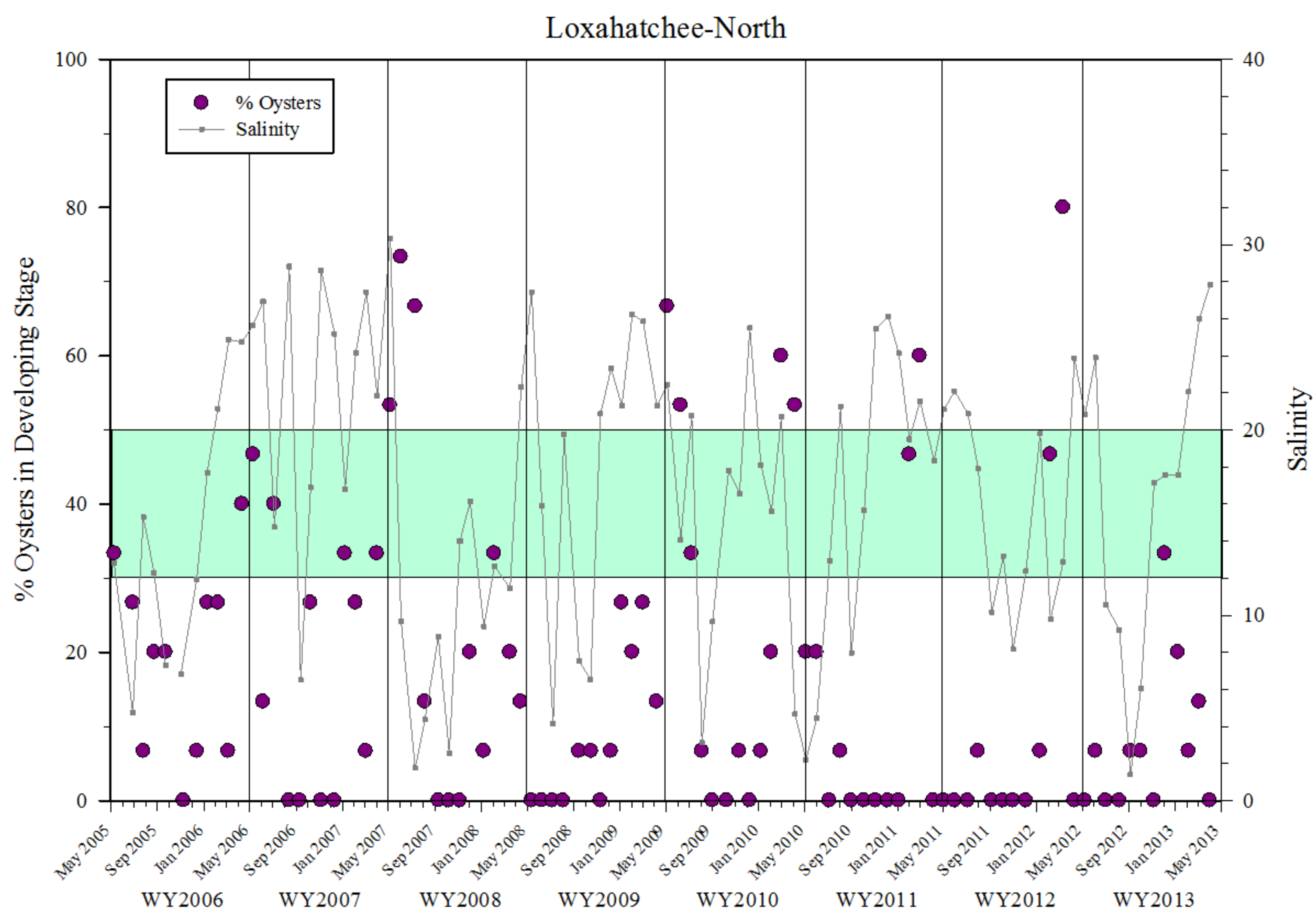

Figure 4-93. Reproductive development, represented as the percentage of oysters in the gonadal development stage, of oysters collected from the Northwest Fork of the LRE each month and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

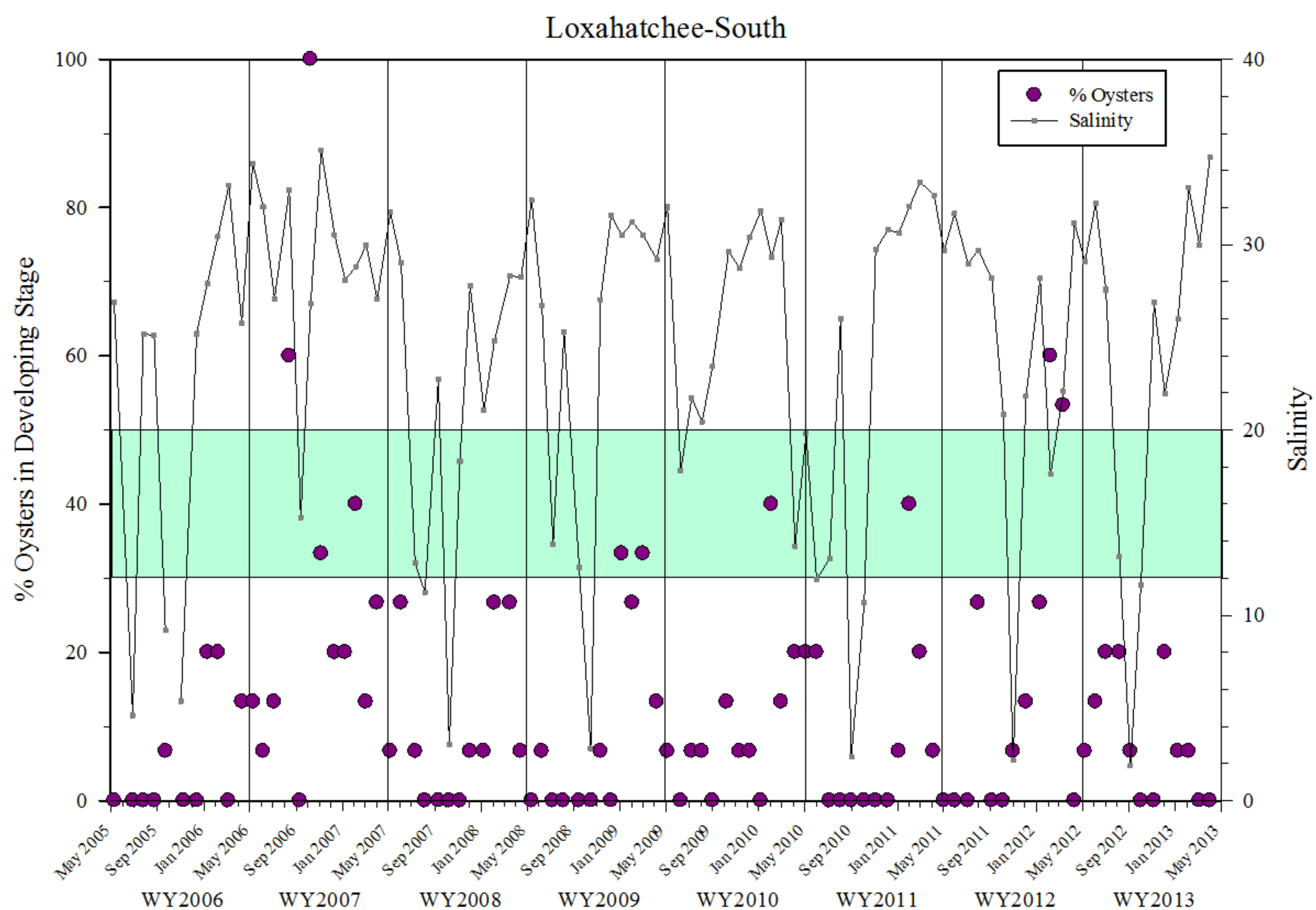

Figure 4-94. Reproductive development, represented as the percentage of oysters in the gonadal development stage, of oysters collected from the Southwest Fork of the LRE each month and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

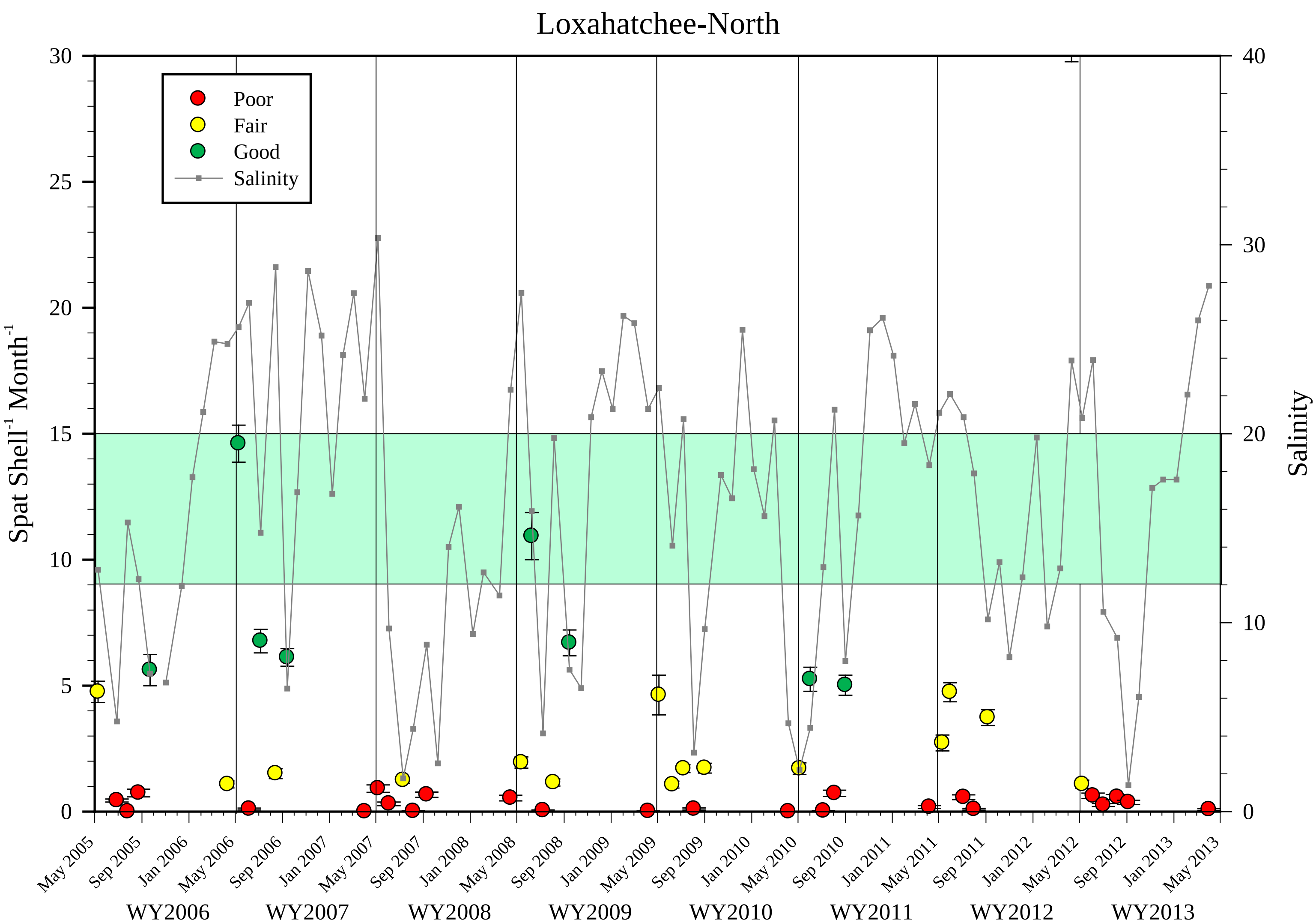

Figure 4-95. Mean number (± standard deviation) of oyster recruits per shell (red, yellow, and green circles) during monthly collections from April through September of each calendar year in the Northwest Fork of the LRE and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

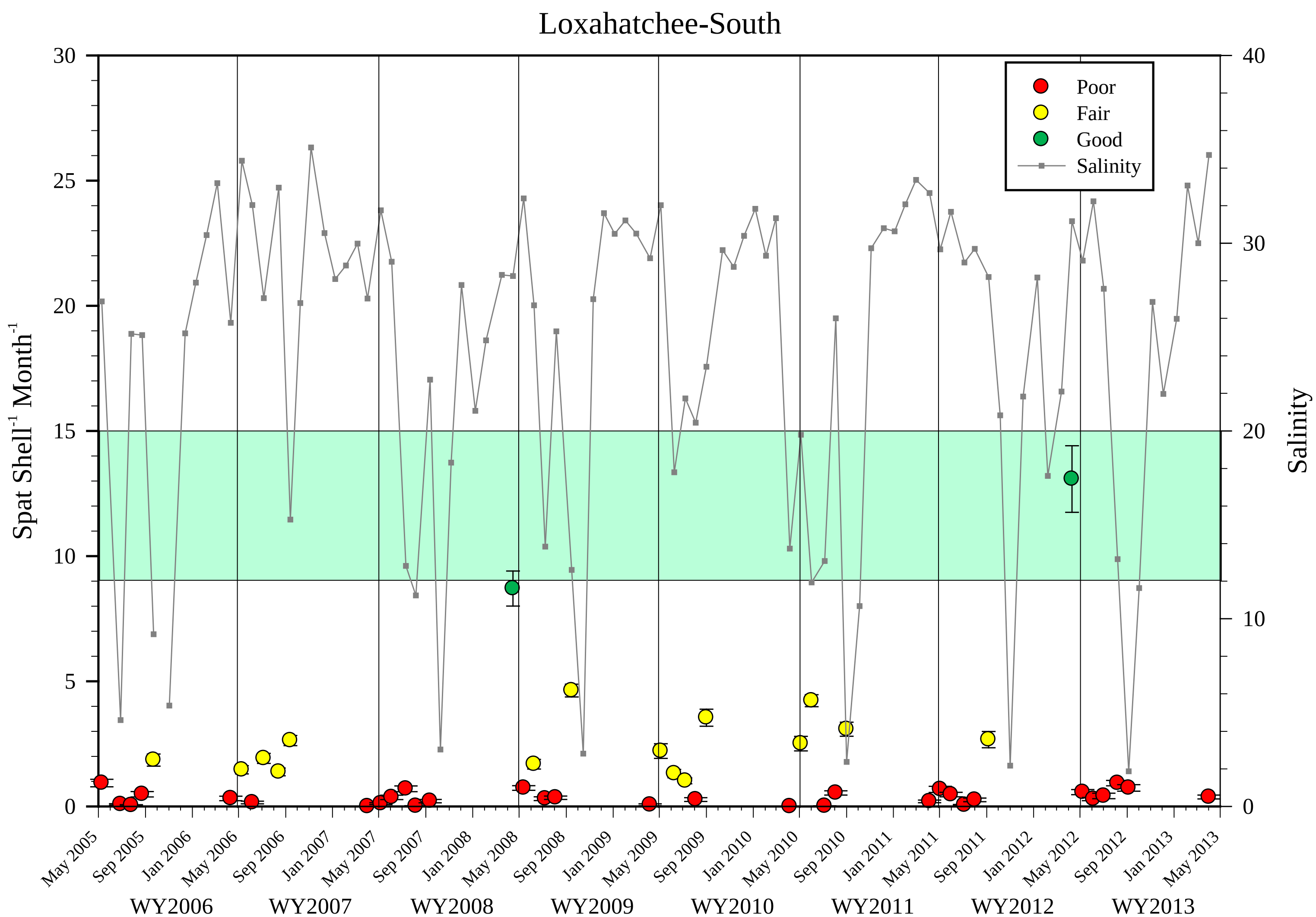

Figure 4-96. Mean number (± standard deviation) of oyster recruits per shell (red, yellow, and green circles) during monthly collections from April through September of each calendar year in the Southwest Fork of the LRE and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

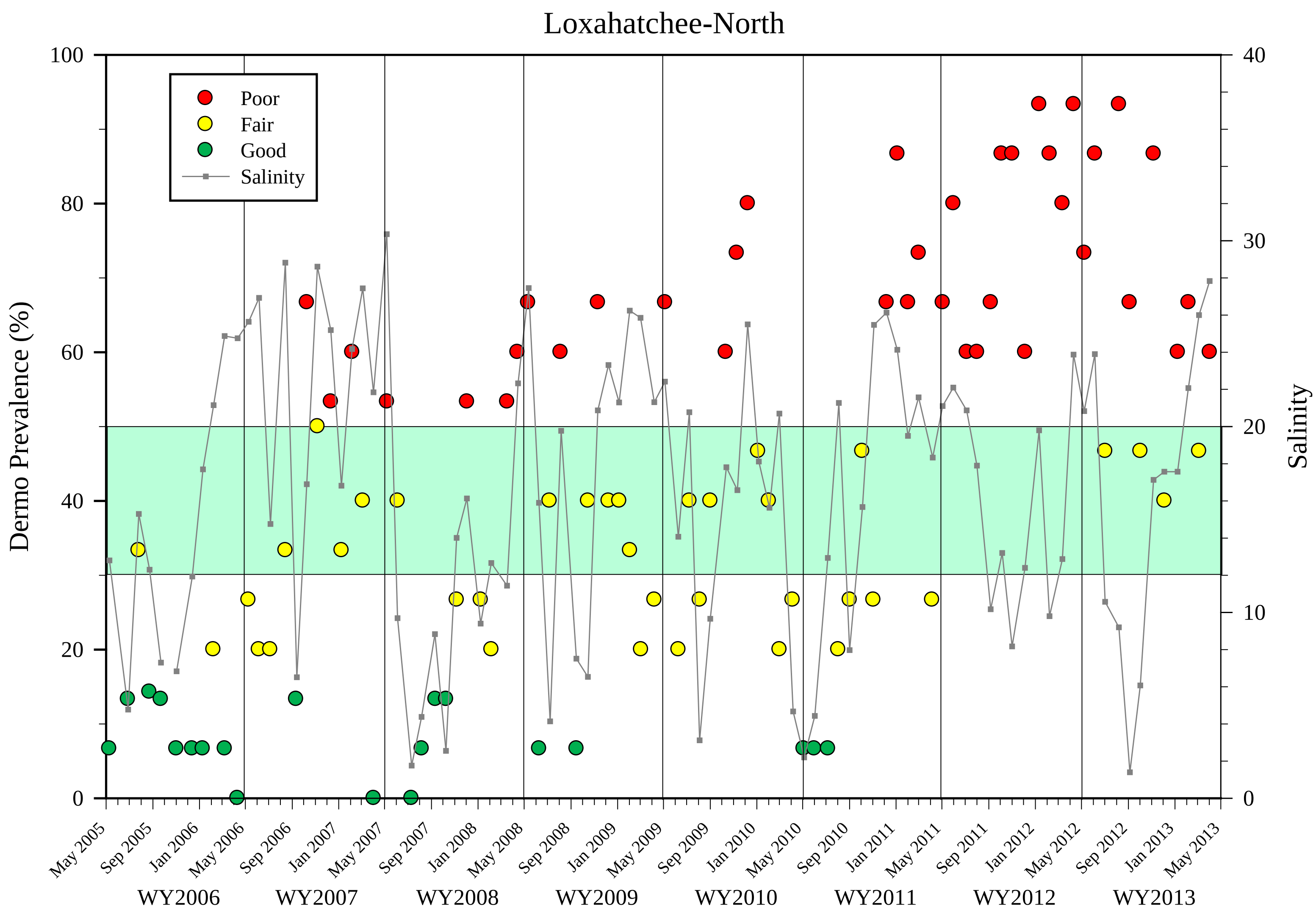

Figure 4-97. Monthly prevalence (%; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the Northwest Fork of LRE and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

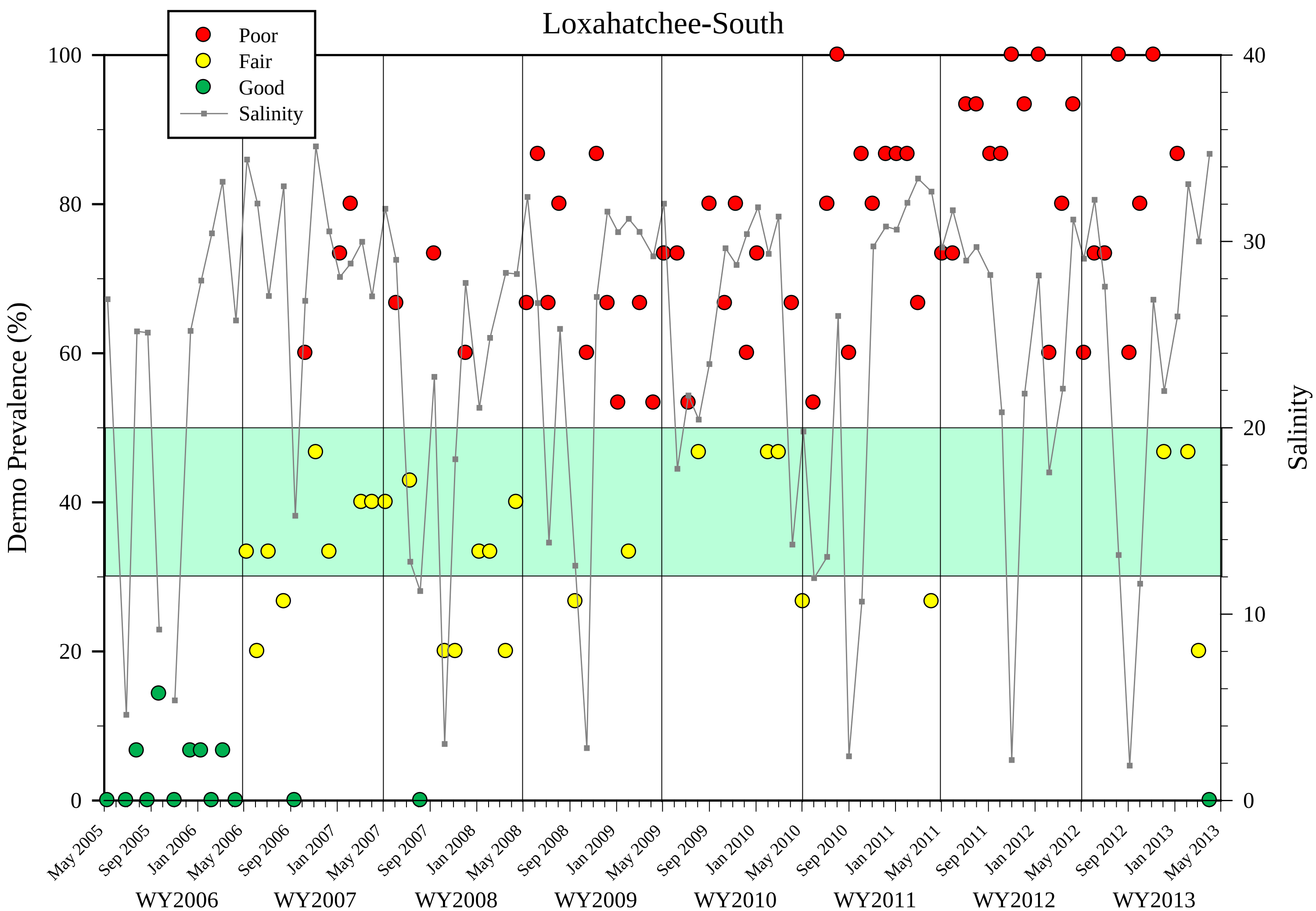

Figure 4-98. Monthly prevalence (%; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the Southwest Fork of LRE and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

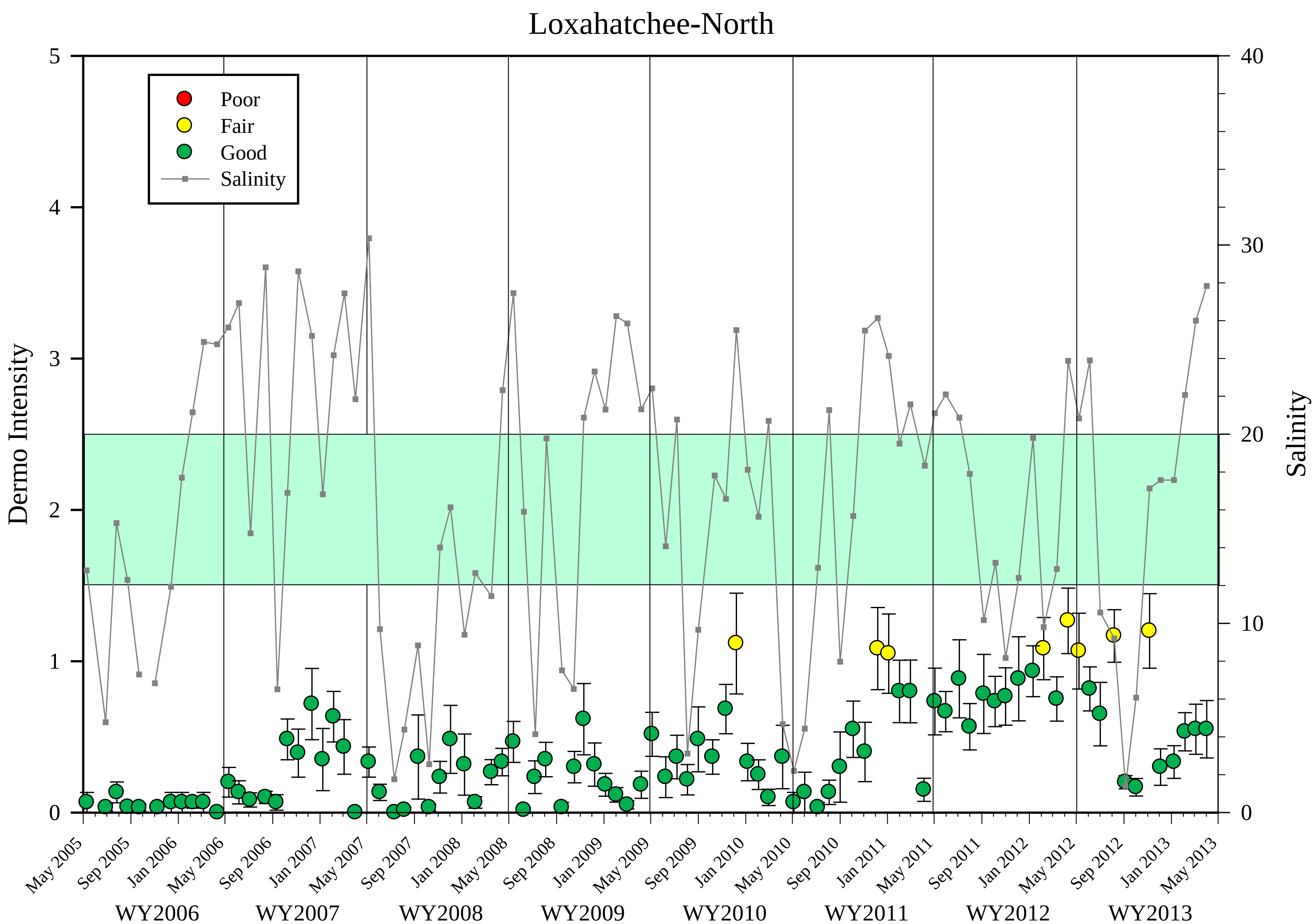

Figure 4-99. Monthly mean infection intensity (± standard deviation; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the Northwest Fork of the LRE and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

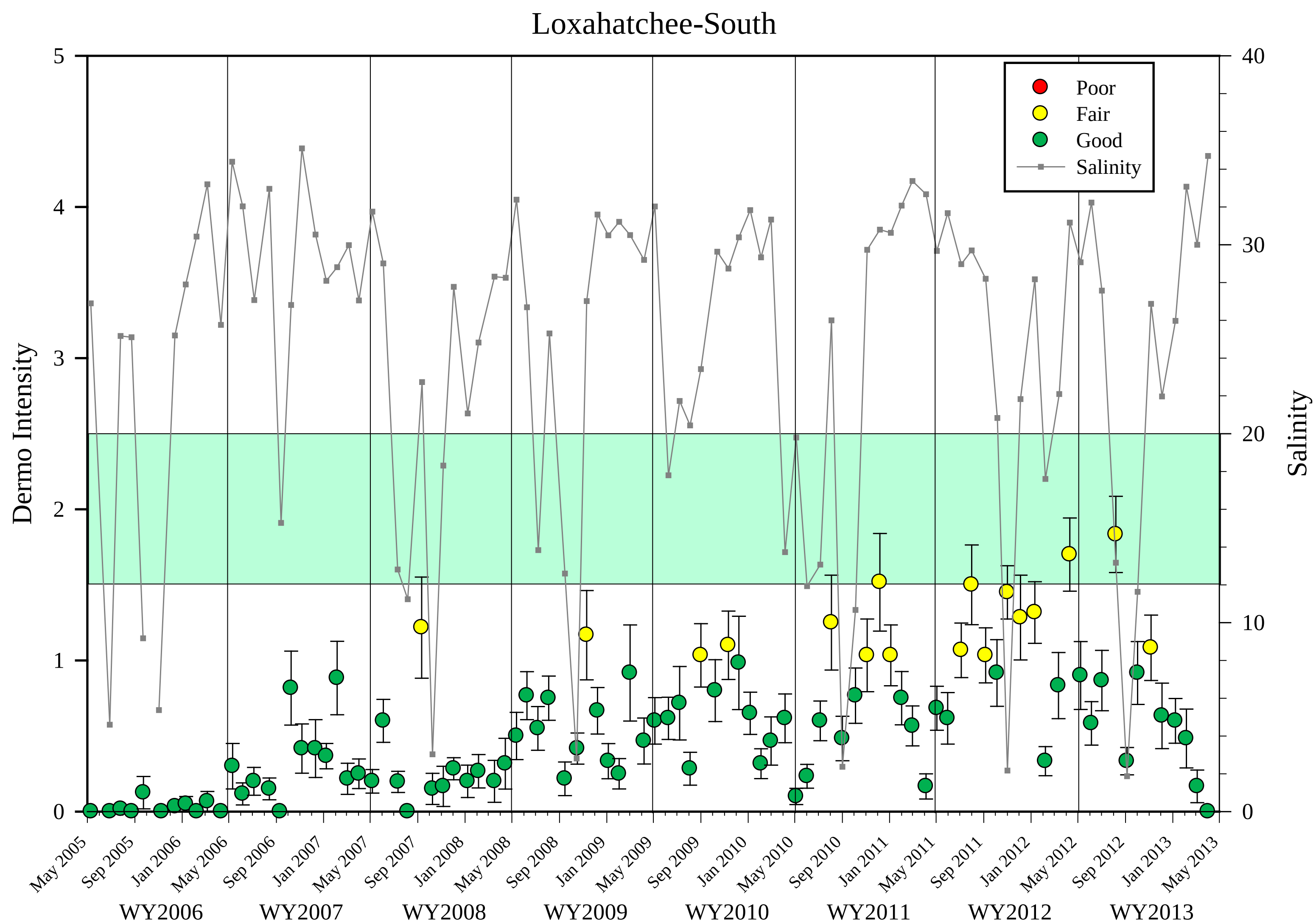

Figure 4-100. Monthly mean infection intensity (± standard deviation; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from the Southwest Fork of the LRE and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the estuary.

Growth rates of tagged oysters planted in the LRE varied in the Northwest and Southwest forks. Oysters of similar size (~30 mm shell height) planted in February 2011 grew at a rate of 1 to 1.5 mm per month over the next 12 months in the Northwest Fork, but at a rate of 2 to 3 mm per month in the Southwest Fork. In the following year, tagged oysters planted in the Northwest Fork exhibited an average growth rate of 3.7 mm per month by April 2013. Oysters planted in the Southwest Fork in 2012 did not fare as well and experienced substantial mortality during the first few months with less than 1% remaining alive at the 6-month mark. The differences in growth rates between years in the Northwest Fork are likely due to differences in salinity patterns between the two years. In 2011, salinities exceeded the upper limit of the optimal range during the first several months of growth, potentially leading to reduced growth rates. The opposite happened in 2012; salinities fell within or below the optimal range during the first seven months of growth and, as a result, growth rates were higher. In both years, mortality of oysters planted in the Northwest and Southwest forks was generally higher in open cages (~10–20%) than in closed cages suggesting that there was substantial macrofaunal predation. The only exception occurred in the Southwest Fork in 2012, when mortalities between open and closed cages were similar and, in fact, slightly higher in the closed cages by April 2013.

Conclusion

Oyster populations in the LRE have been negatively impacted by the highly variable freshwater inflows that are a result of the altered local hydrology. Periods of extremely high flow result in acute damage to oyster populations. Extended periods of reduced flow result in gradual increases in disease and predation rates that result in compromised oyster health and survivorship. The variability in and of itself can compound the problem because rapid shifts between dry and wet regimes reduce the opportunity for acclimatization by the oyster and other estuarine inhabitants. High salinities are a persistent problem in LRE where freshwater inflows have not been of sufficient magnitude or duration to provide relief from disease and predation pressure. Salinities commonly exceed the upper limit of the optimal salinity range (12–20) but brief and intense periods of freshwater inflow have resulted in salinities below 5 and negatively impacted oysters.

Several steps can be taken to strengthen understanding of the relationship between flow and salinity in the estuary. For example, estuarine conditions are commonly summarized by water year, but oysters are often impacted on shorter time scales. In many cases, an extreme low salinity event and a prolonged high salinity event occurred within the same water year; therefore, a more detailed examination will likely yield a better understanding of how changes in salinity impact oyster biology over varying time scales. The estuary also has multiple potential sources of freshwater input and these should be considered in sum when interpreting freshwater inflow impacts on water quality, oysters and other estuarine fauna. Finally, collection of frequent and continuous water quality data in multiple locations within the estuary would allow for development and/or refinement of minimum/maximum flow rates to better prevent extreme salinity changes that lead to conditions that are detrimental to oysters.

Loxahatchee River Estuary Seagrass

Patch-scale Seagrass Monitoring

LRD conducts bi-monthly monitoring of seagrass coverage at 5 sites throughout the estuary (**Figure 4-101**). Prior to 2013, LRD had minimal seagrass data from the region east of the railroad bridge despite knowledge that expansive seagrass beds existed there. In order to better understand seagrass characteristics along the entire salinity gradient, the Inlet site was added in 2013 to extend the seagrass monitoring into this region. The Hobe Sound site, located 8 kilometers north of the Jupiter inlet, was previously monitored bi-monthly as the "reference" seagrass bed and is now monitored on a semi-annual schedule.

Figure 4-101. Locations of bi-monthly seagrass monitoring sites in the Loxahatchee River.

Data from these sites show a clear upstream-downstream gradient of seagrass species composition and cover (**Figure 4-102**). The salinity, water clarity, and ground elevation flux (deposition, accretion) are likely the primary contributors to the varied seagrass species composition and coverage at each of the sites. *H. wrightii* and *H. johnsonii* are the only two species observed at the upstream stations, Northwest Fork and Pennock Point. *H. johnsonii* was greatly impacted at both sites following Tropical Storm Isaac in 2012 (**Figure 4-102**).

The Sandbar site includes primarily *H. wrightii* and *H. johnsonii* with sparse occurrences of *S. filiforme* and *T. testudinum* (**Figure 4-102**). Initially, until summer 2004, *S. filiforme* was observed at much higher coverage than currently observed. This coverage decrease was attributable largely to the storms of 2004 and has yet to return to pre-storm levels. This prolonged degradation, perhaps an alternate stable state, may be due to a number of factors, though changes in bathymetry may be exacerbating recovery. The seasonal pattern observed in coverage of *H. johnsonii* is most apparent at the Sand Bar site, which peaks each spring and then decreases in the summer and fall. These cycles in coverage may be influenced, at least in part, by the freshwater flows.

Slightly downstream, the North Bay site is composed primarily of *H. wrightii*, *H. johnsonii*, *S. filiforme* and *T. testudinum* (**Figure 4-102**). Prior to the 2004 storms, the North Bay site supported a small isolated bed of *H. engelmanni*, which disappeared as a result of the storms and has not been observed since. Manatee grass coverage at North Bay declined significantly following the 2004 storms. Over 8 years, manatee grass returned to 80 percent of pre-storm abundance. It appears that the increase in coverage of Johnson's grass is the result of this species moving into space left vacant by *S. filiforme*. Shoaling is the likely cause of overall decrease in coverage for seagrass at this site as there are now unvegetated portions of exposed sandbar that once were occupied by multiple species of seagrass including *S. filiforme*.

The Inlet site is composed primarily of two species, *H. wrightii* and *H. johnsonii*, with sparse occurrence of *T. testudinum* (**Figure 4-102**). Despite salinity at this site being most "marine-like", *T. testudinum* is very sparse and *S. filiforme* is non-existent; both species are marine-associated, late-stage successional species. The minimal presence of these species suggests that sediment flux due to the high energy nature of this site is occurring too rapidly for these two species to establish a dense bed. Thus, *H. wrightii* and *H. johnsonii*, both known to rapidly colonize disturbed areas, are dominant here.

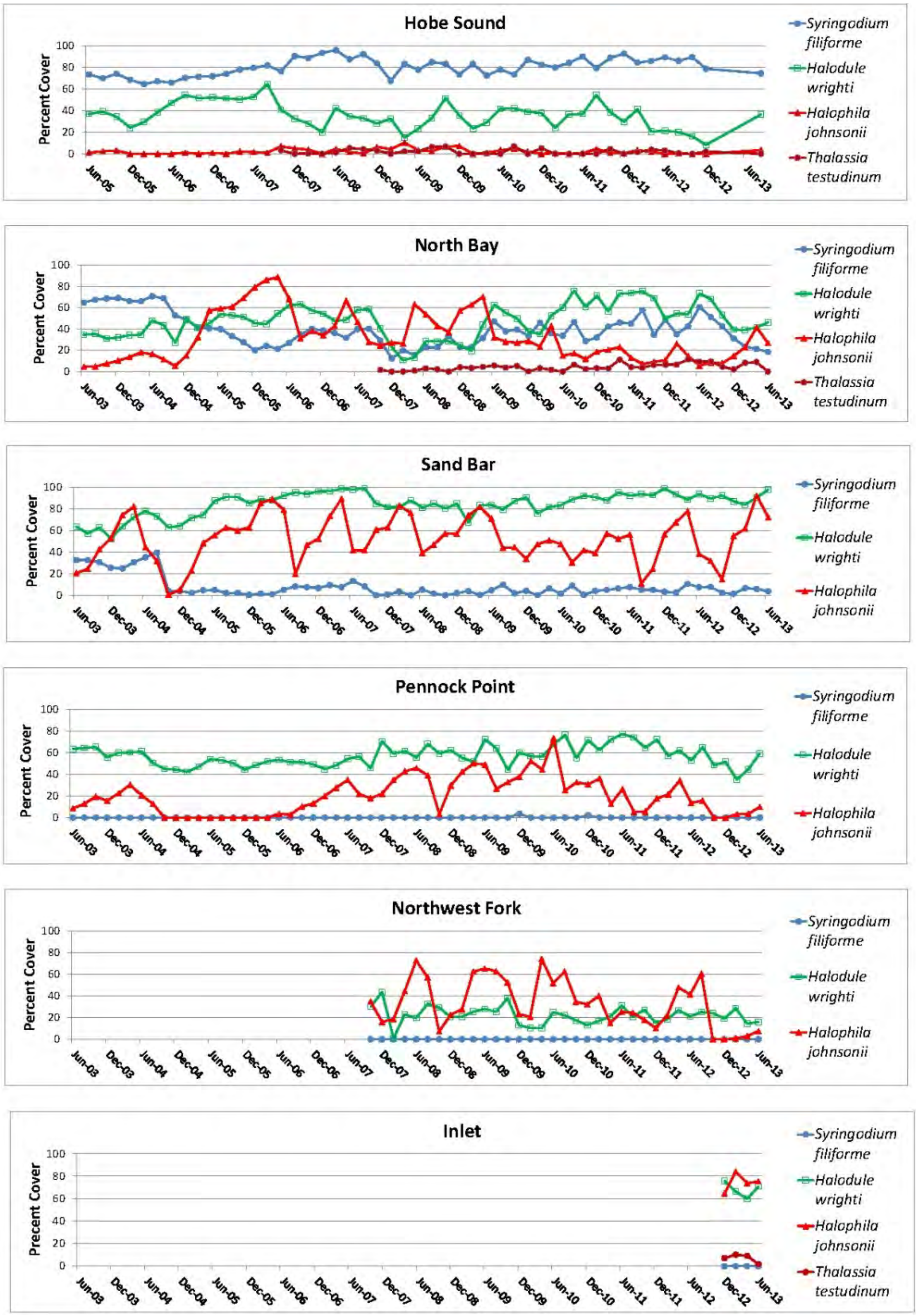

Figure 4-102. Seagrass percent cover by site and species from June 2003 to June 2013 in the Loxahatchee River.

Seagrass Mapping

LRD mapped seagrasses throughout the estuary in 2007 and 2010 using the 9-m^2 quadrant. **Figures 4-103** and **4-140** show the locations of sample points color coded by seagrass coverage for 2010 and 2007. Extensive seagrasses in the estuary indicate a generally functional system.

Seagrasses are most prevalent in the shallow waters of the central embayment, and mostly nearshore in the three forks of the river. The 2007 survey indicated a greater prevalence of *H. decipiens*, a species often associated with low light and higher salinity, in the southwestern portion of the central embayment compared to 2010. Species-specific maps are available on the LRD website: www.loxahatcheeriver.org/reports.php. Salinity, water depth (light attenuation), substrate composition and stability (sand, muck, etc.) are primary factors influencing the distribution of seagrass in the estuary. Upstream, seagrasses appear to be largely limited by water depth/clarity and substrate type as seagrass is mainly confined to the subtidal littoral zone very near shore. Soft, organic rich (muck) sediments are common just off shore in much of the upper estuary and appear to limit seagrass establishment.

Figure 4-103. 2010 seagrass survey sampling points color-coded by number of 1-m^2 cells of 9-m^2 quadrat occupied by seagrass on the LRE.

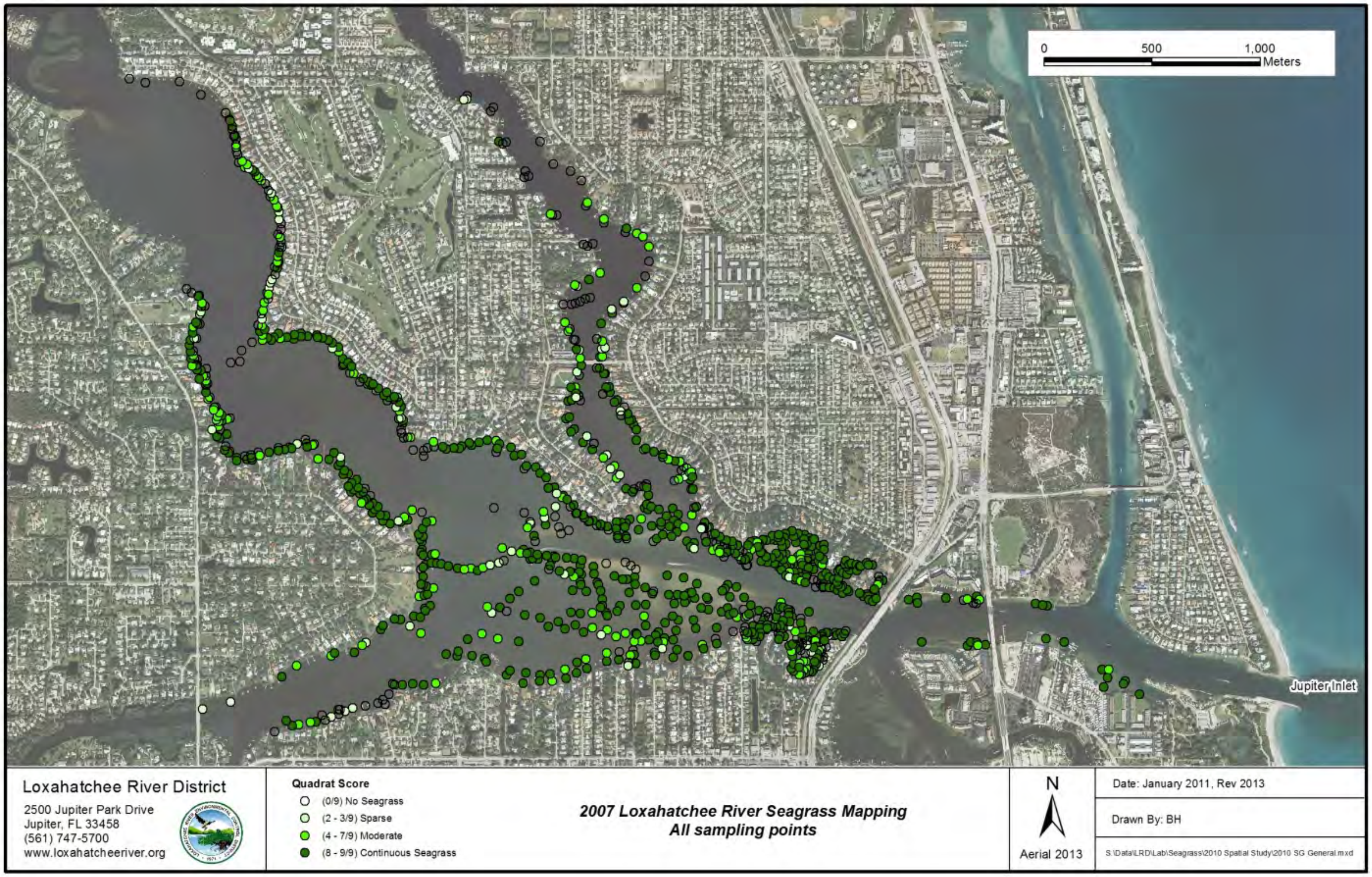

Figure 4-104. 2007 seagrass survey sampling points color-coded by number of 1-m^2 cells of 9-m^2 quadrat occupied by seagrass on the LRE.

In the middle estuary, water clarity is improved with marine waters reaching up to Pennock Point during high tide **Figure 4-105**. Seagrasses in this area extend from the intertidal zone occupied by *H. wrightii* and *H. johnsonii* seagrass to water depths of over 1.5 meters. *S. filiforme*, *T. testudinum*, and *H. decipiens* have all been observed in the central embayment regions just west of the railroad bridge. Sediments in the middle estuary are typically fine sand and there is clear evidence of shoaling that is reducing water depths thus affecting seagrass distribution and composition. Additionally, current velocities in some portions of the lower estuary are high and the coarse sediment substrate is dynamic with extensive tidal-driven sediment transport. In these regions, seagrasses are largely confined to areas protected from the high current velocities. *H. wrightii* and *H. johnsonii* are prevalent in the lower estuary adjacent to where these conditions persist.

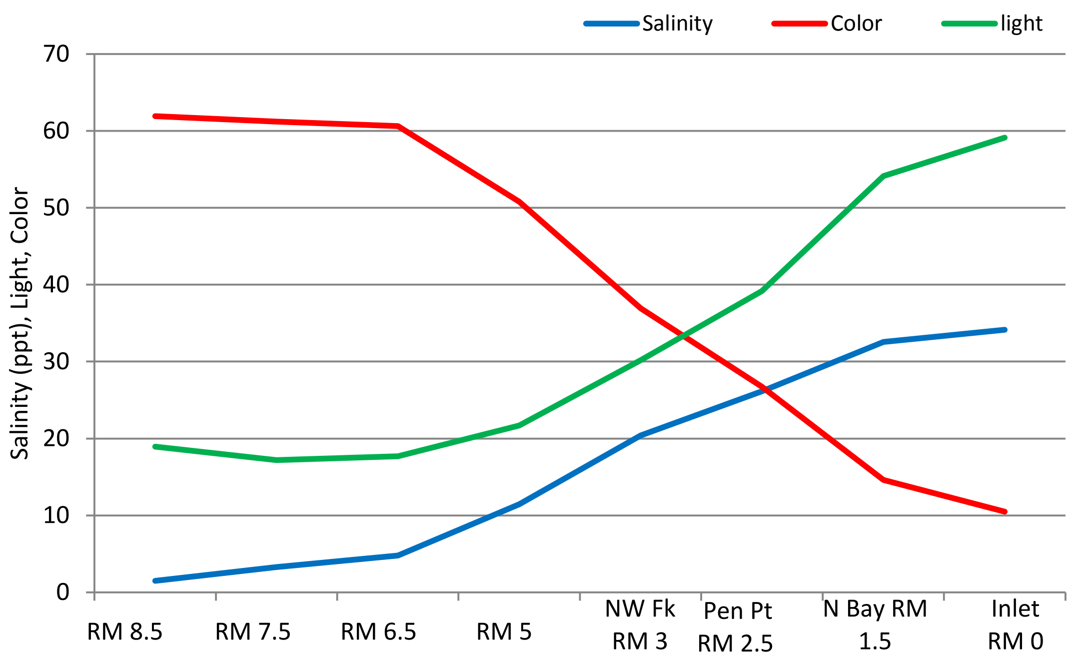

Figure 4-105. Graph showing how salinity, color, and light penetration are related along the salinity gradient. Data are moving from upstream to downstream as graph goes from left to right. Labels show seagrass monitoring sites adjacent to water quality measurements.
(Note: NW Fk – Northwest Fork, Pen Pt – Pennock Point, and N Bay – North Bay.)

<u>*Vallisneria americana*</u>

Vallisneria americana (tape grass) has dramatically increased in distribution, primarily throughout the Wild & Scenic segments of the Northwest Fork (i.e., river miles 15 to 8). LRD and SFWMD staff mapped less than 1 acre of *V. americana* in 2010, when it was first observed, and then approximately 13 acres in 2013 (**Figure 4-106**). Literature and personal communication suggests that salinities in this segment of the river have historically been below lethal thresholds for *V. americana*, so factors other than salinity must be driving this expansion in coverage. Regardless, the *V. americana* is flourishing (even flowering), which is a very encouraging observation for the Loxahatchee River (**Figure 4-107**).

Figure 4-106. Comparison of *V. americana* distribution between 2010 and 2013 as mapped by LRD and SFWMD staff in the Northwest Fork of the Loxahatchee River.

Figure 4-107. Photos of *V. americana* in the Loxahatchee River in 2013.

Loxahatchee River Estuary Water Quality

Staff from the LRD's WildPine Ecological Laboratory collects and analyzes surface water samples for 29 parameters at 39 sites located in the Loxahatchee River, its major tributaries, and associated waters (**Figure 4-108**). Most sites are sampled bi-monthly (round symbols) with a subset of between 10 to 15

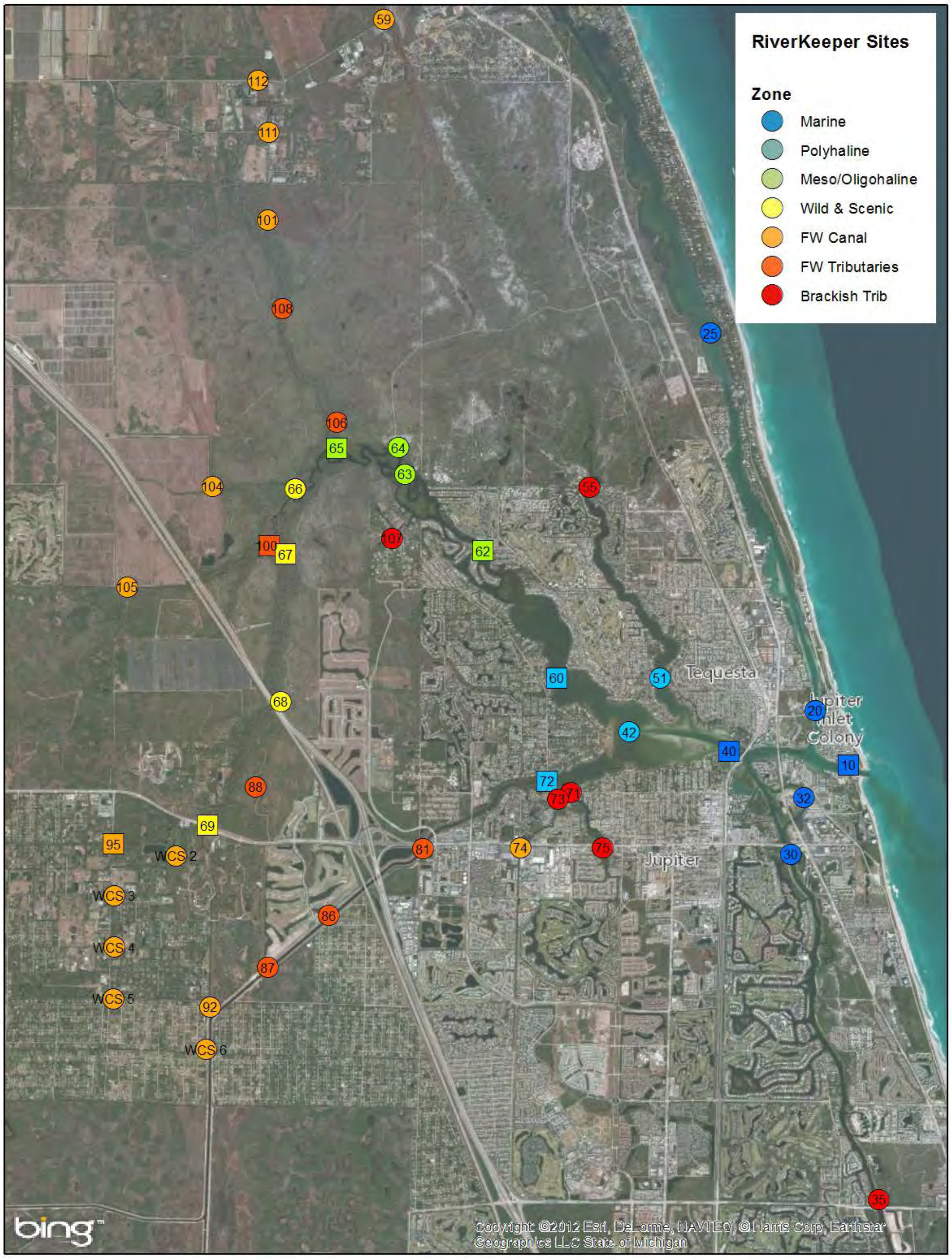

Figure 4-108. RiverKeeper water quality monitoring sites on the Loxahatchee River Estuary. Sites with square symbols are sampled monthly, while round symbols are sampled bi-monthly.

sites (square symbols) sampled every month. This water quality monitoring program, entitled "RiverKeeper", was developed to identify long-term trends, and to assess compliance with water quality targets established in the *Restoration Plan for the Northwest Fork of the Loxahatchee River* (SFWMD 2006, SFWMD et al. 2012). Most recently, the United States Environmental Protection Agency (USEPA) and FDEP have established numerical nutrient criteria water quality targets and serve as the new benchmark. Ongoing results from the water quality monitoring program are used to establish baseline conditions prior to modification of freshwater inflows resulting from the CERP and the *Restoration Plan for the Northwest Fork of the Loxahatchee River* (SFWMD 2006, SFWMD et al. 2012).

The LRD has been providing a simplified characterization and overview of the water quality conditions in the Loxahatchee River watershed using a "stoplight" assessment. Annual water quality results (geometric mean) for TN, TP and chlorophyll *a* are scored to the recently established USEPA/FDEP numeric nutrient criteria (NNC). Fecal coliform bacteria results are scored relative to FDEP's water quality criteria for recreational waters.

Figure 4-109 shows recent TN results for nearly all sites meet the NNC with no more than 2 exceedances in a 3-year period. Site 59, a freshwater ditch in the northern part of the watershed off Bridge Road in Hobe Sound, is the only site not meeting the criteria. Site 101, Hobe Grove Ditch in the northwestern part of Jonathan Dickinson State Park, has also exhibited high TN results.

Figure 4-110 shows that five sites in the Loxahatchee River watershed do not meet the NNC for TP. Sites 55 and 60 are in the polyhaline sections of the North Fork and Northwest Fork and the NNC limit is exceedingly low at 0.03 mg/L. Site 107, a tributary into the Northwest Fork near the River's Edge neighborhood has shown persistent TP issues. In 2012, LRD conducted reconnaissance analysis of key water quality parameters (including phosphorus, fecal coliform bacteria and sucralose) to assess the relative contribution of anthropogenic versus wildlife contributions of nutrients and bacteria. The presence of sucralose at site 107 indicates pollutants from septic systems are making it into surface waters at this site; nonetheless, wildlife (e.g, raccoons and wading birds) also may be contributing to elevated fecal bacteria and TPs concentrations. Station 88, the former agriculture site just west of I-95/Turnpike and north of SR 706/Indiantown Road, also indicates persistently high TP results. Pending site development plans include remediation of legacy pollutants. It will be important to ensure that this site is adequately addressed during redevelopment. Site 75, a brackish tributary named Jones Creek, is an extensive urban drainage basin. Future improvements to increase flushing, adequately treat stormwater inputs, and reduce bacteria counts (e.g., dog waste) would be beneficial.

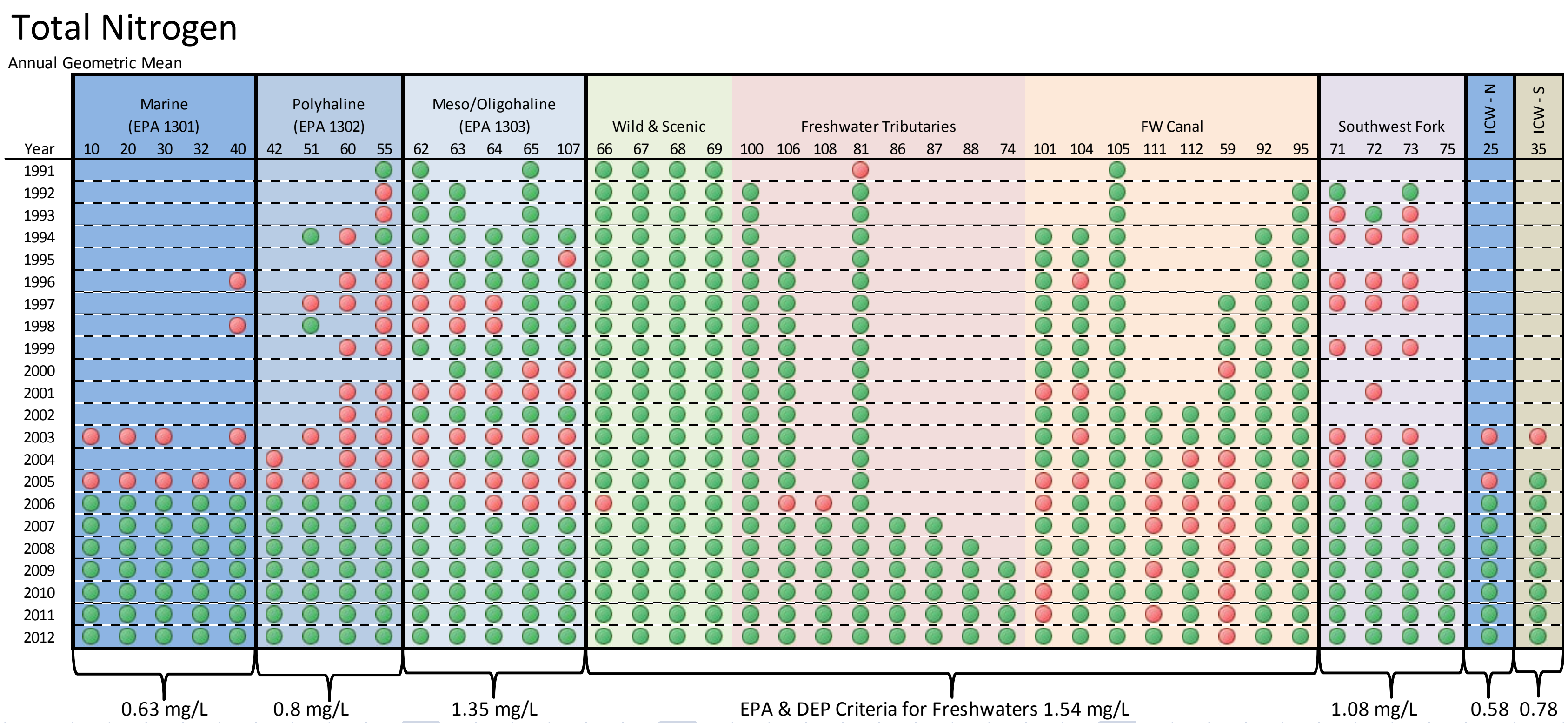

Figure 4-109. Stoplight plot of annual geometric mean TN by site and year scored relative to FDEP (DEP)/USEPA (EPA) NNC values as indicated.

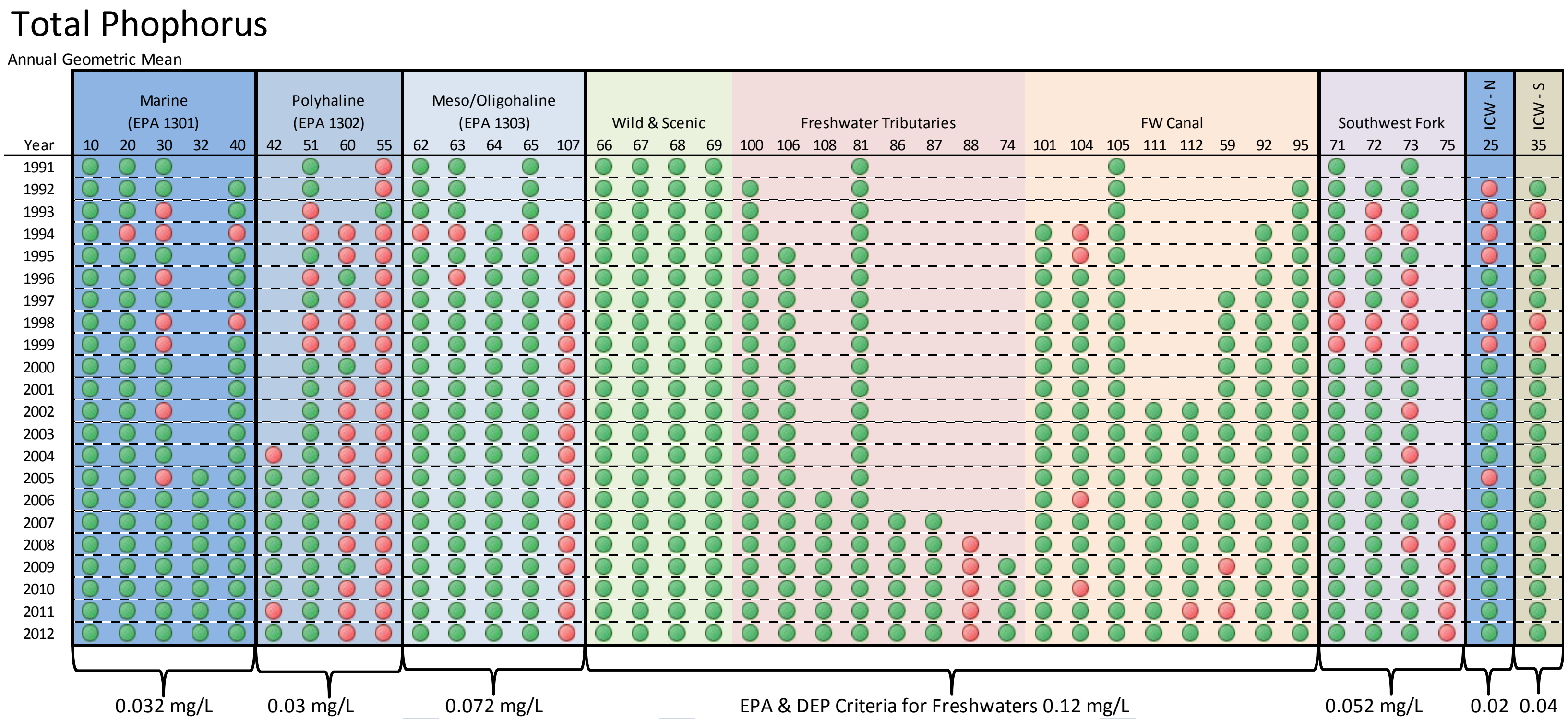

Figure 4-110. Stoplight of annual geometric mean TP by site and year scored relative to FDEP (DEP)/USEPA (EPA) NNC as indicated.

The NNC for cholophyll *a* (**Figure 4-111**) is very restrictive in the marine and polyhaline segments of the Loxahatchee River with limits of 1.8 and 4 microgram per liter (µg/L). As such, the sampling sites in these areas do not score well relative to the NNC. Based on water clarity, relatively low nutrient levels at many sites, and the extent of seagrasses, these criteria may be impractical and unnecessary. Nonetheless, there are sites that are of genuine concern. For example, annual chlorophyll *a* values at site 88, the former agriculture site discussed above, regularly exceeds the NNC of 20 ug/L. Site 72, at the Loxahatchee River Road Bridge over the Southwest Fork, also exhibits high chlorophyll readings, particularly during the spring and summer months when temperatures rise and there is little flow in the Southwest Fork. Further, meaningful amounts of nutrients enter the area from Jones and Sims creeks. Again, additional flushing and stormwater treatment would likely provide benefits to these areas.

Annual fecal coliform bacteria counts (**Figure 4-112**) exceed FDEP's water quality standards for three sites in the watershed. Site 107, the tributary in the Northwest Fork, has shown chronic issues with very high fecal bacteria and TP concentrations as previously discussed. Sites 73 and 75, located in the brackish tributaries that serve as extensive urban drainage areas and flow into the Southwest Fork, also show frequent problems with fecal bacteria counts. USEPA and FDEP have established a total maximum daily load for fecal coliform bacteria in this water body. Improvements will be required as part of future basin management action plans.

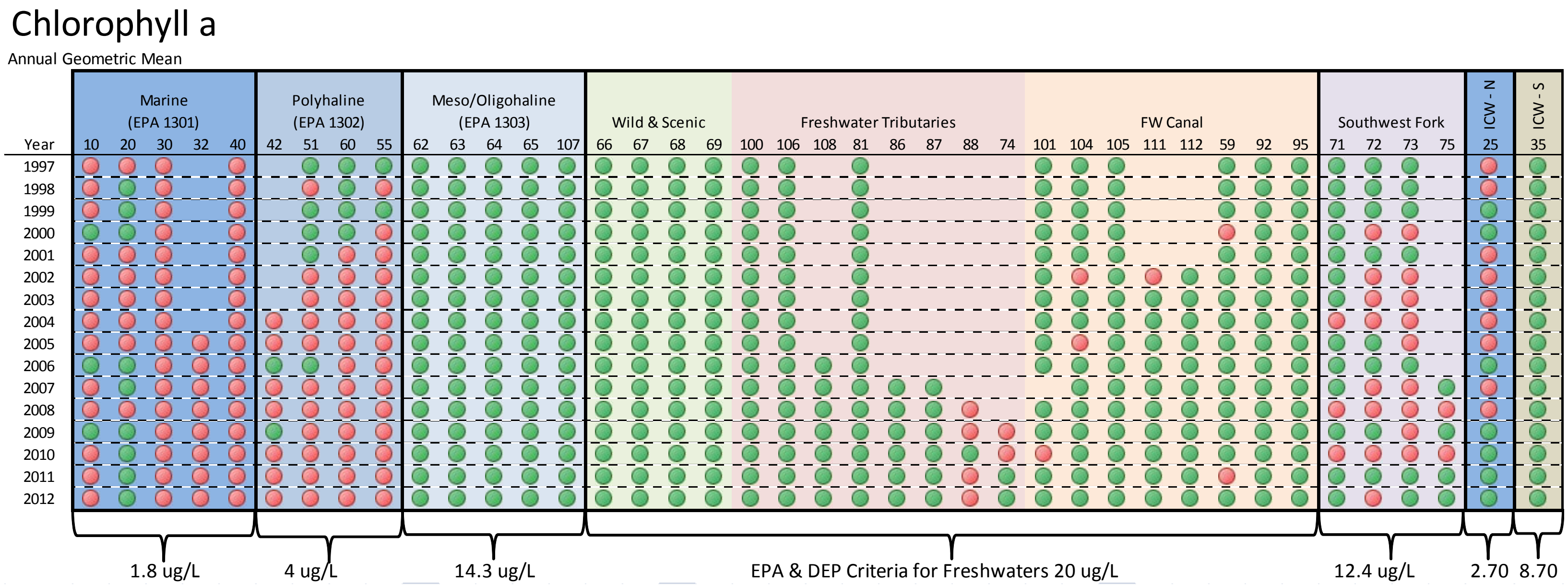

Figure 4-111. Stoplight plot of annual geometric mean chlorophyll *a* by site and year scored relative to FDEP (DEP)/USEPA (EPA) NNC as indicated.

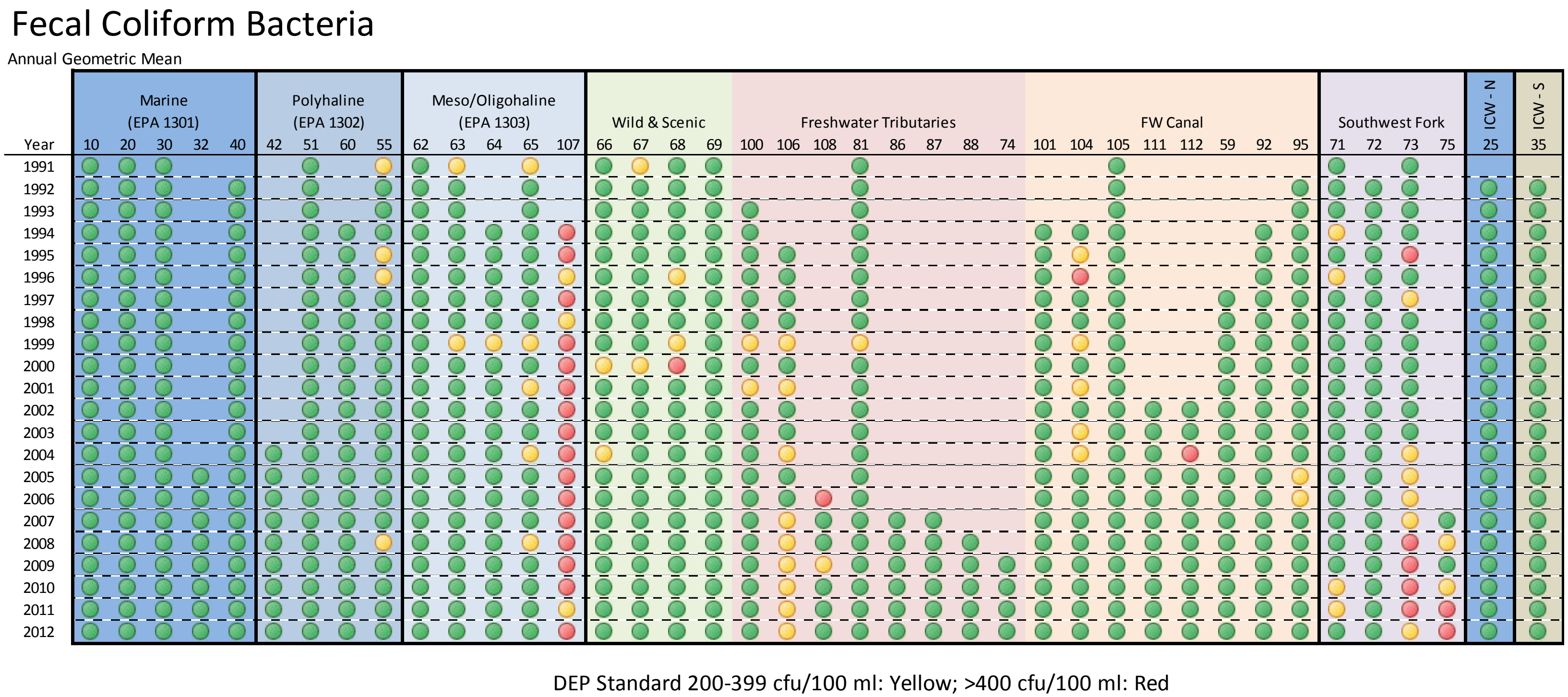

Figure 4-112. Stoplight plot of annual geometric mean fecal coliform bacteria by site and year scored relative to FDEP (DEP) Standard. 200–399 cfs/100 milliliter (ml) – Yellow and > 400 cfs/100 ml – Red.

Significant Findings for the Loxahatchee River Estuary

- Following supplemental flows during spring 2011, 2012, and 2013 to help meet the MFL targets, extensive salinity monitoring indicates the salinity targets in the cypress habitat were met.
- Analysis of salinity data suggests that moderate flood control releases (i.e., < 300 cfs) into the LRE through the S-46 structure can appreciably reduce daily salinity variability, which likely reduces the stress and/or harm experienced by seagrasses and oyster reefs.
- In 2010, Martin County and the LRD, with funding from the American Recovery and Reinvestment Act through NOAA, coordinated the successful restoration of over 5.8 acres of oyster reefs in the Northwest Fork of the Loxahatchee River. Just 20 months after the 5.8-acre oyster restoration project, the reef supported almost 5,000 pounds of nonoyster animal biomass (i.e., small fish, crabs and shrimp) at the restoration site.
- Tropical Storm Isaac in 2012 resulted in more than 10 inches of rain over 3 days in portions of the watershed, and over 14 days of heavy flood control releases (700–2,000+ cfs) from the S-46 structure into the LRE. Oysters in the Southwest Fork were heavily impacted but elsewhere in the estuary oysters were not significantly harmed. Less than 12 months later, it appears the oysters in the Southwest Fork have recovered. Tropical Storm Isaac caused significant and sustained decline in *S. filiforme* at the North Bay and Sand Bar sites. *H. johnsonii* and *H. wrightii* were severely impacted at the Northwest Fork site and have yet to recover. *H. johnsonii* has declined at Pennock Point, but appears generally healthy at the other sites.

LAKE WORTH LAGOON

The LWL's 480-square mile watershed discharges fresh water into the estuary from three major regional canals: 55% of the inflows come from West Palm Beach Canal (C-51, S44), 25% from Boynton Beach Canal (C-16, S41), 8% from the Earman River Canal (C-17, S44), and ~12% from ungauaged stormwater runoff. The freshwater inflows reduce salinity levels, carry large influxes of nutrients, suspended and dissolved organic sediments, and contaminants that can be harmful to oysters, seagrasses, and many aquatic organisms. Between WY 2001 and WY 2012, annual average freshwater inflows to the lagoon ranged from 200 to 1,000 cfs with an overall average of 600 cfs. Water quality analysis from WY 2008 to WY 2012 showed strong interannual variations. In 2010, the estuary received high freshwater inflows resulting in low salinity levels, high TN, and high TP concentrations (PBCDERM et al. 2013). High flow events, like Hurricane Isaac and the 2013 above normal wet season, reduced salinity levels below 10 resulting in record low eastern oyster recruitment rates of 0.54 spat per shell. Seagrasses in the LWL decreased to a record low cover after Hurricane Isaac as a result of the increased freshwater inflows and subsequent increase of fine grained suspended and dissolved organic sediments. These sediments settle out as muck and reduce the light available to seagrasses, a critical factor for seagrass growth and reproduction. Additional water quality information can be found in Chapter 2 of the *2013 Lake Worth Lagoon Management Plan* (PBCDERM et al. 2013; http://www.pbcgov.org/erm/lwli/pdfs/2013ManagementPlan/2013LWLmanagementplanFINAL.pdf).

Lake Worth Lagoon Oysters

Oysters are monitored in LWL by the FWC at locations shown in **Figure 4-32**. Freshwater inflows into the estuary have been altered from a natural state to one in which inflows are more variable and extreme thus adversely impacting local oyster populations. The estuary has experienced impacts as a result of the altered salinity regime: exposure to high freshwater inflows during the wet season, which leads to a rapid decline in oyster health and abundance, and too little freshwater inflow during the dry season or drought periods, which leads to a gradual increase in predation, disease, and mortality. In LWL, salinities typically exceed the optimal salinity range and only fall within the optimal range intermittently during the wet season.

The goals of the salinity performance measure for LWL list a minimum salinity target of 15 for the central region based on the habitat requirements of eastern oysters during all life stages. This is more restrictive than the range of optimal salinities, 12–20, listed for the SLE. While most of the flow into LWL comes from the C-51 Canal, both the C-15 and C-16 canals contribute, especially during times of high flow. Based on minimal monthly data, salinity rarely falls below 10, even with high summed flows of 2,000 cfs or greater (~1,000 cfs from C-51 Canal). When 7-day averages of the summed flows are less than 500 cfs (~200 cfs from C-51 Canal), salinity is typically above optimal (20) at the Florida Wildlife Research Institute station located most closely to the C-51 Canal. Optimal salinities probably occur when summed flows are between 500 and 1,500 cfs, or when C-51 Canal flows are between 250 and 1,000 cfs. Those flow rates are higher than those suggested in the 2011 draft revision of the salinity performance measure.

Based on an analysis by water year, mean salinities in LWL always exceed 15 as well as the upper threshold of the optimal salinity range (**Figure 4-113**). Monthly data reveal that salinity in the central region of LWL rarely drops below 15, with a period of two consecutive months being the longest and most recent occurrence (September 2012–October 2012). LWL salinities typically exceed, and only rarely fall into or below the optimal range (12–20). In an effort to relate the biological responses of oysters to changes in salinity, each measured biological parameter has been categorized into thresholds representing good (green), fair (yellow), or poor (red) for oysters in LWL (**Table 4-12**).

Table 4-12. Thresholds for measured biological parameters of oysters in Lake Worth Lagoon.

Parameter	Good (Red)	Fair (Yellow)	Poor (Green)
Density	0–100 oysters/m^2	101–500 oysters/m^2	> 500 oysters/m^2
Recruitment Rate (May–October)	0–1 spat per shell per month	> 1–5 spat per shell per month	> 5 spat per shell per month
Dermo Infection Prevalence	> 50%	21–50%	0–20%
Dermo Infection Intensity	Heavy	Moderate	Uninfected to Light

The density of live oysters in LWL rarely meets the good (green) threshold and most often falls into the fair category (**Table 4-12** and **Figure 4-114**). On three occasions, live densities in LWL reached the good threshold: during fall 2008 and during the fall and spring surveys of WY 2013. Two of those three surveys coincided with periods when salinity decreased and approached the optimal range due to higher than average flows (1,377 cfs from August 2008–October 2008 and 1,375 cfs from June 2012–November 2012) through all three canals. As with the LRE, low salinity events do not appear to be a major limiting factor for oysters in LWL.

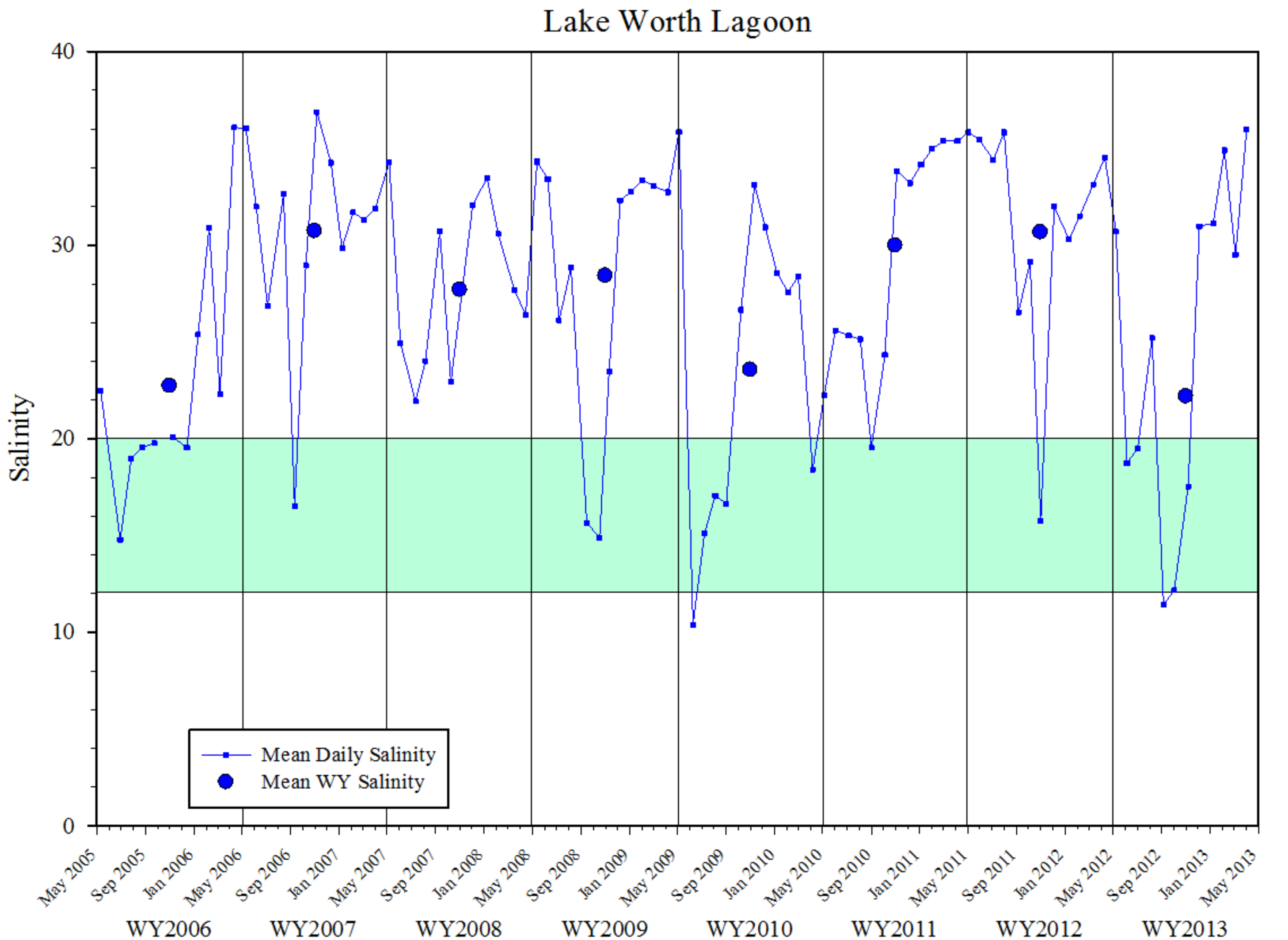

Figure 4-113. Mean monthly salinity and mean water year salinity from LWL. The green band represents the salinity range deemed most favorable for survival and health of juvenile marine fish, oysters, and SAV.

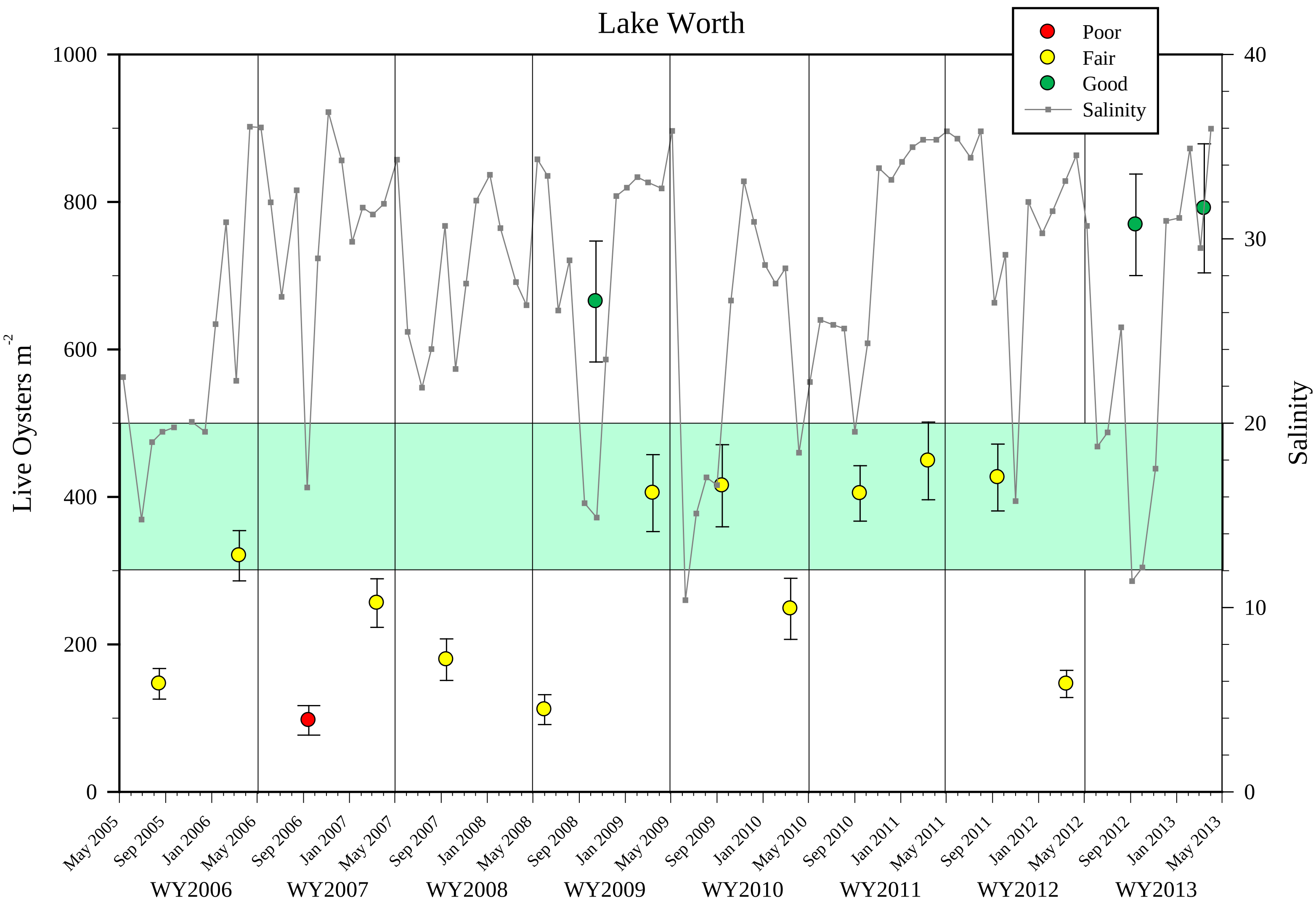

Figure 4-114. Mean number (± standard deviation) of live oysters (red, yellow, and green circles) in LWL during semi-annual surveys and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the lagoon.

Although reproductively developing oysters are typically present in fewer months than in observed in LRE oysters, the percentage of developing oysters during those months is high and similar to LRE oysters (**Figure 4-115**). Recruitment rates in LWL are moderate to high and are commonly classified in the good category (**Table 4-12** and **Figure 4-116**). Among oysters in the three east coast estuaries, LWL oysters have the highest and most consistent recruitment peaks, which typically occur between July and October of each year. The highest recruitment rates to date in LWL were measured during the summer months of WY 2013. Those high recruitment rates coincided with a several month period when salinities decreased and fell within the optimal range.

The impacts from dermo infection on oysters in LWL are similar to those seen in oysters from the LRE. During the first several years of the study (WY 2006–WY 2010), infection prevalence fluctuated between good, fair and poor (**Table 4-12** and **Figure 4-117**) while infection intensity remained within acceptable limits (**Table 4-12** and **Figure 4-118**). There is evidence that lower salinities decreased disease incidence. Specifically, in summer 2009, salinities fell into the optimal range (12–20) and, as a result, dermo prevalence decreased to caution and occasionally optimal levels. However, during the past three years (WY 2011–WY 2013), infection prevalence has increased and remained at failure levels as a result of the higher average salinities present in LWL. This indicates that, like the LRE, freshwater inflows into LWL over the past several years have not been of sufficient magnitude or duration to provide relief from disease pressure.

Growth rates of tagged oysters planted in LWL were much lower in 2011 than in 2012. Oysters planted in 2011 grew at a rate of 1 mm per month, while oysters planted in 2012 grew at a rate of more than 3 mm per month. Mortality rates also differed between years. In 2011, just two months after planting, 100% of the oysters planted in open cages were dead while 56% remained alive in the closed cages. Mortality was also higher in open cages in 2012, but the differential between open and closed cages was smaller than in 2011. Those differences in growth and mortality rates are likely due to differences in salinity patterns between the two years. In 2011, salinities typically exceeded the upper limit of the optimal range producing slower growth rates and increased predation pressure. In 2012, salinities fell within the optimal range for several months and as a result, growth rates were higher and predation pressure was reduced.

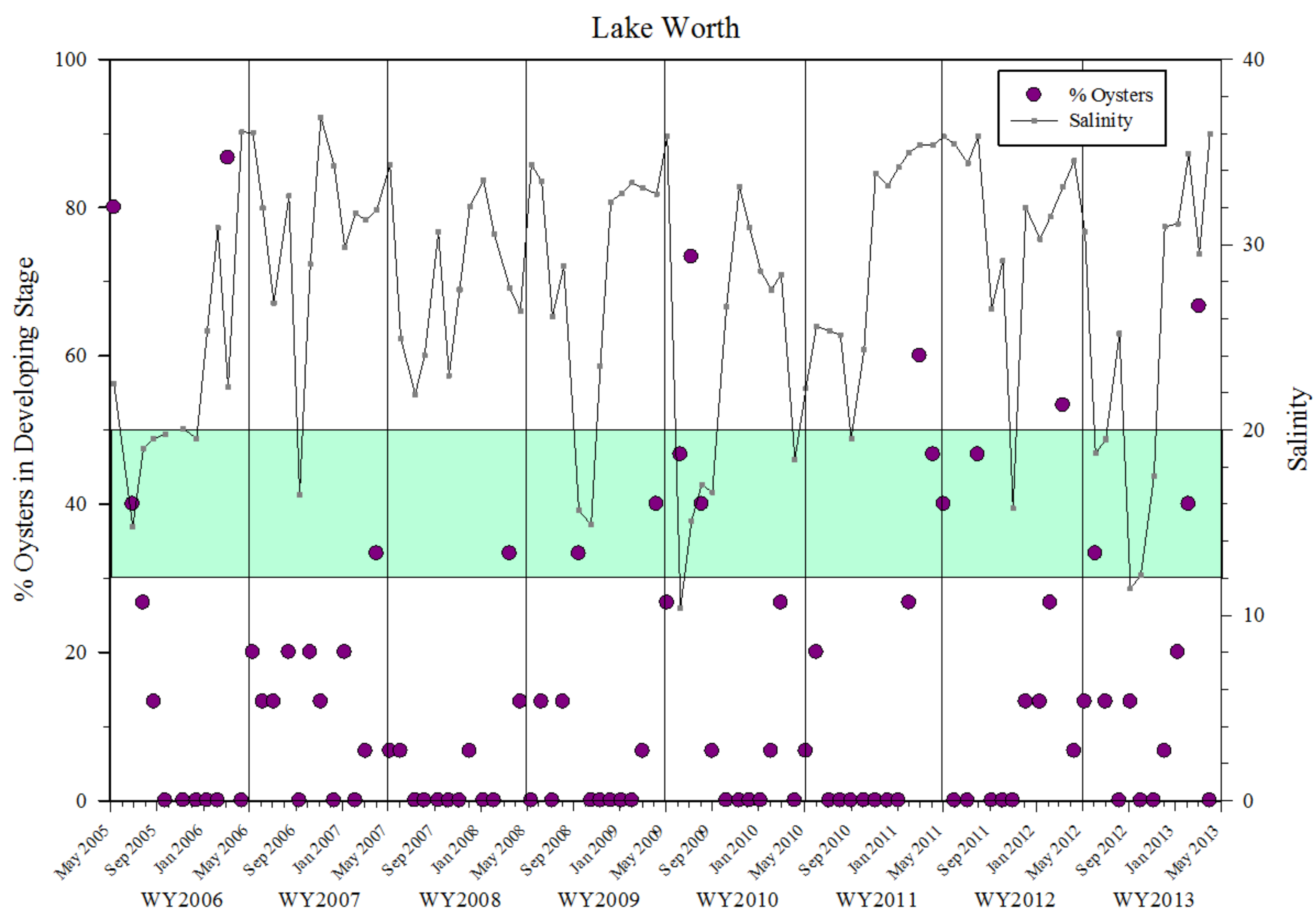

Figure 4-115. Reproductive development, represented as the percentage of oysters in the gonadal development stage, of oysters collected from LWL each month and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the lagoon.

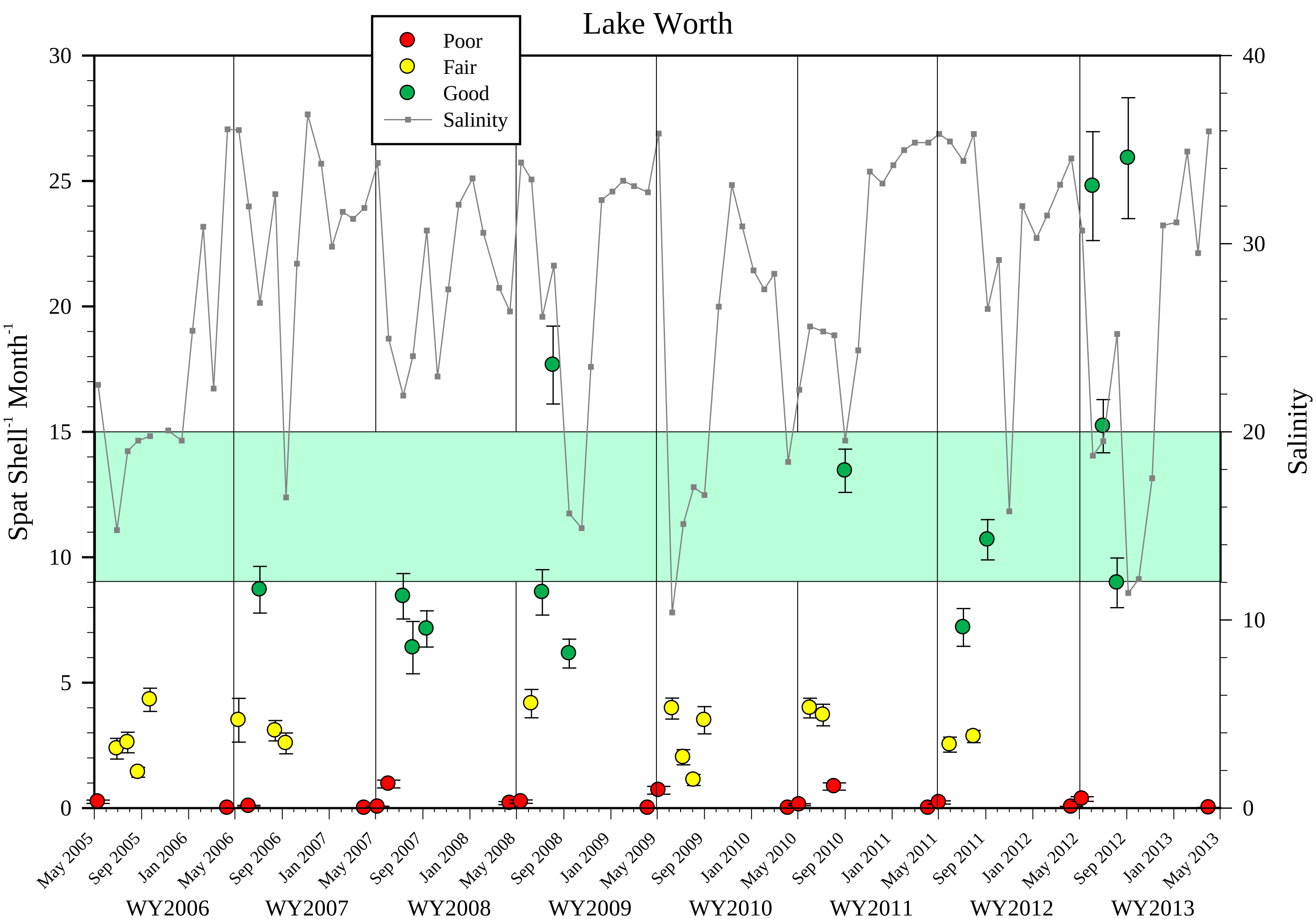

Figure 4-116. Mean number (± standard deviation) of oyster recruits per shell (red, yellow, and green circles) during monthly collections from April through September of each calendar year in LWL and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the lagoon.

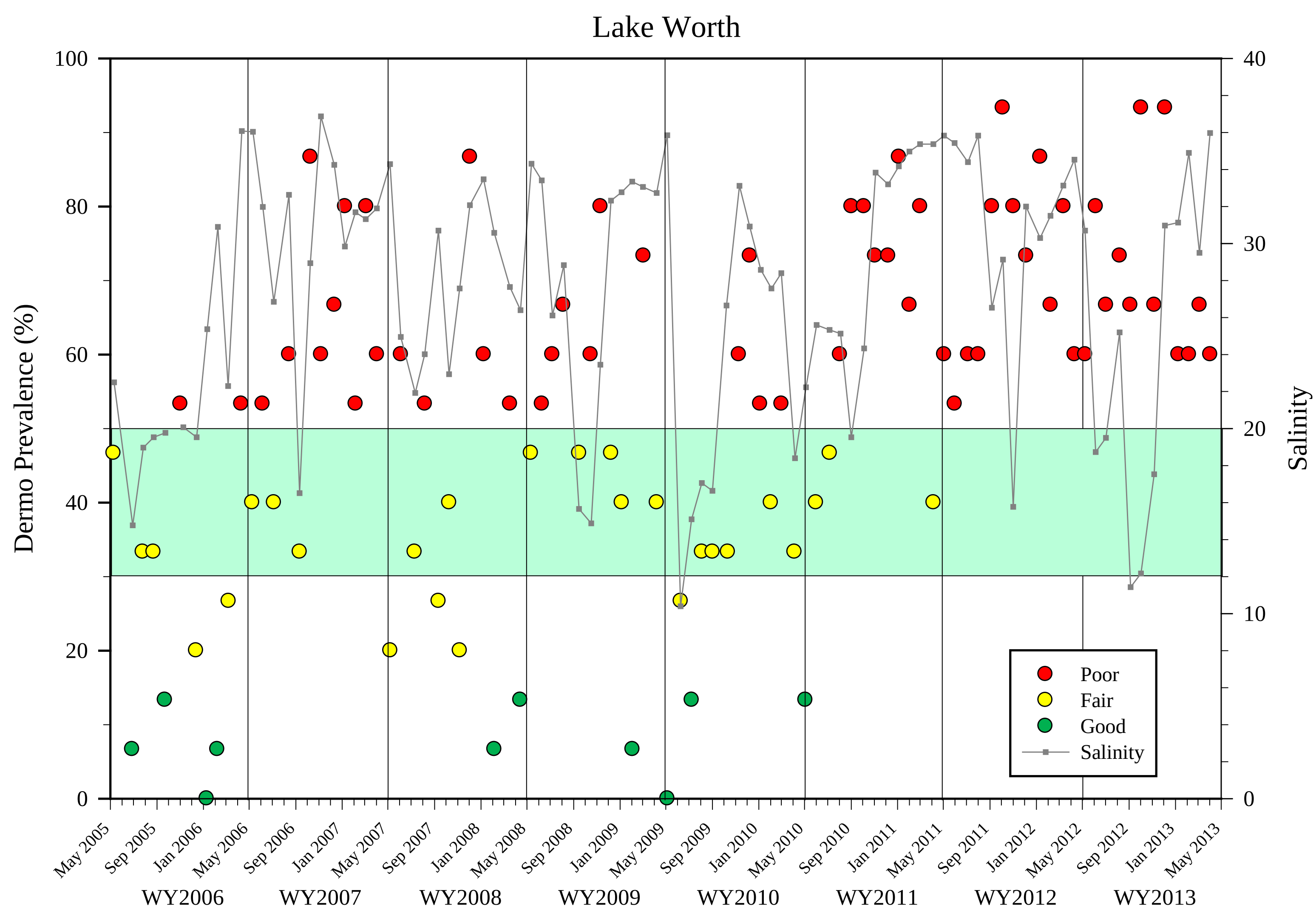

Figure 4-117. Monthly prevalence (%; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from LWL and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the lagoon.

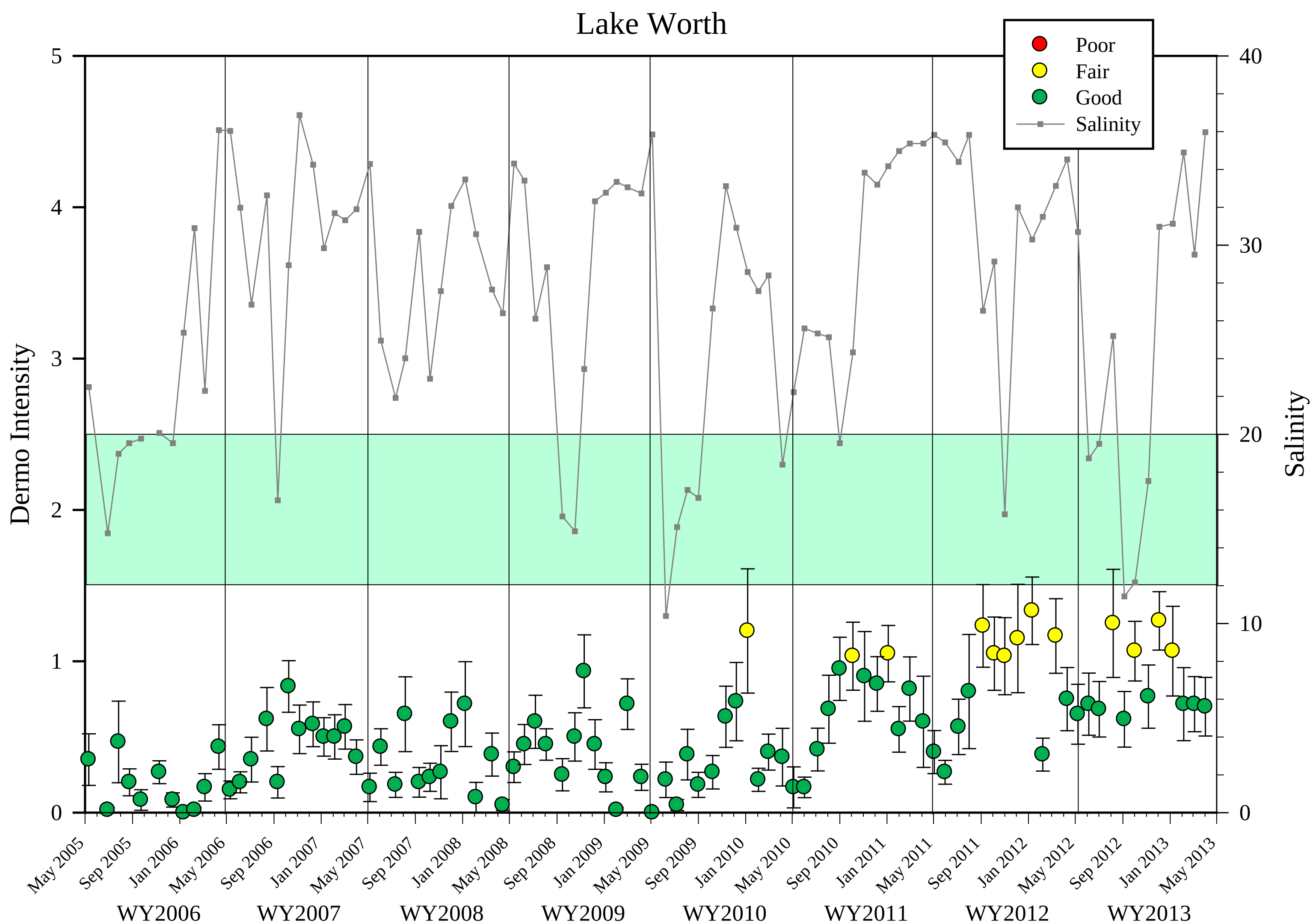

Figure 4-118. Monthly mean infection intensity (± standard deviation; red, yellow, and green circles) of oysters infected with *Perkinsus marinus* during monthly collections from LWL and monthly salinity. The green band represents the salinity range deemed most favorable for oyster survival and health in the lagoon.

Conclusion

Oyster populations in LWL have been negatively impacted by the highly variable freshwater inflows that are a result of the altered local hydrology. Periods of extremely high flow result in acute damage to oyster populations. Extended periods of reduced flow result in gradual increases in disease and predation rates that result in compromised oyster health and survivorship. The variability in and of itself can compound the problem because rapid shifts between dry and wet regimes reduce the opportunity for acclimatization by the oyster and other estuarine inhabitants. Salinities commonly exceed the upper limit of the optimal salinity range (12–20) in LWL and low salinity events have not been a major factor limiting oyster populations. High salinities are a persistent problem in LWL where freshwater inflows have not been of sufficient magnitude or duration to provide relief from disease and predation pressure.

Several steps can be taken to strengthen understanding of the relationship between flow and salinity in the estuary. For example, estuarine conditions are commonly summarized by water year, but oysters are often impacted on shorter time scales. In many cases, an extreme low salinity event and a prolonged high salinity event occurred within the same water year; therefore, a more detailed examination will likely yield a better understanding of how changes in salinity impact oyster biology over varying time scales. The estuary also has multiple potential sources of freshwater input and these should be considered in sum when interpreting freshwater inflow impacts on water quality, oysters and other estuarine fauna. Finally, collection of frequent and continuous water quality data in multiple locations within the estuary would allow for development and/or refinement of minimum/maximum flow rates to better prevent extreme salinity changes that lead to conditions that are detrimental to oysters.

Lake Worth Lagoon Seagrass

LWL, originally a freshwater lake, is now the largest estuarine system in Palm Beach County (PBCERM and FDEP 2008). 81% of the shoreline is bulkheaded (PBCERM et al. 2013). Activities that turned this once freshwater lake into an urbanized lagoon include extensive channel dredging, canal development, industrial and sewage runoff, port development, and shoreline hardening (PBCERM and FDEP 2008). LWL is divided into three segments—northern, central, and southern (**Figure 4-119**)—based on hydrological factors including water quality, circulation, and physical characteristics (PBCERM 2006). Fresh water is discharged into the northern segment of the lagoon from the C-17 canal (the Earman River), the central segment from the C-51 canal (West Palm Beach Canal) and the southern segment from the C-16 canal (Boynton Beach Canal). These discharges carry large influxes of nutrients, suspended and dissolved organic matter, contaminants, and toxins, affecting the flora and fauna (Crigger et al. 2005). Large and often prolonged releases occur from the C-51 canal and are the most influential on the health and status of the lagoon (PBCERM and FDEP 2008).

LWL supports seven species of seagrass: *S. filiforme*, *Thalassia testudinum* (turtle grass), *H. wrightii*, *H. johnsonii*, *Halophila decipiens* (paddle grass), *Halophila engelmanni* (star grass), and *Ruppia maritima* (widgeon grass). While all of these species are most successful in salinities greater than 20, *H. wrightii* can tolerate a wide range of salinity and salinity variations. However, *S. filiforme* is not as tolerant of low salinities or widely varying salinities (Irlandi, 2006). The quantity and quality of freshwater discharges can influence the overall health of seagrass communities (Crigger et al. 2005, PBCERM and FDEP 2008). Seagrass coverage varies throughout the lagoon, with more seagrass found in the northern segment than in the central or southern segments (PBCERM and FDEP 2008). The flushing provided by the Palm Beach Inlet along with decreased urbanization of the northern segment of LWL could be a contributing factor (PBCERM and FDEP 2008).

Performance measures were developed in an attempt to protect seagrass along with other valued ecosystem components throughout the lagoon. These performance measures are based primarily on the salinity requirements of two major valued ecosystem components within the lagoon, seagrass and oysters. Managing the freshwater discharges through the C-51 canal to eliminate flow events of 1,000 cfs and greater based on a 2-day moving average along with eliminating flows greater than 500 cfs based on a 7-day moving average, should help maintain target salinities of $\geq$ 20 in the northern and southern segments and salinities $\geq$ 15 in the central segment.

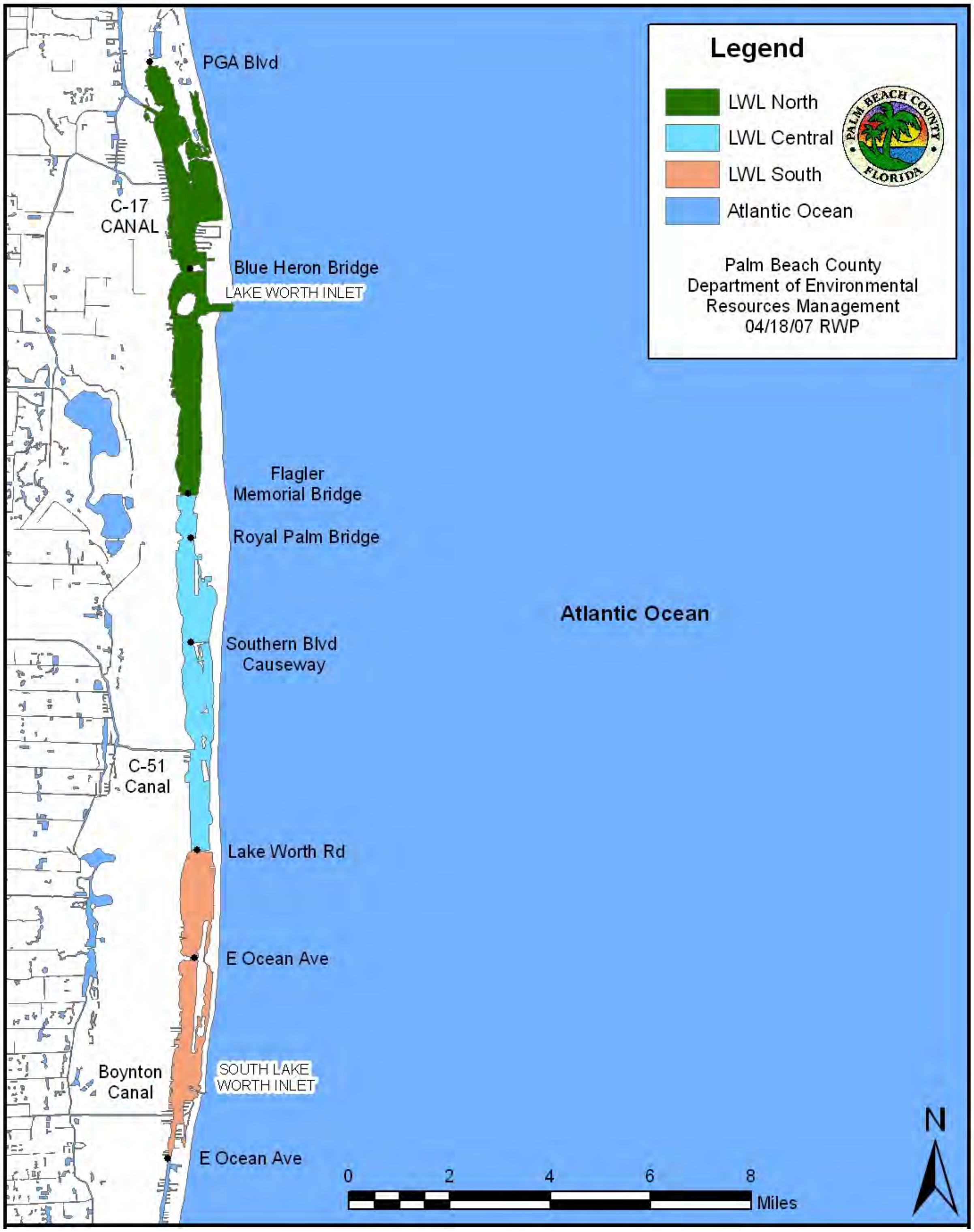

Figure 4-119. LWL segments—northern, central, and southern–based on hydrological factors.

Historically, the seagrass monitoring program in the LWL consisted of two components: (1) landscape-scale mapping collected in 2001 and 2007, and (2) patch-scale fixed transect monitoring collected annually starting in summer 2000. A new seagrass monitoring methodology was developed and implemented in LWL in December 2008. Five seagrass beds ranging from 1.5 to 2.5 acres were selected as sample sites (**Table 4-13**) and monitored once every two months using this new method. Two sites are located in the northern segment of the lagoon, one north and one south of the C-17 canal outlet, while the other three sites were located in the central segment of the lagoon, two north of the C-51 Canal and one south of the C-51 Canal (**Figure 4-120**). The new methods include randomly deploying 30 1-m^2 quadrats within specified boundaries at each site. Percent occurrence of total seagrass and each component species is determined within 25 quadrants of the 1-m^2 quadrats (**Figure 4-121**). Additionally, canopy height and quadrat location are documented (**Figure 4-121**). Species-specific percent occurrence and canopy height are also estimated. In April 2012 the northernmost site around the C-51 canal became inactive and the frequency of sampling was reduced to twice a year at the remaining four sites, once at the beginning of the growing season (April) and once at the end of the growing season (September/October). For more details on the MAP monitoring reductions please refer to **Table 3-10** in Chapter 3: Systemwide Science.

Table 4-13. Site sizes and current SAV species composition of sampling sites in LWL. Species in red is the dominant species.

Site Location	Site Name	Acres	Hectares	Seagrass Found at Site
	17A	2.28	0.9	*Halodule wrightii* (shoal grass) *Halophila decipiens* (paddle grass) *Halophila johnsonii* (Johnson's seagrass) *Syringodium filiforme* (manatee grass) *Thalassia testudinum* (turtle grass)
	17B	2.25	0.9	*Halodule wrightii* (shoal grass) *Halophila decipiens* (paddle grass) *Halophila engelmanni* (star grass) *Halophila johnsonii* (Johnson's seagrass) *Syringodium filiforme* (manatee grass) *Thalassia testudinum* (turtle grass)
	51A (Inactive as of April 1, 2012)	1.97	0.8	*Halodule wrightii* (shoal grass) *Halophila decipiens* (paddle grass) *Halophila johnsonii* (Johnson's seagrass) *Syringodium filiforme* (manatee grass) *Thalassia testudinum* (turtle grass)
North of the C-51 canal outlet	51B	2.27	0.9	*Halodule wrightii* (shoal grass) *Halophila decipiens* (paddle grass) *Halophila johnsonii* (Johnson's seagrass)
South of the C-51 canal outlet	51C	2.05	0.8	*Halodule wrightii* (shoal grass) *Halophila decipiens* (paddle grass) *Halophila engelmanni* (star grass) *Halophila johnsonii* (Johnson's seagrass) *Thalassia testudinum* (turtle grass)

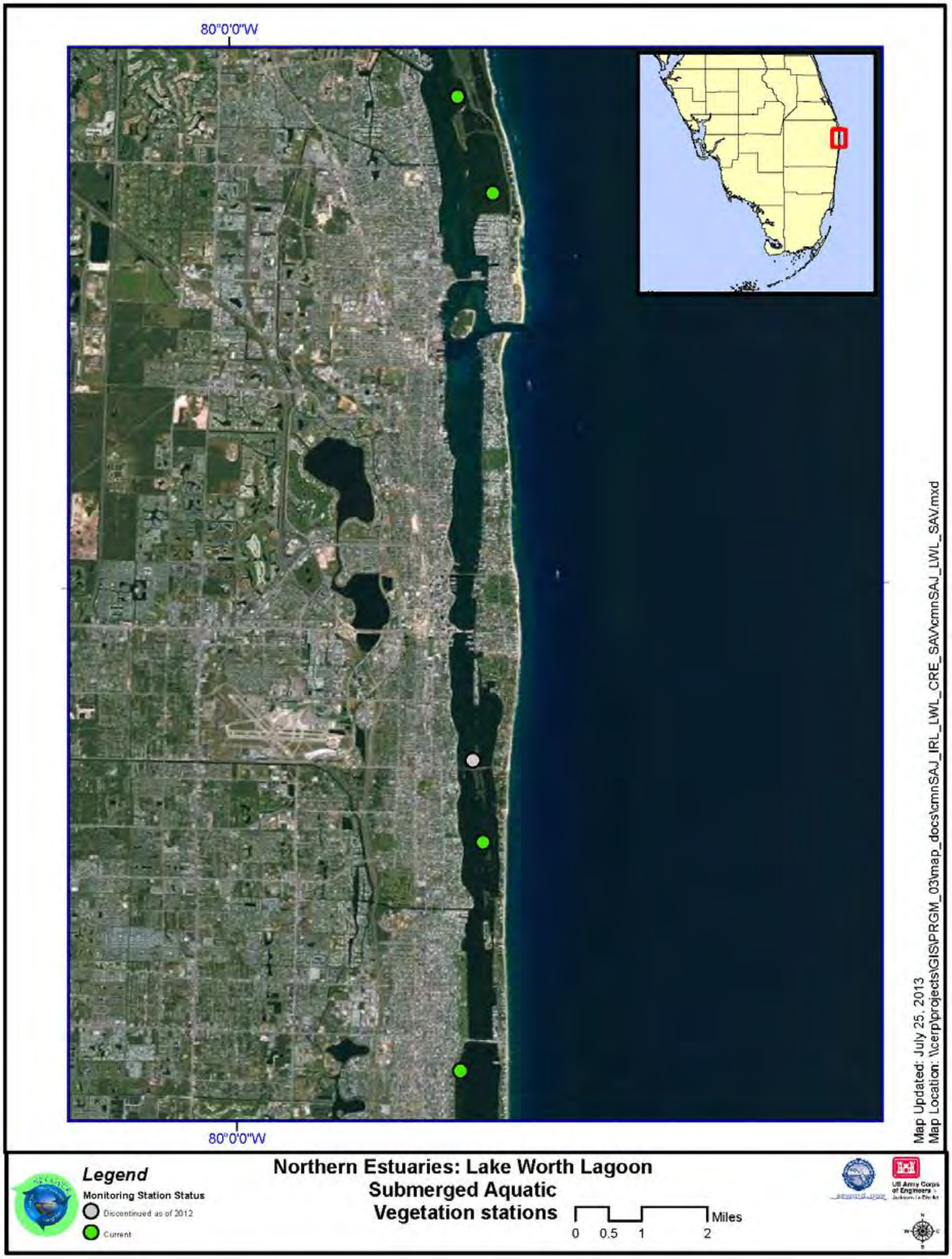

Figure 4-120. Location of monitoring sites for seagrass in LWL.

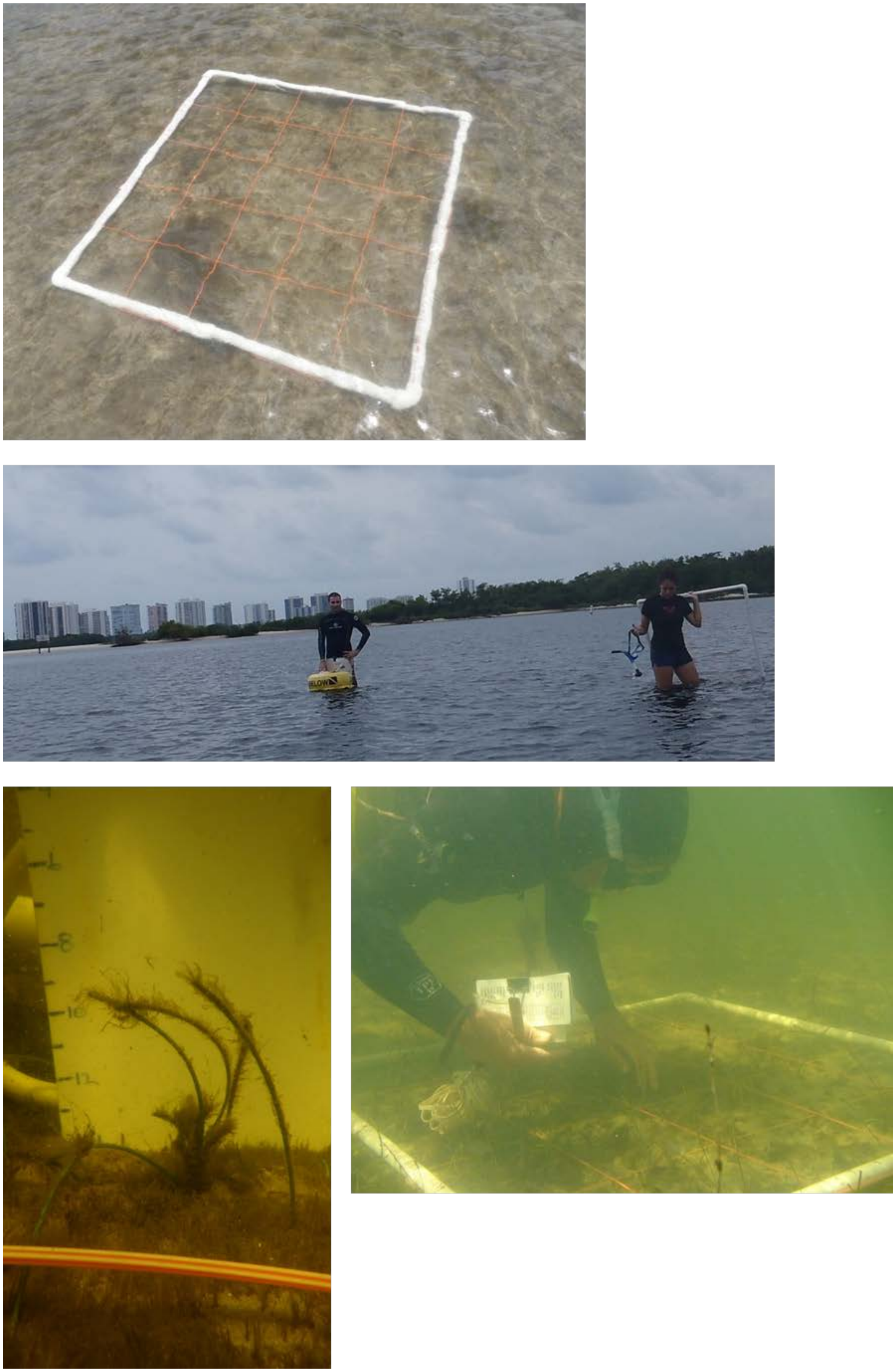

Figure 4-121. Photos of seagrass monitoring.

The period of record (POR) discussed here for this new monitoring program was December 2008–April 2013. During the POR, average monthly salinities in the northern segment of LWL consistently met performance measure targets whereas both the central and southern segments were out of target range at least one month during three during the five-year monitoring period (**Table 4-14**). Freshwater discharge out of the C-51 Canal failed to meet performance criteria at least 5% of the time during the POR for all water years excluding Water Year (WY) 2012 (May 1, 2011–April 30, 2012), which was in compliance all but 1 or 5 days of the water year (**Table 4-15**). Discharges out of the C-51 canal did not meet the 500-cfs 7-day moving average flow target 48%, 28%, and 29% of the time during WY 2010, WY 2011, and WY 2013, respectively. Although WY 2010 did not meet this flow target 48% of the time, seagrass at the sites seemed unaffected; however, drastic declines in percent occurrence of seagrass occurred at the 51B site during WY 2011 and at both the 51B and 51C sites during WY 2013. Percent occurrence rebounded the next monitoring event in 2011 (from 2% to 90%) though recovery of seagrasses after the large discharges in August 2012 (monthly average 1,405 cfs) due to Hurricane Isaac has yet to be seen. Percent occurrence of seagrass at the C-51 Canal sites appears to be highest when freshwater discharge does not exceed the 1,000-cfs 2-day moving average flow performance measure target more than 30 days out of the year (**Figure 4-122**).

Table 4-14. Monthly averages of salinity in the three segments of LWL for WY 2009–WY 2013. [a]

Water Year	Number of Months When Monthly Average Salinity in Northern Segment < 20	Number of Months When Monthly Average Salinity in Central Segment < 15	Number of Months When Monthly Average Salinity in Southern Segment < 20
2009	0	1	1
2010	0	1	0
2011	0	0	1
2012	0	0	0
2013	0	2	2

a. The monthly averages of salinity were calculated over multiple stations of a segment.

Table 4-15. Freshwater inflows from the C-51 canal for WY 2009–WY 2013.

Water Year	2-day Moving Average of Flows from C-51 Canal > 1,000 cfs, Target = 0		7-day Moving Average of Flow from C-51 Canal > 500 cfs, Target = 0	
	Number of Days When Target Not Met	Percentage of the Water Year Target Not Met	Number of Days When Target Not Met	Percentage of the Water Year Target Not Met
2009	52	14%	71	19%
2010	80	22%	177	48%
2011	18	5%	104	28%
2012	1	0.2%	5	1.3%
2013	64	17%	107	29%

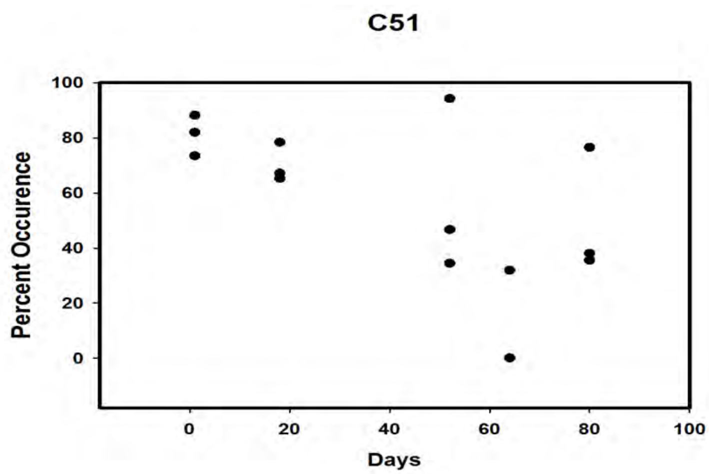

Figure 4-122. Average percent occurrence seagrass at the sites compared to the number of days per water year the freshwater discharge from C-51 canal > 1,000 cfs 2-day moving average of flows.

The two northern lagoon segment sites (17A and 17B) displayed seasonal variability throughout the POR (**Figure 4-123**), even though freshwater discharge increased from both the C-51 and C-17 canals during late August 2012–September 2012 due to Hurricane Isaac (**Figures 4-124** and **4-125**). It appears that the relatively stable salinity conditions at 17A, due to the lack of long-term extreme freshwater inflows, have allowed *S. filiforme*, a canopy-forming grass sensitive to salinity fluctuations, to uniformly dominate this site through time. *H. wrightii*, another canopy-forming species, dominated the 17B site throughout the POR. Seagrass percent occurrence at 17B decreased from 80% in September 2012 to 32% in April 2013. Monitoring performed in September 2013 will hopefully give insight as to whether this decline is due to normal seasonal variation or an after effect of freshwater input from Hurricane Isaac and the extremely wet early summer 2013. Mean seagrass canopy height also followed seasonal trends at both of the northern section sites (**Figures 4-126** and **4-127**).

The three central lagoon sites, 51A, 51B, 51C, which are influenced by freshwater flows from the C-51 canal, were dominated by *H. johnsonii* throughout most of the POR. Seagrass percent occurrence followed a seasonal pattern until September 2012, a month after Hurricane Isaac, during which time freshwater discharges from the C-51 Canal averaged 1,400 cfs. Large negative impacts on the seagrasses were seen both north and south of the C-51 Canal outlet in September 2012 and April 2013 (**Figures 4-123** and **4-125**). Seagrass percent occurrence at 51B (north of the C-51 outlet) decreased from 93% in April 2012 to less than 1% and 0% in September 2012 and April 2013, respectively. During the September 2012 monitoring event, seagrass at Site 51C (south of the C-51 outlet) decreased to 57% from 72% in April 2012. The April 2013 monitoring showed another decline to 7% occurrence. A shift in species dominance from *H. johnsonii* to *H. wrightii* at 51C also occurred during this timeframe. The shift in species dominance and the loss in percent occurrence are most likely due to increased turbidity caused by runoff from the hurricanes, freshwater inflow from the C-51 Canal and burial and scour from wave action.

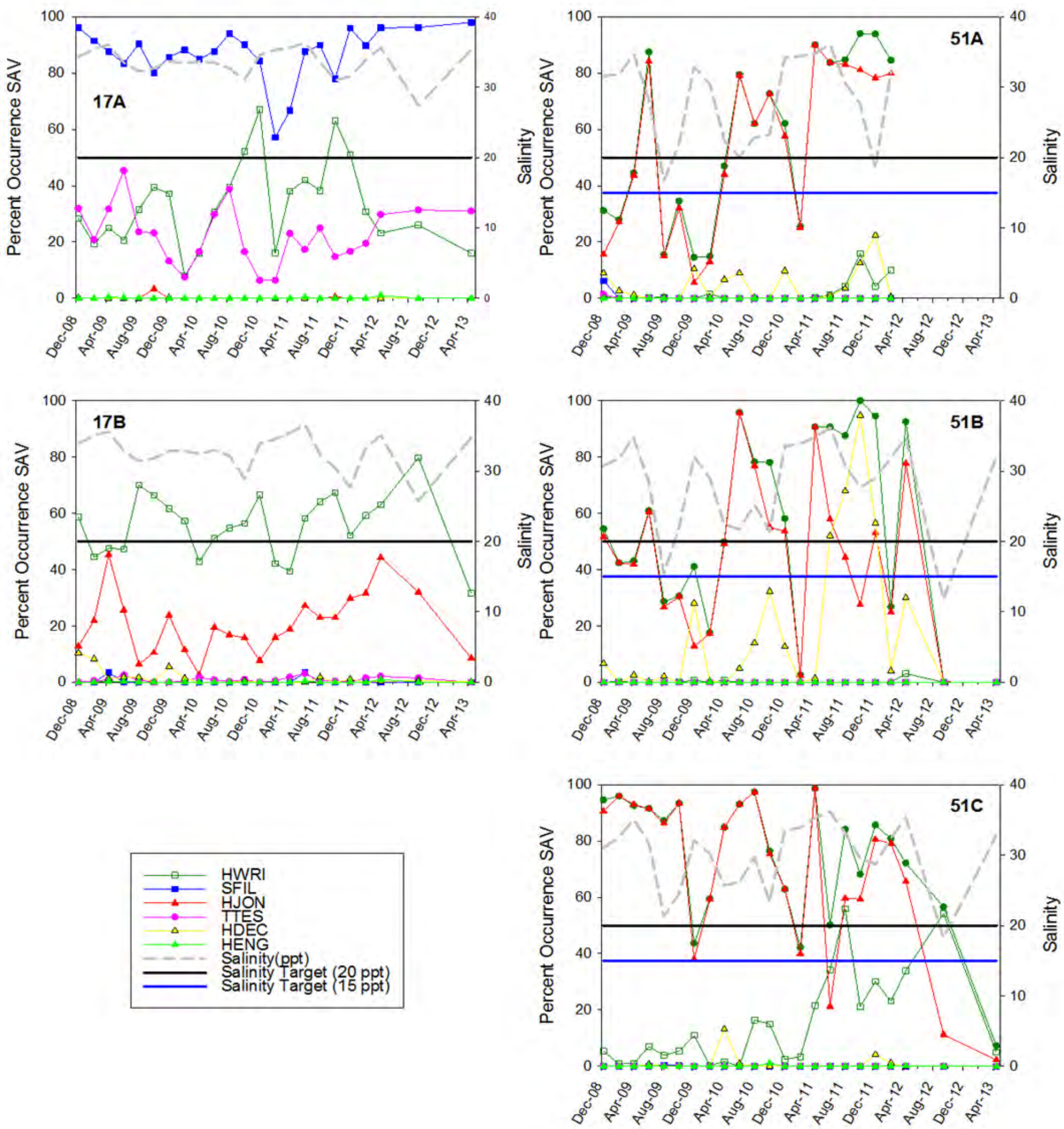

Figure 4-123. Average percent occurrence of seagrass species at each monitoring station along with average monthly salinity within the lagoon segment.

Note: HDEC – *H. decipiens*, HENG – *H. engelmanni*, HJON – *H. johnsonii*, HWRI – *H. wrightii*, SFIL – *S. filiforme*, TTES – *T. testudinum*.

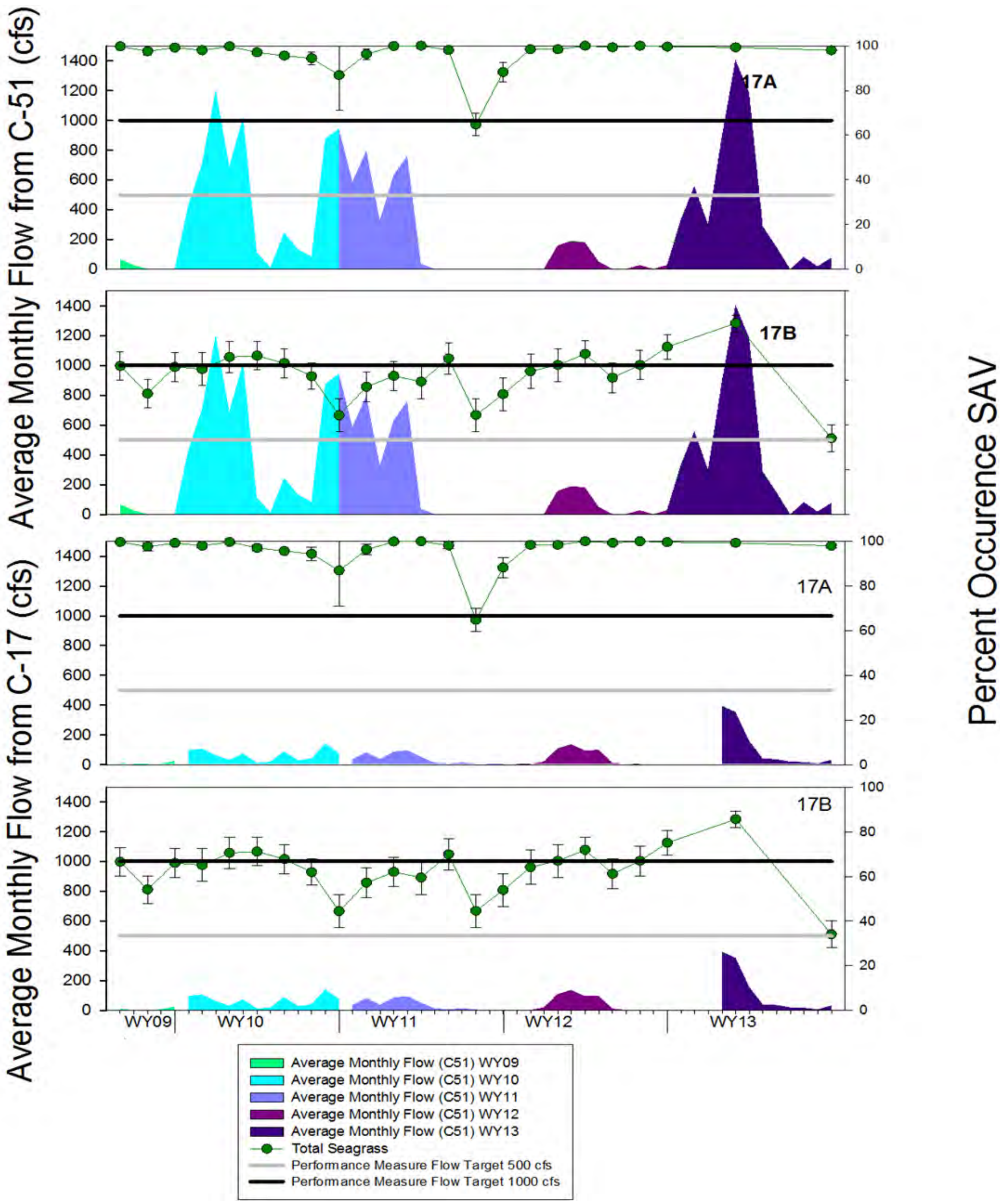

Figure 4-124. Average monthly percent occurrence of total seagrass at each of the two monitoring stations around the C-17 Canal outlet, along with average monthly flows from the C-51 (51A, 51B, 51C) and C-17 (17A, 17B) canals. Restorative flow targets for the LWL include (1) elimination of high flow events of 1,000 cfs or greater, and (2) elimination of flows greater than 500 cfs for extended periods of time (7 days or greater).

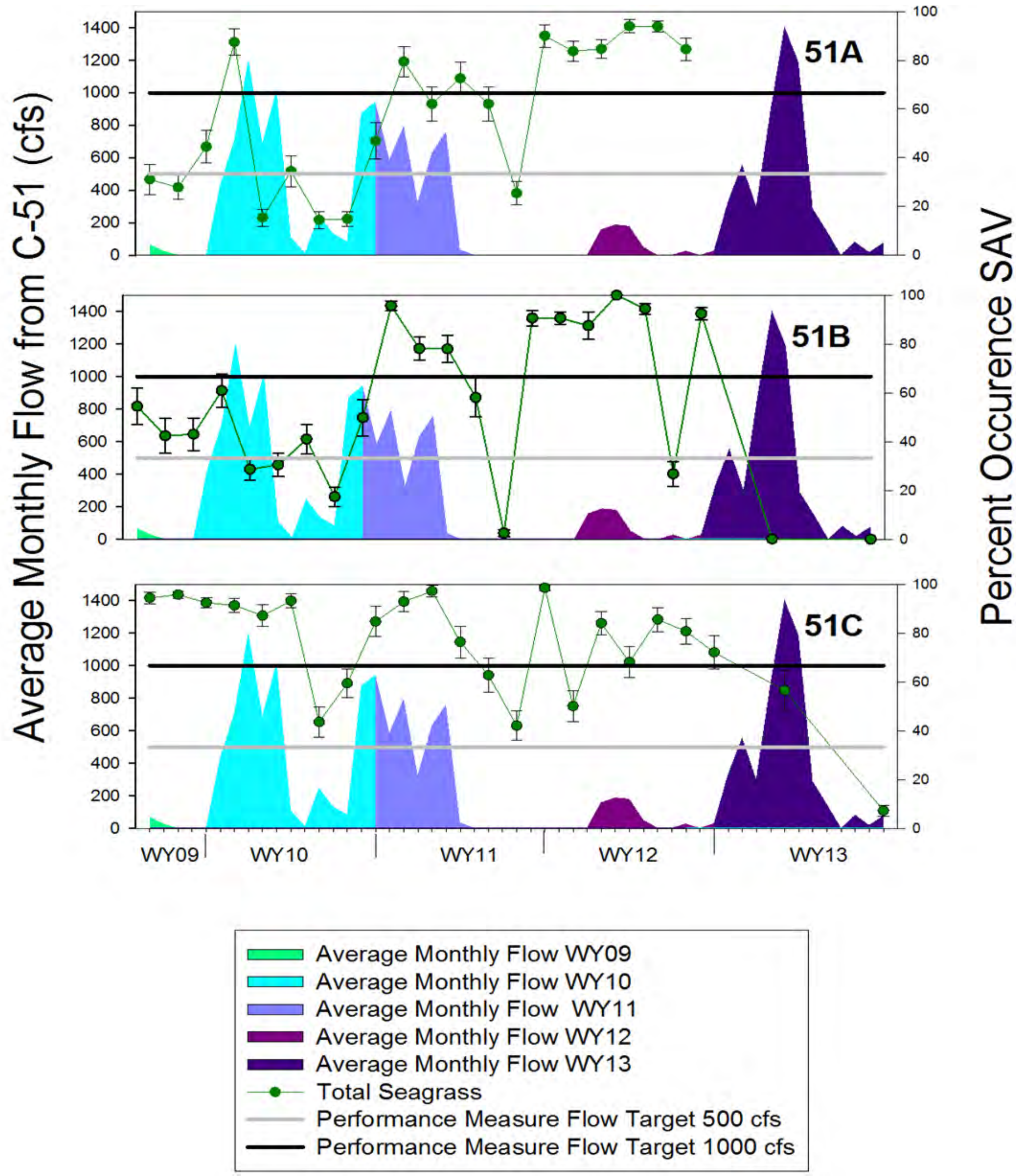

Figure 4-125. Average monthly percent occurrence of total seagrass at each of the three monitoring stations around the C-51 Canal outlet, along with average monthly flows from the C-51 Canal. Restorative flow targets for the LWL include (1) elimination of high flow events of 1,000 cfs or greater, and (2) elimination of flows greater than 500 cfs for extended periods of time (7 days or greater).

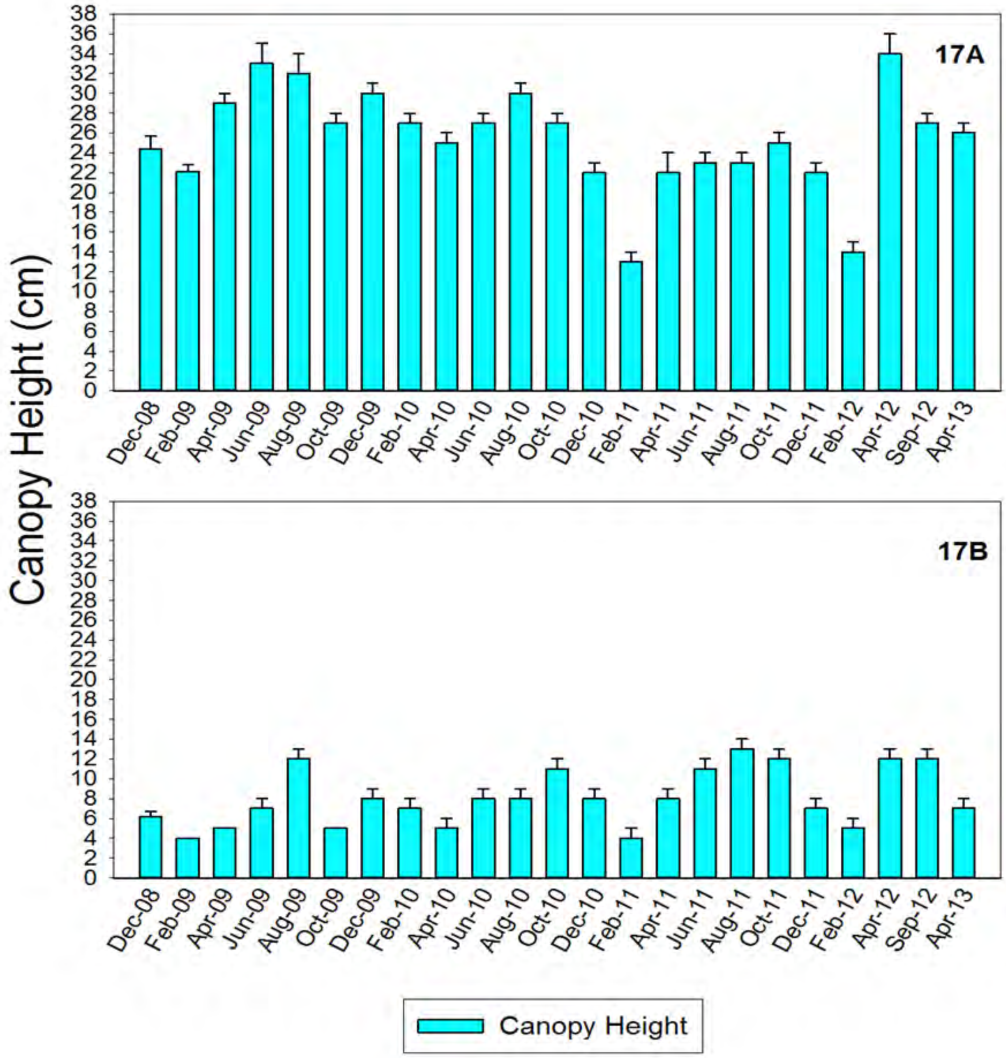

Figure 4-126. Mean canopy height of seagrass at each 17 site during the POR (December 2008–April 2013).

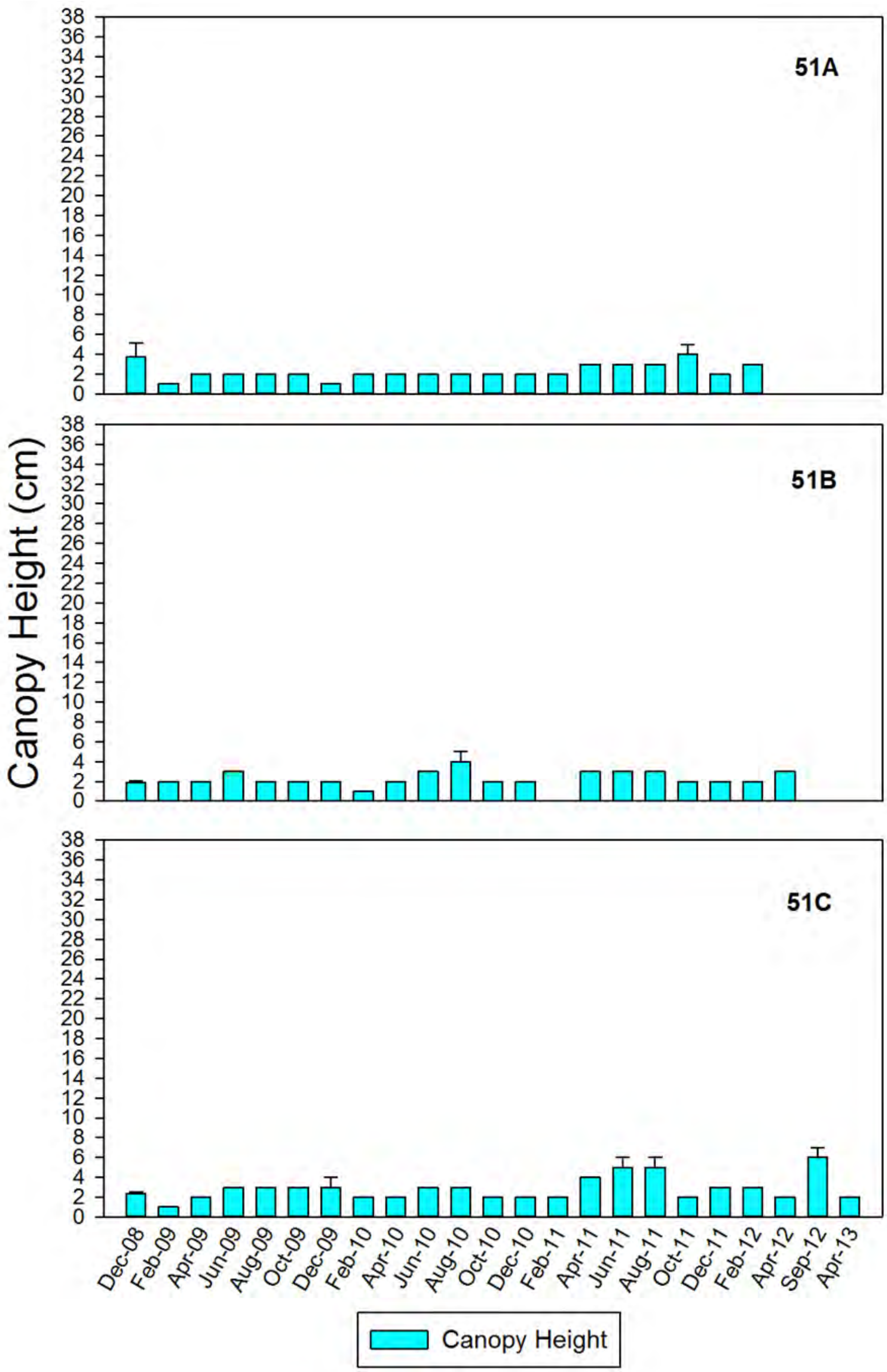

Figure 4-127. Mean canopy height of seagrass at each 51 site during the POR (December 2008–April 2013).

The bimonthly monitoring at the five seagrass beds provides a time series of data that provides a brief glance at the status of the health of seagrass beds in LWL. While the lagoon's seagrass experienced severe stress as a result of Hurricane Isaac and associated freshwater discharge, the status of the seagrass in the northern segment of the lagoon seems to have fared better than the seagrass in the central segment. While *S. filiforme* at the southern site in the northern segment (17A) remains stable, there is a decrease in *H. wrightii* percent occurrence at the north site of the northern segment (17B), which may be due to natural seasonal variability. The central segment of the lagoon is the most severely impacted with respect to turbidity and presence of muck due to the greater freshwater discharge from the C-51 canal then the C-17 or C-16 canals (PBCERM 2003, PBCDERM and FDEP 2008). SAV in the central segment of the lagoon tends to be dominated by *Halophila spp.*, diminutive seagrasses with a canopy height of 1–4 centimeters. As water quality and light attenuation decrease, seagrasses have been known to retreat to shallow depths (Bortone 2000, Orth et al. 2006). Extreme freshwater discharge from the C-51 canal during Hurricane Isaac in August and September 2012 may have influenced these parameters. The apparent loss of seagrasses in this segment may be due to increased loading of sediments associated with the discharge, thereby affecting water quality, clarity, and the overall health of seagrass communities. Also, muck as a result of extreme freshwater input in these areas tends to cover these species and may play a factor in their demise (PBCERM 2003, Orth 2006, PBCDERM and FDEP 2008, PBCDERM et al. 2013). Ongoing monitoring that provides extensive time series data would provide the best assessment of the performance of restoration activities as CERP is implemented.

Although the data did not show a direct relationship with salinity and seagrass percent occurrence, this may not be a true finding. Water quality in the lagoon is measured monthly. This time period of sampling may not be sufficient to capture a true relationship and thus determine if the salinity performance measures of 20, 15, and 20 in the northern, central and southern segments of the lagoon are adequate. Seagrass percent occurrence in the northern segment of the lagoon does not appear to be effected by freshwater discharges from the C-51 canal on which the flow performance measure is based. Data from this period of record is not suitable to determine if target flows will help protect seagrass in the central segment of the lagoon. Flows during WY 2010 did not meet performance measure targets 48% of the year, yet the only decline in seagrass during WY 2010 appears to be seasonal. The declines in percent occurrence seen during WY 2011 and WY 2013 may also be seasonal. This appears to be the case in WY 2011; however, due to reductions in monitoring implemented in April 2012, the frequency of monitoring may not be great enough to determine if the WY 2013 declines were seasonal or due to large freshwater releases associated with Hurricane Isaac.

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CHAPTER 5

LAKE OKEECHOBEE

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TABLE OF CONTENTS

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CHAPTER 5 LAKE OKEECHOBEE

INTRODUCTION

Since the natural shoreline, inflow, drainage and outflow of Lake Okeechobee was altered by the construction of the Herbert Hoover Dike and associated water control structures, water levels within the lake likely fluctuate with increased frequency and amplitude. The increase in lake stage fluctuations and amplitude occurs for several reasons: (1) the connection between the natural northern and western watershed flood plains has been severed, (2) the lake receives large volumes of channelized flow from the northern and western watershed regions, (3) there were probably no channelized flows to the south prior to the dike, water exited to the south as sheet flow, and (4) since dike completion, the lake has been managed primarily for water supply and flood control but also for navigation, regional groundwater recharge, recreational activities and ecological enhancement purposes. When rainfall is above normal in the watershed north and west of the lake, inflows can greatly exceed the ability to release water from the lake. This periodic flow imbalance has resulted in periods when lake stages exceed the stage envelope considered to be ecologically beneficial (12.5 feet–15.5 feet National Geodetic Vertical Datum of 1929 [NGVD]) (Havens 2002). Conversely, during periods of drought conditions, the combination of normal evapotranspiration and withdrawals from the lake for urban groundwater recharge and crop irrigation can lower lake stages to ecologically damaging levels (< 12 feet NGVD) as well.

During the late 1990s, approximately 250,000 acre-feet (ac-ft) of additional water storage were proposed for the Lake Okeechobee Watershed as part of the Comprehensive Everglades Restoration Plan (CERP) (USACE and SFWMD 2004). As part of the Lake Okeechobee Watershed Project analyses, 273,000 ac-ft of north of the lake storage was verified in 2007 as the most cost-effective storage option (USACE and SFWMD 2007). The intent of this additional water storage was to reduce damaging freshwater discharges to the east and west coast estuaries during wet years and to provide additional water supply during dry years. There would also be benefits to Lake Istokpoga, which would be allowed to fluctuate more naturally once its water supply function was transferred, in part, to the new storage features (USACE and SFWMD 2007). Incidental ecological benefits might also accrue in Lake Okeechobee by reducing the extent, frequency and duration of extreme high and low lake stages and by keeping the lake within the ecologically preferred stage envelope more frequently, as detailed in the Resoration Coordination and Verification (RECOVER) Lake Okeechobee performance measures (http://www.evergladesplan.org/pm/recover/perf_low.aspx). Further evaluations of the potential value of additional water storage upstream of Lake Okeechobee also were a feature of the River of Grass, Northern Everglades Initiative, and the *Lake Okeechobee Watershed Construction Project Phase II Technical Plan* (SFWMD et al. 2008) planning efforts beginning in 2007. However, since in the River of Grass planning effort, modeling of various water storage scenarios north of the lake coupled with various types of operational alternatives for treatment and conveyance of water south to the Everglades has made it difficult to isolate northern storage effects on the lake. Further, while modeling and evaluating the effects of additional storage on water supply and flood control appears to be relatively straightforward, the quality of the performance measures used to assess ecological effect-lake stage relationships in Lake Okeechobee have been inadequate, primarily because data to evaluate ecological responses within the lake under various lake stages have been limited.

Over the past 10 to 15 years, various ecological data have been collected under a wide variety of lake stages. As a result, the uncertainties associated with lake stage-ecological response relationships have been reduced. Therefore, ecological responses, based on the identification of analogous lake stage years, to hydrologic modeling output for various north of the lake storage scenarios generated by the monthly time step Reservoir Sizing and Operations Screening (RESOPS) model were examined, to more realistically evaluate how these water storage scenarios might benefit or impact Lake Okeechobee ecology. The goal of this modeling effort was to estimate how much additional water storage north of the lake would be needed to maintain the lake in the ecologically beneficial lake stage envelope more frequently than currently occurs, while still providing adequate water supply and flood protection services. Water supply in the model output is presented as the mean annual amount of water released from the lake (in ac-ft) and the percent of water supply demands not delivered.

Additionally, Lake Okeechobee has been mostly in the ecologically desirable stage envelope since Water Year (WY) 2009 (May 1, 2008–April 30, 2009) (**Figure 5-1**). When lake stage was outside of the ecologically desirable stage envelope during this period, it was generally below this envelope, which is considered to be less ecologically damaging to the nearshore region compared to when lake stage is

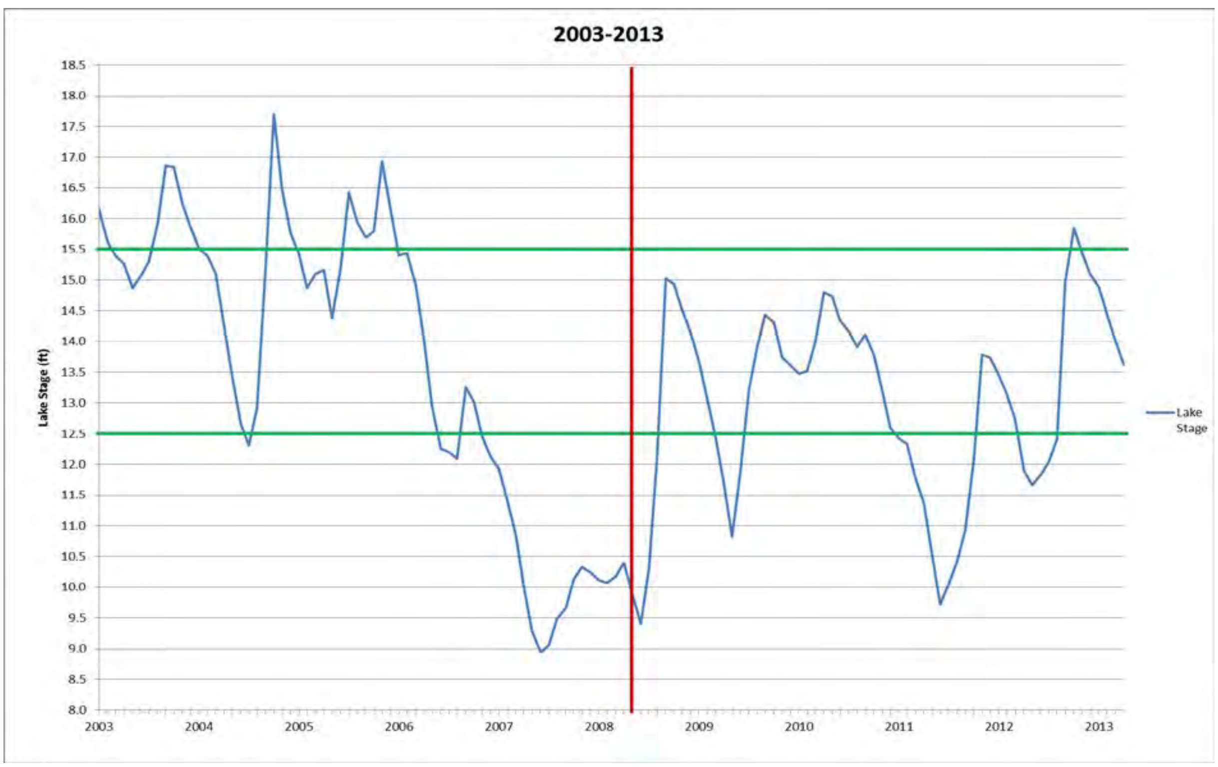


Figure 5-1. Lake Okeechobee stage hydrograph from 2003 to April 2013. The hydrograph to the right of the vertical line denotes the WY 2009–WY 2013 period and the horizontal lines denote the ecologically beneficial stage envelope.

above this envelope. When lake stages are below the ecologically desirable envelope, nearshore region plant communities and associated fish habitat does not appear to be significantly impacted, whereas the reverse has been observed when lake stage is above the ecologically desirable stage envelope for extended periods (e.g., > 3 months). While ecological damage to the aquatic community may occur in the littoral marsh when lake stage is below the ecologically desirable stage envelope, ecological damage can occur to both regions of the lake when lake stage exceeds the top of the ecologically desirable stage envelope. Therefore, lake stage dropping below the ecologically desirable stage envelope may be less ecologically damaging to the lake compared to when lake stage is above the ecologically desirable stage envelope. Overall, this recent water year period may serve as a glimpse at how the lake might respond ecologically if adequate storage capacity was available; minus any large-scale nutrient reduction.

Documenting and reporting ecological responses to the recent lake stage regime may be useful in assessing whether the northern water storage volumes proposed by various restoration programs are on track for meeting ecological goals. Wading birds, fish, submerged aquatic vegetation (SAV) and phytoplankton abundance data served as ecological indicators, and all except for wading birds (published data analyses were used in this case) were used to examine if statistically significant relationships existed with lake stages for each data set. Additionally, with the recent lower lake stage regime, performance measure targets for lake stage regime and water quality variables also were examined to determine if the low to moderate lake stages frequently observed since WY2009, and reflective of the lake stage regime might result in progress toward meeting these performance measures. Lake stages since WY2009 reflect the lake stage regime that might be expected to occur more frequently when all of the CERP projects are operational. Therefore, the performance measure target data for lake stage regime, water quality variables, algal bloom frequency and nearshore SAV coverage for the current reporting period (WY 2009–WY 2013) are compared with previous water years. A range of watershed reservoir storage model scenarios were also examined based on the assumption that additional storage will eventually be constructed as part of CERP and other restoration implementation, thereby enabling the lake to be managed to remain within the ecologically beneficial stage envelope more often than currently occurs.

Details about the water quality, lake stage, macroinvertebrate, phytoplankton, SAV monitoring and performance measures for Lake Okeechobee can be found in the 2007 and 2009 System Status Reports (SSRs) (RECOVER 2007, 2011). Recent periphyton and native fish (nearshore, electrofishing; pelagic, trawl) monitoring details can be found in Chapter 8 of the *2014 South Florida Environmental Report – Volume 1* (Bertolotti et al. 2014). There are no lake performance measures for periphyton or native fish and the lake macroinvertebrate performance measure can be found in the previously referenced SSRs. The status of these variables for WY 2009–WY 2013 and how they compare to the 2009 five-year mean values and performance targets, where applicable, are presented in the next sections.

WATER QUALITY

The five-year averages for nutrient concentrations and total phosphorus (TP) load into Lake Okeechobee for WY 2009–WY 2013 were lower than the previous five-year average values (**Table 5-1**), while the WY 2009–WY 2013 total nitrogen (TN):TP ratio and algal bloom frequency means were slightly higher. All of the mean nutrient concentrations during WY 2009–WY 2013 also exceeded their performance measure targets (**Table 5-1**), while the water column TN:TP and dissolved inorganic nitrogen (DIN):soluble reactive phosphorus (SRP) ratios did not achieve the target ratios. The extreme low lake stage performance measure target (> 10 feet NGVD) was met for each year during this reporting period, except for approximately May and June 2008 and one month during summer 2011. The extreme high lake stage performance measure (> 17 feet NGVD) was not exceeded during any year in this reporting period and the lake was within the ecologically beneficial stage envelope for a majority (77%) of that time.

Table 5-1. Water quality and SAV five-year averages and performance measure targets. (Source: RECOVER 2011 and Bertolotti 2014.)

Variable	Performance Measure Target	Five-Year Average WY 2004–WY 2008	Five-Year Average WY 2009–WY 2013
TP load	140 metric tons per year (mt/yr)	558 mt/yr	450 mt/yr
Pelagic TP	40 micrograms per liter (µg/L)	184 µg/L	122 µg/L
Pelagic TN	not applicable	1.69 parts per million (ppm)	1.50 ppm
Pelagic SRP	not applicable	60 µg/L	39 µg/L
Pelagic DIN	not applicable	302 µg/L	199 µg/L
Pelagic TN to TP	> 22:1	9:1	12.3:1
Pelagic DIN to SRP	> 10:1	5.24:1	5.10:1
Algal bloom frequency	< 5 percent of pelagic chlorophyll *a* exceeding 40 µg/L	4.8%	7.4%
Diatom:Cyanobacteria Ratio	> 1.5:1	10.06:1 (Pelagic) 15.8:1 (Nearshore)	1.86:1 (Pelagic) 3.13:1 (Nearshore)
Nearshore[a] Water Clarity	Secchi disk visible on lake bottom at all nearshore SAV sampling locations from May to September	15%	30%
Nearshore TP	below 40 µg/L	114 µg/L	76 µg/L
Nearshore SAV Coverage	Total SAV ≥ 40,000 acres Vascular SAV ≥ 20,000 acres	26,542 acres 8,926 acres	38,731 acres 30,008 acres

[a] Nearshore SAV sites were replaced with nearshore South Florida Water Management District water quality sites in WY 2012, so the five-year water clarity average values are not directly comparable.

CYANOBACTERIA

Methods

For cyanobacteria, summer (July) abundances (as cell biovolumes) were scored based on correlations with mean monthly prior May lake stages, using the following criteria. If Lake Okeechobee stage is above 14 feet NGVD in May, the probability of having elevated cyanobacteria abundance during the succeeding summer period is high. When lake stage is below 12 feet NGVD in May, the probability of having elevated cyanobacteria abundance in the succeeding summer is low (**Figure 5-2**). When lake stage is between 12 feet NGVD and 14 feet NGVD in May, the probability of having elevated cyanobacteria abundance in the succeeding summer is intermediate.

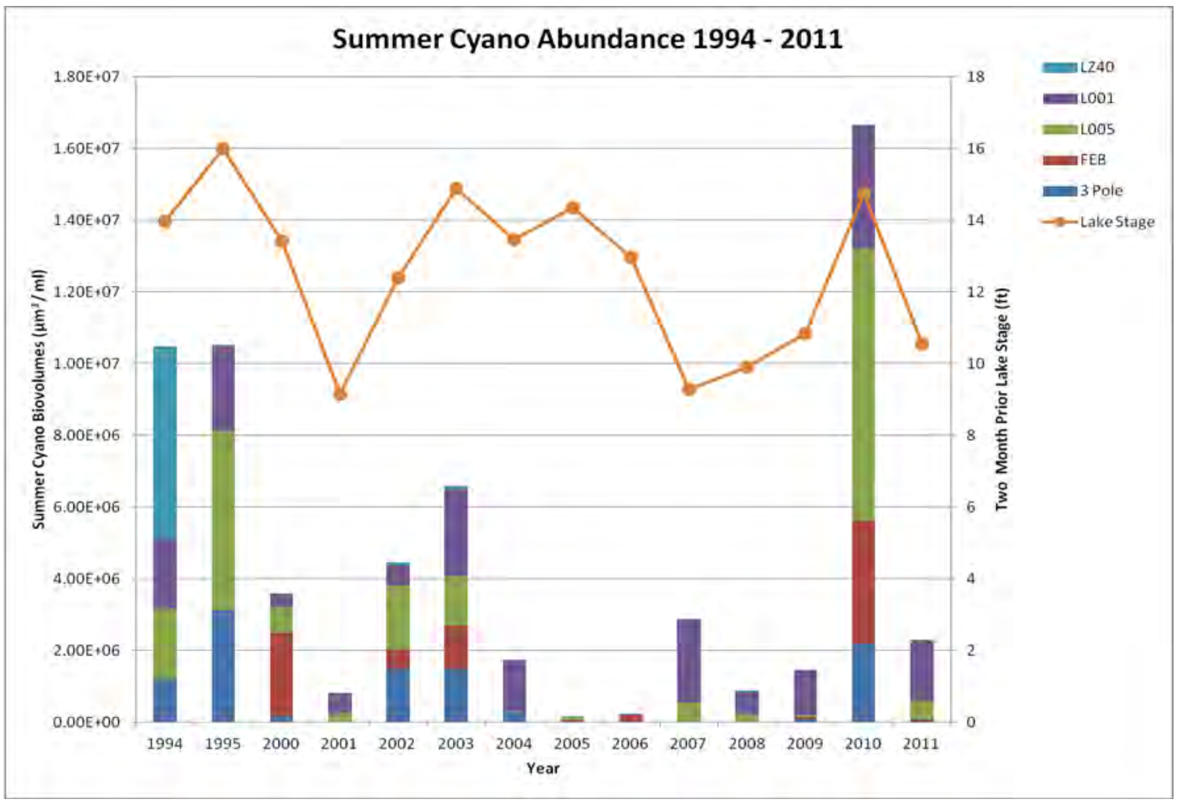


Figure 5-2. The relationship between summer cyanobacteria (cyano) abundance (as biovolumes in cubic micrometers per milliliter [µm^3/ml]) and lake stages during the prior May, for 1994–1995 and 2000–2011 at the lake routine plankton monitoring sites.

The performance measure is graded as follows:

- Greater than 14 feet NGVD results in an increased probability of summer cyanobacteria blooms – 0 points
- Between 12 feet and 14 feet NGVD – Intermediate probability of having elevated cyanobacteria abundance during the succeeding summer – 1 point
- Less than 12 feet NGVD, low probability of summer cyanobacteria blooms – 2 points

Results

The mean diatom:cyanobacteria ratios in both the pelagic and nearshore regions were substantially lower during the WY 2009–WY 2011 period relative to the previous five-year mean values but continued to be greater than the performance measure target (**Table 5-1**). There are no data for 2012 or 2013 due to delays in taxonomic sample processing arising from budgetary constraints.

PERIPHYTON

Nearshore periphyton abundance (as cell biovolumes) and nutrient content data were examined to determine if recent lower lake stages corresponded with significant increases in these periphyton attributes. Spring and fall periphyton data collected between 2003 and 2012 (fall 2012 epiphytic taxonomic samples have not been processed to date) and were compared as the pre-drought higher lake stage (2003–2006) and the drought/post-drought low lake stage (2008–2012) periods. A significant increase in periphyton abundance and nutrient content was hypothesized to be related to the generally lower lake stages and reflected by increased water clarity and light penetration into the water column, which is a factor that can strongly influence periphyton growth in the nearshore region of the lake (Steinman et al. 1997, Carrick and Steinman 2001, Rodusky et al. 2001).

The 2008–2012 mean periphyton abundances were higher for epipelon and *Chara*, *Schoenoplectus* and *Vallisneria* epiphytes, with the highest mean overall epiphytic abundances generally occurring during fall 2011 and spring 2012 (**Figures 5-3** through **5-5**). However, as epiphytic abundances increased with lower lake stages, the amount of associated sample abundance variability likewise increased, as illustrated in the size of the standard deviation bars. This variability likely accounts for the small differences in among-groups epiphytic abundance separation. All epiphytic and epipelic communities have been dominated (> 80 percent) by diatoms throughout the study period and in general, epiphytes on both emergent host taxa have been lower than on the SAV host taxa, the same pattern previously observed in the lake (Zimba 1995).

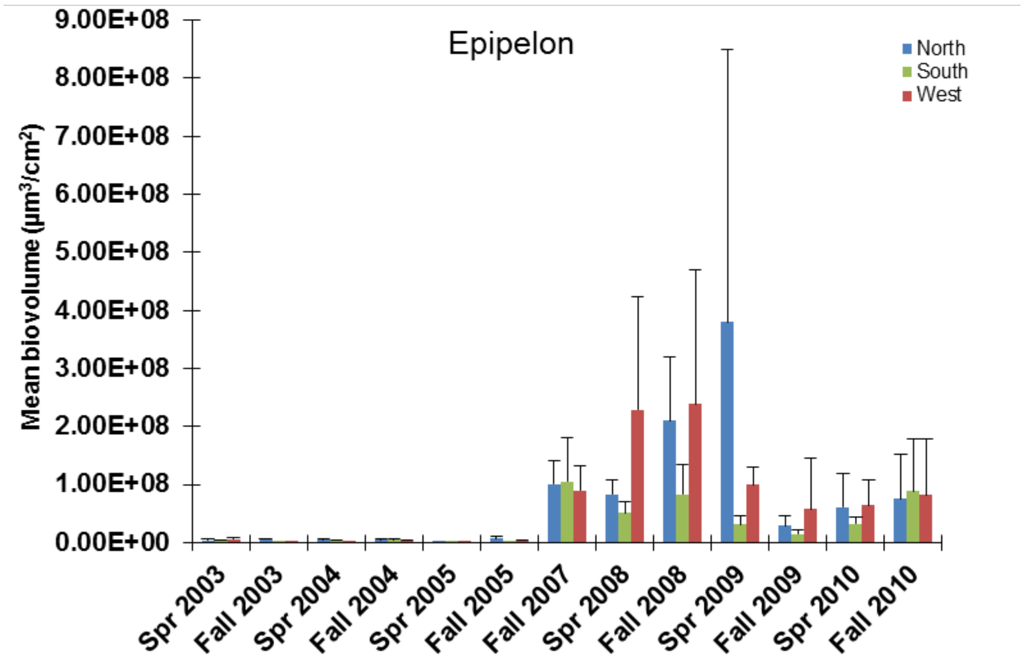


Figure 5-3. Mean epipelic abundances in cubic micrometers per square centimeter ($\mu m^3/cm^2$) +1 standard deviation) for spring and fall 2003–2005 and 2007–2010. Different colored bars represent the three nearshore regions, as indicated in the legend.

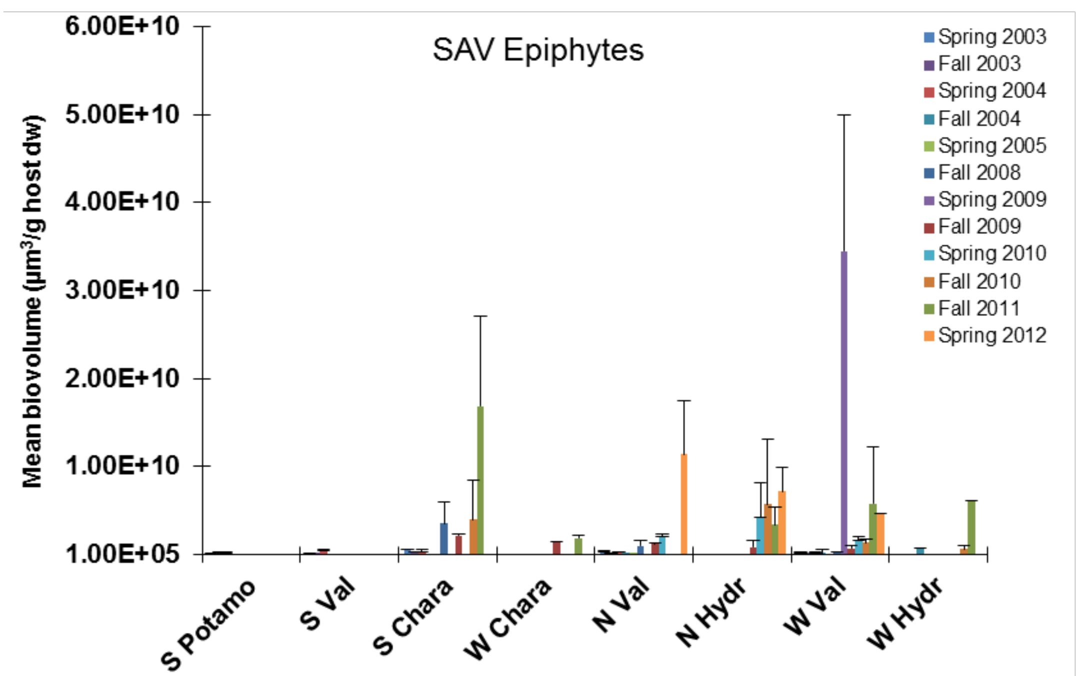


Figure 5-4. Nearshore *Chara*, *Hydrilla* (Hydr), *Potamogeton* (Potamo) and *Vallisneria* (Val) epiphytic mean abundances (+1 standard deviation) in Lake Okeechobee as cubic micrometers per gram host dry weight (μm^3/g host dw). Means are presented by geographic region (north [N], south [S], west [W]).

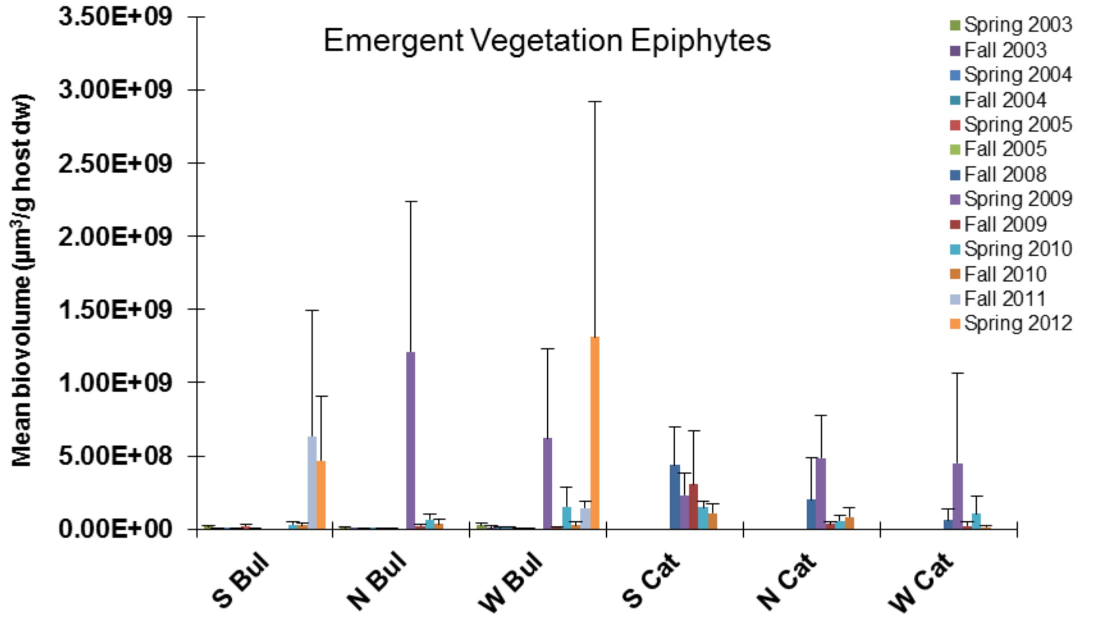


Figure 5-5. Nearshore *Schoenoplectus* (Bul) and *Typha* (Cat) and nearshore region (north [N], south [S], west [W]) epiphytic mean biovolumes (+1 standard deviation) in Lake Okeechobee as cubic micrometers per gram host dry weight (μm^3/g host dw).

Epipelic sampling concluded in 2010 and several SAV and emergent vegetation taxa (e.g. *Hydrilla*, *Typha*) in large areas of the nearshore region were either absent, too sparse to exist as a homogenous unit or did not harbor enough epiphytic biomass to enable nutrient tissue analyses to be conducted. Therefore, although all data are shown, the statistical analyses were only conducted on taxa that were present most spring and fall sampling periods.

Periphyton nitrogen (N) and phosphorus (P) mean concentrations were similar among the two temporal groups for most of the comparable host epiphytes and for the epipelon (**Figures 5-6** and **5-7**). The west region *Vallisneria* epiphytes had significantly more ($p < 0.05$) N content during the 2008–2012 period, while the 2003–2006 P content also was significantly higher ($p < 0.05$) in *Schoenoplectus*. Also, there were no statistically significant differences in the amount of nutrients among the different host-associated epiphytic communities, except for *Chara*-associated epiphytes, which had significantly less nutrient content during the 2008–2012 period. Epipelon likewise had significantly less nutrient content relative to the epiphytic communities during the entire study period. There were no 2002–2006 nutrient content data for *Chara* and *Hydrilla* associated epiphytes.

Mean total N:P tissue ratios varied little during the study and suggested strong N-limitation (8:1–10:1) for all epiphytic and epipelic communities, with two exceptions. *Chara*, associated epiphtyes, and epipelon in the southern nearshore region had mean N:P tissue ratios of 18:1 and 38:1, respectively. These data suggest that P-limitation or conditions approaching P-limitation can occur among some of the periphyton communities in the southern nearshore region of the lake.

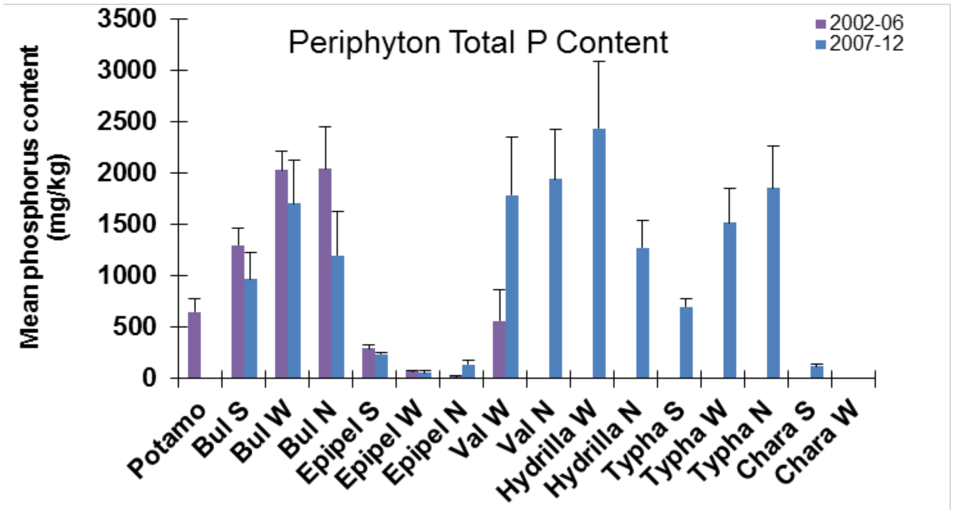


Figure 5-6. Nearshore epipelic (Epipel), SAV and emergent vegetation host epiphytic (*Chara*, *Hydrilla*, *Potamogeton* [Potamo], *Schoenoplectus* [Bul], *Typha* and *Vallisneri*a [Val]) and nearshore region (north [N], south [S], west [W]) mean tissue P content (+1 standard deviation) in Lake Okeechobee as milligrams per kilogram (mg/kg) dry weight.

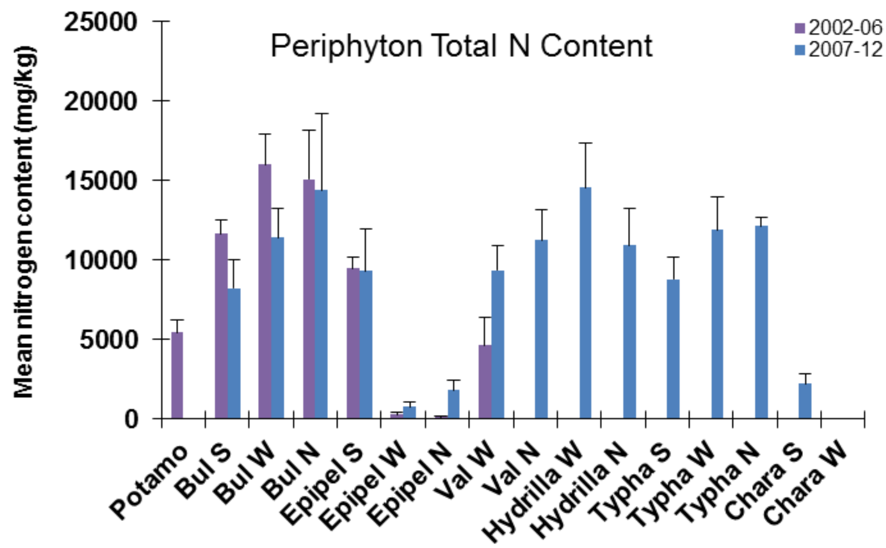


Figure 5-7. Nearshore epipelic (Epipel), SAV and emergent vegetation host epiphytic (*Chara*, *Hydrilla*, *Potamogeton* [Potamo], *Schoenoplectus* [Bul], *Typha* and *Vallisneria* [Val]) and nearshore region (north [N], south [S], and west [W]) mean tissue N content (+1 standard deviation) in Lake Okeechobee as milligrams per kilogram (mg/kg) dry weight.

MACROINVERTEBRATES

For the WY 2009–WY 2013 period, bi-annual benthic macroinvertebrate sampling has been conducted by the Florida Fish and Wildlife Conservation Commission during February and August between 2009 and 2011. During these sampling trips, triplicate samples were collected at each of 18 long-term nearshore and pelagic sites (RECOVER 2011) for a total of 54 samples. During May of both 2009 and 2010, partial sampling was conducted at half of these sites. Due to budgetary constraints, no pelagic zone samples were collected during Fiscal Year (FY) 2012 (October 1, 2011–September 30, 2012) and FY 2013 and during February 2011, samples were collected at only the mud and sand sediment sites. All of the samples collected during this period have been processed, with the exception of 22 samples collected during 2011. The remaining 22 preserved samples have been archived prior to sample processing and taxonomic identification steps. These samples were anticipated to be processed and the data quality assured and quality controlled by September 30, 2013 (G. Warren, personal communication), but the data were not available during this chapter preparation period. The data have not been updated since the 2009 SSR (RECOVER 2011).

FISH

Angler Creel Data

Angler pan fish creel data were collected between 1997 and 2005 in Novmeber and December by the Florida Fish and Wildlife Conservation Commission. Pan fish include bluegill (*Lepomis macrochirus*) and redear sunfish (*Lepomis microlophus*). These data were used since they were found to have a statistically significant relationship with lake stages.

The following criteria were used in developing a scoring rubric. When the November and December lake stages are greater than 16 feet, the successive January and February environmental conditions in the lake are considered to be poor, resulting in lower pan fish catch rates (**Figure 5-8**). November and December lake stages between 15 and 16 feet NGVD or below 12 feet NGVD appear to be tolerable, but suboptimal. Optimal conditions occur when lake stages are between 12 and 15 feet NGVD.

The pan fish catch relationship with lake stage was scored using the following criteria:

- Greater than 16.00 feet NGVD: conditions very poor – 0 points
- Between 15 feet and 16 feet NGVD: conditions intermediate – 1 point
- Between 12 and 15 feet NGVD: conditions optimal – 2 points
- Less than 12 feet NGVD: conditions intermediate – 1 point

In instances where one month is in one lake stage category but the next month is in an adjacent stage category, the lower score was applied (e.g., the lake is > 15 feet one month, and 14 feet the next, pan fish would be scored as a 1 rather than as a 2).

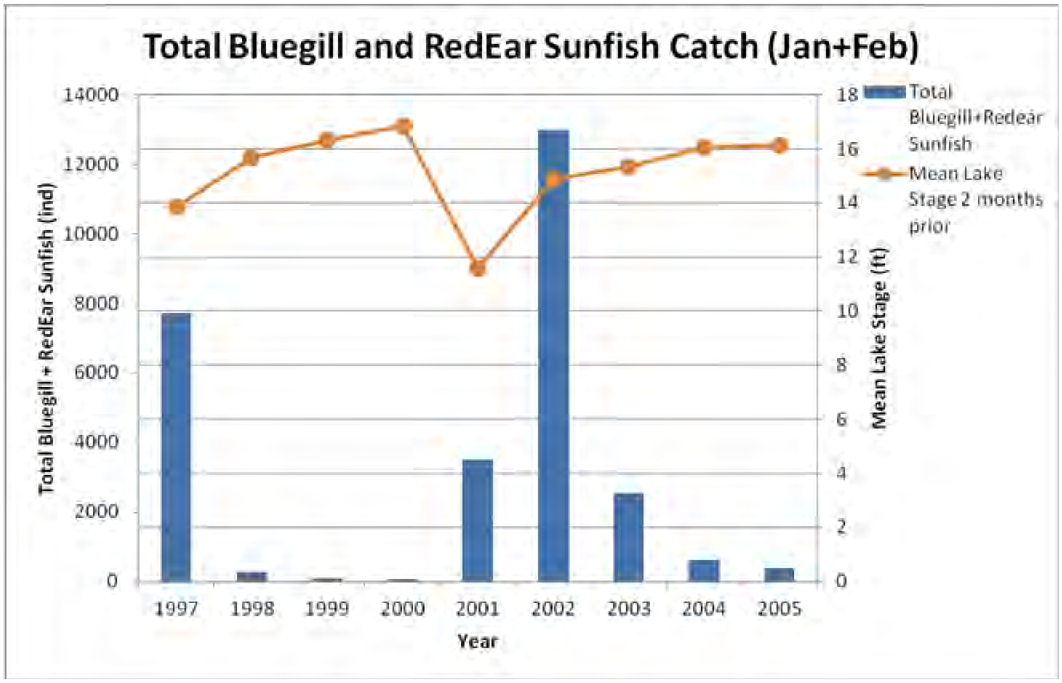


Figure 5-8. The relationship between January–February pan fish creel data (number of individuals [ind]) and lake stages during the prior November and December for 1997–2005.

Electrofishing Data

Electrofishing abundances increased between WY 2009 and WY 2011 and were somewhat lower, but still within historical levels, during WY 2012 and WY 2013. Fish abundances varied between 4,345 individuals (WY 2013) and 7,255 individuals during WY 2011. Combined collected biomass by electrofishing varied between 361 kilogram (kg) and 905 kg during WY 2009 and WY 2012, respectively. Fish abundance was higher during this reporting period than after Hurricanes Frances, Jeanne and Wilma in 2004–2005. Dominant fish taxa during WY 2009–WY 2013 generally consisted of both forage taxa (threadfin shad [*Dorosoma petenense*], gizzard shad (*Dorosoma cepedianum*), eastern mosquitofish [*Gambusia holbrooki*], inland silverside [*Meinidia beryllina*] and sailfin molly [*Poecilia latipinna*]), two piscivorous taxa (largemouth bass, bluegill), and one molluscivorous taxon (redear sunfish). By weight, largemouth bass, striped mullet (*Mugil cephalus*), bluegill and Florida gar (*Lepisosteus platyrhincus*) comprised the majority of the annual catch. Over this reporting period, piscavore abundance increased while forage taxa generally decreased, suggesting that piscavore taxa were becoming more important and exerting increasing predatory pressure on the forage fish taxa.

Trawling Data

Fish abundance in the pelagic region likewise increased between WY 2009 and WY 2011 and then decreased during WY 2012 and WY 2013; although it continued to remain within historical levels. Fish abundances varied between 2,816 individuals during WY 2009 and 13,544 individuals during WY 2011. Combined collected biomass by trawl varied between 221 kg and 653 kg during WY 2009 and WY 2011, respectively. The maximum trawl abundance was attributed to an increase in gizzard and threadfin shad (Bertolotti et al. 2014) although threadfin shad abundance has somewhat decreased over the reporting period, accounting for 31% of the abundance in WY 2013, compared to 41% of the abundance in WY 2009. Even with this decrease in abundance, threadfin shad still ranked as the most abundant taxa in the trawl data during WY 2013, followed by speckled perch (*Pomoxis annularis*), bluegill, and white catfish (*Ameiurus catus*). Conversely, speckled perch accounted for < 5% of the trawl abundance during WY 2009–WY 2011, but have increased in abundance since then, accounting for 29% of the abundance during WY 2013. These data indicate that speckled perch may finally be recovering from their nearly complete absence from the lake during the years immediately following the passage of Hurricanes Frances, Jeanne and Wilma. By weight, white catfish, bluegill, gizzard and threadfin shad, and Florida gar were typically the biggest contributors to the combined collective annual biomasses.

WADING BIRDS

Methods

For wading birds, annual nesting season (January–June) lake stage conditions were used to assess water level suitability for nesting success. Conditions are considered to be good for nesting when the nesting season lake stage is between 10 and 14.4 feet, along with a recession rate of at least 0.5 feet per month. For the performance scoring, one point is assigned to each yes response, for an annual maximum potential total of four points:

- Did the lake remain under 14.4 feet NGVD throughout the period January through June?
- Did the lake remain above 10 feet NGVD throughout the period January through June?
- Was there a mean regular recession rate of between 0.5 and 1.5 feet per month from January to June?
- Were there no reversals > 0.5 feet between January and June?

Results

The generally lower lake stages since 2007 appear to have resulted in a higher and more consistent annual wading bird nesting effort. During WY 2009–WY 2013, nesting effort of all tracked species ranged from 3,079 to 7,461 nests per year. During WY 2006–WY 2008, nesting effort was more variable and declined substantially, from a strong effort of 11,310 nests in WY 2006, to 21 nests, the lowest ever recorded nesting effort, in WY 2008. The decline in nesting effort reflected dry marsh habitat, in which most of the vegetation was burned during several large fires. Extremely low lake stages during 2007 and 2008 also resulted in the loss of the aquatic marsh food web and subsequently, prey for wading birds.

SUBMERGED AQUATIC VEGETATION

Methods

For vascular SAV, the July–August annual nearshore SAV mapping coverage data were used since they had the most significant correlation with lake stage, using the following criteria:

- When lake stage is greater than 15.5 or less than 12 feet NGVD in July, conditions are poor for vascular SAV, so these conditions both score 0 points.
- When lake stage is between 12 and 15.5 feet NGVD in July, conditions are optimal for vascular SAV coverage (**Figure 5-9**), therefore this condition scores one point.

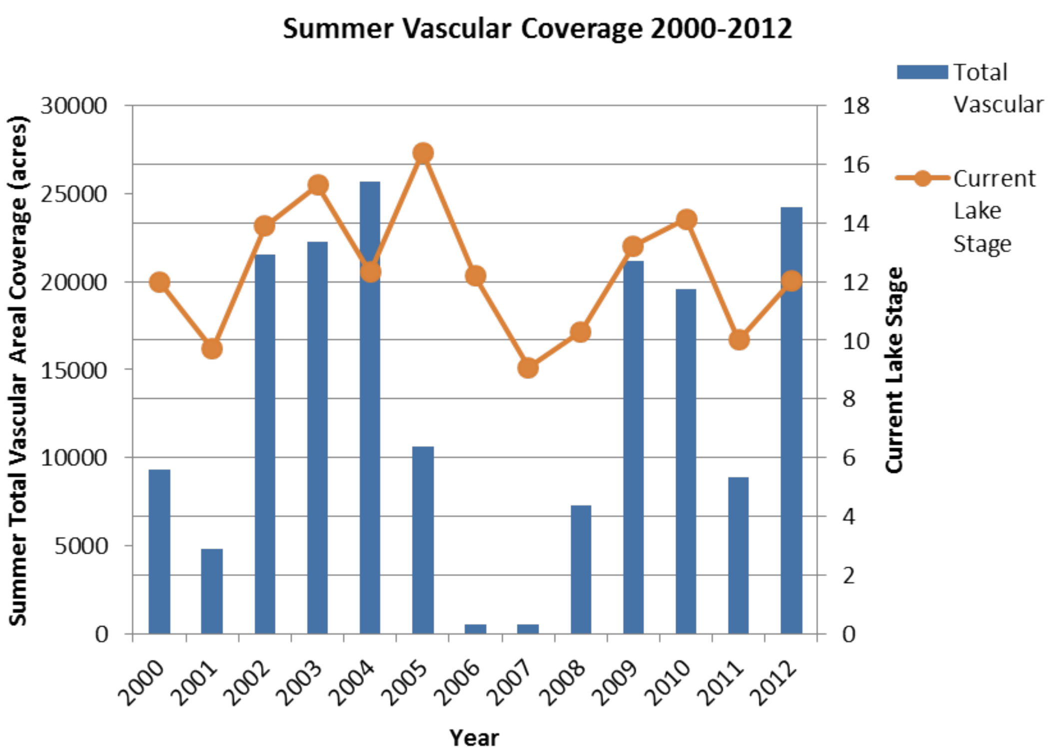


Figure 5-9. The relationship between summer vascular SAV coverage (in acres) and lake stages during the month of annual SAV mapping (July or August) for 2000–2012.

Results

The amount of current mean nearshore total SAV coverage was almost twice that of the previous five-year mean and almost achieved the performance measure target. Vascular nearshore mean SAV coverage was almost six times greater than the previous five-year mean coverage and was greater than the performance measure target.

RESERVOIR STORAGE MODELING SCENARIOS AND ECOLOGICAL INDICATORS

Methods

To evaluate the potential for ecological benefits provided by additional water storage capacity north of Lake Okeechobee, the 1965–2005 period of record (POR) lake stage hydrograph, as simulated using RESOPS, operating according to the current Lake Okeechobee operating schedule (2008 LORS) and associated Lake Okeechobee adaptive protocols (SFWMD 2010), was used for this analysis. The frequency and amplitude of changes in lake stage that might result from adding additional northern watershed storage capacity were examined. A monthly lake stage time step was used and three reservoir storage scenarios were evaluated and compared to the 2008 LORS baseline condition hydrograph (ECB); 500,000 ac-ft (500k) and one (1mil) and two million (2mil) ac-ft of storage. These storage scenarios were used because they are considered to be within a feasible reservoir construction size range. Smaller and larger reservoir storage scenarios were not included in this analysis because smaller reservoirs have previously been evaluated (USACE and SFWMD 2004, 2007) while reservoirs larger than 2mil are likely too large and expensive to receive consideration. For each storage scenario, single compartment reservoirs, approximately 32,500 acres (500k), 64,900 acres (1mil) and 130,000 acres (2mil) in size were used, with each having an average depth of 15 feet and an inflow/outflow capacity equal to the reservoir storage capacity in ac-ft per month. It would therefore take a month to completely fill or drain the reservoir in each storage scenario. A key simulation constraint was that water could only flow into or out of the reservoir in a downstream direction and during periods of excessively high lake stages, water could not be pumped upstream from the lake into the reservoir. Therefore, excess water could only be captured as it flowed towards the lake. Excess water in the lake can only be released under the current management protocol of releases to the St. Lucie and Caloosahatchee rivers, or south into the Everglades. The 12- to 16-foot stage envelope has been used as the lake stage performance measure and was used in these model runs to prevent a significant portion of the stage envelope from being cut off when the minimum and maximum ecologically beneficial stages (12.5 and 15.5 feet) were converted to continuous stage and daily stage values (Otero, personal correspondence). The 12 to 16 foot stage envelope also was used during the model runs as the target to trigger diversion of water into and out of the northern reservoir, with water diversion commencing when lake stage was within 0.25 feet of the top and bottom of this stage envelope. The same evaluation of the actual POR hydrograph was also performed to compare how the historical POR lake stage varied relative to that generated by the model under the 2008 LORS schedule both with and without varying additional volumes of watershed storage.

Pearson and Spearman correlations were conducted with log-transformed lake stages for the month the ecological indicator data were collected, the previous month, the previous two, three, and six months, and the previous year and two-year periods. For all ecological indicators, only the strongest statistically significant correlations ($p \leq 0.05$), generally Pearson correlations, were considered to be significant lake stage-ecological indicator relationships. Performance scores were developed for each of the ecological indicators, based on these statistically significant relationships of long-term Lake Okeechobee ecological monitoring data set correlations with lake stages for each data set POR.

However, in the case of wading birds, since the most recent data set is quite small, scoring was based on the data and results reported in David (1994a), for the data collection that occurred between 1977 and 1988. The resultant occurrence of each individual point score for each ecological indicator was then tallied over the POR and the percentage of time for each point occurrence is presented. Additionally, existing data for each ecological indicator was used to develop associated abundance ranges for each point value, making it possible to evaluate how abundances might have varied among the model run scenarios.

Results

Among the three reservoir storage and ECB scenarios, there were more ecological benefits noted at the lower end of the 12- to 16-foot stage envelope and a smaller range among the upper end of the same stage envelope used as the ecologically beneficial stage envelope for standard performance measure scores (**Table 5-2**). Additionally, Lake Okeechobee would reside in the ecologically beneficial stage envelope 6% (500k), 11% (1mil) and 17% (2mil) more often over the POR as compared to the ECB (**Figure 5-10**).

Table 5-2. ECB and reservoir storage scenario scores for each model run, exceeding the lower (12-foot) and upper (16-foot) lake stage envelope. Higher scores are considered more ecologically beneficial.

