Comprehensive Everglades Restoration Plan

2009 System Status Report

Restoration Coordination and Verification (RECOVER)

September 2010

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EXECUTIVE SUMMARY

BACKGROUND

The 2009 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) in conjunction with historical data and data from non-MAP sources. These 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 to establish pre-Comprehensive Everglades Restoration Plan (CERP) reference conditions and ultimately determine whether the goals and objectives of the CERP are being met. The goal of the SSR is to provide a synthesis of data for each of the four geographic regions (Lake Okeechobee, Northern Estuaries, Greater Everglades and the Southern Coastal Systems) as well as the ecosystem as a whole. A robust system-wide 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 (AMIG).

THE 2009 SSR PROVIDES THE FOLLOWING INFORMATION

1. A geographic and temporal synthesis of MAP findings to provide a holistic description of the status and trends of the defining attributes of the South Florida and Everglades ecosystem.
2. An interpretation of assessment results in relation to hypothesis clusters, performance measures, and achieving system-wide Interim Goals.
3. A summary of those changes in the ecosystem that are consistent with the goals and purposes of the CERP and MAP hypotheses.
4. A discussion, when necessary, of why the goals are not being met and/or why the MAP hypotheses should be revised.
5. Identification of major unanticipated findings that may require attention and correction via processes outlined in the CERP Adaptive Management Strategy.

ASSESSING PRE-CERP CONDITIONS FROM A SYSTEM-WIDE PERSPECTIVE

- Monitoring results summarized in the 2009 SSR confirm and support the detailed hydrological, ecological, and water quality restoration goals and objectives identified in the Water Resources Development Act (WRDA) 2000 (Section 601).

- Ecological components of the South Florida ecosystem identified as Interim Goals continue to be stressed, but still have the capacity for restoration. Examples of such

Interim Goals include juvenile oysters in the Northern Estuaries and crocodiles in the Greater Everglades.

- The South Florida ecosystem is dynamic and continues to respond to hydrologic and water quality drivers. Examples include salinity in the Southern Coastal Systems and phosphorous in Lake Okeechobee.

- The 2009 SSR extensively assesses and validates cause-and-effect relationships described by the MAP hypothesis clusters, such as the relationship between wading bird nest numbers and aquatic prey production in the Greater Everglades.

- The relevance of monitoring to ecosystem restoration was evidenced by the identification of a contiguous swath of marsh running through Water Conservation Areas 3A and 3B, eastern Big Cypress National Preserve, and Everglades National Park. The area has been identified as possessing potential for full restoration of sheetflow.

- Findings from the 2009 SSR reaffirm that a robust and sustainable monitoring program supports the implementation of adaptive management and the integration of science into decision-making.

ASSESSING PRE-CERP CONDITIONS BY GEOGRAPHIC REGION

Lake Okeechobee

The goal of restoration for Lake Okeechobee is to reduce total phosphorus concentrations and to maintain submerged aquatic vegetation in order to provide suitable fish habitat. Over the past decade, total phosphorus concentrations have increased due to sediment and fertilizer runoff from the watershed and resuspension of bottom sediments by the 2004-2005 hurricanes. The continued high total phosphorus concentrations indicate that additional watershed and in-lake phosphorus control projects will be necessary to reduce the frequency of undesirable large-scale algal blooms. Submerged aquatic vegetation cover was extremely variable between 2000 and 2009, reflecting both hurricane activity and the 2001 and 2007-2008 droughts. Maintaining the lake stage between 12.5 and 15.5 feet above National Geodetic Vertical Datum (NGVD) is essential in maximizing the amount of nutrients that submerged aquatic vegetation and attached periphyton (algae growing on surfaces) can store, which will in turn help reduce total phosphorus concentrations. Both watershed and in-lake nutrient management options are being pursued.

Northern Estuaries

Freshwater flows to the Northern Estuaries are typically too high in the wet season (May to October) and vary with rainfall, and too low or infrequent in the dry season (November to April) to maintain optimal salinities and sustain well-balanced estuarine communities. Wet season freshwater releases from canals and watershed runoff after storms overlap with the time of year that oysters produce their offspring (May to October). This increased water flow can flush oyster larvae downstream where salinity and other conditions are not as favorable for growth or survival. Large releases of stormwater due to hurricanes in 2004 and 2005 caused significant declines in salinities and stirred up sediments, damaging submerged aquatic vegetation coverage and density, and altering the locations of several species. Better management options for water coming into and from Lake Okeechobee, including construction of reservoirs to hold stormwater and repairs to Herbert Hoover Dike, are needed to minimize damaging releases in the wet season and increase dry season flows. Adding freshwater to the estuaries during the dry season is necessary to help reduce oyster disease and predation, as well as increase recruitment and survival of the tape grass, *Vallisneria americana*, in the Caloosahatchee Estuary. Re-establishment of favorable salinities in the St. Lucie Estuary revealed that submerged aquatic vegetation and oyster populations can increase in areas currently inhabited, as well as spread to areas left bare by hurricane activity.

Greater Everglades

The status of the Greater Everglades ecosystem over the last few years (2005-2009) reflects that existing infrastructure prevents natural processes from allowing the ecosystem to fully recover. The combination of inherently variable climate conditions and existing infrastructure resulted in very low biomass of prey base fish and crayfish in Everglades National Park in three of the last four years. As a result, wading birds in the Greater Everglades continue to be less than 70 percent of their pre-drainage population size. Alligators and crocodiles continue to exhibit low population numbers in marshes and coastal wetlands relative to their historical numbers. The ridge, slough, and tree island habitats that covered over a thousand square miles of the historical ecosystem continue to degrade due to prolonged periods of higher water depths upstream of levees and overdrained conditions downstream of levees. Additionally, unexpected cattail expansion in Upper Taylor Slough highlights the challenges faced during restoration. Careful physical, chemical, and biological monitoring before, during, and after CERP implementation is critical in order to avoid unintended impacts to hydrologic restoration.

Fortunately, there were positive indicators for restoration in the last four years as well. Water management in the Loxahatchee National Wildlife Refuge led to a consistently productive prey base, successful wading bird nesting, and healthy alligators. Roseate spoonbills in Florida Bay responded positively to a combination of rainfall patterns and recent alterations in C-111 canal operations. Crocodile populations found near the outlet of Shark Slough are increasing following the repair of plugs that block saltwater intrusion into canals. The information learned reinforces the conclusion that implementing restoration projects in an expedited fashion can lead to a restored Everglades ecosystem.

Southern Coastal Systems

Salinity was found to be the most important physical parameter in determining species and community composition in South Florida's coastal waters. Salinity monitoring warrants high priority as it will be directly affected by restoration. In the absence of increased freshwater flows, the existing salinity regime does not support the desired estuarine communities expected under restored conditions. New analyses undertaken to estimate the restored condition in Florida Bay predict that suitable habitat could increase from 25,000 to 44,000 acres once favorable salinities are re-established. Increased nutrients in the water caused widespread algal blooms that persisted for as long as two years. The efficacy of the current submerged aquatic vegetation monitoring strategy to detect restoration-induced change was independently verified. The observed patterns in natural variability underscore the need to better integrate salinity and submerged aquatic vegetation monitoring to evaluate the effects of restoration. Pink shrimp density has been statistically related to hydrology, with higher densities in wet years and lower densities in dry years. Overall, wet-season pink shrimp abundances evidence a significant downward trend from 2005 to 2009.

READING THE 2009 SYSTEM STATUS REPORT

The 2009 SSR is formatted as an interactive webpage accessible at http://www.evergladesplan.org/pm/ssr_2009/ssr_main.aspx. This web-based approach allows managers, stakeholders, and scientists with different degrees of technical expertise to easily explore the SSR according to their interests and desired level of detail. Information is presented in a hierarchical manner, allowing users to initially access very general information about each assessment (e.g., overall trends in wading birds), then slightly more detailed information (e.g., location and number of wading bird nests in Big Cypress) and finally very detailed information (e.g., specific wading bird survey techniques by location).

Housed on the homepage is an interactive map encompassing the four MAP geographic regions and a navigation toolbar with links to major topics addressed within the document including: Data Management, Adaptive Management, Ecosystem Components, and Interim Goals and Interim Targets. Also on the homepage is a link to the Key Findings document, a high-level synthesis highlighting the SSR findings with the greatest implications for restoration.

Users can search for information geographically (via a map) or by ecosystem component (via a navigation toolbar). Both pathways lead to webpages containing summaries of assessments and associated graphics addressing (1) the validity of the hypothesis cluster concept and established functional relationships, (2) the status/trends of ecosystem components, and (3) the use of hypothesis-based assessments to provide the scientific foundation for evaluating Interim Goals and applying adaptive management. Additional hyperlinks are provided (either embedded in the text or at the bottom of the webpage) for related links (*e.g.*, MAP 2006 and 2009, diagrams of conceptual models and hypothesis clusters, Performance Measure documentation sheets, Interim Goals and Interim Targets documentation, etc.).

Although the 2009 SSR is being presented as an interactive webpage, print capabilities (via PDF) will be available for the Key Findings, each geographic region within the MAP, and the document in its entirety.

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ACRONYMS

4DHRH	Four-dimensional High Resolution Hydrology Model
AM	Adaptive Management
AMBI	AZTI's Marine Biotic Index
AMIG	Adaptive Management Integration Guide
AMO	Atlantic Multi-decadal Oscillation
ANOSIM	analysis of similarities
ANOVA	analysis of variance
AOML	Atlantic Oceanographic and Meteorological Laboratory (NOAA)
ASR	aquifer storage and recovery
AT	Assessment Team
ATLSS HRH	Across Trophic Level System Simulation High Resolution Hydrology Model
BMP	Best Management Practices
BNP	Biscayne National Park
C	Carbon
C&SF Project	Central and Southern Florida Project
CDOM	chromophone dissolved organic matter
CEM	Conceptual Ecological Model
CERP	Comprehensive Everglades Restoration Plan
CERP MMS	CERP Model Management System
C.F.R.	Code of Federal Regulation
CH3D	Curvilinear-grid Hydrodynamics Three-Dimensional model
CPUE	catch per unit effort
DASR	Data Access, Search and Retrieval Application
DBHYRDO	The SFWMD's environmental database which stores hydrologic, meteorologic, hydrogeologic and water quality data.

DECODA	Database for Ecological Community Data
DERMO	*Dermocystidium* (oyster disease)
DIN	dissolved inorganic nitrogen
DOI	Department of the Interior
EAA	Everglades Agricultural Area
EdCat	Electronic Data Catalog
EDEN	Everglades Depth Estimation Network
EMAP	Environmental Monitoring and Assessment Program (USEPA)
EML	Ecological Metadata Language
ENSO	El Nino Southern Oscillation
ESM	Everglades Soil Mapping Project
F.A.C.	Florida Administrative Code
FDEP	Florida Department of Environmental Protection
FHAP	Fish Habitat Assessment Program (FWC)
FIAN	Fish and Invertebrate Assessment Network
FWC	Florida Fish and Wildlife Conservation Commission
GIDAST	Geo-referenced Interactive Data Analysis Tool
GIS	geographic information system
GM	Guidance Memorandum
GPS	global positioning system
ha	hectare
HSI	habitat suitability index
IN	inorganic nitrogen
IG/IT	Interim Goal/Interim Target
ITCZ	Inter-tropical Convergence Zone
LILA	Loxahatchee Impoundment Landscape Assessment Area

LOEM	Lake Okeechobee Environmental Model
LOESS	Locally Weighted Scatterplot Smoothing
LORS2008	Lake Okeechobee Regulation Schedule 2008
LOWQM	Lake Okeechobee Water Quality Model
Loxahatchee NWR	Arthur R. Marshall Loxahatchee National Wildlife Refuge
LSU	landscape sampling unit
LTER	Long-term Ecological Research (National Science Foundation)
LWL	Lake Worth Lagoon
MAP	Monitoring and Assessment Plan (RECOVER)
MDS	multi-dimensional scaling
MFL	minimum flow and level
Miami-Dade DERM	Miami-Dade County Department of Environmental Resource Management
MLR	multivariate linear regression
msl	mean sea level
MSX	Multi-nucleated Sphere with unknown affinity "X" (oyster disease)
N	Nitrogen
NAVD 88	North American Vertical Datum of 1988
NGVD	National Geodetic Vertical Datum
NMDS	non-metric multidimensional scaling
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NRC	National Resource Council
NSM	Natural System Model
NWIS	National Water Information System (USGS)
NWR	National Wildlife Refuge

P	Phosphorus
PAR	photosynthetically active radiation
PBCERM	Palm Beach County Environmental Resources Management
PES	Priority Ecosystem Science (USGS)
PI	Principal Investigator
PIR	Project Implementation Report
PLMP	Project-level Monitoring Plan
ppb	parts per billion
ppm	parts per million
ProRegs	Programmatic Regulations
psu	practical salinity units
QAOT	Quality Assurance Oversight Team
RECOVER	Restoration Coordination and Verification Program
REMAP	Regional Environmental Monitoring and Assessment Program (USEPA)
RLG	RECOVER Leadership Group
SAV	submerged aquatic vegetation
SD	standard deviation
SE	standard error
SET	sediment elevation table
SFER	South Florida Environmental Report
SFWMD	South Florida Water Management District
SIMPER	similarity percentage
SFWMM	South Florida Water Management Model
SOFIA	South Florida Information Access (USGS)
SRP	soluble reactive phosphorus
SRSI	Salinity Regime Suitability Index

SSR	System Status Report
STA	stormwater treatment area
SWIM	Surface Water Improvement and Management
TD	Total Depth
TIME	Tides and Inflows in the Mangrove Ecotone (model)
TMDL	total maximum daily load
TN	total nitrogen
TP	total phosphorus
TSI	trophic state index
TSS	total suspended solids
USACE	United States Army Corps of Engineers
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
VEC	valued ecosystem component
WCA	Water Conservation Area
WRDA 2000	Water Resources Development Act of 2000
WS/FP	Water Supply/Flood Protection
WSE	Water Supply and Environmental Regulation Schedule
WY	water year

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UNITS OF MEASUREMENT

%	Percent
°C	degrees Celsius
cfs	cubic feet per second
cm	Centimeter
feet NGVD	feet National Geodetic Vertical Datum
g dw/m^2	grams dry weight per square meter
g/m^2	grams per square meter
g/m^2/year	grams per square meter per year
ha	Hectare
K_d	downwelling light attenuation
kg	Kilograms
kg H_2O/m^2 SA/s	kilograms water per square meter sap area per second
kg/yr	kilograms per year
km	Kilometer
km^2	square kilometers
m^2	square meter
m^3/sec	cubic meters per second
mg/cm^2	milligrams per square centimeter
mg/g	milligram per gram
mg/kg	milligram per kilogram
mg/L	milligrams per liter
mg/m^3	milligrams per cubic meter
mgd	million gallons per day
mL/m^2	milliliter per square meter
mm	Millimeter

mph	miles per hour
ms/sec	millisecond per second
msl	mean sea level
mt	metric tons
mt/yr	metric tons per year
mV	Millivolt
nm	Nanometers
NTUs	nephelometric turbidy units
pcu	platinum-colbalt units
ppb	parts per billion
ppm	parts per million
psu	parts salinity units
μg/g	micrograms per gram
μg/L	micrograms per liter
$\mu m^3/g$	cubic micromoles per gram
$\mu m^3/cm^2$	cubic micromoles per square centimeter
$\mu m^3/mL$	micrometers per milliliter
$\mu mol^3/cm^2$	cubic micromoles per square centimeters
$\mu mol^3/g$	cubic micromoles per gram
$\mu mol^3/L$	cubic micromoles per milliliter
$\mu mol/s/m^2$	micromoles of photons per second per square meter

KEY FINDINGS FROM THE 2009 SYSTEM STATUS REPORT

INTRODUCTION TO THE SYSTEM STATUS REPORT

The 2009 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)[i] in conjuction with historical data and data from non-MAP sources. These 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 to establish pre-Comprehensive Everglades Restoration Plan (CERP) reference conditions and ultimately determine whether the goals and objectives of CERP are being met. The goal of the SSR is to provide a synthesis of data for each of the four geographic regions (Lake Okeechobee, Northern Estuaries, Greater Everglades, and Southern Coastal Systems) as well as the ecosystem as a whole. A robust system-wide 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 (AMIG)[ii].

PURPOSE OF THE KEY FINDINGS

Only findings with the greatest implication for restoration are highlighted in this document. The remaining information is provided on the 2009 SSR web pages at http://www.evergladesplan.org/pm/ssr_2009/ssr_main.aspx. For each key finding, two types of information are provided: (1) new scientific findings garnered from the MAP; and (2) how this information reaffirms key CERP hypotheses, which are the foundation for restoration activities. The key findings are organized by geographic region, hypothesis cluster, and Interim Goal.

Interim Goals are a means by which restoration success is evaluated at specific points throughout the overall planning and implementation process. It is critical that MAP components link to Interim Goals in order to provide incremental evaluations of CERP implementation and to allow for adjustments if necessary. A full list of the CERP Interim Goals can be found in the Programmatic Regulations Interim Goals Agreement (2007)[iii]. The 2009 SSR does not address every Interim Goal listed in the agreement for one of two reasons, either: (a) information about a specific Interim Goal is captured via reporting on another; or (b) the data record is not yet sufficient to allow for reporting.

The South Florida Ecosystem Restoration Task Force has developed a set of stoplight indicators for use in assessing restoration efforts. The status of these indicators were summarized in the System-Wide Indicators for Everglades Restoration 2008 Assessment[iv] (referred to as the Stoplight Indicator Report in this document). As part of the ongoing integration of reporting efforts in south Florida,these stoplight indicators are discussed with the 2009 SSR key findings where appropriate. However, stoplight indictors are often assessed with a slightly different approach and may not always be directly comparable to results in the 2009 SSR. A table has been added at the end of this document to show the linkages between Interim Goals, MAP hypothesis clusters, and stoplight indicators (***Table KF-1***).

DOCUMENT OVERVIEW

Science from the 2009 SSR provides information for CERP projects and/or initiatives such as the: (1) Indian River Lagoon– South Project; (2) Decompartmentalization and Sheetflow Project (DECOMP) Adaptive Management Physical Model; (3) Tamiami Trail Modification (part of Modified Water Deliveries to Everglades National Park); (4) Everglades Restoration Transition Plan (ERTP) – Phase 1; (5) C-111 Spreader Canal Project; and (6) Picayune Strand Restoration Project. Assessments of system-wide monitoring efforts assist with tracking the spectrum of hydrological and ecological responses to operational changes, project implementation, and ecosystem restoration.

The 2009 SSR demonstrates that the ***Northern Estuaries*** continue to be impacted by too much freshwater flow in the summer months and too little freshwater flow in the winter months. Improved Lake Okeechobee management options, wetland rehydration, and the construction and operation of reservoirs and stormwater treatment areas are needed to minimize damaging flows, improve salinity regimes, and improve water quality. Initial steps taken to control flow volumes (*e.g*., Lake Okeechobee Regulation Schedule) have proven to be effective management techniques in improving conditions for oysters and submerged aquatic vegetation (SAV). For example, oyster populations in the St. Lucie Estuary rebounded upon re-establishment of a favorable salinity regime.

For ***Lake Okeechobee***, the 2009 SSR highlights the importance of maintaining the lake stage within the desired envelope to maximize nutrient storage, encourage growth and survival of SAV, and inhibit growth of invasive plant species. Water storage projects north of Lake Okeechobee will provide additonal storage and enable better regulation of lake stage.

In the ***Greater Everglades***, restoration of rainfall-driven multi-year hydroperiods in the slough systems of Everglades National Park will increase the spatial extent and duration of aquatic habitats for prey-base fish, wading birds, and alligators. The disruption of sheetflow, evidenced in impounded or overdrained areas, has degraded ridge, slough, and tree island landscapes, and restoration activities are needed to reduce the risk of further degradation. In addition, the return of rainfall-driven freshwater flows to the coastal wetlands, particularly Florida Bay, will re-establish salinities critical for crocodiles, fish in the marsh-mangrove interface, and roseate spoonbill foraging. In some areas, such as the A.R.M Loxahatchee National Wildlife Refuge, ongoing water management actions are resulting in consistently higher populations of prey-base fish, successful wading bird nesting, and healthy alligators. In other parts of the system, ecosystem impacts (*e.g*., cattail expansion in upper Taylor Slough) highlight the challenges of delivering water with higher nutrient concentrations than historical levels.

In the ***Southern Coastal Systems***, the 2009 SSR found that in Florida Bay, increased nutrient availability can initiate persistent, detrimental algal blooms. Additionally, the current salinity regime exerts adverse impacts on desired species (e.g., shrimp, fish, and SAV) and community composition.

SYSTEM-WIDE KEY FINDINGS

The system-wide key findings from the SSR and their management relevance yield the following broad conclusions that apply to most of the south Florida ecosystem:

- Monitoring results summarized in the 2009 SSR confirm and support the detailed hydrological, ecological, and water quality restoration goals and objectives identified in the Water Resources Development Act (WRDA) 2000 (Section 601).
- Ecological components of the South Florida ecosystem identified as Interim Goals continue to be stressed, but still have the capacity for restoration. Examples of such Interim Goals include juvenile oysters in the Northern Estuaries and crocodiles in the Greater Everglades.
- The South Florida ecosystem is dynamic and continues to respond to hydrologic and water quality drivers. Examples include salinity in the Southern Coastal Systems and phosphorus in Lake Okeechobee.
- The 2009 SSR extensively assesses and validates cause-and-effect relationships described by the MAP hypothesis clusters, such as the relationship between wading bird nest numbers and aquatic prey production in the Greater Everglades.
- The relevance of monitoring results to ecosystem restoration was evidenced by the identification of a contiguous swath of marsh running through Water Conservation Areas 3A and 3B, eastern Big Cypress National Preserve, and Everglades National Park. This area has been identified as possessing potential for full restoration of sheetflow.
- Findings from the 2009 SSR reaffirm that a robust and sustainable monitoring program supports the implementation of adaptive management and the integration of science into decision-making.

NORTHERN ESTUARIES KEY FINDINGS

Northern Estuaries Oysters

Interim Goal:	MAP Hypothesis Cluster:
American Oysters in Northern Estuaries	Oyster Health and Abundance

Interim Goal: The desired restoration condition is to establish approximately 900 acres of living oyster beds in the St. Lucie Estuary and 500 acres in the Caloosahatchee River Estuary. Specific targets for the Loxahatchee River Estuary and Lake Worth Lagoon have not yet been established, but restoration should result in increases in all of the Northern Estuaries.

Key Findings from the SSR:

- The density of living American oysters (*Crassostrea virginica*) is higher in the Caloosahatchee River Estuary sites than the east coast estuaries, a result of higher recruitment at the end of the spawning season (fall). However, during the winter months, juvenile mortality increases due to predation as a result of elevated salinity. Until a more favorable salinity regime is established and maintained, juvenile oyster mortality will continue to be high.

- Oysters in the Caloosahatchee River Estuary actively spawn between May and October when freshwater releases and watershed runoff are at their peak. As a result, oyster larvae are flushed to downstream locations and settle. Subsequent growth rate and survival of juveniles are poor at these locations due to higher than optimal salinity.

- Low salinities in the St. Lucie Estuary during late summer 2008 resulted in widespread oyster mortality; however, with an increase in salinity by November, new recruits exhibited rapid growth. Oyster populations can rebound with a positive trend upon re-establishment of a favorable salinity regime.

- Data collected during the recent drought indicate predation induced by sustained higher salinities may be a substantial stressor. Given that predation pressure is significant in some locations, such information is necessary and a longer data set will statistically provide greater confidence in the ability of the oyster habitat suitability index to predict potential suitable habitat.

- The St. Lucie Estuary has the greatest variation in adult density due to large variability in the magnitude and extent of high flow events, resulting in unfavorable salinity.

2008 Stoplight Indicator:

- The oysters in the Caloosahatchee River Estuary continue to be impacted by too much freshwater in summer and too little freshwater in winter. Too much freshwater adversely impacts reproduction, larval recruitment, survival, and growth, while too little freshwater impacts the survival of oysters due to higher prevalence and intensity of the disease Dermo (caused by the pathogen *Perkinsus marinus)* and predation.

- The 2009 SSR and the Stoplight Indicator Report were in general agreement, although the methods used to derive the results were not identical.

Potential Management Relevance:

- Improvements in Lake Okeechobee management options, rehydration of watershed wetlands, repair of the Herbert Hoover Dike, and construction and operation of reservoirs are needed to minimize damaging releases that adversely impact oyster recruitment, particularly in the summer and fall months.

- Freshwater releases to the Caloosahatchee River Estuary should be less than 2,800 cubic feet per second (cfs) during summer and fall recruitment months to curb flushing of oyster larvae to downstream locations and to create favorable conditions for spat recruitment and survival upstream in areas where they are more likely to survive.

- Mean weekly target flows from the S-79 structure along the C-43 canal should be between 450-2,800 cfs with no flows in excess of 4,500 cfs to sustain the most favorable salinity regime in the Caloosahatchee River Estuary.[v]

- Controlling the volume of late summer/early fall inflows has proven to be an effective management technique resulting in improved rates of recruitment, growth and survival even after extreme wet season conditions.

- Dry season base flows are necessary to help reduce oyster disease and increase recruitment and survival.

Northern Estuaries Submerged Aquatic Vegetation

Interim Goal:	MAP Hypothesis Cluster:
Submerged Aquatic Vegetation in Northern Estuaries	Submerged Aquatic Vegetation

Interim Goal: The desired restoration condition is to increase the spatial extent and improve the functionality of submerged aquatic vegetation in the Northern Estuaries.

Key Findings from the SSR:

- In the upper Caloosahatchee River Estuary, high salinity conditions adversely impacted the historically-present freshwater tape grass (*Valisneria americana*). Subsequent low light conditions associated with freshwater inflows and a lack of propagules into these areas may inhibit the ability of tape grass to reestablish.

- The Southern Indian River Lagoon region was impacted by hurricanes and associated freshwater discharges in 2004 and 2005. Impacts included significant declines in coverage and density, as well as localized losses due to burial by shifting bottom sediments and altered bathymetry.

- Within the Loxahatchee River Estuary, the ephemeral and pioneering species shoal grass (*Halodule wrightii*) and Johnson's grass (*Halophila johnsonii*) are widely distributed. Due to wide salinity fluctuations in the estuary, the more tolerant species shoal grass would be expected to dominate. However, Johnson's grass was surprisingly more prevalent (approximately 354 acres) than shoal grass (256 acres) in 2007. Continued monitoring of SAV, including the threatened Johnson's grass, is important in understanding the complexity and resiliency of the Loxahatchee River Estuary as restoration projects are planned and implemented.

- High turbidity, muck deposits, and poor water quality adversely affected seagrass coverage in the central and southern segments of Lake Worth Lagoon. Seagrass coverage varied throughout the lagoon with more seagrass in the north.

2008 Stoplight Indicator:

- No Stoplight Indicator exists.

Potential Management Relevance:

- Meeting the Caloosahatchee River minimum flow and level (MFL) of 300 cfs at S-79 will allow tape grass to survive and be ecologically beneficial to the estuary.

- Restoration projects, such as reservoirs, stormwater treatment areas, and wetland rehydration will improve salinity regimes and decrease problems associated with turbidity, muck deposits, and poor water quality. Monitoring data, particularly in the St. Lucie Estuary, indicates the potential for expansion of SAV.

Northern Estuaries Water Quality

Interim Goal:	MAP Hypothesis Cluster:
None	Salinity, Water Quality and Nutrients As Primary Stressors Affecting Northern Estuaries

Interim Goal: An Interim Goal for water quality in the Northern Estuaries has not been developed.

Key Findings from the SSR:

- In the Caloosahatchee River Estuary, long-term trends show the majority of the phosphorus load and about half of the freshwater and nitrogen load is run-off from the local drainage basin, not Lake Okeechobee.

- In the St. Lucie Estuary, long-term trends show greater than half of the wet season nitrogen load and greater than two-thirds of the phosphorus load comes from the local drainage basin other than the C-44, not Lake Okeechobee.

2008 Stoplight Indicator:

- No Stoplight Indicator exists.

Potential Management Relevance:

- Documentation of the sources of nutrient loading is required to properly locate and schedule restoration projects that will alleviate adverse water quality conditions.

LAKE OKEECHOBEE KEY FINDINGS

Lake Okeechobee Water Quality

Interim Goal:	MAP Hypothesis Cluster:
Lake Okeechobee Phosphorus	Water Quality and Phytoplankton

Interim Goal: The desired restoration condition for Lake Okeechobee is to reduce pelagic total phosphorus concentrations in the lake to 40 parts per billion (ppb).

Key Findings from the SSR:

- Between 2000 and 2009, large-scale disturbances (*i.e.*, hurricanes, droughts, and lake stage fluctuations) affected the lake and resulted in measurable changes in water quality.
- Mean water column total phosphorus concentrations have increased over the past decade. The five-year (2004-2009) rolling average for mean water column total phosphorus concentrations was greater than 150 micrograms per liter (µg/l).
- Annual total phosphorus loading between 2000 and 2009 consistently exceeded the 2015 target of 140 metric tons. The five-year (2004-2009) rolling average for annual total phosphorus loading was approximately 580 metric tons, or four times the 2015 target.

2008 Stoplight Indicator:

- No Stoplight Indicator exists.

Potential Management Relevance:

- Additional watershed and in-lake phosphorus control projects will be necessary to reduce water column phosphorus concentrations.
- The desired lake stage envelope of 12.5–15.5 feet above National Geodetic Vertical Datum (NGVD) will maximize the nutrient storage potential of SAV and attached periphyton, thus enhancing water column phosphorus concentration reduction.

Lake Okeechobee Submerged Aquatic Vegetation

Interim Goal:	MAP Hypothesis Cluster:
Lake Okeechobee Aquatic Vegetation	Emergent-Submerged Vegetation Mosaic

Interim Goal: The desired restoration condition is to maintain at least 40,000 acres of SAV and at least 20,000 acres of vascular SAV in the nearshore region of the lake.

Key Findings from the SSR:

- Submerged aquatic vegetation areal coverage was extremely dynamic between 2000 and 2009, reflecting the frequent large-scale disturbances that impacted the lake (*e.g.*, hurricanes, droughts, and lake stage fluctuations).

- The target for nearshore submerged plants is for 50 percent of the total coverage to be comprised of vascular plants. As a result of extremely low lake stages from 2006-2008, approximately 80 percent of the SAV was comprised of the non-vascular macroalga *Chara* spp. However, the presence of this species represents a first step towards recovery of nearshore SAV given the ability to manage the lake within target stages.

- As the 2007-2008 drought subsided, recovery of vascular SAV was delayed as compared to previous droughts. This was due to the fact that nearshore areas were not favorable for SAV growth because (1) the inshore portion of the lake had become dominated by terrestrial and emergent vegetation, and (2) the inshore region was subjected to alternating dry and wet cycles.

- Subsequent monitoring during 2009 indicated SAV coverage continued to expand beyond that documented for 2008, with summer SAV coverage of approximately 46,400 acres. Additionally, vascular SAV coverage rapidly expanded during 2009, with summer coverage comprising 66 percent of the total SAV coverage. The Interim Goal of at least 40,000 acres of SAV, with at least 50 percent of that being comprised of vascular taxa, was attained during 2009, the first time the Interim Goal was met since 2004.

2008 Stoplight Indicator:

- Submerged aquatic vegetation coverage, especially vascular plant coverage, decreased dramatically since the fall of 2004. This decline in areal coverage was caused by physical disturbance (uprooting) from three hurricanes (Frances, Jeanne, and Wilma) followed by prolonged water column turbidity. Coverage of the non-vascular macroalga *Chara* spp. dramatically increased during 2007, covering approximately 27,700 acres. However, vascular plants accounted for only approximately 500 total acres.

- The 2009 SSR and the Stoplight Indicator Report were in agreement.

Potential Management Relevance:

- The length of time needed for recovery of SAV reaffirms the importance of maximizing the time the lake is within the stage envelope of 12.5-15.5 feet above NGVD.

- A sufficient number of water storage projects north of Lake Okeechobee will enable lake stage to be better regulated.

- If lower lake stages occur more frequently, a rapid increase in exotic vegetation, such as Hydrilla (*Hydrilla verticillata*), might ensue. If this occurs, alternative management strategies and additional funding to control these types of non-native plants will be required.

GREATER EVERGLADES KEY FINDINGS

Greater Everglades Sheetflow and Hydropattern

Interim Goals:	MAP Hypothesis Cluster:
Sheetflow Hydropattern	Sheet Flow and Water Depth Patterns

Interim Goals: The desired restoration condition for sheetflow is to establish more historic magnitudes and directions of sheetflow in the natural areas of the Everglades.

The desired restoration condition for hydropattern is to restore the natural timing and pattern of inundation throughout the ecological communities of South Florida, matching long-term averages and interannual variability.

Key Findings from the SSR:

- Due to compartmentalization and elimination of sheetflow, variability in water depth and hydroperiod do not correlate with variability in rainfall and topography in the present system. Flow barriers that have been installed perpendicular to pre-drainage patterns of sheetflow have produced ponding of water upstream and over-drainage downstream of barriers.
- The hydroperiod in southern Water Conservation Area (WCA) 3A is currently characterized by prolonged periods of standing water that inundate sawgrass ridges and tree islands for unnaturally long durations, without perceptible flow. This is in contrast to the best understanding of how the pre-drainage system responded to seasonal and interannual variability and rainfall. The resulting pool lasted for over four years spanning a regional drought during 2007 and 2008.
- Even during the wettest years, water deliveries from WCA 3 were inadequate to maintain multi-year hydroperiods in the major slough systems of Everglades National Park (ENP), where pre-drainage hydroperiods exceeded one year.
- In contrast to other areas of the Everglades, water stages in the Loxahatchee National Wildlife Refuge during the last four years approximated beneficial long-term averages, even during the two years of regional drought.

2008 Stoplight Indicator:

- No stoplight indicator exists.

Potential Management Relevance:

- A contiguous swath of marsh running through WCA 3A, WCA 3B, eastern Big Cypress National Preserve and ENP has recently been identified as an area possessing potential for full restoration of sheetflow. Hydrologic restoration merits high prioritization in the future implementation of CERP projects.

- Rainfall-driven multi-year wet and dry cycles, including extended hydroperiods in slough systems of ENP, are key components of hydrologic restoration.

- Full prehistoric hydrologic restoration requires: (1) pre-drainage volume and timing of water deliveries based on antecedent rainfall; (2) sheetflow and water depth patterns not distorted by levees and canals; (3) curtailment of seepage loss across the eastern protective levee system into developed areas; and (4) topographic restoration to pre-drainage conditions.

- Continuing the hydrologic management efforts that are presently in place in the Loxahatchee National Wildlife Refuge will further increase understanding of wading bird predator-prey dynamics and alligator populations.

Greater Everglades Aquatic Fauna Regional Populations and System-wide Wading Bird Nesting Pattern

Interim Goals:	MAP Hypothesis Cluster:
Aquatic Fauna Regional Populations in Everglades Wetlands System-wide Wading Bird Nesting Pattern	Wading Bird Nesting in the Mainland and Coastal Everglades in Relation to the Aquatic Fauna Forage Base

Interim Goals: The desired restoration condition for aquatic fauna regional populations is to achieve late wet season population densities, size distributions, and taxonomic compositions of marsh fishes and other selected groups of aquatic fauna consistent with pre-drainage hydrologic and salinity patterns in the Everglades wetlands and to provide high-density patches of prey availability across the Everglades landscape where wading birds can feed effectively as water levels recede during the dry season. The desired restoration condition for system wide wading bird nesting patterns is for over 50 percent of 80,000 pairs to nest in estuarine locations, wood stork nesting initiation to occur in the December /January time period, and for wading bird super colony events to occur once every four years.

Key Findings from the SSR:

Wet Season Aquatic Prey Production and Dry Season Prey Concentration

- Relatively low prey production across most of the system during water years 2006 and 2007 is consistent with the hypothesis that prey production is limited by hydroperiod.

- Prey production increases during water years 2008 and 2009 are consistent with the hypothesis that drought stimulates pulses in prey production, and post-drought pulses in production involve surges in crayfish populations.

- During water years 2006-2009, aquatic prey biomass produced during the wet season became concentrated three to four-fold during the dry season as surface water receded into isolated pools.

Wading Bird Nesting

- Magnitude and success of wading bird nesting were high during 2006 and 2009 when prolonged water level recession concentrated aquatic prey without hydrologic reversals and/or marsh dry downs during the nesting season.

- Wading bird nest numbers were very high in 2009 after a pulse in aquatic prey production during the preceding wet season. During 2009, nest numbers of white ibis (*Eudocimus albus*) and wood stork (*Mycteria americana*) exceeded the 90th percentile of annual nesting for each species for the period of record beginning in 1931.

- Magnitude and success of wading bird nesting were low during 2007 and 2008 when water level recession patterns were not favorable for nesting, regardless of prey production during the preceding wet seasons.

- Only 6 to 21 percent of the total wading bird nests throughout the Everglades were located in ENP, in comparison to the 70 percent minimum proposed for restored conditions.

- The Loxahatchee National Wildlife Refuge is the only area of the Everglades that has consistently produced high aquatic prey biomass and successful nesting by white ibis and other wading birds over the last four years.

2008 Stoplight Indicator:

Wet Season Aquatic Prey Production and Dry Season Prey Concentration

- Low stoplight scores for total forage-base fish density in ENP are consistent with results in the SSR. The inclusion of crayfish and the extension of monitoring through the 2008 wet season for assessment in the SSR revealed generally low prey biomass throughout much of the Everglades, and strong interannual variation in prey biomass with multi-year wet and dry cycles. These results were not evident in the stoplight assessments.

Wading Bird Nesting

- Red stoplight scores for overall wading bird nesting are consistent with MAP monitoring results indicating a low proportion of total wading birds nesting in ENP. A green stoplight score for the mean interval between exceptional white ibis nesting years is consistent with MAP monitoring results for the 2009 nesting season.

Potential Management Relevance: Monitoring of wading bird nesting in relation to the aquatic fauna forage-base supports the broader management recommendations from hydrology monitoring:

- Restoration of rainfall-driven multi-year wet and dry cycles in the southern Everglades, including extended hydroperiods in the slough systems of ENP, would increase the prey base for wading birds by: (1) supporting population buildup of marsh fish and grass shrimp during sequential wet years; (2) producing surges in crayfish populations after drought years; and (3) decreasing the frequency of rising stage that disrupts wading bird nesting during the dry season.

- The status of wading bird predator-prey dynamics in the Loxahatchee National Wildlife Refuge support the maintenance of the current hydrologic management efforts as recommended above.

Greater Everglades American Alligator

Interim Goal:	MAP Hypothesis Cluster:
American Alligator	American Alligator Density and Body Condition in Relation to Hydrologic Patterns and Artificial Canal Habitats in the Everglades

Interim Goal: The desired restoration condition for the American alligator (*Alligator mississippiensis*) is to restore more natural numbers and distribution patterns for alligators across south Florida's major freshwater and estuarine landscapes.

Key Findings from the SSR:

- Relative density of alligators is extremely low in the ridge and slough landscapes of ENP where naturally occurring multi-year hydroperiods are reduced to less than one year.

- Alligator density is higher in areas where hydroperiod is longer, most likely because of increased extent and duration of the aquatic habitat upon which alligators depend. However, alligator body condition is low throughout most of the Everglades because oligotrophic (low) nutrient status may limit the production of aquatic organisms that alligators feed upon.

- The Loxahatchee National Wildlife Refuge is the only area of the Everglades that has consistently supported restoration targets for alligator relative density during the last four years. This successful outcome was a direct consequence of minimizing withdrawals from the refuge for water supply purposes during droughts, and limiting the rate of water depth increases during wet conditions. These actions maintained favorable habitat conditions for alligators.

2008 Stoplight Indicator:

- Red stoplight scores for alligators in ENP are calcualted using a combination of relative density and body condition. MAP assessments are based only on relative density because body condition appeared less sensitive to contrasting hydrologic regimes. Although these assessment methods for alligators are not directly comparable, results from both are consistent in showing very low relative densities in ENP.

Potential Management Relevance: Monitoring of alligator populations supports the broader management recommendation from hydrology monitoring:

- Restoration of multi-year hydroperiods in the slough systems of ENP would increase the spatial extent and duration of aquatic habitats, which in turn should increase the density and distribution of alligator populations in the park.

- Continuing the hydrologic management efforts presently in place in the Loxahatchee National Wildlife Refuge will increase further understanding of alligator populations.

Ecosystem Characteristics of Everglades Coastal Wetlands and the American Crocodile

Interim Goals:	MAP Hypothesis Cluster:
Freshwater Flow to Florida Bay American Crocodile	Ecosystem Characteristics of Everglades Coastal Wetlands In Relation to Freshwater Inflows

Interim Goals: The desired restoration condition for freshwater flow to Florida Bay is to restore the natural spatial and temporal patterns of salinity, linking freshwater discharges to seasonal rainfall patterns (volume and timing) and natural wetland hydrologic functions. The desired restoration condition for the American crocodile (*Crocodylus acutus*) is to maintain high frequencies of salinities below 20 practical salinity units (psu) for optimal survival and growth of juvenile crocodiles.

Key Findings from the SSR:

Salinity Gradients

- Monitoring of coastal salinity gradients during 2003-2008 indicated an absence of persistent freshwater-oligohaline zones at headwater sites across most of the marsh-mangrove interface of the coastal Everglades.

- The limited spatial extent and duration of the oligohaline zone is caused by altered quantity and timing of freshwater flows to coastal regions due to disrupted sheetflow and connectivity, in combination with operations/management of the system upstream.

- The oligohaline zone is a key driver determining the ecological health of the Everglades coastal wetlands.

<u>Prey-base Fishes</u>

- Monitoring of prey-base fishes in the mangrove zone indicates high salinity values are correlated with low fish biomass.

- Fish located within mangrove zone concentrate in deeper creeks and pools when surface water depths in the surrounding wetlands decrease to 12.5 centimeters. These conditions are favorable for roseate spoonbill *(Platalea ajaja)* foraging.

<u>Roseate Spoonbill Nesting</u>

- The total number of roseate spoonbill nests in Florida Bay increased during nesting years 2005-2006, 2006-2007 and 2007-2008 when compared to past years. During these nesting years, the roseate spoonbill colonies in northeast and northwest Florida Bay produced on average 1.5 chicks per nest resulting in three consecutive successful nesting years.

- The recent increase in roseate spoonbill nesting success is attributed to beneficial rainfall patterns combined with recent (since 2005-2006) revisions to the rules governing C-111 canal operations during the dry season in the Taylor Slough/C-111 basin. Previous operational rules chronically disrupted the seasonal dry downs by out-of-phase releases from the C-111 basin.

<u>American Crocodile</u>

- Growth and survival of juvenile crocodiles drops when salinity exceeds 20 psu.

- Growth, survival and dispersal of juvenile American crocodiles were low in ENP in comparison to other primary crocodile nesting areas (*i.e.*, Crocodile Lake National Wildlife Refuge and Turkey Point Nuclear Power Plant) during the monitoring period beginning in 2005.

- A trend of higher numbers of crocodile nests in ENP since 2000 resulted mainly from increased nesting on artificial substrates on Cape Sable after plugging of the East Cape Canal blocked saltwater intrusion to the interior of the cape.

2008 Stoplight Indicators:

<u>Salinity Gradients</u>

- No stoplight indicator exists.

<u>Prey-base Fishes</u>

- A red stoplight score for prey-base fish community structure in northeast Florida Bay is based on the low abundance of freshwater species at roseate spoonbill foraging sites. This is consistent with the SSR results indicating an absence of persistent freshwater-oligohaline zones at headwater sites across most of the marsh-mangrove interface of the coastal Everglades.

Roseate Spoonbill Nesting

- A red stoplight score for overall nest production and success of roseate spoonbills in Florida Bay is consistent with the continued decline in total nests during the last three consecutive years reported in the SSR. The troubling decline in roseate spoonbill nest numbers is mitigated somewhat by the fact that the few remaining nests have been successfully fledging at least one chick. The hope is that as these young birds reach reproductive maturity, they will return to Florida Bay and increase the number of nests each season.

American Crocodile

- A yellow stoplight score for crocodile juvenile growth and survival in ENP is consistent with low growth and survival in the park compared to other primary crocodile nesting areas during the MAP monitoring period beginning in 2005.

Potential Management Relevance: Monitoring of ecosystem characteristics of Everglades coastal wetlands supports the broader management recommendations from hydrology monitoring:

- Restoration of rainfall-driven volume, timing and distribution of freshwater flow to the Everglades coastal wetlands should re-establish persistent zones of freshwater-to-oligohaline salinity. These zones should include mesohaline areas (less than 20 psu) that encompass wide expanses of the freshwater-marine interface. It must be noted that this interface may eventually move inland with rising sea level.

Greater Everglades Ridge and Slough Pattern and Tree Islands

Interim Goals:

Ridge and Slough Pattern
Everglades Tree Islands

MAP Hypothesis Cluster:

Landscape Patterns of Ridge and Slough Peatlands and Adjacent Marl Prairies in Relation to Sheetflow, Water Depth Patterns, and Eutrophication

Interim Goals: The desired restoration condition for ridge and slough pattern is to restore the ridge and slough landscape directionality and pattern by supporting natural soil forming processes and the restoration of ridge and slough microtopography. The desired restoration condition for Everglades tree islands is to improve the health of islands considered to be stressed or degraded, maintain the status of healthy islands, and prevent areal reductions of tree islands except for islands that have expanded due to over-drainage.

Key Findings from the SSR:

<u>*Ridge and Slough Landscape Patterns*</u>

- The coherence of the ridge and slough patterning has deteriorated in areas characterized by the absence of sheetflow.
- In areas where sheetflow has been disrupted, as in impounded or over-drained areas, the differences in ridge and slough elevation are reduced.
- Altered patterns of sheetflow and related variables have caused disequilibrium of accretion and loss processes. That disequilibrium causes degradation of the ridge, slough and tree island microtopography, driving the trend toward a flattening of the landscape.

<u>*Tree Islands*</u>

- The distribution of woody species across the landscape is related to their flood and drought tolerance. For example, canopy growth rates in ENP indicate relative growth rates are reduced at very wet and very dry conditions.
- Water Conservation Area 3B presents a special case – the surviving tree islands are dominated by plants adapted to very wet conditions, yet are currently experiencing very dry conditions. There is concern these tree islands are likely not sustainable in the existing hydrologic regime.

2008 Stoplight Indicators:

- No stoplight indicators exist.

Potential Management Relevance: Monitoring of landscape patterns of ridge and slough peatlands and tree islands reaffirms the broader management recommendations from hydrology monitoring.

- To slow and potentially reverse the degradation of the Everglades tree islands and ridge and slough landscape, it is necessary to restore sheetflow and allow the naturally occurring, rainfall-driven fluctuation of water levels to pulse across the landscape.
- Landscapes that are overdrained or overhydrated due to impoundment are degrading at higher rates. Important steps to resolve these impairments should be made to reduce the risk of long-term degradation even as restoration is being implemented.

Greater Everglades Total Phosphorus, Periphyton Mat Cover, Structure and Composition

<u>Interim Goals</u>:

Everglades Total Phosphorus
Periphyton Mat Cover, Structure and Composition

<u>MAP Hypothesis Cluster</u>:

Oligotrophic Nutrient Status

Interim Goals: The desired restoration condition for Everglades total phosphorus is water column phosphorus concentrations of 10 micrograms/liter. The desired restoration condition for periphyton mats is to restore periphyton communities that were characteristic of the spatially distinct hydroperiods (short and long hydroperiod) and low nutrient conditions in the greater Everglades wetland communities.

Key Findings from the SSR: The overall synthesis from markers of nutrient status (*i.e*., soil, surface water and periphyton) show similar trends and findings.

- A north to south declining nutrient gradient exists and eutrophication is especially evident at discharge points and near canals.

- The Loxahatchee National Wildlife Refuge and the WCAs are the most impacted by eutrophication as a result of their proximity to surface water inflows.

2008 Stoplight Indicator:

- Red stoplight scores for periphyton-epiphyton in areas close to canal sources of phosphorus, and yellow scores in areas downstream to canal inputs of phosphorus, are consistent with the key findings in the SSR. This indicates nutrients targets have not yet been met as defined by the stoplight indictors and corrective action is required.

Potential Management Relevance:

- A major concern for CERP implementation is that additional water provided for restoration of sheetflow and water depth patterns in the Everglades needs to be of sufficient quality so that it does not further degrade the ecosystem.

- A key management uncertainty is whether the capacity of stormwater treatment areas and water storage facilities is enough to sufficiently reduce total phosphorus concentrations in water needed for Everglades restoration.

- It is critical that total phosphorus concentrations in water intended for restoration meet the scientifically-identified conditions needed to avoid detrimental effects to the contiguous swath of marsh running through WCA 3A, WCA 3B, eastern Big Cypress National Preserve and ENP.

SOUTHERN COASTAL SYSTEMS KEY FINDINGS

Salinity Patterns

Interim Goal:

Salinity Patterns in Florida Bay and Biscayne Bay

MAP Hypothesis Cluster:

Salinity

Interim Goals: The desired restoration condition is to reduce the intensity, frequency, duration, and spatial extent of high salinity events, reestablish common mesohaline to oligohaline conditions in mainland nearshore zones, and reduce the frequency and rapidity of salinity fluctuations derived from pulse releases of fresh water from canals.

Key Findings from the SSR:

- Salinity emerged as the most important physical parameter for determining species and community composition in coastal waters.
- Multi-agency salinity data leveraged in the 2009 SSR was used to enhance salinity modeling efforts. Salinity modeling results focused on restoration conditions in Florida Bay estimate suitable habitat could increase from 25,000 to 44,000 acres once favorable salinities are reestablished.
- Salinity modeling in Florida Bay indicated a reduction of hypersaline events in all basins except for Barnes Sound, which showed a slight increase.
- Evaluation of the Salinity Performance Measure[vi] indicated targets are currently being met in only 3 of 11 zones in Florida Bay. In addition, some performance measure components were found to be inaccurate and must be revised.
- Salinity and flow relationships in the Picayune Strand Restoration Project area were incorporated into the Southern Coastal Systems assessment and reported for the first time in the 2009 SSR.

2008 Stoplight Indicator:

- No Stoplight Indicator exists.

Potential Management Relevance:

- The existing salinity regime does not support the desired species and community composition expected under restored conditions. Increases in freshwater inflows are required.

Florida Bay Algal Blooms

Interim Goals:	MAP Hypothesis Cluster:
Florida Bay Algal Blooms	Water Quality and Phytoplankton

Interim Goals: The desired restoration condition is to sustain good water quality in Florida Bay, minimizing the magnitude, duration, and spatial extent of algal blooms in the bay such that light penetration is sufficient to sustain healthy and productive seagrass habitat.

Key Findings from the SSR:

- The sensitivity of the ecosystem to small increases in nutrient availability was documented. These small changes resulted in a damaging algal bloom in Barnes Sound, Manatee Bay, and Blackwater Bay that persisted from 2005 to 2007 and abated in 2008.

- Of the 19 metric tons of phosphorus that fueled the 2005 algal bloom event, only 14 percent was attributed to C-111 discharges with the remainder attributed to other unquantifiable sources.

2008 Stoplight Indicator:

- Conditions reported for 2008 evidence an overall regional improvement in comparison to 2006 and 2007. No sub-regions were designated as "poor" in 2008.

Potential Management Relevance:

- Small nutrient increases can lead to damaging algal blooms. Early detection and source determination of small changes are critical for an appropriate and timely response in order to mitigate nutrient increases.

Southern Coastal Systems Submerged Aquatic Vegetation

Interim Goals:	MAP Hypothesis Cluster:
Submerged Aquatic Vegetation in Southern Estuaries	Submerged Aquatic Vegetation

Interim Goals: The desired restoration condition is a diverse seagrass community with moderate plant densities, more natural seasonality, and with 65-70 percent of Florida Bay having suitable habitat for seagrass growth.

Key Findings from SSR:

- In nearshore Biscayne Bay, shoal grass had a higher probability of being present at lower salinities, while turtle grass (*Thalassia testudinum*) and manatee grass (*Syringodium filiforme*) predominated at higher salinities.

- An independent review of the current monitoring strategy concluded the design will adequately measure SAV changes as restoration progresses.

2008 Stoplight Indicator:

- In Florida Bay, a "fair" status was designated for the Transition, Central, and Southern Zones and "good" status in Northeast and Western Zones.

- Stoplight scores for SAV were not directly comparable to SSR assessments.

Potential Management Relevance:

- The newly established salinity and SAV relationships further understanding of the connection between hydrology and biological responses. This information will be used in the adaptive management process as restoration progresses.

Juvenile Shrimp Densities in Florida and Biscayne Bay

Interim Goals:	MAP Hypothesis Cluster:
Juvenile Shrimp Densities in Florida Bay and Biscayne Bay	Estuarine Nursery Habitat

Interim Goals: The desired restoration condition is a range of annual densities from 2 to 17 juvenile shrimp per square meter (shrimp/m^2), depending upon region and season.

Key Findings from the SSR:

- Pink shrimp (*Farfantepenaeus duorarum*) density has been statistically related to hydrology, with higher densities in wet years and lower densities in dry years.
- Overall wet season pink shrimp abundance evidences a significant downward trend from 2005 through 2009.
- Variation in annual abundance of juvenile pink shrimp reflects not only annual recruitment but environmental conditions in the nursery grounds.

2008 Stoplight Indicator:

- The SSR presents a complete update to the previously reported stoplight, resolving data errors and refining the methods used to calculate stoplights.
- Dry season differences from 2005 through 2009 have been variable; wet season quality has declined except in south Biscayne Bay, which has consistently remained green.

Potential Management Relevance:

- Ranges for species-specific salinity preferences were established for pink shrimp, gray snapper (*Lutjanus griseus*), spotted seatrout (*Cynoscion nebulosus*), and other species. These relationships can be used to predict and assess various restoration scenarios.

TABLE KF-1. INTERIM GOALS, MAP HYPOTHESIS CLUSTERS, AND STOPLIGHT INDICATORS

Geographic Region	Interim Goals (2007)	MAP Hypothesis Cluster (2009)	Stoplight Indicator Report (2008)
Northern Estuaries	American Oysters in Northern Estuaries	Oyster Health and Abundance	Yes
	Submerged Aquatic Vegetation in Northern Estuaries	Submerged Aquatic Vegetation	No
	Flows to the Northern Estuaries	No	No
Lake Okeechobee	Lake Okeechobee Phosphorus	Water Quality and Phytoplankton	No
	Water Levels in Lake Okeechobee	Water Quality and Phytoplankton	No
	Lake Okeechobee Algal Blooms	Water Quality and Phytoplankton	No
	Lake Okeechobee Aquatic Vegetation	Emergent-Submerged Vegetation Mosaic	Yes
Greater Everglades	Water Volume	No	No
	Sheet Flow	Sheet Flow and Water Depth Patterns	No
	Hydropattern	Sheet Flow and Water Depth Patterns	No
	System-wide Spatial Extent of Natural Habitat	No	No
	Everglades Total Phosphorus	Oligotrophic Nutrient Status	No
	Periphyton Mat Cover, Structure, and Composition	Oligotrophic Nutrient Status and Sheet Flow and Water Depth Patterns	Yes
	Ridge and Slough Pattern	Landscape Patterns of Ridge and Slough Peatlands and Adjacent Marl Prairies in Relation to Sheet Flow, Water Depth Patterns, and Eutrophication	No
	Everglades Tree Islands	Landscape Patterns of Ridge and Slough Peatlands and Adjacent Marl Prairies in Relation to Sheet Flow, Water Depth Patterns, and Eutrophication	No
	Aquatic Fauna Regional Populations in Everglades Wetlands	Wading Bird Nesting in the Mainland and Coastal Everglades in Relation to the Aquatic Fauna Forage Base	Yes
	American Alligator	American Alligator Density and Body Condition in Relation to the Hydrologic Patterns and Artificial Canal Habitats in the Everglades	Yes
	System-wide Wading Bird Nesting Pattern	Wading Bird Nesting in the Mainland and Coastal Everglades in Relation to the Aquatic Fauna Forage Base	Yes
	Snail Kite	No	No
	Flows to Northern Boundaries of the WCAs	No	No
	Flows to ENP	No	No

Southern Coastal Systems	Salinity Patterns in Florida Bay and Biscayne Bay	Salinity	No
	Submerged Aquatic Vegetation in Southern Estuaries	Submerged Aquatic Vegetation	Yes
	Juvenile Shrimp Densities in Florida Bay and Biscayne Bay	Estuarine Nursery Habitat	Yes
	American Crocodile	Monitored in the Greater Everglades along with "Ecosystem Characteristics of Everglades Coastal Wetlands in Relation to Freshwater Inflows"	Yes
	Florida Bay Algal Blooms	Water Quality and Phytoplankton	Yes
	Freshwater Flow to Florida Bay	Monitored in the Greater Everglades along with "Ecosystem Characteristics of Everglades Coastal Wetlands in Relation to Freshwater Inflows"	No
	Freshwater Flow to Biscayne Bay	No	No
System-wide	Quantity of Freshwater Lost to Tide	No	No

[i] RECOVER. 2009. Revised CERP Monitoring and Assessment Plan: Part 1 Monitoring and Supporting Research. c/o United States Army Corps of Engineers, Jacksonville District, Jacksonville, Florida, and South Florida Water Management District, West Palm Beach, Florida.

[ii] RECOVER. 2010. CERP Adaptive Management Integration Guide. c/o United States Army Corps of Engineers, Jacksonville District, Jacksonville, Florida, and South Florida Water Management District, West Palm Beach, Florida.

[iii] U.S. Department of the Army, U.S. Department of the Interior, and State of Florida. 2007. Intergovernmental Agreement Among the U.S. Department of the Army, U.S. Department of the Interior, and the State of Florida Establishing Interim Restoration Goals for the Comprehensive Everglades Restoration Plan.

[iv] Doren, R.F., Trexler, J.C., Harwell, M., and Best, G.R., Editors. 2008. System-wide Indicators for Everglades Restoration 2008 Assessment. Unpublished Technical Report. 43 pp.

[v] RECOVER. 2007. Northern Estuaries Performance Measure: Salinity Envelopes. c/o United States Army Corps of Engineers, Jacksonville District, Jacksonville, Florida, and South Florida Water Management District, West Palm Beach, Florida.

[vi] RECOVER. 2008. Southern Estuaries Performance Measure: Salinity. c/o United States Army Corps of Engineers, Jacksonville District, Jacksonville, Florida, and South Florida Water Management District, West Palm Beach, Florida.

CHAPTER 1 INTRODUCTION

1.1 OVERVIEW

The Comprehensive Everglades Restoration Plan (CERP or Plan) is one of the largest ecosystem restoration programs in United States. Authorized by the Water Resources Development Act (WRDA) of 2000 and the Programmatic Regulations (Pro Regs) (Section 601(h)(3)) (WRDA 2000), the goal of the Plan is to restore the South Florida ecosystem[1] while meeting the other water related needs of the region including water supply and flood protection (WS/FP). Per the Pro Regs, REstoration COordination and VERification (RECOVER) is charged with implementing a system-wide[2] monitoring and assessment program to assess implementation of the Plan. This monitoring and assessment plan is essential to determining the success of CERP and is an integral feature of the CERP Adaptive Management (AM) Program.

Formal assessments of data generated from the RECOVER Monitoring and Assessment Plan (MAP), Part 1 (*Monitoring and Supporting Research*) (http://www.evergladesplan.org/pm/recover/recover_map_2004.aspx) are reported in the System Status Report (SSR), which is developed twice every five years by RECOVER. Assessments of MAP monitoring data (*i.e*., MAP monitoring data is augmented with historical and experimental data as well as non-CERP data provided by partner agencies) are performed using the assessment strategy detailed in the MAP, Part 2 (*2006 Assessment Strategy for the MAP* (http://www.evergladesplan.org/pm/recover/recover_map_part2.aspx). The SSR plays an important role within CERP and represents the accumulation of multiple years of data on the status, condition, and trends of performance measures critical to restoration. Future SSRs will present an assessment of whether the goals and purposes of the Plan are being achieved, including whether the interim goals and targets (IG/IT) are being achieved or are likely to be achieved. Additionally, future SSRs will also address the status of WS/FC, another critical aspect of CERP. While scientific data is currently being assessed for continued refinement of pre-CERP reference conditions, an assessment of whether or not the Plan is meeting its goals and objectives is not yet possible since no CERP projects have yet been fully implemented.

1.2 FORMAT OF 2009 SYSTEM STATUS REPORT

The 2009 SSR will be formatted as an interactive web page accessible from www.evergladesplan.org. This web-based approach allows managers, stakeholders and scientists with many different interests and degrees of technical expertise to easily find the information they want and need. Key findings, which provide a high-level synthesis of the assessments, will be available directly from the SSR home page. Key findings will be organized both geographically and thematicly. Key findings can be accessed from both the SSR homepage as well as from each geographic MAP module webpage. Detailed information about each geographic MAP Module (*e.g*., Lake Okeechobee, Northern Estuaries, Greater Everglades and Southern Coastal Systems) will be housed on the interactive webpage as well. Each MAP

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[1] The South Florida ecosystem includes the Kissimmee Region south through the Florida Keys including

[2] For the 2009 SSR, "system-wide" is synonymous with the definition of the South Florida Ecosystem.

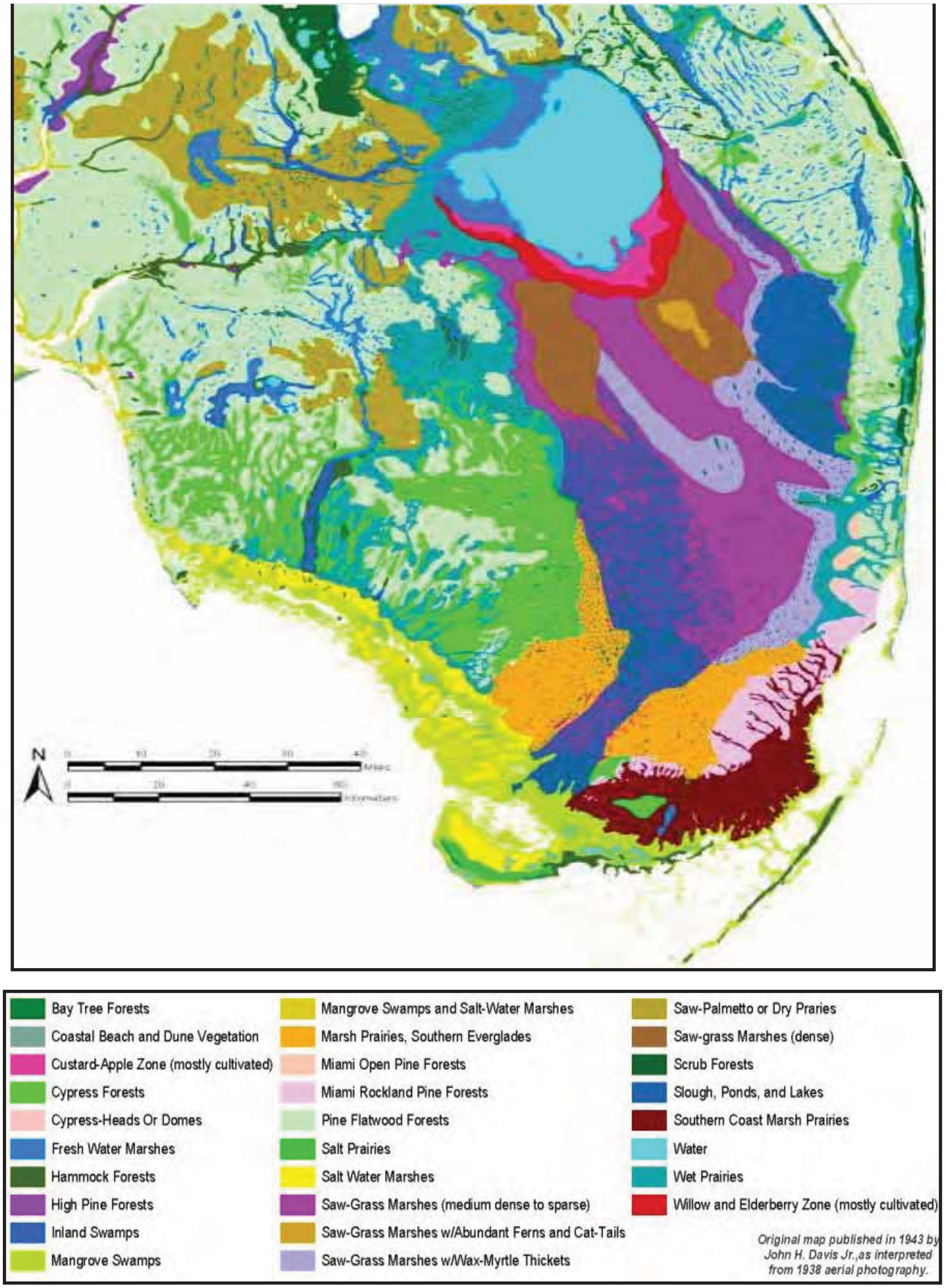
module will address the following topics using the most recent scientific data from the 2009 SSR:

- Validity of the hypothesis cluster concept;
- Metrics and functional relationships;
- Status/trend of key ecological indicators;
- Management issues that communicate high priority areas of concern to the decision makers and managers; and
- Use of hypothesis-based assessments to provide the scientific foundation for evaluating IGs and applying AM.

Information will be presented in a hierarchical way – very general information about each assessment (*i.e.*, general trends in wading birds), slightly more detailed information *(i.e.*, location and number of wading bird nests in Big Cypress) or very detailed information (*i.e.,* specific wading bird survey techniques by location) by downloading reports developed by MAP principal investigators. Even though the 2009 SSR is being presented via an interactive webpage, print capabilities (via PDF) will be available for each MAP module and the document in its entirety.

1.3 BACKGROUND

South Florida was once a diverse mosaic of interconnected landscapes and communities encompassing approximately 3.6 million hectare (ha) that included the expansive, 1.2 million ha freshwater Everglades (Davis and Ogden, 1994). The pre-drainage South Florida ecosystem has been characterized as a hydrologically interconnected, slow flowing, oligotrophic system that extended from the Kissimmee River and Lake Okeechobee southward to the estuaries of Biscayne Bay, Ten Thousand Islands, and Florida Bay (**Error! Reference source not found.**). This system is the product of a unique combination of climate, soil, and topography (Obeysekera et al., 1999). Water depth and distribution, temporally and spatially, are largely determined by seasonal and annual rainfall, evaporation, transpiration, natural topography, overflow through natural streams in the ocean, and the system's capacity for surface- and ground- water storage (SFWMD, 1992). The large water storage capacity that characterized the pre-drainage conditions resulted in a hydrologic system that was much wetter than the current system; this dampened hydroperiod extremes. The system was characterized by alternating high and low water depths, distribution patterns of surface and ground water, and variations in water flow volumes and rates through wetlands into estuaries. These characteristics largely determined the soil and vegetation patterns as well as the distribution, abundance, and seasonal movements and reproductive dynamics of all aquatic and many terrestrial animals in the South Florida ecosystem. The large spatial extent and connectivity of the South Florida system including the Everglades were essential for sustaining populations of species with narrow habitat requirements or large feeding ranges and sustaining regional levels of both primary and secondary aquatic production necessary to support large numbers of higher vertebrates (Ogden et al., 2005).



Published by Davis (1943)

FIGURE 1-1. PRE-DRAINAGE MAP OF SOUTH FLORIDA BASED ON MAP

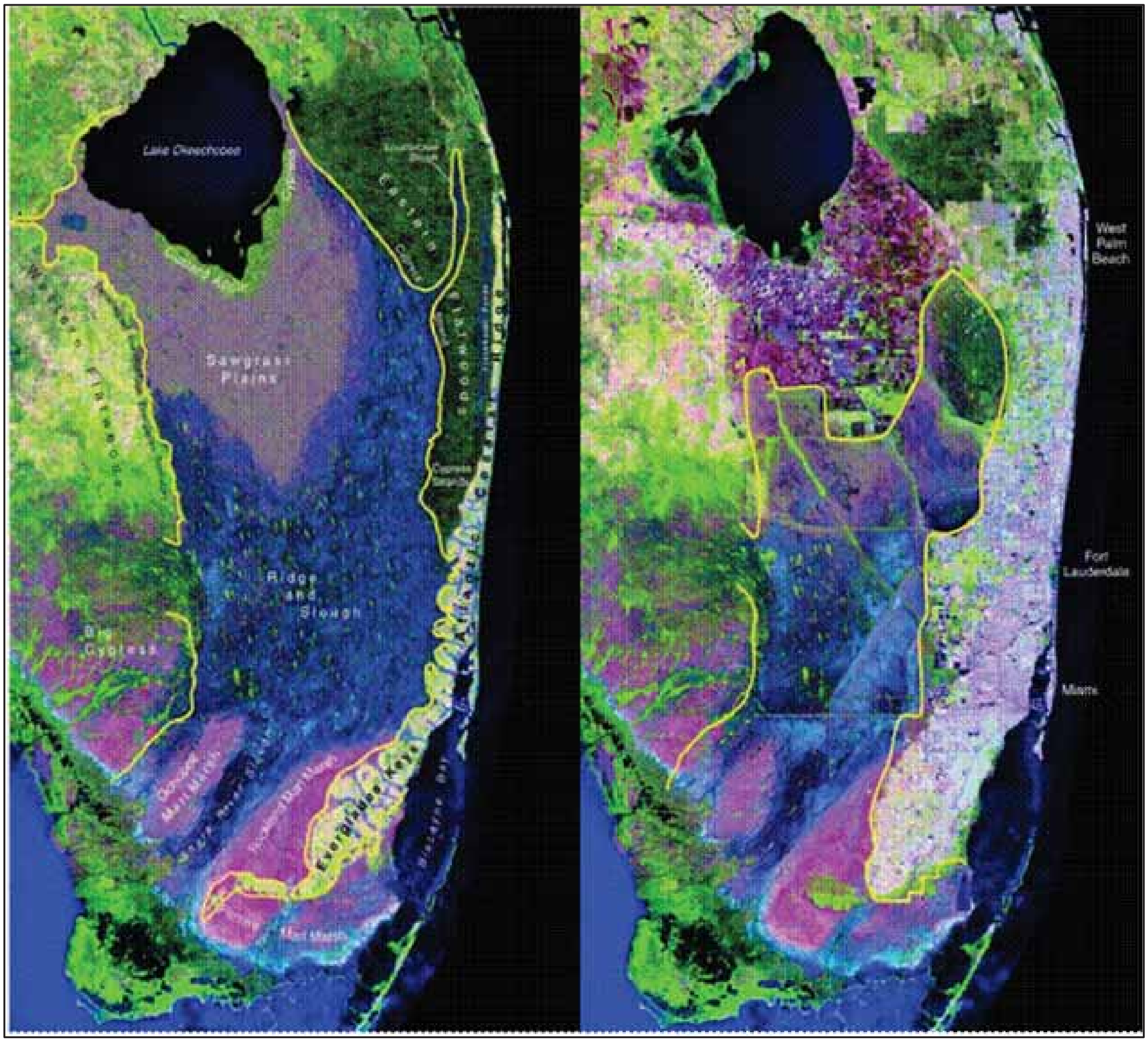
South Florida's rapidly increasing population has resulted in the need for extensive water management particularly in the low-lying wetland areas. Land has been drained for agricultural development and canals and conservation areas constructed for flood control, water retention, water supply, irrigation, and transport. South Florida now contains one of the largest water management systems in the world, the Central and Southern Florida (C&SF) Project (USACE, 1960), which was constructed during the 1950s-1970s to accommodate projected increases in population. Population projections for South Florida are estimated to be approximately 36 million by 2060 (Zwick & Carr, 2006). Presently approximately one-third of the original extent of the greater wetland systems has been lost by conversion to other land uses and the true Everglades has lost 50 percent of it's habitat (***Figure 1-1***) and 70 percent less water flows through the system because more than 1.7 billion gallons are lost to the ocean every day for flood control and water demand for human uses (USACE and SFWMD, 1999). As a result of this environmental degradation and an increasing human population, Congress authorized the CERP in 2000 (USACE and SFWMD, 1999; WRDA 2000) to assist in the restoration of South Florida natural systems (SFERTF, 2000). The primary restoration objectives of the CERP are to increase water storage capacity of the system substantially and distribute water in a manner to reestablish ecologically-desirable patterns of depth, distribution, and flow in freshwater wetlands and desirable salinity regimes in estuaries (Ogden et al., 2003). The plan is based upon the best available science and will employ the concept of adaptive assessment and management to assure the plan is flexible so modifications can be made based upon new information (Ogden et al., 2003).

1.4 COMPREHENSIVE EVERGLADES RESTORATION PLAN

The CERP (www.evergladesplan.org), authorized by WRDA 2000 Section 601(h)(3) (WRDA 2000), is designed to restore the Everglades and South Florida ecosystem while meeting the other water related needs of the region including water supply and flood protection. The ProRegs charge the RECOVER program with developing a system-wide MAP and SSR that is essential to determining the success of CERP and is an integral component of the CERP AM Program.

1.5 MONITORING AND ASSESSMENT PLAN

The goal of the CERP MAP is to establish a framework for measuring and interpreting system-wide responses to CERP, to determine how well CERP is meeting its goals and objectives and, through AM, provide the framework for improving performance of CERP as needed. The MAP (RECOVER, 2004; 2006; 2009) is the primary tool by which the RECOVER program assesses the performance of the South Florida ecosystem as it responds to CERP implementation. The overarching goal for implementation of the MAP is to have a single, integrated, system-wide monitoring and assessment plan that can be used and supported by all participating agencies and tribal governments as the means of holistically tracking and measuring the status of the South Florida and Everglades ecosystem restoration. The intent of MAP is to implement a scientifically robust, long-term monitoring plan with the following objectives: 1) establish a pre-CERP reference condition that includes estimates of condition and variability for each of the attributes used to assess status and trends; 2) provide an assessment of system-wide responses of CERP implementation; and 3) detect unanticipated responses of the ecosystem to changes in stressors resulting from CERP activities. The scientific and technical information generated from MAP implementation will be organized to provide a process for RECOVER to not only evaluate CERP performance and system responses, but to produce assessment reports describing and interpreting these responses.



McVoy, 2005

FIGURE 1-1.
SPATIAL EXTENT OF HISTORICAL GREATER EVERGLADES (LEFT)
SPATIAL EXTENT OF CURRENT GREATER EVERGLADES (RIGHT)

1.5.1 Evolution of the Monitoring and Assessment Plan

Recently the MAP, originally developed in 2004, has undergone additional refinement (http://www.evergladesplan.org/pm/recover/recover_map_2009.aspx). While retaining its focus on long-term system-wide monitoring, assessment and AM, the MAP 2009 provides the added flexibility to address project-level monitoring (PLM). Another key refinement included in MAP 2009 and a direct result of applying the MAP Assessment Strategy (MAP, Part 2) was the development of the hypothesis cluster concept; hypothesis clusters integrate multiple stressor-response pathways (*e.g.*, interface of wading birds, prey-based fish, and flow) contained within a conceptual ecological model (CEM) instead of only addressing a single stressor-response relationship (*e.g.*, the response of wading birds to changes in water levels). The MAP 2009 is focused around hypothesis clusters that address the integration of the important stressor-response elements within a CEM and better capture and represent the complex relationships inherent in the ecosystem.

1.5.2 Monitoring and Assessment Plan Sustainability

Assuring the MAP continues to be based on sound, defensible science is critical to the successful implementation of CERP. The MAP provides this assurance by tightly coupling monitoring, assessment and decision-making through its linkage to AM (see ***Chapter 3***). The MAP addresses five principles central to sustaining and managing healthy ecosystems. These five principles are: 1) socially-defined goals and management objectives; 2) integrated holistic science; 3) broad spatial and temporal scales; 4) adaptable institutions; and 5) collaborative decision making (Machlis et al., 2005). Application of these principles in the MAP allows for the integration of both the ecological and social sciences as an organizing framework with the added benefit of social learning in an AM setting.

1.6 ASSESSMENT APPROACH

The assessment approach utilized by the MAP is outlined in the MAP, Part 2 (http://www.evergladesplan.org/pm/recover/recover_map_part2.aspx). The purpose of the MAP, Part 2 is to provide a systematic framework for analyzing monitoring data and assessing it using a multi-step approach consisting of: (1) monitoring design; (2) data acquisition and management; (3) data analysis; (4) interpretation; (5) assessment; and (6) subsequent evaluation of system-wide performance. The assessment guidance is based on the use of CEMs and hypothesis clusters and utilizes monitoring and assessment to detect changes in the status and trends of ecosystem attributes (including IG/IT – see ***Chapter 4***) and ultimately determine the effectiveness of the Plan. Guidance is provided at the monitoring component, module, and system-wide scales. CEMs were developed for each of the major physiographic regions in South Florida (Wetlands, 2005) and provide an explicit representation of the relationships between natural and anthropogenic activities, the stresses they create, and the resultant ecosystem responses. Hypothesis clusters (see ***Section 1.5.1***) address the integration of the important stressor-response elements contained within a CEM and better capture and represent the complex stressor-response relationships of the system. This strategy provides an assessment of the status and trends of ecosystem attributes, a snapshot in time of "what" is occurring with an attribute which satisfies the need to address IG/IT, and insight to "why" the change is occurring, which is critical to the application of AM (see ***Chapter 3***) (RECOVER, 2006).

1.7 THE SYSTEM STATUS REPORT

The formal assessments of data generated from the MAP are reported in the SSR, which is developed biennially by RECOVER as part of the CERP reporting requirements. The SSR plays an important role within CERP and represents the accumulation of multiple years of data on the status, condition and trends of physical (*e.g.*, hydrology), chemical (*e.g.*, nutrients), and biological (*e.g*., wading birds) performance measures. These metrics can be linked to MAP hypothesis clusters and Interim Goals and are the scientific basis for evaluating the status and progress of restoration. Assessments of MAP monitoring data (*i.e*., MAP monitoring data is augmented with historical and experimental data as well as non-CERP data provided by partner agencies) are conducted using the assessment strategy outlined in the MAP, Part 2. The SSR is a major source of technical information for the RECOVER Technical Report as mandated by the Pro Regs (to be produced no less often than every five years) as well as provides information to the National Research Council's (NRC) Committee on Independent Scientific Review of Everglades Restoration Progress, the Interim Goals Report (as established by the Programmatic Regulations; Section 385.38), and the Five-Year Report to Congress.

The goal of the SSR is to provide a synthesis of current and previously collected hydrological, water quality, and ecological data for each of the geographic modules (Lake Okeechobee, Northern Estuaries, Greater Everglades and the Southern Coastal Systems). This data is compiled, analyzed and interpreted to provide a quantitative assessment of the hypothesis clusters, associated performance measures, and progress toward achieving system-wide IGs. This report identifies those changes from previous year (s) that are inconsistent with the goals and hypotheses and system performance for which corrective action may be required. The SSR is intended to provide the following information:

1. a geographic and temporal synthesis of MAP findings to provide a holistic description of the status and trends of the defining attributes and the validity of hypotheses describing the South Florida and Everglades ecosystem restoration;

2. an interpretation of assessment results in relation to supporting hypothesis clusters, performance measures and achieving system-wide IGs;

3. a summary of those changes in the ecosystem that are consistent with the goals and purposes of the plan and MAP hypotheses;

4. a discussion, when necessary, of why the goals are not being met and/or why the MAP hypotheses should be revised; and

5. identification of major unanticipated findings that may require attention and correction via processes outline in the CERP AM Strategy.

1.7.1 2006 System Status Report

The 2006 SSR was the pilot assessment and represents the initial application of the assessment strategy detailed in the MAP, Part 2. The focus of this 2006 SSR was to use the assessment strategy to determine whether current sampling designs, data quality objectives, variability, power analyses, and relevant spatial-temporal patterns were sufficient to establish a reference state (condition) for important variables and performance measures and to be able to detect change with specified degrees of certainty. The 2006 SSR also characterizes the status of monitoring and data availability for each of the MAP modules and identifies the lessons learned from the assessment process, which will be used to guide future efforts. However, the 2006 SSR is not intended to provide a comprehensive assessment of the ecological condition nor the status of either the MAP modules or the South Florida ecosystem as a whole.

1.7.2 2007 System Status Report

The 2007 SSR was the first comprehensive technical assessment of monitoring data developed using the MAP. Because few CERP projects had been implemented, the 2007 SSR provides estimates of pre-CERP conditions for ecosystem indicators monitored by the MAP, in conjunction with data from other sources. Assessment is done by MAP module. Many of the data sets used in the 2007 SSR are limited to a few years, and thus estimates of pre-CERP reference conditions remain uncertain pending completion of needed monitoring. Sustained multi-year monitoring is a prerequisite for establishing sound estimates of pre-CERP conditions and trends.

1.7.3 2009 System Status Report

The 2009 SSR continues to build upon the comprehensive assessment initiated in 2007 through analysis of additional data; this includes information from new sources (incorporation of non-MAP data) as well as continued compilation of data from ongoing monitoring. Similarly to the 2007 SSR, the 2009 SSR utilizes a hypothesis-based approach to continue establishing pre-CERP reference conditions. Ultimately, the SSR will assess whether the Plan is meeting is goals and objectives but this is currently not possible given that no CERP projects have been fully implemented. The 2009 SSR assesses MAP data by module and begins the complex process of integrating it across geographic regions (*i.e.*, integrated assessment of wading birds in the Greater Everglades as well as the Southern Coastal Systems).

1.8 GLOBAL UNCERTAINTIES AND IMPACTS TO COMPREHENSIVE EVERGLADES RESTORATION PLAN AND THE MONITORING AND ASSESSMENT PLAN

The goal of CERP is “the restoration, preservation, and protection of the South Florida ecosystem while providing for other water-related needs of the region, including water supply and flood protection“(WRDA, 2000). To achieve this goal, the CERP is directed to implement infrastructure changes to enhance water storage, reduce excessive water discharges as a part of existing flood protection procedures, and create a more natural hydrological regime that supports healthy natural communities and enhances regional water supply. A program this complex and extensive in both space and time has uncertainties associated with program planning, implementation, monitoring and assessment. There are a number of approaches that can be used to address uncertainty planning in restoration ecology, including principles of ecological risk assessment (Harwell et al., 2009). While the application of state-of-the-art science and engineering practices can reduce many of the uncertainties, global uncertainties remain. Global uncertainties are factor that have wide-ranging effects and cut across and affect the success of all restoration programs. While it is not the intent here to provide a comprehensive, in depth discussion, it is important to recognize that major uncertainties associated with climate change and sea level rise, invasive and exotic species, and fire can easily re-shape status and trends of ecosystem attributes and impact restoration progress. Clearly South Florida weather is highly dynamic and variable. Within the five-year period covered in this report there have been two consecutive years with four hurricanes followed by two consecutive years of severe drought conditions. The resulting annual, inter-annual, and spatial variability in precipitation has a major affect on hydrology and consequently the status and trends in important ecosystem attributes. Thus it is critical that the success of the MAP and CERP lies in the ability of this program to continue to maintain its long-term monitoring program in order to capture and account for this variability in its trend analysis so that it can effectively discriminate changes that are due to system variability from those resulting from CERP activities.

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CHAPTER 2 ENVIRONMENTAL CHARACTERISTICS OF THE 2008 AND 2009 WATER YEARS

The following is a brief overview of WY 2008 and WY 2009. WY 2008 extends from 1 May 2007 through 30 April 2008 and WY 2009 extends from 1 May 2008 through 30 April 2009. The description of each WY is intended to provide context from a hydrologic perspective for the findings presented in the 2009 SSR. Details about both WYs can be found in the annual *South Florida Environmental Report*, Volume I (https://my.sfwmd.gov/portal/page/portal/pg_grp_sfwmd_sfer/pg_sfwmd_sfer_home) in the 2009 and 2010 Reports.

2.1 WATER YEAR 2008

During WY 2008, the effects of a multi-year rainfall deficit continued to influence south Florida. Following several years of unprecedented hurricane activity and higher-than-normal rainfall, drought conditions continued through WY 2008 and caused far-ranging hydrologic effects in the south Florida ecosystem. Increased rainfall affected the region in late summer 2008 due to the passing of Tropical Storm Fay in August 2008.

<u>Drought Conditions</u> - The 2006–2008 drought ranks in the top six of the most severe regional droughts based on the rainfall received across south Florida during the wet and dry seasons of WY 2007 and WY 2008. Areal rainfall (within the South Florida Water Management District [SFWMD]) in WY 2008 (49.0 inches) was slightly lower than the historical average (52.8 inches) but a marked improvement over the previous WY, which was approximately 12 inches below the average. The Upper Kissimmee Basin, Lake Okeechobee, East Everglades Agricultural Area (EAA), West Everglades Agricultural Area, and the Lower West Coast areas had very low rainfall. Conversely, the Southeast (Broward, Miami-Dade, Everglades National Park), Palm Beach, and Water Conservation Areas (WCAs) 1 and 2 received above-average rainfall. Despite such localized increases, there was not enough rain over the Lake Okeechobee watershed to generate runoff sufficient enough to raise the lake level, thereby placing further constraints on regional water management during the water year.

<u>Drought Effects in South Florida</u> - The 2006–2008 drought caused an imbalance in water inputs and outputs, considerably altering the region's hydrology and reducing surface water flows from the Northern Everglades into the Southern Everglades. During WY 2008, discharges from northern Florida (Lake Kissimmee and Lake Istokpoga) were approximately half of historical average flows, respectively. Lake Okeechobee inflow was half of the historical average and one and a half times that of WY 2007 inflows. Outflows from Lake Okeechobee to the EAA and the Caloosahatchee and St. Lucie estuaries were sharply reduced due to the limited storage in the lake as well as the diminished inflows into the lake. Notably, the WY 2008 outflow from Lake Okeechobee was a record low of 12 percent of the average outflow. Flows into and out of the Everglades Protection area were also drastically reduced.

During WY 2008, water levels in lakes in the Kissimmee and Okeechobee watersheds and the WCAs were lower than their respective historical averages as a result of the extended rainfall deficits. In July 2007, Lake Okeechobee's water level declined to 8.82 feet NGVD – the lowest recorded stage since 1931, the start of the period of record. As lake levels dropped, the amount of available water for storage and supply decreased drastically for all of south Florida. It is important to note that prolonged drought continued through late summer 2008, until Tropical Storm Fay brought drought relief in August 2008.

2.2 WATER YEAR 2009

The hydrology of south Florida for WY 2009 reflects a severe drought with a temporary wet condition in the summer. Meteorologically, WY 2009 was far below average in with respect to rainfall with the exception of the Lower Kissimmee which received average rainfall. There was a distinct difference between dry season (November through May) and wet season (June through October) rainfall. Generally, the wet season was wetter than normal in all rain areas except Palm Beach, Broward County, and the Everglades National Park. Tropical Storm Fay's passage in August 2008 had a significant contribution in making the wet season wetter while the months of September and October during the wet season were drier than average. The dry season was extremely dry with drought return periods of 100-year or more in most areas.

<u>Overview</u> - At the beginning of WY 2009, Lake Okeechobee, continued to show record low water and storage levels. Gravity discharge from the lake was restricted due to persistent low water levels (stage). Since the watersheds of Lake Okeechobee were in continual drought, there was not enough surface water inflow to raise the lake level. Wet season rainfall continued through August, with a major increase in rainfall resulting from Tropical Storm Fay. Rainfall from Tropical Storm Fay was as high as 15.47 inches at some locations. Runoff from Tropical Storm Fay and other rains in August and early September raised Lake Okeechobee's water level by four feet, ending drought conditions temporarily. Rainfall patterns changed to below-average conditions in September 2008 and the pattern continued through the remainder of the WY. As a result, drought conditions returned to the region and drought management was re-initiated. At the beginning of WY 2009, the Lake Okeechobee water level was 10.25 feet NGVD, rising to a maximum of 15.16 feet NGVD in September 2009, but falling back to 11.14 feet NGVD at the end of the WY.

<u>The 2009 Atlantic Hurricane Season</u> – The WY 2009 hurricane season (June 1–November 30, 2008) ended with 16 named storms, eight of which were hurricanes. Four of these systems, Gustav, Hanna, Ike, and Fay, threatened south Florida. Hurricane Gustav passed through the Gulf of Mexico west of the Florida peninsula from August 31–September 1, 2008, and contributed rainfall to south Florida. Hurricane Hanna passed east of south Florida on September 5 and 6, 2008, contributing rainfall to the coastal areas from Palm Beach to the Indian River counties. Hurricane Ike, which devastated Galveston, Texas, contributed rainfall to south Florida as it passed through the Gulf of Mexico from Cuba to Galveston from September 8–13, 2008. Tropical Storm Fay made direct landfall in south Florida, moving across the region longitudinally from the southwest to the northeast and impacting all 16 counties managed by the SFWMD.

<u>Impacts of Tropical Storm Fay</u> - According to the National Oceanic and Atmospheric Administration's (NOAA's) National Hurricane Center, Tropical Storm Fay originated as a tropical wave off the coast of Africa on August 6, 2008. By August 18, the storm had moved northeast of the Florida Keys after a landfall on the keys. On the morning of August 19, 2008, Tropical Storm Fay landed in southwest Florida, near Cape Romano, moving inland across Lake Okeechobee and moving on to the Atlantic Ocean near Melbourne. After lingering for three

days off the coast of the Kennedy Space Center, Tropical Storm Fay made its third landfall in Florida in Volusia County. The storm moved west and northwest across Florida and crossed into Gulf of Mexico. It made a fourth landfall in the panhandle of Florida. Rainfall from the tropical system affected all of the state, with central and south Florida getting storm-related rainfall from August 17–23, 2008.

South Florida had been in severe drought since 2006 when Tropical Storm Fay arrived. The high rainfall and its distribution over the whole region resulted in surface water runoff to fill storage and relieve the drought. Tropical Storm Fay generated enough runoff to fill available storage and created flood conditions in some areas. When the tropical system reached south Florida, the Lake Okeechobee water level was low from an extended drought at 11.25 feet NGVD. The lake water level rose to 15.16 feet NGVD by September 15, 2008, mainly from runoff generated by Tropical Storm Fay. A 3.91 feet water level rise in 30 days for Lake Okeechobee is an extreme event. Inflows to Lake Okeechobee over this period were 1,153,631 acre-feet. These inflows were higher than inflows for WY 2008 (1,012,875 acre feet) and represented 55 percent of the historical average annual inflows.

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CHAPTER 3 ADAPTIVE MANAGEMENT

3.1 BACKGROUND

The WRDA 2000 conveyed the expectation that AM principles would be applied during CERP or Plan implementation. The 2003 Pro Regs specified development of an AM program that includes monitoring and assessment of ecosystem restoration performance, periodic updates of CERP, and continuous improvement of the Plan. These requirements clearly recognized that AM application would be very beneficial to CERP.

By definition, AM is a structured management approach that links science to decision-making, thereby improving the probability of restoration success. Founded on science, AM provides an efficient process to address risk and uncertainty inherent within ecosystem restoration by encouraging flexible plans and designs. When integrated into existing CERP processes, AM improves the implementation of CERP through iterative refinement of project plans, designs and operations.

A multi-agency team, the AM Integration Team, has facilitated development of the CERP AM Program (http://www.evergladesplan.org/pm/program_docs/adaptive_mgmt.aspx), which is composed of two parts: (1) the AM Strategy; and (2) the AM Integration Guide (AMIG). The AM Strategy is used as the restoration framework for CERP. It maps a process for seeking a better understanding of the South Florida ecosystem by using new scientific/technical information to improve the Plan. The AMIG was developed for use by scientists, project delivery teams (PDTs), and managers working on CERP; it is designed to be a more detailed companion document to the AM Strategy. The AMIG identifies: (1) the nine activities required to implement AM at the program and project levels; (2) the significance of those activities; (3) how the activities should be implemented within the existing planning process; and (4) who should conduct the activities.

3.2 THE ROLE OF MONITORING AND ASSESSMENT IN ADAPTIVE MANAGEMENT

There are nine AM activities used to help guide CERP agencies, tribes and stakeholders through the AM process (***Figure 3-1***). The details of each activity are provided in the AMIG. Though RECOVER interfaces directly with each AM activity throughout the implementation process, the RECOVER Assessment Team (AT) is most engaged through participation in the following AM activities:

- Activity 1 Stakeholder Engagement and Interagency Collaboration
- Activity 2 Establish/Refine Restoration Goals and Objectives
- Activity 3 Identify and Prioritize Uncertainties
- Activity 4 Apply Conceptual Models, and Develop Hypotheses and Performance Measures
- Activity 6 Monitoring Ecosystem Restoration
- Activity 7 Assessment
- Activity 8 Feedback to Decision-making

The focus of the AT is on implementing the MAP (Activity 6 – Monitoring) and developing the biennial system status report (Activity 7 – Assessment). There two activities play a critical role in the CERP AM Program because AM advocates using sound science to better inform decision-making and update the Plan as implementation progresses.

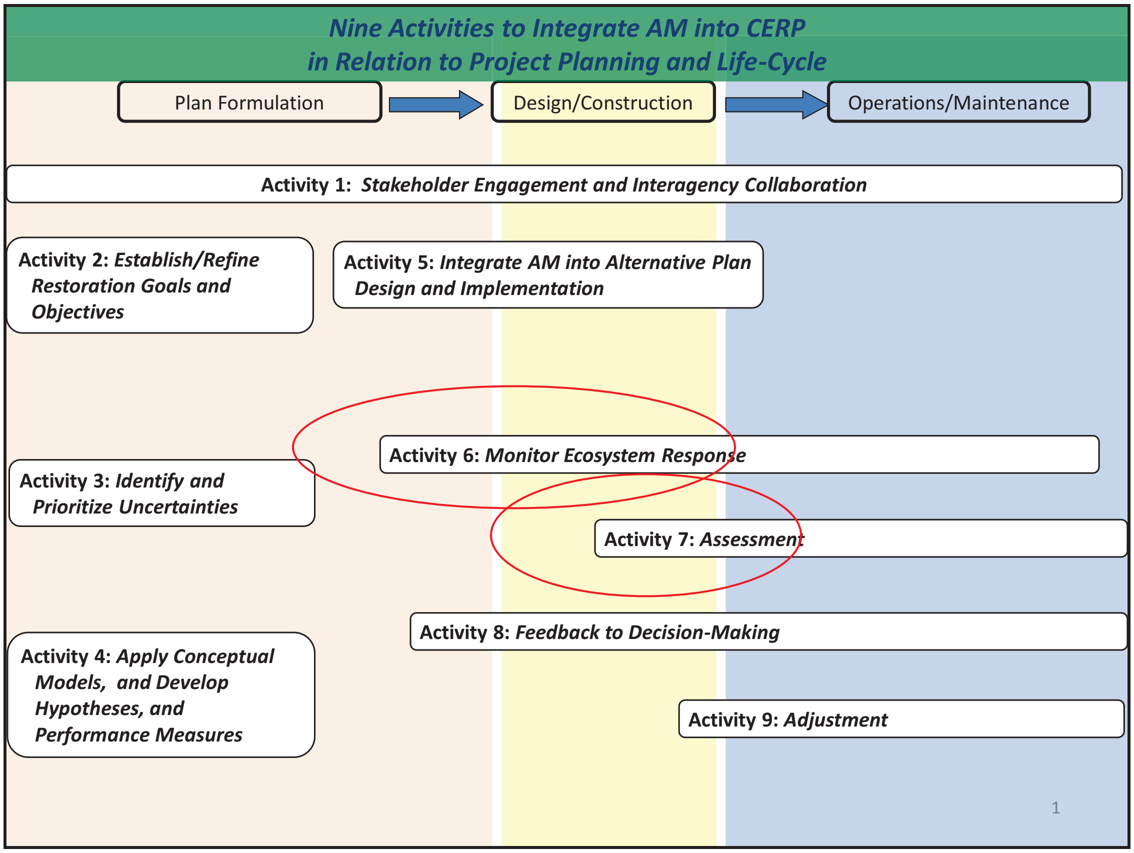


FIGURE 3-1: THE NINE COMPREHENSIVE EVERGLADES RESTORATION PLAN ADAPTIVE MANAGEMENT ACTIVITIES

3.3 ACTIVITY 6 - SYSTEM-WIDE AND PROJECT-LEVEL MONITORING

There are two types of monitoring for CERP: (1) program-level (system-wide); and (2) project-level. An essential element of AM is the development and execution of a scientifically rigorous system-wide/regional monitoring program. The MAP, through implementation of monitoring components, generates information to support understanding of ecosystem responses to CERP implementation. Scientific information is gathered through a comparison of model-based predictions or those based on the best available science, against measurements of ecological response in order to test hypotheses. CERP projects must also address monitoring via development of project-level monitoring plans (PLMP). A PLMP is part of the project implementation report (PIR) and development is a collaborative effort between the PDT and RECOVER. This PDT-RECOVER interaction is critical to both ensure that the PDT is aware of the biological/ecological monitoring implemented by the MAP and for RECOVER to identify existing monitoring being conducted in the project area. This will help ensure project-level and system-wide monitoring are coordinated.

3.4 ACTIVITY 7 - ASSESSMENT

The goals of Activity 7 (Assessment) are to analyze monitoring results and determine if restoration targets are being met in response to CERP implementation, as well as identify potential performance issues. Monitoring information collected via both the MAP and PLMPs must be synthesized, assessed and reported in the appropriate context in order to inform managers about restoration progress; this includes any challenges encountered limiting the achievement of restoration objectives. This information will be used by managers/decision-makers to determine which performance issues should be addressed.

3.4.1 Assessment at the System-wide Scale

Data from the MAP provides the foundation for assessing ecosystem status and trends based on CERP goals and objectives. This data is used to evaluate hypotheses at project-level, regional and system-wide scales. The MAP Module Leads and principal investigators (PIs) are likely to be the first to notice system-wide performance issues (*e.g*., reduction in wading bird population, increased nutrient concentrations) resulting from Plan implementation. Each PI annual report contains a section that clearly identifies ecosystem performance issues, suggests potential scientific explanations for unexpected performance, and recommends general approaches to address needed adjustments to CERP. Potential performance issues may also be identified by the MAP module leads or during development of the SSR since performance issues may be larger than a single geographic region (*i.e*., wading birds are flourishing in the Greater Everglades but as a whole, the population is declining when monitoring from all areas is considered together). These performance issues are incorporated into the SSR and the MAP module lead(s) initiate the steps necessary to validate, or invalidate, them. Module leads elevate performance issues to the AT Chairs and then the AT develops a detailed analysis of the issue and its potential ramifications in coordination with the RECOVER Leadership Group (RLG). The RLG will present this analysis to managers. The majority of the 2009 SSR is focused on assessment at the system-wide scale.

3.4.2 Project-Level Assessment

The project-level assessment process compares data gathered before the project initiation (construction) (*i.e.,* pre-CERP project conditions) to monitoring data gathered after project construction (*i.e.,* post-CERP project conditions). Ecosystem changes observed between pre-CERP and post-CERP project conditions are compared to predicted ecosystem response to project implementation specified during the planning process. The results of the comparisons (*e.g.,* change/no change in conditions, identification of trends) address project hypotheses and guide management. It is envisioned that RECOVER Module Leads will review both system-wide and project-level monitoring and assessment reports. The AT Chairs will coordinate during the course of each year with those coordinating project-level monitoring in order to integrate project-level performance results with the SSR. The SSR will contain a clearly identified section that summarizes project assessment results, clearly identifies performance issues, and suggests potential scientific explanations for any unanticipated performance issues. The 2009 SSR presents limited project-level assessments; primarily for Picayune Strand.

3.5 LINKING MONITORING TO DECISION-MAKING

It is important to note that the identification of performance issues via system-wide assessment can affect the CERP in two ways. First, discovery of an ecosystem performance issue and subsequently the management actions identified to address it can be used to alter CERP implementation (*i.e.*, alter the Plan to ensure its meeting its goals and objectives, including resolution of the performance issue identified). Alternatively, the information garnered from system-wide assessment can be used to refine the MAP and other system-wide/regional monitoring efforts. During implementation of both CERP system-wide/regional and project-level monitoring, it is critical to consider potential outcomes (i.e., the results of monitoring) and how those outcomes could inform decision-making by managers. While this was captured conceptually in the development of the MAP, Part 2 (*2006 Assessment Strategy for the MAP*), management option matrices are needed for interpreting the management actions associated with each monitoring component (***Table 3-1***) for example management option matrix; these matrices are also described in the AMIG). Some management actions may depend on the results of multiple monitoring components. For example, decisions to add or stock oyster larvae require information from various MAP monitoring components, each of which addresses a different variable. Adding or stocking oyster larvae would require information about whether desired salinity ranges are addressed, if the stocking area is part of a defined oyster habitat compatibility area, and if larvae sources are limited to the area.

The table below presents an example of a hypothetical management option matrix for the Northern Estuaries; it includes salinity, water quality, submerged aquatic vegetation (SAV), oysters, macroinvertebrates and fish hypotheses for the Caloosahatchee Estuary. The Caloosahatchee hypotheses are monitored and assessed based on drivers and/or stressors and ecological/biological attributes (*e.g.*, salinity, suitable oyster habitat, water quality sedimentation). Stressors and attributes each have restoration targets and ultimately will be linked to Interim Goals and Interim Targets.

TABLE 3-1: EXAMPLE OF PROGRAMMATIC MANAGEMENT OPTION MATRIX FOR THE CALOOSAHATCHEE ESTUARY

Stressor/Attribute Metric	Target (Timeframe)	Management Action Option 1	Management Action Option 2	Management Action Option 3
Salinity (operational tests may be required to achieve balance between multiple project objectives, i.e., seagrass, oysters, water quality)	5-30 practical salinity units (psu) range (virtually immediate after all projects are operational. Note that this range is the full range over the spatial extent of the entire Caloosahatchee system)	Change operations to meet flow requirements to achieve salinity range and zones		
Oyster HSI	Optimal oyster habitat areas (1-2 years HSI completed)	Conduct management actions in optimal areas		
Oyster Substrate	Acres of suitable habitat (1-2 years surveys completed)	If salinity range is met, add least expensive hard bottom substrates such as concrete rubble if limiting	Add oyster cultch	Dredge muck
Water Quality Sedimentation	Decrease total suspended solids (TSS) entering the estuary (2-3 years)	Adjust flows to minimize sediment transport	Look for opportunities to reduce sediment loads from the watershed such agricultural best management practices (BMPs)	
Water Quality Nitrogen/ Phosphorus	Reduce total nitrogen (TN) and total phosphorus (TP) concentrations to 0.80-0.85 or less and 0.079 mg/L respectively (2-3 years)	Increase stormwater treatment area (STA) acreage in C-43 PIR 2	Decrease overall flows and associated loads	Implement urban and agriculture BMPs
Oyster Recruitment	Presence/absence adults and larvae (2-3 years)	Seed with juveniles	Stock adults	Change operations to avoid too much or too little flow in key months
Oyster Juvenile Growth and Mortality	Attain natural levels of growth and mortality (2-4 years)	If flow/salinity events are affecting growth or mortality, adjust operations to eliminate or minimize events	Excessive predation may require salinity adjustments through operations	Assess the availability of quality food source
Oyster Reef Development	Presence/absence/height of reefs in targeted areas(5-7 years)	Adjust flows to obtain optimal salinities	Add additional cultch	Decrease sedimentation

Oyster Disease	Reduction or Elimination	Operate flows to maintain salinity below maximum threshold	Lower salinity threshold and adjust operations accordingly	
Seagrass	Increase biomass and range of *Valisenaria Halodule* Seagrass (2-5 years)	If water quality targets have not been met, address first	If desired salinity range is met, change operations to adjust flows based on new hypothesis	Implement seagrass plantings in coordination with state, the Department of the Interior (DOI), and National Oceanic and Atmospheric Administration (NOAA)

CHAPTER 4 INTERIM GOALS/INTERIM TARGETS

When the CERP was authorized by Congress in WRDA 2000, establishment of IGs was recommended "to ensure the protection of the natural system is consistent with the goals and purposes of the Plan [CERP]" (Section 601). The Pro Regs (2003) further developed the concept of IGs as "a means by which the restoration success of the Plan may be evaluated throughout the implementation process" (33 Code of Federal Regulations [C.F.R] Part 385.38). Additionally, the Pro Regs required the development of ITs for "evaluating progress towards achieving other water-related needs of the region, including water supply and flood protection" (33 C.F.R Part 385.39). IG/ITs are to be established in five-year increments, beginning with a baseline and ending at full implementation. IG/ITs not only provide a means for agency managers, the State of Florida, and Congress to evaluate progress throughout the overall planning and implementation process of the CERP, but also facilitate the use of AM. The use of AM helps monitor project performance, detect unexpected results, and provides the opportunity for adjustments as necessary.

In 2005, RECOVER used the best science and information available to draft recommendations for IG/ITs. During this process, ecosystem attributes (indicators) were selected that related to the purposes of the CERP, and predictions were made regarding how these indicators would change as CERP projects were implemented. Based upon these recommendations, the State of Florida, DOI, and USACE executed an Intergovernmental Agreement (2007) to establish a set of IGs (Column 2 in ***Table 4-1***), and the State of Florida and the USACE executed an Intergovernmental Agreement (2007) to establish a set of ITs (Column 1 in ***Table 4-2***) for the CERP. It was understood that certain assumptions (e.g., that the hydrologic model used to formulate CERP was accurate, that projects would be fully funded and constructed consistent with the original CERP schedule) and limitations (e.g., uncertainties with models and science, inability of models to provide incremental interim target performance predictions for certain indicator conditions) were associated with RECOVER's recommendations; thus, it was noted that RECOVER would continue to refine IG/ITs, develop predictive measures, and improve incremental performance predictions.

As of the 2009 SSR, no CERP projects have been fully implemented; however, through system-wide monitoring and assessment, pre-CERP status and trends of important ecosystem attributes have been established. Efforts are now focused on revising the existing set of IGs to establish more explicit linkages to ongoing MAP monitoring and assessment activities. In parallel, an initiative is underway to explore merging the SSR (i.e., hypothesis-cluster analyses that describe stressor-response functions) with the System-Wide Indicators for Everglades Restoration Report (i.e., stoplight indicators that describe the status/trends of specific ecological attributes). By creating an interface between the two reporting vehicles, RECOVER will be better able to revise and define the IGs necessary to achieve successful restoration (***Table 4-1***). Although the SSR and MAP are not directly addressing ITs, aspects of the ITs are being addressed in the development of performance measures and assessment models (***Table 4-2***). Since establishment of the Intergovernmental Agreements in 2007, the Integrated Delivery Schedule has evolved. As a result, future activities will be required to explore how the current approach of five-year modeling increments corresponds to future planning efforts for IG/ITs.

TABLE 4-1. INTERIM GOALS, MONITORING AND ASSESSMENT PLAN HYPOTHESIS CLUSTERS, SYSTEM STATUS REPORT, AND STOPLIGHT INDICATORS

Geographic Region	Interim Goals (2007)	Monitoring and Assessment Plan Hypothesis Cluster (2009)	System Status Report (2009)	Stoplight Indicator Report (2008)
Northern Estuaries	American Oysters	Oyster Health and Abundance	Yes	Yes
	Submerged Aquatic Vegetation	Submerged Aquatic Vegetation	Yes	No
	Flows	No	No	No
Lake Okeechobee	Phosphorus Concentrations	Water Quality and Phytoplankton	Yes	No
	Water Levels	No	Yes	No
	Algal Bloom Frequency	Water Quality and Phytoplankton	Yes	No
	Aquatic Vegetation	Emergent-Submerged Vegetation Mosaic	Yes	Yes
Greater Everglades	Water Volume	No	Yes	No
	Sheet Flow	Sheet Flow and Water Depth Patterns	Yes	No
	Hydropattern	Sheet Flow and Water Depth Patterns	Yes	No
	Spatial Extent of Natural Habitat	No	No	No
	Phosphorus Concentrations	Oligotrophic Nutrient Status	Yes	No
	Periphyton	No	Yes	Yes
	Ridge and Slough Pattern	Landscape Patterns of Ridge & Slough Peatlands and Adjacent Marl Prairies in Relation to Sheet Flow, Water Depth Patterns, and Eutrophication	Yes	No
	Tree Islands	Landscape Patterns of Ridge & Slough Peatlands and Adjacent Marl Prairies in Relation to Sheet Flow, Water Depth Patterns, and Eutrophication	Yes	No
	Aquatic Fauna (Fish)	Wading Bird Nesting in the Mainland and Coastal Everglades in Relation to the Aquatic Fauna Forage Base	Yes	Yes
	American Alligator	American Alligator Density and Body Condition in Relation to the Hydrologic Patterns and Artificial Canal Habitats in the Everglades	Yes	Yes
	Wading Bird Nesting	Wading Bird Nesting in the Mainland and Coastal Everglades in Relation to the Aquatic Fauna Forage Base	Yes	Yes
	Snail Kite	No	No	No
	Flows to Water Conservation Areas (WCAs)	Ecosystem Characteristics of Everglades Coastal Wetlands in Relation to Freshwater Inflows	Yes	No
	Flows to Everglades National Park	Ecosystem Characteristics of Everglades Coastal Wetlands in Relation to Freshwater Inflows	Yes	No

Southern Coastal Systems	Salinity in Florida and Biscayne Bays	Salinity	Yes	No
	Submerged Aquatic Vegetation	Submerged Aquatic Vegetation	Yes	Yes
	Juvenile Pink Shrimp	Estuarine Nursery Habitat	Yes	Yes
	American Crocodile	No		Yes
	Algal Blooms in Florida Bay	Water Quality and Phytoplankton	Yes	Yes
	Freshwater Flow to Florida Bay	No	No	No
	Freshwater Flow to Biscayne Bay	No	No	No
System-Wide	Freshwater Lost to Tide	No	No	No

Note: This table lists the IGs by MAP geographic region and describes whether aspects of the IGs are captured within the MAP monitoring and assessed within the SSR and/or the System-Wide Indicators for Everglades Restoration 2008 Assessment (Stoplight Indicator Report).

TABLE 4-2. INTERIM TARGETS, EVALUATION PERFORMANCE MEASURES AND ASSESSMENT METRICS

Interim Targets	Evaluation PMs	Assessment Models
Water Volume	Sheet Flow in the Everglades Ridge and Slough Landscape (Greater Everglades Sheetflow)	EDEN SFWMM
Water Supply for Lower East Coast	Frequency of Water Restrictions for the Lower East Coast Service Area (WS-2)	EDEN SFWMM
Water Supply for Lake Okeechobee	Lake Okeechobee Stage (LO-1, LO-2, LO-3), Frequency of Water Restrictions for the Lake Okeechobee Service Area (WS-1)	Lake Okeechobee hydrodynamic model, Lake Okeechobee Water Quality Model (LOWQM), Lake Okeechobee Environmental Model (LOEM)
Protect Biscayne Bay from Saltwater Intrusion	Prevent Saltwater Intrusion of the Biscayne Aquifer – Meet MFL Criteria for Biscayne Aquifer (WS-4), Prevent Saltwater Intrusion of the Biscayne Aquifer in South Miami-Dade County (WS-5), Southern Coastal Systems Salinity (under revision)	HYCOM, CAFE3D Numerical Hydrodynamic and Mass Transport Model of Biscayne Bay, TABS-MDS, Saltwater Intrusion Model
Flood Control: Root Zone Groundwater Levels in the South Miami-Dade Agricultural Area East of L-31N	Comparison of Stage Differences of Water Levels in South Miami-Dade Agricultural Area (WS-6)	SFWMM
Flood Control: Groundwater Stages for Miami-Dade, Broward, and Palm Beach Counties and Seminole Tribe Surface Water Management Basins	Comparison of Stage Differences of Water Levels in South Miami-Dade Agricultural Area (WS-6)	SFWMM
Flood Control: Flood Water Removal Rate for the Everglades Agrigculture Area (EAA)	Potential for High Water Levels in South Miami-Dade Agricultural Area (WS-3)	No
Surface Water Storage Capacity	Appendix to Band 1	No

Note: This table lists the ITs and describes whether aspects of the ITs are captured within evaluation performance measures and assessment models that have been developed, or are under development.

CHAPTER 5 DATA MANAGEMENT

5.1 INTRODUCTION

The approach for assessing the status of the Everglades ecosystem depends on compiling and integrating biological, chemical and physical data collected by multiple organizations and scientists. A data management system is a necessary component of this effort as data collected to this point are distributed among a variety of information storage systems. Integration of information would greatly facilitate interpreting ecological change as restoration progresses, and as such, is a critical element in the implementation of AM. Tools and applications for supporting data management, system assessment and system evaluation continue to be developed. Many of these tools have been and will be used to produce SSRs.

Standards for data, documentation and coding, established earlier in the program are facilitating the pursuit of seamless data set integration where applicable. Documentation of these standards are part of the CERP Guidance Memoranda (GM) and can be found at www.evergladesplan.org/pm/program_docs/cerp-guidance-memo.aspx. In addition to the GM, the Information and Data Management Program Management Plan (USACE and SFWMD, 2007) can be found at www.evergladesplan.org/pm/progr_data_mgmt.aspx. In order to comprehensively document collected scientific data, Ecological Metadata Language (EML), has been adopted to standardize documentation of ecological data sets.

5.2 SUMMARY OF COMPREHENSIVE EVERGLADES RESTORATION PLAN DATA MANAGEMENT APPLICATIONS

Tools and applications for supporting data management, system assessment and system evaluation have been and are being developed within an information technology environment known as the CERPZone (www.cerpzone.org). The CERPZone provides a collaborative environment for facilitating the multi-agency effort to implement the CERP MAP using an AM approach. The tools and applications fall into several categories based on their use: 1) project and program management, 2) document and data storage and management, 3) geographic information system (GIS), 4) assessment applications, 5) evaluation (predictive and planning) applications and 6) project-level applications. Many of the functions of these tools and applications overlap. As they are being developed, consideration is being given to integration among the tools and applications and their functionality. A summary of these tools and applications is provided here while some are discussed in more detail in the following sections.

The tools and applications that have been developed or are under development for CERP data management are as follows:

1. Project and program management
    - MapTrack

2. Document and data storage and management
    - Documentum
    - Data Access, Search and Retrieval (DASR)
    - Electronic Data Catalog (EdCat)
    - Morpho (metadata tool)

3. GIS
   - Monitoring Locator
   - Gazateer
   - Map Library
   - CERP GIS Data Catalog
   - Survey Monuments Locator
   - Oyster Habitat Suitability Index (HSI)
   - Everglades Depth Estimation Network (EDEN) Client

4. Assessment
   - RECOVER Assessment Application
     - salinity
     - oysters
     - periphyton
     - surface water nutrients
   - Oyster HSI
   - Geo-referenced Interactive Data Analysis System Tool (GIDAST)
   - EDEN Client

5. Evaluation
   - CERP Model Management System (CERP MMS)
   - GIDAST
   - Across Trophic Level System Simulation (ATLSS) High Resolution Hydrology (HRH)
   - Four-dimensional High Resolution Hydrology (4DHRH)
   - Everview

6. Project-level
   - Aquifer storage and recovery (ASR) database

Figure 5-1 and ***Figure 5-2*** both show the relationships between these applications and tools. ***Figure 5-1*** represents the applications and tools that were available and under development at the time the 2007 SSR (RECOVER, 2007) was published. ***Figure 5-2*** presents the applications and tools that are now available or under development. Comparison of these two figures illustrates the progress achieved in the last two years. The overlap of objects in the diagrams represent integration among the applications and their functionality. Dotted lines indicate applications and tools planned for development and implementation. While a great deal of progress has been made, most of these tools and applications are implemented in phases, and require ongoing enhancements and upgrades to support the dynamic needs of the RECOVER program assessment and evaluation initiatives.

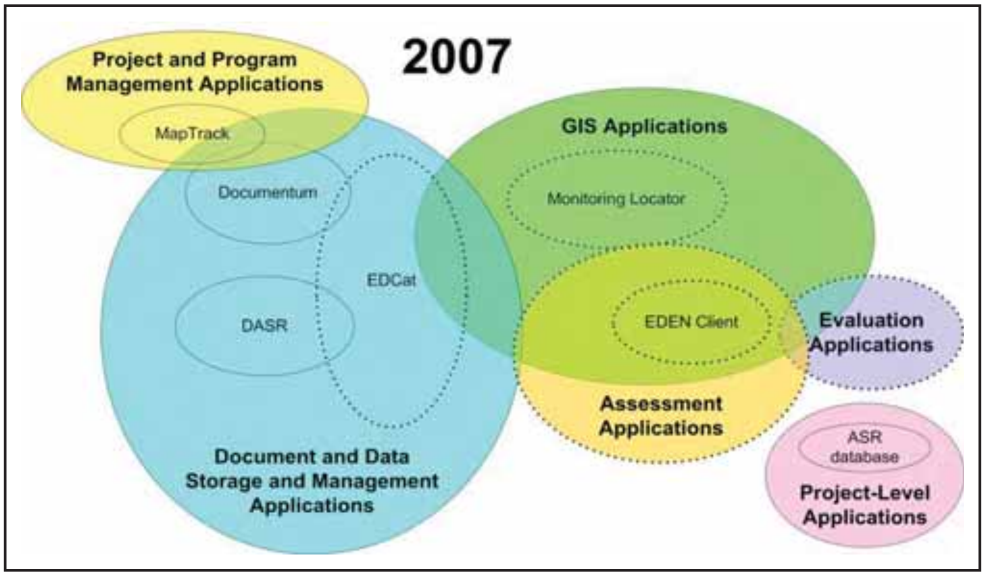


FIGURE 5-1. DATA MANAGEMENT APPLICATIONS AND TOOLS DEVELOPED OR IN DEVELOPMENT IN 2007 AND THEIR RELATIONSHIP TO EACH OTHER

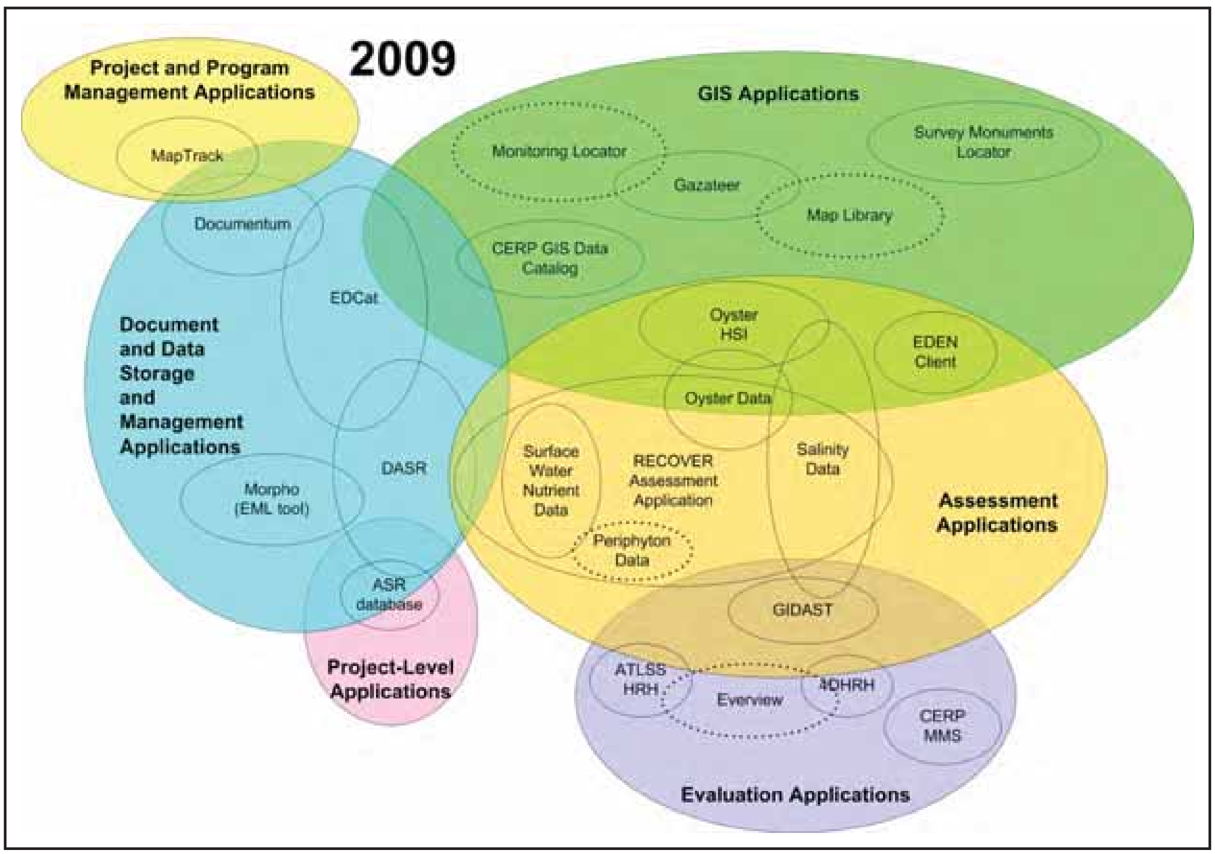


FIGURE 5-2. DATA MANAGEMENT APPLICATIONS AND TOOLS THAT HAVE BEEN DEVELOPED TO DATE OR ARE UNDER DEVELOPMENT AND THEIR RELATIONSHIP TO EACH OTHER

5.3 APPLICATIONS UTILIZED IN DEVELOPMENT OF 2009 SYSTEM STATUS REPORT

The ***EDEN Client*** is considered both an assessment and a GIS application, which provides an interface to the U.S. Geological Survey (USGS) integrated network of real-time water level monitoring, and ground elevation and water surface modeling data. It provides scientists and managers with 1999 to present water depth information for the entire freshwater portion of the Greater Everglades. The target users are biologists and ecologists examining trophic-level responses to hydrodynamic changes in the Everglades. EDEN offers a consistent and documented data set that can be used not only by scientists, but also 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

The ***Oyster HSI*** is also considered both an assessment and GIS application. It is used to predict the locations of future suitable habitat for the eastern oyster (*Crassostrea virginica*). This application supports the study of HSI of an oyster lifecycle given various scenarios of inputs such as flow rate, temperature, salinity and length of oyster life cycle.

The ***RECOVER Assessment Application*** serves as the data integration mechanism for system-wide assessments. Data is integrated across agencies and disciplines for RECOVER module- and system-level analysis. A direct link to SFWMD's DBHYDRO database from the RECOVER Assessment Application supports the integration of hydrology, meteorological and surface water nutrient data. To date, the RECOVER Assessment Application has been used to integrate east coast oyster data obtained from the U.S. Fish and Wildlife Service (USFWS) and west coast oyster data obtained from Florida Gulf Coast University. It automates various statistical analyses and delivers results to graphing and analytical third party software. Flow data from DBHYDRO is integrated with the west coast oyster data. These collective data sets support the Oyster HSI application. Salinity and water quality data from multiple agencies were integrated to produce indices and wet and dry season subsets that were geospatially analyzed (kriged) for the Southern Coastal Systems and Northern Estuaries Modules of this SSR. Integration of periphyton species and nutrient data sets from the South Florida Water Management District (SFWMD) and Florida International University is underway. These data sets will be compiled, aggregated, and processed in the coming year for future integrated assessments for the Greater Everglades Wetlands Module. A pilot spatial interface and viewer for the RECOVER Assessment Application has been developed, but is not yet ready for release into production.

5.4 OTHER APPLICATIONS RELEVANT FOR MONITORING AND ASSESSMENT PLAN IMPLEMENTATION AND FUTURE SYSTEM STATUS REPORT DEVELOPMENT

5.4.1 Project and Program Management Applications

The ***MapTrack*** combines SFWMD and the US Army Corps of Engineers (USACE) financial and project information to facilitate contract management.

5.4.2 Document and Data Storage and Management Applications

RECOVER and CERP documents are stored in a centralized database using ***Documentum*** for organization and version control. Data generated by implementation of the MAP is archived in ***DASR***. Recently implemented in the CERPZone, is a metadata creation and maintenance application, ***Morpho***. This application serves to standardize documentation of ecological metadata for data stored in DASR. ***EDCat***, an electronic data catalog, provides Google-like searches in Documentum, and has been augmented with MetaCarta technology to allow a spatially-based search of Documentum, DASR, the CERP GIS Catalog (discussed below), and the public website www.EvergladesPlan.org. The search results from this tool provide statistical relevance by location and a preview of the data items returned without accessing the storage applications.

5.4.3 Geographic Information System Applications

Several GIS applications have been developed for CERP. The ***Gazetteer*** application is a GIS database that aims to standardize place names and footprints for "officially named" places. The ***CERP GIS Data Catalog*** is a mechanism to facilitate locating CERP GIS data and is a core element of the CERP GIS data publishing process. The ***Survey Monuments Locator*** displays CERP monument locations and allows the user to search for an area of interest and find information such as elevation heights, where the monuments are in relation to roads and highways, and links to data sheets from the National Geodetic Survey website. The ***CERP MMS*** application is a web-based modeling management system that facilitates the accessibility and availability of CERP and non-CERP modeling information, including pre- and post-processed data and modeling documentation, using a GIS-based internet software.

The ***Map Library*** is used by CERP staff as a cataloging application that both stores map products and allows the user community to search and download map metadata and images. The application utilizes Gazetteer place names to enable interactive spatial searches for maps associated with a number of spatial features including projects, counties, cities and water features. The new Map Library utilizing ArcGIS Server technology will be available early in 2010.

The ***Monitoring Locator*** application will allow users to query feature attributes, identifying what data is being collected, where and by whom. It will access metadata, but will not store or display the monitoring data itself. The prototype application was available for review at the end of 2009. The upload data process is currently being programmed and the final application should be available mid-2010.

5.4.4 Evaluation Applications

Several evaluation applications support the predictive and planning portions of the program. This set of applications is intended to be the conduit for processing and integrating model output, as well as integrating assessment information input into models and model analysis.

The ***GIDAST*** tool, which is considered an assessment application, provides an interactive spatial viewer to explore modeled and observed salinity data at specific locations of interest to facilitate interpretation of ecological observations. This tool is currently functional for Biscayne Bay, St. Lucie Estuary and Indian River Lagoon, and Caloosahatchee River Estuary.

A multitude of model output data is used for both evaluation and assessment processes. Various tools have been developed to post-process South Florida Water Management Model (SFWMM) output for use in evaluating plans. One of these tools is the ***ATLSS HRH*** model. From this model a four-dimensional HRH model, ***4DHRH***, has been developed, which encompasses four dimensions: north-south, east-west, depth and time. These tools can be used when developing assessments from monitoring data. Both of these tools have been implemented in the CERPZone with a user interface to facilitate iterative usage of these tools. A pilot version of an enhanced viewer, ***Everview***, is in the final stages of development by the USGS's National Wetlands Research Center and will be implemented in CERPZone in 2010. This will provide a comprehensive and versatile interface for the numerous eco-tools utilized by the evaluation and assessment teams of RECOVER.

CERP MMS enhances communication among project managers, modeling liaisons and modelers by providing modeling support and materials in a timely and efficient manner. Using selection criteria, the user can find information regarding CERP Projects and related models. Once the desired model is identified, the user can download model code, input and output files and documentation through the application interface.

5.4.5 Project-Level Applications

To date, only one application has been developed specifically for CERP project management. A database has been developed in the CERPZone to manage the data associated with the ASR pilot studies. Much of this data is stored in DASR.

5.5 REFERENCES

RECOVER. 2007. 2007 System Status Report. Restoration Coordination and Verification Program c/o U.S. Army Corps of Engineers, Jacksonville, FL, and South Florida Water Management District, West Palm Beach, FL. July 2007.

USACE and SFWMD. 2007. Information and Data Management Program Management Plan. United States Army Corps of Engineers, Jacksonville, FL, and South Florida Water Management District, West Palm Beach, FL. April 2007.

CHAPTER 6 LAKE OKEECHOBEE MODULE

6.1 INTRODUCTION

Lake Okeechobee is a shallow, eutrophic lake located in central south Florida. Historical and background information, including its importance to the south Florida ecosystem and the impacts development has had on the lake can be found in the 2007 SSR (RECOVER, 2007), which can be found at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx. A prolonged drought between 2006 and 2008 resulted in a record low lake stage of 8.82 feet above mean sea level (msl) on July 2, 2008. Vast inshore areas of the lake became exposed to the air and terrestrial emergent vegetation replaced open water habitat. A map of Lake Okeechobee is provided in ***Figure 6-1***.

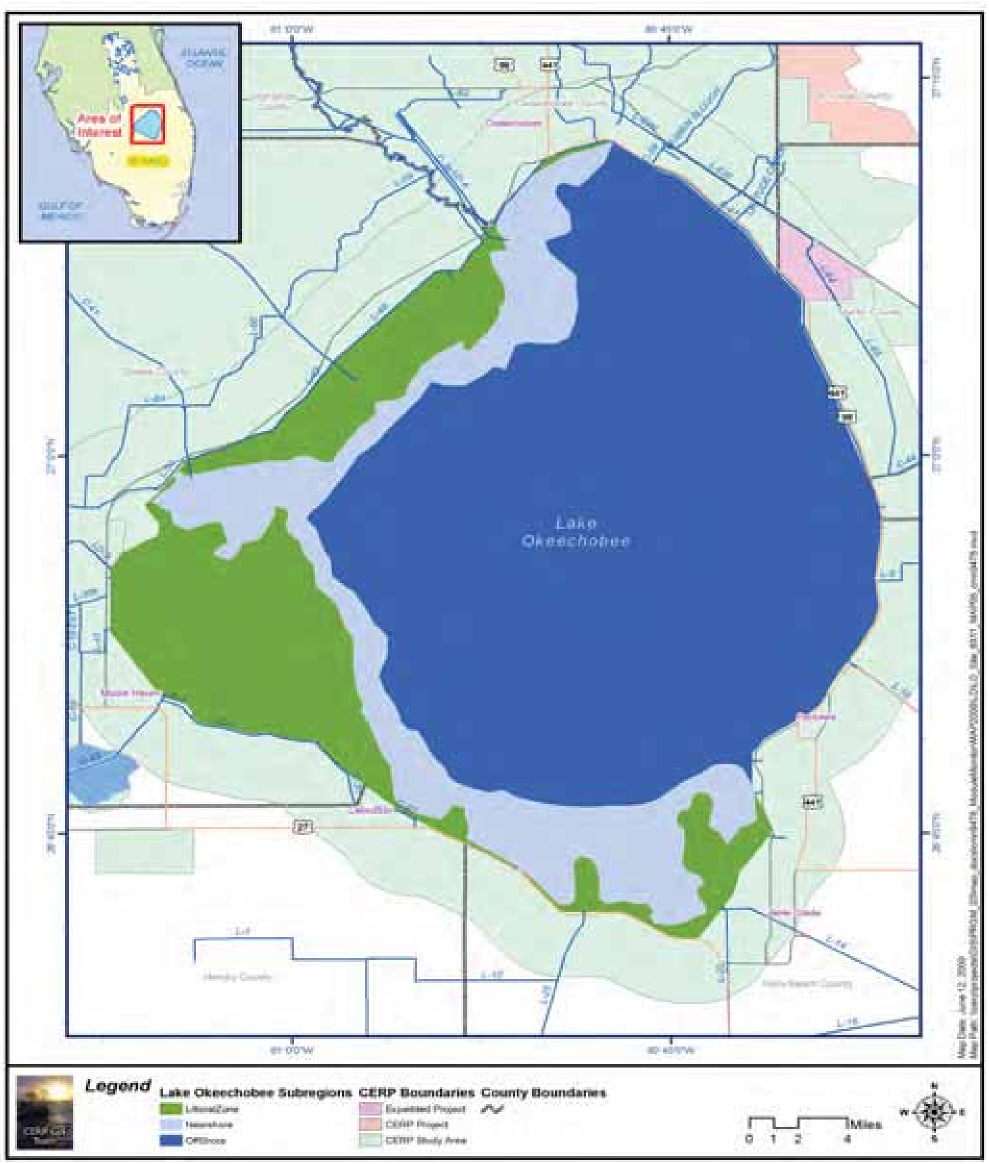


FIGURE 6-1. LAKE OKEECHOBEE SUBREGIONS

As a result of the varied and widely held concerns, CEMs and hypothesis clusters were developed for Lake Okeechobee to provide a science-based path forward toward restoration (RECOVER, 2006). These models and hypotheses succinctly depict the interrelationships that exist between water level and nutrient condition, and those key flora and faunal communities that respond to or are affected by them. The models account for Lake Okeechobee's three subregions that are functionally dissimilar and, as a consequence, may respond to changes in water level and water quality quite differently. These subregions are a littoral marsh, a nearshore region and an open water region (***Figure 6-1)***. The status of Lake Okeechobee is assessed using these CEM and hypotheses.

This module is organized into ten subsections, including this introductory section. The next two subsections provide an overview of the two main stressors on Lake Okeechobee: lake stage and water quality. Following the stressor discussions are subsections for each of the hypothesis clusters for this module: phytoplankton, periphytion, SAV, native fish, and macroinvertebrates. Finally, an overall assessment of the module and references are provided.

6.2 STAGE STRESSOR

6.2.1 Introduction and Background

Water level in Lake Okeechobee is a primary factor affecting both the aquatic vegetation and the community of animals that use these plants for habitat and sustenance (Johnson et al., 2007). It is a stressor in all of the Lake Okeechobee Module hypothesis clusters (***Figure 6-2***). Lake Okeechobee stage history between January 2008 and May 2010, which is the most recent stage plot, is superimposed over the regulatory discharge zones in ***Figure 6-3***. Water discharges can occur in water management zones A-C, while smaller pulse releases can occur in water management zones D and E. The seasonal range in lake stages encompassed by each zone reflects variability in rainfall and watershed runoff typically observed between the dry and wet seasons. The white sub-band lines in Zone D denote the upper and lower boundaries for pulse release consideration. The solid colored line denotes actual lake stage (text box in upper right hand corner shows the most recent lake stage), while the dotted colored lines denote the projected quartile probabilities for lake stage position (***Figure 6-3***). During the fall of 2006, water restrictions were enacted, which continued through 2008 (SFER, 2009). Currently, lake stage operations in Lake Okeechobee follow the Lake Okeechobee Regulation Schedule (LORS) 2008. Lake Okeechobee stage is also a performance measure. Further information and documentation for this performance measure can be found at www.evergladesplan.org/pm/recover/recover_docs/et/lo_pm_stage.pdf. An interim goal has been developed for Lake Okeechobee lake stage, documentation for which can be found at www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_2-2_lakeowaterlevels.pdf.

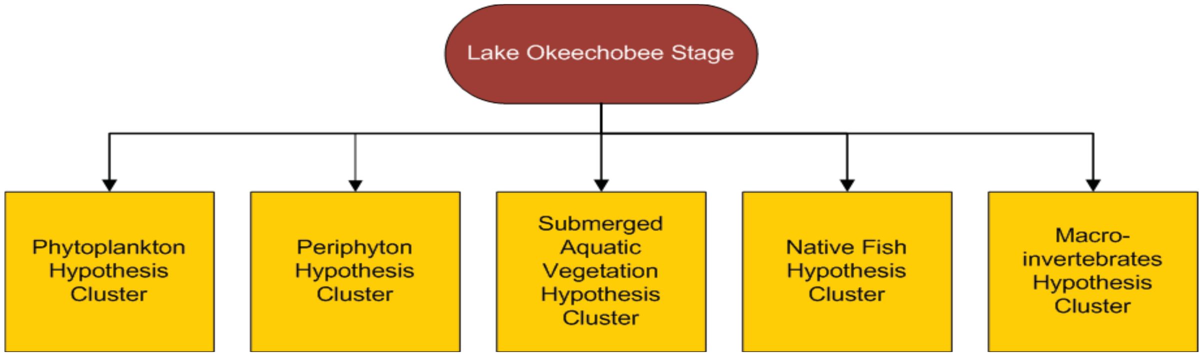


FIGURE 6-2. STAGE AS A PRIMARY STRESSOR AFFECTING ALL LAKE OKEECHOBEE HYPOTHESIS CLUSTERS

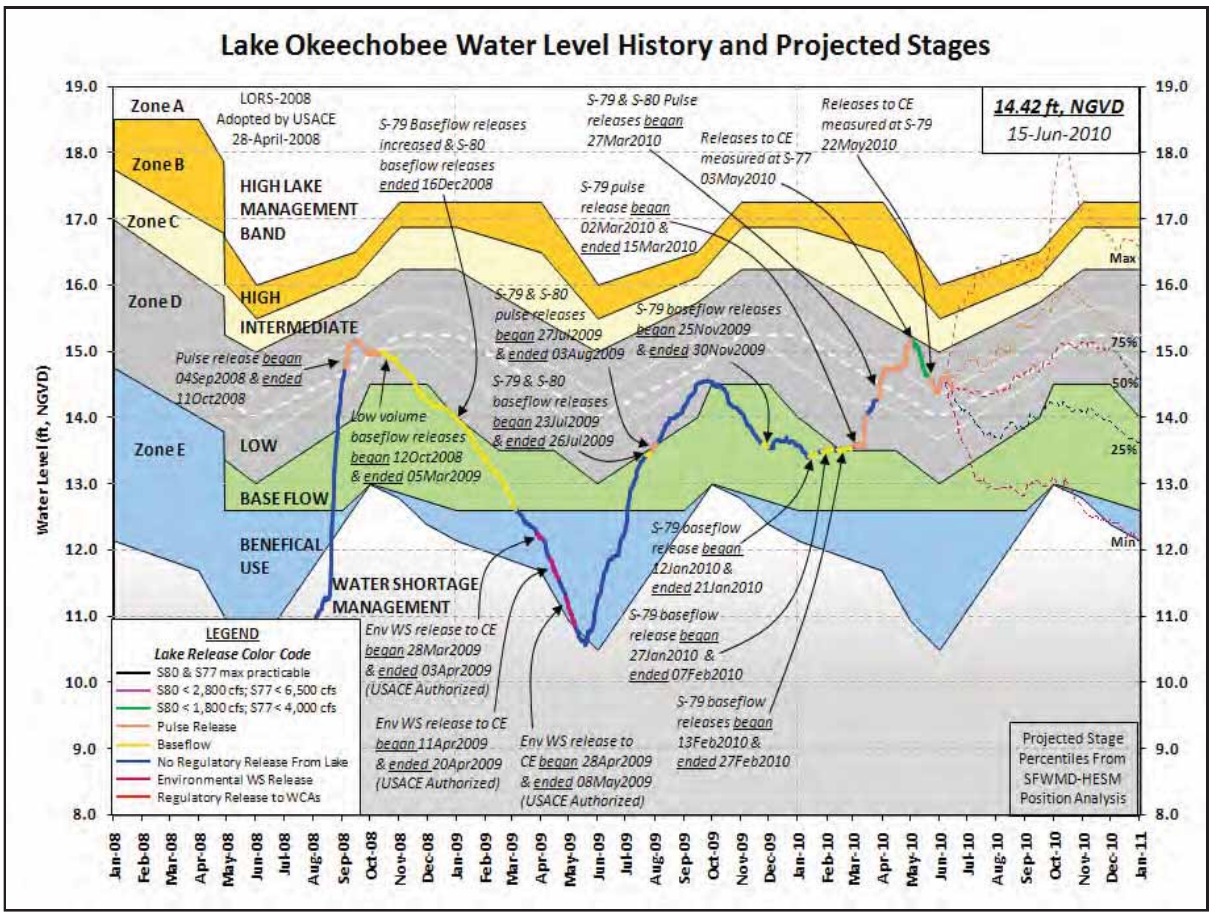


FIGURE 6-3. LAKE OKEECHOBEE 2008 – 2010 STAGE AND REGULATORY DISCHARGE HISTORY AND PROJECTION

Extreme high or low lake levels of any duration, or moderately high or low lake levels of prolonged duration greater than six months can cause significant harm to the lake's ecosystem (Havens and Gawlik, 2005), and results in loss of habitat for fish, birds and other aquatic fauna.

Higher lake stages can result in elevated nutrient concentrations and increased frequency of algal blooms, increased nearshore wave energy, transport of nutrient rich muds to the nearshore region, and drowning of shallow marshes. It is important to note that the nearshore and littoral marsh zones are where most of the beneficial ecosystem functions occur. Higher stages have been shown to result in decreased water clarity, which in turn limits the depth at which SAV can effectively establish (Havens, 2003).

Extreme low lake stage results in the desiccation of the western littoral marsh, which promotes the spread of invasive and exotic vegetation. Lake Okeechobee experienced extreme low stages between fall 2006 and 2008. Extreme low lake stages can be beneficial by encouraging brush fires that help control invasive and exotic species and permit oxidation of organic muck sediments, which will expose the underlying native seed bank and reduce the volume of organic muck material that might otherwise become resuspended.

A certain degree of natural variation in lake stage has been shown to benefit Lake Okeechobee plant and animal communities (Havens et al., 2001a; 2002; 2005; Havens, 2003). Declining water levels in late winter and early spring benefit wading birds by concentrating prey resources (Smith et al., 1995a). Water levels near 12.5 feet msl benefit SAV and emergent vegetation by providing optimal light levels in the summer months (Havens et al., 2004). Variation in the prescribed lake stage range favors development of a diverse emergent plant community (Richardson et al., 1995). A more detailed discussion of the effects lake stage has on the Lake Okeechobee ecosystem can be found in the 2007 SSR (RECOVER, 2007), which can be found at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

The beneficial high, low and varied lake stages have been defined as follows:

- Avoid extreme high water stage greater than 17 feet msl
- Avoid stages greater than 15 feet msl for more than 12 consecutive months
- Avoid extreme low water stage less than 11 feet msl
- Avoid stage less than 12 feet msl for more than 12 consecutive months
- Increase the frequency of spring recessions, which are yearly stage declines from near 15.5 feet msl in January to near 12.5 feet msl in June, with no reversal greater than 0.5 feet.
- One extreme low stage event once per decade

6.2.2 Monitoring

Daily lake stage is recorded at sites around the lake and within the lake itself; current lake stage status may be obtained from www.saj.usace.army.mil/h2o/reports/r-oke.txt. Data regarding lake stage is also maintained in the SFWMD DBHYDRO database.

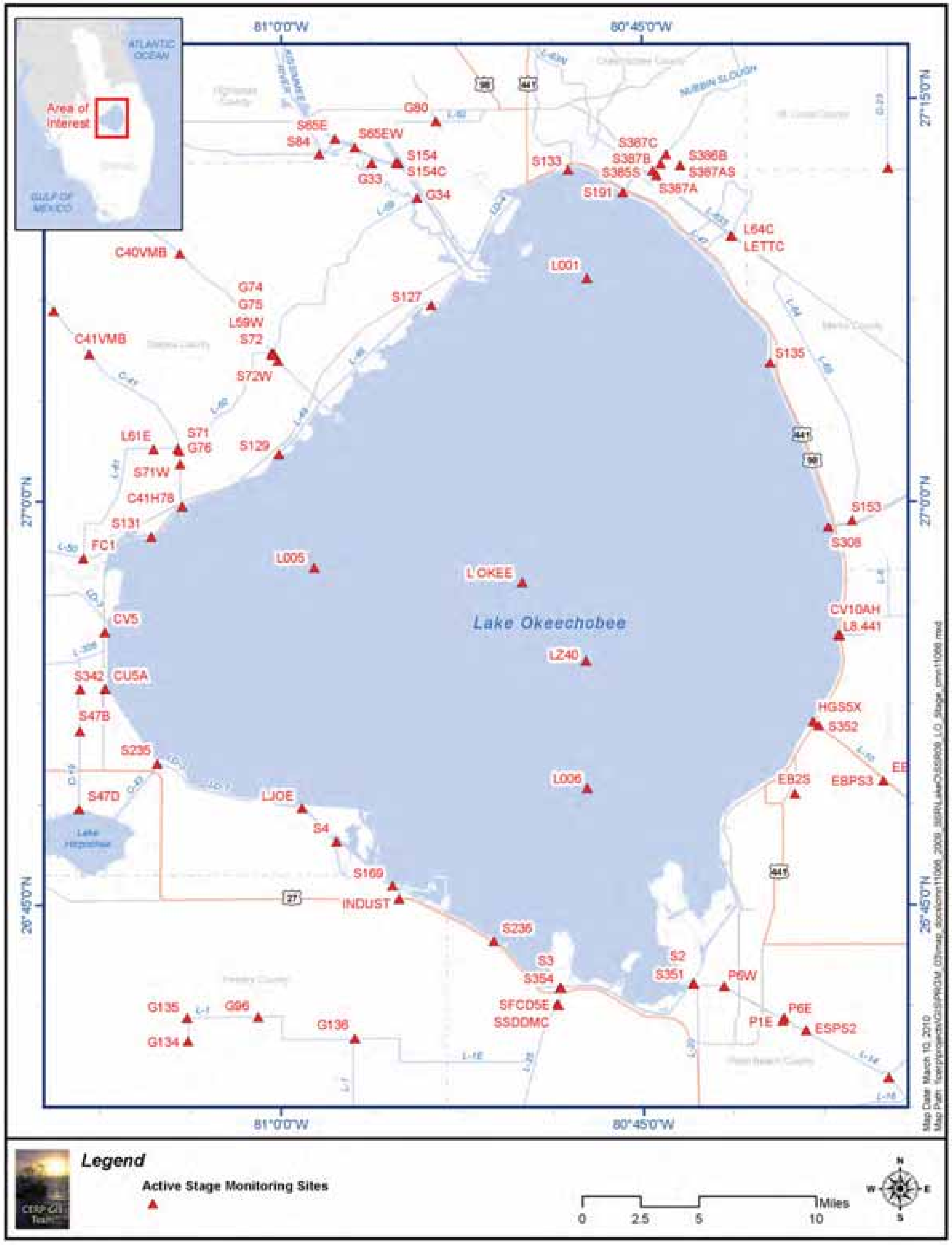


FIGURE 6-4: WATER STAGE SAMPLING LOCATIONS IN LAKE OKEECHOBEE

6.2.3 Results

Maintaining stage within the desired envelope has been hampered by lack of external storage, the need to maintain flood control in the greater lake basin, and the vagaries of south Florida rainfall. As a result, lake stage has been outside of the desired range for a significant amount of time despite efforts to maintain the appropriate stage (***Figure 6-5***).

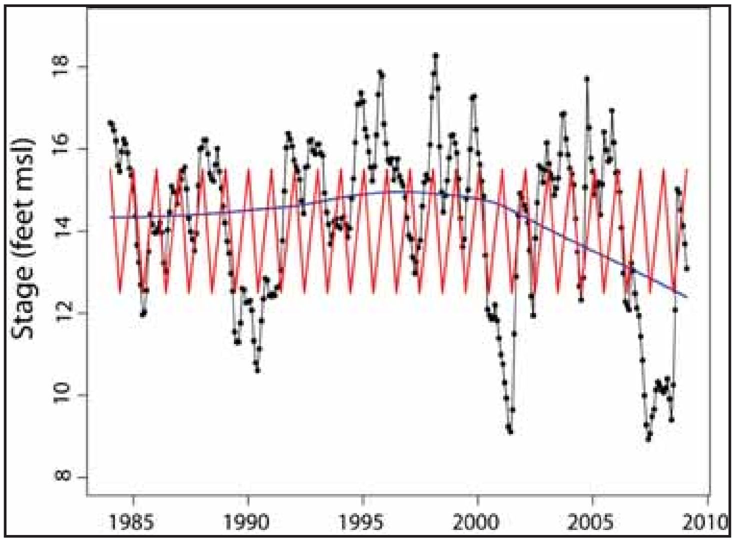


FIGURE 6-5. MEAN MONTHLY STAGE DATA

Key:

red desired recession rates from January high of 15.5 feet msl to June low of 12.5 feet msl
blue smoothed loess of mean monthly stage
black mean monthly stage in feet above mean sea-level 1984 through 2008

The overarching driver affecting lake stage has been, in the absence of alternative controls, watershedrainfall (***Figure 6-6***)), which depicts monthly stage in feet above sea level versus previous two years of rainfall for that month. The apparent decrease in the last few years in the ability to maintain lake stage above an adequate minimum level is a reflection of the perceptible decrease in watershed rainfall over this same timeframe (***Figure 6-7***). The drawdowns that occurred in 2005 and 2006 were made after high lake stages and concerns of dike integrity following hurricanes Frances, Jeanne and Wilma in 2004-2005. These drawdowns preceded the unexpected drought which resulted in unprecedented low lake stages. However, these drawdowns only accounted for approximatley 0.2 feet of additional lowered stage.

In April 2008, the USACE approved a new regulation schedule for Lake Okeechobee, referred to as LORS2008, which replaced the WSE Operating Schedule. LORS20008 was intended to be temporary schedule which focused on public health and general welfare considerations associated with the Herbert Hoover Dike. This plan can be found on the USACE web site at www.saj.usace.army.mil/Divisions/Planning/Branches/Environmental/DOCS/OnLine/Glades/LakeO/LORSS/2007/ACOE_STATEMENT_APPENDICES_A-G.pdf. LORS2008 is expected to be in effect until either the risk of dike failure is reduced with improvements to Reaches 1, 2 and 3 of the dike, or the initial CERP projects are implemented, whichever comes first.

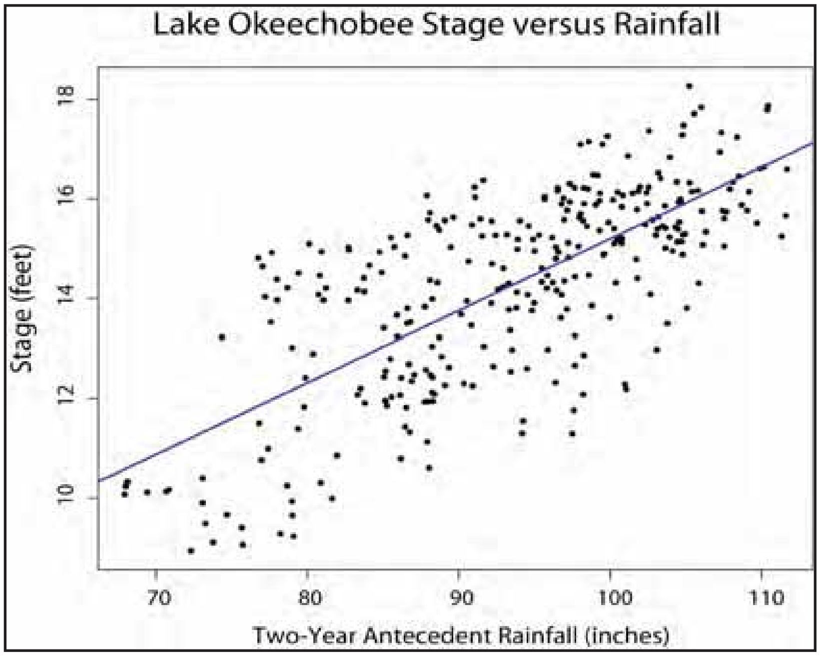


FIGURE 6-6. LAKE STAGE IS SIGNIFICANTLY CORRELATED (P<0.001) TO THE SUM OF RAINFALL FALLING ON LAKE OKEECHOBEE'S WATERSHED FOR THE PREVIOUS TWO YEARS

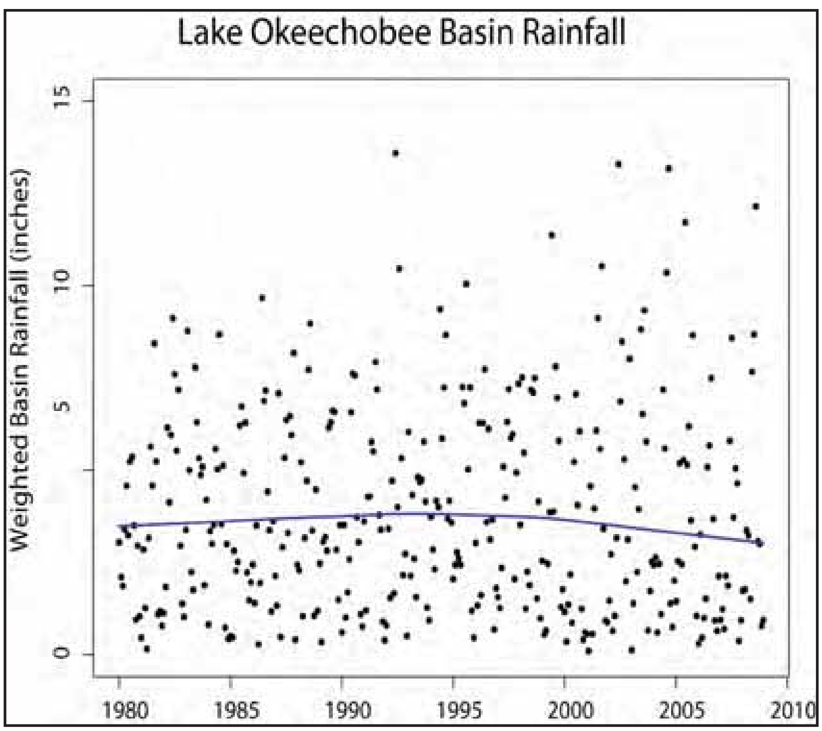


FIGURE 6-7. WEIGHTED LAKE OKEECHOBEE BASIN RAIN FALL

Note blue line denotes smoothed loess average

6.3 WATER QUALITY STRESSOR

6.3.1 Introduction and Background

Lake water quality is a stressor in all of Lake Okeechobee's hypothesis clusters (*Figure 6-8*). In addition, Lake Okeechobee is a primary source of water for Everglades and estuarine restoration and reductions in nutrient concentrations in the lake will improve water quality for all these downstream areas. Development of the lake's watershed has increased phosphorus (P) loading to the lake over the last half century, and has accelerated eutrophication (Engstrom et al., 2006). Additionally, Lake Okeechobee is designated as a Class I drinking water source by the Florida Department of Environmental Protection (FDEP). Approximately 60,000 people rely on Lake Okeechobee as their primary source of potable water. Periodic large-scale surficial blooms since the 1980s are a consequence of the lake's deteriorated water quality regime. The last of these occurred during summer 2005, and has elevated concerns about cyanotoxins and potential adverse health effects for wildlife, livestock and humans.

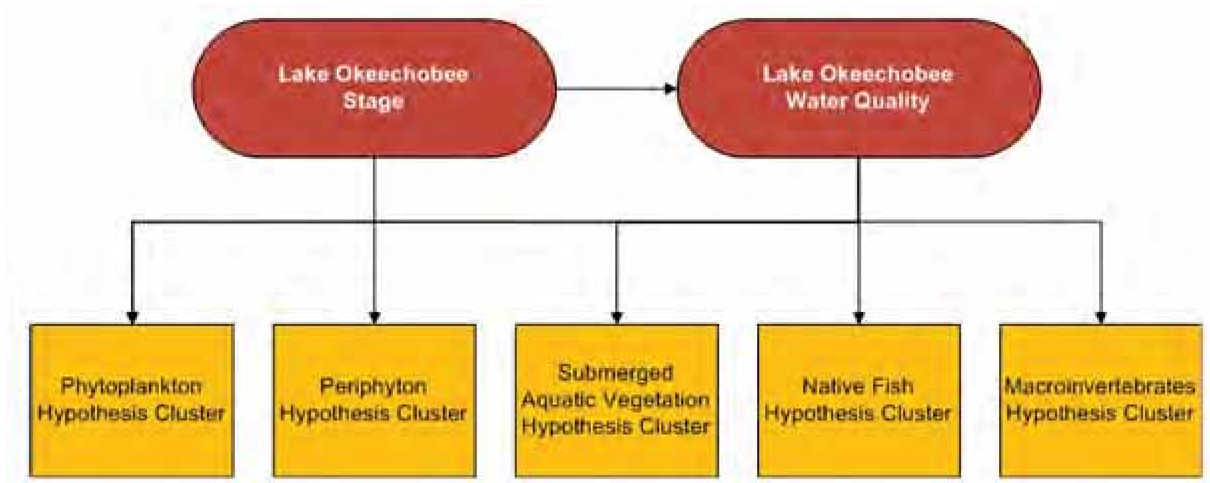


FIGURE 6-8. WATER QUALITY, WHICH IS AFFECTED BY STAGE, IS A PRIMARY STRESSOR AFFECTING ALL LAKE OKEECHOBEE HYPOTHESIS CLUSTERS

Key water quality characteristics of concern for Lake Okeechobee are P concentration, the ratio of nitrogen (N) to P, algal bloom frequency and composition, light climate, turbidity, sedimentation rates, sediment resuspension, and cycling of nutrients sequestered in the bottom sediments. The greatest concern is the increased P concentration within the lake, P has been associated with periodically large algal blooms (Jones, 1987; Havens et al., 2003), and has led to numerous efforts to reduce P loads to the lake.

Excessive P loads to Lake Okeechobee originate from agricultural and residential activities that dominate land use in the watershed. TP loads were 558 metric tons per year (mt/yr) on average from water year (WY) 2003 to WY 2008 (WY is May to April) (Zhang et al., 2009b), a loading that includes several extreme events (e.g., hurricanes). These loads are nearly four times the total maximum daily load (TMDL) of 140 mt/yr considered necessary to achieve the target in-lake P goal of 40 micrograms per liter (µg/L) (FDEP, 2001; Havens and Walker, 2002).

Reducing watershed nutrient loads to Lake Okeechobee is only part of the process leading to a less eutrophic system. As a result of excessive nutrient loading, primarily over the past 60 years (Brezonik and Engstrom, 1998), over 30,000 mt of P is sequestered in Lake Okeechobee's sediment (Reddy et al., 1995). Since Lake Okeechobee is shallow, with an average depth of 2.7 meters, and a large fetch, up to 50 kilometers, wind generated waves can easily resuspend these sediments (Jin and Ji, 2004) and, through equilibrium processes, release P back into the water column. The release of P into the water column through sediment resuspension is of particular concern during hurricanes when massive disturbance of the shallow sediment can result in large spikes in post-hurricane water column P concentration.

Sediment P assimilative capacity appears to be diminishing, thus contributing to increases in P concentration in the water column despite an overall reduction in external P loading since the 1980s (Havens and James, 2005). Clearly, understanding the role that sediments play in the lake's P cycle is paramount in developing restorative measures to reduce in-lake P concentrations. Three alternatives were considered in a sediment management study: dredging, chemical treatment and achieving the lake P-loading goal (Blasland Bouck and Lee Inc., 2003a). Dredging may be cost prohibitive and would take several decades to complete. Chemical treatment of the sediments to prevent P release has never been attempted on a scale that would be needed to treat the central mud zone sediments in Lake Okeechobee. A 32-day laboratory bench test study using chemical compounds to prevent P-release from pelagic mud sediments suggested that ferric chloride at 50 milligrams per liter (mg/L) and alum at 30 to 40 mg/L were the most effective in reducing P release from the sediments (Golder Associates Inc., 2008). TP and soluble reactive P (SRP) release was reduced roughly 50 percent or greater at these doses both in cores where the sediment was periodically resuspended and in those in which the sediments were undisturbed. TP and SRP concentrations were typically between 20 and 100 mg/L depending on the treatment (undisturbed or resuspended) and day. A larger-scale evaluation of these two compounds may be conducted in the future. Expectations from other studies are that if external P loads are reduced, nutrient reduction and ecological recovery will occur over time (Sas, 1989; Jeppesen et al., 2005). The current estimate for positive effects after meeting the watershed loading goal is a few decades (Blasland Bouck and Lee Inc., 2003b).

The presence of easily resuspended mud sediments on the bottom of the central area of Lake Okeechobee presents additional water quality concerns. Resuspended sediments reduce light transmission through the water column. High lake levels enhance the transport of suspended sediments to nearshore regions of Lake Okeechobee, reducing light availability for SAV growth (James and Havens, 2005), which affects animals that use these plant communities as a food source or for habitat (Havens et al., 2005). Wind waves created by hurricanes in 2004 and 2005 resuspended large amounts of mud sediments into the water column and transported these mud sediments throughout the pelagic and nearshore regions. Consequently, turbidity and P concentrations have remained above pre-2004 levels and have negatively affected SAV (James et al., 2008) and fish (Zhang et al., 2009a).

A more detailed discussion of Lake Okeechobee water quality can be found in the 2007 SSR (RECOVER, 2007), which is available on line at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx. Thorough discussions of water quality issues during WY 2005 through WY 2009 surrounding Lake Okeechobee can be found in Chapter 10 of the 2009 South Florida Environmental Report (SFER) (Zhang et al., 2009b), which is available online at my.sfwmd.gov/sfer).

Water quality goals for Lake Okeechobee are as follows:

- Decrease P inputs as part of restoration implementation and basin control efforts
- Pelagic TP concentrations should not exceed 40 µg/L
- Increase the TN to TP ratio to above 22:1
- Chlorophyll *a* concentrations should not exceed 40 µg/L (indicative of algal blooms) in no more than five percent of all samples collected in the pelagic region
- Ratio of dissolved inorganic nitrogen (DIN) to SRP should be greater than 10:1.
- Water transparency clear enough to see a Secchi disk on the lake bottom from May to September in nearshore areas

Two performance measures and two interim goals have been developed for Lake Okeechobee water quality. The performance measures are for water quality mosaic and diatom to cyanobacteria (www.evergladesplan.org/pm/recover/perf_low.aspx). The interim goals are for P and algal blooms (www.evergladesplan.org/pm/recover/igit_subteam.aspx).

6.3.2 Monitoring

A number of studies and long-term monitoring efforts are underway to examine processes occurring in Lake Okeechobee and its watershed. Details of these efforts are reported in the 2009 SFER (Zhang et al., 2009b). Water quality data for Lake Okeechobee are available on the SFWMD's DBHYDRO database. Lake Okeechobee water quality was evaluated with data from the pelagic water quality monitoring stations (***Figure 6-9***) for the period from 1988 to 2008 and from the nearshore monitoring stations for the period of 2004 to 2008.

Correlations among water quality parameters were determined using the nonparametric Spearman rank method using the last ten years of data (1999 to 2008). Data over the last five years (2004 to 2008) were averaged and compared to specific numeric goals as defined in the Lake Okeechobee Protection Plan (SFWMD et al., 2007). These included TP concentration, TN to TP ratio, SRP to DIN ratio, percent algal blooms and TP loads to the lake. Water quality trends for the 1981 to 2007 period were developed by James et al. (2009) using seasonal Kendall's Tau analyses.

Additional water quality data from the nearshore stations (***Figure 6-9***) were averaged for the five-year period 2004 to 2008 and compared to nearshore goals. These included nearshore TP and the percent of samples where Secchi disk transparency was equal to water depth (i.e. the water was clear enough to see a Secchi disk on the bottom of the lake) from May to October. While no TN to TP ratio goals have been set for the nearshore region, this team compared the observed nearshore ratio to the pelagic goals.

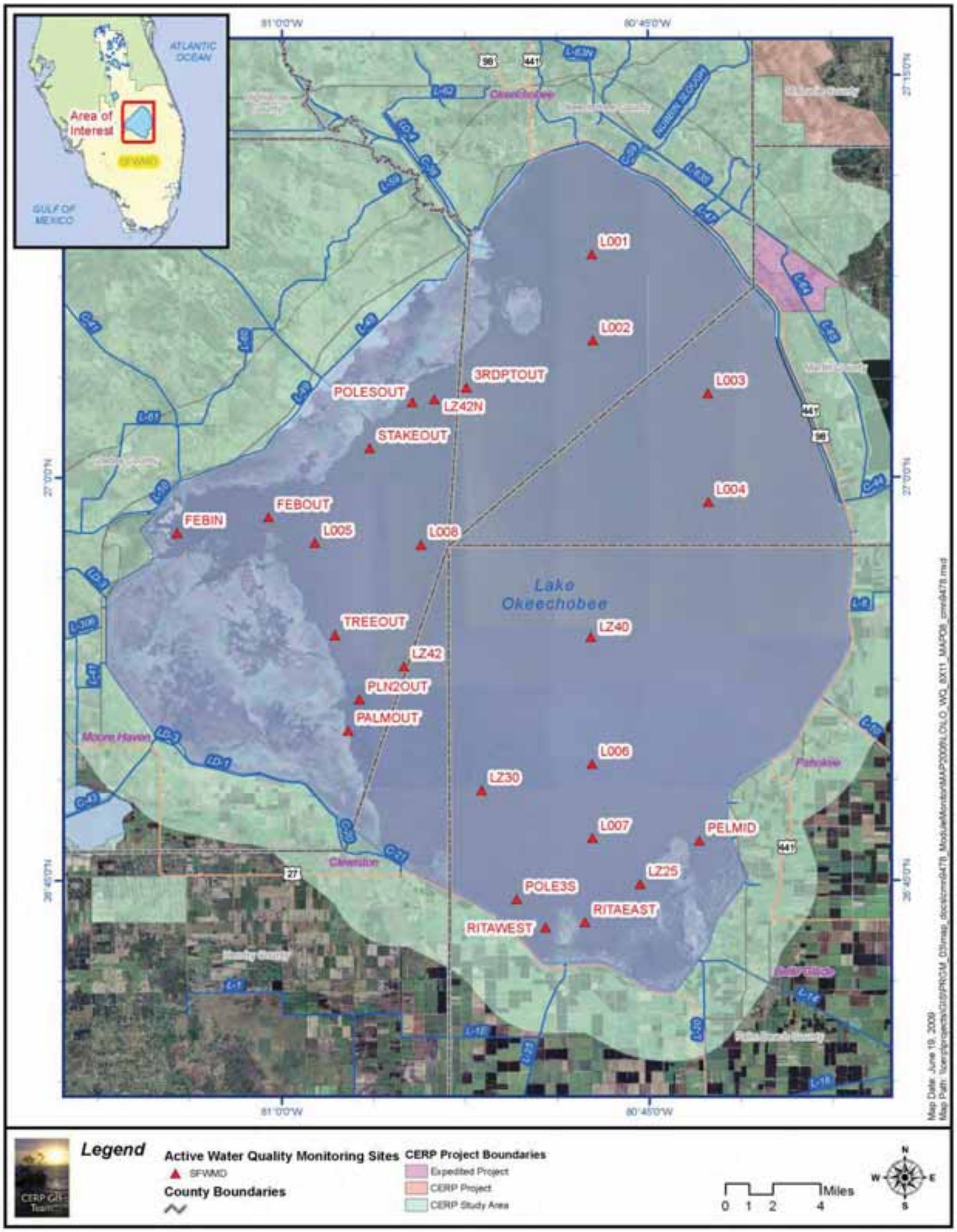


FIGURE 6-9. WATER QUALITY SAMPLING LOCATIONS IN LAKE OKEECHOBEE

6.3.3 Results

The pelagic stations indicate long-term trends in the offshore region of Lake Okeechobee. Sediment water interactions are major factors driving nutrient dynamics in this region. Five-year averages (2004-2008) were compared against quantitative restoration goals. Of the restoration goals for the pelagic region, only the algal bloom (chlorophyll *a*) criteria have been met. The observed reduction in algal bloom frequency is attributed to decreased water clarity caused by disturbances from the 2004 and 2005 hurricanes. Water quality goals for the nearshore region include water transparency clear enough to see a Secchi disk on the lake bottom from May to September and TP less than or equal to 40 µg/L. Neither has been achieved in the past five years. While no TN to TP and DIN to SRP ratio goals have been established for the nearshore region, the observed values were 15.8:1 and 4.6:1, respectively. Trends determined for the offshore region of Lake Okeechobee 1981 to 2007 indicated TP, SRP and turbidity increased significantly, while chlorophyll *a* and TN had no significant change. Because nutrients are in excess, algae in this region of Lake Okeechobee have been mostly light limited. While implementation of restoration projects should improve water quality attributes through reduced water column N, P, TSS and chlorophyll *a* concentrations, the detection of these improvements will require a long-term data set because system responses take time. Restoration success in Lake Okeechobee and its watershed would significantly enhance the restoration success of the Everglades and the coastal estuaries in terms of improved quantity, quality and timing of water leaving Lake Okeechobee.

Phosphorus loads in WY 2007 and WY 2008 were 202 and 246 mt, respectively (***Figure 6-10***). These loads were lower than the five year rolling average because of reduced flows due to drought conditions. In WY 2009, drought conditions eased and P loads rose to 657 mt. There is no detectable trend in P load entering the lake – albeit any changes in load pattern that might be present may be being obscured by the large year-to-year climatic variability. However, there is a significant upward trend (Kendall tau, p=0.009) in P loads being discharged and a corresponding downward trend (Kendall tau, p=0.032) in the fraction of the annual P load being retained in the lake (i.e., "net", or loading into lake minus load out). These trends reflect the reduced assimilative capacity of Lake Okeechobee (Havens and James 1997, 2005). If excessive loads into the lake continue, it is likely that loads out will correspondingly increase. Alternatively, if future loading rates decrease due to the complementary success of various efforts to accomplish exactly that end, the internal recycling from P built up in the sediments may delay the corresponding reduction in loading exiting the lake (Sas 1989; Jeppesen et al. 2005) .

The total inflow to Lake Okeechobee was greater in WY 2008 than in WY 2007 (1,012,785 acre-feet versus 575,283 acre-feet), but inflow P concentration was lower in WY2008 (***Figure 6-11***). The five-year average load (WY 2004 to WY 2008) was 588 mt. This load includes 35 mt from atmospheric deposition as specified by the FDEP (2001). The five-year average load is about four times greater than the goal of 140 mt/yr (***Table 6-1***). Note the decrease in inflow concentration from 1982 through 1997 and increases in some years thereafter. Internal concentrations in 2005 to 2006 were for the first time observed higher than inflow concentrations. The decreased loadings in 2007 and 2008 reflect the prolonged drought that occurred during that timeframe. Detailed nutrient and flow data from the Lake Okeechobee watershed can be found in the annual SFER updates.

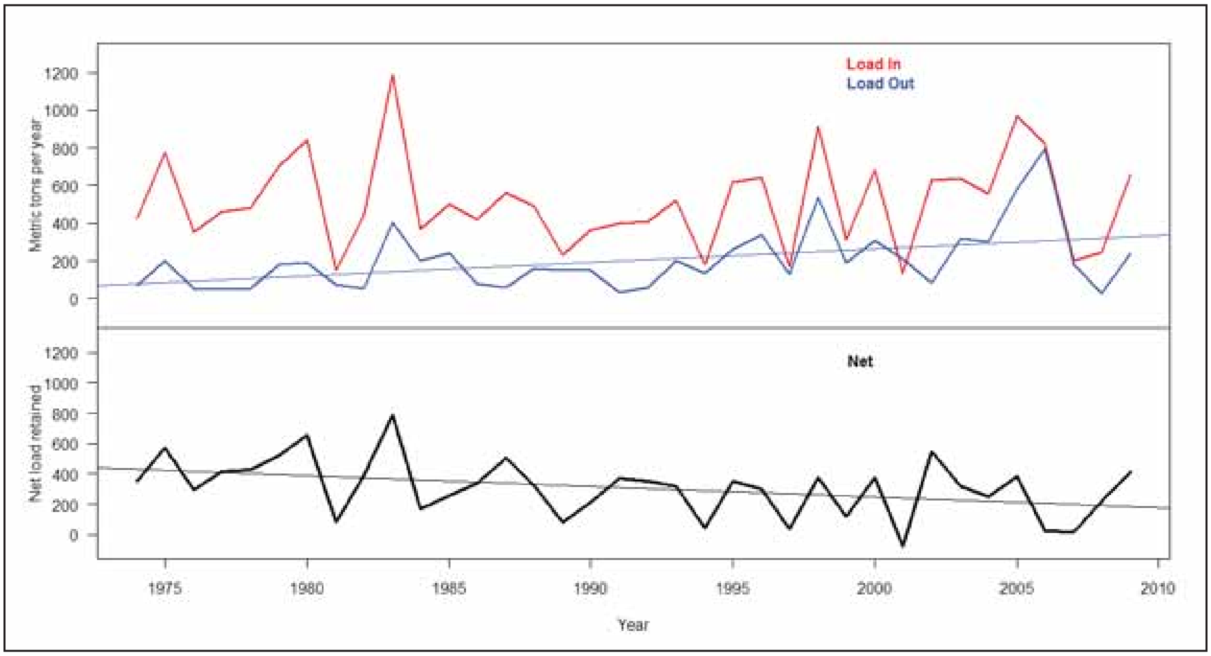
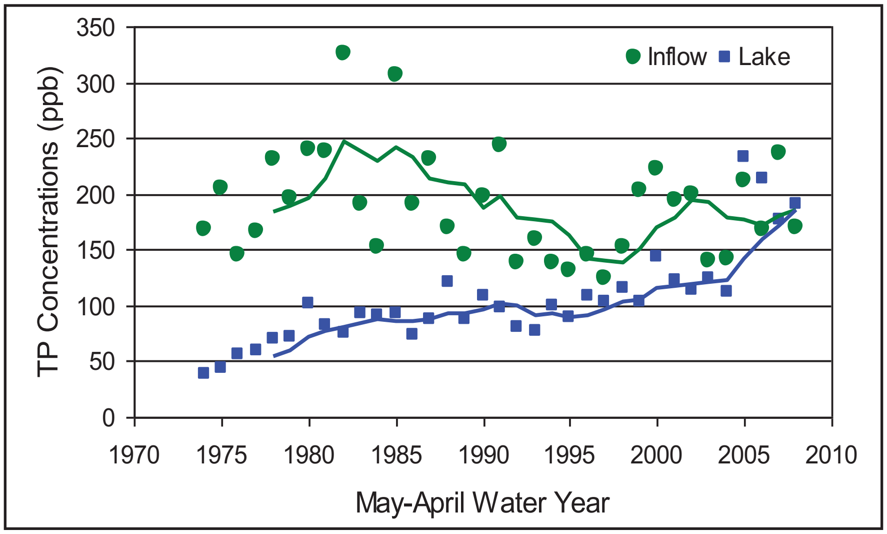


FIGURE 6-10. PHOSPHORUS LOADS INTO AND OUT OF LAKE OKEECHOBEE, AND NET DIFFERENCES BY WATER YEAR



Zhang et al. 2009b

FIGURE 6-11. INFLOW AND IN-LAKE TOTAL PHOSPHORUS CONCENTRATIONS

Note Lines denote five year moving averages
green inflow
blue total phosphorus concentrations (ppb)

TABLE 6-1. GOALS AND FIVE-YEAR AVERAGES FOR WATER QUALITY PARAMETERS

Parameter	Goal	Five-Year Average (WY2005-WY2008)
TP load	140 mt/yr (to be met by 2015)	558 mt/yr
Pelagic TP	40 µg/L	193 µg/L
Pelagic TN	not applicable	1.72 parts per million (ppm)
Pelagic SRP	not applicable	61 µg/L
Pelagic DIN	not applicable	313 µg/L
Pelagic TN to TP	> 22:1	10:1
Pelagic DIN to SRP	> 10:1	5.4:1
Algal bloom frequency	< 5 percent of pelagic chlorophyll *a* exceeding 40 µg/L	4.57%
Water clarity	Secchi disk visible on lake bottom at all nearshore SAV sampling locations from May to September	19.38%
Nearshore TP	below 40 µg/L	129 µg/L
Nearshore TN to TP	not applicable	15.8:1
Nearshore SRP to DIN	not applicable	4.6:1

Among the canals and streams that discharge into Lake Okeechobee, the largest source of surface water inflow is from the Kissimmee River. The combined Upper Kissimmee sub-watershed (above structure S-65), the Lower Kissimmee basins and the Lake Istokpoga sub-watershed comprise about 53 percent of the Lake's watershed, and account for approximately 50 percent of the surface water flows entering Lake Okeechobee. As measured at the lower S-65E structure, the Kissimmee River contributes the majority of P loading to Lake Okeechobee (approximately 30 percent of the total), followed by the Taylor Creek/Nubbin Slough and the C-41 drainage basins. The load from the Kissimmee/Istokpoga basin alone has, in most five-year timeframes since 1994 (***Figure 6-12***), accounted for loads in excess of the regulatory TMDL target of 140 mt/yr for the entire Lake. Although the Kissimmee River Restoration project is an effort separate from Everglades restoration, the two are nevertheless interdependent. The Kissimmee River restoration aims to re-establish hydrologic conditions similar to pre-channelization. Historical flow and volume characteristics will be achieved through increased storage in the headwater lakes. These changes coupled with expanded area and quality of lake littoral zone and river riparian area will result in P load reduction to the lake. Amelioriation of P loads to Lake Okeechobee will ultimately result in trickle-down benefits to the STAs, the Everglades, and South Florida's estuaries, in the form of reduced nutrients, algal bloom frequency and turbidity.

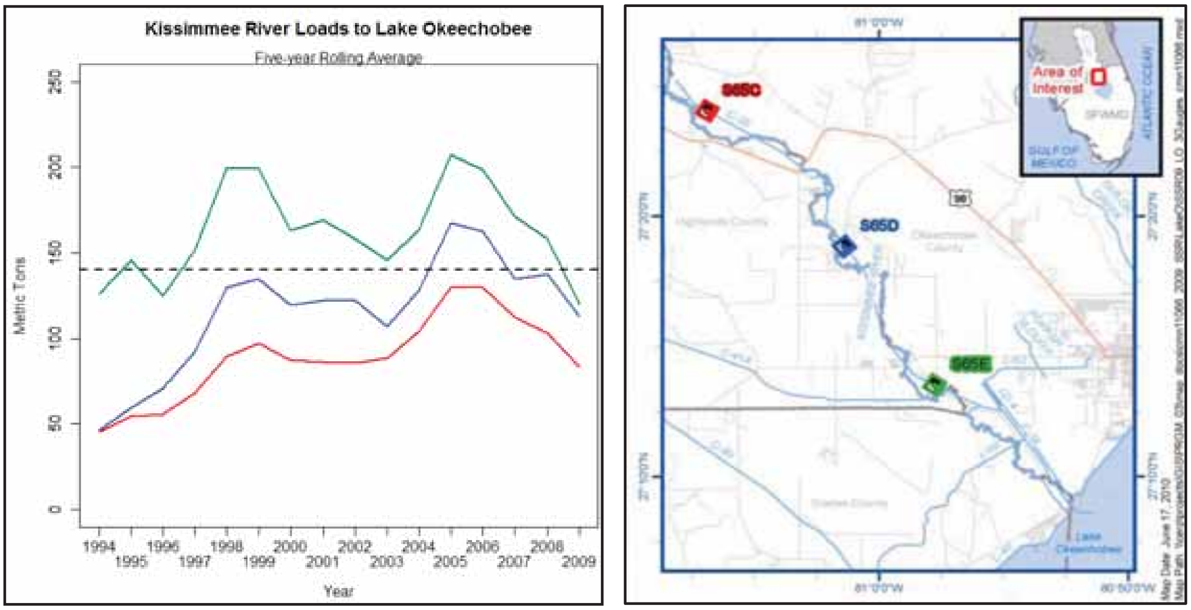


FIGURE 6-12. FIVE-YEAR ROLLING AVERAGE P LOAD AS MEASURED AT THE S-65C, S-65D, AND S-65E KISSIMMEE RIVER STRUCTURES

Note: 1. Loads thru S65E (green) are indicative of kissimmee river contribution of loads entering the lake
2. Dashed line is tmdl

The hurricanes in 2004 and 2005 resulted in an increase of TSS in the water column (***Figure 6-13a***). The values are much higher than before the hurricanes, despite samples being taken weeks after the storms. The amount of suspended sediment in the water column during the storms was most likely orders of magnitude greater than that measured in the weeks after the storms (Jin et al., 2009). Turbidity, which is very closely related to TSS in Lake Okeechobee, has increased significantly over time (***Figure 6-13b***; ***Table 6-2***).

The flocculent mud sediments in this large shallow lake are easily resuspended as demonstrated by spikes in TSS concentration in non-hurricane years (i.e., 1990, 1994, 2000 and 2001; ***Figure 6-13a***). The resuspension occurs primarily from wind-induced waves and secondarily from currents (Jin and Ji, 2004).

The average percent (19.4) of time that nearshore Secchi disk visibility extended to the lake bottom during the May to September period over the last five years has been low (***Table 6-1)***. Variability in this measurement among years is high and related to water levels, with low water years having greater number of measurements with visibility to the lake bottom than high water years (***Figure 6-13e***). Caution should be taken in interpreting these data since the number of observations in the low lake years are small (***Figure 6-13f***).

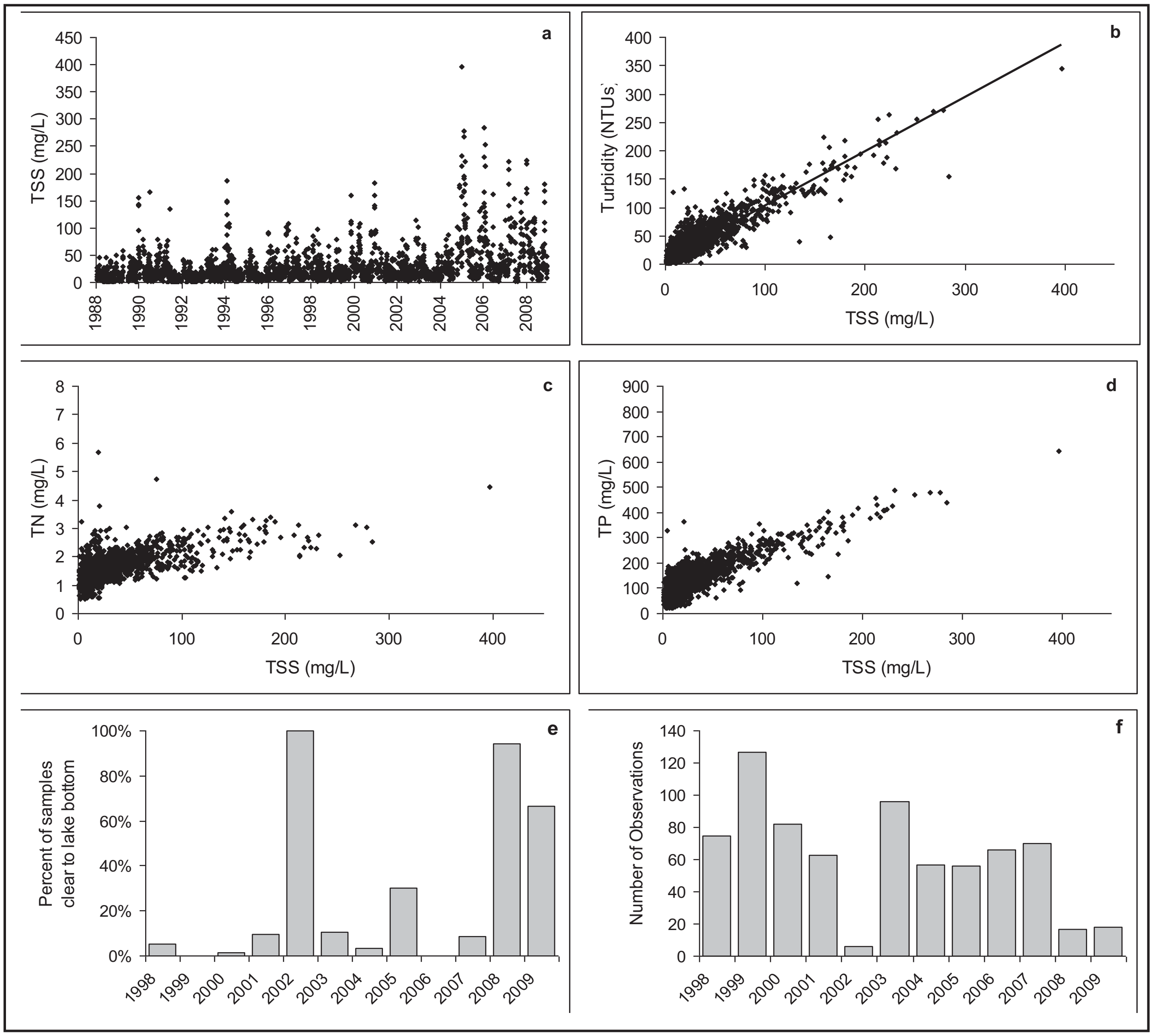


FIGURE 6-13. PELAGIC TOTAL SUSPENDED SOLIDS CONCENTRATION JANUARY 1988 TO DECEMBER 2008

Key:
a) Sampling data
b) TSS versus turbidity
c) TN
d) TP
e) Percent nearshore observations May to September where Secchi disk is visible on lake bed
f) Number of nearshore observations of water transparency

TABLE 6-2. SPEARMAN RANK CORRELATION COEFFICIENTS (TOP VALUE), PROBABILITIES (MIDDLE), AND NUMBER OF COMPARISONS (BOTTOM) FOR SELECTED WATER QUALITY PARAMETERS FROM LAKE OKEECHOBEE FOR 1998 TO 2008

	TP	TN	TSS	Chlorophyll *a*	Turbidity	SRP
TN	0.742					
	<.0001					
	1634					
TSS	0.824	0.743				
	<.0001	<.0001				
	1645	1633				
Chlorophyll *a*	-0.243	-0.026	0.011			
	<.0001	0.291	0.652			
	1645	1633	1648			
Turbidity	0.925	0.742	0.890	-0.229		
	<.0001	<.0001	<.0001	<.0001		
	1646	1634	1649	1649		
SRP	0.757	0.387	0.388	-0.527	0.622	
	<.0001	<.0001	<.0001	<.0001	<.0001	
	1635	1634	1638	1638	1639	
DIN	0.721	0.646	0.478	-0.499	0.697	0.727
	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001
	1634	1634	1637	1637	1638	1638

TSS are closely correlated to turbidity, TN and TP (***Figure 6-13b***, ***c*** and ***d***; ***Table 6-2***). The correlations to TP and TN correspond with resuspended sediments that contain large amounts of P and N. The increases in resuspended sediments also were associated with increases in SRP ($P<0.005$, data not shown). The effect of the 2004 and 2005 hurricanes on P is observed in the marked increased concentration seen in both of those years. P has slowly declined since 2004 (***Figure 6-14b***). However, concentrations had not declined to pre-2004 values by the end of the measurement period. This is attributed to the much more easily resuspended sediments that maintained high P levels in the water column after the 2004 hurricanes (James et al., 2008). A seasonal pattern of higher TP concentrations in the winter than summer (***Figure 6-14***c) is a product of higher wind velocities that occur in winter and spring. The significant upward trend in TP concentration is at a rate of 2 to 3 µg/L per year (***Figure 6-15***; ***Table 6-3***; James et al., 2009). The current five-year average of TP is 193 µg/L or nearly five times the goal of 40 µg/L (***Table 6-1***). The nearshore five-year average of 123 µg/L is three times the target goal. The P trophic state index (TSI), computed using the formula specified in the Florida Administrative Code(F.A.C.) rule 62-302, indicates increasing eutrophic conditions(***Figure 6-14d***).

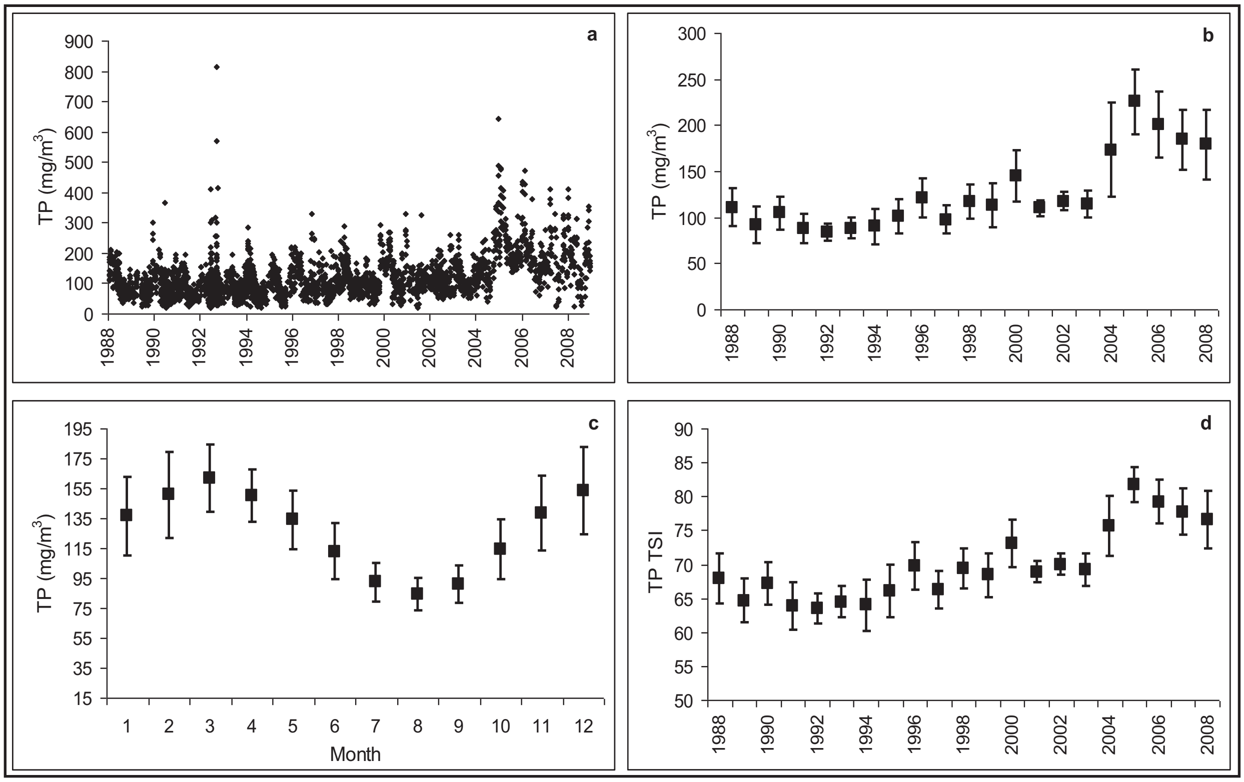


FIGURE 6-14. PELAGIC TOTAL PHOSPHORUS IN MICROGRAMS PER CUBIC METER CONCENTRATION FOR JANUARY 1988 TO DECEMBER 2008

Key: a) Individual sampling data; means and 95 percent confidence intervals by
b) year
c) month
d) annual average TP TSI

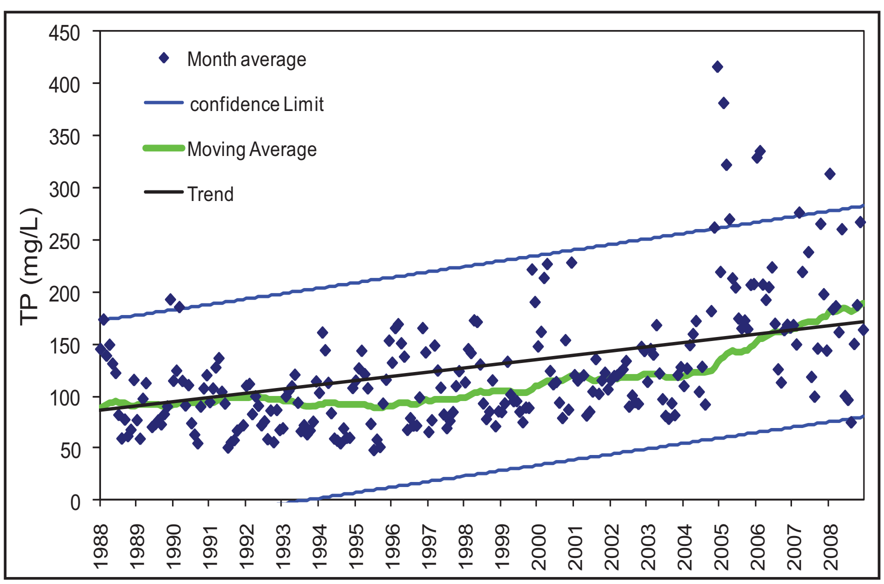


FIGURE 6-15. MONTHLY MEAN TOTAL PHOSPHORUS TIMELINE, FIVE-YEAR MOVING AVERAGE, AND LINEAR TREND OVER TIME INCREASING AT THE RATE OF 0.005 PARTS PER MILLION PER YEAR FROM 1988 TO 2008 (P < 0.001)

TABLE 6-3. SEASONAL KENDALL'S TAU TREND ANALYSIS OF MONTHLY AVERAGED DATA FOR LAKE OKEECHOBEE FROM 1981 TO 2007

Statistic	Tau	Slope per Year	P
Chloride	-0.5555	-1.8074	< 0.0001
TP	0.4321	0.0026	< 0.0001
SRP	0.3848	0.0011	< 0.0001
TN	-0.1207	-0.0062	0.1197
DIN	0.2137	0.0026	0.0133
Turbidity	0.3703	0.8465	0.0001
Secchi transparency	-0.4253	-0.0113	< 0.0001
Chlorophyll *a*	-0.0904	-0.1402	0.2724

The pattern of SRP concentration over time (***Figure 6-16a***) mimics the TP timeline. Both the SRP concentration as well as the SRP to TP ratio (***Figure 6-16***c) exhibit significant ($P < 0.001$) increasing trends. The increasing trend in the SRP to TP ratio indicates that not only is more P present in the water, but more of it is in the more bioavailable form. As with TP concentration, SRP and SRP to TP ratios peaked after the 2004 hurricanes and have declined slowly since, but remain above pre-2004 levels.

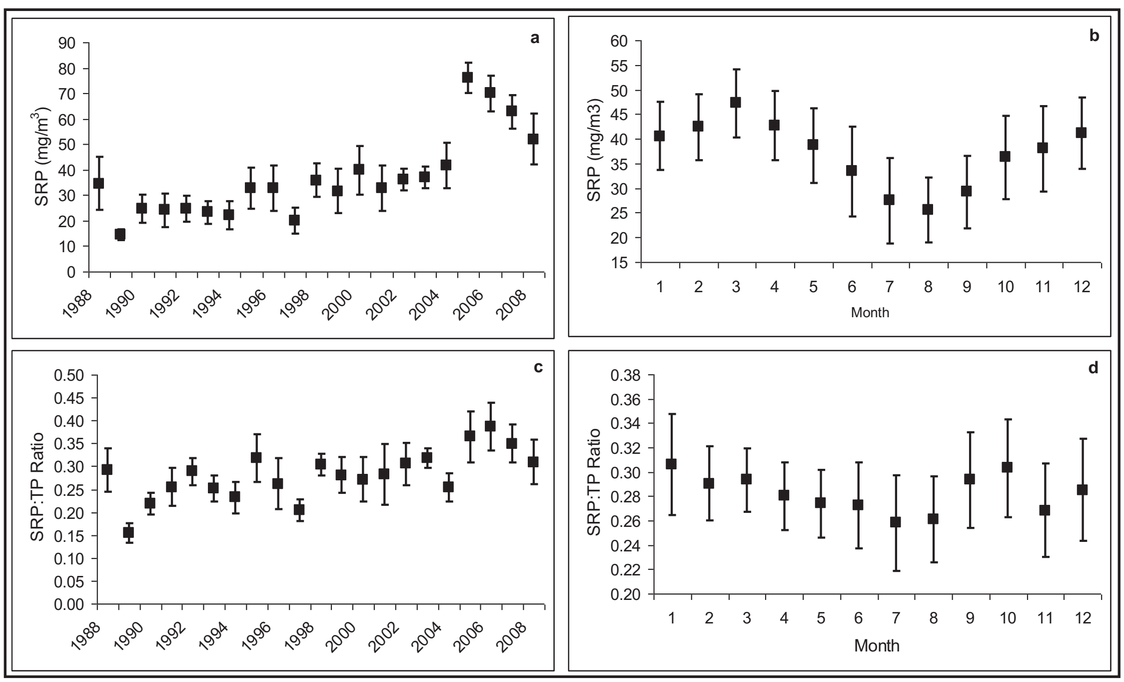


FIGURE 6-16. PELAGIC SOLUABLE REACTIVE PHOSPHORUSMEAN CONCENTRATION FOR JANUARY 1988 TO DECEMBER 2008 MEANS AND 95 PERCENT CONFIDENCE INTERVALS

Key: by
a) year and
b) month; and ratio of SRP to TP by
c) year and
d) month

TN concentrations are correlated with TP concentrations ($P < 0.005$), despite the lack of a significant temporal trend in TN (***Table 6-3***), which suggests that similar seasonal and annual factors that affect P also affect N. TN was also correlated with TSS (***Figure 6-13c***; ***Table 6-2***).

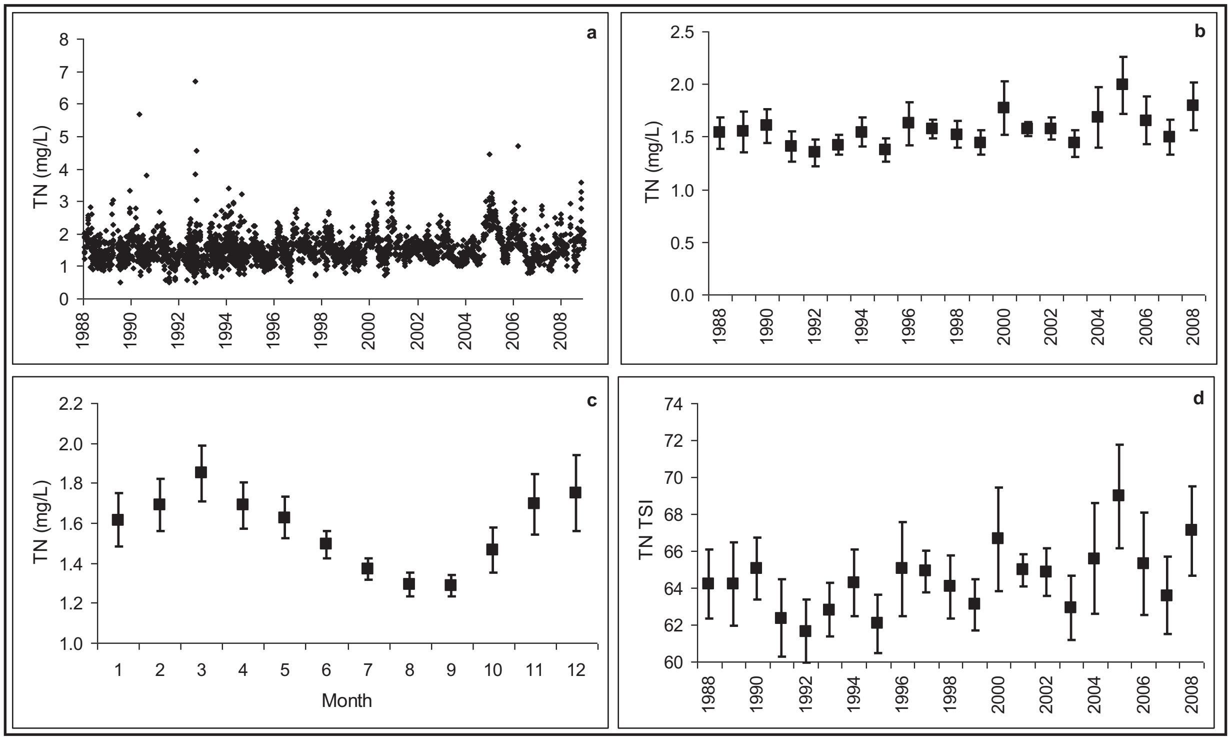


FIGURE 6-17. PELAGIC TOTAL NITROGEN CONCENTRATION FOR JANUARY 1988 TO DECEMBER 2008

Key: a) sampling data; means and 95 percent confidence intervals by
b) year and
c) month; and
d) TN TSI by year

Annual average chlorophyll *a* reached the lowest values in 2005 and 2006 (***Figure 6-18a*** and ***b***). However, the long-term trend is not significant (***Table 6-3)***. The lower values in 2005 and 2006 are attributed to increased light limitation (James et al., 2008). Thus, while Lake Okeechobee met the less than five percent algal bloom frequency restoration target during the past five years, this achievement was a consequence of poor light conditions rather than declining nutrient levels. The seasonal pattern of greater chlorophyll during the summer is related to higher water temperatures and greater light penetration due to less wind-induced mixing during the summer months compared to the cooler, windy winter months (***Figure 6-18c***). The chlorophyll TSI was calculated as referenced in the F.A.C. (***Figure 6-18d***). Values for the lake index above 60 units denote eutrophic conditions. All but three of the years evaluated exceeded this threshold.

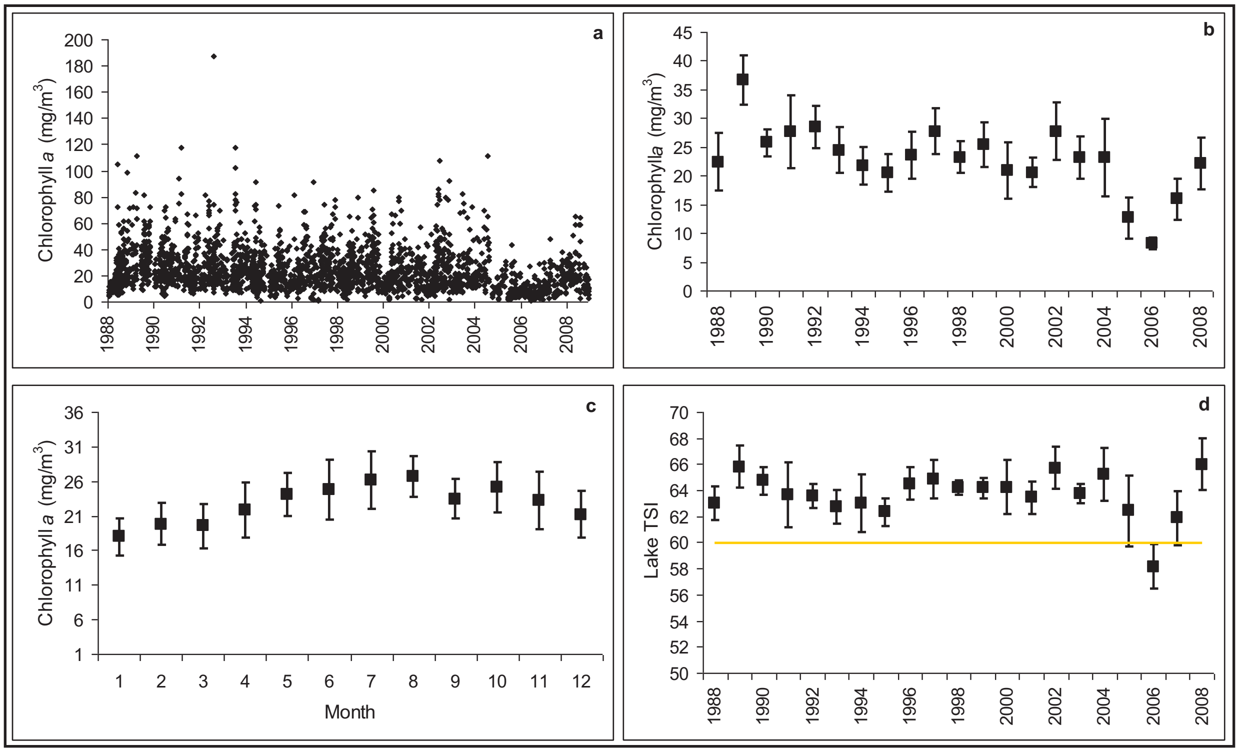


FIGURE 6-18. PELAGIC CHLOROPHYLL *A* CONCENTRATION FOR JANUARY 1988 TO DECEMBER 2008

Key: a) sampling data; means and 95 percent confidence intervals by
b) year and
c) month; and
d) Chl TSI by year (above orange line indicates eutrophic conditions)

6.3.4 Summary

Numerous control efforts are planned or underway in the Lake Okeechobee watershed to capture a percentage of P that would otherwise enter the lake (SFWMD et al., 2007). However, watershed P loads are merely part of the issue, because a large reservoir of P is sequestered in Lake Okeechobee's sediments (Reddy et al., 1995). The decreasing trend in assimilative capacity of Lake Okeechobee suggests that the sediments are losing their capacity to bind P (Havens and James, 2005). When watershed restoration is completed and inflow P concentrations and loads to Lake Okeechobee decline, it is expected to take several decades before in-lake concentrations decline to the in-lake goal of 40 µg/L because of the reduced sediment P assimilation and excess P content in sediments.

Of the four restoration goals for the pelagic region, only the algal bloom criteria have been met. The observed reduction in algal bloom frequency is attributed to decreased water clarity caused by disturbances from the 2004 and 2005 hurricanes. Water quality goals for the nearshore region have not been achieved in the past five years. While no TN to TP and DIN to SRP ratio goals have been established for the nearshore region, the observed values were 15.8:1 and 4.6:1, respectively. Trends determined for the pelagic region of Lake Okeechobee for 1981 to 2007 indicated TP, SRP and turbidity increased significantly, while chlorophyll *a* and TN did not change significantly. Because nutrients are in excess, algae in this region of the lake have been mostly light limited.

In addition to management of water levels, ecological improvements in Lake Okeechobee are dependent on a reduction in nutrient loads and the eventual stabilization of lake sediments through natural processes (e.g., compaction). A significant portion of water quality variability is attributed to sediment water interactions. Long-term increases of P are specifically related to excessive loads to Lake Okeechobee. In-lake responses to incremental reductions in watershed P loads will be undetectable against the backdrop of the sediment water interactions. Even if internal loading does delay full recovery of Lake Okeechobee, observations of shallow eutrophic systems elsewhere around the world demonstrate measurable improvement in water column P and chlorophyll *a* concentrations and increased Secchi disc transparency in as little as ten to 15 years (Sas, 1989; Jeppesen et al., 2005). These improved water quality conditions in terms of reduced nutrient, TSS and chlorophyll *a* concentrations, coupled with maintenance of appropriate water levels as a result of restoration should result in benefits in the nearshore and littoral regions of Lake Okeechobee, where most ecological functions and societal values of the lake reside (Havens et al., 2007). However, since internal loading likely will delay the onset of environmental benefits from improved management of Lake Okeechobee and its basin detecting these changes will require a commitment to water quality and ecological monitoring extending to 2050 and perhaps beyond.

6.4 PHYTOPLANKTON HYPOTHESIS CLUSTER

6.4.1 Introduction and Background

Phytoplankton monitoring is an important component of Lake Okeechobee research for two reasons. First, the lake is designated as a Class I drinking water source by the FDEP. An algal bloom is defined as chlorophyll *a* concentrations greater than or equal to 40 ppb. Second, phytoplankton is one of the primary producers at the base of the pelagic and nearshore food webs and as such may be an important food source for higher trophic-level grazers such as macroinvertebrates and fish.

Data collected as part of a long-term monitoring program indicate that the Lake Okeechobee phytoplankton community has been shifting from one dominated by diatoms in the 1970s to being dominated by cyanobacteria since the 1990s (Havens et al., 1996). Cyanobacteria dominance of the phytoplankton assemblage in Lake Okeechobee is considered to be undesirable since these taxa are not readily grazed by zooplankton, suggesting that energy transfer from phytoplankton up the food web to the higher trophic levels may be reduced. When less energy is transferred up the food web, reduced yields of desirable fish may result (Smith et al., 1995b). A measurement of carbon flow by Work et al. (2005) suggested that a significant fraction of the energy source in Lake Okeechobee was from bacteria. Continued excessive nutrient loading from the watershed, fluctuating climactic events ranging between excessively dry and wet years, and the passage of three hurricanes during 2004 and 2005 may be factors that are influencing changes in the phytoplankton community (***Figure 6-19***). The long-term data set suggests a shift from cyanobacterial dominance between 1994 and 2003, to one dominated by diatoms thereafter. These data are useful in establishing pre-restoration conditions and suggest that over the past decade the phytoplankton community has been dynamic. These data will be useful in assessing if restoration will contribute to a return to a more heterogeneous phytoplankton assemblage that was thought to occur prior to anthropogenic disturbance; one that is dominated or co-dominated by diatoms rather than cyanobacteria. A more detailed discussion of phytoplankton in Lake Okeechobee can be found in the 2007 SSR (RECOVER, 2007a) at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

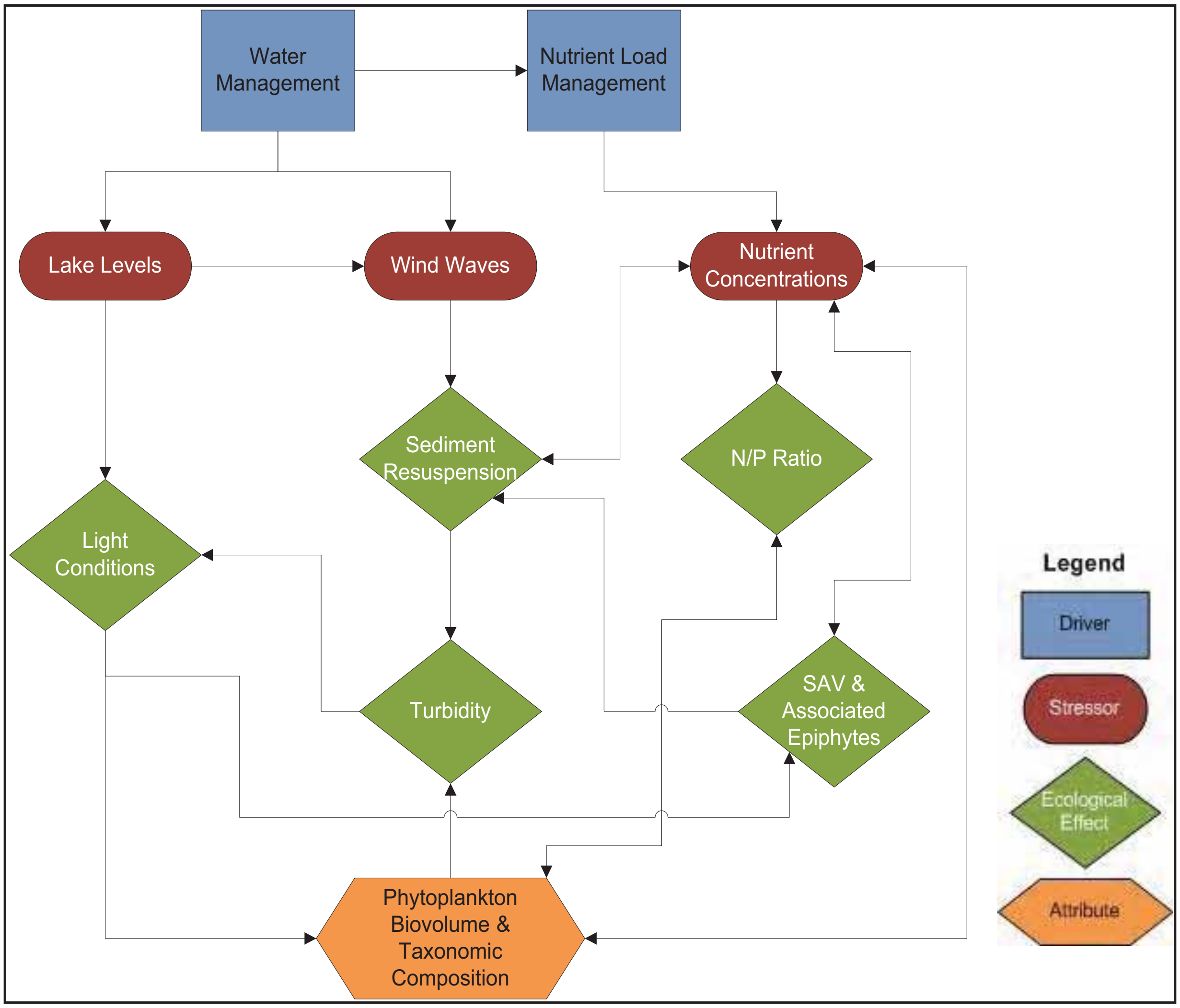


FIGURE 6-19. CONCEPTUAL MODEL FOR PHYTOPLANKTON IN LAKE OKEECHOBEE

Bioassay experiments have been conducted since 1994 on the factors limiting phytoplankton production in Lake Okeechobee. Resource-limitation assays are used to assess trends in the importance of nutrient and light availability for algal blooms in the lake zones most prone to planktonic algal blooms. Integrated samples of the water column were collected quarterly from four stations representing the four ecological zones found in Lake Okeechobee. These bioassays show that from 1997 to 2008 algal growth in Lake Okeechobee is generally light limited but may be periodically limited by N: 55 percent light limitation and 40 percent N or N+P limitation with five percent showing no significant difference.

The RECOVER program has developed a performance measure for the diatom to cyanobacteria ratio, which can be found online at www.evergladesplan.org/pm/recover/recover_docs/et/lo_pm_cyano-diatom.pdf. An interim goal has been developed for algal blooms and can be found online at www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_2-3_lakeoalgalblooms.pdf .

The restoration goals for phytoplankton are as follows:

- Alter bloom composition with cyanobacteria comprising less than 50 percent
- Decrease cyanobacterial bloom frequency
- Reduce cyanobacteria dominance and increase diatoms such that the diatom to cyanobacteria ratio becomes greater than 1.5:1 (data collected since 2003 indicates that the target formulation should be revisited)

6.4.2 Monitoring

Phytoplankton monitoring was initiated in 1994 and is currently conducted on a quarterly basis at the sites shown on ***Figure 6-20*** (East and Sharfstein, 2006). Biomass as chlorophyll *a*, and community taxonomic composition are determined. Laboratory bioassays are conducted on samples to determine whether light or nutrients are potentially limiting phytoplankton growth. Photosynthesis irradiance curves are generated to evaluate how photosynthetic characteristics varied among sites located in ecologically distinct regions of the lake (Phlips et al., 1993a; Maki et al., 2004). Diatom to cyanobacteria ratios are calculated from the percent total biovolumes. Cyanotoxin concentrations were also determined. The monitoring and analysis is discussed in more detail in the 2007 SSR (RECOVER 2007), which can be found online at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

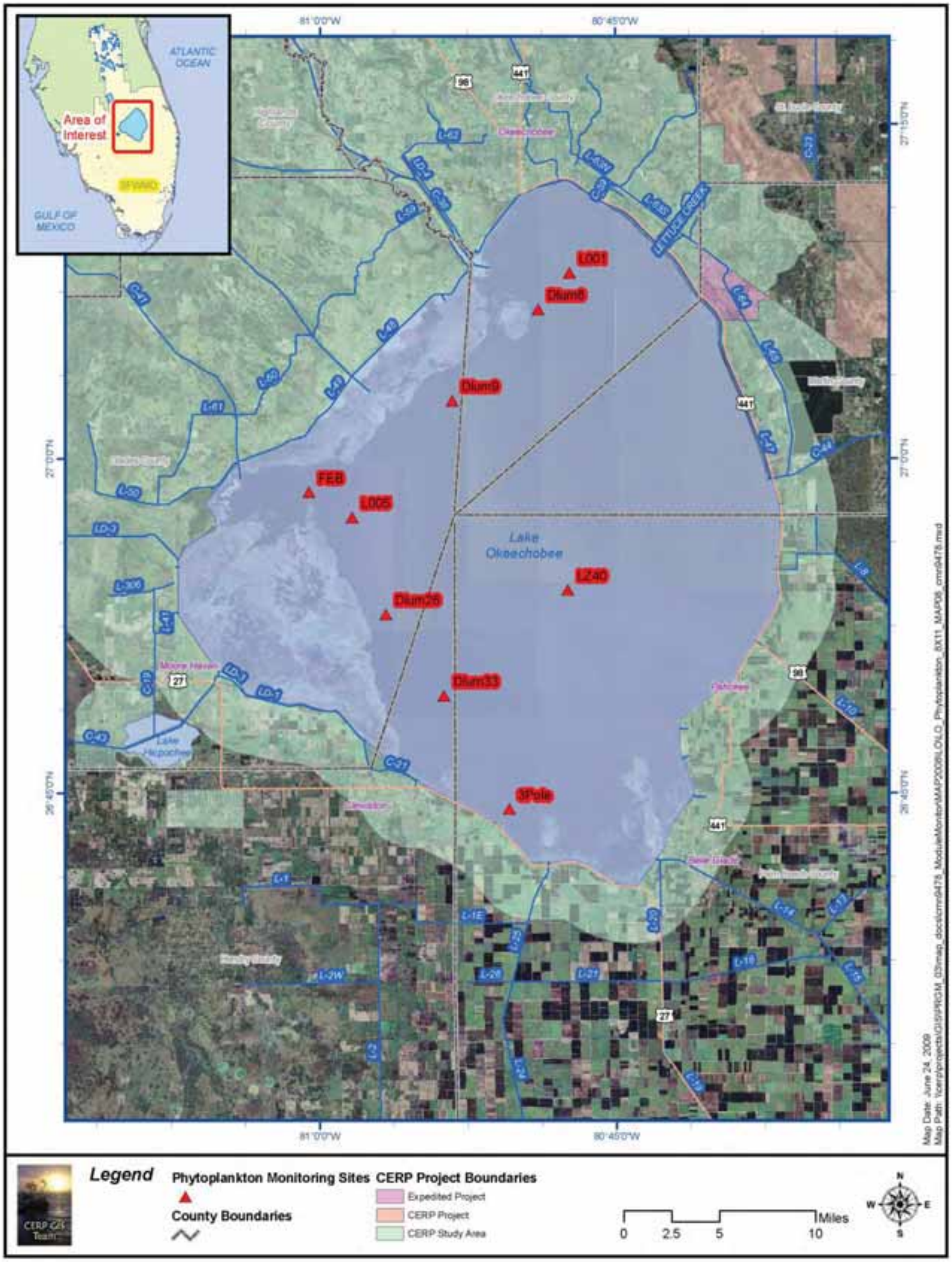


FIGURE 6-20. PHYTOPLANKTON MONITORING SITES IN LAKE OKEECHOBEE

6.4.3 Results

6.4.3.1 Community Composition

Non-metric multi-dimensional scaling ordination analysis of the community data suggests that year and then season were the two most significant factors. The clearest separation (R=0.85, p=0.001) occurred among years and then among seasons (***Figure 6-21***). The group patterns suggest that there was very clear separation (R>0.80) between most years with the differences becoming more pronounced as the interval between years increased. Some of the most significant (R>0.9, p=0.001) among-years differences occurred between 2000, 2001 and each of the subsequent years. During 2001, a lake recession and prolonged drought occurred and lake stage decreased by May of that year to roughly 1.7 meters below the long-term seasonal average, which was, at that time, the lowest lake stage ever recorded, which has since been exceeded in June 2007. Conversely, the phytoplankton communities were not substantially different between 2004 and 2005 (R=0.29, p=0.003), and 2005 and 2006 (R=0.37, p=0.001) (***Figure 6-21***). These years were marked by the passage of hurricanes, lake stage greater than 16 feet above msl, and prolonged high turbidity. As lake stage decreased during the second half of 2006 and remained less than 12 feet for most of 2007 and 2008, differences among the phytoplankton communities became more pronounced. Moderate separation (R=0.54, p=0.001) among the phytoplankton communities was observed between 2006 and 2007 and separation was relatively clear (R=0.77, p=0.001) between 2006 and 2008. Among all but one seasonal classification, the phytoplankton communities were moderately separated (R-value ranges 0.47 to 0.63) (***Figure 6-21***). The assemblages were most different between fall-spring and fall-winter whereas these assemblages were least different among the winter and spring seasons (R=0.25, p=0.001).

The within-year phytoplankton communities had mean similarity percentages that ranged from a high of 59 percent in 1994 and 29 percent in 2005. The higher similarity percentages tended to be earlier in the data period, though they began to increase between 2006 and 2008. These values represent mean taxon contribution to the community structural similarity among samples for each year and suggest that within-year variability was greatest during the period from 2002 to 2006.

In general, cyanobacteria taxa comprised three or four of the top five taxa that contributed most to the greatest among-years relative dissimilarities. Diatoms were less important, although they did comprise one or two taxa that made significant contributions to the among-years dissimilarity values. In these cases, it was diatom taxa that were found primarily in 2001 to 2008. Among the 2000 to 2008 group comparisons, three diatom and one or two cyanobacteria taxa generally contributed most significantly to the among-years dissimilarity values. From 2003 to 2008, diatom taxa contributed most to the among-years dissimilarity values and diatoms contributed most to the within-year similarity values from 2004 to 2008. These results suggest that the phytoplankton assemblage experienced an increase in diatom importance and variability after 2000, while cyanobacteria became less important. Diatoms also became the dominant algal division from 2004 to 2008, while most of the cyanobacteria taxa contributed to a very small proportion of the community similarity and among-communities dissimilarity values. However, some of the among-years community variability also may have been due in part to variability in sample identifications (i.e. taxonomic drift).

Separation among the community on a seasonal basis was less though still somewhat clear (R<0.47, p=0.001, ***Figure 6-21***). The largest separation among phytoplankton communities on a seasonal basis was between winter and fall (R=0.62, p=0.001). The smallest separation, which was marginal (R=0.25, p=0.001), was between the winter and spring phytoplankton communities. Diatom taxa contributed most to the winter and spring community similarity values, while cyanobacteria and a mix of cyanobacteria and diatom taxa contributed most to the summer and fall similarity values, respectively. Between seasons, the highest dissimilarity values were contributed by differences in diatom taxa abundances (fall-winter) and cyanobacteria and diatom taxa (summer-winter). It should be noted that the stress value associated with both the two-dimensional among-years and seasons and the among-years and lake stages plots was sufficiently high to caution their use for anything beyond examination of general trends per guidelines presented in Clarke and Warwick (2001).

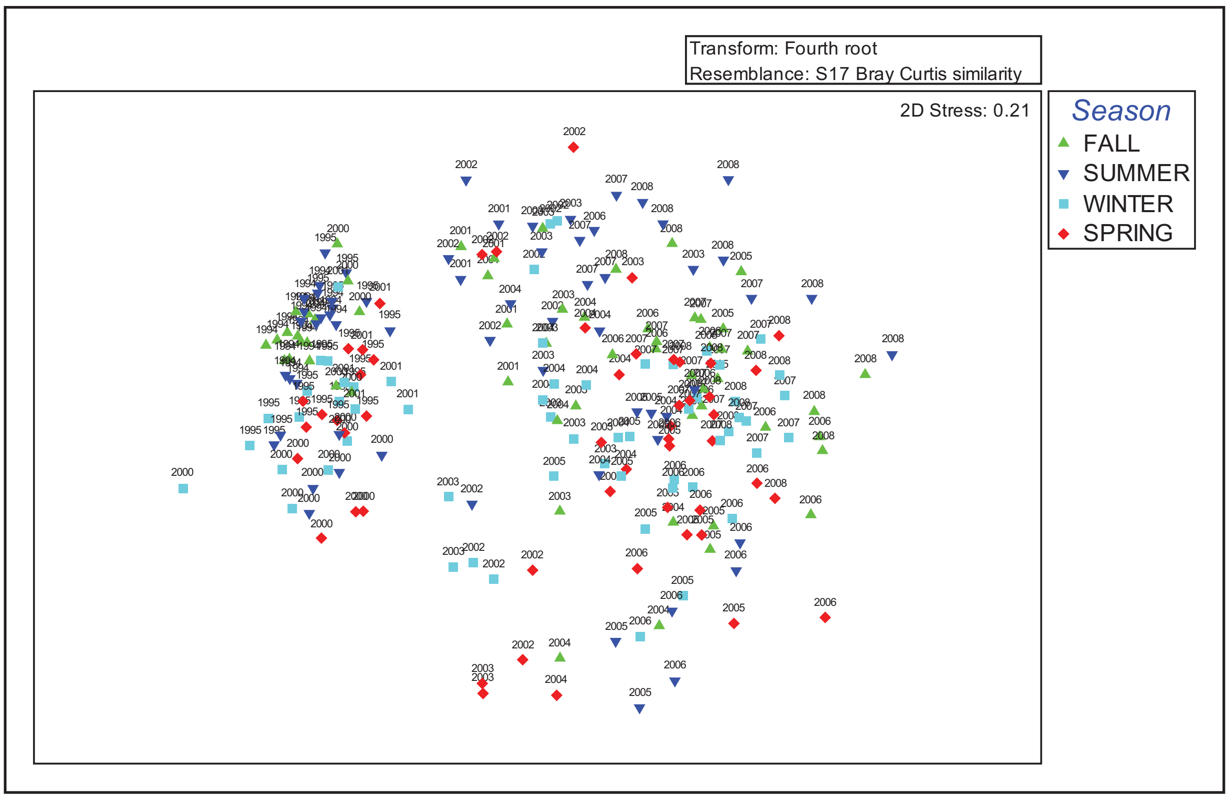


FIGURE 6-21. PHYTOPLANKTON COMMUNITY ORDINATION PLOT BY YEAR AND SEASON

Little difference was found in the phytoplankton communities when classifying the lake stage as either "high" (>15.5 feet msl), "medium" (12.5 to 15.5 feet msl) or "low" (<12.5 feet msl) (R=0.08), although marginal differences were observed (R=0.31, p<0.001) among these stage classifications when year was the second factor (***Figure 6-22***). The greatest separation was observed between medium and low lake stage classifications (R=0.42, p=0.001), suggesting larger changes in the phytoplankton community occurred as the lake varied between medium and low stages, relative to community changes between high and medium, and high and low lake stages. Very little difference occurred among sites (R=0.10, p=0.003), whether examined on an among-years or among-seasons basis. The largest separation was between the communities at 3POLE (near the northwestern tip of Ritta Island in the southern nearshore region) and LZ40 (in the center of the lake), but the amount of separation (R=0.31, p=0.001) was marginal. These comparisons suggest that temporal factors were more important in influencing community structure than variability in either lake stage or geographic location. Since photosynthetic behavior was shown to be homogenous among sites during higher lake stages and heterogeneous under lower lake stages (Maki et al., 2004), it is perhaps surprising that larger differences in the phytoplankton communities were not observed under different lake stages. The marginal separation observed in the phytoplankton community under lake stage classifications may reflect the decreased representation of two of the nearshore sites during periods of low lake stage as sampling was not conducted because these sites were inaccessible.

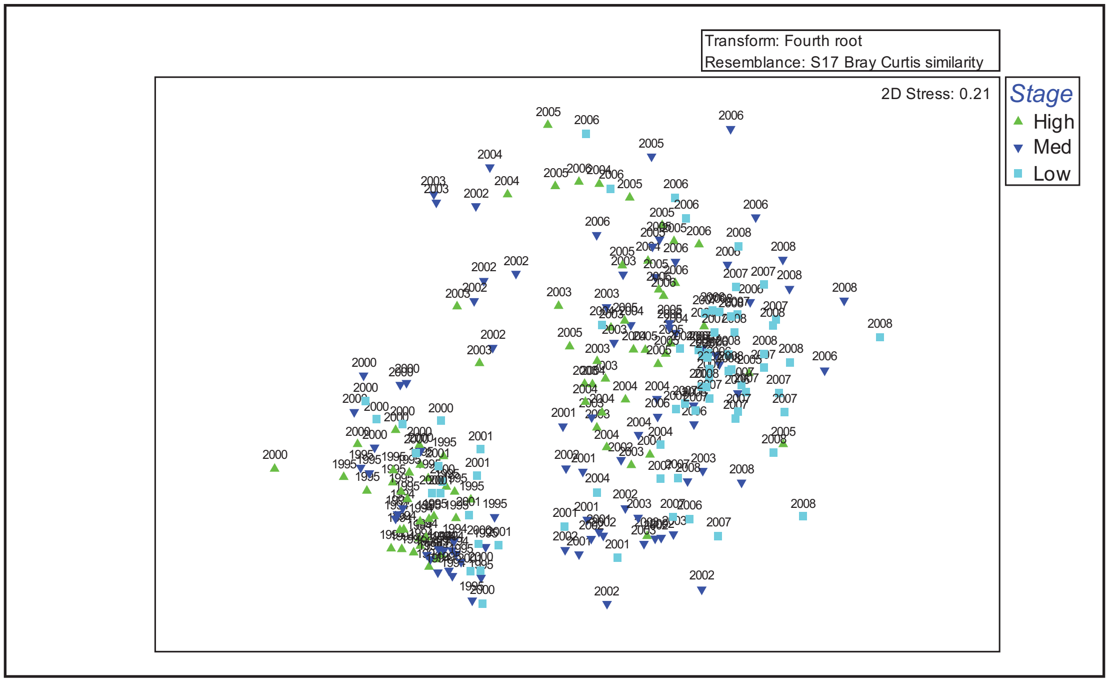


FIGURE 6-22. PHYTOPLANKTON COMMUNITY ORDINATION PLOT BY YEAR AND LAKE STAGE

Stepwise addition of water quality variables suggested a positive but weak relationship (Spearman $\rho=0.29$, P=0.001) between a combination of specific conductance, the ratio of DIN to SRP, ammonia, SRP, TP, and mean daily lake stage and wind speed and the phytoplankton community composition. Similarly weak positive correlations between combinations of subsets of these variables also were observed.

6.4.3.2 Biomass Determination

Biomass defined as mean annual total biovolumes were variable among the nearshore and pelagic sites, and appears to be similar among both regions for most years (***Figure 6-23***). The mean annual pelagic biovolume appears to be significantly lower in 2006 relative to the other years and may be related to extremely low light levels in the water column after the passage of the hurricanes in 2004 and 2005. Mean annual biovolumes varied between 48,000 cubic micrometers per milliliter (μm^3/mL) in 2006 (pelagic sites) to 1,900,000 μm^3/mL in 2001 (nearshore sites).

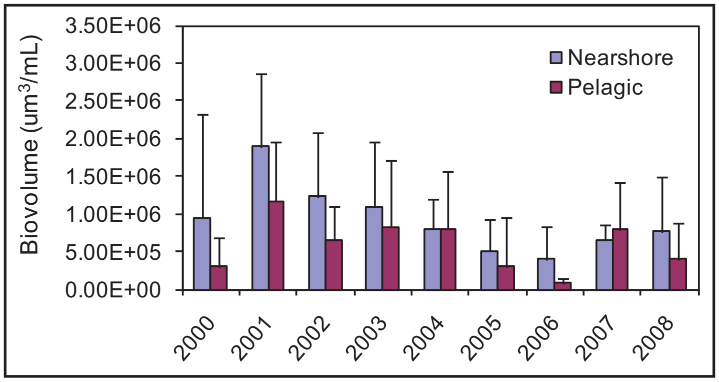


FIGURE 6-23. ANNUAL MEAN PHYTOPLANKTON BIOMASS IN TOTAL BIOVOLUMES ± 1 STANDARD DEVIATION AT BOTH NEARSHORE AND PELAGIC SITES

Key: SD standard deviation

Biomass as mean annual chlorophyll *a* concentrations was less variable and very similar among site types for all years (***Figure 6-24***). Mean annual chlorophyll *a* concentrations were generally between 10 and 20 μg/L. Algal bloom frequency, as previously defined (Havens et al., 1995) was infrequent during this period. Blooms were generally observed on average once a year (from quarterly samples) at either one of the nearshore or pelagic sites. A large-scale surficial bloom has not been observed since August 2005. In the case of that particular bloom, it occurred between the summer and fall quarterly sampling events and was therefore not captured as part of these data.

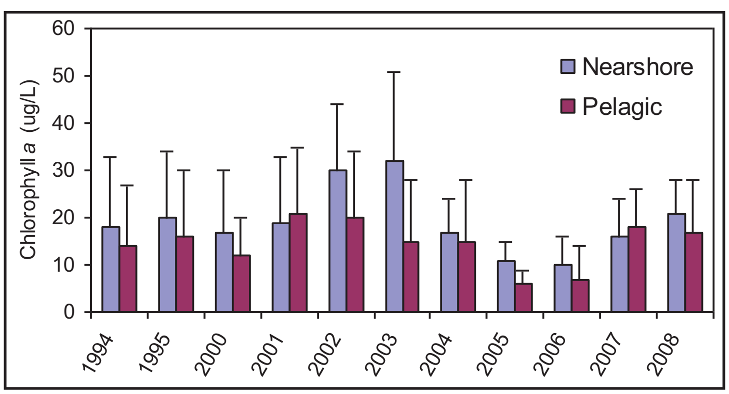


FIGURE 6-24. ANNUAL MEAN PHYTOPLANKTON BIOMASS AS CHLOROPHYLL *A* ± 1 SD AT BOTH NEARSHORE AND PELAGIC SITES

6.4.3.3 Light and Nutrient Bioassays

Long-term (1997-2000) bioassay results indicate that light limited phytoplankton growth approximately 60 percent of the time, while N-limited growth the remaining time (East and Sharfstein, 2006). These bioassays continue to be conducted on a quarterly basis.

6.4.3.4 Diatom to Cyanobacteria Ratio

Diatom to cyanobacteria ratios have been less than 1:1 since the mid-1990s. Since 2003, however, the ratios have been increasing at both the nearshore and pelagic sites, such that since 2004, they have exceeded the desired target ratio of 1.5:1 (red horizontal bar in ***Figure 6-25***). Since 2004, the diatom genera *Fragilaria*, *Aulacoseria* and *Cyclotella* have become increasingly important in both biovolumes and frequency of detection. Meroplankton has been found in the water column during low lake levels and this resuspended meroplankton, rather than the diatom and cyanobacteria assemblage, makes an accurate ratio difficult to obtain. Meeting and exceeding the restoration target prior to a period of time when water quality has not noticeably improved in Lake Okeechobee, perchance as an unrelated result of the hurricanes and water levels, brings into question the validity of the performance measure target. This suggests that the current performance measure should be modified.

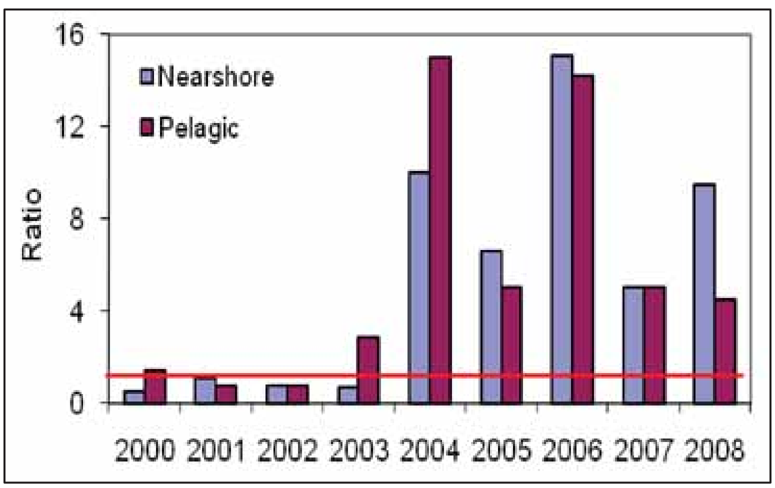


FIGURE 6-25. ANNUAL MEAN DIATOM TO CYANOBACTERIA RATIO AT BOTH NEARSHORE AND PELAGIC SITES

6.4.3.5 Cyanotoxin

Mean monthly microcystin concentrations measured at six sites located in the north, west and southeast areas of the nearshore region of Lake Okeechobee since May 2004 have generally been below the World Health Organization drinking water standard of 1 µg/L. This standard was exceeded during the period after Hurricanes Frances and Jeanne but prior to Hurricane Wilma passing very near or over the lake in the fall 2004 and 2005 (***Figure 6-26***).

The highest mean microcystin concentrations were recorded during a large *Microcystis*-dominated bloom in August 2005, when mean concentrations were between 15 µg/L and approximately 25 µg/L. The passage of Hurricane Wilma, which resuspended bottom sediments, thus increasing TSS concentrations, appears to have disrupted the large algal bloom, which greatly reduced microcystin concentrations.

Previous regression analysis suggested that the best relationship between microcystin and mean chlorophyll *a* concentrations was temporally-lagged between microcystin and the subsequent month's mean chlorophyll *a* concentration. Mean chlorophyll *a* concentrations were measured at the six sites monitored for microcystin, as well as at an additional three sites, one each located at the mouths of the Kissimmee River and Taylor Creek, plus a site on the west side of King's Bar. Mean chlorophyll *a* concentrations at these sites have generally ranged between 5 µg/L and 15 µg/L, though higher concentrations were observed during late spring-early summer 2004 and during the August to September 2005 large surficial bloom.

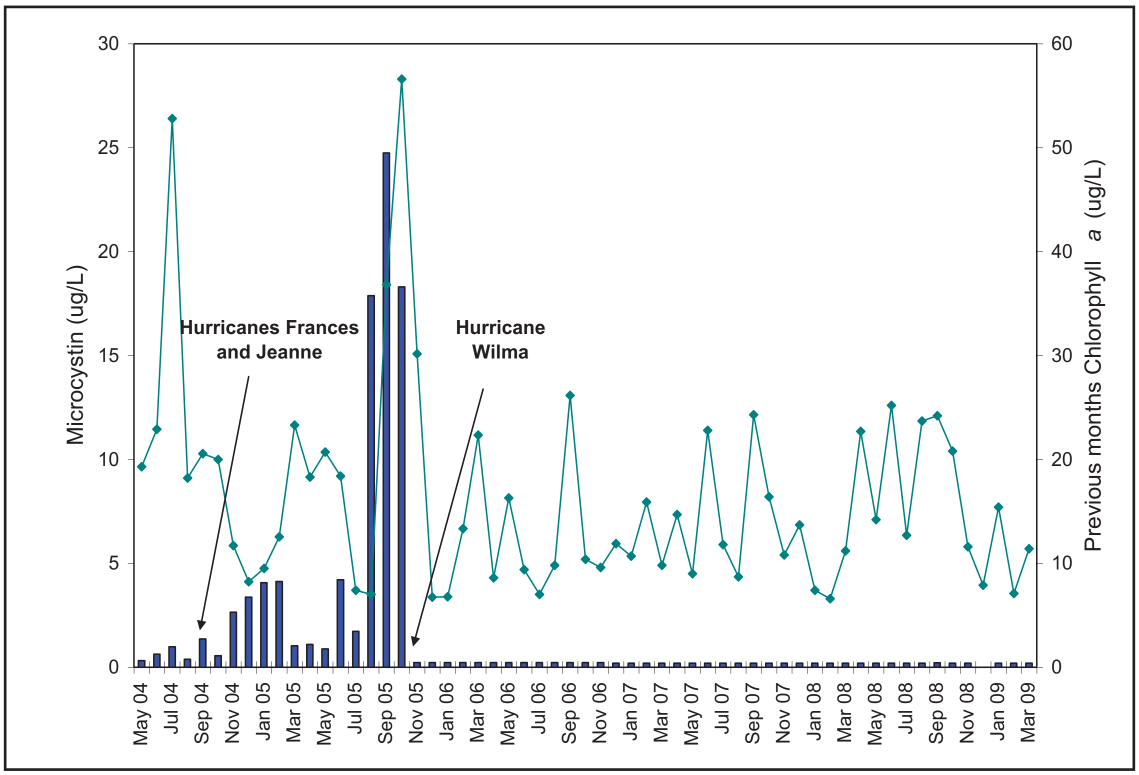


FIGURE 6-26. MEAN MICROCYSTIN AND PREVIOUS MONTH MEAN CHLOROPHYLL *A* CONCENTRATIONS

6.4.4 Summary

The variability in the 1994 to 1995 and 2000 to 2008 community composition data suggest that these changes may be a reflection of the dynamic climatic events experienced by Lake Okeechobee over the past decade. Lake stage has fluctuated between a historical high of 18.5 feet msl during an extremely wet 1995 and a historical low of 8.97 feet msl following a lake recession and prolonged drought in 2001. Since then, a new historical low lake stage of 8.82 feet msl was established during the 2007 drought. The extremely high turbidity, over 100 ppm TSS, after the 2004 and 2005 hurricanes remained until 2006. During 1994 and 1995, taxa that contributed most significantly to within-year similarity values were predominantly the cyanobacteria genera *Lyngbya*, *Anabaena*, and *Oscillatoria* along with the diatom *Melosira* and the cryptomonad genera *Cryptomonas* and *Rodomonas*. While the cyanobacteria taxa *Lyngbya* and *Oscillatoria* continued to play the most significant part in within-group similarity for 2000 through 2002, these groups have been increasingly influenced by diatom genera such as *Fragilaria*, *Aulacoseria* and *Cyclotella*. Since 2004, at least four of the top five most similar within-year taxa have been diatoms, suggesting that they are more consistently being found in samples. Several of these taxa, such as *Thalassiosira proschkinae*, and species of the genera *Aulacoseria, Cyclotella and Stephanodiscus* are nutrient tolerant and indicative of eutrophic or hyper eutrophic conditions (Yang et al., 2005).

Little spatial difference is observed among sites, which suggests that phytoplankton is most strongly influenced by temporal and seasonal factors. This contrasts with previous spatially heterogeneous ecological characterizations of the lake (Phlips et al., 1993a), including phytoplankton (Aldridge et al., 1995; Cichra et al., 1995). However, the lack of significant spatial differences in algal communities among the sites may be in part due to lack of data from nearshore sites when lake stage was so low as to preclude sample collection.

Correlations between water quality variables and community composition remained weak when examined with additional data collected from 2007 and 2008. This continues to suggest that relationships are complex and community structure is likely dependent on dynamically varying individual or composites of water quality factors at different times and under different conditions. An alternative explanation might be that unmeasured variables are more influential in the phytoplankton community structure relative to those that were measured.

It is anticipated that restoration would result in reduced nearshore and pelagic zone nutrient concentrations, with less frequent algal blooms. Blooms could be comprised of a smaller portion of cyanobacteria than has been observed over the past 20 years.

6.5 PERIPHYTON HYPOTHESIS CLUSTER

6.5.1 Introduction and Background

Periphyton may serve two important roles in the nearshore and littoral marsh regions of Lake Okeechobee (***Figure 6-27***). First, periphyton may be an important food source for higher trophic level grazers such as macroinvertebrates and fish (Zimba, 1995; Carrick and Steinman, 2001). Secondly, periphyton may indirectly suppress phytoplankton biomass and bloom frequency through competition for nutrients (Phlips et al., 1993a; Havens et al., 1996). This nutrient competition may occur once light limitation of the periphyton is relieved, typically during the less windy summer months, or during periods of low lake stage.

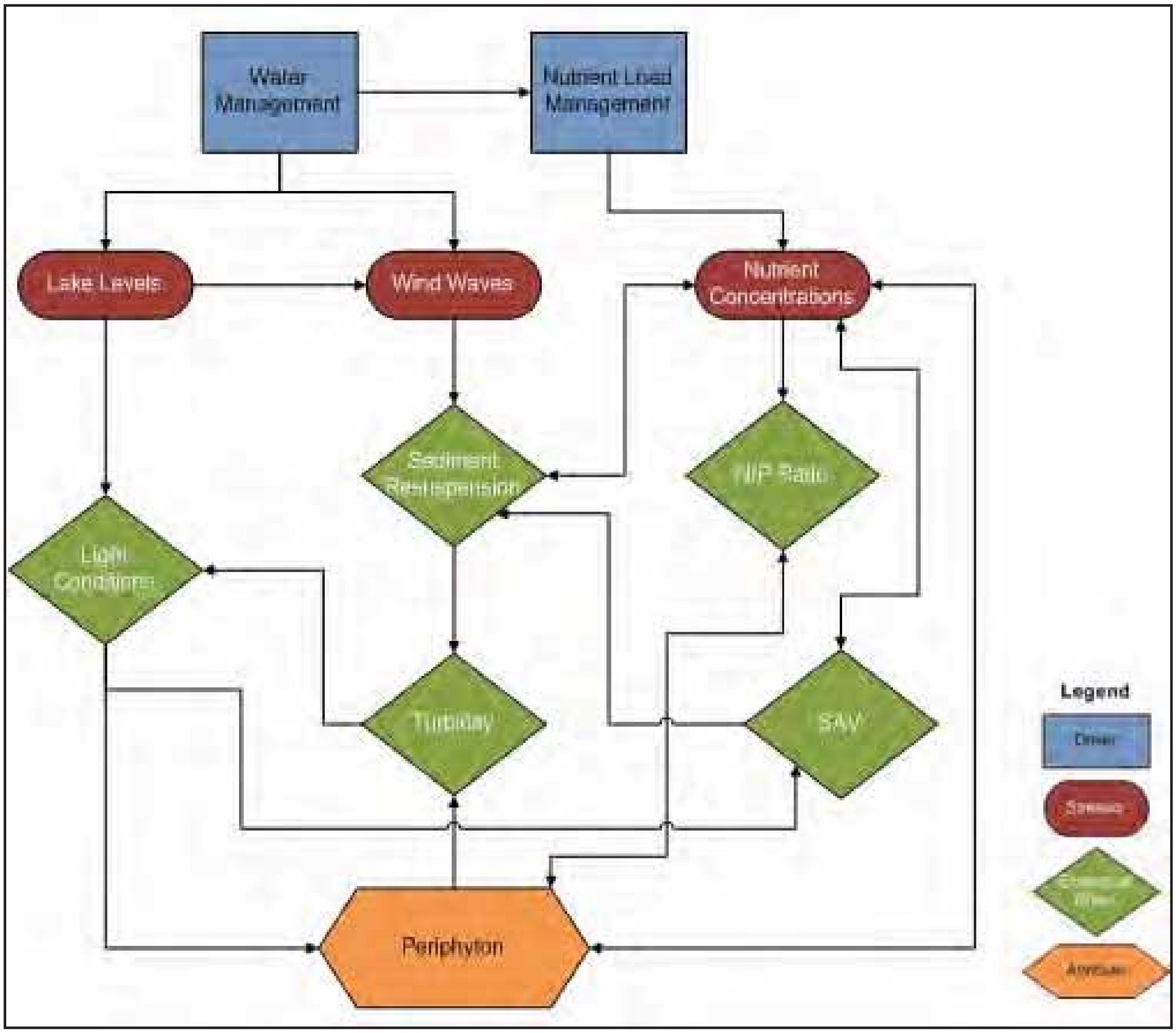


FIGURE 6-27. CONCEPTUAL MODEL FOR PERIPHYTON IN LAKE OKEECHOBEE

Periphyton monitoring and research has been conducted in the nearshore and littoral marsh regions of Lake Okeechobee since the late 1980s. Early studies focused on epiphyte (algae growing on emergent plant and SAV hosts) abundance, community composition, nutrient limitation, and phosphorus P storage capacity (Hopson and Zimba, 1993; Zimba, 1995; Hopson et al., 1998; Zimba, 1998). Two subsequent studies monitored both epiphytic and epipelic (bottom sediment-associated algae) abundance and community composition (Carrick and Steinman, 2001; Rodusky, in press). The more recent study also examined nutrient storage.

Nearshore periphyton abundance has been variable over the past roughly 20 years. Diatoms dominated the epiphytic assemblage during 1990 to 1991 (Hopson et al., 1998), a year in which Lake Okeechobee was generally at low (less than four meters) lake stages. Concurrent stable isotope analyses suggested that epiphytes were potentially important sources of carbon (C) and N to the higher trophic levels (Zimba, 1995). A synoptic study conducted during 1995, which was a year of very high (greater than five meters) lake stage, suggested that diatoms were weakly dominant in the epiphytic communities, while cyanobacteria dominated the epipelon. Periphyton biovolumes were low that year compared to other eutrophic lakes (Carrick and Steinman, 2001). A subsequent study was conducted in the nearshore region after a lake drawdown in 2000, followed by a prolonged drought and the passage of three hurricanes very near the lake during 2004 and 2005. These climatic events resulted in several large lake stage fluctuations and a large decline in nearshore emergent plant and SAV coverage. Both the epiphytic and epipelic communities between summer 2002 and winter 2006 were strongly dominated (greater than 80 percent) by diatoms, but periphyton biovolume was generally less than that found during the 1995 Carrick and Steinman study (Rodusky, in press). Lake stage as it relates to light availability and host substrate areal coverage in the nearshore region of Lake Okeechobee may be the most influential factors of periphyton abundance.

Nearshore and littoral marsh periphyton nutrient dynamics also have been studied in Lake Okeechobee. Both N and silica were found to limit epiphytic growth on artificial plants in a *Hydrilla* bed (Zimba, 1998). A subsequent study found that usually N, but occasionally P, was found to limit periphyton growth on nutrient-diffusing substrates in the nearshore region of the lake once water depths decreased and light limitation was relieved (Rodusky et al., 2001). Both N and P co-limited periphyton and phytoplankton growth at a southwestern site in the nearshore region and in littoral marsh mesocosms, suggesting that periphyton had the potential to indirectly limit phytoplankton biomass via nutrient competition (Havens et al., 1996; Havens et al., 1999). In the case of the marsh mesocosms; however, high N and P loading did not lead to phytoplankton dominance (Havens et al., 1999).

Examination of periphyton nutrient storage indicated that most of the epiphytic P storage was found to be on the SAV host associated substrates relative to that on emergent taxa (Zimba, 1995; Rodusky, in press). A recent estimate of periphyton P storage for the entire nearshore region (Rodusky, in press) was found to be low (8 mt) and approximately nine times lower than an earlier estimate (Zimba, 1995). Several factors may have contributed to the difference in P storage estimates. First, lake stage was generally 0.5 to 1 meter lower and SAV coverage was approximately three times higher during the Zimba (1995) study, relative to the more recent study. Secondly, epiphytic P storage estimates for *Chara* and *Hydrilla*, which comprised approximately 80 percent of the total mean SAV areal coverage during the more recent study, were not included because the samples usually did not contain enough epiphytic material to analyze for nutrients. Finally, the more recent nutrient storage estimates were derived from periphyton tissue samples, whereas the estimates calculated by Zimba (1995) were derived by assuming a 1:1 chlorophyll to phosphorus ratio. It may be that directly measuring periphyton nutrient content provides a more accurate estimate of nutrient storage relative to assuming a biomass to nutrient ratio.

In the case of the marsh and nearshore sites, studies have suggested that greater periphyton abundance led to an overall higher amount of P storage relative to that for phytoplankton during “clear water” phases or periods of low lake stage and increased light penetration into the water column (Havens et al., 2001b). This was observed despite higher phytoplankton P-uptake rates.

A decrease in P inputs to Lake Okeechobee, as part of CERP implementation, is expected to increase the water column N to P ratio above 22:1, decrease phytoplankton cyanobacteria bloom frequency, and alter bloom composition, with cyanobacteria comprising less than 50 percent of future blooms. A decrease in phytoplankton bloom frequency may result in less shading of the water column, thus enabling SAV and emergent plants to maintain higher areal coverage, which would provide more substrate for periphyton growth. Currently, periphyton abundance or nutrient storage targets for CERP have not been established.

6.5.2 Monitoring

Quarterly nearshore periphyton monitoring, which commenced in 2002, was suspended in early 2006 following the loss of nearly all nearshore SAV and most of the nearshore emergent vegetation. Monitoring recommenced in 2007 after a year of generally very low lake stages (e.g. less than 12 feet msl). Periphyton monitoring currently consists of collecting epiphytes from SAV shoots, emergent plant stems and epipelic cores collected from the bottom sediments. Sampling is conducted in the northern, western and southern areas of the nearshore region of Lake Okeechobee (***Figure 6-28***) on a semi-annual basis during March and September or October. During October 2007 and March 2008, no SAV or emergent sites had host plants to sample, so only epipelic data were collected. Some of the inshore sites (Pelbay1, Pelbay2, Southbay1, Ritta E1, RittaW1, RittaW2, KbarW1, KbarW2, EagleBay1 and EagleBay2) have not been sampled because either they were dry, or have upland or emergent plants as dominants. Physical water quality data are also collected. Community biomass, biovolume, composition, and tissue nutrient content (C, N and P) were determined.

6.5.3 Results

6.5.3.1 Biomass

Mean epipelic biomass declined between 2007 and 2008 in the northern and southern regions while it generally increased over the same time period in the western region (***Figure 6-29***). The 2002 to 2006 data are from a single site in each region whereas the 2007 to 2008 data are means for each region. No sampling was conducted between February 2006 and August 2007. Mean biomass between 2007 and 2008 ± SD was highest in the northern (288 ± 88 milligrams per square centimeter [mg/cm^2]) and western (203 ± 130 mg/cm^2) regions and lowest in the southern region (49 ± 30 mg/cm^2). These results are similar to those for the 2002 to 2006 data, wherein epipelic biomass was highest at the northern site and significantly lower at the southern site. Epipelon during 2002 to 2006 varied significantly by both site and season. Seasonally, mean epipelic biomass was significantly higher during spring 2005 and winter 2006 relative to the spring periods in 2003 and 2004.

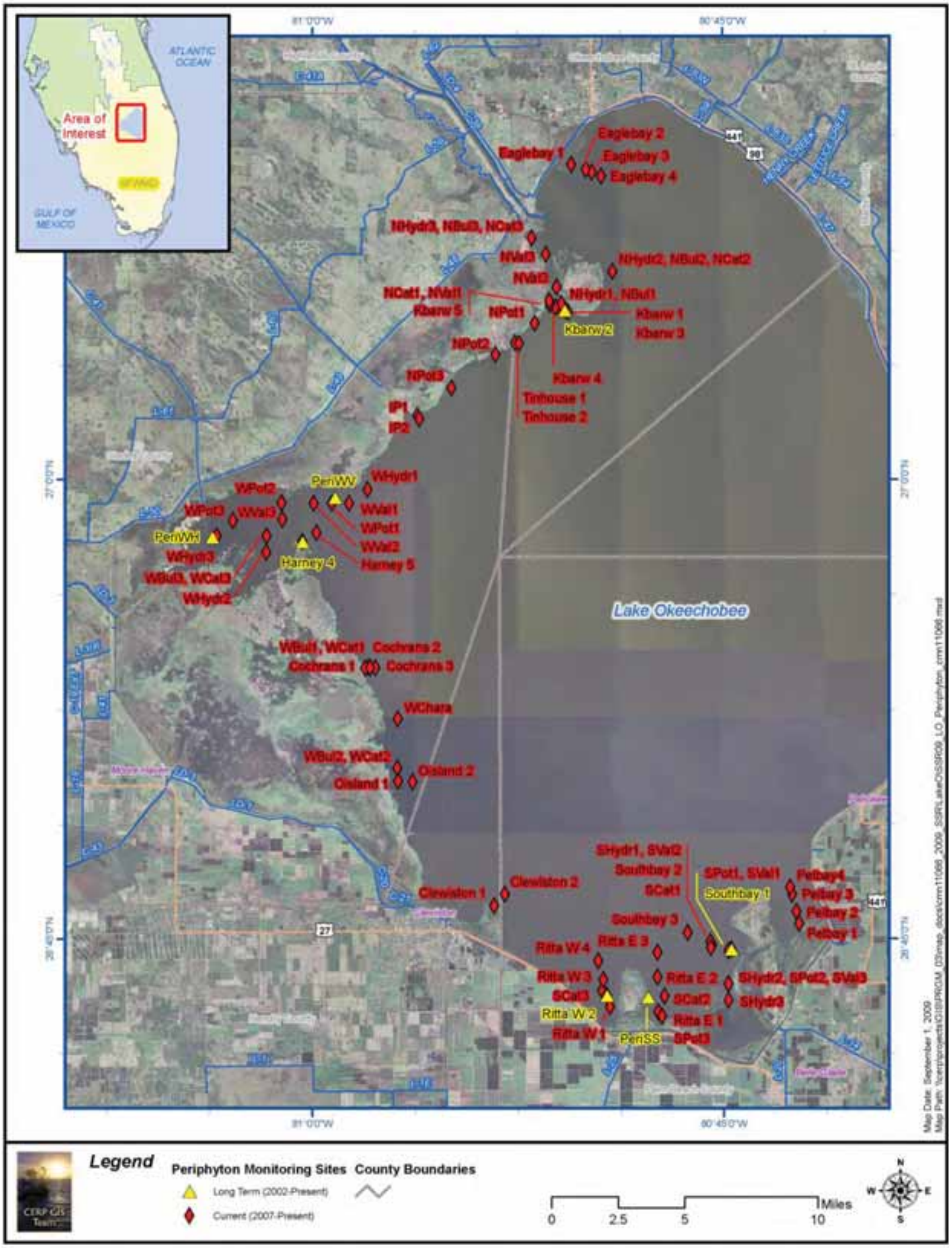


FIGURE 6-28. PERIPHYTON MONITORING SITES IN LAKE OKEECHOBEE

Note: Sites may change year to year as a function of lake stage, plant cover or physical access

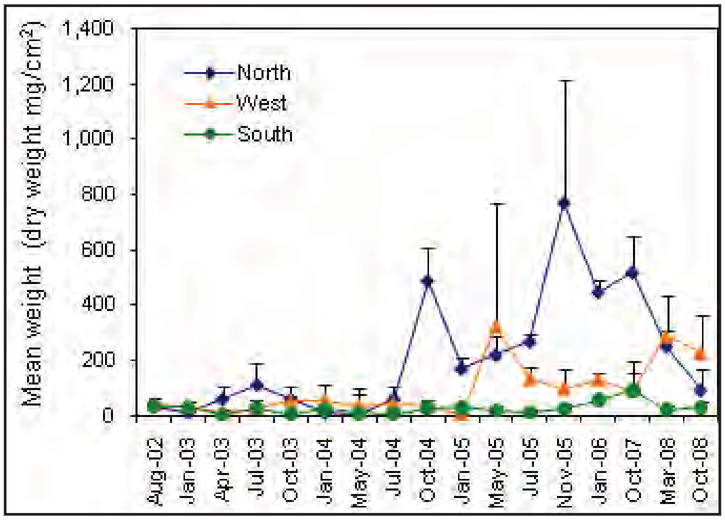


FIGURE 6-29. EPIPELIC BIOMASS +1 STANDARD DEVIATION BY DATE AND REGION

Mean epiphyte biomass for October 2008 was highest on *Vallisneria* at two sites in the western region (311 ± 72 milligram per gram [mg/g]) and lowest and approximately the same (10 mg/g) on *Typha* in all three regions. Mean *Vallisneria*-associated epiphyte biomass from two sites in the northern region (36 ± 18 mg/g) was considerably less than that for the corresponding mean western region *Vallisneria*-associated epiphytes (*Figure 6-30*). Mean *Chara*-associated epiphytic biomass (195 ± 65 mg/g) was higher than that for the remaining epiphytic communities. These results contrast with those from the 2002 to 2006 data, wherein epiphyte biomass was highest on *Chara*, lowest on *Scirpus*, and similar among the vascular SAV taxa (*Figure 6-31*). Scheffe's test on ranked data suggested that epiphytic biomass between 2002 and 2006 varied significantly among dates and among the replicate *Vallisneria* sites (p less than 0.001); the latter result contrasts with the corresponding biovolume data (***Table 6-5***). Epiphytic biomass displayed the same temporal patterns as was observed for their respective host substrates. For the among-site comparisons, the *Scirpus* at site WES and the *Vallisneria* at site WV had significantly higher mean epiphytic biomass than at the *Scirpus* site SS and the remaining *Vallisneria* sites, respectively. *Figure 6-32* presents *Potamogeton* data.

Despite higher than desired lake stages for the majority of the 2002 to 2006 study period, corresponding areal epiphytic biomass ranges were similar to those recently reported in other shallow waters. Data ranged from roughly less than 1 gram per square meter (g/m^2) to 109 g/m^2 for the SAV epiphytes and less than 1 g/m^2 to 12 g/m^2 for *Scirpus*-associated epiphytes. Kiss et al (2003) reported biomass of less than 20 g/m^2 to greater than 40 g/m^2 for epiphytes on differing age classes of *Phragmites* stems in a shallow Hungarian reservoir and restored wetland. Liboriussen and Jeppesen (2006) reported epiphytic biomass of 10 g/m^2 to 23 g/m^2 on plastic strips in shallow temperate Danish lakes of varying trophic status. These comparisons suggest that a consistently lower lake stage in Lake Okeechobee also might result in epiphytic biomass that exceeds biomass reported for these temperate lakes.

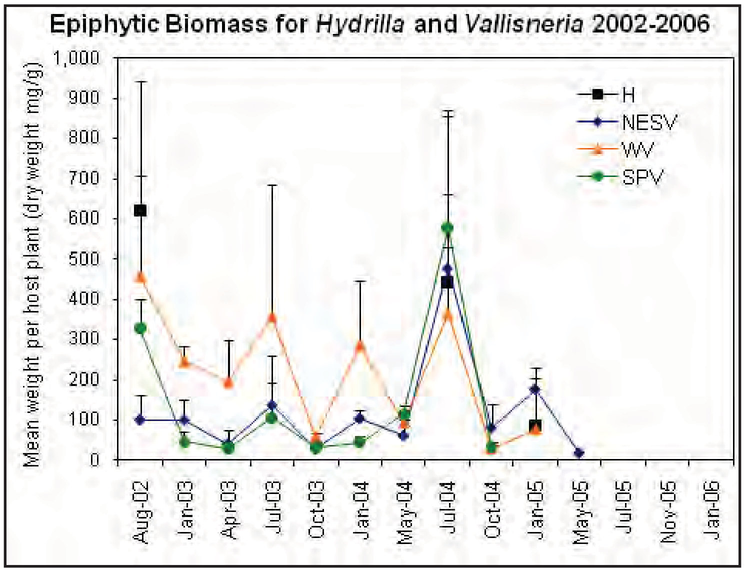


FIGURE 6-30. MEAN 2002-2006 EPIPHYTIC BIOMASS +1 STANDARD DEVIATION PER UNIT HOST, HYDRILLA AND REPLICATE VALLISNERIA, BY DATE AND SITE

Note: site names in legend

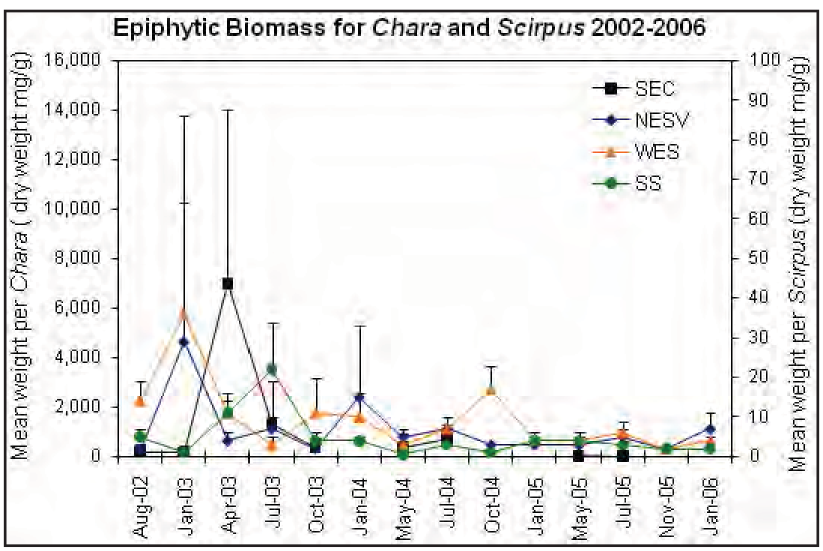


FIGURE 6-31. MEAN 2002-2006 EPIPHYTIC BIOMASS +1 STANDARD DEVIATION PER UNIT HOST, CHARA AND REPLICATE SCIRPUS, BY DATE AND SITE

Note: Site names in legend

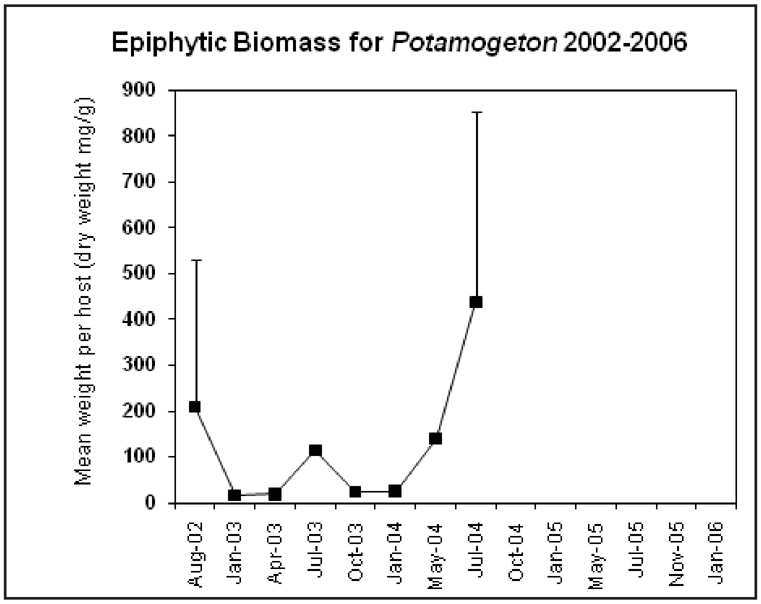


FIGURE 6-32. MEAN 2002–2006 EPIPHYTIC BIOMASS PER UNIT HOST, POTAMOGETON, DRY WEIGHT (MG/G, +1 SD)

Note: By date and site

6.5.3.2 Biovolume

Mean epipelic biovolumes were highest (2.1×10^8 cubic micromoles per square centimeters [$\mu m^3/cm^2$]) in the northern and southern regions during fall 2007 and highest (2.4×10^8 $\mu m^3/cm^2$) in the western region during 2008 (***Figure 6-33***). The lowest mean epipelic biovolumes (5.1×10^7 $\mu m^3/cm^2$) were in the southern region during 2008.

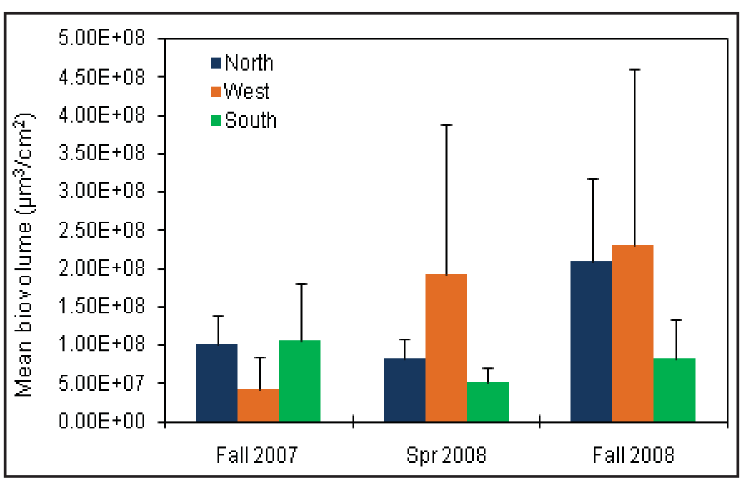


FIGURE 6-33. MEAN REGIONAL EPIPELIC CELL BIOVOLUMES 2007 - 2008

Epipelic data were square root transformed, which yielded slightly greater separation compared to untransformed and fourth root transformed data, in the analysis of similarities (ANOSIM) routine. The ANOSIM tests suggested that lake stage ($R=0.40$, $p <0.01$) and season ($R=0.24$, $p<0.01$) were the most influential combination of factors and this combination was more influential than any single factor on epipelic biovolume between fall 2007 and fall 2008. The R-value for lake stage suggests that the stage fluctuations between "desired" stage (between 12.5 and 15.5 feet msl) and low stages experienced by Lake Okeechboee during this year were marginally influential in the amount of biovolume differences among the epipelic communities. Other factors examined (e.g., region, sediment composition) and combinations of these factors were much less influential (e.g., $R <0.20$) relative to lake stage and season.

The taxonomic similarity percentage (SIMPER) routine suggested that diatom taxa accounted for greater than 80 percent of the most consistently found taxa during both fall 2007 and fall 2008, when lake stage was either below or within the desired lake stage envelope. Diatom dominance during spring 2008 was even more pronounced, comprising greater than 90 percent of the within-group similarity percentage value. Dominant diatom taxa were *Fragilaria* sp*., Navicula* sp. and *Aulacoseira* sp. The few non-diatom taxa that contributed to the within-group similarity percentages were the cyanobacteria *Leptolyngbya* sp and *Oscillatoria* spp. Among-groups most dissimilar taxa were the green algae *Oedegonium* sp. for the among-lake stage comparisons and *Aulacoseira* sp. for fall-spring comparisons.

Total annual epipelic biovolumes between 2003 and 2005 were highest during 2004 (6.4×10^8 $\mu m^3/cm^2$) and lowest during 2005 (1.8×10^8 $\mu m^3/cm^2$). Decreased epipelic biovolumes over this period may have reflected a substantially reduced light regime in the water column following the passage of two hurricanes during fall 2004. The reduced light regime reflected a roughly two meter increase in lake stage and six to eight months of elevated (greater than100 mg/L) TSS concentrations in the nearshore region of Lake okeechobee.

When matching the 2002 to 2008 epipelic assemblage data to available water quality data, the combination of depth, Secchi disc depth to total depth ratio (SD to TD), P and the ratio of DIN to SRP had the strongest Spearman correlation ($\rho=0.34$, $p=0.01$) with the epipelic assemblages. These pattern matches suggest that light-related variables dissolved N and P may have had some influence on the epipelic communities. The periphyton water quality variable data were transformed in the same manner as that for the phytoplankton water quality data. Water column temperature, specific conductance and ammonia data were omitted from this analysis because of missing data.

Fall and spring data were used to compare the 2003 to 2005 and the 2007 to 2008 epipelic data sets, since data was not collected during the summer and winter from 2007 to 2008. The combination of stage ($R=0.78$, $p<0.01$) and season ($R-0.56$, $p<0.01$) (***Figure 6-34***) were the most influential factors and that the among-groups separation was clear when examining by lake stage.

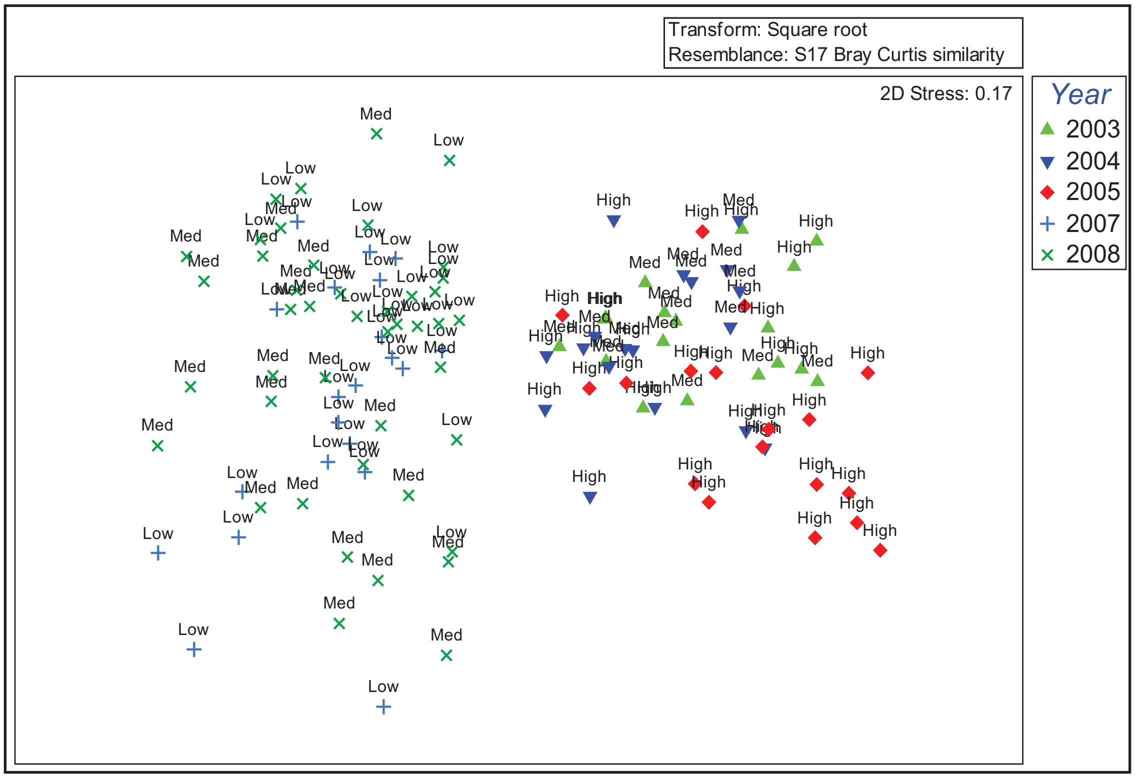


FIGURE 6-34. SEPARATION OF EPIPELIC COMMUNITIES UNDER LOW, DESIRED (MED;12.5-15.5 FEET) AND HIGH LAKE STAGES BETWEEN 2003 AND 2008

The same diatom taxa previously mentioned dominated the within-group similarity percentages, comprising greater than 83 percent of the mean value under low lake stage, to greater than 90 percent at high lake stages. Among the groups, diatom biovolume differences were greatest between the low and high lake stages and most similar among the desired and low lake stages. An inverse relationship existed between the high and low lake stages and the dominant diatom taxa biovolumes, while some variability was observed among the taxa between the desired and low lake stages. Only *Oedogonium* sp., at low lake stages, accounted for more than five percent of the mean dissimilarity percentage value among any of the taxa when grouped by lake stage.

Greater similarity was observed among the epipelic taxa in the fall, and biovolumes of the dominant taxa (diatoms) were typically higher in the fall, relative to those observed for the spring assemblages. Among these two groups, diatom taxa accounted for approximately 80 percent of the dissimilarity percentage value among the fall and spring assemblages.

Mean epiphytic biovolume per unit of host substrate dry weight cubic micromoles per gram ($\mu m^3/g$) for fall 2008 was highest on *Chara* in the southern region of Lake Okeechobee (***Table 6-4***). *Vallisneria*-associated epiphytes had higher mean biovolumes than did those on *Typha* in both the northern and western regions, while mean *Typha*-associated epiphytic biovolume was highest in the southern region and lowest in the western region.

TABLE 6-4. MEAN EPIPHYTIC BIOVOLUMES (µM³/G) FOR FALL 2008 SAMPLING PERIOD

Region		
Northern	**Western**	**Southern**
Typha – 2.1×10^8 *Vallisneria* – 9.8×10^8	*Typha* – 6.4×10^7 *Vallisneria* – 2.0×10^8	*Chara* – 3.5×10^9 *Typha* – 4.4×10^8

Mean fall biovolumes during 2003, 2004 and 2008 for *Chara* (southern region) and *Vallisneria* (northern and western regions) were always highest during 2008 and lowest during 2004 (***Table 6-5***). No Chara epiphyte data is included for fall 2004. Low lake stage and subsequent generally high light penetration (e.g., more than 250 micromoles of photons per second per square meter [µm/sec/m²]) during 2008, and contrasting high lake stage and turbidity (e.g., more than 100 mg/L) following the nearby passage of Hurricanes Francis and Jeanne during September 2004 coincided with these biovolume differences.

TABLE 6-5. MEAN FALL BIOVOLUMES (µM³/G ± SD) BY SITE (2003-2004) AND REPLICATE SITES (2008) FOR *CHARA* AND *VALLISNERIA*-ASSOCIATED EPIPHYTES

Year	*Chara*	Northern *Vallisneria*	Western *Vallisneria*
2003	$2.7 \times 10^8 \pm 1.6 \times 10^8$	$1.7 \times 10^8 \pm 2.0 \times 10^7$	$6.7 \times 10^7 \pm 5.3 \times 10^6$
2004	No Data	$1.7 \times 10^7 \pm 4.8 \times 10^6$	$1.5 \times 10^7 \pm 1.5 \times 10^6$
2008	$3.4 \times 10^9 \pm 2.4 \times 10^9$	$9.8 \times 10^8 \pm 5.4 \times 10^8$	$2.0 \times 10^8 \pm 2.6 \times 10^7$

Water quality variables appeared to be less influential on the epiphytic assemblage composition compared to that for the epipelon. A combination of P and TSS has the highest, but weak ($\rho=0.19$, $p=0.01$) correlation with epiphytic assemblage patterns. The relative weakness of this correlation suggests that water quality variables were not very influential in the epiphytic assemblage patterns. The influence of water quality variables on the periphyton commmunities may have been stronger if water quality data collected during the prior month were used (McCarthy et al. 2009), although this approach was previously used (Rodusky, in press) and the strength of the water quality variable – epiphytic composition correlations was similar to that reported herein. The generally weak correlations between the water quality variables and both the epiphytic and epipelic assemblages lends additional support to the idea that substrate or some other factor such as light limitation rather than water column nutrients is a more important factor in influencing periphyton communities in Lake Okeechobee. Water column chlorophyll *a* concentration, specific conductance and nitrate+nitrite data were omitted from this analysis because of missing data.

The ANOSIM analysis of the 2002 to 2008 epiphytic data suggested that host plant type ($R=0.49$, $p<0.01$) was the most influential factor in the epiphytic assemblage ordination patterns. The combination of plant type ($R=0.52$, $p<0.01$) and season ($R=0.13$, $p<0.01$) provided slightly more differentiation among the epiphytic assemblages. Site location ($R=0.44$, $p<0.01$) and lake stage ($R=0.11$, $p<0.01$), and combinations of all of these factors were less influential relative to plant type alone.

Diatom taxa likewise dominated the epiphytic SIMPER results. The epiphytic assemblages were most similar on the *Potamogeton* and least similar on the *Typha* and *Chara* substrates. Among the plant hosts sampled in at least two of the regions, the most similar epiphytic communities were found on *Scirpus*. Diatom taxa, typically *Cocconeis* spp., *Fragilaria* sp., *Synedra* spp. and *Aulacoseira* spp. were the most consistently found within each plant host epiphytic assemblage. Diatoms accounted for between 85 and 91 percent of the mean within-group percentage similarity value for all host substrates except for *Typha*, wherein diatoms comprised roughly 65 percent of the mean similarity value. In the case of *Typha*, the cyanobacteria *Leptolyngbya* sp. and *Oedegonium* sp. contributed 17 and 5 percent to the overall mean similarity value, respectively. The epiphytic assemblages were most dissimilar among the *Chara* and *Scirpus* hosts and most similar among the *Potamogeton* and *Vallisneria* hosts. In both comparisons, it was primarily diatom taxa that contributed most of the mean dissimilarity values. The second highest mean dissimilarity value was between the epiphytic communities on the two emergent plant hosts (*Scirpus* and *Typha*). *Leptolyngbya sp*., and the chlorophytes *Chaetophora* sp. and *Oedogonium* sp., jointly contributed 23 percent to the dissimilarity value among the two epiphytic assemblages. While diatom taxa still dominated contributions to the among-host dissimilarity value, this was the only instance where non-diatom taxa contributed more than roughly five percent to the assemblage differences. Epiphytic biovolumes were generally higher on the *Typha* host stems relative to that on the *Scirpus.*

6.5.3.3 Nutrient Storage

Epiphytic tissue nutrient content, measured during 2002 through 2006, suggests that on average, approximately 8 mt of P was bound in the nearshore periphyton (Rodusky, submitted). This estimated storage was 10 percent of Zimba's (1995) estimate. However, mean vascular SAV coverage during that study was approximately three times higher than it was during the 2002 to 2006 study period. Additionally, Zimba's (1995) P storage estimate included ten mt from *Typha*-associated epiphytes, which were not measured during the recent study because of low areal coverage. The remaining discrepancy between the two P storage estimates may reflect that the current estimates were derived from direct epiphytic P measurements whereas the Zimba (1995) estimate was derived from an assumption of a 1:1 chlorophyll to P ratio, which was used to estimate P equivalents per unit area. Using this ratio may have overestimated P storage for epiphytic assemblages on some of the SAV taxa. Since the 2002 to 2006 epiphytic P storage data are similar to those measured in epiphytes on artificial substrate carriers in German mesoeutrophic gravel pit lakes (Jöbgen et al., 2004), the current estimates of P storage seem to be reasonable and suggest that measuring epiphytic nutrients rather than using a conversion factor may provide a more realistic estimate of total epiphytic P storage.

While the recent nutrient storage data suggest that periphyton in Lake Okeechobee do not appear to be important as a short-term nutrient sink, it is likely that the actual amount of P storage has been underestimated. The recent estimate does not include P stored by *Chara*, *Hydrilla* and epipelon. While the amount of epiphytic material collected from these two SAV taxa usually was too small to analyze for nutrient content, mean areal coverage for these two SAV taxa were roughly 80 percent of the total SAV mean coverage during the study period. The amount and inter-annual variability in P stored by the epiphytic communities in this amount of SAV may have been significant. For example, nutrient data from the one date where there was sufficient epiphytic material from *Hydrilla* suggested that between 9 and 12 mt of P may have been stored by these epiphytes. Combined with an almost three times increase in areal coverage between 2002 and 2004, the amount of nutrients stored by *Hydrilla*-associated epiphytes alone may have significantly increased during the first half of the study period. Additionally, the amount of P stored by *Hydrilla* was approximately three times higher than that estimated for collective TP storage by epiphytes on *Potamogeton*, *Scirpus* and *Vallisneria*. Nutrient storage estimates for epipelon were not included, since data were lacking on its areal extent and given the lack of a water column depth - epipelic biomass relationship in the nearshore region. When considering that the epipelic community, which had the highest nutrient content, was not included in the nearshore estimate, the overall amount of P storage that could not be accounted for may have been substantial.

6.5.4 Conclusions

Colonizable substrate for epiphytes and the water column light regime for periphyton were greatly reduced after the passage of three hurricanes during 2004 and 2005. Thus, unfavorable environmental conditions appear to have existed for periphytic growth and corresponding nutrient storage capacity after the hurricanes passed. A prolonged drought and subsequent extremely low lake stages during 2007 and 2008 appear to have reversed those conditions and has created favorable conditions for the re-establishment of SAV in Lake Okeechobee. Expansion of SAV in the lake has provided increased substrate for epiphytic assemblages and the extremely low lake stages have facilitated greatly improved light penetration to the sediments, which has greatly improving conditions for epipelic growth. While the amount of post-drought periphyton data is limited, the available data suggest that both periphyton biomass and biovolumes are similar to or exceed those recorded prior to the hurricanes.

Periphyton abundance and P storage may be especially important in the nearshore region of Lake Okeechobee over the next few years since the post-hurricane annual water column mean P concentrations have been greater than 100 µg/L. Reestablishing large SAV areal coverage in Lake Okeechobee appears to be very important, since this P concentration is within the range considered optimal in eutrophic temperate lakes for periphyton growth when host substrate is available to colonize (Liboriussen and Jeppesen, 2006). This P concentration is within the range where shallow subtropical eutrophic lakes might switch from SAV to phytoplankton dominance (Yang et al., 2008).

The importance of periphyton in Lake Okeechobee is still poorly understood. Diatom taxa have dominated the pre- and post-hurricane period data, despite a recent dramatic improvement in the water column light regime, which, coupled with high water column nutrient concentrations, would be considered more favorable for cyanobacteria growth. The role of the nearshore periphyton as a nutrient sink would benefit from additional research, especially under lower lake stages that are expected to occur once CERP water storage projects are completed. Semi-annual monitoring efforts would continue to document the periphyton abundance, assemblage composition, and nutrient storage potential to gain a better understanding of the role and importance of periphyton in the food web and as a nutrient sink in the nearshore region of Lake Okeechobee.

6.6 SUBMERGED AQUATIC VEGETATION HYPOTHESIS CLUSTER

6.6.1 Introduction and Background

SAV plays a key role in shallow lakes, providing diverse spawning and foraging habitat for fish and provides an important food and habitat resource for wading birds, and other wildlife (Havens and Gawlik, 2005). It directly affects water quality attributes such as nutrient concentrations, water column transparency and phytoplankton biomass. Lake level, periodic wind-driven high turbidity, and major physical perturbations such as hurricanes act as external drivers to Lake Okeechobee SAV (***Figure 6-35***). The key characteristics of concern for the spatial extent, density and species composition of SAV in Lake Okeechobee are lake stage, major wind and wave events, nutrient concentrations, light climate, algal bloom frequency and composition, turbidity, sedimentation rates, sediment resuspension and cycling of nutrients sequestered in the bottom sediments. Under existing watershed uses and lake management activities, the spatial extent and abundance of SAV varies widely from year to year. A more detailed discussion of the SAV hypothesis cluster and the role of SAV in Lake Okeechobee is provided in the 2007 SSR (RECOVER 2007) available online at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

RECOVER targets currently specify an annual standing stock of 49,420 acres of total SAV, with at least 50 percent composed of native species. A CERP systemwide performance measure and an interim goal have been developed for Lake Okeechobee vegetation, which are available online at www.evergladesplan.org/pm/recover/recover_docs/et/lo_pm_vegetationmosaic.pdf and www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_2-4_lakeoaquaticveg.pdf, respectively.

6.6.2 Monitoring

SAV and its relationship to the health of Lake Okeechobee is assessed by periodically sampling plant biomass and species composition along strategically located fixed transects, and by large-scale mapping of species specific vegetative coverage. A SAV monitoring program has been in place in Lake Okeechobee since spring 1999 and encompasses data collected over a wide range of hydrological and environmental conditions. A change in collection methodology, however, allows a comparison only of the data collected since summer 2000. Additionally, historical SAV biomass and distribution data exists from transect studies conducted in the late 1980s and early 1990s (Zimba et al., 1995), which can be compared to the current SAV distribution and abundance.

SAV is monitored at two different spatio-temporal scales. Both methods rely on a boat-based sampling methodology, as areas with SAV are generally characterized by water with poor transparency or that is highly colored by dissolved organics, which has thus far prevented the use of remote sensing techniques.

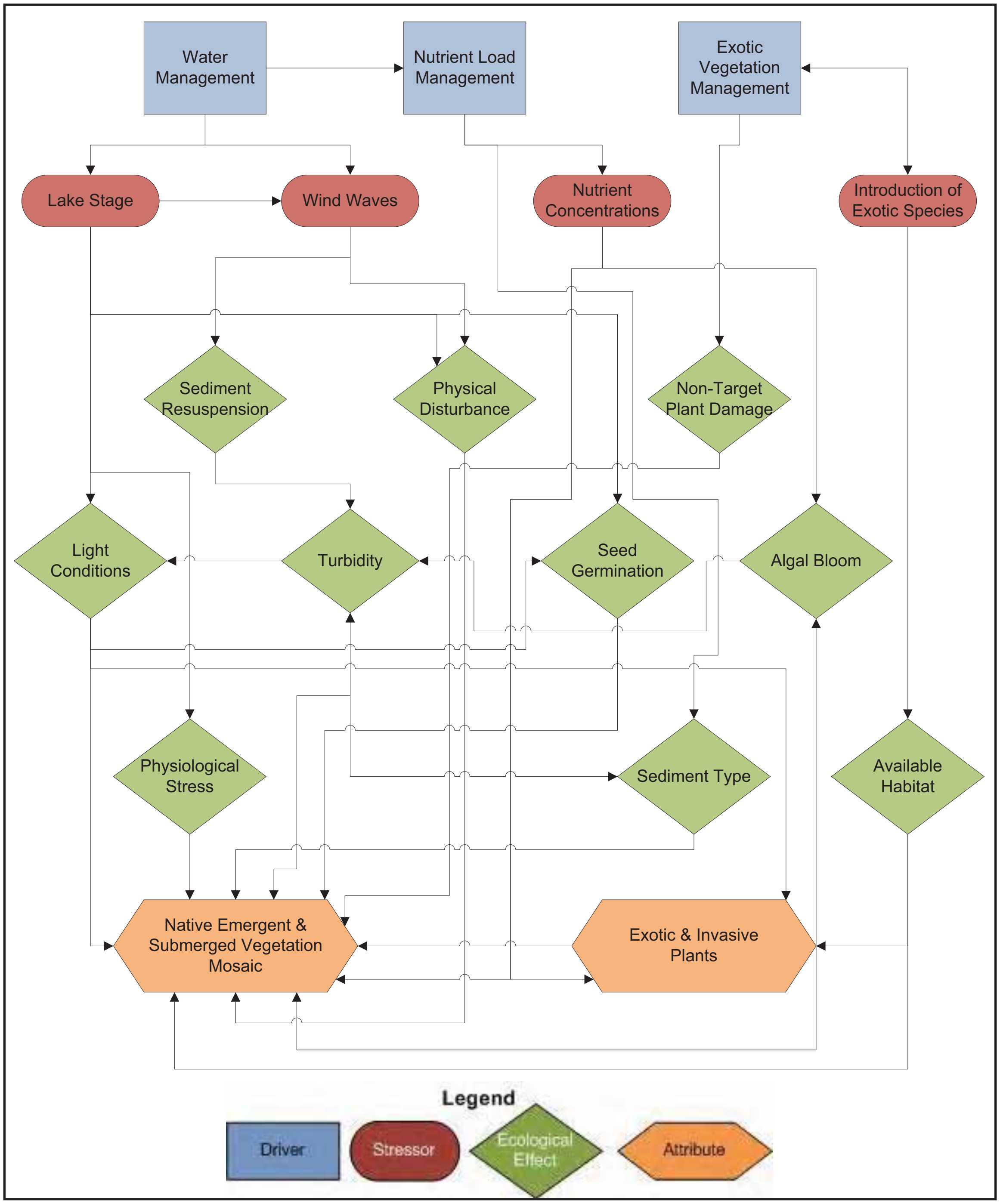


FIGURE 6-35. CONCEPTUAL MODEL OF SUBMERGED AQUATIC VEGETATION IN LAKE OKEECHOBEE

6.6.2.1 Transect Monitoring

In order to obtain relatively rapid quantitative estimates of plant species biomass, sampling is conducted at up to 78 sites located along 16 transects in areas of Lake Okeechobee that support submerged plants (***Figure 6-36***). The sites represent a subset of sites that were sampled in the in the late 1980s and early 1990s (Zimba et al., 1995), allowing for a comparison to historical data. Sampling frequency varies from quarterly to monthly depending on how dynamic anticipated changes in the plant population are expected to be (e.g., more frequent sampling is done during periods of recovery from hurricanes). This sampling effort provides information on plant responses and relative plant distribution and density to changing water levels on a shorter time scale than that for the annual SAV mapping and can be used as input to real-time operations. More details on the transect monitoring methodology can be found in the 2007 SSR, available at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx, and in Rodusky et al. (2005).

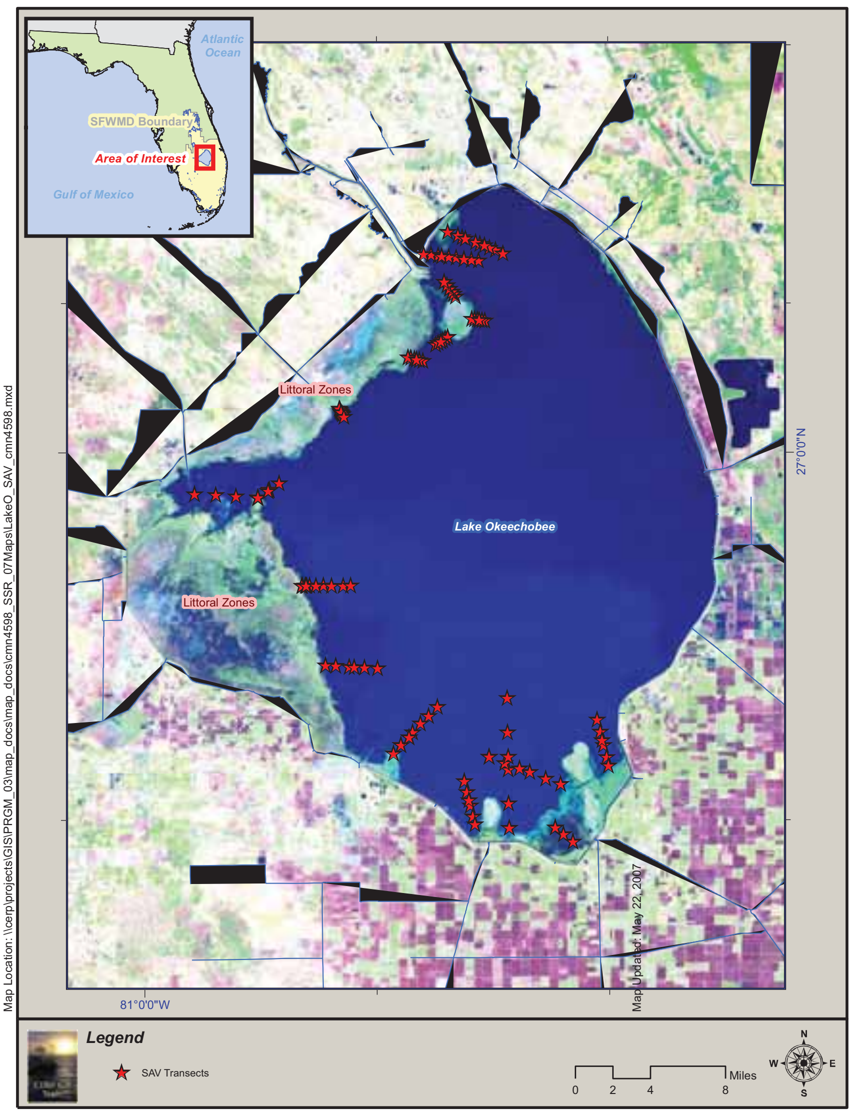


FIGURE 6-36. SUBMERGED AQUATIC VEGETATION TRANSECT LOCATIONS IN LAKE OKEECHOBEE

6.6.2.2 Annual Mapping

The total spatial extent, species distribution, and acreage of SAV is determined by an intensive sampling program that is carried out at the end of the peak SAV growing season, which occurs August through September (Havens et al., 2002). Rather than sampling random locations, the entire nearshore area is evaluated at a spatial scale sufficient to detect significant changes. ***Figure 6-37*** shows the annual mapping sampling grid. More details on the annual mapping can be found in the 2007 SSR, which can be found online at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

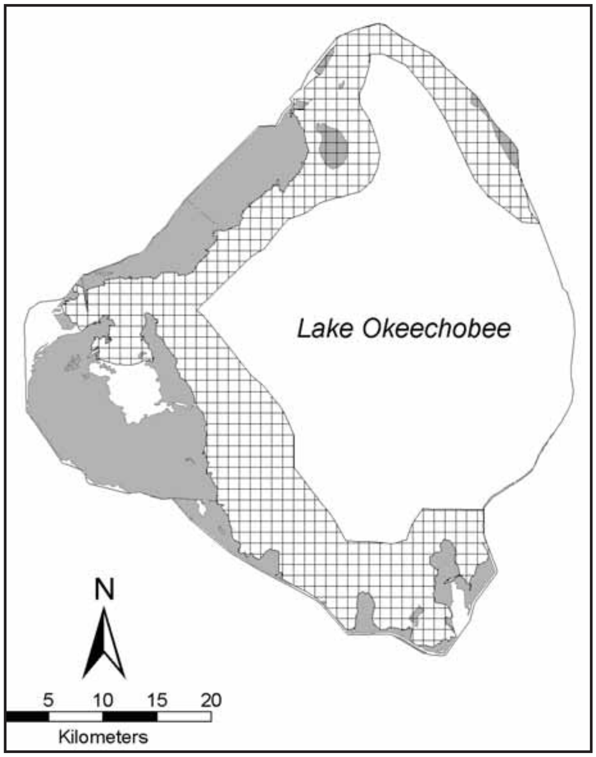


FIGURE 6-37. ANNUAL SUBMERGED AQUATIC VEGETATIVE MAPPING SAMPLING GRID

This sampling effort provides information on the total number of acres of plants that the lake gained or lost under the prevailing hydrologic conditions of a given growth cycle year. Due to annual growth season fluctuations that might result in months other than August and September containing peak SAV abundance (Havens et al., 2002), these data should be used in the context of a coarse temporal scale trend analysis. This information is used to provide a yearly score for the RECOVER performance measure.

The underlying assumption of the timing of the annual large-scale SAV mapping is that the most active growing season would not deviate significantly from the August through September timeframe; however, stage and photoperiod undoubtedly varies year to year. As a consequence, the annual mapping data lends itself most appropriately to evaluation of longer-term trends and should only be cautiously employed as regards to between-year differences. Conversely, sampling along transects is better suited for identifying and understanding short-term changes (Havens et al., 2002). The two approaches are thus complimentary, and sufficiently define the appropriate timescales as to allow interpretation.

6.6.3 Results

6.6.3.1 Transect Monitoring

Over the nine-year sampling period, two multi-year droughts, 2000 to 2001 and 2007 to 2008, and three hurricanes in 2004 and 2005 impacted Lake Okeechobee. Lake stage varied from a high of 17.8 feet National Geodetic Vertical Datum (NGVD) in October 2004 to a record low of 8.82 feet NGVD in July 2007 (***Figure 6-38a*** and ***Figure 6-39a***).

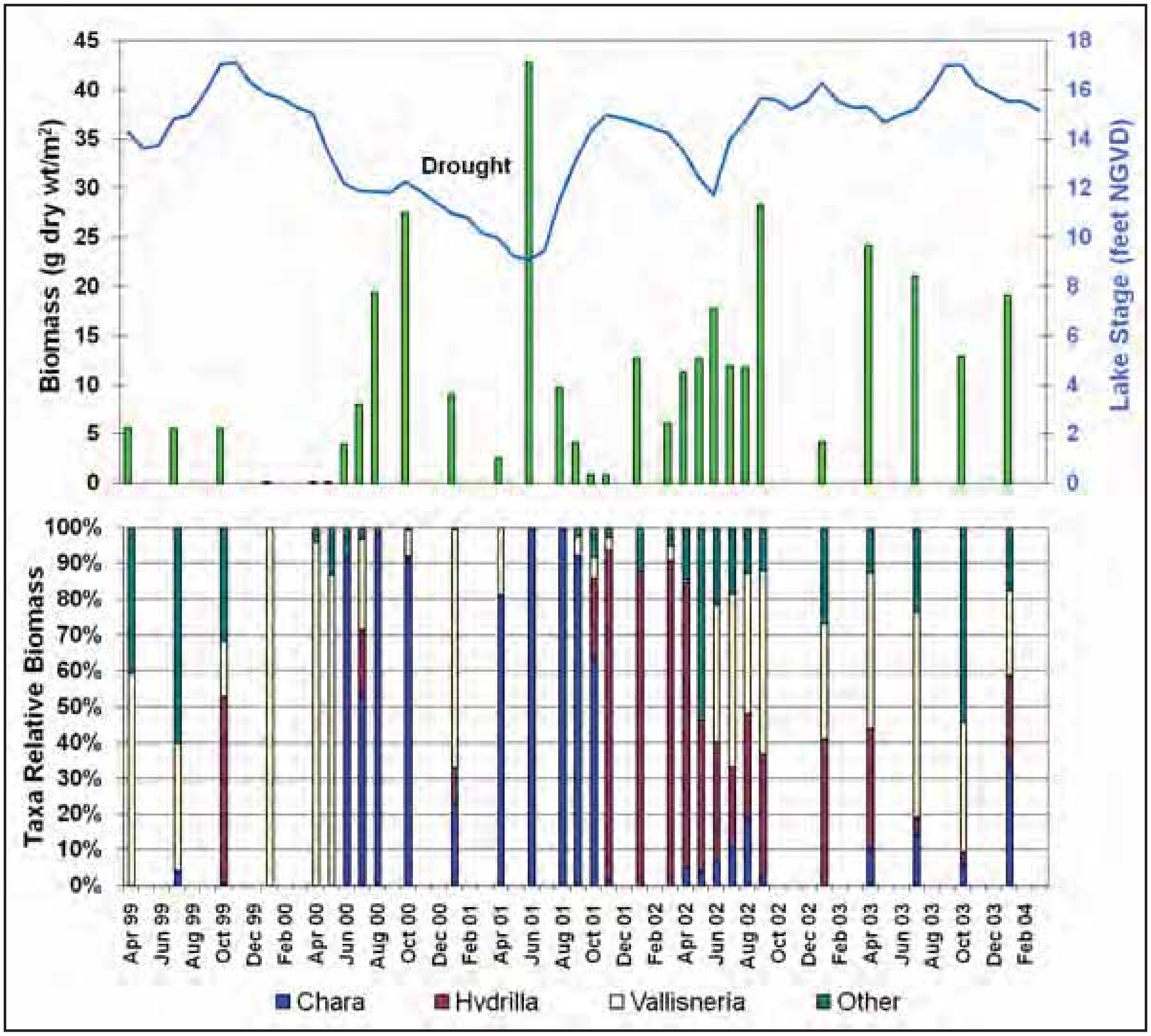


FIGURE 6-38. RESULTS FROM TRANSECT SAMPLING APRIL 1999 TO FEBRUARY 2004

Key: a) variation in SAV biomass
b) relative biomass of plant species

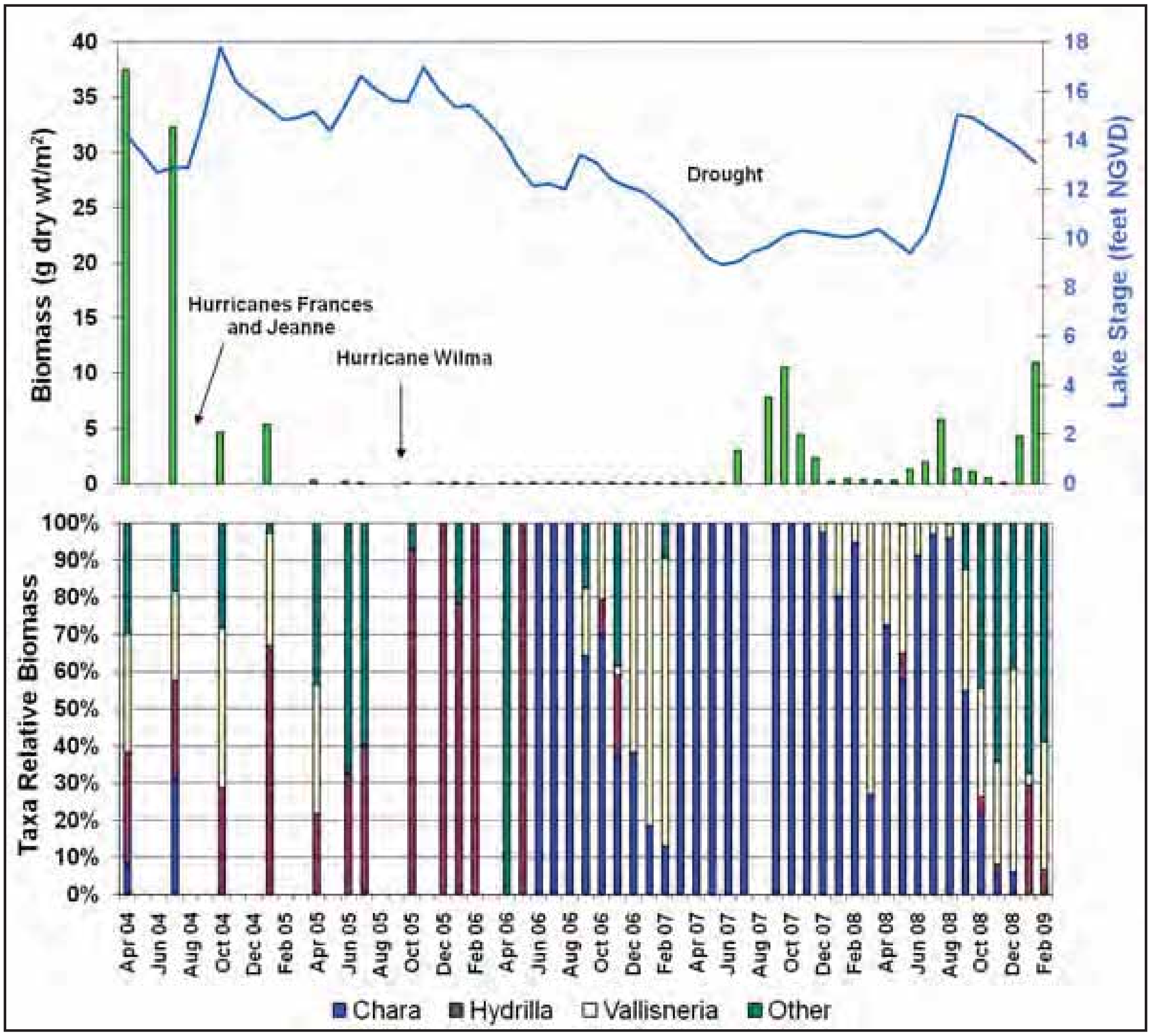


FIGURE 6-39. RESULTS FROM TRANSECT SAMPLING APRIL 2004 TO FEBRUARY 2009

Key: a) variation in SAV biomass
b) relative biomass of plant

6.6.3.1.1 2000-2001 Drought Trends

Prior to the 2000 and 2001 drought, lake water levels were high, average SAV biomass was low and the community was dominated by sparse beds of *Vallisneria* and *Potamogeton.* As water levels decreased during the drought, SAV biomass increased in response to greater irradiance at the sediment surface (***Figure 6-38a***). *Chara* became dominant, accounting for between 50 and 100 percent of SAV biomass (***Figure 6-38b***). The *Chara* beds with scattered and sparse vascular plants began colonizing in the southern end of Lake Okeechobee in summer 2000, and they expanded around the lake shore by fall 2000. This *Chara* dominant phase lasted for over a year until a strong frontal system, with winds in excess of 30 kilometers per hour, passed over the lake in a three-day period in November 2001. From November 2001 until summer 2004, the community structure changed as vascular plants increased and *Chara* dramatically declined to no more than five percent of the relative biomass. The dominant taxa became *Hydrilla*, followed by *Vallisneria*, and a mixed community of *Potamogeten*, *Najas* and *Utricularia* (***Figure 6-38b***).

6.6.3.1.2 2004-2005 Hurricane Trends

A substantial short-term perturbation occurred between July 2004 (pre-hurricanes) and October 2004 (post-hurricanes), which would not have been captured by the annual mapping effort. In the months prior to the 2004 hurricanes, average SAV biomass ranged from 19.1 grams dry weight per square meter (g dw/m^2) to 37.6 g dw/m^2 (***Figure 6-39a***). Immediately after the 2004 hurricanes, average SAV biomass declined to approximately 5.4 g dw/m^2, probably as a result of increased TSS and decreased light penetration as reflected in Secchi to TD ratios brought about by direct wind, wave, seiche (standing wave in an enclosed body of water), and lake stage impacts. Although declines over the winter period are expected due to seasonal conditions such as lower temperatures, increased turbidity and shorter photoperiod, the significant declines observed are primarily a result of long-term light deprivation related to water quality and lake stage effects. Further declines occurred as a result of Hurricane Wilma in 2005 with biomass averaging less than 1.0 g dw/m^2 where it remained throughout 2006 and into summer 2007 (***Figure 6-39***a). *Hydrilla* dominated the community immediately following the hurricanes but by summer 2006 *Chara* accounted for 100 percent of the community (***Figure 6-39b***). The vascular community, specifically *Vallisneria*, began to emerge in the innermost littoral areas during fall 2006 and winter 2007, although the biomass was low.

6.6.3.1.3 2007 - Present Trends

Since the beginning of 2007, Lake Okeechobee has experienced a longer and more severe drought than that of 2000 and 2001, with a record low of 8.82 feet msl recorded in July 2007 (***Figure 6-39a***). The initial biomass and species composition responses were similar to that observed during the previous drought. As lake levels receded, many of the nearshore areas that supported SAV dried out but the shallow water levels resulted in improved light conditions at more lakeward areas and average SAV biomass slowly increased to a peak of 9.1 g dw/m^2 in October 2007 (***Figure 6-39a***). *Chara* was again the dominant species accounting for 100 percent of the biomass and rapidly expanded across the southern and western shorelines (***Figure 6-39b***). During the winter months, *Chara* biomass declined, giving way to sparse beds of vascular species.

Unlike the previous drought, however, lake levels did not immediately increase. They remained far below the long-term mean and fluctuated in a manner that resulted in the alternate wetting and drying of areas where SAV was beginning to recover. This oscillation in lake levels may be the reason SAV biomass has not rebounded as quickly as it did in the previous drought. Alternately, promoting seed germination and seedling desiccation and mortality may have depleted the seed bank in shoreline areas, delaying recovery. If a viable seed bank remains in these areas, then a return to more typical stages, i.e. greater than 12 feet msl but less than 15 feet msl, may result in SAV conditions similar to 2003 and 2004. The full extent of SAV recovery from the current drought conditions may not be known for up to three years following the drought.

6.6.3.2 Annual Mapping

6.6.3.2.1 Current Trends

The most recent SAV map (August 2008) indicates approximately 35,834 acres of SAV in Lake Okeechobee (***Figure 6-40***). This is a substantial increase over the 28,180 acres found in late summer 2007 (***Figure 6-40***). The increase in SAV was a response to low water levels and improved light conditions caused by drought conditions during the year prior to sampling. *Chara,* a macroalgae with a competitive advantage under these conditions, became the dominant species and rapidly expanded across the southern and western shoreline accounting for 79 percent of the SAV community during the August 2008 mapping (***Table 6-6***). These spatial distributions and coverages are similar to those seen during the August 2001 mapping, which was also a drought recovery period.

TABLE 6-6. ACREAGE OF DOMINANT PLANTS IN LAKE OKEECHOBEE 2007 - 2008

Scientific Name	Common Name	Acreage	
		August 2007	August 2008
Chara	shrimp grass	27,686	28,268
Vallisneria	eel grass	494	9,405
Hydrilla verticillata	hydrilla	0	1,150
Ceratophyllum	coontail	0	477
Potamogeton	peppergrass	0	247
Najas (not depicted in figure)	water nymph	0	1,208

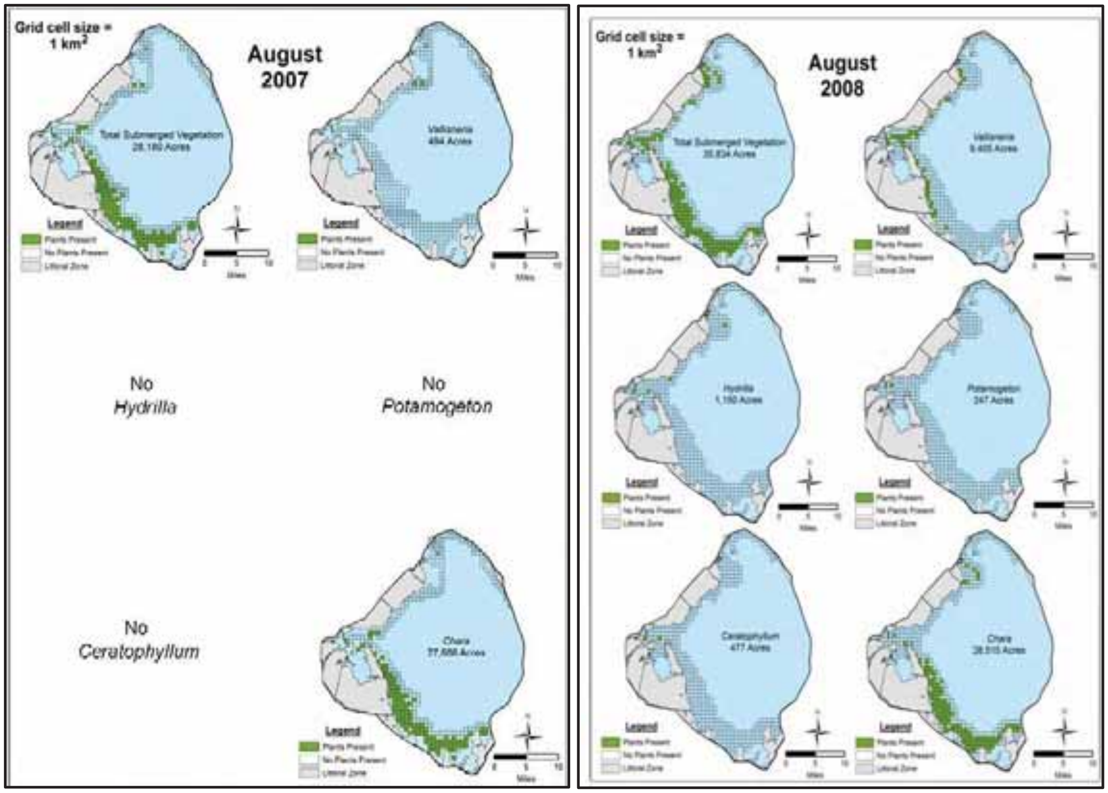


FIGURE 6-40. ANNUAL SUBMERGED AQUATIC VEGETATIVE MAPPING RESULTS FOR AUGUST 2007 AND AUGUST 2008

6.6.3.2.2 Long-term Trends

Over the past nine years, the abundance of SAV has varied from a minimum of less than 3,000 acres in 2006 to a maximum of over 54,500 acres in 2004 (***Figure 6-41a***). In that time, Lake Okeechobee experienced two separate periods of extended drought conditions: 2000 to 2001 and 2007). These droughts resulted in the drying of large areas of littoral and nearshore habitat that normally support SAV growth, but the shallow water levels also resulted in lower turbidity levels and improved light conditions at more offshore sites that do not typically support SAV. SAV slowly responded to the improved conditions in the years following the two droughts; however, the initial increase was due to dominance by *Chara*, a non-vascular species (***Figure 6-41b***).

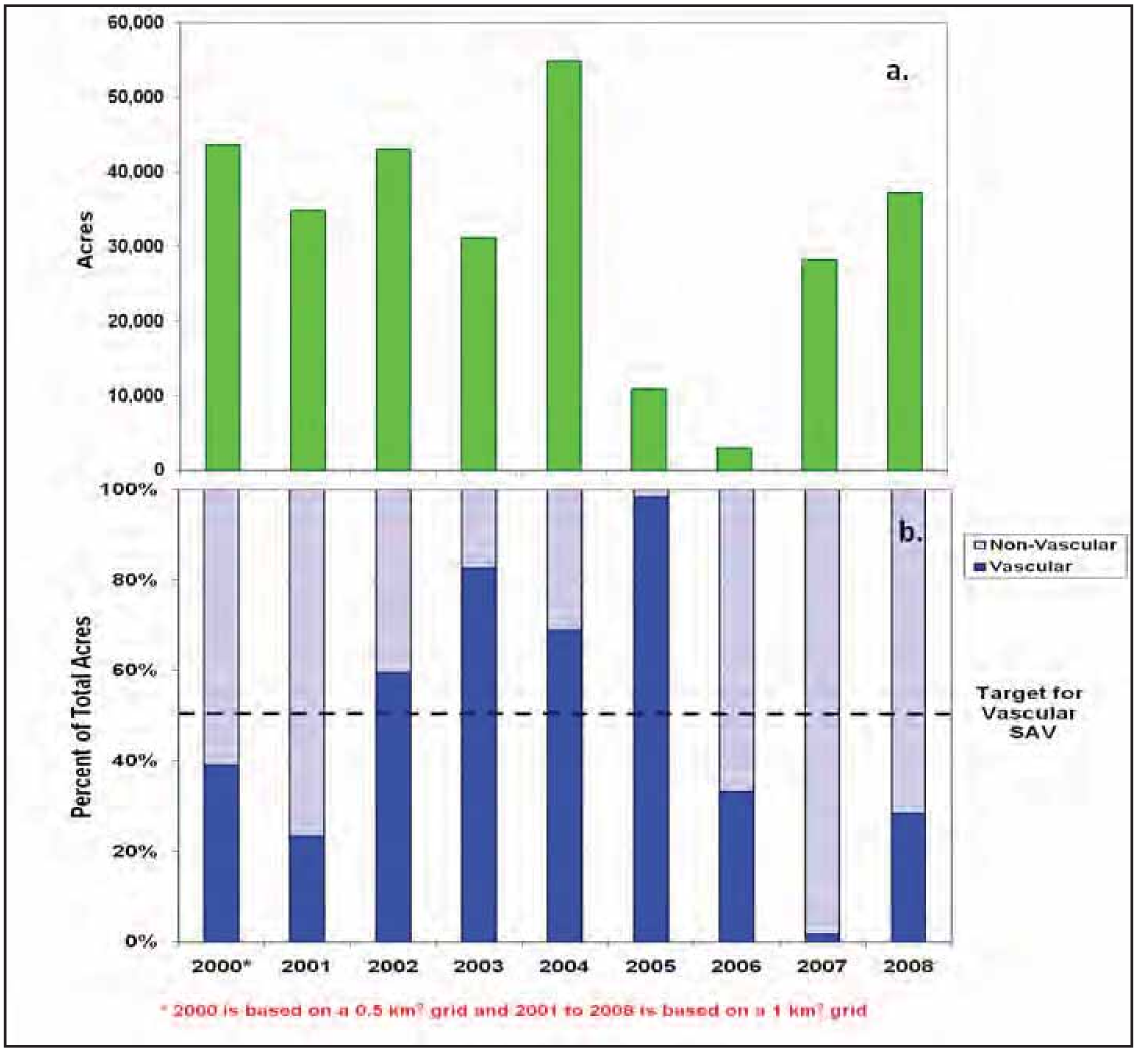


FIGURE 6-41. ANNUAL SUBMERGED AQUATIC VEGETATIVE MAPPING RESULTS 2000 THROUGH 2008

Key: a) total acres of SAV and
b) percent of total acres of vascular and non-vascular species; dotted line represents the performance measure target

A series of three hurricanes directly impacted Lake Okeechobee between the two drought events: Frances and Jeanne in 2004 and Wilma in 2005. These perturbations resulted in dramatic declines in SAV areal coverage from over 50,000 acres in 2004 to less than 3,000 acres in 2006. Physical disturbance (e.g., uprooting of plants) and prolonged turbidity resulted in the decline in SAV coverage, especially that of vascular plants such as *Vallisneria americana*, *Hydrilla verticillata*, and *Potamogeton illinoensis*. *Chara* areal coverage began rebounding in 2006 and by August 2008 both the vascular and non-vascular areal coverages and relative percentages (***Figure 6-41***) were similar to the coverage seen during summer 2001 when conditions were similar.

Information on past SAV mapping and sampling efforts are available for the following:

- Monthly transect assessments: http://www.evergladesplan.org/pm/ssr_2009/ssr_docs/hc_lake_o_sav_monthly_surveys.pdf
- Annual mapping assessments: http://www.evergladesplan.org/pm/ssr_2009/ssr_docs/hc_lake_o_sav_annual_mapping.pdf

- SAV density by species maps for 2008 and 2009: http://www.evergladesplan.org/pm/ssr_2009/ssr_docs/hc_lake_o_sav_density_by_species.pdf

The 2000 to 2007 annual distribution and spatial coverage maps for the major SAV taxa reflect the dynamic nature of SAV distribution and spatial coverage in Lake Okeechobee.

As restoration projects and other complimentary efforts improve conditions within Lake Okeechobee, detectable trends of expansion in SAV areal coverage and increased biomass are expected. Except perhaps for the impacts of major physical perturbations like hurricanes and droughts, the probability for successful utilization of this assessment tool is high.

6.7 NATIVE FISH HYPOTHESIS CLUSTER

6.7.1 Introduction and Background

Fish, besides being the most visible and sought after commodity in most water bodies, are at the trophic pinnacle among aquatic organisms, integrating the effects of both water management and basin development. Fish have been used for many years to indicate whether waters are clean or polluted, doing better or getting worse. Lake Okeechobee has supported valuable commercial and recreational fisheries estimated at times in the hundreds of millions of dollars. Among important species taken from Lake Okeechobee are white catfish (*Ameiurus catus*), bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), black crappie (*Pomaxis nigromaculatus*), and readear sunfish (*L. microlophus*).

Fish require a viable food web and require suitable habitat to avoid predation and ensure reproductive success. A conceptual model has been developed that relates the various stressors and drivers in Lake Okeechobee to responses in the native fish community (***Figure 6-42***). A performance measure has been developed for fish population density, age structure and condition (www.evergladesplan.org/pm/recover/recover_docs/et/lo_pm_fish.pdf).

6.7.2 Monitoring

Native fish populations in the littoral edge, interior marsh and open water areas of Lake Okeechobee were sampled (***Figure 6-43***) to assess relative abundance, and acquire statistics for evaluation of length frequency and length-weight relationship determination. Fish populations from 1973 to 2008 were sampled in open water areas utilizing a trawl methodology (Bull et al., 1995). Fish populations in the littoral edge and interior marsh were sampled utilizing electrofishing techniques using the procedures of Havens et al. (2005) developed for evaluation of the largemouth bass population. Methods used allowed comparison to previous Florida Fish and Wildlife Conservation Commission (FWC) surveys.

6.7.3 Results

Lake-wide electrofishing did not take place in 2007 due to low lake levels. With the increasing lake level due to Tropical Storm Fay, depths were sufficient to complete lakewide electrofishing and lakewide trawling at all sites during fall 2008. Electrofishing resulted in the capture of 4,974 fish with a combined biomass of 360.9 kilograms. Thirty fish species were represented in the catch. Four species collectively comprised 82 percent of the catch by number and were, in order of abundance, threadfin shad (*Dorosoma petenense*), gizzard shad (*Dorosoma cepedianum*), eastern mosquitofish (*Gambusia holbrooki*), and bluegill. Six species collectively comprised 78 percent of the catch by weight and were, in order of biomass, Florida gar (*Lepisosteus platyrhincus*), gizzard shad, bluegill, largemouth bass, bowfin (*Amia calva*), and redear sunfish. Trawl sampling resulted in the capture of 2,816 fish with a combined biomass of 221.2 kilograms. Seventeen fish species were represented in the catch. Three species collectively comprised 84 percent of the catch by number and were, in order of abundance, threadfin shad, white catfish, and bluegill. Three species collectively comprised 79 percent of the catch by weight and were, in order of biomass, white catfish, Florida gar, and bluegill.

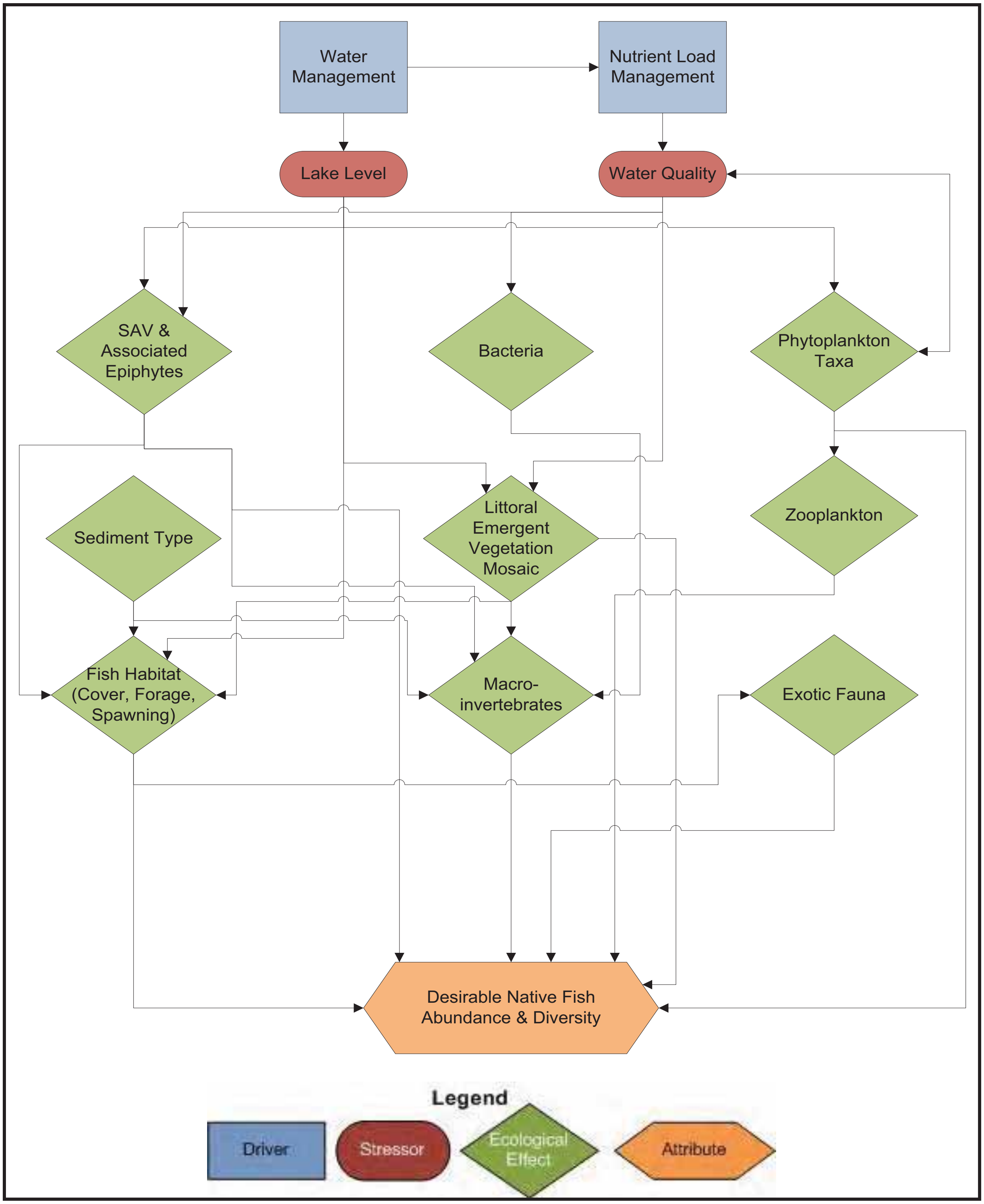


FIGURE 6-42. CONCEPTUAL MODEL FOR NATIVE FISH IN LAKE OKEECHOBEE

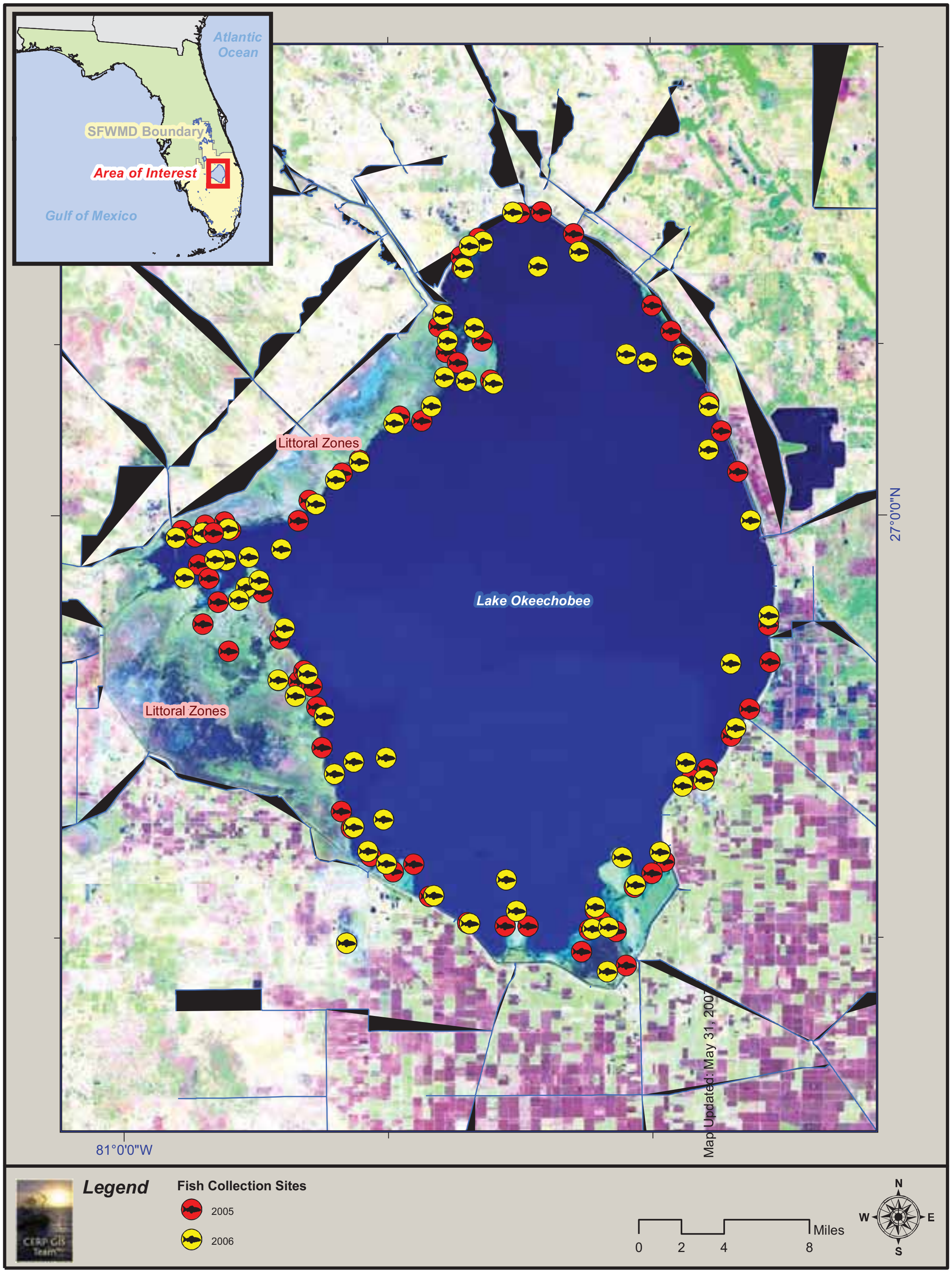
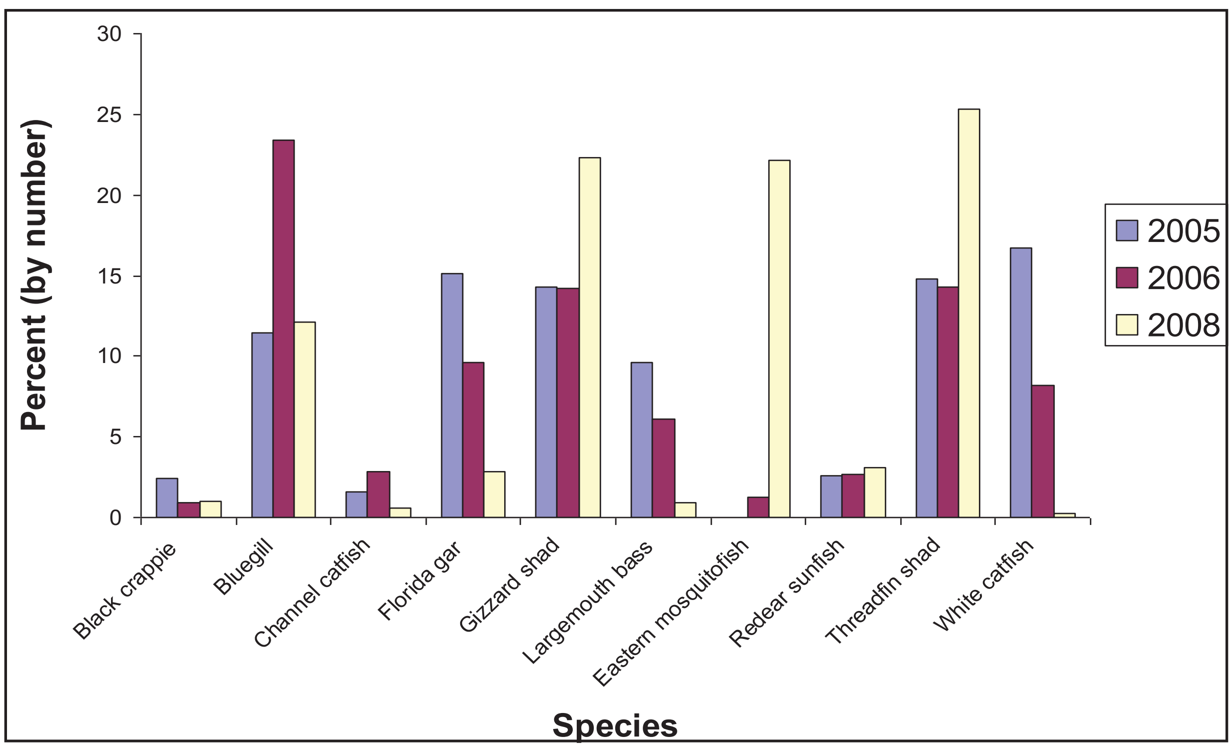


FIGURE 6-43. TYPICAL SAMPLE SITE LOCATIONS FOR NATIVE FISH SAMPLING IN LAKE OKEECHOBEE

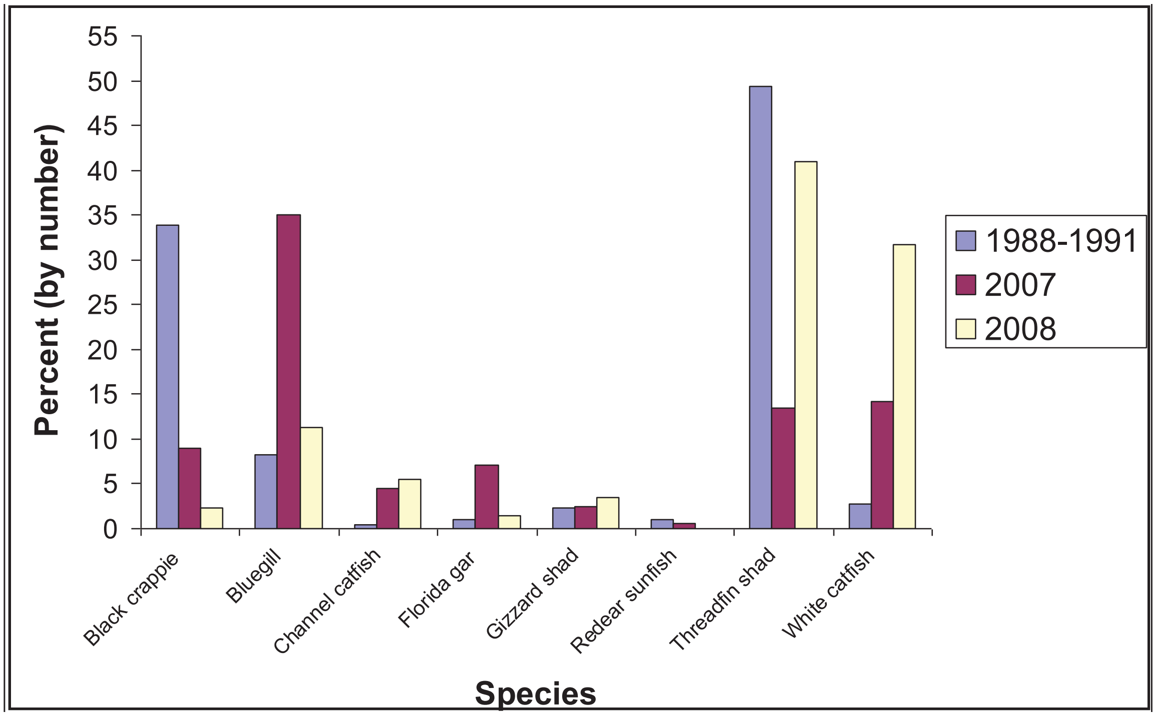
Comparison of lakewide electrofishing data from 2005, 2006 and 2008 shows a change in abundance of prey species while piscivorous species have either declined or remained steady (***Figure 6-44***). Comparison of lakewide trawl sampling data for selected dominant species shows an increase in abundance of threadfin shad and white catfish, while black crappie shows a continued decline in abundance (***Figure 6-45***). The decline in black crappie relative abundance is due to extremely poor recruitment since 2002 and the short-lived nature of the species. The decline in crappie was anticipated and was the reason the FWC implemented the ten-inch minimum in July 2008. The high abundance of white catfish in the 2008 lakewide trawl sampling was skewed by an abnormally large number captured at one site during sampling. A majority of both threadfin and gizzard shad captured in 2008 were young-of-year fish. Threadfin shad abundances have increased since 2005 but remain well below levels observed during 1988 to 1991, a period when black crappie abundances were high. Food habit analyses have shown that young-of-year shad are primary forage of adult black crappie in Lake Okeechobee. Low shad numbers are a major contributing factor to extremely low relative abundance of crappie; thus, the increase in shad observed in 2008 is an important indicator for the potential to rebuild crappie stock levels. This, along with the large number of eastern mosquito fish is key in the recovery of the largemouth bass populations of Lake Okeechobee because without prey, predatory fish populations would continue to decline.



(Graphic courtesy FWC)

FIGURE 6-44. COMPARISON OF LAKEWIDE ELECTROFISHING DATA PERCENT COMPOSITION BY NUMBER FOR SELECTED DOMINANT SPECIES

Key: 2005 (n=1,629 fish)
2006 (n=1,122 fish)
2008 (n=4,974 fish)



(Graphic courtesy FWC)

FIGURE 6-45. COMPARISON OF LAKE-WIDE TRAWL SAMPLING DATA PERCENT COMPOSITION BY NUMBER FOR SELECTED DOMINANT SPECIES

Key: 1988 to 1991 (average n=6,053 fish)
2007 (n=1,172 fish)
2008 (n=2,816 fish)

With a resurgence of these lower trophic level fish species, a project was developed to determine if the largemouth bass population in a relative small area of a much larger water body could be jump started with a supplemental stocking strategy. Stocking is generally reserved for water bodies with limited or no natural reproduction of the fish species being stocked. In January 2009, 80 adult, sexually mature, largemouth bass were collected from the Lake Okeechobee area and transported to the FWC Florida Bass Conservation Center for spawning. Genetic markers for the 80 parent fish were determined at the FWC Fish and Wildlife Research Institute. During March and April 2009, approximately 100,000 largemouth bass fingerlings (one to two inches) were stocked in the Cody's Cove area. This was the first stocking of fish to ever occur in Lake Okeechobee. Fish population sampling will occur throughout 2009 to determine the contribution of the stocked fish to the overall relative abundance of largemouth bass in the Cody's Cove area. Fin clips will be removed from collected juvenile largemouth bass for genetic analysis for determination if they were from stocking or in-lake natural reproduction sources. Food habit analyses will be conducted on a subsample of collected juvenile largemouth bass to determine if variation in food habits occurs between hatchery stocked and natural spawned fingerlings.

An additional important driving factor affecting Lake Okeechobee's fish population density and structure is the reduction in numbers of chironomidae macroinvertebrates (***Figure 6-46***). Chironomid larvae are the primary food source of juvenile black crappie and the decline in the former is another causative factor, along with the decline in threadfin shad, explaining the decline of black crappie. Bluegill, also known as bream or brim, feed on very small fish and invertebrates. In 2005 and 2006, bluegill abundance decreased in comparison to the 1987 to 1991 data by 94 and 92 percent, respectively, which mirrors the decline in invertebrates as their direct prey and that of many of the smaller fish upon which they feed. However, concern regarding these precipitous declines in black crappie population must be tempered by observing that 1986 to 1990 was an unusually productive period (***Figure 6-47***), and, accordingly, the 1987 to 1991 dataset was biased high. Although the 2005 to 2007 timeframe denotes the lowest catch rate on record, other periods of time have been similarly poor. Nevertheless, the preceding clearly illustrates the intertwined relationships among all the lake health attributes (e.g., SAV, water quality, lake stage, macroinvertebrates) and demonstrates the necessity to assess and manage Lake Okeechobee from the widest holistic perspective of balanced ecosystem function

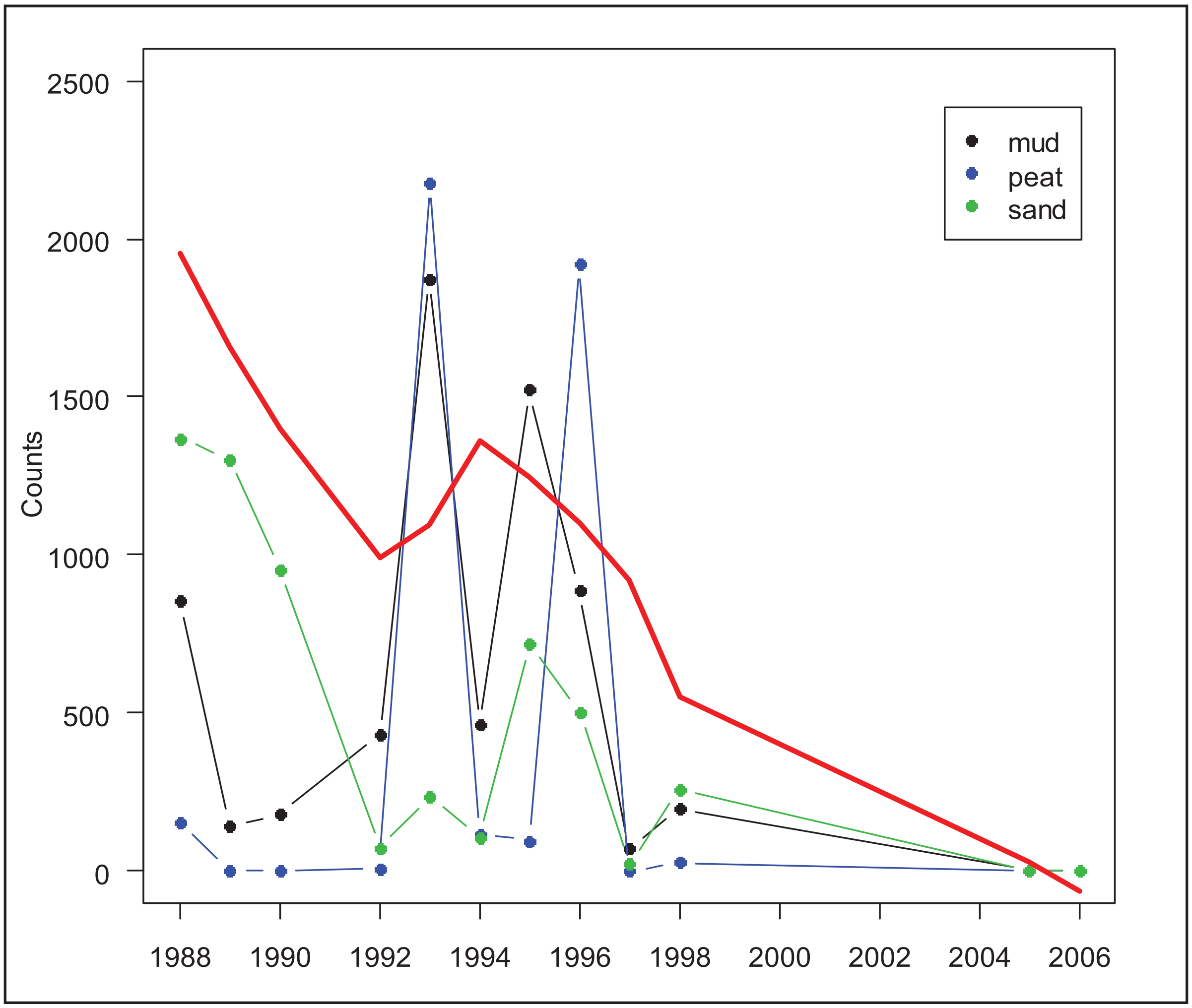


FIGURE 6-46. COUNTS OF CHIRONOMIDAE SAMPLED PER YEAR PER SUBSTRATE TYPE

Note: redline depicts locally weighted scatter plot smoothing (LOESS) trend line for the the three substrates combined

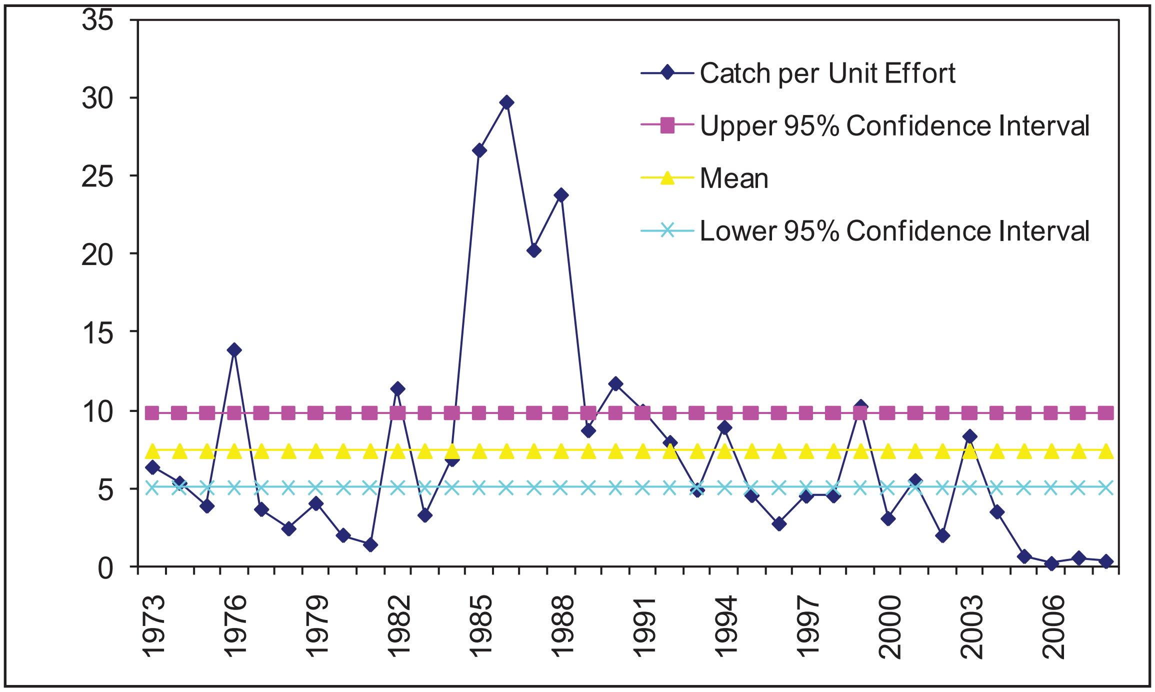


FIGURE 6-47. CATCH RATE OF BLACK CRAPPIE COLLECTED 1973 TO 2008 FROM LAKE OKEECHOBEE

6.7.4 Summary

The abundance and diversity of the native fish population in Lake Okeechobee is dependent on a variety of related factors. Most ostensible among these factors are SAV as habitat and the macroinvertebrate community as the primary food web. Water and basin management directly affect lake stage and lake water quality, with consequence to SAV coverage and algal bloom severity and duration. The fish population evidences considerable year-to-year variation, with periodic very good years interspersed with years less so. Factors related to lake stage and nutrient loading may coincide with changes in the fish community. Extremely high or low lake stage may negatively affect emergent and SAV coverage, thereby reducing the amount of available fish habitat. If lake stages are more frequently within the desirable range of 12.5 – 15.5 feet emergent and SAV coverage might be maximized, thereby increasing the amount of habitat for fish and fish prey to utilize. However, continuous excessive nutrient loading may indirectly negatively impact the fish communities by shifting their macroinvertebrate prey-base from preferred taxa such as chironomids (non-biting midges) to one dominated by less utilized oligochaete (annelid worm) taxa. Recent downturns in fish community health following three hurricanes in 2004-2005 are worrisome, but additional data will be necessary to determine if these are cyclic events.

6.8 MACROINVERTEBRATES HYPOTHESIS CLUSTER

6.8.1 Introduction and Background

Freshwater invertebrate communities are extremely sensitive to existing water quality conditions, and reflect a lake's trophic status because they are unable to escape perturbations (Warren et al., 2007). Species composition, absolute abundance, relative abundance, diversity, species richness and evenness are metrics commonly used to evaluate the ecological condition of lakes. As eutrophication progresses, macroinvertebrate species richness and diversity are reduced, while the community composition shifts to one dominated by pollution-tolerant taxa. As a lake becomes increasingly eutrophic, macroinvertebrates that require higher levels of dissolved oxygen, such as many mussels (*Pelecypoda*), mayflies (*Ephemeroptera),* caddisflies (*Trichoptera),* dragonflies (*Anisoptera*), and damselflies (*Zygoptera*), are eliminated. The invertebrate communities then become dominated by groups of species physiologically adapted to withstand high degrees of organic loading and extended periods of low (less than 4.0 ppm) dissolved oxygen (Brinkhurst, 1974; Warren et al., 2007). If Lake Okeechobee becomes hyper eutrophic, all but the most tolerant segmented worm species, those within Oligochaeta, may be eliminated (Brinkhurst, 1974; Wetzel, 1983). Since macroinvertebrates are an important component of freshwater food webs, adverse changes in the macroinvertebrate community negatively impact fish and other higher trophic level organisms. Principle drivers affecting the quality of the macroinvertebrate community in Lake Okeechobee are water management activities and excessive nutrient loading (***Figure 6-48***). A CERP system-wide performance measure has been developed for macroinvertebrates. The documentation for this measure can be found online at www.evergladesplan.org/pm/recover/recover_docs/et/lo_pm_macroinvertebrate.pdf.

Restoration aims to augment the current infrastructure such that the lake stage can be better controlled, allowing the marsh to be better stabilized, with those improvements being translated and realized in the macroinvertebrate community within the lake. Excessive nutrient loading from Lake Okeechobee's basin as well as internal loading from the sediments causes algal blooms, which augment undesirable characteristics of the lake's sediment in which the macroinvertebrate community must survive. Restoration efforts to decrease external loading, and, to the extent feasible, internal loading, would improve bottom conditions with the attendant improvement in macroinvertebrate community structure.

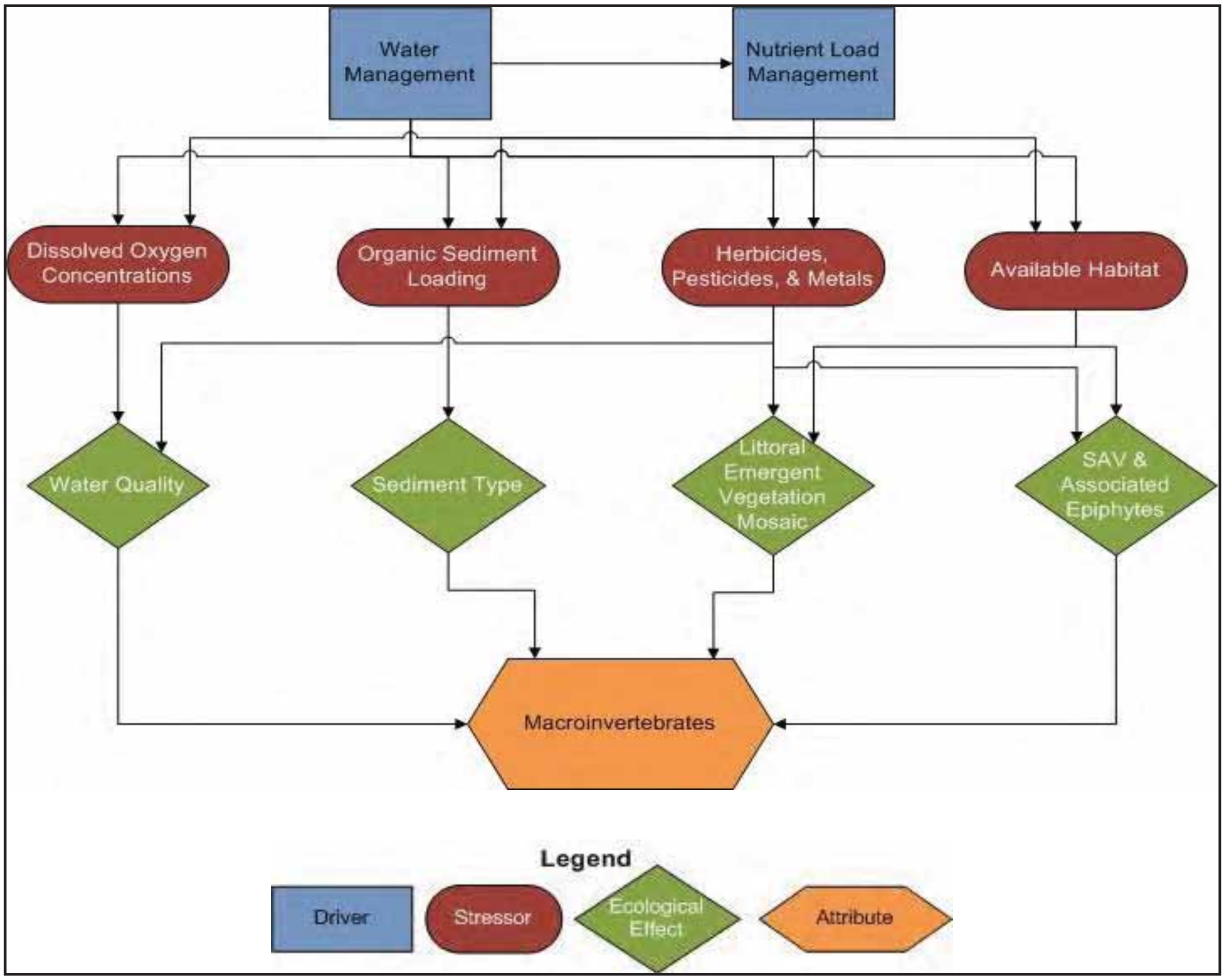


FIGURE 6-48. CONCEPTUAL MODEL FOR MACROINVERTEBRATES IN LAKE OKEECHOBEE

6.8.2 Monitoring

Macroinvertebrates in the pelagic zone of Lake Okeechobee have been intermittently monitored since 1969 (Warren et al., 1995). Currently, monitoring is being conducted at 18 synoptic sites located in peat, mud and sand sediments of the lake (***Figure 6-49***). These are the same sites utilized during previous monitoring efforts. A more detailed description of monitoring methods can be found in the 2007 SSR (RECOVER, 2007) available online at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

6.8.3 Results

6.8.3.1 Overall Lake Community

The 2005 to 2008 pelagic zone data indicated that a total of 118 individual aquatic invertebrate taxa representing 28 major taxonomic groups were collected from Lake Okeechobee. Segmented worms (Oligochaetes) and non-biting larval midges (Chironomids) accounted for 73 of the 118 taxa, were numerically dominant, and comprised 64 percent (lakewide mean = 7,869 individuals per square meter [m^2]) of the total number of organisms collected during the study year. Segmented worms accounted for approximately 41 percent and midge fly larvae accounted for 24 percent of the total number of organisms. Only three other groups (Pelecypoda, aquatic Acari and Amphipoda) accounted for greater than five percent of the total organisms. Other groups typically dominant in Florida lakes, gastropods, isopods, ephemeropterans and trichopterans, accounted for less than two percent each of the pelagic zone abundance total during the 2005 to 2008 study period.

The most abundant macroinvertebrate taxon, at 1,351/m^2 (17.2 percent), was the tubificid worm *Limnodrilus hoffmeisteri*. This worm has been among the most abundant macroinvertebrate taxon in Lake Okeechobee since 1987 and is well adapted to high organic loading. *Corbicula fluminea,* an exotic Asian clam, was the second most abundant at 965/m^2 (12.3 percent). Several taxa that were numerically important during the 1987 to 1997 study period, such as *Pyrgophorus platyrachis, Viviparus georgianus* and *Melanoides* sp (snails); *Hydalella azteca* and *Corophium lousianum* (amphipods); *Cyathura polita* and *Cassidinidea ovalis* (isopods); *Nectopsyche* and *Oecetis* (caddisflies); and several chironomids, were either rare or absent during the 2005 to 2008 study period. The detritivorous chironomid *Chironomus crassicaudatus*, an important prey item for game fish during the past study, also was found at very low densities (0.3/m^2) during the 2005 to 2008 study period.

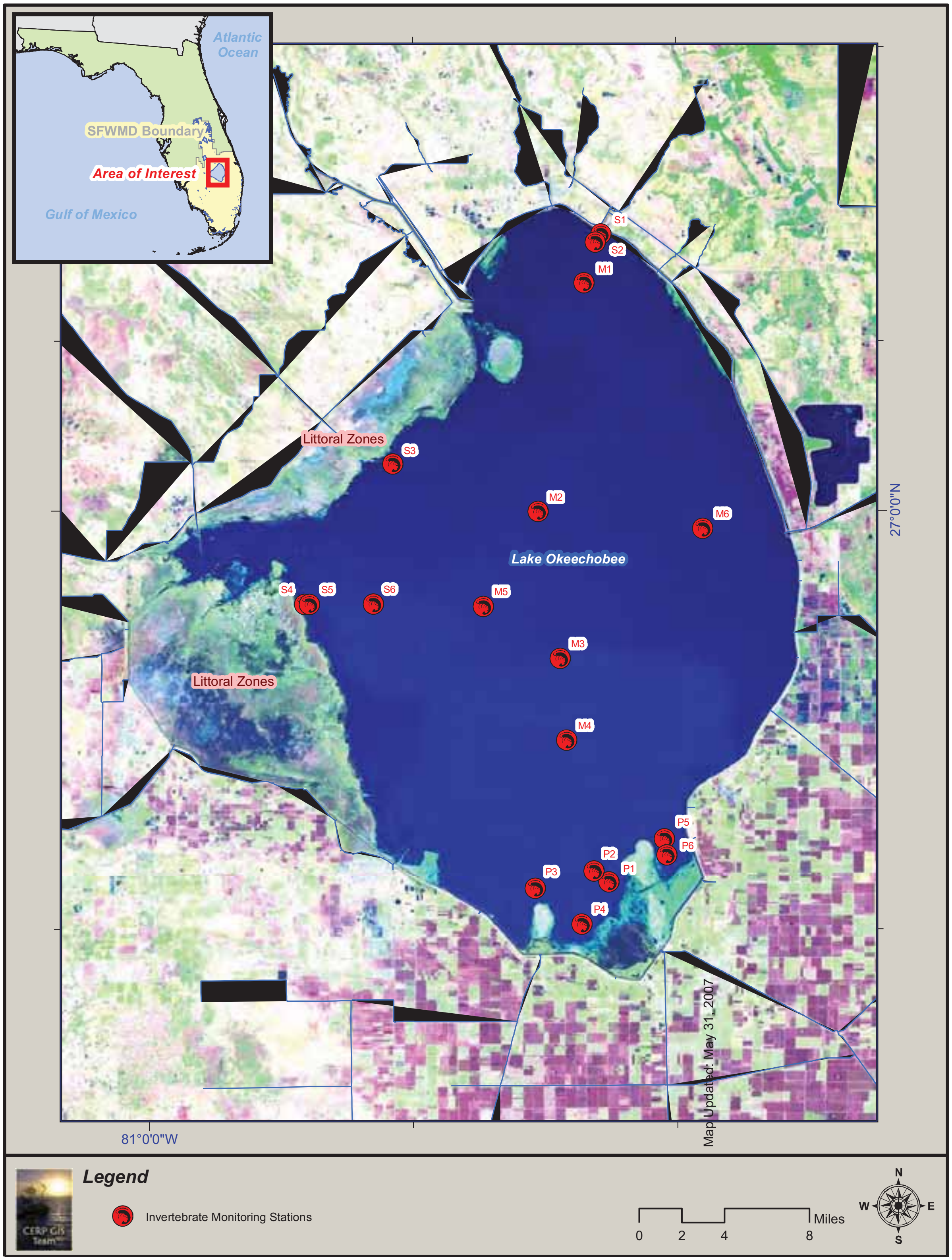


FIGURE 6-49. MACROINVERTEBRATE SAMPLING SITES IN LAKE OKEECHOBEE

Mean species richness (8.8), diversity (1.9), evenness (0.67) and total density were in the low to average range relative to other Florida lakes and Lake Okeechobee between 1987 and 1997 (Warren et al., 2008). These measures of macroinvertebrate community health increased significantly ($p<0.05$) over the three-year study period (***Table 6-7***). Mean evenness did not increase significantly. Overall, it declined from 0.69 to 0.66, but this decline was not significant. While increases in several undesirable taxa (e.g., *L. hoffmeisteri, Corbicula fluminea*) were observed during each year or during successive years of the 2005 to 2008 study period, densities of taxa important for the Lake Okeechobee food web increased. These taxa included gastropods, amphipods, isopods, ephemeropterans (mayflies) and chironomids.

TABLE 6-7. PELAGIC BENTHIC MACROINVERTEBRATE COMMUNITY HEALTH INDICES 2005 - 2008[1]

Descriptor	Study Year 2005-2006	2006-2007	2007-2008
Total taxa	48	68	94
Mean species richness	5.7^{a}	8.9^{b}	11.8^{c}
Mean diversity	1.54^{a}	1.88^{b}	2.18^{c}
Mean evenness	0.69^{a}	0.66^{a}	0.66^{a}
Mean total organisms per m^2	$3{,}338^{a}$	$7{,}591^{b}$	$12{,}678^{c}$

[1] Means with the same superscript are not significantly different (ANOVA, p□0.05). Total organisms per m^2 were transformed by LOG(X+1). Taken from Warren et al., 2008.

6.8.3.2 Habitat and Seasonal Influences

Bottom substrate type is the primary determinant of invertebrate community structure in the Lake Okeechobee sublittoral zone (Warren et al., 1995). Three primary benthic habitat regions characterize the sublittoral zone: mud, sand and peat (Reddy, 1993) (***Figure 6-50***).

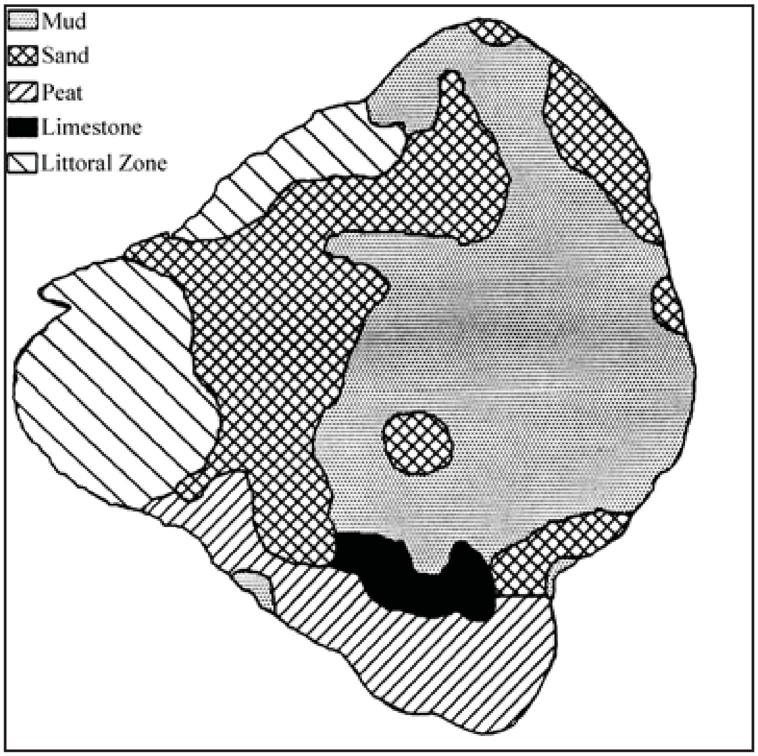


FIGURE 6-50. MAJOR BOTTOM TYPES OF LAKE OKEECHOBEE

The 2005 to 2008 data indicate that the benthic communities associated with the three habitat types continued to remain taxonomically and structurally distinct, but all three also exhibited dynamic change. The macroinvertebrates in all three habitat zones were numerically dominated by taxa that are collectors or filtering detritivores as adults. Total taxa richness and mean species richness and diversity per sample continued to be lowest in the mud zone. The mean number of total organisms per square meter increased significantly ($p \leq 0.05$) in the mud habitat every year and increased by a factor greater than two over the course of the three-year study period (***Table 6-8***). However, the mud habitat macroinvertebrate diversity and evenness declined over this period. These dynamics were attributable to large, statistically significant (analysis of variance [ANOVA], $p \leq 0.05$) increases in the densities of two segmented worm species, *L. hoffmeisteri* and *I. templetoni*, which together accounted for over 70 percent of all organisms collected from the mud habitat each year (***Table 6-9***). Increases in mud habitat taxa abundances during the three-year study period were attributable primarily to the return of other segmented worm taxa that were present in the mud zone during the 1987 to 1997 study period, but were absent during 2005 and 2006.

TABLE 6-8. PELAGIC BENTHIC MACROINVERTEBRATE COMMUNITY HEALTH INDICES 2005 - 2008[1]

Descriptor	Study Year		
	2005-2006	**2006-2007**	**2007-2008**
Mud Habitat			
Total taxa	15	23	22
Mean taxa richness	3.3^{a}	4.3^{b}	4.2^{b}
Mean diversity	1.52^{a}	1.36^{a}	1.24^{a}
Mean evenness	0.71^{a}	0.66^{ab}	0.61^{b}
Mean total organisms per m^2	$1{,}522^{a}$	$2{,}741^{b}$	$4{,}151^{c}$
Sand Habitat			
Total taxa	35	43	61
Mean taxa richness	7.4^{a}	8.5^{a}	12.5^{b}
Mean diversity	1.79^{a}	1.97^{a}	2.38^{b}
Mean evenness	0.65^{a}	0.66^{a}	0.67^{a}
Mean total organisms per m^2	$5{,}841^{a}$	$6{,}081^{a}$	$10{,}058^{b}$
Peat Habitat			
Total taxa	34	54	64
Mean taxa richness	6.3^{a}	14.0^{b}	18.8^{c}
Mean diversity	1.67^{a}	2.32^{b}	2.94^{c}
Mean evenness	0.71^{a}	0.66^{a}	0.71^{a}
Mean total organisms per m^2	$2{,}652^{a}$	$13{,}952^{b}$	$23{,}825^{c}$

1 Means with the same superscript are not significantly different (ANOVA, p□0.05).

Total organisms per m2 were transformed by LOG(X+1). Taken from Warren et al. 2008.

TABLE 6-9. ABUNDANCE AND PERCENT COMPOSITION OF NUMERICALLY DOMINANT (>5 PERCENT OF TOTAL ORGANISMS) INDIVIDUAL TAXA IN THE THREE BENTHIC HABITATS 2005 - 2008[1]

Descriptor	Study Year					
	2005-2006		2006-2007		2007-2008	
	Abundance (no./m^2)	Percent Composition	Abundance (no./m^2)	Percent Composition	Abundance (no./m^2)	Percent Composition
Mud Habitat						
Limnodrilus hoffmeisteri	895[a]	58.8	1,356[a]	49.5	2,710[b]	65.3
Ilyodrilus templetoni	215[a]	14.1	606[b]	22.1	847[b]	20.4
Stephensoniana trivandrana	58[a]	3.8	457[a]	16.7	307[a]	7.4
Coelotanypus sp	95[a]	6.2	74[a]	2.7	96[a]	2.3
Gammarus nr. *tigrinus*	7[a]	0.5	100[b]	3.6	22[a]	0.5
Sand Habitat						
Limnodrilus hoffmeisteri	1,928[a]	33.0	1,664[a]	27.4	2,941[a]	29.2
Corbicula fluminea	577[a]	9.9	1,631[b]	26.8	1,532[b]	15.2
Haber speciosus	1,740[a]	29.8	311[a]	5.1	144[a]	1.4
Djalmabatista pulchra	1[a]	<0.1	0[a]	0.0	1,322[b]	13.1
Stephensoniana trivandrana	257[a]	4.4	274[b]	4.5	578[b]	5.7
Cladotanytarsus sp.	4[a]	<0.1	432[b]	7.1	653[c]	6.5
Gammarus nr. *tigrinus*	212[a]	3.6	717[b]	11.8	101[a]	1.0
Peat Habitat						
Corbicula fluminea	364[a]	13.7	4,318[b]	30.9	212[a]	0.8
Cladotanytarsus sp.	0[a]	0.0	20[b]	0.1	4,701[c]	19.7
Hydracarina	303[a]	11.4	3,283[b]	23.5	1,003[c]	4.2
Pseudochironomus sp.	0[a]	0.0	0[a]	0.0	3,955[b]	16.6
Nais variabilis	0[a]	0.0	890[b]	6.4	2,929[b]	12.3
Gammarus nr. *tigrinus*	207[a]	7.8	1,157[b]	8.3	1,462[a]	6.1

1 Means with the same superscript are not significantly different (ANOVA, p□0.05). Total organisms per m2 were transformed by LOG(X+1). Taken from Warren et al. 2008.

Both the sand and peat habitat zone invertebrate communities were characterized by large annual increases in mean total organisms per m^2, total taxa collected, mean taxa richness, and mean diversity (***Table 6-8***). Most of these increases were statistically significant (ANOVA, $p \leq 0.05$). The increases in total organisms per m^2 can be attributed to rapid growth in the populations of benthic taxa whose numbers may have been negatively impacted by sediment resuspension resulting from the 2004 and 2005 hurricanes.

The macroinvertebrate communities appear to have responded positively to the drought conditions that followed the hurricanes. These drought conditions were characterized by low water levels, low turbidity, improved water column light penetration, low TP, greatly reduced allochthanous organic loading, and transport of mud sediments from the center of the pelagic zone. The taxa that responded favorable to the drought conditions included multi-voltine midges such as *Cladotanytarsus* sp., *Pseudochironomus* sp. and *Djalmabatista pulchra*, as well as the segmented worms *L. hoffmeisteri* and *Nais variabilis*, the Asian clam *C. fluminea*, and water mites (Hydracarina). Added benthic habitat complexity provided by expansion of the macro-alga *Chara* during 2007 and 2008 may have been partially responsible for increases in density and taxa richness in the peat-associated community.

6.8.3.3 Comparison of 2005-2008 and 1987-1997 Study Periods

Data from the 2005 to 2008 study were combined with the FWC 1987 to 1997 dataset to enable a more substantive pre-CERP baseline to be examined. Conclusions drawn from the 1987 to 1997 and 2005 to 2008 data comparisons should be tempered due to the data gap that encompassed several weather-related disturbances such as a drought in 2000 and 2001, high lake stages during 1998, 1999, 2002 and 2003 and hurricane impacts during 2004 and 2005.

Nonmetric multiple dimensional scaling (Ludwig and Reynolds, 1988; Jongman et al., 1995; McCune et al., 2002) ordination analysis was used to examine patterns in the long-term macroinvertebrate community data. Density data from the 1987 through 2008 data sets were converted to percents and arcsine transformed. The data were then sorted by habitat zone (mud, sand or peat), season (August, October and November-wet, February and May-dry), and sample year. Mean values were calculated for each habitat zone by season and year (e.g., 15 mud samples collected in spring 1987). The relativized data was analyzed for assemblage differences using the Sorenson (Bray-Curtis) distance measure in PC-ORD version 4 (McCune and Mefford, 1999). The ordination analysis plotted axis scores for each site and species and the site scores were compared to corresponding water quality data (e.g., water depth, conductivity, dissolved oxygen and temperature) for possible correlations. Extreme dominance (71 percent of total abundance) of February 1997 sand habitat sites by the segmented worm *Haber speciosus* resulted in an outlier that was removed from the analysis.

The resulting ordination suggested that there were differences in the macroinvertebrate communities based on substrate and season (***Figure 6-51***). Taxonomic changes over the entire study period were least at the mud sites and greatest at the peat sites. Differences in the wet season species assemblages compared to those in the dry season for the mud and sand sites were small, while the seasonal differences were more pronounced at the peat sites. Of the individual taxa, 15 were associated with mud sites, six with both mud and sand sites, 46 with sand, 13 with sand and peat, and 90 with the peat sites.

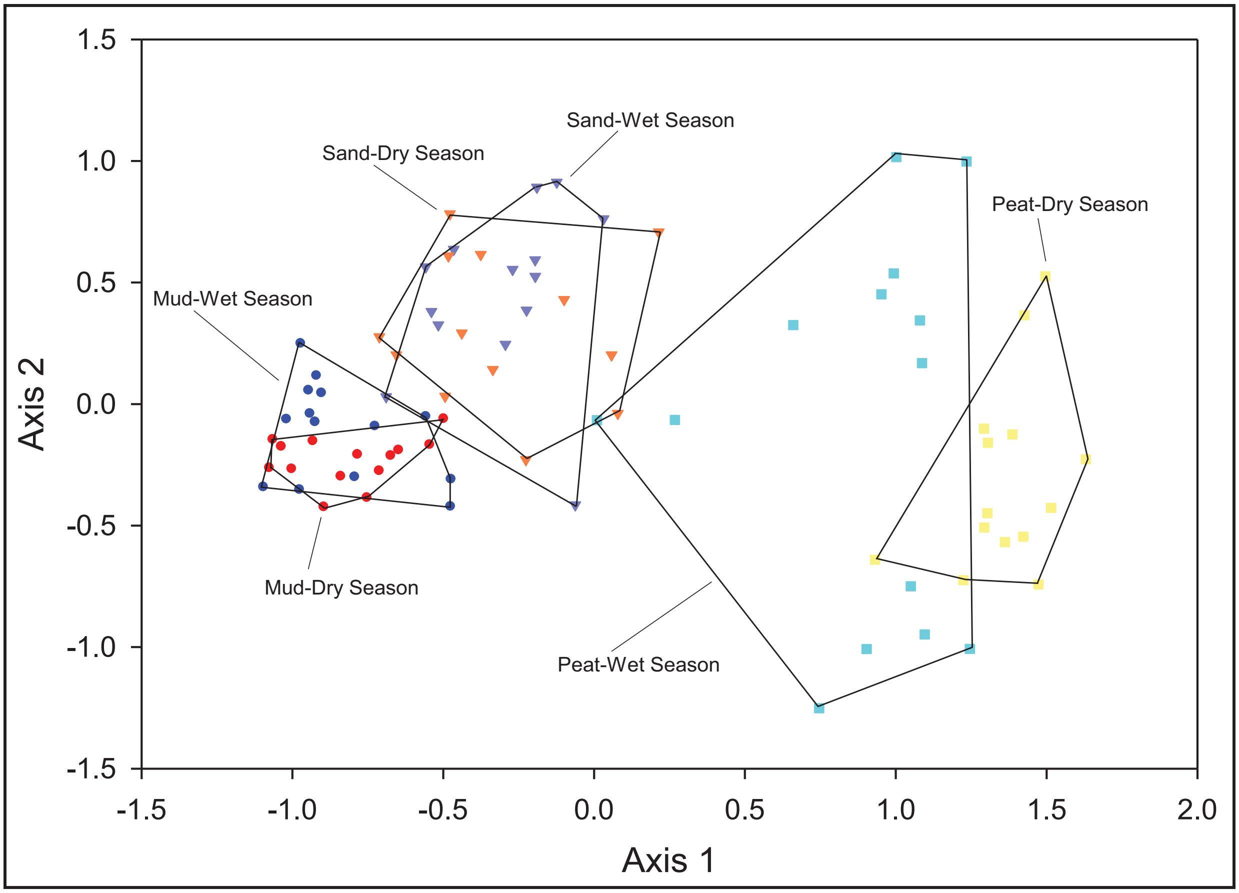


FIGURE 6-51. NON-METRIC MULTI-DIMENSIONAL SCALING OF LAKE OKEECHOBEE RELATIVIZED INVERTEBRATE MEAN TAXA DENSITIES BY SUBSTRATE TYPE (MUD, SAND OR PEAT) AND SEASON (WET OR DRY)

To better illustrate temporal pattern variability, date labels were used for the same ordination (***Figure 6-52***). Annual changes in species composition were least apparent in the mud habitat and greatest in the peat habitat; however, the amount of separation among the 1987 to 1997 and 2005 to 2008 sample dates was small.

Pearson correlations between depth at site/lake stage and axis scores revealed weak but significant ($p=0.05$) correlations. A significant negative correlation was observed between Axis 1 scores and site depths. A significant negative correlation was observed between Axis 1 scores for sand and mean lake stage. A significant negative correlation was observed between Axis 2 scores for peat and mean lake stage. All remaining correlations were not statistically significant ($p>0.05$).

The Lake Okeechobee pelagic benthic macroinvertebrate assemblages exhibited rapid and considerable change during the 2005 to 2008 study period. Macroinvertebrate communities during this time were likely influenced by the weather disturbances (e.g., hurricanes, prolonged drought) that impacted the lake. Wave action during the storms scoured and displaced bottom sediments, and undoubtedly severely impacted embenthic and epibenthic species at all sites. The extended period of high turbidity (approximately six months) following the storms limited primary and secondary production in sublittoral habitats where light typically penetrates to the bottom and indirectly structures invertebrate communities. The pre-hurricane macroinvertebrate communities were not monitored, so it is difficult to say how much change in the community structure was influenced by the preceding weather events. However, data collected immediately following the storms showed that taxa richness, diversity and densities of many important taxa, including the pollution-tolerant worm taxa that dominated the mud sites, were at the lowest levels measured since 1987. The low macroinvertebrate index scores may have been the result of the hurricane impacts on the lake. An additional potential consequence of the storm impacts was disappearance of key species, such as the tubicolous non-biting midge *Chironomus crassicaudatus*. *C. crassicaudatus* was the third most abundant taxon present in the mud zone during 1987 to 1997, but was almost completely absent from all collections during 2005 to 2008. Studies of the black crappie diet preferences in Lake Okeechobee during the early 1990s found *C. crassicaudatus* to be the principal prey item during summer months. Permanent elimination of *C. crassicaudatus* from the food web, therefore, could have consequences to an economically important fish species. Other species with similar ecological value were either absent or rare during the 2005-2008 study period. These included the amphipods *Gammarus* nr. *tigrinus* and *Corophium louisianum*, the isopods *Cyathura polita* and *Cassinididea ovalis*, and the snail *Viviparus georgianus.*

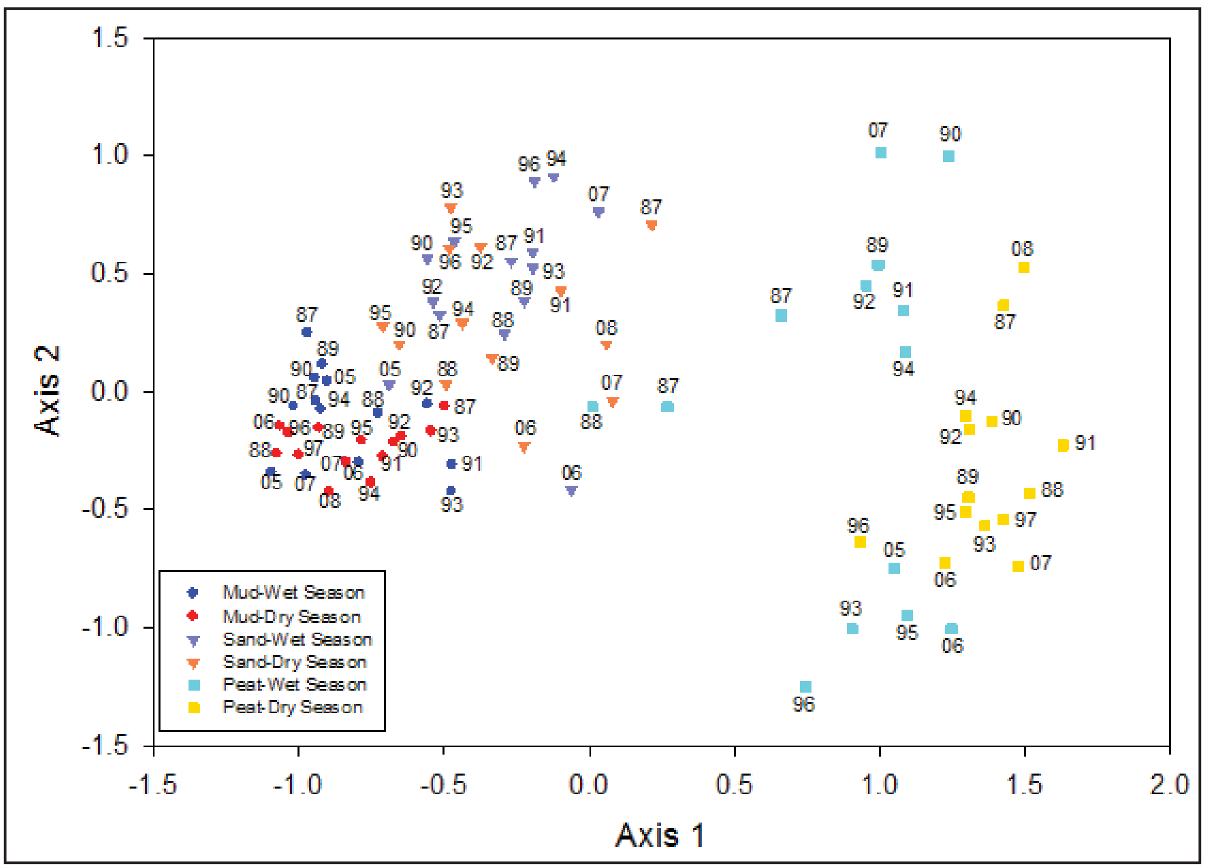


FIGURE 6-52. NON-METRIC MULTI-DIMENSIONAL SCALING OF LAKE OKEECHOBEE RELATIVIZED INVERTEBRATE MEAN TAXA BY SUBSTRATE, SEASON AND YEAR

The prolonged drought that began during the second half of 2006 resulted in lake levels dropping to an historic low monthly mean of 8.8 feet msl in July 2007, greatly reduced turbidity, and expansion of the macro-alga *Chara* in shallow areas of the nearshore region of Lake Okeechobee. The rapid increases in macroinvertebrate pioneer taxa densities at shallower sand and peat sites suggested that environmental conditions conducive to macroinvertebrate growth returned to Lake Okeechobee. The chironomids *Cladotanytarsus* sp. and *Pseudochironomus* sp. and the naided worm *Nais variabilis* are examples of multi-voltine taxa that were rapidly reproducing while water levels were low. Populations of these taxa were very low during 2005 and 2006, but rapidly increased shortly thereafter, illustrating that, despite problems related to eutrophication, invertebrate communities of Lake Okeechobee have retained some level of viability.

However, despite some signs of recovery of the macroinvertebrate community at the sand and peat sites during 2007 and 2008, the benthic macroinvertebrate community as a whole still reflects a less diverse assemblage comprised of a higher percentage of pollution-tolerant taxa, relative to that during 1987 to 1997. The tubificid worms *Limnodrilus hoffmeisteri* and *Ilyodrilus templetoni* have consistently dominated the mud zone community; however, their densities, as well as the densities of most other dominant taxa of the mud zone, appear to have declined by approximately an order of magnitude between the 1987 to 1996 study period and the 2005 to 2008 study period.

The sand sites appear to be more hospitable for macroinvertebrate growth relative to the mud sites. The sand sites appear to provide more heterogeneity for the benthic macroinvertebrates, as evidenced by higher annual and seasonal variation in assemblage composition and higher species richness. The majority of species associated with sand habitat have an affinity for fine substrates and are tolerant of low dissolved oxygen; thus the sand sites are not favorable for pollutant-intolerant taxa growth. The species assemblages inhabiting the sand habitat during the 2005 to 2008 study period were more similar to the peat habitat assemblages compared to previous study years. This response probably reflects that some sand sites were characterized by an intrusion of organic material as a consequence of the hurricane passages.

The coarse texture of the peat sediments zone provides the most heterogeneous habitat for macroinvertebrates, as evidenced by the highest annual and seasonal variation in assemblage composition and the highest values for total and mean taxa richness relative to the other substrate sites. The peat sites are typically inhabited by the most densely populated, species rich and diverse benthic community of the entire pelagic region. It is apparent, however, that despite the increases in some populations during 2007 to 2008, species richness, total organisms and densities of the individual dominant species within this zone have declined since 1987 to 1996. The majority of the species currently associated with the peat habitat are detritivores that have an affinity for fine substrates, and many are tolerant of low levels of dissolved oxygen. The higher annual and seasonal variation in assemblage composition relative to the mud and sand sites reflects the more heterogeneous habitat provided by the peat sediments.

Overall, the results from 2005 to 2008 show that benthic invertebrate communities of the Lake Okeechobee pelagic region have clearly declined in quality relative to the 1987-1997 study period. The poor community quality that characterized the 2005 to 2006 study year appears to be related to impacts of the 2004 and 2005 hurricanes. Taxonomic composition, densities, species richness and diversity documented during 2007 to 2008 suggest that some segments of the macroinvertebrate community are recovering and, at the sand and peat sites, appear to be progressing through successional changes indicative of response to large-scale weather disturbance. If this is the case, the 2005-2008 data may not be suitable to characterize as a pre-CERP baseline. If important prey taxa such as *Viviparus georgianus*, *Corophium louisianum*, *Cyathura polita* and *Chironomus crassicaudatus* continue to be absent or rare in the macroinvertebrate communities, the likelihood of a complete recovery of the lake's invertebrate fauna, even after CERP projects are in place, may be substantially decreased.

6.8.4 Summary

The macroinvertebrate community in Lake Okeechobee has continued to reflect the eutrophic conditions that preceded initiation of benthic sampling. Although there is an apparent large variation in Lake Okeechobee's condition year to year, the range of that variation has continued to swing somewhere between moderate and poor. The macroinvertebrate community monitoring reflects that variability, but continues to portray a stressed community not amenable to long-term establishment of sensitive, pollution-intolerant species. Against this backdrop, a high probability exists that improving conditions within Lake Okeechobee would result in a demonstrably improving benthic community. Such improvements in the foundation of the food web would foment positive effects on fisheries and the ecological health of the lake as a whole. After CERP projects are online and nutrient and sediment loads to Lake Okeechobee are reduced, the dominance by taxa tolerant to organic pollution is anticipated to decrease, while numbers of nutrient and organic intolerant taxa, species diversity, species richness and evenness of distribution are expected to increase.

Future plans include sampling the macroinvertebrate communities in the SAV and emergent vegetation. This monitoring will coincide with the return of SAV (*Hydrilla, Vallisneria*) and bulrush (*Scirpus californicus*), which mostly disappeared from Lake Okeechobee after the hurricanes in 2004 and 2005, and may begin in the near future.

6.9 CONCLUSIONS

Although historic biological data (prior to 1985) for Lake Okeechobee is patchy and often anecdotal, most of the existing evidence suggests that the lake has undergone rapid eutrophication over the past 60 to 80 years. Recently collected paleolimnoligical nutrient and algal data supports this pattern of recent change and suggests that increased nutrient loading to Lake Okeechobee can be attributed to post-1950s anthropogenic watershed alterations (Engstrom et al., 2006). Thus, it is rather certain that Lake Okeechobee has changed from a mesotrophic or mildly eutrophic lake in the early 1900s, to one which is currently strongly eutrophic or may be in the beginning stages of hyper-eutropy.

Prior to development of the watershed, Lake Okeechobee was underlain by sand and peat sediments and ancedotal evidence suggests that the water column was much less turbid than it is presently. This water clarity permitted adequate light penetration to occur to deeper depths than seen today, and as a result the lake quite probably supported very extensive beds of native submerged and emergent vegetation. This widespread aquatic plant community in turn sustained thriving forage and sport fish populations, which likely explained the popularity of the lake with the pre-modern settlement population. Following the development of the Lake Okeechobee basin and nearly complete hydrologic alteration, the lake has accumulated a large flocculent mud sediment zone, and experiences elevated nutrient concentrations, high turbidity and periodic algal blooms. The submerged and emergent plant and fish communities are highly variable and dependent on widely fluctuating conditions and have been supplanted to varying degree by invasive and exotic species. These changes have resulted from a combination of factors including restricted outflow capacity resulting from the construction of the Herbert Hoover Dike, a large input of terrigenous materials from the surrounding highly agricultural watershed, and prolonged excessively high and low lake stages. These severe fluctuations in lake stage reflect the interaction of climactic variability and Lake Okeechobee's role as the key water supply and flood control storage structure in south Florida.

A number of restoration projects are currently in the development phase in the watershed, to reduce nutrient influxes to Lake Okeechobee and to improve lake hydrology. These projects consist of ASR wells, STAs, and water storage reservoirs. Additional projects such as dredging and chemical inactivation of the sediments are being contemplated to reduce the lake's internal nutrient load as a means of accelerating ecological improvements to the system, since internal loading may delay the ecological restoration of the lake on the order of many decades. However, restoration of Lake Okeechobee, extends beyond the boundaries of the lake. Due to the lake's central location and hence, its influence on water quality and quantity in the Northern Estuaries and Everglades, the restoration of these upstream and downstream ecosystems also depends on successfully reducing nutrients in the lake and addressing water storage and lake stage issues.

Currently, periodic monitoring is in place for lake water quality and hydrology, submerged and emergent aquatic vegetation, fish, macroinvertebrates, bacterioplankton, phytoplankton, zooplankton and periphyton. In addition, the littoral portion of Lake Okeechobee is included in system-level monitoring for wading birds and their prey species, and for the Everglades snail kite. A number of other ongoing research studies aim to elucidate key ecological relationships between various ecosystem components. At present, research and monitoring efforts on Lake Okeechobee appear to be sufficient to detect significant changes expected to be brought about by restoration activities. However, only a small proportion of these monitoring and research activities are funded by RECOVER, with the majority being funded through other mandated or non-mandated SFWMD programs or directly by cooperating state agencies (e.g., FWC). This places RECOVER in a role not particularly unique to Lake Okeechobee - as the situation of disparate funding sources for various important monitoring exists throughout south Florida - of holistically compiling data from numerous sources to decipher ecological change and adaptively manage restoration progress.

6.9.1 Lessons Learned

As a consequence of both natural (e.g., climatic) and man-made perturbations, all of the parameters in Lake Okeechobee exhibit a wide range of variability. It is quite clear that extensive, long duration monitoring is required to separate changes due to restoration against this backdrop of variability. Even with all restoration projects in place, there will continue to be good years and bad years and the ability to detect systemwide improvement may depend on factors such as the ability to identify changes in the relative frequency or magnitude of these good and bad years.

Continued monitoring may show that certain monitoring parameters, such as SAV and macroinvertebrates, may prove more responsive to environmental restoration than others. As ongoing research continues to elucidate relationships between key environmental components, it may become possible to monitor fewer parameters without sacrificing assurances that the entire ecosystem is benefiting from restoration activities, and that no untoward effect ensues undetected.

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CHAPTER 7 NORTHERN ESTUARIES MODULE

7.1 INTRODUCTION

The Northern Estuaries Module contains estuaries on both coasts of south Florida with the Caloosahatchee River Estuary on the west coast and the St. Lucie Estuary, Southern Indian River Lagoon, Loxahatchee River Estuary and Lake Worth Lagoon on the east coast (***Figure 7-1***). Detailed descriptions of these individual water bodies can be found in the 2006 SSR (RECOVER, 2007a), which is available at www.evergladesplan.org/pm/recover/recover_docs/at_ssr/020807_rec_at_ssr_3_mod_ne.pdf.

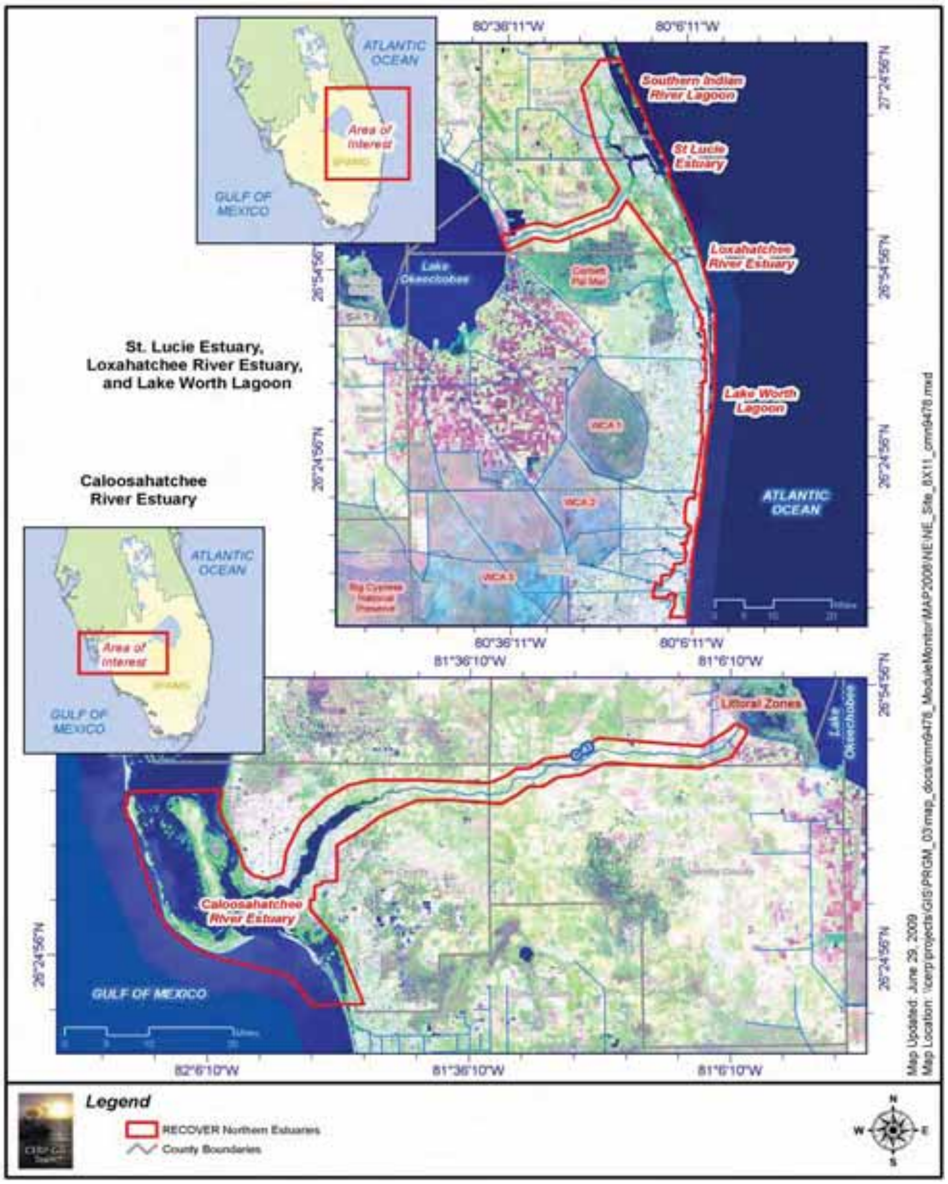


FIGURE 7-1. NORTHERN ESTUARIES BOUNDARIES

Historically, natural freshwater discharges into these water bodies sustained an ecologically appropriate range of salinity conditions to facilitate the presence of healthy floral and faunal communities. The recent urbanization of Florida‟s coastal regions and the ensuing increased demand for water and flood control has led to frequent high and low salinity extremes within the coastal water bodies. Managing for these demands has subsequently resulted in a shift in the ecological components that historically defined the coastal water bodies to communities that have been deemed less desirable. Current management goals of restoration activities are to develop strategies that will mitigate the impacts of historical water management practices by developing more realistic scenarios for controlling 1) water levels for flood management, 2) frequency and duration of water releases and consequent rapid salinity shifts, and 3) introduction of point and non-point source pollutants. Implementation of projects to reach these goals will require a comprehensive approach of both CERP and non-CERP constructed storage and water quality features as well as restoration of natural wetlands and both urban and agricultural BMPs. Removal of anthropogenically produced mucky sediments in the water bodies would be needed to achieve the water quality and clarity goals. These goals, if achieved, should curtail current habitat loss and allow the recovery of more desirable communities.

7.2 WATER QUALITY STRESSOR

7.2.1 Introduction and Background

Key elements defining water quality in the Northern Estuaries are salinity, water clarity and nutrients. These primary factors determine the health of both aquatic vegetation and animal communities that make up the interwoven web of estuarine life, which characterizes and makes unique each of the five estuaries within the Northern Estuaries Module. Water clarity is a stressor in the SAV Hypothesis Cluster, nutrients are a stressor in the SAV, Benthic Macroinvertebrates and Fish Hypothesis Clusters, and salinity is a stressor in all hypothesis clusters (***Figure 7-2***). The RECOVER program has developed performance measures for salinity, water clarity and nutrients. These performance measures can be found online at www.evergladesplan.org/pm/recover/perf_ne.aspx.

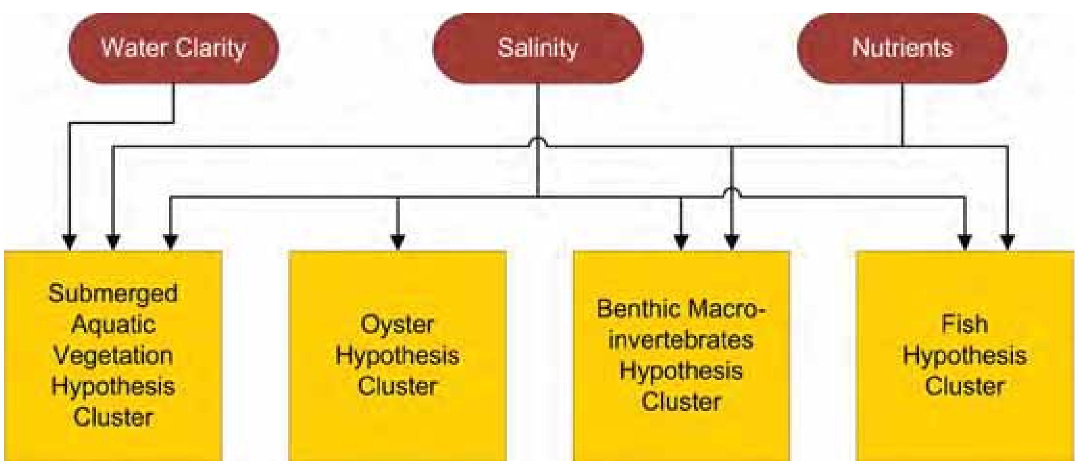


FIGURE 7-2. SALINITY, WATER CLARITY AND NUTRIENTS AS PRIMARY STRESSORS AFFECTING NORTHERN ESTUARIES HYPOTHESIS CLUSTERS

7.2.2 Caloosahatchee River Estuary

7.2.2.1 Monitoring

Water quality monitoring sites within the Caloosahatchee River Estuary are shown in ***Figure 7-3***.

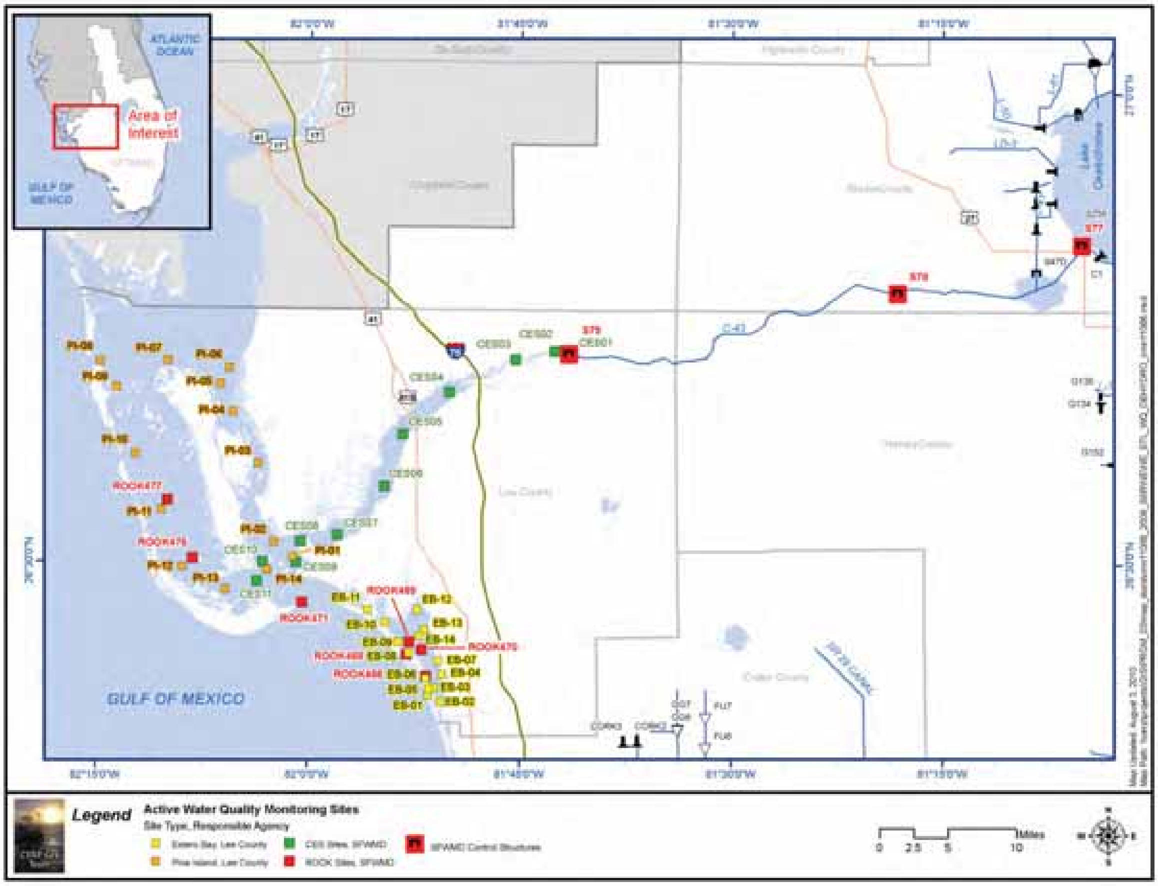


FIGURE 7-3. WATER QUALITY MONITORING SITES IN THE CALOOSAHATCHEE ESTUARY, PINE ISLAND SOUND AND ESTERO BAY

7.2.2.2 Results

Water quality in the Caloosahatchee River Estuary is very dependent on surface water inputs, especially from the S79 water control structure, which conveys water from both the watershed and Lake Okeechobee. When calculated as an 18-year average, the source of the water volume and the N load entering the estuary through the S79 structure is about evenly split between flows that originate from Lake Okeechobee and runoff from the Caloosahatchee River (C-43 Canal) drainage basin (***Figure 7-4***). However, approximately 68% of the calculated P load discharged to the estuary from the S79 structure can be attributed to basin input rather than from Lake

Okeechobee (***Figure 7-4***). Nutrient loads from Lake Okeechobee are calculated at the S77 structure. The difference in loads at S77 and S79 structures reflect the basin nutrient load.

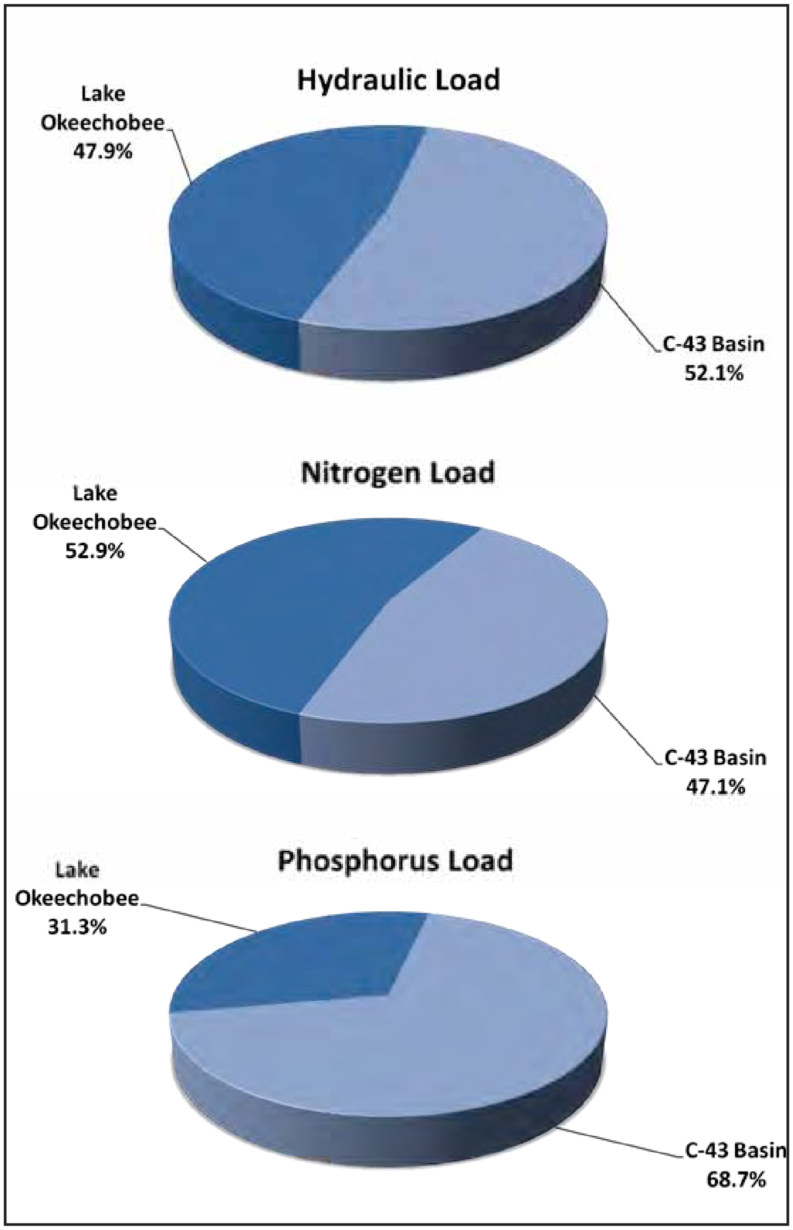


FIGURE 7-4. AVERAGE PERCENT CONTRIBUTION OF THE TOTAL HYDRAULIC, NITROGEN AND PHOUSPHORUS LOADS TO THE CALOOSAHATCHEE RIVER ESTUARY

When examined on an annual basis (***Figure 7-5***), it is clear that the ratio of load coming from Lake Okeechobee versus the load coming from the C-43 Basin changes from year to year. High nutrient loading events, which are typically associated with high flow events, also equate with reduced residence time within the estuary. The direct consequence of high flow events are precipitous drops in salinity and sustained low salinity. As stated previously, loads at S77 represent the loads from Lake Okeechobee while those at S79 represent the sum of loads to the Caloosahatchee River Estuary from both the lake and the drainage basin. The S78 structure is located on the C-43 Canal at approximately mid-point between the S77 and S79 structures.

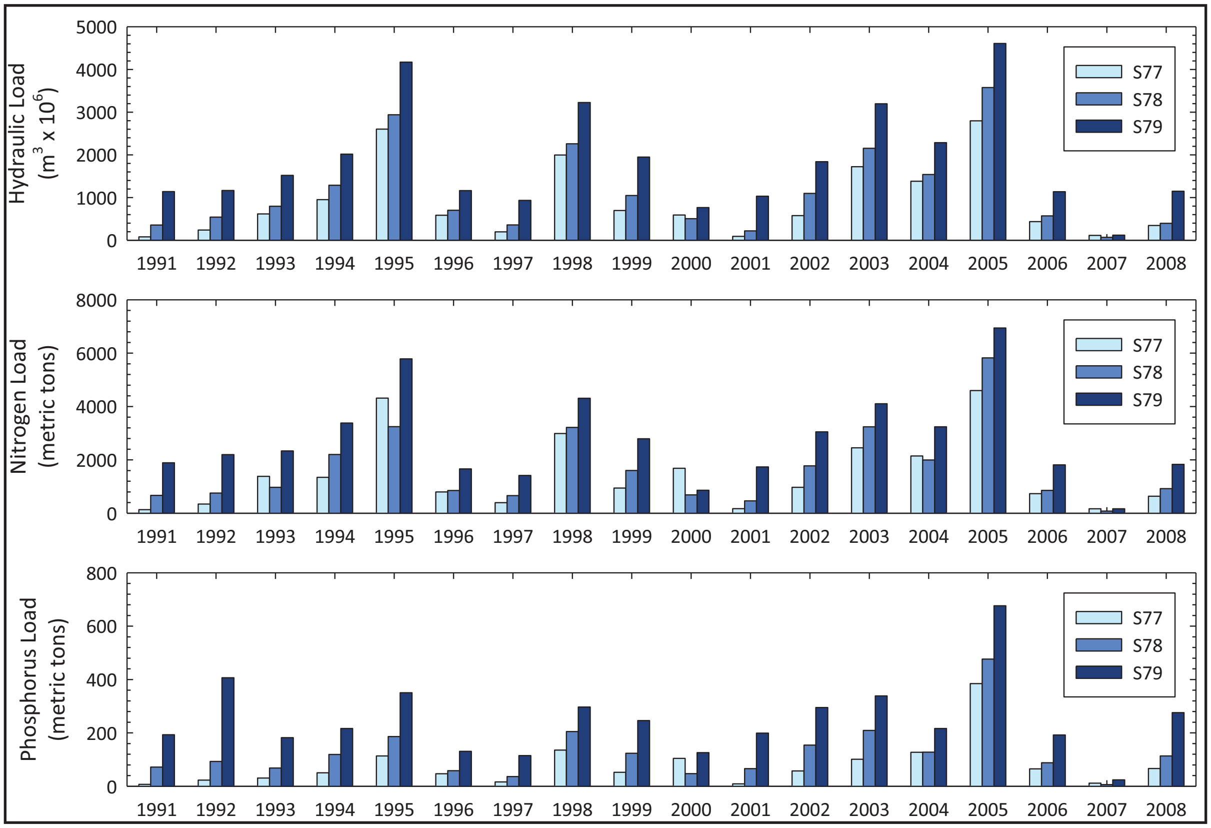


FIGURE 7-5. ANNUAL LOADS OF FRESH WATER, NITROGEN AND PHOSPHORUS FOR EACH OF THE STRUCTURES ALONG THE C-43 CANAL FROM 1991 THROUGH 2008

In consequence of the loading to the Caloosahatchee Estuary being from the S79 as a point source, both P and N concentrations generally decrease with distance from the S79 structure (***Figure 7-6***). However, chlorophyll *a* concentrations exhibit a peak in the upper and mid-estuary. Algal (or phytoplankton) blooms require both nutrients and water clarity, both of which are a function of proximity to the S79 and the Gulf of Mexico, respectively. Thus higher chlorophyll concentration events occur in those intermediate locations where both water clarity and nutrient availability are sufficiently appropriate to promote the observed localized algal response.

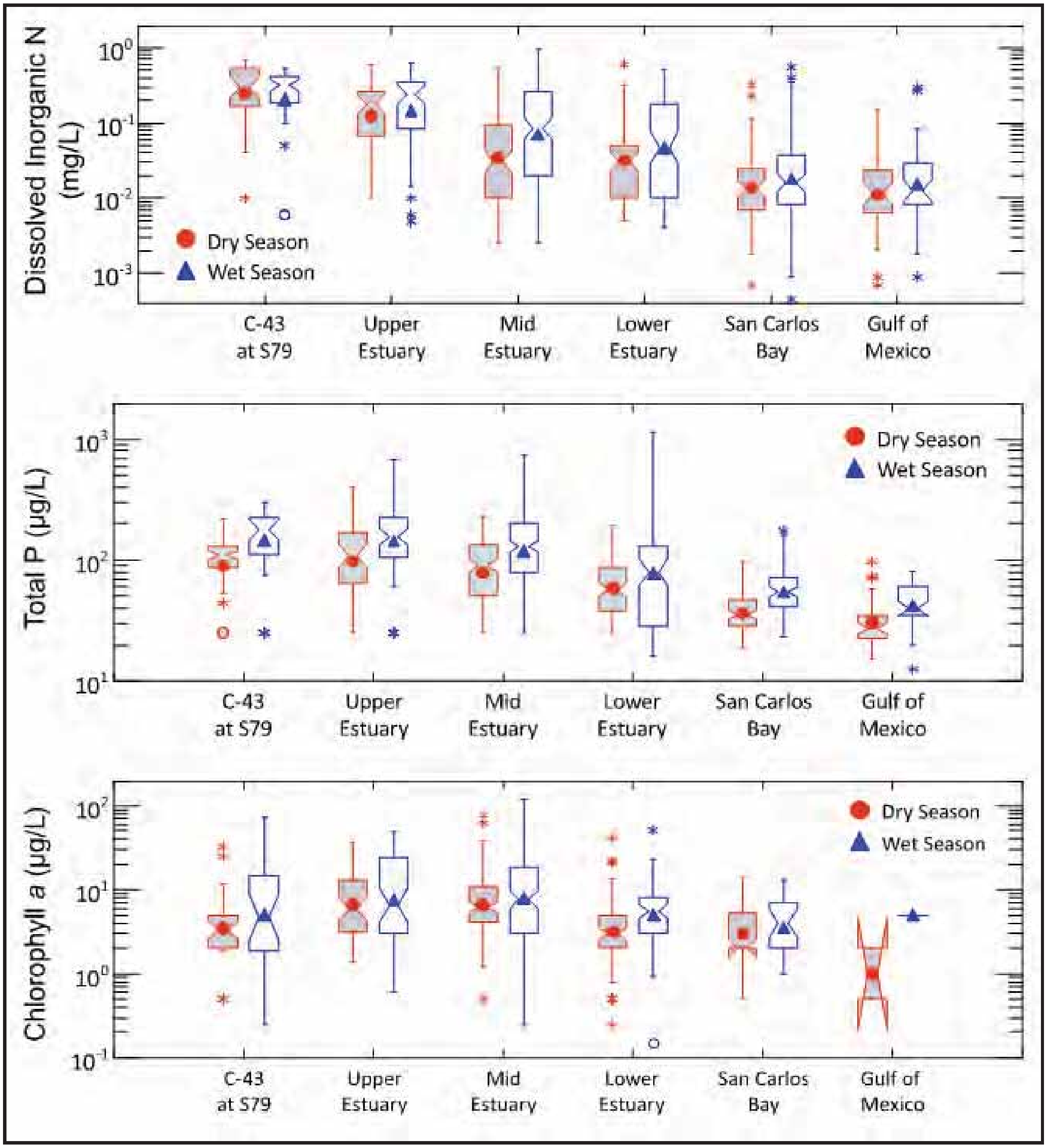


FIGURE 7-6. NOTCHED BOX-AND-WHISKER PLOTS COMPARING SEASONAL CONCENTRATIONS FOR DISSOLVED INORGANIC NITROGEN, TOTAL PHOSPHORUS AND CHLOROPHYLL *A* MEASURED IN THE FIVE REGIONS OF THE CALOOSAHATCHEE RIVER ESTUARY FROM 1991 THROUGH 2008

DIN coupled with adequate water clarity, temperature and available P can result in algal blooms, a concern in the Caloosahatchee River Estuary. When evaluated on the basis of mean condition over the most recent decade (***Figure 7-7***), comparatively elevated N concentrations appear to be sustained in the upper estuary for both dry and wet seasons, suggesting a source originating within the basin, not the lake. Wet season N concentrations are elevated approximately 0.2 to over 0.3 mg/L DIN in both the upper and mid-estuary. The impact from these elevated concentrations extends to the Gulf of Mexico. The „pocket‟ of higher DIN depicted in mid-Pine Island Sound is a consequence of N input from S79 flows and restricted circulation. Estero Bay N inputs appear to be associated with flows coming from Hendry/Mullock Creeks; however, current data with which to evaluate this apparent relationship was not available.

In the dry season, P levels measured in the Caloosahatchee Estuary reflect lower freshwater inflows and longer residence time of P in the narrow confines of the upper estuary (***Figure 7-8***). A slight increase in TP is apparent and is presumably associated with the Imperial River/Spring Creek, which drain the Bonita Springs urban area. In the wet season, the signature of increased loads, primarily from the C-43 Basin, are evident in the upper estuary and dimish to the west. Also suggestive is the plume of elevated P originating in the Peace River/Charlotte Harbor area, which appears capable of affecting the Caloosahatchee River Estuary TP regime; however, this area was outside the SFWMD boundary and circulation patterns and water quality data from the SFWMD was not evaluated in this SSR. The inclusion of such analyses will be evaluated for appropriateness of inclusion in future SSRs. Wet season Estero Bay P regime reflects the inflow of Imperial River/Spring Creek as well as Hendry/Mullock Creeks.

The ten-year mean chlorophyll *a* concentration in the dry season is approximately 10 to 14 µg/L (***Figure 7-9***). These values are a concern since the chlorophyll limit used to delineated impaired estuaries is 11 µg/L (62-303 F.A.C.). Chlorophyll *a* concentrations, and thus the frequency/intensity of blooms, in the rest of the estuary during the dry season are comparatively minimal. In the wet season, a large reach of the Caloosahatchee River Estuary experiences frequent algal blooms, with the average concentration above the impairment criteria. Wet season blooms in the area southwest of Pine Island Sound may be a result of nutrient loads being transported into the area during incoming tide. Nutrients from Charlotte Harbor may have some influence on the region, but the data necessary to evaluate this connection comes from water quality monitoring efforts north of the SFWMD boundary and was not obtained in time for the preparation of this SSR.

As expected, salinity is lowest in the upper estuary (being closer to the source of freshwater) and highest in the lower estuary and San Carlos Bay (***Figure 7-10***). Apparent color decreases in response to increased salinity as the colored fresh water becomes diluted by seawater, whereupon flocullation of the dissolved organic matter responsible for the majority of water color occurs. The apparent maxima in chlorophyll within the mid-estuary are a result of increased water clarity in an area where nutrient concentrations remain moderate. A general decrease in turbidity is observed from the upper estuary to the Gulf of Mexico (***Figure 7-10***), albeit difficult to visualize in the figure due to the log scale necessary to depict the large range of values. Turbidity maxima at the mid-estuary (similar to chlorophyll *a*) suggests that some of the turbidity measured may be associated with algal blooms.

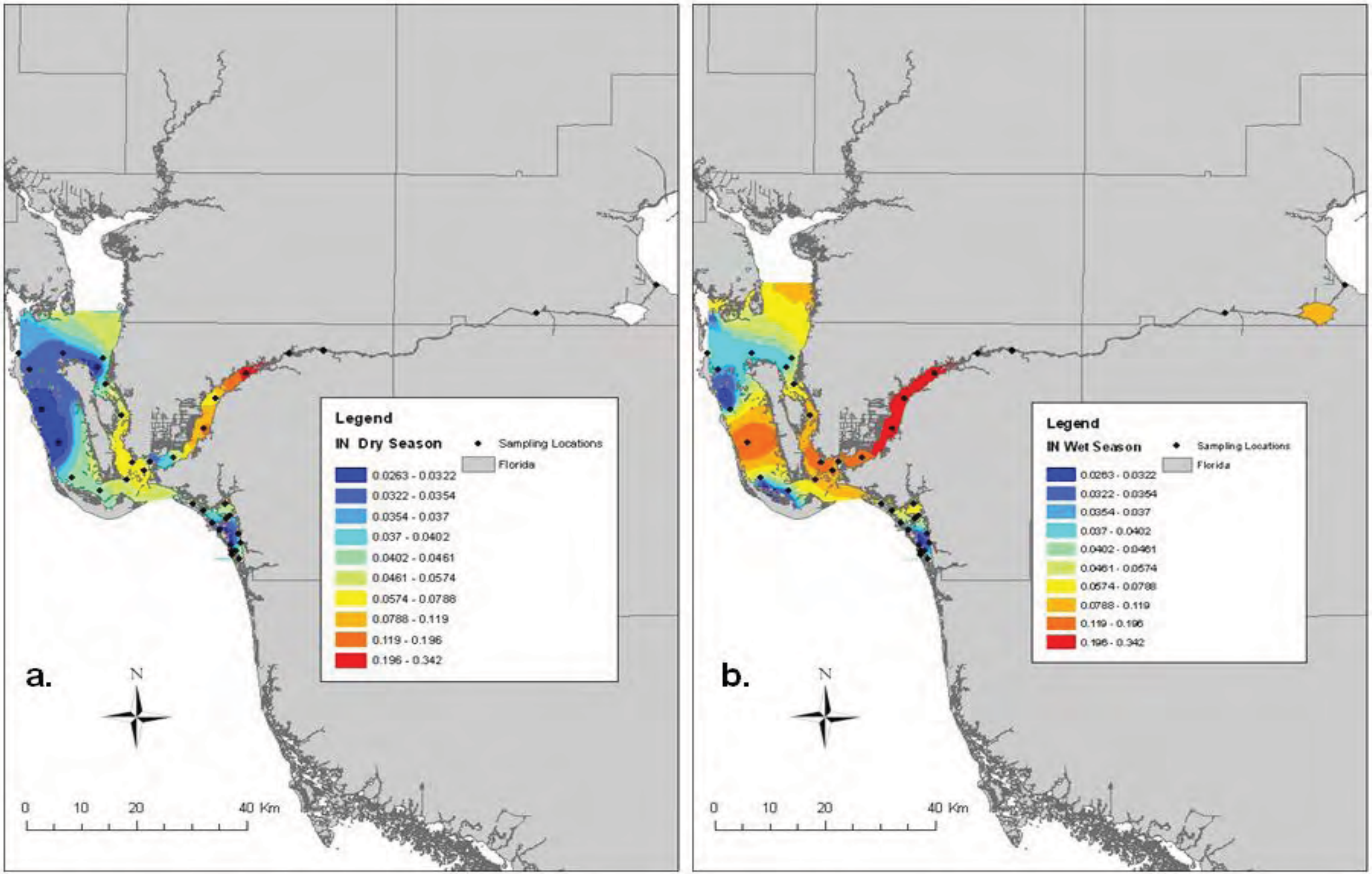


FIGURE 7-7. MEAN A) DRY AND B) WET SEASON DISSOLVED INORGANIC NITROGENCONCENTRATION IN CALOOSAHATCHEE RIVER ESTUARY, PINE ISLAND SOUND AND ESTERO BAY BASED ON 1999-2008 DATA

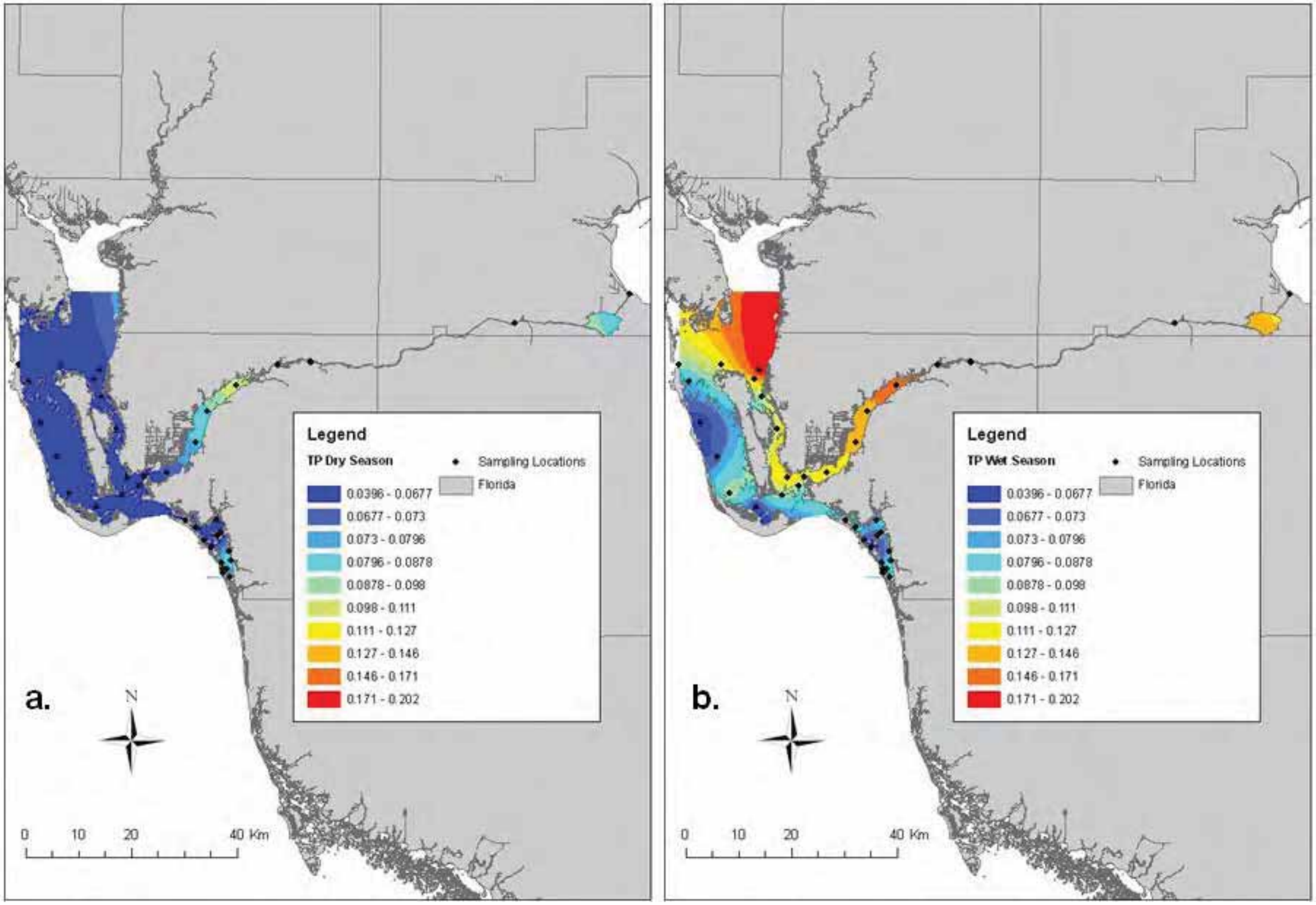


FIGURE 7-8. MEAN A) DRY AND B) WET SEASON TOTAL PHOSPHORUS CONCENTRATION IN CALOOSAHATCHEE RIVER ESTUARY, PINE ISLAND SOUND AND ESTERO BAY BASED ON 1999-2008 DATA

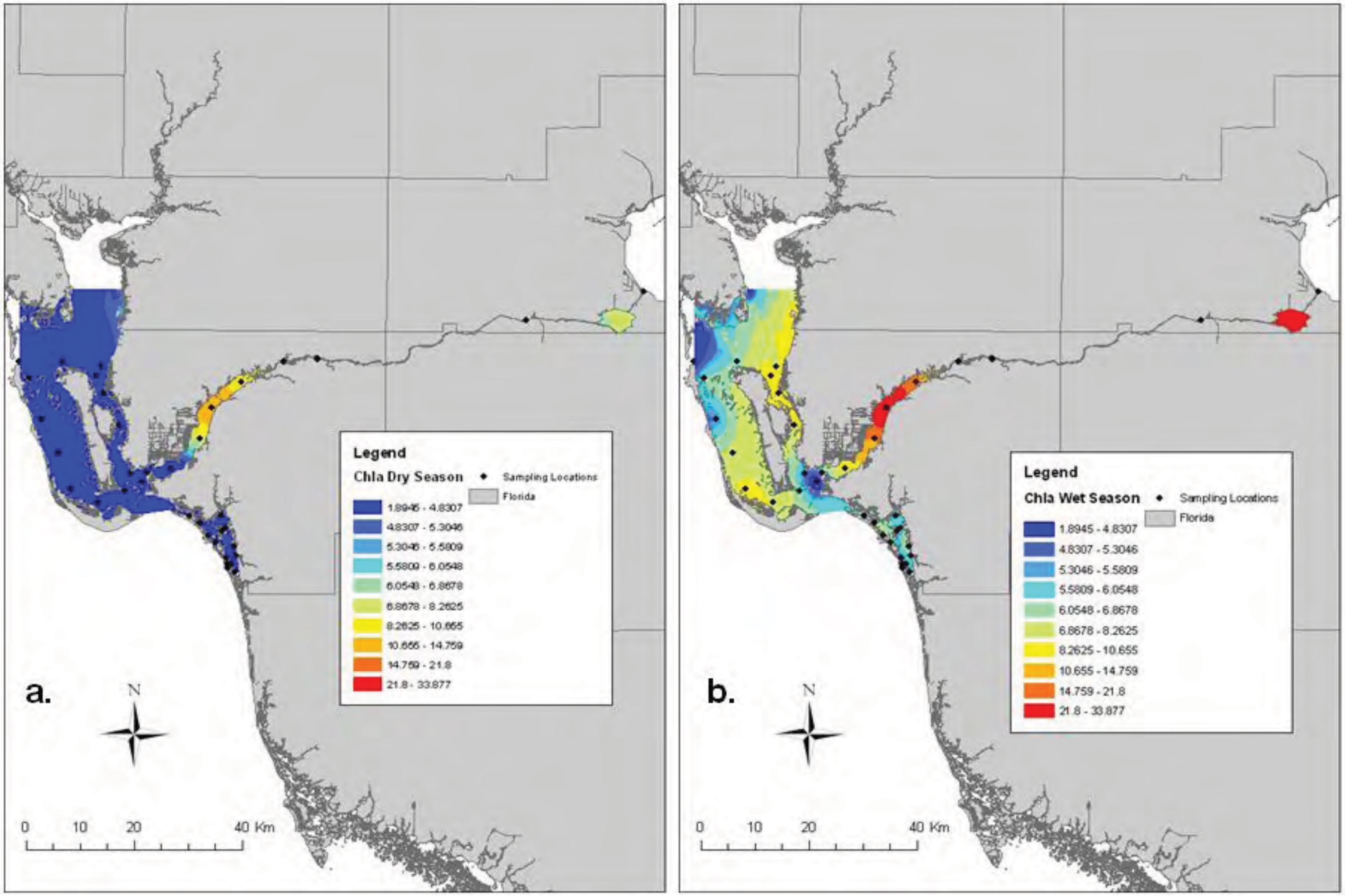


FIGURE 7-9. MEAN A) DRY AND B) WET SEASON CHLOROPHYLL *A* CONCENTRATION IN CALOOSAHATCHEE RIVER ESTUARY, PINE ISLAND SOUND AND ESTERO BAY BASED ON 1999-2008 DATA

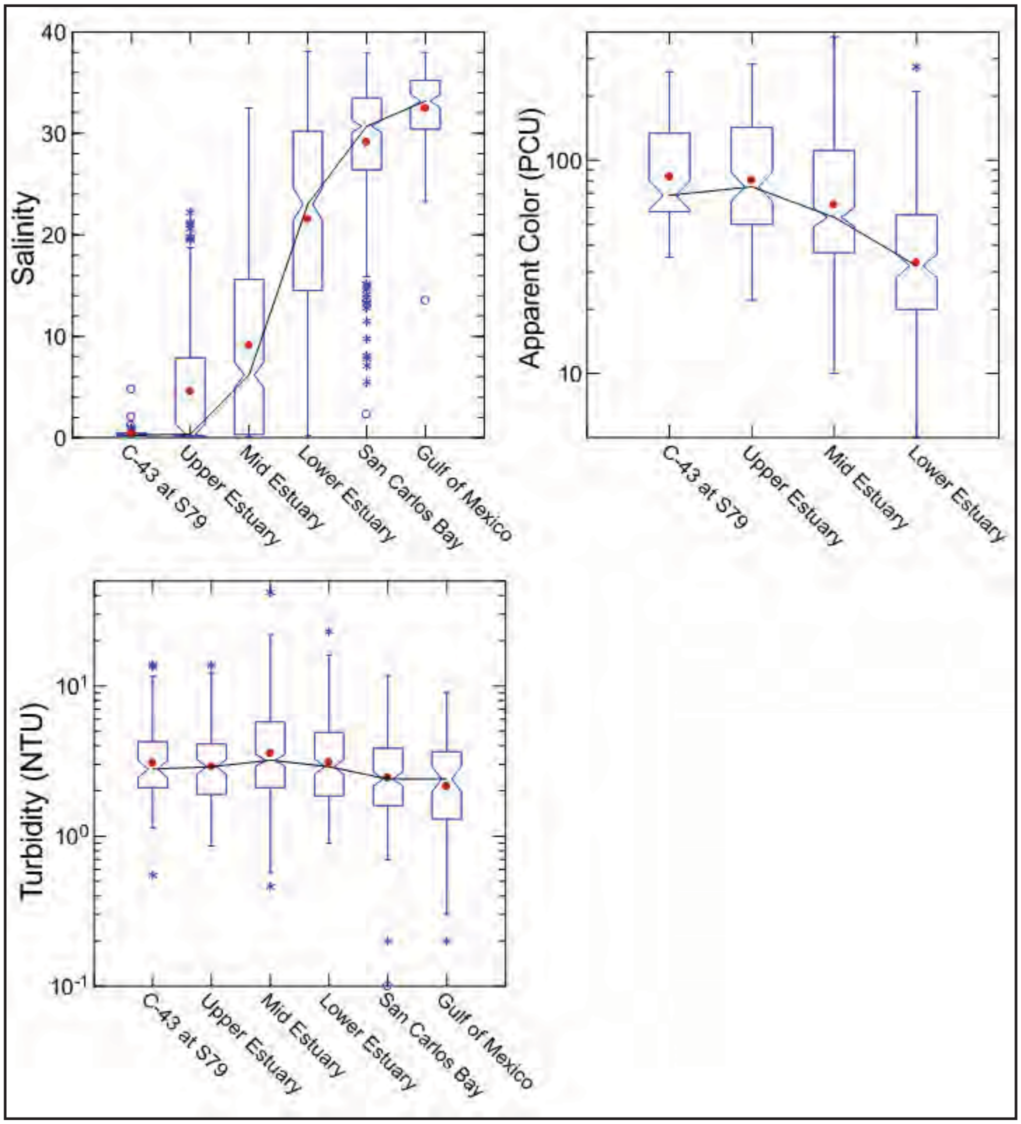


FIGURE 7-10. NOTCHED BOX-AND-WHISKER PLOTS SHOWING MEASURED CONCENTRATIONS OF SALINITY, APPARENT COLOR AND TURBIDITY FOR THE FIVE REGIONS IN THE CALOOSAHATCHEE RIVER ESTUARY FROM 1991 THROUGH 2008

The direct physical consequence of high flow events to the estuary are precipitous drops in salinity and sustained low salinity. Natural variation in freshwater flow and the resultant changes in salinity are part of the estuarine dynamic; however, modification of water delivery patterns necessary to afford flood protection and assure public safety can create unstable salinity conditions that can adversely affect biota. Salinity can rapidly decrease from polyhaline or mesohaline to oligohaline condition in the Caloosahatchee River Estuary. Conversely, as freshwater flows to the estuary diminish, salinity rebounds. These fluctuations in salinity (especially bottom water salinity) make it difficult for benthic organisms to get established, reproduce and, in general, survive. From 1992 to 2009, during three-quarters of those years bottom salinity at the Fort Myers Yacht Basin shifted more than 7 psu in a single week, with shifts as high as 16 psu in a single week recorded (1996). Much of the rest of the estuarine food web is dependent in one way or another on the health of the benthic flora and fauna. The continuous bottom salinity data collected at the Fort Myers Yacht Basin is shown in ***Figure 7-11***. The salinity probe is mounted at a depth of seven (7) feet below the surface, such that flows below 300 cubic feet per second (cfs) typically have little effect on salinities at this depth. Bottom salinity of shallower areas would change more responsively to the input of freshwater; however, the actual salinity regime experienced at any location is a function both of flow volume and duration of flow.

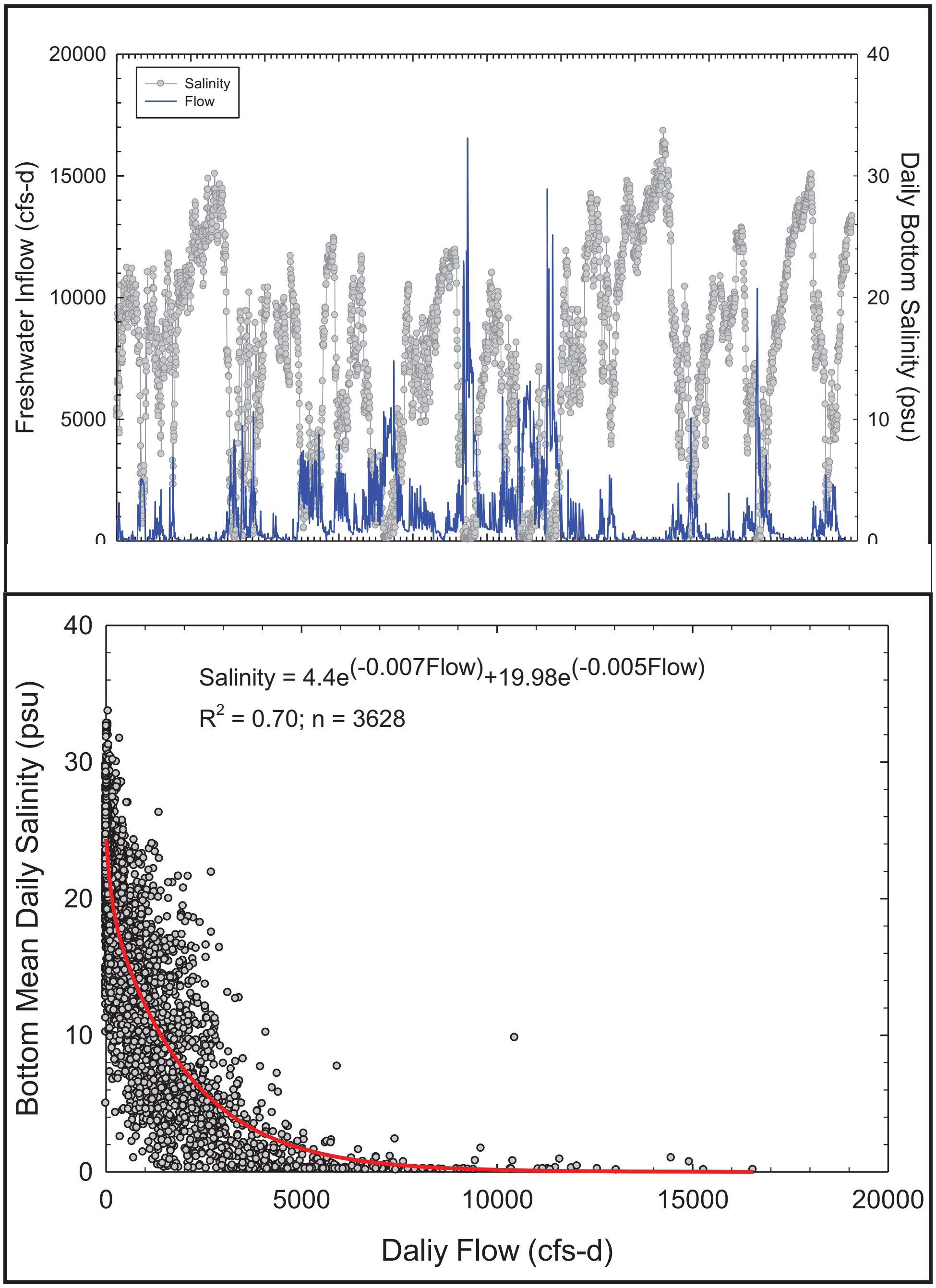


FIGURE 7-11. BOTTOM SALINITY AT FORT MYERS YACHT BASIN AS A FUNCTION OF FLOW VOLUME AT THE S79 STRUCTURE

Note: Red line denotes relationship between flow and salinity

7.2.3 St. Lucie Estuary

7.2.3.1 Monitoring

Water quality monitoring sites within the St. Lucie Estuary and canal inflow structure locations are shown in ***Figure 7-12***.

7.2.3.2 Results

Water flows from Lake Okeechobee into the St. Lucie Estuary via the C-44 Canal, with flows from the local basin entering via the C-44, C-23 and C-24 canals, the North and South Forks, and other minor local tributaries. Water flow and nutrient concentrations are measured at the S80, S48 and S49 structures for the C-44, C-23 and C-24 canals, respectively. The North Fork of the St. Lucie River is a major tributary to the estuary on par with the canals and the wet season flow volume to the estuary has been dominated by runoff from the basin, not Lake Okeechobee, but it is not currently possible to quantify flows or nutrient coming from the North Fork and contributions from this basin are not included in the analyses presented here. Infrastructure necessary to incorporate the North Fork and its basin into future water quality analyses is being implemented.

Figure 7-13 presents hydraulic load (freshwater flow) as well as N and P loads entering the St. Lucie Estuary from the C-44, C-23 and C-24 canals. It is clear from ***Figure 7-13a*** that dry season flow is dominated by that flowing through structure S80 (C-44); however, during the wet season, the S48 (C-23) and S49 (C-24) structures contribute flows whose sum is equal to that from S80.

In both wet and dry seasons, the relative sources of N load (***Figure 7-13b***) appear to follow the same general pattern as does the hydraulic load. During the dry season, most of the TP load (73 percent) comes from the C-44 Canal, but in the wet season, the load from each of the three canals is nearly equal (***Figure 7-13c***). Since concentrations of P in the North Fork is comparable to that in the C-23 Canal, it follows that the wet season load from the North Fork may be fairly equivalent to that from the canals in the wet season. If these assumptions are borne out by impending modeling and monitoring exercises, the C-44 Canal may only be delivering a quarter of the wet season load of P to the estuary, an idea that is surprising given that the C-44 Canal wet season load includes those loads delivered by Lake Okeechobee releases.

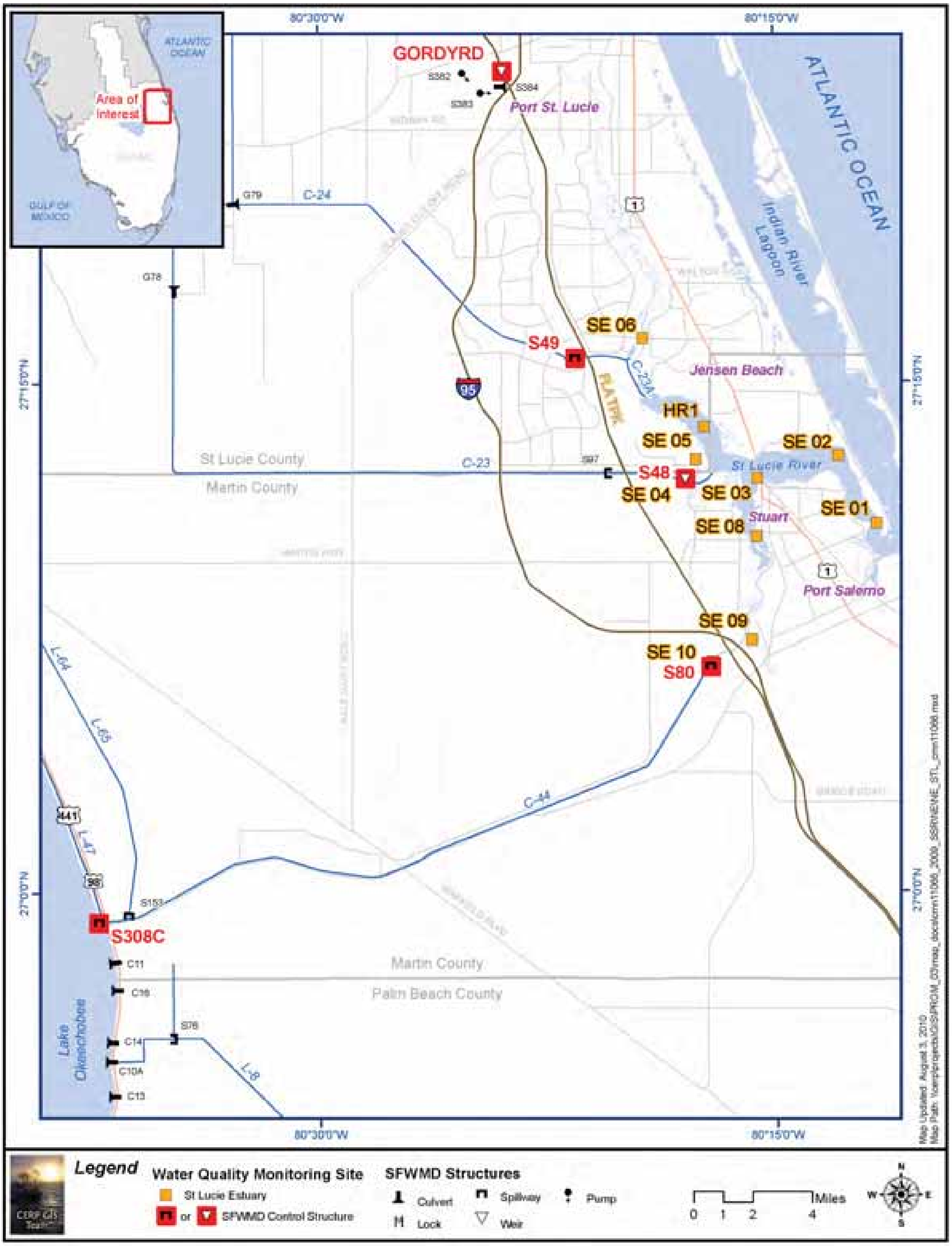


FIGURE 7-12. WATER QUALITY MONITORING SITES IN THE ST. LUCIE ESTUARY

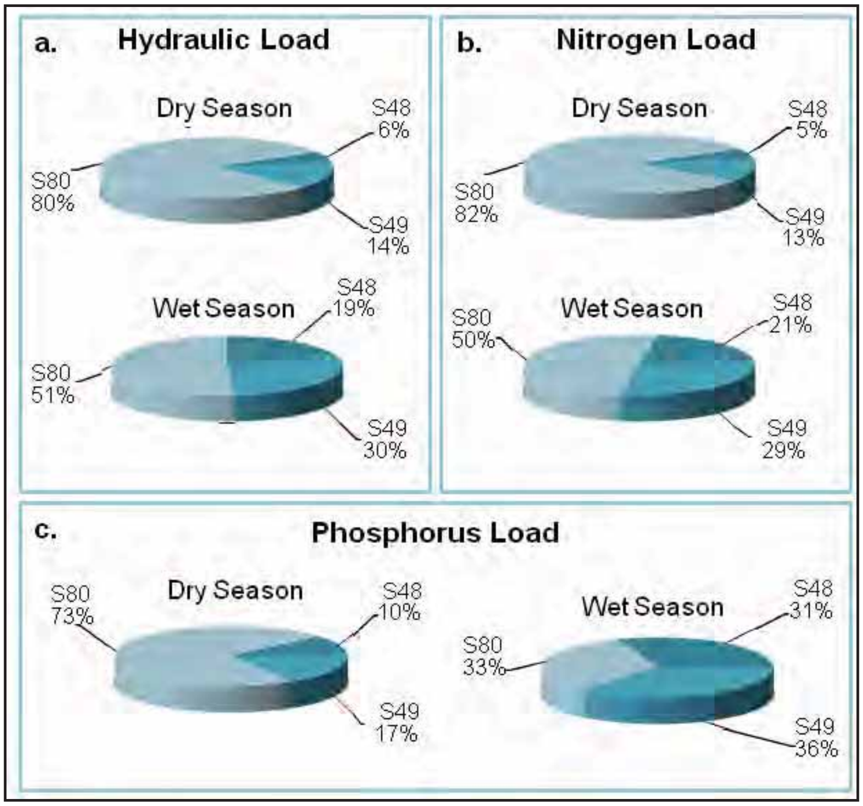


FIGURE 7-13. SEASONAL CONTRIBUTIONS OF TOTAL ANNUAL A) FRESHWATER INFLOWS (HYDRAULIC LOAD) B) NITROGEN LOAD AND C) PHOSPHORUS LOAD BY THE THREE GAUGED STRUCTURES TO THE ST. LUCIE RIVER ESTUARY 1991 -2008

In order to better understand the contribution of loads entering the estuary from S80 structure (C-44), a great local concern, it is necessary to differentiate loads originating from Lake Okeechobee versus those loads coming from the C-44 drainage basin itself (***Figure 7-14***). Percent contribution from the C-44 Basin was calculated as the net difference of flow and loads entering the C-44 Canal from Lake Okeechobee at structure S308, versus those flow and loads entering the St. Lucie Estuary from the C-44 Canal as measured at the S80 structure.

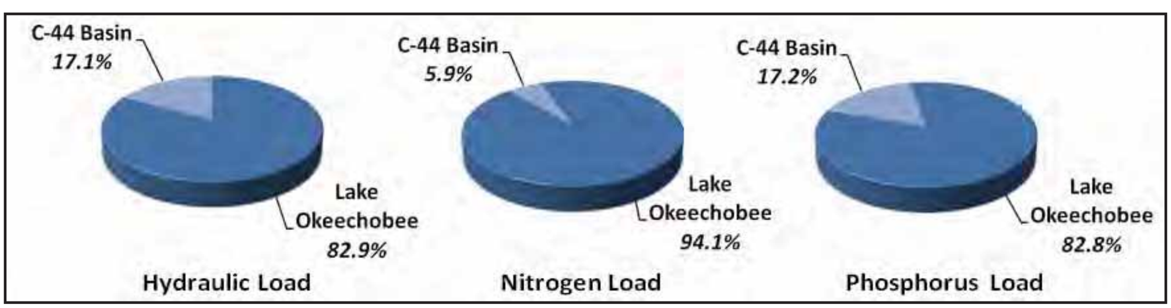


FIGURE 7-14. SOURCE OF FRESHWATER FLOW (HYDRAULIC LOAD) AND NITROGEN AND PHOSPHORUS LOADS FROM THE C-44 CANAL VERSUS LAKE OKEECHOBEE BASED ON 1991-2008 DATA

Under the current scenario of limited ability to store or move water south, the necessity of managing Lake Okeechobee levels through mandated east-west discharges results in the lake being the source of the majority of water and nutrient load entering the estuary from the S80 structure. However, it is important to reiterate that addressing loads from S80 is only part of the problem and in the wet season only around a quarter of the total P load may be entering the system via this route. The difference in the ratio of flow originating from the basin versus the lake between the St. Lucie and Caloosahatchee River Estuaries is due to the much larger C-43 drainage basin; origination of flows over the S79 structure into the Caloosahatchee River Estuary are about evenly split between Lake Okeechobee and the C-43 drainage basin (***Figure 7-15***).

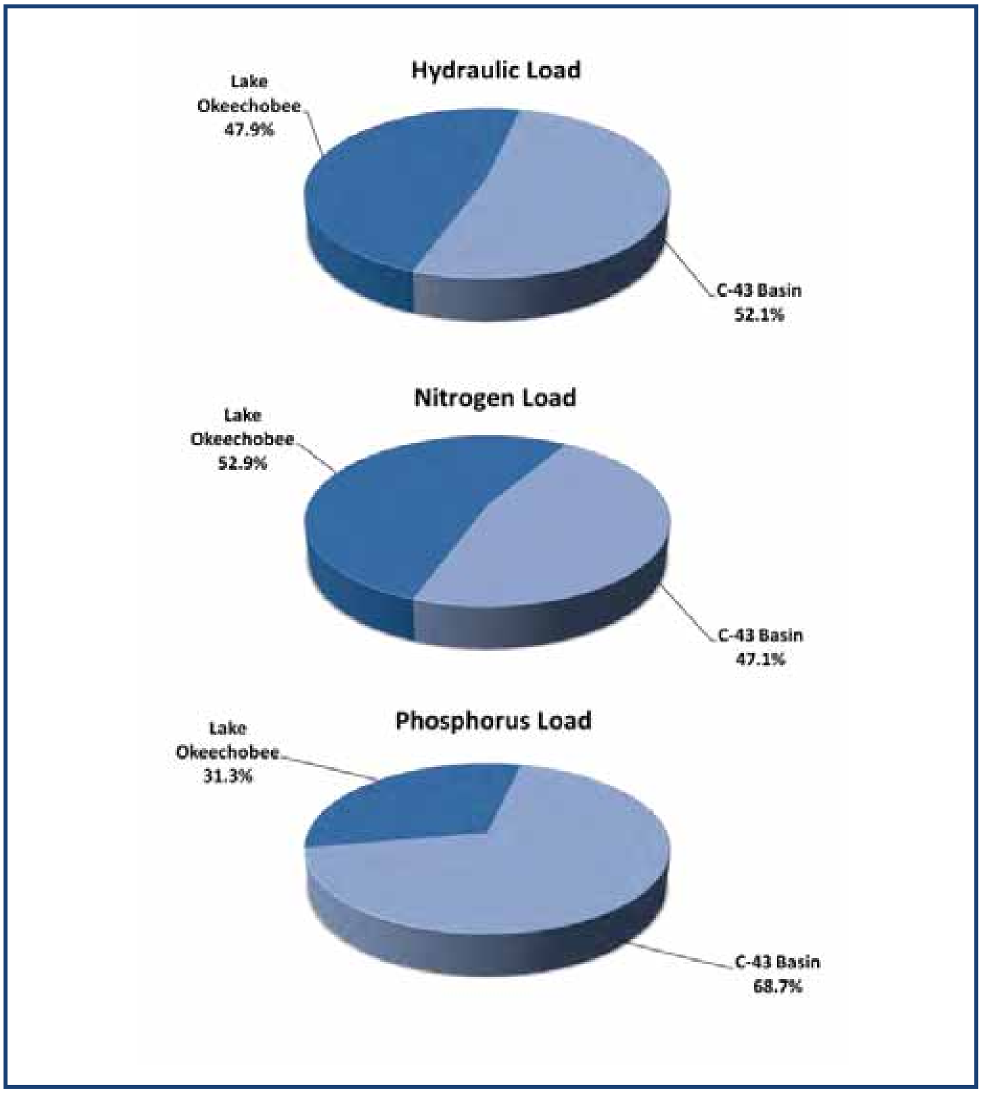


FIGURE 7-15. SOURCE OF FRESHWATER FLOW (HYDRAULIC LOAD) AND NITROGEN AND PHOSPHORUS LOADS FROM THE C-44 CANAL VERSUS LAKE OKEECHOBEE BASED ON 1991-2008 DATA

The breakdown of flows and loads from C-44, C-24 and C-23 canals by year (***Figure 7-16***) indicates that in all but perhaps five of the 18 years, flow volume and nutrient load from the C-23 and C-24 canals has been a significant donor to the elevated estuarine nutrient regime. The median TN concentration is approximately 1.25 mg/L (***Figure 7-17***), which is 1.7 times the median value of TN of all Florida‟s estuaries that are not the recipient of flows from geologic deposits of P, which is 0.74 mg/L (USACE and SFWMD, 2004). In addition, TP concentrations measured at the US Highway 1 Bridge (site SE03) are approximately two to three times higher than the target of 81 parts per billion (ppb) P set for the Indian River Lagoon - South project and TMDLs (***Figure 7-17***).

Of the four parameters presented in ***Figure 7-18*** for SE 03, two (TP and DIN) show an increasing trend while the other two (TSS and chlorophyll *a*) show a decreasing trend for the period from 1991 through 2008. None of the trends presented in this figure are statistically significant; however, the apparent upward trend in estuarine TP at SE0 3 (Sen slope = 0.4 ppb per year, Sen 1968) is arguably due to the observed increases in TP concentration in both the C-23 and C-24 canals.

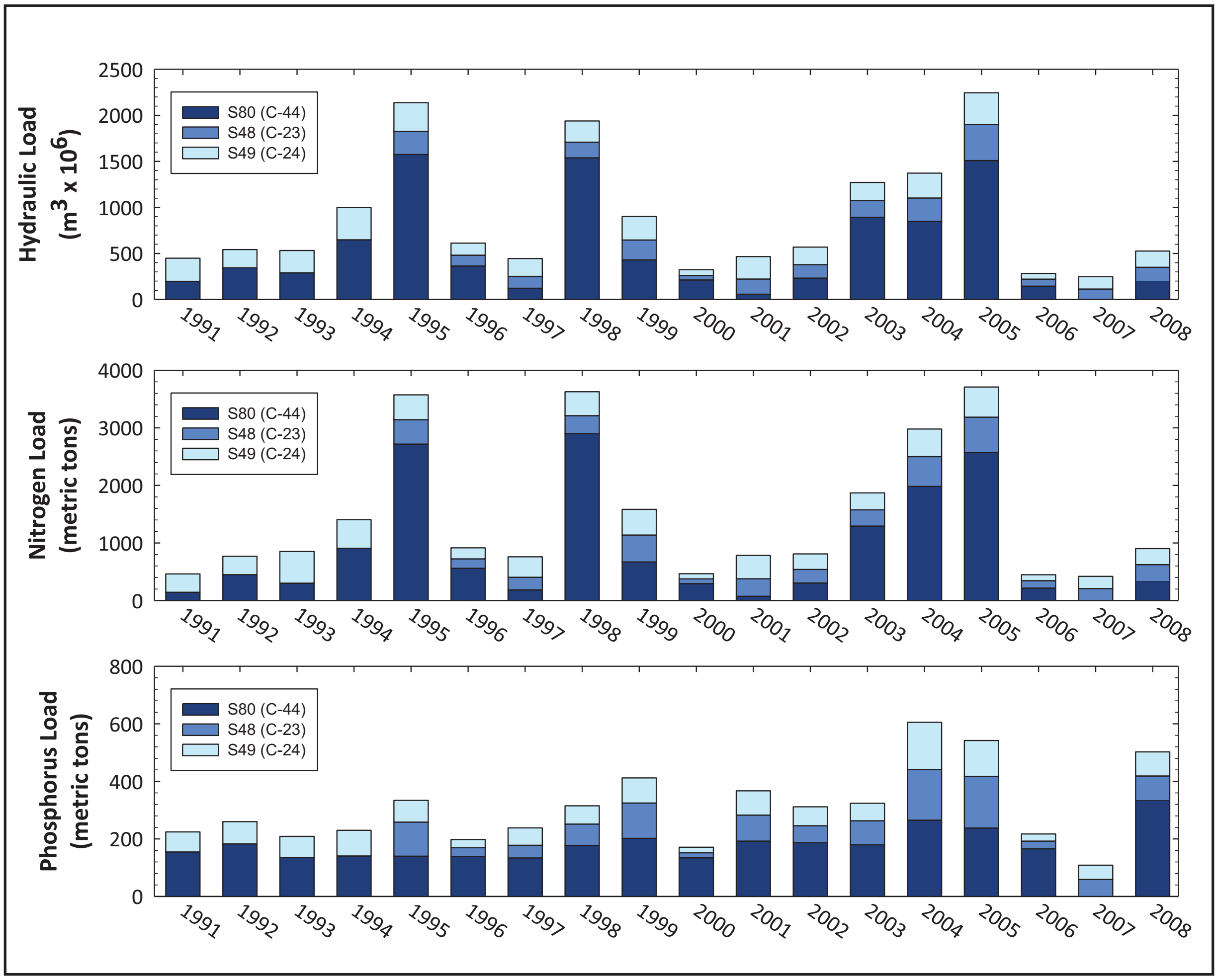


FIGURE 7-16. SUMMARIZATION OF TOTAL ANNUAL FRESHWATER INFLOWS (HYDRAULIC LOAD) AND NITROGEN AND PHOSPHORUS LOADS FROM GAUGED STRUCTURES EMPTYING INTO THE ST. LUCIE ESTUARY FROM 1991 THROUGH 2008

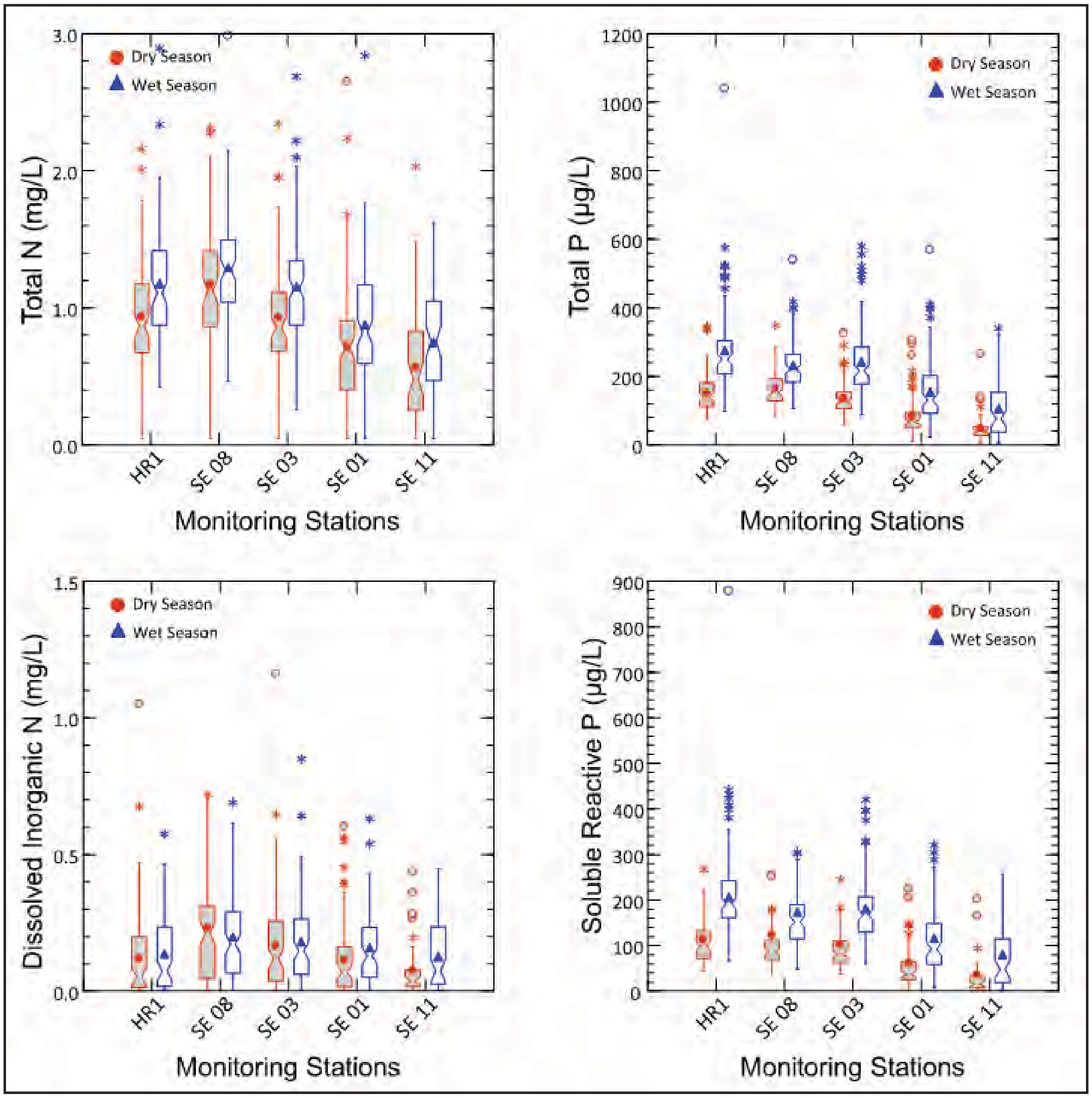


FIGURE 7-17. NOTCHED BOX-AND-WHISKER PLOTS SHOWING DIFFERENCES BETWEEN DRY AND WET SEASON CONCENTRATIONS OF TOTAL NITROGEN, TOTAL PHOSPHORUS, DISSOLVED INORGANIC NITROGEN AND SOLUBLE REACTIVE PHOSPHORUS MEASURED AT FIVE MONITORING STATIONS IN THE ST. LUCIE ESTUARY FROM 1992 THROUGH 2008

Note: Overlapping notches suggest that a statistical difference does not exist at a significant level (a) of 0.05. The filled circle and triange represent the mean concentration. Wet season is defined as the period of the year from May through October.

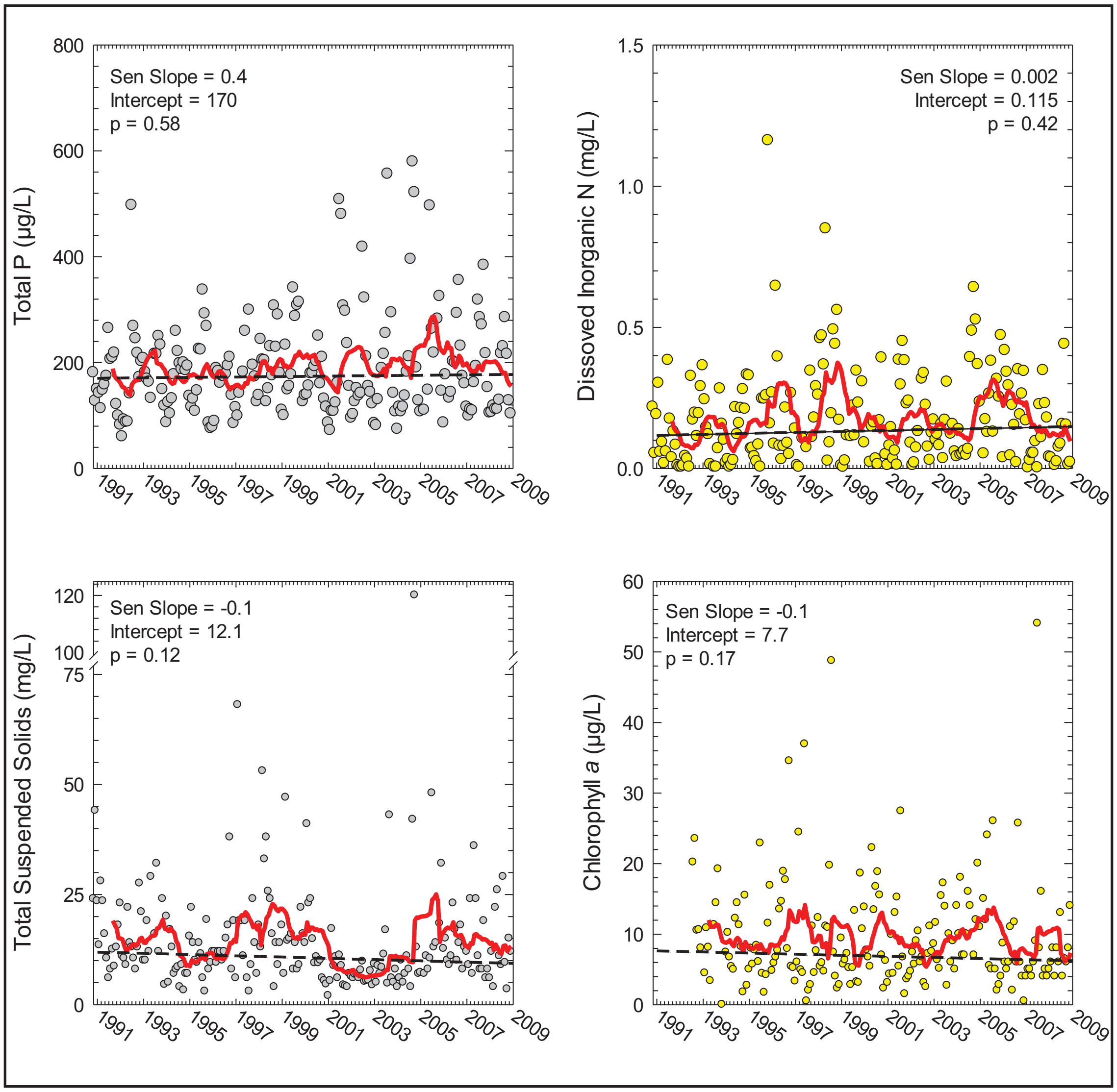


FIGURE 7-18. AVERAGE MONTHLY CONCENTRATIONS OF TOTAL PHOSPHORUS, DISSOLVED INORGANIC NITROGEN, TOTAL SUSPENDED SOLIDS AND CHLOROPHYLL *A* MEASURED AT SITE SE03 IN THE ST. LUCIE ESTUARY FROM 1991 THROUGH 2008

Note: The solid lines in each plot represent moving average concentration. The dashed lines represent the seasonal Kendall Tau trend lines. A positive Sen Slope indicates increasing trend, negative a decreasing trend. All probability (p) values are greater than 0.05, which indicate that trends are apparent and not statistically significant.

Of particular concern in the St. Lucie Estuary, is the impact that freshwater flow volumes have on salinity. Unlike nutrients, which can indirectly influence aquatic plants and animals principally through phytoplankton blooms and other types of secondary stressors, deviations of salinity beyond organism-specific thresholds can cause mortality, lack of reproductive success, loss of habitat, and general avoidance of the area by motile species possessing the option to leave. Organisms such as oysters or other benthic fauna must either „weather it out‟ or die. It is of little surprise that large discharge events (***Figure 7-19a***) result in precipitous drops in salinity

(***Figure 7-19b***), as occurred in 2004, 2005 and 2008. Sustained discharge events as those that occurred in 2001, 2002 and 2003 can also result in adverse and damaging salinity events. Because most of these events occur during the wet season and most of the flow originates in the basin, resolving the salinity stability issue in the estuary (so shifts between higher and lower salinity do not exceed the capacity of plants and animals to survive) require addressing basin flows as well as large releases from Lake Okeechobee.

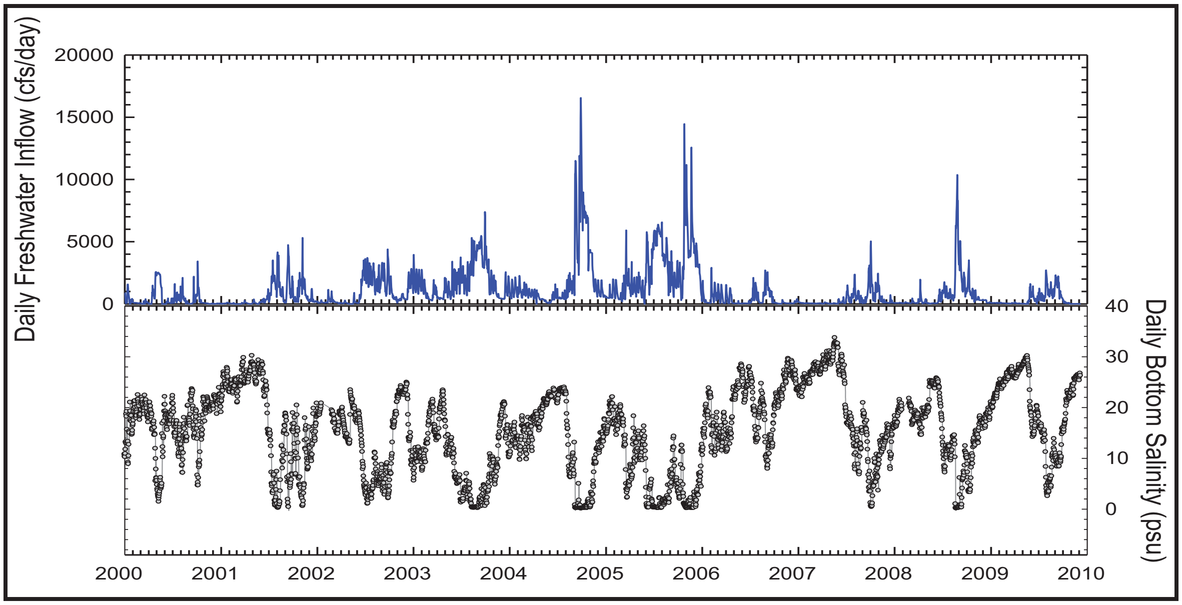


FIGURE 7-19. SUM OF BASIN FLOWS, EXCLUDING NORTH FORK CONTRIBUTIONS, AND BOTTOM SALINITY AT SITE SE03 IN THE ST. LUCIE ESTUARY

7.2.4 Southern Indian River Lagoon

7.2.4.1 Monitoring

Water quality monitoring sites within the Southern Indian River Lagoon as well as canal control structures are shown in ***Figure 7-20***.

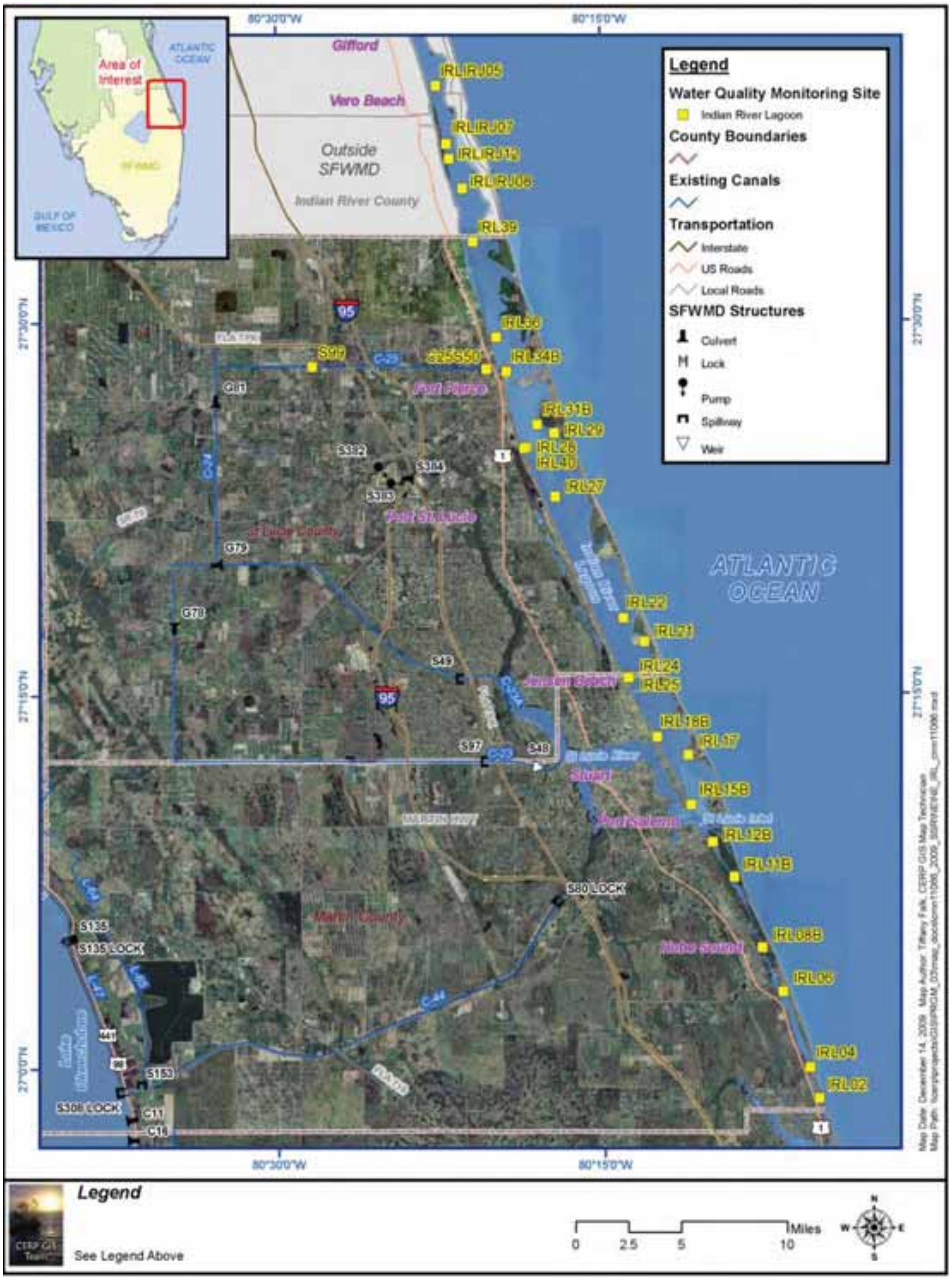


FIGURE 7-20. WATER QUALITY MONITORING SITES IN SOUTHERN INDIAN RIVER LAGOON

7.2.4.2 Results

Current estimates of loading are shown in ***Figure 7-21***. These loadings are based on flow measurements only from gauged structures. Measurements of flow coming from the North Fork of the St. Lucie River and from the Fort Pierce Farms canal (which is located just north of the C-25 Canal in Fort Pierce) are not available. As a result loadings to both the Saint Lucie estuary and Southern Indian River Lagoon are being underestimated. Samples from Fort Pierce Farms indicate that water quality is comparable to that from C-25 (based on available 2001 – 2006 data, mean TP = 0.24 and 0.21 mg/l, and mean TN = 1.05 and 1.31 mg/l, respectively). Assuming that volumes discharged are equally similar, the actual hydraulic and nutrient load transported to the IRL could be approximately twice that shown in ***Figure 7-21*** (i.e., for C-25 alone). To the degree these assumptions reflect true condition, contribution of water and loading from the C-25/ Fort Pierce Farms canals would fairly equal loads from the C-23, C-24 and C-44 canals combined in at least 11 of the 18 years depicted (where C-44 hydraulic and nutrient load includes that due to regulatory releases from Lake Okeechobee). All indications are that these assumptions are correct, indicating that loading from C-25/ Fort Pierce Farms is not an inconsequential concern

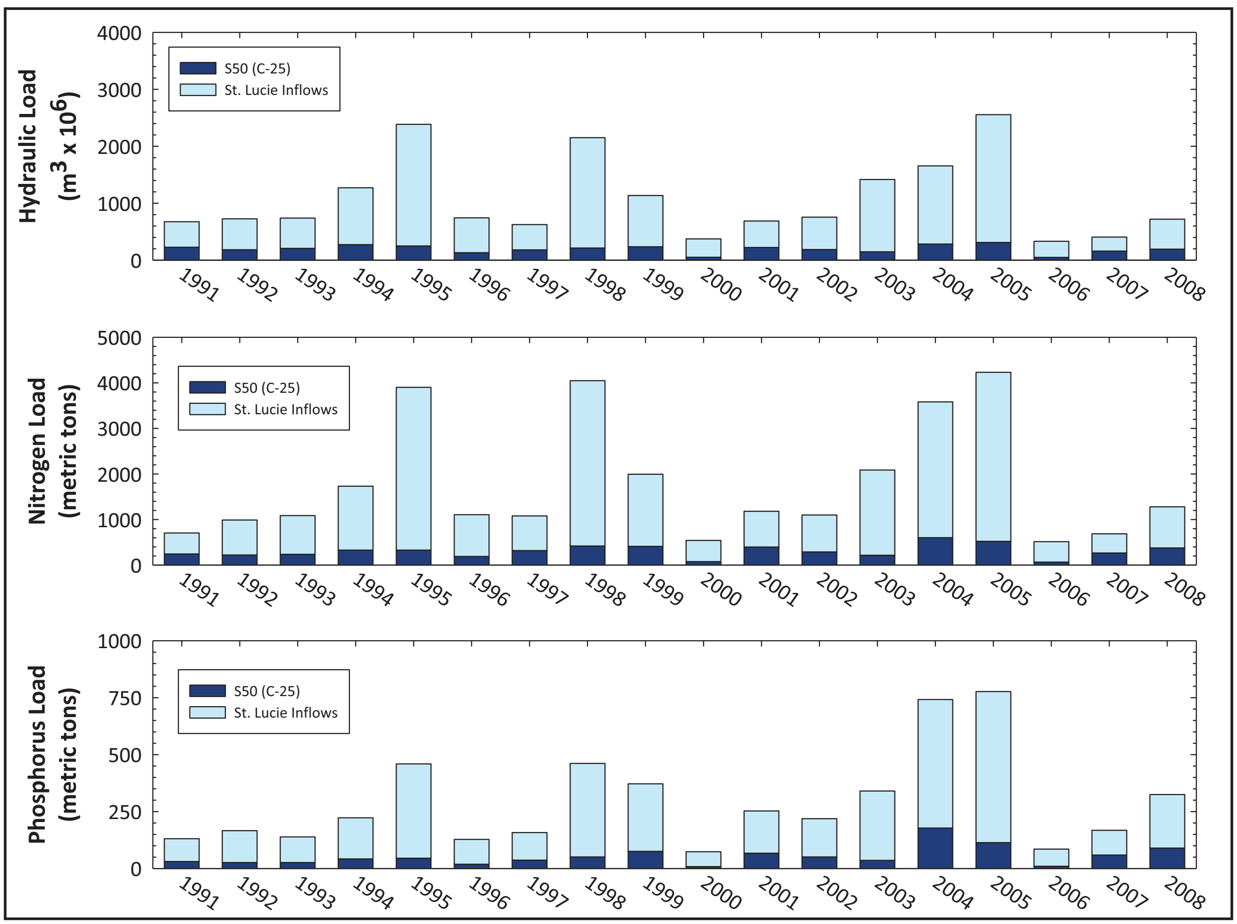


FIGURE 7-21. ESTIMATES OF FRESHWATER INFLOWS (HYDRAULIC LOAD), AND NITROGEN AND PHOSPHORUS LOADING FROM TWO MAJOR SOURCES TO THE SOUTHERN INDIAN RIVER LAGOON

Note: St. Lucie loading is based on measurements at the S80, S48 and S49 structures. C-25 loading is based on measurements at the S50 structure.

Because these large potential sources enter the system near inlets, deciphering the complexities involved requires a mathematical model that can predict the amount of water leaving the inlets and being entrained into the lagoon by tide. The St. Lucie Estuary/Southern Indian River Lagoon Curvilinear-grid Hydrodynamics Three-Dimensional (CH3D) hydrodynamic model (***Figure 7-22***) is well calibrated hydrodynamically throughout the model domain (however, current versions can only evaluate nutrient regime processes in the St. Lucie Estuary).

The morphology of Southern Indian River Lagoon is akin to a strip of water running generally north and south separated from the Atlantic Ocean on the east by barrier islands and bounded by the Atlantic Coastal Ridge on the west along most of its extent. Therefore, it is informative to evaluate water quality as a band where the x-axis represents distance from the Jupiter Inlet heading north and the y-axis represents time (***Figure 7-23***). Examination of the top panel in ***Figure 7-23*** reveals significant freshening of the lagoon where it is impacted by the St. Lucie River in 1992 through 1998 and again in 2005. The constant influence of the Fort Pierce Inlet is shown as a nearly unbroken darker area reflecting depth and width of the inlet as compared to the St. Lucie Inlet. TN (mid-panel) and TP (bottom panel) clearly show (particularly the TP panel) the loads from both the St. Lucie and C-25/Fort Pierce Farms canals, which discharge at a location slightly north of the inlet in both cases, affect water quality predominately north of the inlet; however, it is clear these discharges affect the lagoon both north and south of the inlets and can at times affect large areas (e.g., TP in 2005).

The effect of St. Lucie River and C-25/Fort Pierce Farms discharges on the lagoon‟s nitrate + nitrite and soluble P regime (***Figure 7-24)*** is readily evident. Chlorophyll blooms (top panel) were apparent throughout the length of the lagoon during the 2005 Lake Okeechobee regulatory release. Most of the algal blooms are occurring in the northern most section of the lagoon, north of the Ft. Pierce inlet near the Indian River County boundary, which is a concern due to the biological importance of this area.

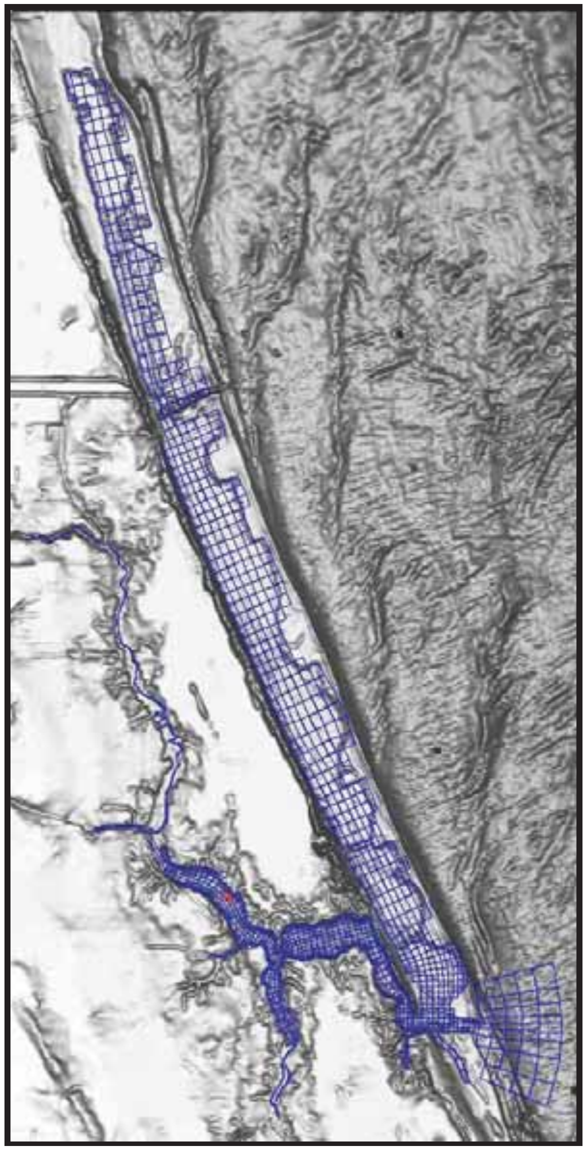


FIGURE 7-22. MODEL DOMAIN OF THE SOUTHERN INDIAN RIVER LAGOON AND ST. LUCIE ESTUARY

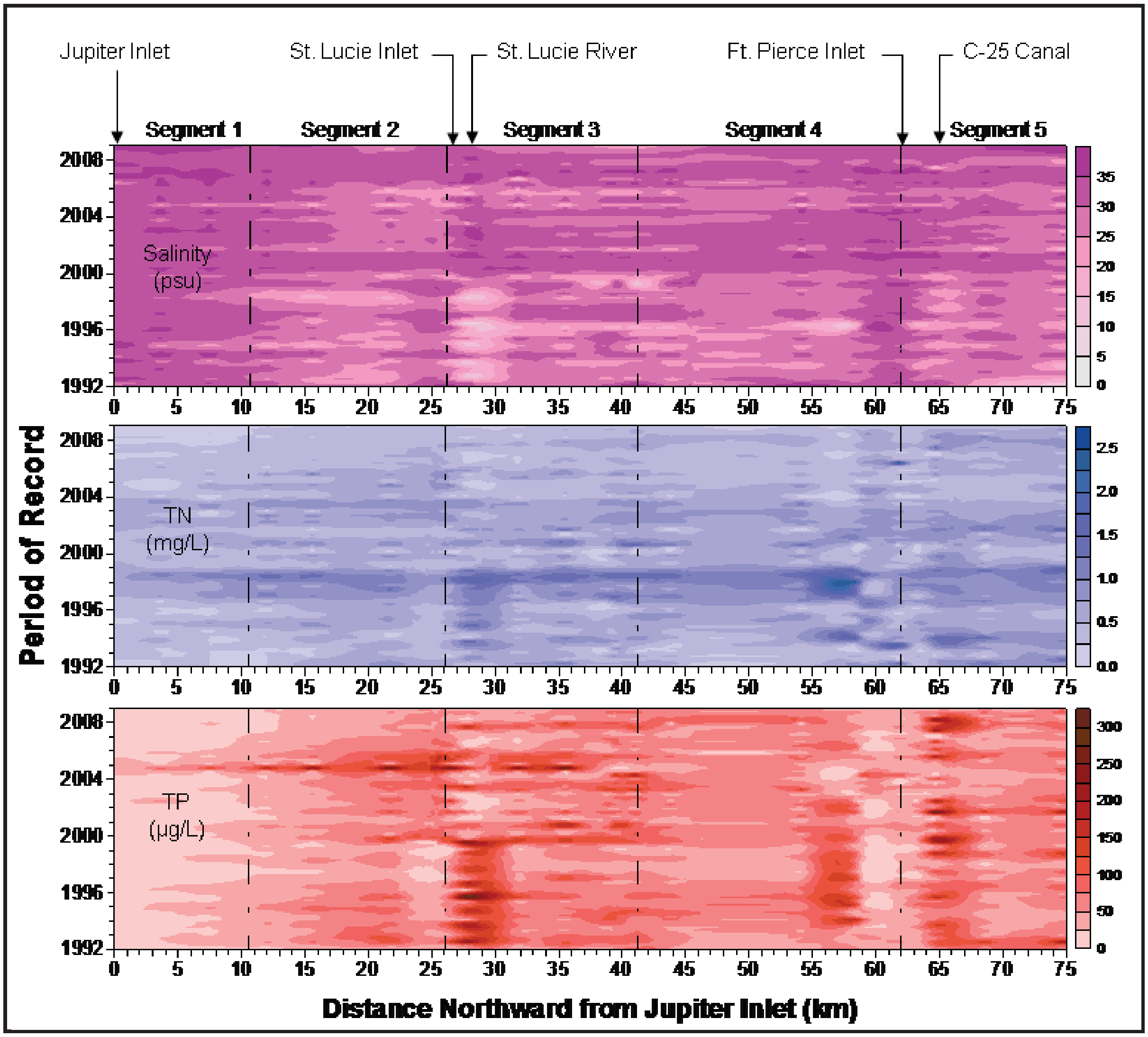


FIGURE 7-23. CONTOUR PLOTS SHOWING CHANGES IN SALINITY (TOP), TOTAL NITROGEN (MIDDLE) AND TOTAL PHOSPHORUS (BOTTOM) CONCENTRATIONS AS A FUNCTION OF DISTANCE FROM JUPITER INLET TO THE NORTHERN BOUNDARY OF THE SOUTHERN INDIAN RIVER LAGOON FROM 1992 THROUGH 2008

Note: Lighter colors are lower concentrations and darker colors are higher concentrations. Dashed lines are lagoon segment boundaries. Freshwater and marine water source locations are indicated by arrows at the top of the figure.

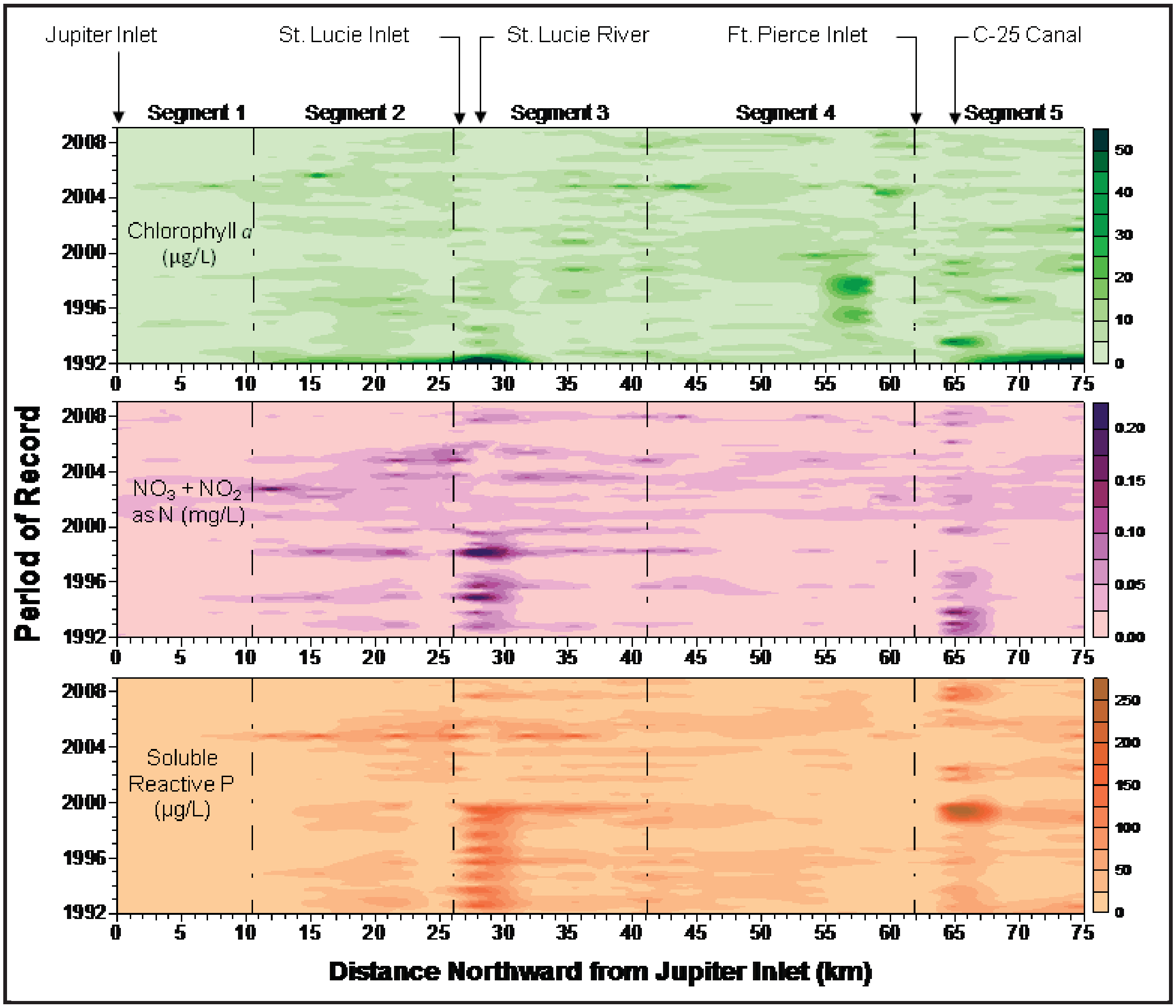


FIGURE 7-24. CONTOUR PLOTS SHOWING CHANGES IN CHLOROPHYLL *A* (TOP), NO3 + NO2 (OR NITRATE + NITRITE; MIDDLE) AND SOLUBLE REACTIVE PHOSPHORUS (BOTTOM) CONCENTRATIONS AS A FUNCTION OF DISTANCE FROM JUPITER INLET TO THE NORTHERN BOUNDARY OF THE SOUTHERN INDIAN RIVER LAGOON FROM 1992 THROUGH 2008

Key: Lighter colors are lower concentrations and darker colors are higher concentrations.
Dashed lines are lagoon segment boundaries.
Freshwater and marine water source locations are indicated by arrows at the top of the figure.