Scenario	Lower Lake Stage Score	Upper Lake Stage Score
ECB	27	79
500k Reservoir	43	83
1mil Reservoir	54	86
2mil Reservoir	64	89

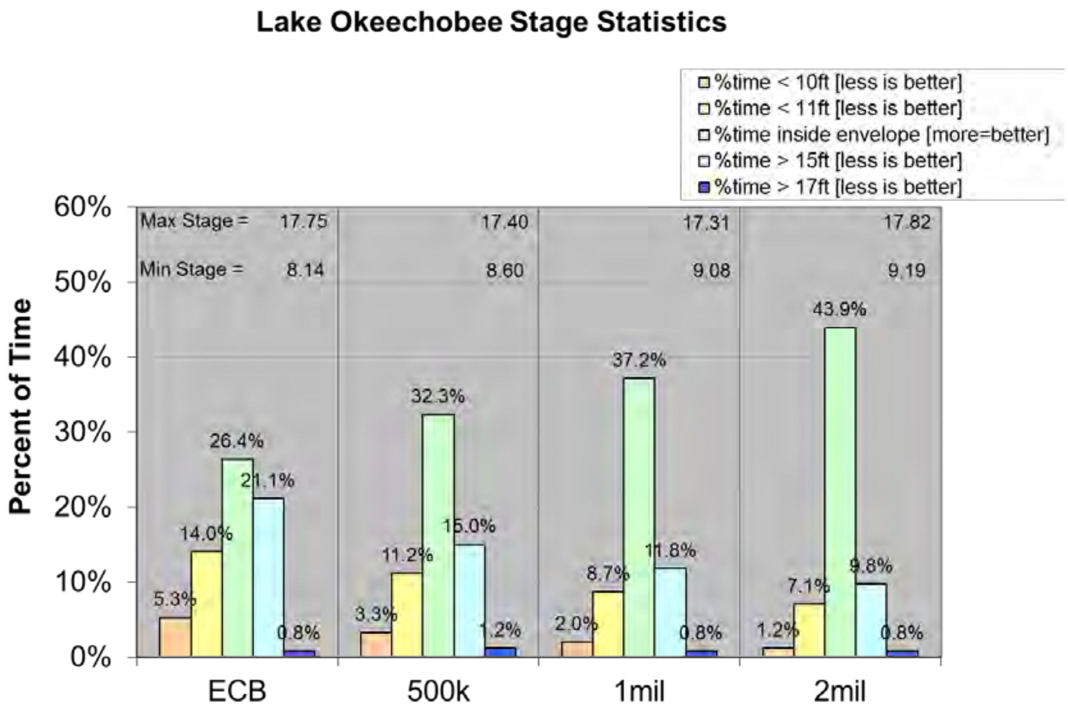


Figure 5-10. Percent of the POR that Lake Okeechobee resides in the ecologically preferred stage envelope (inside envelope) and above and below various lake stages for each of the storage scenarios, as indicated in the legend. The maximum and minimum lake stages for each storage scenario are also listed above the bars. The percent of time not shown is time the lake was between 11 and 12 feet, due to the overlap between percent time > 15 feet and percent time > 17 feet and the percent time that the lake was either above the top or below the bottom of the seasonally variable inside envelope lake stages.

Among the four model run scenarios, the 2mil storage scenario would have resulted in the generally lowest lake stages while the highest lake stages are those of the POR actual stage hydrograph (**Figures 5-11** through **5-18**). The lake operation schedules prior to the implementation of 2008 LORS generally kept the lake at higher lake stages. For example, the Water Supply/Environmental (WSE) Operations schedule in effect from July 2000 to April 2008 called for managing the lake to be one foot higher than under 2008 LORS (SFWMD 2010). Therefore, it is not surprising that the POR actual stage hydrograph was higher than the ECB and storage reservoir projected hydrographs. For the majority of the 41-year POR, the differences in lake stages among the model runs appears to be small (e.g. < 1 foot). However, there are periods where differences in lake stages between the POR actual hydrograph, ECB and the 2mil model runs approached two feet. When the lake was near the lower end or below the bottom of the ecologically beneficial stage envelope, the lake stages under the ECB scenario were often lower than those with reservoir storage, while the reverse condition, with lake stages under the ECB scenario often being higher than those with reservoir storage, would have existed when the lake was near the top of the ecologically beneficial stage envelope.

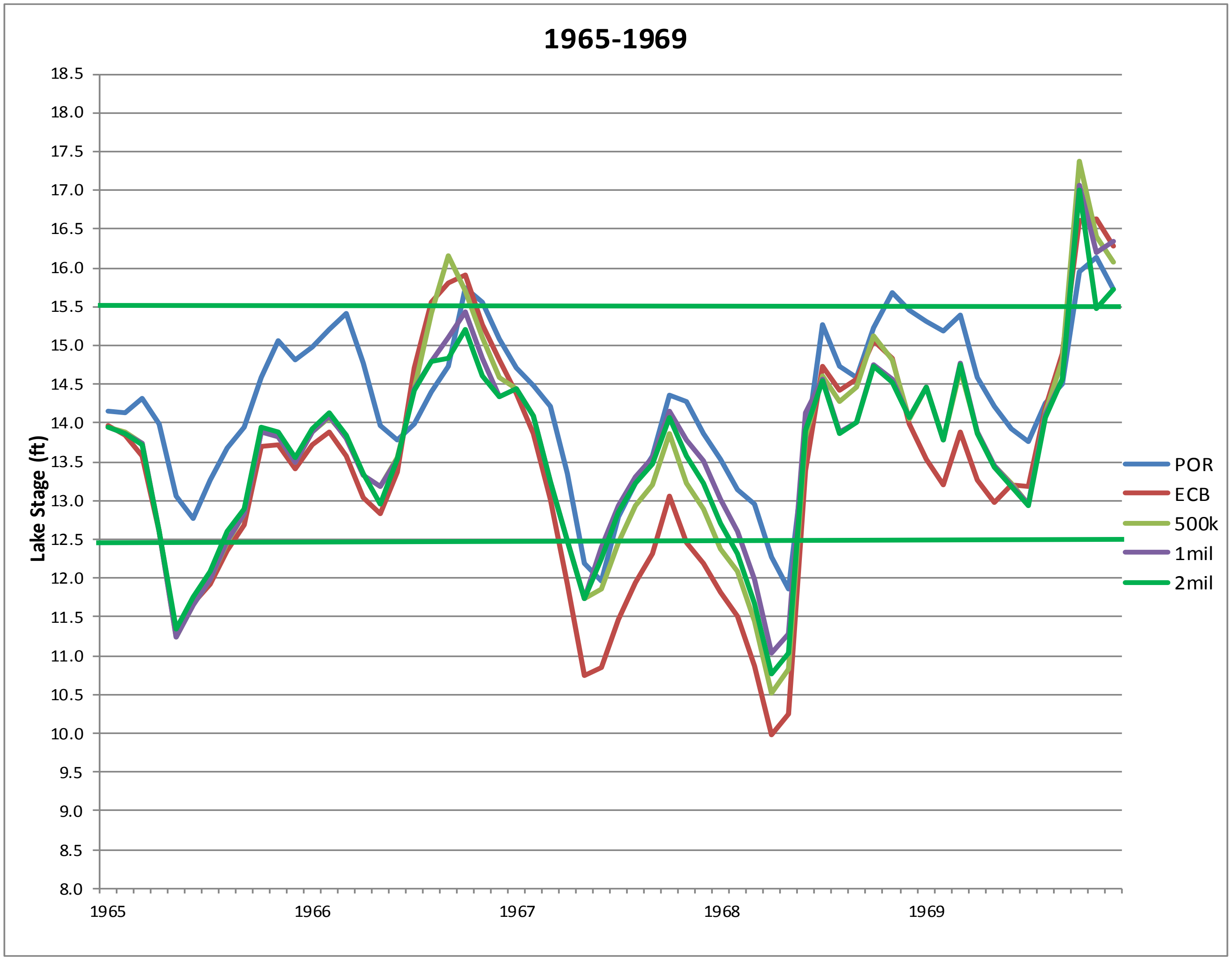


Figure 5-11. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 1965–1969. The green horizontal lines denote the ecologically beneficial stage envelope.

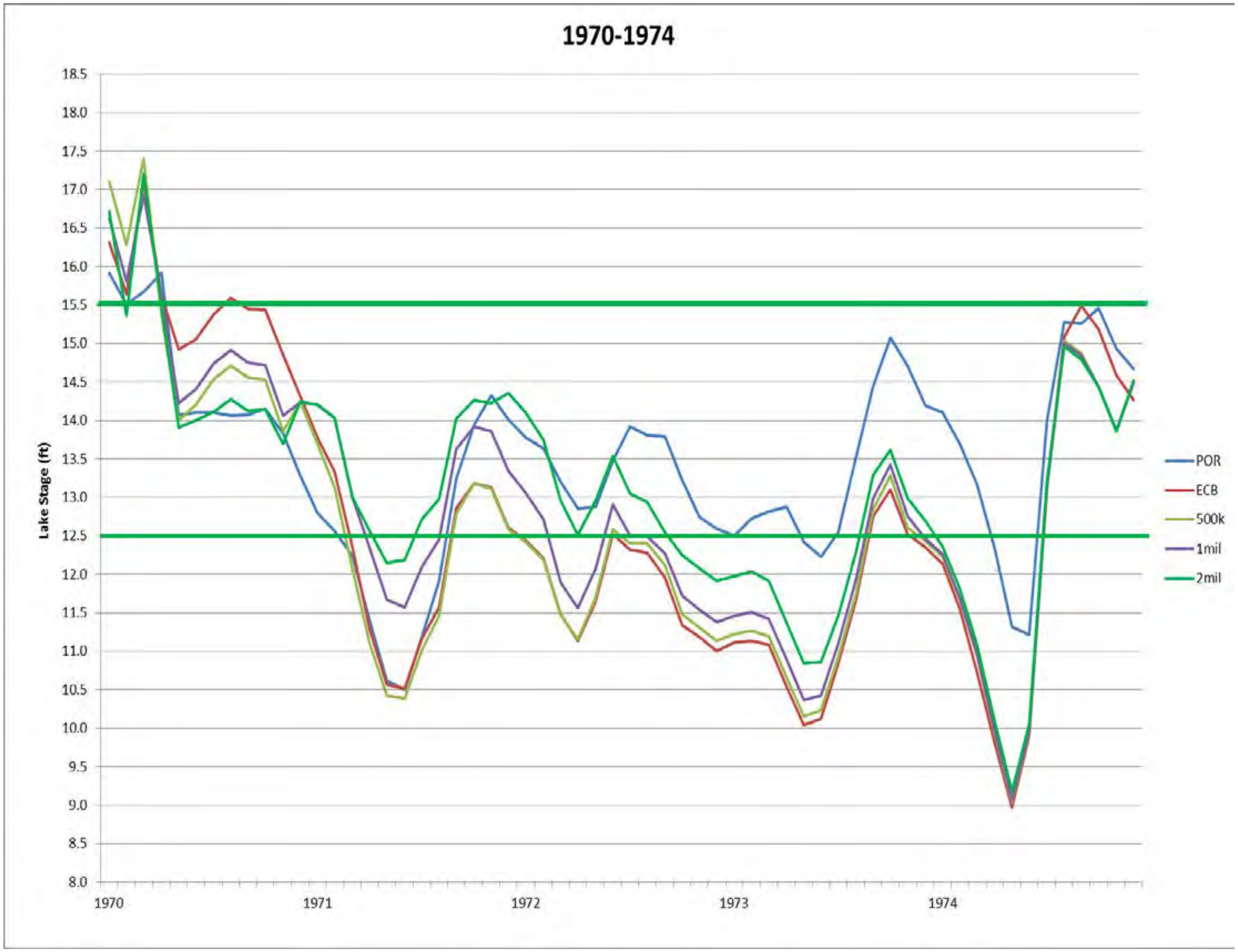


Figure 5-12. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 1970–1974. The green horizontal lines denote the ecologically beneficial stage envelope.

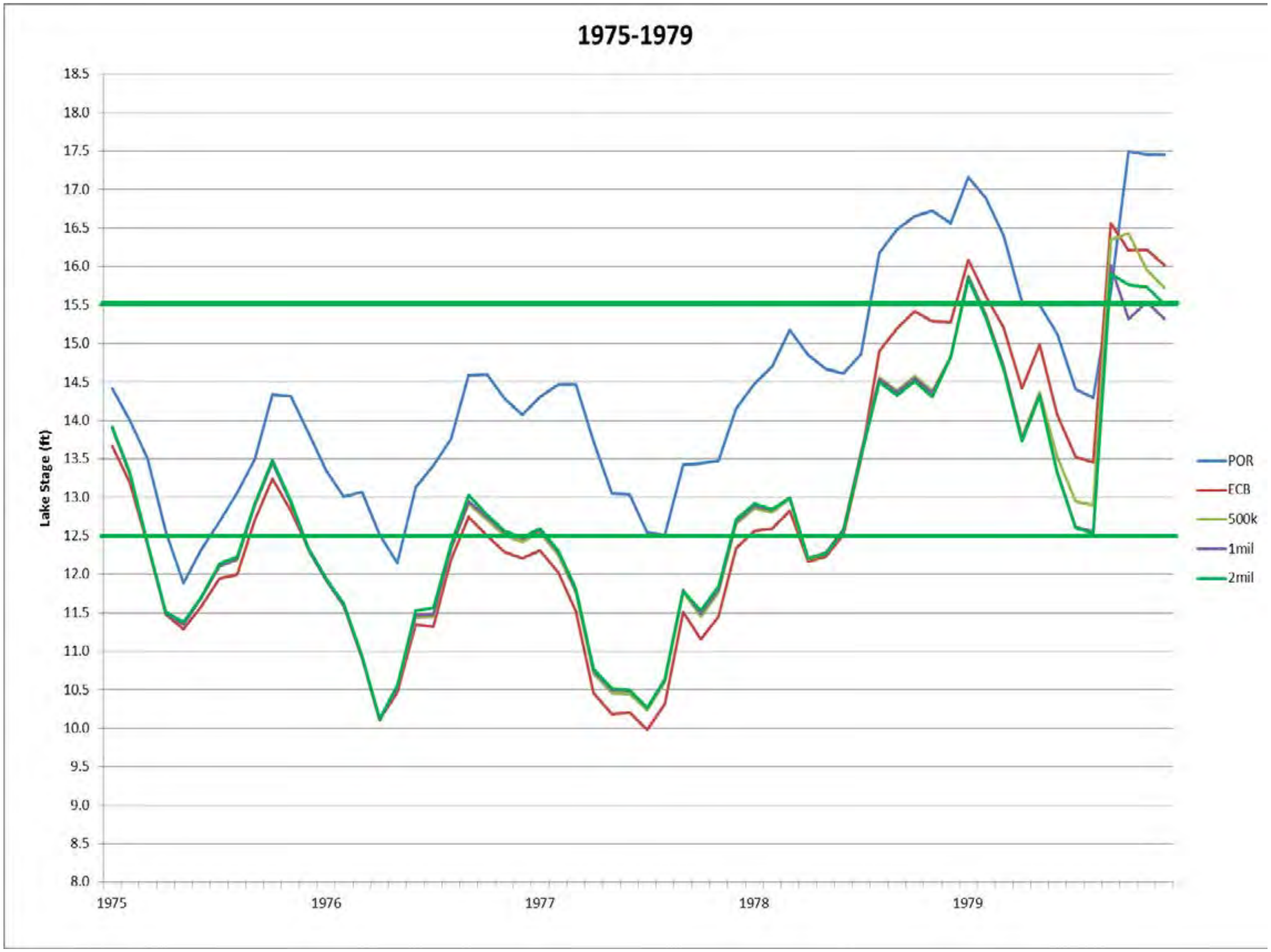


Figure 5-13. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 1975–1979. The green horizontal lines denote the ecologically beneficial stage envelope.

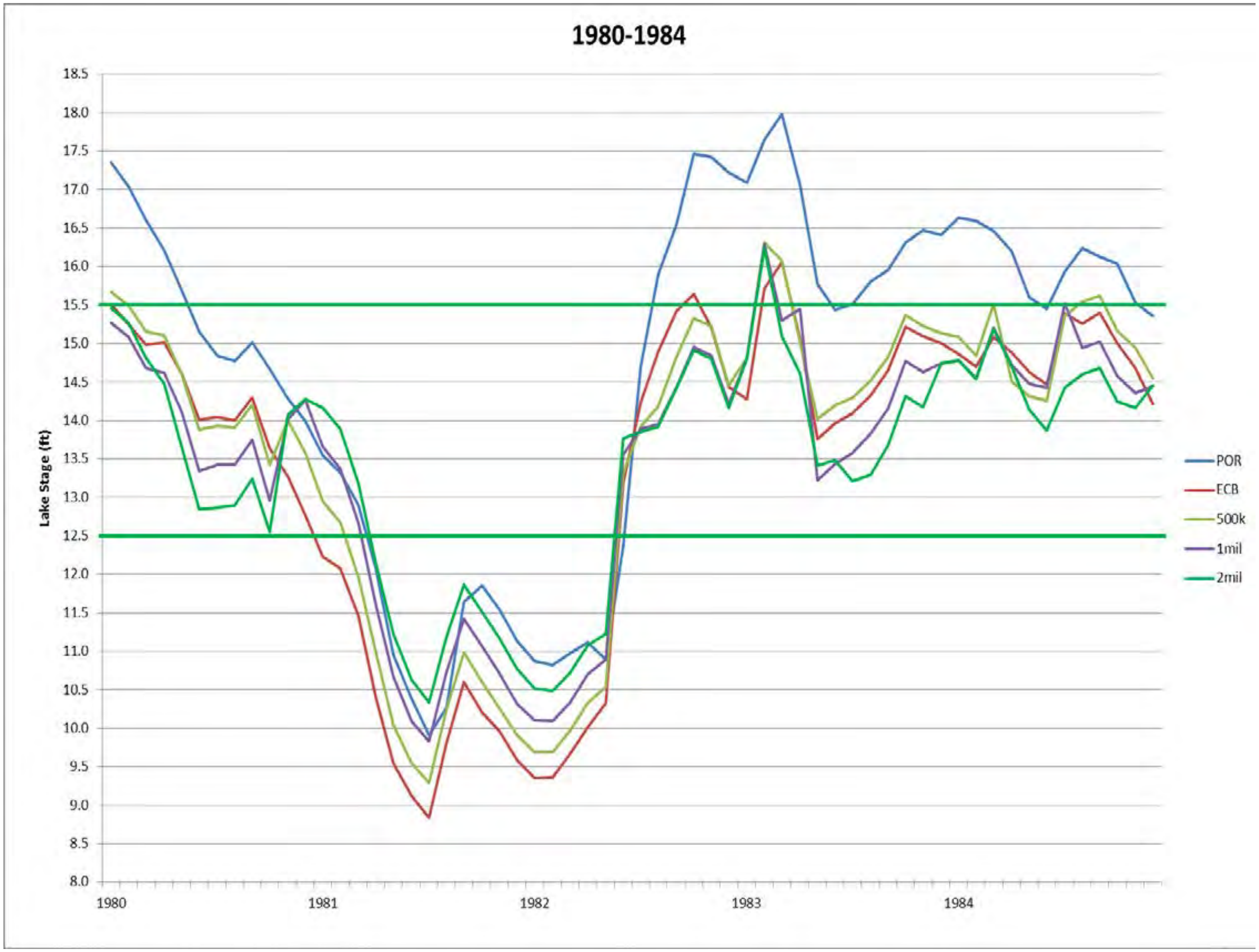


Figure 5-14. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 1980–1984. The green horizontal lines denote the ecologically beneficial stage envelope.

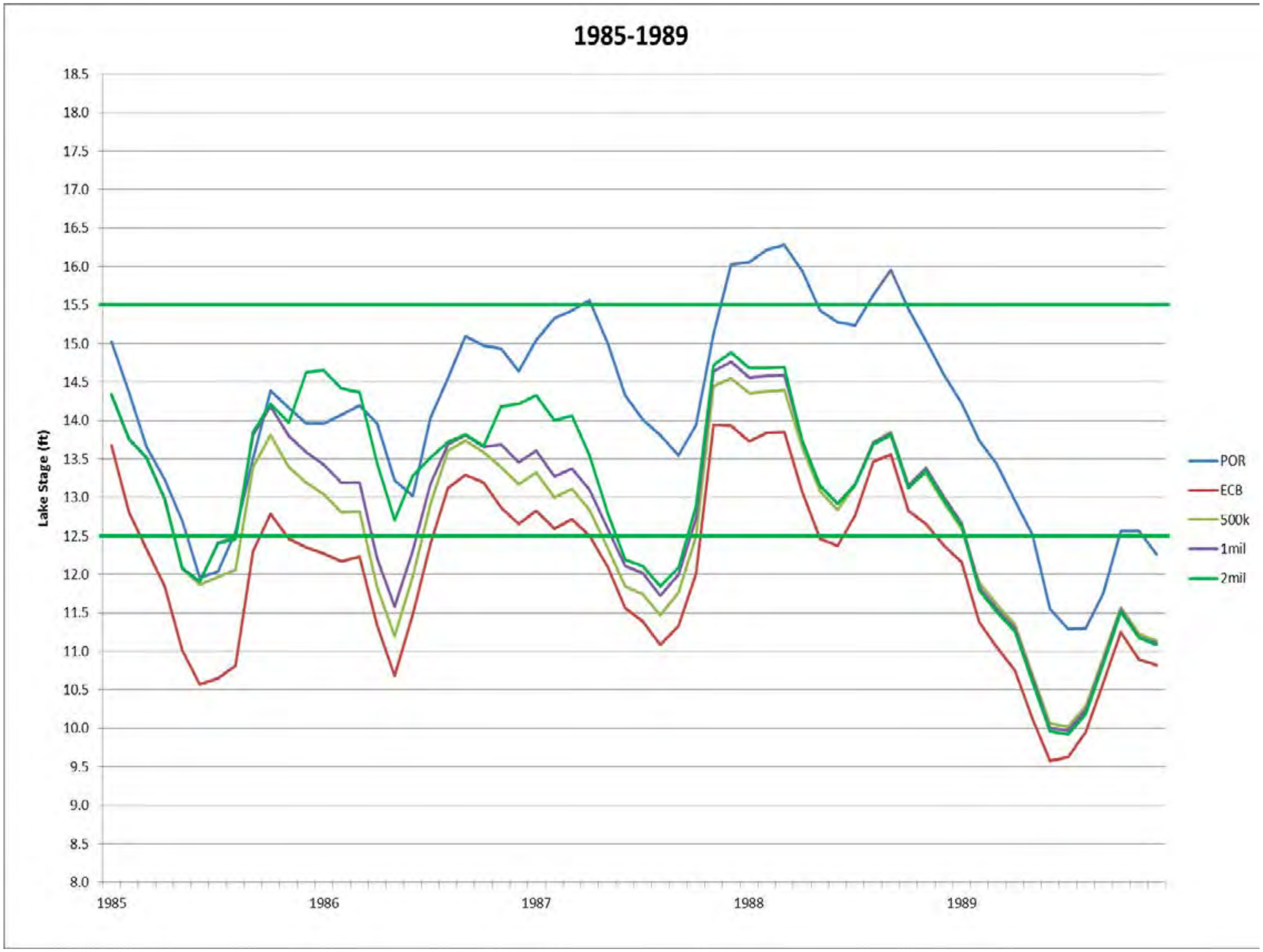


Figure 5-15. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 1985–1989. The green horizontal lines denote the ecologically beneficial stage envelope.

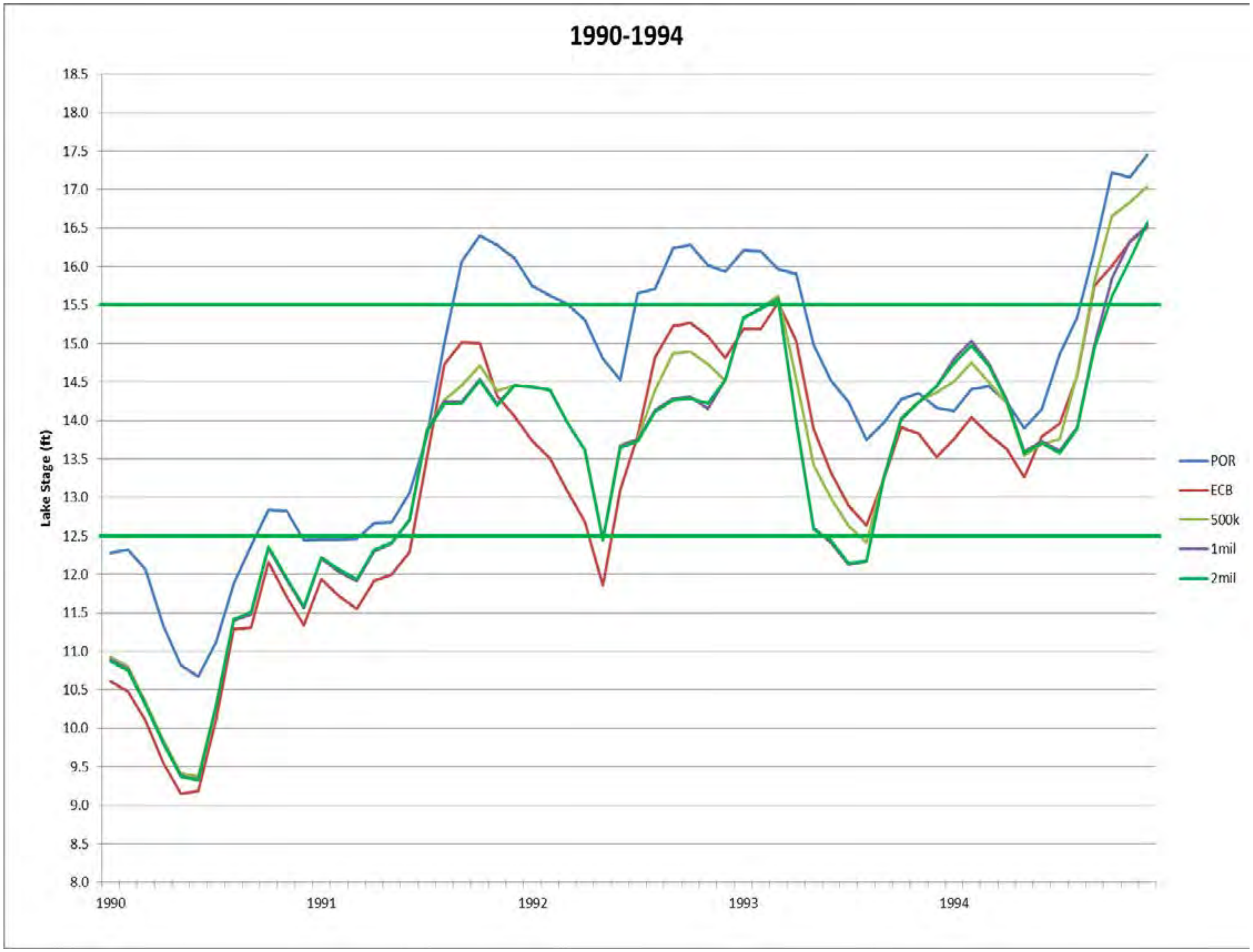


Figure 5-16. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 1990–1994. The green horizontal lines denote the ecologically beneficial stage envelope.

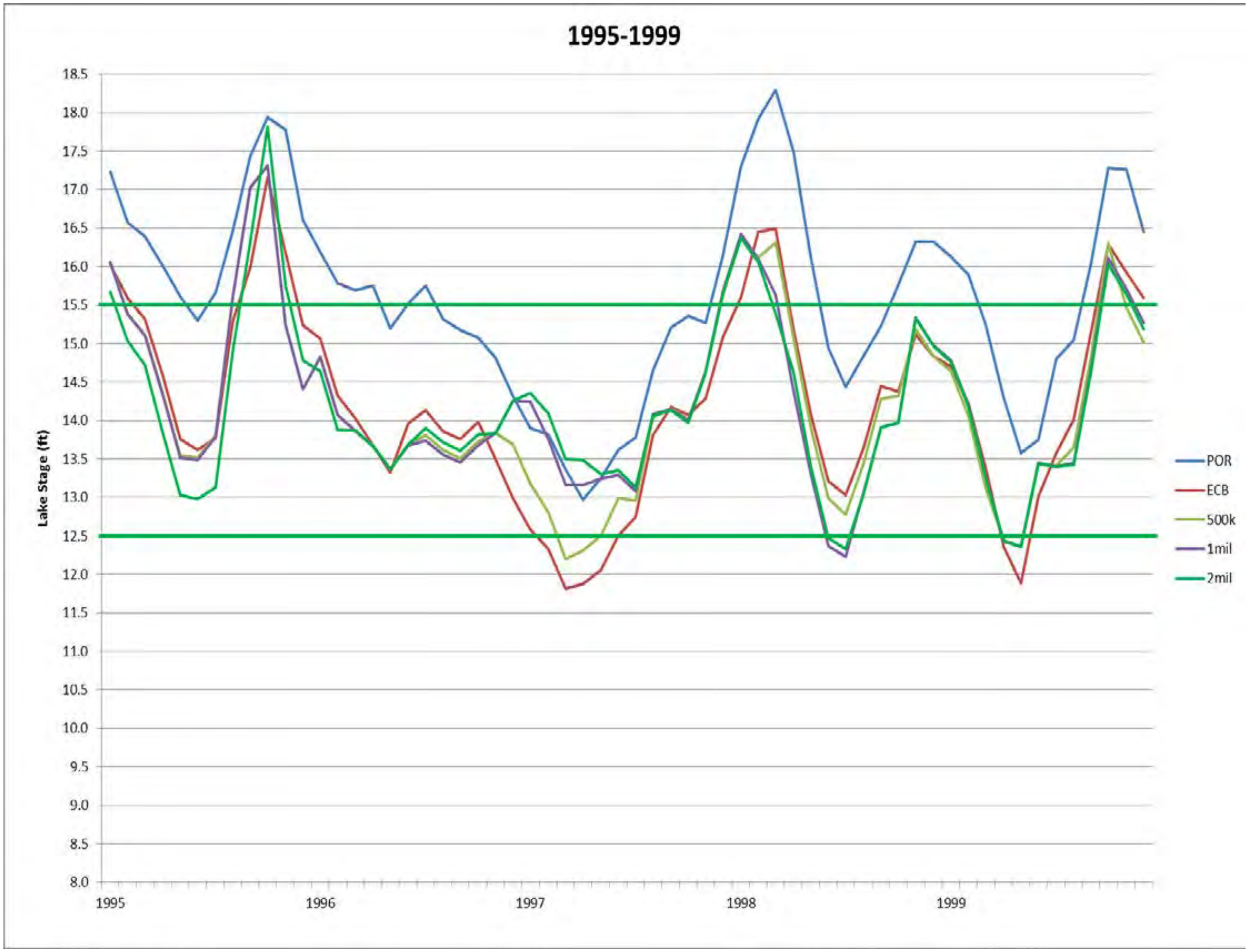


Figure 5-17. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 1995–1999. The green horizontal lines denote the ecologically beneficial stage envelope.

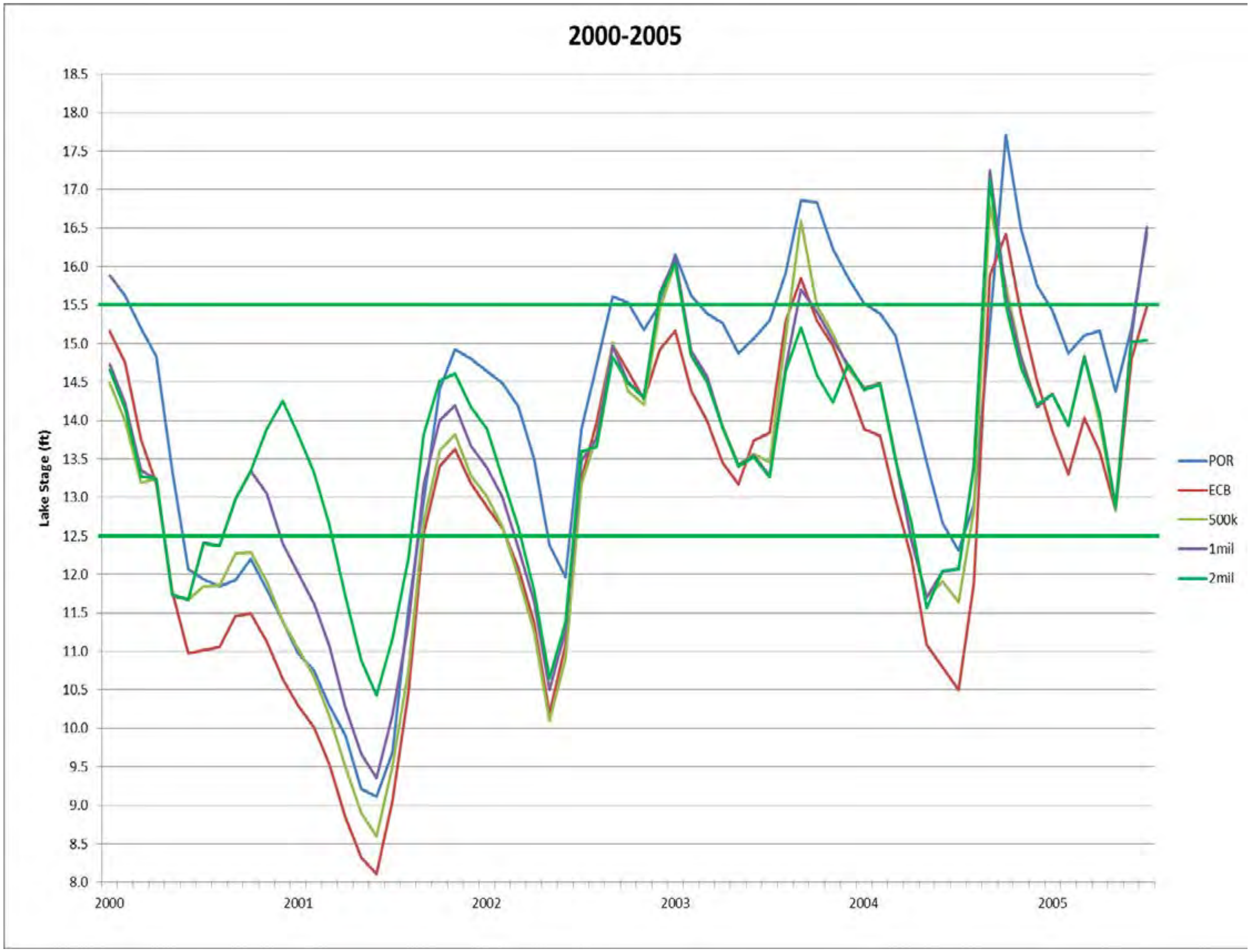


Figure 5-18. Lake stages (feet NGVD) generated by the four model run scenarios and two sets of model runs and the POR actual lake stages for 2000–2005. The green horizontal lines denote the ecologically beneficial stage envelope.

The model runs indicate that that the 2mil reservoir was at maximum capacity roughly 2% of the time, while the 1mil and 500k reservoirs were at maximum capacity approximately 5% and 10% of the time, respectively (**Figure 5-19**). These percentages equate to approximately 7 times (2mil), 12 times (1mil), and 17 times (500k) over the 41-year POR (**Figure 5-20**). The 500k reservoir was projected to be at capacity most often; every 1 to 3 years except during 1972–1978 and 1985–1991, when extended droughts resulted in none of the reservoirs being at capacity. The 1mil reservoir was projected to be at capacity for approximately 3 months to as long as roughly 1.5 years for each event and was projected to be at capacity at a similar frequency as that projected for the 500k reservoir. Conversely, the 2mil reservoir was projected to be at capacity for the shortest duration and lowest frequency, approximately 2 to 4 months per event, with each event occuring between every 2 years and up to 14 years (**Figure 5-20**). Even at the "50% of time storage exceeded" point on the curve for each reservoir, the 2mil reservoir is projected to store roughly only 300,000 ac-ft of water that otherwise would go into Lake Okeechobee, while the 1mil and 500k reservoirs are projected to store approximately 155,000 and 16,000 ac-ft, respectively. These 50% of time storage exceeded point projections are equivalent to only about 3% (500k) and 15% of 1mil and 2mil reservoir capacities.

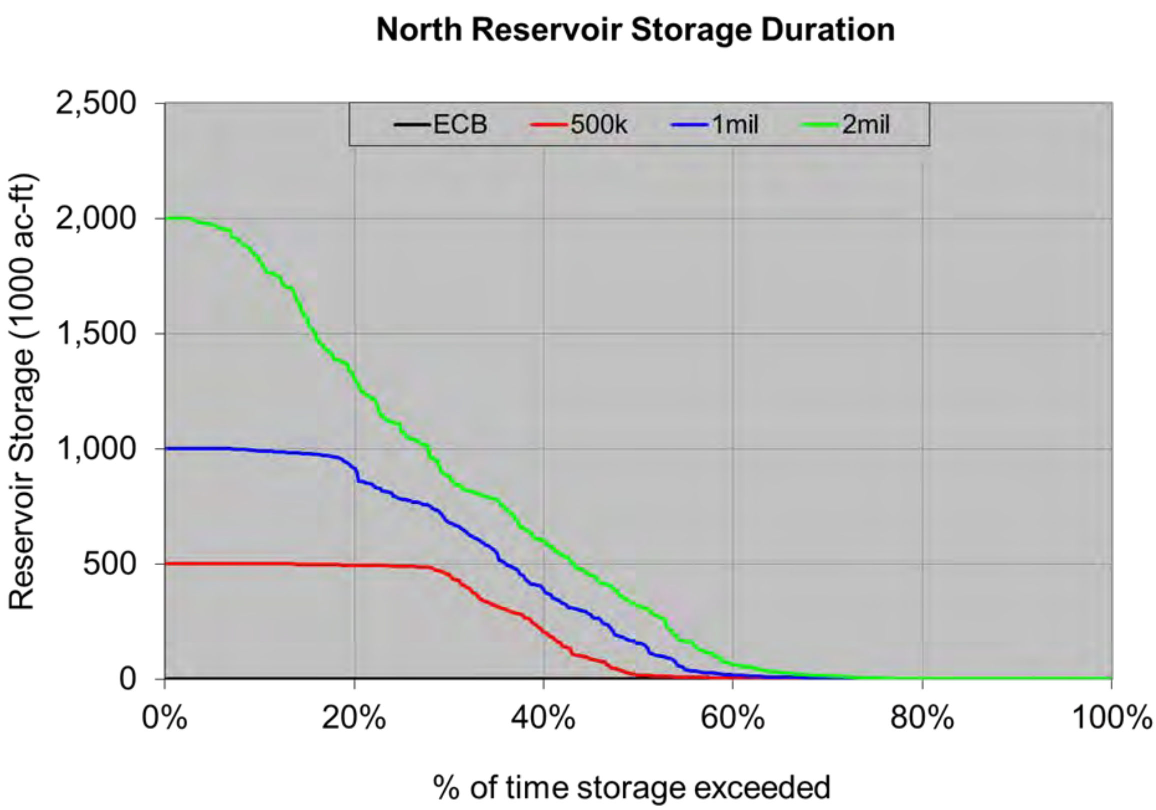


Figure 5-19. Reservoir water storage duration curves over the POR. The vertical axis displays the projected amount of water stored and the horizontal axis represents the percentage of time over the POR for the storage volumes.

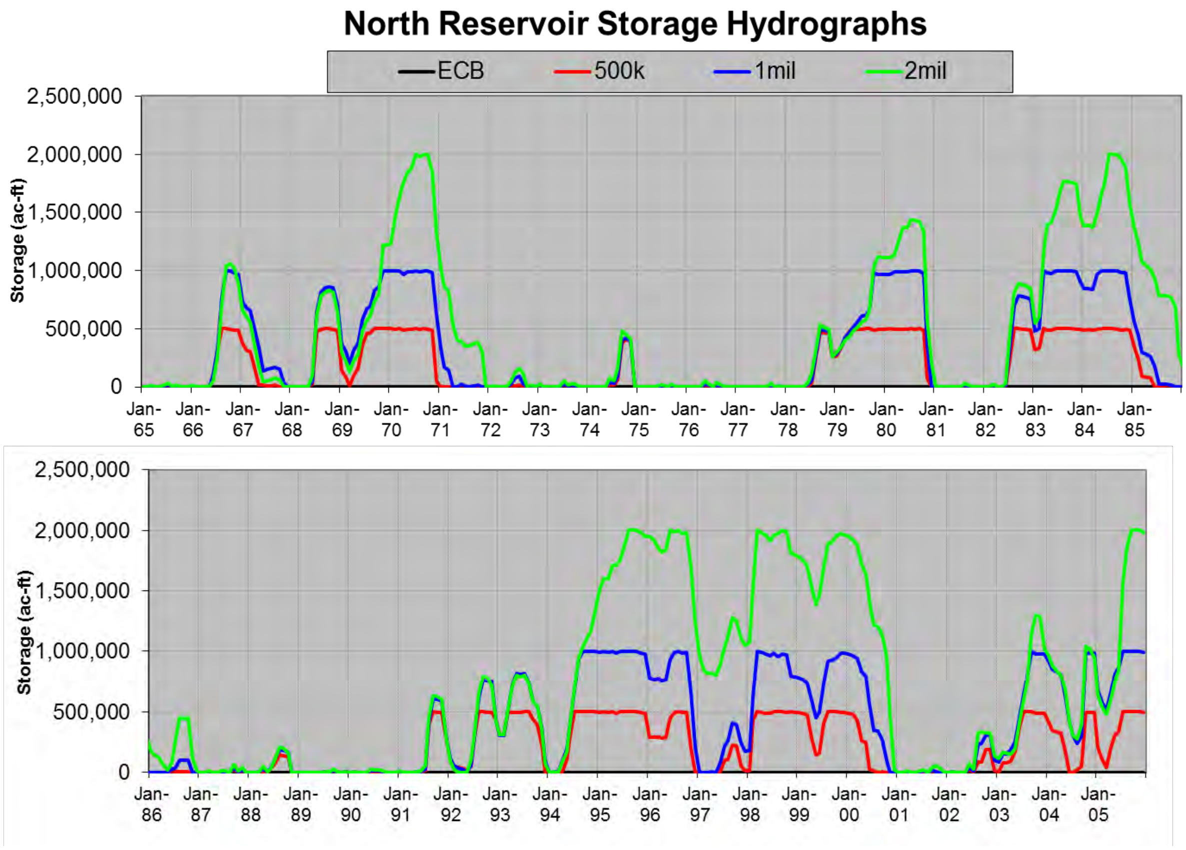


Figure 5-20. Reservoir storage scenario hydrographs over the POR. The vertical axis displays the projected amount of water stored and the horizontal axis displays the 41-year POR.

Higher percentage occurrences of larger point scores over the POR indicate better ecological conditions and higher abundances for the ecological indicators, although in the case of the summer cyanobacteria, this equates to ecological conditions that are less favorable for bloom potential and lower abundances (**Figures 5-21** through **5-24**). Based on our scoring criteria, summer cyanobacteria abundances, and therefore bloom potential, for the POR actual hydrograph, was substantially higher (e.g., a higher percentage of 0 point scores) relative to the ECB and three storage scenario model runs (**Figure 5-21**).

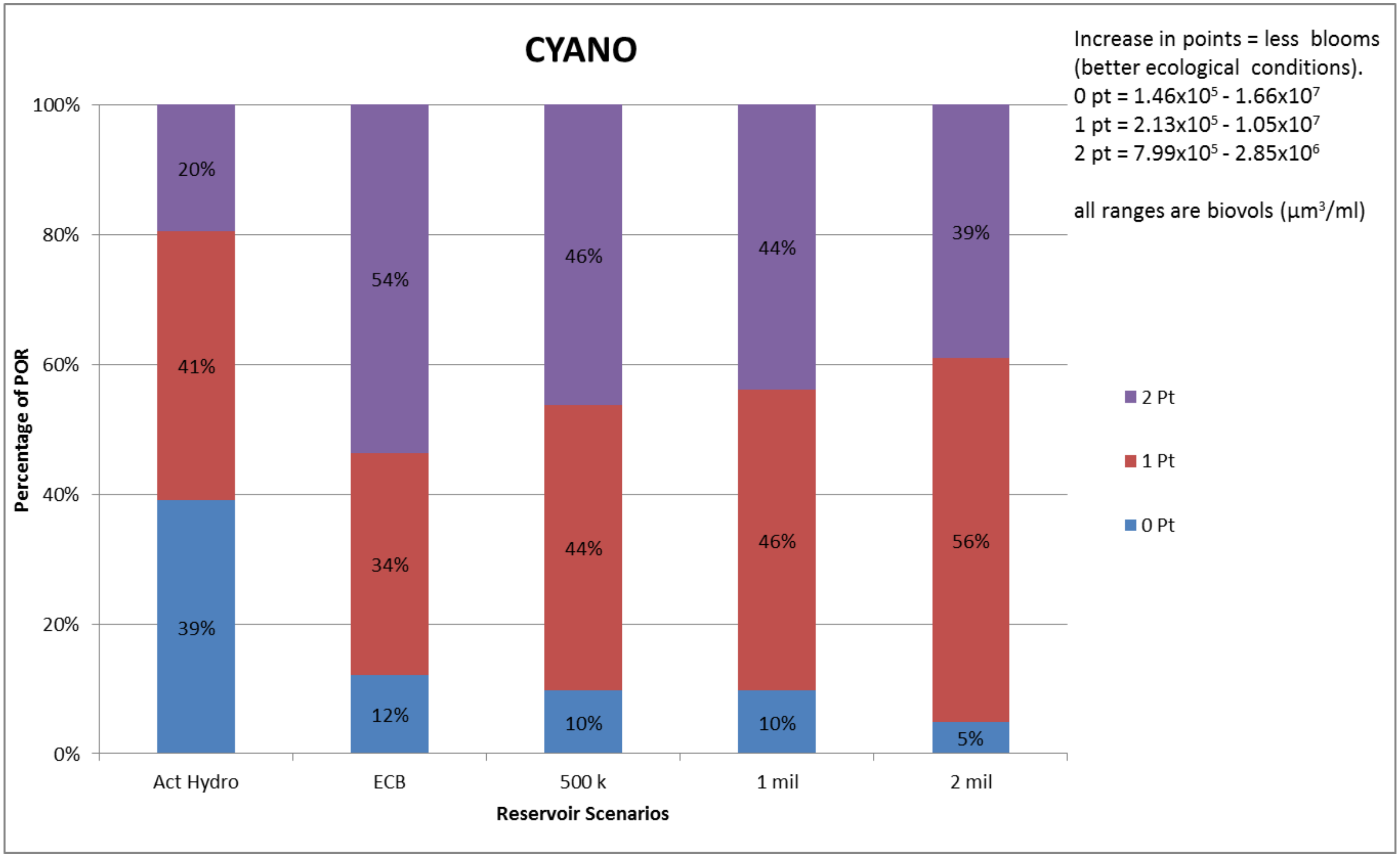


Figure 5-21. Percentage of time over the 40-year POR for the individual point distribution and range of estimated abundances for each point value for summer cyanobacteria. The model run scores are for POR actual hydrograph, ECB, and reservoir storage (500k, 1mil, and 2mil) scenarios. Higher scores for cyanobacteria equates to less abundance and bloom potential. (Note: µm^3/ml; Act Hydro – actual hydrology; biovols – biovolumes; cyano – cyanobacteria; and pt – point.)

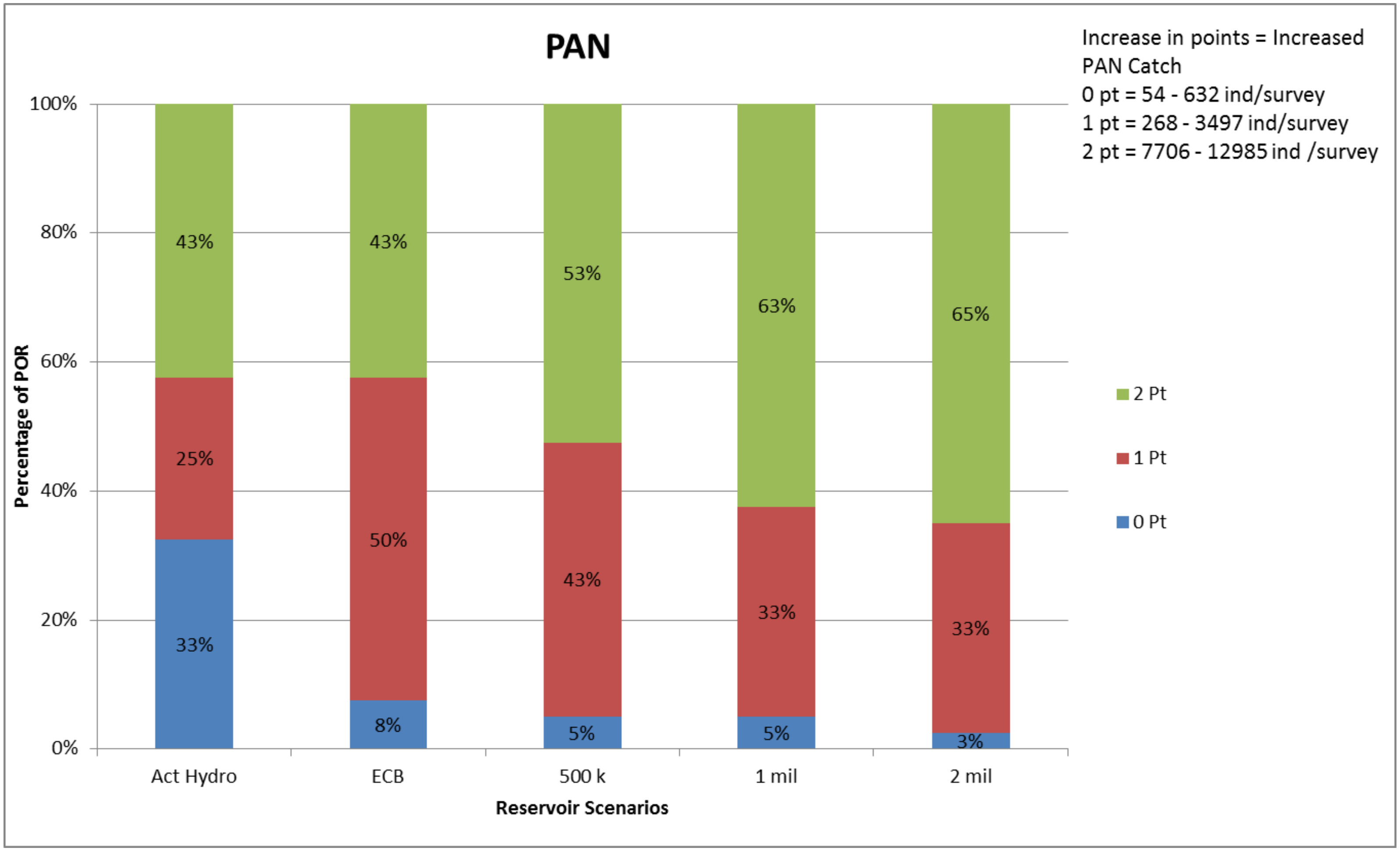


Figure 5-22. Percentage of time over the 40-year POR for the individual point distribution and range of estimated abundances for each point value for pan fish (PAN). The model run scores are for POR actual hydrograph, ECB, and reservoir storage (500k, 1mil, and 2mil) scenarios. (Note: Act Hydro – actual hydrology; ind – individual; PAN – pan fish; and pt – point.)

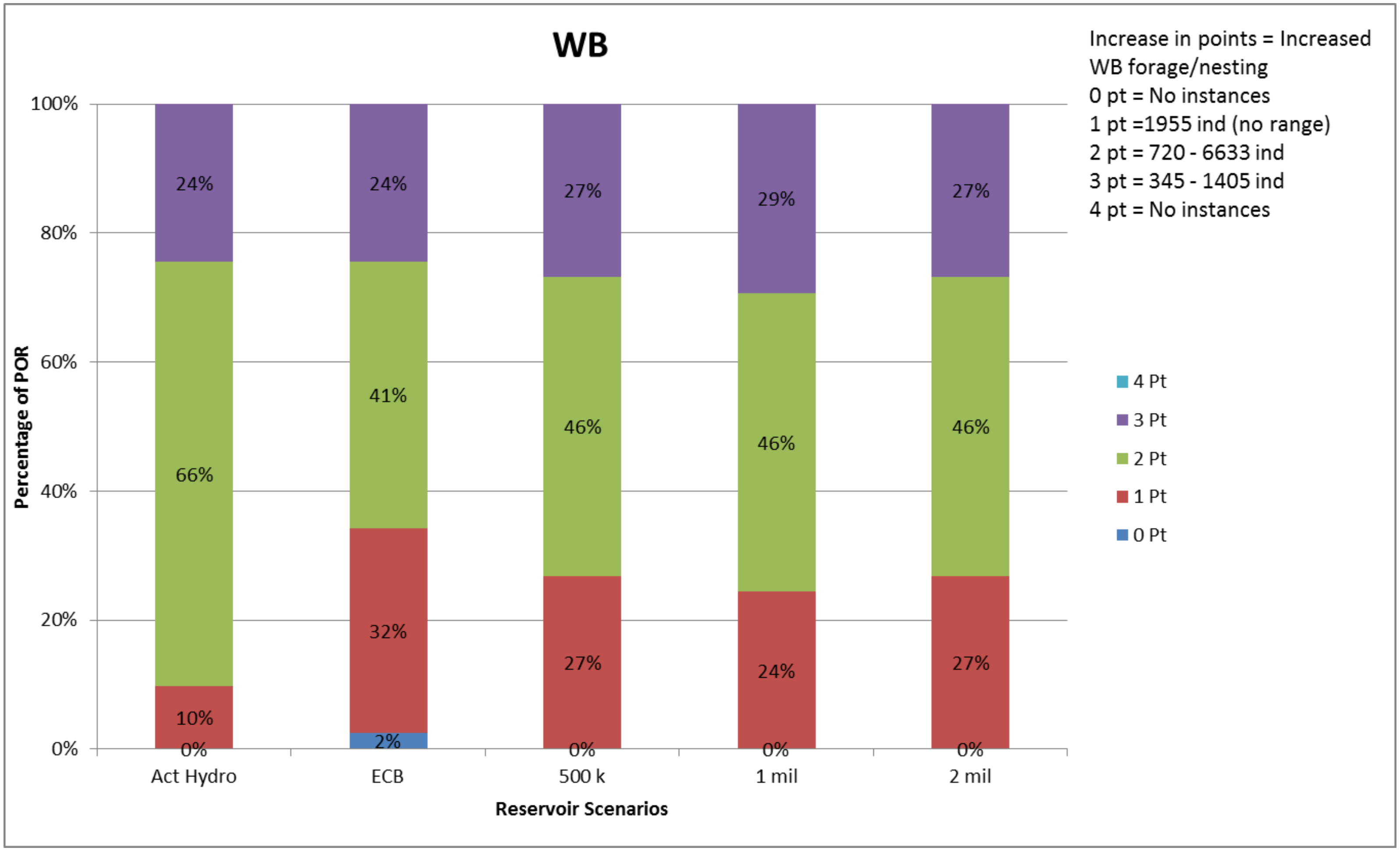


Figure 5-23. Percentage of time over the 40-year POR for the individual point distribution and range of estimated abundances for each point value for wading birds. The model run scores are for POR actual hydrograph, ECB, and reservoir storage (500k, 1mil, and 2mil) scenarios. (Note: Act Hydro – actual hydrology; ind – individual; pt – point; and WB – wading birds.)

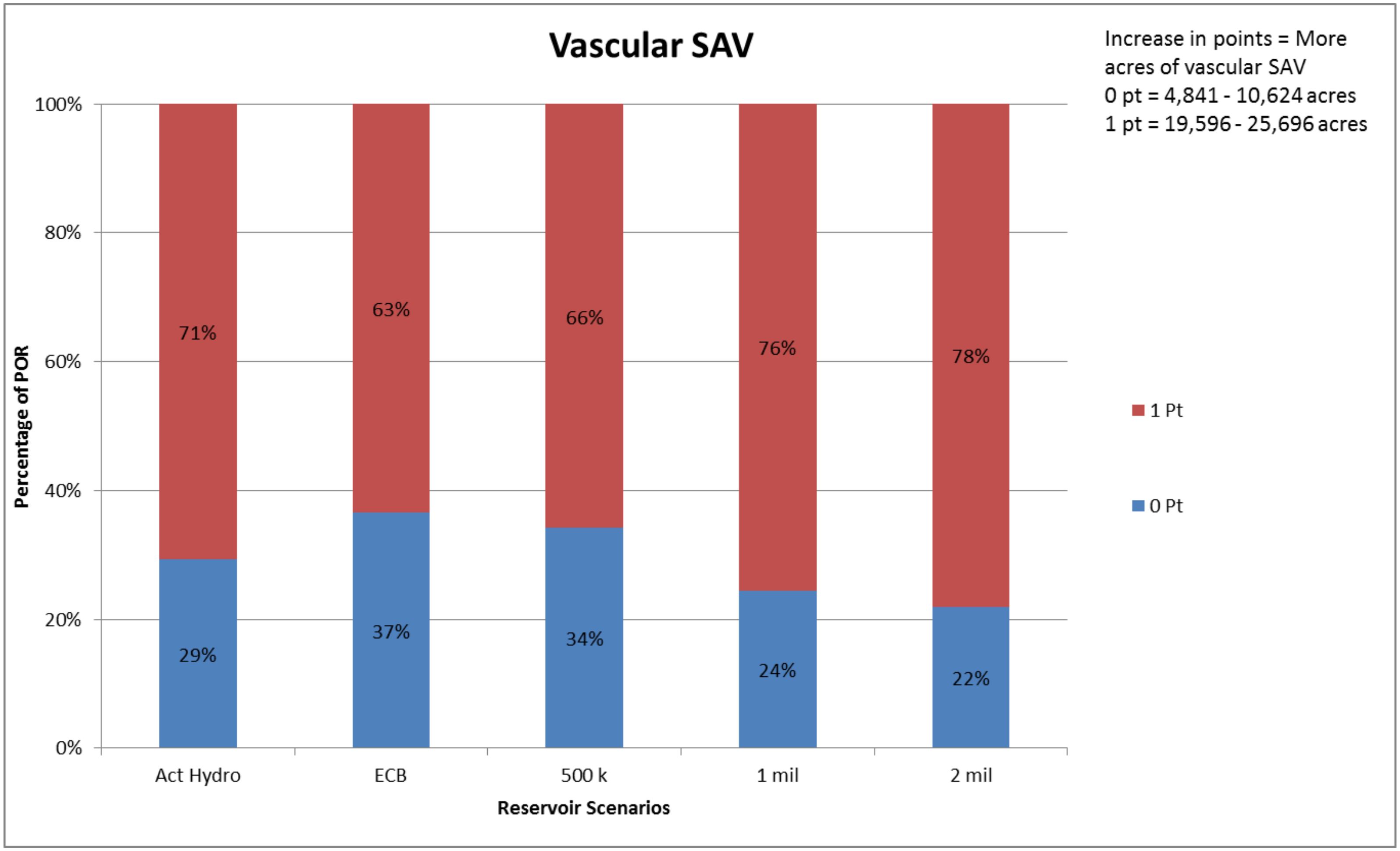


Figure 5-24. Percentage of time over the 40-year POR for the individual point distribution and range of estimated abundances for each point value for SAV. The model run scores are for POR actual hydrograph, ECB, and reservoir storage (500k, 1mil, and 2mil) scenarios. (Note: Act Hydro – actual hydrology and pt – point.)

The amount of potential ecological benefits in terms of increased biotic abundances appeared to be similar for most of the ecological indicators. The percentage occurrence of tallied point scores over the POR also indicate that potential ecological benefits were enhanced for all reservoir storage scenarios and the cyanobacteria bloom potential appeared to be reduced from the addition of any of the storage reservoirs. The largest potential ecological benefits appear under the 2mil storage scenario for cyanobacteria, SAV and pan fish catch. This is perhaps not surprising in the case of SAV and pan fish catch since the model runs were designed to try and keep the lake within the ecologically beneficial stage envelope as much as possible (which was originally based on our understanding of SAV dynamics in Lake Okeechobee), which would be most easily accomplished with northern watershed storage in place using the 2008 LORS operations schedule plus adaptive protocols. In the case of cyanobacteria, the model runs resulted in less time with spring lake stage > 14 ft, which suggests lower summer cyanobacteria abundances and bloom potential. Model runs also took advantage of the operational flexibility built into 2008 LORS to adjust lake discharges to try and optimize reservoir operations.

Conversely, wading birds appear to benefit most from the POR actual hydrograph lake stage regime. It also appears that none of the model runs using 2008 LORS operational parameters provides large ecological benefits to all of the ecological indicators, although the three north of the lake reservoir storage scenarios appear to provide similar or increased ecological benefits for pan fish, wading birds and SAV abundances, relative to those provided by and estimated for the ECB. However, in the case of cyanobacteria and pan fish, the largest increase in potential ecological benefits appears to occur with the ECB scenario. When comparing the three reservoir storage scenarios, the potential ecological benefits appear to be similar under all scenarios for wading birds, while the additional storage provided by the 2mil storage reservoir appears to provide additional potential ecological conditions for reduced cyanobacteria while the 1mil and 2mil storage reservoirs appear to provide additional potential ecological benefits for SAV and pan fish catch.

DISCUSSION

Changes in Biotic Components and Water Quality Performance Measures

During the period of WY 2009–WY 2013, Lake Okeechobee was in or below the ecologically beneficial stage envelope for practically the entire reporting period. The lake is anticipated to remain within the ecologically beneficial stage envelope more often once all of the restoration projects that will influence lake stages are completed. Therefore, how the lake biotic components responded to these generally lower lake stages since WY 2009 may provide a glimpse as to how the lake might appear ecologically once CERP projects are completed. It appears that several biological attributes have responded positively to the overall lower lake stage regime.

Mean annual summer total SAV areal coverage has increased by 46% during FY 2009–FY 2013 relative to the previous five-year mean annual coverage (see **Table 5-1**) and is within < 3,000 acres of the FY 2000–FY 2004 pre-hurricane mean annual summer coverage of 41,536 acres. Prior to the passage of Hurricanes Frances, Jeanne and Wilma, with a generally deeper water column, there was more inshore open water SAV colonizable habitat; some of which has been replaced over the past six years by emergent plants, expanding the littoral marsh. Nevertheless, it appears that the generally lower lake stages between FY 2009–FY 2013 have been beneficial for SAV recovery from the 2004–2005 hurricanes, which dramatically reduced SAV coverage to 2,965 acres by summer 2006. Concomitant with increased SAV host substrate, periphyton abundance has likewise increased, and combined with a lack of large-scale nearshore algal blooms since 2005, suggests that nutrient competition between SAV, periphyton and phytoplankton may be occurring. Competition for nutrients between periphyton and phytoplankton may indirectly be a primary factor in explaining why despite generally favorable conditions for large-scale phytoplankton blooms due to lower lake stages and increased water column transparency, they have not occurred. Although the algal bloom frequency (as chlorophyll *a* concentrations exceeding 40 micrograms per liter) has increased slightly during the WY 2009–WY 2013 period relative to WY 2004–WY 2008, the prolonged high nearshore turbidity and poor water column transparency values following the hurricanes may have been a primary factor in the lower mean algal bloom frequency during WY 2004–WY 2008.

Other biotic components also appear to have benefitted from the generally lower lake stages during WY 2009–WY 2013. With the recovery of spawning and foraging habitat, fish populations, both nearshore and pelagic, were mostly higher during WY 2009–WY 2013 compared to those during 2005 and 2006. Among the important sport fish taxa, largemouth bass and speckled perch (e.g., black crappie [*Pomoxis nigromaculatus*]) catch rates were the highest during WY 2009 and WY 2013 respectively, relative to rates when the current electrofishing and trawl data collection began in 2005 (Bertolotti et al. 2014). Macroinvertebrate populations likewise were rebounding during WY 2009 from extremely low densities in 2005–2007 (RECOVER 2011). Updated abundance data is anticipated to be available during fall 2013 and will hopefully show a continuation of the trend of increased macroinvertebrate populations, especially the nonpollution-tolerant taxa.

Likewise, when the lake is between 10 and 14 feet during the January–June period, as it was during WY 2009, WY 2011, WY 2012, and approximately the first and second half of that period during WY 2010 and WY 2013, respectively, a greater area of the littoral region appears to be suitable for wading bird foraging. The availability of foraging habitat can be a limiting factor throughout the nesting cycle from initiation to the fledging of chicks (Smith 1995). Wading bird nest effort also can be a useful way to follow historical trends since nesting effort has been the only data consistently gathered (Crozier and Gawlik 2003). The limited data used and overlapped in wading bird abundances among the two and three point scores indicate that using bird abundance data only from nesting colonies may somewhat obscure the relationship between lake stages and wading bird littoral marsh utilization, since birds foraging away from the colonies at the time the surveys were conducted would have been missed. Also, David (1994b) indicated that darker colored taxa were probably underepresented and therefore were not included in survey counts. A potentially more suitable metric to track population trends would be nesting success but those data are extremely limited.

While lake stages during WY 2009–WY 2013 may have somewhat reflected those anticipated after systemwide hydrologic restoration is completed, there were no concomitant large-scale nutrient reductions, despite the completion of a number of nutrient abatement projects in the Lake Okeechobee watershed (Bertolotti et al. 2014). This was reflected in the water quality variables, which failed to meet any of the performance measure targets. While the recent prolonged drought conditions temporarily reduced external nutrient loads, water column nutrient concentrations remained high due to internal nutrient loading. Even with reduced watershed runoff, and an approximate 40% decrease in the five-year mean water column TP concentration, relative to the WY 2004–WY 2008 mean concentration (see **Table 5-1**), TP loading during WY 2009–WY 2013 still exceeded the total maximum daily load target (140 metric tons), by a factor of three. Mud sediment resuspension in the pelagic zone likewise continued, resulting in periodic increases in TP concentrations in the water column. While transport of resuspended sediments and nutrients from the pelagic to the nearshore region appeared to occur much less frequently when lake stages were lower (e.g., < 13 feet), water column TP concentrations in both regions over the past five years still exceeded their targets by a factor of at least two to three. Additionally, the pelagic TN:TP ratio still indicates N-limitation, not the P-limitation anticipated once lake restoration is completed.

While no water quality nutrient performance targets were met during WY 2009–WY 2013, there were observations of areas in the nearshore region over the past several years suggesting that even some nutrient reduction can result in improved ecological conditions in the nearshore region of the lake. At these areas, SAV and associated epiphytes are abundant and water column transparency is high, with a Secchi disc transparency often extending to the lake bottom in water columns up to approximately 1.5 meters in depth. Relatively low TP concentrations (20 to 30 micrograms per liter) have been occasionally documented in these areas, along with reduced phytoplankton abundance and an absence of cyanobacteria dominated blooms. Overall, while recent drought conditions, reduced nutrient runoff, generally lower lake stages, frequent uncoupling of water circulation between the nearshore and pelagic regions, or other factors appeared to contribute to a reduction in pelagic and nearshore region nutrient concentrations, regularly meeting or exceeding the performance measure water quality targets will require nutrient reduction on a large scale.

Storage Scenario Comparisons

Given the much larger potential storage capacity of the 2mil storage reservoir relative to that provided by the 500k storage reservoir, it is somewhat surprising that the ecological benefits provided by the largest reservoir were not consistently larger than those provided by the smaller reservoirs and likewise, did not extend to all of the ecological indicators. The largest reservoir appears to provide additional ecological benefits for reduced cyanobacteria bloom potential and abundance, compared to that provided by the 500k and 1mil reservoir. Conversely, ecological benefits provided by the 2mil reservoir for SAV and pan fish catch may not be substantially better than those provided by the 500k reservoir and about the same as those provided by the 1mil reservoir. The failure of the 2mil reservoir to provide ecological benefits proportional to its size is probably attributable to the fact that this reservoir did not appear to have as much influence on lake stages as might be expected for a reservoir that could theoretically store roughly the equivalent of about four feet of Lake Okeechobee stage volume when at capacity. The smaller than anticipated differences in lake stage hydrographs between the three reservoir model scenarios appears to reflect the reservoir storage duration curve projections produced by RESOPS, which projected less differences among the reservoir scenarios than might be anticipated, given the differences in reservoir sizes.

Modeled lake operations also resulted in water being stored in a northern reservoir no more than 71% (500k) to 77% (2mil) of the time. It therefore appears that the current operational procedures for the lake and any reservoirs projected to be constructed north of the lake for water storage will need to have operational parameters adjusted further, if possible, to obtain significant additional lake ecological benefits. Conversely, the RESOPS stage envelope standard scores, if evaluated as providing ecological benefits proportional to their values, suggest that the 2mil reservoir might provide approximately 2.2 to 2.4 times more ecological benefits compared to the ECB scenario based on the low lake stage (< 12 feet) performance measure scores (see **Table 5-2**). The 2mil reservoir also would have generally reduced the amount of time the lake experienced extremely low lake stages, relative to the ECB, during more extreme droughts (e.g., 1953–1956, 2006–2008) than those that occurred during this POR. For the high lake stage (> 16 feet) performance measure, comparisons between the ECB and 2mil reservoir run scores indicate that the additional ecological benefits would be only about 1.3 times greater if the 2mil reservoir was in place (see **Table 5-2**). While the upper lake stage envelope score comparisons between the model output standard scores and the ecological scores are smaller (1.13–1.30) than those for the low lake stage performance measure, in either case the correspondence is not 1:1. These results suggest that using ecological performance measures specific to key ecosystem components and statistical analyses of actual data likely provides a more realistic comparison of potential ecological benefits then does using the standard performance measure post processing scores. As observed in this modeling exercise, using the model output standard scores to evaluate potential changes in ecological benefits would have resulted in an overestimation of the amount of additional ecological benefits provided by the addition of a 2mil reservoir in the Lake Okeechobee northern watershed. Additionally, ecological indicator scoring refinements, especially in the case of wading birds, may provide increased ability to elucidate differences in the additional ecological benefits provided by the various sized reservoirs, along with potentially reducing the overlap in abundance and areal coverage SAV ranges for the individual point scores.

Assuming that additional optimization of reservoir operations could not be achieved, it appears that the potential ecological benefits provided by the 500k reservoir are similar enough to those provided by the 1mil and 2mil reservoirs that building a reservoir larger than 500k probably could not be justified from a cost-benefit perspective. It also appears that some ecological benefits, even if not the primary goal of water storage reservoirs for Lake Okeechobee, may still occur with 500k of additional watershed water storage north of the lake as compared to the ECB. However, while there are periods that a 500k reservoir would keep the lake lower than the ECB during periods of rapid lake stage increases over a short duration (e.g., two months), the 2mil reservoir offers an increased ability to keep the lake lower during these high rainfall periods and higher during droughts. For example, when lake stage increased from approximately 13.25 to 16 feet between June and August 2013, that additional 500k and 2mil of storage north of the lake may have kept the lake up to 0.5 feet and almost 2 feet lower, respectively, and in the ecologically beneficial stage envelope for most of that period, as suggested by a similar summer increase in lake stage during 1966 and 1999 (**Figures 5-11** and **5-17**, respectively). Over the POR, there were two and four occurrences where the largest reservoir would have kept the lake from exceeding the top and from dropping below the ecologically beneficial stage envelope, respectively, whereas the 500k reservoir would not have prevented these occurrences. The largest reservoir during these same occurrences also would have prevented the lake from exceeding the top (once) and dropping below (twice) the bottom of the ecologically beneficial stage envelope, relative to the storage provided by the 1mil reservoir. Therefore, adding storage north of the lake of at least 500k ac-ft in total size is projected to reduce the frequency of high and low lake stage events and provide at least some lake stage amelioration outside the ecologically beneficial stage envelope. Additionally, water supply benefits in terms of increased mean annual acre-feet of water delivered and smaller percentage of water demands not delivered accrue as the reservoir size increases (**Figure 5-25**).

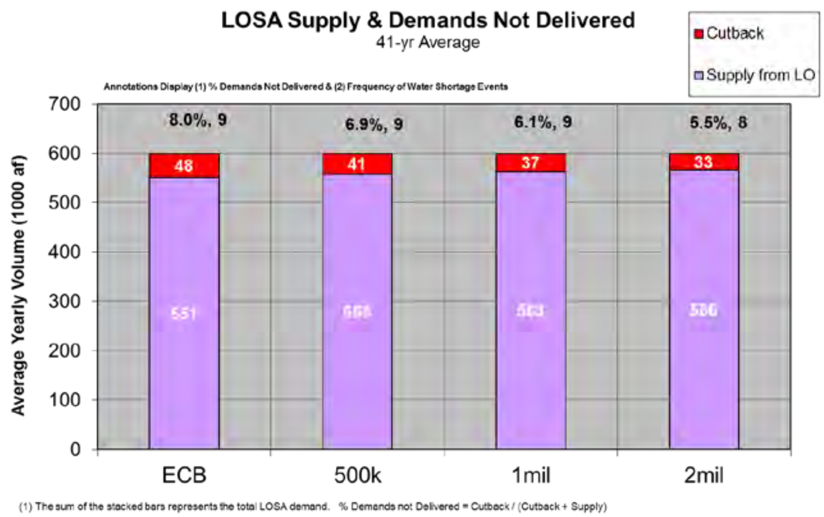
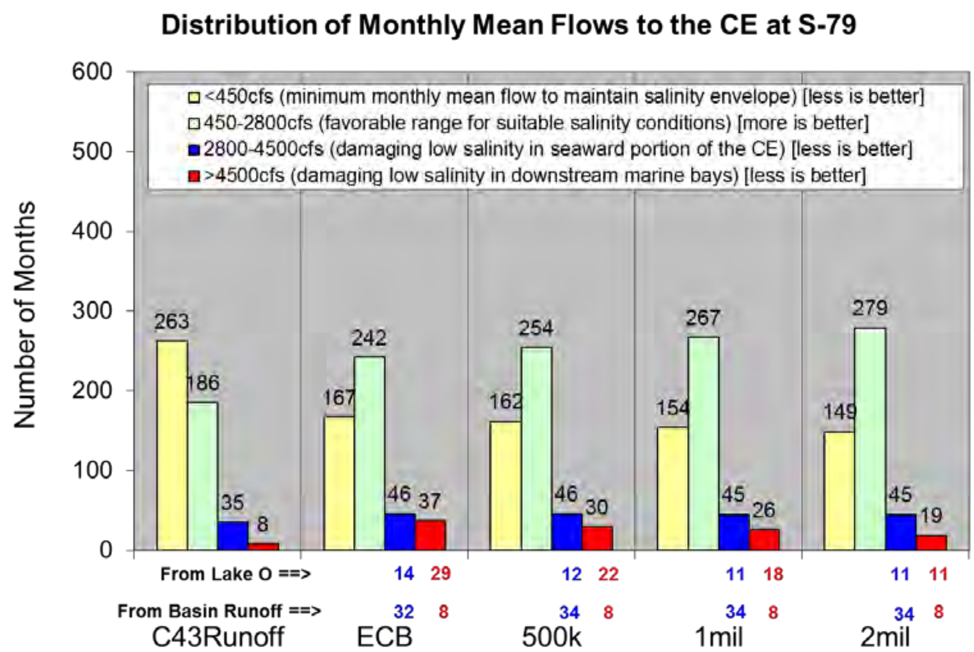


Figure 5-25. Lake Okeechobee Service Area (LOSA) average annual water supply and water demands not delivered (cutback) (both in 1,000 ac-ft) over the POR. A water shortage event consists of at least three months of cutbacks in an October–September period, with at least one of the cutbacks consisting of at least 10% of the water demand not being delivered.

Benefits to the east and west coast estuaries in terms of significantly fewer average months (18% to 42% fewer average months for the 500k and 2mil reservoirs, respectively) of high volume (> 3,000 cfs for St. Lucie Estuary; > 4,500 cfs for Caloosahatchee River Estuary) water release events also would result if a northern storage reservoir were in place (**Figure 5-26**). Additional benefits in terms of a small increase in the number of months that the potential for beneficial small releases (favorable for suitable estuary salinity ranges) from Lake Okeechobee to both estuaries during abnormally dry periods also are projected to occur with the addition of a storage reservoir in the northern watershed. For example, beneficial releases to help maintian desirable salinity conditions in the estuaries are projected to increase by an additional 12 to 37 (Caloosahatchee River) and 10 to 30 (St. Lucie Canal) months over the POR for the 500k and 2mil, respectively, reservoir scenarios.



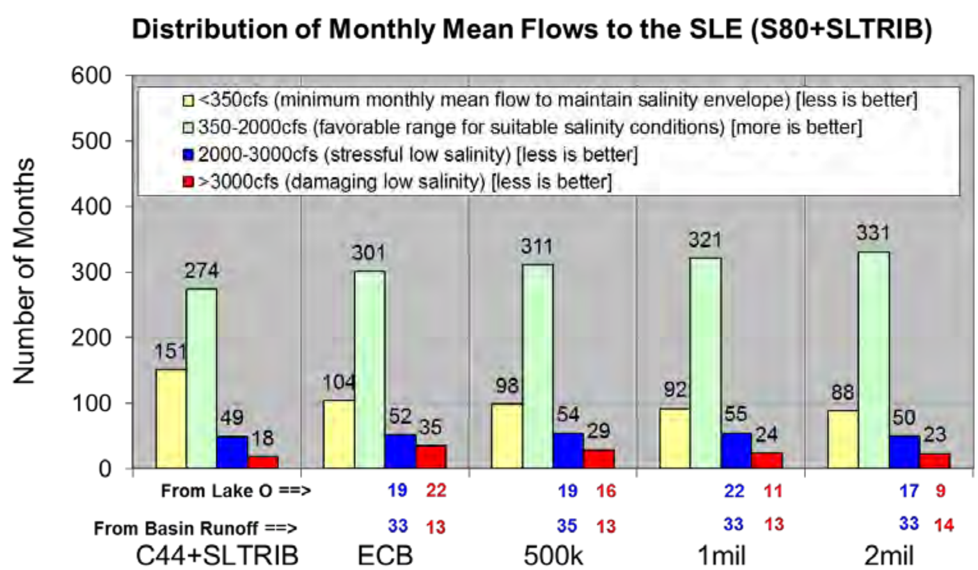


Figure 5-26. Mean monthly flow regimes in cubic feet per second (cfs) over the POR supplied by Lake Okeechobee and from basin runoff under the ECB and three northern storage scenarios for the Caloosahatchee (CE, top) and St. Lucie (SLE, bottom) estuaries. The additional scenario is runoff from the C-43 Reservoir (CE) and the C-44 Reservoir (SLE) and St. Lucie tributary runoff with no additional northern storage outside of the lake. The various flow regimes are described in the legend.

Tradeoffs in ecological benefits provided by storage reservoirs and environmental impacts from their placement also will have to be examined. For example, large surface reservoirs will displace plants and animals otherwise occupying that footprint and may further fragment existing habitat. The tradeoff in potentially increased SAV coverage and subsequent increases in fish abundance in Lake Okeechobee resulting from a large surface reservoir versus the loss of displaced plants and animals in the area where the reservoir will be placed will have to be evaluated to help in determining if the additional benefits from water storage north of the lake exceed the environmental impacts. Large reservoirs (as currently envisioned) would not be expected to contain a significant amount of suitable aquatic ecological habitat due to rapid water elevation changes, and potential for dry out. Another potential benefit of additional water storage north of Lake Okeechobee would be achieving the goals of the Lake Istokpoga Regulation Study Project. The major focus of this project is to increase annual lake stage fluctuations, to help create a balance between environmental needs, water supply, and flood control in the Lake Istokpoga drainage basin. This balance cannot be achieved without additional water storage targeted for Lake Istokpoga. The project would provide benefits for navigation, nutrient reduction, water supply, and levee maintenance. The modeling efforts here did not attempt to include Lake Istokpoga, so additional modeling and further study would be needed to determine linkages between Lake Okeechobee and Lake Istokpoga benefits.

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CHAPTER 6

GREATER EVERGLADES

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TABLE OF CONTENTS

CHAPTER 6 GREATER EVERGLADES

INTRODUCTION

Approximately fifty percent of the Everglades' habitat has been lost. The remaining portion of the Greater Everglades (GE) ecosystem includes a mosaic of inter-connected freshwater wetlands and estuaries located primarily to the east and south of the Everglades Agricultural Area (**Figure 6-1**). This area makes up most of the GE wetlands reported on in this System Status Report (SSR). A ridge and slough system of patterned, freshwater peatlands extends throughout Water Conservation Area (WCA) 1 (which is within the Arthur R. Marshall Loxahatchee National Wildlife Refuge), WCAs 2, 3A and 3B into Shark River Slough (SRS) within Everglades National Park (ENP). The ridge and slough wetlands drain into tidal rivers that flow through mangrove estuaries into the Gulf of Mexico. Higher elevation wetlands that flank either side of SRS are characterized by marl substrates and exposed limestone bedrock. The marl wetland areas located to the east of SRS form the drainage basin for Taylor Slough, which flows through an estuary of dwarf mangrove forests into northeastern Florida Bay. The Everglades marshes merge with the forested wetlands of Big Cypress National Preserve (BCNP) to the west of WCA 3 and ENP.

Restoration Coordination and Verification Program (RECOVER) Monitoring and Assessment Plan (MAP) monitoring over the past 10 years has documented periodic dry years that may have had a positive role in shaping ecosystem structure, providing habitat and concentrating prey for key restoration indicators such as alligators and wading birds. However, successive dry years (droughts) can negatively impact the system. These impacts are exacerbated by the current Central and Southern Florida (C&SF) Project, which compartmentalized the GE and created upstream regions that are too dry and downstream areas that are too wet. The C&SF Project has also disrupted the flow of clean water through the GE that defined and sustained the characteristics that make these wetlands unique in the world.

The following assessment reports are provided for the GE module in the 2014 SSR and build upon previous RECOVER reports (e.g., 2007 and 2009 SSRs [RECOVER 2007a, 20011]):

- Hydrology - GE Sheet Flow and Hydropattern
    - GE hydrology
- Nutrients - Oligotrophic Nutrient Status
    - Oligotrophic nutrient status as indicated by periphyton

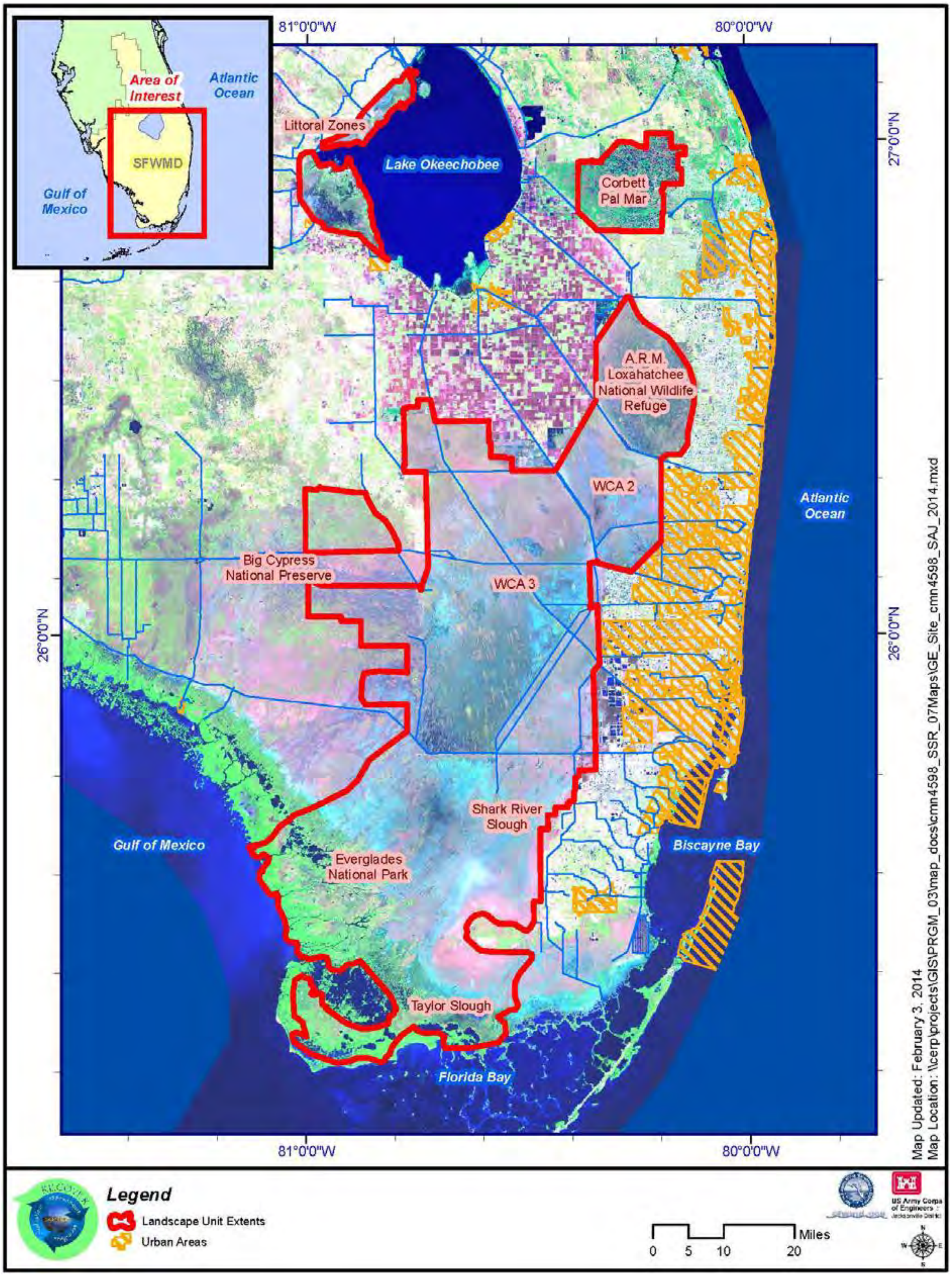


Figure 6-1. Map of the Greater Everglades.

- Landscape - GE Ridge and Slough Patterns and Tree Islands
    - GE vegetation mapping
    - Ridge-and-slough patterning in the interior Everglades peatland
    - Anisotropy and hydrologic feedback generation of ridge and slough
    - Evaporation-Driven Phosphorus Transport from Sloughs to Ridges
    - Hydrologic driven short-term vegetation dynamics in SRS
    - Surface-groundwater interactions in Everglades tree islands
- Predator-Prey - Trophic Hypothesis Update – GE Aquatic Fauna Regional Populations and Systemwide Wading Bird Nesting Pattern
    - Wet season production of aquatic prey populations
    - Dry season prey concentrations
    - Long-term trends of fish and crayfish biomass and community composition in the Southern Everglades
    - Wading bird nesting in the GE and Lake Okeechobee 2010–2013
    - Fish dynamics at the Everglades marsh-mangrove ecotone: effects of seasonal hydrology and extreme climatic events
- Predator - GE American Alligator
    - Trends in alligator relative density 2003–2012

A number of key questions, or uncertainties, regarding restoration from a quantity, quality, timing and distribution objective, spring from the GE conceptual ecological models (Ogden 2005, Ogden et al. 2005, Davis et al. 2005a, 2005b, Duever 2005). These key questions are being addressed through continued monitoring that captures the effects of climate variability and the real world effects of adhering to currently established water management regimes (e.g., regulation schedule for WCA 3A) on the system. The purpose of the monitoring, research, and predictive tool development is to provide an understanding of the degree of impacts and the time required for certain areas to recover once restoration projects are in place. This information has helped to inform operations and improve tools to support the restoration planning and design of the Central Everglades Planning Project (CEPP), which will accelerate Comprehensive Everglades Restoration Plan (CERP) restoration projects in the GE. The purpose of CEPP is to achieve some of the quantity, quality, timing and distribution objectives and help mitigate the impact of successive drought years to the GE and improve the flow of clean water needed to maintain and enhance the unique Everglades ridge and slough system.

Collectively the GE monitoring components are directly applicable to refining the design, implementation and management of CERP, and to assessing pre- and post-project conditions and management issues regarding CEPP, Broward County Water Preserve Areas, and the C-111 Spreader Canal Western Project (C-111 SCWP).

HYDROLOGY

Hydrology in the Everglades is monitored by a network of water level gages operated by multiple agencies including BCNP, ENP, South Florida Water Management District (SFWMD), and United States Geological Survey (USGS). The Everglades Depth Estimation Network (EDEN; Telis 2006), funded in part by RECOVER, integrates this network of gages with models and programs to generate hydrologic data sets including hydroperiod and water level. EDEN is an integrated network of real-time water level monitoring, ground elevation modeling, and water surface modeling that provides scientists and managers with current (1992–present), on-line water depth information for the entire freshwater portion of the Greater Everglades. EDEN offers a consistent and documented data set that can be used by scientisits and managers to (1) guide large-scale field operations, (2) integrate hydrologic and ecological responses, and (3) support biological and ecological assessments that measure ecosystem responses to CERP implementation.

Hydroperiod

Average annual hydroperiod maps for two periods are compared to identify hydrologic differences between them, Water Year (WY) 2000–WY 2008 (May 1, 2000–April 30, 2008), which was presented in the 2009 SSR (RECOVER 2011) and WY 2009–WY 2013, presented in this SSR. Annual average hydroperiods for selected periods show the number of days, on average, that an area is wet (i.e., water level above ground surface). Each area for individual years may be wetter or drier than the average. The hydroperiod map for the period WY 2000–WY 2008 from the 2009 SSR has been revised for this report. The EDEN surface-water model Version 1 was revised in 2012 and all maps presented here are based on the EDEN Version 2 model.

The comparison of the map for WY 2000 through WY 2008 (**Figure 6-2**, left panel) is compared with the average annual hydroperiod map for WY 2009 through WY 2013 (**Figure 6-2**, right panel), hydroperiods are generally shorter (30–60 days) along the southeastern edge of WCA 3B, in Shark River and Lostmans sloughs (60 days or more), and in the northwestern portion of WCA 3A. This is consistent with data that show that three of the past five water years (2009–2013) have been generally drier than average (see the Systemwide Hydrology section in Chapter 4: Systemwide Science).

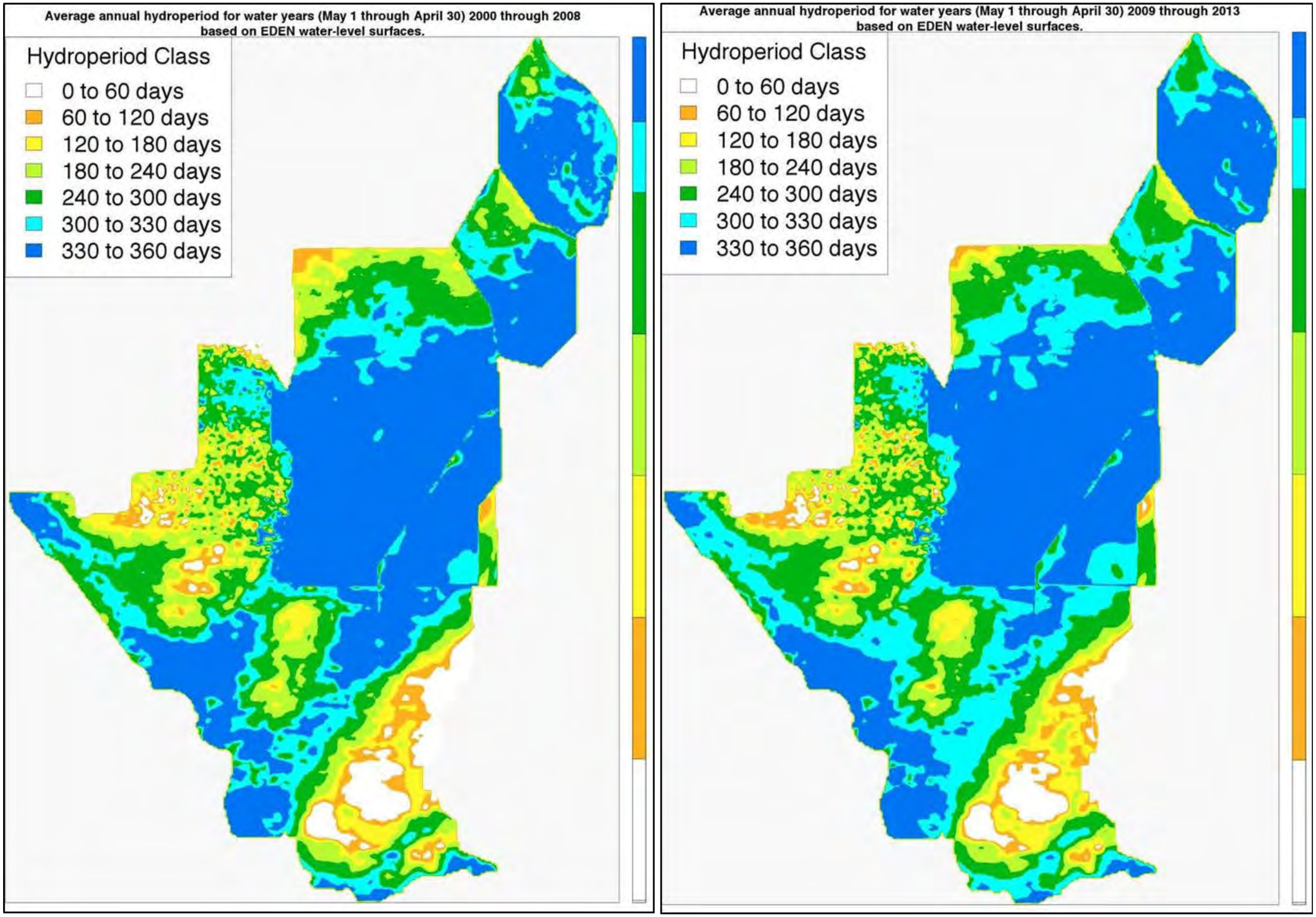


Figure 6-2. Average annual hydroperiods for WY 2000–WY 2008 (left panel) and WY 2009–WY2013 (right panel).

Slightly longer annual average hydroperiods were observed in the past five water years (2009–2013) in northern (30 days or more) and eastern (60 days or more) WCA 1 and in the eastern central portion of ENP (60 days or more). The longer hydroperiods during this drier period in ENP, when compared with the period WY 2000–WY 2008, could be an indication that the C-111 South Dade project is keeping more water in eastern ENP. Other future projects are expected to provide more water to chronically dry areas. For example, CEPP (**Figure 6-3**, left panel) is intended to increase water levels in northern WCA 3A, eastern WCA 3B, and SRS above existing conditions (**Figure 6-3**, right panel). Additionally, the C-111 Spreader Canal Western Project (C-111 SCWP), completed in 2011, is expected to increase water levels in Taylor Slough and the eastern panhandle of ENP.

The computation of days since last drydown is not included in this SSR for comparison to the previous SSR. Challenges in reporting data at specific gages when water goes below ground and differences in handling these data by the agencies both caused inconsistencies in the final computation of days since last drydown. The protocol used by EDEN to compute regional drydowns needs further discussion with agencies that operate these gages. In addition, the number and distribution of gages that measure shallow groundwater levels are not sufficient to accurately model water levels below land surface, which commonly occurs during the dry season in some parts of the Everglades. The paucity of shallow groundwater data is being examined by agency hydrologists and the EDEN team and will be discussed in the next SSR.

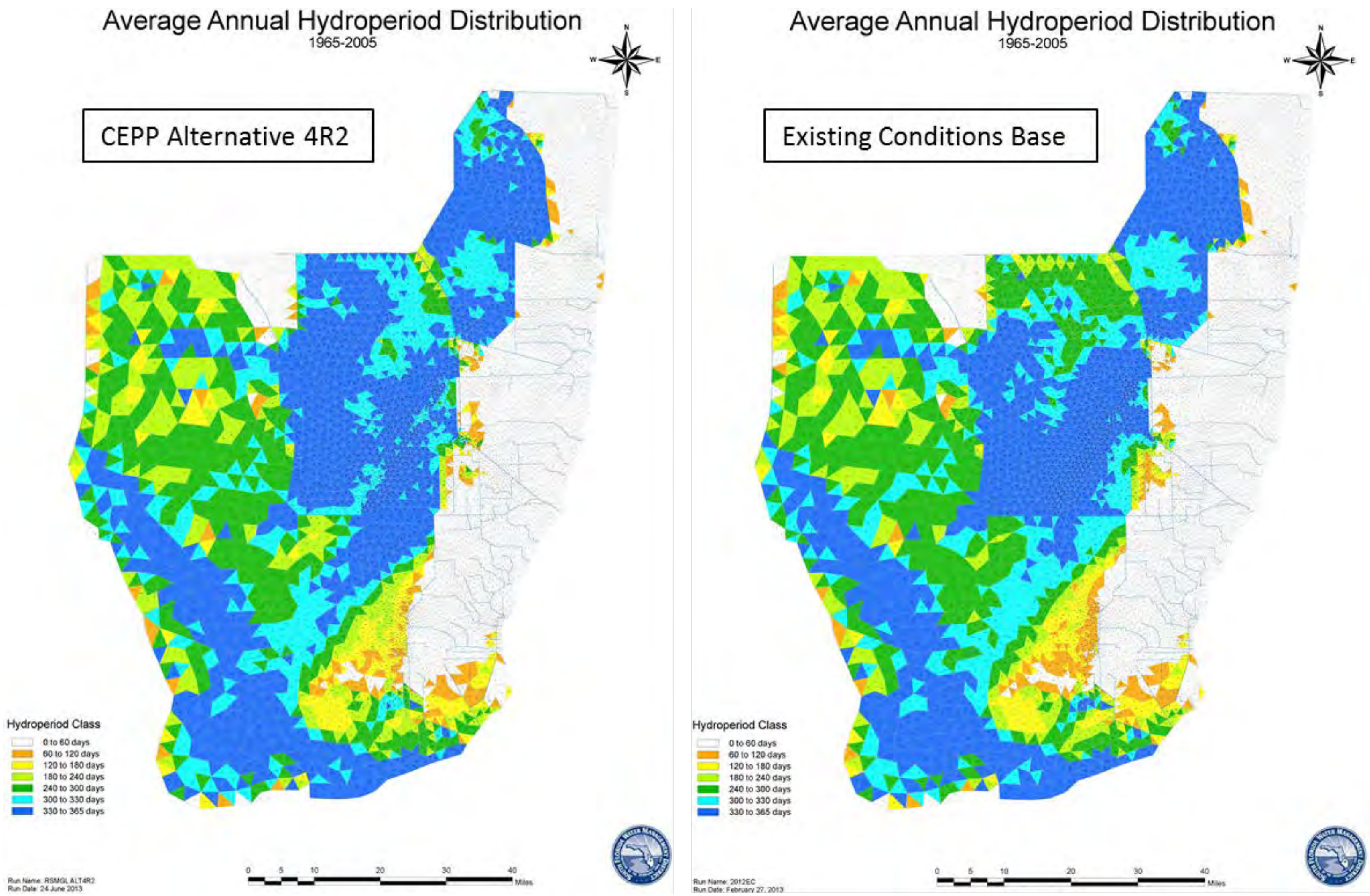


Figure 6-3. Regional Simulation Model output prediction of average annual hydroperiods for WY 1965–WY 2005 for CEPP Alternative 4R2 (left panel) and existing base conditions (right panel).

Water Levels

Introduction

This section describes a new assessment approach for detecting hydrologic changes in the Everglades using the hydrologic data provided by the EDEN network. Water level graphics compare hydrologic status and trends for two periods, WY 2000–WY 2008 (period of analysis used for the 2009 SSR) and WY 2009–WY 2013. To test the ability to detect change in wetland hydropatterns, water level graphics are also used to compare two operational periods with the future intent of adding rainfall analysis to determine what degree operations versus climate might be causing change in the system: (1) Experimental Water Deliveries Operations, which is WY 1992–WY 1999 (note that 1992 is the first year EDEN water level data is available, not the actual beginning of the experimental program) and (2) Interim Structural and Operational Plan (ISOP)/Interim Operations Plan (IOP) (hereinafter referred to as the Interim Operations Period), which is WY 2000–WY 2012). The results presented here are used to convey the current hydrologic status and trends, and to establish baselines for future assessments of the CERP and non-CERP restoration and operations projects (e.g., Modified Water Deliveries, Everglades Restoration Transition Plan, and C-111 SCWP).

Methodology

The GE system was divided into the same zones used in CEPP formulation, which can be referenced easily in the future. Daily water level percentiles are then calculated for one EDEN gage in each area then graphically displayed in a ribbon of one color depicting the 25th to 75th percentile of the daily water level throughout the year for the period of analysis (always light orange) against a ribbon depicting the 25th to 75th percentile of the daily water levels from the Natural System Model (NSM) 4.6.2 (always grey color). Areas where the two ribbons overlap are dark orange and visually represent the degree of similarity. The comparison to NSM 4.6.2 wetland water levels does not communicate that NSM water levels are the correct restoration target, but it does set a theoretical benchmark from which comparisons can be made between different operational and climatic periods. NSM 4.6.2 "envelopes", the 25th to 75th percentile ribbon, are identified as the current working RECOVER target of desired restoration conditions for the Inundation Pattern in GE Wetlands Performance Measure (http://www.evergladesplan.org/pm/recover/recover_docs/et/061807_prev_ge-2.pdf), which provides the rationale for why NSM 4.6.2 was used. Indicator regions, as specified in the performance measure, were not used at this point to simplify the analysis.

Results

Figure 6-4 provides an example of water level percentile plots in northeastern ENP using the northeast SRS 1 gage and how they can be used to compare status and trends. The bottom panel compares 25th to 75th percentile of water levels for WY 2009–WY 2013 (current 2014 SSR) to NSM estimates. Water levels over the past 5 years were drier overall and much more variable during the months of April through June in this location compared to water levels for WY 2000–WY 2008 (2009 SSR) (top panel). In fact, water levels went below ground for a portion of the time from mid-April to June during the most recent 5-year period, and never did over the previous 8 years. Intense dry downs such as these can put parts of the system at risk to severe fires that can damage vegetation, burn peat, and decrease landscape structure heterogeneity (i.e., ridge and slough and tree islands).

Table 6-1 summarizes the results of the water level percentile graphical analysis for each zone and describes the expected change from restoration. Water level graphics for each hydrologic analysis area supporting each trend can be found in **Appendix 6-1**. See **Figure A6-1-1** for a depicition of the gages presented in **Table 6-1**.

Conclusion

In summary, water levels status and trends still remained significantly altered based on comparison with one representation of the pre-drainage Everglades (NSM). Water levels were lower (drier) in northern WCA 3A, near the Miami Canal, WCA 3B, and in northeastern ENP. Water levels were higher (wetter) in southern WCA 3A and slightly higher than NSM in central WCA 1. Water levels were similar to NSM in WCA 2A and WCA 2B, except they were highly variable in both dry and wet seasons. Water levels were only truly similar to NSM in central WCA 3A, except for being higher during the wet season. Implementation of CEPP, including rain-driven operations, is likely to improve water levels where they are too low in WCA 3A, WCA 3B, and ENP, while avoiding making them higher in central and southern WCA 3A. Future SSR assessments of hydropatterns that use this methodlogy combined with rainfall analysis, should be able to detect hydrologic improvements from restoration projects/operations across different parts of the GE system.

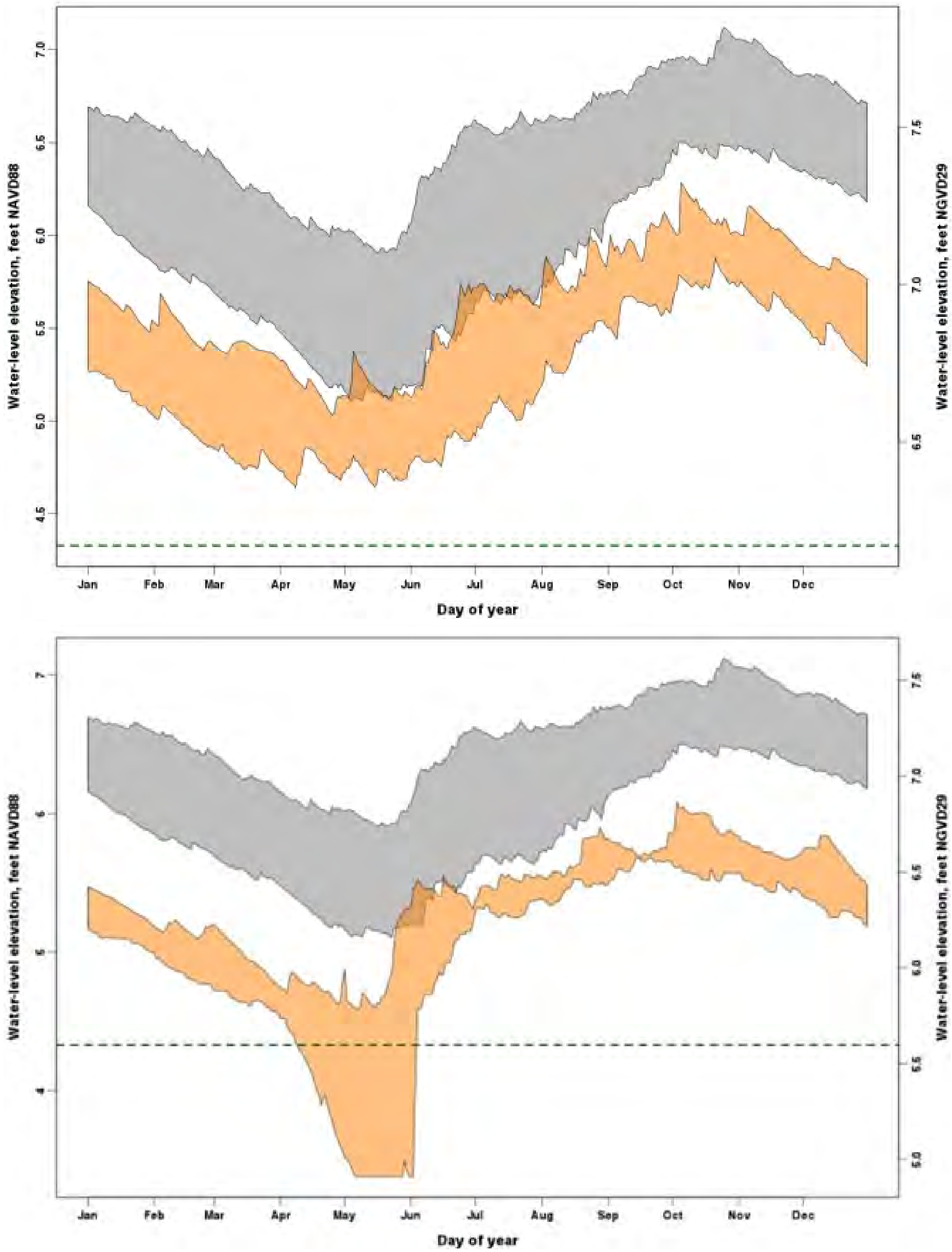


Figure 6-4. The graphs depict 25th to 75th percentile water levels for northeast SRS in ENP for WY 2000–WY 2008 (top panel) and WY 2009–WY 2013 (bottom panel). Measured water levels (light orange ribbon) are compared to water levels estimated by NSM 4.6.2 (grey ribbon) at gaging station NESRS1. Dark orange represents overlap area of observed data and the NSM benchmark. Green dashed line represents the average ground elevation at the gage.

Table 6-1. Summary of water level results for each time period compared to NSM. Actual figures depicting the data are available in Appendix 6-1. Please note NSM is not always the desired target but is a working benchmark used for comparison purposes.

Hydrologic Analysis Area	Hydrologic Change Expectation for Restoration Gage Used for Assessment of Area	Status and Trends		Operational Rainfall Periods	
		WY 2000–WY 2008	WY 2009–WY 2013	Experimental Program (1992–1999)	Interim Operational Plan (2000–2012)
WCA 1 (Site 7)	No Change. Desired hydrology is for ridge, slough and tree island landscape.	Measured lower than NSM.	Measured lower than NSM.	Overlap during start of dry season. Measured lower than NSM end of dry season and during wet season.	Measured lower than NSM.
WCA 2A (Site 17)	More natural water level fluctuation to restore ridge and slough landscape (reduce drainage in north and ponding in south).	Measured lower than NSM during dry season. Measured higher than NSM during wet season.	Highly altered water levels. Similar water levels June to August.	Highly altered water levels.	Measured lower than NSM during dry season. Measured higher than NSM during wet season.
WCA 2B (Site 99)	No Change. Highly impounded area and key to urban water supply.	Measured lower than NSM during end of dry season start of wet season.	Measured lower than NSM during end of dry season. Measured higher than NSM during wet season and early dry season.	Measured higher than NSM.	Measured lower than NSM during dry season. Slightly higher during wet season.
WCA 3A-NW (Site 62)	Increased sheet flow and hydroperiods to reduce soil oxidation and extreme fires and restore ridge and slough mosaic. Area is severely drained and degraded. Area connects to conserved 3A-Central area of ridge and slough landscape.	Measured lower than NSM.	Measured lower than NSM.	Measured lower than NSM.	Measured lower than NSM.
WCA 3A-NE (Site 63)	Increased sheet flow and hydroperiods to reduce soil oxidation and extreme fires. Area highly impacted by soil oxidation, would include mosaic of ridge and slough and sawgrass plain in restored condition.	Measured lower than NSM.	Measured lower than NSM.	Measured lower than NSM.	Measured lower than NSM.
WCA 3A-MC: 3A12 (Site 3A12)	Increased sheet flow and hydroperiods to reduce soil oxidation and extreme fires and restore ridge, slough and tree island mosaic. Historic ridge and slough area is over drained by the Miami Canal.	Measured lower than NSM.	Measured lower than NSM.	Measured lower than NSM.	Measured lower than NSM.
WCA 3A-C (Site 64)	Slightly increased hydroperiods and sheet flow to maintain and enhance conserved ridge, slough and tree island mosaic.	Close to NSM in dry season. Measured higher in wet season.	Close to NSM in dry season. Measured higher in wet season.	Close to NSM in dry season. Measured higher in wet season.	Close to NSM in dry season. Measured higher in wet season.

Table 6-1. Continued.

Hydrologic Analysis Area	Hydrologic Change Expectation for Restoration Gage Used for Assessment of Area	Status and Trends		Operational Rainfall Periods	
		WY 2000–WY 2008	WY 2009–WY 2013	Experimental Program (1992–1999)	Interim Operational Plan (2000–2012)
WCA 3A-SW (Site 65)	Decreased hydroperiods and increased sheet flow to reduce ponding and restore ridge, slough and tree island mosaic.	Measured higher than NSM.	Measured higher than NSM.	Measured higher than NSM.	Measured higher than NSM.
WCA 3B (Site 71)	Increased hydroperiods and sheet flow to reduce soil oxidation and frequency of damaging fires. Transition areas from sawgrass to ridge and slough mosaic while helping maintain and transition tree island communities to longer hydroperiods.	Measured lower than NSM. Slight overlap in wet season.	Measured lower than NSM.	Similar in dry season and wet season. Slightly lower at start of dry season.	Measured lower than NSM.
ENP NE* (site NESRS1)	Increased hydroperiods and sheet flow to reduce soil oxidation and frequency of damaging fires. Restore ridge and slough landscape and tree island mosaic. Affected by operations along Tamiami Trail from WCA 3B to ENP and South Dade Conveyance System.	Measured lower than NSM.	Measured lower than NSM.	Measured lower than NSM. Slight overlap end of dry season.	Measured lower than NSM.
ENP NW* (site NP201)	Increased hydroperiods and sheet flow to restore ridge and slough and marl prairie mosaic. Affected by operations along Tamiami Trail from BCNP and WCA 3A to ENP.	Similar water levels. Measured lower at start of dry season through April.	Similar water levels. Measured lower at start of dry season through April.	Similar water levels, but slightly higher during end of wet season and start of dry season.	Similar water levels. Highly variable at start of dry season.
ENP S - SRS (site P36)	Increased hydroperiods and sheet flow to restore ridge and slough, tree island and marl prairie mosaic and estuarine salinity patterns.	Measured lower than NSM. Slight overlap during wet season.	Measured lower than NSM. Slight overlap during wet season.	Similar water levels, but slightly lower.	Measured lower than NSM. Slight overlap during wet season.
ENP SE - Taylor Slough (site P37)	Increased hydroperiods and sheet flow to restore ridge and slough, tree island and marl prairie mosaic and estuarine salinity patterns.	Similar water levels. Slightly higher during wet season.	Similar water levels. Slightly higher during wet season.	Measured higher during wet season. Overlap on high end.	Similar water levels. Slightly higher during wet season.

*Please note that zone ENP N has been split into ENP NE and ENP NW, as operations and the amount of water moving across these two areas are very different as indicated by CEPP modeling transects 17 and 18.

Coastal Creek Flow

Background

The Coastal Gradients hydrology and water quality monitoring network provides critical hydrologic and water quality data for major coastal creeks and rivers along nine generalized coastal gradients or transects linking freshwater to marine conditions in the southern portion of the Everglades (**Figure 6-5**). In a broader context, the data collected along these transects link the GE and Southern Coastal Systems regional hydrology; in other words, links the upstream drivers (rainfall, water management structure flow, and restoration project changes) and stressors of hydrology (stage and flow) to downstream hydrological effects (flow and salinity). This analysis is uniquely organized to evaluate discrete hydrologic baselines for CERP project changes as well as those related to operations. The southern portion of the Everglades wetlands ecosystem located south of the Tamiami Trail can be separated into two main drainage complexes: SRS and Taylor Slough/C-111. Monitoring for the Taylor Slough/C-111 complex predates the RECOVER MAP program. The USGS monitoring network in SRS and Taylor Slough/C-111 began in 1996 and was expanded with RECOVER MAP funding in 2004 to include additional flow measurements within coastal rivers. Restoration of freshwater inflow from the Everglades across Tamiami Trail via CERP and non-CERP projects is expected to sustain watercourses through the estuary that would closely resemble historic patterns, and reopen some channels that have partly filled because of reduced flow (RECOVER 2007b).

Shark River Slough Drainage Complex

Shark River Slough Drainage Complex – Structure Flow

The SRS drainage complex is comprised of five major rivers located along the southwestern coast of ENP: Lostmans, Broad, Harney, Shark and North rivers. Lostmans River also receives water from Lostmans Slough located northwest of SRS. Coastal creek discharge volumes in this complex are a product of direct rainfall, inflow from the Tamiami control structures (S12 structures) and culverts[1], and watershed runoff originating from BCNP. Flow volumes delivered south to the SRS complex from WCA 3 via the inflow structures and culverts located along the Tamiami Trail averaged 874,674 acre-feet (ac-ft) from 1964 to 2012, which encompasses numerous operational changes managed by SFWMD and the United States Army Corps of Engineers (USACE). The early Experimental Water Deliveries (EWD; WY 1981–WY 1991), late EWD (WY 1992–WY 1999), and IOP (WY 2000–WY 2012) were instrumental in providing water to ENP. These operational periods spanning 32 years were compared to coastal flows spanning only 11 years to establish baseline comparisons. IOP was Initiated in WY 2000, reducing the number of months the S-12s were open, and therefore resulting in a change in the flow distribution from WCA 3 towards ENP (**Figure 6-6**). The Everglades Restoration Transition Plan (ERTP) operations began in 2013.

[1] Culvert data used as part of inflows along Tamiami Trail can be found at http://waterdata.usgs.gov/nwis/inventory?agency_code=USGS&site_no=02289060; http://wdr.water.usgs.gov/wy2012/pdfs/02289060.2012.pdf and http://pubs.usgs.gov/wdr/2005/wdr-fl-05-2a/pdf/pp-186-259-SECoast.pdf

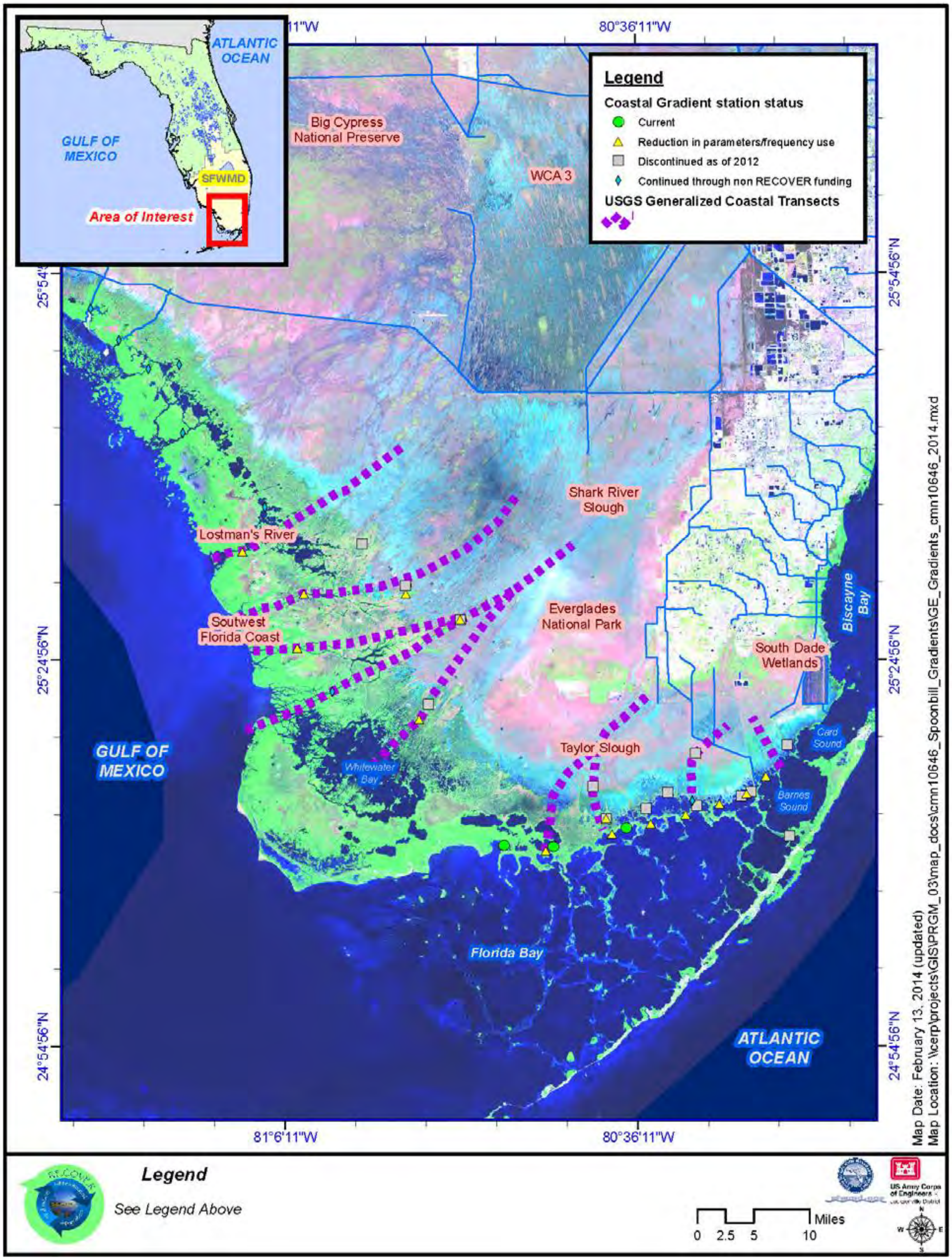


Figure 6-5. Coastal Gradients monitoring stations and transects.

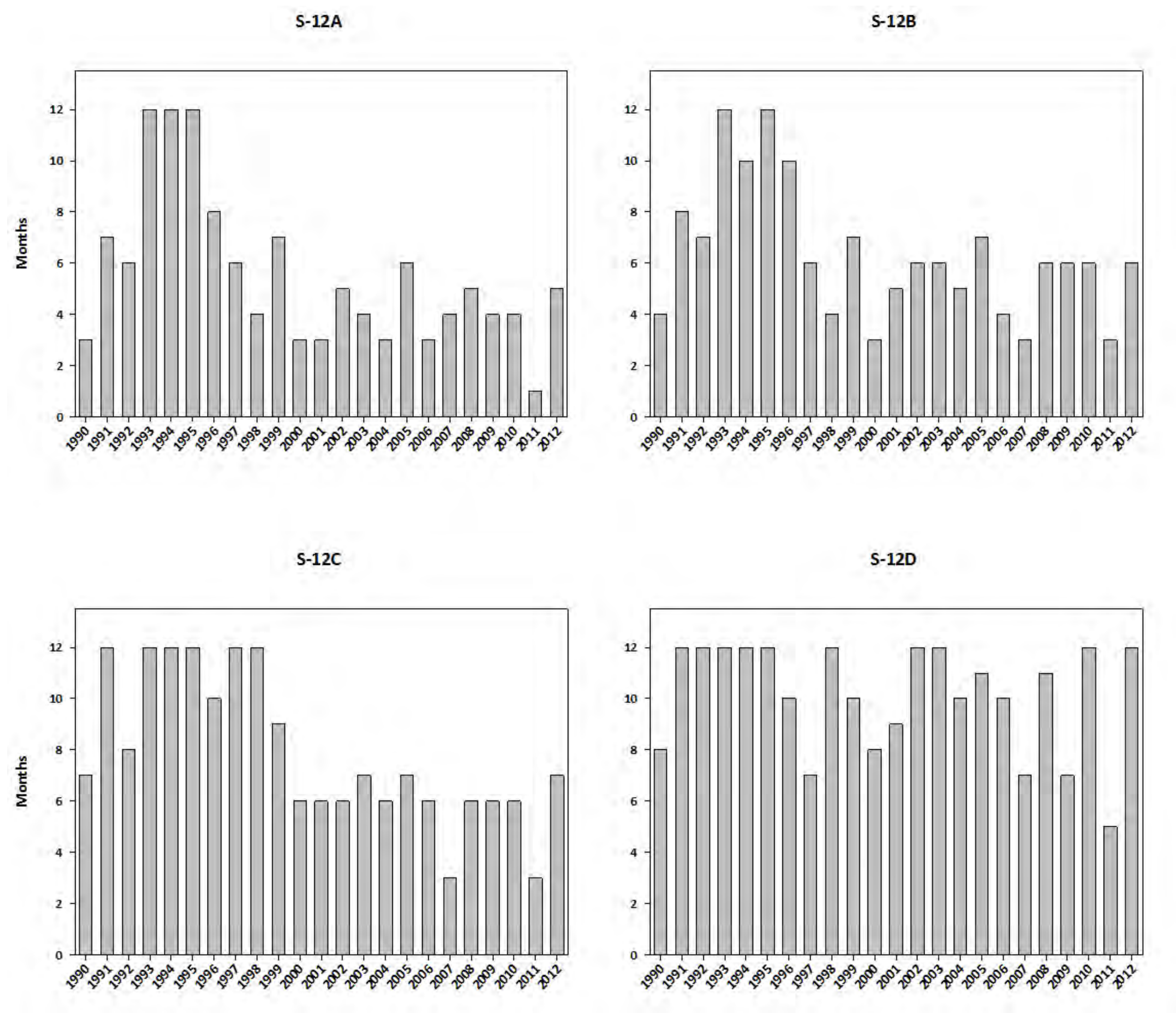


Figure 6-6. Number of months the S-12A–D structures were open from 1990 to 2012.

The status and trends of the annual flows over the last six years, representing the last two SSR reporting periods (WY 2007–WY 2009 and WY 2010–WY 2012) were also compared to each other and to the flow average during the late EWD and IOP (WY 1990–WY 2012). All water years reported in the SSR were below the late EWD and IOP flow average with the exception of WY 2009. WY 2009 would have been classified as a drought year except for Tropical Storm Fay in August 2008. WY 2008 total flow along Tamiami Trail was 197,430 ac-ft, 84% lower than the average flow reported during the late EWD and IOP. This represents the lowest reported flow since WY 1990 when 182,480 ac-ft was recorded. Back-to-back drier water years in WY 2007 and WY 2008 compared to historical averages (7.0 and 8.7 inches less rainfall) for the southwestern coast of Florida potentially explains this dramatic decrease in flow across the Tamiami Trail (Abtew et al. 2008, 2009). Flows across Tamiami Trail in WY 2011 and WY 2012 were similar, both less than the period of record (POR) average reported earlier. However, rainfall estimates in WY 2011 and WY 2012 were quite different measuring 10.9 and 1.7 inches below the average rainfall, respectively. This difference in rainfall did not appear to affect corresponding coastal river flows in

WY 2011 and WY 2012, which were 44% and 47% below the POR (WY 2002–WY 2012) average flow of 1,170,682 ac-ft, respectively (see **Figure 6-8** later in this section). One major difference that may account for the downstream coastal rivers response was the timing of the water deliveries in WY 2011 and WY 2012. Flow to ENP in WY 2011 began in May and eventually tapered off by November while flow during WY 2012 was delayed, eventually starting in September and continuing well into the dry season. In summary, the timing of flow to ENP during the wet season as well as year-to-year variations in rainfall, especially inputs derived from tropical storms or hurricanes, can directly influence the quantity and timing of flow to the coastal rivers along the southwestern coast of Florida.

As noted earlier, the total number of months that the S-12 structures (A, B, and C) were opened over the last 22 years decreased starting in WY 2000 and affected inflow volumes into ENP. Flow volumes were grouped into specific water management operational periods to better understand the effect of changes in water management on flow to ENP. The operational periods include the early/late EWDs (WY 1981–WY 1999) to maximize environmental flows to ENP and IOP (WY 2000–WY 2012) to protect the Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*) nesting grounds by lowering water levels in western SRS (SFNRC 2005). The change in the average flow during the IOP period coincides to the changes in the flow distribution along the S-12 structures (**Figure 6-6**).

The experimental water deliveries shown in **Figure 6-7** were subdivided into two operational periods representing the early (WY 1981–WY 1991) and late (WY 1992–WY 1999) EWDs to be consistent with the operational definitions originally designed to evaluate flows for Taylor Slough, which reflects changes in the C-111 South Dade Conveyance System with the S-332 pump station. An increase in the average annual flow from 725,520 to 1,756,698 ac-ft was observed between the early and late EWDs, while a decrease from 1,756,698 to 1,084,097 ac-ft was observed from the late EWD to IOP (**Figure 6-8**).

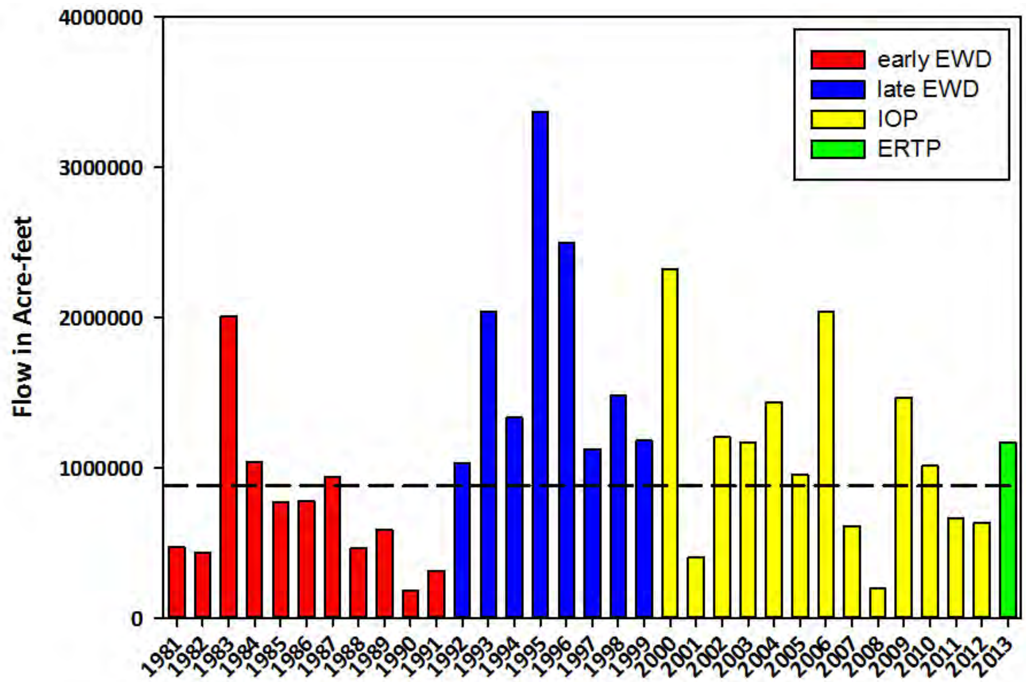


Figure 6-7. Annual flows through the S-12 structures A–D and the culverts along U.S. 41 WY 1981–WY 2013. The dashed black line represents the annual average flow for the period of record from 1964 to 2012 (874,674 ac-ft). WY 2013 data are provisional.

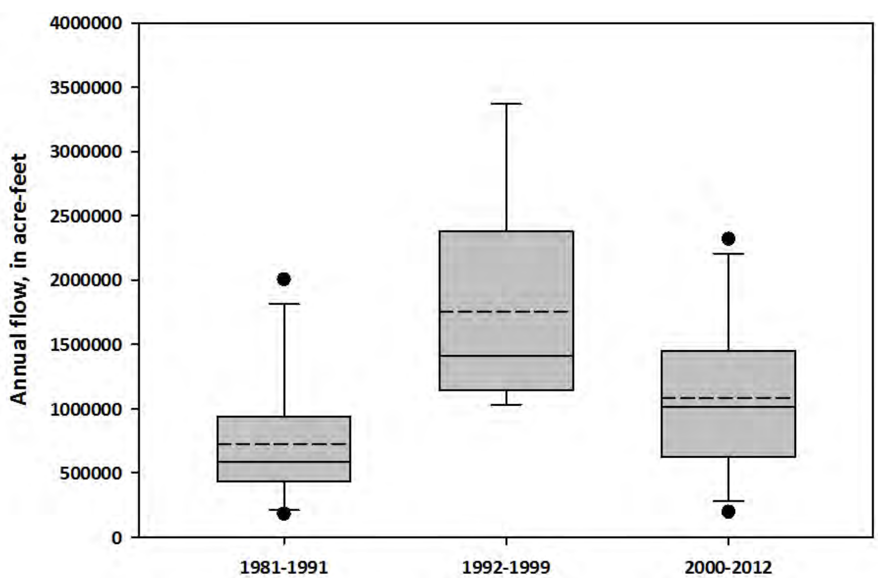


Figure 6-8. Comparison of the average flows representing specific operational periods for the S-12 structures A–D and culverts. The operations are defined as early EWD (WY 1981–WY 1991), late EWD (WY 1992–WY 1999), and IOP (WY 2000–WY 2012). The dashed lines represent the average annual flow, while the solid line represents the median annual flow. The whiskers represent the 10th and 90th percentiles while the circles represent outliers.

Shark River Slough Drainage Complex – Coastal Creek Flow

There are five monitoring sites located near the mouths of the rivers along the southwestern coast of Florida that serve as the downstream flow component of the Coastal Gradients project. The rivers include Lostmans River below Second Bay, Broad River near the Cutoff, Harney River, Shark River below Gunboat Island, and North River upstream of the Cutoff. The annual average flow from WY 2002 to WY 2012 was 1,170,682 ac-ft (**Figure 6-9**).

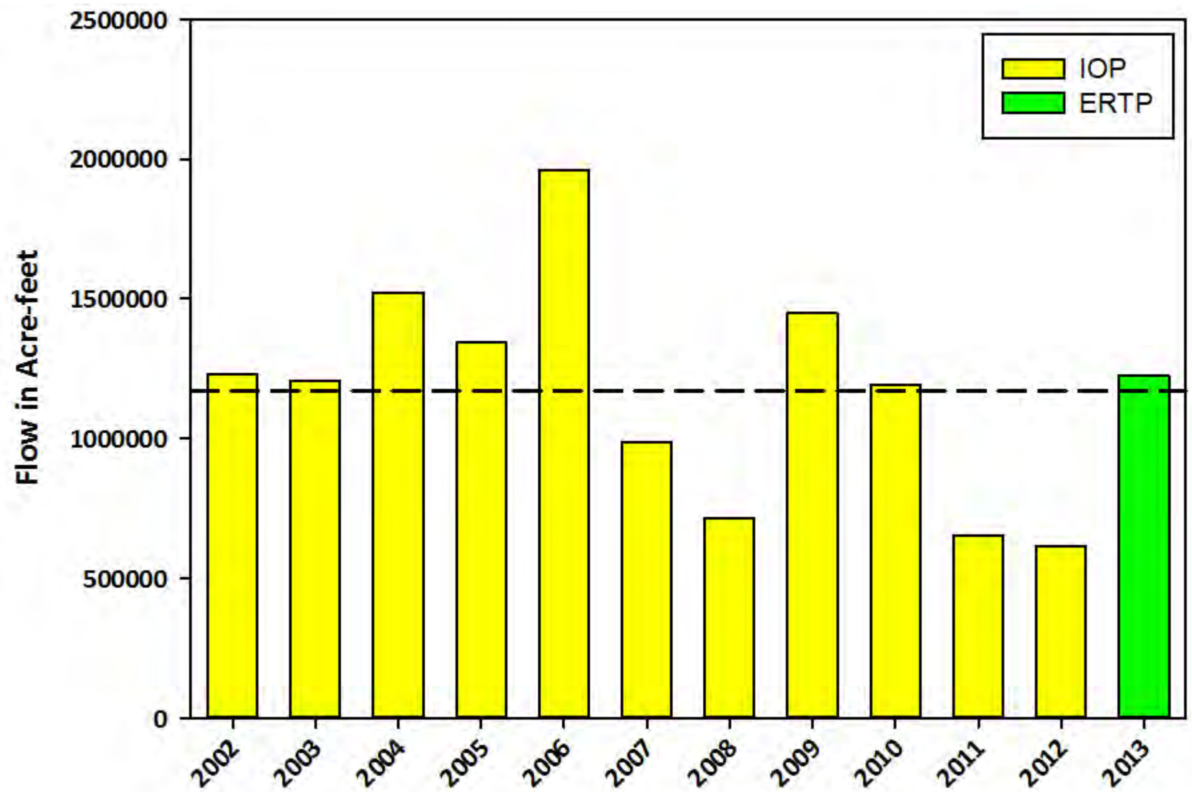


Figure 6-9. Annual flows for the five southwestern coastal rivers WY 2002 to WY 2013. The dashed line shows the average annual flow from WY 2002 to WY 2012 (1,170,682 ac-ft). WY 2013 data represent the start of ERTP operations.

The total annual flow in WY 2012 was 614,580 ac-ft or 47.5% lower than the POR flow average (**Figure 6-9**). Total annual flow in WY 2012 was the minimum for the POR. By comparison, the total annual flow in WY 2011 was 650,800 ac-ft or 44.4% lower than the POR flow average. The total annual flow observed in WY 2011 and WY 2012 corresponded to below average flow recorded at the S-12 structures as well.

Flows during the last two SSR reporting periods were compared to the POR average flow. WY 2007–WY 2008 and WY 2011–WY 2012 were lower than the POR flow average while WY 2009 and WY 2010 were higher. Low flows reported in WY 2007–WY 2008 and WY 2011–WY 2012 were due to droughts and lower reported discharges via the upstream structures. The average flow representing the WY

2007–WY 2009 and WY 2010–WY 2012 equaled 1,051,633 and 819,753 ac-ft, respectively. WY 2007-WY2009 was on the average roughly 231,000 ac-ft higher than the WY 2010-WY 2012. If it were not for to the high flows related to Tropical Storm Fay in August 2008, the average flow for WY 2007–WY 2009 could have been lower.

Distribution of flow during WY 2004–WY 2008 was reported in the 2009 SSR (RECOVER 2011). The distribution of flow during WY 2009–WY 2012 and the POR (WY 2004–WY 2012) are presented and compared against distributions for WY 2004–WY 2008 (revised) in this section. In comparison, the percent distribution of flow at Broad River and Shark River increased while percent distribution at Harney River decreased in the latter period. As a proportion of the total flow delivered to the estuary, flow at Broad River and Harney River decreased in the latter period while flow at Shark River increased. The percent distribution of flow at North River remained unchanged between reporting periods but the total flow decreased in the latter period (**Table 6-2**). The decrease in flow at Harney River could be related to the lower flows along Tamiami Trail, below average rainfall, and reported droughts in WY 2007–WY 2008 and WY 2011–WY 2012. The percent distribution of flow during WY 2004–WY 2012 representing both Lostmans Slough and SRS is presented in **Table 6-3**.

Table 6-2. Percentage of flow along the southwestern coast of Florida representing SRS.

Basin	WY 2004–WY 2008	WY 2009–WY 2012	WY 2002–WY 2012
Broad River	27.8%	32.2%	29.7%
Harney River	42.8%	30.2%	37.2%
Shark River	25.0%	33.8%	29.0%
North River	4.4%	3.8%	4.1%
Average Flow	**785,172 ac-ft**	**627,138 ac-ft**	**714,934 ac-ft**

Table 6-3. Percentage of flow along the southwestern coast of Florida representing Lostmans Slough and SRS.

Basin	WY 2002–WY 2012
Lostmans River	38.0%
Broad River	18.3%
Harney River	23.7%
Shark River	17.3%
North River	2.6%
Average Flow	**1, 170,682 ac-ft**

Taylor Slough/C-111 Drainage Complex

Flow volumes in the Taylor Slough/C-111 drainage complex are a product of direct rainfall and operational contributions from the upstream watershed. These are best characterized by flows measured directly at the Taylor Slough Bridge, the C-111 Canal at the S-18C structure, and the C-111 Canal at the S-197 structure. The Taylor Slough/C-111 drainage complex consists of distintive flow paths to the coast represented by the following estuarine creeks: East Highway Creek, West Highway Creek, Oregon Creek, Stillwater Creek, Trout Creek, Mud Creek, East Creek, Taylor River, McCormick Creek, and Alligator Creek. The pre-S-332i pumping operations (WY 1969–WY 1991), pre-IOP/S-332i pump operations (WY 1992–WY 1999), and IOP/S332D pump operations (WY 2000–WY 2012) were instrumental in providing water to ENP. These operational periods spanning 44 years were compared to coastal flows spanning 17 years to establish baseline comparisons.

Structure Flow to Northeastern Florida Bay and Coastal Creeks

Flow at the C-111 Canal at the S-18C structure can directly impact flow and salinity conditions measured downstream at coastal creeks of northeastern Florida Bay. With its relatively lower stage, this canal receives water via seepage from Taylor Slough in ENP (Kotun and Renshaw 2013). Minimizing this seepage has been a goal of the C-111 South Dade Project and C-111 SCWP. Lag time between managed inflows and outflows to northeastern Florida Bay is minimal because the system is highly coupled due to water management operations and connectivity of the landscape. Flow from the C-111 Canal to the eastern panhandle is approximated using the difference between flow from the C-111 Canal at the S-18C structure and flow released to Manatee Bay via the C-111 Canal at the S-197 structure. Other factors that influence the flow exchange along the C-111 Canal from the S-18C structure to the S-197 structure include flood control operations when the S-197 outflows exceed the S-18C inflows, evapotranspiration (ET), and canal exchange with marshes north and south along the C-111 Canal.

Annual flows from the C-111 Canal at the S-18C structure in WY 2012 were 21% lower than the POR flow average of 125,984 ac-ft (WY 1969–WY 2012) (**Figure 6-10**). The annual flow was 27% lower when compared to the average flow representing the IOP/S-332D period (WY 2000–WY 2012). Flow in WY 2011 was higher than the POR average flow and the IOP/S-332D period by 12% and 4%, respectively. Annual annual flows representing the last two SSR reporting periods from WY 2007–WY 2009 and WY 2010–WY 2012 were compared to the average annual flows during IOP/S-332D (136,652 ac-ft). Flows at structure S-18C were below the IOP/S-332D average flow during WY 2007–WY 2009, and WY 2012. WY 2007 was 42% lower while WY 2010 was 36% higher when compared to IOP/S-332D average annual flows. Annual rainfall totals were below the perod of record rainfall average for four out of the last six years (Zucker et al. 2013).

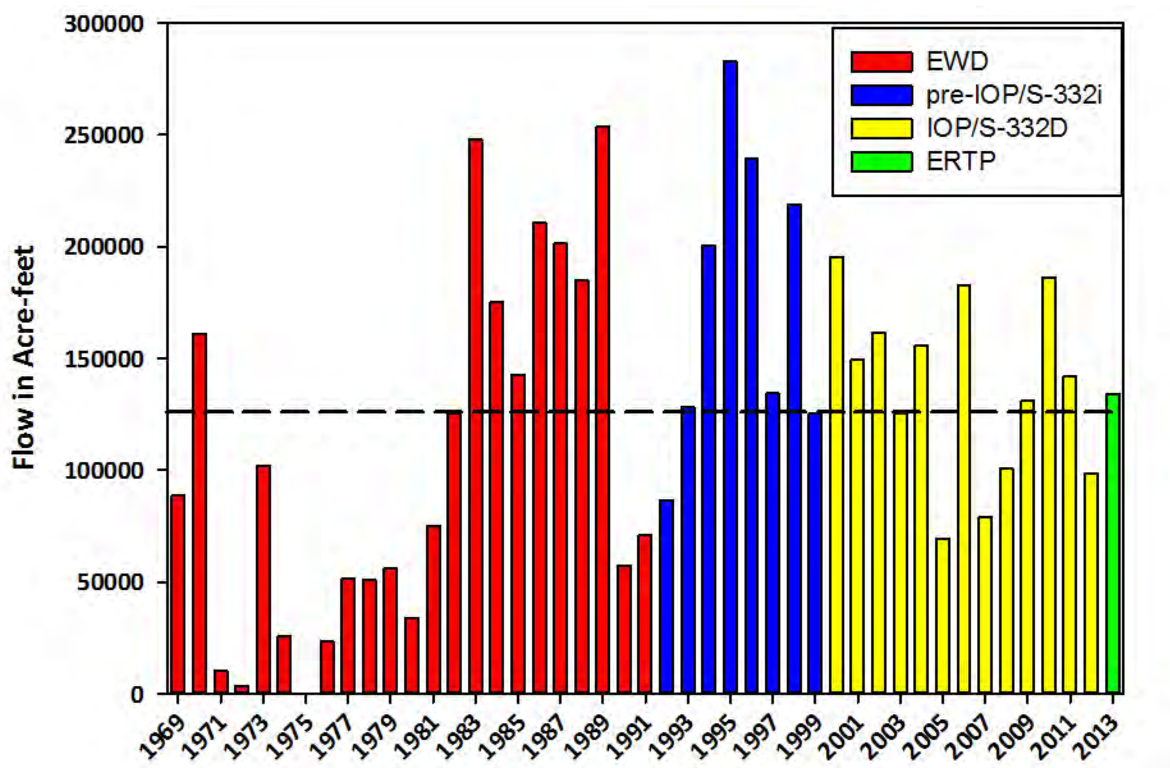


Figure 6-10. Annual flows through the C-111 Canal at the S-18C structure WY 1969–WY 2013. The dashed line represents the average annual flow from WY 1969–WY 2012 (125,984 ac-ft).

The status and trends of the annual average flows at structure S-18C were evaluated during specific operational periods. The average annual flow representative of the pre-S-332i pump operations (WY 1969–WY 1991) equaled 102,226 ac-ft and increased to 176,956 ac-ft coinciding with increased pumping at S-332 during the pre-IOP/S-3332i operations (WY 1992–WY 1999) (**Figure 6-11**), then decreased to 136,652 ac-ft during IOP/S-332D pump operations (WY 2000–WY 2012). The decrease in flow at the S-18C structure during IOP/S-332D pump operations was a direct result of the operations and reported droughts. The average annual flows at S-18C decreased during IOP/S-332D operations as various detention basins were constructed and incrementally operated to reduce seepage out of ENP and to store and deliver water to the headwaters of Taylor Slough (Abtew et al. 2010, Kotun and Renshaw 2013). As a result, flow at Taylor Slough Bridge did not change significantly during the pre-IOP/S332i and IOP/S-332D period.

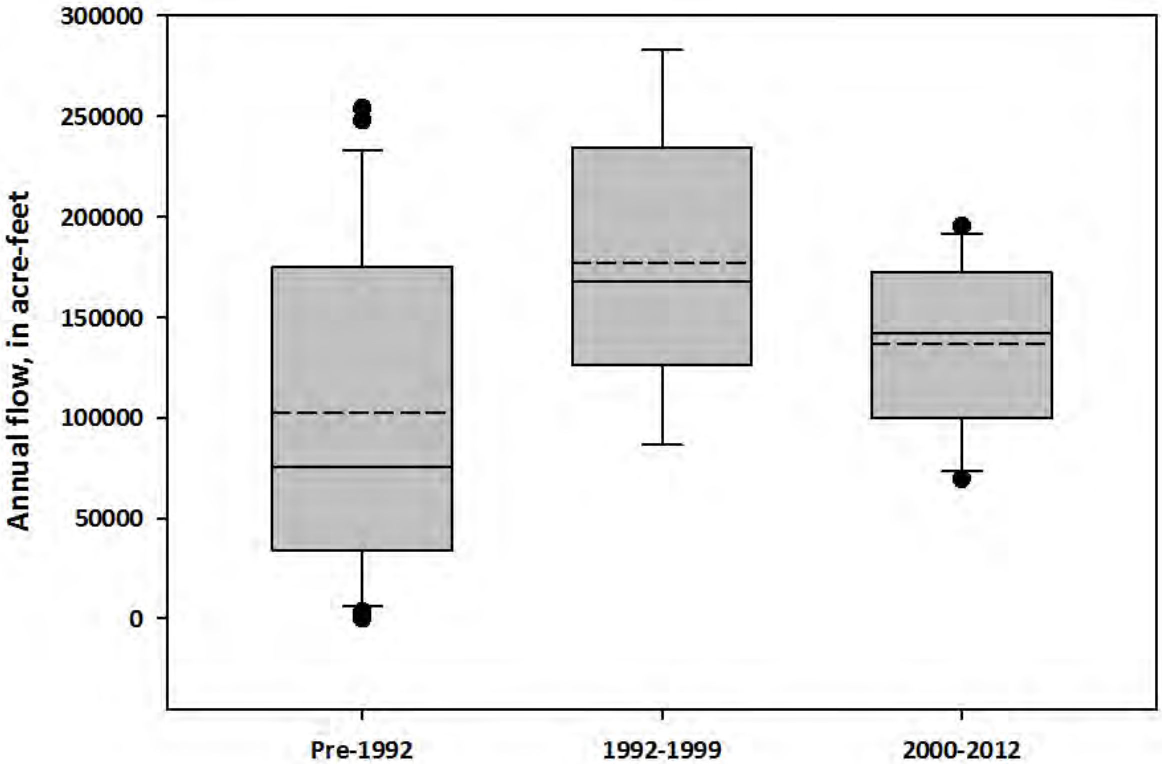


Figure 6-11. Comparison of the average flows in the C-111 Canal at the S18C structure during pre-1992/S332i, pre-IOP/S-332i and IOP/S-332D. The dash line represents the average annual flow while the solid line represents the median annual flow. The whiskers represent the 10th and 90th percentiles while the circles represent outliers.

Flow to Taylor Slough and Northeastern Florida Bay

The coastline of northeastern Florida Bay receives flow from the Taylor Slough drainage basin in addition to the flow through the wetlands south of the C-111 Canal. Flow into Taylor Slough is measured at Taylor Slough Bridge, which is south of the L-31W and C-111 canal structures that control water moving into Taylor Slough. Annual flow at Taylor Slough Bridge during WY 2012 was 27% lower than the POR average flow of 38,982 ac-ft (WY 1961–WY 2012) and 51% lower than the IOP/S-332D average flow of 58,558 ac-ft (WY 2000–WY 2012) (**Figure 6-12**). Annual average flow at Taylor Slough Bridge in WY 2011 was also higher than the flow average representing WY 1961–WY 2012 but lower than the IOP/S-332D period by 29% and 14%, respectively. Annual average flow representing the last two SSR reporting periods from WY 2007–WY 2009 and WY 2010–WY 2012 were compared to the annual average flows during IOP/S-332D. The average annual flow during WY 2007–WY 2009 was 34,095 ac-ft and increased to 49,985 ac-ft during WY 2010–WY 2012. WY 2008 was the lowest recorded flow to Taylor Slough since WY 1991, 75% lower than the IOP/S-332D average flow, while WY 2010 was 21% higher than the IOP/S-332D average flow. Flow at Taylor Slough Bridge for WY 2013 was as high as WY 1999 and WY 2001. WY 2013 initiated the ERTP/C-111 SCWP operations that will be reported on in the next SSR.

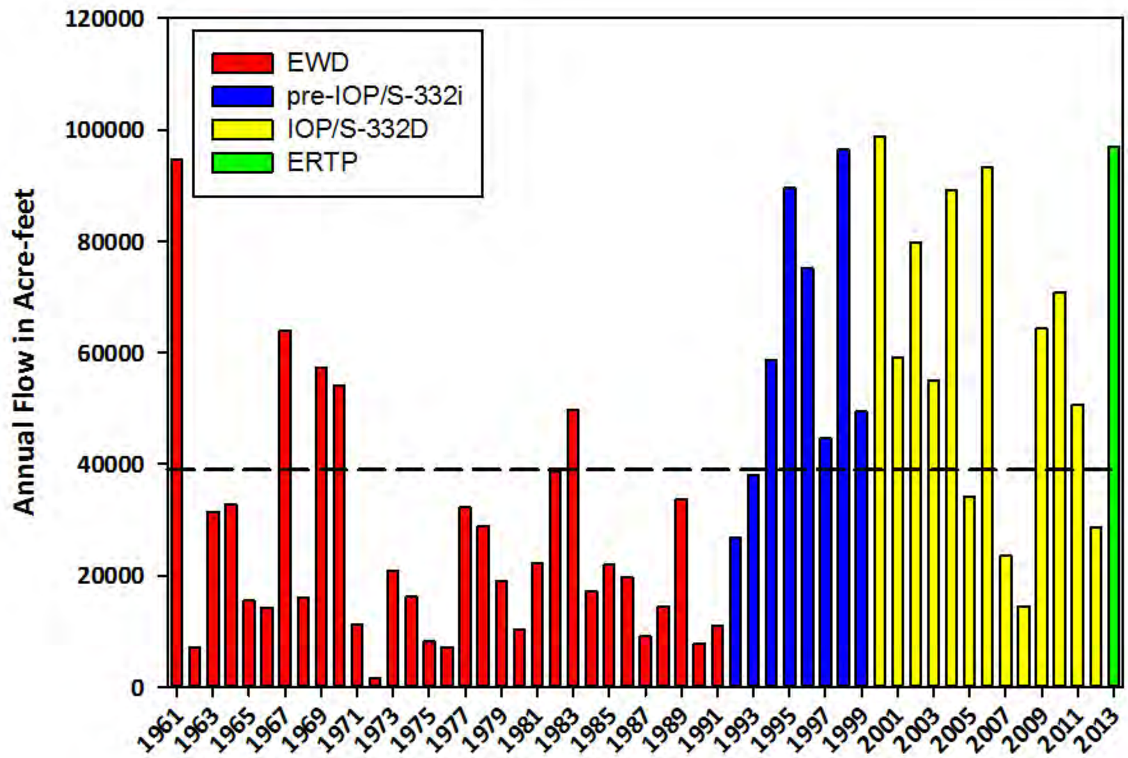


Figure 6-12. Annual flows at Taylor Slough Bridge located on the main park road within ENP. The dashed line represents the average annual flow from WY 1961–WY 2012 (39,982 ac-ft).

The status and trends of the annual average flows at Taylor Slough Bridge were evaluated during specific operational periods (pre-S-332i pumping, pre-IOP/S-332i, and IOP/S-332D). The average annual flow from WY 1961 to WY 1991 equaled 38,982 ac-ft and increased to 59,824 ac-ft coinciding with increased pumping at S-332 during the pre-IOP/S332i period (**Figure 6-13**), while remaining similar at 58,558 ac-ft during the IOP/S-332D period that was drier. Pumping at S-332D and the operation of the various detention basins improved the hydroperiods north of Taylor Slough Bridge, but the effect downstream of the bridge was minor as the C-111 Canal continued to intercept flow from moving into lower Taylor Slough (Kotun and Renshaw 2013).

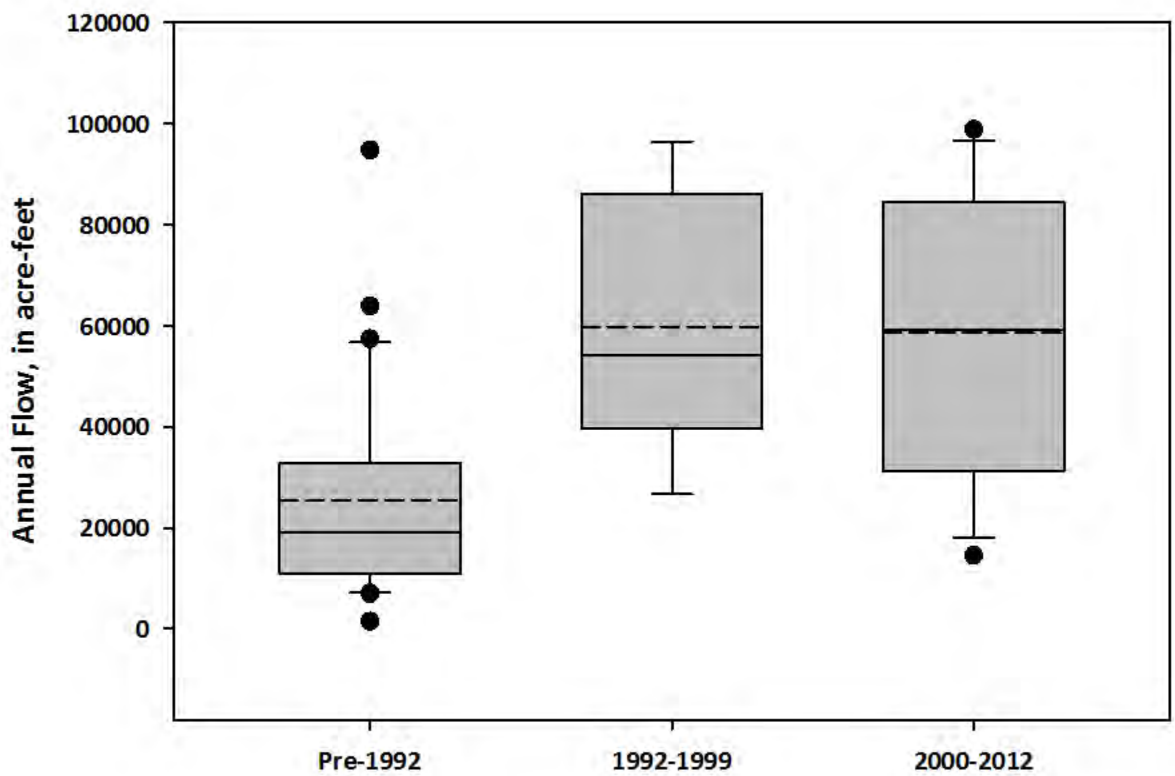


Figure 6-13. Comparison of the average flows at Taylor Slough Bridge representing the pre-1992/S332i operations, pre-IOP/S-332i and IOP/S-332D. The dashed line represents the average annual flow while the solid line represents the median annual flow. The whiskers represent the 10th and 90th percentiles while the circles represent outliers.

Taylor Slough Drainage Complex - Coastal Creek Flow

There are ten monitoring sites located near the mouth of each creek along the coastline of central and northeastern Florida Bay that serve as the downstream components of the Coastal Gradients project. The annual average flow from WY 1997 to WY 2012 was 304,449 ac-ft (**Figure 6-14**). These creek outflow totals exceed the inflow from Taylor Slough Bridge and C-111 Canal most likely due to rainfall, unquantified surface water inputs (sheetflow), and groundwater seepage.

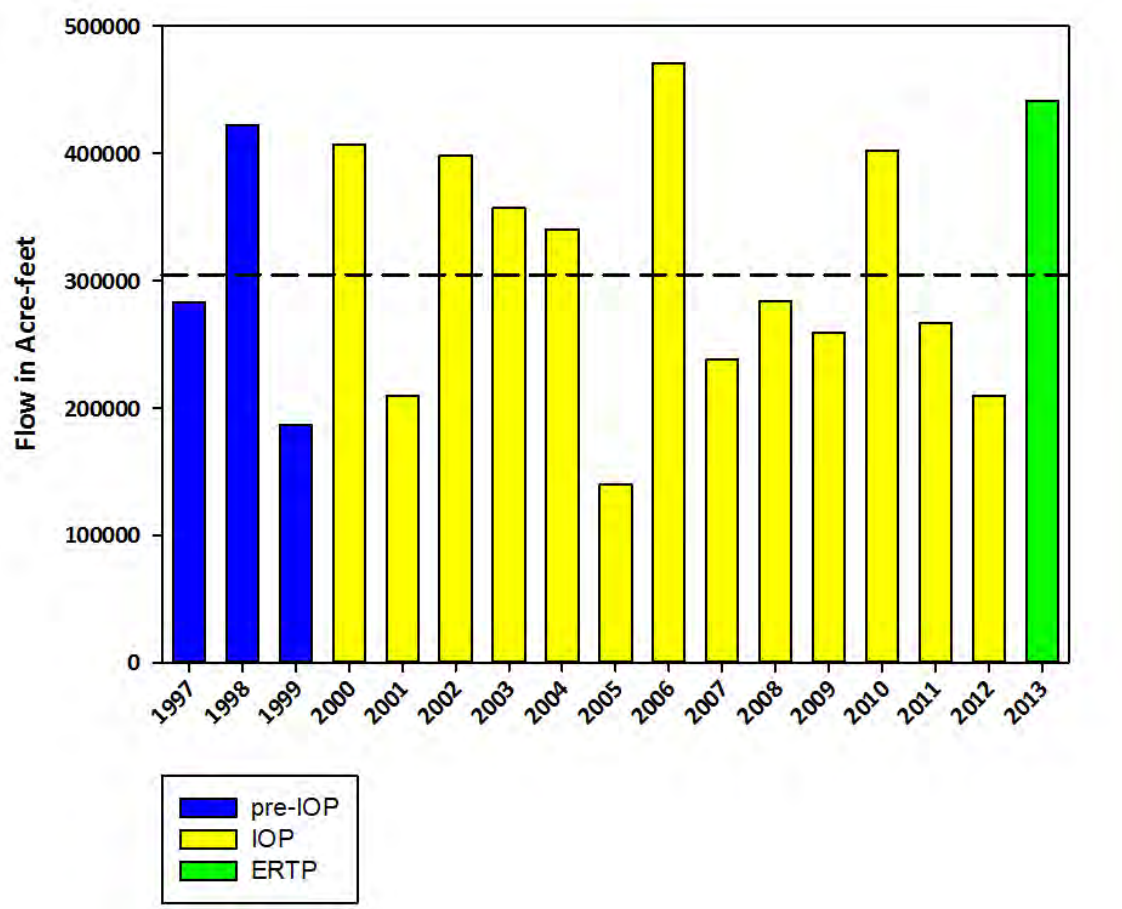


Figure 6-14. Annual flows to northeastern Florida Bay via coastal creeks WY 1997–WY 2013. The dashed line shows the average annual flow from WY 1997–WY 2012 (304,449 ac-ft).

The annual flow in WY 2012 was 209,570 ac-ft, 31% lower than the POR average flow representing WY 1997–WY 2012 (**Table 6-4**). By comparison the annual flow in WY 2011 was 12.5% lower than the flow average for WY 1997–WY 2012. Flow during the last two SSR reporting periods (WY 2007–WY 2009 and WY 2010–WY 2012) were lower than the POR average flow and the average flow condition representing IOP/S-332D (306,110 ac-ft). Only WY 2010 exceeded the average flows reported for the POR and IOP/S-332D. The average flow representing WY 2007–WY 2009 and WY 2010–WY 2012 equaled 260,023 and 292,503 ac-ft, respectively. Although there was an increase in flow in the latter SSR reporting period, the average flow was lower than the average flows for the periord of record and IOP/S-332D.

Table 6-4. Summary statistics of flow representative of the pre-IOP/S332i and IOP/S332D water operations.

Operational Periods	Flow (ac-ft)				
	Minimum	Maximum	Average	Median	Standard Deviation
1997–1999	186,902	421,880	297,254	282,980	118,138
2000–2012	139,350	470,823	306,110	283,660	97,538

The status and trends of the annual average flows to the bay were evaluated during specific operational periods. The average annual flow from WY 1997 to WY 1999 and WY 2000 to WY 2012 equaled 297,254 and 306,110 ac-ft, respectively. The average flow increased by 8,856 ac-ft from the pre-IOP/S-332i to the IOP/S-332D operational period (**Figure 6-15**). When comparing operational periods, a greater percentage of flow was distributed to Taylor Slough during IOP/S-332D (31.4%) compared to pre-IOP/S-332i (22.9%) (**Table 6-5**) (Hittle et al. 2001). Please note that the coastal flows representing pre-IOP/S332i only contain three years of flow data.

Table 6-5. The percentage of flow representative of the pre -IOP/S332i and IOP/S-332D water operations.

Basin	WY 1997 to WY 1999	WY 2000 to WY 2012
Taylor Slough	22.9%	31.4%
Trout Creek	50.7%	48.1%
Long Sound	26.4%	20.5%
Average Flow	**297,254 ac-ft**	**306,110 ac-ft**

Under the IOP/S-332D period, average flow increased in the Taylor Slough basin (~28,000 ac-ft), decreased slightly in Trout Creek (~3,400 ac-ft), and decreased in Long Sound (~a5,700 ac-ft) (**Figure 6-15**). Average flows were compared for the defined operational periods and do not reflect the year-to-year variations in flow to the bay. As reported earlier, the average flow delivered to the bay was slightly higher during the IOP/S-332D period. However, five out of the last six reporting years were below the historical flow average for the bay. WY 2013 flows are shown for comparative purposes only and represent only one year of ERTP/C-111 SCWP operations.

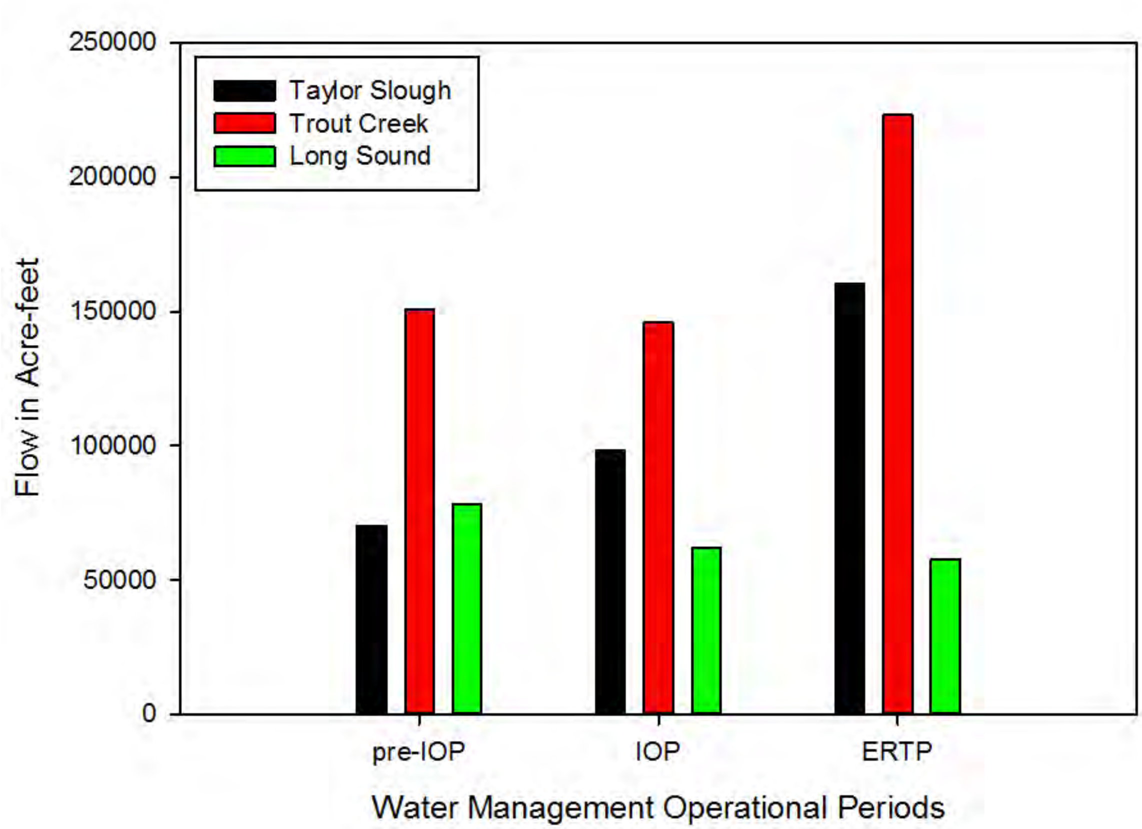


Figure 6-15. Average flow in ac-ft distributed to Long Sound, Trout Creek, and Taylor Slough during pre-IOP/S332i (1997–1999), IOP/S332D (2000–2012), and ERTP (2013 only).

Summary

The Coastal Gradients analysis of structures and downstream coastal creek flows reveal subtle differences in results for SRS and Taylor Slough drainage complexes when comparing between the pre-IOP and IOP operations periods. Rainfall was similar in the IOP period compared to the pre-IOP period, except that rainfall was slightly lower in the dry season, as indicated by the Royal Palm Ranger Station (Kotun and Renshaw 2013) and lower over 5 of the past 7 years (WY 2007–WY 2012, Abtew et al. 2008, 2009, 2010). Similar structure flows might be expected in both the SRS and Taylor Slough complexes between time periods overall, with less flow during individual dry years more recently.

However, S-12 and Tamiami Trail culvert flow was less (see **Figures 6-6** and **6-8**) over the whole IOP period, likely revealing the influence of S-12 operational changes during IOP. No data is available prior to 2001 to compare pre-IOP to IOP downstream coastal creek flows along the lower southwestern coast that might have been affected by these changes. Looking towards the future, a working hypothesis is that any additional operational changes along Tamiami Trail that increase structure flow due to the addition of CEPP reconnecting the rest of GE to ENP and increasing inflows to GE would result in increased downstream coastal creek flows and lower coastal salinities benefiting Florida Bay and the southwestern coastal systems.

In the Taylor Slough drainage complex, only Taylor Slough Bridge flow is similar (**Figures 6-12** and **6-13**), whereas, lower C-111 Canal flow (from S-18C to S-197) was less during the IOP period (**Figures 6-10** and **6-11**). This supports the objective of the C-111 South Dade Project retaining more flow in the detention areas, which would influence Taylor Slough Bridge flow and result in less overall flow down the lower C-111 Canal. Although the POR for coastal creek flows during pre-IOP is limited to three years of data with high rainfall variability, the coastal creek flows also support this objective because a greater percentage of flow was routed to Taylor Slough coastal creeks (9% more) than Long Sound coastal creeks (6% less) (**Table 6-5** and **Figure 6-15**), while the total flow did not change much at all (297,254 ac-ft per year to 306,110 ac-ft per year). Additional flows to Taylor Slough are expected with the C-111 SCWP operations over a longer time period, which would result in lower salinities in central and northeastern Florida Bay. However, unless additional flow enters this part of the system from rainfall or upstream structures, any increases to Taylor Slough could likely mean less flow towards Long Sound.

OLIGOTROPHIC NUTRIENT STATUS AS INDICATED BY PERIPHYTON

Introduction

Concerns regarding preservation of oligotrophic nutrient status under improved hydroperiod and flow conditions expected through CEPP prompted reevaluation of the periphyton-based water quality condition indicator used in “stoplight” reporting (see *Systemwide Ecological Indicators for Everglades Restoration, 2010 Report* [SFERTF 2010]). Current assessment incorporates periphyton total phosphorus (TP) content, diatom species composition and biomass metrics, using region-specific baselines established through surveys that extend along enrichment gradients into minimally impacted marsh. Species composition was dropped from routine monitoring during the 2011 MAP optimization, invoking a reexamination of detection sensitivity using the reduced suite of metrics. Data from a large-scale, long-term phosphorus (P)-enrichment study were used to determine relationships between P exposure and periphyton metrics, and results corroborated the values for baseline conditions employed previously, but also illustrated the power of multi-metric approaches, particularly those including species response information. Specifically, the ratio of endemic diatoms (common to calcareous Everglades periphyton mats) to weedy species (found in enriched areas globally) was the most sensitive metric (85% detection probability when used independently) and adds more than 18% accuracy to metrics based on TP alone (raising detection to 94% from 76%). Diatoms have long been recognized as key water quality indicators, and as a result are incorporated in national stream, wetland and lake water quality risk assessments. In the Everglades, the diatom component costs less than 20% of total periphyton assessment but improves the ability to detect enrichment by more than 18%.

A multi-metric (TP and periphyton community type) approach for WY 2011 showed periphyton alterations in central and southern WCA 3A, and indicated a community shift towards a higher TP-tolerant periphyton communities along the eastern boundary of ENP. Single-metric (TP only) assessments for WY 2012 show similar trends in WCA 3A, and show increasing alteration in WCA 3B, but did not detect periphyton community changes along the ENP boundary (see **Figure 6-17** later in the section). This data set demonstrates that 70% of the periphyton multi-metric condition downstream of inflow structures from WY 2007–WY 2011 can be explained by TP concentrations at the inflows, and suggests that legacy TP and local biogeochemical processes can account for the remaining variability. The strong relationship between periphyton metrics downstream of inflow structures and TP concentrations in inflows from WY 2007– WY 2011 points to the continuing relevance of nutrient dynamics (inputs and cycling) to Everglades ecology. Water TP measurements within the marsh suggest very low and stable ambient levels in WCA 3A and WCA 3B, and with declining concentrations being reported widely in ENP (Julian et al. 2013), did not correspond with periphyton metrics. This is likely due to the ability of the endemic periphyton to quickly absorb most P from the water column, unless the ecosystem is relatively saturated and transitioning to cattail (Gaiser, et al. 2004). Therefore, the periphyton metric is currently the most sensitive metric of community imbalance due to nutrient inflows local biogeochemical processes, and legacy TP mobilization.

Assessment Reevaluation

The periphyton indicator used in stoplight reporting is based on the risk assessment framework adopted by the United States Environmental Protection Agency (USEPA) for streams and wetlands that (1) uses stressor-response relationships in benthic algal communities to identify thresholds and baselines, and (2) employs species-based indicators that have a high sensitivity and low variance in response to stressors (Stevenson 1998, Stevenson and Smol 2003). This approach has a long history of scientific support and application, and benthic algal indicators are now broadly incorporated into water quality risk assessment in most developed nations (Stevenson et al. 2010). The relationship between periphyton attributes and P are well understood for the Everglades, and there is a wealth of support for species-based metrics that conform to USEPA standards for water quality assessment (McCormick and Stevenson 1998), similar to those regularly used in assessments in wetlands elsewhere (Gaiser and Rühland 2010). Nevertheless, species-based assessments were dropped from MAP protocols in the 2011 optimization, prompting a reevaluation of remaining metrics, and their cost-effectiveness relative to accuracy in detecting water quality impairment.

The current periphyton indicator uses periphyton TP content, species composition and biomass to determine water quality (P) impairment, based on comparison to baseline conditions represented in the interior unimpacted areas of marsh (Gaiser et al. 2006, Gaiser 2009). Since exposure histories at sites downstream from control structures cannot be measured directly, the stressor (P)-periphyton response relationship was examined in an experimental P-enrichment study to determine the detection probabilities of the three metrics, and a combination of them. The experimental data set came from the first two years of P enrichment in three flow-through flumes in ENP (conducted in 1998–1999, Gaiser et al. 2004, 2005). Baseline conditions were determined by examining the frequency distribution of control and reference channel values for each metric. Metrics were periphyton TP content, biomass and the ratio of weedy to endemic diatoms as the compositional metric, in accordance with recommended USEPA protocols. The weedy to endemic diatom ratio is an improvement upon a compositional difference metric used in earlier approaches (Gaiser 2009), and is based on the ratio of globally distributed species found in enriched habitats to calcareous mat-forming diatom species found in the Everglades (Gaiser et al. 2011). Resultant mean and variance estimates for baseline conditions were close to those reported from prior studies and incorporated in stoplight reporting (**Table 6-6**).

Table 6-6. Assessment categories for periphyton metrics determined by employing the stressor-response framework to results from the experimental P dosing facility in ENP.

Metric	Measurement	Baseline (Green)	Caution (Yellow)	Altered (Red)
TP	Periphyton TP Content (micrograms per gram [µg/g])	<200	200–250	>250
Species	Weedy:Endemic Species	<1.4	1.5–2.4	>2.5
Biomass	Ash-Free Dry Mass (grams per square meter [g/m^2])	>11	2–10	<1

To determine the relationship between measured condition and exposure, values across treatment channels were then assigned "baseline" (green), "caution" (yellow) and "altered" (red) condition if they were within one, two and greater than two standard deviations from the mean, respectively. Analysis of the detection probability (percent of sites with accurate condition assessment) of each independent metric showed that condition depends on the level of exposure (**Figure 6-16**). Metrics differed in accuracy, with diatoms providing the highest probability, followed by TP content; biomass is a poor condition metric when used alone to detect low P exposure due to high variance in the natural state. Detection probabilities improve appreciably when used in combination, where "caution" is assigned if any metric denotes it or a single metric denotes "altered," and "altered" if two or more metrics fall in that range (Gaiser 2009). The best combination metric was a combination of TP and diatoms, which had both low false positive and high detection probabilities. This finding is in keeping with results from other wetland, stream and lake-based water quality assessment studies that have resulted in national algal-based risk assessments to include both nutrient and compositional information (Stevenson et al. 2012).

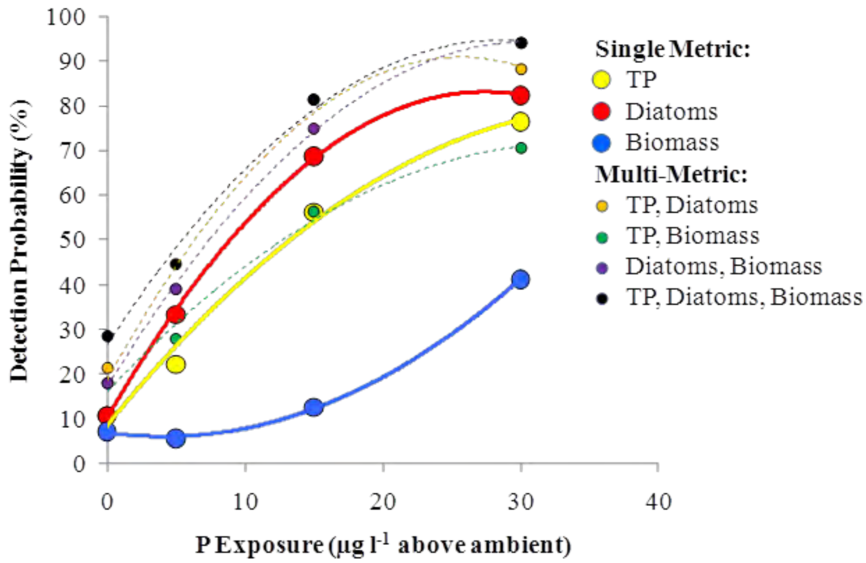


Figure 6-16. Probability of detecting exposure to above ambient P, employing single and multi-metric approaches using data from the experimental dosing facility in ENP. The highest performing metric was the combination of periphyton TP concentration and diatom composition (ratio of weedy to endemic species – orange dotted line), because it provided the best combination of low false positive and highest detection probabilities across exposure levels.

The cost of inclusion of species metrics is commensurate with the improvement in detection accuracy. Periphyton TP and diatom-based assessment used alone yield an 75% and 85% detection probability at highest exposure levels, respectively, and improve accuracy more than 20% when used together (**Figure 6-16**). These metrics represent 10% and 20% of project costs (without including helicopter travel), respectively, since the bulk of assessment costs are in field and laboratory support costs necessary for baseline collections for all metrics. In the recent budget cuts, most of the cost savings were accomplished by dropping the species metric.

Periphyton Assessment of Oligotrophic Status

Application of the improved metrics to the WY 2011 (fall 2010) assessment show similar patterns to those reported in the 2009 SSR (**Figure 6-17**). The TP metric suggests impairment in central WCA 3, while the diatom metric detects impacts along the eastern boundary of ENP likely associated with changes in operations of the L-31W and S-332 structures B, C, and D (see Gaiser et al. 2013 and Bramburger et al. 2013). An important caveat is that the diatom metric also shows impairment in southwestern SRS, but this pattern may be due to the natural increase in P availability in the oligohaline areas of ENP via exposure to marine sources of P (Gaiser et al. 2011; 2012).

The WY 2012 assessment only includes the TP metric, but shows increasing alteration in WCA 3B. However, there is poor detection of water quality impairment in ENP without the diatom information. The side-by-side comparison of the stoplight summary by region, where yellow coding represents areas having >25% yellow or >10% red primary sampling units (PSUs) and red coding represents areas having

>25% red PSUs, clearly shows the level of detection added by diatom inclusion. Temporal patterns of diatoms suggest slow water quality degradation along the SRS and Taylor Slough boundaries, improvement in WCA 2A, and consistent impairment in central WCA 3A (**Figure 6-18**).

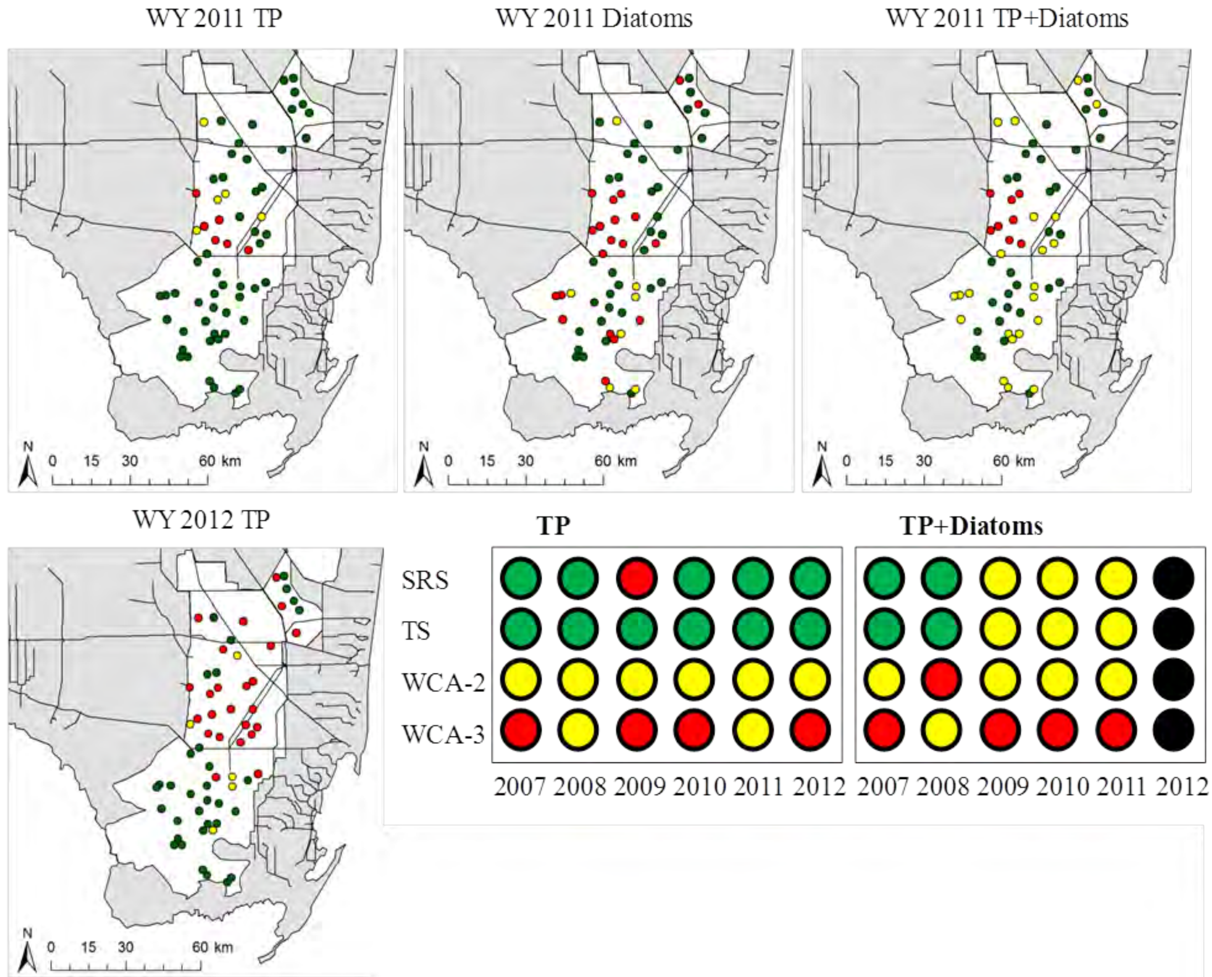


Figure 6-17. Baseline (green), cautionary (yellow), and altered (red) condition at each PSU during wet season sampling using the periphyton TP, diatom and TP + diatom metric in WY 2011 and TP only in WY 2012.

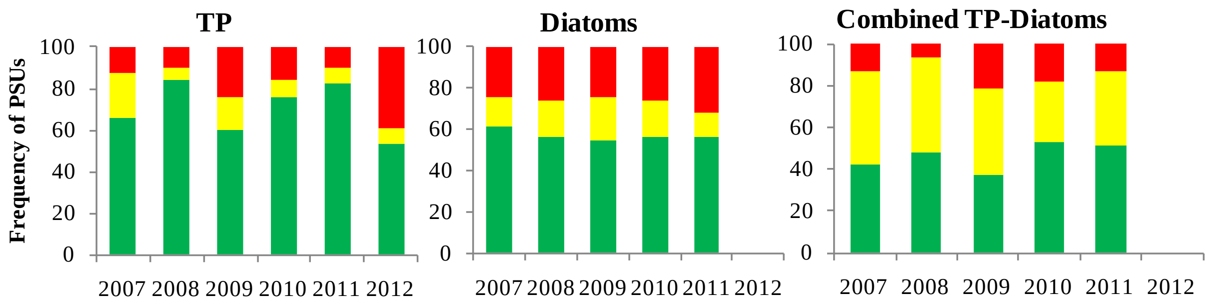


Figure 6-18. Frequency of PSUs (%) in SRS, Taylor Slough, WCA 2 and WCA 3 showing baseline (green), caution (yellow), and altered (red) condition based on TP only, diatoms only, and the combined TP-diatom metric from WY2007–2012.

The interannual pattern in cautionary and altered PSUs was examined relative to water flows, loads and TP concentrations at inflow structures obtained from the SFWMD (**Figure 6-19**). There was no relationship between the condition metrics and flow or load across basins, but there was a strong relationship between the flow-weighted mean inflow TP concentration over the three months prior to sampling and the TP, diatom and TP-diatom metrics (using a second order polynomial model to account for threshold response; R^2=0.29, 0.75, 0.67, respectively). Notably the relationship between inflowing water TP and periphyton TP was stronger than the diatom metric, even though experimental research shows the diatom metric is more sensitive to exposure. Conclusions based on other experimental work suggest that diatoms retain the signal of historical TP exposure (from inflow TP, legacy TP and local biogeochemical cycling of TP) longer than the concentrations within the periphyton mat, suggesting further analysis of lags in response to historical loading, including inflow TP, legacy P and the biogeochemical cyclying of TP, are necessary. The high correlation between inflow concentration and condition status across each wetland is surprising, since it includes PSUs well to the interior. The full interpretation of the periphyton metric for marsh impairment must consider inflow and legacy TP, local biogeochemical processes and other factors (hydroperiod, soil compaction and subsidence) influencing periphyton ecology. Analysis at the PSU level may resolve interpretation of sources for impairment.

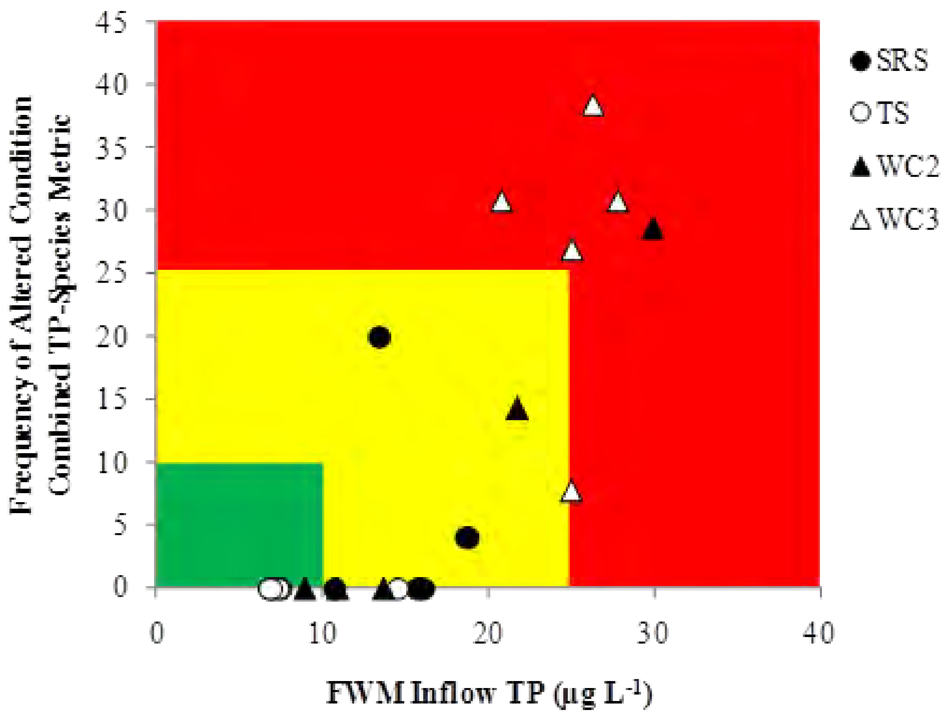


Figure 6-19. Relationship between the frequency (%) of PSUs in altered condition (using the combined TP-species metric) and the flow-weighted mean (FWM) TP concentration (in micrograms per liter [$\mu g\ L^{-1}$]) that the periphyton species have been exposed to 3 months prior to sampling in SRS, Taylor Slough (TS), WCA 2 (WC2) and WCA 3 (WC3) from WY 2007–WY 2011. The exposure concentration is due a combination of inflow TP, legacy P and the biogeochemical cycling of P. Green, yellow and red shading indicate the frequencies necessary to report the entire wetland (or a site over many years) in baseline, cautionary and altered condition.

Developments in Periphyton-based Hydrological Assessment

Much research has been directed at understanding periphyton response to hydrological variability, both through experimental studies and using long-term assessment data that include wet and dry years. Experimental studies have been useful to control for water depth and/or inundation timing while holding nutrient concentrations constant (Thomas et al. 2006, Gottlieb et al. 2006), which help resolve stressor-response relationships in the natural environment where the two drivers are often correlated. These studies show that while biomass often declines with water depth and inundation timing, this relationship has high variance that is often also influenced by a commensurate translocation of nutrients. The strongest independent relationships appear to be in the algal species composition, where filamentous cyanobacteria increase in abundance in dry conditions and diatoms and green algae flourish in longer-hydroperiod settings (Gottlieb et al. 2006). Within the diatom community, Lee et al. (2013) were able to identify a suite of taxa that appear to be reliable indicators of dry conditions and others that are absent from sites with a history of drying. These species responses were very strong: diatoms explain 63% of variance in hydroperiod, and provide inferences with 55-day accuracy. Maps were generated that show the strong correspondence between diatom-predicted hydroperiod and measured hydroperiod for given MAP years (**Figure 6-20**). Because diatom responses to hydrology are now well understood, metrics for hydrologic change assessment can be developed using much the same methodology as the water quality indicators. However, the road block to this development continues to be the absence of secure target/baseline conditions necessary for indicator development. Diatoms are poorly preserved in sediment records, offering little evidence for paleohydrologic interpretations (Sanchez et al. 2013). A solution may be to use targets modeled in natural system approaches to show the deviation in diatom-inferred hydroperiod from what would be predicted given the rainfall (and existing climate conditions) of a particular MAP assessment year(s).

Linkages to Upper Trophic Levels

The conceptual ecological models linking periphyton attributes to ecosystem structure and function focus on periphyton as the base of the food web, supplying energy that ultimately reaches megafaunal indicators such as wading birds and alligators. Research on primary and secondary consumers (which eat periphyton and are prey for higher consumers) has shown that periphyton structure, abundance and composition are important regulators of consumer density and diversity, and explain variance in consumer response to hydrological disturbance (Sargeant et al. 2010, 2011). Notably, the periphyton metrics making the largest contribution to interpreting hydrologic controls on consumer densities are biomass and the ratio of edible (green algae and diatoms) to inedible (filamentous blue-green algae) algae, further corroborating the importance of inclusion of species-based approaches to periphyton assessment in the Everglades.

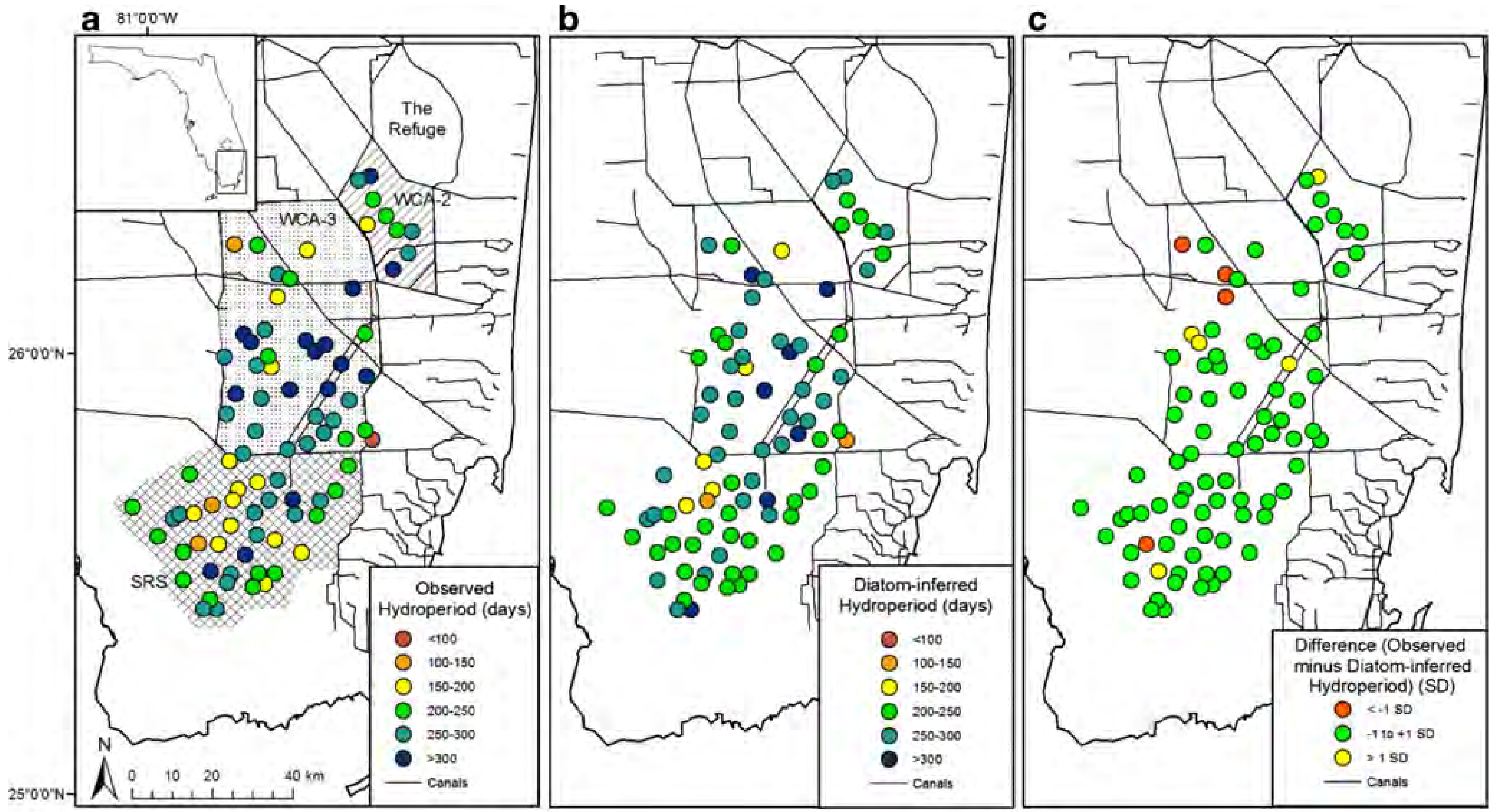


Figure 6-20. (a) Observed, (b) diatom-inferred, and (c) difference (observed minus diatom-inferred values) maps of hydroperiod for WY 2007, from Lee et al. (2013). Black values in difference map (c) indicate values within one standard deviation of observed values. The standard deviation of hydroperiod is 61 days.

VEGETATION MAPPING PROJECT

Introduction

The objective of the RECOVER vegetation mapping component is to produce spatially and thematically accurate prerestoration reference condition vegetation maps to be used in monitoring the spatial extent, pattern and proportion of plant communities within the GE wetlands. A completed vegetation map is an essential tool that would allow resource managers to assess the CERP restoration results (e.g., CEPP), to better understand the long term implications of other environmental impacts (i.e., sea level rise, hurricanes and dry rainfall periods [e.g., WY 2009–WY 2013]), and to answer management questions relevant to individual portions of the Everglades Protection Area (i.e., wilderness areas management, whether overland recreation vehicle use is sustainable, changes in water quality, the presence of invasive species, and long-term operational effects [e.g., ERTP]).

RECOVER vegetation maps have been completed for 4,420 square kilometers (km^2) of the GE, including the WCA 1, Rotenberger Wildlife Management Area (WMA), WCA 2A, WCA 3, the C-111 and southeastern Miami-Dade wetlands, and the northern and western portions of SRS in ENP (**Figure 6-21**). Remaining regions to be mapped include J.W. Corbett WMA, Holey Land WMA, Pal-Mar WMA, the rest of ENP, BCNP, and Biscayne National Park (**Figure 6-21**).

ENP is the third largest national park in the continental United States and the largest park east of the Mississippi, spanning 6,107 km^2 of south Florida. The USACE, in conjunction with the National Park Service, are currently mapping the southern and eastern portions of SRS within ENP (**Figure 6-21**). Immediately adjacent to ENP, BCNP encompasses 2,916 km^2 of critical lands to be mapped immediately following the completion of ENP. Complete and accurate vegetation maps of these critically important regions of the Everglades do not currently exist. Consequently, the successful completion of the current RECOVER vegetation mapping effort within ENP and proposed future mapping of BCNP are vital components for assessing the potential impacts of long-term environmental change and CERP.

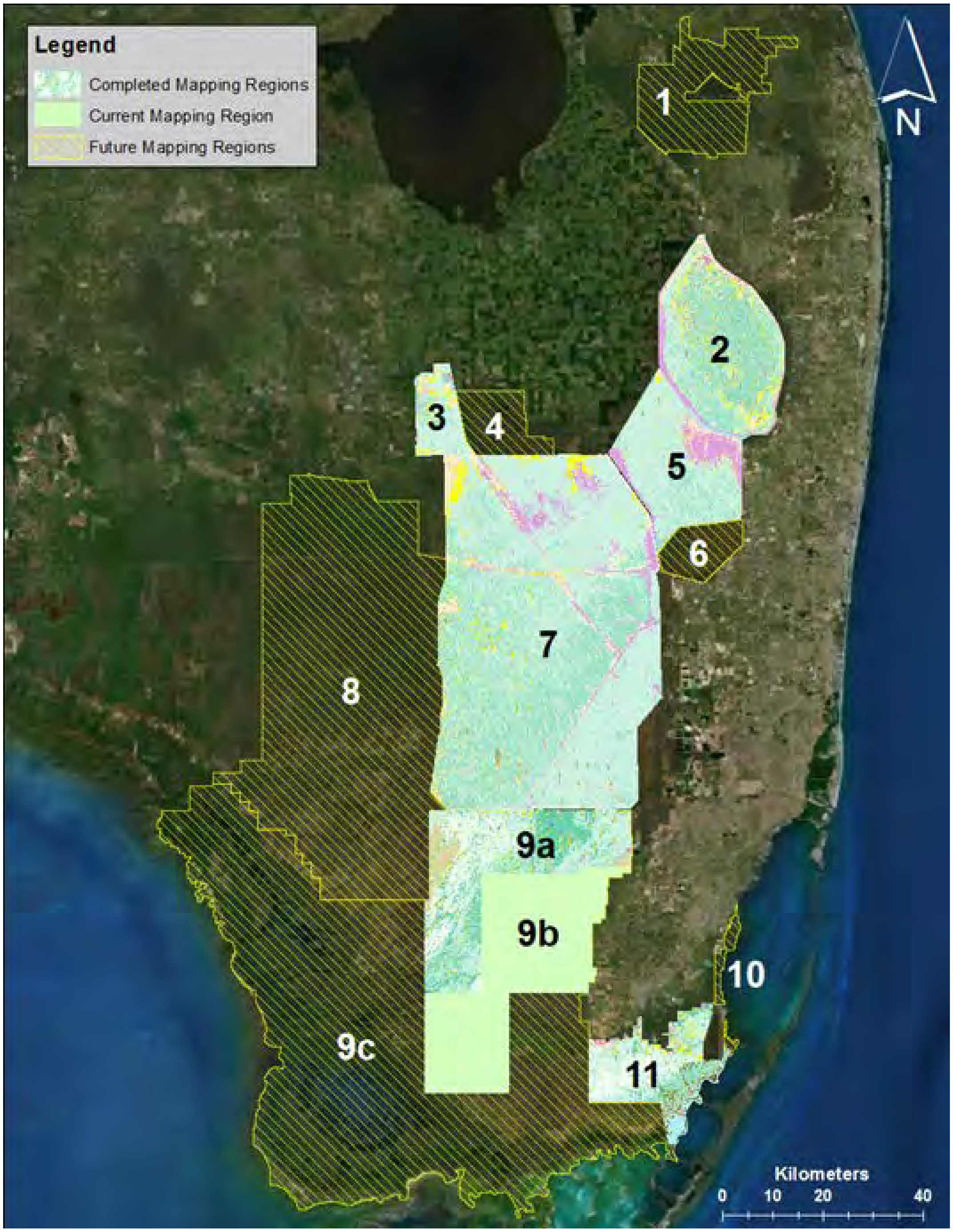


Figure 6-21. Mapping of 4,420 km^2 of the GE has been completed, including WCA 1 (2), Rotenberger WMA (3), WCA 2A (5), WCA 3 (7), the C-111 and southeastern Miami-Dade wetlands (11), and the northern and western portions of SRS in ENP (9a). The current mapping effort is focused on the southern and eastern portions of SRS (9b). Future areas to be mapped include the remaining portion of ENP (9c), the eastern portion of BCNP (8), Biscayne National Park (10), J.W. Corbett and Pal-Mar WMAs (1), the Holey Land WMA (4), and WCA 2B (6).