The lagoon segments shown in ***Figure 7-23*** and ***Figure 7-24*** are the same segments shown in ***Figure 4-55*** in ***Section 4.3 Submerged Aquatic Vegetation Hypothesis Cluster***, which follows this section, but they are numbered differently e.g. segment 1 in ***Figure 7-23*** and ***Figure 7-24*** corresponds to segment 26 in ***Figure 4-55*** segment 2 to segment 25.

All five segments in the Southern Indian River Lagoon exhibited an apparent decreasing trend for chlorophyll *a* from 1992 through 2008 (***Figure 7-25***). Only segment five exhibited a decreasing trend. Chlorophyll *a* levels in this segment decreased at a rate of approximately 0.2 ppb per year. Such small changes in chlorophyll levels may be due to the recent drought.

The extensive seagrass beds in the Southern Indian River Lagoon depend upon adequate water clarity to permit sunlight to promote adequate photosynthesis. In concert with water color, TSS affects the depth to which light can penetrate effectively. All of the lagoon segments exhibited apparent decreasing trend in TSS, with segments 3 and 4 exhibiting a statistically significant decrease (***Figure 7-26***).

The increasing water clarity evidenced by the decreasing trend in TSS is also evident in apparent color (***Figure 7-27***). All segments in the Southern Indian River Lagoon exhibited apparent decreasing trend in color, with segments 2 and 3 exhibiting a statistically significant decrease in apparent color at a significance level (α) of $\leq$0.05. All segments evidenced significance at the α=0.10 (or 90 percent probability) criteria. Together, these reductions in color and TSS should equate with increases in overall water clarity, expansion of seagrass, and to the degree that has occurred, increases in faunal utilization.

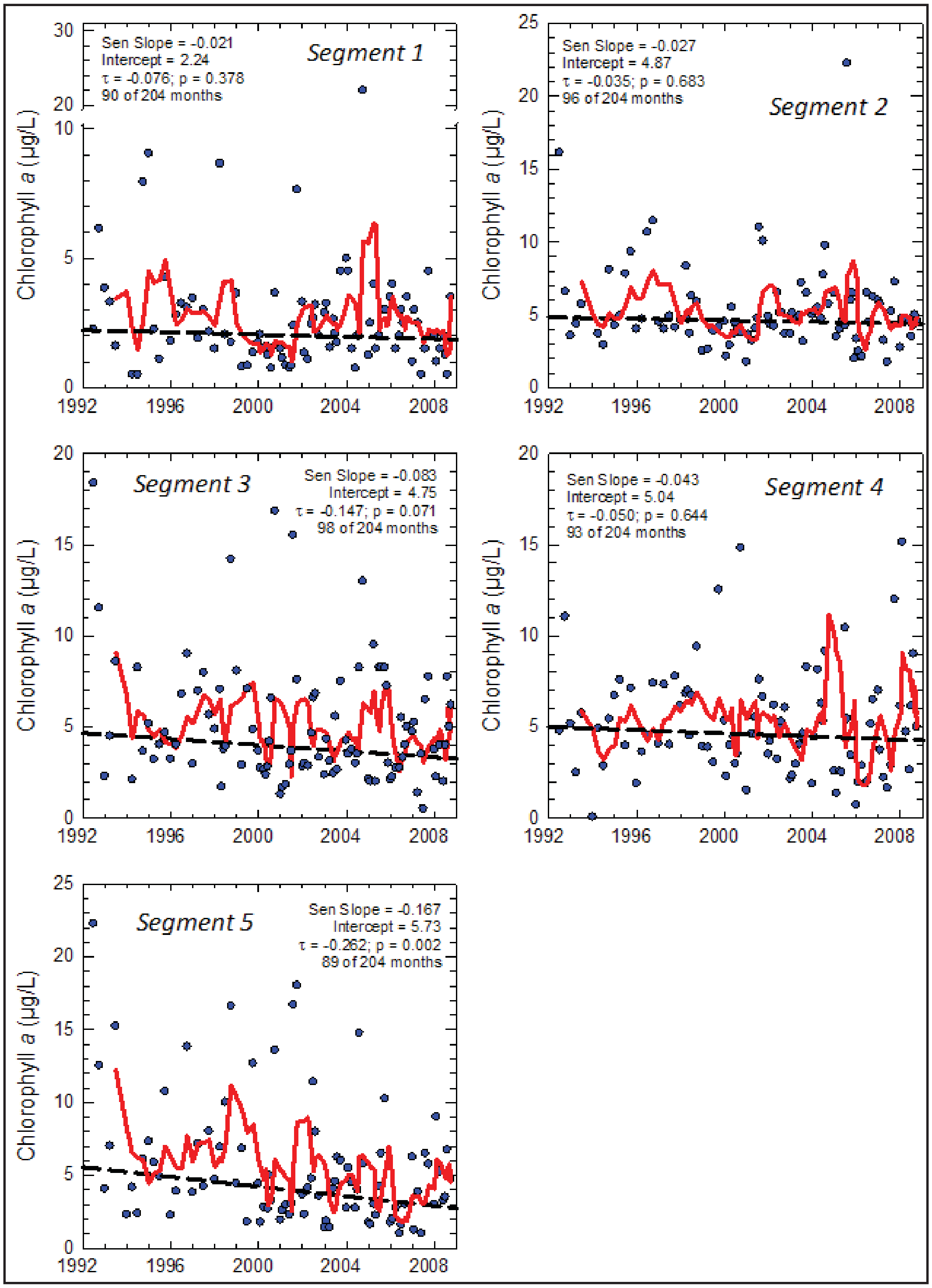


FIGURE 7-25. AVERAGE MONTHLY CHLOROPHYLL *A* LEVELS WITHIN EACH SEGMENT OF SOUTHERN INDIAN RIVER LAGOON

Key: The solid line represents the 12-month moving average concentration.
The dashed line represents the seasonal Kendall Tau trend line.
Probability (p) values less than 0.05 indicate statistical significance.

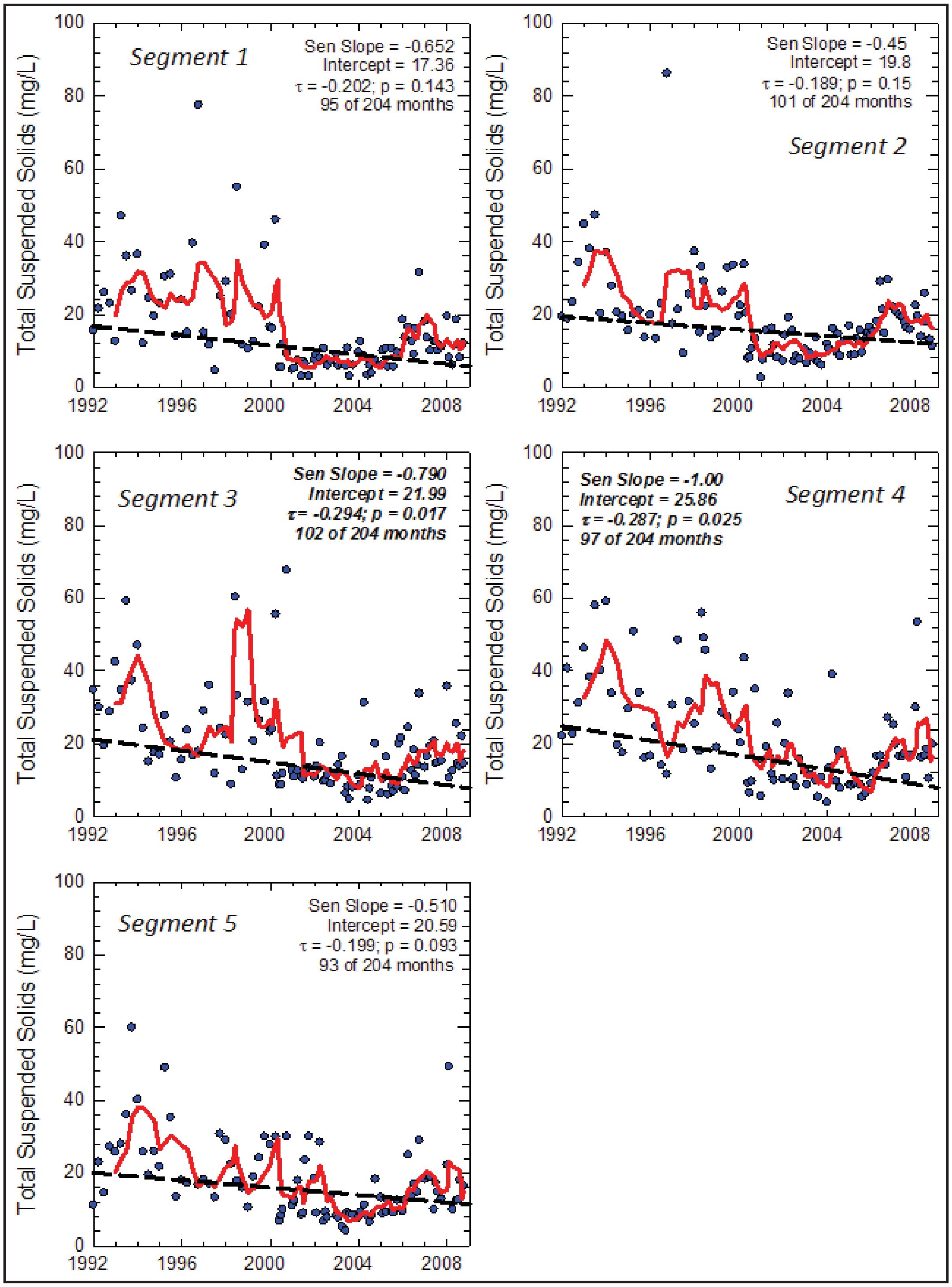


FIGURE 7-26. AVERAGE MONTHLY TOTAL SUSPENDED SOLIDS CONCENTRATIONS WITHIN EACH SEGMENT OF SOUTHERN INDIAN RIVER LAGOON

Key: The solid line represents the 12-month moving average concentration.
The dashed line represents the seasonal Kendall Tau trend line.
Probability (p) values less than 0.05 indicate statistical significance.

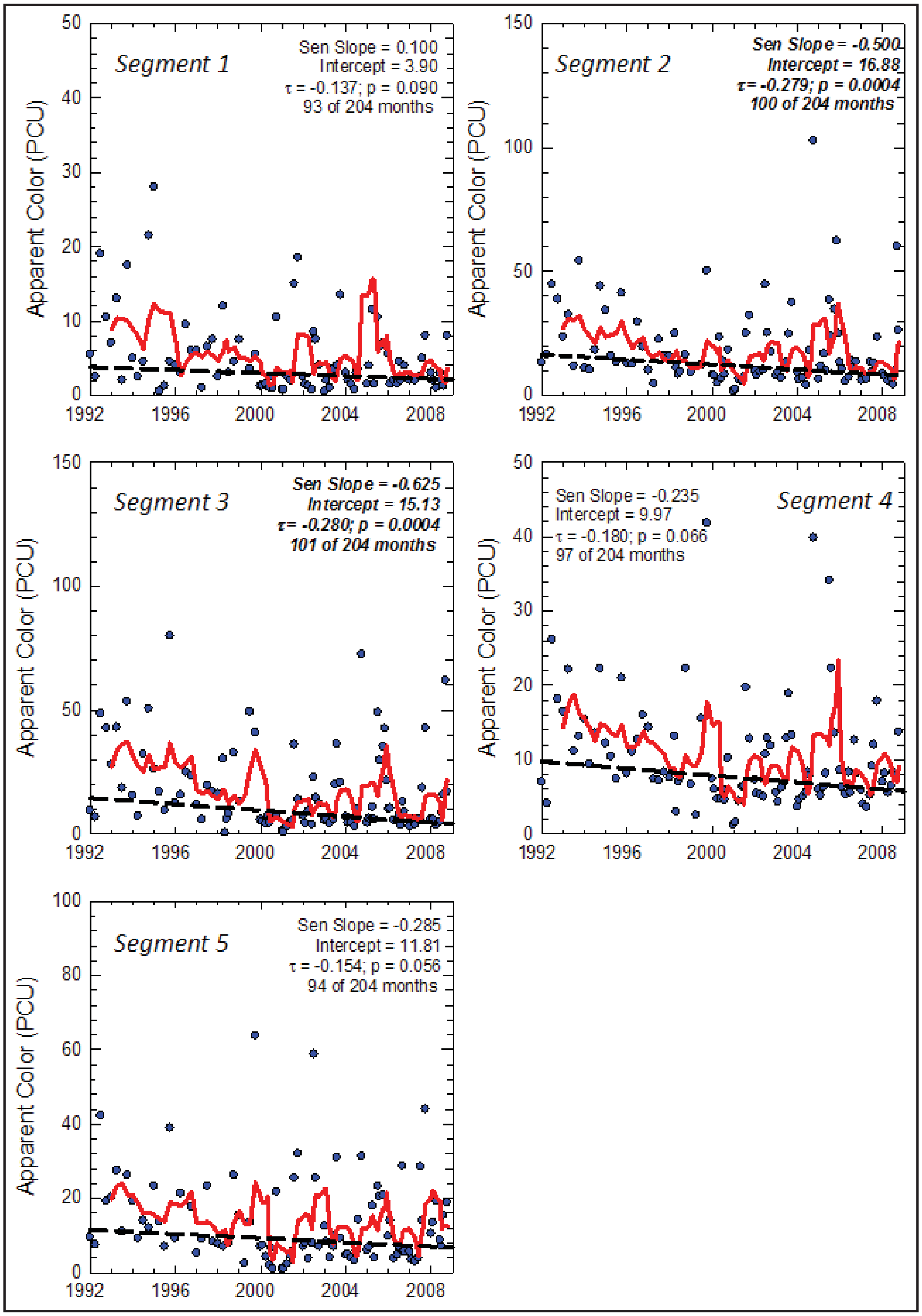


FIGURE 7-27. AVERAGE MONTHLY APPARENT COLOR LEVELS WITHIN EACH SEGMENT OF SOUTHERN INDIAN RIVER LAGOON

Key: The solid line represents the 12-month moving average concentration.
The dashed line represents the seasonal Kendall Tau trend line.
Probability (p) values less than 0.05 indicate statistical significance.

7.2.5 Loxahatchee River Estuary

7.2.5.1 Monitoring

Water quality monitoring sites within the Loxahatchee River Estuary are shown in ***Figure 7-28***.

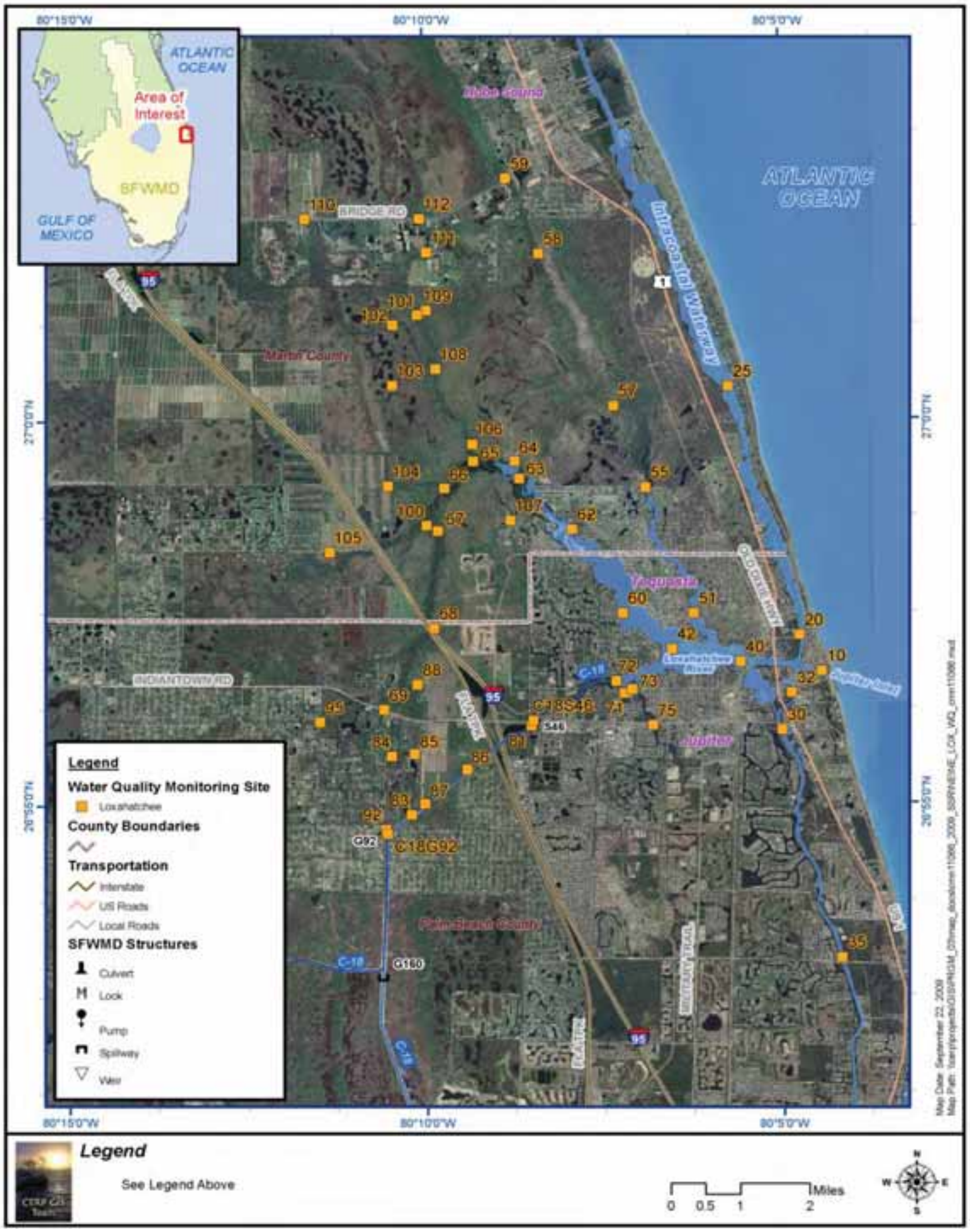


FIGURE 7-28. WATER QUALITY MONITORING SITES IN THE LOXAHATCHEE RIVER ESTUARY

7.2.5.2 Results

To the degree that flows have been sustained in the dry season in the Loxahatchee River Estuary‟s Northwest Fork, the relatively low levels of IN associated with those flows moves down into the open area of the estuary (***Figure 7-29***). In comparison, flows from the Southwest Fork are lower during the dry season and less influx of IN to the estuary is evident. With the advent of the wet season, the concentration of IN decreases, presumably a consequence of warmer temperatures, increased insolation and enhanced biological uptake. This is also observed in the Indian River Lagoon in Indian River County where clear waters and a sustained dry season inorganic source coincide (St. Johns River Water Management District data).

Table 7-1 presents the TP targets established for the Loxahatchee River Estuary and observed mean values based on available data (SFWMD, 2006). The wet and dry season TP regime is the general inverse of that observed in IN: low levels overall in the dry season and an increase in concentration as wet season flows transport the nutrient into the estuary (***Figure 7-30***). In both seasons, the dominant source of IN and TP was from the Northwest and Southwest Forks, with the North Fork contributing comparatively less (***Figure 7-29*** and ***Figure 7-30***).

TABLE 7-1. COMPARISON OF TOTAL PHOSPHORUS TARGETS AND OBSERVED MEAN VALUES BASED ON AVAILABLE 1999-2008 DATA FOR THE LOXAHATCHEE RIVER ESTUARY

	Total Phosphorus (ppb)				
	Marine	Polyhaline	Meso/Oligohaline	Wild and Scenic	Freshwater Tributaries
Target	25	38	56	46	51
Observed	24	37	60	51	51

Dry season algal blooms, as typified by chlorophyll *a* concentration, are higher in the upper reaches of the Southwest Fork, and this pattern extends into the wet season (***Figure 7-31***). The Loxahatchee River Estuary possesses good water clarity with Secchi depths often over a meter. Sunlight is able to penetrate to substantial depth in the water column and the presence of slightly elevated available N and P can result in algal blooms. In the wet season, the estuary west of site 42 evidences a chlorophyll *a* concentration on the order of nine ppb chlorophyll *a* or above, which is worrisome given the criteria for estuarine impairment is 11 ppb. Average annual chlorophyll *a* concentration at site 72, which includes both wet and dry season values, is 11.7 µg/L. As restoration proceeds changes in water quality should be examined to ensure that desirable outcomes in ecological function are realized.

Data from ten sites sampled on a monthly basis since 2007 and bi-monthly from 1999 through 2006 were examined for trends in IN, TP, dissolved oxygen or chlorophyll *a* (***Figure 7-32***). Few were statistically significant (Sen, 1968). Of those that were, the rate of change was, for all ecological purposes, essentially zero (e.g. 10 to 5 mg/L per year). The peak in TP observed at site 65 may be a consequence of P desorbing from particles where fresh and saltier water mix under the current flow regime. The depression in chlorophyll *a* observed at site 67 is likely a consequence of shading. Dissolved oxygen regimes typically exceed the 4 mg/L criteria, and chlorophyll *a* concentrations are generally below the 11 ppb criteria.

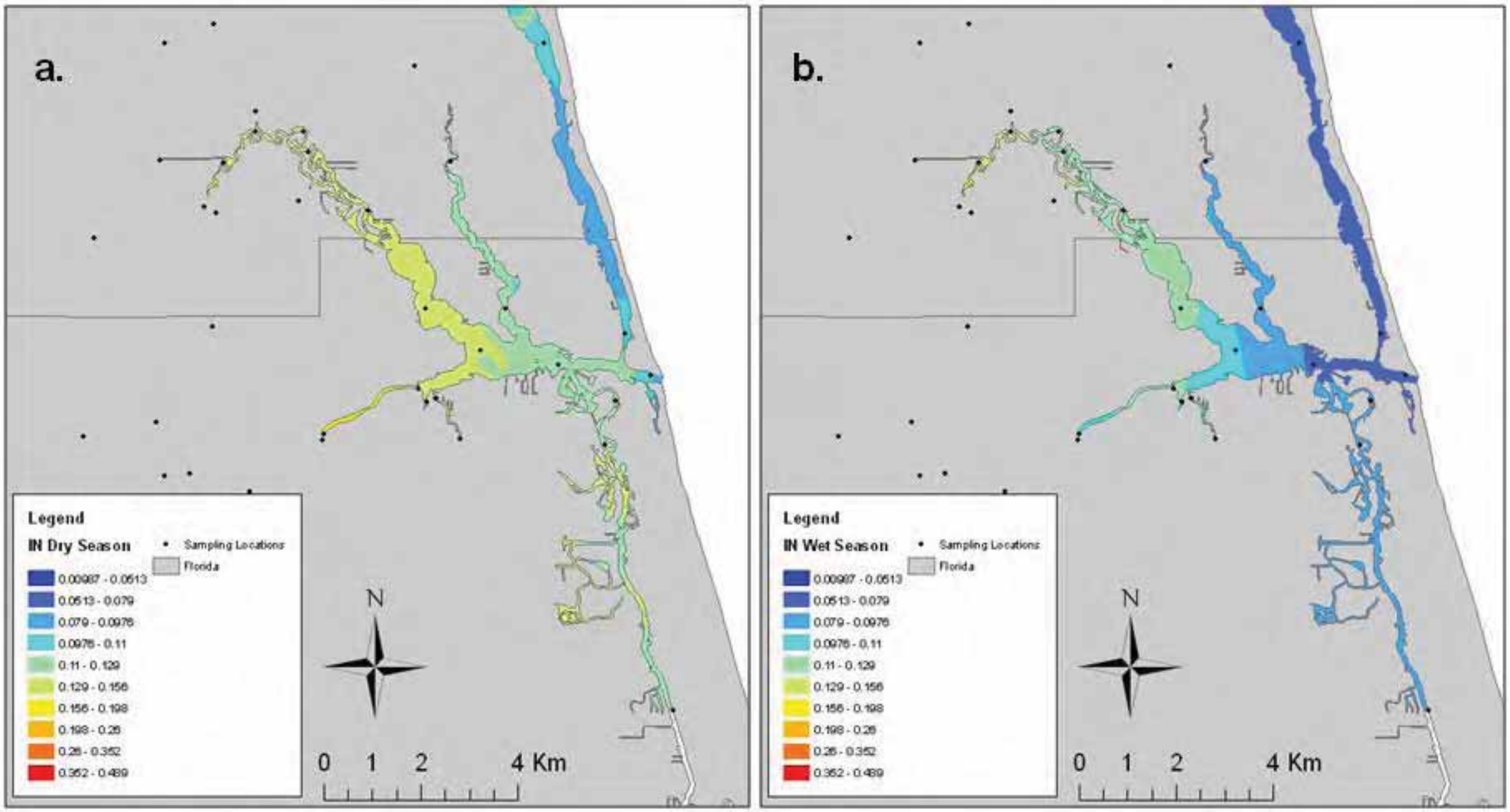


FIGURE 7-29. MEAN A) DRY AND B) WET SEASON IN CONCENTRATION IN THE LOXAHATCHEE RIVER ESTUARY BASED ON 1999-2008 DATA

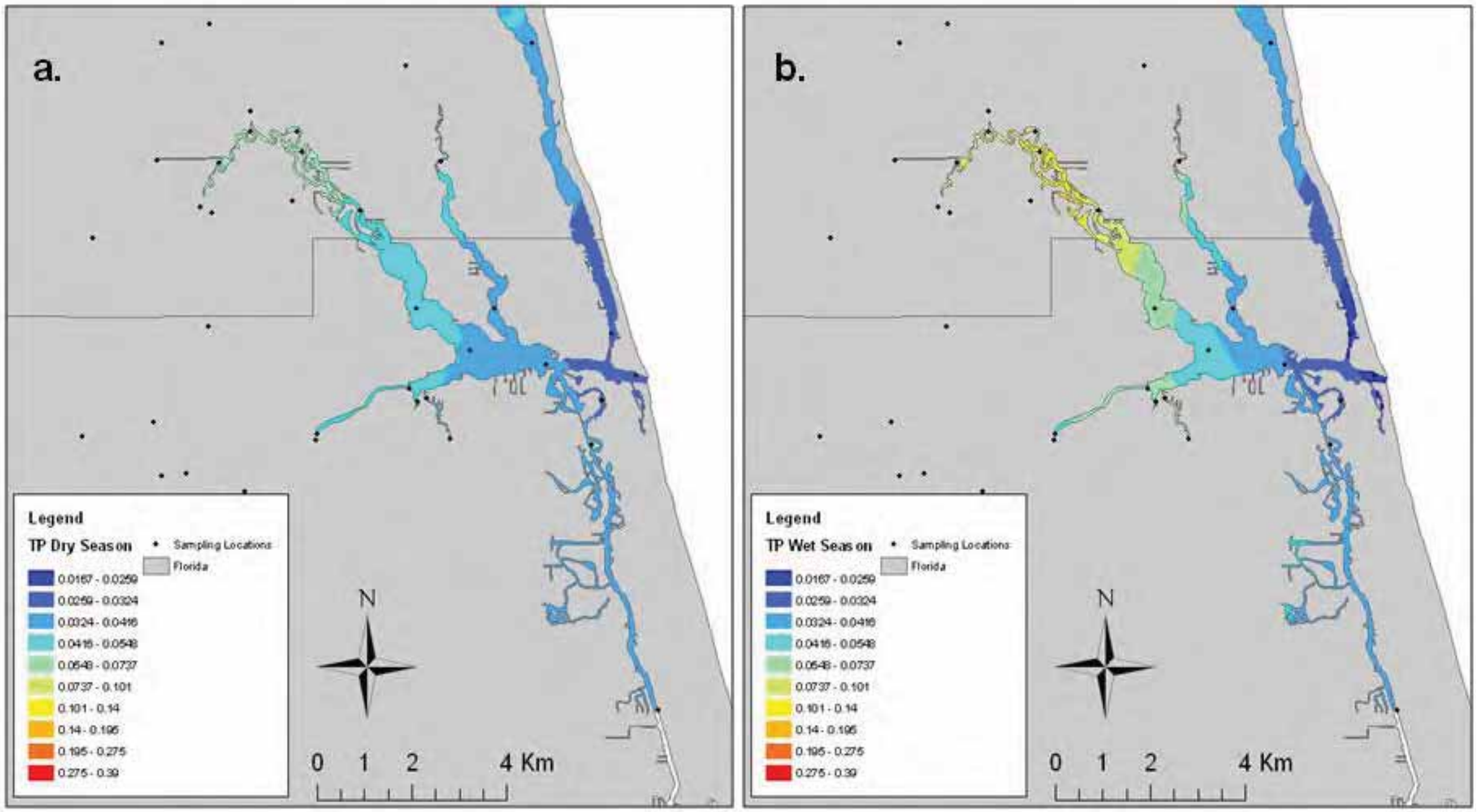


FIGURE 7-30. MEAN A) DRY AND B) WET SEASON TP CONCENTRATION IN THE LOXAHATCHEE RIVER ESTUARY BASED ON 1999-2008 DATA

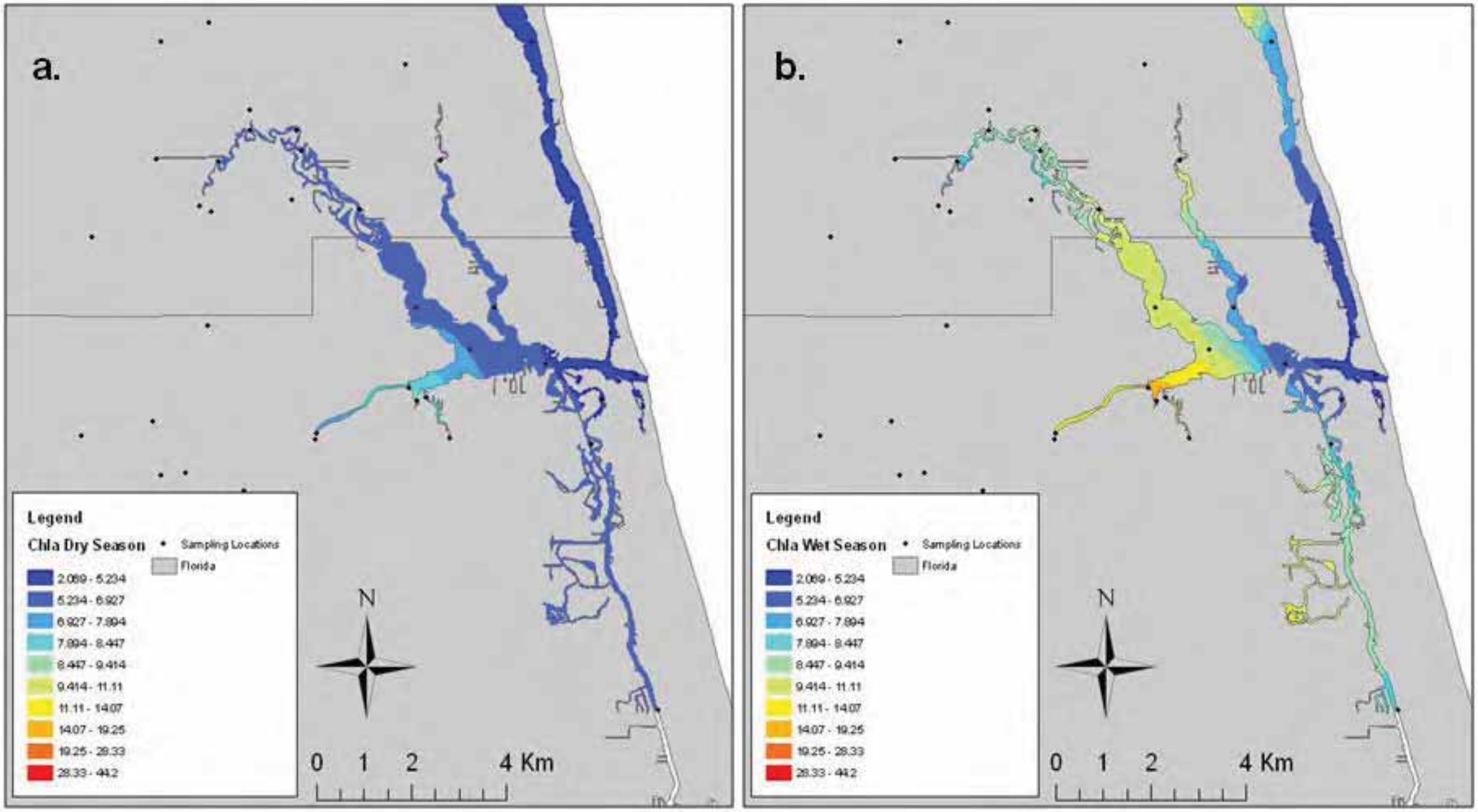


FIGURE 7-31. MEAN A) DRY AND B) WET SEASON CHLOROPHYLL *A* CONCENTRATION IN THE LOXAHATCHEE RIVER ESTUARY BASED ON 1999-2008 DATA

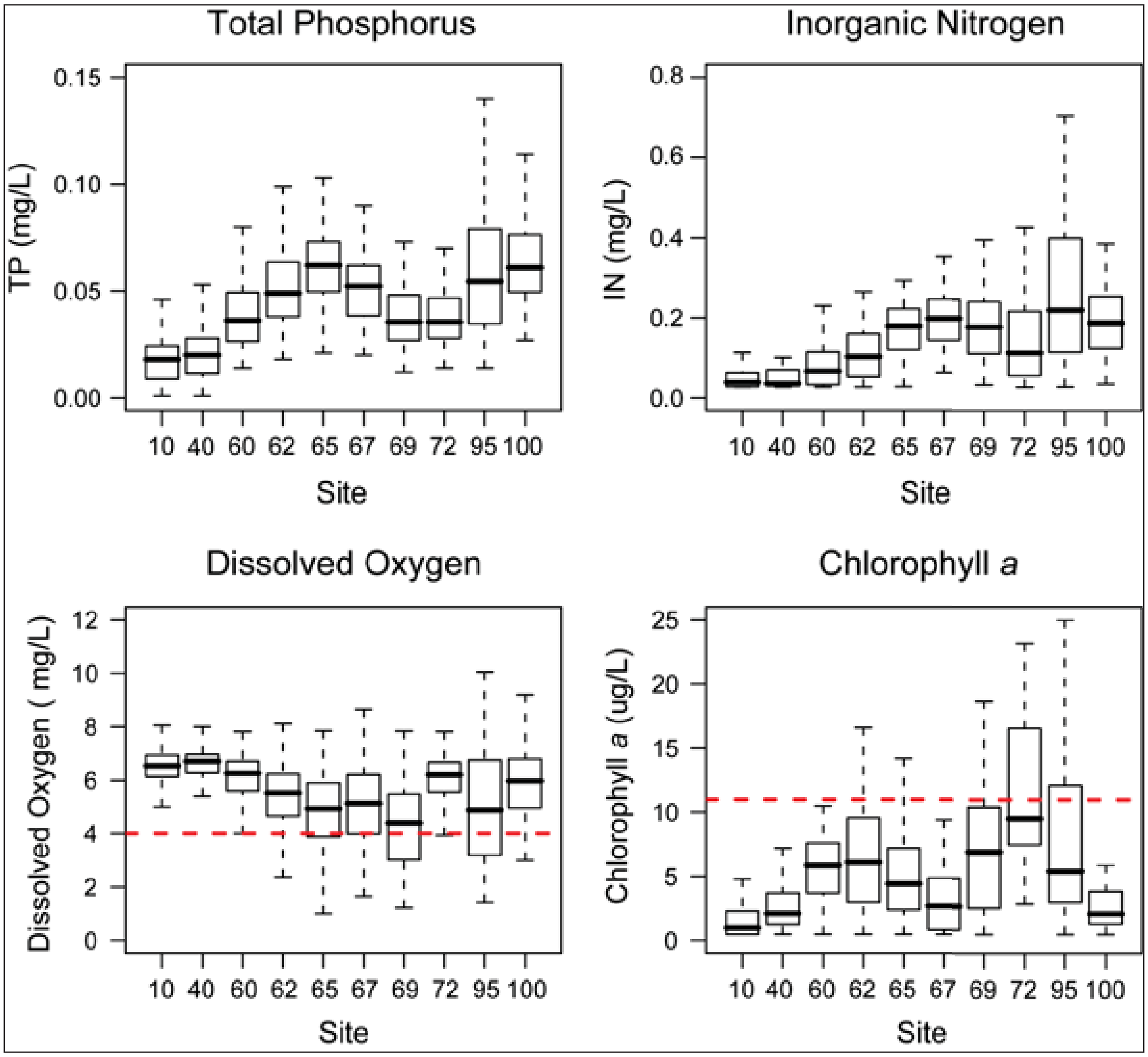


FIGURE 7-32. COMPARISON OF TOTAL PHOSPHORUS, INORGANIC NITROGEN, DISSOLVED OXYGEN AND CHLOROPHYLL *A* IN THE LOXAHATCHEE RIVER ESTUARY BASED ON DATA FROM 1999 THROUGH 2008

Of these selected ten site sites, site 72 evidenced the highest chlorophyll *a* concentration, yet exhibited a TP and IN regime lower than any of the other sites except sites 10 and 40, which are near the inlet. The average Secchi depth at site 72 is 1.1 meters, which seems to support the assertion that relatively low concentration of nutrients in exceptionally clear estuarine water can be problematical insofar as chlorophyll *a* concentrations are concerned. The idea that algae are uniformly distributed in the photic zone when chlorophyll *a* samples are collected is difficult to sustain given that both flagellated and non-flagellated algal species can be highly motile. Samples collected near the top of the water column during daylight may reflect this mobility, i.e., high chlorophyll *a* in moderate nutrient-rich waters due to large volume habitat and advantageous behavior on the part of the algae to obtain more light.

7.2.6 Lake Worth Lagoon

7.2.6.1 Monitoring

Water quality monitoring sites within the Lake Worth Lagoon are shown in ***Figure 7-33***.

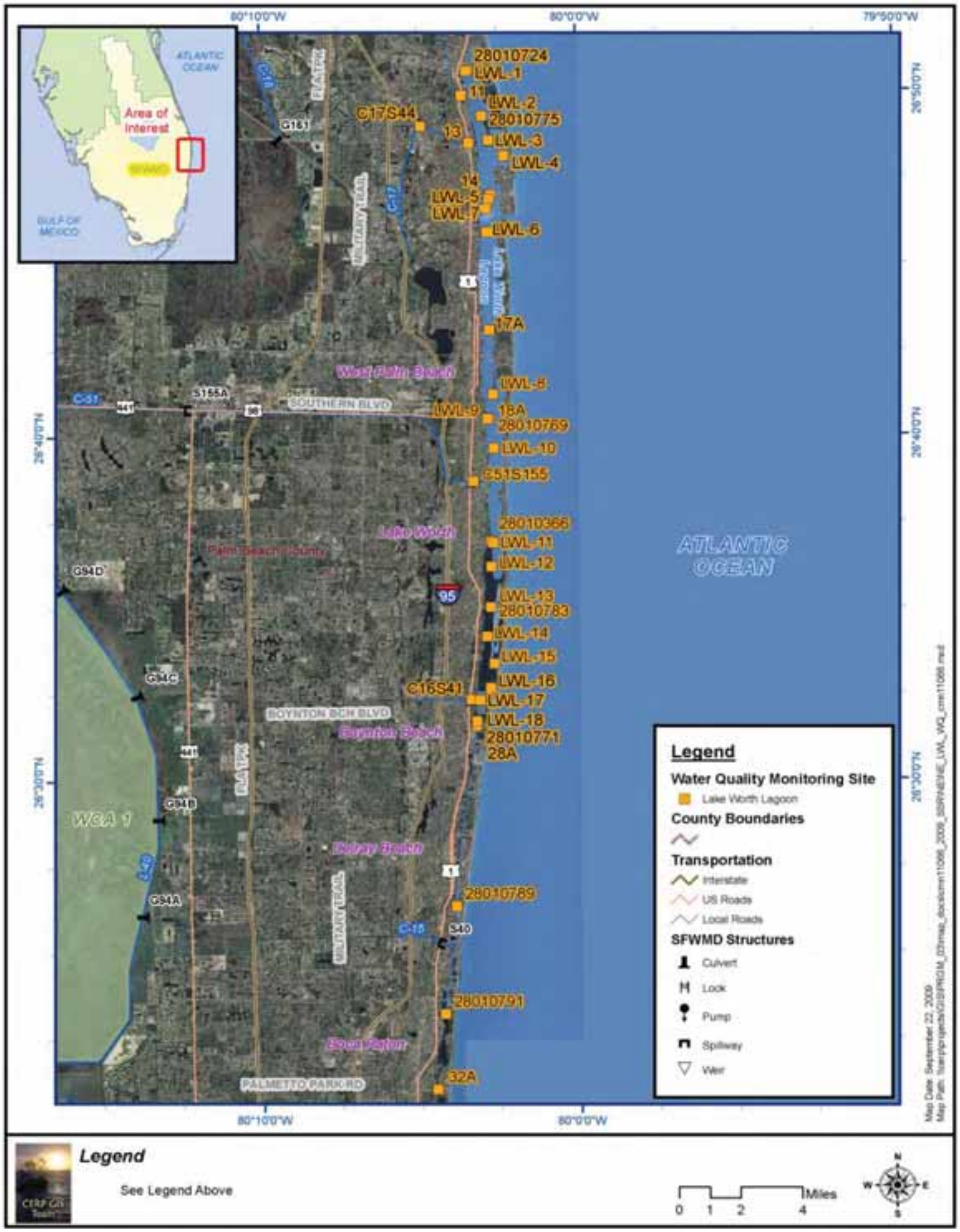


FIGURE 7-33. WATER QUALITY MONITORING SITES IN LAKE WORTH LAGOON

The water quality monitoring design for Lake Worth Lagoon has undergone several iterations. Initially, the monitoring began as an initiative of the Palm Beach County Environmental Resources Management (PBCERM) program in 1990 as part of the NPDES permitting process. The FDEP expanded on the monitoring effort and adopted a slightly different monitoring configuration from 2000 to 2006. More recently (starting in 2007), a joint review of the monitoring program by PBCERM and SFWMD resulted in a reconfiguration of sampling stations and modification of the monitoring plan. As a result of these changes, the present water quality monitoring in the Lake Worth Lagoon (LWL) is more consistent based on parameters measured and methods used. The overall effect of these changes in data collection and analyses resulted in a dataset where interpretation may be more affected by changes in monitoring plans rather than ecological changes. To the degree that long-term individual stations are sustained in the coming years, these problems will increasingly dissipate. ***Figure 7-33*** depicts sites from previous PBCERM and FDEP activities as well as the ongoing PBCERM/SFWMD sites, which are prefixed with “LWL-”.

7.2.6.2 Results

Evaluating the lagoon as a whole over the last decade indicates that the annual oxygen levels are predominantly above the 4.0 mg/L criteria for Class III predominantly marine waters (***Figure 7-34***). Annual chlorophyll a levels are generally below the 11 µg/L (or ppb) criteria for estuarine systems (***Figure 7-34***). A generally decreasing TP trend (Sen slope = 5 µg P/L per year) is apparent. This decrease in P concentration over the later part of the decade is also evident in the chlorophyll *a* concentrations, which also exhibit evidences a decreasing trend (Sen slope = 0.25 µg/L per year). Trends were not apparent for TN nor dissolved oxygen regimes.

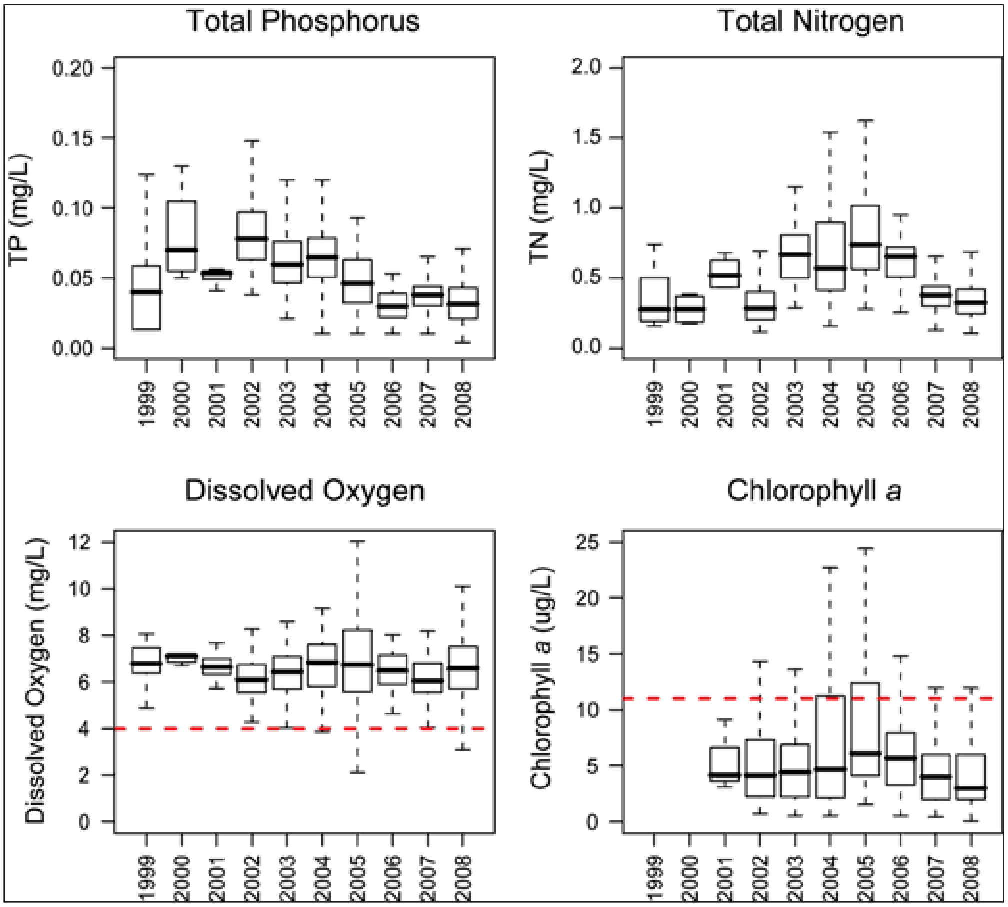


FIGURE 7-34. COMPARISON OF YEAR-TO-YEAR (1999-2008) CHANGES IN TOTAL PHOSPHORUS, TOTAL NITGROGEN, DISSOLVED OXYGEN AND CHLOROPHYLL *A* CONCENTRATIONS FOR LAKE WORTH LAGOON

An examination of seasonal differences in nutrient concentrations suggests that concentrations are slightly higher during the wet season (i.e., May through October) (***Figure 7-35***). In response to increased nutrients and high water temperatures, chlorophyll *a* levels appear to peak in August and continue somewhat abated through October. However, the average chlorophyll levels for any month stays below the 11 µg/L TMDL criteria. Dissolved oxygen is lowest in September. The most variable dissolved oxygen level in the Lake Worth Lagoon are observed during the month of August and may be associated with increased algal levels observed for this month.

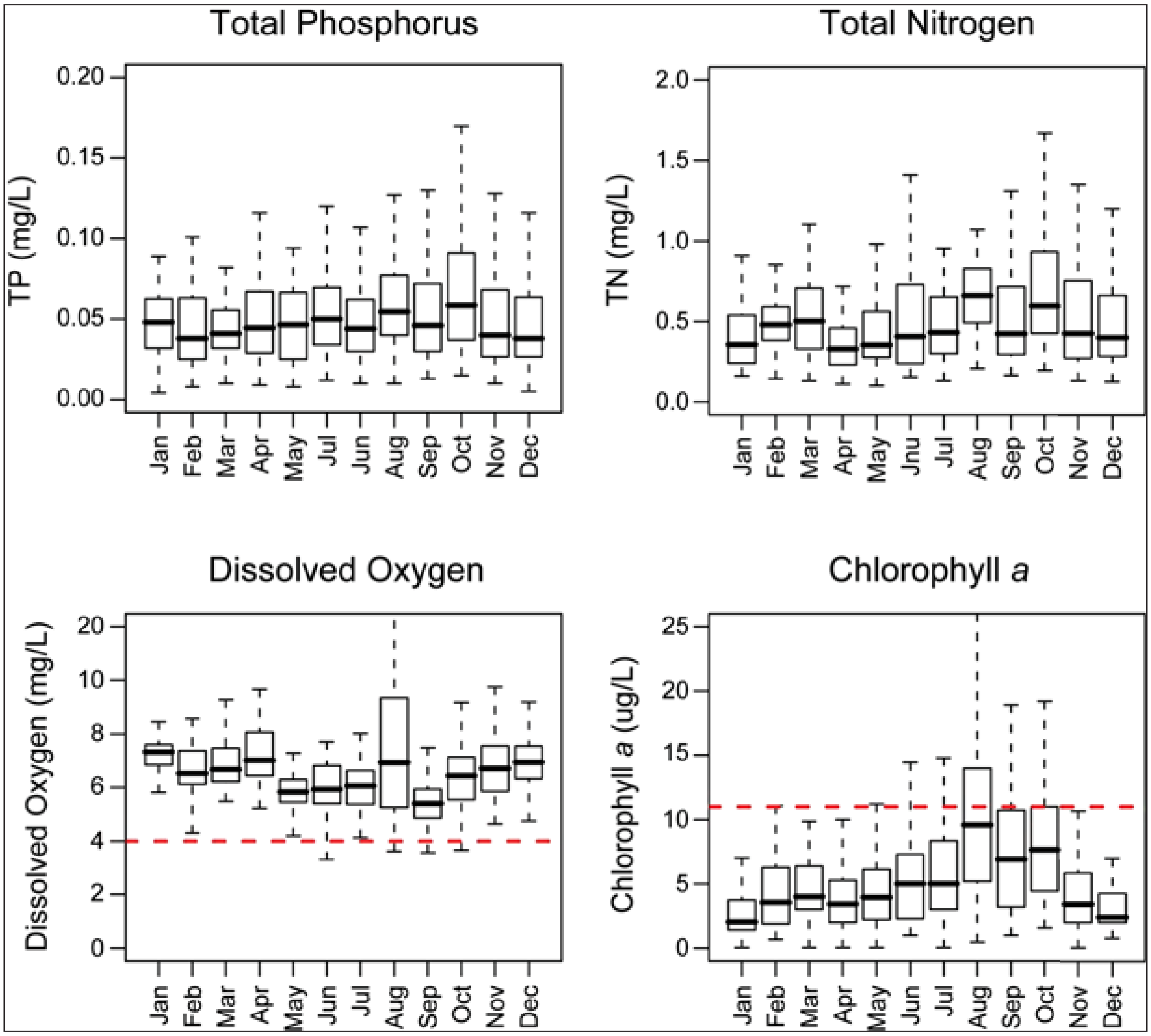


FIGURE 7-35. COMPARISON OF SEASONAL (MONTH-TO-MONTH) CHANGES IN TOTAL PHOSPHORUS, TOTAL NITROGEN, DISSOLVED OXYGEN AND CHLOROPHYLL *A* CONCENTRATIONS FOR LAKE WORTH LAGOON BASED ON 1999-2008 DATA

The impact of the C-17 Canal and/or other sources further north in the Intracoastal Waterway appear to present a slight impact on the lagoon"s DIN regime in both the dry and wet seasons, with little difference in apparent extent (***Figure 7-36***). At the southern end of the lagoon, the C-16 Canal is associated with a similar apparent increase in IN that is maintained in both the dry and wet seasons, but again this pattern may be due to sources of N further south. The pattern due to the C-51 Canal might be explained by low volume releases during the wet season that were relatively high in N coupled with lack of flushing in the dry season, and/or build up of IN in the canal during no flow conditions. In the wet season, the DIN zone around the C-51 Canal is more extensive both to the north and south, but concentrations are lower, suggesting higher flows and increased flushing.

The C-17 Canal and/or other sources that affect the north end of the lagoon have little impact on the P regime in the wet season (***Figure 7-37***). Wet season differences are apparent but the concentration in the lagoon is low, on the order of less than a mean concentration of 0.05 mg/L. As far as TP concentrations are concerned, the southern end of the lagoon exhibits the highest concentrations, particularly in the wet season.

In the dry season, the northern end of the lagoon has few problems with algal blooms (***Figure 7-38***). The high dry season chlorophyll *a* levels near the C-51 Canal as depicted in this figure is due to blooms occurring in the canal itself. During the wet season, the southern two-thirds of the lagoon experiences moderate blooms, albeit on average, with less than a 11 µg/L chlorophyll *a* concentration.

The apparent wet season contribution of P to the lagoon from the C-16 Canal (***Figure 7-37***) is substantiated (***Figure 7-39***, top right panel), where the highest P concentrations observed are being discharged by the C-16 Canal in the wet season. The high concentration in the dry season DIN that appears associated with C-51 Canal in ***Figure 7-36*** is substantiated by the elevated DIN concentrations that occur in the C-51 Canal during the dry season in the middle bottom panel of ***Figure 7-39***. These dry season IN values (e.g., January and December) are higher than those observed elsewhere in these canals.

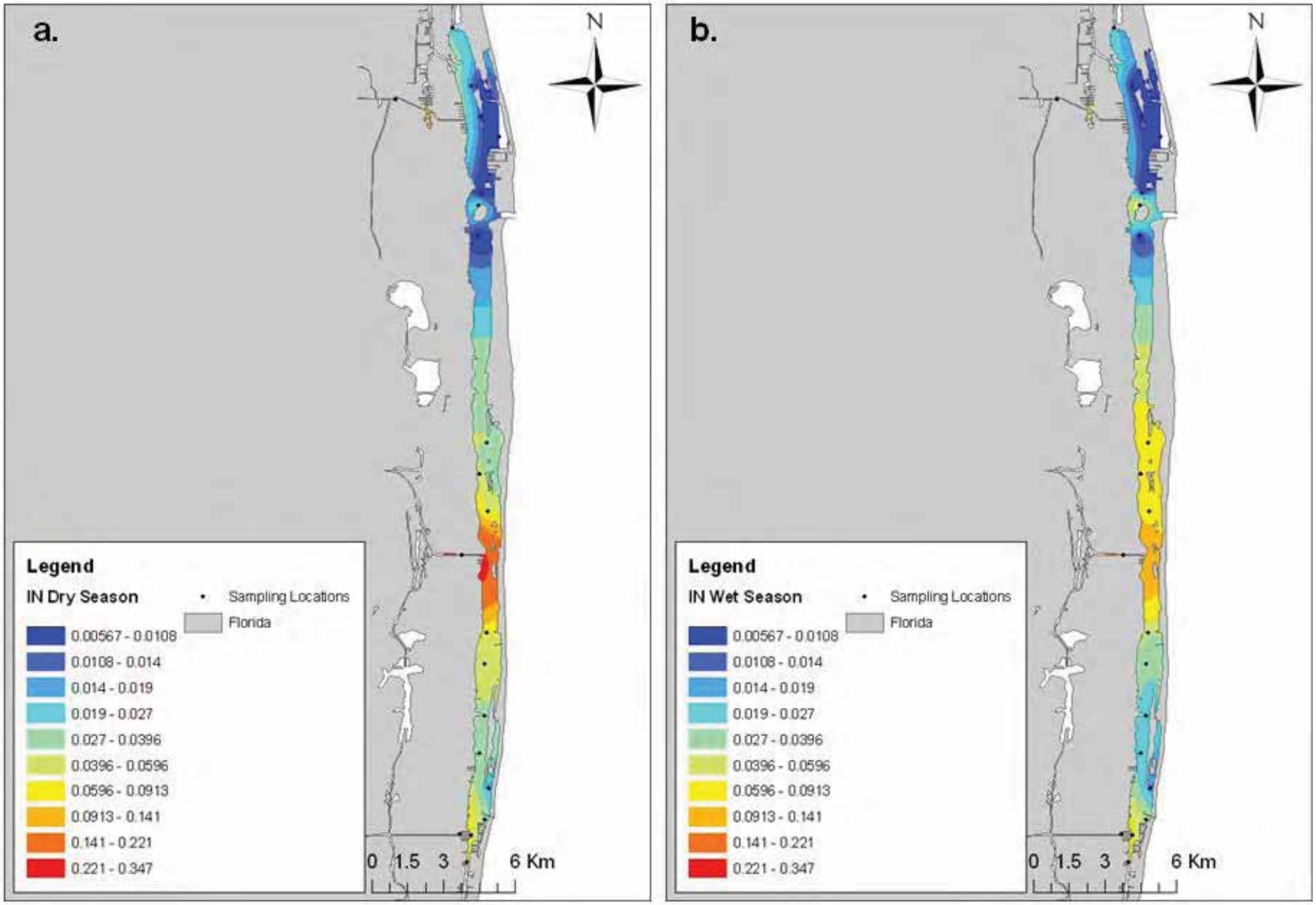


FIGURE 7-36. MEAN A) DRY AND B) WET SEASON IN CONCENTRATION IN LAKE WORTH LAGOON BASED ON 1999-2008 DATA

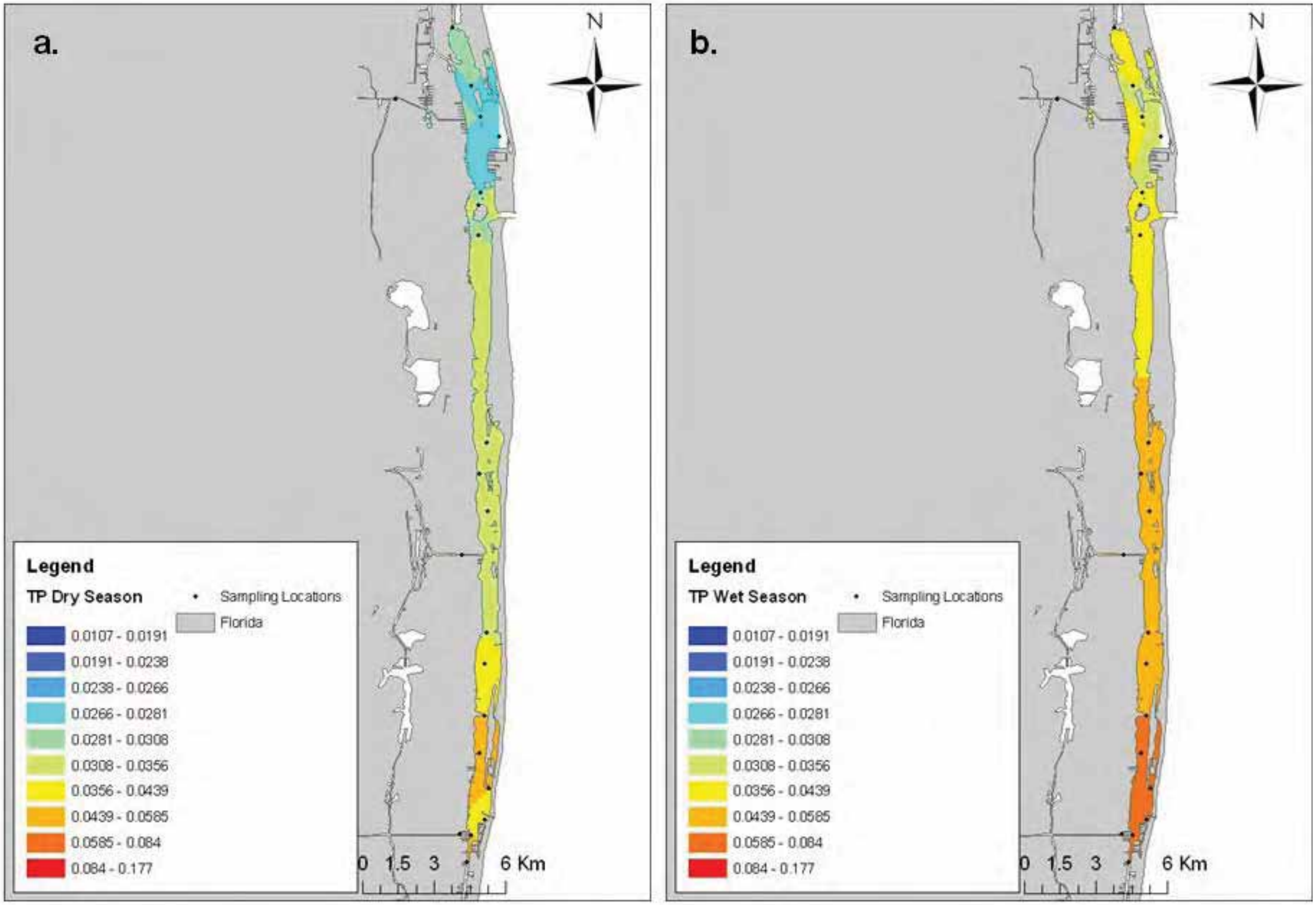


FIGURE 7-37. MEAN A) DRY AND B) WET SEASON TP CONCENTRATION IN LAKE WORTH LAGOON BASED ON 1999-2008 DATA

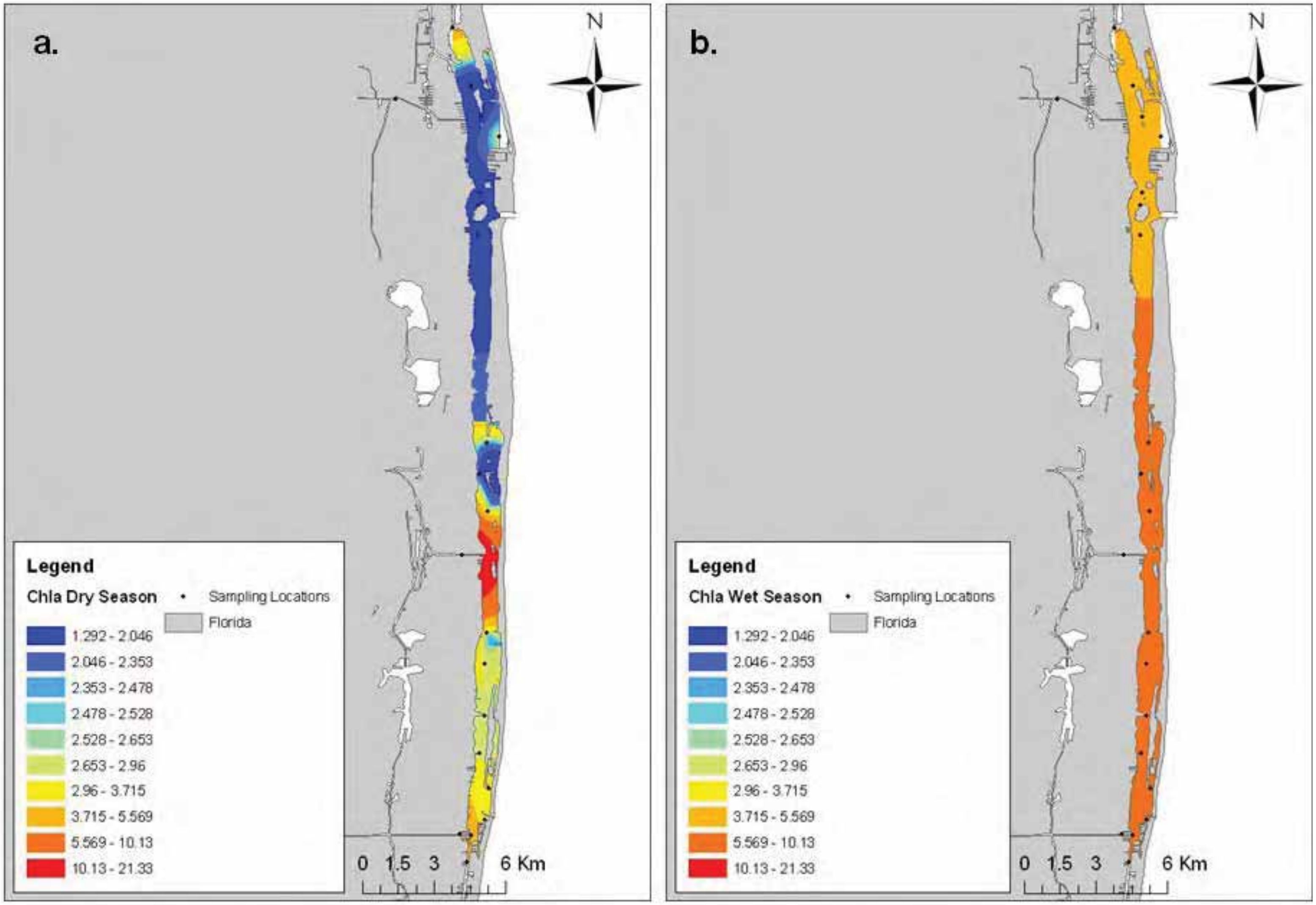


FIGURE 7-38. MEAN A) DRY AND B) WET SEASON CHLOROPHYLL *A* CONCENTRATION IN LAKE WORTH LAGOON BASED ON 1999-2008 DATA

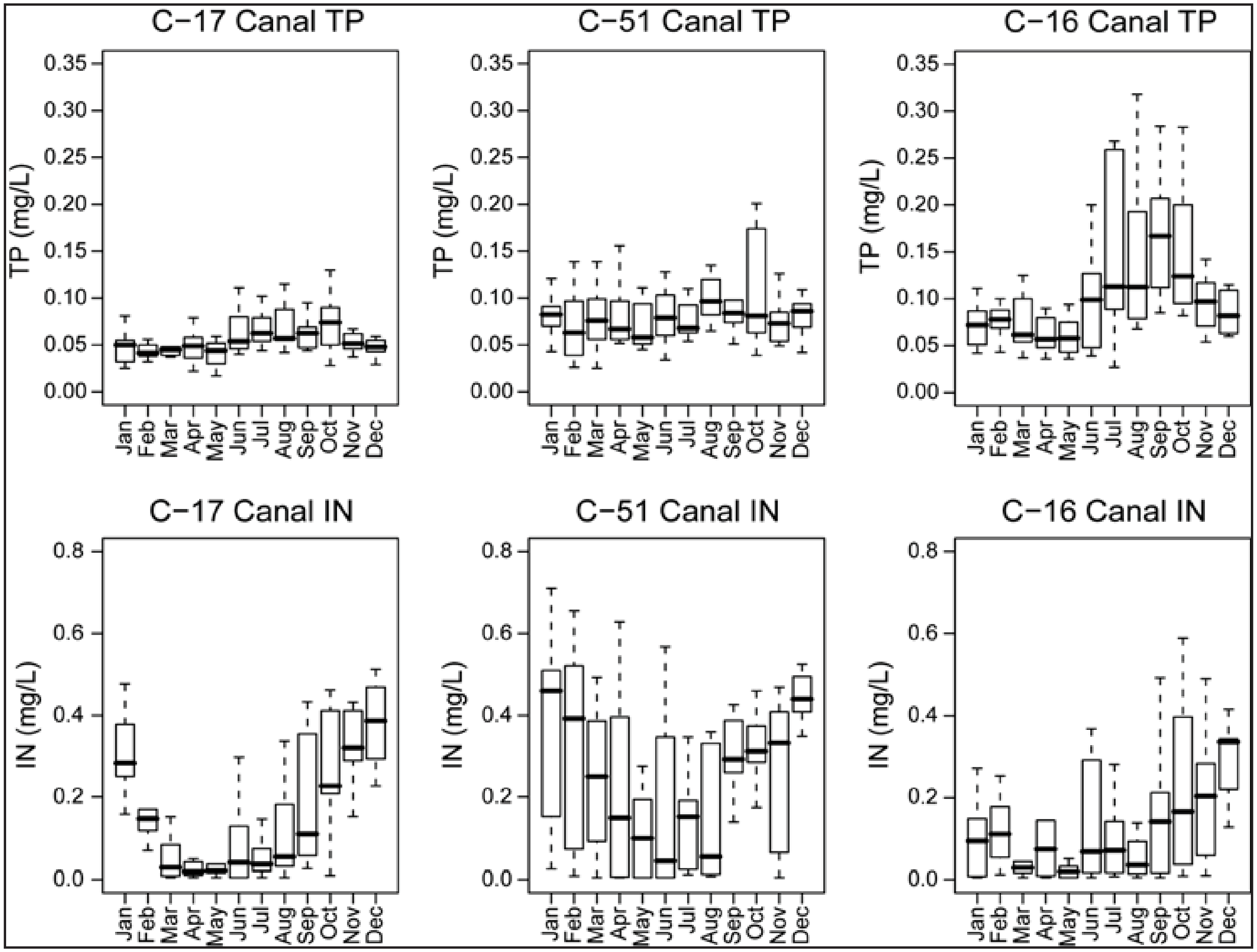


FIGURE 7-39. COMPARISON OF MONTHLY TOTAL PHOSPHORUS AND INORGANIC NITROGEN CONCENTRATIONS IN THE THREE MAJOR CANALS THAT DISCHARGE TO LAKE WORTH LAGOON BASED ON 1999 -2008 DATA

7.3 SUBMERGED AQUATIC VEGETATION HYPOTHESES CLUSTER

7.3.1 Introduction and Background

SAV is a valued ecosystem component (VEC) within the Northern Estuaries. These estuaries are the Caloosahatchee River Estuary, including San Carlos Bay, on the west coast of Florida, and Southern Indian River Lagoon, St. Lucie Estuary, Loxahatchee River Estuary, and Lake Worth Lagoon on the east coast.

The quantity and quality of freshwater discharge influence SAV presence or absence, community structure, aerial extent and productivity (***Figure 7-40***). Salinity is lowered, nutrient loads are increased, and sediments are resuspended by increased freshwater flow during the wet season as water is released into estuaries to provide flood protection. During the dry season, salinities are higher because fresh water that once flowed through the estuaries is now used for water supply. A more detailed description of this hypothesis cluster can be found in the MAP, Part 2 2006 Assessment Strategy for the MAP (RECOVER, 2006; www.evergladesplan.org/pm/recover/recover_map_part2.aspx) and the draft Revised CERP Monitoring and Assessment Plan: Part 1 Monitoring and Supporting Research (RECOVER, 2008; www.evergladesplan.org/pm/recover/recover_map_2008.aspx).

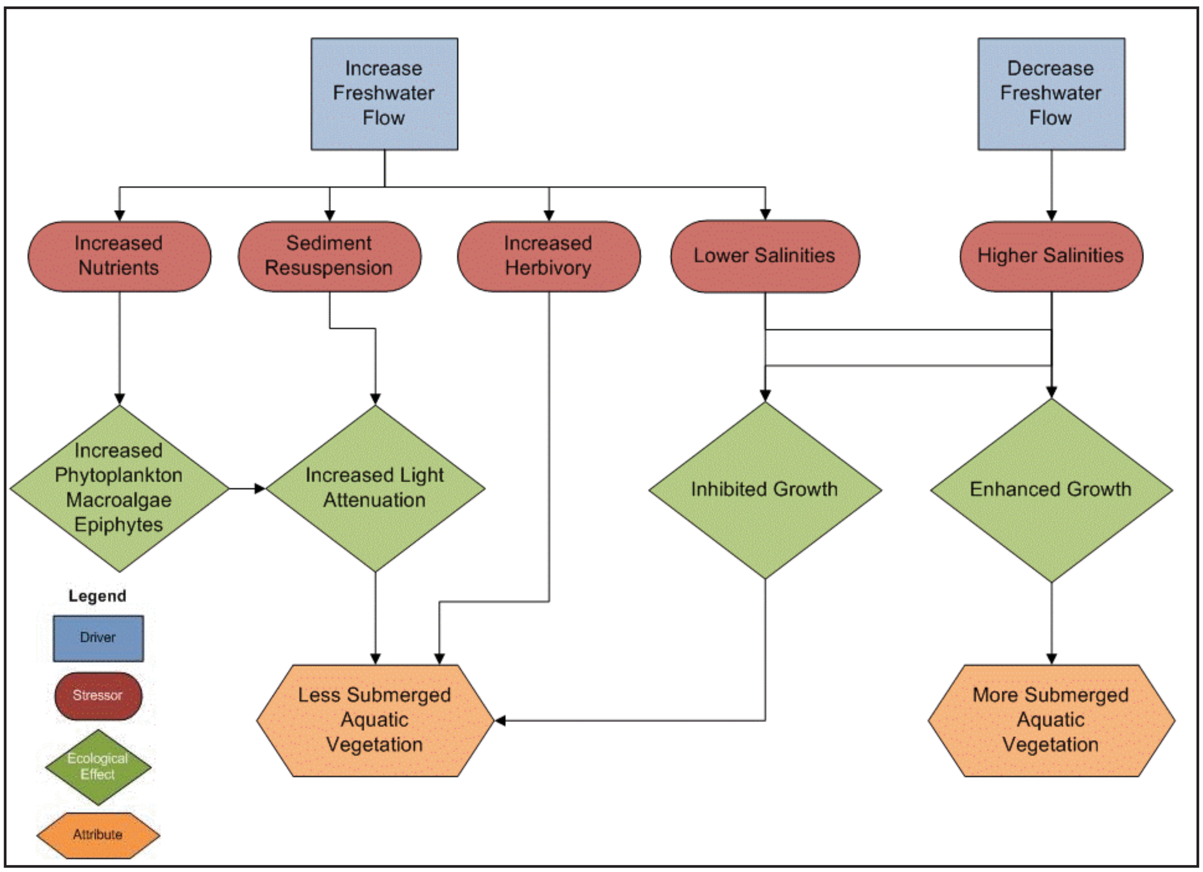


FIGURE 7-40. CONCEPTUAL ECOLOGICAL MODEL FOR SUBMERGED AQUATIC VEGETATION IN THE NORTHERN ESTUARIES

CERP and other restoration projects within the Northern Estuaries and Lake Okeechobee regions are expected to improve the timing, quality and quantity of water released to the Northern Estuaries, which, in turn, is expected to improve the health of SAV within these estuaries. SAV must be monitored and assessed prior to and during restoration implementation to determine if restoration goals for SAV are being met. These goals include an increase in the areal extent of SAV where it has been lost, improvement of SAV functionality, and restoration of SAV natural temporal and spatial dynamics. The RECOVER Program has developed a performance measure for Northern Estuaries SAV, which can be viewed online at www.evergladesplan.org/pm/recover/recover_docs/et/ne_pm_sav.pdf. Northern Estuaries SAV is also an interim goal (RECOVER 2005), which can be viewed online at www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_1-2_nesav.pdf.

7.3.2 Monitoring

Aerial images are used to capture SAV change at a landscape scale. Aerial images should be acquired every five years to calculate total SAV area. This portion of SAV area is compared with an estimate of total area of appropriate SAV habitat. To augment this landscape-scale data, species specific SAV percent cover is assessed every two to three years at the same sites used for patch-scale sampling (described below) using 9-m^2 quadrats.

Patch-scale monitoring is also necessary because SAV response to restoration activities is expected to be species specific. Most of the patch-scale data presented in this report was conducted using fixed transects. A new SAV patch-scale monitoring methodology has been developed and is now being used bimonthly in all of the Northern Estuaries. The new methods include haphazardly deploying 30 1-m^2 quadrats within specified boundaries at each site. Percent occurrence of seagrass species and macro-algal functional groups are determined within 25 quadrants of the 1-m^2 quadrats. Additionally, canopy height and quadrat location are documented. Data collected by both patch-scale methods include species-specific percent cover and canopy height estimates. Ancillary water quality data is also collected at each sampling site.

7.3.3 Caloosahatchee River Estuary and San Carlos Bay

The data presented in this report is from patch-scale SAV and water quality sampling undertaken at seven sites using fixed transects within the upper and lower Caloosahatchee River Estuary and San Carlos Bay (***Figure 7-41***) as part of a monthly sampling program from 2004 to 2008. Landscape-scale mapping has not been done yet for this estuary. Based on salinity gradients and SAV occurrence, the Caloosahatchee River Estuary is divided into three segments: upper, middle and lower estuary. The middle portion of the estuary has not had significant amounts of SAV for a number of years and is not monitored. Sites 1, 2 and 4 are located within the upper estuary. Sites 5 and 6 are located within the lower estuary. Sites 7 and 8 are located within San Carlos Bay.

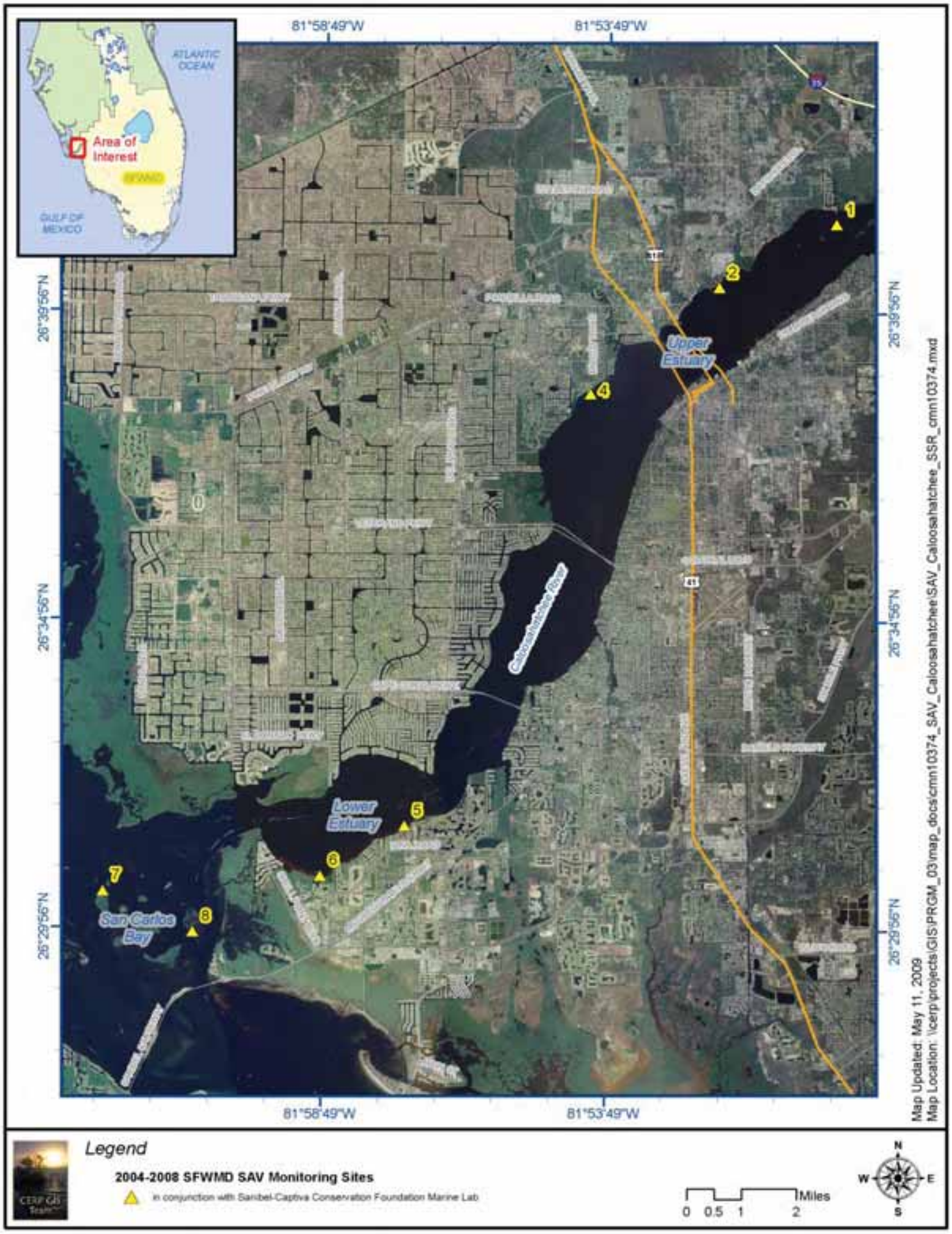


FIGURE 7-41. SUBMERGED AQUATIC VEGETATION AND WATER QUALITY FIXED TRANSECT PATCH-SCALE SAMPLING SITES WITHIN THE CALOOSAHATCHEE RIVER ESTUARY AND SAN CARLOS BAY

7.3.3.1 Patch-Scale Fixed Transects

7.3.3.1.1 Upper Caloosahatchee River Estuary

In terms of distribution and abundance, *Vallisneria americana*, commonly known as tape grass, has been the dominant species in the upper Caloosahatchee River Estuary, colonizing littoral zones in water of less than one meter (Chamberlain and Doering, 1998a). In the early 1990s, SAV covered approximately 1,000 acres and about 60 percent of the coverage occurred within an 8-kilometer stretch between Beautiful Island and the Fort Myers Bridge (Hoffacker, 1994). Total longitudinal cover ranged from 14 to 32 kilometers upstream from Shell Point (Chamberlain and Doering, 1998b). By 2006, increases in salinity had led to the decline and then the absence of *V. americana* in the upper estuary. *Ruppia maritima*, known as widgeon grass, was the dominant species though it never achieved even the minimum abundance recorded for *V. americana* (Burns et al., 2007).

V. americana was the overwhelmingly dominant species observed in Sites 1 and 2 in the upper estuary from 2004 through 2006 (***Table 7-2***; ***Figure 7-42*** through ***Figure 7-45***). *V. americana* can typically tolerate salinities of 3 to 5 psu with few long-term effects if light conditions are sufficient (Haller et al., 1974; French and Moore, 2003; Jarvis and Moore, 2008) so the slightly higher salinities from 2004 to 2006 were possibly not injurious. Dramatic declines in *V. americana* at these two sites were observed beginning in late 2006 as a result of salinities exceeding the species‟ tolerance (Bourn, 1932; Haller et al,. 1974; Doering et al., 1999; Kraemer et al., 1999; Doering et al., 2001).

TABLE 7-2. SUBMERGED AQUATIC VEGETATION DOMINANT SPECIES AT SAMPLING SITES IN THE CALOOSAHATCHEE RIVER ESTUARY

Segment	Site Number	SAV Species
Upper Caloosahatchee River Estuary	1	*Vallisneria americana* *Ruppia maritima*
	2	
	4	*Ruppia maritima* *Vallisneria americana*
Lower Caloosahatchee River Estuary	5	*Halodule wrightii* *Ruppia maritima*
	6	
San Carlos Bay	7	*Halodule wrightii* *Thalassia testudinum* *Ruppia maritima*
	8	

At Site 4, *Ruppia maritime*, commonly known as widgeon grass, was the dominant species though it only occurred at low cover (less than 15 percent) and densities (***Table 7-2***; ***Figure 7-46*** and ***Figure 7-47***). Site 4 is in a transitional section of the estuary where salinities are occasionally too high for the survival of freshwater SAV species such as *V. americana* and during other periods too low for the survival of higher salinity tolerant species such as *Halodule beuadettei* (shoalweed), *H. wrightii* (shoal grass) or *Thalassia testudinum* (turtle grass).

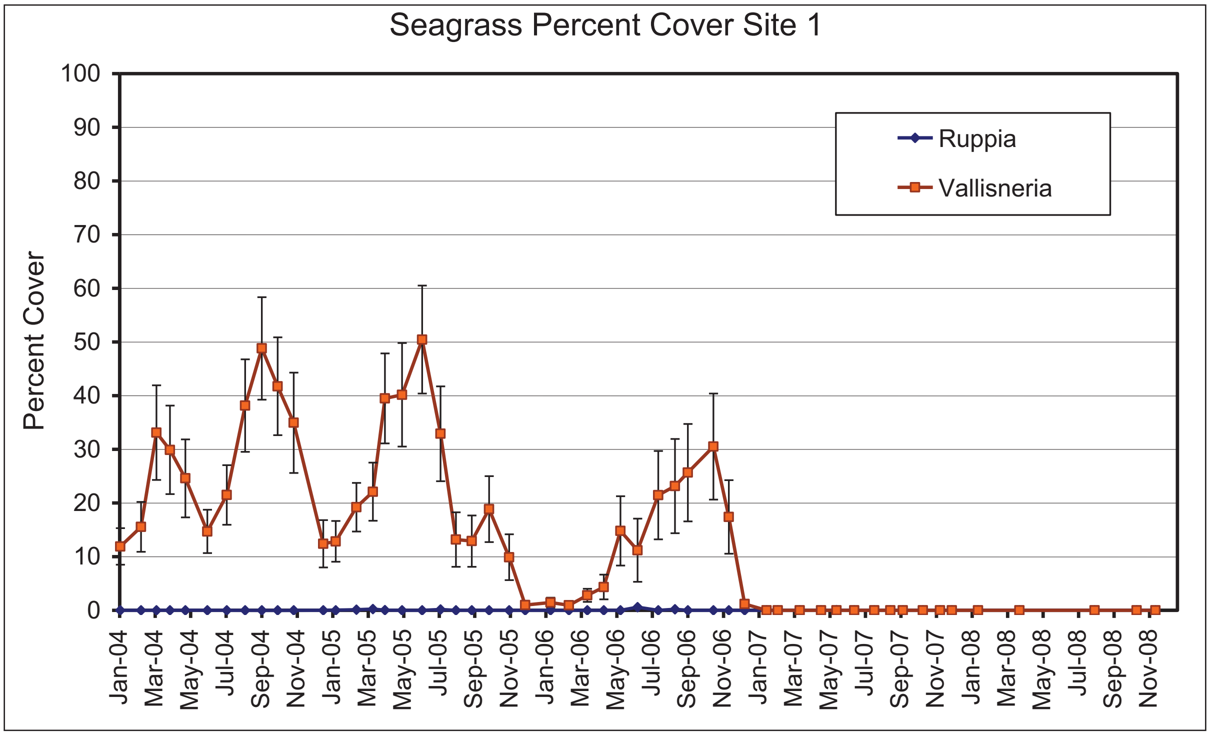


FIGURE 7-42. SUBMERGED AQUATIC VEGETAION MEAN (± STANDARD ERROR) BOTTOM COVER AT MONITORING SITE 1 FROM 2004 TO 2008

Key: SE Standard Error

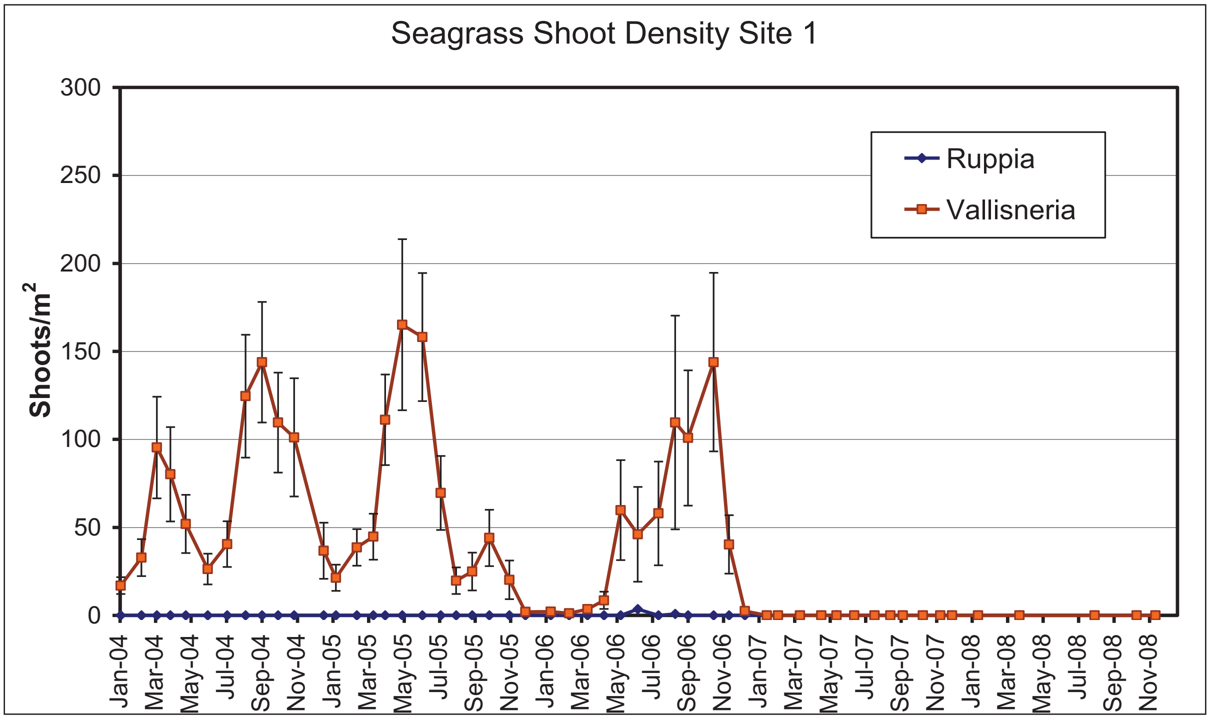


FIGURE 7-43. SUBMERGED AQUATIC VEGETATION MEAN (±SE) SHOOT DENSITY AT MONITORING SITE 2 FROM 2004 TO 2008

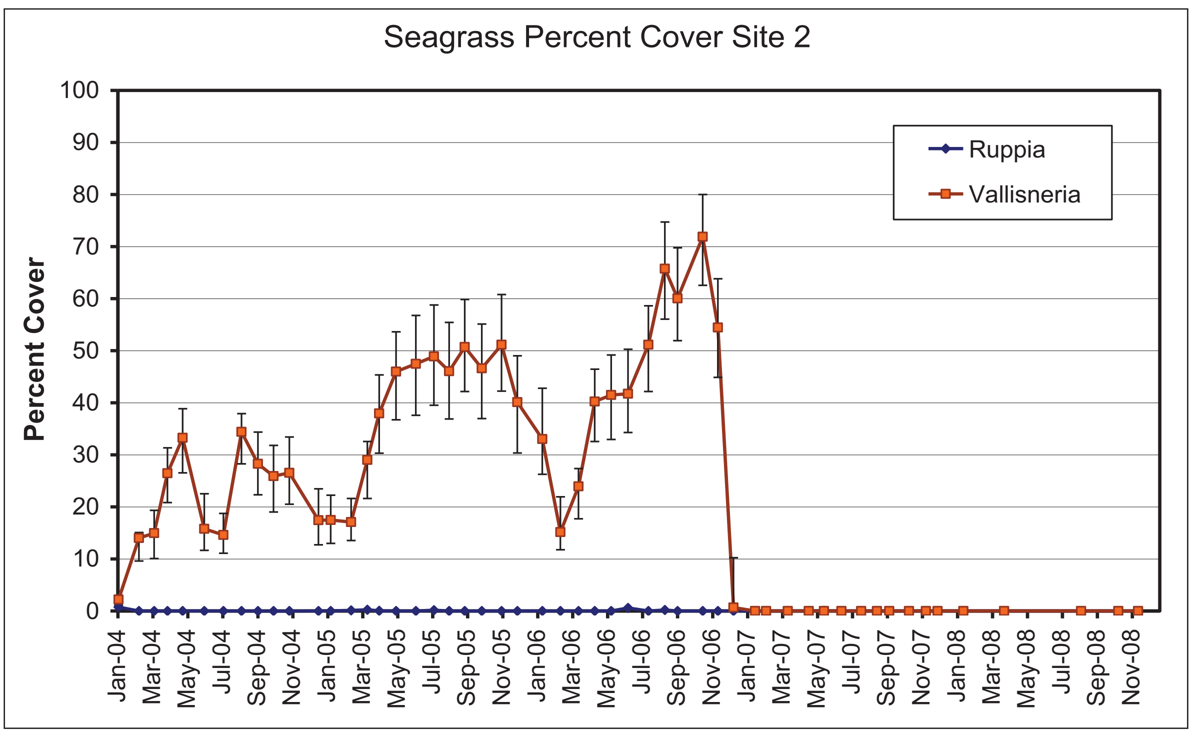


FIGURE 7-44. SUBMERGED AQUATIC VEGETATION MEAN (±SE) BOTTOM COVER AT MONITORING SITE 2 FROM 2004 TO 2008

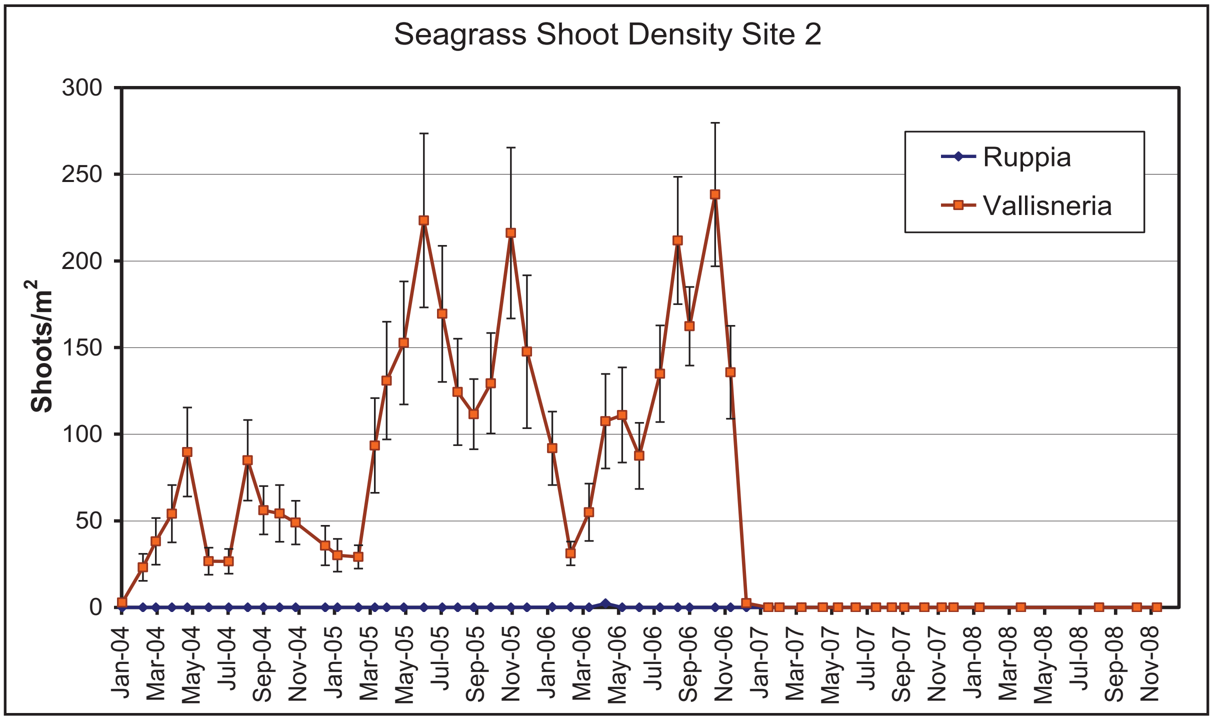


FIGURE 7-45. SUBMERGED AQUATIC VEGETATION MEAN (±SE) SHOOT DENSITY AT MONITORING SITE 2 FROM 2004 TO 2008

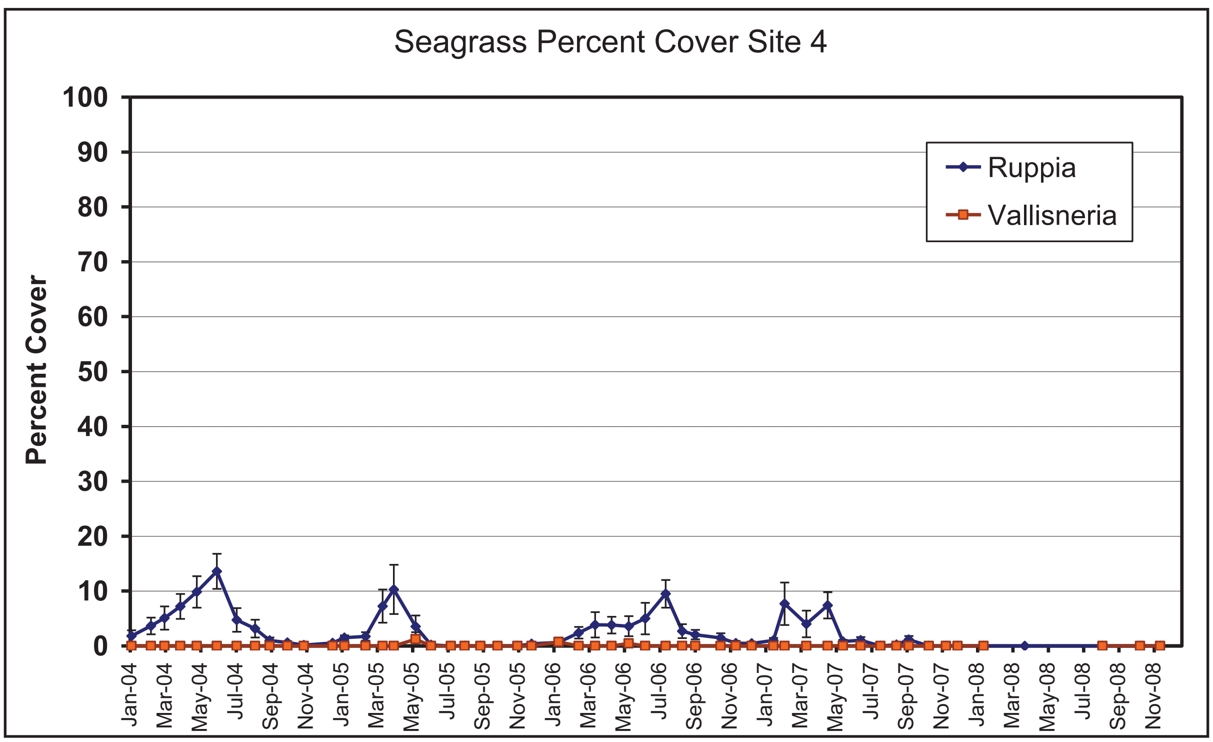


FIGURE 7-46. SUBMERGED AQUATIC VEGETATION MEAN (±SE) BOTTOM COVER AT MONITORING SITE 4 FROM 2004 TO 2008

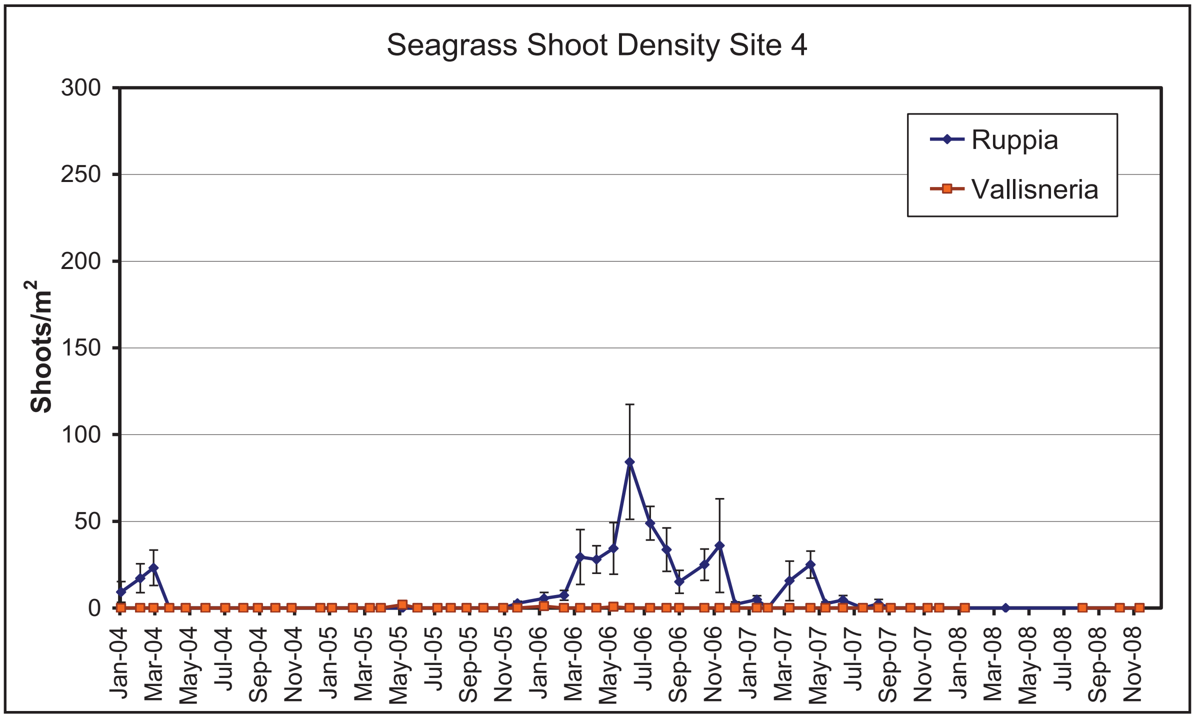


FIGURE 7-47. SUBMERGED AQUATIC VEGETATION MEAN (±SE) SHOOT DENSITY AT MONITORING SITE 4 FROM 2004 TO 2008

The effects of hurricane water releases in 2005 resulted in decreased plant cover and density in Sites 1 and 2 in the latter half of 2005 (***Figure 7-43*** and ***Figure 7-45***). These water releases may have affected SAV in Site 4 as well, but *R. maritima* cover was too low to quantify much effect. Compounding the high turbidity impacts from freshwater releases in 2005, precipitous increases in salinities beginning in October 2006 raised salinity levels to 10 to 25 psu from November 2006 through April 2008. During the December 2005 to April 2006 period, the lower water clarity at Site 1 was associated with lower shoot density and cover compared to Site 2 (***Figure 7-42*** through ***Figure 7-45***). The loss of plants was quite rapid with a significant end-of-year dieback in 2006 followed by no regrowth in spring 2007. Salinities finally declined between April and October 2008, but recovery has been slow. This may be related to a lack of propagules as nearly all the *V. americana* was lost during the 2007 to 2008 high salinity period. It may also be related to herbivory or other impacts on the initial recolonization of recruits into the area. Leaves at these sites were sometimes noted as missing their tips. At Sites 1 and 2, exclosures may need to be planted with either *V. americana* or *R. maritima* following high salinity events in order to produce founder colonies to speed recovery.

In Site 4, 2007 coverage was similar to that in 2006 (***Figure 7-46***). K_d averaged 1.4 m^{-1} in 2007 with no releases and a year of flushing. This suggests there should have been enough light for plant growth at one meter without epiphyte shading. In 2004 and 2005, K_d averaged 2.7 and 3.2 per square meter, which may not allow growth at greater than 0.5 meter. Salinities were often 0 psu at Site 4 until September 2006 and this would prevent survival of seedlings of *Halodule* or other seagrass species. Salinities stayed below 15 psu during wet years, which would support *V. americana*, but much higher salinities during the drought in 2007 prevented long-term survival.

The amount of flow at S79 that would be needed to ensure *V. americana* survival at Site 4 is well over the sustained 300 cfs the MFL rule specifies based on Sanibel Captiva Conservation Foundation River, Estuary and Coastal Observing Network 2008 data (see recon.sccf.org).

7.3.3.1.2 Lower Caloosahatchee River Estuary

Historically, two species of SAV have been routinely reported during surveys in the lower Caloosahatchee River Estuary upstream of Shell Point. These include *H. beuadettei* or *H. wrightii* and *T. testudinum* (Chamberlain and Doering, 1998a; Wilzbach et al., 2000; Burns et al., 2007), which dominates this area. In more recent reports, *Syringodium filiforme*, commonly known as manatee grass, has been reported in San Carlos and Tarpon Bays (Wilzbach et al., 2000; Burns et al., 2007). *H. wrightii* coverage, described as abundant, has been at 300 acres; approximately 75 percent of this occurred between two and eight kilometers upstream of Shell Point (Chamberlain and Doering, 1998b. Species abundance followed the trend of *T. testudinum* greater than *H. wrightii* greater than *S. filiforme*. *T. testudinum* abundance was two and three-times that of the *H. wrightii* and *S. filiforme*, respectively (Wilzbach et al., 2000). However, after October 2005, *H. wrightii* replaced *T. testudinum* as the most abundant species at those sites (Burns et al., 2007).

From 2004 to 2008, the lower estuary was dominated by *H. wrightii* (***Table 7-2***; ***Figure 7-48*** through ***Figure 7-51***). Although *R. maritima* was observed occasionally, it is difficult to separate from *H. wrightii* when it is not flowering, so its abundance could be underestimated slightly. Only very low densities of *R. maritima* were found in the lower estuary when surveys were searching specifically for it. High salinity fluctuations with tides and shading by *H. wrightii* may prevent *R. maritima* from growing. Low salinities during higher rainfall periods and discharge events observed since 2004 likely prevented the survival of seagrass species such as *T testudinum*. *H. wrightii* coverage and densities were lower at Site 6 than the slightly upriver Site 5. This persistently lower abundance, compared to Site 5 since 2004, may be related to site-specific differences in their physical settings or the impacts of storm events.

Water clarity was poor in 2004 and 2005 preventing SAV growth in waters greater than 0.7-meter deep. Water clarity conditions improved in 2007 and were sufficient for growth down to 1.2 meters.

Hurricane effects lowering SAV abundance in 2005 and 2006 and subsequent *H. wrightii* recovery in 2007 are evident with cover in 2007 exceeding 2004 levels (***Figure 7-48*** and ***Figure 7-50***). Salinities of 1 psu or less occurred each year from 2004 to 2006. The large drop in cover and density in fall 2007 prior to the usual winter dieback could have been caused by grazing or bioturbation as other sites in the lower estuary did not show as pronounced a pattern. RECOVER goals for this area of the lower estuary should include minimum salinity goals for this site of above 5 psu to reduce *H. wrightii* damage.

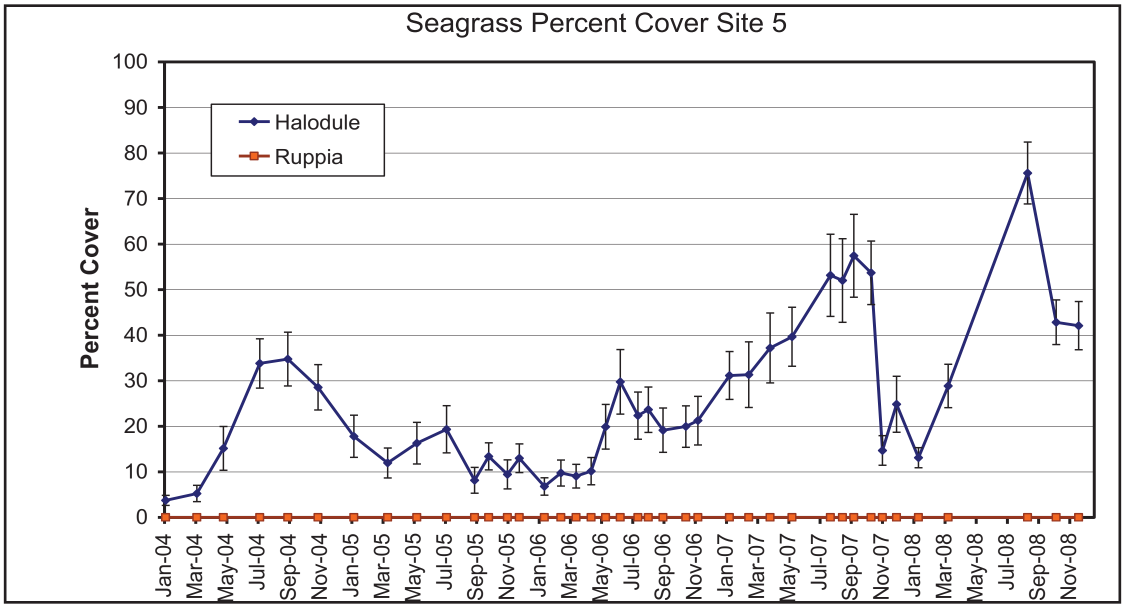


FIGURE 7-48. SUBMERGED AQUATIC VEGETATION MEAN (±SE) BOTTOM COVER AT MONITORING SITE 5 FROM 2004 TO 2008

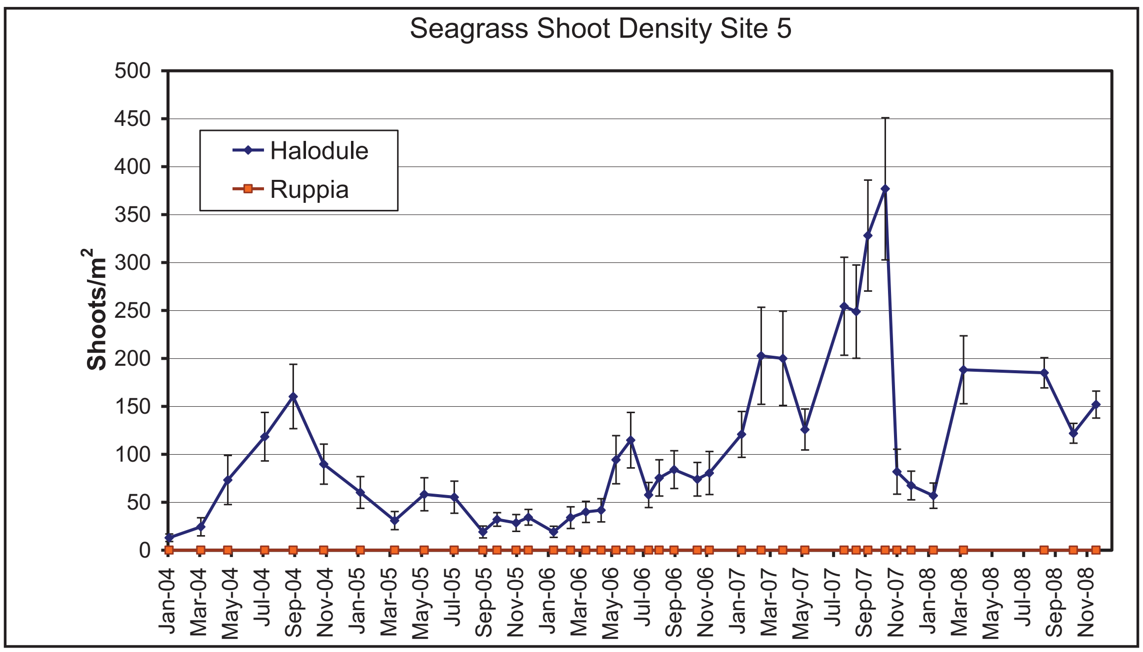


FIGURE 7-49. SUBMERGED AQUATIC VEGETATION MEAN (±SE) SHOOT DENSITY AT MONITORING SITE 5 FROM 2004 TO 2008

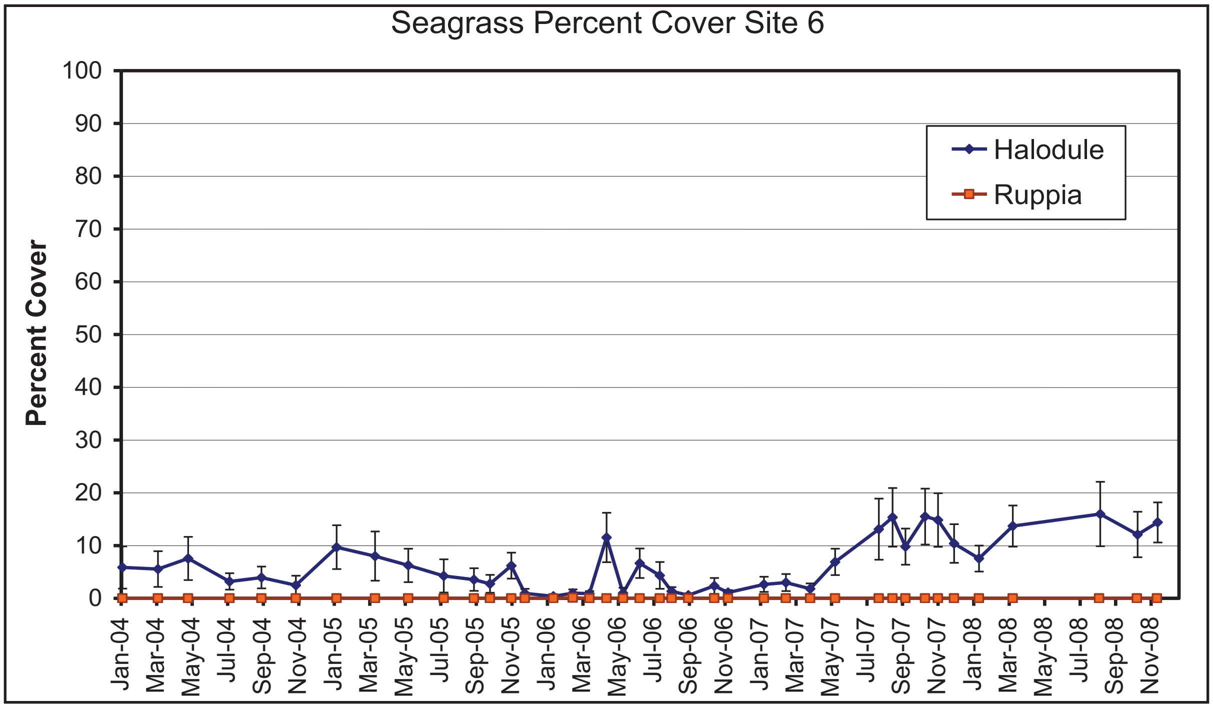


FIGURE 7-50. SUBMERGED AQUATIC VEGETATION MEAN (±SE) BOTTOM COVER AT MONITORING SITE 6 FROM 2004 TO 2008

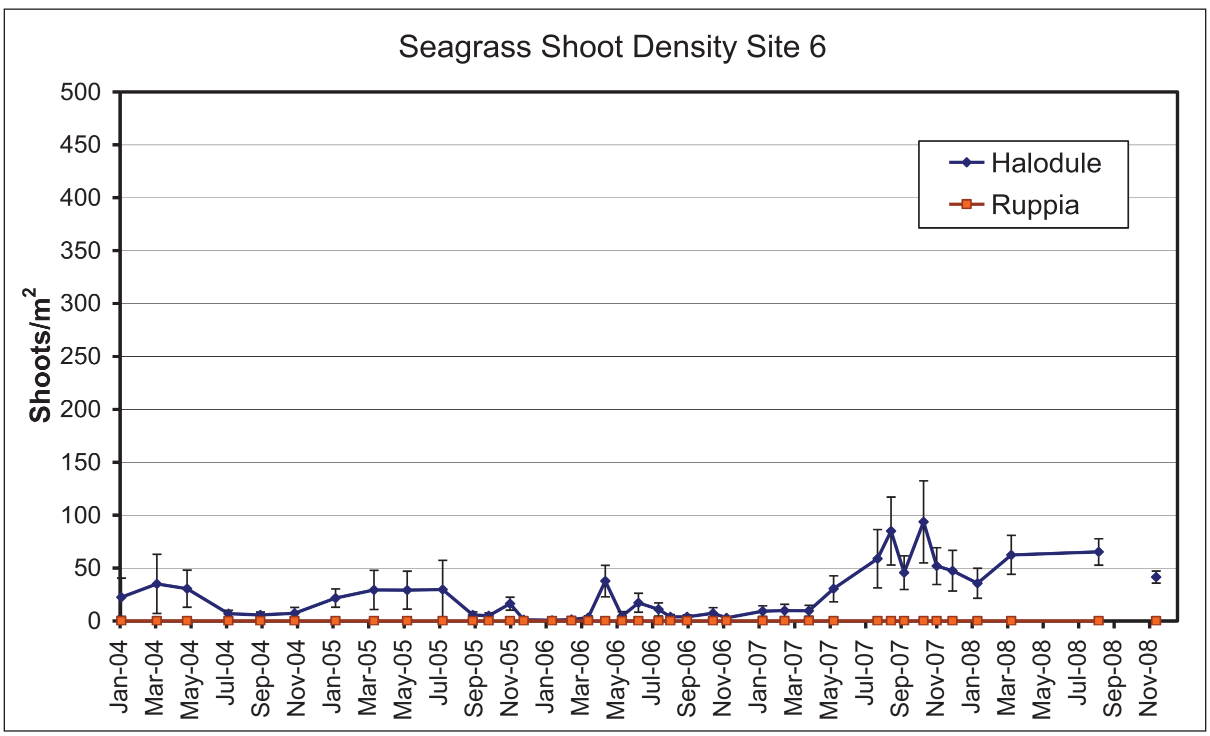


FIGURE 7-51. SUBMERGED AQUATIC VEGETATION MEAN (±SE) SHOOT DENSITY AT MONITORING SITE 6 FROM 2004 TO 2008

7.3.3.1.3 San Carlos Bay

The sites in San Carlos Bay were dominated by *T. testudinum* and *H. wrightii*, though other species, including *R. maritima* and *Halophila engelmannii* (star grass), were found occasionally and *S. filiforme* was observed in the area (***Table 7-2***; ***Figure 7-52*** through ***Figure 7-55***). Percent cover at Site 7 (***Figure 7-52***) was slightly higher than that of Site 8 from 2004 to 2005 (***Figure 7-54***). At Sites 7 and 8, hurricane effects caused salinities as low as 7 and 6 psu, respectively. At Site 7, the low salinity and low light availability were associated with a five-fold decrease in percent cover (***Figure 7-52***) and ten-fold decrease in density (***Figure 7-53***) in *T. testudinum* starting in summer 2005. *H. wrightii* also experienced an approximately two-fold decrease in cover (***Figure 7-52***) and five-fold decrease in density (***Figure 7-53***) during this same period at Site 7. Recovery of *H. wrightii* and a lack of recovery of the climax species *T. testudinum* were observed in 2006. At Site 8, *T. testudinum* was not reduced to as large a degree and it returned to near previous levels near the end of 2007 (***Figure 7-54*** and ***Figure 7-55***). A short-term decrease in salinity observed in September 2006 had little effect on plant coverage at Sites 7 or 8. Higher coverage of *H. wrightii* in 2007 compared to 2004 may be because of the lack of *T. testudinum*, which would be a competitor for space and resources.

The RECOVER MAP goal of 20 percent coverage at 1.75-meter depth was not achieved, but *T. testudinum* coverage at the site depth was slightly greater than 20 percent (32 percent over all years, 22 percent in 2007) for Site 7 and 30 percent (32 percent over all years and 32 percent after 2006) for Site 8. *H. wrightii* coverage was 66 percent for all four years and 84 percent for 2007 for Site 7 and 54 percent for all four years and 73 percent after 2006 for Site 8. RECOVER goals should include minimum salinity goals for these sites above 12 psu to reduce *T. testudinum* damage (Doering and Chamberlain, 2000).

7.3.3.2 Summary

Overall, the pattern of lower salinity and lower light availability may be problematic for SAV regrowth of a species such as *V. americana* in the upper estuary, which are not tolerant of high salinities, but is negatively impacted by low light availability common in this system when salinities are low. Other species observed there, such at *R. maritima* are more tolerant of higher salinities. *V. americana* has been the dominant species historically in the upper estuary (Chamberlain and Doering, 1998a), indicating that the system would support that species given the right combination of conditions, while *R. maritima* has historically not been abundant, suggesting environmental conditions such as low water clarity may be limiting its growth.

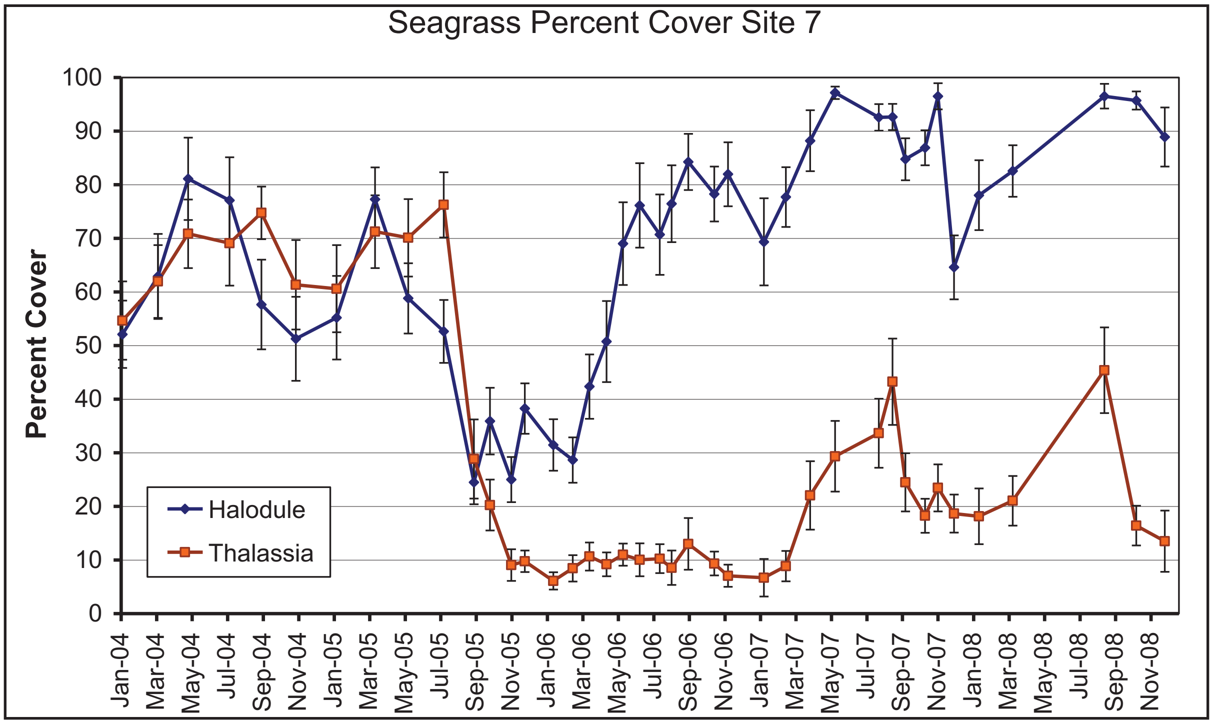


FIGURE 7-52. SUBMERGED AQUATIC VEGETATION MEAN (±SE) BOTTOM COVER AT MONITORING SITE 7 FROM 2004 TO 2008

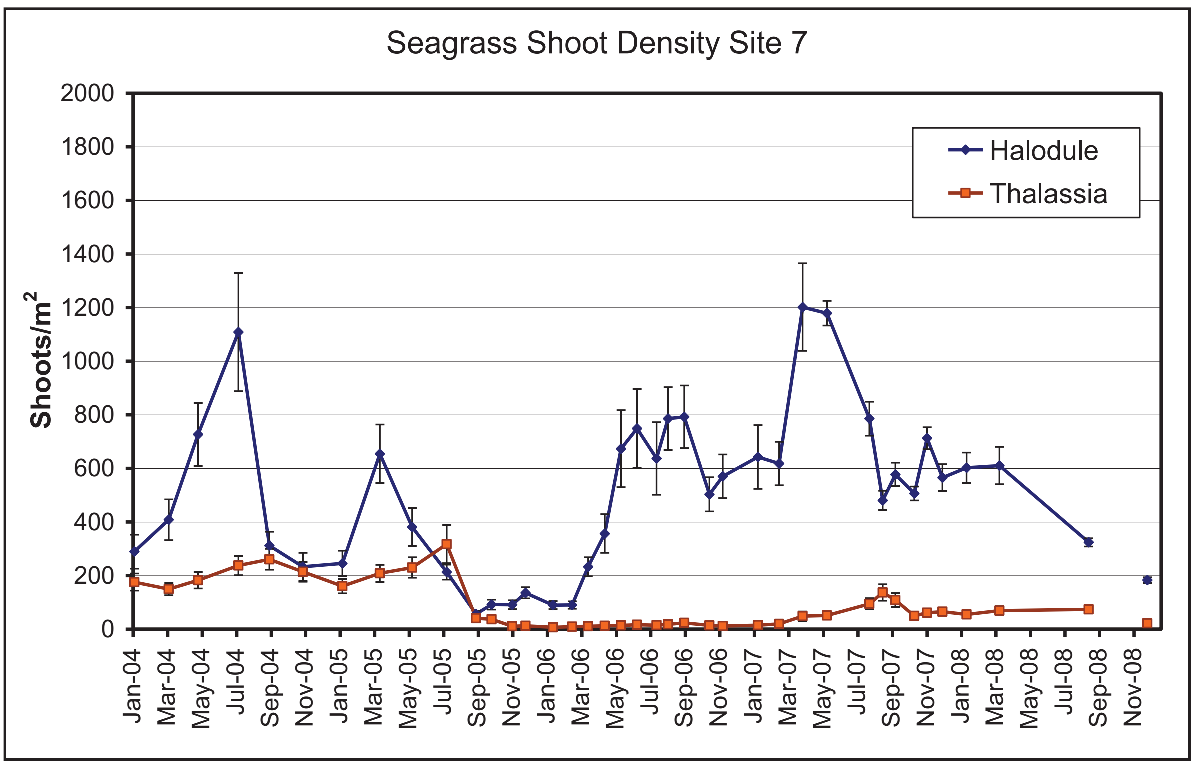


FIGURE 7-53. SUBMERGED AQUATIC VEGETATION MEAN (±SE) SHOOT DENSITY AT MONITORING SITE 7 FROM 2004 TO 2008

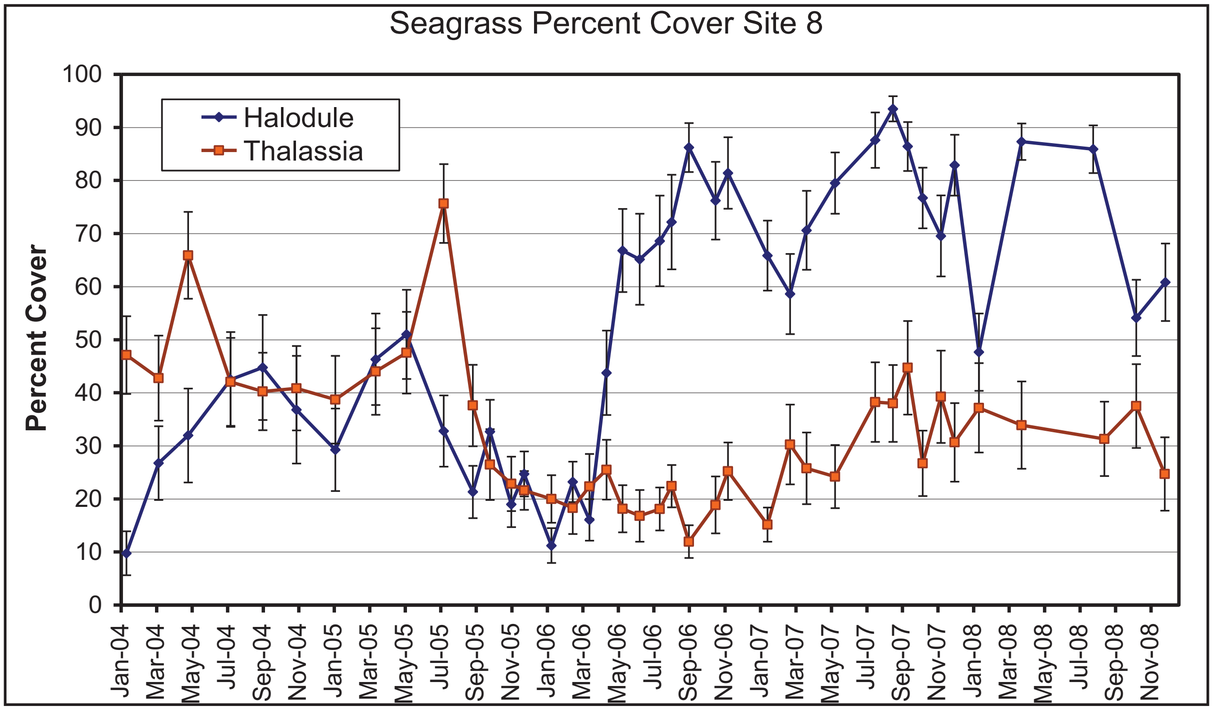


FIGURE 7-54. SUBMERGED AQUATIC VEGETATION MEAN (±SE) BOTTOM COVER AT MONITORING SITE 8 FROM 2004 TO 2008

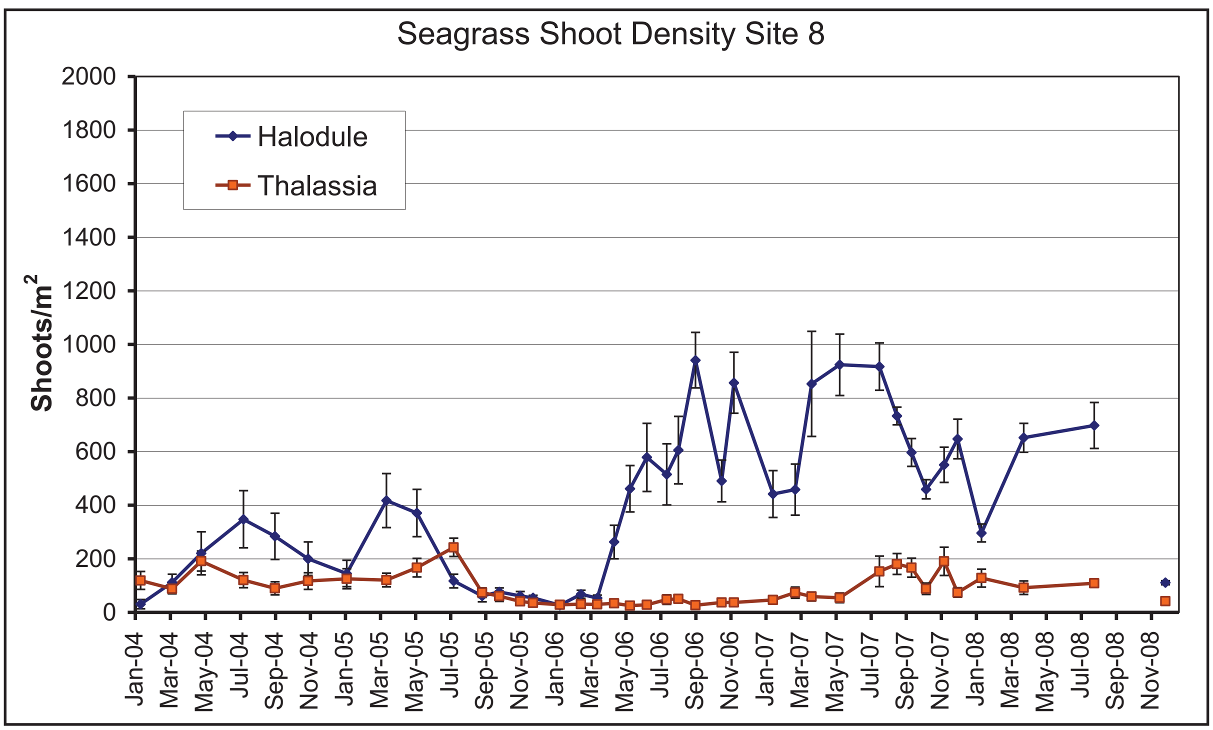


FIGURE 7-55. SUBMERGED AQUATIC VEGETATION MEAN (±SE) SHOOT DENSITY AT MONITORING SITE 8 FROM 2004 TO 2008

In the lower Caloosahatchee River Estuary and San Carlos Bay, the SAV communities have changed dramatically with *T. testudinum* declining to low abundances beginning in 2005 following a period of low salinity and high turbidity. Recovery of the SAV beds has occurred but the community has switched from *T. testudinum* dominance to *H. wrightii* dominance. *H. wrightii* is generally considered a rapid colonizer (Dunton, 1990) and its dominance is not unexpected. Currently (2008), *T. testudinum* is approximately 50 percent of its pre-decline cover and *H. wrightii* is 50 to 100 percent greater than its pre-decline abundance at the study sites. If light availability remains high and salinities increase, increased abundance of *T. testudinum* can be expected.

MFLs have been established based on several reports (Chamberlain et al., 1995; Chamberlain and Doering, 1998a; 1998b) and the tolerance of *V. americana* to elevated salinity (Doering et al., 1999; Kraemer et al., 1999; Doering et al., 2001; 2002). The MFL rule established flows of 300 cfs at S-79 should be maintained when the salinities in *V. americana* areas increase to above the optimum range of 0 to 5 psu (SFWMD 2000a, 2000b). The MFL rule needs to be observed to allow *V. americana* to survive and to ensure the rest of the estuary remains functional (Sklar and Browder, 1998). In order to ensure a year round supply of fresh water, the surficial aquifers with potentiometric surfaces (i.e., the level that water will rise to in the absence of a confining layer) that can bisect the river need to be recharged by allowing water to stand on the overlying land.

The higher than normal flows are also detrimental to estuarine species including seagrasses such as *H. wrightii* (Doering et al., 2002; Irlandi, 2006) and *T. testudinum* (Irlandi et al., 2002; Irlandi, 2006). The variability of flows has also increased from historic levels. High salinity and turbidity variation may negatively affect *R. maritima* (Montague et al., 1988; Kantrud, 1991; Estevez, 2000; La Peyre and Rowe, 2003), the main submerged plant remaining in the upper estuary, and may be one of the factors responsible for its continued low plant densities.

Though the data from this 2004 to 2008 monitoring period demonstrate that the compounding effects of extremes in salinity and water clarity are related to decreases in both *V. americana* abundance in the upper estuary and *T. testudinum* and other species in the lower estuary, high nutrient loadings associated with freshwater inflows also affect submerged plants (Tomasko et al., 1996; Hauxwell et al., 2003) and this should not be ignored. The P-loading rate to the estuary was as high as 8.5 grams per square meter per year ($g/m^2/yr$), which is almost four times greater than the maximum Chesapeake Bay loading rate (Boynton et al., 1995). The TMDL, which is under development, should be determined based on the effect of nutrient loading on all algae, not just phytoplankton, since submerged plants in the estuary are frequently covered with epiphytic micro algae and macro algae and the response of algae to nutrients is time and species dependent. Some cyanobacteria such as *Lyngbya majuscula*, became very abundant and overgrew seagrasses in Pine Island Sound in 2006 and 2007 (Bartleson et al., 2006). Since this species fixes its own nitrogen, a nitrogen-based TMDL alone may not have an effect.

The salinity goal for SAV growth in this region should have average levels above 3 psu with extremes kept below 15 psu. Measures of success of restoration should also include increased *R. maritima* coverage to greater than ten percent, increased maximum depth of plant occurrence to greater than one meter, increased *V. americana* leaf sizes, and increased seed production, which was not observed at these sites, though occasional seedlings of both species were observed in the vicinity of the study site.

7.3.4 Southern Indian River Lagoon and St. Lucie Estuary

Freshwater discharges into the St. Lucie River ultimately flow into the Southern Indian River Lagoon and over SAV communities, which include seagrass and macroalgae. This region was impacted by hurricanes and associated freshwater discharges in 2004 and 2005. Following the hurricanes, observed impacts to Southern Indian River Lagoon SAV communities included large coverage and density declines and smaller direct impacts due to burial by shifting bottom sediments.

The Southern Indian River Lagoon and St. Lucie Estuary support six species of seagrass: *H. wrightii, S. filiforme, T. testudinum, Halophila decipiens* (paddle grass), *Halophila johnsonii* (Johnson‟s grass), and *H. engelmannii. H. wrightii* and *S. filiforme* are the dominant canopy species in the lagoon (Thompson, 1978; Dawes et al., 1995; Morris et al., 2000). While all of these species are most successful in salinities greater than 20 psu, *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).

Southern Indian River Lagoon and St. Lucie Estuary SAV monitoring consisted of water quality monitoring for salinity and water clarity and four SAV components:

1. Lagoonwide, landscape-scale mapping every two to three years since 1986
2. Species-specific landscape-scale mapping in 2007 and 2008 in Southern Indian River Lagoon, and in 1997 and 2007 in St. Lucie Estuary
3. Patch-scale bimonthly monitoring along fixed transects from August 2002 to August 2007
4. Patch-scale bimonthly monitoring using the patch-quadrat method beginning in November 2007

SAV coverage targets have been developed for the Southern Indian River Lagoon and St. Lucie Estuary based on suitable habitat. The target for SAV in the Southern Indian River Lagoon is to increase cover of SAV beds to areas that are less than 1.7 meter msl in depth. The lagoon has a total of 19,799 acres of suitable habitat, of which 7,808 (39 percent) is already colonized by seagrass. For the St. Lucie Estuary, the target is to increase cover of SAV beds to areas that are less than one meter msl in depth. The St. Lucie Estuary has 922 acres of suitable habitat, none of which contain seagrass. The performance measure documentation sheet is available online at www.evergladesplan.org/pm/recover/recover_docs/et/ne_pm_sav.pdf. In addition, a target has been developed for a salinity range most favorable to the estuaries VECs, which include SAV. This salinity is estimated at 12 to 20 psu as measured at the Roosevelt Bridge. The performance measure is available online at www.evergladesplan.org/pm/recover/recover_docs/et/ne_pm_-salinityenvelopes.pdf.

7.3.4.1 Landscape-Scale Mapping

7.3.4.1.1 Lagoonwide Mapping

SAV distribution has been mapped in the Southern Indian River Lagoon every two to three years since 1986, including annual mapping from 2005 through 2007 to assist in hurricane impacts assessment. Areas mapped include the lagoon from the St. Lucie County and Indian River County line south to the Jupiter Inlet. Additionally, the section of the St. Lucie Estuary downstream of the Roosevelt Bridge is included within the map boundaries. Map acreage results are presented by lagoon segments (22-26) (Steward et al., 2003), which are presented in ***Figure 7-56***.

Overall lagoon SAV acreage increased from 8,030 acres in 2006 to 8,847 acres in 2007 (***Figure 7-57***). The 2007 SSR (RECOVER, 2007) noted a decline in the acreage of dense continuous SAV from 2003 to 2006 between the Stuart Causeway and the St. Lucie Inlet. Gains and additional losses were documented in the same area from 2006 to 2007 (***Figure 7-57***). A large area of SAV loss along the west shore previously supported *S. filiforme* (Rebecca Robbins, personal observation). Field work conducted one year later documented the presence of *H. johnsonii* (Avineon, Inc. 2008a), a diminutive pioneer species that colonizes areas left bare by the loss of canopy forming species and may eventually be replaced by more robust species. This indicates seagrass recovery within this portion of the lagoon. Areas of "gain" between the Stuart Causeway and St. Lucie Inlet are also indications of recovery. These areas are largely shallow, on and along shoals. One year after the mapping, *H. wrightii* and *H. johnsonii* (Avineon, Inc 2008a) were found in these shallow areas.

Robbins and Conrad (2001) provide a detailed analysis of lagoon SAV map data from 1986 to 1999. Their report includes a comparison of seagrass acreage to a 1.7-meter depth target established through the Indian River Lagoon Surface Water Improvement and Management (SWIM) planning process (Steward et al., 2003). The SWIM plan states that this target may not be appropriate for all lagoon segments and directs the water management districts to evaluate appropriate targets. The SWIM plan suggests a 1-meter depth target for the St. Lucie Estuary. Crean et al. (2007) compared seagrass and water quality data to evaluate water quality targets for restoring and protecting lagoon seagrasses. Their targets are based on a 1.3-meter benchmark. All three targets (1.0, 1.3 and 1.7 meters) are shown on ***Figure 7-58***.

Figure 7-59 shows acreage by segment over time from 1999 to 2007 compared to these depth targets. From 2003 to 2006, total acreage declines, or declines in the dense SAV category, occurred in all segments except Segment 23. The acreage declines are likely a result of hurricane impacts. All segments showed an increase in overall coverage from 2006 to 2007 indicating recovery of SAV resources.

Segments 24 and 25 would be the most influenced by restoration implementation due to proximity to the St. Lucie Estuary. ***Figure 7-58*** and ***Figure 7-59*** indicate potential for expansion of SAV in both segments. However, runoff from surrounding mangrove wetlands and heavy boat traffic, which suspend bottom sediments, both contributed to the dark waters in Segment 25 and may limit SAV restoration (Robbins and Conrad, 2001).

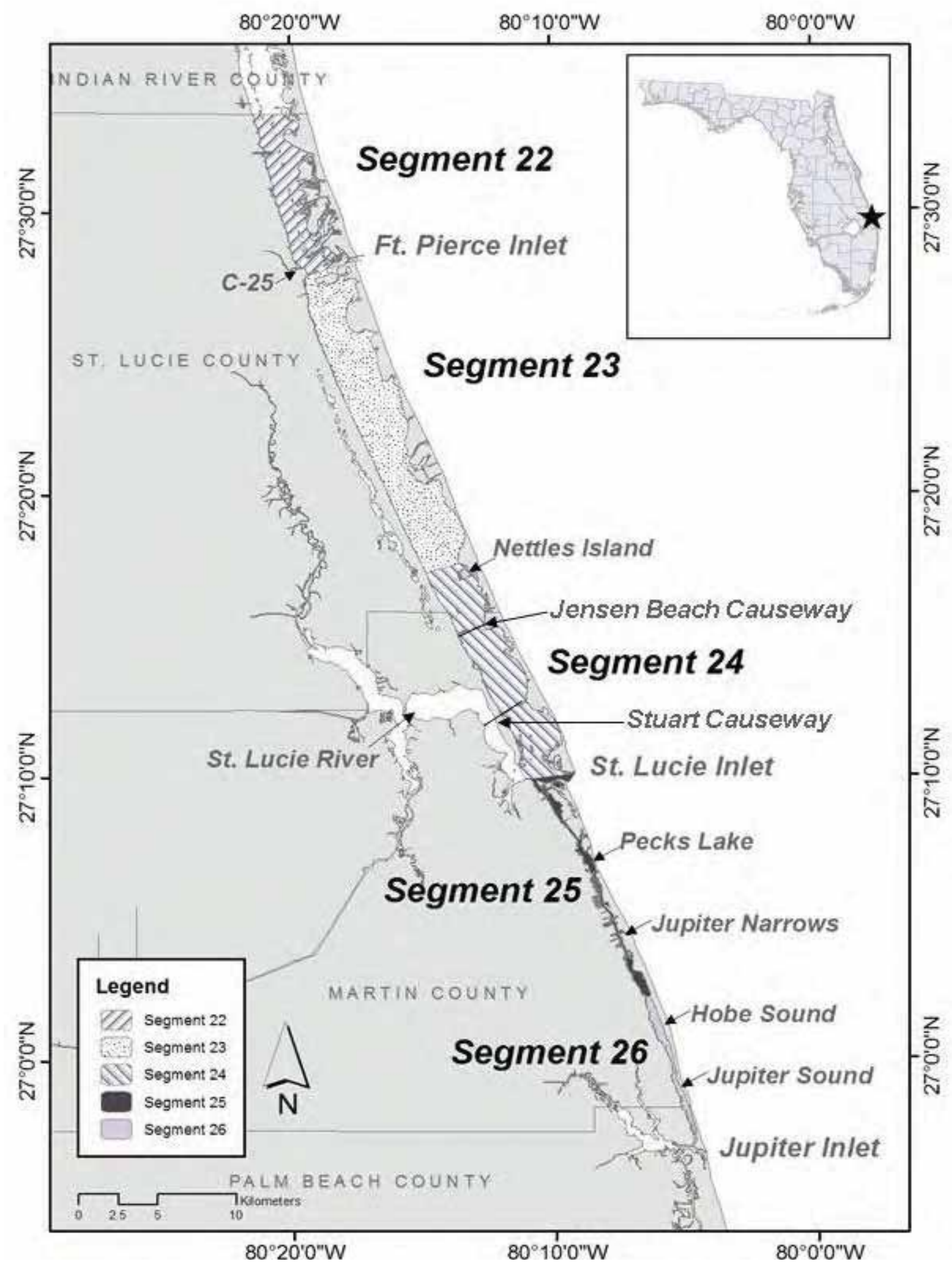


FIGURE 7-56. MAP OF SOUTHERN INDIAN RIVER LAGOON AND ST. LUCIE ESTUARY SHOWING LAGOON SEGMENTS

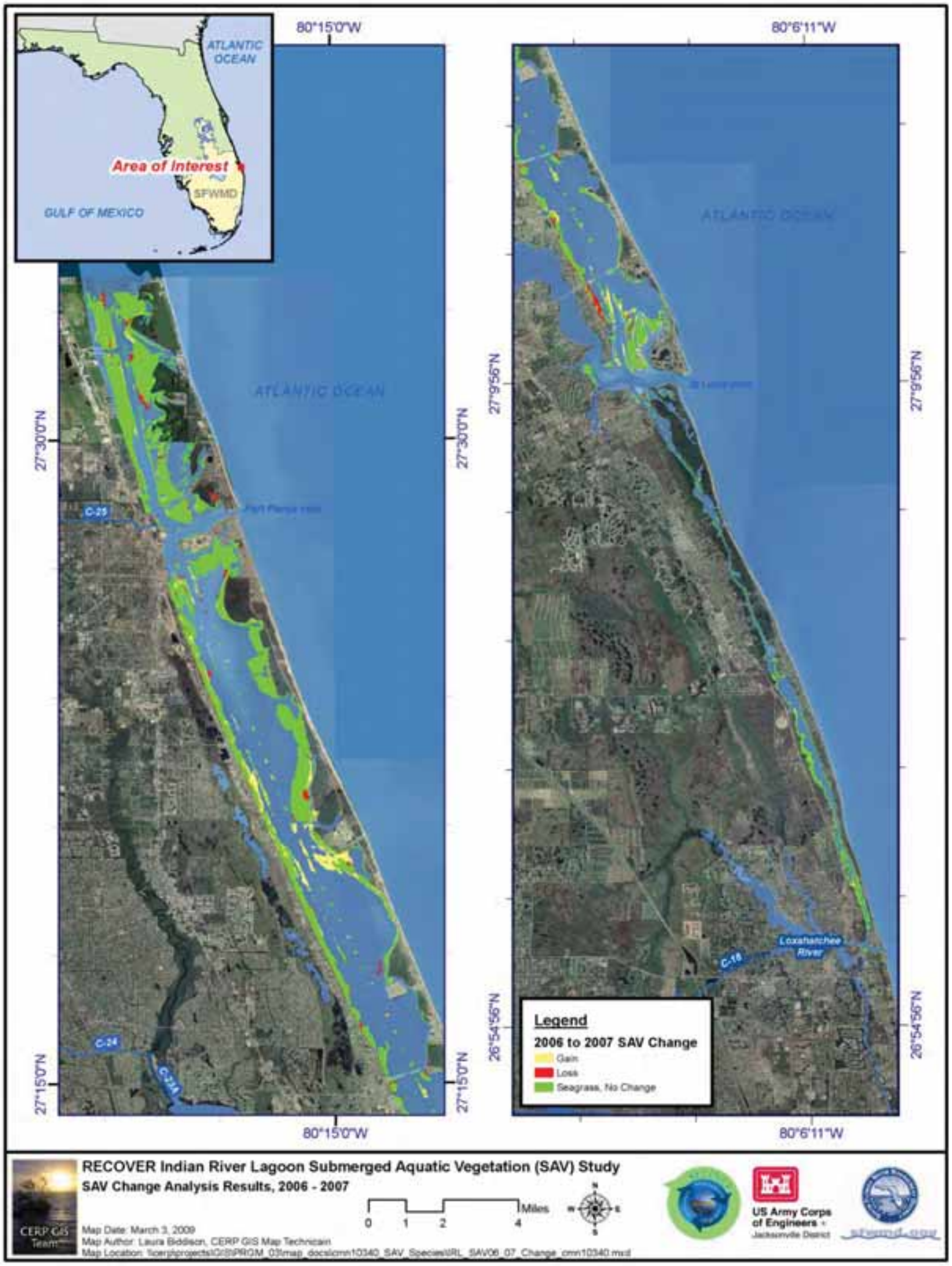


FIGURE 7-57. SUBMERGED AQUATIC VEGETATION CHANGE IN SOUTHERN INDIAN RIVER LAGOON FROM 2006 TO 2007

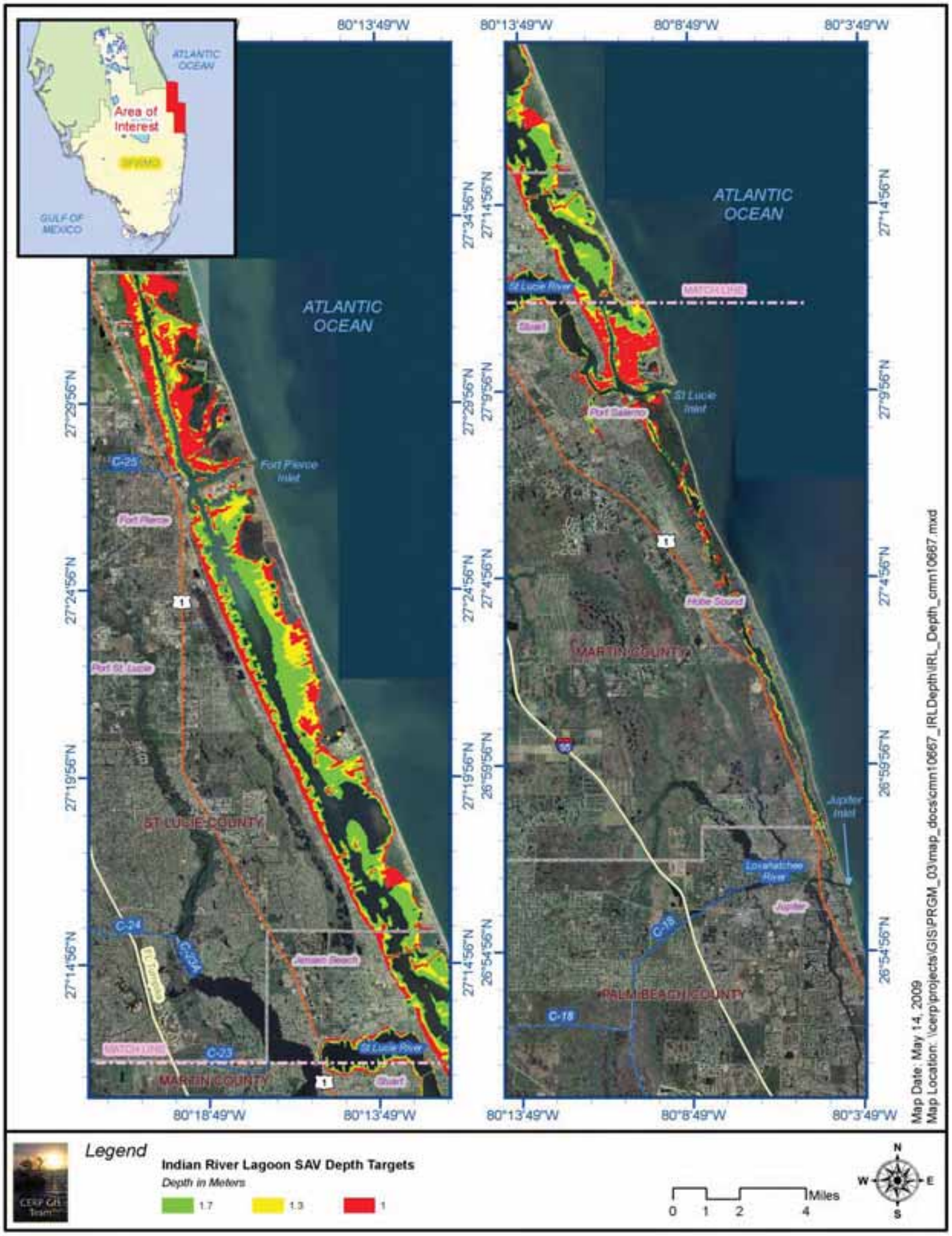


FIGURE 7-58. POTENTIAL SUBMERGED AQUATIC VEGETATION COVER AS FUNCTION OF SUITABILITY AT THREE DEPTHS

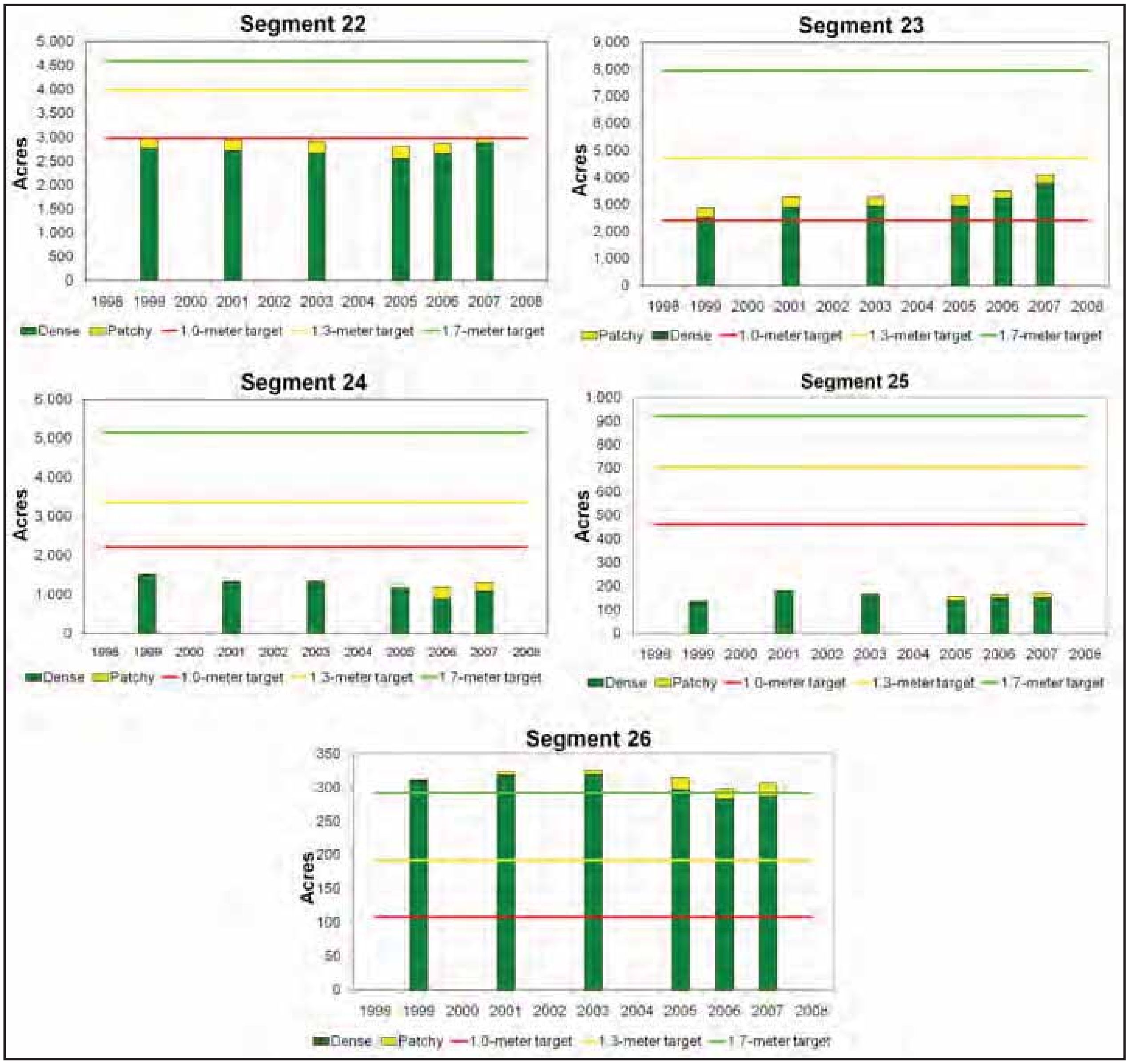


FIGURE 7-59. SUBMERGED AQUATIC VEGETATION ACREAGE AND POTENTIAL COVERAGE TARGETS FROM 1999 TO 2007

Preliminary depth targets per segment are presented in ***Table 7-3***. Water quality around Segment 22 may improve with implementation of the C-25 CERP project (www.evergladesplan.org/pm/projects/proj_07_irl_south.aspx). However, those improvements are likely to be localized near the C-25 Canal discharge since much of it exits the lagoon via the Fort Pierce Inlet. ***Figure 7-58*** shows mapped acreage in this segment often meets the one-meter depth target but significant acreage increases, which are unlikely to result from restoration activities, would be needed to meet the 1.3-meter target. Accordingly, a one-meter depth target is recommended.

For Segment 23, the one-meter target has been consistently exceeded since 1999. Therefore, a target depth of at least one meter should be set for this segment. It is unclear whether restoration efforts will improve water quality in all or parts of this segment. Further evaluation of both water quality and seagrass data are needed for setting a preliminary target for this segment.

TABLE 7-3. DEPTH TARGETS FOR SUBMERGED AQUATIC VEGETATION COVERAGE

	Preliminary Depth Target for SAV Coverage		
Segment	**1.0 meter**	**1.3 meters**	**1.7 meters**
22		X	
23	X	X	
24	X		
25	X		
26			X
St. Lucie Estuary	X		

For Segments 24 and 25, the one-meter target suggested for the adjacent St. Lucie Estuary is recommended for these segments. ***Figure 7-58*** shows that meeting the one-meter target would result in a substantial increase in SAV coverage in Segment 24 and potentially Segment 25. However, as discussed above, factors other than water management strongly influence water clarity in Segment 25. Finally, Segment 26 acreage consistently exceeds the 1.7-meter SWIM target. Accordingly, the target for this segment should be at least 1.7 meter.

7.3.4.1.2 Species Specific Mapping in Southern Indian River Lagoon

Lagoon-wide SAV maps, based on aerial photographs, provide an overall understanding of the SAV coverage and distribution in segments. However, these maps do not provide information on SAV species distribution. Understanding SAV species distribution is important for water management considerations because SAV species found in the Southern Indian River Lagoon have species specific salinity thresholds (Irlandi, 2006). Once restoration projects are completed, species shifts may occur, which cannot be detected from maps created from aerial photographs.

The portion of the lagoon most affected by water management practices is the area that receives discharges from the St. Lucie Estuary. Accordingly, the area of the lagoon adjacent to the estuary mouth was selected for a species specific mapping project. The field work was completed in 2007 and 2008.

Of the area mapped, almost 50 percent was occupied by *H. wrightii* (***Figure 7-60***). Dominance by this species is likely due to bathymetry and salinity tolerance. Other canopy forming species present in the lagoon, *S. filiforme* and *T. testudinum,* are less successful in these very shallow areas, which are less than 0.6 meter msl. Additionally, of the canopy forming species, *H. wrightii* has the greatest tolerance of both salinity range and variability (Irlandi, 2006), and can better adapt to the fluctuating salinities experienced near the mouth of the estuary. As distance from the mouth of the St. Lucie Estuary increased, the percentage of *H. wrightii* declined and *T. testudinum* and *H. decipiens* increased (***Figure 7-61***). The lack of shoals north of the Stuart Causeway and a more stable salinity regime may account for these differences.

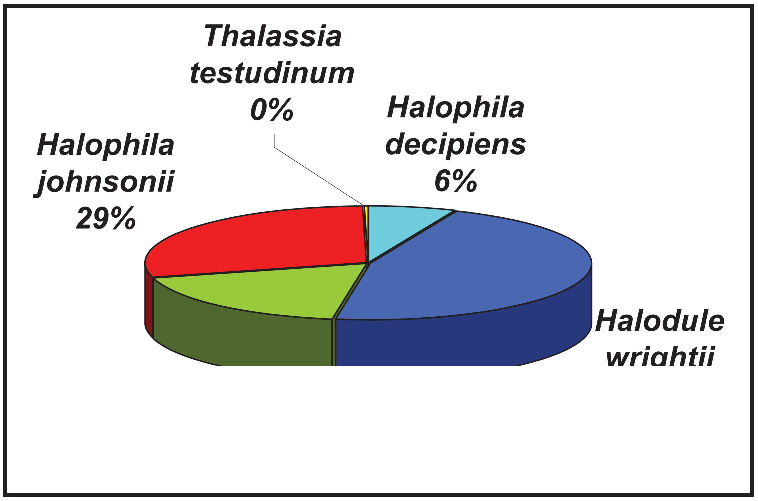
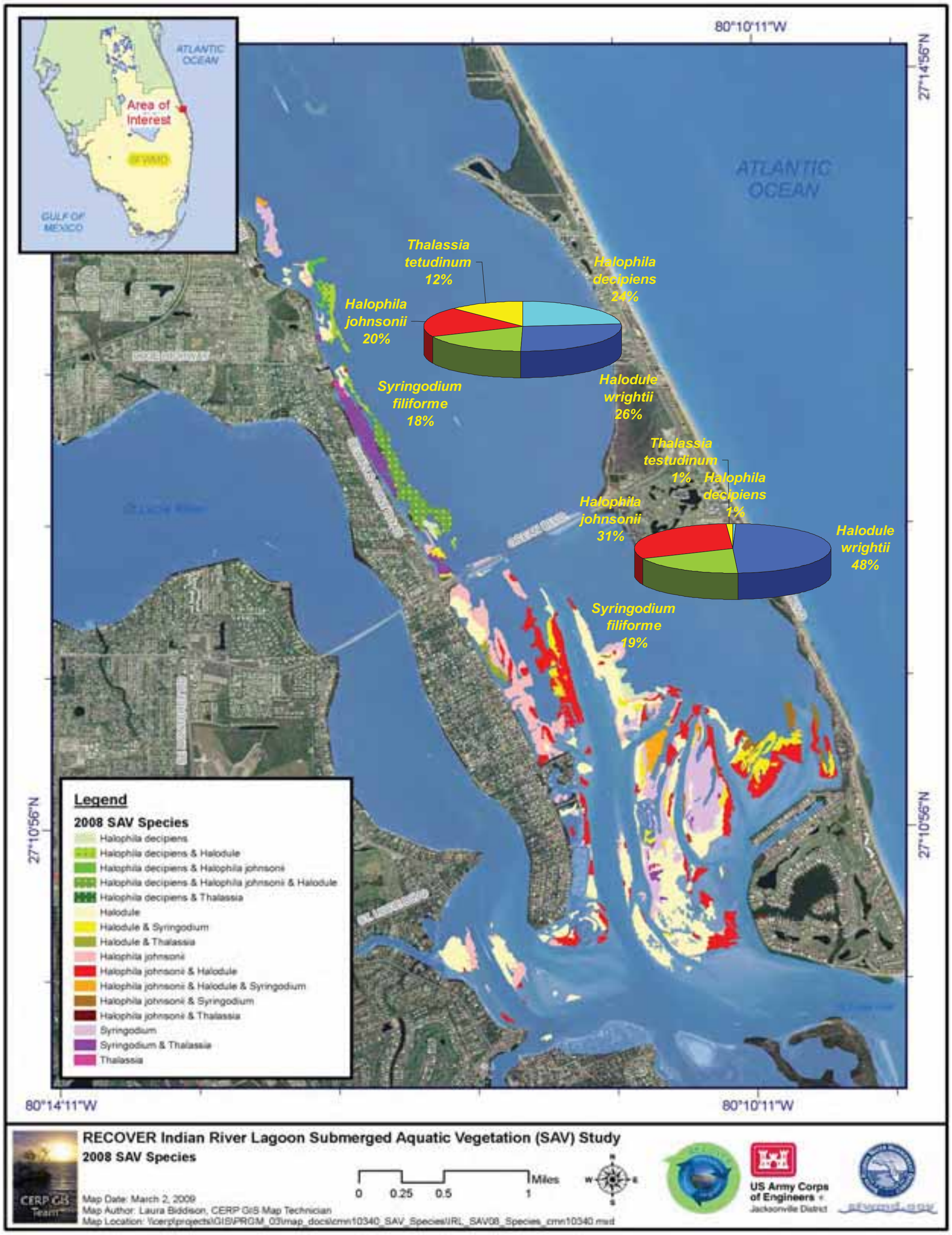


FIGURE 7-60. SUBMERGED AQUATIC VEGETATION SPECIES IN BEDS NEAR THE MOUTH OF THE ST. LUCIE RIVER

Since the species specific mapping field work was done following the 2004 and 2005 hurricane seasons, mapped distribution helped document recovery in some areas. Based on monitoring results discussed below, some beds of *S. filiforme* were impacted, with reduced density and elimination, near the mouth of the St. Lucie Estuary. It is expected, that over time, some areas mapped as *Halophila sp* and *H. wrightii* will shift to *S. filiforme* dominated beds.



DISTRIBUTION FOR THE SOUTHERN INDIAN RIVER LAGOON

7.3.4.1.3 Species Specific Mapping in St. Lucie Estuary

Historic SAV maps (***Figure 7-62a, b*** and ***c)*** show SAV extending throughout the estuary. The most recent maps were completed in 1997 (***Figure 7-62d***) and 2007 (***Figure 7-63***). Most areas inspected during these mapping projects did not support SAV. In 2007, very sparse (less than 10 percent cover in most areas) SAV was present in the lower and middle estuary, but not in either of the forks. Three seagrass species occurred within the estuary: *H. wrightii*, *H. johnsonii* and *H. decipiens*. The majority of the SAV occurred in small isolated patches. The dominant SAV species in 2007 was *H. johnsonii*. It also extended farther upstream than any other SAV species.

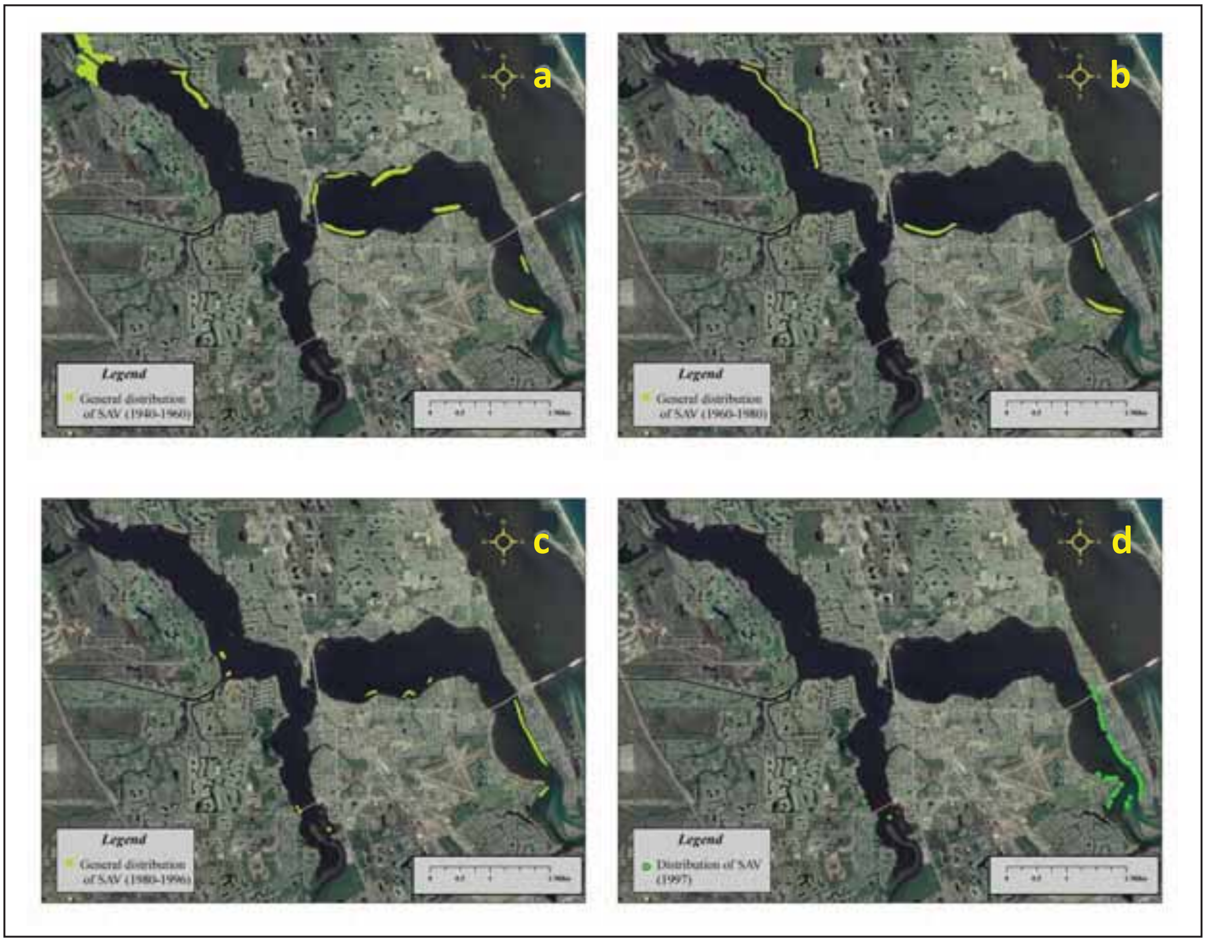


FIGURE 7-62. ST. LUCIE ESTUARY SUBMERGED AQUATIC VEGETATION MAP FOR THE FOLLOWING PERIODS

Note: a) 1940-1960
b) 1960-1980
c) 1980-1996
d) 1997

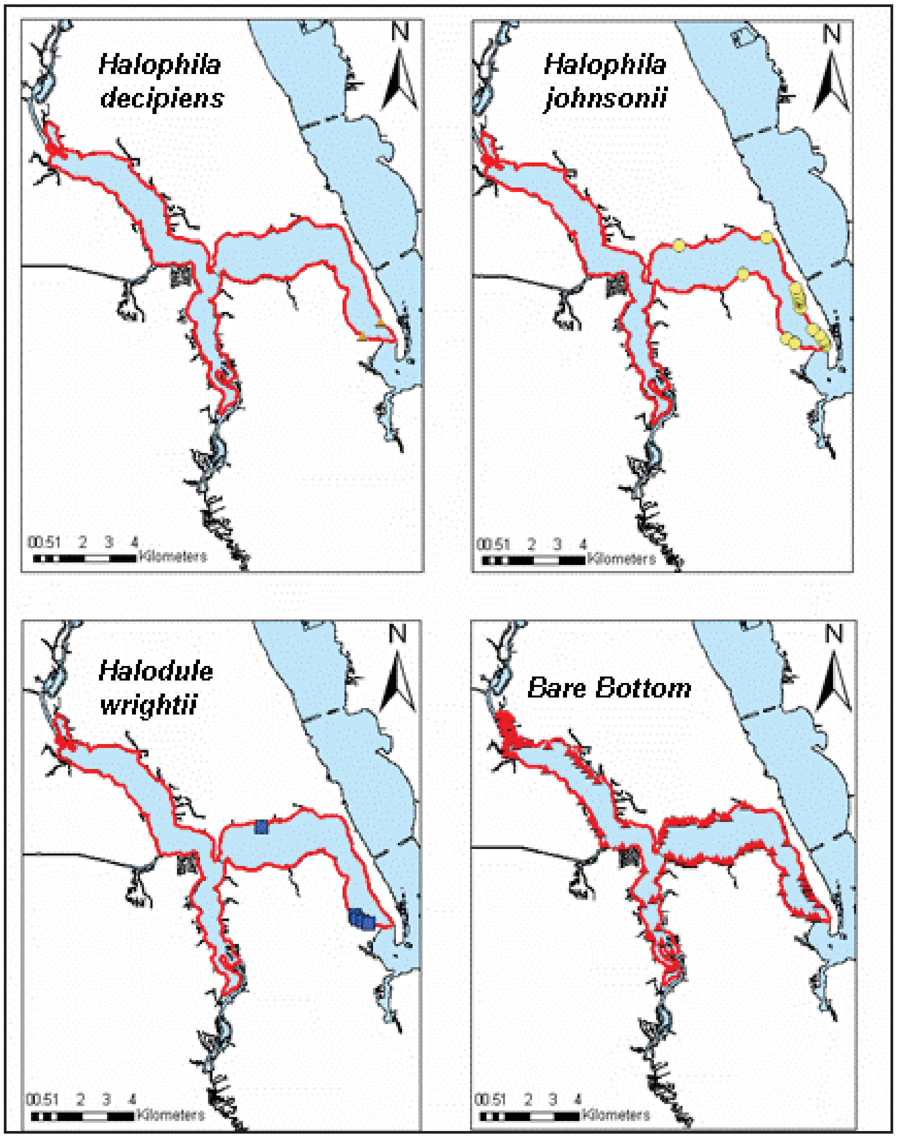


FIGURE 7-63. ST. LUCIE ESTUARY SUBMERGED AQUATIC VEGETATION MAP COMPLETED IN 2007

A previous study determined that a depth target of approximately one meter was recommended for SAV restoration in the St. Lucie Estuary (Steward et al., 1994). Meeting this depth target may result in a thin fringe of SAV throughout much of the estuary. St. Lucie Estuary restoration efforts are focusing on meeting oyster salinity requirements in the middle estuary, so seagrasses with similar requirements should be able to recruit into these areas. *R. maritima* is the most likely SAV species to be successful in the North and South Forks once the restoration salinity regime is in place as this species is adapted to lower salinity conditions (Irlandi, 2006). The middle estuary would most likely support SAV species such as *H. wrightii* and *H. johnsonii*. The lower estuary, where highest salinity and clearest water would occur, would most likely continue to support *H. wrightii* and *H. johnsonii*, and may eventually support *S. filiforme* as observed by Phillips (1960).

7.3.4.2 Patch-Scale Monitoring

7.3.4.2.1 Fixed Transect Method

Patch-scale monitoring documents seasonal changes in seagrass and associated macro-algae (i.e. epiphytes, attached algae and drift algae). The four study sites are shown on ***Figure 7-64***. Sites 1 and 4 were monitored for one year from August 2002 through September 2003. Sites 2 and 3 were monitored for five years from August 2002 through August 2007.

Mean shoot counts and canopy heights for *S. filiforme* prior to hurricane impacts are shown in ***Figure 7-65*** and ***Figure 7-66***, respectively. Peak shoot counts (approximately 1,400 shoots/m^2) and canopy heights (maximum at Site 2 approaching 50 centimeter [cm]) occurred in July for Sites 1 through 3. A significantly lower shoot count peak (approximately 400 shoots/m^2) occurred at Site 4 in September (***Figure 7-65***). Peak canopy height at Site 4 occurred in June (***Figure 7-66***). However, field notes from July indicate that grazing by manatees or sea turtles observed in the area appeared to have reduced the canopy height. Site 4 is a small, patchy site, heavily impacted by boat wakes and may not be representative of most lagoon *S. filiforme* beds.

FIGURE 7-64. FIXED TRANSECT SUBMERGED AQUATIC VEGETATION SAMPLING SITES

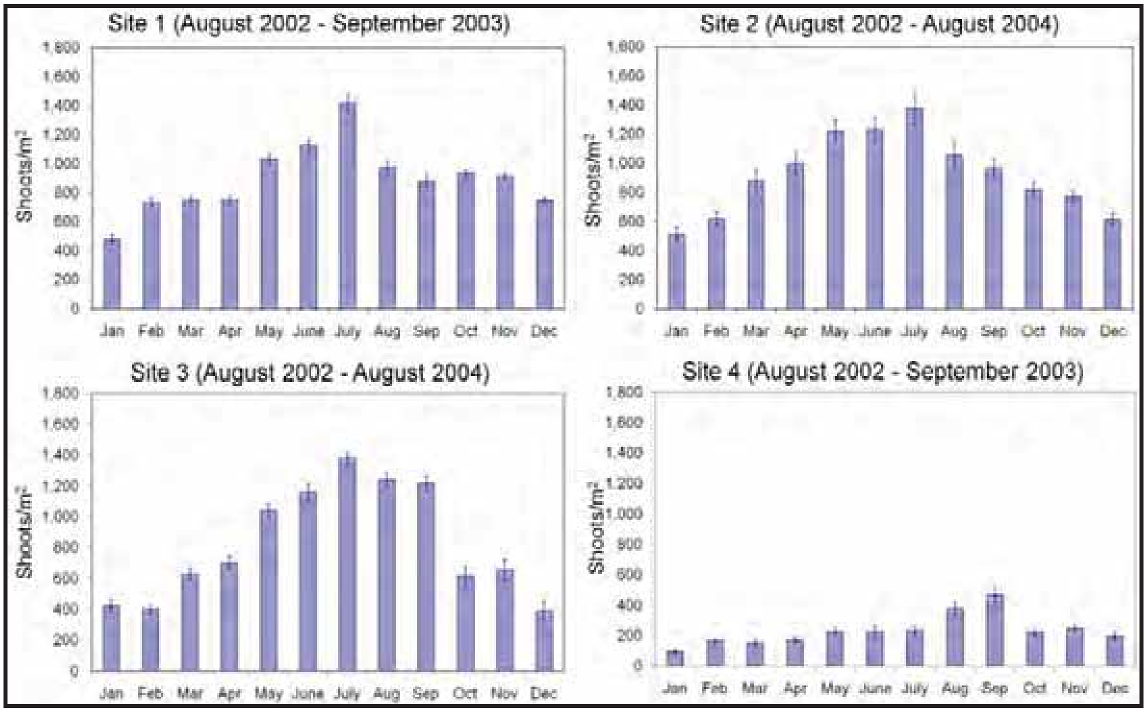


FIGURE 7-65. *S. FILIFORME* MEAN SHOOT COUNTS (±SE) AT SITES 1, 2 AND 4 PRIOR TO HURRICANES

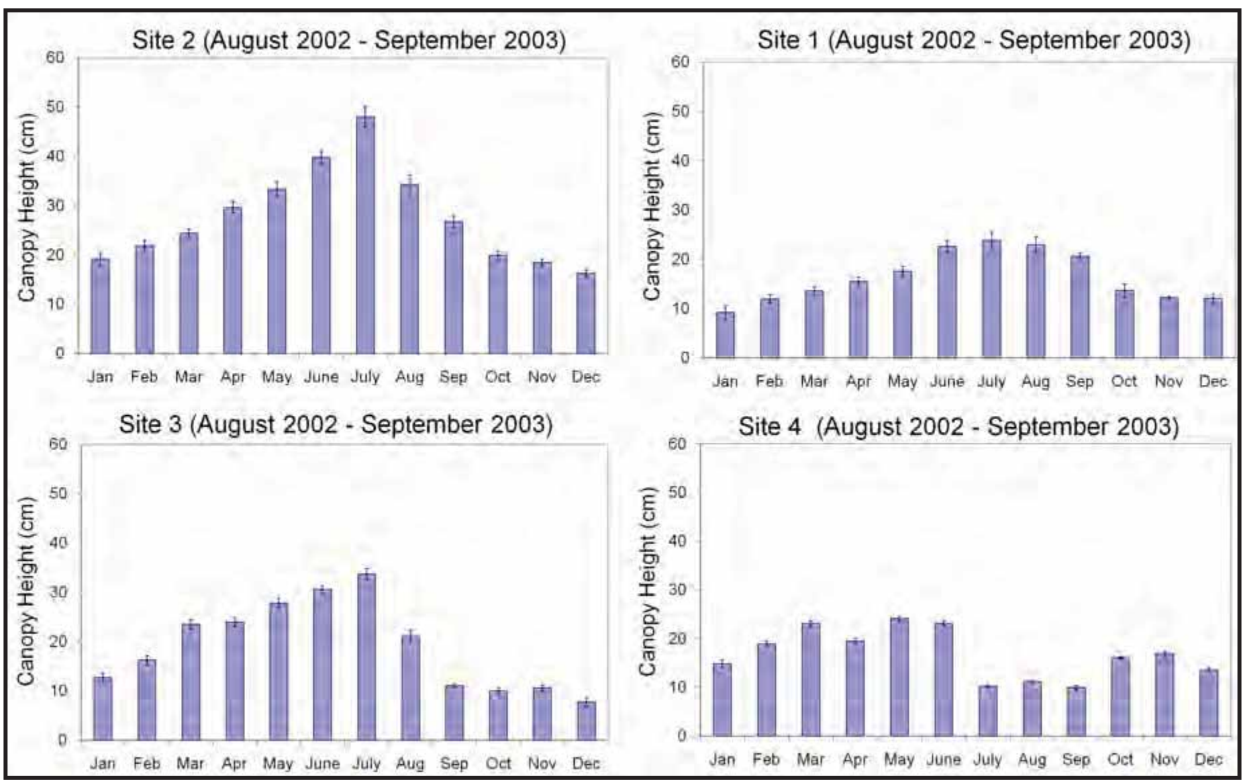


FIGURE 7-66. *S. FILIFORME* CANOPY HEIGHT (MEAN±SE) AT SITES 1, 2 AND 4 PRIOR TO HURRICANES AUGUST 2002 TO SEPTEMBER 2003

Figure 7-67 shows preliminary data on drift (primarily red algae) and attached algae (*Caulerpa* sp.) percent occurrence, as well as epiphyte coverage at Site 2 prior to the hurricane impacts. All months had low occurrence of drift algae with apparent peaks from February through April and another in November. Attached algae occurrence was greatest from March through June. Additional data analysis is needed to fully evaluate algae data collected during this project.

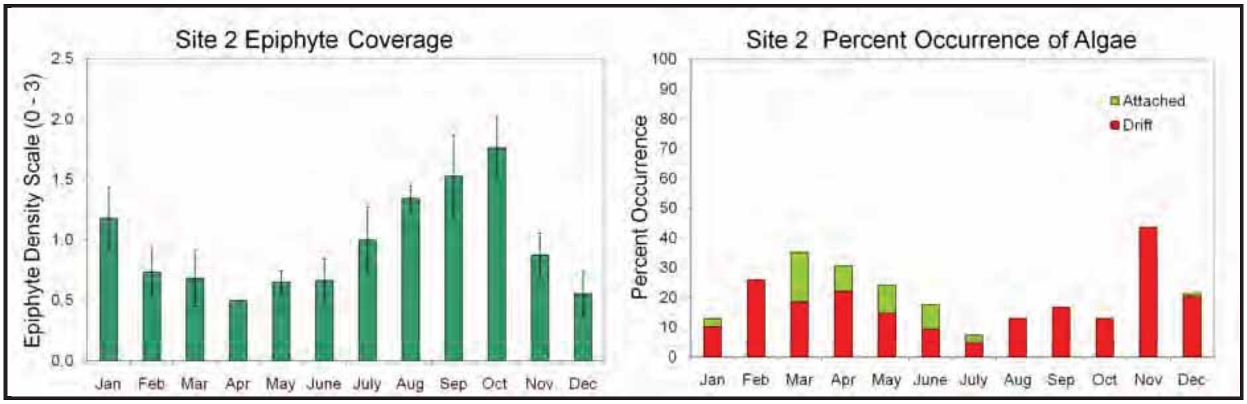


FIGURE 7-67. MACRO ALGAE A) MEAN COVERAGE (± SE) AND B) PERCENT OCCURRENCE PRIOR TO HURRICANE IMPACTS AT SITE 2 PRIOR TO HURRICANES AUGUST 2002 TO AUGUST 2004

In addition to the seasonal decline in water clarity toward the end of the summer, there is also a seasonal rise in water level. ***Figure 7-68*** provides a conceptual diagram based on water level readings from the Jensen Beach Causeway collected during pre-hurricane SAV monitoring, mean canopy height at Site 2, and the elevation of the lagoon bottom near the middle of Site 2. The late summer and early fall decline in canopy height and shoot counts coincides with reduced photoperiod, declining temperatures, increased water levels over reduced canopy heights, and greater light attenuation in the water column due to increased turbidity and color and shallower Secchi depths. All of these factors help define the seasonal variability of lagoon seagrasses.

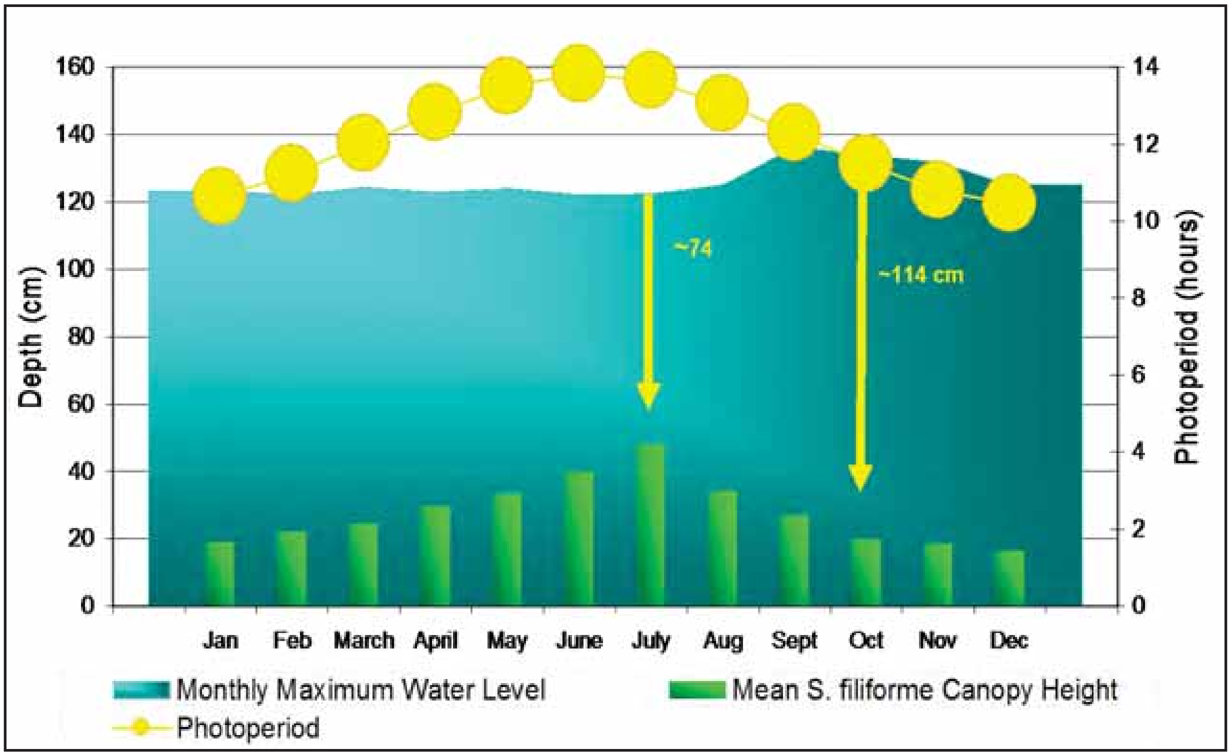


FIGURE 7-68. CONCEPTUAL DIAGRAM OF NATURAL FACTORS DRIVING SEASONAL VARIATION IN SEAGRASS GROWTH

In 2004 and 2005, hurricanes impacted the area. ***Figure 7-69*** summarizes the hurricane effects experienced at Site 2. From project initiation in 2002 through 2004, the site supported a lush *S. filiforme* bed. Low salinities and associated poor water quality greatly impacted *S. filiforme* at this site and likely other beds in the area. The hurricanes also altered bathymetry on the east and west edges of the site, covering seagrasses.

Figure 7-70 documents the decline in *S. filiforme* percent occurrence at Site 2 following the 2004 hurricanes. However, the steepest decline in percent occurrence of this species occurred in 2005 after Hurricane Wilma. Much of the *S. filiforme* was lost from the site. *H. johnsonii* followed by *H. wrightii* colonized the former *S. filiforme* habitat and recruited throughout the site. Available data indicates a clear trend toward recovery of the *S. filiforme* bed at this site and these findings should apply to other areas in the vicinity.

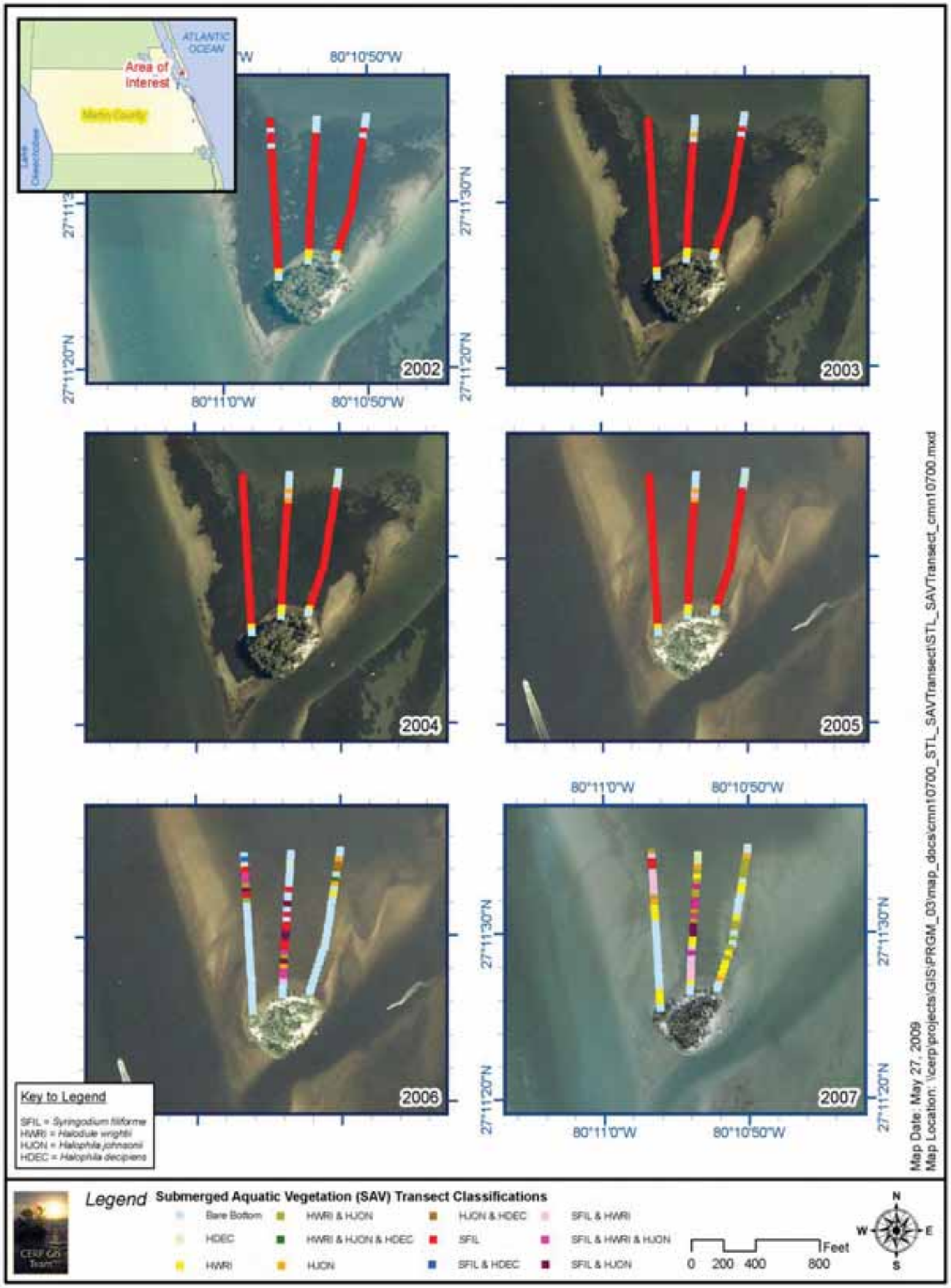


FIGURE 7-69. CHANGES IN SEAGRASS PERCENT OCCURRENCE AND DOMINANT SPECIES FROM 2002 TO 2007 AT SITE 2 SHOWING HURRICANE EFFECTS

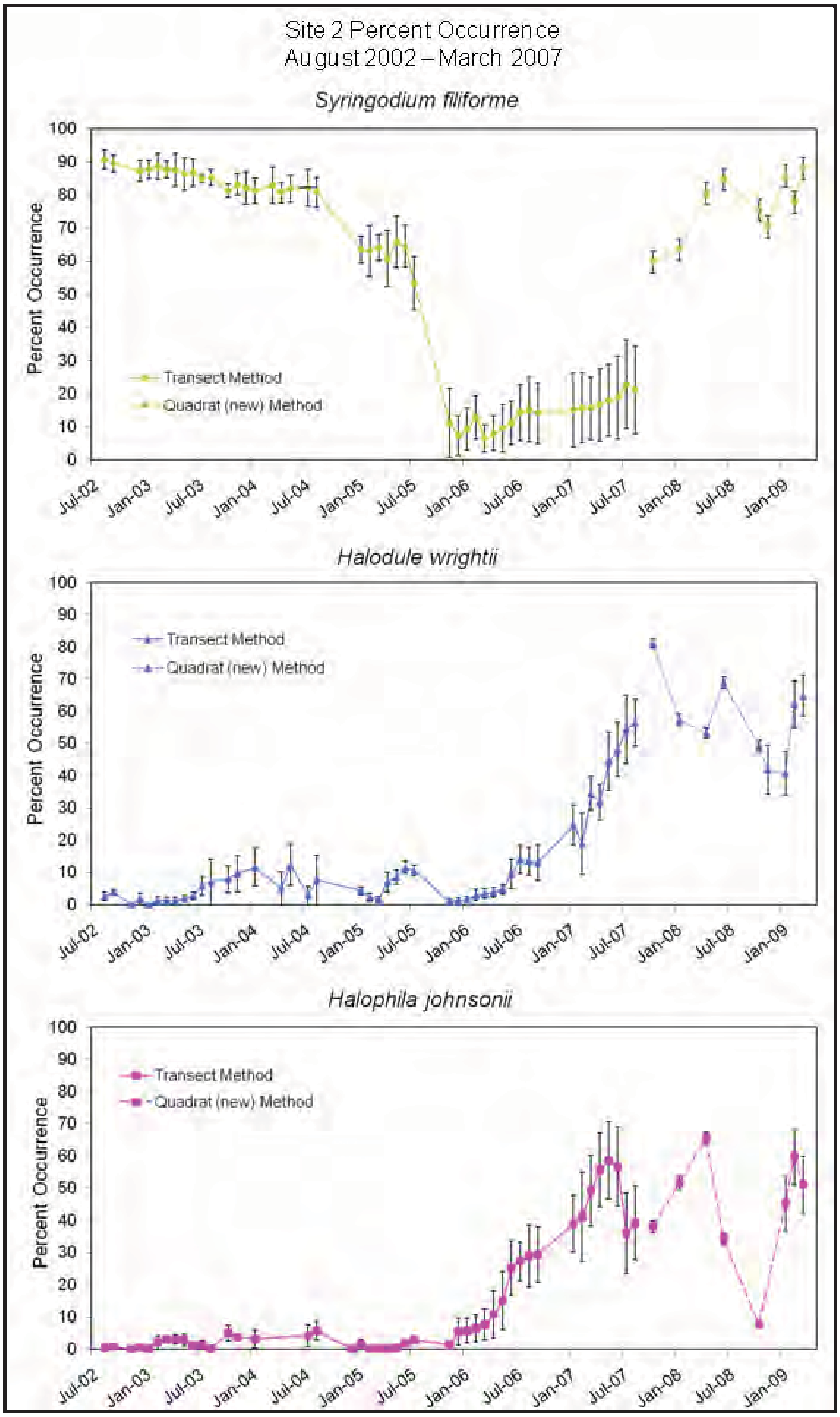


FIGURE 7-70. MEAN (± SE) PERCENT OCCURRENCE OF *S. FILIFORME*, *H. WRIGHTII* AND *H. JOHNSONII* AT SITE 2 FROM AUGUST 2002 TO MARCH 2007

At Site 3 (as depicted in*Figure 7-71*), the *S. filiforme* bed was buried by sediment during the 2004 hurricanes. As of March 2009, very little recovery of *S. filiforme* has occurred at this site (***Figure 7-71***). As observed at Site 2, *H. wrightii* percent occurrence increased following the hurricane impacts, as did percent occurrence of *H. johnsonii*. *H. decipiens* percent occurrence does not show a distinct trend at this site.

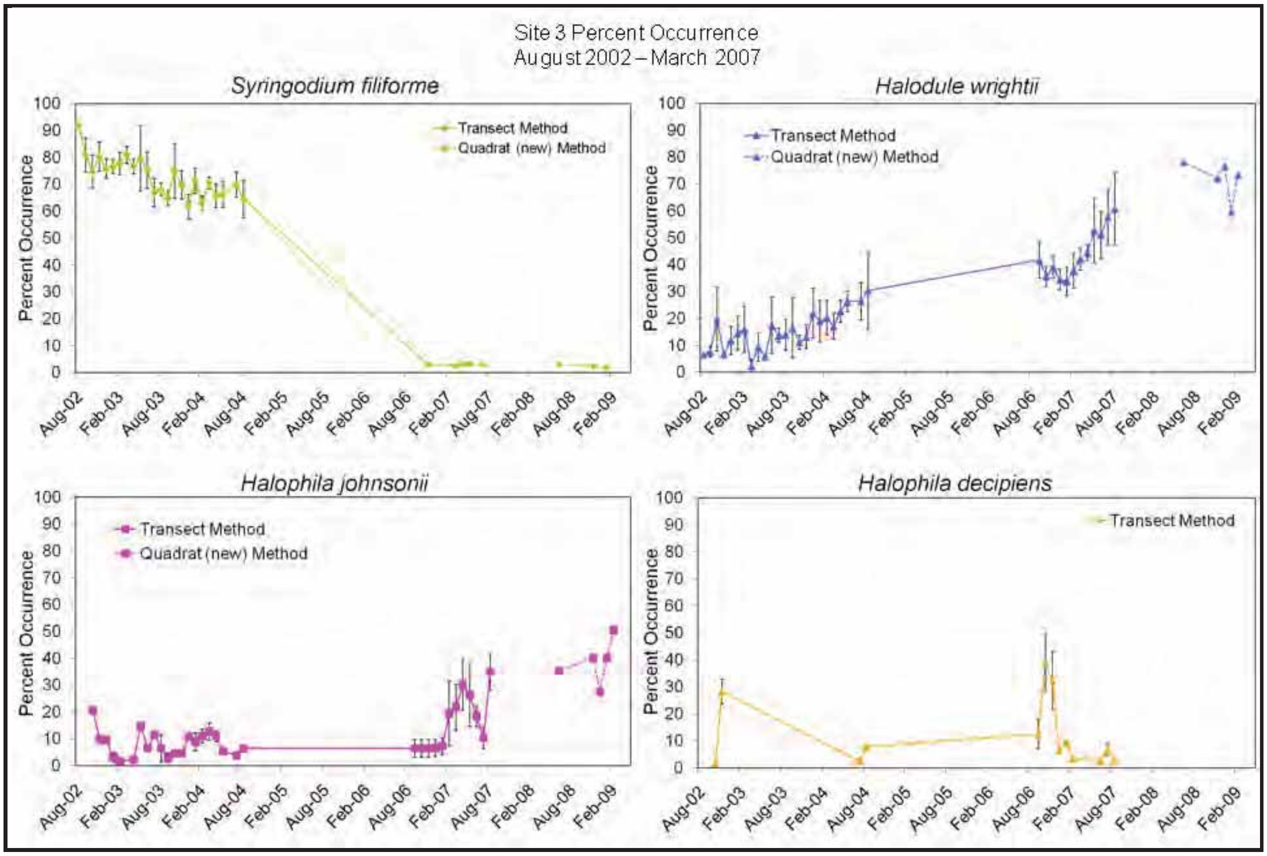


FIGURE 7-71. MEAN (±SE) PERCENT OCCURRENCE OF *S. FILIFORME*, *H. WRIGHTII*, *H. JOHNSONII* AND *H. DECIPIENS* AT SITE 3 FROM AUGUST 2002 TO MARCH 2007

7.3.4.2.2 Patch-Quadrat Method

As the five-year study discussed above was concluding, a new SAV monitoring methodology was developed. This methodology is discussed in more detail at the beginning of the SAV section. Ten monitoring sites were established in Southern Indian River Lagoon (***Figure 7-72***). The site sizes and current SAV species composition are provided in ***Table 7-4***. From October 2007 through November 2008, SFWMD scientists tested the new methods at least once at each of the ten sites. Beginning in December 2008, a team of consultants began monitoring each of the ten sites bimonthly. Three sites, WILL_CK, Site 2 and SLI_SE, are monitored monthly to provide information for SFWMD operation decisions.

TABLE 7-4. SITE SIZES AND CURRENT SAV SPECIES COMPOSITION OF NEW SAMPLING SITES IN SOUTHERN INDIAN RIVER LAGOON AND ST. LUCIE ESTUARY

Site Name	Seagrass Species	Size
BSI (Site 2)	*Syringodium filiforme* *Halodule wrightii* *Halophila johnsonii* *Thalassia testudinum* *Halophila decipiens*	1.87 acres
SLI_NE	*Halodule wrightii* *Halophila johnsonii* *Thalassia tetudinum*	1.90 acres
SITE_1	*Halodule wrightii* *Syringodium filiforme* *Halophila johnsonii* *Halophila decipiens*	1.91 acres
WILL_CK	*Halodule wrightii* *Halophila johnsonii* *Thalassia testudinum*	1.88 acres
SLI_SE	*Halodule wrightii* *Halophila johnsonii*	0.80 acres
FP_NW	*Halodule wrightii* *Syringodium filiforme* *Halophila johnsonii* *Halophila engelmannii* *Thalassia testudinum*	1.71 acres
FP_NE	*Halodule wrightii* *Thalassia testudinum* *Syringodium filiforme*	1.92 acres
SITE_3	*Halophila johnsonii* *Halodule wrightii* *Halophila decipiens* *Syringodium filiforme*	0.69 acres
OC_BR_PK	*Halodule wrightii* *Syringodium filiforme*	1.93 acres
JOES_PT	*Halodule wrightii* *Syringodium filiforme* *Halophila johnsonii* *Halophila decipiens*	1.83 acres

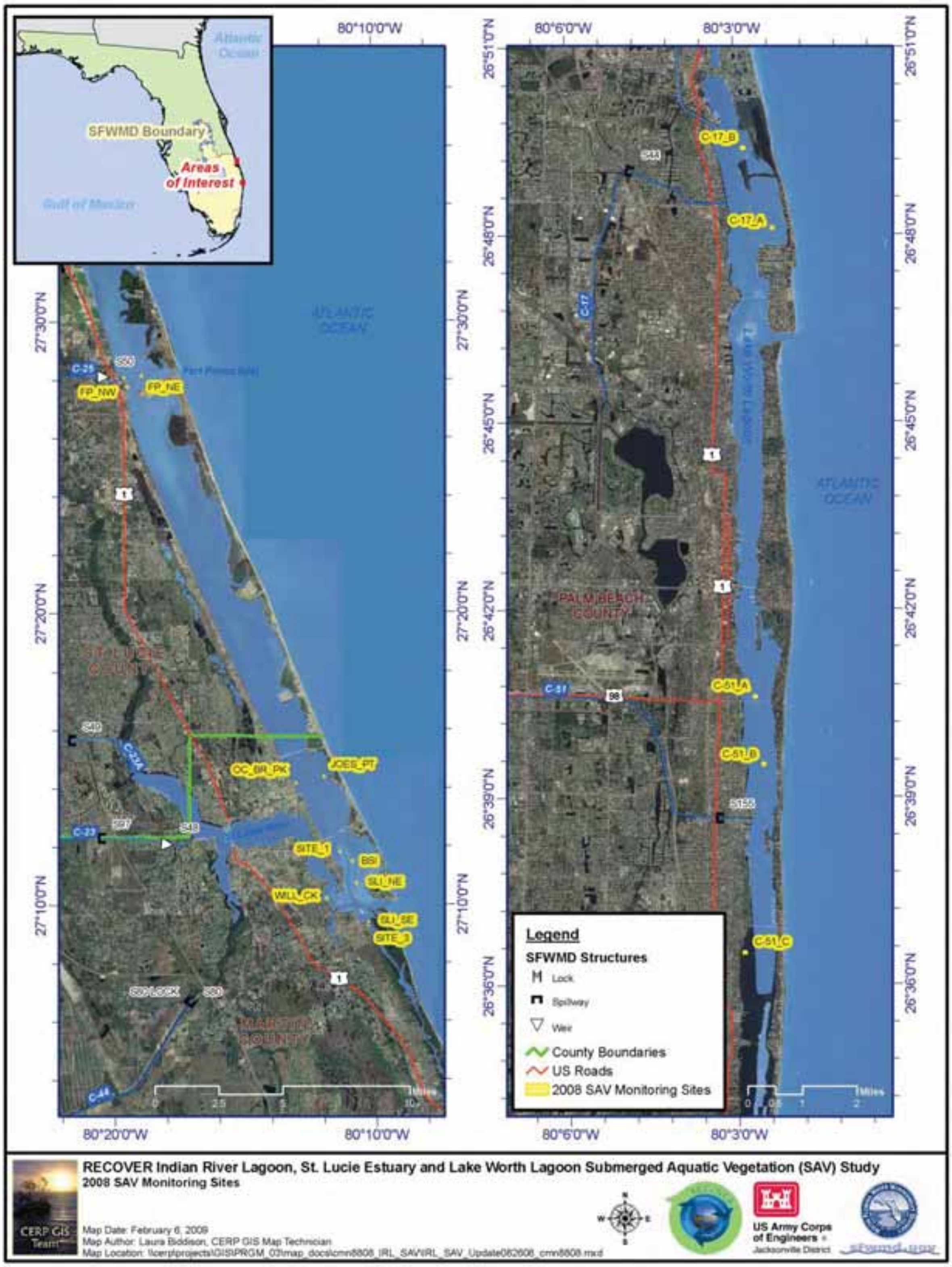


FIGURE 7-72. NEW METHODOLOGY SUBMERGED AQUATIC VEGETATION SAMPLING SITES IN SOUTHERN INDIAN RIVER LAGOON, ST. LUCIE ESTUARY AND LAKE WORTH LAGOON

As discussed above, the new monitoring program was formally started in December 2008. Percent cover and canopy height measured during testing and initial monitoring events are shown in ***Figure 7-73*** and ***Figure 7-74***, respectively, for Site 2. The canopy species had peak canopies in June (monitoring was not conducted in July), which compares well with pre-hurricane data. Peak *S. filiforme* heights (approximately 30 centimeters) are comparable to those measured at Sites 1 and 3 during the monthly monitoring project, but not as high as peak heights at Site 2 prior to the hurricanes. *S. filiforme* canopy height was significantly greater than *H. wrightii* during peak growth.

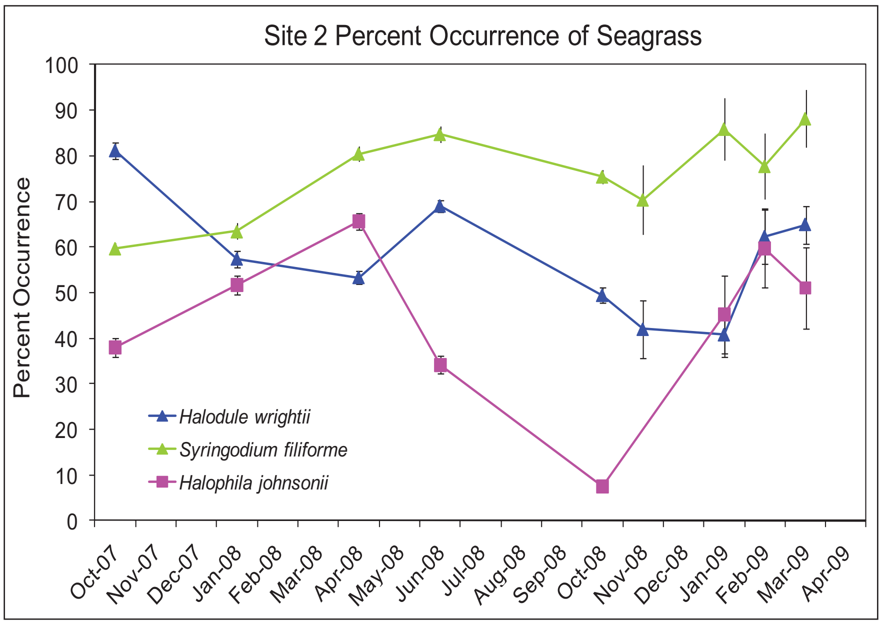


FIGURE 7-73. MEAN (+SE) PERCENT COVER OF SUBMERGED AQUATIC VEGETATION AT SITE 2 DURING TESTING AND INITIAL MONITORING EVENTS USING NEW METHODOLOGY

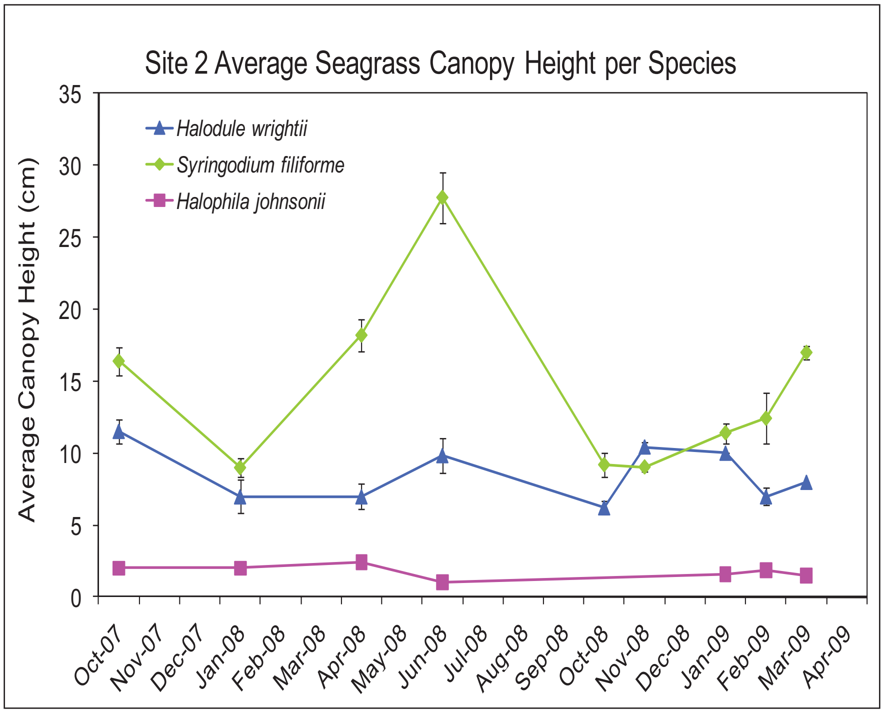


FIGURE 7-74. MEAN (+SE) CANOPY HEIGHT OF SUBMERGED AQUATIC VEGETATION AT SITE 2 DURING TESTING AND INITIAL MONITORING EVENTS USING NEW METHODOLOGY

The new monitoring program includes a site within the St. Lucie Estuary at Willoughby Creek (WILL_CK in ***Figure 7-72***). This site supports the most upstream, persistent SAV bed in the estuary. Initial monitoring occurred in July 2008, during which time the bed supported both *H. wrightii* and *H. johnsonii* (***Figure 7-75***). Following the monitoring event, freshwater discharges into the estuary reduced salinities in the estuary (***Figure 7-77***). The next monitoring event occurred in October 2008. During that monitoring event, no *H. johnsonii* was observed and percent occurrence of *H. wrightii* declined (***Figure 7-75)***. Continued monitoring has documented recovery of *H. wrightii*, but not *H. johnsonii*. ***Figure 7-76*** shows the seagrass canopy height at this site.

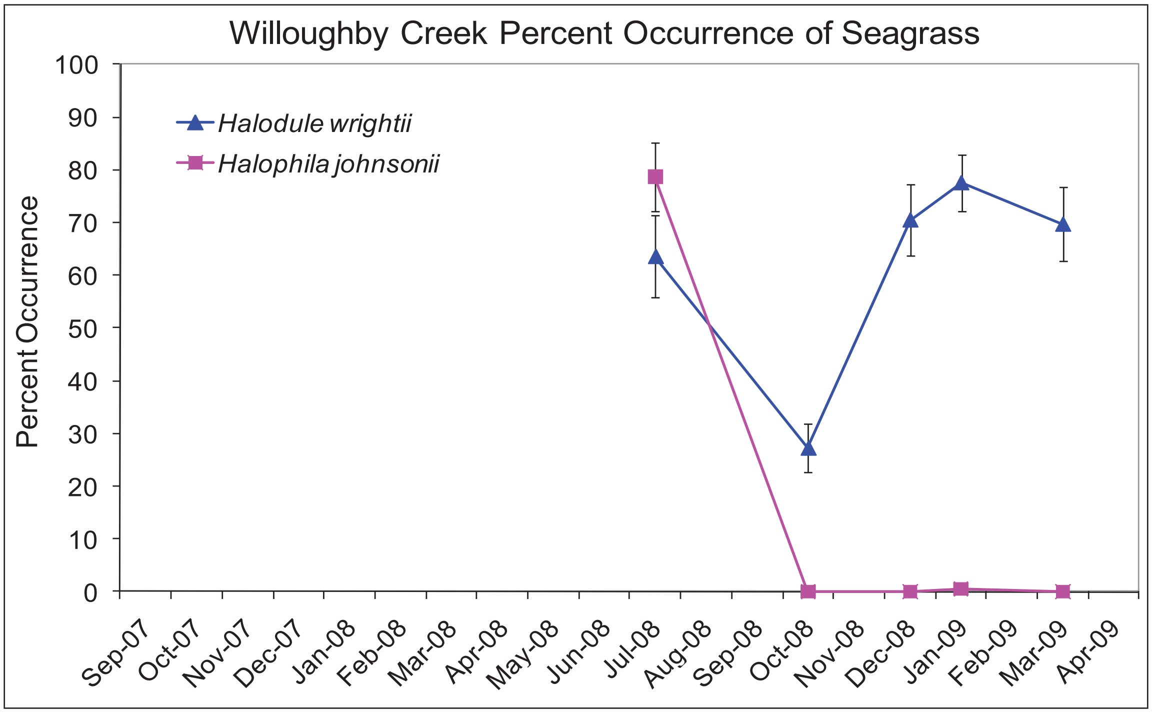


FIGURE 7-75. MEAN (±SE) PERCENT OCCURRENCE OF SEAGRASS AT WILLOUGHBY CREEK IN THE ST. LUCIE ESTUARY FROM JULY 2008 TO MARCH 2009

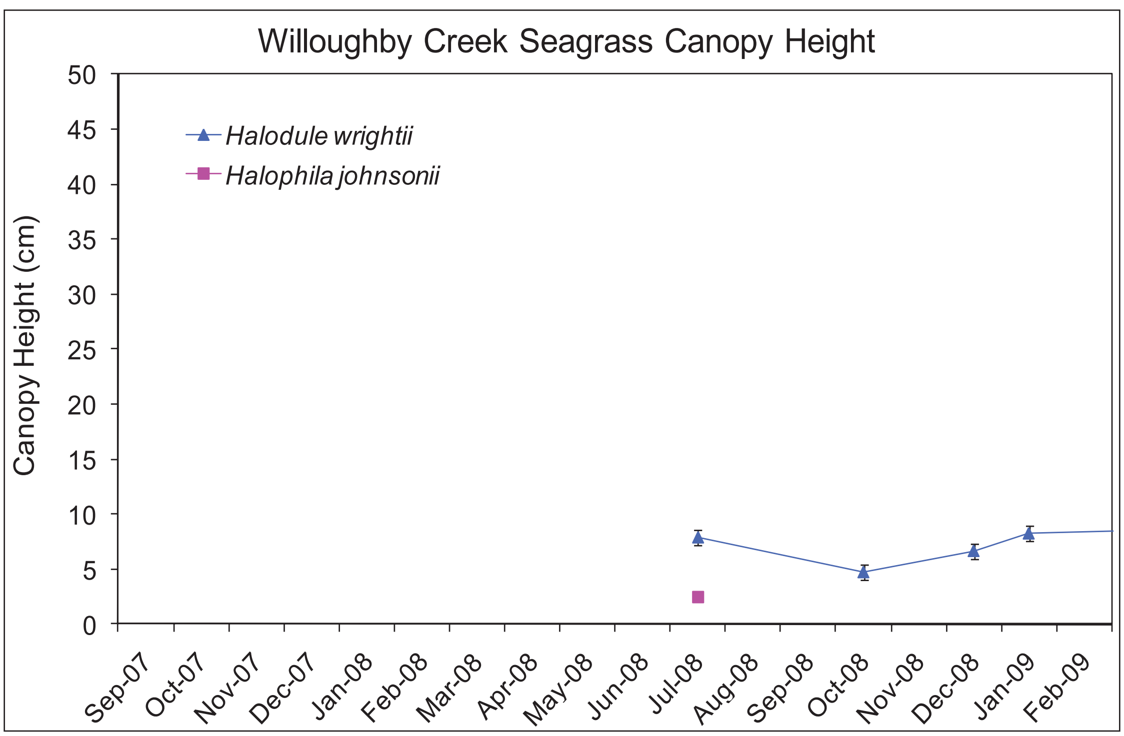


FIGURE 7-76. MEAN (±SE) SHOOT COUNTS OF SEAGRASS AT WILLOUGHBY CREEK IN THE ST. LUCIE ESTUARY FROM JULY 2008 TO MARCH 2009

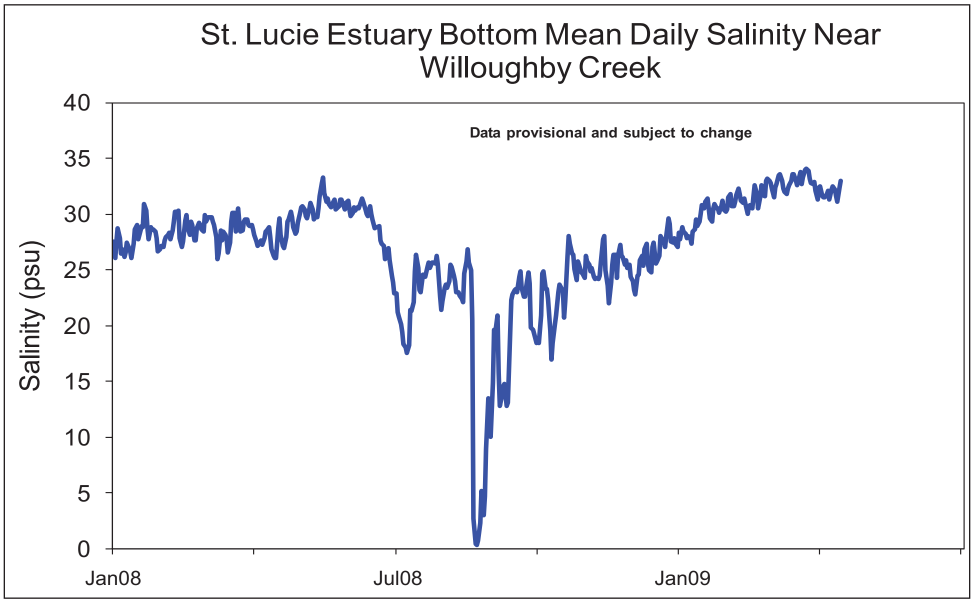


FIGURE 7-77. SALINITY NEAR THE WILLOUGHBY CREEK SITE

7.3.4.3 Summary

While the regions‟ SAV experienced severe stress as a result of the hurricanes and associated freshwater discharge, the system‟s SAV status is improving as documented by increases in mapped acreage, recruitment into areas left bare following the hurricanes, and transition from pioneer to canopy species.

During the next SSR update period, the SFWMD intends to map the Southern Indian River Lagoon from 2009 imagery if funding is available. It is recommended that groundtruthing during this effort include inspections of areas mapped as *H. johnsonii* during the 2007 to 2008 species specific mapping to determine if recovery is continuing with species composition shifting to canopy species.

7.3.5 Loxahatchee River Estuary

The Loxahatchee River District‟s Wild Pine Laboratory has been monitoring seagrasses throughout the Loxahatchee River Estuary since 2003. This monitoring program has consisted of three components:

1. Landscape-scale mapping throughout the river system collected during summer 2007
2. Patch-scale fixed transect monitoring collected bimonthly between June 2003 and August 2007
3. Patch-scale quadrat monitoring collected bimonthly since October 2007

Although the total estuarine area is approximately 1,334 acres, less than 1,062 acres have a salinity regime suitable for seagrass growth. Six species of seagrass have been found in the

Loxahatchee River Estuary: *H. johnsonii*, *H. wrightii*, *H. decipiens*, *S. filiforme*, *T. testudinum* and *H. engelmannii*.

Four seagrass beds in the central embayment of the Loxahatchee River Estuary and a reference seagrass bed in the Southern Indian River Lagoon were selected as sample sites for patch-scale monitoring based on location across the salinity gradient, seagrass cover and persistence. The monitoring sites are Northwest Fork, Pennock Point, Sand Bar, North Bay and Hobe Sound, which is the reference site (***Figure 7-78***). The red polygons represent actual size and shape of seagrass bed monitored using the patch-quadrat methodology.

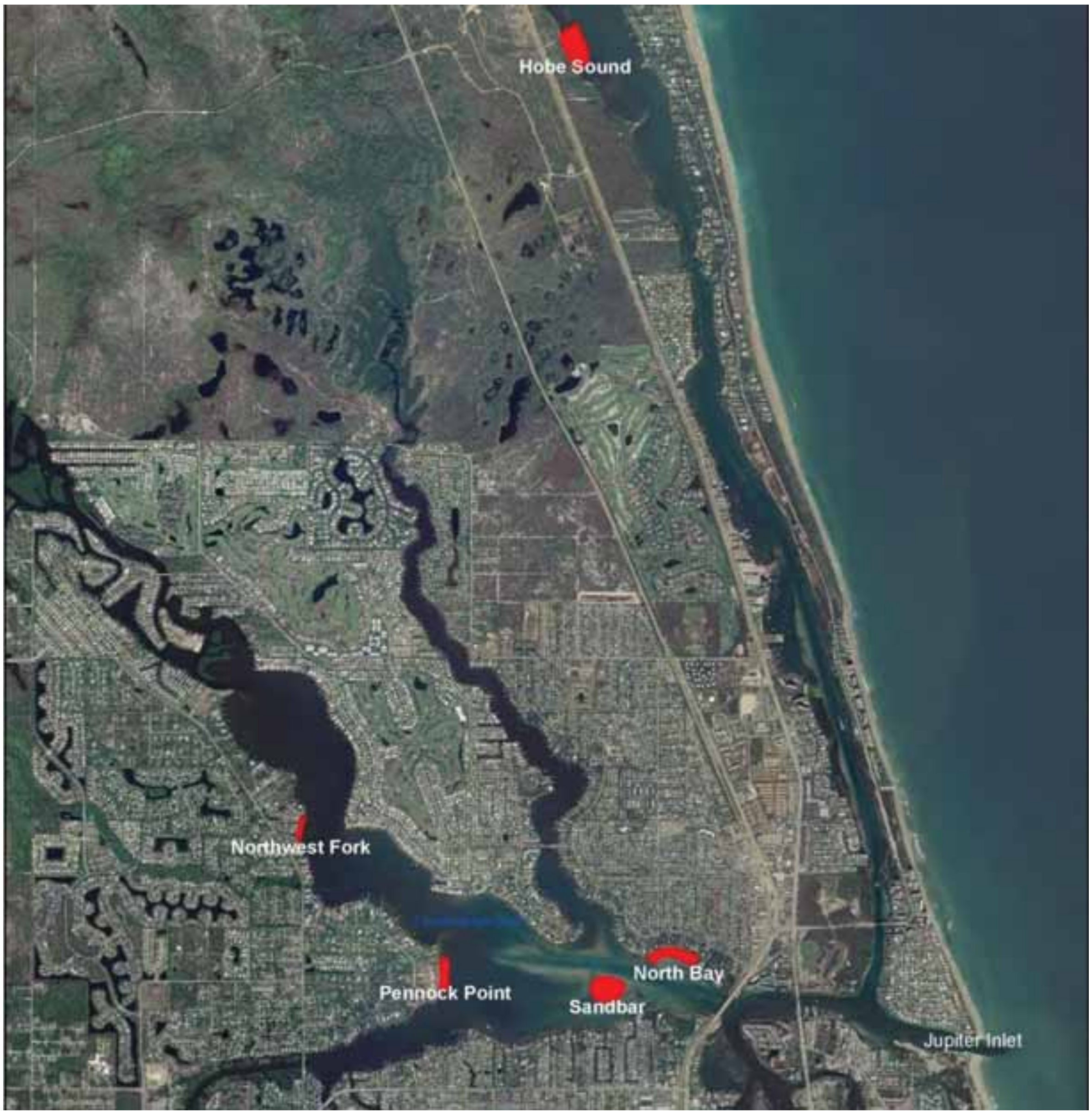


FIGURE 7-78. LOXAHATCHEE RIVER ESTUARY SEAGRASS MONITORING SITES

The Northwest Fork site, added in October 2007, is the most affected by fresh water flowing downstream from the Northwest Fork of the Loxahatchee River. *H. johnsonii* and *H. wrightii* were found at this site during earlier assessments; however, *H. johnsonii* and *H. wrightii* was not encountered in the previous transect-based studies in this area. The Pennock Point site is directly influenced by water flowing downstream from both the Northwest and Southwest Forks. The Sand Bar site is directly influenced by water flowing in from the inlet and downstream from both the Northwest and Southwest Forks. The North Bay is the most downstream site, and therefore, experiences the most stable, marine-like conditions among the four sampling sites. The Hobe Sound site, added in June 2005, is not affected by fresh water discharged from the Loxahatchee River watershed, making it an excellent reference site. It experiences more stable, marine-like salinity conditions than the Loxahatchee River Estuary. Acreage of SAV and species currently present at each site are provided in ***Table 7-5***.

TABLE 7-5. SITE SIZES AND CURRENT COMMON SUBMERGED AQUATIC VEGETATION SPECIES OF SAMPLING SITES IN THE LOXAHATCHEE RIVER ESTUARY

Site Name	Most Common Seagrass Species	Size
Northwest Fork	*Halophila johnsonii* *Halodule wrightii*	0.97 acres
Pennock Point	*Halodule wrightii* *Halophila johnsonii*	2.96 acres
Sand Bar	*Halodule wrightii* *Halophila johnsonii* *Syringodium filiforme*	8.71 acres
North Bay	*Halophila johnsonii* *Syringodium filiforme* *Halodule wrightii*	5.29 acres
Hobe Sound (reference site)	*Syringodium filiforme* *Halodule wrightii* *Halophila johnsonii*	10.79 acres

7.3.5.1 Landscape-Scale Mapping

During summer 2007, seagrass presence and percent cover was determined at 1,085 sites in the Loxahatchee River Estuary (***Figure 7-79***). Seventy-eight percent of samples contained at least one species of seagrass, while seagrasses were absent from 22 percent of samples. *H. johnsonii* was the dominant species, spatially occurring in 71 percent of the samples. *H. wrightii* occurred in 50 percent of samples, while *H. decipiens* was found in 11 percent, *S. filiforme* in four percent, and *T. testudinum* in four percent of samples. Although *H. engelmannii* has been observed in the Loxahatchee River Estuary in previous sampling events (i.e. 2003 mapping study; SFWMD, 2006), it was not found during this mapping effort.

Seagrass distribution was characterized within four major segments of the estuary: central embayment, North Fork, Northwest Fork and Southwest Fork. Interpolated seagrass maps showed more seagrass than expected. In total, seagrasses covered 393 acres of the estuary (***Figure 7-80***), representing 37 percent of the total area identified as potentially suitable for seagrasses. Seagrass percent cover was highest in the central embayment and lowest in the Southwest Fork, through which flood control releases occur. Unshaded areas in ***Figure 7-80*** were deemed unsuitable for seagrass colonization due to depth and were not sampled.

FIGURE 7-79. LANDSCAPE-SCALE SEAGRASS MONITORING SITES DURING SUMMER 2007 IN LOXAHATCHEE RIVER ESTUARY

FIGURE 7-80. SEAGRASS IN THE LOXAHATCHEE RIVER ESTUARY DURING SUMMER 2007

Both *H. johnsonii* and *H. wrightii* were heavily distributed throughout the estuary. Surprisingly, *H. johnsonii*, a threatened species, was the dominant species in each segment, and covered approximately 354 acres (***Figure 7-81***). *H. wrightii*, which tolerates wide salinity fluctuations and is considered a pioneer species, was expected to be dominant but covered only 256 acres. It occurred in healthy beds upstream in each river fork. *H. decipiens* covered approximately 88 acres. This species was commonly found at the greatest depths because of its low light requirements (Kemp et al., 2004) and may have been underestimated in the deeper portions of the system. *S. filiforme* and *T. testudinum* (not pictured in figure) were found in relatively small, isolated patches in downstream segments of the estuary and each covered approximately 14 acres. *S. filiforme* is intolerant of wide salinity fluctuations, and occurred primarily as small beds in the central embayment where water quality was most consistently suitable. *H. engelmannii* was not encountered during the summer 2007 sampling, though it was observed in small, low density patches during 2003 (SFWMD, 2006). It may still occur in the estuary in low densities or it may have gone locally extinct following the hurricanes in 2004 and 2005 (Ridler et al., 2006).

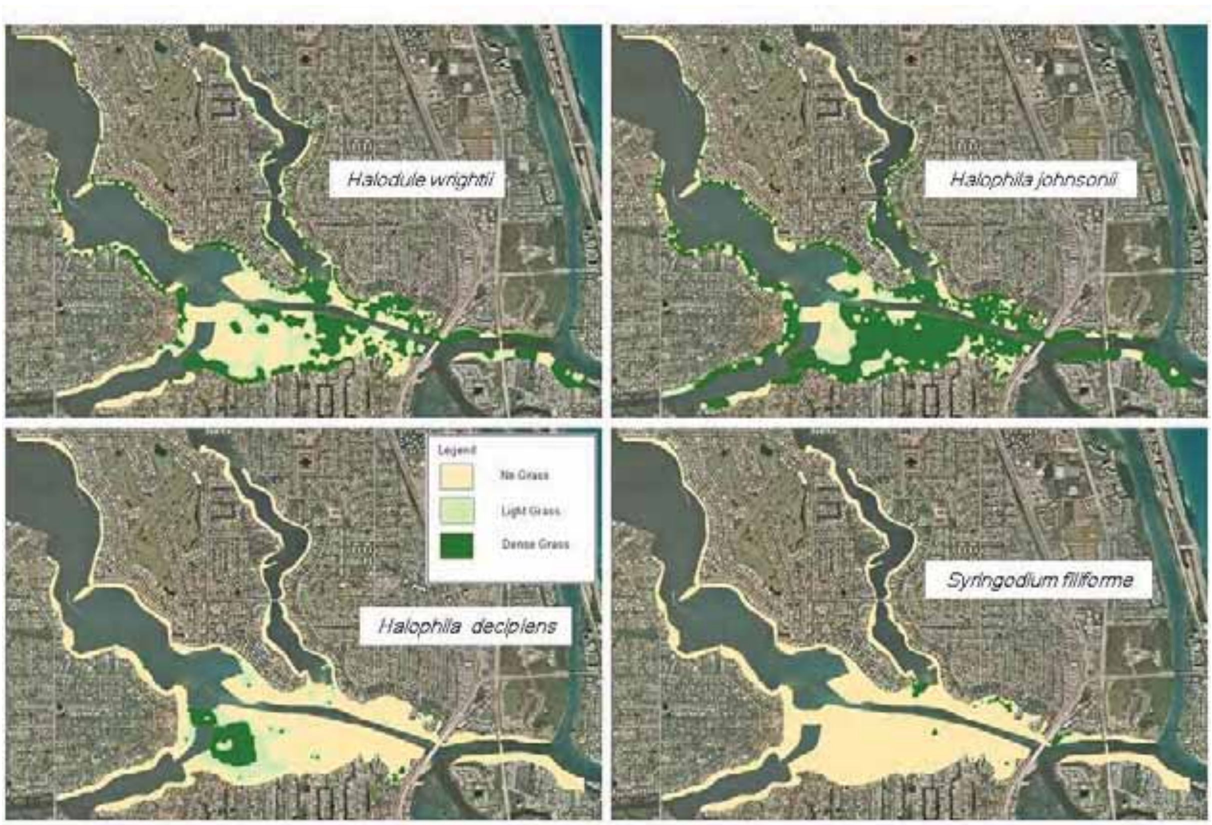


FIGURE 7-81. SPECIES SPECIFIC SEAGRASS COVERAGE MAPS FOR THE FOUR MOST DOMINANT SPECIES IN THE LOXAHATCHEE RIVER ESTUARY DURING SUMMER 2007

Seagrass diversity varied among segments with the central embayment and North Fork being the most diverse with five species present and the Southwest Fork being the least diverse with two species (*H. johnsonii* and *H. wrightii*). *H. johnsonii*, *H. wrightii* and *H. decipiens* were found in the Northwest Fork samples. The central embayment has superior water quality for seagrass with less variable salinities, low light attenuation, relatively low nutrient concentrations, and low chlorophyll *a* concentrations (Loxahatchee River District unpublished data). Such high seagrass diversity in the North Fork was not expected. The landscape mapping data suggest conditions in the North Fork downstream of the Tequesta Drive Bridge were of sufficient quality to support the more sensitive seagrass species *S. filiforme* and *T. testudinum*. Lowest seagrass diversity was observed in the Southwest Fork, which is along the route through which flood control releases are shunted through the C-18 Canal network to the ocean.

Because the 2007 landscape-level monitoring resulted in the most detailed seagrass maps to date for the Loxahatchee River Estuary, it is inappropriate to compare it to results from any previous mapping efforts. This monitoring should continue in 2010.

7.3.5.2 Patch-Scale Fixed Transects and Quadrats

Fixed transects were monitored monthly from June 2003 through August 2007 at Pennock Point, Sand Bar, North Bay and Hobe Sound. Seagrass sampling sites were monitored bimonthly using quadrats from October 2007 through December 2008. Percent cover and canopy height was determined for each seagrass species encountered.

Figure 7-82 presents the percent occurrence and cover for the most common seagrass species found at the sites monitored since 2003. These data suggest strong differences in seagrass health among Loxahatchee River Estuary sites across the upstream-downstream gradient.

Figure 7-83 illustrates the variation in species composition and spatial variation of the patch-quadrat data from the Hobe Sound, the marine dominated reference site, and North Bay, the most downstream sampling site. Stacked bars represent relative percent cover for each species for each individual sample point assessed during this period. The image shows the dominance of *S. filiforme* and *H. wrightii* at the reference site. *H. decipiens* and *H. johnsonii* occurred primarily in the deepest areas. In contrast, the North Bay site in the Loxahatchee River shows the dominance of *H. johnsonii*, *H. wrightii* and *S. filiforme*. Also, shown is the large sandbar that borders the seagrass bed to the south.

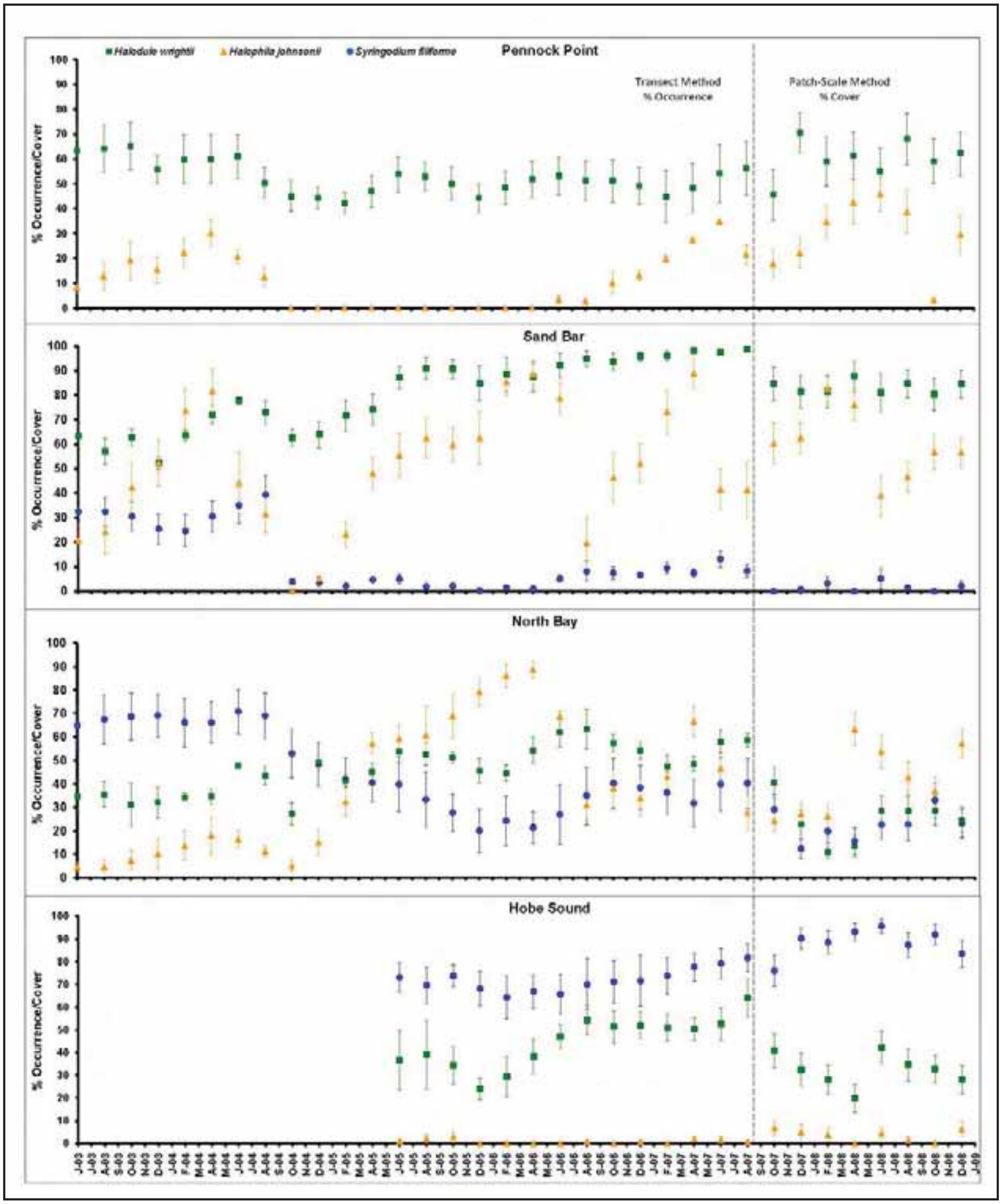


FIGURE 7-82. SUMMARY OF PERCENT OCCURRENCE (±SE) FROM THE TRANSECT METHOD AND PERCENT COVER (±SE) FROM THE QUADRAT METHOD AT EACH SAMPLING SITE, 2003 THROUGH 2008

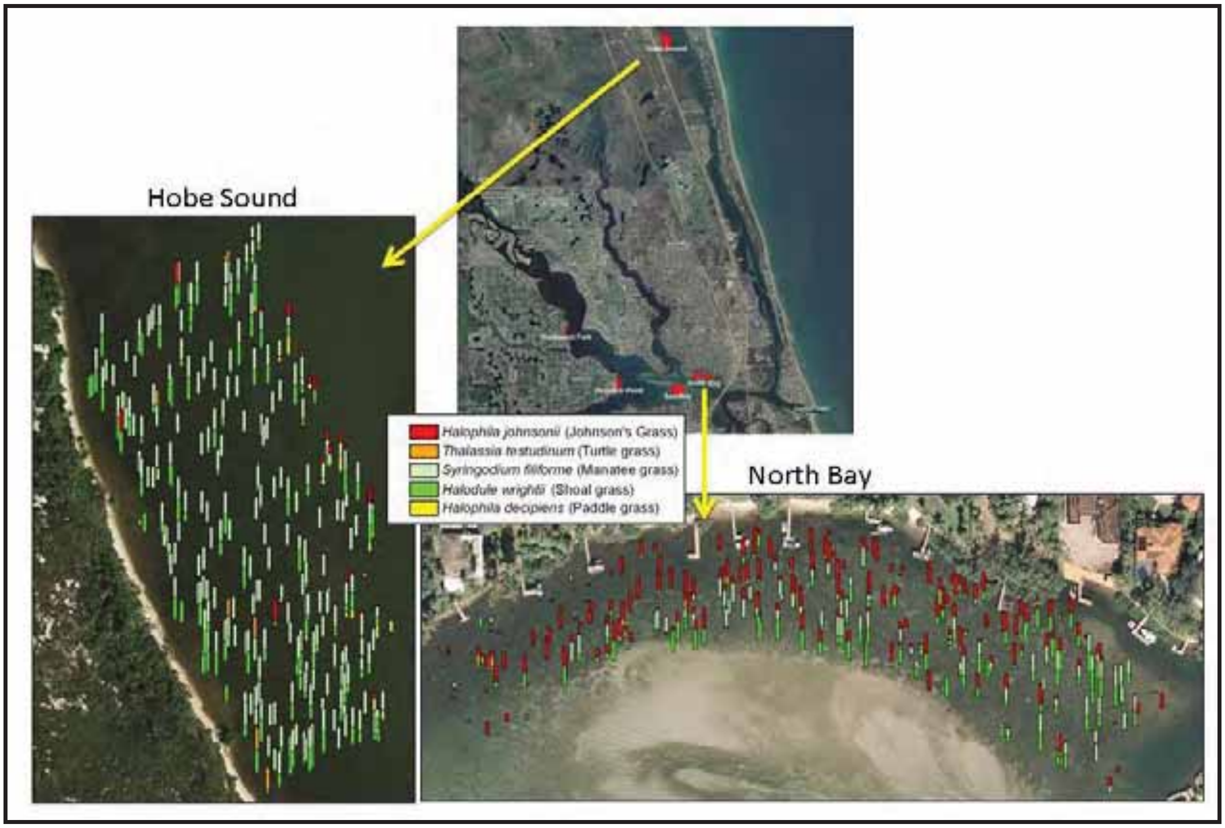


FIGURE 7-83. STACKED BAR GRAPHS SHOW SPECIES SPECIFIC PERCENT COVER SEAGRASS DATA WITHIN EACH SAMPLING SITE ACROSS THE ENTIRE SAMPLING PERIOD (OCTOBER 2007 - DECEMBER 2008)

Figure 7-84 shows the variation in species composition and percent cover across the Loxahatchee River Estuary‟s salinity gradient. Northwest Fork is the most upstream site. The Pennock Point site is influenced by flows from the Northwest and Southwest Forks. The Sandbar and North Bay sites in the central embayment are further downstream and receive more marine water from the Jupiter Inlet during tidal flushing.

The Pennock Point site, influenced by freshwater flows from the Northwest Fork and the Southwest Fork, was dominated by *H. wrightii* and *H. johnsonii* (***Figure 7-82***). Most striking was the nearly two-year absence of *H. johnsonii* following the 2004 hurricanes. Clearly, the volume and duration of freshwater discharges during and immediately following the storms severely impacted *H. johnsonii* at this site. In contrast, the low percent cover of *H. johnsonii* observed in October 2008 suggests an example of a modest impact that may have resulted in loss of leaves. However, based on the substantial recovery observed during the following sampling period, the rhizomes likely survived and the grasses recovered. Spatially, *H. wrightii* was dominant in shallower, nearshore areas and *H. johnsonii* was most commonly found along the deep edge of the bed (***Figure 7-84***). *S. filiforme* was likely absent from this site because of the wide fluctuations in salinity conditions (Loxahatchee River District unpublished data).Further downstream, at the Sand Bar site, *H. wrightii* and *H. johnsonii* dominated seagrass cover throughout the year, though *H. johnsonii* exhibited stronger seasonal declines (***Figure 7-82***). Surprisingly, *H. johnsonii* recovered within months of the hurricanes. In general, a nearly even mixture of *H. wrightii* and *H. johnsonii* characterized this site, though *H. johnsonii* was most prevalent in some of the deeper portions of the seagrass bed (i.e., the southwest corner) (***Figure 7-84***). *S. filiforme* was present at this site in isolated, sparse patches.T

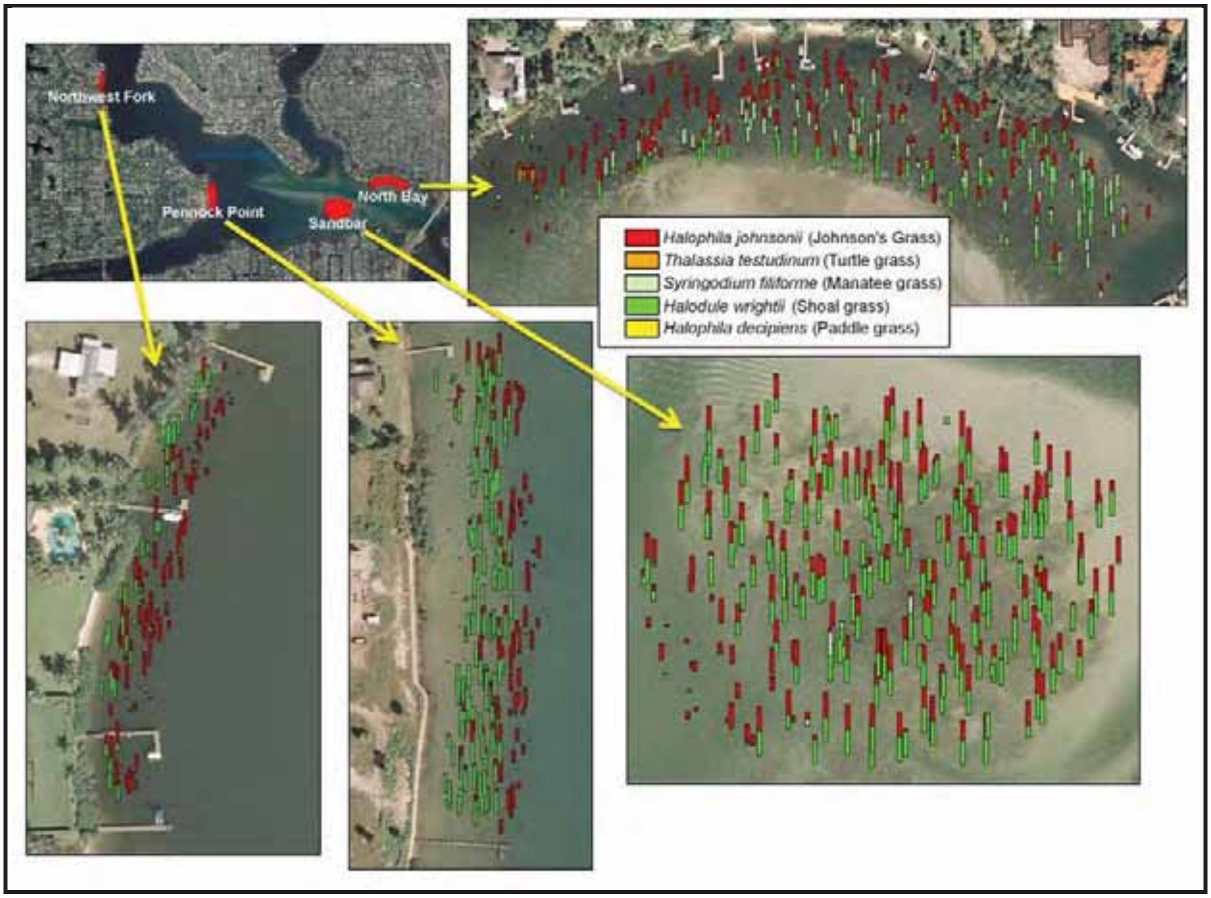


FIGURE 7-84. STACKED BAR GRAPHS SHOW SPECIES SPECIFIC SEAGRASS PERCENT COVER DATA AT EACH SAMPLING SITE ACROSS ENTIRE SAMPLING PERIOD (OCTOBER 2007 - DECEMBER 2008)

The most downstream sampling site in the river, North Bay, has shown the highest species richness (four species) and substantial variation of each species (***Figure 7-82***). *H. johnsonii*, *H. wrightii* and *S. filiforme* were the dominant species observed at this site. In general, *H. johnsonii* occurred in the deeper portions of the nearshore bed, *H. wrightii* was more dominant along the southern edge near the eroding part of the sand bar, and *S. filiforme* was most prevalent in the middle portion of the seagrass bed (***Figure 7-84***). Isolated patches of *T. testudinum* were present at this site, but never comprised more than three percent of the seagrass cover. This site had the most stable salinity regime, which may explain why *S. filiforme* was most abundant compared to the other sites. However, *S. filiforme* percent cover at this site continues to lag far below conditions observed prior to the hurricanes (Ridler et al., 2006). Lastly, *S. filiforme* uniformly dominated the reference site, Hobe Sound, throughout the sampling period. *S. filiforme* covered more than 60 percent of the area during each sampling event (***Figure 7-82***). *H. wrightii* was the second most abundant species at the reference site, and typically covered 20 to 40 percent of the area assessed. *T. testudinum*, another canopy forming species, occurred in isolated patches at this site, though it was locally abundant in these isolated patches. Subcanopy species such as *H. johnsonii* and *H. decipiens* were infrequently encountered and appeared most commonly along the deep edge of the bed (***Figure 7-83***) or in small areas that had been disturbed. It appears that the relatively stable salinity conditions at this site, due to the lack of freshwater inflows, have allowed *S. filiforme*, a canopy forming grass sensitive to salinity fluctuations, to uniformly dominate this site through time.

The variations in species distributions among the sample sites spread across the upstream-downstream gradient are evident. While *H. johnsonii* is the most abundant species, it has experienced substantial fluctuations in coverage. Similarly, *S. filiforme* has experienced dramatic variation in coverage at the downstream sampling sites, particularly following the 2004 storms. For example, *S. filiforme* has failed to recover after the storms at the Sand Bar site, and the North Bay site continues to show less *S. filiforme* since the storms. However, at the reference site in Hobe Sound, *S. filiforme* was not affected by the hurricanes and is the dominant seagrass. In contrast, *H. wrightii*, while sparser than grasses at the other sampling sites, is the most abundant seagrass at the two upstream sampling sites, Northwest Fork and Pennock Point.

7.3.5.3 Summary

The landscape-scale mapping documented seagrasses covering 393 acres of the Loxahatchee River Estuary, the greatest amount of seagrass documented to date. Because sampling occurred near the end of a two-year drought, seagrasses were thriving during the summer 2007 sampling, but the entire gain in seagrass cover was not due to actual increases in seagrass extent. Rather, the new sampling approach employed in the present study generated a more comprehensive and accurate assessment of seagrass cover in the Loxahatchee River Estuary, which included seagrasses being mapped in areas previously thought to be devoid of seagrass. Application of the quadrat sampling approach resulted in detailed, species-specific seagrass maps for the entire estuary, resulting in a more comprehensive understanding of the estuary. Both of these will strengthen the ability to assess and understand changes in seagrass cover across the landscape and through time.

The bimonthly monitoring at the five seagrass beds provides an impressive time series of data that shows interesting fluctuations in the percent cover and occurrence of seagrass species across the upstream-downstream gradient in the river. Two species of seagrass, *H. wrightii* and *H. johnsonii*, dominate the upstream sites. The sites further downstream, Sand Bar and North Bay, have higher species richness and percent cover. The extensive time series shows the profound impact of the 2004 hurricanes at the two downstream monitoring sites, Sand Bar and North Bay, where *S. filiforme* was severely impacted and has yet to recover to pre-storm abundance. The patch-scale monitoring implemented in fall 2007 provides additional detail into the spatial variation within these seagrass beds. Seagrass distributions along depth gradients are now apparent, particularly at upstream sampling sites.

Clearly, freshwater flows influence seagrass in the Loxahatchee River Estuary. Yet the detailed mechanisms that drive seagrass patterns are complex. Salinity, water clarity, temperature and other water and sediment quality parameters influence seagrass success. Because of the complex interactions and durations of these parameters that drive the success or failure of these seagrass, it is presently impossible to accurately predict the cause-and-effect of various water management activities. Ongoing monitoring that provides extensive time series data, coupled with adaptive water management, would provide the best assessment of restoration activities and the influences on the rivers seagrass.

7.3.6 Lake Worth Lagoon

7.3.6.1 Landscape-Scale Mapping

The earliest evaluation of seagrass in Lake Worth Lagoon was compiled from 1940 aerial surveys, which documented 4,271 acres of seagrass. In 1975, a resource inventory found that only 161 acres of seagrass remained in the lagoon. While there is uncertainty about accuracy of the methods used, this indicates a substantial loss of seagrass. The loss of seagrass was hypothesized to be linked to extensive dredging and filling activity and sewage disposal outfalls that directly discharged to the lagoon, which degraded water quality and changed salinity (PBCERM and FDEP, 1998).

In 1990, a natural resource inventory included a detailed in-water survey, which provided the most complete information to date. The survey indicated that there were 2,110 acres of seagrass (PBCERM and Dames and Moore, 1990) or approximately half of the extent of seagrass present in 1940. However, this is a substantial increase of 1,949 acres when compared to results of the 1975 survey.

In 2001, true color aerial photographs were interpreted to determine the seagrass coverage in Lake Worth Lagoon. While this did not include extensive in situ groundtruthing, the goal was to create baseline data for future monitoring. The total 2001 seagrass coverage was determined to be 1,646 acres or approximately 21.2 percent of the lagoon (Avineon, Inc., 2008b). The coverage varied throughout the three segments, Northern, Central and Southern, of the lagoon (***Figure 7-85***). ***Table 7-6*** presents results by segment.

TABLE 7-6. 2001 VERSUS 2007 SEAGRASS ACREAGE AND PERCENT CHANGE IN LAKE WORTH LAGOON

Segment	Acreage: 2001 Seagrass	Acreage: 2007 Seagrass	Acreage: Change	Percent Change
Northern	1,149	1,090	(59)	(5%)
Central	195	205	10	5%
Southern	302	393	91	29%
Totals	**1,646**	**1,688**	**42**	**2.5%**

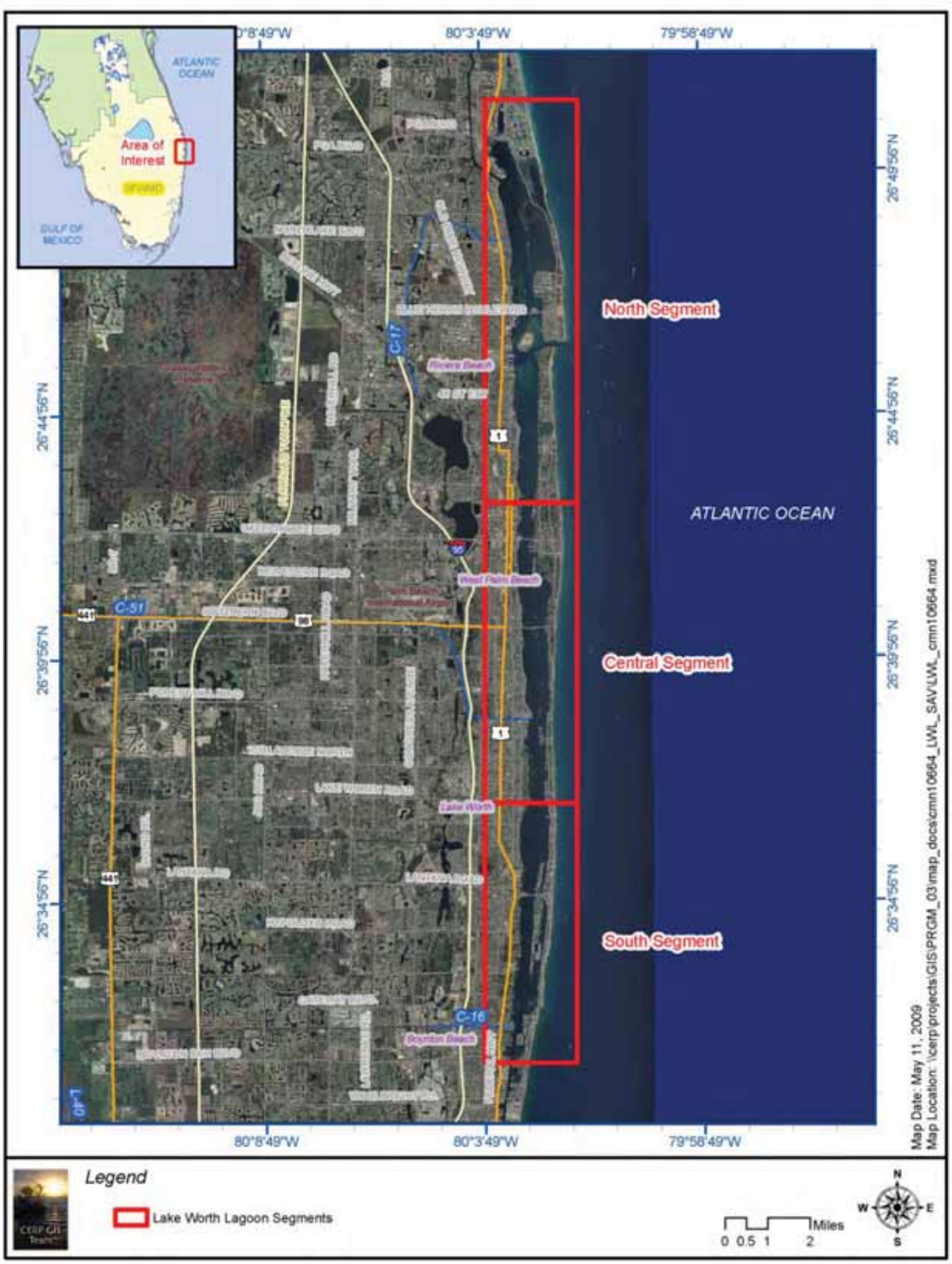


FIGURE 7-85. MAP OF LAKE WORTH LAGOON SHOWING SEGMENTS

Although analysis methods were markedly different between the 1990 and 2001 surveys, seagrass coverage appears to have declined over the 11-year period. The 2001 survey did not include extensive in-water groundtruthing, therefore, sparse seagrass beds, or those located in areas with poor water visibility, were not identified through this mapping (PBCERM, 2008).

The most recent mapping of seagrass was conducted in 2007. This mapping effort implemented the same methodologies that were utilized to map and classify the 2001 seagrass beds. This allowed for the first true large-scale trend comparison between any two years. Results of the 2007 mapping showed that seagrass beds covered at least 1,688 acres, or 21.74 percent, of the lagoon (***Figure 7-86***). This is a 2.5 percent (42 acre) increase over the 2001 calculation of 1,646 acres and is probably not a significant increase given the resolution of the aerials (Avineon, Inc., 2008b). The majority of the increase can be attributed to an increase of patchy seagrass beds throughout the lagoon. More seagrass was identified in the Northern Segment than in the Central or Southern Segments of the lagoon (***Table 7-6***). ***Table 7-7*** presents a summary of Lake Worth Lagoon seagrass coverage to date.

TABLE 7-7. HISTORICAL SEAGRASS COVERAGE IN LAKE WORTH LAGOON

Year	Seagrass (acres)	Percent Change[3]
1940[1]	4,271[2]	-
1975	161	(96%)
1990	2,110	1,210%
2001[4]	1,646	(22%)
2007[4]	1,688	2.5%

Note: [1]Arbitrary date reflects conditions prior to intense urbanization; conditions allow for maximum coverage of seagrass.
[2]Acres is the maximum allowable area of seagrass given pre-War World II conditions.
[3]Due to gross differences in survey methods, these values should only be used to indicate an order of magnitude change.
[4]2001 and 2007 utilized the same methodology to calculate acres.

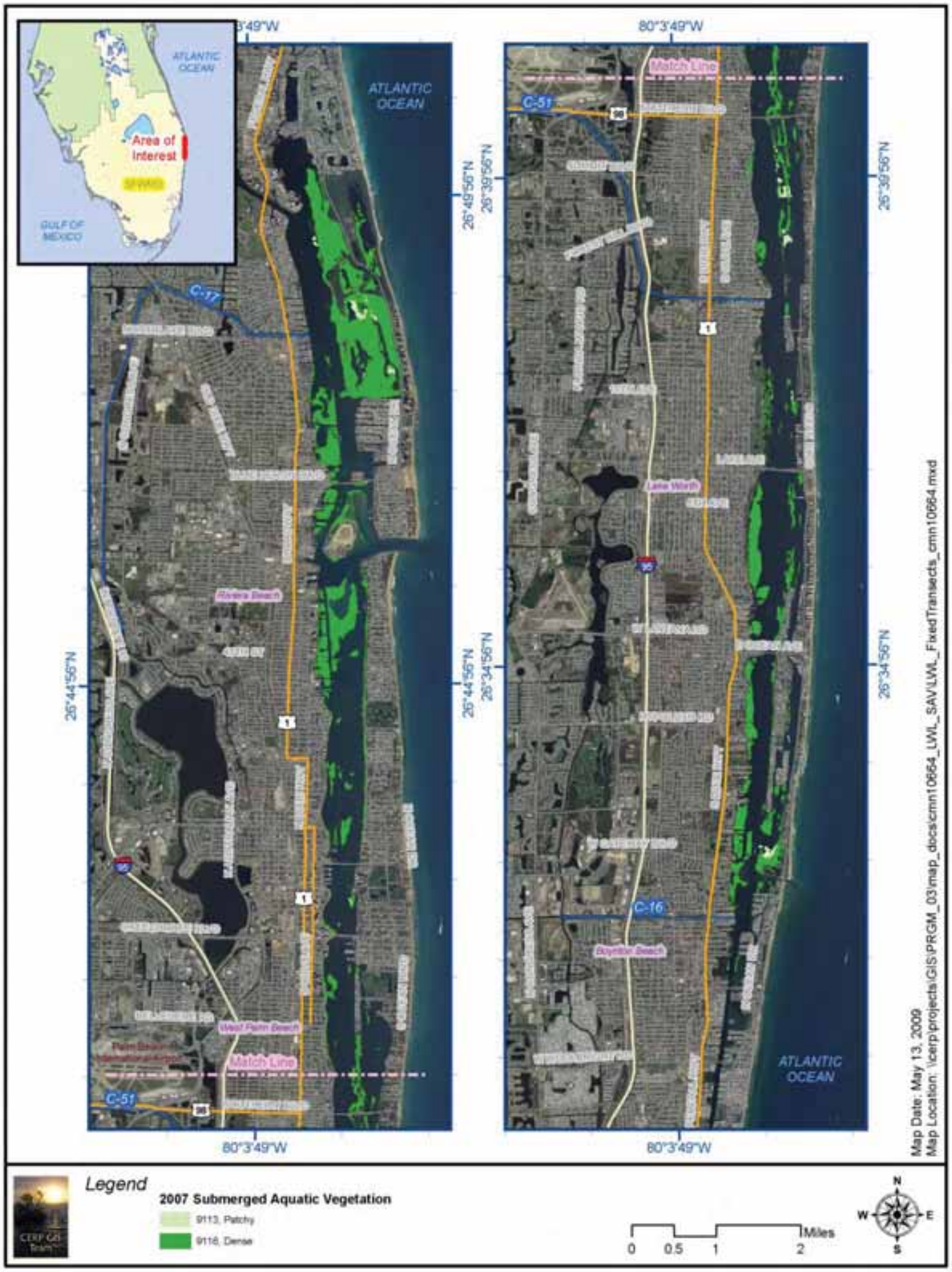


FIGURE 7-86. SUBMERGED AQUATIC VEGETATION AREAL EXTENT AND DISTRIBUTION IN LAKE WORTH LAGOON IN 2007

7.3.6.2 Patch-Scale Fixed Transects

In 2000, PBCERM initiated a long-term seagrass monitoring project that included annual SAV assessment using nine fixed transects throughout Lake Worth Lagoon (***Figure 7-87***). As water quality improves, seagrasses are expected to expand to greater depths and/or increase in density and diversity. The first five years of surveys showed fluctuation in seagrass cover with no obvious pattern of increase or decrease until the 2004 hurricanes. The surveys conducted in June 2005 and June 2006 showed a major decrease in seagrass cover in most areas of the lagoon (Applied Technology and Management, Inc., 2008). This loss is most likely due to increased turbidity caused by runoff from the hurricanes, discharges from Lake Okeechobee, and burial and scour from wave action. Areas suffering the least impact were shallow sites and sites closer to inlets where water quality was least impacted. The 2007 survey reported record highs in terms of total number of cells in which seagrass was observed and percent coverage within the monitoring stations. It also documented the expansion of beds into deeper water than in 2005 and 2006.

The results of the 2008 survey did not exceed the record highs of the 2007 survey; however, the 2008 survey was still the third highest year on record for total number of cells containing seagrass and percent coverage (Applied Technology Management, Inc., 2009). The most impressive results of the 2008 survey was that five of the nine deep water stations recorded highs for number of cells with seagrass presence, and at six of the nine transects, the waterward edge of seagrass was found to have expanded into deeper water than had been observed in 2007 or any previous year. Seagrass was even documented at two of the deep water stations that had never recorded seagrass during the nine years of the previous monitoring.

The cumulative total number of cells with seagrass at all stations of the transects for each monitoring year were compiled and graphed in a smoothed-line format together with the cumulative yearly flow from the C-17, C-51 and C-16 canals (***Figure 7-88***). The graph provides a lagoonwide perspective that shows the general inverse relationship between canal discharge and seagrass presence. This relationship is most evident in the years from 2003 through 2008, which includes the lagoonwide decreases after the 2004 hurricanes.

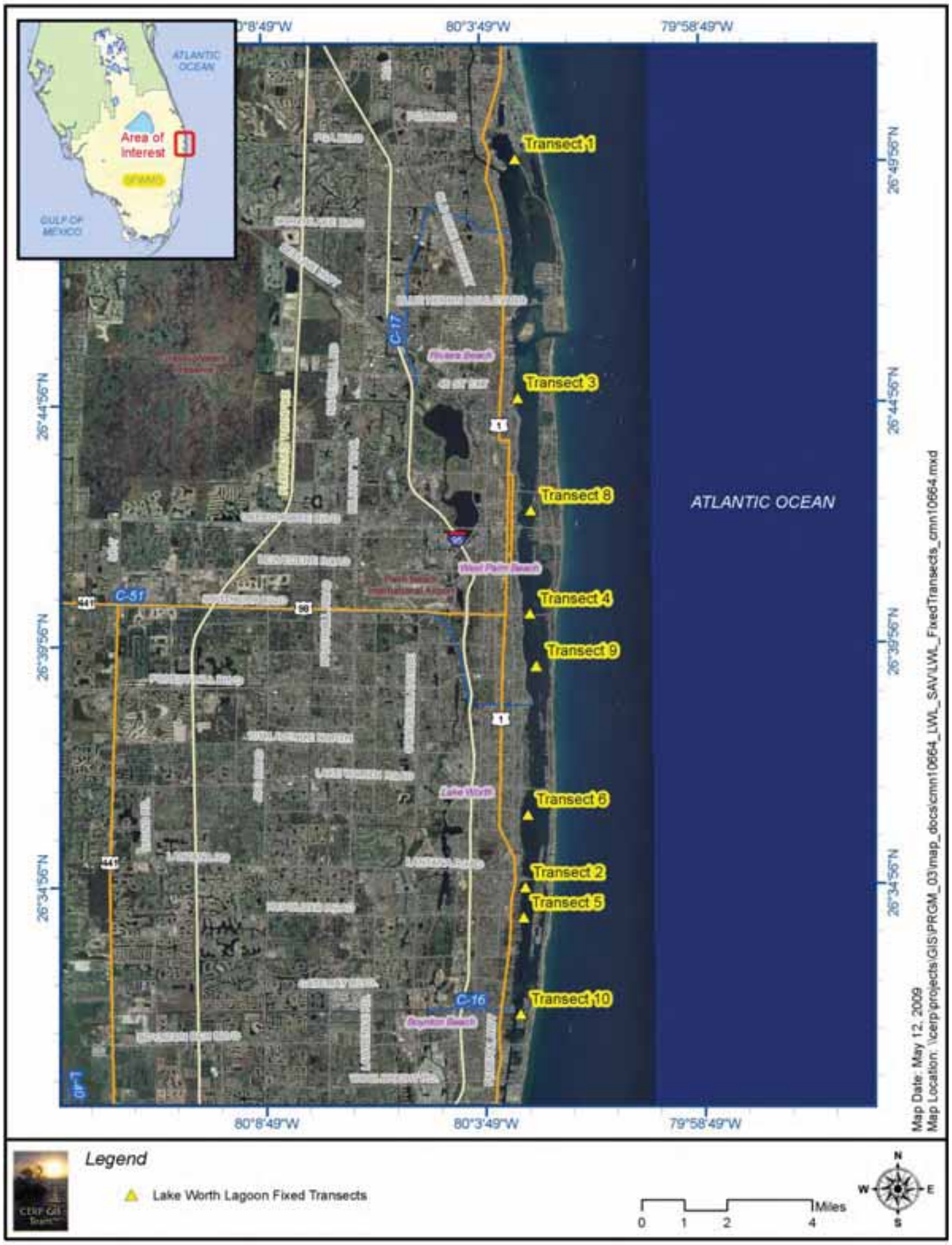


FIGURE 7-87. LOCATION MAP OF LAKE WORTH LAGOON FIXED TRANSECT

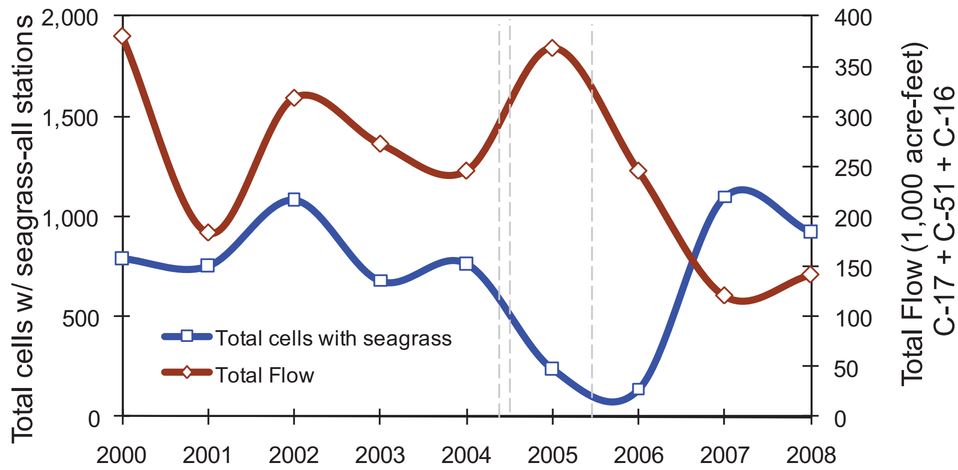


FIGURE 7-88. COMBINED SEAGRASS ABUNDANCE AT ALL LAKE WORTH LAGOON TRANSECTS, 2003 TO 2008, COMPARED WITH TOTAL CANAL FLOW DATA

Seagrass coverage varies throughout the lagoon, with more seagrass found in the Northern Segment than in the Central or Southern Segments. Based on the fixed transect data, *H. decipiens* is usually present in the highest number of stations. In 2008, it was observed in 67 percent of the data sets. *H. johnsonii* is usually present in the second highest number of stations and in 2008 it was observed in 56 percent of the data sets. *H. wrightii* is usually only present in a few of the monitoring stations. It was observed in four percent of the data sets in 2008. While *T. testudinum* and *S. filiforme* are found in the lagoon, they were not observed in any of the monitoring stations associated with the fixed transects.

7.3.6.3 Summary

Preliminary SAV targets have been set for the lagoon based on depths since light availability limits depths at which seagrasses occur. Seagrass distribution by depth was developed using coverage from 2001 aerial mapping and bathymetry data from 2003 (Morgan and Eklund, Inc., 2003). Setting a target depth assumes that water quality would improve allowing increased light attenuation through the water column. These targets would be refined as improved substrate characterization data, as well as photosynthetically active radiation (PAR) data, are collected and analyzed. Using a very conservative approach of increasing the average seagrass depth by one foot in the Northern Segment and 0.6 feet in the Central and Southern Segments, leads to the target depths as presented in ***Table 7-8***. ***Table 7-9*** presents the seagrass acreage and potential acreages in Lake Worth Lagoon.

TABLE 7-8. AVERAGE SEAGRASS DEPTH AND POTENTIAL TARGET DEPTH BY SEGMENT

Segment	Weighted Average Seagrass Depth (feet NGVD)	Target Depth (feet NGVD)
Northern	-5.0	-6.0
Central	-4.4	-5.0
Southern	-3.4	-4.0

TABLE 7-9. SEAGRASS ACREAGE AND POTENTIAL SEAGRASS IN LAKE WORTH LAGOON

Segment	Acreage				Percent Increase
	2001 Seagrass	Potential Seagrass by Depth	Potential Seagrass by Depth and Substrate	Potential Seagrass Target	
Northern	1,149	321	195	1,344	17%
Central	195	537	207	402	106%
Southern	302	236	66	368	22%
Totals	1,646	1,094	468	2,114	28%

The majority of the 1,149 acres of seagrass in the Northern Segment is present at depths of -2.0 to -6.0 feet NGVD(Braun, 2006). ***Figure 7-89a*** presents the distribution of seagrass by depth in this segment.

The Central Segment is most severely impacted with respect to turbidity and presence of muck. As light attenuation decreases, seagrasses can only grow in shallow depths (Bortone, 2000). This segment has only 195 acres of seagrass and the majority of seagrass is found at depths between -3.0 to -5.0 feet NGVD and the average depth is -4.4 feet NGVD (Braun, 2006). ***Figure 7-89b*** presents the distribution of existing seagrass in this segment.

The Southern Segment of the lagoon is also stressed due to poor water quality and is typified by seagrass growing only in shallow depths where light requirements can be met. Based on the 2001 data, this segment had only 302 acres and the majority of seagrass is found at depths between -3.0 to -5.0 feet NGVD (***Figure 7-89c***). The average depth of seagrass is -3.4 feet NGVD (Braun, 2006).

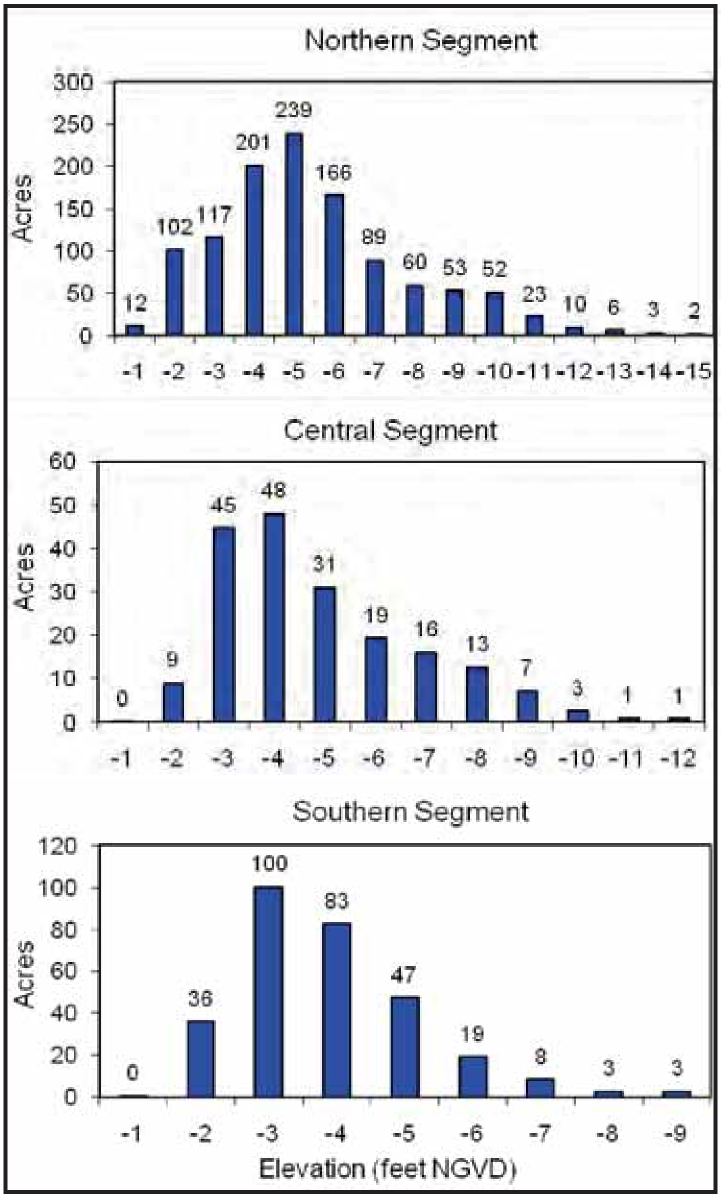


FIGURE 7-89. 2001 SEAGRASS DISTRIBUTION BY DEPTH

Note: A Northern segment
B Central segment
C Southern Segment

7.4 OYSTER HYPOTHESIS CLUSTER

7.4.1 Introduction and Background

The eastern oyster (*Crassostrea virginica*) 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 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 NOAA 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 AM. The oyster has been used by 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. The interim goal can be found at www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_1-1_neoysters.pdf.

CERP implementation and other restoration projects, should restore more natural freshwater inflows by retention in reservoirs and STAs, 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 would promote the re-establishment of healthy oyster beds.

Figure 7-90 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 MAP, Part 2 2006 Assessment Strategy for the MAP (RECOVER, 2006) and the 2007 SSR (RECOVER, 2007b). These documents can be accessed from www.evergladesplan.org/pm/recover/assess_team.aspx. In ***Figure 7-90*** boxes surrounded with dashed lines are not currently being measured. However, depending on the need and the model output, these factors may be included in future monitoring. Predictions of oyster reef development following restoration implementation are made by using a HSI model described in Volety et al. (2005).

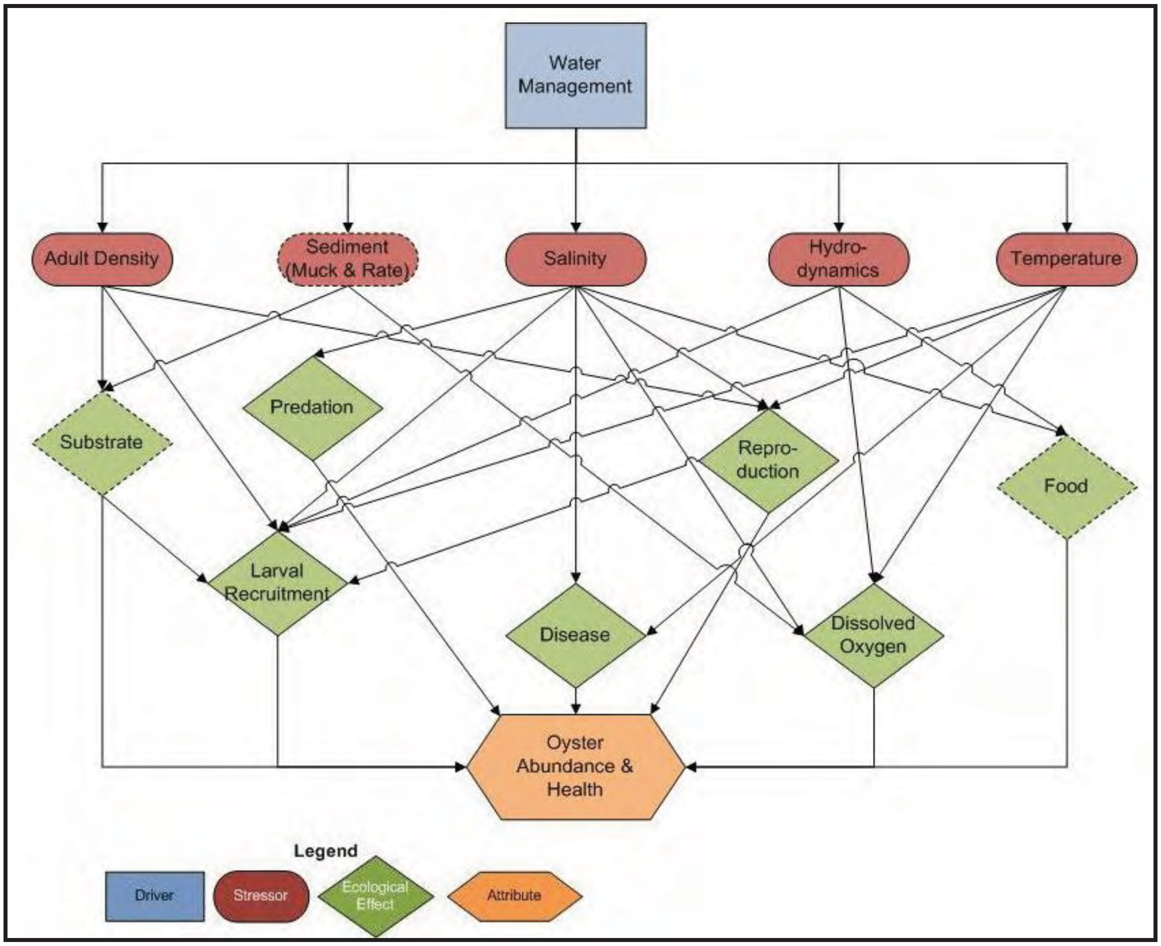


FIGURE 7-90. CONCEPTUAL MODEL FOR INFLUENCE OYSTERS IN THE NORTHERN ESTUARIES

7.4.2 Study Area

Oysters are monitored in the Caloosahatchee River Estuary on the west coast of Florida (***Figure 7-91***) and in the St. Lucie Estuary, Loxahatchee River Estuary and Lake Worth Lagoon on the east coast of Florida (***Figure 7-92***). Detailed descriptions of these individual water bodies can be found in the 2007 SSR (RECOVER, 2007b) at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

7.4.3 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) and, on the east coast only, *Haplosporidium nelsoni* (MSX). Monthly water quality sampling is conducted in conjunction with field sampling at each study site. For details on monitoring techniques please see the CERP MAP: Part 1 Monitoring and Supporting Research (RECOVER, 2004) at www.evergladesplan.org/ pm/recover/recover_map_2004.aspx and the 2007 SSR (RECOVER, 2007b) at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx.

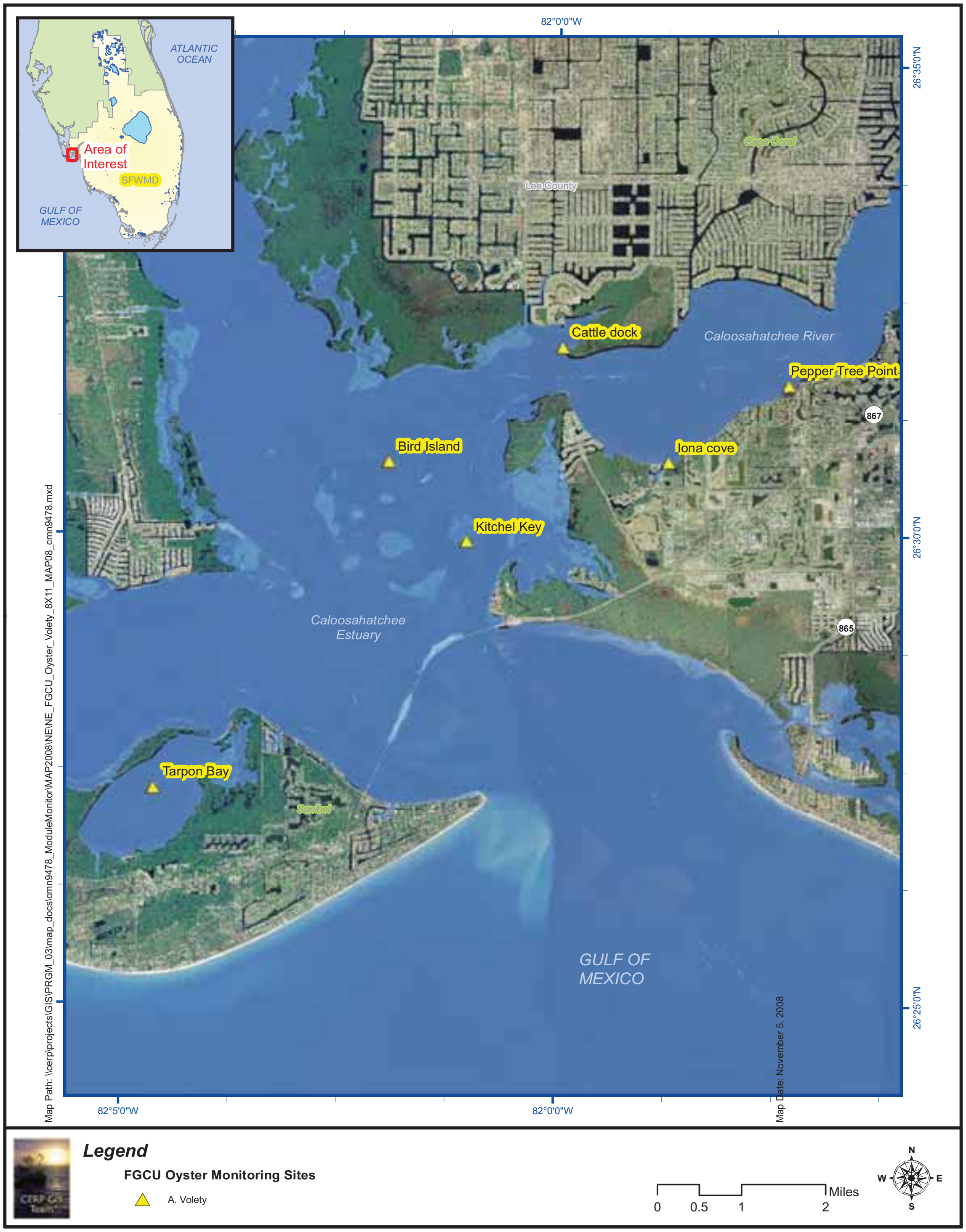


FIGURE 7-91. OYSTER MONITORING SITES IN CALOOSAHATCHEE RIVER ESTUARY

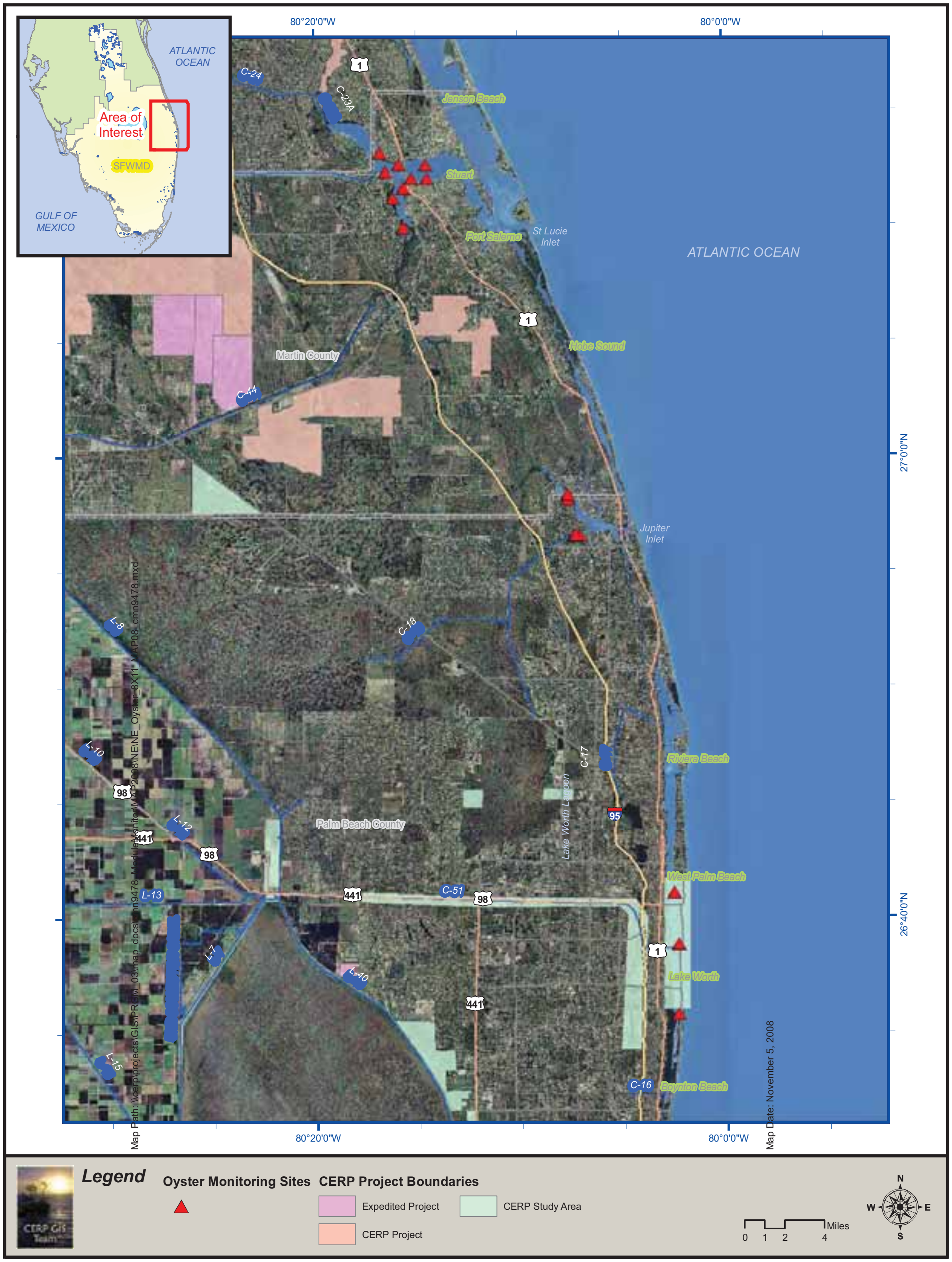


FIGURE 7-92. OYSTER MONITORING SITES IN ST. LUCIE ESTUARY, LOXAHATCHEE RIVER ESTUARY AND LAKE WORTH LAGOON

7.4.4 Results

7.4.4.1 Adult Density

Density of living oysters appears to be higher among the west coast locations compared to the east coast locations (***Figure 7-93a***). Results from all the estuaries suggest that minor differences in living oyster density are apparent between dry (spring) and wet (fall) seasons **(*Figure 7-93b* and *c*)**. The higher living densities are a result of high recruitment of oyster spat (juveniles) at the end of the spawning season (wet season). However, during subsequent months, juveniles and new recruits encounter mortality due to a combination of predation and salinity. Intra-annual variation in oyster abundance within each of the east coast study estuaries is relatively minor compared to among site and among year variations. In most cases, oyster abundance is higher at the end of the dry season (***Figure 7-93b***) than at the end of the wet season (***Figure 7-93c***) due to the abundance of first year animals during the fall survey period. It is expected that many of those small oysters would not survive to achieve maturity and contribute to future populations, so the dry season survey provides the best estimate of population status. Oyster abundance within each site, determined as an average of all sampling data, illustrates the large variation in oyster population density among study sites (***Figure 7-93a***). Of all sites, the St. Lucie Estuary has the largest variation in adult density, with mortality events observed in 2005 and 2008, concurrent with low salinity following storm events.

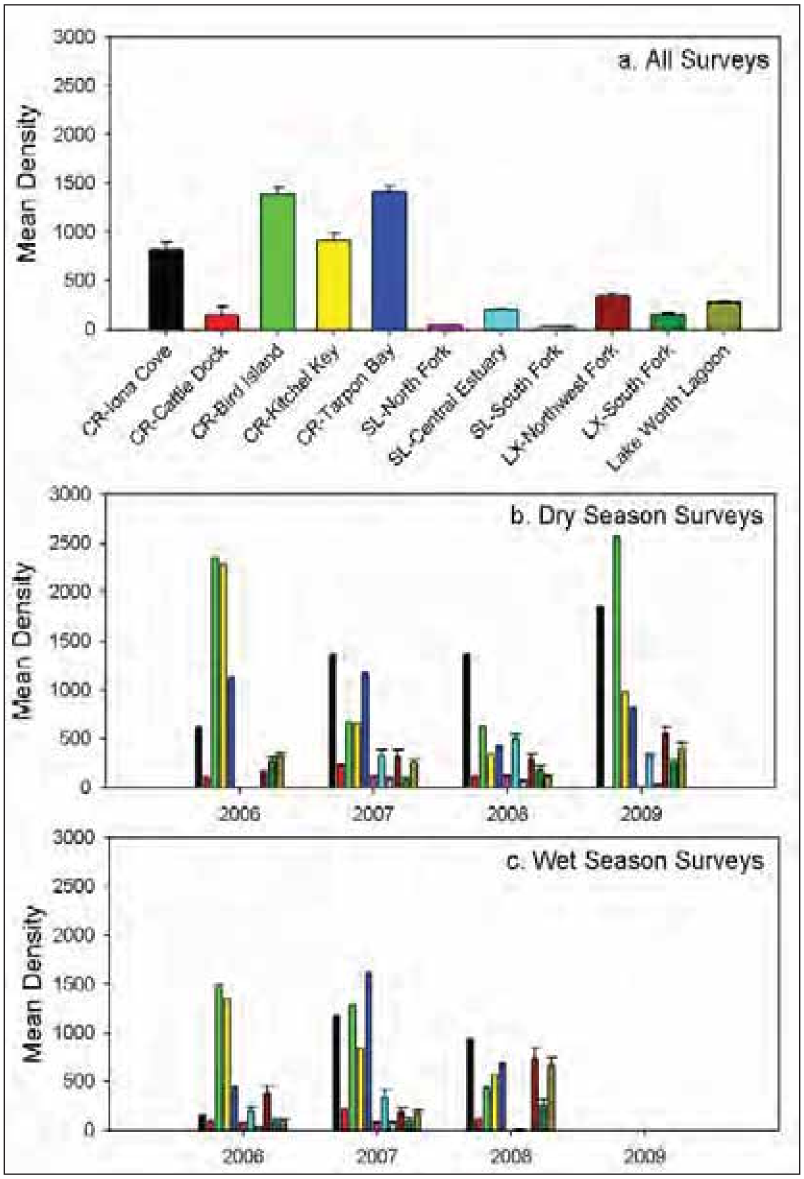


FIGURE 7-93. MEAN (+ SE) ANNUAL ADULT OYSTER DENSITIES (M2) A) AMONG STUDY SITES, B) FOR DRY SEASON SURVEYS AND C) WET SEASON SURVEYS (CR SITES ARE WITHIN CALOOSAHATCHEE RIVER ESTUARY. SL SITES ARE WITH ST. LUCIE ESTUARY, LX SITES ARE WITH LOXAHATCHEE RIVER ESTUARY)

7.4.4.2 Reproduction, Reproductive Potential and Recruitment

Overall mean gonadal index values were relatively constant between sampling locations (***Figure 7-94a***). Oysters in all estuaries showed peak reproductive activity, as evidenced by higher gonadal index values, between April and September (***Figure 7-94b***) coinciding with peak recruitment. Mean gonadal index at most sites begins to drop rapidly beginning in October. Observed bimodal peaks in reproductive index at some sites appear to coincide with minor spring recruitment peaks prior to the major summer peak. Gonadal index values were relatively stable between sampling years (***Figure 7-94c***), although it should be cautioned that the data for 2008 are incomplete. The lower values recorded for oysters collected from northern and southern St. Lucie Estuary sites in 2006 represent young immature animals that had not yet reached reproductive maturity at that time.

Oyster spat recruitment varied between sampling locations, sampling months and sampling years in all the estuaries. Recruitment was higher in the Caloosahatchee River Estuary sites than the east coast estuaries (***Figure 7-95a***) and in higher salinity locations and sites. Recruitment rate within each site reflects adult abundance patterns; sites with relatively high adult densities receive similarly high numbers of recruits. This pattern may reflect the greater substrate availability provided by healthy oyster reefs coupled with the physical flushing effect of freshwater flows into the estuaries during the summer and fall months when when oysters are spawning. However, it is not clear whether recruitment patterns are due to water quality conditions that support both adults and recruits or if it reflects a direct relationship between spawner abundance and availability of recruits. This distinction is important and can be better evaluated upon acquisition of a lengthier data set. Spat recruitment in all of the estuaries occurred between March and November, with peak recruitment occurring between June and October (***Figure 7-95b***). Mean annual spat recruitment varied between years, possibly as a result of previous environmental history (e.g., freshwater release regime) with average values ranging from zero to 11.14 spat per shell per month at most sites (***Figure 7-95c***). Note that 2009 mean recruitment rates are derived from incomplete data that do not include the primary months of recruitment, so the reported values are biased downward.

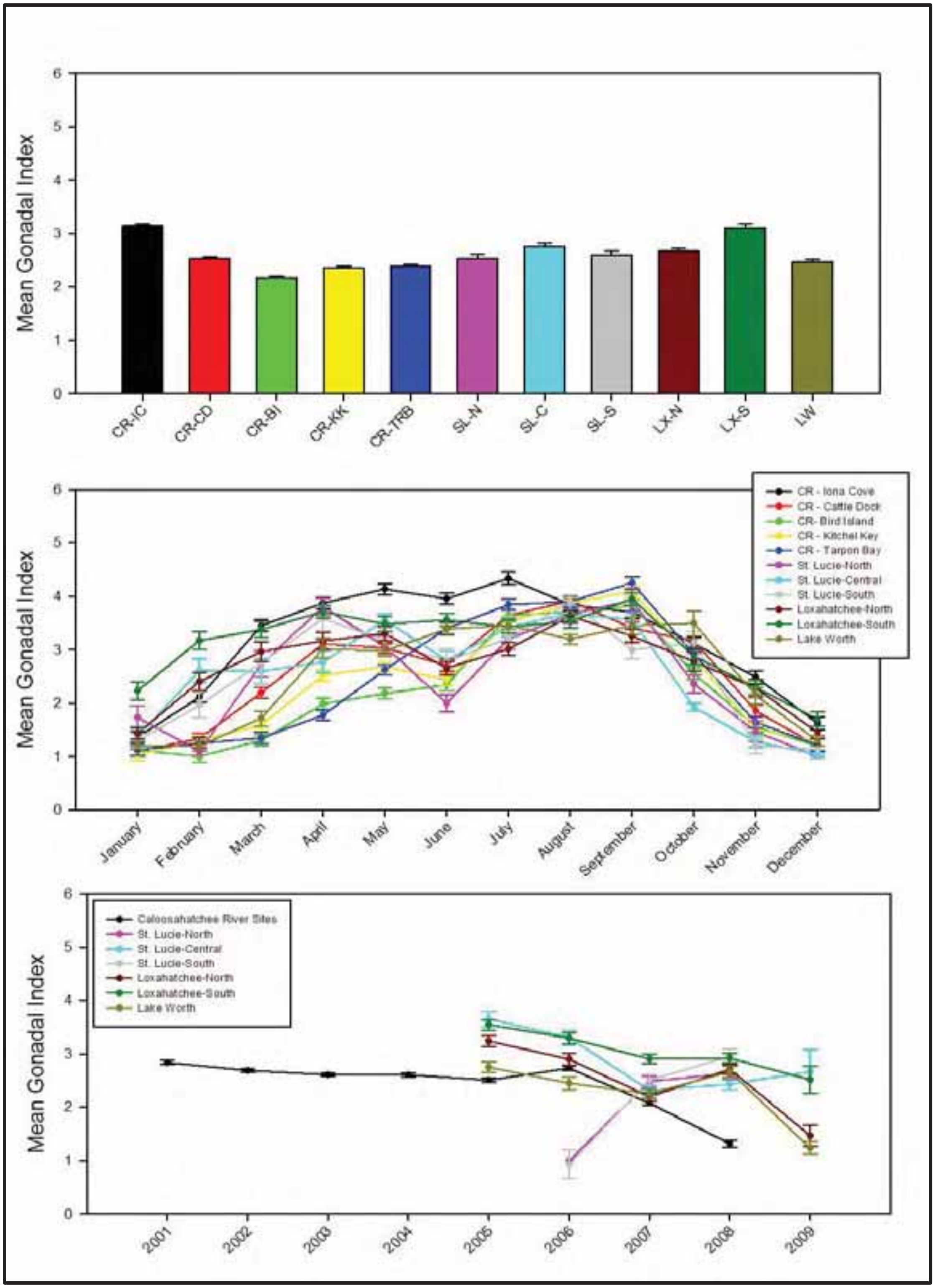


FIGURE 7-94. MEAN (±SE) GONADAL INDEX FOR OYSTERS

Note a) among study sites
b) for each month averaged over all years
c) within each year averaged over all months

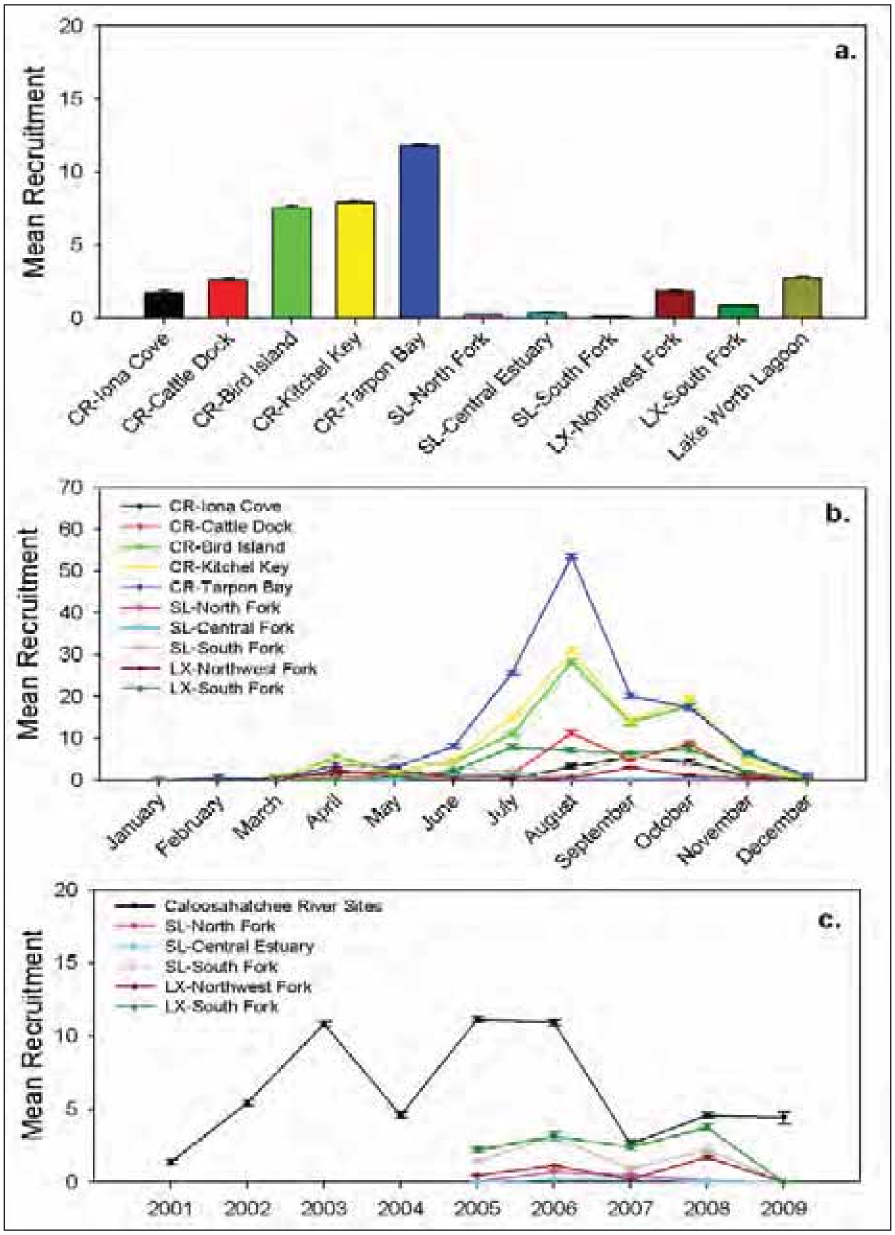


FIGURE 7-95. MEAN (±SE) RECRUITMENT FOR OYSTERS

Note a) among study sites
b) for each month averaged over all years
c) within each year averaged over all months

7.4.4.3 Condition Index

Condition index, a ratio of meat weight to shell weight, varied significantly between sampling locations, sampling months and sampling years in all the estuaries and sampling locations (***Figure 7-96***). For all sites, the overall mean condition index ranged between 2.34 and 3.54 (***Figure 7-96a***). The decrease in condition index in oysters during March through October at most sites is probably a result of spawning activity, where shedding of gametes results in loss of tissue and hence a decrease in condition index. Oysters start to accumulate energy reserves and tissue mass during the cooler winter months, and thus increase of condition index in the cooler winter months (***Figure 7-96b***). Variation in condition index between years is a result of environmental history (i.e. salinity, flow changes, hurricanes) during the year (***Figure 7-96c***).

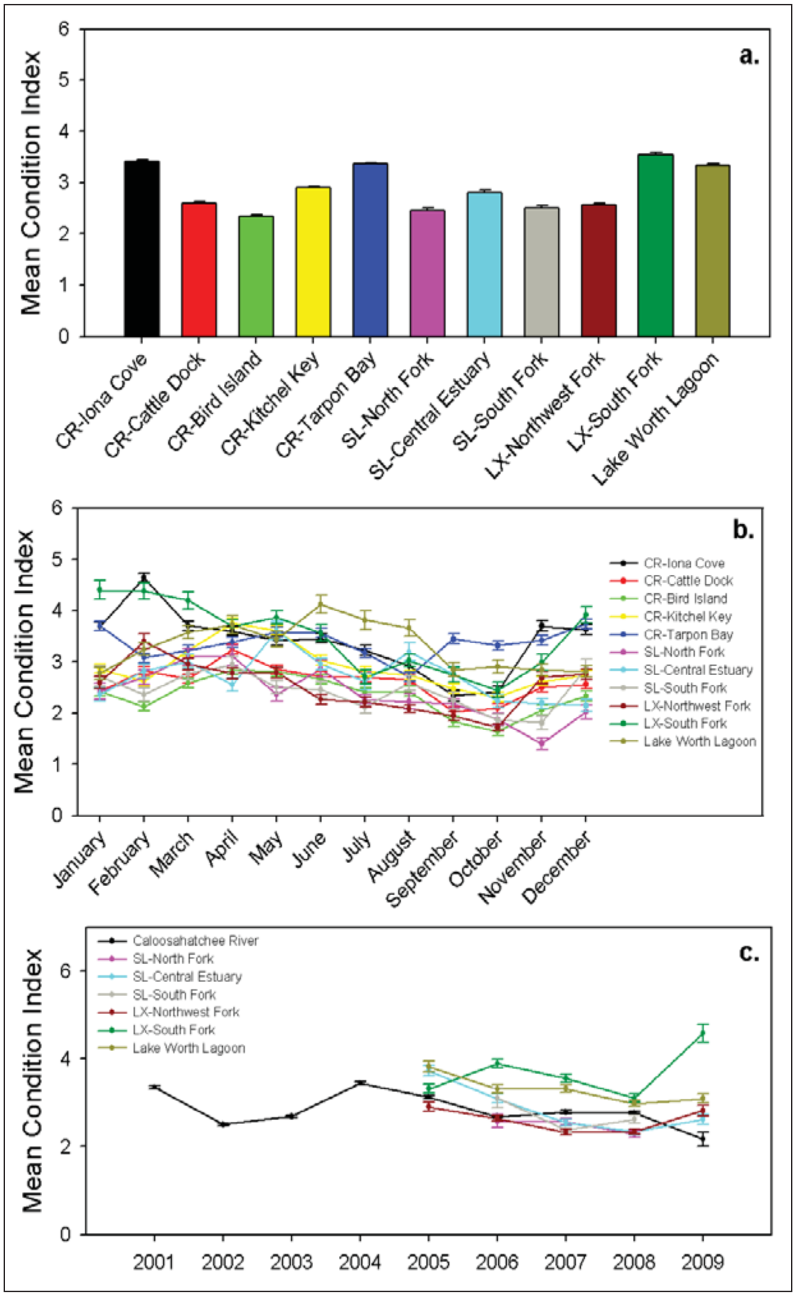


FIGURE 7-96. MEAN (± SE) CONDITION INDEX FOR OYSTERS

Note: a) among study sites
b) for each month averaged over all years
c) within each year averaged over all months

7.4.4.4 Juvenile Growth

In the Caloosahatchee River Estuary, results clearly indicated that juvenile oyster growth responds directly to freshwater releases and resultant salinity prevailing at each location. A representative growth profile of juvenile oysters is presented for elucidating general trends (***Figure 7-97a*** and ***Figure 7-97b***). Typically, given rainfall, resulting watershed runoff and freshwater releases from Lake Okeechobee, low salinities (< 10 psu) prevail in the upper portion of the estuary during the late summer and fall months. When juvenile oysters were deployed after freshwater releases/watershed flows, oysters at the upstream portions, where estuarine salinities prevail, showed the highest growth rate compared to other downstream locations. Most downstream locations, despite receiving the highest spat recruitment, showed poor survival and growth due to higher salinities (> 35 psu) and diseases and predators that dominate those high salinity regions.

At the east coast sites, oysters grew most consistently and most rapidly in the south fork of the Loxahatchee River Estuary and most slowly in Lake Worth Lagoon (***Figure 7-97c***). Juvenile growth monitoring was interrupted at the St. Lucie Estuary sites in the late summer and early fall 2008 due to a drastic decrease in estuarine salinities resulting from a tropical storm event. However, salinities rebounded quickly and juvenile oysters began recruiting to the cages again in October and November. These new recruits exhibited a rapid growth rate, reaching a mean shell height of approximately 30 millimeter (mm) in just four months. This is nearly the same size achieved by oysters growing in both the northern Loxahatchee River Estuary site and Lake Worth Lagoon over 11 months. Oysters growing in the southern Loxahatchee River Estuary site reached a mean shell height of nearly 42 millimeter during that same time period.

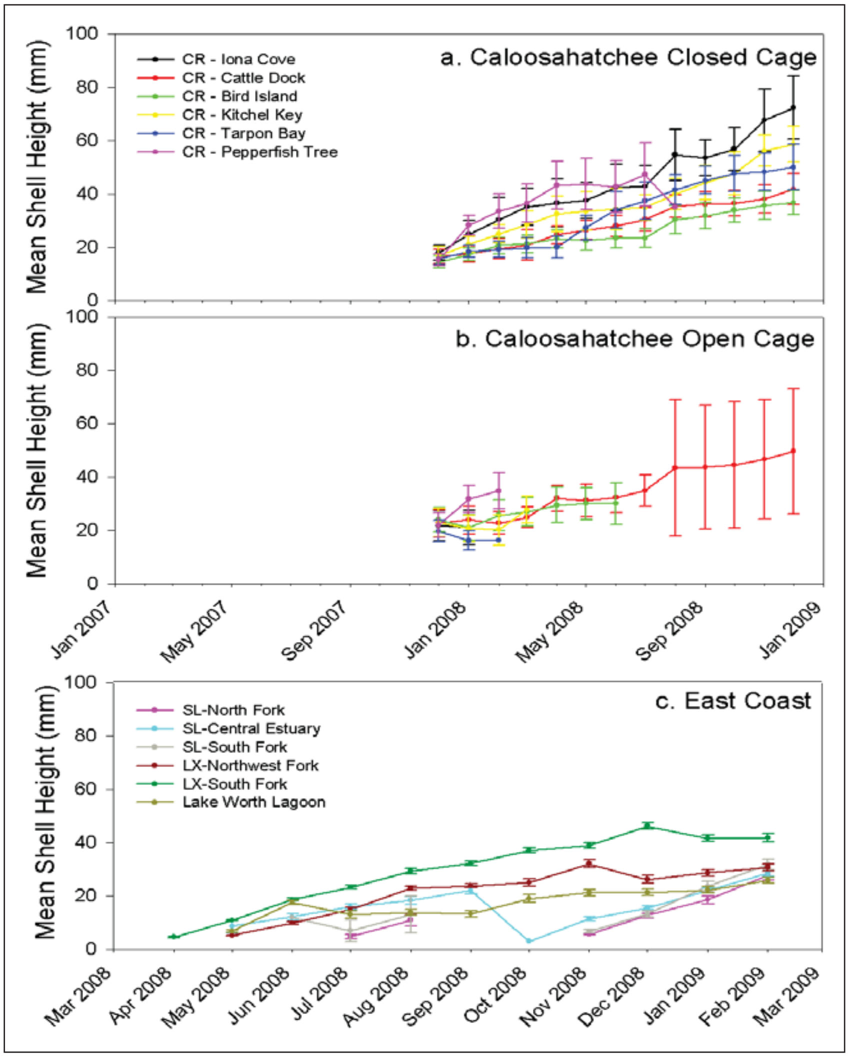


FIGURE 7-97. MEAN (±SE) SHELL HEIGHT OF JUVENILE OYSTERS DEPLOYED WITHIN A) OPEN CAGES AND B) CLOSED CAGES ALONG A SALINITY GRADIENT (DOWNSTREAM TO UPSTREAM) IN THE CALOOSAHATCHEE RIVER ESTUARY AND C) AT VARIOUS SAMPLING LOCATIONS ON THE EAST COAST DURING 2008

7.4.4.4.1 Disease Monitoring

Oysters were collected monthly for the analysis of gonadal condition and for the prevalence and intensity of the oyster diseases *Perkinsus marinus*, referred to commonly as "dermo". Disease intensity, on a scale of 0 to 3 (***Figure 7-98***), and prevalence, as a percent of infected oysters (***Figure 7-99***), of dermo varied significantly between sampling sites, sampling months and sampling years in all estuaries. Overall, mean disease intensity and prevalence were higher in higher salinity areas and also higher on the west coast of Florida sites compared to the east coast locations (***Figure 7-98a*** and ***Figure 7-99a***). Higher salinities and temperatures favor parasite development and proliferation. Given the interaction of temperature and salinity cycles in south Florida estuaries, no distinct seasonal patterns of disease intensity and prevalence were discernable (***Figure 7-98b*** and ***Figure 7-99b***). Although mean prevalence at all the sites ranged between 0 and 58 percent, mean intensity at all sites ranged between 0 and 1.08, an overall light infection. The antagonistic effects of high temperatures and low salinities in warmer months and of high salinities and lower temperatures during winter months keep disease intensity and prevalence of dermo in oysters at low levels in the Florida estuaries (Volety et al., 2003). Dermo intensity and prevalence appear to be increasing over the past five years in the high salinity sites (***Figure 7-98c*** and ***Figure 7-99c***), however it should be cautioned that the overall level of infection remains relatively low (< 1.5).

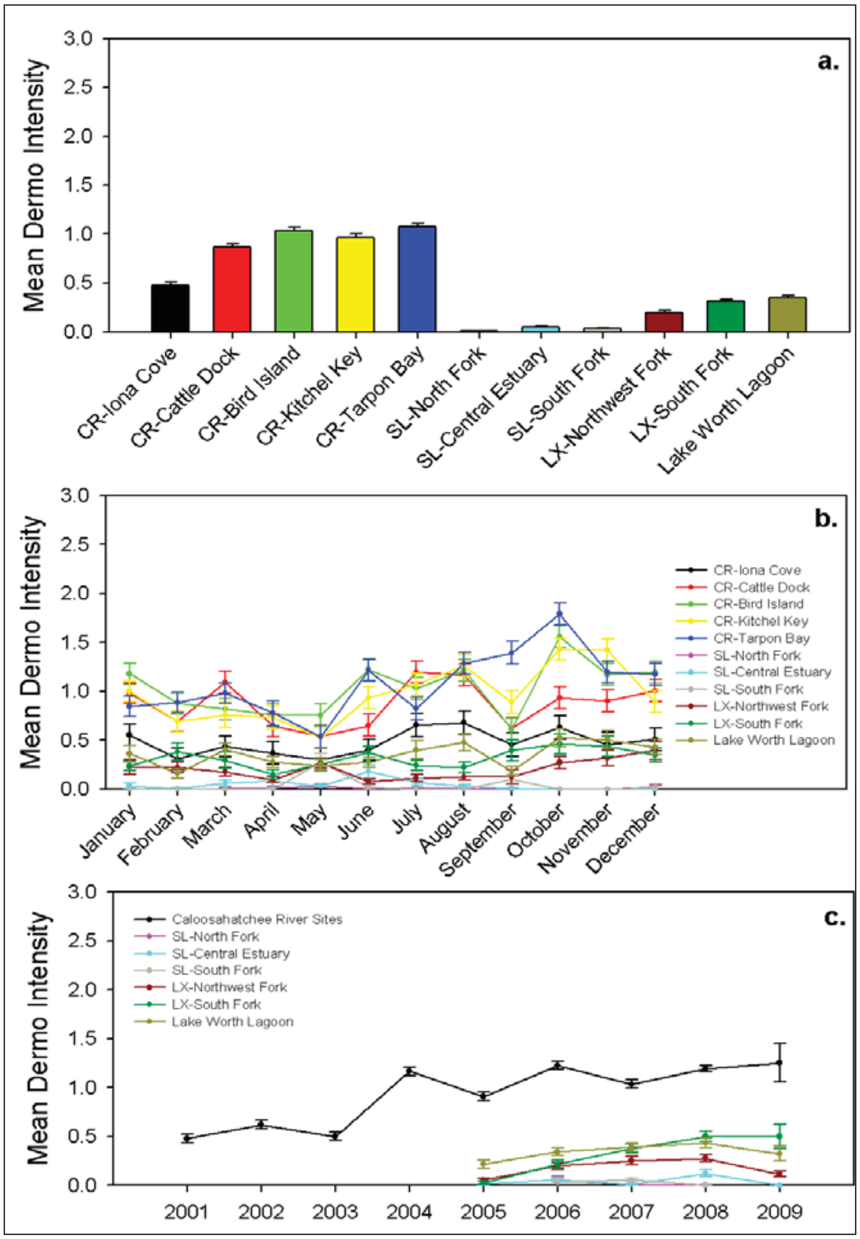


FIGURE 7-98. MEAN (± SE) INTENSITY OF DERMO FOR OYSTERS SAMPLED FROM NORTHERN ESTUARIES

Note a) among study sites
b) for each month averaged over all years c) within each year averaged over all months

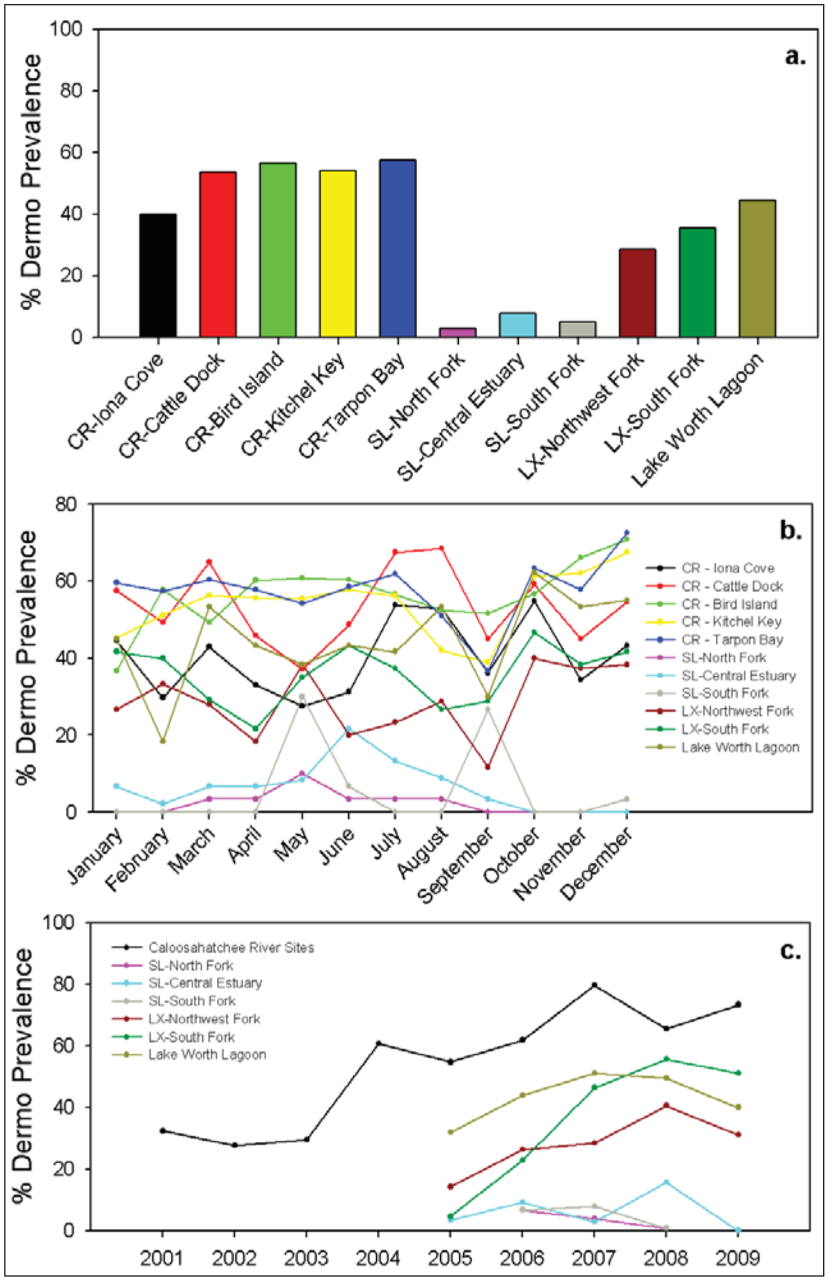


FIGURE 7-99. MEAN (± SE) PREVALENCE OF DERMO FOR OYSTERS SAMPLED FROM NORTHERN ESTUARIES

Note: a) among study sites
b) for each month averaged over all years
c) within each year averaged over all months

7.4.4.4.2 Mapping

A baseline for oysters in each estuary is being established by mapping the existing distribution of reefs and the mean density of living oysters on each bed. Historical distributions, where available, are being used to assist in identifying areas that may have suitable habitat conditions for re-establishment, given predicted changes in the salinity regime. At five-year intervals during restoration implementation, a map of oyster reefs within the various study sites including size distribution, density of living oysters, and height of the oyster reef, when possible, will be prepared. See the 2007 SSR (RECOVER, 2007) for more details.

7.4.5 Overview Discussion

A significant relationship exists between freshwater inflows and salinities at various points in the Caloosahatchee River Estuary. Mean monthly flows between 450 and 2800 with limited or no flows in excess of 4500 cfs from the S-79 structure would result in a beneficial salinity regime. Disease prevalence was lower at upstream locations and increased with distance downstream, suggesting that higher salinities result in increased disease incidence. Limited freshwater releases for durations of less than two weeks would result in lower prevalence and intensity of disease in oysters and higher oyster survival. Oysters in the Caloosahatchee River Estuary 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. Based on nine years of data, regression analyses indicate that limiting freshwater releases to less than 3,500 cfs during these months would curb flushing of oyster larvae to downstream locations and create favorable conditons for spat recruitment and survival. Low disease incidence, high condition index, sufficient spat recruitment and high growth rate at the upstream locations suggest that with the provision of suitable substrate and limiting freshwater flows during the spawning season, oyster reefs would survive and grow in the upstream locations. With restoration implementation and subsequent reduction in freshwater flows, it is anticipated that oyster reef development would 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, need to be studied. The existing sampling design and sampling frequency can adequately assess the direction and magnitude of change in oyster metrics. While water quality (i.e. temperature, salinity and dissolved oxygen) is being measured during each event, more frequent sampling is required to capture episodic events. An attempt is being made to assess the predation pressure in all estuaries, but this parameter had not previously been assessed in the Caloosahatchee River Estuary. The limited dataset over the recent drought shows that predation may be a substantial stressor. 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.

Oyster reefs occupying the various estuaries in south Florida are not isolated entities but are instead linked to one another via exchange of larvae. Each reef is linked to other reefs to a greater or lesser degree depending upon distance, hydrodynamics and environmental factors that promote or defeat larval survival and growth. Previous studies (Hare and Avise, 1996) clearly indicate that, based upon variations in genetic structure among south Florida oyster populations, larval exchange is spatially structured. It is probable that temporal variation in larval exchange characterizes these populations. An understanding of larval exchange is a necessary precursor to the proper management of oyster reefs in Florida because that information would reveal those oyster reefs that act as larval sources and are, therefore, fundamental to the long-term survival of oyster populations. At present, the needed genetic and hydrodynamic information to define these linkages is not available.

7.5 BENTHIC MACROINVERTEBRATE HYPOTHESIS CLUSTER

7.5.1 Introduction and Background

Benthic infauna are secondary producers that occupy a crucial trophic component of the estuarine food web. Benthic infauna provide linkages between primary producers and higher level consumers and represent a large standing stock of carbon in healthy estuaries (Diaz and Schaffner, 1990). Some benthos, such as crabs, shrimp and flatfish, are commercially harvested.

Furthermore, unlike shifting populations of plankton or many pelagic fish species, adults of most benthic invertebrate species are either sessile or of limited mobility. Thus, they are good indicators of locally-induced environmental changes (Boyd, 2002). They respond differentially to varying environmental conditions, and after settlement, most benthos remain within a relatively constrained areas for their entire adult lives (Gray ,1979).

Benthic taxa vary in their responses to changes in water quality (Dauer, 1993), including salinity (***Figure 7-100***). Some are relatively tolerant of organic enrichment and low dissolved oxygen while others are excluded under low dissolved oxygen conditions. Increased nutrient inputs can strongly affect abundances of some species, while not affecting others. By examining shifts in the benthic community over time, an understanding of the major environmental processes affecting the local biota can be gained (Warwick, 1986; Borja et al., 2000).

The post-drainage watershed in the St. Lucie Estuary and Southern Indian River Lagoon system encompasses a much larger surface area than the pre-drainage system. Additionally, an eightfold increase in the quantity of stormwater is now delivered to the coast, accompanied with an increase in nutrient load (Graves et al., 2004). The impacts of these watershed alterations on the estuary have been dramatic. Excess nutrient loading has facilitated organic enrichment, and caused a trophic shift throughout the estuary (Chamberlain and Hayward, 1996). These processes have had a profound effect on the bottom condition of the estuary, leading to accumulation of soft, mucky sediments and increased frequency and duration of low dissolved oxygen conditions (Chamberlain and Hayward, 1996; Doering, 1996) (***Figure 7-100***). Frequent, severe low dissolved oxygen events are a major stress to the system and a source of serious concern (Graves et al., 2002; Iricanin and Crean, 2007).

Water quality parameters in the system exhibit strong seasonality as well as spatial variation (Iricanin and Crean, 2007; Qian et al., 2007). Seasonal signatures are a normal mechanism in tropical and subtropical systems with distinctive wet and dry seasons driving freshwater inflows and nutrient loads. The seasonal and spatial component of water quality characteristics is reflected in patterns of benthic community composition.

Restoration by CERP and other project implementation is aimed at mitigating impacts of historical water management practices. These efforts would elicit a specific response in the macroinvertebrate infaunal community in predictable ways throughout the system. Stabilization of salinity regimes would allow the establishment of healthier, more species rich benthic communities along the salinity gradient in the estuary by reducing the frequency and severity of disturbance events that exclude many estuarine species. Water quality improvement would decrease hypoxic stress and improve habitat quality and, therefore, the health of community composition by reducing the severity of seasonal disturbance and increasing larval recruitment. Reductions in sedimentation rate and, potentially the partial removal of mucky sediments should improve the health of the benthic community by increasing recruitment potential and infaunal diversity (***Figure 7-100***).

7.5.2 Study Areas

Fifteen fixed sites are regularly monitored (***Figure 7-101***). These sites span all salinity regimes within the St. Lucie Estuary and Southern Indian River Lagoon and cover the watershed in such a way that benthic responses to hydrologic events stemming from the system‟s tributaries can be detected and analyzed.

7.5.3 Study Areas

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7.5.4 Monitoring

The benthic sampling program is a fixed site monitoring effort directed at identifying trends in benthic condition. It involves sampling at 15 sites, 13 of which have been sampled since the program‟s inception in February 2005, with two inner estuary sites added in July 2007 (***Figure 7-101***). Each site is visited four times per year, twice (January and April) during months that typically fall into Florida‟s dry season (November - April) and twice (July and October) during months that typically fall into Florida‟s wet season (May - October).

In addition to assessments of the macrofaunal community, sediment quality and water quality are also sampled. The organic content and particle size distribution of sediments have a profound effect on the nature of infaunal communities so organic content is measured every quarter and particle size is analyzed annually.

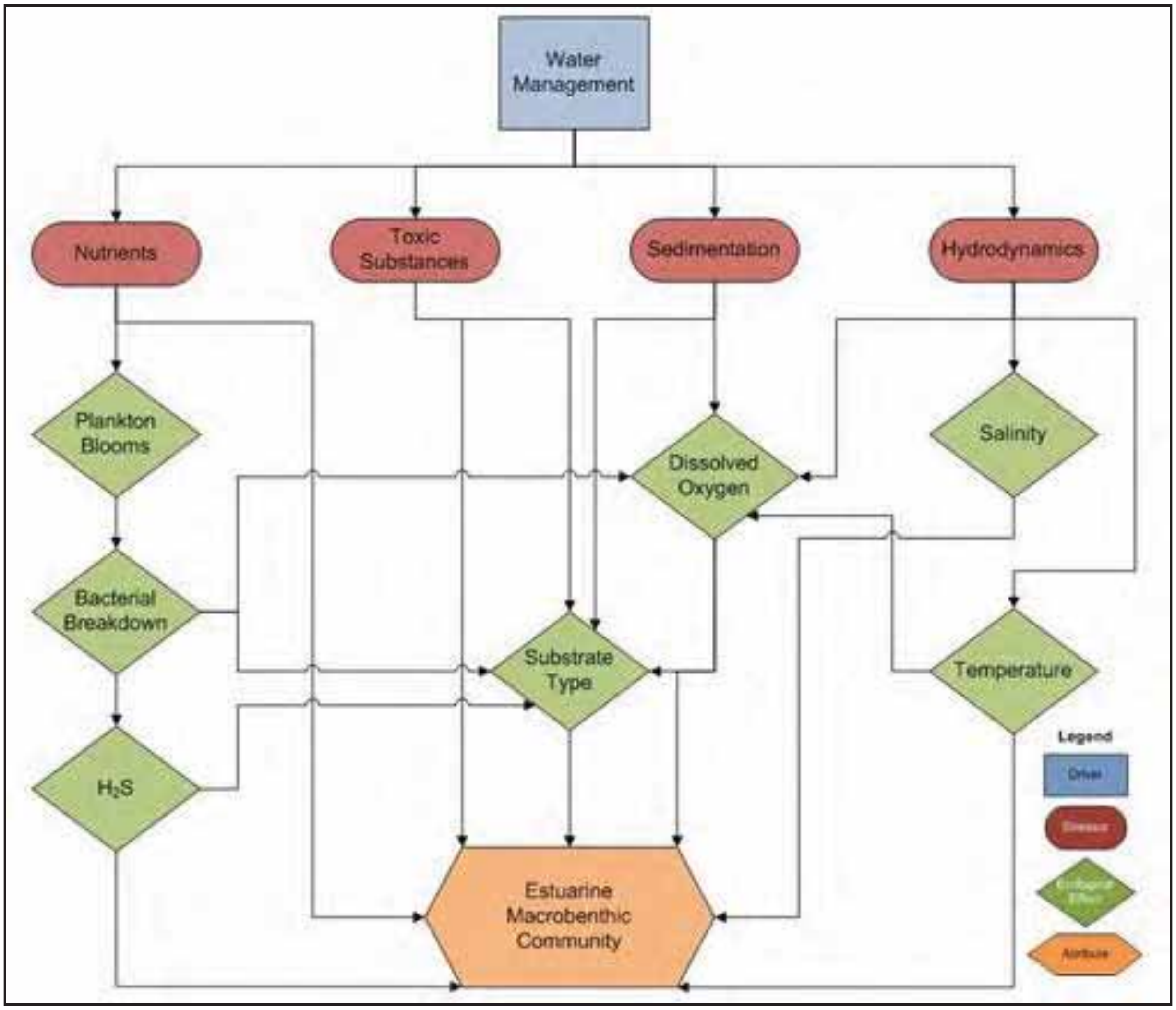


FIGURE 7-100. CONCEPTUAL ECOLOGICAL MODEL FOR BENTHIC MACROINVERTEBRATES IN THE ST. LUCIE ESTUARY AND SOUTHERN INDIAN RIVER LAGOON

Community metrics are used to evaluate status and trends at sites and groups of sites. Changes in abundance, taxonomic richness, and diversity may reveal temporal trends at particular sites or groups of sites with similar salinity regimes. To assess health at all salinity regimes, however, a different type of assessment is needed. The benthic macroinvertebrate monitoring program is working in collaboration with other scientists to apply a robust biotic index to the benthic communities within the Southern Indian River Lagoon and St. Lucie Estuary. AZTI‟s Marine Biotic Index (AMBI) has been applied to marine and estuarine waters worldwide with success (Borja et al., 2008). The premise of AMBI is to place all species in a sample within one of five ecological groups, which are differentiated by their tolerance to stress. The relative distribution of these species in a sample among these five groups is an indication of the level and types of stress that a site has been experiencing over time.

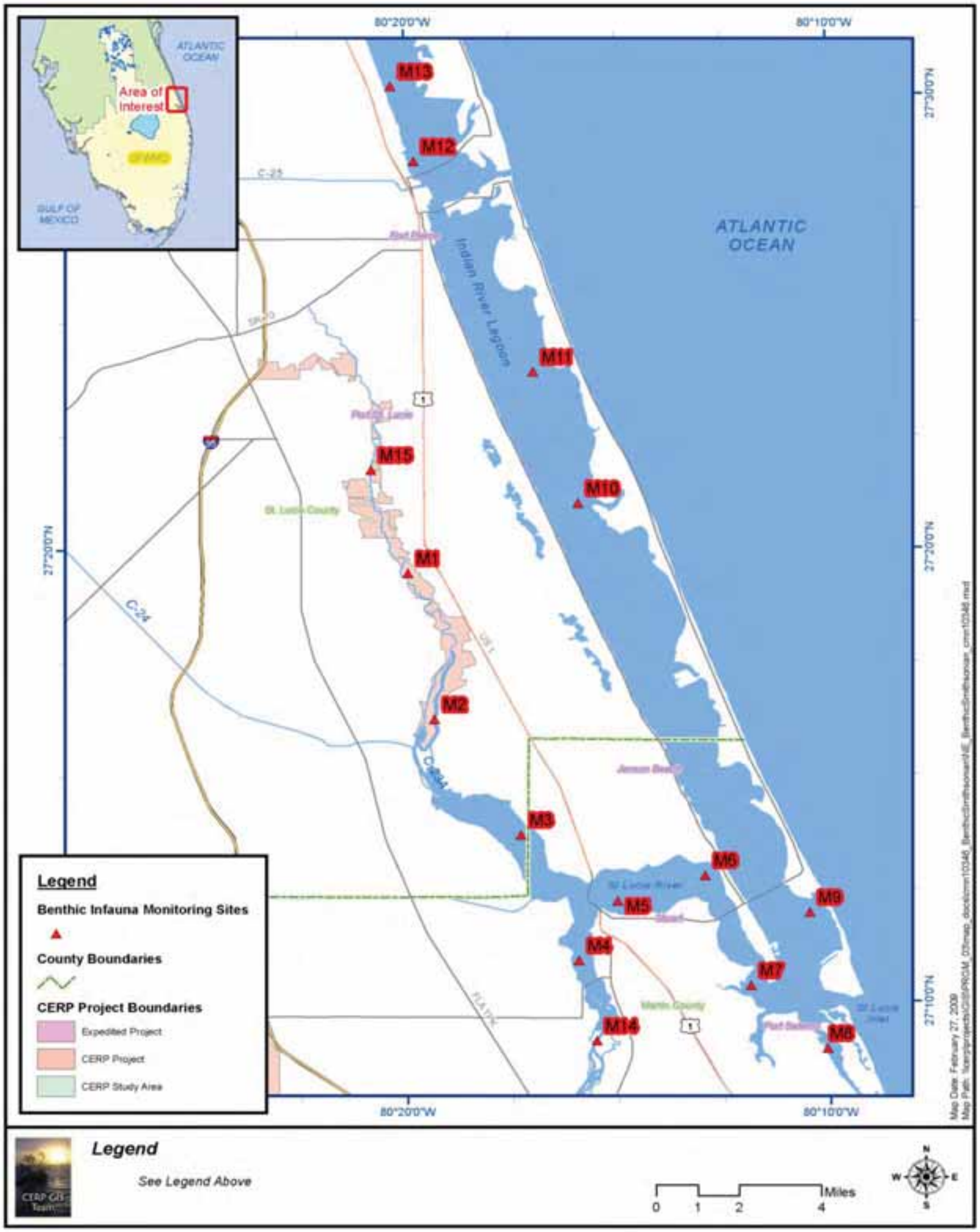


FIGURE 7-101. BENTHIC MACROINVERTEBRATE SAMPLING SITES IN THE ST. LUCIE ESTUARY AND SOUTHERN INDIAN RIVER LAGOON

7.5.5 Results

Conditons in the Saint Lucie Estuary and Indian River Lagoon encompass all four of the major delineations of salinity: oligohaline, mesohaline, polyhaline and euhaline (***Figure 7-102***,

Table 7-10). In this system, patterns of community composition tend to be well differentiated along the salinity gradient. Throughout the time period studied, sites within a particular salinity zone tend to be more similar to each other than sites in other salinity zones. When using MDS analysis to examine all sites sampled within a particular date, oligohaline, mesohaline-polyhaline, and euhaline sites will separate from each other. This pattern is fairly consistent regardless of year or season. ***Figure 7-103*** displays MDS plots for dry season samples and wet season samples for three different years (2005, 2006 and 2008). The points that are close together represent samples that are very similar in community composition and points that are far apart correspond to very different values.

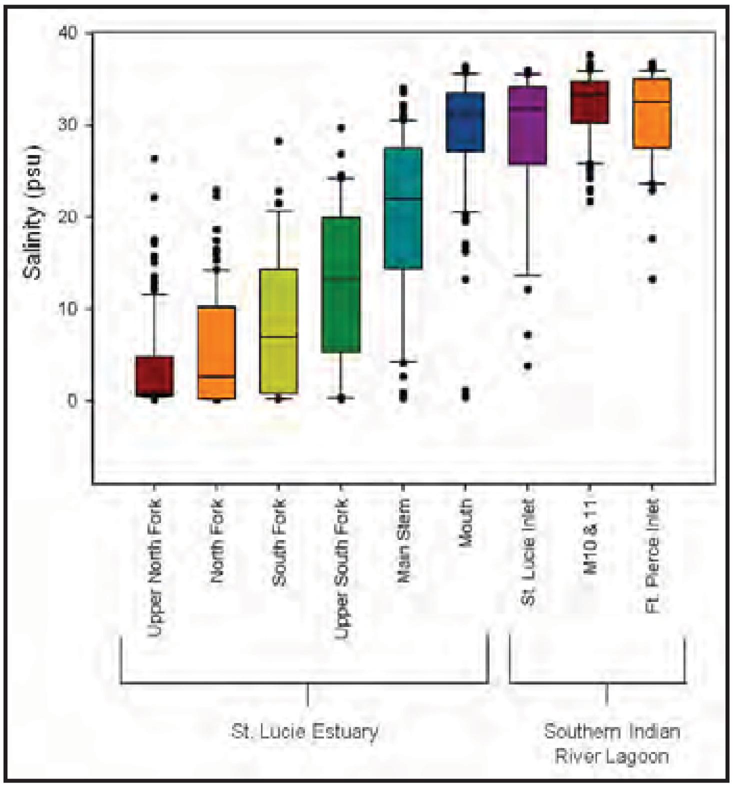


FIGURE 7-102. SALINITY BOX PLOTS WITH MEAN AND INTER-QUARTILE (75TH AND 25TH) RANGES FOR FEBRUARY 2005 TO OCTOBER 2008 DATA

TABLE 7-10. SALINITY ZONES WITHIN THE MACROBENTHIC MONITORING SYSTEM

Region	Sites	Water Quality Sites	Median Salinity	Interquartile (25th and 75th) Salinity Range	Salinity Zone Assignment
Upper North Fork	M1, M2, M15	SE13, SE12, SE06	0.78 psu	<1 - 5 psu	Oligohaline (0 - 5 psu)
North Fork	M3	HR 1	13.07 psu	6 - 20 psu	Mesohaline (5 - 18 psu)
Upper South Fork	M14	SE9, SE10	2.57 psu	<1 - 10 psu	Oligohaline – Mesohaline
South Fork	M4	SE08B	6.93 psu	1 - 13 psu	Oligohaline – Mesohaline
Main Stem	M5, M6	SE03, SE02	21.96 psu	14 - 27 psu	Polyhaline (18 - 30 psu)
Hell‟s Gate (estuary mouth)	M7	SE01	30 psu	26 - 33 psu	Polyhaline (18 - 30 psu)
St. Lucie Inlet	M8, M9	IRL17, IRL12B, SE11	31.93 psu	28 - 34 psu	Polyhaline-Euhaline (30 - 40 psu)
Middle Lagoon	M10, M11	IRL22, IRL27, IRL29, IRL31B	33.33 psu	30 - 35 psu	Euhaline (30 - 40 psu)
Fort Pierce Inlet	M12, M13	IRL34B, IRL36	32.55 psu	28 - 35 psu	Euhaline (30 - 40 psu)

7.5.5.1 Sediment Condition

Salinity limits the distance that a species can penetrate into an estuary, but an infaunal species can only establish when suitable substrates are present. Sediments with a high percentage of organic carbon do not lend themselves well to infaunal community establishment (Gray et al., 2002; Gray and Elliot, 2009). While organic matter in the sediment is an important source of nutrition for most benthic species, too much organic matter can adversely affect species richness and abundance. Microbial breakdown of these materials can potentially release toxic materials and decrease dissolved oxygen concentration at the water-sediment interface where these organisms reside (Hyland et al., 2005). Such conditions act to inhibit certain feeding groups such as filter feeders and select for deposit-feeding organisms with a high tolerance for low oxygen and high sulfide concentrations. Hypoxic soft sediments have a profound effect on larval settlement by eliciting avoidance behavior in many soft bottom species (Marinelli and Woodin, 2002). The relationship between eutrophication, increased organic input, and dissolved oxygen dynamics is a close one. While it may not be possible at this time to isolate a given mechanism that affects benthic community health in the estuary, it is clear that sediment condition can serve as a helpful indicator.

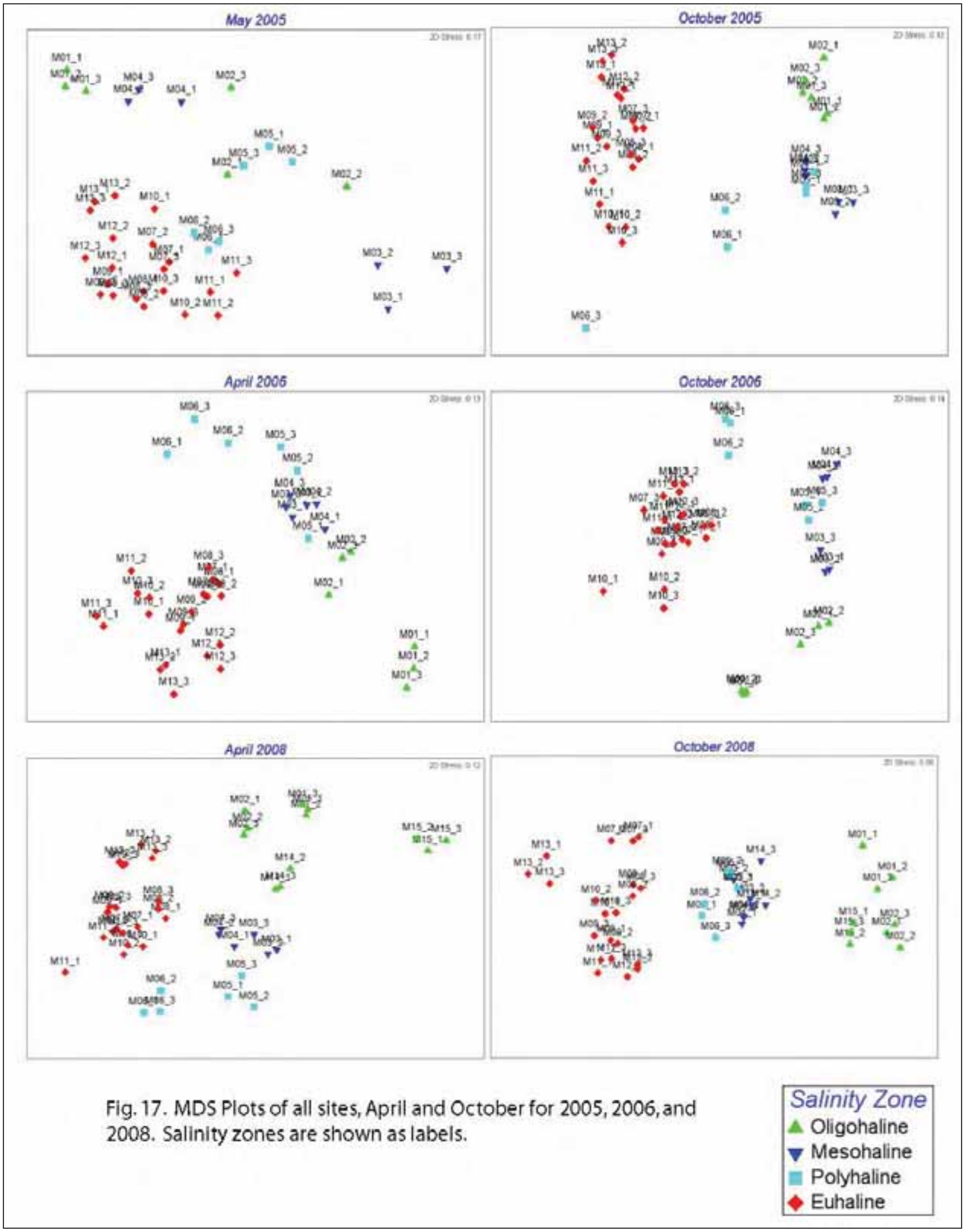


FIGURE 7-103. MDS PLOTS OF ALL SITES FOR APRIL AND OCTOBER IN 2005, 2006 AND 2008

Sediment condition varied significantly throughout the St. Lucie Estuary and Southern Indian River Lagoon (***Figure 7-104*** to ***Figure 7-105***) during the study period. Organic content measured as percent loss on ignition values were highest at the St. Lucie Estuary sites (M3-M6) (***Figure 7-104*** and ***Figure 7-105***). The North Fork (M1, M2 and M15) exhibited elevated values, though not quite as high. Losses on ignition values were lowest at the lagoon sites (M8-M13). The estuary displayed a spectrum of soft bottom community types along the salinity gradient typical of most estuaries.

Quarterly sediment analysis reveal that inner and main stem estuary sites (M3-M6) exhibit an elevated amount of organic carbon, as measured by mass loss on ignition (***Figure 7-104*** and ***Figure 7-105***). Inner estuary sites M14 and M15 also consistently display high amounts of organic carbon while the North Fork sites M1 and M2 tend to vary. These two sites are in erosional-depositional zones of the river where the channel width is much smaller than the basins that contain most of the other sites, which may explain variability in sedimentation rate.

7.5.5.2 Taxa

Number of taxa per sample was very low at sites M3 to M5 throughout the study period (***Figure 7-106***). Numbers of taxa increased slightly in the North Fork (M1 and M2), and increased markedly downstream toward the mouth of the estuary (M6 and M7), with the maximum numbers of taxa per sample recorded in the Southern Indian River Lagoon (M8-M13). Within the St. Lucie Estuary, individuals varied dramatically during the study period (***Figure 7-107***), ranging from near zero to several hundred per sample at a given site. Individuals were lowest on average at sites M3 through M5. Sites within the Southern Indian River Lagoon (M8-M13) contained higher numbers of individuals per sample, where values rarely dropped below 100 individuals per 0.02 m^2. Diversity measures were low at sites M3 through M5, slightly higher in the North Fork (M1 and M2), and increasingly higher and more stable moving downstream in the estuary and into the lagoon (***Figure 7-108***). M14 contained, on average, higher numbers of taxa and individuals than M15 (***Figure 7-109***). The upper south fork site M14 has undergone dramatic shifts in salinity and, consequently, community composition. Over time, sites M3, M4 and M5 display the lowest species richness of any sites in the system (***Figure 7-106***).

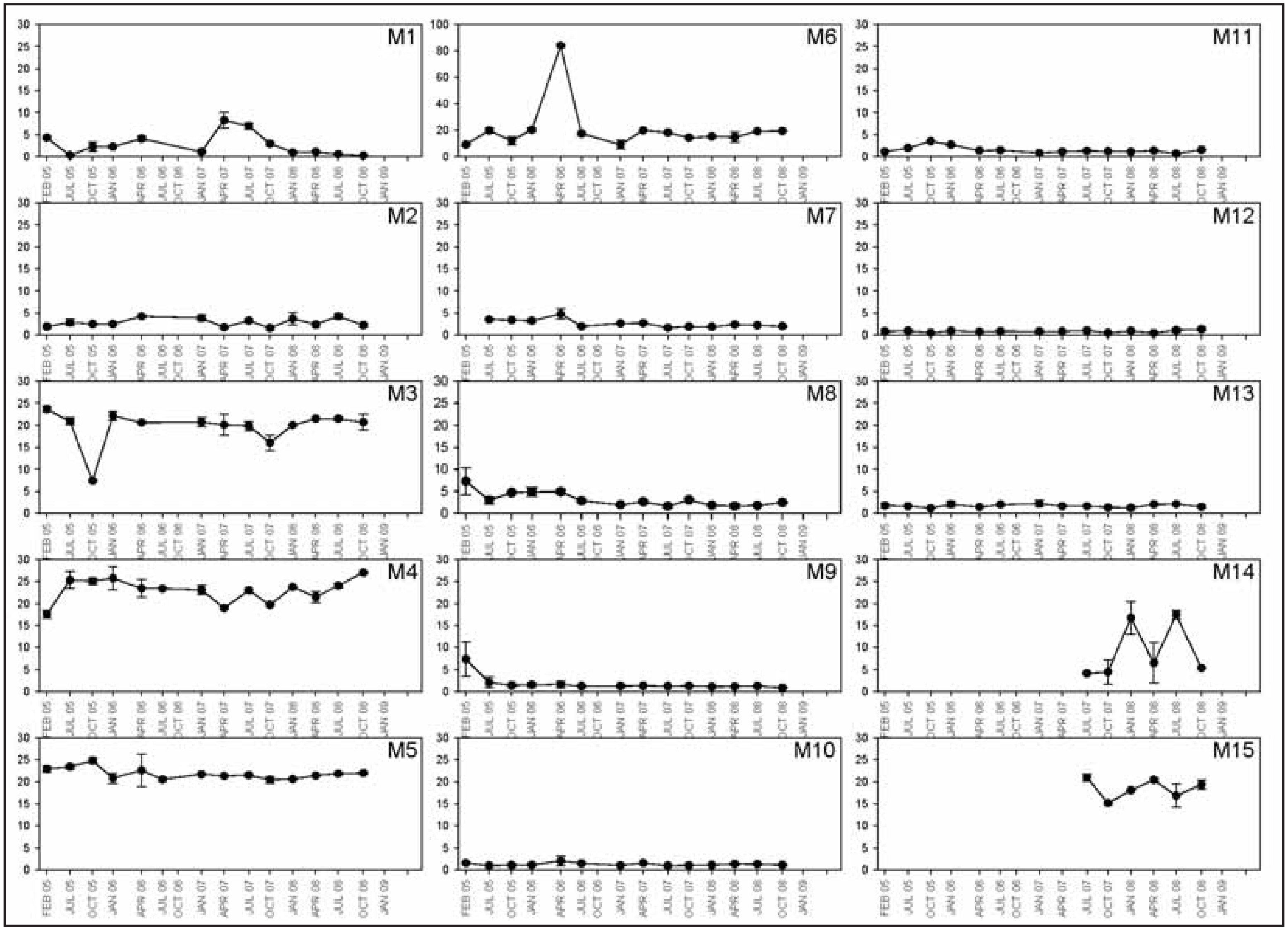


FIGURE 7-104. PERCENT LOSS ON IGNITION (ORGANIC CONTENT) FROM TOP 0 TO 2 CENTIMETERS OF SEDIMENT BETWEEN FEBRUARY 2005 AND OCTOBER 2008

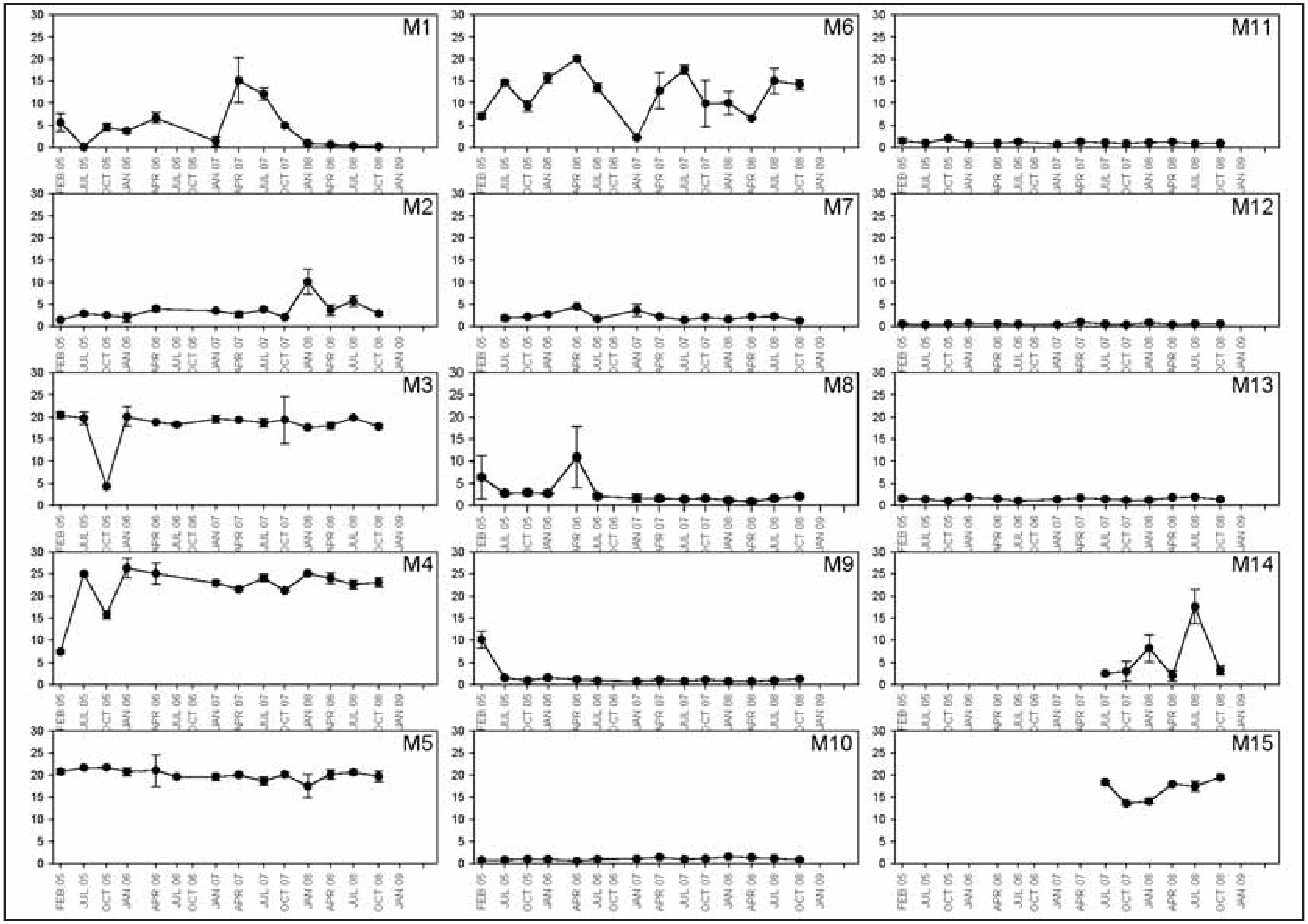


FIGURE 7-105. PERCENT LOSS ON IGNITION (ORGANIC CONTENT) FROM TOP 2 TO 5 CENTIMETERS OF SEDIMENT BETWEEN FEBRUARY 2005 AND OCTOBER 2008

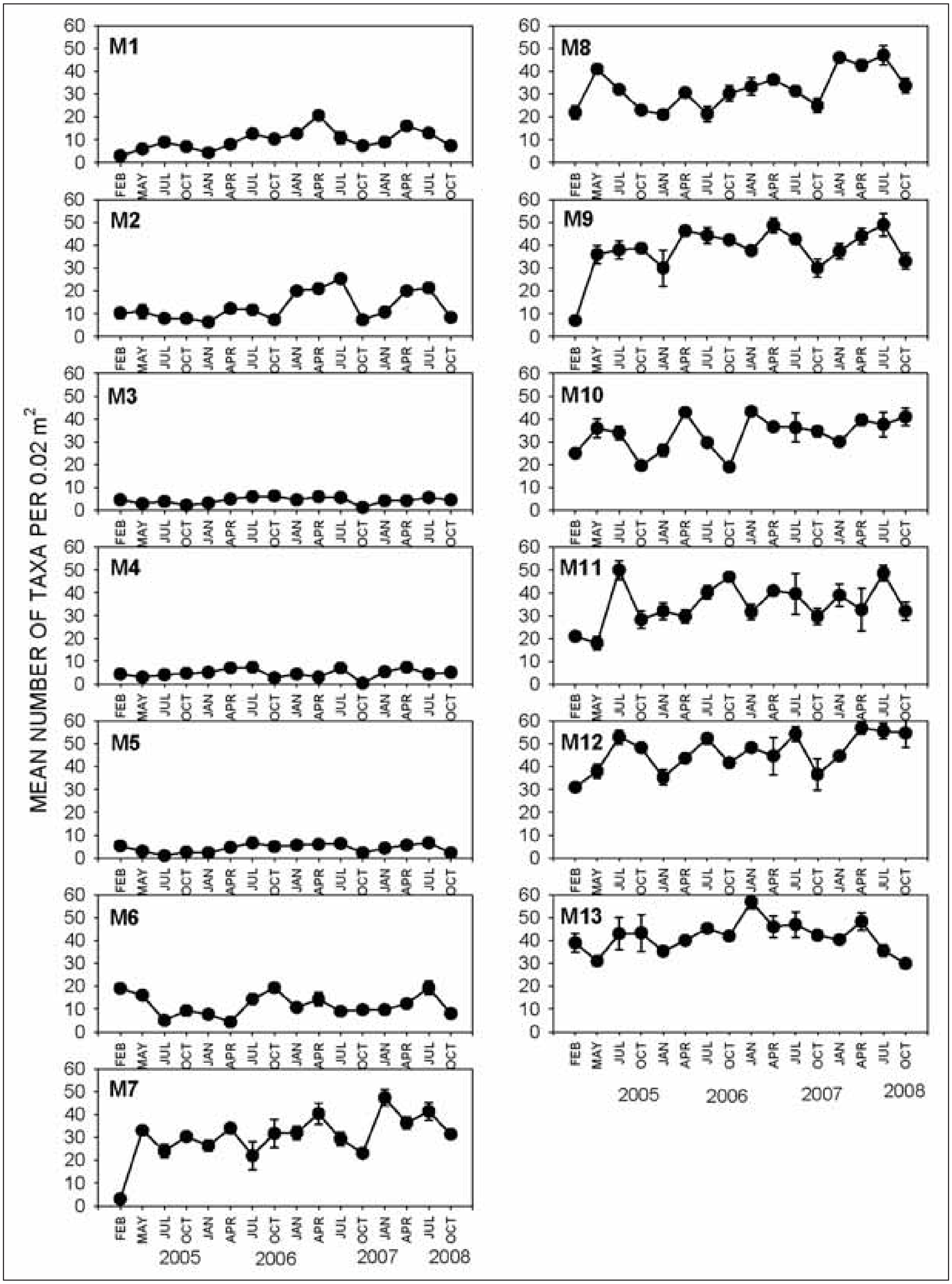


FIGURE 7-106. AVERAGE NUMBER OF TAXA BETWEEN FEBRUARY 2005 AND JANUARY 2009

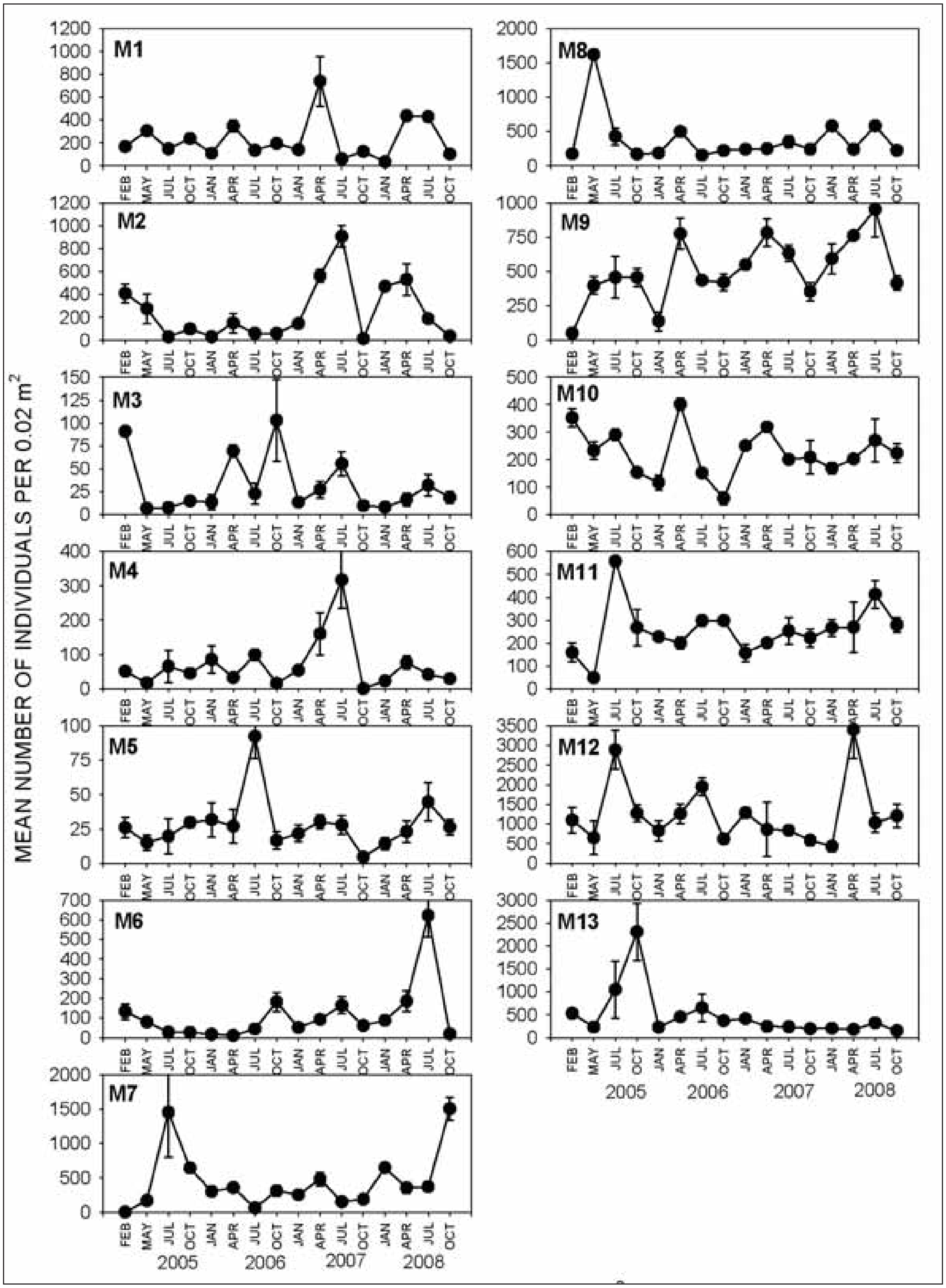


FIGURE 7-107. AVERAGE NUMBER OF INDIVIDUALS BETWEEN FEBRUARY 2005 AND JANUARY 2009

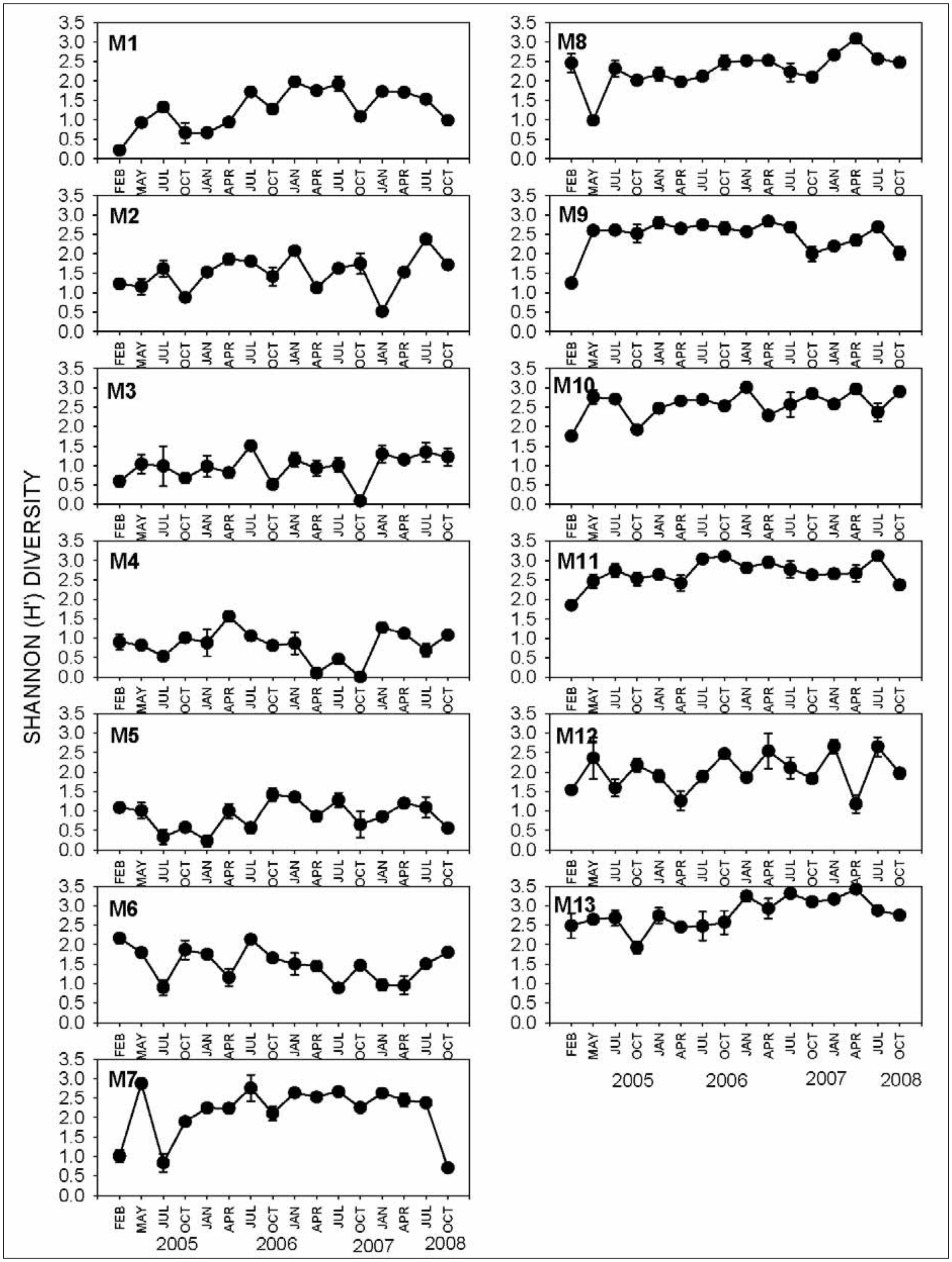


FIGURE 7-108. SHANNON (H' LOG$_E$) SPECIES DIVERSITY BETWEEN FEBRUARY 2005 AND JANUARY 2009

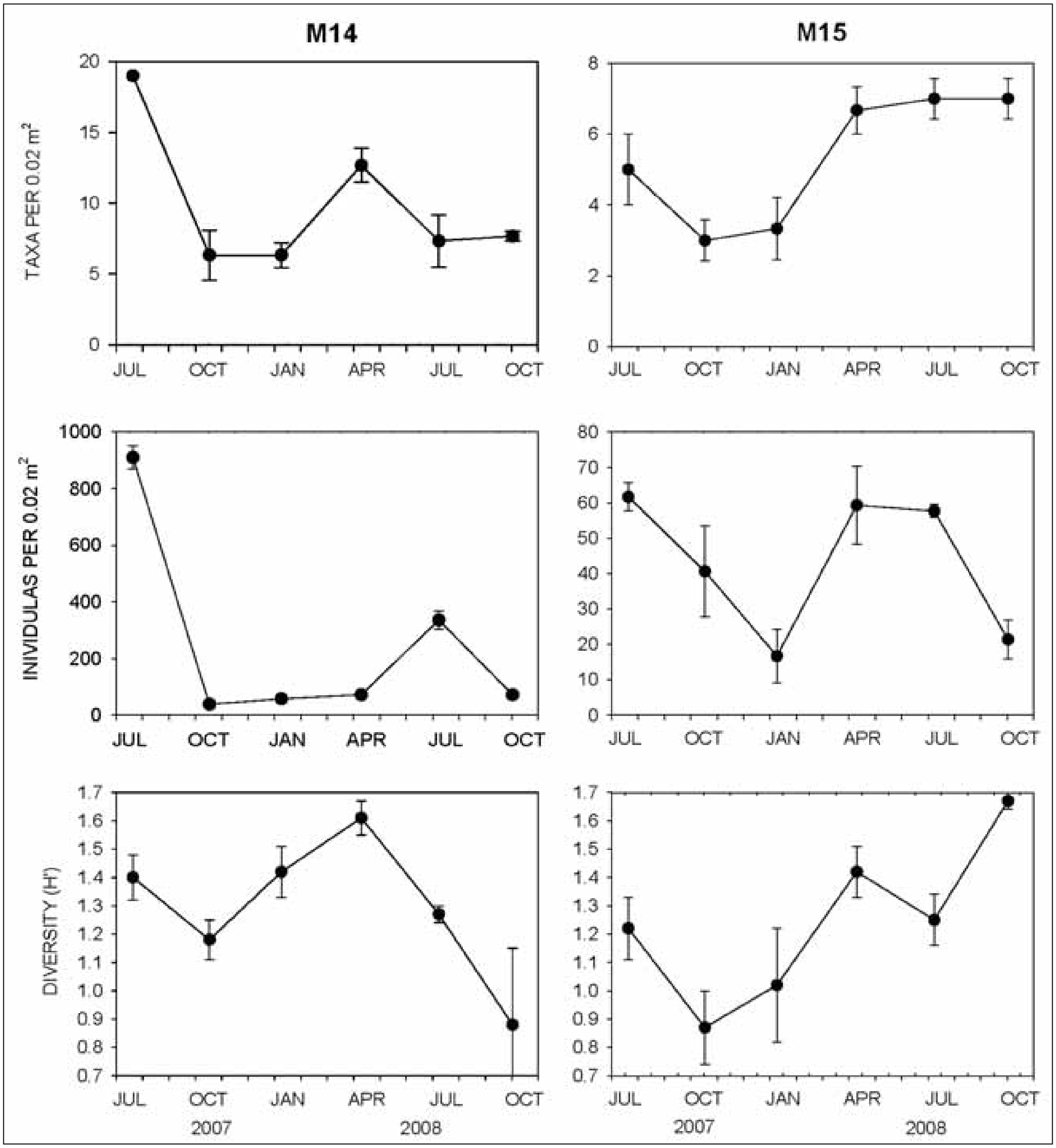


FIGURE 7-109. TAXA, INDIVIDUALS AND DIVERSITY AT SITES M14 AND M15 BETWEEN JULY 2007 AND OCTOBER 2008

When comparing benthic community parameters in the inner and main stem of the St. Lucie Estuary (M3-M6) to freshwater inflow, the seasonal effects of water releases become apparent. Both species richness (number of taxa) and individual abundance tend to decrease in wet months while rebounding in dry months. ***Figure 7-110*** and ***Figure 7-111*** display the change in values (given as standard deviates) of freshwater inflow, taxa, and abundance at sites M3 through M6. Inflow is the cumulative average of all four major inputs to the estuary (C-44, C-23, C-24 and North Fork). The wet season samples that corresponded with high inflow exhibit decreased numbers of taxa and decreased abundance, indicating disturbance events that exclude species recruited during drier months. This effect is less severe in years with no tropical storm activity and below average rainfall, such as in 2006. The inflow event triggered by Hurricane Wilma in October 2005 had a devastating effect on the estuary benthos. Macrobenthic populations did not return to previous levels of abundance and species richness until almost one year later in July 2006. A similar disturbance was observed in the system in October 2007, when a few days of intense rainfall punctuated what was otherwise a notably dry period. Land-based constituents had most likely accumulated during the dry period and were carried to the estuary in high concentration. It is likely that the estuary experienced severe hypoxia during this time. Near-anoxic levels of dissolved oxygen (0.39 mg/L) were recorded at SFWMD water quality site SE08 in the South Fork during a monthly sampling trip. Without in situ monitoring, the extent and severity of events such as this cannot be determined. During fall 2008, normal rainfall levels returned and Tropical Storm Fay deposited prodigious amounts of rainfall. Freshwater inflow was very high and community parameters for October 2008 dropped significantly. The extent of the disturbance event will be evaluated once the post disturbance faunal samples from January 2009 are processed.

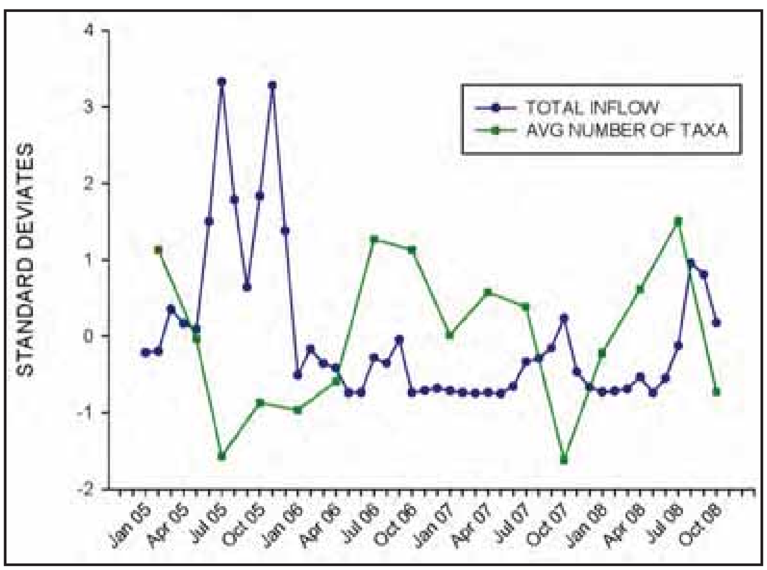


FIGURE 7-110. TOTAL INFLOW AND AVERAGE NUMBER OF TAXA AT SITES M3 THROUGH M6 EXPRESSED AS STANDARD DEVIATES FOR FEBRUARY 2005 TO OCTOBER 2008

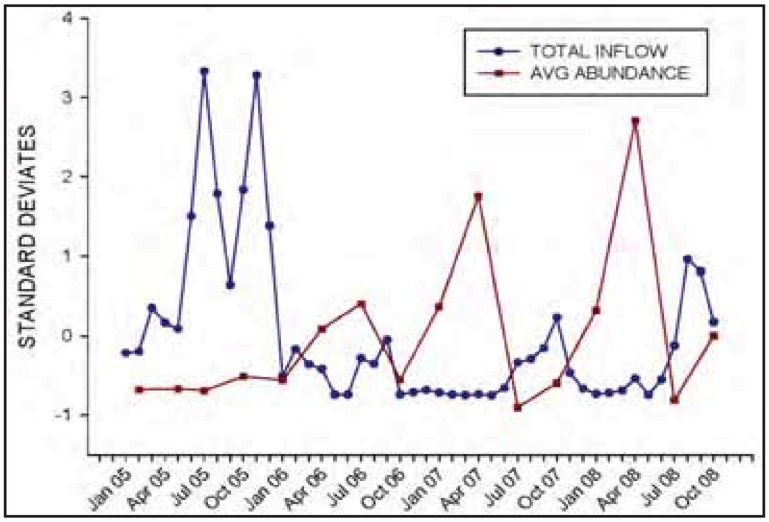


FIGURE 7-111. TOTAL INFLOW AND AVERAGE ABUNDANCE AT SITES M3 THROUGH M6 EXPRESSED AS STANDARD DEVIATES FOR FEBRUARY 2005 TO OCTOBER 2008

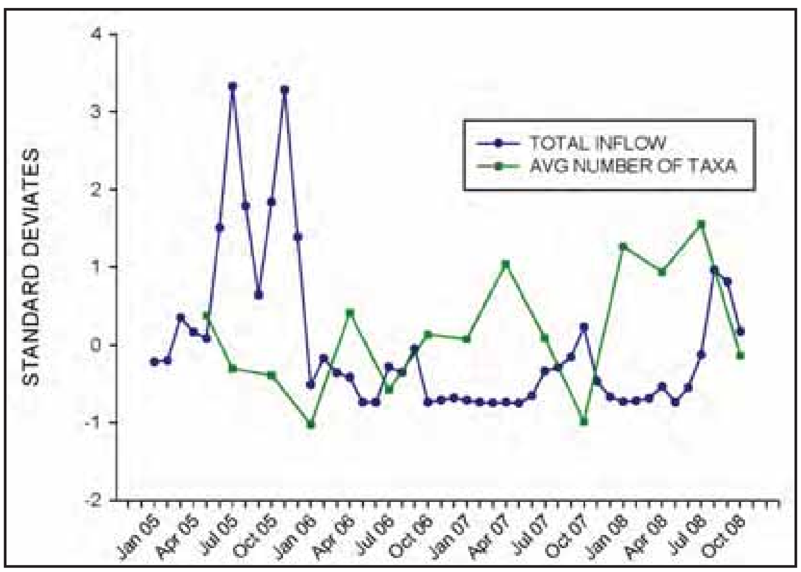


FIGURE 7-112. TOTAL INFLOW AND AVERAGE NUMBER OF TAXA AT SITES M7 THROUGH M9 EXPRESSED AS STANDARD DEVIATES FROM FEBRUARY 2005 TO OCTOBER 2008

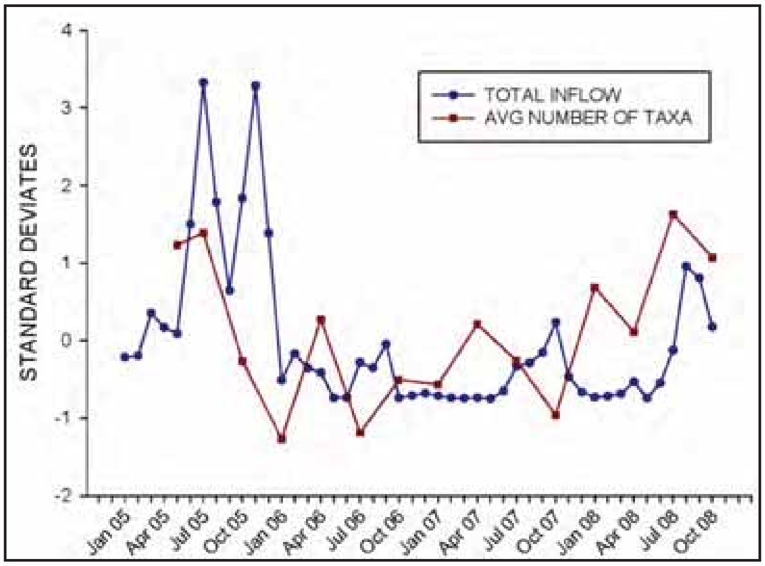


FIGURE 7-113. TOTAL INFLOW AND AVERAGE ABUNDANCE OF TAXA AT SITES M7 THROUGH M9 EXPRESSED AS STANDARD DEVIATES FROM FEBRUARY 2005 TO OCTOBER 2008

The effects of these disturbance events are most severe in the St. Lucie Estuary, where limnic conditions can persist for days, and nutrients and solids can settle out in the basins of the North Fork and the main stem. However, responses are detected in the Southern Indian River Lagoon as well, particularly at sites around the St. Lucie Inlet (M7-M9). While M7 is considered a site within the St. Lucie Estuary, it borders the boundary between the estuary and the lagoon and exhibits similarly high levels of species richness, diversity and abundance. ***Figure 7-112*** and ***Figure 7-113*** display the change in values (given as standard deviates) of freshwater inflow, taxa and abundance at sites M7 through M9. Each increase of freshwater inflow corresponds with a decrease in either taxa or abundance. It is clear that the seasonal fluctuations in freshwater inflow, with its corresponding deliveries of nutrient load and suspended solids, drive the intra-annual dynamic of macrobenthic populations.

Figure 7-114 presents an MDS plot of each sampling date for site M14. Superimposed on the MDS plot for each site is a bubble plot corresponding to the relative proportion of individuals belonging to freshwater groups (in this case, restricted to five psu or less). April and October 2008 had the highest proportion of freshwater taxa and are clearly very similar to each other while being separated from the other dates. The greatest distance between sites and therefore, the greatest dissimilarity between communities, is represented by the gap between July 2007 and April/October 2008. July 2007 represents the conclusion of a period with very little rain and was in the middle of a prolonged zero flow event from the C-44 Canal. Salinities in this part of the estuary spiked as high as 23 psu between March and June 2007. This process of salinity-driven community shifts occurs at most of the inner estuary sites (M14, M3-M6) during sustained regulatory releases and low flow events. Under pre-restoration conditions, salinities at these sites undergo rapid and extreme shifts (Haunert and Startzman, 1985). All other factors excluded, this unstable regime limits the number of estuarine taxa with marine origins that can establish up into the estuary while increasing the distance „downstream" that freshwater taxa can persist. These communities are continually in an early successional state with a high proportion of taxa that are r-selected and/or tolerant, such as the polychaete worm *Streblospio benedicti*, or one of several species of tube-dwelling amphipods, such as *Grandidierella bonneroides* and *Cerapus benthophilus*.

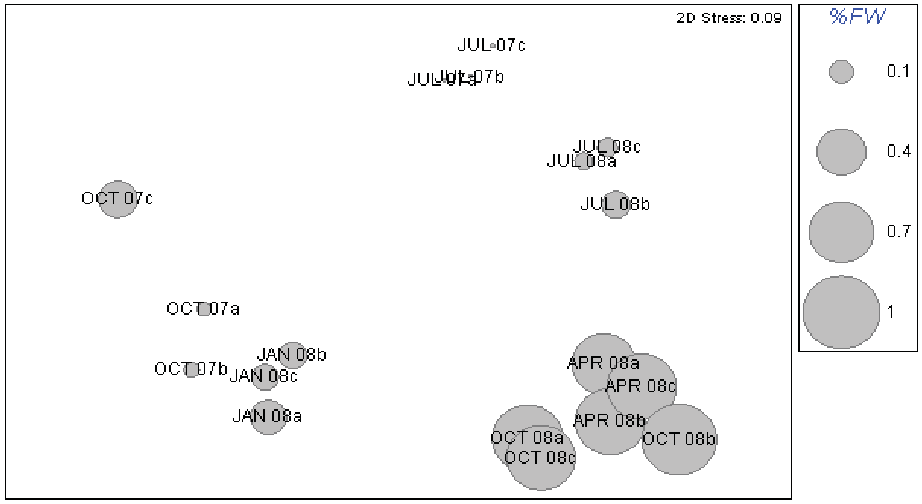


FIGURE 7-114. MDS PLOT OF ALL REPLICATES TAKEN AT M14 FROM JULY 2007 TO OCTOBER 2008 WITH BUBBLES REPRESENTING RELATIVE PERCENTAGE OF FRESHWATER TAXA IN THE SAMPLE

The pattern of improvement over time seems to be related to discharge into the system, as total discharge in the three months previous or two months previous to each sampling affected stations M1 (r: -0.59, p: 0.016) and M7 (r: -0.56, p: 0.023), and M1 (r: -0.56, p: 0.02) and M5 (r: -0.76, p: 0.0006), respectively. Between 2005 and 2008, large changes occurred in total discharge into the system (***Figure 7-114***), and improvement in M-AMBI scores may be related to an overall decrease in discharge over time.

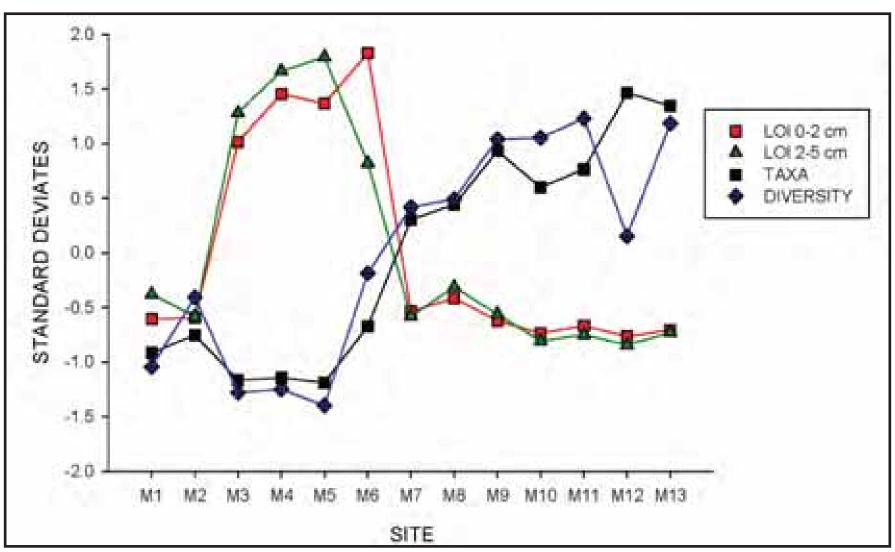


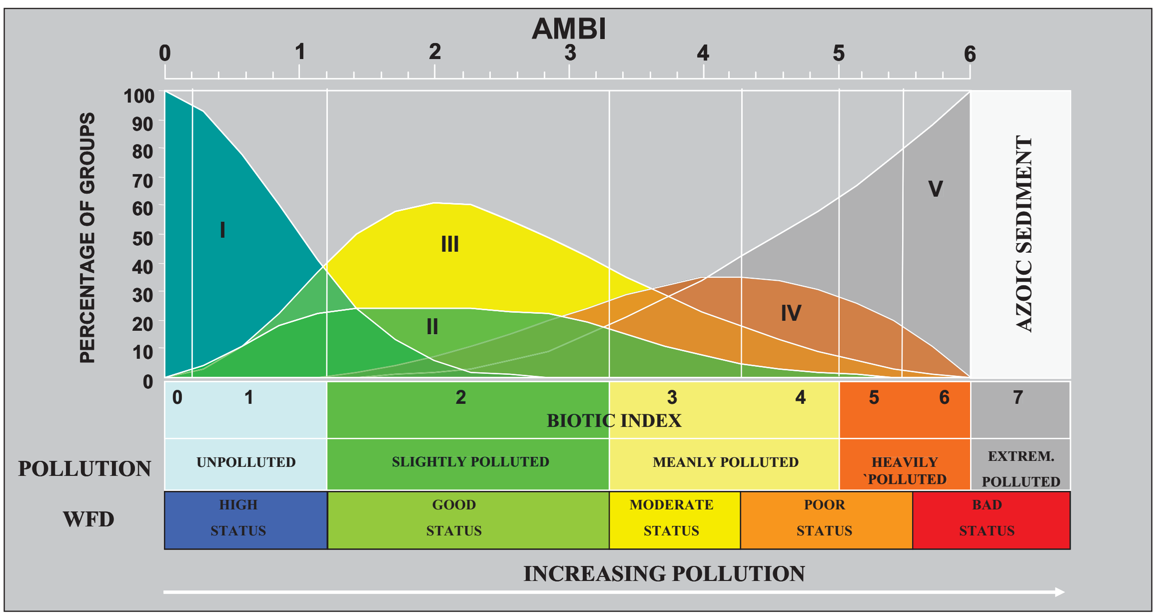
FIGURE 7-115. PERCENT LOSS ON IGNITION, TAXA AND DIVERSITY AS STANDARD DEVIATION UNITS FOR SITES M1 TO M13

Patterns of organic content correspond closely with patterns of species richness throughout the estuary. Sites with elevated organic content tend to be sites with low species richness; sites with low organic content tend to be sites with high species richness (***Figure 7-115***). It is noteworthy that among the low salinity sites M1 through M5, this pattern is consistent as M1 and M2 display higher species richness than M3 through M5 while exhibiting lower organic carbon content. This may discard the notion that the trend in species richness is dictated by salinity alone. Further, it seems apparent that sites M3 through M5 have fewer species than would be expected at this salinity regime, given the capacity for other sites such as M2 from January to July 2007 (***Figure 7-106***) to reach higher mean richness values.

7.5.5.3 Health Index

In order to account for differences between salinity regimes and the varying levels of diversity and richness within each regime, AMBI is used in conjunction with these values to create an ecological quality score, or M-AMBI score ***(Source:*** Bjora et al. 2000

Figure 7-116). In this way, the level of diversity and abundance of invertebrate taxa as well as accounting for the proportion of disturbance-sensitive taxa. Sites are placed on a status gradient: high, good, moderate, poor and bad with reference conditions used as the high boundary.



Source: Bjora et al. 2000

FIGURE 7-116. ORDINATION OF BENTHIC SPECIES INTO FIVE ECOLOGICAL GROUPS ACCORDING TO THEIR SENSITIVITY TO AN INCREASING POLLUTION GRADIENT

Average M-AMBI values indicate that within the St. Lucie Estuary, sites M15, M1, and M3 through M6 are of poor or moderate ecological status (***Figure 7-117***). Sites M14 and M2 are of good ecological status. All of the sites in the Southern Indian River Lagoon are of good or high status. The primary physical difference between good and poor sites within the St. Lucie Estuary is organic content of the sediments; M2 and M14 are comparatively low (***Figure 7-104*** and ***Figure 7-105***). Sites within the St. Lucie Estuary are exposed to a higher degree of nutrient runoff and abrupt salinity changes. Ecological health improves dramatically downstream with M7 at the estuary mouth serving as a threshold point where community health and sediment quality improved dramatically. Southern Indian River Lagoon sites M8 through M13 exhibit high diversity and abundance.

Figure 7-118 presents time series of M-AMBI values for each site. When examining M-AMBI over time, improvement is observed in some of the oligohaline stations (***Figure 7-118a***). For example, M2 improves slightly, M1 improves from bad to good status (this site fluctuates in score, but shows an overall positive trend, the final two months, July and October 2008, represent a decline) and finally, M15 remains between poor and moderate status. Within the mesohaline and polyhaline (***Figure 7-118b***) stretch, M5 shows a significant improvement in M-AMBI values, shifting from bad to poor to moderate status. Other stations show some improvements (M4, M6) although not significant. M14, meanwhile, seems to deteriorate, but not significantly. Sites within euhaline regimes (***Figure 7-118c***) tended to improve, from moderate to good to the boundary between good and high. The only significant trend is that of M11. Several other sites (M7, M8, M10, M12 and M13) show p values near 0.05. Some of these stations show significant increases of richness (M7, M8, M10 and M12) or diversity (M11 and M13), explaining why M-AMBI values seem to improve. As is the case with the AMBI scores, the best M-AMBI scores (good to high status), is found in the euhaline part of the system, which is less affected by discharges and diffuse runoff. Some parts of the oligohaline (M2) and mesohaline and polyhaline (M3, M6 and M14) stretches have climbed to good status more recently. The most degraded areas (poor or moderate status recently) are confined to the St. Lucie Estuary, in oligohaline (M1 and M15) or mesohaline and polyhaline stretches (M4 and M5).

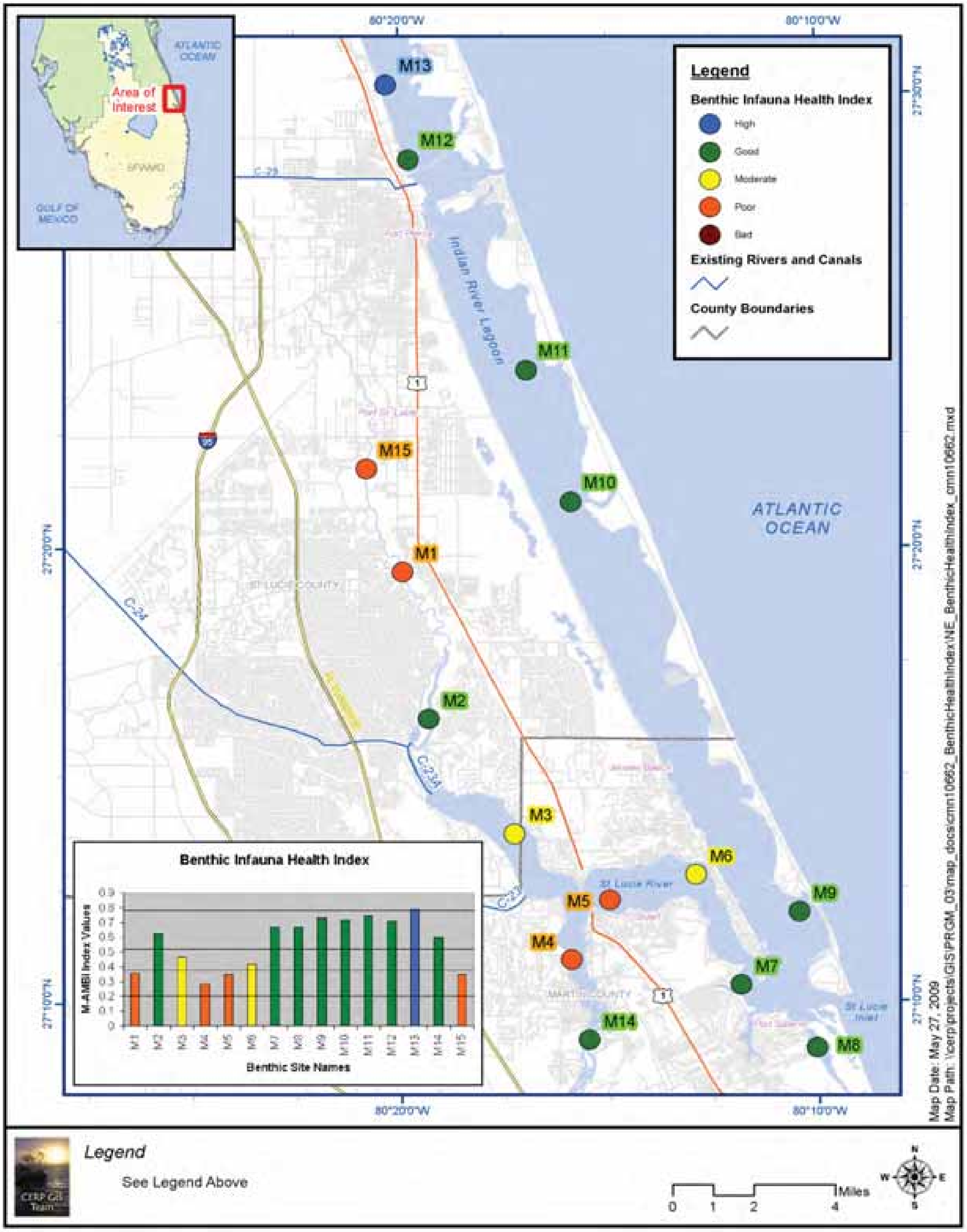


FIGURE 7-117. AVERAGE M-AMBI SCORES FOR BENTHIC MACROINVERTEBRATES IN ST. LUCIE ESTUARY AND SOUTHERN INDIAN RIVER LAGOON

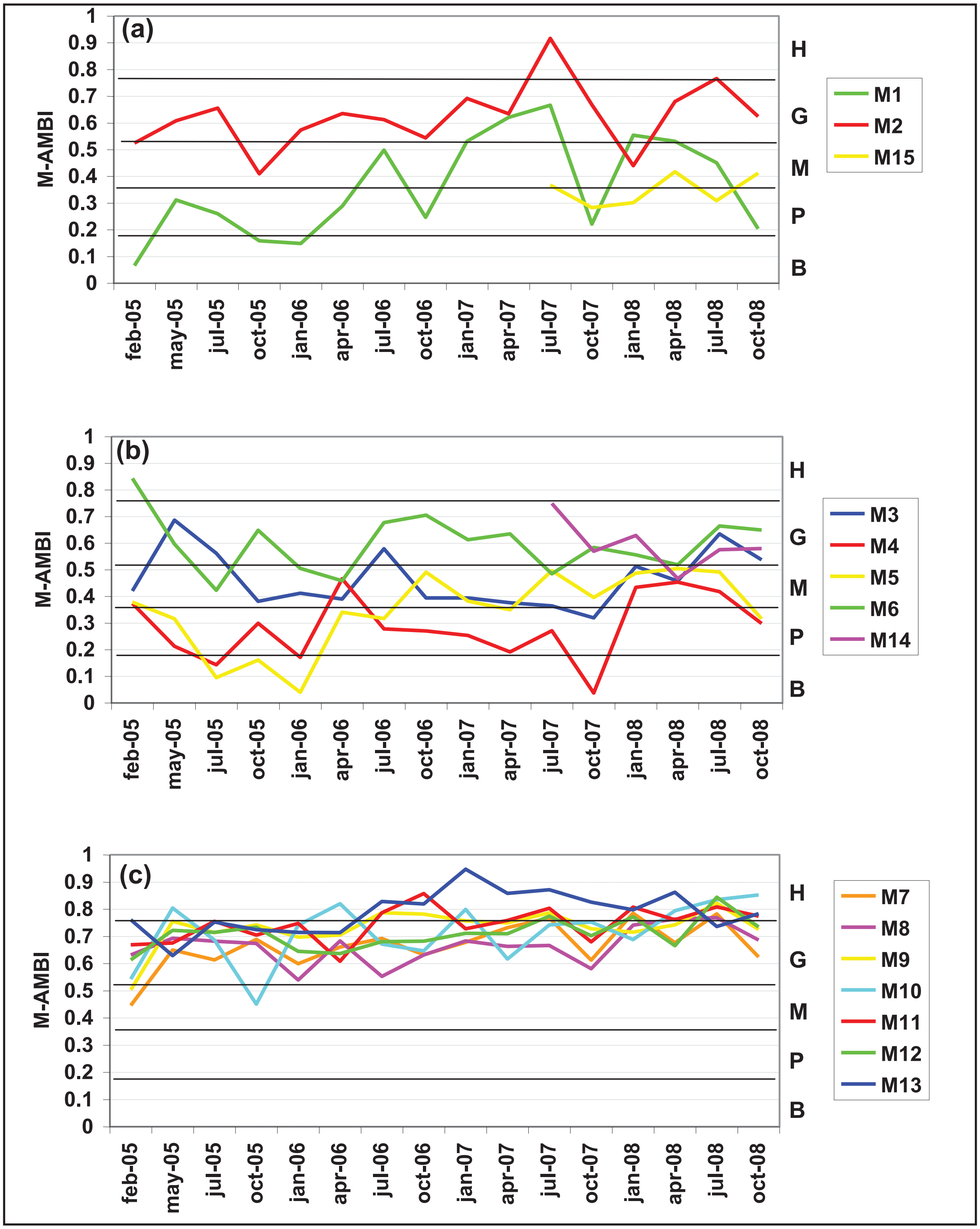


FIGURE 7-118. TIME SERIES OF M-AMBI VALUES FOR EACH SITE FOR FEBRUARY 2005 THROUGH OCTOBER 2008

Key: B=bad G=good
P=poor H=High)
M=moderate

7.6 FISH HYPOTHESIS CLUSTER

7.6.1 Introduction and Background

A CEM for fish in the Northern Estuaries is presented in ***Figure 7-119***. Water management affects hydrodynamics and oceanography; water quality including salinity, temperature, dissolved oxygen and nutrients; and sedimentation bottom type and bathymetry. These in turn affect food availability and habitat. Without appropriate food and habitat, fish cannot reproduce and grow within these estuaries.

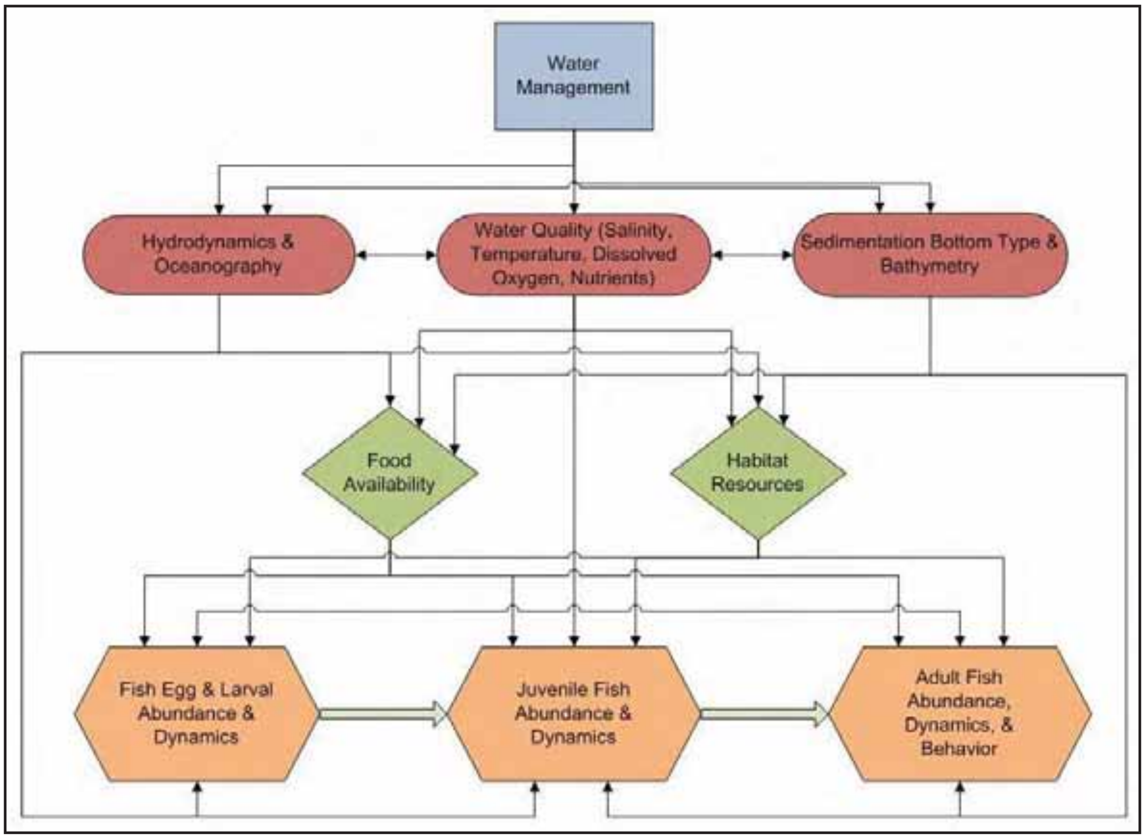


FIGURE 7-119. CONCEPTUAL ECOLOGICAL MODEL FOR FISH IN THE NORTHERN ESTUARIES

7.6.2 Monitoring

From 2003 to 2007, the RECOVER Fish Subteam of the Northern Estuaries module actively met. Their primary objective was to determine how fish can give relevant and timely assessments of performance measures. During this same time period, a collaborative effort by state and federal agencies and private entities tested a wide variety of fish assessment and monitoring techniques and technologies. The Fish Subteam came to a consensus that a variety of techniques should be implemented to target critical life history stages of regional fish populations that are most likely to be impacted by restoration activities. Abundant small species and early developmental stages of larger species are known to be most vulnerable to water quality, predation, food limitations and habitat loss. Therefore, these smaller fishes and the early life history stages of larger species were of particular interest.

Classical capture techniques were suggested including plankton, seine and trawl nets of various sizes and configuration. Shock boats were used in habitats where seine and trawl nets were not practical, such as heavily vegetated shorelines. Spawning and migratory behavior of adult fish were also targeted for performance measures as technologies now allow these behaviors to be readily monitored continuously and remotely using passive acoustic systems. The team recommended that water management scenarios should consider three primary performance measures to determine restoration impacts on fish assemblages: 1) fish spawning intensity monitored continuously with passive acoustic technologies, 2) fish larval abundance and diversity monitored routinely relative to the physical-chemical frontal boundaries associated with freshwater flows, and 3) juvenile fish abundance and diversity in seagrass habitats monitored routinely where riverine systems can influence seagrass habitats.

Pilot projects were designed to evaluate the overall efficiency and effectiveness of several of these methods. These pilots ran from 2003 to 2007. Technique trials included collaborative work with the FWC, FDEP, USGS, NOAA/National Ocean Survey, and the Florida Oceanographic Society. Results from this initial effort can be found in the 2007 SSR (RECOVER, 2007b) at www.evergladesplan.org/pm/recover/assess_team_ssr_2007.aspx .

Due to funding constraints and difficulties related to data management and assessment technique development, the RECOVER funded fish monitoring was put on hold starting in October 2007. Once these difficulties are overcome, the intent is to implement the recommended projects as a long-term fish monitoring program in the Northern Estuaries.

7.6.2.1 St. Lucie Estuary Fish Health

A project funded by the St. Lucie River Issue Team, which was formed by the South Florida Ecosystem Restoration Task Force, has been examining the prevalence of fish with externally visible abnormalities. The prevalence of fish with abnormalities, as a reflection of fish health, appears to integrate the effects of the stressors shown in ***Figure 7-119.*** Conceptual ecological model for fish in the Northern Estuaries**.** Beginning in 2000, a series of retrofit, restoration, and BMP projects were implemented in the St. Lucie Estuary watershed by local government agencies.

The project consists of weekly sampling of fish by rod and reel in the St. Lucie Estuary and nearby reference systems. All fish captured are recorded as normal or having one or more type of abnormality. In addition, the project obtains environmental data including salinity, visibility, temperature, pH, TSS, dissolved organic carbon, chlorophyll-*a* and color.

Thirteen years of St. Lucie data encompassing 54,174 fish in 64 species and covering the period from November 1996 through December 2008 are being examined in relation to environmental variables such as freshwater inflow, salinity, water column visibility, temperature, pH and other parameters. Since the species composition of the samples varies somewhat from month to month, variation in the prevalence of abnormalities among species could potentially confound a monthly time series of prevalence based on the raw data; therefore, a mixed-effect model with species as the random variable is used to produce a monthly time series of predicted prevalence that has been purged of species-specific influences.

7.6.3 Results

7.6.3.1 St. Lucie Estuary Fish Health

In the time series of the prevalence of fish with any externally visible abnormality (***Figure 7-120***), one can see seasonal trends, exceptionally high spikes corresponding to two hurricane periods during 2004 and 2005, and a sustained period of low prevalence, starting about October 2006, coincident with a long drought. These apparent responses to altered conditions provide evidence that the prevalence index is sensitive to environmental variation. While the long period of low prevalence corresponds with a multi-year drought, the area has experienced previous droughts without such a persistently low prevalence of fish with abnormalities as that of the recent period. The watershed management actions funded by the Florida State Legislature through the St. Lucie River Issue Team and implemented by local governments may have contributed to the reduction in the prevalence of fish with abnormalities in the past few years. Continued monitoring and associated analyses would help clarify the reasons for the recent low prevalence. Freshwater flow and salinity associated with this study are provided in ***Figure 7-121*** and ***Figure 7-122.*** Monthly mean salinity at the time of sampling in the St. Lucie Estuary from November 1996 through December 2008, respectively.

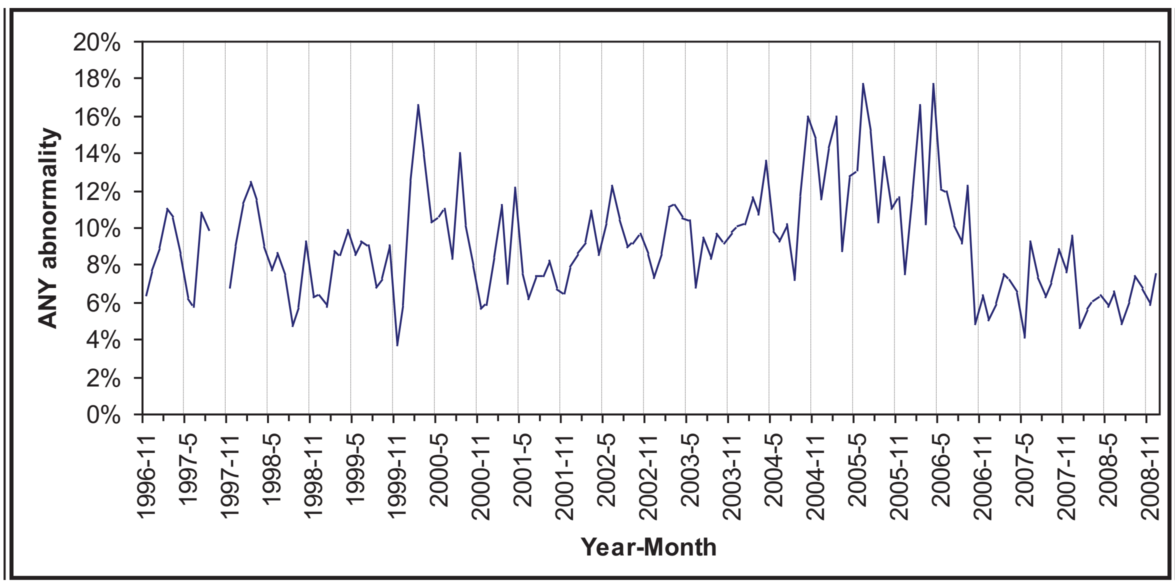


FIGURE 7-120. INDEX OF PREVALENCE OF FISH WITH ANY ABNORMALITY, BY MONTH, IN THE ST. LUCIE ESTUARY FROM NOVEMBER 1996 THROUGH DECEMBER 2008

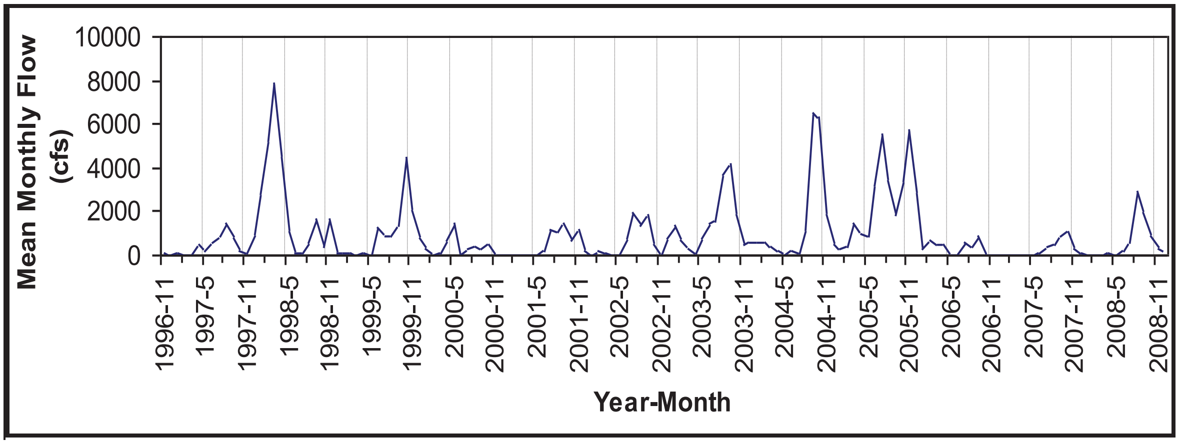


FIGURE 7-121. COMBINED MONTHLY FLOW RATE FROM THE C23, C24 AND C44 CANALS FOR NOVEMBER 1996 THROUGH DECEMBER 2008

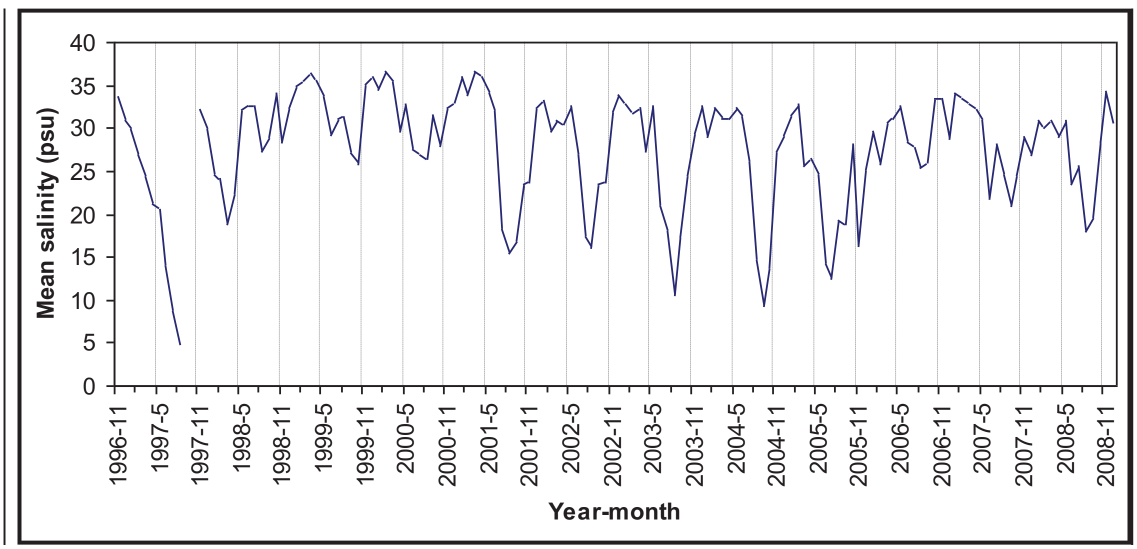


FIGURE 7-122. MONTHLY MEAN SALINITY AT THE TIME OF SAMPLING IN THE ST. LUCIE ESTUARY FROM NOVEMBER 1996 THROUGH DECEMBER 2008

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CHAPTER 8 GREATER EVERGLADES WETLANDS MODULE

8.1 INTRODUCTION

The remaining Everglades encompass a mosaic of inter-connected freshwater wetlands and estuaries located primarily south of the EAA (***FIGURE 8-1***). The Greater Everglades Wetlands Module is organized around defining characteristics of the Everglades, as described in the Total System Conceptual Ecological Model (Ogden et al., 2005), that together distinguished the Everglades ecosystem from other large wetland systems:

- Sheet flow characterized by a milti-kilometer expanse of slowly moving, shallow fresh water, and by multi-year hydroperiods throughout extensive slough systems
- Oligotrophic nutrient status resulting from water inputs primarily from direct rainfall
- A subtropical patterned peatland landscape of sloughs, sawgrass ridges and tree islands intergrading with shorter-hydroperiod wetlands and upland habitats
- Highly productive mangrove estuaries characterized by extensive zones and prolonged periods of oligohaline-to-mesohaline salinity, and by "upside-down" nutrient dynamics of nitrogen enrichment from upstream freshwater marshes and phosphorus enrichment from the Gulf of Mexico
- Large breeding populations of wading birds dominated by tactile-feeding species that depend on the high production and concentration of aquatic prey in shallow water
- The American alligator and its role as a keystone species in the Everglades

The defining characteristics identify a minimum set of goals that must be achieved if the CERP is to be considered successful in restoring the Everglades as a unique ecosystem. Working hypotheses and monitoring results in this report are organized in the context of the defining characteristics.

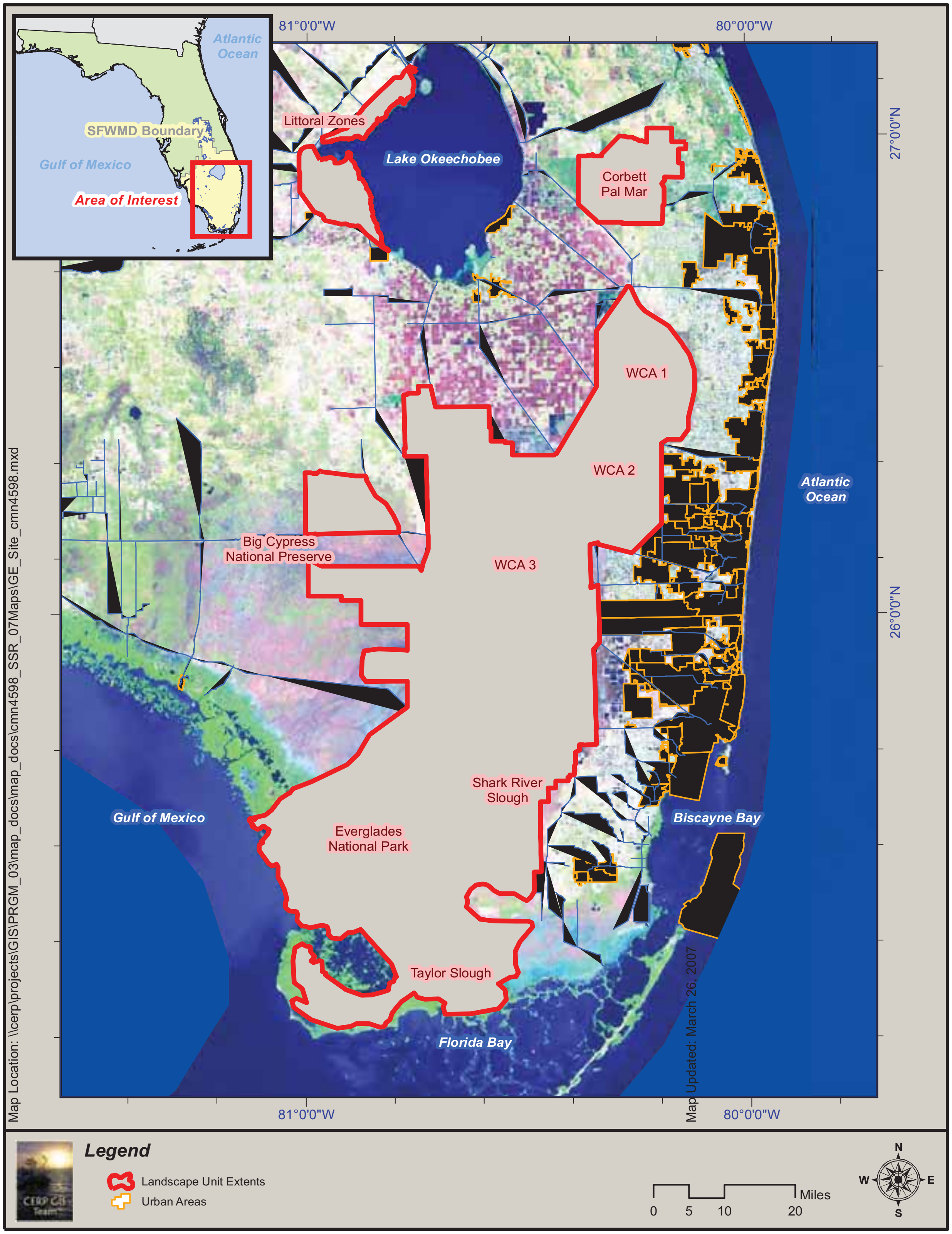


FIGURE 8-1. GREATER EVERGLADES WETLANDS MODULE BOUNDARY

Note: With ridge and slough tree island habitat featured as the patterned landscape present in many portions of the wetlandsSheet Flow and Water Depth Patterns

8.1.1 Introduction and Background

In the absence of point-source water inputs and flow channels, antecedent rainfall in combination with the dynamic storage capacity of the vast south Florida wetland system produced a multi-kilometer-wide expanse of slowly moving, shallow fresh water across much of the ridge and slough landscape of the Everglades habitat pattern (***FIGURE 8-1***) and ***as*** depicted by the Conceptual Ecological Model ***(Figure 8-2)***. The ridge and slough landscape included the remaining wetlands in the Loxahatchee National Wildlife Refuge (NWR), WCAs 2 and 3, the Shark River Slough complex in Everglades National Park (Broad, Harney, Shark and North River drainages), and Lostman's Slough in Everglades National Park. The restoration of the Everglades as an ecosystem requires the resumption of pre-drainage volume, timing and distribution of sheet flow and resulting water depth patterns in a wetland system of large spatial extent, undivided by levees and canals, and connected to downstream mangrove estuaries.

System-wide performance measures that have been developed for sheet flow and water depth patterns in the Greater Everglades Wetlands include measures for extreme high and low water levels, inundation patterns, sheet flow in the ridge and slough landscape, and the number and duration of dry events for Shark River Slough. Documentation for these measures can be found at www.evergladesplan.org/pm/recover/perf_ge.aspx. Interim goals have been developed for water volume, sheetflow and hydropattern. Documentation for these interim goals can be found at www.evergladesplan.org/pm/recover/igit_subteam.aspx.

8.1.2 Monitoring

The EDEN is an integrated multi-agency network of real-time water level monitoring, ground elevation modeling, and water surface modeling that provides ongoing water level and depth information for the freshwater wetlands of the WCAs, Big Cypress National Preserve and Everglades National Park (sofia.usgs.gov/eden).

Surface water level data have been collected daily since 1999 at up to 250 wetland and canal gaging stations operated by the Big Cypress National Preserve, Everglades National Park, SFWMD and the USGS. The EDEN surface water model uses the daily median water levels from the EDEN network of gages to create a spatially continuous interpolation of the daily water surface elevation for the greater Everglades. When the EDEN ground elevation model is subtracted from the daily water level surface, an estimate of daily water depth is computed. EDEN surfaces are created on a 400 x 400 meter grid therefore hydrographs of water level and water depth for individual grid cells represent the mean values for that cell.

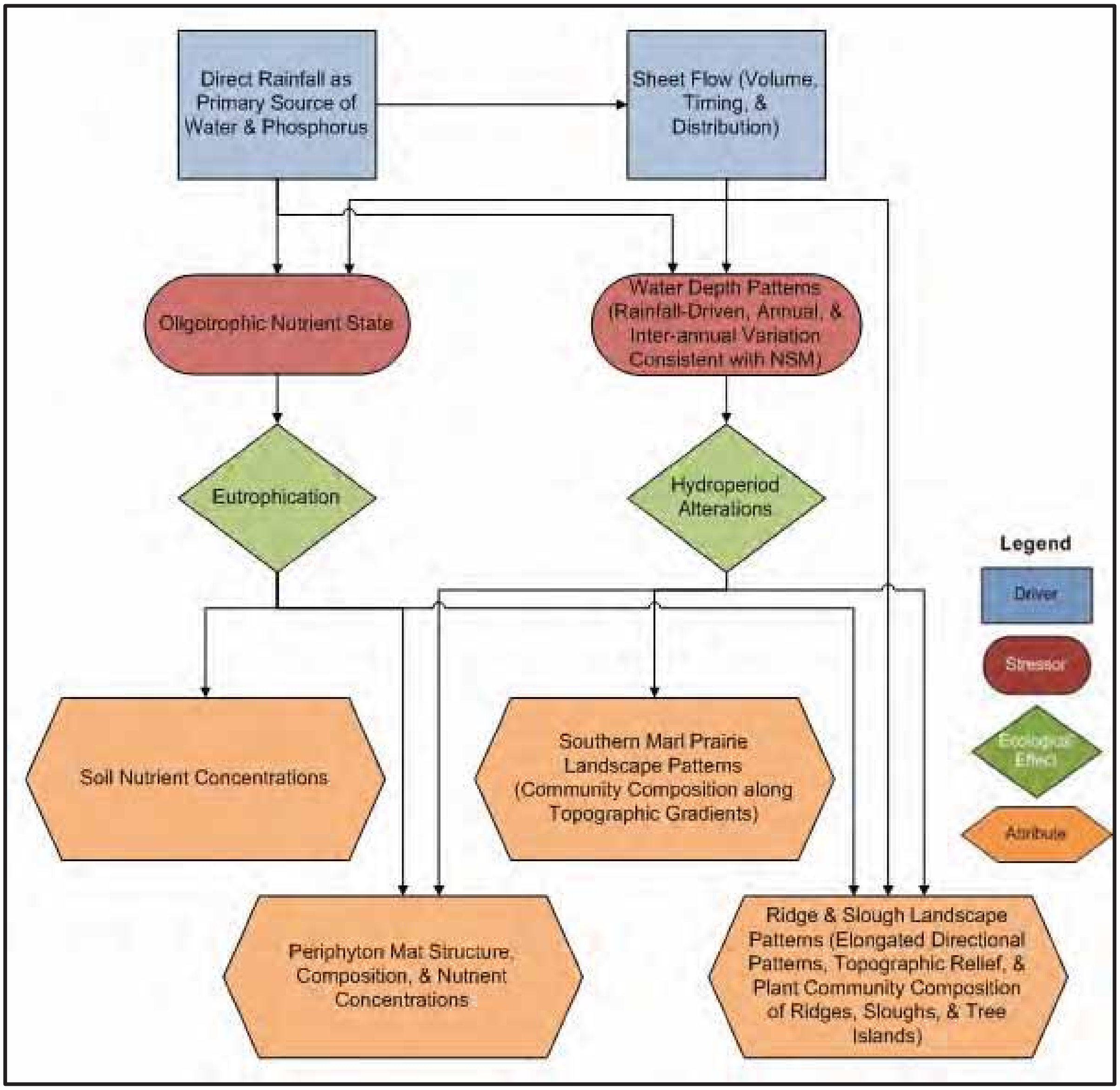


FIGURE 8-2. CONCEPTUAL ECOLOGICAL MODEL FOR SHEET FLOW AND WATER DEPTH PATTERNS IN THE GREATER EVERGLADES WETLANDS

Climate patterns during WYs 2006 through 2009 are summarized by Lori Miller (USFWS, personal communication). Rainfall over the Everglades is taken from the 2006 to 2010 South Florida Environmental Reports (Abtew et al., 2006; 2007; 2008; 2009; 2010). A WY is defined as beginning on May 1 through April 30 of the subsequent year. The wet season extends from May 1 to October 31, followed by the dry season from November 1 to April 30. The 2006 WY is initiated on May 1 2005 and concludes April 30, 2006.

Hydrographs during WYs 2006 through 2009 represent the mean of water depths within 42,415 grid cells of EDEN throughout the Everglades. Rainfall superimposed on the hydrographs represents the mean of 13 gages located throughout the system. As such, the hydrographs depict system-wide average water depths and rainfall throughout the WCAs, Everglades National Park and eastern Big Cypress National Preserve.

8.1.3 Results

The combined influences of climate and water management produced highly contrasting water depth patterns in the Everglades during WYs 2006-2009 (***Figure 8-3***). Given the scale and complexity of summarizing water depth conditions, this section attempts to comprehensively summarize hydrologic conditions from 2006-2009 by starting with a high level summary of spatially aggregated comparisons of WYs. The next step is to characterize spatial variability in hydropatterns that result from the existing infrastructure and water management operations. This section closes with an overview of the hydrologic conditions that are relevant to ecological processes.

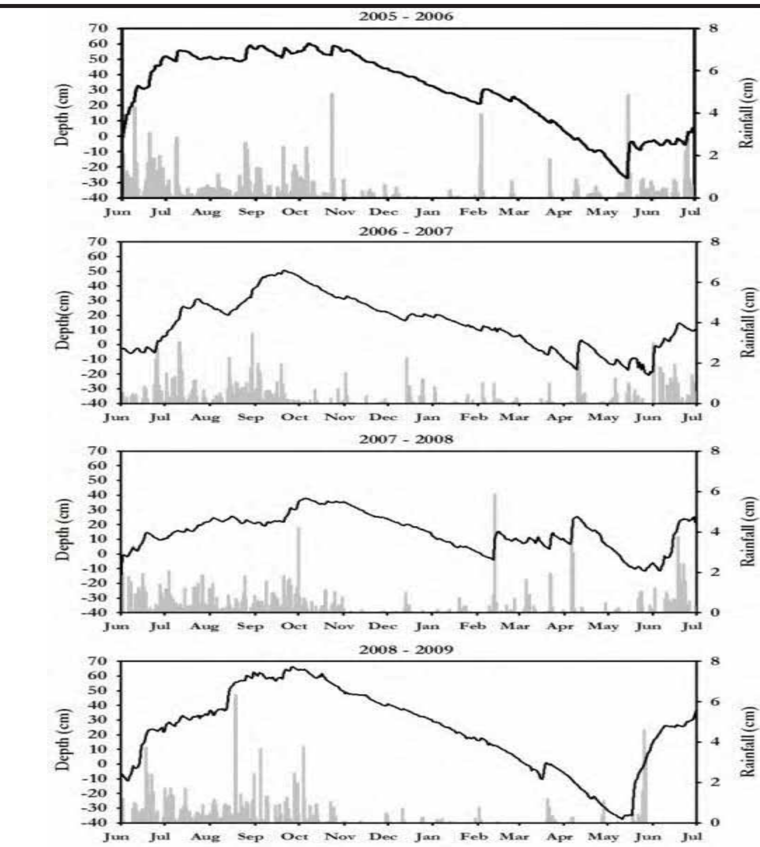


FIGURE 8-3. MEAN WATER DEPTH AND RAINFALL ACROSS GREATER EVERGLADES WETLANDS FOR FOUR WATER YEARS. MEAN DEPTH WAS CALCULATED USING EDEN. RAINFALL REPRESENTS THE MEAN OF 13 GAGES THROUGHOUT THE SYSTEM.

FIGURE **8-3**: MEAN WATER DEPTH AND RAINFALL ACROSS GREATER EVERGLADES WETLANDS FOR FOUR WATER YEARS

Note: 1. Mean depth was calculated using EDEN
2. Rainfall represents the mean of 13 gages throughout the system

The El Nino Southern Oscillation (ENSO) phased in and out with both El Nino and La Nina years during the four WYs. However, the AMO has been in its warm phase since the late 1990s. The AMO is a long-term oscillation in the sea surface temperatures of the North Atlantic Ocean. Climatologists believe that multi-decadal periods of warming and cooling of the North Atlantic Ocean's surface waters ultimately affect precipitation patterns across Florida. The current warm phase of the AMO cycle allows the inter-tropical convergence zone (ITCZ) to move northward. This produces more rain over Florida primarily during the wet season. Also, during warm phases of the AMO, more tropical storms strengthen into major hurricanes than during the cool phase.

8.1.3.1 Water Year 2006 (5/1/2005-4/30/2006)

An El Nino event had a predominant influence on rainfall during WY 2006. Because of El Nino and the weak Bermuda High pressure system in the Atlantic Ocean, most hurricanes that developed in the tropical Atlantic Ocean turned northward well east of Florida. As a result, hurricane activity contributed little to the high wet season rainfall. However, Tropical Storm Ernesto moved across the southern and central portions of the state during late August.

High rainfall and water levels during the wet season continued into the early dry season until mid-February. This pattern was followed by a rapid and persistent recession during the subsequent dry season in WY 2006. Low rainfall after mid-February resulted in rapidly falling water levels through June without hydrologic reversals. This dry season marked the beginning of a regional drought as defined by Abtew (2008, 2009) that resulted in widespread drydown throughout the Everglades. Rapidly falling water levels resulted in extreme low stages at the beginning of the 2007 wet season.

8.1.3.2 Water Year 2007 (5/1/2006-4/30/2007)

El Nino was in effect during the beginning of the WY 2007. As El Nino weakened during the spring and summer months, Atlantic hurricane tracts normalized with approximately 50 percent of the tropical Atlantic storms moving through the Caribbean into the Gulf of Mexico or making landfall in Mexico and the other 50 percent turning northward near Florida. However, no tropical system directly affected southern or central Florida during 2007.

The regional drought continued in this WY (Abtew 2008, 2009), with relatively low rainfall and water levels during the wet and dry seasons and a strong water level recession during the dry season. Low stages at the beginning of the wet season, followed by low rainfall during the wet and dry seasons, resulted in extremely low water levels in April 2007 that continued through June in the WCAs. Suspended water inflows from WCA 3 resulted in extremely low water levels from April through June in the northern Everglades National Park. Marsh drying in many areas of the Everglades during March and April was followed by a rainfall-driven hydrologic reversal in Everglades National Park in mid-April.

8.1.3.3 Water Year 2008 (5/1/2007-4/30/2008)

La Nina developed during the fall of 2007 and extended into 2008. Regional drought conditions continued throughout most of the Everglades during the wet season and early dry season until late February during WY 2008. Loxahatchee NWR and WCA 2A were exceptions, where stages equaled or exceeded means throughout the year. High rainfall events during late February, March and April resulted in rising water to above average stages in all of the WCAs, marking the end of the drought. Heavy rainfall and rising water levels resulted in a hydrologic reversal during March that disrupted wading bird nesting and signaled the end of the regional drought. Continuation of minimal discharges from WCAs to Everglades National Park delayed the rise in water levels in the Park until the following wet season.

8.1.3.4 Water Year 2009 (5/1/2008-4/30/2009)

Hurricane season during this WY was very active for the Caribbean Sea, Gulf of Mexico, and Atlantic Ocean. Tropical Storm Faye was a slow moving system that made landfall in southwestern Florida, moved northwest across central Florida, exited the state, and then made landfall again in northeastern Florida. Tropical Storm Faye averaged five to ten inches of rainfall along its path. However, even with this tropical rainfall, the year ended below normal for the Everglades.

Relatively high rainfall and water levels during the wet season were followed by rapid and continuous water level recession during the dry season in WY 2009. High rainfall resulted in above average water levels throughout the wet season. Extremely low rainfall resulted in a rapid decline in water levels throughout the dry season without hydrologic reversals.

8.1.3.5 Loxahatchee National Wildlife Refuge

In contrast to other areas of the Everglades, water stages in the Loxahatchee NWR during the last four WYs approximated long-term averages, even during the two years of regional drought. This was achieved through proactive regional water management efforts that balanced ecosystem and water supply needs.

8.1.4 Overview

Despite the highly contrasting rainfall and water depth patterns, the overall distribution of hydroperiods across the Everglades showed common patterns each WY that resulted from compartmentalization by the C&SF Project system of levees and canals. ***Figure 8-4*** to ***Figure 8-7*** display the hydroperiods for WYs 2005 through 2008; 2009 data was not available for inclusion in this report. Hydroperiods are back-calculated from the last day of the wet season, which is October 31, of each WY. They represent the period of marsh flooding since the last dry down event.

The system of levees of the C&SF Project produced pools of water with multi-year hydroperiods across both ridges and sloughs in areas located upstream of the levees in each WCA, and drained marshes downstream of the levees. The contrast between upstream pooling and downstream drainage was most widespread and extreme in WCA 3A and Everglades National Park.

An extensive pool of surface water persisted in WCA 3A upstream of L-29 and L-67, where the hydroperiod exceeded four years. The area of prolonged hydroperiods expanded during years when rainfall and water levels were relatively high during the wet season and contracted during dryer years. However, the pool persisted even during the most extreme drought conditions in 2007.

Hydroperiods were constrained to less than one year throughout most of Everglades National Park, including the major slough systems of the Shark Slough complex and Lostman's Slough. For example, discharges through inflow gate S12D produced hydroperiods exceeding one year in length immediately downstream of the structure, to the west of the historic flow corridor of northeastern Shark River Slough. Other than the area immediately south of S12D, hydroperiods in Everglades National Park exceeded one year in length only in pools of water above the marsh-mangrove interface of Lostman's and Shark Sloughs. Ground surface elevations produced by EDEN indicate that those pools form in topographic depressions that occur upstream of higher elevation ridges along the marsh-mangrove interface.

In addition to the water depth and hydroperiod maps, EDEN data can also be used to display surface water elevations across the Greater Everglades. An example of these maps is shown in ***Figure 8-8.*** Sheetflow directions in the Everglades marshes depicted in ***Figure 8-8*** are perpendicular to surface water elevation contours, and EDEN maps such as these can therefore be used to infer flow directions in the marsh. Flow directions are important in the Everglades because the distinct orientation and physiography of the ridge-slough-tree island mosaic developed in response to the historic flow patterns. An examination of the contours in these representative maps shows that surface water flow directions in many areas of the Greater Everglades are not aligned with the historic flow directions or the ridge-slough orientation. For example, in WCA 3B and Northeast Shark Slough in Everglades National Park, a strong west-east gradient in surface water elevations is revealed by the EDEN data. Water in these areas flows from west to east along the gradient, towards the urban areas of Miami-Dade and Broward counties. The gradient persists in wet and dry seasons. Restoring the flow directions in these areas to match the historic north-south directions is a pre-requisite for restoring the ridge-slough-tree island mosaic.

The EDEN surface water elevation maps can also be used to investigate the influences of flow barriers and other structures that modify sheetflow patterns in the marshes, such as Tamiami Trail (U.S. Highway 41) and I-75. Discontinuities in the water surface elevation gradient are typical across these boundaries, and the sharp drop in surface water elevations across Tamiami Trail is indicative of a barrier to flow. In a natural setting, the slope of the marsh water surface approximates the slope of the land surface without sharp discontinuities. The different management strategies applied north and south of the Tamiami Trail means these differences in water surface elevations persist even during very low rainfall periods such as the 2008-09 dry season. During this period, water levels in Everglades National Park just south of Tamiami Trail were significantly lower than those found to the north in WCA 3A. Water levels in WCA 3B were also significantly lower than those found in WCA 3A due to the influence of the L-67A and

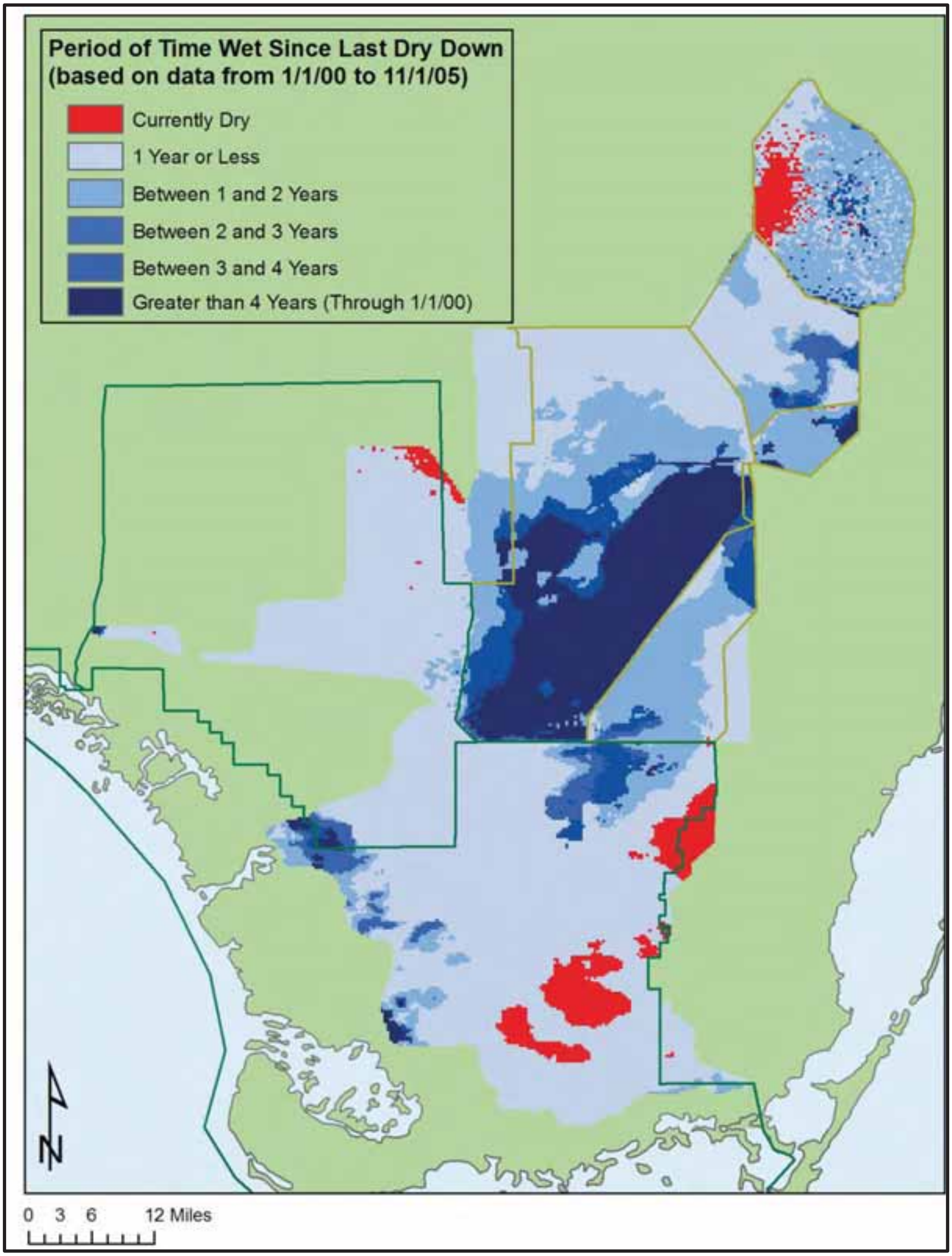


FIGURE 8-4: 2005 HYDROPERIOD MAP REPRESENTING PERIOD OF TIME SINCE LAST DRY DOWN FOR THE GREATER EVERGLADES COMPUTED FROM EDEN DATA BEGINNING IN 2000 THROUGH NOVEMBER 1, 2005

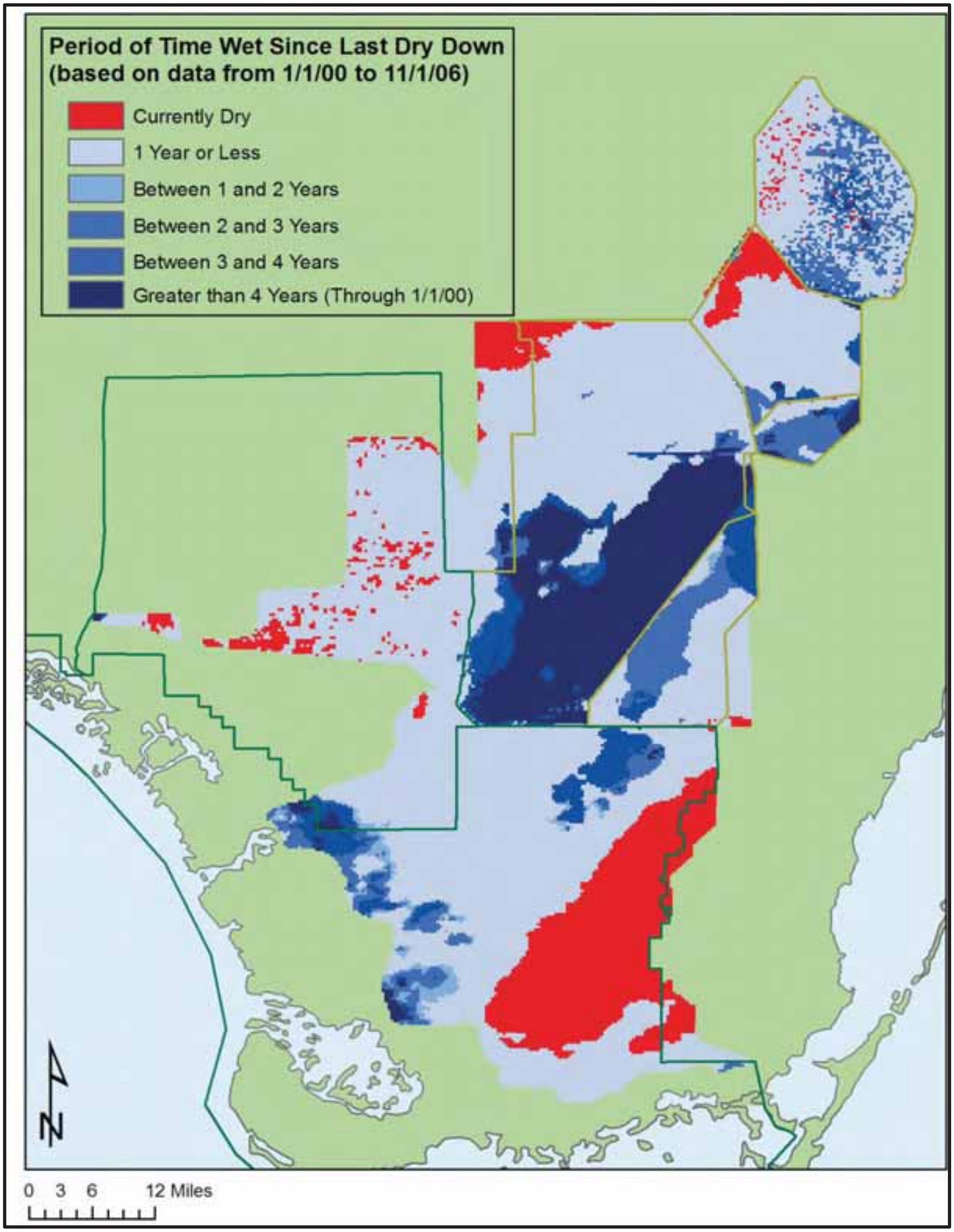


FIGURE 8-5: 2006 HYDROPERIOD MAP REPRESENTING PERIOD OF TIME SINCE LAST DRY DOWN FOR THE GREATER EVERGLADES COMPUTED FROM EDEN DATA BEGINNING IN 2000 THROUGH NOVEMBER 1, 2006

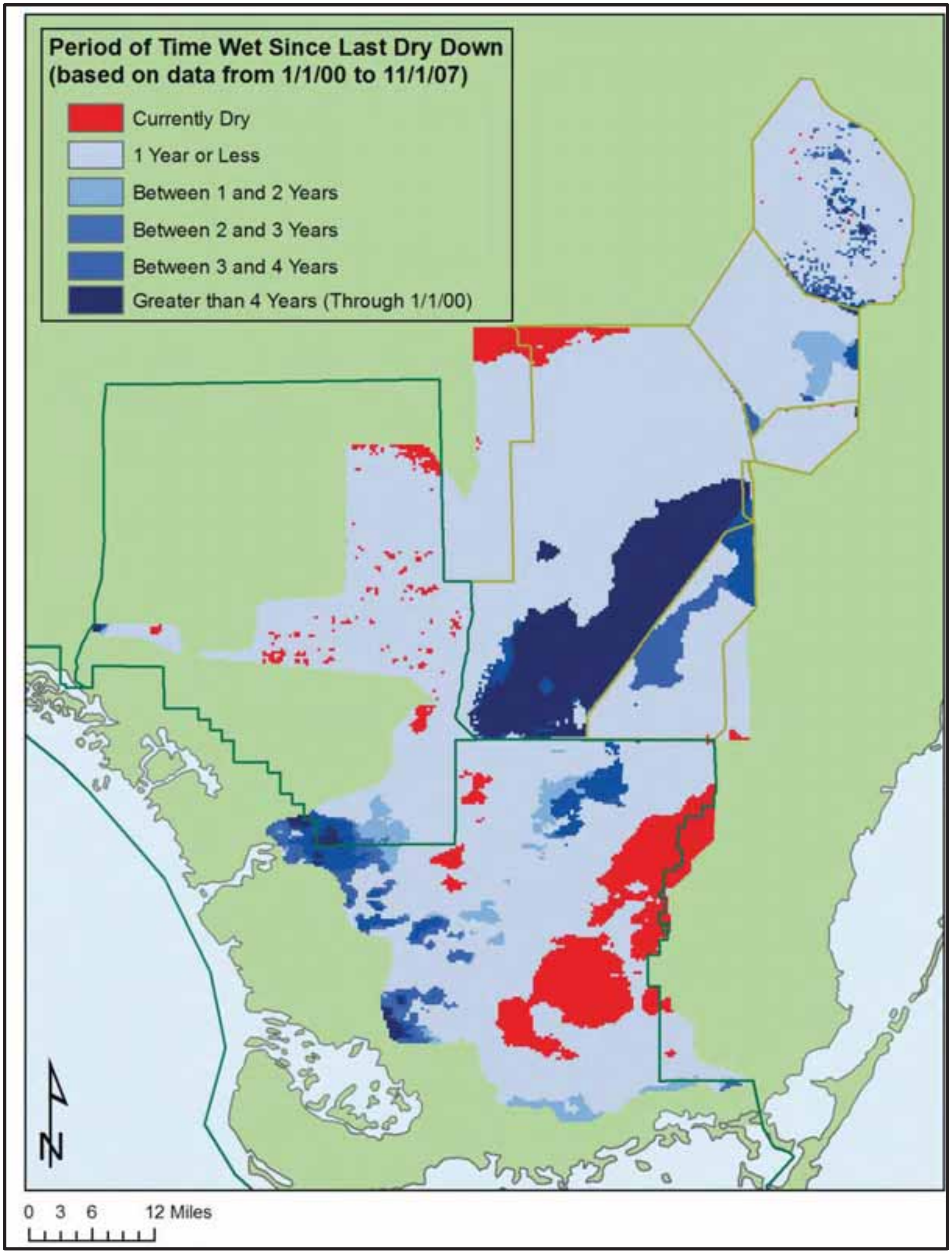


FIGURE 8-6: 2007 HYDROPERIOD MAP REPRESENTING PERIOD OF TIME SINCE LAST DRY DOWN FOR THE GREATER EVERGLADES COMPUTED FROM EDEN DATA BEGINNING IN 2000 THROUGH NOVEMBER 1, 2007

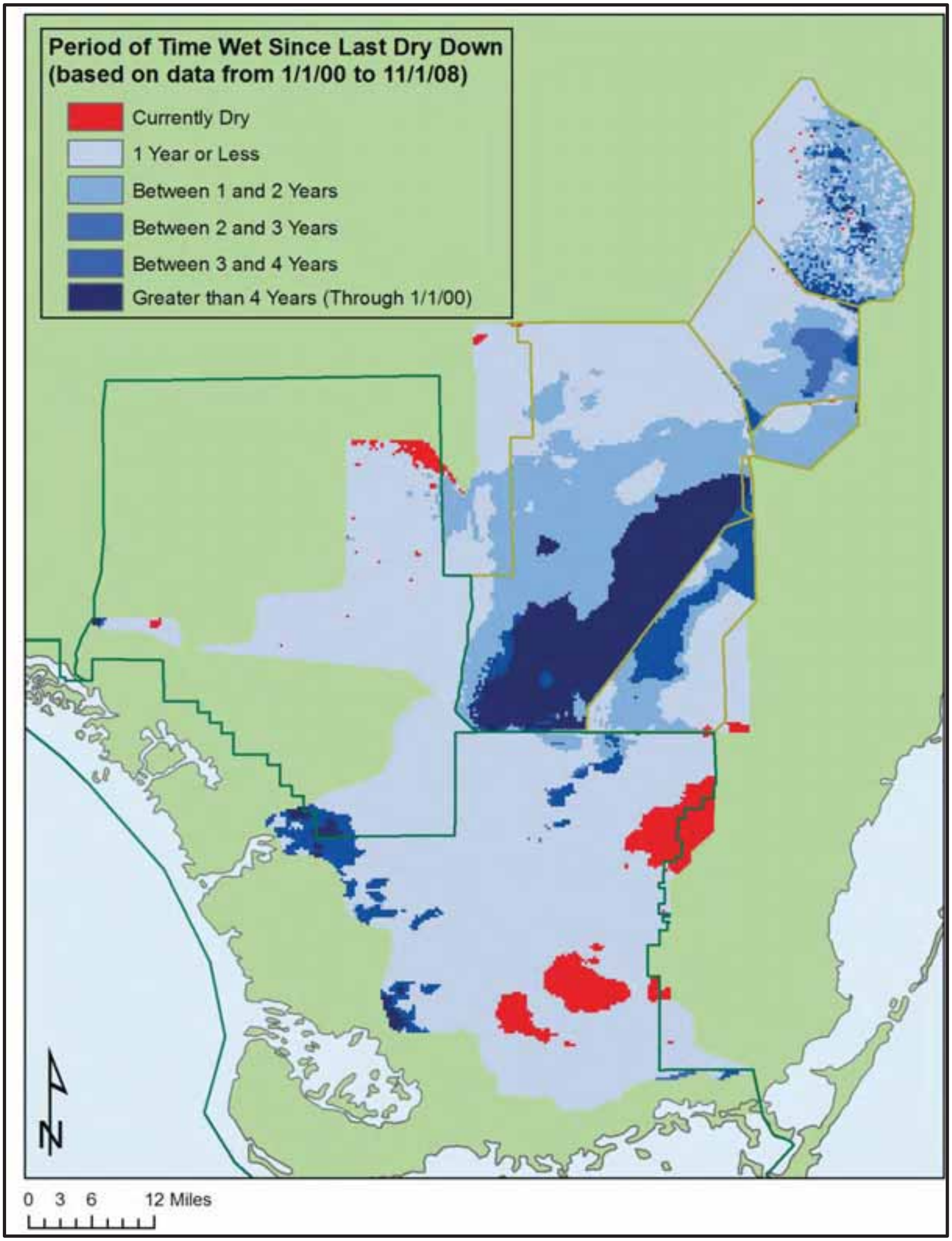


FIGURE 8-7: 2008 HYDROPERIOD MAP REPRESENTING PERIOD OF TIME SINCE LAST DRY DOWN FOR THE GREATER EVERGLADES COMPUTED FROM EDEN DATA BEGINNING IN 2000 THROUGH NOVEMBER 1, 2008

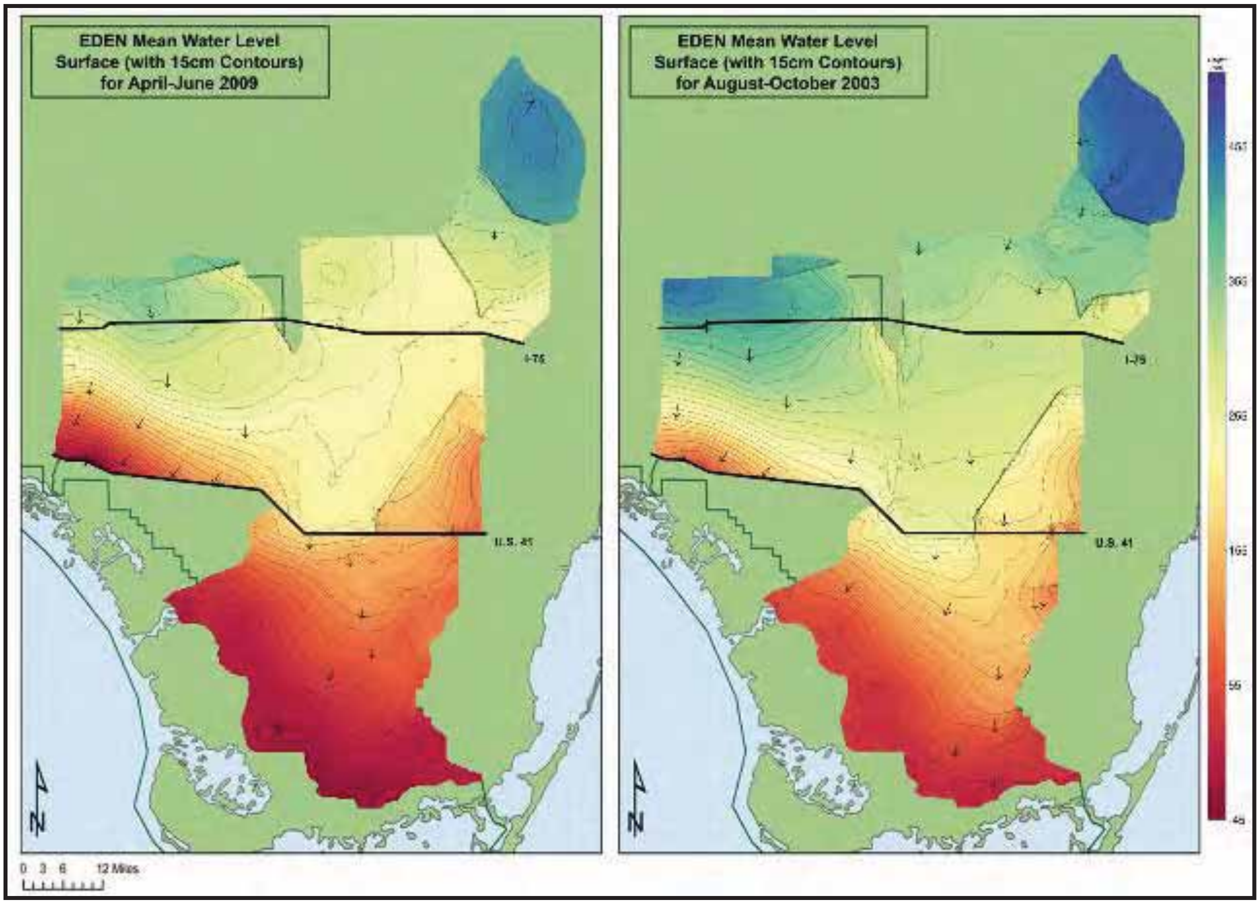


FIGURE 8-8. REPRESENTATIVE CONTOURS OF SURFACE WATER ELEVATIONS IN THE GREATER EVERGLADES DURING LOW (LEFT PANEL) AND HIGH (RIGHT PANEL) RAINFALL PERIODS

C levees. As a result, the drought conditions in Everglades National Park and WCA 3B brought about by the low rainfall in 2009 were exacerbated by the Tamiami Trail and L-67 flow barriers. The discontinuities across the Tamiami Trail and the L-67 levees persist even under high rainfall periods, such as the 2003 wet season. As a result, the water levels in WCA 3A were relatively much higher than those found in other areas. The persistent high water levels in WCA 3A are correlated with the decline in habitat quality, including the loss of tree islands. A restored Everglades ecosystem will not exhibit strong discontinuities in the water surface, and the degree to which these discontinuities are removed either through the physical removal of the barriers or through AM schemes can be considered a measure of restoration success. The degree to which flow directions are also returned to historic directions and the orientation of the ridge slough-tree island mosaic can also be considered an indicator of restoration success.

Strong discontinuities in water surfaces are apparent in the dry season, especially across Tamaimi Trail (U.S. Highway 41). The direction of flow denoted by the black arrows indicate that in many areas, particularly in eastern WCA 3B and northeast Everglades National Park, sheetflow is not aligned with the more north-south orientation of the ridge-slough-tree-island mosaic in these areas.

8.1.5 Summary

8.1.5.1 Assessment of Working Hypotheses and Status and Trends

Hydroperiod and water depth maps based on EDEN data during the past four WYs graphically display how compartmentalization and water regulation schedules of the WCAs have eliminated sheet flow and changed water depth and flow direction patterns throughout most of the Everglades. These effects have been particularly severe and widespread in WCA 3A and Everglades National Park. In the extensive pool of water in WCA 3A, hydroperiods longer than four years persisted even during extreme drought conditions. Even during the wettest years, water deliveries from WCA 3 were inadequate to maintain multi-year hydroperiods in the major slough systems of Everglades National Park.

The hydrologic conditions in the existing system contrast strongly with two independent studies of the pre-drainage system: McVoy et al (in press) and Marshall et al. (2009) both characterize the pre-drainage system as exhibiting greater water depths and longer hydroperiods (with the possible exception of the southernmost portions of the current WCA 1, 2A, and 3A). Marshall et al. 2008 concludes freshwater levels and hydroperiods in major sloughs of Everglades National Park were approximately 0.15 meter higher and two to four times greater, respectively, on the average compared to today's values. When the historical analysis and paleoecological studies (e.g., Willard et al., 2001, 2007; Winkler et al., 2001; Saunders et al., 2006; Bernhardt and Willard, 2009) overlap in time, the two approaches corroborate each other.

Marshall et al. (2008) extends the discussion of hydroperiod and depth by estimating pre-drainage fresh water volumes delivered to the wetlands and estuaries of Everglades National Park to be two and a half to four times greater than modern day flow, with the largest deficit of delivery in the existing system occuring during the dry season. The altered flow patterns in the current system (***Figure 8-8***) are also indicative of flow blockage to Everglades National Park caused by the C&SF Project infrastructure and water management.

8.1.5.2 Key Management Recommendations

The core area of the Everglades that retains potential for restoration of sheet flow and water depth patterns in a system of large spatial extent, undivided by levees and canals, and connected to downstream mangrove estuaries includes WCA 3A, WCA 3B, the eastern portion of Big Cypress National Preserve, and Everglades National Park (RECOVER, 2009).

Hydrology monitoring results to date identify high priority restoration recommendations that apply to the return of connectivity and sheet flow to this core area of the Everglades. Restore rainfall-driven volume, timing and distribution of sheet flow through WCA 3A and WCA 3B into Everglades National Park to produce water depths, hydroperiods, and flow patterns that are consistent with the best understanding of how the pre-drainage system responded to seasonal and interannual variability in rainfall. This includes redistribution of sheet flow through the natural flow corridors of the Shark Slough complex and Lostman's Slough in Everglades National Park in order to restore multi-year hydroperiods in slough systems of the Park. It also requires elimination of unnatural pooling of water in WCA 3A above the L-29 and L-67 levees.

These recommendations support the originally strategy for Everglades restoration described in Section 601 of the WRDA 2000 (U.S. Congress, 2000). Two key expansions of the original recommendations found in this report are the restoration of historical water surface slopes and flow directionality, and identification of the core area of Greater Everglades Wetlands where hydrological and ecological restoration objectives are most likely to be achieved.

8.2 WADING BIRD NESTING IN RELATION TO AQUATIC FAUNA FORAGE BASE

8.2.1 Introduction and Background

The collapse of wading bird nesting colonies in the southern Everglades is attributed to declines in wet season production and dry season concentration of aquatic prey organisms consisting primarily of marsh fish and crayfish as depicted in the CEM (***Figure 8-9***). The overarching hypothesis for this assessment 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.

8.2.1.1 Wet Season Prey Production

The wet season is the period of production of aquatic prey populations. Wet season populations of aquatic prey organisms are directly related to hydroperiod (defined as the period of marsh flooding since the last dry down by Turner et al., 1999; Trexler et al., 2005; Trexler and Goss, 2009). Although responses to hydrology are non-linear and species specific, existing monitoring demonstrates that aquatic prey biomass requires three or more years of standing water to fully develop (Trexler et al. 2004). Thus aquatic prey biomass at the end of each wet season results from net production during that wet season in combination with population build-up during prior years, both of which are the product of the antecedent hydroperiod.

8.2.1.2 Dry Season Prey Concentration

The dry season is a period of concentration of aquatic prey populations. The concentration of prey biomass is controlled by rates of water level recession. In order to support wading bird nesting, water level recession during the dry season must concentrate aquatic prey biomass into isolated pools of shallow water where the birds can forage effectively (Gawlik, 2002). Prey concentration is high system-wide when water level recession creates a progression of isolated pools moving across broad areas of marsh for extended periods of the dry season. Drying patterns may be interrupted by unseasonable rainfall events or regulatory water releases.

Hydrologic reversals are defined as interruptions in drying patterns that cause widespread reduction in wading bird nesting success, as a result of chick starvation and nest abandonment. High rates of water level recession and/or initially low water levels at the beginning of the dry season can cause marsh dry down prior to completion of wading bird nesting, which may also result in reduced nesting success.

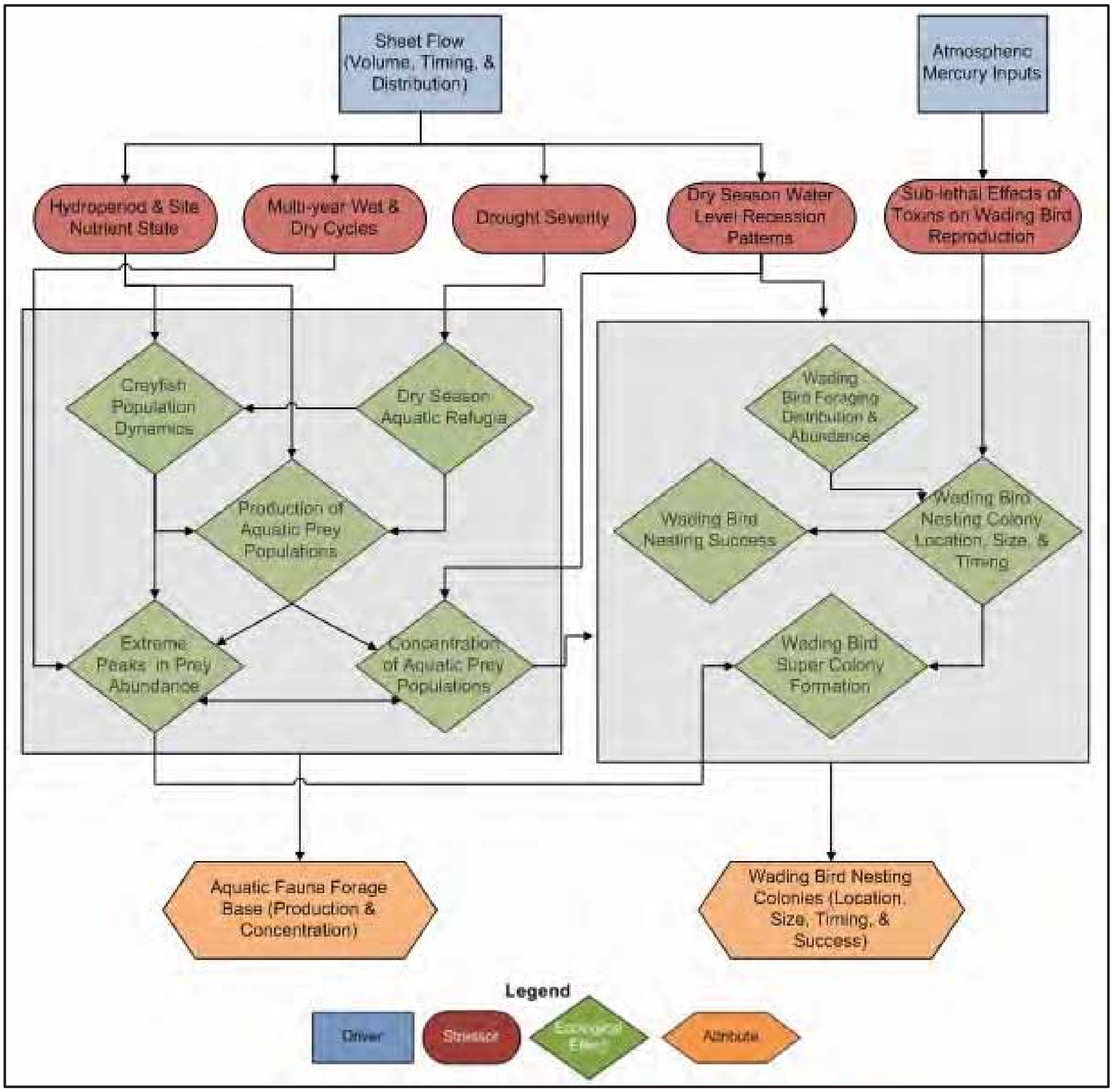


FIGURE 8-9. CONCEPTUAL ECOLOGICAL MODEL FOR WADING BIRD NESTING IN RELATION TO THE AQUATIC FAUNA FORAGE BASE

8.2.1.3 Exceptional White Ibis Nesting Events

White ibis (*Eudocimus albus*) nest in unusually large numbers after periods of drought in multi-year wet and dry cycles in the Everglades (Frederick and Ogden, 2001; 2003; Frederick et al., 2009). An exceptional nesting event is defined as exceeding the 70th percentile of all estimates of annual nestings for the period of record beginning in 1931. The mechanisms contributing to exceptional nesting after drought periods are hypothesized to involve pulses in prey populations, predatory fish population decline, and nutrient dynamics.

Assessment of wading bird predator prey interactions in this report is comprised of four contributions:

- Assessment of sequences of wet season hydroperiod, wet season aquatic prey production, dry season water level recession, dry season aquatic prey concentration, and wading bird nesting in the Everglades for four WYs: 2005-06, 2006-07, 2007-08 and 2008-09. Each WY runs from the official beginning of the wet season on May 1 until the end of the dry season of the following year on April 30.

- Synthesis of data supporting the lower trophic-level linkages between hydrology, periphyton, and aquatic fauna production in the causal network of wading bird predator-prey interactions is provided in the companion document *Monitoring Early and Late Dry Season Aquatic Fauna and Periphyton* prepared by Trexler and Gaiser (2009) (See 2009 SSR Companion Document 5A).

- Synthesis of data supporting the higher trophic-level linkages between hydrology, aquatic fauna production and concentration, and wading bird nesting in the mainland Everglades is provided in the companion document *Predator-Prey Interactions of Wading Birds and Aquatic Fauna Forage Base* prepared by Gawlik et al. (2009) (See 2009 SSR Companion Document 5B).

- Synthesis of data supporting the higher trophic-level linkages between hydrology, aquatic fauna production and concentration, and roseate spoonbill nesting in Florida Bay is provided in the companion document *Verification of the Predator-Prey Conceptual Model for Wading Birds and Aquatic Fauna Forage Base Using Data from Roseate Spoonbill Studies in Florida Bay* prepared by Lorenz et al. (2009) (See 2009 SSR Companion Document 5C).

Three performance measures have been developed based on the trophic relationship between wading birds and their prey. These performance measures focus on the following aspects of this relationship: 1) regional populations of fishes, crayfish, grass shrimp and amphibians, 2) wading bird foraging patterns on over-drained wetlands, and 3) wading bird nesting patterns. Documentation for these performance measures can be found at www.evergladesplan.org/pm/recover/perf_ge.aspx. Interim goals have been developed for Everglades aquatic fauna populations and system-wide wading bird nesting patterns. Documentation for these interim goals can be found at www.evergladesplan.org/pm/recover/igit_subteam.aspx.

8.2.2 Monitoring

8.2.2.1 Wet Season Production of Aquatic Prey Populations

Aquatic prey populations during the late wet season are monitored using one-square meter throw traps (Jordan et al., 1997). Throw trap samples are collected in 149 primary sampling units in freshwater marshes across the Everglades. The primary sampling units are located in landscape sampling units (LSUs) (***Figure 8-10***).

Of the 52 LSUs in ***Figure 8-10***, 32 were identified as feasible for throw trap sampling by Trexler (2004). In this report, aquatic prey populations are assessed in 21 LSUs in the WCAs and Everglades National Park where three or more primary sampling units were sampled during each year (***Table 8-1***).

Selection of primary sampling units follows guidelines from Philippi (2003, 2005) and is based on a spatially balanced recursive tessellation design (Stevens and Olsen, 2004). 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 environmental data (like hydrological characteristics, soil type, or elevation patterns) and then make statistically rigorous comparisons of the biological attributes between specific regions.

8.2.2.2 Dry Season Concentration of Aquatic Prey Populations

Aquatic prey biomass concentrated in isolated pools as water levels receded during the dry seasons of 2006 through 2009. These concentration events were monitored with a multi-stage sampling design that uses throw traps in a subset of the primary sampling units sampled during the wet season as a using a multi-stage sampling design (Cochran, 1977). Dry season prey concentrations were monitored each year in LSUs where surface water receded into isolated pools. The LSUs meeting the criteria that were sampled each dry season included five in 2006, nine in 2007, nine in 2008 and 12 in 2009. Refer to Gawlik and Botson (2008) for detailed sampling methods.

8.2.2.3 Wading Bird Nesting Colonies in the Mainland Everglades

Wading bird nesting colonies in Everglades National Park, the WCAs and Lake Okeechobee are monitored monthly each year between January and June using systematic aerial surveys (Frederick et al., 2006; Cook and Call, 2006). East-west oriented transects spaced 1.6 nautical miles apart are monitored at a flight altitude of 800 feet above ground level, with observers on both sides of the aircraft. Additional north-south oriented transects result in overlapping coverage under a variety of weather and visibility conditions. The method has been used continuously since 1986.

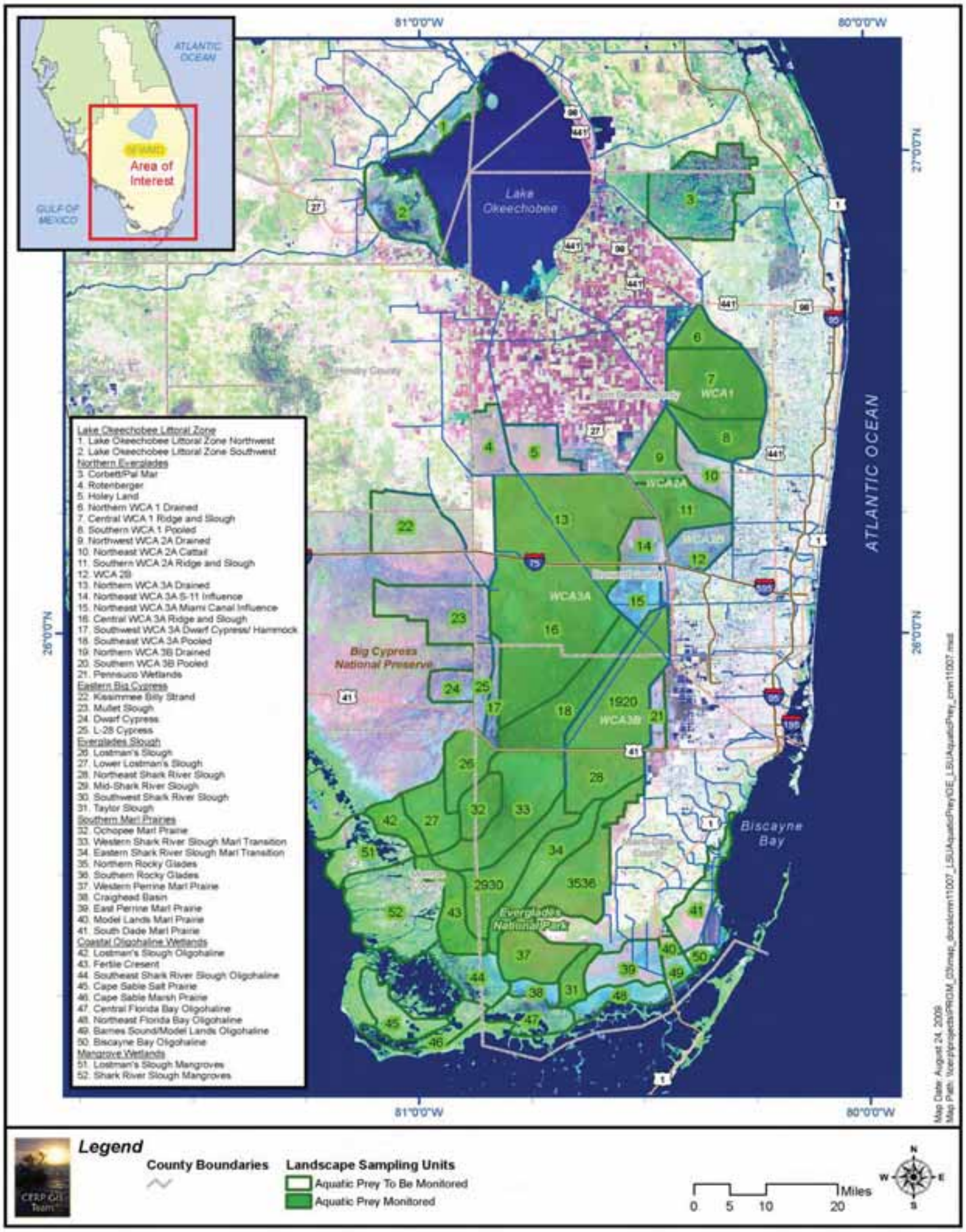


FIGURE 8-10. LANDSCAPE SAMPLING UNITS FOR THE GREATER EVERGLADES WETLANDS MODULE, UPDATED IN 2008; THOSE SHADED IN GREEN ARE USED TO ASSESS TEMPORAL AND SPATIAL DIFFERENCES IN AQUATIC PREY POPULATIONS IN THIS REPORT

TABLE 8-1. LANDSCAPE SAMPLING UNITS USED IN THE ASSESSMENT OF AQUATIC PREY POPULATIONS DURING LATE WET SEASONS OF 2005 THROUGH 2008

Water Conservation Areas	Everglades National Park
Loxahatchee NWR (WCA 1)	**Shark River Slough**
6 Northern WCA 1 (8,186 ha)	28 Northeast Shark River Slough (19,408 ha)
7 Central WCA 1 (32,561 ha)	29/30 Mid- and Southwest Shark River Slough (23,389 ha)
8 Southern WCA 1 (15,334 ha)	
WCA 2A	**Lostman's Slough**
9 Northwest WCA 2A (12,781 ha)	26 Lostman's Slough (14,181 ha)
11 Southern WCA 2A (17,856 ha)	27 Lower Lostman's Slough (14,177 ha)
WCA 3A	**Coastal Wetlands of Lostman's and Shark Sloughs**
13 Northern WCA 3A (61,880 ha)	42 Lostman's Slough Oligohaline (8,732 ha)
16 Central WCA 3A (77,049 ha)	43 Fertile Crescent (15,063 ha)
18 Southeast WCA 3A (27,181 ha)	
	Shark River Slough/Marl Prairie Transition
WCA 3B	33 Western Shark Slough Transition (36,102 ha)
19/20 WCA 3B (39,885 ha)	34 Eastern Shark Slough Transition (19,797 ha)
	Southern Marl Prairies
	32 Ochopee Marl Prairie (10,653 ha)
	35/36 Rocky Glades (28,253 ha)
	37 Western Perrine Marl Prairie (19,401 ha)
	Taylor Slough
	31 Taylor Slough (6,672 ha)

8.2.3 Results

8.2.3.1 Wet Season Production of Aquatic Prey Populations

Aquatic prey biomass within the WCAs and Everglades National Park are reported for the late wet seasons (October-November) of 2005, 2006, 2007 and 2008 in ***Table 8-2*** Mean biomass values in g/m² wet weight include slough crayfish (*Procambarus fallax*), Everglades crayfish (*Procambarus alleni*), marsh fishes and grass shrimp (*Palaemonetes paludosus*) combined. For incidences of high biomass based on temporal patterns, percent composition of crayfish species, fish and grass shrimp is given.

TABLE 8-2. AQUATIC PREY BIOMASS DURING LATE WET SEASONS OF 2005 THROUGH 2008 IN LANDSCAPE SAMPLING UNITS OF THE WATER CONSERVATION AREAS AND EVERGLADES NATIONAL PARK

Landscape Sampling Units	Biomass (g/m^2 wet weight)			
	2005	**2006**	**2007**	**2008**
Water Conservation Areas				
Northern WCA 1 Drained	3.3	1.5	2.2	7.4
Central WCA 1 Ridge and Slough	5.7	4.7	8.6	7.8
Southern WCA 1 Pooled	1.6	2.8	7.1* *P. fallax* 42% *P. alleni* <1% fish 44% shrimp 14%	5.0* *P. fallax* 28% *P. alleni* 0% fish 60% shrimp 13%
Northwest WCA 2A Drained	1.2	2.0	2.7	4.2
Southern WCA 2A Ridge and Slough	2.2	1.0	2.0	3.5
Northern WCA 3A Drained	2.0	2.7	2.3	4.5
Central WCA 3A Ridge and Slough	1.6	1.4	6.3* *P. fallax* 44% *P. alleni* 6% fish 43% shrimp 8%	5.0* *P. fallax* 34% *P. alleni* 4% fish 47% shrimp 16%
Southeast WCA 3A Pooled	2.5	3.1	4.3	4.7
WCA 3B	2.7	2.7	4.8	3.5
Everglades National Park				
Northeast Shark River Slough	2.1	1.6	3.7	3.6
Middle and Southwest Shark River Slough	1.6	4.6* *P. fallax* 7% *P. alleni* 18% Fish 61% Shrimp 14%	3.9* *P. fallax* 22% *P. alleni* 30% fish 36% shrimp 11%	6.9* *P. fallax* 17% *P. alleni* 32% fish 35% shrimp 16%
Lostman's Slough	20.1 * *P. fallax* 0% *P. alleni* 85% Fish 14% shrimp 1%	0.4	8.4* *P. fallax* 24% *P. alleni* 37% fish 36% shrimp 4%	9.2* *P. fallax* 0% *P. alleni* 88% fish 12% shrimp <1%
Lower Lostman's Slough	3.9	2.8	5.5	10.2* *P. fallax* 1% *P. alleni* 74% fish 20% shrimp 4%
Lostman's Slough Oligohaline	0.5	0.1	4.2* *P. fallax* 8% *P. alleni* 37% fish 53% shrimp 2%	2.6

Landscape Sampling Units	Biomass (g/m^2 wet weight)			
	2005	2006	2007	2008
Fertile Crescent	2.6	1.8	4.8* *P. fallax* 7% *P. alleni* 70% fish 21% shrimp 1%	5.3* *P. fallax* 14% *P. alleni* 40% fish 41% shrimp 5%
Western Shark River Slough Marl Transition	3.8	2.2	1.4	6.4* *P. fallax* 2% *P. alleni* 68% fish 26% shrimp 4%
Eastern Shark River Slough Marl Transition	1.7	2.0	1.9	9.9* *P. fallax* 1% *P. alleni* 82% fish 17% shrimp 1%
Ochopee Marl Prairie	9.9 * *P. fallax* 0% *P. alleni* 63% Fish 32% shrimp 4%	2.8	no data dry conditions	1.8
Rocky Glades	4.2	2.0	1.0	2.6
Western Perrine Marl Prairie	4.0* *P. fallax* <1% *P. alleni* 84% Fish 16% shrimp 0%	2.2	1.3	1.6
Taylor Slough	2.1	1.4	1.3	0.9

* LSU had a high biomass based on temporal patterns. Biomass was significantly highest ($P<0.05$) during sampling years in comparison to other years within a given LSU.

8.2.3.2 Temporal Patterns in Aquatic Prey Biomass

Temporal differences in prey biomass are based on pair-wise comparisons among the four sampling years for each LSU. Biomass in a LSU is ranked as high during a particular year if the mean is significantly higher ($p<0.05$) in comparison to means in the same LSU during other sampling years. Biomass within a LSU is ranked as low during a particular year if the mean is significantly lower ($p<0.05$) than high means in the same LSU during other sampling years. ***Figure 8-11*** identifies LSUs where year-to-year variations in prey biomass were significant, and years when biomass was high or low in those LSUs. For incidences of high prey biomass based on temporal patterns, percent composition of marsh fishes, crayfish and grass shrimp is given in ***Table 8-2***.

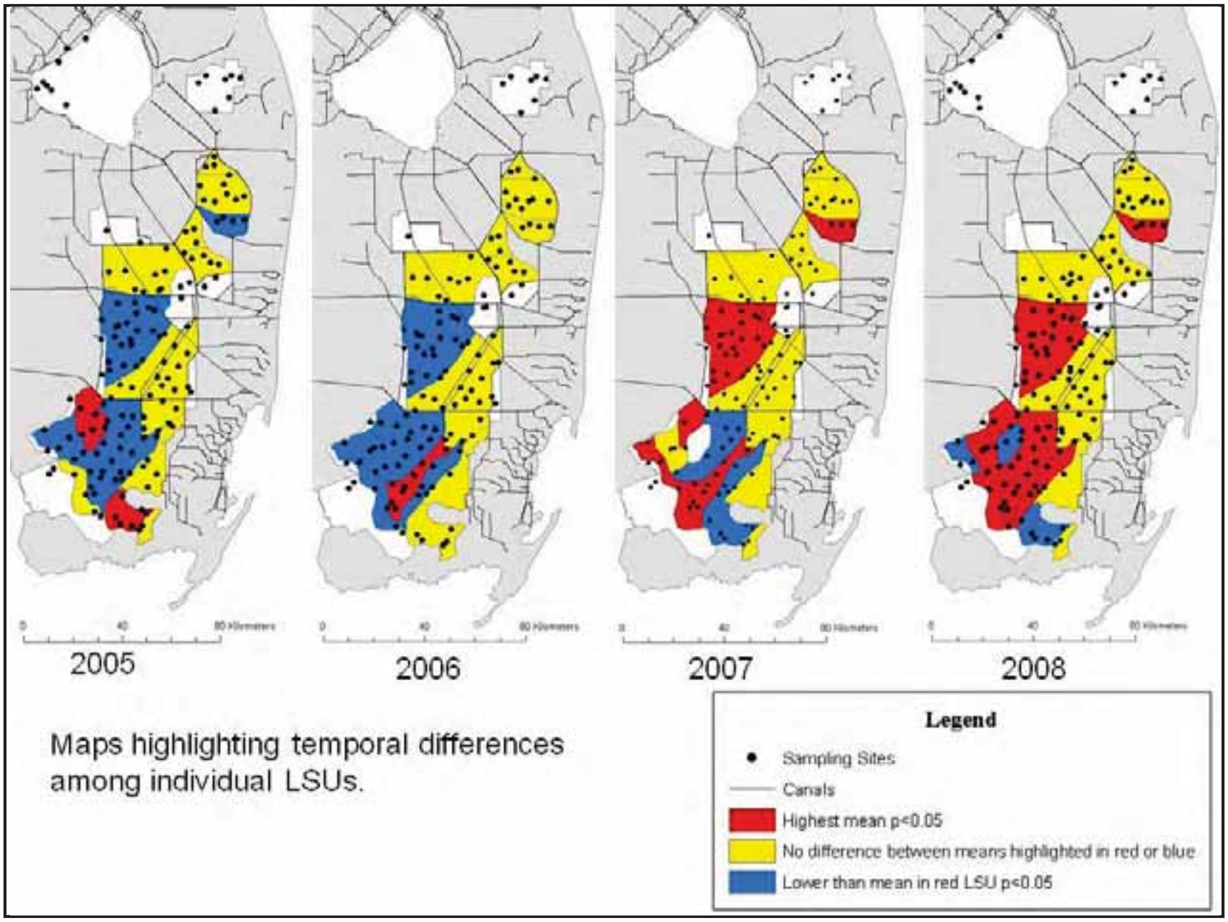


FIGURE 8-11. TEMPORAL DIFFERENCES IN TOTAL PREY BIOMASS BETWEEN SAMPLING YEARS FOR EACH LANDSCAPE SAMPLING UNIT IN THE WATER CONSERVATION AREAS AND EVERGLADES NATIONAL PARK

2005 Wet Season. Prey biomass at the end of the 2005 wet season was low throughout much of the Everglades. Incidences of high biomass occurred in only three LSU's: Lostman's Slough, Ochopee Marl Prairie, and Western Perrine Marl Prairie of Everglades National Park. In each case, Everglades crayfish accounted for most of the high prey biomass.

2006 Wet Season. Prey biomass at the end of the 2006 wet season remained low in much of the Everglades. An incidence of high biomass occurred in only one LSU: Central and Southwest Shark River Slough. In that case, marsh fishes accounted for most of the high prey biomass.

2007 Wet Season. Prey biomass at the end of the 2007 wet season began to increase in LSUs where it had been low during previous years. Incidences of high biomass occurred in southern WCA 1, central WCA 3A, Central and Southwest Shark River Slough, Lostman's Slough, and the coastal oligohaline regions of Everglades National Park. The following taxonomic groups predominated in those cases:

- Marsh fishes and slough crayfish: Southern WCA 1, Central WCA 3A

- Marsh fishes and Everglades crayfish: Middle and Southwest Shark River Slough, Lostman's Slough and Lostman's Slough Oligohaline in Everglades National Park

- Everglades crayfish: Fertile Crescent in Everglades National Park

2008 Wet Season. Prey biomass at the end of the 2008 wet season continued to increase in areas of the Everglades where it had been low at the end of the wet season during previous years. Incidences of high prey biomass occurred in southern WCA 1, central WCA 3A, and most of Everglades National Park. The following taxonomic groups predominated in those cases:

- Marsh fishes and slough crayfish: Southern WCA 1, Central WCA 3A

- Marsh fishes and Everglades crayfish: Middle and Southwest Shark River Slough, Fertile Crescent in Everglades National Park

- Everglades crayfish: Lostman's Slough, Lower Lostman's Slough, Eastern and Western Shark Slough/Marl Transition in Everglades National Park

8.2.3.2.1 Aerial Coverage of High and Low Aquatic Prey Biomass Based on Temporal Patterns

In order to quantify the distribution of high and low biomass events across the Everglades during each wet season, the aerial coverage of LSUs with high and low aquatic prey biomass based on temporal patterns is calculated (***Table 8-3***). These LSUs are identified in ***Figure 8-10***.

TABLE 8-3. AERIAL COVERAGE OF LANDSCAPE SAMPLING UNITS WITH HIGH AND LOW AQUATIC PREY BIOMASS (HECTARES) DURING THE LATE WET SEASON

Year	High Biomass	Low Biomass
2005	44,235 ha	194,580 ha
2006	23,389 ha	195,754 ha
2007	153,748 ha	75,300 ha
2008	215,092 ha	38,786 ha

Key: ha hectares

8.2.3.3 Dry Season Concentration of Aquatic Prey Populations

Aquatic prey production during the wet season and concentration during the dry season were monitored in 34 combinations of LSUs and WYs (***Figure 8-12***). Aquatic prey biomass increased between wet and dry seasons in 31 of the 34 comparisons. Biomass increases between wet and dry seasons during the individual WYs averaged between three-fold and four-fold. These comparisons represent the first estimates of the concentration of aquatic prey biomass that was produced in the wet season into drying pools of water where wading birds forage during the wet season in the Everglades.

Two particularly high concentrations of prey biomass occurred during the 2006 dry season (***Figure 8-12***). A mean dry season biomass of 138 g/m^2 in southern WCA 2A was 62 times the biomass during the wet season, and a mean dry season biomass of 85 g/m2 in northern WCA 3A was 43 times the biomass during the wet season. These two estimates strongly skew the mean increase in biomass between wet and dry seasons during WY 2006**.** The significance of such unusually high prey biomass concentrations to wading bird foraging and nesting is under continued investigation.