Methodology

Vegetative community types are characterized for every 2,500-square meter (0.25 hectare [ha]) grid using high resolution digital aerial photography acquired in April 2009. Photo interpretation of each grid cell is accomplished by superimposing the two dimensional grid over the three-dimensional color-infrared aerial imagery. A hierarchical classification scheme (Rutchey et al., 2006) is used to classify the vegetation of each grid cell into 284 distinct community types. The photo-interpretation portion of the project is being conducted by the USACE Jacksonville District.

The field assessment and botanical training components of this project are being implemented by the National Park Service's South Florida and Caribbean Network. The network's personnel provide necessary training and expertise regarding aspects of south Florida ecology, botany, and plant identification in support of the USACE Jacksonville District mappers in their effort to interpret and classify the land cover within ENP and BCNP. In addition to the interactive training, the network also provides field data that is utilized to both instruct the mapping process and assess the final map accuracy. The field effort will generate over 4,000 spatially explicit training point locations and approximately 1,000 accuracy assessment locations documenting the existing vegetation. The training point data are necessary for the development of the photo interpretation key that serves as a guide for the image classification process. The accuracy assessment field data collection is also of vital importance as it is used to validate the mapping effort and ultimately informs researchers and managers regarding suitable applications of the final vegetation maps. Vegetation maps of ENP and parts of BCNP produced by this project are expected to have an estimated classification accuracy greater than 80% with 90% confidence.

Assessing Landscape Change with RECOVER Vegetation Maps

Pre-RECOVER vegetation maps for WCA 2A and WCA 3 were compared to the completed RECOVER vegetation maps of these respective areas for the purpose of documenting cattail community changes associated with phosphorus enrichment of Everglades' soils. The cattail change analysis was conducted utilizing pre-RECOVER maps indicating cattail conditions within both WCAs for 1995. The 1995 maps were created from aerial photography utilizing a vector mapping approach which required delineation of precise boundaries between distinct landscape features. In contrast, the RECOVER mapping methodology emphasized the characterization of predetermined grid cell boundaries. Quantitative differences between the two distinct mapping methodologies were assessed by converting the 1995 vector map data to the equivalent RECOVER fixed-grid mapping product (**Figure 6-22**).

As a result of the accounting differences between the two distinct mapping methodologies, direct comparisons between the original 1995 vector data and the original 2003/2004 RECOVER products were not attempted. Instead, the 1995 grid-converted map data (RECOVER equivalent product) were compared to the RECOVER maps in order to assess cattail change. This comparative analysis demonstrated that the combined cattail classes had expanded by 2.4% and 0.7% for WCA 2A and WCA 3, respectively (**Figure 6-23**). WCA 2A demonstrated a net gain of 1,023.5 ha of cattail during the period from 1995 to 2003. WCA 3 demonstrated a net gain of 1,609.0 ha of cattail during the period from 1995 to 2004. The location of cattail expansion within both regions appears to be primarily adjacent to previously existing regions of cattail, along levees and canals. Further research is needed to ascertain the driving mechanisms that resulted in the cattail expansion demonstrated in WCA 2A and WCA 3. Nonetheless, these data establish a trend from which future vegetation mapping products can be compared and to help to ascertain if restoration efforts are successful in preserving and restoring predrainage landscape conditions.

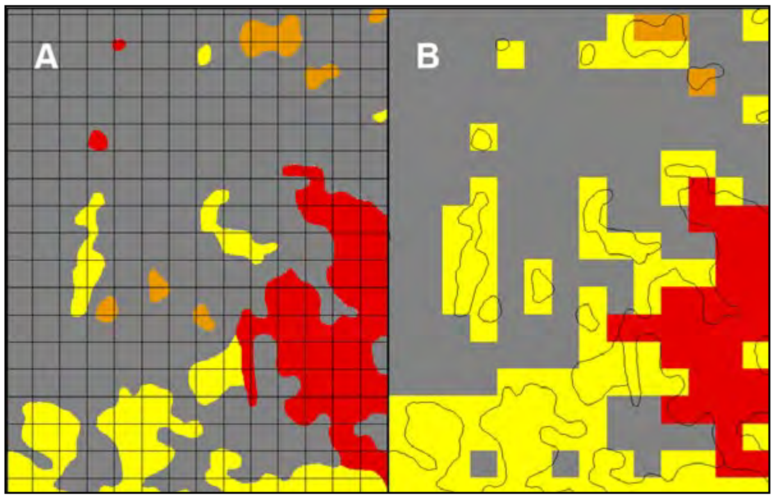


Figure 6-22. (A) The 1995 WCA 2A vector cattail map with the RECOVER mapping grid overlaid and (B) the 1995 WCA 2A cattail data converted to the equivalent RECOVER grid mapping product with the original vector lines overlaid. Land cover classes depicted above include monotypic cattail (red), dominant cattail (orange), sparse cattail (yellow), and other vegetation (gray). Conversion of the pre-RECOVER vector maps to the equivalent RECOVER mapping product resulted in a net increase in the accounting of cattail for both WCA 2A and WCA 3. The mean difference in cattail accounting was determined to be +3.9 percent (Table 6-7). The net increase in cattail accounting is largely due to the apparent expansion of sparse cattail (yellow) that results when precise feature boundaries are expanded to fill a relatively coarse grid.

Table 6-7. Impact of vector versus RECOVER grid mapping methodology on cattail accounting.

Region	Cover Class	1995 Map Data		Net Difference	
		Vector Area (ha)	Grid Area (ha)	Area (ha)	Percent
WCA 2A	Combined Cattail Class	9,310.40	10,773.50	+1,463.1	+3.5%
WCA 3	Combined Cattail Class	20,669.78	30,739.25	+10,069.5	+4.3%
				Mean Difference	+3.9%

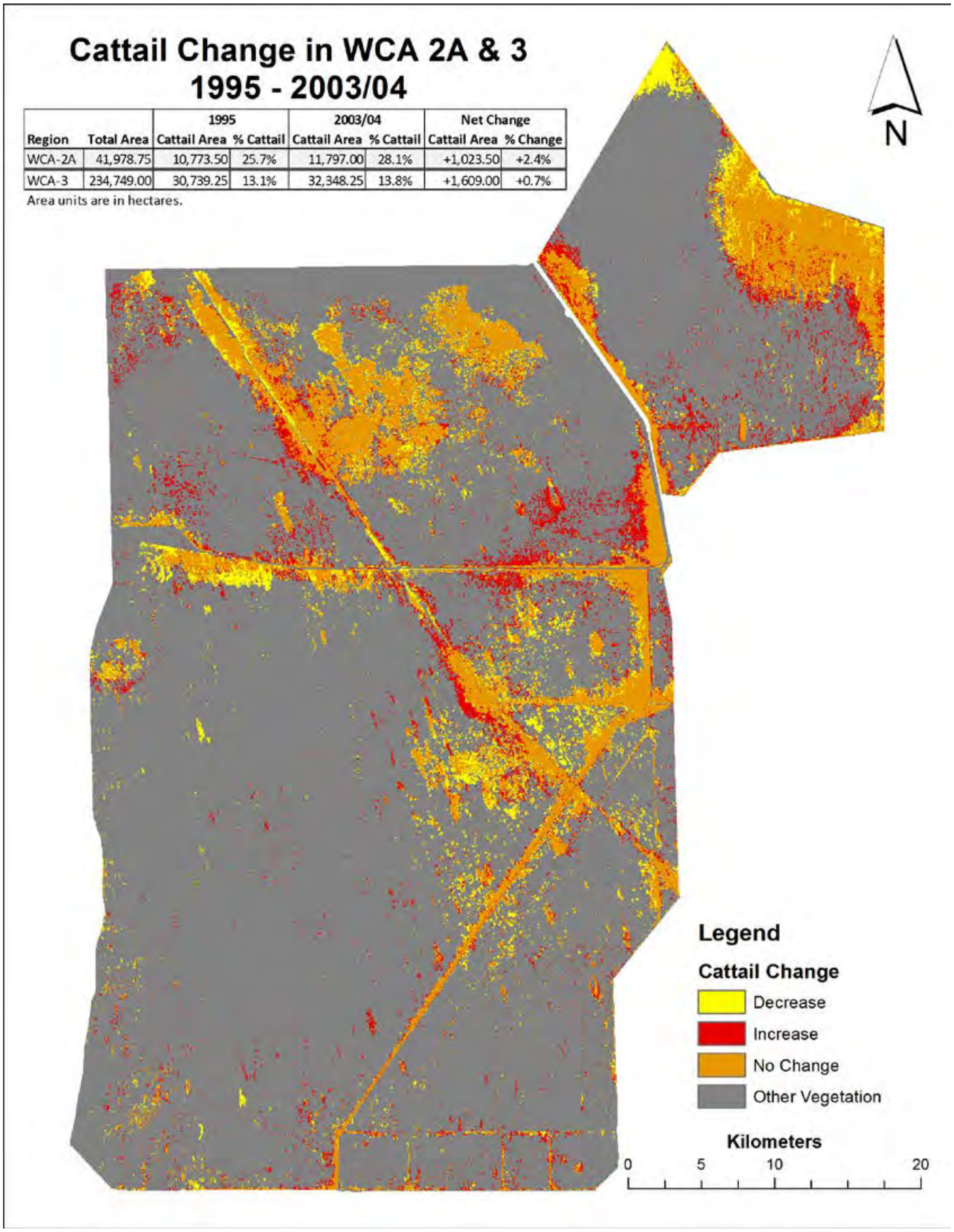


Figure 6-23. Pre-RECOVER vegetation maps for WCA 2A and WCA 3 were converted to an equivalent RECOVER grid mapping product and compared to the RECOVER vegetation maps of these respective areas for the purpose of documenting cattail community change. WCA 2A demonstrated a net gain of 1,023 hectares of cattail or a 2.4 percent gain during the period from 1995 to 2003. WCA 3 demonstrated a net gain of 1,609 hectares of cattail or a 0.7 percent gain during the period from 1995 to 2004.

Additional Uses of the RECOVER Vegetation Mapping Products

RECOVER vegetation maps and site specific field data have been used in a number of concurrent Everglades research projects. The Everglades Landscape Vegetation Succession (ELVeS; SFNRC 2011) model and Florida International University's projects using remotely sensed satellite data to map Everglades plant communities both use the site specific field data created by the South Florida and Caribbean Network to help parameterize their products or to train computer algorithms. The vegetation map products are also being utilized by the Natural Resource Condition Assessment for testing habitat-based differences in exotic species presence within ENP. The RECOVER vegetation maps present a complete vegetation survey of the entire GE and serve as the basis for which site specific vegetation and landscape monitoring and assessment projects can be interpreted within the context of the entire Everglades system. For example, the ridge and slough monitoring primary study units across the GE landscape, Picayune Strand vegetation data (RECOVER 2011), Biscayne Bay Coastal Wetlands vegetation data (RECOVER 2011), Everglades' fire history (Smith et al. in review), and the RECOVER historic tree island change maps (RECOVER 2013), can be compared to the base vegetation map to draw more specific status and trends results and potentially inform why changes are occurring.

RIDGE AND SLOUGH LANDSCAPE

Viewed from above, e.g., from a passing plane, the Everglades landscape displays a streaked, north to south orientation. This is the ridge and slough, a region that features a flow-parallel series of elongated sawgrass strands separated by similarly oriented aquatic marshes with floating-leaved or sparsely emergent plants. The ridge and slough landscape is thought to be at least in part self-organized, i.e., created by feedbacks between the local biota, the peat soils, and the peculiar Everglades water regime (Cohen et al. 2011, Larsen et al. 2011) and is considered to be a defining characteristic of the pre-drainage Everglades (Ogden et al. 2005).

Ridge and Slough Patterning in the Interior Everglades Peatland

Patterned as it is, the Ridge-and-Slough has undergone much structural change over the last century. McVoy et al. (2011) recently showed that (1) vegetation pattern in the pre-drainage Everglades reflected the ecosystem's underlying physiography (sawgrass on relative high elevations, aquatic plants on lower ones); (2) local contrasts in both vegetation and topography in the pre-drainage marsh were sharper than they are today (ridges are typically 30 centimeters (cm) or more above sloughs); and (3) the structure of the ridge and slough has experienced variable degradation across the vast wetland during the late twentieth and early twenty-first centuries.

What mechanism best explains this patterned landscape's initiation, maintenance, and degradation? Proposed models fall into three groups: those that (1) emphasize sediment entrainment and deposition (Larsen et al. 2007, Larsen and Harvey 2010), (2) are based on transpiration-driven nutrient concentration (Ross et al. 2006, Cheng et al. 2011), or (3) simulate the ability of the landscape to route water downstream (i.e., hydrologic competence; Givnish et al. 2008, Watts et al. 2010, Cohen et al. 2011, Heffernan et al. 2013). Understanding how these mechanisms produce patterned ridges and sloughs, either singly or in combination, is important because they suggest different strategies with respect to water management. Given a fixed volume of water, managers may be forced to decide whether to generate high flow periods, which the sediment entrainment hypothesis suggests are necessary, or to forgo these periods in favor of extended inundation, as suggested by the discharge competence hypothesis.

Resolution of the underlying mechanisms that these models describe, which may include elements of all three major theories, requires that they be tested against empirical data from the Everglades itself, i.e., the spatiotemporal patterns in degradation among basins subject to different management. During the last two years, twenty-seven PSUs distributed throughout the peatlands of the Everglades interior were visited. Within each 10-square kilometer (km) PSU, water depth and plant species cover was measured at at least 135 randomly selected locations. These data were used to characterize spatial structure in surface elevation and species composition within each unit, and facilitated comparisons among hydrologic compartments within the Everglades system. The particular interest was in what these spatial patterns might indicate about the dynamics of the degradation process, i.e., whether topographic and vegetation structure declined in tandem, or whether degradation in one was a leading indicator of future degradation in the other (**Figure 6-24**).

A healthy ridge and slough system should have high variation in soil surface elevation, as well as maximum differentiation in vegetation composition between sawgrass ridges and adjacent sloughs. The standard deviation of elevation was used as a metric of topographic condition. Sampled PSUs fell into two groups, one with very high values for the topography metric and the other with much lower values. Developing a metric to express variation in vegetation between ridge and slough involved several steps. A multivariate ordination technique (non-metric multidimensional scaling) was applied to the vegetation data, thereby representing the compositional differences among all sites within a PSU. Next, a procedure called K-means clustering was used to coerce the ordination into two distinct clusters; vegetation community distinctiveness, a metric of ridge and slough vegetation condition, was a statistic that represented the distance between clusters. In all cases, one of the clusters indeed turned out to be sawgrass dominated, justifying the approach. **Figure 6-25** illustrates the landscape condition of the sampled PSUs. Seven were in good condition based on both metrics, fifteen were in poor condition based on both metrics, and five were in good condition with respect to vegetation pattern but poor condition with respect to topographic structure. No area combined degraded vegetation with intact topography. All PSUs in the southern portion of WCA 3A exhibited good ridge and slough vegetation pattern, though two of them had degraded topography. All PSUs in ENP and WCA 3B were characterized by degraded topographic structure, though one of the ENP sites still retained ridge and slough vegetation structure. These results, which are based on a small subset of the full data that will be available once the full set of 80 PSUs has been examined, provide some preliminary support to the view that changes in topography are leading indicators of changes in vegetation composition and structure.

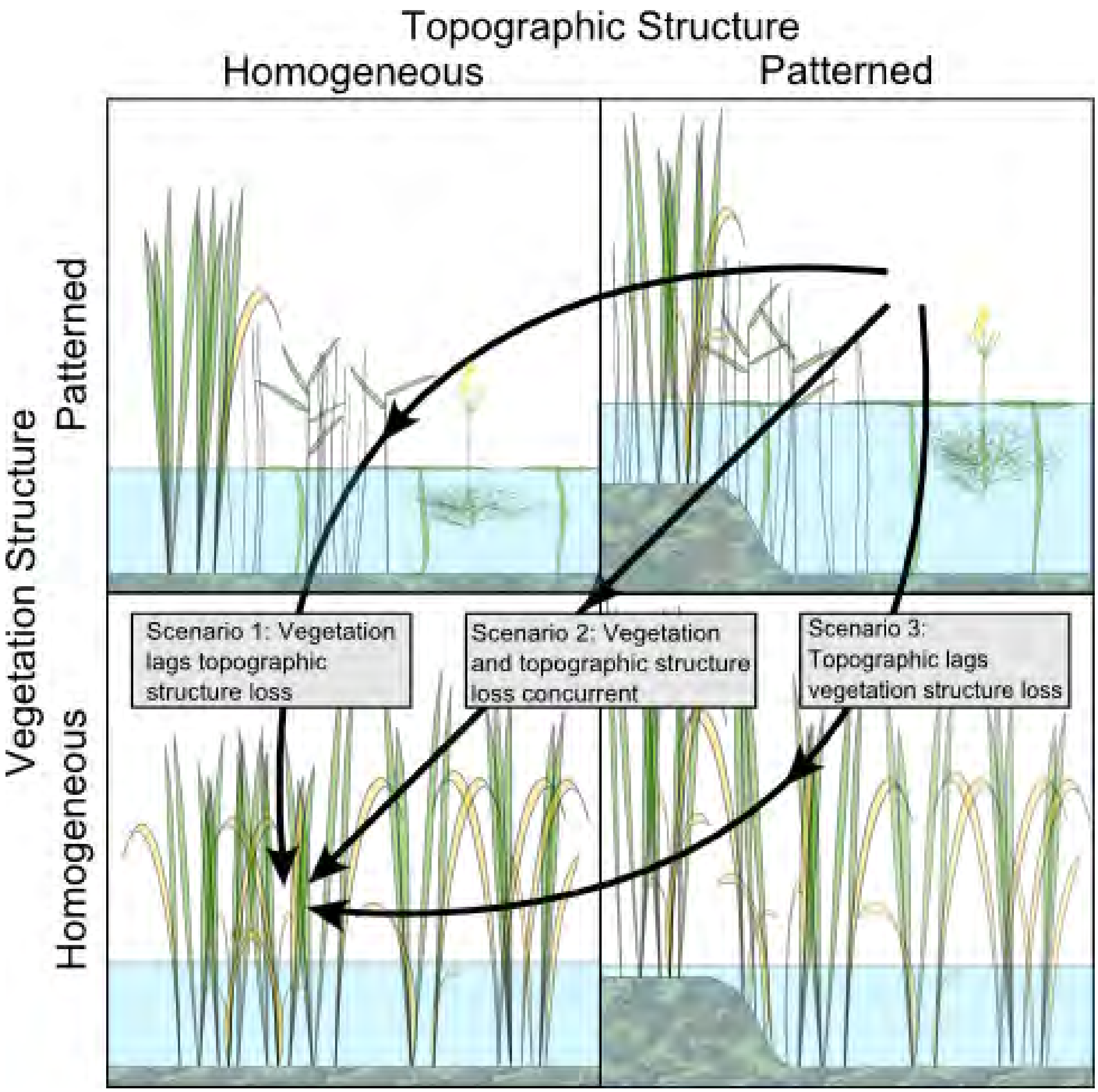


Figure 6-24. Possible pathways of microtopographic and vegetative degradation in the ridge-slough landscape. In one scenario (uppermost arrow) topographic structure is reduced after modification of the hydrologic regime, followed by a lagged response from the vegetation structure; alternatively (lowermost arrow) vegetation patterning may degrade initially in response to modification of the hydrologic regime, followed by a lagged response of topographic patterning; finally (middle arrow) microtopographic flattening and vegetation homogenization may occur, but both lag behind modification of the hydrologic regime. Depending on which pattern accurately describes pathways of ridge-slough degradation, either vegetation or microtopography may serve as a leading indicator of change in the other characteristic.

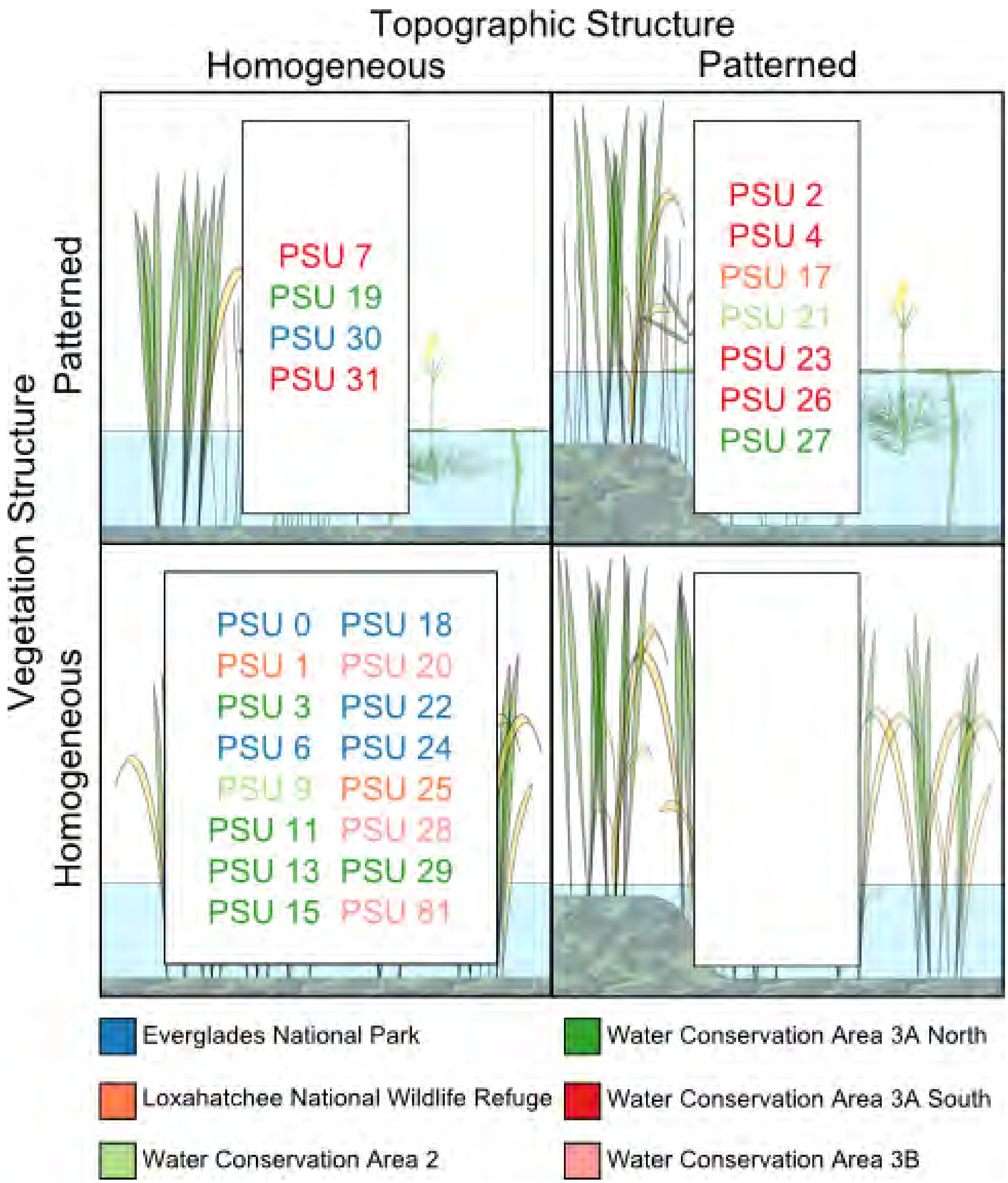


Figure 6-25. Maintenance/degradation of landscape structure in the interior Everglades peatlands: assessment of topographic and vegetation structure in 27 PSUs sampled in 2010–2012.

The rationale behind this study is that fine-scale relationships between hydrology and plant community composition may be amplified and expressed at the landscape scale in one of two stable configurations: (1) a homogeneous wetland lacking in discernible structure or (2) a strongly patterned landscape resembling early surveyors' photos of the Everglades marsh. Models of both the sediment entrainment and hydrologic competence mechanisms suggest the existence and stability of these alternative configurations. Regardless of the mechanism, successful Everglades restoration will need to provide a water regime that favors the second (patterned) alternative. **Figure 6-26** shows the relationship between mean annual water depth in cm over the last 20 years and the vegetation distinctiveness index, the metric of vegetation structure used in this study. These data indicate that good ridge and slough vegetation condition can be found at mean annual water depths of 20 to 50 cm, though a large proportion of sites near the drier end of this range are conspicuously degraded. Based on these albeit preliminary data, maintaining a spatially-averaged long-term mean annual water depth of 35 to 50 cm is recommended for areas in which ridge and slough structure is an important objective.

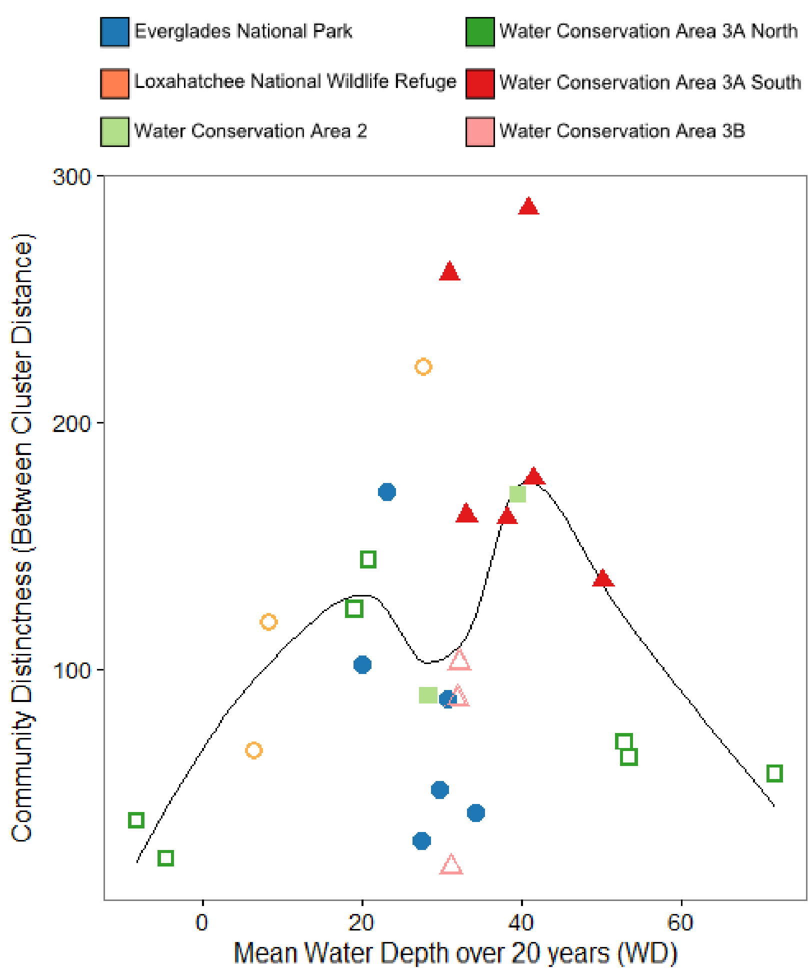


Figure 6-26. Relationship between hydrologic regime (in cm) and vegetation community distinctiveness across PSUs within the historic ridge and slough. Symbols represent PSUs in different hydrologic management basins. Trend line is based on the running mean.

Anisotropic Local Interaction Coupled with Hydrologic Feedback Generate Elongated Ridge-Slough Patterning in the Everglades

The structure and function of many natural landscapes are shaped by complex interactions between biotic and abiotic processes acting at the local and landscape scales. In resource-limited environments, these interactions often give rise to distinct, self-organized patterns (Rietkerk and van de Koppel 2008). The Everglades peatlands is an example of such a resource-limited, patterned landscape in which two broad classes of vegetation communities, ridge and slough, self-organized in regular patches are predominant (**Figure 6-27**).

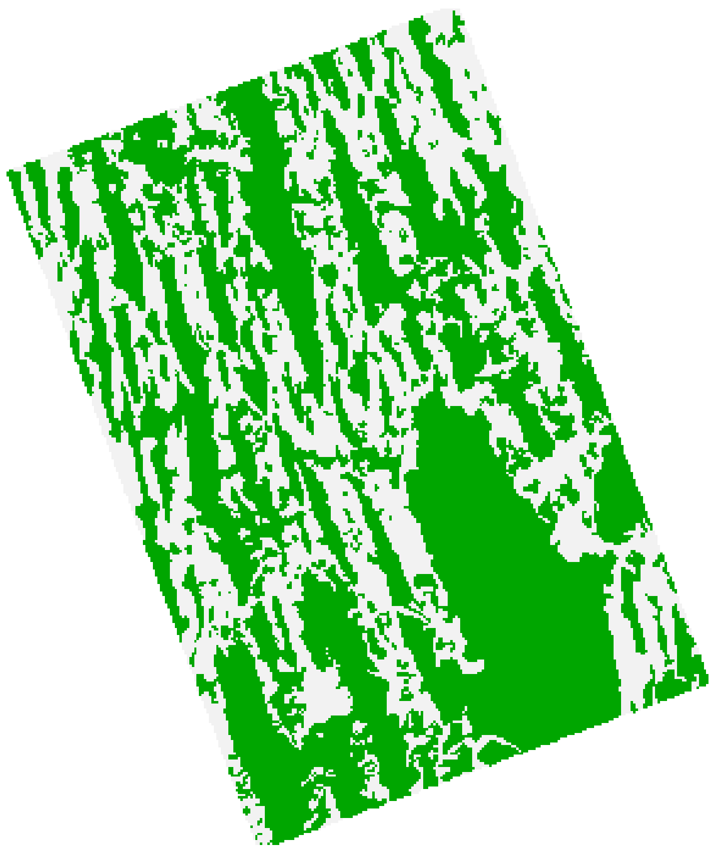


Figure 6-27. The ridge (green) and slough (white) patterned landscape in the best conserved part of the Everglades

In addition to the stable patterning, the Everglades ridge and slough mosaic also consists of a clearly anisotropic (i.e. directional) spatial structure with elongated ridges formed parallel to the direction of the historic flow. While several different hypotheses have been proposed to explain the mechanism behind the formation of stable ridge and slough patterning (e.g., Ross et al. 2006, Larsen et al. 2007, Larsen and Harvey 2010, 2011, Cheng et al. 2011), our study typically focuses on the hypothesis put forth by Cohen et al. (2011). The 'self-organizing canal' hypothesis posits that the anisotropic ridge and slough patterning of the Everglades evolved through coupling of local positive feedbacks with landscape-scale negative feedback operating anisotropically in the direction of the flow. The local feedback at any location is generated by vegetation productivity and is limited to a small spatial kernel (e.g., a 3 x 3 neighborhood of pixels, with a spatial scale of 1 to 10 meters). The global feedback is exerted by the reduced discharge competence of the landscape resulting from the increase in overall density of the ridge-type vegetation.

The Model

Our ridge and slough landscape model is based on a two-state, stochastic (i.e., random) cell-based model, a stochastic cellular automata (SCA) model, in which transition probabilities between ridge and slough stages are dictated by coupled local neighborhood effect and a landscape-scale inhibition function (**Figure 6-28**). The neighborhood effect reflects local facilitation for patch expansion due to primary production and vegetative propagation, while the global inhibition function reflects the negative feedback exerted by change in discharge competence expressed in terms of wetland hydroperiod. **Figure 6-28** presents a schematic of the modeling framework.

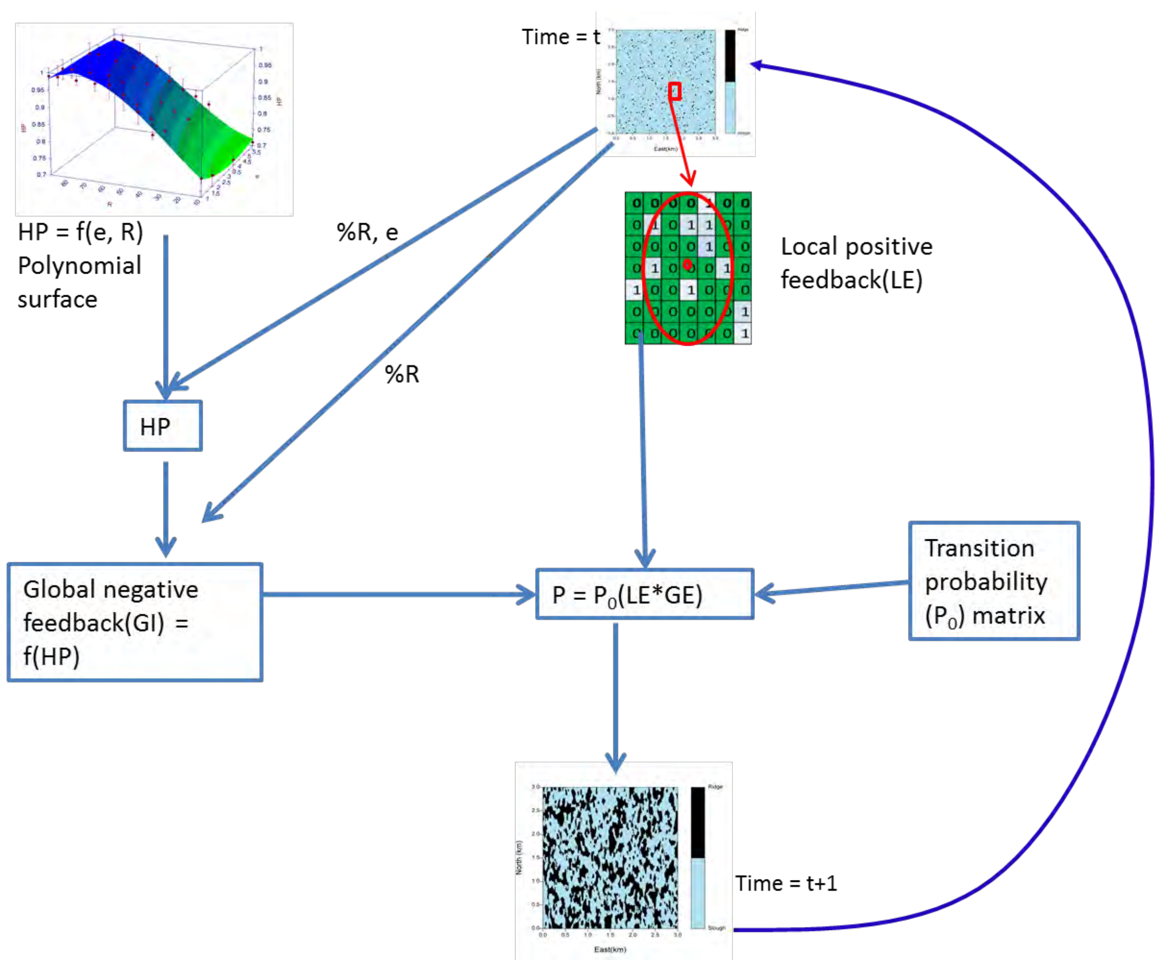


Figure 6-28. Modeling framework of the stochastic cellular automata model for ridge-slough landscape evolution with local and global feedbacks. (Note: see next two pages for an explanation of terms.)

The global feedback operates at the entire landscape scale and reflects the effect of overall patch density and patch orientation on hydroperiod (Kaplan et al. 2012) and the resulting effect on the likelihood of ridge-to-slough and slough-to-ridge transitions. This feedback is based on the theory that wetland hydroperiod (at a given discharge) is a direct result of the landscape's ability to convey water, a property called discharge competence (q). Reduced discharge competence results in elongated wetland hydroperiod, and vice versa. Recent studies have shown that discharge competence in the Everglades is highly controlled by spatial anisotropy (*e*) of ridge and slough patches. In short, pattern exerts control on hydroperiod (Kaplan et al. 2012). In this study, the hydrodynamic modeling procedure outlined by Kaplan et al (2012) was expanded to calculate the hydroperiod of 180 simulated landscapes with various ridge density (*R)* and anisotropy (*e)* combinations. Results suggest that *R* exerts the dominant control on hydroperiod, and that anisotropy is of secondary importance. From the entire set of hydrodynamic models a second-order polynomial function was fited to develop a meta-model of hydroperiod (HP, the fraction of time a location is inundated) as a function of e and *R* (**Figure 6-29**). This model allows us to take any landscape configuration (R and e) and predict the HP that would result.

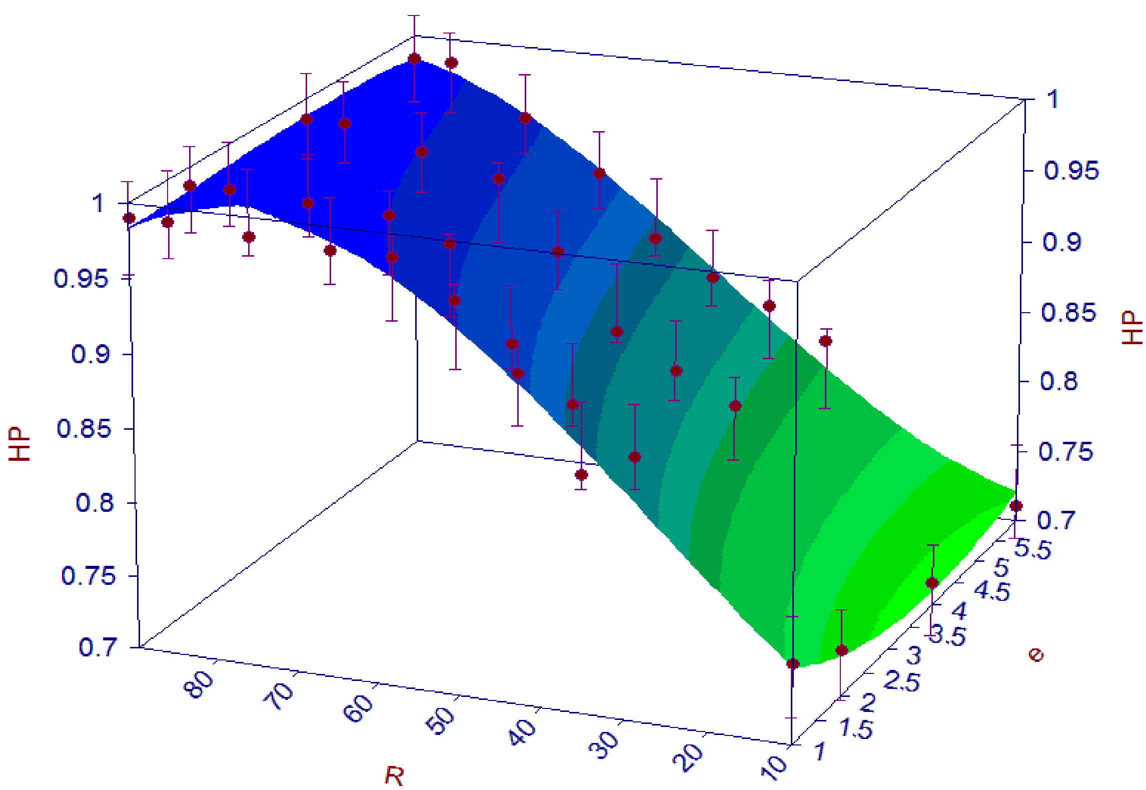


Figure 6-29. Polynomial surface plot of hydroperiod (HP; fraction of time a location is inundated) as function of spatial anisotropy (e) and ridge density (R) based on the results from a hydrodynamic modeling method outlined by Kaplan et al. (2012)

This HP meta-model was then embedded in the cellular automata model as the driver of the global feedback affecting the ridge-to-slough and slough-to-ridge transition probabilities, P_0. The dependence of P_0 on HP is illustrated in **Figure 6-30**, where the equations for ridge-to-slough and slough-to-ridge transition probabilities are as follows:

$$P_{0,R\to S}(HP) = P_{0,max} * \exp(-k_T * (HP_{max} - HP)) \quad (1)$$

$$P_{0,S\to R}(HP) = P_{0,max} * \exp(-k_T * (HP - HP_{min})) \quad (2)$$

where k_T is the decay rate of the base transition probability with changing hydroperiod and HP_{min} and HP_{max} are taken as 0.70 and 1.0, respectively, based on the range of expected and observed HP values in the Everglades (Givnish et al. 2008, Cohen et al. 2011). Note that in this formulation, K_T is assumed to be the same for slough-to-ridge and ridge-to-slough transitions, however the model is flexible to allow for possible nonsymmetry in this relationship.

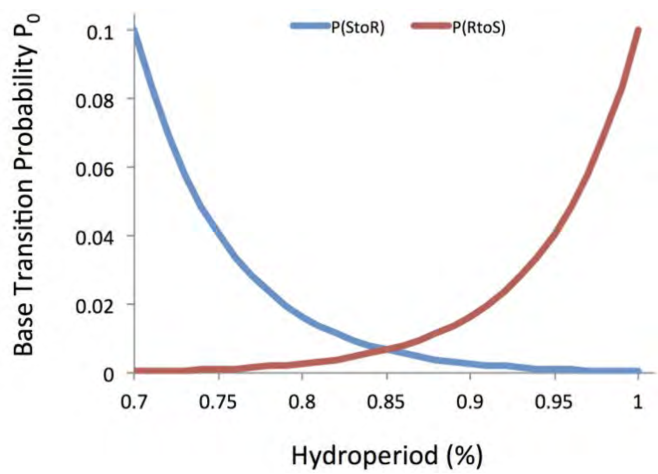


Figure 6-30. Effect of hydroperiod on cell transition probabilities (P_0). Note that the point where the two lines cross is the long-term equilibrium hydroperiod.

The base transition probability predicted by the global HP feedback is then modified by a local feedback function in the SCA model based on the state of the neighboring cells around the cell undergoing transitions, with the overall transitional probability given by the following:

$$P = P_0 * NE \quad (3)$$

where NE is the neighborhood effect. NE is calculated as an exponential function of the fractional similarity of neighboring cells' states to the state of the cell in question (i.e., the percent of cells within the neighborhood that have the same type, denoted as SI), with the strength of the interaction given by the value k_{NE} in **Equation 4** expressed as follows:

$$NE = \exp(-k_{NE} * SI) \quad (4)$$

NE as a function of k_{NE} and SI is shown in **Figure 6-31**. We examined the effect of the local neighborhood shape around each cell, exploring both isotropic (circular) and anisotropic (elliptical) kernels.

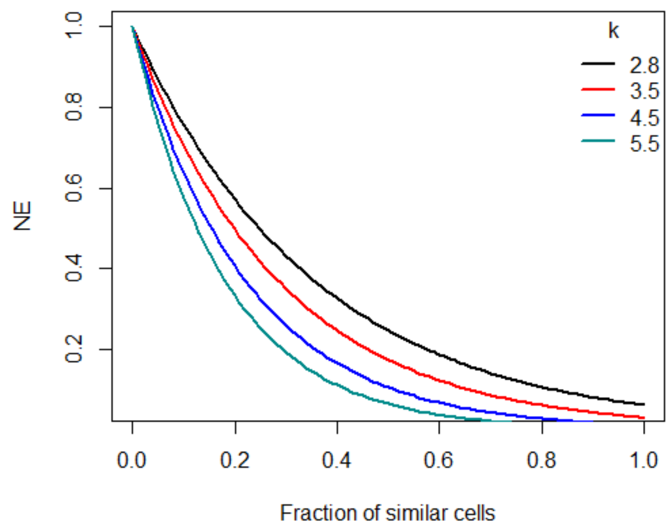


Figure 6-31. Local neighborhood effect (NE) as a function of the fraction of similar adjacent cells within the kernel (k).

At each time-step, the probability of transition of each cell within the domain is calculated based on HP (calculated as a function of R and *e*), which sets the value of P_0, which is finally modified based on neighborhood states according to Equations 1 and 2. At the next time step, a new landscape with different *R* and *e* are produced as some of the cells transition between ridge and slough. This change then results in a new HP and hence the cell transition probabilities for the next time-step. The landscape configuration thus keeps changing, eventually reaching a dynamic equilibrium of *R* and *e*.

Preliminary Results

The preliminary results of the SCA model are presented in **Figure 6-32** and **Figure 6-33**. Note that the resulting landscapes (lower left panel; **Figure 6-32**) closely resemble that of the best conserved ridge and slough landscape of the Everglades (**Figure 6-27**). Furthermore, modeled ridge density and landscape anisotropy (ratio of spatial range in north-south and east-west directions) are in close agreement with real domains (**Figure 6-33**). The statistical agreement between modeled and real landscapes is an area of ongoing MAP/RECOVER funded investigation. One important finding from the initial model is that an elliptical local neighborhood oriented in the north-south direction around each cell is required to produce anisotropic patterning in the SCA model. A suite of potential mechanisms can create this elliptical neighborhood effect, including flow-oriented seed dispersal, sediment entrainment and deposition, and lower P use efficiency in ridges (with emergent vegetation) than in sloughs (with submersed plants). This last mechanism would create slight P enrichment downstream of ridges that would presumably be stimulatory of commensurate ridge expansion.

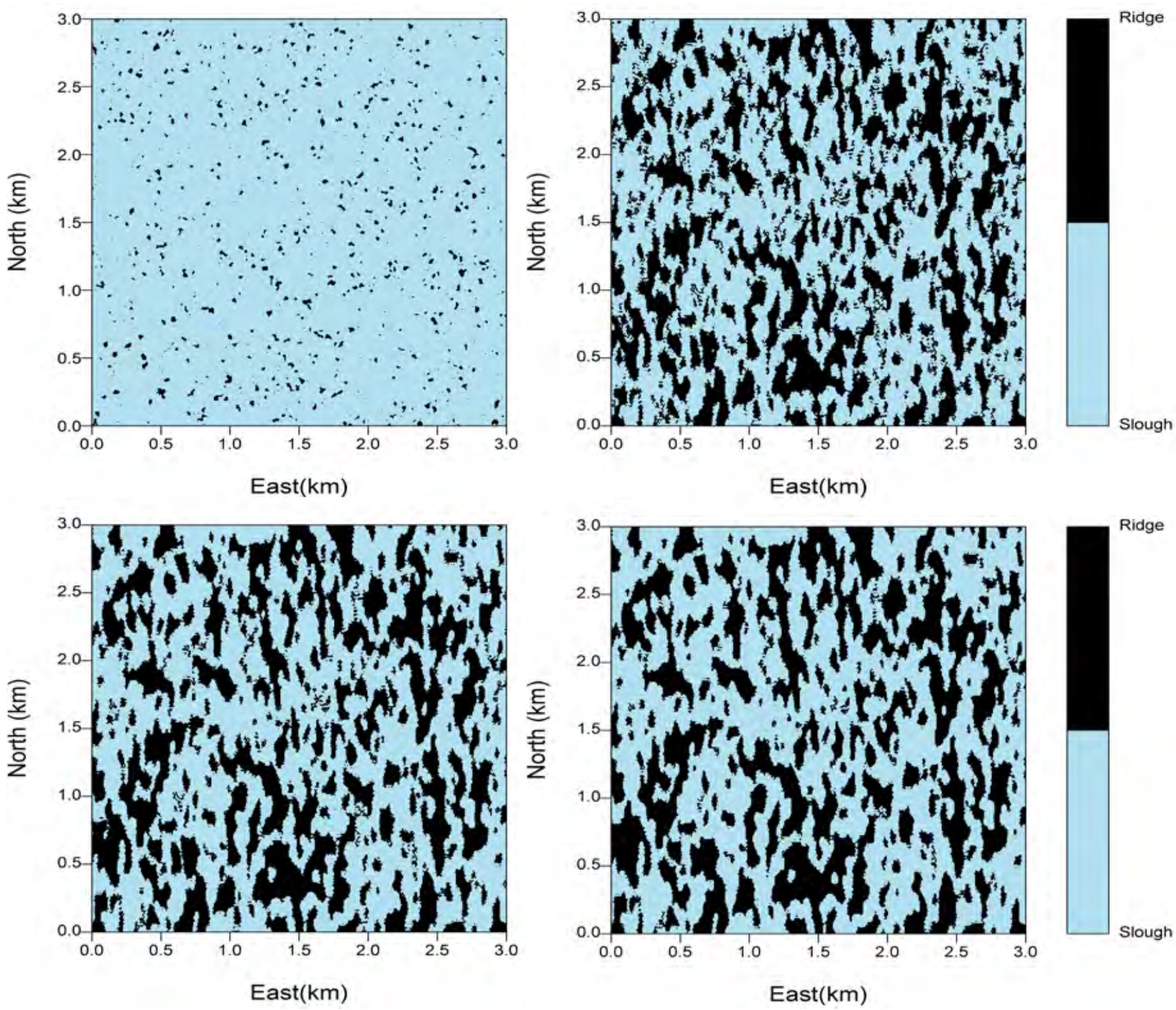


Figure 6-32. Simulated landscapes starting with a random arrangement at time 0 (upper left) and continuing through time (350 at upper right, 750 at lower left) to a stable landscape at 1,500 time steps (lower right). Flow moves from north to south. (Note: km – kilometers.)

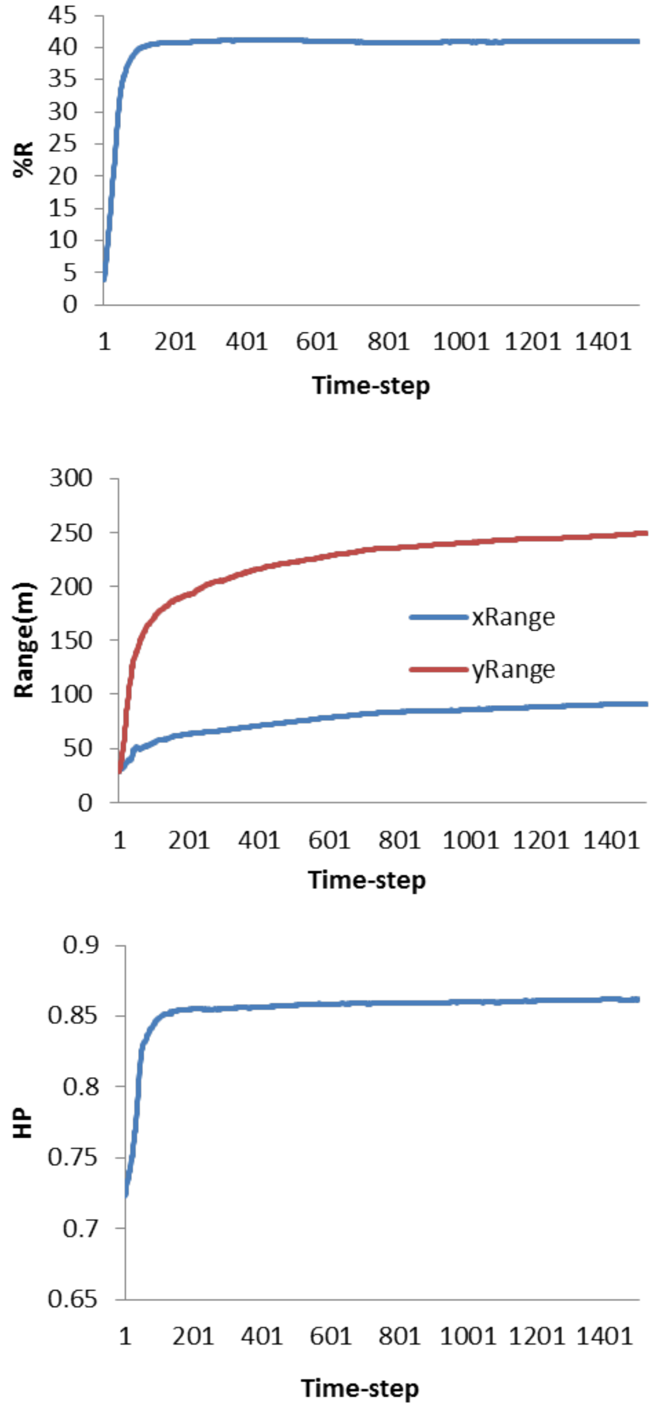


Figure 6-33. Percent ridge density (R) (top), range in meters (m) of spatial autocorrelation in X (East-West) and Y (North-South) directions (middle) and hydroperiod (HP) (bottom) at different model time steps are starting with a random low percent R Landscape.

Evaporation-Driven Phosphorus Transport from Sloughs to Ridges as a Mechanism for Phosphorus Enrichment on Ridges

One mechanism that has been shown to reinforce tree island development is a P subsidy from the adjacent marshes due to the effect of ET on local water level gradients (Ross et al. 2006, Wetzel et al. 2009, Sullivan et al. 2012). In short, tree islands, which are dry when the adjacent marsh (ridges and sloughs) is inundated, pull water from those marshes into the tree islands. As water is “pumped” in from adjacent ecosystems, nutrients move as well, creating a process that focuses phosphorus on tree islands such that total P concentrations can be as much as 10 times higher than in adjacent ecosystems (Wetzel et al. 2009). In the low P environment of the Everglades, this localized P enrichment allows trees to maintain high productivity and keeps tree islands higher than their surrounding habitats. Direct evidence for this process has been observed by measuring water table elevations in tree islands and adjacent ecosystems (ridges and sloughs) during the dry season (Sullivan et al. 2012). Lower water levels under tree islands in response to water demand by trees compared to water levels in adjacent habitats creates a gradient for water and nutrient fluxes.

In 2007, similar relationships were reported between nutrients and soil elevation in ridge and slough habitats (**Figure 6-34**) (Cohen et al. 2007). In short, P concentrations were positively correlated with elevation, and therefore negatively correlated with water depth (**Figure 6-34**). As higher elevations are associated with more productive sawgrass communities (i.e., ridges), it was postulated that higher ET on ridges may also lead to a P “pump”. As with tree islands, this would mean that higher water demand of ridges would lead to water and nutrients moving from sloughs to ridges.

To test this hypothesis, water level recorders were installed in two paired sites (one ridge, one slough at each site, with wells separated by ~150 meters [m]); the elevation difference between ridge and slough soils at the well locations was 17 cm in one case and 9 cm in the other. Measurements of water level fluctuations at 15 minute sampling intervals occurred between summer 2011 and spring 2013 and included periods during which both ridges and sloughs were inundated, periods during which both were exposed, and also periods when ridges were exposed, but sloughs inundated. This range allowed examination of the dynamics of water level differences between well ridge and slough locations under a wide range of hydrologic conditions. Specifically, a hydrologic gradient from the sloughs to the ridges was expected only during the brief period of the year (~30 days per year on average) when ridges are exposed and sloughs are inundated. During this time period, water levels can decline faster in the ridges than sloughs due to differences in specific yield (S_y).

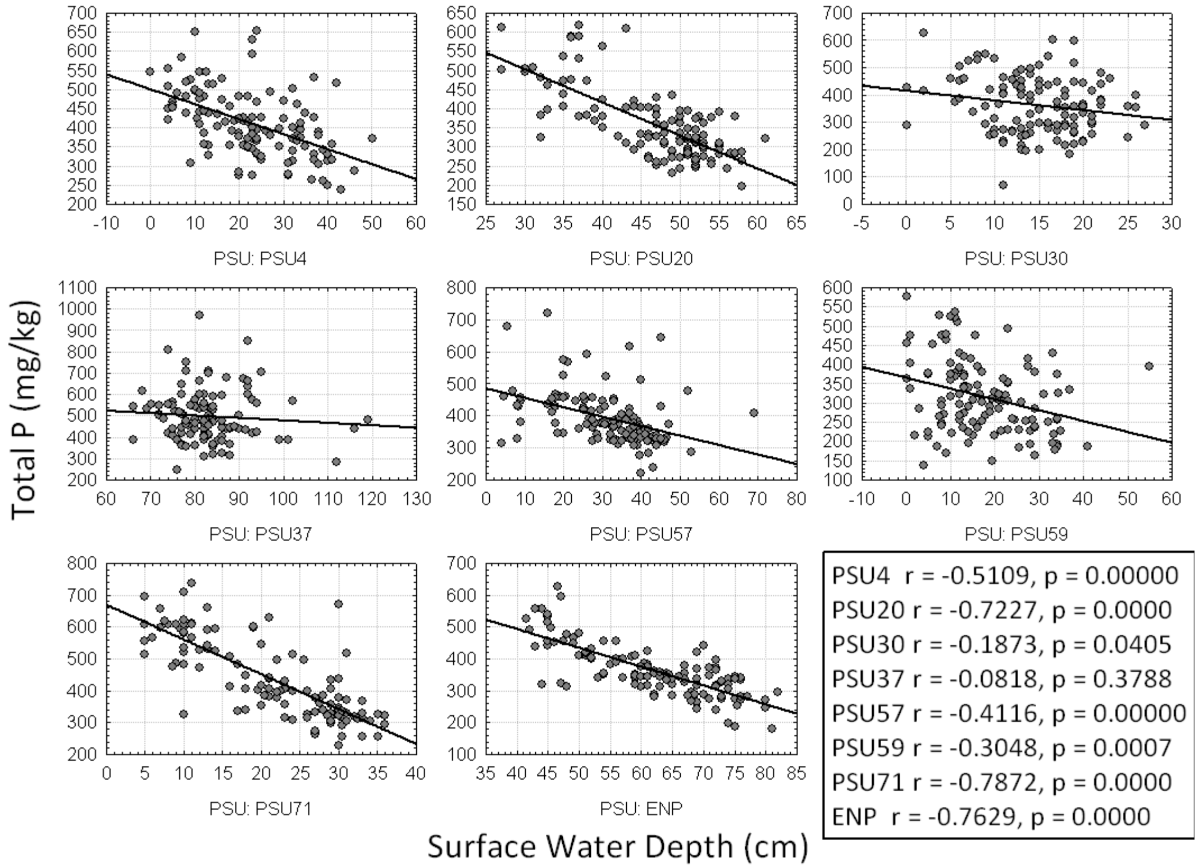


Figure 6-34. Relationship between local instantaneous water depth in cm and soil total P concentrations in milligrams per kilogram (mg/kg) in the ridge-slough landscape; tree islands were excluded. Shallower water means higher soil elevation. These data suggest ridge soils at higher elevation are P enriched compared to directly adjacent sloughs. The association is strongest for locations where the landscape pattern is conserved (PSU 4, 20, 71 and ENP), and weaker where the pattern is degraded.

S_y represents the ratio of a water input (precipitation) or output (ET) to the induced water level change (McLaughlin and Cohen 2013). The soil has large displacement compared to open water so the same magnitude of ET creates a much larger water table decline when the water table is in the soil than when the water table is above the soil. Thus, water levels in the ridges and adjacent slough were expected to be in lock-step during the period when both are inundated (German 2000) (**Figure 6-35A**), but diverge when the sloughs are inundated and ridges exposed (**Figure 6-35B**), since during these times ET demand in the ridges is met by soil water (with specific yield << 1), not surface water (where specific yield ~1). When the water table is above the ridge elevation (**Figure 6-35A**), small diurnal water level variation and strong concordance between ridge and slough water levels was expected. When ridges are exposed and sloughs inundated (**Figure 6-35B**), diurnal water level variation in the sloughs was expected to remain modest but be amplified in the ridges by lower specific yield (i.e., water level variation per water volume change) in the soil than the water. This provides a hydrologic gradient to deliver water from sloughs to ridges, and thereby enrich P on ridges. These high precision water level measurements yield signals where, during daylight hours, plant demand for water means a small but measurable water level decline, while at night, when ET is negligible, water levels rebound from groundwater fluxes or remain constant.

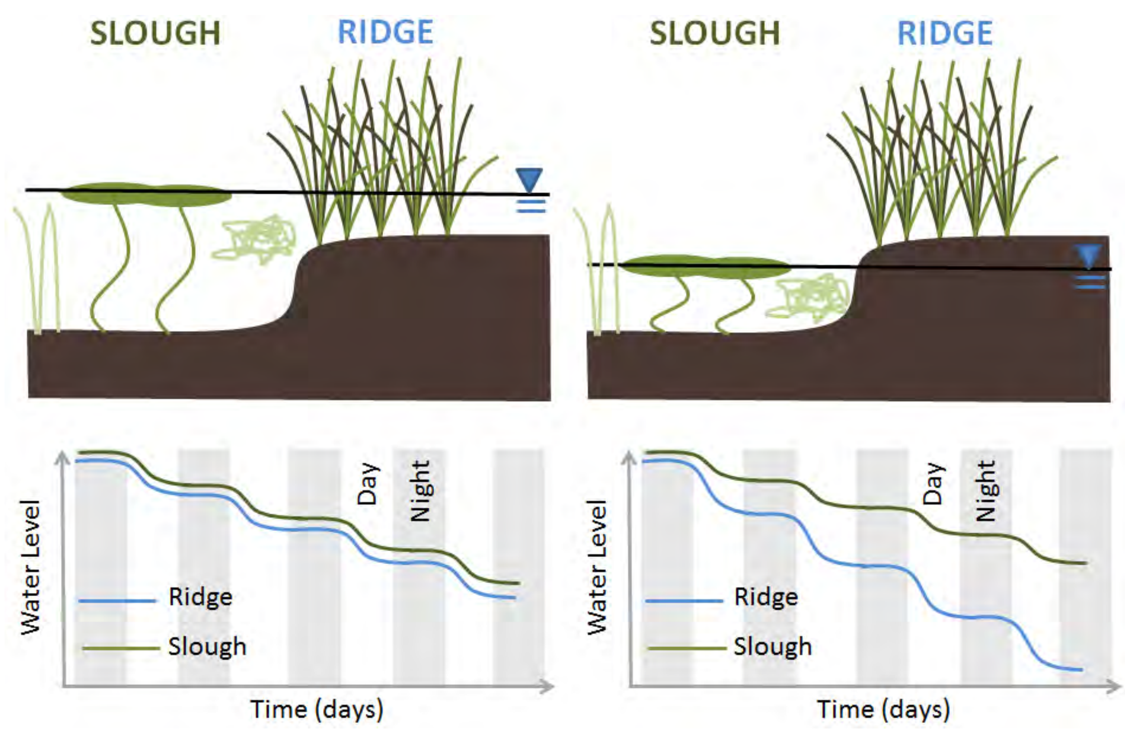


Figure 6-35. When the water table is above the ridge elevation (A), small diurnal water level variation and strong concordance between ridge and slough water levels was expected. When ridges are exposed and sloughs inundated (B), diurnal water level variation in the sloughs was expected to remain modest but be amplified in the ridges by lower specific yield (i.e., water level variation per water volume change) in the soil than the water. This provides a hydrologic gradient to deliver water from sloughs to ridges, and thereby enrich P on ridges.

As expected, water level variation was modest and in lock-step between ridges and sloughs when the landscape was inundated (**Figure 6-36A** showing two representative sets of 3 days from over 500 continuous days of measurements). Diel variation was small (~0.1 cm, consistent with winter ET). However, in contrast to predictions, daily water level variations occurred in lock-step between ridge and slough wells during periods when ridges were exposed (**Figure 6-36B**). Moreover, diel water level variation in both ridge and sloughs was large (~2 cm), which is far larger than expected for springtime open water ET (0.4 to 0.7 cm; German 2000), indicating an effect of S_y. This has important implications for understanding hydrologic connectivity between ridges and sloughs and for understanding the mechanism of P enrichment on ridges. First, it means that there is no period of the year when water levels in the ridges diverge appreciably from water levels in the sloughs. As such, there is no sustained hydrologic gradient for moving water laterally from sloughs to ridges. However, the magnitude of the diurnal variation means that the water levels in ridges and sloughs are responding to ET from both ecosystem components; indeed the magnitude of the observed diurnal variation is in between what would be expected for water table declines from open water and from soil. This means that while there is no sustained hydrologic gradient between ridges and sloughs, there must be lateral exchange from sloughs to ridges that occurs rapidly to equilibrate what would otherwise be divergent water levels. If this were not the case, then the same ET rates in each setting would lead to rapidly declining water levels in the ridge (where specific yield << 1) and slower declines in the sloughs (where specific yield is ~ 1). In short, the observation of lockstep variation in water levels indicates that some evaporative water demand in ridges is being met instantaneously by the connected water table between the two communities. We note that measured hydraulic conductivity rates in the peat soils are extremely low, which means that the most likely flowpath for moving water into the ridge soils during this critical period may be through the underlying carbonate aquifer, with additional implications for ecosystem solute budgets, or possibly through other preferential flowpaths (e.g., pipe flow in the peat).

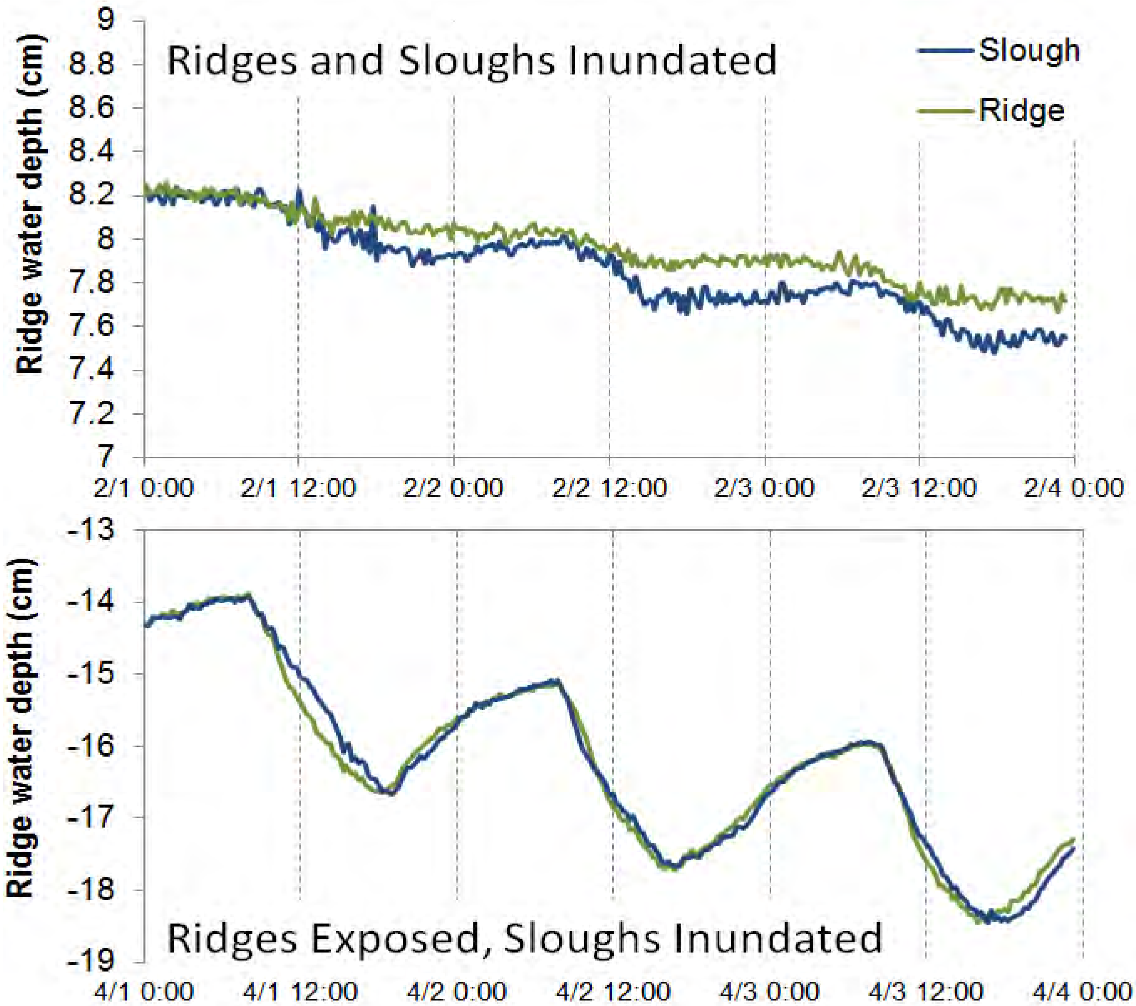


Figure 6-36. Three days (out of 500 measured) showing typical daily water responses during two periods (A) when ridges and sloughs are both inundated, and (B) when sloughs are inundated but ridges exposed. Blue represents the water elevation in sloughs, green represents ridges; both are reported relative to the elevation of the ridge soil such that positive values mean the ridges are inundated, and negative values are the depth of the water table below the ridge surface. Synchronized daily water level variation suggests ridges are drawing water from sloughs during the day, and that the head gradients that were expected to form are immediately attenuated.

From this finding, it follows that the size of the subsidy of water to ridges would scale with the size of the ridge patches as well as the length of time that ridges are dry but sloughs exposed. This means the subsidy can vary substantially from year to year. However, the scale of the subsidy is significant (**Figure 6-37**). Preliminary calculations based on the amount of peat laid down each year, the amount of P in the soil water, and the average length of time ridges are exposed suggest that the subsidy ends up being as much as 14% of the annual P storage in the soils. Accumulating this subsidy over many years may explain the elevation-P relationship (**Figure 6-34**).

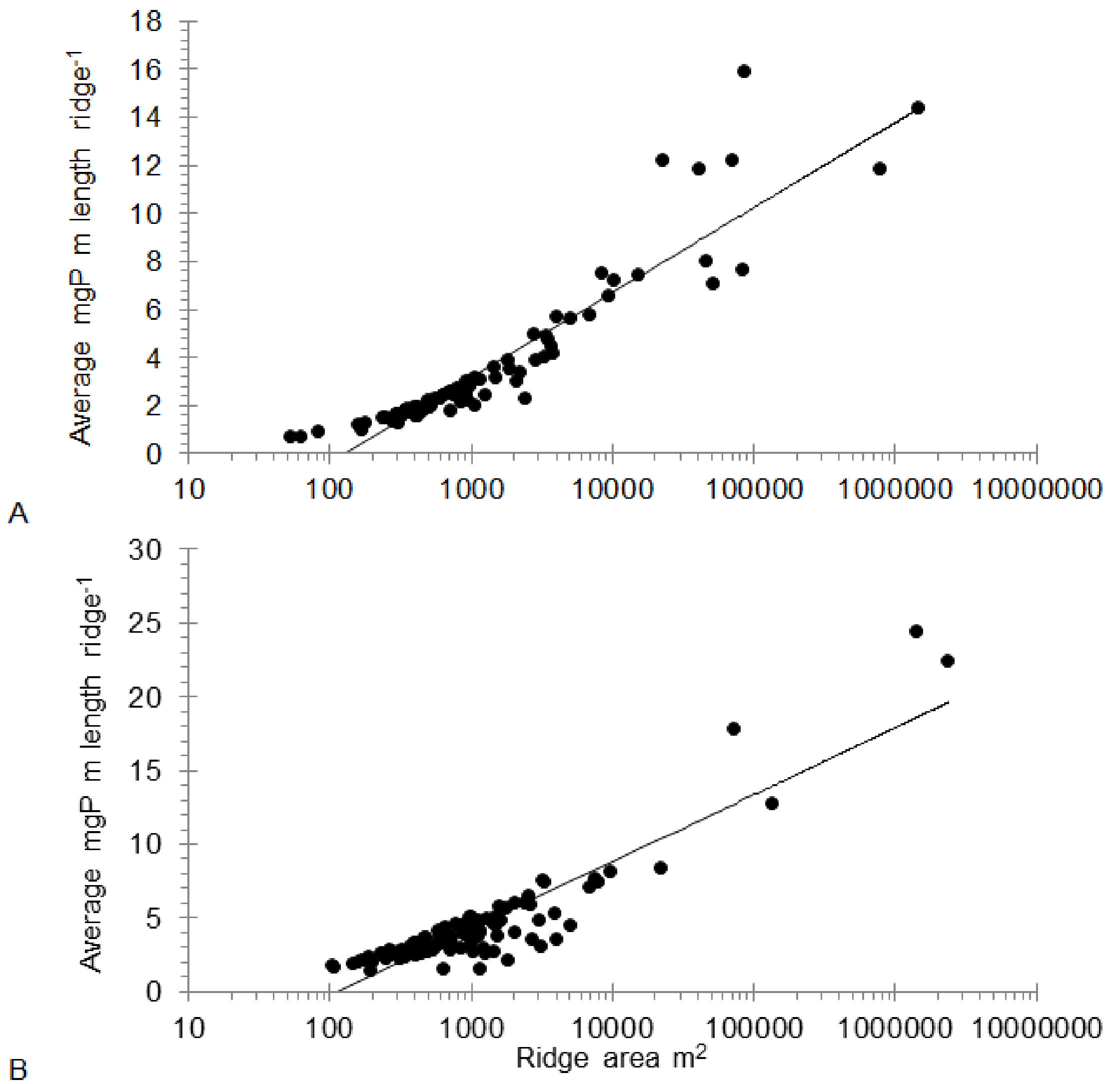


Figure 6-37. The estimated P subsidy to ridges of different sizes due to hydrologic fluxes from sloughs in two locations (A and B). The effect is size dependent because large ridges require more water to maintain water levels in lockstep with adjacent sloughs. The magnitude of the P delivery to the ridges may be sufficient to support P enrichment on ridges (Figure 6-34) and may also lead to faster rates of expansion of big ridges versus small ridges.

Hydrologic-driven Short-term Vegetation Dynamics in Shark Slough

Plant communities arranged along a gradient are the results of ecological processes associated with underlying physico-chemical drivers that vary on both spatial and temporal scales. Temporal changes in those drivers, whether due to natural processes, anthropogenic disturbances or both, often result in a shift in community composition along the gradient and, through feedbacks between community and ecosystem processes, also determine the trajectories of community succession. However, the rate of such a shift in species composition depends on the degree of changes in the underlying drivers, the types and strength of the feedbacks, and the ability of a community to withstand the effects of alterations in those drivers (Wilson and Agnew 1992, Folke et al. 2004).

In the Everglades, where hydrology is the most prominent driver of ecosystem function, plant communities are arranged along a hydrologic gradient, from open water sloughs dominated by water lilies (*Nymphaea* sp.) to spikerush (*Eleocharis cellulosa*) to dense sawgrass (*Cladium mariscus*, *Cladium jamaicense*), and finally to woody communities, bayhead swamps and hardwood hammocks (Gunderson 1994, Armentano et al. 2002). With temporal change in hydrologic regime, including depth and duration of flooding, a change in species composition of these communities often results in a shift in boundaries between self-organized entities, especially those in the ridge, slough and tree islands landscape (Larsen et al. 2011). However, the direction and magnitude of such a change are determined by the extent of hydrologic alterations, with prolonged and extreme wet events even resulting in the complete loss of upland woody vegetation, such as observed in certain tree islands (Patterson and Finck 1999, Sklar and van der Valk 2002). In contrast, prolonged drying conditions set an opposite trend, i.e., the vegetation trajectory proceeds toward an expansion of sawgrass, and to the dominance of trees over herbaceous plants (Kolipinski and Higer 1969, Willard et al. 2006). Historically, vegetation dynamics in the Everglades were governed by natural processes. However, in recent years, hydrologic modifications through the operations of water structures have dramatically impacted vegetation composition and its dynamics (McVoy et al. 2011). Establishment of historical hydrologic regimes is the primary goal of the ongoing restoration efforts under CERP. Within CERP, changes in water management associated with restoration will likely result in changes in the balance and boundaries between herbaceous species and woody communities throughout the marshes within the ridge and slough landscape, while in tree islands, the proportion of flood-tolerant and flood-intolerant woody species will change, resulting in a shift in species assemblages and tree island function.

The climatological records and hydrologic data from the SRS region of ENP suggest that water level during most of the last decade of the twentieth century was well above the 30-year average. In contrast, both the mean annual rainfall and water level were relatively low during the most recent decade (2001–2010). The interaction between hydrology and vegetation was examined over a 12-year period, between 1999–2000 and 2012 within marshes and seasonally flooded portions of tree islands in SRS. It was hypothesized that hydrologic difference between census dates resulted in an increase in abundance of sawgrass and moderately flood-tolerant woody plants depending on position of the study sites along the hydrologic gradient.

In 1999–2000, marsh vegetation was sampled in a series of 10 x 10 m plots on four transects, and in 2001–2002, tree island vegetation was sampled in plots along transects and within two permanently marked plots arranged along a flooding gradient in three islands. In the tree islands, the flooding gradient consisted of a pair of plots, one (20 x 20 m) in bayhead forest (relatively tall, closed forest canopy, short period of flooding) and one (15 x 15 m) in bayhead swamp vegetation (low, open forest with longer flooding duration), separated by 100 meters or more. In the marsh, vegetation was sampled 2 to 3 times between 1999 and 2012, while tree island plots were resampled only in 2011–2012 (**Figure 6-38**). Using a suite of multivariate techniques, including trajectory analysis (Minchin et al. 2005), the direction of vegetation change in SRS marshes over time was examined by quantifying the displacement of sites in relation to the hydrologic gradient in ordination space.

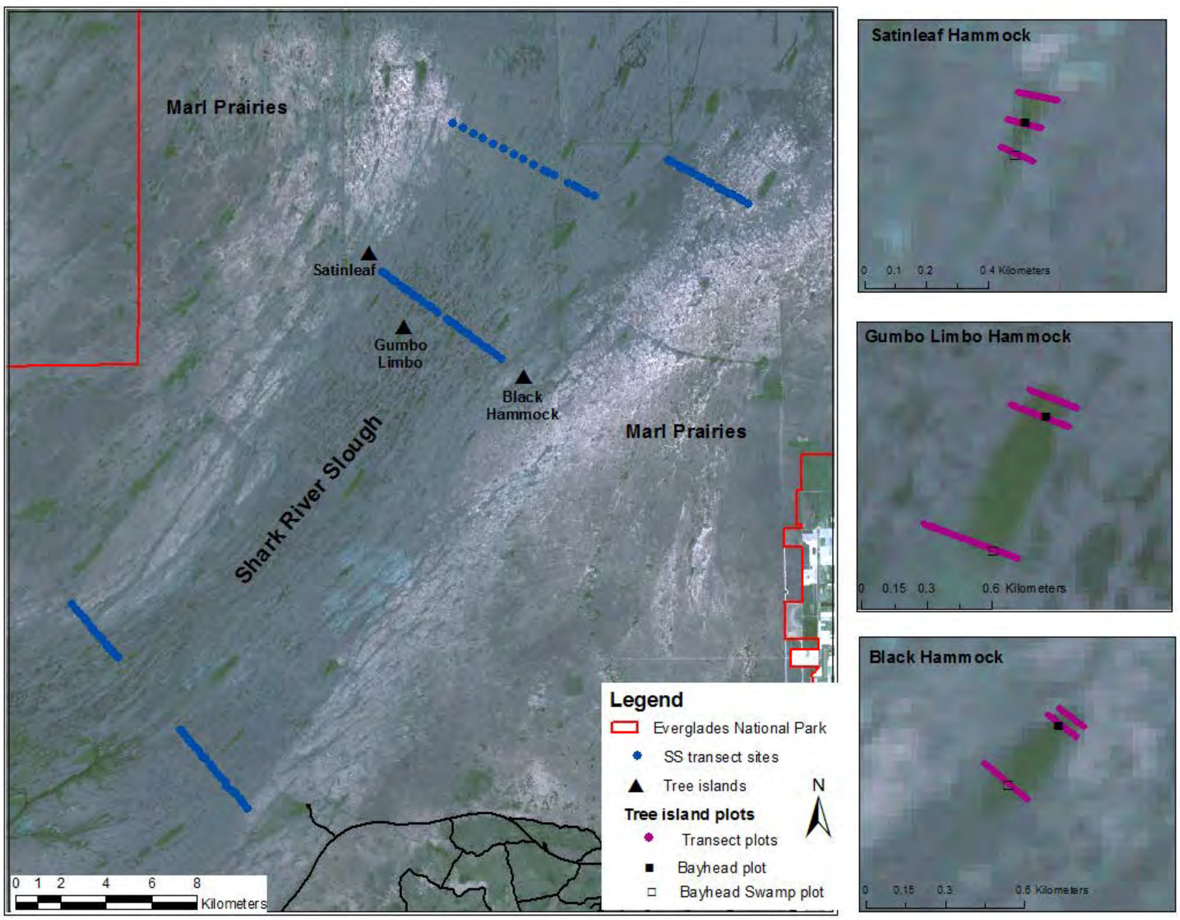


Figure 6-38. Study area map showing the location of SRS (SS) transect sites and tree island transects and plots.

In both SRS marshes and tree islands, vegetation tracked the changes in hydrology over the 12-year period. In marshes, in the vicinity of tree islands as well as in the larger areas within the ridge and slough landscape, vegetation composition changed towards a drier type, as predicted by the prevailing water conditions. However, the intensity of this change was spatially differentiated. The drying trend in marshes decreased from north to south, i.e., the transect in northeastern SRS had the highest percentage of sites showing a significant trajectory towards a drier condition over the period (**Figure 6-39**). Over the study period, the abundance of bladderworts (*Utricularia* spp.), which are indicator of relatively wet conditions, decreased significantly while cover of the emergent sawgrass and spikerush increased (**Figure 6-40**). Sawgrass cover also increased near the boundary between marsh and bayhead, and throughout the bayhead swamps of all three tree islands.

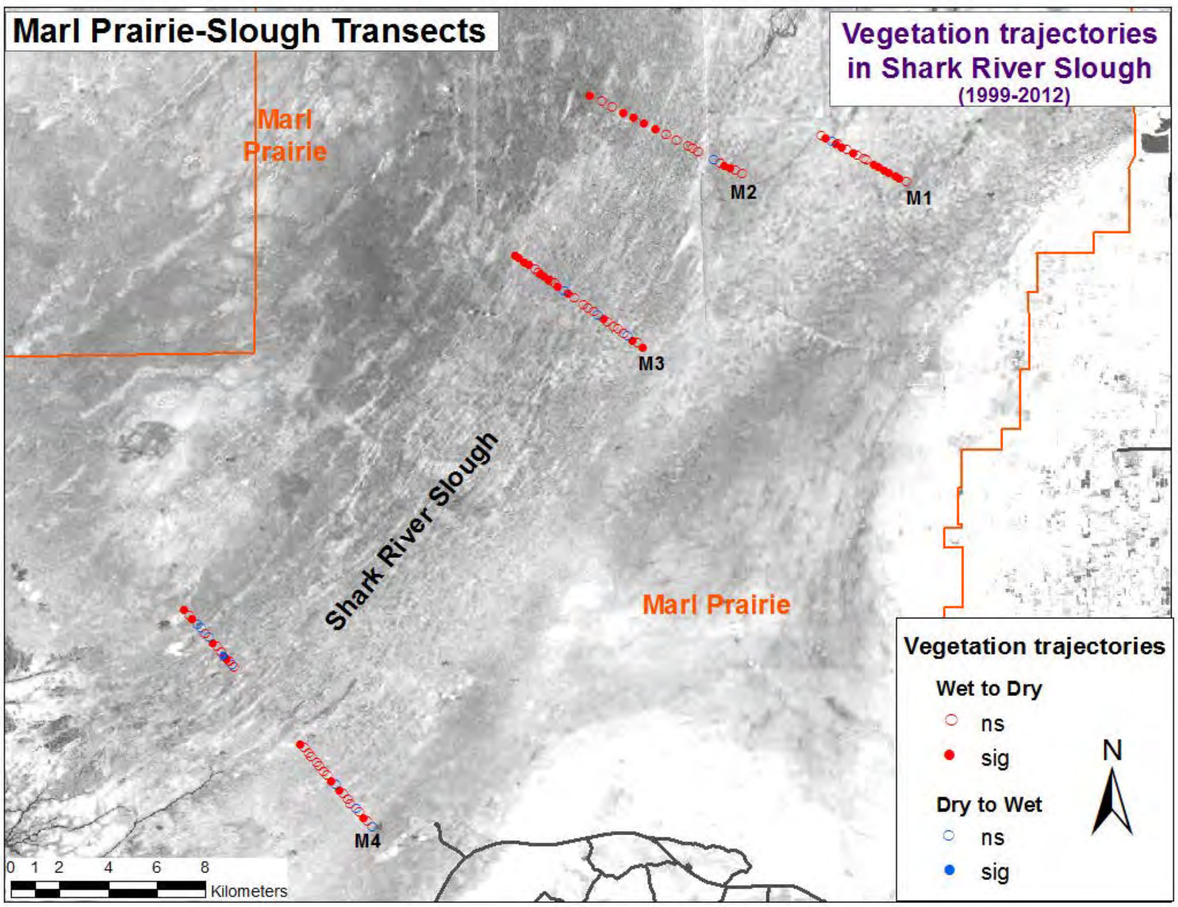


Figure 6-39. Sites in the SRS portion of four transects showing the vegetation trajectory trend that was determined using trajectory analysis on vegetation data collected four times between 1999 and 2012. [Note: ns – not significant; sig = significant.]

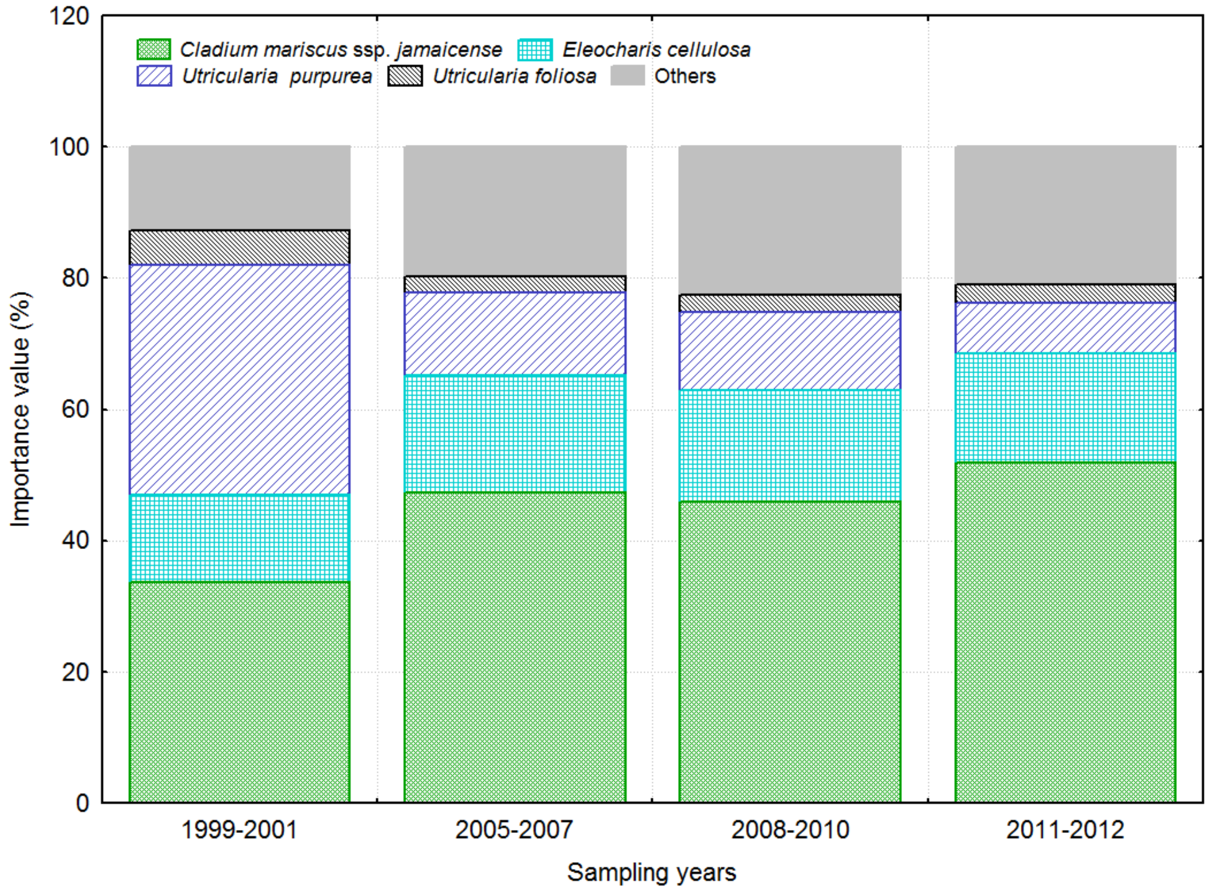


Figure 6-40. Major species' importance value (IV) at the SRS marsh sites sampled four times between 1999 and 2012.

In the tree islands plots, increases in tree density, basal area and species richness as well as canopy volume and height were observed. At the same time, while there was a decrease in sapling density in the bayhead forests, probably in response to intra- and interspecific competition for nutrients and light availability resulting from canopy closure and forest maturation, the bayhead swamp plots experienced an opposite trend. In bayhead swamp, an increase in number of trees—most of which were saplings in 2001–2002—and establishment of a new cohort of saplings were observed, indicating a slow but steady progression in the succession of the bayhead swamp forest toward the more developed canopy of the bayhead forest. Temporal changes in species importance values (IV) reinforced the view that tree island vegetation was undergoing succession, canopy development, and expansion over time in response to below average water levels and flooding duration. In general, flood tolerant species like pond apple (*Annona glabra*) and Carolina willow (*Salix caroliniana*) saw a decline in IV, while moderately flood tolerant species like cocoplum (*Chrysobalanus icaco*), dahoon holly (*Ilex cassine*), and strangler fig (*Ficus aurea*) increased (**Figure 6-41**).

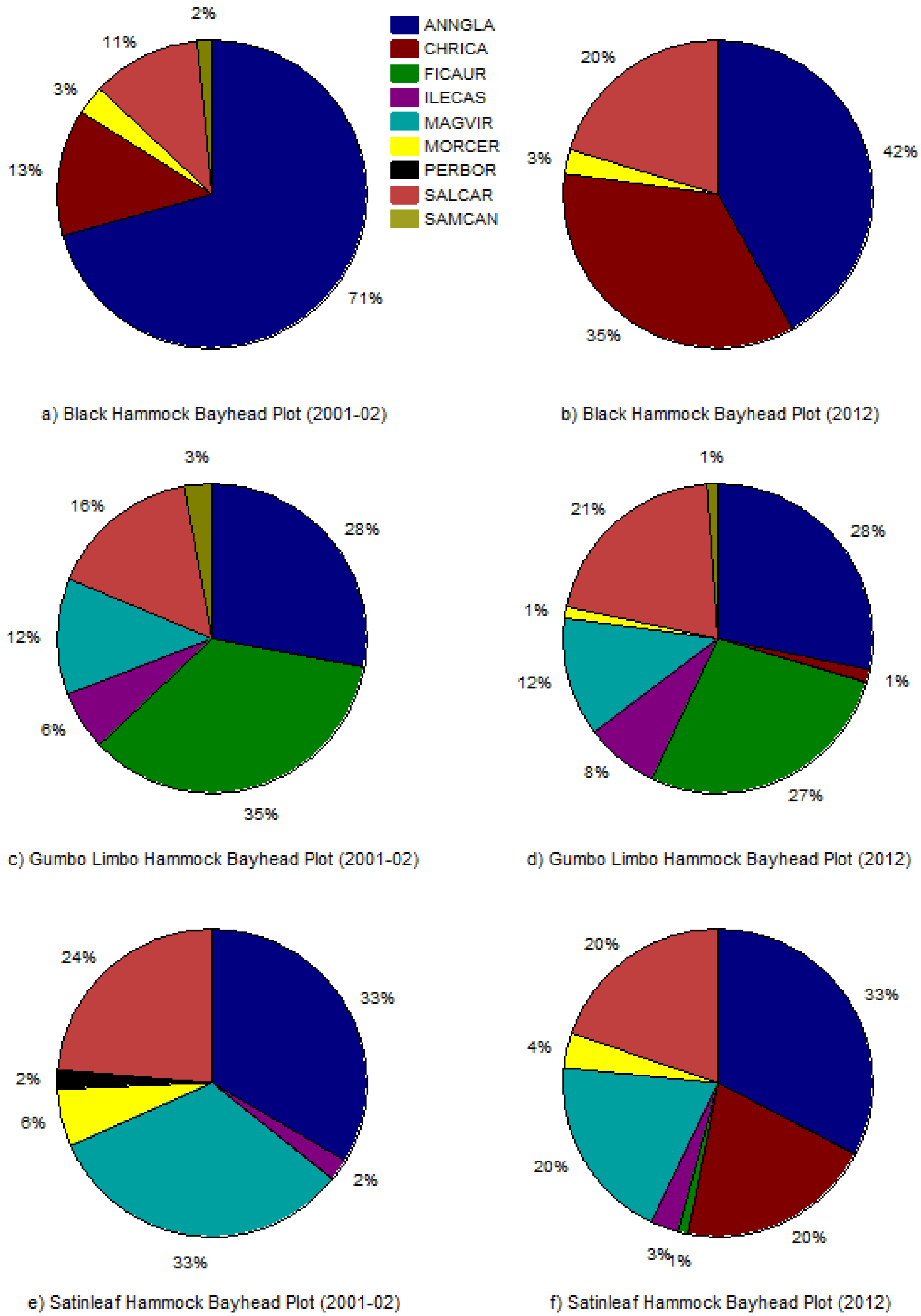


Figure 6-41. Tree species importance value (IV) across all bayhead plots between 2001–2002 and 2012. (ANNGLA = *Annona glabra*, CHRICA = *Chrysobalanus icaco*, FICAUR = *Ficus aurea*, ILECAS = *Ilex cassine*, MAGVIR = *Magnolia virginiana*, MORCER = *Morella cerifera*, PERBOR = *Persea borbonia*, SALCAR = *Salix caroliniana*, SAMCAN = *Sambucus canadensis*)

In conclusion, this work indicates that hydrology is a determining factor with respect to changes in both marshes and tree islands at the population, community, and ecosystem levels, and that these responses are rapid. In this case, long-term fluctuations in hydrologic regime, resulting in below average water levels and shorter hydroperiods in Shark Slough, promoted an increase in spikerush and sawgrass cover at the expense of open water sloughs in the marshes, and the expansion of woody plants across the full suite of communities comprising the tree island gradient, i.e., bayhead forest, bayhead swamp, and sawgrass tail. In the prolonged dry conditions, it is the progression towards sawgrass, and the establishment and growth of trees in the peat environment that drives successional processes towards the expansion, growth, and maturation of tree islands in the ridge and slough landscape.

Surface-groundwater Interactions in Everglades Tree Islands

In the Everglades, tree islands are considered characteristic of the ecological “health” of the landscape (Sklar and van der Valk 2002). P levels in upland tree island soils are more than 100 times higher than the P in adjacent marsh soils (Jayachandran et al. 2004, Wetzel et al. 2005, 2011). Tree islands are hypothesized to be an active sink of P in the landscape, contributing to the P balance of Everglades slough wetlands (Wetzel et al 2005, Troxler et al. 2013). A proposed model for the accumulation of tree island P is the focused nutrient redistribution hypothesis in cm. This hypothesis suggests that among other mechanisms, evapotranspirational pumping indicated by daytime drawdown (i.e., daily daytime drawdown of groundwater levels due to the ET of vegetation, particularly trees), which is enhanced during the dry season, of the tree island soil water table and high concentration of chloride ions are characteristics of “healthy” tree islands. These characteristics are indicators of strong plant-water interactions that build ionic strength and promote mineral soil stability.

This analysis was implemented to compare hydraulic and hydrogeochemical patterns at multiple temporal and spatial scales on four Everglades tree islands in the WCAs and ENP. The tree islands studied are a wet, intact island (3AS3 in WCA 3A); a wet, degraded (Ghost Island in WCA 3A); a dry, intact island (Satin Leaf in ENP); and a dry, degraded island (Twin Heads in WCA 3B) (**Figure 6-42**). The main objectives of this project are (1) to provide an understanding of how biogeochemical pattern and processes can mediate tree island growth and patch maintenance through the interaction of biological, geochemical, and climatic factors and; (2) to test the focused nutrient redistribution hypothesis, which proposes that water movement, mediated by hydraulic processes that are punctuated in the dry season by increased tree island ET, influences the lateral flux of nutrients to downstream components of the tree island ecosystem, thus concentrating P at the intersection between local (shallow rained-derived soil water) and regional (deep groundwater) water pools.

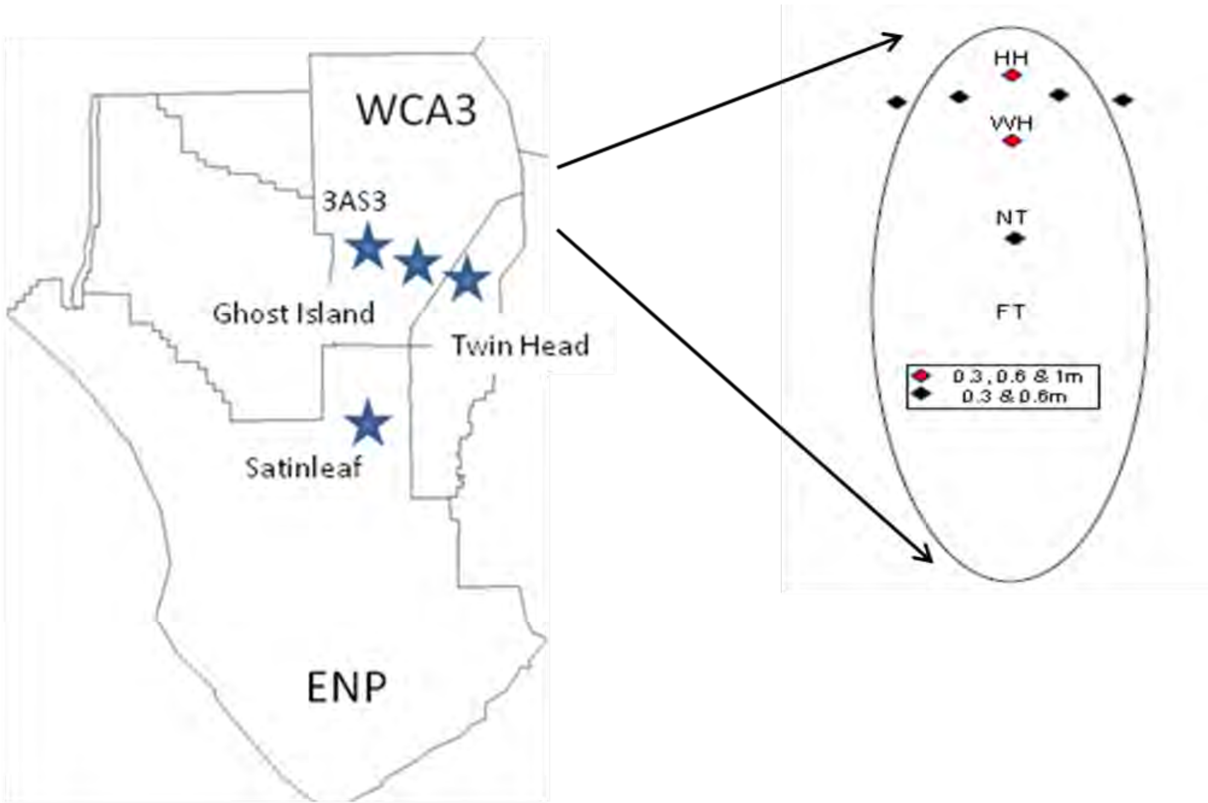


Figure 6-42. Landscape study design. The tree islands are located in WCA 3 and ENP (left). A schematic of well locations and depths is shown on the right.

Hydrologic and geochemical patterns among the four tree islands showed that Satin Leaf, a relatively intact island in ENP, retains distinct hydrologic and hydrogeochemical variation among the three primary tree island vegetation types—high head, wet head, and near tail—and marsh community (**Figure 6-43**). This diurnal hydrologic pattern was more pronounced during the drydown period (February; **Figure 6-43A**) than during the rewetting period (October; **Figure 6-43B**). In contrast, the Ghost Island, an island subject to overflooding, appeared to have a lesser gradient in hydrologic conditions associated with high head, wet head and near tail community types (**Figure 6-44A** and **B**). The dry, degraded Twin Head island is characterized by a low gradient in hydrologic conditions and subjected to low water depth along with short hydroperiods. The degraded tree islands included in this study do retain some key features of an intact island in the WCA 3A, including high soil P concentrations in the high head and the presence of trees. In summary, if these results are confirmed over a longer baseline (Troxler 2013), then regional hydrologic shifts that include higher water levels in drier WCA 3B and lower water levels in ponded, overflooded areas of WCA 3A, should improve nutrient recycling on tree islands, which is important for maintaining tree island health.

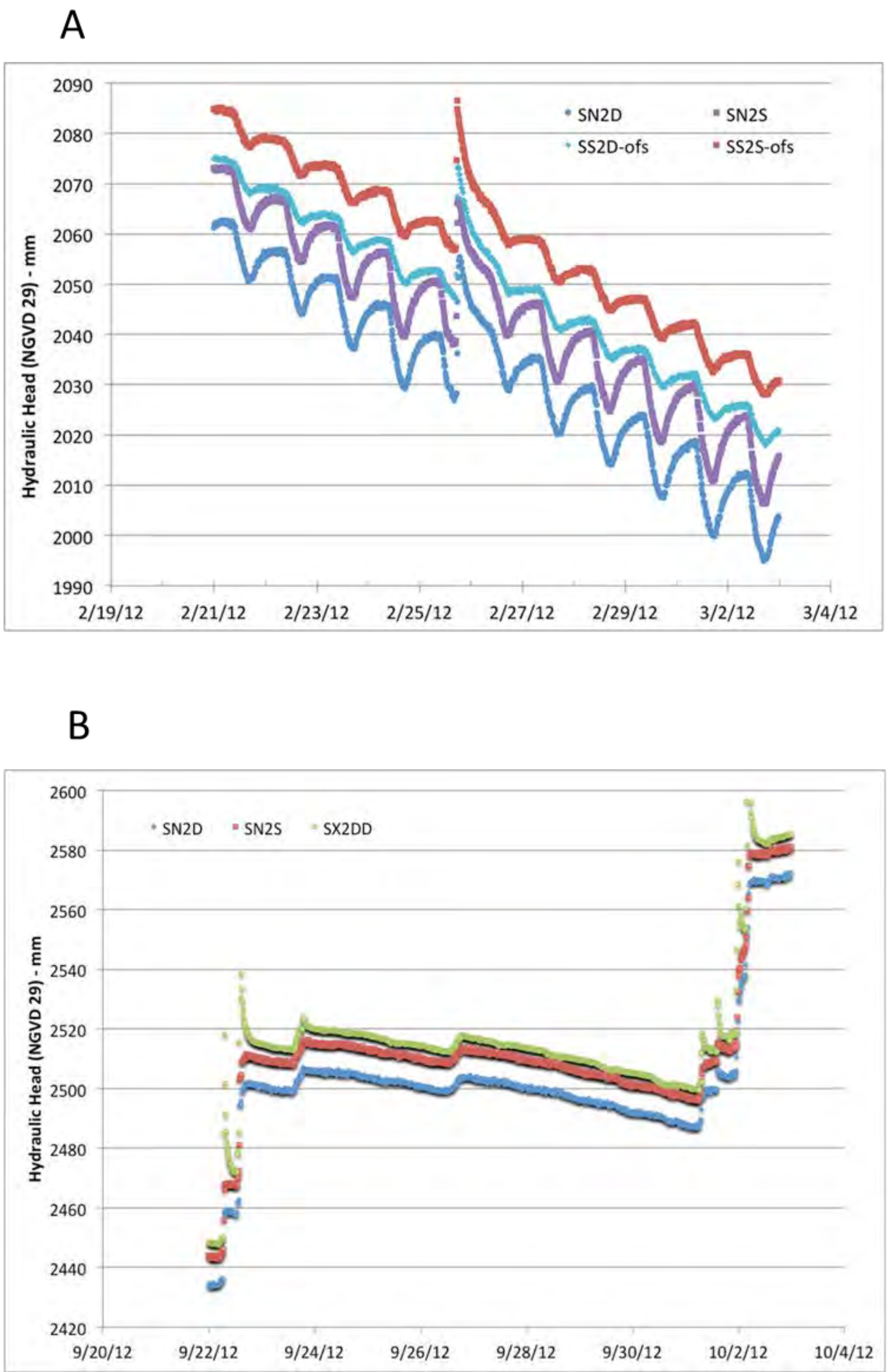
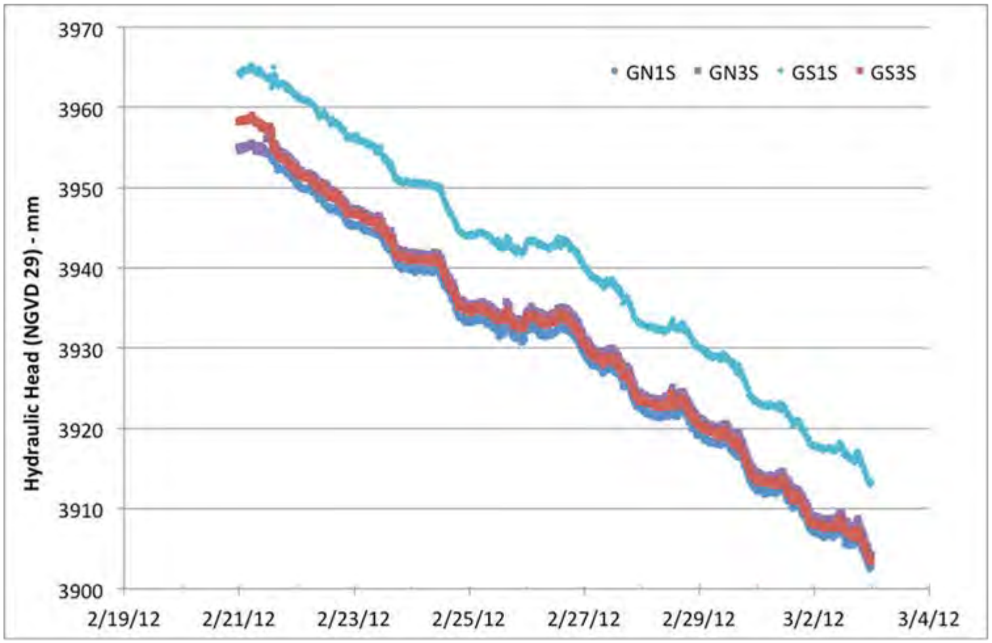


Figure 6-43. Diurnal hydraulic head (millimeter [mm]) relative to National Geodetic Vertical Datum of 1929 (NGVD 29) datum at wells located in Satin Leaf tree island during (A) the drydown period (February–March 2012) and (B) the rewetting period (September–October 2012).

A



B

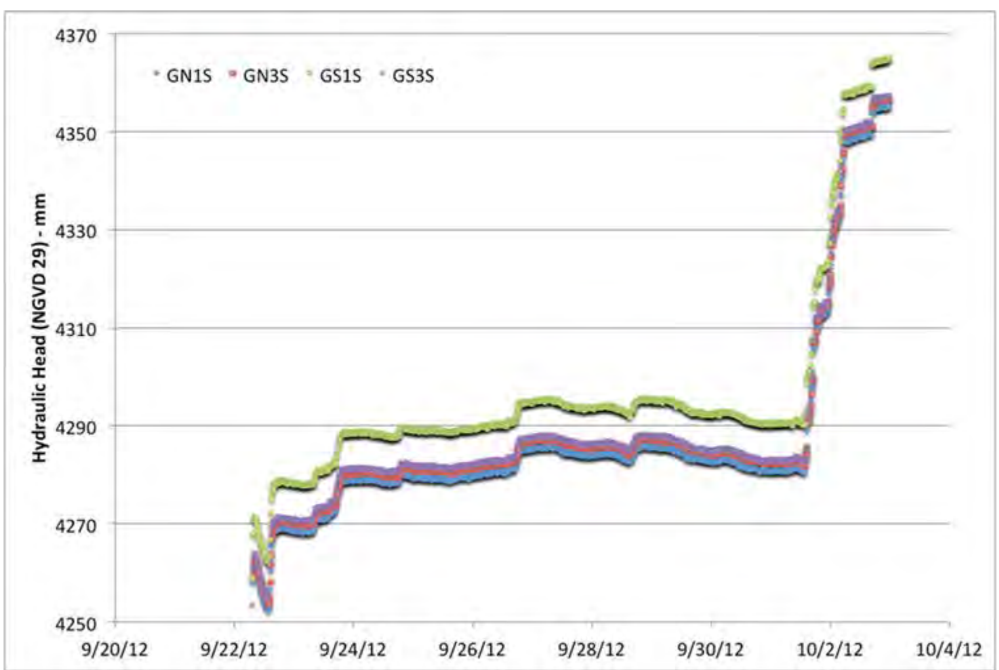
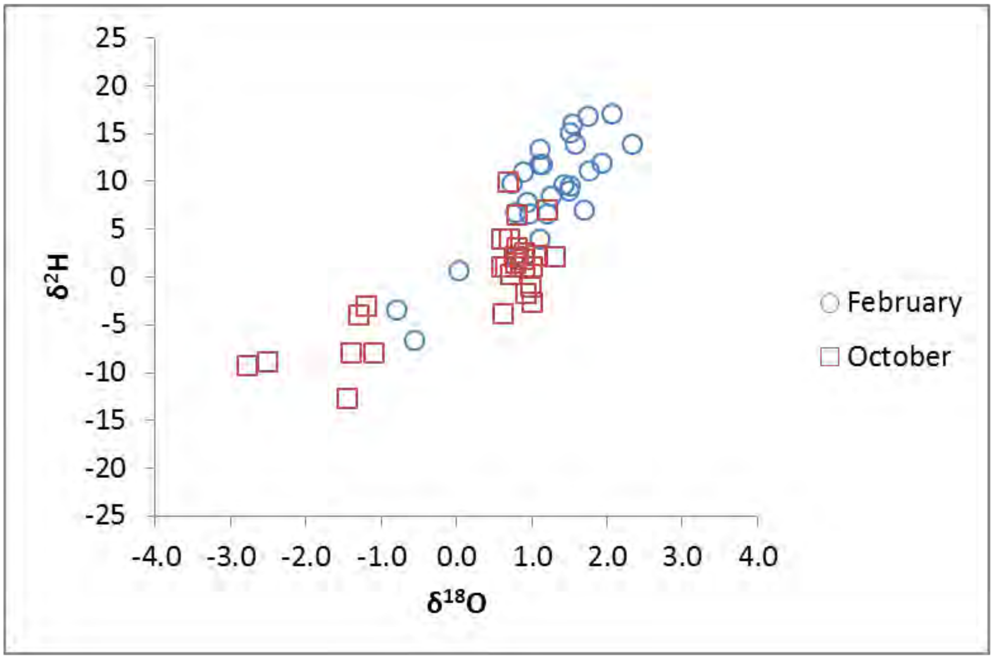


Figure 6-44. Diurnal hydraulic head (mm) relative to NGVD 29 datum at wells located in the Ghost Island during (A) the drydown period (February–March 2012) and (B) the rewetting period (September–October 2012).

These results have produced three main findings that illustrate the mechanisms that lead to the degradation of tree islands and give insight into how tree islands might be restored. These findings are that healthy tree islands: (1) exhibit spatial and temporal variability in plant water uptake that is focused in the shallow soil profile; (2) exhibit ET that increases from wetter communities to drier communities within the tree island, and; (3) actively accumulate ions, especially chloride, in soil water of the drier, high head plant community. At the landscape level, the extent and timing of regional hydrology, including its variability within the WCA compartments, determines the tree island hydrogeochemical pattern as well as the interrelationship between plant community performance, soil mineral precipitation and the balance between accumulation and decomposition of organic matter. The focused nutrient redistribution model, which relates ion composition of tree island water and concentration of total dissolved phosphate for the wet intact tree island, was found to be distinct and robust. This is important because it may prove useful when actions to restore the spatial and temporal pattern of regional hydrology are implemented. The robustness of the model is complemented by evidence that links plant water uptake with tree island water sources, either surface water (wet season) or groundwater (dry season). The linkage is more evident in the dry pristine island where the water source is very seasonal with surface water use during the dry season, indicated by ^{18}O enrichment, and rainwater during the wet season, showing ^{18}O depletion (**Figure 6-45A**), while in the wet degraded island, the water source is always surface-groundwater, regardless of the season, as indicated by moderate ^{18}O enrichment (**Figure 6-45B**).

These results suggest that the degradation of tree islands exposed to extreme and long wet conditions appear to be largely a result of decoupling, in both time and space, of environmental conditions that promote water uptake, mineral P retention and organic matter accumulation, through regional changes in hydrology (Troxler 2013). Thus, the potential of restoring a degraded tree island lies in the restoration of a hydrology targeted to achieve plant performance as well as mineral retention and organic matter accumulation that is stratified across space and time.

A



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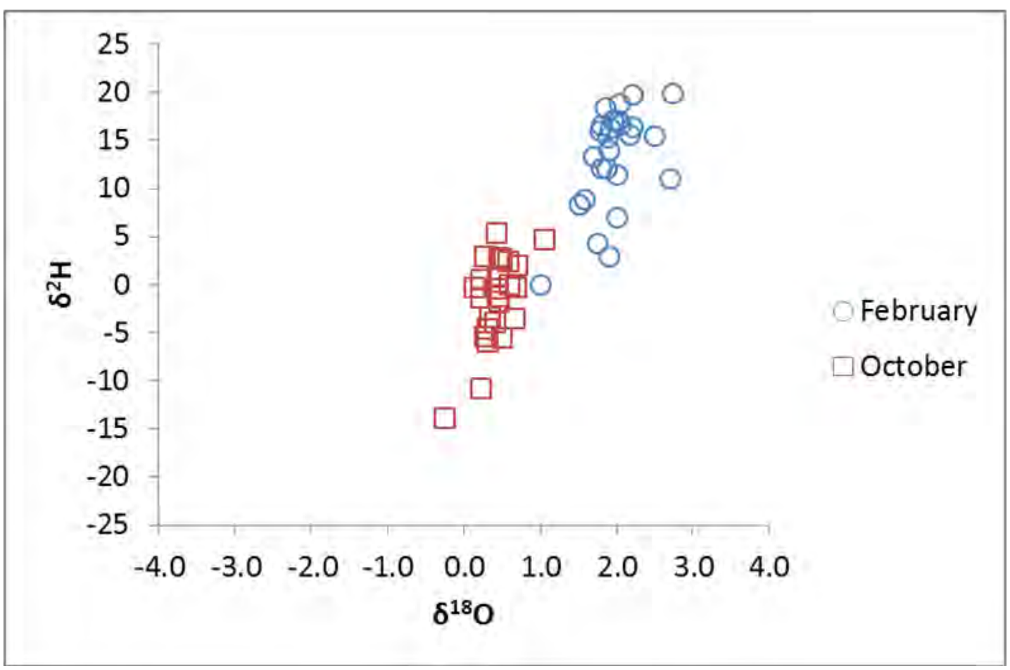

Figure 6-45. Plant stems water $\delta^{18}O$ and $\delta^{2}H$ showing seasonal differences in water use between the drydown period (February) and rewetting period (October) for (A) Satin Leaf Tree Island and (B) Ghost Tree island.

TROPHIC HYPOTHESIS UPDATE

The CERP 2009 MAP included several hypothesis clusters that formed the basis of results-oriented monitoring programs to measure restoration success. One CERP performance goal was the recovery of historical wading bird colonies in numbers approximating historical levels and ideally in historical locations. Monitoring programs were grouped and designed to measure response of ecological indicators to hydrologic stressors affected by restoration actions that will result in wading bird productivity as captured in the "Trophic Hypothesis" (**Figure 6-46**). The key objective of this cluster of monitoring programs was to document status and trends of wading birds, their nesting biology, and the food web supporting their productivity. Multiple lines of evidence collected over three decades indicate that food limitation during the nesting season is a key contributor to nesting success and sustainability of Everglades wading bird populations. Management that produces high quality prey patches throughout the nesting cycle is thus a key to successful Everglades restoration. The trophic hypothesis states that high quality prey patches are created by an interaction of prey production and prey

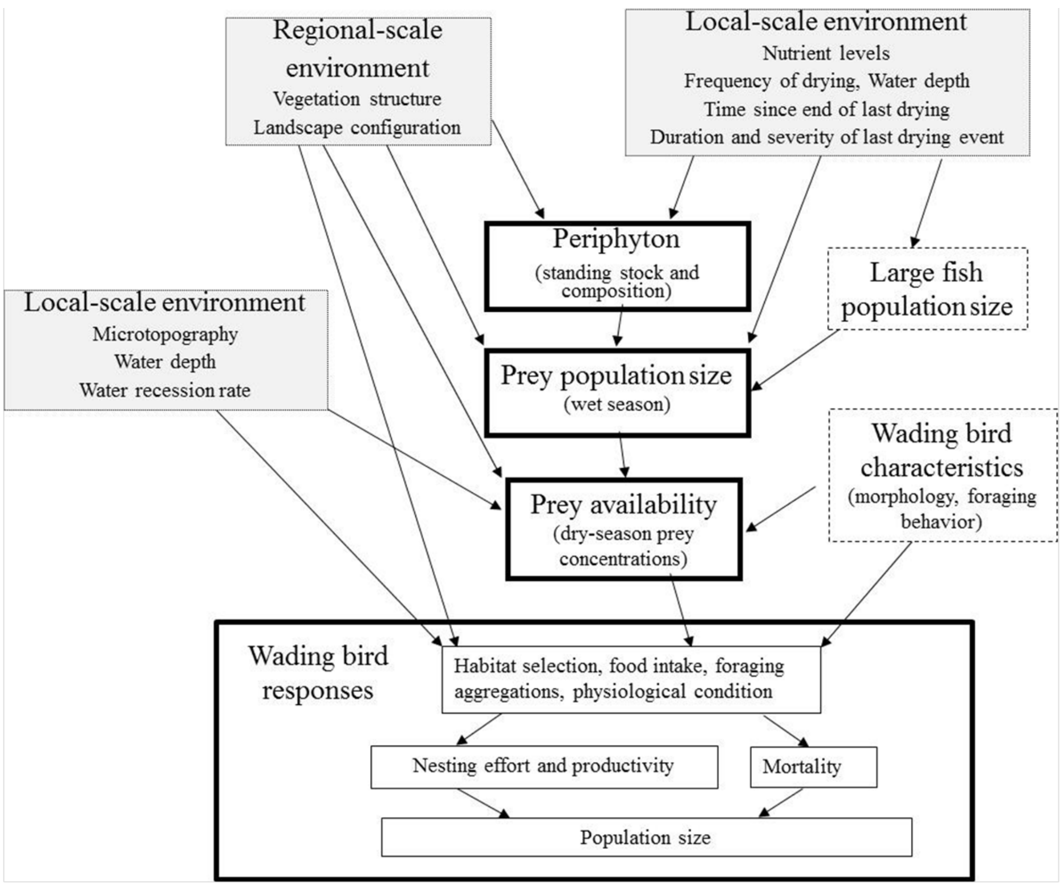

Figure 6-46. Trophic Hypothesis for CERP MAP linking environmental drivers affected by restoration (grey boxes) and wading bird population size. A chain of direct and indirect effect link human actions to wading birds, largely mediated through wading bird food availability during nesting season. Environmental drivers are in grey boxes. Monitored variables discussed in this SSR are in boxes with bold borders. Ecological factors discussed in this SSR, but not monitored, are in boxes with dashed borders.

concentration, and that prey production is tied to periphyton production, hydrology (appropriate water depths and timing), and landscape structure. Ultimately, water management and restoration projects will affect wading bird prey directly through concentrating prey and indirectly by supporting growth of prey populations and the development of microtopographic features in the landscape. The MAP Trophic Hypothesis Cluster includes monitoring periphyton, wading bird prey populations in the wet season, prey populations in drying pools in the dry season, and wading bird foraging and nesting success. Monitoring these ecological attributes in **Figure 6-46** helps validate the hypothesis cluster explaining why changes are occurring and provides data on status and trends of the drivers of nesting success. This monitoring information helps assess if restoration actions are matching CERP goals, interpret the origin of problems if things do not go as planned, and link desirable conditions in the field to restoration actions if appropriate.

This report is organized to follow the flow of the Trophic Hypothesis: (1) first status and trends of wet season prey biomass assessed for the MAP are reported; (2) next, patterns of dry season prey availability and sources of inter-year variation are reported; (3) patterns of wading bird nesting success for the GE region follow; (4) the benefits of continued monitoring are then addressed; and (5) finally, the application and effectiveness of the MAP integrated design is discussed. A discussion of recent results from periphyton monitoring, including the link of hydrological and nutrient drivers to periphyton production and aquatic consumer responses, was addressed in the Oligotrophic Nutrient Status as Indicated by Periphyton section.

Wet Season Production of Aquatic Prey Populations

Aquatic prey of wading birds (particularly small fish, crayfish, and grass shrimp) have been monitored annually since 2005 by sampling in the late wet season (late September through early December) with one-square meter throw traps (Jordan et al. 1997). The late wet season was selected because it is the end of the summer growing season prior to the onset of the dry season, and a time when the entire ecosystem can be characterized in a systematic fashion because of relatively stable high water. Each year, throw trap samples are collected at approximately 150 PSUs in freshwater marshes across the Everglades.

Methods

To facilitate analyses, the PSUs were grouped within landscape sampling units (LSUs) at the outset of the project (**Figure 6-47**). Of the 52 LSUs originally identified for the GE, 32 were considered feasible for throw trap sampling based on dominant vegetation cover (Trexler 2004). Areas of dense cattails or willows cannot be effectively sampled by the throw trap and some of the LSUs are not likely to be affected by CERP projects. Selection of PSUs followed guidelines from Philippi (2003, 2005) and was based on a spatially balanced recursive tessellation design (Stevens and Olsen 2003). A major strength of this sampling design in the context of CERP is its flexibility. The sampling design allows researchers to identify discrete areas of the Everglades landscape based on environmental data (i.e., hydrological characteristics, soil type, or elevation patterns) and make statistically rigorous comparisons of the biological attributes among selected regions.

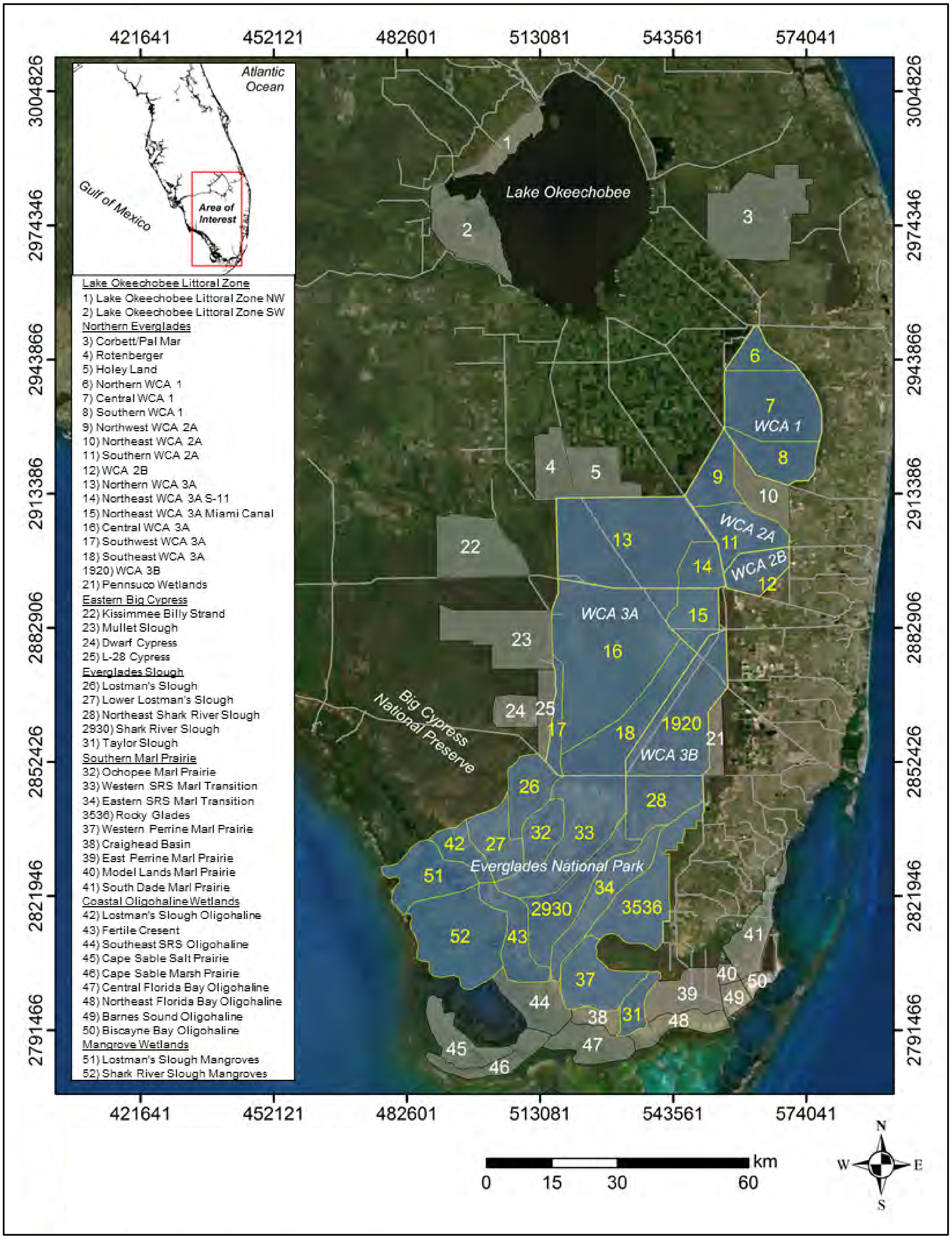

Figure 6-47. LSUs for the Greater Everglades. LSUs shaded in blue were sampled annually from 2005 to 2012.

In this assessment, aquatic prey populations are reported and assessed for spatial and temporal patterns from 13 LSUs in the WCAs and ENP where at least three PSUs were sampled during each year for the late wet seasons (September–November) from WY 2005 to WY 2012 (**Table 6-8**). To simplify the presentation and focus on wading bird prey, mean biomass is reported in grams per square meter (g/m^2) wet weight for crayfish (*Procambarus fallax* and *Procambarus alleni* summed), marsh fishes (all species summed), and grass shrimp (*Palaemonetes paludosus*).

Table 6-8. LSUs used in the assessment of aquatic prey populations from late wet seasons of 2005 through 2012. Numbers to the left correspond to Figure 6-47.

Water Conservation Areas	Everglades National Park
WCA 1 7) Central WCA 1 (32,561 ha) 8) Southern WCA 1 (15,334 ha) WCA 2 9) Northwest WCA 2A (12,781 ha) 11) Southern WCA 2A (17,856 ha) WCA 3 13) Northern WCA 3A (61,880 ha) 16) Central WCA 3A (77,049 ha) 18) Southeast WCA 3A (27,181 ha) 19/20) WCA 3B (39,885 ha)	Shark River Slough 28) Northeast SRS (19,408 ha) 29/30) SRS (23,389 ha) 33) Western SRS Marl Transition (36,102 ha) 34) Eastern SRS Marl Transition (19,797 ha) Southern Marl Prairies 37) Western Perrine Marl Prairie (19,401 ha)

Results and Discussion

The hydrological conditions encountered between 2005 and 2012 have varied and include dry (2009 and 2011), wet (2010), and intermediate (2005–2008, 2012) years. The proportion of PSUs that dried within 365 days before wet season sampling varied from 38% (2010) to 95% (2011) (**Figure 6-48**). The pattern of drying was spatially predictable, with the central and southeastern WCA 3A (LSUs 16 and 18) retaining water even in the driest year.

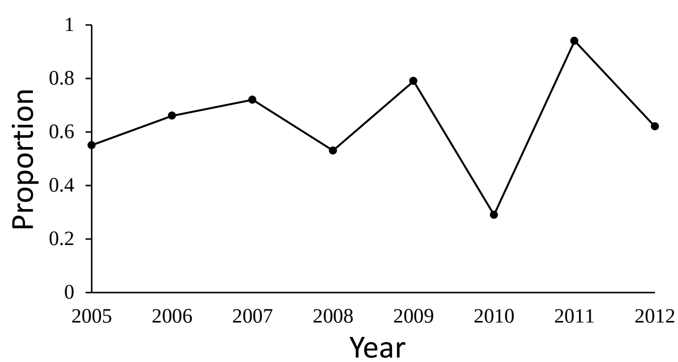

Figure 6-48. Proportion of study sites where the marsh surface dried (water depth < 5 cm in data from EDEN) in the 365 days before sampling plotted by year.

Across the entire eight year data set, area weighted mean prey biomass for the Everglades was 4.1 g/m^2 and ranged from 2.1 to 6.9 g/m^2 for individual LSUs (**Table 6-9**). Generally, lower biomass was observed in the beginning and the end of the time series (2005, 2006, 2011, and 2012) and higher biomass in the middle (2008 and 2009). This does not correspond well to the systemwide patterns of marsh drying.

Table 6-9. Summed biomass of all aquatic prey (g/m^2 wet weight of fish, crayfish, shrimp summed) from 2005 through 2012 in LSUs in which at least three PSUs were sampled in each year. Symbols (+/-) represent instances in which the yearly mean biomass for a given LSU was significantly different (Wilcoxon rank sum, $p < 0.05$) from the annual mean in that LSU. Significant annual deviations are bold. A few extreme values are not different from the annual mean because of high within-LSU variability (e.g., Central WCA 1 in 2007).

Landscape Sampling Units	Biomass (g/m^2 wet weight)								Annual mean
	2005	2006	2007	2008	2009	2010	2011	2012	
Central WCA 1	6.9	5.4	10.5	**8.8**$^{+}$	4.9	**3.9**$^{-}$	**3.4**$^{-}$	**11.1**$^{+}$	**6.9**
Southern WCA 1	1.8	2.9	**7.1**$^{+}$	5.3	4.0	4.0	4.0	3.0	**4.0**
Northwest WCA 2A	**1.2**$^{-}$	2.3	2.7	**4.2**$^{+}$	2.5	4.1	**6.9**$^{+}$	3.5	**3.4**
Southern WCA 2A	3.2	1.1	2.3	**3.5**$^{+}$	**1.1**$^{-}$	2.3	1.8	1.7	**2.1**
Northern WCA 3A	2.4	4.0	2.3	**4.8**$^{+}$	**10.4**$^{+}$	**11.2**$^{+}$	4.5	**2.3**$^{-}$	**5.2**
Central WCA 3A	**1.9**$^{-}$	**1.8**$^{-}$	**6.9**$^{+}$	**5.4**$^{+}$	3.9	4.5	3.0	**2.3**$^{-}$	**3.7**
Southeast WCA 3A	2.9	3.0	**4.7**$^{+}$	**4.8**$^{+}$	2.6	**1.6**$^{-}$	2.5	3.5	**3.2**
WCA 3B	3.3	2.9	**5.3**$^{+}$	**3.7**$^{+}$	3.5	2.4	**2.1**$^{-}$	3.0	**3.3**
Northeast SRS	3.4	2.1	4.2	4.0	4.2	2.5	**1.5**$^{-}$	2.6	**3.1**
SRS	**1.6**$^{-}$	5.0	4.9	**7.3**$^{+}$	**9.0**$^{+}$	5.4	6.9	6.1	**5.8**
Western SRS Marl Transition	4.6	**2.7**$^{-}$	**1.9**$^{-}$	**7.9**$^{+}$	**9.6**$^{+}$	5.2	3.8	4.0	**5.0**
Eastern SRS Marl Transition	**2.2**$^{-}$	2.7	2.0	**9.8**$^{+}$	**8.2**$^{+}$	**2.3**$^{-}$	2.9	4.3	**4.3**
Western Perrine Marl Prairie	4.2	2.2	**1.5**$^{-}$	**1.7**$^{-}$	**7.5**$^{+}$	2.4	3.3	3.0	**3.2**
Total Area-Weighted Mean	**2.8**	**2.8**	**4.3**	**5.3**	**6.0**	**4.8**	**3.5**	**3.2**	**4.1**

Temporal patterns of prey biomass were documented by comparing yearly means to the studywide mean for the LSU using a Wilcoxon rank sum test. Biomass in an LSU was ranked as high during a particular year if the mean was significantly higher ($p < 0.05$) in comparison to the mean in the same LSU for the entire study. Biomass within an LSU was ranked low during a particular year if the mean was significantly lower ($p < 0.05$) than the annual mean in the same LSU for the entire study.

Total prey biomass was low throughout much of the Everglades during 2005 and 2006 (**Table 6-9** and **Figure 6-49**). In 2007, prey biomass began to rise in some of the longer hydroperiod portions of the Everglades (southern WCA 1, central and southern WCA 3A, and WCA 3B), but remained low to average throughout the shorter hydroperiod areas (northern portions of the WCAs and ENP). In 2008, prey biomass peaked across the Everglades with areas of high biomass occurring in most of the WCAs and ENP. The lone instance of low biomass in 2008 occurred in the western perrine marl prairie region of ENP. Prey levels began to drop in 2009 with high biomass generally occurring in the shorter hydroperiod regions (ENP and Northern WCA 3A). From 2010 to 2012, prey biomass was variable throughout the Everglades with few strong spatial patterns in total prey biomass emerging.

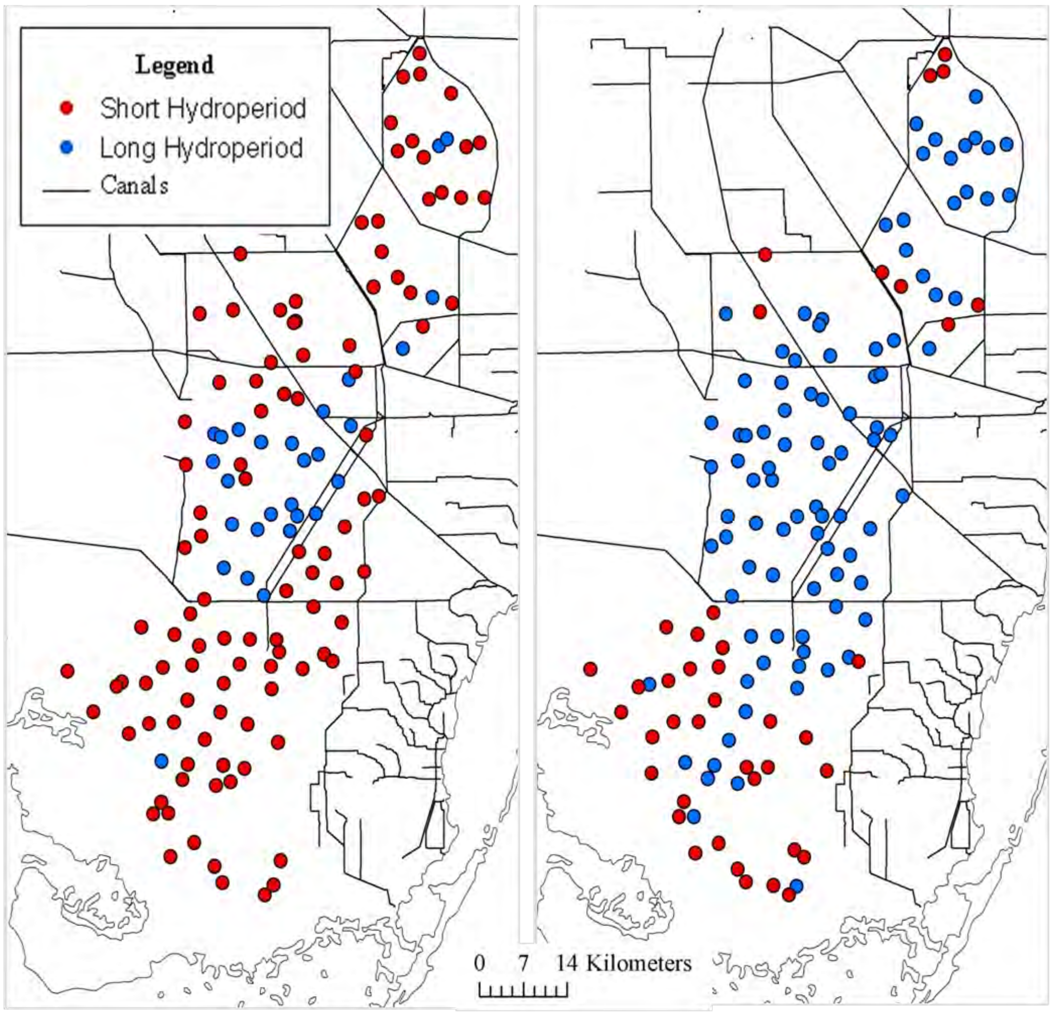

Figure 6-49. Illustration of the distribution of sites that dried in a dry year (left panel 2009) and wet year (right panel 2010). Short hydroperiod was defined as having dried within 365 days of sampling while long hydroperiods was defined as not having dried in that period.

The lack of pattern in these data arose because the three major wading bird prey types were combined. To explore this, trends in biomass composition were analyzed using non-metric multidimensional scaling of Jaccard similarities (results available upon request). Environmental vectors were fitted to explore the drivers of biomass composition. PSUs yielding samples dominated by fish and grass shrimp generally fell near each other and were positively correlated with hydrologic variables such as days since last dry down, water depth, and hydroperiod. This suggests that high biomass of fish and shrimp are often found together and these taxa are responding similarly to hydrologic conditions. Samples from PSUs dominated by crayfish were well separated from those dominated by fish and grass shrimp, and negatively correlated with the hydrologic variables. Systemwide crayfish biomass was well correlated with the proportion of our sites that dried in the year prior to sampling (**Figure 6-50**), possibly as a result of reduced predation pressure (Dorn and Trexler 2007, Dorn 2008, Kellogg and Dorn 2012).

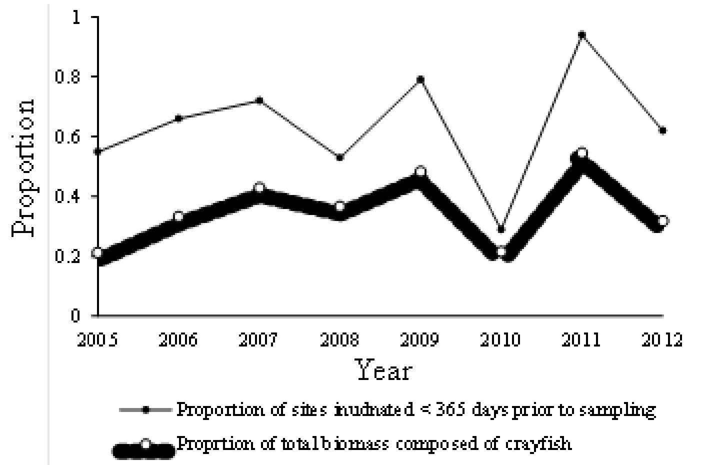

Figure 6-50. Proportion of sites where the marsh surface dried (water depth < 5 cm in data from EDEN) in < 365 days prior to sampling and proportion of total wading bird prey biomass composed of crayfish from 2005–2012 across all sites sampled in the Everglades.

Individual adult crayfish weigh more than most wading bird forage fish and are presumably a higher quality prey item for those wading bird species that can consume them. However, fish biomass exceeded crayfish biomass by a ratio of 7.4:1 over the entire Everglades study area during this study (**Table 6-10**). Consistent with finding that crayfish tend to have lower biomass in long hydroperiod conditions, fish comprised over twice the proportion of prey biomass in 2010, a wet year, than over the entire study (16.5 versus 7.4, **Table 6-10**). Crayfish biomass was high in 2011, a very dry year preceded by a very wet one (**Figure 6-50**), and the systemwide ratio of fish to crayfish biomass was almost half the average for the study (3.8 versus 7.4, **Table 6-10**; area-weighted average ratio for 2011 drops to 2.4 when outlier southern WCA 1 is excluded). Proportionally high biomass of fish appeared locally in some years (central and southern WCA 1; northern, central, and southeastern WCA 3A; northeastern SRS; and western SRS marl transition). In most of these instances, total prey biomass was not unusually high (compare to **Table 6-9**) and there is no correlation between total prey biomass and the ratio of fish to crayfish biomass ($r = -0.01$, $P = 0.92$, $n = 104$). The first two years of the study had low prey biomass (**Table 6-9**), though these were not particularly wet years (**Figure 6-48**). The prey composition was different in these years; 2005 had a high proportion of fish biomass in two large regions (southern WCA 1 and central WCA 3A, **Table 6-10**), while 2006 had proportionally low fish biomass systemwide (1.3 versus 7.4, **Table 6-10**). The apparently low systemwide productivity of 2005 and 2006, and fluctuation in prey type, is not explained by systemwide drying and requires further analysis. The last year reported here, 2012, was hydrologically similar to the first two years of the study, with an intermediate proportion of the study sites drying (**Figure 6-48**). That year had the third lowest prey biomass, after 2005 and 2006 (**Table 6-10**).

Table 6-10. Ratio of fish to crayfish biomass (g/m^2 wet weight) from 2005 through 2012 in LSUs in which at least three PSUs were sampled in each year.

Landscape Sampling Units	Fish/Crayfish Biomass								Annual Mean
	2005	2006	2007	2008	2009	2010	2011	2012	
Central WCA 1	11.5	1.4	1.2	1.0	22.7*	13.9*	1.8	1.2	**6.8**
Southern WCA 1	69.5*	2.9	5.4	2.4	30.6*	1.7	40.0*	3.9	**19.6**
Northwest WCA 2A	2.2	1.9	8.2*	2.1	0.8	2.7	0.4	2.6	**2.6**
Southern WCA 2A	4.9	0.3	1.4	4.0	1.5	12.6*	4.2	11.1*	**5.0**
Northern WCA 3A	3.2	0.3	3.1	9.3	1.8	31.2	1.7	4.0	**6.8**
Central WCA 3A	30.7*	1.0	1.9	5.1	3.1	4.5	4.1	10.3	**7.6**
Southeast WCA 3A	1.0	0.8	21.7*	2.8	22.6*	3.7	2.0	4.3	**7.4**
WCA 3B	3.2	2.1	6.2	1.5	3.0	11.4*	5.5	3.3	**4.5**
Northeast SRS	9.6	1.6	1.1	9.6	31.1*	25.4*	0.5	19.9	**12.4**
SRS	2.4	6.1	3.6	7.0	2.6	30.5*	2.1	0.9	**6.9**
Western SRS Marl Transition	1.8	0.3	1.0	7.7	0.3	50.1*	0.4	53.0*	**14.3**
Eastern SRS Marl Transition	1.2	0.4	0.9	0.7	0.3	3.3*	0.2	1.0	**1.0**
Western Perrine Marl Prairie	0.3	0.0	3.9*	0.2	1.9*	0.3	0.3	0.4	**0.9**
Total Area-Weighted Mean	**11.5**	**1.3**	**4.1**	**4.8**	**7.6**	**16.5**	**3.8**	**9.9**	**7.4**

*High fish biomass years, indicated by ratios approximately double or more than the annual mean ratio.
Marginal means are bold.

In the 8 years of this systemwide study, wet years yield high fish biomass and very dry years yield high crayfish biomass, somewhat compensating for each other in total wet season prey biomass. Though 8 years of study has been a major investment, systemwide patterns are just being seen from a diversity of year types to serve as the basis for restoration and operation recommendations. Questions remain. For example, why did 2007 and 2008 yield much higher systemwide biomass, particularly in WCA 3A and SRS, than 2005 and 2006? Years with intermediate levels of systemwide drying may be ones where intervention by hydrological management and restoration could have the greatest impact in shaping prey biomass; in drought years, there is very little water in the ecosystem for operations to use to influence recession rates, while in very wet years the entire system is inundated and there is limited capacity to drain water and alter local drying patterns. Future analyses will focus on these intermediate year types. The sequence of year types effects the time for prey biomass to recover from drought events. For example, in 1999 and 2000, two consecutive drought years led to a long lasting and profound effect of fish community composition and density unmatched in Everglades fish study records extending at some locations from 1977 to the present (Trexler et al. 2005). Longer time series will be required to provide opportunities to understand how the sequence of years affects prey production.

Dry Season Prey Concentrations

The dry season prey concentrations project monitors the spatial patterns of aquatic fauna densities in drying pools just as they become available to wading birds. Of particular interest are the inter-annual variation and correlations with local site characteristics (e.g., microtopography and vegetation types), hydrologic patterns, wet season fauna production, and ultimately, wading bird nesting numbers.

Prey density in drying pools has been monitored since 2005 during the wading bird nesting seasons (about January–May) using 1-m by 1-m throw traps. Estimates of wet season fauna production come from the calendar year preceding the dry season fauna and wading bird nesting data. Sampling sites across the Everglades landscape were arranged using a multi-stage sampling design with strata of LSUs, PSUs, sites, and throw traps. LSUs were delineated based on hydroperiod (number of days inundated with water) and vegetative characteristics. Each LSU contained a minimum of seven 500-m by 500-m randomly placed PSUs. Each PSU contained two randomly placed sites. Within each site, aquatic fauna were sampled from two randomly placed throw traps cleared with bar seines. Microtopography was characterized by taking water depth measurements every meter along a 100-m transect, centered on the first throw-trap, at each site. Additional water measurements were derived remotely from EDEN. From 2010 to 2013, dry season prey concentrations were characterized with 590 throw trap samples: 129 traps in 2010, 237 traps in 2011, 105 traps in 2012, and 119 traps in 2013. The discussion below provides comparisons between the 2006–2009 and 2010–2013 sampling years that are relevant to water management.

Temporal Patterns

The 2007 and 2008 sampling years provided examples of the hydrologic patterns that were associated with low prey availability (i.e., prey density and prey vulnerability) and wading bird nesting effort. Both the 2007 and 2008 dry seasons were preceded by relatively dry wet seasons, and thus began with low water levels (**Figure 6-51**). These low water levels resulted in low wet season prey production. Whereas steady water recession rates during the dry season usually give wading birds access to aquatic fauna, both 2007 and 2008 experienced periodic increases in water levels during the dry season, primarily due to rainfall, which made what little prey existed unavailable to wading birds. The result was low wading bird nesting effort.

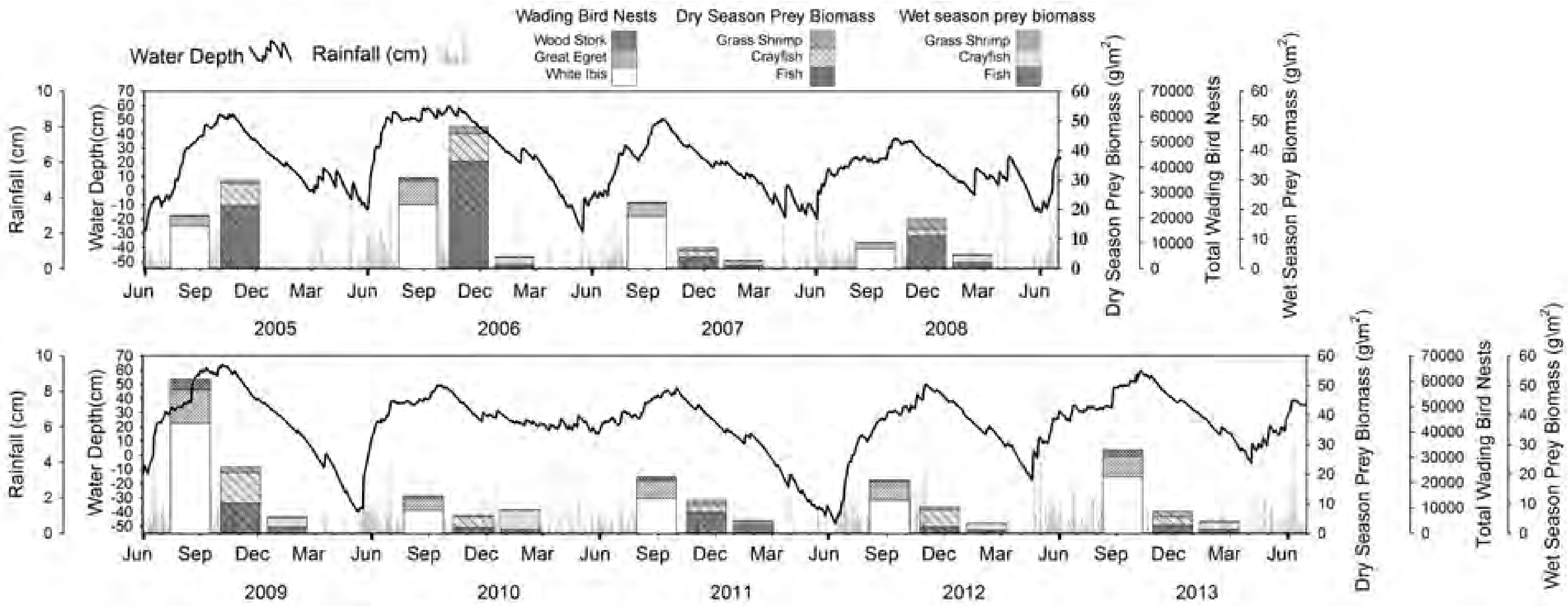

Figure 6-51. Mean rainfall, mean water depth, number of wading bird nests and wet and dry season prey biomass throughout the Florida Everglades from June 2005 to July 2013. Depth values represent the mean of 46,818 EDEN grid cells throughout most of the freshwater portion of the Everglades. Rainfall represents the mean of 9 gages throughout the Everglades. Medium gray bars represent grass shrimp, light gray are crayfish and dark grey are fish. Dry season biomass is differentiated by diagonal bars. Water years are demarcated by a long tic mark at the month of June. Wet season biomass (from Joel Trexler's Aquatic Fauna Regional Population study) is pooled mean biomass for the same LSUs within which dry season biomass was sampled.

In contrast, 2006 and 2009 demonstrated hydrologic patterns that produced high prey availability and wading bird nesting. Both 2006 and 2009 began with high water levels and experienced a steady dry down with little interruption from reversals. As a result, wading bird nesting effort was high as birds had access to high prey concentrations throughout the dry season. Whereas 2006 and 2009 had similarities in hydrologic patterns, the main driver producing high prey concentrations differed. In 2006, the preceding wet season experienced a long period of inundation with high water levels facilitating a moderate production of prey, particularly fish, at the start of the dry season. This moderate fish production, coupled with a steady recession rate, produced high prey availability. In contrast, 2009 did not start with particularly long inundation period, but had a steady and extended dry down throughout the dry season, which created a vast amount of foraging habitat available throughout the entire season. This suggests that there are multiple interacting mechanisms that can produce high prey availability for wading birds. Not only do different mechanisms affect prey availability, but these mechanisms can produce different prey compositions. In 2006, the long inundation period provided time and space for fish production, although, curiously, wet season biomass was only moderately high. However, the extended dry down in 2009 resulted in drought-like conditions, which hampered fish survival and mobility. Instead, an increase in the proportion of crayfish was observed. Crayfish have the ability to cope with droughts by burrowing in the sediment, and are more likely to produce and survive with a decrease of predatory fish. One key uncertainty that remains is to what degree the different wading bird species take advantage of high crayfish densities.

The 2010–2013 sampling seasons were years with extreme wet or dry hydrological conditions (**Figure 6-51**). All years had low prey concentrations and wading bird nesting, but the conditions that inhibited prey production and/or concentrations differed among years. The 2010 dry season was characterized by a lack of a water level recession, frequent reversals, and overall high water levels, which created minimal available habitat for foraging wading birds. Although water levels were high during the 2010 dry season, the following wet season had below average water levels. The low starting water levels in 2011 coupled with a rapid water level recession caused much of the landscape to become available and desiccate before it could be used by foraging wading birds. Even the longest hydroperiod regions of the Everglades dried almost completely, leaving very little habitat available to nesting birds later in the dry season. The dry seasons of 2012 and 2013 were both preceded by low wet season water levels that hindered prey production, and were marked by rainfall during the dry season, which interrupted the water level recession and diminished the concentration of an already low prey base.

Important Variables for Producing High Quality Foraging Patches

Results from models show that recession rate and microtopography are critical contributors for creating high quality foraging patches for wading birds. Additionally, this analysis confirms the importance of wet season standing stock to the concentration of fish the following dry season and shows that the density of submerged vegetation also affects fish concentrations. Microtopographical relief creates shallow depressions that allow fish to concentrate before the marsh dries completely and rapidly receding water is a mechanism that distributes fish and macroinvertebrates into these depressions. The relationship of prey production during the wet season with high quality prey patches during the following dry season was poorly understood prior to this study, although it was assumed to be important. Strong support for wet season fish and crayfish biomass in the top models illustrates the first quantitative link between wet-season standing stock and concentrations of fish and crayfish the following dry season. However, the importance of abiotic factors like recession rate and microtopography show that fish production alone does not produce the high quality foraging patches on which wading birds depend. The models also highlighted differences in how hydrologic fluctuations affect fish and crayfish. White ibises prefer to eat crayfish, while wood storks (*Mycteria americana*) prefer to eat larger fish (≥ 2.0 cm). Recession rate and microtopography were less important for crayfish than fish, underscoring how particular hydrologic patterns could increase food availability for one wading bird species and not another. An important test of the trophic hypothesis is whether the integrated wet and dry season prey data can be used to predict wading bird nesting. This study shows that wet season prey biomass is not related to nesting in a direct way (**Figure 6-51**); however, the models confirm that wet season prey are a key component of dry season prey availability, which is directly related to wading bird nesting (**Figure 6-51**). This is a strong confirmation of the trophic hypothesis but leaves a number of uncertainties related to the response by individual wading bird species to particular prey communities.

Implications for Management

The contradictions between the abundance patterns in fish-eating wading bird species and their prey in 2009 led to the proposal that the landscape availability hypothesis, which states that the area of suitable habitat in the landscape can sometimes be the primary determinant of wading bird prey availability. A very high proportion of the landscape became available in 2011, but there was not correspondingly high wading bird nesting effort, as expected. It could be that the temporal pattern of the dry down is as important as the spatial extent. Although water levels began to recede around the same time in 2009 and 2011, the peak wet season water level was much lower in 2011, causing the location of the drying front to be much farther down the elevation gradient by the time breeding wading birds were searching for suitable foraging patches. In 2012 and 2013, low water levels following the massive 2011 dry down persisted and slowed the recovery of prey populations. Subsequent wading bird nesting was low, suggesting that wet season prey production, the amount and quality of dry season prey concentrations, and the spatial and temporal pattern of prey availability across the landscape all can influence wading bird nesting, with the relative contribution of each varying from year to year based on hydrological conditions.

The amount of prey that becomes available during the dry season is partially dependent on wet season production; however, an optimal water level recession across a large spatial area can offset poor production. Thus, the highest prey availability occurs when a long period of inundation that promotes prey production is followed by a prolonged and strong recession rate. Because the ecosystem cannot maintain large areas with multiyear hydroperiods and have extensive areas that dry every year, it is reasonable to expect that particularly good nesting years may be followed by years with lower nesting effort. As hydroperiods are restored across a larger spatial extent of the Everglades, poor nesting years should become less frequent because good foraging conditions are more likely to exist somewhere in the system.

Long-term Trends of Fish and Crayfish Biomass and Community Composition in the Central and Southern Everglades

Fish and crayfish biomass and hydrological data collected in SRS, Taylor Slough, WCA 3A and WCA 3B between 1977 and 2012 were analyzed to evaluate long-term trends at 19 study sites (Trexler et al. 2013) (**Figure 6-52**). Analysis focused on fishes within the size range consumed by wading birds (between 18 and 80 millimeter standard length) and crayfish to evaluate dynamics of wading bird prey. The actual length of the time series analyzed varied among the study sites. Nineteen sites provided 16-year time series from 1996 through 2012. Additional years of study at three of those sites in SRS provided 27-year time series starting in 1985, while two sites in SRS provided 33 years of data beginning in 1977. Marsh drying is known to reduce biomass of aquatic animals in the Everglades by forcing them to move across the landscape to seek hydrological refuges to avoid death by desiccation or predation. The recovery of aquatic animal biomass following marsh drying in long hydroperiod areas can take several years and may not be complete before a subsequent drying event resets the recovery process. The concentration of aquatic animals, particularly fishes and crayfish, caused by periodic drying is a key process in sustaining high numbers of nesting wading birds. Management to recover historical nesting populations of wading birds must include spatially structured dynamics created by seasonal cycles of rainfall and inter-annual variation in the severity of marsh drying. This study reports analysis of long-term trends in biomass of wading bird prey.

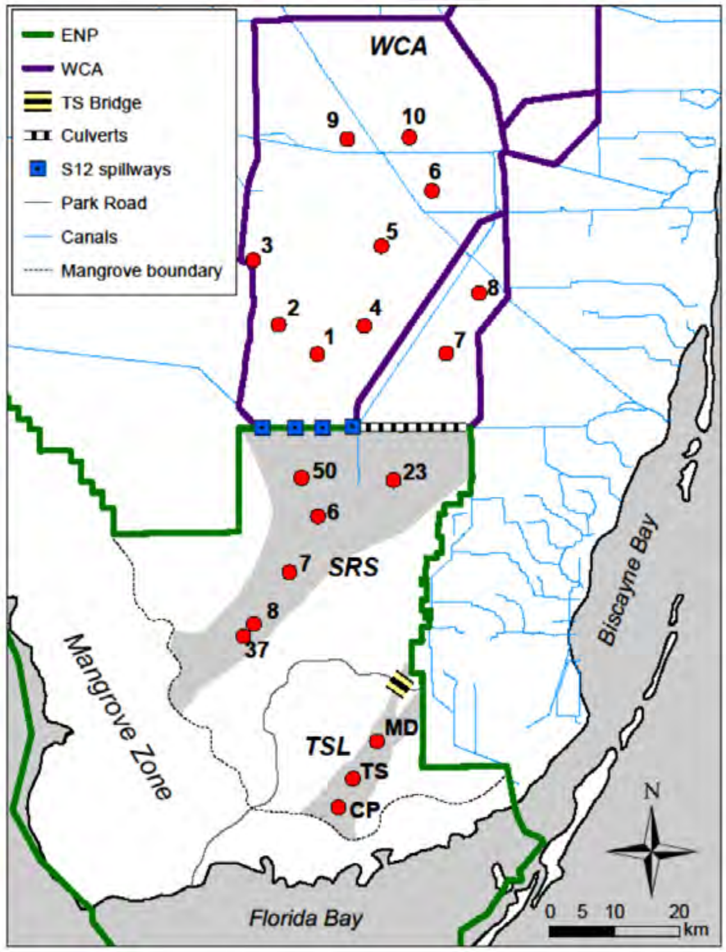

Figure 6-52. Map of study sites showing management structures (S-12 spillways, culverts, and the Taylor Slough Bridge). Approximate boundaries of SRS and Taylor Slough are shown in gray.

Fish biomass declined at five of six long-term study sites in the SRS, three of three sites in Taylor Slough, five of eight sites in WCA 3A, and two of two sites in WCA 3B (**Table 6-11** and **Figure 6-53**). The rate of decline varied among sites, in part related to their location within each study region, but on average, an 11.2% decline was noted in Taylor Slough, 9.5% in SRS, and 13.2% in WCA 3A and WCA 3B between 1996 and 2012. The steep decline in the WCAs was largely the result of large biomass reductions at sites north of Alligator Alley. The modest drop in SRS was influenced by increasing biomass between 1978 and 2000 at one site in northeastern SRS (SRS site 23), consistent with a past report on benefits from Water Delivery Test 1 (Trexler et al. 2005); however, biomass at that site has declined since 2000. The long-term decline in biomass noted overall in SRS began before the start of widespread sampling in 1996. One plot at site 6 was sampled continuously beginning in 1978 (SRS site 6). Over this 33-year time period, fish biomass at SRS site 6 exhibited a decreasing trend with a 14.1% overall drop, despite a temporary increase in the late 1990s likely associated with an unusually high amount of precipitation (**Figure 6-54**). In many cases, fish species composition has changed to reflect an increase in the relative abundance of species that thrive in short hydroperiod conditions. Crayfish biomass has also declined since 2000, though in more of a step-wise change because of a switch in species dominance. Prior to 2000, slough crayfish (*Procambarus fallax*) dominated SRS collections and was the most common species in Taylor Slough. The slough crayfish species is indicative of long hydroperiods, in contrast with the short-hydroperiod Everglades crayfish (*P. alleni*). Since 2000, the Everglades crayfish has become the most common crayfish in both regions, where it maintains lower biomass populations than the slough crayfish. As a result, crayfish biomass has decreased over the study period at several sites in SRS and Taylor Slough. In WCA 3A and 3B, the biomass of slough crayfish has also declined at several sites in these regions, while Everglades crayfish continues to hold low numbers compared to slough crayfish.

Table 6-11. Mean fish biomass (g/m²) in 1996 and 2012, difference, and percent change reported for each site.

Region	Site	1996	2012	Total Change	% Change
SRS	6	1.9	1.5	-0.3	-18.1
SRS	7	2.0	2.0	0.0	-2.4
SRS	8	2.0	1.8	-0.3	-12.5
SRS	23	1.3	1.4	0.1	10.2
SRS	37	4.1	3.2	-0.9	-22.5
SRS	50	0.9	0.8	-0.1	-11.8
Taylor Slough	CP	1.0	0.8	-0.1	-13.3
Taylor Slough	MD	1.7	1.6	-0.1	-4.6
Taylor Slough	TS	1.2	1.0	-0.2	-15.8
WCA	1	1.3	1.3	0.0	3.5
WCA	2	1.8	1.6	-0.2	-12.5
WCA	3	2.0	1.6	-0.4	-21.1
WCA	4	1.5	1.5	0.0	0.7
WCA	5	2.1	2.1	0.0	-2.2
WCA	6	1.4	1.5	0.1	9.0
WCA	7	2.1	1.9	-0.2	-8.0
WCA	8	2.0	1.5	-0.5	-23.5
WCA	9	2.5	1.8	-0.8	-30.4

WCA	10	2.5	1.3	-1.2	-47.3

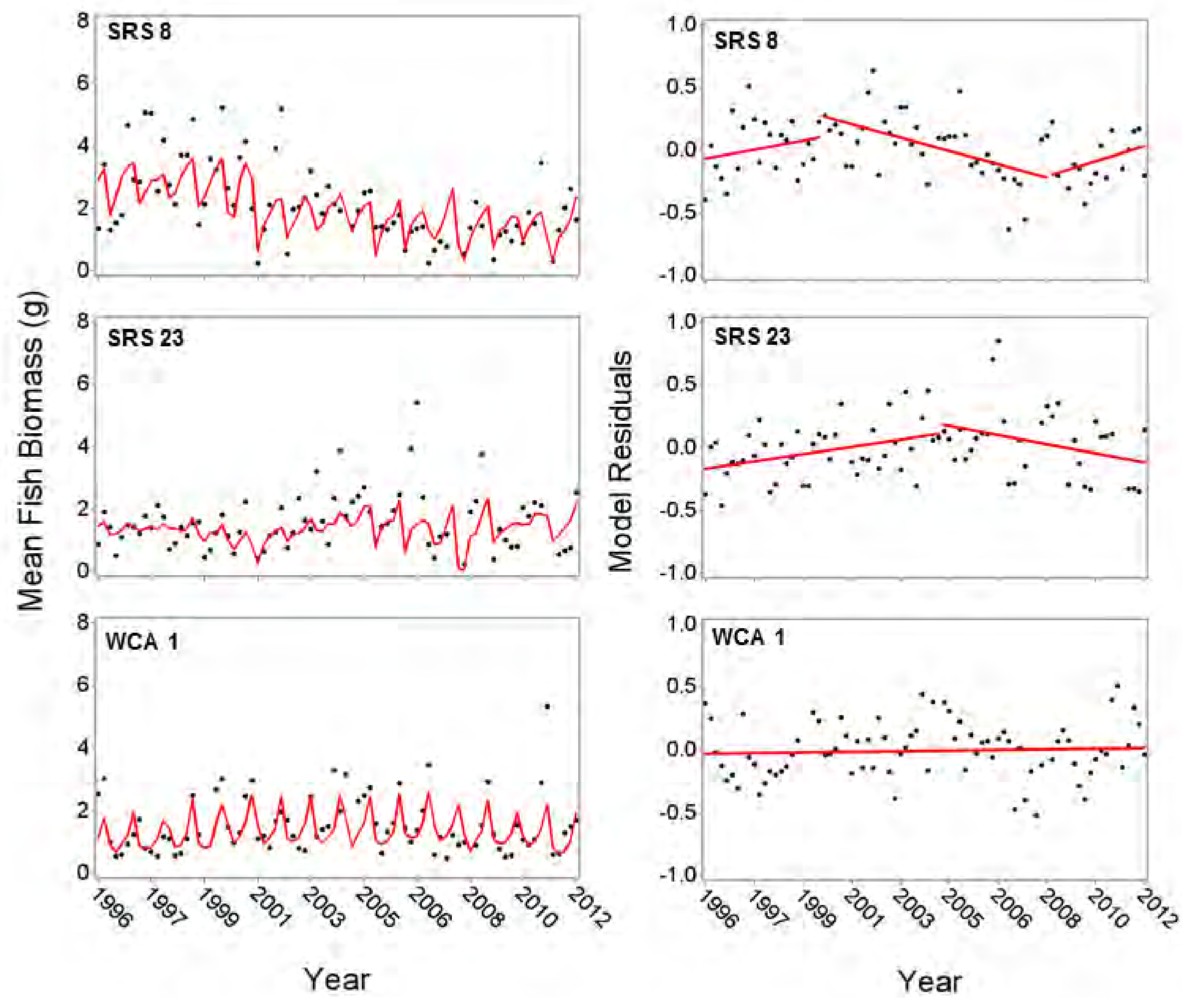

Figure 6-53. Fish community biomass time series from February 1996–April 2012 for three example monitoring sites. Column 1: observed values (black dots) and model predicted values (red line) of total fish biomass (grams [g] per square meter) for SRS sites 8 and 23 (northeastern SRS) and WCA 3A site 1. Predicted function from the full model including all hydrologic variables. Column 2: fish biomass model residuals (black dots) and piecewise regression functions (red lines) fit to unexplained variation in each time series. SRS sites 8 and 23 are in different spatial areas of SRS where hydrologic conditions were modified during the time series. WCA site 1 is north of Tamiami Trail where hydrologic conditions have remained relatively stable.

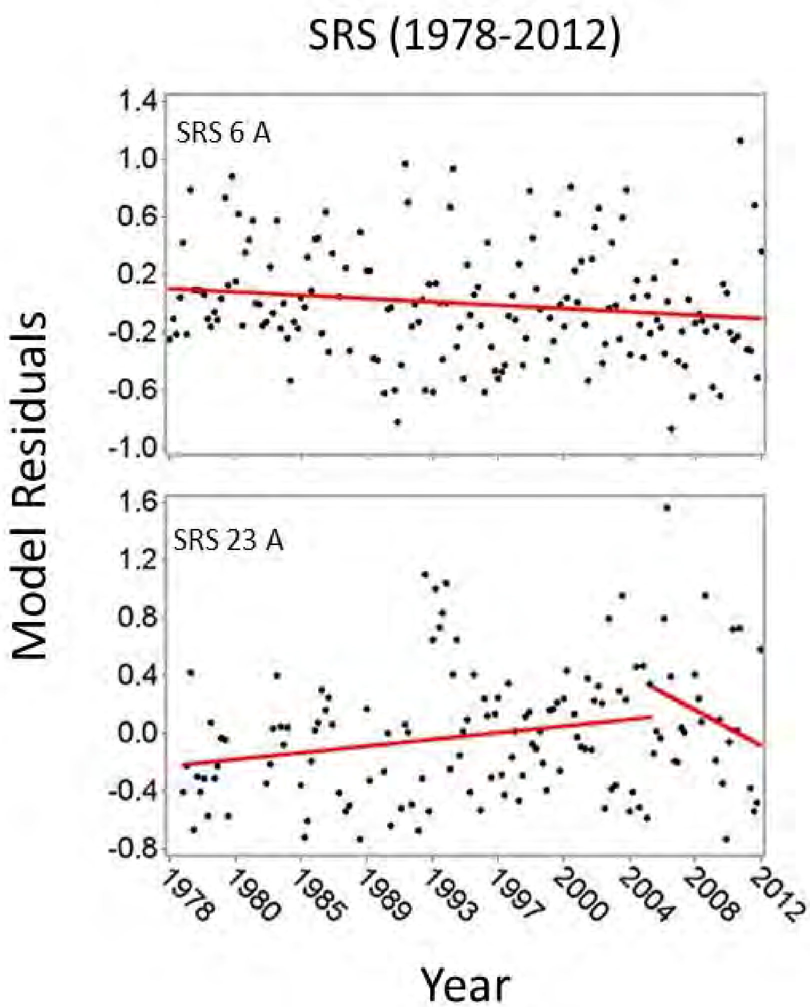

Figure 6-54. Fish biomass model residuals (black dots) and piecewise regression functions (red line) fit across the time series (1978–2012) for SRS sites.

The effects of management control of hydrology on these patterns were then evaluated using a two-step analysis. First, flow from upstream water structures was determined to explain the majority of variation in water depth at long-term monitoring sites in SRS (flow from the S-12 structures) and Taylor Slough (flow metered at the Taylor Slough Bridge). Though variable, water depth did not display trends over the course of the study (e.g., one or more rainy years were followed by one or more dry years, to yield no net trend). Regional rainfall explained very little of the observed variation in depth. However, water flow from upstream structures did show long-term trends toward lower flows, particularly in the January through March dry season months. Over the POR for EDEN data (1992–2012), a general trend toward increased probability that the marsh surface would dry for some period each year was found at fish-crayfish monitoring sites. This pattern could not be explained by rainfall changes, which was a covariate in the analysis. In summary, the operation of structures upstream from the monitoring sites appeared to result in a greater frequency of marsh drying between 1992 and 2012 than would be expected by rainfall.

The second step in evaluating the source of trends in fish and crayfish biomass was to incorporate hydrological parameters from the study sites into models of biomass. For fishes, the parameters were (1) days since the site was last dry; (2) water depth at the time of sampling; (3) rate of change in water depth from 30 days prior to sampling; and (4) season (coded from wet season to dry season: 1 for July, 2 for October, 3 for December, 4 for February, and 5 for April). Biomass generally increased to an asymptote as time passed after a drying event, decreased (slightly) as depth increased, decreased if water depth increased and increased if water depth decreased, and varied seasonally from a low in July (many of the fishes were small recruits at this time and were excluded from this analysis) to a high in February and April. Declining trends in fish biomass remained at all but one site after accounting for these local hydrological factors known to affect fish biomass. The results suggest that in addition to the immediate effect of increasing the frequency of marsh drying, forcing fish communities into a perpetual recovery condition, emergent negative effects on biomass were also present.

Two possible explanations for this emergent effect have been identified. First, changes in fish community composition from species favored in long hydroperiod marshes to species favored under short hydroperiod conditions may be favoring species that maintain lower biomasses. This is also the case with the crayfish, in which long hydroperiod slough crayfish are typically found at higher density than short hydroperiod Everglades crayfish. Second, it is possible that each drying event forces a lottery of survival that requires some minimal period of recovery; if these high mortality lotteries occur too frequently in succession, populations may not be able to recover adequately to have enough individuals to fare well in the next event. Therefore, each successive drying event further depletes the pool of survivors. It seems likely that there is a minimal return time between systemwide drying events that can sustain a highly productive aquatic community.

The hydrological analyses focused on SRS and Taylor Slough. Future work should explore the controls of hydrological variation at monitoring sites in WCA 3A and WCA 3B. Study sites located north of Alligator Alley, in western WCA 3A south of Alligator Alley, and in WCA 3B displayed long-term declines that should be examined in more detail.

This project examined the standing crop of fishes and crayfish with the focus on key prey items for wading birds. The CERP trophic hypothesis proposes that food availability for predatory birds is limited by the pattern of water recession. Recession patterns determine how potential prey are provided in high quality patches at water depths that allow birds to feed efficiently at key times in the nesting cycle. The trophic hypothesis proposes that the abundance of potential prey is a necessary antecedent to making prey available, but alone is not sufficient to assure abundant prey needed to sustain nesting. It is possible that the increased frequency of drying in SRS and Taylor Slough makes prey more available in the short term, while simultaneously depleting the store of prey regionwide and in subsequent years. Essentially, under such a scenario, wading bird consumption of prey resources could outpace the rate of prey replenishment. Wet season prey biomass contributes to, but does not alone explain, the formation of high quality prey patches, but low prey abundance may hamper their formation. At present we can only hypothesize that multi-decadal, slow declines in prey biomass will adversely affect dry season prey availability. Continued monitoring of prey biomass in sloughs and in drying pools where birds forage is warranted to better understand the implications for these long-term trends.

Wading Bird Nesting in the Greater Everglades 2010–2013

The four years of the current reporting (2010–2013) presented a number of different patterns in rainfall, hydrology, and temperature, with direct and indirect effects on prey base and nesting by wading birds (**Figure 6-55** and **Table 6-12**). Prior to the current four years, total numbers of nests increased dramatically beginning in the late 1990s, and that increase has been sustained through the present time. During the past four years, (2010–2013) compared to the previous nine (2000–2009) numbers of nests have decreased somewhat for all species, though the level of decrease may not be biologically significant.[2]

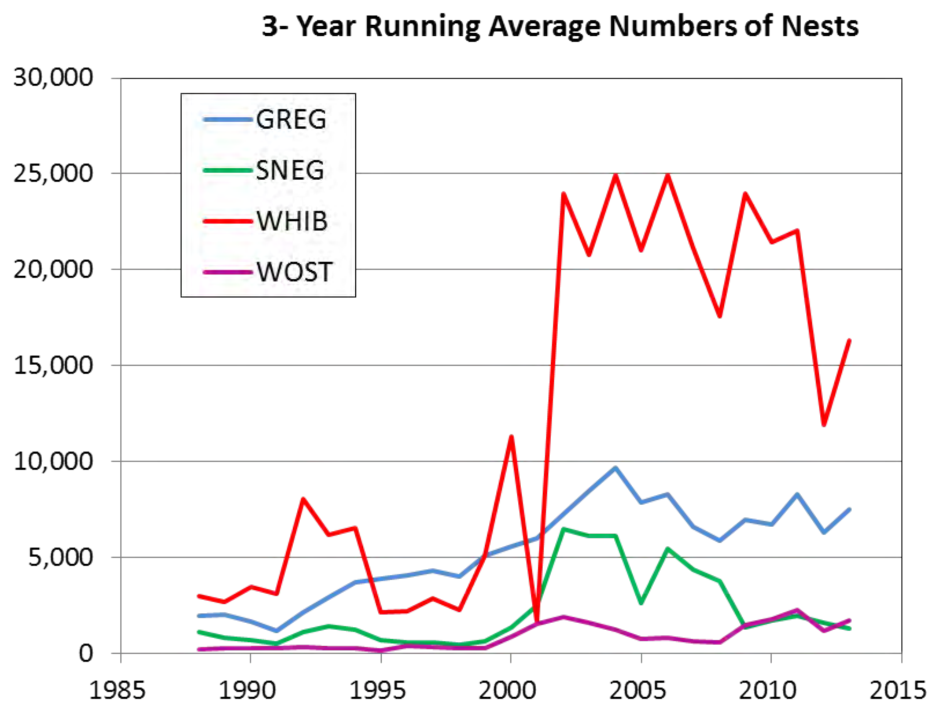

Figure 6-55. Total numbers of nests in the GE by four indicator species, 1986–2013.
Great Egrets = GREG, Snowy Egrets = SNEG, White Ibis = WHIB, Wood Storks = WOST.

[2] Three- (or four- or five-) year running averages of nests damp the volatility of single-year fluctuations and allow the effect of major management actions and weather events to emerge.

Table 6-12. Numbers of nest starts and nest success of wading birds[a] in GE, 2010–2013.

Year	Everglades National Park	WCAs	Total Everglades	Nest Starts Percentile Ranking [b]	Nesting Success
		White Ibises			
2010	3,975	5,072	9,047	low	high
2011	220	13,379	13,599	moderate	moderate
2012	5,050	7,972	13,022	moderate	low
2013	6,600	15,626	22,226	high	low
		Wood Storks			
2010	1,000	0	1,000	moderate	low
2011	1,257	470	1,727	moderate	low
2012	820	0	820	moderate	low
2013	2,005	506	2,511	very high	moderate
		Total Wading Birds			
2010	7,443	57,564	65,007	high	moderate
2011	4,468	12,431	16,899	low	moderate
2012	9,559	20,816	30,375	moderate	low
2013	12,505	14,629	27,134	moderate	low

a. Total wading birds include tricolored herons, little blue herons, black-crowned night herons, great blue herons, and roseate spoonbills as well as the species listed in **Figure 6-55**. Total nest numbers, however, are completely dominated by those four species figures in **Figure 6-55**.

b. POR 1931–2013; "Low" ≤ 50th percentile; "moderate" = 50th to 69th percentile, "high" = 70th to 89th percentile, "very high" ≥ 90th percentile.

During summer and fall 2010, most of the region received greater than usual rainfall, resulting in a long growing season for aquatic prey, and high stages at the start of the nesting season. Though drying rates were initially promising for concentration of prey in November 2009–January 2010, temperatures in December through February were very much colder than usual, and a series of rainfall events kept stages nearly flat during January and February, when water levels are usually falling. Early breeders (great egrets [*Ardea alba*] and wood storks) were stimulated to nest in large numbers, and wood storks began nesting relatively early (January). Roseate spoonbills (*Platalea ajaja*) also nested in dramatically larger numbers in WCA 3, and circumstantial evidence suggests they moved from Florida Bay due to poor nesting conditions there. However, early nesters like wood storks and great egrets had very poor nest success and high nest abandonment rates in response to a series of reversals. Later breeders (white ibis [*Eudocimus albus*]) were generally much more successful because they encountered warmer conditions and falling water. In addition to water level reversals, the dramatically colder weather of this season led to very low prey availability (Frederick and Loftus 1993) and consequently, high nesting failure. In addition, cold weather and poor food availability may also have affected adult birds, some of which were found moribund on particularly cold mornings.

The 2011 nesting season was preceded by lower than average rainfall in summer and fall, and lower stages at the beginning of the nesting season. While drying rates were high throughout the November–March period, stages went below ground level in ENP and many parts of the WCAs by mid-April, leaving nesting birds with little area for foraging. Storks abandoned nearly all of their nests during the latter part of the season, and ibises had both poor nesting success and low numbers of nesting attempts. When water disappears from wetlands surrounding colonies, mammalian predators often can invade, which was documented by large-scale predation on chicks and eggs by raccoons in these colonies, even in normally wetter parts of central WCA 3.

The 2012 season was the second in a row to be preceded by lower than average rainfall, resulting in short hydroperiods and short intervals between dry downs in many wetlands. Time since drying is a functional predictor of prey populations in the Everglades. The most dramatic (and predicted) impact was that nearly all species nested between one and two months later in 2012. This effect was evident for wood stork, great egret, and white ibis. In addition to nesting beginning late, a series of reversals in mid- and late April resulted in an interruption in falling water and concentration of prey at a time when both early and late nesters had young of various ages. This resulted in widespread abandonment by storks and very poor nest success generally.

The 2013 nesting season was preceded by relatively more rainfall than 2010 or 2011. Drying rates in the November–February period were rapid, and this stimulated a relatively large number of storks to initiate nesting in WCA 3 and ENP. However, this nesting was interrupted by a series of reversals in mid- and late March that led to intermittent parental feeding patterns, some abandonments, and generally low nest success by storks, especially in ENP. Although ibises initiated nesting in large numbers, nesting success was poor, with major abandonments at the Alley North and 6th Bridge colonies. The large number of ibis nesting attempts counted was likely to have been an overestimate of the numbers of pairs nesting, since there was evidence that many individuals failed in their first nesting attempt and then renested. For this reason, numbers of nest starts in this case may be somewhat misleading as an indicator of high quality nesting conditions.

Taking the broader view of progress towards restoration, the proportion of wading birds nesting in the coastal sections of ENP is thought to be a success indicator by restoring estuarine conditions which historically supported the majority of nesting in the predrainage Everglades (Frederick et al. 2009). This proportion has been variable during the past two decades, but remains markedly increased (mean 33%) by comparison with the mid-1990s (mean 9.7%) (**Figure 6-56**). A second indicator, the interval between exceptional nesting (> 70th percentile) by white ibises varied between 0 and 4 years during 2010–2013, well within the restoration target average of 1.6 years. Several other indicators suggest poor or only slight progress towards restoration indicators. In the predrainage period, nesting wading birds were dominated by tactile foragers (white ibises and wood storks) and visual foragers like egrets were much less common. This ratio of tactile to visual foragers is a restoration target because tactile foragers tend to need high or very high densities of prey in order to forage. The ratio of tactile to visual foragers (3-year running average) during the study period was 2.55, much below the restoration target of 35. Earlier wood stork nest initiation is expected with restoration of flows to coastal regions as a result of (1) the creation of food in coastal areas with appropriate hydroperiod, and (2) earlier availability of this food in coastal habitats because these habitats are higher elevation. The 4-year running average of stork nest initiation remains an early to mid-February initiation date, considerably later than the target of November and December. It should be clear that since much of the Everglades has not been

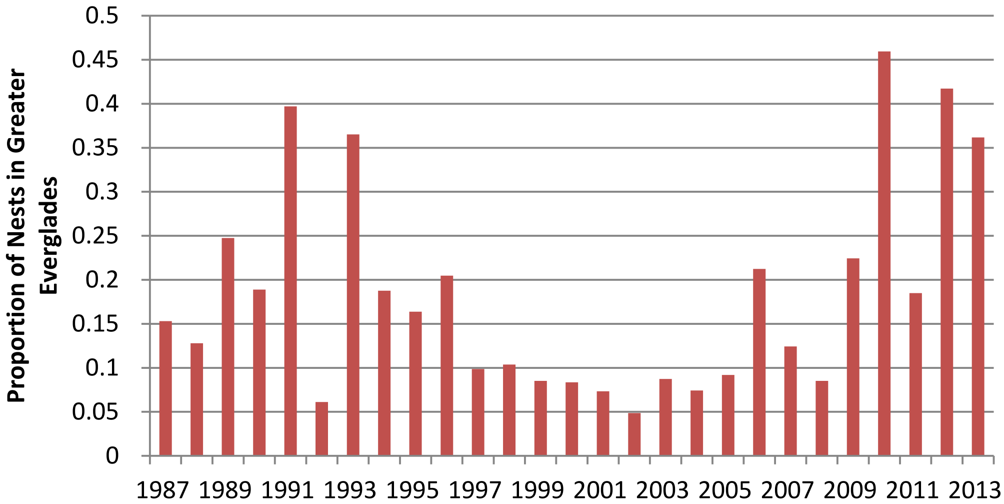

Figure 6-56. Proportion of all nests in GE (ENP + WCAs) that were found in coastal colonies 1987–2013. Target for restoration is > 70% of all nesting in the Everglades to be located in ENP.

hydrologically restored yet, much progress should not be expected as measured by these indicators. Essentially, the lack of positive responses are expected and are consistent with the current predictions of the hypothesis cluster.

Wading Bird Nesting in Lake Okeechobee 2010–2013[3]

Lake Okeechobee water levels in the 2010 dry season were atypical in that they stayed constant from January to March and then peaked in May, rather than declining from January to June, as in the past. Nest effort, defined as the cumulative total of the peak number of nests for great egret, great blue heron (*Ardea herodias*), white ibis, and snowy egret (*Egretta thula*), was the eighth highest on record. Lake Okeechobee's 33 years of nesting record dates back to 1957 (David 1994, Smith and Collopy 1995, Zaffke 1984). The period of reference for hydrologic patterns is from 1977–2013, which corresponds to the time period for which there are wading bird nesting data on the lake. Nest success in 2010, with the late start to the nesting season, was mediocre with many nests fledging on average ≤ 1 young (**Table 6-13**). In 2011, levels in Lake Okeechobee during the dry season were extremely low, reaching the lowest point on 24 June at 2.89 m (9.5 feet). Nest effort in 2011 was moderate, probably because of high prey availability and a prolonged dry down that began in September of the previous year. The majority of nests monitored fledged at least one chick by the end of May. Water levels in the 2012 dry season were higher than in 2011, but were still below average. Nest success in 2012 suffered from an exceptionally dry preceding wet season, below average water levels during the nesting season, and storms with high winds and rain; as a result, many colonies were abandoned and consequently experienced reduced fledging rates. Low nest numbers in 2012 were presumably due to the low prey production during the

[3] The littoral zone of Lake Okeechobee was defined in the 2004 MAP (RECOVER 2004) as being part of the GE in the trophic hypothesis cluster. In this SSR, the status of wading birds nesting on Lake Okeechobee is linked to both models.

short time that marshes were inundated during the 2011 wet season. In 2013, lake levels were high during the preceding wet season and then normal with an extended dry down throughout the nesting season. Nesting peaked in early April and nest success was high, with many nests fledging 2 to 3 young. Water levels pre- and during nesting illicit different responses from wading birds. If environmental conditions are poor during the pre-nesting period (December–February), nest effort is low as many birds forego nest initiation. If conditions are poor during nesting (March–June), wading birds will abandon their nests.

Table 6-13. Lake Okeechobee hydrologic conditions and wading bird nesting 2010–2013.

Year	Pre-nesting Hydrology	Lake Level during Nesting	Nest Effort	Nest Success
2010	Moderate	Average but Atypical	5,600	Low
2011	Low	Extremely Low	4,257	Moderate
2012	Extremely Low	Below Average	2,090	Low
2013	High	Average	6,919	High

A comparison of nesting between the Everglades and Lake Okeechobee showed that in 2010, nesting success was poor in both regions, but the timing of peak nesting was later at the lake. In 2011, wading birds in the Everglades were affected by dry conditions resulting in low nest effort as well as poor nesting success. Wading birds at the lake, on the other hand, managed far better with moderate nest effort and success. The 2012 nesting season suffered from below average water levels and a series of reversals, which resulted in widespread abandonment in both the Everglades and the lake. Nesting patterns in 2013 differed substantially between the two regions. Early in the season, relatively large numbers of wading birds initiated nesting in the Everglades. However, their nesting was interrupted by a series of reversals that led to nest abandonment, and low nest success. At Lake Okeechobee, water levels were ideal, nesting peaked in early April, and nest success was high.

Multiple Pathways by Which Hydrologic Patterns Affect Nesting

SFWMD records show that environmental conditions during the last decade have been relatively dry; a situation that may have elevated the influence of prey production during the preceding wet season. For example, the 2011 dry season had low lake levels similar to three of the five worst nest effort years on record (1971, 1981 and 2007), yet nest effort was within the top ten on record. On the other hand, 2012 water levels during the dry season began a foot higher than 2011, but 2012 had only average amounts of nesting effort. The main difference between these two nesting years was their respective preceding wet seasons; water levels in the WY 2011 wet season were high with good conditions for prey production and water levels in the WY 2012 wet season were exceedingly low. Likewise, nest effort in 2013 was the fourth highest on record and most likely the result of above average water levels during the preceding wet season.

Water levels during the breeding season can also affect nest effort. In 2010, nest effort would have likely been low if birds had not taken the unusual step (for south Florida) of initiating nests while water levels were rising. Birds in south Florida depend heavily on receding water to concentrate prey into high quality patches (Gawlik 2002) whereas this is not the rule in river ecosystems (Kingsford et al. 2010). Nest initiation that was independent of receding water levels suggests that suitable foraging conditions in Lake Okeechobee were produced by different mechanisms than in the Everglades. The average prey density in the lake was 133% higher than in the Everglades (**Table 6-14**), probably because productivity of the lake is higher.

Table 6-14. Mean prey density (standard error) found within 1-square meter throw-traps throughout Lake Okeechobee's littoral zone and the Florida Everglades in the dry seasons of 2011–2013.

Year	Lake Okeechobee[1]	Everglades[2]
2011	165 (21)	77 (8)
2012	87 (7)	32 (4)
2013	104 (9)	44 (5)
Mean	119 (6)	51 (3)

[1]Chastant and Gawlik, unpublished data
[2]Botson et al. 2013

Water levels affect not only foraging conditions, but also colony location and nesting substrate. Higher lake regulation schedules result in deeper water throughout the littoral zone, a situation that is not conducive to wading bird foraging nor is it beneficial for emergent or upland vegetation. About half as many wading birds nested in the lake when the water schedule was higher than it is today (1979–1992; David 1994). During the 1980s, water levels remained high, causing massive die-offs of willows (*Salix* sp.) and the encroachment of cattail (*Typha* sp.). Concurrently, nest effort declined and wading birds formed smaller more dispersed colonies on the edges of the marsh (David 1994). Willow head colonies in the marsh are characteristically large, usually more than 1,000 nesting birds, whereas spoil island colonies typically accommodate a few hundred nesting birds. More recently, 2011 and 2012 were dry years in which nest effort was low and colonies were located on spoil islands surrounded by deep water. In the wet years of 2010 and 2013, nest effort was high and birds nested on willow heads in the more traditional marsh locations.

The first attempt at linking lake levels and wading bird nesting using direct statistical models was not satisfactory because the models generally had low predictive power. Alternatively, a habitat suitability model that was driven by hydrologic patterns was developed, which showed a strong connection between habitat conditions on Lake Okeechobee and wading bird foraging and nesting patterns (Chastant et al 2012; **Figure 6-57**). Habitat suitability was based on factors that contribute to the accessibility, vulnerability, and abundance of wading bird prey based on functions derived from marsh elevation, lake stage, and vegetation (i.e., proxies for prey availability). A digital elevation model was used based on light detection and ranging (LIDAR) data collected by Merrick & Company from September through December 2007 for the Florida Division of Emergency Management. Lake stage was accessed through the SFWMD's hydrometeorological database, DBHYDRO, and vegetation was based on

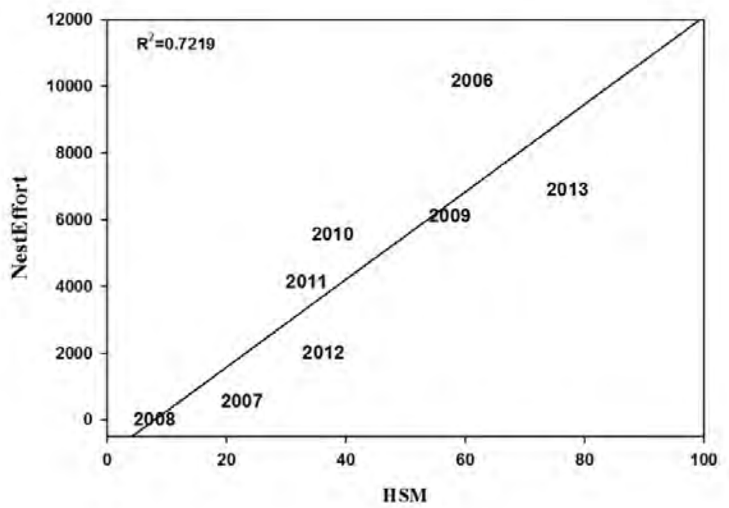

Figure 6-57. Regression of habitat suitability model values on yearly nest effort for white ibis, snowy egret, and great egret combined.

the 2007 vegetation mapping of Lake Okeechobee by SFWMD. Depth was calculated by subtracting the digital elevation model from lake stage, and recession was calculated by subtracting the present lake stage from the lake stage 14 days prior. A formal validation of the model using the locations of foraging flocks showed that the model could be characterized as "useful" as defined by Bradley (1997). Additional validation of the model based on correlations between nest effort for the great egret, snowy egret, and white ibis combined, and the mean habitat suitability index value of highest quartile during a dry season showed a strong positive relationship ($R2 = 0.7219$, $p < 0.0076$; **Figure 6-57**). Nest effort ranged from 10,176 pairs in 2006 to 20 pairs in 2008, and corresponded to habitat suitability index values of 0.61 and 0.08, respectively.

Value of Continued Monitoring Associated with the Trophic Hypothesis

The Everglades has the longest and largest continuously running wading bird survey program in the world (1965–present for ENP). In addition, the dimensions of the ecological inference of the resulting data set are vastly expanded by the inclusion of intermittent periods of historical information, beginning during the predrainage period over a century ago. The combination of historical and recent information has been instrumental not only in understanding the relationship between bird nesting, hydrology and prey species, but in offering a comprehensive picture of pre-drainage ecology. The long-term monitoring program has shown several important shifts by the birds that could be robustly interpreted as indications of ecological change resulting from management actions, some of which were never anticipated under any scenarios. Four examples are as follows:

- The movement of bird nesting away from the coast was one of the first indicators that the estuary was impaired (Ogden 1994), and subsequent work has demonstrated the relationship between freshwater flows, estuarine productivity, and avian nesting in GE. This trend was recently witnessed (2010 and 2011) through a general and unpredicted exodus of roseate spoonbill from Florida Bay, settling in the freshwater WCAs. This also illustrates a key feature of the birds as early indicators of restoration success, as they are rapid responders to improved hydrologic conditions.

- Wading bird nesting pulses over a 70-year period indicated that severe droughts play a special role in causing a pulse of prey availability critical to both bird populations and aquatic community dynamics (Frederick and Ogden 2001). This understanding has helped explain long-term dynamism in wading bird population responses, and suggested strongly that interannual variability was a crucial part of the restoration policy.

- Avian reproductive trends, and bird and fish tissue analysis showed both the dramatic rise, and equally dramatic fall, of a major and completely unpredicted mercury contamination problem in the 1990s (Frederick et al. 2004). These studies have also demonstrated a persistent area of contamination in the coastal zone of ENP. The exact magnitude of the risk to white ibises and wood storks is not known at this time, but it has strong ramifications because this is the exact zone predicted to be recolonized as a result of CERP hydrological restoration.

- Short-term and anthropogenic effects have been shown. The long-term nature of the wading bird record has, in several cases, allowed the effects of several unexpected issues of anthropogenic origin to be studied. These include the landfill near WCA 1, the Tamiami Trail road widening (1990s), construction of water control structures at the south end of WCA 3B, planned wind farms and power lines, Tamiami bridges, setback distances for ecotourism, and perceived conflicts between management for different endangered species, not to mention the likely effects of seasonal and long-term water management operations decisions.

In short, the existence of immediate, up-to-date information, and the ability to compare with a long-term record have been the two necessary components to turn wading bird reproductive information into an effective management tool that informs about the likely effects of a number of sometimes unexpected anthropogenic effects that may not have originally been envisioned as part of CERP. The larger ramifications are that coupled avian/aquatic prey studies have demonstrated a series of “ecological surprises”, the elucidation of which has greatly enhanced confidence in the basic hydrological restoration hypothesis that is integral to CERP. Since 2000, this linked program has used increased understanding of ecosystem variability to directly test and validate multiple facets of the trophic hypothesis, and confidence in the hypothesized links between restored water/restored aquatic prey/restored avian populations has been strengthened enormously.

Benefits of the MAP Trophic Hypothesis Cluster Integration

The MAP is a powerful integrative tool for guiding Everglades management and restoration. The most innovative feature of the plan is that it was designed to test hypotheses about the effects of management actions rather than just monitor changes in a set of indicators. The performance measures are aligned vertically through the food web along mechanistic pathways so that a response can be traced at multiple levels, each providing a confirmation or a rejection of the underlying hypothesized cause. In true adaptive fashion, the resulting knowledge can be fed directly back into management plans to adjust future scenarios or policies.

One property of the MAP that was not known at the onset was the synergistic effect of having principle investigators applying new results from their models and focused research studies to the MAP data, thereby enhancing its interpretation and pointing the way toward the next generation of focused studies. The increased rate of progress has allowed for the development of evaluation models for screening restoration alternatives, and assessment models for providing real-time measures of habitat conditions.

FISH DYNAMICS AT THE EVERGLADES MARSH-MANGROVE ECOTONE: EFFECTS OF SEASONAL HYDROLOGY AND EXTREME CLIMATIC EVENTS

Natural ecosystems typically have characterized pulsed disturbance regimes (e.g., seasonal hydrology), which strongly affect their ecology, and the Everglades are no exception. Recurrent drydowns affect inundation patterns, and the amount, extent and quality of habitat available to aquatic organisms, playing a major structuring role on fish communities of the Everglades. The monitoring of fishes (initiated in 2004) at the southern end of SRS points to the same structuring role of seasonal hydrology occurring in the marsh-mangrove ecotone (**Figure 6-58**). This is an important foraging and nesting habitat for wading birds, and a region of interest for restoration efforts.

The key objective of the monitoring effort is to gain a better understanding of how freshwater inflows affect freshwater and estuarine fishes, both the smaller fishes that form the prey base for wading birds and the larger taxa that constitute economically important recreational fisheries (largemouth bass [*Micropterus salmoides*] and snook [*Centropomus undecimalis*]) and thus a key ecosystem service provided by the coastal Everglades (**Figure 6-59**). Monitoring involves electrofishing at 10 fixed sites in the upper 12 kilometer of the Shark River in the southern Everglades (**Figure 6-58**).