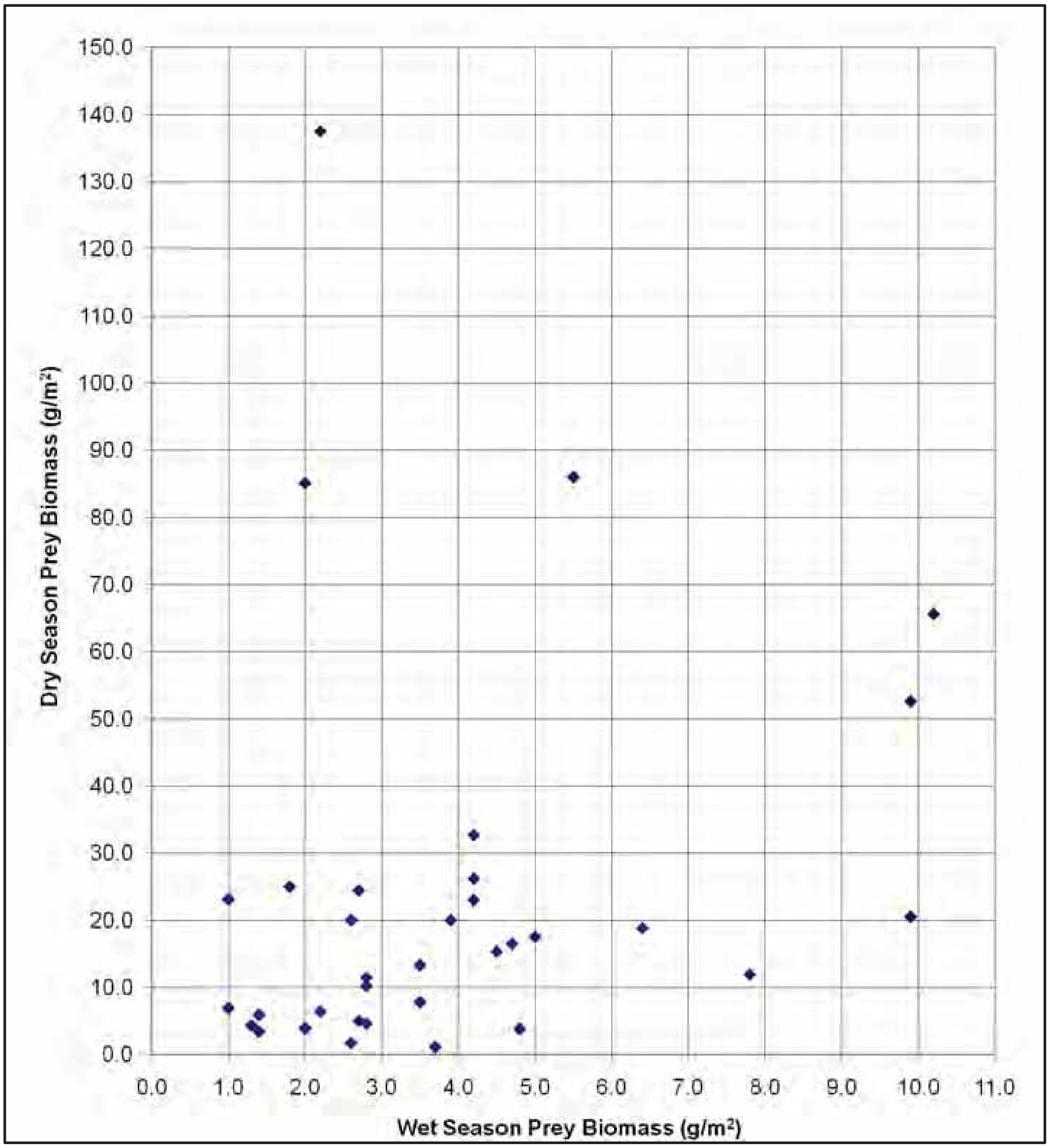


FIGURE 8-12. AQUATIC PREY BIOMASS DURING LATE WET SEASON AND IN DRY SEASON CONCENTRATION PATCHES FOR THE 34 COMBINATIONS OF LANDSCAPE SAMPLING UNITS AND WATER YEARS SAMPLED DURING BOTH WET AND DRY SEASONS

8.2.3.4 Wading Bird Nesting Colonies in the Mainland Everglades

8.2.3.4.1 White Ibis and Wood Stork Nest Numbers and Nesting Success

Table 8-4 reports wading bird nesting results for four nesting seasons, which begin in February and end in June of 2006, 2007, 2008 and 2009. Assessment of wading bird nesting focuses on white ibis and wood stork (*Mycteria americana*) as indicators of restoration success in the Everglades, based on recommendations of Frederick et al. (2009). These tactile feeders depend on concentrated prey in shallow water, and they have become proportionately less common than other wading bird species in the Everglades. The distribution and size of nesting colonies of white ibis and wood stork each year are mapped in ***Figure 8-13*** and ***Figure 8-14***.

Nest numbers of white ibis and wood stork during the four years are characterized as low to very high based on the percentile of all estimates of annual nestings for each species for the period of record beginning in 1931: low less than 50th percentile, moderate 50th to 69th percentile, high 70th to 89th percentile, very high more than 90th percentile. The high ranking is consistent with the 70th percentile threshold for exceptional white ibis nesting proposed by Frederick et al. (2009). Nesting success, which is the probability of any nest start fledging at least one chick 55 days in age, is characterized as high, low or not successful.

TABLE 8-4. NUMBERS OF NESTS OF WHITE IBIS, WOOD STORK AND TOTAL WADING BIRDS IN EVERGLADES NATIONAL PARK AND THE WATER CONSERVATION AREAS DURING THE NESTING SEASONS OF 2006 THROUGH 2009

Nesting Season	Everglades National Park	WCAs	Total Park + WCAs	Nesting Success
White Ibis				
2006	4,430	20,892	High 25,322	High
2007	1,458	19,203	High 20,661	High
2008	550	7,232	Low 7,782	High
2009	8,150	35,265	Very High 43,415	High
Wood Stork				
2006	1,124	190	Moderate 1,314	High
2007	340	0	Low 340	not successful
2008	145	0	Low 145	not successful
2009	2,602	1,461	Very High 4,063	High
Total Wading Birds				
2006	10,120	39,667	49,787	High
2007	3,281	32,032	35,313	Low
2008	950	15,204	16,154	Low
2009	15,432	57,564	72,996	High

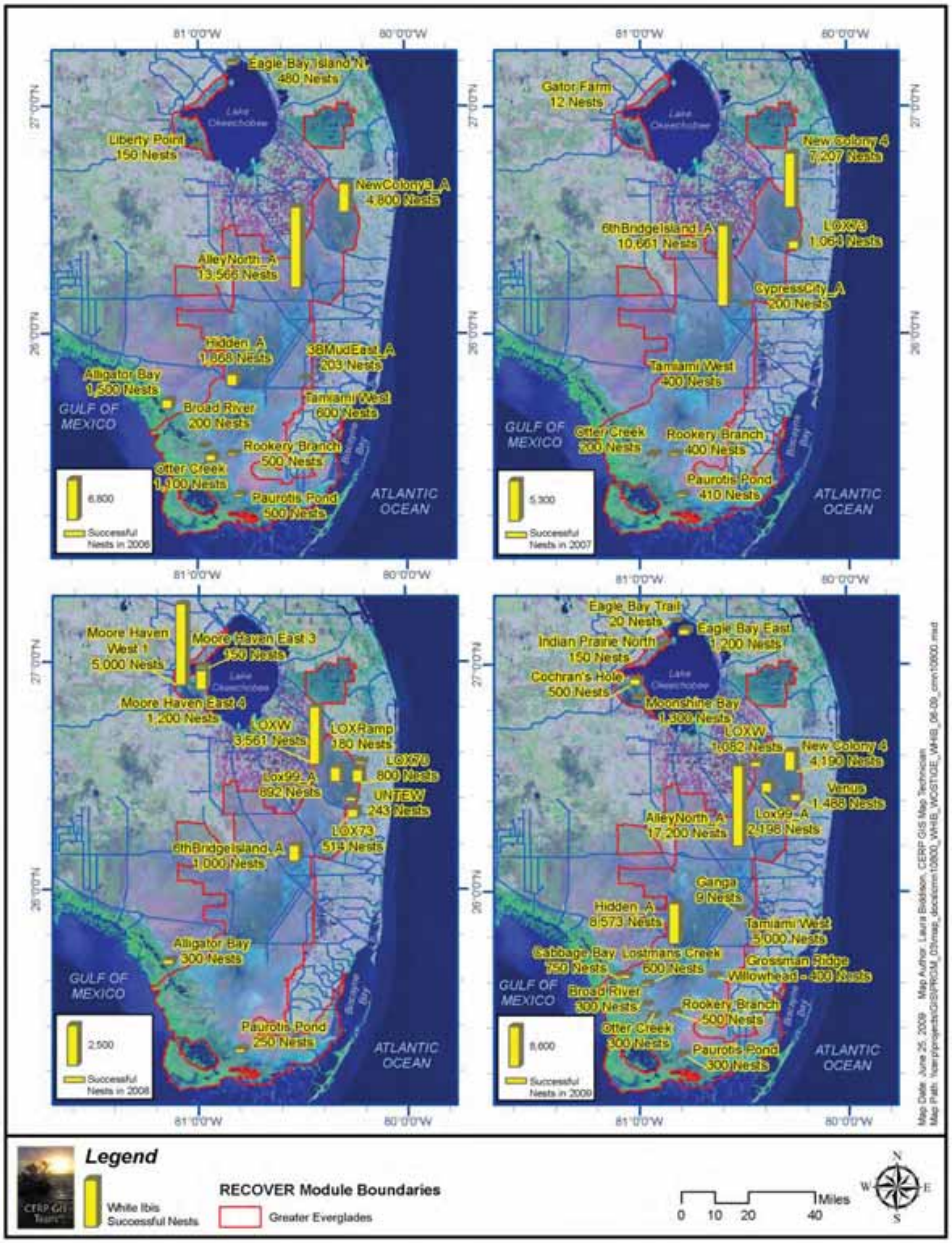


FIGURE 8-13. WHITE IBIS NESTING COLONY LOCATIONS AND NUMBERS OF NESTS DURING NESTING SEASONS OF 2006 THROUGH 2009

Note: Bar diagrams represent white ibis colonies with 50 or more nests.

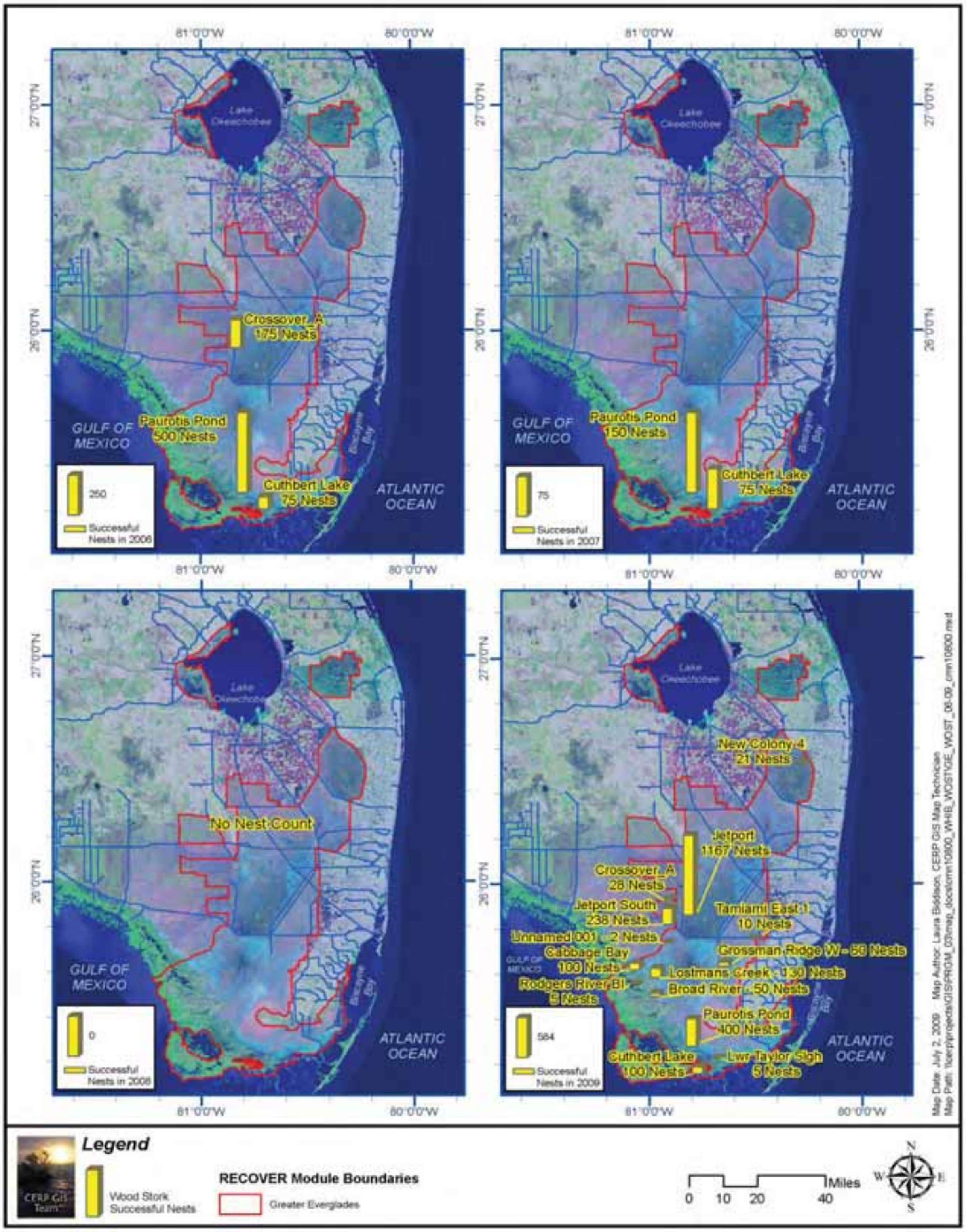


FIGURE 8-14: WOOD STORK NESTING COLONY LOCATIONS AND NUMBERS OF NESTS DURING NESTING SEASONS OF 2006 TO 2009

Note: Bar diagrams represent all wood stork colonies

8.2.3.4.2 Proportion of Wading Bird Nests in Water Conservation Areas versus Everglades National Park

Assessment of wading bird nesting also examines the proportion of all wading bird nests that occur in Everglades National Park in comparison to the WCAs during each nesting season. Based on the weight of evidence from the pre-drainage Everglades and from the comparable Usumacinta wetland system in the Yucatan, Frederick et al. (2009) propose a proportion of 70 percent of wading bird nesting in Everglades National Park in comparison to the WCAs as a conservative minimum for restored conditions in the Everglades.

The proportion of nesting in Everglades National Park during the last four years for all wading bird species combined ranged from 6 to 21 percent, in comparison to the desired 70 percent threshold proposed by Frederick et al. (2009) (***Table 8-5***).

TABLE 8-5. PROPORTION OF WADING BIRD NESTS THAT OCCURRED IN EVERGLADES NATIONAL PARK AND THE WATER CONSERVATION AREAS DURING EACH NESTING SEASON FROM 2006 THROUGH 2009

Nesting Season	Everglades National Park	WCAs
2006	20%	80%
2007	9%	91%
2008	6%	94%
2009	21%	79%

The very low proportion of wading bird nesting in Everglades National Park corresponds to hydroperiods of less than one year in slough systems where they were two to four times greater under pre-drainage conditions (Marshall et al., 2009). It also is consistent with the strong relationship of populations of wading bird prey organisms to hydroperiod (Turner et al., 1999; Trexler et al., 2005; Trexler and Goss, 2009). Proportion of nesting in Everglades National Park was higher during WYs with prolonged wet seasons and strong wading bird nesting. Restoration of wading bird nesting in the Everglades appears to depend on the return of hydrologic conditions favorable to nesting in Everglades National Park.

8.2.4 Summary

Assessment of Status and Trends in Context of Working Hypotheses: Wading Bird Nesting in the Context of Hydrology, Wet Season Prey Production and Dry Season Prey Concentration

Sequences of hydrology, wet season prey biomass, dry season prey concentration, and subsequent wading bird nesting in the Everglades during four WYs prior to the influence of CERP are examined. The combination of hydrology, prey populations and wading bird nesting was different each year.

8.2.4.1 Hydrology

The following hydrologic characteristics of the four WYs would be expected to influence aquatic prey populations and wading bird nesting based on the working hypotheses:

- Water levels dropped below the ground surface during each of the four WYs throughout extensive areas of the ridge and slough landscape, where multi-year hydroperiods appear to have occurred in the pre-drainage ecosystem.

- WYs 2006 and 2009 both had relatively high rainfall and water levels during the wet season, after which water level recession was prolonged and uninterrupted during the dry season. The two WYs differed in that 2009 followed a regional drought, while 2006 did not.

- WYs 2007 and 2008 both had relatively low rainfall and water levels during the wet season, after which water level recession was interrupted by reversals during the dry season. WY 2007 differed from 2008 in that widespread areas of marsh began drying in March, and water level reversals were limited to ENP. WY 2007 was the first of two consecutive years of regional drought, while 2008 was the second consecutive year.

- In contrast to most of the Everglades, water levels in Loxahatchee NWR remained at or above normal during WYs 2007 and 2008.

8.2.4.2 Wet Season Aquatic Prey Production and Dry Season Prey Concentration

The annual marsh drying that occurred during each of the four WYs would be expected to limit aquatic prey biomass at the end of each wet season, since three or more years are generally required for aquatic prey populations to fully develop (Trexler et al. 2004). Relatively low prey biomass of 2-3 g/m^2 during WYs 2006 and 2007 is consistent with the hypothesis that prey populations are limited by hydroperiod. However, higher prey biomass of 4-5 g/m^2 during the last two WYs is not consistent with that hypothesis.

Prey biomass increases during WYs 2008 and 2009 are consistent with the hypothesis that periodic drydowns to the soil surface results in pulses in crayfish populations. This type of response should not be confused with the undesirable effects of intense drought where large areas of marsh experience dried soils and water table over one meter below the surface (Acosta and Perry, 2001). Aquatic prey biomass of 4 g/m^2 during the last year of the drought represented a widespread increase across 154,000 ha of LSU's compared to past years. Aquatic prey biomass of 5 g/m^2 during the year following the drought represented a pulse across 215,000 ha of LSU's that was unprecedented compared to past years.

The prevalence of crayfish in areas where prey biomass increased during WYs 2008 and 2009 is consistent with the hypothesis that drydowns are associated with increases in crayfish populations. Crayfish comprised 54 percent of the aquatic prey biomass in the 154,000 ha of population increase during the 2007 wet season, and 61 percent of the aquatic prey biomass in the 215,000 ha of population increase during 2008.

Aquatic prey biomass produced during the wet season resulted in increased threefold to more-than-fourfold higher densities during the dry season as surface water receded into isolated pools during the four WYs.

8.2.4.2.1 Wading Bird Nesting in the Context of Hydrology, Wet Season Prey Production, and Dry Season Prey Concentration

Magnitude and success of wading bird nesting were high during 2006 and 2009 when prolonged water level recession concentrated aquatic prey without hydrologic reversals and/or marsh drydowns during the nesting season.

Wading bird nest numbers were very high in 2009 after a pulse in aquatic prey production during the preceding wet season. Nest numbers of white ibis and wood stork during 2009 exceeded the 90th percentile of all estimates of annual nesting for each species for the period of record beginning in 1931.

The pulse in prey biomass during the 2008 wet season, followed by very high wading bird nest numbers and nesting success during 2009, is consistent with the hypothesis that white ibis nest in unusually large numbers after periods of drydown in the context of multi-year wet and dry cycles in the Everglades. Mechanisms contributing to exceptional nesting after drought periods may involve pulses in prey populations, and are likely to be affected by the intensity of dry conditions. Wood stork and other wading birds also nested in unusually high numbers with high success during 2009.

Magnitude and success of wading bird nesting were low during 2007 and 2008 when water level recession patterns were not favorable for wading bird nesting, regardless of prey production during the preceding wet season.

8.2.4.2.2 Loxahatchee National Wildlife Refuge

The Loxahatchee NWR is the only area of the Everglades that consistently produced high aquatic prey biomass and successful nesting by white ibis and other wading birds during the last four years. Water stages that approximated long-term averages appeared to be beneficial to prey production and wading bird nesting in the Refuge.

8.2.5 Key Management Recommendations

8.2.5.1 Restore Multi-Year Hydroperiods in Slough Systems of Everglades National Park

The close association between restoration of wading bird nesting and the return of hydrologic conditions favorable to nesting in Everglades National Park identifies the return of multi-year hydroperiods in the slough systems of the park as a high priority management objective for CERP. This supports the broader management recommendation from the sheet flow and water depth patterns hypotheses to redistribute sheet flow through the natural flow corridors of the Shark Slough complex and Lostman's Slough in Everglades National Park in order to restore multi-year hydroperiods in slough systems of the park. Multi-year hydroperiods in sloughs would increase the prey base supporting wading bird nesting in Everglades National Park by increasing populations of aquatic prey organisms that require three or more years to fully develop, including marsh fish and grass shrimp. Monitoring results also identify the wood stork in particular as heavily dependent on Everglades National Park for nesting, and as particularly vulnerable to diminished prey availability due to frequent drying and hydrologic reversals.

8.2.5.2 Implement Rainfall-Driven Multi-Year Wet and Dry Cycles in Water Conservation Area 3 and Everglades National Park

The pulse in aquatic prey production and wading bird nesting during WY 2009 is consistent with the working hypothesis that drydowns can contribute to events of high prey abundance and wading bird nesting. This supports the broader management recommendation from the sheet flow and water depth patterns hypotheses to restore rainfall-driven volume, timing and distribution of sheet flow through WCA 3A and WCA 3B into Everglades National Park to produce water depths, hydroperiods, and flow patterns that are consistent with the best understanding of how the pre-drainage system responded to seasonal and interannual variability in rainfall. Pre-drainage patterns of multi-year hydroperiods, interrupted by periodic droughts would increase the prey base for wading birds, both by supporting population buildup of aquatic prey organisms such as marsh fish and grass shrimp during sequential wet years, and by producing surges in crayfish populations after drought years.

8.2.5.3 Maintain Current Water Depth Patterns in Loxahatchee National Wildlife Refuge

Loxahatchee NWR is the only area of the Everglades that has consistently worked to produce successful nesting by white ibis and other wading birds during the last four years. The Loxahatchee NWR has consistently supported high biomass of aquatic prey during the last four years, which, in part, has masked significant differences among sampling years. Until there is further understanding of why this is working, it is recommended to continue the hydrologic regime that is now in place.

8.2.5.4 Maintain Monitoring of Wading Bird Nesting and Aquatic Prey Abundance in the Water Conservation Areas and Everglades National Park

Highly variable and unexpected fluctuations in prey biomass and wading bird nesting during the last four years indicate the utility of long-term monitoring in quantifying ecological responses to hydrology in the Everglades, both prior to and following CERP implementation. Therefore, it is recommended that the monitoring of wading bird nesting and aquatic prey abundance be continued in the WCAs and Everglades National Park. This is key information needed to support AM.

8.3 AMERICAN ALLIGATOR DENSITY AND BODY CONDITION IN RELATION TO THE HYDROLOGIC PATTERNS IN THE EVERGLADES

8.3.1 Introduction and Background

Density and body condition of the American alligator (*Alligator mississippiensis*) in remaining Everglades' wetlands are currently reduced due to altered water depth patterns, salinity distributions, and prey abundance, which have resulted from compartmentalization and disrupted sheet flow as indicated by the CEM depicted in ***Figure 8-15***. Also, canals draw alligator populations from adjacent marshes and reduce the abundance and survival of juvenile alligators due to increased predation. Resumption of natural volume, timing and distribution of sheet flow in the WCAs and Everglades National Park would affect alligator populations as follows:

Restoration of multi-year hydroperiods and natural water level recession patterns in the ridge and slough landscape would result in increased alligator density and body condition.
Restoration of hydropatterns (depth, duration, distribution and flow) in the southern marl prairies and rocky glades landscapes of Everglades National Park would expand the distribution and abundance of reproducing alligators and active alligator holes.
Restoration of estuarine salinity regimes in Everglades National Park would expand the distribution and abundance of reproducing alligators into oligohaline portions of estuaries.

A performance measure has been developed for American alligator distribution, size, nesting and condition. The documentation for this performance measure can be found at www.evergladesplan.org/pm/recover/recover_docs/ret/pm_ge_alligator.pdf. An interim goal has been developed for the American alligator. The documentation sheet for this interim goal can be viewed at www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_ report/ig_3-10_alligators.pdf

8.3.2 Monitoring

8.3.2.1 Alligator Relative Density

Alligators are counted via spotlight surveys along ten survey routes in Loxahatchee NWR, the WCAs, Everglades National Park, and eastern Big Cypress National Preserve (***Figure 8-16***), following guidelines in the Alligator Spotlight Survey Protocol (Appendix 1 in Rice and Mazzotti, 2007). Surveys are conducted twice during both spring and fall, at least 14 days apart, to achieve independent counts. Details of monitoring methods are given in Mazzotti et al. (2009). Relative density is calculated by dividing the total number of non-hatchling animals encountered on each survey by the total length in kilometers of the survey route.

8.3.2.2 Alligator Body Condition

To determine body condition of alligator populations, semi-annual capture surveys are performed along the same routes as described for spotlight surveys (***Figure 8-16***). A minimum of 15 alligators greater than one meter total length are captured from each survey route in the fall and spring of each year. Body condition, which is a ratio of body length to body volume, is calculated using a Fulton's K condition factor (Zweig, 2003).

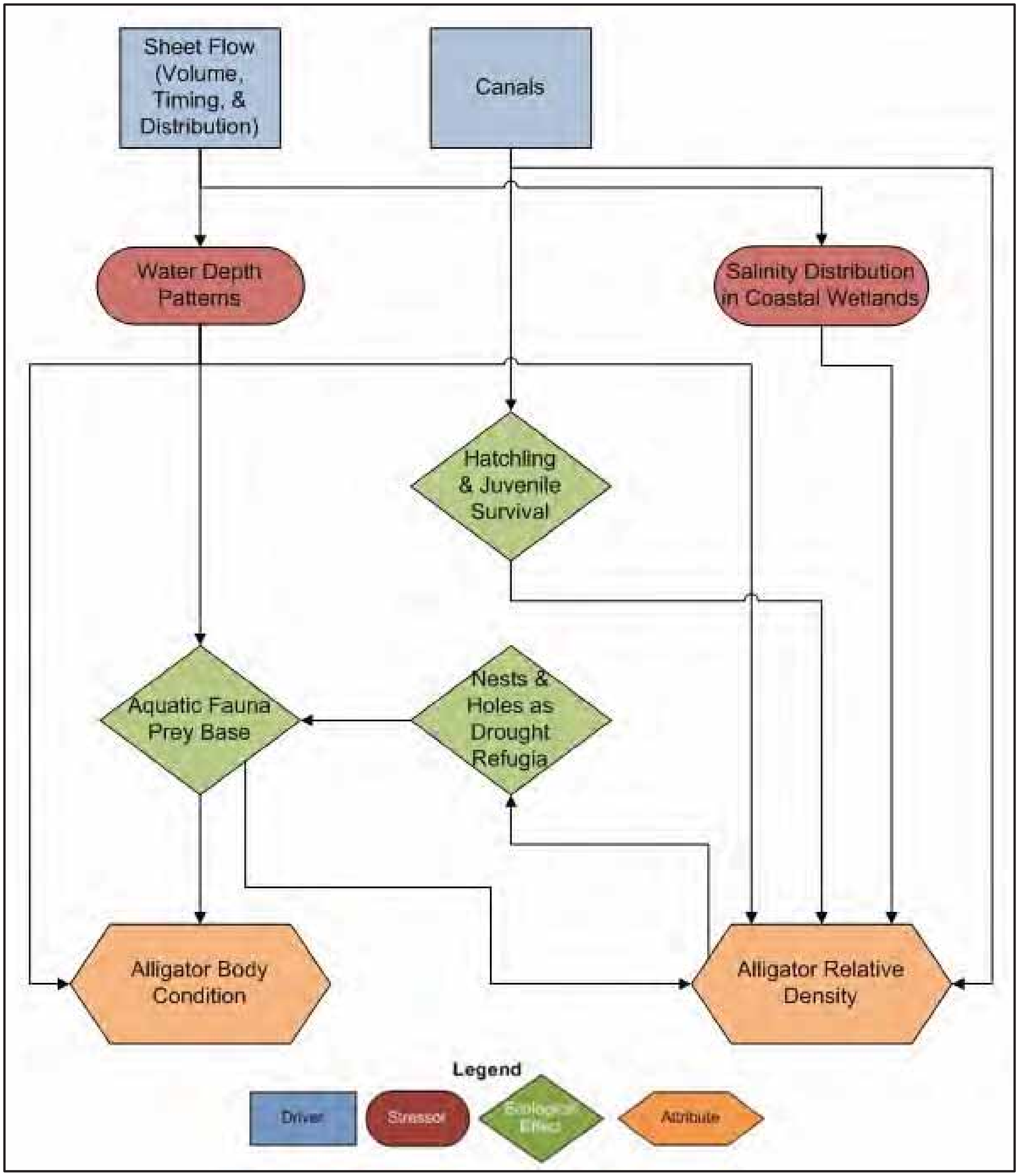


FIGURE 8-15: CONCEPTUAL ECOLOGICAL MODEL FOR DENSITY AND BODY CONDITION OF ALLIGATORS IN THE EVERGLADES

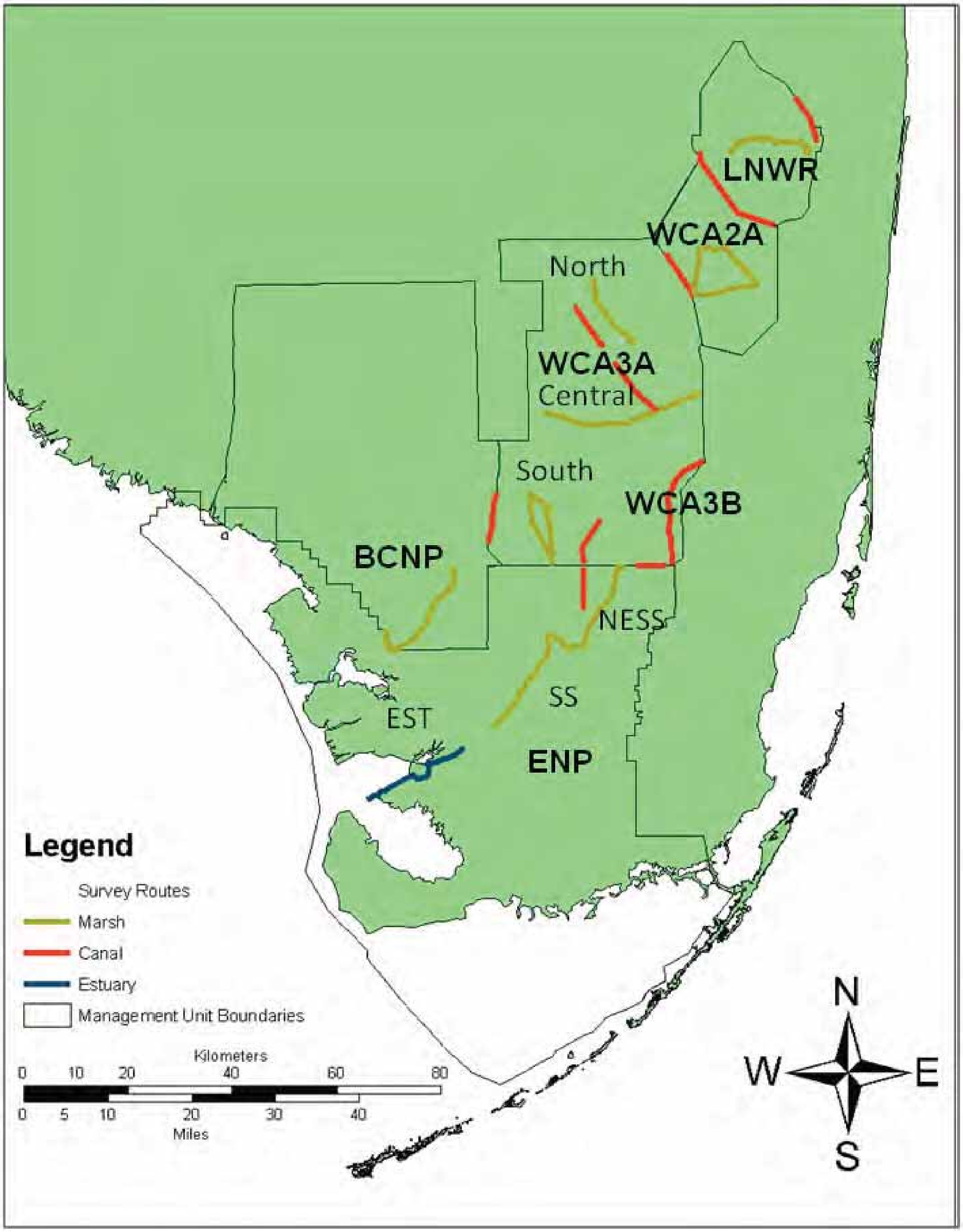


FIGURE 8-16. LOCATION OF ALLIGATOR SURVEY ROUTES IN LOXAHATCHEE NATIONAL WILDLIFE REFUGE, THE WATER CONSERVATION AREAS, EVERGLADES NATIONAL PARK, AND EASTERN BIG CYPRESS NATIONAL PRESERVE

Key:				
	LNWR	Loxahatchee NWR	SS	Shark Slough,
	ENP	Everglades National Park	EST	estuary
	NESS	Northeast Shark Slough	BCNP	Big Cypress National Preserve

8.3.2.3 Supplemental Monitoring and Supporting Research

Surveys for alligator hole occupancy were conducted via standard reconnaissance flights in Everglades National Park during 2005 and 2006. Details of monitoring areas, methods and results are given in Rice and Mazzotti (2007).

8.3.3 Results

Restoration targets for alligator relative density and body condition are set as constant for all areas of the Everglades where spotlight surveys are located. The assumption is made that all those areas can attain equally high alligator density and condition values if hydrology restoration is successful. Restoration targets for alligator relative density and body condition are based on the upper fourth quartile of the distribution of values from all survey routes for the 1999 to 2006 period of record. Results are taken from Rice and Mazzotti (2008) and Mazzotti et al. (2009).

Relative density of alligators during 2006 to 2008 was highest in impoundments and canals of the WCAs (***Figure 8-17***). Loxahatchee NWR supported densities of four to ten alligators per kilometer (/km), which corresponded to 4th quartile restoration targets for the Everglades. Densities in the pool of water in central and southern WCA 3A mostly ranged between 1.4 and 2.8 alligators/kilometer, which fell short of 4th quartile targets but exceeded extreme low densities in the 1st quartile. Relatively higher alligator densities in Loxahatchee NWR and WCA 3 predictably occurred where impoundment produced extensive areas of multi-year hydroperiods.

In contrast to densities in impounded marshes of the WCAs, relative density of alligators in Everglades National Park was consistently low during all three years and along all survey routes (***Figure 8-17***). Densities of less than 1.4 alligators per kilometer in the park fell within the lowest quartile for the Everglades, indicating populations were extremely low compared to restoration targets. Relatively low alligator densities in Everglades National Park predictably occurred where withholding of water deliveries reduced hydroperiods to less than one year in a ridge and slough landscape characterized by multi-year hydroperiods under natural conditions.

Differences between the WCAs and Everglades National Park were not evident for body condition from 2006 to 2008 (***Figure 8-18***). Fulton's K values for alligator body condition consistently ranged within 2nd and 3rd quartile values of 9.4 and 11.3 during most years and along most survey routes throughout the Everglades. No relationship was evident between relative density and body condition of alligators.

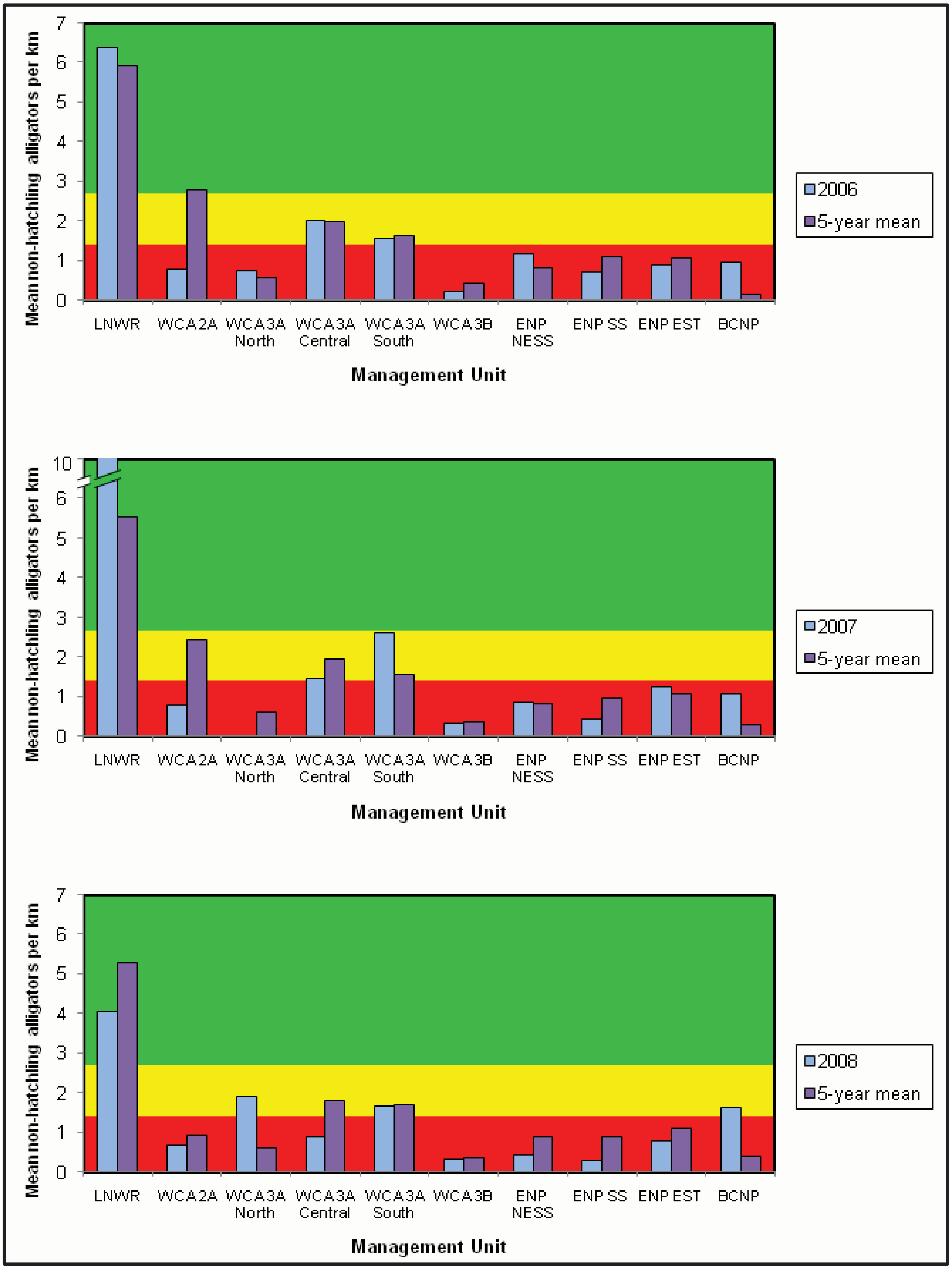


FIGURE 8-17. MEAN RELATIVE DENSITY OF ALLIGATORS DURING 2006 TO 2008 FOR EACH OF THE SURVEY ROUTES

Key:		
	1st quartile	red
	2nd quartile	yellow
	3rd quartiles	yellow
	4th quartile	green

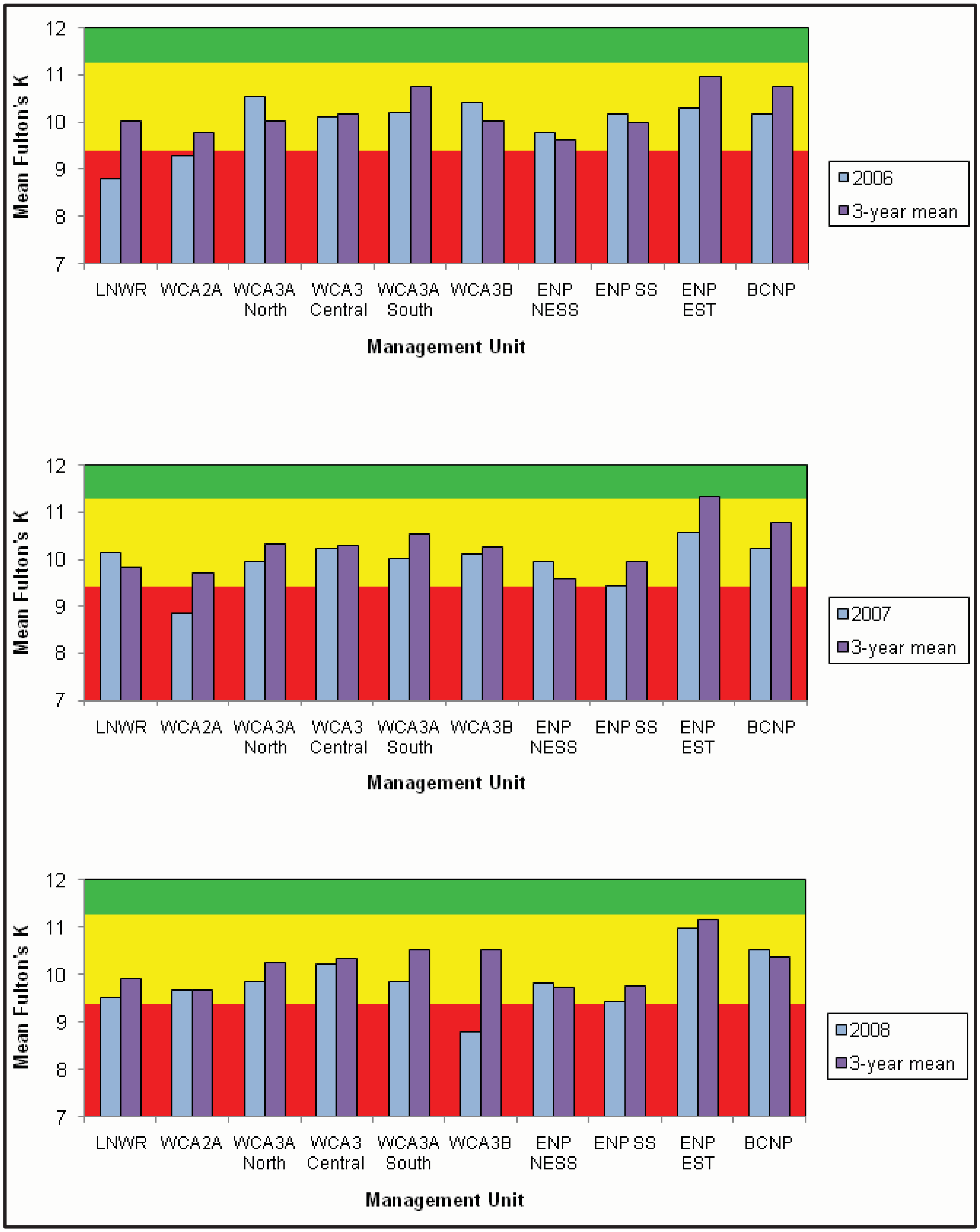


FIGURE 8-18. MEAN BODY CONDITION OF ALLIGATORS DURING 2006 TO 2008 FOR EACH OF THE SURVEY ROUTES

Key:	1st quartile	red
	2nd quartiles	yellow
	3rd quartiles	yellow
	4th quartile	green

8.3.4 Summary

8.3.4.1 Assessment of Status and Trends of Alligator Populations in Context of Working Hypotheses

Low relative density of alligators in Everglades National Park in comparison to the WCAs supports the working hypothesis that altered hydrologic patterns have reduced alligator populations in the Everglades. Relative density of alligators is extremely low in ridge and slough landscapes of the park where naturally occurring multi-year hydroperiods are reduced to less than one year (Marshall et al., 2009), due to reduced volume and altered distribution of inflows from WCA 3A.

Low body condition of alligators throughout the Everglades suggests that alligator population density responds more sensitively to contrasting hydrologic regimes than body condition.

The Loxahatchee NWR is the only area of the Everglades that has consistently supported restoration targets for alligator relative density during the last four years.

8.3.4.2 Key Management Recommendations

8.3.4.2.1 Restore Multi-Year Hydroperiods in Slough Systems of Everglades National Park

Extremely low relative density of alligators in ridge and slough landscapes of Everglades National Park where naturally occurring multi-year hydroperiods are reduced to less than one year, in comparison to high relative density of alligators in longer hydroperiod pools in the WCAs, identifies the return of multi-year hydroperiods in the slough systems of the park as a high priority management objective for CERP. This recommendation supports the broader management recommendation from the sheet flow and water depth patterns hypotheses to redistribute sheet flow through the natural flow corridors of the Shark Slough complex and Lostman's Slough in Everglades National Park in order to restore multi-year hydroperiods in slough systems of the park. It is consistent with management recommendations from the wading bird nesting in relation to aquatic fauna forage base hypotheses. Restoration of multi-year hydroperiods in the Everglades National Park would increase the spatial extent and duration of aquatic habitats, which in turn would support alligator populations comparable to those in longer hydroperiod areas of the WCAs.

8.3.4.2.2 Maintain Current Water Depth Patterns in Loxahatchee National Wildlife Refuge

Loxahatchee NWR is the only area of the Everglades that has consistently supported 4^{th} quartile restoration targets for alligator relative density during the last four years. Until there is better understanding of how it is working, monitoring should continue with the hydrologic regime that is in place. This is identical to the recommendation based on results from the wading bird nesting in relation to aquatic fauna forage base hypotheses.

8.4 ECOSYSTEM CHARACTERISTICS OF EVERGLADES COASTAL WETLANDS IN RELATION TO FRESHWATER INFLOWS HYPOTHESIS CLUSTER

8.4.1 Introduction and Background

The volume, timing and distribution of freshwater flow to coastal wetlands of the Everglades from Barnes Sound to Lostman's Bay have been altered by upstream diversions of water, compartmentalization, and disrupted sheet flow (RECOVER, 2009). The potential ecological effects include disruption of estuarine predator-prey interactions, roseate spoonbill nesting success, and American crocodile (*Crocodylus acutus*) juvenile growth and survival as indicated by the CEM depicted in ***Figure 8-19***.

Five performance measures relating to this hypothesis cluster have been developed and are being refined by RECOVER. These include measures on: 1) coastal salinity gradients, 2) tidal creek sustainability, 3) mangrove forest production/soil accretion, 4) roseate spoonbill nesting patterns and 5) American crocodile juvenile growth and survival. Documentation for these performance measures can be found at http://www.evergladesplan.org/pm/recover/perf_ge.aspx.

Interim goals were developed for indicators and are included within this hypothesis cluster. An interim goal has been developed for the American crocodile (www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_4-4_crocodiles.pdf) and roseate spoonbills are included in the system-wide wading bird nesting patterns interim goal (www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_3-11_wadingbirds.pdf).

The study area is comprised of coastal wetlands of the Everglades from Barnes South to Lostman's Bay including northeast and northwest Florida Bay (***Figure 8-20***). This map shows monitoring stations for all of the data used in the assessments discussed in this chapter.

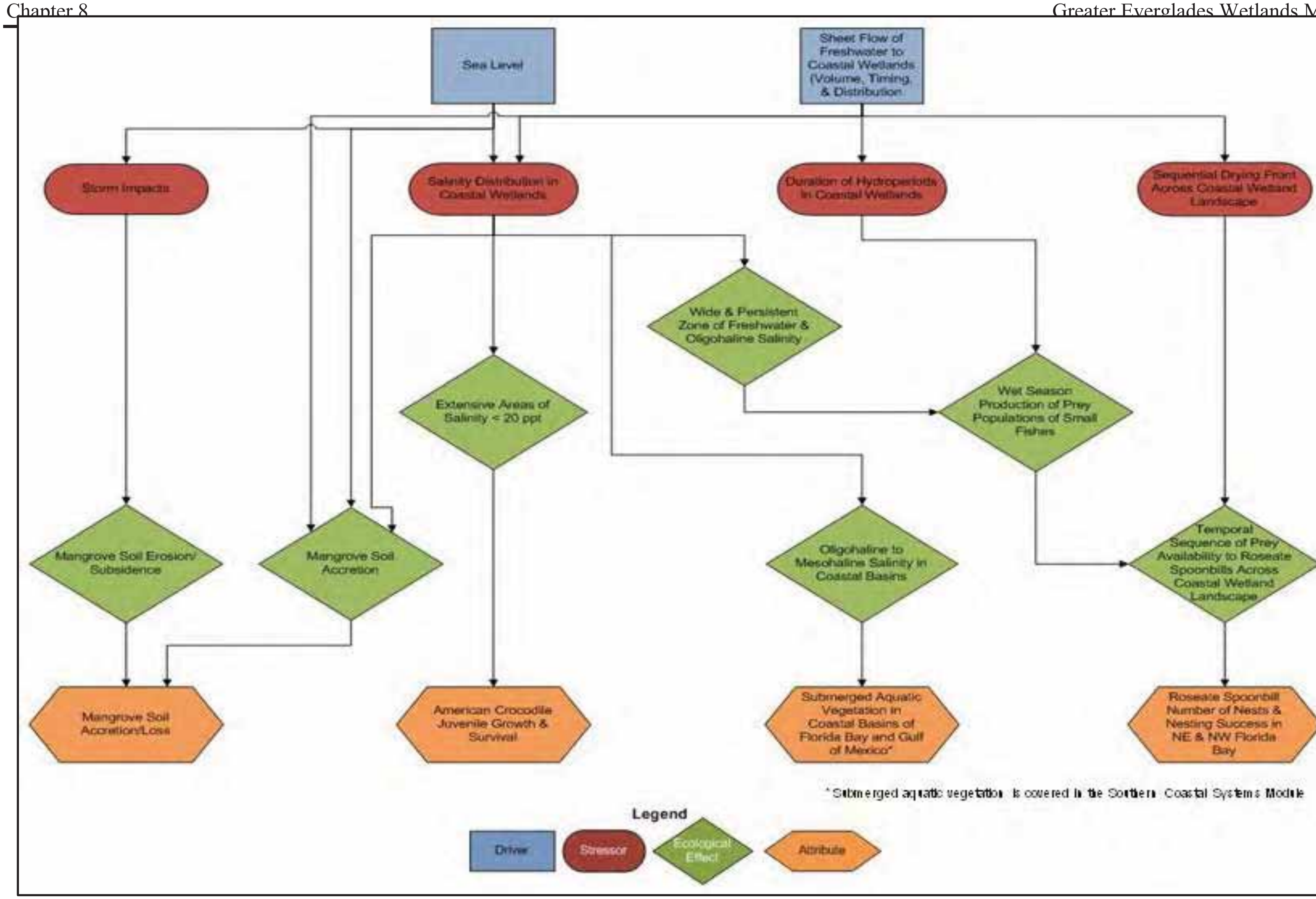


FIGURE 8-19. CONCEPTUAL ECOLOGICAL MODEL FOR EVERGLADES COASTAL WETLANDS IN RELATION TO FRESHWATER INFLOWS

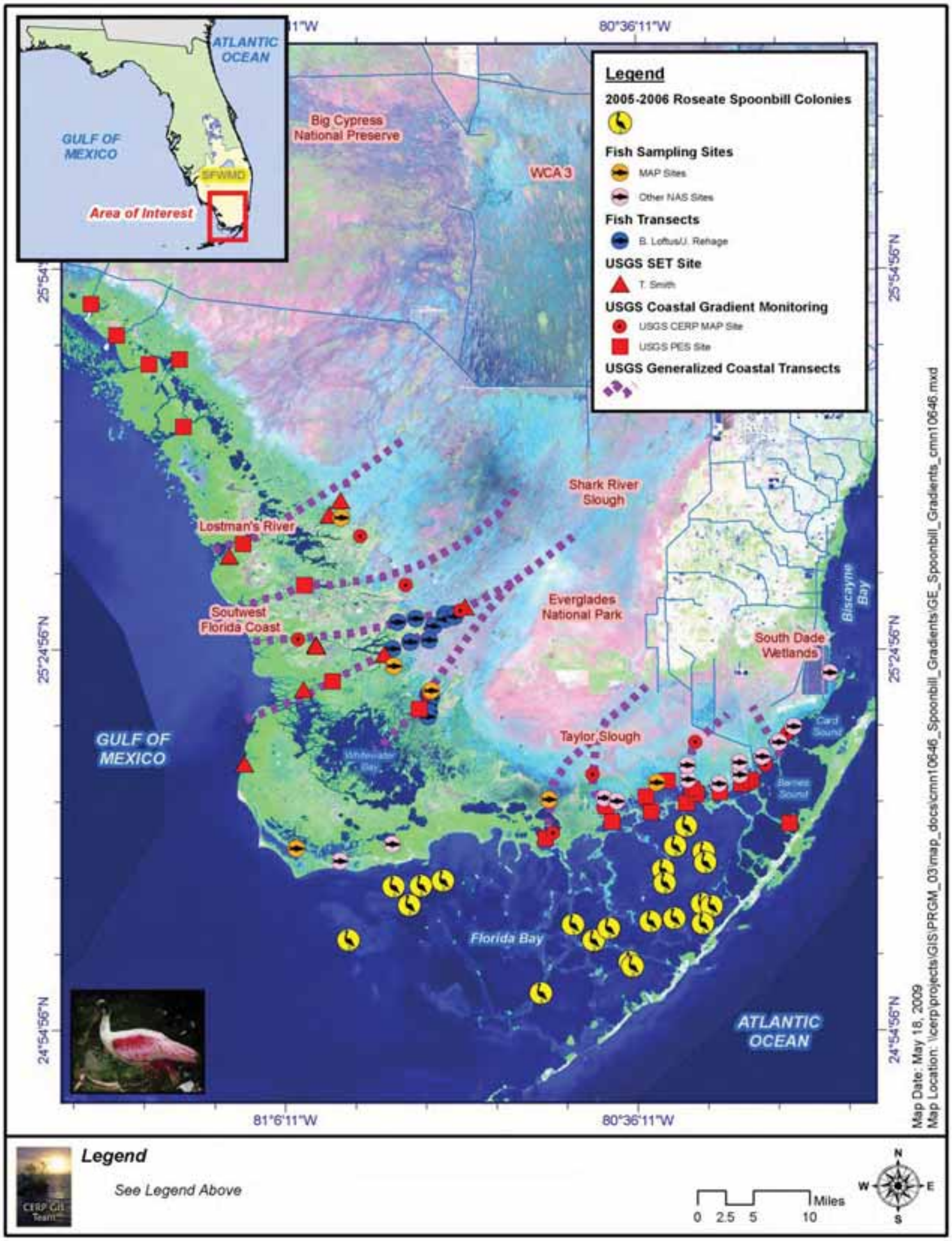


FIGURE 8-20. LOCATION OF TRANSECTS AND MONITORING STATIONS FOR SALINITY GRADIENTS, SEDIMENT ELEVATIONS, FISH BIOMASS AND ROSEATE SPOONBILL NESTING IN THE COASTAL EVERGLADES

8.4.2 Freshwater Inflows, Hydrologic Patterns, Salinity Distributions, and Surface Water Nutrient Concentrations

8.4.2.1 Monitoring

An integrated network of 36 monitoring stations located in major coastal creeks and rivers are combined to provide data along nine generalized coastal gradients or transects extending from freshwater to marine conditions (***Figure 8-20***). These sites have been established with MAP and USGS Priority Ecosystem Science (PES) funding. Additional information regarding the USGS coastal monitoring sites and the PES program is available through the USGS South Florida Information Access (SOFIA) web page (www.sofia.usgs.gov). In addition, direct access to the provisional real-time data may be accessed via the USGS National Water Information System (NWIS) web page (waterdata.usgs.gov/nwis). Detailed information regarding the exact methodology and quality assurance plans may be found in the 2008 Annual Report submitted to RECOVER by Woods and Zucker (2008).

The southern portion of the Everglades wetlands ecosystem located south of the Tamiami Trail can be separated into two main drainage complexes named for the two well-known sloughs: Shark and Taylor/C-111. Monitoring for the Taylor/C-111 complex predates the RECOVER MAP program and has been in place since 1996. Consistent flow measurements within coastal rivers of the Shark complex began with RECOVER MAP funding in 2004.

The Shark drainage complex is comprised of five major rivers located along the southwest coast of Everglades National Park: Lostman's, Broad, Harney, Shark and North Rivers. Discharge volumes in this complex are a product of direct rainfall, inflow from the Tamiami control structures (S12s) and culverts, and watershed runoff originating from Big Cypress National Preserve. Data for the Tamiami Trail structures includes flows for S12A, B, C and D; the 29 outlets and S14 to the west of the S12s; and the 19 outlets plus S12E to the east of the S12s. Data for the C-111 is based on S18C flows.

8.4.2.1.1 Freshwater Discharge

It is hypothesized that freshwater inflow from the Everglades is expected to sustain watercourses through the estuary that would more closely resemble historic patterns, and re-open some channels that have partly filled because of reduced flow. (Refer to the Tidal Creek Sustainability performance measure documentation sheet, which can be found at www.evergladesplan.org/pm/recover/recover_docs/et/ge_16.pdf, for more information.)

Flow volumes delivered south to the Shark complex from WCA 3 via the inflow structures and culverts located along the Tamiami Trail averaged 1,209,159 acre-feet per year, noting that in 2007, a drought year, flow volumes totaled 166,860 acre-feet; the lowest volume of flow recorded in 30 years (Woods and Zucker, 2008). Tamiami Trail inflows accounted for about 55 percent of the discharge volume from the Shark complex (***Table 8-6***) for 2004 through 2008, with precipitation and Big Cypress National Preserve watershed runoff accounting for the rest. In 2005, local precipitation events from three major storms (Hurricanes Katrina, Rita and Wilma) affected water depth (see ***Section 8.2 Sheet Flow and Water Depth Patterns Hypothesis Cluster***) and discharge from Everglades National Park.

TABLE 8-6. TOTAL AND MEAN ANNUAL DISCHARGE FROM SHARK AND TAYLOR/C-111 DRAINAGE COMPLEXES AND FLOW MEASURED AT TAYLOR SLOUGH BRIDGE AND C-111 CANAL FROM 1996-2008[1]

Year	Tamiami Trail Inflows (acre-feet)	Shark Drainage Complex (acre-feet)	Lostmans River Drainage Complex (acre-feet)	C-111 Inflows (acre-feet)	Taylor River/C-111 Drainage Complex (acre-feet)	Taylor Slough Bridge Inflows (acre-feet)
1996	1,344,200	no data	no data	138,600	301,340	51,903
1997	1,145,920	no data	no data	167,100	332,900	81,573
1998	1,233,890	no data	no data	173,500	275,680	62,544
1999	2,267,500	no data	no data	164,200	376,370	82,751
2000	679,340	no data	no data	187,100	230,340	84,573
2001	1,052,900	no data	no data	149,100	362,680	82,991
2002	1,181,040	no data	717,700	131,500	380,010	64,596
2003	1,440,200	no data	741,300	157,300	338,650	91,121
2004	1,051,230	829,800	650,400	70,920	152,850	35,316
2005	1,970,500	1,175,460	749,100	183,000	501,290	92,016
2006	707,290	883,610	342,500	79,660	222,510	29,797
2007	166,860	564,040	162,300	97,470	280,540	14,959
2008	1,478,200	1,110,120	584,400	132,400	266,060	62,777
Mean	1,209,159	912,606	563,957	140,912	309,325	64,378

The Shark drainage complex accounted for over 80 percent of the mean annual discharge from Everglades National Park comparing years 2004 to 2008 (***Table 8-6***). The total flow volume discharged through all five rivers ranged from a low of 768,180 acre-feet in 2007 to a high of 2,011,070 in 2005. This flow volume corresponds with the 2007 drought and 2005 active storm season. Lostman's and Harney Rivers discharges comprised over 60 percent of the flow volume leaving the Shark complex for 2004 through 2008, while discharges from Shark River, which are weakly correlated to flows measured at Tamiami Trail (Woods and Zucker, 2008), on average comprised only 14 percent of the total (***Figure 8-21***). Percent discharges were relatively stable through the period of record excepting 2007 when Lostman's River percent discharges dropped to 23 percent, less than the overall mean, while Harney River percent discharges rose to 44 percent, which was greater than the overall mean.

The Taylor/C-111 drainage complex consists of nine major discharge paths: east Highway Creek, west Highway Creek, Oregon Creek, Stillwater Creek, Trout Creek, Mud Creek, Taylor River and McCormick Creek. Discharge volumes in the Taylor/C-111 complex are a product of direct rainfall and operational contributions from the upstream watershed that are best characterized by flows measured directly at the Taylor Slough Bridge, and the S18C structure located along the C-111 Canal (***Table 8-6***). Structure S197 discharges directly into Manatee Bay but is opened only during extreme events, such as in 2005 following Hurricane Wilma. Overall annual mean calculations do not include 2008 Taylor Slough Bridge values.

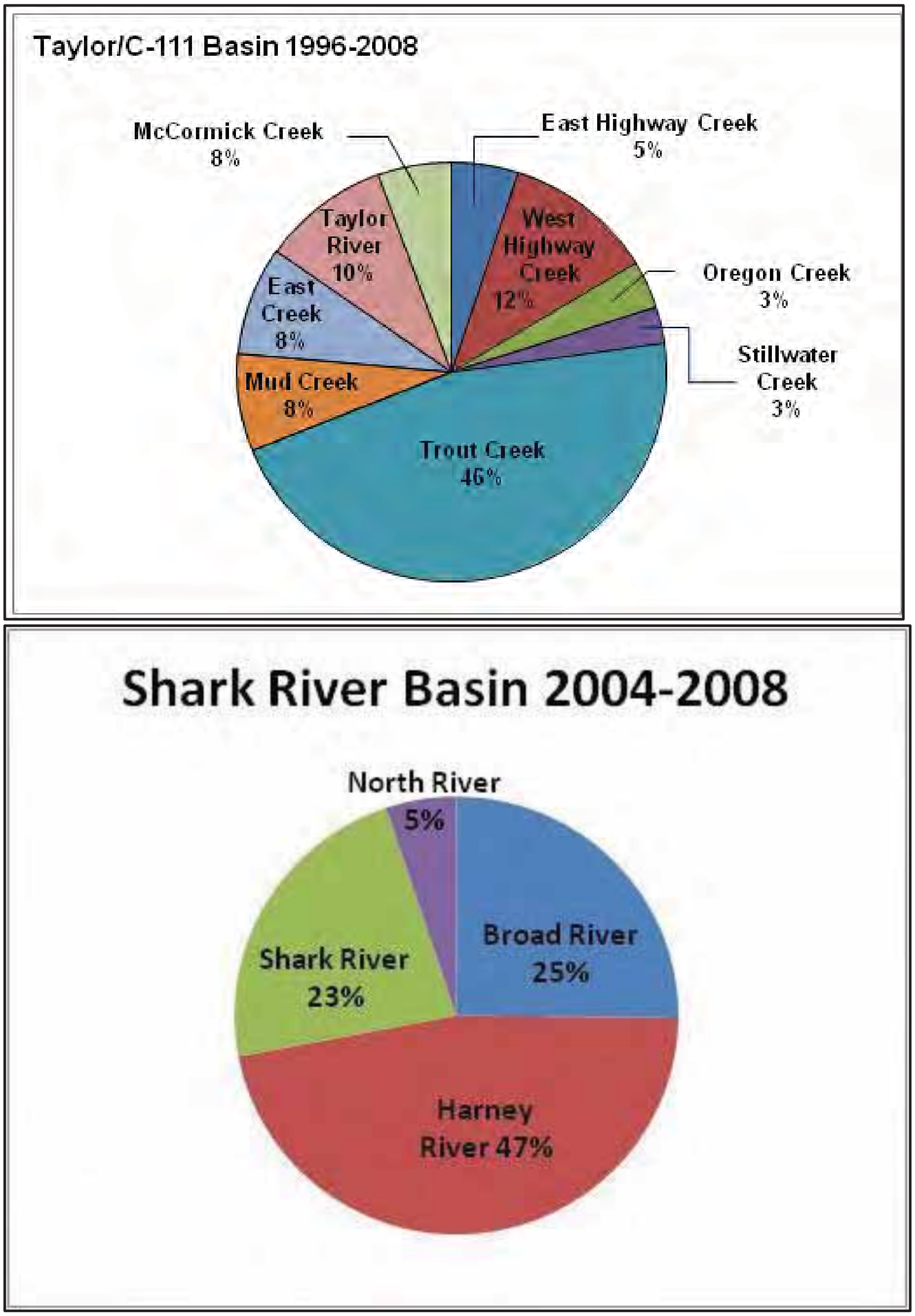


FIGURE 8-21. MEAN ANNUAL FLOW DISTRIBUTION FROM SHARK DRAINAGE COMPLEX FOR THE PERIOD 2004 THROUGH 2008 AND TAYLOR /C-111 DRAINAGE COMPLEX FOR 1996 THROUGH 2008

Flow volumes delivered to the Taylor/C-111 drainage complex via the Taylor Slough Basin and the C-111 pathways had a combine annual average of 205,290 acre-feet, representing 68 percent of the Taylor/C-111 discharge volume, except in 2000 when Taylor Slough Basin and C-111 combined flows exceeded Taylor/C-111 discharge volumes (***Table 8-6***). Combined inflow volumes during 2007 were 112,429 acre-feet, which was lower than average but not the minimum recorded value of 106,236 acre-feet, which was recorded in 2004. ***FIGURE 8-22*** shows the timing of flow volumes from the 1960s as recorded at the Taylor Slough Bridge. While the total flow volume delivered during the 1990s and 2000s is similar, at 566,979 and 558,146 acre-feet, respectively (note that 2009 flow volumes are not included), a clear shift in the timing of flow is observed, with almost no flow delivered from January through May during the 2000s. The flow volumes represented in January during the 2000 decade results almost entirely from flow delivered during 2000.

The Taylor/C-111 drainage complex accounted for approximately 15 percent of the annual mean discharge volumes from Everglades National Park for the 2004 through 2008 period (***Table 8-6***). For the entire period of record (1996 - 2008), the discharge ranged from a low of 154,130 acre-feet in 2004 to 506,170 acre-feet in 2005. In 2007, the Taylor/C-111 complex total discharge was 267,880 acre-feet, which is lower than mean volume of 303,096 acre-feet but not the minimum recorded value. Trout Creek discharges accounted for 46 percent of the discharge volume for the Taylor/C-111 complex for the period of record (1996 - 2008), while discharges from Taylor River on average comprised only ten percent of the total (***Figure 8-21***).

8.4.2.1.2 Salinity

It is hypothesized that under pre-drainage conditions headwater sites along the marsh-mangrove interface of the coastal Everglades had persistent freshwater to oligohaline salinities. (Refer to the Coastal Salinity Gradients performance measure documentation sheet, which can be found at www.evergladesplan.org/pm/ recover/recover_docs/et/ge_12.pdf, for more information.)

Mean monthly discharge and salinity at upstream and downstream coastal creek sites are presented for October 2003 through 2008 for the set of transects previously described (***Figure 8-23*** and ***Figure 8-24***) (Woods and Zucker, 2008). Five transects comprise the Lostman's River and Shark Slough drainage complex and primarily deliver fresh water to the Gulf of Mexico. The nine discharge pathways previously used for the Taylor/C-111 drainage complex are re-aggregated into five transects for this discussion and primarily deliver fresh water to Florida Bay.

The coastal gradients during this period continue to indicate an absence of persistent freshwater-oligohaline zones at headwater sites across most of the coastal Everglades as previously reported in the 2007 SSR (RECOVER, 2007b). 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 8-23c*** and ***Figure 8-23d***). Freshwater-to-oligohaline conditions also prevailed at the Broad River headwaters, although salinity rose to more than 10 psu during no discharge periods of 2004, 2005 and 2008 (***Figure 8-23b***). Upstream and downstream salinities fluctuated together in the

Lostman's River, North River, and all Florida Bay drainages, where oligohaline conditions occurred only during high discharge periods (***Figure 8-24a*** and ***Figure 8-24b***).

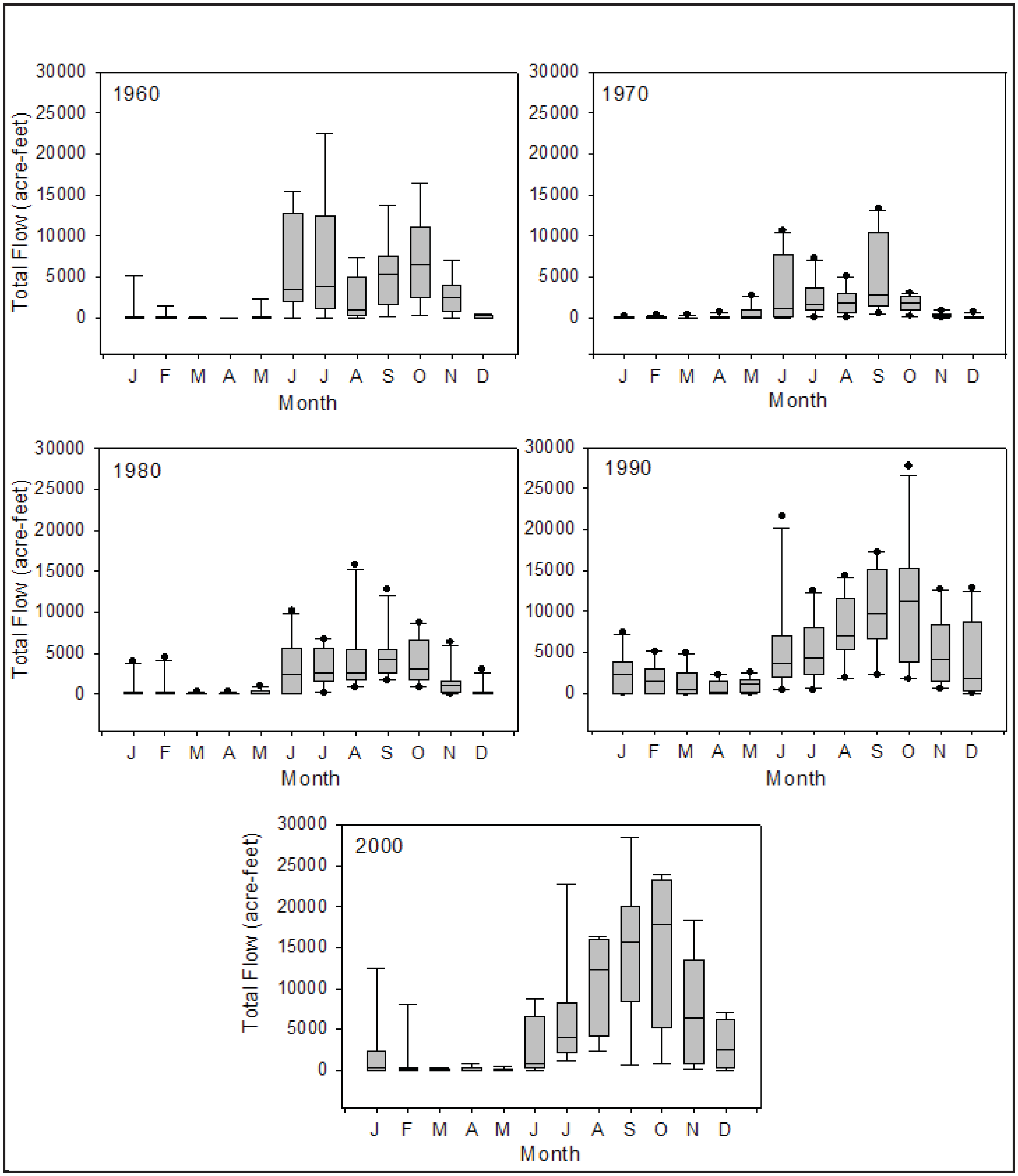
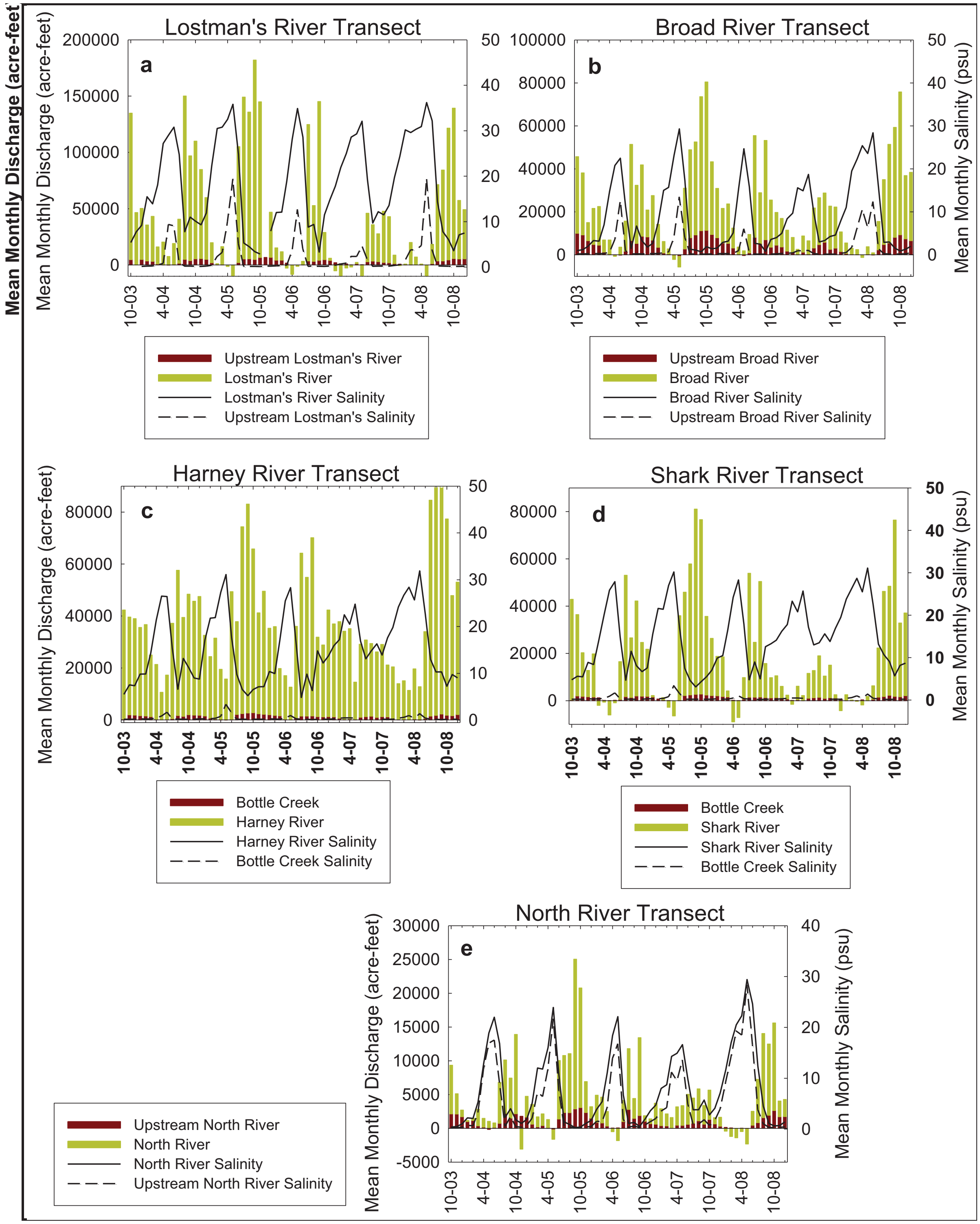


FIGURE 8-22. BOX PLOTS OF MONTHLY TOTAL FLOWS AT TAYLOR SLOUGH BRIDGE FROM OCTOBER 1960 THROUGH DECEMBER 2008

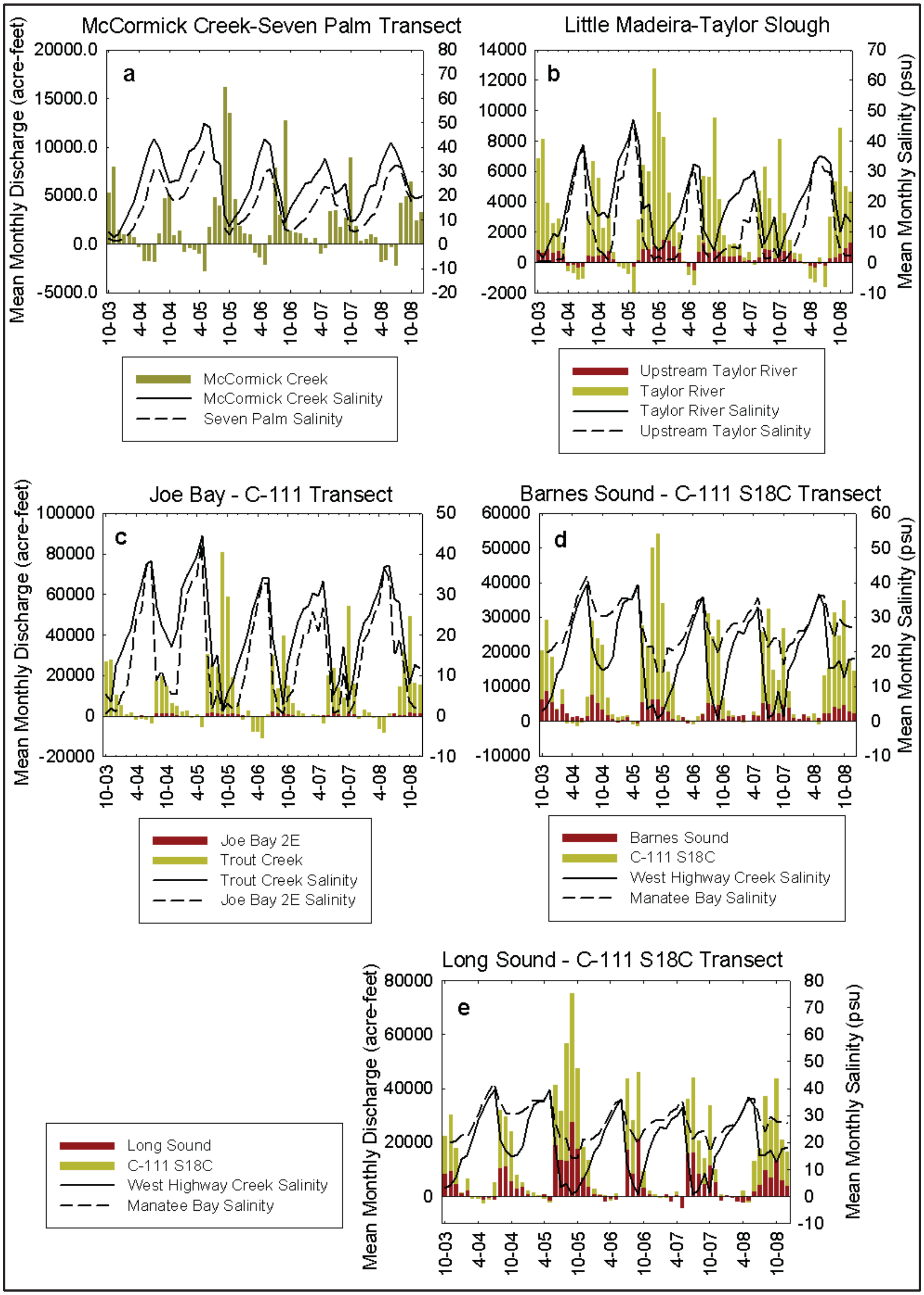
Note: black line is the median; decades 1960 and 2000 are each missing one full year of flow values, 1960 and 2009, respectively



(Woods and Zucker 2008)

FIGURE 8-23. MEAN MONTHLY DISCHARGE AND SALINITY AT UPSTREAM AND DOWNSTREAM MONITORING STATIONS OF THE LOSTMANS RIVER AND THE SHARK DRAINAGE COMPLEX FOR 2003 THROUGH 2008

Note: Measured in Practical Salinity Units (PSU)



(Woods and Zucker 2008)

FIGURE 8-24. MEAN MONTHLY DISCHARGE AND SALINITY AT UPSTREAM AND DOWNSTREAM MONITORING STATIONS OF THE TAYLOR/C-111 DISCHARGE COMPLEX FOR 2003 THROUGH 2008

The salinity regime for the southwest coast differed from that of northeast Florida Bay. For the southwest coast, salinities are usually fresh throughout the wet season and rise during the dry season and during major tidal surges. 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.

8.4.3 Mangrove Forest Soil Accretion/Subsidence

8.4.3.1 Monitoring

Wetland surface elevation and accretion/subsidence are measured utilizing surface elevation tables (SETs) (Smith et al., 2009). Sediment elevation sites are co-located with hydrologic monitoring of surface water and groundwater levels. Conductivity is also monitored. Additionally, permanent and non-permanent vegetation plots are established at each site. Study sites are arranged along transects situated within the mangrove forests along the Shark-Harney River system and Lostman's River. Sites are located along a salinity gradient, in upstream freshwater wetlands, in a brackish marsh-mangrove community, and downstream in pure stands of a mangrove forest (***Figure 8-20***).

It is hypothesized that the primary production of mangrove forests supports the aquatic fauna that ultimately sustains reproduction by higher vertebrates. Mangrove forest production, as measured by soil accretion rate, is considered to be a regional indicator of the functional base of food webs in the mangrove ecotones of the Greater Everglades Wetlands. A performance measure has been developed for mangrove forest production/soil accretion, for which documentation can be found at www.evergladesplan.org/pm/recover/recover_docs/ret/pm_ge_mangrovesoil.pdf.

Figure 8-25 presents sediment elevation patterns. Change is measured relative to the elevation when monitoring began; beginning elevation was set at zero. Elevation change represents the change in table pin measurements between sampling events, accretion represents changes in sediment accumulation as measured on feldspar markers, and shrink-swell represents the expansion or compaction of deeper sediment layers.

Soil accretion rates and sediment surface elevation at the freshwater sites (SH1 and LO1) have been more variable compared to the estuarine sites (SH3 and LO3) (***Figure 8-25***). Additionally, the small gains in sediment elevation obtained at the SH1 site cannot be attributed to accretion but maybe related to the seasonal shrink-swell of the peat. Conversely, accretion rates and sediment elevation are much less variable at the estuarine sites (SH3 and LO3), with major gains in sediment elevation and accretion rates directly attributable to Hurricane Wilma, which deposited large amounts of sediment in the mangrove forests on the southwest coast of Florida (Smith et al., 2009). Prior to Hurricane Wilma, the accretion rate at SH3 was steadily rising but the sediment elevation did not show the same gains, indicating this area was experiencing shallow subsidence-compaction. At this point, the extent surface water and groundwater fluctuations has affected wetland surface elevations cannot be determined.

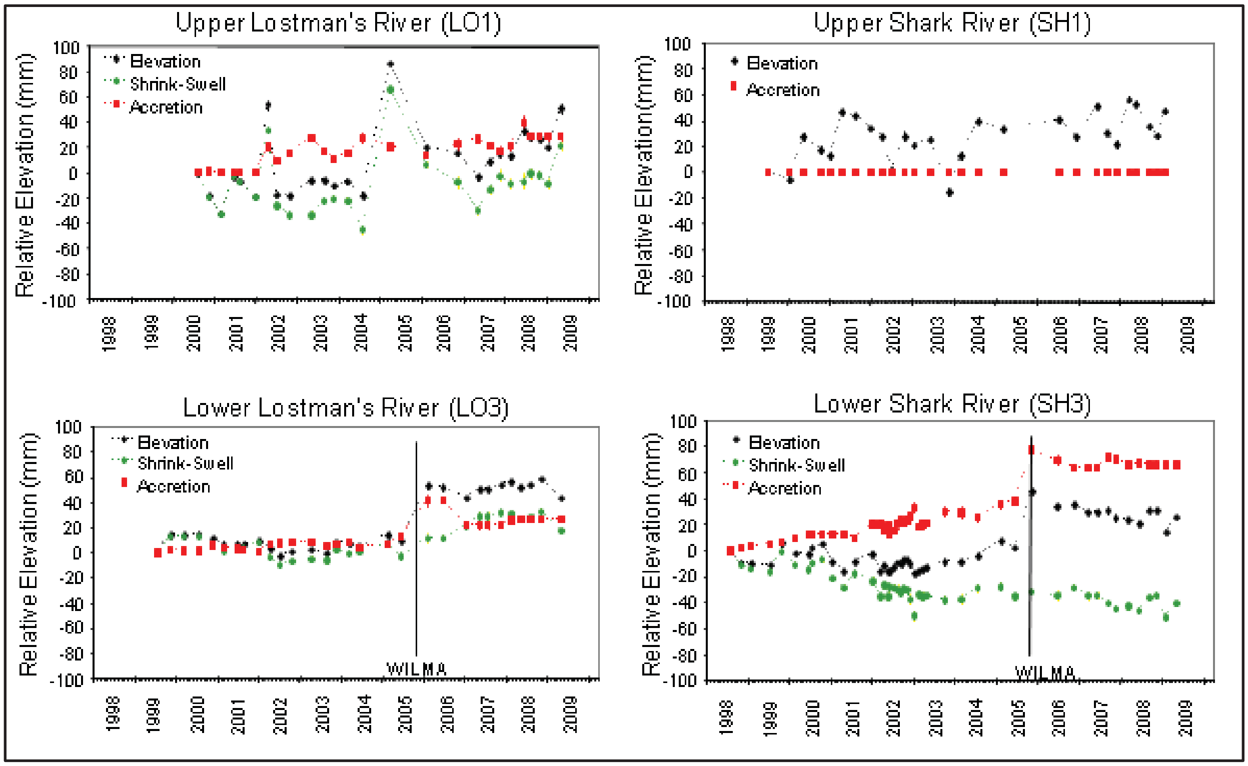


FIGURE 8-25. PATTERNS OF SEDIMENT ELEVATION CHANGE AS MEASURED AT THE FRESHWATER SITES (LO1 AND SH1) AND ESTUARINE SITES (LO3 AND SH3) LOCATED ALONG LOSTSMAN'S AND SHARK RIVERS FROM 1999 THROUGH 2008

8.4.4 Mangrove Creeks as Dry Down Refuges in Everglades Coastal Wetlands

8.4.4.1 Monitoring

Fishes and macro-invertebrates are monitored along the marsh-mangrove ecotone in the southwestern region of Everglades National Park (***Figure 8-20***). This monitoring is used as support data for the Wading Bird Nesting in Relation to the Aquatic Fauna Forage Base Hypothesis Cluster discussed in ***Section 8.3*** Physio-chemical parameters, including water temperature, dissolved oxygen, water clarity, bottom type, turbidity and specific conductance/salinity, are also measured.

Sampling takes place in 15 creeks associated with two drainages: the Rookery Branch, Otter Creek and Squawk Creek drainage basin, which forms the Shark River headwaters, and the North and Roberts Rivers, which flow into Whitewater Bay (***Figure 8-26***). Sampling extends into the uppermost stretches of ecotonal creeks accessible by electrofishing motorboat in an effort to document the role of these creeks as dry season refuges for freshwater fishes.

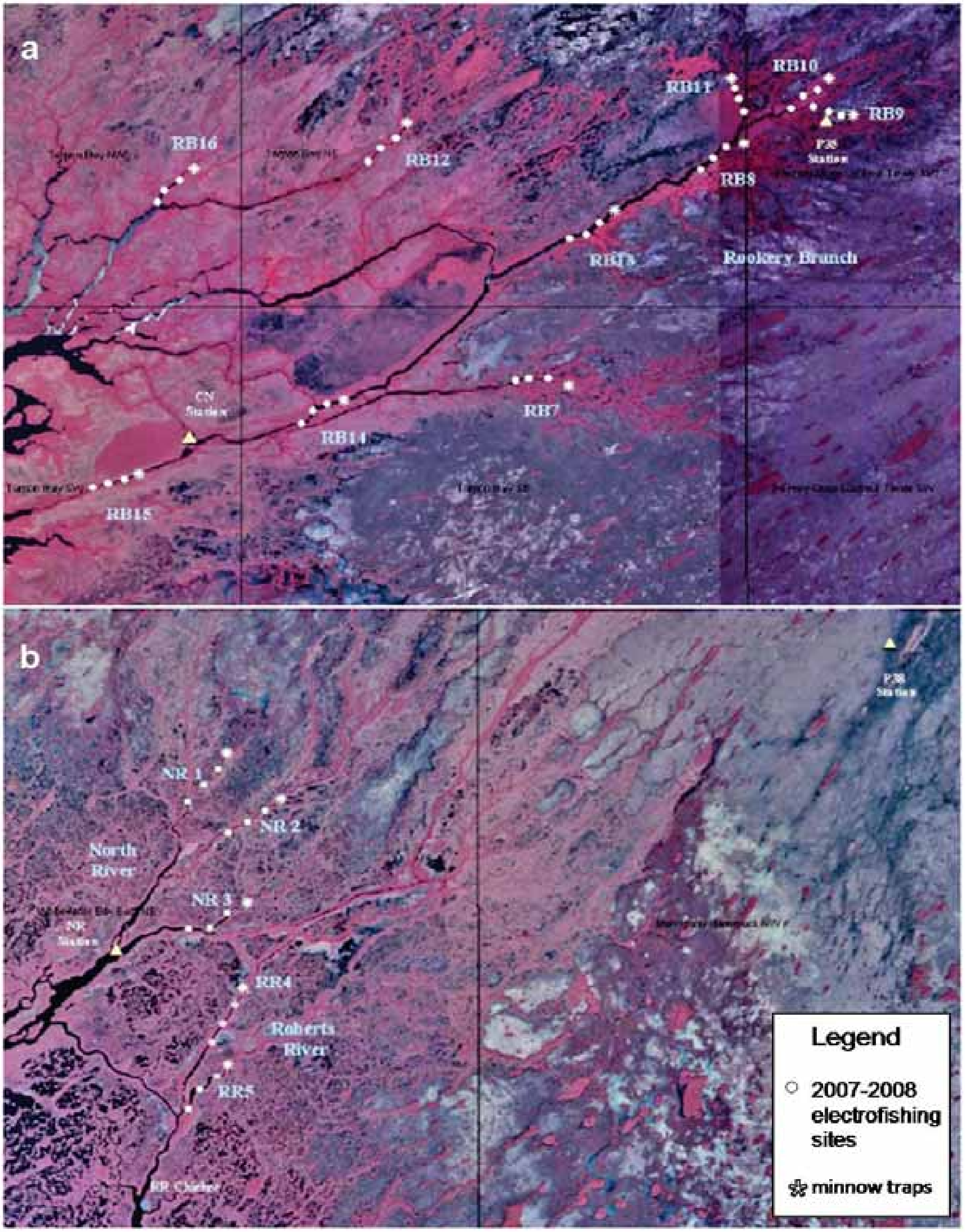


FIGURE 8-26. DETAILED SATELLITE IMAGES OF A) THE 10 STUDY SITES CURRENTLY SAMPLED IN THE ROOKERY BRANCH, OTTER CREEK AND SQUAWK CREEK DRAINAGE BASIN, AND B) THE FIVE SITES SAMPLED IN THE NORTH AND ROBERTS RIVERS

8.4.4.2 Results

It was hypothesized that restoring marsh-creek connectivity would allow these systems to serve as dry season habitats. Another hypothesis is the pattern and timing of marsh recession affects not only the downstream salinity but the timing and spatial extent of freshwater movement into the creeks, possibly shifting the energy flow from avian to piscine predators. Two performance measures pertain to these hypotheses: 1) Wetland Landscape Patterns - Tidal Creek Sustainability and 2) Wetland Trophic Relationships - Mangrove Forest Production/Soil Accretion. Documentation for these measures can be found online at www.evergladesplan.org/pm/recover/recover_docs/et/ge_16.pdf and www.evergladesplan.org/pm/recover/recover_docs/ret/pm_ge_mangrovesoil.pdf, respectively.

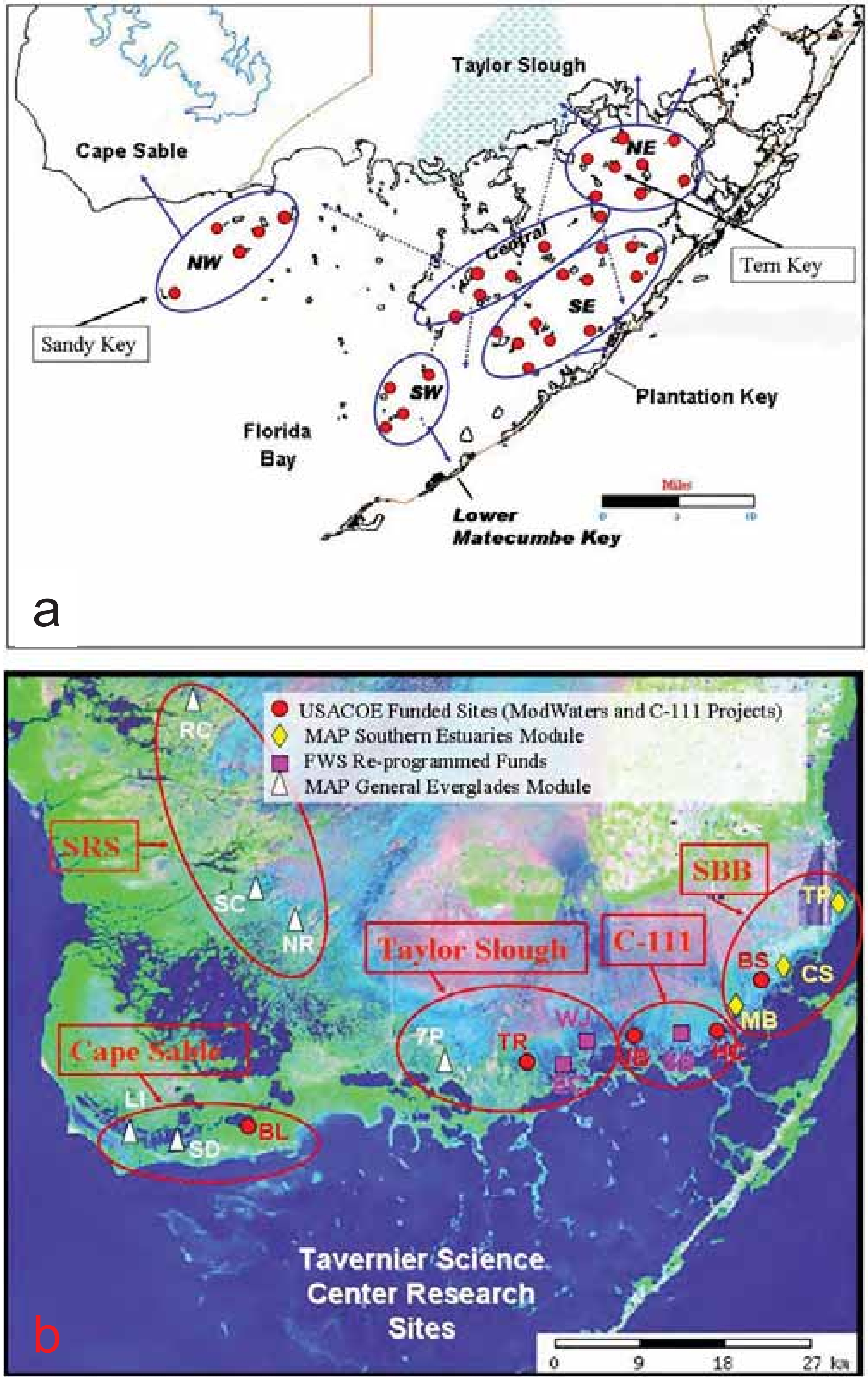
Findings from the methodologies and strategies testing showed that electrofishing catch-per-unit effort (CPUE) provided a reliable estimator of large-bodied fish abundance and species richness at salinities below 15 psu. Minnow traps CPUE provided an adequate estimate of smaller forage fish and macroinvertebrate abundance for the mangrove prop root microhabitat (Loftus and Rehage, 2007).

Results confirm a significant biotic connectivity in the Rookery Branch/Otter Creek/Squawk Creek site between freshwater marshes and the oligohaline/mesohaline mangrove habitats indicating that these habitats serve as important dry season habitat for a variety of freshwater taxa (Loftus and Rehage, 2007; Rehage and Loftus, 2008). This connectivity was not as visible in the data from the North and Roberts Rivers site. In both regions, marsh conditions, including depth and hydroperiod, were not a key factor in explaining any of the variation in the smaller fish taxa. Salinity and dissolved oxygen were more important in explaining the variability, especially in the electrofishing CPUE. The timing of when smaller marsh fish migrate to the creeks does appear to be closely tied to the pattern of water recession in the upstream marshes, with peak CPUE delayed to late spring (April and May) rather than earlier (February and March) in wet years, such as 2005. Rehage and Loftus (2008) documented a negative relationship between fish abundance and salinity in the ecotonal mangrove region (Lorenz, 1999; Lorenz and Serafy, 2006; Lorenz et al., 2009).

8.4.5 Roseate Spoonbill Nesting and Estuarine Predator-Prey Interactions

8.4.5.1 Monitoring

To determine roseate spoonbill nesting and estuarine predator-prey interactions, several parameters must be monitored. These include: 1) prey base fishes, 2) local hydrological conditions such as water depth, 3) point salinity values, 4) SAV surveys and abundance estimates co-located with prey monitoring, 5) roseate spoonbill nest counts at 39 colonies, and 6) estimates of roseate spoonbill nesting success (nest production) for select regions. Regional salinity, hydrologic and operational data were provided by various agencies, including, but not limited to, the USGS, the SFWMD, and Everglades National Park. This data integrates with ***Section 8.3 Wading Bird Nesting in Relation to the Aquatic Fauna Forage Base Hypothesis Cluster***.
Prey base fishes are monitored along the coastal floodplain wetlands from Biscayne Bay to Lostman's River. Roseate spoonbill nests and colonies are located within Florida Bay (***Figure 8-20*** and ***Figure*** 8-27).



Top Key:
Central – central Florida Bay
NE – northeast Florida Bay
NW – northwest Florida Bay
SE – southeast Florida Bay

Bottom Key:
USACOE – US Army Corps of Engineers
MAP – Monitoring & Assessment Plan
FWS – US Fish & Wildlife Service
7P – Seven Palms Lake
BL – Bear Lake
BS – Barnes Sound
C-111 – C-111 Canal
CS – Card Sound
EC – East Creek
HC – Highway Creek
JB – Joe Bay
LI – Lake Ingraham
MB – Manatee Bay
NR – North River
RC – Rocky Creek Bay
SB – Sunday Bay
SBB – south Biscayne Bay
SC – Squawk Creek
SD – Slaggie's Ditch
SRS – Shark River Slough
TP – Turkey Point
TR – Taylor River
TS – Taylor Slough
WJ – West Joe Bay

FIGURE 8-27. LOCATIONS OF A) ALL ROSEATE SPOONBILL COLONIES WITHIN FLORIDA BAY WITH ARROWS INDICATING THE PRIMARY FORAGING GROUNDS FOR EACH REGION, AND B) 17 FISH AND HYDROLOGY SAMPLING SITES DESIGNATED BY FUNDING AGENCY AND GROUPED BY FORAGE AREA

8.4.5.2 Results

It is hypothesized that successful nesting of roseate spoonbills in Florida Bay requires both the production of small marsh fishes during the wet season and a temporal sequence of prey availability, concentrating the fishes as the dry season progresses between November and April. Freshwater inflows from the watershed additionally affect the salinity and water depth in the mangrove ecotone, the primary foraging grounds of nesting spoonbills. A performance measure has been developed for roseate spoonbill nesting patterns. Documentation for this measure can be found at www.evergladesplan.org/pm/recover/recover_docs/ret/pm_ge_roseatespoonbillnesting.pdf

Hypersaline conditions in Florida Bay during the 2004-05 season resulted in a major die-off of SAV at all monitoring sites, with recovery of SAV abundance over the next few years (

Figure 8-28) as a result of localized improved conditions (Lorenz et al., 2009), despite an absence of persistent freshwater-oligohaline zones along the major creeks across most of the coastal Everglades as reported above in ***Section 8.5.2 Freshwater Inflows, Hydrologic Patterns, Salinity Distributions, and Surface Water Nutrient Concentrations***. Additionally, during this period, preliminary findings indicate an increasing number of freshwater fishes with the greatest increase in the Taylor Slough Basin. Lorenz and Serafy (2006) concluded that lower salinity levels in the estuaries of northeastern Florida Bay result in a greater biomass of prey base fish per unit area and larger proportion of freshwater prey base fish species in the prey base community.

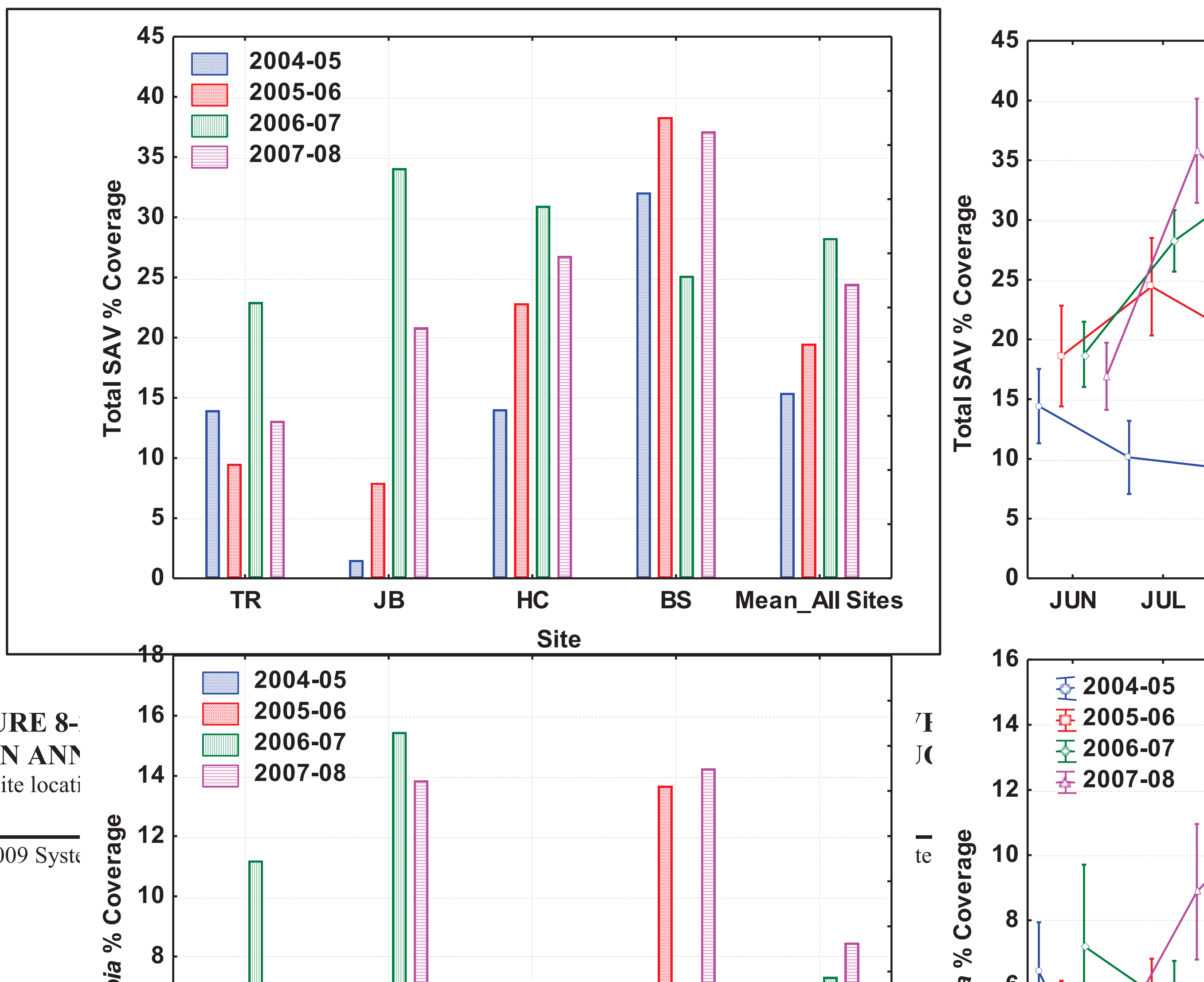


FIGURE 8-
ON AN ANN
Note: site locati

Lorenz and Frezza (2007) reported that from 1982 to 2004 only 7 of 19 primary nesting cycles could be classified as successful with success defined as more than one chick per nesting attempt. During this reporting period (2005-2008) the NE and NW spoonbill colonies (***Figure 8-27a***) produced on average 1.5 chicks per nest resulting in three consecutive successful nesting years (***Table 8-7***) (Lorenz et al, 2009).

TABLE 8-7. MEAN NUMBER OF CHICKS PER NEST FOR THE TWO LARGEST SPOONBILL NESTING COLONIES, NW AND NE, LOCATED IN FLORIDA BAY FROM 2005 TO 2008

Colony Name	Nesting Success (chicks per nest)		
	2005-2006	**2006-2007**	**2007-2008**
NW	1.3	1.7	1.8
NE	1.5	1.0	1.8

Florida Audubon researchers have been tracking spoonbill nests in Florida Bay since 1935, when, in 1935, five spoonbill nests were found on Bottle Key. Spoonbill recovery in Florida Bay was gradual with more than half of the nests located in the NE subregion (***Figure 8-29***). The number of nests in this region has followed a declining trajectory since 1981, dropping to approximately 100 nests since the 2001-02 nesting season. The drop in nests coincides with the completion of the South Dade Conveyance System in 1982. Since spoonbills live roughly 25 years and tend to return to their birthplaces to nest once they reach reproductive maturity, the trend of fewer nests since the installation of the South Dade Canal System is attributed to the continued mortality of breeding age birds combined with an absence of young birds maturing into the breeding population. Recent increases in nesting success lead researchers to hypothesize that more bird nests should appear over the next few years now that a water management strategy that consistently results in successful fledging of spoonbills in NE Florida Bay has been developed for this area (Lorenz et al. 2010). The primary purpose of the South Dade Conveyance System was to: 1) provide agricultural water supply to south Miami-Dade County, and 2) deliver mandated minimum water deliveries to Taylor Slough. The total number of nests in Florida Bay for this report was 547, 457 and 292 for nesting years 2005-06, 2006-07 and 2007-08, respectively.

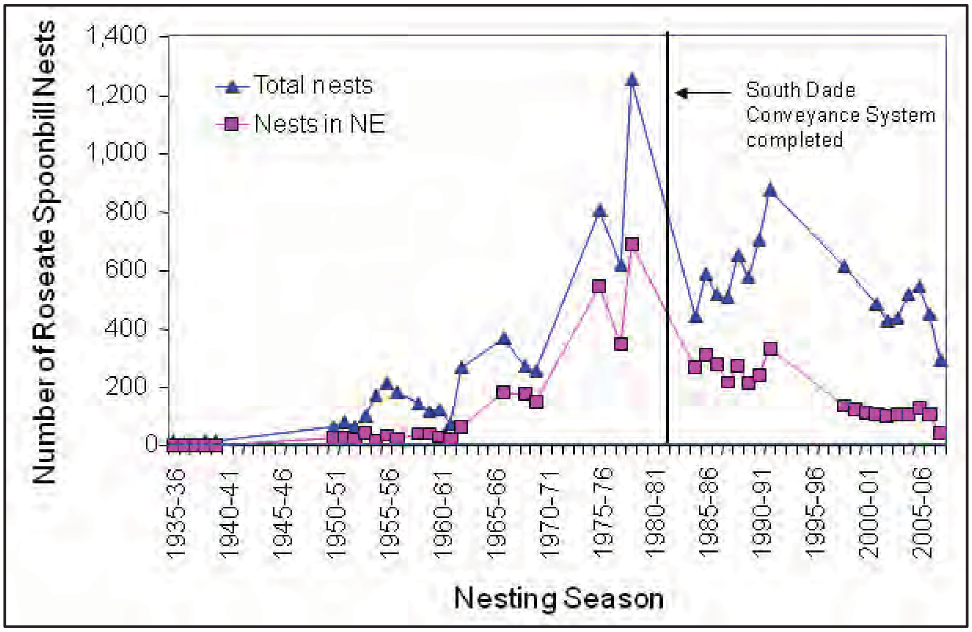


FIGURE 8-29. TOTAL NUMBER OF SPOONBILL NESTS FOR FLORIDA BAY AND NE SUBREGION COLONIES FROM 1935 TO 2006

8.4.6 American Crocodile Juvenile Growth and Survival

8.4.6.1 Monitoring

American crocodile (*Crocodylus acutus*) growth and survival along with nest location are monitored. Survey areas included Key Largo, Key Biscayne, and most of the accessible coastal and estuarine shoreline from southwest Florida around the coast to the mouth of the Miami River. This encompasses the three known nesting sites at Turkey Point Power Plant, Crocodile Lake NWR, and Everglades National Park. The surveyed area is divided into two main complexes for discussion: Everglades National Park and Biscayne Bay (***Figure 8-30***).

Florida Power and Light Company conducted nest surveys at the Turkey Point Power Plant site, and FWC conducted nest surveys at the Crocodile Lake NWR. Surveys within Everglades National Park were supported with RECOVER monitoring initiatives.

Growth and survival of juvenile crocodiles were estimated by periodic recapture of individuals that were marked between 1978 and 2002. Growth rate was determined by a change in total length for crocodiles marked at hatchlings and recaptured as juveniles. Minimal survival was defined as proportion of hatchling crocodiles known to have survived for at least 12 months.

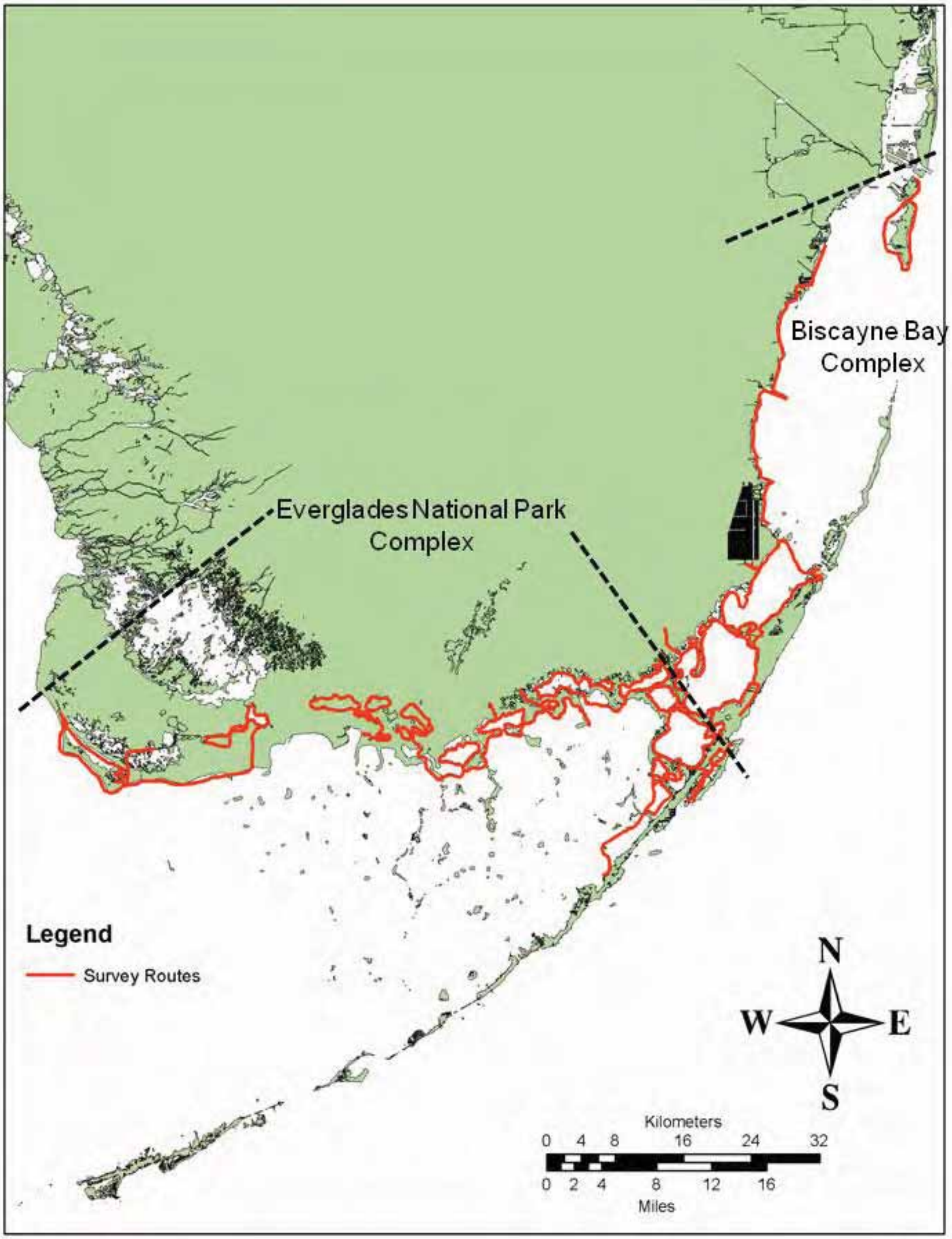


FIGURE 8-30. SPOTLIGHT SURVEY ROUTES FOR THE AMERICAN CROCODILE IN SOUTH FLORIDA IN 2008

8.4.6.2 Results

It is hypothesized that restoration of freshwater flows into the estuaries would increase growth and survival of crocodiles. A performance measure has been developed for this hypothesis: American Crocodile - Juvenile Growth and Survival. Documentation for this measure can be found at www.evergladesplan.org/pm/recover/recover_docs/ret/pm_ge_americancroc.pdf.

Growth, survival and dispersal of juvenile crocodiles were low in Everglades National Park in comparison to other primary crocodile nesting areas of south Florida during the MAP monitoring period beginning in 2005, which may be a factor of the distance hatchling crocodiles must travel from nesting locations to nursery habitats (Rice and Mazzotti, 2008). In Everglades National Park, most hatchlings were produced from shoreline nests that may be located kilometers, as opposed to meters as in Turkey Point, from nursery habitat (***Figure 8-31***).

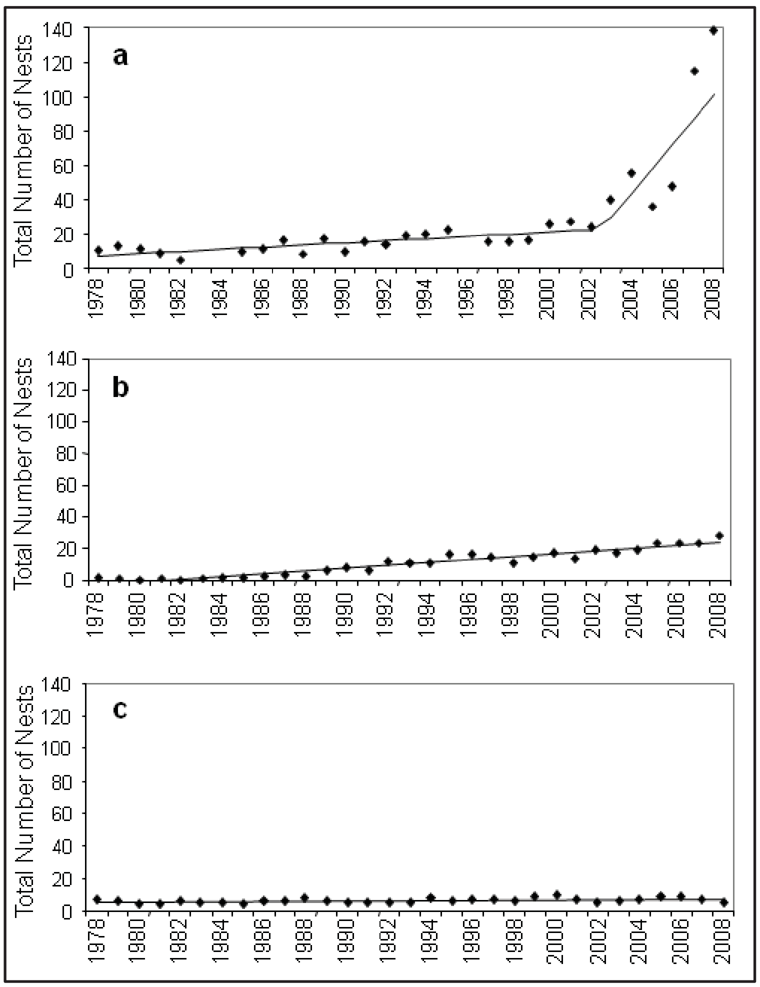


FIGURE 8-31. TOTAL NUMBER OF AMERICAN CROCODILE NESTS FOUND BETWEEN 1978 AND 2008 IN A) EVERGLADES NATIONAL PARK, B) TURKEY POINT POWER PLANT, AND C) CROCODILE LAKE NATIONAL WILDLIFE REFUGE

A trend of higher numbers of crocodile nests in Everglades National Park since 2000 resulted mainly from increased nesting on artificial substrates on Cape Sable after the plugging of the East Cape Canal, which blocked saltwater intrusion into the interior of the cape. Nest numbers in other primary crocodile nesting areas of south Florida did not show the magnitude of increase that was observed on Cape Sable.

8.4.7 Summary

8.4.7.1 Assessment of Validity of Working Hypotheses

The EDEN-derived hydroperiod and water depth maps presented in ***Section 8.2 Sheet Flow and Water Depth Patterns Hypothesis Cluste***r provides evidence that during this reporting period, compartmentalization and water operations/management of the system have disrupted sheet flow and connectivity throughout most of the Everglades ecosystem, especially affecting the hydroperiods of the Shark Slough complex. This is further supported by the discharge distribution values presented in this chapter (***Figure 8-21***) indicating that Shark River currently provides less than 15 percent of the volume discharged from this complex. Likewise, the discharges from Taylor River account for approximately ten percent of the total discharge from the Taylor River/C-111 drainage complex.

This reduction in quantity and timing of fresh water to the coastal gradient area results in a disruption of the oligohaline zone, which is supported by salinity data showing the lack of a persistent low salinity gradient along the marsh-mangrove interface. The data further indicates that the disruption in the quantity and timing of fresh water to the coastal regions affects both SAV cover (

Figure 8-28) and fish biomass. In agreement with the findings of Lorenz and Serafy (2006), this monitoring data corroborates that, in the mangrove coastal ecotone, fish biomass was inversely correlated to salinity values (Rehage and Loftus, 2008; Lorenz et al., 2009).

Lorenz (2000) and Lorenz et al. (2009) documented that prey-based fishes located within the spoonbill primary feeding grounds begin to concentrate in deeper creeks and pools when surface water depths on the surrounding wetlands drops to 12.5 centimeters, validating that quantity and timing of flows is a critical aspect of spoonbill foraging success.

Mazzotti et al. (2007) documented that most crocodile sightings are in water of salinity values less than or equal to 20 psu and that increasing fresh water to crocodile nesting sites will increase the number of nests (***Figure 8-31***) and improve survival rate.

8.4.7.2 Assessment of Status and Trends

Crocodile and spoonbill population health and stability in the Everglades coastal wetlands ecosystem are at risk. Causes of the existing risk are altered hydrology (Lorenz et al., 2002; Mazzotti et al., 2007; Lorenz and Frezza, 2007; Lorenz et al., 2008; Doren et al., 2009). MAP monitoring corroborated this assessment and validated the linkages between hydrology and these indicator species. Growth, survival and dispersal of juvenile crocodiles were low in Everglades

National Park in comparison to other primary crocodile nesting areas of south Florida during the MAP monitoring period beginning in 2005. The number of spoonbill nests in Florida Bay has been declining since 1984, dropping to approximately 100 nests since the 2001-02 nesting season. The total number of nests in Florida Bay for this report was 547, 457 and 292 for nesting years 2005-06, 2006-07 and 2007-08, respectively.

However, indications for optimism are observed. Lorenz et al. (2009) established that during this reporting period (2005-2008) the NE and NW spoonbill colonies produced on average 1.5 chicks per nest resulting in three consecutive successful nesting years (***Table 8-7***), which is greater than the minimum indicator to be rated as successful. Preliminary data from the 2009 nesting season indicated another successful nesting year – resulting in four consecutive successful nesting years. Lorenz et al. (2009) attributed this recent increase of nesting success directly to naturally ideal rainfalls in addition to recent (since 2005-06) changes in water deliveries in the Taylor Slough/C-111 Basin that have resulted in persistent dry downs with no reversals caused by out-of-phase releases from the C-111 Basin.

The trend of higher numbers of crocodile nests in the Everglades National Park since 2000 resulted mainly from increased nesting on artificial substrates on Cape Sable. After the plugging of the East Cape Canal, which blocked saltwater intrusion into the interior of the cape. Mazzotti et al. (2007) hypothesized that plugging the canals in the Cape Sable area would result in reduced saltwater intrusion thus creating more suitable nesting habitat for the crocodiles in this area.

In both cases, the measurable response to a management action demonstrates the ability to detect change in crocodile and spoonbill nesting resulting from altered hydrology and salinity regimes.

8.4.7.3 Key Management Recommendations

Ecological results from Everglades coastal wetlands support the broader management recommendation from the sheet flow and water depth patterns hypotheses to restore rainfall-driven volume, timing and distribution of sheet flow through WCA 3A and WCA 3B into Everglades National Park to produce water depths, hydroperiods, and flow patterns that are consistent with the best understanding of how the pre-drainage system responded to seasonal and interannual variability in rainfall. This includes redistribution of sheet flow through the natural flow corridors of the Shark Slough complex and Lostman's Slough in Everglades National Park in order to restore multi-year hydroperiods in slough systems of the park.

Mazzotti et al. (2007) define restoration as Taylor Slough being the main source of fresh water for natural hydropatterns restoration for crocodiles in the eastern and central Florida Bay areas rather than freshwater inputs from the C-111 Canal. Specifically, these flows are needed to restore early dry season flow (October to January) to Florida Bay. Measurable hydropattern objectives of success would be a fluctuating mangrove back country salinity that rarely exceeds 20 psu in northeastern Florida Bay.

Lorenz et al. (2009) described recent changes in water deliveries to the Taylor Slough/C-111 Basin and the resulting persistent dry downs with no reversals caused by out-of-phase releases from the C-111 Basin. Lorenz et al. also stated that long-term achievement of this goal requires restoring natural rain-driven patterns.

It must also be understood that assessment of the RECOVER monitoring data is able to detect ecological responses to recent operational changes in water deliveries only because of years (decades in many cases) of long-term uninterrupted ecological monitoring from several agencies. This data was compiled and utilized to quantify the natural variability due to climatologic drivers, such as decadal rainfall patterns or intrinsic behavioral patterns not yet fully recognized.

The Everglades system is not operating in a sleep pattern awaiting restoration, but is a dynamic system that continues to respond to hydrologic drivers. Monitoring is needed to track these changes if documentation of restoration is an ultimate objective.

8.5 LANDSCAPE PATTERNS OF RIDGE AND SLOUGH PEATLANDS AND ADJACENT MARL PRAIRIES IN RELATION TO SHEET FLOW, WATER DEPTH PATTERNS AND EUTROPHICATION HYPOTHESIS CLUSTER

8.5.1 Introduction and Background

The landscape monitoring projects described herein have received MAP support over the last one to three years. This natural grouping of monitoring activities are designed to achieve a synthetic understanding of the changes to the landscape as they occur and a clear understanding of the causes of these changes to the maximum possible extent. Paleoecological studies provide valuable insight into how the regional system responded to the changing climatic conditions that led to the formation and development of landscape features and the distribution of community types prior to the establishment of regional water management infrastructure. When coupled with historical aerial photos, this information provides a strong basis for developing hypotheses about the causes of changes in the landscape that have occurred subsequent to the construction and operation of the C&SF Project (Nungesser, 2009) (***Figure 8-32***). However, it is important to complement these monitoring efforts with specific investigations of the biogeochemical processes that support the establishment and sustainability of Everglades communities. Therefore, the objectives of the MAP landscape monitoring program are to identify: 1) the processes that lead to the formation and persistence of the ridge and slough, tree islands and marl prairie habitats of the Everglades, and 2) a set of indicators that effectively represent the functioning of these processes and can be used to quantify CERP effects.

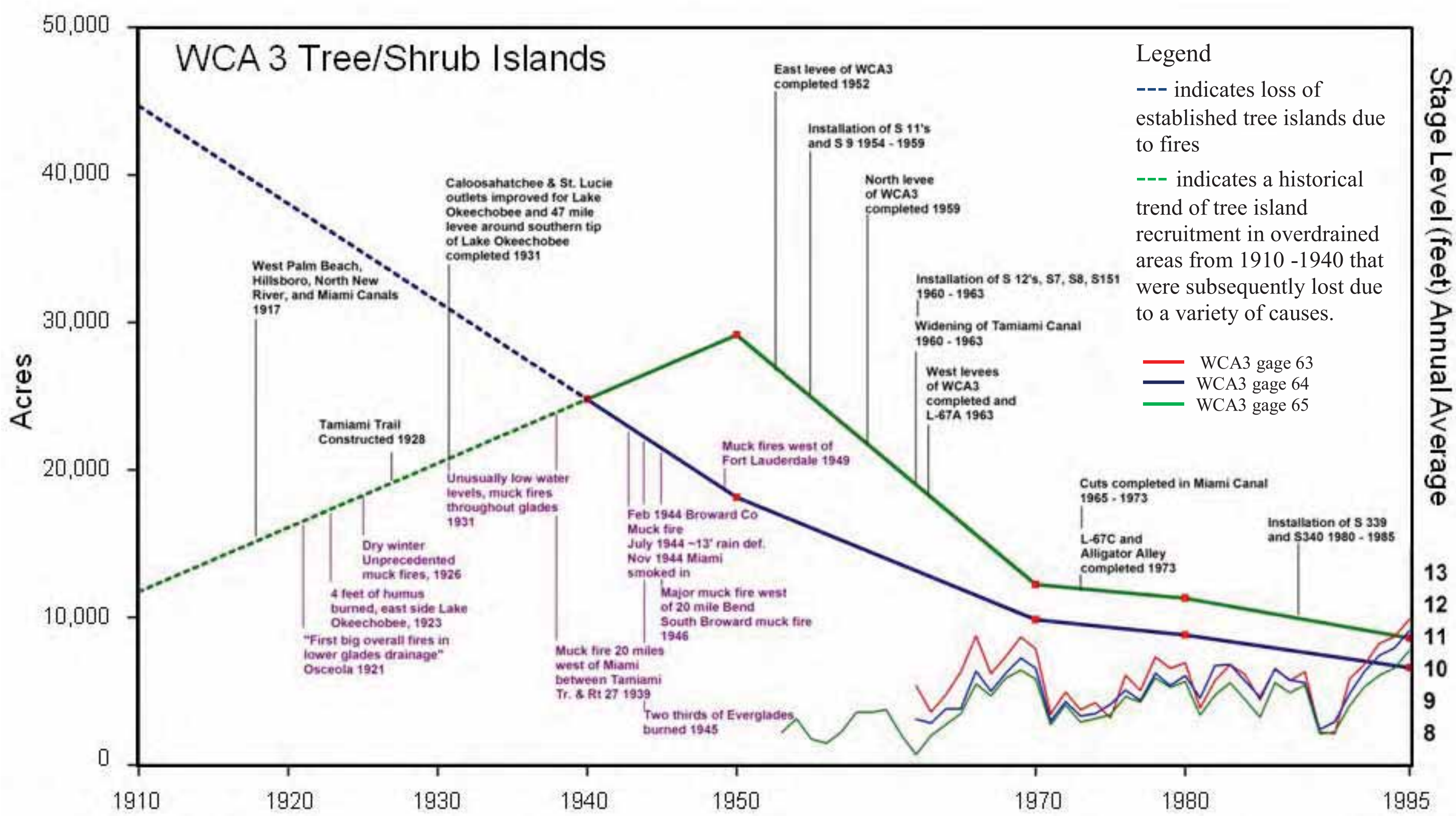


FIGURE 8-32. THE CHANGES IN TREE ISLAND AERIAL EXTENT IN WATER CONXERVATION AREA 3 SINCE 1910 ARE CORRELATED WITH THE COMPLETION OF CENTRAL AND SOUTHERN FLORIDA PROJECT FEATURES

(F. Sklar and K. Rutchey, SFWMD, personal communication)

Figure 8-2 in ***Section 8.2 Sheet Flow and Water Depth Patterns Hypothesis Cluster*** describes the effects sheet flow, water depth and eutrophication have on landscape patterns of ridge and slough peatlands and adjacent marl prairies. Restoration of the Everglades ecosystem depends on restoring the volume, timing and distribution of sheet flow and on restricting inputs of P and other chemical constituents to levels approximating those in direct rainfall. Sheet flow and inputs of P fundamentally affect all the hypotheses described below. Hydrology and water quality data associated with the MAP underpin, and are integrated into, the assessment of all other hypothesis clusters in the Greater Everglades Wetlands. A primary product of the integrated hydrology monitoring effort is EDEN. Another product is the system-wide map of soil nutrients reported in the 2006 SSR (RECOVER, 2007a). This was followed by a recent analysis of variance in the soil nutrient data (presented below) to ascertain the implications for interpreting repeated system-wide monitoring results.

Another CEM describes Everglades ridge and slough landscape dynamics (***Figure 8-33***). Degradation of microtopography, changes in hydroperiod and water depth, eutrophication, changes in fire and vegetation, especially non-native and invasive species, can reduce the diversity and stability of habitats that were previously long-term, large-scale features of the ridge and slough landscape. Organic soil accretion and loss also affect microtopography. Sheet flow interacts with hydroperiod, water depth, fire and nutrient dynamics to maintain organic soil accretion and loss in a state of dynamic equilibrium.

Figure 8-34 presents another CEM that describes cause-and-effect relationships relating to plant community dynamics along elevation gradients. The composition and distribution of plant communities along elevation gradients are determined by the interaction of several factors including hydroperiods, seasonal average water depth conditions, maximum and minimum water depths, nutrient dynamics, and fire frequency and intensity throughout freshwater wetlands of the Greater Everglades.

An additional hypothesis has been developed for N dynamics in the Everglade**s**. N dynamics, such as rates of nitrification, denitrification, uptake of inorganic N (IN) species, and production of organic N, are dominated by local cycling and processing under natural conditions in the Everglades. N dynamics play a big role in determining how the wetland system contributes to or offsets greenhouse gas emissions. A CEM has not been developed specifically for this hypothesis.

Performance measures have been developed for landscape patterns in the Greater Everglades Wetlands. These include measures for freshwater and estuarine vegetation mosaics, ridge and slough community sustainability, wet prairie, and marl prairie Cape Sable seaside sparrow habitat. The performance measure for TP concentrations in soil also pertains to this chapter. Documentation sheets for these measures can be found online at www.evergladesplan.org/pm/recover/perf_ge.aspx.

Interim goals were developed for: 1) ridge and slough patterns and 2) tree islands. Documentation for these interim goals can be found online at www.evergladesplan.org/pm/recover/igit_subteam.aspx.

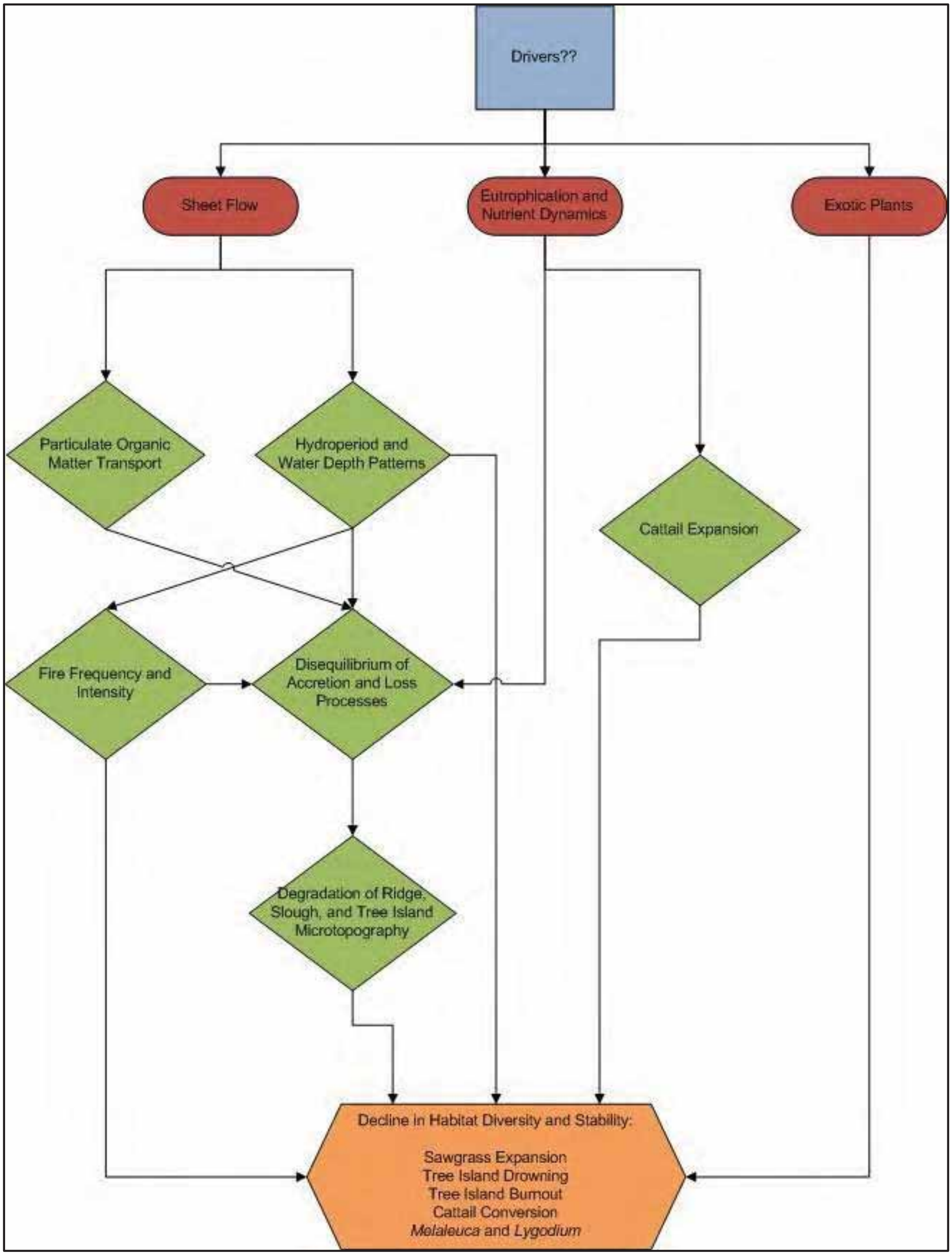


FIGURE 8-33. CONCEPTUAL ECOLOGICAL MODEL FOR RIDGE AND SLOUGH LANDSCAPE DYNAMICS IN THE GREATER EVERGLADES WETLANDS

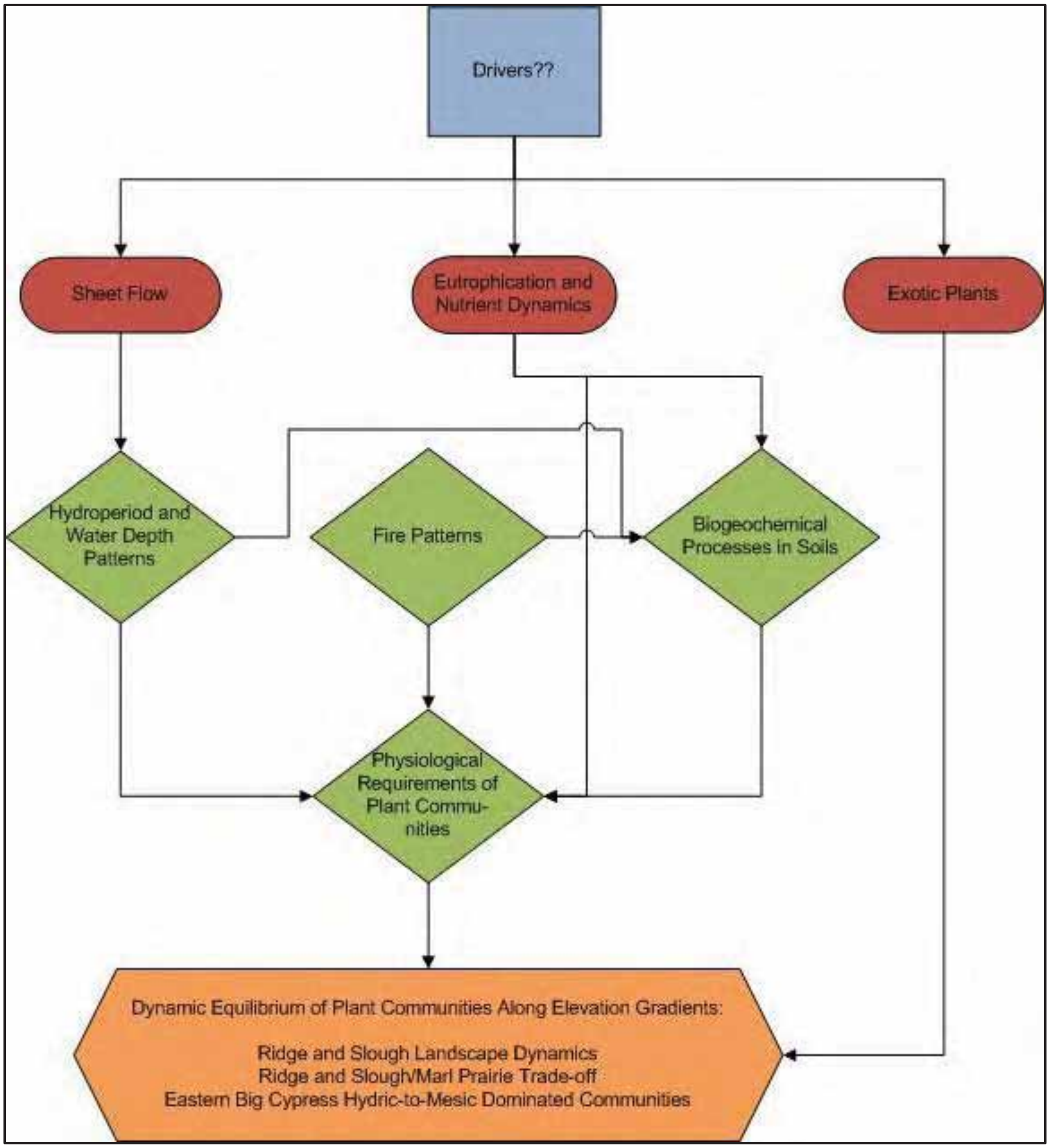


FIGURE 8-34. CONCEPTUAL ECOLOGICAL MODEL FOR PLANT COMMUNITIES ALONG ELEVATION GRADIENTS IN THE GREATER EVERGLADES WETLANDS

8.5.2 Decomposition Dynamics, Community Composition and Ridge-Top Senescence in the Ridge and Slough Mosaic

8.5.2.1 Monitoring

Vegetation-hydrology-nutrient interactions in the ridge and slough habitat of the WCAs and Everglades National Park are monitored using a combination of spatially structured measurements conducted at eight disparate locations throughout the Everglades Protection Area, and the execution of a designed and replicated soil oxidation experiment. The objective is to categorize species composition and patterning in several locations distributed throughout the Greater Everglades Wetlands, and relate these observations to the peat soil accretion processes of ridge and slough formation. Soil oxidation rates are hypothesized to increase as a function of reduced water depths, shortened hydroperiods, and increased nutrient loading. The magnitude of increased oxidation is likely to change based on the interaction of the factors listed above, and the designed/replicated soil oxidation experiment allows for detection of significant differences between oxidation rates in the full array of hydrologic and nutrient treatment combinations. The initial three-year pilot study is nearing completion. The results from this study and from several RECOVER workshops have been used in the design and implementation of a broader, multi-year landscape monitoring program (***Figure 8-35***). Over the next five years, the landscape monitoring program will expand to include marl prairie regions, tree islands, Loxahatchee NWR, and coastal freshwater wetlands in Everglades National Park.

The pilot study was conducted in the southern Everglades spanning several hydrologic conditions (***Figure 8-36***) to provide the basis for a peat soil accumulation performance measure. The photos in the figure were taken in 2004. Light tan colors are ridges. Sloughs are either dark when periphyton is absent or very light when periphyton is present. Green areas are tree islands. The stabilized flow site has a fire scar across the middle of the sampling block. Sampling was done in the WCAs on a single day between September and December 2007, and Everglades National Park was sampled in November 2008

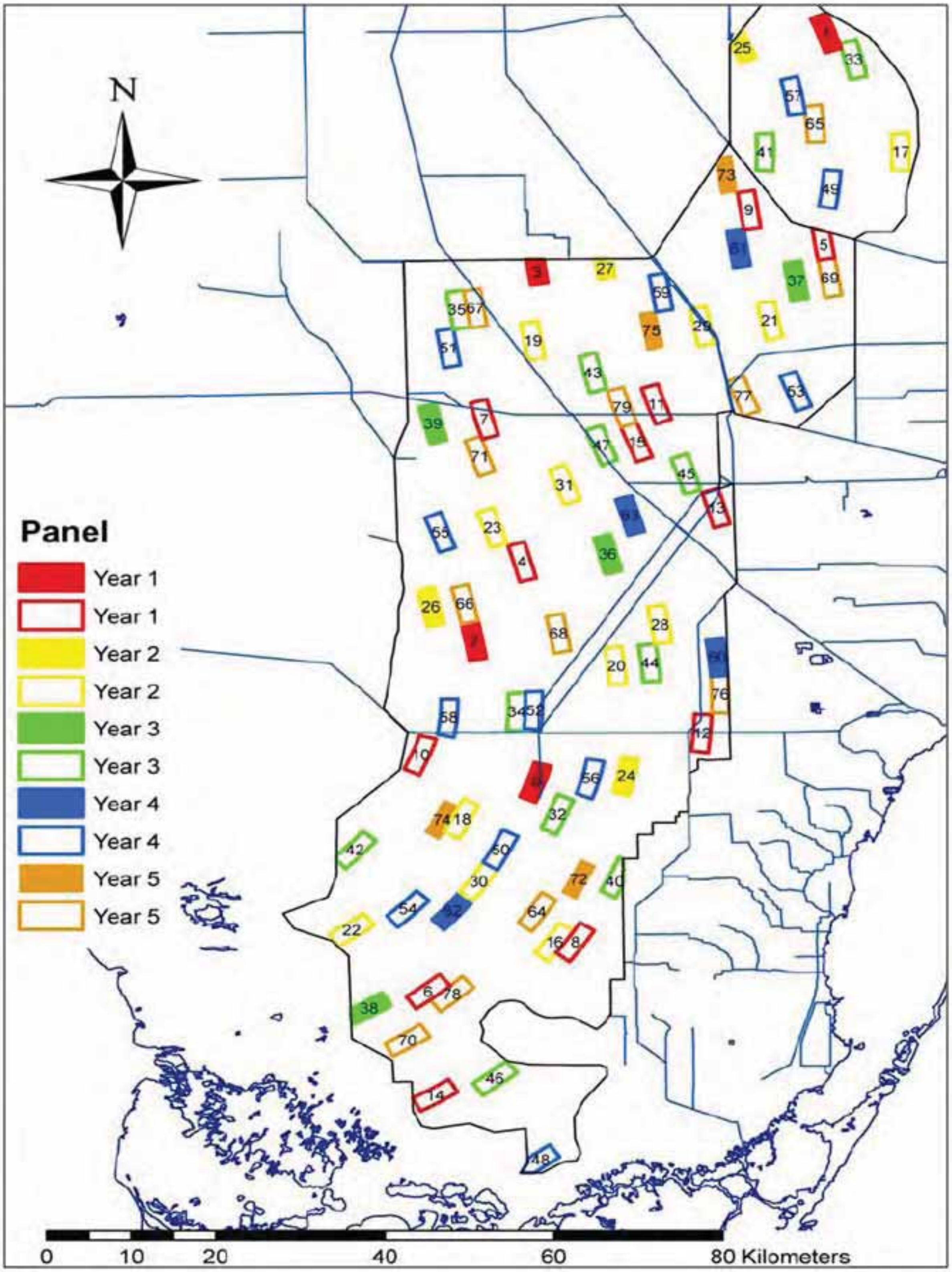


FIGURE 8-35. SAMPLING DESIGN FOR MONITORING RIDGE, SLOUGH AND TREE ISLAND DYNAMICS IN THE RIDGE AND SLOUGH LANDSCAPE

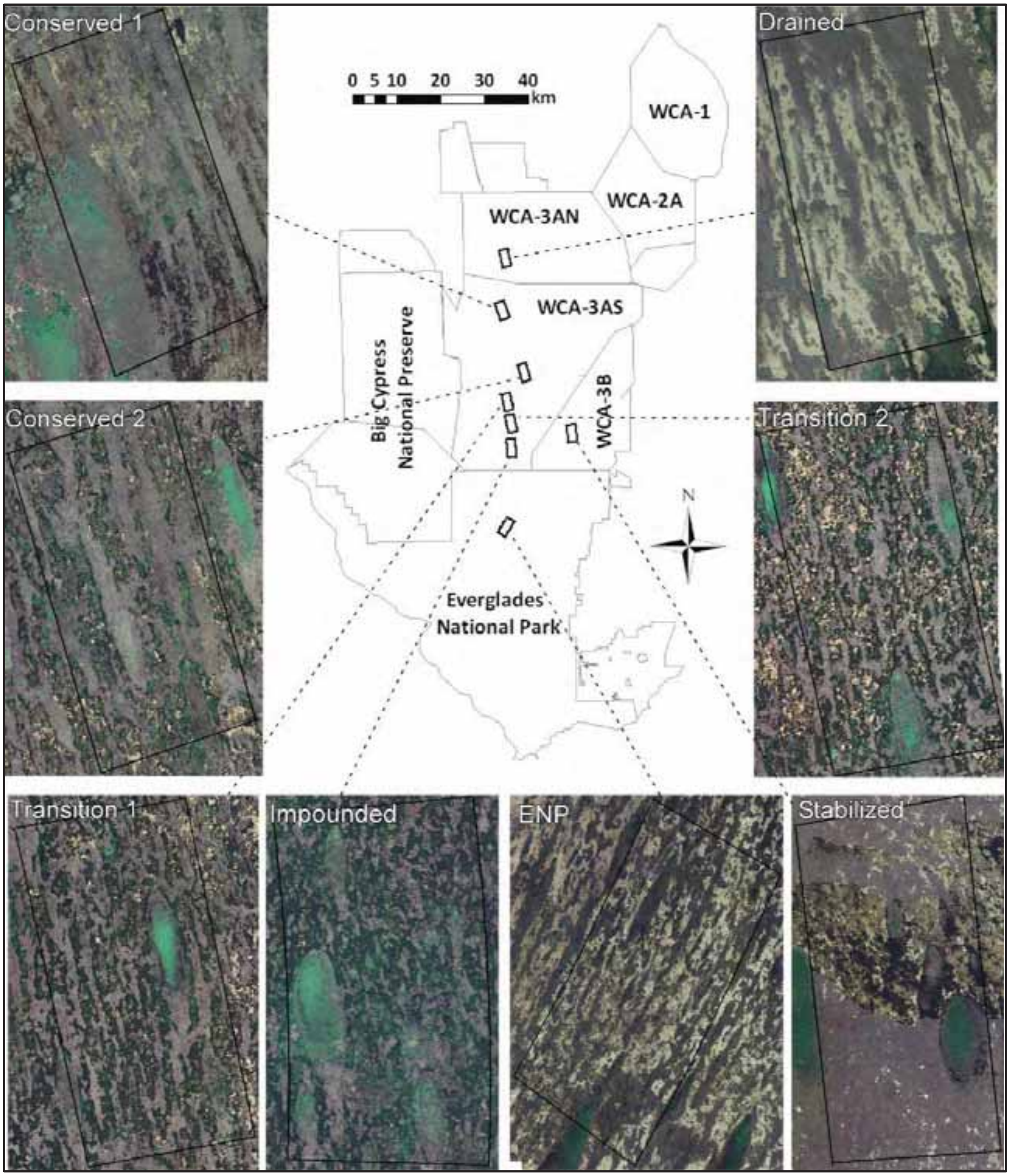


FIGURE 8-36. LANDSCAPE PRIMARY SAMPLING UNITS FOR VEGETATION-HYDROLOGY-NUTRIENT INTERACTIONS IN RIDGE AND SLOUGH LANDSCAPES

At randomly selected sites (tree islands were not sampled), water depth measurements were made and all species and percent cover of dominants were noted. Soil and porewater samples were collected from four primary sampling units, representing drained, conserved (n=2) and impounded conditions. At each site, four gradient transects were established spanning from ridge center to the adjacent slough center. Two gradient transects were established lateral to prevailing flow, one at the ridge head and one at the ridge tail (***Figure 8-37***). In addition, two center transects were established in the interior of the ridge and adjacent slough.

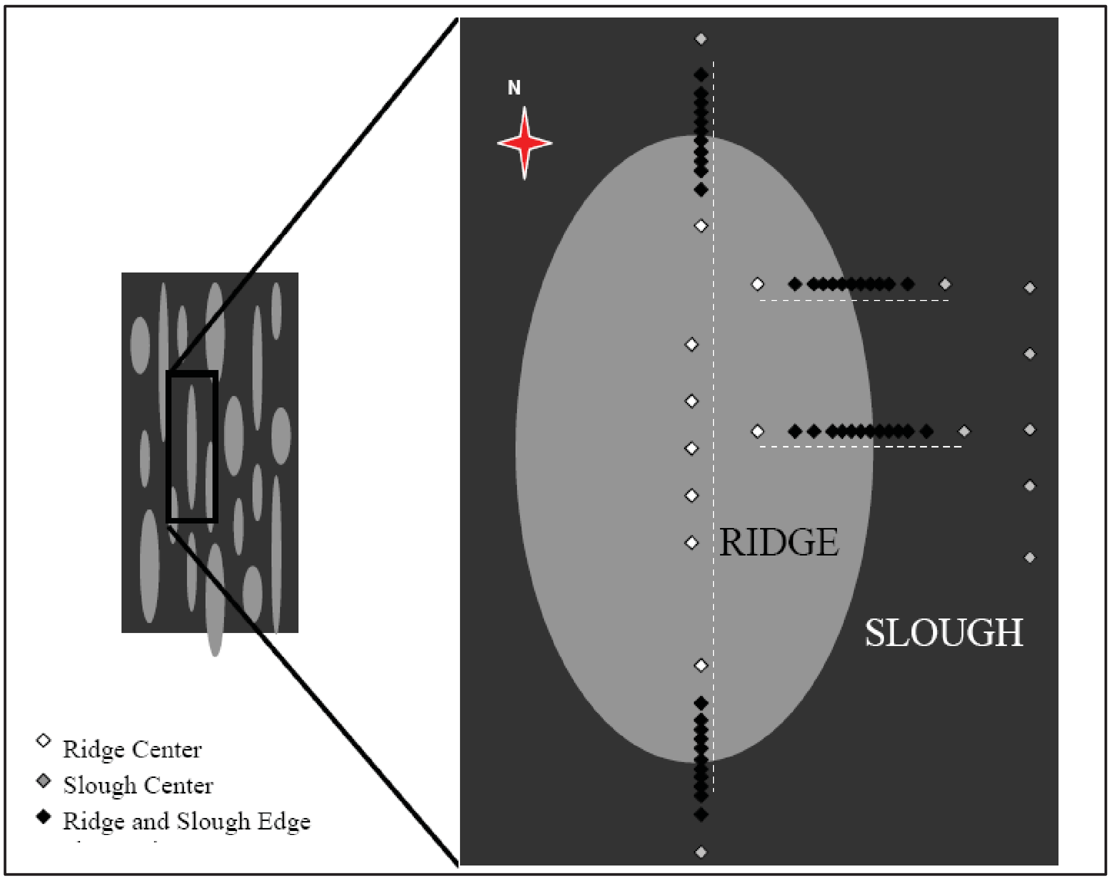


FIGURE 8-37. SCHEMATIC OF SAMPLING DESIGN FOR EVALUATING RIDGE EDGE MAINTENANCE

Mean daily water depth data for the eight years preceding sampling, June 30, 2000 through June 29, 2008, were obtained from interpolated data between EDEN monitoring stations (sofia.usgs.gov/eden/). Both real observation and model-hind casted predictions were used (Conrads and Roehl, 2007). Water depths were converted to water surface elevations in North American Vertical Datum (NAVD) 88 using the EDEN landscape model. The difference between water elevation and measured water depth at each sampling location is the peat elevation. All subsequent analyses used this peat elevation. Historical water elevation data were used to compute a cumulative elevation exceedance probability curve, which, when compared with the peat elevation probability density function, yield the inundation probability, and thus long-term mean hydroperiod, for each peat elevation.

A recently developed ridge and slough CEM proposes that autogenic feedbacks maintain patterning in this system. To maintain landscape patterning, these interactions should lead to similar long-term accretion rates in ridges and sloughs, despite strongly differential primary productivity. This conceptual model suggests that changes in hydrologic conditions can cause shifts between the two ecosystem states defined by ridges and sloughs. The bi-modality in the distribution of peat elevations in the ridge and slough habitat was tested by comparing the probability density function of observed peat elevations to distributions predicted by: 1) a single normal and 2) a mixed double normal model. Patterning and spatial structure was explored using semi-variograms, which describe the degree of spatial dependence between paired observations as a function of distance between them (Goovaerts, 1997).

8.5.2.2 Results

In conserved and transitional sites, the peat elevation density function exhibited distinct bi-modality (***Figure 8-38***). This bimodal signature is lost under drained and impounded hydrologic conditions. Kurtosis, which is the degree of flatness or peakedness of the elevation frequency distribution, follows a similar trend (***Figure 8-38***, values of g), in which drained blocks have positive values, conserved blocks have negative values (i.e. density functions in these areas are bi-modal) and impounded blocks exhibit a shift back towards positive values. Model selection for the Everglades National Park site (labeled ENP in ***Figure 8-36***) was less clear and suggested nearly equivalent data consistency with single and mixed normal distributions.

Significant differences in mean peat elevation were observed between ridge and slough communities across all sampling blocks, with differences between community types ranging from 6 to 25 centimeters. However, the separation of means decreases with hydrologic impairment (i.e. drainage and impoundment). This indicates greater overlap in the range of water depths between communities. The variance in peat elevations in both communities increases with impoundment, but appears to decrease under drained conditions.

The accuracy of the site designations was evidenced by the hydrologic conditions in each landscape block. In blocks where bi-modality is well conserved (Conserved 2, Transition 1 and 2; all in WCA 3A south), the inundation probability for sloughs is 100 percent, and the probability of exposure increases markedly over the extant range of peat elevations. Inundation probabilities at the drained site (WCA 3A north) match expectations; nearly the entire unimodal peat elevation distribution is exposed at some point (<100% inundation probability), though the inundation probability for the mean block elevation still exceeds 80 percent. In contrast, frequent peat exposure is not observed in the shallow stabilized flow site in WCA 3B where sawgrass encroachment is nearly complete. That site is almost permanently inundated, suggesting, surprisingly, that sawgrass cover can expand under those conditions. Moreover, the median inundation depth, which is the depth at 50 percent probability of water level exceedance in ***Figure 8-38*** is 30 centimeters, a value strikingly similar to median inundation depths observed in Conserved 2 and Transition 1 and 2 sites. Conversion to an increasingly slough-dominated landscape, as appears to be occurring at the impounded site, is similarly related to the water depths and duration. In this case, nearly all peat elevations within the block are permanently inundated, and the median inundation depth is 20 centimeters higher than that of the other sites.

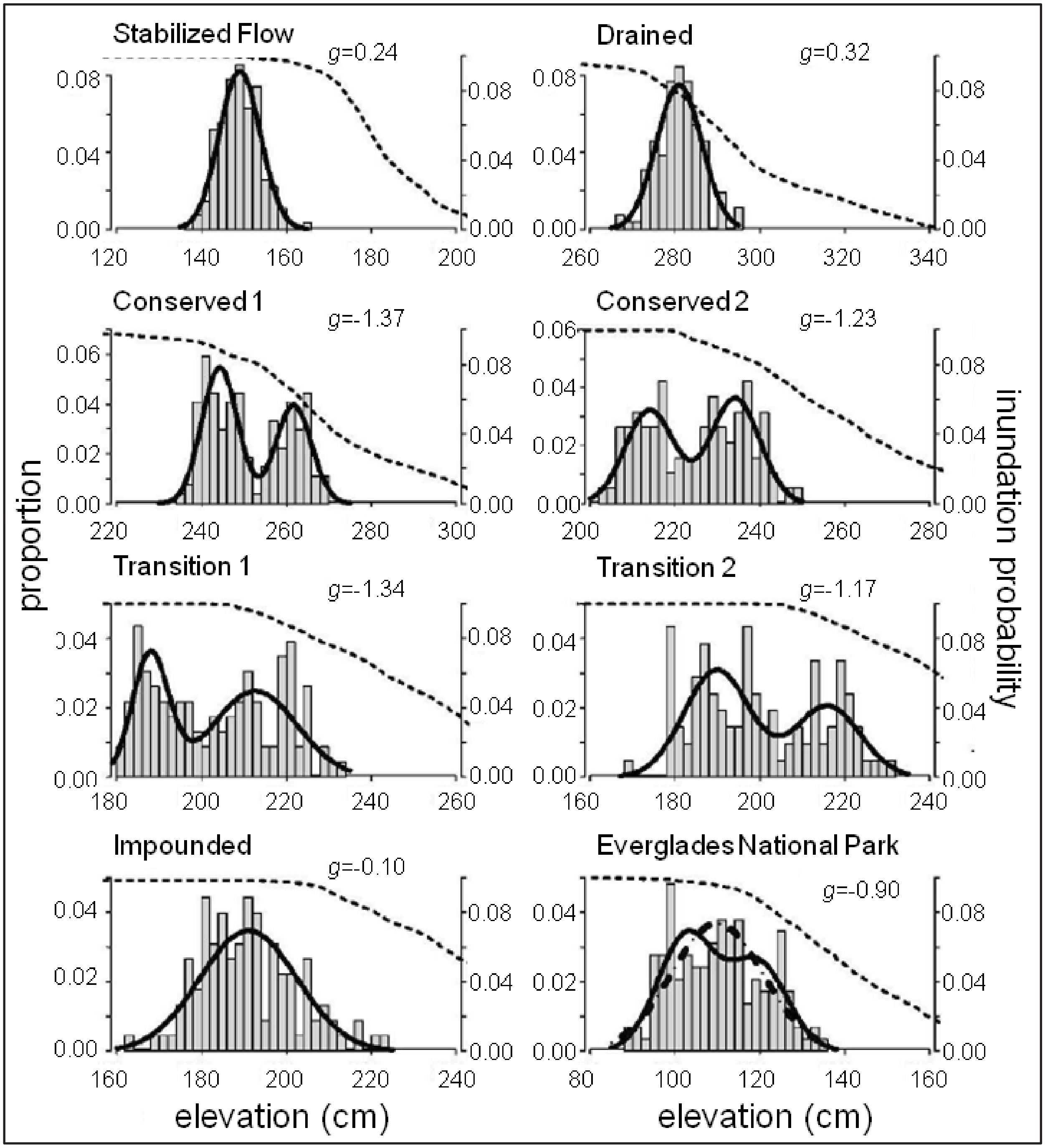


FIGURE 8-38. PEAT ELEVATIONS WITHIN EACH LANDSCAPE BLOCK ARE SHOWN AS HISTOGRAMS WITH SOLID LINES INDICATING THE PROBABILITY DENSITY FUNCTION OF THE BEST MODEL, "G" INDICATING KURTOSIS, AND DASHED LINES INDICATING THE WATER ELEVATION EXCEEDANCE PROBABILITY

The landscape prevalence of vegetative community types varied with position within the hydrologic gradient. Ridges were substantially more prevalent in the stabilized flow site (WCA 3B) compared to other sites (***Figure 8-39***). Ridge prevalence decreases systematically with impoundment, while slough prevalence increases. When sloughs are partitioned into wet prairies (dominated by emergent vegetation) and deep sloughs (submerged and floating-leaved aquatics), a clear decline in the prevalence of wet prairies is observed from conserved through impounded conditions.

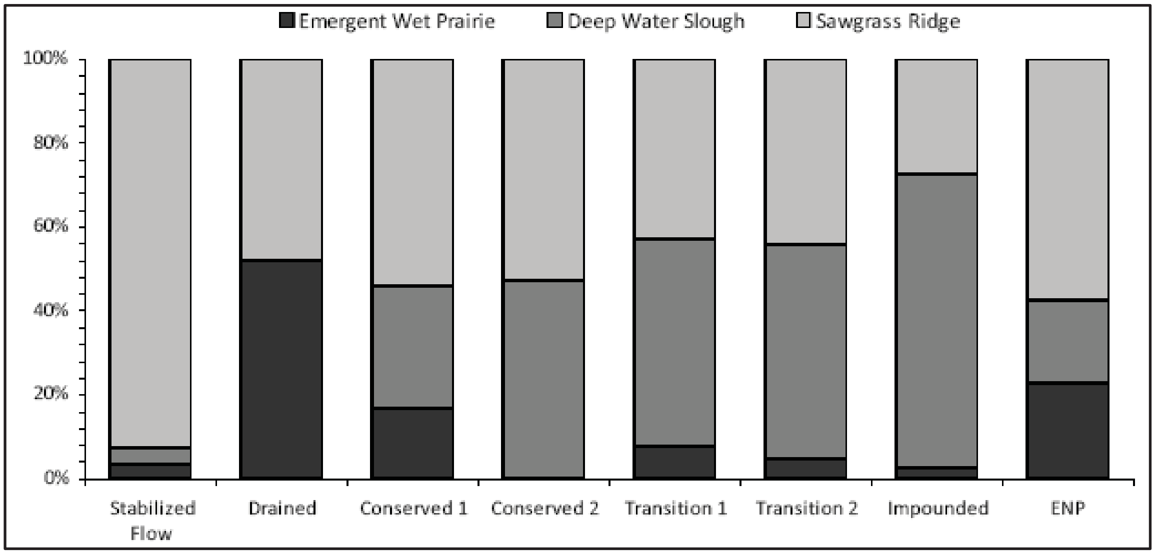


FIGURE 8-39. RELATIVE INCIDENCE OF VEGETATION COMMUNITIES BY SAMPLING UNIT

Short-range autocorrelation in marsh elevation was high for the conserved blocks and the Transitional 1 block (between 0.83 and 1); in contrast, short-range autocorrelation was lower in the drained and impounded blocks (between 0.5 and 0.75) (***Figure 8-40***). Anisotropic autocorrelation (defined as varying uniquely as a function of direction of measurement) trends generally differ when measured in either parallel or orthogonal to the prevailing flow directions as indicated by the directional orientation of the ridges and sloughs. In all cases, autocorrelation drops off to values not significantly different from zero with some periodicity. Under more conserved hydrologic conditions (Conserved 1, 2; Transition 1) autocorrelation becomes significantly negative, and at differing distances in the parallel and orthogonal to flow directions. Denser sampling may have yielded a clearer pattern. Observations of the autocorrelation range, which is the duration of one complete cycle of an oscillation, was short (<100 meters) in the impounded sites, moderate (100-200 meters) in the stabilized and drained sites, and longest (>220 meters) in the sites that are relatively well conserved. Given the spatial proximity of Transition 1 and Transition 2, this area is clearly one worthy of regular monitoring.

Transect data were collected to test the plausibility of several hypotheses related to the development of ridge and slough microtopography in relation to organic soil accretion and loss**.** The results reveal soil TP values along ridge-slough gradients are a suitable indicator of hydrologic disturbance, but the patterns observed do not support the hypothesis that differential ridge formation is the predominant mechanism driving the formation of ridge-slough habitats.

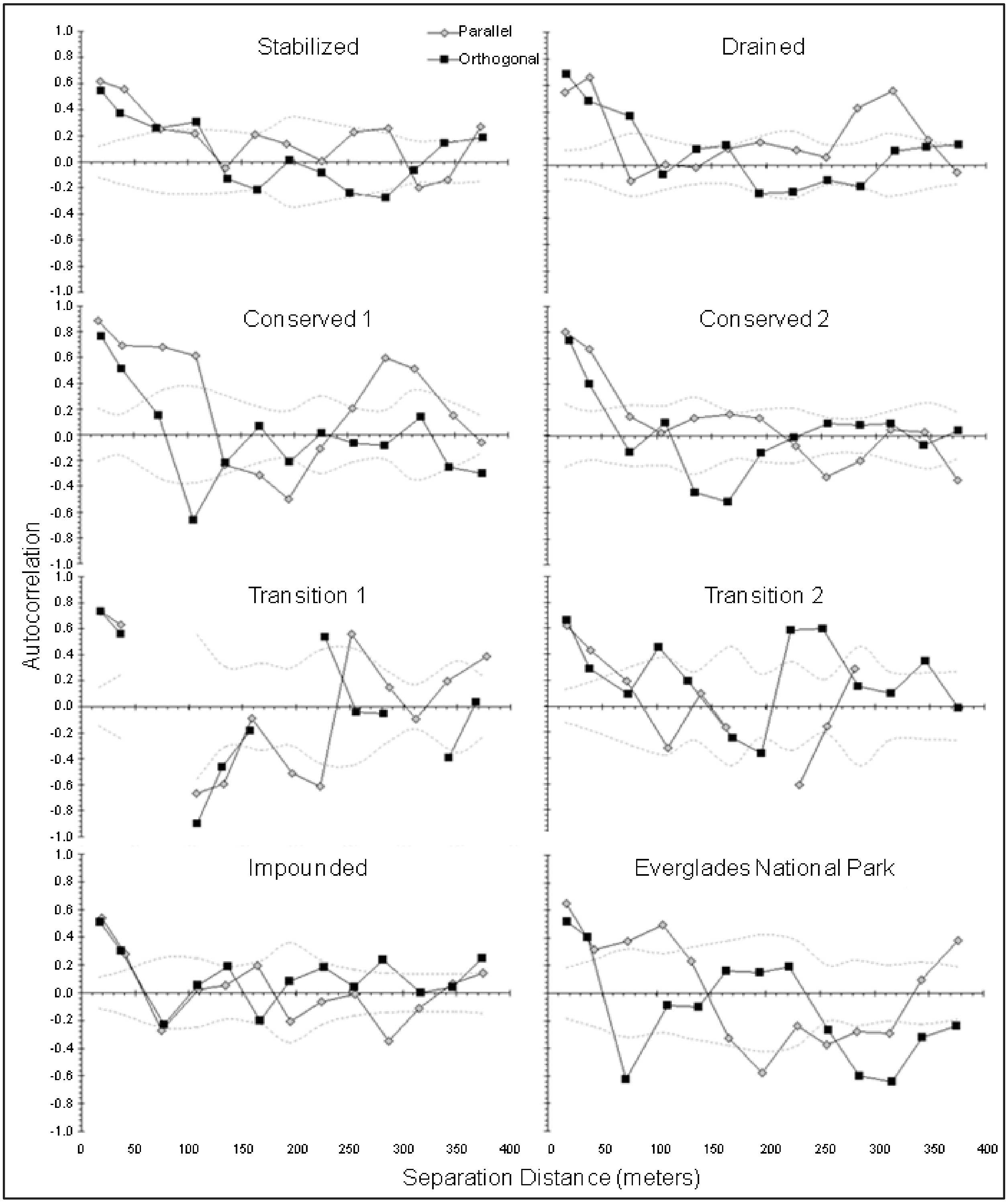


FIGURE 8-40 CORRELOGRAMS OF WATER DEPTHS WITH LAG DISTANCES OF H = 30 METERS WITH THE 95% CONFIDENCE LIMITS OF THE CROSS-CORRELATION VALUES BASED ON THE NUMBER OF PAIRS WITHIN EACH LAG CLASS H AND THE DISTANCES ALONG THE X-AXIS ARE THE AVERAGE DISTANCES AMONG ALL PAIRS WITHIN EACH LAG CLASS

Transect data show no evidence to support ridge formation mechanisms through inorganic material subsidy. While there are strong differences between ridge and slough calcium content, they are not structured in the manner that would be expected if calcium subsidy was a significant component of ridge edge maintenance.

A third hypothesis for why discrete edges are observed between ridges and sloughs deals with entrainment of organic material in the sloughs, and deposition in ridge edges. The data show strong stoichiometric differences between ridge and slough vegetation and soils, and carbon (C):N and N:P ratios from ridge and slough edges suggest of a mixture of the two end members (***Figure 8-41***). These observations, therefore, support the hypothesis that organic matter exchange between the ridges and sloughs is a primary mechanism in edge formation. However, since the differences among the end members is driven by stoichiometric differences among dominant vegetation, it remains unknown whether the intermediate values in C:N and N:P observed at edge sites are due to mixing between organic material or because of stoichiometric differences in the vegetation at edges vis-à-vis the centers. These data analyses are forthcoming. Data on differential decomposition is currently being analyzed, with the results expected to provide insight into the role of differential peat decomposition and accretion in maintaining ridge and slough patterns, and how these processes may change with hydrologic disturbance. The results of the soil oxidation experiment are likely to be integrated with the results from the WCA 3 Decompartmentalization Physical Model to provide conclusive evidence about the relative roles of differential peat accumulation and entrainment of organic material in supporting the sustainability of the ridge-slough and tree island landscape type.

8.5.3 Tree Island Condition in the Southern Everglades

8.5.3.1 Monitoring

Tree island elevations have been measured on more than 250 islands in WCA 3 and Everglades National Park. These elevation data are used to calculate tree island hydroperiod and water depths using time series of the EDEN water surfaces at each of the tree island locations. Basal area and density of trees and shrubs were measured on 64 islands in Everglades National Park. These islands are referred to as TI_EXT. Sixteen of these islands, referred to as TI_INT, have been chosen for more repeated monitoring and more intensive measurements, including soil nutrients, changes in basal area, stem density and seedling recruitment. On three of these islands, continuous measurements of meteorological variables such as irradiance, temperatures and rainfall are also recorded, along with transpiration rates and soil moisture on automated data loggers. Litter fall and seedling recruitment are measured bi-monthly on these three islands. N and C isotopes in plant tissues are measured bi-annually on these islands and on several others distributed throughout the marl prairie and ridge and slough habitats of Everglades National Park.

8.5.3.2 Results

The tree island monitoring program collected data across several spatial scales. Patterns of tree island characteristics at the landscape scale across WCA 3 and Everglades National Park are presented first, followed by comparisons of individual islands across the marl prairie-Shark Slough hydrologic gradient in Everglades National Park, and then by comparisons of species-level physiological responses to changing hydrology.

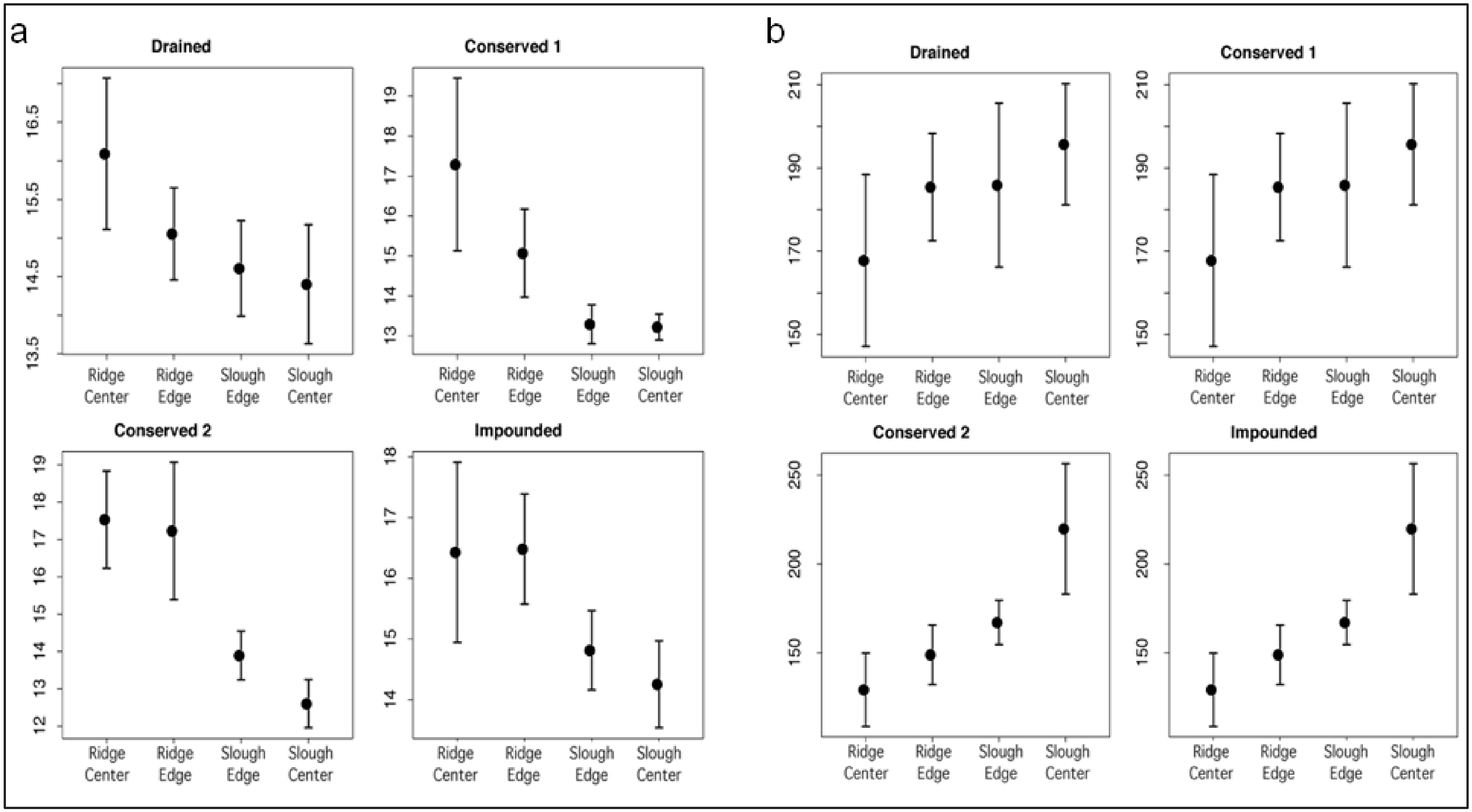


FIGURE 8-41. SUMMARY OF SOIL NUTRIENT RATIOS FOR RIDGE TO SLOUGH TRANSITIONS FOR FOUR PRIMARY SAMPLING UNITS ACROSS A GRADIENT OF HYDROLOGIC DISTURBANCE

8.5.3.2.1 Patterns of Elevation and Inundation at the Landscape Scale

Tree island height above the surrounding marshes varies significantly across WCA 3 and the Everglades National Park (***Figure 8-42***). This creates significantly variability in hydroperiods on Everglades tree islands (***Figure 8-43***). This variation reflects the differences in island heights relative to the surrounding marshes and the larger-scale hydropatterns across WCA 3 and the Everglades National Park. At the landscape scale, inundation periods on the islands generally increase with ponding depths from north to south in WCA 3A. In comparison, the islands in WCA 3B show significantly shorter hydroperiods, resembling the values observed on the islands in the Everglades National Park.

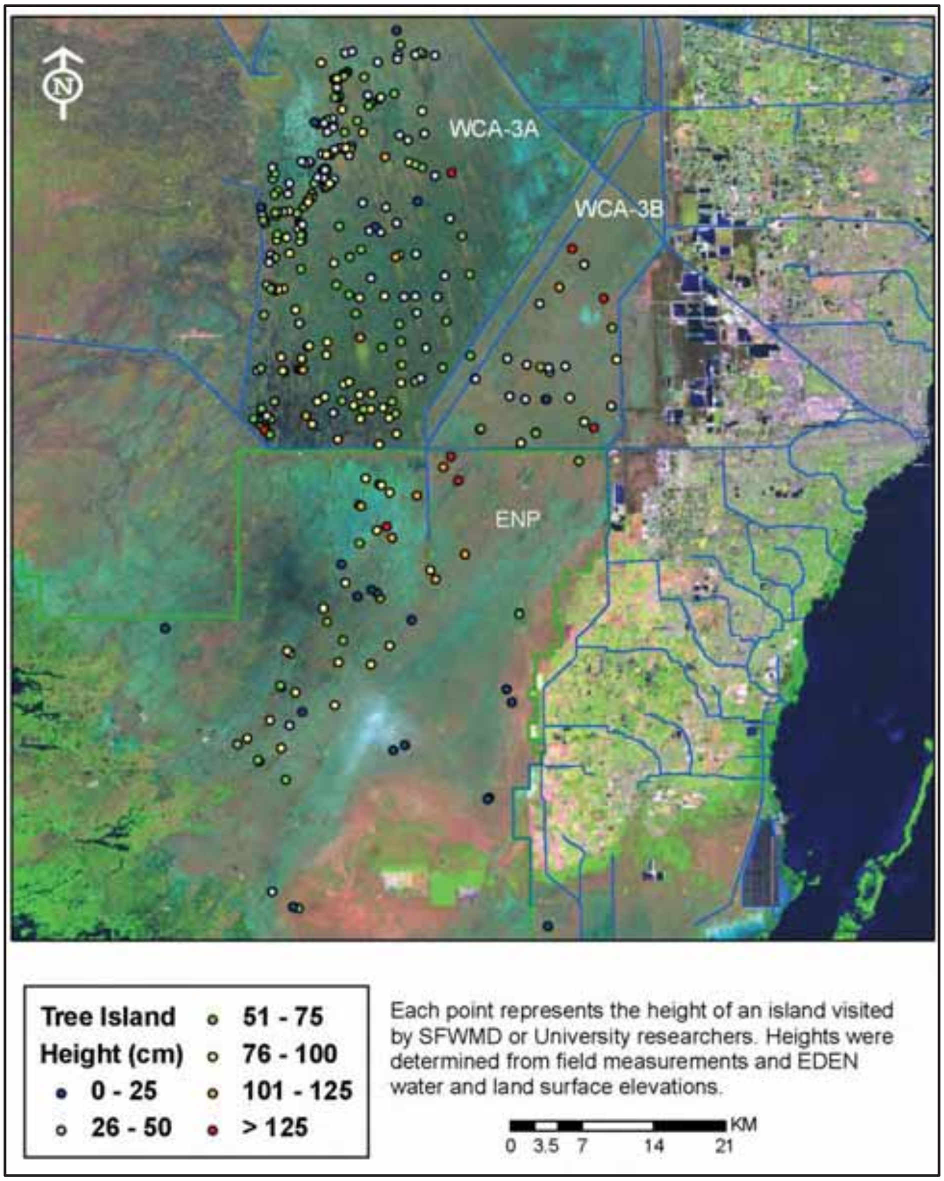


FIGURE 8-42. HEIGHT OF TREE ISLANDS IN WATER CONSERVATION AREA 3 AND EVERGLADES NATIONAL PARK

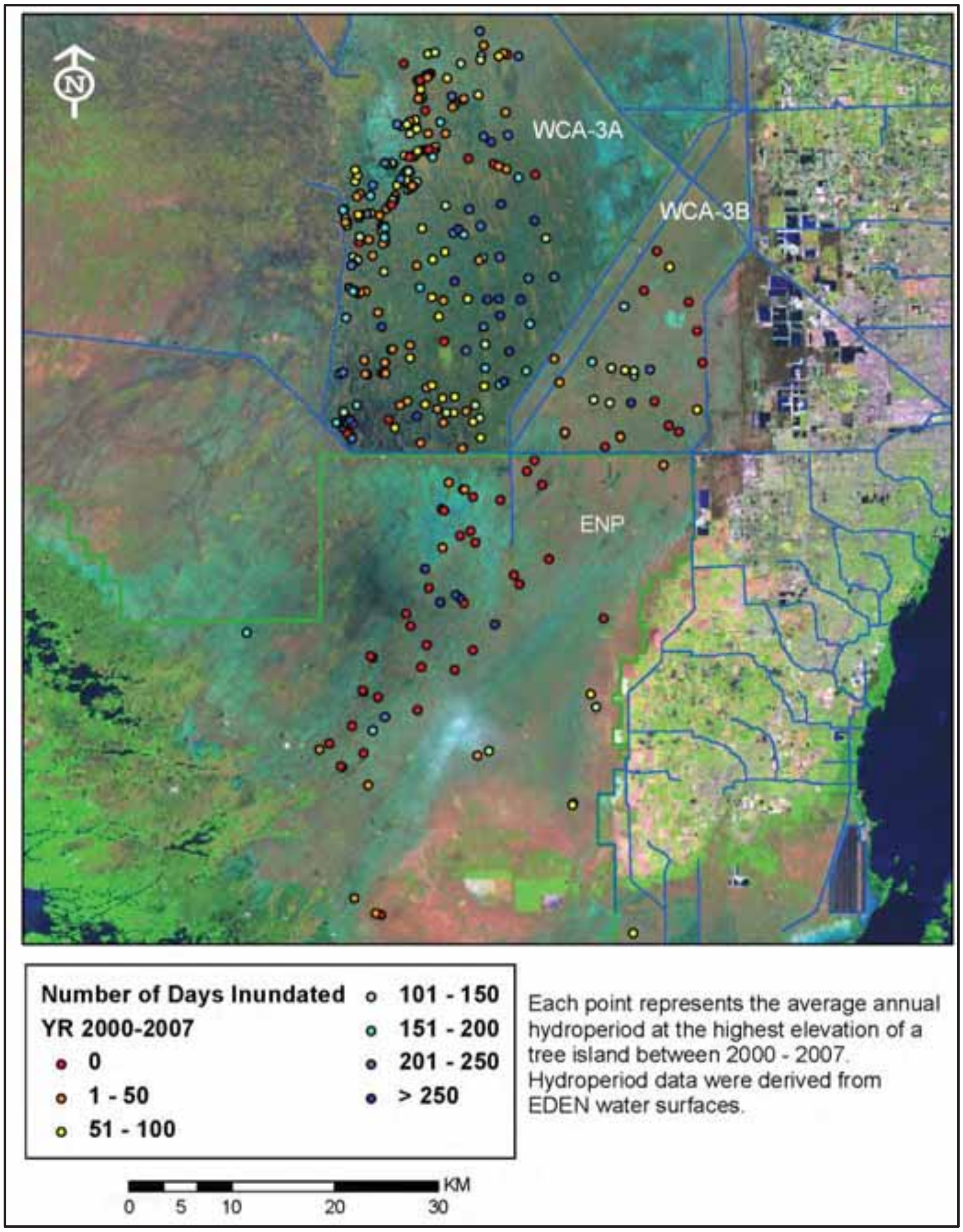


FIGURE 8-43. AVERAGE ANNUAL HYDROPERIODS AT THE HIGHEST ELEVATION OF TREE ISLANDS WITHIN WATER CONSERVATION AREA 3 AND EVERGLADES NATIONAL PARK BETWEEN 2000 AND 2007

In Everglades National Park, tree island height is positively correlated with the average water depths in the marshes of Shark Slough (***Figure 8-44***). The data show the difference between tree island heights and marsh water levels increases with marsh depths. As a result, the islands found in the deepest marshes may actually represent the driest upland habitats. The marl prairie islands are generally "shorter" than the slough islands, but these islands are also among the driest because the water table in these habitats is significantly lower than those found in Shark Slough.

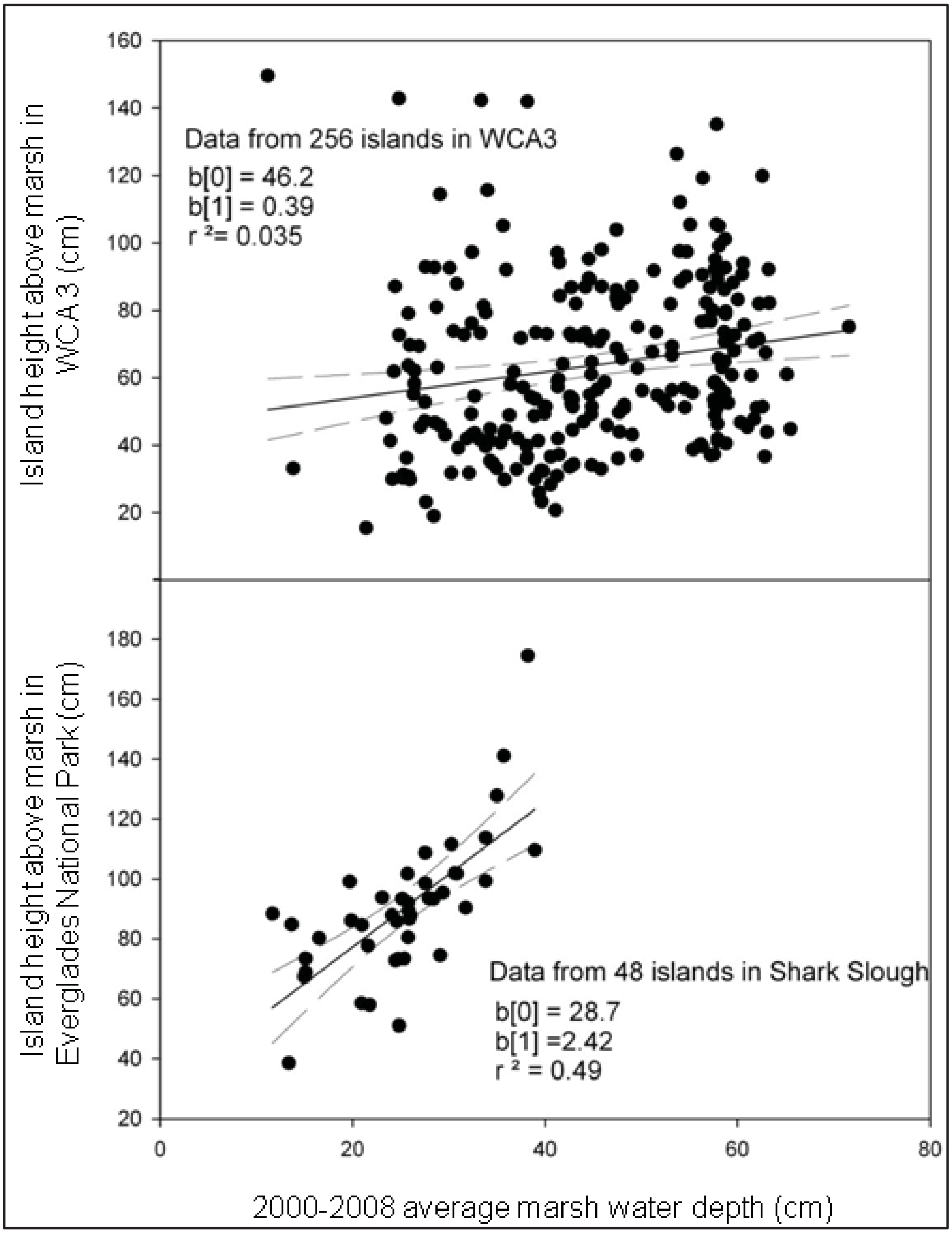


FIGURE 8-44. TREE ISLAND HEIGHTS ABOVE THE SURROUNDING MARSH ARE CORRELATED WITH MARSH WATER DEPTHS IN SHARK SLOUGH WITHIN EVERGLADES NATIONAL PARK, BUT NOT IN WATER CONSERVATION AREA 3

It is hypothesized that the relationship between island height and marsh water depths in Shark Slough is indicative of the biogeochemical processes that lead to tree island formation and persistence. This relationship is not found in the impounded regions of WCA 3 where tree islands have been lost. The absence of this pattern may reflect the mechanisms that have led to island decline in WCA 3.

8.5.3.2.2 Canopy Composition and Growth in Relation to Hydrology

Weighted-averaging regression is used to determine hydrological niches of tree species found in tree islands in the Everglades (***Figure 8-45***). The concept behind weight-averaging regression is tree species whose hydrologic "optima" are close to conditions experienced on a given island would tend to be the most abundant taxa there and on islands of similar hydrology (see Birks et al., 1990 for analogy). Therefore, hydrologic optima and tolerances, which is an expression of dispersion around the optima, of a taxon were calculated from species abundances across several islands for which hydrological variables were known. This work expands on the analyses presented in Sah (2004). In the current study, the hydrologic tolerances of tree species are calculated using data from the most elevated positions on 73 islands. Hydroperiod is defined as the number of days per year when water level was above the ground surface. Negative values result from the subtraction of a large tolerance value from a smaller mean and should be ignored. In general, the species that are found in the wettest environments exhibit the largest tolerances to variable hydrologic conditions. *Taxodium distichum* was found on only one island.

Extensive survey data show tree islands in WCA 3A and 3B are generally colonized by flood tolerant species (***Figure 8-46*** and ***Figure 8-47***), and that the hardwood hammocks islands in Everglades National Park are generally inhabited by species that occur in drier habitats (***Figure 8-48***). The color codes on the stack graphs correspond to the hydrologic tolerance of the dominant tree species on the island, which is shown in the legend, and the height corresponds to their relative importance value (importance value – relative basal area * relative density). Individual graphs for the island heads and tails are shown. The island elevation and the mean annual hydroperiod for 2000 to 2007 derived from EDEN are shown next to each island.

The effects of hydrologic restoration on the islands in WCA 3B is of concern to several agencies and interest groups involved in CERP. The long-term effects of restoration on these islands are unknown, and will be determined by factors such as fire frequency, rates of seedling recruitment, natural and man-made disturbances, and competition from invasive exotics such as *Lygodium*. However, the information on the hydrologic tolerances of the extant species can be compared to the current range of conditions these islands typically experience for an initial estimate of how these communities may respond to increased water levels.

The islands in WCA 3B are generally inhabited by flood-tolerant tree species and the average annual hydroperiods on these islands are significantly lower than the hydroperiods in other areas where these species are typically found. In other words, the species that dominate the relatively dry islands in WCA 3B are also known to inhabit much wetter environments in WCA 3A and Everglades National Park.

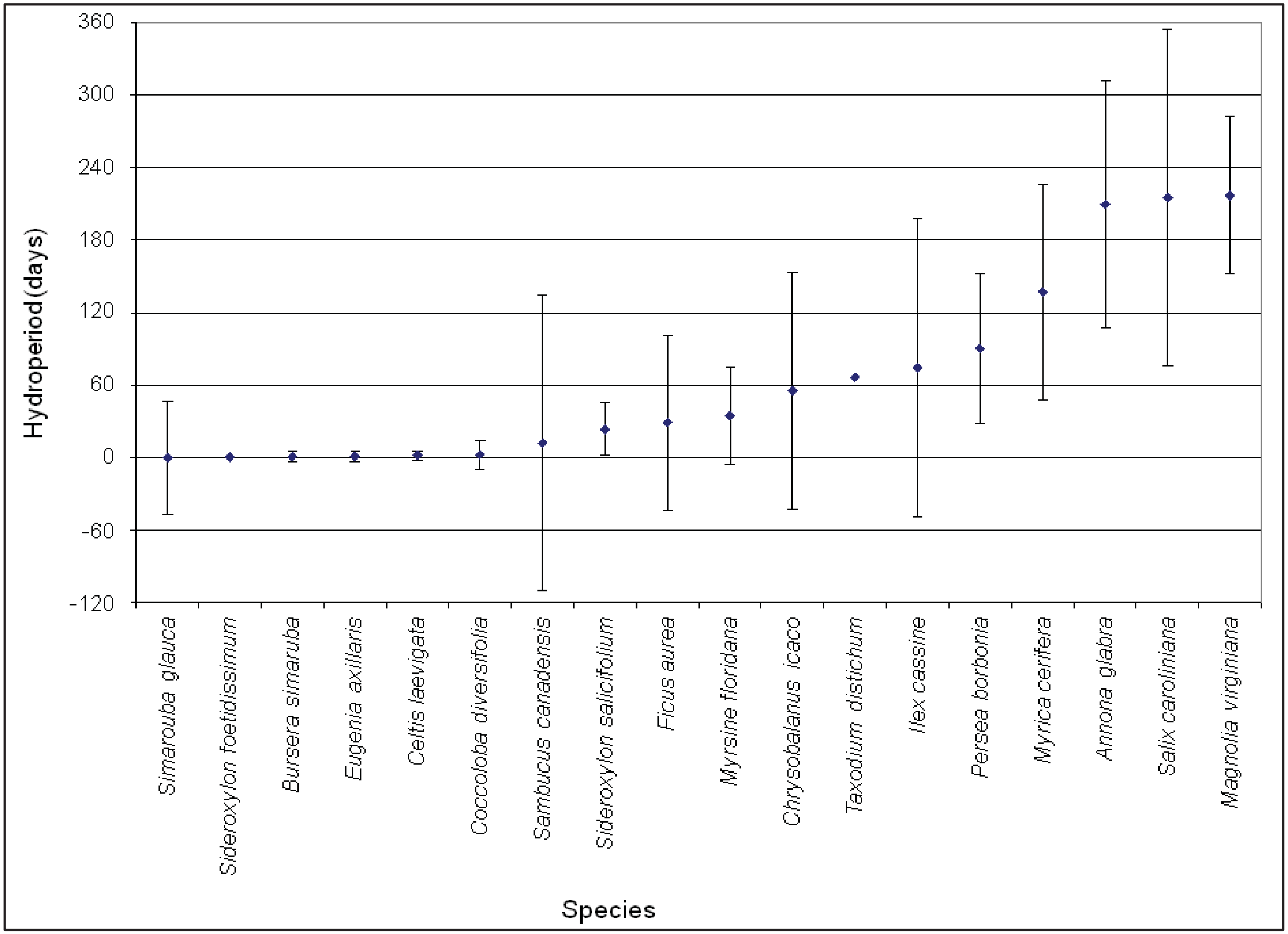


FIGURE 8-45. MEAN ANNUAL (OPTIMA) AND RANGE OF HYDROPERIODS (TOLERANCE) OF 16 OF THE MOST COMMON TREE SPECIES IN THE EVERGLADES

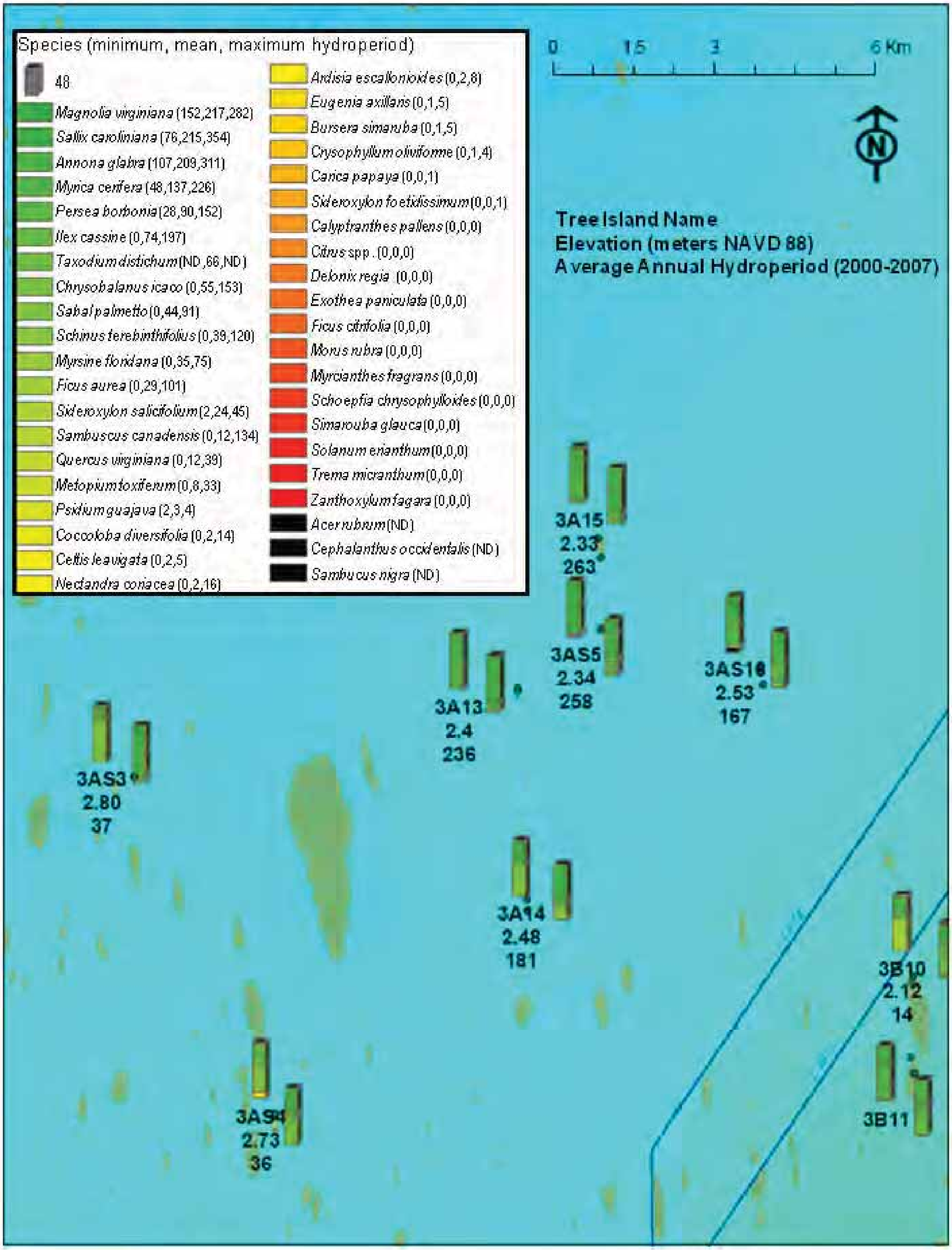


FIGURE 8-46. CANOPY COMPOSITION OF SELECT TREE ISLANDS IN WATER CONSERVATON AREA 3A

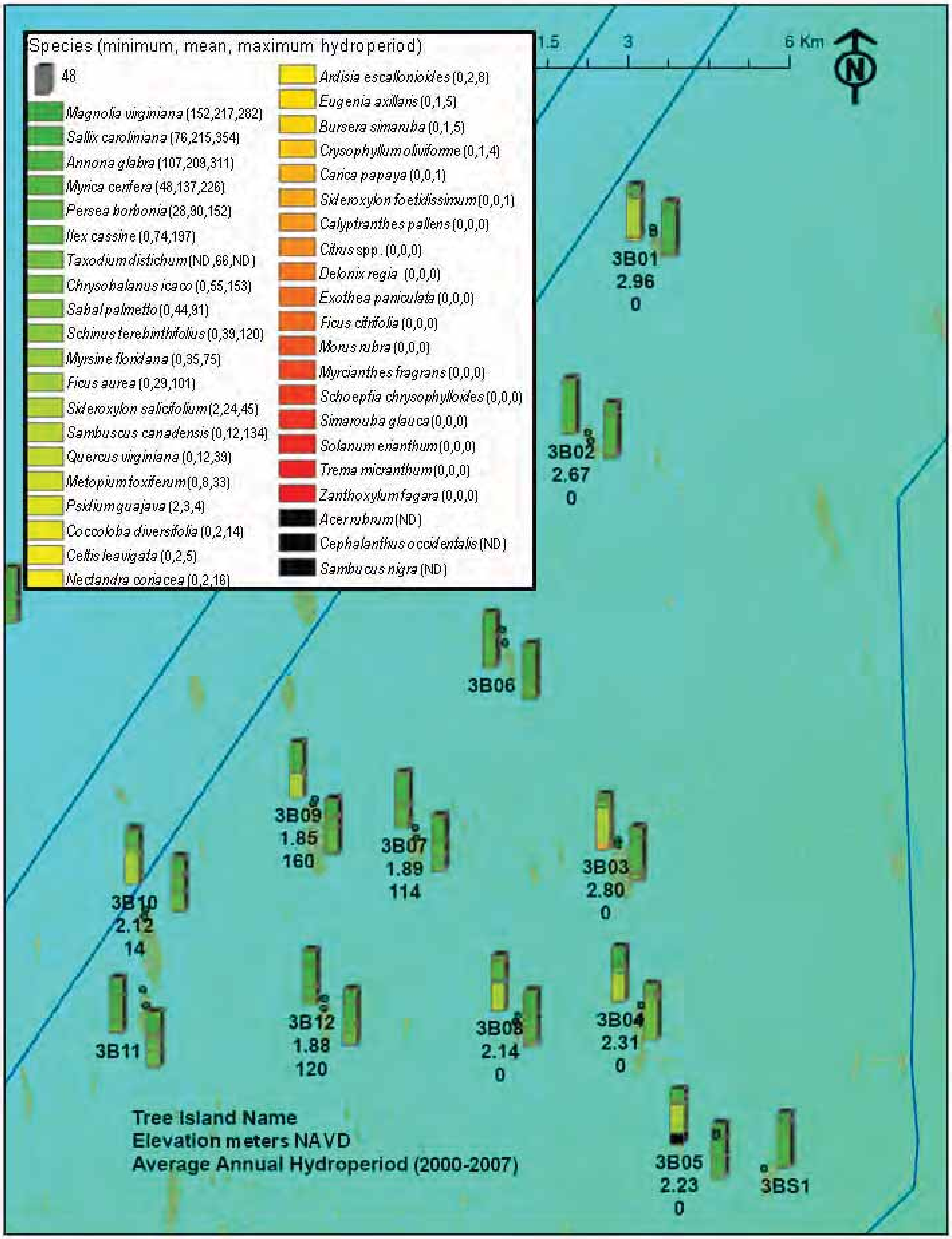


FIGURE 8-47. CANOPY COMPOSITION OF SELECT TREE ISLANDS IN WATER CONSERVATION AREA 3B

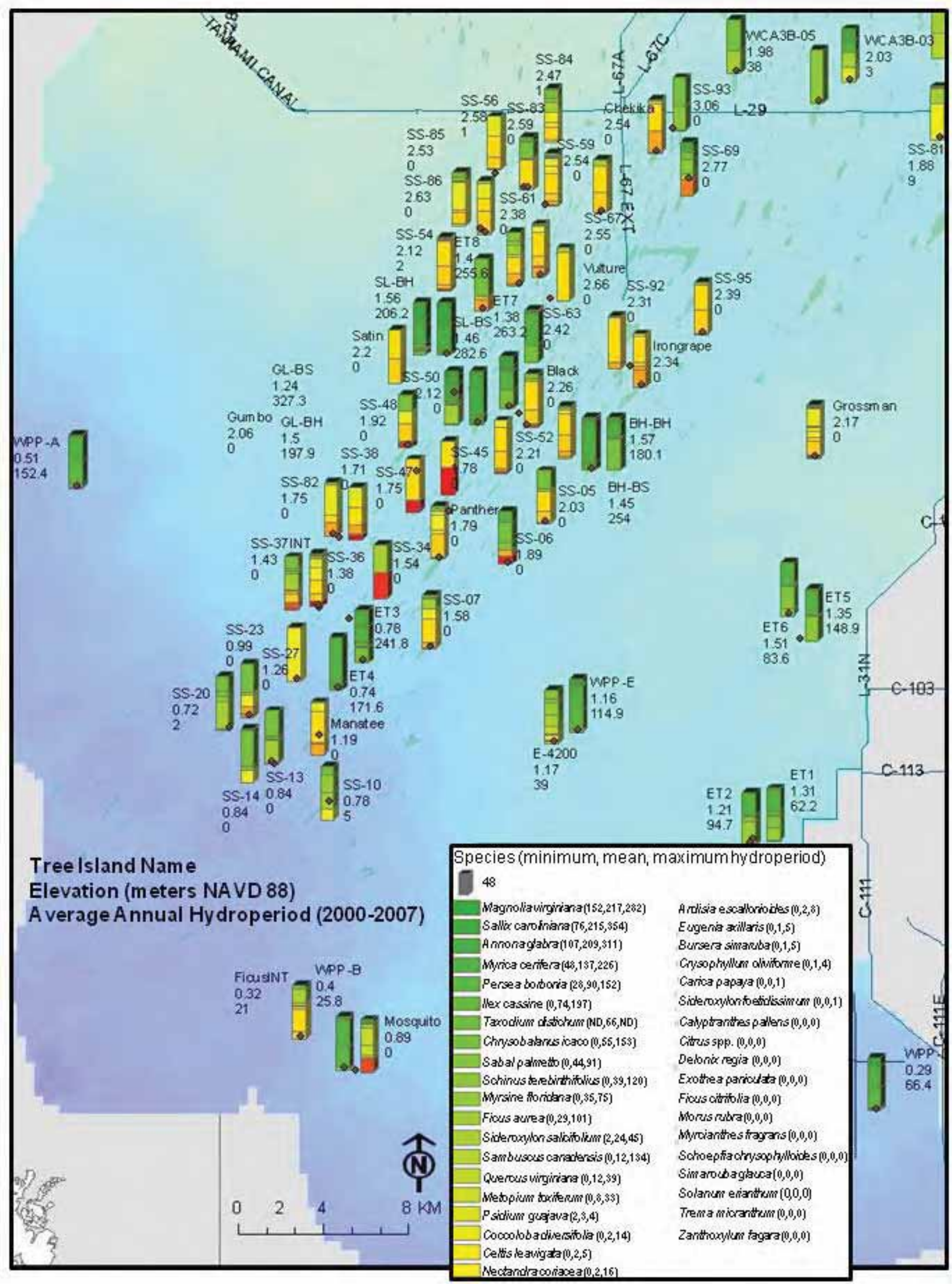


FIGURE 8-48. CANOPY COMPOSITION OF SELECT TREE ISLANDS IN EVERGLADES NATIONAL PARK

Tree growth and canopy development on tree islands is expected to vary with water depths. Plot-level data from 2007 through 2009 on ten islands (TI_INT) in Shark Slough show the changes in total basal area follows a quadratic relationship with water depths measured from the highest point on the island (***Figure 8-49***). In this case, the changes are expressed as relative growth rates, which are the percent change in basal area, in order to make direct comparisons across islands that host trees of different size or stem density.

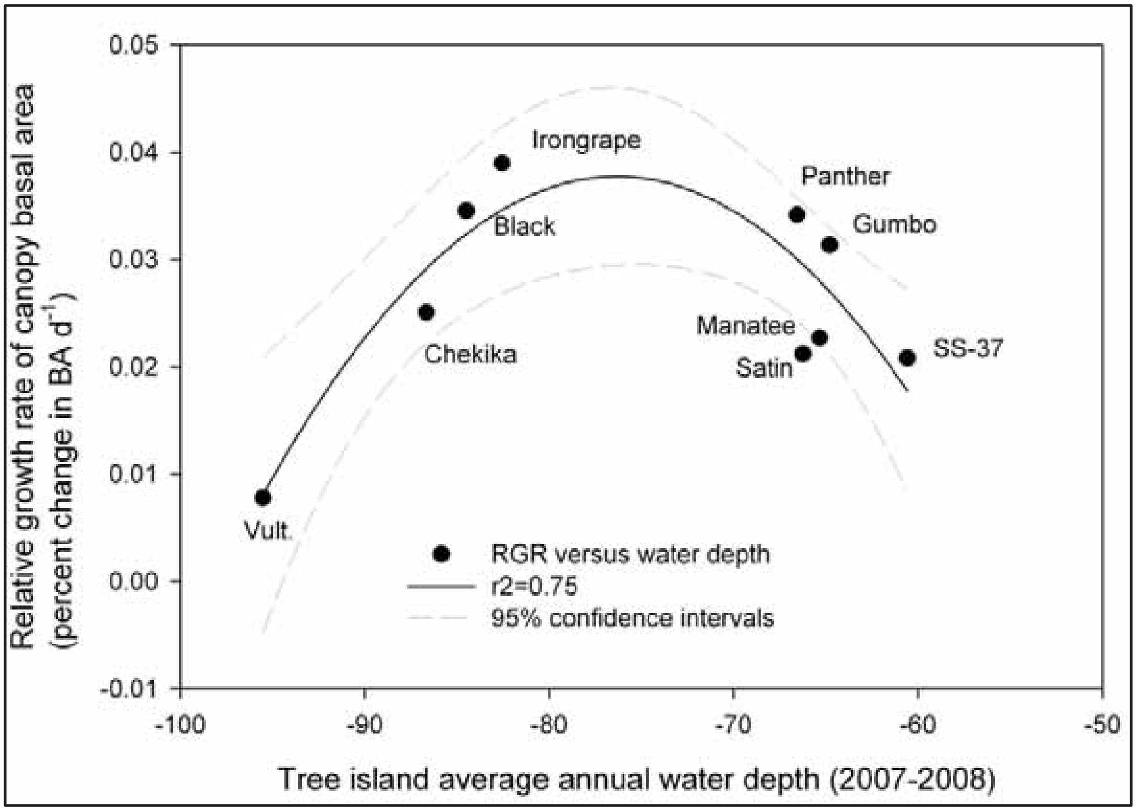


FIGURE 8-49. PLOT-LEVEL DATA ON 10 ISLANDS IN SHARK SLOUGH SHOW CHANGES IN TREE BASAL AREA, EXPRESSED AS RELATIVE GROWTH RATE, FOLLOWS A QUADRATIC RELATIONSHIP WITH WATER DEPTHS AS MEASURED FROM THE HIGHEST POINT ON THE ISLAND

As shown above, tree island height is positively correlated with marsh water depths, and as a result the total canopy relative growth rate shows a quadratic relationship with island height similar to that observed with water depths. The relationship between canopy growth rates and tree island height may reflect some recent results by Wetzel et al. (2009), which show the P content on tree islands in WCA 3 generally increases with height above the marsh. Wetzel et al. (2009) suggests the correlation between island height and P content may reflect a process driven by high transpiration rates whereby tree islands sequester nutrients from the surrounding marshes (Ross et al., 2006a; Wetzel et al., 2005). The relative growth rates of several species appear to be a function of height, with some species showing positive and others negative relationships (***Figure 8-50***). In all cases, the correlation between relative growth rate and island height for

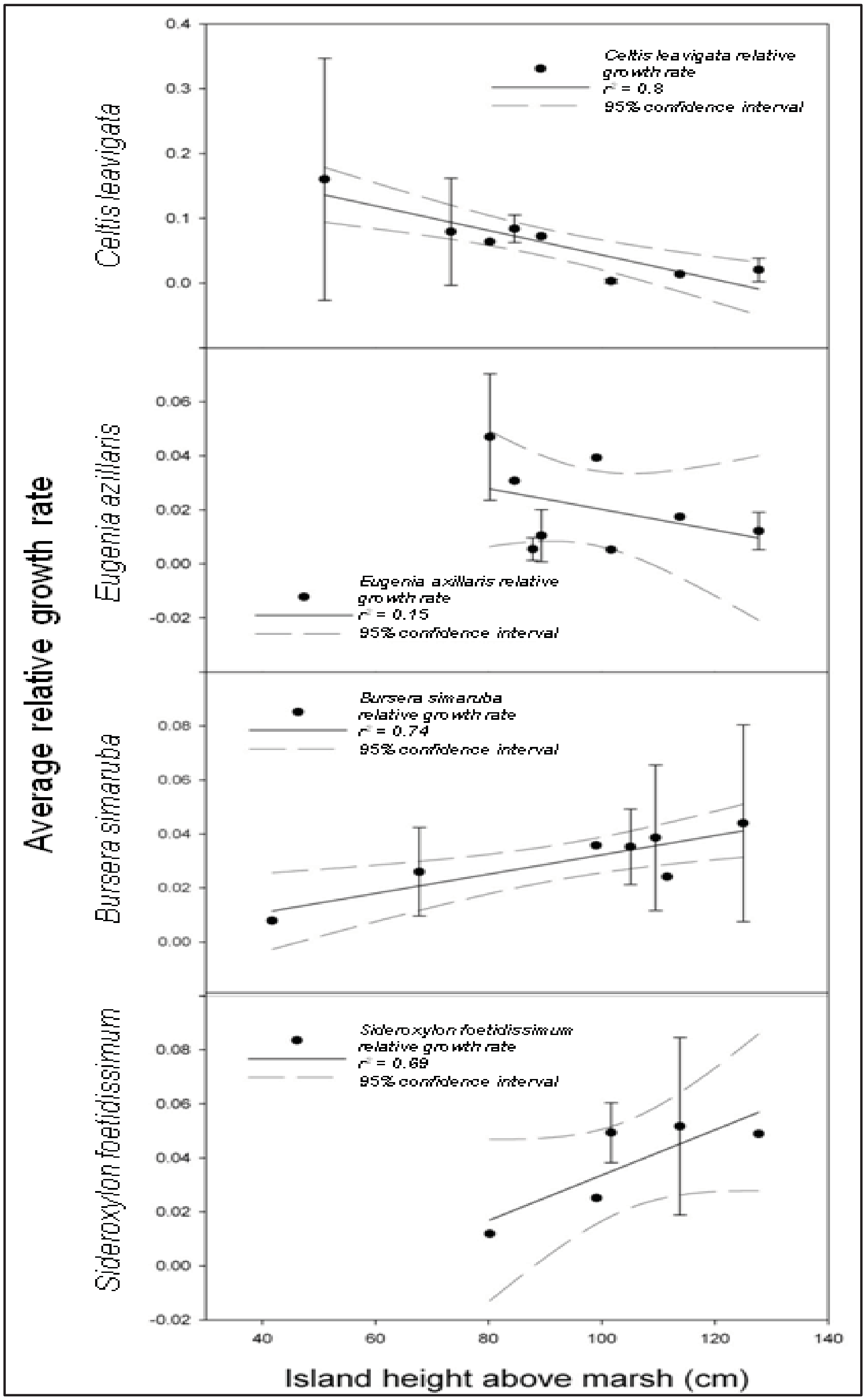
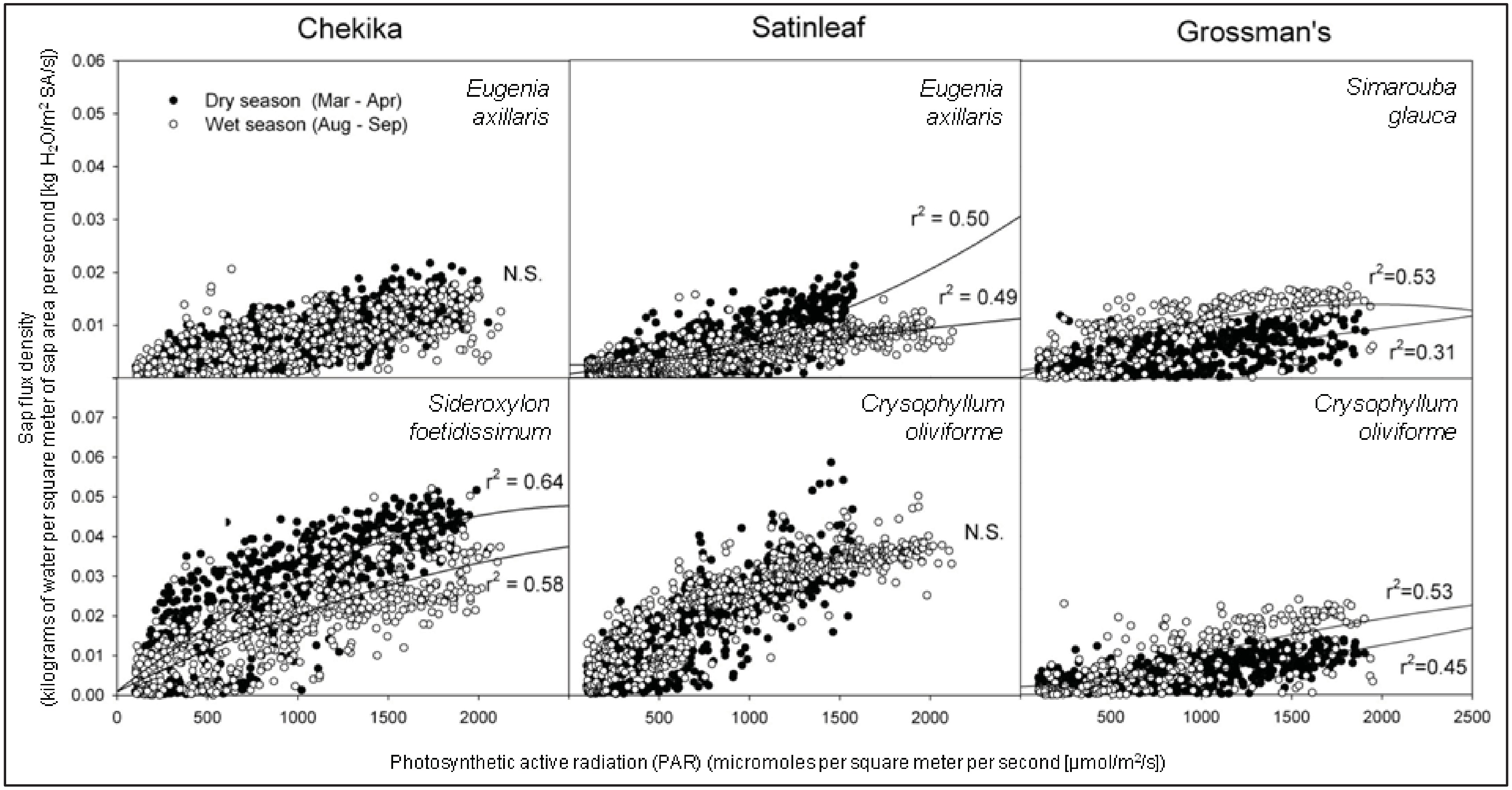


FIGURE 8-50. RELATIVE GROWTH RATES OF SEVERAL SPECIES ON EVERGLADES NATIONAL PARK TREE ISLANDS APPEAR TO BE A FUNCTION OF ISLAND HEIGHT ABOVE THE SURROUNDING MARSH

individual species was better than the correlation than between relative growth rates and water levels. The species that showed a positive response to increasing island height are adapted to slightly drier conditions than those that showed a negative response. This result is somewhat expected since taller islands are drier islands. However, all of these species are commonly found coexisting in elevated hardwood hammocks in Everglades National Park and it is not known if the differences in growth can be attributed strictly to island height or to hydrologic conditions. Competition for light or nutrients, or disturbance history, may also play a role. Larger sample sizes are needed to investigate these relationships further.

8.5.3.2.3 Physiological Responses to Hydrologic Conditions

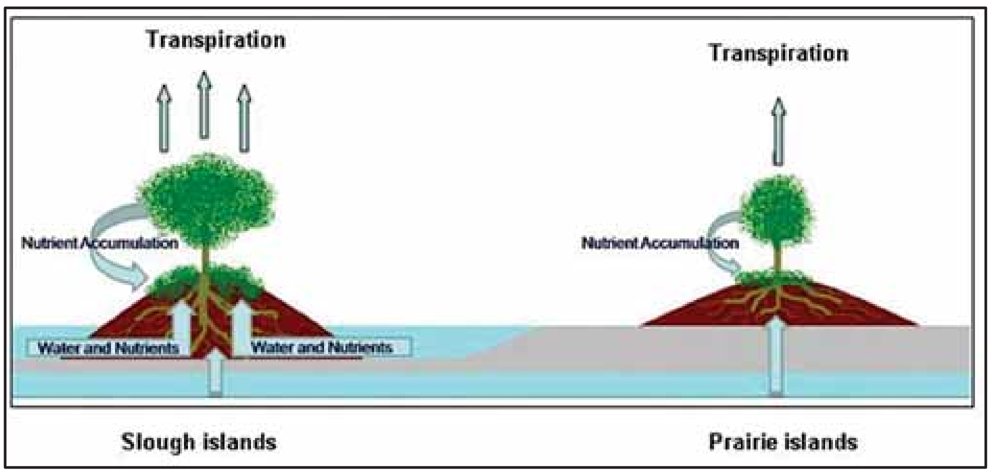
Transpiration rates are often considered an indicator of growth and photosynthesis in trees. Transpiration rates were different during periods of high and low water depths in several tree species (***Figure 8-51***). Wet season (high water) transpiration rates were lower than dry season rates in two species, white stopper (*Eugenia axillaris)* and false mastic *(Sideroxylon foetidissimum)* on two islands, Satinleaf and Chekika. However, *Eugenia* did not respond consistently on these two islands. Marsh water depths around Satinleaf (*Chrysophyllum oliviforme)* and Chekika are similar, but Satinleaf is shorter than Chekika relative to the surrounding marsh and the island can be considered wetter. *Eugenia* has low tolerance for flooding, and the wetter conditions on Satinleaf may have caused some stress in this species that was not observed on Chekika. However, *Eugenia* occurs in the subcanopy on Satinleaf where several large and deciduous *Bursera* and *Celtis* are also found, and these species may increase shading of the lower canopies layers during the wet season. Thus, the effect could be due to light competition not to flooding stress. The other dominant species on Satinleaf showed no transpiration response to higher water levels. Conversely, *C. oliviforme* and paradise tree *(Simarouba glauca)*showed increased transpiration rates during summertime high water conditions on the marl prairie tree island Grossman's hammock. Pigeon plum *(Coccoloba diversifolia)* showed a slight positive response to higher water levels on this island. These results suggest that drought stress regulates growth on the marl prairie islands in eastern Everglades National Park.



data courtesy of M. Ross, J. Sah and S. Oberbauer, Florida International University

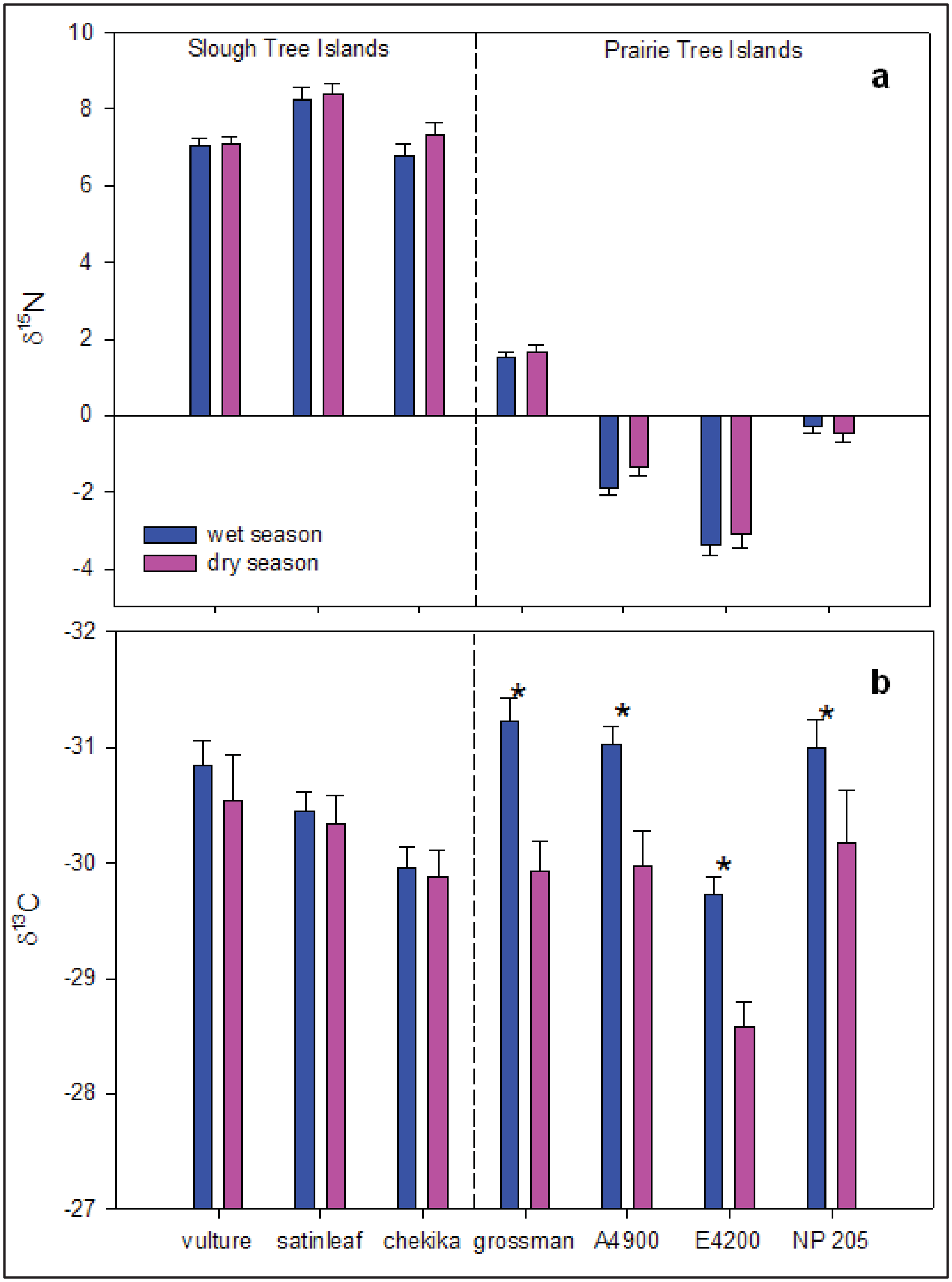
FIGURE 8-51. TRANSPIRATION RATES IN 2008 DURING DIFFERENT PERIODS OF HIGHER VERSUS LOWER WATER DEPTHS IN SEVERAL TREE SPECIES

A recent CEM developed by L. Sternberg and X. Wang at the University of Miami links the processes of transpiration, nutrient uptake and water stress on tree islands using data on stable isotopes of C and N (***Figure 8-52***). According to this model, those tree islands occurring in marshes that are flooded year-round would experience less water stress and would exhibit increased growth compared to islands subjected to periodic drydowns such as those occurring in the marl prairies. The high transpiration rate helps sequester P from the surrounding marshes through groundwater uptake. This additional P contributes further to increased tree growth which, in turn, creates increased demand for the limiting nutrient N. Thus, N demand should be higher on the wetter islands, and ^{15}N in the leaf tissues should increase (***Figure 8-53a***). On the other hand, water stress is expected to increase on those islands that experience dry downs, and this should result in increases in ^{13}C in leaf tissues during the dry season (***Figure 8-53b***). Data collections are being expanded in 2009 and 2010 to test this model further.



(courtesy of X. Wang and L.S.L. Sternberg, University of Miami)

FIGURE 8-52. CONCEPTUAL MODEL OF THE CHEMOHYDRODYNAMIC HYPOTHESIS OF TREE ISLAND INTERACTIONS WITH THE SURROUNDING MARSHES



courtesy of X. Wang and L. Sternberg, University of Miami

FIGURE 8-53. AVERAGE FOLIAR A) Δ^{15}N (‰) AND B) Δ^{13}C ± STANDARD ERROR (SE) IN WET AND DRY SEASONS FOR THREE SLOUGH TREE ISLANDS AND FOUR PRAIRIE TREE ISLANDS

Note: stars (*) indicate significant differences between wet and dry season at α=0.05

8.5.4 Marl Prairies and Slough Gradients

8.5.4.1 Monitoring

Taylor Slough consists of a relatively narrow, sediment-laden channel that widens in the south bordered by marl prairies which are 10 to 30 centimeters higher in elevation and much broader than the slough itself. Marl prairies in the Taylor Slough Basin support the most species-rich wetland plant assemblages in the Everglades. Under normal conditions, vegetation composition and structure in this community remain in harmony with natural variation in hydrologic regimes. The hydrology of the area was altered by the completion of Tamiami Trail in the 1920s and 1930s, and, subsequently, by completion of the C&SF Project. Since the early 1990s, hydrologic conditions in Taylor Slough have been affected by the operations of the S332 or S332D structures, resulting in rapid spatio-temporal changes in species composition. Water delivery from S332 started in the early 1980s, but delivery was not substantial until approximately 1992. In 2000, operation of S332 was fully replaced by S332D, located north of S332. The intent of the change in operation was to switch flow patterns from point delivery to seepage flow. In this endeavor, water is raised east of a levee along the border of Everglades National Park, creating a hydrologic head that allows water to seep into the park. The question addressed here is, "How closely do temporal trends in vegetation composition mimic such hydrologic manipulations?" It is hypothesized that vegetation composition in Taylor Slough and adjacent prairies south of the S332 shifted towards a wetter type until 1999 and, thereafter, shifted towards a drier assemblage, probably because less water actually reached the park through seepage from S332D than had arrived by direct hydration from S332. To examine the effects of hydrologic alterations, total and species-specific macrophyte cover and plant species composition have been periodically monitored since 1979.

Study areas are in the Shark Slough and Taylor Slough basins in Everglades National Park. In these basins, plant communities along marl prairie and slough gradients are monitored. Vegetation in Shark Slough is monitored along five transects, varying in length from 9 to 35.8 kilometers, and in Taylor Slough along six transects ranging from few hundred meters to two kilometers (***Figure 8-54***).

In Taylor Slough, vegetation is monitored along six transects: two transects (T4 and T5) are in the headwaters of Taylor Slough, two transects (T1 and T2) are in the upper slough, and two transects (T3 and T6) are in the middle slough below State Road 9336. T1, T2 and T3 were established in 1979, T4 and T5 in 1997, and T6 in 2007. T4 and T5 are currently upstream of water deliveries from S332 and S332D, and are used as reference sites. Vegetation data collected from these transects will be increasingly relevant once the operational plan for water delivery to the northeast Everglades through opening(s) under Tamiami Trail is fully implemented. On each transect, plant species composition, total and species specific cover, and hydrologic conditions are recorded.

To facilitate direct comparison of sequential surveys in Taylor Slough dating back to 1992, identical vegetation sampling protocols have been maintained during each sampling event. In 1992, only T2 was sampled, and in 1995 both T1 and T2 were sampled. T3 was sampled in 1996 and T4 and T5 were sampled for the first time in 1997. Since 1999, the sites on all five transects (T1-T5) were sampled every four years.

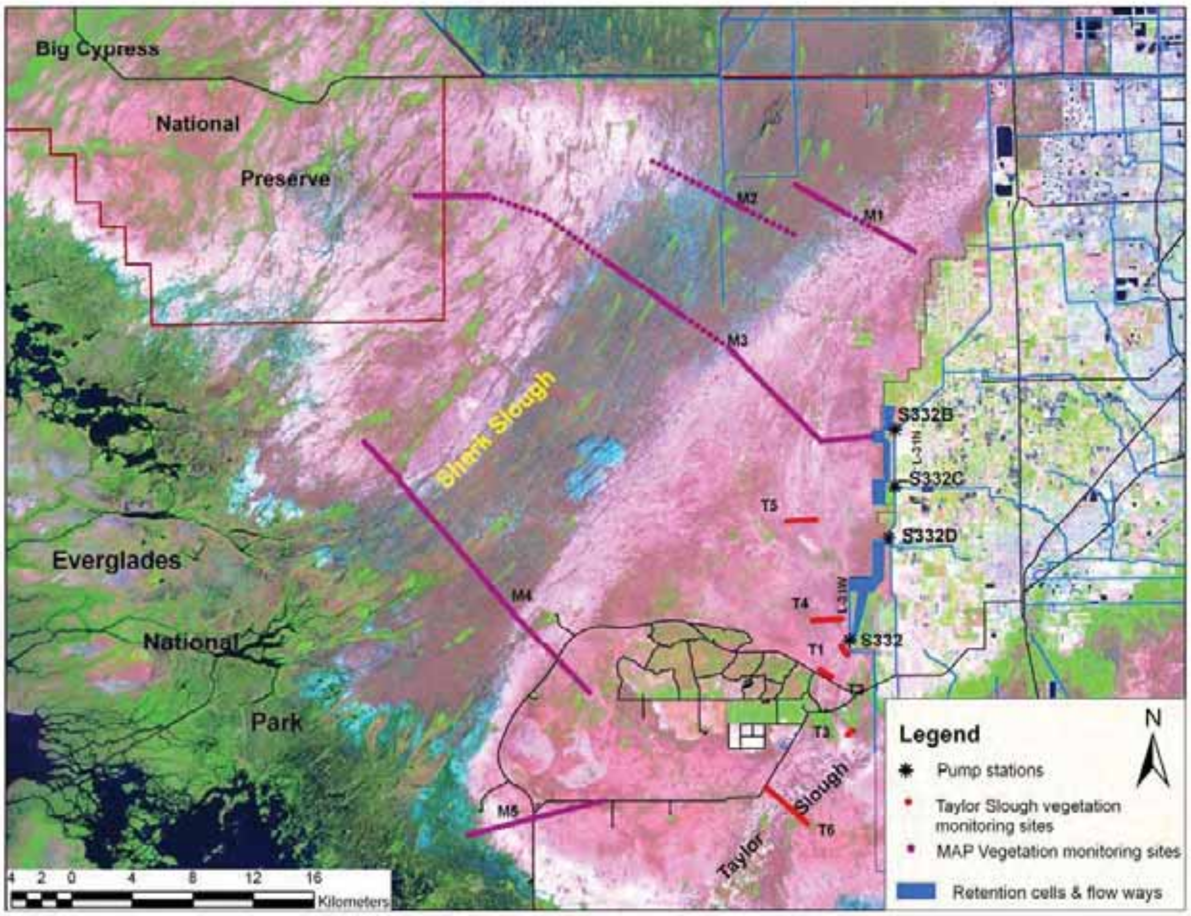


FIGURE 8-54. SITE MAP SHOWING THE VEGETATION MONITORING TRANSECTS IN SHARK SLOUGH AND TAYLOR SLOUGH BASINS IN EVERGLADES NATIONAL PARK AND BIG CYPRESS NATIONAL PRESERVE

In order to relate Taylor Slough vegetation dynamics to changes in hydrology or soil nutrients, data from the Cape Sable seaside sparrow habitat study (Ross et al., 2006b) were utilized. In this study, shoot cover was estimated. Also, hydrology and soils data were collected allowing them to serve as reference sites for the Taylor Slough study. Surface soils were collected for chemical and physical analysis.

In the marl prairies and slough gradients study, patterns of soil TP, which is generally the limiting nutrient in Everglades marshes, were examined. Hydroperiod was calculated using water level data obtained from the EDEN database and mean plot elevation was determined. Hydroperiod was defined as the number of discrete days per year when mean water level was above the ground surface.

Change in vegetation composition along hydrologic and P gradients was analyzed using trajectory analysis (Minchin et al., 2005). First, the vegetation data were summarized by a non-metric multidimensional scaling (NMDS) ordination, followed by examination of the time trajectory of each site along hydrologic and P gradients. The reference vectors for the gradients were defined by including the reference sites from four Cape Sable seaside sparrow vegetation transects with known hydroperiod and a set of Cape Sable seaside sparrow vegetation sites on those transect for which soil P data were available, and using vector fitting technique in the Database for Ecological Community Data (DECODA) (Minchin, 1998). Statistics were calculated to quantify the degree and rate of change in vegetation composition and to determine the degree to which each site's trajectory was aligned with the reference vectors, and their statistical significance was tested using Monte Carlo simulations with 10,000 permutations.

8.5.4.2 Results

8.5.4.2.1 Hydrologic Changes in Taylor Slough

Hydrologic models developed using multiple linear regression reveal that water level in Taylor Slough Basin is affected by water pump operations. When a precipitation-based-only hydrologic model developed from the data for the pre-S332 (1961-1980) period was used to estimate water levels during the S332 (1980-1999) and S332D (2000-2008) periods, observed water levels were much higher than predicted in both periods. In both the S332 and S332D periods, however, the water level in Taylor Slough was strongly related to deliveries from the pumping stations. Water delivery from S332 was relatively low (3.8 million cubic meters per month) between 1980 and 1991, but delivery significantly increased after 1991 (13.6 million cubic meters per month), resulting in water levels 20 centimeters higher in the 1990s than in the 1980s, and, in general, 25 to 50 centimeters higher than they would have been in a precipitation driven system (Armentano et al., 2006). After 2000, even though water was not directly delivered into the slough, as it was during the operation of the S332 pumping station, water level in the slough was 15 to 40 centimeters higher than it would have been in a precipitation-driven system, suggesting a strong influence of S332D. The effects of this structure are particularly evident in the dry season when seasonal mean water level downstream from Taylor Slough Bridge clearly tracks the amount of water delivered through S332D (***Figure 8-56***). In recent years, water has been delivered from the L31N Canal into the L31W Canal, then by a series of retention ponds in the Frog Pond region and a flow way cell into Everglades National Park near S332. However, it is likely that water from the retention ponds also seeps into the L31W Canal from where it enters as surface and subsurface flows into Everglades National Park, and then, passing through the expanse of marl prairies, water goes into the slough. This linkage is weak in the wet season (***Figure 8-55***) when surface flow from Shark Slough and northeast Everglades National Park influences the water level in Taylor Slough, reducing the impact of the local water budget.

In the upper Taylor Slough Basin, water levels in marl prairies are influenced by water pumps, as is evident by how well stage at marsh recorder CR2 tracks the total flow from S332B (***Figure 8-57***). In this region, the S332B and S332C pump structures deliver from the L31N Canal into a series of inter-connected retention ponds of the primary pond near S332B, which has a large fixed-crest weir on the western levee that allows surface water to enter Everglades National Park marl prairies.

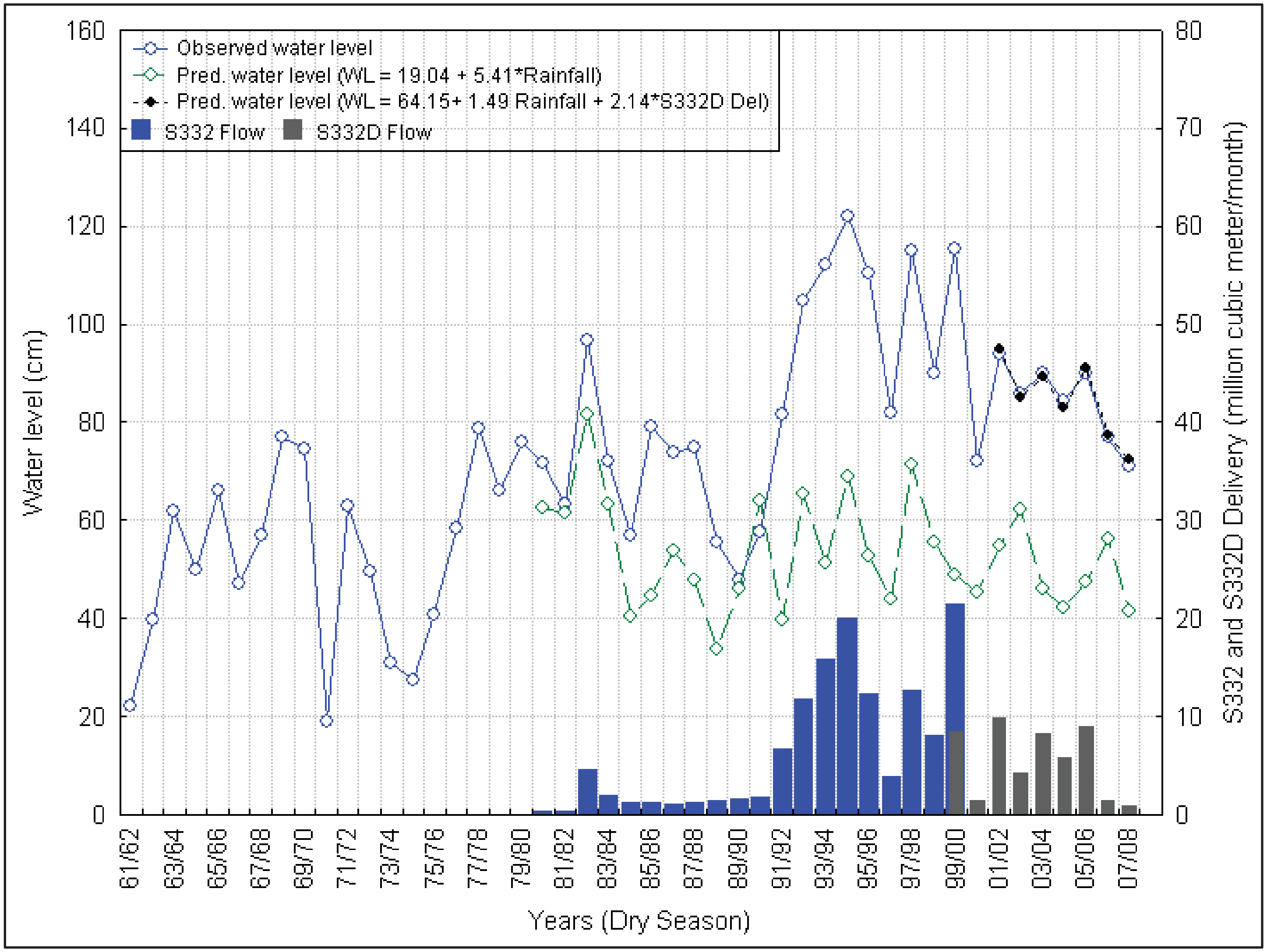


FIGURE 8-55. OBSERVED AND PREDICTED WATER LEVEL AND SEASONAL TOTAL WATER FLOW THROUGH S332 INTO TAYLOR SLOUGH AND THROUGH S332D INTO A RETENTION POND DURING THE 1961 THROUGH 2008 WET SEASONS

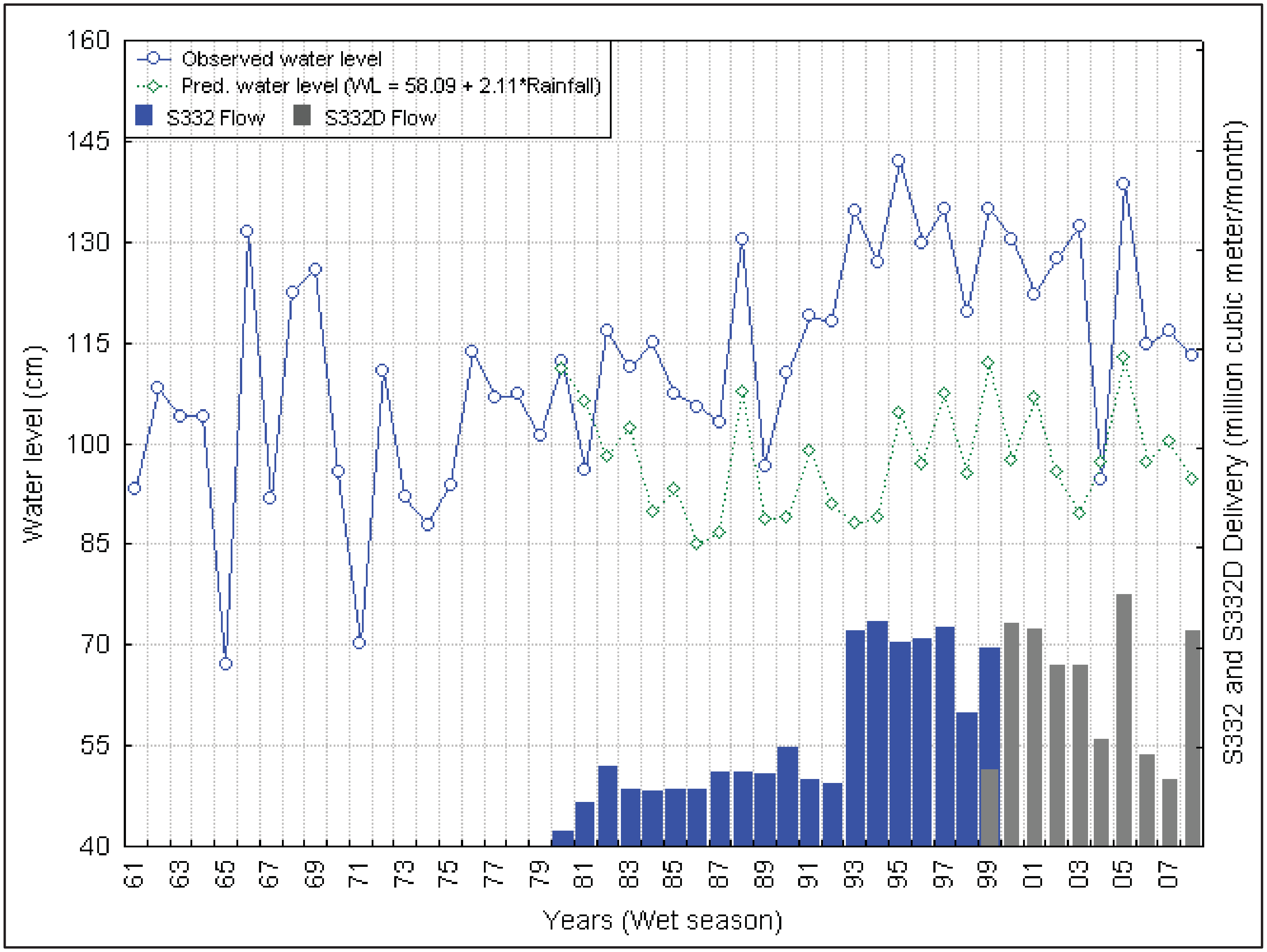


FIGURE 8-56. OBSERVED AND PREDICTED WATER LEVEL AT THE TAYLOR SLOUGH BRIDGE STAGE RECORDER AND WATER FLOW FROM L31W CANAL THROUGH S332 INTO TAYLOR SLOUGH AND FROM L31N THROUGH S332D INTO A RETENTION POND DURING THE 1961 THROUGH 2009 DRY SEASONS

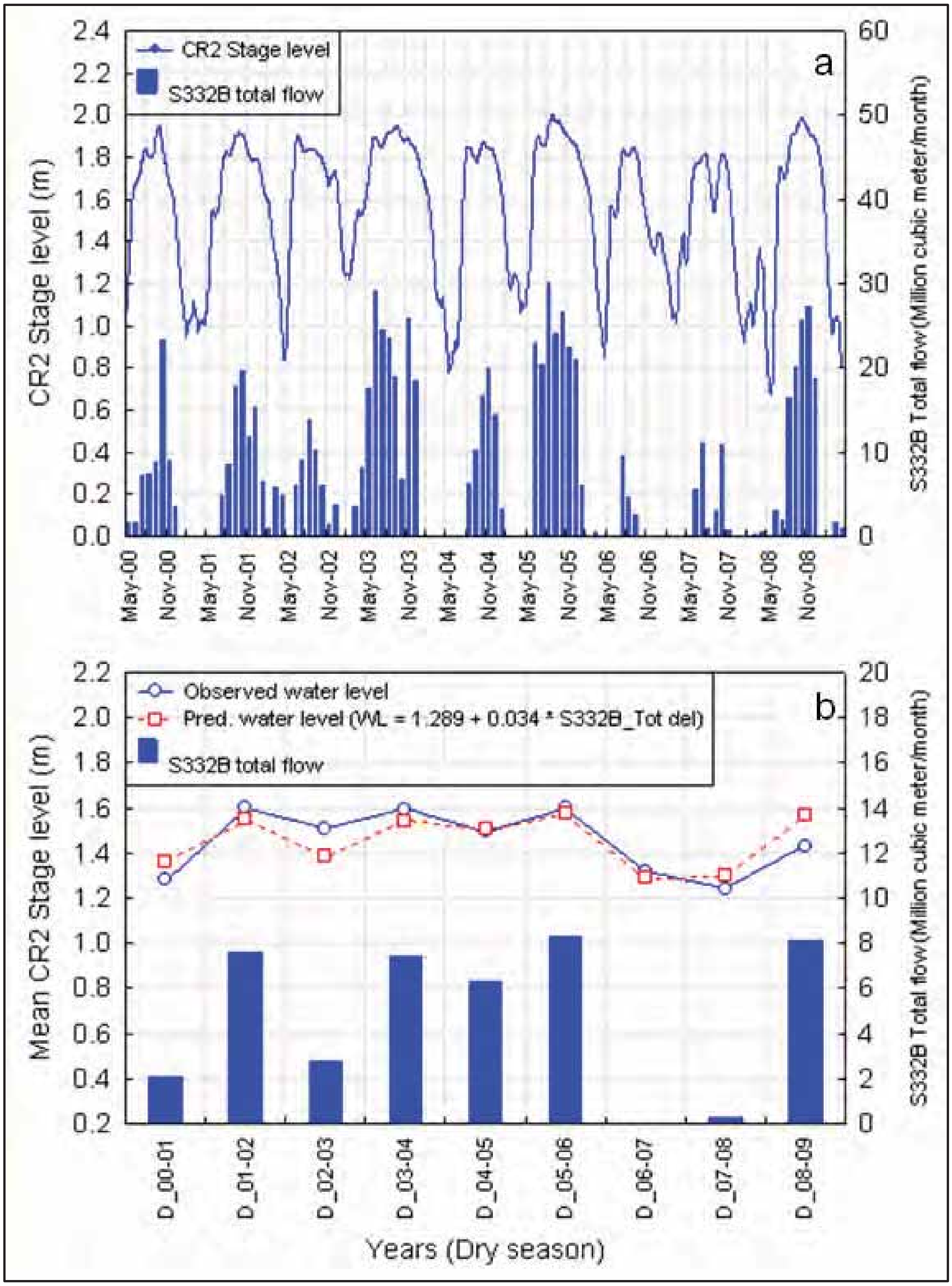


FIGURE 8-57. A) 30-DAY MOVING AVERAGE WATER LEVEL AT CR2 STAGE RECORDER AND MONTHLY MEAN S332B TOTAL FLOW AND B) OBSERVED AND PREDICTED WATER LEVEL AND SEASONAL TOTAL WATER FLOW THROUGH S332B NORTH AND WEST

As envisioned in the authorization of the construction and operation of water pumps S332BN, S332BW, S332C and S332D, and the associated retention ponds, whose goal is to restore more natural hydrologic conditions in Taylor Slough Basin, marl prairies in Taylor Slough Basin have become wetter in recent years. In turn, one may reasonably expect that these operational changes have also lead toward vegetation assemblages characteristic of wetter environments than those present in Taylor Slough in the 1980s and earlier decades.