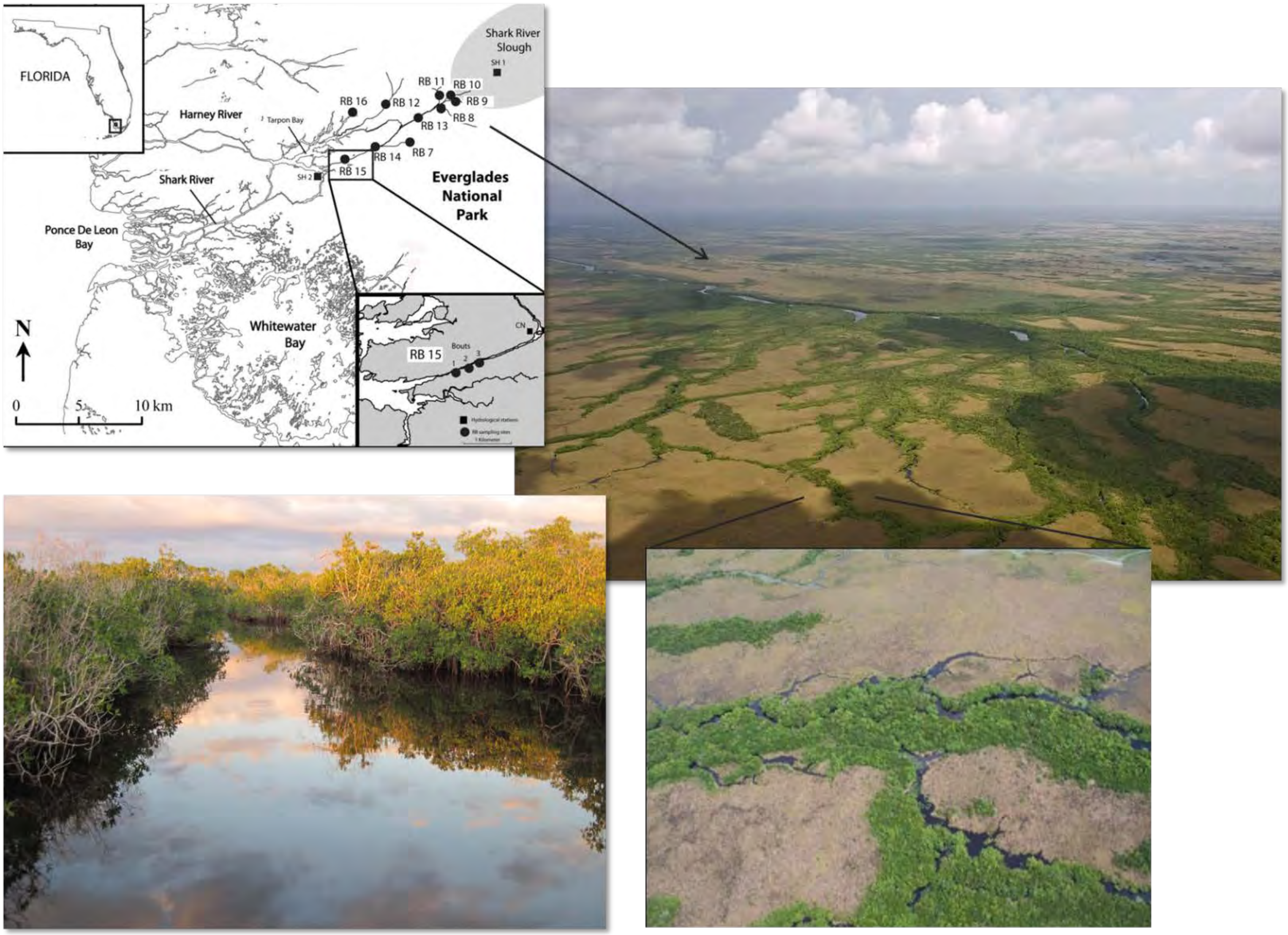

Figure 6-58. Map and images of sampling sites at the SRS-Shark River estuary interface, ENP. Shown are the location of the 10 fixed electrofishing sites and the location of the 3 replicate bouts per site in insert.

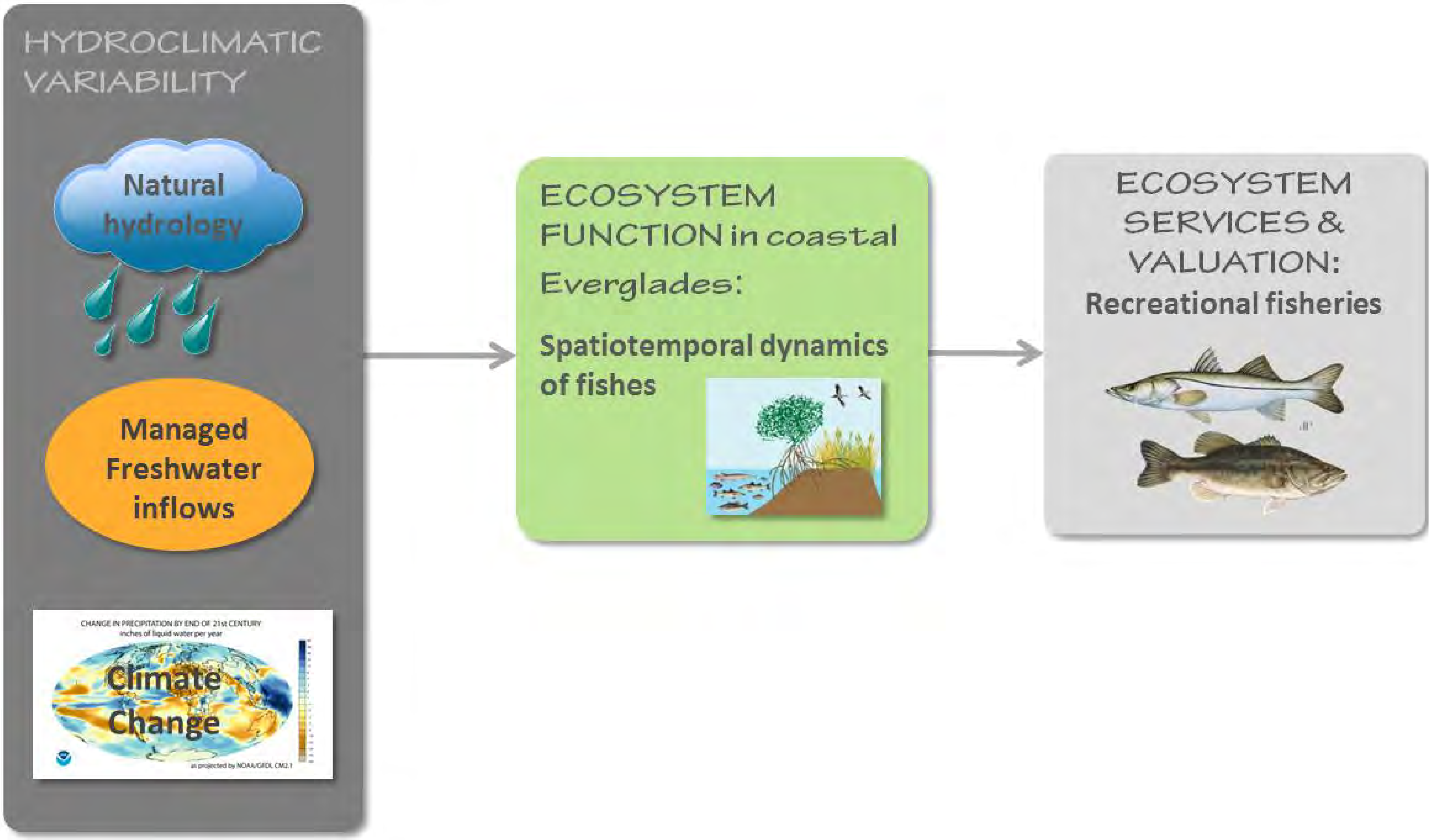

Figure 6-59. Broad conceptual framework developed based on the past 8 years of monitoring of fishes in ecotonal habitats in the southwestern Everglades. The hypothesis is that hydroclimatic variability is a main driver of the spatiotemporal dynamics of fishes and recreational fisheries in the mangrove zone.

Monitoring in the headwaters of the Shark Rivers shows that mangrove creeks are inhabited by a diverse and dynamic fish community composed of transient freshwater fishes, resident estuarine species (snook), transient marine taxa (tarpon [*Megalops atlanticus*]) and nonnative fishes (Mayan cichlids [*Cichlasoma urophthalmus*], African jewelfish [*Hemichromis letourneuxi*], andspot-finned spiny eel [*Macrognathus siamensis*]). In the dry season, fish abundance in mangrove creeks increases 5 to 6 fold, driven primarily by increases in the abundance and richness of freshwater fish (**Figure 6-60**). These freshwater species include both small-bodied prey species (sunfishes [*Lepomis* spp.]) and larger predatory taxa (largemouth bass, bowfin [*Amia calva*] and Florida gar [*Lepisosteus platyrhincus*]), which pulse into the estuary from upstream marshes at the onset of the dry season. Increases in abundance and species richness, however, are limited to the uppermost headwater sites of the Shark River; downstream, no evidence of this seasonal influx is found coming from drying marshes (**Figure 6-61**). This increase in the abundance of freshwater fishes is accompanied by increases in the abundance of estuarine predators, dominated by snook (**Figure 6-62**). Previous work has shown that small marsh fish form an important prey base for resident snook, which triple in abundance at the headwaters upon marsh drying. Interestingly, nonnative fishes show no increases in abundance in the dry season, but their numbers are consistently higher at headwater sites relative to sites downstream in the Shark River (**Figure 6-62**).

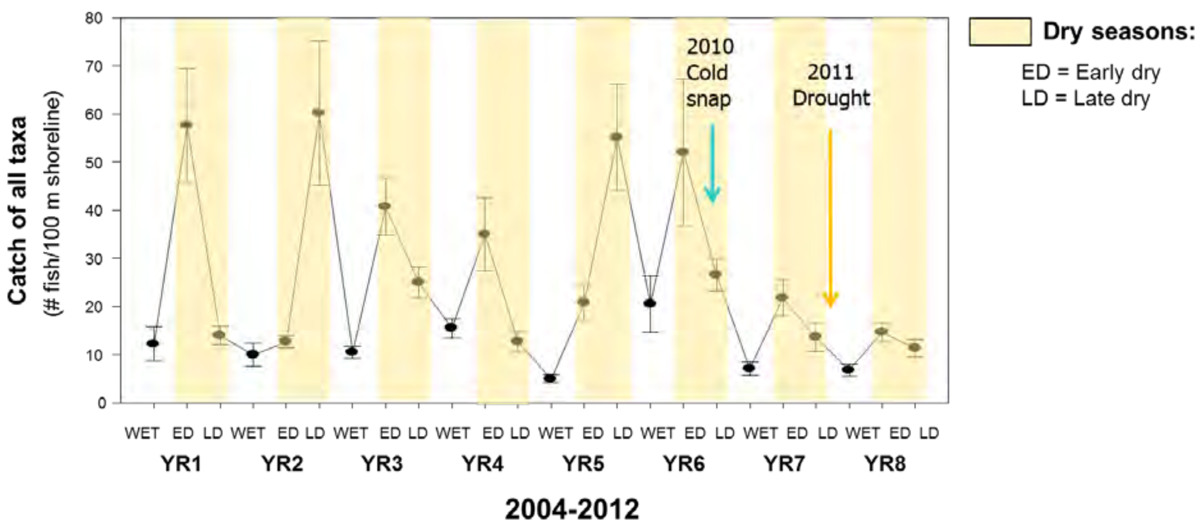

Figure 6-60. Seasonality in electrofishing catches (number of all fish caught per 100 m of creek shoreline, ± standard error [SE]) across the three sampling events: wet (November, white shading), and the early dry (February–March) and late dry seasons (April–May, both in yellow shading). Shown also are the timing and effects of the two extreme climatic events on fish abundance: the 2010 cold snap and the 2011 drought. (YR – year.)

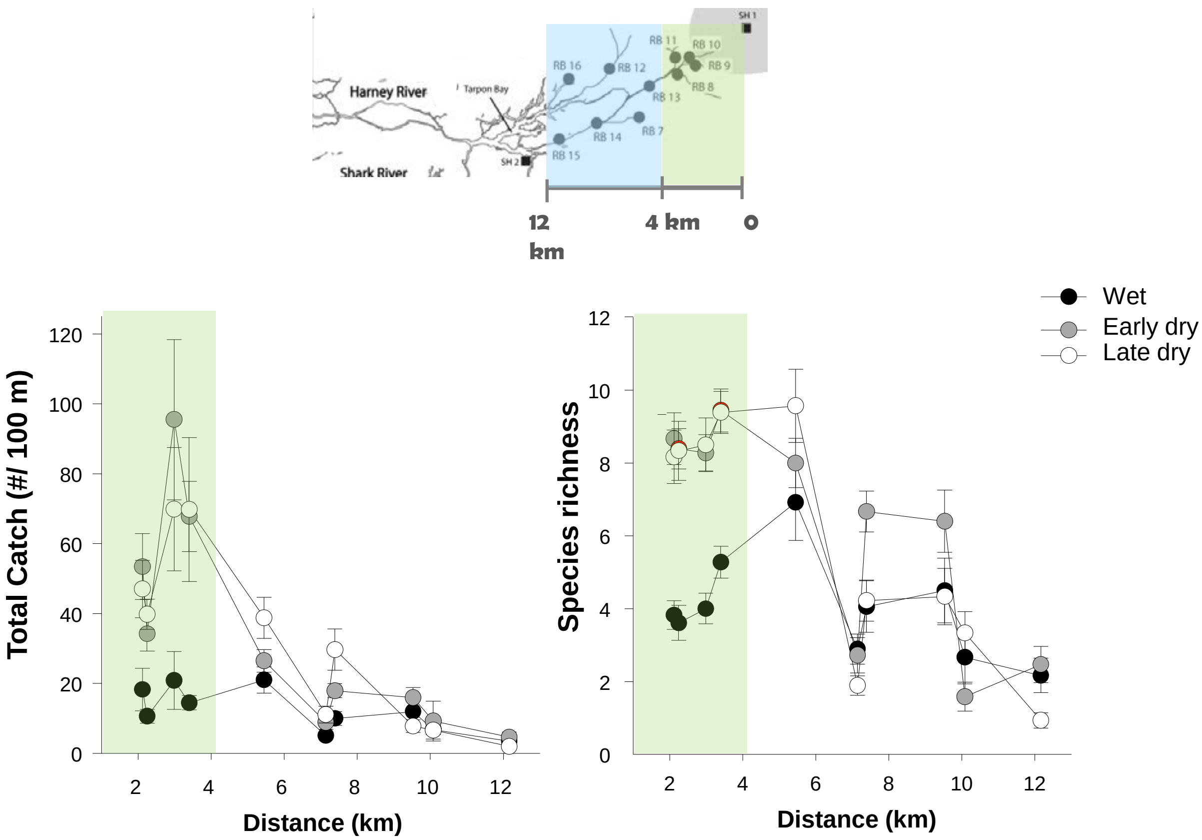

Figure 6-61. Spatial variation in total electrofishing catches (all species combined, number of fish per 100 m of mangrove creek shoreline) and species richness (means ± SE) as a function of distance to upstream marshes for the three seasonal samples. Green shading indicates sites located in the upper 4 km of the estuary and closest to freshwater marsh habitats.

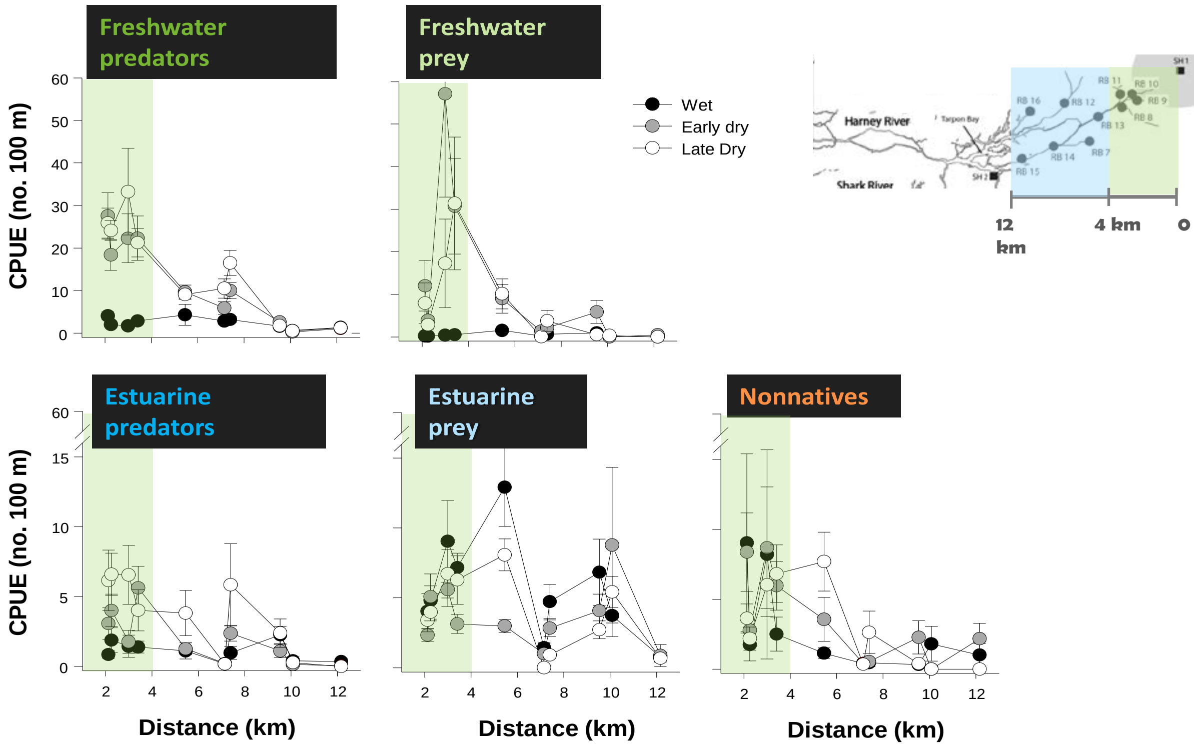

Figure 6-62. Abundance (number of fish per 100 m of mangrove creek shoreline) of the five main functional groups of fishes found in the ecotone (mean ± SE): freshwater or marsh predators, freshwater or marsh prey, estuarine predators, estuarine prey, and nonnative taxa as function of distance to marshes for the three seasonal samples. Green shading indicates sites located in the upper 4 km of the estuary and closest to freshwater marsh habitats.

Analyses of time series shows that patterns of fish abundance and richness are primarily driven by a combination of regional hydrological disturbance (seasonality, marsh water depth and days since marshes upstream of study creeks last dried) and local conditions experienced by fish in creeks (i.e., salinity, creek water depth, dissolved oxygen and temperature; **Figure 6-63**). But the relative importance of regional versus local processes varies across different components of the community, whether fresh water or estuarine and whether larger-bodied predators or smaller prey taxa. For instance, the number one predictor of total fish abundance in ecotonal mangrove creeks is regional hydrological variation. Close to 60% of the variation in total fish numbers can be explained: 21% is solely attributed to regional hydrological disturbance parameters, 12% is accounted for by local creek conditions, 4% is due to the spatial location of creek sites (distance to upstream marshes), and 19% is due to some combination of these factors (**Figure 6-63**). In contrast, fish numbers varied little as a function of sampled years. Overall, regional hydrological disturbance played a stronger structuring role on the freshwater and nonnative fishes, while the abundance patterns of estuarine taxa were more heavily influenced by local abiotic conditions, particularly salinity. However, for estuarine predators, regional hydrology still played a major structuring role, accounting for 30% of the explained variance in their abundance patterns. Regional hydrology had no effect on the abundance of estuarine prey

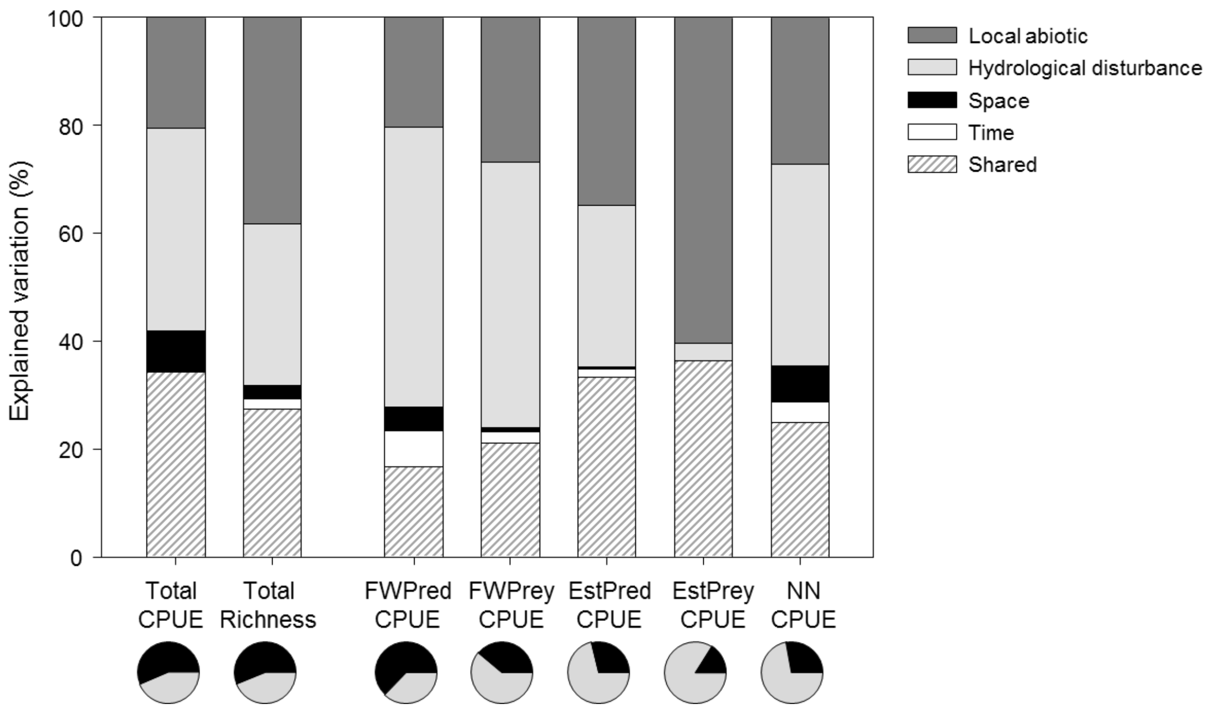

Figure 6-63. Percent of explained variance attributable to 4 sets of predictor variables: local abiotic creek conditions (salinity, temperature, dissolved oxygen and water depth), regional hydrological disturbance (marsh water level, days since last dry down and season), spatial factors (distance to upstream marshes), temporal factors (year of sampling), and to combinations of these (shared) for total fish abundance (catch per unit effort [CPUE]), total species richness, and for the abundances (CPUEs) of freshwater predators (FWPred), freshwater prey (FWPrey), estuarine predators (ESTPred), estuarine prey (EstPrey), and nonnative (NN) taxa. Pie charts show the total amount of variance explained (black) and the residual unexplained variance (grey).

(mojarras [*Eucinostomus* spp. and *Eugerres* spp,], mullet [*Mugil cephalus*], and hogchoker [*Trinectes maculatus*]).

Overall, results point to the biological coupling of freshwater and estuarine Everglades habitats, driven by the movement of marsh fishes into mangrove creeks during the dry season, and by the movement of estuarine predators between upstream and downstream regions of the Shark River estuary. This hydrological-driven displacement of marsh fishes provides foraging opportunities for important recreational fisheries such as snook, and trophically links both habitat types. Thus, mangrove creeks act as critical dry down habitats for marsh fishes, similar to the role of alligator holes, solution holes and canals elsewhere in the ecosystem. These findings are in agreement with previous work on Everglades fishes showing the overwhelming effect of seasonal hydrology on their abundance and distribution patterns. But, they also point to how seasonal drydown in marshes can spill over to influence population dynamics of fishes in an adjacent habitat type, the coastal mangrove zone.

AMERICAN ALLIGATORS

Trends in Alligator Relative Density 2003–2012

Introduction

Alligators integrate biological responses of hydrological operations throughout their life. Their reproduction and survival is dependent upon suitable hydrologic patterns and current populations in Everglades marshes and estuaries are suppressed because of altered hydrology and salinity (Mazzotti and Brandt 1994, RECOVER 2009, Shinde et al. 2013, but see http://www.cloudacus.com/simglades/alligator.php for documentation in review). In addition, alligators play a key role in shaping plant and animal communities through creation of deeper water (alligator holes and trails) and higher ground (nest sites) habitats. They also act as both prey (when they are small) and top predators and have a role in enhancing wading bird nesting success in areas where they may inhibit predation in colonies (Burtner 2011).

Restoration of hydrological and salinity patterns that more closely mimic what occurred naturally, including less frequent and intense dry downs in areas that currently dry down almost every year, and changes in water levels that correspond to seasonal patterns are expected to result in an increase in relative density of alligators and alligators that are in better health as measured by body condition.

In this summary, information from WY 2003 through WY 2012 is provided on patterns of alligator relative density in six areas that reflect different hydrological patterns. A water year begins on May 1 and ends on April 30 of the subsequent year.

Methods

Alligators are surveyed for relative density twice during spring and twice during fall according to protocols established and documented for the MAP (Mazzotti et al. 2010; **Figure 6-64**). Here we report on data from six areas: the WCA 1 (Lox on map), WCA 2A (WCA2A), northern WCA 3A (WCA3A-Tower), central WCA 3A (WCA3A-HD), WCA 3A southwest (WCA3A-N41), and northeastern SRS in ENP (Frog City). We selected these areas to report on because they represent contrasting hydrology with Lox, WCA3A-HD, and WCA3A-N41 being longer hydroperiod sites and WCA2A, WCA3A-Tower, and Frog City being shorter hyrodoperiod sites (**Table 6-15** and **Figure 6-65**). Our analysis uses data from WY 2003 through WY 2012 to look at the ten-year trend and trends in five-year increments.

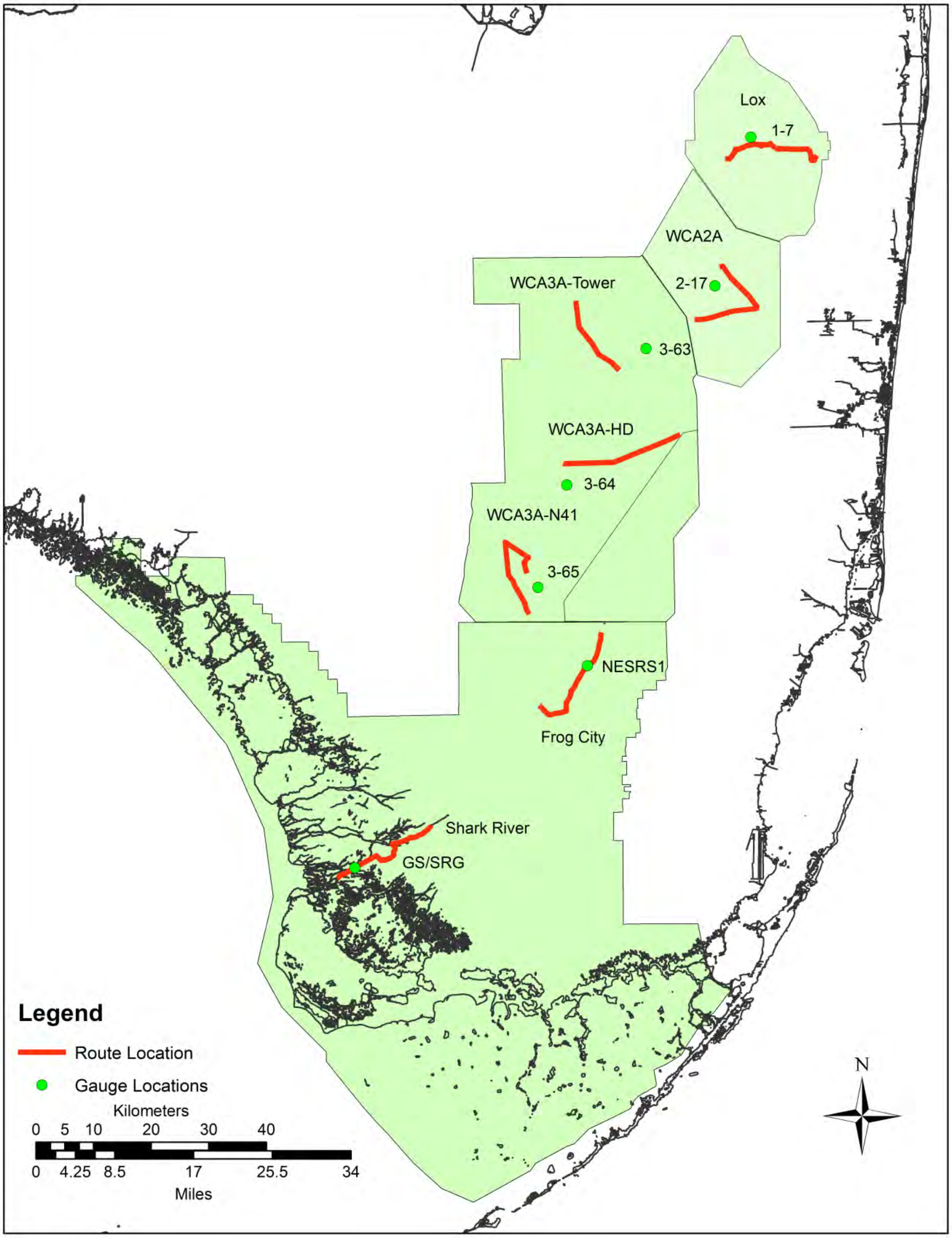

Figure 6-64. Location of alligator survey routes and gauges used for WY 2003–WY 2012 analysis of trends in alligator relative density. Results for the Shark River route are presented in the Southern Coastal Systems chapter.

Table 6-15. Hydrologic characteristics for WY 2003–WY 2012 for six survey areas listed from longest to shortest average hydroperiod. Gages used for analysis are 3-65, 1-7, 3-64, 2-17, 3-63, and NESRS1, respectively (Figure 6-64). Hydroperiod was calculated as number of days in each water year where the gage reading was equal to or above 15 cm (6 inches), below which is the depth that makes it harder for alligators to move around the marsh to feed and mate (Rice et al. 2004; Shinde et al. 2013). Average days since dry and average length dry are in reference to survey date.

	WCA3A-N41	Lox	WCA3A-HD	WCA2A	WCA3A-Tower	Frog City
Average Water Year Hydroperiod (Days ≥ 15 cm)	359	354	332	317	294	292
Average Days Since Dry	2075	850	303	249	249	204
Average Length Dry	10	34	25	63	78	93
Number of water years in last ten (WY 2003–WY 2012) that it was dry	2	4	8	10	10	9
Number of water years in last ten (WY 2003–WY 2012) that it was dry during courtship and mating (April 16–May 15)	0	1	3	4	7	6
Number of water years during WY 1993–WY 2002 that it was dry	2	2	3	7	6	5

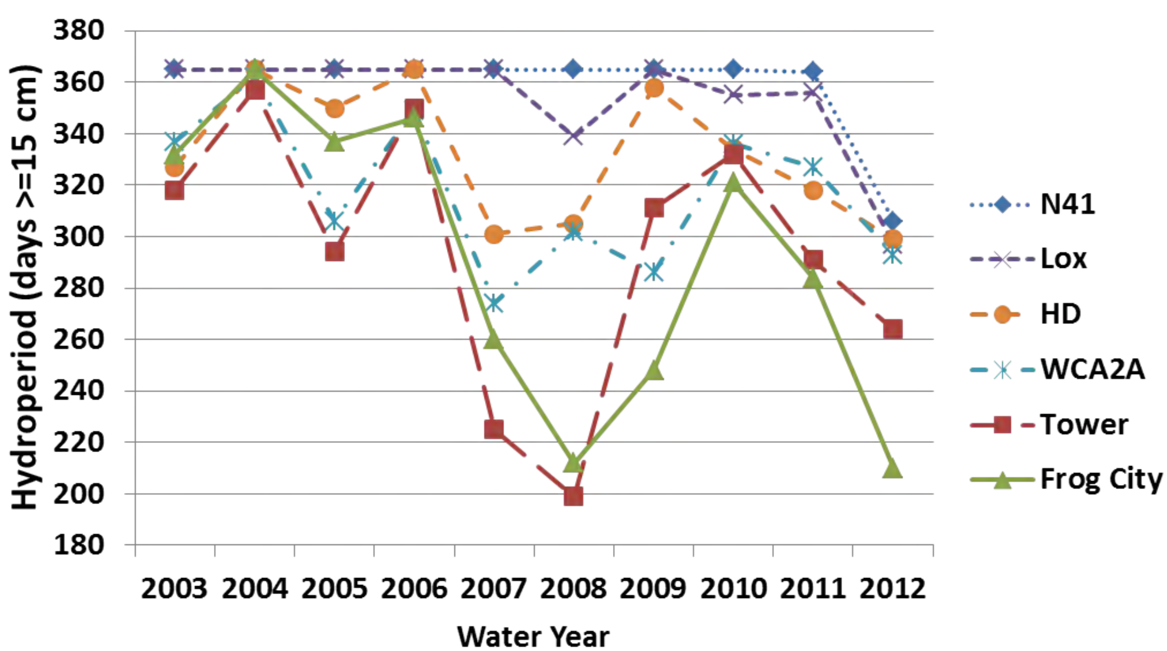

Figure 6-65. Yearly hydroperiod by water year for six areas of the Everglades where alligators are sampled (see Figure 6-64 for locations). Hydroperiods are calculated "through the eyes of an alligator" with 15 cm considered "dry."

Multiple regression analysis was used to examine trends in non-hatchling alligator abundance (also called total population). Trends from WY 2003–WY 2012 and in five-year increments were looked at starting in 2003. The model regresses log-transformed counts of alligators per km (dependent variable) on water year, season (fall and spring), transect, average water depth (AWD), and average water temperature (AWT):

$$\text{Log}((n+1)/\text{transect length})=\text{water year}+\text{season}+\text{transect}+\text{AWD}+\text{AWT} \quad (5)$$

Where n is the count of alligators, and AWD and AWT are the average of the values measured on the surveys.

Results

There were significant negative trends from WY 2003 to WY 2012 ($p < 0.05$) for WCA2A, Tower, and Frog City (the three shorter hydroperiod sites) and no significant trend for N41, HD, and Lox, the longer hydroperiod sites (**Figure 6-66** and **Table 6-16**).

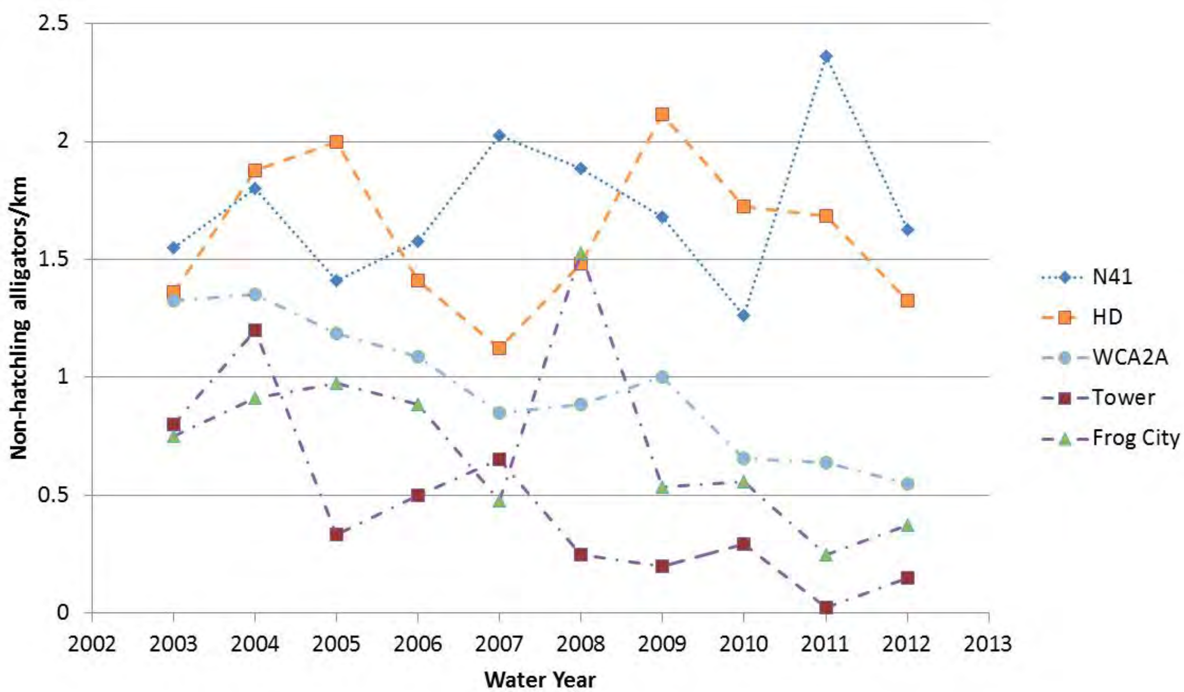

Figure 6-66. Average alligators/km of two spring and two fall surveys by water year for five areas where alligators are monitored. Lox is not included on the graph because densities are much higher (ranging from yearly averages of 4.0–8.2 alligators/km).

Periods of decline varied by area (**Table 6-16**). Negative trends for WCA2A were observed in three of four of the five-year increments from 2003 through 2010. Negative trends were observed for WCA3A-Tower in the five-year increments from 2006 through 2011 and negative trends were observed for Frog City in all five-year increments from 2006 through 2012. The WCA3A-HD site showed a negative trend in one five-year increment from 2004 through 2008.

Table 6-16. Modeled percent change per water year in trends in relative density of nonhatchling alligators. Empty cells indicate that a significant trend did not occur in that increment.

Water Year Increment	WCA3A-N41	Lox	WCA3A-HD	WCA2A	WCA3A-Tower	Frog City
2003–2012				-9.7%	-15.1%	-12.6%
2008–2012						-40.5%
2007–2011					-33.7%	-37.1%
2006–2010				-8.2%	-18.7%	-17.0%
2005–2009						
2004–2008			-14.2%	-13.2%		17.5
2003–2007				-7.9%		

Discussion

With the exception of the WCA 1 (Lox) route, all of the areas surveyed have alligator abundance below restoration targets > 1.7 alligators per km. The southern WCA 3A (N41) and the central WCA 3A (HD) routes fluctuate around the 1.7 alligators per km with yearly averages ranging from 1.2 to 2.4 alligators per km. These three areas (Lox, N41, and HD) are also the three areas that have not experienced declines since 2003, have the longest hydroperiods, and experience less frequent and intense dry downs.

In contrast, in the other three areas (WCA 2A, northern WCA 3A [Tower], and Northeastern SRS in ENP [Frog City]), alligator densities were low and have declined since 2003. These areas have shorter hydroperiods and experience more frequent and intense dry downs.

These data support the hypotheses that multi-year hydroperiods are important for maintaining alligator populations in the Everglades and are consistent with the hypothesis that dry downs on average of once every five years would be optimal. Repeated and intense dry downs affect both the ability of alligators to reproduce if they occur during April–May and the survival of hatchling and juvenile alligators regardless of when they occur. Based on the information above, areas that experience dry downs that last longer than two months (60 days) or repeatedly occur at intervals more frequently than once every five years are not likely to support populations of alligators that are at or approaching restoration targets.

It is fortunate that there is at least ten years of data over various hydrologic conditions that allow a comprehensive review of patterns of change and better definition of appropriate hydrologic targets. Because the hypothesis is that dry down events on average of once every five years is optimal, a minimum of five years of data ae needed before and after the event to test this. In addition, long-term data give a big picture perspective on system responses and allow temporal lags to be incorporated into ecosystem responses. Some monitoring attributes will respond within months, such as alligator body condition and relative density. Others, such as growth and survival, are measured over years. Numbers of nests and clutch size respond over decades. Only long-term monitoring programs will tell if increases in body condition and relative density today will lead to increased reproduction in ten years.

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APPENDIX 6-1: EVERGLADES DEPTH ESTIMATION NETWORK WATER LEVEL RESULTS FOR FOUR TIME PERIODS COMPARED TO THE NATURAL SYSTEM MODEL WATER LEVEL ESTIMATES

Most of the figures in this appendix compare measured water levels from the Everglades Depth Estimation Network (EDEN) and water levels estimated by the Natural System Model (NSM) at various gaging stations in each zone depicted in **Figure A6-1-1** and for Water Conservation Area (WCA) 1, WCA 2A and WCA 2B. The four discrete time periods of measured water levels are (1) Water Year (WY) 2000–WY 2008 (2009 System Status Report [SSR] period); (2) WY 2009–WY 2013 (2014 SSR period); (3) Experimental Program (1992–1999); and (4) Interim Operations Period (2000–2012). A water year begins on May 1 and ends on April 30 of the following year. Period 1 and 2 are compared to identify changes in hydrology trends reported in the 2009 and 2014 SSRs. Period 3 and 4 compare two different operations periods with the future intent of adding rainfall analysis to determine what degree operations versus climate might be causing change in the system.

Gray ribbons in the remaining figures (**Figures A6-1-2** through **A6-1-53**) within this appendix represent the mid-range distribution (25^{th} to 75^{th} percentile) of water level from NSM (1965–2005). The light orange ribbon represents the mid-range distribution (25^{th} to 75^{th} percentile) for each time period of measured water levels. The dark orange ribbon represents the overlap area of the observed data and the NSM restoration benchmark. The green dashed line represents the average ground elevation at the gage. The left-hand y-axis in these figures shows water level elevation in feet North American Vertical Datum of 1988 (NAVD88). the right hand y-axis shows water level elevation in feet National Geodetic Vertical Datum (NGVD29).

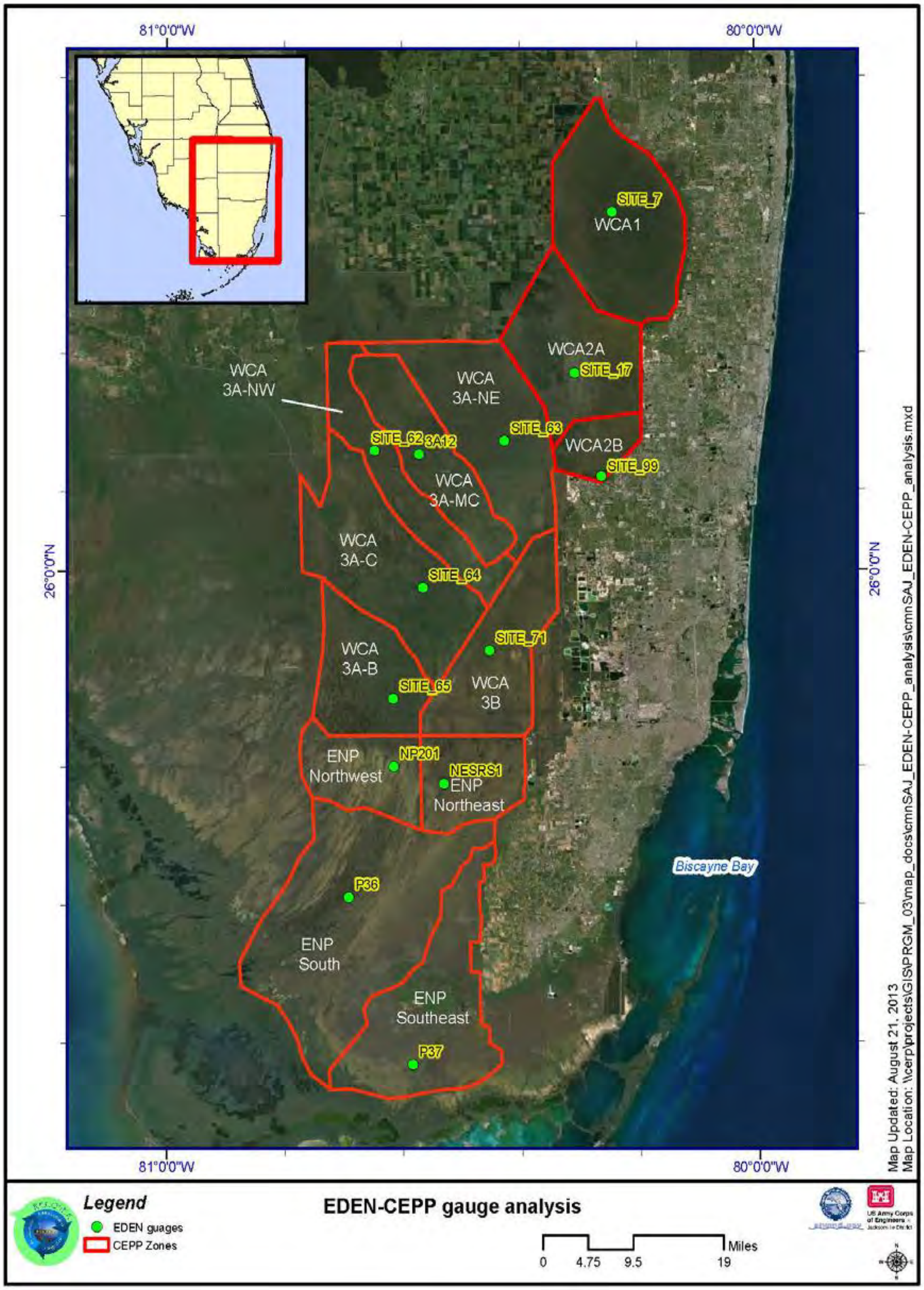

Figure A6-1-1. Map of EDEN hydrologic analysis zones in WCA 3 and Everglades National Park (ENP) that matches Central Everglades Planning Project (CEPP) analysis zones, except that ENP-N is split into ENP Northeast (NE) and Northwest (NW).

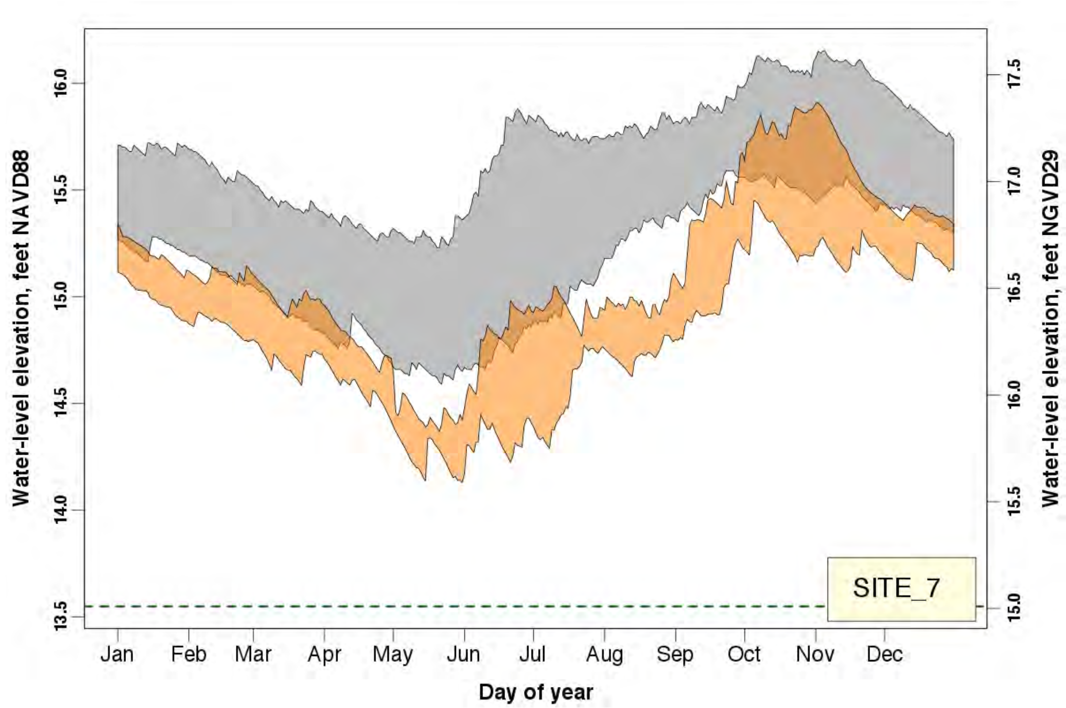

Figure A6-1-2. WCA 1 – Site 7 gage for WY 2000–2008 (2009 SSR period).

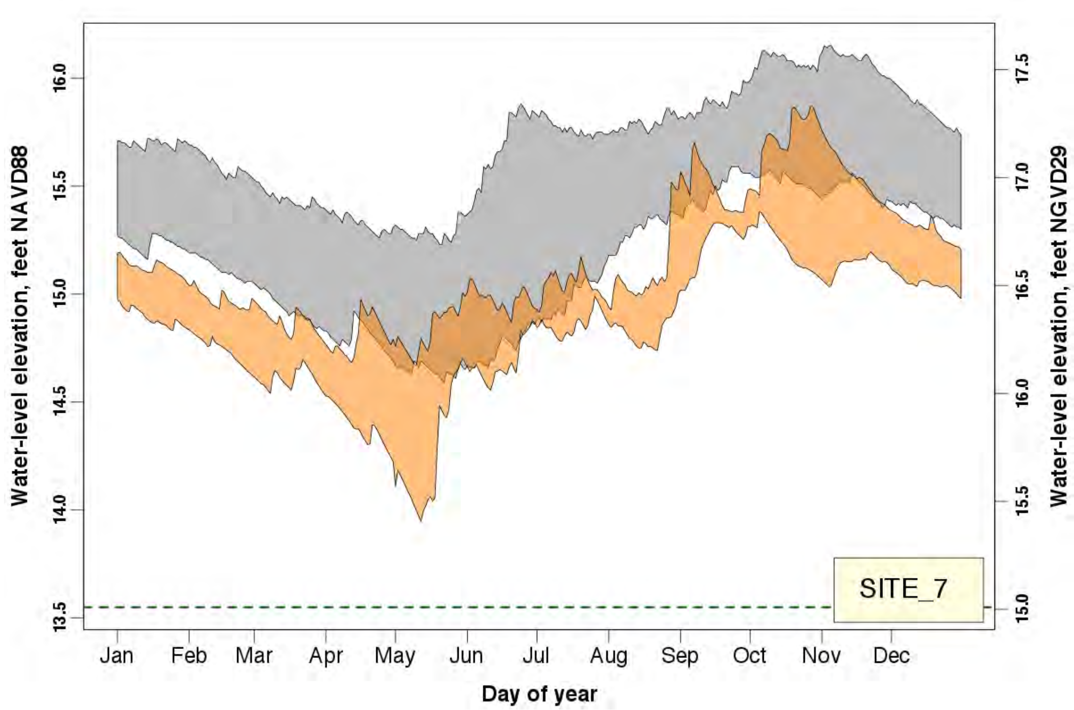

Figure A6-1-3. WCA 1 – Site 7 gage for WY 2009–WY 2013 (2014 SSR period).

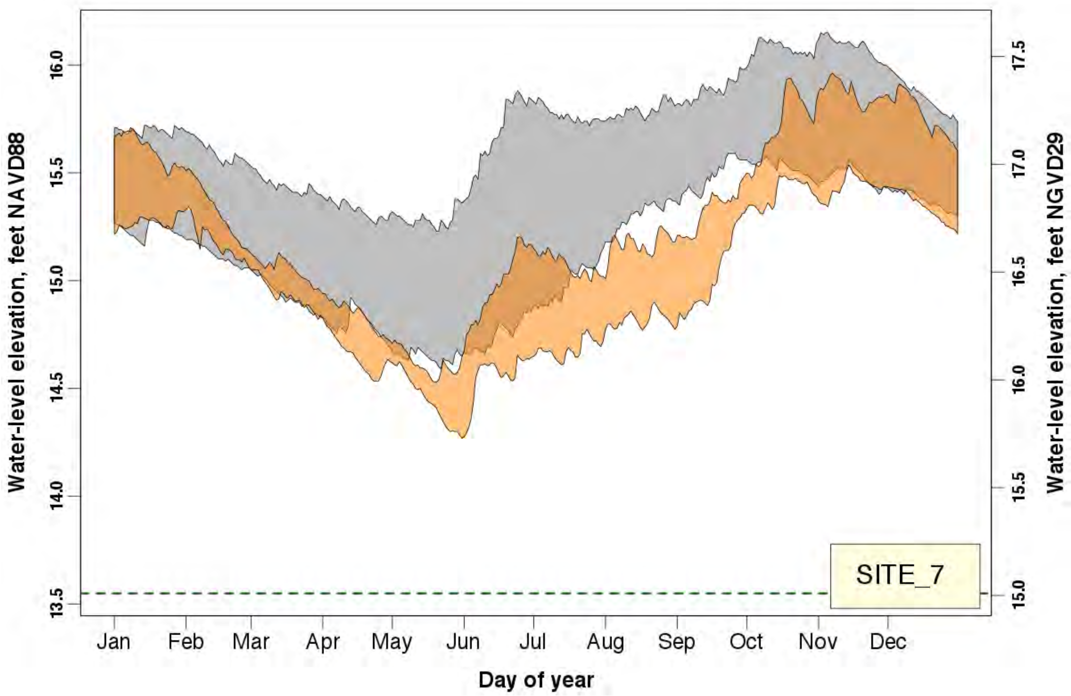

Figure A6-1-4. WCA 1 – Site 7 gage for the Experimental Program (1992–1999).

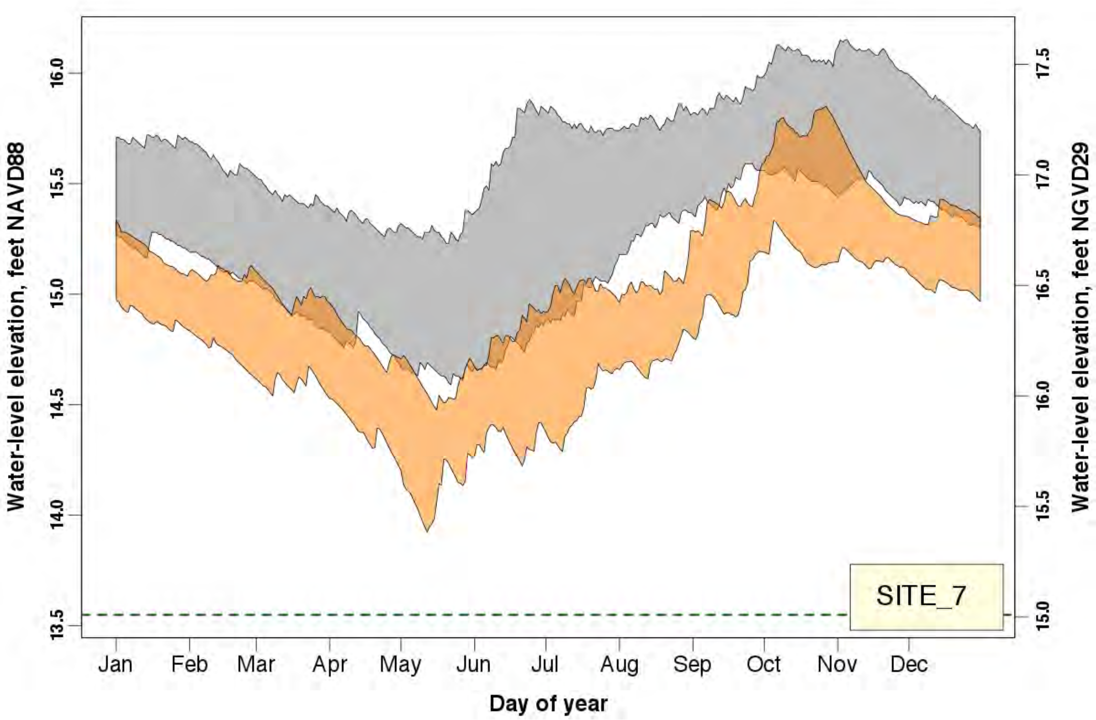

Figure A6-1-5. WCA 1 – Site 7 gage for the Interim Operations Period (2000–2012).

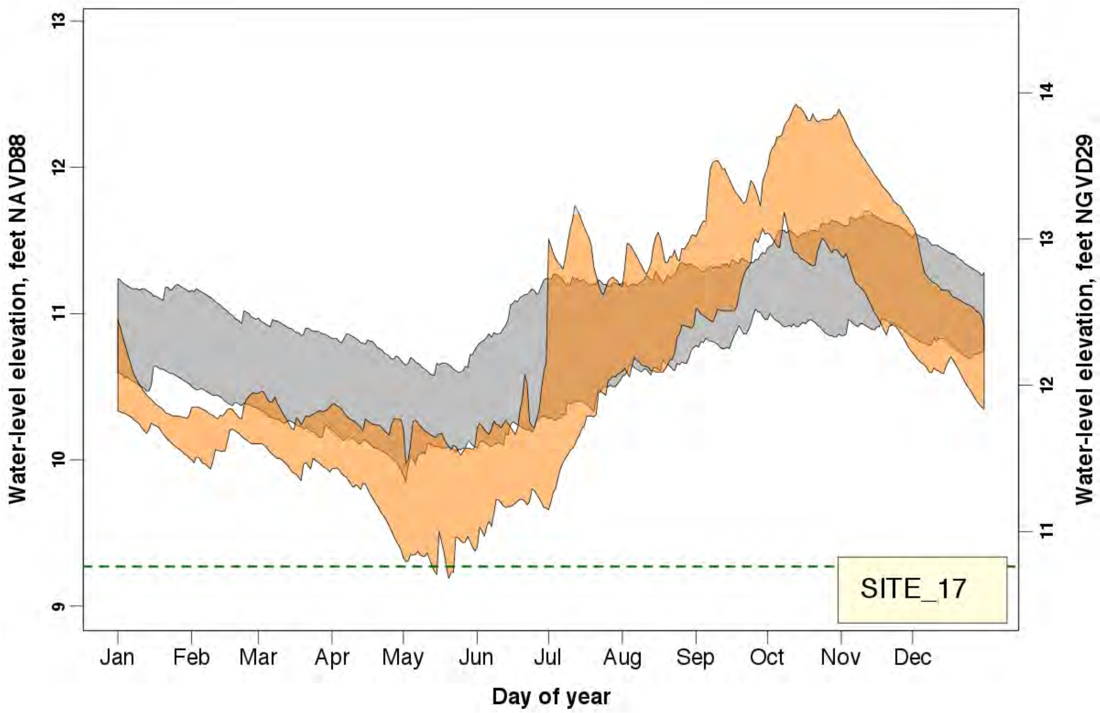

Figure A6-1-6. WCA 2A – Site 17 gage for WY2000–WY2008 (2009 SSR period).

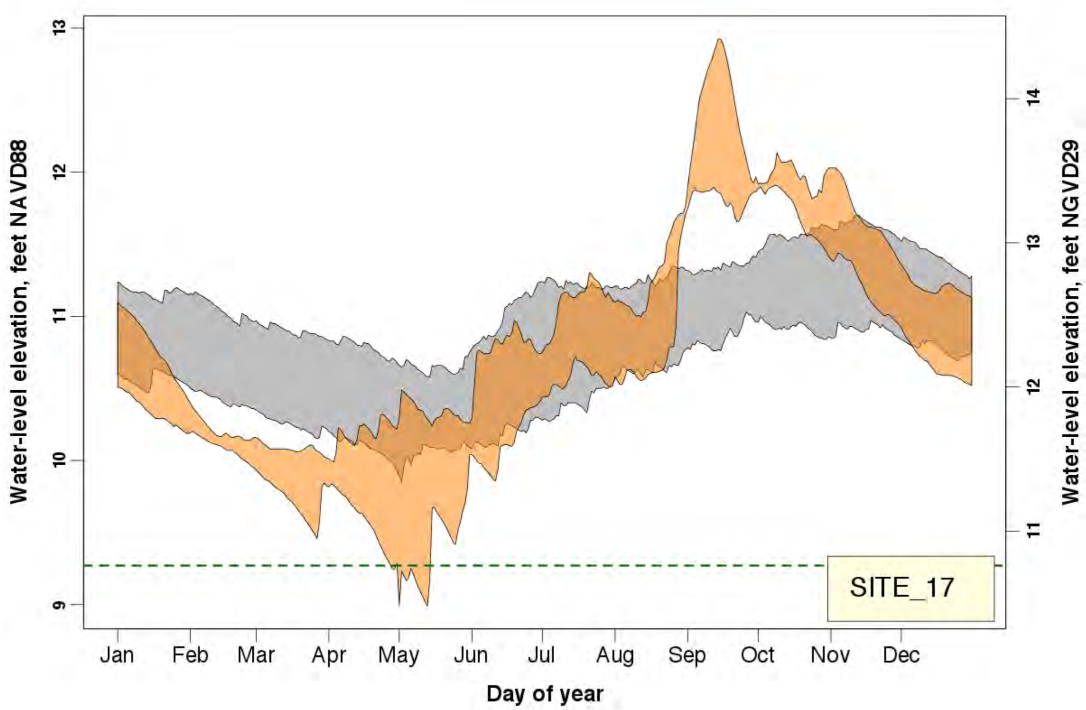

Figure A6-1-7. WCA 2A – Site 17 gage for WY2009–WY2013 (2014 SSR period).

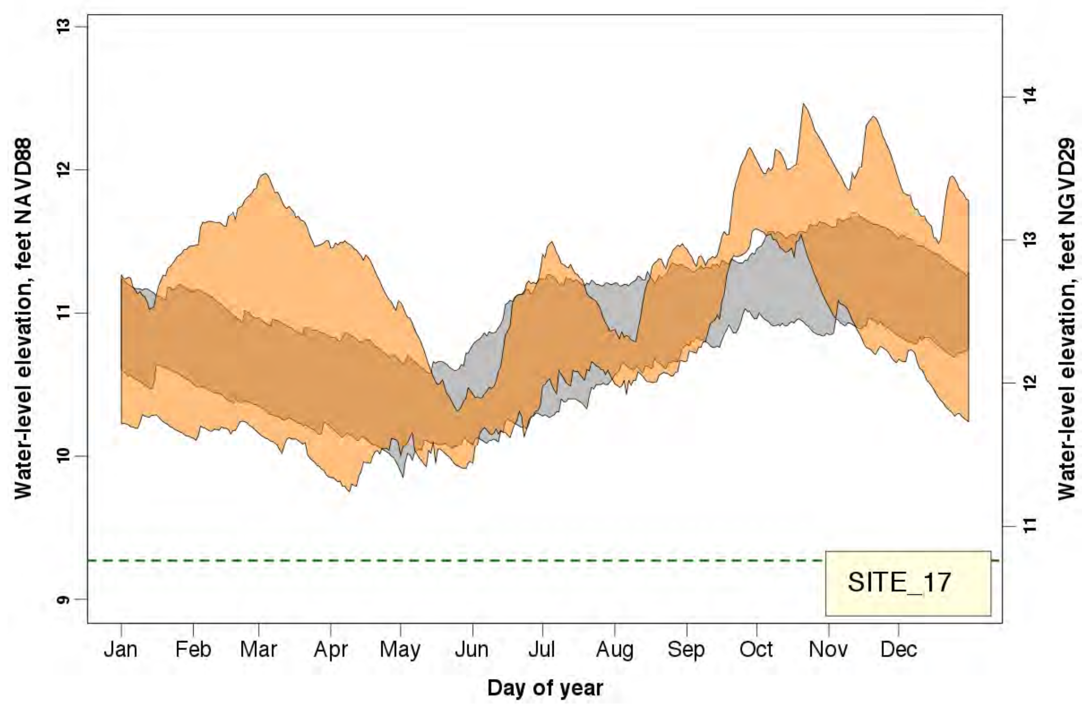

Figure A6-1-8. WCA 2A – Site 17 gage for Experimental Program (1992–1999).

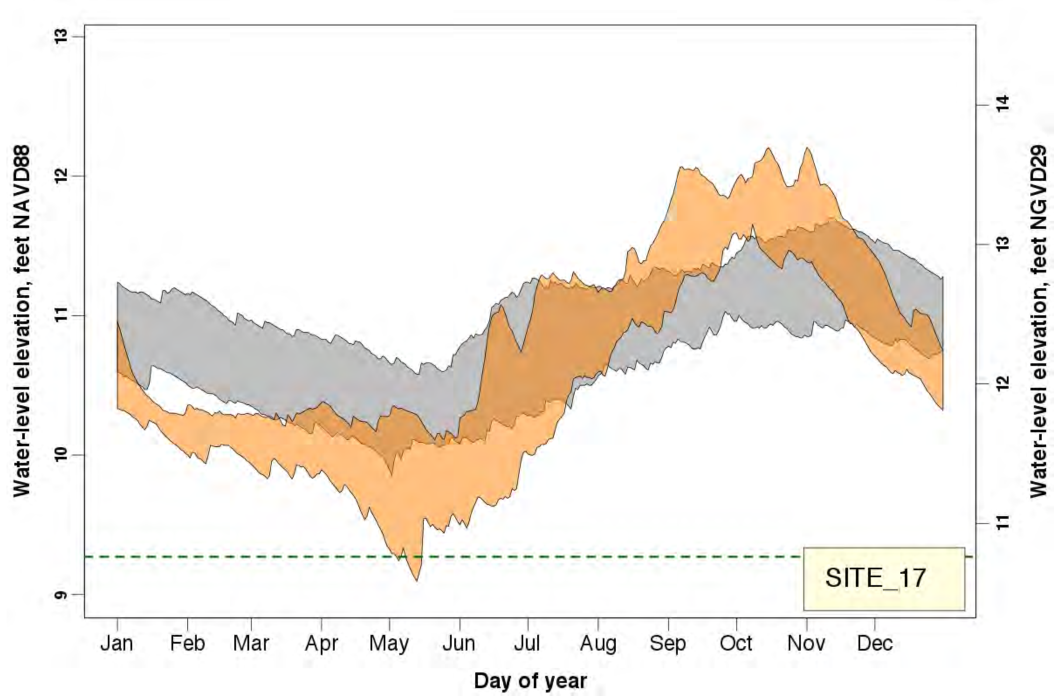

Figure A6-1-9. WCA 2A – Site 17 gage for Interim Operations Period (2000–2012).

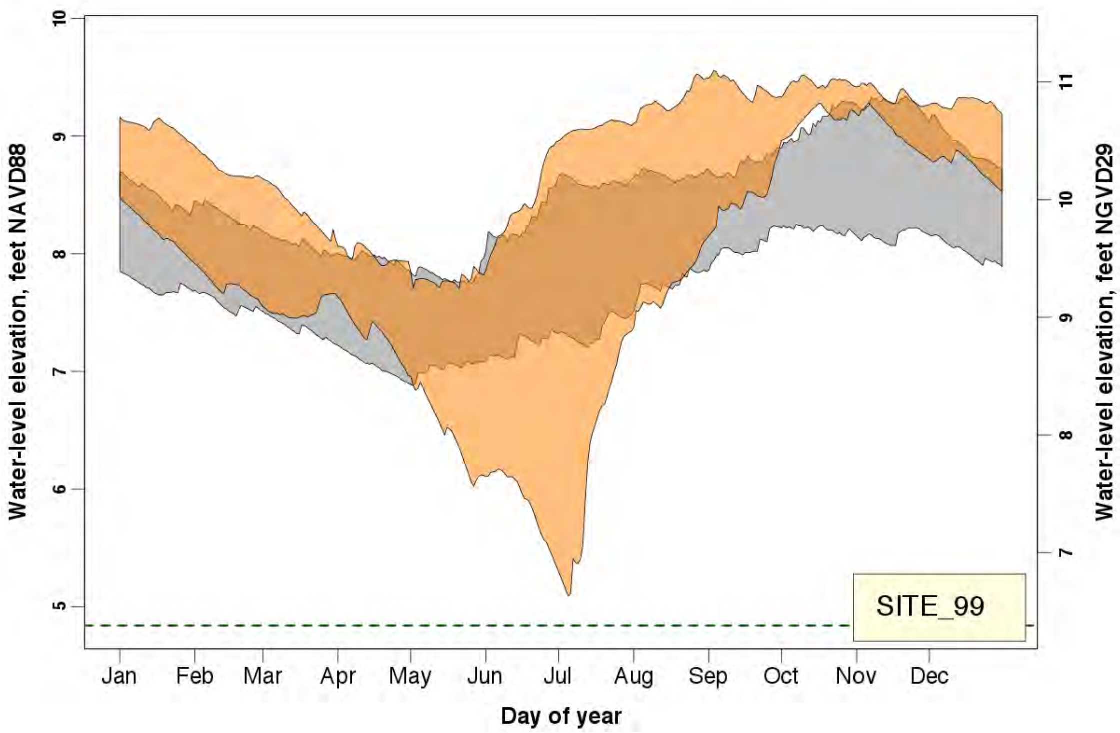

Figure A6-1-10. WCA 2B – Site 99 gage for WY 2000–WY 2008 (2009 SSR period).

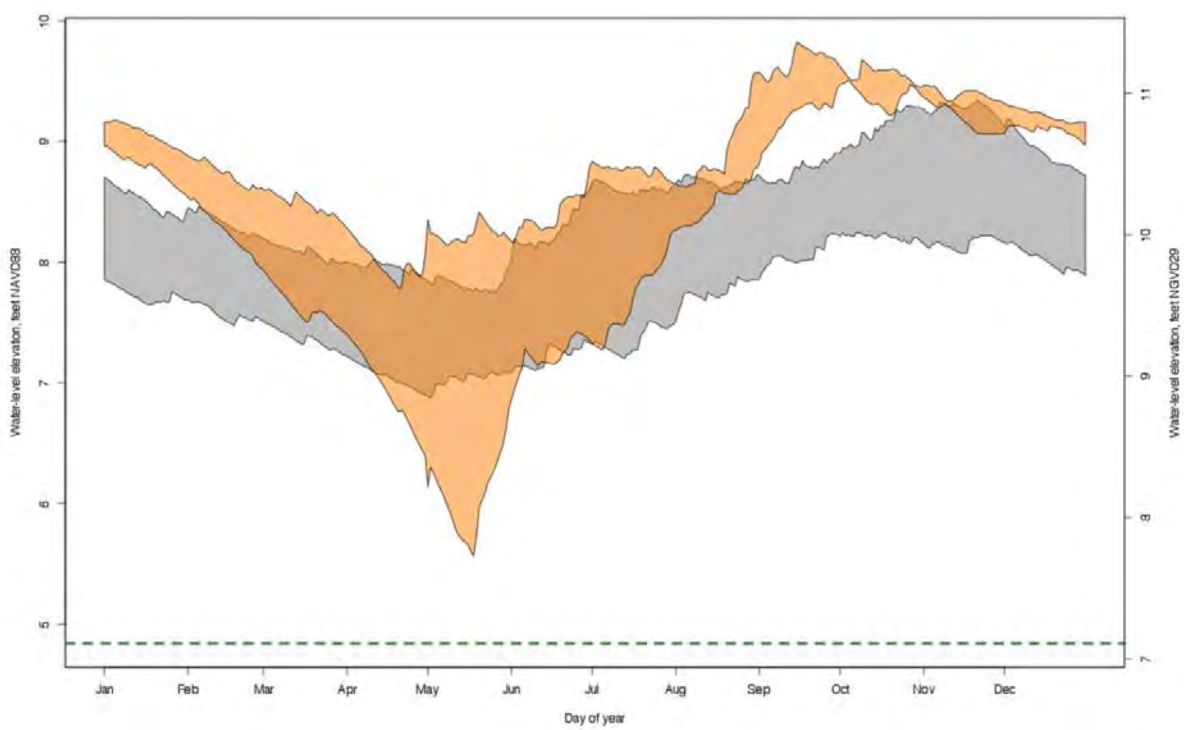

Figure A6-1-11. WCA 2B – Site 99 for WY 2009–WY 2013 (2014 SSR period).

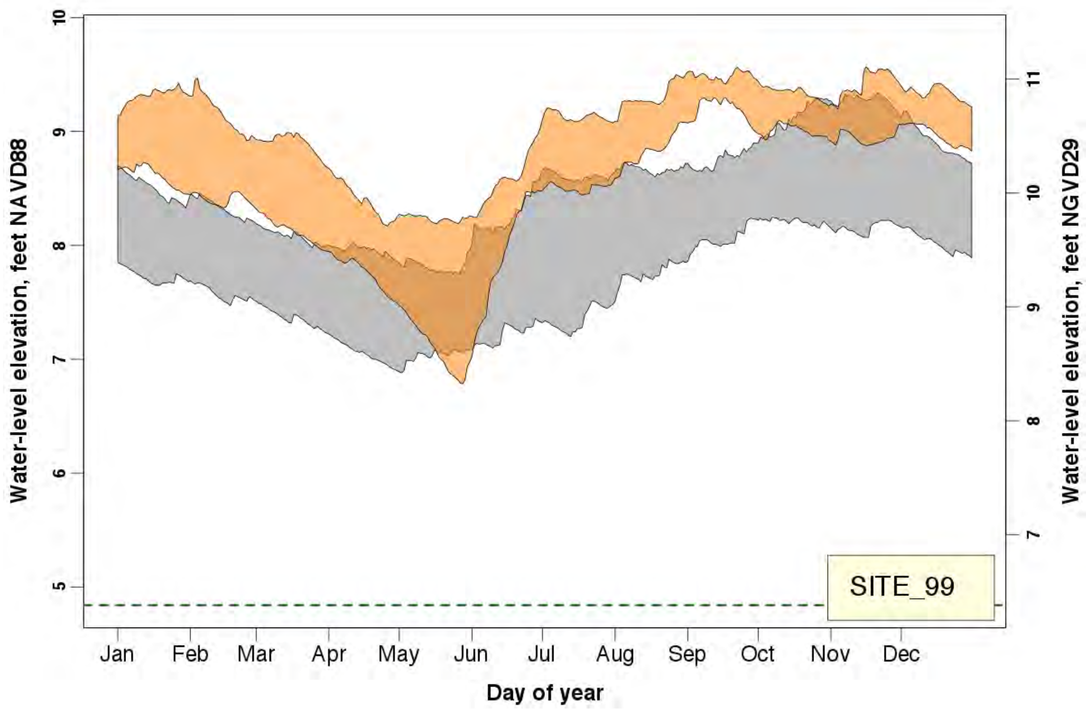

Figure A6-1-12. WCA 2B – Site 99 gage for Experimental Program (1992–1999).

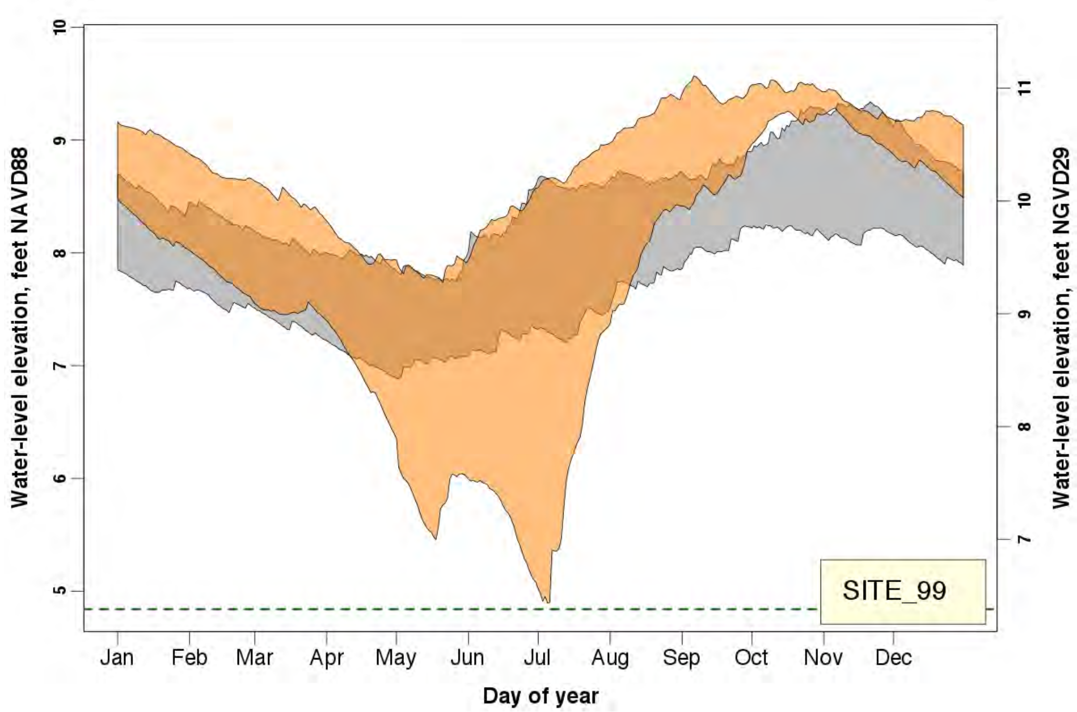

Figure A6-1-13. WCA 2B – Site 99 gage for Interim Operations Period (2000–2012).

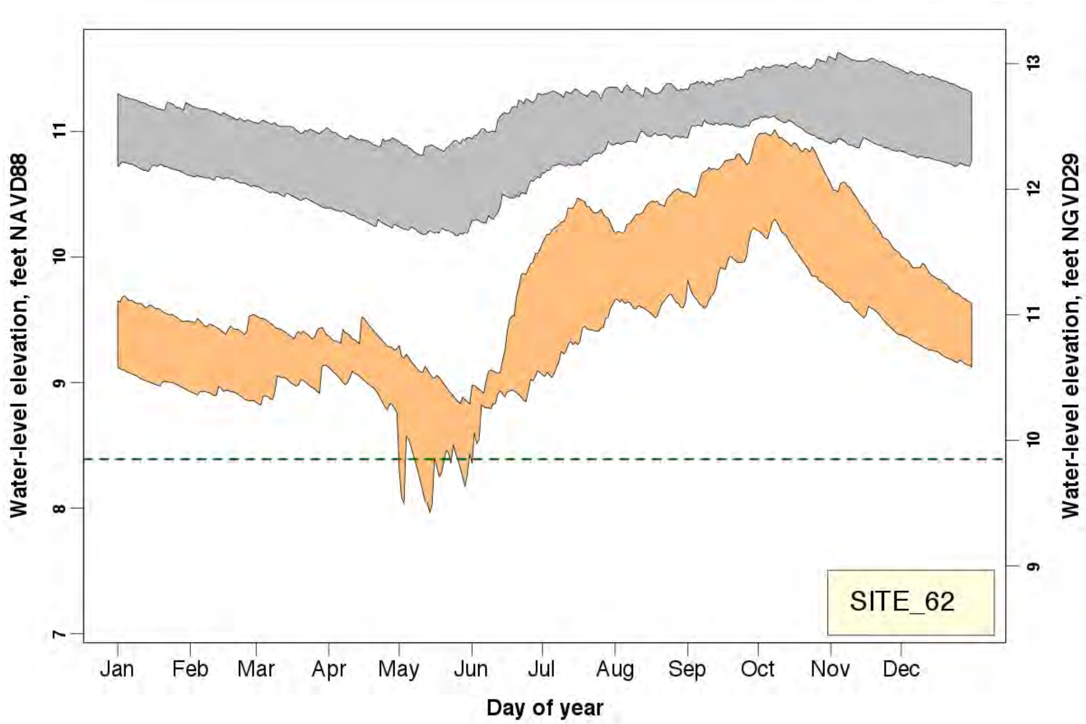

Figure A6-1-14. WCA 3A NW – Site 62 gage for WY 2000–WY 2008 (2009 SSR period).

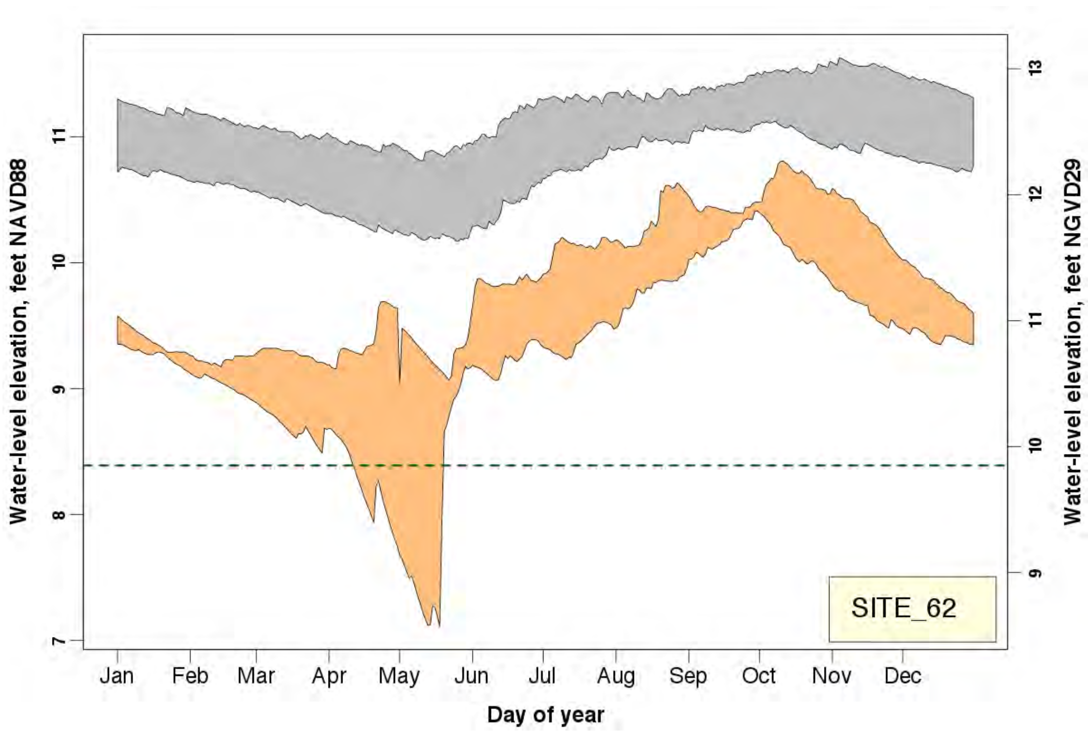

Figure A6-1-15. WCA 3A NW – Site 62 gage for WY 2009–WY 2013 (2014 SSR period).

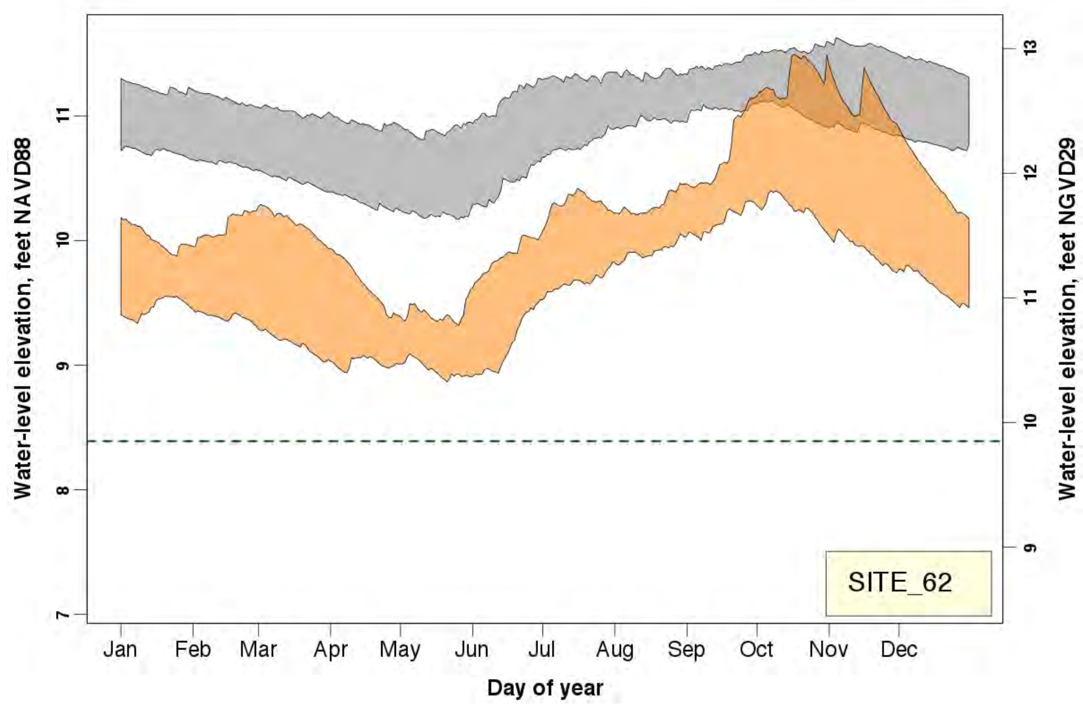

Figure A6-1-16. WCA 3A NW – site 62 gage for the Experimental Period (1992–1999).

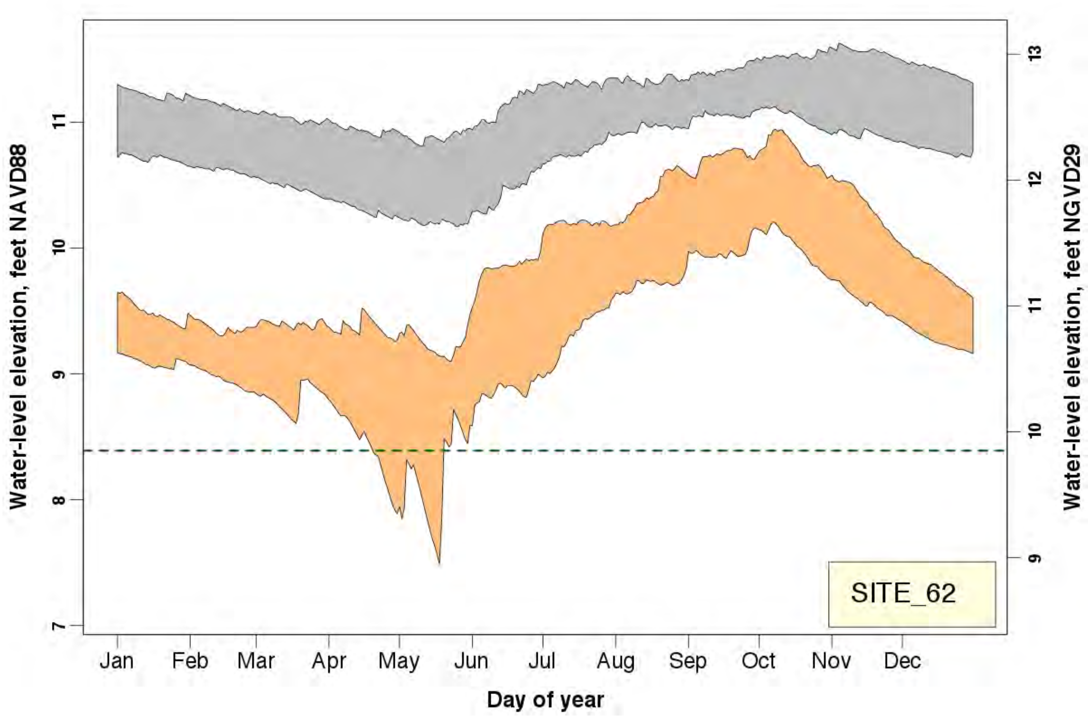

Figure A6-1-17. WCA 3A NW – site 62 gage for the Interim Operations Period (2000–2012).

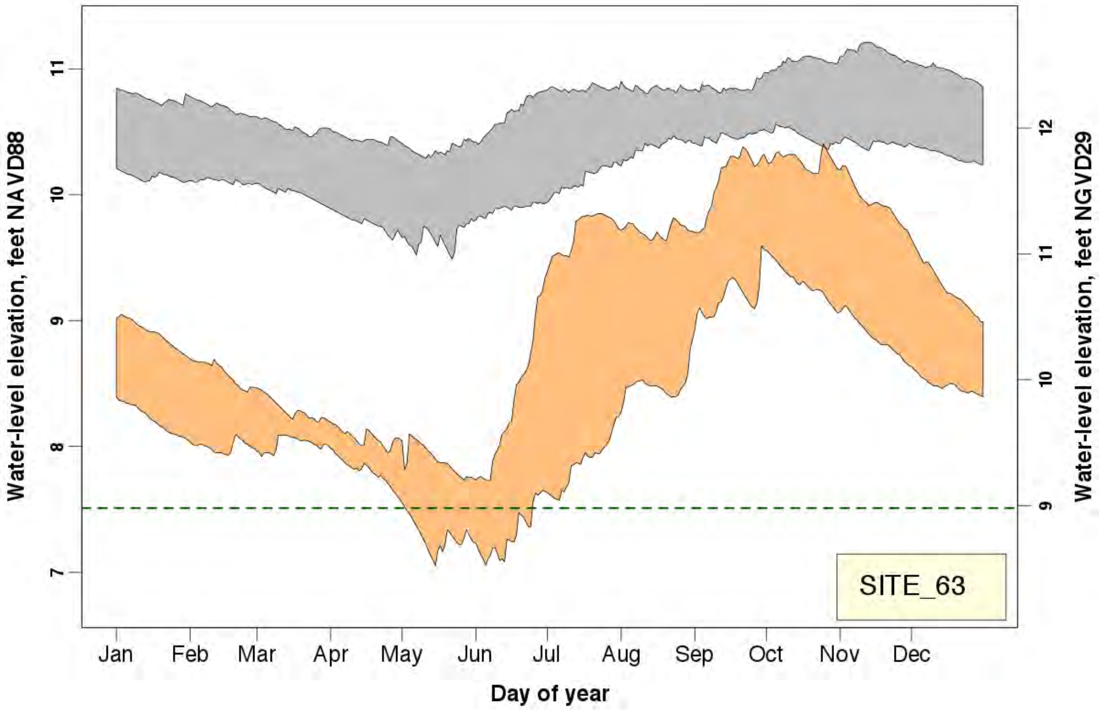

Figure A6-1-18. WCA 3A NE – Site 63 gage for the WY 2000–WY 2008 (2009 SSR period).

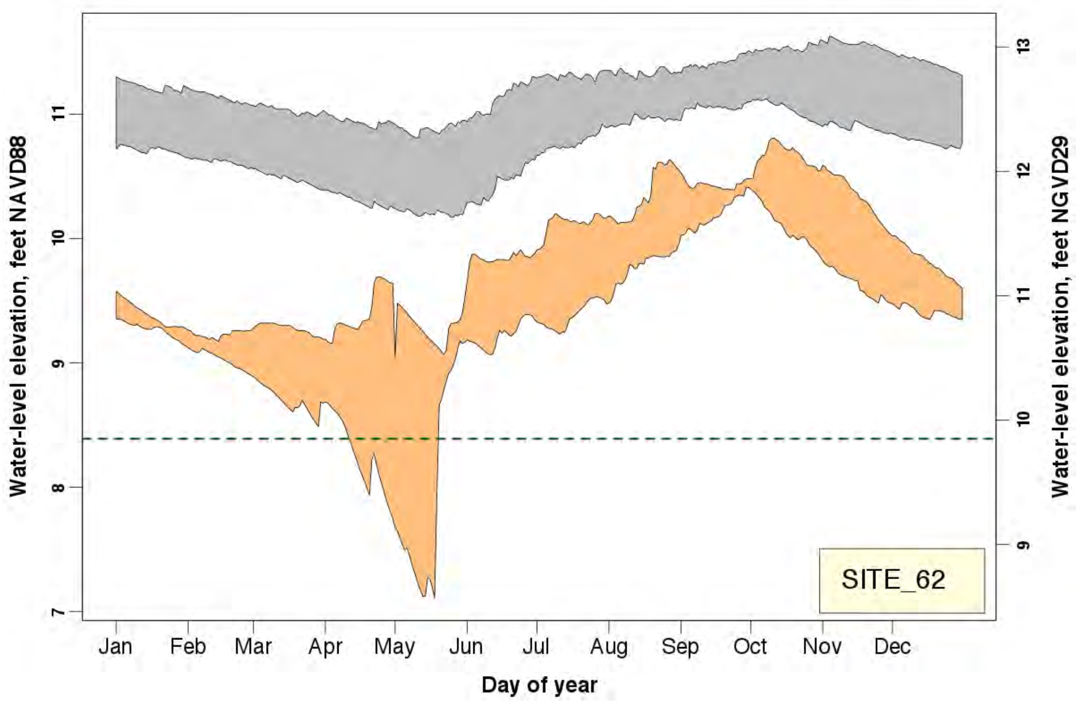

Figure A6-1-19. WCA 3A NE – Site 63 gage for the WY 2009–WY 2013 (2014 SSR period).

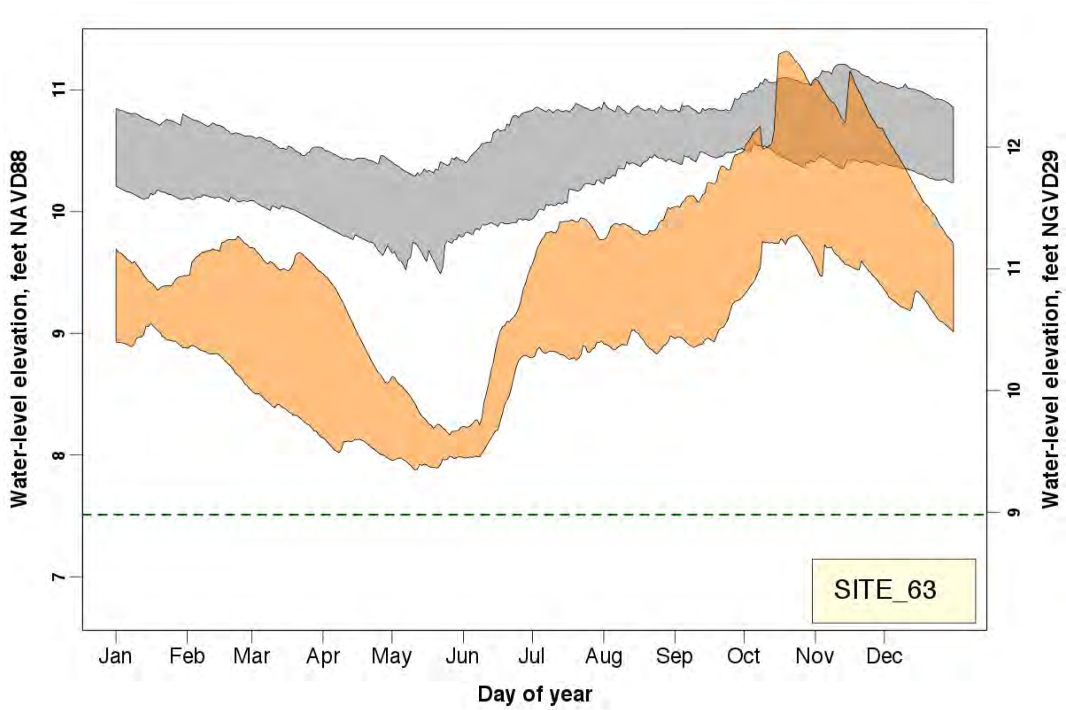

Figure A6-1-20. WCA 3A NE – Site 63 gage for the Experimental Program (1992–1999).

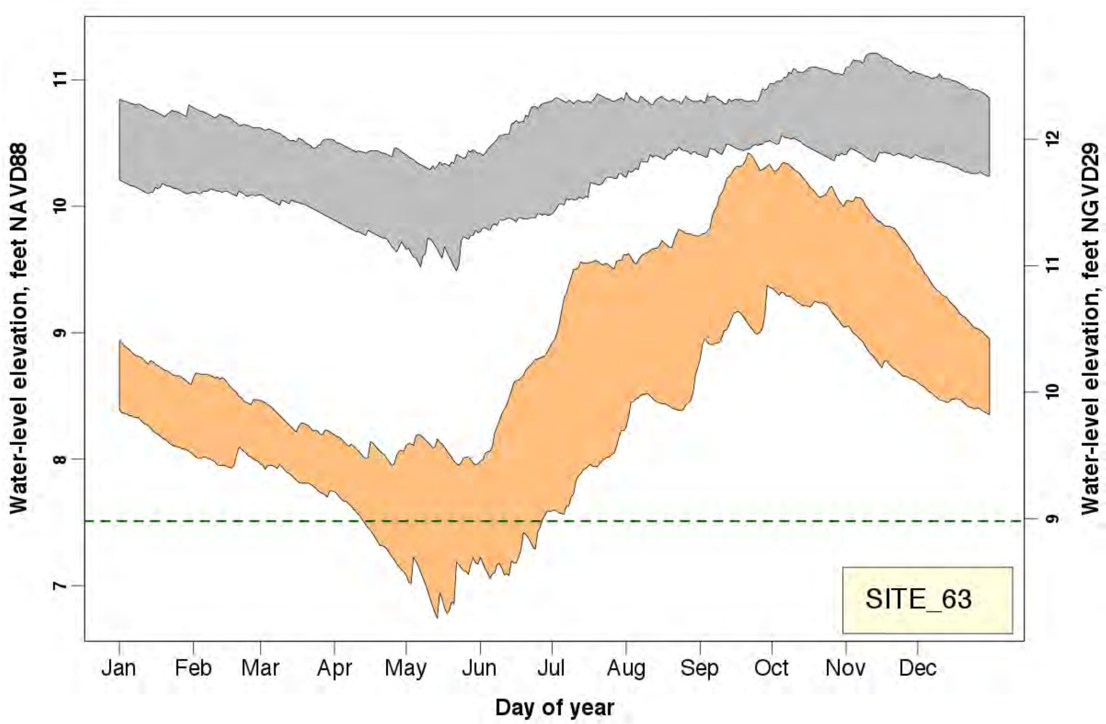

Figure A6-1-21. WCA 3A NE – Site 63 gage for the Interim Operations Period (2000–2012).

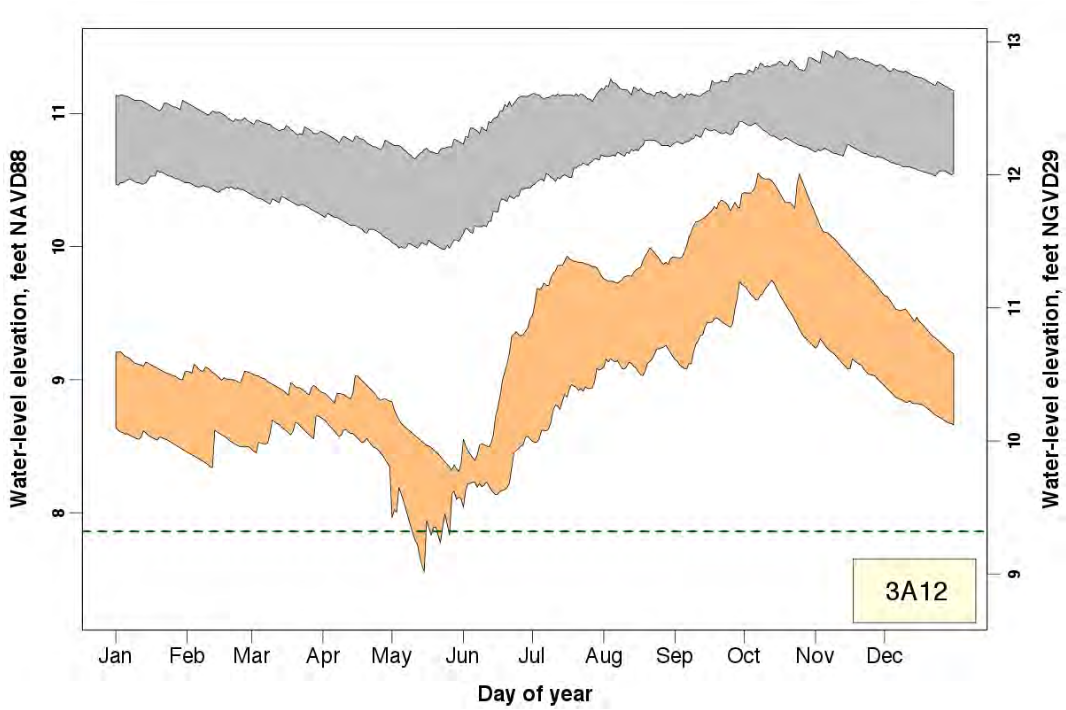

Figure A6-1-22. WCA 3A MC – Site 3A12 gage for WY 2000–WY 2008 (2009 SSR period).

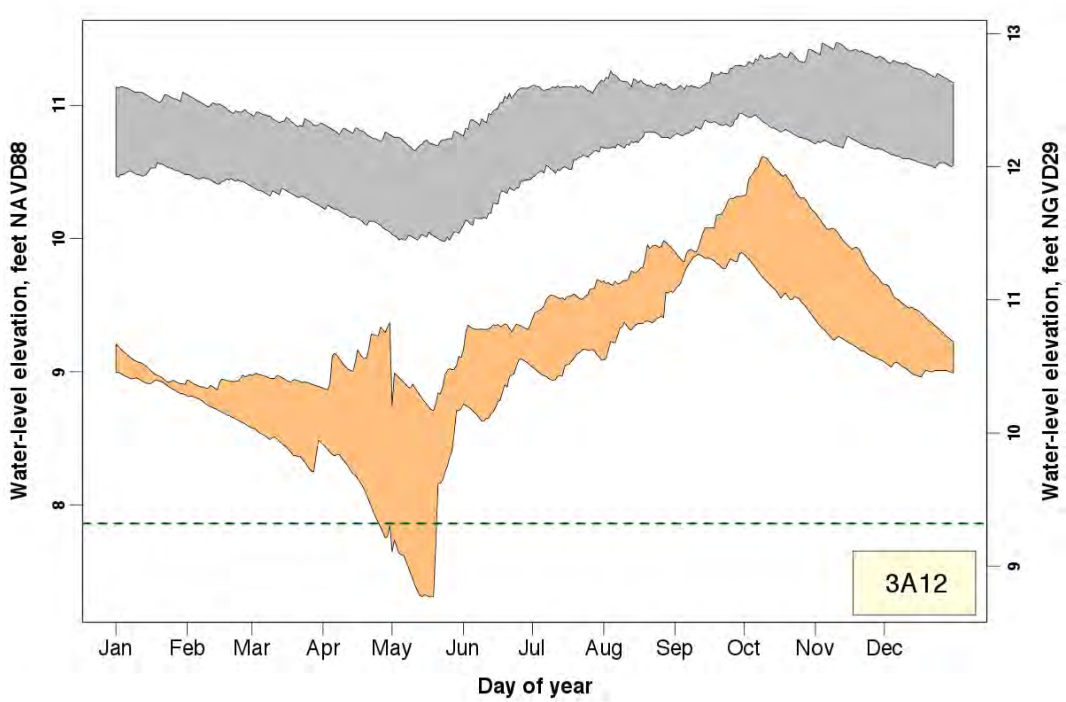

Figure A6-1-23. WCA 3A MC – Site 3A12 gage for WY 2009–WY 2013 (2014 SSR period).

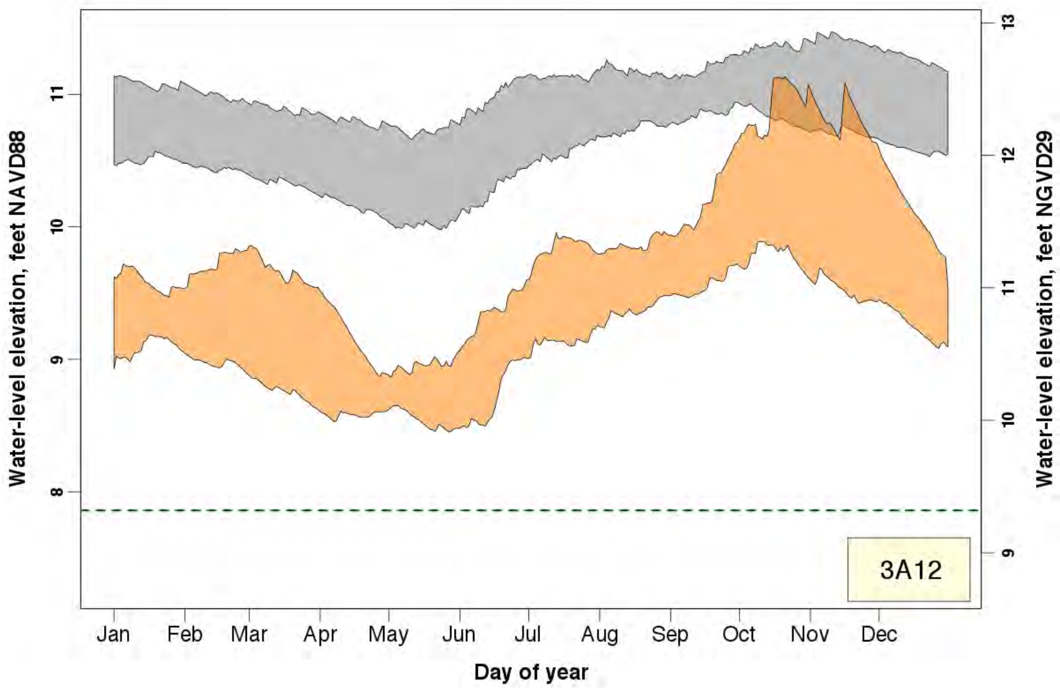

Figure A6-1-24. WCA 3A MC – Site 3A12 gage for the Experimental Program (1992–1999).

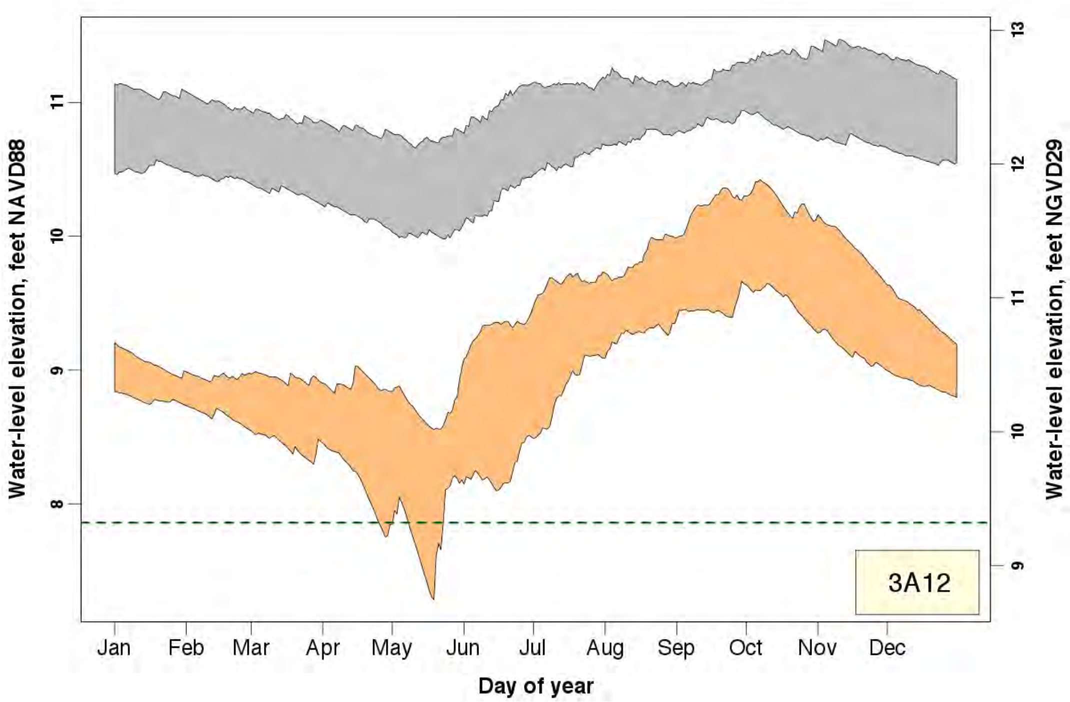

Figure A6-1-25. WCA 3A MC – Site 3A12 gage Interim Operations Period (2000–2012).

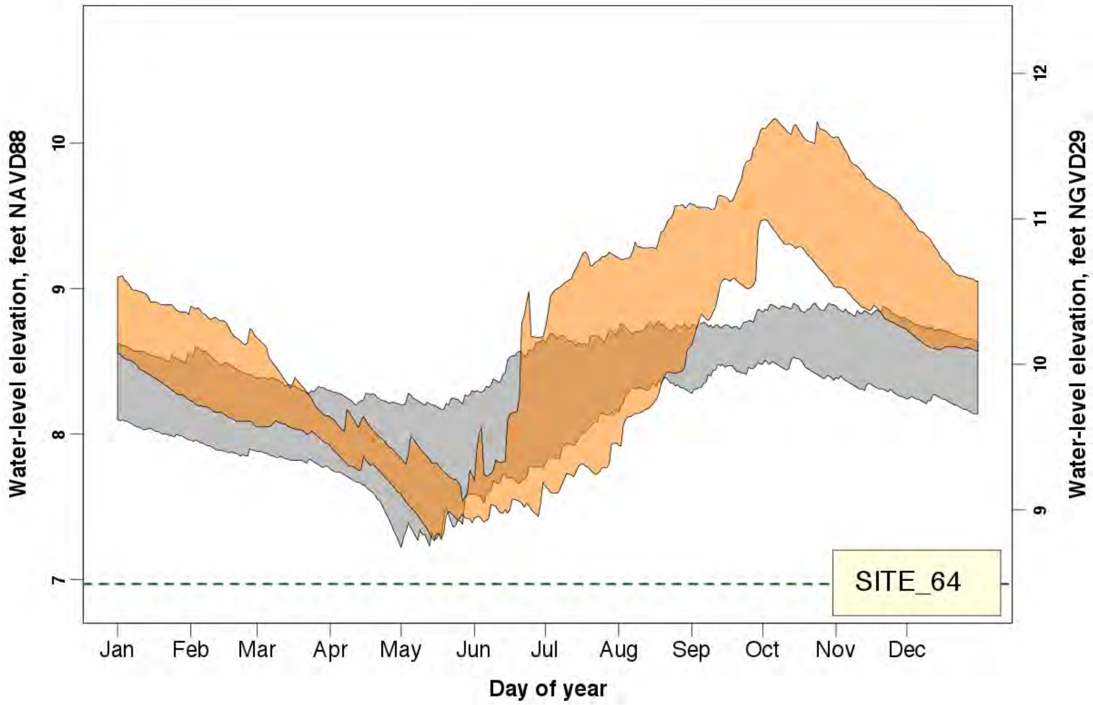

Figure A6-1-26. WCA 3A C – Site 64 gage for WY 2000–WY 2008 (2009 SSR period).

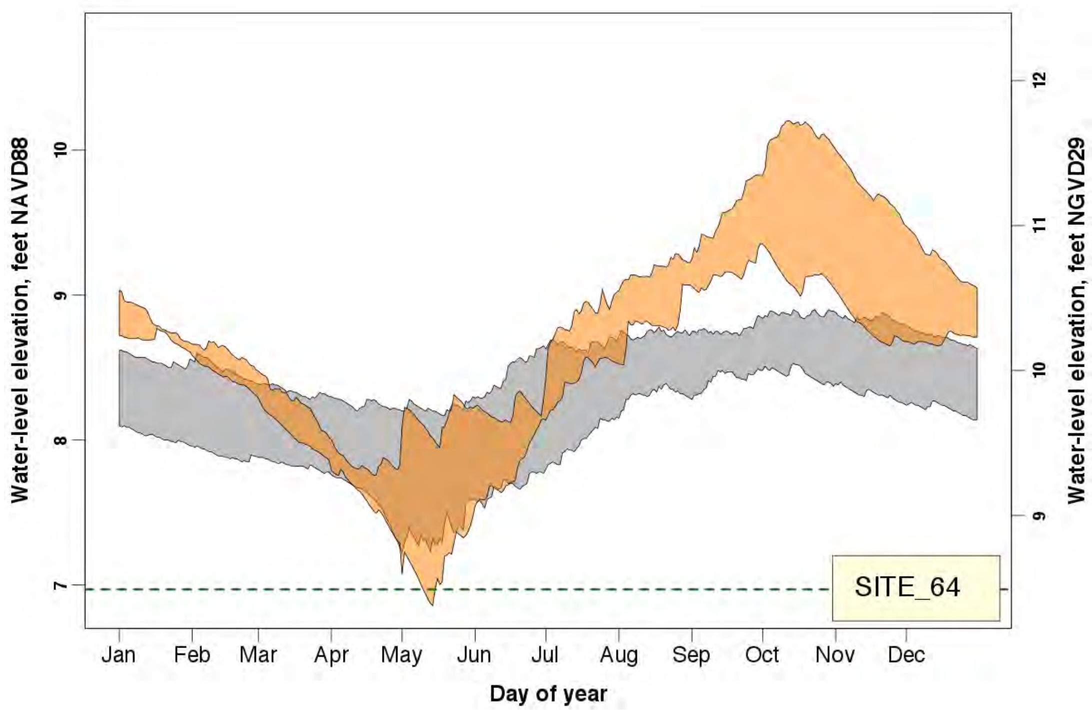

Figure A6-1-27. WCA 3A C – Site 64 gage for WY 2009–WY 2013 (2014 SSR period).

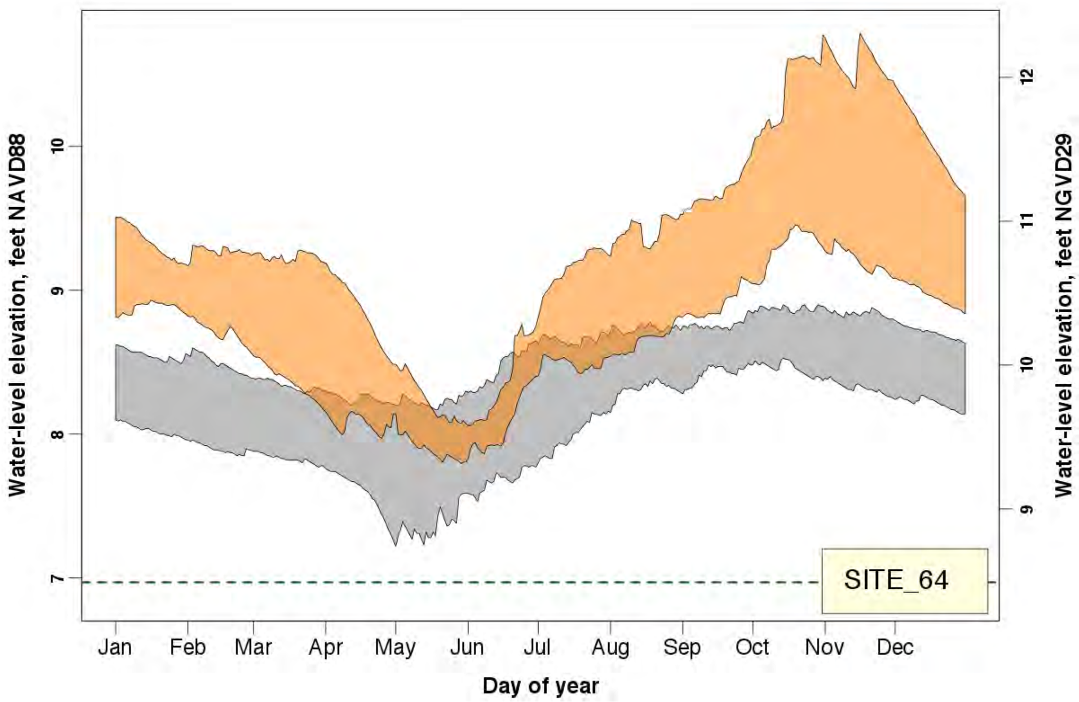

Figure A6-1-28. WCA 3A C – Site 64 gage for Experimental Program (1992–1999).

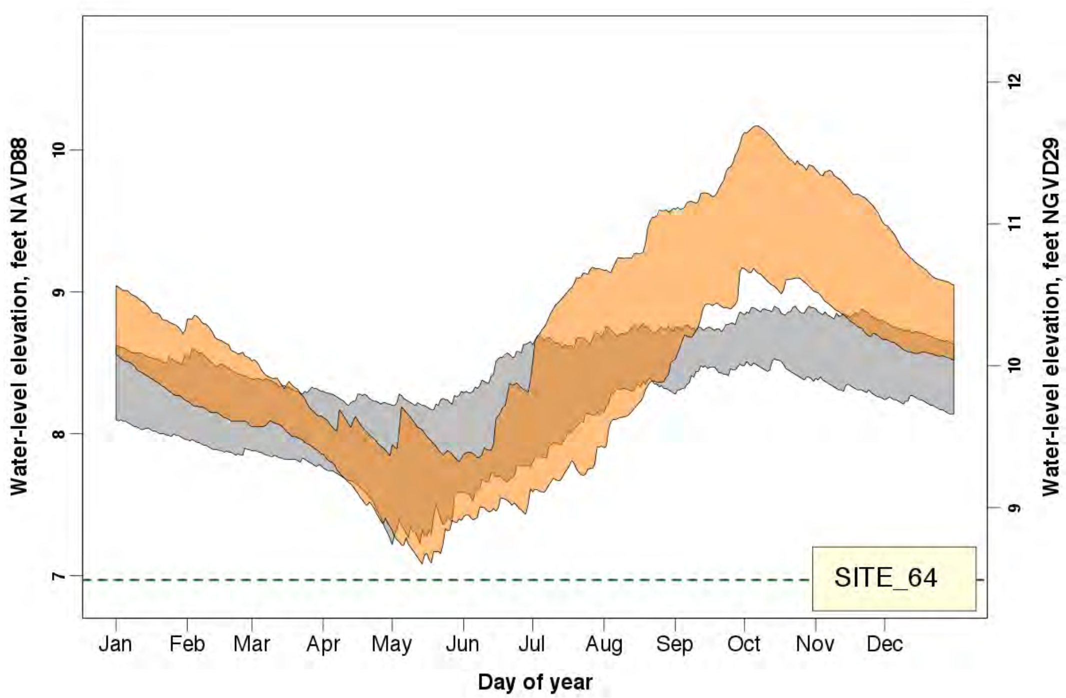

Figure A6-1-29. WCA 3A C – Site 64 for Interim Operations Period (2000–2012).

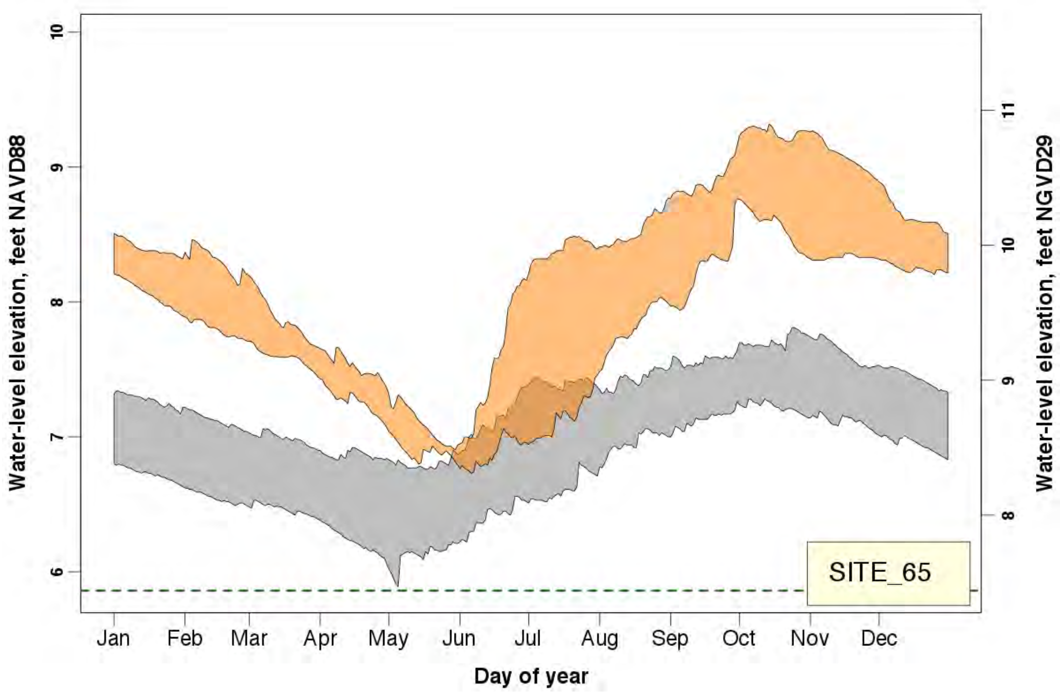

Figure A6-1-30. WCA 3A SW – Site 65 gage for WY 2000–WY 2008 (2009 SSR period).

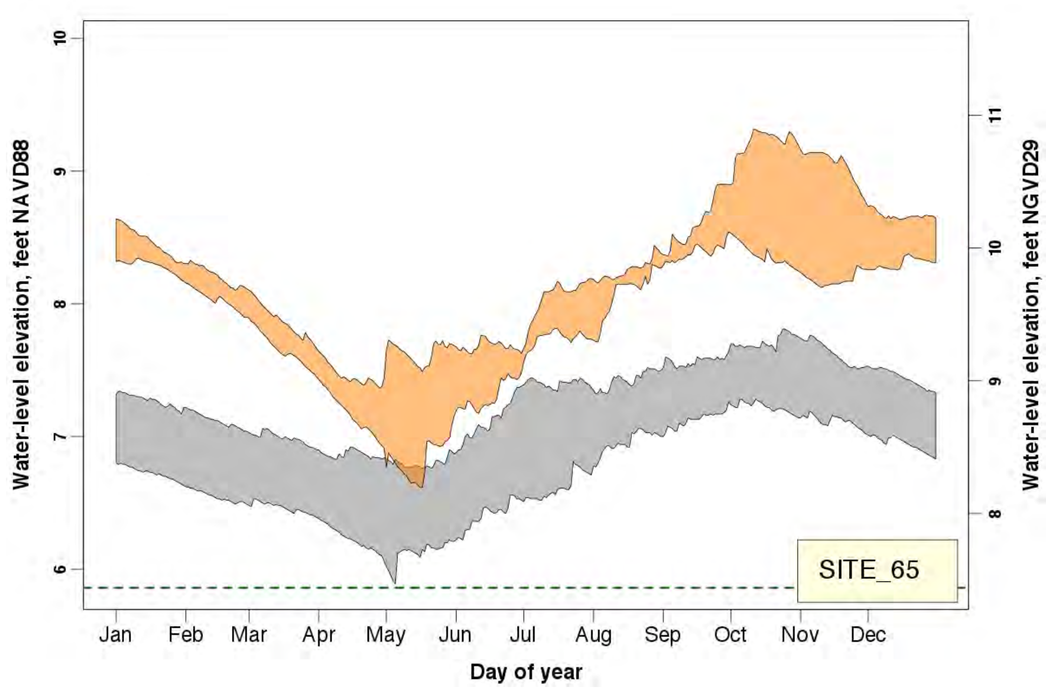

Figure A6-1-31. WCA 3A SW – Site 65 gage for WY 2009–WY 2013 (2014 SSR period).

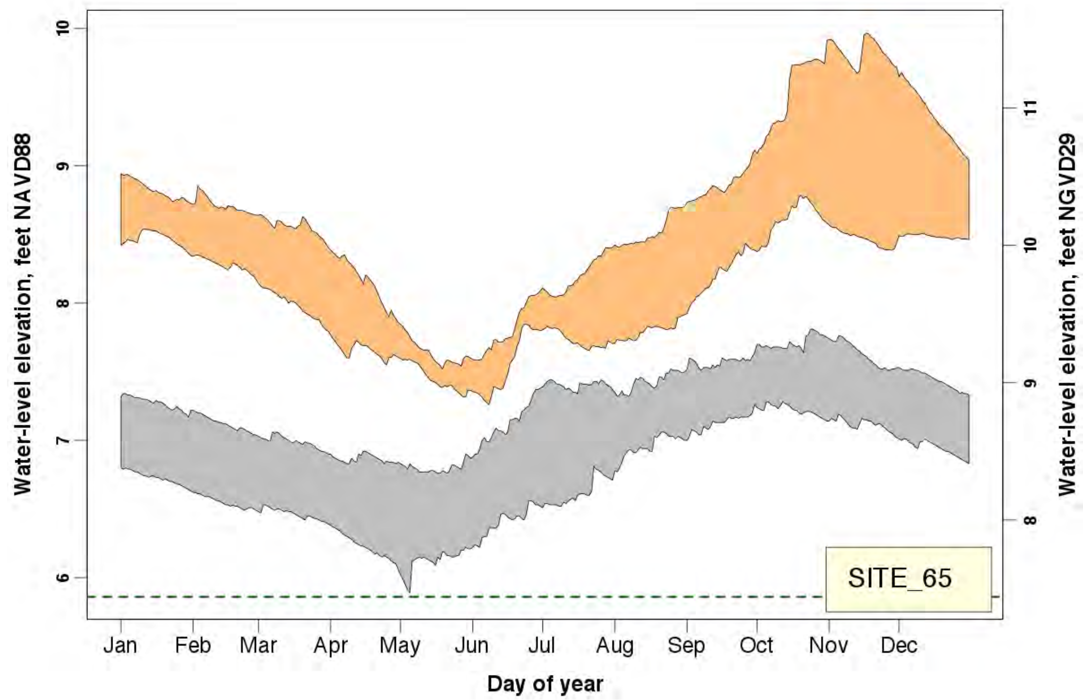

Figure A6-1-32. WCA 3A SW – Site 65 gage for Experimental Program (1992–1999).

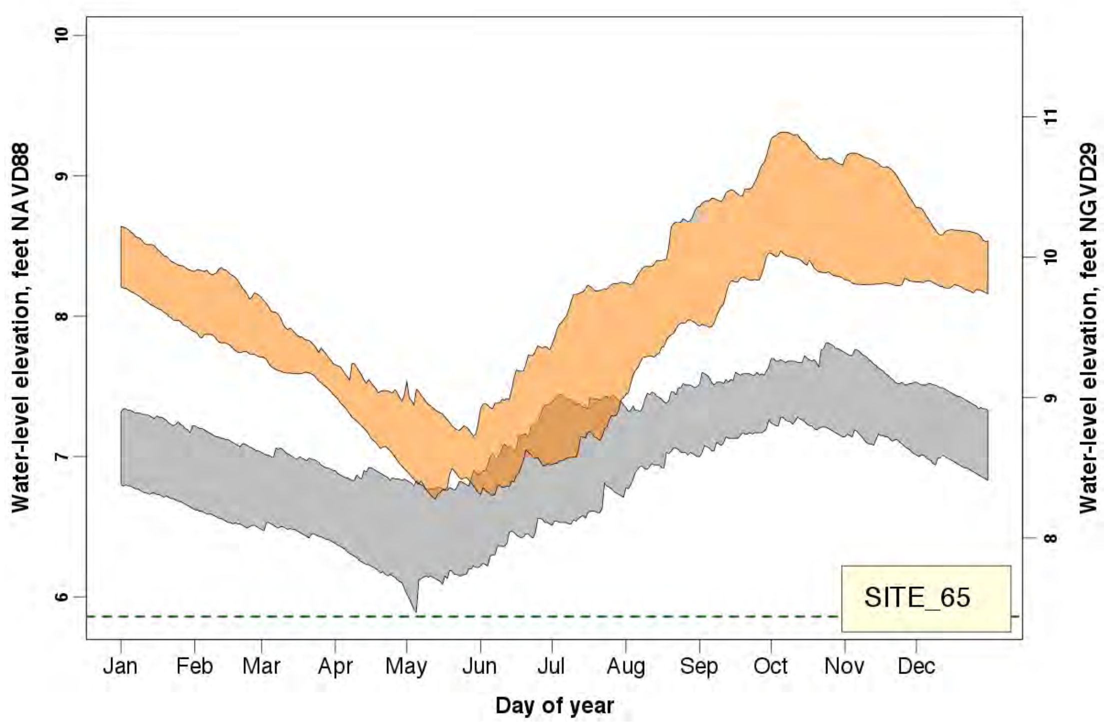

Figure A6-1-33. WCA 3A SW – Site 65 gage for Interim Operations Period (2000–2012).

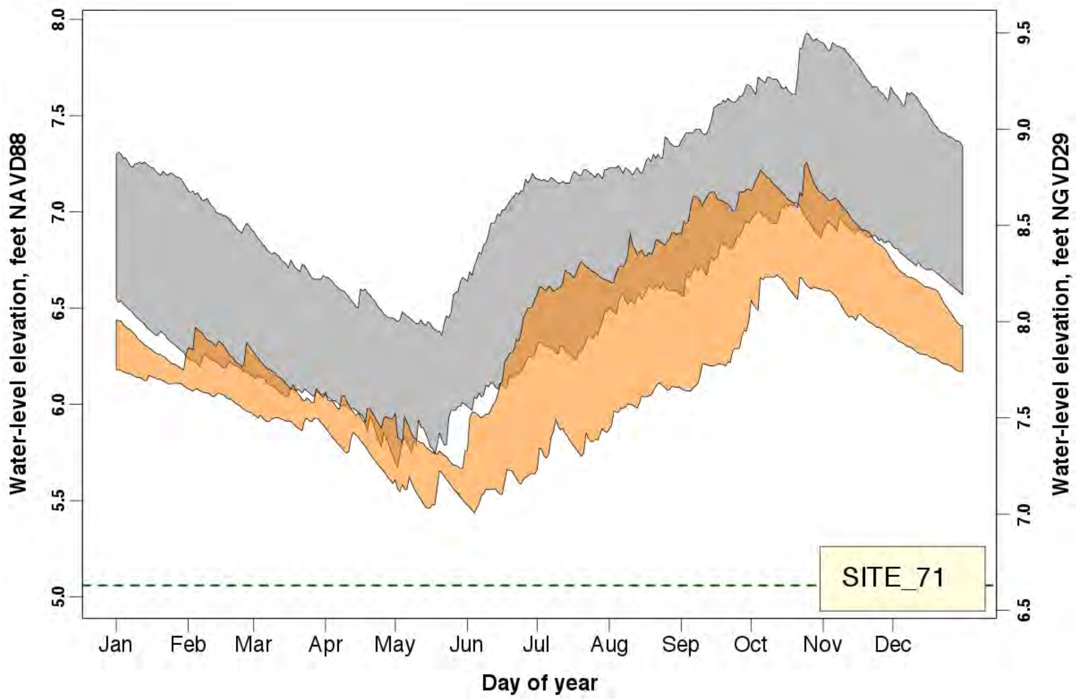

Figure A6-1-34. WCA 3B – Site 71 for WY 2000–WY 2008 (2009 SSR period).

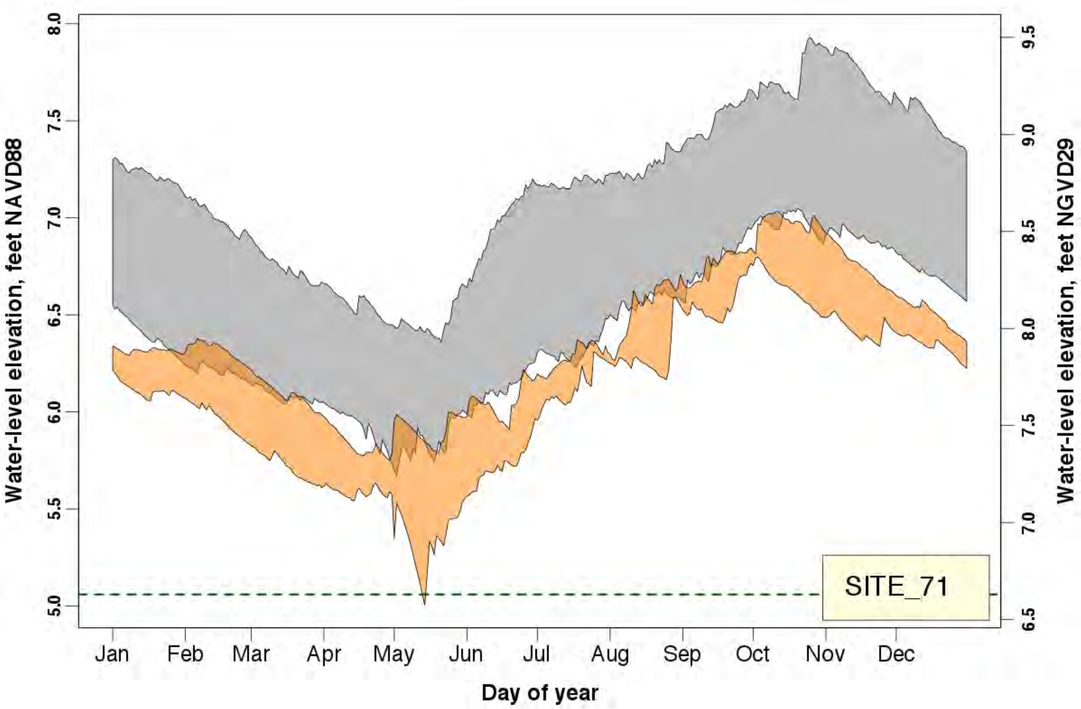

Figure A6-1-35. WCA 3B – Site 71 for WY 2009–WY 2013 (2014 SSR period).

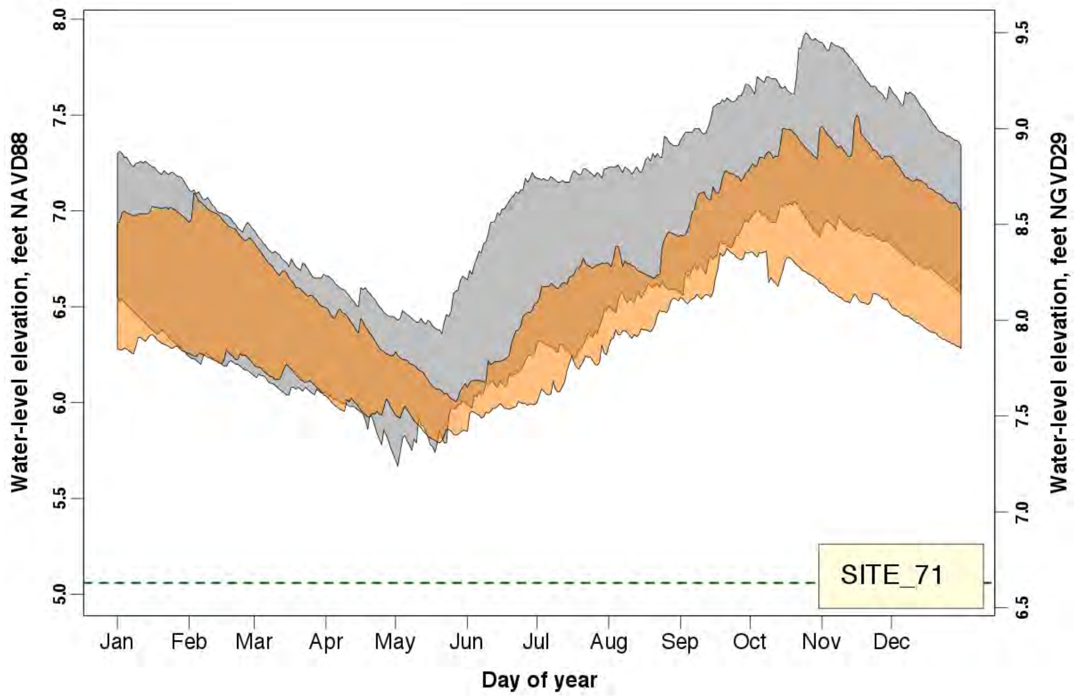

Figure A6-1-36. WCA 3B – Site 71 for the Experimental Program (1992–1999).

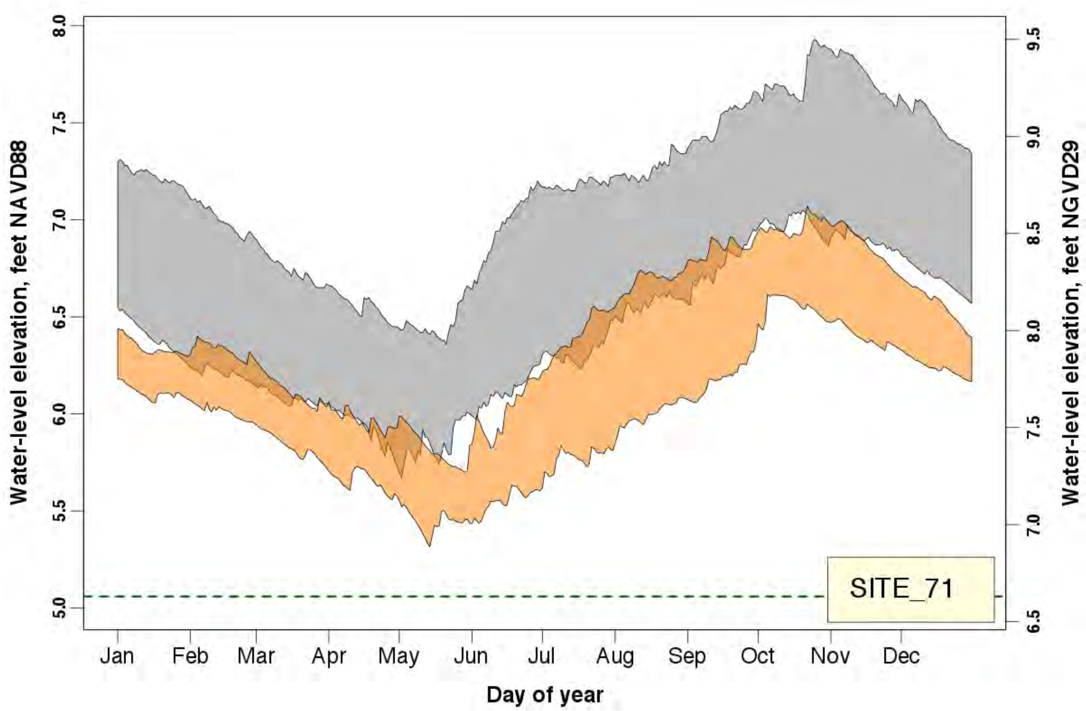

Figure A6-1-37. WCA 3B – Site 71 for the Interim Operations Period (2000–2012).

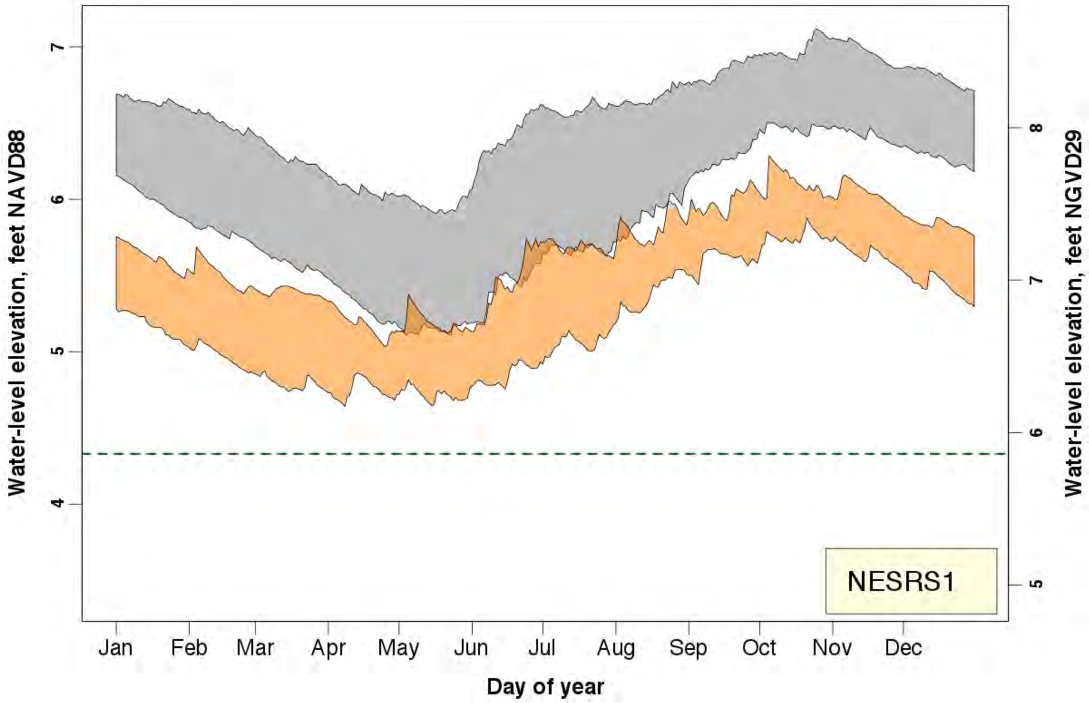

Figure A6-1-38. ENP NE – NESRS 1 gage for WY 2000–WY 2008 (2009 SSR period).

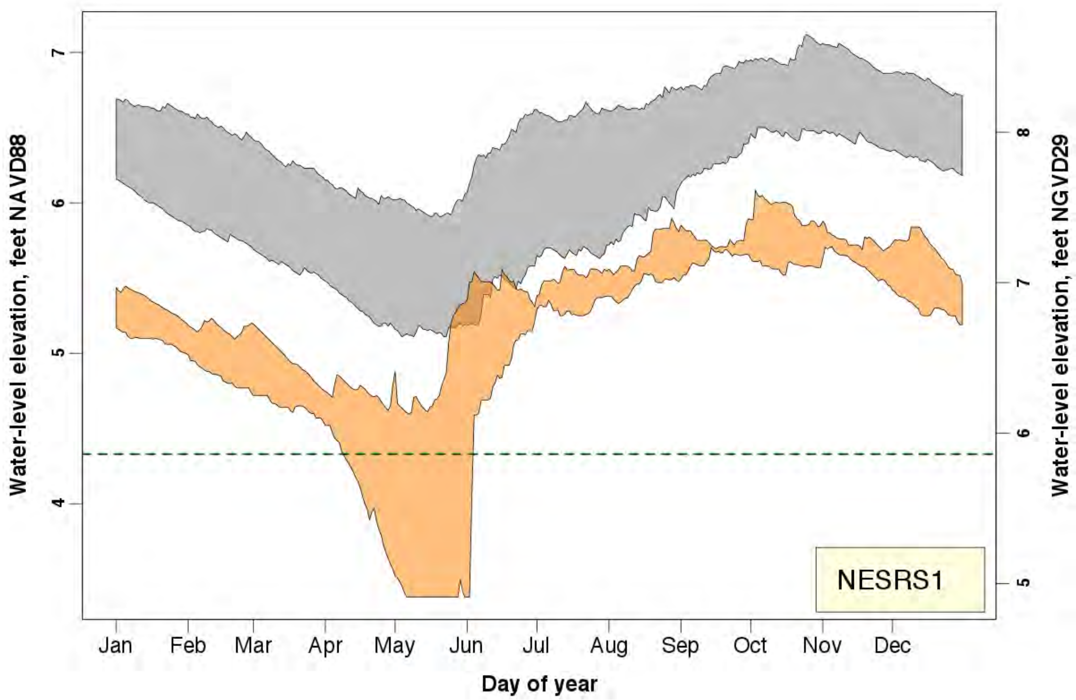

Figure A6-1-39. ENP NE – NESRS 1 gage for WY 2009–WY 2013 (2014 SSR period).

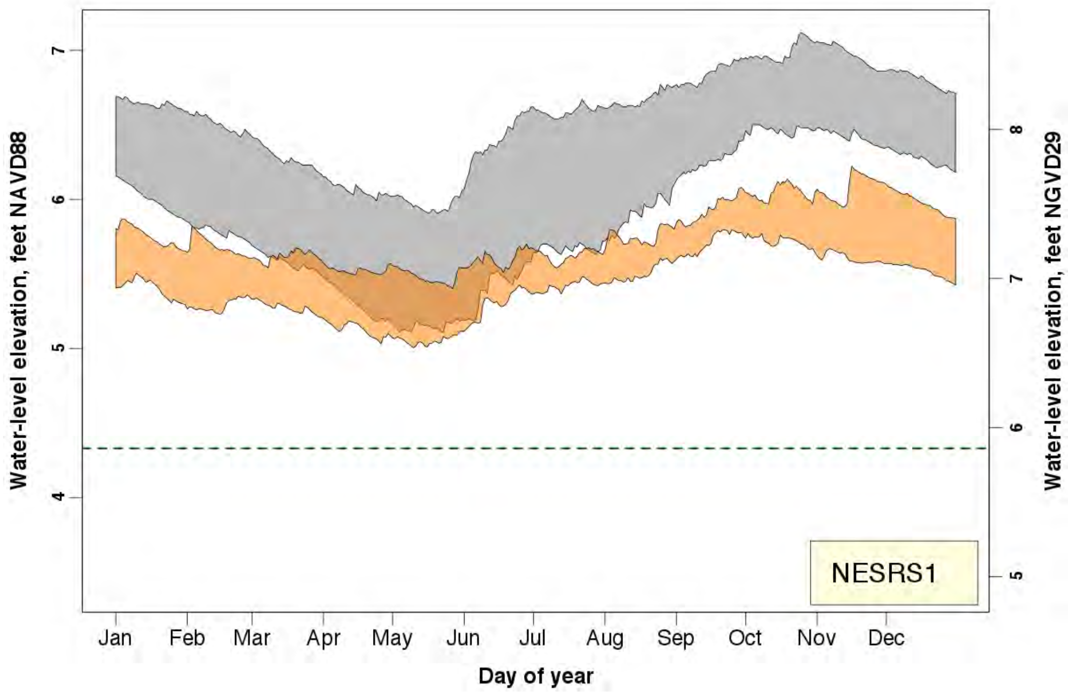

Figure A6-1-40. ENP NE – NESRS 1 gage for the Experimental Program (1992–1999).

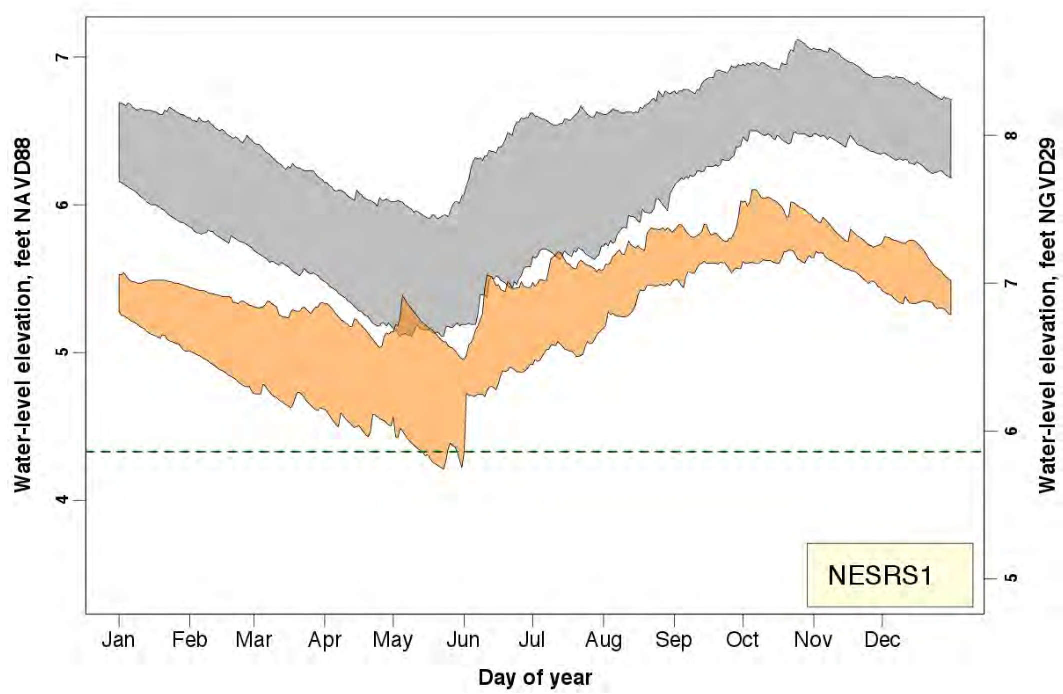

Figure A6-1-41. ENP NE – NESRS 1 gage for the Interim Operations Period (2000–2012).

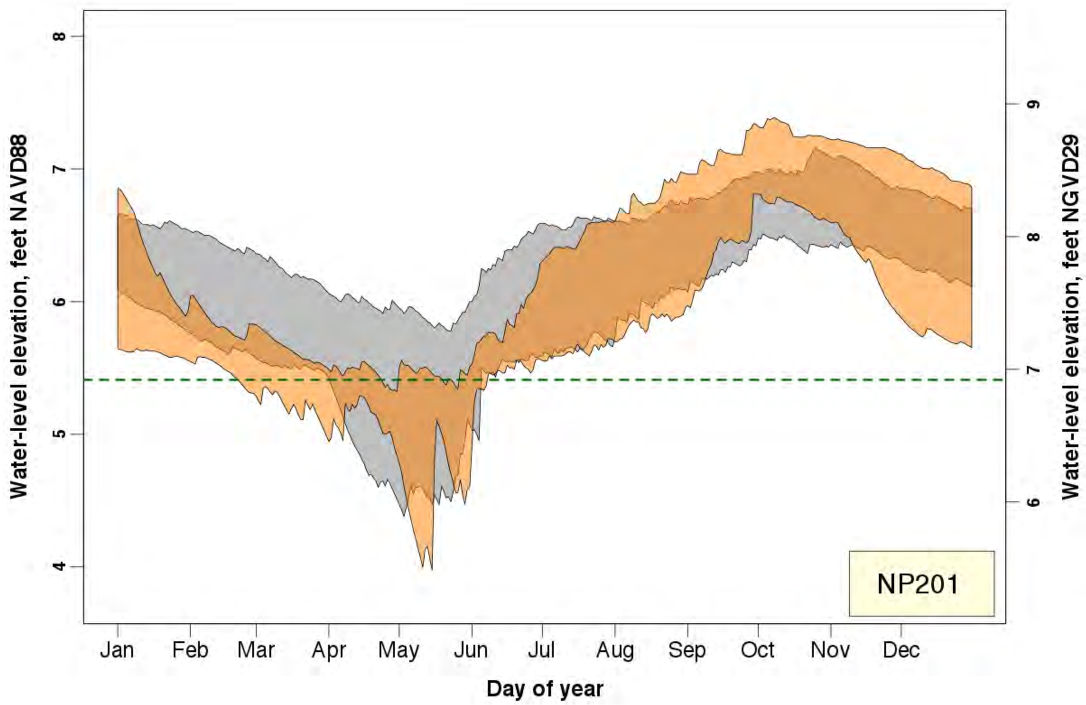

Figure A6-1-42. ENP NW – NP201 gage for WY 2000–WY 2008 (2009 SRR period).

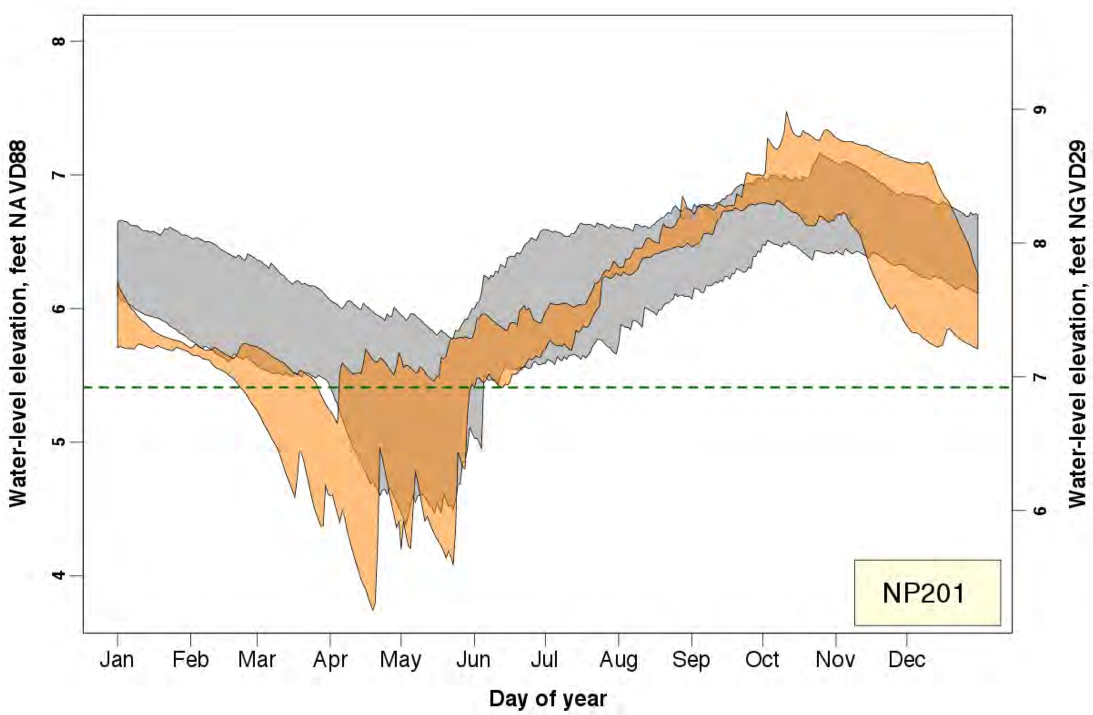

Figure A6-1-43. ENP NW – NP201 gage for WY 2009–WY 2013 (2014 SSR period).

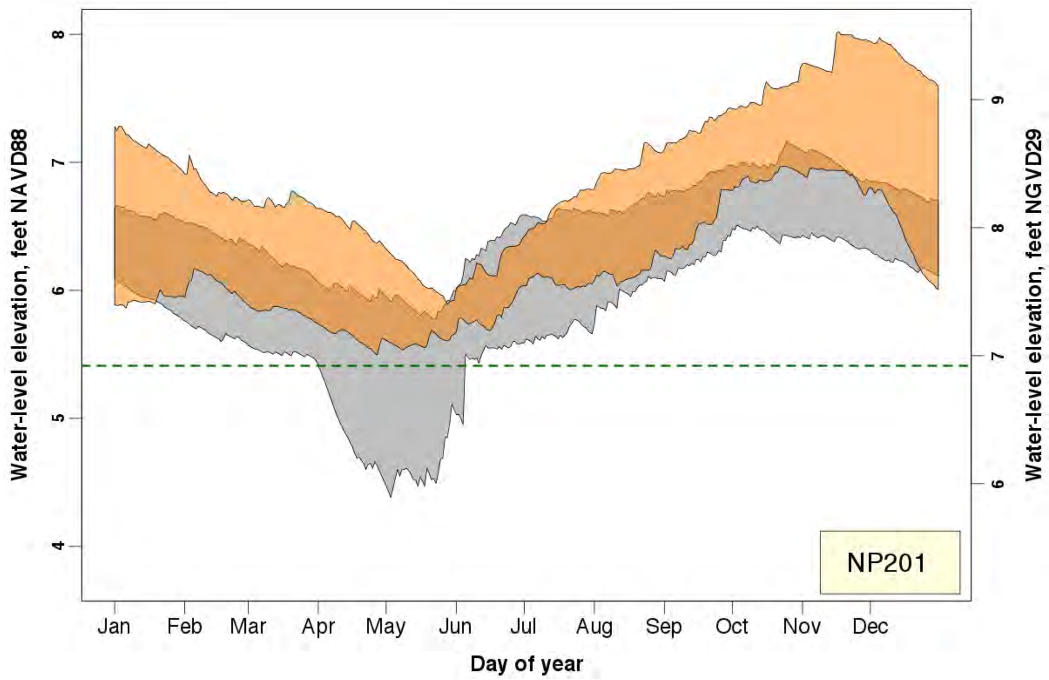

Figure A6-1-44. ENP NW – NP201 gage for the Experimental Program (1992–1999).

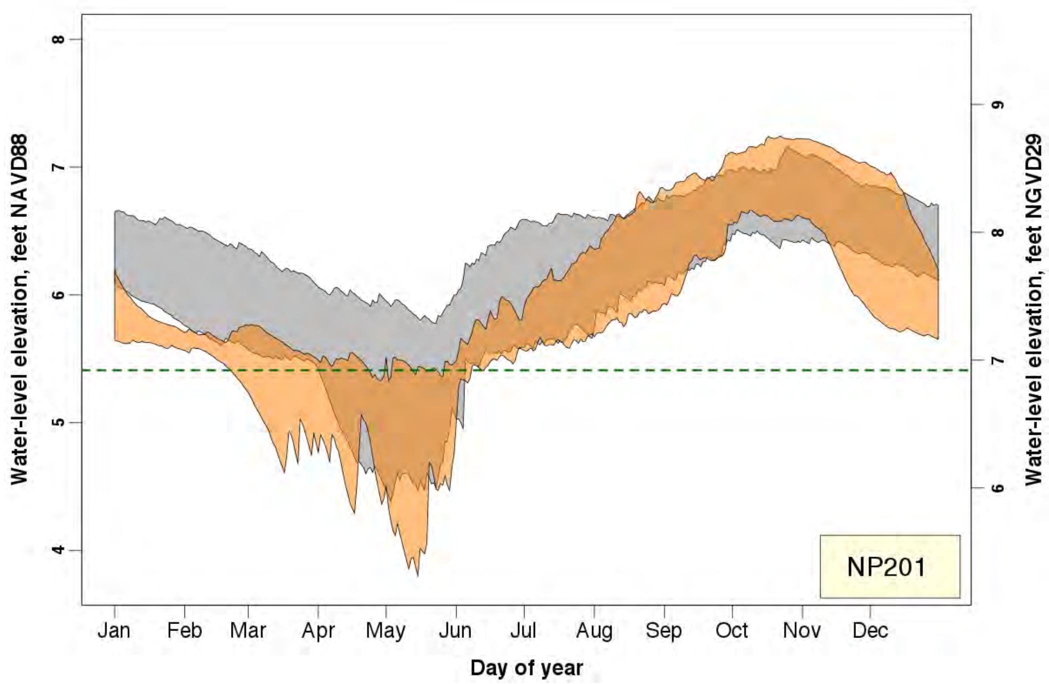

Figure A6-1-45. ENP NW – NP201 gage for the Interim Operations Period (2000–2012).

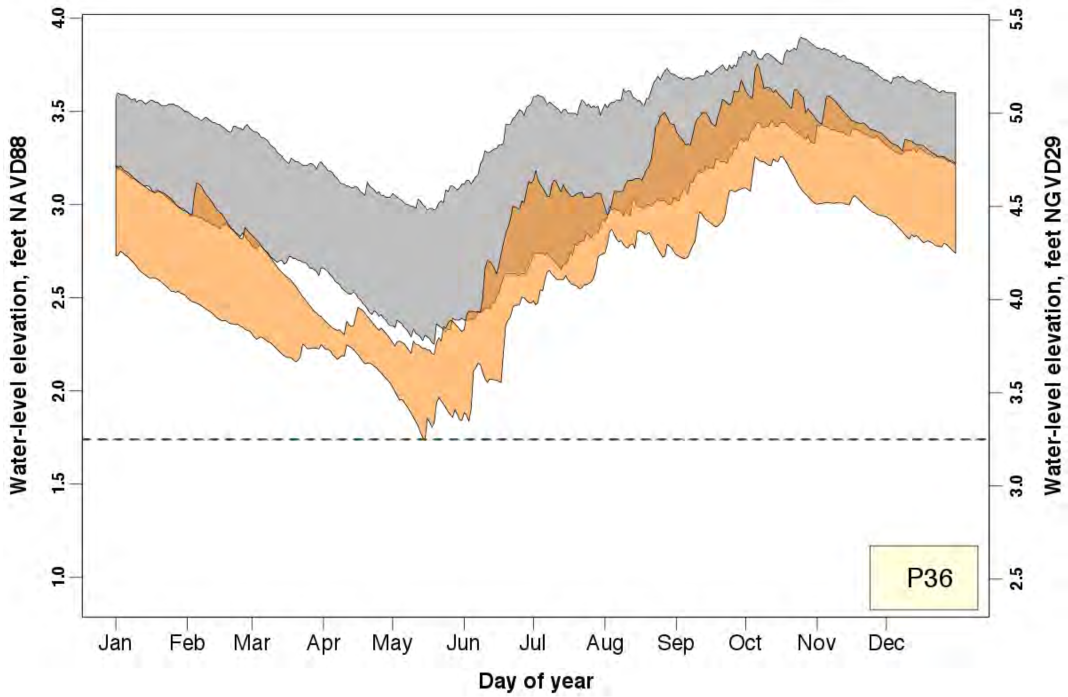

Figure A6-1-46. ENP S Shark River Slough – P36 gage for WY 2000-WY 2008 (2009 SSR period).

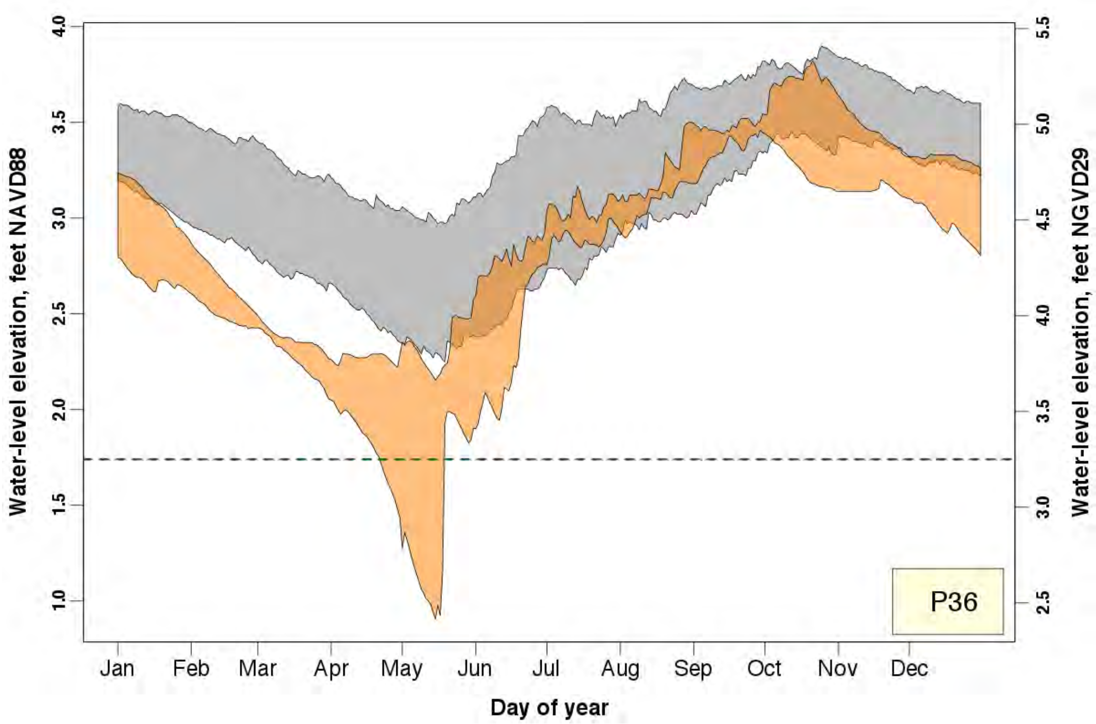

Figure A6-1-47. ENP S. Shark River Slough – P36 gage for WY 2009–WY 2013 (2014 SSR period).

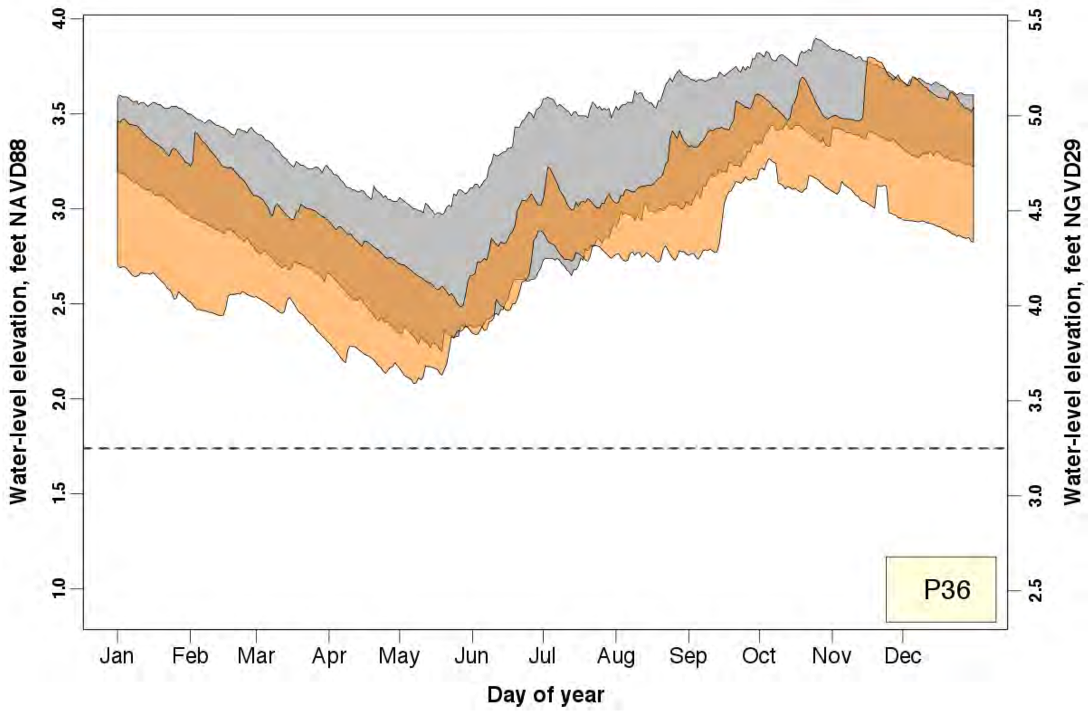

Figure A6-1-48. ENP S Shark River Slough – P36 gage for the Experimental Program (1992–1999).

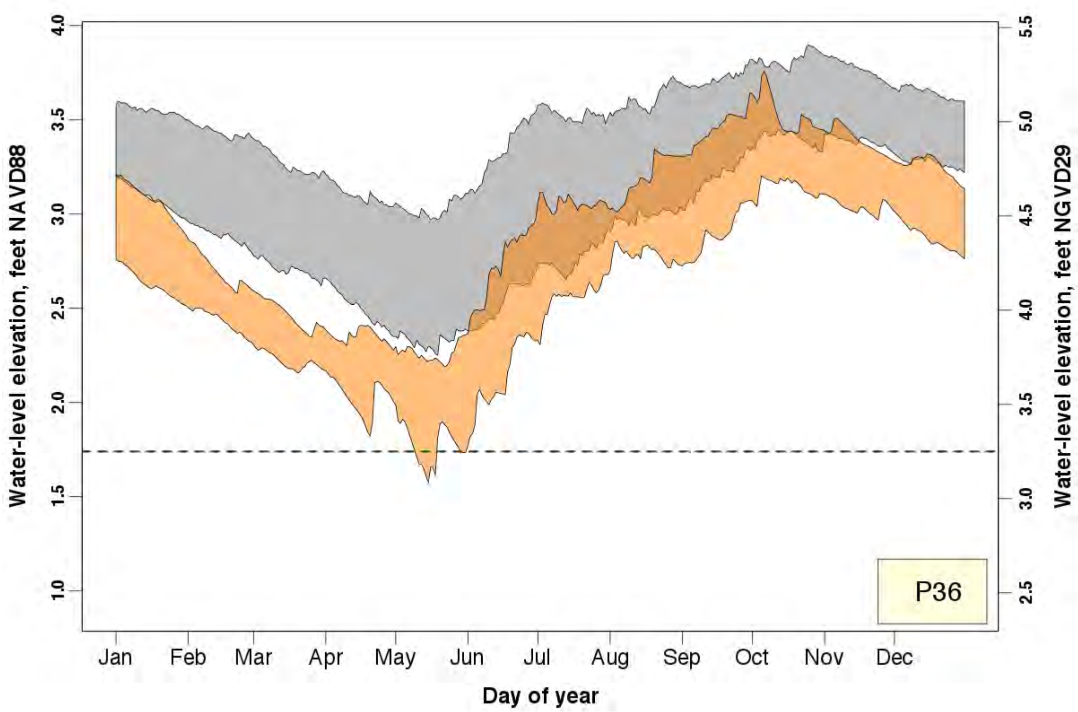

Figure A6-1-49. ENP S Shark River Slough – P36 gage for the Interim Operations Period (2000–2012).

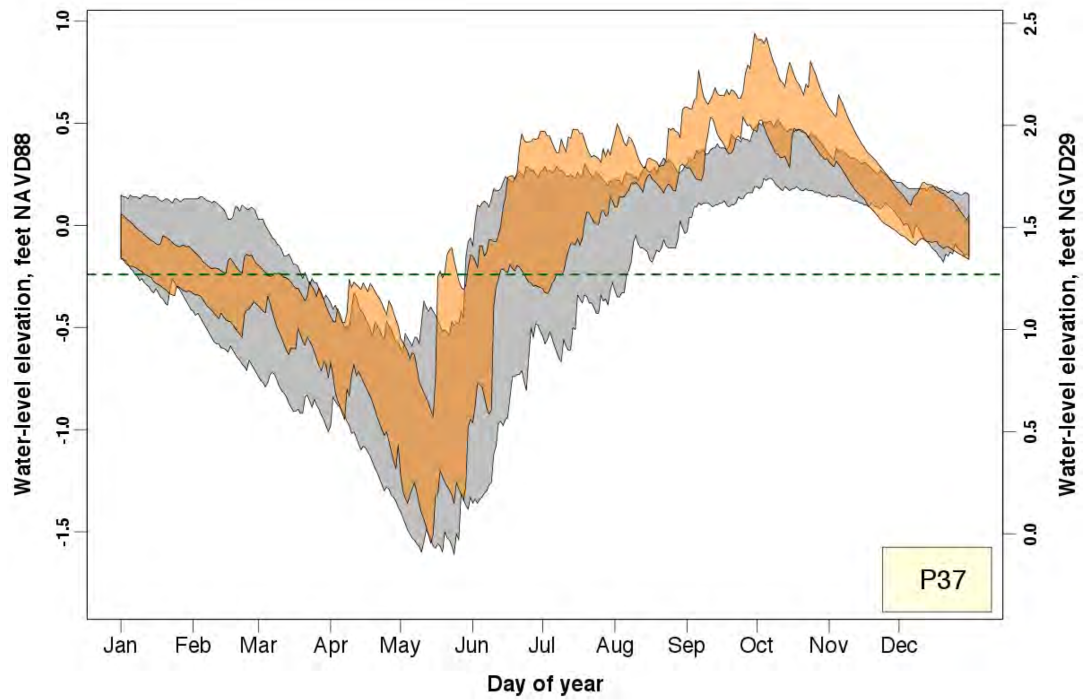

Figure A6-1-50. ENP SE Taylor Slough – P37 gage for WY 2000–WY 2008 (2009 SSR period).

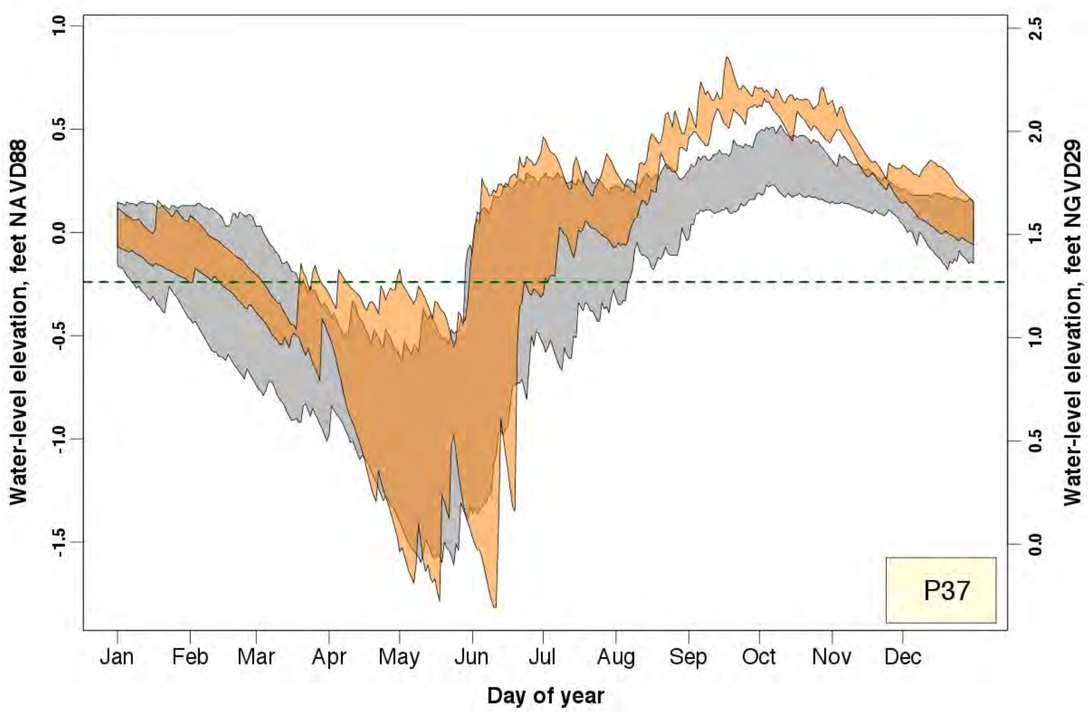

Figure A6-1-51. ENP SE Taylor Slough – P37 gage for WY 2009–WY 2013 (2014 SSR period).

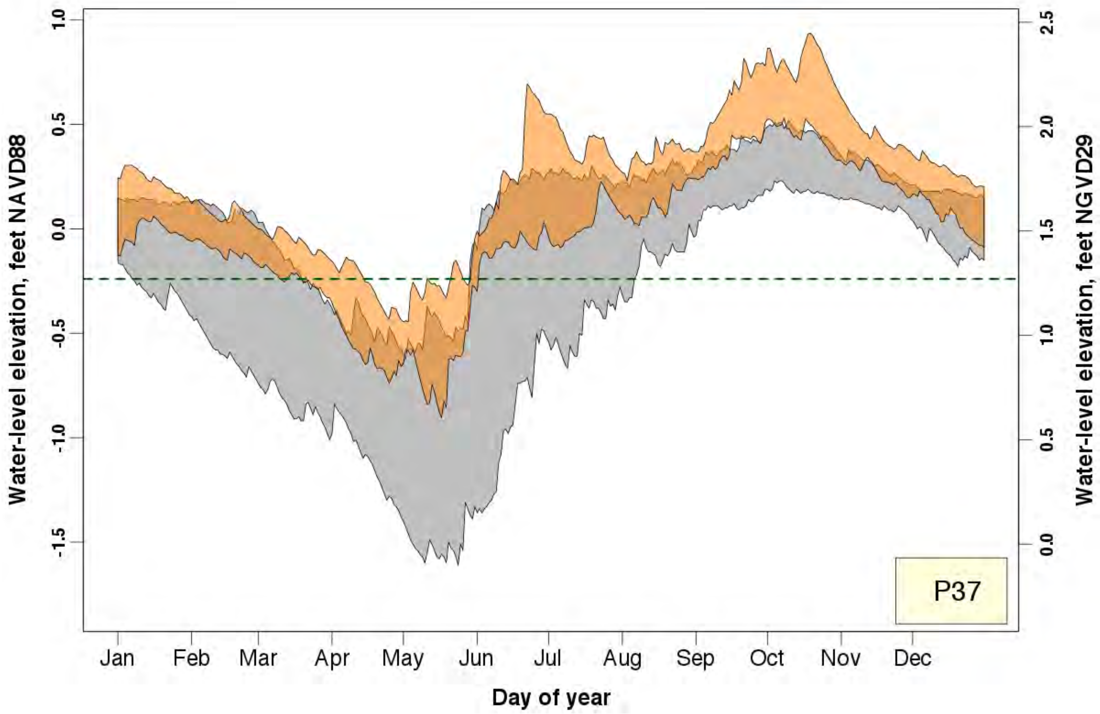

Figure A6-1-52. ENP SE Taylor Slough – P37 gage for the Experimental Program (1992–1999).

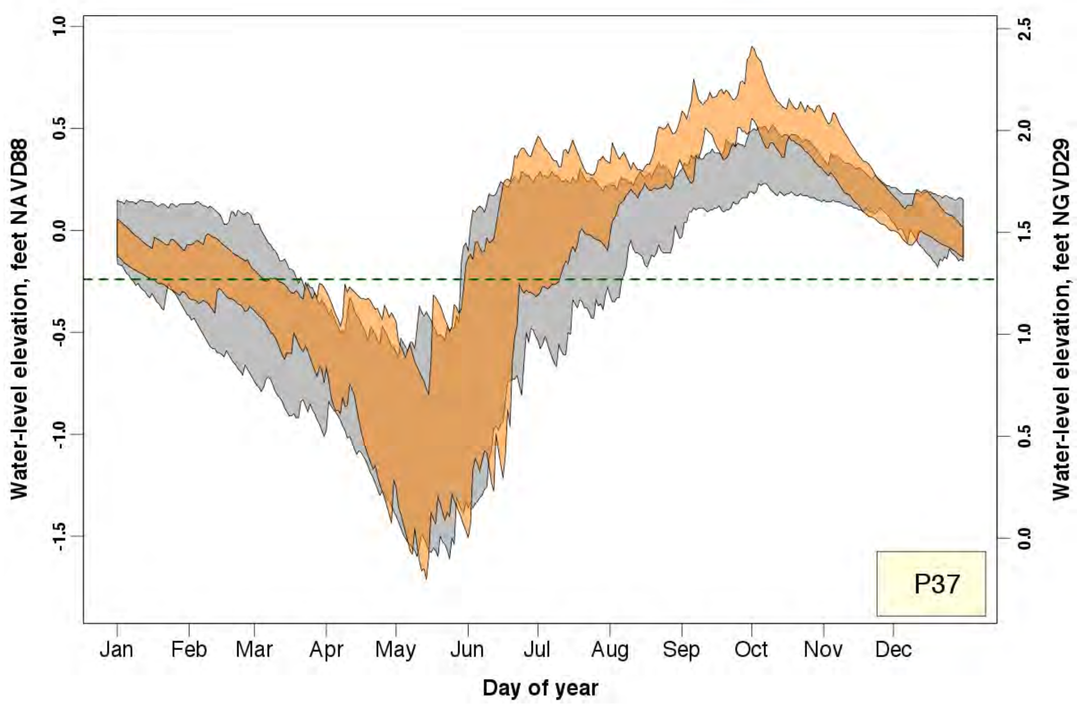

Figure A6-1-53. ENP SE Taylor Slough – P 37 gage for the Interim Operations Period (2000–2012).

CHAPTER 7

SOUTHERN COASTAL SYSTEMS

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TABLE OF CONTENTS

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CHAPTER 7 SOUTHERN COASTAL SYSTEMS

INTRODUCTION

The Southern Coastal System (SCS) region encompasses a large ecologically and economically important area that includes Biscayne Bay; Florida Bay and the Lower Southwest Coast; and the Upper Southwest Coast and Ten Thousand Islands coastal environments (**Figure 7-1**). This report is organized by each of these three subregions, which are distinguished by regional hydrology and monitoring program design. Over the past century, water management practices and agriculture/urban development have disrupted the availability, timing, and distribution of fresh water to the SCS, which has significantly altered the structure and function of the regional ecosystem. A primary goal of the Comprehensive Everglades Restoration Plan (CERP) for the SCS is to restore and sustain the highly productive estuaries and adjacent coastal wetlands of the SCS via the restoration of freshwater flows to the extent practical. Reestablishing more natural flows will restore estuarine salinity conditions, resulting in improved habitat for fish and wildlife resources. A brief description of the geographic setting and biological resources of each SCS subregion are provided in the following sections. A more thorough description of the SCS area setting can be found in the *2009 System Status Report* (SSR) (RECOVER 2011) and viewed at http://www.evergladesplan.org/pm/ssr_2009/mod_scs.aspx.

Biscayne Bay

Biscayne Bay is a shallow subtropical estuary on the southeastern coast of Florida and it is located primarily in Miami-Dade County. The average natural depth is three to nine meters. The bay is bordered to the west by the mainland of Florida and to the east by a series of barrier islands and the northern Florida Keys. The bay can be divided into four major areas: North Bay, Central Bay, South Bay, and Card and Barnes sounds (refer to **Figure 7-19** for the four major areas). Each of the four areas has distinct physical and ecological characteristics. Eleven major conveyance canals discharge fresh water into the bay from the mainland. The majority of the tidal exchange with the Atlantic Ocean occurs through an area known as the "Safety Valve[1]", a wide series of shoals and shallow cuts through barrier islands that separate Central Biscayne Bay from the Atlantic, and through narrow cuts and creeks in other parts of the bay. The Biscayne Bay Coastal Wetlands (BBCW) Project area of influence lies entirely within this SCS subregion.

[1] For location of the safety valve, please refer to the following link: http://www.nature.nps.gov/geology/inventory/publications/s_summaries/BISC_scoping_summary_20050421.pdf

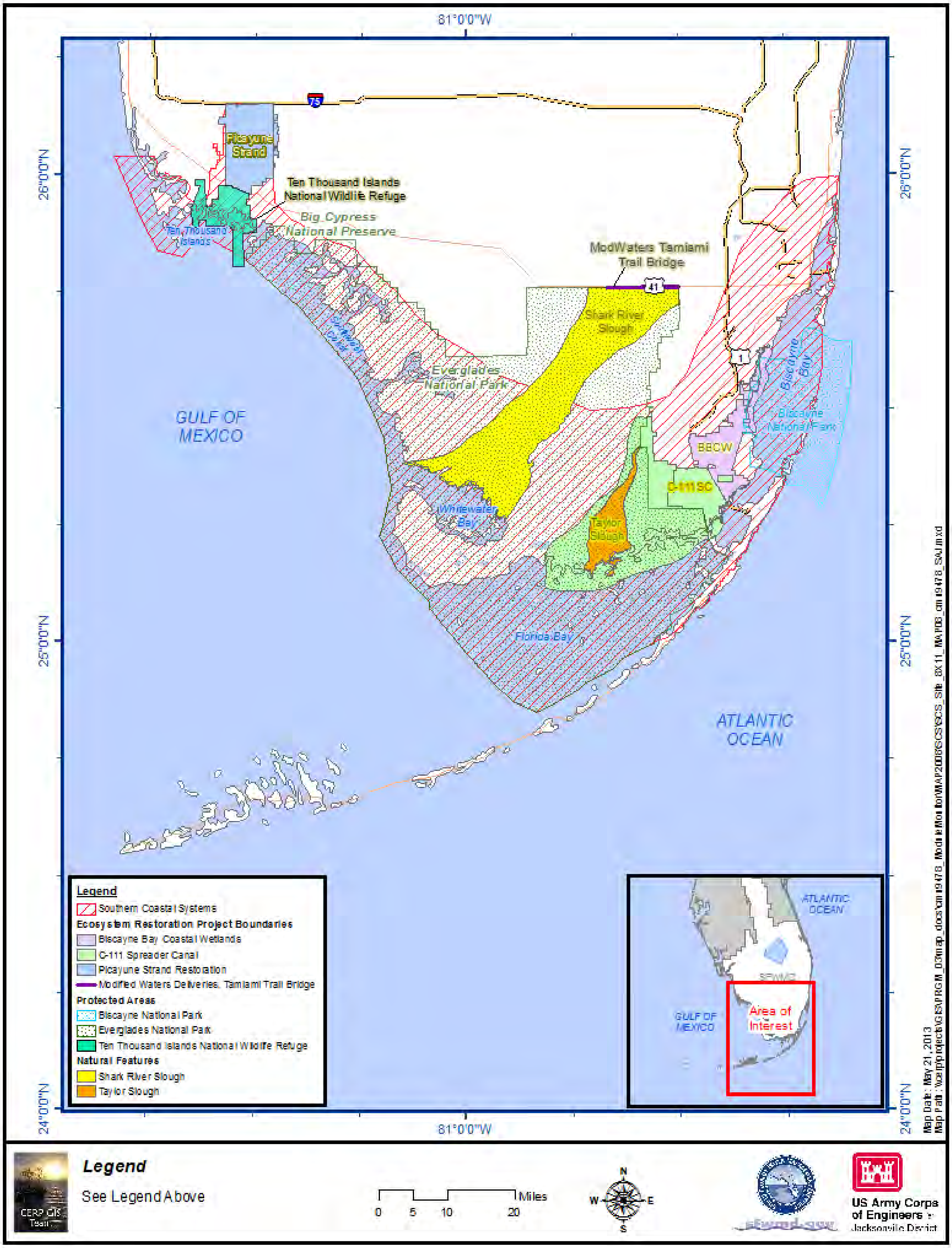


Figure 7-1. Map showing the SCS geographic extent (red boundary) and associated features.

Biscayne Bay is a naturally clear water bay with subtropical flora and fauna typical of the region. Prior to the urban and agricultural development, much of the bay was bordered by mangroves and herbaceous wetlands. The bay is hydrologically connected to the Greater Everglades ecosystem, historically, through tributaries, sloughs, and groundwater flow and beginning in the twentieth century, through conveyance canals. Both marine and estuarine habitats are part of the Biscayne Bay ecosystem. The productivity of the bay is largely benthic-based because of the bay's shallow depths and naturally clear waters (Roessler and Beardsley 1974). Although the area along Biscayne Bay from the Broward County line through the City of Miami is heavily impacted by adjacent urban development, benthic communities exist and are dominated by seagrasses intermixed with calcareous green algae. Development along Biscayne Bay south of the City of Miami grades from suburban to agricultural to park land where much of the natural mangrove wetlands are still intact along the western shore and eastern barrier islands because they lie within Biscayne National Park.

Florida Bay and Lower Southwest Florida Coast

Florida Bay is located at the southern tip of the Florida peninsula and it is characterized by a mosaic of banks, basins and small islands (**Figure 7-1**). A defining feature of the bay is its shallow depth, which averages about 1 meter (Schomer and Drew 1982). Light sufficient to support photosynthesis can reach the sediment surface in almost all areas of the bay (Kelble et al., 2005). Basins are as deep as 3 meters, and are separated by a network of shallow, flat-topped banks. Basins are connected with one another via a series of channels. These irregular mud banks are a conspicuous feature of Florida Bay, covering nearly 75 percent of Western Florida Bay and about 10 percent of the northeastern part of the bay. The bay also includes over 200 small islands, many of which are rimmed with mangroves. Over 85 percent of Florida Bay's 2,200 square kilometer (km^2) area lies within Everglades National Park (ENP); much of the remainder lies within the Florida Keys National Marine Sanctuary. The C-111 Spreader Canal Western Project (C-111 SCWP) area of influence lies entirely within this SCS subregion.

The shallowness of Florida Bay also affects its circulation and salinity regime. Exchanges with the Gulf of Mexico occur across the relatively open and broad western side of the bay, while exchanges with the Atlantic Ocean occur through narrow passes between the Florida Keys. Except for basins near the northern coast, which are near freshwater sources, the bay's water column is vertically well mixed and usually isohaline. In contrast, its complex network of shallow mud banks restricts horizontal water exchange amongst the bay's basins and between these basins and the Gulf of Mexico (Smith 1994, Wang et al. 1994). In areas of Florida Bay with long residence times, the salinity of the water can rise rapidly during drought periods due to an excess of evaporation relative to precipitation and freshwater inflow (Nuttle et al. 2000, Kelble et al. 2007). Salinity levels as high as twice that of seawater have been measured (McIvor et al. 1994).

The Lower Southwest Coast estuary extends from Cape Sable at the southwestern tip of the Florida peninsula up the southwestern coast of Florida to Lostmans River (**Figure 7-1**). The areas of Whitewater Bay and Shark River Slough (SRS) are included in this SCS subregion. Whitewater Bay and the rivers connecting SRS to the lower Southwest Florida Shelf (i.e., Shark, Harney and Lostmans) are critical components of the integrated SCS ecosystem. Much of the Everglades freshwater outflow did and still does affect the Lower Southwest Coast domain via discharge from SRS through various outlets.

Seagrass habitat is prevalent throughout Florida Bay and the Lower Southwest Coast. Various sponge species contribute significantly to the bay's benthic habitat. Florida Bay provides important habitat for many commercially important species, such as spiny lobsters (*Panulirus argus*), stone crabs (*Menippe mercenaria*), juvenile pink shrimp (*Farfantepenaeus duorarum*), and finfish species. Florida Bay and the Lower Southwest Coast, as well as the other SCS subregions, support numerous imperiled species including the West Indian manatee (*Trichechus manatus*), American crocodile (*Crocodylus acutus*), roseate spoonbill (*Ajaja ajaja*), and several species of sea turtles.

Upper Southwest Florida Coast and the Ten Thousand Islands

The Upper Southwest Coast extends from Lostmans River to the Ten Thousand Islands region just south of Naples, Florida (**Figure 7-1**). The area encompasses numerous small bays and includes one of the largest mangrove forested regions in the western hemisphere. This SCS subregion includes the northwestern coast of ENP, the Cape Romano-Ten Thousand Islands Aquatic Preserve, and the Rookery Bay Aquatic Preserve. The southwest coastline is dominated by mangroves (USACE and SFWMD 2004). The Picayune Strand Restoration Project (PSRP) area of influence lies entirely within this SCS subregion.

Currently, water from the Big Cypress National Preserve and local basin runoff dominates flows entering this SCS subregion. Historically, the Everglades and the Big Cypress Swamp extended as a continuous wetland across the southern part of the peninsula south of Lake Okeechobee (McPherson and Halley 1996), but subsequent redirections of flows and lowering of stage resulted in a decreased volume of water delivered to the estuarine systems.

The Upper Southwest Coast and Ten Thousand Islands subregion supports a variety of submerged aquatic vegetation (SAV) and oyster reefs near the mouths of some rivers. However, seagrasses are not nearly as prevalent as in Florida Bay or Biscayne Bay. This subregion supports a wide variety of fish and invertebrate species along with protected species, such as the West Indian manatee. Of particular note is the relatively high abundance of juvenile and adult stone crabs in coastal waters from Lostmans River northward.

Goals and Objectives

The focus of the Restoration Coordination and Verification Program (RECOVER) Monitoring and Assessment Plan (MAP) for the SCS has been to (1) understand the pre-restoration salinity and water quality variability in south Florida's coastal marine ecosystem; (2) gain the insight necessary to predict how this ecosystem may be altered by Everglades restoration; and (3) distinguish changes resulting from restoration from the backdrop of natural variation, cyclical patterns and episodic events in weather, and climate change. Understanding the present variability within the coastal ecosystem is essential to formulate sound working hypotheses regarding the effect of Everglades restoration projects on the downstream coastal ecosystem; to assess the actual effects of restoration on the coastal ecosystem; and to provide timely feedback to managers necessary to successfully implement iterative adaptive restoration.

The current salinity regime in the SCS can create significant stress on its inhabitants via hypersaline events, changes in salinity regimes in areas with reduced freshwater discharge, and rapid reductions in salinity due to pulsed freshwater flows from canals. As a consequence, the current nearshore biotic communities differ from historical nearshore communities. One of the primary restoration goals for salinity is to reestablish a more natural estuarine salinity gradient from nearshore to offshore by returning to a more diffuse runoff pattern. Accomplishing this goal is expected to allow biota to return to a more natural species composition, distribution, and abundance. Moreover, decreasing the intensity, duration, frequency, and spatial extent of high salinity events should allow a more productive upper trophic level biotic community. For example, juvenile spotted seatrout (*Cynoscion nebulosus*), a fisheries indicator species, has significantly lower abundance during high salinity events in North-central Florida Bay.

Water quality in the SCS is dependent upon the volume, distribution, and quality of fresh water flowing to the system. The quality of fresh water flowing to the estuaries is heavily influenced by both natural nutrient inputs from biogeochemical wetland processes transported through overland flow or creeks and anthropogenic nutrient inputs from agricultural and urban areas runoff primarily transported in conveyance canals. In general, the primary restoration goal is to protect water quality, maintaining current conditions. Algal blooms that can be stimulated by nutrients from the watershed are a continuing concern in SCS waters. Such blooms can decrease light penetration, impacting seagrass habitat. The desired condition is to minimize the magnitude, duration, and spatial extent of algal blooms to sustain healthy and productive habitat. It will be critical for CERP to strike a balance whereby the benefits afforded by increased flow and improved salinity regime for faunal utilization are not undone by adverse effects from potentially increased nutrient loading; CERP's goal to "get the water right" includes protecting all the south Florida ecosystems, including its coastal systems.

Restoration goals for SAV include a more diverse seagrass community with greater spatial coverage of seagrass beds containing mixed species, such as *Thalassia testudinum* (turtle grass), *Halodule wrightii* (shoal grass), and *Ruppia maritima* (widgeon grass). For purposes of this report, SAV generally includes seagrass and benthic macroalgae *(Chara* sp., *Najas* sp., and *Utricularia* sp.). In Florida Bay, *T. testudinum* should have lower density and biomass than during the period prior to the die-off in the late 1980s, when turtle grass beds had extremely high blade density. In the Everglades mangrove estuaries, there should be an increase in cover and seasonal duration of *R. maritima* in coastal lakes and basins compared to the last 20 years. For South Biscayne Bay, there should be an increase in cover of seagrass beds, consisting primarily of species tolerant of lower salinity (i.e., *H. wrightii* and *R. maritima*) in nearshore areas. For North Biscayne Bay, the existing seagrass communities should be maintained.

Restoration of salinity regimes and suitable habitats (i.e., SAV) should result in the reestablishment of lower trophic levels to desired community composition, and species abundance and diversity. Currently, a key indicator in this group for the SCS is juvenile pink shrimp. The desired restoration condition for pink shrimp in Florida Bay and Biscayne Bay is an increase in peak abundance during the August to October period in optimal habitat in three regions of Florida Bay, Ponce de Leon Bay (on the lower southwestern mangrove coast), and in southwestern nearshore Biscayne Bay compared to current conditions.

Restoration goals for higher trophic levels include enhanced function of the SCS as nursery grounds for fishery species, and increasing the diversity and density of fish and invertebrate assemblages along the mangrove shorelines of the SCS. A more specific goal includes increasing the distribution, abundance, growth, and survival of juvenile spotted seatrout in North-central and Western Florida Bay compared to current conditions. In South Biscayne Bay, the desired condition is to increase productivity of nearshore mesohaline fish and invertebrates, resulting in higher abundances of mesohaline forms and lower abundances of purely marine forms, such as some coral reef fishes. Another goal is to restore habitat for apex predators, such as the imperiled American crocodile.

More detailed information on the desired restoration condition and CERP goals and objectives for the SCS can be found in the RECOVER performance measure documentation sheets (http://www.evergladesplan.org/pm/recover/perf_se.aspx), the CERP MAP (RECOVER 2009; http://www.evergladesplan.org/pm/recover/recover_map_2009.aspx), and the 2007 and 2009 SSRs (RECOVER 2007, 2011). Spatially explicit SAV restoration targets for the Florida Bay ecosystem are discussed in detail in the Florida Bay and Florida Keys Feasibility Study Draft Performance Measures (USACE and SFWMD 2004) and to a lesser extent in the Florida Bay and Everglades Mangrove Estuaries conceptual ecological models (Rudnick et al. 2005, Davis et al. 2005). Updates on the three CERP projects currently being constructed and implemented within the SCS region are provided later in this chapter.

BACKGROUND INFORMATION

The following section provides background information applicable to the SCS region. Descriptions of relevant CERP and non-CERP projects and operational and structural changes over the last three decades are included. In addition, background information on salinity (a key driver of the SCS) and other environmental factors are included in this section. For more detailed information see **Appendices 7-1** through **7-3**.

Operational and Structural Regimes and Changes

Since the 1980s, there have been a number of structural and operational changes that have affected freshwater inflow to the SCS region. These include the following:

- Increased flow to Biscayne Bay via the C-1 Canal by 25% beginning in 2000.
- Increased flow to Taylor Slough via increased pumping at the S-332 structure (1991–1999) and subsequently (beginning in 2000) via operation of ENP eastern boundary detention areas with S-332B, C, and D structures.

- Removal of dredge spoil from the south bank of the C-111 Canal in 1997 that increased marsh stage downstream of the C-111 Canal and increased southerly water flow to Northeast Florida Bay.
- Construction of ENP road bridges across northern Taylor Slough in 1999 in order to allow more north-to-south flow down the slough.
- Increased connection between Long/Blackwater/Little Blackwater Sounds and Manatee Bay/Barnes Sound as a result of the U.S. Highway 1 widening project (2005–2009).
- Increased flow per unit rainfall across the Tamiami Trail near the S-12 water control structures (early 1990s) that is likely due to an increase in flows entering Water Conservation Area (WCA) 3 from the Everglades Agricultural Area. It is unclear how much of this additional water exits SRS in the estuaries of the southwest Florida coast and Florida Bay.
- Increased flow at the Faka Union water control structure beginning in 1999, possibly due to the construction of the Lucky Lake Water Control Structure at the northern section of PSRP, which was completed in July 1999 and is an adjustable crest weir with 8 operable steel gates. The purpose of the weir was to reduce the drainage impacts of the Southern Golden Gage Estates (SGGE) canal system by holding stages higher in the Lucky Lake Strand area north of Merritt Canal.

For a more detailed description of these operational and structural changes, refer to **Appendix 7-1**. It is unclear if any of these operational and structural changes have altered downstream salinity in the estuaries because no specific studies have been conducted to determine the effects. Where possible and practical, evaluations of MAP data in relation to these structural and operational changes are provided in this report.

Much of the period assessed in this report, WY 2000–WY 2013 (May 1, 1999–April 30, 2013), falls under the water management operational plan called the Interim Operational Plan (IOP). This plan was instituted in late 2001, finalized with the release by the United States Army Corps of Engineers (USACE) of the June 2002 *Final Environmental Impact Statement for the Interim Operational Plan (IOP)* (USACE 2002), and replaced in October 2012 by the Everglades Restoration Transition Plan (ERTP) (USACE 2011). Prior to the formal institution of IOP in June 2002, water management operations fell under the Interim Structural and Operational Plan (ISOP), which consisted of two generations of plans: ISOP 2000 implemented from December 1999 to January 2001 and ISOP 2001 implemented from January 2001 to July 2002 (USACE 2000). ISOP, IOP, and ERTP are the operational plans that are most likely to affect conditions in Florida Bay and the Lower Southwest Coast. An early assessment of IOP by the National Park Service (NPS) (SFNRC 2005) found that under the combined ISOP/IOP operational plans, salinity conditions significantly increased during the late dry season as a function of ISOP/IOP at two Florida Bay stations downstream of SRS and in Highway Creek. Model simulations of ISOP/IOP compared to 1995 baseline conditions also showed significantly higher dry season salinity at several Florida Bay stations scattered throughout the bay. The actual and simulated dry season increases in salinity are counter to CERP efforts to restore favorable estuarine conditions to Florida Bay and the Lower Southwest Coast.