8.5.4.2.2 Hydrology and Vegetation Change

The dynamics of vegetation composition in the marl prairies and sloughs of Taylor Slough have in fact tracked hydrologic changes within the basin. During the 1990s, vegetation composition south of S332 became more like that of long hydroperiod marshes (Armentano et al., 2006), but after 1999 it shifted towards a drier type with a community characteristic of less prolonged flooding. Changes were noticeable on T2 (refer to ***Figure 8-54*** for transect location), which is located near the Taylor Slough Bridge, where vegetation was surveyed more frequently than other transects in the network. Between 1992 and 1999, almost all sites on this transect followed a trajectory that paralleled the hydrologic gradient from dry to wet conditions (***Figure 8-58***), and this compositional shift was statistically significant at more than half of the sites surveyed. After 1999, however, several sites, mostly in the narrow slough itself or along its western fringe farthest from the canal system, took on an opposite trajectory and trended from wetter to drier conditions (***Figure 8-59)***. The initial point on each site trajectory represents the 1992 sampling and the end of the arrows represents the 1999 sampling. A few prairie sites adjacent to the slough along T1 showed a similar trend.

Marl prairie vegetation north of S332 was unaffected by its direct delivery into the Taylor Slough. Since 2000 and 2001, however, operation of new water pumps and delivery of water into a series of retention ponds west of the canal has influenced vegetation in this region. More than half of all sites along T4 and T5, which are the first slightly north and the second slightly south of S332D, respectively, showed a significant change in vegetation composition from drier type to wetter type. Vegetation south of S332D (T5) appeared to be affected by seepage from the L31W Canal, and vegetation to the north along T4 seemed to be influenced by surface and subsurface flow from the retention ponds receiving water from S332B and S332C.

Through the construction and operation of a series of water pumps and retention ponds, more natural surface flow was expected to replace the direct delivery of water into the slough. Water was expected to seep into the Biscayne Aquifer reducing eastward ground water seepage from Everglades National Park. Vegetation monitoring results indicate, in recent years, the operation of water pumps and seepage from adjacent retention ponds have significant effects on marl prairie and slough vegetation, mediated through changes in hydrologic conditions in the Taylor Slough Basin. Within the slough portion of the basin, reduced annual mean water level, partially resulting from replacement of direct delivery through S332 by surface flow, caused a shift in vegetation composition indicative of drying conditions. In contrast, seepage from canal and retention basins into the eastern marl prairies caused a shift to vegetation adapted to relatively wetter conditions. In a recent study, Gaiser et al. (2008) suggested that in the vicinity of the L31N and L31W canals and adjacent basins, the groundwater contours are parallel to the L31N Canal, indicating a predominant direction of groundwater flow toward the east and southeast away from Everglades National Park. However, seasonal variation in groundwater contours and the presence of groundwater mounds, or “the “bulls-eye” associated with water control structures on the L31N and L31W canals, suggest water also flows towards the west during certain periods, particularly in the dry season (Gaiser et al., 2008). These flow patterns may be affected by operational changes necessitated by climatic extremes, such as the record-setting drought of 2006 and 2007 (Meeker, 2008). This two-year drought may have impacted the trajectory of vegetation change.

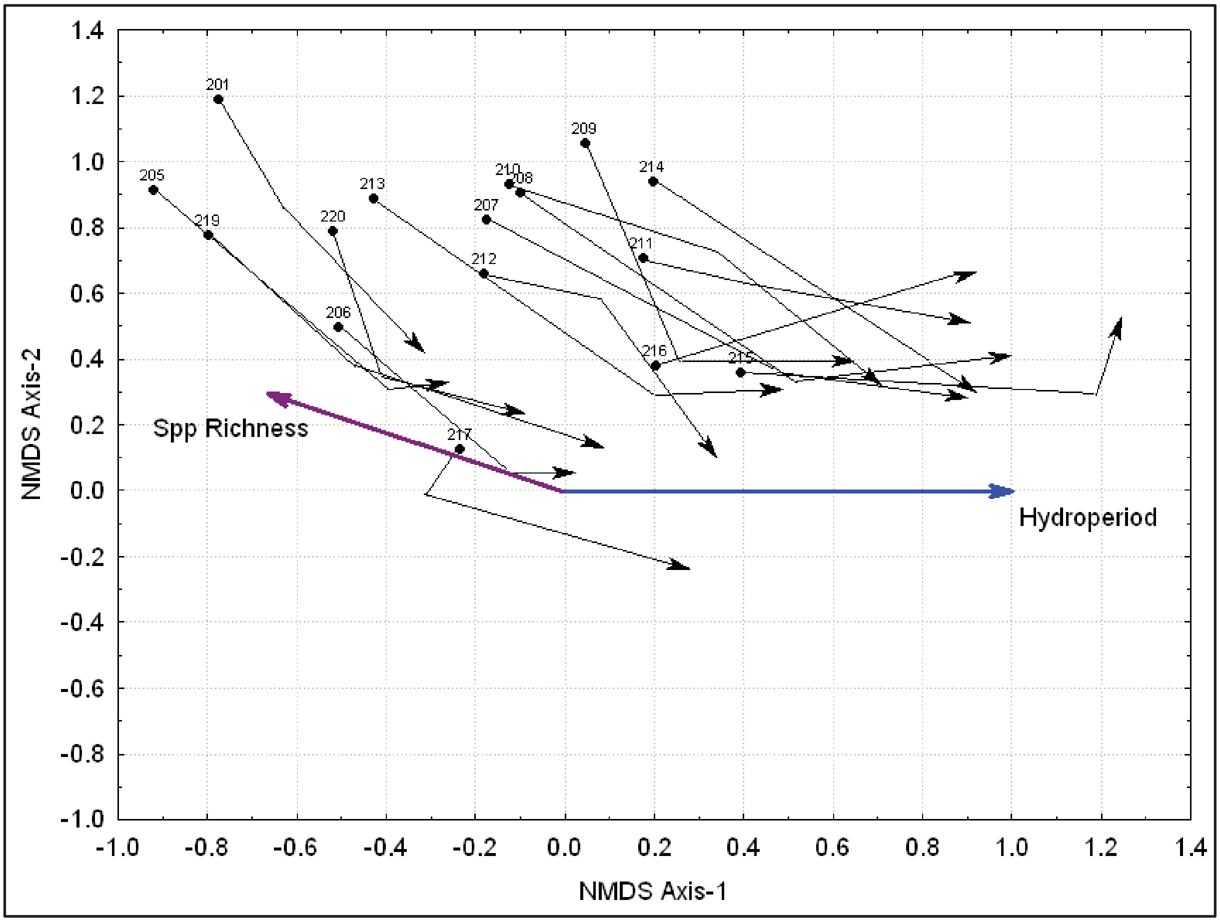


FIGURE 8-58. THE NMDS ORDINATION SHOWING THE TRAJECTORY OF SITES FROM T2 SAMPLED IN 1992, 1995 AND 1999 FOR SITES THAT SHOWED SIGNIFICANT RATE OF CHANGE (HALF CHANGE PER YEAR) IN SPECIES COMPOSITION ALONG THE HYDROLOGY GRADIENT

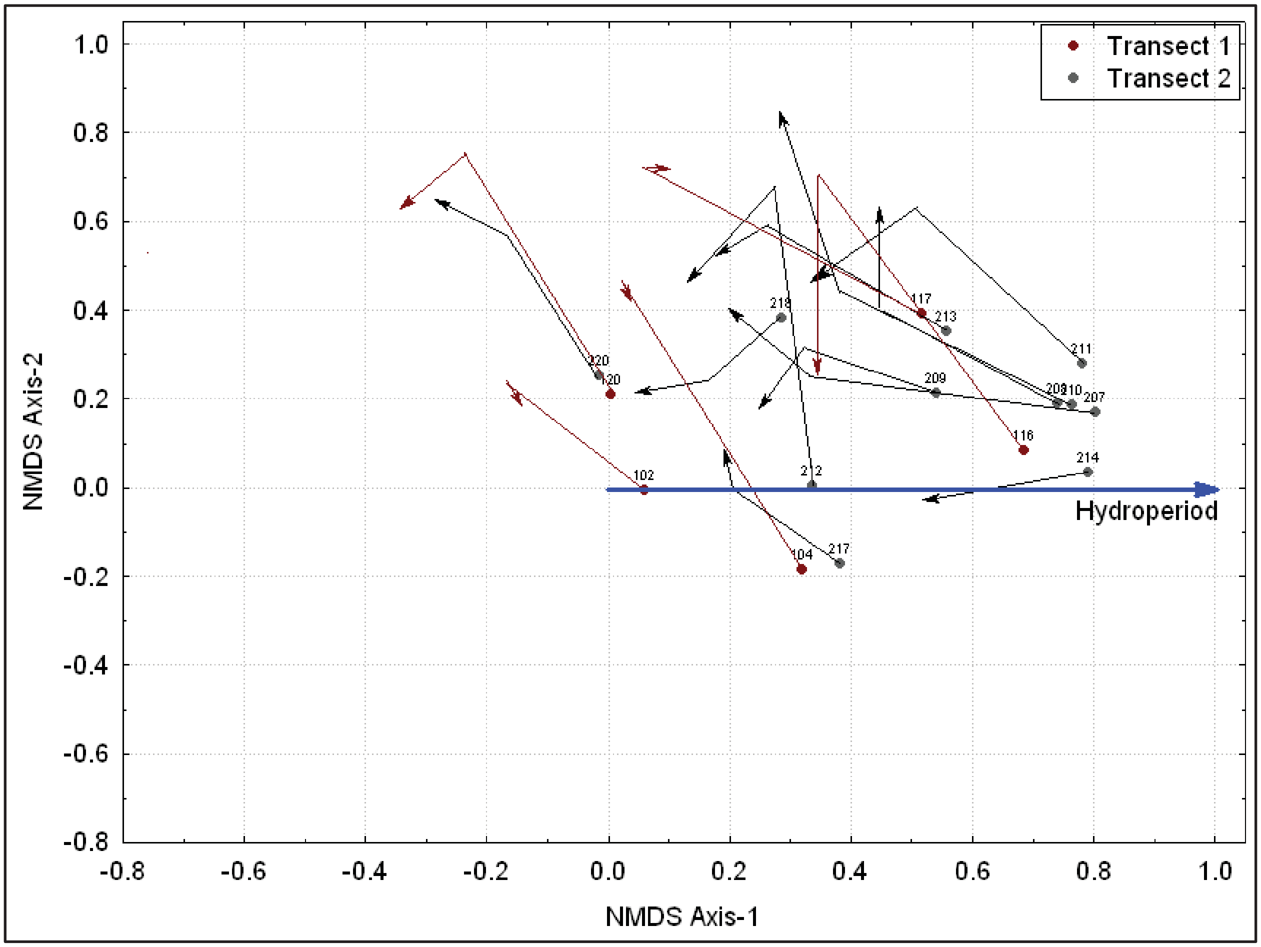


FIGURE 8-59. THE NMDS ORDINATION SHOWING THE TRAJECTORY OF SITES FROM T2 SAMPLED IN 1999, 2003 AND 2007 FOR SITES THAT SHOWED SIGNIFICANT RATE OF CHANGE (HALF CHANGE PER YEAR) IN SPECIES COMPOSITION ALONG THE HYDROLOGY GRADIENT

8.5.4.2.3 Hydrology, Phosphorus and Vegetation

It has been hypothesized by Childers et al. (2003) that long-term seepage of water along the canal levee and Everglades National Park interface, preceded by direct point source input of water into Taylor Slough, has already or may in the future cause soil P enrichment in Taylor Slough resulting in a compositional shift toward species indicative of P enriched soil. Vegetation composition at some sites, particularly along T1, T4 and T5, has recently shown such a change. The sites have shifted in the ordination space along trajectories that are positively correlated with increasing P content in the soil (***Figure 8-60***). Such shifts of the sites in the direction of increasing P were more obvious after 1999. In 2007, for the first time since initiation of the Taylor Slough vegetation monitoring program in 1979, cattail (*Typha domingensis*) was observed in five plots along T1 in 2007. This transect is in closest proximity to S332, from which canal water flowed directly into Taylor Slough, and all five of the cattail-invaded plots are in or immediately adjacent to the central drainage of Taylor Slough (***Figure 8-61***).

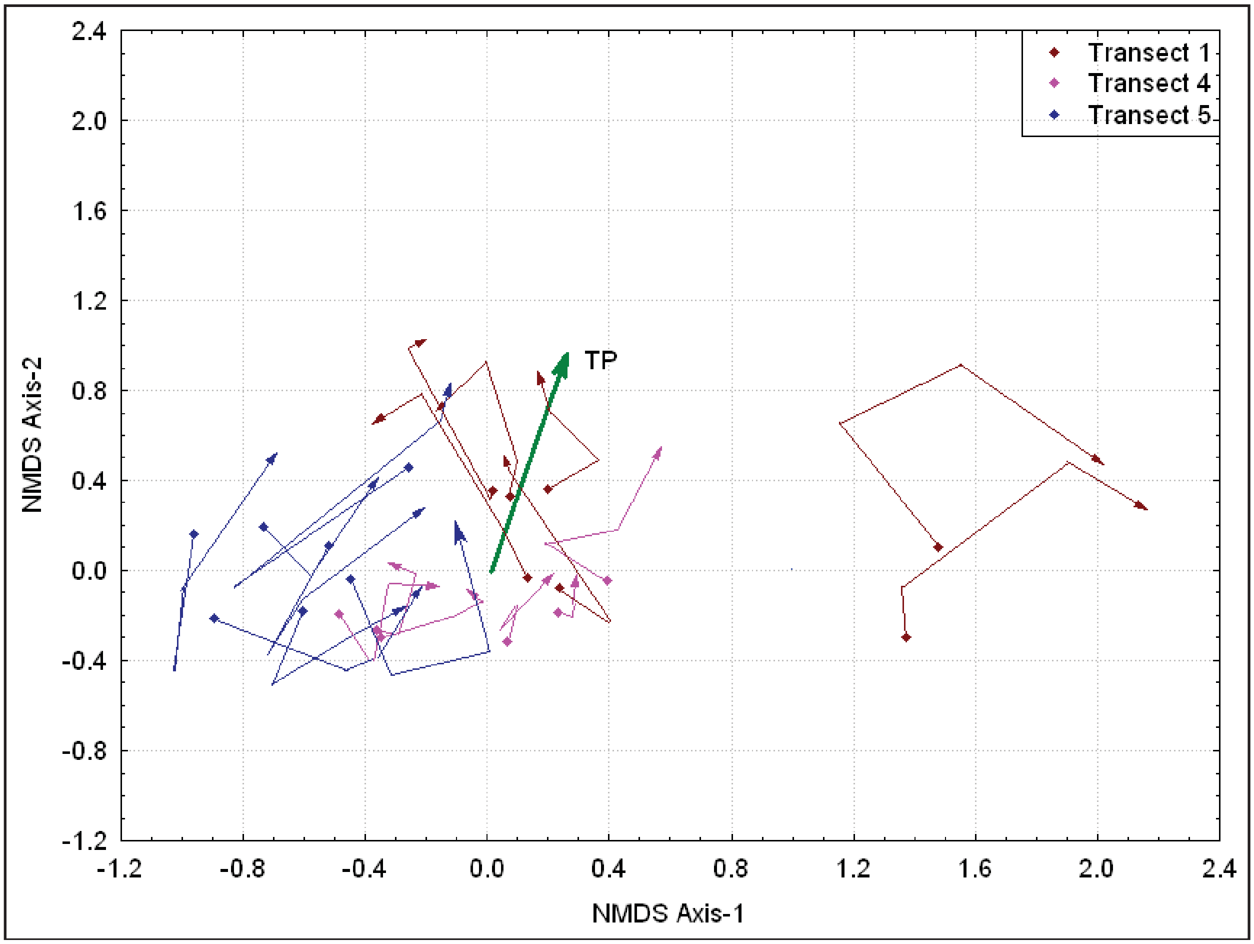


FIGURE 8-60: THE NMDS ORDINATION SHOWING THE TRAJECTORY OF SITES FROM T1, T4 AND T5 SAMPLED IN 1995 (T1) OR 1997 (T4 AND T5), 2003 AND 2007 THAT SHOWED SIGNIFICANT (P≤0.01) RATE OF CHANGE (HALF CHANGE PER YEAR) IN SPECIES COMPOSITION ALONG A TP GRADIENT

Currently, soil P content is unknown for the Taylor Slough survey points themselves, and must be inferred from vegetation patterns observed in similar locations elsewhere in the Everglades. Other studies within Everglades National Park have demonstrated that water input from the canals has altered soil P in the marsh. For instance, water delivered through the S12 structures from the Tamiami Canal into Shark River Slough caused soils to be enriched with P, resulting in altered plant communities (Childers et al., 2003). In the eastern Everglades, the P content in periphyton was higher at the marl prairies sites near the L31W Canal and in detention basins near the L31N Canal than in adjacent marl prairie sites to the west, reflecting the long-term exposure of the canal-side sites to seepage (Gaiser, 2006; Gaiser et al., 2008). Periphyton is known to show a quick response to increased P concentration in surface water, and is a precursor of P enrichment of soil (Gaiser et al., 2004a). Therefore, a time lag in P enrichment in the soil is inevitable.

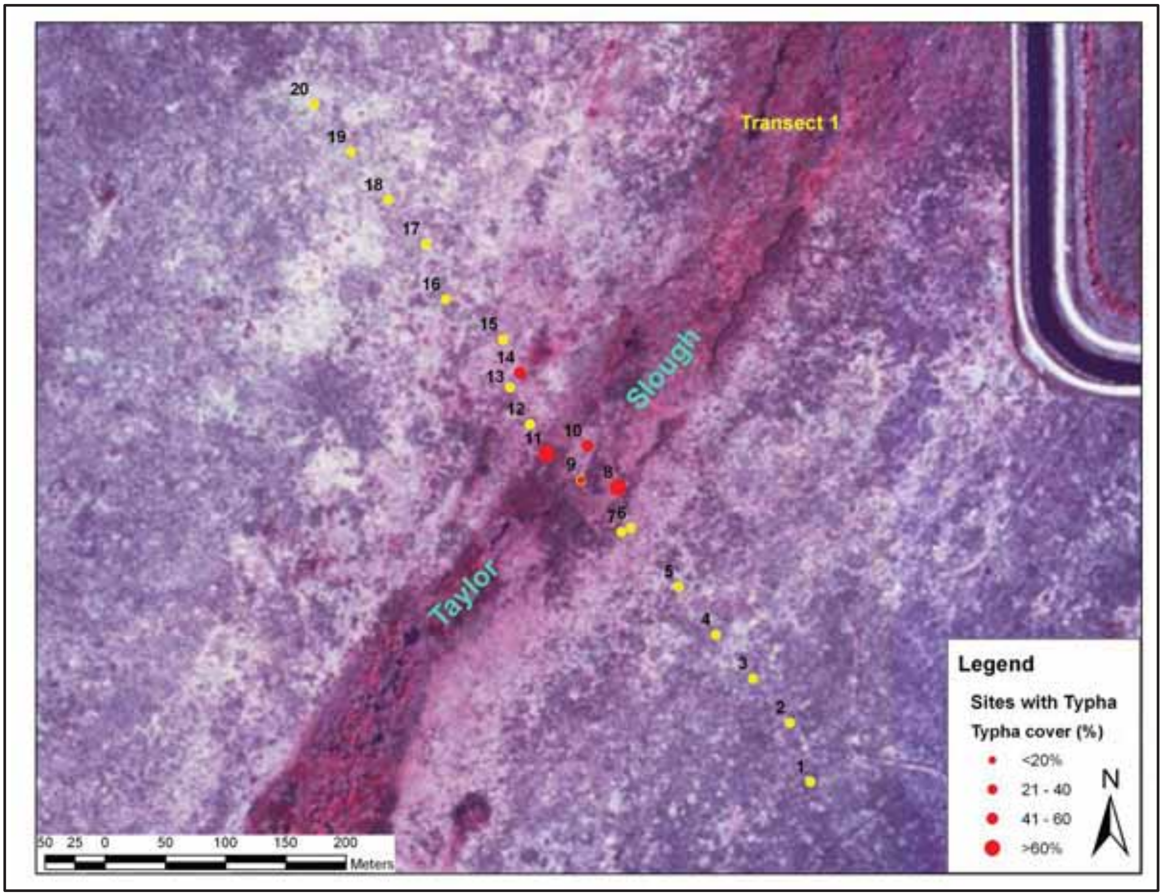


FIGURE 8-61. T1 SHOWING THE SITES THAT HAD CATTAIL IN 2007

In the Taylor Slough Basin, P concentrations in surface water have been monitored since 1989. The long-term trend suggests that even though the 12-month flow-weighted mean TP concentrations in inflows to Everglades National Park through Taylor Slough and other coastal basins decreased after 2000, when S332D replaced S332 as the means of delivering water into the basin (Mo et al., 2007), frequent spikes exceeding the long-term limit of 11 parts ppb P were observed after 2003. Moreover, for several years during the S332 inflow period, these concentrations either exceeded or remained close to the limit (Mo et al., 2007). Hence, it is likely that the cumulative effects of P loading in canal outflows have enriched adjacent soils in P, resulting in an increase in the abundance of species that respond, as cattail does, to elevated P. The extent of P-enrichment in the soil may also depend on the total flow of water. For instance, soil P in the S332B, C and D basins were correlated with water delivered from the canal into the basin (Gaiser et al., 2008). Therefore, it is important to minimize the P loading in the water directly entering Everglades National Park from the canal. Minimization of P loading is of course linked to the effectiveness of the detention ponds in functioning as STAs.

Long-term monitoring of vegetation and other indicators of change in hydrologic conditions and associated parameters is essential to provide feedback for the effective and AM of marl prairies in Everglades National Park. Effectively leveraging of these monitoring efforts requires an analytical tool capable of extracting broad-level patterns and assessing site-specific responses of biological communities to the changes in physical and chemical drivers and stressors. Trajectory analysis is a new multivariate analysis technique effective in extracting the underlying pattern in community data. It is also appropriate for testing the extent and direction of a change in community composition with reference to restoration targets. Therefore, it has been used to analyze long-term changes in Taylor Slough vegetation.

8.5.5 Vegetation Mapping

8.5.5.1 Monitoring

Vegetation mapping is conducted across all of the Greater Everglades Wetlands, including the WCAs, Everglades National Park, Big Cypress National Preserve, and the Biscayne Bay coastal wetlands. Vegetative land cover and community types are characterized over 50-square meter pixels using high resolution digital aerial photography and state-of-the-art stereo-photogrammetry. A hierarchical classification scheme (Rutchey et al., 2006) is used to classify the vegetation into 284 distinct community types. New digital aerial imagery encompassing nearly 2,000-square miles was acquired during April 2009 for Everglades National Park, southeastern Miami-Dade coastal wetlands, and Biscayne Bay coastal wetlands for the MAP vegetation mapping project (***Figure 8-62***).

Digital aerial photography for another 361 square miles was acquired under the same contract in order to take advantage of economies of scale and the resulting cost savings. These other areas included the Loxahatchee Impoundment Landscape Assessment Area (LILA), STAs, northern WCA 2A, Lake Istokpoga, Lake Josephine, Kissimmee Upper Chain of Lakes, and the Kissimmee river floodplain. Ground pixel resolution is approximately one square foot for the entire project with the exception of the LILA area, which was collected at a one-quarter square foot resolution. Improved color intensity range and ground resolution, makes this data set superior to any products to be utilized to date for conducting vegetation mapping in the Greater Everglades (***Figure 8-63).***

8.5.5.2 Results

Vegetation maps for WCA 1 and WCA 2 (Rutchey et al., 2008) have been completed (and ***Figure 8-64***, ***Figure*** *8-65* and ***Figure 8-65)***. These images were acquired over time and will document the changes in cattail coverage and any shifts in community types that are concurrent with alterations to hydrology and/or nutrient inputs associated with restoration, climate and management.

The current status of the MAP vegetation mapping project is on schedule and a new 2004 aerial photo-based vegetation map of WCA 3 will be completed by the end of 2009. This data provides an update to the earlier map done for this impoundment in 1995 (Rutchey et al., 2005). The next area to be mapped, based on priorities, is northeastern Shark River Slough (***Figure 8-66***), is scheduled to be completed in 2010.

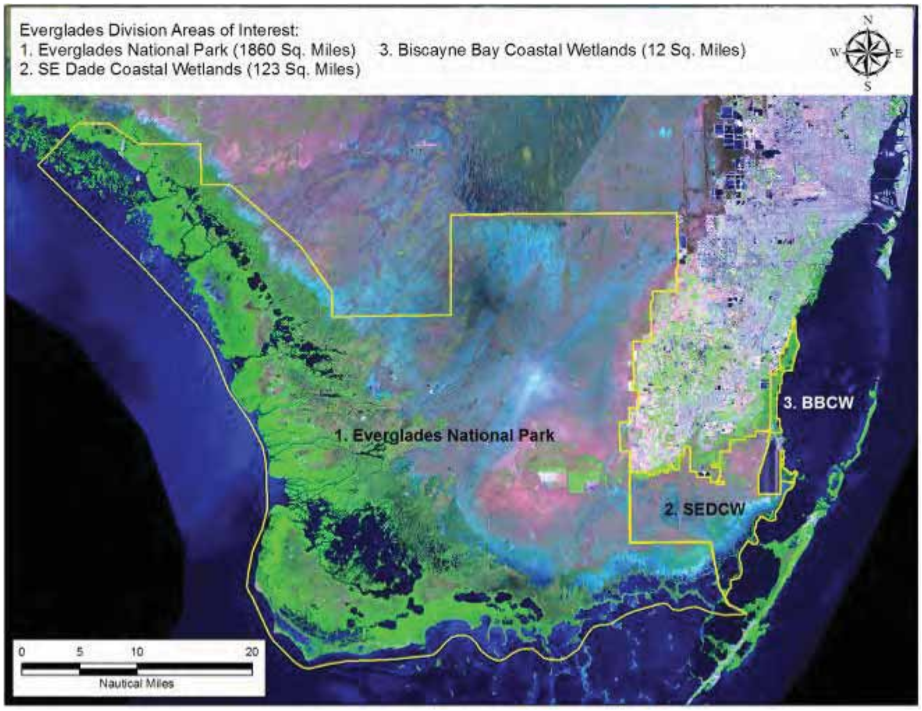


FIGURE 8-62. 2009 DIGITAL AERIAL PHOTOGRAPHY ACQUISITION INCLUDES EVERGLADES NATIONAL PARK, SOUTHEASTERN MIAMI-DADE COASTAL WETLANDS, AND BISCAYNE BAY COASTAL WETLANDS

FIGURE 8-63. AN EXAMPLE OF A COLOR INFRA-RED COMPOSITE OF DIGITAL AERIAL PHOTOGRAPHY LOCATED AT THE FLAMINGO MARINA IN EVERGLADES NATIONAL PARK WITH THE ENLARGEMENT SHOWING A BOAT BEING LOADED ONTO A TRAILER AND THE BOAT'S PROP WASH

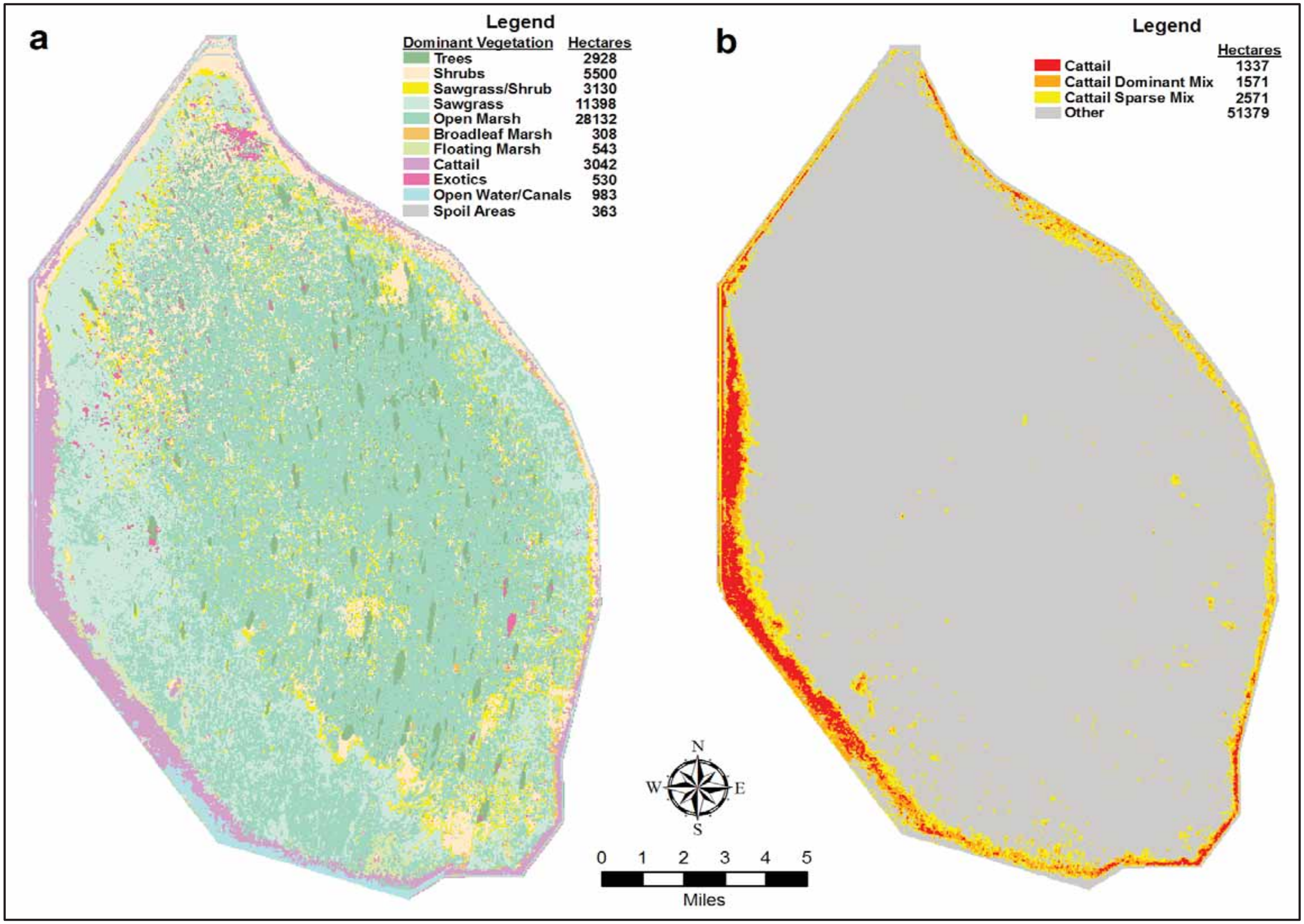


FIGURE 8-64. HIGH RESOLUTION A) VEGETATION COVER AND COMMUNITY TYPES IN WATER CONSERVATION AREA 1 AND B) A DETAILED FOCUS ON EXTENT ON INVASIVE CATTAIL EXPANSION INTO THE REGION

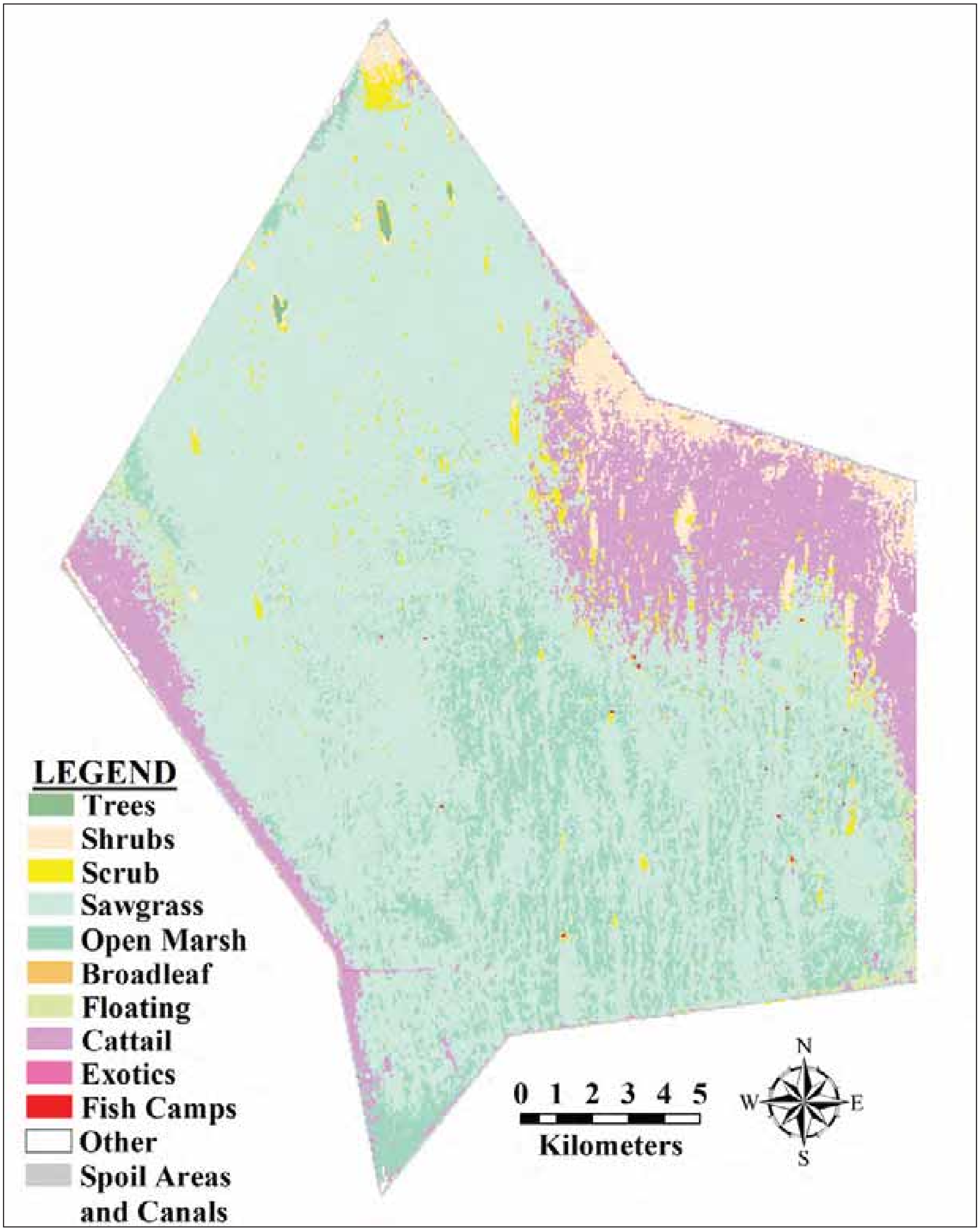


FIGURE 8-65. HIGH RESOLUTION AERIAL IMAGERY SHOWS VEGETATION COVER AND COMMUNITY TYPES IN WATER CONSERVATION AREA 2

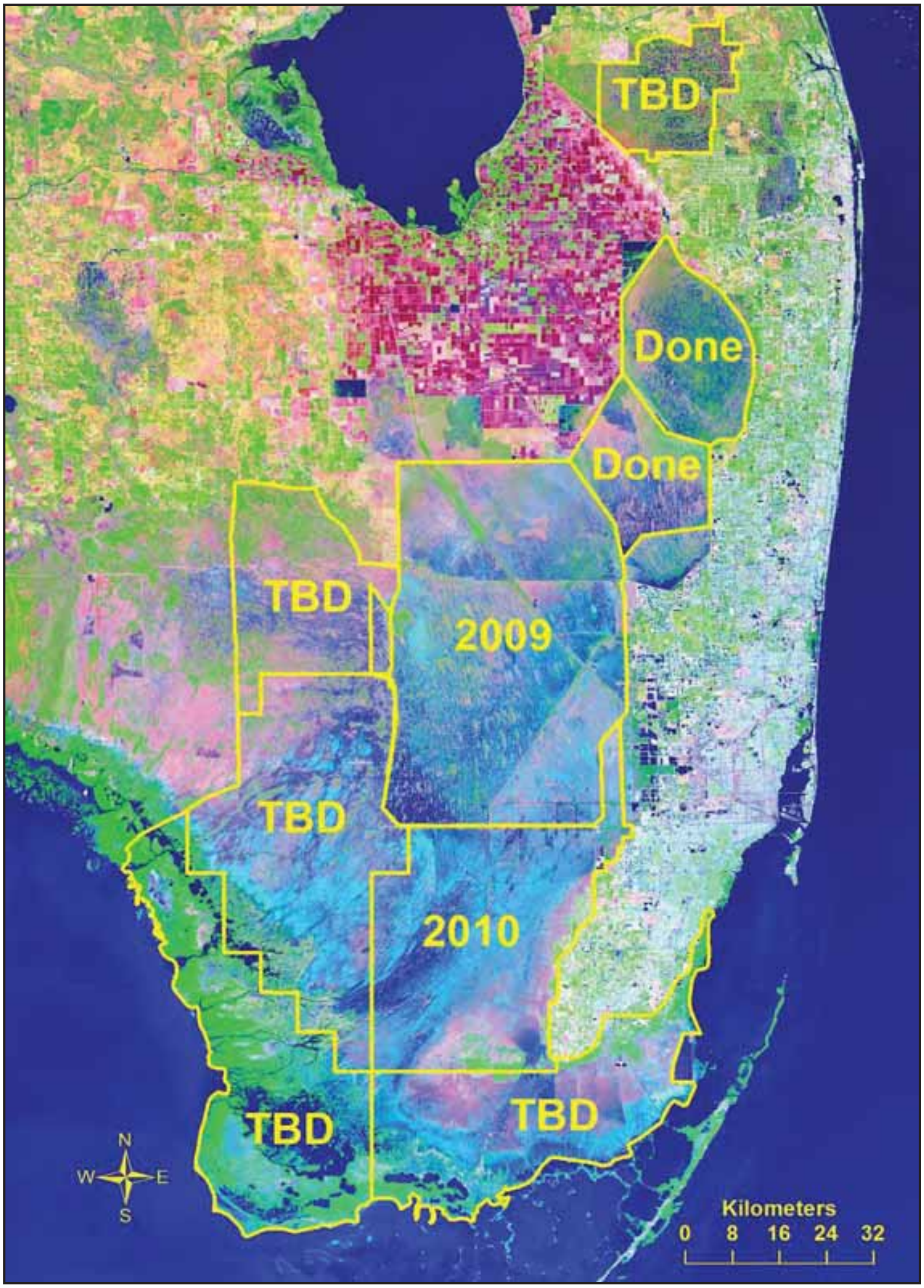


FIGURE 8-66. VEGETATION MAPPING AREAS THAT HAVE BEEN COMPLETED, ARE PROJECTED TO BE COMPLETED, AND TO BE DECIDED

8.5.6 Regional Distribution of Soil Nutrients

8.5.6.1 Monitoring

Systematic and comprehensive baseline soil mapping for the Greater Everglades Wetlands was undertaken in 2003 by the Everglades Soil Mapping Project (ESM) (Reddy et al., 2005) as well as under the USEPA Regional Environmental Monitoring and Assessment Program (REMAP), which was initiated in 1993 (Scheidt and Kalla, 2007) (***Figure 8-67)***. Subsequent studies have made comparisons of the 2003 survey to previously collected data and suggest observations of differences in soil attributes is possible. However, whether to attribute that difference to natural variability or systematic change is unknown.

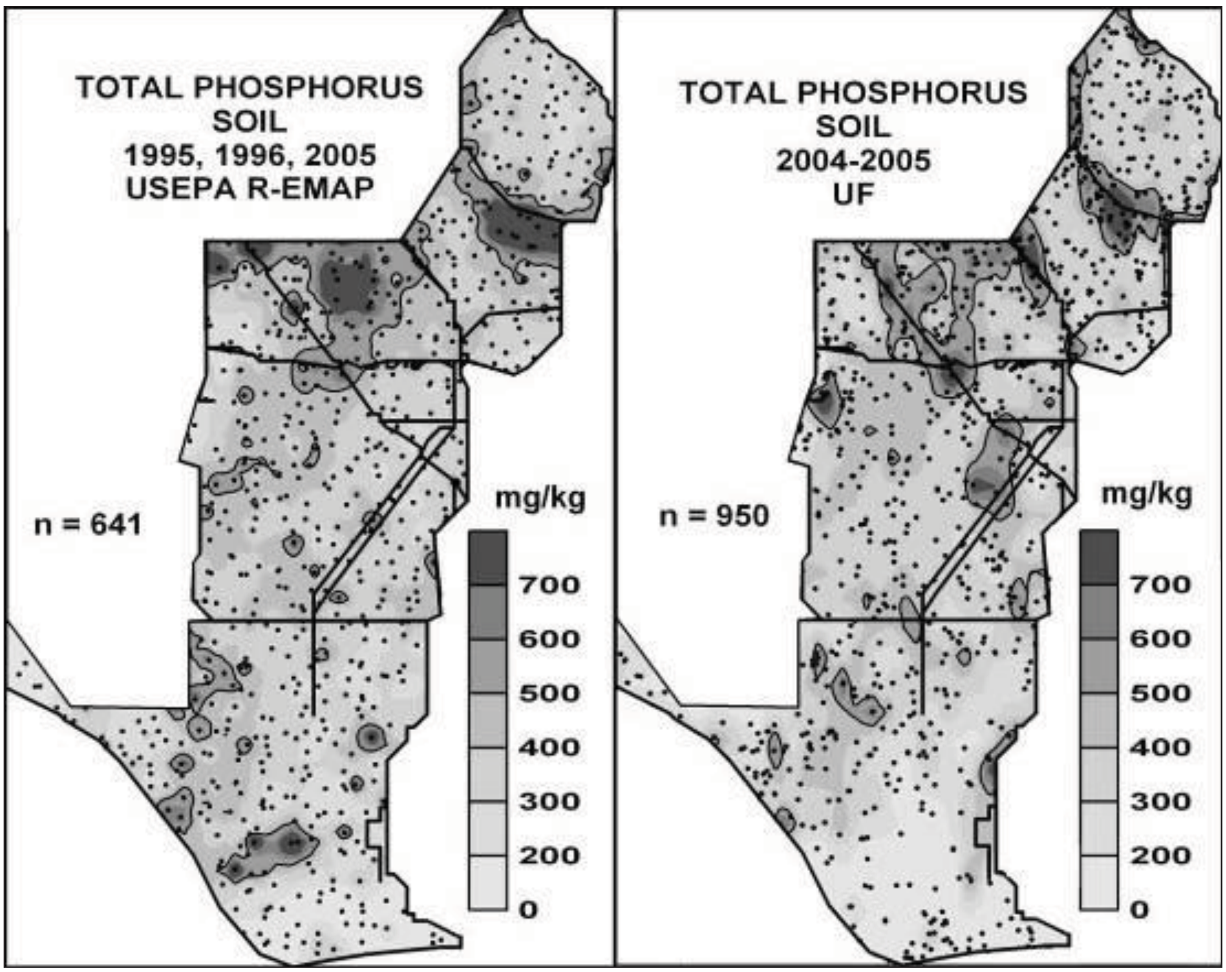


Figure courtesy of Osborne et al. in review

FIGURE 8-67. COMPARISON OF TOTAL PHOSPHORUS MAPS GENERATED FROM BOTH REGIONAL ENVIRONMENTAL MONITORING AND ASSESSMENT PROGRAM AND EVERGLADES SOIL MAPPING EFFORTS

The aim of the current soil nutrient mapping project is to determine if the differences observed are due to natural variability or systematic changes. Two key questions are being addressed for future mapping efforts:

- How much short range variability can be expected across the Greater Everglades landscape and how does that compare with regional variation gradients?
- What are the ecological drivers of short-range variability and can these be accounted for to improve the likelihood of successful change detection after future sampling?

As mentioned earlier in this chapter, 80 primary sampling units have been established across the Greater Everglades region to guide implementation of regional monitoring efforts. For this study, a subset of eight primary sampling units was selected to represent the range of ecological and hydrologic conditions across the system (***Figure 8-68***). Sampling was initiated in August 2007 and was completed in December of 2008. Soil samples were analyzed for TP, loss on ignition, total carbon and TN.

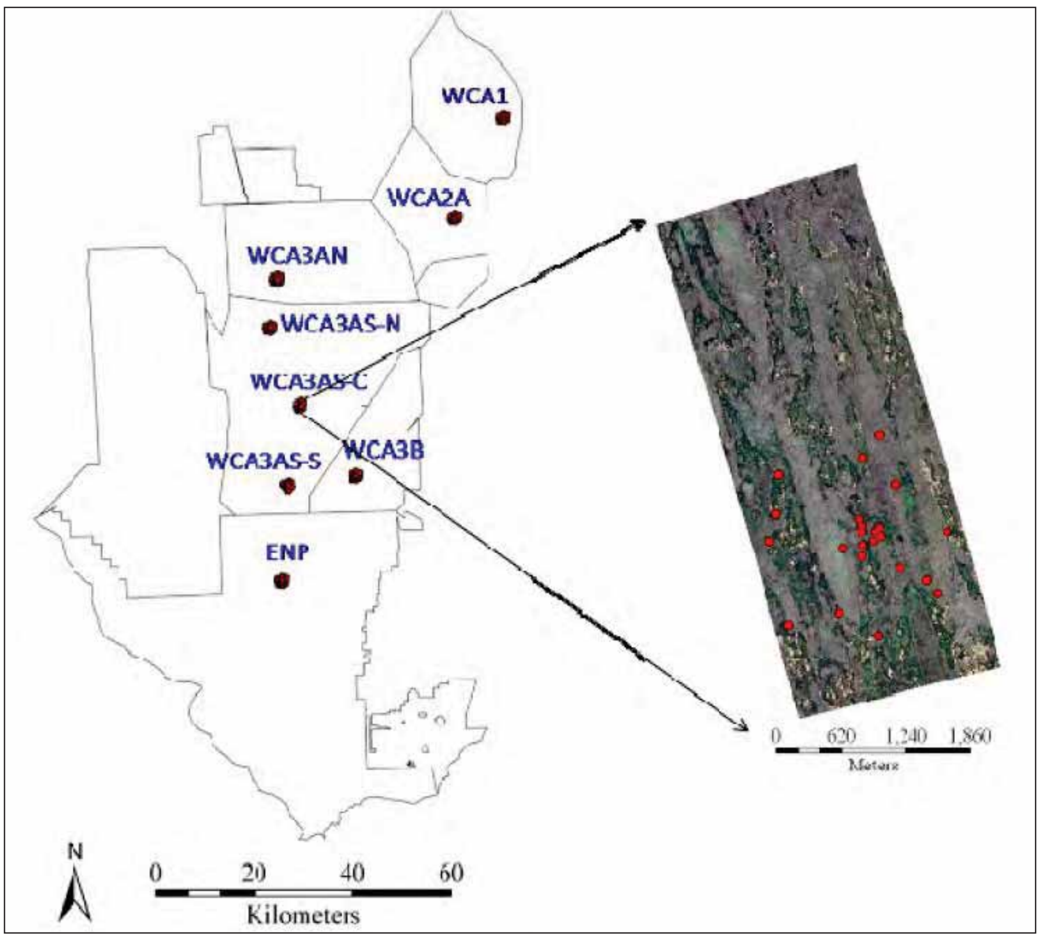


FIGURE 8-68. PRIMARY SAMPLING UNITS USED IN SOIL MAPPING WITH AERIAL IMAGERY AT RIGHT SHOWING SAMPLE LOCATIONS FOR SOUTH-CENTRAL WATER CONSERVATION AREA 3A

8.5.6.2 Results

Fine-scale spatial variability in soil TP is generally large. Strong spatial structure was observed in nearly all eight of the primary sampling units with ranges varying between 15 and 931 meters. To determine the relative magnitude of the short range variability, spatial model parameters were compared at the scale of primary sampling units with results from ESM at the scale of entire hydrologic partitions: WCA 1, WCA 2A, Everglades National Park. As expected, fine-scale semi-variances were generally much smaller than that observed in the ESM results, but overall variability was comparable to ESM values. These results suggest strongly that fine-scale processes generate substantial local variability, and that this variability is realized over ranges between 50 and 500 meters.

As mentioned above, the soil TP values from the fine-scale study can be comparable to those found in the larger-scale ESM study (***Figure 8-69***). There is a strong concordance of mean values, with high TP levels observed in WCA 2A, and low levels observed in Everglades National Park, WCA 1 and WCA 3B. However, it is clear that the standard deviations are far larger for the ESM dataset, as might be expected given large-scale variability in soil forming factors and anthropogenic enrichment gradients that might be missed by sampling only within a 2 kilometer by 2 kilometer primary sampling unit.

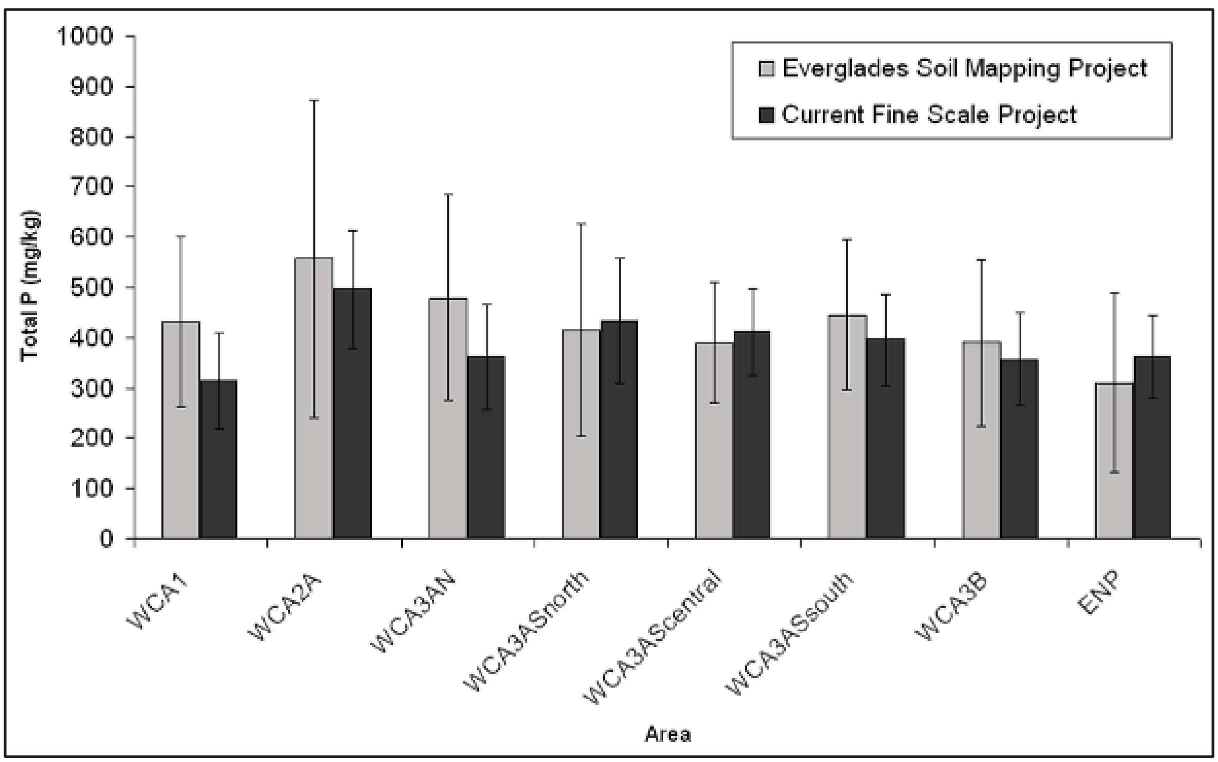


FIGURE 8-69. COMPARISON OF MEAN TOTAL PHOSPHOURS (± SD) AS MILLIGRAM PER KILOGRAM VALUES BY HYDROLOGIC PARTITION FROM THE LARGE-SCALE EVERGLADES SOIL MAPPING PROGRAM AND THE CURRENT FINE-SCALE MAPPING PROJECT USING PRIMARY SAMPLING UNITS WITHIN EACH HYDROLOGIC PARTITION

Of particular interest, however, is the observation that the measures of dispersion are correlated between the regional- and primary sampling unit-scale analyses (***Figure 8-70***). While the association is poor in some areas, such as the central WCA 3AS site, the slope of the line indicates that the sample variability is roughly half (53 percent) as large at the small scale as at the large scale. This can be interpreted in several ways. Clearly, it suggests that substantial variability occurs over small scales, particularly for primary sampling units above the fitted line in ***Figure 8-70***, that is, where the dispersion at the fine scale is even larger than 50 percent of dispersion at the regional scale. It also suggests the processes that principally control variability are the same at both scales, and regional gradients may be of less importance than previously expected. A primary conclusion is fine-scale processes would generate significant uncertainty about what constitutes actual change when future maps are compared to baseline maps. Notably, this effect was not uniform across all hydrologic partitions, meaning that the interpretation of any future change maps will need to take into consideration geographic location.

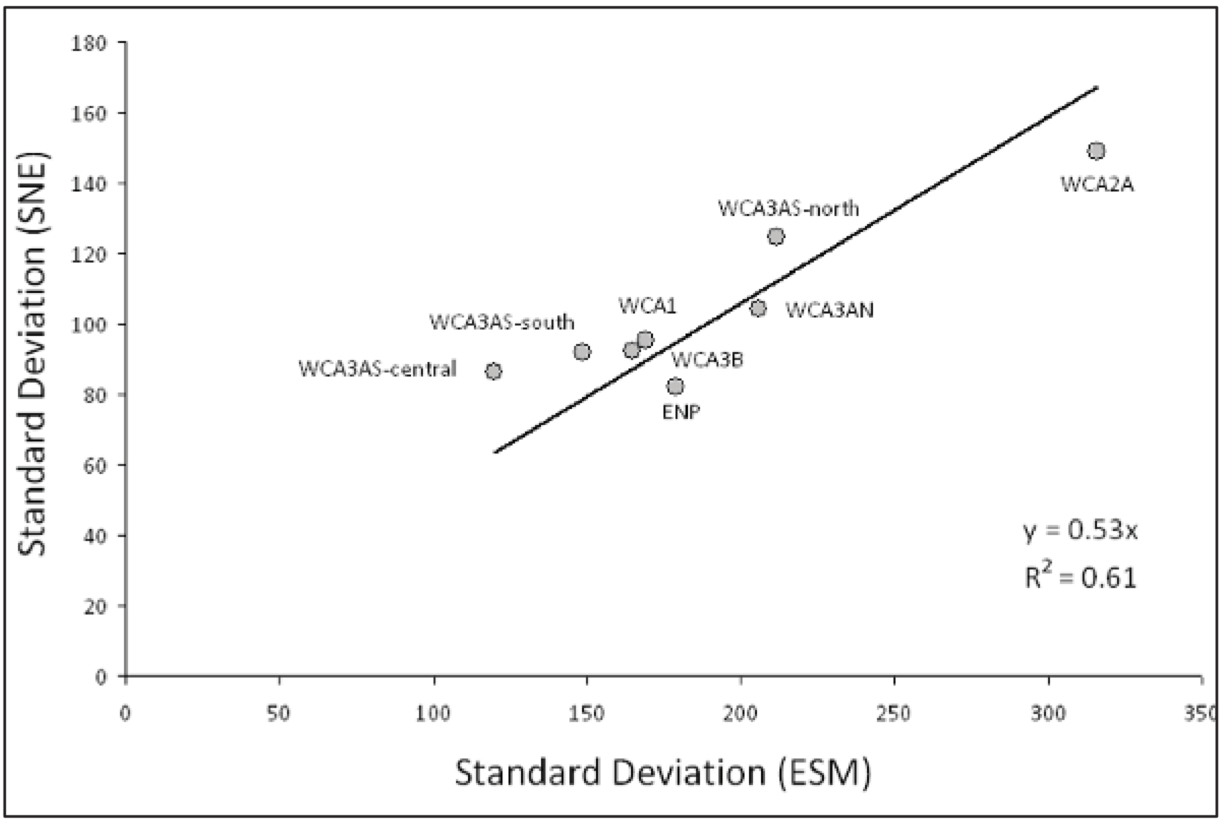


FIGURE 8-70. THE DISPERSION IN THE POPULATION (SD OR TP VALUES) AT THE FINE SCALE (SOIL NUTRIENT EXAMINATION) IS OVER 50 PERCENT OF THE DISPERSION AT THE LARGE SCALE (EVERGLADES SOIL MAPPING PROGRAM)

Fine-scale spatial variability in soil TP appears to be strongly controlled by local and soil properties. Specifically, a strong association between TP and organic matter content was observed in areas of the Everglades with a moderately to severely reduced hydroperiod. No such association was observed in areas with a regular or extended hydroperiod. More importantly, strongly significant negative correlation between TP and water depth was observed in areas of the Everglades where the ridge and slough landscape is relatively well conserved. This is perhaps the first demonstration of a significant association between soil accretion processes and TP. This suggests that spatial structuring of TP concentrations is influenced by patterns of elevation variability, which is in turn modified by ecosystem state. No effect of proximity to tree islands was observed in these data, but fine-scale analyses of tree island soil nutrients suggest soils in the immediate vicinity of tree islands are P enriched. Overall, the data suggest that biological processes create large self-organized fine-scale gradients in nutrient content that can substantially impair inference from coarse-scale maps.

8.5.7 Summary

8.5.7.1 Assessment of Hypothesis Clusters

The relatively short duration of the landscape monitoring program prevents a thorough evaluation of the landscape pattern hypotheses cluster. A full evaluation of many of these hypotheses requires implementation of the full landscape monitoring program (***Figure 8-35***). However, with a few exceptions, initial results show a high degree of hypothesis fidelity. For example, the hypothesis for ridge and slough landscape dynamics suggests the disequilibrium of organic matter accretion and loss processes cause degradation of the ridge and slough habitat and a flattening of the landscape (***Figure 8-33***). The data show clear relationships between landscape patterning and peat microtopography (***Figure 8-38*** and ***Figure 8-39***). These results indicate the differences between ridge and slough peat surface elevations decrease with the loss of patterning and hydrologic disturbance. Complementing these results, and in agreement with the hypothesis, the initial monitoring and analyses of the soil nutrient within the ridge and slough landscape have provided key insights into the processes that maintain peat surface microtopographic differences in the ridge and slough landscape. C:N and N:P ratios suggest ridge edges are composed of organic material representing a mixture of ridge and slough end-members (***Figure 8-41***). This suggests the transport of organic matter between sloughs and ridges is an important mechanism in ridge edge formation. Another conclusion of this study is these soil data are not consistent with alternative hypotheses that assert ridge formation is caused by transpiration-driven nutrient concentration or preferential deposition of inorganic particles on ridges. Forthcoming data analyses will provide information of relative importance of organic soil accretion and loss in ridge and slough formation and persistence in disturbed areas.

This hypothesis cluster asserts the process of ridge and slough landscape pattern degradation is exacerbated by eutrophication, and restoration of sheet flow, in combination with related hydrology, water quality and fire patterns, will re-establish feedback conditions that sustain the microtopography of this habitat. While the short duration of the landscape monitoring program raises the level of uncertainty associated with accepting or rejecting this hypothesis, the data do suggest the causes of degradation and the conditions required for restoration of this habitat as described in the hypothesis are accurate. For example, in those areas where nutrients are low and sheet flow patterns and water depths are close to the historic condition (e.g. central WCA 3A), the microtopographic differences in ridge and slough peat surface elevations are maximized. In areas where sheet flow has been disrupted, as in impounded or over-drained areas, the differences in ridge and slough elevation are minimized. This suggests a disequilibrium in peat accretion and loss rates that, in combination with the altered hydrology, has also led to shifts in vegetation. Analysis of spatial autocorrelation (***Figure 8-40***) shows unequivocally how the coherence of the ridge and slough patterning has changed in areas characterized by the absence of sheet flow. Soil mapping results (see ***Section 8.6.6. Regional Distribution of Soil Nutrients***) support this hypothesis, and show a highly significant negative correlation between TP and water depth in areas of the Everglades where the ridge and slough landscape is relatively well conserved. This is perhaps the first demonstration of a significant association between soil accretion processes and P budgets in the Everglades. However, the role of eutrophication in the degradation of the ridge and slough landscape proposed in the oligotrophic nutrients hypothesis cluster (see ***Section 8.7*** below) has not yet been fully tested. Nutrient impacts in the system are also described in the sheet flow and water depth patterns hypothesis cluster presented in ***Section 8.2*** earlier in this module. The cattail expansions into nutrient enriched areas such as WCA 2 and into the upper reaches of Taylor Slough are just a few examples of landscape data that provide support for this hypothesis. Another example of how the field data support this hypothesis is found in the 15N signatures on tree island species (***Figure 8-53***).

The results of the ridge and slough, tree island and marl prairie-slough gradients monitoring discussed in this chapter are in general agreement with the plant community dynamics along elevation gradients hypothesis cluster (***Figure 8-34)***. This hypothesis cluster asserts composition and distribution of plant communities along elevation gradients are determined by patterns of hydroperiod, water depth, nutrient dynamics and fire throughout freshwater wetlands of the Greater Everglades. The ridge and slough monitoring shows the relative cover of vegetation types throughout the system generally reflects the remaining microtopographic differences (***Figure 8-39***). Similarly, the results from the tree island monitoring clearly show that the distribution of woody species across the landscape is related to their flood and drought tolerance (***Figure 8-46*** through ***Figure 8-49***). On the tall, dry islands of Everglades National Park, flood-intolerant species dominate, while in the flooded regions of WCA 3A, flood tolerant species are most prevalent. Interestingly, canopy-scale growth rates in Everglades National Park exhibit a quadratic relationship with water depths (***Figure 8-50***), suggesting the relative growth rates are reduced at both very wet and very dry conditions. This has not previously been documented for Everglades tree islands. Growth rates of many species themselves reflect the position along elevation gradients (***Figure 8-51***).

Perhaps the most rigorous support for this hypothesis is shown in the vegetation surveys along the marl prairie-slough transects in Taylor Slough. The changes were most noticeable on T2, near the Taylor Slough Bridge (***Figure 8-54***). Between 1992 and 1999, nearly all sites on this transect followed a trajectory that paralleled the hydrologic gradient from dry to wet conditions (***Figure 8-58***), and this compositional shift was statistically significant at more than half of sites surveyed. After 1999, however, several sites, mostly in the narrow slough itself or along its western fringe of the slough farthest from the canal system, took on an opposite trajectory and trended from wetter to drier conditions (***Figure 8-59***). A few of the marl prairie sites adjacent to the slough along T1 showed a similar trend.

This hypothesis cluster states that in most of the Greater Everglades Wetlands where hydroperiods and water depths have decreased, the hydrologic tolerances of the surviving plant communities should be adapted to wetter conditions. The results from the tree island monitoring suggest this hypothesis is correct. For example, the tree islands in WCA 3B are generally dry, but they are dominated by species that are typically found in much wetter habitats (***Figure 8-47***). This suggests the tree communities on these islands are not at equilibrium with their surroundings, and water depths in this area could be increased while still maintaining conditions within the hydrologic tolerance of the dominant tree species. As with the tree island monitoring, results from the ridge and slough monitoring also suggest, in general, plant community distributions in disturbed areas are not in equilibrium with current hydrologic conditions. For example, one expectation of the bi-modality exercise in the ridge and slough monitoring (***Figure 8-38***) is that the variance of water depths within each community should increase with both drainage and impoundment as the cover of sawgrass and/or slough species expand or contract as a fast variable, while peat surface elevations change much more slowly. However, observed effects were different in impounded versus drained areas. In partial agreement with this hypothesis, the variance in water depths found in both the ridge and slough communities does increase with impoundment. This may reflect a condition whereby sawgrass persists with inundation, but does not expand. It is possible sawgrass has differential die-off patterns with inundation versus drainage. In contrast, the smoother response with drainage may be an artifact of the system being more adapted to periodic drought. Sawgrass, in particular, is more adaptable to drought, as the species exhibits physiological stress responses with increasing inundation that limit its spatial expansion (Pezeshki et al., 1996; Brewer, 1996; Weisner and Miao, 2004), and decreased growth at higher water levels (Newman et al., 1996). Slough species, on the other hand, are much less drought tolerant and they are quickly lost when dry downs occur.

Overall, the results to date indicate the landscape pattern dynamics hypotheses cluster captures many of the key functional relationships between drivers, stressors and the ecological indicators in this system. Future work associated with the full landscape monitoring program will provide more comprehensive tests of this hypothesis.

8.5.7.2 Assessment of Status and Trends

Five performance measures are currently used to evaluate CERP projects: 1) extreme high and low water levels, 2) sheet flow in ridge and slough landscape, 3) wet prairie habitat, and 4) dry events in Shark River Slough, and 5) inundation pattern. These measures use hydrologic targets based on the NSM. The hydrologic conditions in the current system do not yet match those predicted by the NSM, which is expected since no large-scale restoration projects have been completed. Thus, it is not yet possible to determine the status and trends of the indicators relative to their restoration targets. However, the status and trends of the indicator relative to current hydrologic conditions is presented below.

One of the principal ecological indicators for the Greater Everglades Wetlands Module is sawgrass expansion. The vegetation mapping shows large areas of cattail expansion into WCA 1 and WCA 2 as a result of nutrient inputs (***Figure 8-64*** and ***Figure 8-65***). The marl prairie-slough gradients monitoring and analyses have revealed an expansion of cattail into Taylor Slough. As predicted by the ridge and slough landscape pattern dynamics hypothesis cluster (***Figure 8-33***), the expansion of cattail would continue as long as TP inputs exceed the amount coming only from direct rainfall. The hypotheses suggest rainfall should be the primary determinant of hydrologic status in the freshwater marshes. However, the hydrologic conditions in most areas of the Greater Everglades Wetlands do not match the conditions expected from rainfall patterns alone. The canal and levee system and water management operations create this imbalance, which has resulted in extended periods of high water in some areas, and extended periods of low water in others. As a result, several of the other indicators continue to show decline. For example, ridge and slough microtopography and landscape patterning has degraded throughout most of the former Everglades (Nungessar, 2009). This decline is attributed to the effects of altered hydrology and, in some areas, by eutrophication and cattail expansion. In ***Section 8.6.4 Marl Prairie and Slough Gradients*** above, shows how the decline in landscape patterning is revealed in statistical measures of anisotropy in the ridge and slough landscape. The altered hydrology has led to tree island drowning and burnout over time, two of the other primary attributes and indicators of the disturbed system. Upcoming aerial photogrammetry will determine current trends in tree island aerial extent in relation to marsh water levels and operations throughout the Greater Everglades Wetlands.

In one positive outcome, the marl prairie-slough gradients monitoring has detected a shift in vegetation indicative of wetter conditions in Taylor Slough. This area had been considered too dry relative to historic conditions and the shift to wet adapted species can be attributed to the effects of the recently completed detention basins and operations of the S332 structures.

8.5.7.3 Key Management Recommendations

Results from landscape monitoring support the broader management recommendation from the sheet flow and water depth patterns hypotheses to restore rainfall-driven volume, timing and distribution of sheet flow through WCA 3A and WCA 3B into Everglades National Park to produce water depths, hydroperiods, and flow patterns that are consistent with the best understanding of how the pre-drainage system responded to seasonal and interannual variability in rainfall. This includes redistribution of sheet flow through the natural flow corridors of the Shark Slough complex and Lostman's Slough in Everglades National Park in order to restore multi-year hydroperiods in slough systems of the park. It also requires elimination of unnatural pooling of water in WCA 3A above the L-29 and L-67 levees.

The only way to slow the continued degradation of the Everglades landscape system is to restore sheet flow and allow the water table and nutrients to fluctuate with rainfall similarly across the landscape.
The data show overwhelmingly the landscape patterns that characterized the historic Everglades continue to decline and have revealed important process-level information regarding the mechanisms by which these declines are precipitated. Altered patterns of sheet flow and related variables in the present system have caused disequilibrium of accretion and loss processes, which is exacerbated by eutrophication. That disequilibrium causes degradation in the ridge, slough and tree island microtopography toward a flattening of the landscape. Degradation of microtopography interacts with hydrology, eutrophication, fire and exotic plants to reduce the diversity and stability of habitats that were pre-drainage, large-scale features of the Everglades ridge and slough landscape. Declines in ridge and slough habitat diversity and stability during the last century have included expansion of sawgrass into sloughs and wet prairies, tree island

drowning, tree island burnout, conversion to cattail under eutrophic conditions, and takeover by exotic plant species.

Another key management recommendation is to increase water levels in over-drained areas. This recommendation is supported by several of the studies described above. For example, in over-drained areas such as in WCA 3B, the tree island monitoring data suggests these communities are not in equilibrium with the current hydrologic conditions (***Figure 8-47***). The known hydrologic tolerances of the Everglades tree species suggest that the hydroperiods on many of these islands could be increased substantially while still maintaining conditions within the known tolerances of the dominant species. Similarly, the ridge and slough monitoring shows widespread expansion of sawgrass into sloughs in over drained regions. This drying has lead to the oxidation of ridge soils and a loss of ridge and slough microtopography.

A third key management recommendation is to eliminate or minimize nutrient inputs into the system. The vegetation mapping program shows extensive cattail expansion into WCA 1 and WCA 2 (***Figure 8-64*** and ***Figure 8-65***). Similarly, and for the first time since the surveys began in the 1990s, cattail has been found in the northern reaches of Taylor Slough. This is attributed to nutrients entering into the system from a series of detention basins and canals to the north. With the on-going expansion of cattail into the once oligotrophic regions, it is vital nutrient inputs are monitored regularly throughout the system. This monitoring must take into account local-scale variability in soil nutrient content in order to develop more accurate assessments of changes in wetland trophic status.

8.6 LIGOTROPHIC NUTRIENT STATUS

8.6.1 Introduction and Background

The pre-drainage Greater Everglades Wetlands ecosystem was characterized by hydrologic inputs primarily from direct rainfall and by extended hydroperiods, which generally resulted in an oligotrophic, P-limited ecosystem. Increased P loads and concentrations in surface water runoff, along with replacement of sheet flow with canal flow and point source discharges, has shifted portions of the ecosystem from oligotrophic to eutrophic states (as depicted in the CEM ***Figure 8-2*** in ***Section 8.2 Sheet Flow and Water Depth Patterns Hypothesis Cluster***).

The *CERP Monitoring and Assessment Plan: Part 1 Monitoring and Supporting Research* (RECOVER, 2004) delineated monitoring initiatives for two major attributes important to ecosystem functioning that are affected by overall nutrient quality and loading rates: 1) interior gradients of surface water quality; and 2) regional distribution of soil nutrients. The follow-up document, *Monitoring and Assessment Plan, Part 2 2006 Assessment Strategy for the MAP* (RECOVER, 2006) advanced the monitoring framework presented in MAP Part 1, adding periphyton mat as an indicator of total system water quality (Refer to ***Section 9.2.3 Integrated Hydrology and Water Quality in RECOVER 2006***).

The draft *Revised CERP Monitoring and Assessment Plan: Part 1 Monitoring and Supporting Research* (RECOVER 2009) further restructures the Greater Everglades Wetlands Module MAP based on the Total System Conceptual Ecological Model (Ogden et al., 2005), further utilizing and integrating all available data, regardless of funding agency, thus maximizing RECOVER monitoring dollars. To fully assess the oligotrophic status of the Greater Everglades Wetlands ecosystem, the revised MAP Part 1 capitalizes on water quality data collected through other initiatives (i.e., USEPA REMAP, SFWMD surface water quality permit and research, USGS Priority Ecosystems Science program, and National Science Foundation Long-term Ecological Research [LTER]) while pursuing the use of periphyton mat structure and community composition as an early responder used to detect system-wide alterations in both hydrology and water quality. However, as currently sampled, the periphyton monitoring network alone is inadequate to fully capture the original intent of characterizing the nutrient status of the system originally outlined by the MAP. The 2009 revisions to the MAP focuses on placing future emphasis on investigating ways to better integrate the soil, periphyton, surface water nutrient concentration, and vegetation data to determine the nutrient status of the system.

During the MAP Part 1 revision process, the CEM for oligotrophic nutrient status and sheet flow was revised, as depicted in ***Figure 8-2***, presented earlier in this module in ***Section 8.2 Sheet Flow and Water Depth Patterns Hypothesis Cluster***. The CEM depicts the current working hypotheses for oligotrophic nutrients in the Greater Everglades Wetlands ecosystem. Nutrient state and changes in hydroperiod and water depth pattern affect periphyton mat biomass, composition and structure. Nutrient enrichment causes an elevation in periphyton nutrient content, a reduction in the proportion of calcareous floating and epiphytic periphyton mats, and a replacement of native species by non-mat forming filamentous species. In addition, P and N concentrations in soil and flocculent organic matter reflect patterns and trends in surface water concentrations when integrated over time scales of months to years. Restoration of hydrology toward a pre-drainage state would maintain or reduce nutrient (i.e., P and N) loads from inflow structures, resulting in surface water nutrient concentrations in the open marsh that do not expand zones of eutrophication in the Greater Everglades Wetlands ecosystem.

The following performance measures, which are currently being discussed and may be revised, pertain to oligotrophic nutrients in the Greater Everglades Wetlands:

- Greater Everglades Wetlands TP Concentrations in Surface Water (www.evergladesplan.org/pm/recover/recover_docs/ret/030807_ge_tp_conc_sw.pdf)

- Greater Everglades Wetlands Basin-Wide TP Loading and Flow-Weighted Mean Concentration Inflows (www.evergladesplan.org/pm/recover/recover_docs/ret/030807_ ge_tp_loading.pdf)

- Greater Everglades Wetlands Nutrient TN Concentrations in Surface Water (www.evergladesplan.org/pm/recover/recover_docs/ret/030807_ge_tn_conc_sw.pdf)

- TN Loads/Flow-Weighted Mean Concentration in Inflows to the Greater Everglades Wetlands (www.evergladesplan.org/pm/recover/recover_docs/ret/030807_ ge_tn_loads_flow.pdf

- TP Concentrations in Soil (www.evergladesplan.org/pm/recover/ recover_docs/ret/030807_ge_tp_conc_soil.pdf)

- Greater Everglades Wetlands Sulfate Concentrations in Surface Water (www.evergladesplan.org/pm/recover/ recover_docs/et/ge_10.pdf)

- Greater Everglades Wetlands Conductivity in Surface Water (www.evergladesplan.org/ pm/recover/recover_docs/et/ge_11.pdf)

- Wetland Trophic Relationships - Periphyton (www.evergladesplan.org/pm/recover/ recover_docs/ret/pm_ge_periphyton.pdf)

In general, the study area for the monitoring described below extends through the remaining freshwater portion of the Greater Everglades Wetlands located within the Loxahatchee NWR, WCAs and Everglades National Park (refer to ***FIGURE 8-1*** At the beginning of this module).

8.6.2 Monitoring and Results

8.6.2.1 Periphyton

8.6.2.1.1 Monitoring

Periphyton biomass, nutrient concentrations, and species composition were measured within the freshwater portion of the Greater Everglades Wetlands and were collected in conjunction with the aquatic fauna monitoring that supports assessment of the Wading Bird Nesting in Relation to the Aquatic Fauna Forage Base Hypothesis Cluster (see ***Section 8.3*** of this report). The sampling design follows guidelines described by Philippi (2003, 2005) and is based on a spatially-balanced generalized recursive tessellation (Stevens and Olsen, 2004). Note that, by definition, this statistically-based design was not intended to capture near-field or interior transects of water quality gradients, although an individual sampling location may end up geographically close to a water inflow point.

Periphyton biomass (dry mass, ash free dry mass, chlorophyll *a*), TP and species composition were determined (Gaiser et al., 2004b). Periphyton collection was completed for 2008 but analytical results documenting the relative abundance of different periphyton species were not completed in time for inclusion in this report.

8.6.2.1.2 Results

Analysis of periphyton metrics of abundance, including dry biomass (g/m^2), wet biovolume (mL/m^2), and ash free dry weight, demonstrate a clear, consistent north to south gradient in all sampled years (2005, 2006 and 2007 are reported here) (***Figure 8-71, Figure 8-72*** and ***Figure 8-73***), while measures of periphyton TP content (micrograms per gram [µg/g]) and percent organic content (100-percent mineral content) decreased along this same gradient (***Figure 8-74*** and ***Figure 8-75***). Both of these large scale, geographic gradient trends are consistent with the hypothesis that there is a large scale, regional trend of more nutrients in the system in the north, and fewer nutrients in the south.

In general, P values were highest in the flocculent periphyton of Pal Mar and the Loxahatchee NWR (also referred to as WCA 1), although effects and patterns of P enrichment are evident in all the WCAs especially around the inflow water control structures. Patterns of periphyton TP in the WCAs generally correspond to patterns of eutrophication as described by Gaiser et al. (2006). The majority of the landscape south of Tamiami Trail had TP concentrations less than 200 µg/g, with the notable exception being elevated periphyton TP in southern Everglades National Park above the marsh-mangrove ecotone of the Gulf of Mexico drainages.

While temporal trends are difficult to discern with only three years of data, and need to be assessed within the context of the overall hydrologic patterns experienced by the system, it appears that overall periphyton TP concentration in 2007 decreased across the system relative to the two previous years, except within the Loxahatchee NWR.

The research conducted by Trexler and Gaiser (2009) found strong spatial patterns in algal species among the primary sampling units and the data were used to determine TP optima and tolerances for the most common taxa. The resultant models indicate that periphyton P content explains about half the variability in periphyton species distribution.

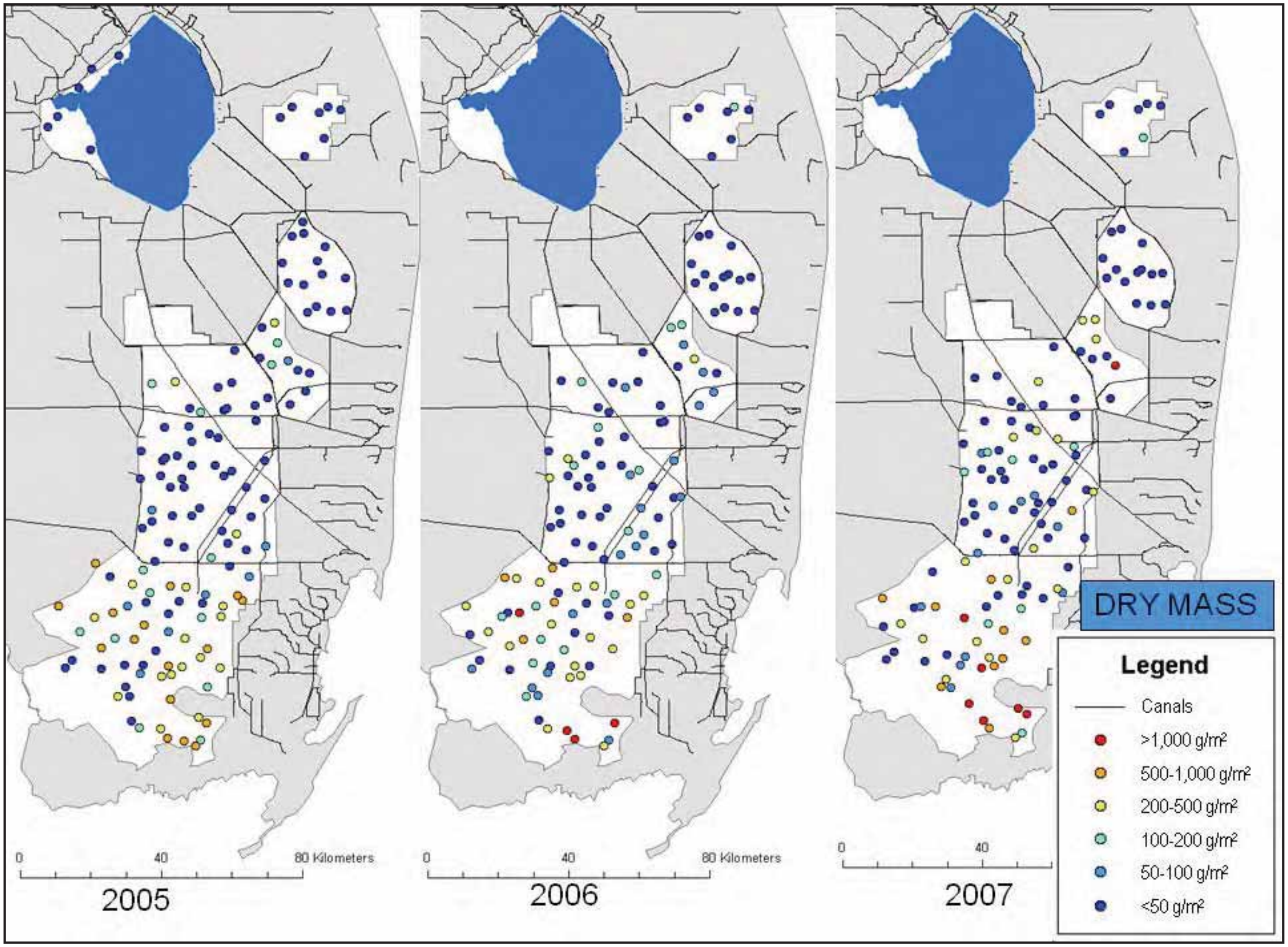


FIGURE 8-71. PATTERN OF DISTRIBUTION OF PERIPHYTON ABUNDANCE EXPRESSED AS DRY BIOMASS ACROSS THE GREATER EVERGLADES WETLANDS ECOSYSTEM IN FALL 2005, 2006 AND 2007

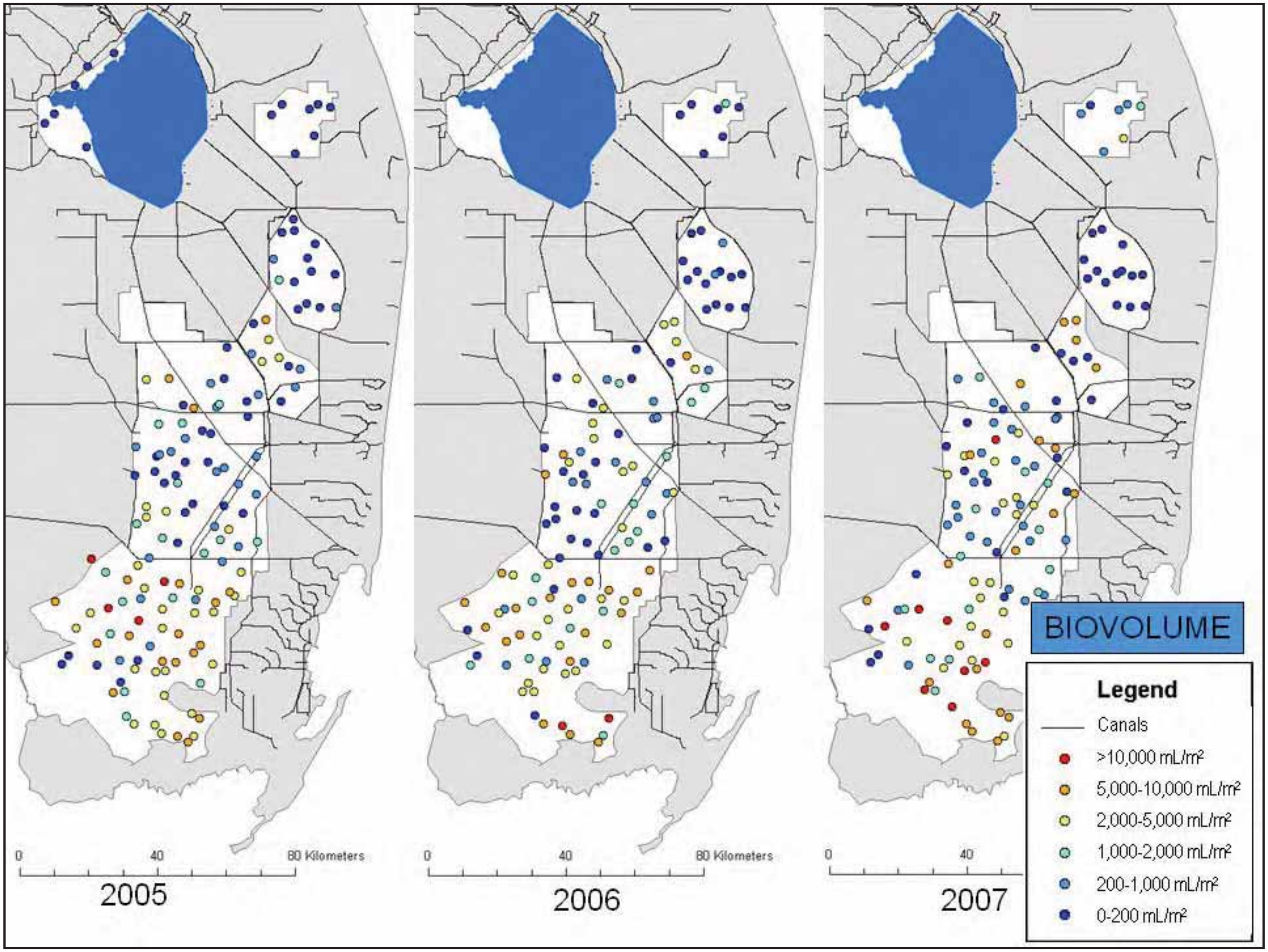


FIGURE 8-72. PATTERN OF DISTRIBUTION OF PERIPHYTON ABUNDANCE EXPRESSED AS WET BIOVOLUME ACROSS THE GREATER EVERGLADES WETLANDS ECOSYSTEM IN FALL 2005, 2006 AND 2007

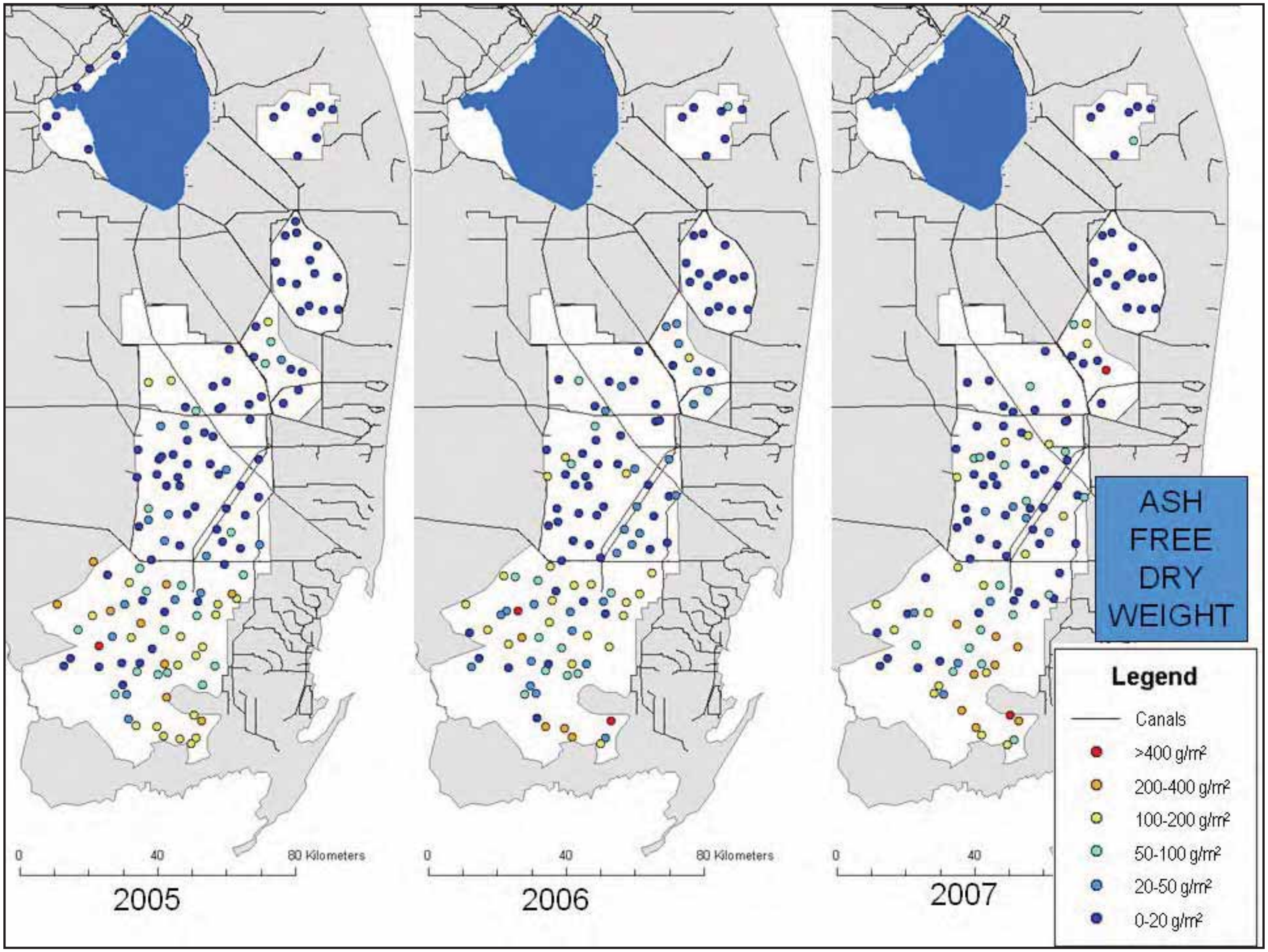


FIGURE 8-73. PATTERN OF DISTRIBUTION OF PERIPHYTON ABUNDANCE EXPRESSED AS ASH FREE DRY WEIGHT ACROSS THE GREATER EVERGLADES WETLANDS ECOSYSTEM IN FALL 2005, 2006 AND 2007

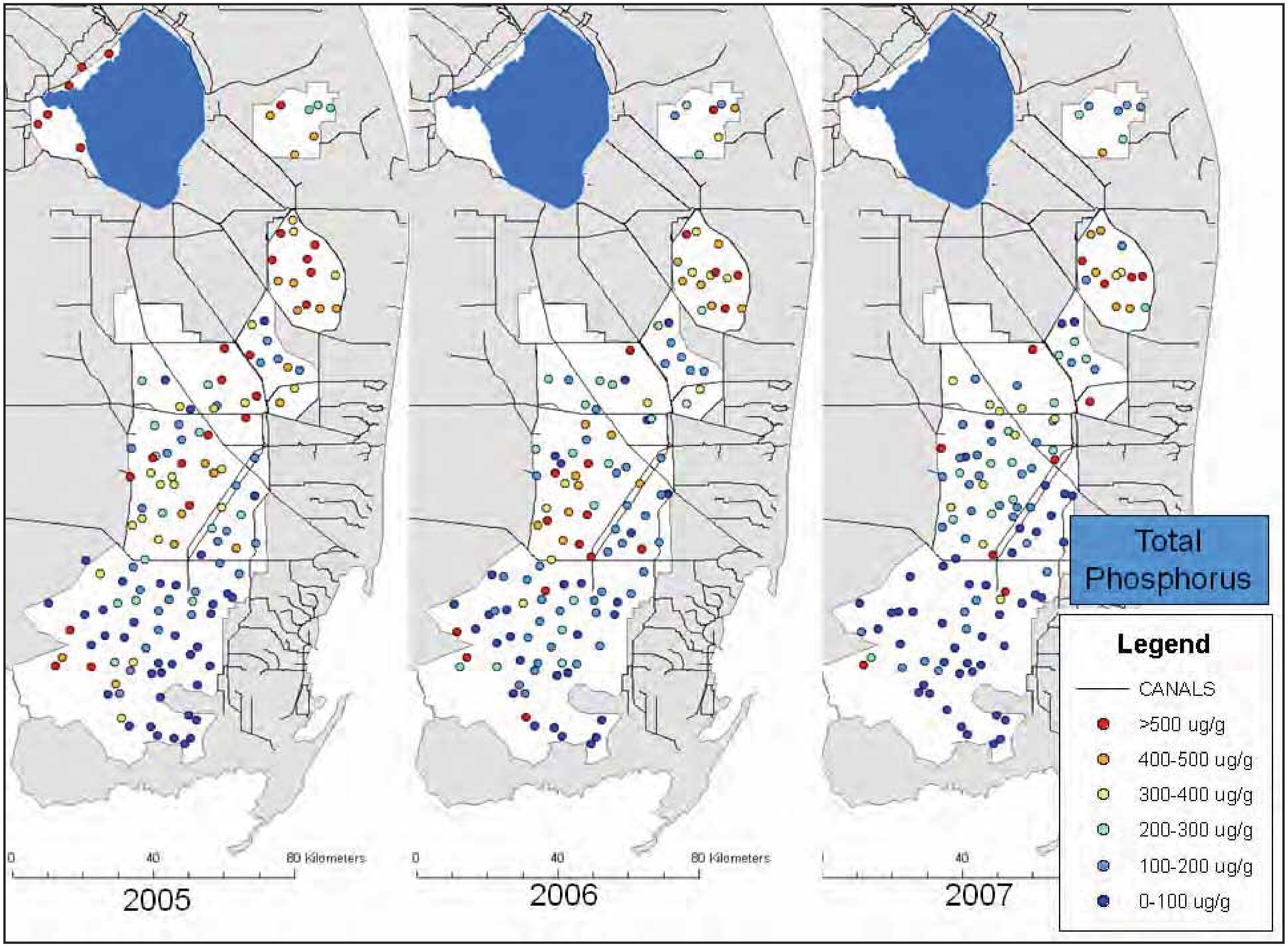


FIGURE 8-74. PATTERN OF DISTRIBUTION OF PERIPHYTON QUALITY EXPRESSED AS TOTAL PHOSPHORUS CONCENTRATION ACROSS THE GREATER EVERGLADES WETLANDS ECOSYSTEM IN FALL 2005, 2006 AND 2007

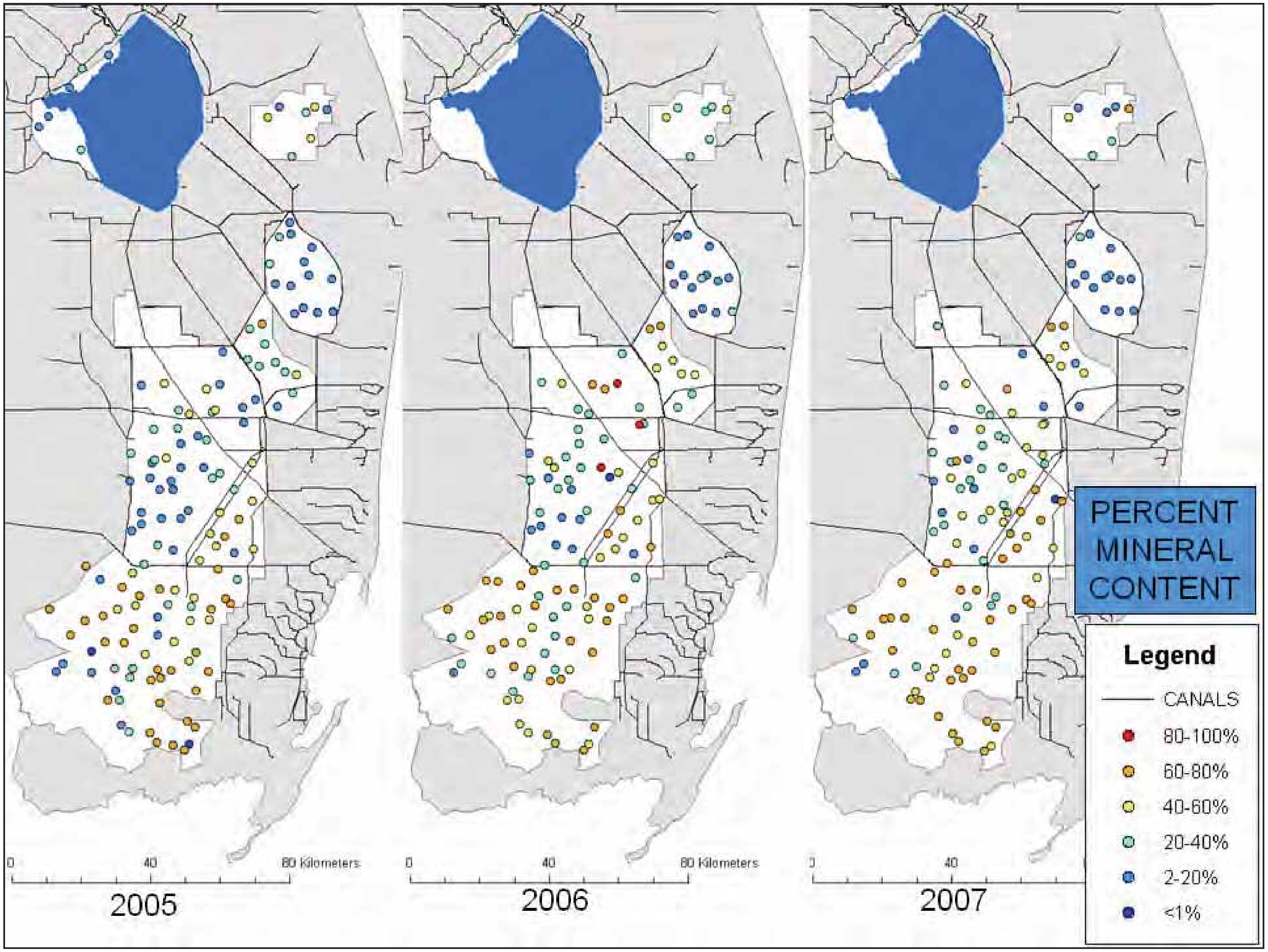


FIGURE 8-75. PATTERN OF DISTRIBUTION OF PERIPHYTON QUALITY EXPRESSED AS PERCENT MINERAL CONTENT ACROSS THE GREATER EVERGLADES WETLANDS ECOSYSTEM IN FALL 2005, 2006 AND 2007

8.6.2.2 Interior Water Nutrient Concentrations

8.6.2.2.1 Monitoring

USEPA REMAP (most recent data from 2005) and SFWMD permit-related monitoring are two main sources that offer extensive temporal and spatial surface water quality data. RECOVER proposes to utilize these datasets in future, more in-depth assessments of the nutrient status of the Greater Everglades Wetlands ecosystem (RECOVER, 2009). This report briefly summarized the system-wide findings previously reported by both these efforts.

The REMAP nutrient data is collected in support of a long-term research, monitoring and assessment effort that provides critical, timely, scientific information needed for management and restoration decisions. The REMAP program utilizes a statistical, probability-based sampling strategy to select sites for sampling (Scheidt and Kalla, 2007). The program area extends from Lake Okeechobee southward to the mangrove fringe on Florida Bay and from the ridge along the urbanized eastern coast westward into Big Cypress National Preserve. The data summarized in this report were primarily collected as part of the REMAP 2005 Phase III biogeochemical sampling. Three REMAP data collection efforts have occurred since 1993, but no current resource plans are funding additional REMAP sampling.

The SFWMD surface water nutrient data is collected to support annual water quality assessment of the Everglades Protection Area (Figure 3A-1 in Payne et al., 2009) as required under the Everglades Forever Act permit. The goal is to provide a synoptic view of water quality conditions on a regional scale, including the Loxahatchee NWR, WCAs 2 and 3, and Everglades National Park (Payne et al., 2009). The water quality evaluation utilized a network of water quality monitoring stations selected from existing long-term SFWMD monitoring projects. These stations were carefully selected to be representative of either the Everglades Protection Area boundary conditions (i.e., inflow or outflow) or ambient marsh conditions (interior). The SFWMD reports annual geometric means for water years (May 1 through April 30) in the South Florida Ecosystem Reports (my.sfwmd.gov/portal/page/portal/pg_grp_sfwmd_sfer/pg_sfwmd_sfer_home). For consistency with the MAP data presented throughout the Greater Everglades Wetlands Module, this report summarized the findings from the 2006 through 2009 South Florida Environmental Reports (Weaver and Payne, 2006; Weaver et al., 2007; 2008, Payne et al., 2009).

8.6.2.2.2 Results

Using the REMAP data, Scheidt and Kalla (2007) report that, in November 2005, 72.8 ± 7.5 percent of the entire Greater Everglades Wetlands (excluding canals; see Figure 14 in Scheidt and Kalla, 2007) had a TP concentration less than or equal to ten µg/L, which is a large improvement over 1995 values, where only 42.2 ± 7.8 percent of the ecosystem had measured TP concentration values less than or equal to 10 µg/L. However, fixed interior stations measured in 2006 indicate that mean values for the Loxahatchee NWR and WCA 2A TP concentrations values exceed 10 µg/L, with values of 15 and 18 µg/L, respectively. Additionally, surface water TP concentrations during 2005 (Figure 43 in Scheidt and Kalla, 2007) exhibited a general north to south decreasing gradient.

SFWMD data indicates a general north to south decreasing TP gradient (***Table 8-8*** and ***Table 8-9***) for both inflow and interior areas of all regions reported (Loxahatchee NWR, WCA 2, WCA 3, and Everglades National Park) for WY 2005 through 2008. In all regions, the interior area geometric means were substantially lower than the inflow region geometric means. The Loxahatchee NWR and WCA 2A inflow annual geometric means have decreased every year since 2005, while WCA 3A and Everglades National Park TP inflow concentrations have remained essentially unchanged during this period. However, the interior TP concentrations exhibit persistent intra-annual variability but no discernable temporal trend over this same period.

TABLE 8-8. SUMMARY OF ANNUAL GEOMETRIC MEAN TOTAL PHOSPHORUS SURFACE WATER CONCENTRATIONS (AND STANDARD DEVIATIONS) FOR INFLOWS INTO THE EVERGLADES PROTECTION AREA REGIONS FOR WATER YEARS 2005 THROUGH 2008

Region	Annual Geometric Mean TP Surface Water Concentrations (µg/L)			
	Water Year 2005	Water Year 2006	Water Year 2007	Water Year 2008
Loxahatchee NWR	68.6 (1.9)	79.7 (1.6)	57.6 (1.7)	46.9 (2.2)
WCA 2	26.6 (2.3)	26.7 (1.8)	25.4 (1.8)	19.7 (1.5)
WCA 3	23.9 (1.9)	24.3 (2.1)	23.8 (1.9)	20.1 (1.7)
Everglades National Park	10.3 (2.2)	8.2 (1.6)	11.1 (1.9)	11.2 (1.9)

Note: see text for data sources

TABLE 8-9. SUMMARY OF ANNUAL GEOMETRIC MEAN TOTAL PHOSPHORUS SURFACE WATER CONCENTRATIONS (AND STANDARD DEVIATIONS) FOR INTERIOR SITES WITHIN EVERGLADES PROTECTION AREA REGIONS FOR WATER YEARS 2005 THROUGH 2008

Region	Annual Geometric Mean TP Surface Water Concentrations (µg/L)			
	Water Year 2005	Water Year 2006	Water Year 2007	Water Year 2008
Loxahatchee NWR	12.8 (1.9)*	9.8 (1.8)	9.3 (1.9)	10.6 (1.9)
WCA 2	17.9 (2.9)	13.5 (2.3)	13.3 (2.6)	12.2 (2.1)
WCA 3	9.6 (2.4)	9.1 (2.4)	9.7 (2.4)	8.5 (1.8)
Everglades National Park	5.1 (2.4)	5.7 (2.1)	6.3 (2.0)	5.2 (1.7)

* The 2006 South Florida Environmental Report (Weaver and Payne 2006) reported the 2005 Loxahatchee NWR geometric mean as 14.6 ± 2.1

Note: see text for data sources

Overall, neither the most recent REMAP data (2005) nor SFWMD regulatory compliance water quality monitoring provides the temporal or spatial resolution that are needed to fully understand ecosystem responses resulting from CERP restoration efforts, especially as site- or date-specific water quality data are often required to examine ecosystem responses to a particular change.

8.6.2.3 Soil Nutrient Concentration

8.6.2.3.1 Monitoring

A stratified random sampling design was used to produce maps of freshwater soil nutrient values across the Everglades (Stober et al., 2001; Sklar et al., 2006; Scheidt and Kalla, 2007; Bruland, 2007). The methods and resulting soil nutrient distribution maps from the 2003 sampling effort were reported as the pre-CERP condition in the 2006 SSR (RECOVER, 2007a) and Chapter 6 of the 2006 South Florida Ecosystem Report (Sklar et al., 2006). These data combined with the 2005 REMAP soil TP data reported by Scheidt and Kalla (2007) comprise the most recent ESM project results.

It was originally estimated that this regional soil core collection would be repeated at approximately five-year intervals (RECOVER, 2004); however, at the time of this reporting, there is no active funding for a REMAP sample collection in 2010, which will be five years after the most recent REMAP soil sampling effort was undertaken. The Greater Everglades Wetlands Module has recently completed a study of this soil data to assess the spatial variance associated with this data to determine if the sample number is adequate to detect regional changes in soil P concentration (Cohen et al., 2009). The detailed results of this study will be used to determine the temporal and spatial scope of future soil cores collected for nutrient analysis.

8.6.2.3.2 Results

In general, P enrichment of the peat soils was most evident in the northern portion of the system, with the highest concentrations associated with discharge points or proximate to canals (refer to Figure 45 in Scheidt and Kalla, 2007 and Figure 4-25 in Weaver and Payne, 2006). Specifically, WCA 3A north of Alligator Alley, northern WCA 2A and the edges of the Loxahatchee NWR have elevated soil TP values relative to other peat-based areas of the system.

Cohen et al. (2009) found that while it is likely that processes controlling the variability at both fine and coarse scales are principally the same, substantial variability can exist at small scales influencing interpretation of data sets designed to detect large scale changes over time. While finer resolution maps generate information relevant for understanding localized soil nutrient changes (e.g, related to surface water inflows), the time series of coarse, landscape-scale mapping efforts (e.g., REMAP) are valuable for assessing large spatial/temporal scale changes in soil nutrient conditions.

8.6.3 Summary

8.6.3.1 Assessment of Working Hypotheses and Status and Trends

The overall synthesis from the various reports for the three markers of ecosystem nutrient status (soil, water and periphyton) show similar trends and findings: 1) a north to south declining nutrient gradient exists; 2) eutrophication is especially evident at discharge points and near canals; and 3) the Loxahatchee NWR and WCAs are the most impacted.

Research has shown that assessment of water column nutrient concentrations alone does not generally capture the level of ecosystem impact, restorative or deconstructive, that results from changes in nutrient loads over a period of time (McCormick and Stevenson, 1998; Smith and McCormick, 2001; Gaiser et al., 2004a) because wetland water column nutrients are highly variable (Reddy et al., 1999) with excess nutrients quickly assimilated by vegetation. The long-term fate of surface water nutrient concentrations is then tied to how they are assimilated, including sequestration into the flocculent material (Koch and Reddy, 1992; Kadlec and Knight, 1995; Harwell et al., 2008). This high spatial and temporal variability makes water column nutrient concentration a poor marker for assessing immediate changes in nutrient status, which is critical for applying AM principals to the restoration effort. Still, water quality monitoring is very useful in: 1) calculating nutrient load for structures and/or basins; 2) assessing compliance of applicable water quality standards; 3) calculating ecosystem nutrient budgets; and 4) defining temporal and spatial trends in nutrient impacted areas. To date, RECOVER efforts have not focused on expanding further than these four categories.

Soil nutrient concentration is a long-term indicator of nutrient loading and, in the Everglades, is often associated with ecosystem change (Craft and Richardson, 1993; Doren et al., 1996; Newman et al., 1997; McCormick et al., 1999; Scheidt and Kalla, 2007). Reddy et al. (1999) reported that sediment response to nutrient loading occurs over years (10 to 15 years) in the top ten centimeters of nutrient impacted wetlands. The USEPA REMAP (Scheidt and Kalla, 2007) and the ESM project undertook a systematic, probabilistic soil mapping effort of the Everglades ecosystem. Combined, these efforts provide the first comprehensive baseline soil map for the ecosystem. This soil map is expected to serve as a benchmark of soil conditions in the Everglades as well as to provide a complimentary dataset to be used to correlate changes in vegetation and landscape pattern to the effects of restoration. Several studies have utilized the various soil maps to quantify spatial autocorrelation (Newman et al., 1997; DeBusk et al., 2001) and detect changes over time (Bruland et al., 2007), but the issue of short-range variability and spatial variability were not explicitly addressed (Grunwald et al., 2004; Cohen et al., 2009). Cohen et al (2009) found that while it is likely processes controlling the variability at both fine and coarse scales are principally the same, substantial variability exists over small scales, which can generate uncertainty about inferring change across different datasets.

Research has shown that periphyton can provide a rapid (weeks to months) and accurate indication of water quality changes (McCormick et al., 1996; Reddy et al., 1999; Gaiser et al., 2004a, 2005; Gaiser, 2006) making periphyton an excellent first response, signal of both restorative and deconstructive ecological responses to nutrient and hydrological changes if periphyton is monitored in the appropriate locations. Gaiser (2009) developed a periphyton-based tool that effectively indicates landscape-scale ecosystem changes relative to a natural un-impacted system. Currently, predictions on smaller spatial scales are not yet possible because of the unique natural qualities periphyton exhibits along the ecosystem gradient.

The *Monitoring and Assessment Plan, Part 2* 2006 Assessment Strategy for the MAP (RECOVER, 2006) began moving toward a more integrated assessment of nutrient status that could be used to support AM, but did not go far enough to bring about a holistic, integrated assessment resulting in three separate assessments of nutrient trends. The revised MAP Part 1 (RECOVER, 2009) advances the integration of all three data layers in an effort to provide total predictive and assessment measures to assess on both short and long spatial and temporal scales. However, it recognizes that the CERP Monitoring and Assessment Plan: Part 1 Monitoring and Supporting Research (RECOVER, 2004) vision of characterizing water quality gradients especially relevant to project-level monitoring have not been fully addressed. Future efforts will move these integration efforts forward, updating the performance measures to better reflect the holistic assessment of oligotrophic nutrient status.

8.6.3.1.1 Key Management Recommendations

A major concern in the implementation of CERP is that additional water that is provided for restoration of sheet flow and water depth patterns in the Everglades needs to be of sufficient quality that it does not further degrade the system that we are trying to restore.

The capacity of the STAs and CERP water storage facilities to sufficiently reduce TP concentrations in the immense volumes of water needed for Everglades restoration represents a key management uncertainty.

This report identifies a core area of southern Everglades that retains the greatest potential for restoration of defining characteristics of the ecosystem, which includes portions of WCA3, eastern Big Cypress and Everglades National Park. Loxahatchee NWR represents another area where the potential for restoration is high, because it appears to remain functional in many respects under its current management regime.

It is particularly critical that TP concentrations in restoration water drop to 10 ppb or less before reaching these areas.

RECOVER should consider establishing a standing nutrient status hypothesis cluster team to further enhance our ability to understand system-wide nutrient status as called for in the 2009 revised MAP.

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Monitoring Early and Late Dry Season Aquatic Fauna and Periphyton

Prepared by Joel Trexler and Evelyn Gaiser, Principal Investigators, towards validation of the Wading Bird Nesting in Relation to Aquatic Fauna Forage Base Hypothesis Cluster

Introduction

The conceptual model for wading bird nesting in relation to the aquatic fauna forage base (***Figure 8-9*** in ***Section 8.2*** of Greater Everglades) links physical drivers of hydrology and nutrients to wading bird nesting success by way of the aquatic food web of Everglades marshes (RECOVER 2004, Ogden et al. 2005). Here, we report data that support the lower trophic level linkages of the causal network, emphasizing those provided by the Monitoring and Assessment Plan (MAP) program. The ecological detail of the trophic hypothesis cluster increases as it describes the linkage of aquatic consumer communities to the specific prey patches or concentrated prey pools, where wading birds forage. Several rather general hypotheses state that periphyton production is primarily controlled by two drivers, hydrology and nutrients, and production, in turn, controls secondary productivity of wading bird prey species, which is ultimately selectively filtered into drying pools where wading birds can catch their food. Our report is divided into two sections, one addressing periphyton communities in general and documenting their linkage to hydrology and nutrients based on our monitoring data. This section is followed by a section that 1) focuses on the relative abundance of algae generally considered most palatable (green algae and diatoms), 2) identifies linkages of the two focal environmental drivers to the relative abundance of these edible algae, and 3) establishes correlative relationships between the relative abundance of edible algae and aquatic consumers.

Hydrology and Phosphorus Gradients are Correlated with Periphyton Communities

The utility of periphyton in exposing ecological ramifications to restorative or deconstructive change in the Everglades is due its bearing several of the most desirable features of a reliable ecological indicator including 1) being distributed throughout the system of study, 2) having rapid response to environmental change that is 3) readily quantifiable at several levels of biological organization (individual, species, population and community) with 4) consequences to levels above and below its placement on the food web (Karr 1999). Reliance on periphyton to indicate environmental change has been well justified by scientific research conducted in the Everglades (McCormick and Stevenson 1998, Gaiser et al. 2006, In Press), which adds regional applicability to the existing body of literature in aquatic sciences that has supported the widespread employment of periphyton monitoring in aquatic ecosystem management (Hill et al. 2000, Stevenson 2001). Specifically, we anticipate that patterns of periphyton production, nutrient content and composition among and within the primary sampling units sampled in this mapping assessment will provide reliable indication of changes driven by hydrology and nutrient enrichment. Alterations in periphyton attributes then cascade through the system to affect higher organisms through changes in food quality, composition and concentration of gasses and nutrients in the water column and ecosystem structure (i.e., soil formation and quality, physical habitat structure). The following hypotheses were formulated using data from descriptive and experimental studies. We list the hypotheses and follow each with a discussion of supporting data and progress in application to this project.

Effects of Nutrient Enrichment

Hypothesis

Nutrient enrichment causes an elevation in periphyton nutrient content, a reduction in the proportion of calcareous floating and epiphytic periphyton mats, and a replacement of native species by non-mat forming filamentous species.

Supporting Data

Throughout the system, periphyton has been proven to provide rapid and accurate indication of water quality changes; periphyton responses were critical to establishing the phosphorus (P) criterion for freshwater sloughs (McCormick et al. 1996, Gaiser et al. 2004), have been used to indicate rates of coastal saltwater encroachment in mangroves (Ross et al. 2001, Gaiser et al. 2004) and for detection of nutrient enrichment in adjacent offshore seagrass beds (Frankovich et al. 2006). Several studies have shown that periphyton not only respond to but also regulate water quality (Thomas et al. 2006b, Gaiser et al. 2006) by quickly and efficiently removing excess P from the water column. Gaiser et al. (2005) recommends using periphyton P content as a metric of P enrichment history, rather than water or soil P, because it has been shown repeatedly to provide a much more reliable indication of P load history. This has been adopted in most large-scale monitoring programs in the Everglades (i.e., this study, the U.S. Environmental Protection Agency Regional Environmental Monitoring and Assessment Program [REMAP] assessment, National Science Foundation Long-term Ecological Research [LTER] research).

Progress

We found very strong and temporally consistent spatial patterns in periphyton biomass that showed a general increase from northern to southern primary sampling units (***Figure 1***). Periphyton biomass, measured by biovolume, cover, and dry and ash-free dry mass all increased to the south, becoming highest in Shark River Slough, the southern Marl Prairies and Taylor Slough (***Figure 1***). Patterns of biomass distribution measured by ash-free dry mass were plotted across primary sampling units for fall 2005 and 2006 collections (***Figure 2***). In this plot, the degree of shading indicates biomass quantity while the color of the collection site indicates the degree of departure from expected background values, defined for each landscape sampling unit according to Gaiser et al. (2009). Values exceeding one standard error of the expected range are coded yellow while red points indicate a departure exceeding two standard errors. While two years of data are insufficient to detect temporal trends, we do observe decreased biomass in the southern Everglades in 2006 compared to 2005, perhaps reflecting an effect of prolonged drying of this end of the system driven both by climate and water management practices. However, modeling efforts are ongoing to determine whether these trends are driven mainly by hydrological change or by altered water quality, as, in both years, areas of the Arthur R, Marshall Loxahatchee National Wildlife Refuge (Loxahatchee NWR), and Water Conservation Areas (WCAs) 2 and 3 have ash-free dry mass values much lower than expected, a probable response to a long history of P enrichment experienced in this system.

Accordingly, the pattern of periphyton quality variables (total phosphorus [TP], organic and chlorophyll *a* content) was opposite that of the quantity estimates (***Figure 1***). We plotted TP values spatially in the same manner as for ash-free dry mass, above, and found that values were highest in the flocculent periphyton of Lake Okeechobee, Pal Mar, Holeyland and Loxahatchee NWR, which is also referred to as WCA 1 (***Figure 3***). In 2005, these values showed an increase in the oligohaline zone, where P delivery from marine and groundwater is likely (Childers 2006, Price et al. 2006), although this pattern was not observed in the 2006 survey. Effects and patterns of P enrichment are clearly seen in the WCAs as well as northern parts of Shark River Slough, where values are highest around peripheral canals and downstream of water control structures. No notable seasonal patterns were observed in these values, although biomass has normally found to be highest in the wet season (Iwaniec et al. 2006). We suspect that continued sampling will reveal this weaker temporal trend.

Figure 6. Patterns in periphyton total phosphorus content, biomass, organic and chlorophyll a content among locations in the dry and wet seaons of 2005 and 2006. See Figure 1 for locality codes.

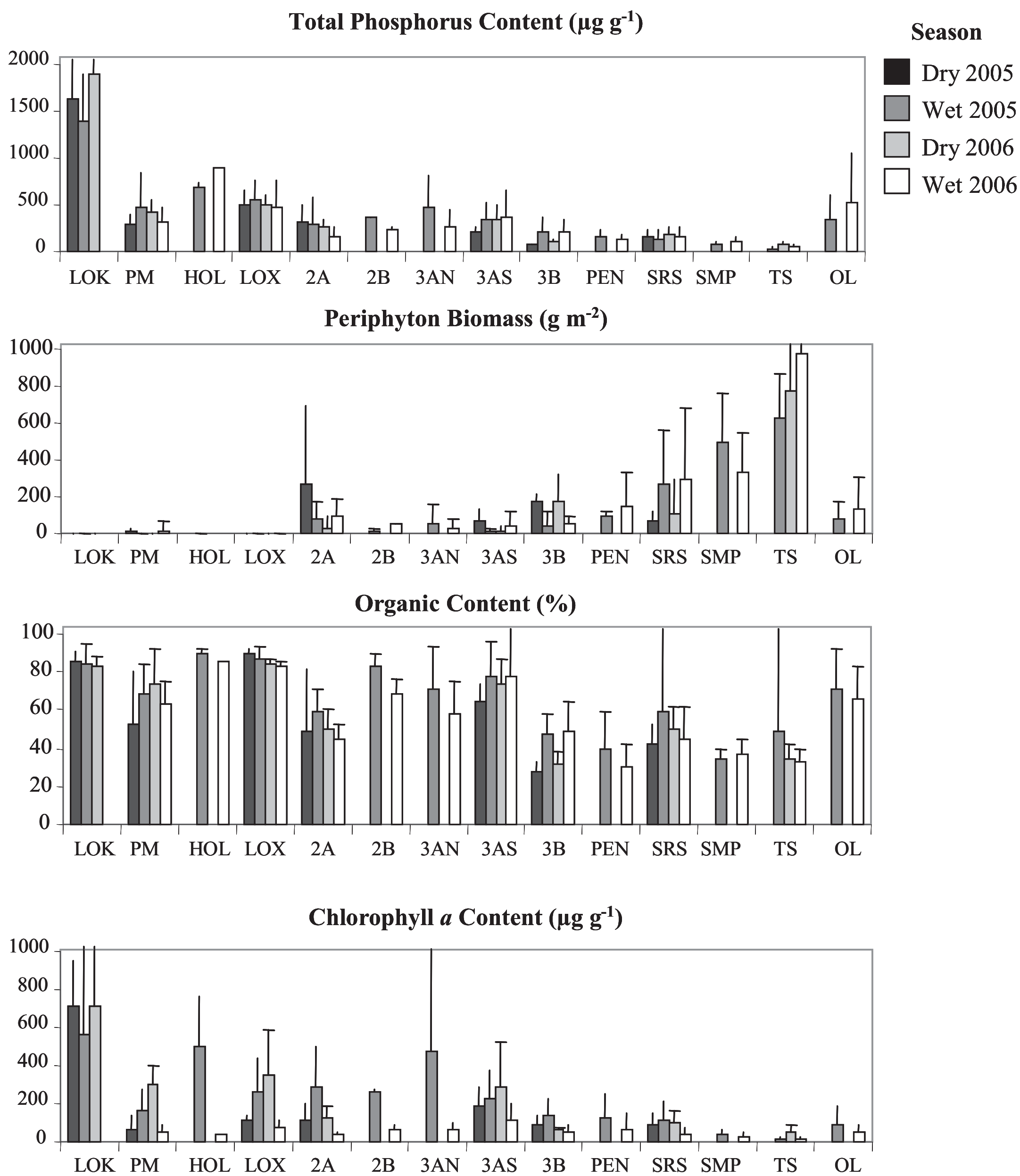


Figure 1. Patterns in periphyton TP content, biomass, organic and chlorophyll *a* content among locations in the dry and wet seasons of 2005 and 2006

µg g^{-1} = micrograms per gram, g m^{-2} = grams per square meter, % = percent)

Figure 8. Pattern of distribution of periphyton ash-free dry biomass across the Greater Everglades in fall 2005 and 2006. Sites are coded by the degree of departure from expected values (green = within 1 SE of mean in natural system, yellow = > 1 SE of natural system mean and red = >2 SE of natural system mean).

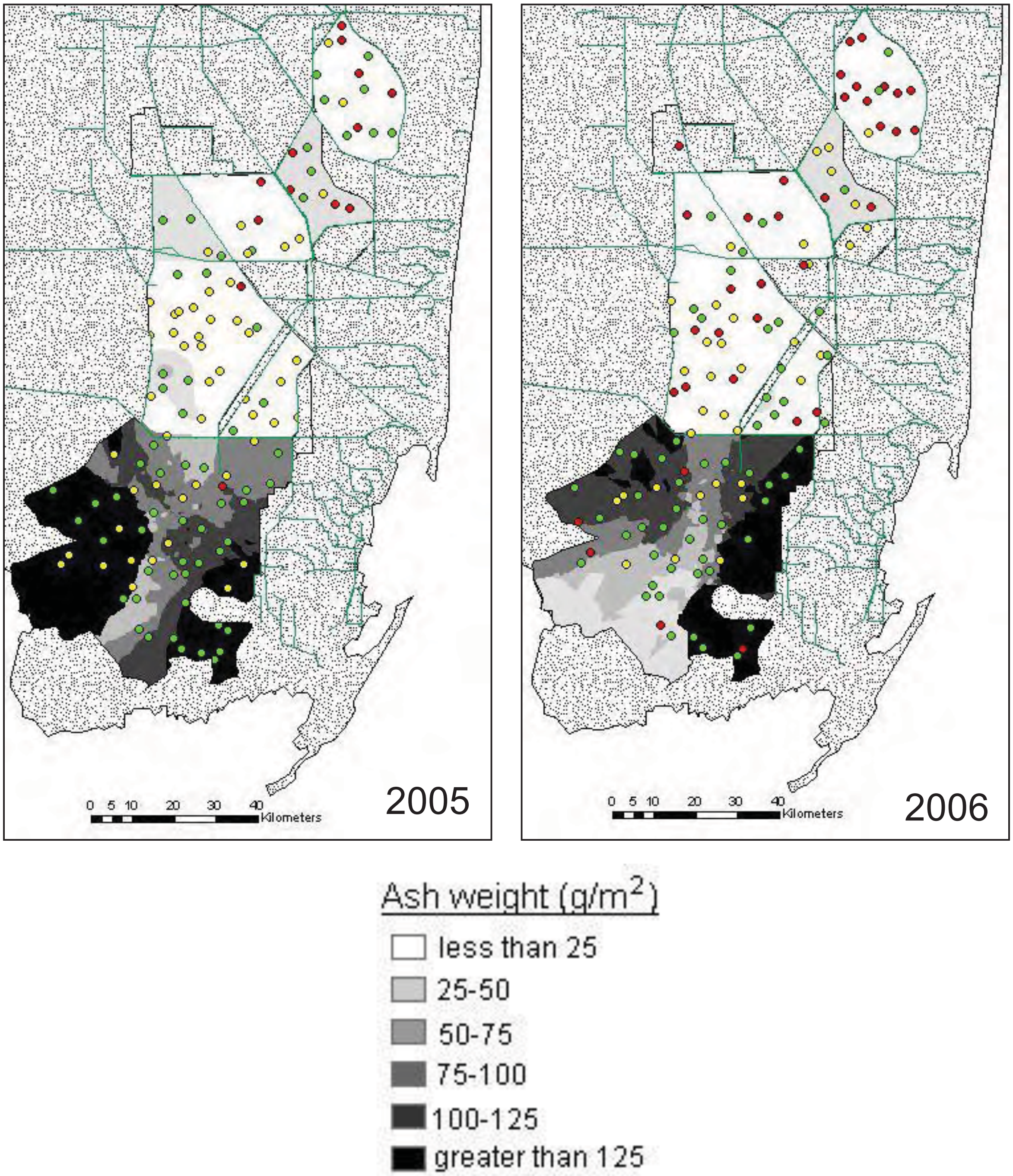


Figure 2. Pattern of distribution of periphyton ash-free dry biomass in grams per square meter (g/m^2) across the Greater Everglades in fall 2005 and 2006

Sites are coded by the degree of departure from expected values: green = within 1 standard error [SE] of mean in natural system, yellow = > 1 SE of natural system mean, and red = >2 SE of natural system mean.

Figure 9. Pattern of distribution of total phosphorus concentration in periphyton across the Greater Everglades in fall 2005 and 2006. Sites are coded by the degree of departure from expected values (green = within 1 SE of mean in natural system, yellow = > 1 SE of natural system mean and red = >2 SE of natural system mean).

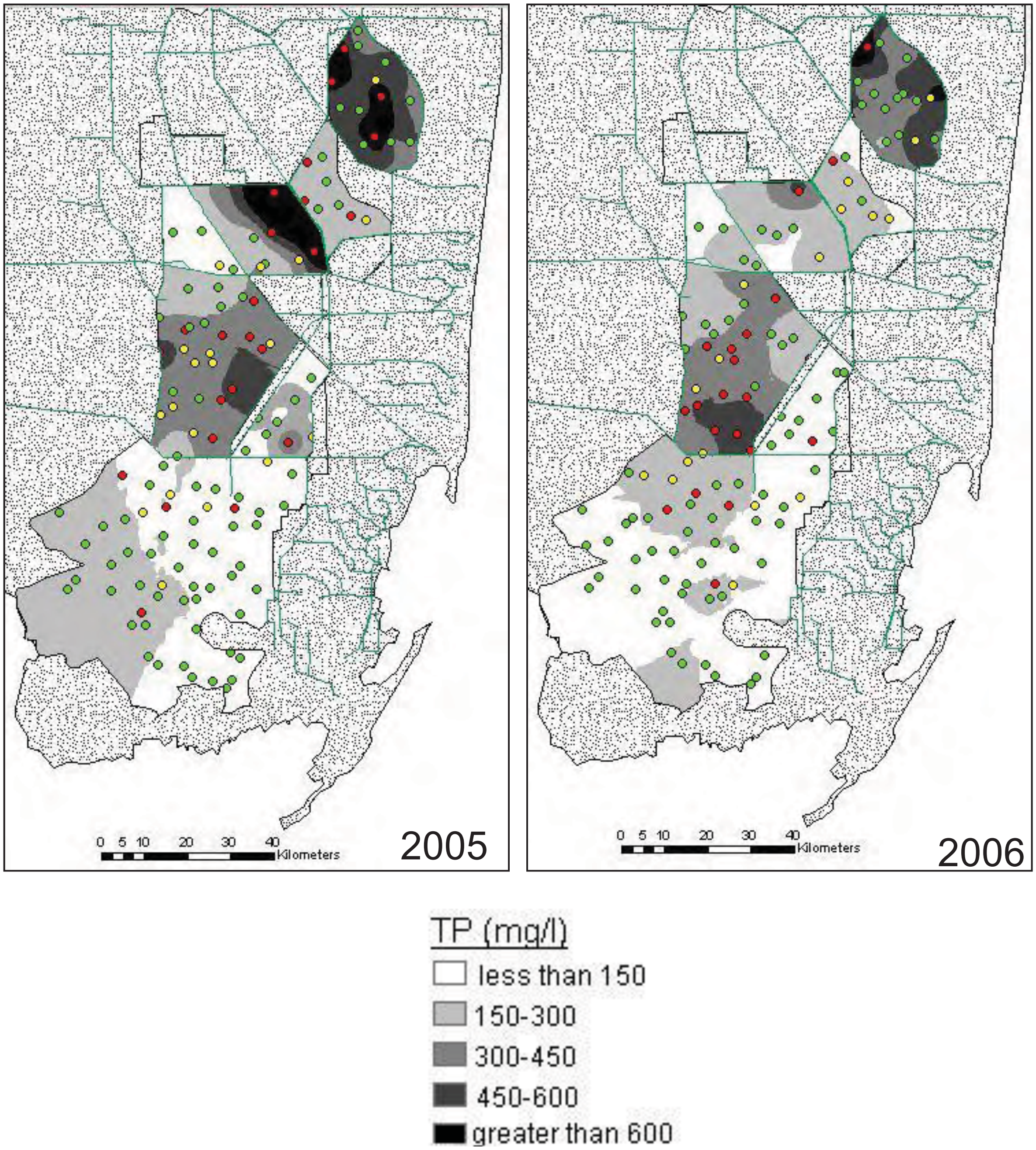


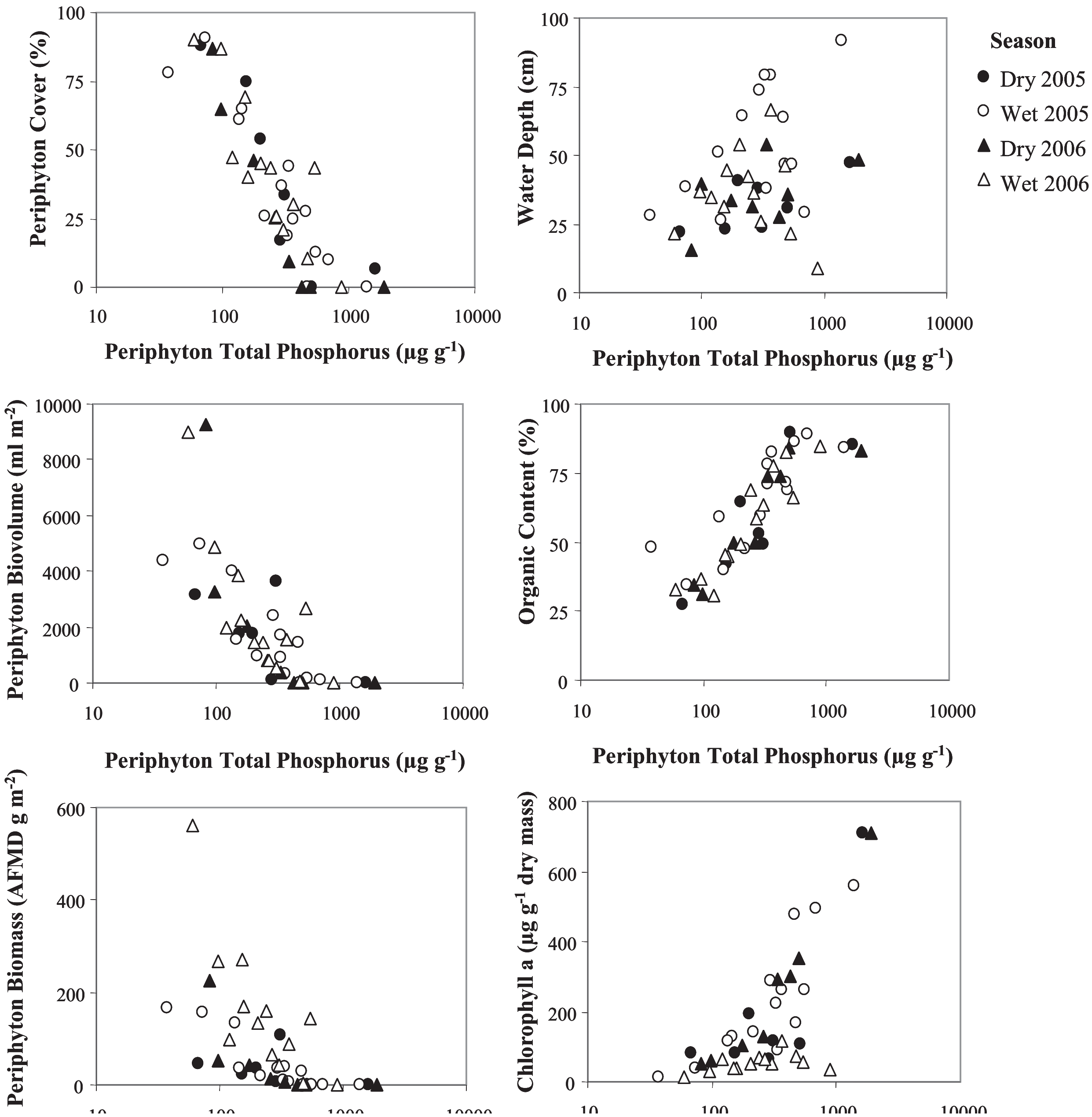
Figure 3. Pattern of distribution of TP concentration in periphyton across the Greater Everglades in fall 2005 and 2006

(See Figure 5A-2 for color coding)

The spatial patterns in periphyton attributes and TP content were highly correlated in expected ways. We found a strong and temporally persistent decrease in periphyton cover, biomass and biovolume with increasing TP content among sites (***Figure 4***). The organic content increased with TP availability, which is commonly observed as both a consequence and driver of the concomitant change in periphyton biomass (Gaiser et al. 2006). Notably, however, it is not just a

Figure 4. Relationships of periphyton TP to cover, biovolume, biomass, water depth and periphyton organic and chlorophyll *a* content averaged within primary sampling units

Figure 10. Relationships of periphyton total phosphorus to cover, biovolume, biomass, water depth and periphyton organic and chlorophyll a content averaged within PSU's during the dry and wet seasons of 2005 and 2006.



during the dry and wet seasons of 2005 and 2006

loss of the calcitic matrix that occurs with P enrichment but a loss of biomass as well (Gaiser et al. 2005), as shown here in the strong negative correlation of ash-free dry periphyton biomass with TP content. An increase in the chlorophyll *a* content of that biomass with increased P availability is expected, as P availability increases productivity of cells in species with high P requirements. We observed a strong positive association of periphyton chlorophyll *a* with P, especially in 2005 wet season and the subsequent 2006 dry season. A slightly positive correlation was observed between P availability and water depth, although this trend was only significant in the 2005 wet season.

We also found strong spatial patterns in algal species composition among the primary sampling units. A total of 229 non-diatom algae and 155 diatom taxa were found in the 2005 and 2006 surveys. The most common taxa are shown in ***Tables 1*** and ***2***. Because of a strong relationship between composition and TP availability, we were able to determine TP optima and tolerances for the most common taxa by fitting a model to their responses along the P gradient using the calibration program C2. The optima reflect the average value of TP or hydroperiod where the taxon occurred and the tolerance reflects the breadth of occurrence, centered around the average.

The values reported in ***Tables 1*** and ***2*** (Gaiser et al. 2006, 2009) can be used to predict the total periphyton P content for sites in which these taxa are found by weighting the contribution of each taxon by its relative abundance in the sample. Following this approach, we used a leave-one-out cross-validation procedure to estimate the periphyton TP content of each site and then compared that to real measured values using regression. We performed this procedure separately for the 2005 and 2006, soft algae and diatom samples, respectively. The 2005 models produced an R^2 of 0.56 and 0.46, respectively, with a root-mean-squared error of 218 and 241 micrograms per gram (µg/g), respectively (***Figure 5A***). The 2006 models produced an R^2 of 0.46 and 0.34, respectively, with a root-mean-squared error of 150 and 166 µg/g, respectively (***Figure 5B***). The 2005 model produced a better result because a longer TP gradient was represented. The prediction accuracy of the 2006 models is actually higher than that of 2005, shown by the lower root-mean-squared error of prediction. These models tell us that about half of the variability in species distributions can be explained by P, and P predicted at any given site by species is within 150-200 µg/g of its actual value. Mean TP values indicative of natural conditions vary among wetland primary sampling units but an error of this size will produce estimates that will be within 10 to 25 percent of the mean, suggesting a high predictive power of this model (Gaiser et al. 2006). These models are useful in tracking effects of historical P loading on species composition that may not be reflected in TP values. They also provide a mechanism interpreting cascading effects of P enrichment on algal species composition and, subsequently, the rest of the food web. Strong trends were observed in the residuals of these models that are important in evaluating model accuracy. The trends are related to the gradient reflected in the second axis of the non-metric multidimensional scaling, which is correlated with pH, water and soil depth. Deeper-water areas tend to encourage formation of deeper, peat soils of reduced pH, which is known to be a strong driver of algal community composition.

Table 1. Most common algal taxa in samples from fall 2005 and 2006 surveys

Taxa	Percent Occurrence Across Sites	Maximum Relative Abundance (%)	Total Phosphorus (µg/g periphyton dry weight)		Hydroperiod (days)	
			Optimum	Tolerance	Optimum	Tolerance
Achnanthes caledonica	8	2	283	140	271	55
Achnanthes gracillima	2	1	302	82	259	48
Amphora sulcata	4	8	183	262	199	94
Anabaena spp.	3	0	565	541	270	91
Aphanothece spp.	47	36	384	369	243	88
Brachysira brebissonii	9	23	542	353	208	94
Brachysira neoacuta	2	0	345	319	248	85
Brachysira neoexilis	40	16	321	289	261	80
Bulbochaete sp. 1	8	8	306	285	263	78
Chroococcidiopsis sp.	14	8	1022	704	212	98
Chroococcus morph large	18	2	232	186	226	81
Chroococcus morph small	37	1	240	189	236	80
Closterium spp.	1	1	271	137	295	49
Coelastrum spp.	23	3	235	254	237	80
Coelosphaerium spp.	2	0	288	221	239	80
Cosmarium commisurale	3	10	456	407	275	81
Cosmarium contractum	3	1	344	163	254	105
Cosmarium isthmium	1	0	363	208	222	60
Cosmarium ocellatum	2	0	207	93	220	81
Cosmarium pyramidatum	9	7	515	346	265	63
Cosmarium reniforme	2	19	682	462	275	62
Cosmarium spp.	23	1	425	322	242	89
Cyclotella meneghiniana	2	4	686	551	221	64
Desmidium baileyii	8	7	248	212	240	75
Desmidium schwartzii	3	5	356	218	230	113
Diadesmis confervacea	0	3	1681	815	315	76
Diploneis oblongella	2	0	192	178	241	69
Diploneis parma	4	21	1267	944	270	59
Encyonema evergladianum	40	7	226	174	275	58
Encyonema sp. 1	32	29	391	284	252	72
Encyonema silesiacum var. elegans	6	4	525	226	246	91
Encyonema sp. 3	2	8	260	194	300	63
Encyopsis sp. 1	6	5	594	335	169	110
Encyonopsis microcephala	24	3	462	339	253	87
Encyonopsis subminuta	4	0	563	337	282	53
Euastrum cornubiense	6	1	383	300	235	99
Euastrum pectinatum	2	0	389	299	301	19
Euastrum small morph	7	0	280	173	287	56
Euastrum spp.	3	0	332	179	262	59
Eunotia flexuosa	3	4	1400	736	244	53
Eunotia incisa	1	0	976	610	198	99
Eunotia monodon	1	0	528	125	279	84
Eunotia naegelii	5	4	1039	512	232	91

Table 1. continued

Taxa	Percent Occurrence Across Sites	Maximum Relative Abundance (%)	Total Phosphorus (µg/g periphyton dry weight)		Hydroperiod (days)	
			Optimum	Tolerance	Optimum	Tolerance
Eunotia rabenhorstiana var. elongata	1	1	981	605	246	92
Eunotia spp.	4	1	505	409	239	60
Fragilaria nana	15	1	434	319	263	61
Fragilaria spp.	1	0	1410	958	34	54
Fragilaria synegrotesca	32	11	274	210	259	59
Fragilaria ulna	1	5	633	522	274	55
Frustulia rhomboides var. crassinerva	7	6	556	311	178	104
Frustulia rhomboides	1	2	441	213	245	71
Genicularia elginensis	6	5	241	224	235	88
Gloeocapsa spp.	8	4	319	289	224	83
Gloeothece spp.	45	12	281	261	234	87
Gomphonema gracile	2	1	237	163	250	60
Gomphonema intricatum var. vibrio	10	2	401	301	267	72
Gomphonema spp.	1	1	1310	731	266	22
Gomphonema vibrioides	2	8	244	119	271	43
Gomphosphaeria spp.	16	26	385	324	269	81
Johannesbaptista sowerbyi	3	31	180	108	221	56
Lagynion spp.	3	0	251	208	255	101
Lyngbya spp.	19	29	314	349	258	74
Mastogloia lanceolata	1	2	142	167	194	103
Mastogloia smithii	39	45	251	198	264	64
Micrasterias pinnata	1	8	288	183	306	54
Micrasterias spp.	23	1	450	535	224	88
Mougeotia large morph	5	66	357	165	260	72
Mougeotia small morph	9	10	337	238	251	83
Navicula cryptocephala	11	1	355	191	268	65
Navicula radiosa	7	1	287	166	245	82
Navicula radiosafallax	5	0	270	156	244	83
Navicula spp.	1	2	1630	1006	253	37
Navicula subtilissima	9	0	295	183	268	82
Nitzschia amphibia	5	1	1054	700	208	90
Nitzschia nana	2	2	1358	622	177	129
Nitzschia palea	10	2	423	433	226	73
Nitzschia palea var. debilis	39	2	401	351	210	99
Nitzschia serpentiraphe	13	3	176	114	239	75
Nitzschia spp.	1	1	770	735	171	25
Oedogonium (large)	7	29	334	199	225	98
Oedogonium (small)	17	2	434	301	231	93
Onychonema spp.	1	1	417	277	249	116
Oocystis spp.	12	0	237	166	256	64
Oscillatoria spp.	4	10	419	290	244	68
Peridinium spp.	18	5	254	198	257	70
Pinnularia microstauron	1	3	879	518	234	99

Table 1. continued

Taxa	Percent Occurrence Across Sites	Maximum Relative Abundance (%)	Total Phosphorus (µg/g periphyton dry weight)		Hydroperiod (days)	
			Optimum	Tolerance	Optimum	Tolerance
Pinnularia viridis	3	7	832	579	130	137
Pleurotaenium minutum var. excavatum.	28	53	314	271	228	79
Pleurotaenium minutum var. attenuatum	18	38	322	275	254	75
Rhabdoderma linearis	44	9	235	210	242	82
Rhabdoderma sigmoidea	29	4	210	206	235	79
Rhopalodia gibba	1	1	211	112	190	67
Scenedesmus spp.	7	4	347	328	240	68
Schizothrix spp.	46	59	252	246	237	80
Scytonema hofmannii	39	96	238	249	223	87
Sellaphora laevissima	2	0	163	102	197	94
Staurastrum connatum	3	1	484	337	278	44
Staurastrum excavatum	19	2	411	218	278	73
Staurastrum longebrachiatum	5	15	451	302	248	98
Staurastrum ophiurum f. cambriatum	1	14	487	226	260	88
Stauroneis phoenicenteron	1	7	360	227	218	90
Staurastrum cf. sonthalium	1	6	375	225	256	34
Staurastrum spp.	2	9	405	348	70	118
Stipiticoccus spp.	25	3	264	222	244	75
Teilingia spp.	2	1	408	98	274	30
Tetraedron caudatum	1	0	547	406	273	95
Tetraedron pentaedricum	1	0	293	148	307	36
Tetraedron spp.	6	0	339	218	268	67
Triploceras spp.	1	70	334	202	253	55

Table 2. Most common diatom taxa in samples from fall 2005 and 2006 surveys

Taxa	Percent Occurrence Across Sites	Maximum Relative Abundance (%)	Total Phosphorus (µg/g periphyton dry weight)		Hydroperiod (days)	
			Optimum	Tolerance	Optimum	Tolerance
Achnanthes caledonica L-B	13	12	479	474	252	86
Achnanthes cf. *minutissima* var. *gracillima* (Meist.) L-B	2	2	218	161	213	70
Amphora sulcata Bréb.	6	46	225	225	195	76
Brachysira spp.	24	18	159	192	212	92
Brachysira brebissonii Ross in Hartley	11	78	430	236	199	104
Brachysira neoexilis Lange-Bertalot	45	51	277	280	219	95
Brachysira vitrea (Grun.) Ross in Hartley	6	5	404	351	183	113
Cyclotella meneghiniana Kütz.	20	4	380	317	231	88
Diploneis oblongella (Naegelii ex. Kütz.) R. Ross	10	2	186	179	196	70
Diploneis parma Cleve	24	31	654	681	193	84
Encyonopsis sp.	6	47	546	269	215	114
Encyonopsis microcephala (Grun) Krammer	26	56	375	255	218	92
Encyonema evergladianum Krammer	40	68	186	169	246	83

Table 2. continued

Taxa	Percent Occurrence Across Sites	Maximum Relative Abundance (%)	Total Phosphorus (µg/g periphyton dry weight)		Hydroperiod (days)	
			Optimum	Tolerance	Optimum	Tolerance
Encyonema sp. 2	42	53	399	327	252	85
Encyonema silesiacum var. *elegans*	19	31	462	293	249	73
Encyonema sp.3	4	2	212	162	304	38
Eunotia camelus Ehr.	4	40	527	423	277	59
Eunotia flexuosa Bréb. ex. Kütz.	12	13	769	578	250	65
Eunotia incisa W. Smith ex. Gregory	9	28	695	361	214	100
Eunotia monodon Ehr.	3	2	528	220	258	62
Eunotia naegelii Migula	11	27	970	543	242	78
Eunotia rabenhorstiana v. *elongata* (Patrick) Metz. & L-B	4	4	526	229	217	105
Fragilaria nanana Lange-Bertalot	13	13	461	277	274	79
Fragilaria synegrotesca Lange-Bertalot	43	73	318	251	253	68
Fragilaria ulna [danica complex] (Nitz.) Lange-Bertalot	3	5	628	370	255	64
Frustulia rhomboides (Her.) de Toni	8	39	449	277	172	115
Gomphonema affine Kütz.	2	4	868	720	236	93
Gomphonema gracile Ehrenberg emend. Van Heurck	4	5	703	558	155	90
Gomphonema intricatum var. *vibrio* Ehr. (Cl.)	34	10	430	385	240	77
Gomphonema vibrioides Reichardt et Lange-Bertalot	10	5	298	248	260	71
Mastogloia lanceolata Thwaites ex. W. Sm.	2	7	421	322	247	88
Mastogloia smithii var. *lacustris* Grunow	10	58	283	190	190	84
Mastogloia smithii Thwaites ex. W. Sm.	43	88	220	199	239	76
Navicula cryptotenella Lange-Bertalot	26	16	403	377	234	83
Navicula radiosa Kütz.	23	7	320	237	247	83
Navicula subtilissima Cl.	16	9	219	156	232	90
Nitzschia amphibia var. *amphibia* Grun..	14	20	811	596	221	79
Nitzschia nana Grun. in Van Heurck	6	42	879	577	157	74
Nitzschia palea var. *debilis* (Kütz.) Grun.	39	36	216	271	192	88
Nitzschia palea (Kütz.) W. Sm.	10	24	339	478	221	58
Nitzschia serpentiraphe Lange-Bertalot	28	33	102	105	183	92
Pinnularia gibba Ehr.	2	1	302	225	174	86
Pinnularia gibba Ehr. Morph 2	4	5	837	609	204	107
Pinnularia microstauron (Ehr.) Cl.	6	4	530	391	207	95
Pinnularia stomatophora (Grun.) Cl.	2	2	609	495	140	41
Pinnularia viridiformis Krammer	5	3	602	390	163	121
Rhopalodia gibba (Ehr.) O. Müll.	4	2	610	585	235	67
Sellaphora laevissima (Kütz.) Mann	9	9	372	349	221	88
Sellaphora pupula (Kütz.) Mereschk	3	3	849	535	187	111
Stenopterobia curvula (W. Sm.) Kr.	4	4	689	304	203	80
Stauroneis phoenicenteron (Nitz.) Ehr.	5	6	672	430	174	116

Figure 14. Relationships between measured periphyton total phosphorus content and that predicted from the total phosphorus optima and tolerances of diatoms and soft algae from the Fall A) 2005 and B) 2006 surveys.

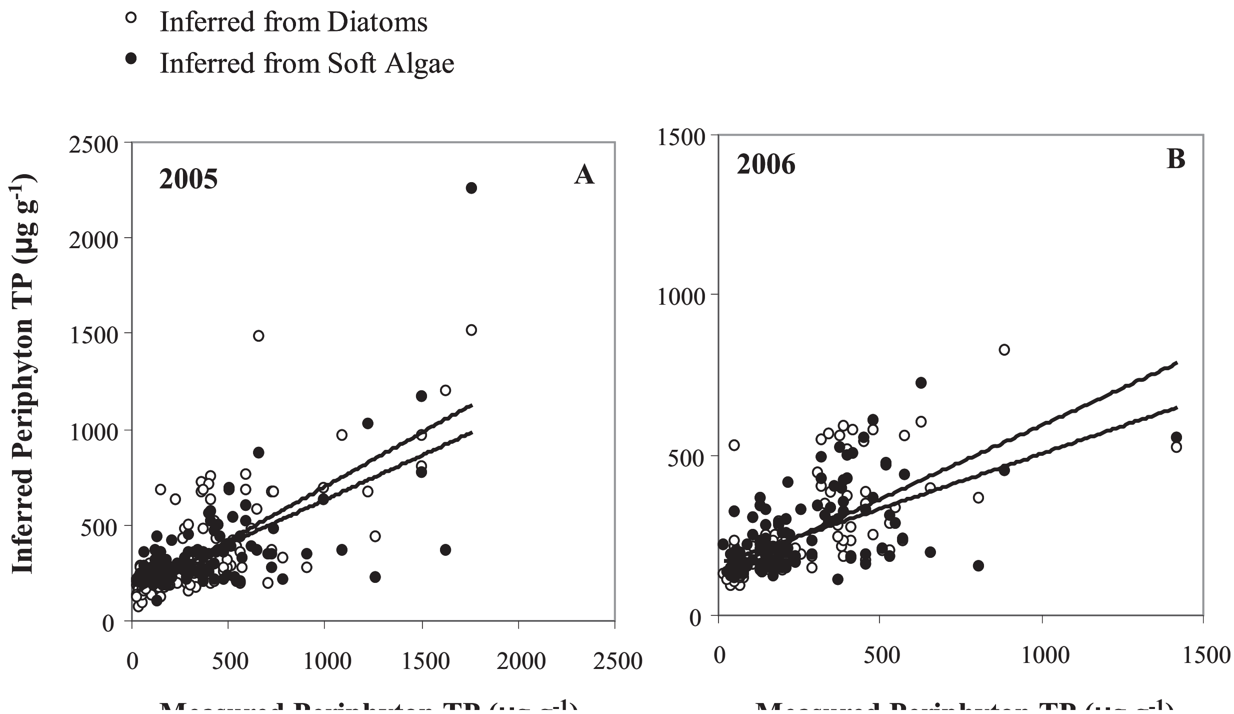


Figure 5. Relationships between measured periphyton TP content and that predicted from TP optima and tolerances of diatoms and soft algae from fall A) 2005 and B) 2006 surveys

Shortened Hydroperiod

Hypothesis

Shortened hydroperiods cause a reduction in the proportion of diatoms and green algae and an increase in calcareous blue-green algae, possibly reducing food value of periphyton, and affecting overall productivity of the Everglades.

Supporting Data

Compositional responses of periphyton to hydrologic change were quantified in field and laboratory studies by Gottlieb et al. (2005, 2006) and Thomas et al. (2006a). Gottlieb et al. (2006) found marked differences between algal communities in long and short hydroperiod marshes within Everglades National Park and derived hydrologic optima and tolerances for the most abundant species. Thomas et al. (2006a) and Gottlieb et al. (2005) conducted drying and rewetting experiments to determine the length of time it takes the community to be measurably altered when exposed to an alternative hydroperiod and found significant change within days to weeks of exposure. Further, studies by Geddes et al. (2003) and Dorn et al. (2006) documented the connection between periphyton composition and consumers showing that the two are connected partly through the periphyton-derived detrital food web and also through nutrient regeneration by the animals.

<u>Progress</u>

Periphyton biomass metrics (biovolume, dry and ash-free dry biomass) tended to increase with decreasing hydroperiod and increasing number of days since last dry, while quality metrics (TP, chlorophyll *a* and organic content) tended to decrease with decreasing hydroperiod and increasing number of days since last dry (***Figure 6***). Periphyton at deeper sites also had a higher TP, organic and chlorophyll *a* content (***Figure 4***).

Hydrology also strongly controlled compositional differences among sites. To determine the magnitude of effect of hydrology on species composition, we created weighted-averaging regression models relating diatom and soft algae species composition to hydroperiod in a similar manner to those developed for TP described above. Hydroperiod optima and tolerance ranges reported in ***Table 1*** and ***2*** for diatoms and soft algae can be used to predict the hydroperiod of the site. The 2005 models produced an R^2 of 0.34 and 0.29, respectively, with a root-mean-squared error of 73 and 76 days, respectively (***Figure 7A***). The 2006 models produced an R^2 of 0.26 and 0.24, respectively, with a root-mean-squared error of 64 and 66 days, respectively (***Figure 7B***). This means about one-third the variability in hydrology can be explained by diatom and soft algae species composition and hydrology estimates derived from the distribution of these taxa will be within about 70 days of the measured value. Weaker results for 2006 are again expected due to the shortened length of the hydroperiod gradient in this year when much of the Everglades were very dry. We expect that these models will improve with time once legacy effects of changes in hydrology can be examined. We also are exploring new approaches to weighted-averaging regression/calibration being developed to take interactive effects, like that of hydrology and P availability, into account. Once fully developed, these models can be used to guide expectations of algal assemblage composition from the Natural System Model (NSM) or other targets.

Future Directions

Water quality and hydrologic inference models may also be strengthened when created and employed in a regionally-specific manner. We cannot yet employ these models as our within-primary sampling unit replication is below the level needed for weighted-averaging regression/calibration. However, with each year of sampling, we add within-primary sampling unit replication and should soon have plenty of sites to be able to employ these region-specific models. Basin-specific P-prediction models presented in Gaiser et al. (2006) had much higher predictive power than whole system models, suggesting that model development on a smaller scale is also an appropriate approach in this survey. Conversely, multiple models can also be more cumbersome so our next steps are to 1) develop a multi-parameter model for the entire system that explains residual trends driven by large-scale variability and 2) develop explicit P and hydrology prediction models for each primary sampling unit and, by cross-validating across primary sampling units, determine the optimal scale for a univariate prediction model. We believe that the latter approach will be most powerful, as it will facilitate direct interpretation of eutrophication trends from periphyton species data.

Figure 16. Relationships between periphyton attributes and hydrperiod and days since last dry.

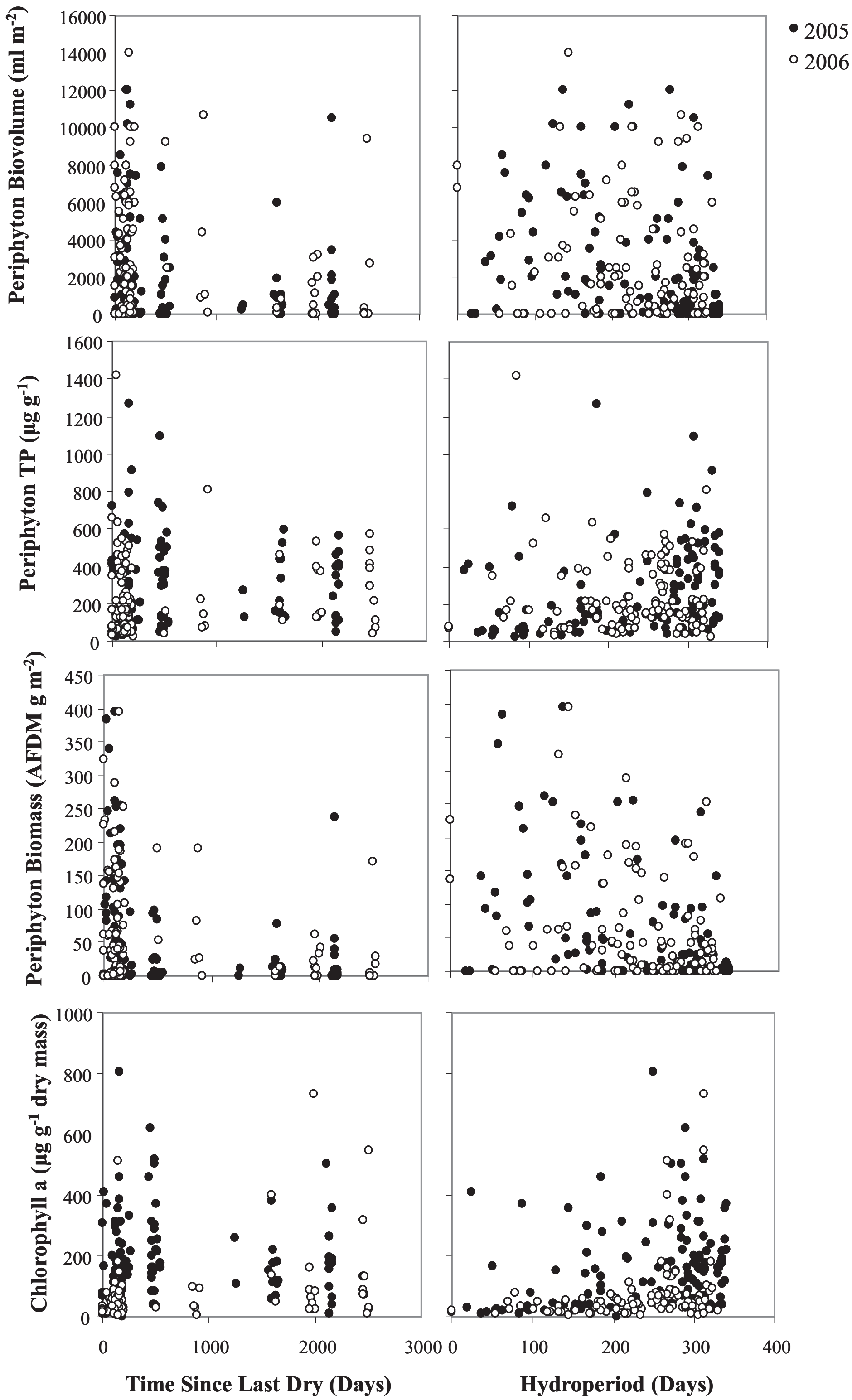


Figure 6. Relationships between periphyton attributes and hydroperiod and days since last dry

Figure 17. Relationships between measured hydroperiod and that predicted from the hydroperiod optima and tolerances of diatoms and soft algae from the Fall A) 2005 and B) 2006 surveys.

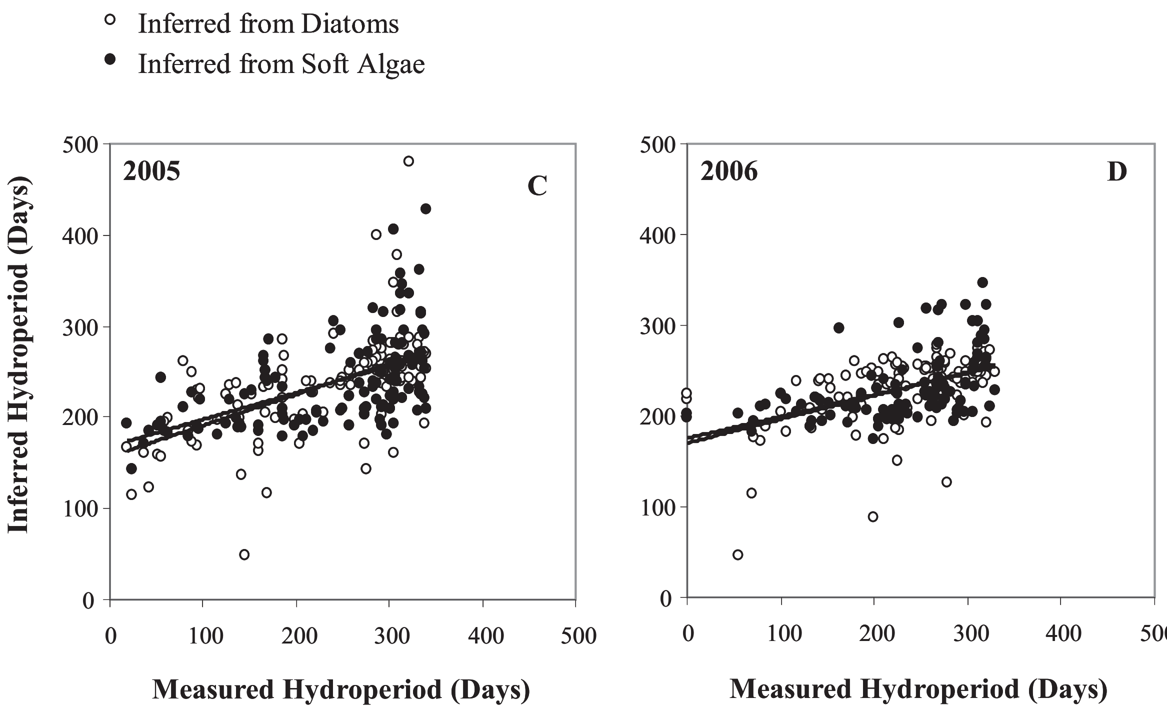


Figure 7. Relationships between measured hydroperiod and that predicted from TP optima and tolerances of diatoms and soft algae from fall A) 2005 and B) 2006 surveys

Periphyton Mats and Consumer Dynamics

Everglades wetlands have among the highest rates of primary production recorded from wetlands and similarly high standing crops of periphytic algae (Goldsborough and Robinson 1996, Ewe et al. 2006). The remarkable amount of algal productivity in the Everglades would seem to support a large standing crop of aquatic consumers, but in fact the secondary productivity appears to be relatively low compared to other aquatic systems (Turner et al. 1999). We have hypothesized that this low apparent ecological efficiency is tied to the composition of Everglades periphyton mats (Geddes and Trexler 2003, Chick et al. 2008). Mature periphyton mats in Everglades marshes are held together by a calcareous matrix secreted by filamentous cyanobacteria and may reach thicknesses between 1.9 and 10 centimeters (cm) (Van Meter-Kasanof 1973, Browder et al. 1994). These mats host a complex community of algae, microbes, protozoans and invertebrates (Browder et al. 1994, Donar et al. 2004, Gaiser et al. 2004). Everglades periphyton has very high stoichiometric ratios (average carbon [C]:P = 4,759; C:nitrogen [N] = 21.8; N:P = 208.7; n = 159 compare to Redfield ratio of C:N:P = 106:16:1), indicative of very low quality food resources because of P limitation and a ‘stoichiometric bottleneck’ that limits trophic efficiency (Van Donk et al. 2008). Evidence suggests that edible algae (diatoms and green algae) inhabiting these mats gain an ‘associational refuge’ (Hay 1986, Duffy and Hay 1990) from grazers resulting from physical and chemical defenses of cyanobacteria. Geddes and Trexler (2003) demonstrated that destroying the physical structure of mature periphyton mats allowed more effective grazing of edible algae by omnivorous eastern mosquitofish (*Gambusia holbrooki*). Green algae and diatoms are generally considered more edible by animals than blue-green algae (Lamberti 1996, Steinman 1996, Sullivan and Currin 2000). Enrichment of Everglades wetlands by P serves to dramatically change these periphyton communities and, at moderate to high levels, leads to the complete disassociation of the mat complex followed by increased abundance of phytoplanktonic forms (McCormick et al. 2002, Gaiser et al. 2005).

Periphyton is the major source of primary production in Everglades marshes (Ewe et al. 2006). Formation of mature periphyton mats with their characteristic physical and biotic structure may be critical to trophic interactions and food web dynamics in Everglades marshes. Furthermore, it is likely that direct consumption of algae is not the primary route of energy flow in marshes of the southern Everglades; analysis of stable isotopes of C and N suggests detrital routes of energy flow dominate the energy budget of unenriched Everglades marshes (Williams and Trexler 2006). The source of this detritus is heavily weighted toward senescing algal mats, so the periphyton mats are appropriately considered the 'base' of the Everglades food web, but via detrital production that is colonized by bacteria that are consumed by animals. We hypothesize that suppression in the Everglades of grazing, important in controlling algal dynamics in most aquatic systems (Steinman 1996, Lamberti 1996), permits the accumulation of algae found here.

Algal mats can influence aquatic animal communities by providing food and habitat. Everglades periphyton mats are home to rich infauna that both consume it and use it as a refuge from predation (Liston and Trexler 2005, Liston 2006, Liston et al. 2008). Liston and Trexler (2005) observed most Everglades macroinvertebrates inhabit periphyton mats as infauna. When these mats were eliminated by nutrient addition in field mesocosms, the macroinvertebrate density in the benthos failed to change, indicating a marked net loss of these small animals attributed to consumption by small fish and aquatic invertebrates (Liston et al. 2008). Path analysis of sampling data also supported consumption of enhanced macroinvertebrate production by small fishes and large invertebrates along nutrient gradients (Sargeant, Gaiser and Trexler unpublished analyses). In the latter data, P enrichment enhanced periphyton production and small fish and large macroinvertebrate density, but not periphyton mat dwelling macroinvertebrate density (chironomid larvae, amphipods, etc).

Evidence that algal production limits consumer dynamics in the southern Everglades is provided by analysis of effects of nutrient gradients in the field. Increasing density, biomass and changes in age structure with increasing nutrient availability are all evidence of food limitation in the unenriched Everglades. Failure to find such increases must be interpreted with caution because increased local secondary productivity could be consumed by mobile predators that are not sampled at the same scale or by immigration of secondary consumers from the local site. We have avoided estimating secondary productivity from our data because most such calculations require an assumption of samples from a closed system while landscape-scale movements of animals tied to water fluctuation is probably of great importance in this system (Trexler et al. 2001, Ruetz et al. 2005).

Stoichiometry and Everglades Consumers

We used the 2005 REMAP data (Scheidt and Kalla 2007) to evaluate the relative power of algal mat stoichiometric composition and algal species composition to explain patterns of density of small omnivorous fishes and herbivorous macroinvertebrates and fish. REMAP is a systemwide sampling study organized by the USEPA that focuses on biogeochemical parameters and included throw-trap sampling of aquatic consumers in 2005. REMAP samples were collected between September 23 and December 7, 2005 using a stratified random protocol similar to that used in MAP sampling. Because the focus of REMAP is on biogeochemistry, more detailed analysis of nutrient parameters from periphyton were recorded than is conducted during MAP monitoring. These data permitted an analysis of the relative impact of hydrological parameters and resources parameters, including stoichiometric ratios and relative abundance of edible algae, on aquatic consumers. We used multiple regressions that included parameters hypothesized to be informative for these dependent variables in other analyses. We focused on the N:P ratio because the accumulation of inorganic carbon in Everglades periphyton complicates estimates of stoichiometric ratios. Eliminating inorganic carbon makes the high C:N and C:P ratios reported above even more extreme. The N:P ratio is unaffected by partitioning of organic and inorganic components.

The N:P ratio in periphyton tissues was linearly related to TP in periphyton tissues in a semi-log plot (***Figure 8A***) suggesting marked P limitation at low P availability or luxury uptake of P as it becomes more available relative to N (see Sterner and Elser 2002). A negative correlation exists between the relative abundance of edible algal taxa in periphyton tissues and the N:P ratio in the REMAP data ($r = -0.447$, $n = 158$, $P < 0.001$). While the relative abundance of edible algal types (green algae and diatoms) decreases with increasing N:P (***Figure 8B***), a triangle pattern is apparent (Thomson et al. 1996). This is revealed by a quantile regression that fits a separate slope for selected quantiles of the dataset (Terrell et al. 1996, Cade and Noon 2003). In this case, the highest 10^{th} quantile of the data display a slope of -0.131 ($t_{1,15} = -3.64$, $P < 0.001$) and the lowest 10^{th} quantile displays a slope of -0.032 ($t_{1,15} = -1.87$, $P = 0.063$). This pattern suggests that the maximum values of edible algae are more strongly set by P limitation (indicated by the N:P ratio) than are the lower values; other factors may be interacting with P availability to determine algal edibility when the N:P ratio is relatively low. Adding hydrological parameters, though improving the model fit, does not remove the changing slope with quantile of the N:P ratio as an explanatory variable for algal edibility.

The REMAP data permit us to ask if algal edibility or algal nutrient dynamics better explain patterns in density of aquatic consumers in the Everglades. This provides a test of the hypothesis that P limitation per se is transferred up the food web, or alternatively if food limitation is an outcome of changing algal composition (blue-green algae with grazer deterrents versus non-defended species). We regressed the density of herbivores and, separately, omnivorous fishes, on several independent variables that describe resource availability and hydrology (***Table 3***) to test these hypotheses. Herbivores were the summed density of all snails, grass shrimp, crayfish, and two species of herbivorous fishes, flagfish (*Jordanella floridae*) and sailfin molly (*Poecilia latipinna*). While crayfish and grass shrimp are omnivores, we included them because stable isotope and gut content analyses indicate plant matter is important in their diets (see Loftus 1999). Omnivorous fishes were comprised of all fishes collected by throw trap except the two herbivorous species.

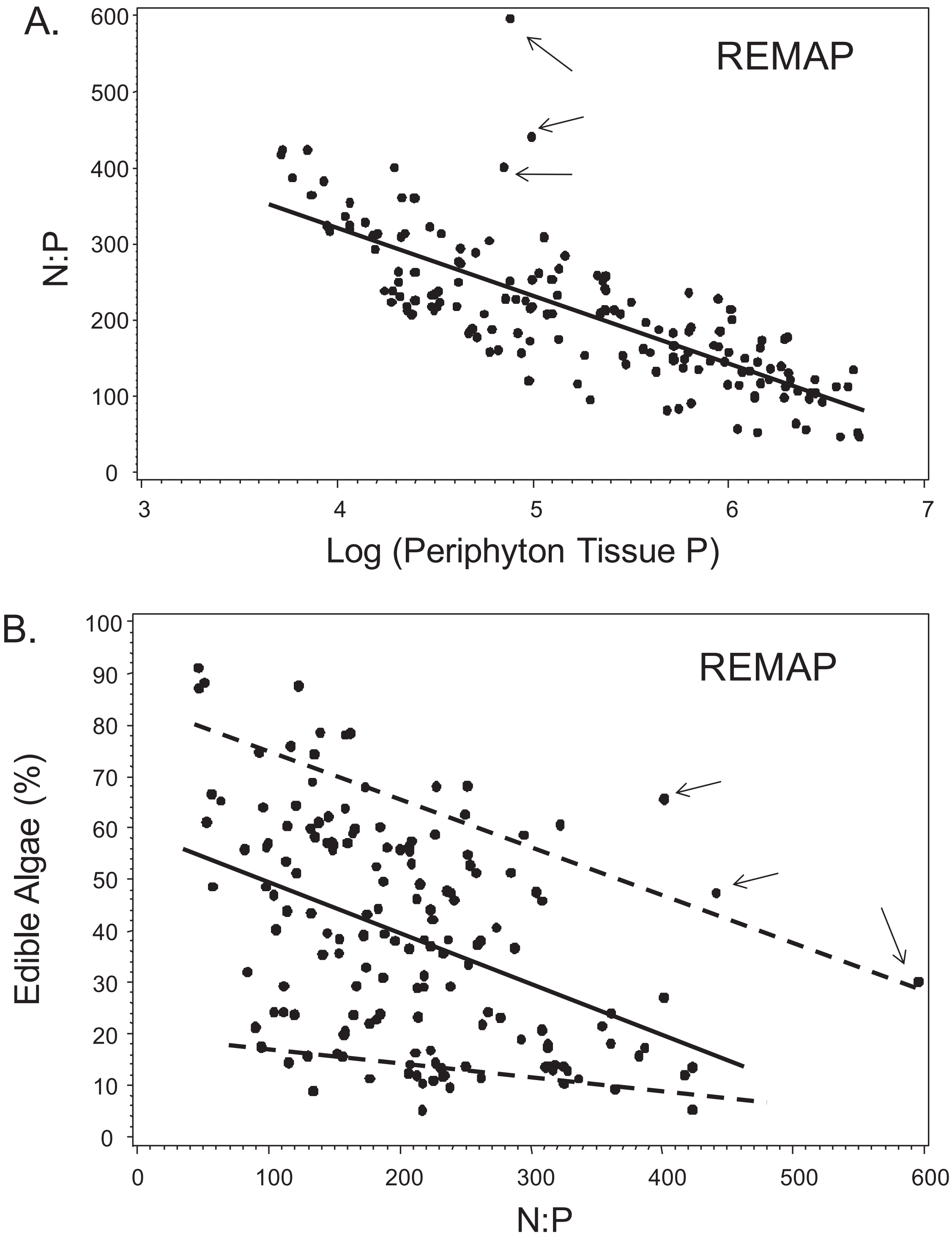


Figure 8. A) N:P of periphyton versus tissue P (ln µg/g + 1) with arrows identifying three apparent outliers and B) relationship between relative abundance of edible algae and N:P from REMAP data collected during 2005 wet season with the solid line indicates a simple least-squares regression of these data, the dashed line indicating a quantile regression slope from the highest 10% quantile, and arrows indicating same three points from A

Table 3. Regression analysis of data collected for herbivore and omnivorous fish density during 2005 REMAP

Variable	Estimate	Error	t	P
Herbivore Density Adj R-Sq = 0.49; n = 155				
Intercept	-4.19	0.90	-4.67	<0.0001
Edible algae (%)	0.04	0.004	8.21	<0.0001
N:P molar	0.0005	0.001	0.51	0.612
Hydroperiod (days)	-0.01	0.002	-2.53	0.013
Days since reflooding (log days)	0.87	0.19	4.50	<0.0001
Stem density (log #/m^2) *	0.11	0.09	1.23	0.220
Omnivorous Fish Density Adj R-Sq = 0.45; n = 155				
Intercept	-2.25	0.86	-2.61	0.010
Edible algae (%)	0.03	0.004	7.92	<0.0001
N:P molar	-0.00043	0.00091	-0.47	0.639
Hydroperiod (days)	-0.004	0.002	-2.03	0.045
Days since reflooding (log days)	0.61	0.18	3.31	0.001
Stem density (log #/m^2) *	0.12	0.09	1.44	0.152

* #/m^2 = number per square meter

Both hydrological parameters and the relative abundance of edible algae contributed to explaining variation in herbivore density in the REMAP 2005 data (***Table 3***). Interestingly, N:P was not a significant effect in this model that explained 49 percent of the variation in the dependent variable. When edible algae were dropped from the statistical model, the adjusted R^2 dropped to 0.263 and N:P was a significant parameter with a negative slope consistent with P limitation ($t_{1,154} = -2.89$, $P = 0.004$). However, the R^2 adjusted only dropped to 0.227 when N:P was excluded, leaving only hydrological parameters. A similar weak effect of N:P was noted for analysis of omnivorous fishes (***Table 3***). The $R^2_{adjusted}$ was 0.45 for the model of their density that included both resource and hydrological parameters. Dropping edible algae relative abundance yielded an $R^2_{adjusted}$ of 0.230, and a model with only hydrological parameters had $R^2_{adjusted}$ of 0.163, suggesting only a slightly better performance of N:P with this dependent variable.

These results indicate that food edibility is a factor limiting consumer density in the Everglades. P limitation is an important factor in shaping Everglades periphyton mats. However, the effect of oligotrophy on the food web and consumer dynamics in general appears to be mediated through shaping the patterns of relative abundance of algal types in mats. It would seem that production of physical and chemical defenses against grazers by blue-green algae would be energetically costly and cause a trade-off between defense and other life functions for algae living with limited P availability. Presumably, these costs render them inferior competitors to green algae and diatoms when P is even slightly less limited, leading to changes in Everglades periphyton mats that ultimately culminate in their breakdown as a cohesive matrix. At elevated P availability, green algae and diatoms must be able to reproduce at rates permitting them to outgrow consumption by grazers. These suggestions could be cast as hypotheses for tests in controlled experimental conditions.

Periphyton Edibility and Hydrology Drive Aquatic Communities

Data collected for the MAP provide independent information to test hypotheses about the linkage of Everglades consumers and periphyton edibility derived from the REMAP data. Aquatic consumer and periphyton samples were collected in the wet season (late September through early December) of 2005 and 2006 (sampling has continued to the present, but algal composition is only available for these years) using a spatially stratified sampling Generalized Random Tessellated Grid (GRT) design (Philippi 2003, Stevens and Olsen 2004). MAP periphyton data include information on algal species composition and algal tissue P, but not percent C and N, so stoichiometric analysis is not possible. However, analysis of REMAP data suggested that algal tissue P and relative abundance of edible taxa provide the best predictors for consumer dynamics. We focus on independent variables tied to periphyton volume and edibility, vascular plant density, and hydrology to evaluate how resource availability, mediated through algal productivity and edibility, and hydrology affect consumer dynamics. We have included emergent plant stem density and periphyton mat volume as indices of habitat structure.

We noted geographic patterns in the relative abundance of edible taxa, as well as in consumer communities such as fish collected by throw traps, in the MAP data. Edible algae was estimated as the percent of all algae that was green or diatoms. Fish density is the sum of all species collected by throw trap. For example, edible algae relative abundance differed among water management areas such as Loxahatchee NWR, WCA 2A and WCA 3a (***Table 4***; Region 2005: $F_{1,7} = 7.97$, $P < 0.001$; 2006 $F_{1,7} = 13.54$, $P < 0.001$), as did fish density (***Table 4***; Region 2005: $F_{1,7} = 8.76$, $P < 0.001$; Region 2006:$F_{1,7} = 7.18$, $P < 0.001$). Both relative abundance of edible algae and fish density generally decrease from north to south, though with exceptions (***Figure 9***, ***Table 4***). We found that regional differences were confounded with differences in periphyton tissue TP and other possible environmental drivers so we did not include spatial data in subsequent statistical analyses. We will discuss this spatial confounding in our concluding section.

Table 4. Regional means of edible algae and fish density

Region	Edible Algae (%)		Fish Density (log #/m^2)	
	2005	2006	2005	2006
Pal Mar	54.2	41.6	2.08	1.96
Loxahatchee NWR	50.9	53.3	2.88	3.02
WCA 2A	60.6	32.3	2.95	2.64
WCA 3A	44.4	34.1	2.81	2.35
WCA 3B	49.1	49.9	2.70	2.95
Shark River Slough	33.9	28.1	2.23	1.95
Southern Marl Prairie	16.8	9.5	1.63	1.58
Taylor Slough	10.2	9.6	1.47	0.44

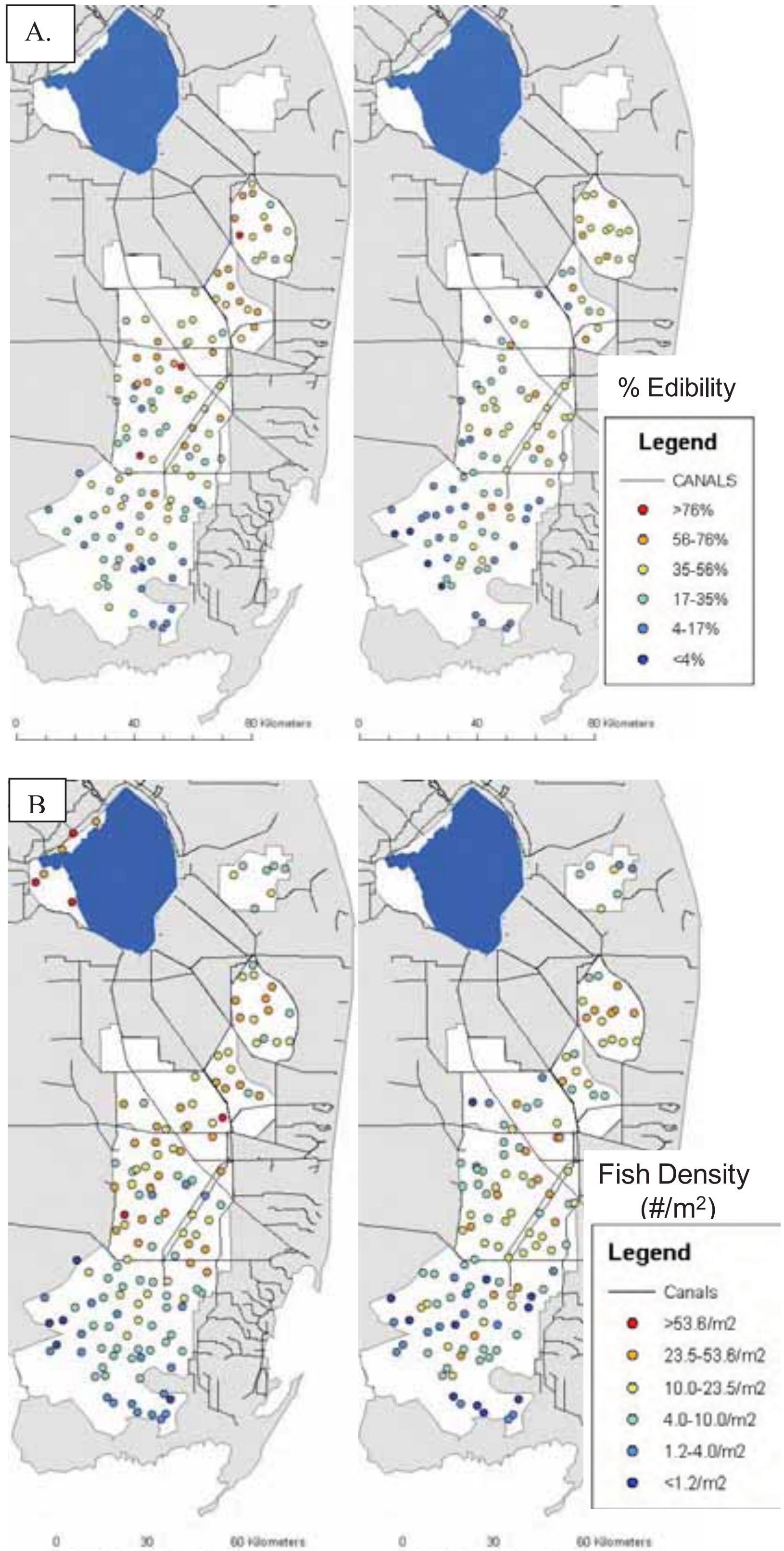


Figure 9. Maps illustrating distribution of A) relative abundance of edible algae (%) in 2005 and 2006. and fish density (#/m2) in 2005 and 2006

Prior to analysis of consumer data, we examined patterns of covariance of the independent variables to identify environmental gradients that could be treated as independent patterns in the search for potential drivers of community dynamics. The pattern of correlation among independent variables was roughly similar in the two years we are examining (***Table 5***). However, every correlation that was significant in 2005 was weaker in 2006 (because of high sample sizes, relatively weak and possibly trivial correlations of $r = 0.15$ are 'significant' in these data); no correlation over $r = 0.4$ changed sign between the two years. Weaker coherence of the data between years was also revealed in principal components analyses (***Table 6***). In 2005, the first factor explained 39.5 percent of the total variance and the first two factors explained 58.6 percent, while in 2006 the first factor explained only 27.5 percent and first two only 49.4 percent. In both years, the first factor from the principle component analysis was primarily tied to algal characteristics, though edibility (proportion of green algae and diatoms) was only strongly correlated to axis one (we arbitrarily define this as $r = 0.4$) in 2005. Though reversed between years, the patterns of correlations to axis 1 were the same in both years (sign of correlations between variables and factors). Thus, tissue P, organic matter (%), chlorophyll *a*, and edibility were inversely related to mat ash-free dry mass and volume. With the exception of depth at the time of sampling, which was positively correlated to tissue P, organic matter, etc. in both years, the hydrological parameters were primarily correlated with factor 2 in both years. Only factor 3 consistently revealed some common variability between algal and hydrological parameters. This is a surprising result because we generally expect periphyton volume and ash-free dry mass to be inversely correlated with hydroperiod.

In both years, periphyton edibility was correlated with periphyton tissue TP, hydroperiod, and stem density of emergent plants (***Table 7***). Thus, periphyton mats have a higher relative abundance of edible taxa in sites with more P, longer hydroperiods, or higher stem density. Simple plots of the dependent variable against an explanatory variable can be misleading when models contain multiple independent variables because the effects of other independent variables have not been accounted for, as they are in the regression model. We correct for this by illustrating the results of our regression models with partial regression plots that display the residuals of the dependent variable against the residuals of each explanatory variable after it has been regressed on the other explanatory variables in the model (Belsley et al. 1980). Although the axes are not on the same scale as the original variables, the plots accurately show the relationships fit by regression (i.e. the slope equals the partial regression coefficient reported in our tables) and allow visual inspection of the scatter of points and possible influential cases and lack-of-fit. In both years, our statistical models predicted a 20 percent change in relative abundance of edible algae between the lowest and highest levels of algal tissue TP (***Figure 10***). Though this relationship was significant, there was as much as a 75 percent range of in relative abundance of edible algae in both datasets (***Figure 10***), which was only partially explainable by other independent variables we measured (R^2_{adj} was < 0.3 in both years, notably in 2006; ***Table 7***).

Table 5. Correlation matrix of independent variables

Units are listed on the left side of the table. Pearson correlations are listed in the first row for each variable, with P values on the second row, and sample size on the third.

	Stem Density	Hydroperiod	Edible Algae	Periphyton Ash-Free Dry Mass	Periphyton Volume
Wet Season 2005					
Periphyton Tissue P (log µg/g)	-0.088	0.456	0.397	-0.838	-0.708
	0.294	<0.0001	<0.0001	<0.0001	<0.0001
	143	126	143	144	144
Stem Density (log #/m^2)	1	-0.035	0.115	0.102	0.032
		0.697	0.166	0.215	0.696
		128	146	148	148
Hydroperiod (days)		1	0.452	-0.466	-0.184
			<0.0001	<0.0001	0.037
			127	129	129
Edible Algae (%)			1	-0.432	-0.166
				<0.0001	0.044
				147	147
Periphyton Ash-Free Dry Mass (log g/m^2)				1	0.707
					<0.0001
					149
Wet Season 2006					
Periphyton Tissue P (log µg/g)	0.027	0.079	0.254	-0.712	-0.640
	0.761	0.394	0.003	<0.0001	<0.0001
	131	119	132	133	130
Stem Density (log #/m^2)	1	0.069	0.122	0.017	0.011
		0.455	0.166	0.841	0.903
		120	131	134	133
Hydroperiod (days)		1	0.215	-0.058	0.091
			0.019	0.525	0.323
			119	122	119
Edible Algae (%)			1	-0.250	-0.098
				0.004	0.267
				133	130
Periphyton Ash-Free Dry Mass (log g/m^2)				1	0.695
					<0.0001
					133

Table 6. Principle components analysis for 2005 and 2006

Correlations of factors to variables are reported. Total explained variance for each factor is reported at the bottom of each column, separately by year. Grey backgrounds indicate correlations exceeding 0.4.

Independent Variables	2005 Factors 1	2005 Factors 2	2005 Factors 3	2006 Factors 1	2006 Factors 2	2006 Factors 3
Periphyton Tissue P (log µg/g)	0.911	0.137	0.024	-0.867	0.073	-0.019
Periphyton Organic Matter (%)	0.86	0.111	0.042	-0.668	0.203	0.215
Ash-Free Dry Mass Periphyton Mat (log g)	-0.858	-0.169	0.129	0.882	0.031	0.023
Floating Mat Volume (log mL/m^3)*	-0.743	0.091	0.148	0.791	0.294	0.115
Chlorophyll *a* (µg/g)	0.577	0.068	0.168	-0.501	0.388	0.189
Average Hydroperiod (days)	0.319	0.881	-0.016	0.078	0.879	-0.064
Days Since Last Flooding (log days)	0.109	0.831	0.259	0.020	0.917	-0.077
Depth Recession Rate (cm/day)	-0.18	0.807	-0.201	0.057	0.039	-0.79
Depth (cm)	0.44	0.546	0.442	-0.298	0.681	0.46
Emergent Plant Stem Density (log #/m^2)	0.098	-0.019	-0.885	0.173	-0.078	0.55
Edible Algae (%)	0.499	0.347	-0.294	-0.219	0.19	0.623
Total Variance Explained	39.5%	19.1%	10.9%	27.5%	21.9%	14.8%

* mL/m^3 = milliliters per cubic meter

Table 7. Regression results for analysis of periphyton edibility (%) reported separately by year

N=124 in 2005 and N=116 in 2006

Variable	Estimate	Error	t	P
2005	R^2_{adj} = 0.299			
Intercept	-19.33	9.59	-2.01	0.046
Periphyton Tissue TP (log µg/g)	5.52	1.77	3.11	0.002
Hydroperiod (days)	0.08	0.02	4.26	<0.0001
Stem Density (log #/m^2)	3.38	1.26	2.68	0.008
2006	R^2_{adj} = 0.108			
Intercept	-17.14	12.54	-1.37	0.175
Periphyton Tissue TP (log µg/g)	5.53	2.01	2.75	0.007
Hydroperiod (days)	0.05	0.02	1.99	0.049
Stem Density (log #/m^2)	2.84	1.44	1.97	0.051

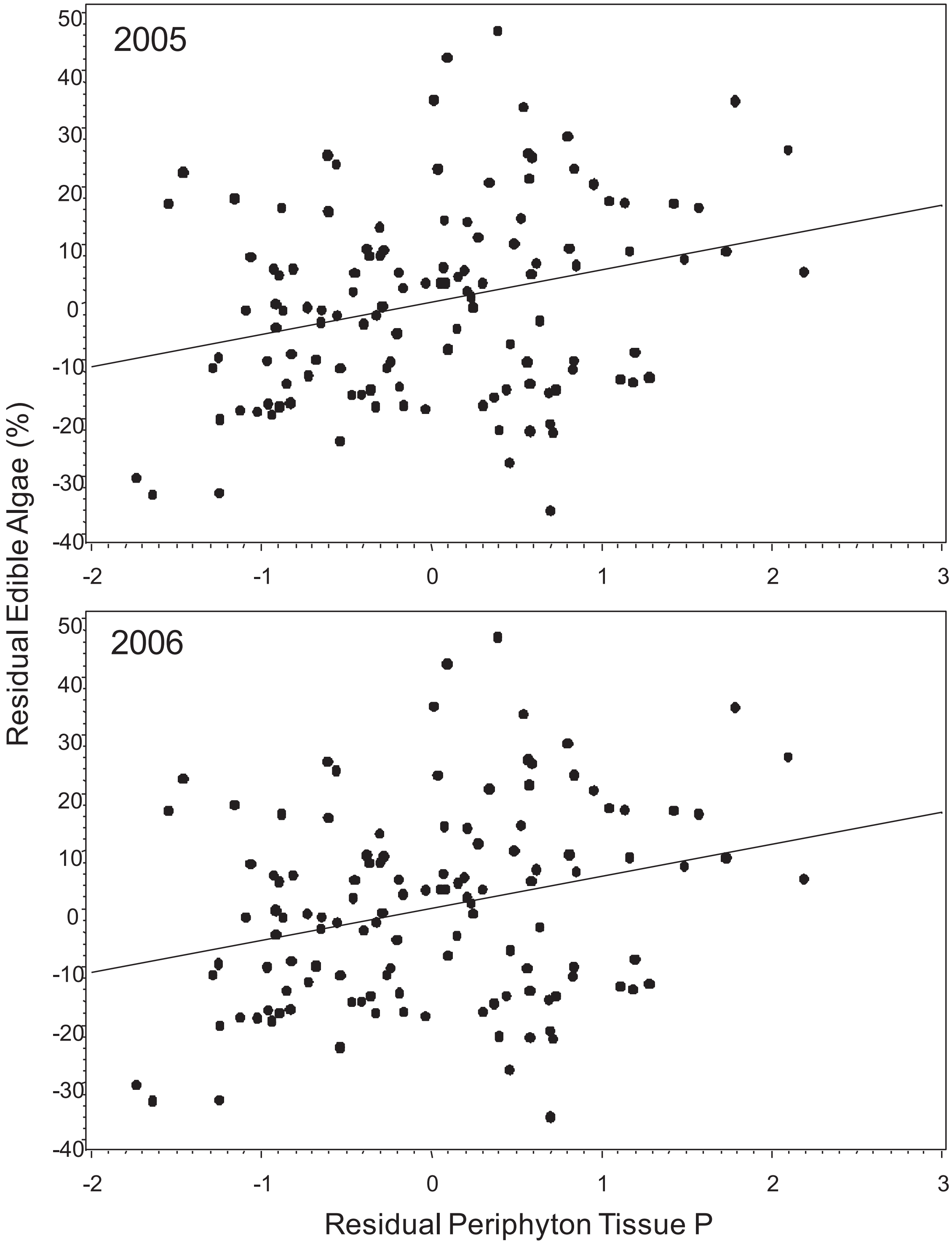


Figure 10. Partial regression plot of edible algae (%) versus periphyton tissue P reported separately for each year

The mean percentage of edible algae was 39.8 in 2005 and 32.6 in 2006. Residuals from a regression with hydroperiod (days) and stem density (#/m^2) are plotted to indicate the unique pattern attributable to tissue P

Consumer density in general increased with increasing relative abundance of green algae and diatoms in periphyton mats. Herbivore density was correlated with algal edibility and hydroperiod in both 2005 and 2006 (***Table 8***). The amount of variation explained by our models were similar in both years (R^2_{adj} = 0.237 and 0.256 in 2005 and 2006, respectively), though much variation in their density was unexplained. Edible algae relative abundance, hydroperiod and emergent plant stem density were all positively related to herbivore density in 2005, but only edible algae relative abundance and hydroperiod were positively related to herbivore density in 2006. Though significant both years, the effect of algal relative abundance was stronger in 2006 than in 2007 (***Figure 11***; partial regression slope was three times steeper in 2006 than in 2005). Omnivorous fish density also increased with relative abundance of edible algae in both years and the partial regression of hydroperiod also yielded positive slopes both years though the effect was marginal in 2005 (***Table 8***). Stem density was significant in 2005 and not in 2006. The model explained more variation in 2006 than in 2005 (R^2_{adj} = 0.249 in 2005 and 0.321 in 2006). The effect of algal relative abundance was 50 percent stronger in 2006 than in 2005 (***Figure 12***); partial regression slope in ***Table 8***). Note that both herbivore and omnivore density were log transformed, so that the effect would appear more marked and curvilinear on the arithmetic scale. Also, measures of periphyton mat volume and ash-free dry mass were not reported in these regression analyses because they never explained significant variation in either consumer dependent variable, neither in models with other periphyton and hydrology parameters nor alone.

Table 8. Regression results for analysis of herbivores and omnivorous fish density (#/m^2) reported separately by year

Herbivore N=126 in 2005 and N=117 in 2006; omnivorous fish N=126 in 2005 and N=117 in 2006

Variable	Estimate	Error	t	P
Herbivores				
2005	R^2_{adj} = 0.237			
Intercept	-0.75	0.40	-1.91	0.059
Edible Algae (%)	0.01	0.01	2.18	0.031
Hydroperiod (days)	0.004	0.001	3.87	0.000
Stem Density (log #/m^2)	0.18	0.08	2.26	0.026
2006	R^2_{adj} = 0.256			
Intercept	-0.37	0.43	-0.86	0.389
Edible Algae (%)	0.03	0.01	5.17	<0.0001
Hydroperiod (days)	0.004	0.001	2.75	0.007
Stem Density (log #/m^2)	-0.01	0.09	-0.10	0.922
Omnivorous Fish				
2005	R^2_{adj} = 0.249			
Intercept	0.48	0.34	1.41	0.162
Edible Algae (%)	0.02	0.00	3.79	0.001
Hydroperiod (days)	0.002	0.00	1.94	0.055
Stem Density (log #/m^2)	0.19	0.07	2.77	0.007
2006	R^2_{adj} = 0.321			
Intercept	0.63	0.34	1.85	0.067
Edible Algae (%)	0.03	0.00	6.40	<0.0001
Hydroperiod (days)	0.003	0.00	2.47	0.015
Stem Density (log #/m^2)	0.005	0.07	0.07	0.947

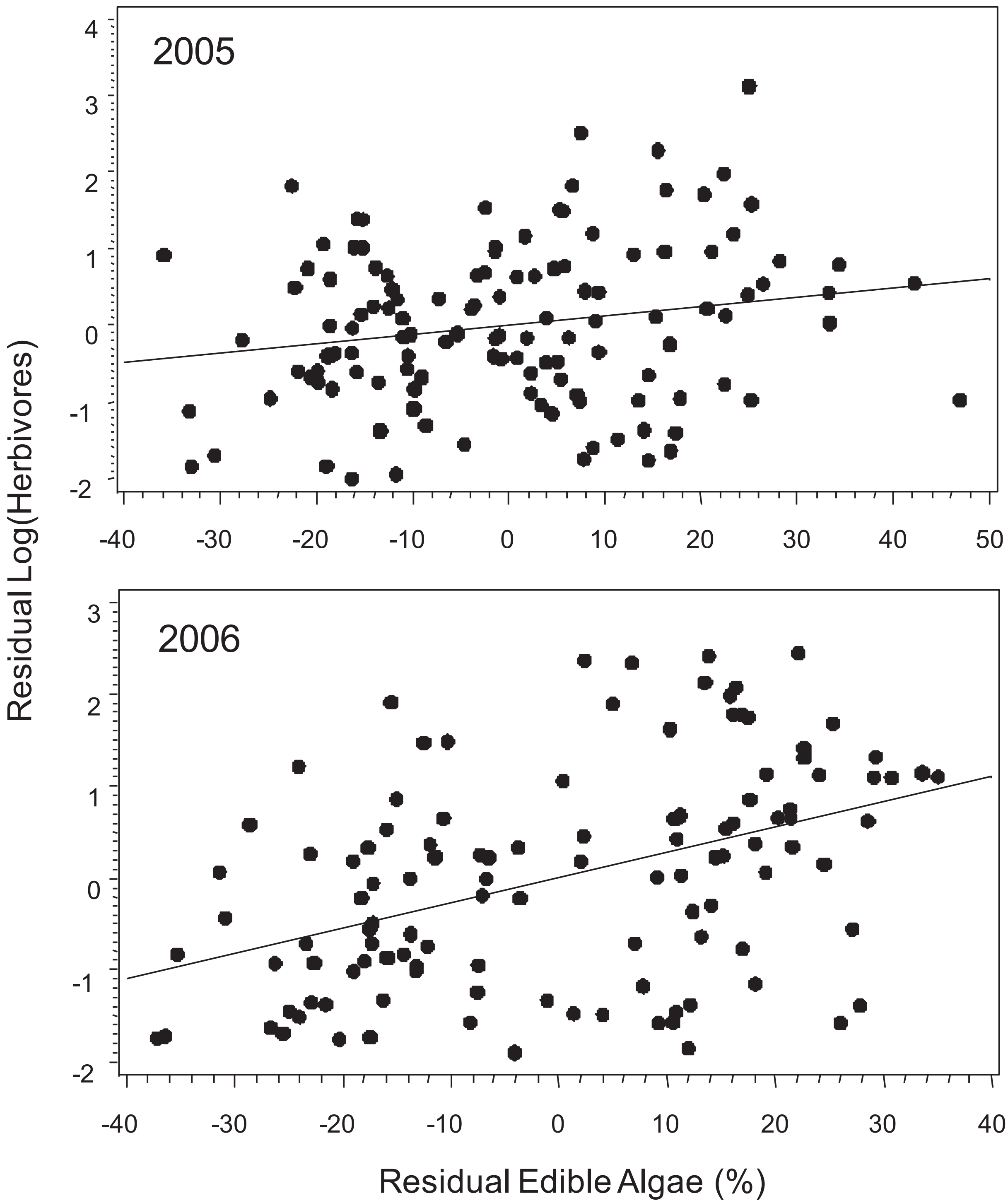


Figure 11. Partial regression plot of herbivore density versus edible algae (%)

Residuals from a regression with hydroperiod (days) and stem density (#/m^2) are plotted to indicate the unique pattern attributable to tissue P.

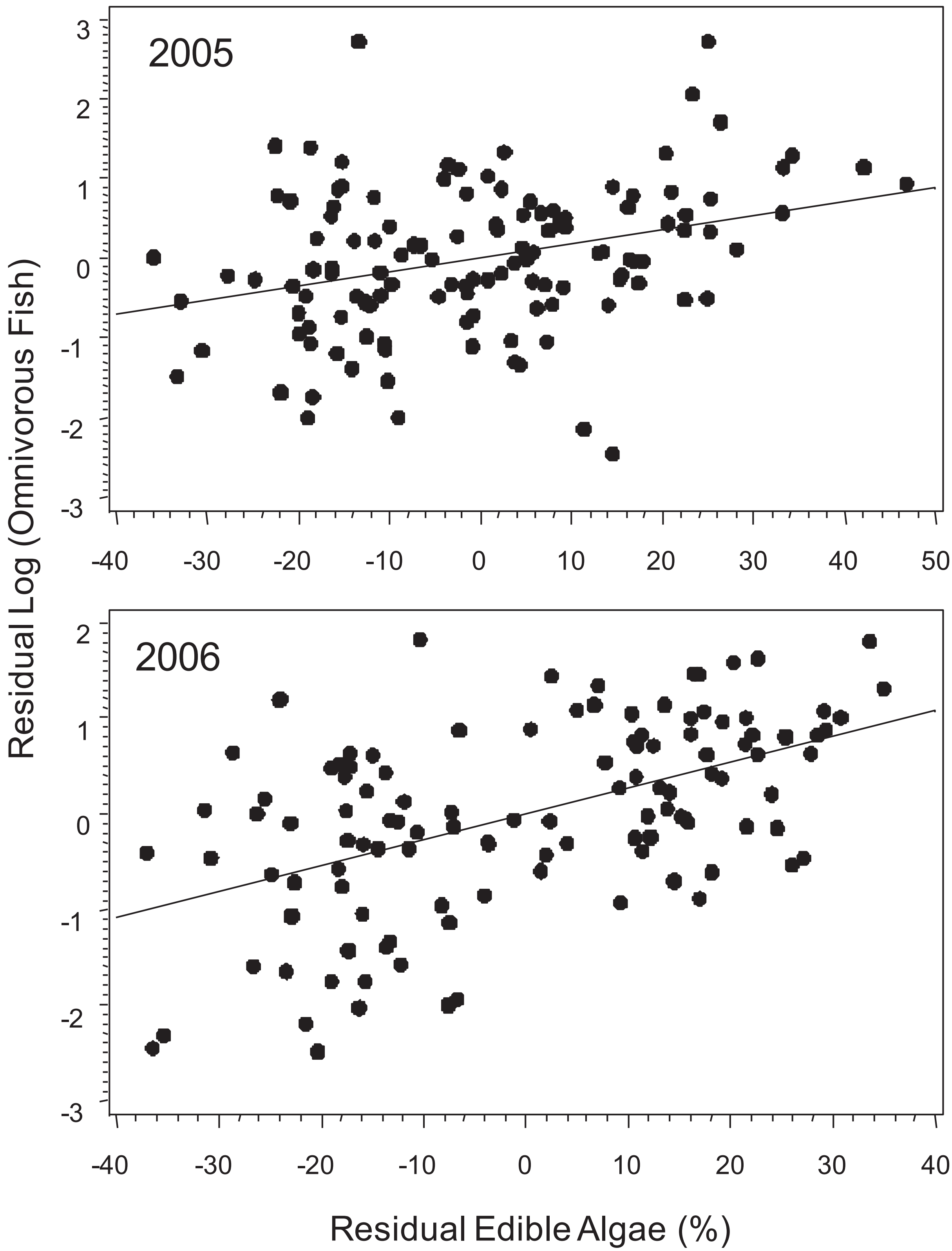


Figure 12. Partial regression plot of omnivore density versus edible algae (%)

Residuals from a regression with hydroperiod (days) and stem density (#/m^2) are plotted to indicate the unique pattern attributable to tissue P.

We examined the two species of crayfish inhabiting Everglades wetlands in similar statistical models because past work demonstrated that their dynamics were controlled differently than fish (Dorn and Trexler 2007). In both 2005 and 2006, edible algae failed to explain a significant amount of variation (***Table 9***). Only depth-related hydrological parameters provided significant relationships and these were most apparent for the Everglades crayfish (*Procambarus alleni*). Dorn and Trexler (2007) provide data supporting the hypothesis that slough crayfish (*Procambarus fallax*) are more affected by the density of large aquatic predators than by physical parameters. Everglades crayfish are often the only crayfish in short hydroperiod marshes or in long hydroperiod wetlands that dried within the past year (Hendrix and Loftus 2000), but slough crayfish are the dominant (and only) species over much of the Greater Everglades ecosystem. Our statistical models fit the data better for both species in 2005 than in 2006 (***Table 9***), consistent with our data on periphyton but not other aquatic consumers.

Table 9. Regression results for analysis of crayfish density (log #/m^2) reported separately by year

N=126 in 2005 and N=117 in 2006

	Slough Crayfish				Everglades Crayfish			
Variable	**Estimate**	**Error**	**t**	**P**	**Estimate**	**Error**	**t**	**P**
2005	R^2_{adj} = 0.283				R^2_{adj} = 0.332			
Intercept	-0.65	0.21	-3.15	0.002	1.60	0.24	6.71	<0.0001
Edible Algae (%)	0.003	0.003	1.05	0.294	-0.005	0.003	-1.59	0.113
Hydroperiod (days)	0.005	0.001	5.29	<0.0001	-0.003	0.001	-3.12	0.002
Stem Density (log #/m^2)	0.08	0.04	1.99	0.049	-0.02	0.05	-0.54	0.592
Depth (cm)	0.001	0.003	0.27	0.785	-0.01	0.003	-3.14	0.002
Average Depth Past 60 Days (cm)	0.001	0.01	0.25	0.803	0.01	0.01	2.02	0.046
Average Depth Past 90 Days (cm)	-0.01	0.005	-2.40	0.018	-0.01	0.01	-1.27	0.208
2006	R^2_{adj} = 0.037				R^2_{adj} = 0.145			
Intercept	0.26	0.21	1.25	0.215	1.00	0.23	4.40	<0.0001
Edible Algae (%)	0.002	0.003	0.71	0.478	0.001	0.003	0.43	0.671
Hydroperiod (days)	0.001	0.001	1.31	0.193	0.000	0.001	-0.09	0.932
Stem Density (log #/m^2)	0.02	0.04	0.48	0.631	-0.03	0.05	-0.70	0.486
Depth (cm)	-0.01	0.003	-2.97	0.004	-0.01	0.003	-3.91	<0.0001
Average Depth Past 60 Days (cm)	0.001	0.003	0.23	0.820	-0.001	0.003	-0.39	0.699
Average Depth Past 90 Days (cm)	-0.004	0.004	-1.16	0.248	0.0001	0.004	0.000	0.997

Conclusions

We report the results of analyses of three independent datasets collected in the wet season of 2005 and 2006. Though the strength of some relationships varied across years and some of the weaker relationships were only significant in one or two of the datasets, we observed consistent results regarding the linkage of P in periphyton tissues to changing periphyton mat, with implications for standing crops of herbivores and omnivores throughout the Great Everglades ecosystem. Our analysis of REMAP data further supported mat algal composition as the key driver of consumer dynamics, rather than stoichiometric relationships per se. To our knowledge, no other studies of food web function in the Everglades have been conducted at such a comprehensive scale. Past work indicates that the spatial separation of our sampling units was appropriate to treat each as an independent sample and appropriate for inclusion in regression analyses of the type we report. That noted spatial autocorrelation is pervasive in the distribution of nutrient enrichment in the Everglades because the delivery of P is made through water flowing into the ecosystem from essentially point sources of canals and water control structures. The most marked and longest sustained enrichment is in the northern portion of the ecosystem, with lower levels of P in the southern Everglades, particularly in Taylor Slough and the oligohaline zone. We demonstrated that this oligotrophy has implications for the quality of periphyton as a food source, a conclusion that is consistent with past experimental studies. Adding P changes the periphyton community to favor more edible algae leading to higher standing stocks of consumers. Interestingly, consumer density was not highly correlated with stoichiometric patterns unless algal relative abundance was excluded from the model. This suggests that algal defenses trump stoichiometric considerations in limiting consumer productivity.

The hypothesis cluster for wading bird nesting in relation to the aquatic fauna forage base is based on a causal linkage of aquatic consumer dynamics and hydrological and nutrient management. Our study provides strong support for these causal linkages. Both measures of algal edibility tied to nutrient gradients and hydrological variation remained in statistical models of aquatic consumer dynamics. These environmental drivers collaborated in determining aquatic animal production that ultimately feeds wading birds and alligators. We already have data from two more years of aquatic consumer dynamics and will soon have complementary algal data. These data can be used to test the hypotheses reported here and contribute to models that can be used to set targets for CERP restoration and for evaluation efforts required in the process of choosing steps forward in the restoration process. Finally, we observed some differences in model fits between years, and some weaker relations were only noted as significant in one of the two study years. We look forward to adding more years to the dataset to better identify robust relationships, and those that are dependent on climatic conditions affecting the region and ecosystem as a whole.

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Predator-Prey Interactions of Wading Birds and Aquatic Fauna Forage Base

Prepared by Dale E. Gawlik, Peter C. Frederick, Bryan Botson, Garth Herring, James M. Beerens, Joel C. Trexler and Mark I. Cook, towards validation of the Wading Bird Nesting in Relation to Aquatic Fauna Forage Base Hypothesis Cluster

Introduction

One of the key sets of hypotheses underlying Everglades restoration is collectively termed the trophic hypothesis. Coarsely summarized, it states the collapse of wading bird nesting colonies in the southern Everglades is attributed to declines in population densities and seasonal concentrations of marsh fishes and other aquatic prey organisms (RECOVER 2006, Trexler and Goss 2009). Restoration of natural hydrologic conditions is predicted to re-establish distributions of prey densities and concentrations across the landscape that in turn will support the return of large, successful wading bird nesting colonies to the southern Everglades. Four hypotheses that fall under this overarching hypothesis are listed below and are described in more detail in the Monitoring and Assessment Plan (MAP), Part 2 2006 Assessment Strategy for the MAP (RECOVER 2006).

1. Aquatic Fauna Wet Season Prey Population

The wet season density, size structure and relative abundance of marsh fishes and other aquatic wading bird prey are directly related to the time since the last dry down and the length of time the marsh was dry. Aquatic prey populations are further affected by salinity in coastal ecotones and by site nutrient status. Responses are non-linear and species specific.

Resumption of natural volume, timing and distribution of freshwater flow will restore historical hydroperiods and salinity distributions to the southern Everglades. These changes are expected to increase wet season density and size structure of wading bird prey in the southern Everglades within a four to seven year timeframe.

2. Aquatic Fauna Dry Season Prey Concentration

The concentration of marsh fishes and other wading bird prey into high density patches where wading birds can feed effectively is controlled by the rate of dry season water level recession and local topography/habitat heterogeneity.

Resumption of natural volume, timing and distribution of freshwater flow to the Everglades is expected to restore historical patterns of water level recession and drying edges, which will concentrate wading bird prey into a succession of high density patches throughout the dry season in the remaining ridge and slough, marl prairie and coastal landscapes.

3. Wading Bird Nesting Colony Location, Size and Timing

The collapse of wading bird nesting colonies in the tributary headwaters and southern mainland of the Everglades, the abandonment of roseate spoonbill nesting colonies in islands of northeastern Florida Bay, and delay in the annual initiation of wood stork nesting have been caused by the following:

- Decreased population densities of marsh fishes and other aquatic prey organisms in the southern Everglades, as described under Hypothesis 1
- A shift in the location and timing of seasonal concentrations of marsh fishes and other aquatic prey organisms, as described under Hypothesis 2
- Reduced shallow water foraging options for wading birds along elevation gradients

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 roseate spoonbill nesting colonies in northeastern Florida Bay. This restoration is also expected to increase numbers and success of nesting wading birds, wood storks and roseate spoonbills and to cause wood storks to initiate nesting no later than January in most years.

4. Wading Bird Super Colony Formation

Unusually large aggregations of nesting wading birds, referred to as super colonies, consisting of mostly white ibis formed in the pre-drainage system in response to the effects of extreme, natural patterns of drought prior to colony formation.

Resumption of natural patterns of volume, timing and distribution of flow to the southern Everglades, in combination with interannual variation in rainfall, will restore natural multi-year wet and dry cycles as they would have occurred prior to drainage of the southern Everglades. This is expected to stimulate pulses of secondary productivity that will likely involve Everglades crayfish populations. The pulses in secondary production are expected to result in an increase in the return frequency and size of ibis-dominated super colonies in the tributary headwaters of the Shark River and other Gulf of Mexico mangrove estuaries at a frequency of two or more events per decade.

Existing Evidence for the Trophic Hypothesis

The trophic hypothesis rests on the premise that 1) food is limiting nesting populations of wading birds and 2) hydrology controls the production and availability of aquatic prey animals. This premise has been articulated as the prey availability hypothesis (Gawlik 2002, Frederick et al. 2009). Indirect evidence to suggesting nesting wading birds in the Everglades are limited by food is considerable (1 above; Kahl 1964, Kushlan et al. 1986, Bancroft et al. 1990, Frederick and Collopy 1989, Frederick and Spalding 1994, Frederick and Ogden 2001). These studies did not measure prey availability directly, but rather examined correlations between a wading bird nesting parameter and a hydrologic pattern, assuming that hydrologic patterns control either prey production or availability. The strongest relationships are 1) birds often fail to nest or abandon nests following a reversal in the seasonal drying trend and 2) birds fail to nest if water levels are extremely high (Frederick 1995). Frederick and Ogden (2001) found a correlation between wading bird nest effort and hydrologic patterns, which they also interpreted as being controlled by food availability, however, the hydrologic pattern that produced the highest nest numbers was a drought, which reduces fish populations to their lowest levels (Loftus and Eklund 1994, Trexler et al. 2002). It was suggested that crayfish rather than fish were the primary food items following droughts. Other studies have shown an indirect link between hydrologic patterns and foraging patterns, again assuming hydrologic patterns control either prey production or availability

(Kushlan 1976a, Strong et al. 1997, Hoffman et al. 1994, Bancroft et al. 2002, Russell et al. 2002).

We are aware of thirteen South Florida wading bird field studies (Kahl 1964, Kushlan 1976a, Kushlan 1977, Ogden et al. 1976, Powell 1983, Surdick 1998, Gawlik 2002, Cook and Herring 2007, Garrett 2007, Gawlik and Botson 2007, Harris 2007, Beerens 2008, Lantz 2008) and one lab study (Adams and Frederick 2008) that measured directly some aspect of prey availability, prey density, or vulnerability to capture. Most of these studies related prey density to changes in wading bird foraging patterns. To our knowledge, only Kahl (1964) with the wood stork, Kushlan (1977) with the white ibis, Powell (1983) with the great white heron, Cook and Herring (2007) with the white ibis, and Herring (2008) with white ibis and great egret, demonstrated a direct relationship between prey density and wading bird nesting in the field.

Recent studies have shown that indicator species for the Everglades restoration, such as the white ibis and great egret (Frederick et al. 2009), differ in their sensitivity to hydrologic conditions (Gawlik 2002, Herring 2008) and they exhibit different population trends (Crozier and Gawlik 2003). The great egret has increased rather steadily over the period of record whereas the white ibis is considerably less abundant now than it was historically (Crozier and Gawlik 2003), although nesting effort has improved in recent years. Evidence is growing that the population responses can be linked to how constrained a species is in its choice of habitats (Gawlik 2002, Beerens 2008). Searchers like the white ibis are more limited in their selection of foraging sites and tend to select high quality patches and abandon them quickly, whereas exploiters like the great egret are opportunistic and tend to minimize searching effort by staying at foraging areas longer until prey densities are very low (Gawlik 2002). An understanding of species-specific responses to prey availability is necessary if the intent is to evaluate hydrologic changes on population-level patterns of individual wading bird species. We rely heavily in this report on the information gained by comparing a searcher and exploiter species in years with differing amounts of food.

The evidence for hydrologic control on aquatic prey production (2 above) is strong (Loftus and Eklund 1994, Trexler et al. 2002, DeAngelis et al. 2005, Ruetz et al. 2005, Trexler and Goss 2009). Populations of different prey fish species tend to peak after 1 to 8 years of continually flooded marsh (Trexler et al. 2002, Trexler et al. 2005), with the fish and macroinvertebrate community showing at least three different periods of responses that correspond to life history strategy (Trexler and Goss 2009). Hydrologic patterns also affect the populations of aquatic predators, which influence the density of small fishes (Kushlan 1976b, Chick et al. 2004) as well as by affecting competitive interactions (DeAngelis et al. 2005). Increased nutrient levels increase fish production (Turner et al. 1999), but increased nutrients also increase plant density, which can actually reduce prey availability (Crozier and Gawlik 2002). How wet season production of aquatic prey species is related to small-scale concentrations of prey a few months later in the dry season is poorly understood.

Although contaminants like phosphorus affect wading birds by affecting the production of their aquatic prey or altering the vegetation, mercury is a contaminant that can cause direct negative effects on the health or behavior of wading birds. The mercury hypothesis states mercury contamination effects on avian reproduction patterns are large enough to affect population responses to hydrological restoration. If this notion is supported, the mercury hypothesis might yield a partial explanation for the association between increased nesting and decreased mercury, and so reduce uncertainty about the influence of hydrology on wading bird nesting responses. If, for example, mercury at ambient Everglades levels have little effect on reproduction, then there is a considerable amount of unexplained variation in our hydrological/nesting models. Because there was little underlying data from which to test this hypothesis, a project was initiated as part of the MAP to elucidate key uncertainties in the ability to predict responses by wading birds to hydrological restoration.

Approach and Datasets Used to Evaluate the Trophic Hypothesis

Despite the abundance of evidence for various pieces of the trophic hypothesis (Trexler and Goss 2009), no comprehensive set of studies has shown a simultaneous relationship among hydrologic patterns, aquatic prey production, aquatic prey availability, and wading bird nesting (i.e., the prey availability hypothesis). This is not surprising given the large spatial extent of the study area and the multiple trophic levels involved. The MAP, however, was designed as a comprehensive and coordinated effort to quantify the pathway linking the elements of the trophic hypothesis and to guide the restoration process. This report examines monitoring data from 2005 to 2007 for the MAP studies of aquatic prey production, seasonal concentrations of aquatic prey, and wading bird nesting. Data on hydrologic patterns come from a powerful new monitoring tool called the Everglades Depth Estimation Network (EDEN; Telis 2006). This new real-time network of water level gages produces a daily water depth surface for most of the Everglades ecosystem. This report also draws ancillary research studies conducted by principal investigators on related questions to aid in the interpretation of MAP data. Four such studies are effects of mercury on wading birds (Adams and Frederick 2008), effects of food levels on wading bird habitat selection (Beerens 2008), effects of food supplementation on white ibis chick growth and survival Cook and Herring (2007), and effects of a food levels on wading bird physiology and reproduction (Herring 2008).

Our approach for evaluating the trophic hypothesis was to examine the relationships between each successive component of the prey availability hypothesis. First, we quantified the hydrologic patterns of the years of study using the EDEN hydrologic network. The study years of 2005 to 2008 provided contrasting hydrological conditions and levels of prey availability thereby providing us with a natural framework for examining the trophic hypothesis. Second, we qualitatively compared annual measures of prey availability with wading bird nesting patterns just to see if there was concordance at a landscape level. Because only four years of systemwide field data were available from the MAP, and wading bird nesting totals only produce one estimate per year, we compared visual plots of prey availability and wading bird nesting. This analysis did not identify a pathway by which food could affect wading birds but it did provide a check on the notion that food and nesting are related, and it is more direct than the previous comparisons between the endpoints of the trophic hypothesis, which are the hydrologic driver and a wading bird reproductive response. This approach also allowed us to characterize the years of study in terms of levels of prey availability and nesting.

The link between the hydrologic conditions and prey concentrations came solely from MAP studies. We conducted a statistical analysis of the relative effects of wet season prey production and environmental variables factors that could affect seasonal prey concentrations. We had little basis from which to form preconceived ideas of the shape of the relationship between prey concentrations and prey production, hydrology, and other abiotic variables. Thus, we used a model building approach that estimated the parameters, rather than to conduct pair wise tests of a null hypothesis.

Quantifying the relationship between prey concentrations and wading bird populations was more complicated. Although a relationship between the endpoints of the hypothesis has long been known, little information was available on the mechanistic pathway by which food affects reproduction. Documentation of this pathway would be strong support for the trophic hypothesis, and would provide a mechanistic basis for the previous observations of the endpoints of the hypothesis being correlated. The degree to which the pathway involved a behavioral or physiologic response to food was unknown prior to this study.

We started by comparing the behavioral response of a searcher species (white ibis) and an exploiter species (great egret) in terms of habitat selection. If the searcher species is more food limited, as their population trends would suggest, then we could expect to find greater constraints on habitat selection for the searcher. Again, we used a model building approach that estimated the parameters. We followed radio tagged great egrets and white ibises and developed resource selection models to identify the important landscape variables and associated temporal scales that influence foraging habitat selection.

Next, we measured the physiological condition and productivity of adult egrets and ibises in response to landscape-level prey availability and differences in foraging strategies between the species. We measured physiological markers and radio tracked for the remainder of the breeding season the adult great egrets and white ibises captured in the pre-breeding season and referred to in the previous section. We recorded clutch size and then monitored nest success and fledging rates, and sampled chicks to determine their physiological condition and growth rates for both radio tagged and random nests. We focused on corticosterone and stress protein physiological markers, which generally increase during periods of increased stress.

The final study is a report on the role that mercury might have played in affecting wading bird nesting patterns historically. If mercury levels were affecting wading birds, their population response might not match predictions from the trophic hypothesis. This study used an experimental approach whereby young white ibises were collected from a colony in the Everglades and placed into mercury exposure groups in a large free-flight aviary in Gainesville, Florida.

A New Evaluation of the Trophic Hypothesis

1. Everglades Fauna Concentrations and Production during Wading Bird Breeding Season

To quantify the availability of prey for wading birds across the Everglades landscape as part of the MAP, we sampled fish and macroinvertebrates with a one-square meter (m^2) throw trap during the dry seasons from December to May. We also sampled at sites where large flocks of wading birds (more than 30) were foraging to compare used versus available sites. See Gawlik and Botson (2007) for a detailed description terminology, sampling design and sampling methodology.

Water levels at the start of the 2005 dry season experienced a steady and rapid recession through the end of February. A series of rain events in early March reversed the seasonal water recession (***Figure 1***), inhibiting the concentration of prey and producing widespread nest failure by most wading bird species (Cook and Call 2005). These reversals initiated a longer than usual period of rising water levels in which prey were produced and subsequently concentrated in the 2006 drydown. Hydrologic conditions in the 2006 dry season were close to optimal for wading bird nesting; meaning water levels were high at the start of the dry season, thereafter receding uninterrupted over the remainder of the dry season (Gawlik 2002).

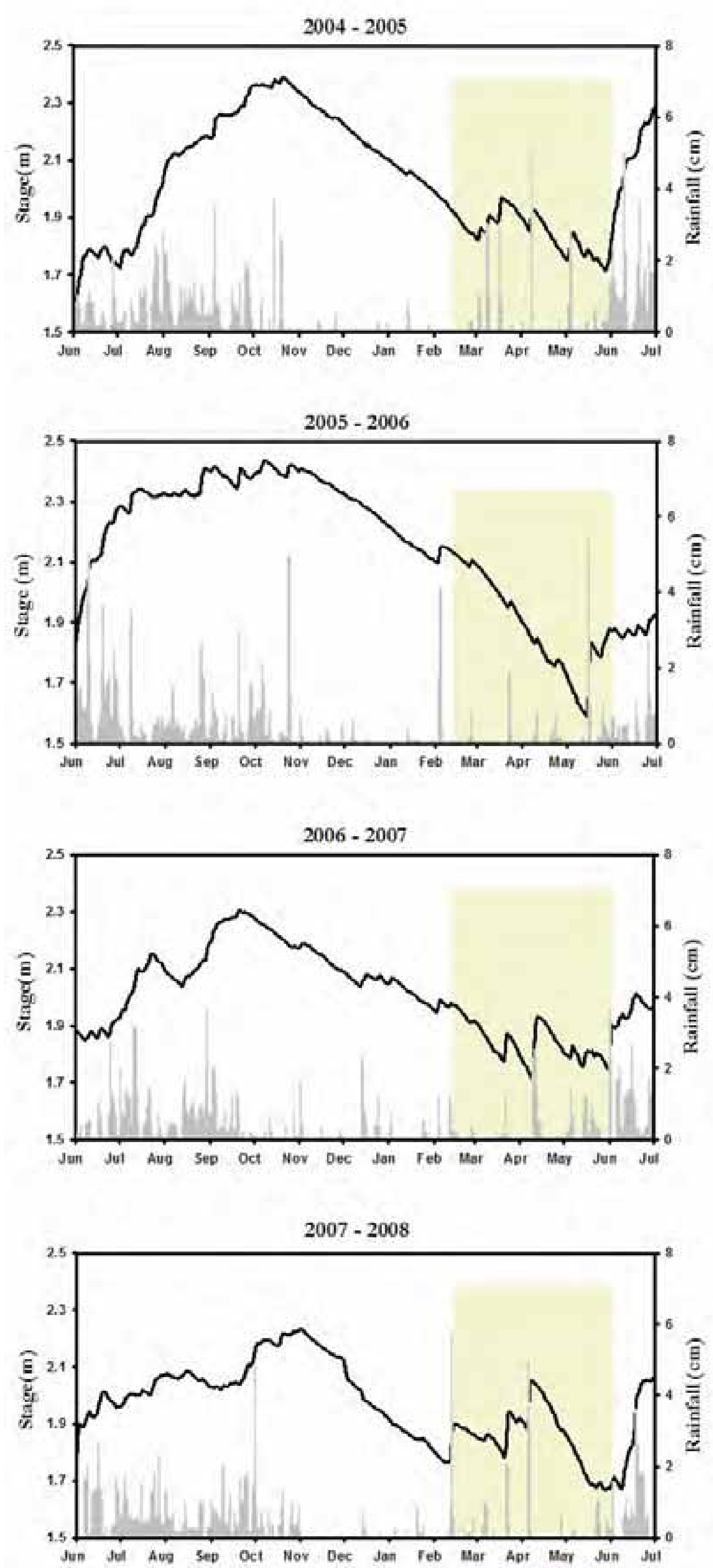

Figure 1. Mean rainfall and stage level throughout the Everglades from June 2004 to July 2005, June 2005 to July 2006, June 2006 to July 2007 and June 2007 to June 2008

Shaded areas represent approximate wading bird nesting season. Stage values represent the mean of 23 gages (site 7, site 8t, site 9, 2A site 17, 2A site 19, site 62, 3ANE, site 64, site 65, site 71, SRS1, NE4, NP202, NP203, P33, CR2, A13, NP206, NP62, P36, P34, TMC, NP205). Rainfall represents the mean of 13 gages (NE4, NP202, NP203, P33, CR2, A13, NP206, NP62, P36, P34, TMC, NP205, BCA18, BCA19, BCA20, MDTS, S174, S20).

Furthermore, the late onset of the wet season in 2006 continued to provide ample foraging patches for fledging birds late in the season. This steady, prolonged recession in 2006 fostered the highest dry season prey densities (***Figure 2***) and the most wading bird nests of the four years. The long recession period, partly due to an unusually dry wet season in 2006, produced abnormally low water levels during the 2006 wet season and 2007 dry season, at least in the northern Everglades (***Figure 1***).

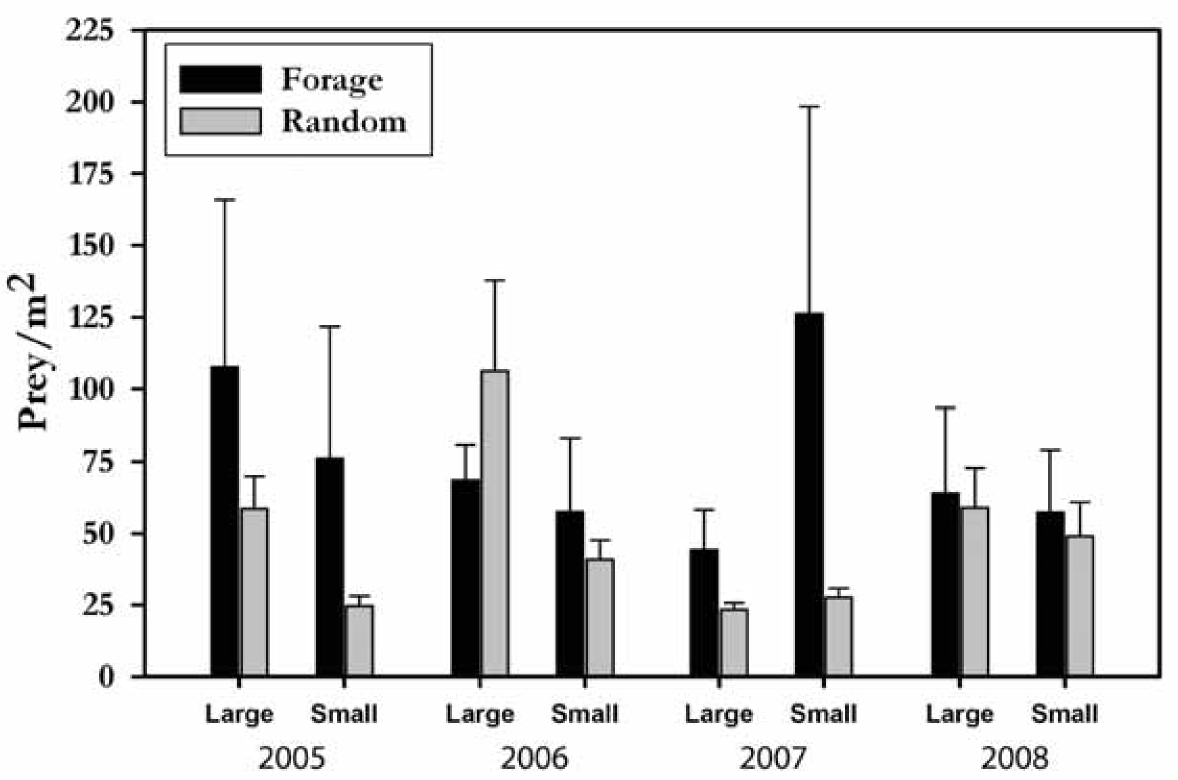

Figure 2. Mean density of all prey items in 1-m^2 throw traps at random sites and wading bird foraging sites throughout the Everglades during the 2005, 2006, 2007 and 2008 dry seasons

Error bars are ± 1 standard error (SE).

Like 2006, the 2007 dry season had fewer reversals than in 2005 and 2008, but mean prey density and biomass at random sites declined significantly (***Table 1**, **Figure 2***), suggesting the low water levels of the preceding wet season constrained the growth and reproduction of prey populations leading into the 2007 dry season. The drought conditions of 2007 extended into the early dry season of 2008, which began with extremely low water levels, especially in the southern Everglades (***Figure 1***). Starting in mid-February and extending through April, a series of rainfall events drastically increased water levels systemwide (***Figure 1***). Overall, hydrologic conditions and wading bird nesting effort were again poor in 2008 (***Figure 3***).

Table 1. Mean prey density and mean biomass found within 1-m^2 throw traps for random and foraging sites throughout the Florida Everglades in the dry seasons of 2005, 2006, 2007 and 2008

Data shown as the mean ± 1 SE.

Sample Type	Mean Prey Density (range)				Mean Prey Biomass in grams			
	2005	2006	2007	2008	2005	2006	2007	2008
Random	81 ± 14 (0-798)	142 ± 36 (1-3198)	51 ± 5 (0-633)	108 ± 25 (0-3169)	32 ± 5	48 ± 12	8 ± 1	18 ± 3
Forage	184 ± 98 (1-4124)	126 ± 34 (4-832)	170 ± 84 (1-3590)	121 ± 50 (0-1197)	25 ± 12	31 ± 8	12 ± 3	19 ± 6

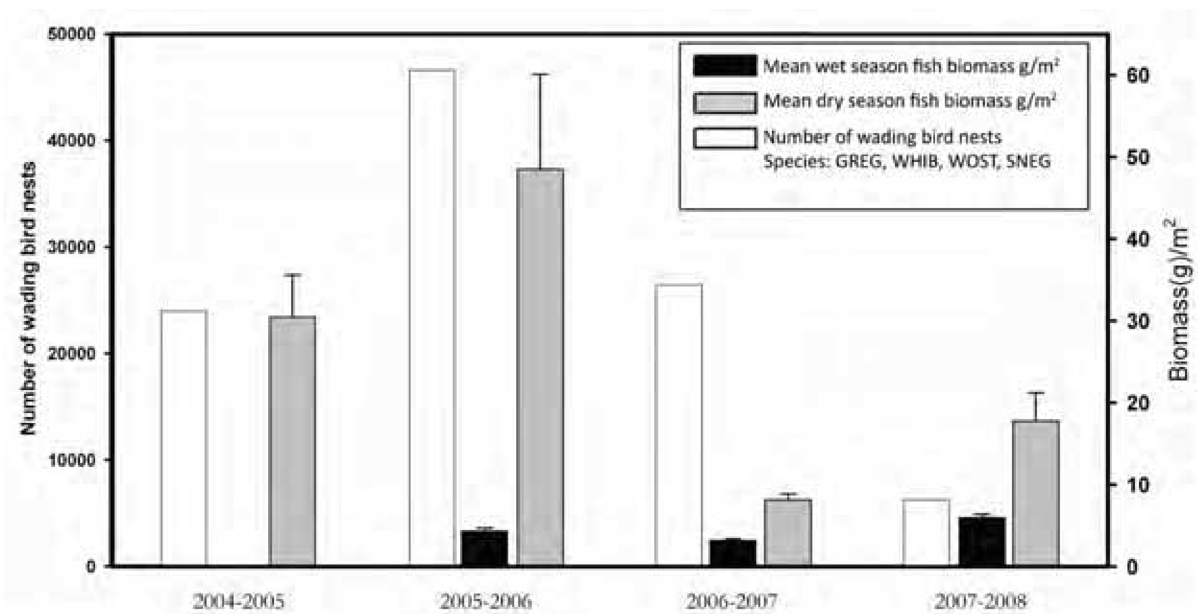

Figure 3. Mean dry season prey biomass (grams per m^2), mean wet season prey biomass grams perm^2), and number of wading bird nests in 2005, 2006, 2007 and 2008

Means includes all prey items captures in LSUs where wet and dry season samples were collected in the same year. No wet season data was available for 2004. Wading bird nests represent the pooled nest numbers for the great egret, white ibis, wood stork and snowy egret for Everglades National Park and the Water Conservation Areas (WCAs).

During 2005 and 2007, two years with poor hydrology and low wading bird nesting effort, birds were selecting sites with greater prey densities than what was available in the landscape, suggesting that prey availability was low overall, but that birds were able to find some sites with high prey densities. In 2006, prey densities at foraging sites were similar to what they were in 2005 and 2007, however, the key distinction in 2006 is that there was no discernable difference in prey density between random and foraging sites, suggesting that high quality foraging patches were simply more common in the landscape. Prey densities in 2008 were similar between random and foraging sites but they were much lower than in 2006, suggesting that landscape prey availability was so low that wading birds were not able to locate relatively rare sites with high prey densities.

During the first two years of this study, it was apparent that one difference between the fish community in drying pools and the fish community in deeper water is that the former is dominated by large fish (greater than 2 centimeters (cm) total length) rather than small fish. This novel pattern is opposite of what is typically seen when sampling in deeper water (Loftus and Eklund 1994, Trexler et al. 2002). It was interesting that this pattern did not persist in 2007 or 2008 (***Table 1**, **Figure 2***). During the drought, the number of small fish declined only slightly whereas the large fish declined greatly. This suggests that the primary impact of the drought was to reduce the density of large fish. It also suggests that the mechanism that produced a greater proportion of large fish in the drying pools, as compared to the wet season, did not occur in 2007 and 2008, as it did in the previous years. The proportion of large prey in the landscape remained low in 2008, indicating that prey populations had not yet fully recovered following last year's drought. The lack of large prey items in the landscape likely dramatically reduces the quality of prey patches, particularly for larger birds like the wood stork, which did not nest successfully in 2007 or 2008.

Experimental studies with white ibis nestlings suggested that food limited growth and survival for this species in 2007 but not in 2006 (Cook and Herring 2007). Physiological data of adults and chicks also suggested that birds were in poorer body condition in 2007 than in 2006 (Herring 2008). Cumulatively, this evidence led to our characterization of 2006 as a year with good habitat conditions for wading birds and 2007 as a year with poor habitat conditions for wading birds.

<u>Wet Season versis Dry Season Prey Biomass</u>

Mean biomass pooled across all landscape sampling units (LSUs) of prey collected during the dry season tended be much higher than mean biomass of prey collected during the preceding wet season (***Figure 3***). The magnitude of the difference was especially evident from 2005 to 2006, when hydrological conditions were best for wading bird foraging. Dry season and wet season biomass increased from 2007 to 2008, but dry season biomass was still significantly lower than in 2006. The increase in wet season biomass may indicate an overall increase in production; however, the numerous reversals during the 2008 dry season likely reduced prey availability. Some of the longer hydroperiod LSUs did not dry during 2006 and thus had more time for fish growth and reproduction prior to the 2007 dry season. This was the case in LSU 7 in the Arthur R. Marshall Loxahatchee National Wildlife Refuge (Loxahatchee NWR), which not only was the one area that had high prey production and concentrations during the 2006 to 2007 drought year but also had the largest numbers of nesting wading birds in 2007 (***Figure 3***). Longer hydroperiod parts of the southern Everglades may have been spared of a severe dry down and water levels appeared to rebound more rapidly than in the northern Everglades. This extra time for growth and reproduction fostered the high wet and dry season prey biomass at LSU 27 in 2008 relative to the rest of the system (***Figure 4***).

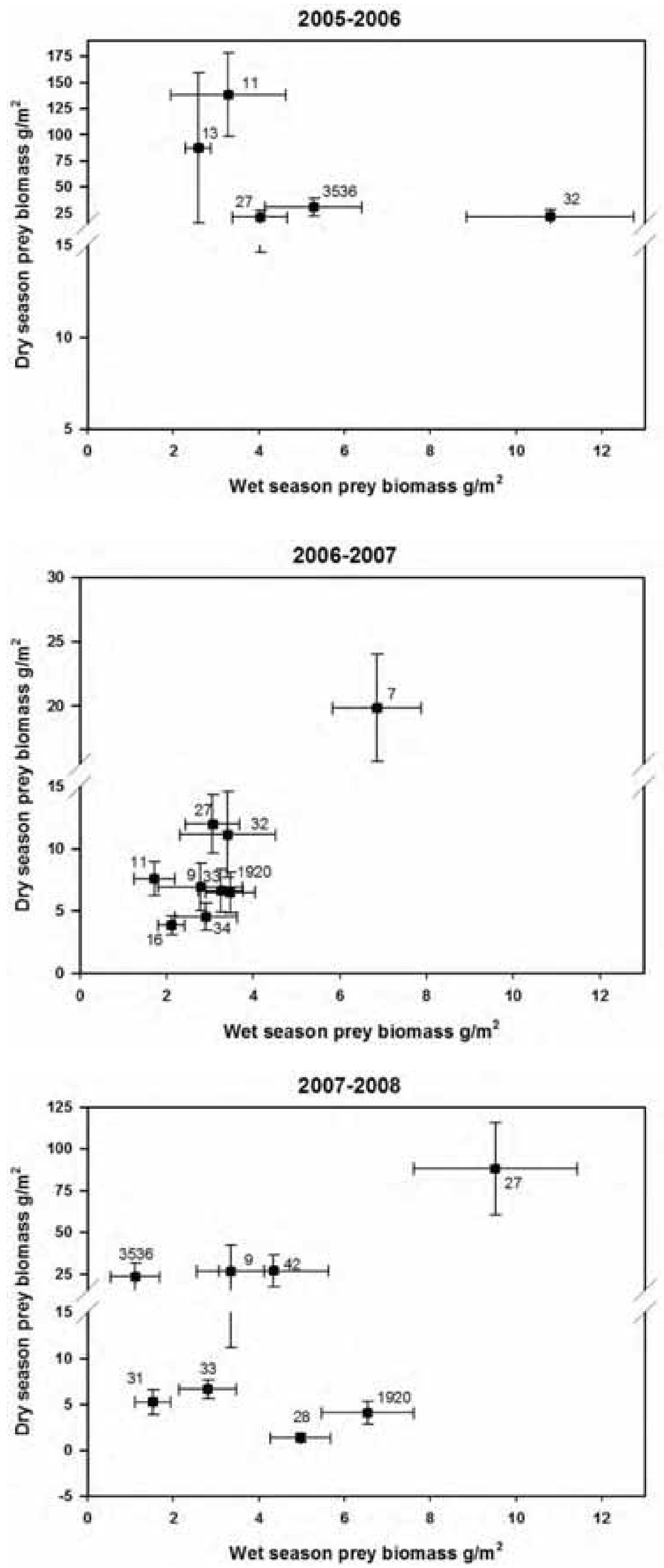

Figure 4. Mean wet season versus dry season biomass (grams per m^2) at LSUs sampled in the wet and dry season from 2005 through 2008

Wet season values represent prey biomass from the wet season that preceded the reported dry season.

Model Selection

We quantified relationships between wading bird prey concentrations and the parameters wet season biomass from the preceding wet season (WETBIO), recession rate (REC), number of days since last drydown (DSD), and transect depth coefficient of variation (representing microtopographical variation; DEPTHCV) using a generalized linear model computed with the procedure PROC MIXED in SAS version 9.2. Recession rate was calculated using the EDEN surface as the mean drying rate during the two weeks preceding a prey sampling event. We included landscape sampling unit (LSU) and year (YEAR) as class variables in every model to account for spatial and temporal differences in prey biomass. In general linear models, the parameter estimate for one level of a class variable is set to 0, thereby providing a relative comparison for parameter estimates at other levels. The reference level for YEAR and LSU were 2008 and 9, respectively. We tested for colinearity among explanatory variables via a correlation analysis and calculated Moran I statistics using the VARIOGRAM procedure in SAS software version 9.2 to test for spatial autocorrelation. We used the information theoretic approach to investigate competing models (Burnham and Anderson 2002) and to identify variables of importance. We developed a priori candidate models based on relevant literature and our current understanding of factors the affect prey concentrations. To identify the most parsimonious set of a priori models we used Akaike's Information Criterion for small sample sizes (AICc). The ΔAICi values were used to determine separation between the best model and the other candidate models in the set. At this point, we eliminated all models with a ΔAICi greater than seven because they had essentially no empirical support (Burnham and Anderson 2002). We then calculated model probabilities (wi) to further evaluate the support for the top set of models.

To assess the relative importance of each predictor variable in the candidate set, we summed Akaike weights (wi) for each model containing a variable of interest. Additionally, we calculated model-averaged parameter estimates, which gauge how well a parameter explains variation in the response variable. To account for model selection uncertainty, we calculated the unconditional standard error of the parameter estimates.

Our results showed that the model with the most support for explaining variation in prey biomass included the parameters wet season biomass and recession rate (***Table 2***). This model is over four times more likely than the second best model, which also included the parameters wet season biomass and recession rate, as well as the term days since last drydown. All five top models included the variable recession. Wet season biomass was included in the top two models and the fourth best model, further supporting the strength of the top model. Summed Akaike weights (w_i) demonstrate the importance of recession rate and wet season biomass relative to the other continuous parameters (***Table 3***). There is considerably less support for day since drydown, and almost no support for variation in microtopography at a site (DEPTHCV) in explaining variation in prey biomass.

Table 2. Akaike's Information Criterion model selection for prey concentration biomass
All candidate models within seven AICc values from the top model are presented ($\Delta i < 7$).

Hypothesis	K	Δ_i	w_i
LSU + YEAR + WETBIO + REC	6	0	0.65263
LSU + YEAR + WETBIO + REC + DSD	7	2.951	0.14923
LSU + YEAR + REC	5	3.62719	0.10642
LSU + YEAR + WETBIO + REC + DEPTHCV	7	4.4697	0.06984
LSU + YEAR + REC + DSD	6	6.79117	0.02188

Table 3. Model-averaged parameter estimates, unconditional SE of model-averaged parameter estimates, and summed Akaike weights (wi) from all candidate models of prey concentration biomass with $\Delta i < 7$

Parameter	N	Estimate	SE	w_i
Intercept	5	47.08781177	27.0604	1
LSU				
11	5	57.82301373	34.9844	1
13	5	42.31752767	44.3177	1
15	5	-8.654088253	77.8899	1
16	5	-28.81127145	41.0439	1
1920	5	-36.20405553	36.9591	1
27	5	-1.845039397	31.4924	1
28	5	-62.57550259	50.5602	1
2930	5	46.00688663	56.7147	1
31	5	-48.14495192	40.3158	1
32	5	-20.01256057	34.3255	1
33	5	-31.67490633	30.2892	1
34	5	-11.05788274	39.9429	1
3536	5	-32.41494654	36.3486	1
42	5	-42.15907197	48.3581	1
7	5	3.496357893	42.5576	1
Year				
2006	5	8.818700394	24.123	1
2007	5	-39.73879502	20.2679	1
days since drydown	2	0.00921213	0.0271	0.17111
recession	5	9.228749298	17.3691	1
microtopographical variation	1	-0.00179481	0.0098	0.069837
wet season biomass	3	0.575732471	2.2556	0.8717

This analysis provides further evidence that recession of water levels during the dry season is an important mechanism for concentrating aquatic wading bird prey. It has long been known that as the annual drying front moves through the landscape from higher to lower elevations, it leaves behind isolated pools that trap fish and macroinvertebrates (Kushlan 1976a, Gawlik 2002, DeAngelis 1994). These high quality foraging patches are crucial to wading birds during the breeding season. Recession rate also affects how many prey are concentrated within any given drying patch, as illustrated by our analysis above.

A key assumption of the trophic hypothesis is prey production during the wet season also affects the concentration of fish during the dry season (Trexler and Goss 2009). Until this study, however, no direct evidence of this effect was available. The strong empirical support for wet season prey biomass in the top models illustrates the first quantitative link between dry season prey concentrations and prey production during the wet season. This is important evidence in support of the trophic hypothesis.

2. Habitat Selection of Two Wading Bird Species with Divergent Foraging Strategies

The previous section showed that wading bird prey production and rates of receding water levels strongly affected prey concentrations during the dry season, which is a parameter that corresponds, at least qualitatively, to annual wading bird nest numbers. As more years of data are collected that relationship can be quantified. As of yet we have not discussed how the same level of prey availability in the landscape can lead to different nesting patterns among wading bird species. Here we extend the trophic hypothesis to the idea that differential foraging constraints of bird species would interact with different levels of prey availability in ways that are consistent with observed differences in nesting trends among species. This section describes the relationship between habitat selection and prey availability.

Great egrets (n = 77) and white ibises (n = 127) were captured using a net gun and a modified flip trap (Herring et al. 2008) and radio tagged prior to the initiation of breeding in 2006 and 2007. Birds were captured in the central and northern Everglades and, thereafter, a subset were relocated three to four times a week from a plane (Beerens 2008). The habitat at foraging locations (n = 1,217) were compared with habitat parameters at random locations (n = 206,726) generated daily with ArcMap. Foraging and random locations representing available foraging sites were classified in ArcMap using five hydrological variables calculated from daily EDEN water depths, five vegetation classes, vegetation diversity and soil phosphorus. These variables represent processes that occur over a wide range of temporal scales to capture daily to decadal influences on habitat selection. These variables were used to develop resource selection functions (RSFs), which identified habitats selected by great egrets and white ibises while accounting for changing habitat availability. The cumulative hazard function for each species and year combination were plotted to illustrate the seasonal change in probability of use of the landscape. An increasing function is evidence that overall probability of use of the landscape is increasing.

<u>Great Egrets, 2006 (Good Habitat Conditions)</u>

Great egrets selected shallow water depths (***Figure 5***), low soil phosphorus levels, presence of cattail-dominated vegetation, and increased days since dry down, in order of descending importance based on the likelihood scores.

White Ibises 2006 (Good Habitat Conditions)

White ibises selected shallow water depths (***Figure 5***), low soil phosphorus levels, slow recession rates and short hydroperiods, respectively. White ibises preferred sites where water was receding slightly slower (0.254 cm per day ± 0.024 SE) than systemwide recession rates (0.317 cm per day ± .002 SE; ***Figure 6***).

Great Egrets 2007 (Poor Habitat Conditions)

Great egrets selected, in order of descending importance, rapid recession rates, low soil phosphorus levels, sites that were still wet when a reversal occurred, shallow water depths (***Figure 5***), cattail-dominated vegetation, open water/high impact urban dominated vegetation, and freshwater marsh and wet prairie dominated vegetation (Beerens 2008). In the food unlimited year, great egrets were more selective of optimal water depths, in contrast to 2007, when they selected a broader range of less optimal water depths (***Figure 7***). Selectivity for rapid recession rates dramatically increased in the food limited year (P < .0001; ***Figure 6***), receiving the highest likelihood score.

White Ibises 2007 (Poor Habitat Conditions)

White ibises selected, in order of descending importance, low soil phosphorus concentrations, rapid recession rates, shallow water depths, open water/high impact urban dominated vegetation, short hydroperiods and sites that were still wet when a reversal occurred. Like the great egret, white ibises were less selective of optimal water depths in the food limited year than in 2007 (***Figure 7***). Analysis of water depth at foraging locations indicated selection of foraging sites similar to depths selected by great egrets in the same year (***Figure 5***).

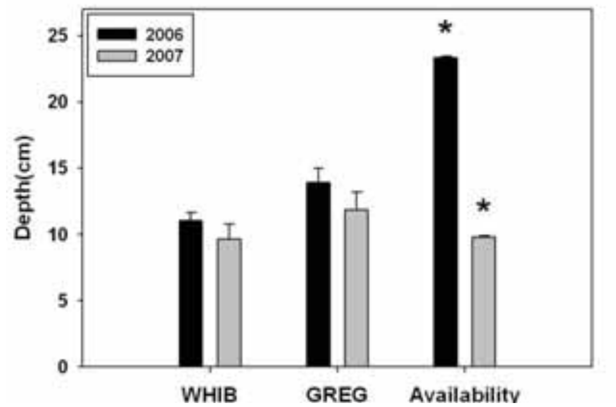

Figure 5. Yearly mean (± SE) EDEN depths, by great egret (GREG) and white ibis (WHIB) use and availability

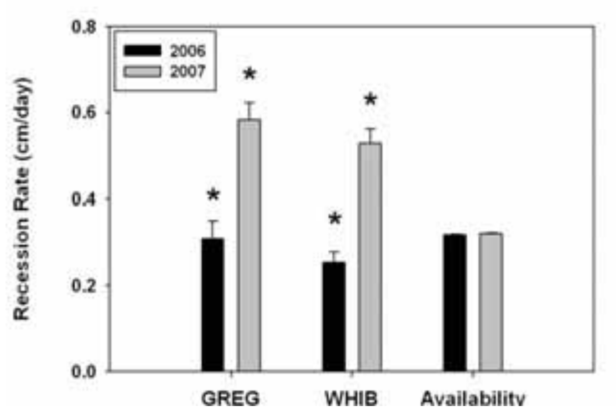

Figure 6. Yearly mean (± SE) recession rates at available and used sites by the great egret (GREG) and white ibis (WHIB)

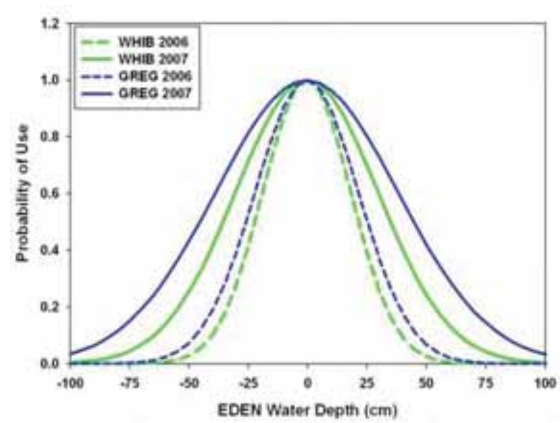

Figure 7. Relative probability of use for water depth with depth selectivity is highest for white ibises (WHIB) in 2006, followed by great egrets (GREG) in 2006, white ibises in 2007 and great egrets in 2007

Cumulative Hazard Functions

The cumulative hazard function indicated that the probability of birds using the landscape increased by Julian date in both years, suggesting that landscape features selected by great egrets and white ibises were increasingly more suitable as the season progressed (***Figure 8***). This pattern continued until the first major reversal (***Figure 8***), which caused a decrease in the probability of use. Great egrets in 2006 were an exception because probability of use simply stopped increasing after the reversal.

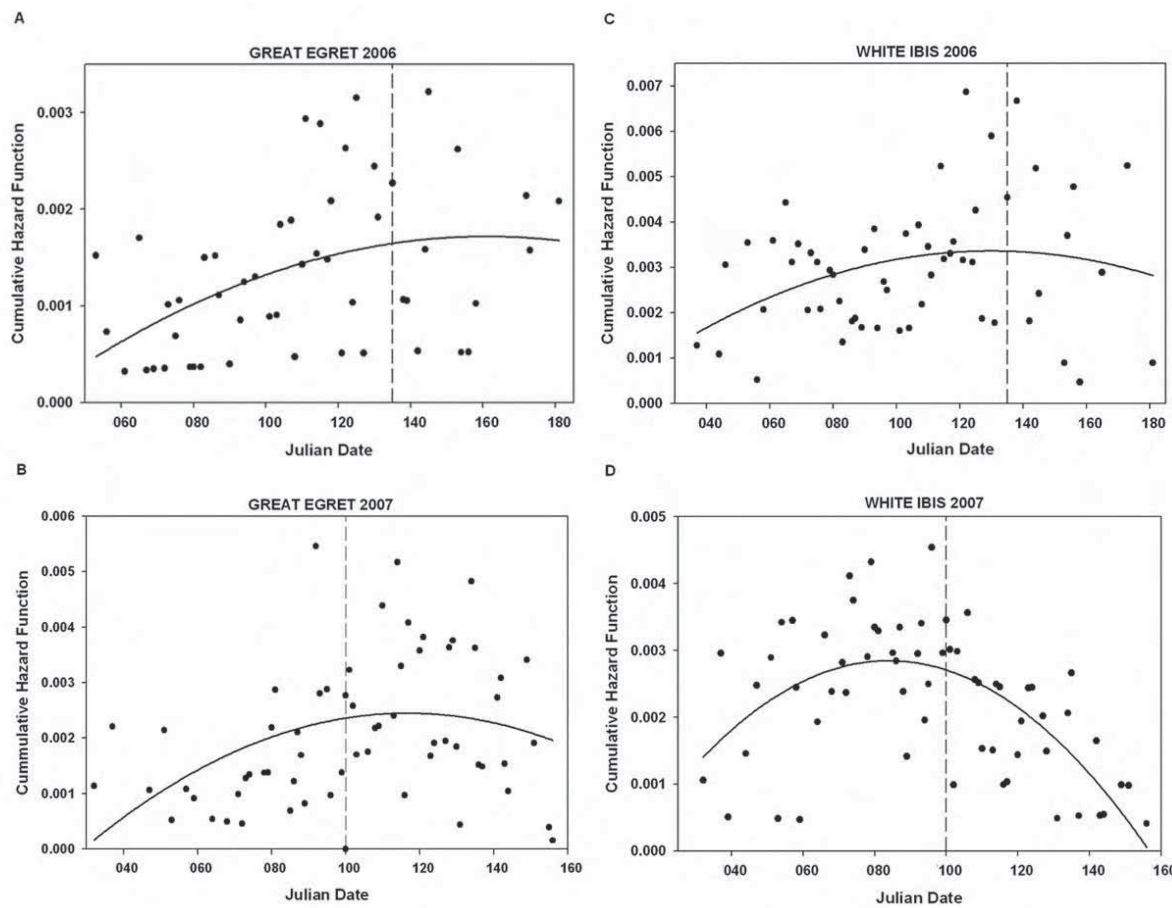

Figure 8. Cumulative hazard function (-log survivor function) depicting daily changes in the probability of use of the landscape for great egrets in 2006 (A) and 2007 (B) and white ibises in 2006 (C) and 2007 (D)

The dashed lines show the timing of a major reversal. An increasing cumulative hazard function is evidence for an increasing probability of use of the landscape.

As expected, adult great egrets and white ibises did not randomly select foraging locations, but rather displayed a clear preference for distinct habitat features. There appeared to be a relationship among resource availability, the temporal scale of the independent variable, and whether the response was similar or different between species. One set of independent variables differed strongly between years as a function of resource availability. Within this set, variables that change over short time scales, such as surface water dynamic (e.g. water depth, recession rates and site reversal) tended to produce a similar response by both species. In contrast, longer term processes involved in prey productivity such as days since dry down and hydroperiod produced a different response between species. Great egrets consistently selected longer days since drydown, whereas white ibises selected sites with a shorter time since dry down.

A second set of independent variables that change so slowly they were invariant over the two years of our study produced a consistent selection pattern between years, although it sometimes differed between species. For example, both species avoided areas of high soil phosphorus concentrations and white ibises consistently selected areas with short hydroperiods. The cumulative hazard functions suggest that in a good year, a reversal is less detrimental to great egret foraging than white ibis foraging, and supports the notion that the latter species is more constrained in its use of habitat (Gawlik 2002, Herring 2008). Modeling studies will be needed to confirm whether these differences in responses to food limitations are enough to explain the divergent population trends, but the evidence here is at least supportive of the direction of the differences between species.

3. Differential Sensitivity of Great Egret and White Ibis to Prey Availability and Hydrology

Tests of the trophic hypothesis in this section are whether we observe the predicted response to prey availability from contrasting species with regard to nesting success, and whether there is a physiological response in adults and chicks to different levels of prey availability. Evidence of such a response would provide the mechanism by which different trends in nesting among bird species could result from the same levels of landscape prey availability.

We measured physiological markers and radio tracked for the remainder of the breeding season the adult great egrets and white ibises captured in the pre-breeding season and described in the previous section. Radio tagged adults were located in nesting colonies, we recorded clutch size and then monitored nest success and fledging rates, and sampled chicks to determine their physiological condition and growth rates for both radio tagged and random nests. In this study, we focused on corticosterone and stress protein physiological markers, which generally increase during periods of increased stress, which can lead to reproductive failure.

We used a logistic exposure approach in a mixed model setting to model nest survival. We then used an information theoretic approach with PROC GENMOD in SAS software version 9.2 to build and rank competing models (Burnham and Anderson 2002). Competing models were developed based on a biologically meaningful understanding of the nesting responses of wading birds from previous studies (see Frederick and Collopy 1989). We evaluated the goodness-of-fit of the global models and verified that the models fit the data before proceeding with additional model analyses. Prior to model selection we visually examined the residuals of model variables to identify outliers or other patterns that might require transformation. Given the large differences in the time great egret and white ibis chicks spend at nests before fledging, we used a conservative fledging chick age of 15 days.

Pre-breeding adult physiology measures suggested that in a year with high prey densities both great egrets and white ibises were in good physiological condition (low levels of stress proteins and fecal corticosterone). During a year with poor habitat conditions (2007), ibis physiological condition declined compared to 2006; stress protein 60 and fecal corticosterone metabolites were higher during the 2007 pre-breeding period in ibis (***Figures 9** and **10***). However, great egret stress levels remained stable between the two years.

Nesting results (n=473 nests) suggest that white ibises modify their clutch size during years with poor habitat conditions (18 percent lower during 2007) in accordance with the life history traits of a long lived species, whereas great egrets maintained similar clutch sizes during years with poor and good habitat conditions. Model selection identified rain, water depth, Julian date, year, and prey biomass, respectively, as parameters that most influenced the daily survival rates (DSR) of white ibis nests (***Figure 11***). Great egret nest DSRs were most influenced by nest stage, region, Julian date, water depth, and the quadratic form of recession rate (***Figure 12***), respectively. Daily survival for both great egret and white ibis nests was higher during 2006 (DSR = 0.992 and 0.999, respectively) than 2007 (DSR = 0.981 and 0.979, respectively).

Great egret daily nest survival increased an additional one percent more than white ibis nest success with relatively small (+1 grams per m^2) changes in landscape prey biomass during the week before a nest survival estimate. This difference, when scaled up to an estimate of nest success over the entire nesting period, could account for up to a 20 percent difference between the two species, suggesting that white ibises may be more food limited than egrets. Great egrets may be either using resources more efficiently, or minimizing their energy expenditure when obtaining prey. Giving-up-densities at foraging sites suggest foraging costs increase less for egrets than ibis as water depths at foraging sites move further from an optimal depth (Gawlik 2002). This means great egrets can exploit a wider variety of habitat conditions (e.g. depth; Gawlik 2002, Beerens 2008) than do ibises for the same foraging costs. Indeed telemetry data show that great egrets foraged closer to colonies than did ibises despite large differences in weekly water levels in the area (Beerens 2008).

The strategy of the great egret is advantageous in years with poor habitat if birds are more likely to experience brood reduction rather than total nest failure, or if habitat conditions improve after the onset of nesting, allowing for the third egg to hatch and or chick to fledge (Lack 1947). This bet hedging approach appears to allow great egrets to produce successful nests in poor years and may maximize their reproductive efforts during average to above average years. The searcher strategy of white ibises makes them more dependent on high quality foraging patches and less able to withstand changes in hydrological conditions. Poor pre-breeding prey availability may be a cue for this species to lower its clutch size in preparation for increased costs of locating suitable high quality foraging patches to provision chicks.

Chick physiology showed that white ibis chicks were in poorer physiological condition in 2007 than in 2006 based on measures of long-term stress (SP60) and growth rates (mass). Great egret chicks, however, had increased levels of fecal corticosterone but no response in the growth rates (mass) of first- and second-hatched chicks during the poor year with lower prey biomass. Perhaps the increase in corticosterone in great egret chicks is not above a threshold that results in deleterious effects. Alternatively, the increase in great egret chick corticosterone levels may facilitate increased begging and provisioning (Kitaysky et al. 2001).

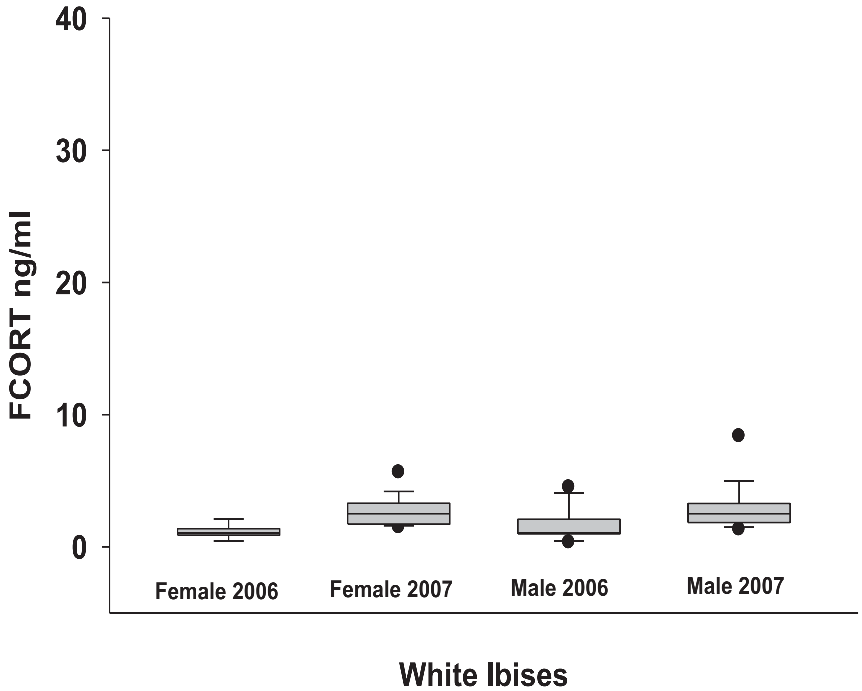

Figure 9. Adult white ibis fecal corticosterone levels during the 2006-2007 pre-breeding

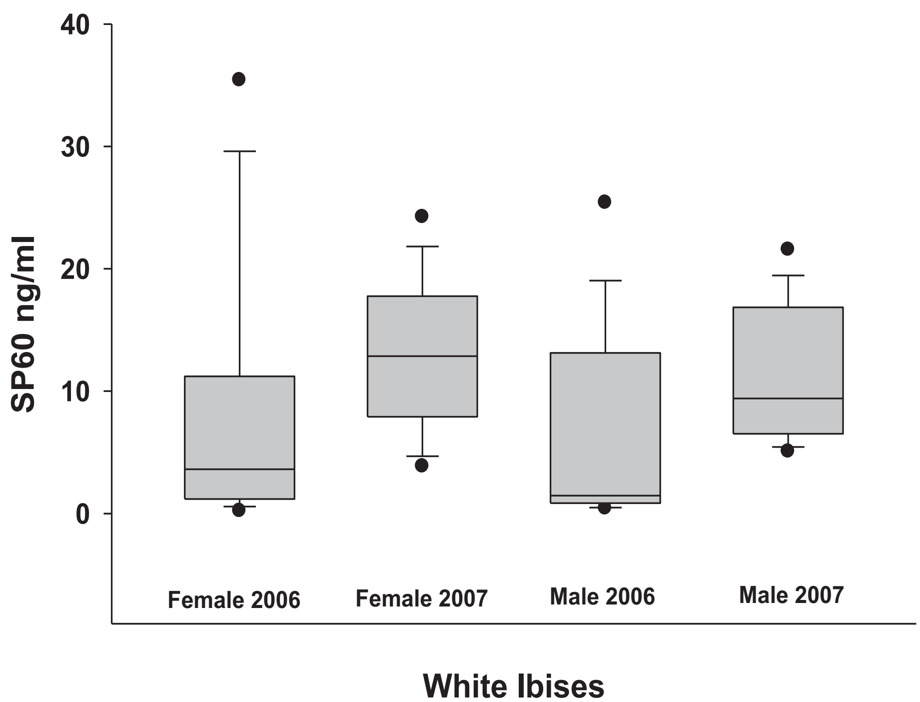

Figure 10. Adult white ibis stress protein 60 levels during the 2006-2007 pre-breeding period in the Loxahatchee NWR and WCA 2A and 3A

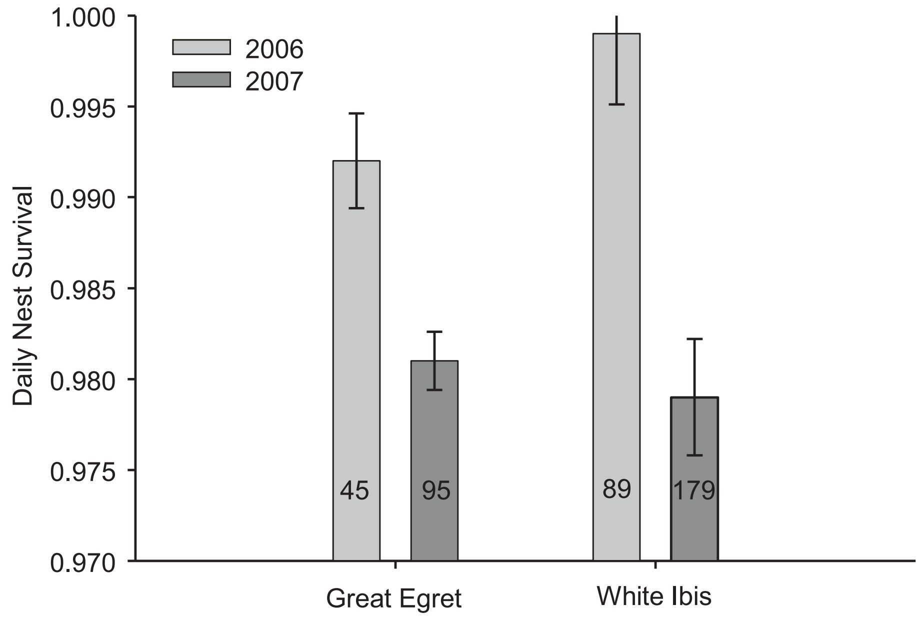

Figure 11. Me **and**
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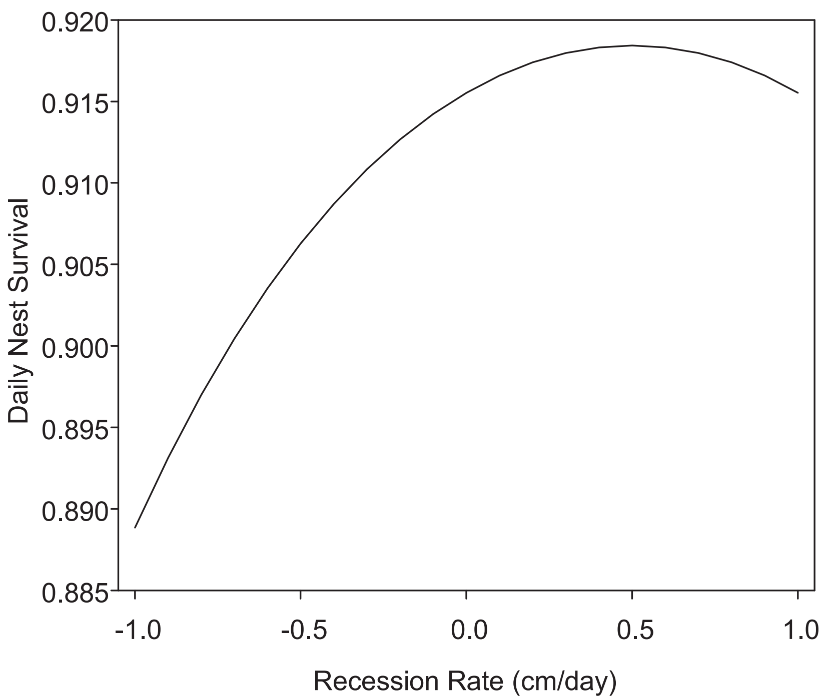

Figure 12. Variation in daily survival probability of great egrets relative to recession rate within the foraging flight range of nesting colonies during the 2006 - 2007 breeding seasons in the Everglades

The quadratic parameter estimates for recession rate show exceeding an optimal daily recession rate could negatively impact both great egret and white ibis daily nest survival rates. However, the manner in which egrets and ibises are impacted may differ. In the gradually sloping Everglades landscape surface, high quality patches tend to occur in bands perpendicular to the plane of the landscape slope. If the energetic costs of searching for new patches is higher for great egrets than white ibises, as we suspect, then egrets will be impacted first if recession rates are so high as to move the high quality foraging patches beyond an energetically profitably distance from a colony. The second way in which excessive recession rates could affect wading birds is by moving the band of high quality patches so quickly that birds are only able to capture a small portion of the concentrated aquatic prey before the patch dries. The net result would be that the quality of an individual patch would actually be lower than what prey density would indicate. The ecosystem did not experience extremely high recession rates during the two years of this study relative to past years; however, the quadratic form of recession rate that best described great egrets nest success responses to recession rate suggests that it may have been higher than optimal for that species.

Taken as a whole, this study demonstrated the significant effects of landscape-level prey availability and the habitat variables that influence prey availability on pre-breeding physiological condition of great egrets and white ibises and their reproductive responses. The responses of the two species is consistent with the notion that food availability has played an important role in the long-term nesting trends of both species, with ibis responding more acutely to lower prey availability. Results from adult, chicks and nest survival show that both prey density and factors that make prey vulnerable to bird predation, such as recession rates and hydrological reversals, affect wading bird productivity, but that the response between bird species will differ as a function of their foraging strategy.

4. Mercury Contamination Dynamics and Effects of Methymercury on Wading Bird Reproduction

Methylmercury became a contaminant of interest in the Everglades during the late 1990s, when high concentrations were found in tissues of nearly all predatory vertebrates. A concerted research program was necessary both to understand the source of the mercury (municipal and medical waste incineration in the urban east coast of south Florida; Frederick et al. 2005), and the extremely efficient methylation environment posed by the Everglades freshwater marsh (Gilmour et al. 1992). Beginning in the early 1990s, the deposition of mercury in the Everglades was decreased dramatically through use of pollution control devices, and the removal of mercury from the formulation of household batteries, which were thought to constitute over 70 percent of the mercury in the waste stream (Frederick et al. 2004).

At about the same time that mercury deposition was falling drastically, mercury concentrations in fish and wading birds were also beginning to decline significantly (***Figure 13***). For birds, the very sharp decline in mercury concentrations in feather tissue occurred at the same time as a sharp increase in numbers of birds breeding (***Figures 13** and **14***; Heath and Frederick 2005). This increase probably occurred for several reasons, and favorable hydrology was certainly more common during the years of the increase than prior to that time. However, there remains a statistical effect of annual mercury concentration on wading bird nesting numbers, even when the effect of hydrology is controlled. This suggested that high levels of mercury depressed the numbers of breeding birds at part of the hydrological response curve.

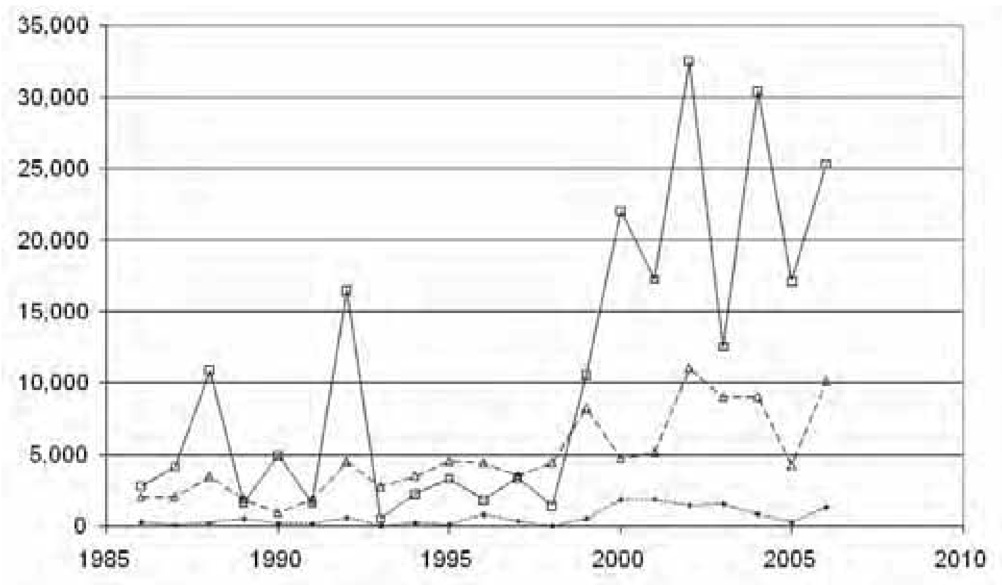

Figure 13. Mercury concentrations in feathers of nestling great egrets from 1994 through 2007

Each line/symbol group represents a single colony location sampled repeatedly. Not all colonies were active in all years. Despite large intra-annual variation in mercury exposure among colony sites, note a declining trend across colonies from 1994 through 1999. No data are presented for 2008 because of very small numbers of nesting birds.

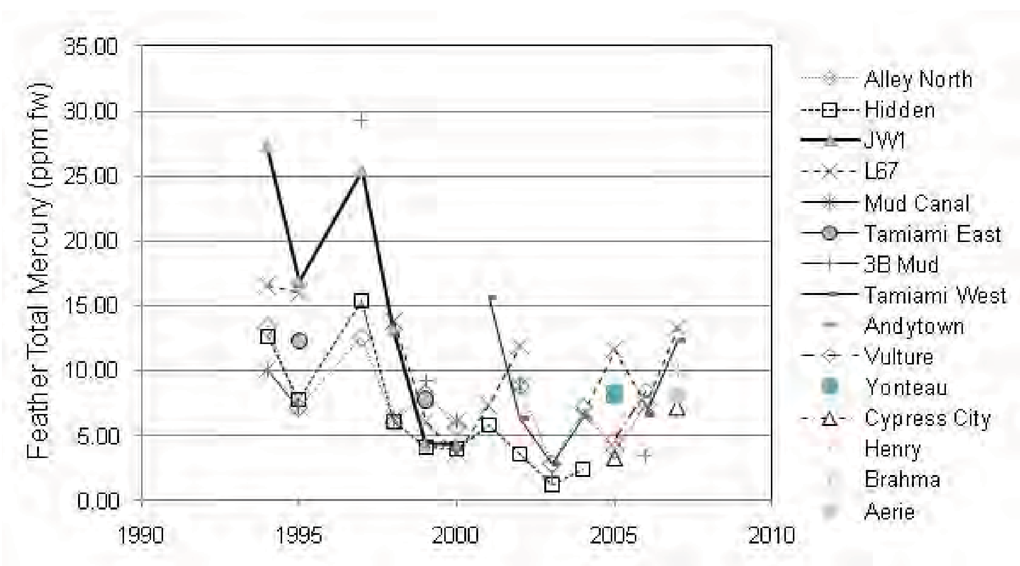

Figure 14. Numbers of nesting pairs of wood storks (diamonds), great egrets (triangles), and white ibises (squares) in the Everglades from 1986 through 2006 compared to total mercury in feathers

Note the large increase in nesting beginning in 1998. Compare this time sequence with Figure 1. Mercury measured as parts per million feather weight (ppm fw).

Mercury contamination is known to directly affect avian reproduction in various ways, including developmental abnormalities, decreased parental attentiveness, inappropriate reproductive behavior, decreased appetite, altered endocrine function and immunocompetence (Heinz 1979, Heinz et al. 2009, Spalding et al. 2000a, b). In wild common loons (*Gavia immer*), methylmercury concentrations in fish at 0.14 parts per million wet weight (ppm ww) is associated with a 40 percent decrease in reproduction (Evers et al. 2008). These effects levels are all well within the probable exposure range of Everglades wading birds during the mid-1990s (Loftus 2000).

Beginning in 2005, young white ibises were collected from a colony in the Everglades and randomly placed into one of four mercury exposure groups: 0, 0.05, 0.1 and 0.3 ppm ww in diet in a large free flight aviary in Gainesville, Florida. These diets spanned the range of dietary exposure in the Everglades. Dose groups comprised approximately 20 males and 20 females (sexed by DNA) and group locations were switched annually.

We found no evidence of differences in pre- or post-dosing appetite, size or size corrected mass, juvenile hormone expression (Adams and Frederick 2008), foraging behavior (Adams et al. 2009), health parameters or growth (***Table 4***). The birds were bred in captivity during three springs from 2006 through 2008. During each year, we found large proportions of pairs in the mercury dosed groups were male-male pairs, and the proportion of those pairing homosexually increased with dose group in all years (***Figure 15***). The differences in proportion pairing homosexually were greatest in 2007. The range was two percent of pairs for controls, 27 percent for low dose, and 43 to 44 percent for medium and high dose groups. The overall proportions nesting homosexually declined with age and, by the 2008 breeding season, the proportions were 0 percent, 16 percent, 24 percent and 20 percent in the control, low, medium and high exposure groups, respectively. Not surprisingly, this pattern resulted in significantly fewer of the dosed pairs producing eggs than the control pairs (***Table 5***).

Table 4. Summary of results of methylmercury exposure on captive juvenile and adult white ibises

Parameter	Age	Power of detection	Finding
Health	Juvenile and adult	Good to mediocre	No effects
Appetite	Juvenile and adult	Excellent	No effects
Foraging behavior	Juvenile	Excellent	No Effects
Developmental abnormalities	Juvenile	Mediocre	No Effects
Growth	Juvenile	Good	No Effects
Reproductive behavior	Adults	Good	No Effects
Clutch Size	Adults	Good	No Effects
Hatchability	Adults	Good	No Effects
Developmental abnormalities	Embryos and chicks	Good	No Effects
Pairing behavior	Adults	Excellent	Strong effects
Reproductive success	Adults	Good	Strong effects
Estradiol expression	Breeding Adults	Excellent	Effects on females
Testosterone expression	Breeding Adults	Excellent	No Effects
Corticosterone expression	Breeding Adults	Good	No Effects

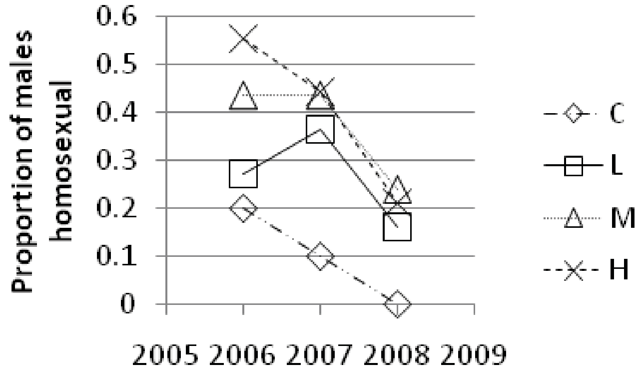

Figure 15. The proportion of males nesting homosexually in each dose group in 2006, 2007 and 2008

Table 5. Proportion of nests that had eggs, by year and treatment group

Entries with different letters within the same year show significant differences (Chi squared, $p<0.05$).

Year	Control	Low	Medium	High
2006	0.80a	0.79a	0.60b	0.50b
2007	1.00a	0.72b	0.80b	0.82b
2008	1.00a	0.93b	0.93b	0.91b

Male-male pairings were not related to a shortage of females. In fact, the highest level of homosexuality was observed in the high dose cage, where there was a female-biased sex ratio. Although no eggs were ever laid in male-male nests, male-male pairings occupied a very large proportion of the total pair days. One of the effects of homosexual pairings may therefore have been to reduce the numbers of females in reproduction, largely in proportion to mercury dose.

We did not find differences in clutch size or hatchability attributable to dose group. No differences were observed among the number of chicks fledged per active nesting attempt, or when considering all nesting attempts, but a difference was observed in individual performance in fledging success. The proportion of females in the high dose group successful in fledging at least one young during the 2008 breeding season was significantly lower when compared with the controls. The high dose group males showed a similar trend. This effect was not seen in the other dose groups, nor was it seen in 2007. It is possible that prolonged exposure to high levels of mercury resulting in high body burdens is somehow affecting parental behavior in the high dose group, maybe above a certain threshold.

This work has illustrated experimentally that chronic exposure to even very low concentrations of mercury typical of the Everglades can have strong effects on pairing behavior and consequently on reproductive success. The magnitude of this effect in the aviary can be expressed by several parameters. The proportion of nests with eggs reduced increased 30 percent. However, this measure is likely to be an underestimate of the effect in the field, since the proportion was affected by second and third nestings, which tended to be more heterosexual than homosexual. In the field, second and third nestings are rare unless the pair abandons nesting quickly, which the homosexual pairs in our study did not. Another way to view the effect is from the point of view of females unable to pair with a male. Homosexual pairings occupied up to 28 percent of the available breeding time in our studies. Again, this number is also probably conservative since second and third pairings are included. Finally, it is generally more difficult to estimate the likely effect of mercury contamination on reproduction in a field situation because our study did not include most of the stressors that would be evident in the field. These include much larger energetic demands of foraging and flying, exposure to disease, interruptions in food supply and predation pressure.

Summary

The hydrological disparities among 2005, 2006, 2007 and 2008 were associated with differences in prey concentrations and wading bird nesting in a way that supports the key trophic hypothesis, which is a foundation of the Comprehensive Everglades Restoration Plan (CERP). The fauna concentration study suggests both prey production and concentration act to limit wading bird nesting in different years. The important question now is under what conditions are wading birds limited by one or the other. This is a significant contribution to our understanding of restoration and wading bird nesting and it will allow for much better interpretation of historic nesting data. As more information on prey production and concentrations are collected and the degree to which one factor operates over the other becomes better known, our understanding of how the Everglades hydrologic restoration is linked to upper trophic levels will increase significantly. Although modeling and experiments have suggested such relationships exist, only now are data available to show a linkage between hydrology, prey availability and wading bird nesting in the field, which support this key restoration hypothesis.

These studies illuminate an important pathway for how hydrological conditions can influence prey availability during the wading bird breeding season, differential habitat use and physiological responses of foraging searcher and exploiter species, and, ultimately, their nesting responses. The fact that white ibises were more selective of foraging sites, particularly after hydrological reversals, lowered their clutch size, and fledged chicks in poorer physiological condition relative to great egrets in poor habitat condition years, suggest that they do not respond similarly to management of the Everglades. Poor foraging conditions will likely produce earlier, and larger negative responses (e.g. foraging site abandonment, increased stress and nest failure) in white ibises than great egrets, and may partially explain the difference in population trends between the two species. Given 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 of a response in nesting patterns for great egrets as for white ibises, although it seems current hydrologic patterns in the Everglades suite the great egret well.

The role of seasonal water recession has been clarified somewhat. Receding water is more important to both species in a year with poor habitat than in a year with good habitat; however, only great egret nesting success and habitat selection were affected by recession rate in both years. It has been noted previously that recession rate is more important to primarily fish eating birds, like the great egret and wood stork, than to white ibises, which eat primarily crayfish. The fauna concentration study confirmed crayfish densities do not increase as much as do fish and freshwater shrimp during the seasonal drydown.

The shape of the response to recession has a much stronger quantitative basis now than previously. It supports the idea that recession rates below 5 millimeters (mm) per day may result in nest failure, but also suggests that if the rate exceeds 7 mm per day it will increase nest failures. Habitat selection studies demonstrated when prey availability is low, water level recession is more important to both searchers and exploiters. If this pattern was extended to a reduction in prey availability over decades, it would support the suggestion that the role of the water level recession in the Everglades today is an artifact of the ecosystem being in a degraded state (Frederick and Spalding 1994).

Experimental studies of mercury dosing suggest that mercury levels in the Everglades during the early and mid-1990s were high enough to negatively affect breeding behavior and possibly cause a reduction in nesting numbers. Had mercury levels not been reduced to their current levels, this response could have confounded a response by wading birds to increased prey availability. Continued monitoring will also ensure that levels do not again get to the point where impacts from mercury on wading bird reproduction are likely.

A key effect of drought on wading bird prey availability was apparent from our studies. The main impact on birds does not appear to be a reduction in total prey density as much it is a decrease in the larger aquatic animals (greater than 0.2 cm in total length), which make up the bulk of wading bird diets, particularly for large birds like the wood stork and great egret. A smaller reduction in density of crayfish, which are the main prey of the white ibis, results from drought. A refined understanding of the effects of droughts and floods on the concentrations of individual prey species, and subsequently on the associated wading bird predators, will greatly improve our ability to predict the effects of short-term water conditions after a specific sequences of years with varying hydrologic conditions.

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Verification of the Predator-Prey Conceptual Model for Wading Birds and Aquatic Fauna Forage Base Using Data from Roseate Spoonbill Studies in Florida Bay

Prepared by Jerome J. Lorenz, Peter E. Frezza, Michelle Robinson, Luis Cañedo and Karen Dyer of Audubon of Florida towards validation of the Wading Bird Nesting in Relation to Aquatic Fauna Forage Base Hypothesis Cluster

Introduction

This report is the first to incorporate data from six related projects (***Table 1***) into a single cogent analysis of roseate spoonbill nesting dynamics, the abundance and availability of prey fish on their primary foraging grounds during nesting, the habitat at their foraging locations, and the affect of hydrologic parameters and water management practices on these parameters.

Table 1. Agencies, agreements, funding sources and data collected for projects that contributed to this report

Agency	Agreement Number	Funding Source of Project	Data Collected[1]
U.S. Army Corps of Engineers (USACOE in Figure 3)	W912EP-07-C-0021	Modified Water Deliveries to Everglades National Park Project (Mod Waters in Figure 3)	hydrology, fish and SAV data collection at BL, TR, JB, HC and BS
U.S. Fish and Wildlife Service (FWS in Figure 3)	40181G010	Endangered Species Act	hydrology and fish data collection at EC, WJ and SB, and spoonbill monitoring, banding and tracking
University of Florida and U.S. Geological Survey	00HQAG00212/DOI/ USGS SUB 00059613	Joint Ecological Modeling (JEM) Laboratory	spoonbill and fish habitat suitability modeling effort
Florida International University and South Florida Water Management District	205002512-01	CERP MAP	water level and fish data collection at RC, SC, NR, LI and 7P
National Oceanic and Atmospheric Administration National Marine Fisheries Service and South Florida Water Management District	WC133F-07-CN-0204	CERP MAP	hydrology and fish data collection at MB, CS and TP
Everglades National Park and U.S. Army Corps of Engineers	J5297 07 0204 Mod 1	CERP MAP	spoonbill data collection

[1]Site locations are shown in Figure 3.

Predator-Prey Interactions of Wading Birds and Aquatic Fauna Forage Base Conceptual Ecological Model

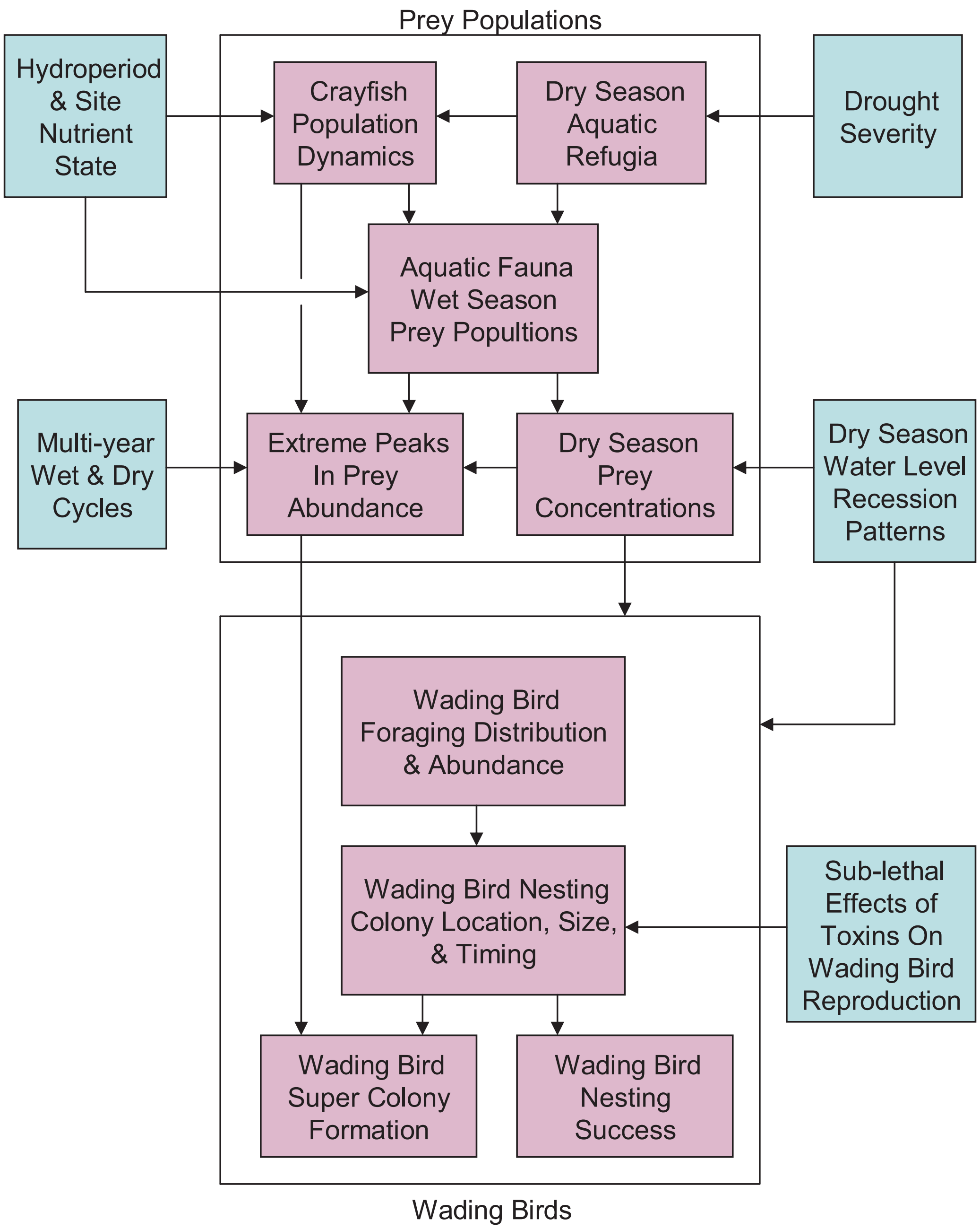


Figure 2 is a streamlined version of the conceptual model that is tailored to the trophic linkages specifically associated with roseate spoonbill nesting activities in Florida Bay. The main drivers are the effects of rainfall and water management practices on salinity and water level in the coastal estuaries (***Figure 2***). We use freshwater flows at Taylor Slough Bridge and discharge from the C-111 canal as parameters for these drivers. As also indicated by the model, the effects of rainfall and water management on salinity are also dependant on the antecedent conditions at the time of the start of the reporting period. The period of this report comprises three hydrologic years, each defined as June 1 to May 31, and nesting cycles. These years are 2005-06, 2006-07 and 2007-08. Data for 2004-05 is also presented so antecedent conditions that influenced conditions during the reporting period can be assessed.

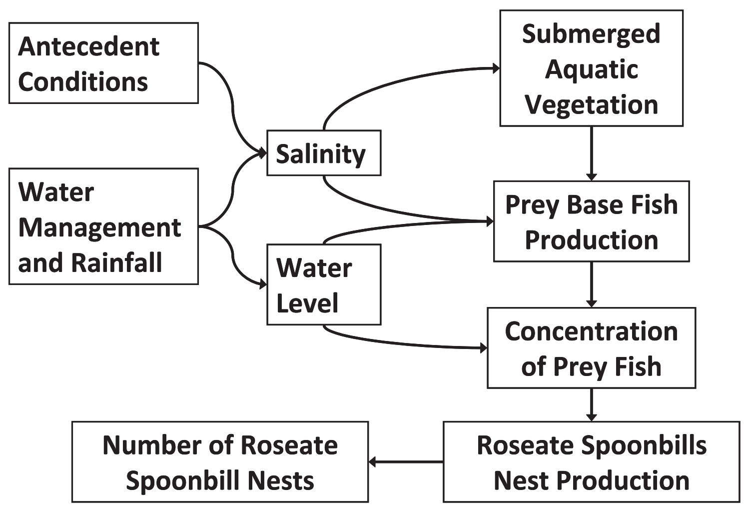


Figure 2. Conceptual model adapted from Figure 1 for spoonbills nesting in Florida Bay

Salinity and water level are the principal stressors of the model, as these parameters directly affect the biota at the lower trophic levels, specifically submerged aquatic vegetation (SAV) and prey base fish production and availability in the coastal ecotone (***Figure 2***). Lorenz and Frezza (2008) documented the direct statistical relationship between salinity and SAV and also made inferences about the effects SAV has on prey base fishes by providing food and habitat. The model also shows the direct effects of salinity on prey fish production, as documented by Lorenz and Serafy (2006). The model indicates water level has a direct impact on prey fish production by determining the amount of time (i.e., hydroperiod) and space available for fish to reproduce during the wet season (Loftus and Ecklund 1994, Lorenz 2000, Trexler et al. 2002). During the dry season, receding water levels further influence prey fishes by concentrating them into shallow pools and creeks where they are subject to predation by higher trophic levels (Kushlan 1980, Loftus and Kushlan 1987, Lorenz 2000, Gawlik 2002). In the case of the model, these higher trophic levels are represented by roseate spoonbills, with the parameter being nesting productivity. Lorenz (2000) and Lorenz et al. (2002) clearly demonstrated this link. Over time, nesting production determines the number of spoonbill nests in Florida Bay. This report presents the first attempt to demonstrate this linkage. The goal of this report is to test the hypotheses in this cluster by analyzing each of the links described in the model.

Methods

Fish collections were made at 17 locations known to be used by, or at least suitable for, spoonbill foraging (***Figure 3***). These 17 sites are divided into five watersheds: Shark River Slough (SRS in figure), Cape Sable, Taylor Slough, C-111 Basin and Southern Biscayne Bay (SBB in figure). Collections are funded through four different but related government agreements with Audubon of Florida (***Table 1***). In addition, SAV is monitored along transects at four of the fish sampling locations (***Figure 3***; TR, JB, HC and BS) and all 17 sites are equipped with a hydrologic monitoring station on-site or within one kilometer of the site. These stations provide hourly water level and salinity data, and it is these parameters that may be affected by water management practices.

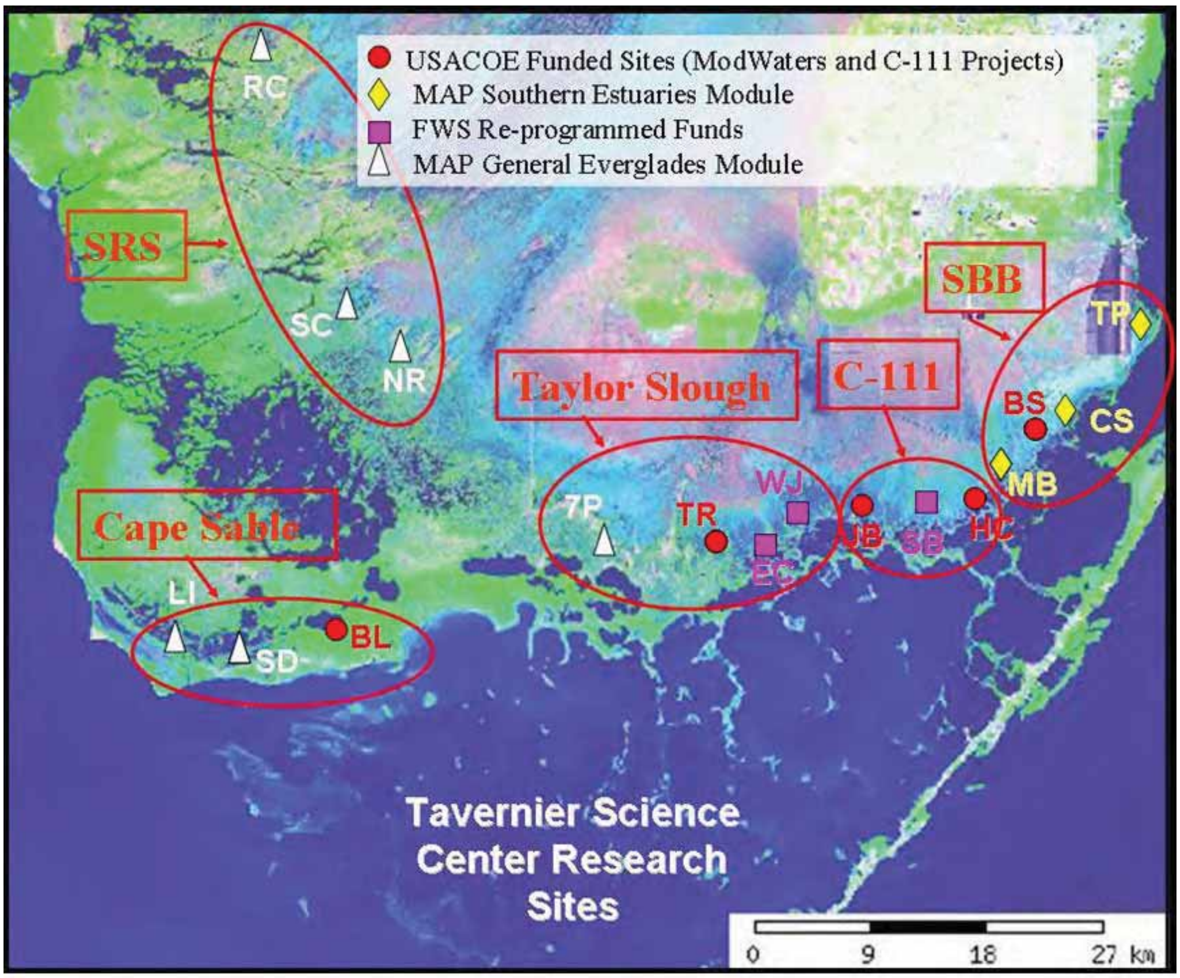


Figure 3. Location and supporting agency/funding source for 17 fish and hydrology sampling sites

Submerged aquatic vegetation surveys were performed at sites co-located with four of the fish sampling sites (***Figure 3***; TR, JB, HC and BS). Overall abundance (Morrison 1988) and total percent cover was estimated. Percent cover was estimated for each species as well.

Prey based fish collections were made in June, September and monthly from November through April. Lorenz et al. (1997) and Lorenz (1999) provide complete details regarding fish collections and statistical treatment of these data.

Flow rates are measured at Taylor Slough Bridge (TSB; ***Figure 4***) and at the S-18C and S-197 structures on the C-111 canal (***Figure 4***). Discharge from the C-111 canal toward Florida Bay was calculated by subtracting the flow at the S-197 structure from that at the S-18C structure. Salinity data were acquired for Duck Key and Middle Butternut Key (***Figure 4***). Water levels and salinity were collected at all of the sites in ***Figure 3***.

During the spoonbill nesting cycle (November-May), complete nest counts were made at 39 Florida Bay colonies. In addition, nest production was estimated for the northwestern (NW) and northeastern (NE) subregions of Florida Bay (***Figure 5***) by monitoring the largest colony within each region. This entailed marking up to 50 nests per colony shortly after full clutches had been laid, and then revisiting the nests on an approximate 7 to 10 day cycle to monitor nest success.

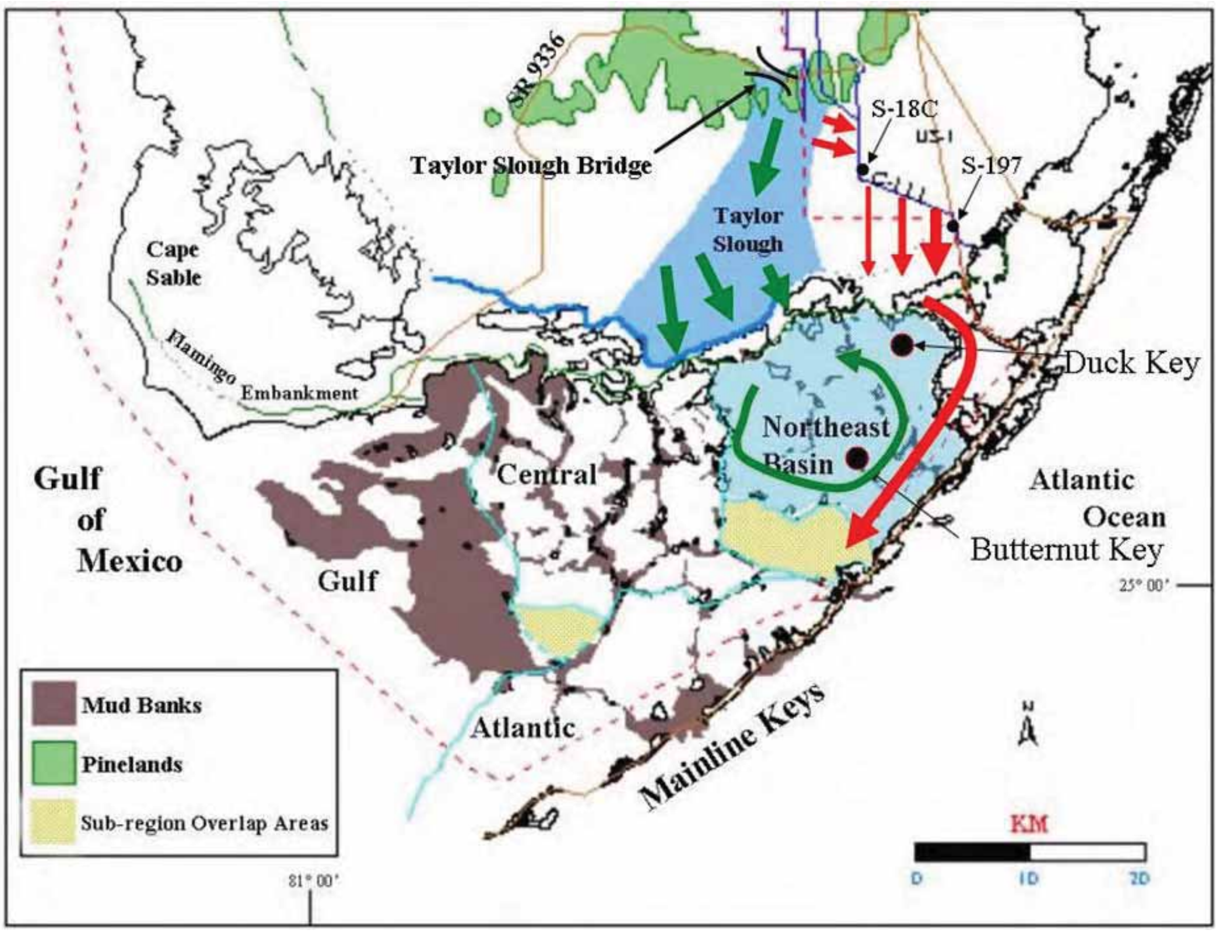


Figure 4. Freshwater flow into northeastern Florida Bay from Taylor Slough (green arrows) and the C-111 canal (red arrows) as well as locations of Taylor Slough Bridge, C-111 canal, water management structures S-18C and S-197, and Butternut and Duck Key hydrostations

The birds within the northeastern and northwestern subregions principally use wetlands on the mainland as their primary foraging grounds. These foraging grounds are influenced by water management to varying degrees. The northeastern foraging grounds are directly influenced by management practices. In comparison, the northwestern foraging grounds are relatively minimally affected.

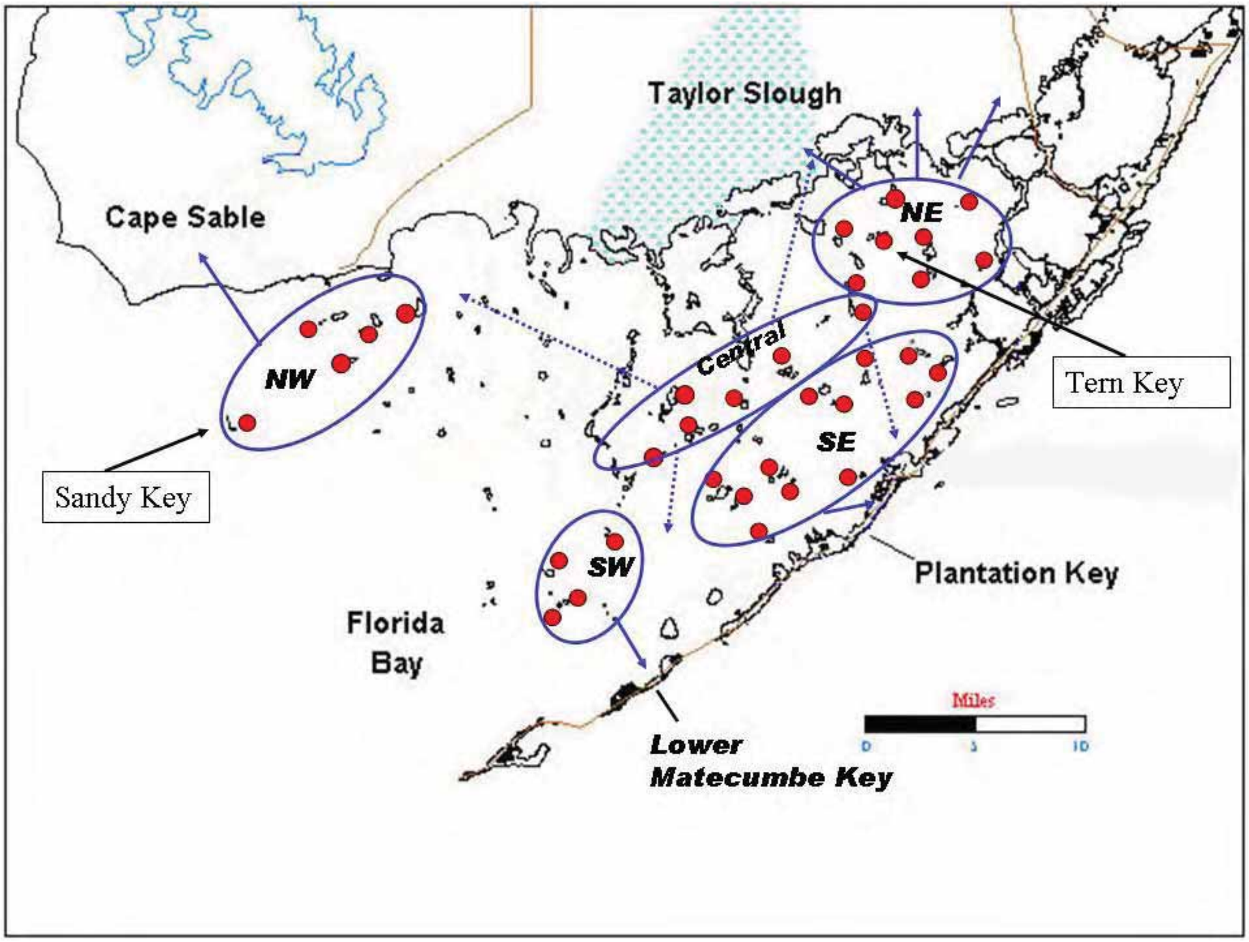


Figure 5. Location of all spoonbill colonies by region of Florida Bay with arrows indicate the primary foraging grounds for each region

Results and Discussion

Freshwater Flow into Florida Bay

Freshwater flows into Florida Bay by two main sources: Taylor Slough sheet flow and C-111 canal discharge (***Figure 4***). Currently, the majority of water delivered to the bay is through the C-111 canal (***Figure 6***), which draws water from the upper reaches of Taylor Slough and redirects it to the far eastern corner of Florida Bay, rather than into central Florida Bay (Lorenz 2000). By examining flow rates and salinity at two sites in northeastern Florida Bay (DK, BN; ***Figure 3***), Lorenz (2000) presented evidence that dry season salinity in the northeastern basin of Florida Bay is more a factor of total wet season flow through Taylor Slough than of flow through the C-111 canal.

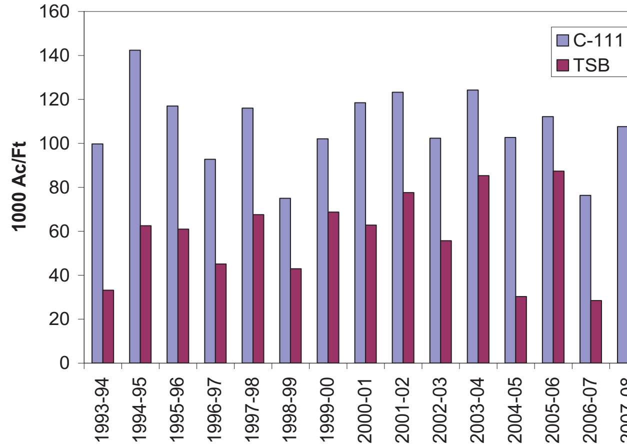


Figure 6. Comparison of annual freshwater flow volumes into Florida Bay via Taylor Slough and the C-111 canal from 1993 through 2008

Figure 7 summarizes these findings by showing that minimum salinity for the month of January, which is the month when salinity typically begins to rise sharply as freshwater flow diminishes, at two locations in the northeastern basin was more strongly correlated with wet season flows at Taylor Slough Bridge in Everglades National Park than with discharge from the C-111 canal (all correlations significant at $p<0.05$). Furthermore, the minimum January salinity in the northeastern basin determines the maximum dry season salinity at roseate spoonbill foraging locations (TR, JB and HC – the sites with data back to 1993) in the mangrove ecotone (***Figure 8***; all correlations significant at $p<0.05$). These analyses lead us to the conclusion that annual maximum salinity in the mangrove ecotone of northeastern Florida Bay is a factor of total flow through Taylor Slough during the preceding wet season, even though the vast majority of water enters Florida Bay via the C-111 canal (***Figure 6***). It appears that water entering Florida Bay from the C-111 canal bypasses the central bay as depicted in ***Figure 5***, thereby not adding to the salinity buffering capacity of the northeast basin.

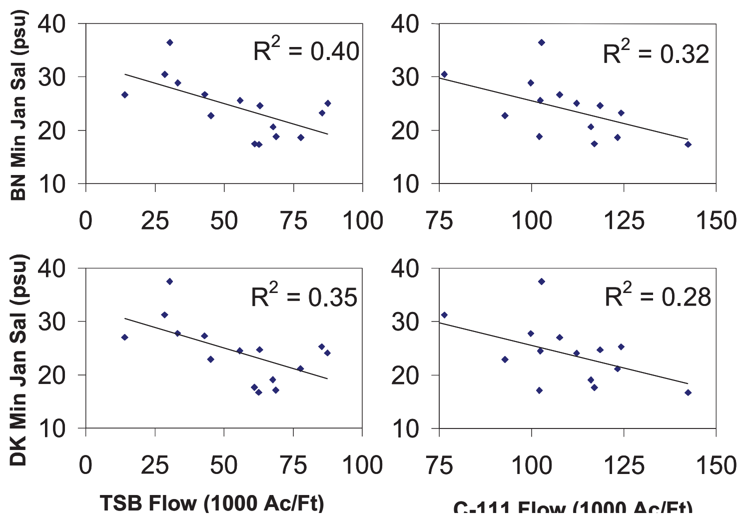


Figure 7. Correlations of freshwater flow from Taylor Slough and the C-111 canal with minimum January salinity at Duck Key (DK) and Butternut Key (BN) based on data from 1993 through 2008

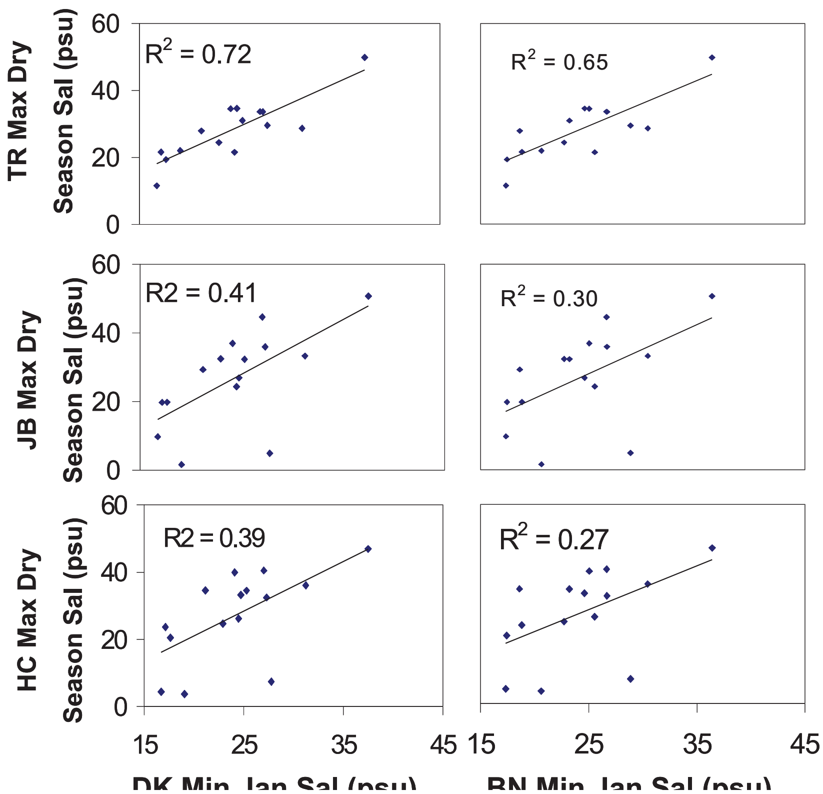


Figure 8. Correlations of minimum January salinity at Duck and Butternut Keys (DK, BN) with maximum dry season (December-May) salinity at three fish sampling sites (TR, JB and HC), which suggest dry season salinity is a product of the salinity buffering capacity of the northeast basin

Water Level Is Also Affected by Flow

During the wet season, flow and direct rainfall dictate water level. C-111 discharges also have an impact on water level, although the effect is relatively short lived. When large releases are made through the C-111 canal, water levels in the ecotonal areas south of the canal increase for several days following the discharge (Lorenz 2000, Lorenz and Frezza 2007). During the wet season, the effects of these rises are ecologically unimportant, as the water levels are already high. During the dry season, however, these pulse increases can have a devastating impact on predators including spoonbills and other wading birds. During periods of low water, fish become concentrated into small creeks and pools where are they are readily available to higher consumers (Lorenz 2000, Gawlik 2002). Lorenz (2000) documented that this concentration effect occurs when water levels on the ephemeral wetlands drop to 12.5 centimeters (cm) above the substrate level of those wetlands. Spoonbills, and other species that rely on this resource, time nesting to coincide with this fish concentration. If discharges raise water levels to the point prey fishes can disperse across the landscape, nesting birds cannot find enough food to meet the high energetic demands of their chicks. Even if the reversal event only lasts for a few days, this can result in high or even complete, mortality of the chicks.

For all three years of the reporting period, significant differences were observed in wet season flow at the Taylor Slough Bridge, with 2005-06 having more than twice the flow of 2006-07, and 2006-07 having almost twice the flow of 2007-08 (***Figure 9***). No differences in dry season flow were observed between the three years (***Figure 9*** insets). Discharge from the C-111 canal was significantly lower in the 2006-07 wet season when compared with the other two years, but no other significant differences in seasonal C-111 discharge was observed (***Figure 9*** inset).

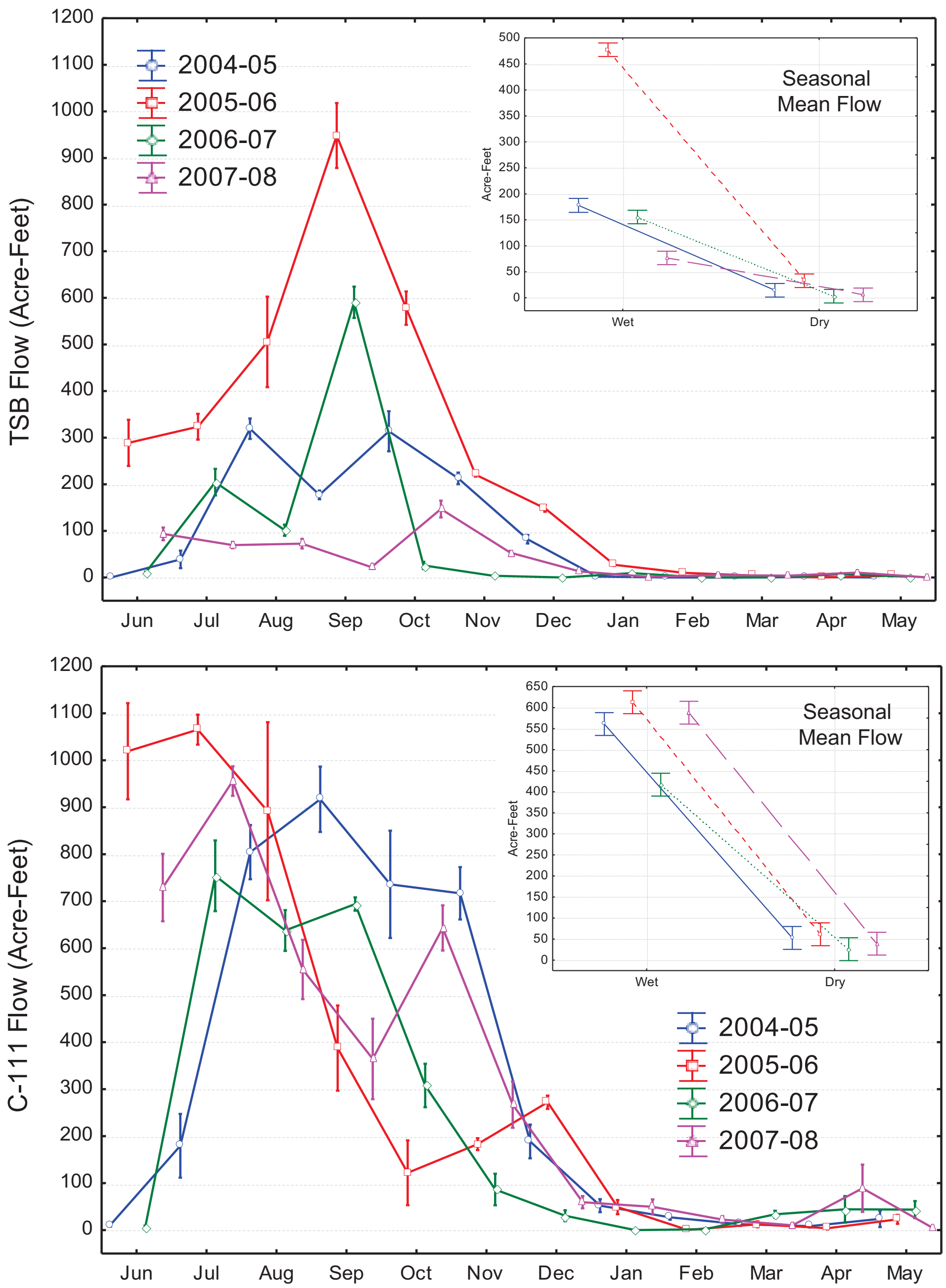


Figure 9. Mean monthly flow (± Standard Error [SE]) at Taylor Slough Bridge (top) and the C-111 canal (bottom) for June 2004 to May 2008 with insets providing mean flow at these locations by season

Salinity and Water Level at Fish Sampling Sites as a Function of Rainfall and Freshwater Flow

The mangrove ecotone of northeastern Florida Bay does not experience a lunar or diurnal tide (Holmquist et al. 1989, National Academy of Science data collection for this report). The annual cycle of water depth in the mangrove swamp is controlled by four factors that vary seasonally: sea level, wind, rainfall and flow (i.e., water management). Starting in approximately June, water levels increase throughout the summer months, peaking in late September or early October. Water levels then typically decline through October and November, culminating in dry season conditions from January through April or May (***Figure 10***). Salinity follows a similar but inverted pattern. Salt concentrations are typically highest in late May or early June and rapidly decline with the onset of the wet season in June (***Figure 11***). With the exception of relatively brief pulses in salinity in the early wet season, which usually only occur in dry years, salinity remains low throughout the wet season and is typically at or near freshwater conditions from September through December (***Figure 11***). Salinity begins to pulse upward in December and a steady and sustained climb typically begins in January and continues through to the beginning of the wet season (***Figure 11***). Because the natural breaking point in the annual hydrologic cycle is the initiation of the wet season, all analyses focus on the hydrologic year, which runs from June to May. These stereotypic cycles are affected by a high degree of spatial, interannual and intra-annual variation of physical conditions in the mangrove zone.

At three of the sampling sites, TR, JB, HC and BS (see ***Figure 3***) we have at least a 20-year record of salinity and water level. At site BS, 16 years of salinity and water level data are available. All other sites have periods of record less than six years for these parameters. We used the sites with long-term records (TR, JB, HC and BS) to characterize the reporting years. We also used these sites to characterize the year prior to the reporting years, which is 2004-05, so as to understand prior conditions.

Daily water levels for the three reporting years, the year prior to the reporting years and the 20-year mean are presented in ***Figure 10***. The year prior to the reporting years, 2004-05, was a severe drought year (Frezza et al. 2008) and the consequences of this drought are readily apparent. Water levels in 2004-05 were well below average as late as September at HC, and only exceeded the mean for relatively short periods of time. A great deal of interannual variability occurs during the wet seasons. Most striking is the peaks in water level from August to November of 2005. The two largest peaks in water level were the result of Hurricanes Katrina (August 25) and Wilma (October 24), which made landfall in southern Florida. Two lesser, but significant, peaks were a near miss by Hurricane Rita in late September and a non-tropical storm event in early October. These four storms not only caused the observed spikes but also resulted in relatively large flows through Taylor Slough (***Figure 9***). These conditions ended the lingering hydrologic effects (salinity in particular) of the 2004-05 drought but the ecological effects of the drought lingered for well over a year. Aside from these events, the dry seasons were remarkable similar for the three reporting years and the long-term mean water level.

Daily salinity for the four years and the 20-year mean are presented in ***Figure 11***. Again, the drought that occurred prior to the reporting years is evident with all four sites having consistently higher salinity during the early wet season and the dry season in 2004-05 than the three reporting years. Furthermore, the low salinity periods during the wet season were much shorter. The conditions in 2004-05 resulted in very high salinities relative to the 20-year mean in the early wet

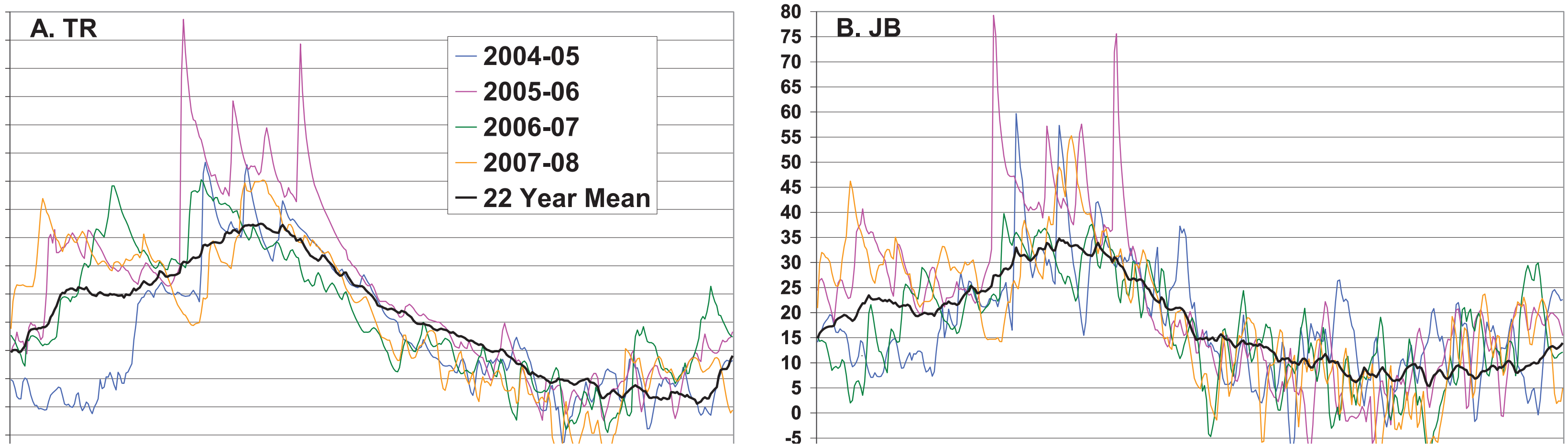


Figure 10. Comparison of daily water level at four fish sampling sites for each hydrologic year from 2004-05 to 2007-08 to the 20-year mean salinity at these sites

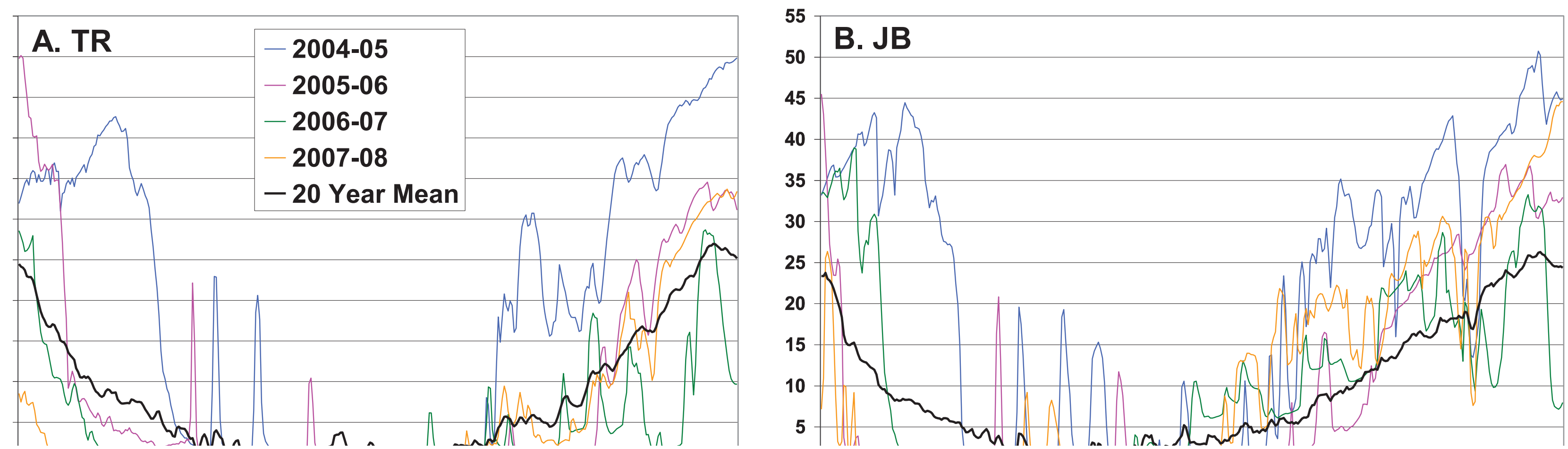


Figure 11. Comparison of daily water salinity in parts salinity units (psu) at four fish sampling sites for each hydrologic year from 2004-05 to 2007-08 to the 20-year mean salinity at these sites

season of 2005-06, but this was short lived as salinity dropped below the mean by mid-to-late June 2005. This coincides with the initiation of a relatively high flow period at the Taylor Slough Bridge and high C-111 discharges as a result of high rainfall, which subsequently ended the drought (***Figure 9***; Frezza et al. 2008). With the exception of the stark differences between the three reporting years during June, salinity patterns in all three years were remarkably similar to one another and to the 20-year mean.

Our conclusion from the examination of salinity and water level from the four long-term sites are the three reporting years had very similar hydrologic conditions, particularly during the dry season, and that over the three years, the ecotonal areas of Florida Bay experienced typical hydrologic conditions based on comparisons to long-term means. Based on these four sites, we also make the assumption that the other watersheds we examined also experienced typical hydrologic conditions during this three-year period. In lieu of reporting daily water levels and salinity for all of the remaining 13 sites, we examine hydrologic data from these sites by comparing monthly means for each watershed between years using analysis of variance (ANOVA) to determine what, if any, differences occurred between the three years. These watersheds are Taylor Slough, C-111, Southern Biscayne Bay, Cape Sable and Shark River Slough (***Figure 3***)

The Taylor Slough and C-111 watersheds are the primary foraging grounds for spoonbills nesting in northeastern Florida Bay, but spoonbills also forage in the Southern Biscayne Bay watershed. As with water level and salinity at the long-term sites, the ANOVAs of these parameters statistically showed few significant differences between years, with the majority of the differences occurring during the period of transition from the wet season to the dry season (May and June, **Figures** ***12***, ***13*** and ***14***). These differences can be explained by variation in the timing of the onset of the wet season between years. These data were so similar, only one example is found in which all three years were different during a given month for either parameter in all three watersheds; all three years had significantly different salinity means for June in Taylor Slough. These results bolster the conclusion that these three years were hydrologically very similar.

In the Taylor Slough and C-111 watersheds, all three reporting years had consistently low salinity during the wet season (***Figures 12*** and ***13***), which, according to the spoonbill conceptual model (***Figure 2***), should result in higher primary production of SAV and a more robust fish prey base. These conditions were the result of the 2004-05 drought ending by the 2005-06 tropical storms and water managers sending Taylor Slough more fresh water (***Figure 9***). An examination of the salinity in the Southern Biscayne Bay watershed (***Figure 14***) provides further evidence water management practices contributed to low salinity conditions in the Taylor Slough and C-111 Basin watersheds. The Southern Biscayne Bay watershed is isolated from the managed flows to Florida Bay by the US 1 and Card Sound roadbeds and, therefore, acts as a control for the effect of these flows on the Taylor Slough and C-111 watersheds. Southern Biscayne Bay salinity did respond to the storms, with the lowest salinity of 2005 following Hurricanes Katrina and Rita in September (***Figure 14***). By October, however, salinity had increased to above pre-storm conditions. In the absence of managed flows, we expected the same to occur in the Taylor Slough and C-111 Basin watersheds. Flows through Taylor Slough were kept relatively high during this period (***Figure 9***), suggesting management flows resulted in salinity remaining low at the Taylor Slough and C-111 Basin sites.

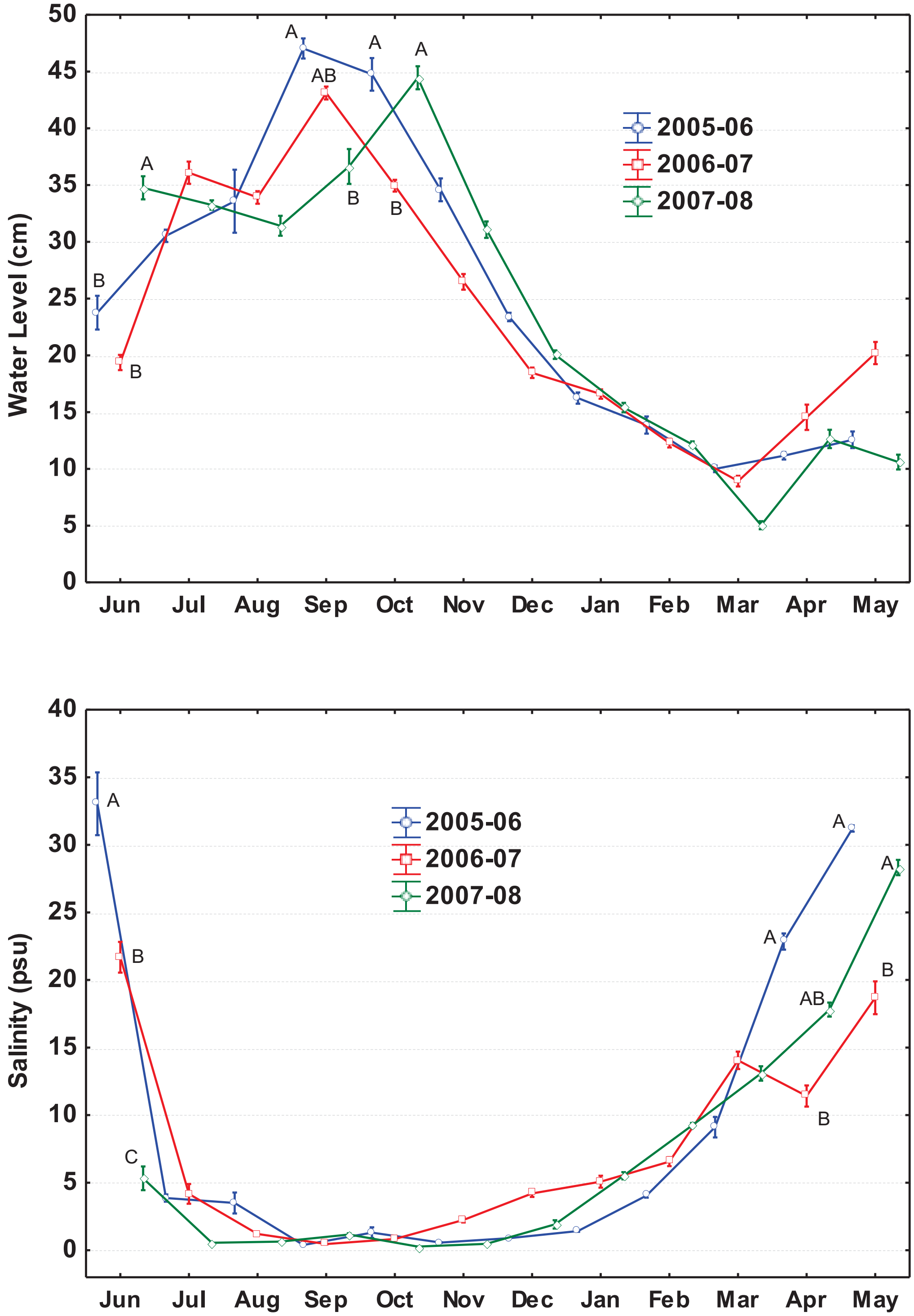
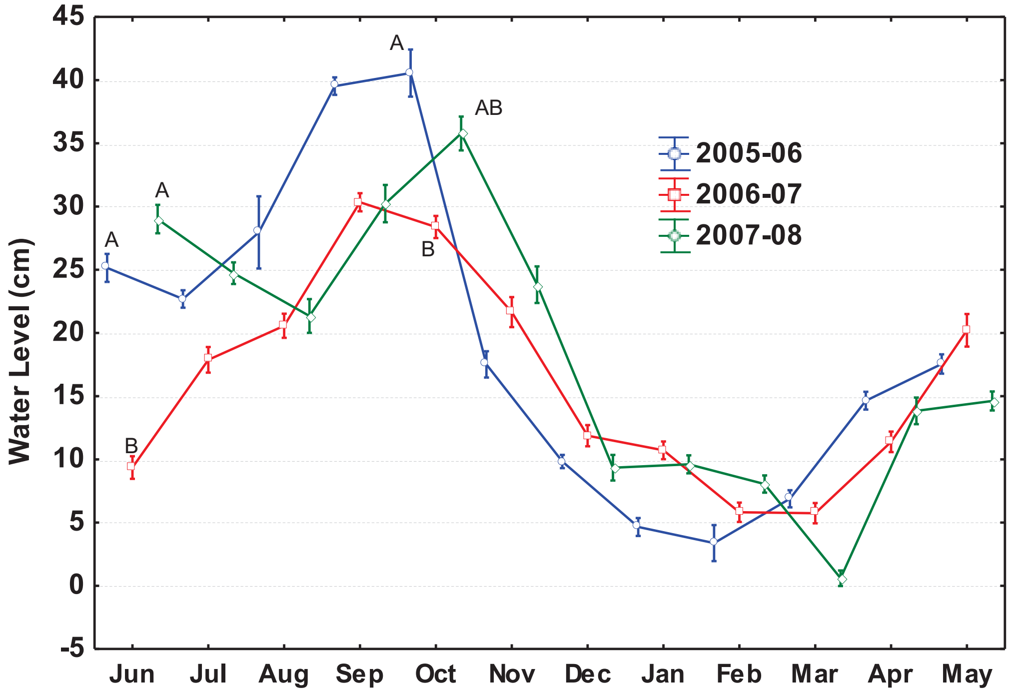


Figure 12. Mean (± SE) monthly water level (top) and salinity (bottom) for the Taylor Slough watershed (sites 7P, TR, EC and WJ combined) for the three reporting years

Points labeled with different letters indicate a significant difference between sites within months.



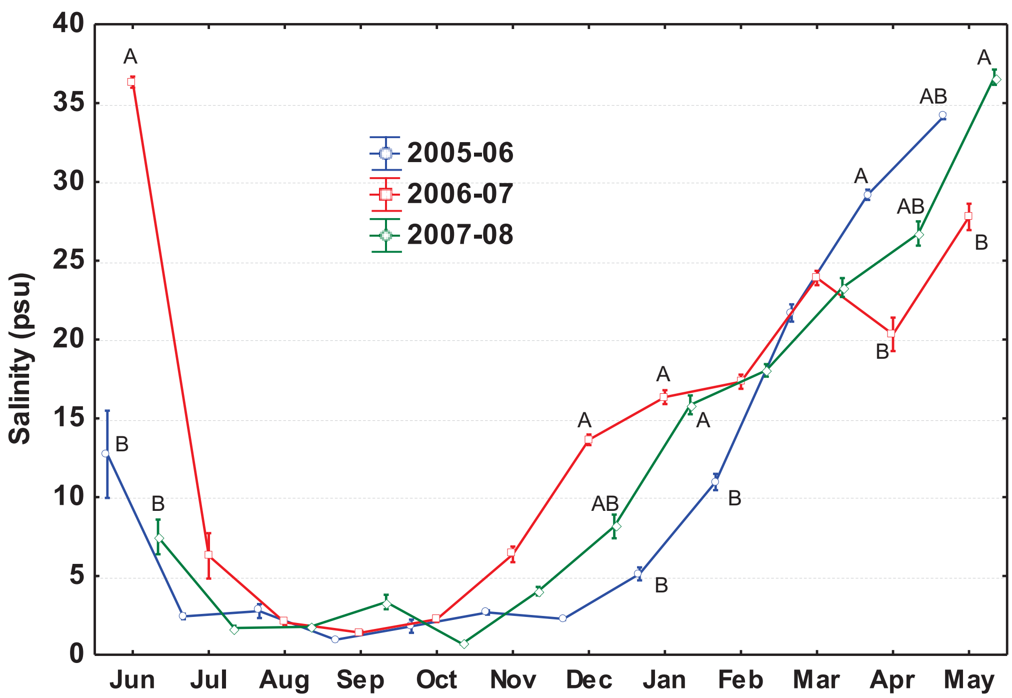
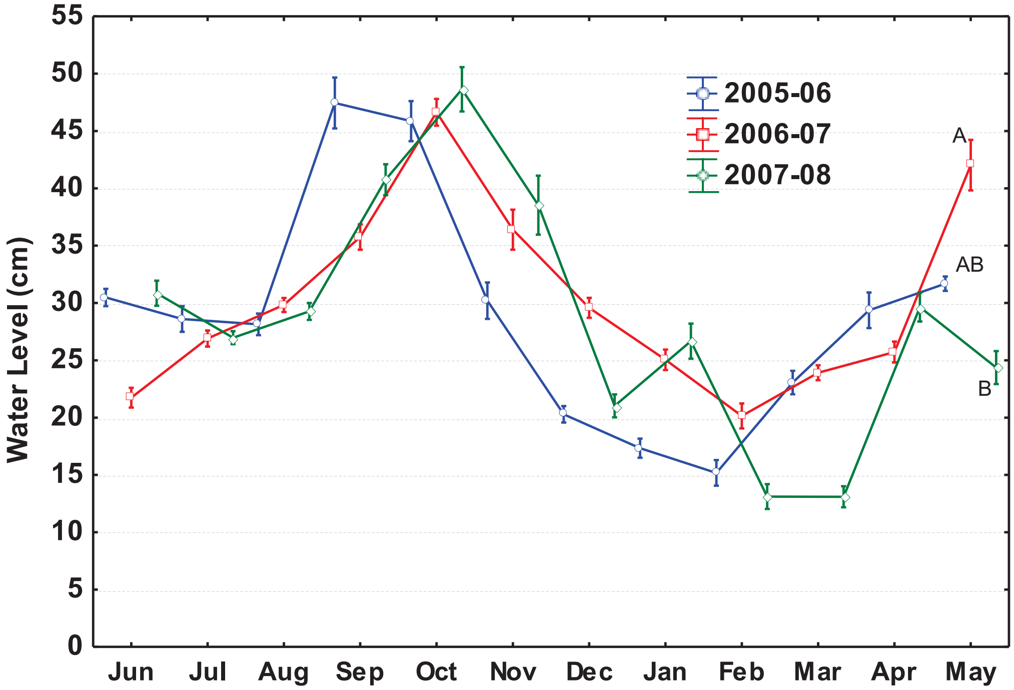


Figure 13. Mean (± SE) monthly water level (top) and salinity (bottom) for the C-111 watershed (sites JB, SB and HC combined) for the three reporting years

Points labeled with different letters indicate a significant difference between sites within months.



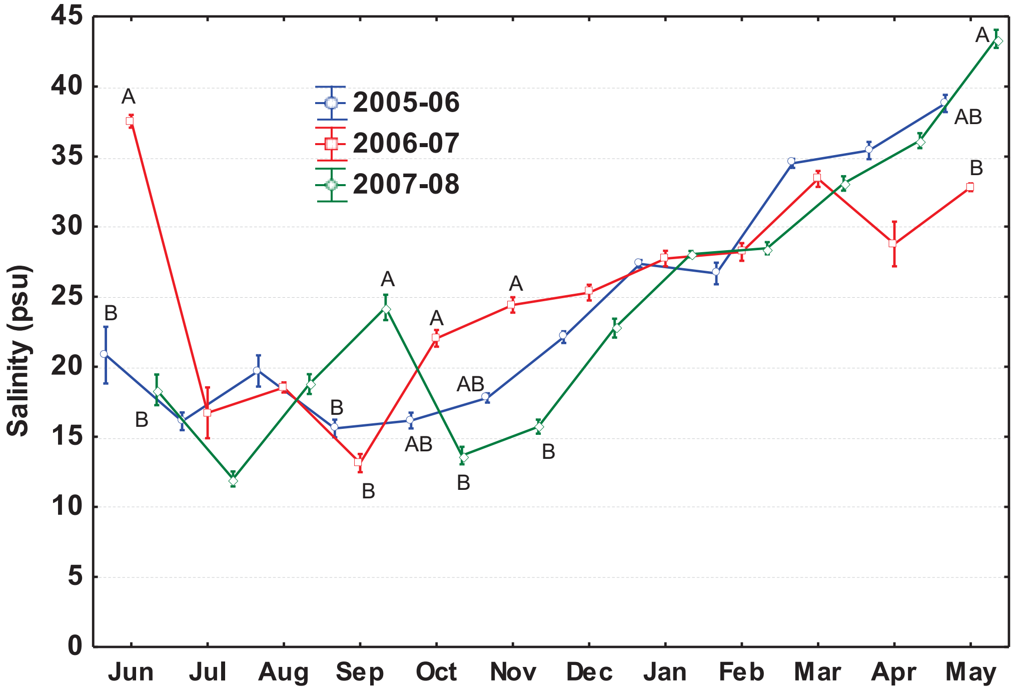


Figure 14. Mean (± SE) monthly water level (top) and salinity (bottom) for the Southern Biscayne Bay watershed (sites MB, BS, CS and TP combined) for the three reporting years

Points labeled with different letters indicate a significant difference between sites within months.

Rainfall and flow resulted in 2005-06 having as long a hydroperiod as the other two years. This is particularly apparent in the C-111 watersheds (***Figure 13***) where water levels were significantly lower in June 2005 than those of June 2006 or 2007. When flows through Taylor Slough increased in June 2005 due to changes in management practices in response to the first tropical storms of this record breaking year for storm activity (***Figure 9***), the effect on downstream water level was apparent in both the Taylor Slough and C-111 watersheds. This ending of the 2004-05 drought conditions resulted in 2005-06 having very similar hydroperiods as the other two reporting years rather than being a short hydroperiod year. Based on the spoonbill conceptual model (***Figure 2***), this should result in a higher abundance of prey fishes for all three reporting years.

C-111 discharges and water levels affect the foraging grounds during the breeding season. Spoonbills typically begin nesting in November, with the chicks hatching in December. For several months following hatching, spoonbills need fish to be concentrated on their foraging grounds but discharges from the C-111 canal can disrupt this concentration of prey. In all three reporting years, discharges from the C-111 canal were negligible during the critical nesting months (***Figure 9***). The spoonbill conceptual model (***Figure 2***) suggests these conditions would prove favorable for spoonbill nesting success.

The Cape Sable watershed is the principal foraging ground for spoonbills nesting in northwestern Florida Bay, but these spoonbills also forage in the Shark River Slough watershed. These wetlands are less directly affected by water management practices than the northeastern colonies, and, therefore, act as a control for the impact of water management practices on the interaction proposed in the spoonbill conceptual model (***Figure 2***). ***Figures 15*** and ***16*** provide the results of the ANOVAs that examine differences between reporting years for water level and salinity for the Cape Sable and Shark River Slough watersheds, respectively. For Shark River Slough sites, we only have data beginning in November 2005. The results of these ANOVAs support the findings in the eastern watershed that these years had very similar patterns in water levels with very few months with significant differences (***Figures 15*** and ***16***). These years were very different, however, when it came to salinity. The consistently higher salinity in 2005-06 at Cape Sable sites was likely a lingering effect of the drought during the preceding year (2004-05). This was not the case with Shark River Slough sites, however, where salinity was typically lower in 2005-06. This may simply be due to the fact that the Shark River Slough watershed receives a great deal of upstream flow in addition to rainfall while the Cape Sable watershed is isolated from upstream flow by Whitewater Bay and the entire Cape Sable peninsula is rainfall driven. With the active tropical storm season in 2005, the Shark River Slough sites would have been completely flushed of salt water due to high rainfall upstream. In the case of the Cape Sable watershed, the rainfall from these storms was also accompanied by high storm surges that pushed marine water over the coastal ridge; the storm surge was approximately three meters at the LI site during Hurricane Wilma. These differences in how storms impacted these very different watersheds likely explain the observed differences in salinity patterns between years.

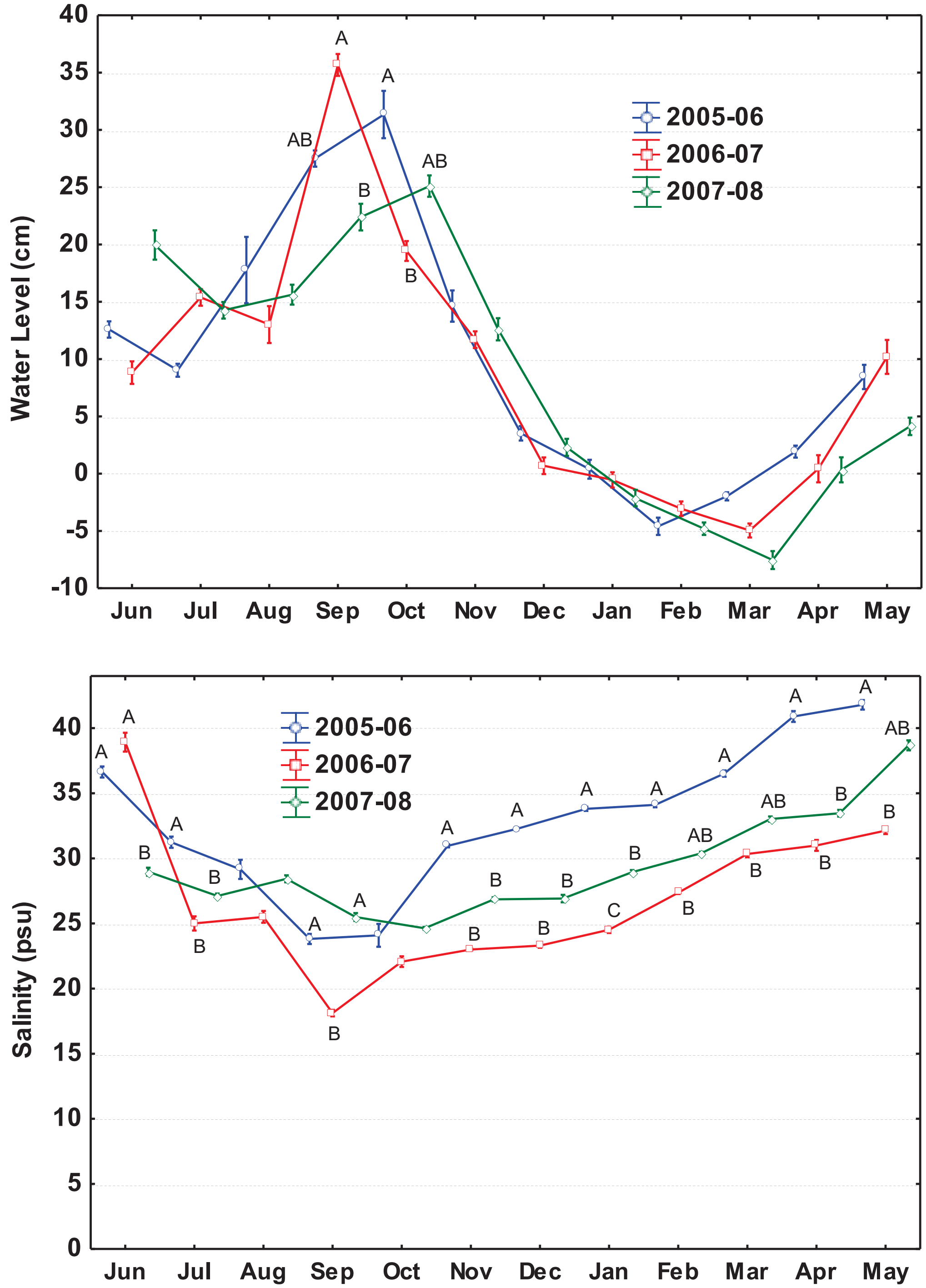


Figure 15. Mean (± SE) monthly water level (top) and salinity (bottom) for the Cape Sable watershed (sites LI and BL combined) for the three reporting years

Points labeled with different letters indicate a significant difference between sites within months.

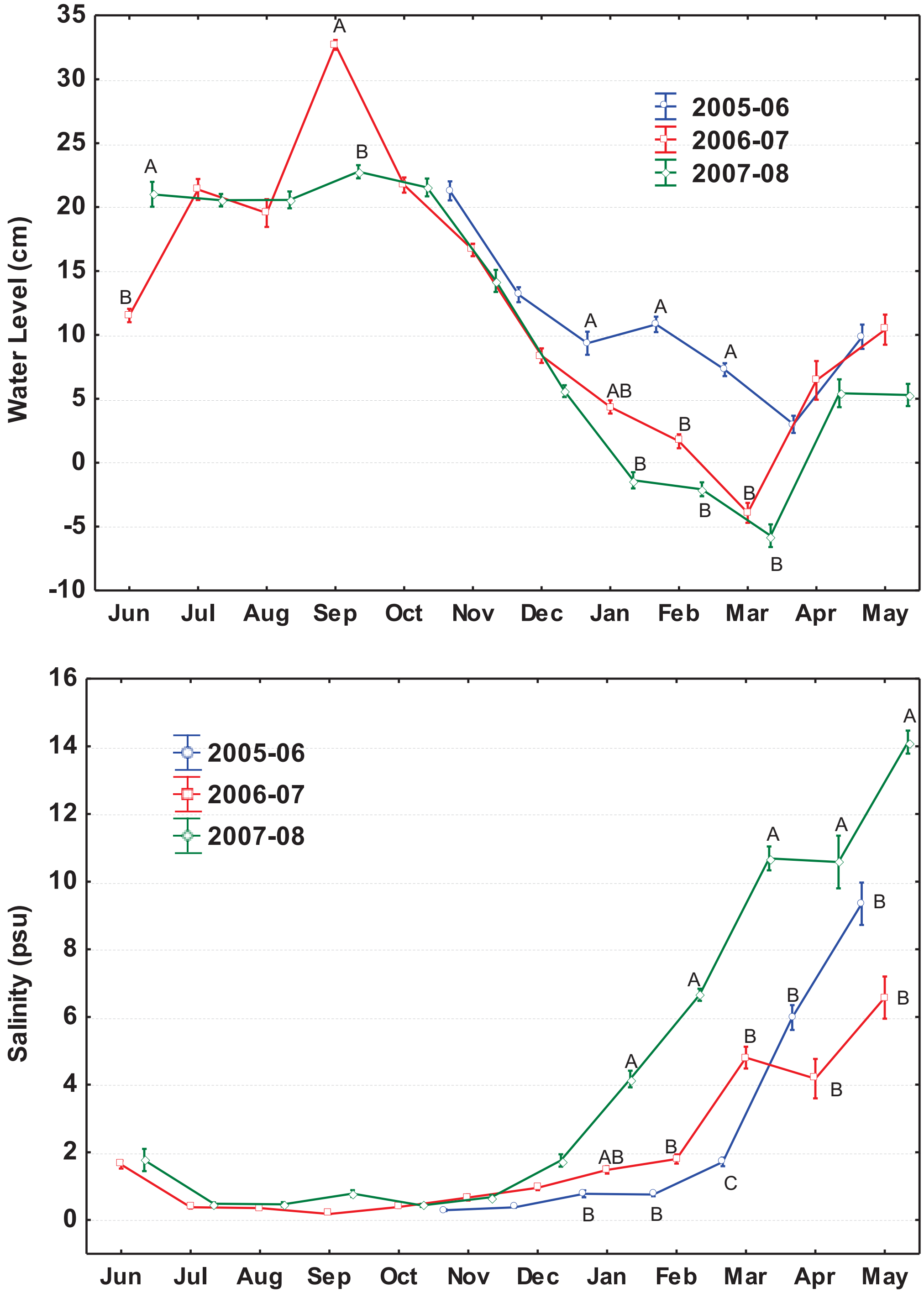


Figure 16. Mean (± SE) monthly water level (top) and salinity (bottom) for the Shark River Slough watershed (sites SC, RC and NR combined) for the three reporting years

Points labeled with different letters indicate a significant difference between sites within months.

SAV data is not collected at any of the Cape Sable or Shark River Slough sites, so we cannot examine the relationship described by the spoonbill conceptual model (***Figure 2***) between salinity and plants for these watersheds. The conceptual relationship between fish abundance and salinity from the model would suggest fish production should be lower in 2005-06 due to higher salinity in the Cape Sable watershed (***Figure 15***). At the Shark River Slough sites, a higher abundance of fish should be present in 2005-06 than in 2007-08, and possibly 2006-07 as well, due to significantly lower salinity in 2005-06 (***Figure 16***). As for the conceptual relationship between hydroperiod and fish abundance, the years were very similar in hydroperiod and water depth (***Figure 15*** and ***16***) so this aspect of the model suggests production is similar. Finally, all three reporting years seem to have ideal water level conditions for fish concentration, with depth below 12.5 cm in all breeding months (***Figure 15*** and ***16***), which should be favorable to successful spoonbill reproduction.

Salinity and Submerged Aquatic Vegetation

An analysis of our complete SAV dataset dating back to 1996 shows a high degree of variability in total SAV coverage on both a seasonal and interannual level. A number of die-off and re-growth events have been documented throughout the period of record. In order to better understand this variation in SAV coverage, Frezza et al. (2007) examined the relationships between total SAV coverage and salinity, water temperature and water level. Significant negative relationships exist between salinity and total SAV percent cover on both a seasonal and annual level. Also, years with higher, more variable salinity corresponded to years with less total SAV coverage and years that had a lower, more stable salinity regime responded with greater total coverage and a more diverse SAV assemblage.

As discussed above, salinity from the four long-term SAV monitoring sites (TR, JB, HC and BS) had very similar hydrologic conditions over the three reporting years and the ecotonal areas of Florida Bay experienced „typical‟ hydrologic conditions based on comparisons to long-term means. Of particular ecological importance to the SAV community is the three reporting years had salinity conditions that approximated „normal‟ compared to the long-term, 20-year mean of salinity from these sites. Furthermore, through much of the wet season, all three years had lower than normal salinity conditions and experienced dry season conditions that did not reach hypersaline conditions (greater than 35 psu) for extended periods. Given this hydrologic scenario, we would expect that the SAV community would be similar between years and that total percent coverage of SAV would be typical compared to what we have seen over our period of record. We also would not expect the SAV community to suffer a die-off event during this three-year period.

Differences in total SAV abundance and *Ruppia maritima* abundance between hydrologic years were examined using ANOVA. *R. maritima* was included in these analyses because this species has been identified as a keystone indicator of seagrass community health and a target species for salinity optimization in Florida Bay as Everglades‟ restoration proceeds (SFWMD 2006). Two-way ANOVA between mean total SAV coverage (four sites combined), month, and the three years under investigation revealed a significant difference ($F_{14,1129} = 2.0307$, $p = .01328$) in plant coverage between the three reporting years (***Figure 17***). However, Tukey‟s HSD post hoc test revealed that there were no individual months that had significantly different coverage between years. Although not statistically significant at the predetermined level ($p < 0.01$), ***Figure 17*** reveals that between September and January, noticeably less SAV was present during 2005-

06. ***Figure 17*** also shows that total SAV coverage during 2005-06 decreased throughout the productive wet season months. This is a very atypical trend for the SAV community in these wetlands, which normally show increasing coverage throughout the wet season as seen during the 2006-07 and 2007-08 hydrologic years. These substantial differences were most likely enough to have ecological significance and this warranted further investigation.

A review of antecedent hydrologic conditions prior to our reporting period reveals that during the 2004-05 hydrologic year, the coastal mangrove zone of northeastern Florida Bay experienced drought conditions. These minor drought periods occur one out of every five years (Frezza et al. 2005). A substantial rainfall deficit in the Taylor Slough watershed during 2004-05 prolonged hypersaline conditions in this estuary (***Figure 11***). A major die off of SAV at all of our monitoring sites occurred concurrently with the increased and highly variable salinity during 2004-05 (***Figure 17***). The lowest total SAV and individual species (*R. maritima*, *Chara hornemanii*, *Utricularia spp.* and *Najas marina*) coverage was recorded during the 2004-05 hydrologic year at all sites since monitoring began in 1996 (Frezza et al. 2005). Our previous studies have shown it takes at least two years following a major die-off event for SAV to recolonize to pre-die-off coverage (Frezza et al. 2006). While hydrologic conditions were „normal‟ during the 2005-06 hydrologic year, it can be assumed that the SAV community was in a recovery state following the die-off event, explaining the low abundance detected in that year. Due to the impact the 2004-05 hydrologic year had on the SAV community, SAV data from 2004-05 was included in all figures for this report to better illustrate the changes in community composition that ensued. SAV data from 2004-05 was not, however, included when reporting results of the ANOVAs.

An analysis of the SAV community at the site level indicated that the two-way ANOVA for year and site was also significant ($F_{6, 1141}=14.786$, $p<.001$). Three out of the four research sites had less coverage in 2005-06 than the other two reporting years. ***Figure 18*** shows total SAV coverage by month for the four monitoring sites for the three reporting years. Only the control site for freshwater flow (BS) had more coverage in 2005-06 (***Figures 17*** and ***19***). Post hoc tests revealed the differences in total coverage at TR and JB,between 2005-06 and the other two years was significant ($p<0.01$; ***Figure 18***). Total mean plant coverage (four sites combined) was also lowest in 2005-06 as indicated in ***Figure 17***.

Along with total SAV, the abundance of *R. maritima* at all monitoring sites was also analyzed (***Figure 19***). Two way ANOVA between mean *R. maritima* coverage (four sites combined), month, and the three reporting years revealed no significant difference in *R. maritima* coverage ($F_{14, 1129}=1.3001$, $p=0.20$) between years. ***Figure 19*** show *R. maritima* coverage by month for the four monitoring sites for the three reporting years. Post hoc tests on these data showed no significant ($p<0.01$) differences in *R. maritima* between any of the months surveyed during these three years (***Figure 19***). However, as with total SAV coverage, annual mean *R. maritima* coverage (four sites combined), was the lowest during 2005-06 (***Figure 17***). The two-way ANOVA between mean *R. maritima* coverage (four sites combined), site and the three reporting years revealed significantly ($p<0.01$) less *R. maritima* coverage at the TR and JB sites during 2005-06 (***Figure 19***).

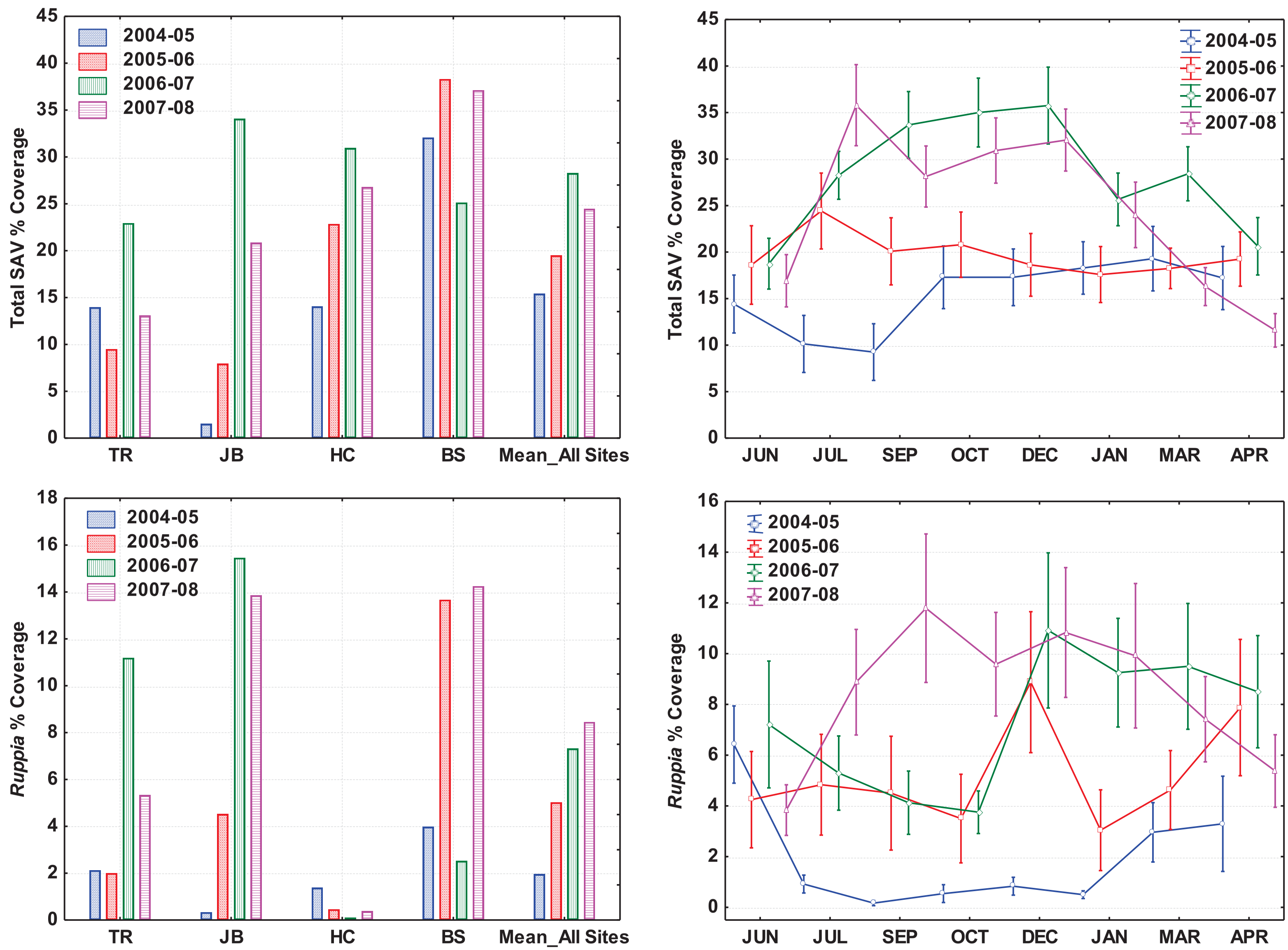


Figure 17. Mean total SAV coverage (top panels) and *R. maritima* coverage (bottom panels) on an annual basis for each site (left panels) and on a monthly basis (± SE) for each year (right panels)

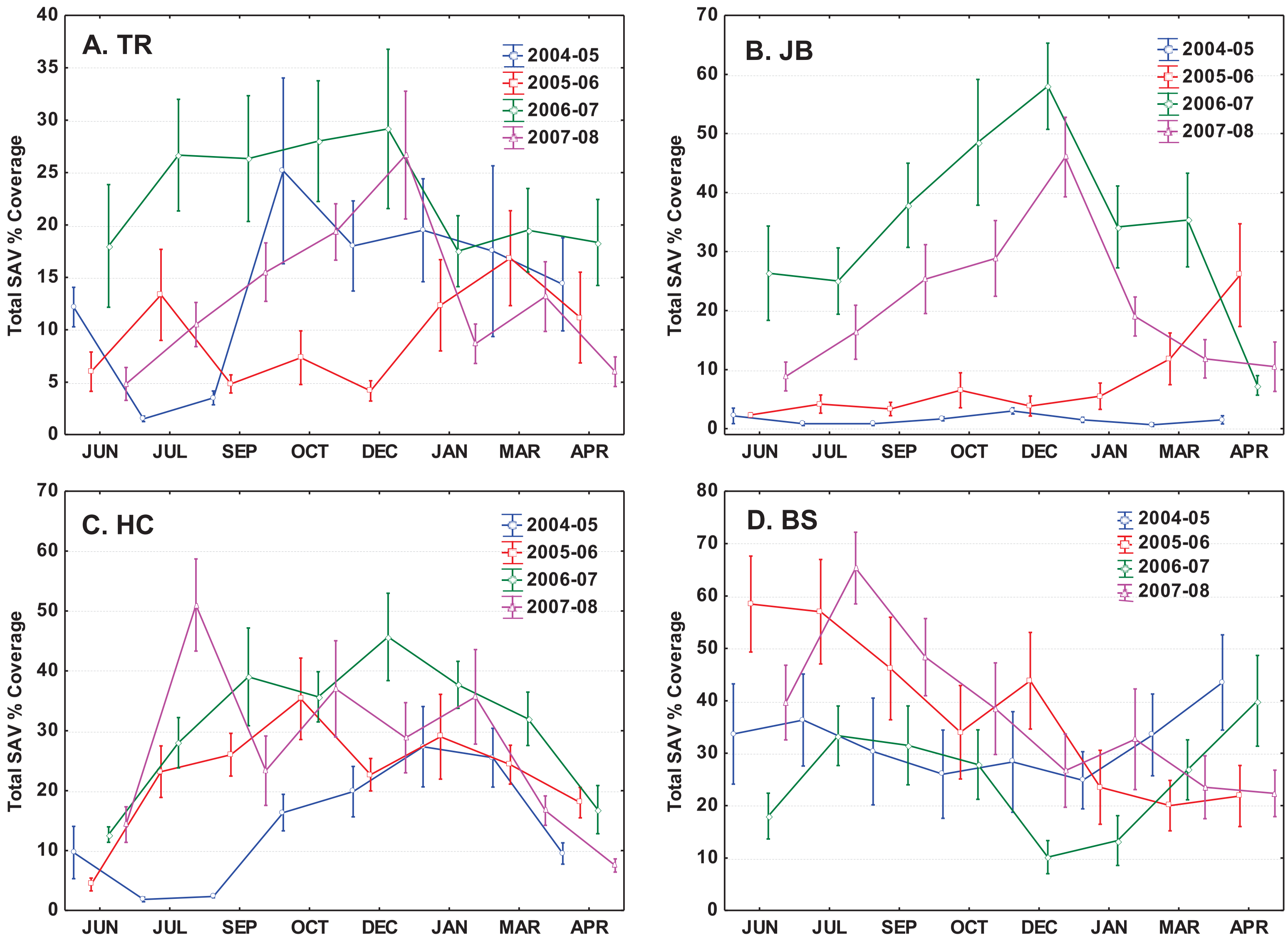


Figure 18. Mean monthly total SAV coverage (± SE) for each year of the reporting period plus the year prior to the reporting period for each of the four sites where SAV data were collected

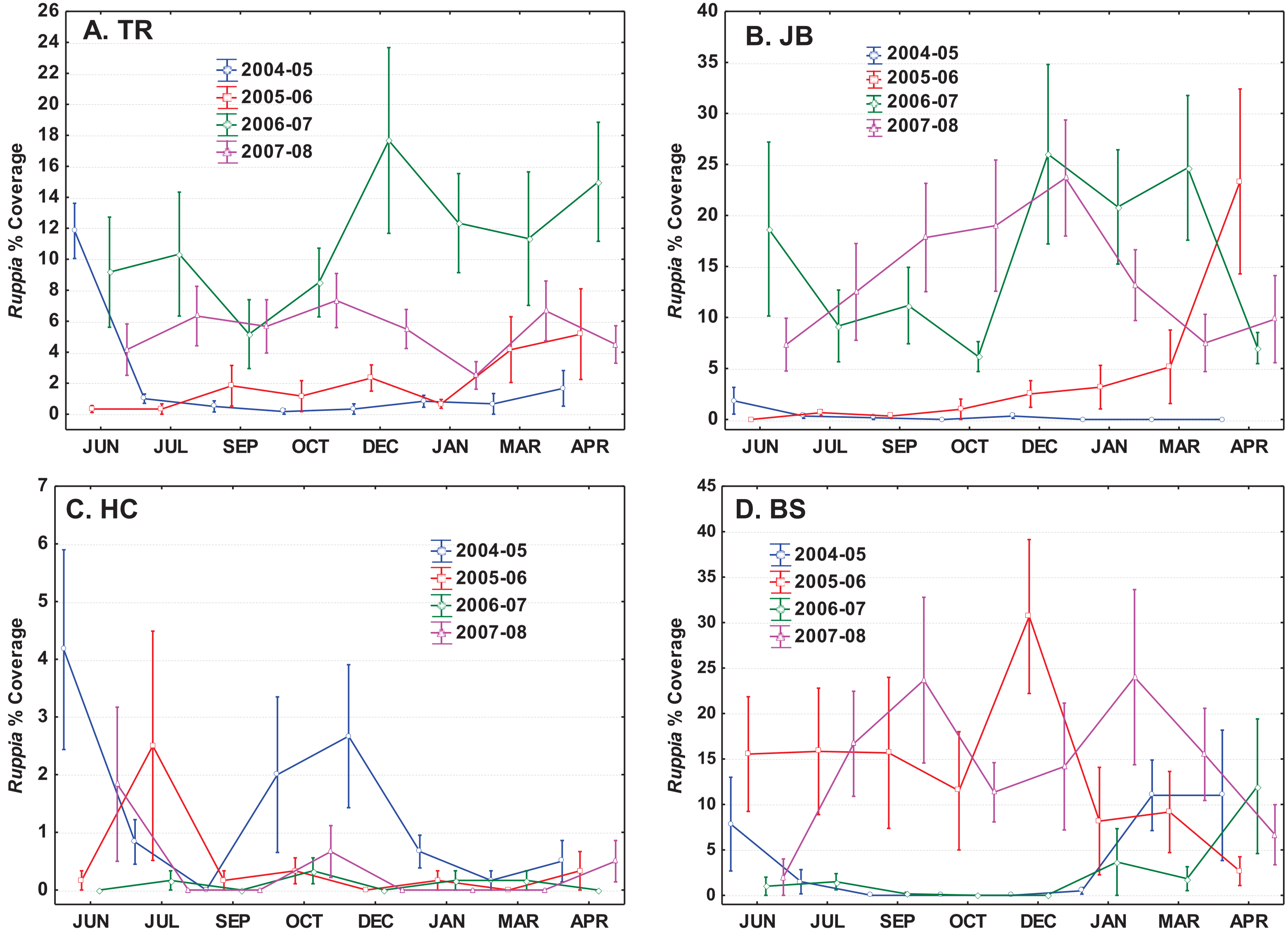


Figure 19. Mean monthly *R. maritima* coverage (± SE) for each year of the reporting period plus the year prior to the reporting period for each of the four sites where SAV data were collected

Results of the ANOVA for site, month and year (***Figures 18*** and ***19***) reveal SAV coverage, for the most part, was similar between years. Monthly analysis indicated no statistically significant monthly differences in total and *R. maritima* coverage between all three reporting years when all sites were combined. Annual total mean SAV and *R. maritima* coverage also revealed similarities between years. While not statistically significant, noticeably less coverage of SAV, however, was detected in 2005-06 than the following two years. The reasoning for this is easily explained by the drought conditions of 2004-05 that caused hypersaline conditions resulting in substantial SAV die off described previously.

The 2005-06 hydrologic year began with salinity conditions much higher than what is indicated by the long-term mean. This fact is most evident at the TR site. Elevated rainfall and flow in the 2005-06 wet season subsequently ended the drought and hypersaline conditions. Following the drought and SAV die off of 2004-05, longer periods of freshwater conditions ensued through the reporting period and salinity conditions followed a „nomal" pattern between June 2005 and May 2008. These conditions allowed for an increasing trend in SAV production throughout the reporting period (***Figure 17***, ***18*** and ***19***) although the mean coverage from all sites (***Figure 17***) shows slightly less coverage in 2007-08 than in 2006-07 when all sites were combined. However, *R. maritima*, an indicator of more freshwater conditions, shows the increasing trend evenly throughout the reporting period (***Figure 17***). Based on hydrologic conditions, as predicted, we did not see a major die off of SAV during the reporting period.

Salinity and Prey Fish Production

The conceptual relationship between higher fish production with longer hydroperiods and lower, more stable salinity is based on the work of Lorenz (1999) and Lorenz (2000). Although these studies demonstrated hydroperiod and salinity are determinants of prey fish abundance, the relationships were highly complex and non-linear. These complexities make it difficult, if not impossible to simply look at, for example, mean salinity verses fish biomass. The question becomes; "which mean salinity?" Do you compare fish biomass with the 90d mean (or median) salinity prior to the sample or do you use the 30d mean (or median)? Is the number of consecutive days prior to the sample that salinity was less than some threshold (e.g. less than 1 psu) more important than mean or variance? These questions also arise for the effect of hydroperiod on fish abundance. Lorenz and Serafy (2006) used community analyses as a way to examine fish abundance-based species community structure in relation to salinity, thereby bypassing the complexities of direct relationships between the independent physical parameters and dependent biological variables. The data they used were collected as part of the same long-term monitoring program being examined in this report so we repeated their analyses for the reporting period to determine if data from the reporting period were consistent with their conclusions.

First, Lorenz and Serafy (2006) classified every species of fish collected over an eight-year period as either freshwater (collected in water with 30d mean salinity prior to the sample collection under 1 psu), oligohaline (1 to 8 psu), mesohaline (greater than 8 to 16 psu) and polyhaline (greater than 16 psu) based on the by Venice System of Estuarine Classification (Bulger 1993). Using this classification of individual species, they then classified each sample collected according to the parameters as follows, which we used as well:

1. Any sample with greater than 50 percent representation of freshwater species was placed in the freshwater category.

2. Any sample with between 10 to 50 percent representation of freshwater species was placed in the transitional category. The reason for creating a transitional category was based on the well documented finding that freshwater fishes are generally stenohaline and more constrained by the freshwater/oligohaline interface than their saline-tolerant counterparts (Myers 1949, Darlington 1957, Moyle and Cech 1988, Wagner 1999). Consequently, any significant (greater than 10 percent) representation of freshwater species indicated a distinctive community different from samples with a preponderance (greater than 90 percent) of euryhaline species.

3. Any sample with greater than 50 percent representation of oligo-, meso- or polyhaline species fell into these respective categories.

4. When no salinity group made up the majority of the given sample, the sample was categorized within the centrist category (e.g. if oligo-, meso- and polyhaline species were in approximate equal abundance, they were categorized as mesohaline).

Lastly, ANOVA was performed on fish biomass by the sample classification. Lorenz and Serafy (2006) found the samples classified as freshwater and transitional had significantly greater biomass than the other categories. They concluded lower salinity leads to a fish community of greater biomass in these habitats. An ANOVA of fish biomass by sample classification for the three reporting years of our study supports their finding that samples with freshwater and low salinity species tend to have greater biomass (***Figure 20***). We conclude increased freshwater flow to the ecotone reduces salinity and results in a fish community of greater biomass, thereby supporting the spoonbill conceptual model (***Figure 2***). Increased flow also leads to higher water levels and longer hydroperiods, thereby supporting the effect of water level on the fish abundance component of the conceptual model.

Lorenz and Serafy (2006) also documented that it could take up to three low salinity years following a high salinity event before freshwater species made up a significant portion of the species collected. ***Figures 11***, ***14*** and ***15*** indicate that both the Southern Biscauyne Bay and Cape Sable watersheds had consistently high salinities and these watersheds had a total of 38 individual fish classified as freshwater combined during the entire three-year reporting period so these watersheds were not used in the following analysis. In the other three watersheds combined, a total of 47,065 fish were collected from June 2004 through May 2008. ***Figure 21*** indicates the total percent of this catch that were freshwater species for each hydrologic year.

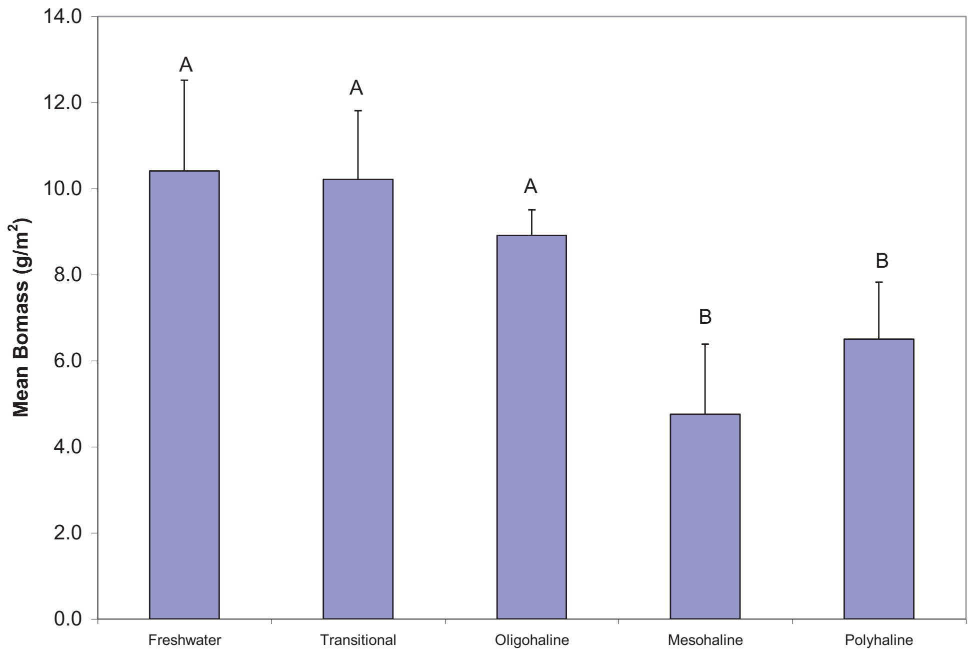


Figure 20. Mean (± SE) biomass of fish collections classified by community salinity category

Categories with different letters were significantly different.

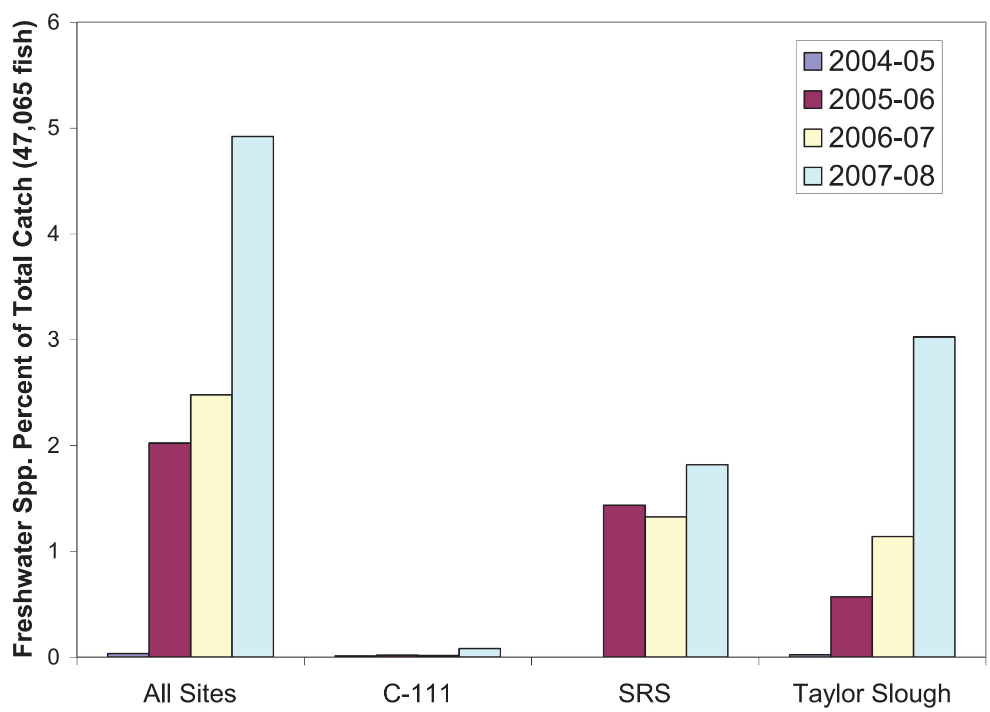


Figure 21. Percent of total catch from a four-year period that were classified as freshwater species by year and watershed

These data support the findings of Lorenz and Serafy (2006) in that during the drought preceding the reporting period, when salinity was relatively high and variable (***Figure 11***), freshwater species made up less than 0.2 percent of the total catch. With the end of the drought in 2005-06, and the onset of longer periods of fresh water and low salinity conditions, the percentage of freshwater species increased in each consecutive reporting year, both overall and in each of three watersheds. This supports the finding of Lorenz and Serafy (2006) that it can take up to three years of low salinity conditions for these species to return to significant representation following a high salinity event. We conclude overall abundance of prey increased in each successive year of the reporting period as the percent of freshwater species, which is an indicator of fish abundance based on the data in ***Figure 21***, increased over this period.

Dry Season Low Water Levels, Fish Concentrations and Spoonbill Nest Production

Lorenz and Frezza (2007) examined 25 spoonbill nesting cycles in Florida Bay from 1982 to 2004 in effort to determine what management operations favored nesting success or nesting failure. This was done by comparing nest production in the northeastern and northwestern colonies (***Figure 5***) using the northwestern colonies as a control since they are largely unaffected by operations of the South Dade Conveyance System of which the C-111 canal is part (***Figure 4***). Of the 25 cycles, 19 occurred during the traditional nesting period (November to March) while six were second nesting attempts that occurred later in the season. Out of the 19 primary nestings investigated for the northeastern subregion, only seven were successful. A successful nesting cycle is defined as having nest production greater than one chick per nesting attempt. Successful nesting only occurred during drought years or the year following a drought. As discussed by Frederick and Ogden (2001), droughts and the year following droughts appear to present ideal conditions for wading birds throughout the Everglades landscape. Since the completion of the Southern Dade Conveyance System in 1984, the northeastern colonies were only successful under drought related conditions. Prior to construction and operation of the system, these colonies were frequently successful in non-drought circumstances as is currently experienced in the northwestern colonies (Lorenz et al. 2002). This led to the conclusion that operation of the South Dade Conveyance System negatively affects spoonbills nesting in northeastern Florida Bay except under ideal conditions. Of the 12 failed primary nestings, it appeared that half were naturally occurring events that may have been exacerbated by operation of the South Dade Conveyance System. The other six failed attempts appeared to have resulted directly from events that were caused or augmented by operation of the South Dade Conveyance System.

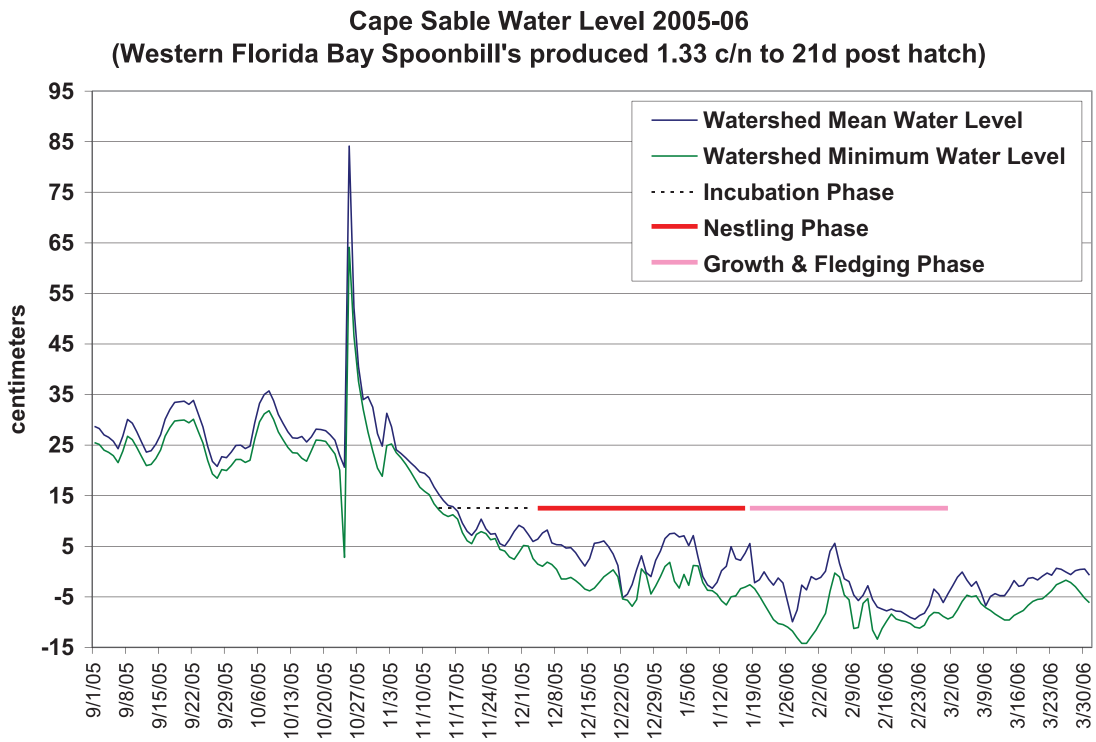
Lorenz (2000) documented prey based fishes on the spoonbills primary foraging grounds begin to concentrate in dry season refuges (i.e. deeper creeks and pools in the wetland) when surface water levels on the ephemeral wetlands surrounding these refuges drops to 12.5 cm, referred to as the prey concentration threshold. He also documented if water levels increase above the prey concentration threshold while spoonbills are feeding their chicks, the nesting attempt fails because adult spoonbills can not acquire enough food to sustain the high energetic demands of their rapidly growing young. Lorenz and Frezza (2007) indicated three ways conveyance and discharge from the South Dade Conveyance System, specifically through the C-111 canal, can affect water levels at the foraging sites, thereby reversing the drying front and increasing water levels above the prey concentration threshold. Reversals in the drying front occurred when conveyance and canal releases were pulsed resulting in abrupt increases in water levels. Although there were clear examples of anthropogenic reversals, most were preceded or accompanied by natural events that confounded the determination of the causes of these reversals. The second type of impact caused by water management operations occurred when conveyance was more constant through time resulting in a steady release of water from the canal, which slowed or stopped the natural draw down process. It appeared that slowing of the draw down was the most common result of conveyance that caused mortality in spoonbill chicks. The third way that the conveyance system affected the foraging grounds was by creating artificially low water levels early in the dry season. When certain criteria are met, managers abruptly stopped water deliveries to Taylor Slough and the C-111 canal resulting in rapidly falling water levels on the foraging grounds. This stimulated spoonbills to begin nesting earlier in the season during the transitional period between wet and dry seasons than under more natural conditions. This created a scenario in which spoonbills prematurely nested while there was still a strong possibility of severe rainfall events. When this occurred, the rainfall and subsequent management activities (i.e. further releases into Taylor Slough and the C-111 canal) increased water levels, resulting in eggs hatching during a period of undesirably high water conditions and, ultimately, in nest failure.

For the three-year reporting period, we examined nest production as a function of water levels on the various foraging grounds during the spoonbill nesting cycles to see if conditions were favorable for fish concentrations. We also examined fish density at the sampling sites to document the availability of prey during the nesting period. This information was then used to make inferences about nest production based on water levels and prey availability. We used the conditions and nest production of the northwestern colonies (***Figure 5***) as a control for the effects of water management on the northeastern colonies since the northwestern colonies are unaffected by operations of the South Dade Conveyance System.

It must be noted that the direct use of prey availability by presenting total fish number or biomass per m^2 is confounded by the fact each of the sites have dry season deep water refuges that drain different sized watersheds. For example, the dry season refuges we sample at the BL site is the catchment for an ephemeral wetland more than 100 times the size the one at the CS site, however, the CS site has many more catchment refuges than the BL site. As a result, the catch at BL is consistently much higher than at CS. From the perspective of a foraging spoonbill trying to feed chicks, both situations may be invaluable. In the area of CS, numerous small packets of concentrated prey are present over a longer time period as the drying front moves through the wetland. The higher overall concentration of fish at BL may last longer than any of the individual small packets at CS but the sequence of drying refuges at CS may last far longer than at BL, providing a more consistent food resource through time. In an effort to standardize the size of the catchment area that each of the sites drained, we calculated the total biomass collected over the three-year period at each site and used the percent of this total collected in each individual sample. Within each watershed, we calculated the mean percent catch across the sites within the watershed to get an index of available biomass for each sample collection within the watershed.

In 2005-06, spoonbills nesting in the northwestern colonies produced 1.3 chicks per nesting attempt (c/n). ***Figure 22*** presents the mean and minimum water levels at the fish sampling sites for the Cape Sable and Shark River Slough watersheds, which serve as the foraging grounds for spoonbills nesting at these colonies during the 2005-06 nesting cycle. The minimum water level is important because it indicates whether at least one of the sites within a watershed were below the prey concentration threshold). ***Figure 22*** indicates that from the time the first egg hatched through to when the chicks fledged, water levels for the Cape Sable sites were well below the prey concentration threshold and at the Shark River Slough sites, water levels were at or below the prey concentration threshold for most of this period. These conditions likely explain the success of the spoonbill nesting in 2005-06. In stark contrast to this observation, however, the fish samples collected during the nesting period indicate incredibly low availability of prey (***Figure 23***) During the nesting period (December 2005 - February 2006) fish were virtually absent from the Cape Sable sites and were some of the lowest collected during the reporting period at the Shark River Slough sites (***Figure 23***). The likely reason for the lack of fish during 2005-06 was the lingering effects of the drought conditions of 2004-05. Fish production in 2004-05 was low as indicated by the discussion involving ***Figure 21***. Furthermore, the number of fish that survive the dry season of a drought year in the Everglades landscape are relatively few compared to normal or high water years (Loftus and Kushlan 1987, Loftus and Eklund, 1994). This is likely due to high fish mortality caused by extremely low water levels that result in dry season refuges drying up or through above average predation pressure on the few remaining wetted areas on the landscape (Loftus and Kushlan 1987, Loftus and Eklund, 1994). ***Figure 24*** shows fish numbers were the lowest at the end of the 2004-05 dry season (April) than in the three reporting years. Furthermore, the number of fish at the start of the 2005-06 wet season (June), were also the lowest of the four-year period. Simply put, very few fish were present at the start of 2005-06 so the increases in number over the course of 2005 wet season were relatively modest. With fewer fish in the system, the concentrations of fish during the 2005-06 nesting period were miniscule. Without examining the antecedent year to the reporting period, this finding might be misinterpreted.

If no prey were available, then why did the spoonbills have such a successful year? No prey was available at sites where they traditionally forage (i.e. our fish sampling sites) so the birds must have been feeding successfully at non-traditional locations. Researchers who collect the fish samples make notes of spoonbill observations at the sites during sample collections. No spoonbills were observed at BL in 2005-06 and the only observations made at LI were in February and these birds were only observed roosting not feeding. Usually, sightings of spoonbills feeding are numerous at these sites during the breeding season. Another anecdote is we had one adult spoonbill from the northwestern subregion outfitted with a satellite tracking device. This bird fed almost exclusively in Lake Ingraham during the nesting season and did not feed in the inland wetlands of Cape Sable. Frederick and Ogden (2001) attributed the success of wading birds following a drought year to two biological mechanisms: release of nutrients when a drought is broken, which stimulates productivity, and the relatively small number large fish species that compete with wading birds for prey following a drought because many would have been killed by the drought conditions. The latter is unlikely in this case because most of the larger predatory fish species at these sites are estuarine and would simply go to deeper estuarine waters and returned immediately after the drought. Given the low number of fish at the sites in 2005-06, the former seems unlikely to have affected these particular sites. The only possible explanation is some mechanism caused by the drought resulted in higher prey availability at some, as yet unknown, alternative feeding locations for spoonbills than at their traditional foraging grounds where we sample the prey base. A continuation and expansion of the satellite tagging program would help resolve the issues of where and when spoonbills utilize their foraging habitat.



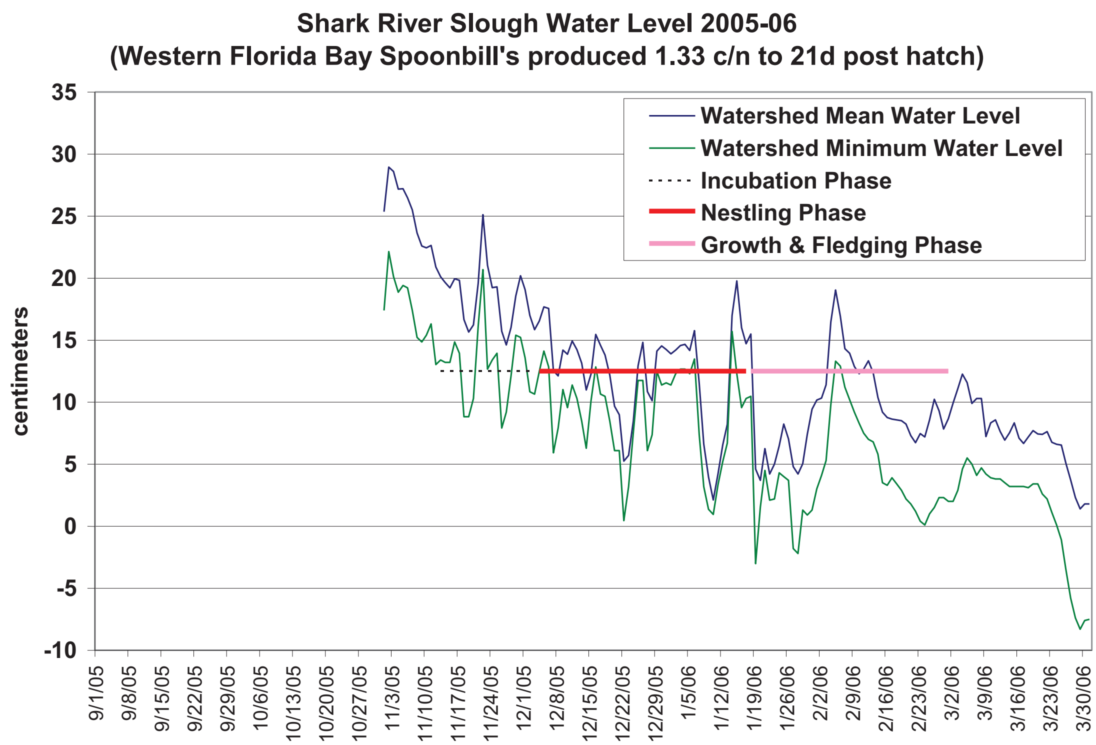


Figure 22. Daily mean and minimum water level for Cape Sable (top) and Shark River Slough (bottom) fish sampling sites from September 2005 to March 2006

The timing of spoonbill nesting for the northwestern colonies is indicated at the prey concentration threshold of 12.5 cm. Sampling did not begin in Shark River Slough until October 2005.

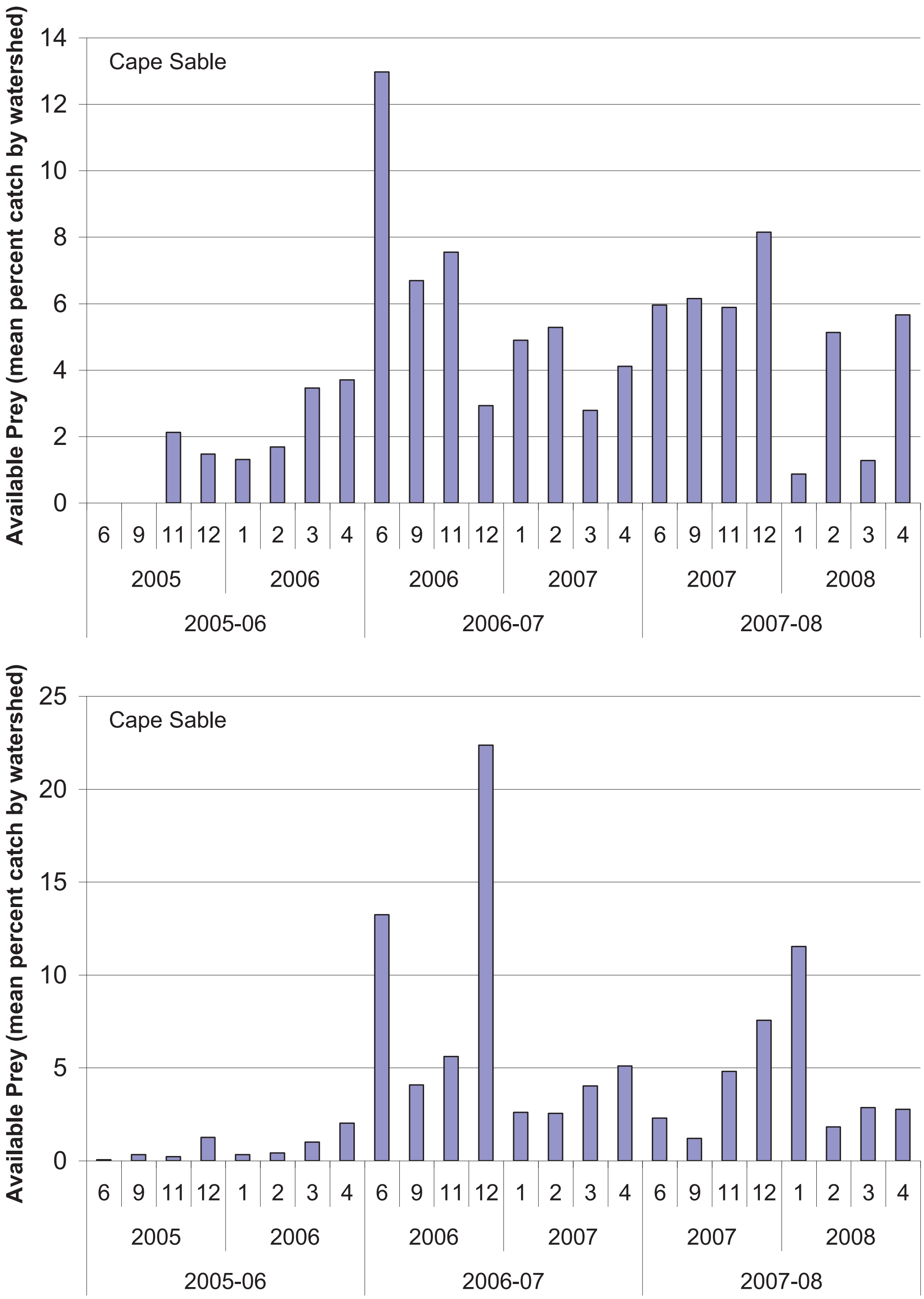


Figure 23. Mean available prey by watershed for Cape Sable (top) and Shark River Slough (bottom) expressed as a percentage of total available fish biomass for the entire three-year reporting period from June 2005 to April 2008

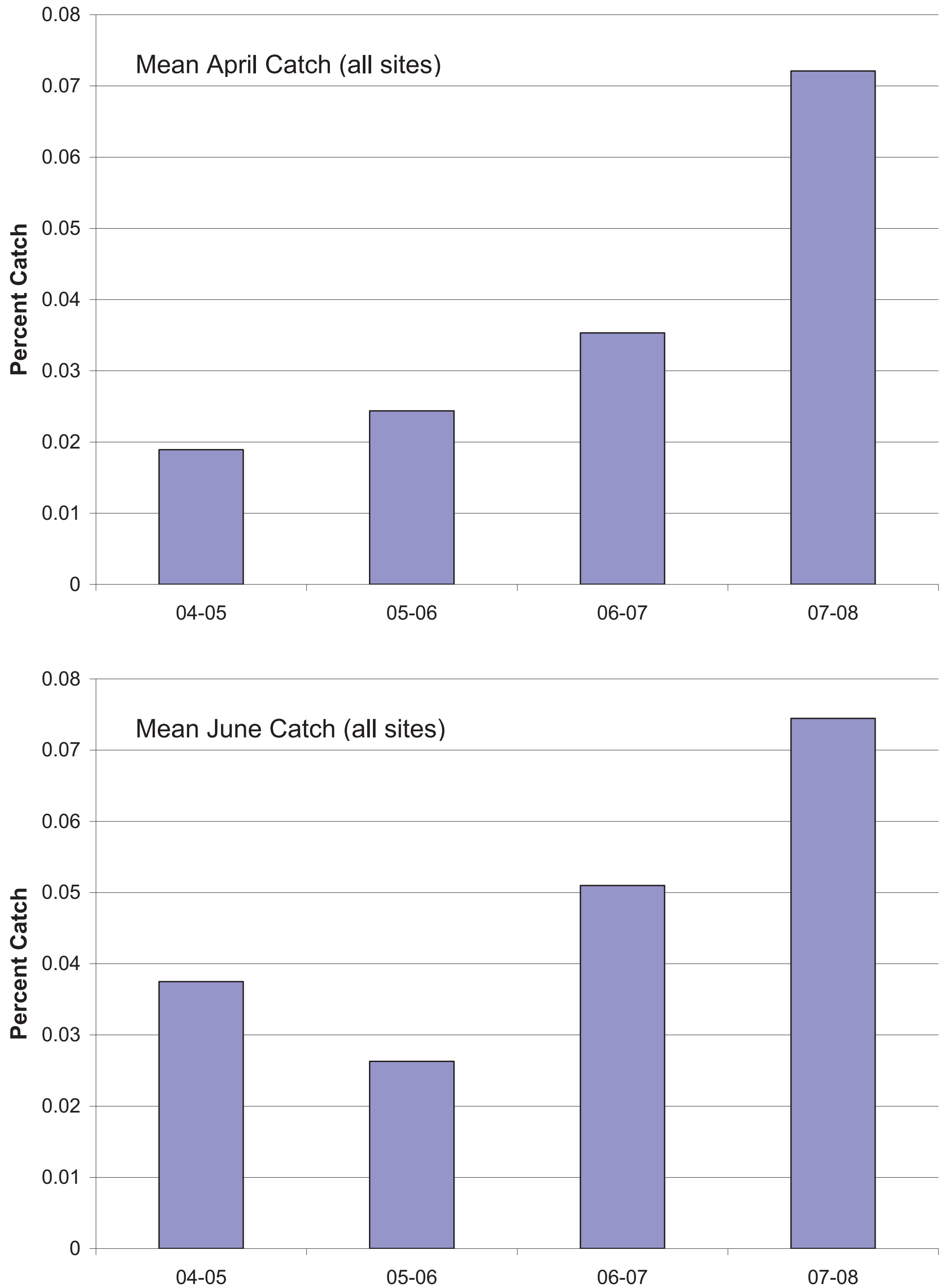
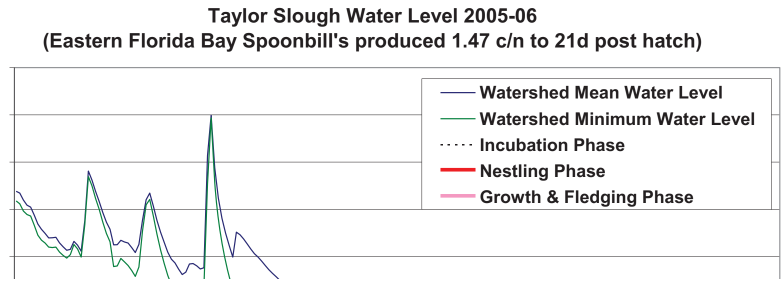
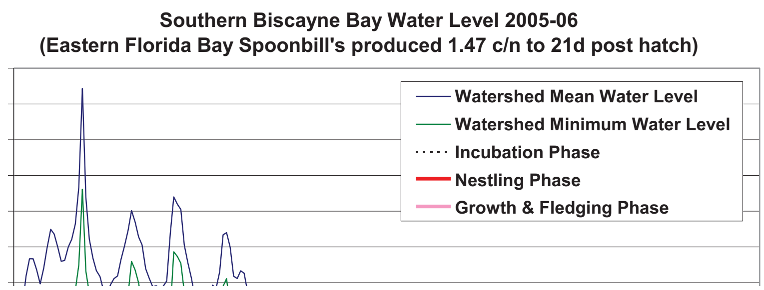


Figure 24. Percent of total fish catch collected in April (top) and June (bottom) for each year

Percent of total catch was calculated from the total number of fish collected at all sampling sites for the four year-period combined. These data indicated very few fishes were present across the landscape at the end of 2004-05 and the beginning of 2005-06.

The northeastern colonies were also successful in 2005-06, producing 1.5 c/n. From the time the first egg hatched through to fledging of the chicks, the Taylor Slough and Southern Biscayne Bay watersheds had at least one site at or below the prey concentration threshold and the minimum water level only exceeded the prey concentration threshold in the C-111 watershed for a few days during the early fledging phase of development (***Figure 25***). These data suggest ample suitable foraging areas were available for these birds during the 2005-06 nesting cycle.





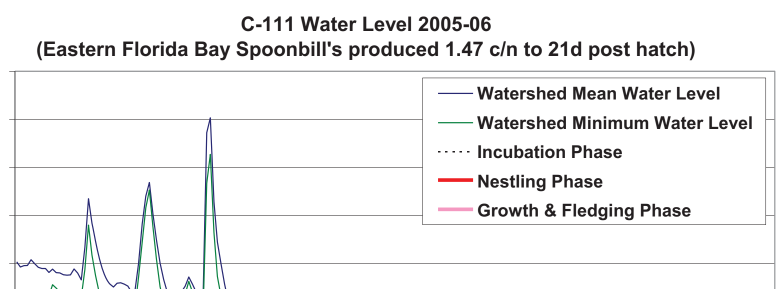


Figure 25. Daily mean and minimum water level for Taylor Slough (top), Southern Biscayne Bay (middle) and C-111 (bottom) fish sampling sites from September 2005 to March 2006

The timing of spoonbill nesting for the Northwestern colonies is indicated at the prey concentration threshold of 12.5 cm.

Figure 26 indicates that in the early phases of nesting (December and January), relatively high numbers of prey were available in Southern Biscayne Bay and similarly high numbers of fish were available in February and March of the nesting cycle in the Taylor Slough watershed. We placed seven satellite tracking devices on adult spoonbills from the northeastern colonies late in the 2005-06 nesting season (March) and all of these birds foraged in the area of Taylor Slough between our 7P and TR sites, further indicating these birds were taking advantage of the fish concentration we observed at these sites during this period.

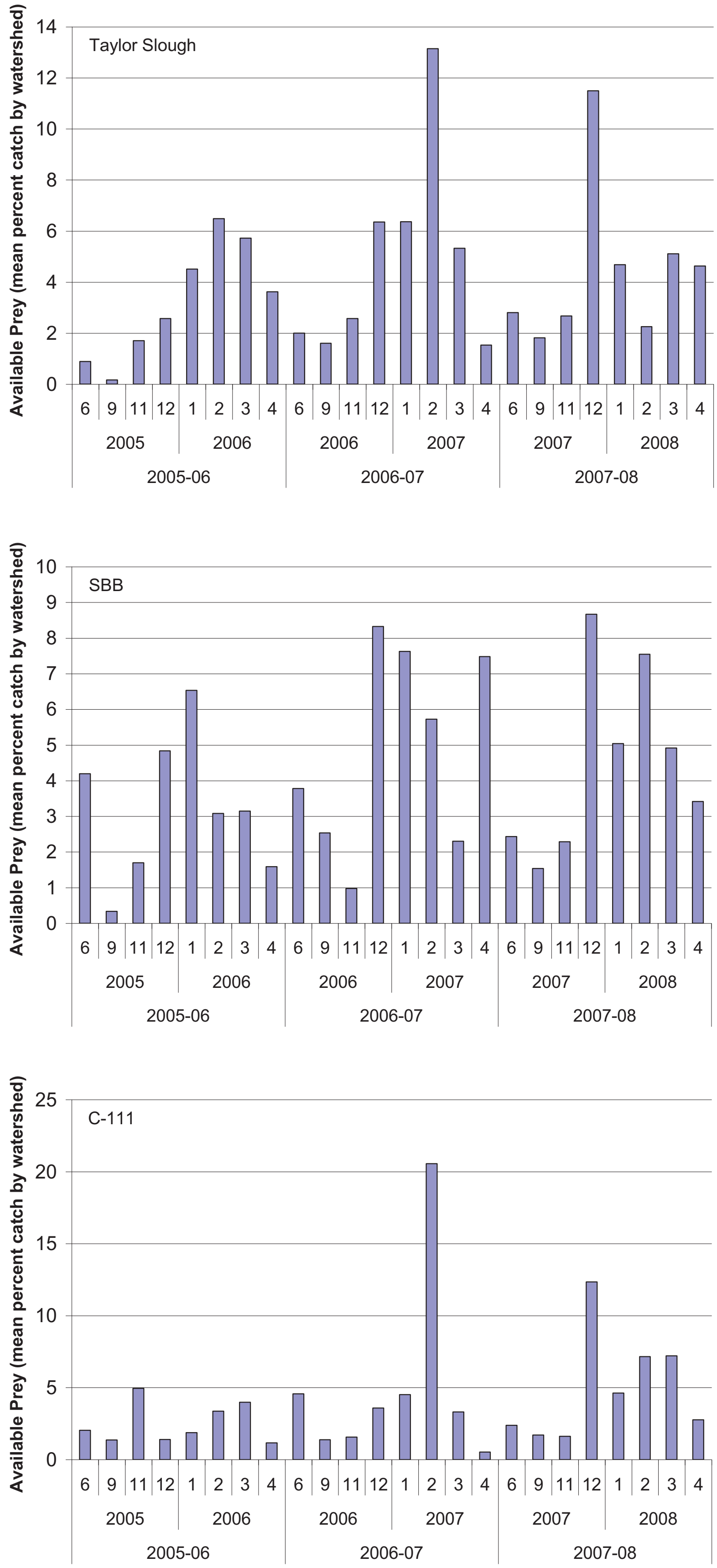
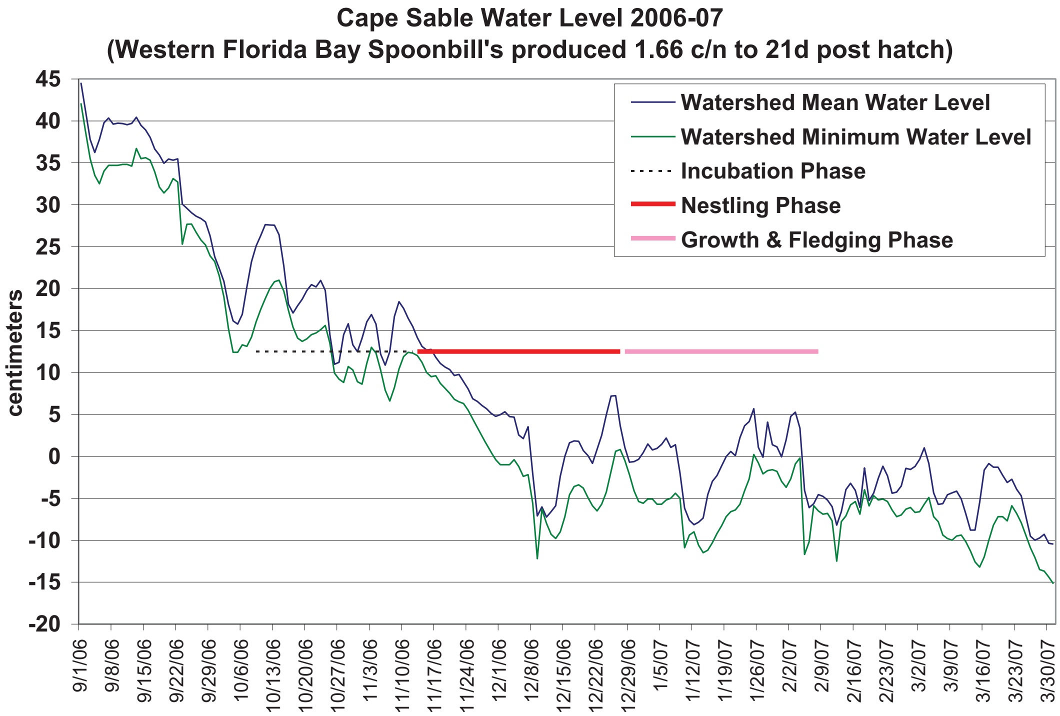


Figure 26. Mean available prey by watershed for Taylor Slough (top), Southern Biscaune Bay (middle) C-111 (bottom) expressed as a percentage of total available fish biomass for the entire three-year reporting period from June 2005 to April 2008

In both the 2006-07 and 2007-08 nesting cycles, spoonbills in the northwestern subregion were successful in nesting producing 1.7 and 1.8 c/n respectively. As in 2005-06, hydrologic conditions were ideal in both the Cape Sable and Shark River Slough watershed with water levels at or below the prey concentration threshold at least at one site within each watershed for the entire nesting period with the exception of the first few days after the first egg hatched in 2006-07 in the Shark River Slough watershed (***Figures 27*** and ***28***). In 2006-07, nesting activity ran from November through January and fish availability was relatively high in Shark River Slough in November and January of the nesting cycle and was the highest for the reporting period in December at the Cape Sable sites (***Figure 23***). There was also moderately high availability of prey at Cape Sable sites in November (***Figure 23***). Use of a combination of these watersheds through the nesting cycle likely explains the high production of spoonbills in 2006-07. In 2007-08 the nesting activity ran from December through February. The Cape Sable sites had moderately high fish availability in December and the third highest for the reporting period in January and then one of the lowest number of available fish in February (***Figure 23***). In contrast, the second highest estimate of fish availability for Shark River Slough occurred in December but then dropped to its lowest point in January and then rebounded in February (***Figure 23***). These complementary periods of high and low fish availability between the two watersheds likely explains the high success of spoonbills in 2007-08.

At the northeastern colonies in 2006-07, the nesting period ran from December to March and was considered successful although the production was just under 1.0 c/n (0.96 c/n). Although ***Figure 29*** indicates that all three watersheds were above the prey concentration threshold early in the nestling phase. The periods above the threshold in the Taylor Slough and Southern Biscayne Bay watershed were out of phase with these periods above the threshold in the C-111 watershead. At least one site in the three watersheds was below the prey concentration threshold during this period. For the remainder of the nesting period, the Taylor Slough watershed had at least one site below the threshold while the C-111 and Southern Biscayne Bay watersheds fluctuated above and below the threshold (***Figure 29***). At the Taylor Slough sites, prey availability was relatively high for the entire four months of the nesting cycle (***Figure 26***) with the highest availability of prey during the reporting period occurring in February. Similarly ern Biscayne Bay had high rates of available fish in all months except March. In the C-111 watershed, all of the months during breeding were low with the exception of February when the highest rate of fish availability for the entire reporting perio occurred in this watershed. This would suggest the spoonbills nesting at the northeastern colonies were foraging principally in the Taylor Slough watershed with some in Southern Biscayne Bay watershed and maybe some activity in the C-111 watershed during February.

In 2007-08, spoonbill nesting activity in the northeastern colonies occurred from mid-December to mid-March and produced 1.8 c/n. ***Figure 30*** indicates that at least one site in the Taylor Slough watershed was below the prey concentration threshold for this entire period, while the C-111 sites fluctuated above the threshold during the first few days post hatching and then remained at or below the threshold for the remainder of nesting except for on two-day period where the threshold was slightly exceeded. In contrast, sites in Southern Biscayne Bay exceeded the prey concentration threshold for most of the critical nestling phase then remained below the threshold for the development phase. The second highest rate of prey availability for the reporting period in the Taylor Slough and C-111 watersheds occurred in December (***Figure 26***).



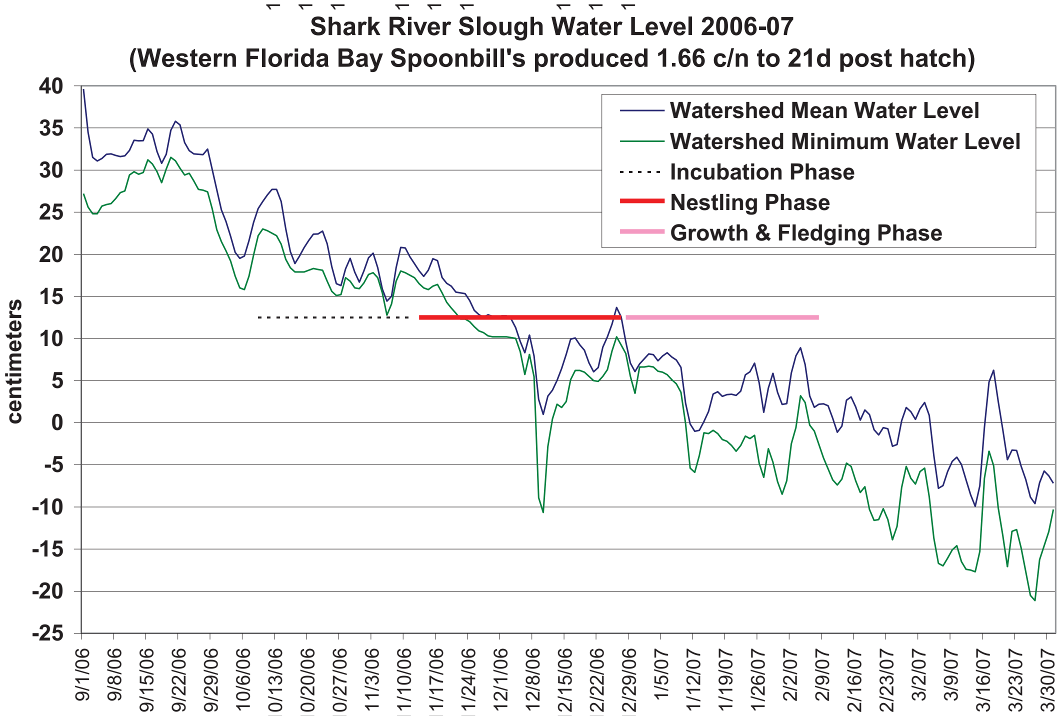
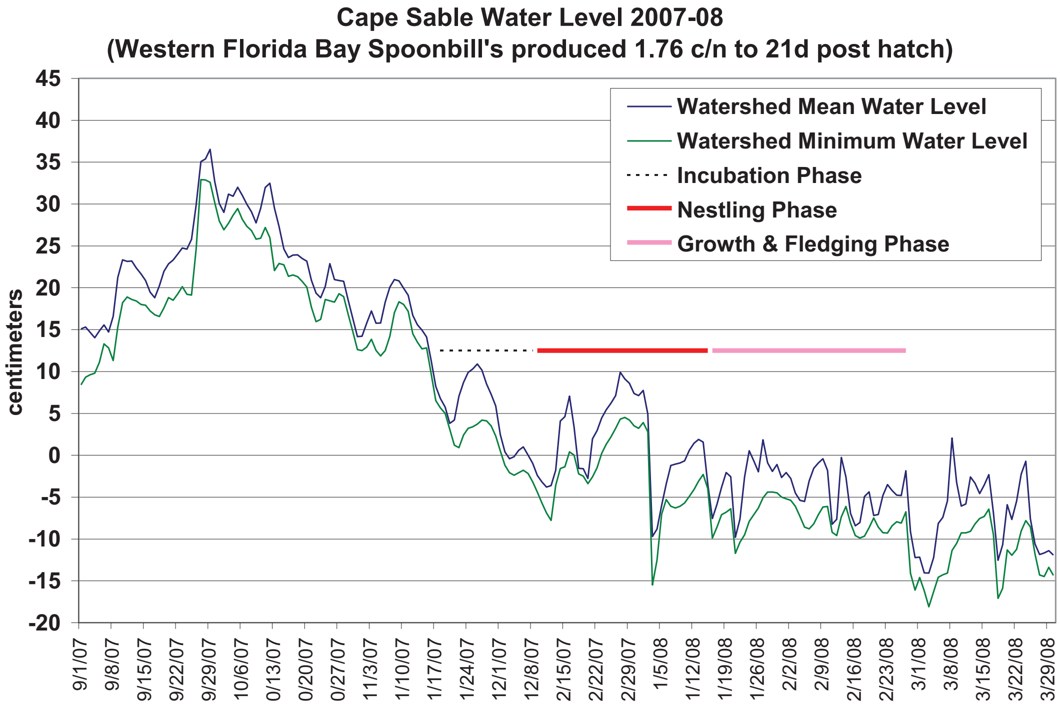


Figure 27. Daily mean and minimum water level for Cape Sable (top) and Shark River Slough (bottom) fish sampling sites from September 2006 to March 2007

The timing of spoonbill nesting for the Northwestern colonies is indicated at the prey concentration threshold of 12.5 cm.



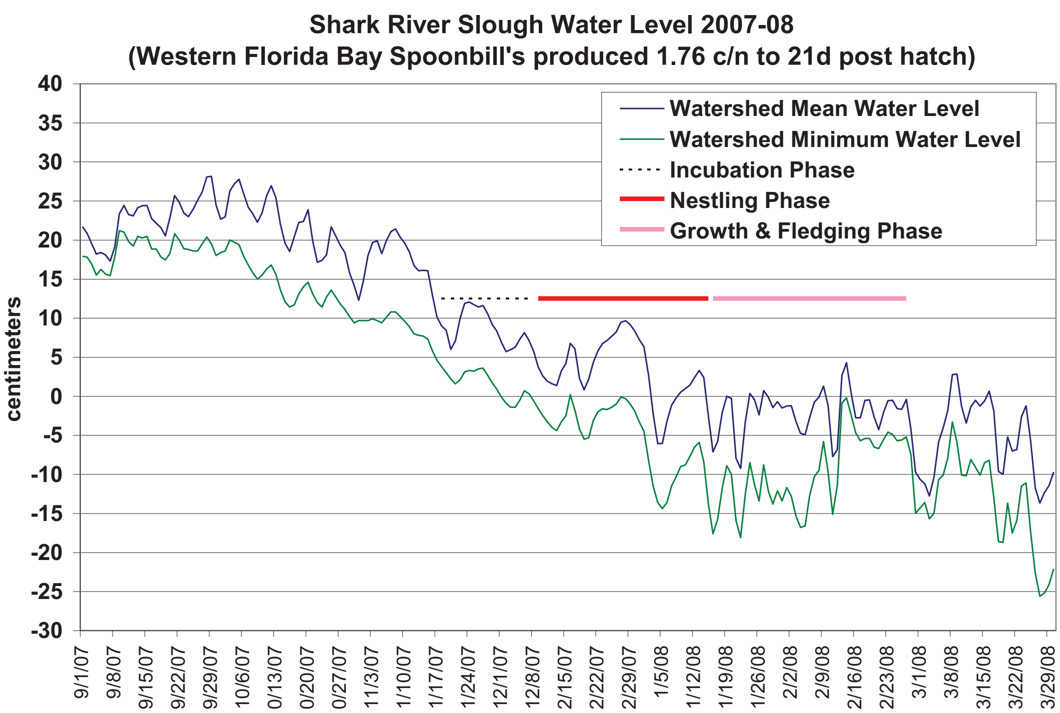
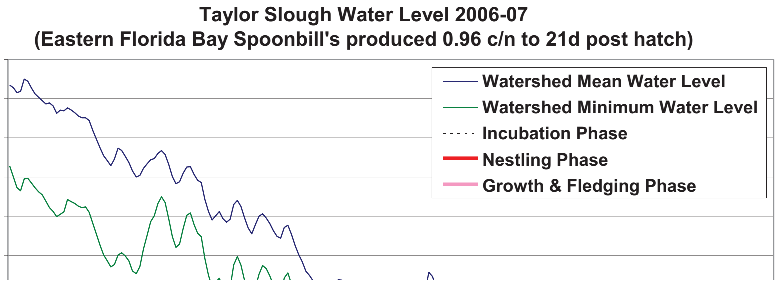
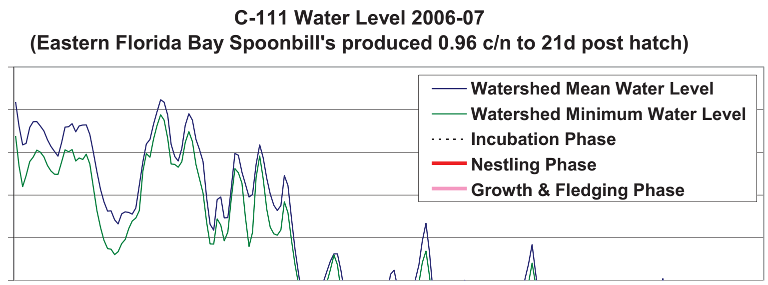


Figure 28. Daily mean and minimum water level for Cape Sable (top) and Shark River Slough (bottom) fish sampling sites from September 2007 to March 2008

The timing of spoonbill nesting for the northwestern colonies is indicated at the prey concentration h of 12.5 cm.





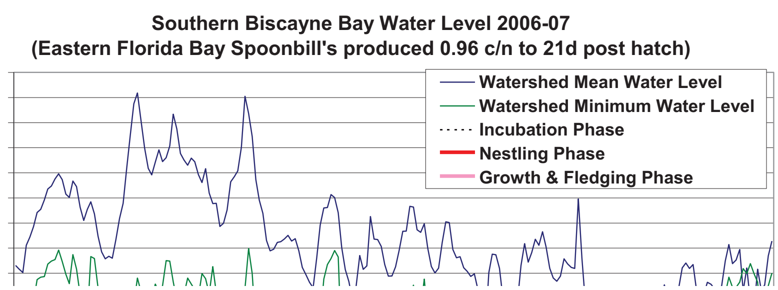
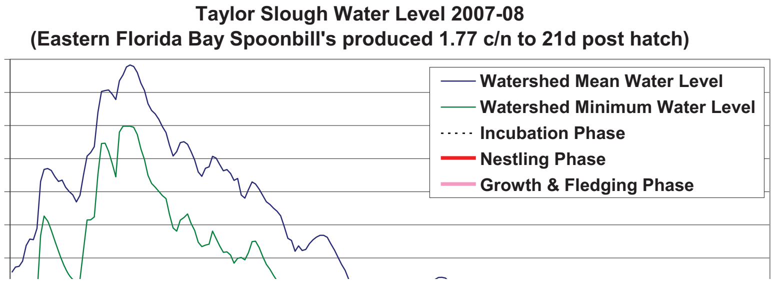
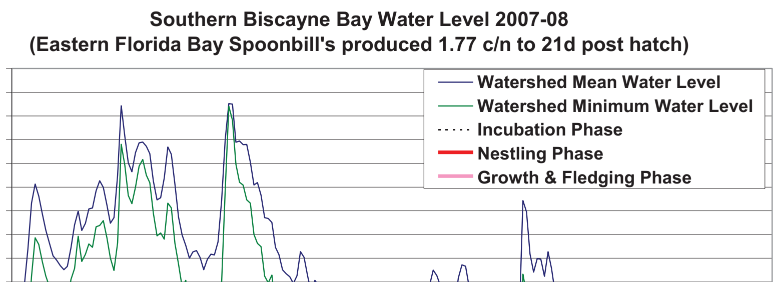


Figure 29. Daily mean and minimum water level for Taylor Slough (top), Southern Biscayne Bay (middle) and C-111 (bottom) fish sampling sites from September 2006 to March 2007

The timing of spoonbill nesting for the northwestern colonies is indicated at the prey concentration threshold of 12.5 cm.

The Taylor Slough watershed had moderately high availability in January and March and the C-111 watershed had moderately high levels of prey in February and March. ***Figure 26*** indicates Southern Biscayne Bay had high to moderately high levels of prey availability throughout the nesting cycle. This was likely due to our unwittingly making fish collections at the Southern Biscayne Bay sites in December and January during the low water periods of these months (***Figure 30***). These data suggest the mosaic of habitats within these three watersheds provided good foraging conditions for spoonbills throughout the 2007-08 nesting cycle although each of the watersheds may have been used at different times.





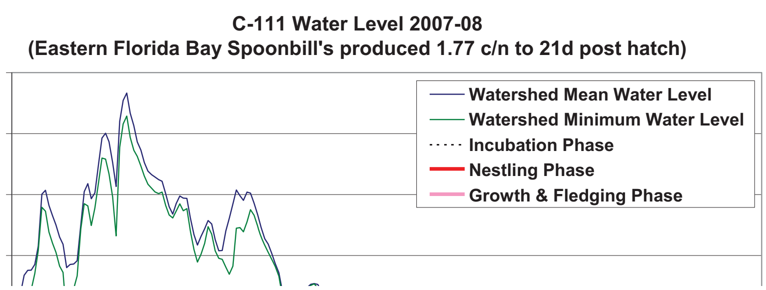


Figure 30. Daily mean and minimum water level for Taylor Slough (top), Southern Biscayne Bay (middle) and C-111 (bottom) fish sampling sites from September 2007 to March 2008

The timing of spoonbill nesting for the northwestern colonies is indicated at the prey concentration threshold of 12.5 cm.

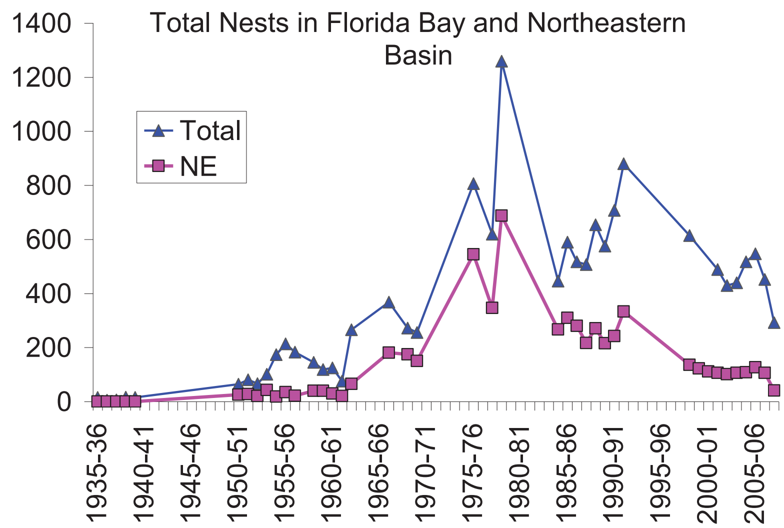


Figure 31. Total number of nests for Florida Bay and the northeastern colonies since 1935, indicating the decline in nest numbers since the completion of the South Dade Conveyance System in 1982, especially in the northeastern region

According to Lorenz and Frezza (2007), the northeastern colonies spoonbills failed in 12 of 19 primary nesting cycles between 1982-83 and 2004-05 and were only successful in drought years and years following a drought. This report indicates three successive years of successful nesting beginning in 2005-06. Although we have not reported the data elsewhere, the just completed 2008-09 nesting period in the northeastern subregion was one of the highest production years since the completion of the South Dade Conveyance System, making four consecutive successful years with only one of those years being a drought year or the year following a drought. Although much of the reason for this change is due to naturally ideal rainfall patterns for spoonbills (high rains during the wet season with little or no rain during the dry), changes were also made in water management decision making that included consideration for periods of spoonbill nesting. Although the Lorenz and Frezza (2007) report regarding the 1982 to 2004 nesting data was not completed until 2007, the findings were presented to scientists and engineers at the South Florida Water Management District prior to the 2005-06 nesting cycle. After that presentation, the decision making team that decides weekly operations was in conference with the spoonbill scientist during breeding season before any operational changes were made with the specific purpose of not disrupting the prey concentration-foraging relationship. These conferences resulted in predictably persistent dry downs with no reversals caused by out-of-phase releases from the C-111 canal (**Figure 9**). Although this is great step in recovering spoonbills and all the higher trophic levels they represent as an indicator species, it is not the ideal situation. Ultimately, restoration should result in these processes occurring without such conferences, however, in the interim, this is a good way to minimize disruptions to the Taylor Slough and C-111 watersheds.

Spoonbill Nest Numbers and Locations

Spoonbills were extirpated in Florida by the late 1800s during the bird plume trade era (Lorenz et al. 2002). In 1935, five spoonbill nests were found on Bottle Key in Florida Bay by Audubon wardens (Allen 1942). Since that time, Audubon researchers have been keeping track of the number of spoonbill nests in the bay. Although there are gaps in these data, spoonbills gradually recovered from being hunted and, by the late 1970s, more than 1,250 nests were located throughout the bay with more than half of these nests in the northeastern subregion (***Figure 31***). Following the completion of the South Dade Conveyance System in 1984, the number of nests began to decline and in 2001-02 the total nests dropped below 500 with only about 100 nests in the northeastern subregion (***Figure 31***). Lorenz et al. (2002) made convincing arguments that these declines were attributable to the effects that operations of the South Dade Conveyance System had on the primary foraging grounds of the northeastern colonies and the subsequent decline in nest productivity as a result. For the reporting period, the total number of spoonbill nests in Florida Bay was 547, 457 and 292 in 2005-06, 2006-07 and 2007-08, respectively. The number of nests in the northeastern subregion was 127, 106 and 41 nests for the respective years. 2007-08 was the lowest nesting year in Florida Bay since 1969-70 and the lowest number in the northeastern subregion since 1962-63, a year when spoonbills had not yet recovered from the plume hunting era (***Figure 31***).

In 1935, spoonbills were one of only a few species selected by the National Audubon Society for intense protection and study because they had not responded to the post-plume era like other birds by recovering to numbers of the pre-plume era (Allen 1942, Ogden 1994). By the time the original five spoonbill nests were discovered, all of the other wading bird species that historically nested in the Everglades had rebounded and were periodically forming super colonies of more than 100,000 nests in the Shark River Slough and Taylor Slough estuaries (Ogden 1994). Although Audubon researchers continued to study spoonbills and documented their relatively slow increases compared to other wading bird species, the reason the spoonbills were so slow to recover was never fully understood. In the early 1970s, spoonbills began to nest in Tampa Bay on Florida"s east coast. Audubon researchers kept track of the number of nests in Tampa Bay as well (***Figure 32***). Similar to the recovery in Florida Bay, the increase in nest numbers in Tampa was slow but steady and peaked in 2005-06 with almost 600 nests. Tampa nest numbers exceeded Florida Bay number in 2006-07 for the first time (***Figure 32***).

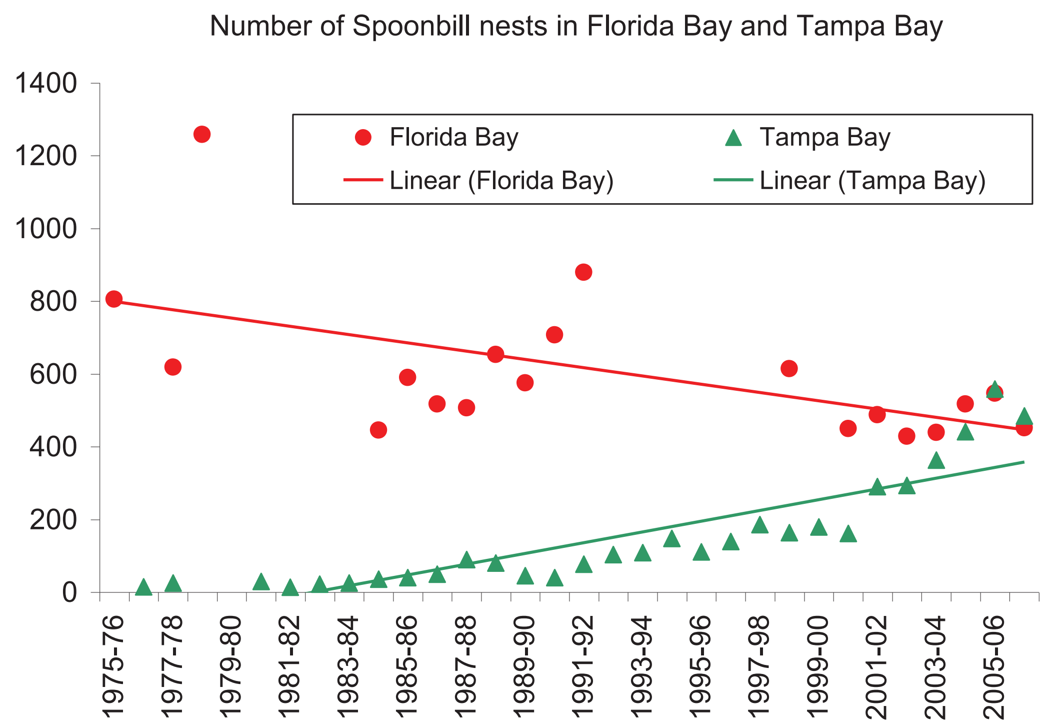


Figure 32. A comparison of nest numbers in Florida and Tampa Bays since spoonbills returned to nesting colonies in Tampa Bay in 1975

Beginning in 2003-04, we began banding spoonbills in the nesting colonies of Florida and Tampa Bays and from March 1995 to March 2006, 20 adult spoonbills from Florida Bay were outfitted with satellite tracking devices. The results of this study have bearing on why spoonbills recover their numbers so slowly compared to other species and why the numbers in Florida Bay continue to decline even while, overall, nest numbers in the rest of Florida are increasing. One of the hypotheses that spurred this study was spoonbills nested in Florida Bay in November or December, completed nesting by March and then moved up the coast to Tampa Bay to nest again beginning in April. The data collected by this study disproved this hypothesis with none of the 20 telemetered birds nesting in Tampa Bay and very few banded birds from Tampa Bay being re-sighted in Florida Bay and vice versa.

Another goal of the banding/tracking program was to collect demographic data such as age at maturity and life span of spoonbills. Previous published estimates of age at sexual maturity were 3.5 years and a lifespan of seven years but these estimates were based on inference rather than data. Based on re-sightings within nesting colonies, we now estimate that spoonbills reach sexual maturity in 2 to 5 years after hatching and that, although highly variable within individual birds, the mean age at sexual maturity is 3.5 years after hatching. de Magalhães et al. (2007) developed a calculation to estimate maximum age of birds from mean age at sexual maturity by examine these known parameters from 69 bird species. Using their calculation and a mean age at sexual maturity of 3.5 years, maximum spoonbill lifespan is estimated to be 32 years. In support of this life span estimate, the closely related Eurasian spoonbill (*Platalea leucorodia*) has been documented to live 28 years in the wild (De le Court and Aguilera 1997). Furthermore, as part of the satellite tagging process, we coincidentally captured a bird on its nest at Tern Key that was previously banded by Audubon researchers as a nestling at Tern Key in 1990. The bird was 16 years old and the tracking data verified that it nested again at Tern Key the following year making it a breeding adult at age 17. The transmitter failed shortly after, but the bird was re-sighted in Florida Bay two years later at age 19 and presumably breeding again. This estimate and the longevity record of 19 years prove that this species is longer lived than previously thought.

The 16-year old bird recaptured at Tern Key as a breeding adult was also the first record of natal colony nesting fidelity in spoonbills. Preliminary results from the banding data also suggest spoonbills have a high degree of fidelity to nesting at their natal colonies. At the time of this report, we have re-sighted 23 spoonbills in adult breeding plumage during the breeding season for both Tampa and Florida Bays combined. Of these 23, all but two were documented as having some close affiliation with their natal colony (***Figure 33***). Although other wading bird species have been documented to nest at their natal colony, there is not a strong fidelity to do so (J. Ogden, D. Gawlik pers. comm.) and other species are highly opportunistic in nest sight selection based on local conditions (Bancroft et al. 1994). In a banding study of Eurasian spoonbills, De le Court and Aguilera (1997) documented that 94 percent of the 590 total re-sightings were within 100 kilometers (km) of the natal colony while only 2.4 percent of the re-sightings were of birds further than 100 km from their natal colony during the reproductive season. Their conclusion was that spoonbills use the region where they we hatched preferentially for reproductive sites as adults and relatively few birds chose other suitable regions for nesting (these birds they called dispersers). These findings support our preliminary conclusions that spoonbills seem to be unique among the wading bird species of southern Florida in their behavior of returning to their natal colony to nest. This fidelity to natal colonies likely explains why spoonbills responded to post-plume hunting so much more slowly than the other wading bird species because colonization of new areas would be rare. It also explains the slow increase in numbers at Tampa Bay since re-colonization in 1972.

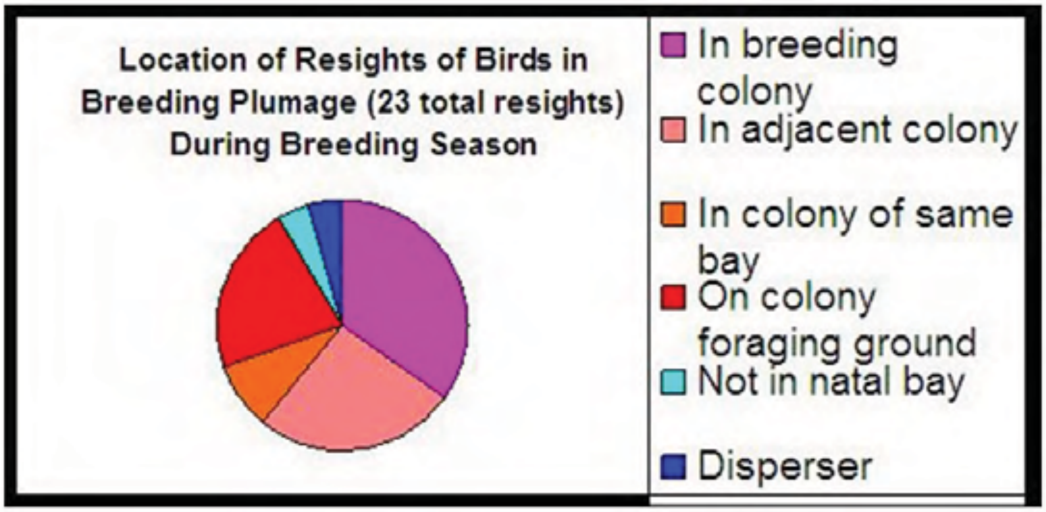


Figure 32. Preliminary results from spoonbill band re-sighting of adult birds in breeding plumage during the breeding season in relation to the natal colony of the individual bird re-sighted

Disperser indicates birds that were observed nesting at a location with no relation to the bird‟s natal colony.

Given that estimated maximum age of roseate spoonbills is 32, we suggest the average age at mortality is probably closer to 25 years. Based on this, most of the spoonbills hatched in Florida Bay prior to completion of the South Dade Conveyance System (before nest numbers began to decline) are now dead. Also, since there were only seven successful nesting years between 1982 and 2004 (Lorenz and Frezza 2007), it is highly unlikely that the adults from the pre-conveyance system period are being replaced at the same historical rate that occurred during the initial recovery period from 1935 to 1978. Given that the number of chicks reaching adulthood is less than the loss of adults to mortality, and nesting population in Florida Bay is largely closed to immigration, we conclude this explains the decline in nest numbers and the lowest number on record during the reporting period. We predict the four consecutive years of successful reproduction reported here should result in a reversal in the decline of nesting activity over the next few years as the chicks hatched over the last 4 years reach sexual maturity and return to their natal colonies. If these preliminary results hold true, the actions of the South Florida Water Management District in considering spoonbills in water management decisions, (which, at least in part, led to those years of success) may have prevented this iconic species of Florida Bay from becoming locally extinct. Although these efforts are much appreciated for the time being, ultimately CERP‟s goal is to restore the system such that spoonbills, and the other species for which they serve as an umbrella indicator species, successfully reproduce with greater regularity without the intervention of water managers. In the short term, completion of Phase 1 of the C-111 Spreader Canal Project would be the first productive step in achieving that goal.

Conclusions

By careful analysis and inference of the data we have collected, we feel that we have demonstrated that the linkages proposed in the conceptual models (***Figures 1*** and ***2***) have been authenticated.

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CHAPTER 9 SOUTHERN COASTAL SYSTEMS MODULE

9.1 INTRODUCTION

The Southern Coastal System Module, formerly known as the Southern Estuaries Module, encompasses a very large ecologically and economically important area that includes Biscayne Bay, Florida Bay and the southwest Florida coastal environments (***Figure 9-1***). Over the past century, water management practices and agriculture/urban development have disrupted the availability, timing and distribution of fresh water to the Southern Coastal Systems, which has significantly altered the structure and function of these ecosystems. The objective of restoration is to restore freshwater flows to the Southern Coastal Systems to the extent practical.

9.1.1 Biscayne Bay

Biscayne Bay is a shallow coastal lagoon located along the southeastern coast of Florida. 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 in four major areas: north bay, central bay, south bay and Card and Barnes Sounds. Each of the four areas has distinct physical and ecological characteristics. Twelve major conveyance canals discharge fresh water into the bay from the mainland. Tidal exchange with the Atlantic Ocean occurs through the Safety Valve, a wide series of shoals and shallow cuts in central Biscayne Bay, and through narrow cuts and creeks in other parts of the bay.

Biscayne Bay is naturally a clear water bay with tropically-enriched flora and fauna. Prior to the development of Miami-Dade County, much of the bay was bordered by mangroves and herbaceous wetlands. The bay was once hydrologically connected to the Greater Everglades ecosystem through tributaries, sloughs and groundwater flow. It possessed not only marine habitats, but also a substantial area of estuarine habitat. Because of the bay‟s shallow depths and naturally clear waters, its productivity is largely benthic-based (Roessler and Beardsley, 1974). Although the north bay is heavily impacted by adjacent development, benthic communities exist and are dominated by seagrasses intermixed with calcareous green algae. Development along the central and south bay is not as pronounced and much of the natural mangrove wetlands are still intact along the western shore and eastern barrier islands. Benthic communities in the central and south bay consist of several species of seagrasses, primarily turtle grass, shoal grass and manatee grass, and algal communities. An area of hard grounds, communities of hard and soft coral, sponges and other benthic organisms, is found in distinct patches along the middle of the bay‟s north-south axis.

Construction of major canals through the Everglades and dredging of natural tributaries and transverse glades that carried fresh water to Biscayne Bay lowered regional and coastal water tables (Parker et al., 1955), reduced water storage in the watershed, decreased groundwater flow, and eliminated natural tributaries. Drainage of the watershed and opening inlets greatly affected the natural salinity gradients and reduced or eliminated critical estuarine habitat for bay species requiring low-to-moderate salinity waters. In addition, constructed drainage systems result in pulsed, point source discharge that degraded estuarine habitat near canal mouths by creating biologically damaging zones of bottom scouring and rapid salinity fluctuation.

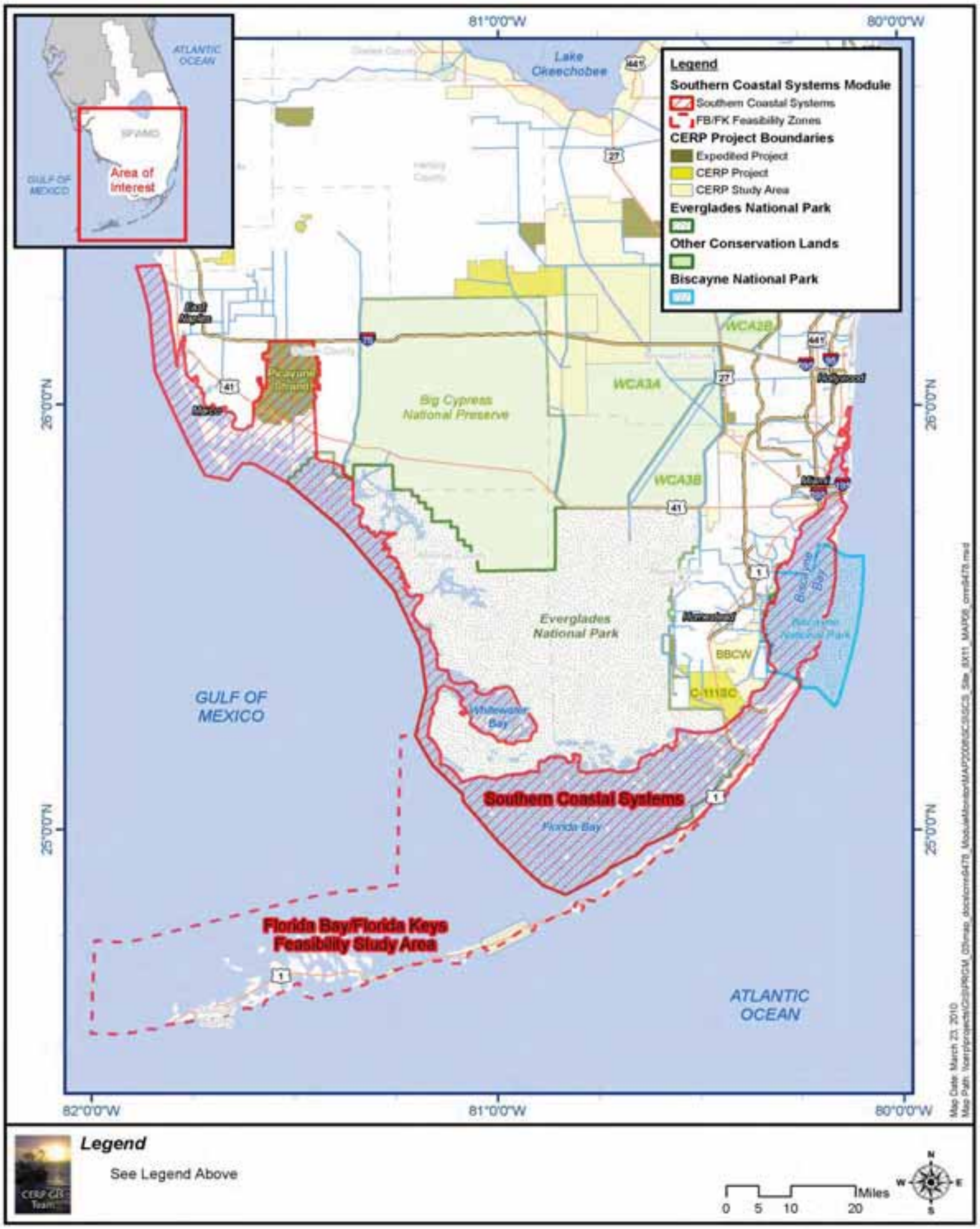


FIGURE 9-1. SOUTHERN COASTAL SYSTEMS MODULE BOUNDARY AND SURROUNDING AREAS

The bay has also been significantly affected by watershed development (Alleman et al.. 1995). Today, most new development is occurring in former wetlands. Bottom dredging resulted in the loss of seagrass beds in northern Biscayne Bay and has affected the stability of bay sediments and the capacity to assimilate nutrients and trap particulates. Stormwater runoff from urban development has increased the bay‟s exposure to contaminants and excessive nutrients. The filling and destruction of coastal wetlands has eliminated natural filtering capacity. In the past 30 years, Biscayne Bay‟s water quality has improved substantially because of the elimination of direct discharge of sewage into the bay and other pollutant control measures (McNulty, 1970: Alleman et al., 1995; Miami-Dade County DERM, 2005).

The Biscayne Bay CEM provides an in-depth view of the interactions of the bay‟s ecological drivers, stresssors, effects and attributes (Browder et al., 2005a), and it is a valuable tool for guiding restoration efforts. Altered freshwater flow into Biscayne Bay is the stressor that the CERP will most directly affect by modifying flow volume, timing and spatial distribution. CERP may indirectly affect the input of solids, nutrients, toxicants and pathogens.

9.1.2 Florida Bay

Florida Bay is a mosaic of banks, basins and small islands. A defining feature of the bay is its shallow depth, which averages about one 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), resulting in the dominance of seagrass beds as both a habitat and a source of primary production. Basins are as deep as three meters, and are separated by a network of shallow, flat-topped banks. These irregular mud banks are a conspicuous feature of Florida Bay covering nearly 75 percent of western Florida Bay and about ten percent of the northeastern part of the bay. The bay includes over 200 small islands, many of which are rimmed with mangroves.

The shallowness of Florida Bay 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). Additional background information on Florida Bay salinity is provided in ***Section 0***

Salinity Stressor.

Many basins of Florida Bay are carpeted with a variety of seagrasses, that include turtle grass, shoal grass, manatee grass and widgeon grass. Various sponge species, particularly loggerhead sponges, contribute significantly to the bay‟s benthic habitat. Florida Bay provides important habitat for many commercially important species, such as spiny lobsters, stone crabs and many important finfish species, and it serves as the principal nursery for the offshore Tortugas pink shrimp fishery. The bay supports numerous imperiled species including the West Indian manatee, American crocodile, roseate spoonbill and several species of sea turtles.

Until the 1980s, Florida Bay was perceived by the public and environmental managers as being a healthy and stable system, with clear water, lush seagrass beds, and highly productive fish and shrimp populations. However, in the mid-1980s catches of pink shrimp decreased dramatically (Browder et al., 1999), and in 1987, a mass mortality of turtle grass beds began (Robblee et al., 1991a). By 1992, the ecosystem appeared to change from a clear water system, dominated by benthic primary producers, to a turbid water system, with algal blooms and resuspended sediments in the water column. These ecological changes are attributed to multiple causes (e.g., water management, development, sea level rise). The interaction of ecological drivers, stressors, effects and attributes in the bay are well described in the Florida Bay CEM (Rudnick et al., 2005).

9.1.3 Southwest Florida Coast

The southwest coastal environment includes Whitewater Bay and the Ten Thousand Islands region, which encompasses numerous smaller bays both named and unnamed. This area includes one of the largest mangrove forested regions in the western hemisphere. This Southern Coastal System subregion extends northward from Cape Sable, which forms the southwest tip of the Florida mainland, to link with the Northern Estuary Module boundary at the Lee-Collier County line. Shorelines in the estuarine southwest coast were, and still are, generally lined with mangroves (USACE and SFWMD, 2004). Whitewater Bay and the rivers connecting Shark River Slough to the southwest Florida shelf (e.g., Shark, Harney and Lostman‟s) are critical components of the integrated Southern Coastal System ecosystem. Whitewater Bay is an important nursery area for many recreationally and commercially significant fisheries species. The subregion includes the Picayune Strand Restoration Project area and its downstream estuarine zone, which is within the Ten Thousand Islands region.

Much of the Everglades freshwater outflow did and still affects the southernmost part of the southwest coastal domain via discharge from Shark River Slough through various outlets. Water from Big Cypress Swamp and local basin runoff dominates flows entering the northern part of the southwest coastal 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 decreased volume of water delivered to the estuarine systems. For additional discussion of coastal flows see Section 5.5 Ecosystem Characteristics of Everglades Coastal Wetlands in Relation to Freshwater Inflows Hypothesis Cluster, which is in the Greater Everglades Wetlands Module.

The Picayune Strand Restoration Project area is west of and adjoins Fakahatchee Strand Preserve State Park. The federal and state preserves and parks surrounding the Picayune Strand Restoration Project area (***Figure 9-2***) are expected to function as one regional and mutually beneficial ecosystem. Prior to anthropogenic impacts, flat topography, marl soils and seasonal rainfall cycle were principal influences on hydrology of the Picayune Strand area. This natural sheet flow system absorbed floodwater, promoted groundwater recharge, sustained wetland vegetation, rejuvenated freshwater aquifers, assimilated nutrients, and removed suspended materials. Fresh water reached the Ten Thousand Islands estuaries and associated salt marsh and mangrove swamps through a combination of overland sheet flow and groundwater seepage (USACE and SFWMD, 2004). The quantity and timing of freshwater inflows determined many characteristics of estuarine habitat by shaping salinity gradients and influencing other key aspects of water chemistry. The pre-development, slow year-round influx of fresh water maintained salinity in the natural range that estuarine species require.

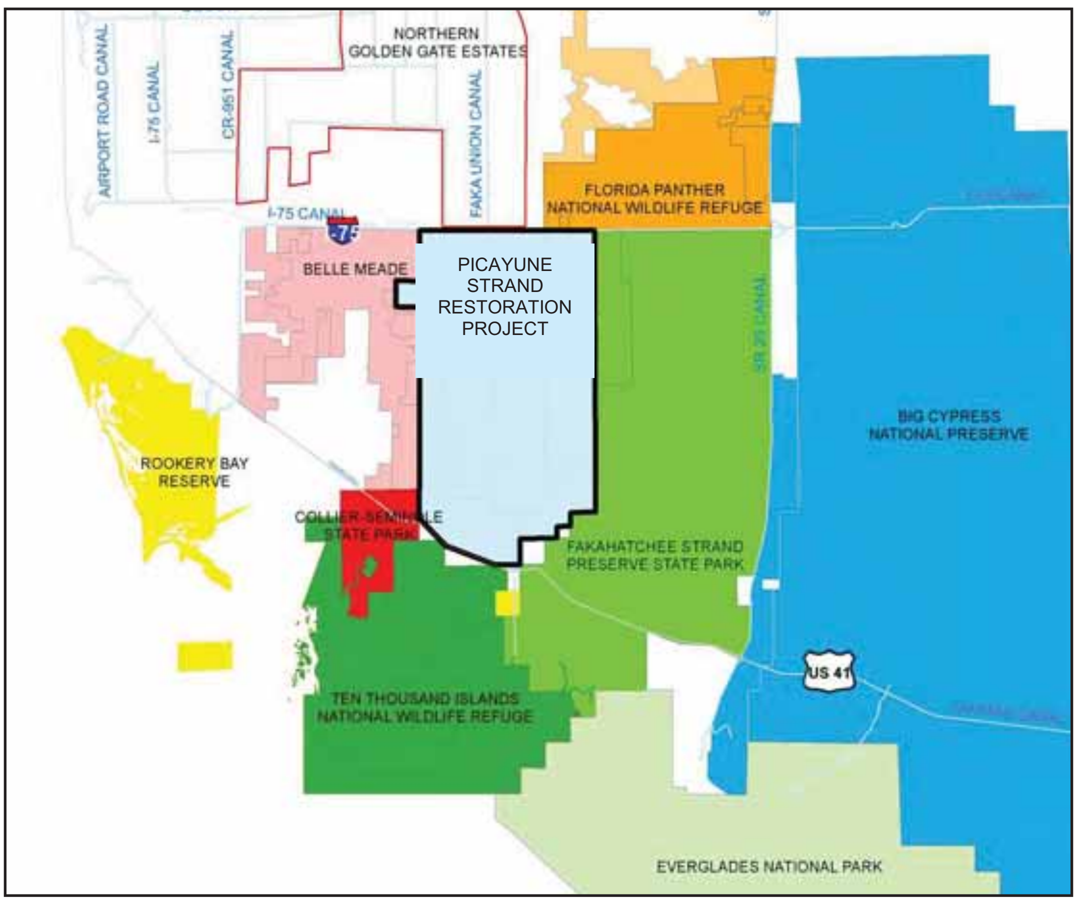


FIGURE 9-2. PICAYUNE STRAND RESTORATION PROJECT AREA AND SURROUNDING AREAS

9.2 SALINITY STRESSOR

9.2.1 Introduction and Background

The focus of the multiagency effort to evaluate salinity in the Southern Coastal System for this report has been to 1) understand the pre-restoration salinity regime 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 in weather, and climate change. Understanding the present variability within the coastal ecosystem is essential if a sound working hypotheses is to be formulated regarding the effect of Everglades restoration projects upon the downstream coastal ecosystem, to assess the actual effects of restoration upon the coastal ecosystem, and to provide the timely feedback to managers necessary to successfully implement iterative adaptive restoration.

Estuaries are dynamic coastal ecosystems where salt water from the ocean meets fresh water from the watershed (Gross, 1972; Davis, 1991). As such, salinity in estuaries usually varies significantly over time and space, and it is arguably the most important physical parameter for determining species and community composition in coastal waters. 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 identified as a stressor in virtually all CEMs developed for Florida‟s estuaries and adjacent mangrove wetlands (Browder et al., 2005a; Davis et al., 2005; Ogden et al., 2005; Rudnick et al., 2005). Further, salinity is the physical parameter most likely to be altered by restoration, and the primary causative agent for corresponding changes in flora and fauna. Because salinity is such an important parameter for determining the ecology of the Southern Coastal Systems and plays a key role in restoration planning, evaluation and implementation, a separate CEM was developed for it (***Figure 9-3***).

This conceptual model (***Figure 9-3***) presents the hypothesis that salinity patterns in the coastal wetlands and estuaries change in response to changes in the quantity, timing and distribution of freshwater inflow. Restoration would 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 would 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. For Biscayne Bay, maintaining a higher groundwater table in southeastern Miami-Dade County might provide groundwater flow during the early dry season, potentially extending the duration of nearshore estuarine conditions. For Florida Bay, diverting fresh water from northeastern into north-central Florida Bay and maintaining a higher water table in the southern Everglades would decrease the magnitude, duration and spatial extent of hypersalinity in north-central Florida Bay. Estuarine salinity patterns are also affected by circulation, which is driven by winds, tides, sea level rise and estuarine morphology (***Figure 9-3***). Restoration would alter circulation via changes in morphology as a result of biological feedbacks within estuaries (e.g. sediment accretion and stabilization, oyster reefs, mangrove islands).

Marine and estuarine communities are affected by short- and long-term salinity changes. Salinity can affect these communities by being too high, too low or excessively variable. 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. 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 timing and location. Changes in salinity resulting from restoration are expected to result in changes in biomass, distribution, species composition and diversity of seagrass, benthos, fish, invertebrates, birds and crocodiles. Overall for the Southern Coastal System, plant and animal diversity would increase as more natural freshwater flows and salinity regimes are restored.

Specific salinity patterns within each estuarine region as well as changes expected in these patterns following restoration are discussed in the following sections. A more thorough characterization of historic and current salinity and flow patterns, anthropogenic alterations to the system, and desired restoration conditions can be found in the documentation sheet for the RECOVER Southern Estuaries Salinity Performance Measure. This documentation sheet can be found at www.evergladesplan.org/pm/recover/recover_docs/perf_measures/090108_se_salinity.pdf.

9.2.1.1 Biscayne Bay

Salinity in Biscayne Bay varies on both temporal and spatial scales, particularly along the western shore and in the relatively isolated northern and southern sections. In the northern and southern areas, salinity tends to be lower during the wet season mainly due to canal and groundwater discharge. During the dry season, conditions in the southwestern part of the bay and in Card and Barnes Sounds are typically hypersaline (greater than 35 psu). Along the southwestern nearshore area, salinity can range from 15 psu to 46 psu during the year. Salinity tends to more stable, and generally near oceanic levels in the central-eastern area. The main factors that determine salinity in the bay are freshwater inflows from coastal canals, groundwater influx, the difference between rainfall and evaporation over the bay, and tidal exchanges with the Florida Straits primarily through the Safety Valve shoals.

Construction of major canals through the Everglades and dredging of natural tributaries and transverse glades that carried fresh water to Biscayne Bay resulted in lowered regional and coastal water tables (Parker et al., 1955), reduced water storage in the watershed, decreased groundwater flow to the bay, and elimination of many tributaries. Drainage of the watershed greatly affected the natural salinity gradients and ecotones from the Everglades through coastal wetlands and tidal creeks into the bay, and reduced or eliminated critical estuarine habitat for bay species requiring low-to-moderate salinity waters. In addition, constructed drainage systems result in pulsed, point source discharge degrading estuarine habitat near canal mouths by creating biologically damaging zones of bottom scouring and rapid salinity fluctuation (Chin and Wang, 1987; Montague and Ley, 1993; Serafy et al., 1997). The general lowering of the water table on the east coast ridge and diversion of both surface and ground water into canals has degraded not only estuarine habitats within the bay, but also adjacent coastal wetland communities, including herbaceous freshwater marshes and coastal mangrove wetlands that were once functionally connected to estuarine habitats.

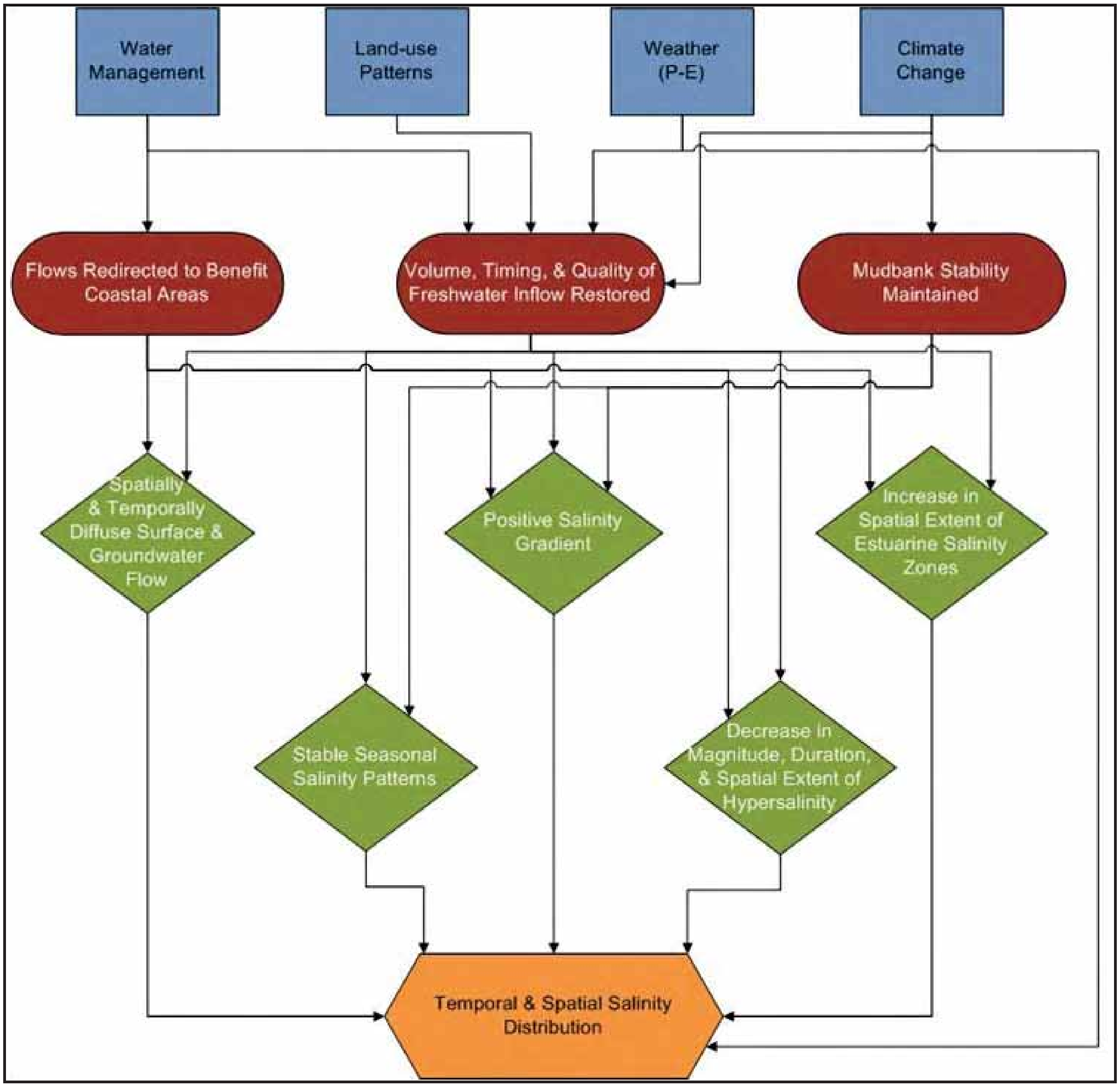


FIGURE 9-3. CONCEPTUAL ECOLOGICAL MODEL FOR SALINITY IN SOUTHERN COASTAL SYSTEMS

Modifying freshwater inflow volume, timing and spatial distribution would alter salinity regimes along the southwestern shore of the bay. It is anticipated that redirecting canal flows would provide sustained mesohaline salinity patterns in the nearshore environment and lower salinity in the mouths of tidal creeks. The restoration goals in Biscayne Bay are to: 1) reduce the intensity, frequency, duration and spatial extent of high salinity events; 2) reestablish common mesohaline to oligohaline conditions in mainland nearshore zones; and 3) reduce the frequency and rapidity of salinity fluctuations resulting from pulse releases of fresh water from canals.

9.2.1.2 Florida Bay

Florida Bay‟s salinity regime also varies greatly over time and space. This variation ranges from coastal areas that can be nearly fresh during the wet season, to large areas of the central bay that can have salinity levels near 70 psu during prolonged droughts, to nearly stable marine conditions of approximately 35 psu on the western boundary of the bay or near the Florida Keys‟ passes. The main factors that determine the salinity regime in the bay are the inflow of fresh water from the Everglades, the difference between rainfall and evaporation over the bay, and exchange with marine waters of the Gulf of Mexico and Florida Straits (Lee et al., 2006: Kelble et al., 2007).

Hydrologic alteration began in the late 1800s but accelerated with construction of drainage canals by 1920, the Tamiami Trail by 1930, and the C&SF Project and the South Dade Conveyance System from the early 1950s through 1980 (Light and Dineen, 1994). As a consequence of the loss, diversion, and impediments to flow, fresh water that would otherwise have made its way southward into the bay diminished. These changes in volume, timing, and distribution accordingly altered the bay‟s salinity regime (Swart et al., 1999 Brewster-Wingard et al., 2001; Dwyer and Cronin, 2001). For example, flows would historically cross the Buttonwood Embankment, a coastal levee averaging 1.5 feet in height that separates the peninsula of Florida from Florida Bay (Craighead 1964), and thereby decrease coastal salinity; however, such flows are now only observed after the passing of tropical cyclones or similar high precipitation events (Kelble et al., 2007). Although records indicate that significant hypersalinity events have occurred every decade since 1950, a prolonged regional drought 1987–1991 resulted in sustained hypersaline conditions. Isotopic examination of coral skeletons indicated that salinities observed 1989 - 1990 were the highest over the period 1824 – 1993 (Swart et al., 1999). A massive seagrass die-off began in 1987 and sparked much public and media concern regarding the future health of the resource (Fourqurean and Robblee, 1999). Although hypersalinity is a stressor to seagrass, it is not the only stressor (e.g., hypoxia and/or sulfide toxicity, among others). The multiplicity of potential causative factors coupled with the realization that the die-off began at the beginning of the drought, not after the drought when hypersaline conditions were fully engaged, suggests that stresses on seagrass were already at play when the die-off initiated. What this demonstrates is the complexity of the bay as a functional entity, and the importance of science as the necessary safeguarding element to ensure that watershed restoration brings about desired and beneficial change to a system as sensitive and potentially fragile as Florida Bay.

9.2.1.3 Southwest Florida Coast

In general, salinity along the southwest Florida coast is relatively stable compared to Florida and Biscayne Bays due to the direct connection to the Gulf of Mexico (Lee et al., 2002). However, localized salinity patterns are influenced by large volumes of direct freshwater runoff. Numerous creeks enter this area carrying water that originated in the Big Cypress Swamp as well as local basin runoff. Expected restoration changes in water stage upstream would result in changes in the salinity regime of the coastal areas.

Salinity along the southwest Florida coast varies seasonally with peak salinities occurring in the early summer. The peak salinity is typically close to that of the Gulf of Mexico, around 36.3

psu. However, during periods of extended drought, such as in 2007, conditions can become hypersaline, approaching 40 psu. A month or two after the onset of the wet season, the salinity in the nearshore regions begin to decrease. By late in the wet season, decreased salinities spread further offshore and reach their minimum values of below 30 psu. At the end of the wet season, much of the southwest Florida coastal region has salinities less than that of the Gulf of Mexico (Ortner et al., 2008). Upon cessation of the wet season, salinities slowly begin to increase and the cycle begins anew. Periodic meteorological phenomena, such as El Niño, can disrupt this seasonal pattern by decreasing the seasonal variability in precipitation (Kelble et al., 2007).

Faka Union Bay is one of several mangrove-lined indentations in the southwest Florida mainland that are partially separated from the Gulf of Mexico by many small mangrove islands. Other nearby bays of similar aspect and configuration include the larger Fakahatchee Bay immediately to the southeast, which has a direct connection with Faka Union Bay, and Pumpkin Bay immediately to the northwest.

Faka Union Bay was once supplied with upland flow through a natural flowway. This has been replaced by the Faka Union Canal. Fakahatchee Bay receives fresh water through two natural rivers, Fakahatchee River and East River, carrying flows that primarily originate as natural overland sheet flow from a watershed that is, for the most part, a protected wildlife conservation area, Fakahatchee Strand Preserve State Park. Pumpkin Bay receives fresh water through a natural flowway. Additionally, two small rivers, Wood River and Little Wood River, enter the estuarine system between Faka Union and Pumpkin Bays. The Whitney River, the Blackwater River, and Royal Palm Hammock Creek also feed fresh water to the general area.

Unlike most of south Florida, Collier County‟s estuarine areas remained virtually unaltered until the 1960s when the Faka Union Canal System was constructed to provide drainage for the Northern and Southern Golden Gate Estates developments. The development of Southern Golden Gate Estates was soon halted and this area is now the site of the Picayune Strand Restoration Project. The network of canals drains 190 square miles of southwest Florida, with approximately 185,000 acre-feet of water being discharged annually into Faka Union Bay, a small embayment of approximately 580 acres (SFWMD, 1996). As a result, too much fresh water is delivered too quickly to Faka Union Bay, particularly during the wet season. In addition, freshwater flows to adjacent bays are reduced because of surface flow diversion, which resulted in higher, less suitable salinity. These alterations in the timing and distribution of fresh water resulted in undesirable shifts in food availability, predation pressure, and reproductive success in Faka Union Bay when compared to Fakahatchee Bay (USACE and SFWMD, 2004). It is expected that faunal utilization of Faka Union Bay would become similar to that of Fakahatchee Bay as flows and salinity patterns improve.

9.2.1.4 Salinity Report Structure

For purposes of this report, monitoring results for salinity are presented in four major sections. They begin with an overview of wet and dry season conditions for both current (monitoring data) and model-predicted historical or restoration conditions. Results from a salinity HSI are then presented, followed by an evaluation of the RECOVER salinity performance measure using monitoring data and model output. Lastly, results from synoptic surveys of the southern coastal system are described.

9.2.2 System-wide Salinity Overview

9.2.2.1 Monitoring

The current evaluation of the Southern Coastal Systems salinity regime was based on assembling and evaluating the salinity data collected by Biscayne National Park (BNP), Everglades National Park, the FDEP, the NOAA, the USGS, and SFWMD. Although the MAP program provides funds to only the BNP and USGS efforts, the strategy of the Southern Coastal Systems Module, in accordance with that of the original MAP (RECOVER, 2004), is to appropriately integrate quality datasets regardless of funding source. This allowed a more robust evaluation of salinity in the large Southern Coastal Systems domain than would otherwise have been possible. An example of the long-term high-frequency datasets available is shown in ***Figure 9-6***. Such datasets will prove invaluable in detecting and interpreting restoration-induced changes in the salinity regime.

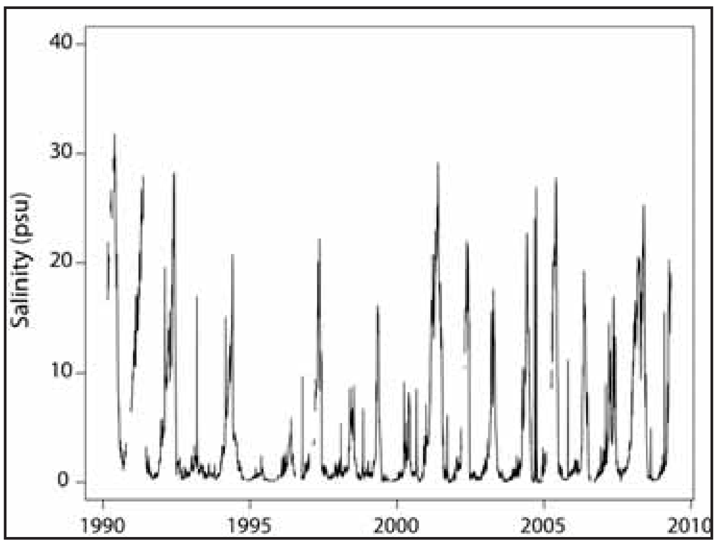


FIGURE 9-4. EXAMPLE OF LONG-PERIOD SALINITY TIME SERIES (BROAD RIVER) INVALUABLE TO UNDERSTAND SALINITY REGIME

High temporal resolution salinity time series were generated by self-contained units (sondes) deployed and maintained at 114 fixed sites throughout the Southern Coastal System (***Figure 9-6***) by the agencies listed above. Data were assembled and integrated into a dataset that could be easily queried. Frequency of measurement varied across efforts, from every 15 minutes, half-hour or hour. Analyses were performed at hourly time steps for comparability. Although some sites have been operational since 1990, some efforts were not initiated until 2004. To account for these differences in timeframes the analysis of empirical data was constrained to the 2004 to 2008 period. The 2004 to 2008 five-year variability in salinity was examined to determine the impact of water management and meteorological phenomena on the physical environment of the Southern Coastal Systems. Because few restoration projects are presently in place to produce a detectable response in salinity, the goal is to depict the current pattern of variation and suitability for faunal habitat and reproduction. It is anticipated the Picayune Strand, Biscayne Bay Coastal Wetlands, and C-111 restoration projects are the most likely candidate efforts that would begin to alter the current pattern of salinity within the next few years. Given the projects proceed and monitoring is sustained, a goal of the next system status report would be to detect responses to these efforts.

In addition to portraying current salinity conditions, an attempt was made to compare current salinity to estimates of pre-drainage salinity, as a test of target setting. Accomplishing this required utilizing simulated results from predictive models that were available at the time this SSR was prepared. Natural System Model (NSM) forecasts (Marshall et al., 2004; Marshall 2005a, 2005b, 2008) were used to simulate pre-drainage conditions in Florida Bay. The hydrodynamic model developed by John Wang of the University of Miami for Biscayne Bay (Wang et al., 2003) was employed to estimate restoration conditions in Biscayne Bay based on the assumption that both the Biscayne Bay Coastal Wetlands Alternative O, Phase 1 and Miami-Dade wastewater reuse projects are in place, i.e. a presumed restoration condition. Forecasted time series data were then extracted spatially to coincide with the current network of monitoring sites. The timeframe utilized for model output for both areas was 1996 to 2000 so that the comparisons with real world empirical data would at least be within the same positive phase of the Atlantic Multi-decadal Oscillation (AMO) (since predictions for the 2004-2008 time period are presently not available and most of the current time period for NSM represents a negative AMO phase). No predictive (i.e., model-based) data are currently available for the Florida southwest coast, thus the gap in spatial coverage in the following figures for simulated salinity. Efforts are underway to rectify the modeling gap along the Southwest Florida coast. Although this approach is less than ideal, future SSRs are expected to improve upon this approach as better information becomes available.

RECOVER is developing new software tools to generate time series salinity data at any desired point to coincide with the location of ecological monitoring data, such as SAV or fish. These extrapolations can be based on either the actual salinity data or from model output. This capability in the Southern Coastal Systems Module is currently limited to Biscayne Bay. All indications are that the overall current monitoring design is sufficiently robust to distinguish restoration-induced changes from those due to climatic change and variability, given the caveat that revisions will continue to be implemented as restoration, monitoring, and analysis of monitoring data advances.

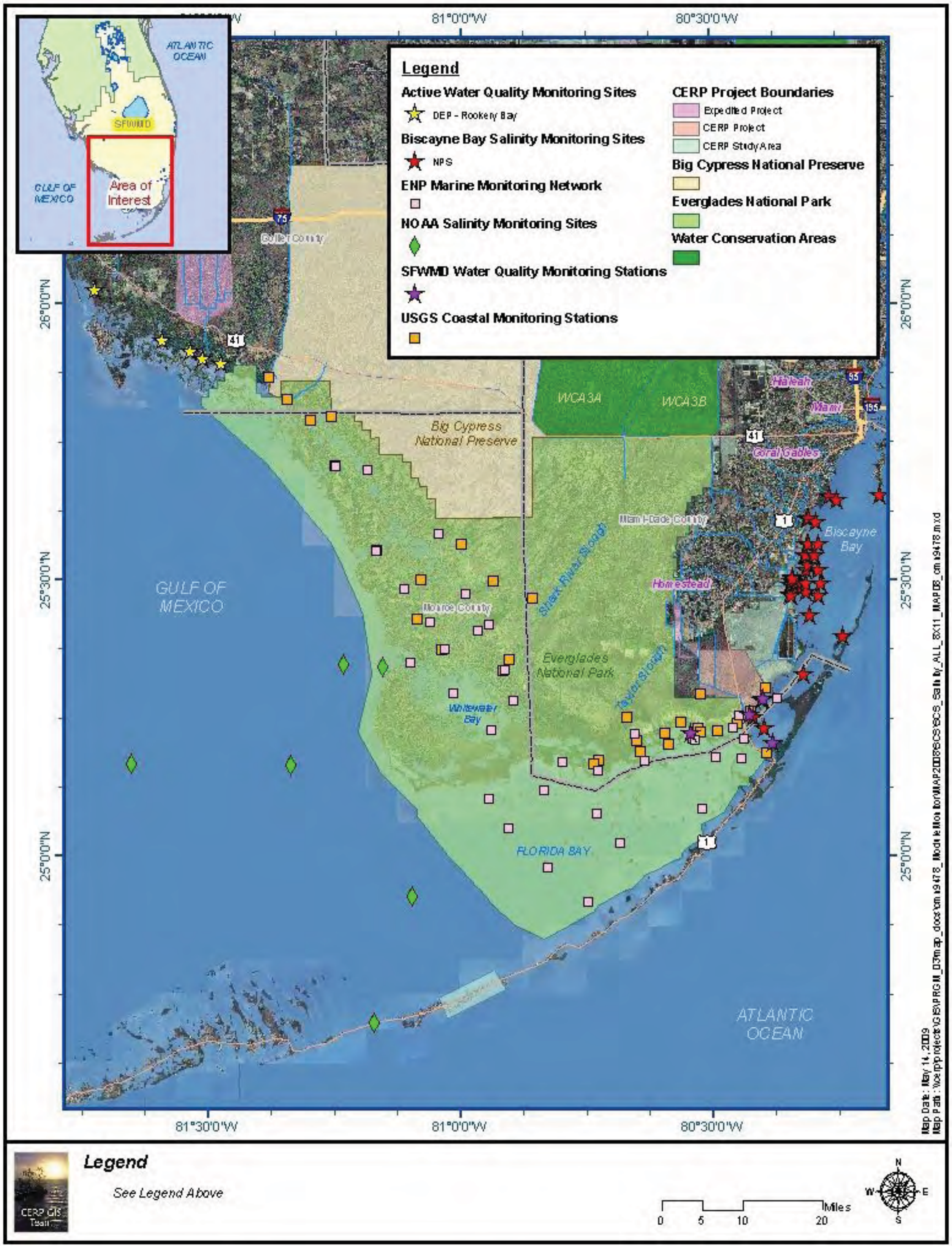


FIGURE 9-5. LOCATIONS OF SITES WHERE SALINITY TIME SERIES ARE COLLECTED IN THE SOUTHERN COASTAL SYSTEMS

(Collections by Biscayne National Park and Everglades National Park, US Geological Survey, National Oceanic and Atmospheric Administration, and Florida Department of Environmental Protection)

9.2.2.2 Results

The current dry season salinity regime (***Figure 9-6a***) reflects the delivery of water via the C-111 Canal to Barnes Sound and Manatee Bay, which historically received more water than at present, as did Joe Bay and Long Sound in northeast Florida Bay. The potential benefit of water delivery to these areas is apparent in the predicted graphic provided in ***Figure 9-6b*** **(**i.e., decreases in salinity of 5-10 psu in the restored condition). The key point in ***Figure 9-6a*** and ***Figure 9-6b*** is that more water is needed and the distribution of this water needs to be broader – further west toward Flamingo and further east toward Card Sound – in order to approach the NSM-based estimates. Restoration thus requires more water flowing through Taylor Slough (not seeping east into the C-111), along with spreading more water into the Model Lands. See Lorenz et al. (2009) for a more complete discussion of C-111 Canal flow patterns.

The predicted scenario also shows a decrease in salinity from north central Florida Bay to Key Largo due to rainfall-driven runoff from the approximately 50,000-acre Model Land basin that gently slopes in this direction and is dominated by marl soils prone to promote surface runoff (***Figure 9-6***). Dry season salinities in Whitewater Bay may be improved by sustained higher stages inland. Little apparent difference can be seen between current and predicted Biscayne Bay salinities, which are presumably a consequence of the absence of storage and additional water in the restoration plan. Also, there may be improvements in the nearshore areas along western Biscayne Bay under the restored condition that are important to flora and fauna, but not easily detectable on the scale at which ***Figure 9-6*** is depicted. Wastewater reuse does not appear to have a large effect on nearshore salinity during the dry season; however, once updated models become available, a more thorough analysis should be made of the effects of 130 million gallons per day (mgd) of reuse water being added to the system.

Currently, the Florida Bay wet season salinity (***Figure 9-7a***) is, in general, higher than dry season because of the lag time between wet season rains and outflow of that water delayed several months into the dry season. Also apparent is the expansion of the area of hypersalinity which is a result of increased isolation and evaporation, and is potentially enhanced by Shark River and C-111 circulation patterns isolating this area further from freshwater input. The predicted regime for Florida Bay shows a much larger area of lowered salinity and the area of hypersalinity is absent (***Figure 9-7b***).

Current condition in Biscayne Bay shows little difference between wet and dry seasons (***Figure 9-6a*** and ***Figure 9-7a***). Presumably this is because water is discharged to the bay in the wet season for flood control purposes creating a little lag, and discharges are made in the dry season to manage stage for agriculture. In addition, groundwater inputs continue beyond the end of the wet season for several months due to the extreme porosity of the Biscayne aquifer. The predicted regime (***Figure 9-6b*** and ***Figure 9-7b***) for Biscayne Bay shows a much larger area of reduced salinity compared to the current regime (***Figure 9-6a*** and ***Figure 9-7a***).