Physical Environment

As with other regions in the Everglades ecosystem, the physical environment plays a critical role in determining community composition and health of the SCS region. These physical factors include topography and bathymetry, geology, hydrology and hydrodynamics, fire, hurricanes, climate, weather (i.e., temperature extremes and droughts), water quality, light availability, and salinity. This section will describe these factors in three components: (1) the physical connectivity of the SCS subregions to the upstream Greater Everglades RECOVER region, (2) the importance of salinity as an ecological driver and stressor in the SCS, and (3) the effects of other environmental factors listed above.

Connectivity

SCS is irrevocably linked to the upstream environment of the Greater Everglades. This linkage provides the fresh water that establishes and maintains brackish salinity that is critical to the health and well-being of the native estuarine flora and fauna of the region. Each of the SCS subregions has a unique physical connectivity to the upstream environment. For example, the Biscayne Bay subregion is separated from the Greater Everglades by the Atlantic Coastal Ridge and the connectivity to it was historically via transverse glades (McVoy et al 2011). Today, that connectivity is via major drainage canals that mostly lie in the footprint of the historic transverse glades. Other subregions connect to their upstream environment in different ways. It is important to understand the physical connectivity between the southern estuaries and the upstream watershed and the factors that control freshwater movement between the regions. These connections are described in detail in **Appendix 7-2**. Both historical and present day connectivity are provided.

Salinity

Marine and estuarine communities are tied closely and directly to salinity conditions. Salinity can affect these communities by being too high, too low or too variable. Salinity patterns directly influence productivity, population distribution, community composition, predator-prey relationships, and food web structure in the inshore marine habitat (Myers and Ewel 1990, Kennish 1990). Salinity is arguably the most important physical parameter for determining species and community composition in coastal waters. Departures from natural salinity patterns are ecologically damaging to many species because salt concentration affects growth, survival, reproduction, and other critical physiological processes in both plants and animals. Salinity is identified as a stressor in virtually all conceptual ecological models developed for Florida’s estuaries and adjacent mangrove wetlands (Browder et al. 2005, Davis et al. 2005, Ogden et al. 2005, Rudnick et al. 2005, Nuttle and Fletcher 2013a,b,c). Further, estuarine salinity is the physical parameter most likely to be altered by Everglades restoration, and the primary causative agent for corresponding changes in flora and fauna.

To maintain appropriate salinity conditions for flora and fauna, estuarine communities rely on a balance of fresh water delivered in the appropriate amounts with proper quality, timing, and location.

Restoration will substantially affect freshwater delivery patterns and the resultant coastal wetland and estuarine salinity. The restoration of more natural distribution of freshwater flows to coastal wetlands will result in spatially and temporally diffuse surface and groundwater flow into the estuaries producing a gradient of increasing salinity from interior wetlands into the nearshore zone with a greater spatial extent of estuarine salinity zones and more gradual seasonal changes in salinity patterns. Changes in salinity resulting from restoration are expected to translate into changes in biomass, distribution, species composition and diversity of seagrass, benthos, fish, invertebrates, birds, and crocodiles. In general, systemwide plant and animal diversity will increase as more natural freshwater flows and salinity regimes are restored. A thorough characterization of historic and current salinity and flow patterns, anthropogenic alterations to the system, and desired restoration conditions in the SCS can be found in the 2009 SSR (RECOVER 2011) and in the documentation sheet for the RECOVER Southern Estuaries salinity performance measure, which can be viewed online at www.evergladesplan.org/pm/recover/recover_docs/perf_measures.

Other Environmental Effects

A variety of other environmental factors impact the SCS including sea level rise, hurricanes, drought, fire, nonnative species, and climate change. These factors and their effects on the SCS are described succinctly in **Appendix 7-3** of this report. Given its subtropical climate, another important physical factor affecting the SCS is the occurrence of cold events. Since the last SSR, a severe cold event significantly affected south Florida in 2010. A special segment is included in this report describing the effects of this cold event on the SCS flora and fauna.

ASSESSMENTS BY SUBREGION

The 2009 SSR (RECOVER 2011) provided assessments by physical or biological component. Each component included results from all geographic subregions of the SCS. For the 2014 SSR, assessments are provided by subregion. In that way, the connection between upstream stage/flow to salinity to habitat to upper trophic levels can be conveyed to the extent practicable.

Assessments are generally provided on a monthly, seasonal, and/or annual temporal resolution. Seasonal assessments are by water year periods and follow the South Florida Water Management District (SFWMD) definition, which characterizes each water year as beginning on May 1 and ending on April 30 of the following year. For example, Water Year (WY) 2013 began on May 1, 2012 and ended on April 30, 2013. The term “wet season” is synonymous with “rainy season” and runs May 1 through October 31, which may not be coincident with the period of lowest salinity in receiving waters.

Biscayne Bay

Assessments for Biscayne Bay include five primary components. The first will be results from the Integrated Biscayne Bay Ecological Assessment and Monitoring Project (IBBEAM), a RECOVER MAP project. This is a collaborative effort by NPS, the National Oceanic and Atmospheric Administration (NOAA), and the University of Miami to assess conditions in the nearshore waters of CERP’s BBCW Project. Parameters include salinity, SAV, epifauna, and mangrove fish communities. The second component is the status and trends of salinity in Biscayne Bay outside the area included in the IBBEAM project. The third component is provided by the Miami-Dade County Department of Environmental Resources Management (Miami-Dade DERM) and includes assessments of water quality throughout Biscayne Bay. The fourth component, also provided by Miami-Dade DERM, includes SAV assessments throughout the bay. A fine-scale assessment of a CERP project affecting Biscayne Bay is found later in this section. Note that assessment of crocodile status and trends in Biscayne Bay are found in the Florida Bay section.

Integrated Biscayne Bay Ecological Assessment and Monitoring Project

The IBBEAM combines four previously individual projects funded by the RECOVER MAP: Salinity Monitoring Network (3.2.3.3), Nearshore SAV (3.2.3.3), Alongshore Epifauna (3.2.4.7), and Mangrove Fish (3.2.3.6). As a means of continuing the projects assessing Biscayne Bay, IBBEAM takes advantage of efficiencies derived from combining all four efforts and sampling cooperatively to maintain a reduced level of monitoring with the allotted funding, albeit at a reduced spatiotemporal scale due to the funding reduction (~50%) and staff reduction.

IBBEAM objectives, supporting assessment of CERP success, are to (1) facilitate comparison of past and present salinity regimes, SAV communities, SAV-associated fish and invertebrate assemblages, and mangrove- associated fishes to determine status and trends and enable before-and-after CERP comparisons; (2) quantify key relationships between salinity and the diversity, distribution, and abundance of SAV, epifaunal fishes and invertebrates, and mangrove shoreline fishes; and (3) provide a scientific basis for formulating performance measures and targets to assess the effectiveness of CERP projects and assisting with adaptive management.

Data and Methods

The focal study area is the alongshore strip between Shoal Point and Turkey Point, which coincides with the nearshore area of the BBCW Project where changes may be most easily detected (**Figure 7-2**). An analysis of the salinity database was conducted in conjunction with biotic program salinity values to link the sampling sites of the four projects for coordinating field collection efforts, integrating databases, and conducting future analysis. Sampling visits are scheduled to coincide in time and space and to share field personnel.

Seventeen pre-established Biscayne National Park water quality (salinity, temperature, and depth) stations are sampled. These are located along the shoreline in the first areas that will be affected by changes in freshwater delivery to the bay as part of the BBCW Project (**Figure 7-2**). The water quality station locations represent “hubs” around which all biotic sampling takes place.

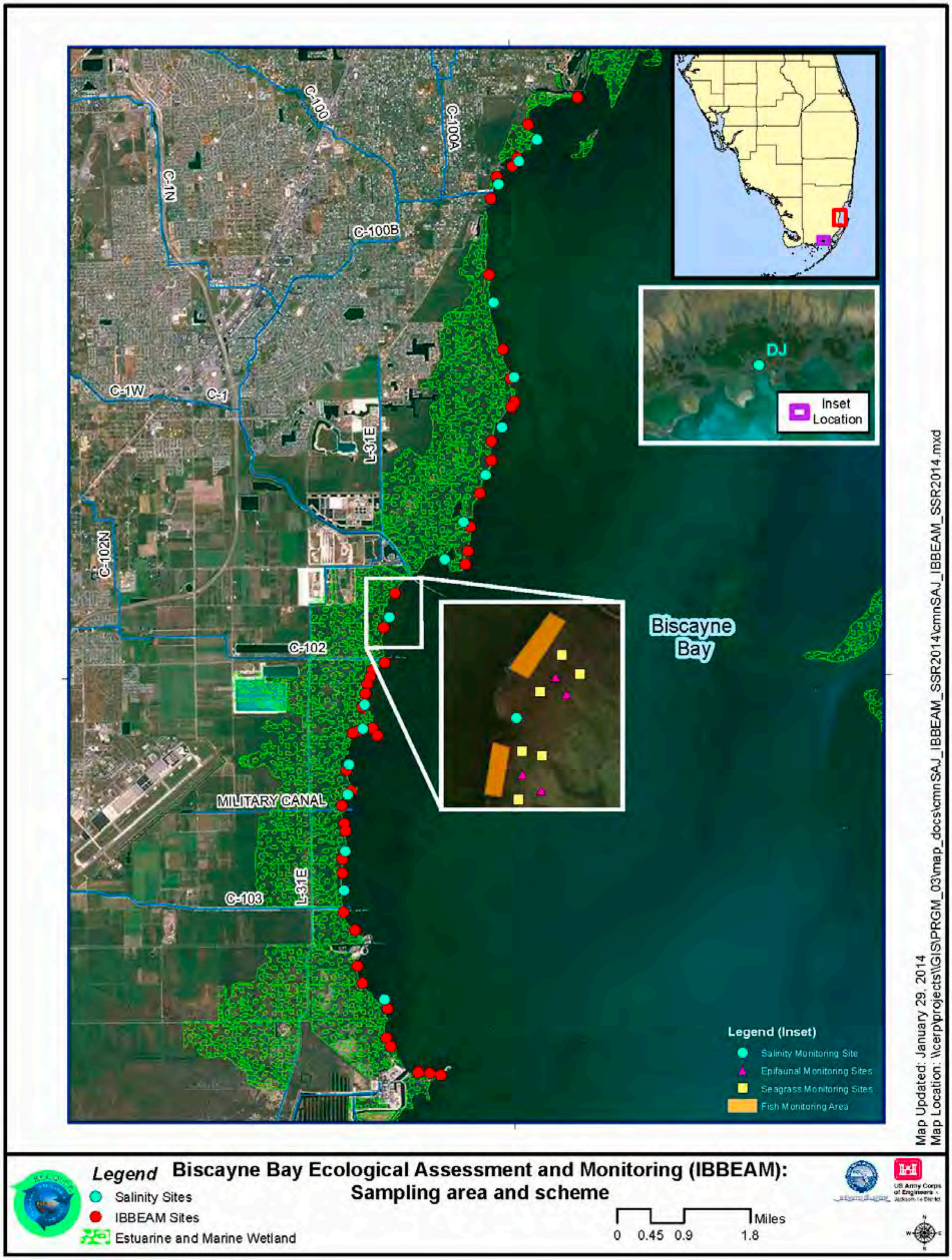


Figure 7-2. IBBEAM sampling area and sampling scheme.

There are 47 co-located biotic sampling sites. Mangrove sites start at the mangroves and extend 30 meters (m) along the shore. Associated epifaunal and SAV sites are within 50 m of the shoreline at < 1 m of depth. The data collected at each site include SAV (identity, percent cover of seagrass and macroalgae, and seagrass canopy height), epifauna (identity, abundance, and size of all fish, decapods, and echinoderms captured), and mangrove fish (identity, abundance, and size-structure of all fishes observed). These basic data are used to calculate community-based metrics such as taxonomic richness and dominance and taxon-specific abundance metrics (occurrence, concentration, and density). Biological sampling takes place within the dry (November–April) and wet (May–October) seasons, specifically in January–March and July–September. Details of data collection and methods are provided in the 2013 IBBEAM Annual Report (Bellmund et al. 2013).

Results

Salinity Indices

Three salinity regime indices were used to characterize salinity: (1) mesohaline index (proportion of salinity observations ≤ 20), (2) hypersalinity index (proportion of salinity observations > 38), (3) salinity variability index (proportion of days where salinity range is > 5 within a day). For purposes of this section, mesohaline refers to a 0 to 20 range and hypersaline is defined as > 38. Indices 1, 2 and 3 were aggregated using the geometric mean to yield a salinity regime suitability index (SRSI).

The salinity regime at a site in Joe Bay in Northeast Florida Bay (site DJ) was used as a restoration target against which all nearshore Biscayne Bay salinity data were compared using the above indices. The present day salinity regime at this reference site was considered a surrogate for the salinity regimes that existed along Biscayne Bay's western shoreline prior to construction and operation of the coastal canal system. The IBBEAM Team believes salinity conditions at this selected reference site appear to provide a realistic view of conditions that might be associated with the performance measures (http://www.evergladesplan.org/pm/recover/perf_se.aspx). In addition to having a long-term high-resolution water quality data set and desired salinity conditions, the DJ station (25.21665N and -80.55563W) has biotic (seagrass, fish, and invertebrate) data that demonstrate the floral and faunal communities associated with such conditions.

Index values were color-coded green, yellow, and red to signify optimal, adequate, and unsuitable conditions, respectively. Color blends (i.e., between red, yellow, and green) were used to highlight differences in index values in space and time. The color scaling was such that optimal values (green) were the seasonal mean values at the DJ site; unsuitable (red) was the minimum (or maximum, depending on the metric) value in the entire matrix; and adequate (yellow) was the mid-way (50%) value between optimal and unsuitable. Therefore, color blends of green, yellow, and red were used, as appropriate, for intermediate index values.

The salinity indices differed spatially and seasonally (**Figure 7-3**, and **Tables 7-1** through **7-4**). The mesohaline index revealed that none of the sampled areas are optimal in terms of the preferred water quality restoration characteristics identified by RECOVER for nearshore Western Biscayne Bay. The best performing sites, according to the index, were in the area south of Black Point, primarily between Military Canal and the C-103 Canal; these sites had near-optimal mesohaline conditions throughout the entire WY 2006 and optimal mesohaline conditions during the dry season in WY 2012 and wet season in WY 2013. Hypersaline events occurred (**Figure 7-3** and **Table 7-2**), but were mainly confined to the wet seasons (May–October periods) of WY 2005 and WY 2012. The worst conditions (see brightest red), according to the hypersaline index, were recorded at site D2 in the wet season (May–October) of WY 2012.

The spatial pattern of the mean annual variability index differed slightly between wet and dry seasons (May–October and November–April) (**Figure 7-3** and **Table 7-3**). Some sites around and south of Black Point experienced more than 20% to 40% of days with a > 5 within-day salinity change in the wet season (May–October) (**Figure 7-3** and **Table 7-3**). Wet season canal operations and rainfall likely were responsible for this high variability. In contrast, some sites north of Black Point had stable salinity through some years. This area has only one canal with very low flow and receives only rainfall as an alternative source of fresh water. Water quality site 14, located nearshore off Mowry Canal, experienced within-day salinity variability > 5 throughout all water years and seasons and was obviously heavily and directly impacted by canal operations. The SRSI values, which are composites of the mesohaline, variability, and hypersaline indices, indicated poor salinity habitat suitability at the northern and southern extremes of the study domain during the wet season (May–October) and greater suitability for the southern part of the bay during the dry season (November–April) (**Figure 7-3** and **Table 7-4**). The poor composite index was likely due to the lack of mesohaline habitat and the high variability at some sites.

An initial analysis of the salinity data recorded by IBBEAM since 2004 suggests that, compared to conditions recorded in the Florida Bay reference site, salinity environments of Western Biscayne Bay are presently not adequate to support the target biological communities. While some limited areas south of Black Point do experience near-desired mesohaline conditions, the persistence of these conditions is limited and highly restricted spatially. These inadequate conditions highlight the need for future modifications to the water delivery system to achieve CERP goals. The salinity metrics developed here provide tools to analyze and present a complex and dynamic data set in an efficient manner. Future efforts will evaluate these metrics further and relate the salinity metrics with the biological data being collected concurrently.

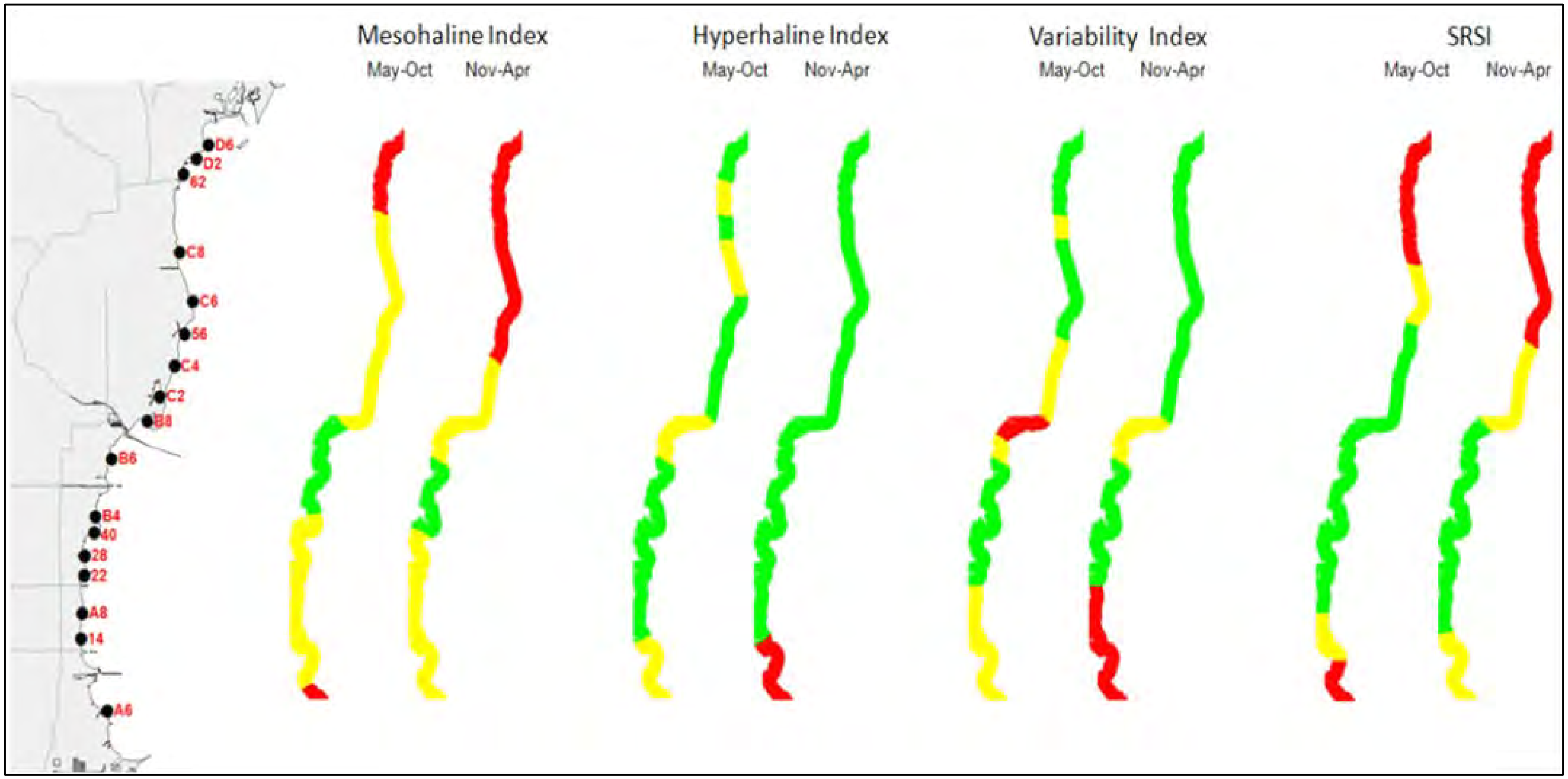


Figure 7-3. Salinity indices averaged by season for 2004–2011, November–April (dry season) and May–October (wet season) along the Biscayne Bay shoreline. The colored ribbon represents salinity within 100 m of shore. Red denotes poor conditions (not suitable), green denotes conditions equivalent to target conditions (optimal), and yellow denotes conditions in between red and green (adequate).

Table 7-1. Mesohaline index by water year (WY), calendar year (CY), and season (Wet = May–October; Dry = November–April). Mesohaline index = proportion of salinity observations ≤ 20. Red denotes poor conditions (not suitable), green denotes conditions equivalent to target conditions (optimal), and yellow denotes conditions in between red and green (adequate). Cells not color-coded (i.e., white) represent absent or incomplete (gray values) data sets.

WY	2004	2005		2006		2007		2008		2009		2010		2011		2012		2013		Mean	
CY	2004	2004	2005	2005	2006	2006	2007	2007	2008	2008	2009	2009	2010	2010	2011	2011	2012	2012	2013		
Month	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr
Site/Season	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**
D6														*0.000*	0.000	0.016	0.026	0.199	*0.000*	0.108	0.013
D2														*0.000*	0.000	0.012	0.035	0.278	*0.000*	0.145	0.017
62	*0.004*	0.122	0.002	0.351	0.002	0.206	0.005	0.196	0.003	0.154	0.003	0.081	0.011	0.160	0.006	0.039	0.072	0.474	*0.000*	0.198	0.013
C8														*0.203*	0.000	0.045	0.072	0.509	*0.000*	0.277	0.036
C6														*0.264*	0.020	0.075	0.063	0.725	*0.000*	0.400	0.041
56	*0.098*	0.226	0.062	0.588	0.000	0.358	0.008	0.492	0.013	0.334	0.008	0.190	0.031	0.298	0.011	0.175	0.089	0.776	*0.000*	0.382	0.028
C4														0.394	0.039	0.237	0.151	0.811	*0.000*	0.481	0.095
C2														*0.480*	0.078	0.306	0.191	0.872	*0.018*	0.589	0.134
B8														*0.465*	0.089	0.153	0.274	0.903	*0.058*	0.528	0.182
B6														0.847	0.204	0.446	0.528	0.966	*0.544*	0.753	0.366
B4														*0.571*	0.215	0.338	0.673	0.942	*0.397*	0.640	0.444
40	*0.278*	0.345	0.306	0.719	0.511	0.529	0.372	0.483	0.302	0.526	0.419	0.370	0.366	0.666	0.260	*0.481*	0.659	0.947	*0.337*	0.573	0.399
28	*0.233*	0.293	0.356	0.658	0.462	0.471	0.309	0.435	0.281	0.475	0.364	0.304	0.292	0.643	0.205	0.316	0.535	0.901	*0.295*	0.500	0.351
22	*0.000*	0.269	0.323	0.598	0.389	0.466	0.311	0.432	0.278	0.587	0.323	0.335	0.299	0.703	0.193	0.322	0.561	0.868	*0.375*	0.509	0.335
A8														0.552	0.147	0.272	0.516	0.723	*0.185*	0.516	0.331
14	*0.206*	0.229	0.279	0.580	0.387	0.444	0.314	0.532	0.387	0.513	0.373	0.331	0.307	0.564	0.173	0.273	0.651	0.769	*0.272*	0.471	0.359
A6														*0.151*	0.051	0.098	0.287	0.295	*0.063*	0.196	0.169
DJ														0.984	0.704	0.768	0.314			0.876	0.509

Table 7-2. Hyperhaline index by water year (WY), calendar year (CY), and season (Wet = May–October; Dry = November–April). Hypersaline index = proportion of salinity observations > 38. Red denotes poor conditions (not suitable), green denotes conditions equivalent to target conditions (optimal), and yellow denotes conditions in between red and green (adequate). Cells not color-coded (i.e., white) represent absent or incomplete (gray values) data sets.

WY	2004	2005		2006		2007		2008		2009		2010		2011		2012		2013		Mean	
CY	2004	2004	2005	2005	2006	2006	2007	2007	2008	2008	2009	2009	2010	2010	2011	2011	2012	2012	2013		
Month	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr
Site/Season	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry
D6														0.000	0.059	0.202	0.000	0.000	0.000	0.101	0.030
D2														0.000	0.011	0.397	0.000	0.000	0.000	0.198	0.006
62	0.000	0.098	0.000	0.023	0.000	0.000	0.000	0.000	0.001	0.000	0.003	0.059	0.000	0.010	0.012	0.255	0.000	0.000	0.000	0.049	0.002
C8														0.000	0.038	0.263	0.000	0.000	0.000	0.132	0.019
C6														0.000	0.053	0.276	0.000	0.000	0.000	0.138	0.027
56	0.011	0.390	0.029	0.034	0.000	0.000	0.008	0.000	0.002	0.000	0.082	0.123	0.000	0.000	0.031	0.270	0.000	0.000	0.000	0.091	0.019
C4														0.000	0.047	0.255	0.000	0.000	0.000	0.085	0.023
C2														0.000	0.028	0.240	0.000	0.000	0.000	0.120	0.014
B8														0.000	0.032	0.256	0.000	0.000	0.000	0.128	0.016
B6														0.000	0.025	0.366	0.000	0.000	0.000	0.122	0.012
B4														0.000	0.054	0.306	0.000	0.000	0.000	0.153	0.027
40	0.000	0.219	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.022	0.031	0.122	0.000	0.000	0.062	0.143	0.000	0.000	0.000	0.045	0.012
28	0.000	0.255	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.076	0.000	0.000	0.049	0.235	0.000	0.000	0.000	0.063	0.006
22	0.000	0.234	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.028	0.079	0.000	0.000	0.017	0.355	0.000	0.000	0.000	0.074	0.006
A8														0.000	0.062	0.260	0.000	0.000	0.000	0.087	0.031
14	0.000	0.267	0.000	0.023	0.000	0.000	0.000	0.000	0.000	0.019	0.046	0.094	0.000	0.002	0.076	0.362	0.000	0.000	0.000	0.085	0.015
A6														0.000	0.122	0.384	0.066	0.000	0.000	0.192	0.094
DJ														0.000	0.000	0.000	0.120			0.000	0.060

Table 7-3. Variability index by water year (WY), calendar year (CY), and season (Wet = May–October; Dry = November–April). Variability proportion of observations where daily salinity range > 5. Red denotes poor conditions (not suitable), green denotes conditions equivalent to target conditions (optimal), and yellow denotes conditions in between red and green (adequate). Cells not color-coded (i.e., white) represent absent or incomplete (gray values) data sets.

WY	2004	2005		2006		2007		2008		2009		2010		2011		2012		2013		Mean	
CY	2004	2004	2005	2005	2006	2006	2007	2007	2008	2008	2009	2009	2010	2010	2011	2011	2012	2012	2013		
Month	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr
Site/Season	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**
D6														0.000	0.011	0.043	0.060	0.163	0.115	0.103	0.036
D2														0.000	0.006	0.027	0.055	0.120	0.000	0.073	0.030
62	0.188	0.293	0.089	0.332	0.188	0.281	0.227	0.266	0.033	0.174	0.105	0.234	0.127	0.408	0.177	0.130	0.082	0.359	0.000	0.275	0.129
C8														0.116	0.000	0.049	0.088	0.174	0.000	0.111	0.044
C6														0.137	0.077	0.103	0.071	0.377	0.000	0.240	0.074
56	0.057	0.147	0.055	0.223	0.033	0.060	0.083	0.179	0.022	0.179	0.033	0.022	0.066	0.163	0.056	0.125	0.071	0.239	0.000	0.149	0.053
C4														0.072	0.000	0.087	0.044	0.076	0.000	0.079	0.022
C2														0.532	0.188	0.402	0.253	0.435	0.049	0.418	0.220
B8														0.287	0.028	0.239	0.170	0.353	0.066	0.296	0.099
B6														0.268	0.072	0.228	0.214	0.299	0.082	0.265	0.143
B4														0.290	0.081	0.212	0.209	0.207	0.115	0.209	0.145
40	0.014	0.087	0.024	0.109	0.011	0.095	0.006	0.049	0.022	0.114	0.022	0.078	0.033	0.082	0.022	0.102	0.176	0.082	0.049	0.087	0.039
28	0.014	0.076	0.127	0.196	0.043	0.070	0.061	0.082	0.038	0.136	0.055	0.038	0.088	0.207	0.387	0.484	0.082	0.103	0.000	0.154	0.110
22	0.067	0.120	0.114	0.152	0.061	0.130	0.055	0.098	0.093	0.231	0.061	0.130	0.072	0.473	0.193	0.190	0.121	0.212	0.026	0.193	0.096
A8														0.120	0.077	0.098	0.055	0.190	0.049	0.136	0.066
14	0.239	0.310	0.239	0.416	0.503	0.554	0.326	0.565	0.610	0.543	0.344	0.467	0.436	0.567	0.232	0.413	0.408	0.620	0.410	0.495	0.387
A6														0.120	0.077	0.098	0.055	0.190	0.049	0.144	0.066
DJ														0.021	0.011	0.000	0.011			0.010	0.011

Table 7-4. SRSI by water year (WY), calendar year (CY), and season (Wet = May–October; Dry = November–April). This index is a composite of the mesohaline, hypersaline and variability indices presented above. Red denotes poor conditions (not suitable), green denotes conditions equivalent to target conditions (optimal), and yellow denotes conditions in between red and green (adequate). Cells not color-coded (i.e., white) represent absent or incomplete (gray values) data sets.

WY	2004	2005		2006		2007		2008		2009		2010		2011		2012		2013		Mean	
CY	2004	2004	2005	2005	2006	2006	2007	2007	2008	2008	2009	2009	2010	2010	2011	2011	2012	2012	2013		
Month	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr	May-Oct	Nov-Apr
Site/Season	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry
D6														*0.000*	0.000	0.232	0.292	0.550	*0.000*	0.391	0.146
D2														*0.000*	0.000	0.193	0.321	0.626	*0.000*	0.409	0.160
62	*0.147*	0.427	0.117	0.612	0.123	0.529	0.158	0.524	0.144	0.503	0.135	0.388	0.211	0.454	0.166	0.294	0.404	0.672	*0.000*	0.489	0.182
C8														*0.564*	0.000	0.316	0.403	0.749	*0.000*	0.533	0.202
C6														*0.611*	0.260	0.365	0.387	0.767	*0.000*	0.566	0.323
56	*0.450*	0.490	0.385	0.761	0.048	0.696	0.196	0.739	0.232	0.650	0.194	0.546	0.306	0.629	0.213	0.482	0.436	0.839	*0.000*	0.648	0.251
C4														0.715	0.333	0.544	0.524	0.908	*0.070*	0.722	0.429
C2														*0.608*	0.394	0.518	0.522	0.790	*0.257*	0.654	0.458
B8														*0.729*	0.403	0.448	0.595	0.863	*0.386*	0.656	0.499
B6														0.740	0.578	0.553	0.810	0.907	*0.706*	0.733	0.694
B4														*0.846*	0.572	0.618	0.759	0.855	*0.798*	0.736	0.666
40	*0.649*	0.627	0.669	0.862	0.796	0.782	0.718	0.771	0.666	0.770	0.735	0.669	0.707	0.849	0.620	*0.718*	0.816	0.955	*0.684*	0.786	0.716
28	*0.612*	0.587	0.677	0.809	0.762	0.760	0.662	0.736	0.646	0.743	0.700	0.646	0.644	0.799	0.492	0.500	0.789	0.931	*0.666*	0.723	0.672
22	*0.000*	0.566	0.659	0.797	0.715	0.740	0.664	0.730	0.631	0.767	0.665	0.645	0.652	0.718	0.535	0.552	0.790	0.881	*0.715*	0.711	0.664
A8														0.739	0.504	0.538	0.740	0.797	*0.553*	0.692	0.622
14	*0.539*	0.488	0.597	0.692	0.578	0.583	0.596	0.614	0.532	0.613	0.616	0.542	0.557	0.625	0.498	0.467	0.728	0.664	*0.543*	0.587	0.588
A6														*0.510*	0.346	0.379	0.633	0.620	*0.391*	0.499	0.489
DJ														0.988	0.886	0.916	0.649			0.952	0.768

Submerged Aquatic Vegetation

Figure 7-4 presents the seasonal occurrence and average percent cover of *Halodule wrightii* and *Thalassia testudinum* from the 47 IBBEAM sites surveyed between 2008 and 2012 at shoreline habitats of western Biscayne Bay (< 50 m from shore). *H. wrightii* is the dominant species in terms of occurrence (found at > 75% of samples in all surveys) and cover, compared to *T. testudinum* that is found in 60% to 80% of samples. Average percent cover of *H. wrightii* was 16.5% (standard deviation [SD] = 17.9) compared to 8.1% (SD = 12.2) for *T. testudinum*. The occurrence of *H. wrightii* is less variable than that of *T. testudinum*, which can oscillate between seasons and years. This pattern is consistent with the life history and stress response of *H. wrightii*, which is considered a pioneer species that thrives in environments exposed to low and variable salinity and increased nutrient inputs. Both species have shown, since 2008, consistent seasonal fluctuations in cover, with peaks in abundance corresponding to the warmer summer months. The exception to this pattern was the lack of seasonal increase in *H. wrightii* cover following the 2010 cold water anomaly (January 2010) when extreme temperatures seem to have affected the physiology of this species. *H. wrightii* has shown a recovery trend since this cold event, but wet season high values are still below those recorded in 2009 prior to this event. Another notable finding is the lack of seasonal increase in cover of both species in summer 2012, when both species registered percent cover values below their dry season values. Continued monitoring will reveal whether this is a transient decline. Considering the significant relationships documented between seagrasses cover and salinity in this project, future analyses will be conducted to determine whether the observed temporal trends correlate directly with antecedent, short-term changes in salinity.

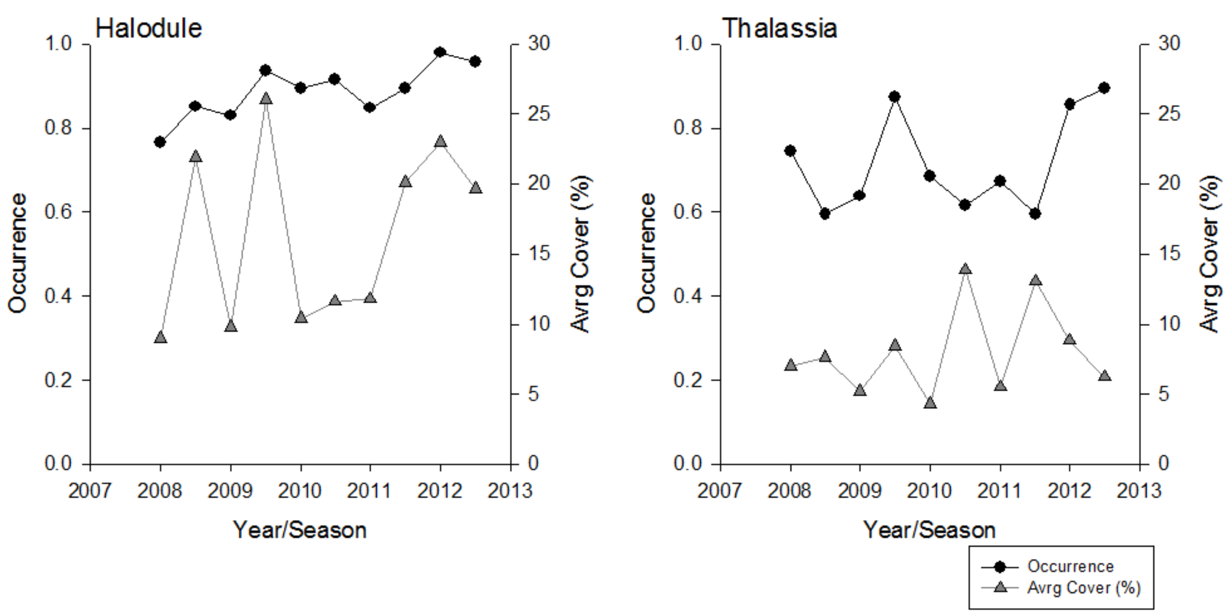


Figure 7-4. Biscayne Bay nearshore occurrence and mean cover of *H. wrightii* (left panel) and *T. testudinum* (right panel) by season and year.

The percent cover data for the two dominant seagrass species, *T. testudinum* and *H. wrightii,* collected from 2008 to 2012 from the 47 sites included in IBBEAM were incorporated into interpolated surface contours shown in **Figure 7-5**. These contours highlight the distribution of higher (yellow to green in **Figure 7-5**) and lower (black to red in **Figure 7-5**) cover for both seagrass species along the western shoreline of Biscayne Bay. Higher cover for *H. wrightii* occurs during the wet season, likely due to the influx of fresh water and nutrients (in addition to increased light availability and higher temperatures). Areas of higher cover (shown as yellow and green in the contours) are concentrated in areas with higher freshwater inflow from canals (Snapper Creek, south of Black Point to Military canal). The opposite pattern appears to be true for *T. testudinum*, where higher cover is more prevalent in areas removed from these freshwater influences, such as areas north of Black Point and south of the C-103 Canal where *T. testudinum* cover was > 15 %.

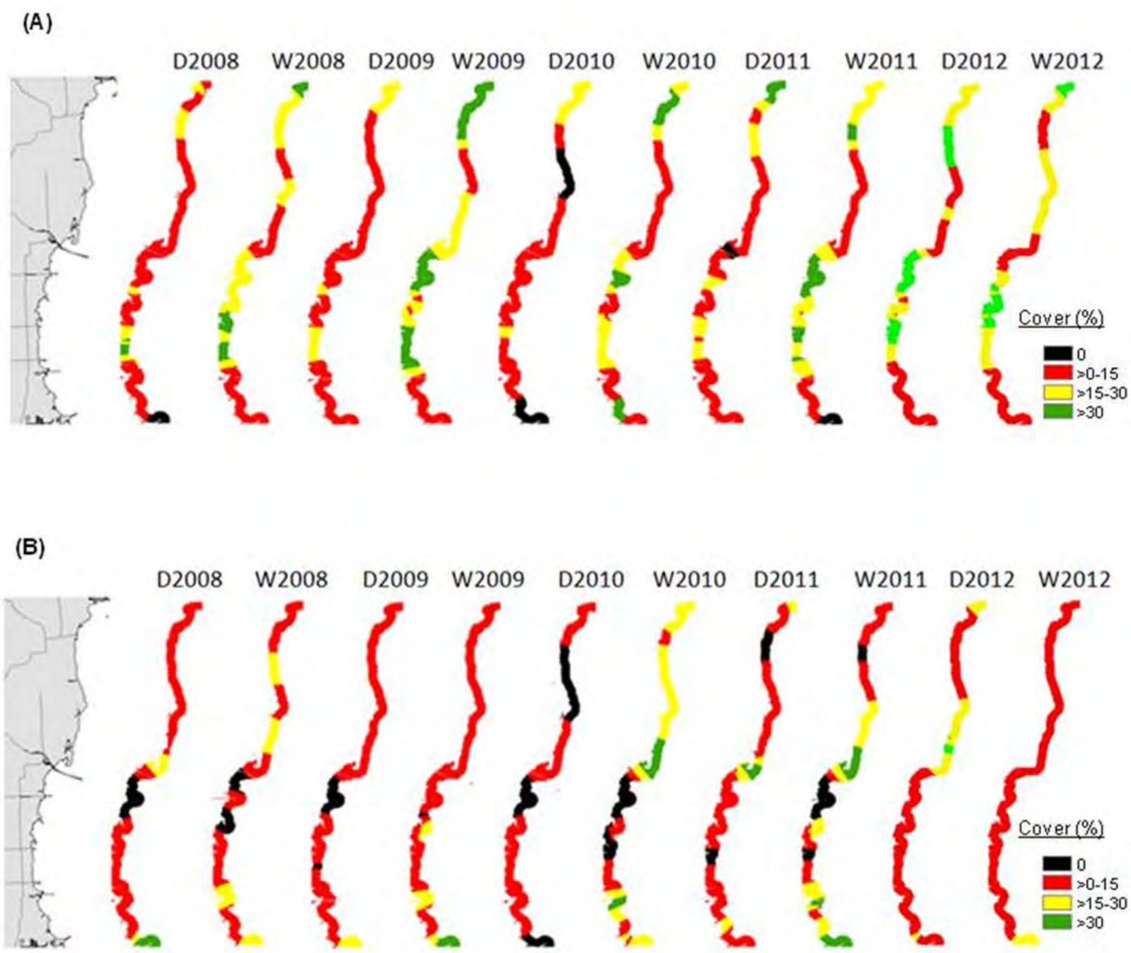


Figure 7-5. Spatial analysis nearshore (< 50 m from shoreline) of (A) *H. wrightii* cover and (B) *T. testudinum* cover. (D = dry season, W = wet season, Calendar Years 2008–2012.)

The contours presented are spatial interpolations and do not take into account sediment depth. Shallow sediments such as commonly found in hard bottom areas may not be suitable for seagrass growth. Also, it is worth noting that the sediment depth influences species composition; *T. testudinum* thrives in deeper sediments compared to *H. wrightii*.

Epifaunal Community

The occurrence and average density of four epifaunal indicator species over time, starting with the dry season of Calendar Year 2005 can be seen in **Figure 7-6**. All species, except goldspotted killifish (*Floridichthys carpio*), showed a seasonal trend, with higher occurrence and density during the dry season compared to the wet season. Conversely, goldspotted killifish abundance indices were higher during the wet season and decreased in the dry season. The goldspotted killifish pattern changed with the 2010 dry season, when south Florida experienced a cold water anomaly that appeared to affect some species. The pattern of higher density in the wet season may have become reestablished with the 2012 wet season. The cold water anomaly did not affect gulf pipefish (*Syngnathus scovelli*), pink shrimp, or grass shrimp (*Palaemonetes* spp.).

Pink shrimp occurrence was relatively stable over the period of record, but densities were highly variable (**Figure 7-6**). No clear response in either metric appears attributable to the 2010 cold water anomaly. The seasonal variation in gulf pipefish and grass shrimp occurrence and density appear closely correlated, with highest values of all metrics occurring in the dry season. Both abundance metrics of gulf pipefish declined by year over most of the monitoring period. No particular pattern in variation of pink shrimp or grass shrimp abundance was evident across sampling years.

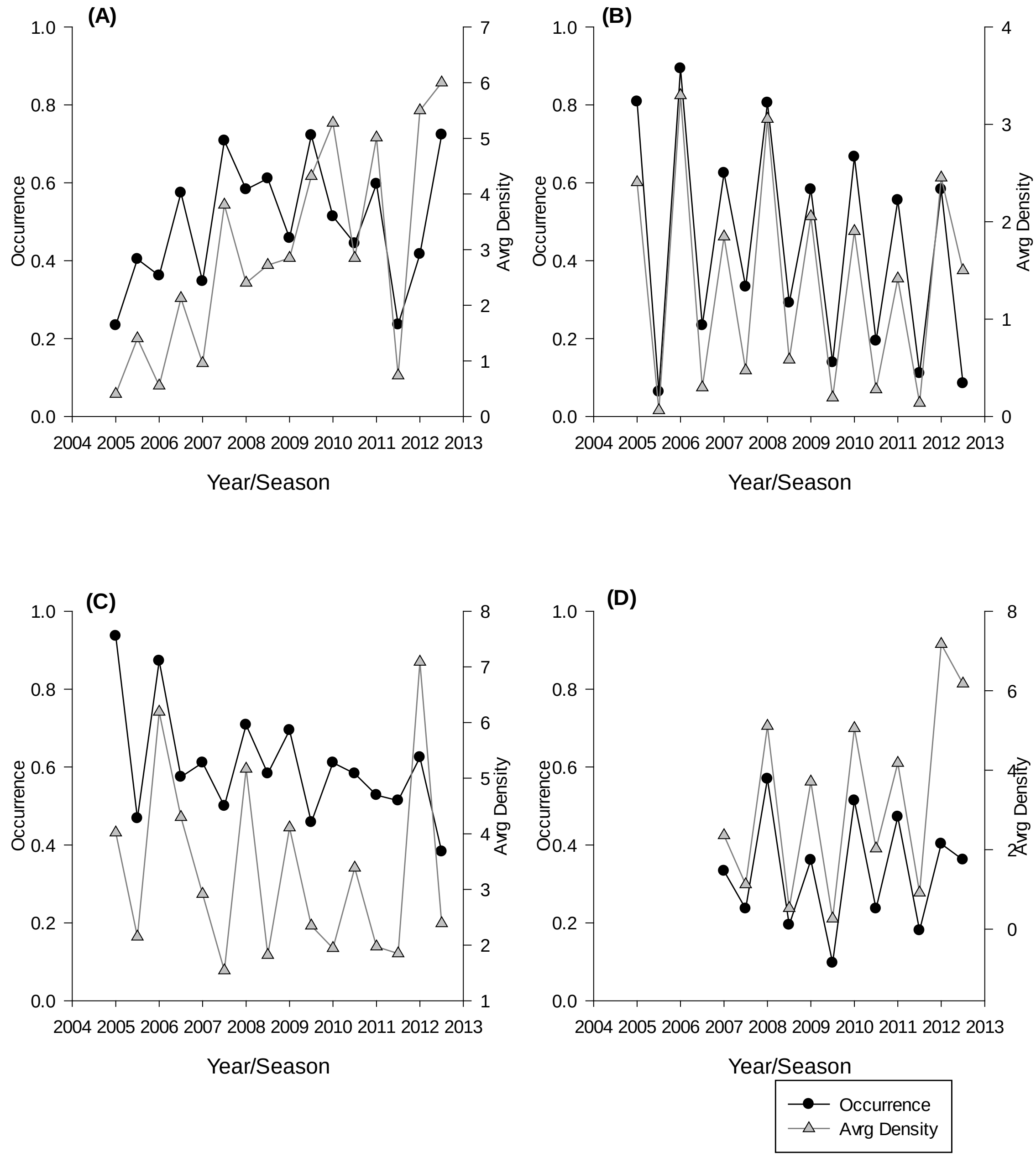
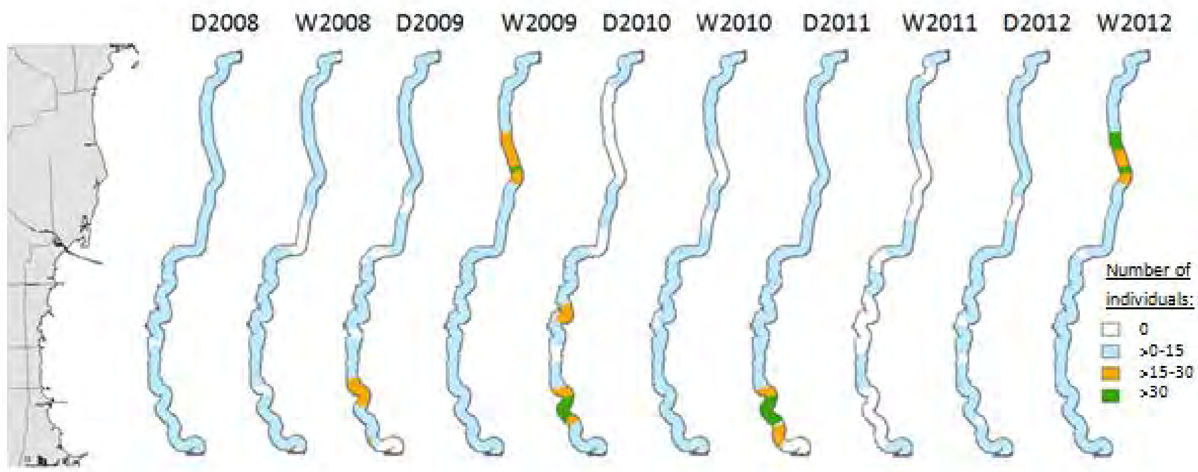


Figure 7-6. Occurrence and mean density of nearshore epifaunal (SAV-associated): (A) goldspotted killifish, (B) gulf pipefish, (C) pink shrimp, and (D) grass shrimp by season and year. (Avrg – Average.)

The spatial distributions of the four epifaunal indicator species are shown by year and season in **Figures 7-7** and **7-8**. In general, goldspotted killifish were concentrated at densities of greater than 15 individuals per 3 square meters (m^2) in certain locations during dry seasons and were absent from many locations during wet seasons (**Figure 7-7**). The 2009 and 2012 wet seasons were notable exceptions, with higher goldspotted killifish densities in an extensive strip of shoreline immediately north of Black Point. Strips of shoreline with no goldspotted killifish were especially extensive in the 2011 wet season, possibly due to the lack of mesohaline salinity during this year. The general pattern of lesser abundance during the wet season was even more pronounced in gulf pipefish, but concentrations of greater than 15 individuals were not seen in this species, even in the dry season.

(A)



(B)

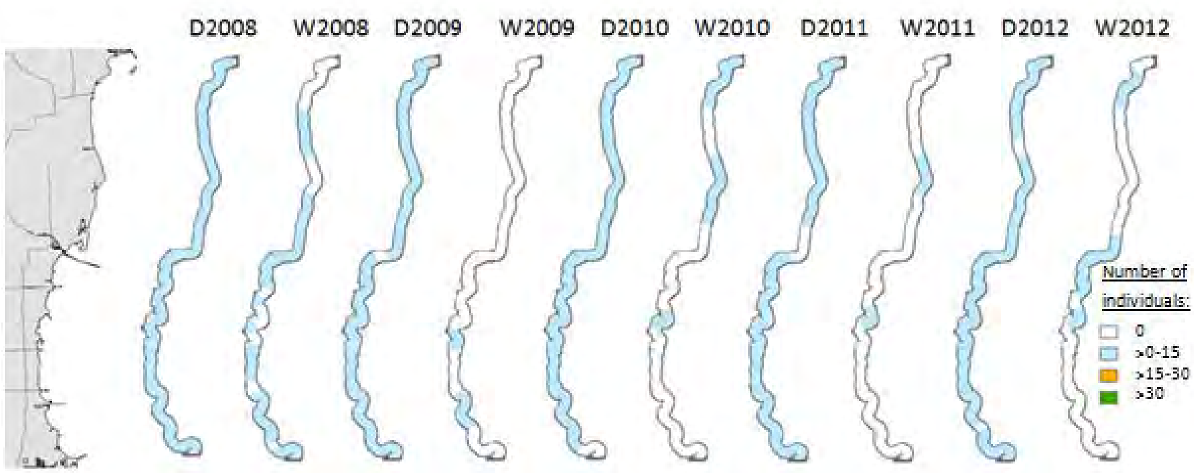


Figure 7-7. Spatial analysis of nearshore epifaunal abundance (SAV-associated, < 50 m from shoreline): (A) goldspotted killifish; and (B) gulf pipefish. (D = dry season, W = wet season, Calendar Years 2008–2012.)

Pink shrimp were found almost everywhere along the shoreline of Biscayne Bay with a count between 0 and 15 per 3 m^2 and little difference between seasons or years (**Figure 7-8**). *Palaemonetes* spp. followed a similar seasonal pattern as the gulf pipefish, with most areas yielding from 1 to 15 individuals per 3 m^2, while, during wet season, most areas were devoid of grass shrimp (**Figure 7-8**).

Assessments on status and trends of epifauna in the other Biscayne Bay regions (North and Central) can be found in the Florida Bay section. The assessment within the Florida Bay section utilizes data collected by the Fish and Invertebrate Assessment Network (FIAN), which is a monitoring program that encompasses and integrates both Biscayne and Florida bays.

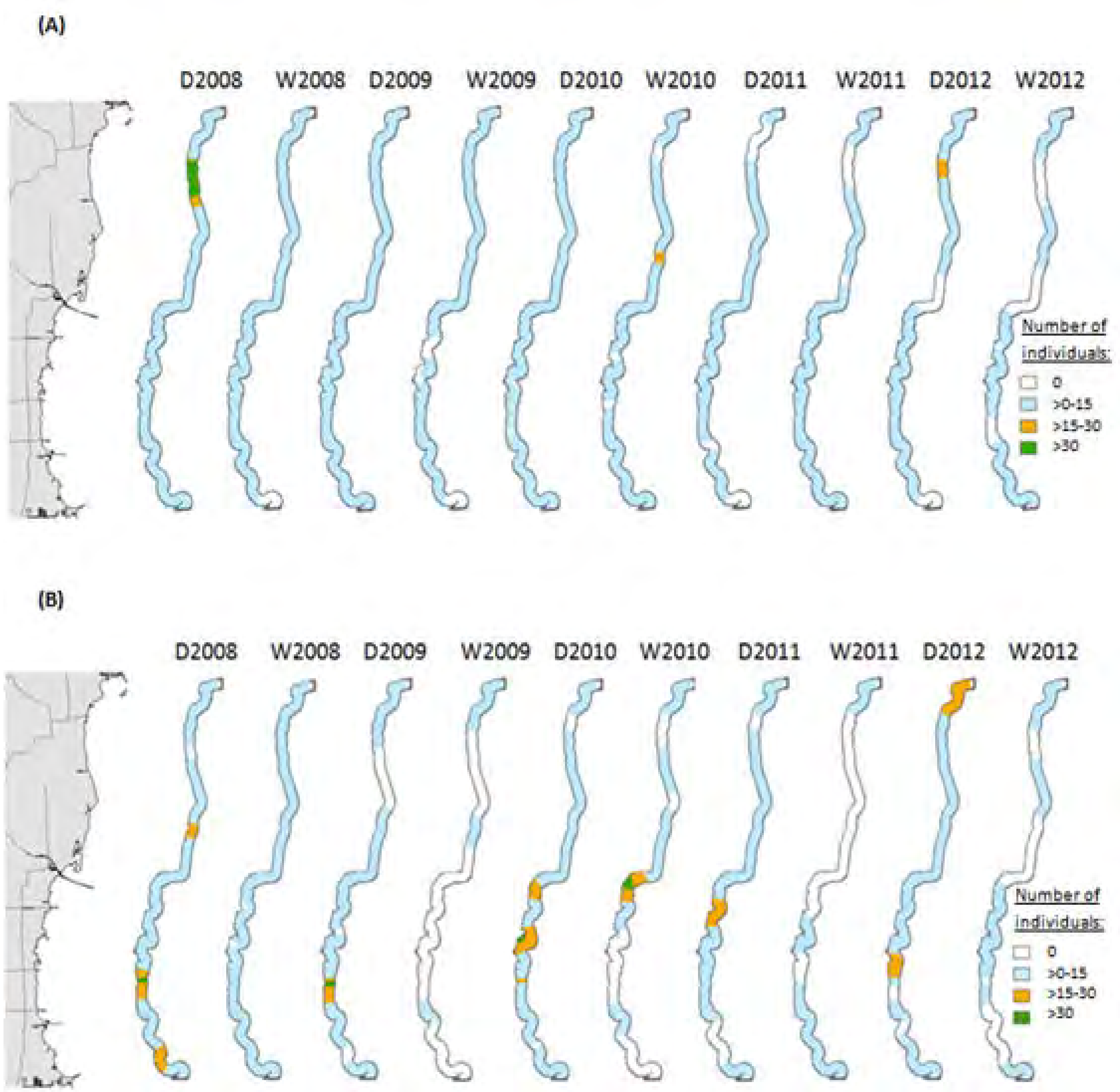


Figure 7-8. Spatial analysis of nearshore epifauna abundance (SAV-associated, < 50 m from shoreline): (A) pink shrimp; and (B) grass shrimp abundance. (D = dry season, W = wet season, Calendar Years 2008–2012.)

Mangrove Fish Community

Figure 7-9 shows the temporal trend of the three mangrove fish species in terms of their occurrence and average density from 1998 to 2012 and complements the spatial results in **Figures 7-10** and **7-11**. Goldspotted killifish in the mangroves occurred most frequently during the dry season and seldom in the wet season, and its density followed the same general pattern (**Figure 7-9**). This is the opposite pattern observed in SAV-associated (epifaunal) goldspotted killifish, possibly indicating seasonal movement between mangrove and seagrass habitats. Both gray snapper (*Lutjanus griseus*) and yellowfin mojarra (*Gerres cincereus*) occurred more frequently and at higher density in the wet season than the dry season. Occurrence and density of yellowfin mojarra appear to have decreased over the period of record (POR), a trend possibly driven by the 2010 cold event.

All three mangrove fish indicator species displayed substantial spatial variation, being concentrated in some areas and absent from others (**Figures 7-10** and **7-11**). Goldspotted killifish, the most abundant of the three species, were concentrated in some locations in dry seasons, with over 100 individuals observed at a site, and absent from extensive areas in wet seasons, especially in 2010, 2011 and 2012. Gray snapper was more abundant in the wet season than in the dry season. Few gray snapper were observed during dry 2010 and 2011. Yellowfin mojarra were also observed in higher abundance during the wet season than in the dry season, except in the 2010 wet season. The decrease of individuals was probably caused by the cold anomaly that happened December 2009–January 2010. Starting in the 2011 wet season, yellowfin mojarra abundance appears to recover, with as many as 20 yellowfin mojarra observed per site.

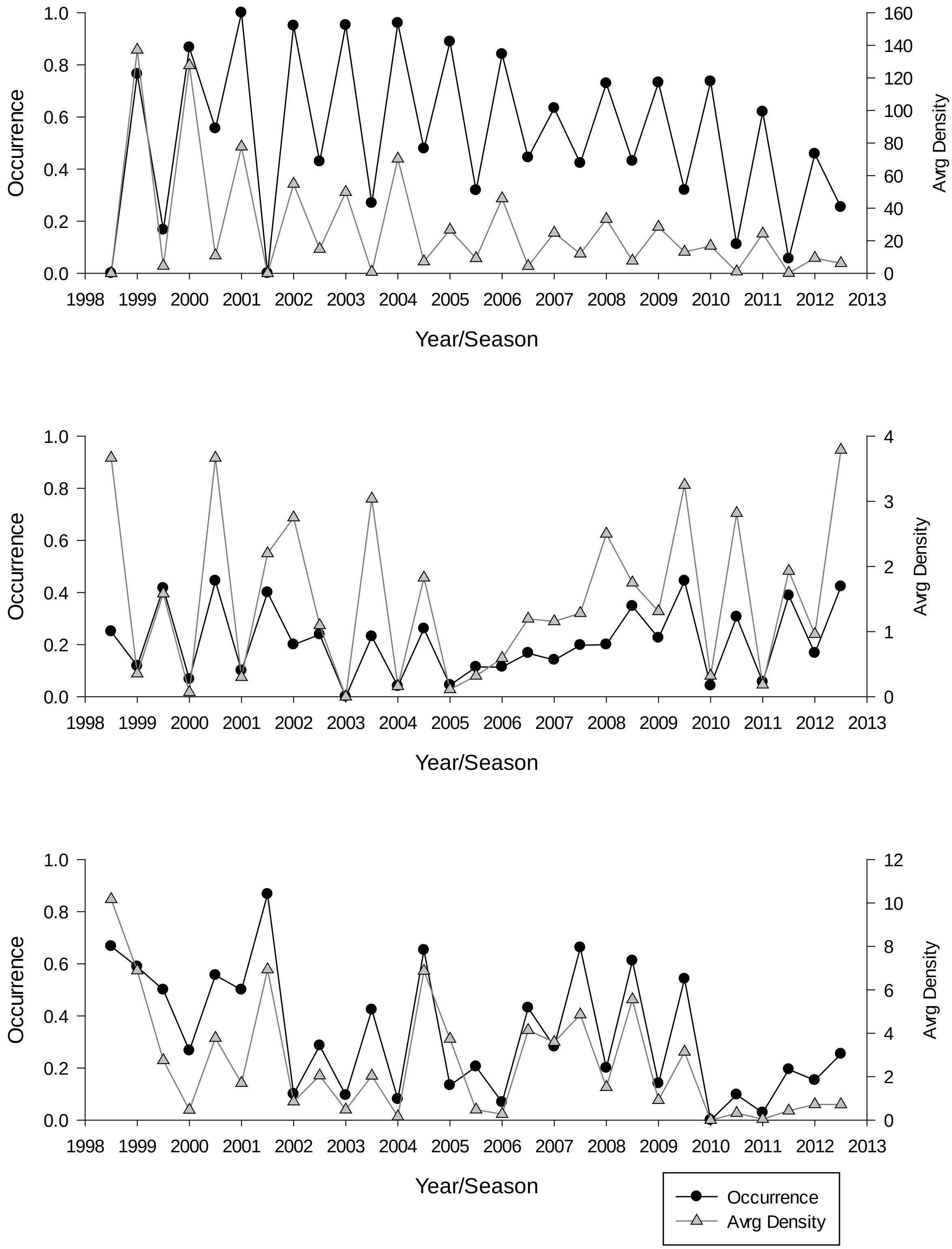


Figure 7-9. Occurrence and mean density of mangrove-associated goldspotted killifish (top panel), gray snapper (middle panel), and yellowfin mojarra (bottom panel) by season and year. (Avrg = Average.)

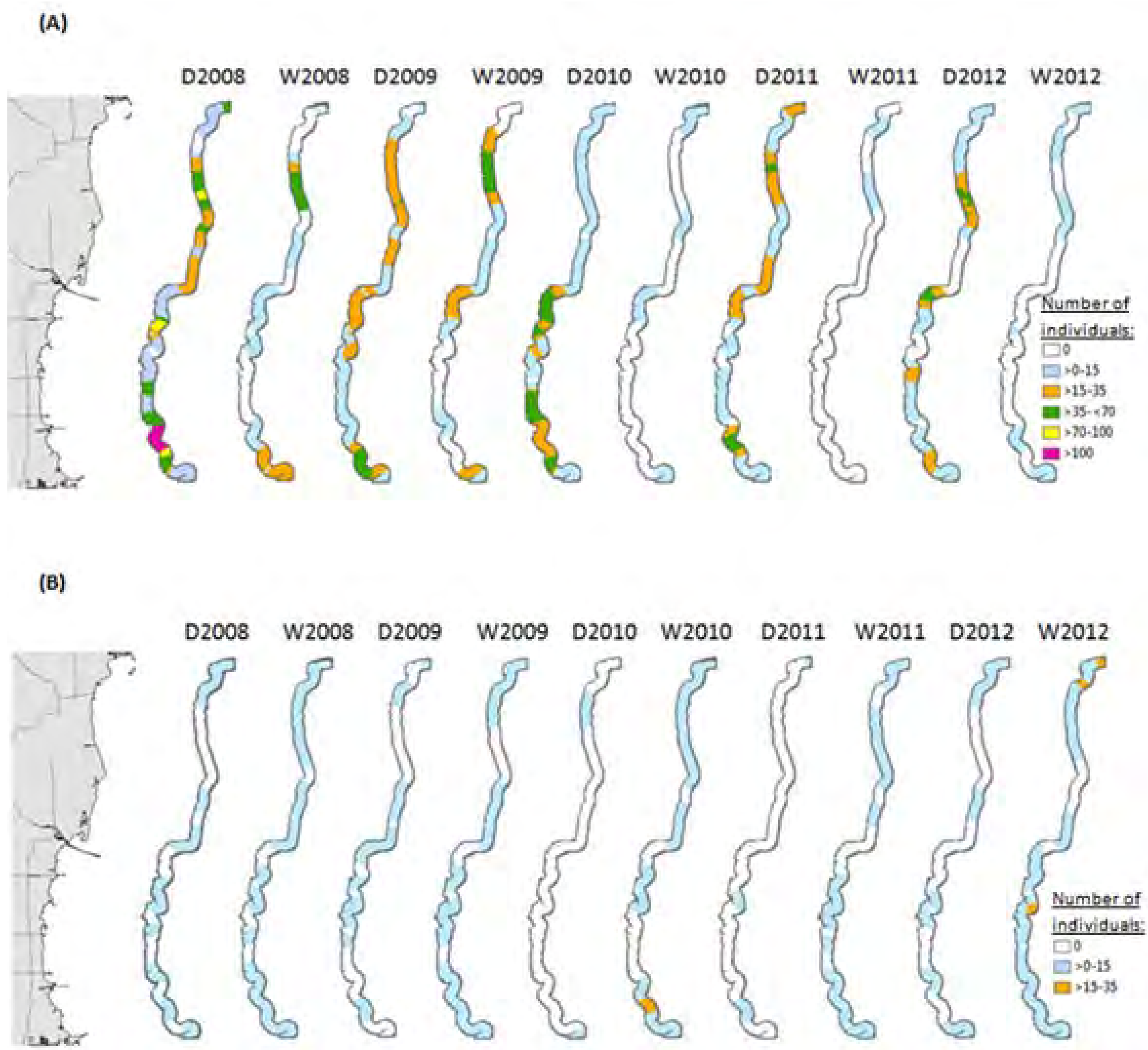


Figure 7-10. Spatial analysis of mangrove-associated (A) goldspotted killifish and (B) gray snapper abundance. (D = dry season, W = wet season, Calendar Years 2008–2012.)

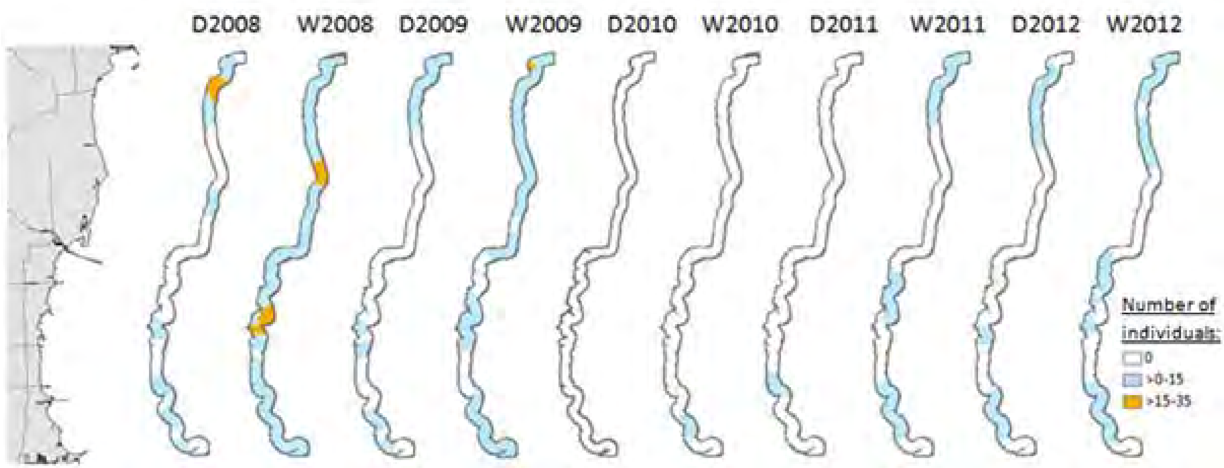


Figure 7-11. Spatial analysis of mangrove-associated yellowfin mojarra abundance. (D = dry season, W = wet season, Calendar Years 2008–2012.)

Ecological Tool Development

Biotic and abiotic data were merged to explore statistical relationships among variables using general linear and logistic regression models. SAV data and mangrove fish data were regressed against salinity, temperature, and depth. Epifaunal data were merged with the SAV data set to test the relationship of epifaunal parameters to different seagrass species. Results indicate that *H. wrightii* occurrence and density are robust indicators of nearshore salinity conditions. Negative linear relationships were documented between *H. wrightii* occurrence and salinity and depth (**Figure 7-12**).

Epifaunal results are consistent with previous analyses showing that gulf pipefish and grass shrimp correlate well with salinity and are robust indicators of nearshore faunal abundance in relation to salinity regime. **Figure 7-13** shows that grass shrimp occurrence and density exhibit a significant negative relationship with salinity and a positive relationship with *H. wrightii* cover. Grass shrimp reach their maximum density at lowest salinity; whereas, gulf pipefish reach their maximum density at intermediate salinity. Both species are expected to increase in abundance when more persistent and widespread mesohaline salinity are reintroduced to the nearshore area, and both are ecologically important in the nearshore ecosystem as energy converters and energy sources for higher trophic level species.

For the mangrove fish community, occurrence and density of mangrove-associated goldspotted killifish and yellowfin mojarra are responsive to nearshore salinity conditions, and thus are indicators of ecological status in Biscayne Bay. **Figure 7-14** shows goldspotted killifish occurrence and density were significantly related to salinity with maxima at approximately 25 . For all models, only statistically significant ($P < 0.05$) model terms were included in final models.

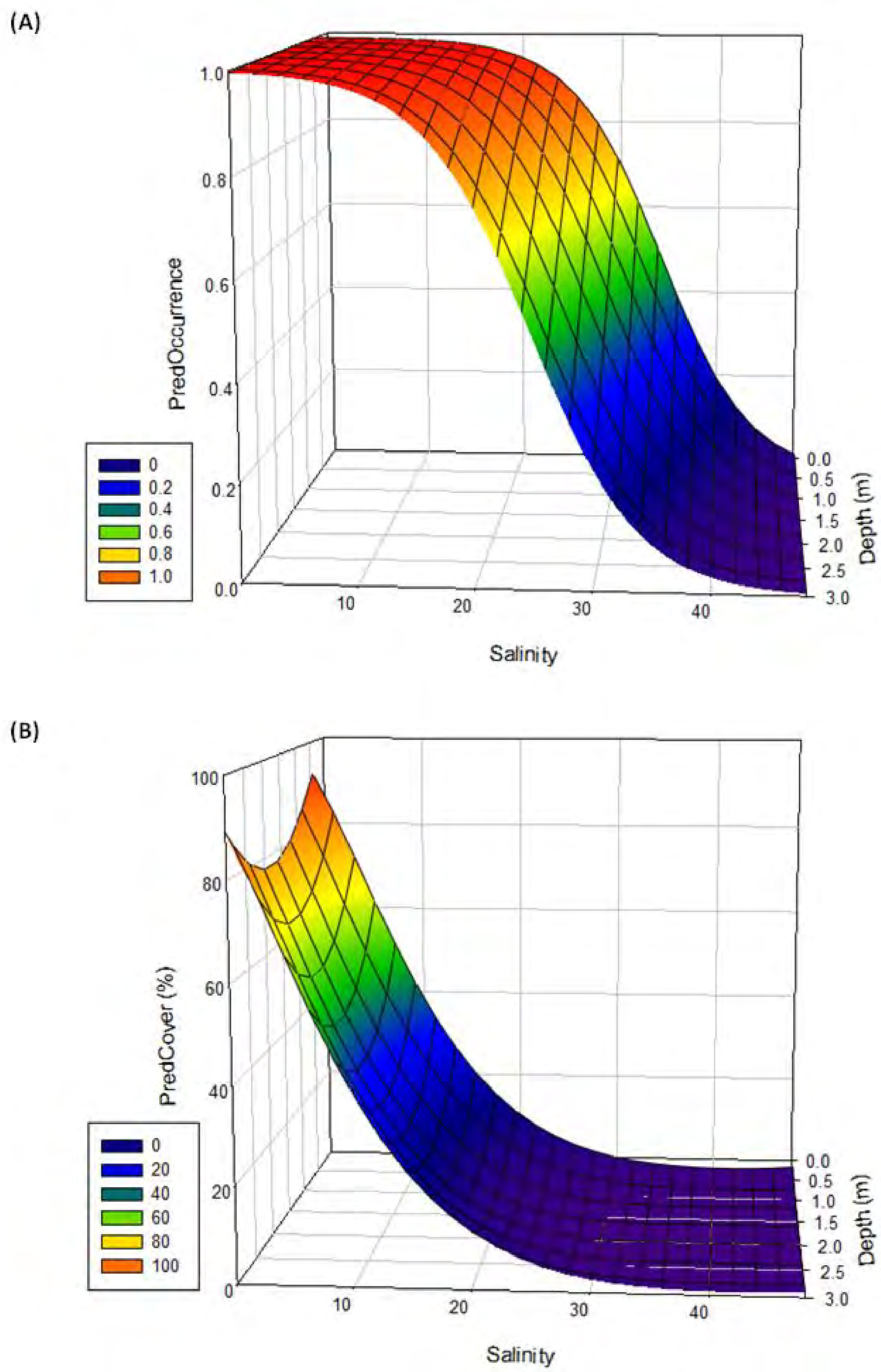


Figure 7-12. Predicted (Pred) response surface plots of the seagrass, *H. wrightii* (A) occurrence and (B) cover (%) with respect to salinity and depth. Only statistically significant ($P < 0.05$) model terms were included in final models. Model goodness of fit indices and coefficients are found in the IBBEAM 2013 Annual Report (Bellmund et al. 2013).

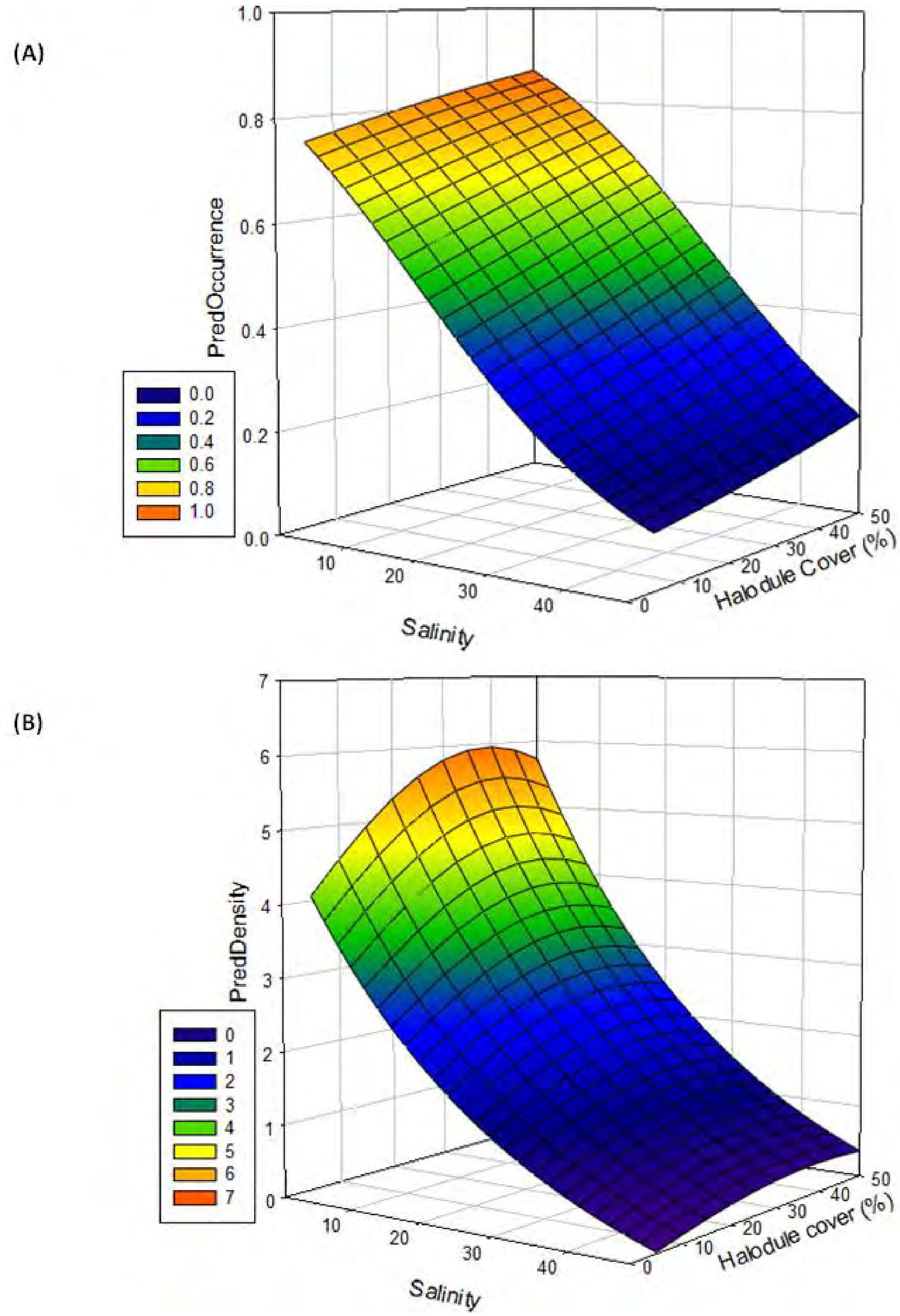


Figure 7-13. Predicted (Pred) response surface plots of grass shrimp (A) occurrence and (B) density plotted with respect to salinity and temperature. Prediction model for occurrence kept *H. wrightii* cover constant at 30%. Only statistically significant ($P < 0.05$) model terms were included in final models. Model goodness of fit indices and coefficients are found in the IBBEAM 2013 Annual Report (Bellmund et al. 2013).

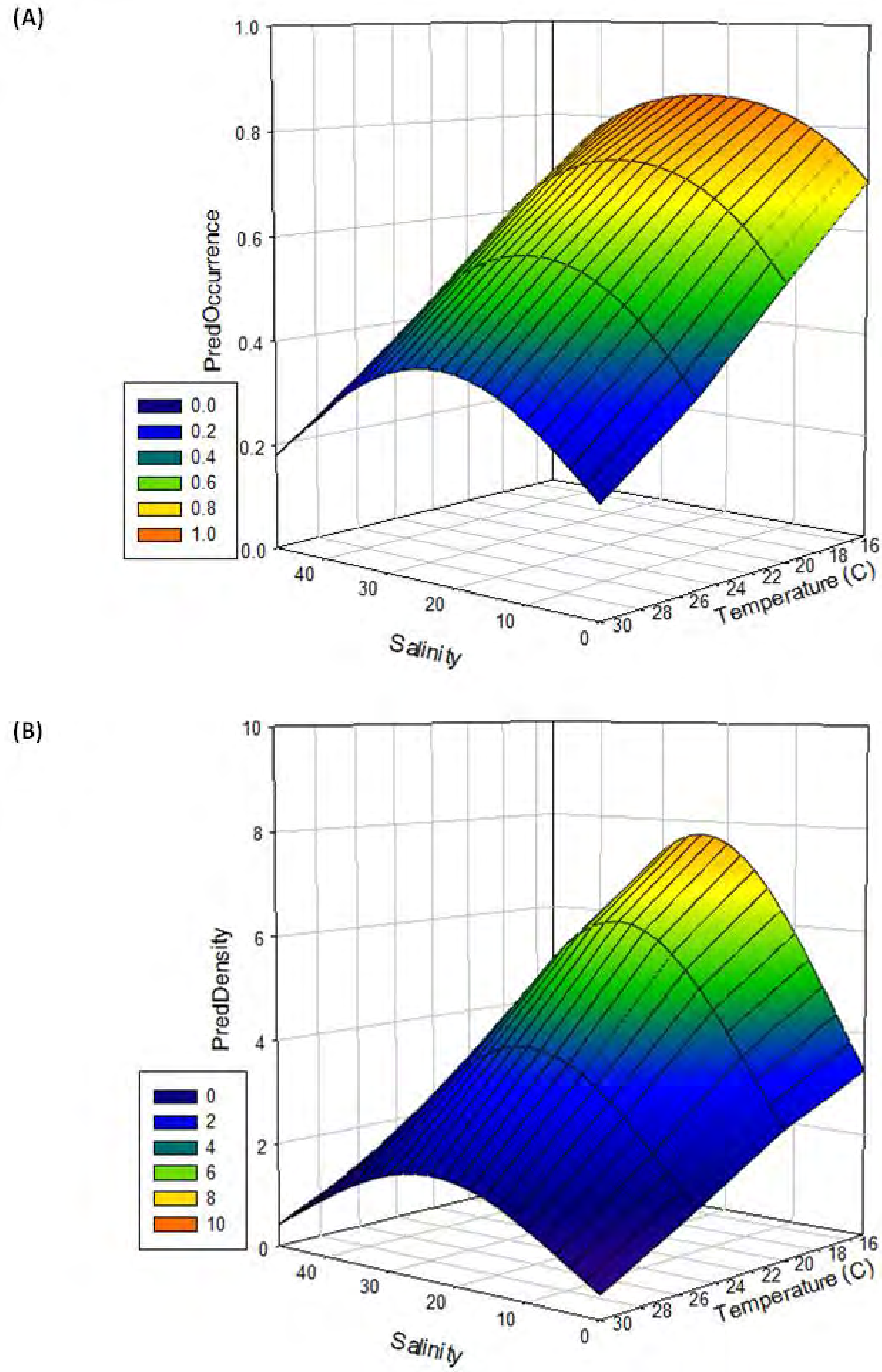


Figure 7-14. Predicted (Pred) response surface plots of mangrove-associated goldspotted killifish (A) occurrence and (B) density with respect to salinity and temperature (degrees Celsius [°C]). Water depth was kept constant at 50 centimeters. Only statistically significant ($P < 0.05$) model terms were included in final models. Model goodness of fit indices and coefficients are found in the IBBEAM 2013 Annual Report (Bellmund et al. 2013).

Results of relationships between other biotic and abiotic data can be found in the IBBEAM 2013 Annual Report (Bellmund et al. 2013). These relationships are critical to the development of habitat suitablity indices and ecological models—valuable tools for assessing and evaluating CERP.

Summary and Conclusions

A summary of IBBEAM results include the following:

- Spatiotemporal analyses suggest impairment in the nearshore salinity regime in comparison to desired conditions, especially in the north part of the study domain and near the end of the period of record. Salinity conditions in Biscayne Bay's nearshore environment fall short of those present at the Florida Bay reference station, especially in terms of the occurrence of mesohaline (≤ 20) salinity conditions needed to restore an estuarine community. Salinity is most variable in the portion of the study area with the largest canal input. Increased freshwater flow to maintain mesohaline salinity, for at least part of the year (i.e., 3 to 5 months), is needed to shift to a more diverse and abundant estuarine flora and fauna.

- The occurrence of *H. wrightii* and *T. testudinum* in the study domain remained largely stable over the POR. Seagrass cover and spatial distribution fluctuated seasonally and varied across years, but no clear temporal trend in seagrass cover was apparent. The data suggest that increased mesohaline conditions, a desired target of CERP activities, will increase overall seagrass abundance and expand habitat suitability for *H. wrightii* as hypothesized and desired.

- Strong seasonality was evident in the overall abundance of the four focal epifaunal taxa (goldspotted killifish, gulf pipefish, pink shrimp, and grass shrimp). A gradual declining trend in gulf pipefish abundance was observed, whereas no clear trend in the other taxa was apparent over the POR. For the epifaunal community, occurrence and density of SAV-associated goldspotted killifish and grass shrimp and density of gulf pipefish are responsive to nearshore salinity conditions and indicators of ecological status.

- Overall abundance patterns of the three focal mangrove fishes (gray snapper, goldspotted killifish, and yellowfin mojarra) indicated substantial species-specific seasonality and spatial variation. Recent abundance declines in yellowfin mojarra appear to coincide with extremely low temperatures in winter 2009–2010. Results indicate that the abundances of goldspotted killifish and yellowfin mojarra will increase if persistent mesohaline conditions are reestablished; the abundance of gray snapper, a fish of high fisheries value, will either remain stable or increase in response to such a salinity regime change.

The IBBEAM project has increased understanding of estuarine hydrologic-ecologic relationships. The development of ecological models, including habitat suitability models, and other predictive tools will be invaluable for understanding, assessing, and forecasting the present and future impacts due to changes in hydrology and water quality of the Everglades watershed as CERP activities and projects proceed. Because the power to detect change is highly dependent on sample size and environmental and biological variability, continued monitoring and data analysis are essential for CERP assessment. Maintaining sufficient sampling is particularly important during the initial post-construction period, when the measureable effects on salinity of upstream changes might be small and hard to detect. Every additional year of data collection substantially increases the power to distinguish CERP effects from non-CERP effects, improves predictive models, and better informs the adaptive management process.

In conclusion, IBBEAM results indicate that Biscayne Bay's current nearshore environment does not constitute the consistent, expansive mesohaline habitat that CERP seeks to reestablish. The bay's littoral zone is currently occupied by floral and faunal species assemblages operating below their productive potential. In part, this deficiency is due to inadequate and unnatural freshwater flows that limit the duration and spatial extent of mesohaline conditions, thus limiting the abundance of species characteristic of south Florida estuaries.

Biscayne Bay Salinity

Data used in this status and trends analysis were provided by Miami-Dade DERM. All stations are reported as depth-averaged monthly salinity. The stations were selected to represent the various areas/subbasins (i.e., Oleta River, Julia Tuttle Basin, 79th St. Causeway Basin, Rickenbacker Basin, etc.) within Biscayne Bay and those that had data covering the full period assessed (May 1999–April 2013). For this report, WY 2000 through WY 2008 (i.e., May 1999 through April 2008) are lumped and compared to WY 2009 through WY 2013. **Figures 7-15** through **7-18** provide box-and-whisker plots showing a comparison of monthly salinity between the assessed periods for four regions of the bay using 9 stations. Note that the region of Biscayne Bay that includes the footprint of the BBCW Project was covered already and is not included here. Also, it should be noted that due to lack of resources, the relationships between salinity patterns, precipitation, and canal discharge were not examined. Therefore, the causation of differences in patterns between WY 2000–WY 2008 and WY 2009–WY 2013 described below are not discussed.

Overall, seasonal salinity patterns throughout Biscayne Bay for WY 2009–WY 2013 have remained relatively similar to WY 2000–WY 2008 patterns. Salinity tends to peak in April/May and is lowest in September/October. In north Biscayne Bay (stations BB04 and BB09), salinity generally ranges between 19 and 37 based on 10th and 90th percentile values (**Figure 7-15**). There are notable monthly differences between WY 2000–WY 2008 and WY 2009–WY 2013. For example, at station BB04, WY 2009–WY 2013 median salinity was about 2 higher in February, April, and August compared to WY 2000–WY 2008; whereas, median salinity was as much as 6 lower during the months of May, June, September, and November for WY 2009–WY 2013 compared to WY 2000–WY 2008. Monthly comparisons at BB09 showed a very different pattern. July median salinity for WY 2009–WY 2013 was about 4 higher than WY 2000–WY 2008; October and November median salinity for WY 2009–WY 2013 was about 2 lower than WY 2000–WY 2008.

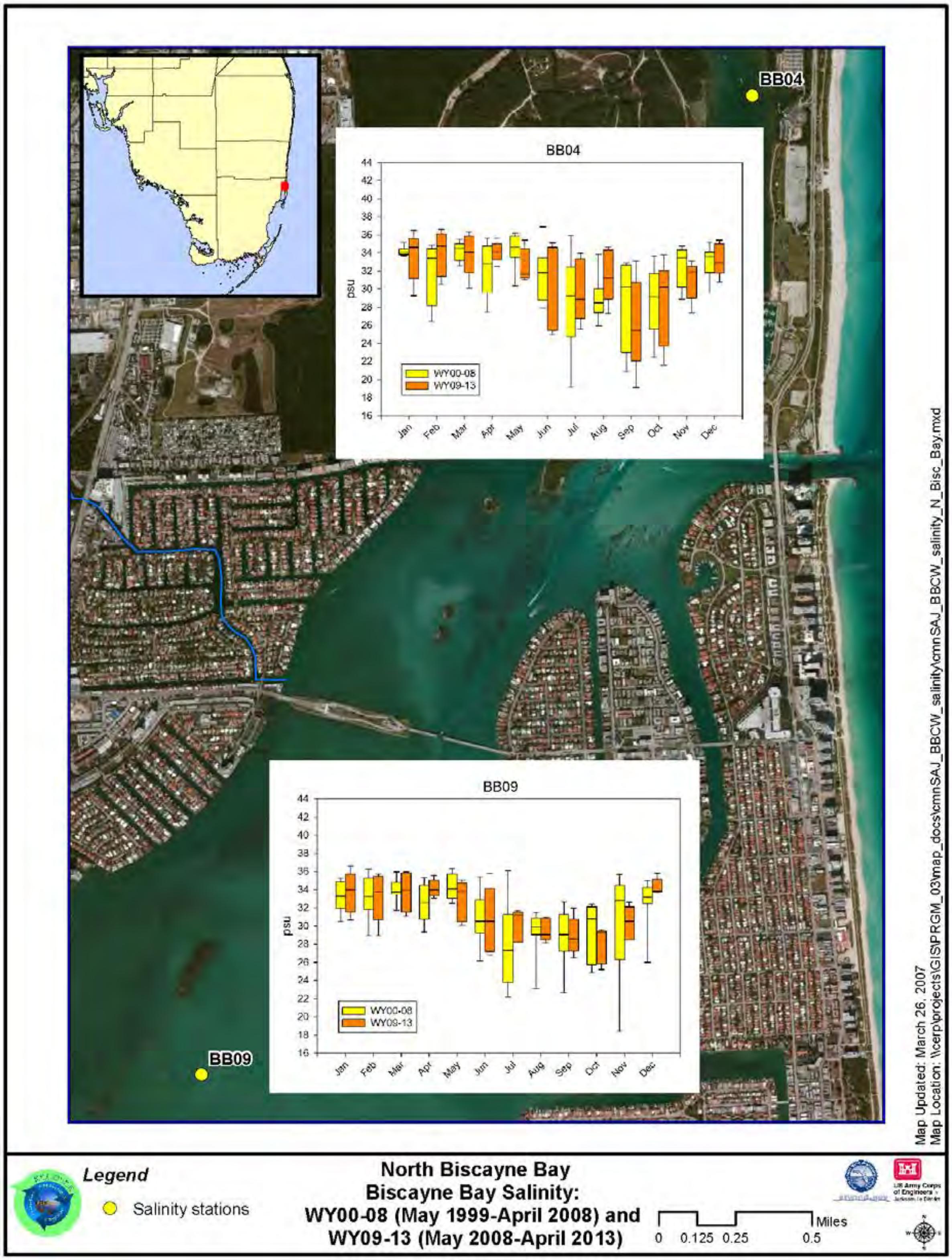


Figure 7-15. WY 2000–WY 2008 and WY 2009–WY 2013 box-and-whisker plots of monthly salinity in North Biscayne Bay.

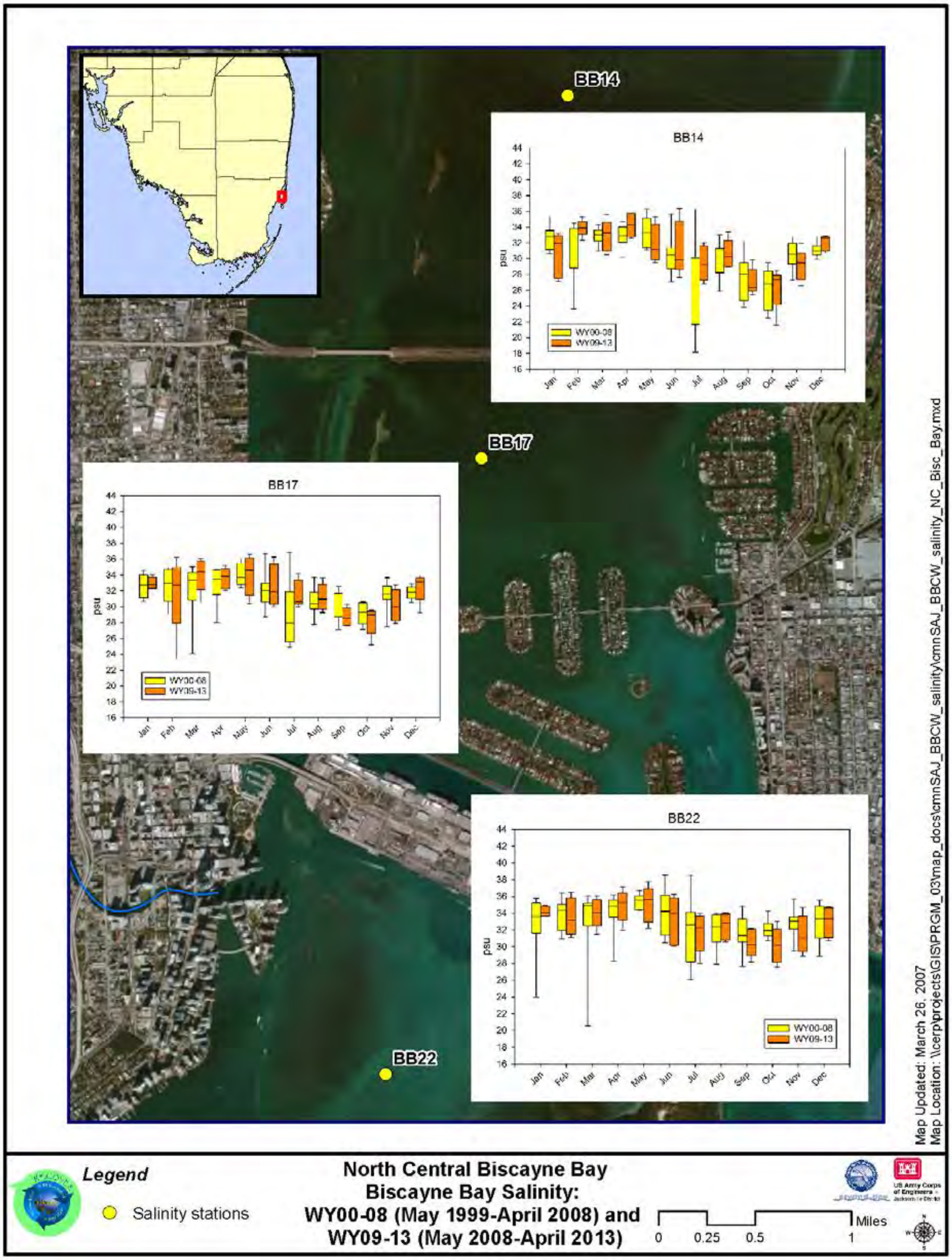


Figure 7-16. WY 2000–WY 2008 and WY 2009–WY 2013 box-and-whisker plots of monthly salinity in North-central Biscayne Bay.

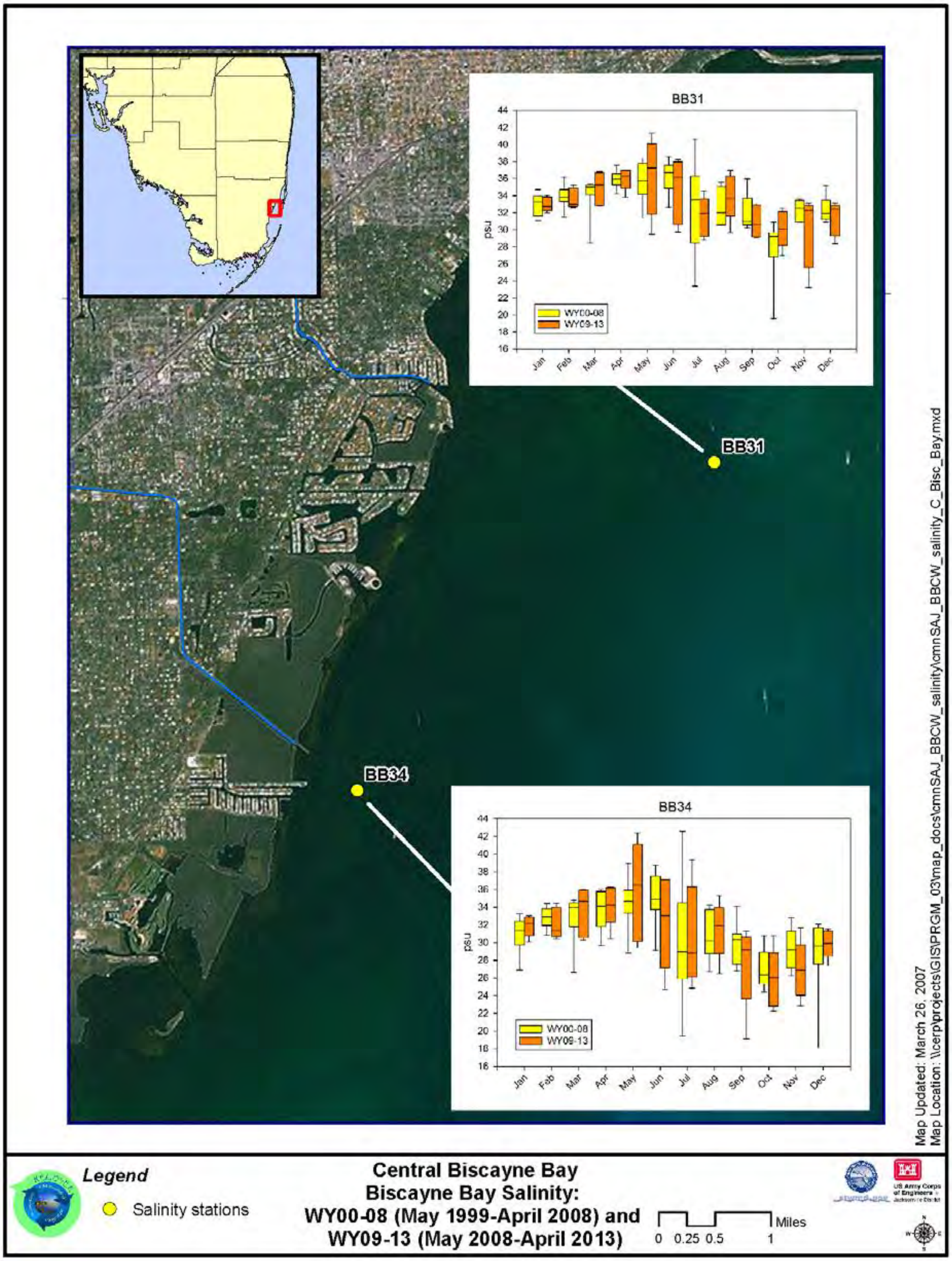


Figure 7-17. WY 2000–WY 2008 and WY 2009–WY 2013 box-and-whisker plots of monthly salinity in Central Biscayne Bay.

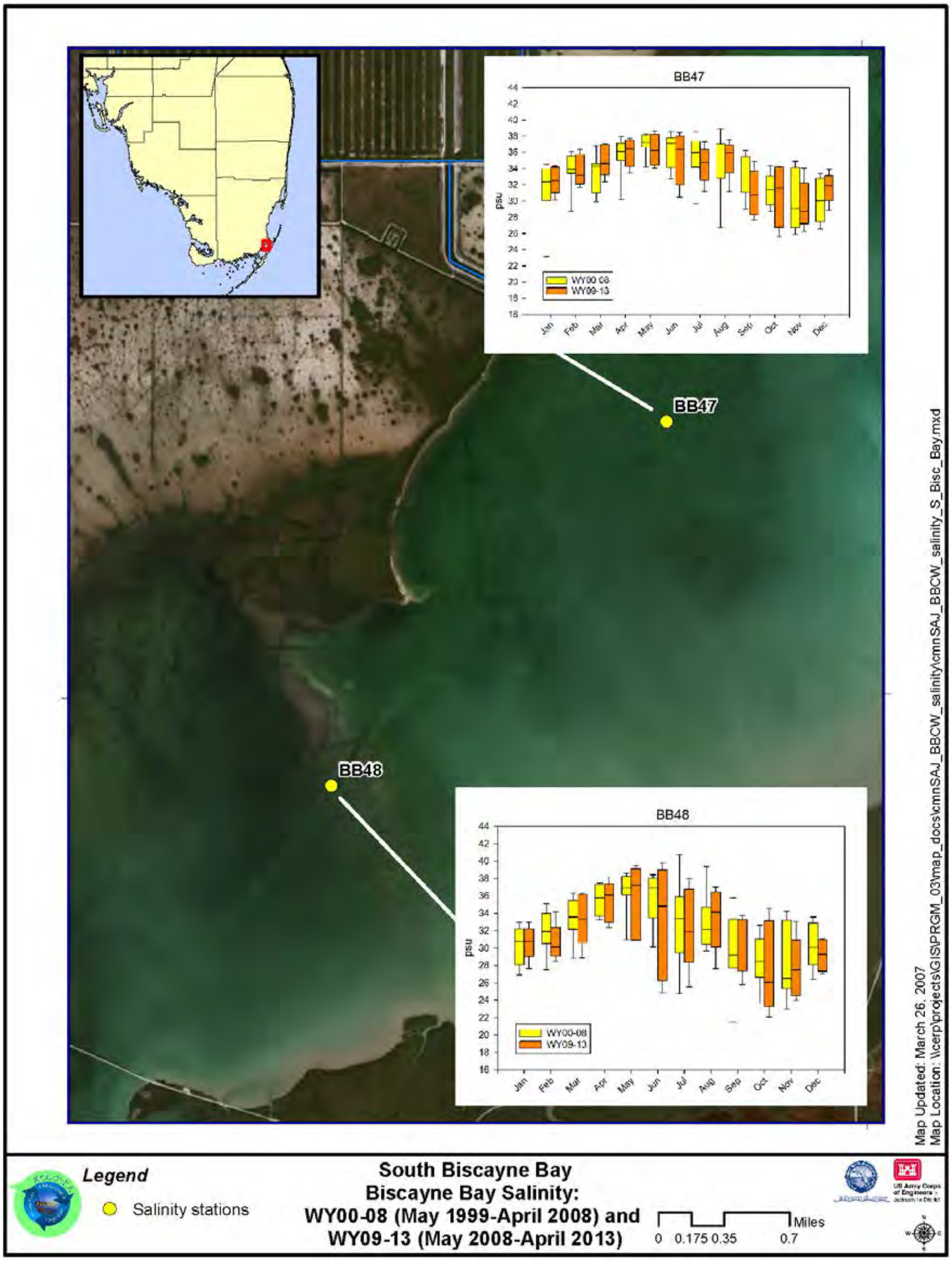


Figure 7-18. WY 2000–WY 2008 and WY 2009–WY 2013 box-and-whisker plots of monthly salinity in South Biscayne Bay.

In north-central Biscayne Bay (stations BB14, BB17, and BB22), the lower end of the salinity range (10^{th} percentile) is noticeably higher than for the northern region, which may be due to the closer proximity of the north-central bay monitoring stations to ocean inlets (i.e., Government Cut; **Figure 7-16**). Salinity generally ranges between 18 and 39 based on 10^{th} and 90^{th} percentile values. Monthly differences between WY 2000–2008 and WY 2009–WY 2013 at BB14 were very similar to those observed at BB04. At BB17, there are only slight differences in monthly medians between WY 2000–WY 2008 and WY 2009–WY 2013. The notable exception is the month of July when median salinity is about 3 higher for WY 2009–WY 2013 compared to WY 2000–WY 2008. At BB22, monthly median salinity between the two comparison periods also is similar. When differences occur (February, March, October, and November), median salinity for WY 2009–WY 2013 is about 2 to 3 lower than WY 2000–WY 2008.

In the Central Biscayne Bay region (stations BB31 and BB34), the salinity range (between 18 and 43 based on 10^{th} and 90^{th} percentile values, **Figure 7-17**) is generally about the same as the northern bay regions with one notable exception. For WY 2000–WY 2008 July and WY 2009–WY 2013 May, salinity is appreciably higher, with the 90^{th} percentile values reaching 40 to 43 during those two months. This is a somewhat surprising occurrence given that these two stations are located in an area of the bay that is well flushed with Atlantic Ocean waters via the nearby Safety Valve. At BB31, median monthly salinity for WY 2009–WY 2013 is about 2 higher than WY 2000–WY 2008 during May and August and about 2 lower during July. At BB34, median monthly salinity for WY 2009–WY 2013 is about 2 higher than WY 2000–WY 2008 during May and August and about 2 to 3 lower during February, July, September, and November.

For the southern region of Biscayne Bay (i.e., Card Sound; stations BB47 and BB48), salinity generally ranged between 21 and 40 , with BB48 exhibiting a larger range and variability than BB47 (**Figure 7-18**). The overall higher salinity in this region compared to other regions is not surprising given its relative isolation from Atlantic waters. At BB47, median salinity during WY 2009–WY 2013 is 1 to 2 higher than WY 2000–WY 2008 during August and December, and about 2 lower during July. BB48 showed a similar difference pattern for July and August. BB48 also showed median salinity that was about 2 lower in WY 2009–WY 2013 compared to WY 2000–WY 2008 during February, June, and October.

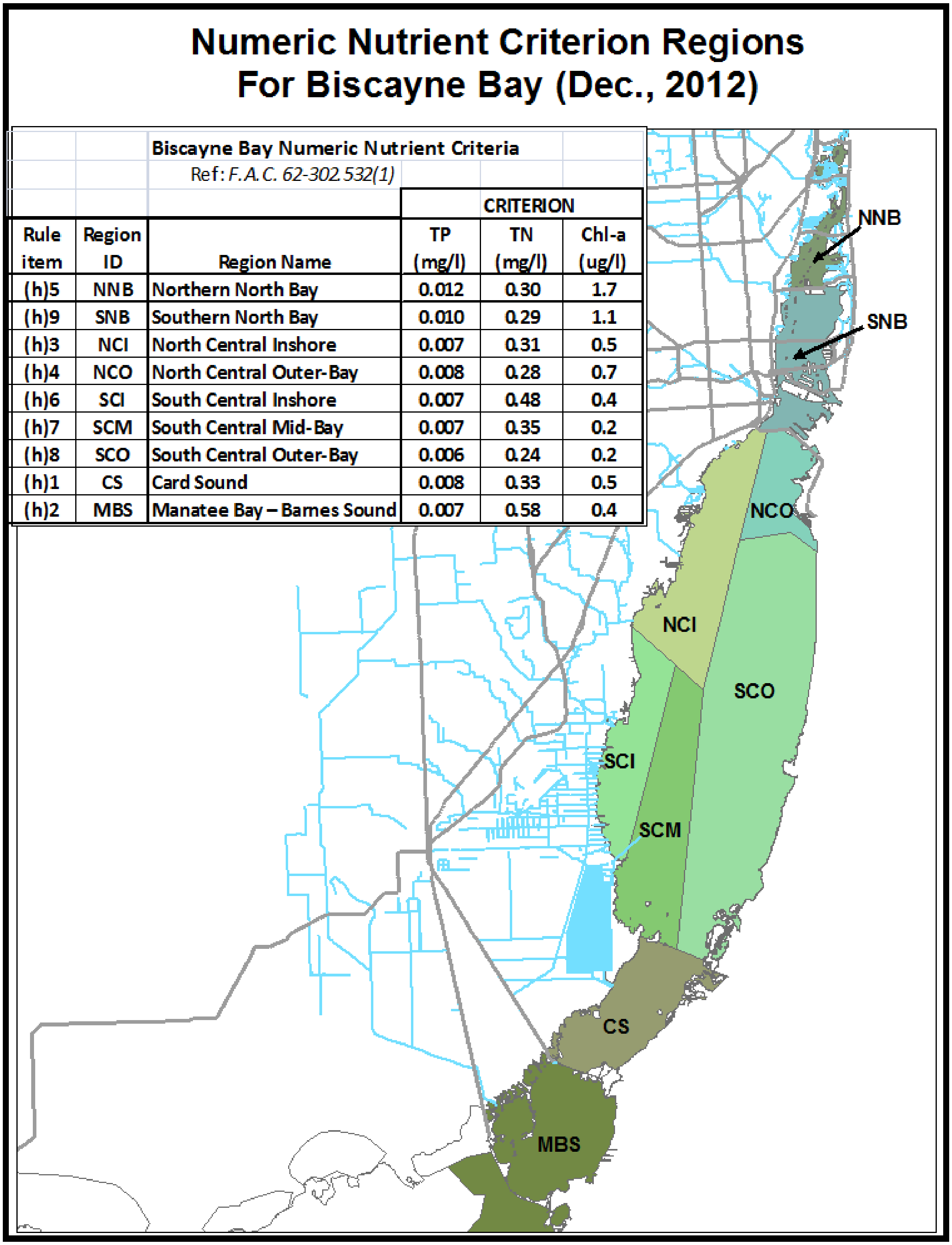
Assessments of salinity status and trends in Manatee Bay and Barnes Sound can be found in the Florida Bay section later in this chapter.

Biscayne Bay Water Quality

The Biscayne Bay Water Quality Monitoring Program, a cooperative program with Miami-Dade County and SFWMD, collects various water quality parameters to determine status and trends, identify pollution sources and regions of impact, as well as support multiple management and regulatory programs, initiatives, and requirements. Construction of major canals through the Everglades, dredging of natural tributaries and transverse glades, urban development and agriculture, and opening inlets to the Atlantic Ocean in areas historically isolated from oceanic influences greatly affected the natural salinity gradients of the bay and introduced increased levels of anthropogenic sources of nutrients and turbidity, all of which influence and affect water quality.

Increased nutrient availability will result in increased frequency, severity, duration, and spatial extent of algal blooms. Algal blooms can adversely affect other water quality parameters, such as light penetration and dissolved oxygen regimes, which subsequently affect the bay biota with cascading effects through the ecological web. Open areas of Biscayne Bay generally exhibit nutrient concentrations from 0.001–0.015 milligrams per liter (mg/L) of total phosphorus (TP) and 0.04–1.07 mg/L of total nitrogen (TN). In December 2012, the State of Florida established numeric nutrient criteria (62-302.532(h) Florida Administrative Code) for TP, TN, and chlorophyll-*a* (Chla; used as an indicator for overall water quality conditions) based on various hydrologic and water quality statistical models. Nine indicator regions were defined throughout the bay based on a combination of physical, chemical, and biological parameters (salinity, temperature, pH, dissolved oxygen, phosphorus, nitrogen, and Chla). Criteria were established for each region based on the specific conditions and natural variation within each region (**Figure 7-19**).

Compliance with the established numeric criteria is based on a single exceedence of the annual geometric mean in a three-year period. **Table 7-5** shows TP, TN, and Chla compliance with the numeric nutrient criteria for the period 2009–2012 for all nine indicator regions shown in **Figure 7-19.** Annual geometric mean values are shown in each cell and the cell color reflects compliance (green) or noncompliance (red) status. The majority of indicator regions for the entire assessed period fell well below (30%–80%) the established criterion value. The majority of the Chla values for the assessed period fell slightly above or below the established criterion value with every year assessed for Manatee Bay/Barnes Sound, South Central Mid-Bay, and South Central Outer-Bay in a non-compliance status. South Central Inshore fell into noncompliance status in 2010–2012. North Central Inshore came into and remained in compliance in 2011–2012. The non-compliance status of Manatee Bay/Barnes Sound may reflect the recovery status of the indicator region from the unprecedented phytoplankton bloom event of 2005–2008 (Boyer et al. 2009). The red status and annual geometric mean values slightly above the criterion in the South Central Mid-Bay and South Central Outer-Bay are cause for concern as they may be indicating water quality conditions that merit attention.



Biscayne Bay Numeric Nutrient Criteria
Ref: *F.A.C. 62-302.532(1)*

Rule item	Region ID	Region Name	CRITERION TP (mg/l)	TN (mg/l)	Chl-a (ug/l)
(h)5	NNB	Northern North Bay	0.012	0.30	1.7
(h)9	SNB	Southern North Bay	0.010	0.29	1.1
(h)3	NCI	North Central Inshore	0.007	0.31	0.5
(h)4	NCO	North Central Outer-Bay	0.008	0.28	0.7
(h)6	SCI	South Central Inshore	0.007	0.48	0.4
(h)7	SCM	South Central Mid-Bay	0.007	0.35	0.2
(h)8	SCO	South Central Outer-Bay	0.006	0.24	0.2
(h)1	CS	Card Sound	0.008	0.33	0.5
(h)2	MBS	Manatee Bay – Barnes Sound	0.007	0.58	0.4

Figure 7-19. Nine Biscayne Bay Numeric Nutrient Criteria indicator regions and their associated numeric criteria. (µg/L – microgram per liter.)

Table 7-5. Biscayne Bay Numeric Nutrient Criteria compliance 2009–2012. Green cells indicate compliance with numeric nutrient criteria. A region must not exceed the criteria more than once in any three-year period to be considered "compliant". (AGM – annual geometric mean.)

AGM for Total Phosphorus Concentration (mg/l)

Bay Region	MBS	CS	SCI	SCM	SCO	NCI	NCO	SNB	NNB
Criterion	0.007	0.008	0.007	0.007	0.006	0.007	0.008	0.01	0.012
2009	0.002	0.002	0.003	0.002	0.002	0.003	0.003	0.003	0.005
2010	0.003	0.003	0.003	0.002	0.003	0.003	0.003	0.004	0.005
2011	0.003	0.002	0.003	0.003	0.002	0.003	0.002	0.004	0.005
2012	0.003	0.003	0.003	0.002	0.002	0.003	0.003	0.005	0.006

AGM for Total Nitrogen Concentration (mg/l)

yr	MBS	CS	SCI	SCM	SCO	NCI	NCO	SNB	NNB
Criterion	0.58	0.33	0.48	0.35	0.24	0.31	0.28	0.29	0.3
2009	0.13	0.31	0.30	0.16	0.13	0.18	0.15	0.16	0.17
2010	0.11	0.27	0.30	0.13	0.10	0.13	0.09	0.11	0.14
2011	0.18	0.29	0.27	0.17	0.15	0.19	0.15	0.12	0.15
2012	0.15	0.23	0.34	0.18	0.14	0.16	0.15	0.14	0.17

AGM for Chlorophyll-a Concentration (ug/l)

Bay Region	MBS	CS	SCI	SCM	SCO	NCI	NCO	SNB	NNB
Criterion	0.4	0.5	0.4	0.2	0.2	0.5	0.7	1.1	1.7
2009	0.51	0.31	0.31	0.22	0.26	0.53	0.47	0.73	1.36
2010	0.66	0.47	0.49	0.35	0.35	0.51	0.68	0.87	1.74
2011	0.66	0.34	0.47	0.31	0.25	0.42	0.51	0.85	1.61
2012	0.70	0.38	0.46	0.28	0.24	0.46	0.48	0.88	1.65

Bay Regions: BSMB=Barnes Sound-Manatee Bay; CS=Card Sound; SCI=South Central Inshore; SCM=South Central-Mid; SCO=South Central Outer; NCI=North Central Inshore; NCO=North Central Outer; SNB=Southern North Bay; NNB=Northern North Bay

= Region is compliant with Numeric Nutrient Criteria Standard
= Region is not compliant with Numeric Nutrient Criteria Standard

While water quality generally has met federal, state, and local criteria, three recent events highlight the sensitivity of the system to minor perturbations. The first occurred following a series of hurricanes in 2005 and activities associated with the U.S. Highway 1 expansion (see **Appendix 7-1**, Significant Structural and Operational Changes Affecting the Southern Coastal System over the Last 30 Years and the C-111 Spreader Canal Project, U.S. Highway 1 Expansion section later in this chapter for additional information and other analyses). An algal bloom of microscopic blue-green algae reached unprecedented concentrations and duration in Southern Biscayne Bay, Card Sound, Barnes Sound and Eastern Florida Bay. Chla concentrations remained elevated for a period of three years. Maximum levels exceeded background concentrations by a factor of almost 100 times. Reduced water clarity also impacted the associated seagrass community (see the Submerged Aquatic Vegetation in Central and Southern Biscayne Bay section for details). The bloom was most likely triggered by introduction of higher than normal levels of phosphorus from increased canal discharges and the bay's internal sources (in sediments and soils) due to storm-related water releases and soil/sediment disturbance related to U.S. Highway 1 construction and storm events (Rudnick et al. 2007).

The second event involved parts of the western shoreline of Biscayne Bay just south of the Rickenbacker Causeway. A macroalgae bloom composed of mainly *Anadyomene* spp. was detected initially in 2002 and still persists to this day. It has resulted in a shift from a seagrass dominated community to a macroalgal dominated community in the areas affected. More information on this macroalgae event can be found in the RECOVER 2012 SSR Update, Southern Coastal Systems, Submerged Aquatic Vegetation Update, New Science (RECOVER 2012).

The third event began in June 2013 and is detailed in the next section. Algal blooms are generally considered to be an indicator of degraded conditions in the areas in which they occur and underscore the sensitivity of the bay to both human and natural disturbances.

2013 Biscayne Bay Bloom

In mid-June 2013, reports from Biscayne National Park staff were received of cloudy, foul smelling water in Biscayne Bay near the areas of Black Point, Convey Point, and Elliott Key. A multi-agency response—NPS, NOAA's Atlantic Oceanographic and Meteorological Laboratory, NOAA's National Marine Fisheries Service South East Fisheries Science Center, Florida Fish and Wildlife Conservation Commission (FWC), Florida Department of Environmental Protection (FDEP), University of Miami, Florida International University, Miami-Dade DERM, and SFWMD—of both aerial and boat surveys was initiated to determine the spatial extent of the potential bloom and to collect water quality samples. Shipboard continuous fluorometry transects indicated the presence of an algal bloom in four distinct regions of Biscayne Bay (Card Sound, Convoy/Turkey Point, Elliott Key, and Coral Gables Waterway) on July 16, 2013 (**Figure 7-20**). It was not clear whether the bloom originated in one location and was later separated by stormy weather into the four regions or whether four separate blooms formed.

This bloom event was dominated by diatoms. The highest Chla measurements of 27 micrograms per liter (µg/L) were recorded in Card Sound region where the primary genus was the diatom *Chaetoceros* spp. This same diatom dominated the bloom around the Coral Gables Waterway (C-3 Canal), with Chla of 12 µg/L. The area between Turkey Point and Convey Point showed the lowest Chla levels. A second baywide survey was conducted on July 23, when bloom conditions had considerably dissipated (**Figure 7-21**).

This event was nearly as intense as the cyanobacteria bloom that occurred in 2007 associated with the 2005 hurricane season and U.S. Highway 1 widening. Both bloom events had recorded Chla levels over 20 times baseline conditions. By the August 8 survey, the bloom had largely dissipated. Throughout the bloom, all identified phytoplankton were nontoxic with no associated fish kills or reports of human illness and water quality data (Lindsey Visser, NOAA, personal communications 2013) suggest it was likely caused by higher than average rainfall driving increased coastal runoff coupled with nutrient loading and temperature/salinity combinations favorable for diatom production.

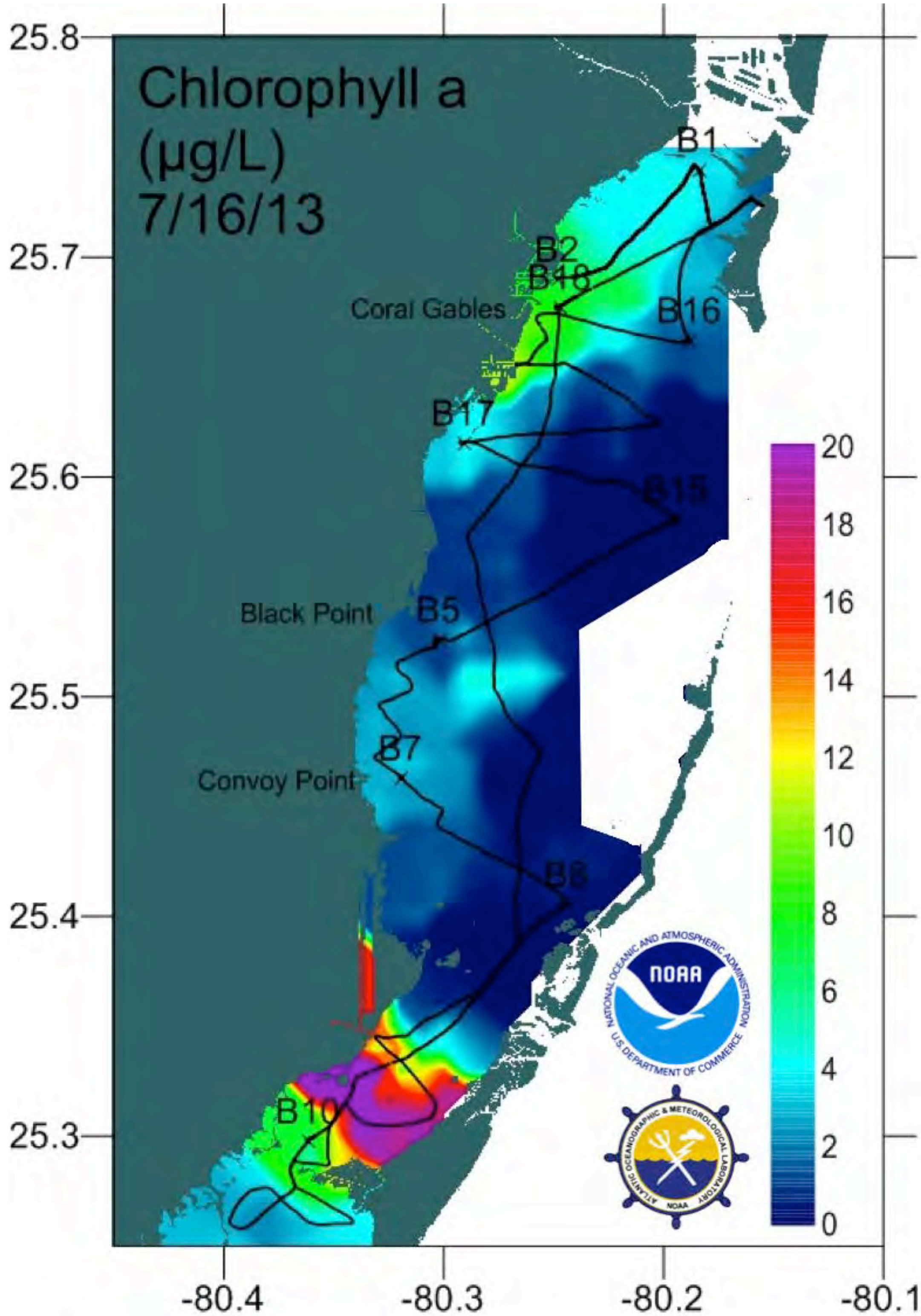


Figure 7-20. Chla concentrations in Biscayne Bay on July 16, 2013. The black line within the bay shows the survey's boat track of continuous-flow water quality analysis.

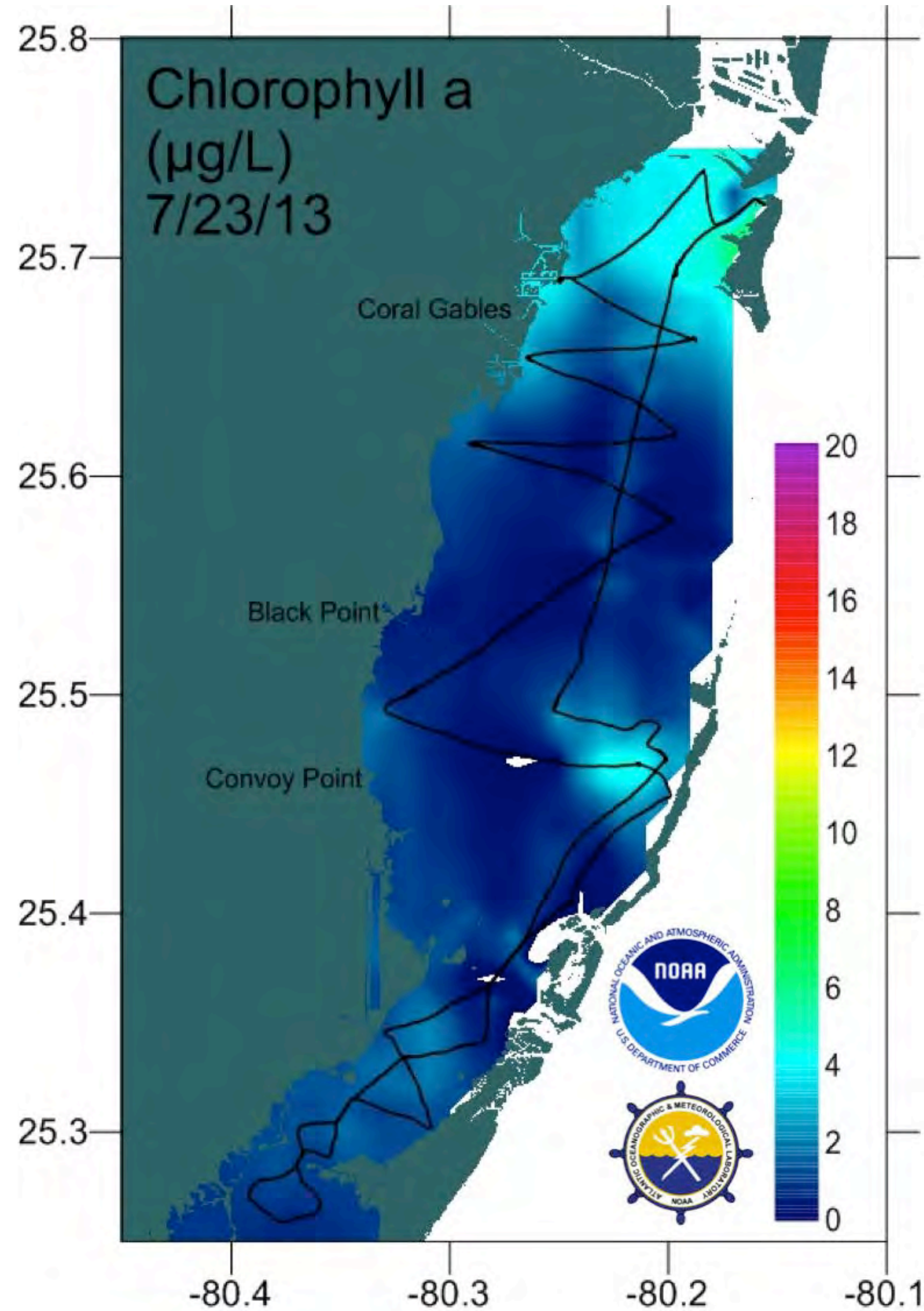


Figure 7-21. Chla concentrations in Biscayne Bay on July 23, 2013. The black line within the bay shows the survey's boat track of continuous-flow water quality analysis.

Central and Southern Biscayne Bay Submerged Aquatic Vegetation

Since 1999, Miami-Dade DERM, in partnership with SFWMD, has conducted a broad, spatially stratified random sampling program (annually) to assess SAV from the Rickenbacker Basin south to Little Card Sound. Additionally, as part of a separate program, SAV in Manatee Bay and the southern half of Barnes Sound has been monitored since 1998. The sampling design employs a 100-m buffer from major shorelines, and primarily utilizes the Braun-Blanquette Cover/Abundance (BBCA) scale as the metric to assess SAV (Braun-Blanquett 1932).

Central - Southern Biscayne Bay Seagrass and SAV Distribution

In Central-Southern Biscayne Bay, depth, bottom type, salinity regimes, and exchange with ocean waters are determining factors in the distribution of seagrasses. The distribution and coverage of three primary species of seagrass in Biscayne Bay, *Thalassia testudinum*, *Syringodium filiforme*, and *Halodule wrightii*, are illustrated for the most recent three years (2010–2012) in **Figure 7-22**. Overall, *T. testudinum* is the dominant seagrass in Central-Southern Biscayne Bay area. The eastern region of the sampling area consistently has high coverage (≥ 75 %) and low year-to-year deviation. The hard bottom region south of the Feather Bed Shoal Complex as well as localized areas with greater depth (+4 m) in the northeastern and southeastern have patchy *T. testudinum* with average low coverage (< 5%). *H. wrightii* has consistently been found in the northern and southern regions, along the western shore line, and in the hard bottom areas. *S. filiforme* has also been primarily found in the northern and southern region, with infrequent records near the western nearshore area and in the eastern Safety Valve region. In summary, the overlap of these three seagrasses coupled with the diverse macroalgal components collectively form a total SAV community with moderate to high coverage (> 50%) throughout the majority of Central-Southern Biscayne Bay.

During the past five years, the three most abundant species of seagrass have been remarkably stable in Southern Biscayne Bay (**Figure 7-23**), except for two areas that were impacted by phytoplankton (Card Sound to Manatee Bay) or macroalgal blooms (central Biscayne Bay; see below). *H. wrightii* and *S. filiforme* abundance in southern Biscayne Bay is normally less than 25% where found; however, due to their limited distribution, their overall coverage is < 5%. *T. testudinum* is the obvious dominant seagrass in this region. The variation in *T. testudinum* abundance noted in the last five years, outside of areas impacted by algal blooms, has been slight and is consistent with levels of variability noted in the past. However, two disturbance events—a phytoplankton bloom in 2005–2008 and a macroalgal bloom from 2006 to present—had significant local impacts on seagrass and the associated benthic communities. The phytoplankton bloom occurred in Card Sound south through Barnes Sound and Manatee Bay and into Northeast Florida Bay.

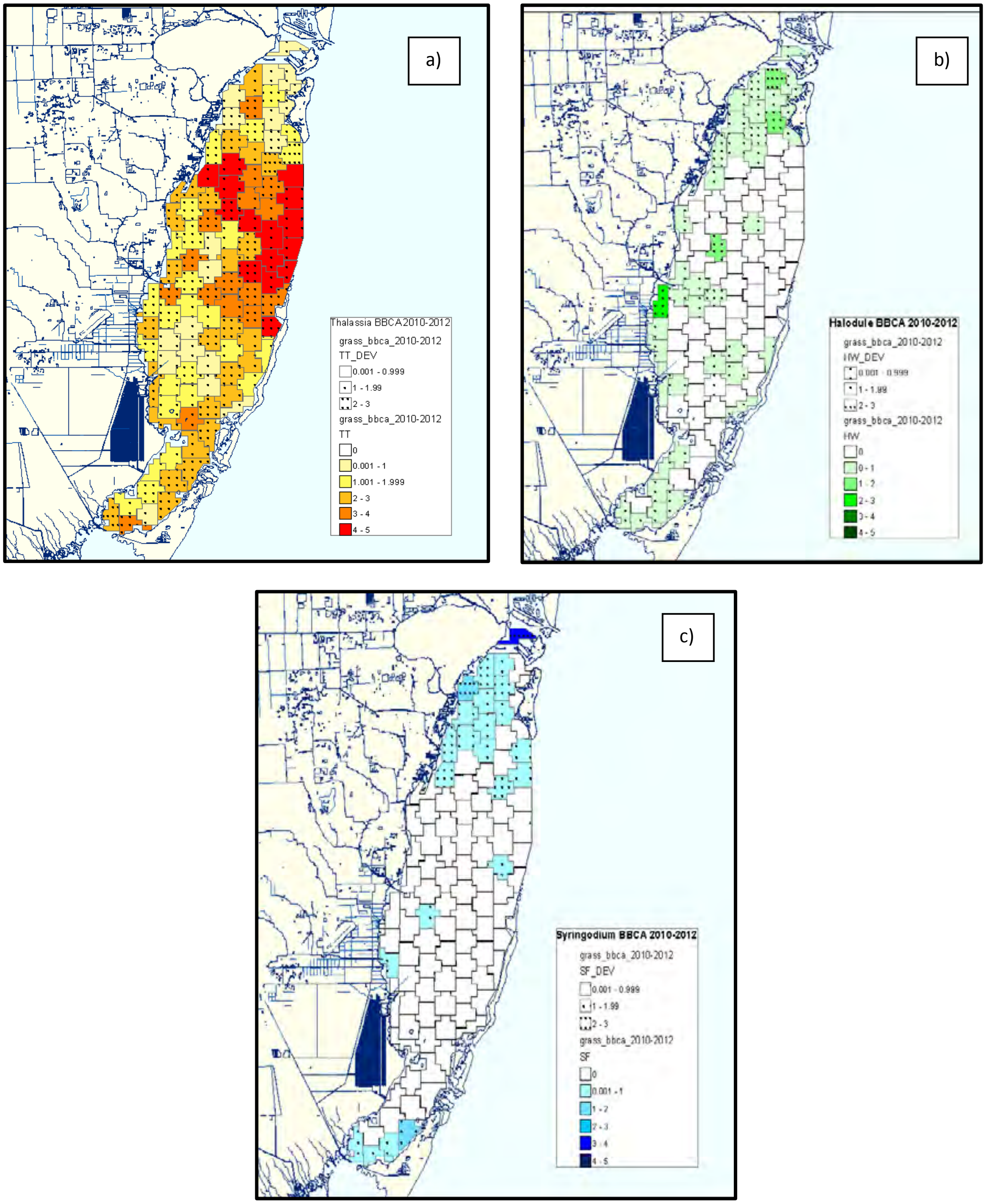


Figure 7-22. Maps of Central-Southern Biscayne Bay bottom cover by three primary species of seagrass in 2010–2012, based on mean BBCA scale values in the sampling design's stratified random polygons for a) *T. testudinum*, b) *H. wrightii*, and c) *S. filiforme*.

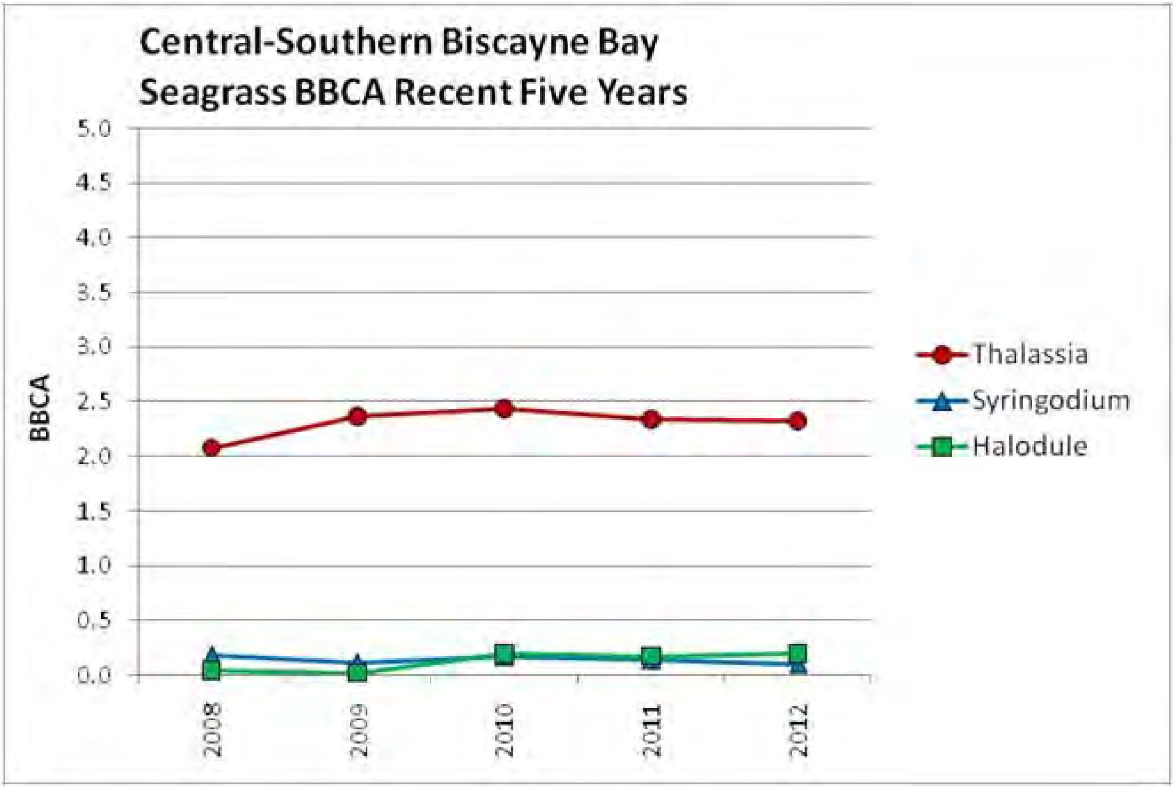
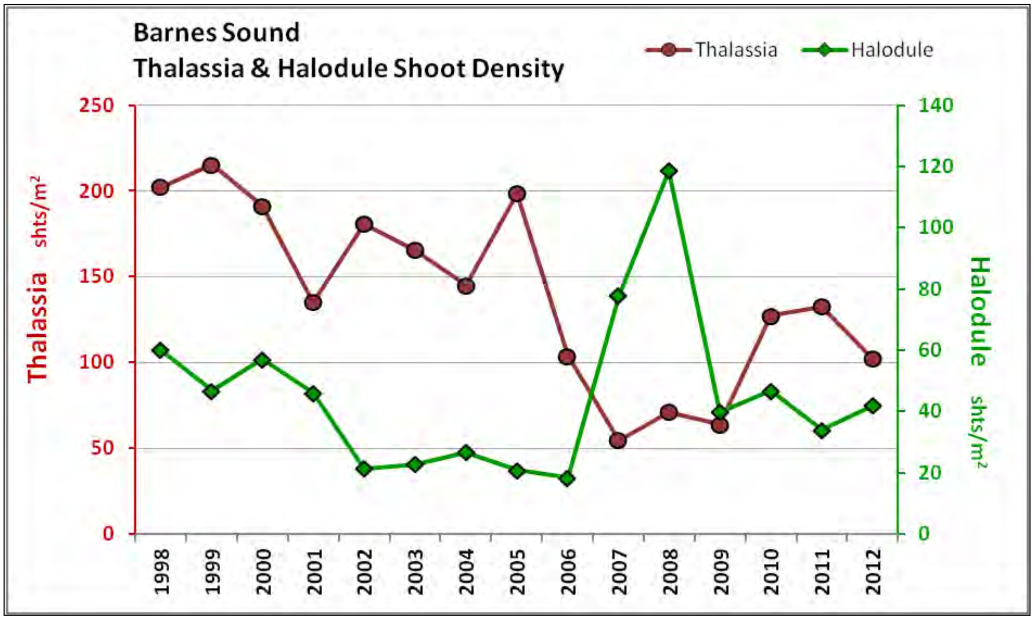


Figure 7-23. Annual mean bottom cover (BBCA values) for the three primary species of seagrass in Central-Southern Biscayne Bay.

<u>Southern Basins Phytoplankton Bloom and Seagrass Trends</u>

A series of events and a combination of factors have been related to the development of a phytoplankton bloom that impacted southernmost Biscayne Bay and Northeast Florida Bay basins (Rudnick et al. 2007). Preceding the bloom event were two successive years, 2004–2005, of record high salinity in the region, with peak values in Manatee Bay and Barnes Sound exceeding 40 . In October 2005, immediately following Hurricane Katrina and prior to the initiation of the bloom, dissolved oxygen was > 1.0 mg/L and oxidation-reduction potential values were strongly negative (indicative of chemical consumption of oxygen by hydrogen sulfide) throughout the area. Concurrent observations were made of stressed seagrass and sponges. By the time a second hurricane, Wilma, impacted the region, the phytoplankton bloom had initiated. Declines in *T. testudinum* shoot density were measured through 2009, with concurrent increases in *H. wrightii* densities (**Figure 7-24**). From 2010 through 2012, *T. testudinum* shoot density has been recovering (increasing) in both basins. During that time, a similar inverse relationship was observed between the density of *T. testudinum* and *H. wrightii*—the density of *H. wrightii* decreases concurrently with the increase in *T. testudinum*.



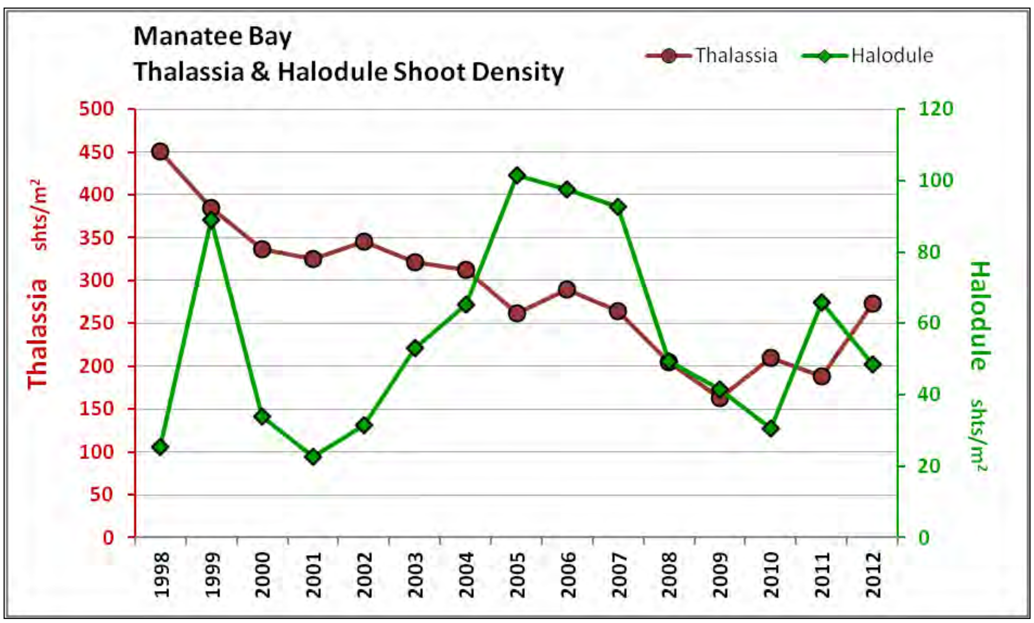


Figure 7-24. Annual mean shoot density since 1998 in Barnes Sound and Manatee Bay, showing the response of *T. testudinum* and *H. wrightii* during and after the phytoplankton bloom period of 2005–2007.

Biscayne Bay Macroalgae Bloom and Seagrass Losses

A bloom of two species of green macroalgae—*Anadyomene stellata* and *A. linkiani*—has been, and is continuing to be, documented in Central Biscayne Bay. Drifting mats of *Anadyomene* greater than 1 m in height were sufficient to smother adjacent seagrass beds (principally *T. testudinum*). Supplemental BBCA surveys, in addition to the routine annual monitoring during 2010–2012, mapped the spatial distribution and cover of the algae (**Figure 7-25**). *Anadyomene stellata* had been detected in low abundance within the bay since the beginning of the SAV monitoring (1999), but in the 2005–2007 period, the percent cover of these species increased dramatically in this region. Impacts to seagrass were evident in the data from this region. During 1999–2004 (pre-bloom period), green algae had a low percent cover, with most sites showing a BBCA of 0.1 to < 2 (solitary to < 5%; **Figure 7-26 top panel**). During this time, the opposite pattern was detected for *T. testudinum*, the dominant SAV, within the bloom area (> 75% coverage; **Figure 7-26 bottom panel**). From 2005–2012, when the bloom increased considerably, sites with *T. testudinum* were found to have BBCA scores predominantly less than 2 (< 5%; **Figure 7-26 bottom panel**), while green algae had BBCA scores of 4 to 5 (> 75%; **Figure 7-26 top panel**) at over half of the sites within the area. The bloom area has not expanded beyond the North Central Inshore Region. The biomass of *Anadyomene* remains high, with only a few locations showing senescence. These areas are seen as nonvegetated areas with unconsolidated sediments, and no detectable recovery of SAV has been documented in these areas to date. See Collado-Vides et al. (2013) for details and findings from the ongoing efforts to understand this bloom.

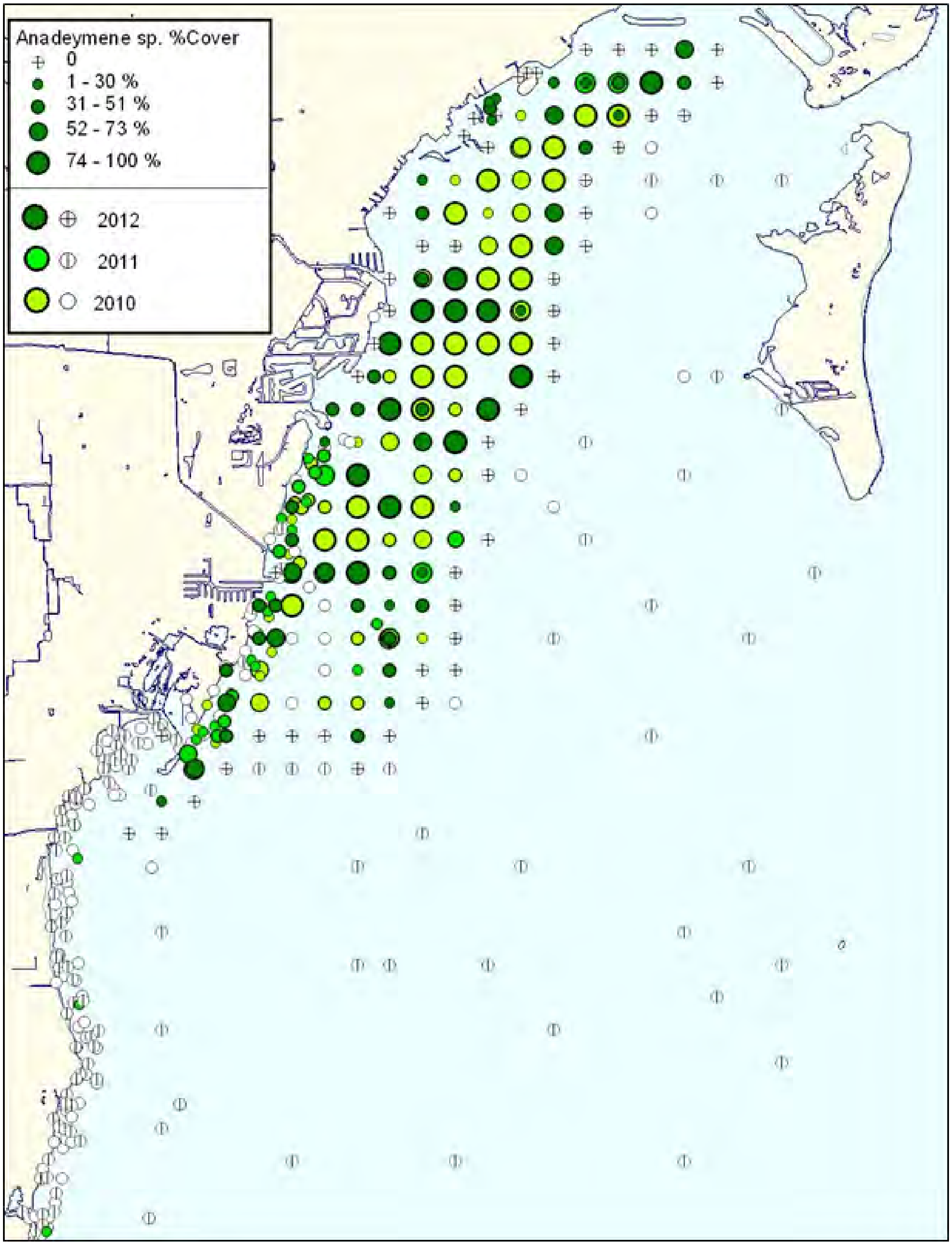
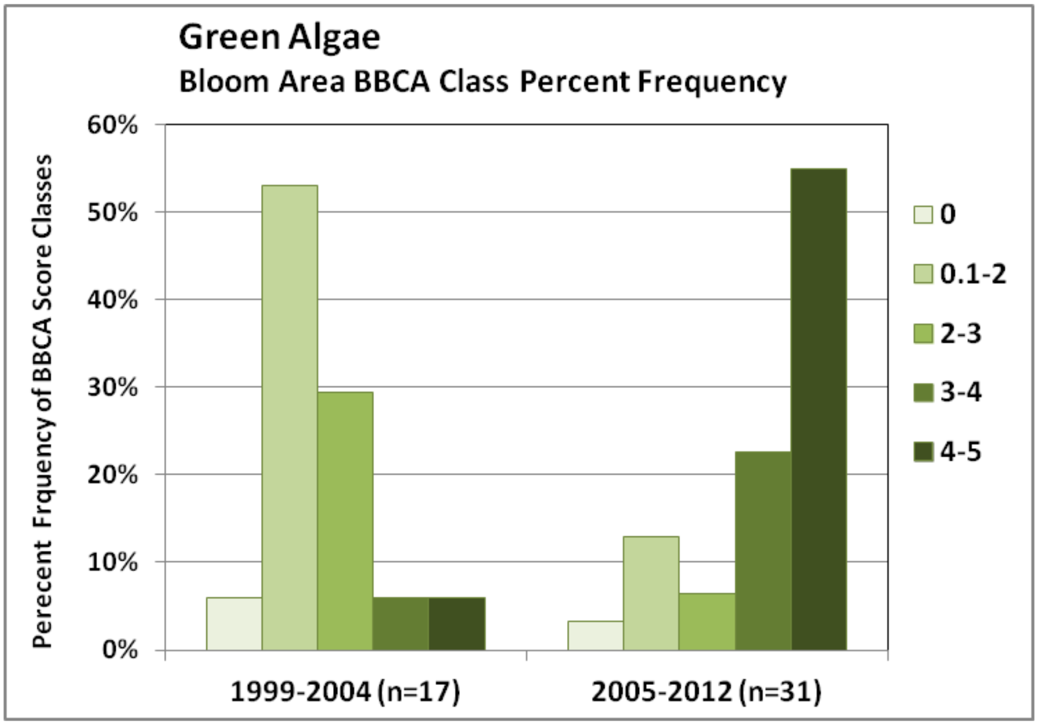


Figure 7-25. Distribution and percent bottom cover by green macroalgae *Anadyomene* spp. during 2010–2012. Data were mapped using combined data sets from Miami-Dade DERM and the University of Miami's Shallow Water Positioning System program.



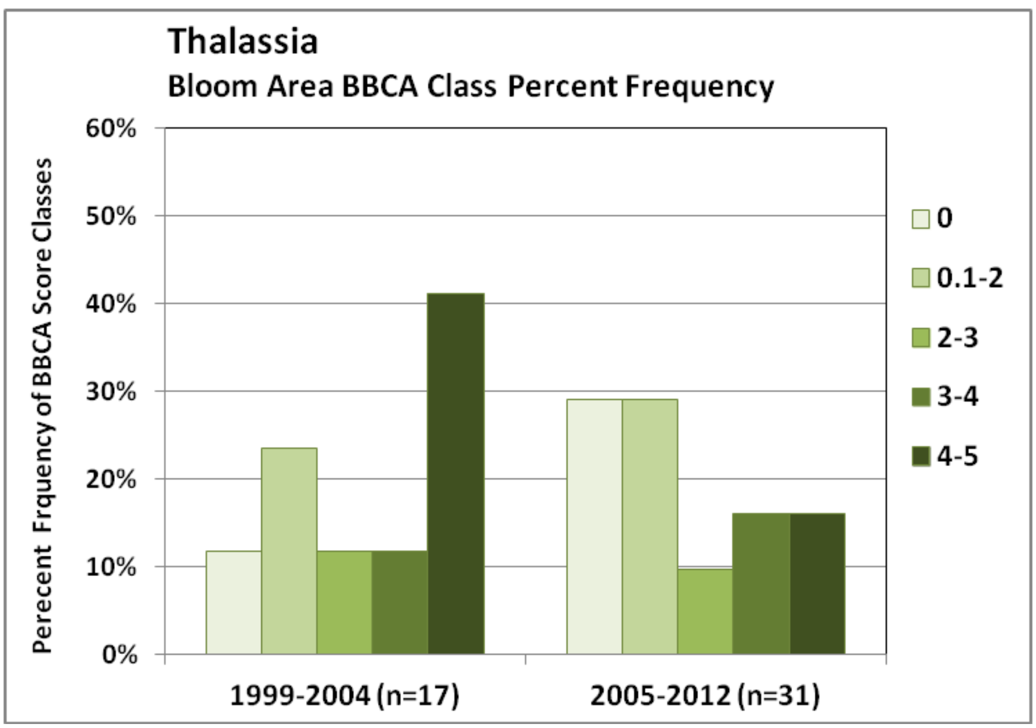


Figure 7-26. Frequency of five BBCA bottom cover categories for green macroalgae (top) and *T. testudinum* (bottom) in the Biscayne Bay macroalgal bloom area, comparing prebloom 1999–2004 and bloom 2005–2012 periods.

Florida Bay and the Lower Southwest Coast

Assessments for Florida Bay and the Lower Southwest Coast include seven primary components. The first describes hydrology/salinity through the United States Geological Survey (USGS) Coastal Gradients MAP project, status and trends at the NPS Marine Monitoring Network (MMN) stations in Florida Bay and Manatee Bay/Barnes Sound, and the application of the RECOVER performance measure for salinity in Florida Bay. The second component assesses water quality status and trends throughout Florida Bay and the Lower Southwest Coast, with data provided by NOAA's Atlantic Oceanographic and Meteorological Laboratory and SFWMD. The third component assesses SAV status and trends in Florida Bay, Manatee Bay/Barnes Sound, and the Lower Southwest Coast with data provided by both the Florida Habitat Assessment Program (FHAP), conducted by FWC's Florida Marine Research Institute and Miami-Dade DERM. The fourth component assesses pink shrimp and other epifauna status and trends in Florida Bay with data provided by NOAA, NOAA's National Marine Fisheries Service, and USGS. The fifth component assesses juvenile sportfish status and trends in Florida Bay with data provided by NOAA's Atlantic Oceanographic and Meteorological Laboratory. The sixth component assesses status and trends of roseate spoonbills along the coastal areas surrounding Florida Bay and the Lower Southwest Coast with data provided by Audubon Florida. The final component assesses status and trends of crocodile and alligator populations along the coastal areas surrounding Biscayne Bay (northernmost boundary at Key Biscayne), Card/Barnes Sounds, Manatee Bay, and Florida Bay to Cape Sable with data provided by the United States Fish and Wildlife Service in cooperation with the University of Florida. A fine-scale assessment of projects affecting Florida Bay (C-111 SCWP and U.S. Highway 1 Expansion) can be found later in this chapter.

Florida Bay and Lower Southwest Coast Hydrology/Salinity

Coastal Gradients

This RECOVER MAP project is assessing hydrologic responses to CERP in the mangrove-dominated transitional wetland between the Everglades and the SCS coast from Lostmans River to Long Sound. It is hypothesized that under pre-drainage conditions, headwater sites along the marsh-mangrove interface of the coastal Everglades had persistent freshwater to oligohaline salinity. For more information on historic conditions, please refer to the Coastal Salinity Gradients performance measure documentation sheet, which can be found online at www.evergladesplan.org/pm/recover/recover_docs/et/ge_12.pdf.

Mean monthly discharge and salinity at upstream and downstream coastal creek sites in Florida Bay and the southwest Florida coast for October 2003 through December 2012 are shown in **Figures 7-27** and **7-28** (Woods and Zucker 2011, Zucker and Boudreau 2013). Five transects comprise the Lower Southwest Coast drainage complex, including SRS, and deliver fresh water toward the Gulf of Mexico. The nine discharge pathways previously used for the Taylor Slough/C-111 drainage complex are reaggregated into five sets of creeks for this discussion and deliver fresh water to Florida Bay. Three continuous salinity monitoring stations (upstream Lostmans River, upstream Broad River, and upstream North River), representing the upstream component of the southwest coast transect, were discontinued in October 2011 (**Figure 7-27a**, **b**, **e**).

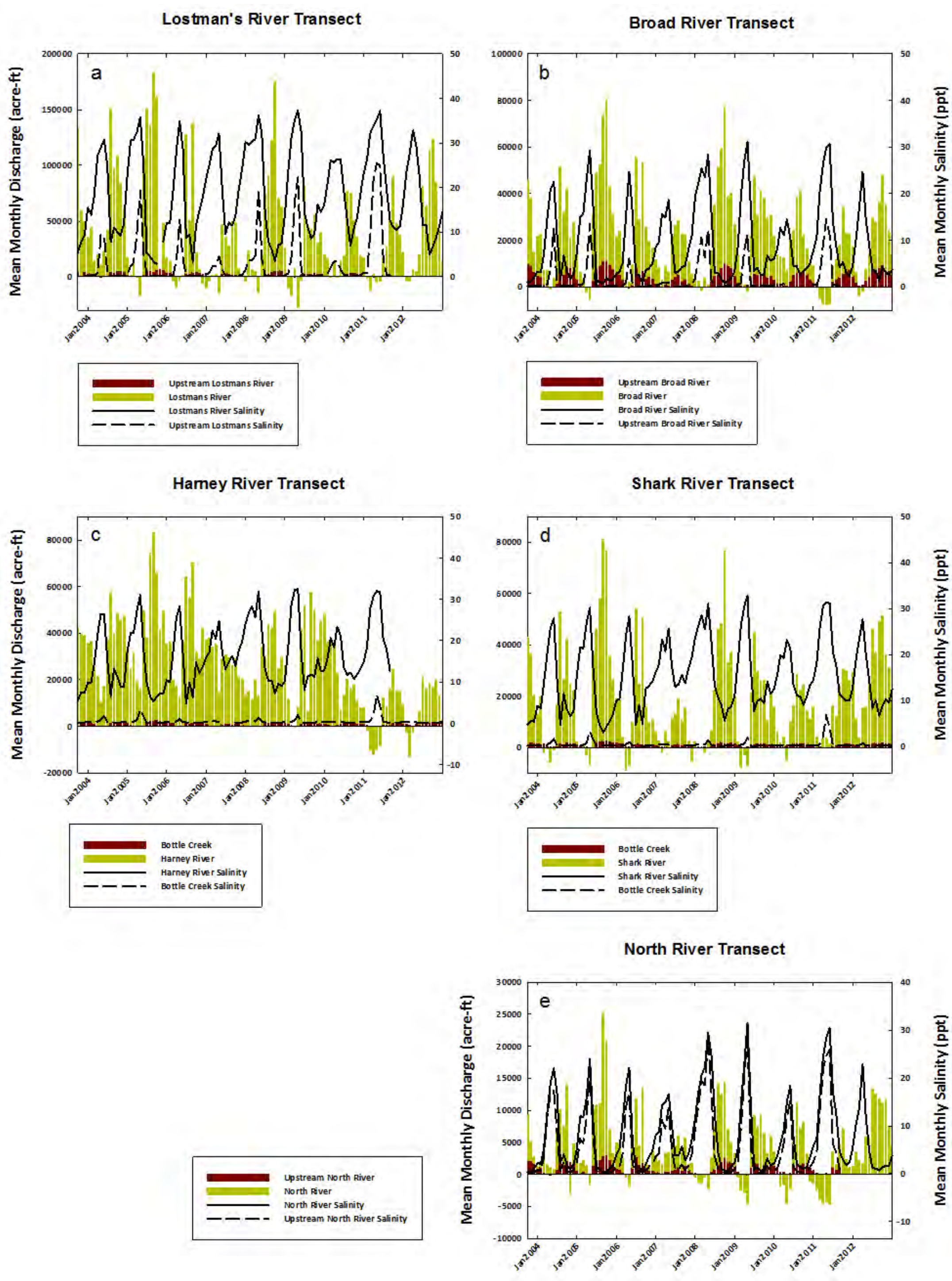


Figure 7-27. Mean monthly water discharge and salinity at upstream and downstream monitoring stations on Lower Southwest Coast rivers during 2003–2012. (Note: acre-ft – acre-feet; ppt – parts per thousand, which is an outdated salinity unit. Salinity is now considered unitless.)

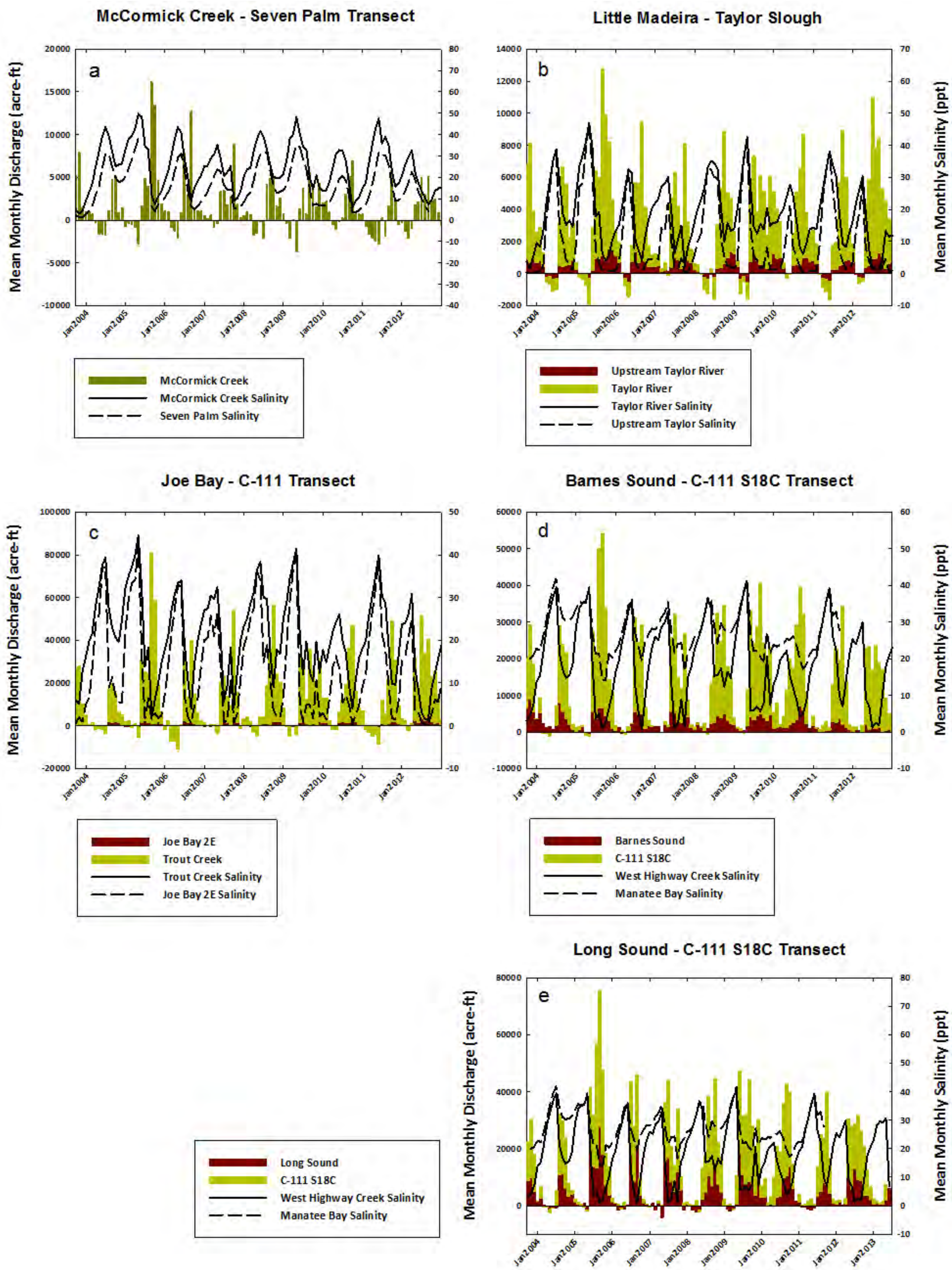


Figure 7-28. Mean monthly water discharge and salinity at upstream and downstream monitoring stations on Taylor Slough and C-111 basin coastal creeks during 2003–2012. (Note: acre-ft – acre-feet; ppt – parts per thousand, which is an outdated salinity unit. Salinity is now considered unitless.)

Salinity along the Coastal Gradients project transects indicated an absence of persistent freshwater-to-oligohaline zones at headwater sites across most of the coastal Everglades over the last four years. Seasonal occurrence of mesohaline salinity (5–18 ; Venice System 1959) was previously measured and reported in the 2007 and 2009 SSRs (RECOVER 2007, 2011). Salinity predictably dropped during periods when discharge increased in each drainage area. The only site with persistent freshwater-to-oligohaline conditions was the Bottle Creek headwaters of the Harney and Shark Rivers, which is located in Everglades herbaceous wetlands, well above the marsh-mangrove interface (**Figure 7-27c, d**). The exception occurred in May and June 2011 when salinity increased, reaching a maximum of 13.3 on May 18, 2011 and remained elevated above 5 until freshwater runoff and rainfall reduced salinity to fresh conditions in late June. Freshwater-to-oligohaline conditions generally prevailed at the Broad River headwaters, although salinity rose to more than 10 in 2004, 2005, 2008, 2009, and 2011 as a result of net negative flow due to tide (**Figure 7-27b**). Upstream and downstream salinity fluctuated coherently and routinely exceeded 10 in the Lostmans River, North River, and all Florida Bay creeks, where oligohaline conditions occurred only during high discharge periods (**Figures 7-27a**, **e**, and **7-28**).

The salinity regime for the southwest coast differed from that of Northeast Florida Bay. For the southwest coast, salinity at the upstream locations is usually fresh throughout the wet season and rises during the dry season due to tidal flow usually in May or June. By comparison, downstream salinity conditions of the southwest coastal rivers range from fresh to brackish in the wet season and remain brackish to saline in the dry season. When these sites are compared to Northeast Florida Bay, saltwater intrusion via surface water in the Taylor Slough and C-111 Basin occurs more rapidly during the transitional period between the wet and dry seasons and occasionally persists to July or August. Peak salinity values in downstream southwest coast rivers were typically near 30 and less than 40 , while peak salinity values in Northeast Florida Bay typically were near 40 , but ranged from about 20 to 50 .

Florida Bay: Status and Trends at the NPS Marine Monitoring Network

Status of Florida Bay in 2009 and trends from 1990 through 2009 have been assessed by NPS (SFNRC 2012). **Figure 7-29** shows the location of the monitoring stations and the water quality zones for Florida Bay and southernmost Biscayne Bay. A key finding in this report was an increasing salinity trend in the bay from 1996 through 2009, with the most pronounced increases in the Central and East bay. These trends are also evident in **Figures 7-30** through **7-35**, but note that delineated zones here are not identical to those of the NPS report. This figure shows the daily, 180-day moving, and three-year moving average salinity for the water quality zones depicted in **Figure 7-29** from May 1990 to April 2013.

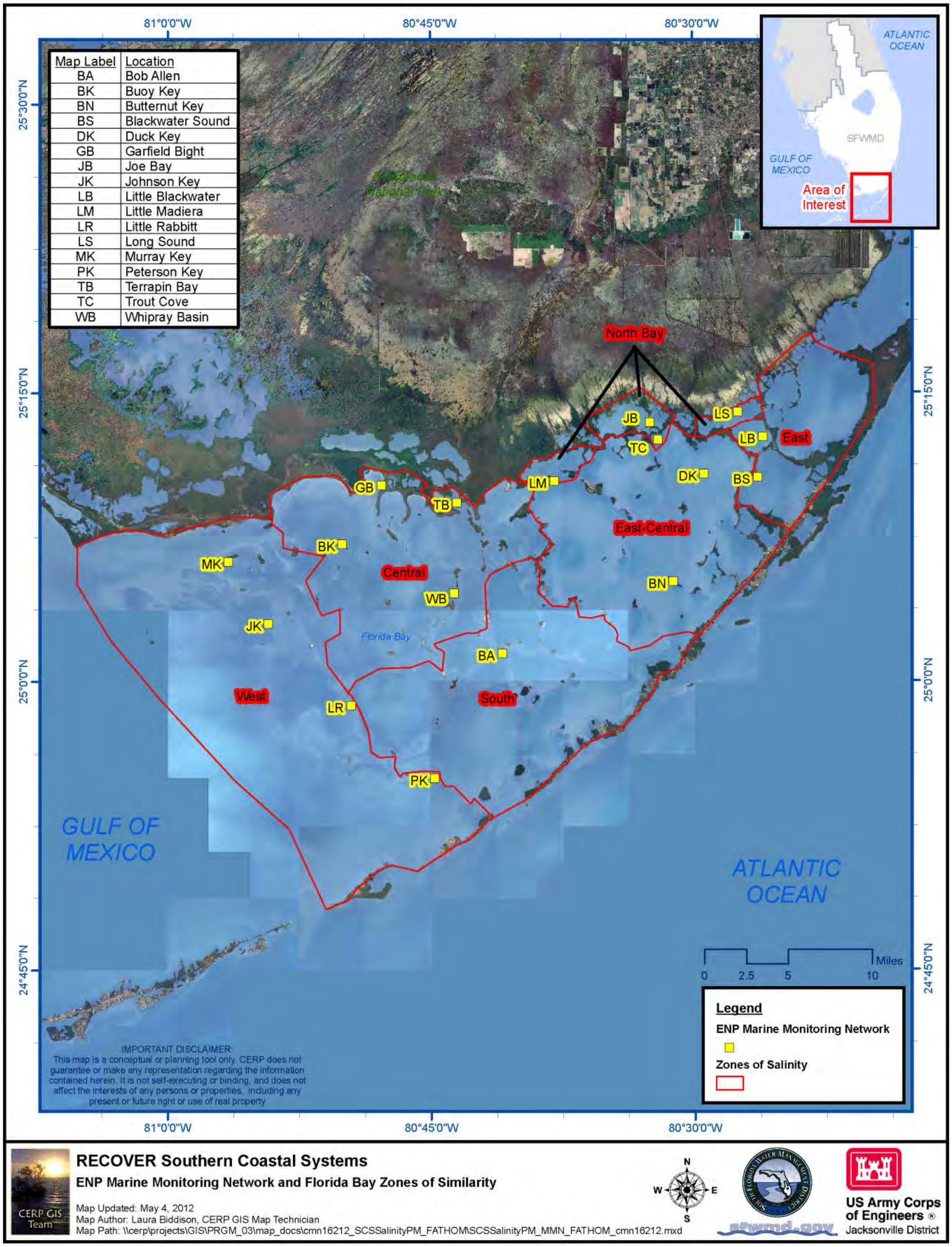


Figure 7-29. RECOVER SCS Performance Measure – Salinity in Florida Bay, NPS MMN assessment stations and water quality zones of similarity.

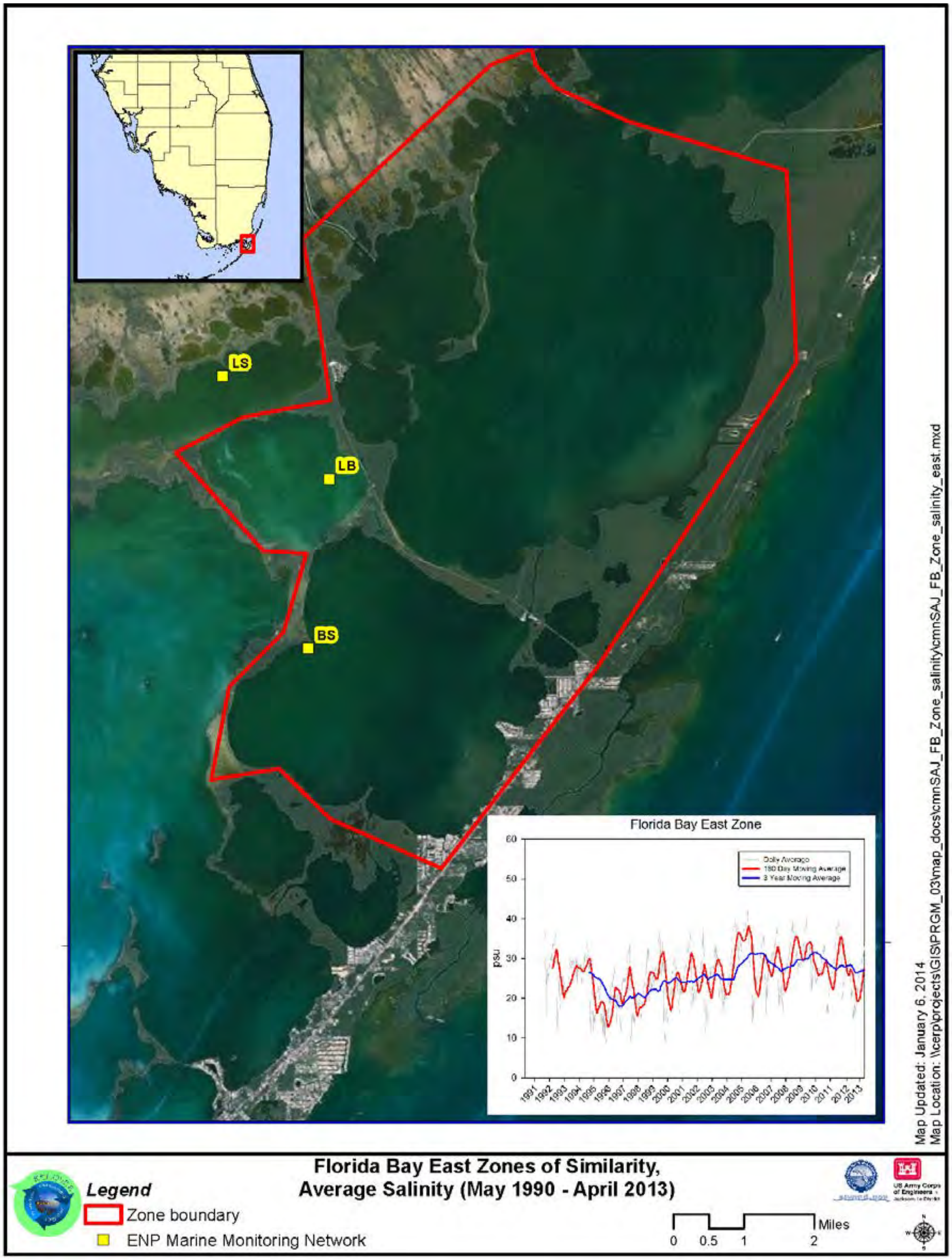


Figure 7-30. Daily, 180-day moving, and three-year moving average salinity time series in the East Florida Bay water quality zone of similarity. (Note that only stations that fall within the red boundary of the zone are included in the assessment for that zone.)

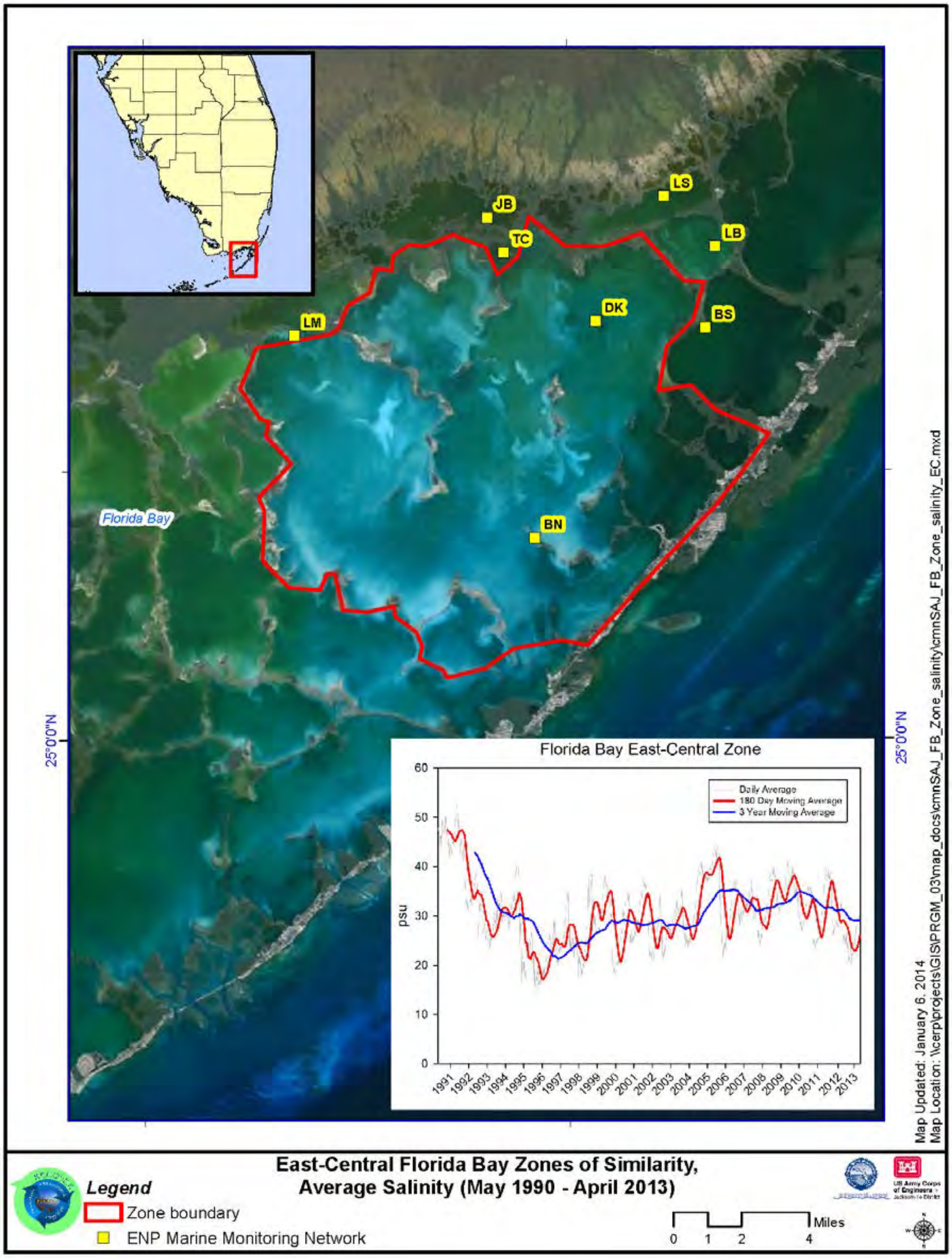


Figure 7-31. Daily, 180-day moving, and three-year moving average salinity time series in the East-Central Florida Bay water quality zone of similarity. (Note that only stations that fall within the red boundary of the zone are included in the assessment for that zone.)

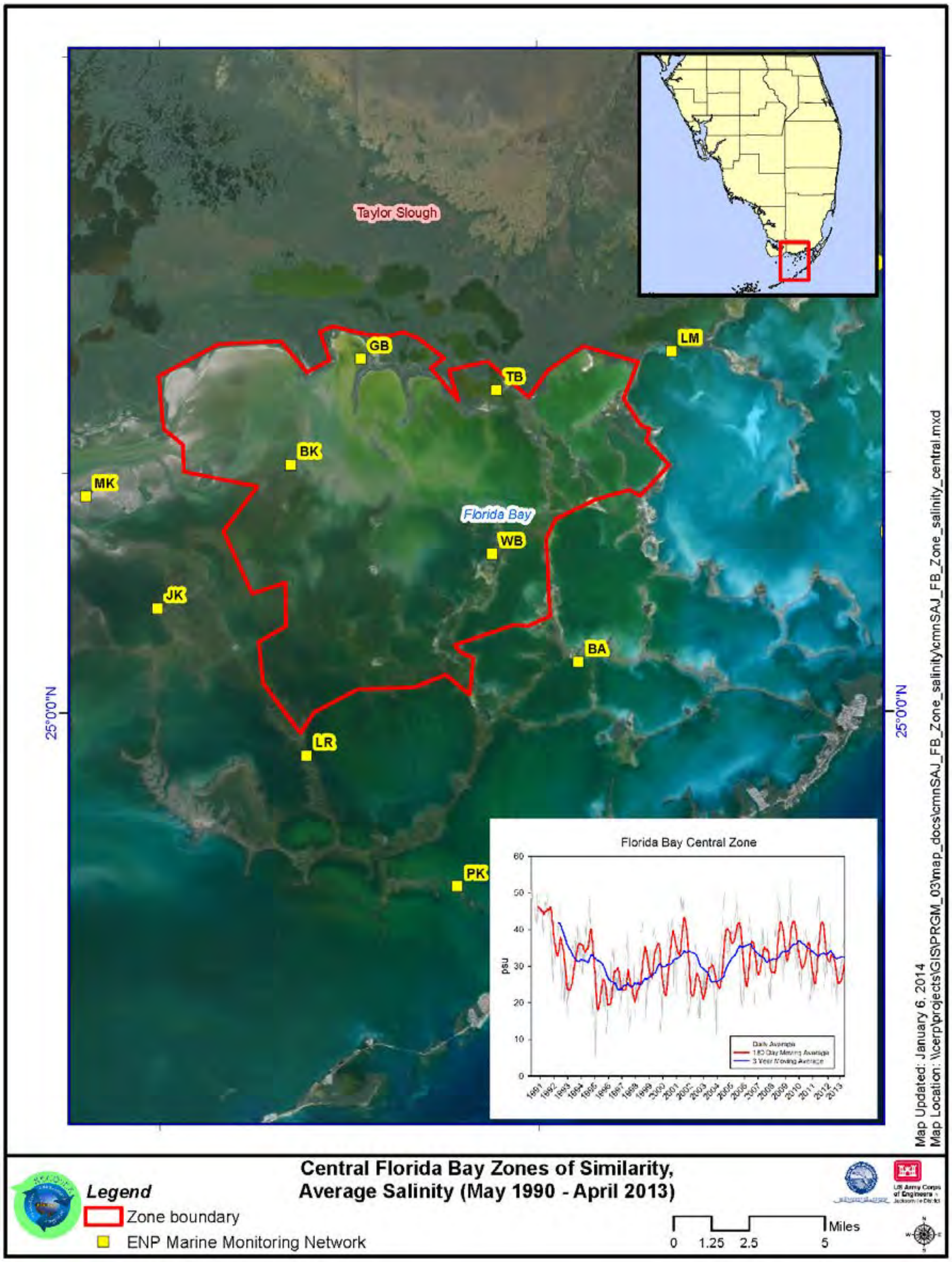


Figure 7-32. Daily, 180-day moving, and three-year moving average salinity time series in the Central Florida Bay water quality zone of similarity. (Note that only stations that fall within the red boundary of the zone are included in the assessment for that zone.)

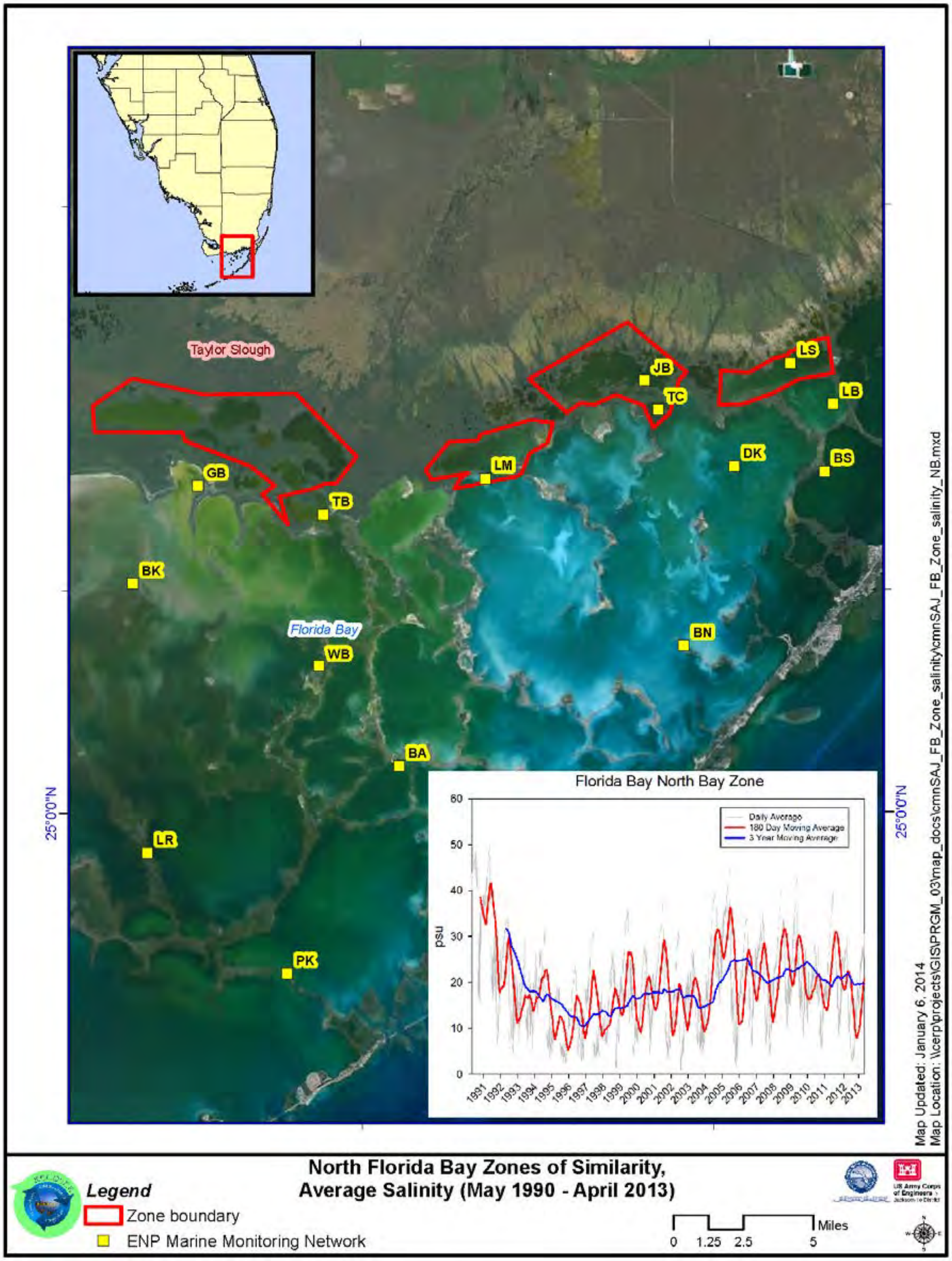


Figure 7-33. Daily, 180-day moving, and three-year moving average salinity time series in North Florida Bay water quality zone of similarity. (Note that only stations that fall within the red boundary of the zone are included in the assessment for that zone.)

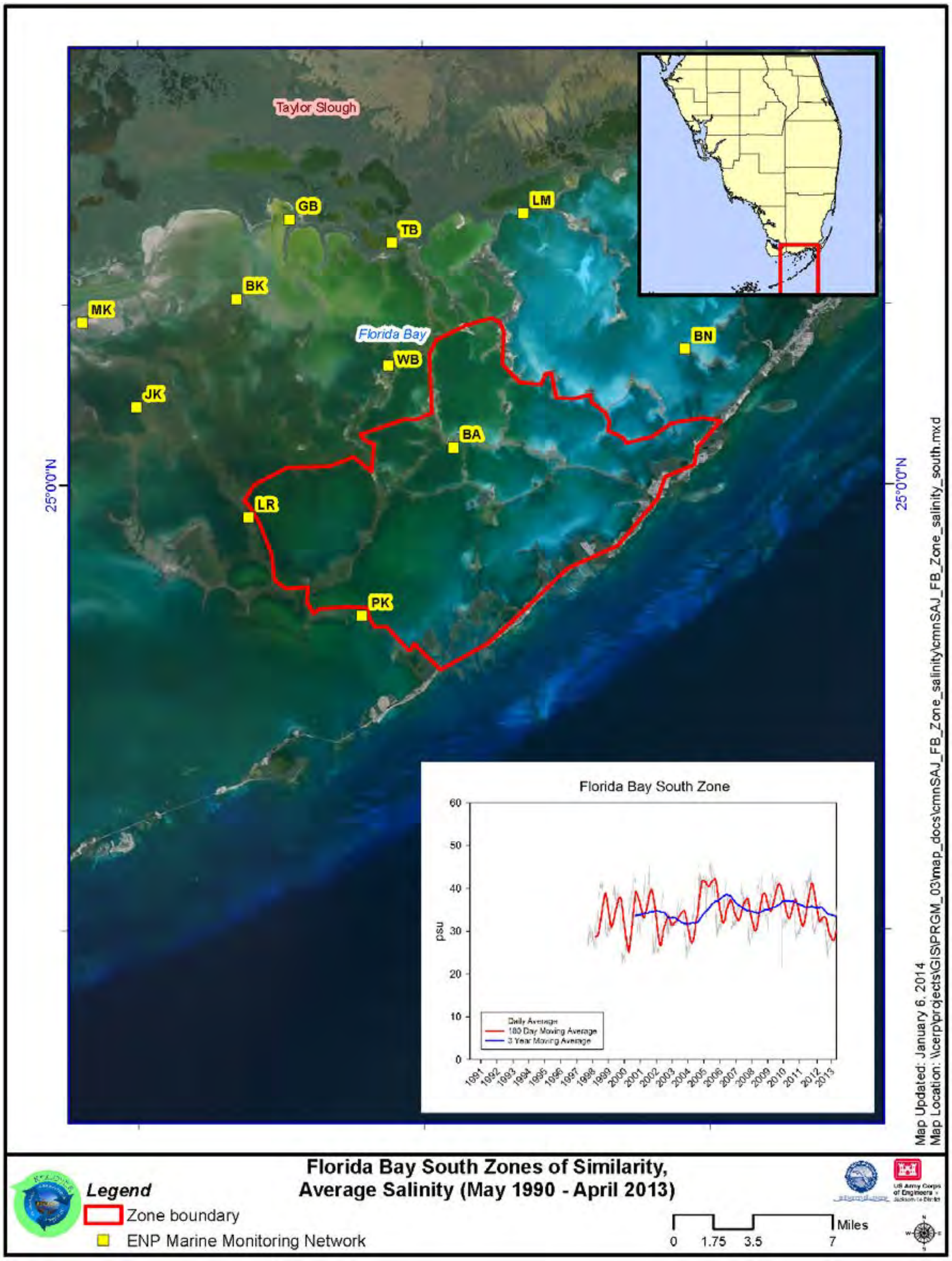


Figure 7-34. Daily, 180-day moving, and three-year moving average salinity time series in the South Florida Bay water quality zone of similarity. (Note that only stations that fall within the red boundary of the zone are included in the assessment for that zone.)

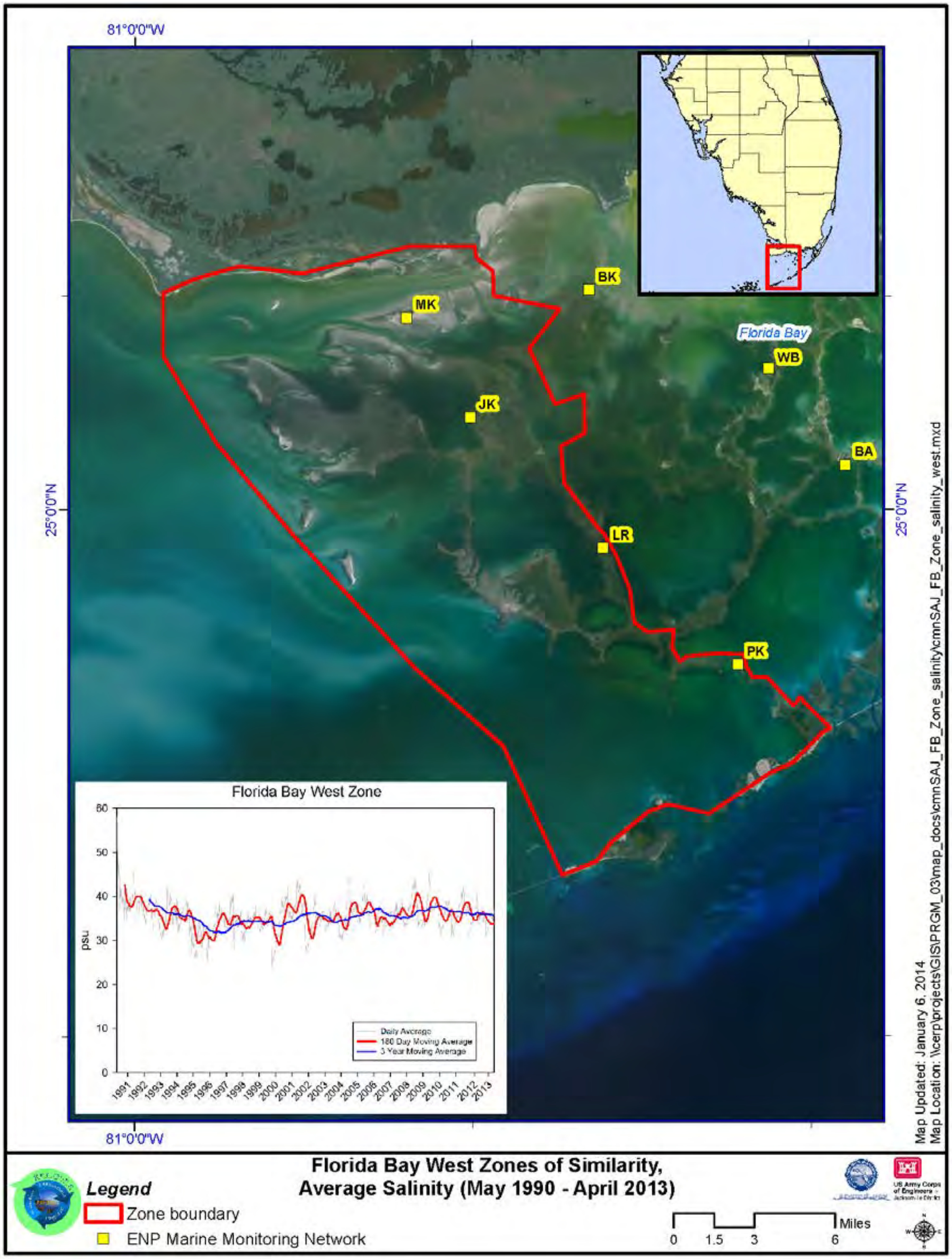


Figure 7-35. Daily, 180-day moving, and three-year moving average salinity time series in the West Florida Bay water quality zone of similarity. (Note that only stations that fall within the red boundary of the zone are included in the assessment for that zone.)

Salinity throughout the bay in the early 1990s was hyperhaline (> 40 per Venice System), with initial daily values often above 50 due to the severe 1989–1990 drought. The mid-1990s brought wetter precipitation conditions, which resulted in a dramatic drop in salinity conditions on the order of 10 to 30 for the daily average. As drier conditions prevailed through the late 1990s and early 2000s, salinity levels increased. The return of wetter precipitation conditions in the mid- to late 2000s results in the leveling off or slight decrease in salinity throughout the bay (most evident from three-year moving average in the North, East, East-Central, and Central zones). However, while the recent trend may have been toward slightly lower salinity, absolute values remained high, especially in the late wet season and early dry season; annual minima in 2008, 2009 and 2011 (evident in daily and 180-day moving average) were unusually high for most of the bay (especially in the East and North zones), and in most cases, only were exceeded by 1991 or 1992 values. Quantitative attribution of these patterns to numerous potential causes is uncertain, but is related to large-scale climatic patterns and their resultant local precipitation patterns, and may be related to sea level rise and water management; further data analysis is needed.

Figures 7-36 through **7-42** shows box-and-whisker plots of the average monthly salinity at 21 MMN stations throughout Florida Bay and Manatee Bay/Barnes Sound for two periods, WY 2000–WY 2008 and WY 2009–WY 2013. The box provides the 25th, 50th (median), and 75th percentiles and the whiskers describe the 10th and 90th percentiles. West zone stations for both time periods generally had salinity values between 30 and 40 with low variability. South zone (Bob Allen) salinity for both time periods trended between 25 and 40 with the time periods' salinity overlapping at the 10th–90th percentiles. Nearshore Central zone stations (Garfield Bight and Terrapin Bay) show greater variability over the year and between periods. Garfield Bight had higher median salinity values for every month in the WY 2009–WY 2013 period than the it had for the WY 2000–WY 2008 period. Nearby Buoy Key and Murray Key sites had a similar pattern, each with 11 monthly salinity medians higher in the recent period. East-central zone stations trend between 20 and 40 and generally overlap at the 10th–90th percentiles between periods. The greatest difference between the periods appeared to be at northeastern coastal sites; Highway Creek, Long Sound, and Little Blackwater Sound had higher late wet season and early dry season salinity in WY 2009–WY 2013 than WY 2000–WY 2008. These sites are strongly influenced by C-111 Canal management and this finding may reflect recent successful efforts to minimize seepage from Taylor Slough towards the C-111 Canal, including C-111 SCWP implementation in 2013. This inference should be considered tentative and needs to be strengthened by monitoring over a longer period of project operation. Note that these sites are in close proximity to new culverts and bridges along the U.S. Highway 1 "18-mile stretch" connecting these basins to Manatee Bay and Barnes Sound. Recent data collected at Highway Creek indicate that tidal circulation has increased in creeks that previously did not exhibit a diurnal tidal signal due to damming effect of the highway. More information can be found later in this chapter, in the U.S. Highway 1 Expansion section.

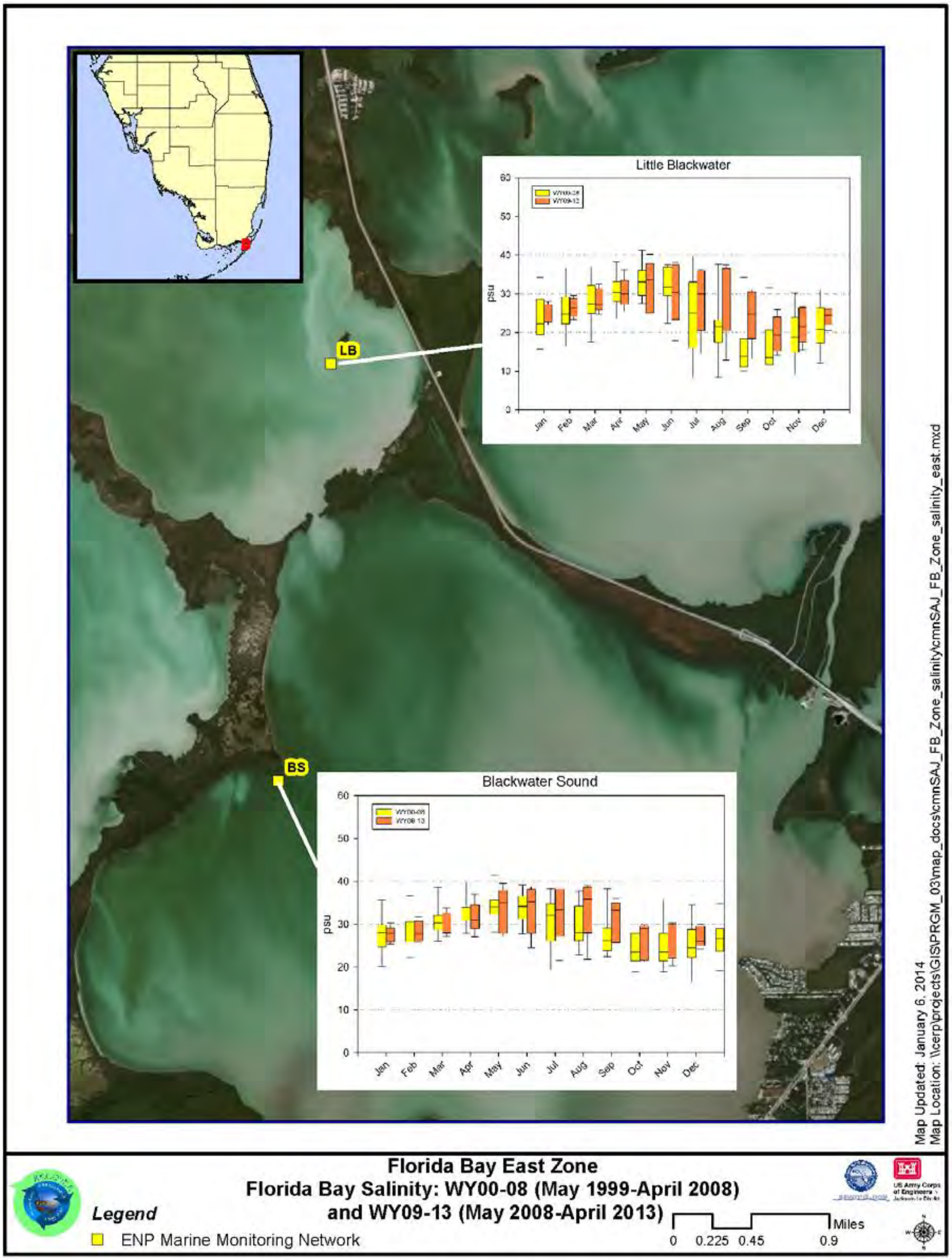


Figure 7-36. Box-and-whisker plots of monthly salinity for WY 2000–WY 2008 and WY 2009–WY 2013 at Little Blackwater (LB) and Blackwater Sounds (BS), within the East Florida Bay water quality zone of similarity.

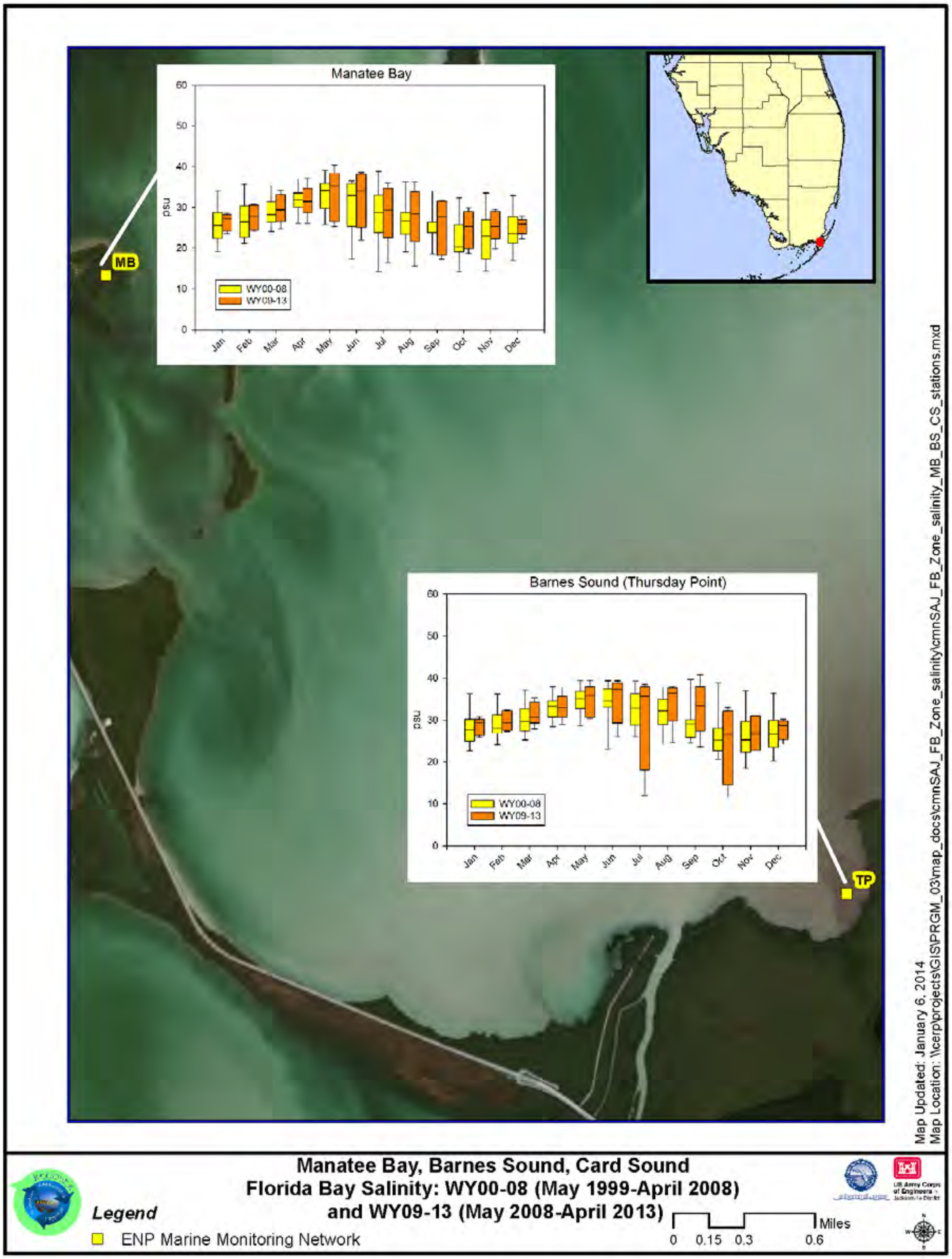


Figure 7-37. Box-and-whisker plots of monthly salinity for WY 2000–WY 2008 and WY 2009–WY 2013 at Manatee Bay (MB) and Barnes Sound (TP).

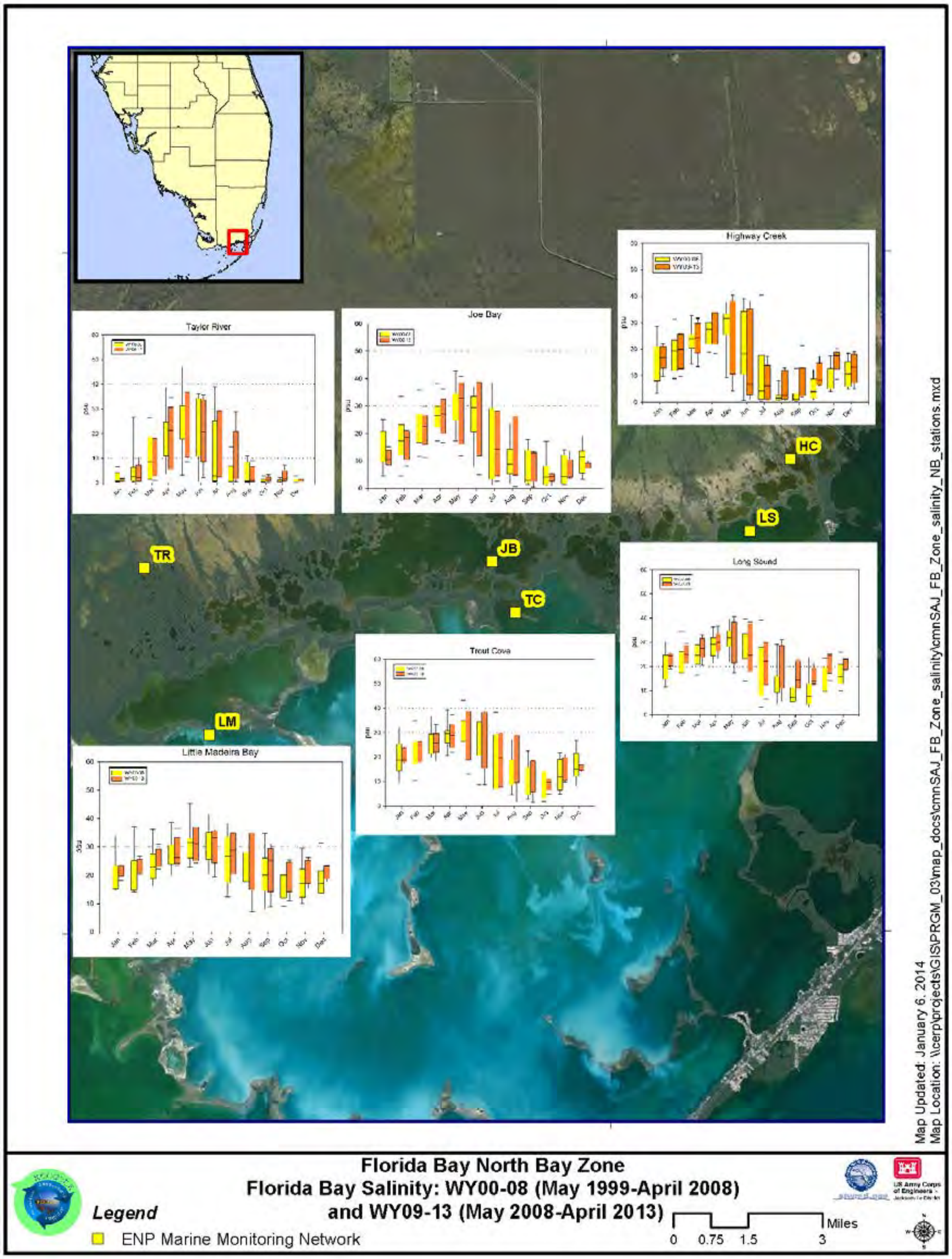


Figure 7-38. Box-and-whisker plots of monthly salinity for WY January 2000–December 2008 (in yellow)and January WY 2009– December WY 2013 (in orange) at Taylor River (TR), Joe Bay (JB), Highway Creek (HC), Little Madeira Bay (LM), Trout Cove (TC), and Long Sound (LS), in the North Bay Florida Bay water quality zone of similarity.

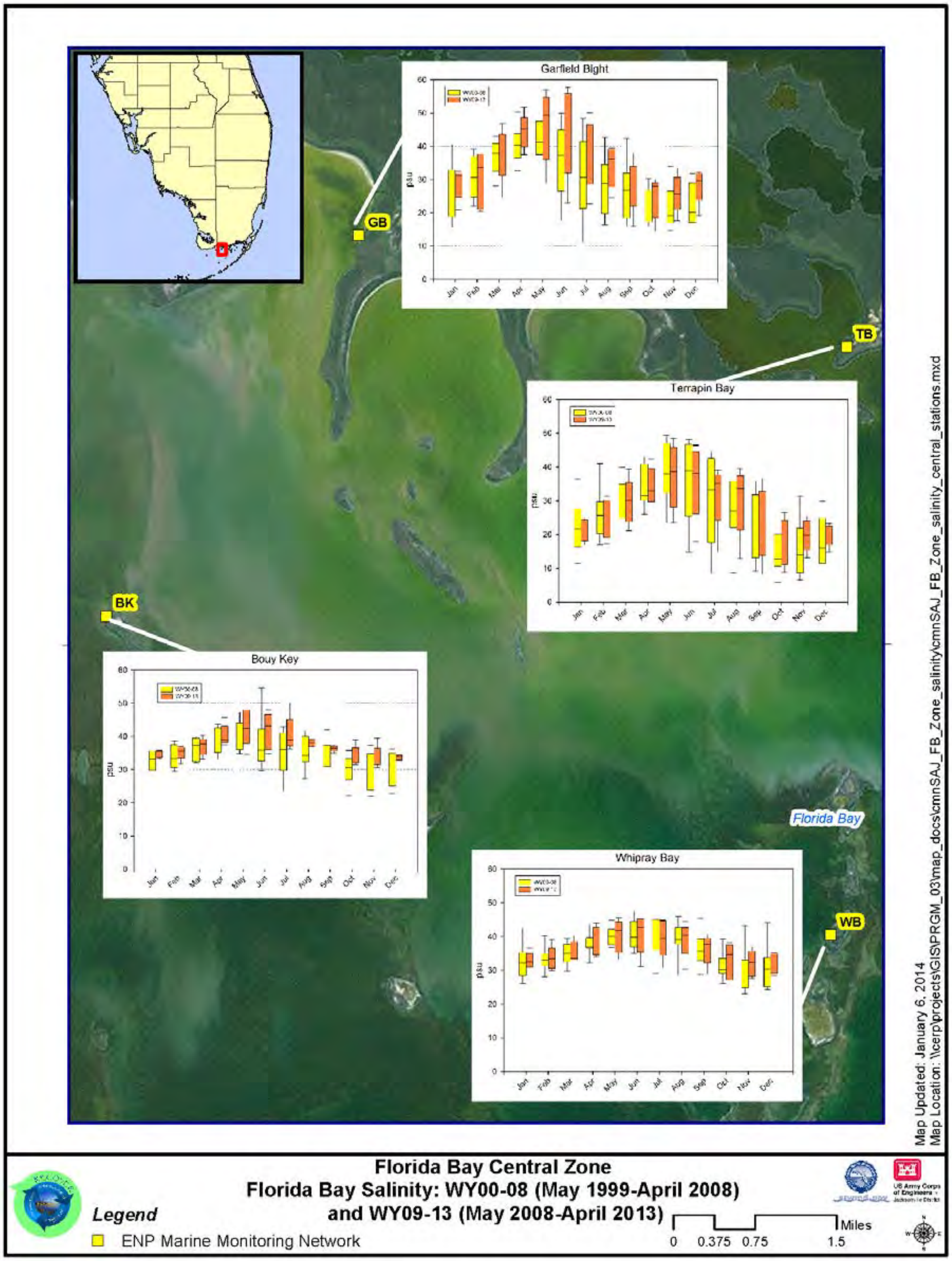


Figure 7-39. Box-and-whisker plots of monthly salinity for WY 2000–WY 2008 and WY 2009–WY 2013 at Garfield Bight (GB), Terrapin Bay (TB), Bouy Key (BK), and Whipray Bay (WB), in the Central Florida Bay water quality zone of similarity.

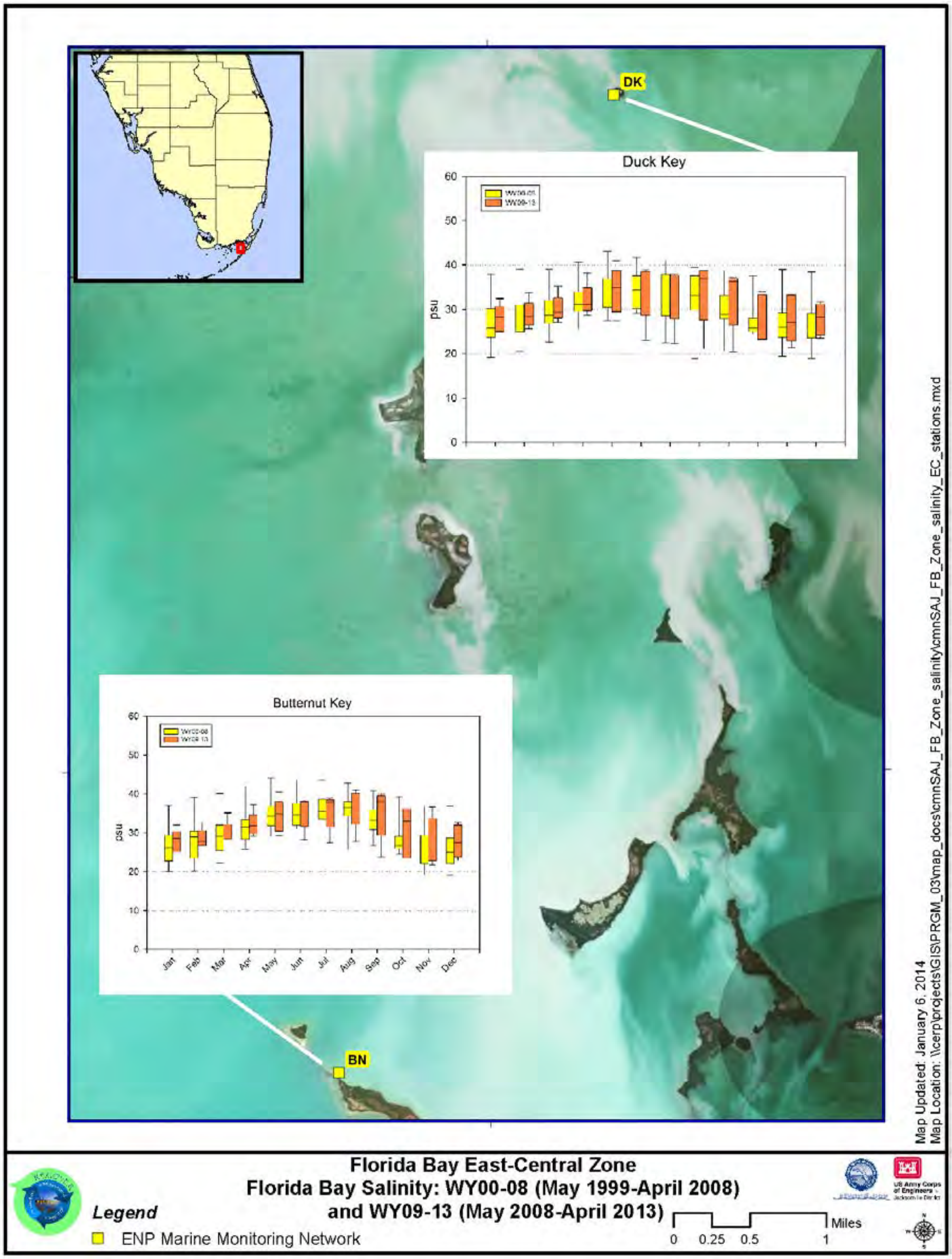


Figure 7-40. Box-and-whisker plots of monthly salinity for WY 2000–WY 2008 and WY 2009–WY 2013 at Duck Key (DK) and Butternut Key (BN), in East-Central Florida Bay water quality zone of similarity.

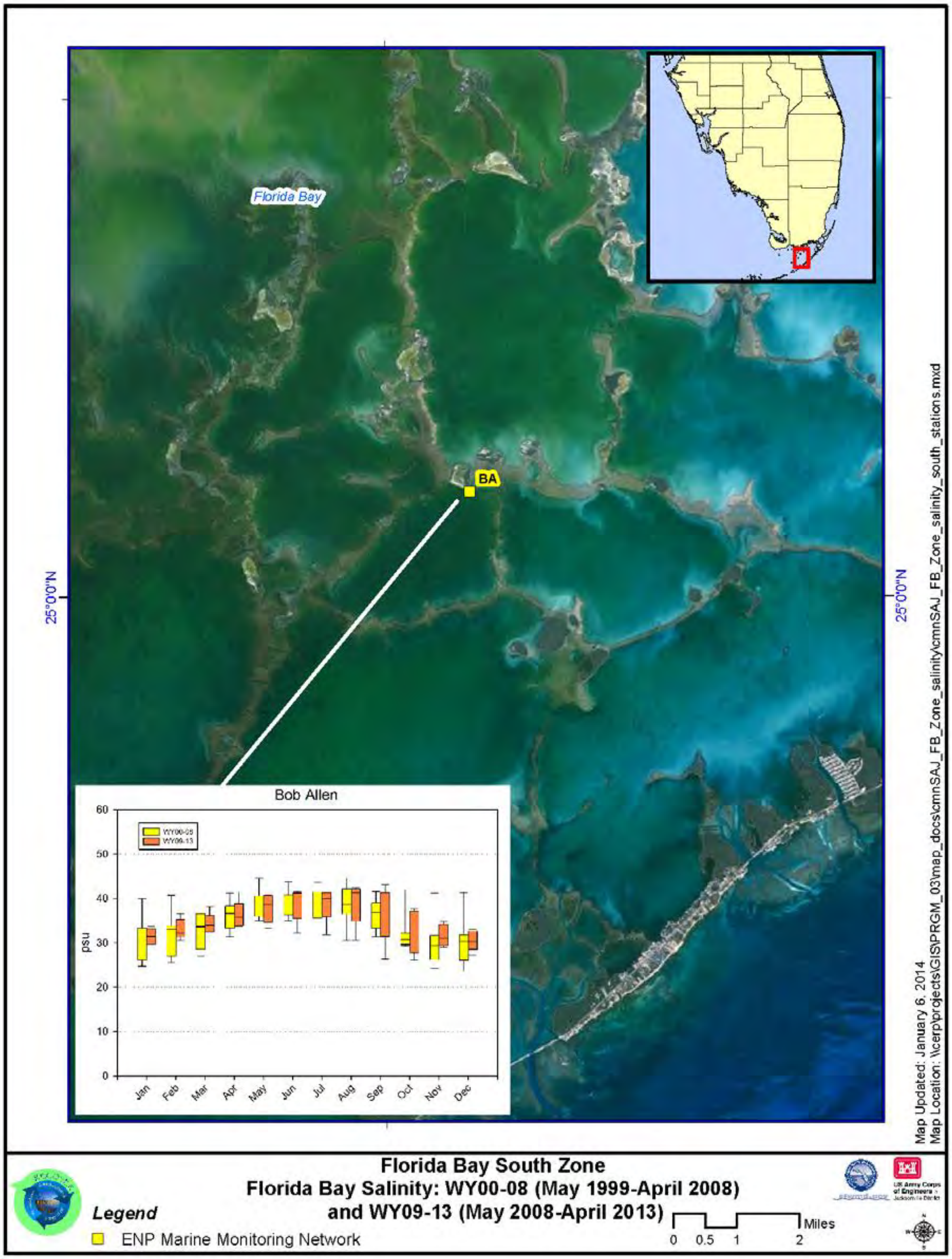


Figure 7-41. Box-and-whisker plots of monthly salinity for WY 2000–WY 2008 and WY 2009–WY 2013 at Bob Allen (BA), in the South Florida Bay water quality zone of similarity.

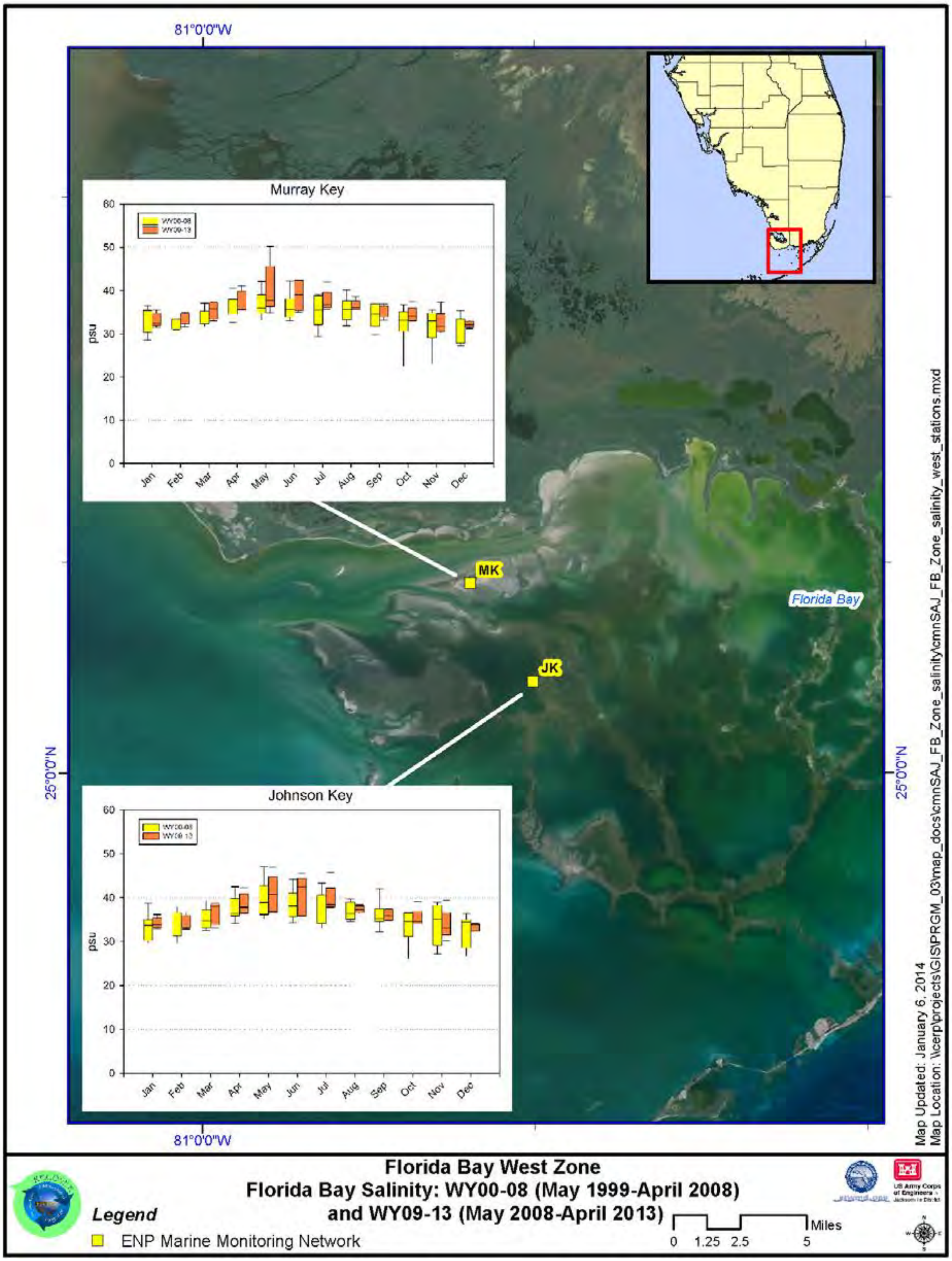


Figure 7-42. Box-and-whisker plots of monthly salinity for WY 2000–WY 2008 and WY 2009–WY 2013 at Murray Key (MK) and Johnson Key (JK), in the West Florida Bay water quality zone of similarity.

Florida Bay: RECOVER Salinity Performance Measure

Salinity conditions in Florida Bay were assessed at 17 MMN stations in Florida Bay (**Table 7-6**) for the periods WY 2000–WY 2008 (May 1999–April 2008) and WY 2009–WY 2013 (May 2008–April 2013). The *RECOVER Southern Coastal Systems Performance Measure – Salinity in Florida Bay* was applied to both periods and comparisons were made between periods. **Table 7-6** shows the actual index scores for both wet and dry seasons for both assessed periods. These results, with almost all red or yellow scores, show Florida Bay salinity conditions remained far from the RECOVER restoration target since 2000 in all parts of the bay. The regime and overlap index scores are on a scale from zero to a restoration target of 1. The offset index score is the magnitude of the salinity difference between mean salinity at a field site and the restoration target salinity at that site; the index target score is zero. **Table 7-7** shows the difference between the index scores for WY 2009–WY 2013 and WY 2000–WY 2008 (WY 2009–WY2013 minus WY2000–WY2008 for the regime and overlap scores; reversed for the offset scores). The scores show either little change or increasing salinity and decreasing performance between these two time periods for all Florida Bay zones. The only exception was Joe Bay's positive dry season change between the periods for the high salinity and offset metric scores. Wet season performance in those zones closest to the Everglades and expected to be most affected by CERP—North, East, East-Central, and Central zones—had negative wet season performance at almost all sites for all three metrics. The general decline in the metric scores baywide is of concern as much of the bay was already in a red "merits action" or yellow "merits attention" condition. Closer examination of the regime overlap scores in **Table 7-6** reveals that Long Sound, one of the only two stations reported as yellow condition during the wet season for WY 2000–WY 2008, dropped to red, with an overlap index score of 0.00, indicating that the entire assessed period failed to meet the restoration target. In fact, almost all of the regime overlap scores for both wet and dry seasons for WY 2009–WY 2013 dropped to an index score of less than 0.10 with the highest reported "red" score being 0.25 at Trout Cove during the wet season.

Table 7-6. Output from the salinity performance measure in Florida Bay for WY 2000–WY 2008 and WY 2009–WY 2013. The actual metric value is shown in each cell. The cell color indicates the salinity status score relative to a restoration target. Red indicates substantial deviations creating severe negative conditions that merit action. Yellow indicates conditions do not meet restoration targets and merit action. Green indicates conditions have met restoration targets.

	WY00-08						WY09-13					
	Regime Overlap		High Salinity		Offset		Regime Overlap		High Salinity		Offset	
MMN Station	Wet Season	Dry Season	Wet Season	Dry Season	Wet Season	Dry Season	Wet Season	Dry Season	Wet Season	Dry Season	Wet Season	Dry Season
Joe Bay (JB)	0.51	0.01	0.61	0.27	5.09	11.58	0.45	0.01	0.56	0.40	5.38	9.71
Little Madeira Bay (LM)	0.13	0.13	0.62	0.60	6.84	6.37	0.00	0.00	0.43	0.44	8.78	7.73
Long Sound (LS)	0.40	0.00	0.74	0.29	4.24	11.14	0.05	0.00	0.37	0.12	8.93	13.45
Trout Cove (TC)	0.30	0.00	0.41	0.18	5.14	12.95	0.25	0.00	0.27	0.14	7.13	12.45
North Bay average	0.34	0.03	0.59	0.34	5.33	10.51	0.17	0.00	0.41	0.28	7.56	10.83
Blackwater Sound (BS)	0.16	0.00	0.61	0.47	4.74	5.03	0.03	0.05	0.38	0.57	6.67	4.69
Little Blackwater Sound (LB)	0.32	0.00	0.69	0.38	4.29	8.11	0.04	0.00	0.42	0.33	8.21	8.65
East average	0.21	0.00	0.65	0.43	4.51	6.57	0.03	0.03	0.40	0.45	7.44	6.67
Butternut Key (BN)	0.07	0.28	0.41	0.73	7.33	3.22	0.02	0.11	0.38	0.68	8.04	3.87
Duck Key (DK)	0.00	0.00	0.44	0.60	7.55	5.04	0.00	0.00	0.38	0.45	8.50	5.69
East-central	0.03	0.14	0.43	0.66	7.54	4.13	0.01	0.06	0.38	0.57	8.57	4.78
Buoy Key (BK)	0.15	0.04	0.57	0.45	4.49	4.76	0.00	0.00	0.31	0.34	7.93	6.55
Garfield Bight (GB)	0.32	0.10	0.65	0.47	5.48	9.06	0.02	0.00	0.58	0.42	10.28	11.62
Terrapin Bay (TB)	0.27	0.08	0.62	0.54	8.43	9.64	0.03	0.01	0.41	0.48	10.84	9.52
Whipray Basin (WB)	0.11	0.13	0.49	0.65	6.32	3.65	0.04	0.30	0.46	0.74	6.94	4.08
Central average	0.21	0.09	0.58	0.53	6.18	6.78	0.02	0.05	0.44	0.50	9.00	7.94
Bob Allen Key (BA)	0.05	0.21	0.50	0.69	6.08	2.91	0.00	0.06	0.42	0.71	6.04	3.39
South average	0.05	0.21	0.50	0.69	6.08	2.91	0.00	0.06	0.42	0.71	6.94	3.39
Johnson Key (JK)	0.03	0.21	0.45	0.57	4.72	3.04	0.00	0.06	0.29	0.55	5.65	4.00
Little Rabbit Key (LR)	0.05	0.20	0.38	0.56	4.29	2.16	0.03	0.09	0.28	0.60	4.00	2.96
Murray Key (MK)	0.01	0.12	0.44	0.62	3.82	2.50	0.00	0.00	0.28	0.56	5.36	4.02
Peterson Key (PK)	0.03	0.17	0.37	0.56	2.98	1.15	0.00	0.00	0.35	0.43	3.53	2.37
West average	0.01	0.17	0.41	0.58	3.96	2.21	0.01	0.02	0.30	0.54	4.86	3.34

Table 7-7. Difference in salinity performance measure indices of WY 2009–WY 2013 from WY 2000–WY2008. The actual difference value is shown in the cell. The magnitude of change from WY 2000–WY 2008 to WY 2009–WY 2013 (difference of WY 2009–WY 2013 minus WY 2000–WY 2008) for regime and high salinity indexes: WY 2000–WY 2008 minus WY 2009–WY 2013 for offset index) are denoted by red (≤ -0.1 for the regime overlap and high salinity indices and ≤ -1.0 for the offset metric), no cell color (+0.1 < x < -0.1 for the regime and high salinity indices and +1.0 < x < -1.0 for the offset metric), and green (≥ +0.1 for the regime overlap and high salinity indices and ≥ +1.0 for the offset metric).

	WY00-08 vs WY09-13					
	Regime Overlap		High Salinity		Offset	
MMN Station	Wet Season	Dry Season	Wet Season	Dry Season	Wet Season	Dry Season
Joe Bay (JB)	-0.06	0.00	-0.05	0.13	-0.30	1.86
Little Madeira Bay (LM)	-0.13	-0.13	-0.18	-0.15	-1.95	-1.36
Long Sound (LS)	-0.40	0.00	-0.37	-0.17	-4.69	-2.31
Trout Cove (TC)	-0.06	0.00	-0.14	-0.04	-1.98	0.50
North Bay average	-0.16	-0.03	-0.18	-0.06	-2.23	-0.33
Blackwater Sound (BS)	-0.07	0.05	-0.23	0.09	-1.94	0.34
Little Blackwater Sound (LB)	-0.28	0.00	-0.27	-0.06	-3.92	-0.54
East average	-0.17	0.03	-0.25	0.02	-2.93	-0.10
Butternut Key (BN)	-0.05	-0.17	-0.03	-0.05	-1.11	-0.65
Duck Key (DK)	0.00	0.00	-0.06	-0.15	-0.95	-0.65
East-central	-0.02	-0.08	-0.04	-0.10	-1.03	-0.65
Buoy Key (BK)	-0.15	-0.04	-0.25	-0.11	-3.44	-1.79
Garfield Bight (GB)	-0.30	-0.10	-0.07	-0.05	-4.80	-2.57
Terrapin Bay (TB)	-0.24	-0.07	-0.21	-0.06	-2.41	0.12
Whipray Basin (WB)	-0.07	0.07	-0.04	0.09	-0.62	-0.43
Central average	-0.19	-0.04	-0.14	-0.03	-2.82	-1.17
Bob Allen Key (BA)	-0.05	-0.15	-0.08	0.03	-0.86	-0.48
South average	-0.05	-0.15	-0.08	0.03	-0.86	-0.48
Johnson Key (JK)	-0.03	-0.21	-0.16	-0.02	-0.92	-0.96
Little Rabbit Key (LR)	-0.02	-0.11	-0.10	0.04	-0.61	-0.80
Murray Key (MK)	-0.04	-0.12	-0.15	-0.06	-1.54	-1.52
Peterson Key (PK)	-0.03	-0.17	-0.02	-0.13	-0.55	-1.22
West average	-0.03	-0.15	-0.11	-0.05	-0.91	-1.13

Florida Bay and Lower Southwest Coast Water Quality

Water quality monitoring and assessment in the SCS continues providing information on the status and trends of nutrient concentrations, phytoplankton blooms, and light conditions. These water quality attributes are influenced by the Everglades watershed, including water management and restoration projects. These attributes are also influenced by other factors external to coastal waters, such as dynamics on the ocean boundary and climate, especially local rainfall, as well as internal ecological dynamics, including seagrass bed growth or mortality and the feedback loops of salinity, nutrient cycling, phytoplankton growth, light availability, and seagrass condition (Rudnick et al. 2005). Given these complexities, attribution of a given water quality pattern to a single cause (particularly restoration) is difficult, requiring long-term assessment of freshwater flow and associated nutrient inputs, along with assessment of internal bay dynamics. In this report, an overview is presented of the status and trends of Florida Bay water quality, focusing on Chla concentrations as a proxy of phytoplankton biomass and an indicator of overall water quality conditions. Results from WY 2013 are relevant to the assessment of the initial effects of the C-111 SCWP, which began operations in July 2012, but given the short operational period of record, must be considered only as preliminary.

Water quality monitoring in the SCS is not currently funded by RECOVER. RECOVER partially funded NOAA water quality monitoring until 2010, with regional synoptic mapping via an array of shipboard sensors and continuous-flow sampling (Kelble et al 2007). SFWMD has independently funded coastal water quality monitoring of fixed stations with grab sampling in Florida Bay since 1991. Late in 2011, sampling frequency was decreased from monthly to bimonthly. SFWMD researchers also synoptically measure water quality parameters using a continuous-flow system (Madden et al. 1992) near the northern Florida Bay shoreline on at least a quarterly basis to assess fine-scale spatial patterns in association with wetland hydrologic features (creeks and ridges).

Water quality in the SCS was assessed based on the stoplight indicator of Chla used by the South Florida Ecosystem Restoration Task Force. Here, directional trends are incorporated to provide a more comprehensive analysis with potential signs of degradation and restoration captured by the trend in this data. The indicator was assessed using data collected by SFWMD, NOAA, and Miami-Dade DERM. The geometric mean of each water year from 2005 through 2013 was compared against baselines medians and 75th percentile according to Boyer et al. (2009). In addition, the entire data set was analyzed for long-term linear trends in Chla and five-year running linear trends for each year from WY 2005 through WY 2013 to determine whether conditions were improving (e.g., Chla was decreasing) or degrading (e.g., Chla was increasing) in ten subregions delineated statistically within the SCS (Boyer et al. 2009).

The Chla indicator in WY 2012 and WY 2013 was dominated by yellow (caution) stoplights throughout much of the SCS (**Figure 7-45**). This indicates Chla concentrations were above the baseline median, but below the baseline 75th percentile in most subregions. The indicator was red (severe negative conditions) in the Southwest Florida Shelf in WY 2012 and Northeast Florida Bay in WY 2013. Restoration targets, which in this case are to cause no harm to the system via any increase in phytoplankton blooms, were only met (with green indicator result) for both years in West Florida Bay.

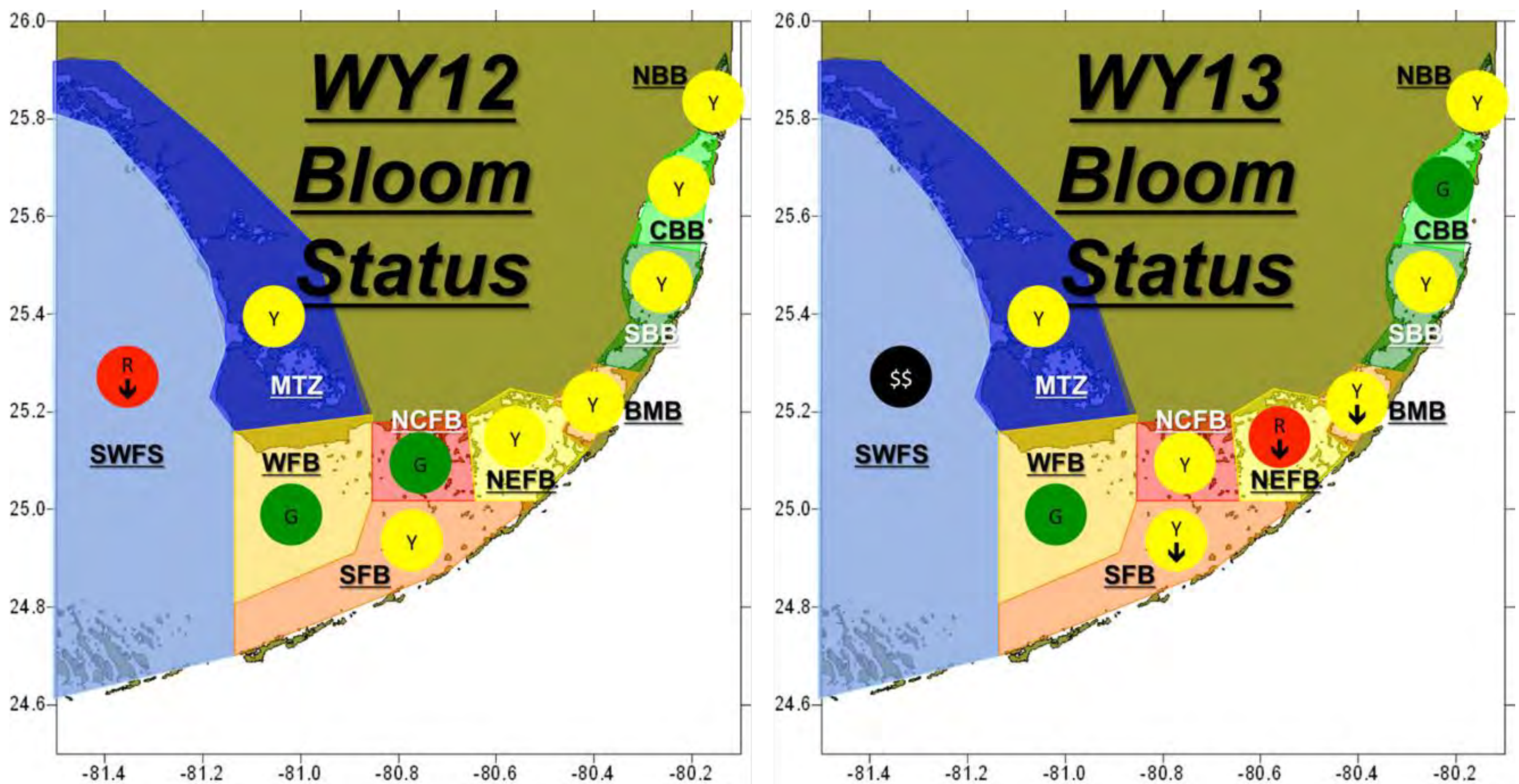


Figure 7-43. Map showing the spatial distribution of the water quality indicator status (red, yellow and green stoplights) throughout the SCS in WY 2012 and WY 2013. The arrows indicate trends of degrading water quality (downward arrows) and improving water quality (upward pointing arrows). (NBB – North Biscayne Bay, CBB – Central Biscayne Bay, SBB – South Biscayne Bay, BMB – Barnes, Manatee, and Blackwater sounds, NEFB – Northeast Florida Bay, NCFB – North-Central Florida Bay, SFB – South Florida Bay, WFB – West Florida Bay, MTZ – Mangrove Transition Zone, and SWFS – Southwest Florida Shelf.)

To provide temporal context to this indicator, it was assessed in each subregion for all years since WY 2005 (**Table 7-8**). Subregions of particular concern are those with negative trends (degrading conditions) or severe negative conditions (red): Northeast Florida Bay; Barnes Sound, Manatee Bay, and Blackwater Sound; South Florida Bay; and the Southwest Florida Shelf. All had significant trends of degradation and were not meeting targets.

A subregion of serious concern is the Southwest Florida Shelf, which is influenced by freshwater inflow from the Caloosahatchee River Estuary, the Big Cypress region (via Lostmans River), and SRS, as well as by the Gulf of Mexico. This subregion had severe negative conditions in WY 2012 and a significant trend of degradation over the past five years. Additionally, the long-term trend shows significant linear increases in Chla throughout the period of record suggestive of long-term degrading water quality conditions (**Figure 7-46**). However, in WY 2013, data were only collected in June and August prior to the cessation of NOAA's water quality monitoring program in south Florida. This level of data is insufficient to assess the indicator for WY 2013 and water quality monitoring in this subregion has terminated. The adjacent Mangrove Transition Zone of the adjacent southwest coast, where bimonthly sampling continues, also had a significant long-term increase in Chla, but the indicator for this region remains yellow, without a significant five-year trend.

Another subregion of serious concern is Northeast Florida Bay, which is strongly influenced by freshwater flow from Taylor Slough and the C-111 Canal. While the long-term trend in this subregion shows significant improvement, largely because of high Chla in the early 1990s (**Figure 7-46**), there has been a significant degradation of water quality in this subregion over the past five years (see trend line in **Figure 7-46**; transition from green to red indicator score from WY 2010 through WY 2013 in **Table 7-8**).

Northeastern Florida Bay's negative status and trend may be associated with a combination of climatic conditions and possibly with the initial operations of the C-111 SCWP. Rainfall patterns in the region were unusual in WY 2012 and WY 2013, with high rains in April and May. The relationship between such late season rain, coastal creek discharge, and nutrient export from the wetland remains to be investigated. It is notable that an inverse relationship between salinity and Chla existed in Northeast Florida Bay (**Figure 7-47**) in the early WY 2013 wet season, with highest Chla in areas closest to freshwater flow sources (especially Little Madeira Bay and Joe Bay). The fine-scale continuous-flow mapping results in this figure also show that Chla concentrations in areas of these two bays were higher than those measured in the same month in grab samples at monitoring sites (with values up to 9 parts per billion [ppb]). This difference likely reflects the spatial complexity and temporal variability of this nearshore region; the bimonthly grab sample sites may not be representative of the entire area at a given time. Current bimonthly grab sampling for water quality monitoring and CERP assessment appears problematic in this dynamic zone. While C111 SCWP operations started in summer 2012, concurrent with a period of increasing Chla (**Figures 7-46** and **7-47**), any attribution of salinity and water quality changes to these operations is premature; this requires a longer period of record and extensive comparisons with pre-project data sets.

Table 7-8. Chla stoplight indicator chart. The color indicates the Chla concentration relative to the restoration target (baseline median). Red indicates the concentration was above the baseline 75^{th} percentile and merits action. Yellow indicates the concentration was between the baseline median and 75^{th} percentiles and merits attention. Green indicates the concentration was at or below the baseline median and has met the restoration target. The directional arrow within some of the colored circles indicates if the Chla concentrations exhibit a positive (up arrow) or negative (down arrow) trend.

Chlorophyll *a* Indicator	WY 2005	WY 2006	WY 2007	WY 2008	WY 2009	WY 2010	WY 2011	WY 2012	WY 2013	Current Status
Southwest Florida Shelf	Y	Y	G	G	Y	Y	Y ↓	R ↓	$$	Southwest Florida Shelf was showing serious signs of water quality degradation with annual concentrations greater than 75% of baseline data and a significant increasing trend for the past 2 years. Unfortunately, data collection in the Southwest Florida Shelf ceased in August 2012 making it impossible to assess this subregion for WY 2013.
Mangrove Transition Zone	G	Y	Y	Y	Y	Y	Y	Y	Y	Mangrove Transition Zone values have been in the caution range for the past 8 years and there is a long-term increasing trend when examining the entire period of record.
West Florida Bay	G	G	G	G	G	G	G	G	G	For all 9 years of the assessment, West Florida Bay has had ideal Chla concentrations and a decreasing trend for the period of record indicating no signs of degradation.
South Florida Bay	G	R	R	R ↓	G	Y	Y	Y	Y ↓	South Florida Bay experienced a significant cyanobacterial bloom from WY 2006 through WY 2008. Conditions returned to baseline in WY 2009, but have since been degrading and are currently in the caution zone with a significant linear trend of degradation.
North-Central Florida Bay	G	Y	Y	G	G	G	G	G	Y	North Central Florida Bay concentrations showed a slight sign of degradation in WY 2013; moving just above the baseline for the first time in 6 years.
Northeast Florida Bay	Y	Y	Y	G	G	G	Y	Y	R ↓	Chla has been increasing linearly in Northeast Florida Bay since WY 2009 and is currently significantly above baseline values and a cause for serious concern.
Barnes Sound, Manatee Bay, & Blackwater Sound	G	R	R ↓	R	G	G	Y	Y	Y ↓	Barnes Sound, Manatee Bay, and Blackwater Sound experienced an unusual (compared to the other years of data; pre-2005 conditions not shown) cyanobacterial bloom from WY 2006 through WY 2008. This bloom was initiated by a spike in phosphorous that was a result of a combination of adjacent road construction and canal releases associated with the 2005 hurricane season. This region is currently showing signs for concern with a significant trend of degrading water quality and concentration greater than the baseline median.
South Biscayne Bay	Y ↓	R ↓	R ↓	Y	Y	Y ↑	R	Y	Y	Chla in South Biscayne Bay has been consistently above baseline every year since assessment began. However, it currently is not showing a degradation trend and remains less than 0.5 ug/L; a value considered oligotrophic.
Central Biscayne Bay	Y	R ↓	Y	Y	Y	Y	Y	Y	G	Chla in Central Biscayne Bay was below the baseline median value in WY 2013 for the first time since assessment began in WY 2005.
North Biscayne Bay	Y	Y	Y ↓	Y	Y	Y	Y	Y	Y	Chla in North Biscayne Bay has remained above the baseline median concentrations since assessment begin in WY 2005, but it continues to be below the 75^{th} percentile of baseline concentrations with no significant linear trend, except WY 2003 through WY 2007.

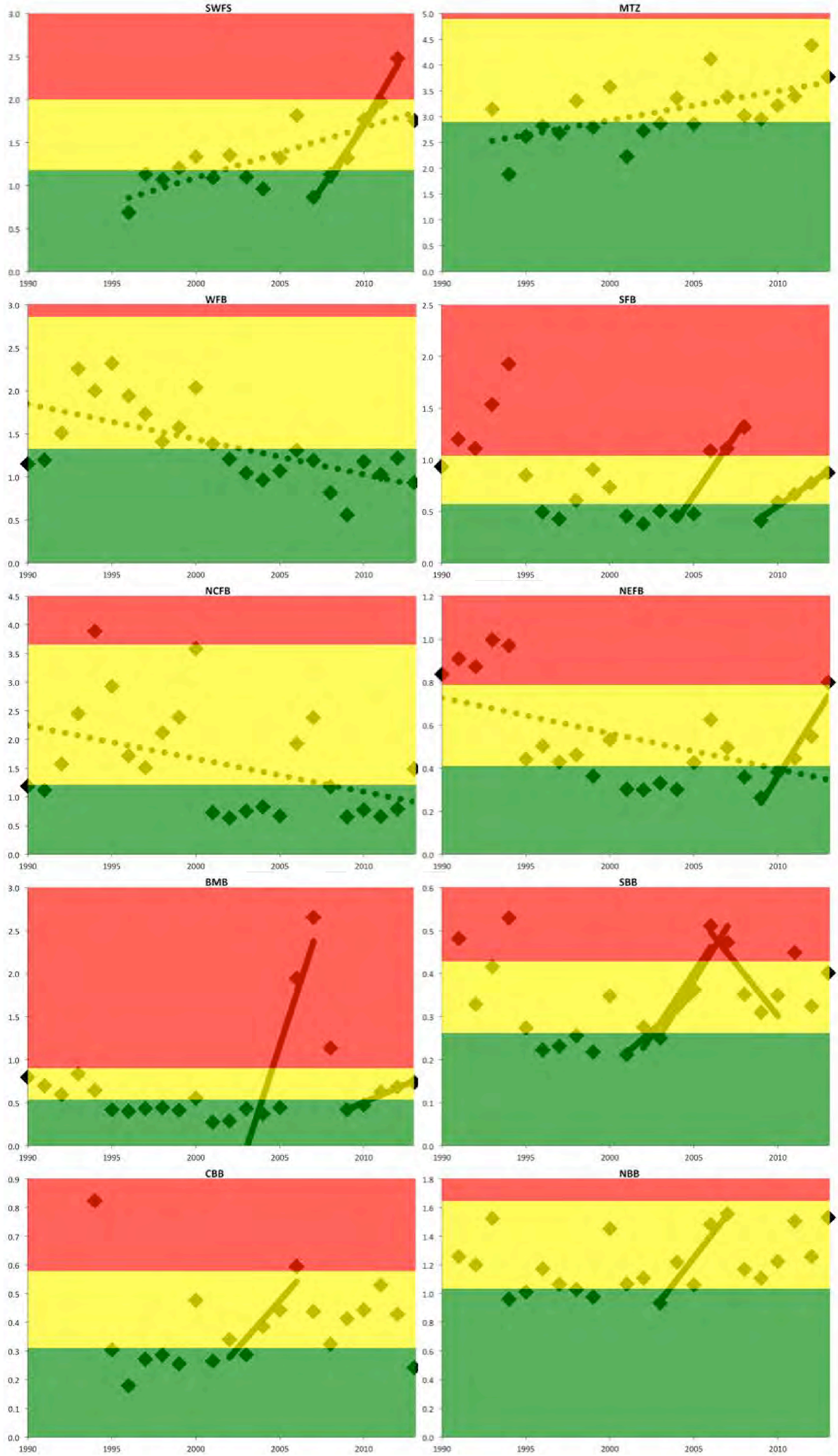


Figure 7-44. These plots show the annual geometric mean Chla concentrations in 10 subregions of the SCS. The green, yellow, and red shading show where each year scores relative to the indicator criteria established for Chla in SCS (Boyer et al. 2009). The solid lines depict statistically significant 5-year linear trends in Chla from WY2005 through WY2013. The dashed lines depict statistically significant long-term linear trends in Chla. (SWFS – Southwest Florida Shelf, MTZ – Mangrove Transition Zone, WFB – West Florida Bay, SFB – South Florida Bay, NCFB – North-Central Florida Bay, NEFB – Northeast Florida Bay, BMB – Barnes, Manatee and Blackwater sounds, SBB – South Biscayne Bay, CBB – Central Biscayne Bay, and NBB – North Biscayne Bay.)

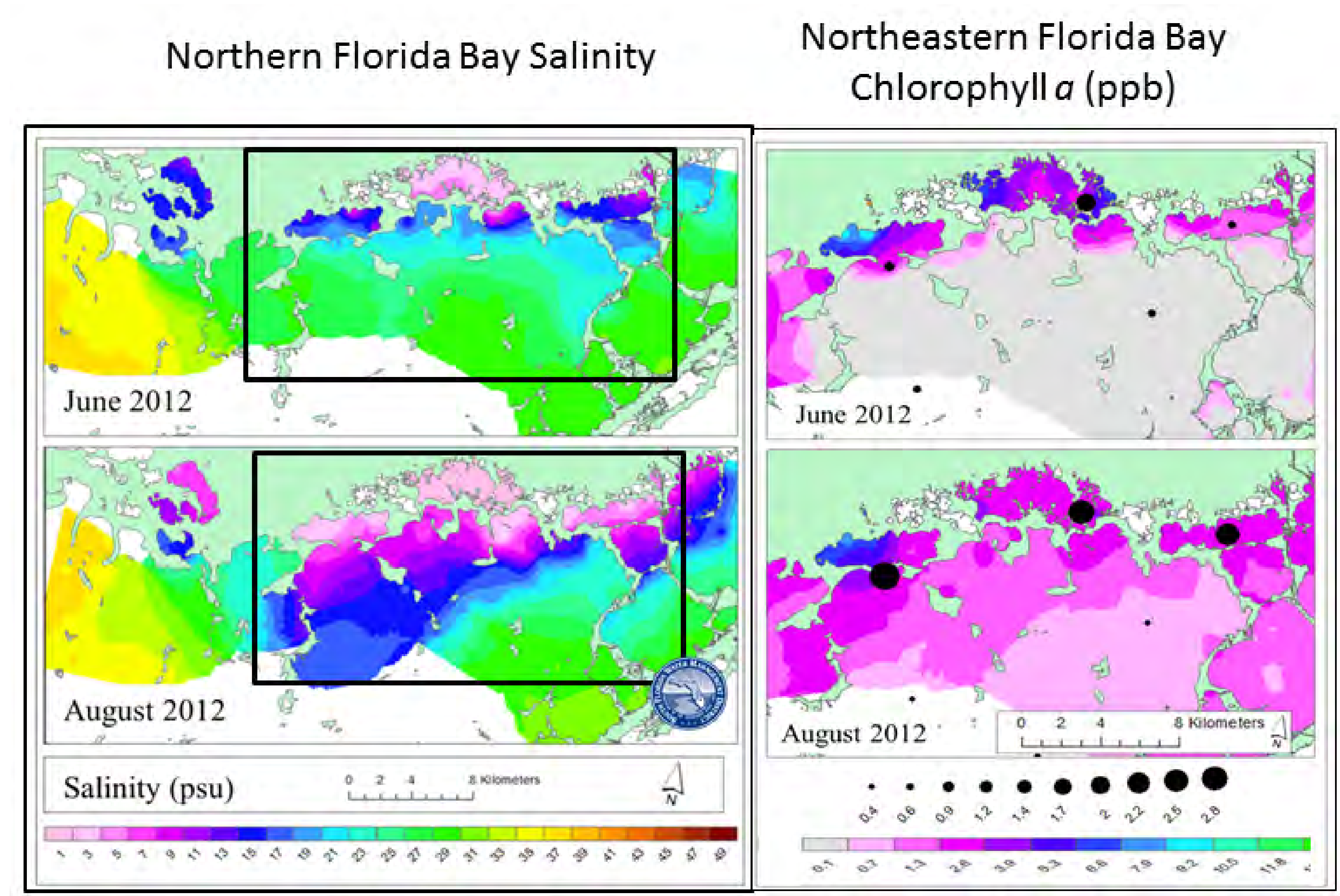


Figure 7-45. Continuous flow mapping of Florida Bay waters' specific conductivity and in-vivo Chla fluorescence, with estimated salinity and Chla concentrations, in the early WY 2012 wet season. Chla map domain shows as black rectangle on salinity maps. Black dots show values from SFWMD monitoring network grab samples. From Madden and colleagues, SFWMD.

Lastly, it should be noted that North Central Florida Bay, which experienced ecologically damaging cyanobacterial blooms during the 1990s, has experienced a period from WY 2008 through WY 2012 when it met water quality targets. In WY2013, Chla concentrations increased in North Central Florida Bay to exceed the baseline median, but only did so by approximately 0.2 ppb and there was no significant trend associated with this increase.

Overall, these results point toward the continuing vulnerability of the SCS estuaries and the South Florida Shelf to phytoplankton blooms. The finding of high Chla in 2012 and 2013 in Northeast Florida Bay should raise awareness of the importance of sustained water quality monitoring with appropriate temporal and spatial scales downstream of restoration projects. Initiation of such projects entail ecosystem disturbance as the system transitions from its current state to a restored state. This includes changes in hydrology, salinity, and potentially nutrient availability, and these changes may promote phytoplankton growth in some estuarine areas. Such responses are expected to be transitional and temporary. For now, the findings cannot point to such a transitional disturbance associated with CERP; further evidence is needed.

Florida Bay and Lower Southwest Coast Submerged Aquatic Vegetation

Florida Bay supports one of the world's most extensive seagrass meadows, and is located at the base of the Everglades hydrological system. Because seagrasses integrate net changes in water quality parameters that tend to exhibit rapid and wide fluctuations when measured directly (e.g., salinity, light availability and nutrient levels), Florida Bay seagrasses were chosen as an indicator for SCS, and as a systemwide indicator for Everglades restoration assessment (Doren et al. 2008).

The South Florida FHAP has provided data on the community characteristics of SAV, which includes seagrasses and macroalgae, in ten basins within Florida Bay since 1995. As a RECOVER MAP component, the geographic scope of the South Florida FHAP was expanded in 2005 to include three additional basins in Florida Bay and four locations in the southwestern Everglades, an area also expected to reflect CERP-associated water quality changes (**Figure 7-48**). Additionally, Miami-Dade Regulatory and Economic Resources provides data on the SAV community within twelve coastal embayments along the shoreline of northeastern Florida Bay. Spatially comprehensive data regarding the status and trends of SAV distribution, abundance and species composition will be used to assess ecosystem responses to CERP implementation on a near real-time basis, and to weigh alternative restoration options.

Data and Methods

South Florida FHAP conducts SAV surveys annually, at the end of the dry season (May–June) when salinity stress on seagrasses is typically highest. Monitoring stations are determined using a systematic-random sampling design. Each basin is divided into 30 tessellated hexagonal grid cells (**Figure 7-48** inset), and a single station position is randomly chosen from within each grid cell during each monitoring event. Each station is subsampled with eight 0.25-m^2 quadrats that are placed haphazardly on the bottom.

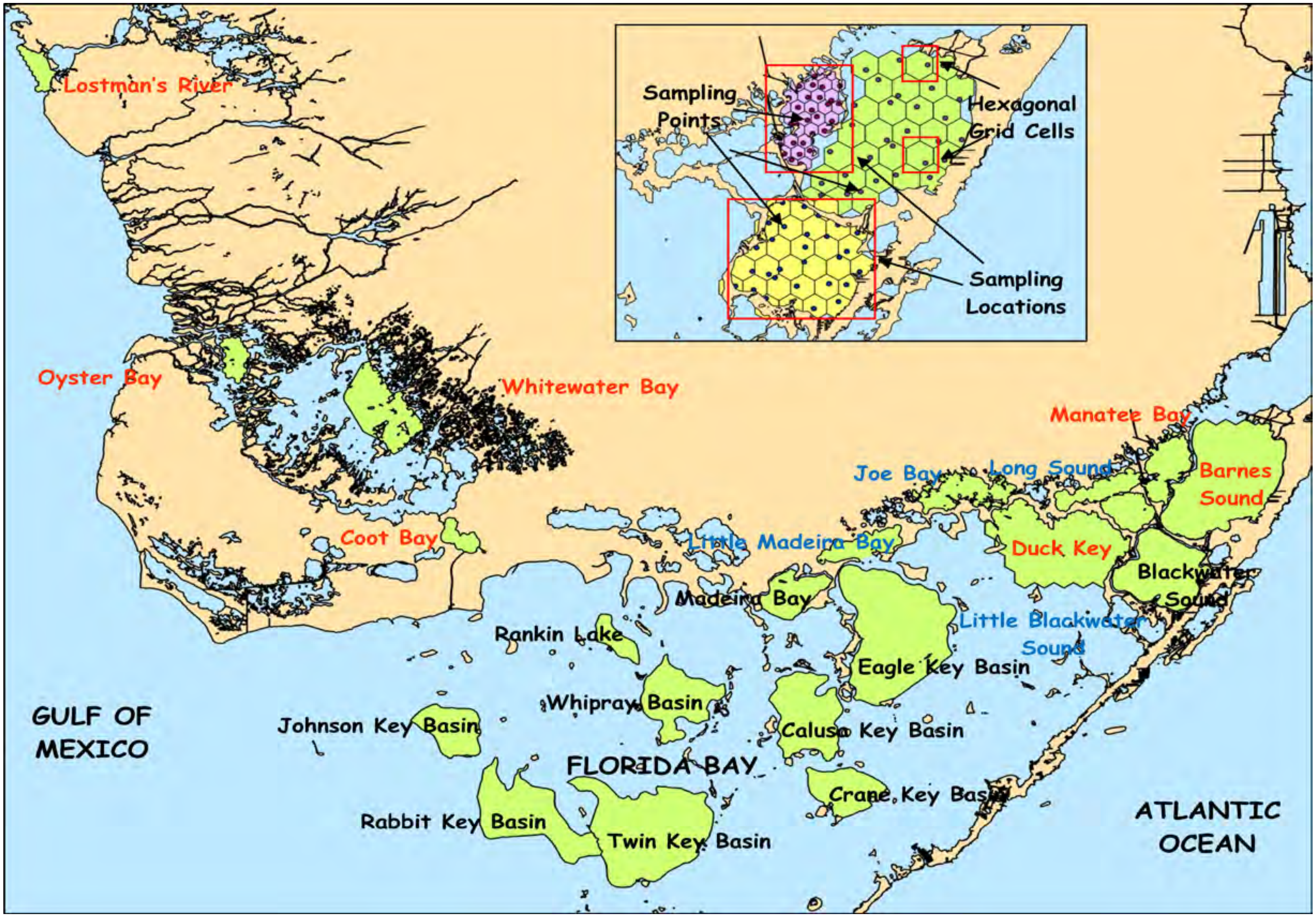


Figure 7-46. Map depicting the South Florida FHAP sampling locations and sampling design (see inset map). Locations with black labels have been sampled since 1995; those with red labels since 2005; and those with blue labels are sampled by Miami-Dade Regulatory and Economic Resources.

Miami-Dade Regulatory and Economic Resources surveys SAV quarterly within each of twelve nearshore embayments. Depending on basin size, each basin is divided into twelve or four subbasins, each containing 9 possible fixed sampling stations. One sampling station per subbasin is randomly selected for each monitoring event. The four basins used to describe seagrass community structure in the section below each have twelve subbasins. Each station is subsampled with four 0.25-m^2 quadrats placed haphazardly on the bottom.

Benthic cover (e.g., bottom occlusion) is visually estimated using a modified BBCA (Fourqurean et al. 2002). All species occurring within each 0.25-m^2 quadrat are assigned a BBCA value according to the following scale: 0 = absent; 0.1 = solitary, < 5% cover; 0.5 = few, < 5% cover; 1 = numerous, < 5% cover; 2 = 6%–25% cover; 3 = 26%–50% cover; 4 = 51%–75% cover; and 5 = 76%–100% cover. In addition to individual SAV species, total seagrass cover and total macroalgal cover are also assessed, and when *Thalassia testudinum* is present in South Florida FHAP basins, short-shoots are collected to determine leaf epiphyte biomass and shoot morphological characteristics.

Results

Seagrass Community Structure in Florida Bay – Spring 2012

Contour plots illustrating seagrass distribution, abundance, and species composition in spring 2012 were produced by kriging BBCA data using ArcGIS in the thirteen South Florida FHAP basins, plus an additional four basins sampled by Miami-Dade Regulatory and Economic Resources (**Figure 7-49**). *T. testudinum*, *Halodule wrightii*, and *Syringodium filiforme* were the three most abundant seagrass species in Florida Bay. *T. testudinum* was clearly the dominant species, and was widely distributed throughout the system (**Figure 7-49b**). *H. wrightii* was also present throughout the bay, but usually in substantially lower densities than those of *T. testudinum* (**Figure 7-49c**). *S. filiforme* was the most geographically limited of the three species, occurring chiefly in western and extreme northeastern Florida Bay (**Figure 7-49d**).

Unlike most estuarine and coastal systems, large-scale patterns of seagrass community structure in Florida Bay do not reflect differential light availability within various bay regions. Florida Bay is typically a shallow, clear-water system, and there is usually a sufficient amount of light to support seagrasses in all but the extreme northwestern bay (Kelble et al. 2005). Patterns in seagrass distribution, abundance, and species composition in Florida Bay are driven mainly by phosphorus availability, sediment depth and salinity regime (Fourqurean et al. 2003). However, since the onset of phytoplankton blooms in the 1990s following a major *T. testudinum* die-off in western and central Florida Bay, light reduction from periodic algal blooms have had localized, occasionally substantial, effects on seagrass community dynamics (Phlips et al. 1995, Hall et al. 1999, Rudnick et al. 2007).

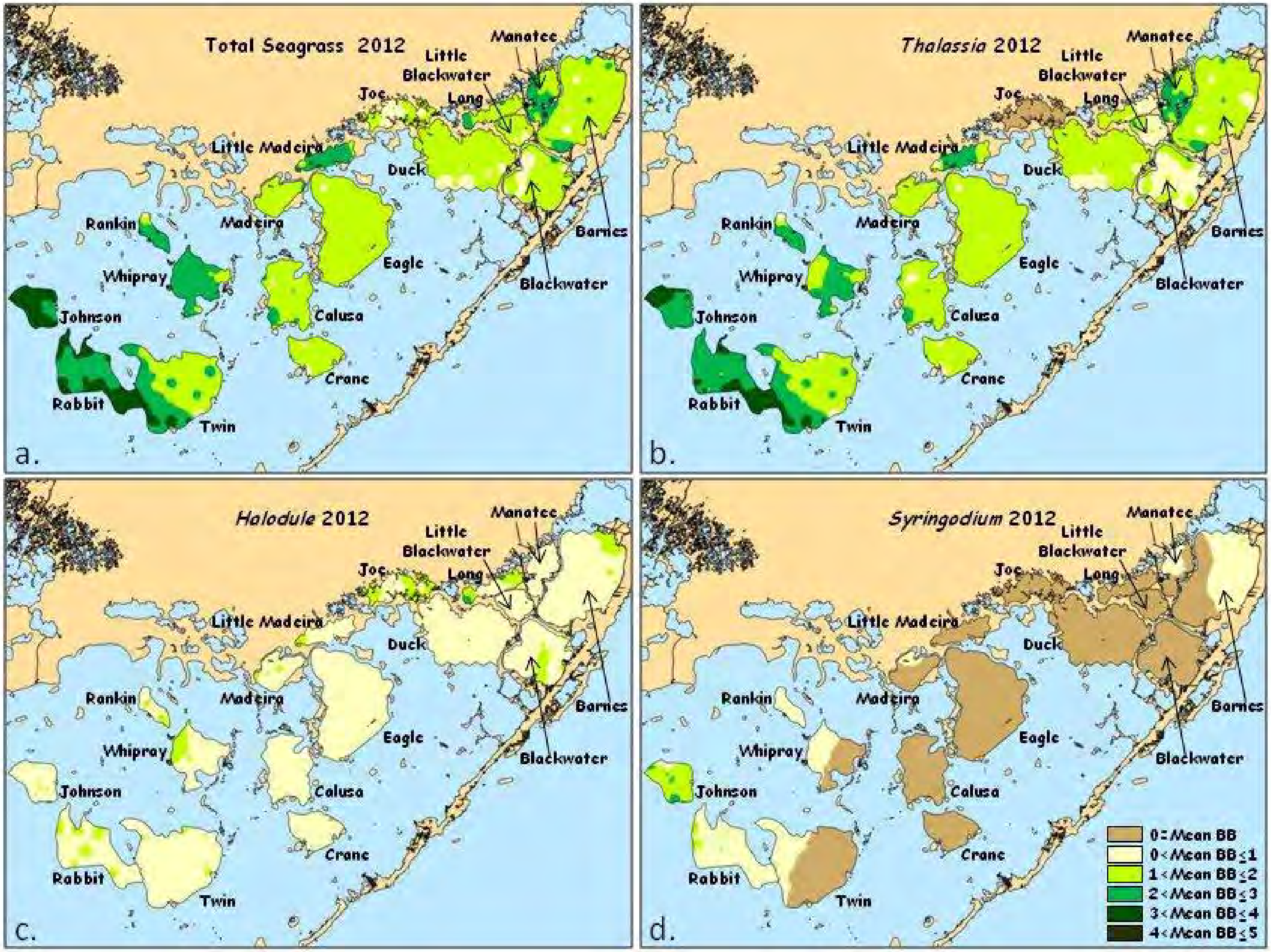


Figure 7-47. Contour plots illustrating 2012 Braun-Blanquet bottom cover values for a) Total seagrass, b) *T. testudinum*, c) *H. wrightii*, and d) *S. filiforme* in Florida Bay study areas.

The general increase in seagrass cover from northeastern to southwestern Florida Bay (**Figure 7-49a**), coincides with gradients in both sediment depth (Zieman et al. 1989) and phosphorus availability (Fourqurean et al. 1992). Many areas in northeastern Florida Bay have an insufficient amount of sediment to support dense seagrass communities; however mudbanks become broader and the amount of basin sediment increases towards the southwest (Zieman et al. 1989). Phosphorus limits seagrass growth in subtropical areas with biogenic carbonate sediments such as Florida Bay (Short et al. 1990). The major source of phosphorus for Florida Bay is Gulf of Mexico water, thus the gradient of phosphorus availability also increases from the northeast to the west (Fourqurean et al. 1992).

Because *T. testudinum* is the dominant seagrass species in the system, its pattern of distribution and abundance within Florida Bay is very similar to that for total seagrass (**Figure 7-49a**, **b**). *T. testudinum* densities are higher than those of *H. wrightii* in most of the bay because salinity is consistently that of seawater or above (**Figure 7-49b**, **c**). *T. testudinum* favors marine salinity; however, *H. wrightii* favors estuarine conditions when salinity is lower and more variable. Basins along the mainland receiving the greatest amount of fresh water from Taylor Slough are dominated or co-dominated by *H. wrightii* (e.g., Joe Bay and Long Sound, respectively). The brackish water species *Ruppia maritima* is also commonly observed in Joe Bay during the wet season when salinity is lowest. *S. filiforme* forms extensive meadows in the shallow Gulf of Mexico waters adjacent to western Florida Bay, and reaches its highest abundance within the bay in nearby Johnson Key Basin (**Figure 7-49d**).

Current patterns of seagrass distribution and abundance in Florida Bay are very similar to those observed in 1984 (Zieman et al. 1989), thus *T. testudinum* has dominated the system for at least the past 30 years. However, earlier reports from the 1960s and 1970s suggest that seagrass communities in northern and eastern Florida Bay were previously dominated by *H. wrightii*, and the seagrass community in the western and central bay consisted of a *T. testudinum-H. wrightii* mix (Schmidt 1979, Zieman 1982). Shifts in species prevalence since the 1960s have been attributed to consistently higher salinity in Florida Bay due primarily to upstream drainage and water management projects associated with Everglades land reclamation, which began in the 1950s, including construction of the canal system in south Florida (Light and Dineen 1994, Kelble et al. 2007).

Eastern Florida Bay Algae Bloom Impacts

From fall 2005 through 2008, an unprecedented algal bloom dominated by the cyanobacteria *Synechoccocus* spp. persisted along the eastern boundary of Florida Bay and along either side of U.S. Highway 1. The initiation of the bloom coincided with construction along U.S. Highway 1 and the onslaught of three hurricanes with their associated stormwater releases. A more thorough discussion about the bloom and its causes can be found in a white paper that was included as part of the *2007 South Florida Environmental Report* (Rudnick et al., 2007). A short summary of the impacts on the benthic community are described below.

One impact of the bloom was a lack of sponges in the benthic community for more than a year. In the October 2005 benthic survey, no sponges were found in any of the basins immediately along U.S. Highway 1 (Long Sound, Little Blackwater Sound, Blackwater Sound, Manatee Bay, and Barnes Sound). This survey coincided with the first water quality sampling that detected elevated Chla, thus it is not known if the loss of the filter feeders contributed to the bloom initiation through removal of top-down control on phytoplankton or if the sponge loss was caused by the bloom. However, the absence of sponges persisted in this area until spring 2007, when they started to repopulate (**Figure 7-50**).

SAV also declined in this region after fall 2005 and has not yet fully recovered. The frequency of quadrats with > 5% SAV cover decreased from almost 100% to as low as 35% in some areas (**Figure 7-51**). In all basins along U.S. Highway 1, the frequency of quadrats with > 5% cover has become more variable and has exhibited a wider range of values since 2005 when compared to prior years, suggesting that the benthic vegetation is still recovering from the disturbance. For an evaluation of the effects of this bloom using different SAV metrics, see the Southern Basins Phytoplankton Bloom and Seagrass Trends section.

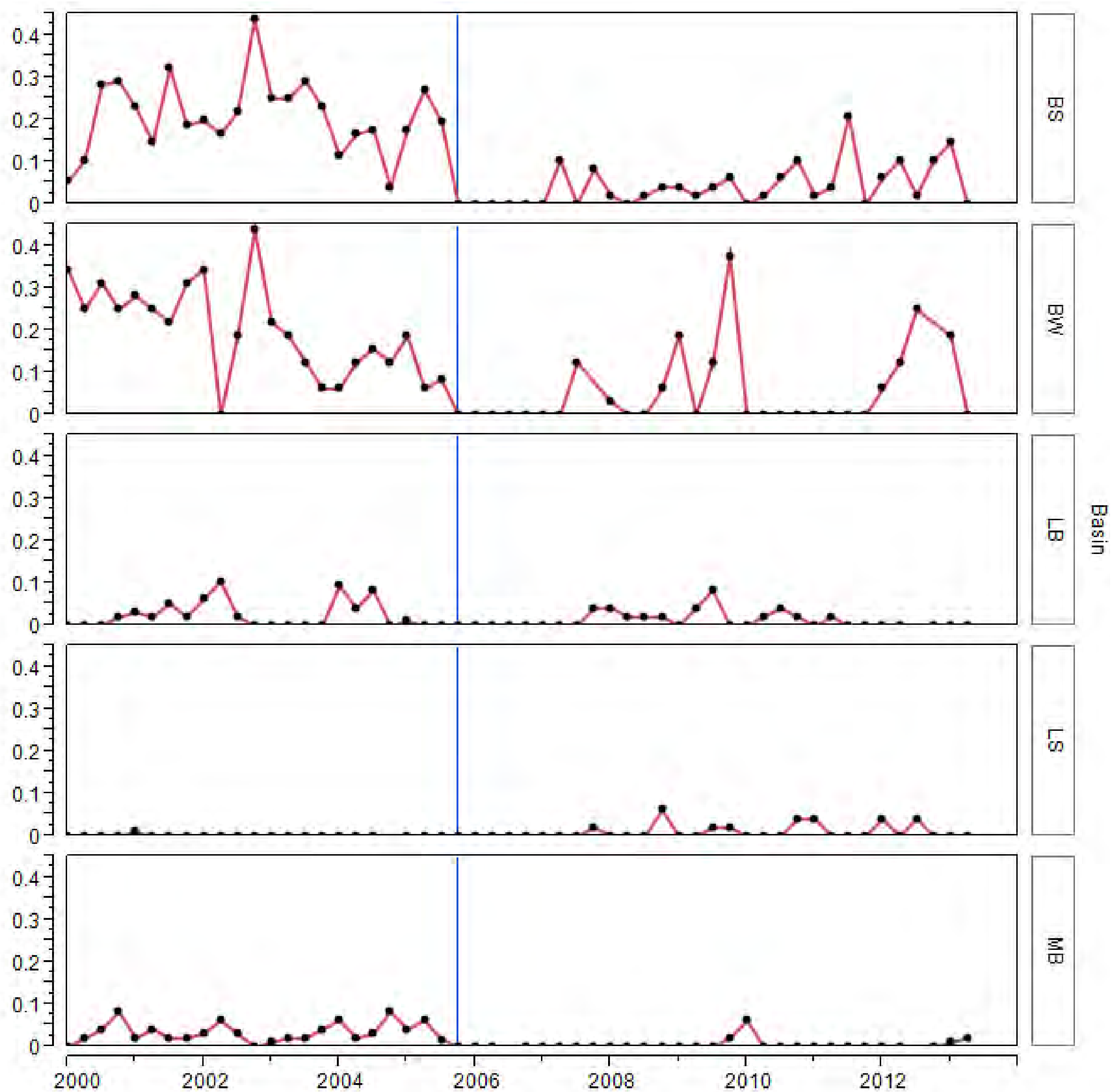


Figure 7-48. Sponge frequency of occurrence in basins along the U.S. Highway 1 corridor before and after the 2005–2007 algal bloom. Data was collected by Miami-Dade DERM 2000–2013. The vertical reference line marks October 2005, which was the first sampling event when no sponges were observed in any basin. This also coincides with the first measurement of elevated Chla in this area. Basins: BS – Barnes Sound, BW – Blackwater Sound, LB – Little Blackwater Sound, LS – Long Sound, MB – Manatee Bay.

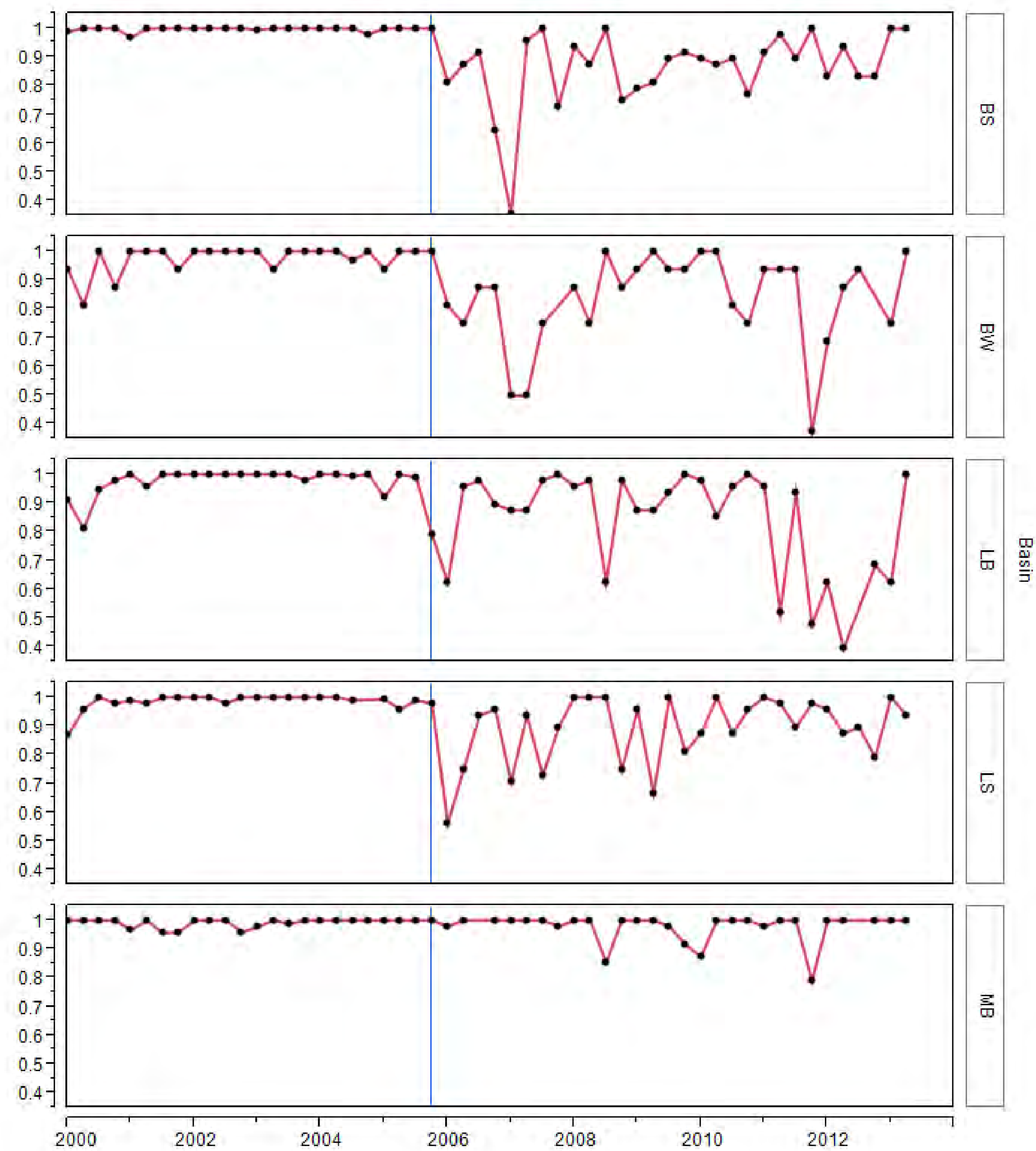


Figure 7-49. Seagrass and benthic macroalgae frequency of occurrence in basins along U.S. Highway 1. Occurrence is defined as when the more than 5% of the bottom in a quadrat is occluded. Data was collected by Miami-Dade DERM 2000–2013. The vertical reference line marks October 2005, which is the first sampling event where elevated chlorophyll was detected in this area. Basins: BS – Barnes Sound, BW – Blackwater Sound, LB – Little Blackwater Sound, LS – Long Sound, MB – Manatee Bay.

Central Florida Bay Community Shift

In May 2008, a significant increase in *H. wrightii* was observed in Madeira Bay, which is an embayment in north central Florida Bay with no visible creeks for freshwater inflow. Madeira Bay is within the area expected to benefit from C-111 Spreader Canal Western Projects operations. The percentage of quadrats where *H. wrightii* was present increased from 26% in 2007 to 49% in 2008, which was the highest measured since the beginning of South Florida FHAP in 1995. In 2009, *H. wrightii* frequency increased again to 65% before declining slightly to 59% in 2012 (**Figure 7-52**). *T. testudinum*, the dominant species within this embayment, had occurred in nearly 100% of quadrats from 1998 through 2005, but decreased from 100% in 2005 to occurrence in only 74% of quadrats in 2008. This species shift could possibly indicate decreased salinity levels or decreased light availability caused by phytoplankton blooms. Salinity data collected coincident with the SAV sampling shows high variability year to year, but this sampling is one day each year and does not give information on antecedent conditions, variability, or persistence. Thus, the cause of this species shift in Madeira Bay has been difficult to identify because there are no routine water quality monitoring stations within this embayment.

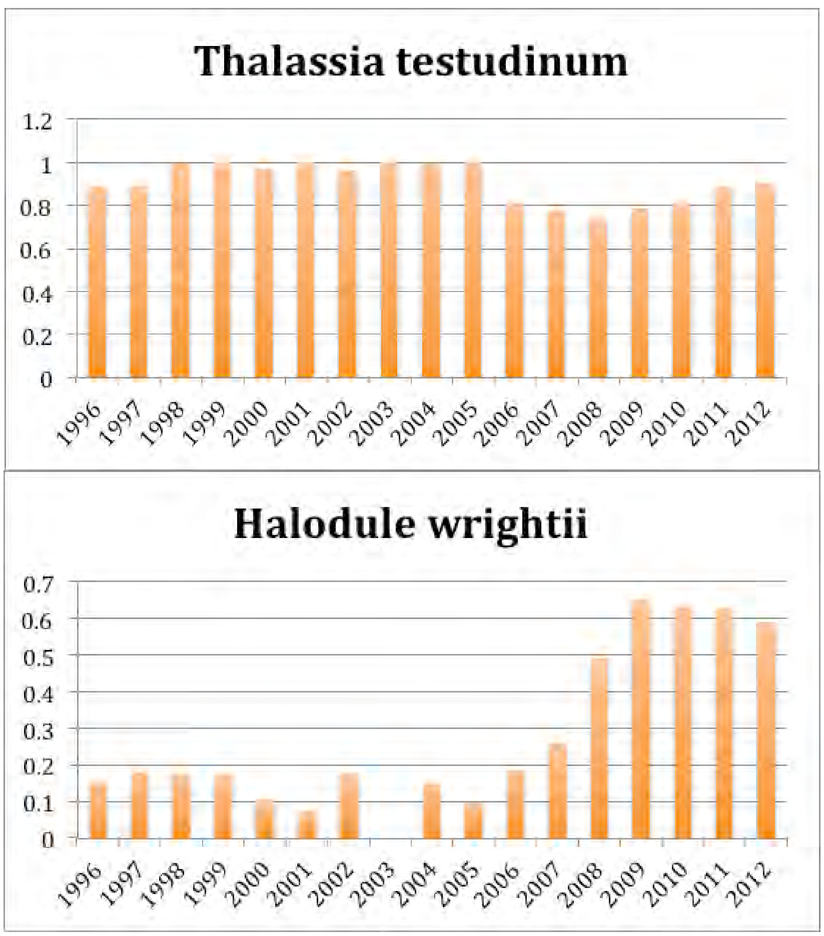


Figure 7-50, Frequency of occurrence for *T. testudinum* and *H. wrightii* in Madeira Bay from 1995 through 2012.

Evidence of *T. testudinum* Die-off in Western Florida Bay

In 1987, dense *T. testudinum* beds in western and central Florida Bay began dying rapidly. Although *T. testudinum* die-off slowed considerably by 1991, it was followed by several years of algal blooms and water column turbidity that negatively affected all seagrass species (Hall et al. 1999). Long-term changes in the salinity regime of Florida Bay may have contributed to the *T. testudinum* overgrowth, which left the system vulnerable to die-off (Rudnick, et al. 2005). Braun-Blanquet data collected from 1995 thorugh 2012 showed that cover of *T. testudinum* generally increased in the basins most heavily affected by the die-off (e.g. Rankin Lake, Whipray Basin, and Johnson Key Basin; **Figure 7-53**). However, *H. wrightii* increased in the basins through approximately 2000, but since has had a variable pattern in Rankin and Whipray basins and a declining trend in Johnson Key Basin. These changes in species abundance followed a secondary successional pattern typical for subtropical seagrass systems. Although the recovery of seagrasses indicates the resiliency of the Florida Bay seagrass community, scientists and managers have long been concerned that widespread die-off might reoccur if *T. testudinum* densities increased to pre-die-off levels. Small die-off patches (one to two meters diameter) have been noticed periodically since 1995, but in spring 2012 relatively large (several hundred meters diameter) die-off patches were observed in dense *T. testudinum* beds on banks at the periphery of Rankin, Whipray, and Johnson Key basins. These die-off patches were “active” (e.g., die-off in progress), as evidenced by our ability to easily remove apparently healthy shoots from the edges of the patches because the meristematic tissue was dead. Die-off patches were revisited in August 2012, and were no longer active.

The large die-off patches observed in Rankin, Whipray, and Johnson Key basins in May 2012 suggest that *T. testudinum* abundance in western Florida Bay has increased to the point where the community is potentially vulnerable to another large-scale die-off event. The original die-off and subsequent ecological disturbances (phytoplankton blooms, sponge die-offs, etc.; Butler et al. 1995) largely precipitated the initiation of CERP. Changes in freshwater delivery associated with CERP activities are expected to result in more diverse seagrass communities with moderate plant densities, making the system less susceptible to future *T. testudinum* die-off events.

<u>*Summary and Conclusions*</u>

SAV community distribution data over the past decade indicate the system is potentially vulnerable to another large scale die-off event similar to the one that occurred in 1987 which, along with subsequent ecological disturbances, largely precipitated the initiation of the CERP. These observations include the following:

- Seagrass distribution and abundance are similar to conditions recorded in 1984.
- Large die-off patches were observe in several central and westerly located stations.

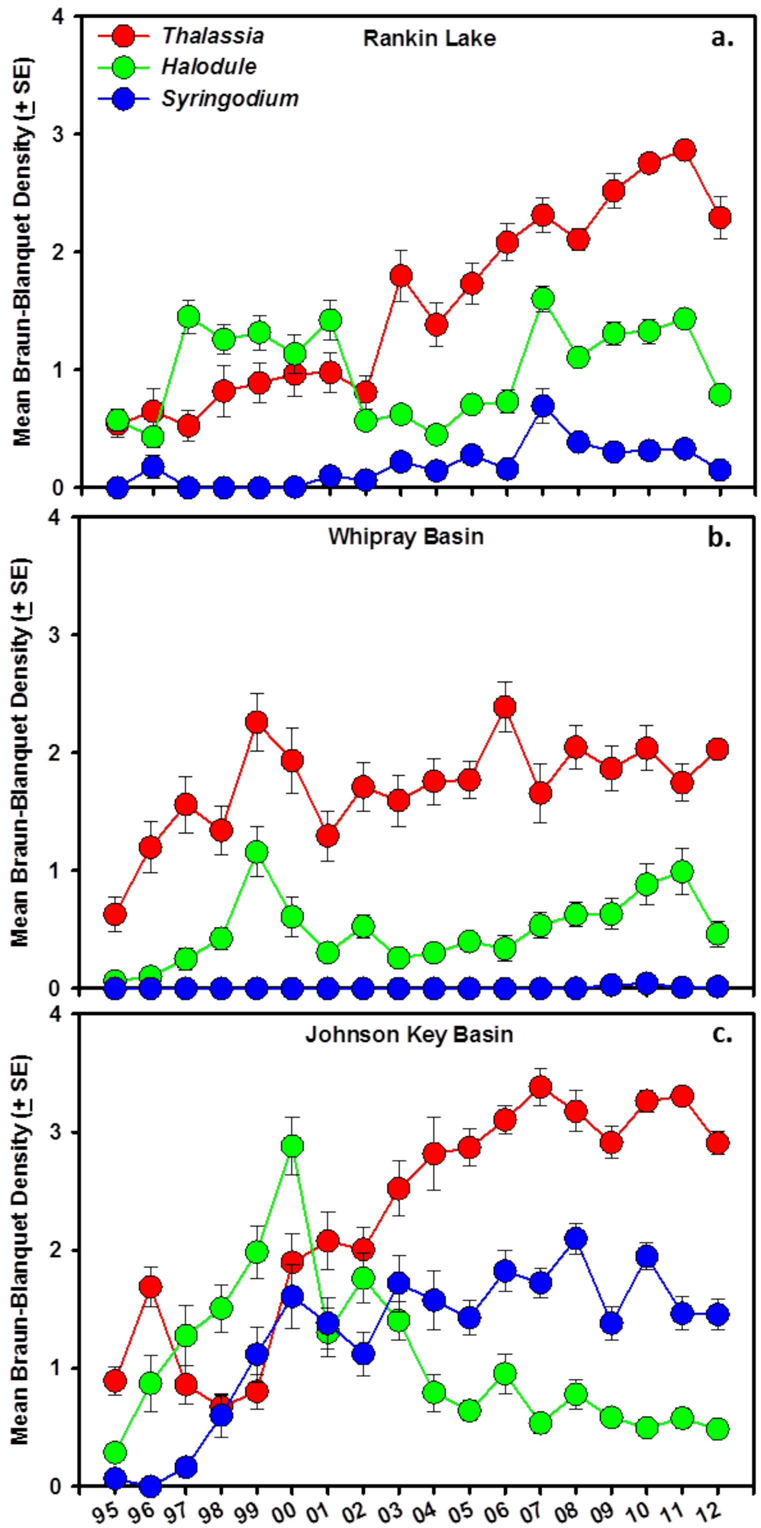


Figure 7-51. Mean Braun-Blanquet bottom cover values for *T. testudinum* (red), *H. wrightii* (green) and *S. filiforme* (blue) in a) Rankin Lake, b) Whipray Basin, and c) Johnson Key Basin from 1995 through 2012. (Note: SE – standard error.)

Determination of the cause for observed SAV population composition shifts in basins anticipated to be first affected by the constructed and operational features of the CERP requires spatially coherent information on freshwater flow, salinity, and water quality. Such information is absent for Madeira Bay where dramatic SAV changes have been observed. Assessment of C-111 SCWP would benefit from the monitoring described in the C-111 SCWP project implementation report (USACE and SFWMD 2011) for this region.

Florida Bay and Lower Southwest Coast Epifauna and Pink Shrimp

The FIAN is a regional-scale monitoring project designed to assess the condition of Florida's southernmost estuarine ecosystems based on the abundance of juvenile pink shrimp (*Farfantepenaeus duorarum*) and other epifauna in four major taxa, fish, caridean shrimp, penaeid shrimp, and crabs. The pink shrimp is an indicator species in CERP because it is ecologically and economically important, it is common and widespread, and several studies, including Browder (1985) and Sheridan (1996) suggested a relationship with nursery ground salinity regime. Shrimp production from these estuaries supports game fish (i.e., spotted seatrout and gray snapper) and other predators, as well as substantial shrimp fisheries offshore and in Biscayne Bay.

FIAN was organized on the premise that habitat and environmental conditions, particularly salinity, are critical factors structuring nearshore fish and crustacean communities. It was hypothesized that secondary production in nursery areas will increase as the overlap of favorable salinity conditions with optimal seagrass habitat increases (Browder and Moore 1981). The 19 FIAN sampling locations are distributed from northern Biscayne Bay through Florida Bay into the Lower Southwest Coast estuaries (**Figure 7-54**). FIAN sampled across all habitats and environmental gradients within a 30-cell sampling grid at each sampling location so that faunal samples were distributed in proportion to the area of habitat and conditions present. For details on sampling methods and faunal taxa identified (over 200 species) see Robblee and Browder (2013).

The four major FIAN taxa were found over wide ranges of salinity and in several types of habitat, including seagrasses (*T. testudinum*, *H. wrightii*, *S. filiforme*, and others) and macroalgae. Distribution patterns of total fish, caridean shrimp, and crabs were significantly related to canopy height, which was representative of seagrass community development. Caridean abundance was positively related to salinity, probably because the caridean group of shrimp was dominated by *Thor* (71% of total carideans), a marine genus associated with shallow water seagrass meadows. Two caridean groups, *Ogyrides* and the *Palaemoninae*, were relatively abundant in low salinity waters, and new analyses indicate that their abundance was significantly related to salinity (parabolically for *Ogyrides* and negatively for *Palaemoninae*).

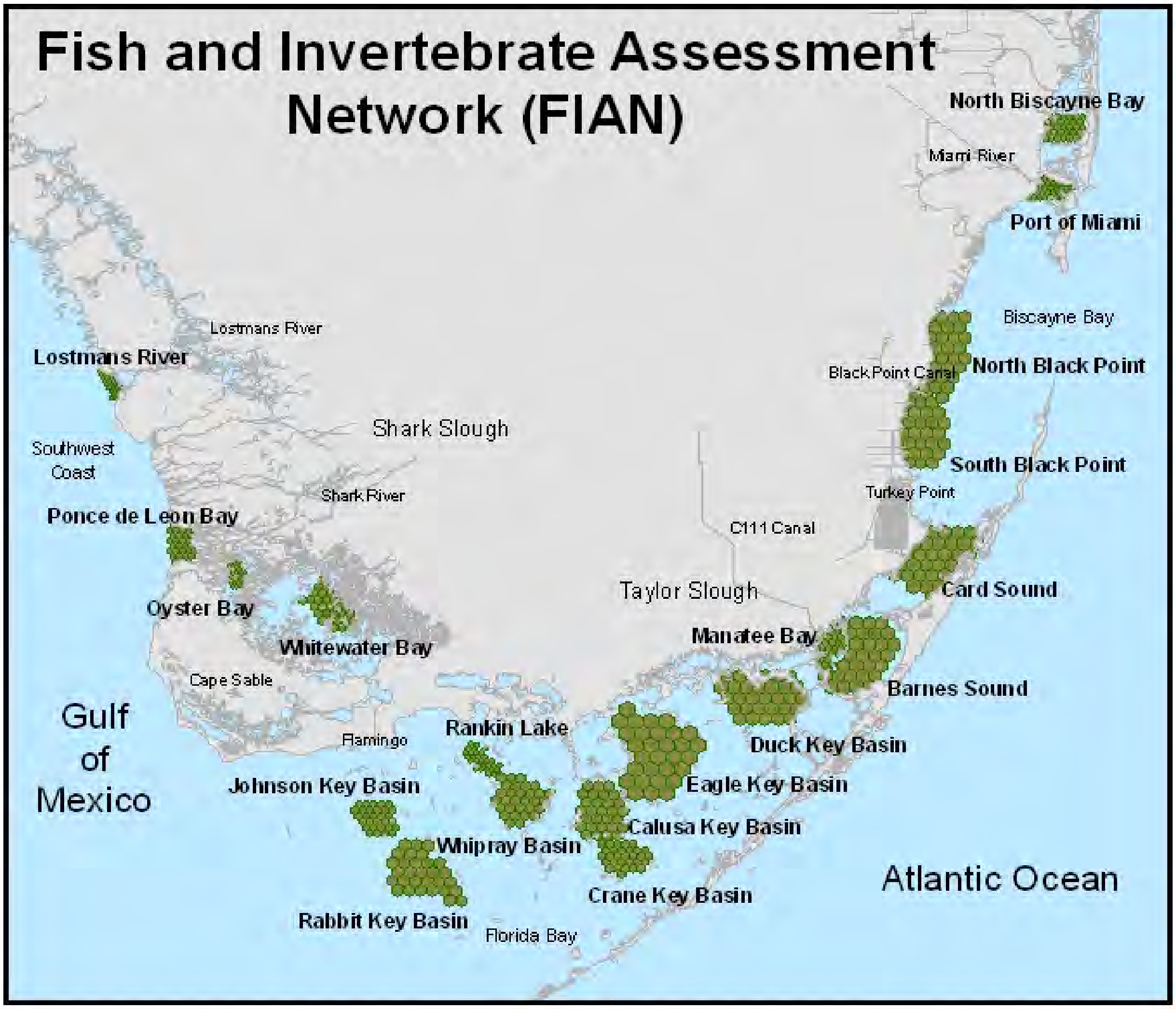


Figure 7-52. The 19 FIAN sampling locations with sampling grids (green) superimposed.

<u>*Pink Shrimp Indicator*</u>

Pink shrimp has been the indicator species for annual CERP assessments by FIAN since 2005. Assessment of 2010 and 2011 conditions for pink shrimp were based on location-specific data from the first five years of FIAN sampling, 2005–2009, and covered all 19 sampling locations. Assessment of 2012 conditions was not conducted due to termination of funding.

Assessment results for spring and fall 2010 and 2011 are mapped in **Figure 7-55** and portrayed using stoplight indicator color scheme (red = substantial deviations from restoration targets creating severe negative condition that merits action; yellow = current situation does not meet restoration targets and merits attention; green = situation is good and restoration goals or trends have been reached [Doren et al. 2009]). There are more red stoplights than green ones on all four maps. System performance, relative to juvenile pink shrimp, was especially poor in spring 2010. Best overall performance was in fall 2011, when there were more green lights, coupled with fewer red lights, than for the other three collections.

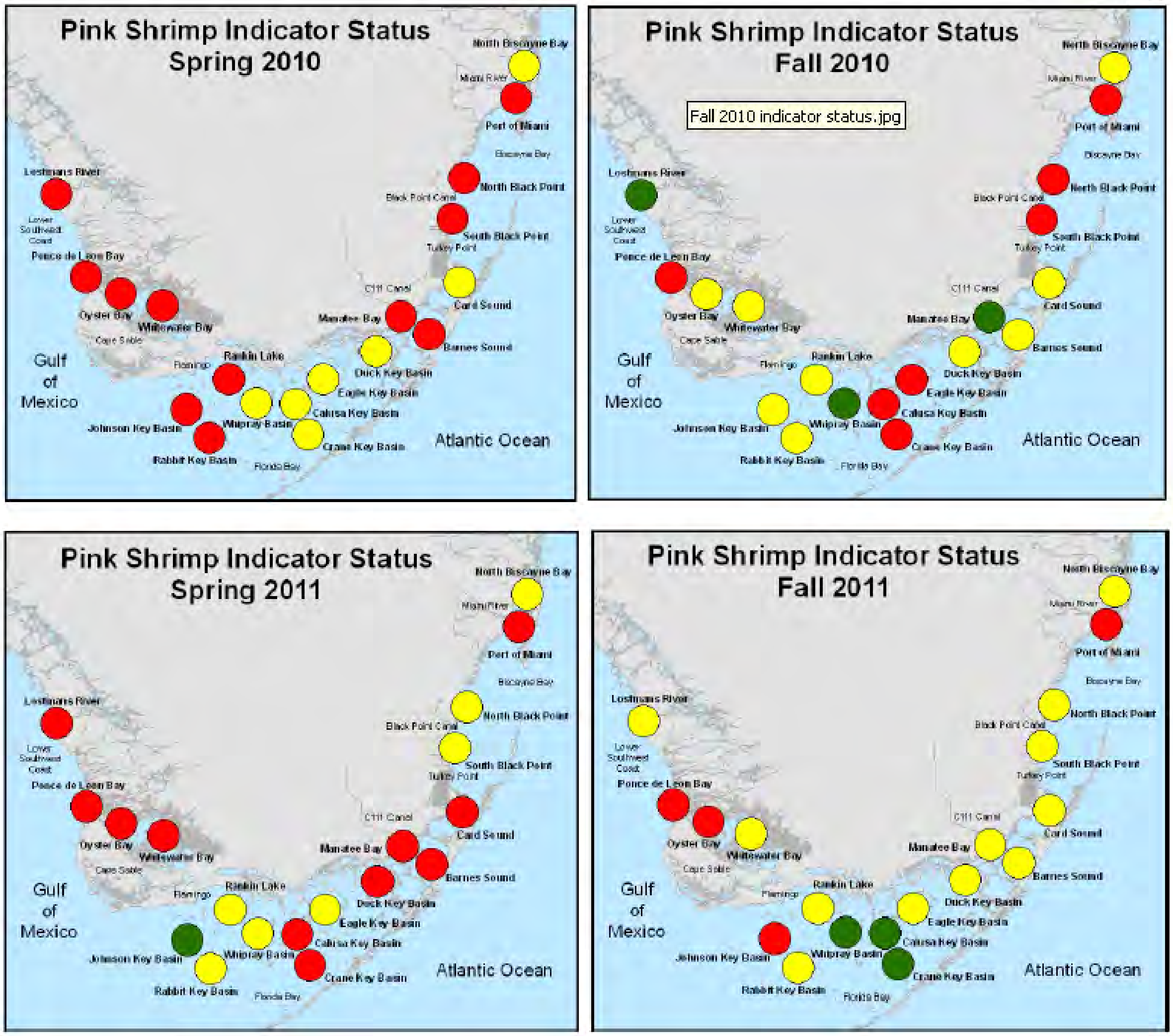


Figure 7-53. Pink shrimp indicator status at SCS locations in wet and dry seasons of 2010 and 2011. Data derived from FIAN.

Fall performance was green in Whipray Basin, indicating relatively good conditions for pink shrimp, in both years. Whipray Basin is an area where conditions for pink shrimp and other epifaunal species may improve with hydrologic restoration that reduces the intensity, frequency, and duration of hypersaline conditions, although no significant relationship was found between shrimp density and salinity in either the combined or separate spring and fall grid average Whipray data.

The poor performance in Johnson Key Basin in 2011 is of special concern because of this area's relative importance as a nursery ground (i.e., overall pink shrimp density about twice as high as in any of the other 19 sampled locations). Average salinity over this grid at sampling time was slightly higher in 2011 than in most of the reference years. A significant negative relationship over the range 33 to 42 was found in Johnson grid average data for both fall and fall and spring combined for 2005–2011. A significant negative relationship within the range of 28 to 45 was previously found in the Johnson Key Basin historic (pre-FIAN) data by Browder and Robblee (2009).

Annual spring and fall abundance estimates of juvenile pink shrimp at the 19 locations, organized by South Florida bay system, are shown in **Figure 7-56** (Biscayne Bay), **Figure 7-57** (Florida Bay), and **Figure 7-58** (Southwest Coast). The abundance index was delta density (number per m^2), calculated as described in the 2012 final FIAN report (Robblee and Browder 2013). The first five points in each time series (i.e., the years 2005–2009, plotted with open triangles) were used to calculate the 1st, 2nd, 3rd, and 4th quartiles for assessing 2010 and 2011 conditions for pink shrimp (plotted with closed triangles). The quartiles, as used in the assessment, are portrayed by the background stoplight colors of red (1st quartile, representing poor performance), yellow (2nd and 3rd quartiles, representing neutral performance), and green (4th quartile, representing good performance). The positions of the 2010 and 2011 triangles on the red, yellow, and green backgrounds in each plot indicate the assessment score.

The plots in **Figures 7-56** through **7-58** allow annual density estimates for each season to be screened for obvious trends over time and commonalities or contrasts in temporal patterns among locations. Collectively and individually, these assessments indicate annual conditions in the south Florida inshore pink shrimp nurseries.

Spring abundance at North Black Point, South Black Point, and, to a lesser extent, Card Sound and even Barnes Sound and Manatee Bay, varied similarly across years, but not with any trend. All sites had three exceptionally poor years (2005, 2010, and 2011) in common. North Miami, North Black Point, and South Black Point also shared high spring abundance in 2006. These characteristics made North Miami's temporal pattern more similar to North Black Point and South Black Point than to the nearer Port of Miami.

Spring abundance at Rankin Lake, Johnson Key Basin, and Rabbit Key Basin in Florida Bay varied similarly across years, declining generally from 2005 through 2010 and increasing in 2011. Whipray Basin was similar to Rankin Lake except that 2006 was poorer, and 2010 and 2011 better than Rankin relative to other years.

Ponce de Leon Bay, Oyster Bay, and Whitewater Bay, which form a salinity gradient from outer to inner estuary, varied similarly over time with abundance exceptionally high in spring 2009 and exceptionally low in springs 2006, 2010, and 2011. The temporal pattern in spring abundance at Lostmans River, another outer estuary, was similar to the others with the exception that 2008 was even higher than 2009. 2005 was a year of exceptionally high fall abundance in most locations in all three coastal systems. Notable exceptions were Lostmans River, Ponce de Leon Bay, Card Sound, Barnes Sound, Manatee Bay, Duck Key Basin, and Eagle Key Basin, where fall abundance in 2005 was either low or not remarkably high.

Salinity is a possible factor influencing the seasonal and annual variation in abundance of pink shrimp, and FIAN was consistent with Browder et al. (2002) and Browder and Robblee (2009) in results suggesting avoidance, or lack of tolerance, of hypersalinity by pink shrimp (see Figure 17 in the 2012 FIAN report [Robblee and Browder 2013]). However, a significant relationship of pink shrimp abundance with salinity was not found in the regional-scale FIAN data.

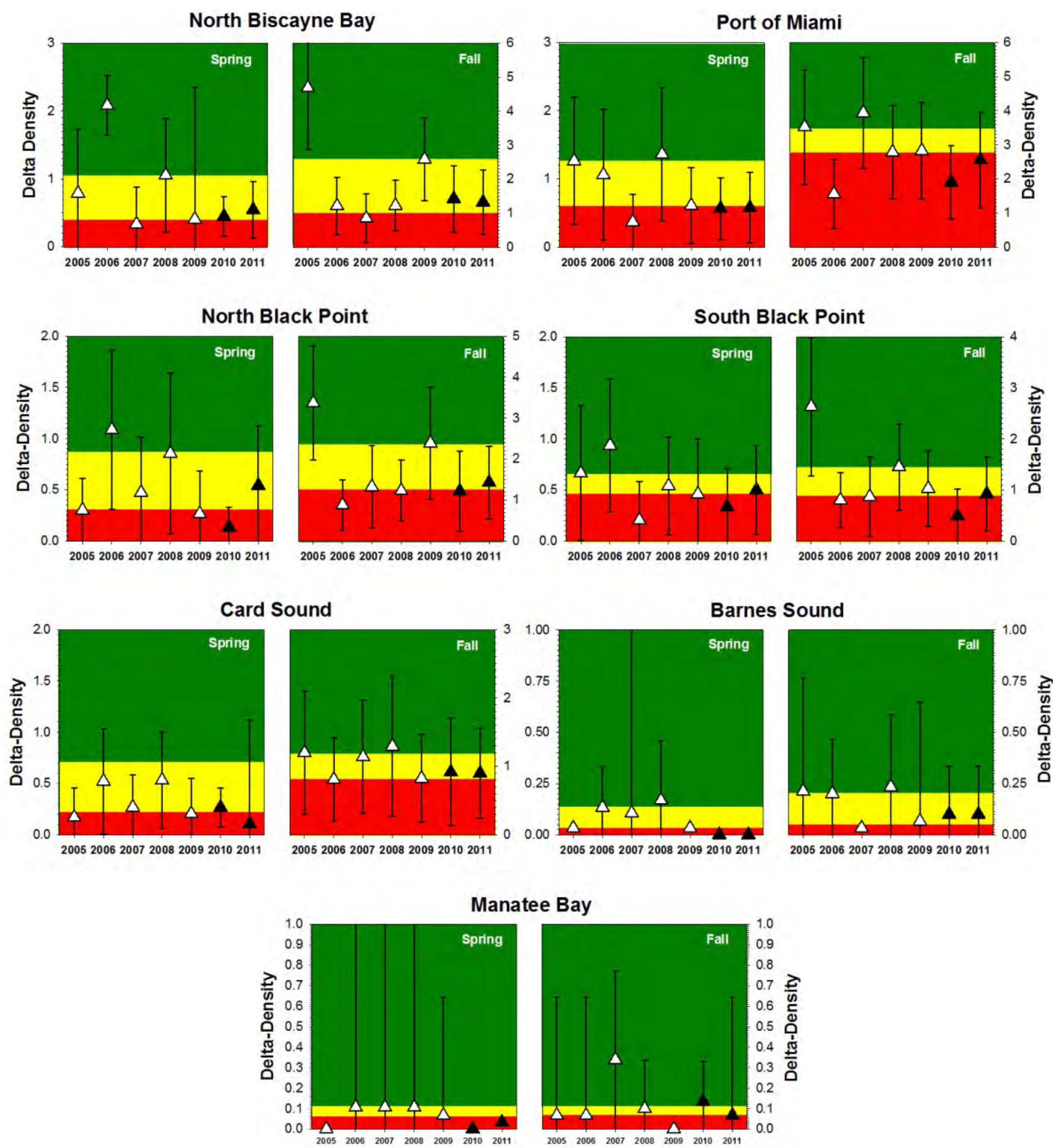


Figure 7-54. Seasonal and annual pink shrimp abundance (as delta density, number per m^2) at Biscayne Bay assessment locations.

FIAN found that fall shrimp abundance in Whitewater Bay, where the fall salinity regime tended to be mesohaline, matched that of Rabbit Key Basin and was higher than that in any other FIAN location except Johnson Key Basin, where abundance was about double that of Whitewater Bay and Rabbit Key Basin. Whitewater and Coot bays had been viewed as pink shrimp nursery grounds in studies of the 1950s and 1960s that quantified variation in pink shrimp abundance by monitoring ingress and egress through the Buttonwood Canal (i.e., Tabb et al. 1962). FIAN, with its consistent sampling design in all 19 locations, provided the first data allowing valid comparison of the two nursery areas, Johnson Key Basin and Whitewater Bay.

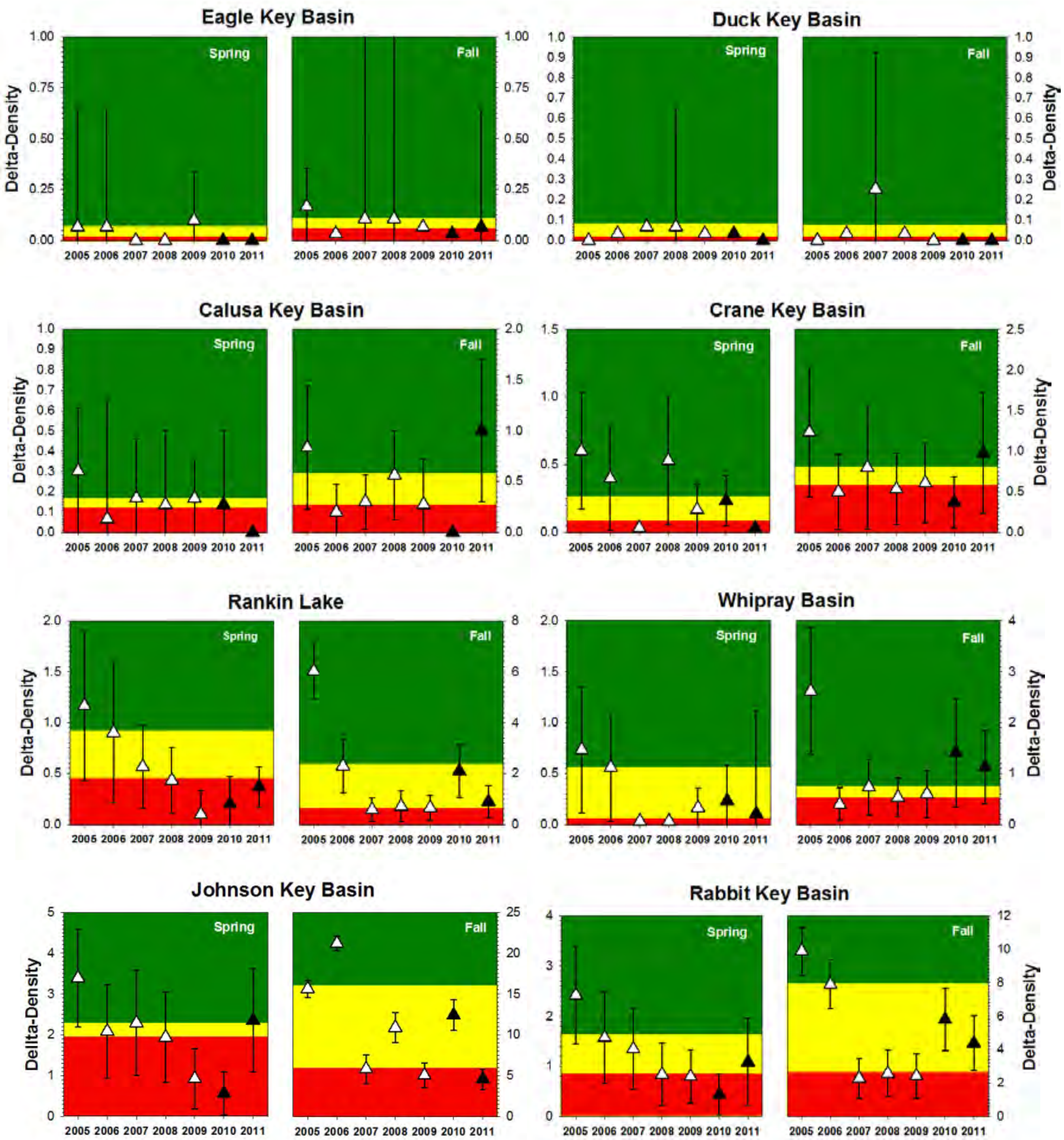


Figure 7-55. Seasonal and annual pink shrimp abundance (as delta density, number per m^2) at Biscayne Bay assessment locations.

Results of the comparison raised the question of how Whitewater Bay could support young pink shrimp so well. Several hypotheses were formed to address this question. Hypothesis 1 is that the pink shrimp is a "plastic" species whose salinity tolerance range and optimum are shaped at early settlement stage by "conditioning" to the salinity regime where the individual will grow up. Hypothesis 2 is that two or more physiological phenotypes, or "morphs"—each with a different salinity tolerance range—exist within the south Florida shrimp population, and local conditions determine which morph survives on entering a given estuary; the polymorphs are maintained in the breeding population by the difference in salinity regimes from one area to another.

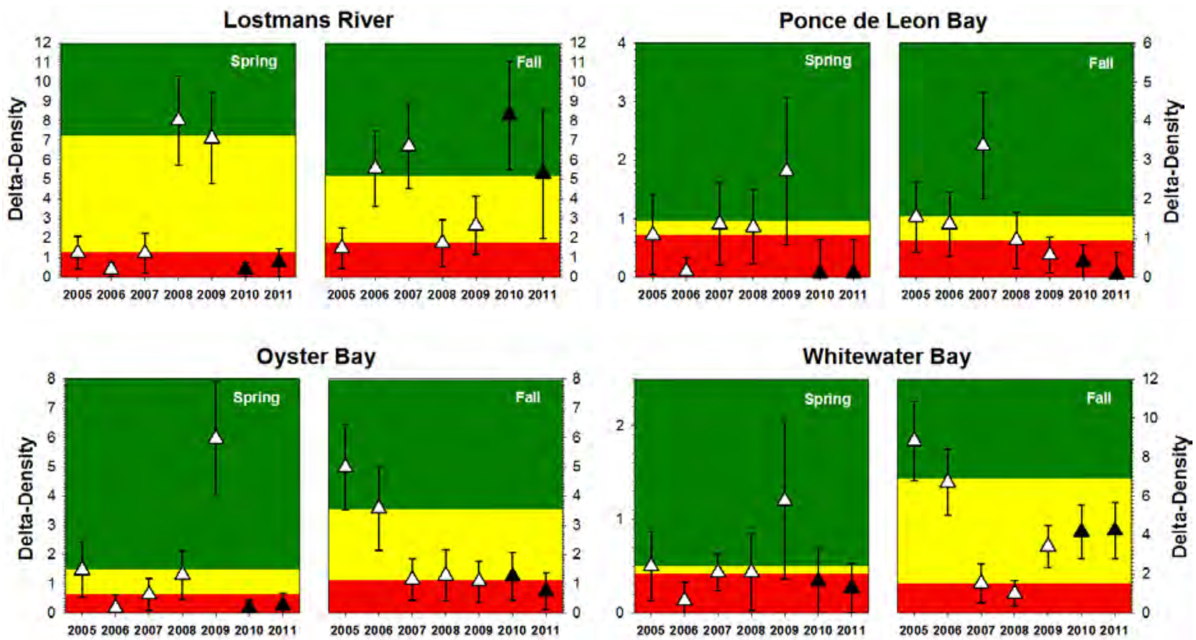


Figure 7-56. Seasonal and annual pink shrimp abundance (as delta density, number per m^2) at Biscayne Bay assessment locations.

Another surprising finding from FIAN was that macroalgal and/or sparse *H. wrightii* habitat can be an effective nursery ground, supporting substantial densities of pink shrimp (see Figure 14 in the 2012 FIAN report [Robblee and Browder 2013]). Clearly, the pink shrimp is not the obligate seagrass associate it was thought to be in previous studies that sampled in the extensive seagrass habitats of Florida Bay (Costello et al 1986, Holmquist et al 1989, Robblee et al. 1991).

Postlarval transport success is another possible factor influencing seasonal, annual, and spatial variation in pink shrimp abundance (Criales et al. 2005, 2006). Synchronization of annual variation across several locations open to the same recruitment pathway, i.e., across the Southwest Florida Shelf or up the east coast in Gulf Stream-associated eddies might suggest an immigration role. An influence of offshore spawning success (Ehrhardt et al. 1999) might be suggested by seasonal differences in abundance at the same location or by synchronization of annual highs across most locations, especially those most exposed to fluxes from offshore. On the other hand, synchronization of annual patterns of abundance at locations exposed to similar sources of fresh water (i.e., the Whitewater Bay to Ponce de Leon Bay system or Whipray Basin and Rankin Key Basin) might suggest rainfall or water management effects, especially if related to salinity. Broadscale synchronization of temporal patterns across the region was not noted in the time series data. However, the low abundance at interior locations, where tidal affects were lowest, suggests that differential postlarval access influences the spatial distributions of abundance within estuarine systems.

Summary of Pink Shrimp

When last viewed by FIAN, conditions for pink shrimp in the south Florida coastal system overall suggest an opportunity for much improvement by CERP. The recent data indicate overwhelming red and yellow status for this indicator for 2010–2011. The variation seen by FIAN in the magnitude of pink shrimp abundance across the south Florida coastal region confirms the need to use location-specific scales to perform regionwide assessments. The relative independence of temporal patterns across locations and conformity to local salinity variation suggest that salinity may influence local abundance in at least some locations.

Florida Bay and Lower Southwest Coast Juvenile Sportfish

Sportfish in Florida Bay represent a significant economic and ecological component of south Florida. These populations are subject to numerous anthropogenic pressures and will respond to the salinity changes that will occur as CERP is implemented. The analyses presented here are for juvenile spotted seatrout (*Cynoscion nebulosus*) and gray snapper (*Lutjanis griseus*). These two species account for greater than 60% of the recreational fishery catch in Florida Bay (Tilmant 1989).

An otter trawl was used to sample juvenile sportfish and their prey in Florida Bay monthly from May through October each year, since 2004 (Kelble et al. 2013; **Figure 7-59**). Highest abundances and

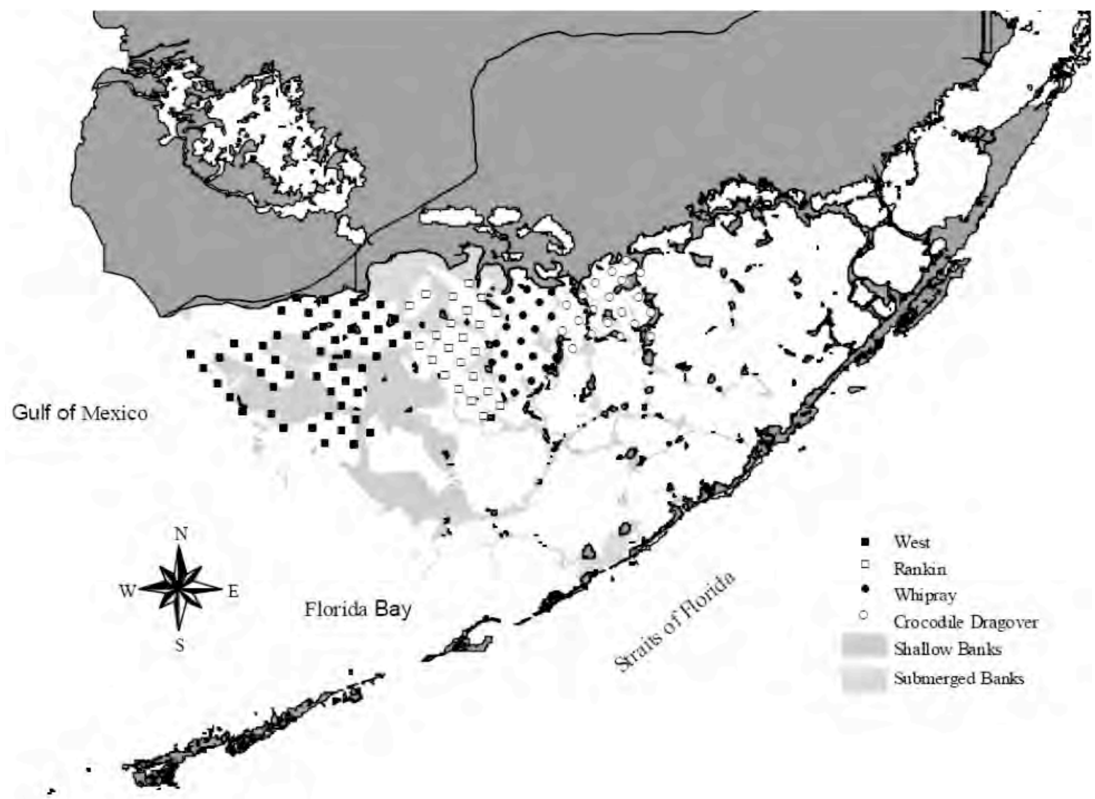


Figure 7-57. Sampling area of juvenile sportfish trawls in western and northern Florida Bay. Location of sampling areas are shown. A subset of these areas from each region is randomly sampled per sampling event.

frequencies of occurrence of both species were observed in the West subregion and the lowest abundances and frequencies of occurrence were observed in Crocodile Dragover, the easternmost subregion sampled (**Figures 7-60** and **7-61**). Both species showed a large degree of interannual variability, but unlike spatial distributions there were significant differences between the interannual distributions. Juvenile spotted seatrout have had low populations in all subregions since 2008 (**Figure 7-60**). However, juvenile gray snapper have only experienced low populations in all subregions since 2010 (**Figure 7-61**). These low populations of juvenile gray snapper may be a negative response to record cold weather in January 2010 (Lirman et al. 2011). The lowest population numbers, at least in the three central subregions, were observed in 2010 and have since been increasing (**Figure 7-61**).

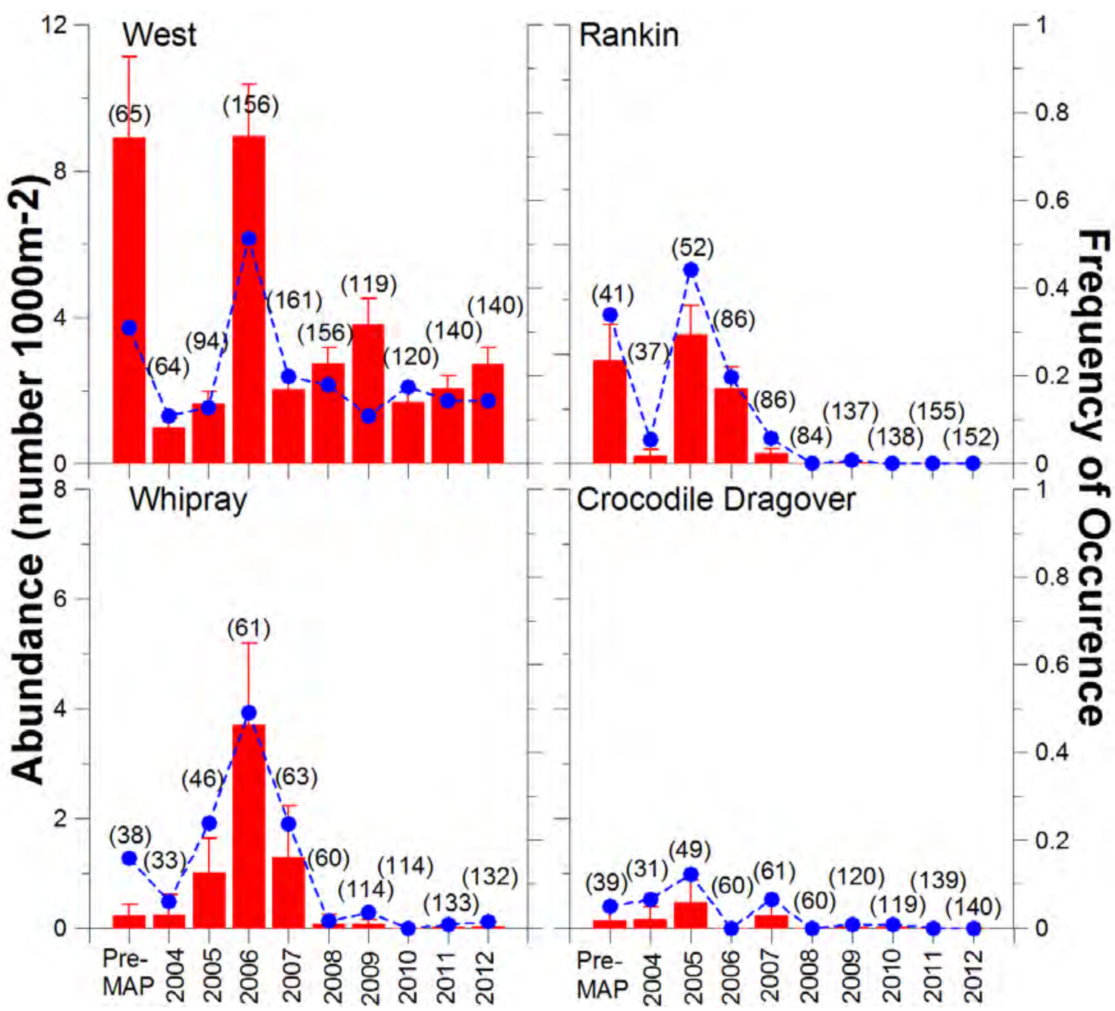


Figure 7-58. Density (number 1000 m^2 + standard error) as a bar chart with error bars representing the standard error and frequency of occurrence as a scatter plot for juvenile spotted seatrout by subregion and year in Florida Bay. Values in parentheses indicate the number of stations sampled. Data from 1984 through 2000 ("pre-MAP") are combined.

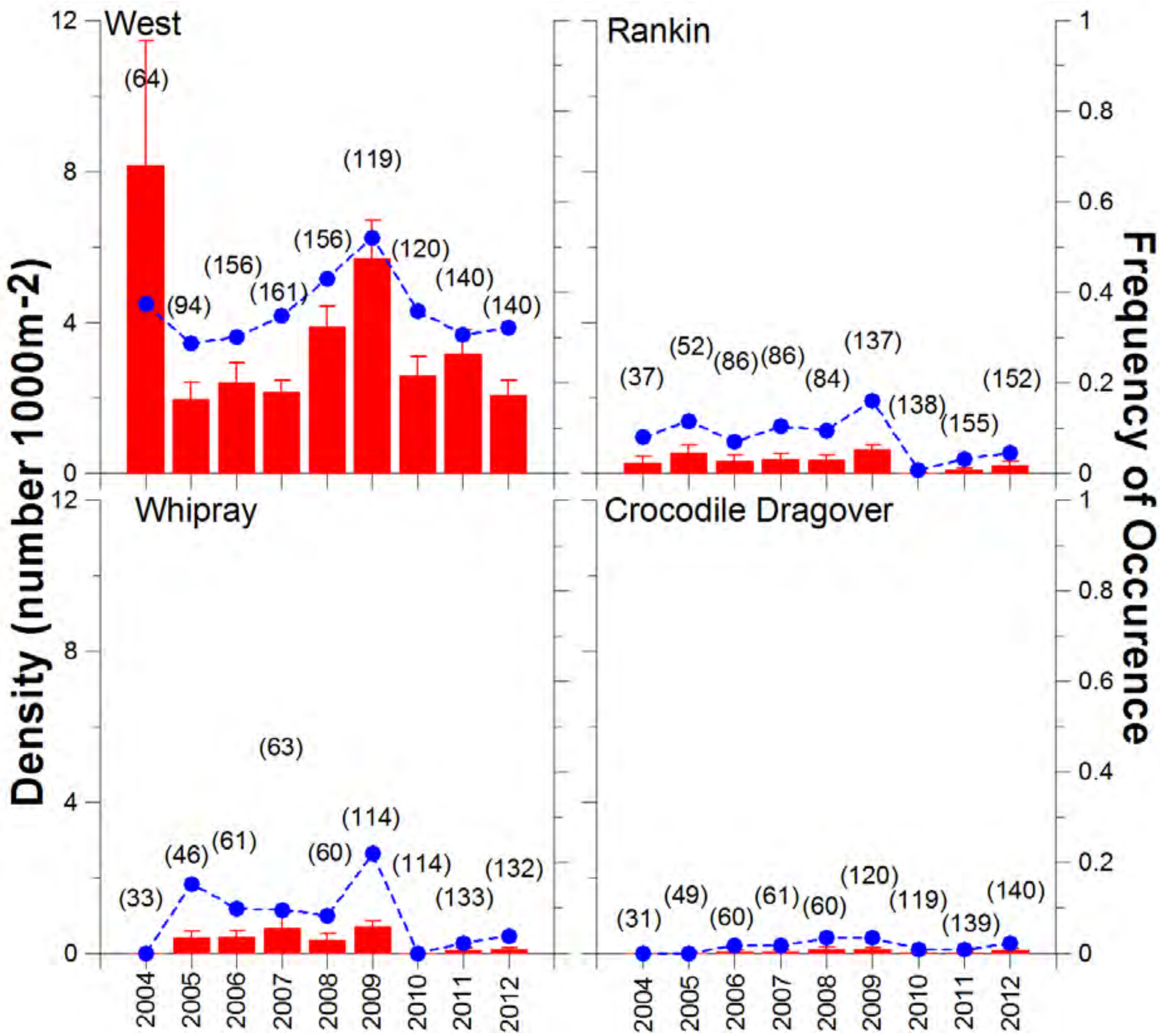


Figure 7-59. Density (number 1000 m^2 + standard error) as a bar chart with error bars representing the standard error and frequency of occurrence as a scatter plot for juvenile gray snapper by subregion and year in Florida Bay. Values in parentheses indicate the number of stations sampled.

The response of these two populations to CERP will largely be determined by their response to salinity. Current responses to the salinity variability in each subregion were determined to prepare for predicting and assessing the effects of CERP. The two species showed disparate salinity responses in all subregions, with the exception of Whipray, where occurrences and abundances of both species had inverse linear relations with salinity (**Figures 7-62** and **7-63**). Other than the Whipray subregion, juvenile gray snapper had no other significant linear relationships of their population parameters with salinity. Conversely, juvenile spotted seatrout had significant inverse linear relationships of at least one population parameter with salinity in each subregion. This could lead to the conclusion that juvenile gray snapper are not affected by salinity and thus unlikely to be affected by CERP. However, these results merely suggest that in all subregions, juvenile spotted seatrout prefer the lower end of the range of observed salinity; whereas, juvenile gray snapper only show a preference in Whipray.

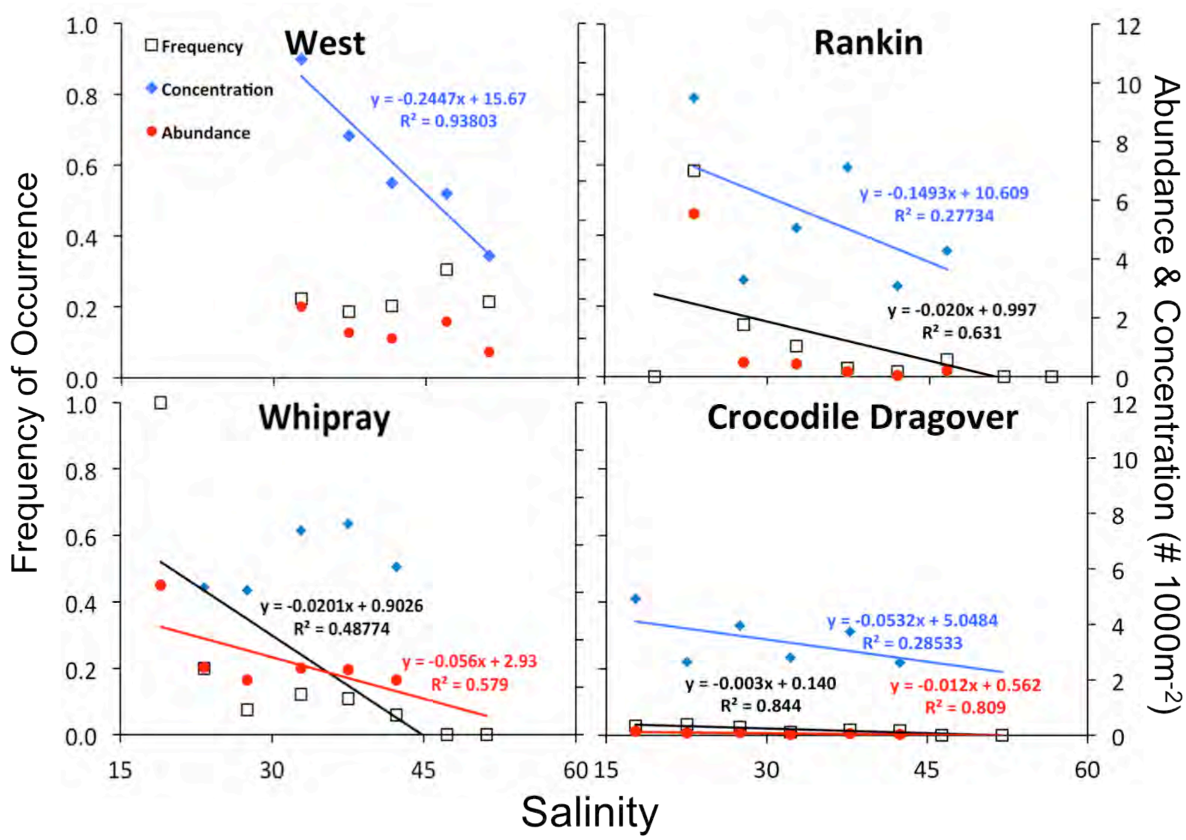


Figure 7-60. Scatter plots depict the correlation of the juvenile spotted seatrout populations with salinity within each subregion. The black squares are frequency of occurrence, the blue diamonds are concentration of fish when present and the red circles are abundance as delta-density. Only statistically significant linear regressions are depicted.

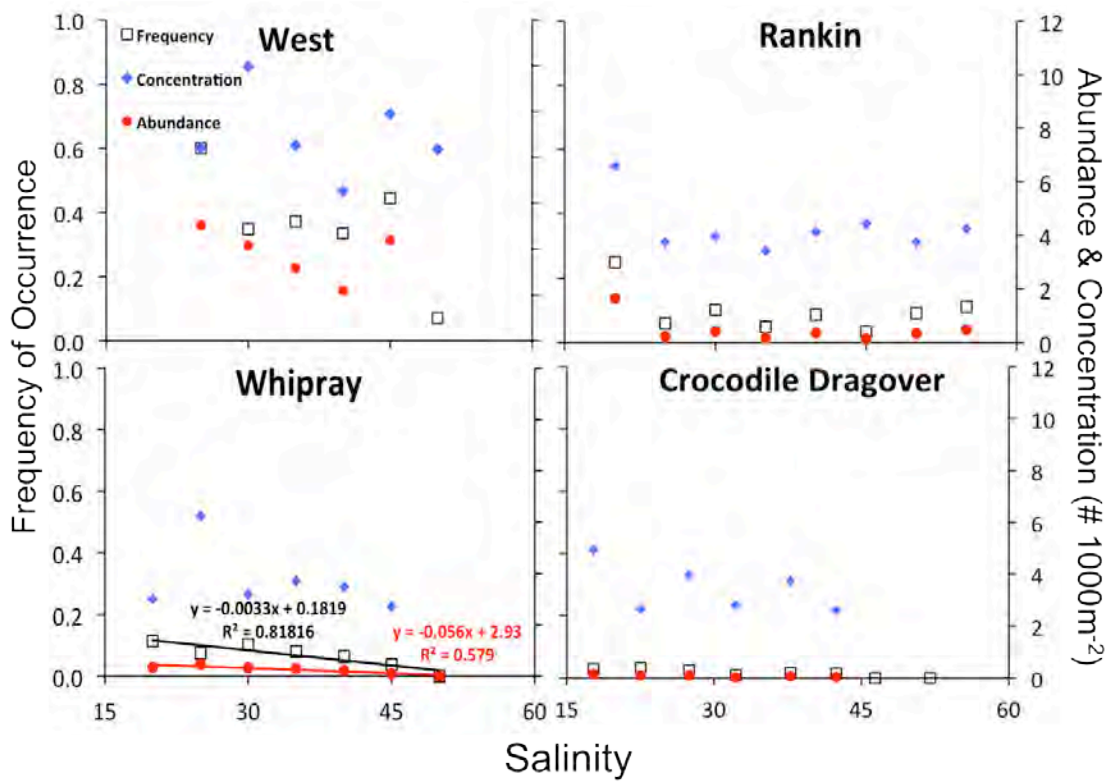
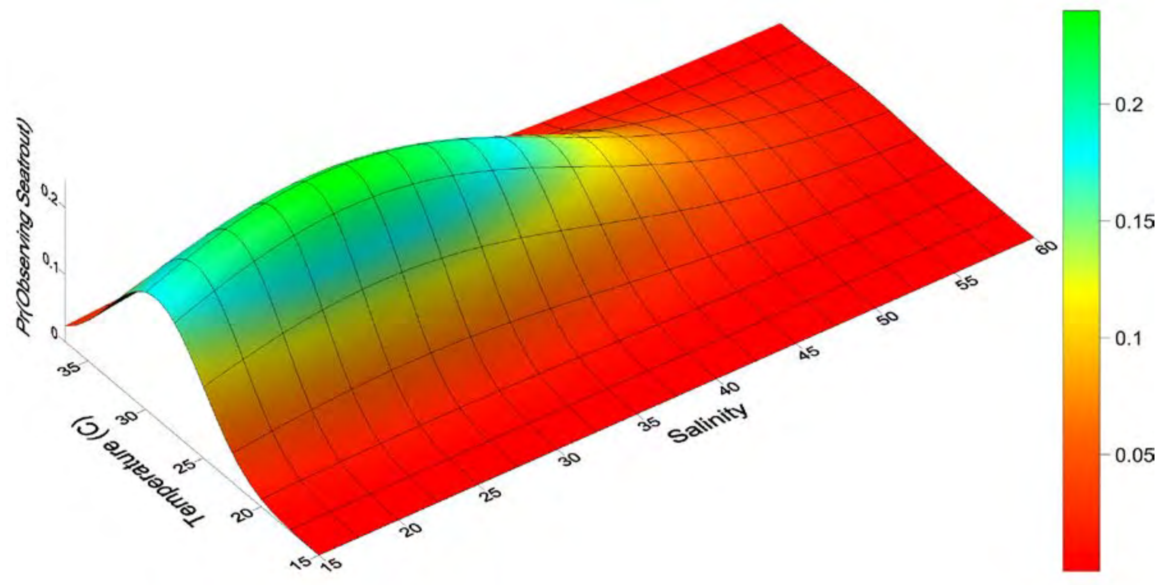


Figure 7-61. Scatter plots depict the correlation of the juvenile gray snapper populations with salinity within each subregion. The black squares are frequency of occurrence, the blue diamonds are concentration when fish are present and the red circles are abundance as delta-density. Only statistically significant linear regressions are depicted.

To further investigate the relationships among juvenile spotted seatrout and gray snapper, with temperature and salinity, logistic regressions were employed to develop habitat suitability index models. These models determine how temperature and salinity at time of observation contribute to the habitat quality for these species. The models suggest a strong effect of salinity on both species, but disparate responses (**Figure 7-64**). Juvenile spotted seatrout prefer lower salinity, as evidenced by their highest quality habitat occurring at salinity less than 37 and a peak between 25 and 30 , which is on the low end of currently observed salinity. Juvenile gray snapper prefer salinity between 30 and 45 , with a peak between 35 and 40 . This is much closer to the mid-range of current salinity and thus explains the lack of a linear response in the observational data. However, if CERP's water alterations minimize hypersalinity, particularly the occurrence of salinity greater than 40 , this should benefit both species.

It has been well documented that both juvenile spotted seatrout and gray snapper are highly dependent upon seagrass for habitat. To investigate this relationship, the population parameters for both species were compared to the mean seagrass Braun-Blanquet score recorded. For both species, frequency of occurrence, abundance, and concentration increased linearly with seagrass Braun-Blanquet scores (**Figure 7-65**). This suggests that improving seagrass habitat, in particular, the coverage of seagrass in Florida Bay will provide substantial benefits to the sportfish populations.



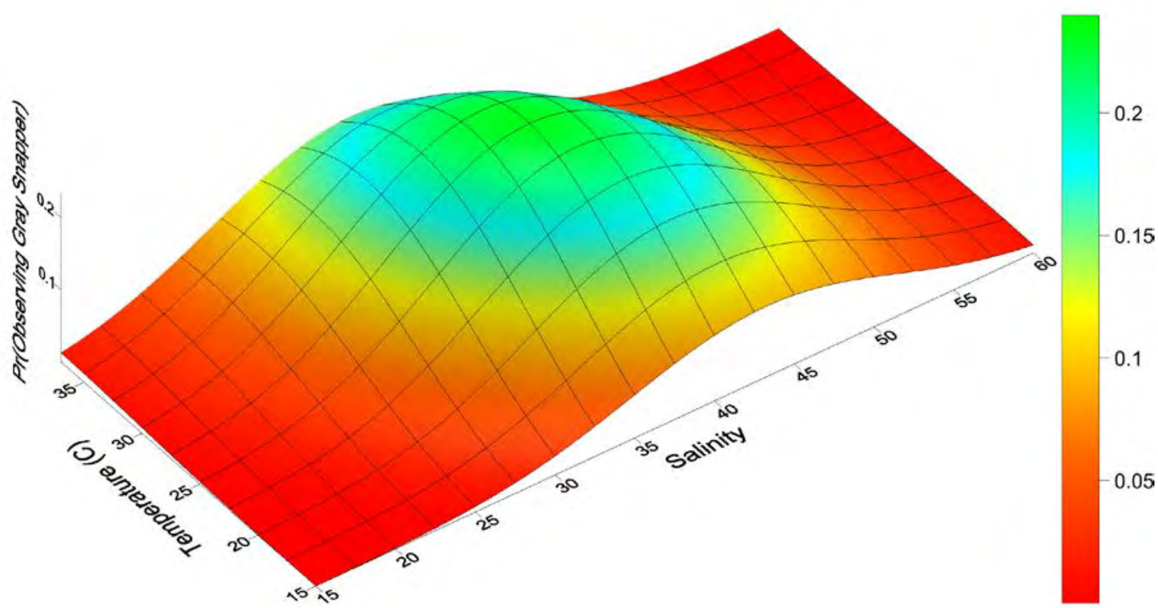


Figure 7-62. The probability of observing juvenile spotted seatrout (top panel) and gray snapper (bottom panel) across the range of salinities and temperatures in Florida Bay extracted from logistic regression analyses.

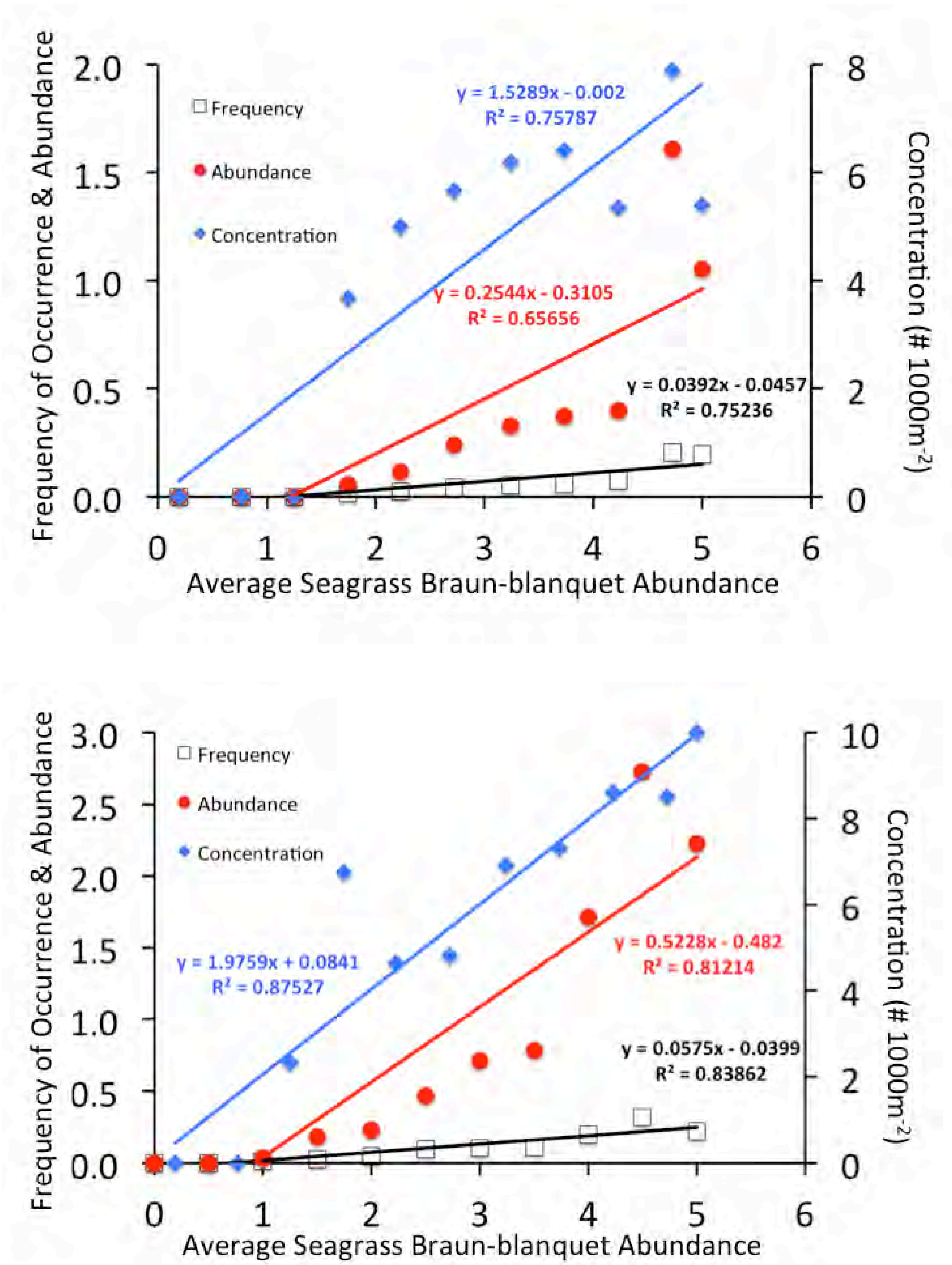


Figure 7-63. Scatter plots depicting the correlation of the juvenile spotted seatrout population (top panel) and gray snapper (bottom panel) with the average seagrass Braun-Blanquet abundance for all subregions combined. The black squares are frequency of occurrence, the blue diamonds are fish concentration when present, and the red circles are abundance as delta density. Only significant linear regressions are depicted.

Florida Bay and Lower Southwest Coast Roseate Spoonbills

Roseate spoonbills (*Platelea ajaia*) are a CERP indicator species because their reproduction is dependent on aquatic food that abnormally fluctuates due to altered hydrologic and water quality conditions in the Greater Everglades wetlands (Lorenz et al. 2009). As a result, the foraging distributions and nesting patterns of spoonbills have been altered due to the redistribution of high concentrations of prey organisms (Lorenz et al 2009). Restoration of historic spatio-temporal patterns of prey production and concentration is expected to reestablish wading bird nesting colonies in the coastal and tributary regions of the southern Everglades and, specifically, reestablish roseate spoonbill nesting colonies in northeastern Florida Bay, which should also increase the numbers and success of nesting spoonbills.

Since the 2009 SSR, the Florida Bay roseate spoonbill population has modestly increased according to nest numbers, with the exception of the 2010–2011 season, which had the lowest number of nests baywide (87) since the early 1960s (**Figure 7-66**). A continuing downward trend had been occurring since the late 1980s. Although the 2010–2011 low nest numbers was of concern, the positive aspect of this was that spoonbills nested successfully (defined as > 1 chick per nesting attempt) that same year producing about 2 chicks per nest attempt (c/n) making it the fifth time in six years that spoonbills nested successfully in Florida Bay (**Figure 7-67**). Decreased nesting effort resulted in low overall productivity of the population during the period from 2006 through 2011. It was hoped that the chicks raised during this period would enter the breeding population thereby reversing the declining nest numbers (the reproductive age of spoonbills ranges from 2 to 5 years, averaging 3.5 years). This appears

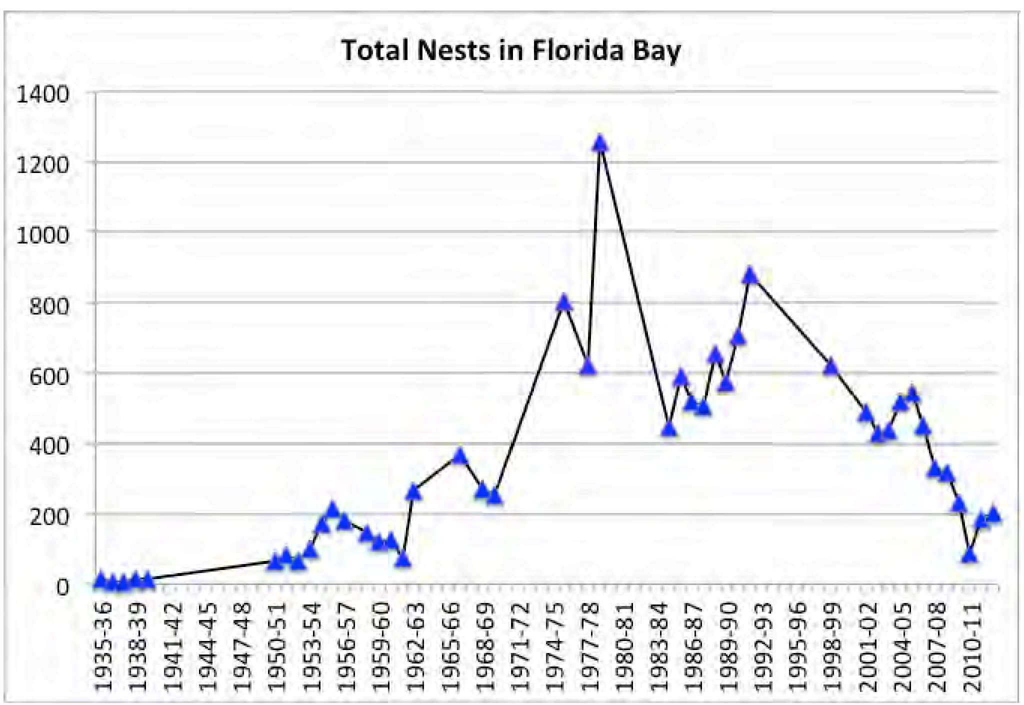


Figure 7-64. Number of spoonbill nest observed in Florida Bay since the species became re-established following their extirpation from Florida during the plume hunting era.

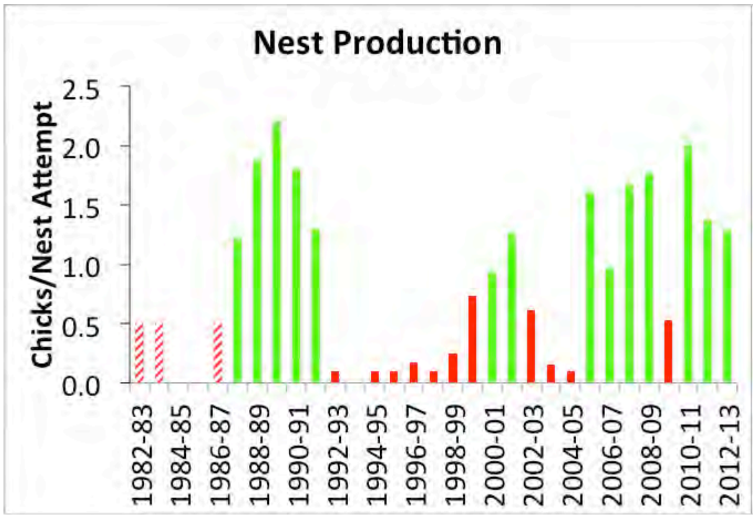


Figure 7-65. Spoonbill nest production in northeastern Florida Bay since 1982–1983. No surveys were performed in 1984–1985, 1985–1986, and 1993–1994. Red indicates a failed nesting attempt (> 1 chick per nest), green indicates successful nest attempt. Cross-hatching indicates that the nesting attempt was recorded as failed, however, no numerical value was available.

to have happened in 2011–2012 and 2012–2013, when 184 and 203 nests were recorded, respectively (**Figure 7-66**). Another positive sign for spoonbill recovery was that the Florida Bay population successfully nested in both 2011–2012 and 2012–2013, bringing the success rate to seven times in eight years.

The recent nesting success is in stark contrast to the poor nesting success that occurred from 1982–1983 to 2004–2005 (**Figure 7-67**). During that period, nest success was documented during 20 breeding seasons and spoonbills were successful in only 7 of those seasons. Spoonbills, like other wading birds are believed to need an average production rate of 1 c/n per nesting cycle for the population to remain stable and greater than that to increase population numbers. The low reproductive success from the early 1980s to 2004–2005 explains the declining nest numbers during this period (**Figure 7-67**). Also, like other tactile feeding wading birds, successful nesting seasons in the currently managed Everglades system is largely restricted to drought years and the years following a drought (Frederick and Ogden 2001). All 7 successful nesting seasons during this period occurred during drought years or the year following a drought (there was a prolonged 4 year drought from 1987–1988 to 1990–1991 and 2000–2001 was a drought year). This drought-dependent nesting success was not the case before modern water management practices began to impact Florida Bay (circa 1960). Prior to the completion of the C-111 canal, spoonbills in Florida Bay produced 2 c/n on average and were successful in almost every year for which data is available (Lorenz et al. 2002) regardless of rainfall. Between 1984 (when the South Dade Conveyance System began to heavily affect water flow to Florida Bay) and 2002, spoonbills only produced 0.7 c/n on average and were successful in only one-of-three years (Lorenz et al. 2002). It appears that until 2005–2006, the operations of the South Dade Conveyance System resulted in adverse drying patterns except during very low rainfall years.

Successful nesting seasons from 1987–1988 through 1991–1992 resulted in a sharp increase in the nest numbers in the early 1990s. The recent nesting success has followed this same pattern. This is a logical succession as young birds raised during the successful years enter the breeding population several years later and swell the number of nesting pairs. For example, between 1985–1986 to 1989–1990 spoonbill nest numbers fluctuated around 600, but then as the spoonbills that hatched during the successful nesting years (1987–1988 and 1988–1989) entered the reproductive population, numbers jumped to more than 700 nests in 1990–1991 and almost 900 nests in 1991–1992.

Beginning with the 2005–2006 breeding season, conditions have been favorable for nesting spoonbills. Based on rainfall data collected at spoonbill foraging sites, only two of the successful years were drought years or years following a drought year (2005–2006 was a year following a drought while 2008–2009 was a drought year). Interestingly, 2009–2010 was a year following a drought and was the only year of the last eight years that spoonbills were not successful but this failure was attributed more to the record cold event of January 2010 and subsequent extreme rainfall.

Although myriad conditions play into whether spoonbills are successful or not, two key factors may account for the recent success. One is rainfall. Although there were only two years during this period that could be called drought years, most years had moderate rainfall patterns, i.e., there were very few high rainfall events that would result in poor foraging conditions. The single year that had high precipitation was 2009–2010, the year that nesting failed. Moderate rainfall years reduce the likelihood that either natural or anthropogenic reversals in the drying pattern of the wetland that serve as spoonbill foraging grounds would occur (reversals result in poor foraging conditions as prey switch from high to low concentrations because they spread out over the newly wetted habitat). Secondly, beginning in 2005, water managers began consulting with field biologists in general and specifically with spoonbill biologists before operational decisions were made. Operational options were evaluated and previously ecologically damaging operations were avoided or mitigated to the extent possible. Under less moderate conditions, such ecologically damaging decisions would still likely have to be made (as was the case in 2009–2010); however, these consultations, at least in part, have likely contributed significantly to the recent success. To date, the effect of these consultations on actual hydrologic conditions has not been evaluated numerically; this will hopefully be done in the near future.

A surprising and fortuitous recent finding was that, beginning in 2009–2010, spoonbills had returned to a colony site on the mainland near Taylor River that had been inactive since the early 1980s. This particular site was never reported as a spoonbill nesting stronghold. This colony is extraordinarily difficult to access and it was not surveyed by ground in 2009–2010 or 2010–2011. However, aerial surveys during that time suggested a high degree of spoonbill nesting activity. The colony was surveyed twice during 2011–2012, resulting in an estimate of 164 nests with a high degree of success (> 1 chicks per nest). The colony was surveyed three times in 2012–2013. A thorough nest count could not be performed because the presence of the field biologists caused the adults to leave their nests, opening up eggs and chicks to predation by numerous crows that followed researchers into the colony. Researchers were in the colony less than 10 minutes before this threat manifested and the survey was abandoned, but that was sufficient time to assess that the colony was likely as large or larger than in 2011–2012 and nesting was successful. The fact that this colony is located just 3 kilometers (km) north of Florida Bay and that adults in breeding plumage that were banded as nestlings in Florida Bay colonies were resighted throughout the colony indicate that these nests should be included with the Florida Bay population estimate. However, if this colony is included in that population estimate, then nest estimates for Florida Bay were inaccurately reported as low in 2009–2010 and 2010–2011 and the current population is now approaching 400 nesting pairs nearly doubling the previous count of overall nest numbers.

In summary, spoonbill numbers appear to be increasing as a result of the recruitment of young birds into the adult population in response to high levels of nesting success over the last eight years. This may be an optimistic statement because two years of increasing nest numbers does not make a trend. It appears that open communication between water managers and ecologists have contributed substantially to improvements in operation for the benefit of spoonbills in Florida Bay. Since spoonbills are an accepted indicator for the ecological health of Florida Bay (Lorenz et al. 2009), this suggests that current management practices are more beneficial to Florida Bay than they had been in the past.

Florida Bay and Lower Southwest Coast Crocodilians

Crocodilians, which include crocodiles and alligators, integrate biological impacts of hydrological operations throughout their life and are critical in the food web as top predators, influencing abundance and species composition of prey (Mazzotti and Brandt 1994, Mazzotti et al. 2009).

The distribution and abundance of crocodiles in estuaries is directly dependent on timing, amount, and location of freshwater flow (Dunson and Mazzotti 1989, Mazzotti and Dunson 1989). Therefore, responses of crocodiles are directly related to suitability of environmental conditions and hydrologic change (Mazzotti and Brandt 1994, Mazzotti et al. 2007, Mazzotti et al. 2009). This section reports on the status and trends of crocodiles in SCS, and includes an assessment of crocodile response to an earlier restoration project in ENP to validate the CERP hypothesis that crocodiles will respond positively to decreased salinity conditions.

Alligator reproduction and survival is dependent upon suitable hydrologic and salinity patterns. Current populations in Everglades marshes and estuaries are suppressed because of altered hydrology and salinity (Mazzotti and Brandt 1994, Shinde et al. 2012, RECOVER 2011). Historically, they were abundant in the freshwater mangrove zone (Beard 1938, Craighead 1968, Brown 1993, Simmons and Ogden 1998) in areas of lowest salinity, < 20 (Mazzotti 1983, Rosenblatt and Heithaus 2011). When patterns of freshwater flow change salinity conditions, alligators change their location (Birkhead and Bennett 1981). Nesting rarely occurs in areas where salinity is greater than 10–12 (McNease and Joanen 1978, Wilkinson 1983).

<ins>*American Crocodiles*</ins>

Positive or negative trends of populations of American crocodiles relative to hydrologic changes permit assessment of positive or negative trends in restoration. Restoration success or failure can be evaluated by comparing recent and future trends and status of crocodile populations with historical or reference population data, and model predictions.

CERP hypotheses for crocodiles are as follows (RECOVER 2004a, Section 3.1.2.6):

- Restoration of freshwater flows and salinity regimes to estuaries will increase growth and survival of crocodiles.
- Restoration of location of freshwater flow will result in an increase in relative density of crocodiles in areas of restored flow, such as Taylor Slough/C-111 Canal drainage.

Water salinity affects populations of crocodiles (Dunson and Mazzotti 1989, Mazzotti and Dunson 1989). Although there are higher numbers of crocodiles in more places today than when they were declared endangered in 1975, virtually all of that increase is due to crocodiles occupying and nesting in man-made habitats such as the Turkey Point Power Plant site and along the Buttonwood and East Cape canals in ENP (Mazzotti et al. 2007). The mangrove back-country of northeastern Florida Bay has consistently been considered core habitat of the American crocodile in Florida (Ogden 1978, Kushlan and Mazzotti 1989, Mazzotti 1999, Mazzotti et al. 2007). Today this physically unaltered area suffers from diversion of fresh water (McIvor et al. 1994). Of the locations where crocodiles are found, growth rates and survival of crocodiles are lowest in this area (Mazzotti et al. 2007).

For this SSR, the relationship of salinity patterns relative to density of crocodiles during WY 2004 to WY 2012 was examined. Also, the effects of an early ecosystem restoration project (begun in 1986) to restore salinity patterns in the Flamingo/Cape Sable area of ENP were assessed. In that project; the Buttonwood, East Cape, and Homestead canals in ENP were plugged to prevent saltwater intrusion and loss of fresh water to tide.

Methods

Relative Density. Two measures of relative density were used to track changes in the American crocodile population: (1) encounter rate measured as crocodiles per km, and (2) nesting effort measured as number of nests per year. Nesting effort responds on a longer time step (10 to 15 years) than relative density (2 to 3 months). Nesting effort and success were determined using nighttime spotlight surveys and surveys for signs of nesting activity according to established protocols (Mazzotti et al. 2010). Spotlight surveys conducted between January 2004 and December 2012 were initially performed quarterly but were reduced to three times per year in 2010 (Mazzotti et al. 2010). Relative densities of crocodiles were examined throughout the study period, across the study area. Unfortunately, limited resources have precluded the collection of salinity data to support the assumption that canal plugging has reduced salinity in the wetlands.

Approximately 550 km of the south Florida coastline was surveyed, including interior creeks and rivers as potential crocodile habitat. Survey routes are grouped into zones (**Figure 7-68**) and habitat is characterized as canal, cove, pond, creek/river, or exposed shoreline. Metrics compared included number of animals observed over time, encounter rates, the number of nests observed, and nest success along survey routes.

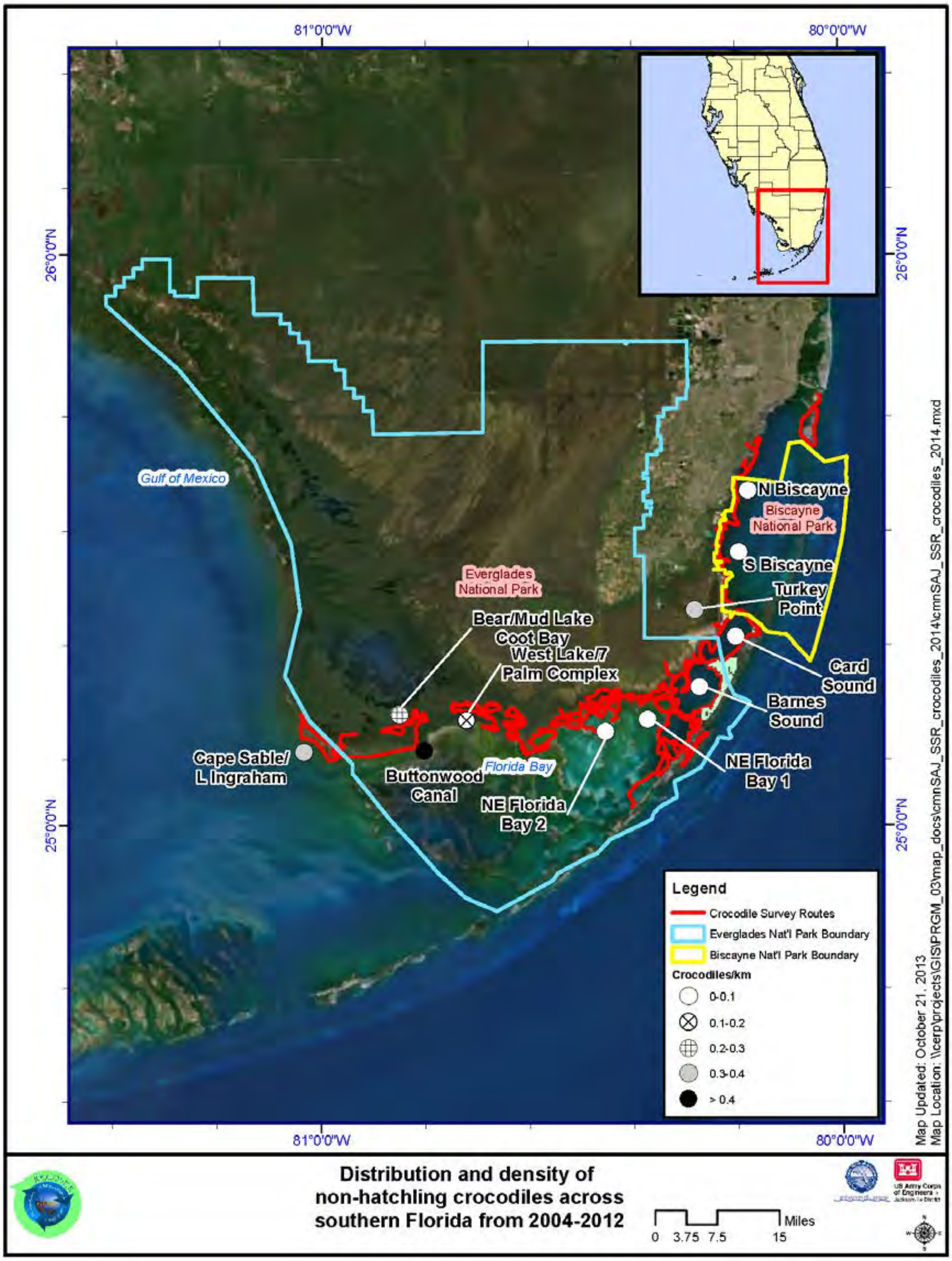


Figure 7-66. Distribution and density of nonhatchling crocodiles across south Florida from WY 2004 to WY 2012 reported as number of crocodiles encountered per km of survey route. Survey routes are grouped into zones: North (N) Biscayne, South (S) Biscayne, Turkey Point, Card Sound, Barnes Sound, Northeast (NE) Florida Bay, West Lake/Seven Palm Complex, Buttonwood Canal, Mud/Bear Lake/Coot Bay, and Cape Sable/Lake Ingraham.

Evaluating effects of plugging Buttonwood, East Cape, and Homestead canals addressed two questions. Did the number of nests per year increase in response to lower salinity, especially in comparison to northeastern Florida Bay where diverted freshwater flow and salinity patterns are currently the target of restoration? Is the relative density of crocodiles greater in areas with restored salinity patterns compared to areas with diverted freshwater flow?

Data Analyses. Differences in relative density throughout the study period and across the study area were determined using linear regression, and ordered logistic models were used to test for trends over the years. A one-way analysis of variance (ANOVA) was used to analyze the distribution of crocodiles across habitat types. Density estimates in relation to environmental conditions, e.g., salinity, was calculated using linear regression. A one-way ANOVA was used to assess density estimates in response to restoration plugs by comparing areas nearest the plugs (Buttonwood Canal and Cape Sable/Lake Ingraham) relative to Joe Bay/Little Madeira Bay in Northeast Florida Bay.

Results

Relative Density. A total of 2,177 crocodiles, of which 1,595 were nonhatchling crocodiles, were observed throughout the study area across the nine water years surveyed. Areas from Northeast Florida Bay to Cape Sable account for more than 80% of crocodile density in south Florida (**Figure 7-69**). Buttonwood Canal, Turkey Point, and Cape Sable/Lake Ingraham, however, had the highest relative densities of nonhatchling crocodiles across the nine water years (0.43, 0.38, and 0.33 crocodiles per km, respectively, **Figure 7-69**). Buttonwood Canal had the highest relative crocodile density during a single

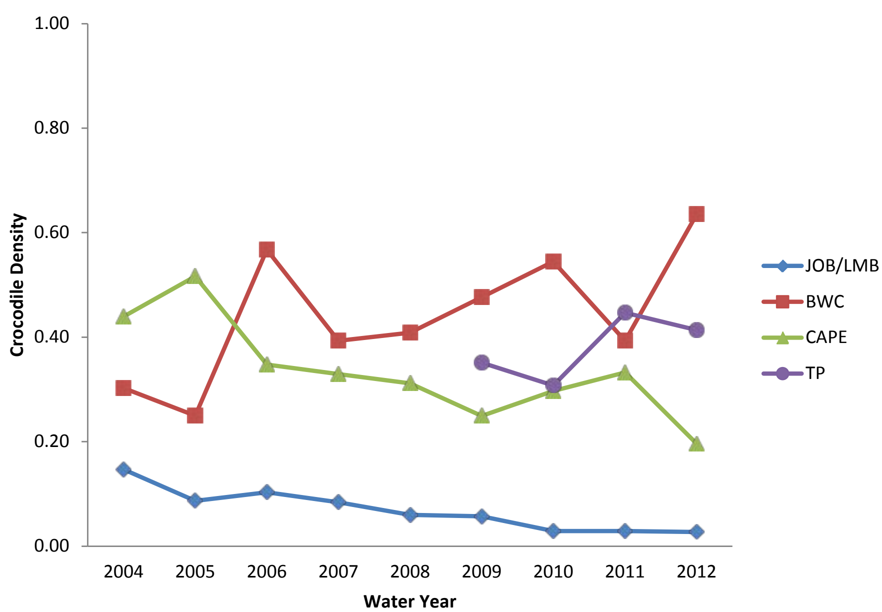


Figure 7-67. Non-hatchling crocodile density calculated as encounter rate per km of route surveyed during study period of 2004–2012. JOB/LMB = Joe Bay/Little Madeira Bay, BWC = Buttonwood Canal, CAPE = Cape Sable/Lake Ingraham, TP = Turkey Point was surveyed from 2009 to 2012.

survey (2.54 crocodiles per km). Across all nine water years, there is a decrease in relative density of crocodiles (β = -0.0049; P = 0.018). However, from WY 2007–WY 2012, crocodile density was remarkably stable (**Table 7-9**). Also, there was a difference in habitat use by crocodiles ($F = 30.168$, df = 3, 32, $P < 0.001$) whereby 85% of all crocodile sightings occurred in canals and coves and 15% in ponds, creeks/rivers, or exposed shorelines.

Table 7-9. Overall nonhatchling crocodile density throughout the study site between WY 2004–WY 2012.

Water Year	Crocodile Counts	Total Distance Surveyed (km)	Density (crocodiles per km)
2004	260	1,917.9	0.14
2005	198	1,859.6	0.11
2006	248	2,060.7	0.12
2007	182	2,000.6	0.09
2008	170	1,869.0	0.09
2009	173	2,010.6	0.09
2010	121	1,457.3	0.08
2011	140	1,521.7	0.09
2012	95	1,020.1	0.09

The number of crocodile nests per year has been generally increasing over the past twenty-five years (**Figure 7-70**). Crocodile nests peaked in the Flamingo/Cape Sable area in 2008 and in Northeast Florida Bay in 2012. Initiation of crocodile nesting along East Cape and Buttonwood canals was coincident with the plugging of those canals in the 1980s (**Figure 7-70**, Mazzotti et al. 2007). Nesting effort has increased faster in the Buttonwood and East Cape canals than in Northeast Florida Bay (**Figure 7-70**).

Evaluation of the Flamingo/Cape Sable Canal Plugs. To assess restoration efforts, relative density of crocodiles was compared between areas nearest the canal plugs (Buttonwood Canal and Cape Sable/Lake Ingraham [including Homestead Canal]) to Joe Bay/Little Madeira Bay in Northeast Florida Bay). There were higher crocodile encounter rates in both Buttonwood Canal and Cape Sable/Lake Ingraham relative to Joe Bay/Little Madeira Bay ($F = 36.677$, df = 2, $P < 0.001$; **Figure 7-69**). Encounter rates of crocodiles were highly correlated with survey areas ($r = 0.766$, $P < 0.001$) but were not correlated with water year ($r = -0.057$, $P = 0.776$; **Figure 7-69**). On average, there is a decrease in relative crocodile density in Joe Bay/Little Madeira Bay ($\beta = -0.0137$; $P = 0.00021$) and Cape Sable/Lake Ingraham ($\beta = -.0285$; $P = 0.0070$), but an increase in Buttonwood Canal relative crocodile density ($\beta = 0.03017$; $P = 0.059$).

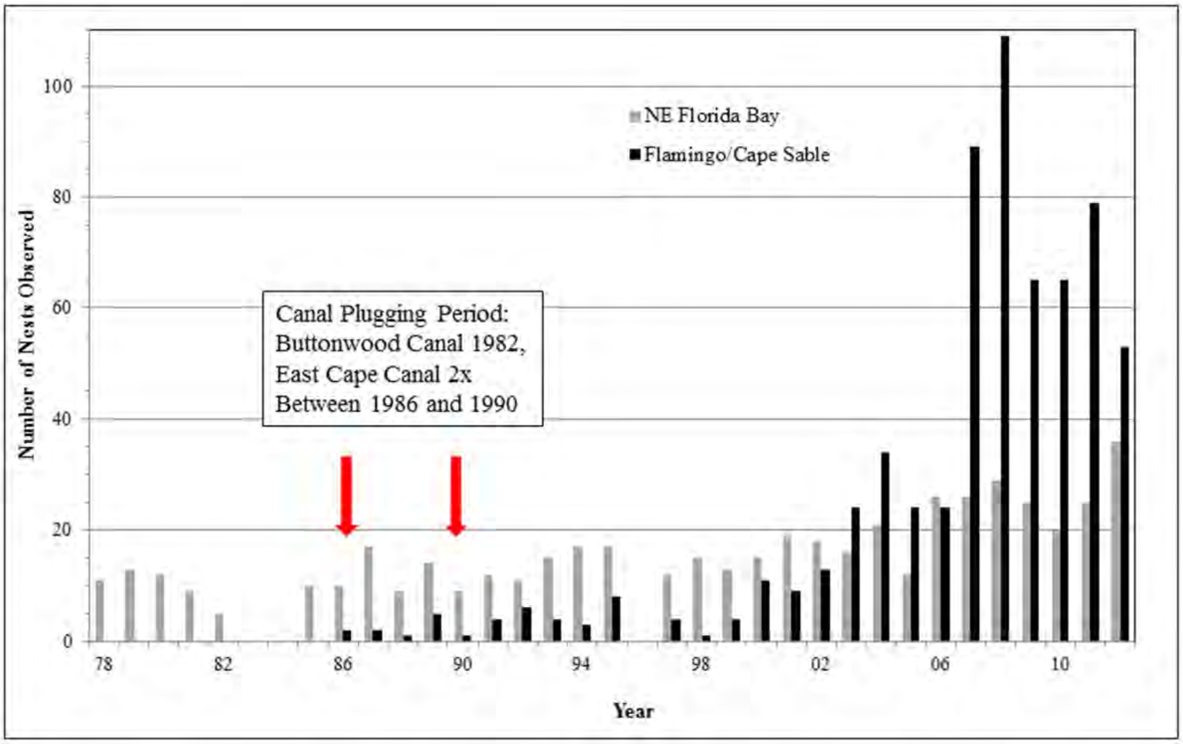


Figure 7-68. Number of American crocodile nests per year found in Northeast Florida Bay (Little Madeira and Joe bays) and the Flamingo/Cape Sable area (Buttonwood and East Cape canals) during 1978–2012. Crocodile nests were first discovered in the Flamingo/Cape Sable area after Buttonwood and East Cape canals were plugged in the early 1980s. Since then, the number of nests per year has increased more rapidly in the Flamingo/Cape Sable area.

Implications

As noted above, salinity data are not available for the Cape Sable area however, it is logical to assume that plugging of canals resulted in reduced salinity in wetlands upstream of the plugs because of significant reductions in saltwater intrusion from Florida Bay and the Gulf of Mexico. The three locations with the highest densities of crocodiles were artificial habitats (canals) with presumably restored salinity regimes (East Cape and Buttonwood canals) or with artificial juxtaposition of low salinity habitats near saline habitats (Turkey Point Power Plant site, see Brandt et al. 1995 for description). This confirms the CERP hypothesis that restoration of salinity regimes will benefit crocodiles. Restoration of salinity regimes in the Flamingo/Cape Sable area also was coincident with an increase in crocodile nesting. That the increase in crocodile nests was more rapid in areas with restored salinity regimes (Flamingo/Cape Sable) than in areas with static salinity regimes (Northeast Florida Bay and Turkey Point) also supports the CERP hypothesis. The rapid increase in nesting that began in 2000–2002 is about 15 years post restoration activities in the Flamingo/Cape Sable area (**Figure 7-70**). This is the time step expected for nesting response to environmental change and is related to the amount of time it takes for a surviving hatchling to enter the breeding population.

Encounter rates for crocodiles respond on a much shorter time step. It is hypothesized that the decline in encounter rates for crocodiles on Cape Sable after 2005 is a response to Hurricanes Katrina and Wilma overrunning and compromising an already damaged plug in East Cape Canal. This would result in increased salinity and a quick response in terms of encounter rates for crocodiles. The plug in Buttonwood Canal was not compromised by those hurricanes and, hence, restoration benefits of a restored salinity regime continue. In Northeast Florida Bay, continued diversion of fresh water to tide has resulted in a continued decline in relative density of crocodiles, similar to that reported for estuarine alligators in this report.

Having crocodile attributes that respond on both short (relative density) and longer (nesting) time scales and having a long-term data set to work with make it possible to test and confirm the hypotheses about the importance of freshwater flows to the ecological functioning of SCS. Continued monitoring of crocodilians in these areas will continue to inform on progress toward restoration.

Alligators

This section reports on the trends in alligator relative density from 2003 to 2012 in SCS. The CERP hypothesis for alligators is that restoration of estuarine salinity regimes will expand the distribution and abundance of reproducing alligators into oligohaline portions of the estuaries. This section provides information from WY 2003–WY 2012 on patterns of alligator relative density in Shark River, ENP, one of the areas where alligators were once relatively abundant that will ultimately benefit from restoration of freshwater flows.

Alligators were surveyed for relative density twice during spring and twice during fall according to protocols established and documented for the RECOVER MAP (Mazzotti et al. 2010; **Figure 7-71**). Results are provided for the estuarine route in Shark River. Additional results for routes in the freshwater wetlands are presented in Chapter 7: Greater Everglades. The analysis uses data from WY 2003 through WY 2012 to look at the 10-year trend and trends in five-year increments.

Multiple regression analysis was used to examine trends in nonhatchling alligator relative density (also called total population). Trends from WY 2003–WY 2012 and in five-year increments starting in 2003 were examined. The model regresses log-transformed counts of alligators per km (dependent variable) on water year, season (fall and spring), transect, average water depth (AWD), and average water temperature (AWT):

$$Log\left(\frac{n+1}{transect\ length}\right) = \mathrm{f}(water\ year, season, transect, AWD, AWT)$$

Where n is the count of alligators and AWD and AWT are the average of the values measured on the surveys.

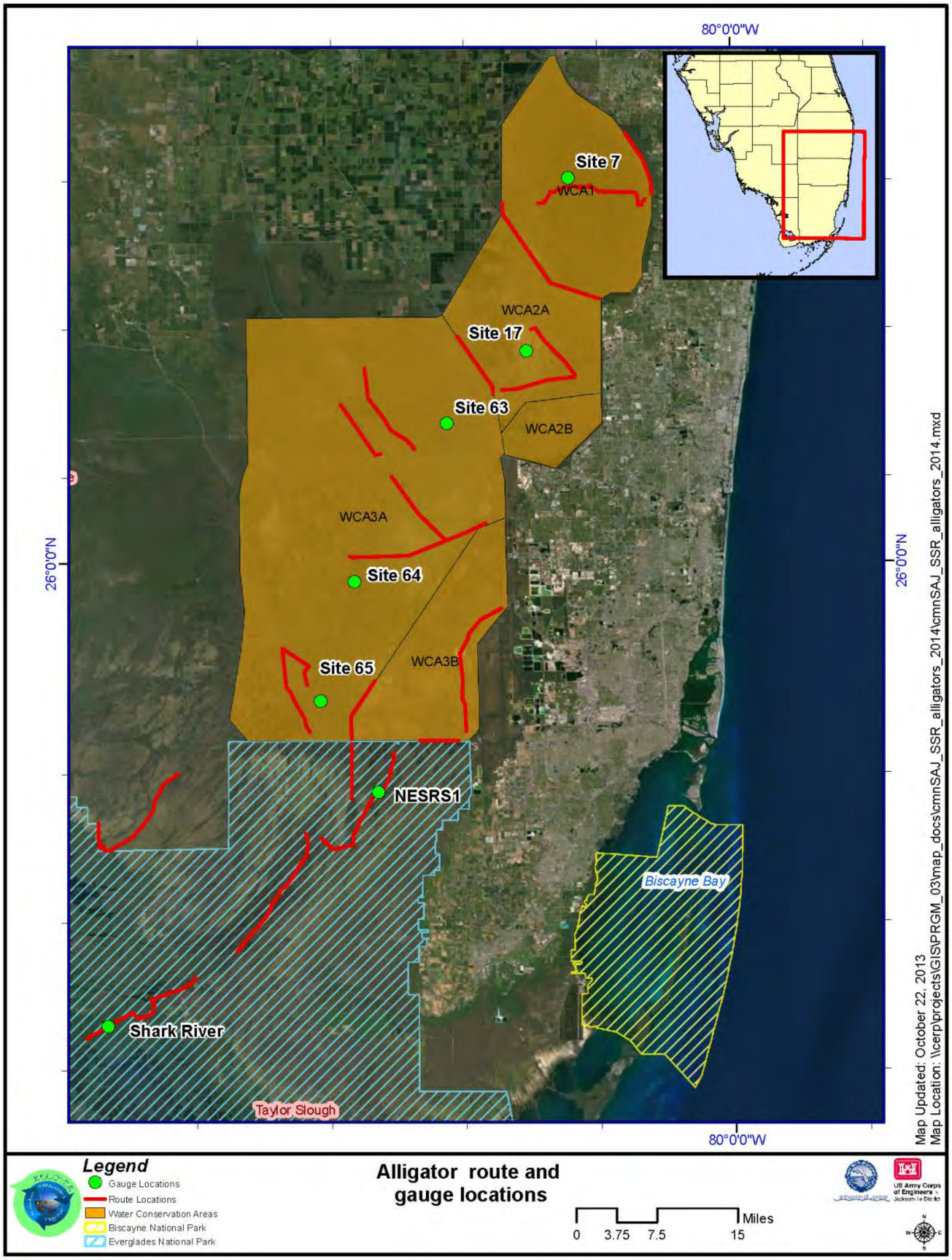


Figure 7-69. Location of alligator survey routes and gages used for WY 2003–WY 2012 analysis of trends in alligator relative density. Results for the Greater Everglades routes are presented in the Greater Everglades section.

The only significant trend was a negative trend in the WY 2003–WY 2012 time period when there was an average annual decrease in alligator density of 8.1% per year (**Figure 7-72**) in the Shark River system. Average annual salinity ranged from 8.7 in 2004 to 20.0 in 2008. Mean salinity from WY 2002–WY 2006 (12.5) was significantly lower than the WY 2007–WY 2012 mean salinity (17.0 ; t = -3.138; $p < 0.05$), and these differences likely explain the negative trend in the WY2007–WY 2011 time period. No other five-year time period showed a significant trend.

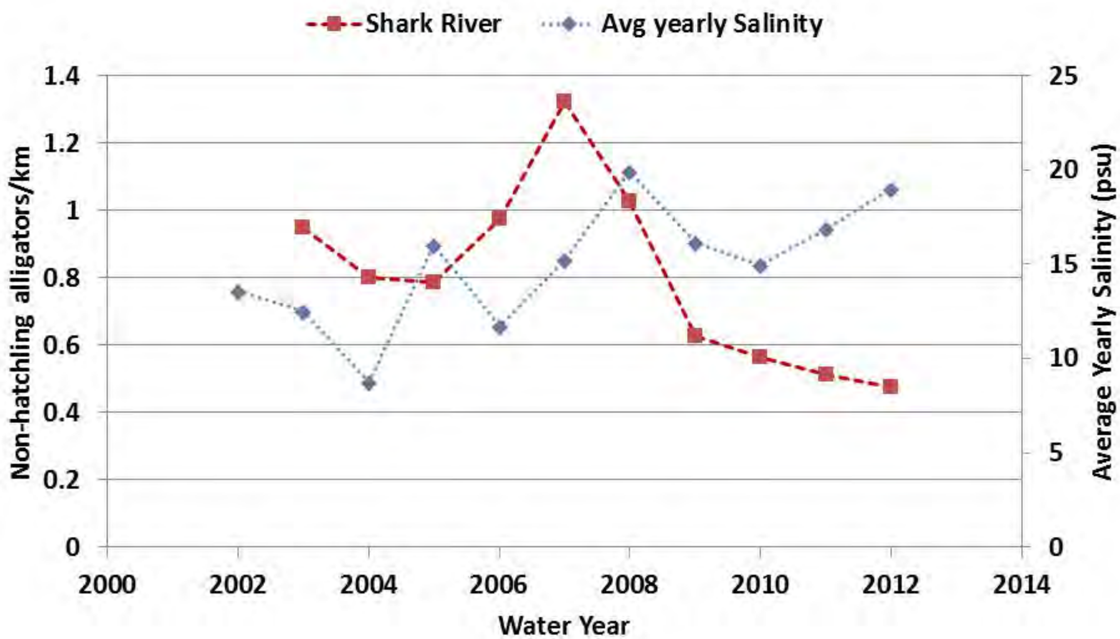


Figure 7-70. Average alligators per km of two spring and two fall surveys and average (Avg) yearly salinity by water year in Shark River. Salinity data are from Shark River site 'GS'/SRG. (Practical salinity units (psu) is an outdated unit used with salinity measurements. Salinity is now considered unitless.)

Even at its peak in WY 2007, alligator abundance in Shark River was below the restoration target of > 1.7 alligators per km. This is likely a reflection of average salinity in this area that was generally higher than historical salinity. The correspondence of higher salinity in the later study years (WY 2007–WY 2012) with a decreasing trend in alligator relative density is further evidence of the role salinity has in explaining alligator relative density patterns. Changes in relative density are likely a result of alligators moving out of higher salinity areas (Rosenblatt and Heithaus 2011) and limited successful reproduction in areas that have salinity greater than 10 to 12 . These data support the hypotheses that increasing freshwater flows resulting in reduced salinity is an important factor in restoring historic densities of alligators to estuarine areas.

Upper Southwest Coast and Ten Thousand Islands

Assessments for the Upper Southwest Coast and Ten Thousand Islands area include three primary components. The first will be the status and trends from flow and salinity monitoring by the USGS and Rookery Bay National Estuarine Research Reserve (RBNERR). The second component assesses the estuarine nekton community (RBNERR), oyster dynamics (Florida Gulf Coast University), and American crocodile presence and nesting (FWC, RBNERR, United States Fish and Wildlife Service). The last component is found in the Picayune Strand Restoration Project section later in this chapter and is a fine-scale assessment of effects from constructed components of the PSRP.

Upper Southwest Coast and Ten Thousand Islands Salinity

The historical watershed of the Ten Thousand Islands (TTI) estuary has been altered by the construction of the SGGE Project, the Barron River Canal, and the Tamiami Trail (US-41). With restoration projects underway to improve freshwater delivery to TTI estuary, a collaboration of federal and state, CERP and non-CERP funding has supported a network of hydrologic monitoring (water level [tide], discharge, salinity, and temperature) stations in Blackwater River, Pumpkin River, Faka Union Canal, and East River (**Figure 7-73**). For the purpose of this assessment, discharge and salinity data shown are limited to USGS monitoring stations at rivers flowing into Fakahatchee Bay (East River), Faka Union Bay (Faka Union Canal), and Pumpkin Bay (Pumpkin River).

River Discharge

Discharge magnitudes at Faka Union Canal overwhelm those of East River and Pumpkin River (**Figure 7-74**). No significant or apparent changes in discharge were observed due to water management practices or restoration efforts. Discharge trends appear to follow rainfall patterns.

The data gap from late 2009–early 2010 is due to loss of PSRP pre-construction project funds and delays in implementation of RECOVER MAP funds. Measurements of discharge at East River were discontinued in April 2012 as a result of a decrease in RECOVER MAP funding, but plans are in motion to restore these data in Fiscal Year 2014.

River Salinity

Salinity at East River, Faka Union River, and Pumpkin River follow rainfall patterns of the dry and wet seasons (**Figure 7-75**). During the dry season, higher salinity values are observed at Pumpkin River for longer periods of time as compared to East and Faka Union rivers. This is most likely due to the fact that a significant portion of upstream flow to Pumpkin Bay has been redirected eastward as a result of the drainage canals constructed as part of the SGGE Project (i.e., the PSRP area). At the peak of wet season flows, Pumpkin River salinity rarely drops below 10 , while both East and Faka Union rivers drop below 5. The data gap from late 2009–early 2010 is due to loss of PSRP pre-construction project funds and delays in implementation of RECOVER MAP funds.

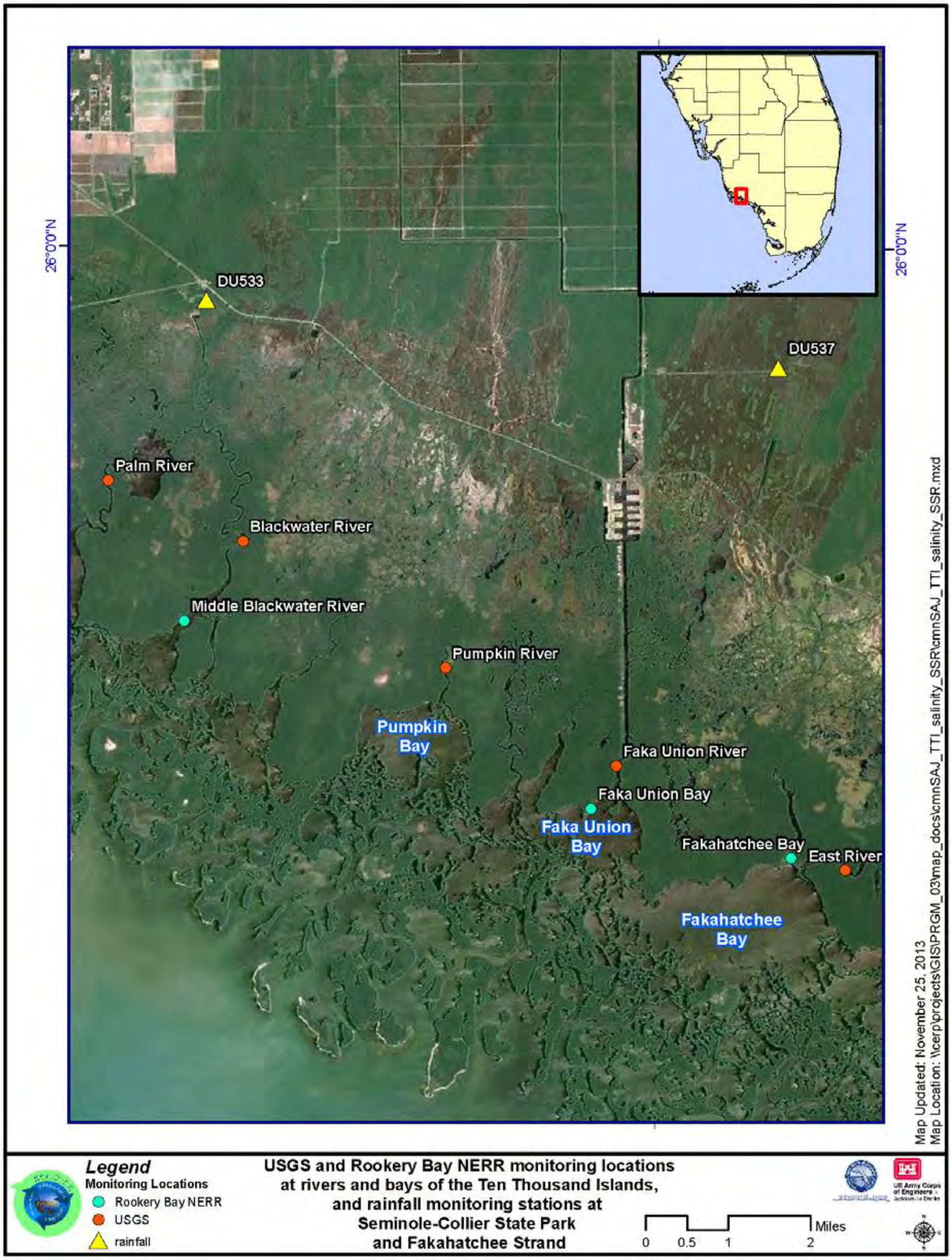


Figure 7-71. USGS and RBNERR monitoring locations at rivers and bays of the TTIs, and rainfall monitoring stations at Seminole-Collier State Park and Fakahatchee Strand Preserve State Park (from SFWMD's DBHYDRO database).

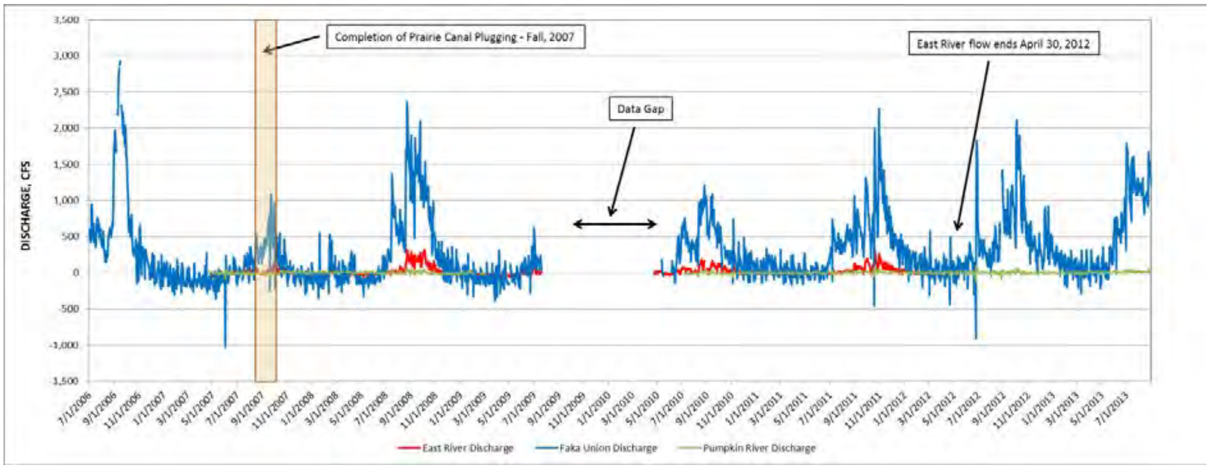


Figure 7-72. Mean daily discharge for USGS monitoring stations at East, Faka Union, and Pumpkin rivers for the period of July 1, 2006 to August 31, 2013.

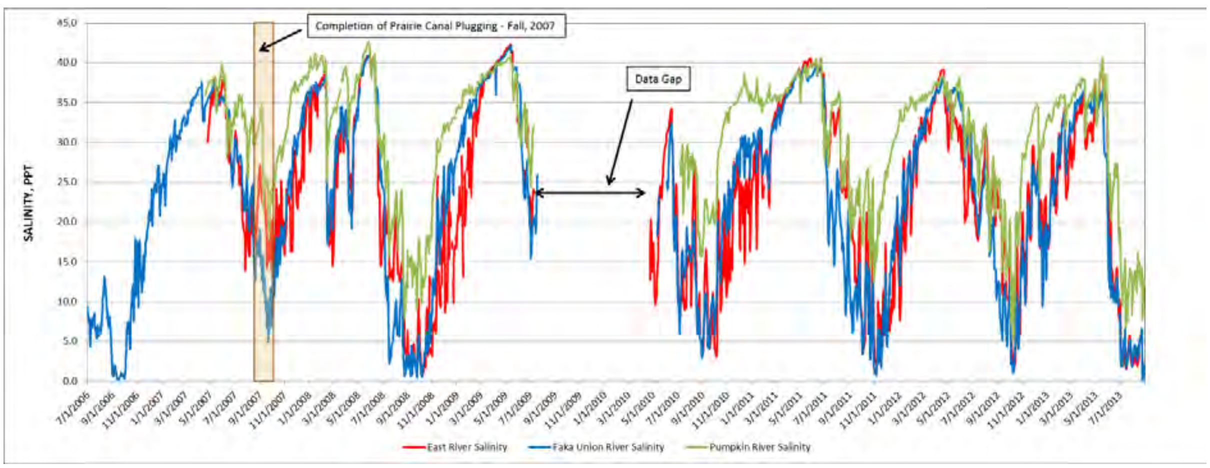


Figure 7-73. Mean daily salinity for USGS monitoring stations at East, Faka Union, and Pumpkin rivers for the period of July 1, 2006 to August 31, 2013.

It is important and surprising to note that salinity in the highly impacted Faka Union Canal closely tracks salinity in the relatively undisturbed East River. Given that flow in Faka Union Canal is at times several orders of magnitude greater than flow in East River, one would expect to see undesirably severe and prolonged low salinity conditions in Faka Union River, but this does not appear to be the case. However, the high volume of water entering Faka Union Bay causes low salinity to extend further out in the bay towards the Gulf of Mexico compared to the target salinity condition in Fakahatchee Bay (E. Patino, personal communication, September 17, 2013). Also, East River appears to maintain lower salinity into the dry season due to more gradual and prolonged sheet flow and river flow from upstream, perhaps mimicking a more natural pattern. No significant or apparent changes in salinity are observed due to water management practices or to restoration efforts. It is noteworthy that previous data show that effects from Faka Union Canal freshwater outflows extend east and west to neighboring bays (Soderqvist and Patino, 2010; http://pubs.usgs.gov/ds/501/pdf/ds501_report.pdf). Additionally, monthly salinity readings from oyster studies within Pumpkin Bay show that a reversed salinity gradient forms during high freshwater flow conditions at Faka Union Canal, suggesting that the effects from these outflows reach further west into Pumpkin Bay (P. Goodman, personal communication, November 19, 2013).

Dates and magnitude of high river salinity peaks vary depending on when heavy and/or frequent rainfall begins during the wet season. **Figures 7-76** and **7-77** show salinity and rainfall data during the period of April 1 to June 30, for 2008 (dry May) and 2012 (wet May). Note that river salinity during dry May 2008 was at or above a hypersalinity condition of 40 for 3 to 4 weeks; whereas, during wet May 2012, salinity at the end of the dry season never reached hypersaline conditions.

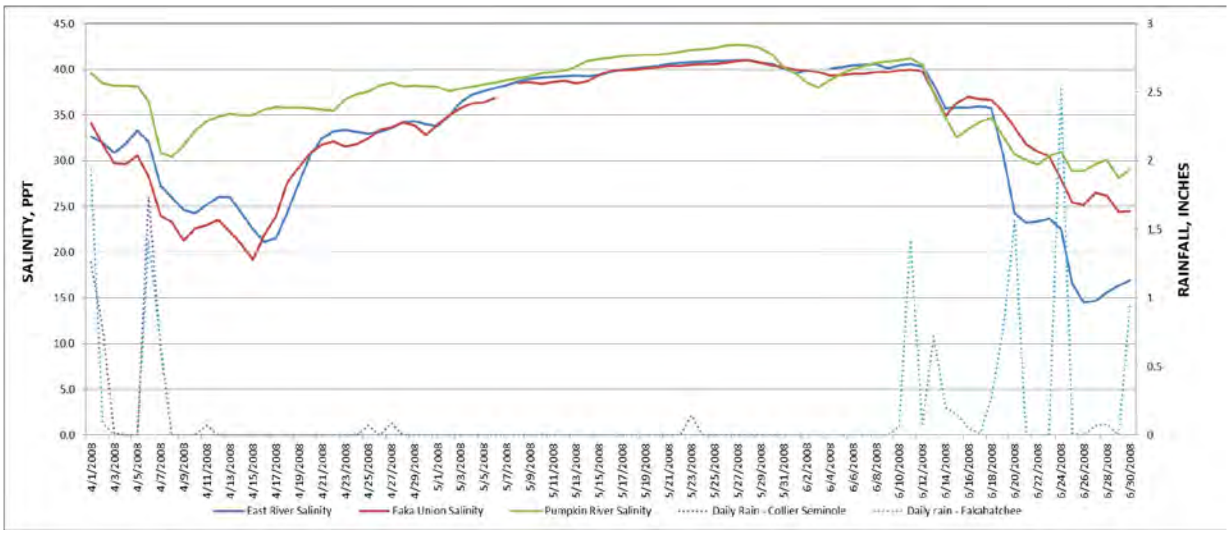


Figure 7-74. Daily rainfall at Collier-Seminole State Park and Fakahatchee Strand Preserve State Park stations and mean daily salinity at East, Faka Union, and Pumpkin rivers, for the period of April 1 to June 30, 2008 (dry).

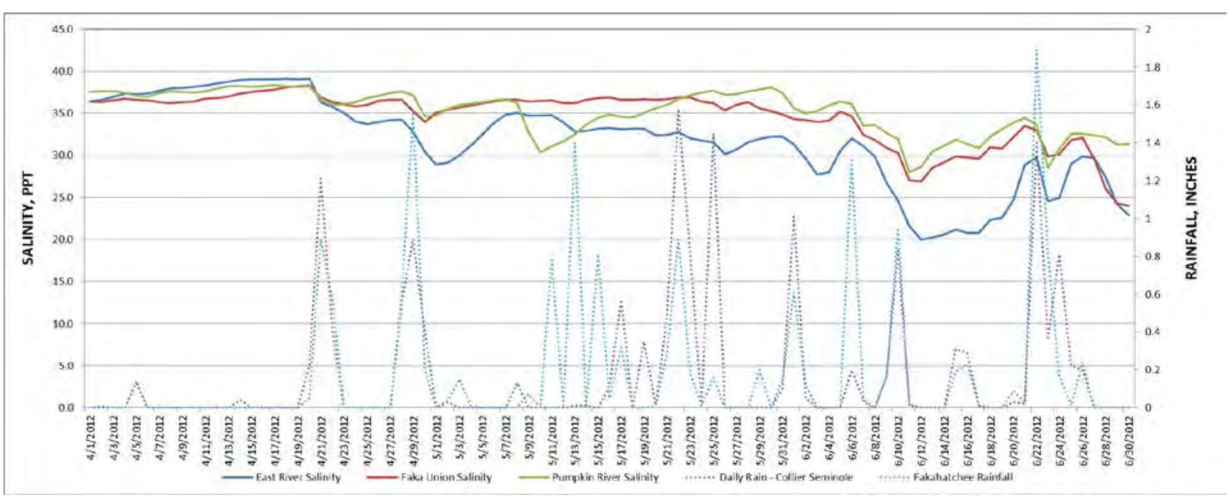


Figure 7-75. Daily rainfall at Collier-Seminole State Park and Fakahatchee Strand Preserve State Park stations and mean daily salinity at East, Faka Union, and Pumpkin rivers, for the period of April 1 to June 30, 2012 (wet).

Summary and Comments

No significant or apparent changes in river discharge and salinity trends are observed due to water management practices or to restoration efforts. Changes in river discharge and salinity trends are expected when all components of PSRP are completed. Additional high density salinity monitoring locations within Pumpkin, Faka Union, and Fakahatchee bays would improve the understanding of current conditions and help evaluate post-construction changes due to PSRP restoration efforts. Completion of all PSRP components as originally described in the project implementation report (USACE and SFWMD 2004) is dependent on additional authorization and appropriation by congress. Additional information can be found in the Picayune Strand Restoration Project section and the *Draft Picayune Strand Restoration Project Limited Reevaluation Report and Environmental Assessment* (USACE 2013).

Upper Southwest Coast and Ten Thousand Islands Fish Nekton

Fish nekton are directly and indirectly influenced by freshwater inflow to estuaries and thus serve as indicators of estuarine condition (Gilmore et al. 1983, Whitfield 1999). The natural freshwater inflow to most estuarine areas in the TTI has been altered by anthropogenic channelization of wetland sheet flow, which has substantially changed salinity regimes in affected areas. Altered freshwater inflow has been identified as the factor most affecting the health and biodiversity in the TTI estuaries (Shirley et al. 1997).

RBNERR conducted baseline nekton monitoring in the TTI area from 2000 to 2009. This effort established a pre-construction baseline for nekton that will later be used to evaluate changes after the PSRP has been completed. Downstream estuaries are still at pre-construction conditions. The primary goal of the nekton monitoring is to examine nekton species composition linked to habitat quality in three estuaries: Fakahatchee, Pumpkin, and Faka Union bays (**Figure 7-73**). Faka Union Bay receives unnaturally excessive freshwater volumes in pulsed releases during the rainy season. Pumpkin Bay has diminished flows due to manmade hydrologic barriers resulting from development in the watershed. By comparison, nearby Fakahatchee Bay is relatively unimpacted. The watershed immediately upstream of this bay has been relatively undisturbed other than the construction of Tamiami Trail. Fakahatchee Bay is, thus, representative of an unimpacted estuary in the TTI complex that has been modified primarily by natural evolutionary processes. This estuary serves as a reference or target estuary against which the disturbed Faka Union and Pumpkin bays are compared.

The monitoring has been modeled from the performance measure and target for nekton species identified in the Ecological and Water Quality Monitoring Plan in the project implementation report for PSRP (USACE and SFWMD 2004). The specific target is for the abundance and composition of nekton in Faka Union and Pumpkin bays to be similar (75% by Bray-Curtis) to the composition in Fakahatchee Bay. As with studies in other estuaries with extreme seasonally dependent freshwater inflow (Kanandjembo et al. 2001), the nekton studies found significant differences in species composition correlated with seasonal changes in salinity. From December 2000 to December 2009, staff at the RBNERR conducted bottom trawls to sample nekton populations monthly within Fakahatchee, Faka Union, and Pumpkin bays. Data analyses showed significant differences between Faka Union and Fakahatchee/Pumpkin bays (analysis of similarities [ANOSIM] procedure). Cluster analysis (**Figure 7-78**) indicates that Fakahatchee and Pumpkin bays are similar (above the 75% level); whereas, Faka Union Bay was slightly less similar (73% similarity) compared to the other two bays.

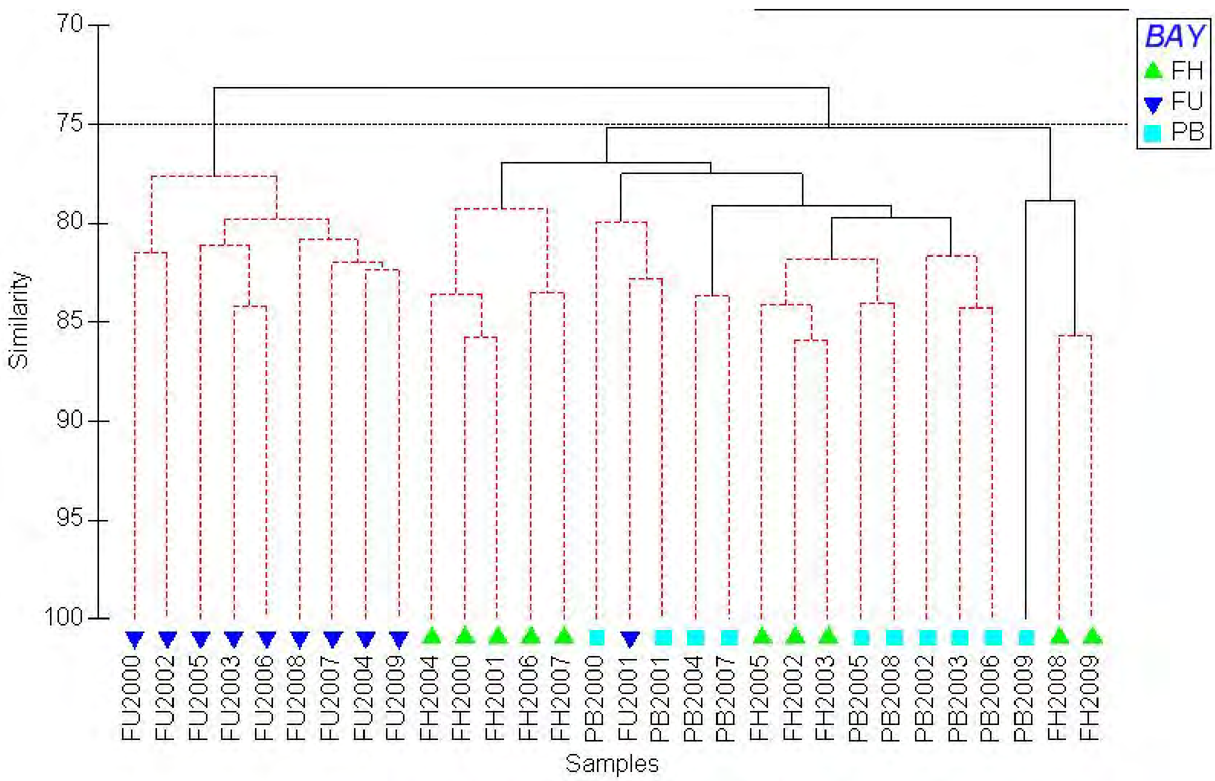


Figure 7-76. Dendrogram of nekton species composition similarity for Fakahatchee Bay (FH), Faka Union Bay (FU), and Pumpkin Bay (PB). The ANOSIM procedure indicates that Fakahatchee Bay species composition was significantly different from the two other estuaries. Prior to analysis, the data were fourth-root transformed in order to down weight dominant species influence and increase influence from rarer species. The dashed line indicates a Bray-Curtis similarity value of 75.

The baseline monitoring suggests that nekton species composition patterns are a sensitive indicator of altered freshwater inflow. Not surprisingly, Faka Union Bay nekton was significantly different from the other two bays. As noted above, Faka Union Bay receives large, unnatural point source discharges from Faka Union Canal that alters salinity in the bay. However, the fact that both Faka Union and Pumpkin bays' similarity is at or near the 75% target suggests that the target may be set too low. Unfortunately, no fish nekton data are available for the area prior to watershed alteration (Shirley et al. 2005), so the scientists involved in setting the target have used best professional judgment. Given the known impact to Pumpkin and Faka Union bays and the proximity of their nekton species composition to the current restoration target, it would be prudent for the PSRP Project Delivery Team to revisit the target.

Upper Southwest Coast and Ten Thousand Islands Oysters

The eastern oyster (*Crassostrea virginica*) is a valued ecosystem indicator for the TTI estuaries. Altered salinity regimes due to anthropogenic-induced changes in freshwater delivery to these estuaries have reduced or eliminated many eastern oyster reef areas and have impacted both the timing and extent of oyster reproduction (Berrigan et al. 1991). The diverse faunal community associated with oyster reefs has been correspondingly impacted. Restoration of more natural freshwater inflows to the TTI estuaries and the corresponding reestablishment of mesohaline and polyhaline salinity resulting from restoration should provide appropriate habitat conditions to restore healthy oyster beds and associated communities.

From June 2012 through May 2013, oyster responses in three estuaries within the TTI—Pumpkin, Faka Union and Fakahatchee bays—were studied concurrently. Live oyster density estimation in all three estuaries was conducted once in fall 2012 and once in spring 2013. This represented the first time that oysters were monitored in all three estuaries simultaneously by either RECOVER or the PSRP. As noted in the Fish Nekton section above estuarine conditions in Faka Union and Pumpkin bays have been significantly altered due to anthropogenic changes in freshwater flows. Fakahatchee Bay is relatively unimpacted by water management practices and serves as the reference or target bay condition. Typically, April through October is considered the wet period while November through March is considered the dry period for southwest Florida. It should be noted that during the study period, the first half of the wet season received little to no rain (until late August 2012) with heavy rainfall September through November. This is in contrast to 2009–2011, which were relatively dry to normal years.

<u>Live Densities</u>

Of the three estuaries studied, oyster reefs are relatively scarce in number and small in aerial extent within Faka Union Bay when compared to the estuaries of Henderson Creek, Blackwater River, Pumpkin Bay and Fakahatchee Bay. In Faka Union Bay, the distribution of reefs and the region of maximum mean live density are displaced seaward relative to the other two estuaries, which is a result of higher freshwater discharges to Faka Union Bay that shifts favorable salinity seaward compared to the other two bays. Faka Union Bay exhibited the highest estuarywide mean live densities in both the wet (2,613 per m^2) and dry (2,197 per m^2) seasons (**Figure 7-79**). Such high density is likely due to the fact that for most of the study period, conditions were relatively dry and the freshwater discharged from the Faka Union Canal resulted in a salinity regime in Faka Union Bay that was more favorable for oysters compared to higher salinity conditions in the other two bays. Unlike previous observations, the outermost region of Faka Union Bay had the highest live density (4,646 per m^2). Faka Union Bay exhibited the greatest productivity further upstream in the wet season, while Fakahatchee Bay had greater density upstream in the dry season (1,451 per m^2), presumably a result of changes in the availability and delivery of fresh water, which is due to operation of the Faka Union Canal.

Of the oysters measured during live density monitoring, Pumpkin Bay oysters obtained the greatest wet season mean length (33 millimeters [mm]); Faka Union Bay had the greatest dry season mean length (32 mm). The greatest mean length for individuals, 38 mm, was recorded at the most upstream station of Faka Union Bay during dry season monitoring.

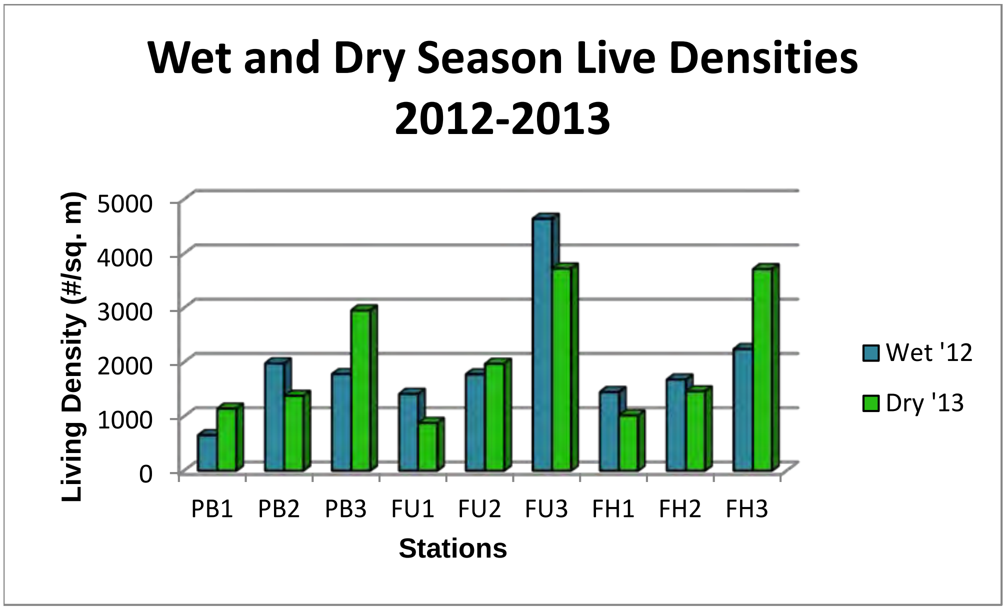


Figure 7-77. Live oyster densities at various sampling locations in Pumpkin (PB), Faka Union (FU) and Fakahatchee (FH) bays during two sampling periods.

Condition Index

Comparison of oyster condition index, a ration of dry meat weight to dry shell weight (Volety 2008), among the oyster reefs along a salinity gradient is used as an indicator of oyster health and the influence of salinity and disease. Pumpkin and Fakahatchee bays experienced nearly identical condition index values (2.421 and 2.424, respectively), while Faka Union Bay had a slightly lower condition index value (2.385) of the three estuaries and exhibited the greatest level of variation (+0.796). Mean annual condition index values above 2 are considered healthy. These values are in line with the median condition index of 2.240, which have been reported previously for oysters on the southwest coast of Florida (Volety 2008, Volety et al. 2009).

In the TTI estuaries, condition index values of oysters were consistently higher at locations experiencing higher salinity. Condition index for oysters in Faka Union decreased significantly from August through October (mean condition index 1.825 + 0.550), compared to the remainder of the monitoring period (2.660 + 0.918). Oysters from the most seaward site in the Faka Union Bay (2.649 + 0.705) had a higher condition index when compared to those in the middle (2.22 + 0.795) and upstream (2.280 + 0.888) sites for most of the monitoring period. Condition indices dropped only slightly during the months of September and October in Pumpkin and Fakahatchee bays, suggesting an adverse impact to the nearshore area of Faka Union Bay, which received higher volumes of freshwater inputs during the wet season via Faka Union Canal.

Spat Recruitment

Mean spat recruitment during the period of study was significantly higher in Pumpkin Bay (11.15 + 2.990) than in Faka Union (7.800 + 1.517) or Fakahatchee (8.720 + 1.876) bays (**Figure 7-80**). There are several factors, which may be contributing to the higher recruitment values in Pumpkin Bay. The plentiful rainfall experienced during the wet season was late in arriving, but was sufficient to affect salinity in both Faka Union and Pumpkin bays. Intense salinity stress during the wet season and a smaller brood stock population may have contributed to the low recruitment rates in Faka Union Bay. Simultaneously, Pumpkin Bay's most seaward station experienced reduced salinity due to hydrologic influence from a connection to Faka Union. This likely presented a more desirable salinity regime in Pumpkin Bay for recruitment and growth. As previously noted, the condition index for oysters at Pumpkin Bay was nearly identical to the reference estuary, which receives freshwater inputs from two rivers. Recruitment was lowest in the upstream end (2.088 + 0.416) of all three study estuaries, and significantly higher at the most marine end of the systems (18.369 + 4.290); again suggesting that salinity plays a significant role in the survival of the recruits. When PSRP is completed, and the hydrology of the upstream watershed improves timing and distribution of freshwater inputs, it is expected that a more normal salinity regime will occur in Faka Union and Pumpkin estuaries leading to improved recruitment further upstream rather than at the most marine location of the estuary where higher salinities allow for higher predation pressure and salinity-related stress for recently settled oyster larvae and juveniles.

Disease

In reference to disease, the significant role that salinity and freshwater inputs play in the survival and health of oysters in the TTI was patently apparent in Faka Union and Pumpkin bays. Mean *Perkinsus marinus* (dermo) infection intensities were low for oysters within all TTI estuaries. Oyster disease was more prevalent in Fakahatchee Bay, an estuary that experiences consistently higher salinity than Faka Union and Pumpkin bays.

Perkinsus marinus (a parasite) thrives under high temperature/high salinity conditions. Florida oysters experience climatic circumstances that fortunately create conditions that limit the intensity and prevalence of *Perkinsus marinus*. In this case, large freshwater discharges from the manmade system upstream depressed salinity in all of Faka Union Bay and the lower reaches of Pumpkin Bay, and limited the prevalence and intensity of *Perkinsus marinus*, while simultaneously creating abnormal and undesirable salinity regimes in both estuaries. The two upstream monitoring stations in Pumpkin Bay can experience higher salinity similar to Fakahatchee Bay. A reverse salinity gradient was observed in Pumpkin Bay numerous times due to freshwater influence from Faka Union Canal discharge at the most seaward station in Pumpkin Bay. This most seaward station in Pumpkin Bay regularly experienced significantly lower salinity conditions than the station immediately upstream particularly during the wet season (**Figure 7-81**), and while it had the lowest rate of prevalence in Pumpkin Bay, it was near the mean for the level of intensity of *Perkinsus marinus.*

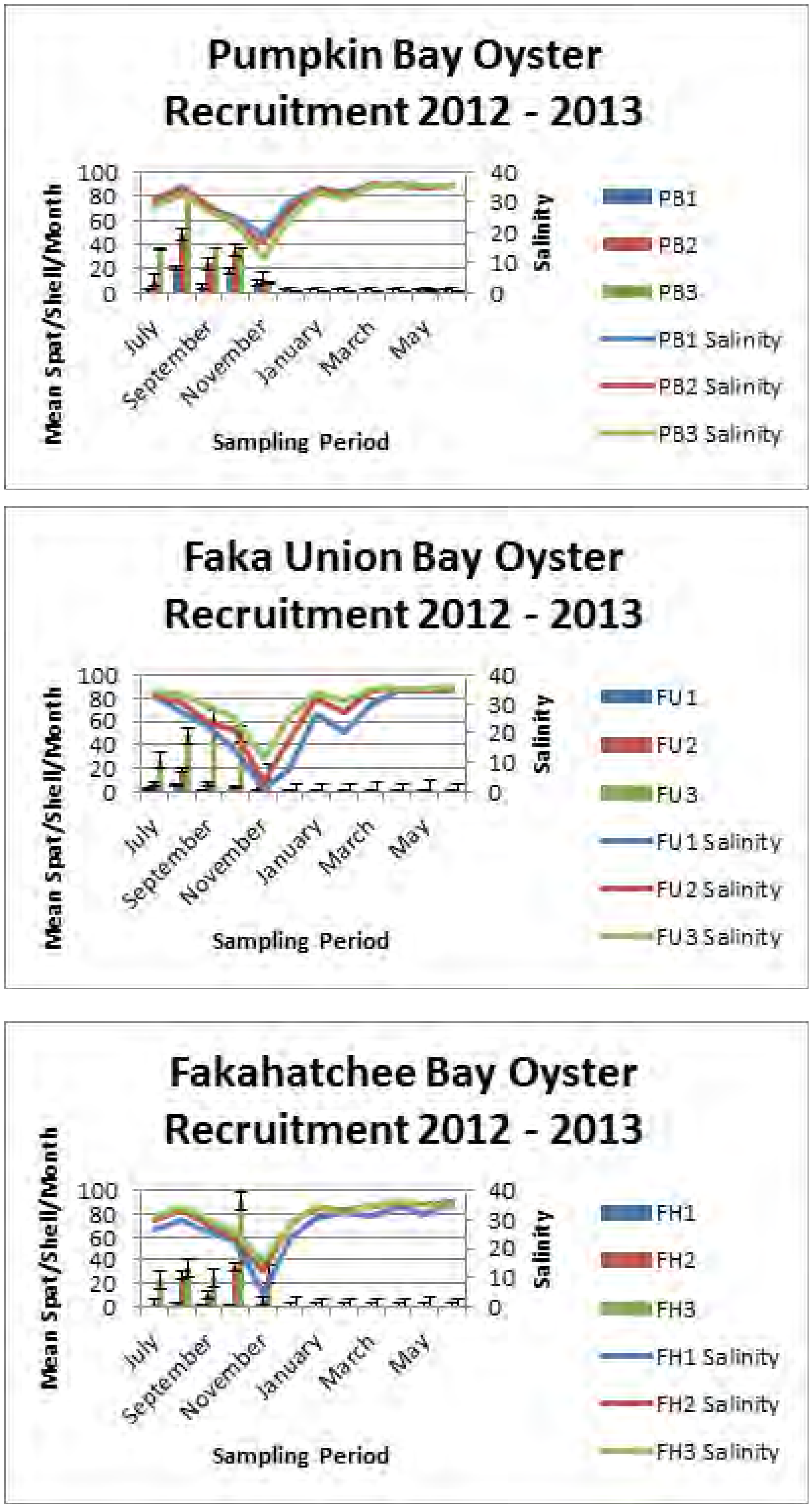


Figure 7-78. Mean oyster spat recruitment per month at various sampling locations in Pumpkin (PB), Faka Union (FU) and Fakahatchee (FH) bays over the sampling period.

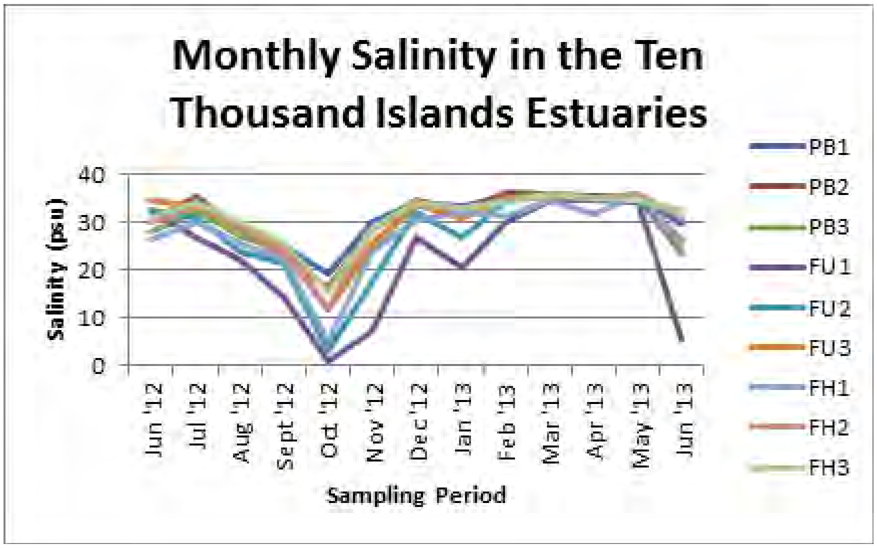


Figure 7-79. Monthly salinity at various sampling locations along the salinity gradient in Pumpkin (PB), Faka-Union (FU) and Fakahatchee (FH) bays during the sampling period.

Summary

This study illustrates that oysters are an important indicator species for restoration of the SCS ecosystem. Oysters are a critical component of the ecosystem, building up reefs that form habitat for a number of other organisms; they respond quickly to environmental changes, particularly salinity fluctuations, and their responses are easy to measure. In this case, the brood stock was present and the substrate for settlement was available along the salinity gradient. When salinity regimes changed to more favorable conditions, the live density of oysters and recruitment metrics improved. This work can assist in devising or refining restoration plans, which may offer significant benefits via infrastructure or operational components that provide more desirable salinity regimes. In many cases, restoration plans do not anticipate that the first element to respond to restoration will be the downstream estuaries. This study also illustrates that due to sensitivity to changes and rapid response, restoration changes may be detected by this indicator sooner than other ecological indicators. It should be cautioned that results presented herein are only from one year and represent oyster metrics given the environmental conditions for that year. Continued monitoring will give us information of natural changes while establishing more robust baseline values from which restoration impacts can be measured over time.

Upper Southwest Coast and Ten Thousand Islands American Crocodiles

American crocodiles were first reported in southwestern Florida in the 1940s (Kushlan and Mazzotti 1989). The occurrence of those crocodiles was attributed to storm displacement or human transport. Crocodiles continue to be sighted (see FWC nuisance crocodile database, RBNERR, and United States Fish and Wildlife Service files) in southwestern Florida today in locations likely to be affected by Everglades restoration projects including regulated freshwater discharges down the Caloosahatchee River from Lake Okeechobee and the 86-square mile PSRP, which will restore seasonal sheet flow to downstream estuaries and eliminate a point-discharge of fresh water to Faka Union Bay. Affected areas include estuaries within RBNERR, TTI National Wildlife Refuge, Collier-Seminole State Park, Fakahatchee Strand Preserve State Park and ENP in Collier and Monroe counties).

Crocodiles have also deposited clutches of eggs at the Marco Airport in Collier County since the 1990s, none of which have hatched successfully. Eggs collected and analyzed in 2004 were infertile, explaining why they failed. This pattern of presence of crocodiles in the area, but no successful reproduction, is similar to that found around Flamingo and Cape Sable prior to plugging canals in the 1980s (see **t**he Florida Bay and the Lower Southwest Coast Crocodilians section earlier in this document). Monitoring consistent with what is conducted in other parts of the SCS is recommended for the TTI and upper southwest coastal areas to determine the crocodilian response to ecosystem restoration (increased freshwater flow and lower salinity).

CERP PROJECT ASSESSMENTS

This section provides assessments for the three CERP projects currently being constructed and operated in the SCS region—BBCW Project, C-111 SCWP, and PSRP (**Figure 7-1**). These assessments include projects-level monitoring data complemented with RECOVER MAP monitoring data. Generally, it is early in the life of each of these projects, but some restoration benefits are being realized.

Biscayne Bay Coastal Wetlands Project

The CERP BBCW Project Phase 1 is composed of three components: Deering Estate Flow-way, Cutler Wetlands Flow-way, and L-31E Flow-way (**Figure 7-82**). In advance of congressional authorization and appropriations, SFWMD constructed the Deering Estates Flow-way and a portion of the L-31E Flow-way (the L-31E Culverts). Monitoring is currently taking place for these two components.

The Deering Estate Flow-way redistributes excess freshwater runoff, directing it away from existing canal discharges and spreading it out as sheet flow prior to discharging into Biscayne Bay. SFWMD completed construction in April 2012. The project became operational in November 2012.

The L-31E Flow-way is designed to reestablish historical sheet flow and wetland hydroperiods (the number of days per year water stands at or above the ground surface) downstream of the project area to the extent possible. This will be accomplished by connecting L-31E to the wetlands east of L-31E using a series of culverts. This component may also provide the additional benefit of mitigating impacts of discharging fresh water via the existing canals. Construction for the culverts began on January 11, 2010 and was completed on June 10, 2010.

The Cutler Wetlands Flow-way has yet to be constructed. It includes a pump station on the C-1 Canal, a lined conveyance canal, a spreader canal system, and box culverts under roadways, as well as the plugging of mosquito control ditches. The pump station will deliver water to the spreader canal located in the saltwater wetlands via a lined conveyance canal. The Cutler Wetlands Flow-way final design was completed in November 2009. Funding for initiating SFWMD expediated components of Cutler Flow-way is subjected to SFWMD Governing Board approval.

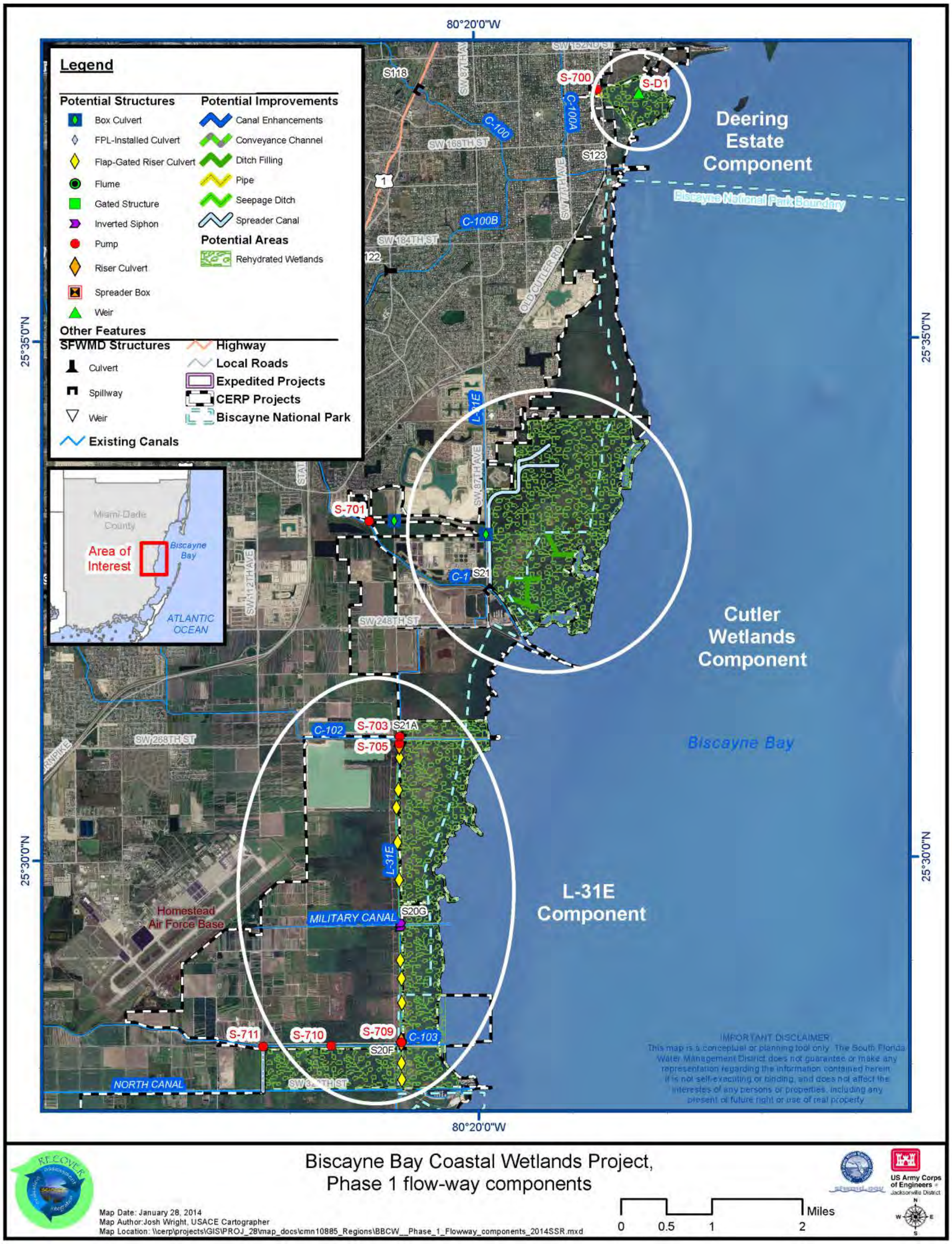


Figure 7-80. BBCW Project Phase 1 flow-way components.

Deering Estate Flow-way Assessments

The following section provides assessment of Deering Estate Flow-way monitoring data. For more detailed information on all BBCW Project monitoring results please see Charkhian (2014).

<u>*Stage Effects from Deering Estate (S-700) Pumping*</u>

Figure 7-83 shows the project features of the Deering Estate Flow-way, including relevant monitoring sites. Parameters monitored include stage, flow, wetland inundation, water quality, nearshore salinity, and vegetation. From the beginning of operation on December 17, 2012 through August 7, 2013, the S-700 Deering Estate pump station (S-700) has diverted approximately 12,867 acre-feet (ac-ft) of fresh water from the C-100 Canal to historic remnant wetlands near Cutler Creek east of Old Cutler Road, which represents about 50% of the available fresh water that could have been diverted from the C-100A Canal. The remaining 50% was diverted through the S-123 Spillway to Biscayne Bay, almost all of which was released from May through July. **Figure 7-84** shows stage conditions during pump operations. There are several clear examples when stage increases as pumping is initiated or increased (February 15, April 6, and June 10). However, there are other times when stage fluctuated significantly when there was no pumping (end of May), which is likely due to local rainfall events (**Figures 7-84** and **7-85**). The rainfall monitoring station at S-123 recorded 20.45 inches from December 2012 through June 2013 (**Figure 7-85**).

Figure 7-86 indicates that surface water levels at the two staff gages in Cutler Slough (Gage 1 upstream of the weir and Gage 3 downstream of the weir; **Figure 7-83**) respond quickly and appreciably to flow diversions from S-700. For example, water levels at both staff gages were higher in January and the end of February after the pumps had run continuously for several weeks; following a three week period of no pumping in March, stages noticeably decrease. Since operation of S-700 began, surface water levels within the slough show an increasing trend through the period of record ending in late June 2013. Groundwater also noticeably rose during the initial test period, and water levels varied according to pump operation (not shown). Not surprisingly, the groundwater response became progressively muted as distance from the diversion point (S-700) increased. Monitoring Station 2 is located about 360 feet downstream of pump station S-700 discharge. Statistical analyses indicate high correlations between groundwater levels and S-700 flow (**Table 7-10**). Analyses also revealed low correlations between groundwater levels and rainfall, between groundwater salinity and S-700 flow, and between groundwater salinity and rainfall.

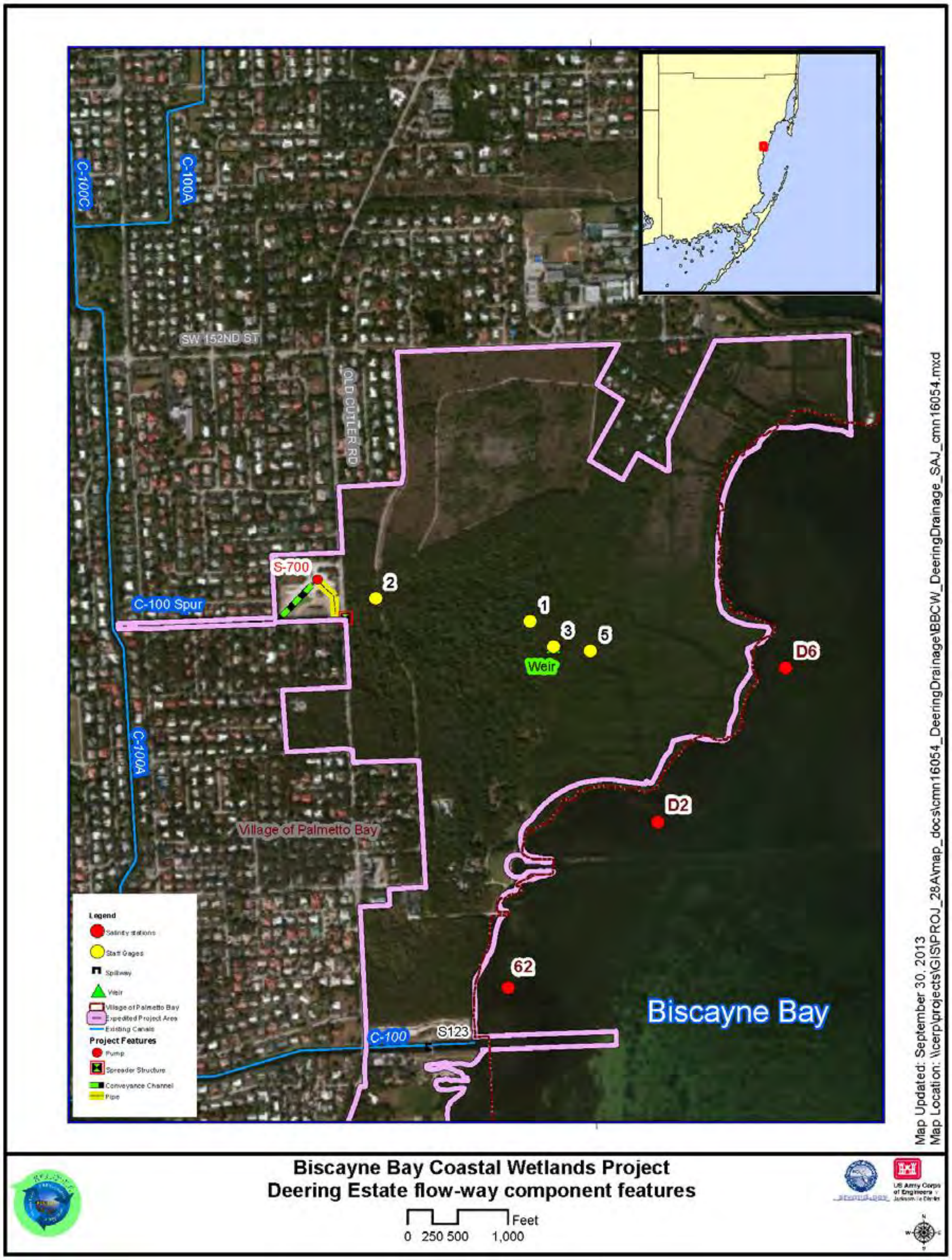


Figure 7-81. Deering Estate Flow-way showing project features, including pump station (S-700) and weir.

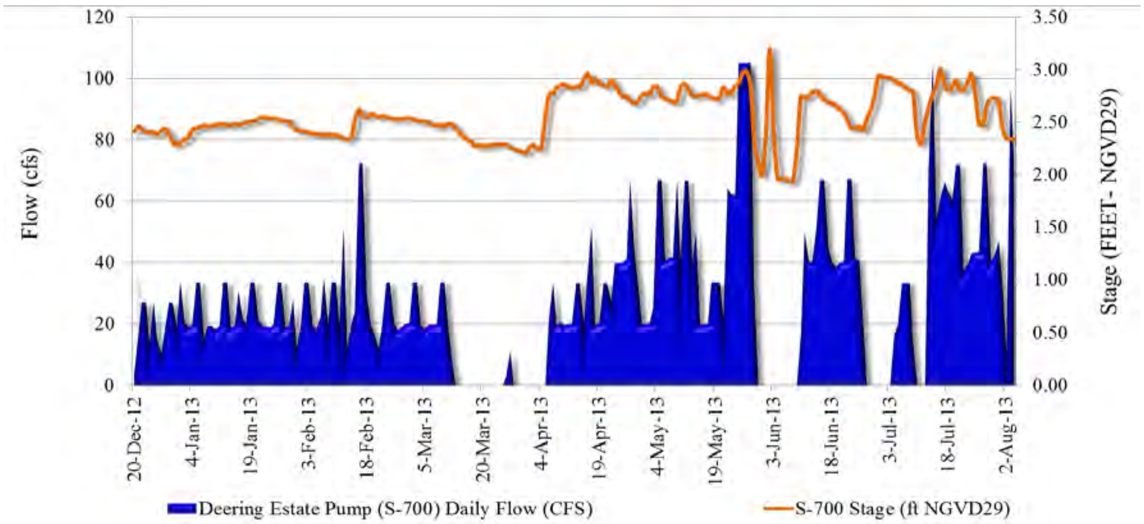


Figure 7-82. Time series of flow (cubic feet per second) and stage (National Geodetic Vertical Datum of 1929 [NGVD29] vertical datum) for pump station S-700.

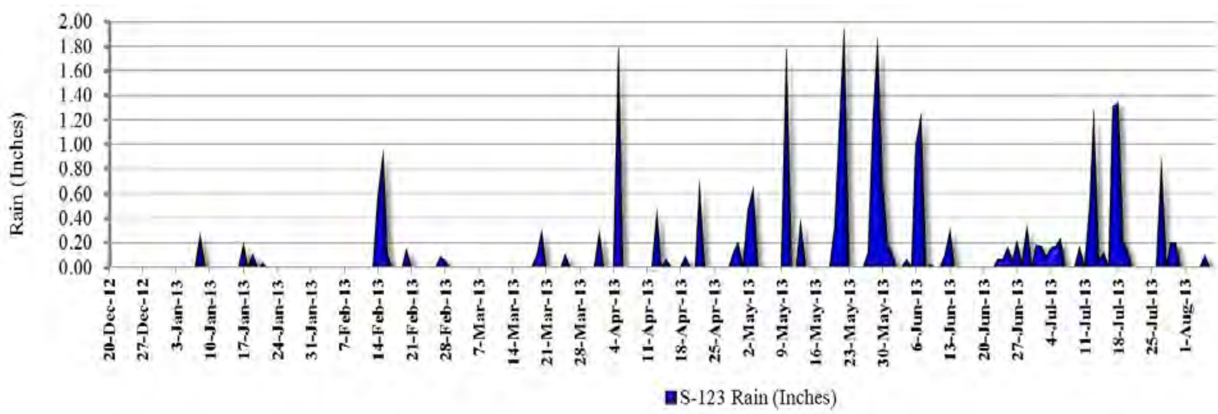


Figure 7-83. Rainfall at Station S-123 on the C-100 Canal.

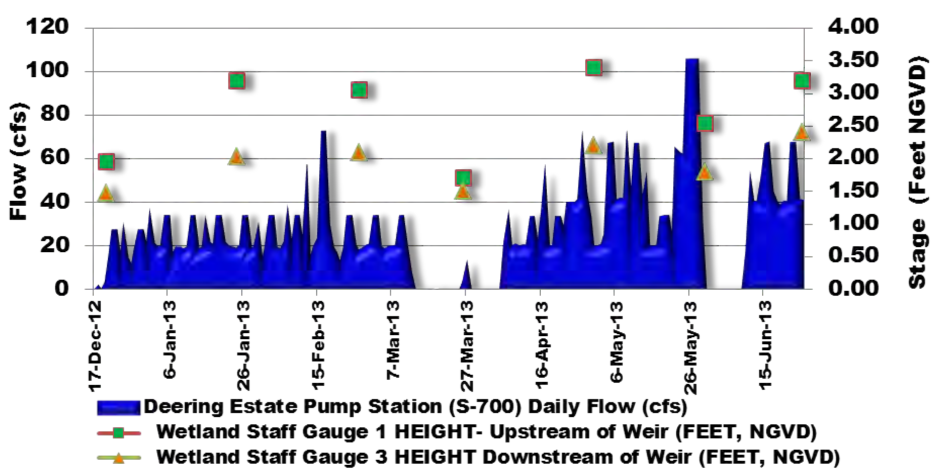


Figure 7-84. Comparison of water levels in feet National Geodetic Vertical Datum of 1929 (NGVD) at Deering Estate staff gages 1 and 3 versus pump station S-700 daily flow.

Table 7-10. Pearson correlation matrix for water levels at wells 2 and 5, flow at S-700, rainfall at S-123, and salinity in groundwater well 5. (Sample size was 187.)

Station	Water Level Well 2	Water Level Well 5	Flow at S-700	Rainfall at S-123	Salinity Concentration in Groundwater Well 5 (Downstream of Weir)
Water Level Well 2	1				
Water Level Well 5	0.841	1			
Flow at S-700	0.836	0.775	1		
Rainfall at S-123	-0.044	0.018	0.044	1	
Salinity Concentration in Ground Water Well 5 (Downstream of Weir)	-0.072	-0.431	-0.243	-0.259	1

Wetland Inundation

Between January and February 2013, the spatial extent of wetland inundation within the slough under various pumping rates was determined (**Table 7-11** and **Figure 7-87**). Results indicate that approximately 19 acres of wetland are inundated under a pumping rate of 25 cubic feet per second (cfs), which is about 58% of the historic slough area. Under a 100-cfs rate, 31 acres of wetland were inundated—approximately 94% of the historic slough area. Typically, these wetlands are dry during this time of year because they have been recharged only by rainfall since the C-100 Canal was built.

Table 7-11. Estimated acreage of impounded surface water under different pumping/flow rates within Deering Estate. The table includes the percentage of the historic remnant slough inundated by various pumping rates.

Pumping Rate	Duration of Testing	Estimated Acres of Impounded Surface Water	Percentage of Inundated Historic Remnant Wetlands within Cutler Creek during 5 Hours of Testing
0	5	0	0%
25	5	19	58%
50	5	25	76%
75	5	27	82%
100	5	31	94%

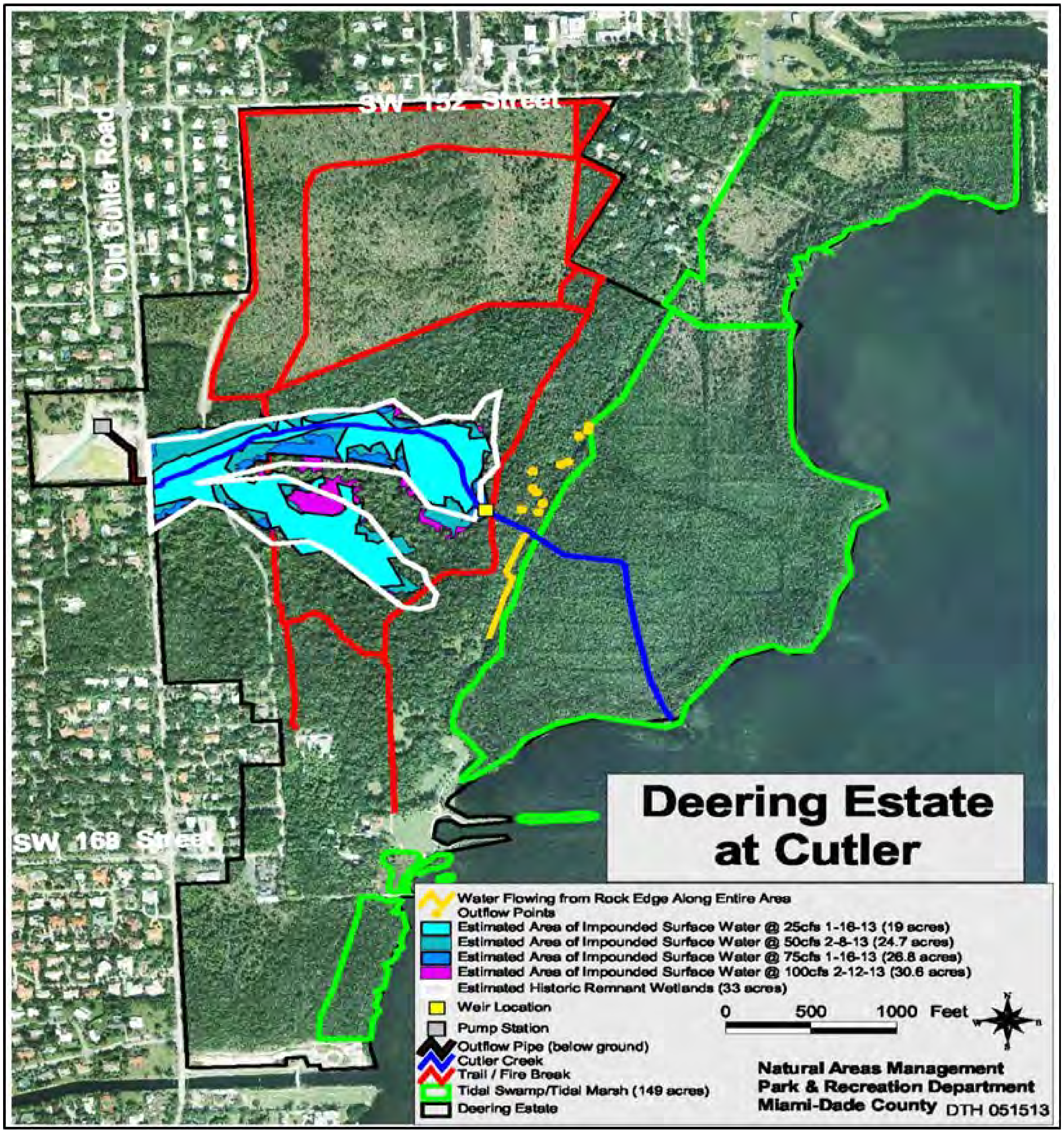


Figure 7-85. Delineation of the historical freshwater wetland slough in Deering Estate and areas of inundation at different pump rates.

Water Quality

Water quality sampling at S-700 began on May 24, 2012 in accordance with Specific Conditions of FDEP Permit Number 0271729-002 for the Deering Estate. Water quality was sampled on a biweekly basis when pumps were operated (parameters included ammonia, total phosphorous, nitrate-nitrite, total Kjeldahl nitrogen and physiochemical parameters included specific conductance, dissolved oxygen, pH, and temperature). In order to establish the state of the system for WY 2013, Class III criteria standards for fresh waters were monitored. Excursions of Class III freshwater criteria occurred for two of the analyzed parameters: dissolved oxygen and specific conductivity. In south Florida, due to the high water temperatures, the amount of dissolved oxygen tends to be low in many freshwater bodies. Such is the case for measurements taken at pump station S-700 where, during WY 2013, all measured water temperatures are above 22° Celsius and 20 percent of dissolved oxygen measurements are under 5 mg/L. During WY 2013, specific conductivity exceeded the accepted freshwater criteria one time, on November 27, 2012, during the dry season when a long period of no flow occurred. No water quality samples were taken before May 24, 2012 at pump station S-700 and therefore, at this time, no comparisons with previous water years can be performed.

Table 7-12 summarizes monthly TP and TN loads discharged from pump station S-700 during WY 2013. Due to the short period of record, any estimations of nutrient loads temporal trends may be premature. The variability of the data could be caused by seasonal changes rather than project related effects. A minimum of two years of monitoring data are required for a rigorous seasonal Kendall tau test.

Table 7-12. Summary of estimated monthly nutrient load for pump station S-700. [a]

Month	Total Monthly Flow (cfs)	Total Monthly Flow (ac-ft)	Total Monthly NO_X Load (kg)	Total Monthly NH_4 Load (kg)	Total Monthly TKN Load (kg)	Total Monthly TN Load (kg) [b]	Total Monthly TORGN Load (kg) [c]	Total Monthly TP Load (kg)
December 2012	230.85	457.88	95.30 [d]	16.31 [d]	143.19 [d]	238.49 [d]	126.88 [d]	7.39 [d]
January 2013	743.70	1,475.13	287.37	144.78	970.13	1,257.50	825.35	26.78
February 2013	782.48	1,552.04	404.54	122.01	1,081.22	1,485.76	959.21	29.21
March 2013	295.28	585.68	140.04	43.77	423.15	563.18	379.37	10.20
April 2013	789.82	1,566.61	249.96	56.12	1,150.36	1,400.32	1,094.24	27.05
May 2013	1,478.27	2,932.15	240.26	41.15	1,714.23	1,954.48	1,673.07	55.52
June 2013	776.00	1,541.17	228.12	9.50	779.40	1,007.52	769.90	43.72
July 2013	136.48	270.70	40.07	1.67	136.90	176.97	135.23	7.68 [e]
Total	**5,233.87**	**10,381.37**	**1,687.00**	**435.00**	**6,398.60**	**8,084.22**	**5,963.25**	**207.65**

a. Key: cfs – cubic feet per second, kg – kilograms, NH_4 – ammonium, NOx – nitrogen oxide, TKN – total Kjeldahl nitrogen, TN – total nitrogen, TORGN – total organic nitrogen, and TP – total phosphorus.

b. TN load = TKN load + NO_x load.

c. TORGN load = TKN load + NH_4 load.

d. Month is incomplete; flow data start on December 2012.

e. Month is incomplete; flow data ends on July 2013.

Salinity

Continuous salinity monitoring in the coastal region near Deering Estate, along the shore of Biscayne Bay north of the S-123 Spillway, has been conducted at site 62 since 2004 and was augmented with two additional nearshore stations, sites D2 and D6 in fall 2010 (**Figure 7-83**). Bottom salinity is collected at 15-minute intervals at these stations located within 500 meters of the shore.

Salinity along the shore in the Deering Estate region was statistically within the range of the longer-term time series. No discernible trend was observed in salinity in recent years even though there were periods of higher salinity, such as from the 2011 dry season into the 2012 wet season, and slightly lower periods in WY 2013. These year-to-year variations are a result of variations in precipitation in the region but may also have water management related influences that will require more detailed analysis to separate out. The analysis is limited to a brief statistical summary of conditions at Stations 62, D2, and D6.

For WY2013, dry season salinity was 27.43 ± 3.84 while wet season salinity was 20.98 ± 5.19 . S-700 operations had begun during the WY 2013 dry season but occurred during time periods where additional water was also being moved from Water Conservation Area 3A toward the east coast. Dry season salinity was similar to the 26.58 ± 4.38 observed in WY 2012 and within the range of salinity observed in this region over the period of record for continuous salinity observations. The wet season values were appreciably lower than the 33.64 ± 5.80 observed in the WY 2012 wet season, likely due to differences in precipitation.

Surface water salinity responded to the pumped inputs of fresh water from pump station S-700 into the historic remnant wetlands within the vicinity of Cutler Creek (**Figure 7-88**). Salinity at both wetland staff gages 1 and 3 decreased to less than 1 approximately six weeks after pumping had been initiated. Salinity remained at or near zero through most of the POR after late January. The only exception was an increase to about 8 at staff gage 3 in late March, which is likely in response to the cessation of S-700 pumping during that time. It is noteworthy that salinity remained at or near zero in the latter part of May when no pumping was occuring. This is likely due to the exceptionally high rainfall conditions experienced during this time.

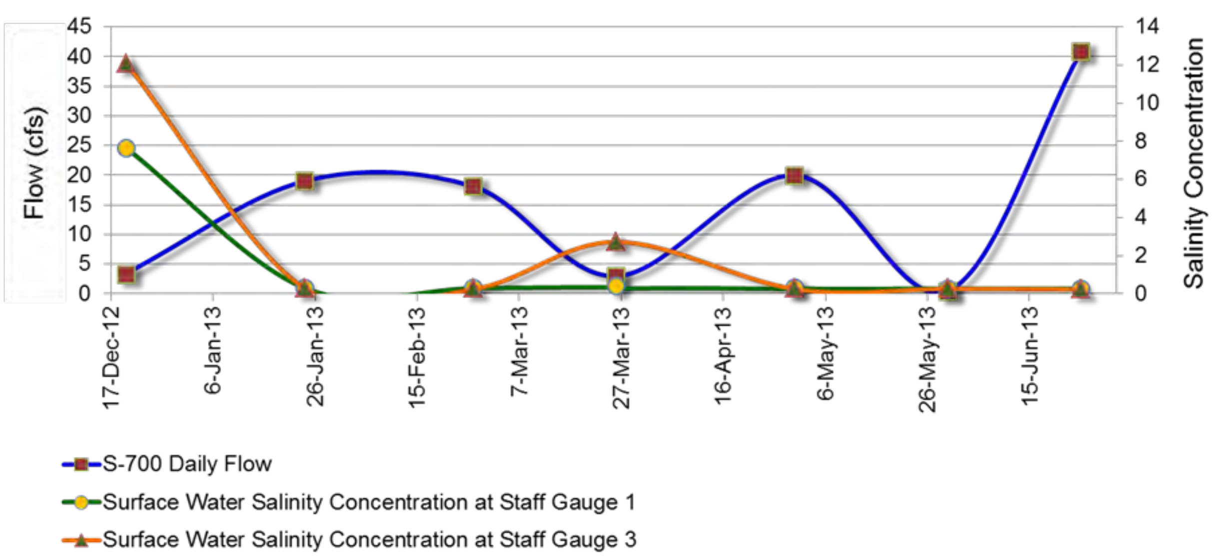


Figure 7-86. Salinity at wetland stage gages 1 and 3 together with daily flow at S-700.

Figure 7-89 shows the salinity near the mouth of Cutler Creek compared to pumping at S-700 and local rainfall from April 25 through early August 2013. The plot indicates that salinity is responding to pumping and rainfall. There are instances of decreases in salinity when pumping was occurring during a time period of no rainfall (i.e., end of first week of May). Also, salinity remains relatively low (around 12) during mid-June when there was no precipitation and pumping was continuous and between 40 and 65 cfs. However, there are other time periods when salinity appears to respond to rainfall more than pumping, such as the sudden decrease in salinity at the end of May after a significant rainfall event even though a prolonged, significant S-700 flow had stopped. There are other instances in the plot that show salinity responding more to rainfall than pumping (i.e., May 10).

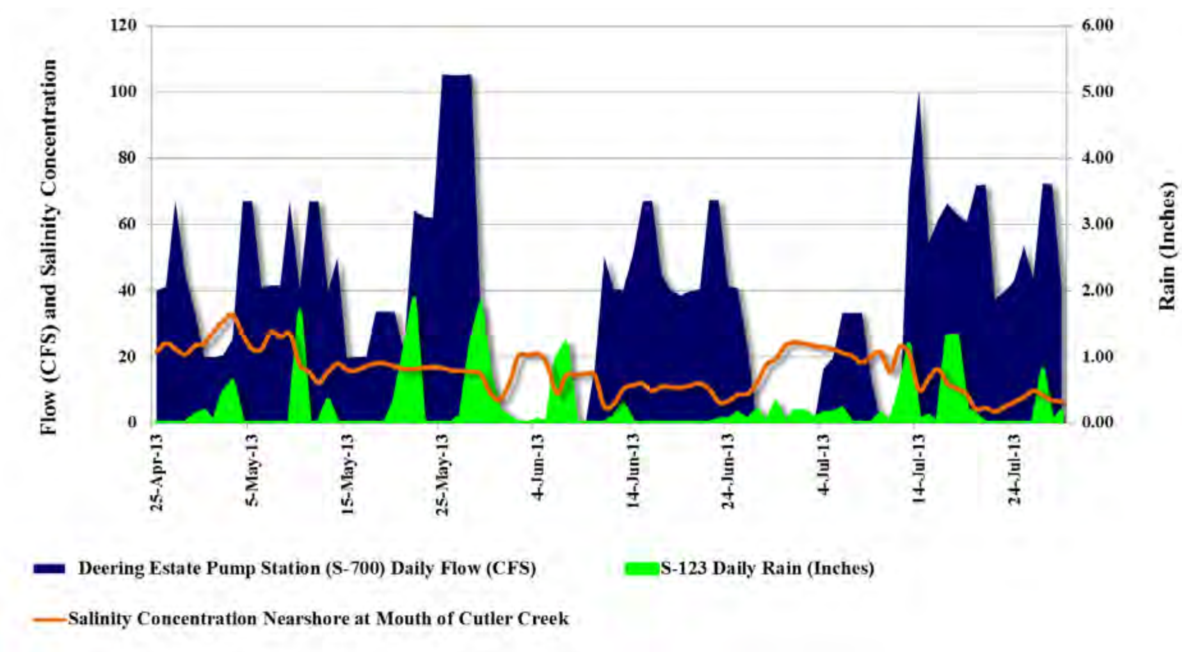


Figure 7-87. Comparison of surface water salinity concentrations nearshore at the mouth of Cutler Creek (YSI unit deployed on April 23, 2013) versus pump station S-700 daily flow and daily rain at the S-123 Rain Station.

Groundwater salinity may also have shown a response to the inputs of fresh water from pump station S-700 into the historic remnant wetlands (**Figure 7-90**). Generally, salinity decreased steadily throughout the dry season after S-700 came online and there was an abrupt decrease in salinity in response to large pumping rates in late May. However, it should be noted that salinity continued to gradually decrease through most of March, when there was no pumping from S-700. Also, salinity remained relatively constant during the continuous pumping from early April through mid-May. So, groundwater salinity exhibits a variable response to S-700 pumping. Statistical analysis also indicates a correlation between salinity concentration in groundwater well 5 downstream of the weir, surface water salinity concentration at the monitoring station in mouth of Culter Creek nearshore, rainfall at S-123, and S-700 flow.

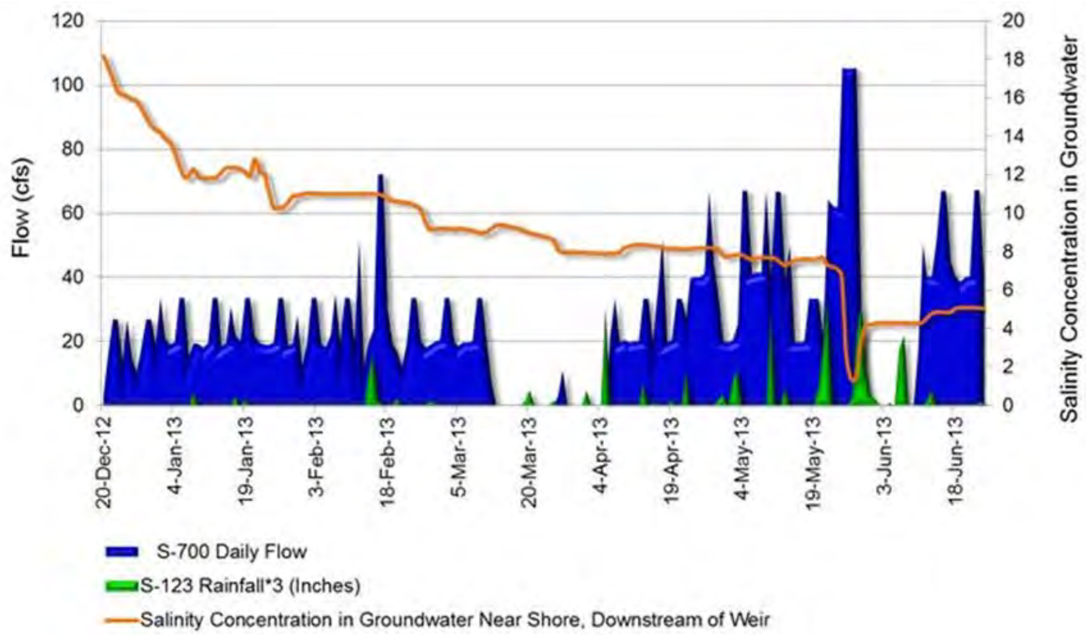


Figure 7-88. Comparison of salinity concentration in groundwater within the vicinity of historic remnant wetlands of Deering Estate versus pump station S-700 daily flow.

<u>*Vegetation Changes within Historic Remnant Cutler Creek*</u>

Changes have occurred in the vegetation communities within the historic remnant wetlands in the vicinity of Cutler Creek in Deering Estate. Upland trees that had grown into the historic wetlands over the past several decades are beginning to die back (**Figure 7-91**). Now that the rehydration project is bringing increased water levels back to the historic wetlands, many of these trees should be displaced by wetland species. The project team is in the process of establishing fixed photographic stations to document changes in vegetation communities within the vicinity of the historic remnant wetlands. Also, Fairchild Tropical Botanic Garden has established baseline vegetation in the slough from 29 randomly placed, 50-meter transects centered on Culter Creek. Monitoring to detect vegetative changes in the historical remnant Cutler Creek and associated wetlands may begin in 2015 and occur every three years. During this reporting period, chemical and mechanical treatment of exotic plant species was conducted within approximately 33 acres of the historic remnant freshwater wetland in Deering Estate.

Figure 7-89. Changes in the vegetation communities and the beginnings of die-back of upland trees in the historic remnant wetlands within the vicinity of Cutler Creek in Deering Estate in July 2013.

L-31E Flow-way Assessments

Figure 7-92 shows the L-31E Flow-way Component area of the BBCW Project. Water level and flow, water quality, nearshore salinity, and vegetation are all monitored relative to the culverts in accordance with Special Conditions 9 through 12 of USACE Permit Number SAJ-2007-1994-1327(IP-TKW) for the L-31E Flow-way and Specific Conditions of FDEP Permit Number 0271729-002 for the L-31E Culverts. For more detailed information on any of these monitoring results please see Charkhian (2014).

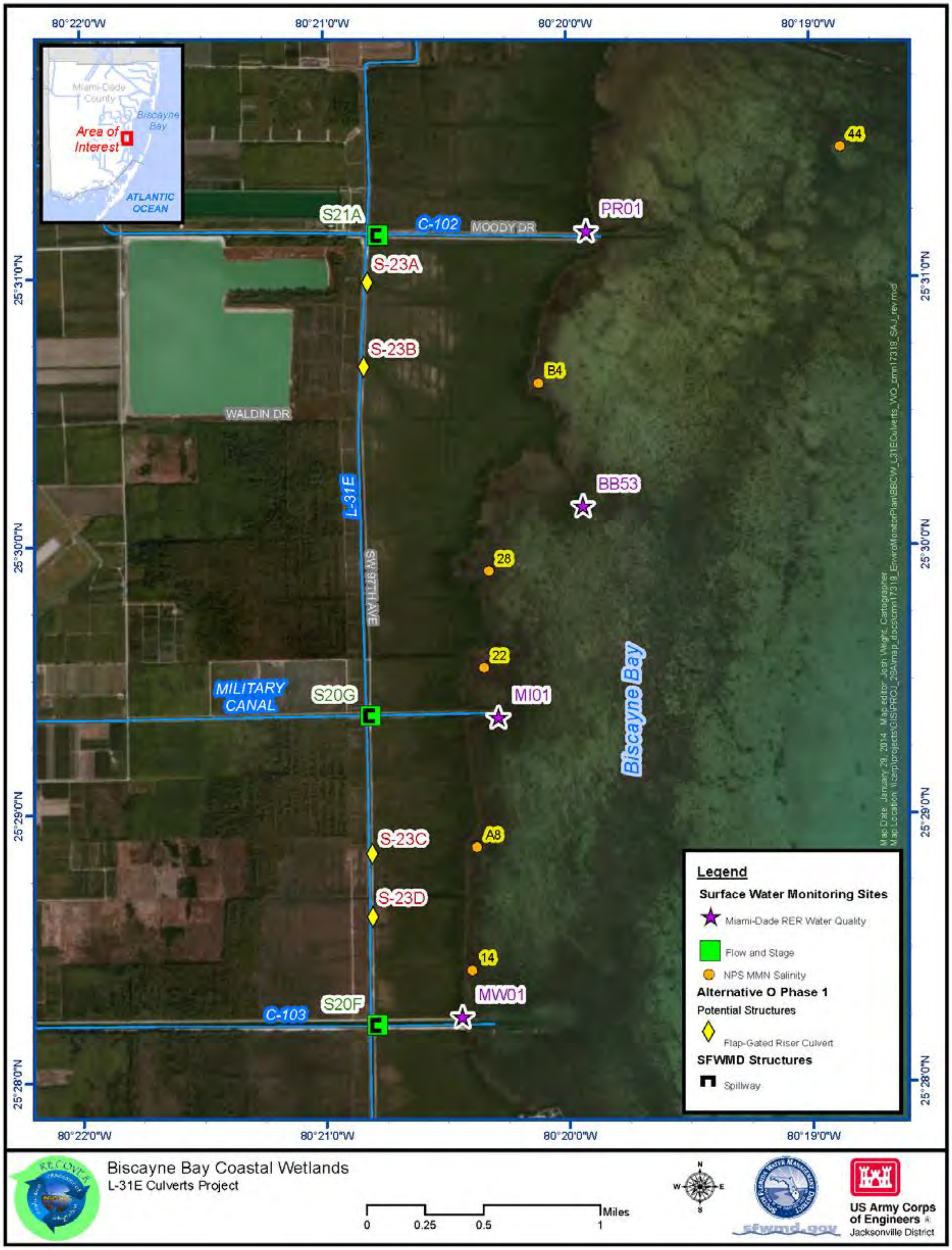


Figure 7-90. Location of L-31E culverts and related flow, stage and water quality monitoring stations.

Water Level and Flow

The L-31E culverts are passive structures intended to redirect flow from the L-31E Canal and provide hydrologic enhancement to the adjacent BBCW Project. Beginning in August 2010, water levels were recorded on a monthly basis within 50 m downstream of the S-23A and S-23D culverts. These water level sites are located along the vegetation sampling transects associated with the S-23A and S-23D culverts. Daily flow rates for culverts S-23A and S-23B were calculated based on mean daily stages at structure S-21A and design rating curve, while daily flow rates for culverts S-23C and S-23D were calculated based on mean daily stages at structure S-20F and the design rating curve (see Charkhian 2014 for more information). During the first annual reporting period, south Florida experienced extreme drought conditions. Beginning in October 2010, water levels along the transect downstream of culvert S-23A fell below the minimum level required for sampling and remained at or below this level through June 2011. From December 2010 through June 2011, November 2011 through December 2012, and November 2012 through March 2013, water levels along the transect downstream of culvert S-23D fell below the minimum level required for sampling. Once water began to flow in July 2011 through September 2011, and May 2012 through October 2012, inundations were observed along the entire length of the transects with water depths approximately 12 to 24 inches along the transect downstream of culvert S-23D and 6 to 12 inches along the transect downstream of culvert S-23A.

The target for diversion by the L-31E Culverts is 4% of the total available water for diversion (as determined by the flow through the S-21A and S-20F structures) in accordance with Special Conditions 10-C of USACE Permit Number SAJ-2007-1994-1327(IP-TKW). The project was able to divert 3 percent (5,110 ac-ft) of the total available water in WY 2012 and 2% (4,705 ac-ft) of the total water available in WY 2013 (**Figure 7-93**). The project was able to meet or exceed its 4 percent diversion target during 6 months of the last two water years. While the "4% of available water diverted through the culverts" is a regulatory-based target, the BBCW Project goals for this element discuss a volume (in ac-ft) of water needed by the system. Thus, utilization of the volume of water diverted may provide a more direct restoration-based target than the percentage of available water.

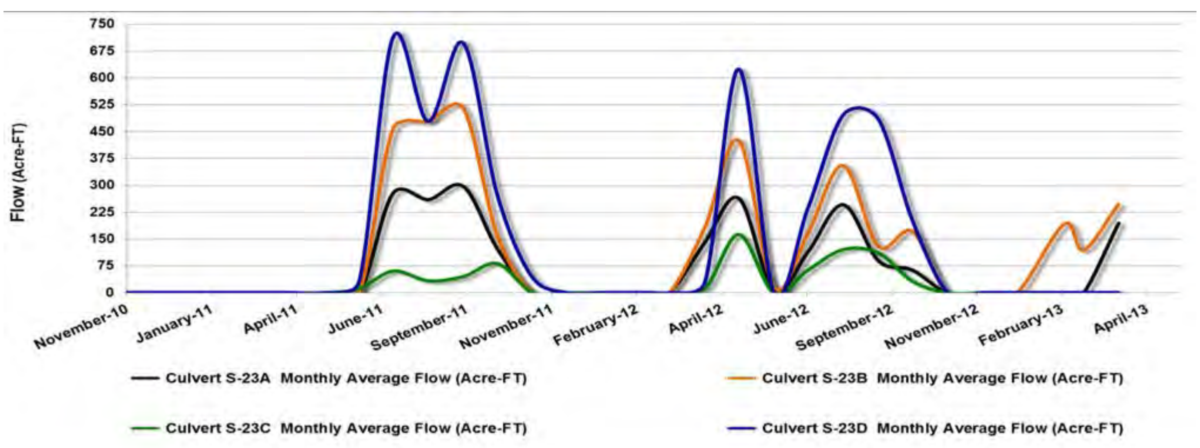


Figure 7-91. Comparison of fresh water in acre-feet (Acre-FT) diverted from the L-31E Canal through the L-31E Flow-way to wetlands east of the L-31E Levee.

Figure 7-94 and **Table 7-13** provide the monthly and daily average flow of available fresh water that would have been diverted through the L-31E Culverts if stage in the L-31E Canal was held at 2.2 feet National Geodetic Vertical Datum of 1929 (NGVD) versus estimated volumes actually diverted. The estimated total culvert flows that would have occurred at a constant 2.2-foot stage in L-31E were 19,450 ac-ft for WY2012 and 26,467 ac-ft for WY2013 whenever fresh water for diversion was available. The estimated actual flow through the culverts for WY2012 and WY2013 was 26% and 18% of these larger diversions, respectively. These results suggest that minor operational changes to the control structures S-21A and S-20F (a reduction of gate openings) could substantially increase the volume of available fresh water for diversion to the wetlands east of the L-31E levee through the L-31E culverts. Therefore, it appears reasonable that diversion targets could be attained with greater frequency by maintaining the L-31E Canal stage level greater than the current 2.12 feet NGVD.

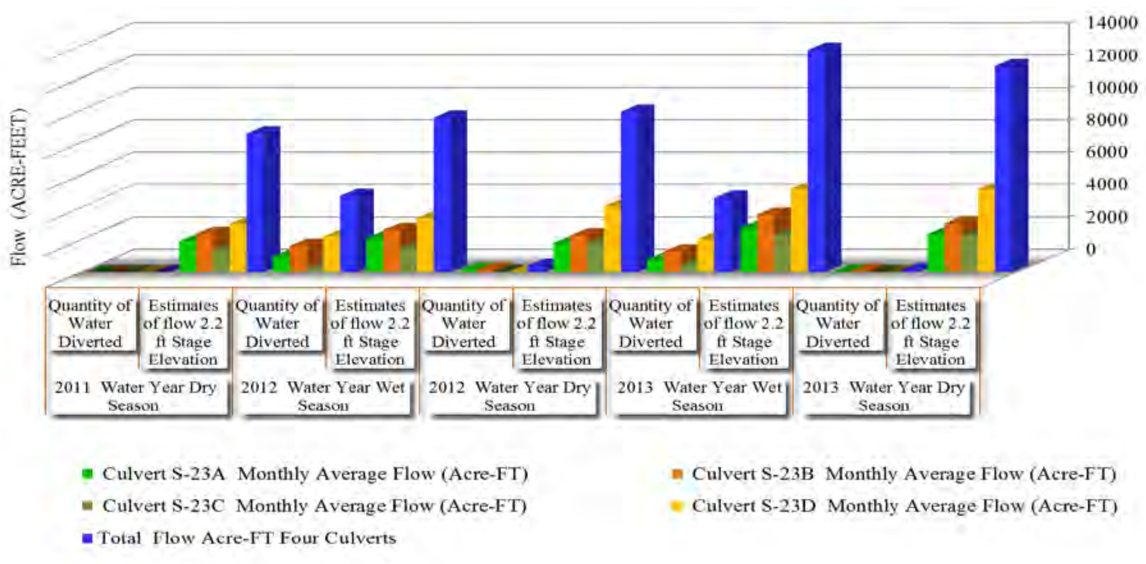


Figure 7-92. Monthly diversions versus estimates of flow in acre-feet (Acre-FT) through the L31E Culverts if stage in the L-31E Canal had been higher (up to top of bank capacity, which is 2.2 feet NGVD).

Table 7-13. Daily flow delivered and estimated based on high water stage, which is 2.2 feet NGVD.

	Monthly Average Flow (ac-ft)									
	WY 2011 Dry Season		WY 2012 Year Wet Season		WY 2012 Dry Season		WY 2013 Wet Season		WY 2013 Dry Season	
Culvert [a]	Quantity of Water Diverted [b]	Estimates of flow 2.2 feet NGVD Stage Elevation [c]	Quantity of Water Diverted [b]	Estimates of flow 2.2 feet NGVD Stage Elevation [c]	Quantity of Water Diverted [b]	Estimates of flow 2.2 feet NGVD Stage Elevation [c]	Quantity of Water Diverted [b]	Estimates of flow 2.2 feet NGVD Stage Elevation [c]	Quantity of Water Diverted [b]	Estimates of flow 2.2 feet NGVD Stage Elevation [c]
S-23A	0.00	1,887.48	956.52	2,081.52	140.53	1,785.56	784.59	2,758.70	32.34	2,328.48
S-23B	0.00	2,369.84	1,609.94	2,613.46	182.87	2,248.90	1,264.59	3,563.87	93.13	3,022.12
S-23C	0.00	1,329.66	231.67	1,491.95	12.54	1,806.34	487.26	2,279.87	0.00	2,270.46
S-23D	0.00	2,991.74	2,189.99	3,358.66	70.36	4,064.26	2,043.30	5,135.59	0.00	5,108.54
Total Flow	**0.00**	**8,578.72**	**4,704.12**	**9,545.59**	**406.31**	**9,905.06**	**4,579.74**	**13,738.03**	**125.47**	**12,729.61**

a. Headwater elevation from structures S-21A, S-20G and S-20F were utilized to calculate discharges for culverts S-23A, S-23B, S-23C and S-23D (see Charkhian 2014 for more information).

b. Available fresh water diverted through the L-31E Culverts to wetlands east of the L-31E Canal.

c. Estimated flow through the L-31E Culverts that would have occurred if stage in the L-31E Canal was higher at 2.2 feet NGVD.

Water Quality

Monthly water quality samples collected at stations BB53, MI01, MW01, and PR01 (**Figure 7-92**) were collected and analyzed for WY 2013 and compared to data collected and analyzed the previous two water years. Two types of water quality parameters were analyzed: (1) profile parameters measured along water depth and (2) surface parameters measured close to the water surface at approximately an 0.5-meter depth. The most important change noticed for profile parameters is the statistically significant decrease in salinity and specific conductivity at stations BB53, MW01 and PR01 during WY 2013, especially when compared to WY 2012. This finding is not surprising given that WY 2012 was a relatively dry year; whereas, WY 2013 was a normal to wet year.

There are statistically significant changes for all surface water parameters. Most of the parameters decreased during WY 2013. Color decreased at MW01. Dissolved ammonia decreased at MW01 and PR01. Total suspended solids decreased at BB53 and MW01. Turbidity decreased at MI01 and PR01. Total phosphorous values are already low and have a small variability. Their decrease, even when it seems statistically significant, have the order of magnitude of the laboratory method detection limit and, therefore, are environmentally insignificant. During WY 2013, the only parameter that increased in concentration is nitrate + nitrite; there was a statistically significant increase in concentration at MW01 and PR01.

Nearshore Salinity

Continuous salinity monitoring has been conducted since 2006 in the coastal region near the L-31E Culverts, along the shore between Turkey Point and the C-1 Canal. Salinity stations used for the L-31E Culverts component nearshore region are 14, 22, 28, 44, A8 and B4 (**Figure 7-92**). Salinity is collected near the bottom at these stations at 15-minute intervals within 500 meters of the shore.

For WY2013, salinity exhibited similar conditions for this region as it has during the period of record. Specifically, dry season salinity showed a slightly higher average salinity with a narrower range of conditions than wet season at 29.21 ± 2.10 relative to 21.80 ± 3.84 , respectively. No statistically significant difference in the salinity ranges or seasonal patterns was observed in 2013 relative to the previous two years. Recent apparent trends downward in the mean and median salinity are a reflection of a return of relatively normal precipitation compared to the drier periods observed in the 2011 dry season and 2012 wet season. Note that wet and dry season average salinity was above the RECOVER salinity performance measure target of 20 (see http://www.evergladesplan.org/pm/recover/perf_se.aspx). More detailed analyses of the factors influencing salinity, particularly how water management in the coastal zone is influencing salinity, is necessary to interpret these differences or to ascertain any ecologically significant trends.

Vegetation

The SFWMD has been monitoring vegetation on a monthly basis since October 2010 along two 100-meter transects downstream of culverts S-23A and S-23D to document the composition and coverage of vascular plant species found in the groundcover and canopy strata. The project area is also monitored and treated for Category I and II invasive exotic plant species.

The plant communities found downstream of culverts S-23A and S-23D differ significantly in their vegetative composition, structure, soil type, elevation, and hydrology. The vegetation located downstream of the culvert S-23D consists entirely of dwarf red mangroves whereas the vegetation downstream of culvert S-23A is characteristic of a mature transitional mangrove forest ecotone that has both halophytic and freshwater wetland plant species present. Data collected during the last two years indicate the following: (1) a slight decrease in areal coverage of *Rhizophora mangle* (red mangrove), possibly due to freezing and cold temperatures in winter 2011 (**Figure 7-95**); (2) a slight decrease in percentage of vegetation coverage of *R. mangle* within transect S-23A downstream of culvert S-23A; and (3) no change in the percentage of canopy coverage of vegetation downstream of culverts S-23A except a slight increase in the percentage of canopy coverage of *R. mangle*.

Figure 7-93. Cold weather damage to dwarf *R. mangle* in the vicinity of the L-31E Culverts during winter 2010.

The height of scrub (dwarf) red mangroves are measured at specific distances along the 100-meter transect downstream of culvert S-23D to monitor the effect of increased flow and longer hydroperiod on the height of these mangroves. The red mangroves at this location exhibit a stunted growth (~1 meter tall), which may be the result of hydrologic stress and/or nutrient limitation. The red mangroves downstream of culvert S-23A vary widely in size, and range from seedlings approximately 15 centimeters (cm) tall to mature trees upwards of 15 meters tall. Given the variability in red mangrove height and the feasibility of obtaining accurate height measurements of mature trees within a forested system, the data collected may not provide useful insight or a basis for comparison between the two sites.

Based on field observations and review of the monitoring data collected during the last two years, there has not been a major change in the composition and coverage of vascular plants downstream of culverts S-23A and S-23D.

During this reporting period, the percent of coverage of Category I and II invasive plant species based on the Florida Exotic Pest Plant Council's 2009 List of Invasive Plant Species was less than 5 percent. Chemical and mechanical treatment of exotic plant species was conducted over the past three calendar years to date within approximately 291 acres of the project site. Dominant exotic species removed included *Ardisia elliptica* (shoebutton ardisia), *Casuarina equisetifolia* (Australian pine), *Lygodium microphyllum* (Old World climbing fern), *Cestrum diurnum* (day-blooming jasmine), and *Schinus terebinthifolia* (Brazilian pepper).

Sawgrass Mapping

In April 2012, the project team initiated field mapping of *Cladium jamaicense* (sawgrass) communities within a 370-acre area in the vicinity of the L-31E culverts. The objective of this task is to create a baseline map showing the distribution and relative abundance of *C. jamaicense* within the effective area of culverts S-23A and S-23B (**Figure 7-96**). This reporting period established the sawgrass habitat baseline, which consisted of 42 acres of patchy and scattered sawgrass communities within the area of surface water redistribution and removal of exotic plants (northern culverts).

Biscayne Bay Coastal Wetlands Project Summary

Monitoring results demonstrate substantial diversions of fresh water from the C-100 canal complex by the S-700 pump station as part of the Deering Estate Component, resulting in a clear improvement of hydrologic conditions in Cutler Slough. With pumping at 100 cfs, the wetland inundation in the slough is at 95% of the target. Vegetation in Cutler Slough already appears to be responding to the changed hydrology, with upland tree species in the slough beginning to die back as expected. Salinity in surface water at staff gages within the slough decreased in response to S-700 pumping; whereas, salinity responses near the mouth of Cutler Creek responds sporadically to S-700 pump operations. For the L-31E Component, water diversions are well short of the target and downstream hydrologic and ecologic responses have not been detected. For WY2012 and WY2013 combined, approximately 10,000 ac-ft is estimated to have been diverted into the wetlands east of the L-31E canal, which is short of the flow target of 46,000 ac-ft. However, the water that was diverted by this component of the project previously would likely have been released as damaging pulsed, point source discharge to the bay. Calculations indicate that a modest increase in stage in the L-31E would divert significantly more water into the wetlands through the project culverts. The modification to the project proposed by the SFWMD that would increase L-31E stage would improve project performance. Also, although the Cutler Flow-way Component has not been constructed, it is anticipated that construction of this component may provide more benefits than the Deering Estate and L-31E components combined.

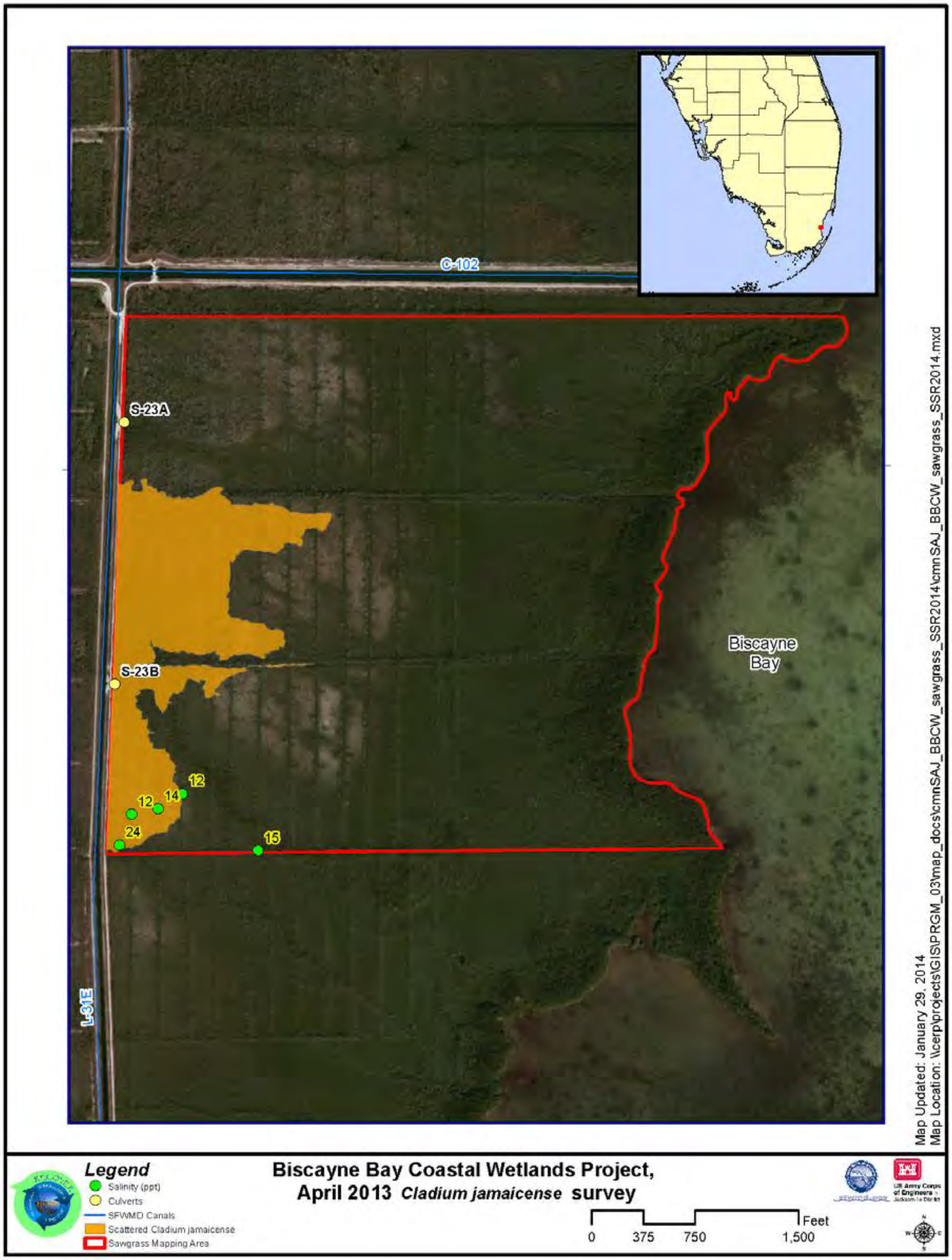


Figure 7-94. April 2013 *C. jamaicense* spatial coverage within the influence of flow from culverts S-23A and S-23B.

C-111 Spreader Canal Western Project

The C-111 Canal is the southernmost canal of the Central and Southern Florida (C&SF) Project and is located in south Miami-Dade County (**Figure 7-97**). The C-111 Canal courses through extensive marl wetland prairie and coastal mangrove marsh before it empties into Manatee Bay. The canal serves a basin of approximately 100 square miles and is the final segment of the South Dade Conveyance System and functions primarily to provide flood protection and drainage for the agricultural areas to the west and south of Homestead, Florida. The canal has had unintended effects on groundwater levels in Taylor Slough, and has contributed to the reduced discharge to northeastern Florida Bay and increased discharges to Manatee Bay and Barnes Sound. Taylor Slough is a natural drainage feature of the Everglades that flows southwest into numerous tributaries that eventually empty into Florida Bay. The C-111 SCWP focuses on the restoration of flows to Florida Bay via Taylor Slough as well as the restoration of the Southern Glades and Model Lands and coastal zone of Florida Bay.

In January 2012, the SFWMD completed construction on several important project components (**Figure 7-97**) that were fully operational in June 2012:

- ***Frog Pond Detention Area*** **(constructed) –** A 225-cfs pump station (S-200) downstream of the existing S-176 structure now routes excess water to an approximately 500-acre aboveground detention area, which would otherwise be discharged down the lower C-111 Canal via the S-177 structure.

- ***Aerojet Canal*** **(constructed) –** A second 225-cfs pump station (S-199) was constructed immediately upstream of the existing S-177 structure and downstream of State Road 9336. The pump station is intended to work in tandem with and mirror the Frog Pond Detention Area pump operations, and routes water to the Aerojet Canal via a northerly extension of the canal. Same pumping constraints apply to the S-199 structure as they do to the S-200.

- ***One Plug at S-20A*** **(constructed)** ***and Operational Changes at Existing Structure S-20*** **(adaptive implementation that began January 2014) –** Construction of a permanent plug at existing structure S-20A in the L-31E Canal, and operational changes at existing the S-20 structure. The proposed plug near S-20A and proposed operational changes at S-20, specifically raising the “open and close” triggers to 0.5 feet, are intended to restore hydroperiods within the wetlands east of Card Sound Road.

- ***Ten Plugs in the C-110 Canal*** **(constructed) –** Construction of earthen plugs at key locations within the C-110 Canal was done in order to promote sheet flow within the Southern Glades.

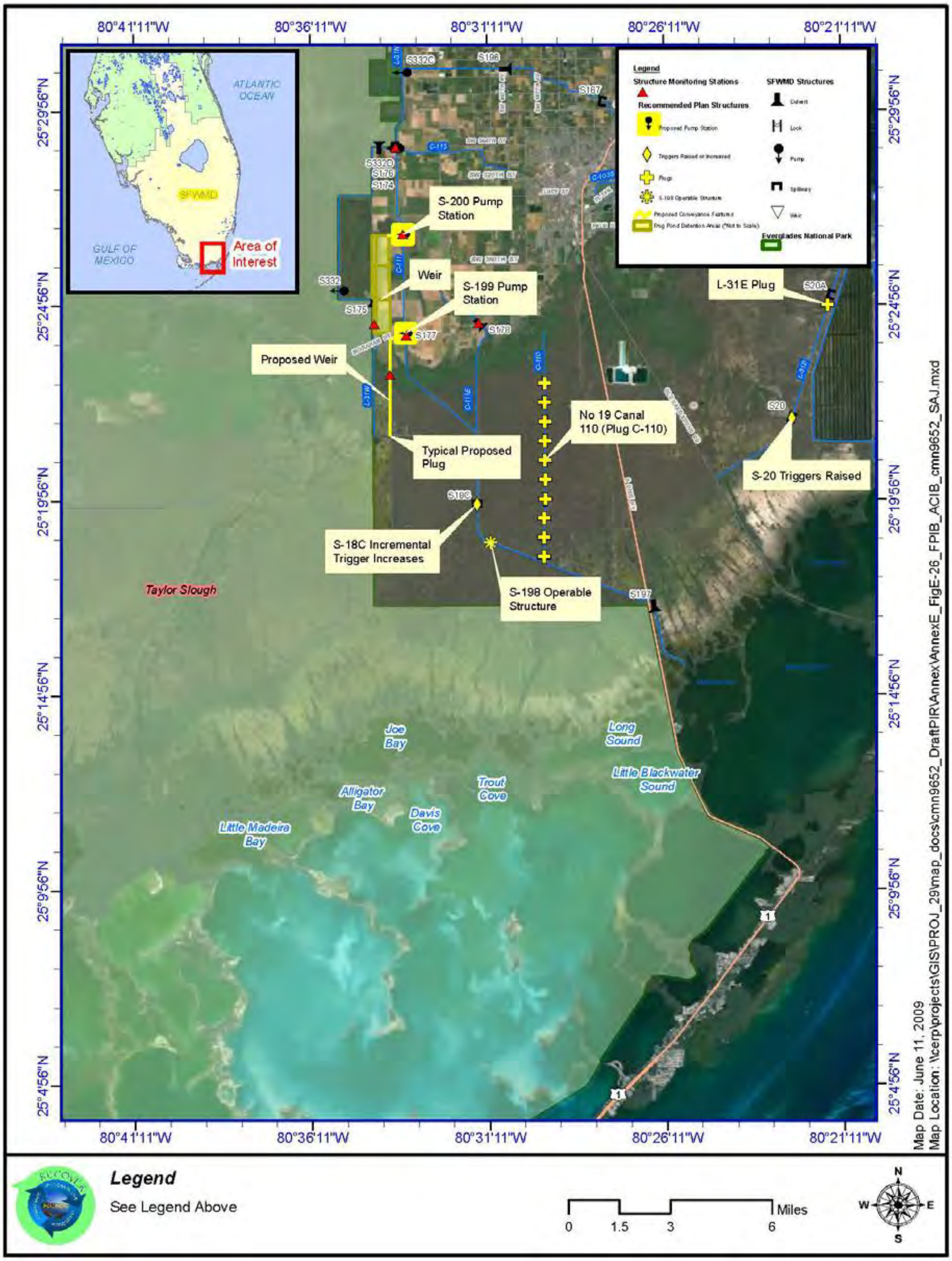


Figure 7-95. Map of C-111 SCWP components.

- ***A New Operable Structure in the Lower C-111 Canal* (adaptive management option in future) –** The proposed S-198 structure is an operable gated spill way to reduce current levels of seepage from the lower C-111 Canal and was included as part of the C-111 SCWP project implementation report (USACE and SFWMD 2011). It will not be constructed until the project is federally authorized for cost share, budgeted for construction by the USACE, and monitoring and assessment feedback from the first constructed components of the C-111 SCWP components continues to support the need.

This report is intended to capture a baseline assessment for comparison in future SSRs to determine restoration success (i.e., achievement of project goals and objectives) of the C-111 SCWP and the need for any adaptive management actions to improve performance. Project restoration goals and objectives and performance measures can be found in the project implementation report (USACE and SFWMD 2011). Because project-level monitoring has been scaled back for the state-constructed portion of the project, this report focuses heavily on the RECOVER-funded MAP data, as well as data funded by non-MAP related programs. In addition, this report presents data for the initial year of C-111 SCWP operations, which may be used in future SSRs to show project performance improvements in Taylor Slough, Southern Glades, and Model Lands.

C-111 Spreader Canal Western Project Structure Flows

Water control structures, S-200 and S-199, work in tandem to fill the Frog Pond Detention Area and release excess water (above 2.5 feet) into the Aerojet Canal to create a hydrologic ridge during the wet season and moving into the early dry season. Even though this report is focused on reporting up to WY 2013, data through August 2013 of WY 2014 reveals these features were not operational during the same June to July timeframe as the previous year due to extremely wet conditions already in the system. In WY 2013, the flows occurred during the wet season and into the early dry season (June to January) to minimize seepage from Taylor Slough to the C-111 canal. (**Figure 7-98**).

Because of the higher water levels due to relatively high rainfall in WY 2013, detecting project effects in the marsh was not conclusive and will likely require another four years of data with the S-200 and S-199 project components operating over a variety of rainfall conditions (wet, average, and dry) to determine project effects. However, analysis of S-331 flow compared to S-177 flow reveals that flow is being diverted into the Frog Pond Detention Area via the S-200 structure (**Figure 7-99**; see last two years on graph), which indicates the project is adjusting flow in the water management system as expected.

The S-331 structure was installed in 1983 and sends water south along the L31 Canal and to the C-111 Canal via the S-176 structure, then through the S-177 structure, through the S-18C structure, and out of the water management system (into Manatee Bay) via the S-197 structure. **Figure 7-100** reveals that, in WY 2013, water (with C-111 SCWP flows) is divergent from the past 30-year trend of S-331 inflow compared to S-177 flow. S-331 flow was close to 150,000 ac-ft compared to approximately 25,000 ac-ft of flow through the S-177 structure, about 30 percent of what it would have been normally. Ultimately, the flow that would have gone directly through S-177 passed through the South Dade

Detention areas into the C-111 Spreader Canal Frog Pond Detention Area and out the Aerojet Canal to keep stages high on the eastern edge of ENP and reduce seepage out of ENP, as intended by the project.

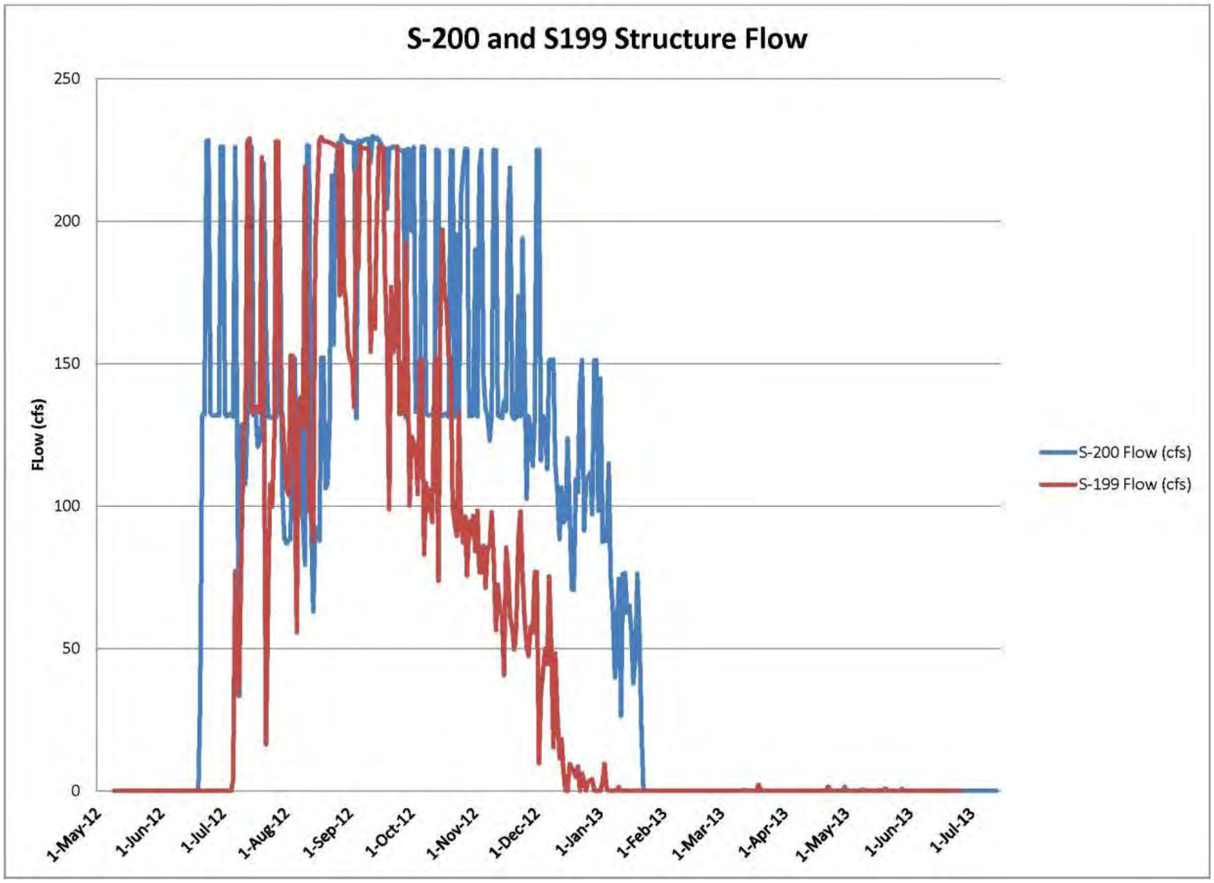


Figure 7-96. C-111 SCWP water control structure S-200 and S-199 flow for WY 2013.

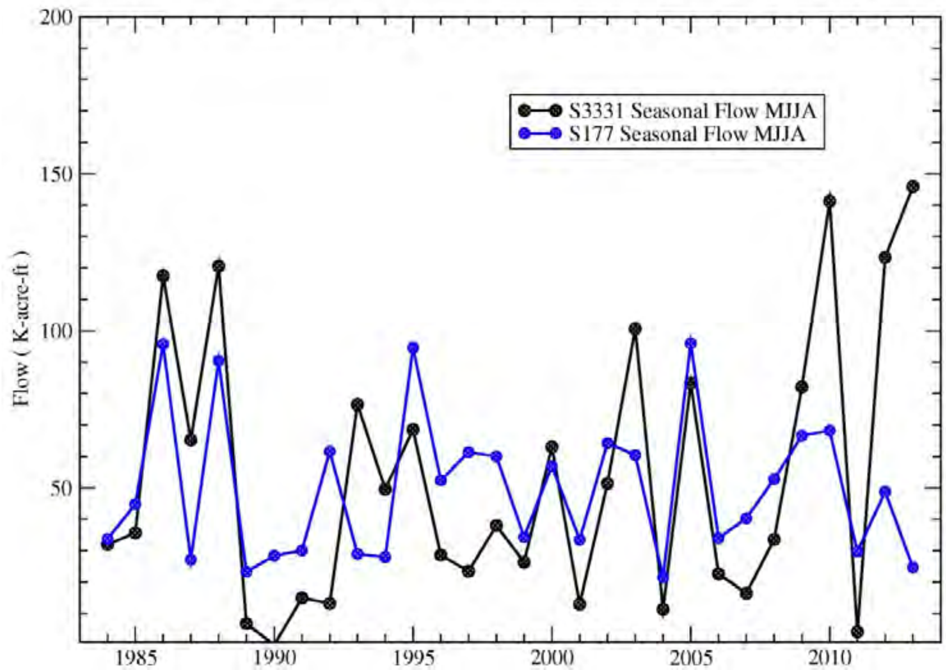


Figure 7-97. S-331 and S-177 flows from WY 1983 to WY 2013.

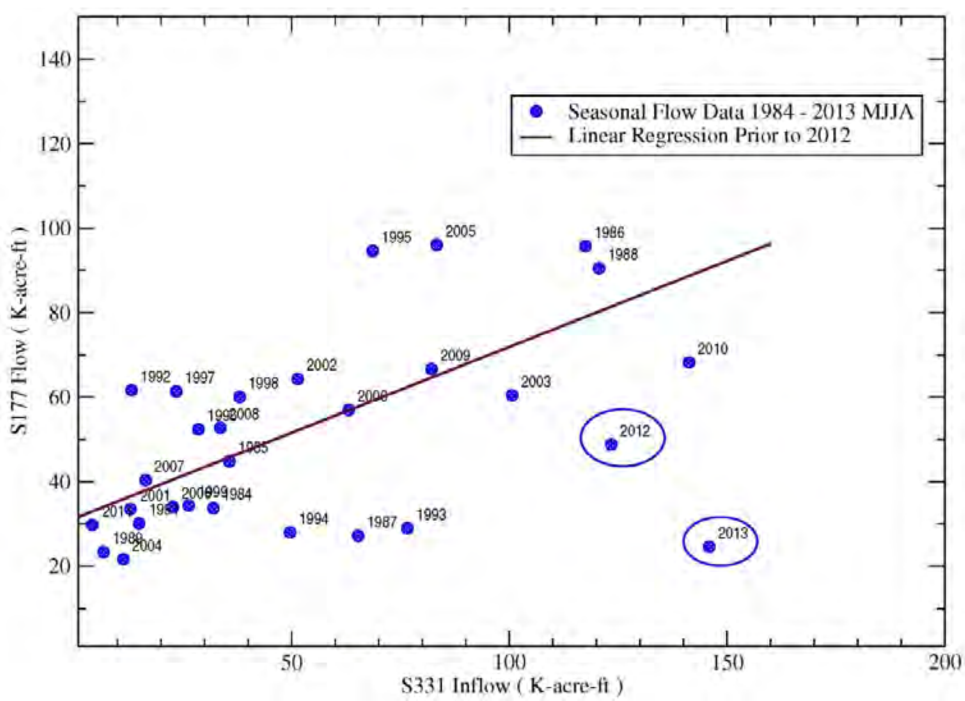


Figure 7-98. Scatter plot of S-177 flow versus S-331 flow with WY 2012 and WY 2013 circled. Linear regression line includes data from WY 1984 to WY 2012, before C-111 SCWP operations took effect in WY 2013.

C-111 Spreader Canal Western Project Hydrology

Hydroperiods

Hydroperiods are calculated based on Everglade Depth Estimation Network (EDEN) data sets for the C-111 SCWP area. The project features have only been in place for one year; therefore data are not available yet to adequately assess the resulting hydrologic changes. Average annual hydroperiod maps are presented here for the IOP (2000–2012) (**Figure 7-101 left panel**) and pre-IOP (1992–1999) (**Figure 7-101 right panel**) periods and serve as baseline conditions for comparison with post-project conditions in future SSRs. During the IOP period (2000–2012), the average annual hydroperiod in Taylor Slough ranged from 240 to 300 days and, in the headwaters of Taylor Slough, ranged from 60 to 180 days.

Comparison of average annual hydroperiods during IOP and pre-IOP periods reveals two important changes that occurred (**Figure 7-101 middle panel**). First, shorter hydroperiods in Shark River and Taylor Sloughs during the IOP period reflect, in part, the drier rainfall conditions during the late wet season and throughout the rest of the dry season during that period. Second, hydroperiods in the circled area of the middle panel of **Figure 7-101** are on average 60 days longer during the IOP period compared to the pre-IOP period. These changes may be associated with the C-111 South Dade Conveyance Project that was implemented and became fully functional during the IOP period for the purpose of reducing seepage eastward into the urban areas, thereby (1) increasing flows in Taylor Slough, (2) increasing coastal creeks flows to Florida Bay, and (3) lowering salinity in northeastern Florida Bay.

Flows at S-331 increased by 20 percent during the wet season and decreased by 5% in the dry season between IOP and pre-IOP periods. This difference reflects changing patterns of wet season and dry season rainfall, combined with changing flood control and water supply operations. Flow at S-176, the next structure downstream, decreased by 70 percent, reflecting the movement of water from S-331 into the South Dade detention areas along the L-31N Levee.

Hydrologic impacts along the L-31N Levee (circled areas in **Figure 7-101**) suggest that seepage toward the L-31N canal was reduced in this area but seepage from Taylor Slough toward the C-111 Canal was not reduced. This conclusion is supported by Kotun and Renshaw’s (2013) more detailed analysis. Ultimately, the C-111 SCWP is expected to provide additional seepage management for the southern portion of Taylor Slough by reducing the effects of the C-111 Canal and increasing hydroperiods and flow through the slough and coastal creeks to central and northeastern Florida Bay.

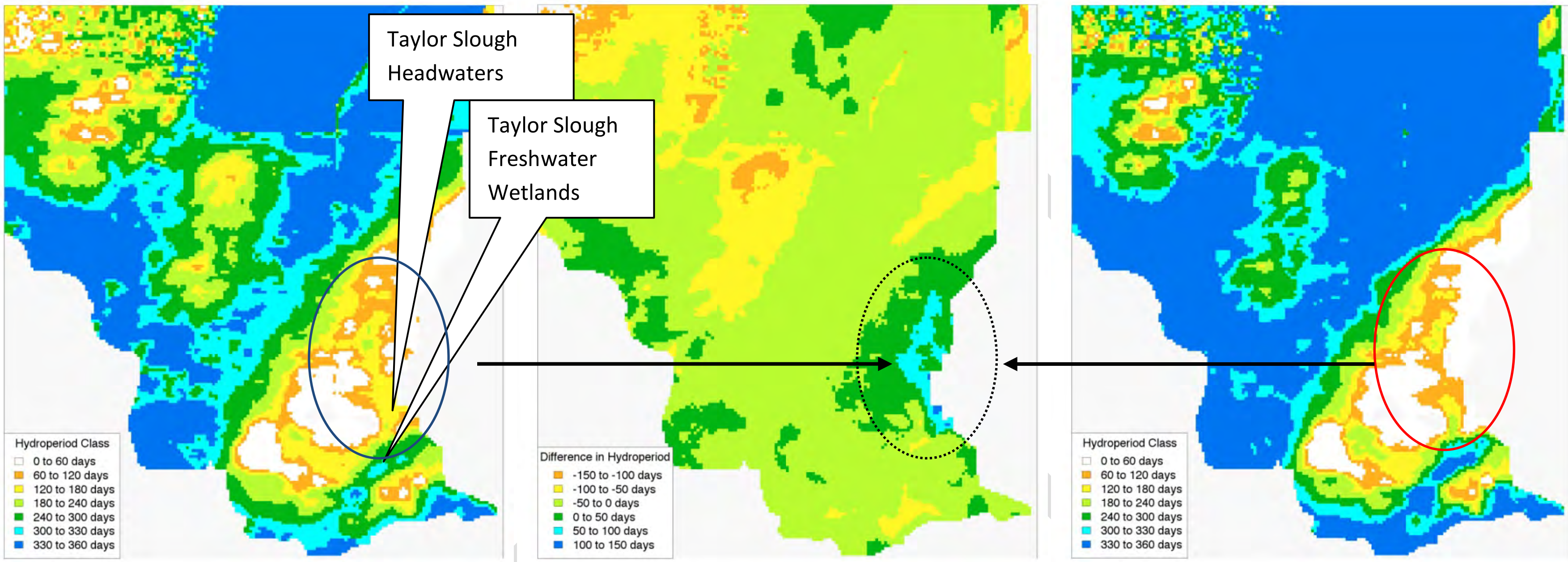


Figure 7-99. Left panel depicts average annual hydroperiods in ENP for the IOP period (2000–2012). Hydroperiod classes range from 0 to 60 days (white), 60 to 120 days (orange), 120 to 180 days (yellow), 180 to 240 days (light green), 240 to 300 days (light blue), and 330 to 360 days (dark blue). Right panel depicts average annual hydroperiods in ENP for the pre-IOP period (1992–1999). Center panel depicts the difference between average annual hydroperiods for IOP less pre-IOP period. Hydroperiod difference classes include -150 to -100 days shorter (orange), -100 to -50 days shorter (yellow), -50 to 0 days shorter (light green), 0 to 50 days longer (dark green), 50 to 100 days longer (light blue), 100 to 150 days longer (dark blue). The circled area in each figure represents the same area of interest that indicates increased hydroperiods between the two time periods.

Water Levels

Average water levels are compared at select gages in Taylor Slough (TS2 [Taylor Slough Bridge]; R127; TSH [Taylor Slough Headwater]; NP67; and E146) and Southern Glades (EPSW and EVER 6) (**Figure 7-102**) where stages are expected to increase in the future with C-111 SCWP operations. Sites in SRS (NP203 and P33) are used as controls to be able to determine if changes in Taylor Slough are due to the project or rainfall patterns. NP206 represents the western edge of the C-111 South Dade Conveyance Project structure influence. Mid-range (25th to 75th percentile) water levels were calculated using EDEN gage data for two baseline operational periods (Pre-IOP period 1991–1999, IOP period 2000–2012 with C-111 South Dade Conveyance Project in place) before the project was constructed and operational.

Figure 7-100. Map of EDEN gages analyzed.

Baseline average annual water level data indicates that TS2, R127, NP67, and E146 experienced a decrease in water levels from January to June during the IOP period (2000–2012) compared to the pre-IOP period (1991–1999) (see **Figure 7-103** which shows NP67 as an example). However, water levels in the ENP panhandle below the C-111 Canal did not change that much over the two baseline periods, reflecting continued seepage from Taylor Slough during the IOP POR (see **Figure 7-104** which shows EPSW water level comparison). The control sites in SRS (i.e., NP203 depicted in **Figure 7-105**), which are not expected to be influenced by the C-111 South Dade Conveyance Project, did experience a similar decrease in water levels from January to June.

In general, there are slower recession rates closer to the detention areas in the central-eastern part of ENP, which is reflected by the NP206 gage (**Figure 7-106**) effect shown in the hydroperiod maps. Dry season differences shown in Taylor and SRSs are likely due to rainfall differences between the two periods. Rainfall during IOP was drier during the winter and spring and slightly wetter during the summer and fall compared to pre-IOP conditions (**Figure 7-107**). This does not explain the lack of difference in water levels in the southeastern (panhandle) portion of ENP, represented by EPSW water levels (**Figure 7-104**), especially during the wet season. It is likely that dry season water inputs to the ENP panhandle were supplemented by the C-111 Canal relatively more during the IOP period than the pre-IOP period. This supplement could have been derived from water supply conveyance via the L-31 Canal or via seepage from nearby marshes (likely from Taylor Slough). The finding of higher stages in the C-111 Canal downstream of S-18C (**Figure 7-108**) during the IOP period than the pre-IOP period, especially during the wet season, is consistent with this explanation. The above results illustrate the strong influence of both precipitation and the C-111 Canal on hydrology and highlights the need for a comprehensive monitoring program and careful evaluation of the data.

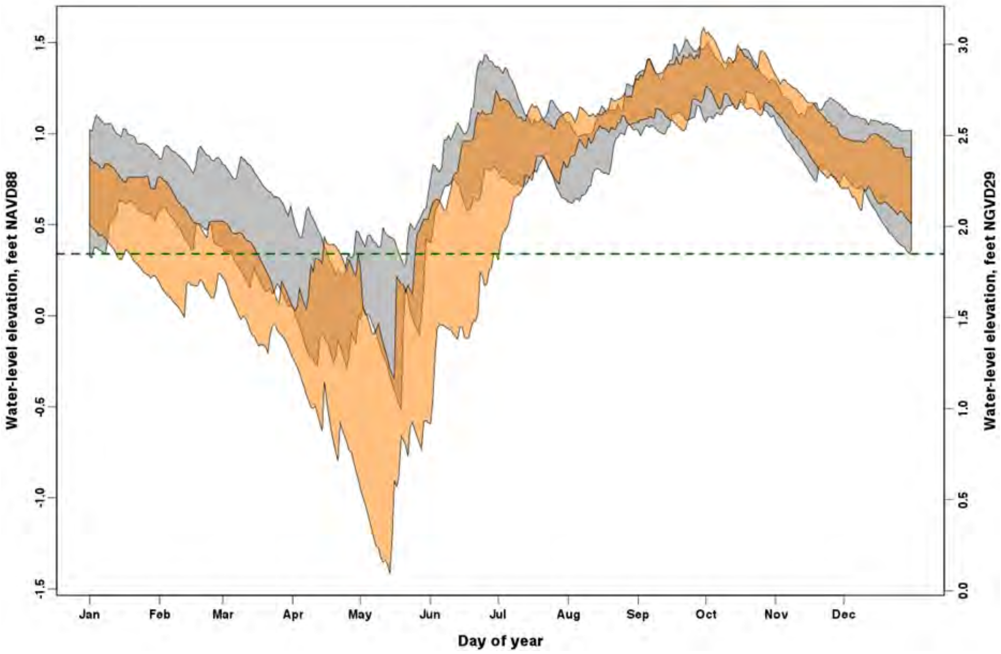


Figure 7-101. Comparison of interquartile (25th percentile to 75th percentile) stages between pre-IOP period (1991–1999 as gray ribbon) and IOP period (2000–2012 as light orange ribbon) at NP67. The overlap of data distributions between these periods is shown in dark orange. The dotted line is ground elevation.

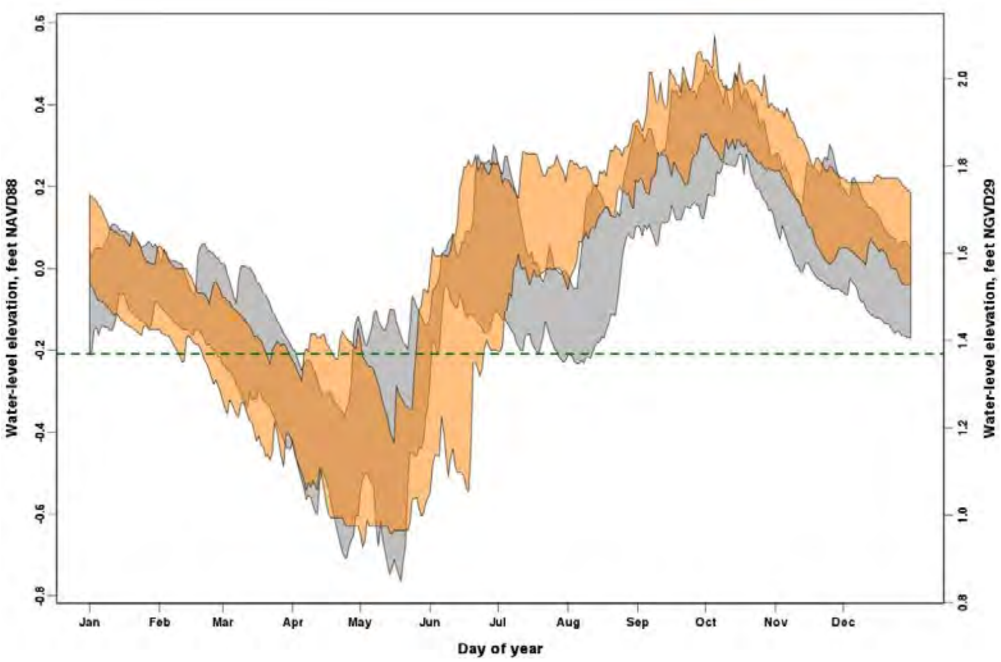


Figure 7-102. Comparison of interquartile (25th percentile to 75th percentile) stages between pre-IOP period (1991–1999 as gray ribbon) and IOP period (2000–2012 as light orange ribbon) at EPSW. The overlap of data distributions between these periods is shown in dark orange. The dotted line is ground elevation.

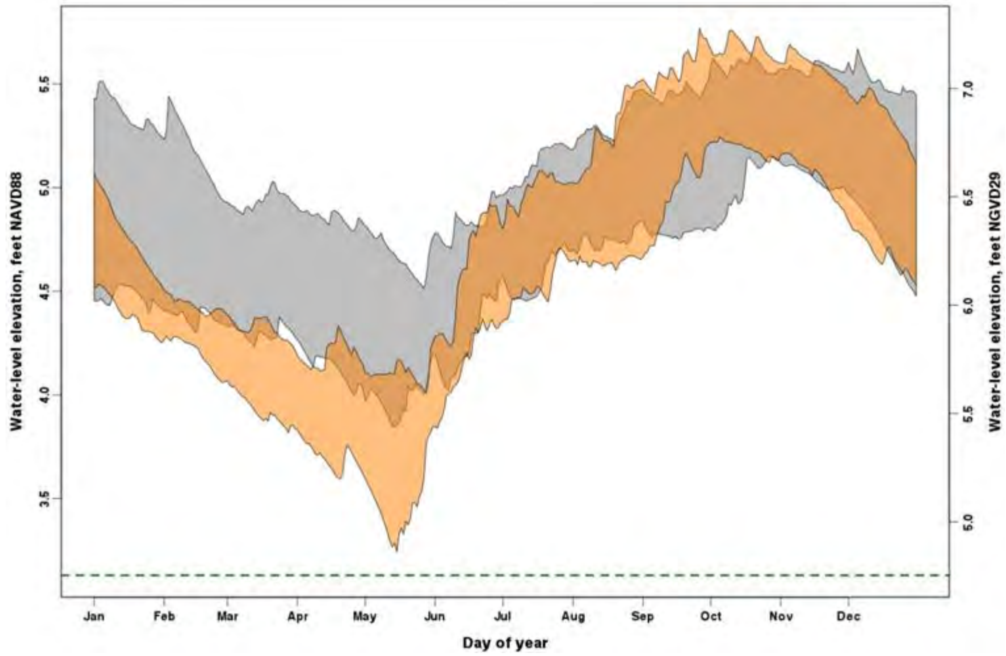


Figure 7-103. Comparison of interquartile (25th percentile to 75th percentile) stages between pre-IOP period (1991–1999 as gray ribbon) and IOP period (2000–2012 as light orange ribbon) at NP203. The overlap of data distributions between these periods is shown in dark orange. The dotted line is ground elevation.

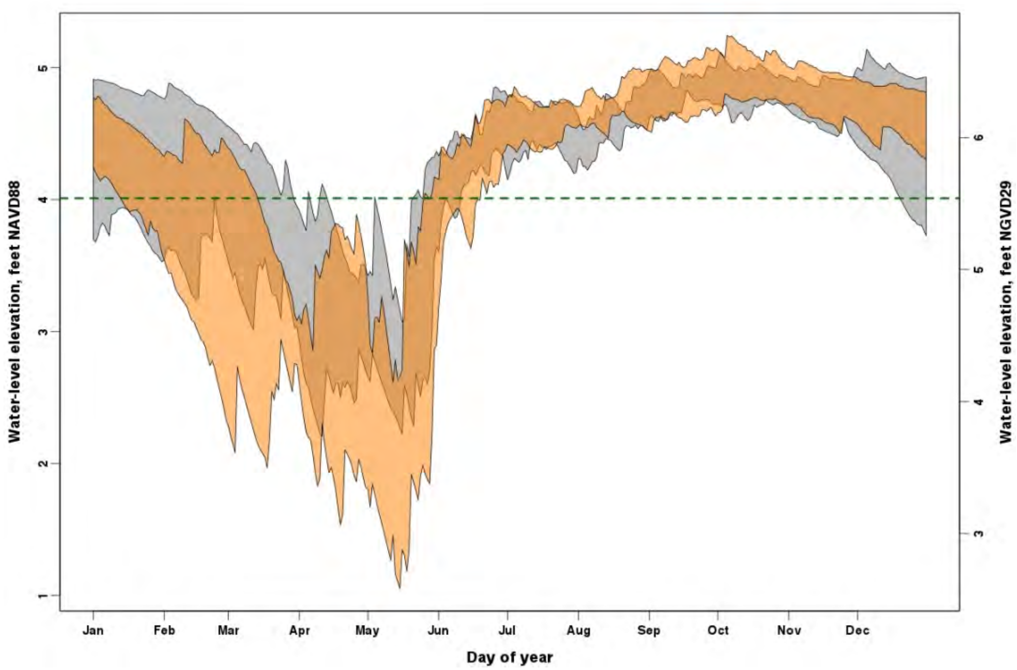


Figure 7-104. Comparison of interquartile (25th percentile to 75th percentile) stages between pre-IOP period (1991–1999 as gray ribbon) and IOP period (2000–2012 as light orange ribbon) at NP206. The overlap of data distributions between these periods is shown in dark orange. The dotted line is ground elevation.

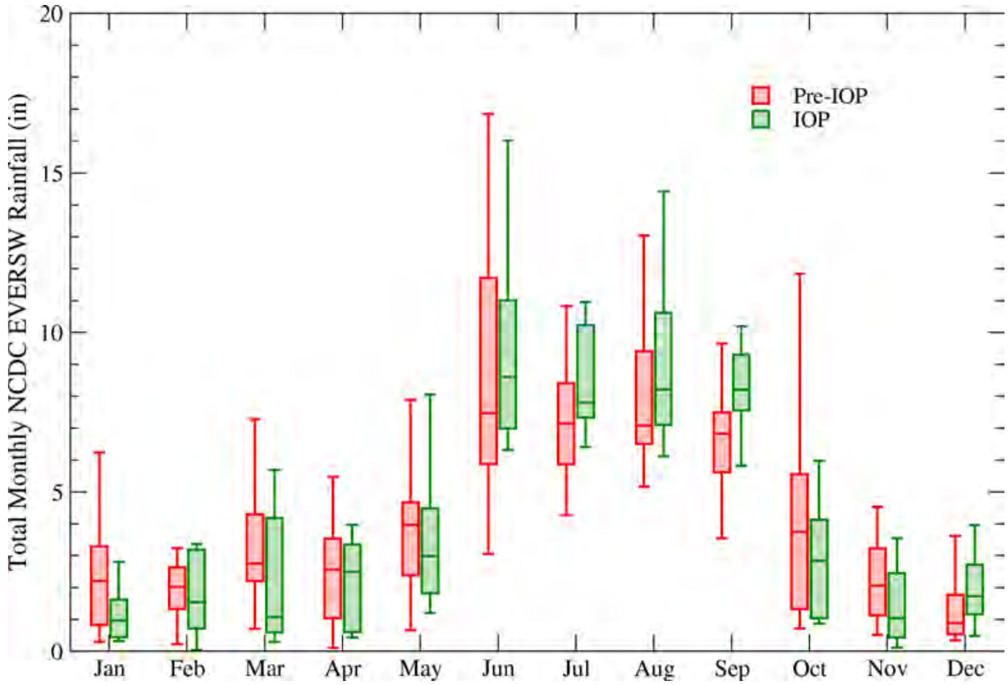


Figure 7-105. Precipitation at station EPSW in the ENP panhandle prior to IOP (1991–1999) and following IOP implementation (2000–2012).

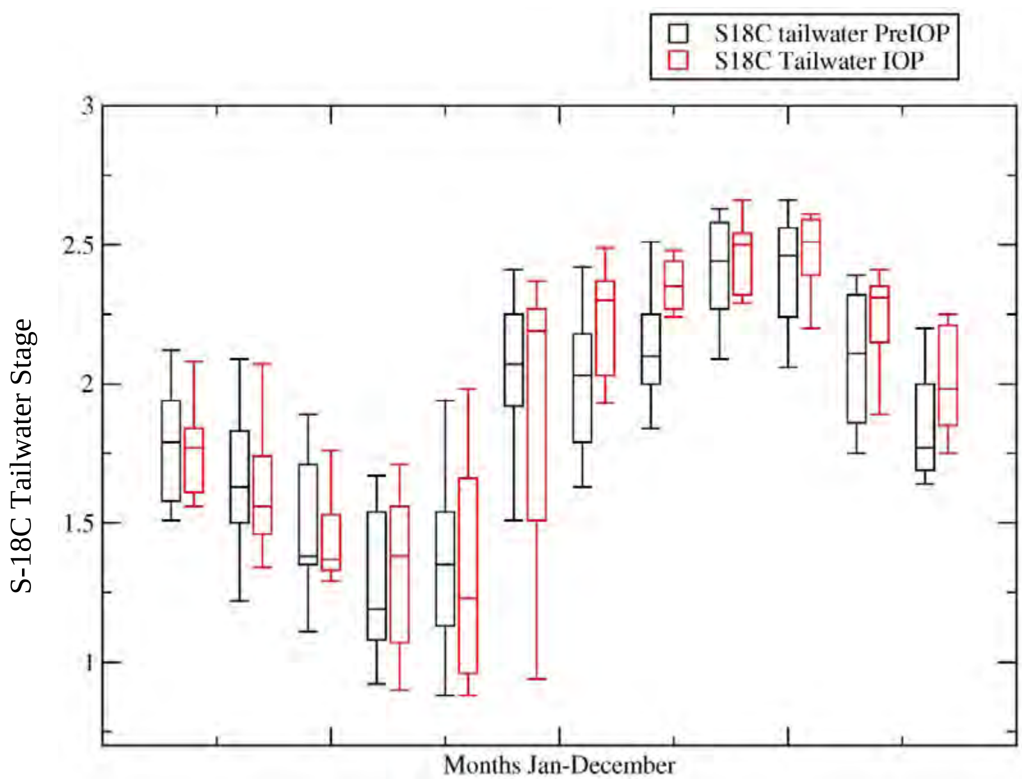


Figure 7-106. S-18C tailwater stages pre-IOP (1991–1999 in black) and during IOP (2000–2012 in red) box-and-whisker plots.

Coastal Creek Flow

Coastal creek flow is monitored by the RECOVER Coastal Gradients integrated monitoring network of hydrology and water quality data for major coastal creeks and rivers along generalized coastal gradients or transects. Coastal creek flow links freshwater hydroperiods and stages affected by the C-111 SCWP to downstream salinity conditions affecting Taylor Slough and Florida Bay, as well as vegetation and freshwater and estuarine prey and predators.

Flow from the C-111 Canal to the eastern panhandle of ENP and northeastern Florida Bay is approximated using the difference between flow from the C-111 Canal at S-18C structure and flow released to Manatee Bay via the C-111 Canal at the S-197 structure. Flow into Taylor Slough is measured at Taylor Slough Bridge, where constricted flow has enabled long-term estimates of flow down the slough. Taylor Slough Bridge is not a control structure but is a proxy for controlled flow from a series of upstream pumping stations along the L-31W Canal. Between WY 2000–WY 2012, flows at Taylor Slough Bridge and S-18C averaged 58,558 and 136,652 ac-ft annually, respectively. These flows, when combined with rainfall, evapotranspiration, canal leakage into ENP, and southward flow through the C-111 Canal, ultimately affect downstream coastal creek flows. Coastal creek flows for the full Taylor Slough drainage complex averaged 306,110 ac-ft annually. This total was distributed among the major transects as follows: 31.4% to Taylor Slough, 48.1% to Trout Creek, and 20.5% to Long Sound. The C-111 SCWP is expected to increase flows to coastal creeks in Taylor Slough and Trout Creek.

C-111 Spreader Canal Western Project Salinity

Salinity measured in the Coastal Gradients transects in the Taylor Slough-C-111 drainage complex still fluctuates greatly in creeks north of Little Madeira (0–40), Joe Bay (5–40), and Long Sound (5–40) (**Figure 7-109**). While there appears to be a downward trend in coastal creek salinity between WY 2011 and the provisional data of WY 2013, the time series is not long enough to determine if lower salinity is due to the C-111 SCWP.

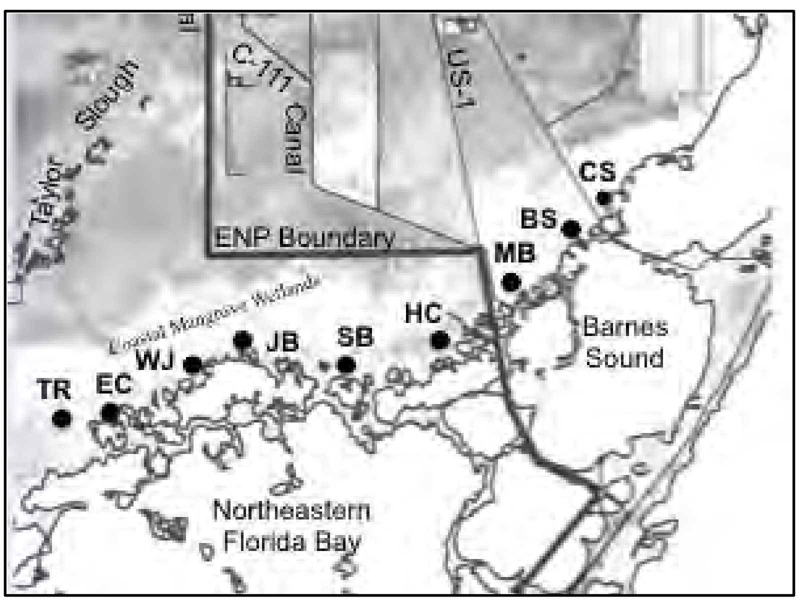


Figure 7-107. Map depicting Florida Audubon monitoring sites along northeastern Florida Bay and adjacent coastal mangrove wetlands. (Note: TR – Taylor River; EC – East Creek; WJ – West Joe Bay; JB – Joe Bay; SB – Sunday Bay; HC – Highway Creek; MB – Manatee Bay; BS – Barnes Sound; and CS – Card Sound.)

C-111 Spreader Canal Western Project Freshwater/Estuarine Prey

One of the stated goals of C-111SCWP was to increase freshwater flows toward Florida Bay via Taylor Slough. This would result in lowered salinity in northeastern Florida Bay and associated wetlands to the north as indicated by data collected at sites shown in **Figure 7-109** (USACE and SFWMD 2011). Lorenz et al. (2009) demonstrated that roseate spoonbills and their prey were excellent indicators for this area and that they are representative of the overall ecological health of this region. Spoonbills that nest on the islands in northeastern Florida Bay forage primarily on the small (< 6.5 cm) demersal fishes that live in the coastal wetlands north of this area. Lorenz (1999) and Lorenz and Serafy (2006) demonstrated that these fish are more productive under relatively low salinity conditions (**Figure 7-110**). Spoonbills were found to need a high abundance of these fishes in order to successfully raise and fledge chicks (**Figure 7-111**; Lorenz 2013a).

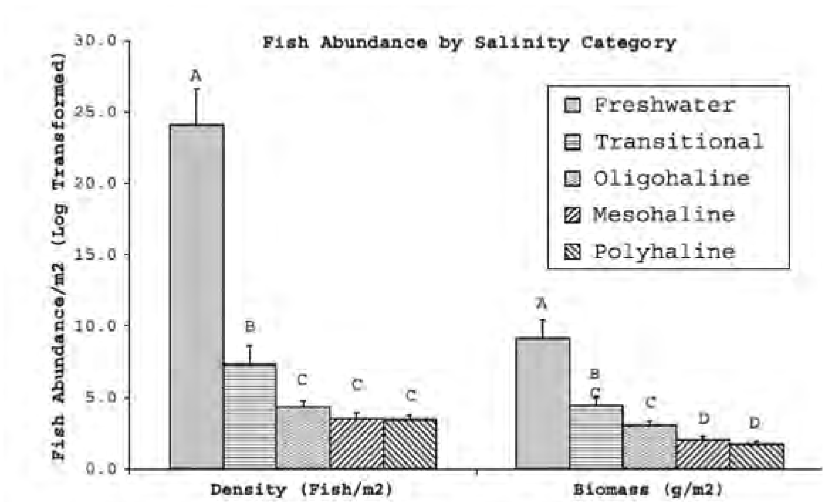


Figure 7-108. Fish abundance by salinity category. The lower the salinity category, the greater the production (from Lorenz and Serafy 2006).

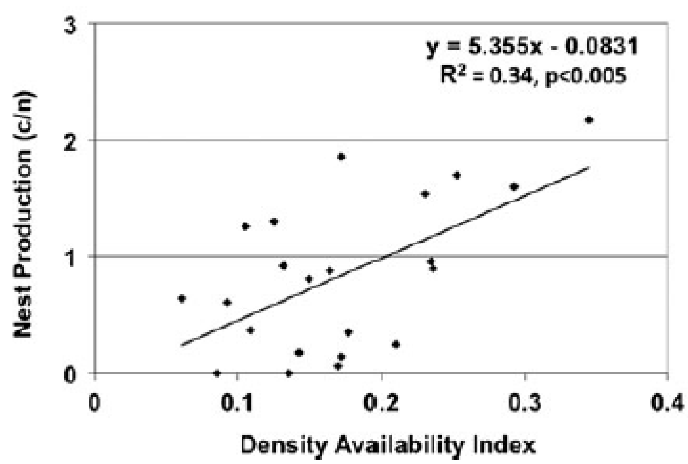


Figure 7-109. Average annual spoonbill nest production as a function of the annual index of prey fish availability (from Lorenz 2013a).

In addition to the relationship between fish and spoonbills, Madden et al. (2013) demonstrated that lower salinity also results in higher cover of submerged aquatic plants, specifically the species *Ruppia maritima* (**Figure 7-112**). *R. maritima* provides food and habitat for the prey fishes that spoonbills exploit. Spoonbill nest numbers in northeastern Florida Bay have been declining since completion of the South Dade Conveyance System in 1984 (**Figure 7-113**; Lorenz 2013a) probably due to the salinity stresses that the system created in northeastern Florida Bay (Lorenz 2013b). This explanation is supported by the fact that ideal conditions for *R. maritima* and prey fish production have only occurred infrequently since the South Dade Conveyance System came on line and this is translated up the food chain. In short, salinity stress resulting from the operation of the system resulted in lower primary (**Figure 7-112**) and secondary (**Figure 7-110**) productivity in the coastal wetlands resulting in lower spoonbill nest production (**Figure 7-111**) and declining spoonbill nest numbers (**Figure 7-113**). If the C-111 SCWP is successful at moving more fresh water through Taylor Slough, then lower and more stable salinity is expected that will result in greater ecosystem production.

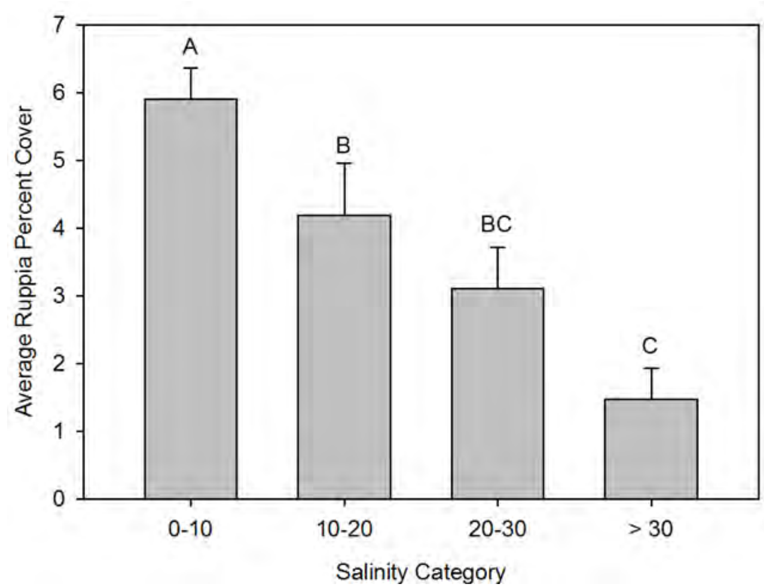


Figure 7-110. *R. maritima* percent cover with standard error along the Joe Bay and Taylor River transects with respect to salinity category. Category C is statistically lower than Categories B and A, and salinity in the range of 0–10 (Category A) is significantly more favorable for *R. maritima* than higher salinity categories.

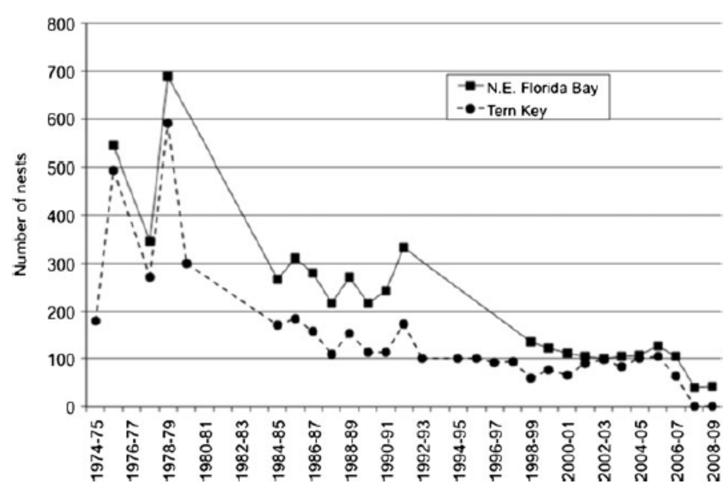


Figure 7-111. Number of roseate spoonbill nests in northeastern Florida Bay 1975–2009. Note that after construction began on the South Dade Conveyance System (1982–1984) numbers dropped precipitously.

Audubon Florida has a network of specific sites within these wetlands (**Figure 7-109**) and monitors hydrology and salinity, aquatic vegetation, and prey abundance at these sites. They also survey spoonbill nesting colonies to gage nesting success and overall nest numbers. This information has been useful to help establish baselines before the C-111 SCWP project was operational, as well as identify restoration benchmarks and determine restoration success.

C-111 Spreader Canal Western Project Roseate Spoonbills

Lorenz et al. (2009) set targets for restoration efforts for spoonbills and prey fish in northeastern Florida Bay based on previous findings. These include at least 688 spoonbill nests in northeastern Florida Bay (based on peak nesting numbers in 1978; **Figure 7-113**) and an average nest production rate of 1.38 chicks per nesting attempt (based on pre-1984 estimates) and spoonbills should produce, on average, greater than 1 chick per nest attempt in 7 of every 10 years. Current trends over the past three years have seen increasing numbers of nests from 87 in WY 2011 to 203 in WY 2013 (see the Florida Bay and the Lower Southwest Coast Roseate Spoonbills section earlier in this chapter). In addition, the nesting success rate has increased to 7 times in the past 8 years. These improving trends are expected to increase with additional years of C-111 SCWP operations.

The prey fish community target was that at least 40% of the community should be classified as freshwater species using the techniques of Lorenz and Serafy (2006). In the past several years, oligohaline species have dominated Taylor River (91.8%), Joe Bay (94.7%), and Highway Creek (78.5%) sites (Lorenz, et al., 2013) and Taylor River and Joe Bay have experienced downward trends in freshwater species. It is possible the changes noted above are in response to the implementation of the C-111 SCWP; however, it is too early in the implementation period to fully capture and correlate findings to the project.

C-111 Spreader Canal Western Project Estuarine Submerged Aquatic Vegetation

The C-111 SCWP monitoring plan was revised by the SFWMD in 2013 from the C-111 SCWP project implementation report ecological monitoring plan (USACE and SFWMD 2011) and no longer includes SAV monitoring for restoration success. Given the reduction in funding of project-related SAV monitoring, this assessment focuses on other SAV monitoring funded by SFWMD, RECOVER, and Miami-Dade County developed to address different objectives but is located downstream of the C-111 SCWP area of influence. This monitoring is used to provide a baseline for future C-111 SCWP assessment to detect changes from restoration. In addition to the spoonbill and fish targets mentioned above, increased SAV cover by *R. maritima* (**Figure 7-112**) is expected with the lowered salinity regimes from restoration. In the last three hydrologic years, *R. maritima* has averaged less than 5% at the Taylor River and Joe Bay stations (**Figure 7-109**), while the Highway Creek station fluctuated between 3% to 12% *R. maritima* coverage. Data analyses not shown here (Lorenz, unpublished) identifies a clear relationship between salinity lower than 30 on a long-term annual average and increased percent area covered by *R. maritima*. In addition, salinity variability is negatively correlated with *R. maritma* coverage (R^2 = -0.815, $p < 0.05$, Pearson; **Figure 7-114**), which is also supported by other studies in Florida Bay on macrophyte biomass and salinity as well as *R. maritima* seedling survival (Montague and Ley 1993, Strazisar et al.

2013) (**Figure 7-115**). Ultimately, the C-111 SCWP is expected to result in increased freshwater flows to Taylor Slough, thereby lowering salinity and favoring *R. maritima* growth, reproduction, and recruitment. Ideally, operations will also continue to focus on reducing the variability in salinity in northeastern Florida Bay to encourage increases in *R. maritima* coverage. Increased *R. maritma* combined with *Thalassia testudinum* and *Halodule wrightii* SAV species will provide more habitat niches in north central and eastern Florida Bay, resulting in a broader assemblage of valued estuarine fauna, including juvenile and adult stages of several economically important species (pink shrimp and spotted seatrout) (RECOVER 2004b).

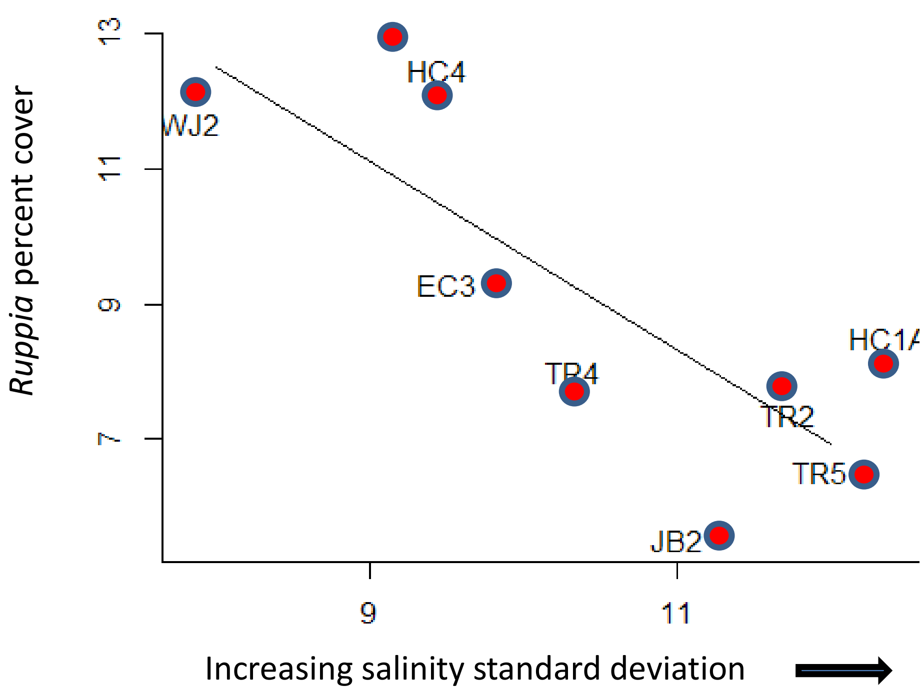


Figure 7-112. Relationship between salinity variability (standard deviation) and *R. maritima* coverage (R = -0.82, p < 0.05). Analysis was limited to sites with > 5% average *R. maritima* coverage. Site JB3, which had an exceptionally robust population, was excluded from this analysis.

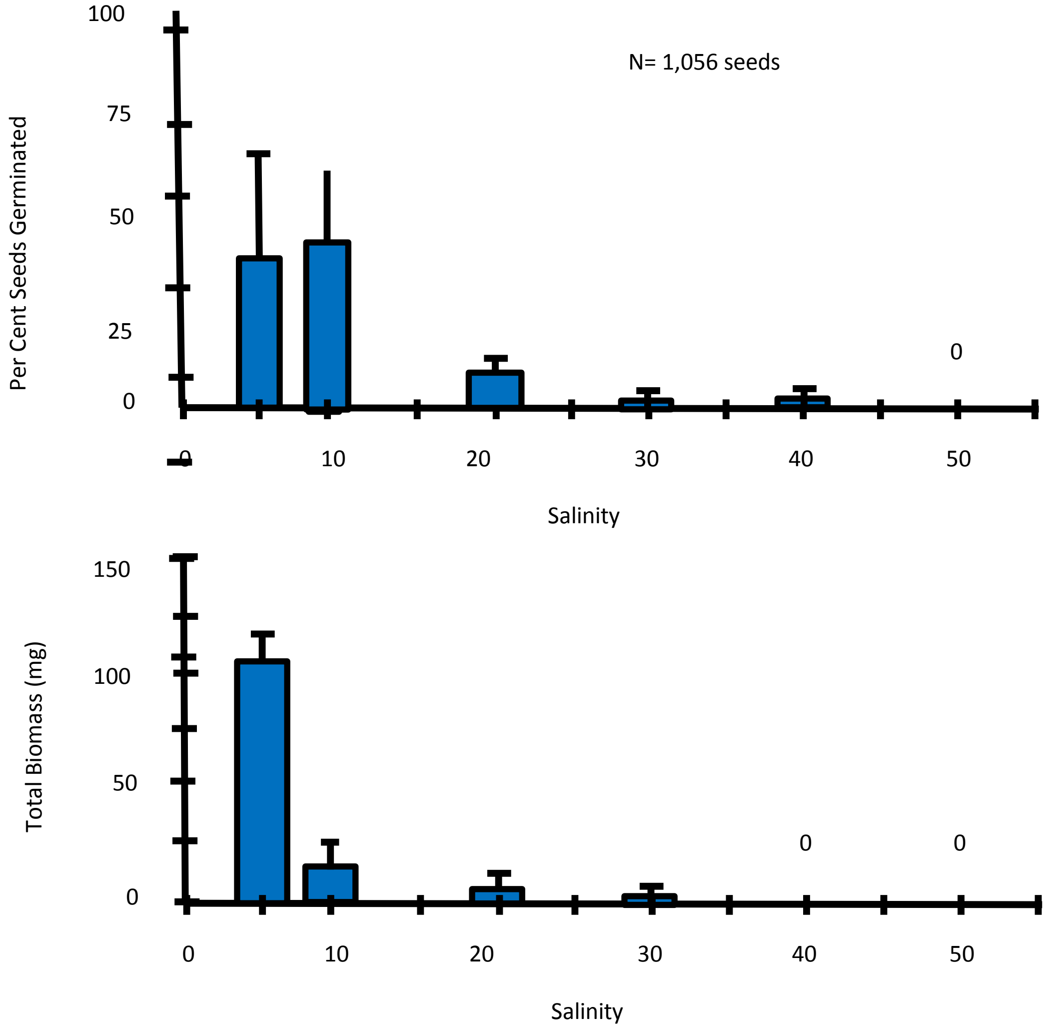


Figure 7-113. ***R. maritima*** **seed germination and seedling biomass at different salinity levels (from Strazisar et al. 2013).**

Increased freshwater flow to northeastern and central Florida Bay is expected to lower salinity overall and shift *T. testudinum* dominated SAV beds to *T. testudinum-H. wrightii* dominated. *R. maritima* densities are expected to also increase in coastal embayments as salinity conditions become fresher. In northeastern Florida Bay near U.S. Highway 1, SAV occurrence declined with the 2005–2007 algal bloom event (see **Figure 7-51**). Fluctuating improvements occurred over the past six years and future SAV recovery in this area, as well as SAV dynamics throughout northeastern and north-central Florida Bay are expected to be influenced by C111 SCWP operations.

U.S. Highway 1 Expansion

Over the last decade, the Florida Department of Transportation expanded the 18-mile stretch of U.S. Highway 1 from Florida City to Key Largo (**Appendix 7-1**, **Figure A7-1-7)** and its hydrologic and ecologic effects will likely influence the interpretation of effects from the C-111 SCWP. Mitigation efforts were incorporated into the project in attempts to mimic a more natural hydrologic connection between northeastern Florida Bay and southern Biscayne Bay. This mitigation effort included the enlargement and/or creation of 14 box culverts under the roadbed and the construction of a bridge, which replaced a box culvert at the location of Manatee Creek at the Monroe/Miami-Dade County line. This work was conducted along a 5.4-mile section of U.S. Highway 1 from the Cross Key portion of Little Blackwater Sound north to the intersection of the C-111 Canal and U.S. Highway 1. Work on this project was finished by the third week of January 2008. This section assesses the effects on the hydrology and ecology from the road construction project.

Annual salinity and water level at Audubon Florida study sites situated near the construction area in the years prior to and after construction of the culverts and bridge were analyzed to determine if this mitigation project had an effect on local hydrology at these locations. For the southern Biscayne Bay region, the Manatee Bay (MB) and Barnes Sound (BS) sites were used as 'impact' sites and the Card Sound (CS) and Turkey Point (TP) sites were used as 'control' sites due to proximity near and away from construction area (**Figure 7-116)**. For the northeastern Florida Bay region, the Highway Creek (HC) site

Figure 7-114. Location of sites used to analyze the affects of culverts and bridges along US1.

was used as an ‘impact’ site and the Joe Bay (JB) site was used as a ‘control’ site. The hydrologic year when the bridge and culverts were under construction, 2007–2008, was eliminated from analysis. ANOVA was used to determine if ‘years’ before and after construction had an effect on salinity and water level in comparison to control and impacted sites. Individual ‘year’ interactions ($p < 0.01$) were investigated through a Tukey’s Honest Significant Difference post hoc test.

In the northeastern Florida Bay region, post hoc tests of ANOVA revealed that ‘years’ before and after construction had a slight effect on annual salinity at the Highway Creek site in comparison to the control site at Joe Bay ($F_{(8,\ 6554)} = 1.55$, $p = 0.14$). Salinity was significantly higher at the Highway Creek site after expansion of U.S. Highway 1 (**Figure 7-117**).

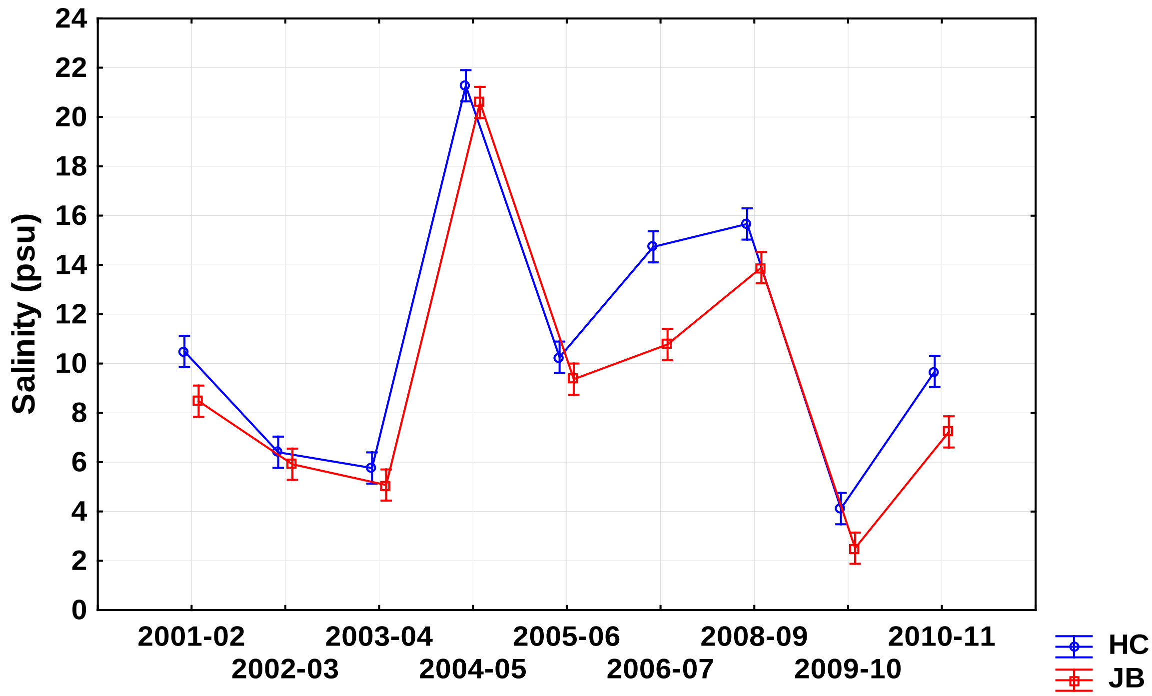


Figure 7-115. Comparison of annual mean salinity (± standard error) at the impacted Highway Creek (HC) site and the control Joe Bay (JB) sites before (2003–2007) and after (2008–2011) construction along the 18-mile stretch section of U.S. Highway 1.

Anecdotal observations within Long Sound indicated that water clarity within this body of water also was significantly greater in the years following the construction of the culverts and dams. This is assumed to be a result of the increased Atlantic tidal and wind driven flow that has resulted with the enhanced connection between Long Sound and southern Biscayne Bay. Possibly in response to increased water clarity at our HC6 (Highway Creek) SAV monitoring site located in Long Sound (**Figure** 7-116), coverage of *H. wrightii* at the site increased from a ten-year prior mean of approximately 50% to near 100% since the construction project was completed. Coverage has remained near 100% since that time. Concurrently, there has been no change in the abundance of *H. wrightii* at our JB6 (Joe Bay) monitoring site located in Florida Bay in Trout Cove. Since construction was completed, coverage of *H. wrightii* at this site has remained near the period of record average of about 50%.

The higher salinity at the Highway Creek sampling site and increase in the amount of *H. wrightii* at the HC6 SAV monitoring site indicated that there could be a change in the fish community at the Highway Creek prey base fish sampling site. An analysis of fish density at the Highway Creek, Joe Bay, and Taylor River sites indicates a decrease in response to freshwater events at Highway Creek after 2008 compared to years prior to construction. **Figure 7-118** shows that generally the Highway Creek site has a similar pattern of fish density compared to the Joe Bay and Taylor River control sites. Increases in density at Highway Creek occur during the same time periods as increases at either Joe Bay and/or Taylor River, although the increase is not as pronounced at Highway Creek (i.e., 1999 and 2005). These increases are normally in response to periods of very low salinity at these sites. However, after the U.S. Highway 1 construction, Joe Bay and Taylor River experienced record fish density levels in response to the freshwater event of WY 2010–WY 2011. Six additional fish sampling sites in Taylor Slough and SRS that were not included as part of this U.S. Highway 1 analysis also experienced record fish density levels. In comparison, fish density at Highway Creek showed virtually no response to this event and density levels have remained relatively low and steady since construction was completed (**Figure 7-118**). Similar results were observed for fish biomass. So, it appears that the conditions found throughout the mangrove habitats west of the Highway Creek were not realized at Highway Creek in WY 2010–WY 2011 probably as the result of increased salinity stemming from increased Atlantic tidal flow through the bridges and the culverts, thereby culminating in lowered freshwater prey fish production at this site compared to lower salinity sites. The fish community at Highway Creek in northeastern Florida Bay is now more characteristic of the fish community in southern Biscayne Bay.

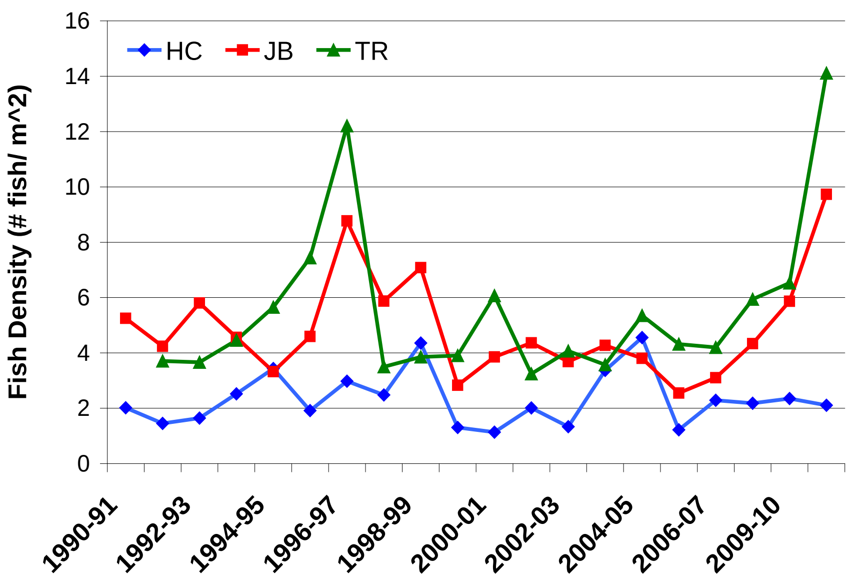


Figure 7-116. Comparison of annual mean prey base fish density in number of fish per square meter at the control sites, Joe Bay (JB) and Taylor River (TR), and the impacted site, Highway Creek (HC), before (1990–2007) and after (2008–2011) construction along the 18-mile stretch section of

In the southern Biscayne Bay region, ANOVA revealed that water level increased after expansion of U.S. Highway 1 (2008–2010) compared with water levels before the expansion (2003–2007) ($F_{(17,\ 9831)}$ = 5.37, $p < 0.01$). Post hoc tests revealed that, compared to control sites and especially the Turkey Point (TP) site, water level tended to be higher at the Manatee Bay (MB) site after the U.S. Highway 1 expansion (**Figure 7-119**). Post hoc tests revealed there were no differences in annual water level between years at the Barnes Sound (BS) site in comparison to the control sites. The Barnes Sound site is located farther from the construction zone than the Manatee Bay site, which may have affected these results. Post hoc analyses of the ANOVA, which test changes in annual salinity (pre- versus post-expansion of U.S. Highway 1), found no significant differences in salinity between control sites and the impacted sites (results not shown). It is possible that no changes in salinity profile were detected at these two sites after construction because they are located too far upstream into the wetlands to be affected.

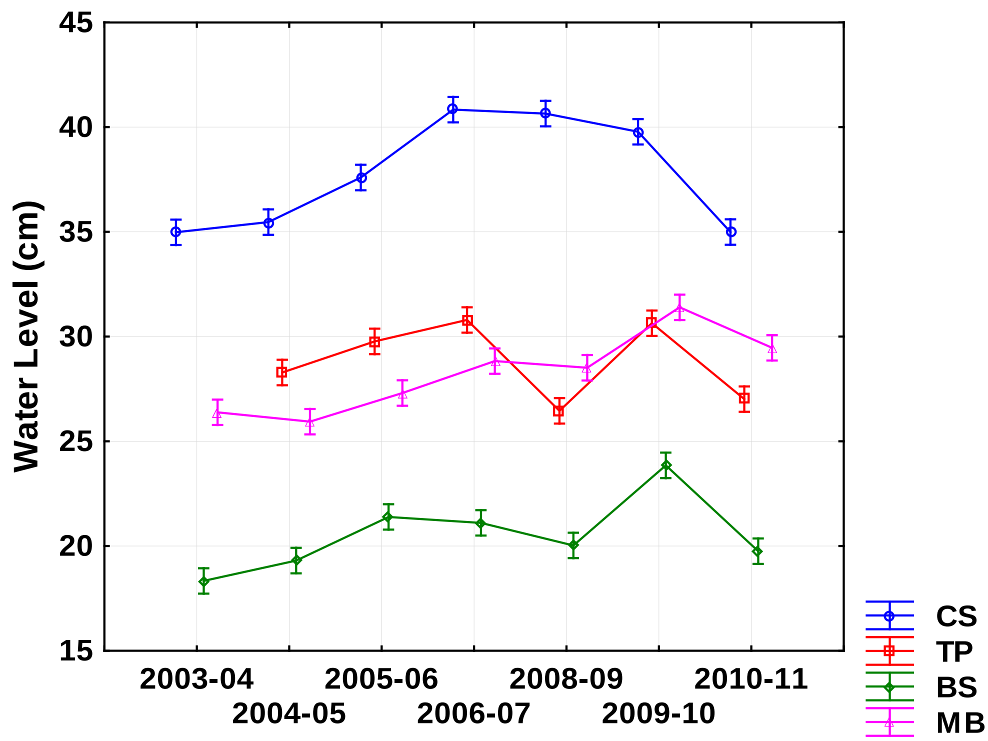


Figure 7-117. Comparison of annual mean water levels (± standard error) at impacted, Barnes Sound (BS) and Manatee Bay (MB), and control, Card Sound (CS) and Turkey Point (TP), sites before (2003–2007) and after (2008–2011) construction along the 18-mile stretch section of U.S. Highway 1.

Although there was a moderately significant effect on the annual water level at the Manatee Bay site from before and after construction, the fish community did not seem to be affected. Neither density nor biomass at Manatee Bay or the other sites found in the southern Biscayne Bay watershed were altered due to the construction of the U.S. Highway 1 culverts. In general, the control sites, Turkey Point (TP) and Card Sound (CS) had higher percentages of oligohaline species and lower percentages of mesohaline species than the impact sites, Manatee Bay (MB) and Barnes Sound (BS) for both the before and after construction periods. Rainfall differences between the before and after construction period were not considered and may play a significant role.

C-111 Spreader Canal Western Project Conclusion

The C-111 SCWP has only been in operation since mid-summer 2012 so monitoring results are not yet conclusive and cannot indicate how well the project is performing. In addition, the lack of focused monitoring in the C-111 SCWP area makes it challenging to set the needed baselines and select key restoration indicators for comparisons of ecosystem components before and after the expansion of U.S. Highway 1. This assessment summarizes the baseline and one year of C-111 SCWP operational data and suggests restoration benchmarks to be compared with project results in future SSRs to measure expected restoration outcomes. Preliminary results indicate that the C-111 SCWP features are working to adjust flows in the water management system as designed. The next SSR reporting will include an additional four to five years of with-project operations data and should indicate whether these water management flow changes result in the desired restoration outcomes as described in the project implementation report (USACE and SFWMD 2011).

Picayune Strand Restoration Project

Picayune Strand Restoration Project Description

The PSRP is a component of CERP that is designed to restore hydrological and ecological function to SGGE and its surrounding public conservation lands (**Figure 7-120**). SGGE was once a portion of a large southwest Florida residential development initiated in the 1960s with the creation of over 270 miles of roads and 48 miles of major canals that were directly connected to the Gulf of Mexico. In addition, cypress logging in the 1940s and 1950s left about 35 miles of raised railroad beds within the boundary of SGGE that were impeding natural flows through the area. Subsequent to the completion of the road and canal system within the northern and southern portions of Golden Gate Estates, many private residences were built between 1970 and 2000. Most of the residences were built in Northern Golden Gate Estates (NGGE), although about 140 structures were located in SGGE.

The primary objective of the PSRP is to establish the pre-development hydrologic regime, including wet and dry season water levels, overland sheet flow, and hydroperiod. Hydrologic restoration of SGGE involves filling at least 50% of the length of the larger canals and several smaller ditches draining the area and eliminating impediments to reestablishing sheet flow by removing raised roads, and logging trams (**Figure 7-121**). In addition, all spoil along the roads, logging trams, and canals will also be brought down to natural grade, and any excess will be placed in the canals. Residences within SGGE have been removed and any associated raised land surfaces will be brought down to natural grade. While the hydrologic regime of most of the lands in SGGE will be returned to their pre-development condition, current levels of drainage must be maintained in NGGE. This has required the construction of three large pump stations on the three western canals at the north end of SGGE. The pump stations will maintain existing levels of drainage in NGGE, while sending water from the canals downstream as sheet flow across SGGE.

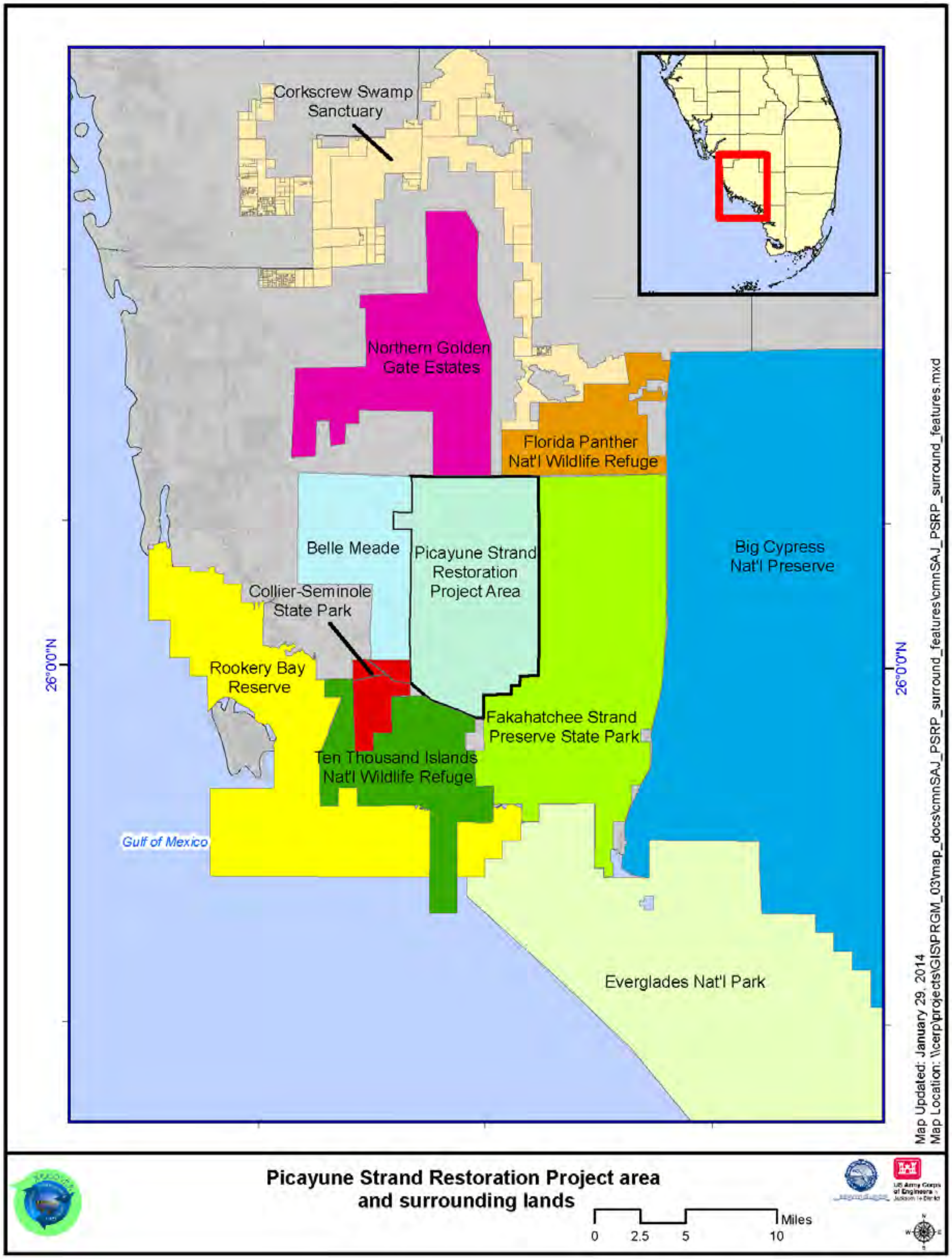


Figure 7-118. PSRP area and surrounding lands.

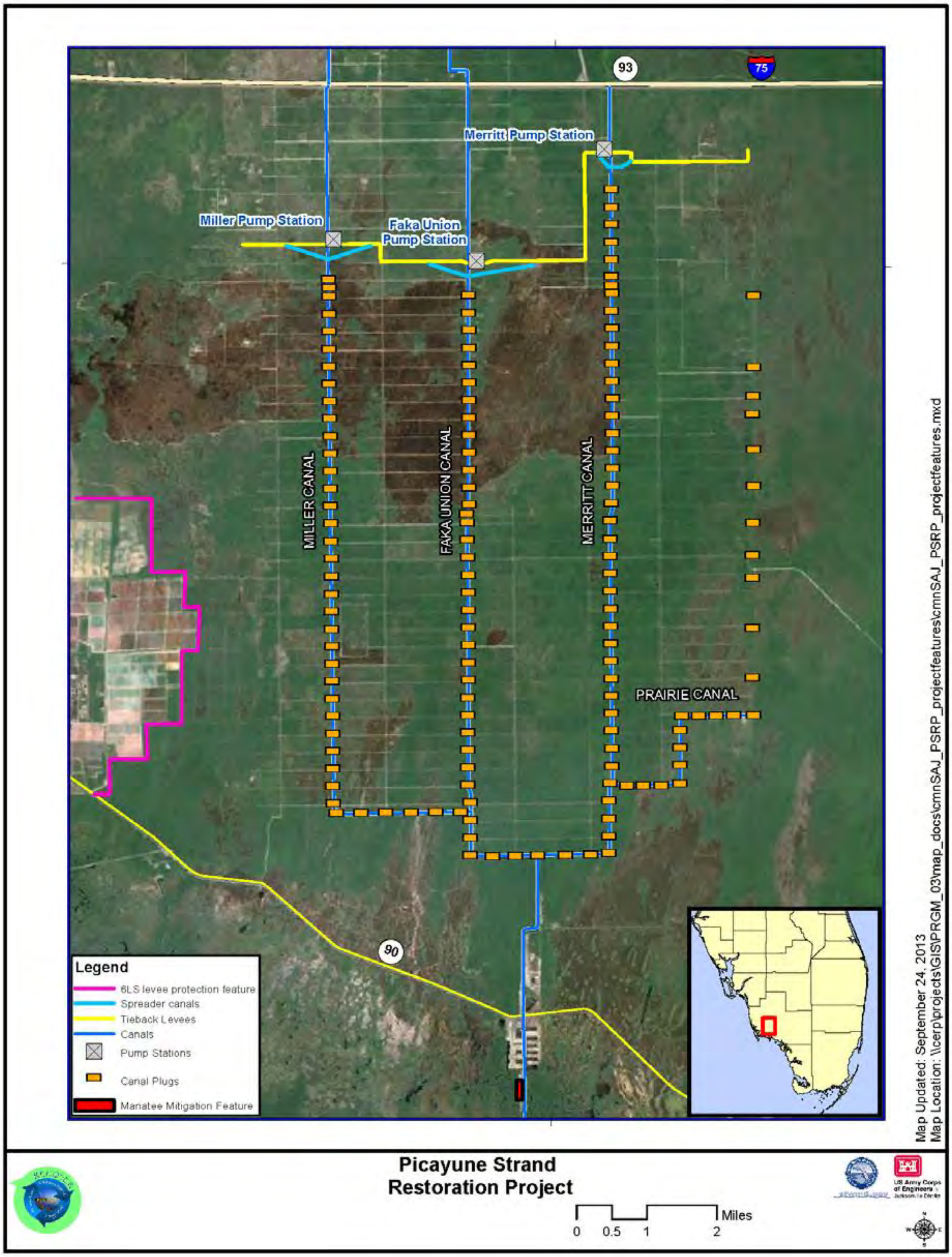


Figure 7-119. The major features of PSRP.

Although there were numerous bridges under Tamiami Trail, water flows from Picayune Strand to the TTI along the Gulf of Mexico have been impeded by Tamiami Trail since the 1920s. With the development of Golden Gate Estates in the 1960s, all of the surface and groundwater flows from SGGE and nearby lands to the east and west that originally flowed as sheet flow to the estuaries were captured by the canal system and routed as a point discharge to Faka Union Bay. One component of the PSRP involved reestablishing the original flowways across Tamiami Trail by identifying natural flowways and constructing additional conveyance under the trail in these locations. In addition, plugs were placed in the canal along the upstream side of the trail to reestablish the original watershed boundaries. The second component was reestablishing natural sheet flow across Picayune Strand and its downstream estuaries as described above with the elimination of the erratic and sometimes very large freshwater point discharges into Faka Union Bay. This would also reestablish more natural salinity gradients in the brackish wetlands and coastal estuaries.

Picayune Strand Restoration Project Status

The first canal filled was Prairie Canal, the easternmost canal in SGGE (**Figure 7-121**), which did not require construction of a pump station. The upper two miles of Prairie Canal was filled with adjacent spoil in spring 2004. The lower five miles of the north-south portion of Prairie Canal was plugged in fall 2006, with substantial amounts of additional fill added from fall 2006 through June 2007 as the roads east of Merritt Canal were degraded. Additional degrading of spoil along the roads east of Merritt Canal and along Prairie Canal was done in winter–spring 2012. The logging trams east of Merritt Canal were degraded during winter–spring 2011.

Road removal began in spring 2010 on land between Merritt and Faka Union canals and was substantially complete by December 2012, although some work remained in the area around the Merritt Canal pump station. Logging trams in this phase were degraded in winter–spring 2011. The Merritt pump station is scheduled to be operational starting in winter 2013–2014, and Merritt Canal is scheduled to be plugged in spring 2014 down to where it joins Prairie Canal.

Road removal began in fall 2011 in the area between Faka Union and Miller canals, and as of summer 2013, most of the work has been completed. Degrading of the logging trams in this area was done from summer 2011 through winter 2012. The Faka Union pump station is scheduled to be operational in summer 2014. However, timing of plugging the Faka Union Canal down to where it joins Miller Canal will be dependent on a hydrologic analysis currently being done on the potential effects on private lands to the west of SGGE.

Work on the lands west of Miller Canal commenced in winter 2013–2014 and is expected to be complete in 2017. When the north-south portion of the Miller Canal is plugged, the East-West Stair-Step Canal from Prairie Canal west to Miller Canal will also be plugged, leaving only the portion of Faka Union Canal open south of where all four of the SGGE canals come together. Most roads north of the tieback levee will be removed, as will spoil along the canals and any remaining roads slated for retention. Protection levees or other structures needed to protect adjacent private lands will also be built. A manatee mitigation site will be located in the vicinity of the Port of the Islands community along the south end of the Faka Union Canal below Tamiami Trail.

As of the drafting of this document, the project requires additional authorization and appropriation by congress to complete: plugging of the north-south portion of Miller Canal, plugging of the East-West Stair-Step Canal from Prairie Canal west to Miller Canal, road removal north of the tieback levee, spoil removal along canals and remaining roads, protection levees/features for private lands, and the manatee mitigation site located in the vicinity of the Port of the Islands community. Additional information can be found in the PSRP *Draft Limited Reevaluation Report and Environmental Assessment* (USACE 2013).

Picayune Strand Restoration Project Operations Plan

The operations plan for the PSRP pump stations is still being developed, and will continue to be modified as each additional pump station comes on line. However, there are three primary components of the operating plan: routine day-to-day operations, emergency operations, and structure maintenance operations. Structure maintenance operations would occur during routine scheduled maintenance and repairs to the pump stations.

Routine day-to-day operations involve maintenance of target water levels at the I-75 bridges over each canal, although the location of where these targets will be measured may be moved to just above the pump stations for easier access. These target water levels have been in place for years, and have been maintained by a series of weirs designed to limit excessive drainage by the Golden Gate Estates canal system. There are two water levels targets, one for wet season water levels and another for dry season water levels, and the pumps on each canal will be operated to maintain these water levels. The dates on which the water level targets switch from wet season to dry season targets and vice versa can vary from year-to-year depending on when the summer rains begin and end, the goal being to retain as much water in the system as possible without flooding residential areas. However, the elevation of the wet and dry season targets are inflexible because increasing it would not maintain existing levels of drainage in NGGE, and decreasing it would increase drainage of remaining wetlands in NGGE. Actually, while wet or dry season water levels are being maintained post-restoration, flows into Picayune Strand will be operating on a slightly skewed rain-driven regime, since when there is more rainfall, there will be more flow in the canals, and when there is less rain, there will be less flow in the canals. It will be slightly skewed because more water will be coming downstream into Picayune Strand in the wet season to maintain drainage in NGGE and less water will be coming down during the dry season because of the loss of wet season surface and groundwater storage in NGGE.

Emergency operations are primarily associated with tropical storms. The current standard procedure when the arrival of a hurricane is imminent is to lower water levels in canals beginning about three days before landfall. This would not be a potential problem for Picayune Strand restoration when a high-rainfall storm passes through the area or the event occurs well before the end of the summer wet season and there is time to refill the upstream watersheds. However, when a late wet season hurricane does not produce much rain or misses the area where water levels have been lowered, this could lead to undesirably low water levels going into the dry season. Since there will be high capacity pumps available at Picayune Strand, it has been suggested that the decision to lower water levels in the canals could be postponed until one day before a hurricane landfall, at which time the location of landfall and severity of the storm would be more certain. A drawback to the shorter and more rapid canal drawdown is the possibility of producing higher water levels and more stressful flows rates downstream of the pump station than would occur with the three-day drawdown.

Picayune Strand Restoration Project Monitoring Assessments

Water Quality

Water quality has been measured in the Golden Gate Estates canals for decades, and no significant problems have been documented in the PSRP area. As a result, water quality improvement was not an objective of the PSRP. However, for a number of years after the pump stations become operational, water quality will be monitored in the canals upstream of the pump stations and at the downstream end of the project above Tamiami Trail. Water, sediment, and fish samples will also be taken at these sites looking for possible mercury contamination.

Water Levels

Water level monitoring at 23 wells in Picayune Strand north of Tamiami Trail commenced in October 2003, and at three additional wells in brackish marshes south of Tamiami Trail in November 2006 (**Figure** 7-122). Monthly water level monitoring has also been conducted since 1987 at 24 wells along two transects across Fakahatchee Strand by Fakahatchee Strand Preserve State Park staff.

Hydrologic restoration began when most of Prairie Canal was filled in fall 2006. Prior to 2006, water levels at four of the Picayune Strand wells near and between Merritt and Prairie canals fluctuated in a similar range around each other (**Figure 7-122**). Prairie Canal was filled in fall 2006, after which water levels in the wells near Prairie Canal and between the two canals were consistently higher than those in the well near the unfilled Merritt Canal; usually by about 2 feet during the dry season and by 1–2 feet in the wet season. The main flow way through Picayune Strand enters from the north at Merritt Canal, arcs to the east near Prairie Canal midway through the project area, and then turns back west to cross Tamiami Trail at the Faka Union Canal. Thus, water levels in the well near Prairie Canal are expected to be substantially higher during the wet season following the plugging of Merritt Canal. Dry season water levels should also be somewhat higher, since 26 years of monitoring data from a 6.5-mile transect across Fakahatchee Strand show that dry season water level drawdowns can extend two to three miles from a SGGE canal, and the well near Prairie Canal is approximately two miles east of Merritt Canal and only 1.25 miles north of the unfilled East-West Stair-Step Canal. Water levels in a well located about one mile from Prairie Canal in Fakahatchee Strand had fairly consistent high wet season water levels from 2003 through 2012 and a range of annual water level fluctuation of about 5 to 6 feet (**Figure 7-123**). Its seasonal water levels were consistently higher than the other three wells, even after Prairie Canal had been mostly filled. It is expected that following the plugging of Merritt Canal, water levels in the three Picayune Strand wells will fluctuate within the same range as the Fakahatchee Strand well.

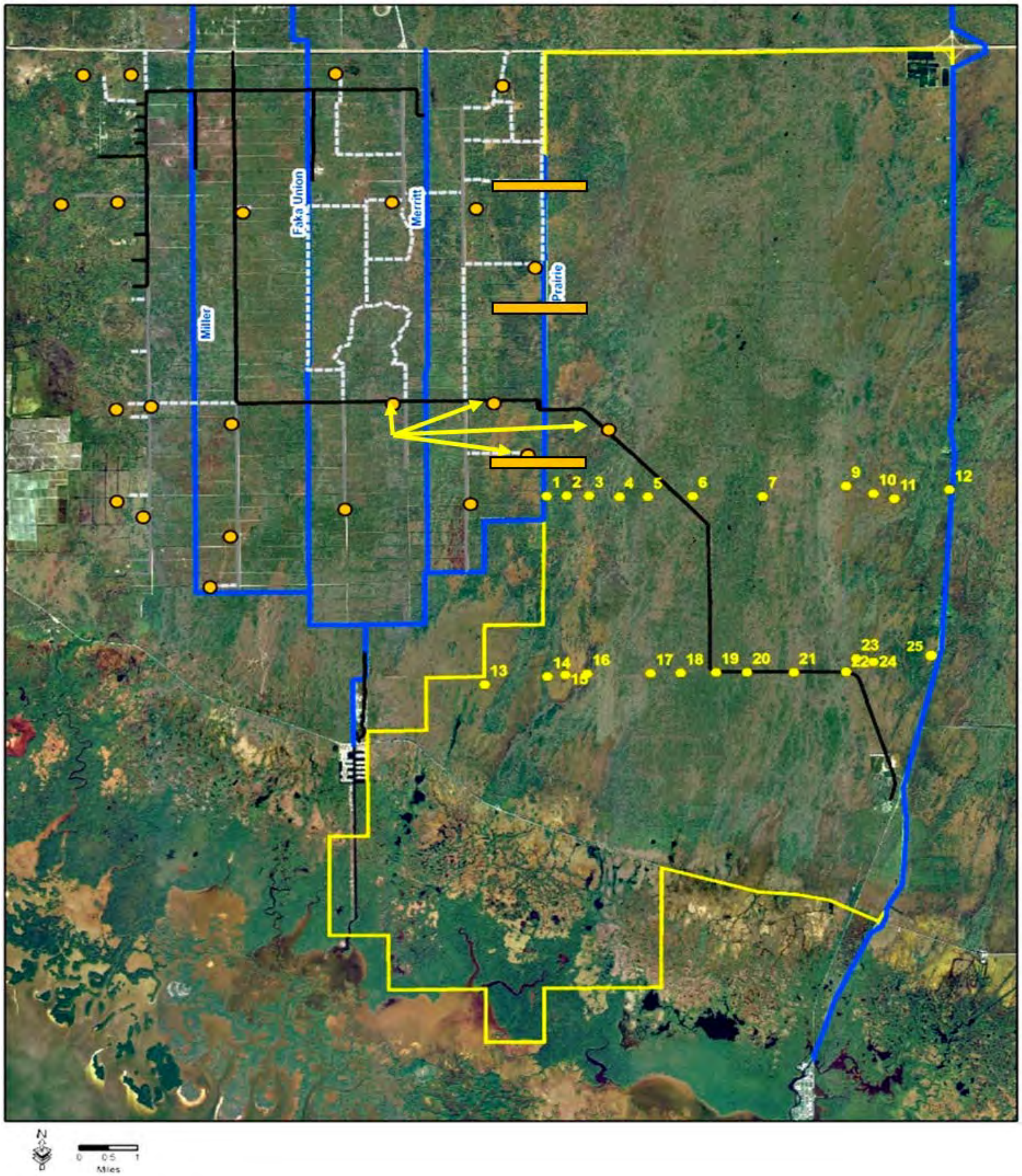


Figure 7-120. Monitoring wells within the PSRP (orange dots) and Fakahatchee Strand Preserve State Park (yellow dots). Also indicated by the arrows are four wells for which water levels from October 2003 through January 2013 are shown in Figure 7-123. The location of the 2-km vegetation transects across Prairie Canal are also shown (orange bars).

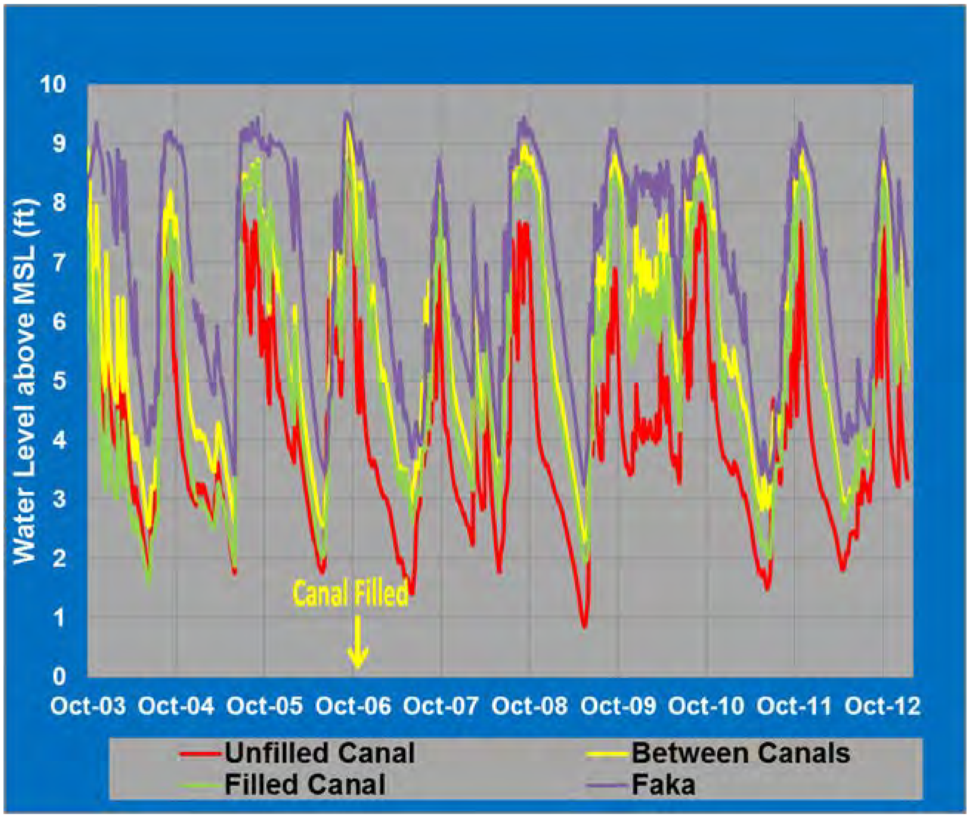


Figure 7-121. Water levels in four wells (see Figure 7-122 for well locations) next to filled and unfilled canals, approximately midway between a filled and unfilled canal, and about a mile into Fakahatchee Strand from the easternmost PSRP canal.

Water level data from the northern well monitoring transect across Fakahatchee Strand shows four distinct water level periods associated with four distinct types of management of Prairie Canal between 1987 and 2012 (**Figure 7-124**). The first period from 1987 to about 1992 was when Collier County was responsible for management of the Golden Gate Estates canals, and was doing little aquatic weed control. During this period, wet season water levels were typically about 1 foot below those in the interior of Fakahatchee Strand and dry season water levels would drop to about 4 feet lower. The second period was from about 1992 until the late 1990s when the responsibility for management of the canals was transferred to the SFWMD and they instituted a very effective aquatic weed management program. During this period, wet season water levels were about 4 feet lower than the interior of Fakahatchee Strand and dry season water levels were about 5.5 feet lower. In response to these disturbingly low water levels, the SFWMD installed control structures in the canals to reduce drainage impacts. The third period showed the beneficial effects of the control structures during the wet season, but it also demonstrated how ineffectual the structures were during the dry season when water just passed through the very permeable surrounding bedrock. During this period, wet season water levels were often raised to within about 2 feet of the water level in the interior of Fakahatchee Strand, while dry season water levels remained at about 5.5 feet below those in the interior of Fakahatchee Strand.

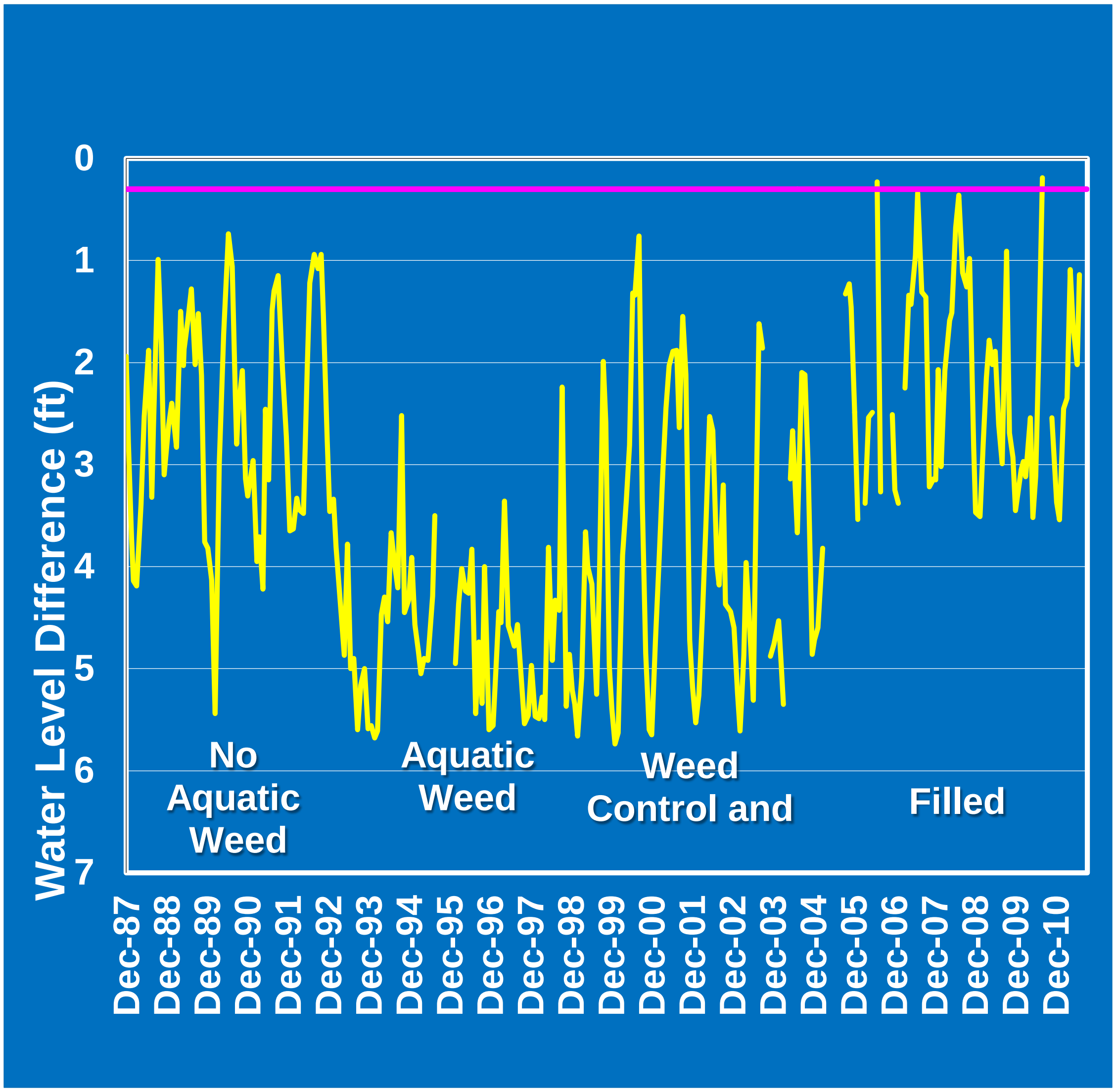


Figure 7-122. Differences in water levels between well 1 next to Prairie Canal and well 6, which was 2.5 miles east of Prairie Canal into Fakahatchee Strand, under different management conditions from 1987 to 2011. The red line at the top of the figure is the difference in ground surface elevations between the two wells.

Of particular interest in this report is the fourth period after most of Prairie Canal had been filled, and water levels rose significantly. During the wet season, they could reach similar elevations to those in the interior of Fakahatchee Strand. However, during the dry season, they were still about 3.5 feet lower than the water levels in the interior of Fakahatchee Strand. This is partially explained by the fact that the Merritt Canal is still not plugged, but probably more importantly by the fact that the East-West Stair-Step Canal is not yet filled and is only 0.5 miles from the north transect well adjacent to Prairie Canal. Hydroperiods have lengthened since Prairie Canal was filled in 2006, despite drought conditions during four of the six years between 2007 and 2012 when there was relatively low rainfall.

Plant Communities

Sampling for the Year 4 post-restoration plant communities in the Prairie Canal and the reference plots was conducted between May and July 2011. Sampling was completed on a total of 36 transects, including 11 reference plots. Ten previously monitored plots were not sampled in 2011 because they burned in a wildfire on May 12. Sampling methods included belt transects, line-intercepts, and quadrats to identify and quantify plants within the canopy, subcanopy, shrub, and groundcover strata. Comparisons were made of data collected in 2011 with data collected during the dry seasons of 2008 and 2009.

Dominant tree species show minimal changes in basal area and density indicative of hydrologic alteration from 2008 to 2011. Changes in this stratum are expected to be slow, with the exception of mortality of species sensitive to inundation. The dual effects of prolonged drought and as yet incomplete hydrologic restoration likely contributed to the lack of changes in this stratum. Density and basal area of *Taxodium ascendens* (cypress) in cypress habitats remains lower at restoration sites than at reference sites, which is thought to indicate increased mortality rates in the drained communities prior to restoration. Relative growth rates were lower for *T. ascendens* and *Fraxinus caroliniana* (pop ash) at restoration transects, while *Pinus elliottii* (pine) growth rates were significantly higher at restoration transects, which is consistent with known ecology of these species. When hydrologic restoration is complete, these trends are expected to reverse.

To date, restoration has not altered the densities of *Sabal palmetto* (cabbage palm), a major nuisance native species in the drained Picayune Strand, with the exception of some mortality of seedlings and young *S. palmetto* in *T. ascendens* plots close to the Prairie Canal footprint. The overall trend for *S. palmetto* density at both reference and restoration plots since 2004 has been a slow and steady increase. This increase may have been exacerbated by drought conditions, although the densities and rates of change vary considerably by habitat and land management regime. High *S. palmetto* densities and rates of increase were recorded in pinelands within both restoration and reference plots. Cypress habitats had high *S. palmetto* densities in restoration plots, but not in reference plots. All wet prairie plots remain low in *S. palmetto* densities. Variability in *S. palmetto* densities from year to year is greatest in the younger strata (seedlings and pre-trunk palms) in cypress habitats, suggesting that fluctuating water levels and varying hydroperiods cause some mortality in the restoration plots.

Differences in overall shrub cover, as measured using the line intercept method, primarily reflected recent fire history. A notable exception is mortality of invasive *Schinus terebinthifolius* (Brazilian pepper) due to flooding in plot PC26, which is adjacent to the canal footprint and thus hydrologically more affected by restoration. Changes in shrub cover are expected to be slow, but eventually species sensitive to flooding should be eliminated from plots at relatively low elevations.

In the cypress plots, wetlands affinity index (WAI; Barry and Saha 2008) values in restoration plots are increasing and converging with WAI values in reference plots, while in the wet prairies both reference and restoration plots have maintained relatively constant WAI values from 2008 through 2011 (**Figure 7-125**). WAI values for pineland plots show greater variability, possibly due to the plots occurring

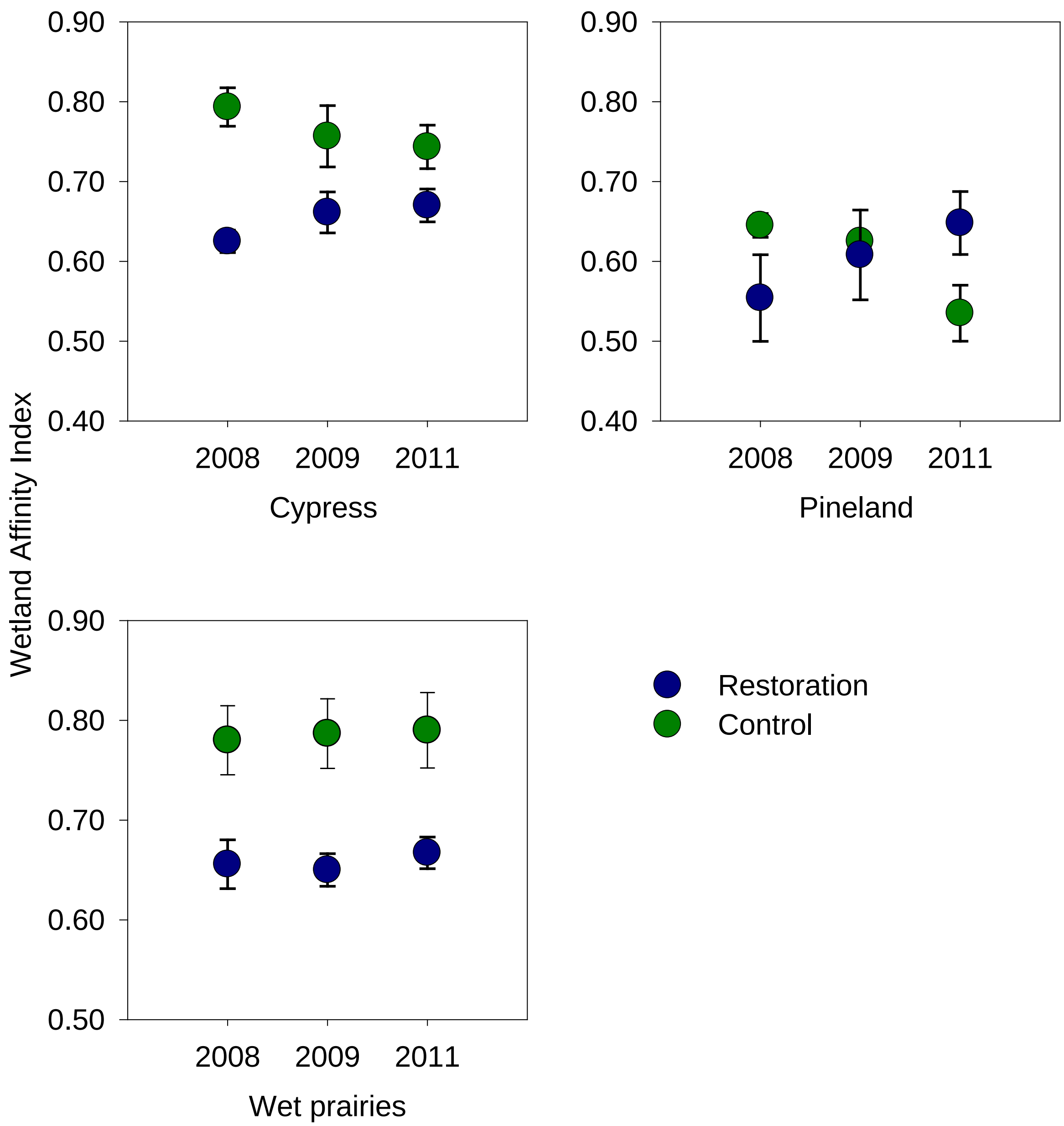


Figure 7-123. WAI by habitat group.

over an elevation range from mesic to hydric pinelands, with reference plot WAI mean values decreasing, and the restoration WAI mean values increasing and surpassing reference values. Considering recent drought conditions, it is encouraging that all restoration plots have exhibited increases in WAI, suggesting hydrologic restoration is having a positive effect on groundcover vegetation. Plots closest to the canal footprint exhibited the greatest increases.

Classification and ordination analyses for most plots show no discernible trend relating to hydrologic restoration (not shown). However, data reveal that groundcover strata in restoration plots adjacent to the Prairie Canal footprint are becoming more similar to the reference plots. These plots, being closest to the canal, were more severely drained. As analyzed separately by habitat groupings, plots in cypress and wet prairie showed greatest similarity to plots within the same management regime (reference versus restoration). Pineland plots also showed similarity within a management regime, particularly for sites that had burned within the past seven years. Plots that had not burned in more than seven years were less similar to the other pineland sites.

Aquatic Macroinvertebrates

A total 20 (64.5%) of 31 impacted wetlands and all of the 11 reference wetlands were inundated sufficiently to produce aquatic macroinvertebrates during the baseline assessment period. A total of 7,123 individual macroinvertebrates representing 182 species were identified from the baseline collections. Aquatic macroinvertebrate community structure was significantly different ($p < 0.05$ or 95% confidence level) between impacted and reference wetlands (**Figure 7-126**). The macroinvertebrate

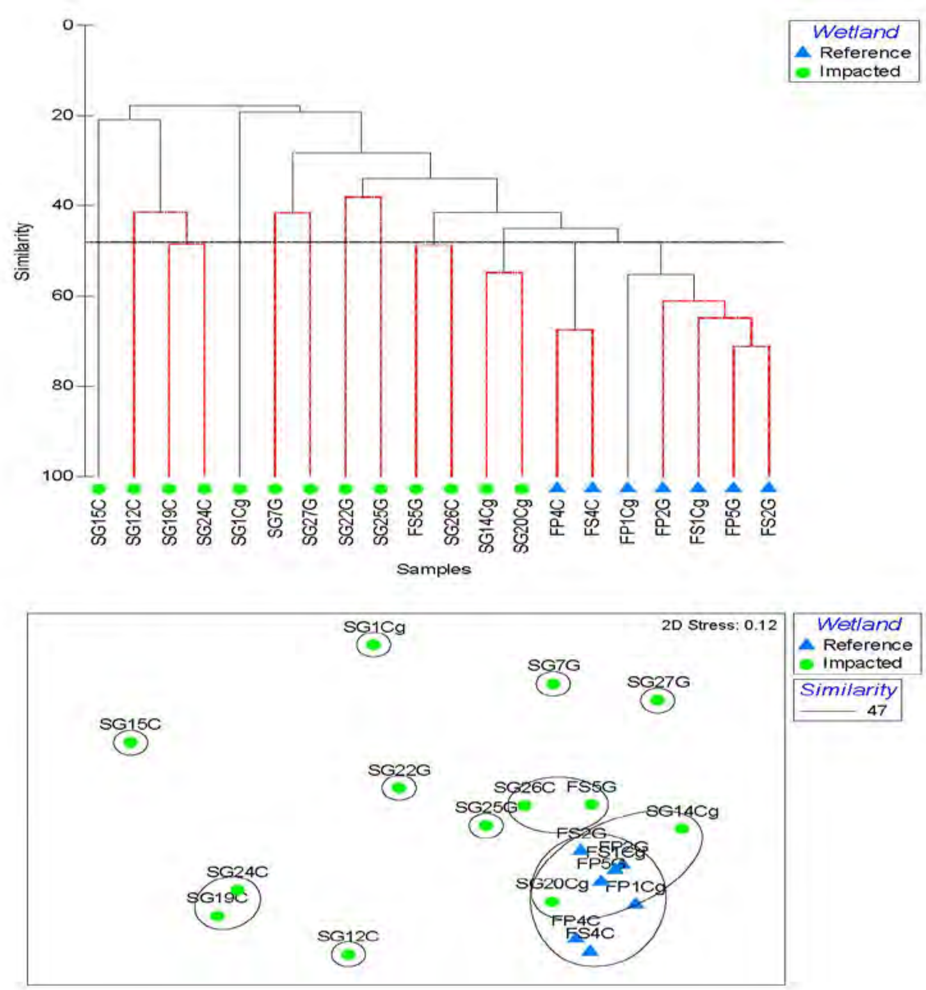


Figure 7-124. Hierarchical clustering (top) and multi-dimensional scaling ordination (bottom) of cypress, cypress-graminoid, and wet prairie macroinvertebrate communities. Slice (top) and overlay (bottom) illustrates groupings at 47% similarity where all reference sites group together.

communities in many impacted sites within cypress, cypress graminoid, wet prairie, marsh, and pine were significantly different from others in that group, especially cypress. With the exception of one hydric-mesic pine site in Fakahatchee Strand (FS3Ph), all reference wetlands of all habitat types, shared greater than 45% similarity in community structure. Six of the ten reference sites (FS1G, FS2C, FP2G, FP5G, FP3Ph and FP6Ph) shared greater than 60% similarity and as a group, were not significantly different from each other but were significantly different from all other groups. In the cluster diagrams, the groups show high percent similarity and in the multi-dimensional scaling ordination plots they appear close together in 2-dimensional space. The macroinvertebrate taxa that were major contributors to the dissimilarity between reference and impacted wetlands included mayflies, *Callibaetis* (7.3%) and *Caenis* (4.4%), the damselfly, *Ischnura* (4.9%), amphipod *Hyalella* (4.0%), and snails *Hatia* (3.4%) and *Planorbella* (2.8%). In addition, several taxa were also identified that may serve as indicators of hydrologic recovery over time, including dragonflies, crayfish, freshwater shrimp, and apple snails because of their life history requirements and relative abundance in reference wetland habitats of the region.

Fishes

A total of 6,381 individual fish representing 25 species were collected by Breder traps during the baseline assessment, including five nonnative species. Seventeen (52%) of 31 impacted wetlands and 10 (91%) of 11 reference wetlands produced fishes during the baseline sampling period. Reference wetlands had more diverse fish communities than impacted wetlands, with a more even distribution of species abundances and percent composition within and between sites. Everglades pygmy sunfish (*Elassoma evergladei*) and warmouth sunfish (*Lepomis gulosus*) were only collected from reference wetlands and may be considered as indicators of restoration success.

Multivariate clustering analysis results show a generally similar pattern as macroinvertebrates (i.e., generally reference sites group closely together and impacted sites do not group closely with each other or with reference sites). Four significantly different groupings ($p < 0.05$) were identified with one large group that included almost all reference sites and several of the impacted sites (**Figure 7-127**). There is a clear break in the group at 52% similarity such that only five impacted sites group closely with eight of the reference sites (hierarchical clustering not shown). These "impacted" sites include one wet prairie site in Fakahatchee Strand (FS5G) and a deep freshwater marsh in Picayune Strand (SG4Mf) along with other sites (SG6Pm, SG7G and SG25G), which were located either on the fringes of Picayune Strand or on local areas that had not been as severely affected by canal drainage as was the rest of Picayune Strand. Reference sites average 58% similarity (42% dissimilarity) of fish community structure, with 7 species contributing to > 93% of the similarity between reference sites. Of the impacted wetlands that contained fishes, there was an average of 45% similarity (55% dissimilarity). Five species contributed to > 93% of that 45% similarity between sites; the tolerant and ubiquitous eastern mosquitofish (*Gambusia holbrooki*) contributed the most (>66%).

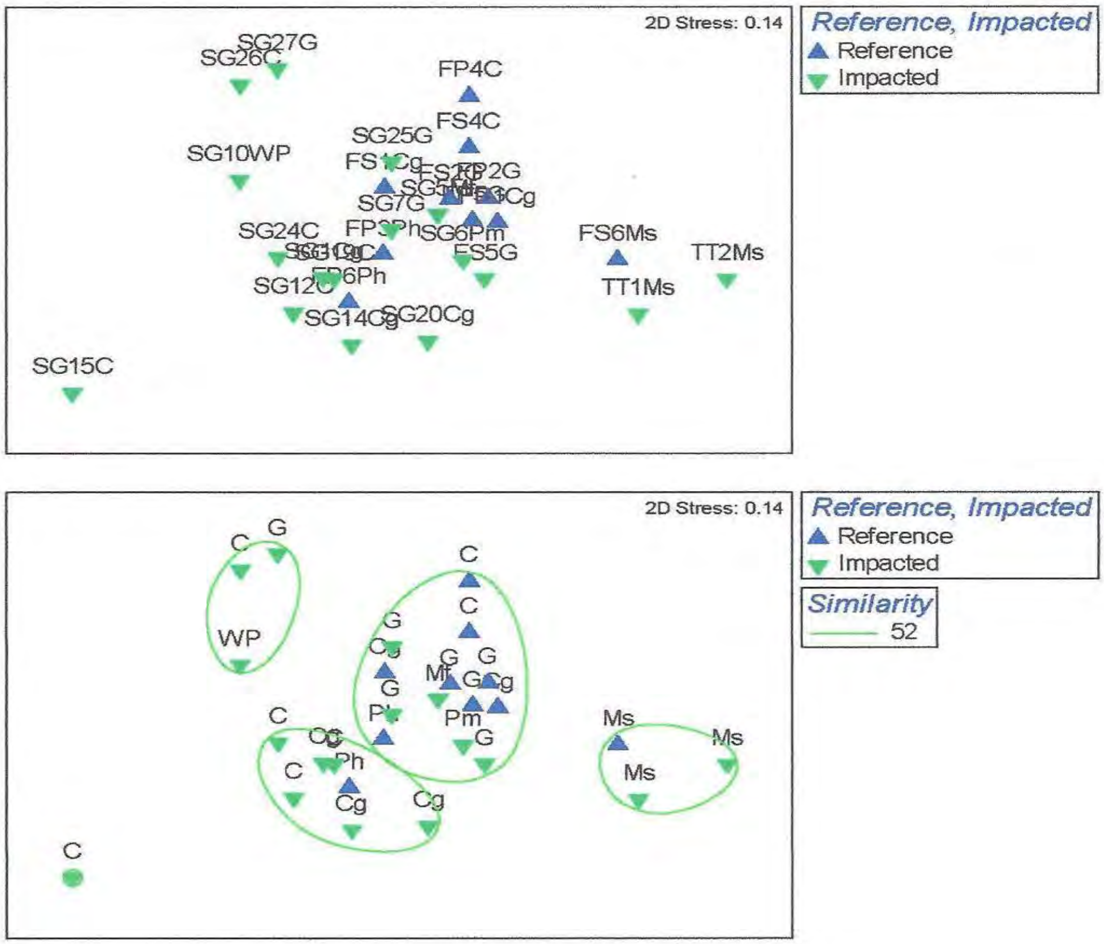


Figure 7-125. Multi-dimensional scaling ordination of fish communities from reference and impacted wetlands for the Picayune Strand baseline assessment. They are labeled by site and reference/or impacted status (top) and by habitat type with a cluster overlay of important groupings.

Treefrogs

Treefrogs were monitored monthly from August 2005 through April 2007 using PVC pipes placed vertically in one or two aquatic fauna sampling sites located close to each of the Picayune Strand water level monitoring wells and in reference sites in Fakahatchee Strand Preserve State Park and Florida Panther National Wildlife Refuge. Cuban treefrogs (*Osteopilus septentrionalis*) were the dominant species collected at the Picayune Strand restoration sites, while green (*Hyla cinerea*) and squirrel (*Hyla squirella*) treefrogs were the most abundant species captured at the eleven reference sites (data not shown). As Cuban treefrogs commonly utilize disturbed areas, it is not surprising that this species has been able to exploit the altered landscape in Picayune Strand. The reference sites were not devoid of Cuban treefrogs and their presence in Fakahatchee Strand Preserve State Park could be attributed to its proximity to Picayune Strand. Furthermore, many exotic species utilize roads as a means for dispersal and Janes Scenic Drive directly connects Picayune and Fakahatchee strands.

Of the three species collected during this study, green treefrogs were more frequently documented at the three coastal brackish marshes (FS6, TT1 and TT2), which was unexpected since Cuban and squirrel treefrogs tend to be more tolerant of brackish waters than green treefrogs. However, none of these sites exceeded the mesohaline range; they were within the oligohaline range during 38% of the sampling events.

Hydrological restoration of Picayune Strand is expected to create conditions more favorable for green and squirrel treefrogs. However, any shift in the populations of these two native species will also be contingent on their ability to compete with a well established population of Cuban treefrogs, which utilize native treefrogs as prey. This transition could take a long time in those portions of Picayune Strand where the native plant communities have been devastated by many years of drainage and fires that have severely altered them. The return of other native species of treefrogs would also be expected, even if only in low numbers compared to the populations of squirrel and green treefrogs.

Summary

While there are portions of the PSRP area that have already shown significant hydrologic improvement, most of these improvements to date are near Prairie Canal and in Fakahatchee Strand to the east of SGGE. Based on 26 years of monthly water level monitoring documenting the effects of Prairie Canal across Fakahatchee Strand, it is known that the effects of the SGGE canals can extend 1 to 1.5 miles from the canal during the wet season and 2 to 3 miles during the dry season. The SGGE canals are only 2 miles apart. Thus, even though Prairie Canal has been mostly filled since 2007, lands to the west of the canal are still being affected by drainage from Merritt Canal, and lands to the south are still being affected by the unfilled East-West Stair-Step Canal. In addition, the main flow-way through SGGE enters from the north in the vicinity of Merritt Canal, and arcs close to Prairie Canal in the north-south middle of the project area before curving west again to where the Faka Union Canal crosses Tamiami Trail. Thus, when Merritt Canal is filled, additional hydrologic restoration is expected along Prairie Canal, and there will be additional improvement in the southern portion of the area when the East-West Stair-Step Canal is plugged. While plugging Prairie Canal has resulted in preliminary restoration of cypress swamps, many of the other faunal indicators have not yet responded to the observed hydrologic and vegetative changes. No benefits have been observed in the estuarine areas from the newly created overland flow as most of it is recaptured by the East-West Stair-Step Canal and routed as point discharge into Faka Union Bay via Faka Union Canal.

EFFECTS OF THE 2010 COLD EVENT ON EVERGLADES BIOTA

Continuous and long-term monitoring efforts throughout the Everglades system not only allow the building of robust time series that can help better understand how the ecosystem 'works', and thus track its response to water management decisions and restoration efforts, but also affords an opportunity to detect the effects of other important drivers of ecosystems, such as climatic extremes. Climatic extremes are defined as statistically rare events that result in conditions that abruptly and substantially exceed the acclimation capacity of organisms. They often trigger 'punctuated killing events' for vulnerable species, and these can dramatically alter the structure and functioning of ecosystems, and at times even push ecosystems into novel trajectories outside their normal bounds.

Why are climatic extremes important in general and to CERP in particular? Global climate change is predicted to not only result in shifting trends in average conditions (i.e., increases in global temperatures), but also to increase the frequency and intensity of climatic extremes (i.e., droughts, hurricanes, and cold events). These predicted changes in climatic extremes are expected to have profound ecological impacts, yet studies on the impacts of climate extremes lag well behind those of gradual climate change, and opportunities to track their effects on biota are rare. Of particular interest is understanding the combined effects of climatic extremes with other underlying chronic ecosystem stressors (i.e., eutrophication). The 2010 cold event provides us with an opportunity to examine the effects of an extreme cold event on Everglades biota. For CERP assessments, climatic extremes may complicate the interpretation of Everglades restoration responses by causing significant perturbations in the monitored parameters of key indicators, as described below.

Severity of the 2010 Cold Event

In January 2010, a record negative measurement of the Arctic Oscillation allowed for the cold air of the jet stream to be deflected further south than usual, resulting in unusually frigid temperatures that reached south Florida and the Everglades. South Florida experienced extremely low temperatures, including a record low at Key West of 6 degrees Celsius (° C), the second lowest temperature since 1873.

In Miami, air temperatures reached 1.7° C, but dipped below -2° C at the Royal Palm Station in ENP and below -3° C in Everglades City (**Figure 7-128**). Temperatures this low at the Everglades City station were last recorded in winter 1996 (-5.0° C) and winters 1989 and 1940 (-4.4° C). Hydrological stations throughout ENP recorded water temperatures as low as 5 to 6° C in freshwater marshes and coastal habitats (**Figure 7-128**). One station in Highway Creek recorded a low temperature on January 11, 2010 of 1° C (Dr. Jerome Lorenz, personal communication). However, what was most notable about this cold event was its duration. Air temperatures did not go above 11.5° C for a record 12 consecutive days, the coldest period since 1940. Mangrove regions in ENP experienced sustained water temperatures below 15° C for over 2 weeks (**Figure 7-129**). Similarly, inshore reef areas of Biscayne Bay and the Florida Keys experienced sea temperatures below 18° C for 2 weeks.

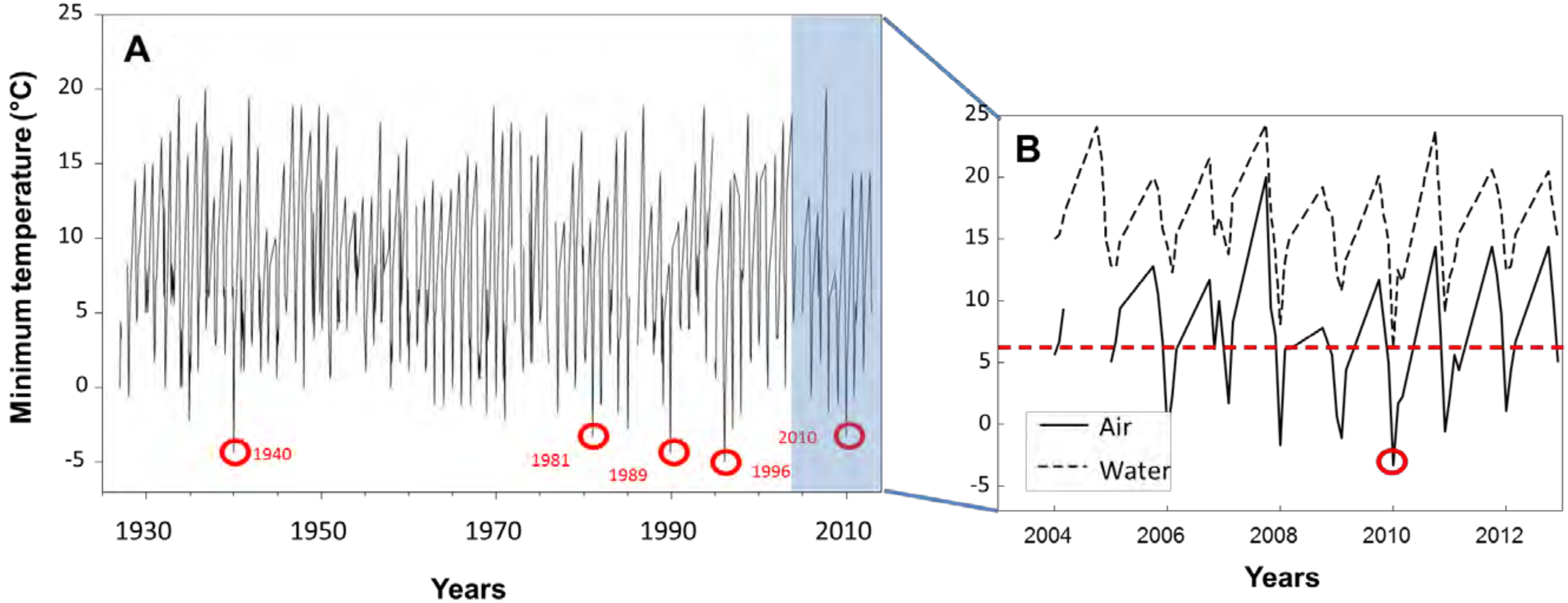


Figure 7-126, Minimum monthly air temperatures (from coldest months of each year: October–March) for the POR, 1927–2012, at NOAA Everglades City station (http://www.ncdc.noaa.gov/cdo-web/)(A). Red circles show previous major cold event events in the time series. Insert shows minimum monthly Everglades City air temperatures (B, solid line) plotted along with minimum monthly water temperatures from USGS hydro station Bottle Creek in ENP (dotted line, http://sofia.usgs.gov/eden/) for 2004 to 2012 (also for coldest months of each year: October–March). Red circle in the insert shows the 2010 cold event, while the red line indicates the lethal limit for a large number of tropical-originated Everglades biota (6° C).

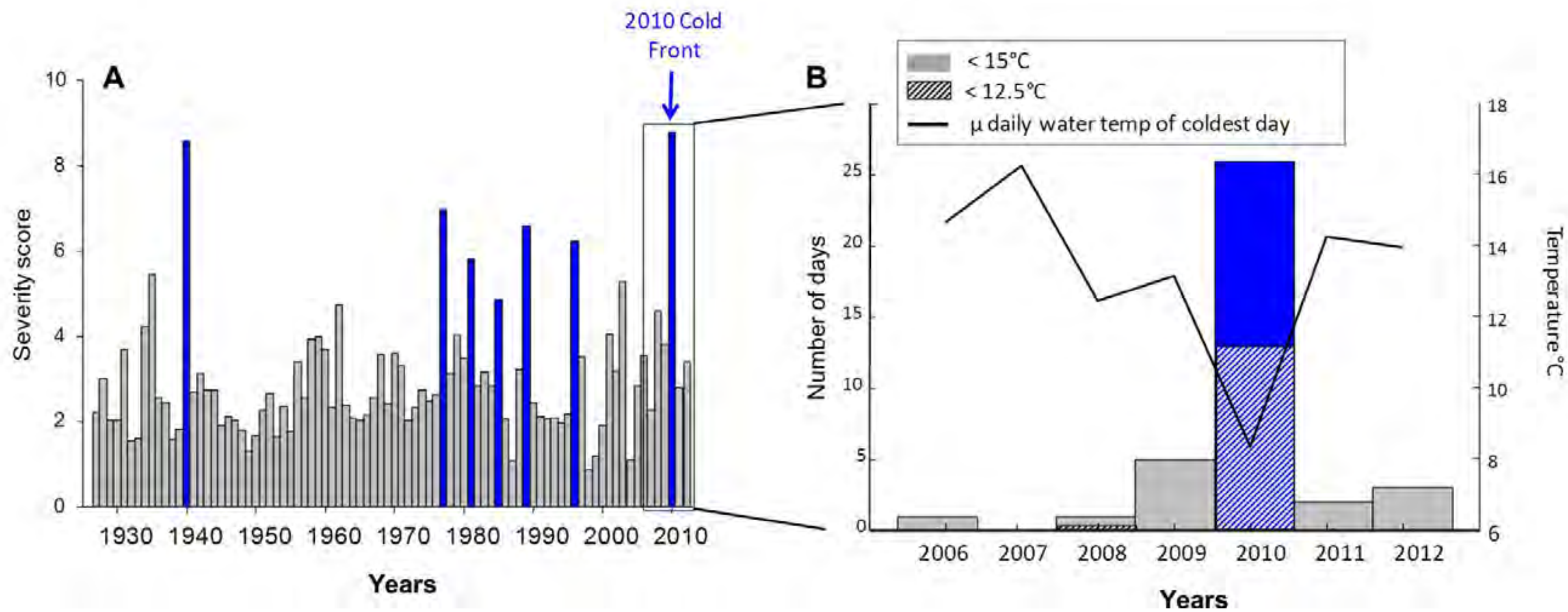


Figure 7-127. Severity score of the most severe cold front of each year for the period 1927–2012, calculated from air temperatures at the NOAA Everglades City station (http://www.ncdc.noaa.gov/cdo-web/) (A). Blue bars indicate cold events known to have caused fish kills in Florida. The insert (B) show the duration of low water temperature conditions at USGS Bottle Creek hydro station in ENP over the past few years (2006–2012, http://sofia.usgs.gov/eden). The average temperature of the coldest day of each year, and the number of days below 15° C and 12.5° C are highlighted, since these temperatures correspond to sublethal and lethal levels for key coastal fishes such as common snook (*Centropomus undecimalis*).

A close look at the 2010 cold event's minimum temperature and duration shows that this cold event was one of the most severe cold events on record (**Figure 7-129**). The history of cold fronts was identified using one of the longest coastal air temperature records in the Everglades region (Everglades City, 1927–2012; http://www.ncdc.noaa.gov/cdo-web/). Since both magnitude and duration are important in dictating the severity of disturbances and climatic extremes, a severity score was developed that incorporated both the length and minimum temperature of cold fronts over this 85-year time period (**Figure 7-129A**). A total of 803 cold fronts were identified between 1927 and 2012. On average, these cold fronts lowered temperatures to a daily minimum of 11.5° C and lasted 5.6 days. The 2010 cold front lasted for 15 days, and brought air temperatures down to a minimum of -3.3° C, with an average air temperature of 2.9° C over its duration. Across these 85 years, eight cold fronts of high severity were identified, with the 2010 cold event having the highest severity score of all (**Figure 7-129**), followed by cold fronts in 1940, 1989, 1977, 1981, and 1996—all known to have caused major fish kills in coastal areas.

Effects of the 2010 Cold Event

The effects of the cold event were particularly strong in the estuarine and marine region of the Greater Everglades, which fall within the RECOVER SCS region, largely due to the tropical and subtropical origin of many resident species. Florida is the northern extreme of the distribution for many of these taxa, and therefore they experience hypothermal stress and high mortality during occasional extreme cold temperature events. Thus, a large number of plant and animal species were impacted by the severity of the 2010 cold event. Previous research documented mortality and frost damage on mangroves, buttonwood stands and hardwood hammocks, as well as wetland and pineland vegetation as a result of the 2010 cold event and of similar previous cold events (Ross et al. 2009, Hallac et al. 2010). Mangrove damage was also noted above in the BBCW Project area. Similarly, researchers and NPS personnel documented mortality effects on butterflies, corals, West Indian manatees, bull sharks (*Carcharhinus leucas*), and nonnative pythons (Hallac et al. 2010, Lirman et al. 2011, Matich and Heuthuas 2012).

Here, effects from the 2010 cold event on estuarine flora and fauna within the footprint of the SCS region are described, including seagrasses, nonnative snails, coastal native and nonnative fishes, and American crocodiles. Mortality effects were severe on 28 species of native estuarine and marine fishes, including those supporting many economically-important recreational fisheries (Hallac et al. 2010). Impacts were also severe also on many of the nonnative fish species presently abundant throughout South Florida—a potentially positive effect of the 2010 cold event. For example, lethal temperature limits were reached for 14 of the 17 nonnative fishes established in ENP, causing mass mortality of these taxa throughout natural habitats. Canals are known to harbor a large number of nonnative fish species. Temperatures in canals in the region stayed relatively warm throughout the cold event (i.e., above 18° C in the L31 West Canal). The ability of the canals to remain warm over cold events emphasizes their roles as both sources and refuges for nonnative fish species (Schofield et al. 2010, Rehage et al. 2013).

Effect on Nearshore Seagrass in Biscayne Bay

Seasonal surveys of nearshore (< 500 m from shore) benthic habitats of Biscayne Bay were conducted in the dry (January–March) and wet (July–September) seasons in the region between Matheson Hammock and Manatee Bay to document seagrass cover patterns. Nearshore habitats were divided into five 100-m strata at increasing distances from shore (< 100 m, 100–200 m, 200–300 m, 300–400 m, and 400–500 m from shore). Surveys spanned eight consecutive dry and wet seasons between 2008 and 2011 (**Figure 7-130**). The average cover of the three seagrass species over these four years was 22.8% cover (± 18.7 standard deviation). For both *Thalassia testudinum* (20.1% versus 13.2%) and *Halodule wrightii* (6.6% versus 4.4%), cover was significantly higher in the warmer wet season than in the cooler dry season; no seasonal pattern was evident for *Syringodium filiforme* cover (0.8% versus 1.0%, respectively).

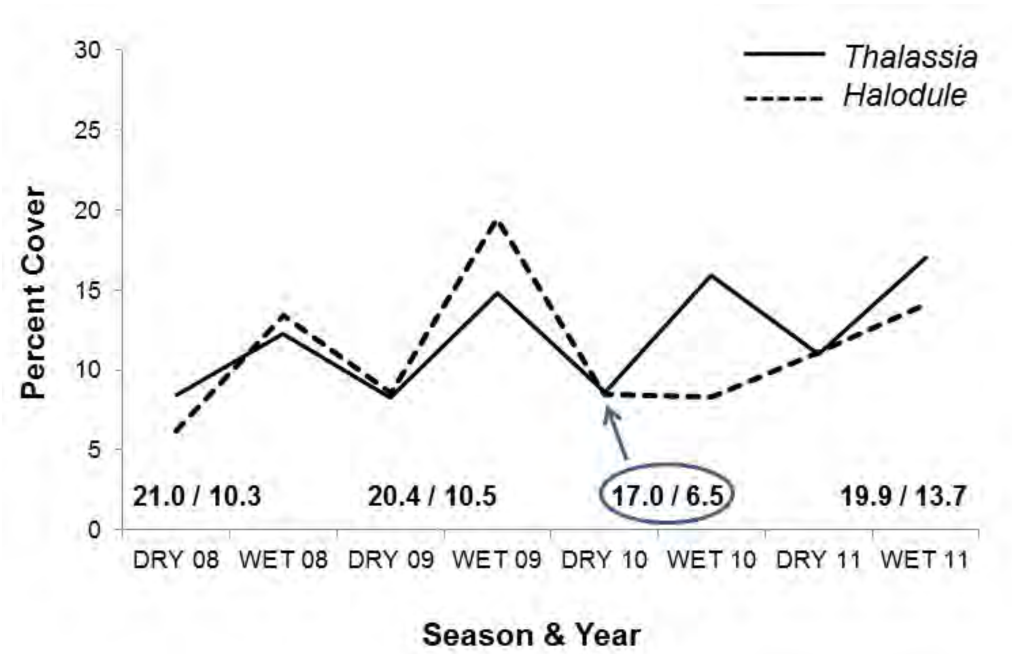


Figure 7-128. Seasonal and interannual patterns in average percent cover of the two dominant seagrass taxa (*T. testudinum* and *H. wrightii*) from inshore sites (< 100 m from shore) in central and south Biscayne Bay. The numbers in bold show mean and minimum salinity values recorded during the month of January each year. Timing of the 2010 cold event is indicated by the arrow.

No major physically-impacting events, such as hurricanes or tropical storms were recorded during the study period, except for the extreme January 2010 cold event, when seawater temperatures reached 6.5° C in the shallow nearshore environments of western Biscayne Bay. While this cold water anomaly took place just before the 2010 dry season SAV surveys, no significant impacts were documented on seagrass cover. In fact, percent cover values for *S. filiforme* and *T. testudinum* in the 2010 dry season were higher than those recorded in preceding dry seasons.

The only direct impact observed post-cold event was the depression of the wet season cover of *H. wrightti* in 2010 (i.e., lack of recovery of *H. wrightii* cover to wet season highs) in the extreme nearshore environments (< 100 m from shore, **Figure 7-130**), which experienced the most drastic temperature declines. However, data from 2011 show a recovering trend in *H. wrightii* cover after this temporary depression, highlighting the short-lived impacts of this anomaly on seagrasses. Seagrasses in nearshore extreme environments of Biscayne Bay are commonly exposed to dramatic temperature changes due to the shallow nature of these environments, and are thus able to survive acute temperature anomalies such as those recorded during January 2010 without drastic declines in abundance.

Effects on Non-native Snails in Biscayne Bay

The nonnative gastropod red-rimmed melania (*Melanoides tuberculatus*) was first reported in the United States in the 1950s and in south Florida in 1971 (Russo 1973). USGS teams working in Biscayne National Park first noticed the snail in 2003 in large numbers near Black Point (**Figure 7-131**). It is a species of concern because in its natural habitat in tropical and subtropical regions of Asia and Africa, it is known to carry parasites that can be harmful to humans (Wingard et al. 2008). In addition, it can harm native animal populations by transmitting parasites and by displacing native species.

Between 2004 and 2007, site assessments were done approximately once per year at Black Point and the density of live *Melanoides* was recorded in order to monitor the spread of their population (**Table 7-14**). Based on USGS observations of *Melanoides'* response to salinity and temperature, and from reports in the literature, the USGS team hypothesized that a significant cold event might be the most likely way to curtail the population. Onsite assessments done in the months following the cold event (February 2010 and April 2010), and again in April 2011 support this idea. No live *Melanoides tuberculatus* were found post-cold event, whereas pre-cold event abundances averaged 532 live snails per m^2 (**Table 7-14**).

Table 7-14. Estimated density of live *Melanoides tuberculatus* (#/m^2) along transects conducted at Black Point, Biscayne National Park (see Figure 7-131 for site locations).

	Pre-cold Event			Post-cold Event		
Sites	**10/26/2004**	**12/7/2006**	**3/31/2007**	**2/24/2010**	**4/29/2010**	**4/14/2011**
TR1	1,739	7	14	0	0	
TR2	1,377	424	1,043	0	0	0
TR3	1,174	14	1,333	0	0	
TR4	1,290	0	0	0	0	0
TR5	812	0	0	0	0	
TR6	348	0	0	0	0	0

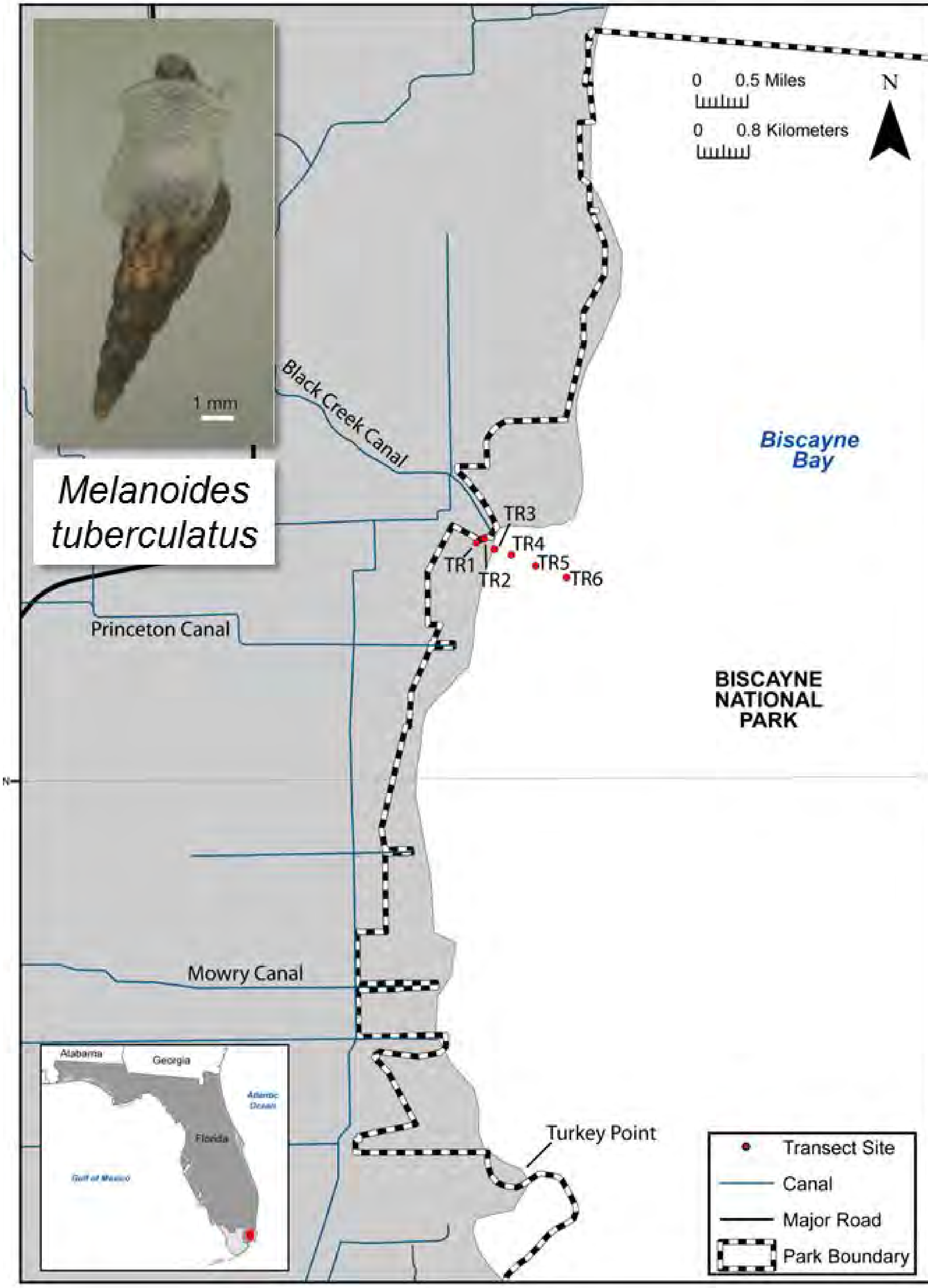


Figure 7-129. Map showing location of transect points surveyed for nonnative *Melanoides tuberculatus* in the Black Point area, Biscayne National Park.

Effects on Epifauna in Biscayne Bay

Community and population abundance metrics from the RECOVER MAP alongshore epifauna projects in Biscayne Bay were used to explore effects of the January 2010 cold event on the epifaunal community. Details of sampling methods are described in annual reports, the most recent of which is Robblee and Browder (2012). The sampling area of this analysis extended from roughly Shoal Point to Turkey Point. Fish richness and abundance indices of dominant fish and shrimp taxa were examined in the analysis. Seasonal averages for each of the years following the cold event, 2010, 2011, and 2012 were compared to the lower 95% confidence level for the years preceding the cold event, 2005–2009. If seasonal averages in 2010 fell below the 2005–2009 lower 95% confidence level, then richness or abundance indices (occurrence and density) were considered significantly decreased. Seasonal data for 2011 and 2012 provide information on recovering trends. Note the 2010 dry season sampling occurred just a few days following the cold event.

Species richness of all epifaunal fish declined significantly in the 2010 dry season and again in the 2010 wet season (**Figure 7-132**). Richness was still significantly lower in both seasons of 2011 and 2012. Four of the eight dominant fish and invertebrate species (pink shrimp, gulf pipefish, clown goby [*Microgobius gulosus*], gulf toadfish [*Opsanus beta*]) declined significantly from previous levels for both abundance indices in the 2010 dry season collection, which immediately followed the 2010 cold event (**Table 7-15**, **Figure 7-133** and **Figure 7-134**). One of these four species (gulf toadfish) also declined in the 2010 wet season collection, compared to previous years. Some of these four species rebounded or varied in abundance in subsequent years, but others continued to decline. For instance, gulf pipefish continued to decline in 2011 and 2012; whereas, gulf toadfish increased in 2011 and then decreased in 2012 during the wet season and showed an increasing trend after 2010 during the dry season. The other four dominant epifaunal species examined (rainwater killifish [*Lucania parva*], goldspotted killifish, code goby [*Gobiosoma robustum*], and grass shrimp) showed mixed results (**Table 7-15**).

The species richness results suggest a reduction in overall species present, both dry season and wet season, which may have been due to the cold event. Significantly, fewer species were present in the monitoring area in 2010 and 2011 compared to the previous four years. Results support an immediate cold event response on some epifaunal species and a possible longer-term effect on gulf pipefish and gulf toadfish. Results also show high within-season year-to-year variability.

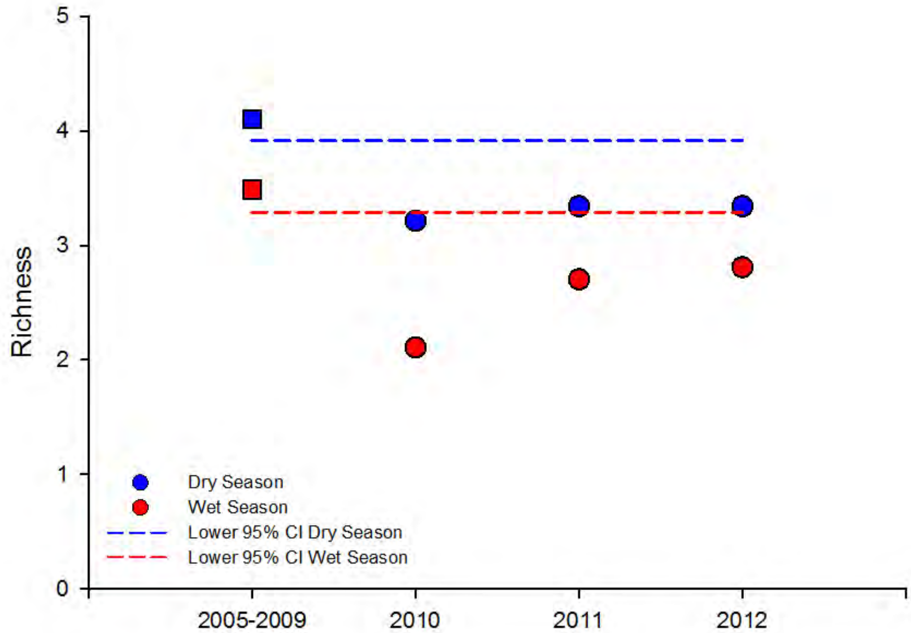


Figure 7-130. Fish species richness (average number of fish species per unit area). (Note: CI – confidence interval.)

Table 7-15. Significant decline in abundance (from 2005–2009 mean) after 2010 cold event.

Species	Scientific name	Density		Occurrence	
		Dry	Wet	Dry	Wet
Gulf pipefish	*Syngnathus scovelli*	Yes	No	Yes	No
Clown goby	*Microgobius gulosus*	Yes	No	Yes	No
Gulf toadfish	*Opsanus beta*	Yes	Yes	Yes	Yes
Pink shrimp	*Farfantepenaeus duorarum*	Yes	No	Yes	No
Rainwater killifish	*Lucania parva*	No	No	No	Yes
Goldspotted killifish	*Floridichthys carpio*	No	No	No	Yes
Code goby	*Gobiosoma robustum*	No	No	No	Yes
Grass shrimp	*Palaemonetes* spp.	No	No	No	No

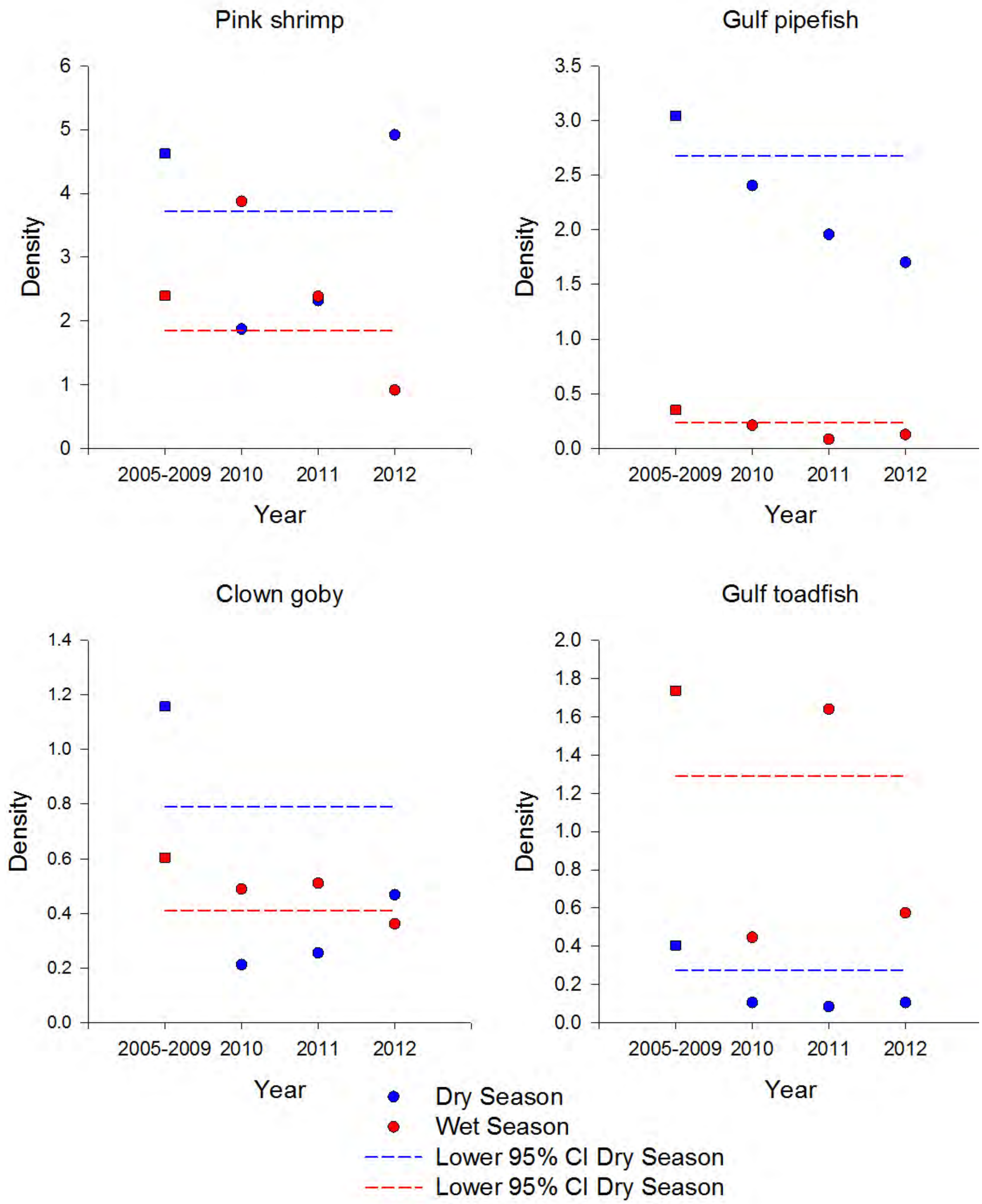


Figure 7-131. Average density (number per 3 m^2) of epifaunal species in Biscayne Bay sampling. Dots below the dashed line of same color are significantly different from the 2005–2009 mean. (Note: CI – confidence interval.)

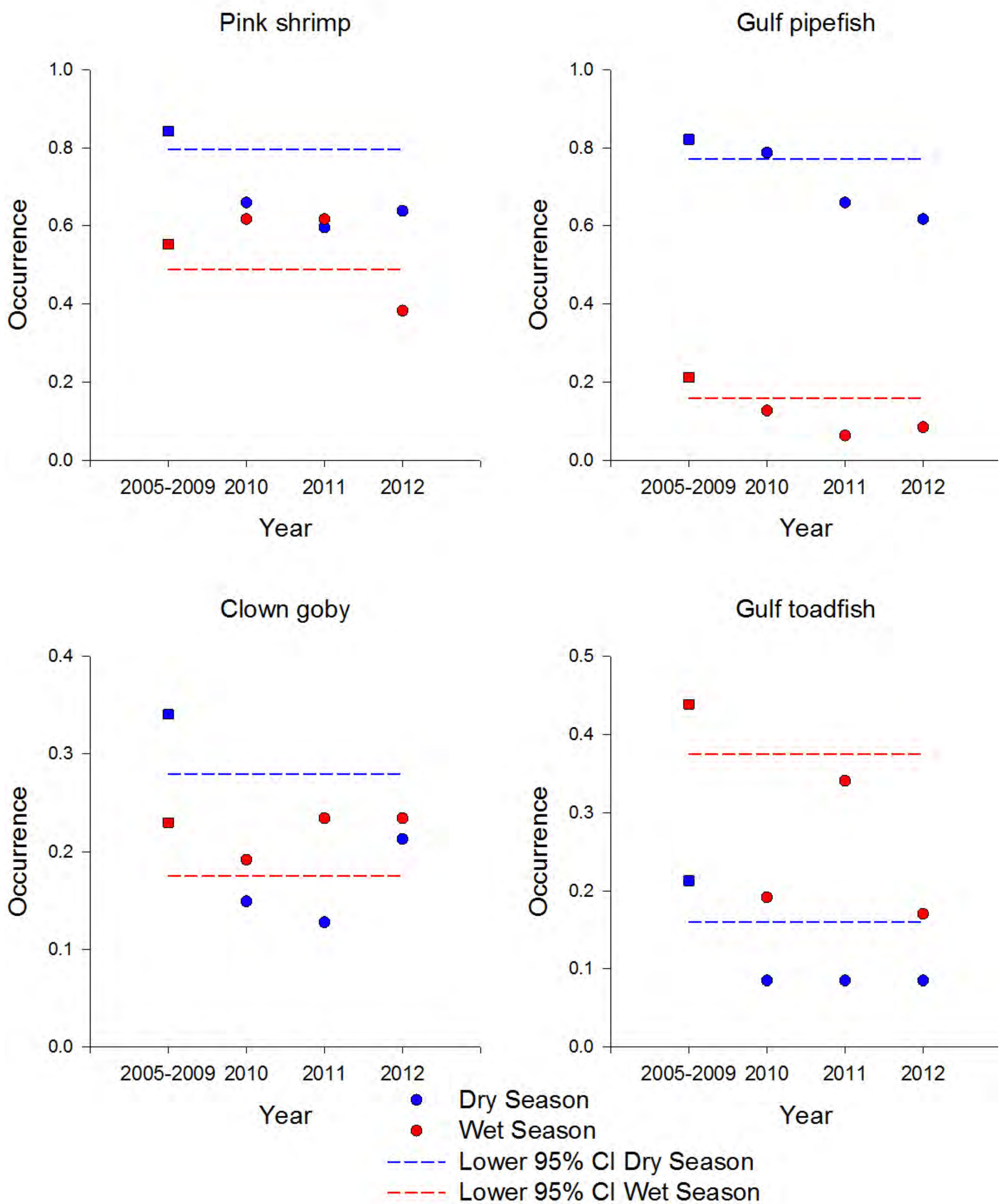


Figure 7-132. Occurrence (percent of sampled sites where encountered) of epifaunal species in Biscayne Bay sampling. Dots below the dashed line of the same color are significantly different from the 2005–2009 mean. (Note: CI – confidence interval.)

Effects on Coastal Fish

<u>Northeastern Florida Bay Mangrove Areas Fish</u>

The demersal wetland fish community in the mainland mangrove zone of northeastern Florida Bay was strongly affected by the 2010 cold event—fish densities just after the cold event (2011–2012) were higher than in years prior to the cold event, but biomass was unchanged.

To assess the effects of the 2010 cold event, a 9-m^2 drop trap was used to assess fish abundance and biomass at four study sites (**Figure 7-135**) just after the cold event (2011–2012). The data were then compared to analogous data collected during two years that had similar rainfall patterns prior to the cold event. At the local level, 1999–2000 was a similar rainfall year (based on rainfall data collected at sampling sites) while 2001–2002 was similar at the regional level (i.e., at the scale of 40 NOAA rainfall stations across south Florida). These two years were not affected by cold events, and thus have nonnative fish densities (primarily Mayan cichlid [*Cichlasoma urophthalmus*]) that are likely more representative of this habitat. Using these two similar-rainfall years also controls for rainfall-induced variation in hydrological parameters, known from previous work to affect fish patterns (Lorenz and Serafy 2006).

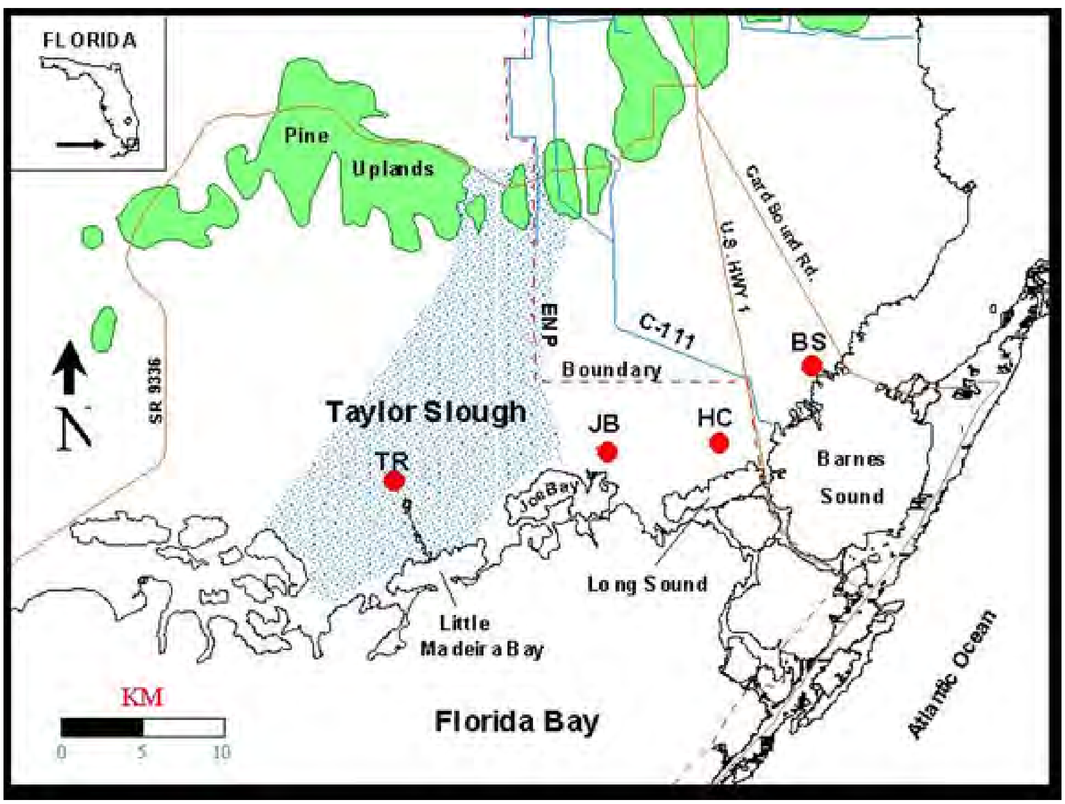


Figure 7-133. Map showing location of four fish sampling sites in the mainland mangrove zone of northeastern Florida Bay, ENP: Taylor River (TR), Joe Bay (JB), Highway Creek (HC), and Barnes Sound (BS).

Abundance and biomass of non-native Mayan cichlids and affected native fish species in 2011–2012 were also compared to the two pre-cold-event years. Mayan cichlids are of particular importance in this area, since they can make up as much as 50% of fish numbers, and their presence is inversely related to the abundance of native fishes (Trexler et al. 2001). Potentially affected native species include sheepshead minnow (*Cyprinodon variegatus*), marsh killifish (*Fundulus confluentus*), eastern mosquitofish, rainwater killifish, and sunfishes. Previous work showed that the density of Mayan cichlids at these sites is negatively related to the density of sheepshead, marsh killifish, and eastern mosquitofish (Harrison et al. 2013). Sunfishes have been reported as both positively and negatively affected by the presence of Mayan cichlids, while sailfin mollies (*Poecilia latipinna*) have been shown to be positively correlated with Mayan cichlid abundance.

Total fish density was consistently higher across sites in 2011–2012 relative to the two pre-cold-event years, but the same consistency was not seen in total fish biomass (**Figures 136A** and **7-137A**). The same pattern was observed in the density and biomass of affected native species (**Figures 7-136** and **7-137B**), except at the Barne Sound (BS) site. For Mayan cichlids, there was a notable difference between years in both density and biomass, particularly at the Joe Bay (JB) and Taylor River (TR) sites (**Figure 7-136C** and **7-137C**). Mayan cichlids were most abundant in 1999–2000, comprising 21.3% of the annual population at Taylor River site and 15.6% at the Joe Bay site. When abundant, Mayan cichlids made a greater contribution to fish biomass due to their relatively larger size. For example, Mayan cichlids comprised 20%, 9% and 1.4% of the Taylor River catch by number in 1999–2000, 2001–2002 and 2011–2012, respectively, but accounted for 54%, 32% and 8% of the biomass, respectively.

Marked increases were detected in densities post-cold event for five of the six individual fish species known to be affected by Mayan cichlids, but the patterns in biomass were less consistent (not shown). In 2011–2012, sheepshead minnow accounted for 25%, and rainwater killifish for 35% of fish density, much higher values than the densities recorded in pre-cold event years. The contribution of these taxa to biomass, however, was smaller than those of Mayan cichlids due to their smaller size. For example, rainwater killifish comprised 35% of the density, but only 18% of the biomass in 2011–2012. Post-cold event increases in sailfin molly density in 2011–2012 contradict previous results showing a positive correlation between sailfin molly and Mayan cichlid numbers. It is suspected that the absence of Mayan cichlids post-cold event resulted in a release from predation or competition for native fishes, contributing to their higher densities. Similar increases in biomass were not seen due to the relatively smaller body size of native taxa, and thus their smaller contribution to total fish biomass.

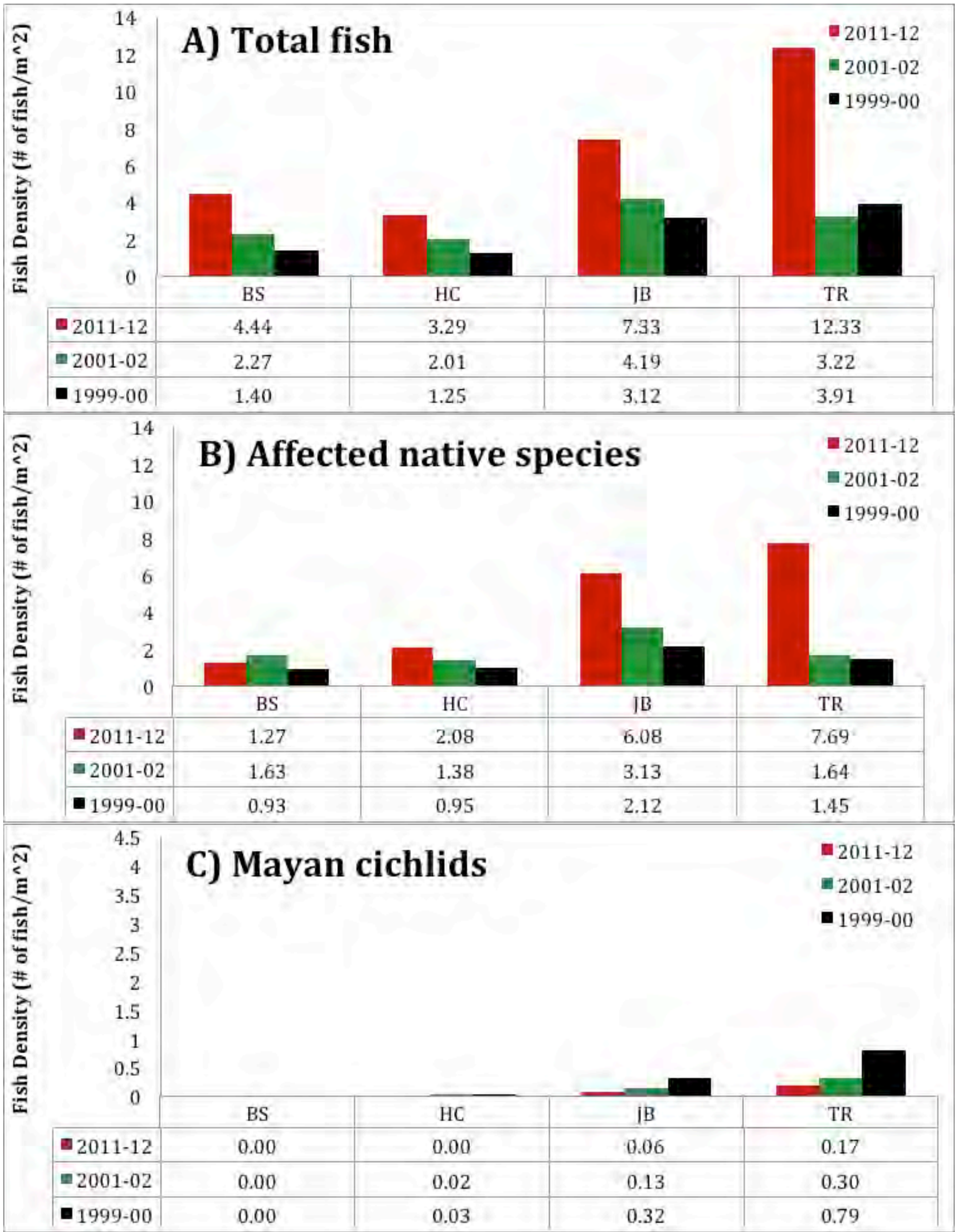


Figure 7-134. Annual mean fish density for (A) all species collected, (B) sheepshead minnow, eastern mosquitofish, and rainwater and marsh killifishes combined, and (C) nonnative Mayan cichlids for 2011–2012 and the two pre-cold event years (2001–2002 and 1999–2000) across Florida Bay sites (see Figure 7-135 for locations). Note variable scale across panels.

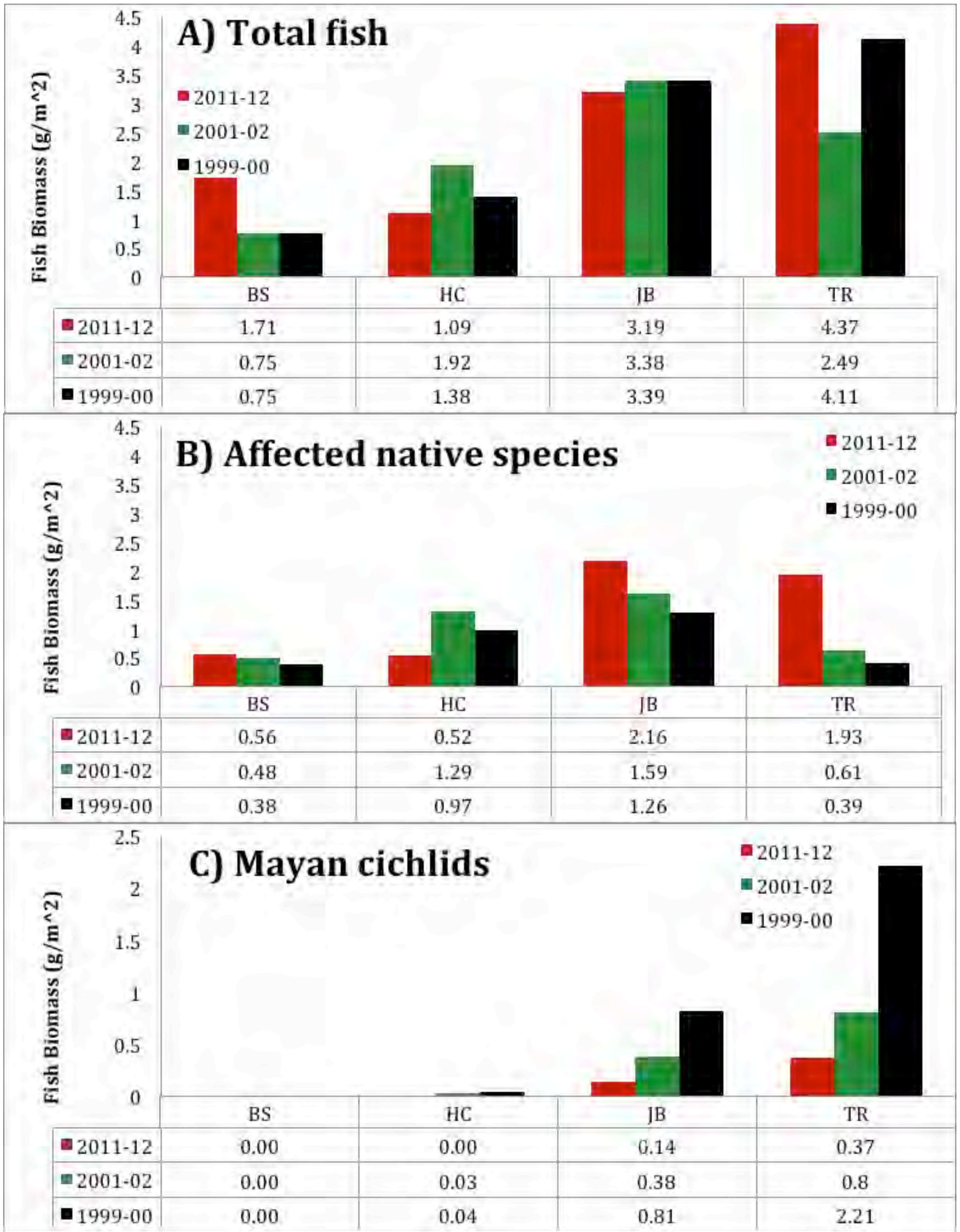


Figure 7-135. Annual mean fish biomass for (A) all species collected, (B) sheepshead minnow, eastern mosquitofish, rainwater and marsh killifishes combined, and (C) nonnative Mayan cichlids for 2011–2012 and the two pre-cold event years (2001–2002 and 1999–2000) across Florida Bay sites (see Figure 7-135 for locations). Note variable scale across panels.

Biscayne Bay Mangrove Areas Fishes

Largely supported by RECOVER, visual belt-transect fish surveys have been conducted along the mangrove-lined shorelines of Biscayne Bay and adjacent sounds since 1998. Therefore, this monitoring effort provides several years of seasonally-resolved (dry: January–March; wet: July–September) data with which to examine fish responses to acute disturbances, such as the 2010 cold event, and to assess recovery. Here, the analyses were restricted to data collected from January 2005 forward since sample sizes prior to 2005 were limited (i.e., < 100 transects per season). The emphasis was on variation in several fish metrics, including a community-level metric (i.e., mean species richness per transect) and a selection of species-specific density (log_e-transformed) metrics. Specifically, on a seasonal basis, average yearly metrics post-cold event (2010–2013) were compared to values averaged across the pre-cold event period (2005–2009). Post-event metric values falling below the lower 95% confidence interval of the 2005–2009 average were considered significant departures that are potentially attributable to the very low temperatures that prevailed in winter 2010. This exercise also provides insight into recovery rate variation in fish diversity and density following this unusual disturbance event.

Species Richness

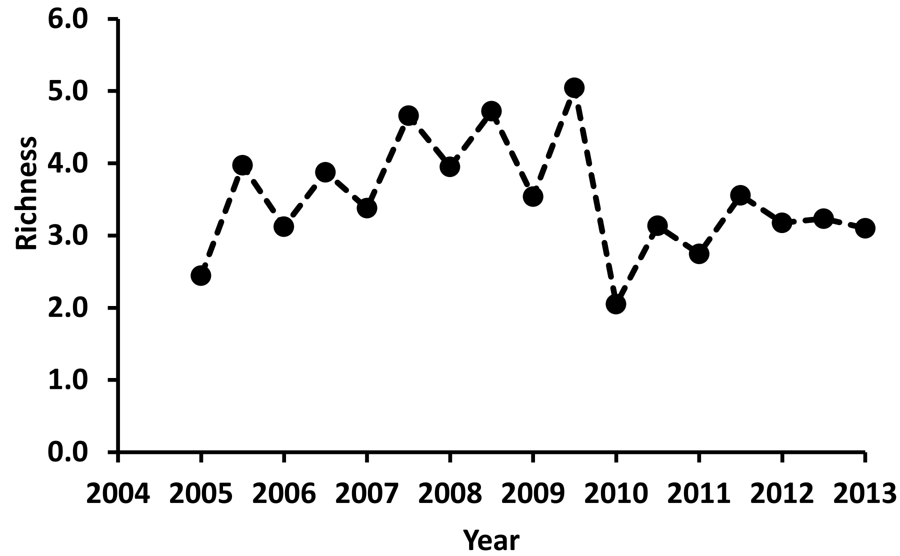
Species richness is a useful, straightforward measure of biodiversity within a system. The mean mangrove fish species richness level observed immediately following the event (2010 dry season) was the lowest seasonal average ever observed (**Figure 7-138**). During subsequent dry seasons, significantly reduced levels of species richness persisted until 2012. During subsequent wet seasons, species richness levels were still below pre-event wet season levels by the end of the 2012 sampling season.

Small Mojarras

The mixed-species group of small (< 10 cm total length) mojarras (*Gerreidae*) constitutes an important forage base for predatory fishes, wading birds, and other fauna. The mean small mojarra density immediately following the event (2010 dry season) was the lowest seasonal average observed between 2005 and 2013 (**Figure 7-139B**). Subsequent dry season means were not significantly different from pre-event values. However, mean wet season densities were consistently below the significance threshold each year after the 2010 cold event.

Gray Snapper

Gray snapper (*Lutjanus griseus*) is among the most ecologically and economically important species in the region. Mean gray snapper density immediately following the event (2010, dry season) was the lowest seasonal average ever observed (**Figure 7-138**). A significantly reduced density was observed in the subsequent year (2011 dry season), but this metric returned to pre-event levels by the beginning of 2012. Wet season patterns of gray snapper density were less clear, with values either very close to, or above, our significance threshold from 2010 forward.



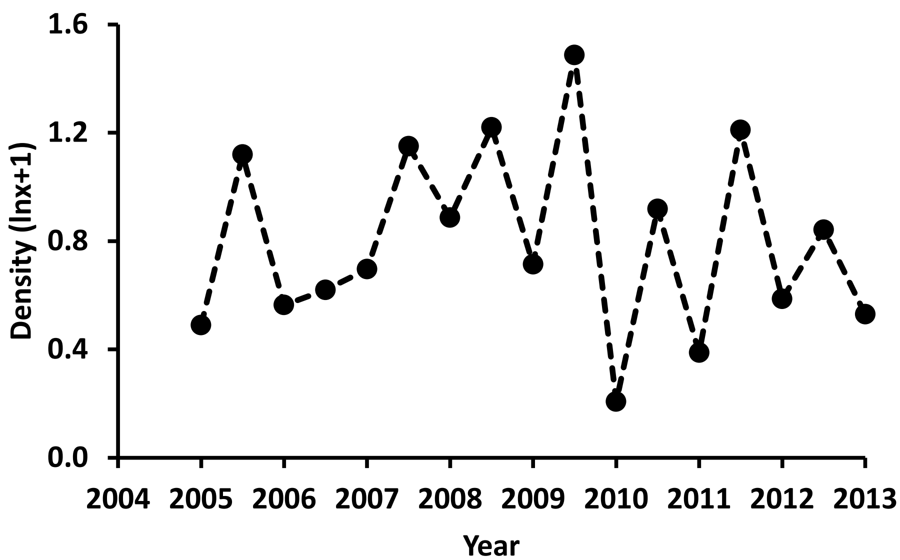


Figure 7-136. Taxonomic richness (upper panel) and gray snapper density (lower panel) time series along the mainland mangrove shoreline habitats surveyed. Note that the lowest values of both fish metrics were observed immediately following the cold 2010 event.

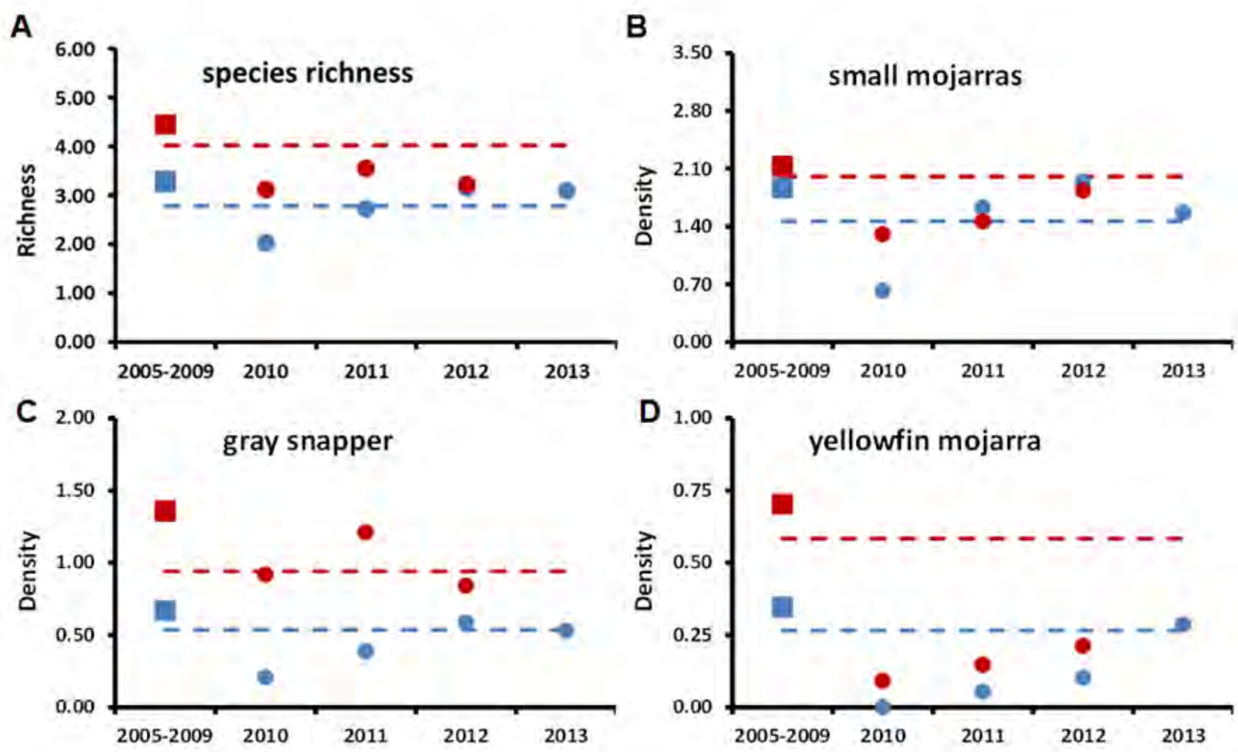


Figure 7-137. Effects of the 2010 cold event on mangrove-associated fish assemblages in Biscayne Bay and adjacent waters across (A) species richness, (B) small mojarras, (C) gray snapper, and (D) yellowfin mojarra (*Gerres cinereus*). Red and blue symbols and lines represent wet (January–March) and dry (July–September) season values, respectively. Squares indicate seasonal mean values for the five-year period (2005–2009) prior to the cold event. Dashed lines indicate lower 95% confidence intervals associated with the pre-event means. Species richness units are mean numbers of fish taxa per 60 m^2. Density units are ln(x+1) individuals per 60 m^2.

Yellowfin Mojarra

Yellowfin mojarra over 10-cm total length are readily identifiable by their coloration; small (< 10 cm) yellowfin mojarra form an unknown fraction of the mixed-species mojarra group described above. This species feeds on benthic organisms and is a preferred prey of lemon sharks (*Negaprion brevirostris*), a top predator along regional mangrove shorelines. No yellowfin mojarra was identified in any surveys immediately following the 2010 cold event (**Figure 7-139D**); a mean density of zero was the lowest mean density ever observed in the program's history. Densities following the event were significantly depressed for the following two years. Not until the 2013 dry season, was mean yellowfin mojarra density at a level equivalent to pre-event levels.

Goldspotted Killifish

The goldspotted killifish is the most abundant cyprinodontid observed along the mangrove-shorelines of the region. A primary benthic consumer, this species constitutes important forage for wading birds and juvenile seatrout, snappers, grunts, and great barracuda (*Sphyraena barracuda*). Patterns of killifish density suggest a general trajectory of decline that is not readily attributable to the cold event (**Figure 7-140A**), although it is possible that they indicate that the response was delayed by one or more seasons.

Great Barracuda

Juvenile great barracuda in mangroves are important predators of mojarras, killifishes, silversides, pinfish, and young grunts and snappers. Dry season barracuda density was significantly depressed immediately after the event, as well as during 2011 (**Figure 7-140B**). By 2012, dry season density was within pre-event levels. Barracuda density during the wet season remained significantly below 2005–2009 levels from 2010 forward.

Small Water Column Fishes

The small water column fishes group includes silversides, herrings and anchovies that often form large mixed-species schools. This group primarily feeds on micro/macroscopic invertebrates including zooplankters and invertebrates that inhabit algal, seagrass, and prop-root surfaces. They are a key food item of gray snapper. Mean small water column fishes density immediately following the event (2010 dry season) was the lowest seasonal average ever observed (**Figure 7-140C**). Densities were still depressed a year later (i.e., during both the 2011 wet and dry seasons). By 2012, however, densities had returned to pre-event levels.

Summary

In summary, for most of the mangrove-fish metrics examined, the lowest values on record were observed immediately following the 2010 cold event. Values of many metrics were still depressed a full year later. With a few exceptions, recovery to pre-event, dry season levels was achieved by 2012. Metrics that by 2012 had not reached pre-event, wet season levels were species richness and densities of small mojarras, yellowfin mojarra and great barracuda. Analyses that examine whether similar cold event “signals” are present in other long-term fish monitoring data sets from south Florida are warranted. Further investigation is required to explore whether (1) low densities of mojarras and barracuda since the cold event have had negative consequences for their immediate predators and positive consequences for their immediate prey in the system; and (2) the winter 2010 “cold shock” to nearshore nursery habitats has reduced the 2010 cohort of a suite of species that eventually made their ontogenetic migration from the shallows to the coral reef tract offshore. Additional years of monitoring are needed for a more complete picture of impacts and resilience.

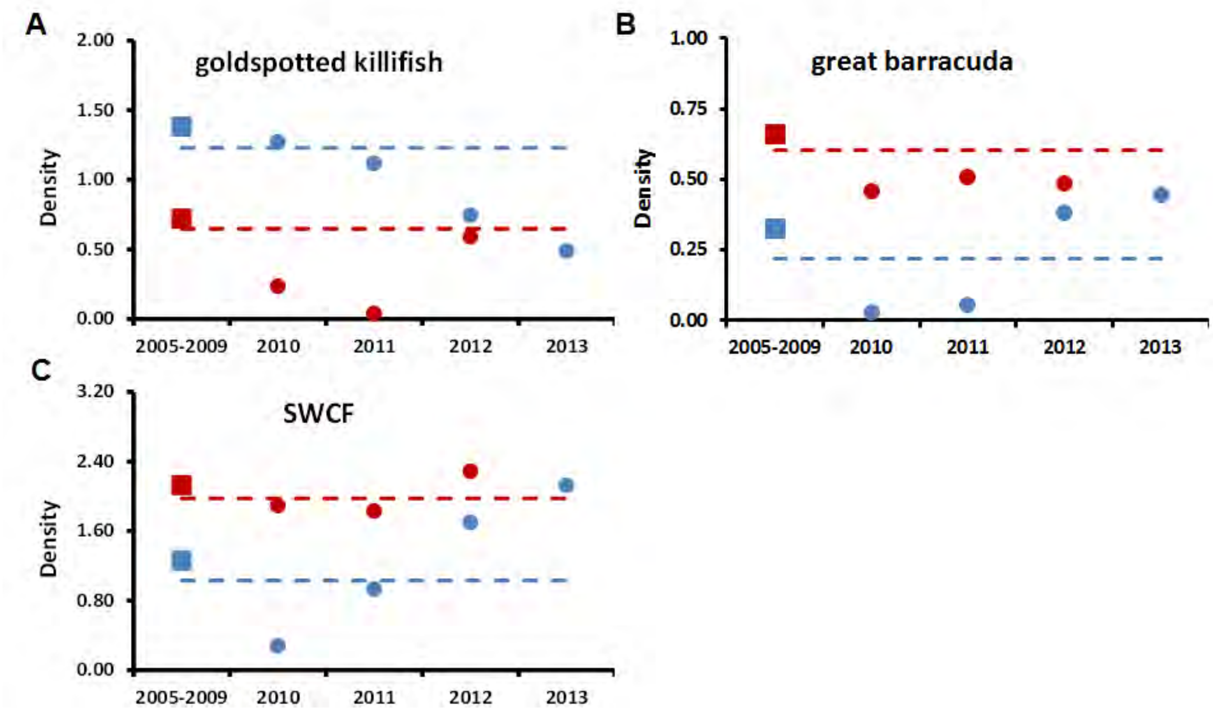


Figure 7-138. Effects of the 2010 cold event on mangrove-associated fish assemblages in Biscayne Bay and adjacent waters across (A) goldspotted killifish, (B) great barracuda, and (C) small water column fishes (SWCF—silversides, herrings and anchovies). Red and blue symbols and lines represent wet (January–March) and dry (July–September) season values, respectively. Squares indicate seasonal mean values for the five-year period (2005–2009) prior to the cold event. Dashed lines indicate lower 95% confidence intervals associated with the pre-event means. Species richness units are mean numbers of fish taxa per 60 m^2. Density units are ln(x+1) individuals per 60 m^2.

Effects on Fishes in Mangrove Areas of the Upper Shark River

Ecotonal fish communities in the upstream region of the Shark River Estuary in ENP (**Figure 7-141A**) are composed of freshwater taxa of a temperate origin, and of estuarine/marine and nonnative species of a subtropical/tropical origin, and thus vulnerable to thermal stress. Responses of this fish community to the 2010 cold event were examined by tracking changes in the abundance (**Figure 7-141B, C, D**) and the functional trait structure (**Figure 7-142**) of the community across pre- (2005–2009) and post-cold event years (2010–2012).

The cold event had major negative effects on the estuarine and nonnative components of this fish community. Approaches that examine functional trait structure are taxon and system-independent, directly link traits to environmental drivers, and allow for mechanistic prediction of changes in species composition and abundance (Mouillot et al. 2010). Changes were plotted in the abundances of the 15 dominant species (which account for 84% of total fish numbers) in trait-space biplots, defined by each species' salinity and temperature lethal limits.

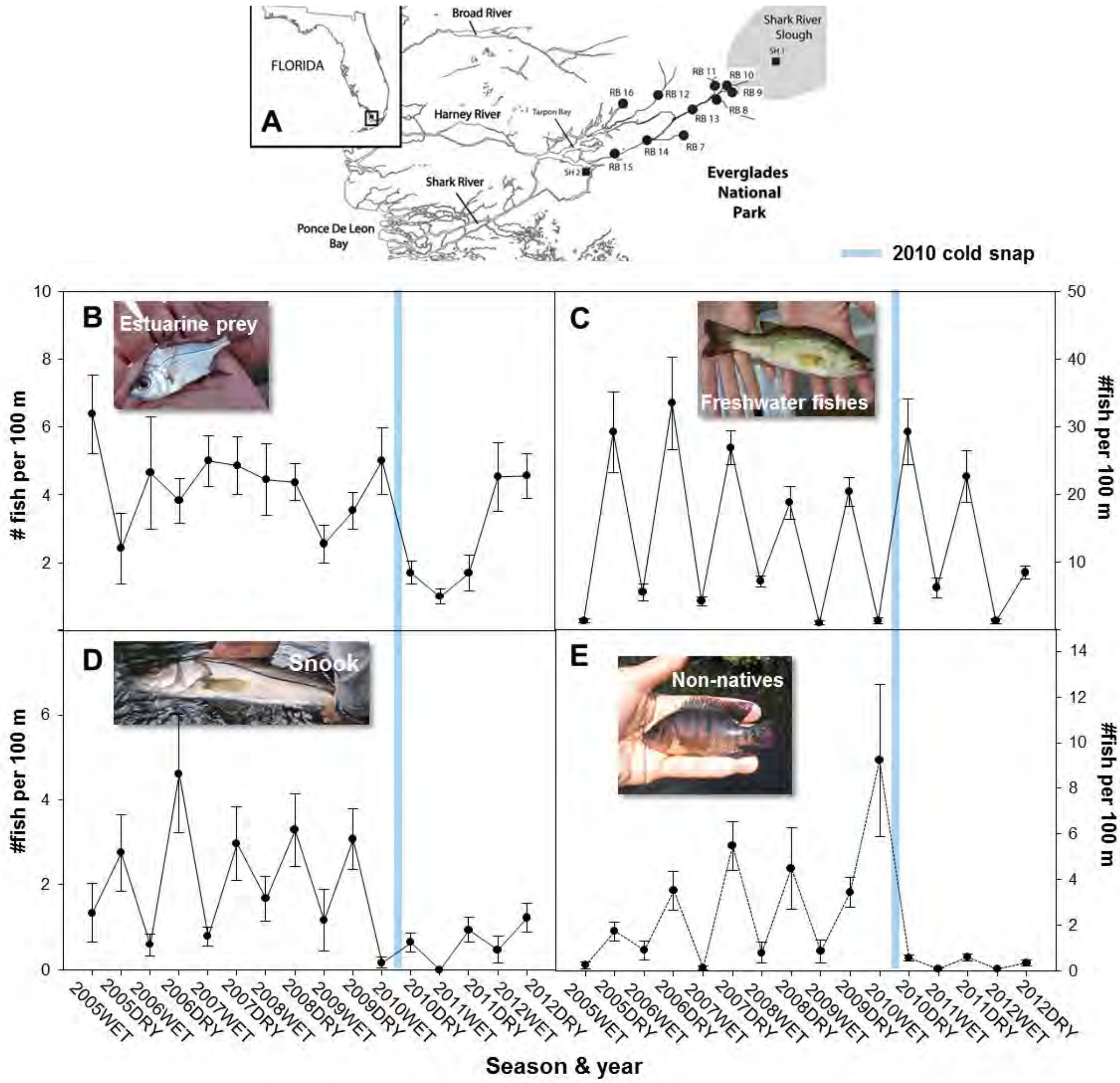
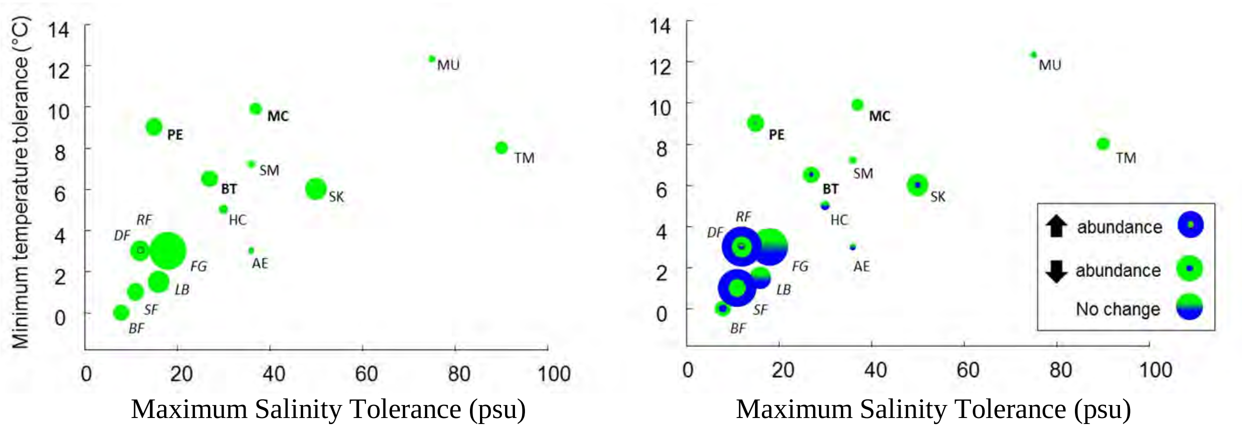


Figure 7-139. Seasonal and yearly abundance of (B) estuarine prey, (C) freshwater marsh fishes (both larger predators and smaller prey), (D) snook (*Centropomus undecimalis*), and (E) nonnative fishes across (A) long-term ecotonal mangrove creek sites in the upper Shark River, ENP. Data are from boat electrofishing and reflect the number of fish caught per 100 m of mangrove creek shoreline at 10 fixed sites (round symbols in A). Wet season samples correspond to November–December, while dry season samples aggregate early (February–March) and late (April–May) samples. The timing of the 2010 cold event is shown by the blue line.



Freshwater fishes	Estuarine fishes	**Nonnative fishes**
BF = Bowfin	AE = American eel	**BT = Blue tilapia**
DF = Dollar sunfish	HC = Hogchoker	**MC =Mayan cichlid**
FG = Florida gar	MU = Striped mullet	**PE = Peacock eel**
LB = Largemouth bass	SK = Snook	
RF = Redear sunfish	SM = Striped mojarra	
SF = Spotted sunfish	TM = Tidewater mojarra	

Figure 7-140. Changes in the functional structure of the ecotonal fish community in the upper Shark River (Figure 7-141A) across functional trait space defined by the maximum salinity and minimum temperature tolerance of the dominant 15 taxa. The size of circles corresponds to each species' relative abundance. Panels show abundance patterns across temperature and salinity trait space: (A) pre-cold event (2006–2009, green symbols) versus (B) post-cold event (2010, blue symbols). In (B), color coding indicates increases, decreases or no change in a species' abundance. Two-letter abbreviations indicate individual species: italics are freshwater, plain text are estuarine, and bold are nonnative fishes, see table insert for fish names.

Seven species experienced at least an 80% decrease in abundance relative to their pre-cold event abundances. These included 4 estuarine taxa (snook, striped [*Eugerres plumieri*] and tidewater [*Eucinostomus harengulus*] mojarras, and striped mullet [*Mugil cephalus*]) and the 3 dominant nonnatives (blue tilapia [*Oreochromis aureus*], Mayan cichlid and spot-finned peacock eel [*Macrognathus siamensis*]). Interestingly, the cold event had a positive effect in the abundance of certain freshwater marsh species. Dollar sunfish (*Lepomis marginatus*) and spotted sunfish (*Lepomis punctatus*) increased in abundance 500% and 400%, respectively, in 2010 relative to the pre-cold event period. It is hypothesized that their increases relate to a release from predation. Diet studies by Florida International University have shown both of these species to be readily consumed by snook (Boucek and Rehage 2013), which suffered high mortality and were extremely rare at study sites post-cold event (**Figure 7-141D**). Continued monitoring after this climatic extreme showed rapid recovery only in the estuarine prey (i.e., striped mullet). Estuarine prey numbers bounced back to pre-cold event as early as

November 2012 (**Figure 7-141B**). In contrast, the recovery of snook and the nonnative taxa have been much slower (**Figure 7-141D** and **E**).

Of particular interest is the recovery of common snook, an economically-important recreational fishery in the Everglades and statewide. Recreational fishing in Florida accounts for $7.5 billion of economic impact annually, with 21% of Florida anglers fishing the Greater Everglades, and generating $1.2 billion in annual economic activity. In the Everglades, snook are one of the main recreational fisheries; it is estimated that 40% of Everglades anglers target snook. Findings from this study show that in the upper Shark River, snook numbers have not fully recovered and remain at about 50% of abundances pre-cold event, although they appear on a recovery trajectory, particularly in the dry season samples (**Figure 7-141D**). Snook numbers are known to increase in the dry season, likely due to local (i.e., from downstream in the Shark River) or regional immigration. The increasing trend in their dry season numbers may indicate a faster recovery for transient relative to resident individuals.

Effects on the American Crocodile

The American crocodile is a primarily coastal crocodilian, occurring in parts of Mexico, Central and South America, the the Caribbean, and reaches the northern extent of its range in south Florida. Crocodile mortality was evaluated in response to the cold event using a combination of aerial surveys from a helicopter and boat surveys between January 20 and February 19, 2010. Location, size, and, when possible, sex was determined for each crocodile mortality encountered. Also, encounter rates (crocodile per km) in established surveys and nesting effort (number of nests per year) were compared before and after the 2010 cold event.

A total of 150 dead crocodiles ranging in age class from hatchling to adult were documented. Prior to this cold event there were 143 American crocodile mortalities recorded in Florida between 1967 and November 2007, a number surpassed by mortality in this single 2010 cold event. The maximum cold related mortality reported in any previous year was four individuals in 1990 following a hard freeze. Relative density of crocodiles on our survey routes decreased between 2006 and 2007, but remained relatively constant before and after the cold event. It is hypothesized that changes in salinity caused the decrease in relative density. Relative density may be more sensitive to changes in salinity patterns, whereas overall distribution of crocodiles may be more affected by cold temperatures.

Numbers of nests in Florida reached an all-time high in 2008, but then decreased in the following years. In 2009, the decrease in number of nests was related to storm activity (especially flooding), and in 2010 due to the cold event (see **Figure 7-70**). While still fluctuating, nesting appears to be on a general increasing trend since 2010, especially in northeastern Florida Bay.

Summary of the Cold Event and Its Effects

In summary, the 2010 cold event was an extremely severe event, with profound impacts on Everglades fauna and flora. Community-wide effects were documented, including extensive mortality effects in several species. For instance, the event coincided with the highest observed occurrence of crocodile mortality on record for south Florida. Also, decreases in nonnative Mayan cichlids and snook in coastal mangroves resulted in increases in the abundance of native small fishes, likely a result of a release from predatory and competitive interactions, showing the cascading effects of changes in the abundance of just a few member species. Long-term monitoring efforts provided an opportunity to both track the responses of key taxa to this climatic extreme, as well as to closely examine their recovery trajectories. Monitoring shows that most affected native species either quickly recovered or are on a recovery trajectory. Of interest is the variation in the recovery time of species, relatively quickly for many of the monitored taxa (i.e., seagrasses and gray snapper), yet slower for other fauna (i.e., snook and yellowfin mojarra), and yet to be shown for others (i.e., gulf pipefish and nonnatives). Additional monitoring and research is needed to better understand this variation, particularly since slower-recovery species are recreationally important. An important exception to positive recovery trajectories is the lack of or significantly slower recovery in the nonnative taxa. At least two years post-cold event (2012), recovery of nonnative snails in Biscayne Bay and fishes in both Shark River and Florida Bay mangrove regions is not apparent, although expected. Given the abundance of thermal refugia (i.e., canals) in the system, recovery and recolonization, albeit perhaps slower than for native taxa, is fully expected for nonnatives in the next few years.

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APPENDIX 7-1: SIGNIFICANT STRUCTURAL AND OPERATIONAL CHANGES AFFECTING THE SOUTHERN COASTAL SYSTEM OVER THE LAST 30 YEARS: A BRIEF SUMMARY OF AVAILABLE INFORMATION

Since the 1980s, there have been a number of structural and operational changes that have affected freshwater inflow to the Southern Coastal Systems region. Some of the structural changes have also affected local circulation in northeastern Florida Bay. This appendix highlights some of those changes and effects.

Figures A7-1-1 through **A7-1-5** show water managerment structures that are referred to in the following sections.

Biscayne Bay

To the best of our knowledge, operational changes affecting flow to central and southern Biscayne Bay over the last few decades have been limited to a change affecting flows through the C-1 Canal, which occurred in 2000. Prior to 2000, the S-335 structure at Tamiami Trail rarely opened during the wet season. Pressure from private property interests in the Pennsuco Wetlands to maintain L-30 Canal stage at 6 feet required opening the S-335 during the wet season. This additional wet season water was shunted to tide through the C-1 Canal. Effects on flow through the C-1 Canal due to this operational change were examined by evaluating flow data at the S-21 coastal structure prior to and following 2000. Results are provided below:

- Average annual flows through the C-1 Canal at S-21 prior to 2000 (period of record: January 1, 1979–December 31, 1999) = 163.6 cubic feet per second (cfs)
- Average annual flows through the C-1 Canal at S-21 beginning in 2000 (period of record: January 1, 2000–December 31, 2012) = 205.2 cfs

These calculations indicate that after 2000, flows through the C-1 Canal at the S-21 structure increased 25%, which is a substantial increase in freshwater deliveries to the central-southern area of Biscayne Bay. Additional analyses are needed to better understand how these flow increases have affected the regional salinity regime. A similar flow analysis conducted at structure S-21A (the coastal structure associated with the adjacent C-102 basin to the south of the C-1 basin) indicated only a slight increase (< 4%) in flow for the 2000–2012 time period compared to the previous time period (1979–1999). An analysis of Division 5 rainfall data indicates that rainfall was not a contributing factor to the observed flow increase in the C-1 Canal (average annual rainfall: 1979–1999 = 52.9 inches; 2000–2009 = 52.5 inches).

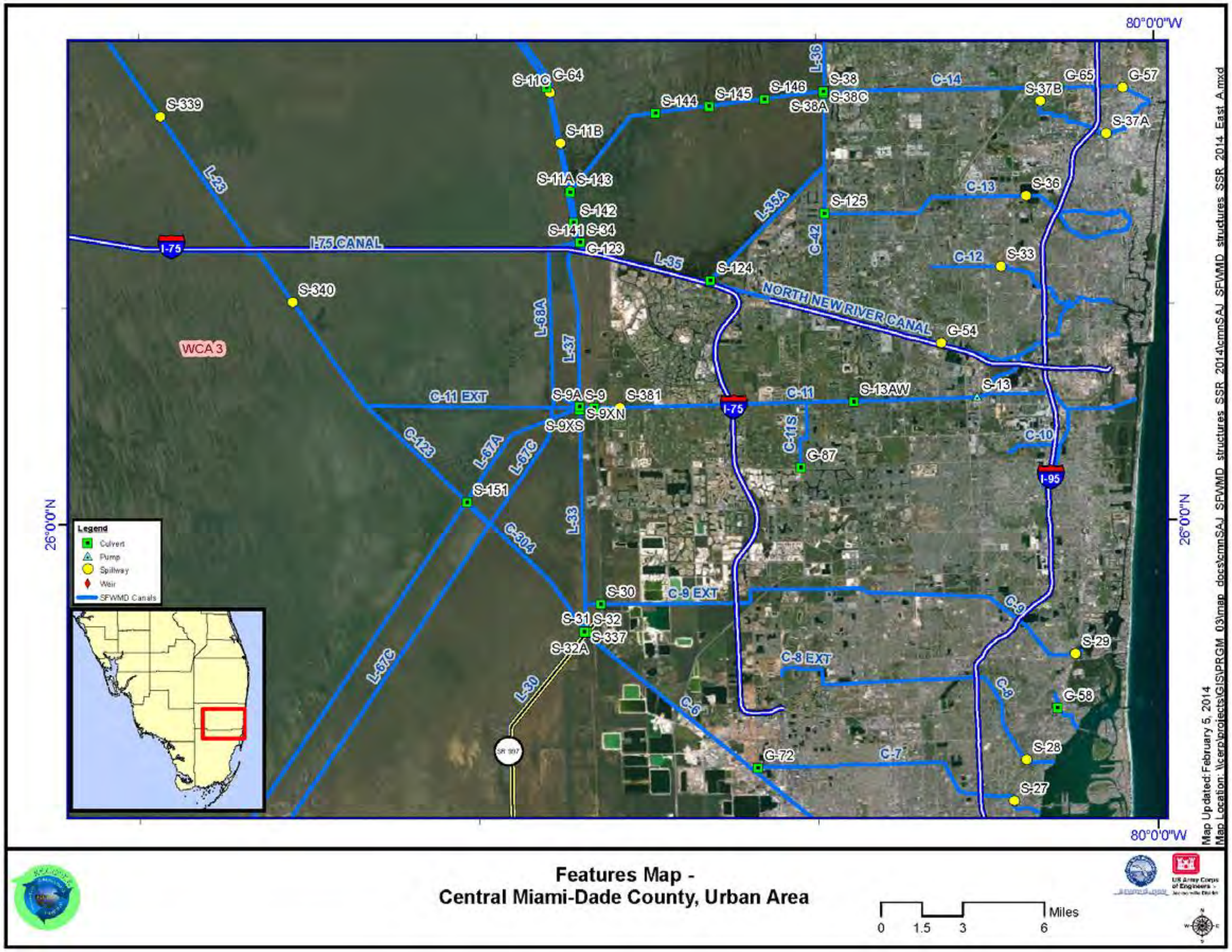

Figure A7-1-1. Structures in Central Miami-Dade County.

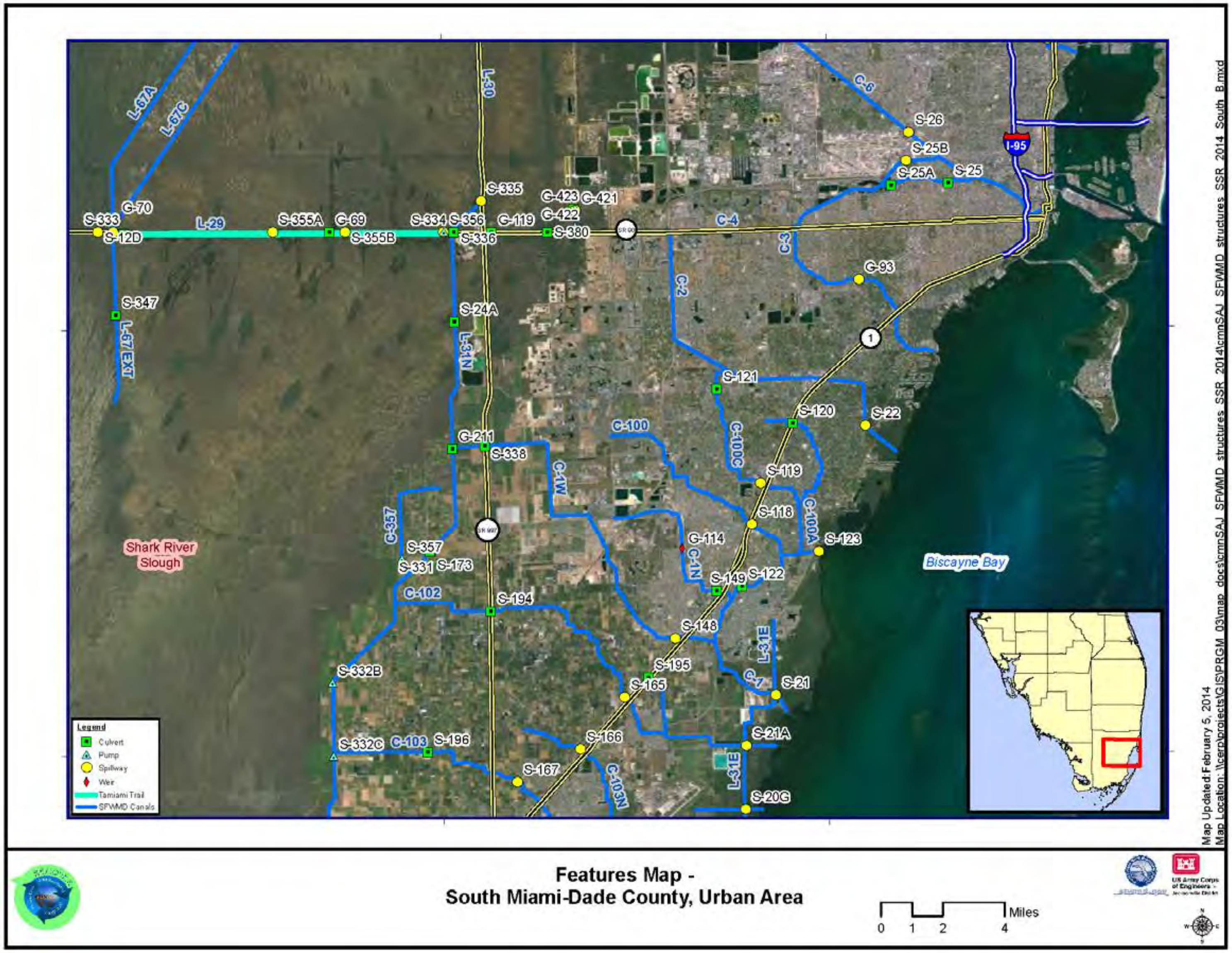

Figure A7-1-2. Structures in South Miami-Dade County.

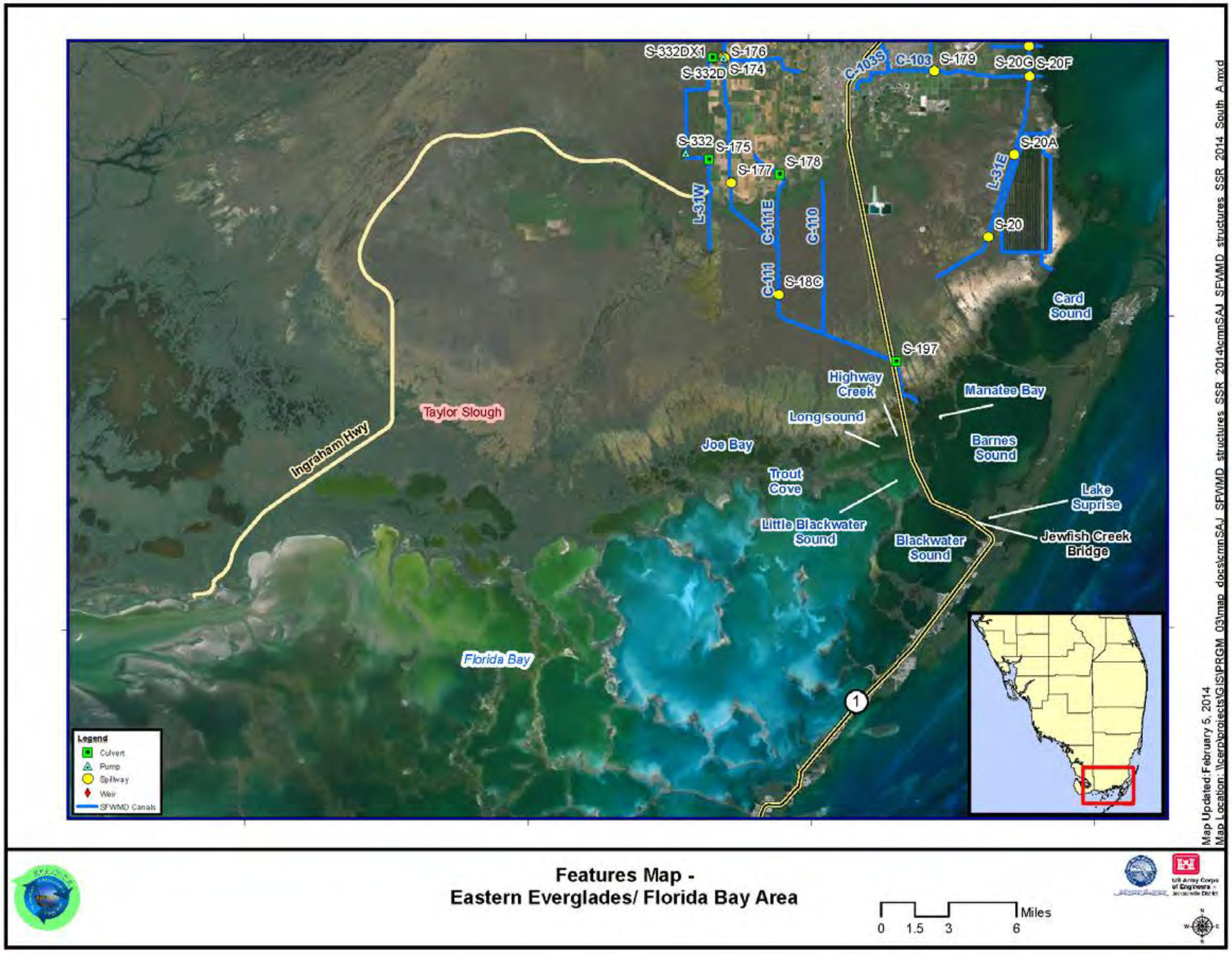

Figure A7-1-3. Structures in the eastern Everglades National Park and Florida Bay area.

Figure A7-1-4. Structures within the Big Cypress National Park and western Everglades National Park area.

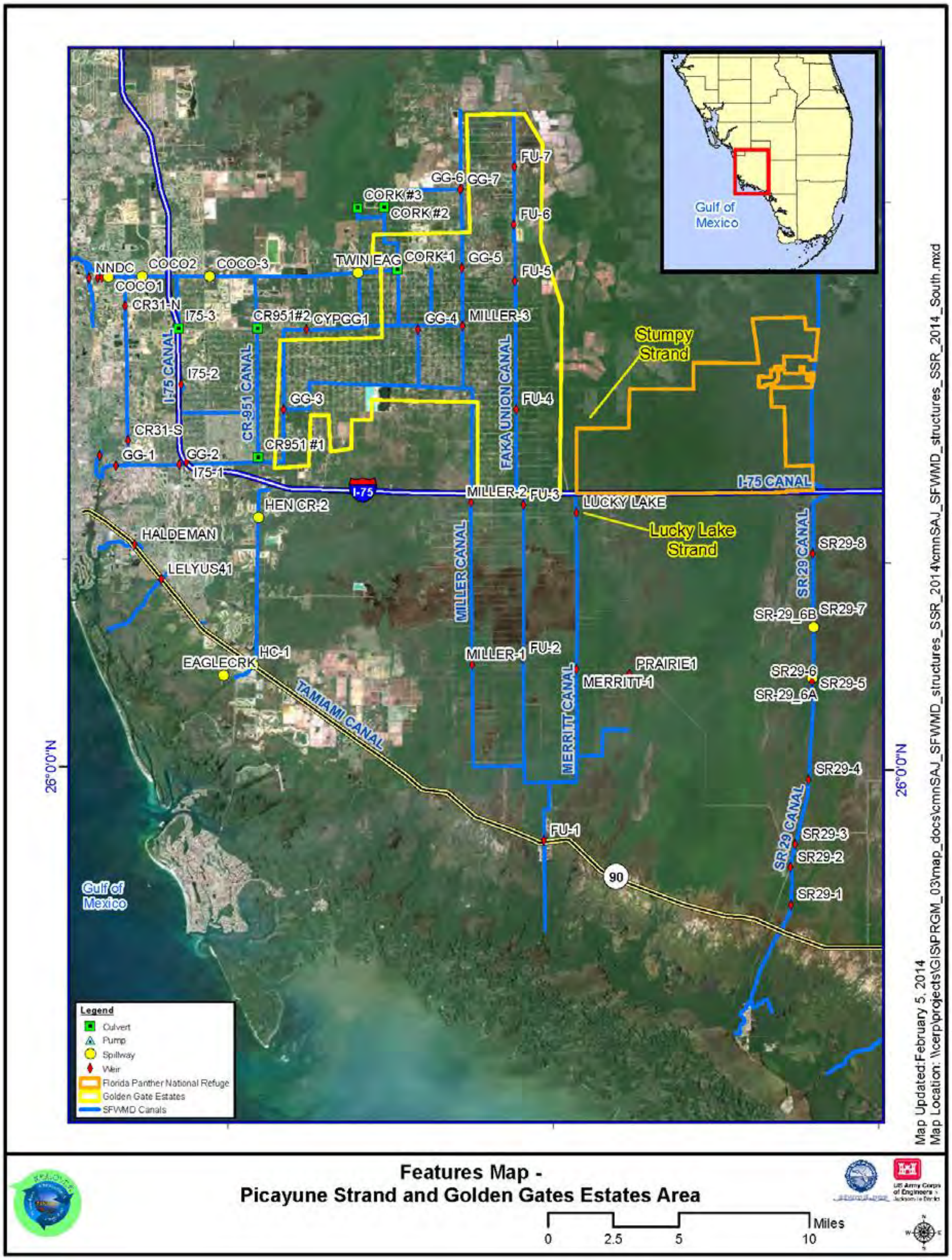

Figure A7-1-5. Structures in the Picayune Strand and Golden Gates Estates area.

Central and Northeastern Florida Bay

Four changes have occurred over the last 25 years that have affected central and northeastern Florida Bay: (1) upstream operational changes affecting flow to Taylor Slough and the C-111 Canal, (2) removal of dredge spoil from the south bank of the C-111 Canal, (3) modification of the Everglades National Park (ENP) road where it crosses Taylor Slough, and (4) increased connection between Long/Blackwater/Little Blackwater Sounds and Manatee Bay/Barnes Sound as a result of the U.S. Highway 1 Widening project.

Operational Changes

Kotun and Renshaw (2013) and Lorenz and Kotun (in preparation) identify five time periods of distinct water management changes that have affected Taylor Slough hydrology since 1961. The effects of some of these changes have been limited to upper Taylor Slough. Because the Comprehensive Everglades Restoration Plan (CERP) Monitoring and Assessment Plan (MAP) data does not begin until the 1990s at the earliest, the operational changes of interest for this System Status Report are limited to the last two operational changes identified by the papers noted above, 1991–1999 and 2000–2010. Pumping to the west into northern Taylor Sough increased from 165 cfs to 500 cfs at the S-332i pump on L-31W Canal (identified as 1991–1999 period in Kotun and Renshaw) and the S-332B, C, and D pumps were brought online (2000–2010). Of those two operational regimes, the major operational change that affected flow to northeastern and central Florida Bay occurred when the S-332i capacity increased, which caused a 3-fold increase in flow to upper Taylor Slough. The increase in pumping above the earlier 165 cfs began with a slight increase in October 1992, with the major increase in pumping capacity initiating in late 1993 (**Figure A7-1-6)**. Much of the additional water in the slough drained back into the C-111 Canal south of the pump station resulting in increased flow at the S-177 and S-18C structures. The increased pumping capacity of S-332i resulted in greater water deliveries to northeastern and central Florida Bay after 1992.

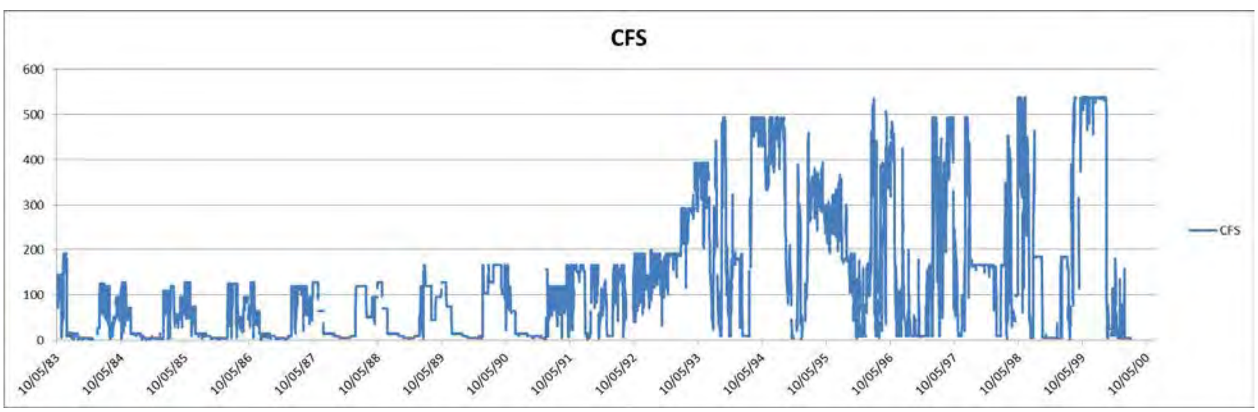

Figure A7-1-6. Time series of S-332 pumping volumes, October 1983–September 2000.

An excerpt from the Kotun and Renshaw paper explains the effects of the 1991–1999 operational period in greater detail:

> During this period, basin inflow at S331 was similar to the previous period, but flow in Taylor Slough at SR9336 increased by a factor of three and flow per unit rainfall increased dramatically indicating a major change in basin hydrology. This was made possible by inflow at S174 which doubled while outflow at S175 remained steady. As a result, virtually all of the increased flow from S174 was directed into Taylor Slough with the higher capacity S332i pump. The result was a substantially higher water level and increased hydroperiod at all monitoring sites downstream of the S332i pump station. The hydroperiod at the NTS1 monitoring site increased by 78 days but still only averaged 80 days which demonstrates how the headwaters were still being drained by the canal system during this period. As flow through S174 increased, flow through S176 decreased by almost 50%. However, further downstream at S177 and S18C, flow increased during this period. This suggests that much of the water pumped into Taylor Slough made its way back into the canal system. The water level difference data, which showed the strongest water level gradient towards the canal during this period, support this assertion. Even though much of the water was recaptured by the canal system, it did spend time in the marsh substantially increasing hydroperiod throughout Taylor Slough.

The most recent operational change affecting the Taylor Slough area occurred in 2000 when the S-332B, C, and D pumps (each 500-cfs capacity) gradually became operational (D – late 1999, B – April 2000, C – June 2002). Operation of all three pumps was limited until early 2009 when the detention area system was completed. The operation of these pumps substantially changed hydrology in Taylor Slough north of the ENP road (SR9336). However, their operation had relatively little affect south of the road. Flow data at Taylor Slough Bridge and the C-111 Canal at structures S-177 and S-18C indicate about a 10% decrease after the beginning of 2000 compared to the 1992–1999 time period. Those decreases are attributed to decreases in average annual rainfall during the 2000–2010 time period compared to 1992–1999.

Table A8-1-1 provides a brief summary of the effects of recent operation changes in the Taylor Slough/C-111 Canal area. For a thorough description of water management operational and structural changes affecting the Taylor Slough/C-111 region, see Kotun and Renshaw (2013) and Lorenz and Kotun (in preparation).

Structural Changes

C-111 Spoil Removal

When the C-111 Canal was constructed in 1964–1966, spoil was deposited in mounds on the south bank of the canal, creating an irregular berm downstream of the southeastward fifty degree bend in the canal (between canals S-18C and S-197). When deposited, there were breaks between mounds for water to flow southward. In August of 1993, the gaps between spoil mounds on the western end of this segment of the C-111 Canal (between the S-18C structure and C-110 Canal) were cleared of debris and vegetation to augment flow toward Joe Bay and adjacent embayments. All spoil mounds were removed between January and October 1997. During that time, the 15 easternmost mounds were degraded to 2 feet National Geodetic Vertical Datum of 1929 (NGVD); the 35 westernmost mounds were degraded to 1 foot NGVD.

Table A7-1-1. Major operational change effects in central and northeast Florida Bay since 1981

Period	Flow in Taylor Slough (at Taylor Slough Bridge)	Flow in C-111 Canal	Stage	Hydroperiod	Rainfall/Taylor Slough Flow Relationship
S-332 (1981–1991)	Average annual flow was 28 million cubic meters (mcm); slightly higher than previous period.	Average annual flow at S-18C was 200 mcm. Wet and dry season flows increased significantly compared to previous period.	Increase at Taylor Slough Bridge from previous operation period due to S-331 pumping.	NTS1 = 2 days R3110 = 40 days TSB = 130 days P37 = 270 days (annual average)	Same as previous 2 operational periods.
S-332i (1992–1999)	Increased 3 times from previous period via increased flow at S-174 (flow at S-176 decreased by 50%).	Increased at S-177 and S-18C. So, some of flow increase at Taylor Slough Bridge ended up back in C-111 Canal.	Increased from previous period at all monitoring sites downstream of S-332i pump station.	Significant increase from previous period at all monitoring sites downstream of S-332i pump station. NTS1 = 80 days R3110 = 146 days TSB = 249 days P37 = 327 days (annual average)	Relationship changes; increased flow relative to rainfall due to increased pumping at S-332i and inflows from S-331.
S-332D (2000–2010)	Average annual flow decreased slightly (~10%) from previous period. Decrease due to slightly lower rainfall during the S-332D period.	S-332B,C,D pumps reduced flow at S-176 by 70%.	L31W stage at highest level of any period, pushing water into Taylor Slough near the detention areas. This effect diminishes quickly south of the detention areas. Much of the water recaptured by canal system south of detention areas.	Average hydroperiod increased significantly in the Rocky Glades. Water above ground surface entire wet season. NTS1 = 138 days R3110 = 164 days TSB = 177 days P37 = 314 days (annual average) Decrease at Taylor Slough Bridge by 72 days and by 13 days at P37, but still longer than any other period except S-332i.	Relationship same as S-332i period.

Results from a study conducted on the effects from the berm removal indicate that higher marsh stages were observed for the central (Eastern Panhandle Station almost due south of the C-110 and C-111 canals confluence) and western (upper and lower Joe Bay) marsh study sites after berm removal (Anderson et al. 1999). Differences in stage appear to be greatest at the lower Joe Bay site—marsh water levels after berm removal are approximately 0.1 to 0.4 feet higher than prior to berm removal, with before/after differences decreasing as marsh stage increases. However, the eastern marsh site (lower Highway Creek) indicated slightly lower stages (< 0.1 feet) after berm removal. Results suggest that more water is flowing into the Joe Bay and Trout Cove area and less water is moving to the eastern area of Long Sound following berm removal. It should be noted that the period of record from this study is limited to 19 months for the four stage stations after berm removal and 13 months to 7 years prior to berm removal. Also, the authors noted that local rainfall effects should be analyzed to further evaluate the effectiveness of removing the berm. To our knowledge, no analysis was ever conducted to determine the salinity effects of the spoil mound placement and their subsequent removal.

Taylor Slough Bridge Upgrade

In 1999, the ENP Road was modified where it crosses the narrow northern end of Taylor Slough. The culverts under the park road were replaced with two short bridge spans. More specifically, the two low bridges were built parallel to the old road and connected to the old road. The old roadbed and culverts were then removed. The project was completed in 2000. This modification presumably allows more flow along Taylor Slough, but no analysis shows how this change affected flow. One complication for isolating effects of the road modification is that upstream inflow patterns changed at the same time when the S-332B and S-332D pumps came online, as described above.

U.S. Highway 1 Widening Project

Circulation changes in the area of northeastern Florida Bay and adjacent wetlands to the north have been altered relatively recently by the widening of U.S. Highway 1 between Florida City and Key Largo. Culverts and wildlife passages under the road in both the bay and wetland sections of the project area were installed sometime between April 2005 and July 2009 (see **Figure A7-1-7** for locations of culverts and bridges). Most of the work in the Barnes/Blackwater/Little Blackwater Sound/Manatee Bay area occurred from 2005 to 2007. The small bridge between Manatee Bay and Long Sound opened about 2007 (**Figure A7-1-7**). The replacement of the drawbridge over Jewfish Creek with a much longer bridge span allowed the removal of the causeway that split Lake Surprise, allowing for water exchange through a creek on the north side of the lake and a "new" pass on the west side of the lake. This mostly affects the lake itself and it is unclear how much it affects exchanges between Barnes Sound and Blackwater Sound. In fact, it is unclear how the hydrology and circulation have been affected by any of the passages and structures associated with the road widening as limited analyses of these affects have been conducted. Anecdotal evidence indicates that exchanges between the embayments east/north of the road and those west/south of the road have increased since construction was completed (see the U.S. Highway 1 Expansion section in the main Southern Coastal Systems chapter).

Figure A7-1-7. Map showing locations of bridges and box culverts associated with the U.S. Highway 1 Widening project south of the C-111 Canal. Red dots show wildlife crossings north of the C-111 Canal.

Lower Southwest Florida Coast

A number of upstream structural and operational changes have affected flows exiting southwest SRS into the adjoining coast estuaries over the past century. Over the last three decades, the most significant change affecting SRS flows downstream of the Tamiami Trail occurred in the early 1990s. **Figure A7-1-8** shows the cumulative flow across the Tamiami Trail between the Forty Mile Bend and the L-67 Levee (i.e., flows at the S-12 water control structures) plotted as a function of cumulative rainfall between 1941 and 2008. Note that flow per unit rainfall almost doubled around 1990. Additional analyses indicates that this increase is likely due to an increase in flows entering Water Conservation Area 3 from the Everglades Agricultural Area, and is not a change in operations of the S-12 structures (personal communication with Kevin Kotun, ENP, April 16, 2013; Kotun 2012). It is unclear how much of this additional water exits SRS in the estuaries of the southwest Florida coast. Also, how much salinity may have changed in the southwest coast estuaries after 1990 cannot be assessed. The ENP Marine Monitoring Network initiated operations in 1991. Continuous salinity data are not available prior to 1991.

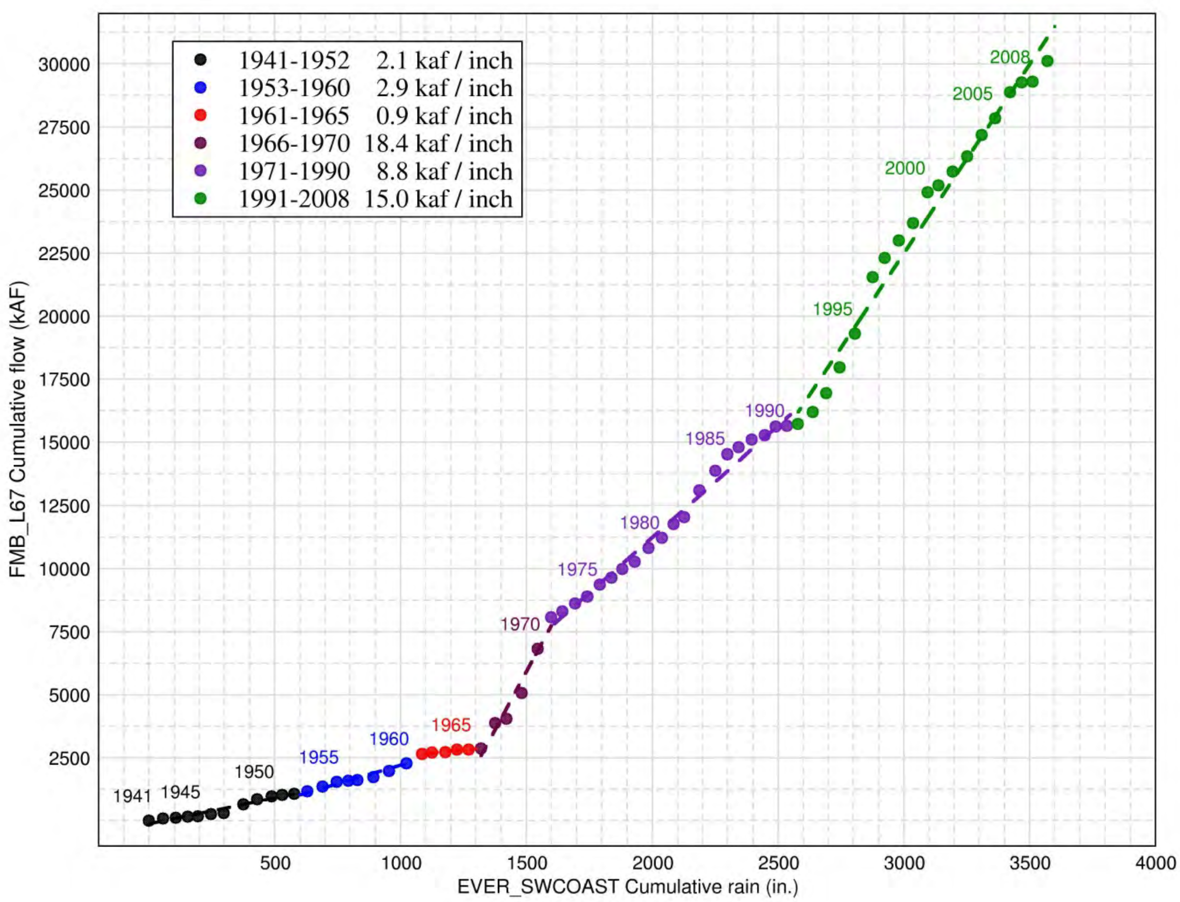

Figure A7-1-8. Cumulative flow across the Tamiami Trail between the Forty Mile Bend and the L-67 Levee (i.e., flows at the S12 water control structures) plotted as a function of cumulative rainfall between 1941 and 2008 (shown in thousands of acre-feet per inch).

Picayune Strand and Ten Thousand Islands Area

Southern Golden Gate Estates

The greatest changes to this area began in the 1960s with the development of the Southern Golden Glades Estates (SGGE) subdivision. The area was subdivided into rectangular plots and a network of 279 miles of roads was laid out on a quarter-mile grid. Roads were built above surrounding ground by excavation of borrow ditches on each side. The roads and ditches, oriented north-south and east-west, intercepted historic shallow flow paths, which were generally oriented in a north-northeast – south-southwest direction. To maintain a lower groundwater table and provide flood drainage, 48 miles of large artificial channels were built. The Merritt, Faka Union, Miller, and Prairie canals ultimately delivered all drainage to the lower end of the Faka Union Canal. These canals additionally provided conveyance for water drained from Northern Golden Gate Estates, located outside the main study area, north of Interstate 75. The operation of these large canals lowered groundwater throughout the SGGE landscape. Pre-drainage sheet flow was virtually eliminated; aquifer storage was reduced due to the generally lowered water table. Channeling all of the flow caused “shock load” discharges to the estuaries, releases of very large quantities of fresh water over a relatively small cross-sectional area of Faka Union Bay, during a relatively short time. Subdivision and road construction, as well as some land clearing on purchased lots inside the SGGE Project Area further changed the landscape by promoting invasion of upland and exotic vegetation, changes in species dominance in native communities, and increased wildfire. Beginning in 1985, the State of Florida started purchasing the SGGE lands with the purpose of restoring the hydrology and ecology of the area and combining the surrounding natural preserve units (USACE and SFWMD 2004).

Lucky Lake Strand, Stumpy Strand and Merritt Canal

Lucky Lake Strand and Stumpy Strand are found in the western portion of the National Panther Wildlife Refuge. Approximately half of Lucky Lake Strand is on private property. Lucky Lake and Stumpy strand's several thousand acres are chronically drained by Merritt Canal, which begins at the refuge's southwest corner and runs 20 miles south to the Gulf of Mexico. Eventually it merges with Lucky Lake Strand before flowing through refuge lands until its terminus at Interstate 75, where Merritt Canal takes inflows to the Gulf of Mexico through the Faka Union Canal System. At the time the refuge was established (June 1989) construction of I-75 (conversion from two-lane SR-84 to a four-lane interstate) was ongoing including various modifications in adjacent canals and canal interceptions. Although the Merritt Canal facilitated drainage of Lucky Lake and Stumpy strands prior to highway modifications when SR-84 was only a two-lane highway, post-Interstate conditions may have accelerated drainage. In short, Merritt Canal is an effective drain to the current wetland system. Once wet season rains diminish, relatively rapid removal of sheet flow water through Lucky Lake Strand is accomplished, shortening the hydroperiod of that system to a significant degree. Subsurface drainage also occurs when canal elevations drop sufficiently (USFWS 2001).

The purpose of the Lucky Lake Strand Restoration Project was to restore historic wetland functions within Lucky Lake and Stumpy strands. Since 1990, the need to return historic hydroperiods to the strands was observed as a result of chronic overdrainage caused by the Faka Union Canal System, including Merritt Canal, along with the construction of SR-84 in 1966 and conversion to I-75 in 1990 (USFWS 2001).

The South Florida Water Management District and the United States Fish and Wildlife Service cooperated in an effort to restore the hydroperiods within the Lucky Lake and Stumpy strands to their historic levels. In September 1994, the United States Fish and Wildlife Service signed a Grant Agreement with the South Florida Water Management District to provide assistance to construct a low-head water control structure on the south side of I-75 on Merritt Canal. The United States Fish and Wildlife Service provided funds for the construction of this Lucky Lake Water Control Structure (**Figure A7-1-9**). Construction costs above this amount and structure maintenance were carried by the South Florida Water Management District. A site approximately 3,600 feet south of I-75 was chosen where the land adjacent to the canal was state-owned, allowing construction access and maintenance. In July 1999, the Lucky Lake Water Control Structure was completed. It is is an adjustable crest weir with 8 operable steel gates (USFWS 2001). Its location is indicated in **Figure A7-1-9**.

Figure A7-1-9. Map showing locations of Lucky Lake Water Control Structure (Lucky Lake Weir) and Merritt Pump Station.

Faka Union Canal

The present day discharge from the Faka Union Canal (**Figure A7-1-10**) experiences extreme seasonal variations, resulting in large fluctuations in the salinity levels of the receiving bays. Canal discharge records measured at the United States Geolgogical Survey gauging station (FU-1), located upstream from the outfall weir of the Faka Union Canal, are available starting in 1969. The average discharges for the period of record were 115 cfs during the dry season (November through May) and 460 cfs during the wet season (June through October), with an extreme discharge of 3,200 cfs occurring right after the canals were built (USACE and SFWMD 2004).

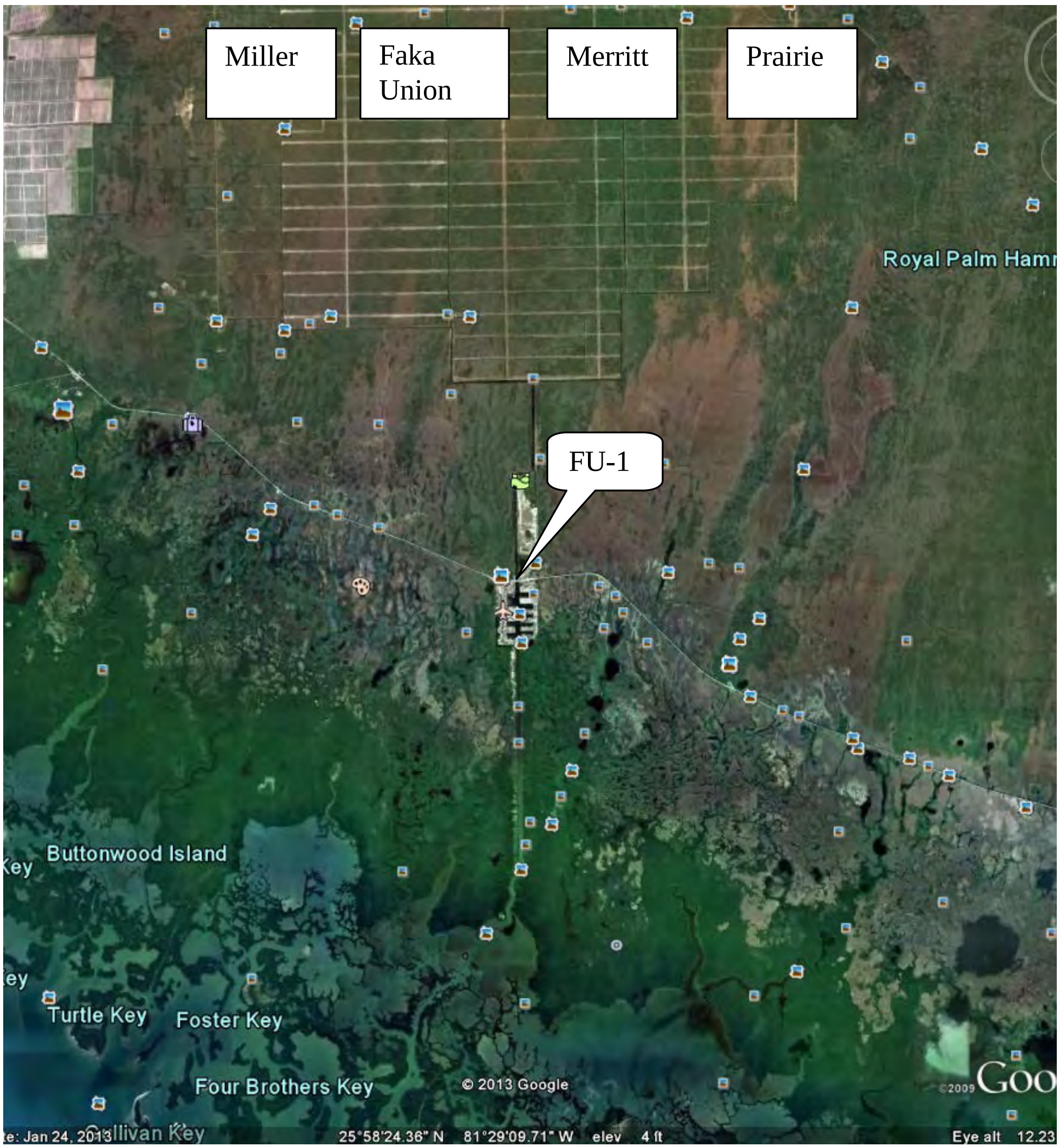

Figure A7-1-10. Picayune Strand Restoration Project area showing drainage canals (top of image) and Faka Union outlet structure (FU-1).

Current point source discharge patterns cause enormous shocks to the aquatic biota of Faka Union Bay and deliver too little freshwater input to the saline areas in surrounding bays. During drainage pulses, estuarine salinity rapidly declines to near freshwater conditions. These conditions have caused prolonged salinity stresses and have eliminated or displaced a high proportion of the benthic, midwater, and fish plankton communities from the bay. Such suppressed plankton development has resulted in very low relative abundance of midwater fish and also a considerable drop in shellfish harvest levels. The impact on commercial and recreational fisheries has been very significant. Comparisons of oyster physiology and ecology clearly demonstrate that water management practices, specifically the impacts of freshwater inundation from the uncontrolled draining and channeling of the wetlands within SGGE, have adversely affected oysters and the development of oyster reefs (USACE and SFWMD 2004). Seagrass meadows are no longer a prevalent habitat type in the bay. Instead, bare, sandy mud and algal areas predominate (USACE and SFWMD 2004).

Water levels in the Faka Union Canal, which have been regulated by Faka Union Canal Weir #1 (FU-1), Faka Union Canal Weir #2 (FU-2) and Faka Union Canal Weir #3 (FU-3), will be regulated by the Faka Union Pump Station possibly beginning in October 2014. FU-3 and FU-2 on Faka Union Canal will be removed. FU-1 is a non-adjustable fixed crest weir that will remain after project construction. The Faka Union Canal features of the Picayune Strand Restoration Project (PSRP) include the Faka Union Pump Station (S-487) and its tieback levee and plugs in the Faka Union Canal downstream of the pump station. The pump station will be located on the Faka Union Canal approximately 2.5 miles south of I-75. It will maintain the upstream flood protection levels and lift water to the distribution canal and spreader berm weirs for the restoration of historic sheet flow downstream. The pump station has a total capacity to pump 2,650 cfs and was designed to approximate the function of the existing FU-3, which will be removed as part of the project. It will compensate for the loss of gravity drainage due to canal plugging. However, its construction is not yet complete. Also, the Miller Pump Station and Miller Canal features have not yet been constructed.

Prairie Canal

The longest reach of Prairie Canal, which runs north-south, was plugged fall 2006 resulting in rehydrated wetlands immediately adjacent to Prairie Canal. The other segments of Prairie Canal will be plugged as part of the overall PSRP. Near the south end of the PSRP project area, the Prairie Canal merges with the Merritt Canal, and the Merritt Canal and Miller Canal merge with the Faka Union Canal. Prairie 1, a fixed crest weir with stop logs, was removed during the Prairie Canal plugging.

Merritt Canal

The Merritt Canal features of PSRP include the Merritt Pump Station (S-488) and its tieback levee and plugs in the Merritt Canal downstream of the pump station. The pump station is located on the Merritt Canal approximately 0.8 miles south of I-75. It will maintain the upstream flood protection levels and lift water to the distribution canal and spreader berm weirs for the restoration of historic sheet flow downstream. It has a capacity of 810 cfs and was designed to replace the function of the Lucky Lake Weir. The pump station will compensate for the loss of gravity drainage due to canal plugging. Water levels in the Merritt Canal, which have been regulated by the FU-1, Merritt 1 and Lucky Lake weirs, will be regulated by the Merritt Pump Station possibly beginning in October 2013. The Lucky Lake Weir and Merritt 1 structures on Merritt Canal will be removed.

Picayune Strand Restoration Project Outflow

The outlet structure for the PSRP area is FU-1. Results of analysis of headwater stage and flow data at FU-1 from October 1984 to July 2013 are shown in **Table A7-1-2**. FU-1 flow and FU-1 headwater stage data from station "Faka_H" are shown respectively in **Figure A7-1-11** and **Figure A7-1-12** for the periods from October 1984 to July 2013.

Table A7-1-2. Major headwater stage and flow effects at FU-1 since 1984.

Period	FU-1 Flow (cfs)	FU-1 Headwater (station Faka_H)
October 1984–April 1998	Average: 289 cfs Standard Deviation: 333 cfs Peak flows tend to be lower than in below period	Average: 2.44 Standard Deviation: 0.54 Stages tend to vary less widely than in below period
May 1999–July 2013	Average: 389 cfs Standard Deviation: 461 cfs Peak flows tend to be higher than in above period	Average: 2.48 Standard Deviation: 0.70 Stages tend to vary more widely than in above period

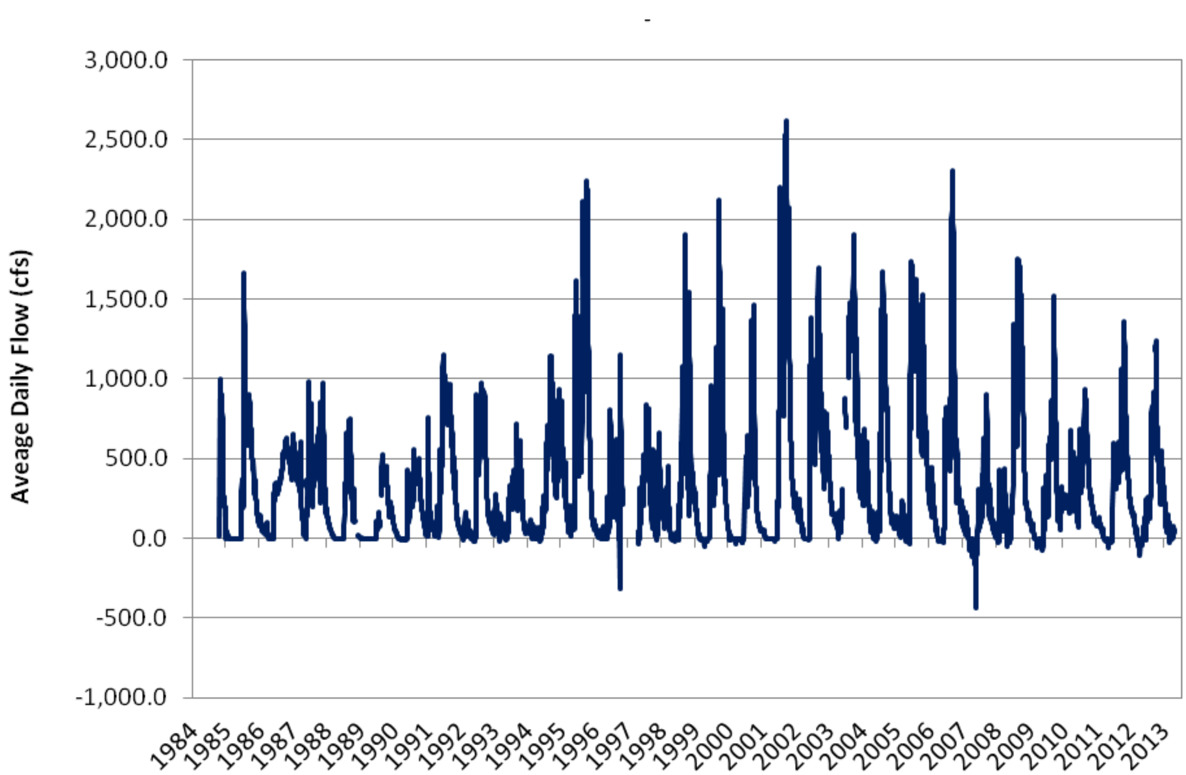

Figure A7-1-11. FU-1 flow graph (May 1, 1984 to April 30, 2013).

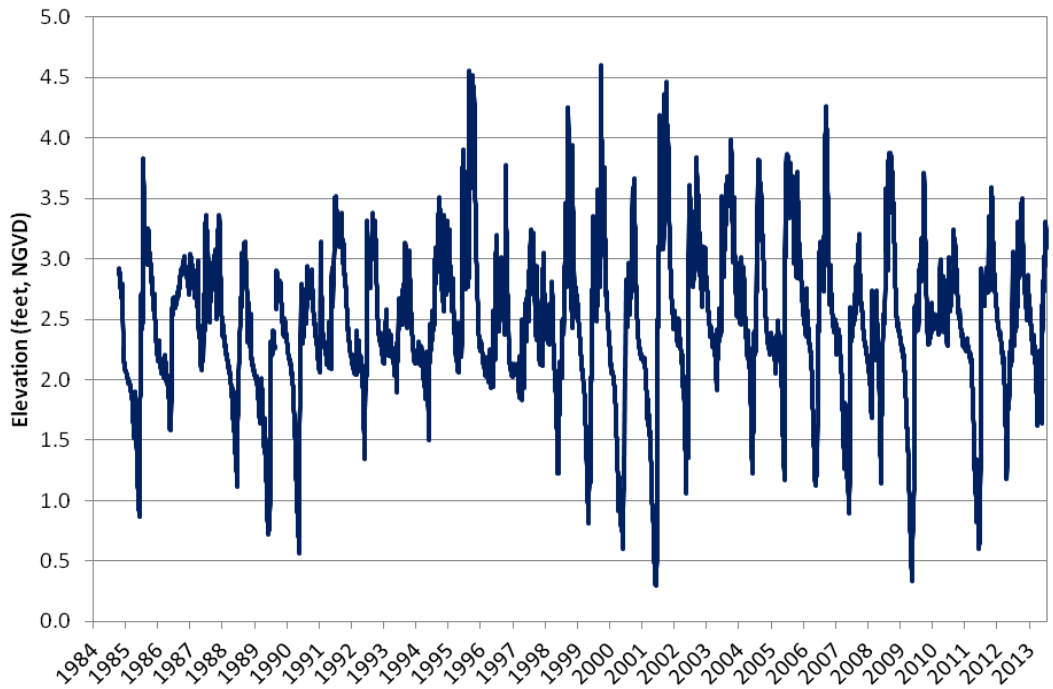

Figure A7-1-12. FU-1 headwater stage graph (May 1, 1984 to April 30, 2013).

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APPENDIX 7-2: INTER-CONNECTIVITY OF SOUTHERN COASTAL SYSTEMS WITH UPSTREAM ENVIRONMENT

The Southern Coastal Systems region is inextricably linked to the upstream environment of the Greater Everglades. This linkage provides the fresh water that establishes and maintains brackish salinities that are critical to the native estuarine flora and fauna of the region. This appendix describes the physical inter-connectivity between the southern estuaries and the upstream watershed to provide a better understanding of factors controlling freshwater movement between the regions. Both historical and present day connectivity are provided, and each subregion of the Southern Coastal Systems is treated separately as each subregion has different connectivity characteristics.

Biscayne Bay

The Biscayne Bay watershed region can be characterized by four primary physiographic zones situated contiguously from east to west: the mangrove and coastal wetlands, the Atlantic Coastal Ridge, the Rocky Glades, and interior Everglades (Sonntag 1987, Fish and Stewart 1991) (**Figure A7-2-1**). The bay and its associated coastal glade wetlands are mostly separated from the Everglades by the Atlantic Coastal Ridge. This ridge generally parallels the coast and has a width of about two to ten miles. The ridge varies in elevation from eight feet above sea level in the south to 22 feet in the north and forms a natural barrier to drainage of the interior Everglades, except where breached by canals, rivers, or sloughs (also called transverse glades). East and south of the ridge are the Coastal Glades and mangrove wetlands. This zone was formerly characterized by low-lying wetlands but was drained for farming and urban development. West of the ridge, there is a flatland area of freshwater marshes over thin rocky soils about four miles wide (Rocky Glades), then the interior Everglades extends 40 or so miles inland. The interior Everglades elevations range from four to thirteen feet above sea level.

Historically, water levels at and just west of the Atlantic Coastal Ridge were several feet higher than they are today (Parker et al. 1955). This drove flow eastward to the bay through the rivers and transverse glades (**Figure A7-2-1**). Once on the east side of the ridge, the water spread out into broad sheet flow across the extensive Coastal Glades wetlands, then rechannelized somewhat into a complex network of creeks near the bay. Since the morphology of the creeks was shallow, narrow, and sinuous, flow velocity was not large and did not ordinarily vary greatly from day to day. The continuous flow resulted in a range of salinity patterns from fresh to brackish near the coast. Also, the sheet flow across the Atlandic Coastal Ridge and Coastal Glades was relatively slow and steady, which maintained brackish salinity conditions in the nearshore waters of the bay well into the dry season.

Groundwater is an important component of the Biscayne Bay regional hydrology and it is greatly influenced by the highly permeable surficial Biscayne Aquifer. Under pre-development conditions in western Miami-Dade County, water entered the limestone aquifer by lateral movement from Broward and Collier counties and by downward seepage from the Everglades and the Biscayne Aquifer, and moved southward and southeastward into Miami-Dade County to coastal discharge areas. Groundwater flow direction in the Biscayne Aquifer inland was primarily to the south and southeast. In eastern Miami-Dade County, the seasonal groundwater ridge that formed under pre-development conditions supported both easterly and westerly groundwater flow away from the ridge axis (Fish and Stewart 1991). Overall, surface water makes up the largest input of fresh water to Biscayne Bay; however, groundwater influence becomes proportionally greatest at the end of the dry season, typically in April and May. Groundwater discharge to Biscayne Bay occurs in two ways: seepage from the aquifer and flow through subsurface leakage channels (Parker et al. 1955).

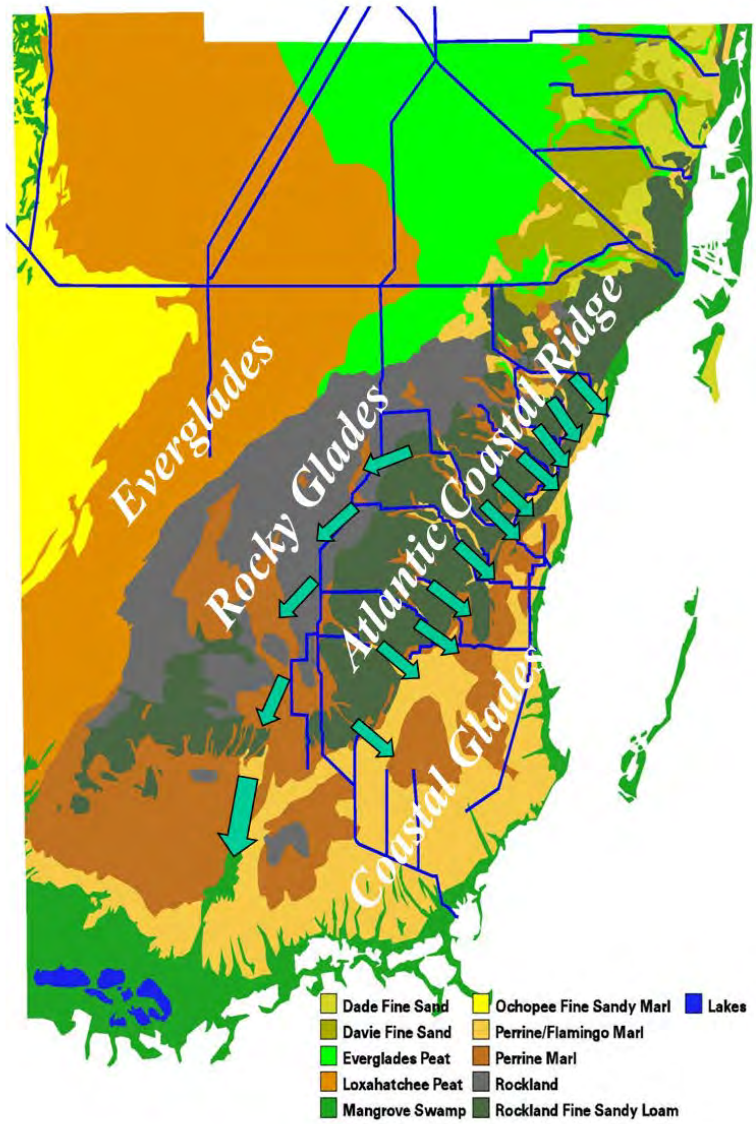

Figure A7-2-1. Schematic showing the four primary physiographic zones of southeastern Everglades and adjacent area, including historic freshwater flow pathways. The present day canal system (blue lines) is included for reference.

Currently, the water elevations at the ridge are controlled at less than three to four feet above mean sea level year round, and flow from the Everglades to the coast is heavily managed through a system of canals, levees, and water control structures. Drainage as a result of extensive canal systems and large-scale pumping from municipal well fields has greatly altered the pre-development flow system in eastern Miami-Dade County by (1) eliminating or greatly reducing a seasonal and coastal groundwater ridge; (2) reducing groundwater flow in the lower portion of the Biscayne Aquifer; (3) reducing or eliminating seasonal westward movement of groundwater; (4) changing the pattern of freshwater discharge into the bay from long, slow releases over a broad front to “pulse” releases from canals following rain events; and (5) lowering the water table and inducing saltwater intrusion. In addition, this management has created an abrupt difference between saline and freshwater habitats. The very low and moderate salinity habitats have become uncommon along the bay’s western shore. Rapid drainage afforded by the canals and the reduced water table now maintained in the study area have reduced the functional habitat value of the remaining freshwater wetlands. Hydroperiods that were once as long as an entire year have been reduced to days or weeks.

Saltwater intrusion into the aquifer remains a continuing problem due to the introduction of drainage canals and increased groundwater withdrawal to satisfy potable water demand. Water management programs have been implemented to control the intrusion rate. A network of canals and control structures provides for water and salinity control in the area. Salinity monitoring sensors near the coastal structures indicated that the installation of salinity control structures were effective in controlling saltwater intrusion decades ago. Wellfields, which are the source of municipal water supplies, are significantly recharged by water from the Water Conservation Areas that are located northwest of the project area. Water stored in the Water Conservation Areas is used to maintain groundwater levels in the coastal area for public water supply, to irrigate the vast agricultural areas interspersed within the project area, and to maintain a freshwater head along the lower east coast for salinity control. Minimum stages are maintained in lower east coast canals, principally to provide the volume of water needed to protect the Biscayne Aquifer from saltwater intrusion. The head created in the canals raises groundwater levels, recharging the aquifer and the urban wellfields

Card Sound, Barnes Sound, Manatee Bay, Long Sound, Little Blackwater Sound and Blackwater Sound

Card Sound, Barnes Sound, Manatee Bay, Long Sound, Little Blackwater Sound, and Blackwater Sound are the southeastern-most estuaries on the coast of Florida, and are located between Biscayne Bay and Florida Bay (**Figure A7-2-2**). Card Sound has an open connection to Biscayne Bay. The others are mostly isolated from each other and from Florida Bay to the south and west (Marshall et al. 2008). Barriers to flow include natural mangrove covered shoals and the Card Sound Road and U.S. Highway 1 causeways. Several relatively small breaks in the shoals and causeways exist allowing some exchange of water, the largest opening being the dredged channel for the Intracoastal Waterway.

Figure A7-2-2. Map of Manatee Bay and Barnes Sound.

The Everglades and the Coastal Glades (**Figure A7-2-1**) are sources of fresh water to Card Sound, Barnes Sound, Long Sound, and Manatee Bay. Under natural conditions, the southernmost transverse glades through the Atlantic Coastal Ridge provided a direct connection for the Coastal Glades to the freshwater marshes of the Rocky Glades in the Greater Everglades. Fresh water in the Coastal Glades also comes from precipitation or emergent groundwater. Intruding salt water into fresh coastal water table aquifers can force fresh or brackish water in the water table aquifer to emerge above ground and flow overland to the estuarine receiving water (Nuttle and Portnoy 1992, Nuttle and Harvey 1995). Before the diversion of fresh water for agricultural or urban uses, when water levels were high in Shark River Slough (SRS) within the Greater Everglades and rainfall was abundant, fresh water could stage in the Rocky Glades then flow through the transverse glades into the Coastal Glades. Under natural conditions, the combination of local precipitation and Greater Everglades-sourced fresh water discharged through local, small-scale coastal creeks and streams into Card Sound, Barnes Sound, Manatee Bay, and Long Sound. The very low slope of the Coastal Glades meant that fresh water could pond there under natural conditions as the wet season progressed, to be discharged into Card Sound, Barnes Sound, Manatee Bay, and Long Sound, continuing well into the dry season.

The currently managed freshwater flows to the southeastern-most estuaries are dominated by the influence of the C-111 Canal and the extension of the L-31E Canal. In the northern areas of the Card Sound watershed, the management of water for the existing power plant has altered the connection of Card Sound to the hydrology of the watershed. These drainage ditches remove ponded water in the Coastal Glades and groundwater from the porous substrate, collecting like freshwater horizontal wells. The general result throughout most of the Coastal Glades is drier conditions than the historic, nonmanaged system.

Under the managed regime, fresh water is concentrated in the C-111 Canal, controlled by the S-18C structure, then directed to the southeast and discharged through the S-197 structure into Manatee Bay. When the S-197 is open, it can rapidly reduce the salinity in Manatee Bay to less than 10 practical salinity units. Some of the water in the C-111 Canal flows over the southern bank towards estuarine Long Sound. The C-111 Canal also collects and diverts ponded Coastal Glades water towards the east that may have naturally flowed into Joe Bay. Fresh water is also trapped behind (west of) U.S. Highway 1 creating much wetter local conditions.

In the current managed-flow system, the volume and timing of freshwater flows being discharged to the estuaries is not always based on the amount and pattern of precipitation, as was the case for the natural system. For example, the needs of drainage for the agricultural activities west and southwest of Homestead may require a discharge of fresh water from the C-111 Canal when wet season precipitation is ending, which can release large quantities of fresh water into Manatee Bay and Long Sound over a short period of time. In the meantime, Joe Bay is starved for fresh water. Under natural conditions, fresh surface water in the Coastal Glades would have been stored as ponded water and released slowly into Barnes Sound, Manatee Bay, and Long Sound. Many believe that this unnatural temporal and spatial distribution of fresh water being discharged from the Coastal Glades has negatively- affected the water-based ecology of the Coastal Glades, Joe Bay, Long Sound, Barnes Sound, and Manatee Bay.

Fresh water was also historically stored as a water table aquifer in the sands of the Atlantic Coastal Ridge. Some of this groundwater emerged in the Coastal Glades. Water management has depleted this freshwater source. The effect of loss of this freshwater source is exacerbated by the function of the C 111 Canal because the remaining Coastal Glades fresh water, which now comes primarily from precipitation, is collected and directed away, instead of forming a dispersed, slower moving freshwater front towards Long Sound, Barns Sound, and Manatee Bay.

Therefore, in these southeasternmost areas of coastal Florida the natural, relatively slow-reacting, dispersed hydrologic connection between fresh water in the Greater Everglades and the estuaries has been replaced by a short-circuiting flow path for fresh water flowing to the estuaries in a more concentrated manner. This has reduced the storage of fresh water in the Coastal Glades, concentrated fresh water in some areas, and starved fresh water flow to other areas, resulting in altered spatial and temporal patterns of salinity in the estuaries.

Florida Bay

The natural connection of Florida Bay to the Greater Everglades was primarily through the discharge of fresh water from Taylor Slough and freshwater contributions from the west end of the Coastal Glades into the nearshore embayments of eastern and central Florida Bay (Joe Bay, Little Madeira Bay, and Terrapin Bay; **Figure A7-2-3**). Freshwater flow from Taylor Slough into Florida Bay has been impacted not only by the reduction of water in the Greater Everglades by regional water management but also by the loss of emerging groundwater from the water table under the Atlantic Coastal Ridge. Joe Bay naturally received some fresh water from the Coastal Glades that is now diverted by the C-111 Canal.

Ponded water in the Rocky Glades of the Greater Everglades north of the pineland keys in Everglades National Park (ENP) contribute to the flow in Taylor Slough as a result of high stages in SRS and/or local precipitation on the Rocky Glades. Water management upstream has reduced water levels in SRS within the Greater Everglades, thereby reducing the opportunity for this source of water to contribute to freshwater flows in Taylor Slough, and consequently to discharge into Florida Bay. Loss of the Atlantic Coastal Ridge groundwater source of fresh water to the Rocky Glades has also contributed to lower stages in this wet prairie. It is inferred from paleoecological evaluations in Florida Bay that water levels in SRS before the alteration of natural freshwater flows in the Greater Everglades may have been 0.5–0.8 feet higher than the current observed levels, which would require about two times more flow than is currently being discharged (on average) across Tamiami Trail into SRS (Marshall et al. 2009, Marshall and Wingard 2012, Marshall et al. in press).

Paleoecological evaluations also indicate that the flow of fresh water into Taylor Slough may be three to four times less than natural conditions (Marshall et al. 2009, Marshall et al. in press). Even so, recent work by Zapata-Rios and Price (2012) indicates that groundwater discharges into Taylor Slough still contribute about 27% of the freshwater inputs under current conditions. Therefore, Taylor Slough still has a relatively important connection to groundwater flowing from upstream sources.

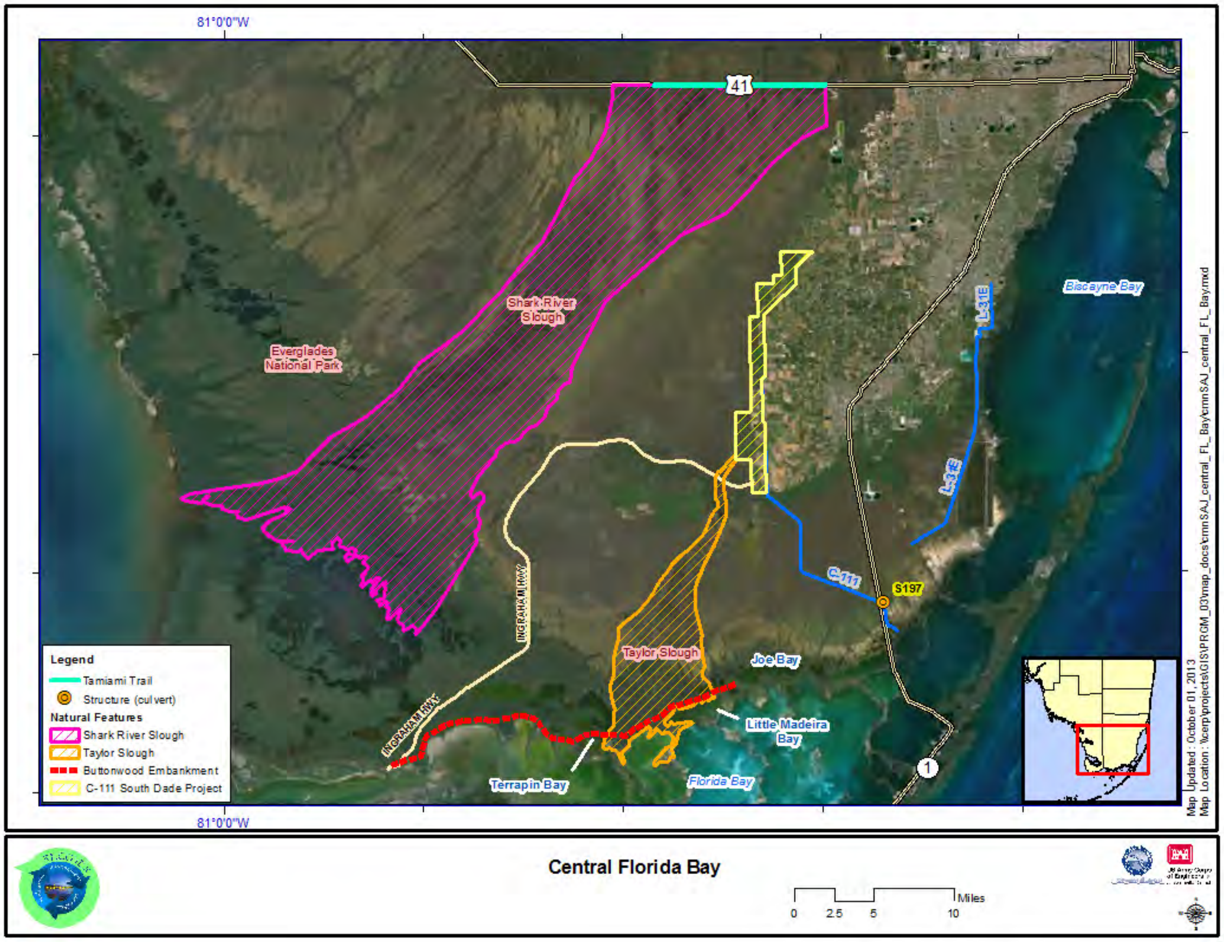

Figure A7-2-3. Map of central Florida Bay.

The loss of freshwater flow into Joe Bay, Little Madeira Bay, and Terrapin Bay has increased the salinity throughout central Florida Bay, although the increased flow of managed fresh water into east Long Sound may have moderated this loss of fresh water to some extent. However, managed freshwater flows directed into extreme northeastern Florida Bay typically come at an unnatural time to be beneficial to the ecology of central Florida Bay. The estuarine ecology of Florida Bay benefits from the sustained, dispersed flow of fresh water past the end of the wet season, thereby allowing for the existence of oligohaline to mesohaline conditions well into the dry season.

To the west, freshwater flows from western Taylor Slough can pond in the coastal prairie behind the Buttonwood Embankment along the shoreline of central Florida Bay because of the lack of coastal creeks such as those that naturally drain the Coastal Glades to the east. The only locations of discharge into central Florida Bay for this trapped western Taylor Slough water are McCormack Creek discharging into Terrapin Bay and Alligator Creek discharging into Garfield Bight. The residence time for this trapped water in the mangrove areas and open waters of Seven Palm Lake, Cuthbert Lake, Middle Lake, Monroe Lake, The Lungs, Henry Lake, and West Lake is naturally high. During the dry season, and particularly during drought periods, hypersaline conditions have been routinely recorded. The extent to which natural freshwater flows from the Greater Everglades via Taylor Slough mitigated the creation of hypersaline conditions in this area in the past is only now being understood (Frankovich et al. 2011, 2012). The oligohaline conditions in the observed data from Roberts River and North River just to the west may indicate that the direct influence of fresh water discharging from SRS on the salinity of the West Lake/Cuthbert Lake/Seven Palms Lake area was historically small.

The results of Florida Bay paleoecological evaluations indicate that the salinity in Terrapin Bay and Little Madeira Bay has been impacted greatly (i.e., increased) by the reduction in Taylor Slough flows. Inferred results from these studies suggest that, under natural conditions, Taylor Slough flows may have been two to three times higher than the current conditions (Marshall et al. 2009, in press, Marshall and Wingard 2012).

Lower Southwest Coast

The largest natural channel for freshwater flows in ENP is SRS, which discharges into the Gulf of Mexico and Whitewater Bay via fresh and brackish water rivers that are characterized by vast stands of mangroves along the banks (**Figure A7-2-4**). SRS is a dominant feature in the Greater Everglades landscape and freshwater flows into SRS are controlled by the Tamiami Trail causeway and the structures that allow and control flow underneath the roadway. However, once freshwater enters SRS in ENP it is mostly unaffected by any water management features. The mangrove-lined rivers ("mangrove rivers") of the ENP west coast are wider, deeper, and longer than the coastal creeks that drain the Coastal Glades into the nearshore embayments of northeastern Florida Bay. Reasons for this may be the very large flow volume of SRS compared to Taylor Slough flow, and the addition of emergent groundwater from the Biscayne Aquifer and local precipitation. Compared to the relatively small tidally-induced flow and stage in Florida Bay due to the dampening effects of mud banks, tidal effects along the lower southwest coast estuaries are larger. These larger tidal effects, combined with larger freshwater outflows from SRS, likely contribute to the lack of a slightly higher elevation belt along the shoreline as is seen along the Florida Bay shoreline. Differing wind patterns and sediment composition may also play a role. In addition, the effect of local rainfall on the estuarine reaches of the mangrove rivers is small due to the small surface area of the stream. For these reasons, there is a more direct effect of Greater Everglades hydrology on the salinity in the estuarine reaches of the mangrove rivers on the Gulf Coast compared to the nearshore areas of Florida Bay.

Preliminary paleoecological investigations in the estuarine areas of the ENP Gulf Coast indicate that the salinity regimes in the mangrove rivers of the Gulf Coast have also been affected by the reduction of stages in the Greater Everglades (SRS) due to water management diversions (Frank Marshall, personal communication). Initial results indicate that the effect of water management on salinity is not as severe as the effect of water management on salinity in Florida Bay. These paleoecological analyses are currently underway and should be completed in time for the results to be included in the next System Status Report.

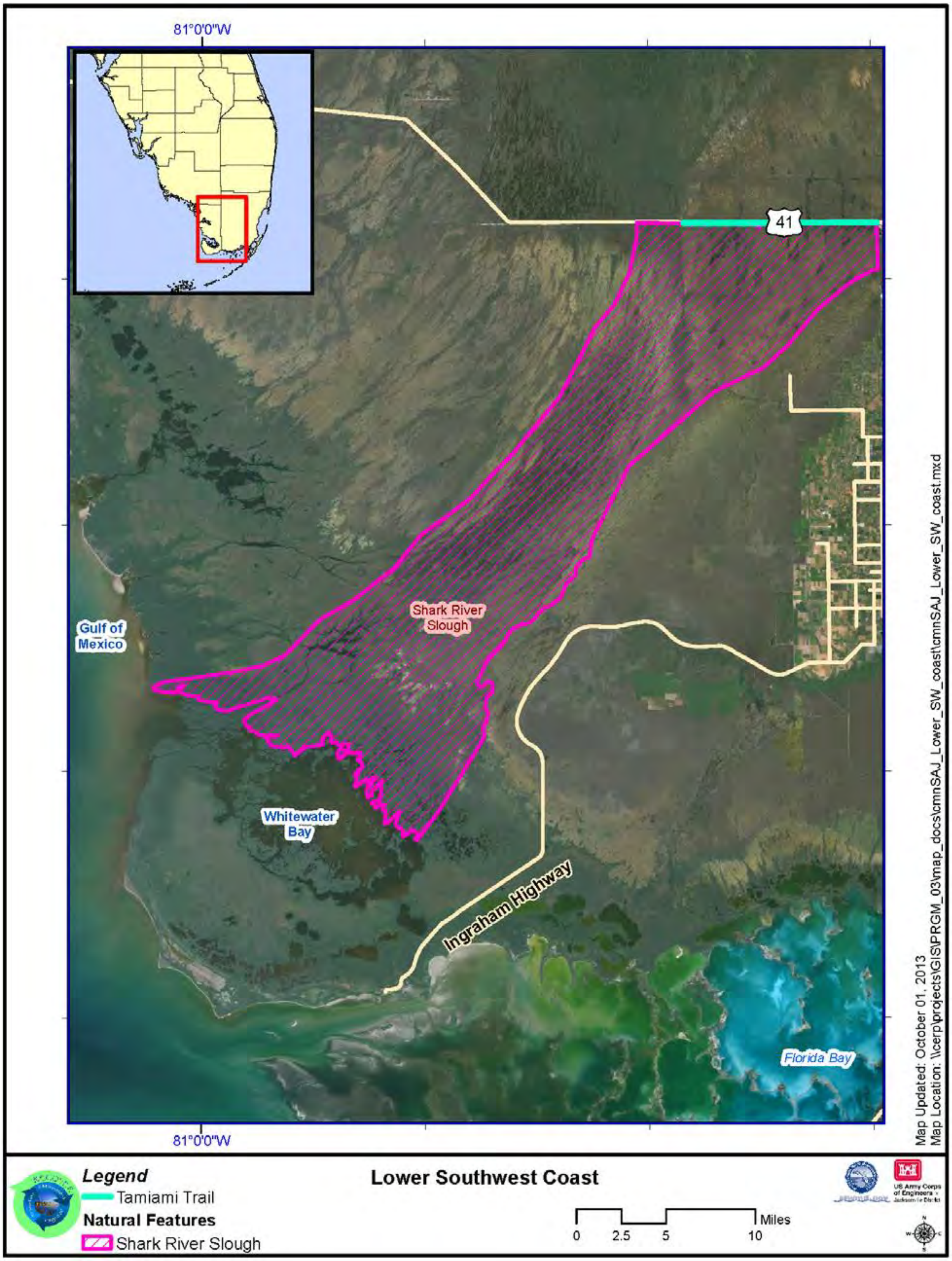

Figure A7-2-4. Map of the Lower Southwest Coast embayments.

Ten Thousand Islands

The Ten Thousand Islands area of the upper southwest Florida coast receives drainage primarily from Big Cypress National Preserve, although there is no defined freshwater slough similar to SRS (**Figure A7-2-5**). The hydrologic connection between the upstream Big Cypress Preserve is still relatively unchanged from historic conditions. Although the freshwater drainage patterns of this area have not been widely studied, there are some small coastal creeks that direct fresh water into a series of open water areas within the mangrove zone that ultimately flow through other creeks and rivers to discharge into the Gulf of Mexico. The connection between the upland drainage area in Big Cypress and the downstream estuarine portions of the discharging rivers and bays in the Ten Thousand Islands area is direct but dispersed. Emergent groundwater may also be an important freshwater source in this part of the Gulf coast.

The Picayune Strand Restoration Project area lies upstream from and discharges into the Ten Thousand Islands area. The connectivity between the project area and the downstream estuaries has been significantly altered by man. Picayune Strand is an upland/wetland area with a historically persistent high water table that is located north of U.S. Highway 41. Drainage for this area was created by a series of canals that were intended to reduce the water table to the point where a large urban development project could be sited. However, the development failed, and the land went into public ownership and became the Picayune Strand Restoration Project.

The Picayune Strand water management system includes Faka Union Canal that provides a point source discharge into Faka Union Bay. Discharges from this canal can drastically reduce the salinity in Faka Union Bay. Therefore, there is a direct connection between the freshwater flows from the Picayune Strand water management system and the salinity in Faka Union Bay and the adjacent, connected estuarine areas. The water management system in Picayune Strand is in the process of being retooled to reduce the impact of the freshwater discharge on the estuarine areas of Faka Union Bay and adjacent areas.

Figure A7-2-5. Map of the Ten Thousand Islands.

Summary

There is significant evidence from the observed data, the output from hydrologic and salinity models, vegetation maps, and the results of numerous studies that there is a connection between the hydrology (stage and flow) in the Greater Everglades and the salinity in the estuarine areas of south Florida. Natural connections in the east, where urban and agricultural development has occurred, have been altered by water management.

Significant alterations to the connection between the Greater Everglades and the salinity in Barnes Sound, Manatee Bay, Long Sound, and eastern Joe Bay include the loss of underflow from the water table aquifer that once existed near the Atlantic Coastal Ridge and the diversion of large volumes of water from the Coastal Glades by the C-111 Canal. In western Joe Bay, Little Madeira Bay, and Terrapin Bay (and perhaps Garfield Bight) the large reduction in Taylor Slough flows and stored fresh water has increased average salinities, which translate downstream to elevated average salinity values in central Florida Bay. Under current conditions, the connection between the Greater Everglades and Taylor Slough remains but is significantly altered, and the impact appears to be the greatest in the Seven Palms Lakes region of ENP where hypersaline conditions occur regularly.

On the ENP west (Gulf) coast, the connection of SRS to the mangrove rivers is still relatively strong, although paleoecological studies indicate that water levels and flows in SRS are significantly lower than pre-water management times. However, the hydrologic connection between surface water in SRS /Rocky Glades and surface water in Taylor Slough, which may have only been intermittent historically, is now weak and mostly associated with groundwater.

In the Ten Thousand Islands area, the hydrologic connection between the upstream Big Cypress National Preserve is still relatively unchanged from historic conditions. In Picayune Strand, the man-made drainage system creates a streamlined surface connection between the hydrology in Picayune Strand and the salinity in Faka Union Bay. Historically, that connection would have been primarily through groundwater and perhaps slow meandering surface flow via small creek networks. Any increase or decrease in discharges from the drainage system can rapidly and significantly affect the salinity in the downstream estuarine waters.

The complexity and importance of restoring the original hydrological and salinity regimes of Greater Everglades-Florida Bay ecosystem necessitates coordination and cooperation between researchers and managers of the Greater Everglades and Southern Coastal Systems regions for the success and sustainability of the Comprehensive Everglades Restoration Plan and the ecosystem as a whole. Performance measures and targets for both regions should be implemented with both freshwater and estuarine ecology objectives in mind. The alternative could result in the loss of the unique, interconnected water-based ecology that provides so many ecosystem services to all.

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APPENDIX 7-3: OTHER ENVIRONMENTAL EFFECTS

A variety of other environmental factors impact the Southern Coastal Systems including sea level rise, hurricanes, drought, fire, non-native species, and climate change. These factors and their effects on the region are summarized in this appendix. Another important environmental factor affecting the region is the occurrence of cold events. Since the last System Status Report, a severe cold event significantly affected south Florida in 2010 (see the Effects of the 2010 Cold Event on Everglades Biota section of the main Southern Coastal Systems chapter).

Sea Level Rise

The marshes and mangrove forests that form much of the natural coastline of south Florida developed and stabilized during the last 3,000 years (Willard and Bernhardt 2011)—a period of relatively slow rates of sea level rise averaging approximately 4 centimeter (cm) per century in the region (Wanless et al. 2000). Since 1930, however, this rate has accelerated to approximately 20-40 cm per century (Wanless et al. 1994), which has led to a destabilization of the coastline. As the marine waters transgress into the marshes and mangroves, the ecotones shift landward (Wanless et al. 2000, Krauss et al. 2011, Jiang et al. 2011).

The relative rate of rise will determine the response of the coastal system. Model results suggest that the global rate of sea level rise may accelerate in the twenty-first century. The Intergovernmental Panel on Climate Change projects that worldwide sea level will rise between 20 cm and 60 cm during the twenty-first century (IPCC 2007). At a rate of about 30 cm per 100 years, the seaward margin of all coastlines in south Florida will be erosional. Although mangroves may be able to maintain elevated areas by forming embankments, saline marshes will migrate landward, coastal embayments will deepen and their salinity will increase, and the formation of new inlets through the Florida Keys will negatively impact corals (Wanless et al. 1994). Intergovernmental Panel on Climate Change projections, however, do not include factors such as ice sheet flow dynamics that could significantly increase the rate of sea level rise (IPCC 2007). The more recent Copenhagen Diagnosis (Allison et al. 2009) states that the Intergovernmental Panel on Climate Change 2007 report underestimated sea level rise and that it may be as much as twice what has been projected. At a rate of 60 cm per 100 years, saline water will overstep the coastal embankments encroaching on freshwater wetlands. A rapid loss of transitional and freshwater environments will occur; erosion of the coastal embankment will increase turbidity and nutrient levels in the coastal zone; mud banks within the estuaries will erode, creating a more open water but more turbid environments; and water depth in the coastal zone will increase and cause substantial changes to benthic communities (Wanless et al. 1994).

Hurricanes

Hurricane frequency is greater in south Florida than in any other part of the United States (Gentry 1984). The National Oceanic and Atmospheric Administration reports that for the period between 1900 and 2010, Florida had the highest hurricane strike density for the United States, with 32 hurricanes recorded for Monroe County, 25 for Miami-Dade County, and 22 for Broward County (http://www.nhc.noaa.gov/climo/). South Florida also received the highest number of category 3 or higher storms during the 1900 to 2010 time period, compared to the rest of the country. The south Florida ecosystem has evolved and adapted to these periodic storms and it is believed that the absence of any major hurricane since the 1960s may have contributed to the decline of conditions in Florida Bay in the late 1980s (McIvor et al. 1994). These storms play an important role in the hydrologic cycles of south Florida. Storms in June contribute to the onset of the wet season and a rapid rise in the water table, and storms in October or November prolong the wet season (Duever et al. 1994). In addition, the storms are responsible for a significant amount of movement of fresh water, nutrients, and sediments between the wetlands and the coastal ecosystem (Duever et al. 1994).

Hurricanes have the biggest impact on natural areas of the coastal region (Duever et al. 1994) and the direction of approach, duration, wind speed, rainfall, and path of the storm determine the nature of those impacts (Davis et al. 2004, Pitts 2001, Rogers and Beets 2001). The effects of hurricanes and tropical storms include changes in sediment dynamics, salinity, water quality, nutrient fluxes, vegetative cover and biotic community structure (Davis et al. 2004, Tilmant et al. 1994). The complex interactions of these factors can have both positive and negative effects on the coastal ecosystem. Winds, runoff, storm surges and currents associated with hurricanes can move large amounts of water, sediments, and pollutants. Onshore transport of sediments can contribute to the buildup of embankments and sediment accretion. The influx of salt water into mangroves and freshwater marshes can cause substantial die-offs, which in turn can increase carbon and nutrients in the estuaries (Davis et al. 2004, Smith et al. 1994, Wanless et al. 1994). Water quality is affected by resuspension of sediments and by the introduction of nutrients from runoff and pollutants from damaged structures such as landfills, water treatment plants and marinas (Davis et al. 2004, Tilmant et al. 1994). Runoff and debris can rapidly and significantly change dissolved nitrogen, oxygen, organic carbon, and phosphorous concentrations in estuaries and can lead to phytoplankton blooms (Davis et al. 2004, Tilmant et al. 1994). Winds moving perpendicular to the coast can create pressure gradients from bay to shelf or shelf to bay (depending on wind direction) and can contribute to significant short-term mixing of water between outer estuaries and shallow shelf environments (Pitts 2001). Within bays and shallow marine environments, seagrass beds and corals can be harmed by currents that scour seagrass beds or break off corals, or by transport of sediments that can smother seagrasses, corals, and sponges (Rogers and Beets 2001, Tilmant et al. 1994). Damage to corals can have particularly long-term effects because growth rates of corals are slow and because reduction in spatial relief alters available habitat for fish and invertebrates (Rogers and Beets 2001). However, these “periodic perturbations,” may play an important role in promoting the coexistence of different species within these environments (Tilmant et al. 1994).

Drought

Seasonal patterns of rainfall and evapotranspiration are the major factors in determining droughts in the wetlands of the Everglades (Duever et al. 1994). In addition, regional and global weather patterns affect seasonal rainfall in south Florida. Winter and spring precipitation tends to be below average in cool El Niño Southern Oscillation/La Niña years (Davis et al. 2004, Weaver 2005). Although the impact of seasonal rainfall on the wetlands and estuaries has been greatly altered by water management practices, these regional patterns still play a key role in the balance between fresh water and salt water in the freshwater-saltwater transition zone (Jiang et al. 2011, McIvor et al. 1994, Shomer and Drew

1982). Extended periods of hypersalinity in Florida Bay have been linked to drought conditions. The multiyear drought from 1987 to 1991, for example, led to salinities near 70 practical salinity units in significant portions of central Florida Bay (Fourqurean and Robblee 1999, McIvor et al. 1994).

Droughts are a natural part of climate variability; however, it is the duration, extent, severity, and recurrence interval of periods of low rainfall that can impact coastal ecosystems and drive changes to the system (Gilbert et al. 2012, Petes et al. 2012). Low freshwater influx into estuaries can affect the stratification of the water column and mixing, leading to hypoxia, and alteration of freshwater flow rates can contribute to the buildup of nutrients or pathogens and alter water quality (Gilbert et al. 2012). Harmful algal blooms and introduction of parasites in estuaries also have been linked to periods of drought (Gilbert et al. 2012, Petes et al. 2012). Changes in salinity, water quality, and freshwater influx associated with droughts can alter species composition, distribution, abundance and health (Gilbert et al. 2012). Studies of the long-term cycles of drought and their impacts on estuaries are lacking, and very few studies have looked at impacts of drought on the marine components of coastal ecosystems (Gilbert et al. 2012).

Fire

Fires in the mangrove forests and marshes of the coastal wetlands can result in subsidence by affecting the rates of peat accretion, either by removing materials that will contribute to the peat or in the case of extreme fires, by directly burning the peat (Davis et al. 2005, Smith et al. 1994). Accretion is important in order to maintain the wetland habitat as sea level rises and/or the land subsides. The altered landscape following a fire can provide an opening for invasives to take hold in the landscape, or for a different native flora to move in (Gunderson 1994), leading to loss of nursery habitat for juvenile estuarine organisms. Conversely, periodic fires clear out dense shrubs and invasives, deterring succession and creating open areas where graminoid marshes can form (Hofstetter 1984). These alterations to the coastal zone caused by fires change the amount of organic matter that enters the estuarine food web and becomes available to juvenile fish (Nyman and Chabreck 1995) and invertebrates. For example, the burning of leaf littler in mangroves can prevent nutrient exportation to estuaries by tides or storm surges. Alternatively, some studies indicate that periodic fires can actually increase the nitrogen and phosphorous content of vegetation and soils and can double the standing crop of some marsh grasses (Nyman and Chabreck 1995), thus increasing the nutrients that will ultimately become available to the coastal ecosystem.

Nonnative Species

A recent compilation of invasive marine species (Molnar et al. 2008) ranked the Floridian ecoregion (the coastal zone surrounding the lower half of peninsular Florida) as 15th out of 232 world-wide marine ecoregions in numbers of invasive species. The report identifies 45 invasive marine species, of which 23 are considered harmful based on a scoring criteria that measures impact on native species and overall biodiversity (Molnar et al. 2008). A study of Florida coastal waters identified 31 nonindigenous marine fish (Schofield 2009), but only the lionfish (*Pterois*) was considered to be established and to pose a threat to native species. Nonnative species are a threat to marine ecosystems because they can displace native species and change community structure through competition and predation, and by altering ecological processes such as nutrient cycling (Bax et al. 2003, Molnar et al. 2008). These changes to the natural fauna and flora affect local ecosystem services such as tourism, recreation, commercial fisheries, and in some cases human health (Bax et al. 2003). The collapse of local ecosystems and related fisheries has been linked to the introduction of invasive marine species (for example, an invasive comb jelly in the Black Sea and an Asian clam in San Francisco Bay).

Worldwide the most common pathway for the introduction of nonnative marine organisms is shipping, either through ballast water or as fouling organisms (Bax et al. 2003, Molnar et al. 2008). At any given time, 10,000 species are estimated to be in transit around the world in ballast water (Bax et al. 2003). The shipping lanes off the coast of Florida, the number of ships that enter south Florida ports make south Florida coastal waters susceptible to these types of introductions. Aquaculture is the second most common pathway for introductions when species escape the areas where they are being farmed (Molnar et al. 2008, Schofield et al. 2009). In south Florida, release of aquarium fish, either accidently or intentionally, is considered the primary source of many of introductions of nonnative fishes. The occurrence of major storms may contribute to these releases from both aquaria and aquaculture facilities (Schofield et al. 2009).

Climate Change

The predicted changes in global climate over the next century will alter many of the factors that shape south Florida's coastal ecosystem, including fire, drought, hurricanes, sea level rise and invasive species. The Intergovernmental Panel on Climate Change Summary Report for Policymakers (2007, page 12) states that "it is likely that future tropical cyclones (typhoons and hurricanes) will become more intense, with larger peak wind speeds and more heavy precipitation associated with ongoing increases of tropical SSTs [sea surface temperatures]." Models predict that rising tropical sea surface temperatures can lead to an increase in number and intensity of tropical storms in the North Atlantic (Mann et al. 2009). An increase of one degree Celsius in global sea surface temperature could result in a 30% increase in category 4 and 5 storms worldwide (Elsner et al. 2008). The Copenhagen Diagnosis (Allison et al. 2009) discusses evidence of increased hurricane activity over the past decade and a global increase in the number of category 4 and 5 hurricanes. Such an increase in frequency and intensity of storms would accelerate the impacts described above, particularly damage to corals and loss of the coastal wetlands. Changes in the delivery of fresh water, nutrients, and sediments will alter estuarine productivity and eutrophication (Scavia et al. 2002). In addition, synergistic effects of multiple storms in quick succession can lead to more substantial changes in hydrologic patterns (Davis et al. 2004).

Despite the predicted increase in major storms, the Intergovernmental Panel on Climate Change 2007 report indicates that there will be a likely net decrease in precipitation in subtropical land regions over the next century. However, the relationship between overall precipitation and precipitation associated with increased likelihood of hurricanes and changes in atmospheric temperature is unclear (Scavia et al. 2002). Predicted increases in global temperatures (Allison et al. 2009, IPCC 2007) will lead to increased evaporation rates. Less rainfall and increased evaporation rates, combined with rising sea level, indicate there will be less fresh water available to enter the coastal ecosystem in the future and potentially more periods of drought.

Increases in the frequency of thunderstorms, particularly in the tropics and southeastern United States also have been predicted in association with increased atmospheric temperatures (Aumann et al. 2008, Trap et al. 2007). In addition to wind and rainfall, lightning associated with thunderstorms play a role in fire generation in south Florida (Gunderson and Synder 1994), and southwest Florida currently has one of the highest incidences of lightning strikes in the United States (Michaels et al. 1987). These factors will have indirect effects on the south Florida's coastal ecosystems.

As discussed above, Intergovernmental Panel on Climate Change 2007 projections for worldwide sea level rise range from 20 to 60 cm during the twenty-first century; however, when ice sheet dynamics are factored in, sea level rise could approach 100 cm in the next century (Allison et al. 2009). When the effects of sea level rise are combined with reduced freshwater flow and the potential of major hurricanes to cause substantial erosion, it is likely that "overstepping of coastal wetland margins" will occur (Wanless et al. 1994). Mangrove propagules are unable to take hold in areas that have been eroded and deepened by these combined processes and the areas may become intertidal mud flats that eventually convert to shallow estuarine environments (Smith et al. 2009, Wanless et al. 1994).

Changes in biotic composition will occur in the coastal ecosystem as the climate changes. Increased ocean temperatures and ocean acidification will impact the ability of corals and other marine calcifiers to secrete new growth, and higher ocean temperatures have been linked to coral bleaching (Scavia et al. 2002). Disturbances in the habitat structure caused by sea level rise, changes in freshwater supply, fire, storms or other factors related to climate change can provide opportunities for nonnative species to become established in the coastal environment (Scavia et al. 2002). The direct impacts of these invasive plant and animal species on the coastal ecosystem are not well understood, but a study of invasive plant species in Florida has found that they can alter the geomorphology, hydrology, biogeochemistry, and community composition of an area (Gordon 1998).

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