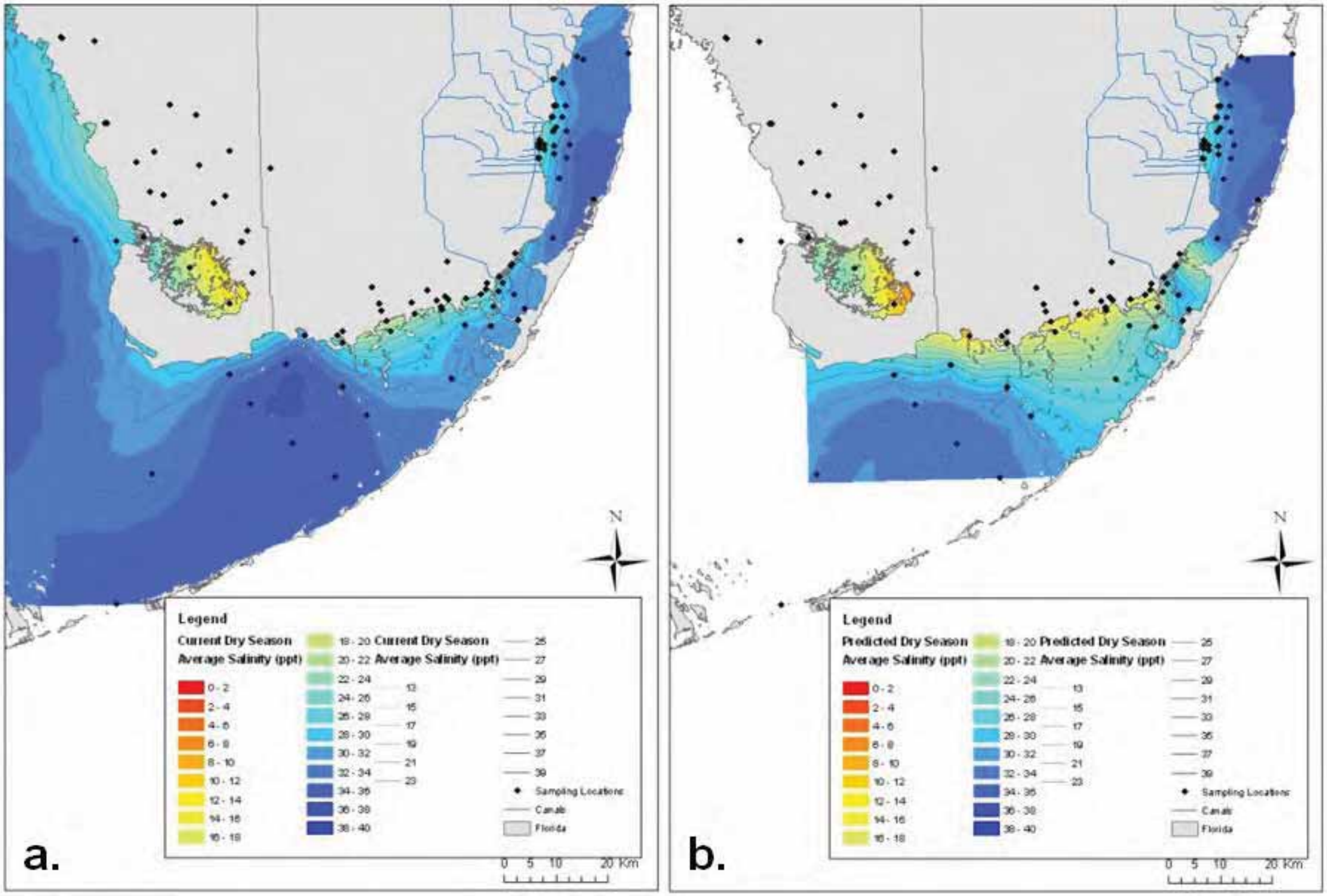


FIGURE 9-6. AVERAGE DRY SEASON SALINITY (NOVEMBER-MAY)

Note: salinity based on a) observed (current) and b) model (predicted) data

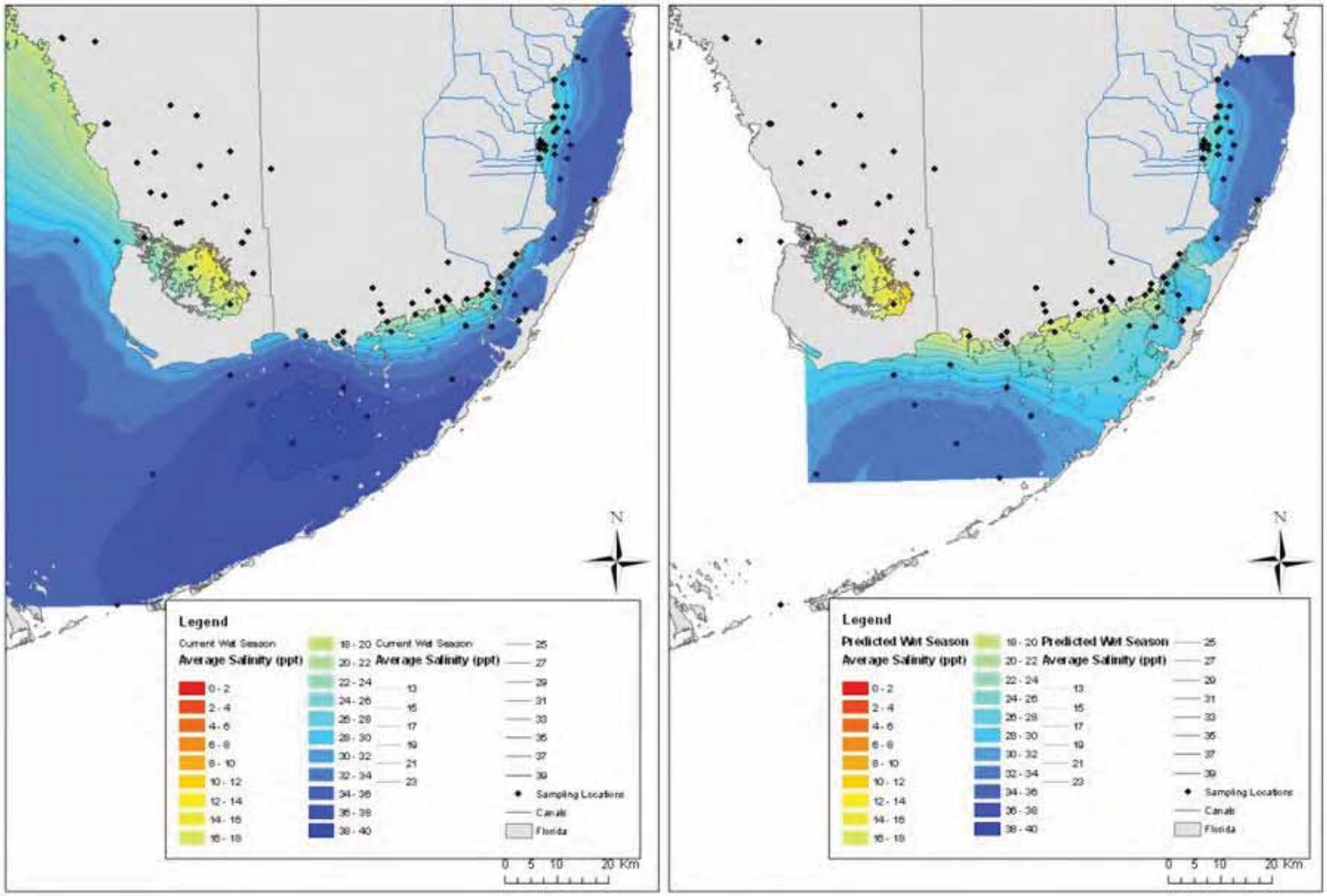


FIGURE 9-7. AVERAGE WET SEASON SALINITY (JUNE-OCTOBER)

Note: salinity based on a) observed (current) and b) model (predicted) data

9.2.3 Salinity Regime Suitability Index

9.2.3.1 Monitoring

The available salinity time series data for the most recent five years, from January 1, 2004, through December 31, 2008, for all sites (***Figure 9-9***) were used to evaluate three suitability criteria, which were defined as: 1) minimal incidence of hypersaline conditions where salinity does not exceed 37 psu; 2) minimal incidence of abrupt salinity changes (stability) where salinity does not vary by more than 5 psu over the course of a day; and 3) a preponderance of observations that are 25 psu or less, which is the optimal estuarine salinity range. These criteria were developed by the SSR Southern Coastal Systems Module Sub-team and they reflect salinity conditions that are either unsuitable (hypersalinity and abrupt changes) or optimal (less than or equal to 25 psu) to flora and fauna desirable in the nearshore areas. Values for each index were scoreded on a zero-to-one scale based on the percentage of observations that satisfied the criteria, where zero denoted that an undesirable condition existed at all times (i.e. always hypersaline or unstable or above 25 psu), and a value of one denoted that desirable condition existed at all times during the period of record evaluated. For example, if salinity was greater than 37 psu for 30 percent of the days, then the hypersalinity criterion index would equal 0.7 for that site, that is, 70 percent of the time less than 37 psu.

These indices provide insight into the current condition and potential improvements in condition that may be forthcoming as restoration progresses, and constitute an assessment strategy to evaluate the suitability of salinity regimes. As conditions improve and as biological-salinity relationships are refined, these criteria will be revisited. Furthermore, a Salinity Regime Suitability Index (SRSI) is proposed in this SSR as a tentative approach to facilitate implementation of AM. As proposed herein the SRSI equals the geometric mean of the three sub-indices, given the caveat that the geometric mean approach as utilized herein is exploratory and would be revised as further examination deems it necessary. AM utilization of the SRSI would in any case depend on full examination of the underlying salinity conditions. For example, consider two hypothetical areas that have the same low SRSI score. Area A may have a low score primarily because it is subject to large salinity changes (low stability index score), but is otherwise benign. Area B, on the other hand, might have a low SRSI primarily because of frequent hypersalininity. For Area A, an appropriate management action might be to change the timing or location of water delivery such that additional water may not be necessary. For Area B, additional water during the dry season may be necessary.

9.2.3.2 Results

9.2.3.2.1 Hypersalinity

Figure 9-8 shows the spatial depiction of the hypersalinity subindex. The scale shows the percent of time an area is not affected by hypersalinity, i.e., percent of days where the mean daily salinity was less than or equal to 37 psu. ***Figure 9-8a*** represents current observed conditions (2004-2008) while ***Figure 9-8b*** represents predicted conditions after restoration plans are implemented. Increased blue within the figures denotes decreased prevalence of hypersaline condition. The apparent predicted large benefit in Biscayne Bay following restoration is likely attributable to the addition of supplemental water (reuse water) during the dry season when hypersaline conditions are problematic. The wetlands would then retard the timing and spatially redistribute flows to the extent that nearshore and central Biscayne Bay hypersalinity events are greatly reduced or eliminated. A similar marked reduction in the incidence of hypersalinity is predicted in Florida Bay, especially in central Florida Bay (an area particularly prone to hypersalinity), Long Sound, and Joe and Little Madeira Bays. Hypersaline conditions are predicted to still occur in Barnes Sound, which is somewhat isolated. The change in hypersaline condition in Whitewater Bay between the current observed condition (***Figure 9-8a)*** and the predicted restoration condition (***Figure 9-8b)*** is not significant (an approximate two percent apparent increase).

9.2.3.2.2 Stability

Figure 9-9 shows the spatial depiction of the stability subindex. The scale shows the percentage of days where salinity varies by more than 5 psu (i.e, an undesirable condition). Increased blue within the figures denotes improved stability. Improvements in the occurrence of instability between the current observed condition (***Figure 9-9a***) and the predicted restoration condition (***Figure 9-9b***) are apparent throughout Biscayne and Florida Bays. Salinity instability is a key factor adversely affecting the quality of floral and faunal habitat and is often overlooked, particularly in the coastal ponds upstream of Florida Bay. Unstable conditions can stress organisms, disrupt reproductive cycles, result in avoidance of the area entirely by organisms that can opt to do so, and/or result in mortality. The importance in the apparent potential for restoration to shift from an unstable salinity environment to an appropriately stable system cannot be overstated.

9.2.3.2.3 Optimum Range

A spatial depiction of the current observed (2004-2008) and predicted restoration optimum range indices are provided in ***Figure 9-10a*** and ***Figure 9-10b***, respectively. The scale used is the fraction of time the area shown has a mean daily salinity of less than 25 psu. Increased blue within the figures denotes increased occurrence of mean daily salinity less than 25 psu. Obviously, areas far from shore and sources of fresh water and closer to the Atlantic Ocean or Gulf of Mexico would be expected to have higher salinity. In the case of the optimum range index, decreased incidence far from shore (red) should not be interpreted negatively.

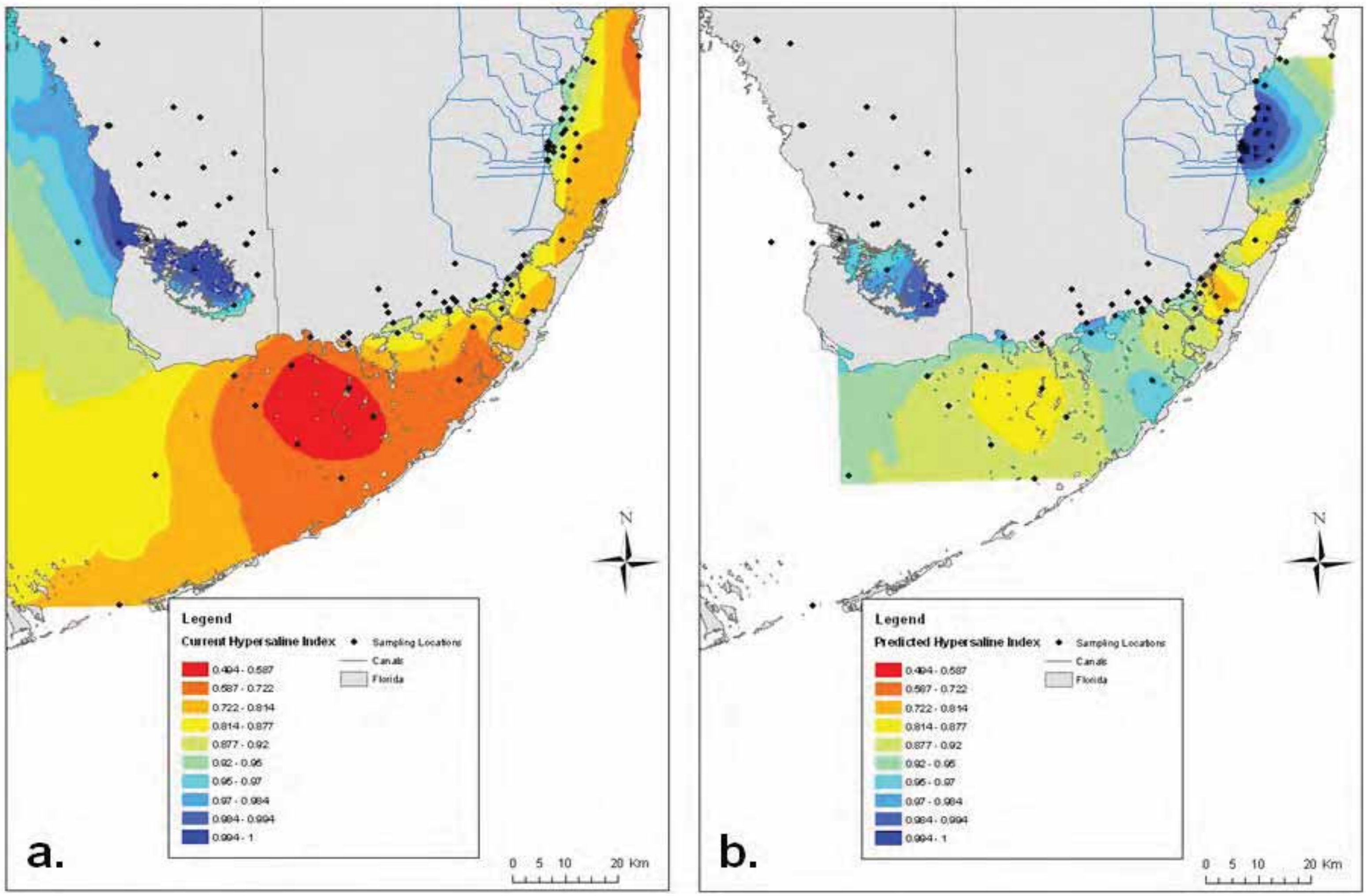


FIGURE 9-8. HYPERSALINITY SUB-INDEX FOR A) CURRENT OBSERVED CONDITIONS (2004-2008) AND B) PREDICTED RESTORATION CONDITIONS

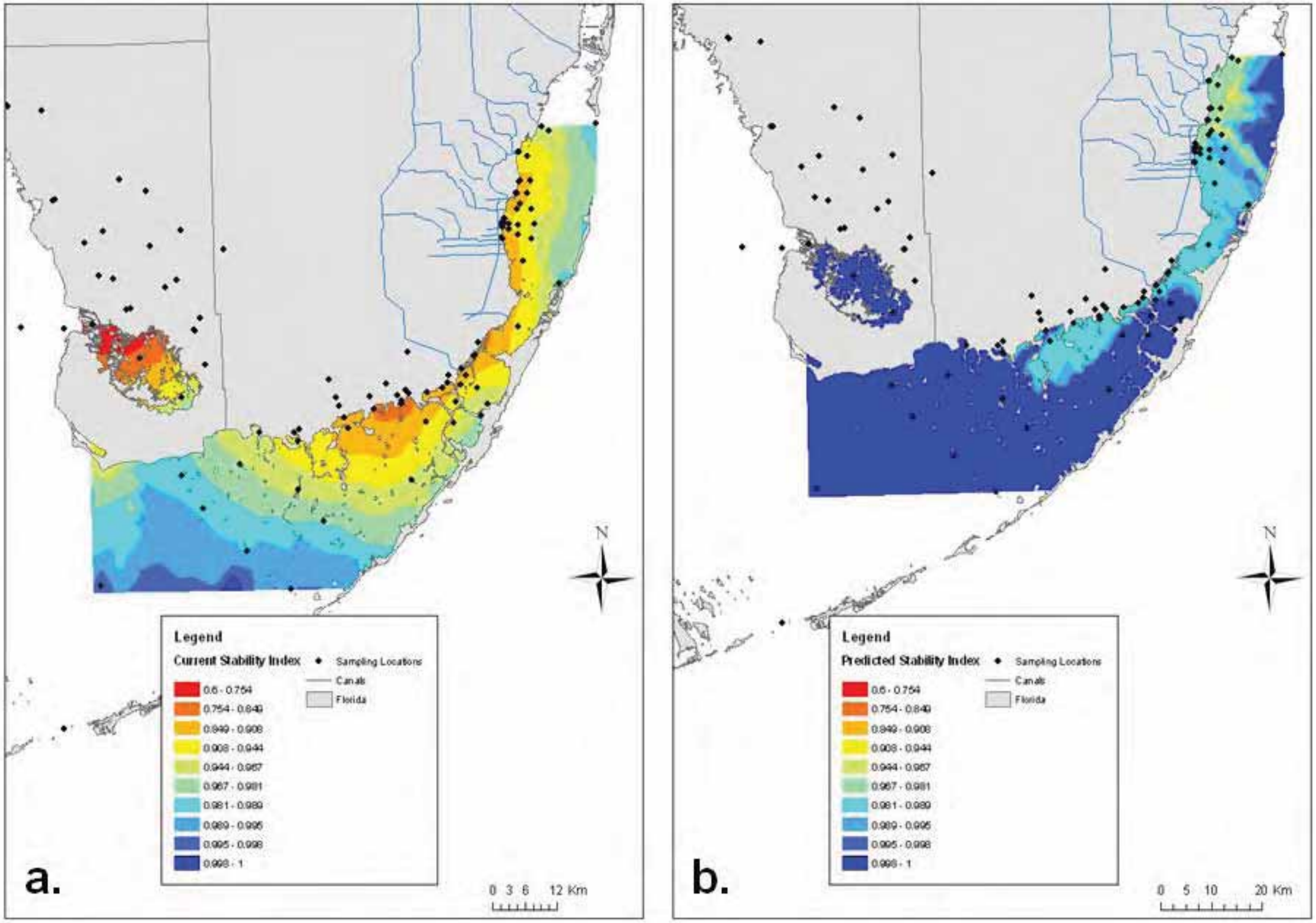


FIGURE 9-9. STABILITY SUB-INDEX FOR A) CURRENT OBSERVED CONDITIONS (2004-2008) AND B) PREDICTED RESTORATION CONDITIONS

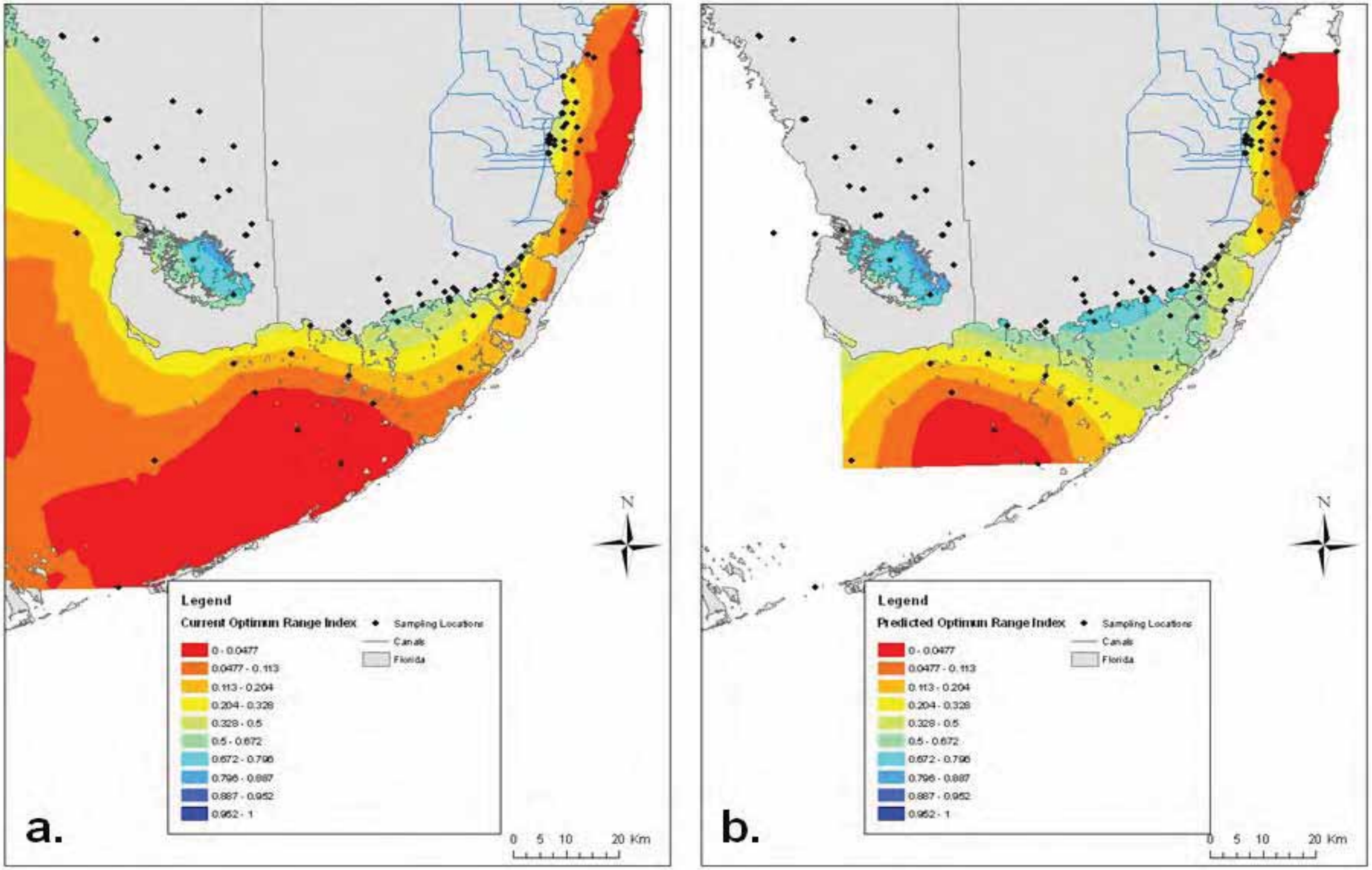


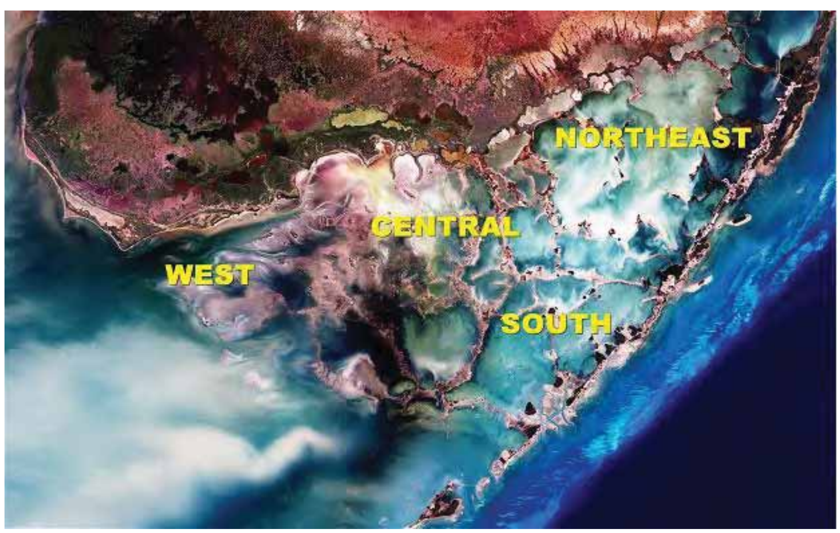
FIGURE 9-10. OPTIMUM RANGE SUB-INDEX FOR A) CURRENT OBSERVED CONDITIONS (2004-2008) AND B) PREDICTED RESTORATION CONDITIONS

The incidence of lower nearshore Florida Bay salinities is less desirable under the current conditions (***Figure 9-10a***) versus the model-based prediction for restoration conditions (***Figure 9-10b***). The area in Florida Bay where the incidence of lower salinity may occur following restoration is quite expansive, which could equate with numerous ecological benefits over a large area. The predicted incidence of lower salinity in Whitewater Bay following restoration is only marginally increased.

Biscayne Bay shows little change between the observed current optimum range index (***Figure 9-10***a) and the predicted restoration optimum range index (***Figure 9-10b***). This is presumably a result of the absence of additional water, other than wastewater reuse, being available under the final alternative (Alternative O, Phase 1) chosen for the Biscayne Bay Coastal Wetlands project. The benefits provided by reuse may be offset by the implementation of seepage barriers along the L31-W levee. The same amount of water is depicted as resulting in the same salinity regime when examined as a five-year average. In Card and Barnes Sounds, the current incidence of lowered salinity, which is 5 to 30 percent of the time, (***Figure 9-10a***) is less than the modeled predicted restoration regime, which is 20 to 50 percent of the time (***Figure 9-10b***).

9.2.3.2.4 Full Salinity Regime Suitability Index

The purpose of including the SRSI in a SSR is to show how these analyses might be used in future SSR to track change from the current condition to the restored condition that would provide suitable faunal habitat. While the SRSI is a useful assessment tool, a caveat must be applied to these analyses. The current condition graphic provided below does not reflect the segmented reality of Florida Bay due to numerous banks and shallows that restrict circulation (***Figure 9-11***); however, the condition at the individual sampling points is an accurate indicator. Depicted condition in areas between sites should be interpreted with caution.



(Florida Bay Science Program 2003)

FIGURE 9-11. LANDSAT-7 EXTENDED THEMATIC MAP IMAGE OF FLORIDA BAY SHOWING ITS SHALLOW BANK BATHYMETRY AND FOUR PRINCIPAL REGIONS

Spatial results of the geometric mean of the three subindices for both the observed current (2004-2008) condition and for predicted restoration condition based on modeling are shown in ***Figure 9-12b***. Under current conditions (***Figure 9-12a***), 25,000 acres are deemed suitable when the index is achieved at least 80 percent of the time. The area under the predicted conditions meeting the same criteria is around 69,000 acres. Assuming these estimates are reliable indications of potential restoration benefit, 44,000 acres of additional restored habitat are possible, a 280 percent improvement.

Restoration has the potential for meaningful improvements in habitat, particularly in Florida Bay. Moving away from the current condition (***Figure 9-12a***) to a more restored condition (***Figure 9-12b)*** would result in widespread ecological improvements. Since these areas are linked to one another, and linked to the coastal reef system, improvements in one area will equate with system-wide improvements.

The SRSI does not indicate much improvement in Biscayne Bay, except perhaps the very nearshore area (***Figure 9-12***). The lack of change may reflect the loss of water in the north part of the bay via the Miami River and the S-27 and S-28 structures as shown in the analysis of the Biscayne Bay salinity performance measure provided below. Improvements in Biscayne Bay nevertheless seem to occur and are much more apparent in the individual subindices used to calculate the SRSI.

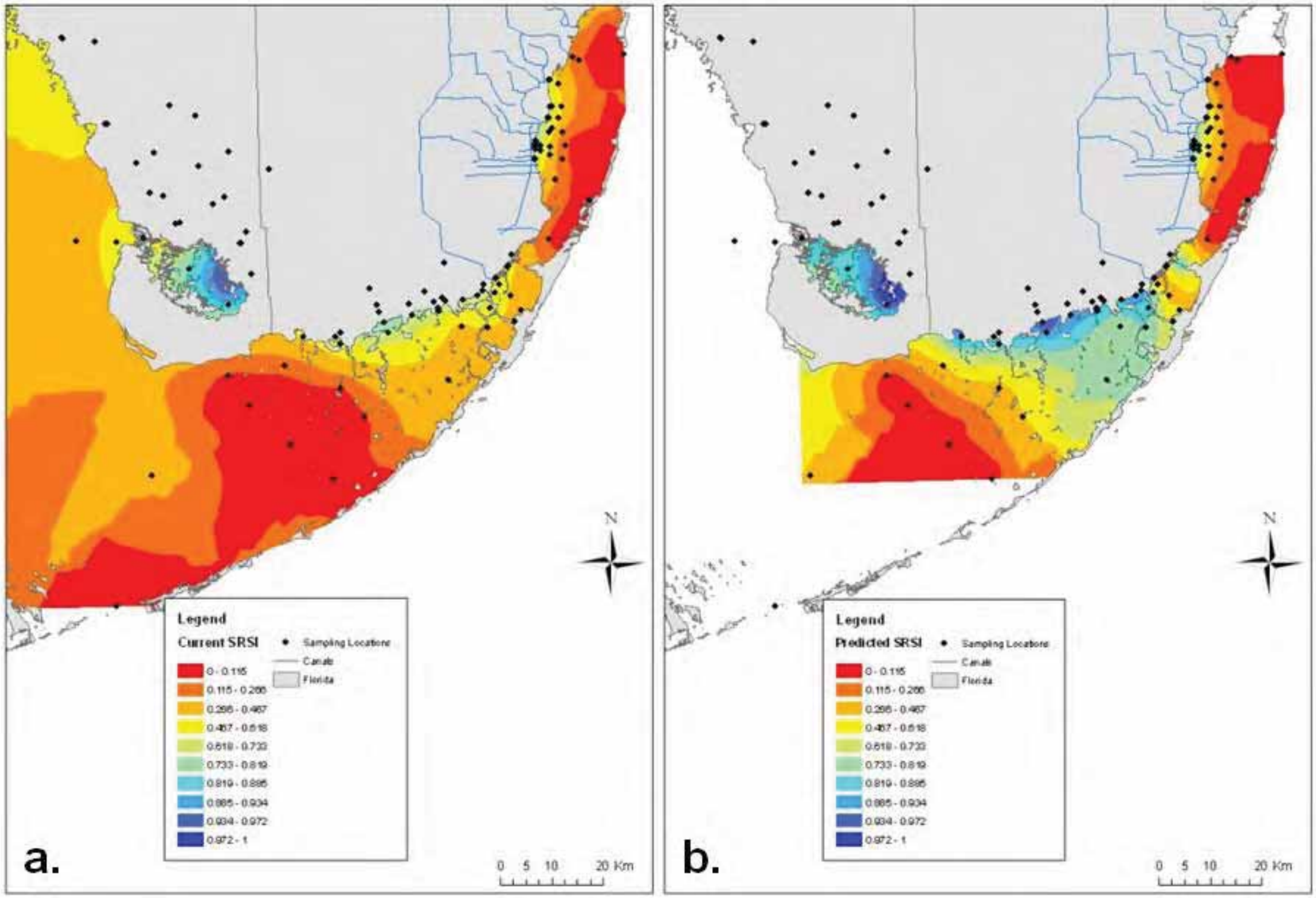


FIGURE 9-12. SALINITY REGIME SUITABILITY INDEX FOR A) CURRENT OBSERVED CONDITIONS (2004-2008) AND B) PREDICTED RESTORATION CONDITIONS

9.2.4 Synoptic Salinity Data

9.2.4.1 Monitoring

Surveys are conducted to determine the broad spatial patterns of circulation, water mass properties (including salinity), and distribution of dissolved and particulate constituents during different seasons, and to help quantify volume transports through the open boundaries of Florida Bay. The data are also used to determine the advective-dispersive pathways of freshwater riverine discharges and their associated nutrient inputs. Synoptic surveys of interest for restoration purposes are periodically executed in Florida and Biscayne Bays, and the southwest Florida coastal waters. The surveys utilize a flow-through system equipped to measure temperature, salinity, percent light transmission at λ=660 nanometers (nm), chlorophyll fluorescence, and chromophoric dissolved organic matter (CDOM) fluorescence. These data are collected at seven-second intervals and are stamped with the time and global positioning system (GPS) position of the measurement. In addition to the underway measurements, there are 40 discrete sampling stations in Florida Bay and 17 stations in Biscayne Bay. These stations are used to calibrate the underway instrumentation as well as to collect data on parameters not readily measured via underway methodologies (e.g. dissolved nutrients). Data has been used to partition salinity variation throughout the different subregions of Florida Bay (Kelble et al., 2007), and combined with information on the circulation in Florida Bay to suggest a possible restoration approach to mitigate the magnitude, extent and duration of hypersalinity in north-central Florida Bay (Lee et al., 2006).

9.2.4.2 Results

The results of the synoptic salinity study are available as graphic products and tabulated data. A recent example of the cruise tracks for the bay and coastal surveys and surface salinity contours are shown in ***Figure 9-13***. The cruise tracks are shown in white. This survey shows the anomalously high surface salinity that was caused by the extreme drought in central and northern Florida during spring 2008.

During the study period encompassing May 2006 through September 2008, 13 one-day surveys were conducted of Biscayne Bay and 11 two-day surveys were conducted of Florida Bay. Bay surveys were conducted approximately bimonthly and regional-scale surveys were conducted approximately quarterly during the study period. See http://www.aoml.noaa.gov/sfp for further details. A brief synopsis is given below.

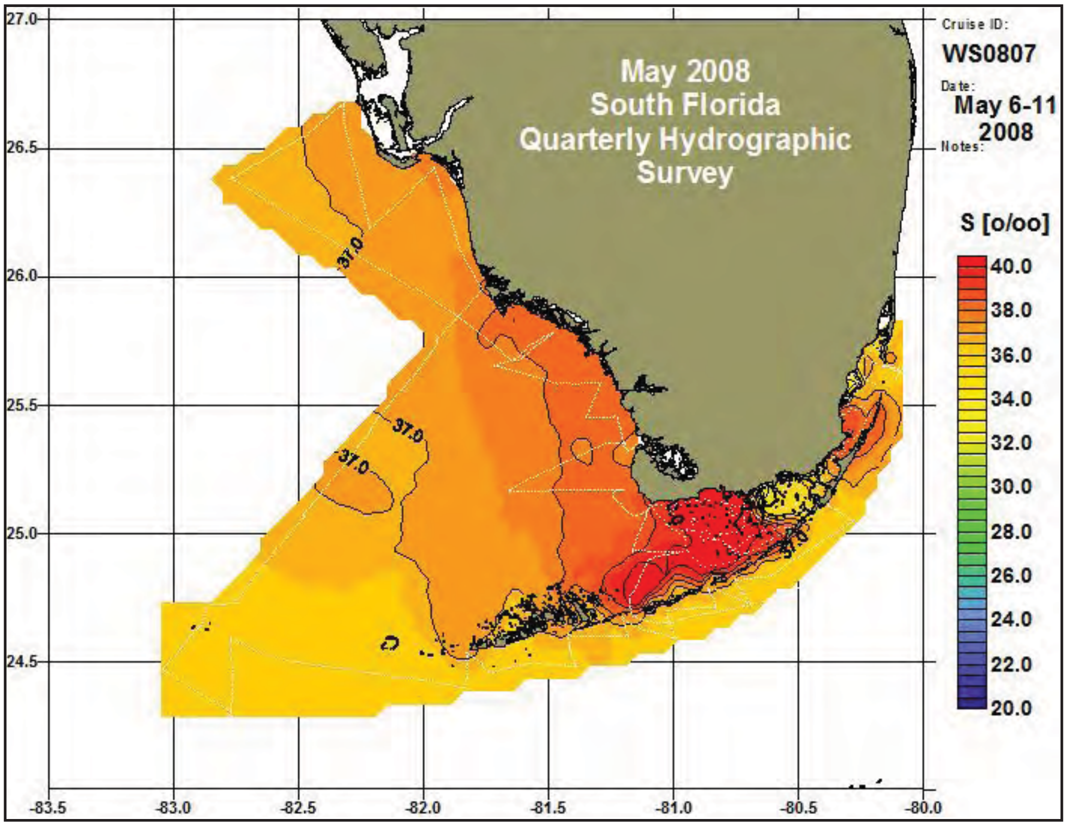


FIGURE 9-13. REPRESENTATIVE MAP OF SURFACE SALINITY FROM COMBINED COASTAL SURVEYS

Note: The cruise tracks are shown in white

9.2.4.3 Biscayne Bay, Barnes Sound, Blackwater Sound and Manatee Bay Shipboard Surveys

During 2007, the overall seasonal salinity pattern in Biscayne Bay, Barnes Sound, Blackwater Sound and Manatee Bay was similar to that observed during 2006. In February 2007, during the early part of the dry season, salinities in Biscayne Bay began to rise and by May 2007 values were uniformly higher (approximately 36 psu). The July 2007 survey showed a rapid transition to wet season fresher conditions, with point sources of extremely fresh (near 0 psu) inputs. The October 2007 cruise showed fresher conditions overall, but with a more diffuse salinity field and less sharp salinity gradients. By November 2007, salinities were again higher offshore and more uniformly mid-range over much of Biscayne Bay and the southern estuaries. The first two 2008 cruises (February and May) showed typical dry season high salinities, followed by an overall freshening, especially along the coast in central Biscayne Bay during summer and fall 2008.

9.2.4.3.1 Florida Bay Shipboard Surveys

During May 2007 salinities were higher, especially in the north-central Florida Bay. This pattern is typical of annual salinity variability in Florida Bay, which peaks in the mid-summer due to increased evaporation rates (Kelble et al., 2007). After the initiation of the rainy season, Florida Bay again became fresher overall, especially in the northeast corner. By February 2008, Florida Bay salinities were still typical of the end of the wet season, in the mid-range, but by May 2008, the extreme regional drought conditions had again caused the north-central bay to become hypersaline, with values exceeding 50 psu. Finally, by September 2008, the bay was fresher overall due to rains, especially along the northeast and northwest coasts.

9.2.4.3.2 Florida Regional-Scale Shipboard Surveys

During the study period encompassing May 2006 through April 2008, nine 6- to 10-day surveys were conducted off Florida coastal waters from Charlotte Harbor on the southwest Florida coast, to the Dry Tortugas, and the Florida Keys area to Miami. Overall, the salinity showed a typical dry season-wet season pattern during each of the study years. In May 2006, the salinity was uniformly high, similar to typical offshore Gulf of Mexico values. In August 2006, the values were fresher near the Ten Thousand Islands and Shark River areas, and by November 2006, these fresher values had spread offshore and filled the broader Cape Romano to Florida Bay area.

9.2.4.3.3 Synopsis

During the 2006 through 2008 time period, Florida experienced a drought that was the most severe since the late 1920s and early 1930s. This is documented in National Climatic Data Center records (www.ncdc.noaa.gov) of the Palmer Hydrological Drought Index for the State of Florida (***Figure 9-14***). Rainfall in south Florida responds to the larger-scale interannual El Nino-La Nina cycle (see National Centers for Environmental Prediction, www.ncep.noaa.gov), most noticeably with wetter than normal winters during the El Nino phase, and drier than normal spring and summer months during the La Nina phase. The La Nina of 2006 to 2008 is one of the likely causes of the severe drought in Florida observed during the same time period, while the active hurricane seasons of 2004, 2005 and 2008 caused increased monthly rainfall totals in the state.

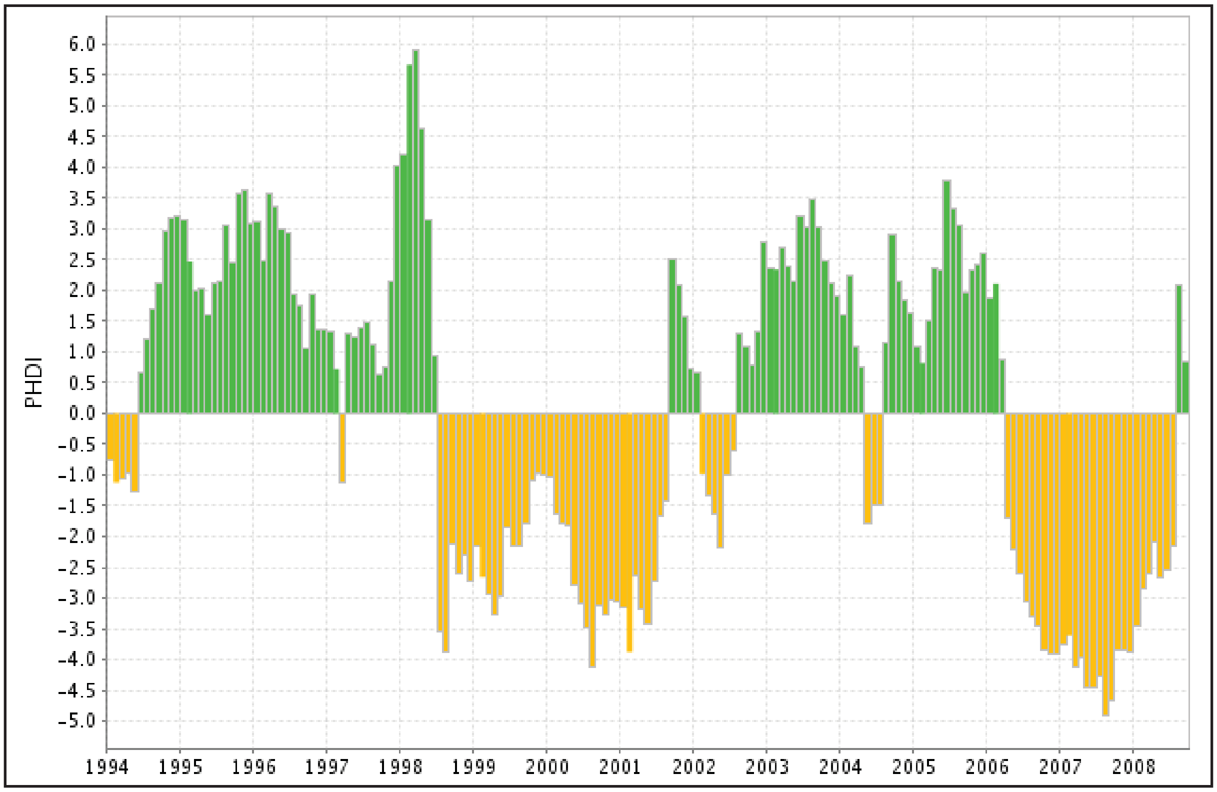


FIGURE 9-14. NATIONAL CLIMATIC DATA CENTER RECORDS OF THE PALMER HYDROLOGICAL DROUGHT INDEX FOR THE STATE OF FLORIDA

This interannual variability in the precipitation-drought pattern in Florida was manifested also in the surface salinity field of the coastal waters and estuaries, as shown in the salinity time series of the regional-scale cruise average (***Figure 9-15***). Although the overall variability is rather small, with a total range of approximately 34.5 to 37.5 psu, the trend from mid-2006 to the present is definitely for increasing salinity due to the drought. The second to last two data points, February and May 2008, are the highest values observed in the period of record. This area typically leads the State in degree of climatic change (R. Alleman, personal communication) which may affect interpreting restoration-induced change.

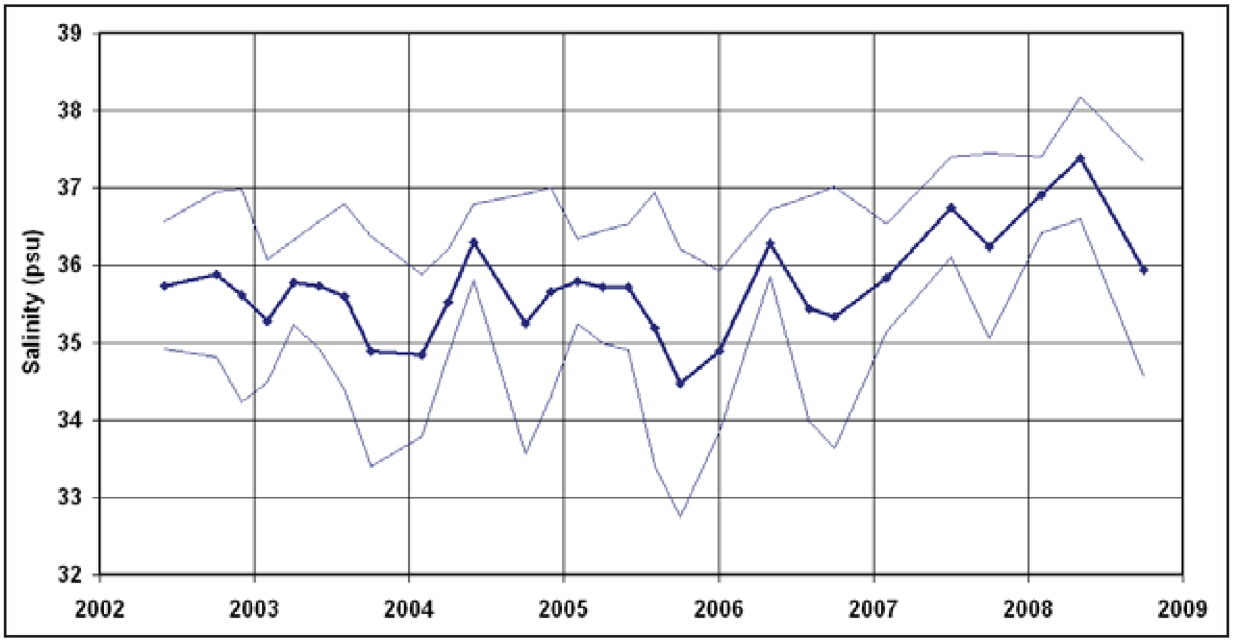


FIGURE 9-15. CRUISE-AVERAGED SURFACE SALINITY FROM THE COASTAL SURVEYS 2002-2008

Note: Light blue lines represent ±1 standard deviation

The more southern portions of the regional-scale cruise track, the Dry Tortugas, the lower and upper Florida Keys, and Florida and Biscayne Bays, did not show as much of an extreme high salinity response in 2008 to the Florida drought as did the overall cruise average. The reason for this is, as evident in the National Centers for Environmental Prediction climatological division time series shown in ***Figure 9-14***, the 2006 to 2008 drought was primarily a northern and mid-Florida event, with much less extreme drought conditions in the lower part of the state. This pattern of freshwater shortage was manifested in the surface salinity time series with the most extreme hypersaline conditions found in the northern part of the regional cruise track.

Further analysis aimed at quantifying the relationship between precipitation patterns and coastal surface salinity is in progress (Johns et al., 2008). It should be noted that events such as the 2006 to 2008 drought are, to a great extent, unpredictable. Thus, the only way to capture them observationally is to maintain a monitoring program. In this way, baseline conditions are established, means and SDs can be computed, and the degree to which these events are truly anomalous can be assessed. Long time series of observations become more valuable as the time series increases in length. Only long time series datasets are capable of differentiating effects due to restoration from climatic variation.

9.2.5 Picayune Strand Restoration Project Salinity Data

The Picayune Strand Restoration Project is one of a few system-scale restoration projects currently in the implementation phase. The goal of the project is to rehydrate more than 55,000 acres of land in southwestern Collier County that was partially developed. Roads and drainage canals reduced aquifer recharge; changed the quantity, quality and timing of discharges to estuarine receiving waters; and adversely affected the former wetland habitat by decreasing hydroperiod and increasing frequency and severity of forest fires. At this time, roads are being removed, canals are being plugged, and pump stations are being constructed. These activities would restore the wetland mosaic within Picayune Strand. In addition, flows would be redirected overland from the Faka Union Canal, which should improve conditions in the Ten Thousand Islands estuaries downstream of the project area by providing a more stable salinity regime.

9.2.5.1 Monitoring

Salinity monitoring sites within the estuaries are shown in ***Figure 9-16***. Yellow diamonds in ***Figure 9-16*** represent locations of FDEP high resolution (collected daily) salinity monitoring sites. Stars represent locations of SFWMD synoptic sampling sites, which are collected monthly. Faka Union Canal flow is measured at the US Highway 41 weir.

9.2.5.2 Results

Table 9-1 summarized statistics for data that were collected at the sites shown in ***Figure 9-16***. Values shaded in the table are the sites most directly influenced by Faka Union Canal discharge and, consequently, have the lowest salinity and highest variability (SD). It may be important to appropriately caveat results since resolution differs among these two sets of data (FDEP salinity monitoring sites generate data on a 15 minute timestep and SFWMD synoptic sites are sampled monthly), and significant gaps occur in both datasets over the period of record.

TABLE 9-1. SALINITY MEASURED WITHIN TEN THOUSAND ISLANDS ESTUARIES 2002-2007

Station	Mean	Standard Deviation	25%	Median	75%	Sample Size (n)	Resolution
Fakahatchee Bay	26.8	8.2	20.8	28.1	33.7	2012	daily
Faka Union Bay	22.9	11.1	13.0	24.4	32.7	2097	daily
Middle Blackwater River	29.6	7.9	24.8	32.4	35.6	1899	daily
Pumpkin Bay	31.4	6.2	27.8	33.0	36.4	1070	daily
TTI65	29.0	5.9	25.3	29.1	33.7	99	monthly
TTI66	30.8	4.9	27.6	31.2	34.6	99	monthly
TTI67	31.3	4.8	28.3	31.9	34.8	99	monthly
TTI69	28.9	7.3	25.3	30.1	34.0	99	monthly
TTI70	22.1	12.0	10.1	24.2	32.9	99	monthly
TTI71	32.1	4.6	29.8	33.5	35.5	99	monthly
TTI72	30.7	5.7	27.3	32.3	35.0	99	monthly
TTI74	32.6	4.9	31.2	34.1	35.9	99	monthly
TTI75	30.8	7.0	28.4	33.4	35.6	97	monthly

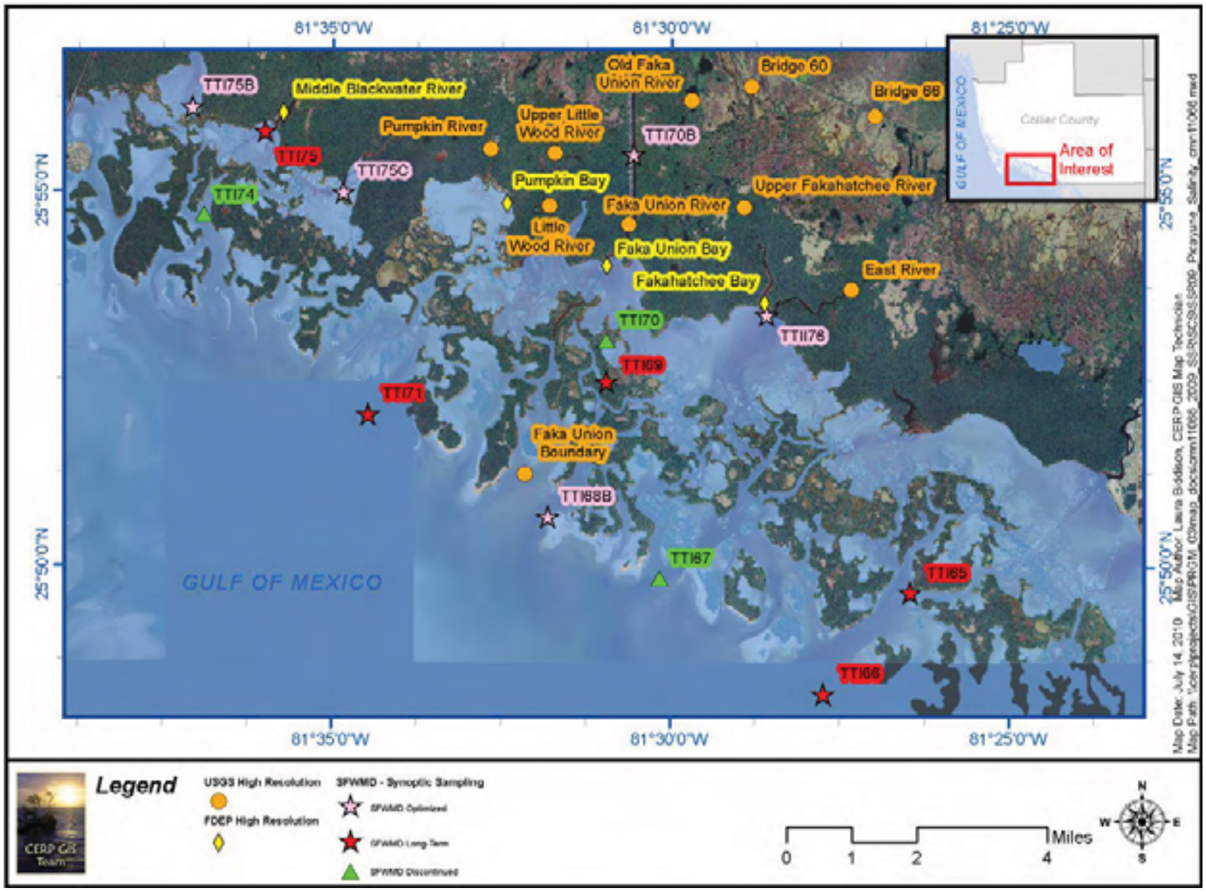


FIGURE 9-16. SALINITY STATIONS IN THE NEARSHORE ESTUARINE ZONE AFFECTED BY PICAYUNE STRAND RESTORATION PROJECT

A box-and-whiskers plot (***Figure 9-17***) presents the observed data graphically. The similarities between the salinity distributions of Faka Union Bay and TTI70 can be seen, along with the difference between these two sites and the other salinity monitoring stations. ***Table 9-2*** summarizes the mean salinity values by wet season (June-October, five months) and dry season (November-May, seven months). Faka Union Bay and TTI70 display the lowest mean salinity values and the greatest difference between wet and dry season mean salinity values. This is evidence that these stations are directly affected by the flow of fresh water from the Faka Union Canal. The diversion of water from the secondary canal system into spreader swales as part of the restoration is intended to redirect the fresh water into the water table aquifer and correct, to the extent possible, this seasonal impact on salinity.

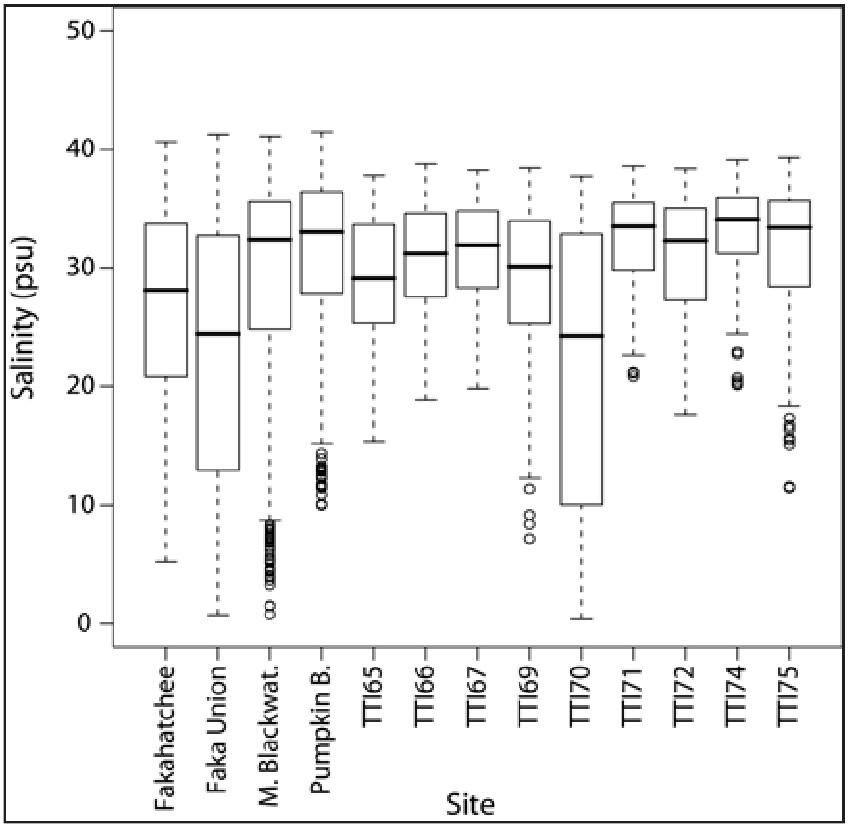


FIGURE 9-17. SUMMARY OF 2002-2007 PERIOD OF RECORD SALINITY

TABLE 9-2. COMPARISON OF SEASONAL SALINITY VALUES MEASURED 2002-2007

Station	Salinity (psu)		
	Dry Season	Wet Season	Difference (dry-wet)
Fakahatchee Bay	30.9	21.5	9.4
Faka Union Bay	28.6	15.5	13.1
Middle Blackwater River	34.1	24.2	9.9
Pumpkin Bay	34.4	27.0	7.4
TTI65	31.5	25.5	6.1
TTI66	32.8	28.0	4.8
TTI67	33.4	28.4	4.9
TTI69	32.3	24.0	8.3
TTI70	27.7	14.1	13.6
TTI71	34.2	29.1	5.1
TTI72	33.4	26.9	6.5
TTI74	35.0	29.2	5.8
TTI75	34.3	26.0	8.3

The salinity data were analyzed for correlations between the individual salinity monitoring stations and canal flow data. ***Table 9-3*** presents the Pearson (linear) correlation coefficient matrix for the SFWMD monthly data. These values are hampered somewhat by the small number of values (36) used to estimate correlation. It can be seen that all of the monthly salinity stations are correlated at a relatively high level, generally greater than r = 0.85. Correlation between salinity and Faka Union Canal flow are somewhat lower at all stations, with the highest correlation coefficient values, which occur at several stations, of 0.69. There does not appear to be any difference in the level of correlation between TTI70 and canal flow and the level of correlation seen at the other stations.

TABLE 9-3. PEARSON CORRELATION COEFFICIENT (R) FOR THE MONTHLY SOUTH FLORIDA WATER MANAGEMENT DISTRICT SALINITY DATA AND FAKA UNION CANAL FLOW DATA

	TTI65	TTI66	TTI67	TTI69	TTI70	TTI71	TTI72	TTI74	TTI75	Canal
TTI65	1.00	0.98	0.93	0.88	0.86	0.92	0.91	0.85	0.80	-0.62
TTI66	0.98	1.00	0.92	0.87	0.85	0.90	0.89	0.81	0.77	-0.59
TTI67	0.93	0.92	1.00	0.93	0.91	0.96	0.95	0.87	0.83	-0.66
TTI69	0.88	0.87	0.93	1.00	0.95	0.96	0.97	0.91	0.91	-0.69
TTI70	0.86	0.85	0.91	0.95	1.00	0.94	0.96	0.90	0.88	-0.69
TTI71	0.92	0.90	0.96	0.96	0.94	1.00	0.99	0.96	0.91	-0.69
TTI72	0.91	0.89	0.95	0.97	0.96	0.99	1.00	0.97	0.94	-0.69
TTI74	0.85	0.81	0.87	0.91	0.90	0.96	0.97	1.00	0.97	-0.68
TTI75	0.80	0.77	0.83	0.91	0.88	0.91	0.94	0.97	1.00	-0.69
Canal	-0.62	-0.59	-0.66	-0.69	-0.69	-0.69	-0.69	-0.68	-0.69	1.00

Table 9-4 presents the Pearson (linear) correlation coefficient matrix for the daily data from the FDEP high resolution stations. Correlation between salinity stations is similar to the level of correlation seen for the synoptic SFWMD monthly data, generally above r = 0.90. Correlation between flow and salinity is higher for the daily values compared to the monthly stations. Linear correlation between daily canal flow and Faka Union Bay salinity appears less than that seen at other stations because the relationship is highly non-linear (***Figure 9-18b***).

TABLE 9-4. PEARSON CORRELATION COEFFICIENT (R) FOR THE DAILY FLORIDA DEPARTMENT OF ENVIRONMENTAL PROTECTION SALINITY DATA AND FAKA UNION CANAL FLOW DATA

	Fakahatchee Bay	Faka Union Bay	Middle Blackwater River	Pumpkin Bay	Cana l
Fakahatchee Bay	1.00	0.95	0.91	0.96	-0.91
Faka Union Bay	0.95	1.00	0.87	0.90	-0.89
Middle Blackwater River	0.91	0.87	1.00	0.92	-0.94
Pumpkin Bay	0.96	0.90	0.92	1.00	-0.91
Canal	-0.91	-0.89	-0.94	-0.91	1.00

The pre-restoration discharge pattern from the Faka Union Canal includes high flow in the wet season and much lower, at times zero, flows in the dry season (***Figure 9-18a***). As a result, salinity in Faka Union Bay is highly variable, responding to moderate and larger discharges from the Faka Union Canal (***Figure 9-18b***). The Faka Union Bay salinity monitoring site is situated near the mouth of the canal. The non-linear effect of canal flow on Faka Union Bay salinity is clear in ***Figure 9-18b***, and the difference in the pattern at this station compared to the pattern at Fakahatchee Bay or Pumpkin Bay (nearly linear) can be seen.

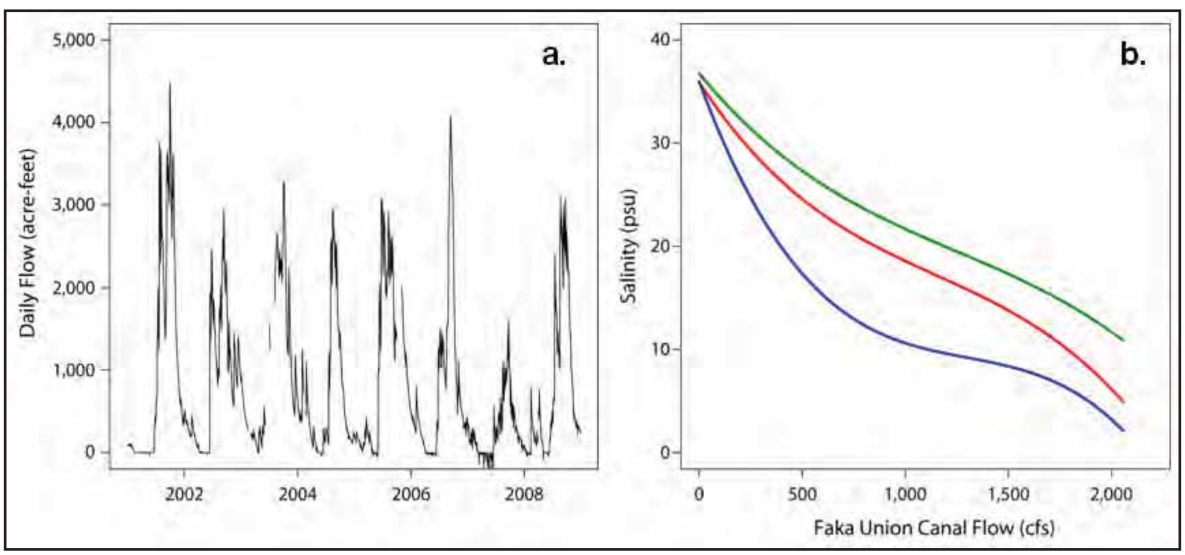


FIGURE 9-18. SEASONAL PATTERNS IN FLOW FROM FAKA UNION CANAL (PANEL A) AND SALINITY IN FAKA UNION (BLUE), FAKAHATCHEE (RED) AND PUMPKIN (GREEN) BAYS AS A FUNCTION OF FAKA UNION CANAL DISCHARGE (PANEL B)

Figure 9-19 graphically compares Faka Union Canal flow and the SFWMD monthly salinity data. The broad range of freshwater influence, both near shore and off shore, is likely due to a composite of wet season-dry season flow from the combined seasonal output of groundwater and surface flows. ***Figure 9-20*** presents differences in the salinity versus flow plots for Faka Union Bay and TTI70. A distinct non-linear pattern can be seen in the Faka Union Bay daily data, behavior that is not readily apparent in the TTI70 plot. The variability in daily flow is higher than the variability at the monthly level because high flow events typically last on the order of days, meaning several high flow events and subsequent lowering of salinity can occur each month and have short durations. This variability in salinity is not captured in the monthly grab samples.

High frequency salinity data (***Table 9-5)*** indicates that swings in salinity of five psu in one day most frequently occur in Faka Union Bay and Middle Blackwater River, both of which are downstream of a canal or river. Larger swings in daily salinity of 10 psu rarely occur in Fakahatchee Bay, supporting its choice as a reference area. Hypersalinity occurs most often in Pumpkin Bay, presumably due to its shallowness, and because much of the freshwater runoff that once flowed into Pumpkin Bay was diverted to the east by the Faka Union Canal.

TABLE 9-5. PROPORTION OF TIME DURING WHICH SALINITY REGIME WAS STABLE (LESS THAN 5 OR 10 PSU PER DAY) AND DEGREE OF FREEDOM FROM HYPERSALINE CONDITIONS BASED ON 2004-2007 CONTINUOUS SALINITY MONITORING DATA

Location	<5 psu/day Variation	<10 psu/day Variation	Hypersalinity (percent of days mean salinity <37 psu)
Faka Union Bay	0.73	0.78	0.91
Fakahatchee Bay	0.81	0.97	0.91
Pumpkin Bay	0.87	0.91	0.82
Middle Blackwater River	0.68	0.83	0.86

A summary of the Spearman‟s nonparametric correlation of flow and salinity compared to the Pearson (linear) correlation coefficient values is presented in ***Table 9-6***, and indicate that regression model (linear and non-linear) would provide a reasonable method to extrapolate model-estimated flows to specific locations within the region. This comparison indicates greater non-linear correlation to flow compared to linear correlation in salinity data from Faka Union Bay (***Figure 9-18b***) and TTI70, and suggests the use of non-linear or multivariate linear regression models for salinity at this site instead of simple linear regression models. Prominent curvature in the salinity-versus-flow plot at the Faka Union Bay monitoring site nearest the canal mouth (***Figure 9-20***) suggests that small increases in flow result in a significant decrease in salinity. Salinity values less than 10 psu are only seen when canal flows are greater than about 800 cfs. This curvature is not evident in the next nearest monitoring site, TTI70, nor is it particularly prominent at the FDEP monitoring sites in Pumpkin and Fakahatchee Bays (***Figure 9-18b***).

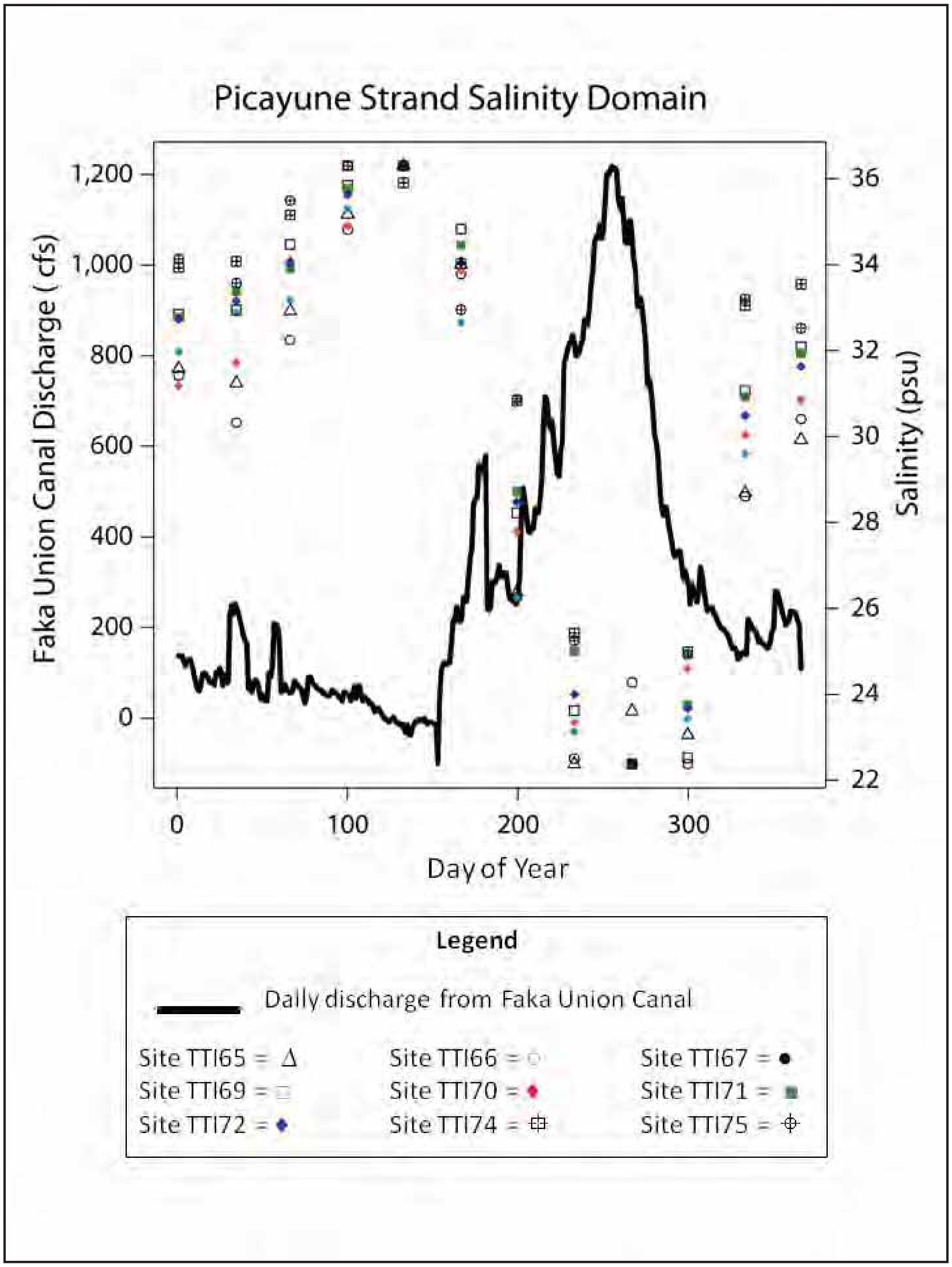


FIGURE 9-19. MEAN DAILY SALINITY (2002-2007) FOR SOUTH FLORIDA WATER MANAGEMENT DISTRICT TEN THOUSAND ISLAND WATER QUALITY NETWORK IS STRONGLY RELATED TO MEAN DAILY DISCHARGE FROM FAKA UNION CANAL

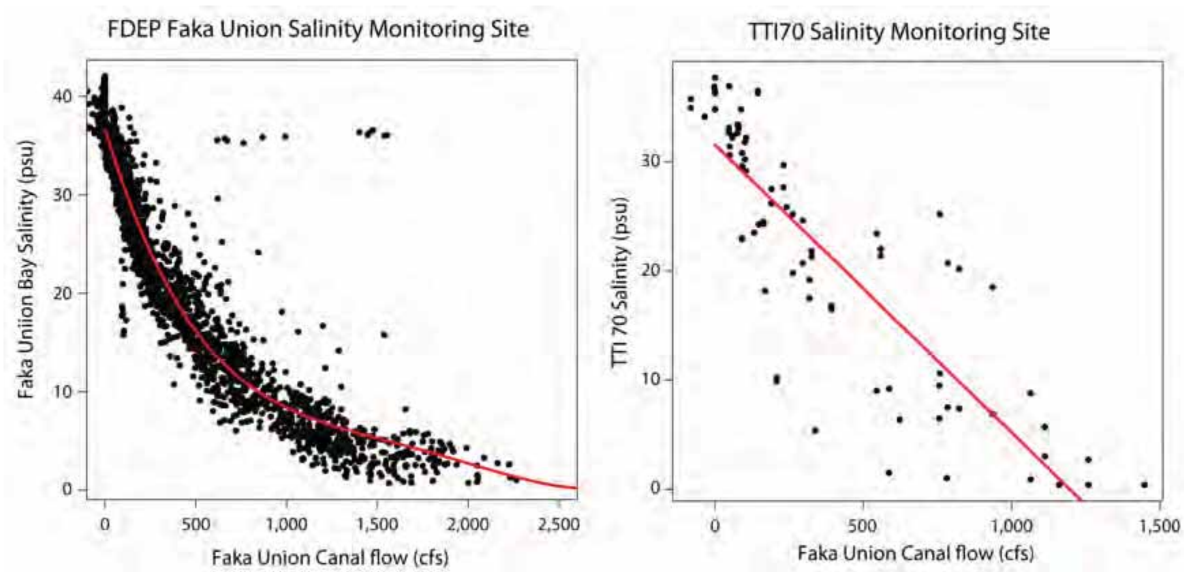


FIGURE 9-20. RELATIONSHIP BETWEEN FAKA UNION CANAL FLOW AND FAKA UNION BAY SALINITY AS MEASURED AT FLORIDA DEPARMENT OF ENVIRONMENTAL PROTECTION CONTINUOUS MONITORING SITE NEAR CANAL MOUTH AND SOUTH FLORIDA WATER MANAGEMENT DISTRICT TTI70 MONTHLY SAMPLING SITE AT END OF CUT CONNECTING FAKA UNION AND FAKAHATCHEE BAYS

TABLE 9-6. SPEARMAN'S NONPARAMETRIC RANK CORRELATION AND PEARSON CORRELATION COEFFICIENT (R) VALUES FOR FAKA UNION CANAL DISCHARGE VOLUME AND SALINITY DATA

Salinity Station	Spearman's rho	Pearson r	Observed Data Time Step
Fakahatchee Bay	-0.950	-0.91	daily mean
Faka Union Bay	-0.957	-0.89	daily mean
Middle Blackwater River	-0.928	-0.94	daily mean
Pumpkin Bay	-0.890	-0.91	daily mean
TTI65	-0.904	-0.62	monthly
TTI66	-0.885	-0.59	monthly
TTI67	-0.900	-0.66	monthly
TTI69	-0.911	-0.69	monthly
TTI70	-0.905	-0.69	monthly
TTI71	-0.898	-0.69	monthly
TTI72	-0.923	-0.69	monthly
TTI74	-0.874	-0.68	monthly
TTI75	-0.873	-0.69	monthly

Note: all correlations are significant at $p<0.0001$

The relationship between Faka Union flow and salinity at the monthly Ten Thousand Islands monitoring sites is more linear, even those located a distance away, such as TTI75, which is near Middle Blackwater River, suggesting a regionally seasonal hydrologic influence, as opposed to strictly being the consequence of Faka Union Canal flow patterns. A more gradual reduction in salinity, or a more linear relationship, due to seasonal rainfall increases may be the more desired, more natural condition. As restoration unfolds, an indicator of progress toward more natural condition may be a reduction in curvature of the salinity response to flow at the Faka Union Bay monitoring site. The relationship between salinity and flow from other streams and the associated proximate monitoring sites should be explored to resolve these issues.

Redistribution of flows from Faka Union Canal to the wetland mosaic once Picayune Strand Restoration Project is fully implemented is intended to reduce flows to Faka Union Bay and increase freshwater flows to Fakahatchee and Pumpkin Bays, and Middle Blackwater River through numerous tidal creeks and freshwater sloughs discharging to the local estuarine areas. The expected effects of this restoration project have been modeled by the USGS using the Ten Thousand Islands Model. This model is a separate model from the Tides and Inflows in the Mangroves of the Everglades (TIME) Model but is similar in computational structure. It was originally developed to evaluate temperature effects of the Picayune Strand Restoration Project in manatee refuge areas during cold weather (Swain and Decker, 2009). The USGS provided Ten Thousand Island Model output for post-restoration discharge from the Faka Union Canal as daily time series for the analysis below.

The USGS analysis of the Picayune Strand Restoration Project output (Swain and Decker, 2009) indicates that the simulation produces, in general, lower Faka Union Canal flows during normal dry seasons even though higher flows may occur during normal wet seasons. An analysis of the observed salinity and estimates of post-project daily mean salinity indicate that location is a significant factor in salinity variations. At the US Highway 41 weir, the dry season mean salinity values are higher by 2.0 psu and the wet season mean salinity values are higher by 0.3 psu. This reflects similar results to the regression analysis. In Faka Union Bay, however, the model results indicate that the rerouted water in the wetlands helps supply the bay with fresh water not coming down Faka Union Canal. The dry season mean salinity values are lower by 0.46 psu and the wet season mean salinity values are lower by 1.3 psu. So the numerical model indicates that salinity changes expected following restoration vary a great deal spatially.

Linear regression (uni- and multivariate) models were developed to perform a preliminarily evaluation of the hydrologic performance of the Picayune Strand Restoration Project as a function of the available observed data for canal flow and salinity. Daily data were used to develop daily models and monthly data were used to develop monthly models. R^2 values ranged from 0.63 to 0.91. USGS Ten Thousand Island Model output data for the restoration project at the appropriate resolution were used as input to the regression models to forecast post-project salinity regime at various monitoring stations over the period of the analysis for which data were available, which was January 2002 through December 2004 at both FDEP and SFWMD stations. The results of this analysis are presented in

Table 9-7, ***Table 9-8*** and ***Table 9-9***. Results from these tables are presented in ***Figure 9-22***, ***Figure 9-23*** and ***Figure 9-24***, respectively.

The regression model estimates indicate that post-project mean salinity values are approximately 2 to 6 psu higher than the observed data for a period, assuming that hydrology is similar to the period of analysis (***Table 9-7***). For the wet season, mean values are estimated be 4 to 9 psu higher (***Table 9-8***), and dry season mean values are estimated to be higher by 0.2 to 2 psu (***Table 9-9***). Regression model error (root mean square) is approximately 2 to 4 psu, similar to the error produced by most salinity models developed for estuaries in south Florida, regardless of the resolution or complexity of the model. This means that the wet season difference seen in regression model output between existing conditions and post-project conditions is greater than model error, which substantiates the difference between current and post-project condition wet season salinities. Regression models indicate that dry season conditions will not be much different than current conditions, which is reasonable given the reduced influence of freshwater supply during the dry season in general. Overall, the project is likely to increase salinity in the estuarine areas adjacent to the canal and in the other embayments in the Ten Thousand Island region. The salinity of the estuarine area away from the direct influence of the canal is likely affected more by the increase in groundwater and overland flows from adjacent freshwater wetlands and tidal creek discharges rather than the direct effect of a reduction of flow in the Faka Union Canal, which is in agreement with the USGS analysis reported above. Spreader swales in the Picayune Strand Restoration Project are designed to redirect flow from the secondary canal system into the aquifer, which will ultimately increase dispersed freshwater flow down-gradient at the coast.

TABLE 9-7. COMPARISON OF ANNUALIZED MEAN SALINITY VALUES FROM OBSERVED DATA, REGRESSION MODEL OUTPUT, AND USGS TEN THOUSAND ISLANDS MODEL RESTORATION OUTPUT

Location	Observed	Regression Model	Difference Observed - Regression	USGS Model Restoration	Difference Regression-USGS	Difference Observed-USGS
Faka Union Bay	20.5	22.0	-1.4	25.9	-3.9	-5.3
Fakahatchee Bay	24.7	26.2	-1.5	29.2	-3.0	-4.5
Middle Blackwater River	27.7	29.1	-1.4	32.0	-2.9	-4.3
Pumpkin Bay	29.8	29.4	0.4	31.8	-2.4	-2.0
TTI65	27.6	27.6	0.1	30.7	-3.2	-3.1
TTI66	29.5	30.1	-0.6	32.9	-2.8	-3.4
TTI67	30.2	30.5	-0.3	33.2	-2.7	-3.0
TTI69	27.7	27.9	-0.2	31.6	-3.7	-3.9
TTI70	20.4	21.2	-0.8	27.0	-5.8	-6.6
TTI71	31.0	31.0	0.0	33.2	-2.2	-2.2
TTI72	29.5	29.3	0.2	31.6	-2.3	-2.1
TTI74	31.5	31.4	0.1	33.3	-1.9	-1.8

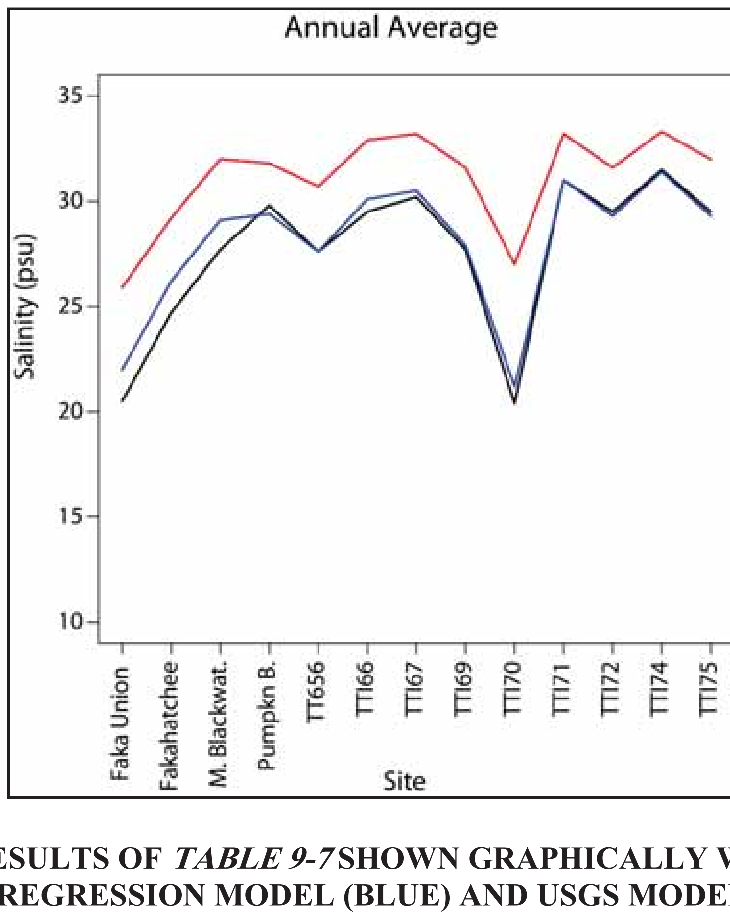


FIGURE 9-21. RESULTS OF *TABLE 9-7* SHOWN GRAPHICALLY WITH OBSERVED DATA (BLACK), REGRESSION MODEL (BLUE) AND USGS MODEL RESTORATION (RED)

TABLE 9-8. COMPARISON OF OBSERVED, REGRESSION MODEL OUTPUT, AND US GEOLOGICAL SURVEY TEN THOUSAND ISLAND MODEL RESTORATION OUTPUT FOR WET SEASON SALINITY

Location	Observed	Regression Model	Difference Observed - Regression	USGS Model Restoration	Difference Regression-USGS	Difference Observed-USGS
Faka Union Bay	13.7	14.7	-1.0	22.3	-7.7	-8.7
Fakahatchee Bay	19.7	20.7	-1.0	26.5	-5.8	-6.8
Middle Blackwater River	23.1	23.6	-0.5	29.4	-5.7	-6.3
Pumpkin Bay	25.2	26.0	-0.8	29.6	-3.6	-4.4
TTI65	23.8	25.1	-1.3	28.2	-3.1	-4.4
TTI66	26.4	28.4	-2.0	30.7	-2.2	-4.3
TTI67	26.9	28.3	-1.5	31.0	-2.7	-4.2
TTI69	22.3	23.4	-1.1	28.6	-5.1	-6.3
TTI70	12.1	14.9	-2.7	21.6	-6.7	-9.4
TTI71	27.4	27.8	-0.4	31.4	-3.6	-4.0
TTI72	25.0	24.7	0.2	29.6	-4.8	-4.6
TTI74	27.3	27.0	0.4	31.6	-4.6	-4.3
TTI75	23.7	23.1	0.5	29.6	-6.5	-5.9

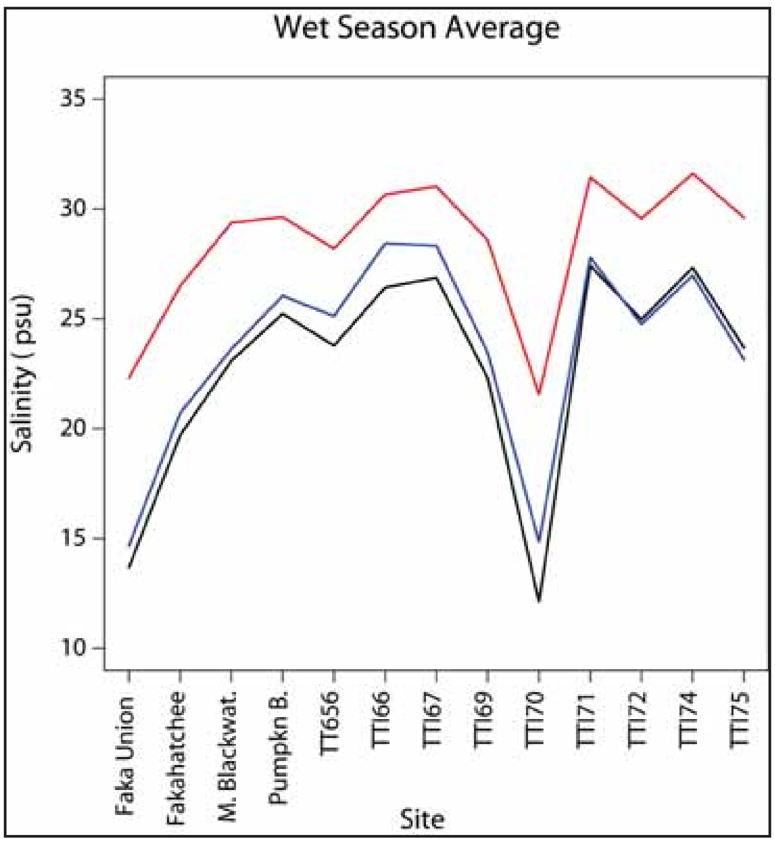


FIGURE 9-22. RESULTS OF TABLE 9-8 SHOWN GRAPHICALLY WITH OBSERVED DATA (BLACK), REGRESSION MODEL (BLUE), AND US GEOLOGICAL SURVEY MODEL RESTORATION (RED)

TABLE 9-9. COMPARISON OF OBSERVED, REGRESSION MODEL OUTPUT, AND US GEOLOGICAL SURVEY TEN THOUSAND ISLAND MODEL RESTORATION OUTPUT FOR DRY SEASON SALINITY

Location	Observed	Regression Model	Difference Observed - Regression	USGS Model Restoration	Difference Regression-USGS	Difference Observed-USGS
Faka Union Bay	26.4	27.3	-0.8	28.4	-1.1	-2.0
Fakahatchee Bay	29.3	30.1	-0.8	31.1	-1.0	-1.8
Middle Blackwater River	32.6	33.0	-0.4	33.9	-0.9	-1.3
Pumpkin Bay	31.9	31.5	0.5	33.4	-1.9	-1.4
TTI65	30.4	28.9	1.5	32.8	-3.9	-2.4
TTI66	31.7	30.8	0.9	34.6	-3.8	-2.9
TTI67	32.5	31.7	0.8	34.9	-3.2	-2.3
TTI69	31.5	30.9	0.6	33.9	-3.0	-2.4
TTI70	26.4	25.4	0.9	31.4	-5.9	-5.0
TTI71	33.5	33.5	0.0	34.6	-1.1	-1.1
TTI72	32.7	32.7	0.0	33.0	-0.3	-0.4
TTI74	34.5	34.8	-0.4	34.6	0.2	-0.2
TTI75	33.7	34.0	-0.4	33.9	0.2	-0.2

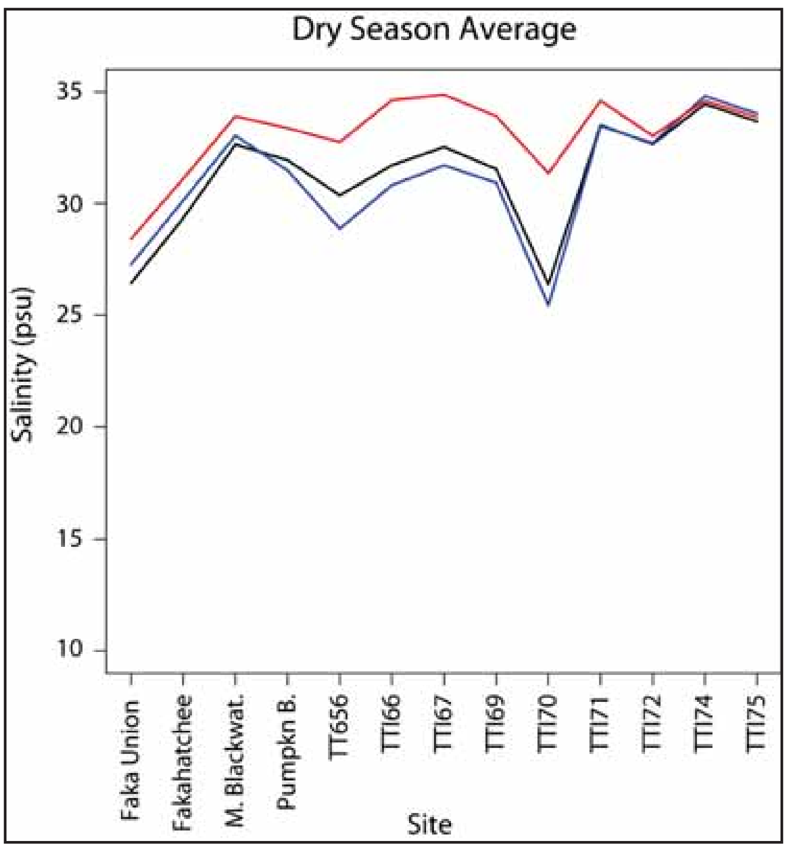


FIGURE 9-23. RESULTS OF TABLE 9-9 SHOWN GRAPHICALLY WITH OBSERVED DATA (BLACK), REGRESSION MODEL (BLUE), AND US GEOLOGICAL SURVEY MODEL RESTORATION (RED)

The utilization of the instrumental data from these two salinity monitoring programs (FDEP daily; SFWMD monthly) with accepted models (USGS Ten Thousand Island numeric model; linear regression models) for this analysis shows the value of the monitoring program as a way of assessing CERP progress on individual projects. The use of the observed data and regression models linked to output from USGS Ten Thousand Islands Model output for the Picayune Strand Restoration Project provides useful information to gauge the progress of restoration in this part of south Florida. Separation of the role that the Faka Union Canal plays in the overall salinity regime from the seasonal hydrology signal is benefited greatly from the use of the USGS model.

Model statistics for both regression modeling exercises (one using salinity at the closest station to the canal discharge point; one using canal flow directly) show the root mean squared error for salinity estimates produced by the models for both daily and monthly resolution is between 2 and 4 psu. Because the difference in mean salinity for the post-Picayune Strand Restoration Project compared to the current condition is approximately the same as the level of uncertainty, it is unclear the degree to which the project may improve salinity, but indications are that it will be small. The tentative findings that salinity would slightly increase as a result of the project seems substantiated given an equivalent volume of water distributed over a wider front and evapotranspiration losses resulting from spreading water over thousands of acres.

The use of the observed data and regression model output for the Picayune Strand Restoration Project can provide useful information to gauge the progress of restoration in this part of south Florida. However, the USGS Ten Thousand Island Model will likely be available in time for the next SSR (i.e., 2012), and will facilitate determining the exact role Faka Union Canal plays in the overall salinity regime.

9.2.6 Assessment of Salinity Performance Measures

Since RECOVER salinity targets for Florida and Biscayne Bays are being employed in a variety of restoration venues (i.e., planning and evaluation of restoration project alternatives and long-range planning) it was deemed prudent to evaluate the underlying assumptions that led to the development of these targets by evaluating their applicability using real-world monitoring data and updated predictive model data recently made available. This assessment marks the first in-depth attempt to evaluate the targets, and was performed for the purpose of initiating proactive correction as may be advisable. RECOVER has not yet developed targets for most of the southwest Florida coast. The targets for Biscayne Bay are based on canal discharge volumes, which were derived from relationships between desired salinity conditions in the bay and observed flow rates. Florida Bay targets were based on desired salinity conditions using target ranges for 17 salinity zones (***Figure 9-24***, ***Table 9-10***) established as part of the Florida Bay Florida Keys Feasibility Study (www.evergladesplan.org/pm/studies/fl_bay.aspx).) As an improved version of the NSM becomes available, or is replaced by a methodology which consensus indicates is a substantial improvement, the following analyses would accordingly be updated; however, the following represents the best available information at the time of preparation of this SSR.

9.2.6.1 Monitoring

Biscayne Bay salinity was evaluated by extracting flow data from the SFWMD‟s DBHydro data repository for the 2004 through 2008 timeframe and comparing it to project targets. Targets have been set for three different regions of Biscayne Bay (north, central and south), Manatee Bay, Barnes Sound, and for two inflow points to Biscayne Bay (Miami River and Snake Creek). Note that Barnes Sound and Manatee Bay are also included in the Florida Bay performance measure, which provides additional information of CERP performance in this important region.

The RECOVER performance measure subdivides Florida Bay into 17 separate zones, with four of those further divided into subzones. For Florida Bay salinity was evaluated using the daily means from 2004 through 2008 for all monitoring data within a given zone and compared to the average annual salinity range target for that zone. For example, Zone 1 has 11 monitoring stations which were combined to obtain a daily overall mean for the zone. The daily overall mean time series was then plotted for each zone and the percent daily values within the target range was determined. ***Figure 9-24*** shows the location of salinity monitoring sites by zone. Note that salinity monitoring data was not collected in five of the zones (Zones 9, 10, 11, 12 and 16).

In addition to analyzing the performance measure targets with real-world monitoring data, output from the SFWMM for the NSM version 4.6.2 scenario was post-processed using multivariate linear regression (MLR) models which utilized water level, wind speed and direction, and sea surface level to estimate pre-drainage salinities at select locations in Florida Bay (Marshall et al., 2003, 2004; Marshall, 2008). The restoration objective is that the restored salinity regime would resemble the predevelopment condition. This analysis was performed as a check on the adequacy of the performance measure targets to reflect desired future condition. It is important to note that the same five-year time span (2004-2008) used for the empirical data was not available for the NSM output, and that NSM output was only available for 22 sites located within 11 of the 17 performance measure defined zones. Since the AMO entered its current phase in 1996, NSM output from 1996 through 2000 was regarded as the most comparable.

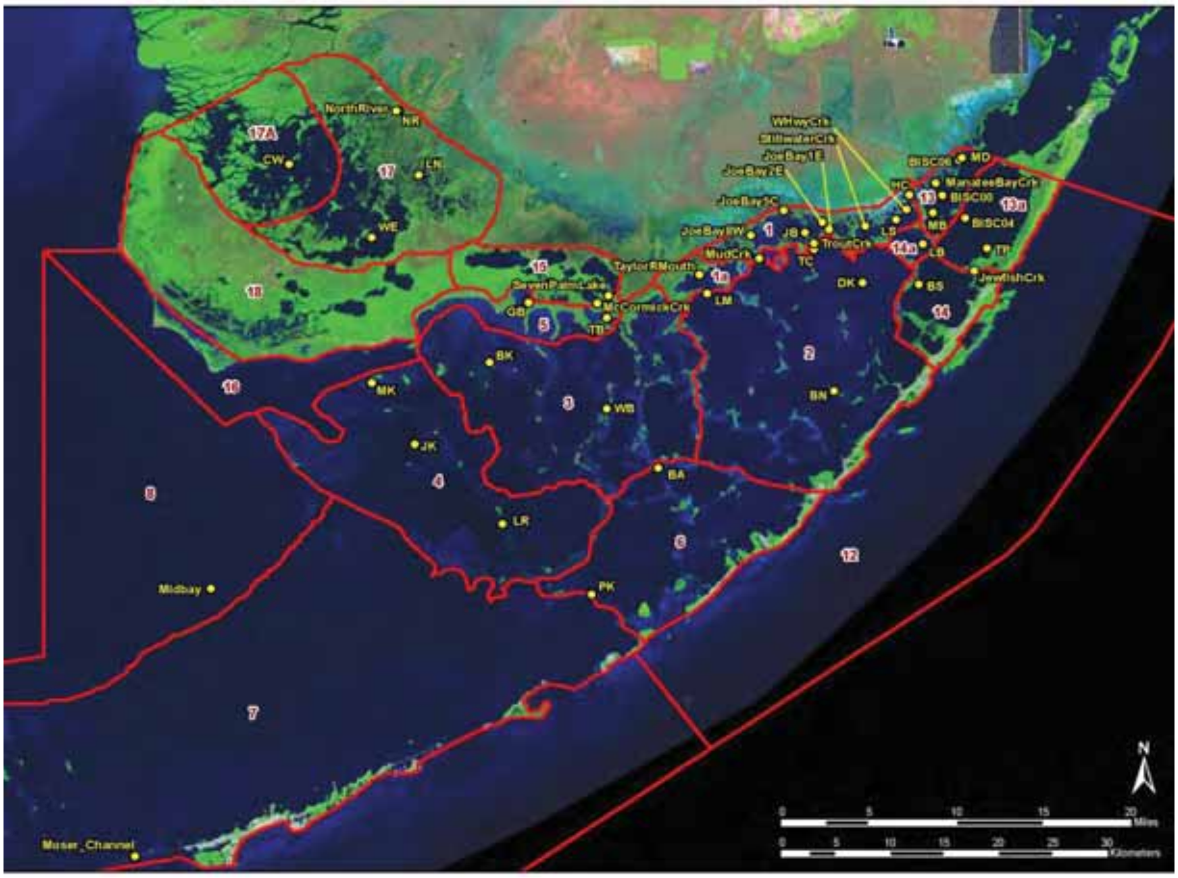


FIGURE 9-24. SALINITY ZONES IN FLORIDA BAY ON WHICH THE PERFORMANCE MEASURE IS BASED AND CONTINUOUS SALINITY MONITORING SITES WHERE CURRENTLY AVAILABLE

TABLE 9-10. PERFORMANCE MEASURE SALINITY ZONES AND REGION NAMES WITHIN FLORIDA BAY

Zone(s)	Region(s)
1	Highway Creek, Long Sound and Joe Bay
1a	Little Madeira Bay
2	Butternut Key and Duck Key
3	Buoy Key, Whipray Basin, and Bob Allen Key
4	Murray Key, Johnson Key, Rabbit Key, and Peterson Key
5	Terrapin Bay and Garfield Bight
6	South central Florida Bay
7,8,9	Western Florida Bay
10,11,12	Offshore Florida Keys
13	Manatee Bay
13a	Barnes Sound
14	Blackwater Sound
14a	Little Blackwater Sound
15	Seven Palm Lakes
16	Cape Sable
17	Whitewater Bay
17A	Clearwater Pass

9.2.6.2 Results

9.2.6.2.1 Biscayne Bay

The desired restoration conditions for targeted geographic regions of Biscayne Bay are:

1) north Biscayne Bay, maintain the current salinity patterns, which should conserve the current seagrass communities and associated faunal communities that exist in this section of the bay;

2) central Biscayne Bay, maintain mesohaline conditions near the western shore to allow for maintenance or establishment of faunal communities consisting of mullet, shoal grass and other estuarine species;

3) south Biscayne Bay, provide mesohaline salinity patterns in the nearshore environment of the bay and lower salinity at the mouths of tidal creeks;

4) Miami River, increase freshwater inflow into the bay providing expanded estuarine conditions; and

5) Snake Creek, maintain and improve conditions for the oyster bed growth at the mouth of Snake Creek through controlling the level of freshwater discharge through the S-29 control structure.

To achieve these desired restoration conditions, a more natural salinity regime should be established. For some areas, a specific salinity range has been determined, while additional monitoring needs to be conducted in others to determine the specific salinity range. For all areas, flows requirements have been set. ***Table 9-11*** summarizes these targets. Manatee Bay and Barnes Sound are discussed below.

TABLE 9-11. BISCAYNE BAY SALINITY AND FLOW REQUIREMENTS

Area	Structures or Canals	Salinity Range (psu)	Flow Requirement (acre-feet)
North Biscayne Bay	S-27, S-28	current	107,000 annual wet season 49,000 annual dry season
Central Biscayne Bay	S-22	current mesohaline	22,392 - 50,360 average monthly
South Biscayne Bay	C-1, C-100, C-102 C-103, Military	≤ 20	321,000 annual wet season 146,000 annual dry season
Miami River		oligohaline/ euhaline	3,000 7-day total 80% of time < 99 per day < 10% of time
Snake Creek	S-29	5 - 25	1,120 - 41,470 average monthly

North Biscayne Bay. The goal in north Biscayne Bay is to preserve, in healthy condition, the existing seagrass beds by maintaining the current levels of fresh water flowing through the S-27 and S-28 control structures and, thereby, current salinity levels. The flow targets are an annual average of 107,000 acre-feet for the wet season (June through October) and 49,000 acre-feet for the dry season (November through May). The dry season target is achieved in all years, while the wet season target was achieved in all years except 2006 (***Figure 9-25***).

Central Biscayne Bay. The salinity target for central Biscayne Bay is to maintain mesohaline conditions. Restoration targets for surface water discharges from the Snapper Creek Canal (C-2) through the S-22 structure to central Biscayne Bay is based on the average monthly flow, which should be maintained between 22,392 acre-feet per month and 50,360 acre-feet per month. Targets are not achieved in the dry season and are only achieved during some months of the wet season (***Figure 9-26***). The performance measure target was met approximately 13 percent of the time during 2004 to 2008.

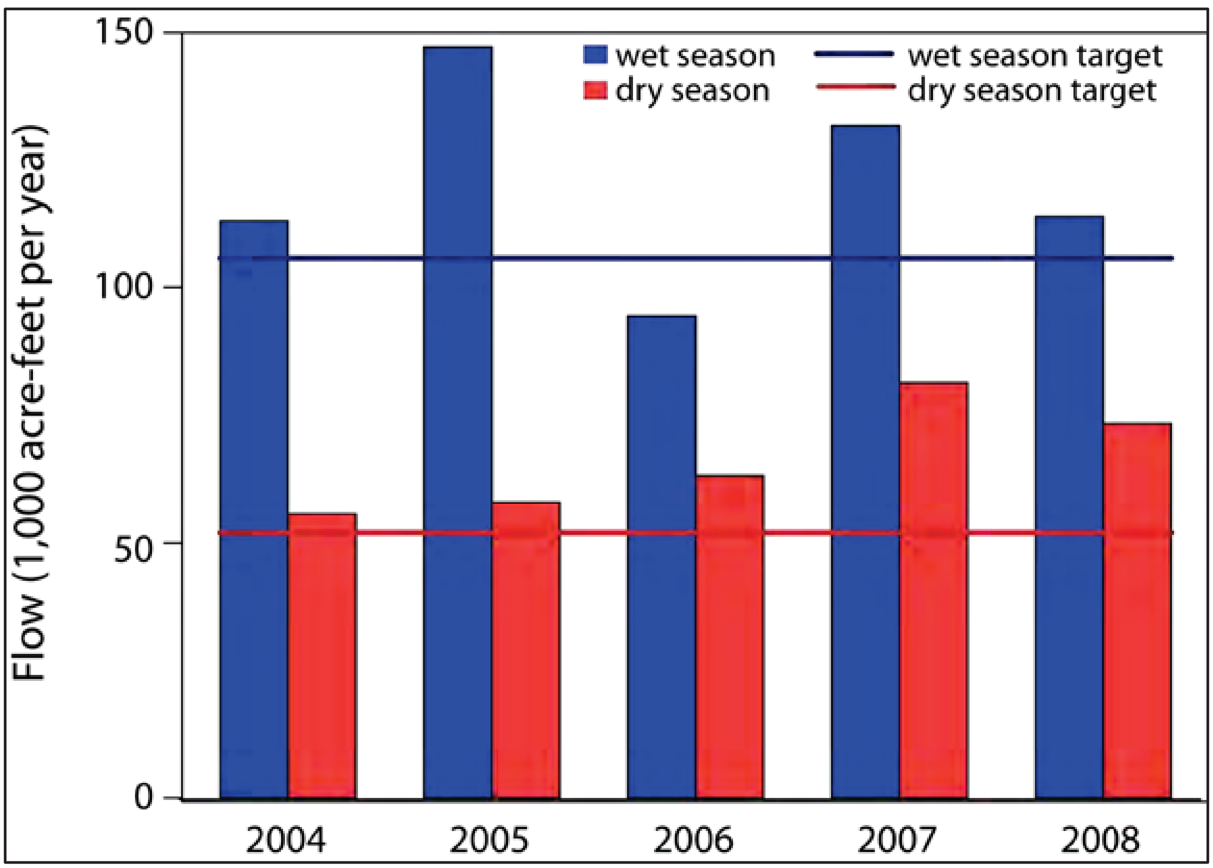


FIGURE 9-25. AVERAGE ANNUAL WET SEASON AND DRY SEASON FLOWS FROM THE S-27 AND S-28 CONTROL STRUCTURES TO NORTH BISCAYNE BAY COMPARED TO THE WET AND DRY SEASON TARGETS

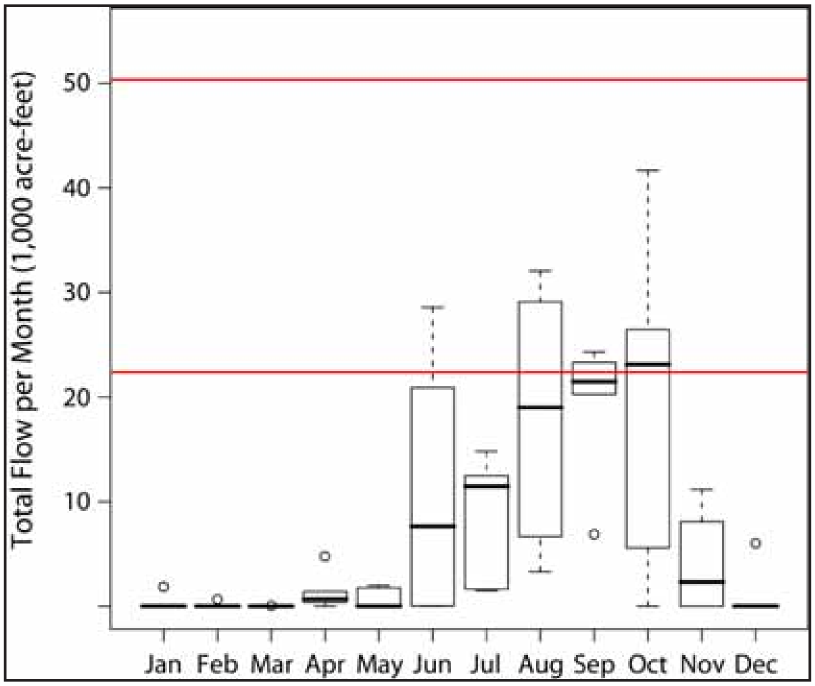


FIGURE 9-26. BOX-AND-WHISKER PLOTS OF AVERAGE MONTHLY DISCHARGES FROM SNAPPER CREEK CANAL (C-2) THROUGH S-22 TO CENTRAL BISCAYNE BAY COMPARED TO UPPER AND LOWER MONTHLY MEAN TARGET RANGE (RED LINES)

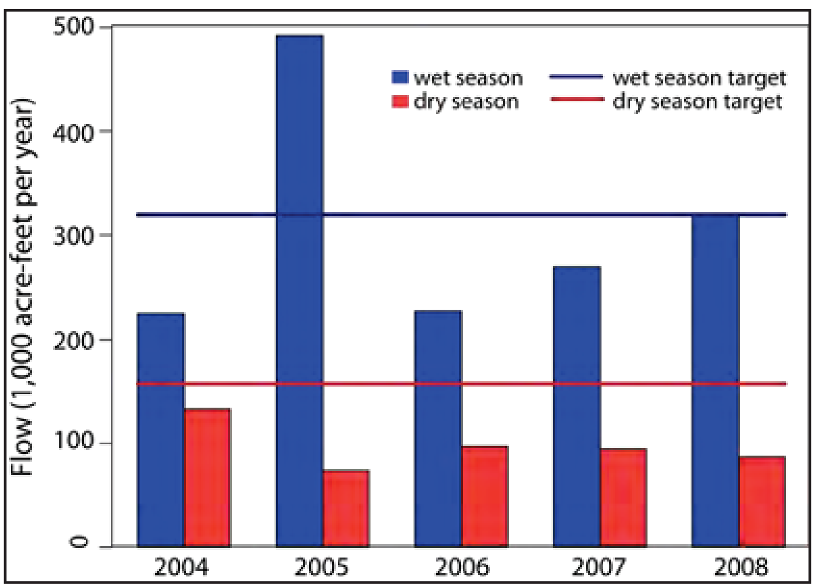


FIGURE 9-27. AVERAGE ANNUAL WET AND DRY SEASON DISCHARGES TO SOUTH BISCAYNE BAY COMPARED TO THE WET AND DRY SEASON TARGETS

South Biscayne Bay. Restoration targets for south Biscayne Bay are based on creation of a persistent positive salinity gradient from freshwater wetlands into the bay with an average bottom salinity of 20 psu in a zone extending 500 meters from shore in the wet season, and 250 meters from shore in the dry season. Meeting these salinity targets requires a total wet season flow of 321,000 acre-feet, and total dry season flow of 146,000 acre-feet, where that total is the sum of flows being discharged into the bay from C-1, C-100, C102, C-103, and Military Canals. The wet season target was only met in one year (2005) while the dry season target was not met in any year (***Figure 9-27***).

The seasonal distances from shore have been converted to acres (red zone) in ***Figure 9-28***.

Estimates of the actual acreage exhibiting a bottom salinity of 20 psu or less was spatially interpolated from the BNP high resolution salinity monitoring network (blue line in ***Figure 9-28***). Although the wet season acreage is partially achieved in some years, it is not sustained, and accordingly equates with faunal stress. For the most part, dry season acreage meeting the 20 psu criteria is generally non-existent, an artifact of the lack of storage necessary to extend wet season water deliveries into the dry season.

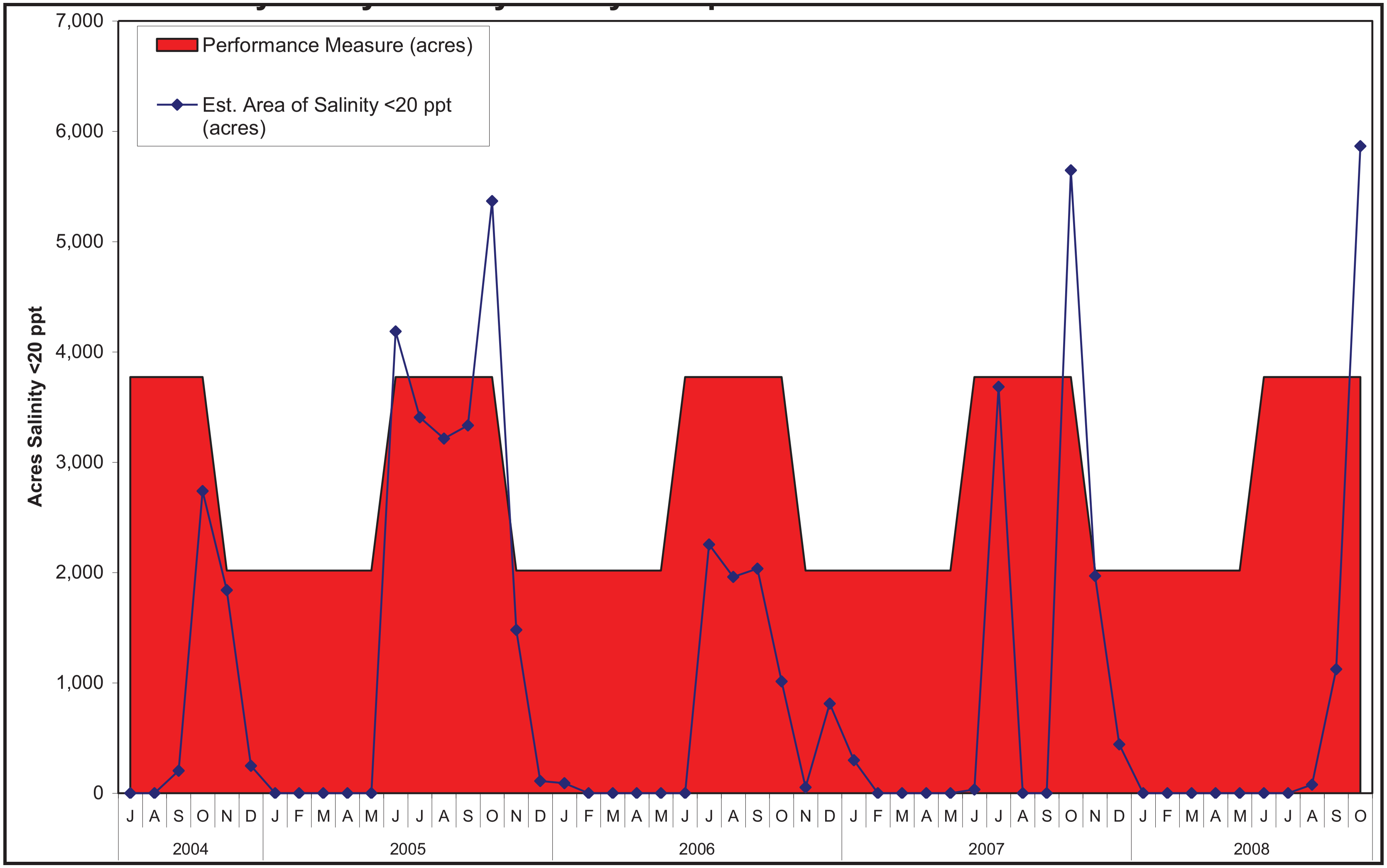


FIGURE 9-28. ACRES IN SOUTH BISCAYNE BAY THAT MEET THE SALINITY TARGET OF 20 PRACTICAL SALINITY UNITS OR LESS DURING THE WET AND DRY SEASON 500 METERS AND 250 METERS FROM SHORE, RESPECTIVELY

Miami River. The target is oligohaline conditions within the Miami River and euhaline conditions within Biscayne Bay. Flow from the Miami River to Biscayne Bay targets are 1) a total weekly flow of greater than 3,000 acre-feet 80 percent of the time, and 2) a daily flow rate less than 99 acre-feet less than ten percent of the time. Based on 2004-2008 flow data, total weekly flow was greater than 3,000 acre-feet in 64 percent of the time, while daily flow was less than 99 acre-feet 30 percent of the time (***Figure 9-29***). The average volume discharged to tide in excess of the desired performance measure target (3,000 acre-feet per week) was around 236,000 acre-feet per year during this five-year period.

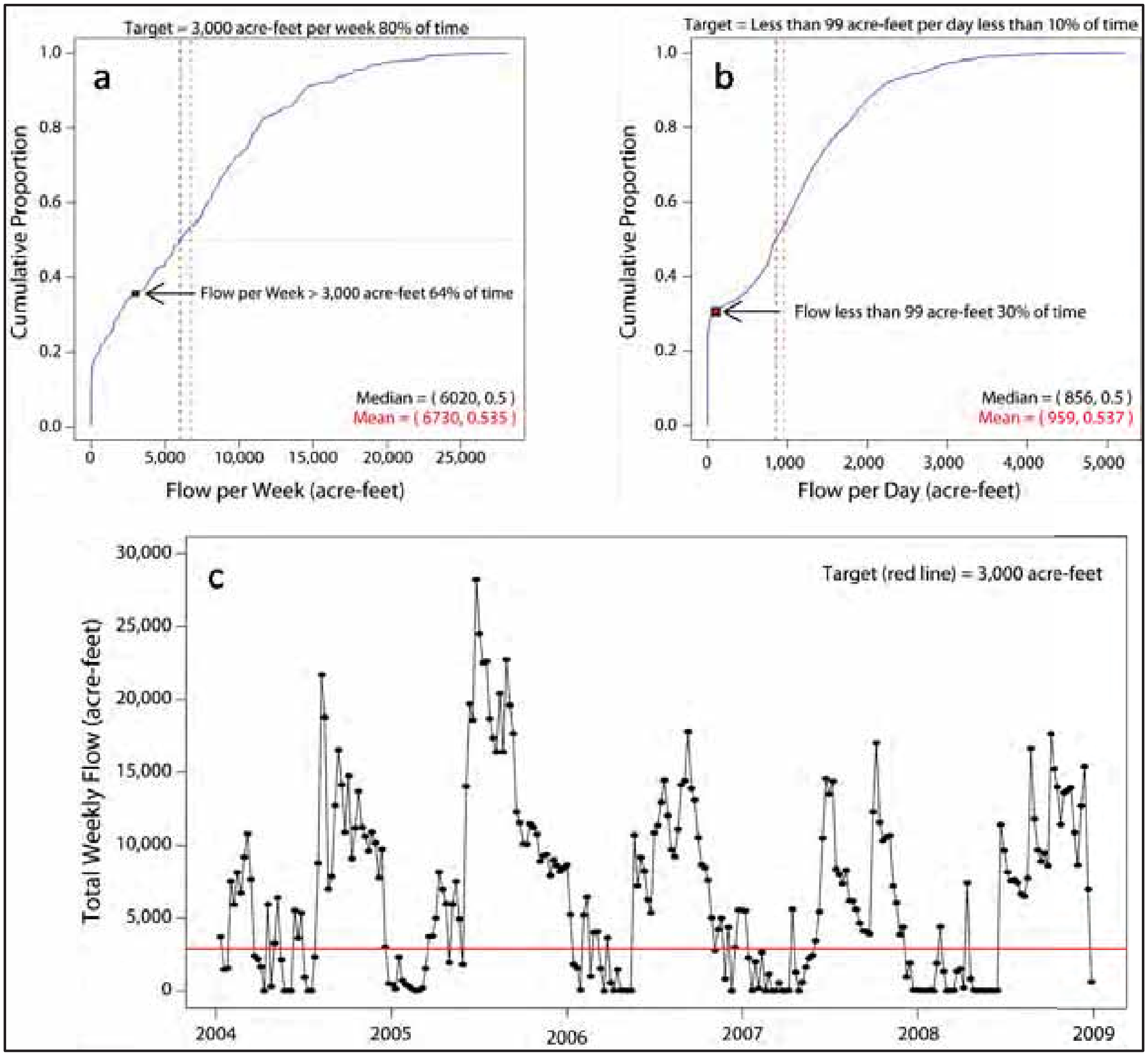


FIGURE 9-29. PERFORMANCE MEASURE EVALUATION OF CURRENT PATTERN OF MIAMI CANAL FLOW TO BISCAYNE BAY. DISTRIBUTION OF FLOW BY WEEK (A), BY DAY (B), AND WEEKLY TOTAL FLOW BY MONTH 2004-2008 (C).

Snake Creek. The target for Snake Creek is to maintain salinity downstream of the S-29 structure between five and 25 psu. The flows required to maintain this salinity are 1,120 to 41,470 acre-feet per month from the S-29 control structure on Snake Creek Canal (C-9) to north Biscayne Bay. Flow from Snake Creek under current conditions meets this target 75 percent of the time (***Figure 9-30***).

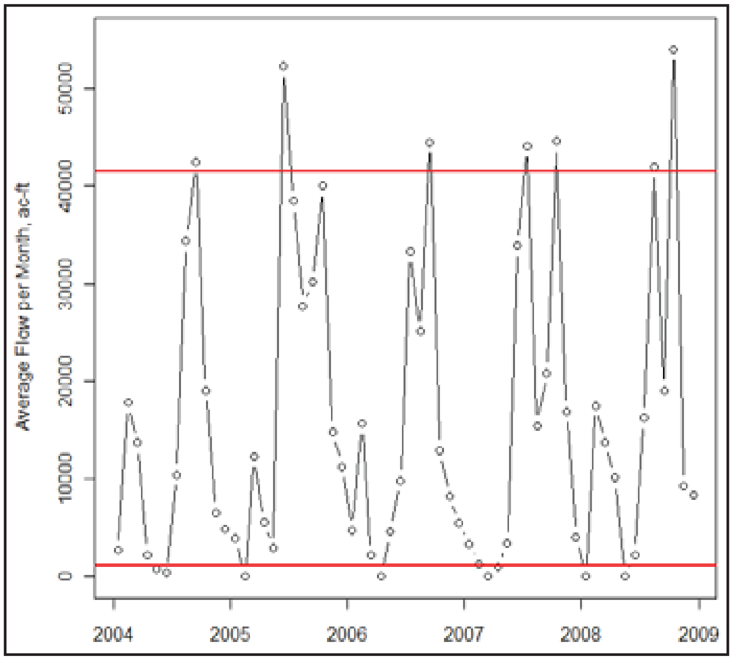


FIGURE 9-30. AVERAGE MONTHLY FLOW FROM THE S-29 CONTROL STRUCTURE ON SNAKE CREEK CANAL (C-9) TO NORTH BISCAYNE BAY FROM 2004-2008 WITH UPPER AND LOWER MONTHLY MEAN TARGETS (RED LINES)

Manatee Bay and Barnes Sound. ***Table 9-12*** shows the seasonal salinity targets have been set for Manatee Bay and Barnes Sound (see ***Figure 9-24*** for location). These targets need to be met 90 percent of the time. The targets for Manatee Bay are met zero percent of the time during the wet season and 0.3 percent of the time in the dry season (***Figure 9-31***). The targets for Barnes Sound are achieved 22 percent of the time during the wet season and 53 percent of the time in the dry season (***Figure 9-32***). In addition to average salinities being consistently higher than the restoration targets, extreme salinities are often recorded. The salinities maintain a core range between 22 and 40 psu. When comparing the data from dry season to wet season a strong correlation between rainfall, canal flow and large salinity changes is not evident.

TABLE 9-12. SALINITY TARGETS FOR MANATEE BAY AND BARNES SOUND

Area	Wet Season Salinity	Dry Season Salinity
Manatee Bay	5 – 15 psu	10 – 19 psu
Barnes Sound	15 – 30 psu	20 – 32 psu

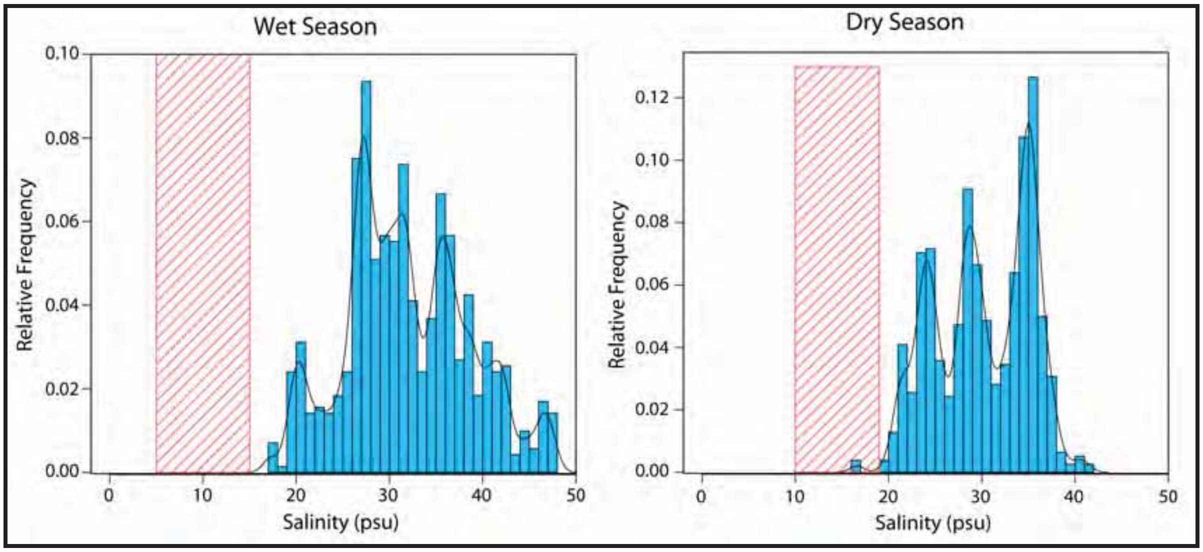


FIGURE 9-31. MANATEE BAY SALINITY DISTRIBUTION FOR 2004-2008 COMPARED TO THE TARGET SALINITY RANGE (RED SHADED AREA)

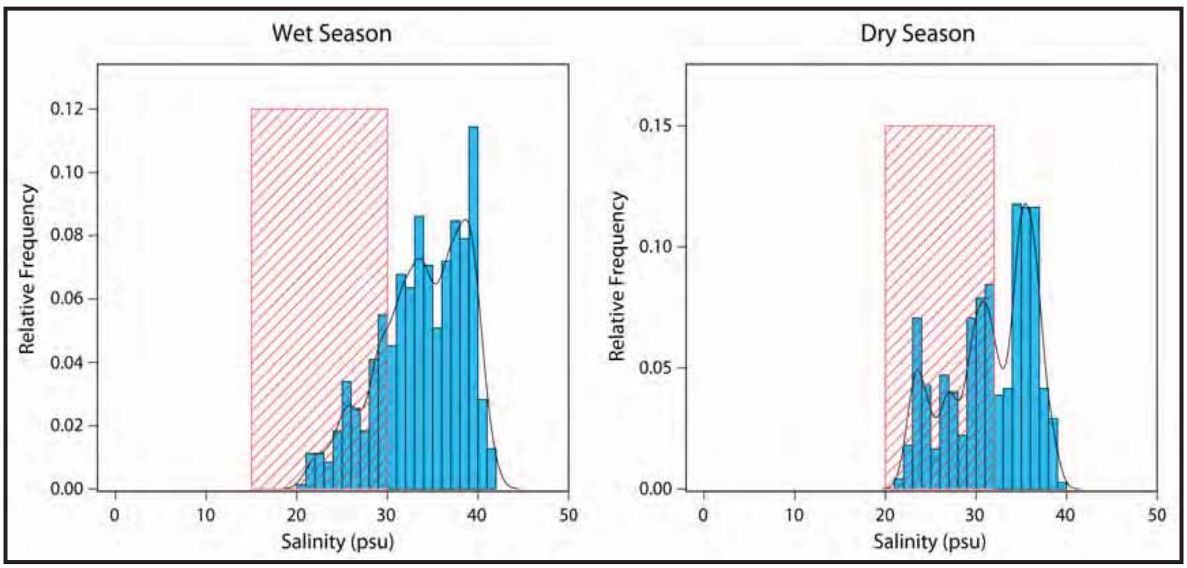


FIGURE 9-32. BARNES SOUND SALINITY DISTRIBUTION FOR 2004-2008 COMPARED TO THE TARGET SALINITY RANGE (RED SHADED AREA)

9.2.6.2.2 Florida Bay

The salinity restoration objectives for Florida Bay are to:

1) provide less abrupt and less extreme decreases in salinity in the northeastern bay;

2) reduce the frequency, duration, magnitude and extent of hypersaline conditions throughout the bay;

3) increase the frequency and extent of lower salinity conditions in the bay; and

4) lower salinity to oligohaline levels in coastal lakes and basins in the mangrove estuarine areas.

Restoration of natural salinity patterns include extending the duration of relatively low, wet season salinities into the late fall and early winter. This would be accomplished by restoring the up-gradient Everglades, which would effectively act as a hydrologic storage area extending the duration of flow necessary to reestablish and sustain a more biologically functional salinity regime.

Based on best professional judgment and studies of bay history, the area has been subdivided into geographic zones with desired annual salinity target ranges defined for each (***Table 9-13***). The salinity range specified is not a prescribed maximum and minimum; but rather atargeted central tendency for the majority of values. The magnitude of extremes varies from location to location, season to season, and year to year depending on water depth, seawater flushing, bay water circulation, rainfall, evaporation, freshwater runoff from the Everglades, and groundwater influx. ***Table 9-13*** also indicates those locations for which NSM output is available. A side-by-side analysis of observed and NSM-based salinity evaluations at all zones in which a comparison was possible are provided in ***Figure 9-33*** through ***Figure 9-43***. Upper and lower salinity ranges (targets) are indicated, respectively, by the blue and red horizontal lines. Analysis of the currently proposed targets has revealed that some do not adequately reflect desired condition (the caveat being that the current version of NSM suitably defines the desired condition).

TABLE 9-13. ZONES, SITE LOCATIONS AND SALINITY TARGETS USED TO ASSESS FLORIDA BAY SALINITY, ALONG WITH LOCATIONS FOR WHICH NSM OUTPUT IS AVAILABLE

Zone*	Salinity Target (psu)	Site Location	NSM Output Locations
1	5-15	Highway Creek	X
		Long Sound	X
		Mud Creek	
		Stillwater Creek	
		Trout Creek	
		West Highway Creek	
1a	15-25	Taylor River Mouth	
2	15-30	Butternut Key	X
		Duck Key	X
		Little Madeira Bay	X
		Trout Cove	X
3	25-35	Buoy Key	X
		Whipray Basin	X
4	30-35	Johnson Key	X
		Little Rabbit	X
		Murray Key	X
5	15-35	Garfield Bight	X
		Terrapin Bay	X
6	25-35	Bob Allen	X
		Peterson Key	X
7	≤ base	Moser Channel	
8	≤ base	Western Florida Bay	
13	5-15	Manatee Bay	
		Middle Key	X
		Manatee Bay Creek	
13a	10-30	Barnes Sound	
		Thursday Point	
		Jewfish Creek	
14	15-30	Blackwater Sound	X
14a	10-20	Little Blackwater	X
15	0-5	McCormick Creek	
		Seven Palm Lake	
17	5-15	Lane Bay	X
		Whitewater Bay East	X
		North River	X
17a	5-20	Clearwater Pass	X

* See ***Figure 9-24*** for location of zones.

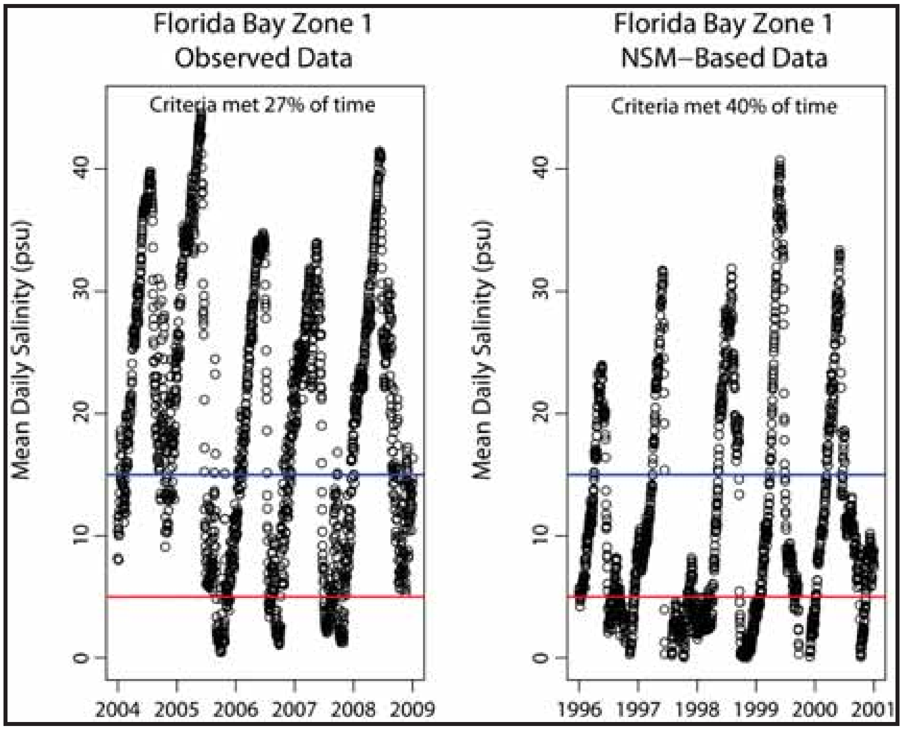


FIGURE 9-33. ZONE 1 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

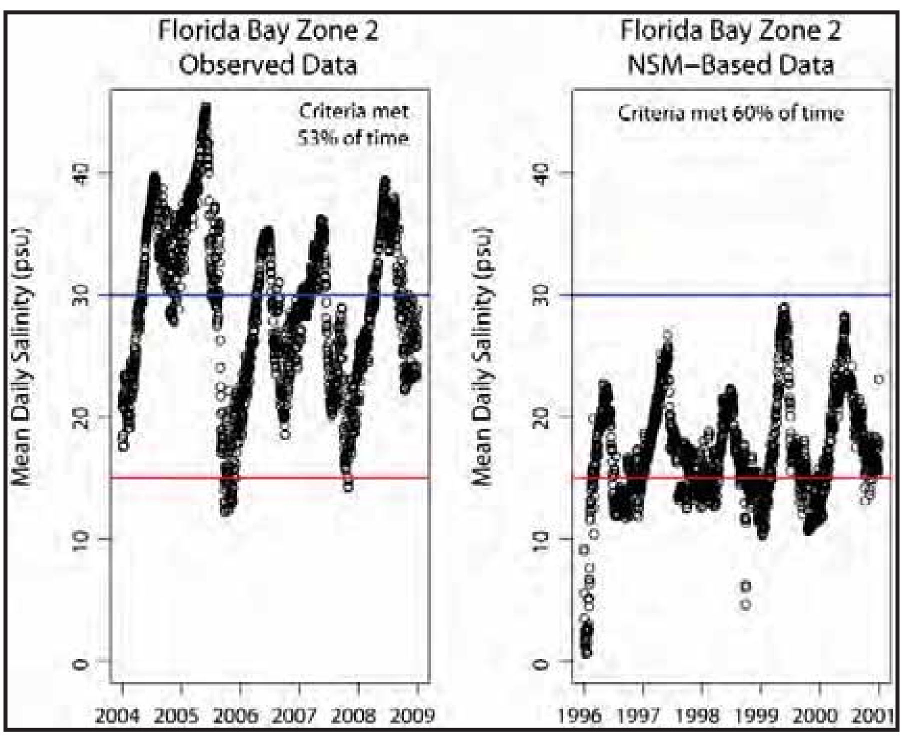


FIGURE 9-34. ZONE 2 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

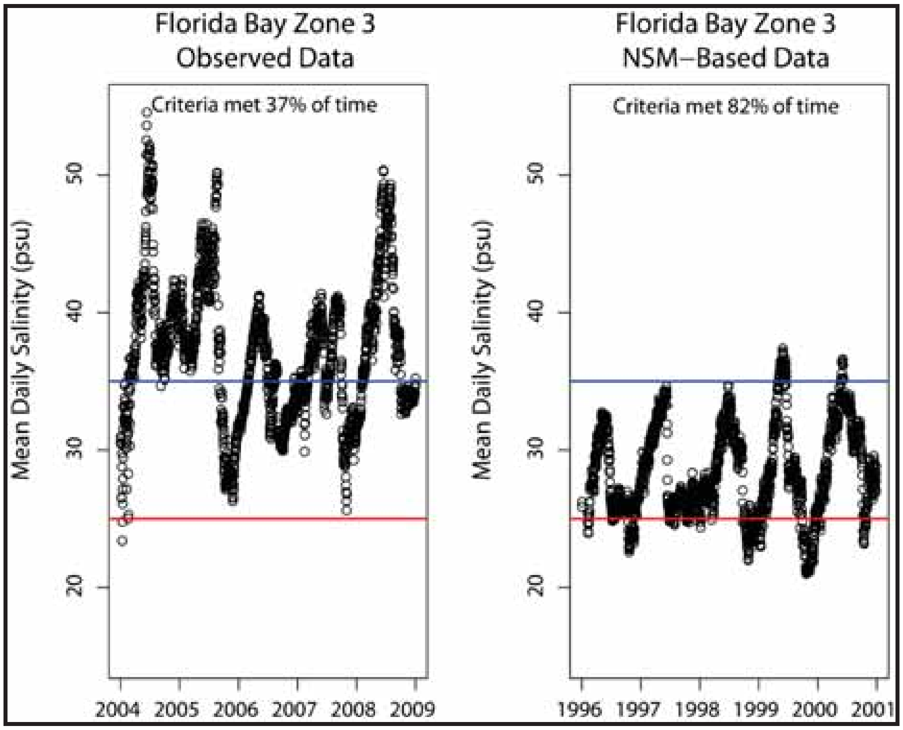


FIGURE 9-35. ZONE 3 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

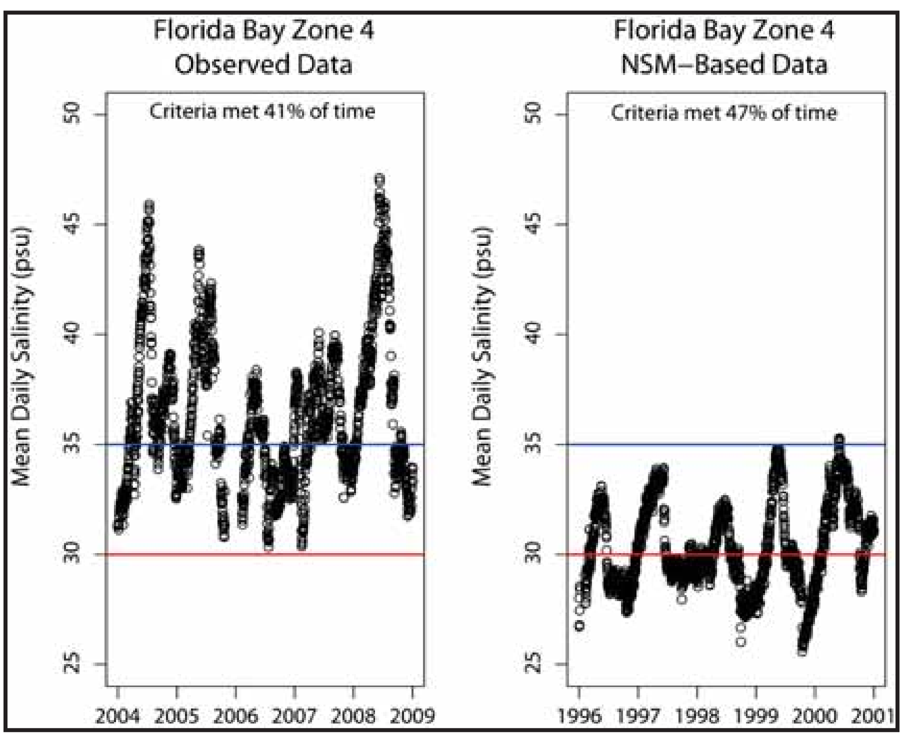


FIGURE 9-36. ZONE 4 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

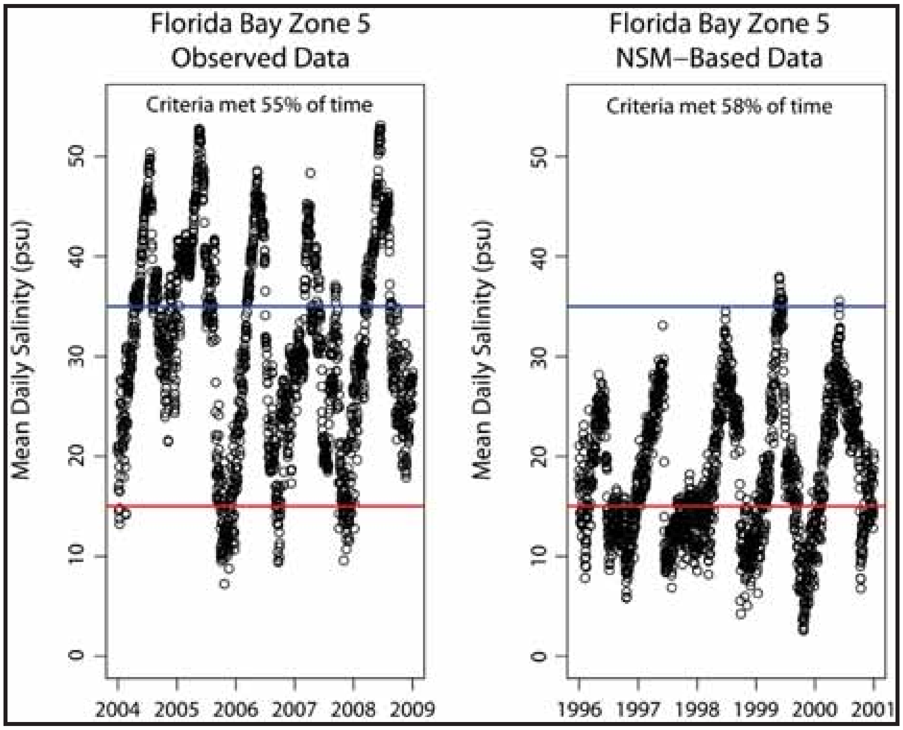


FIGURE 9-37. ZONE 5 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

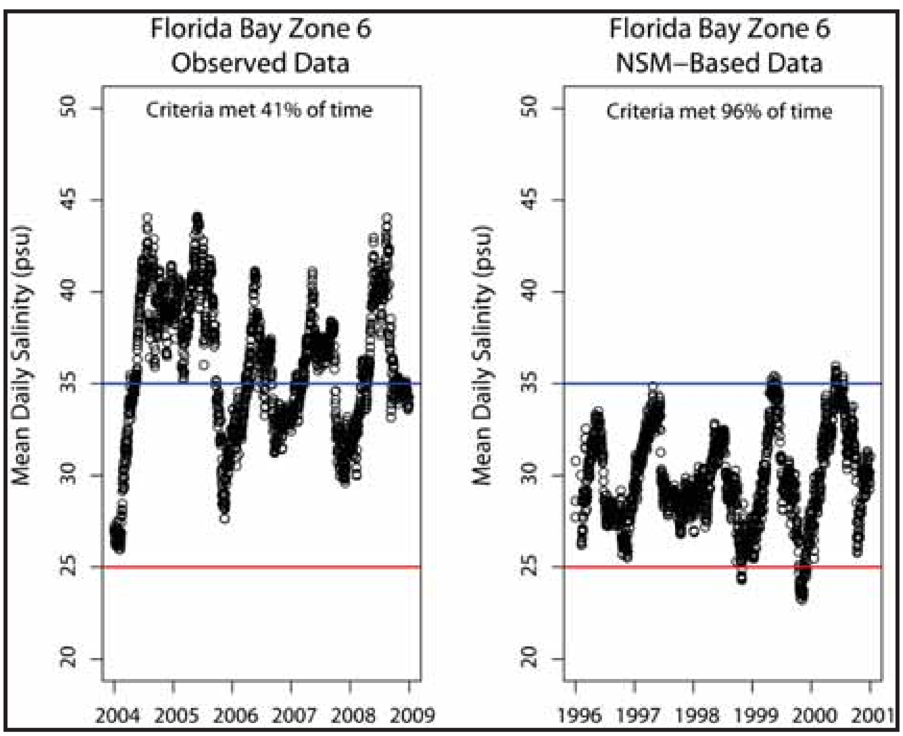


FIGURE 9-38. ZONE 6 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

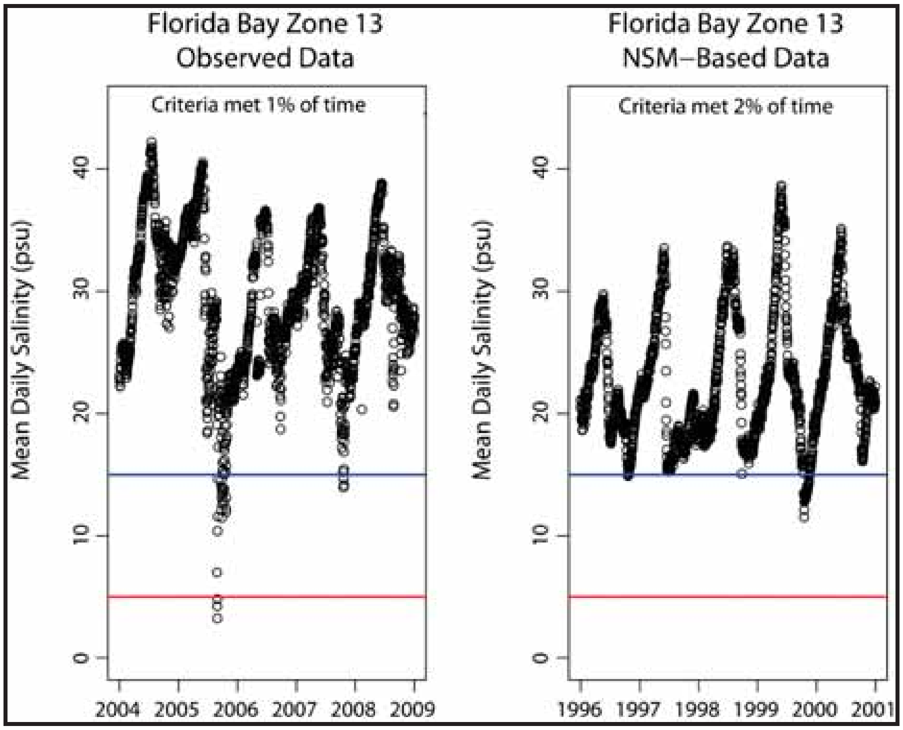


FIGURE 9-39. ZONE 13 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

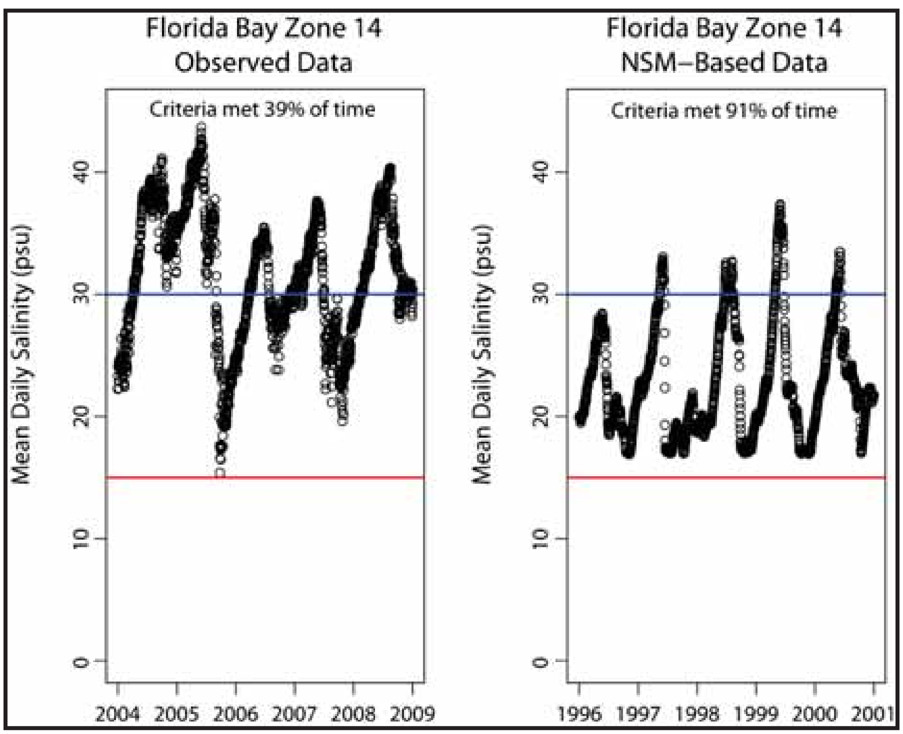


FIGURE 9-40. ZONE 14 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

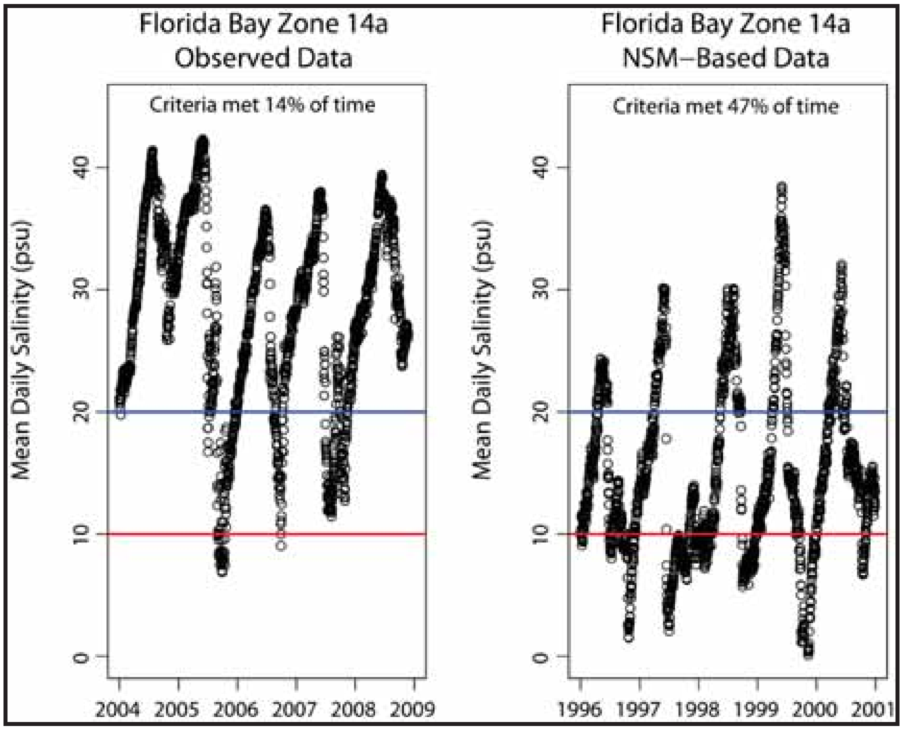


FIGURE 9-41. ZONE 14A MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

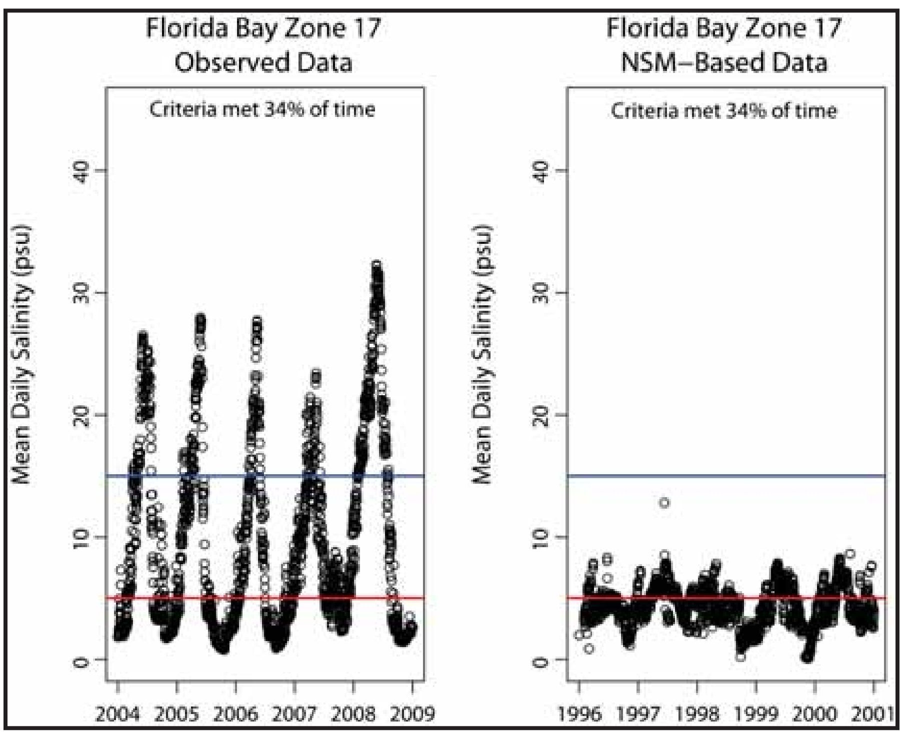


FIGURE 9-42. ZONE 17 MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

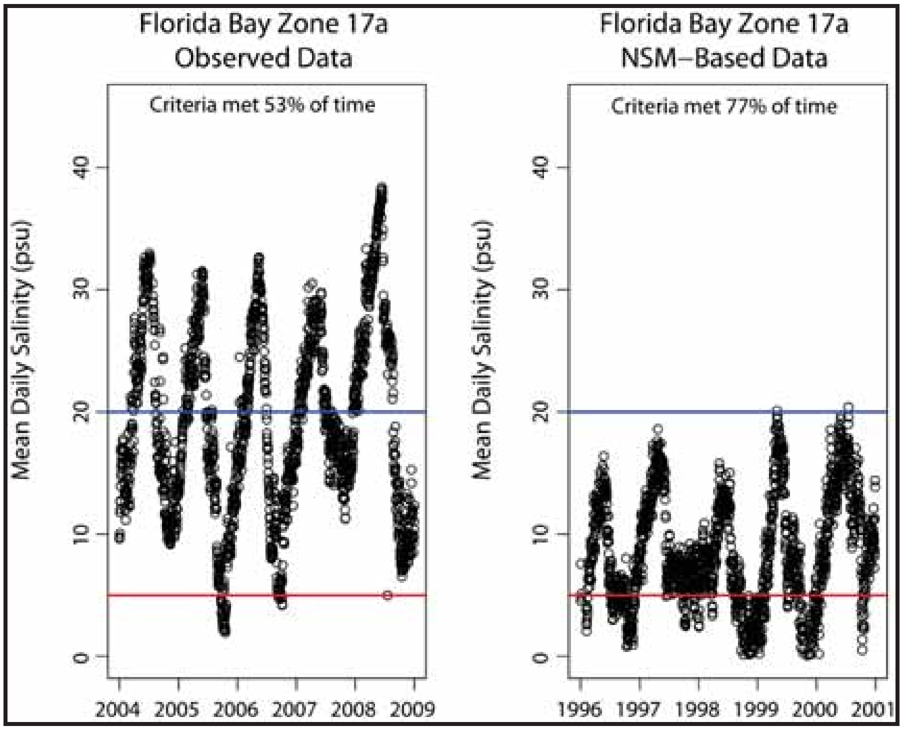


FIGURE 9-43. ZONE 17A MEAN DAILY SALINITY OBSERVED DATA (LEFT) COMPARED TO NSM-BASED DATA (RIGHT)

Data appropriate for evaluation were available for only 11 of the 17 zones. ***Table 9-14*** presents the results of the target evaluation for each zone. It is important to note that the purpose of this evaluation was to examine the target, not whether the target was being achieved. With only the C-111 and Biscayne Bay Coastal Wetlands Projects poised to make substantive changes, and both of these only partially implemented in the Phase I build-out, salinity regime changes due to project implementation are currently in their formative stage. Properly evaluating these changes, as the changes materialize, requires appropriate targets, i.e., the purpose of this exercise.

TABLE 9-14. EVALUATION OF SALINITY TARGETS AND WHETHER CURRENT SALINITY CONDITIONS (OBSERVED) MEET TARGET FOR THE 11 ZONES FOR WHICH DATA WERE AVAILABLE.

Zone	Salinity Target[1] (psu)	Expected[2]	Percentage Within Target NSM Evaluation[3] (1996-2000)	Percentage Within Target Observed[4] (1996-2000)	Consistent Target[5]	Currently Meets Target[6]
1	5-15	> 50	40	27	No	
2	15-30	> 50	66	53	Yes	Yes
3	25-35	> 50	82	37	Yes	No
4	30-35	> 50	47	47	No	
5	15-35	> 50	58	55	Yes	Yes
6	25-35	> 50	96	41	Yes	No
13	5-15	> 50	2	1	No	
14	15-30	> 50	91	39	Yes	No
14A	10-20	> 50	47	14	No	
17	5-15	> 50	34	34	No	
17A	5-20	> 50	77	53	Yes	Yes

1) predetermined, zone-specific salinity targets (i.e., salinity ranges);
2) the percentage of NSM values that are expected to fall within each target, which is set at greater than 50 percent for all zones;
3) the percentage of values predicted by the NSM to fall within each target for the 1996 thorugh 2000 period of record;
4) the percentage of current observations falling within the target;
5) a determination (yes/no) of whether the target is appropriate; and
6) for those zones in which the target appeared appropriate, a determination of whether current conditions fall within this target.

Performance measure evaluation results indicate that approximately half of the zone-specific salinity targets may be inappropriate as less than 50 percent of NSM-predicted values fell within the target, suggesting that the targets may warrant revision. Salinity targets appear appropriate for the remaining six zones. For example, Zone 6 (***Figure 9-40.*** Zone 14 mean daily salinity observed data (left) compared to NSM-based data (right) appears to represent a performance measure target that captures NSM-based restoration expectations; conversely, NSM-predicted salinity in Zone 13 (***Figure 9-41.*** Zone 14a mean daily salinity observed data (left) compared to NSM-based data (right)) rarely achieves the current RECOVER target and warrants further evaluation either in the NSM model, the target, or both.

Evaluation of the RECOVER performance measure targets using current monitoring data indicate that among the six zones exhibiting NSM-appropriate targets, salinity targets appeared to be achieved under current conditions in Zones 2, 5 and 17A. However, these findings tell only a part of the necessary story - although the current condition may meet the target greater than 50 percent of the time, salinity conditions may not be meeting faunal requirements due to lack of sufficient stability. Wide swings from mesohaline to hyperhaline condtion may exceed metabolic limitations resulting in failed reproduction, mortality, or avoidance. As an example, see Zone 5 (***Table 9-14.*** Evaluation of salinity targets and whether current salinity conditions (observed) meet target for the 11 zones for which data were available.).

The objectives of the performance measure targets are to provide a feedback mechanism to trigger AM. Presently, guidance on CERP AM implementation is being drafted. If the targets have been appropriately defined and are not being met, changes in operation or other considerations would be explored. If it is determined that the targets have not been appropriately designed – as is apparent – the science underlying these targets would have to be re-evaluated and the targets updated.

Another difficulty that became apparent as a result of these analyses was the realization that performance measure zones were generally laid out without consideration of the location of monitoring sites with which to evaluate them. As a result, some zones had multiple monitoring sites while some had none. This situation may be overcome to a degree with calibrated models based on observed data; however, this solution would require that model runs are maintained reasonably up-to-date. Currently, model output lags real-time observations by around five years. Furthermore, the current layout of zones requires averaging nearshore with farther from shore monitoring locations, a bias that the model alone could not overcome. These sundry problems, as revealed by analyzing the performance measures against the best available data, have prompted an initiative to overhaul the RECOVER Southern Coastal Systems salinity performance measures.

9.2.7 C-111 Project Salinity Data

As the phased implementation of the C-111 Project progresses, water deliveries will be altered and result in changes in salinity. Thorough evaluations and reports on the continuous salinity monitoring sites that may detect these changes will be addressed in the next SSR (e.g., 2012). It appears that detecting these potentially subtle changes should be possible by looking at flows, seasonal differences in salinity, and rainfall pattern.

9.2.8 Summary and Conclusions

The south Florida coastal ecosystem is economically and environmentally important. Understanding the circulation and water property patterns of Florida Bay and surrounding waters is of vital importance if the health of the coastal ecosystem is to be factored into an iterative adaptive restoration strategy. Salinity is arguably the most crucial factor driving the ecological health and suitability as nursery, refuge and food web habitat for aquatic and aquatic-dependent fauna. Deciphering and tracking restoration induced changes in salinity is being achieved through a sustained research and monitoring program that integrates analyses from moored instrument arrays, regular cruises, drifter deployments and numerical modeling. The primary outcomes are rigorous quantification of the pre-restoration baseline conditions, testable hypotheses, predictive models and alternative management options. The desired restoration outcome is a reduction in the intensity, frequency, duration and spatial extent of high salinity events; reestablishment of mesohaline to oligohaline conditions in nearshore zones; and a reduction in the frequency of rapid salinity fluctuations resulting from pulse releases of fresh water from canals.

9.3 WATER QUALITY STRESSOR

9.3.1 Introduction and Background

Water quality in the Southern Coastal System is dependent upon the volume, distribution and quality of fresh water flowing to the system, along with anthropogenic nutrient inputs to the coastal waters from runoff from agricultural and urban areas being primarily transported in canals (***Figure 9-44***). Other sources include atmospheric deposition and exchange with the Atlantic Ocean and/or Gulf of Mexico. The latter is affected by inputs from Florida‟s large geologic deposits of phosphate and urban development from Tampa to Naples. Different regions and basins are differentially influenced by these local or remote sources, depending on the magnitude of input of different nutrients, internal cycling pathways and rates, and water residence time (Boyer et al., 1999; Rudnick et al., 1999; Childers et al., 2006), freshwater runoff patterns (Nuttle et al., 2000; Kelble et al., 2007), circulation (Lee et al., 2006), sediment biogeochemistry (Zhang et al., 2004), nutrient inputs (Rudnick et al., 1999), grazer biomass (Peterson et al., 2006), and phytoplankton species composition (Phlips and Badylak, 1996). Biotic components of estuaries are sensitive to nutrient loading rates, and these rates may be modified by efforts to restore the overall south Florida ecosystem.

The degree to which external N and P inputs historically affected the Southern Coastal System estuaries is unclear as no pre-development water quality data exists. Historical volumes of water delivered to Biscayne and Florida Bays and to the southwest coast were undoubtedly higher than they are currently and, to the degree possible, restoring these flows is a goal of restoration. However, higher flows within the context of static N and P concentrations equates with increased nutrient loading. Increased freshwater flows, if not of an appropriate quality, may result in more frequent, intense and/or persistent phytoplankton blooms (CROGEE, 2002; Brand, 2002; Jurado et al., 2007). This is of great ecological importance because of the potential negative cascades that can occur due to increased frequency or intensity of undesirable algal blooms (Butler et al., 1995). It follows that a balance exists whereby the benefits afforded by increased flow and improved salinity regime for faunal utilization are not undone by potential adverse effects from increased nutrient loading. This is a true conundrum that must be carefully monitored, changes fully documented, and the resultant implications deciphered as restoration unfolds.

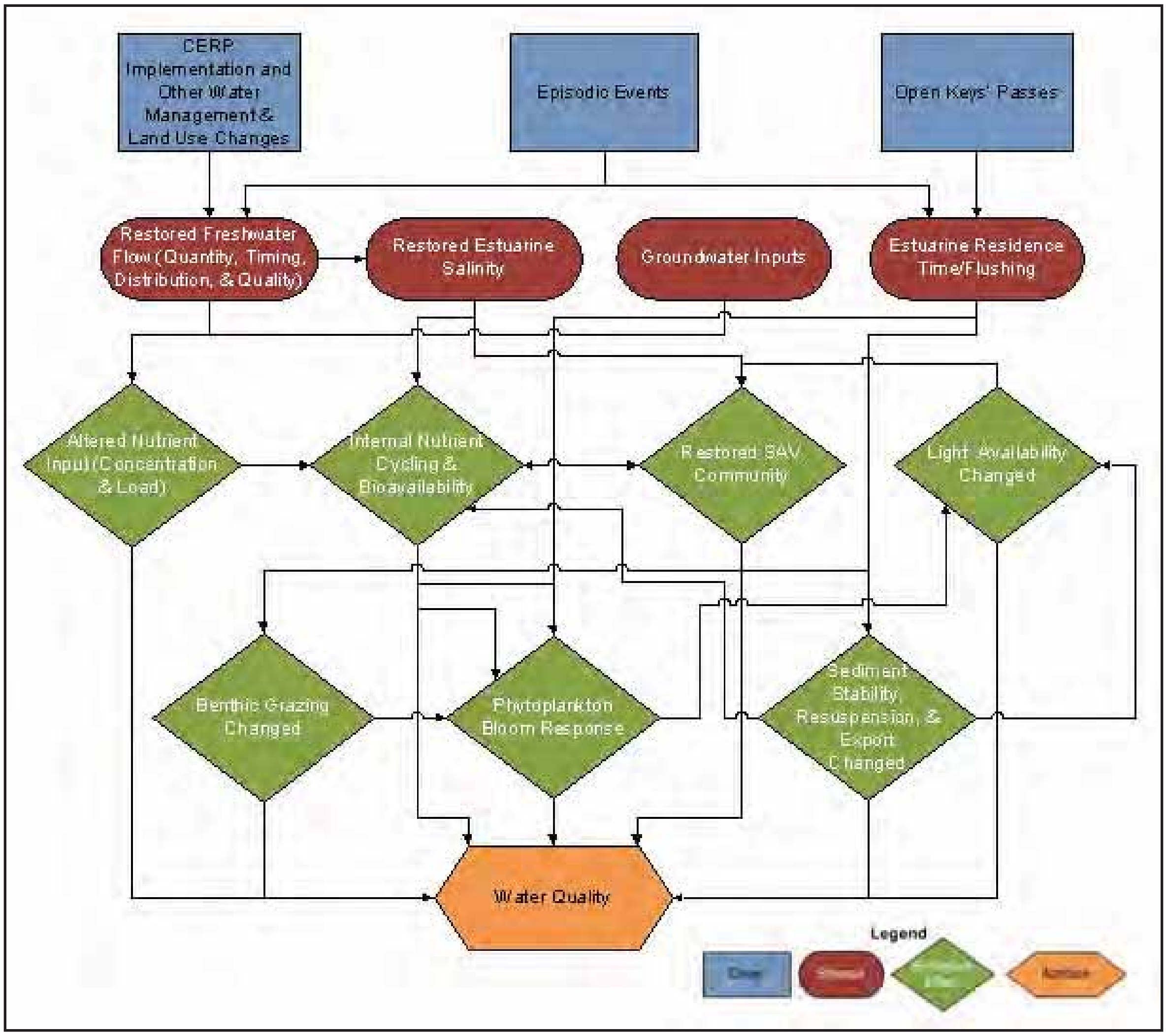


FIGURE 9-44. CONCEPTUAL ECOLOGICAL MODEL FOR WATER QUALITY IN SOUTHERN COASTAL SYSTEMS

RECOVER has developed a performance measure for Southern Coastal Systems water quality and is available on line at www.evergladesplan.org/pm/recover/recover_docs/ret/pm_se_waterquality.pdf.

9.3.2 Monitoring

Systematic monitoring of water quality at fixed stations in the Southern Coastal System has been ongoing since late 1989 as part of Florida International University‟s Southeast Environmental Research Center‟s water quality monitoring network, which has been funded by the SFWMD from 1991-2008. This effort began in Florida Bay and by the mid-1990s had expanded to the entire Southern Coastal Systems, including the mangrove transition zone. Also, beginning in the mid-1990s NOAA‟s Atlantic Oceanographic and Meteorological Laboratory (AOML) began monitoring water quality and circulation throughout the Southern Coastal Systems via fixed station and continuous synoptic sampling. All of the fixed stations except those located on the southwest Florida shelf were sampled monthly by both programs until 2005. In 2008, the Florida International University monitoring program was transferred to the SFWMD for continuation. Funding issues caused the SFWMD to discontinue sampling on the southwest Florida shelf in 2007. The Miami-Dade County Department of Environmental Resource Management (Miami-Dade DERM) has maintained a water quality monitoring effort in Biscayne Bay since the early 1990s. More recently initiated efforts by USGS and FDEP generate water quality data, respectively, at select coastal discharge sites and from four water quality monitoring stations in the vicinity of the Picayune Strand Restoration Project area (***Figure 9-45***).

Typical parameter coverage at each of the fixed stations includes chlorophyll *a*, N and P nutrients, and varying combinations of TSS, turbidity, dissolved and total organic carbon, pH, conductivity or salinity, dissolved oxygen, and/or alkaline phosphatase activity, depending on the agency and the focus of the monitoring program. In addition, as part of their continuous synoptic (i.e. cruise) sampling, AOML measures sea surface temperature, salinity, chlorophyll *a* fluorescence, which can be converted to biomass estimates, beam transmission (λ=660), which can be used to estimate TSS, and CDOM fluorescence. These measures can then be used to estimate light attenuation along the underway track, which is useful for determining if phytoplankton and/or seagrass growth is light-limited within specific regions of the Southern Coastal Systems. Recent peer reviewed publication pertaining to water quality in the Southern Coastal Systems include Boyer et al. (1997, 1999) for Florida Bay and mangrove transition zone water quality distributions and trends, Rudnick et al. (1999) for Florida Bay nutrient loading, Kelble et al. (2005) for Florida Bay light attenuation, Caccia and Boyer (2005) for Biscayne Bay water quality distributions, Kelble et al. (2007) for Florida Bay salinity variability, Jurado et al. (2007) for bloom dynamics on the southwest Florida shelf and Zhang et al. (2009) for the impact of tropical cyclones on Biscayne Bay water quality.

A detailed analysis of chlorophyll *a* dynamics is presented because chlorophyll *a* concentrations are indicative of phytoplankton, and phytoplankton blooms are a major concern to the overall health of the Southern Coastal Systems (Rudnick et al., 2005; Boyer et al., 2009). Other water quality analyses will be restricted to a high-level system-wide overview of nutrient regime. This was accomplished by combining water quality data from the cooperating agencies into one queriable database.

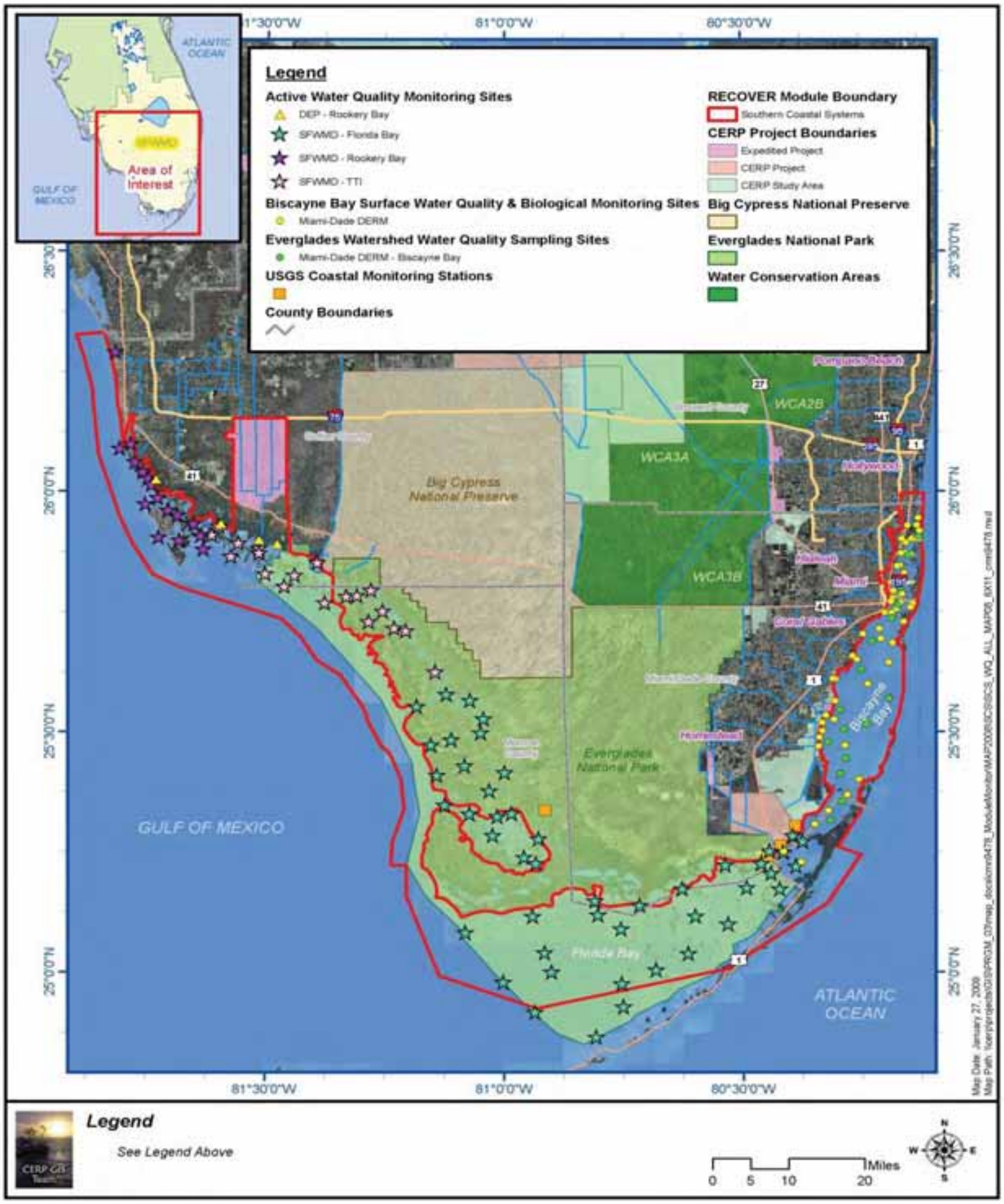


FIGURE 9-45. WATER QUALITY MONITORING STATIONS RELEVANT TO THE SOUTHERN COASTAL SYSTEMS AS SAMPLED BY SOUTH FLORIDA WATER MANAGEMENT DISTRICT, FLORIDA DEPARTMENT OF ENVIRONMENTAL PROTECTION ROOKERY BAY, NATIONAL OCEANIC AND ATMOSPHERIC ADMINISTRATION, MIAMI-DADE COUNTY DEPARTMENT OF ENVIRONMENTAL RESOURCE MANAGEMENT AND US GEOLOCIAL SURVEY

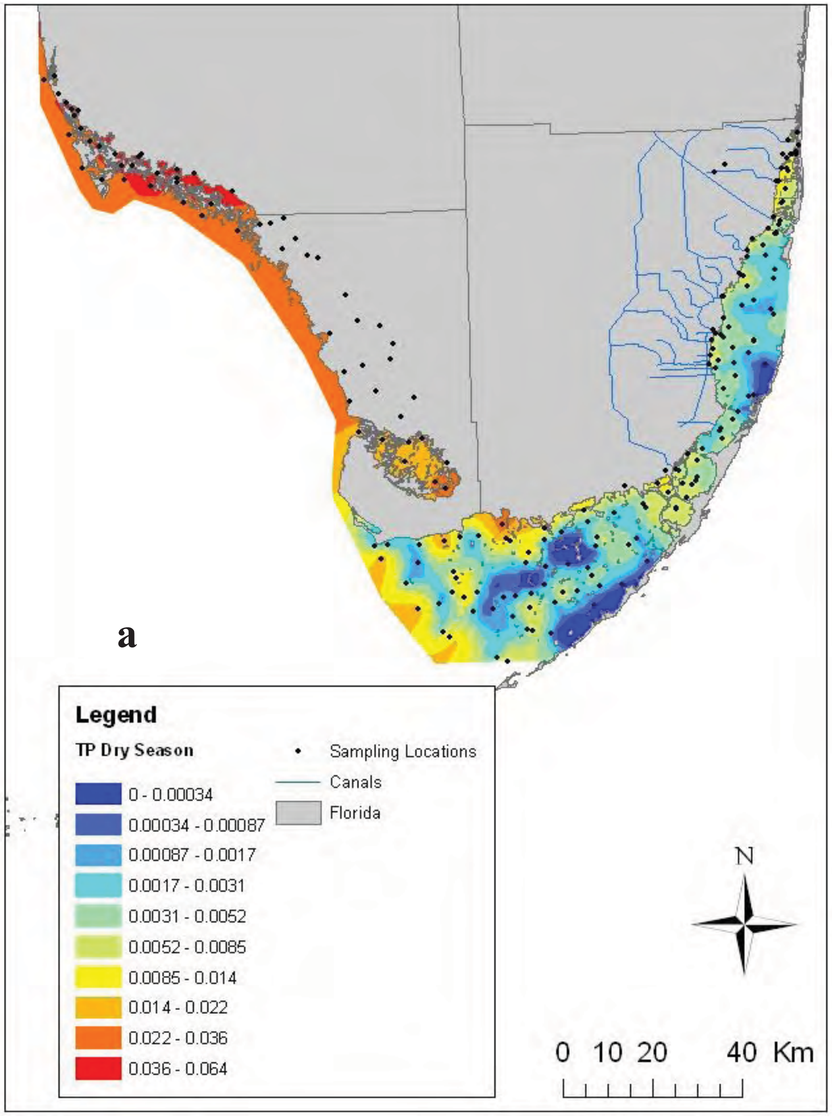
9.3.3 Results

Biscayne Bay and eastern Florida Bay are relatively poor in P (Boyer et al., 1997). This is, at least partly, a function of natural biogeochemical processes (e.g., P retention by the bay‟s carbonate sediments) and thus may have existed prior to recent human influences. In comparing dry versus wet season averages (***Figure 9-46***), there is a slight apparent increase in TP concentration in Biscayne Bay during the wet season. This is presumably due to canal discharge necessary to maintain flood protection during the wet season, transporting TP from the nearly completely developed basin into the bay. Diverting water from point source canal discharge to the bay‟s coastal wetlands, which will sequester more P than do the canals, may reduce these differences.

In eastern and central Florida Bay, seasonal changes in TP concentration are slight. It is important to note that Florida Bay, and particularly eastern Florida Bay, is, to a degree, compartmentalized as a result of numerous banks and shallows, and is more or less hydrologically isolated (***Figure 9-46***Figure 9-46. Average ten-year dry and wet season total phosphorus (mg/L) concentration (1999-2008)). As a consequence, graphical representations of the type employed here require a number of caveats. Conditions at individual sampling points in eastern Florida Bay may be accurate indicators, but areas between sites as depicted in these graphics should be interpreted with some caution.

The two small areas of elevated TP against the northern shore of west central Florida Bay are apparent near Flamingo and Alligator Creek. Alligator Creek is connected to The Lungs, Long Lake and West Lake, where TP concentrations average over 0.08 mg/L and chlorophyll *a* concentrations of 20 µg/L, indicative of substantial algal blooms, are not uncommon (Frankovich et al., 2008). These same areas of elevated TP in Florida Bay were also identified by Boyer et al. (1997), Brand (2002) and others. Western Florida Bay evidences higher TP in the dry season versus the wet, presumably due to influence from the Gulf of Mexico. Average TP concentrations in Faka Union Canal, Little Blackwater River, and Henderson Creek are approximately 0.02, 0.06 and 0.04 mg/L, respectively, based on recent results; however, periods of record differ. Since the red area in ***Figure 9-46*** covers a range of TP concentration starting at 0.036 mg/L, TP from Faka Union Canal discharges do not account for the apparent seasonal difference, which are likely due to sources further north. The Peace and Myakka Rivers have been estimated to deliver 11 million kilograms per year (kg/yr) of P into the Gulf of Mexico (McPherson et al. 2000; pubs.usgs.gov/circ/circ1207/major_findings.htm) which could conceivably be carried south by the Gulf of Mexico loop current.

The C-102 and C-103 canals discharge into Biscayne Bay, carrying an average inorganic nitrogen (IN) load, most of which is in the nitrate form, of over 800,000 kg/yr combined (Graves et al., 2004). The effect that this quantity of nitrate has on Biscayne Bay is clearly evident (***Figure 9-47***), with the zone of influence showing further evidence of expansion in the wet season. The higher N concentration in eastern Florida Bay is in the same general location as previously reported (Brand, 2002) and increases during the wet season. Whitewater Bay and the southwest nearshore also have higher IN during the wet season in comparison to the dry season.



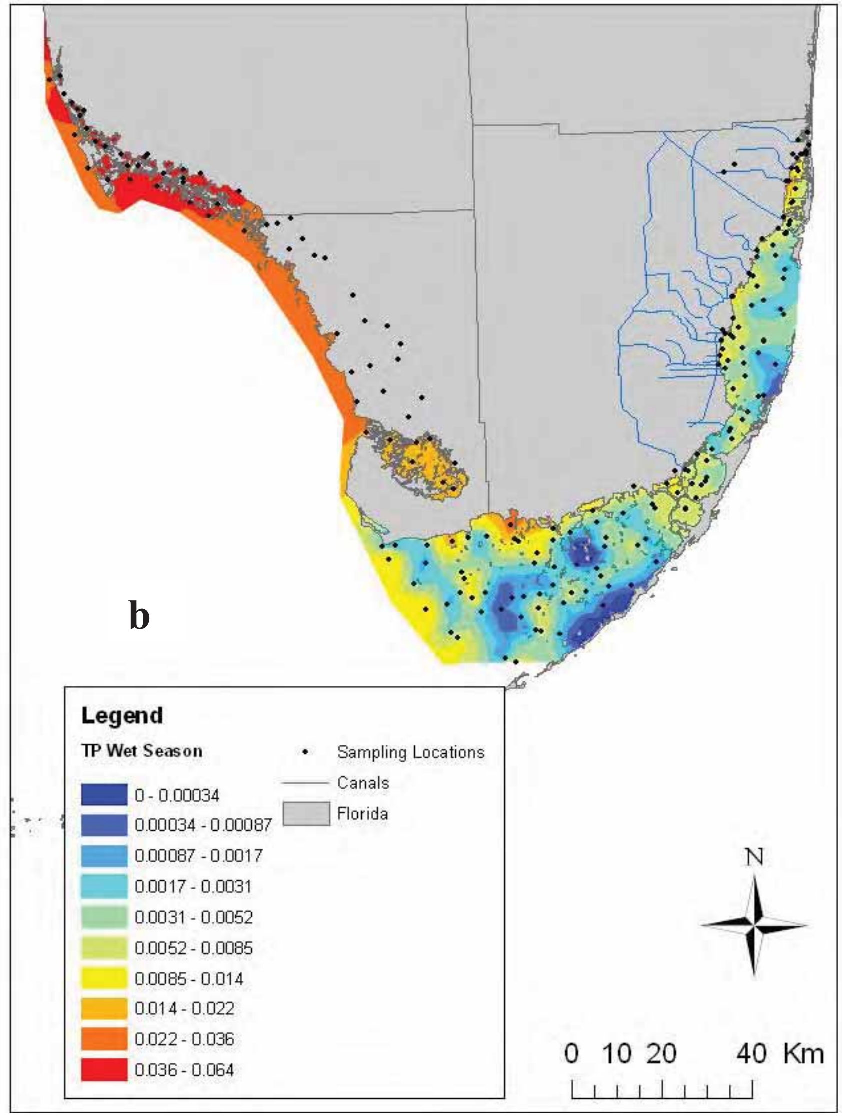


FIGURE 9-46. AVERAGE TEN-YEAR DRY AND WET SEASON TOTAL PHOSPHORUS (MG/L) CONCENTRATION (1999-2008)

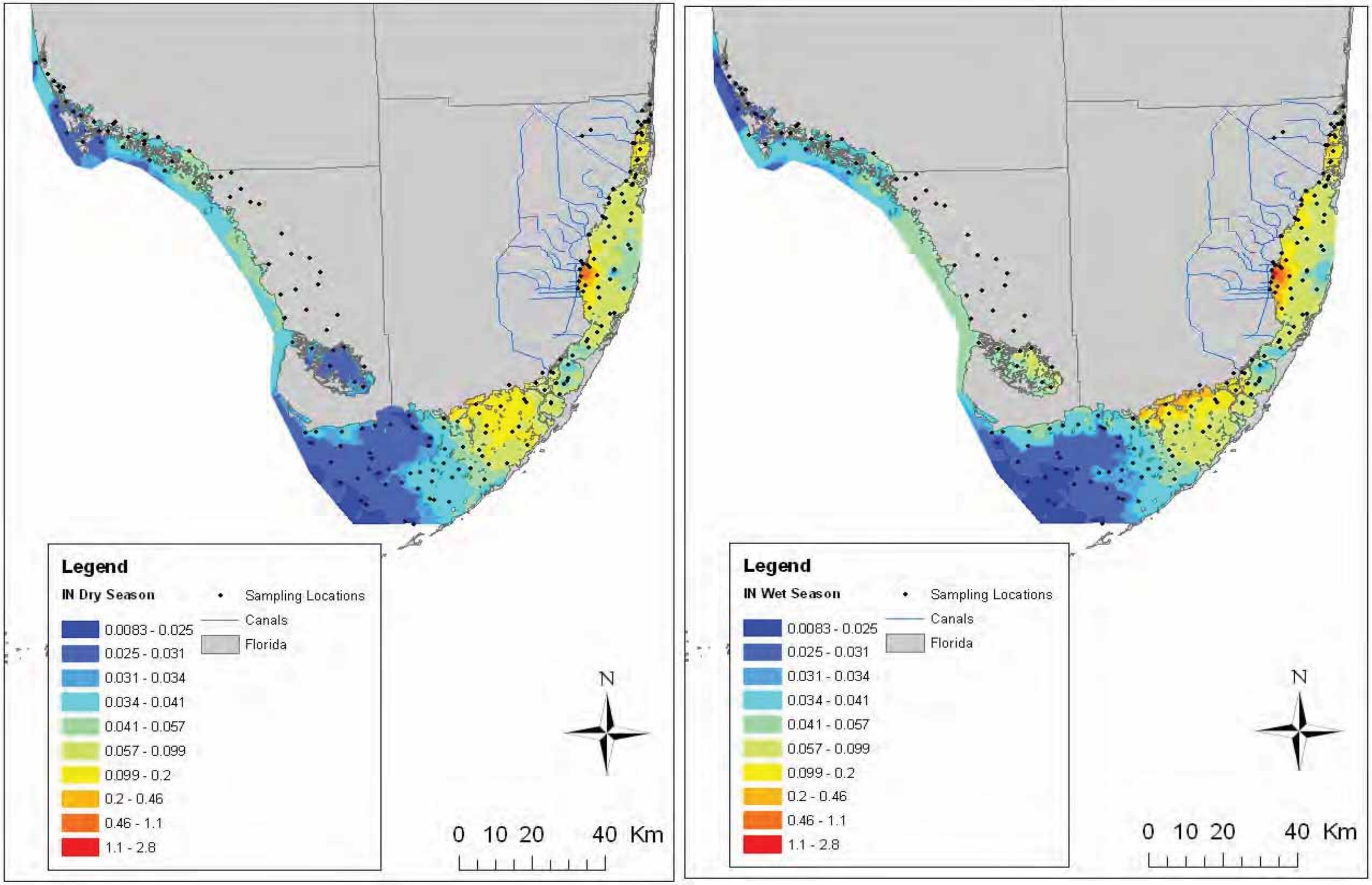


FIGURE 9-47. AVERAGE TEN-YEAR DRY AND WET SEASON INORGANIC NITROGEN (MG/L) CONCENTRATION (1999-2008)

Chlorophyll *a* concentration is an indicator of phytoplankton biomass, and has been chosen by both the U.S. Environmental Protection Agency (USEPA) (2001) and FDEP (Chapter 62-303 of the F.A.C.) for determining nutrient impairment in their regulatory efforts to classify and protect national and Florida water bodies. Based upon the major relevant Southern Coastal Systems restoration objectives and water quality hypotheses, chlorophyll *a* will be similarly employed as the primary measure with which water quality status and trends are assessed (Boyer et al., 2009). What is remarkably atypical in the oligotrophic Southern Coastal Systems waters is that a chlorophyll *a* concentration of 3 µg/L is considered indicative of a noteworthy bloom; a value that might elsewhere be considered a sign of exceptional water quality.

As expected, chlorophyll *a* concentrations are higher in the wet season, during times of higher water temperatures, longer photoperiod, and generally higher nutrient regime than the dry season (***Figure 9-48***). Also expected are the higher chlorophyll *a* concentrations in the wet season along the upper southwest coast where higher TP and IN nutrients occur. Biscayne Bay is P-limited and the small increase in chlorophyll *a* concentration there merely reflects the small increase in TP in the wet season, and not to the aforementioned loading of N from the C102 and C103 canals. The blue and light blue areas south of Taylor River and Trout Creek and extending southward to the Florida Keys denotes a zone where chlorophyll *a* has remained, on average, 1 µg/L or less. This area of very low productivity exists where seasonally higher concentrations of IN are unable to be utilized due to a lack of TP, since there is no real indication of sustained seasonal algal blooms, at least on a decadal timescale. To the east, in Manatee Bay, Little Blackwater Sound, Blackwater Sound and southwestern Barnes Sound, a seasonal chlorophyll *a* signal is apparent, an artifact of the 2005-2008 bloom are discussed in detail in the following sections.

To facilitate the following detailed discussion of the chlorophyll *a* dynamics, the Southern Coastal Systems module was divided into ten subregions based upon statistical methodologies (Boyer et al., 1999; Caccia and Boyer, 2005) and analysis of circulation patterns (Lee et al., 2006; 2007). The distribution of chlorophyll *a* concentrations was not normal in any of these subregions, always being heavily weighted towards lower concentrations, and accordingly non-parametric statistical tests were employed to analyze the data. USEPA guidelines were applied to establish the reference conditions for chlorophyll *a* concentrations and to establish criteria for determining what constitutes elevated levels of chlorophyll *a* (USEPA, 2001; 2008). Data from 1989 through 2005 was employed to establish reference conditions for each subregion (***Table 9-15***). Box-and-whisker plots provide a visual representation of each year in each subregion (***Figure 9-49***). The center of the notch is the median. The extent of the notch is the 95 percent confidence interval of the median. Top and bottom of the box are the 25th and 75th percentiles, respectively, with the whiskers denoting the 10th and 90th percentile. The top of the green zone denotes the 1989-2005 reference median for the subregion. The top of the yellow zone is the reference 75th percentile.

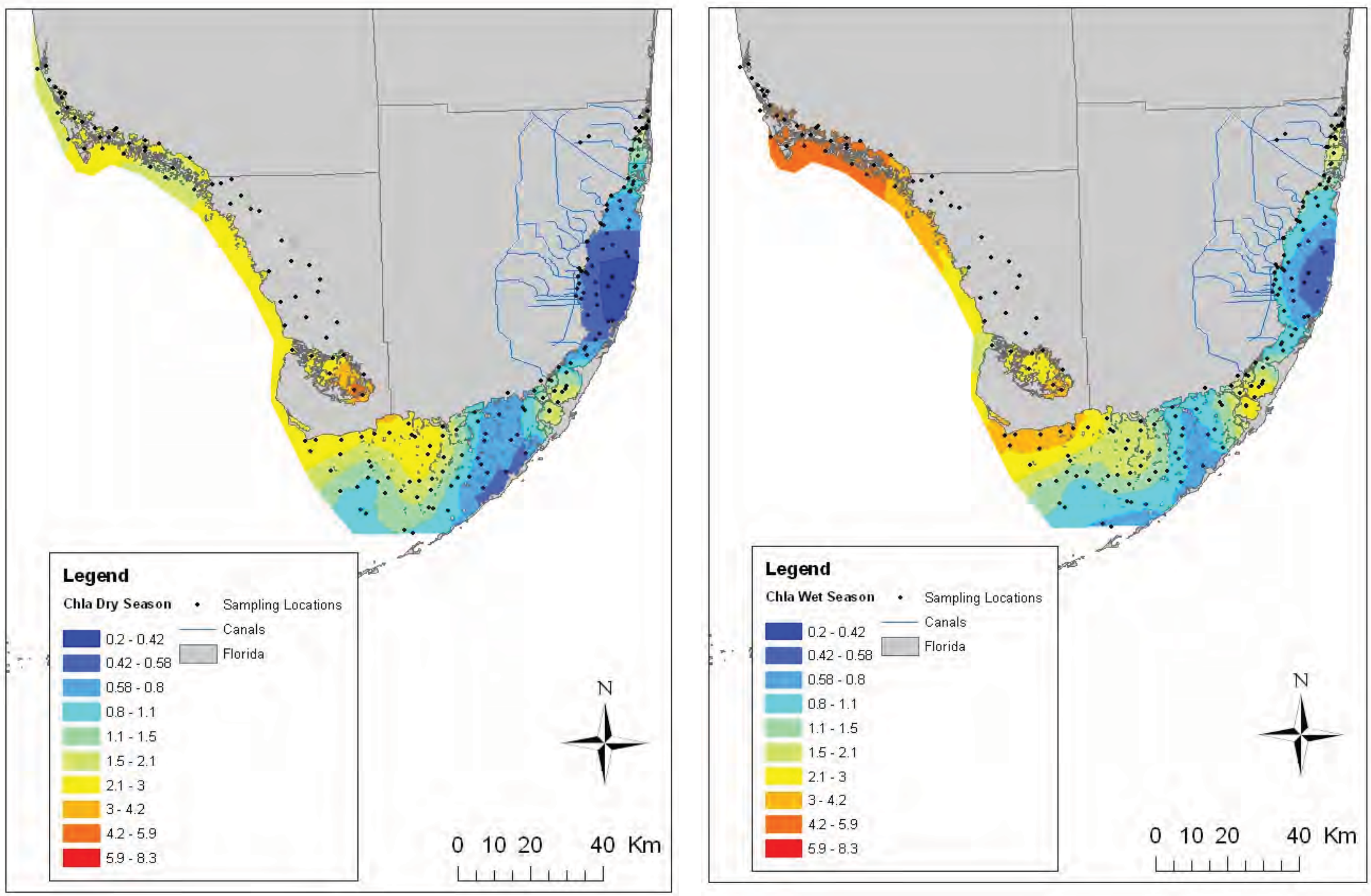


FIGURE 9-48. AVERAGE TEN-YEAR DRY AND WET SEASON CHLOROPHYLL *A* (µG/L) CONCENTRATION (1999-2008)

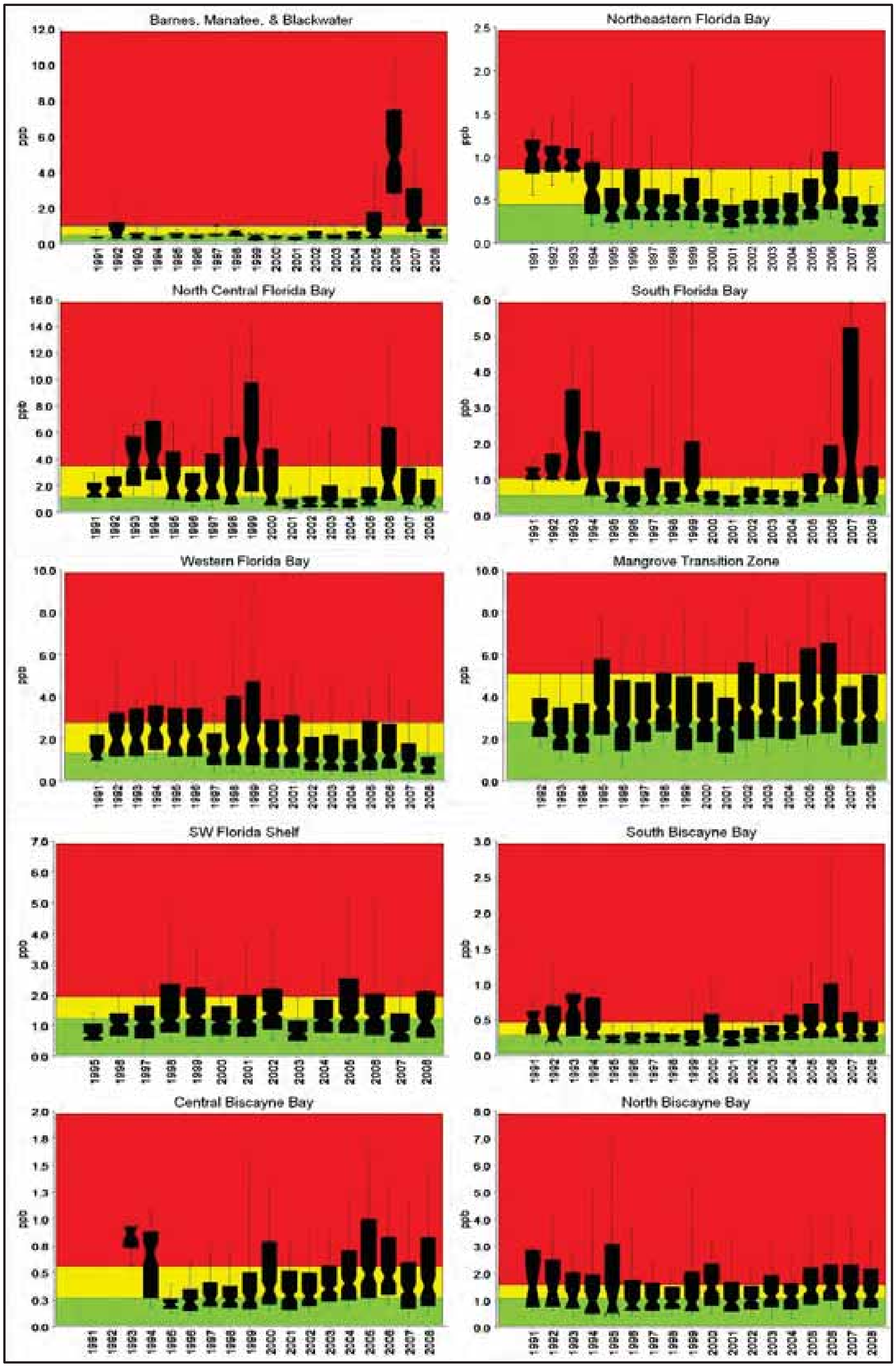


FIGURE 9-49. BOX-AND-WHISKER PLOTS OF CHLOROPHYLL *A* IN EACH SUBREGION FOR EACH YEAR

TABLE 9-15. MEDIAN AND PERCENTILE CHLOROPHYLL *A* CONCENTRATIONS FOR EACH OF THE TEN SUBREGIONS

Subregion	Sample Size	25th Percentile	Median	75th Percentile
Blackwater Sound, Manatee Bay, Barnes Sound	1,704	0.306	0.526	0.910
Central Biscayne Bay	1,673	0.200	0.313	0.566
Mangrove Transition Zone	3,803	1.690	2.863	4.903
North Biscayne Bay	6,35	0.670	1.048	1.648
North-Central Florida Bay	1,399	0.585	1.216	3.710
Northeast Florida Bay	1,979	0.254	0.417	0.790
South Biscayne Bay	2,257	0.181	0.264	0.426
South Florida Bay	1,695	0.327	0.533	1.059
Southwest Florida Shelf	1,297	0.739	1.180	1.976
West Florida Bay	2,304	0.653	1.345	2.845

Note: calculated from the available 1989-2005 data; units in parts per billion

Annual median values obtained each year subsequent to 2005 for each subregion were compared against this reference timeframe to assign relative status (***Figure 9-50***). Green denotes median concentration is below the 1989 to 2005 reference median, yellow is a median concentration greater than the reference median, but less than the 75th percentile, and red denotes the median concentration is greater than the reference 75th percentile. Furthermore, Kruskal-Wallis tests were employed to statistically test for differences in chlorophyll *a* between 2007 and 2008 to all data collected prior to 2006. If any differences were measured, more detailed analyses were undertaken to identify underlying changes in water quality parameters and determine the ultimate cause(s) of the observed change. Since no pre-development data exists with which to perform the calculations the monitoring data from 1989 to 2005 were used as a reference point to capture pre-CERP conditions. A more appropriate indicator of restoration would be a comparison to pre-development chlorophyll *a* concentrations. Thus, this indicator should be viewed as a conservative value because the reference period was from the altered system.

In 2007, five of the ten subregions were rated green, three yellow and two red (***Figure 9-50***). The two red subregions both had the second highest median chlorophyll *a* concentrations of any year on record. The red subregion that incorporates Blackwater Sound, Manatee Bay and Barnes Sound has actually had the intensity of the bloom subside significantly from 2006. In 2006, the entire 95 percent confidence interval of the median was located in the red region of the graph, indicating a substantial increase in chlorophyll *a*. However, in 2007 the lower limit of the 95 percent confidence interval extended into the yellow. South Florida Bay was the other subregion with degraded water quality. Moreover, chlorophyll *a* levels in south Florida Bay were of similar magnitude to the peak of the post-die off cyanobacterial bloom of the early 1990s. The other eight subregions were all well within their respective historic ranges of chlorophyll *a*. In 2008, five of the subregions were yellow and five were green and all were within their historic ranges. The two subregions that had been rated red in 2007 had subsided by 2008.

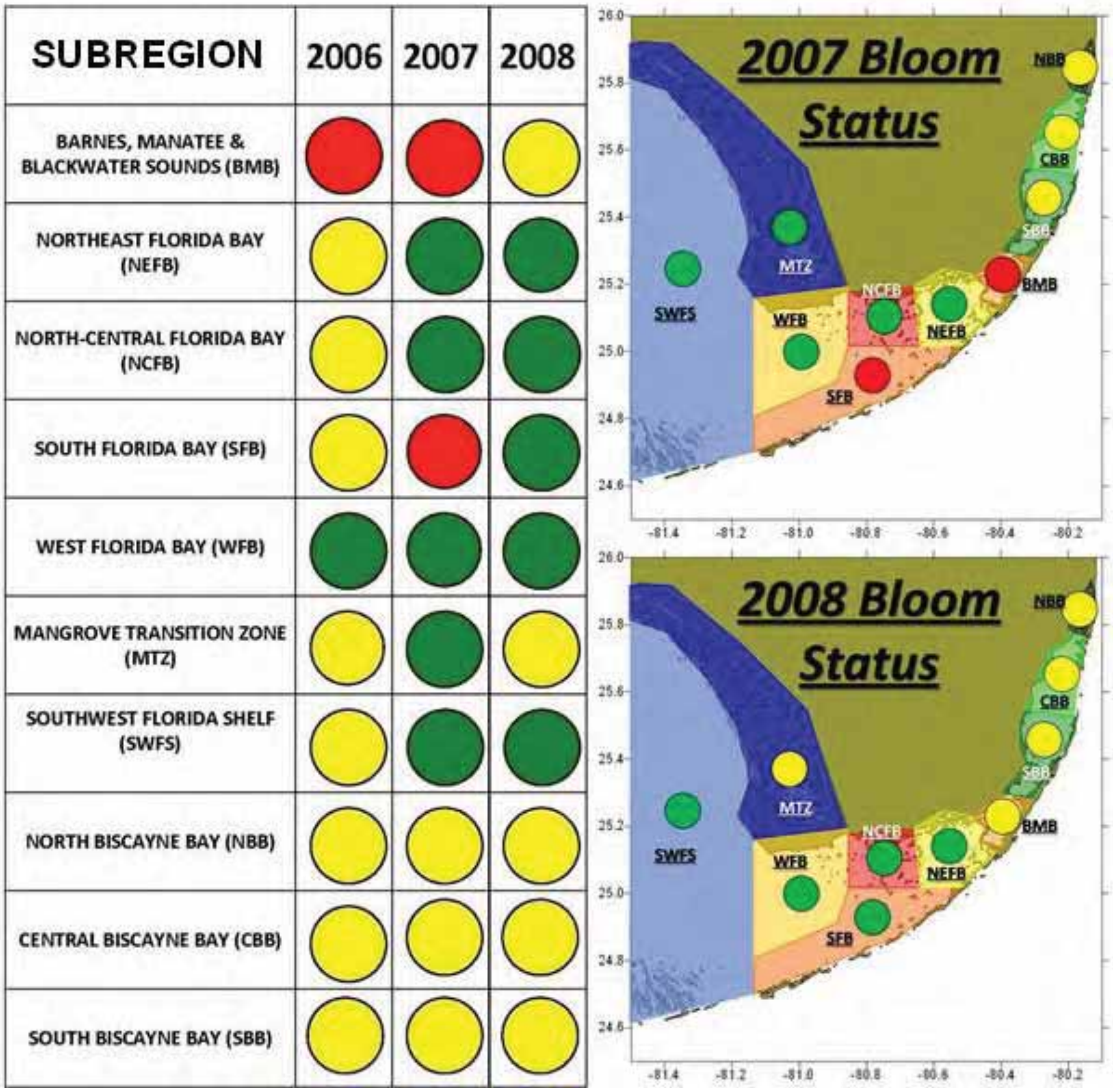


FIGURE 9-50. THE STATUS OF CHLOROPHYLL *A*, THE INDICATOR FOR WATER QUALITY IN THE SOUTHERN COASTAL SYSTEMS FOR 2007 AND 2008

The bloom in Barnes Sound, Manatee Bay and Blackwater Sound began in the fall of 2005 when the area was subject to significant disturbances unrelated to restoration. The 2005 Atlantic hurricane season was the most active in recorded history, with three hurricanes impacting the region, namely Katrina, Rita and Wilma. Earlier that year, in April, construction had begun to expand US Highway 1 which involved a significant amount of cutting and mulching of mangroves and manipulating the soil in the nearshore waters. On August 25 Katrina made landfall near the Miami-Dade/Broward County border. In order to minimize flooding associated with Katrina‟s high rainfall, the terminal S-197 structure of the C-111 Canal was opened the following day, August 26. A water quality sample collected August 27 indicated a concentration of 0.116 mg/L of P, the highest value recorded during the period 1999-2008 (average TP at this site with this sample removed was 0.007 mg/L) - likely the result of physically re-entraining unconsolidated sediments into the water column in this canal that typically discharges infrequently. On September 20, Hurricane Rita skirted the keys, followed on October 24 by Wilma striking the west coast of Florida at Cape Romano, crossing the state just south of Lake Okeechobee. By November, a widespread algal bloom was underway that extended from Barnes Sound to Duck Key in eastern Florida Bay, with chlorophyll a concentrations far exceeding previously measured values for this region. Mass balance calculations indicated that an estimated 19 mt of TP (measured on October 6, 2005) fueled the region-wide bloom, of which 2.6 mt (14%) could have been due to the Katrina C-111 discharge. What cannot be explained by the discharge is that the TP peak occurred more than a month later (after Rita), and that a regional-scale bloom followed that peak. Given the magnitude and distribution of the TP peak, additional sources of TP are required, and likely include some combination of P mobilized by physical disturbance of sediments by hurricane-driven wave action, groundwater influx, and internal cycling between the water column and sediment. The bloom itself was possibly sustained by limited circulation and long residence times, efficient P utilization, and characteristically high ambient inorganic N concentrations from the watershed and other sources. TP remained elevated throughout the bloom (***Figure 9-51***), which suggests that a short-term increase in TP availability, perhaps only marginally, can result in a bloom that can last over a year. For more information on this event see Rudnick et al. (2007).

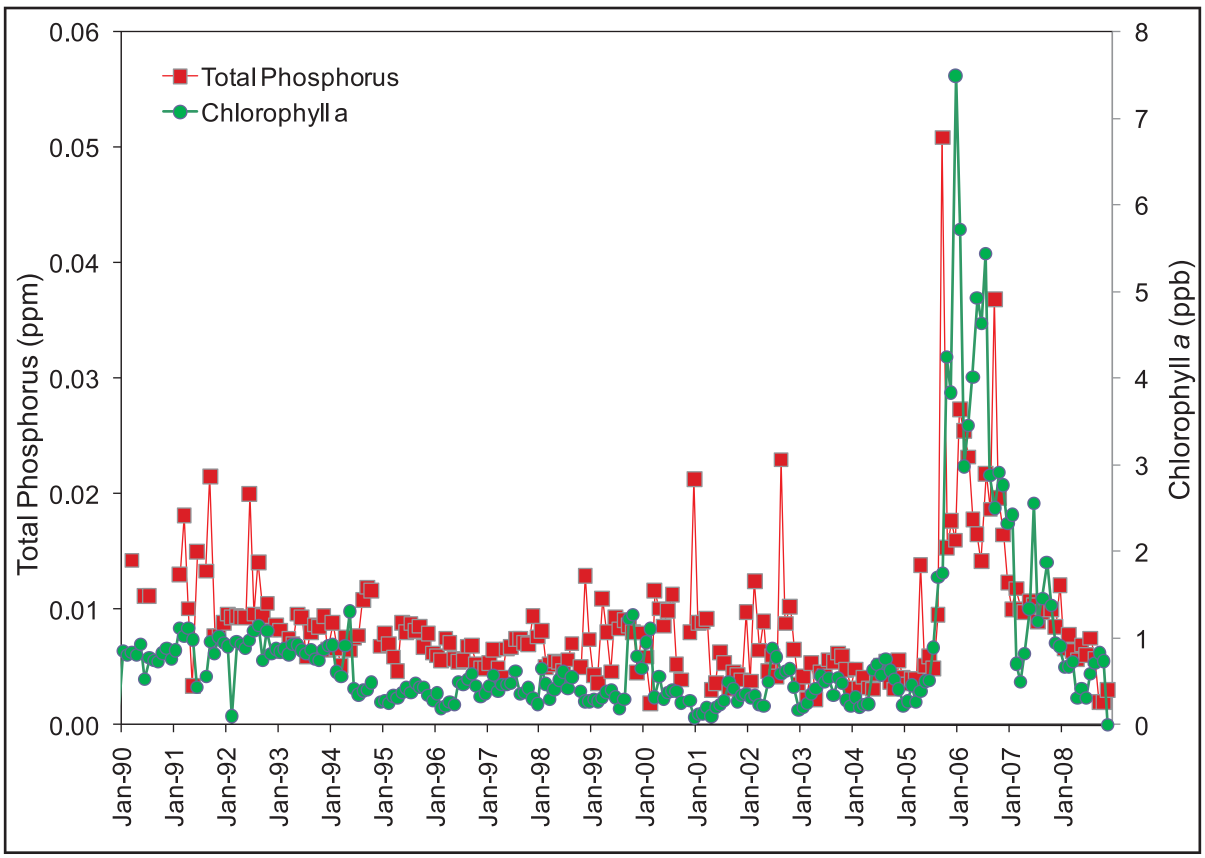


FIGURE 9-51. TIME SERIES OF MEDIAN MONTHLY TP IN PARTS PER MILLION AND CHLOROPHYLL *A* IN PARTS PER BILLION WITHIN THE BLACKWATER SOUND, MANATEE BAY AND BARNES SOUND SUBREGION

Key: ppb parts per billion
ppm parts per million

Elsewhere, in south Florida Bay, phytoplankton blooms are observed sporadically throughout the period of record most notably in 1993 (***Figure 9-52***). From 2000 to mid-2005 blooms were not detected, and chlorophyll *a* concentrations remained less than 1 µg/L. Following passage of the 2005 hurricanes, an intense bloom occurred and chlorophyll *a* spiked to 7.5 µg/L; the highest value observed for the period of record. This subsided and then again rose to around 3 µg/L at the end of the wet season in 2006, and to above 5 µg/L near the end of the wet season in 2007. This late wet season rise observed in 2005, 2006 and 2007 was not repeated in 2008. Although there was a net increase in P concentration during the 2005 to 2007 blooms, even higher P concentrations in 2001 and 2002 resulted in no significant bloom formation.

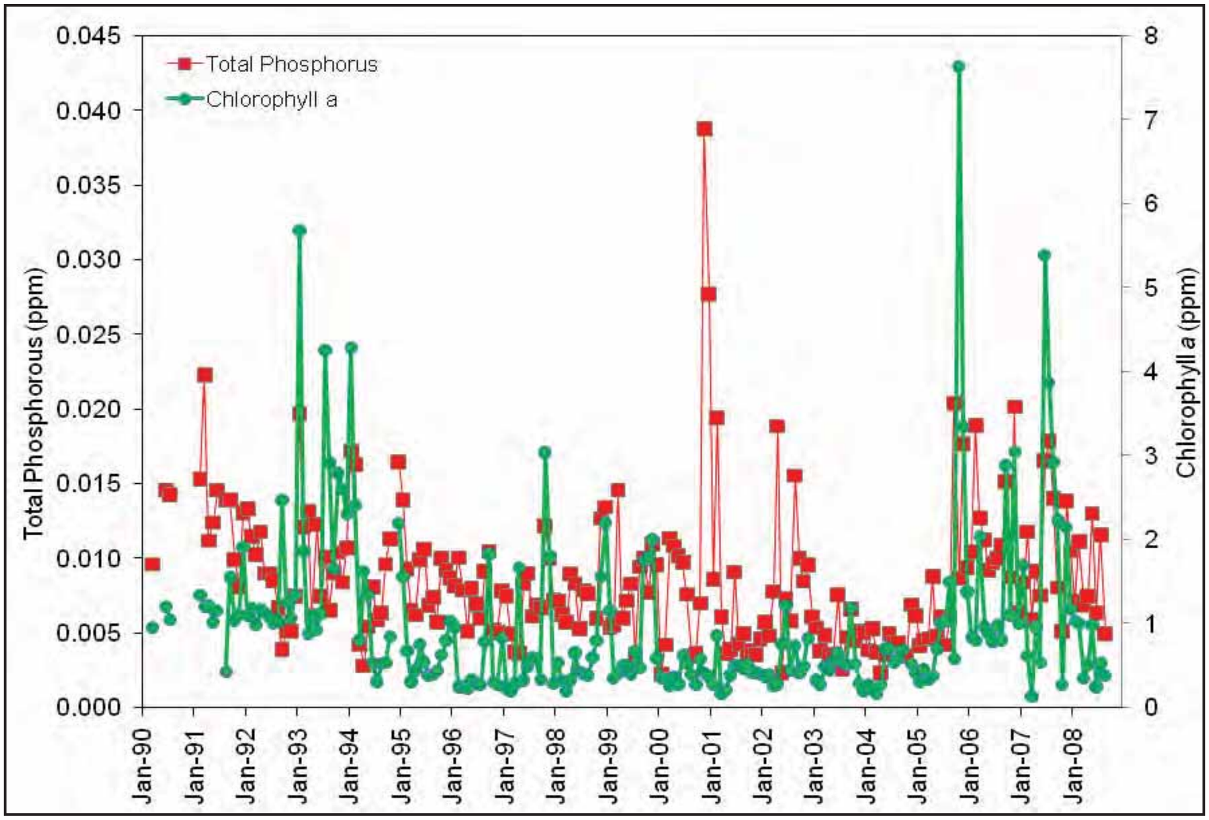


FIGURE 9-52. TIME-SERIES OF CHLOROPHYLL *A* AND TOTAL PHOSPHORUS IN SOUTH FLORIDA BAY

The 2007 bloom in south Florida Bay was particularly noteworthy, as this bloom was eventually transported eastward over the Florida Keys reef tract, and was sufficiently intense to be visible to the naked eye (***Figure 9-53***, courtesy of U.S. Coast Guard, acquired from Jon Fajans, Florida Institute of Oceanography). The unusual bloom and the resultant media alarm prompted the Florida Keys National Marine Sanctuary to assemble a panel of experts to analyze the available data and determine the causes and consequences of the blooms occurring in Florida"s southern estuaries. This was the first bloom during the period of record that peaked in south Florida Bay in the summer; all other blooms observed occurred from November to February. Increased summer phytoplankton growth rates and internal nutrient recycling may have allowed the bloom to maintain a relatively high concentration even as it was advected through the nutrient poor coastal waters of the Florida Keys and out to the reef tract.

Despite the short duration and relatively moderate increase in chlorophyll *a* associated with this bloom, a cascade of detrimental ecological effects ensued. The bloom was largely composed of cyanobacteria that formed mucilaginous masses. These masses clogged sponges, which typically circulate and filter large volumes of water to survive. The ensuing mortality of sponges resulted in a loss of lobster habitat. The commercial catch of spiny lobsters is the second most valuable Florida shell fishery, second only to shrimp. For more information on the potential initiation mechanisms and ecological effects resulting from this bloom please consult Donahue and Diersing (2008), which can be found online at floridakeys.noaa.gov/pdfs/bloom_workshop.pdf.

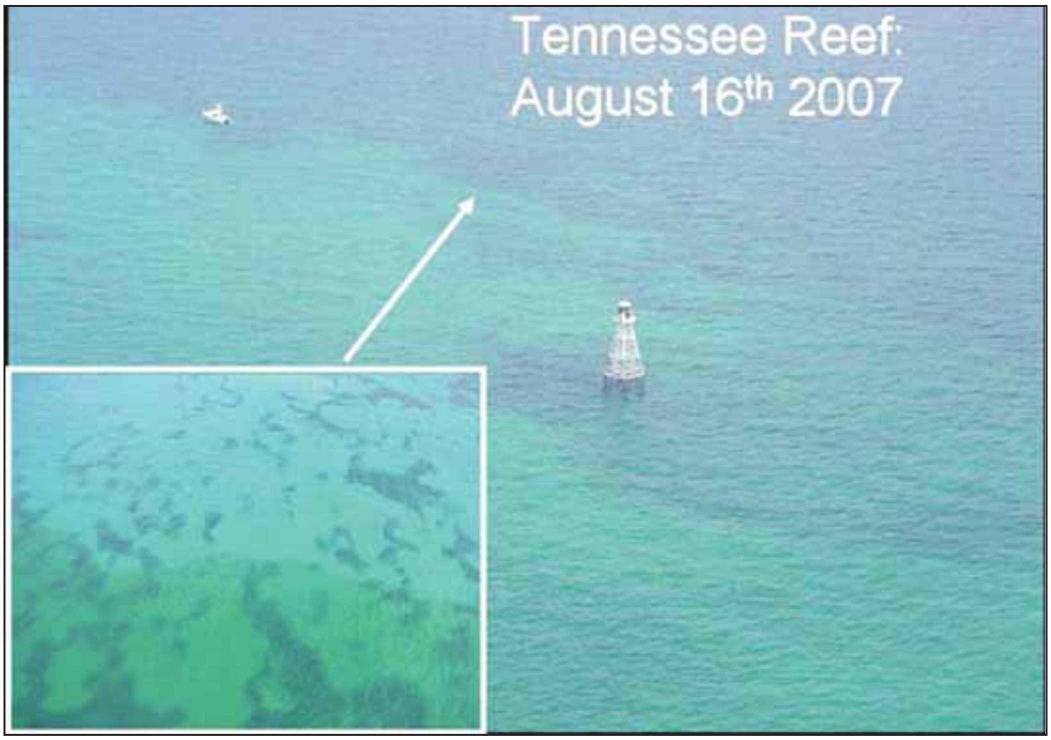


FIGURE 9-53. TENNESSEE REEF AND LIGHTHOUSE, LOCATED EAST OF THE FLORIDA KEYS, ON AUGUST 16TH, 2007 WITH THE INSET SHOWING A CLOSE-UP OF THE ALGAL FRONT MOVING ACROSS THE REEF

The initiation of a phytoplankton bloom occurs in its most basic sense when net primary productivity is positive for a sufficient period of time. Net primary productivity is the difference between respiration and gross primary productivity, such that net primary productivity equals gross primary productivity minus respiration. NOAA"s AOML measured productivity and respiration in southern Twin Keys Basin. The net primary productivity was positive from February to July 2007. In February 2007, the elevated chlorophyll *a* and increased productivity was an extension of the north-central Florida Bay bloom, which died back by May. In July, the chlorophyll *a* levels were again elevated although now it was not an extension of the north-central Florida Bay bloom, but instead centered in south Florida Bay. By July 2007, respiration values had increased and net primary productivity had decreased significantly; however, gross primary productivity and chlorophyll *a* values remained elevated. The peak of the 2005-2008 bloom occurred in July 2007, when the bloom became so widespread as to encompass the entire basin and surrounding waters. Therefore, the bloom was initiated at some point between June and July 2007, although favorable bloom conditions may have occurred earlier as indicated by the extension of the north-central Florida Bay bloom into this basin in February 2007 (***Figure 9-54***).

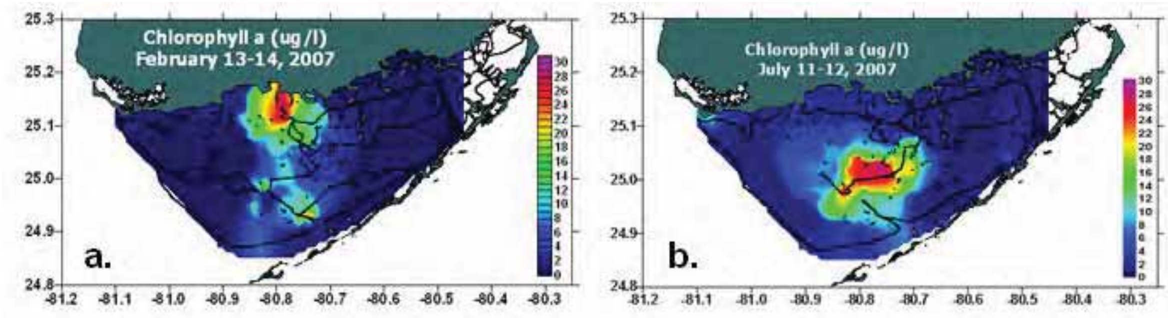


FIGURE 9-54. SPATIAL DISTRIBUTION OF THE ALGAL BLOOM IN FLORIDA BAY, AS MEASURED IN A) FEBRUARY AND B) JULY 2007

The initiation of the bloom is not entirely clear but was likely due to multiple conditions resulting in a favorable environment for increasing growth rates and decreased dilution (Donahue and Diersing, 2008). Circulation, transport and sea level in Florida Bay are significantly altered by large, sustained wind events (i.e. tropical cyclones, cold fronts and high pressure systems). These wind events can cause significant exchange between basins within Florida Bay, as well as between Florida Bay and the adjacent waters on the southwest Florida shelf and the Atlantic coastal zone (Lee et al., 2006; 2008). Sustained wind events result in a net transport of water downwind during the wind event and a net transport in the reverse direction shortly after the wind event has ceased as sea level returns to normal.

A period of consistent east winds from 18 to 27 miles per hour (mph) occurred from May 20 until June 1, 2007 that likely transported water west onto the southwest Florida shelf and decreased water depths within Florida Bay. When winds relaxed on June 1, sea level within Florida Bay returned to normal, and it is probable that some fraction of shelf water was transported into south Florida Bay. Since water on the southwest Florida shelf has higher P concentrations than that typically present in the south Florida Bay, this water exchange may have constituted a significant P influx. Increased wave height due to the sustained strong wind may have resulted in sediment resuspension and increased nutrient availability from the sedimentary pool (Zhang et al., 2004), which added to or helped sustain favorable bloom conditions. This scenario may have been further exacerbated by the southward extension of the north-central algal bloom into the south Florida Bay in February 2007 (***Figure 9-54***).

The recently observed blooms highlight the sensitivity and trophic instability of the Southern Coastal Systems to relatively small environmental perturbations. A relatively small change in the nutrient regime of these coastal systems can result in persistent phytoplankton blooms that may have far-reaching consequences. The temporal and spatial extent of the monitoring programs in place from 1999 through 2008 proved to be of the utmost importance in quantifying the ecological impact of the phytoplankton blooms; however, deciphering causality of these events remains challenging despite the depth of monitoring that was available at the time.

Reductions in the spatial and temporal depth of data will undoubtedly make such interpretive tasks more difficult, and as such, more unlikely to achieve the desired successful outcome of a clear science base upon which to make informed decisions.

A more detailed discussion of Florida and Biscayne Bay water quality can be found in Chapter 12 of the 2010 South Florida Environmental Report (Alleman et al., 2009) at my.sfwmd.gov/pls/portal/docs/PAGE/PG_GRP_SFWMD_SFER/PORTLET_SFER/TAB2236041/2009REPORT/report/v1/chapters/v1_ch12.pdf.

A description of Picayune Strand water quality can be found in the 2009 South Florida Environmental Report in Appendix 7A-2 (Chuirazzi et al., 2009), which can be found on the web at my.sfwmd.gov/pls/portal/docs/PAGE/PG_GRP_SFWMD_SFER/PORTLET_SFER/TAB2236041/2009REPORT/report/v1/appendices/v1_app7A-2.pdf.

9.3.4 Summary

The data and knowledge generated by the monitoring programs in the Southern Coastal System is proving invaluable to identifying patterns and disentangle the causes and consequences of observed water quality phenomenon. It will be necessary to fully resolve reasons why current patterns in water quality are present in order to understand changes in those patterns as restoration progresses. Possessing all of the water quality data in a queriable dataset, which has been largely accomplished in the preparation of this systems status report, should greatly facilitate future inquiries.

How these various efforts will be able to continue to produce a viable dataset upon which to interpret findings and make decisions is of concern. Funding shortfalls are affecting many of the participating agencies. It is important to note that there is a threshold below which data becomes so sparse or infrequent as to be unusable. Unusable data means that whatever costs were associated with generating that data were for all intents and purposes wasted. For example, AOML was only able to perform three sampling surveys along the southwest Florida shelf, far below the minimum recommendation for coastal and estuary water quality monitoring (ACWI and NWQMC 2006; acwi.gov/monitoring/network/design/). AOML is attempting to increase their sampling to bi-monthly, which constitutes a significant improvement, but still falls short of the monthly recommendation.

For this SSR, the current spatial and temporal density of data has proven capable of detecting changes such as the 2006 and 2007 blooms in Blackwater Sound, Manatee Bay and Barnes Sound; the 2007 blooms in south Florida Bay; and the improvements that occurred in 2008. These blooms underscore the highly sensitive and oligotrophic nature of the Southern Coastal Systems, revealing its vulnerability to nutrient loading and stochastic events.

The Quality Assurance Oversight Team (QAOT) has a large and pressing role to play as regards both salinity and water quality data. Methodologies and datasets need to be reliable, and equally importantly, comparable to avoid after-the-fact inability to use data from differing sources in a complimentary way (i.e. combining them to form a more robust backdrop against which to measure and interpret change). The QAOT has been making progress in this regard since 2008, but any issues that may remain need to be identified and addressed sooner rather than later.

9.4 SUBMERGED AQUATIC VEGETATION HYPOTHESIS CLUSTER

9.4.1 Introduction and Background

SAV communities composed of seagrasses and macroalgae are characteristic of shallow coastal waters worldwide; however, few areas contain meadows as extensive as those found in south Florida (Fourqurean et al., 2002). SAV communities provide key ecological services, including organic carbon production, nutrient cycling, sediment stabilization, food sources and habitat structure that enhance local biodiversity (Orth et al., 2006). These plants are not only a highly productive food web base, but they also provide principal habitat for higher trophic levels.

Because SAV live in close proximity to the interface between land and sea, they are subject to physical disturbances and water quality changes associated with anthropogenic changes. Rooted SAV is generally consider to integrate net changes in water quality parameters that tend to exhibit rapid and wide fluctuations when measured directly (e.g., salinity, light availability, nutrient levels). To a large extent, SAV abundance determines public perception of the health of Florida coastal waters (Goerte, 1994; Boesch et al., 1995). For these reasons, seagrasses have been deemed one of the best indicators of change in the Southern Coastal System Module (Fourqurean et al., 2002).

A conceptual model has been developed to depict the factors that influence SAV community structure (***Figure 9-55***). The drivers of the system include water management, land use and episodic events. Restoration would alter the volume, timing and spatial distribution of freshwater inflow into the Southern Coastal Systems. Changes in both salinity and water quality resulting from restoration are expected to result in changes in seagrass and macroalgae cover, biomass, distribution, species composition and diversity through the combined and interrelated effects of light penetration, epiphyte load, nutrient availability, sediment depth, salinity, temperature, hypoxia/anoxia, sulfide toxicity and disease. Significant changes in benthic algae and seagrass distribution can affect susceptibility of sediments to becoming resuspended and the stability of mud banks as well as nutrient availability to other primary producers. Changes related to restoration are expected to include an expansion of areas with *Halodule wrightii* (shoal grass) and *Ruppia maritima* (widgeon grass) cover and a reduction in areas of *Thalassia testudinum* (turtle grass) monoculture along the northern third of Florida Bay. Based on forecasted changes in hydrology, seagrass density and species composition in the southern two-thirds of Florida Bay and the eastern half of Biscayne Bay are not expected to change.

SAV field data and concomitant water quality information are being collected to establish baselines (i.e. reference conditions) against which the extent of system change will be measured once restoration is implemented. The location of SAV monitoring sites is provided in ***Figure 9-56***. This data is being collected through four programs: 1) nearshore benthic habitat monitoring in Biscayne Bay conducted by Lirman, 2) benthic community monitoring in Biscayne Bay conducted by Miami-Dade DERM and SFWMD, 3) SAV monitoring conducted by Miami-Dade DERM in northeast Florida Bay, and 4) the South Florida Fisheries Habitat Assessment Program (FHAP) throughout northern and central Florida Bay.

In-depth analyses of data being collected are important both from the standpoint of utilizing the data to document ecological processes, but also to correct inadequacies or refine sampling design to ensure continued quality. Accordingly, a comparative analysis was performed on the DERM and FHAP monitoring efforts in Florida Bay (Christman and Holt, 2009). The analysis determined that:

- For most years and basins, there has been a significant difference between the two agencies, with DERM reporting higher *Thalassia* cover in most instances. Following the 2008 inter-agency calibration these differences appeared to have been resolved.

- Prior to 2008 methodological improvements, the two agencies did not have completely comparable estimates for the magnitude of either spatial or temporal trends. However, both agencies did detect similar directions of change for *Thalassia* cover in time and among basins.

- The sampling approach is appropriate for detecting change in *Halodule* (a target species in the performance measure), and robust for *Thalassia* since the latter is more common.

- It is an expected outcome of restoration that species richness will increase. However, neither agency observed Chara during the study periods at any overlap location, and Syringonium and Ruppia were observed so rarely that they could not be evaluated. As a result, “diversity” as used in the current RECOVER performance measure may not be a realistic index given only a few species. The performance measure is currently being revised.

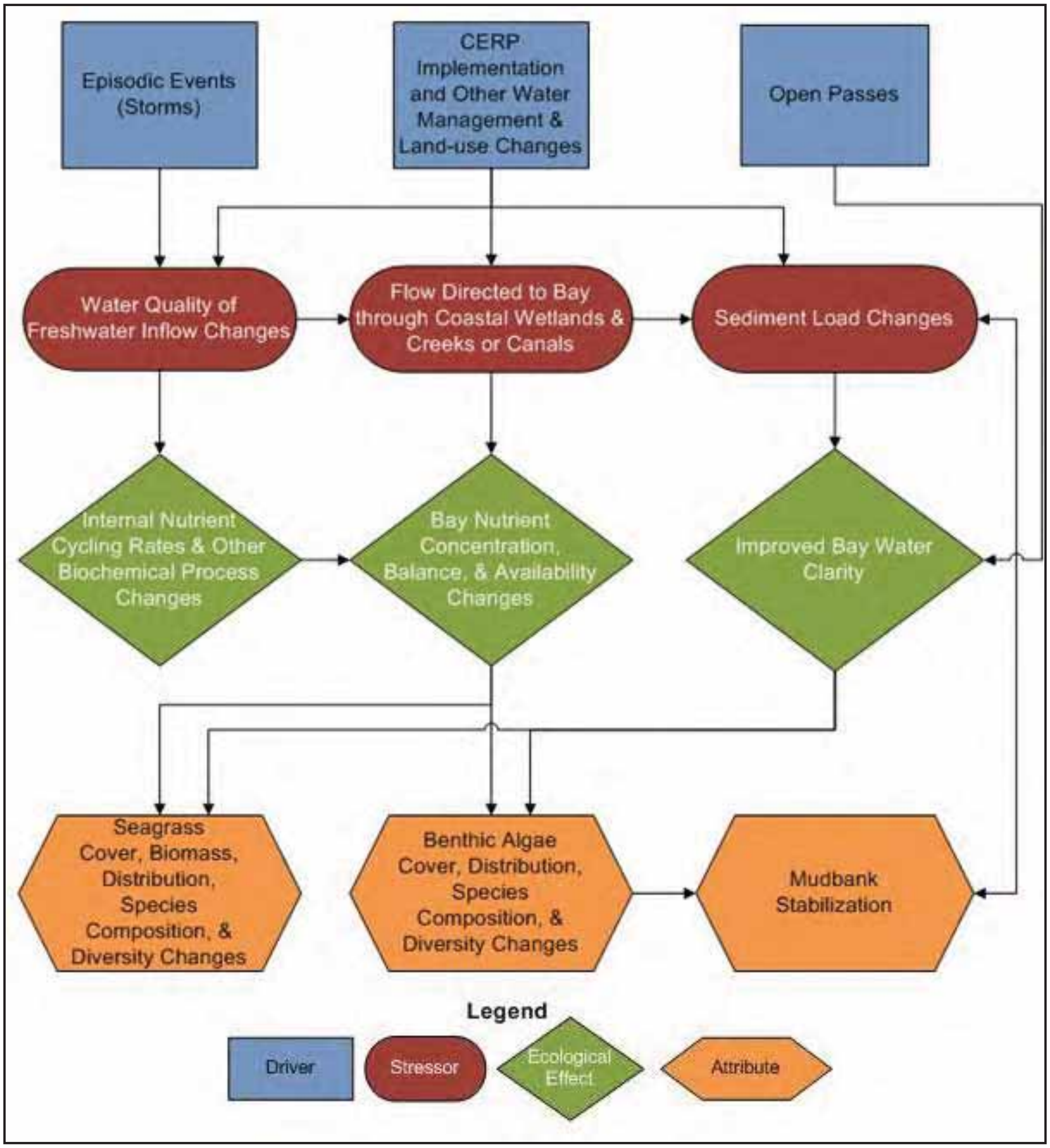


FIGURE 9-55. CONCEPTUAL ECOLOGICAL MODEL OF SUBMERGED AQUATIC VEGETATION IN SOUTHERN COASTAL SYSTEMS

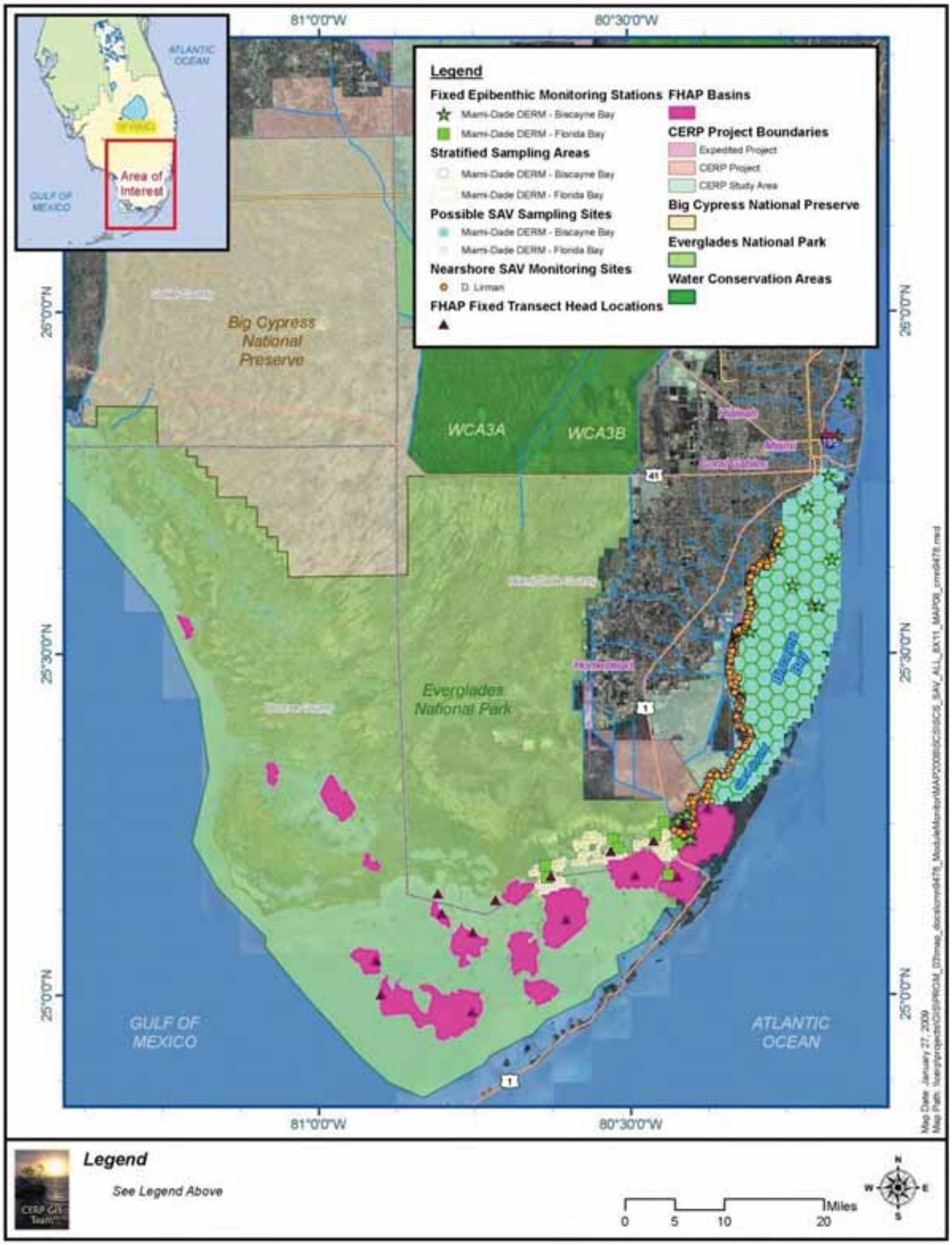


FIGURE 9-56. MONITORING SITES AND PROGRAMS FOR SUBMERGED AQUATIC VEGETATION IN SOUTHERN COASTAL SYSTEMS

9.4.2 Nearshore Benthic Habitats in Biscayne Bay

Since 2003, nearshore benthic habitats of Biscayne Bay have been monitored by the University of Miami and NOAA‟s National Geodetic Survey to evaluate spatial patterns of abundance of SAV in relationship to distance from shore and inflow of fresh water from canals, groundwater, and overland sources (Lirman et al., 2008b; 2008c). One of the most profound changes expected as a result of restoration is the alteration of salinities within western Biscayne Bay. It is unknown what ecological effects this alteration would have on benthic organisms. The shallow areas along the mainland shoreline, which are most sensitive to these changes, are critical nursery habitats for pink shrimp (Diaz, 2001) and economically-valuable fishes (Faunce et al., 2002; Serafy et al., 2003). This project provides a comprehensive, spatially-explicit, baseline database on the seasonal species composition, distribution and abundance of SAV in these susceptible habitats against which ecological responses to changes in water quality and salinity regimes can be measured.

9.4.2.1 Monitoring

In 2008, wet and dry season surveys of nearshore benthic habitats, those less than 500 meters from shore, were conducted between Matheson Hammock and Manatee Bay. These surveys expand previous efforts conducted in 2003 and 2005, which concentrated on the area between Matheson Hammock and Turkey Point (***Table 9-16***). Site selection followed a stratified random sampling design that included distance-to-shore buffers (less than 100, 100-200, 200-300, 300-400 and 400-500 meters from shore) as well as four distinct zones/basins delineated based on hydrodynamic and salinity patterns (Lirman et al., 2008c). The survey domain encompassed 132 km^2, and the minimum distance between sites averaged 300 meters. The indicators of SAV status used include seagrasses and macroalgae percent cover, abundance, frequency of observation and probability of occurrence in relationship to salinity (Lirman et al., 2008b).

TABLE 9-16. NUMBER OF NEARSHORE SUBMERGED AQUATIC VEGETATION SITES SURVEYED IN BISCAYNE

		2003	2005		2008	
Zone	**Location**	**Dry**	**Dry**	**Wet**	**Dry**	**Wet**
Zone 1	Matheson Hammock to North Black Point	66	100	100	124	124
Zone 2	Black Point to Turkey Point	63	140	140	68	68
Zone 3	Turkey Point t to Manatee Bay	-	-	-	145	145
Zone 4	Manatee Bay	-	-	-	34	34
	Total	129	240	240	371	371

9.4.2.2 Results

A synopsis of the monitoring data is provided; a full description of the data collected in 2008 is provided in Lirman et al. (2008a).

9.4.2.2.1 Percent Cover

Two seagrass species, *H. wrightii* and *Syringodium filiforme* (manatee grass), showed a stable pattern of benthic cover since 2003, while *T. testudinum* has shown a steady decline, from greater than 40 percent in 2003 to less than 20 percent in 2008 (***Figure 9-57***). Macroalgal cover remained stable between 2003 and 2008, except for a large increase in the 2005 wet season associated with the bloom of species with a large affinity for fresh water such as *Chara* and *Batophora*. It is unclear whether the declining trend in *T. testudinum* cover is just temporary or a more persistent pattern. However, considering that previous research in Florida Bay suggests that dense *T. testudinum* beds may be more susceptible to mortality due to reduced oxygen conditions, it is crucial that monitoring of these habitats be continued over multiple years to separate natural variability from intentional improvements resulting from restoration induced changes.

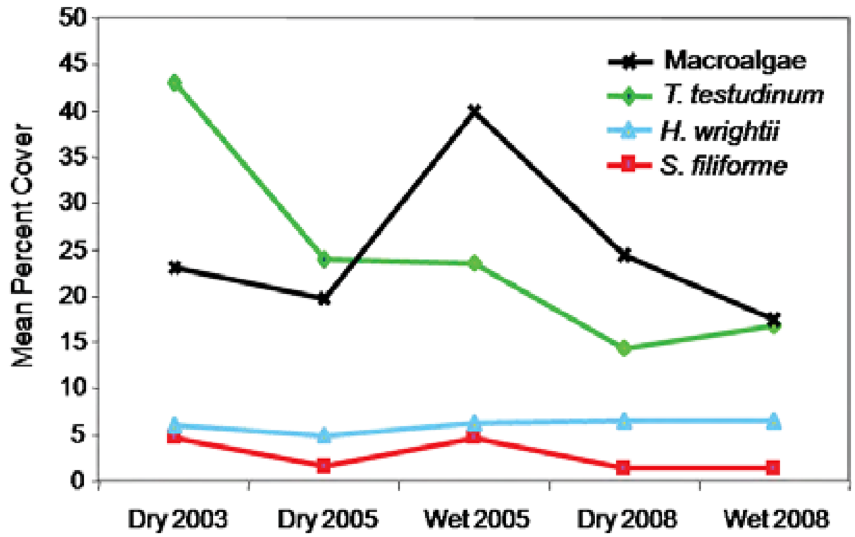


FIGURE 9-57. PERCENT COVER OF SUBMERGED AQUATIC VEGETATION IN NEARSHORE HABITATS OF BISCAYNE BAY 2003 - 2008

9.4.2.2.2 Abundance

The percent cover of *T. testudinum* was similar throughout the study region. *S. filiforme* was only found in zones 1 and 3, reaching its highest cover in zone 1 (Matheson Hammock to north of Black Point). *H. wrightii* was found in all zones, reaching its highest cover in zone 2, the area with the largest inflow of fresh water from canals (i.e., south of Black Point to Convoy Point). Drift macroalgae were abundant throughout the study region, reaching highest percent cover in zones 1 and 3, the zones with the highest mean salinity values. Attached macroalgae were most abundant in zone 2, the area with the lowest salinity. This pattern is influenced greatly by the high cover of *Batophora*, *Acetabularia*, *Chara*, and other macroalgal species commonly associated with more estuarine-like conditions in the Black Point area.

The abundance of *T. testudinum* increased linearly with increasing distance from shore and depth, while the abundance of *H. wrightii* showed the opposite pattern. The abundance of

S. filiforme was highest within buffer 4 (300-400 meters from shore), the buffer with the deepest mean depth in the zone where this species is found. Finally, "dead zones", zones devoid of any macrophyte tissue, are extremely limited in western Biscayne Bay. In fact, no sites completely devoid of SAV were documented in 2008.

9.4.2.2.3 Frequency of Observation

The proportion of images without any SAV decreased from the dry season to the wet season as productivity increased. For seagrasses, the proportion of plots with dense cover greater than 70 percent increased as seagrass growth peaked in the warmest months of the year. While the proportion of sparse seagrass plots increased between 2005 and 2008, the proportion of the densest patches, which had greater than 70 percent cover, declined greatly in this period, suggesting that the decline in cover was especially pronounced within the densest *T. testudinum* beds.

9.4.2.2.4 Probability of Occurrence in Relation to Salinity

The probability of occurrence of SAV in relationship to salinity was tested using logistic regression where seagrass species were coded within sites as either present or absent. All three species showed significant relationships with mean salinity (logistic regression, p less than 0.05 for the three seagrass species). *H. wrightii* has a higher probability of occurrence at low mean salinity, while *T. testudinum* and *S. filiforme* have a higher probability of occurrence at high mean salinity (***Figure 9-58***).

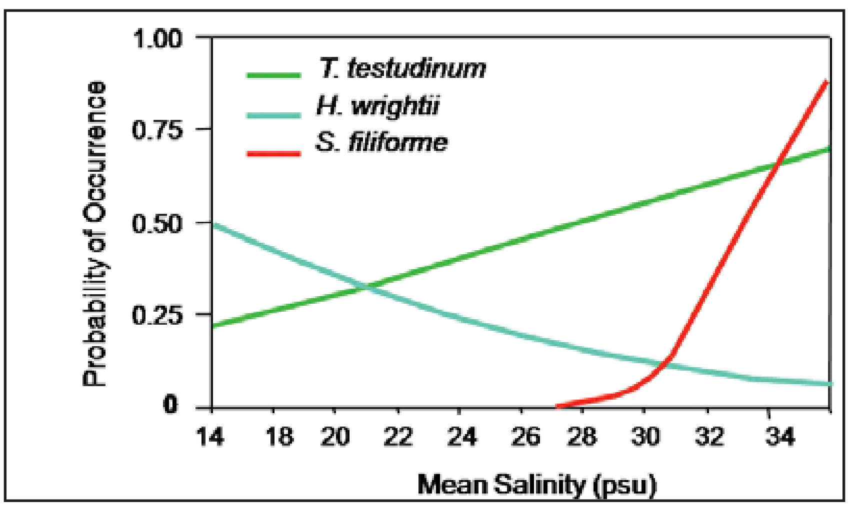


FIGURE 9-58. PROBABILITY OF OCCURRENCE OF SEAGRASSES IN RELATION TO MEAN SALINITY DURING THE WET SEASON FITTED WITH LOGISTIC REGRESSION

9.4.2.3 Summary

The SAV data collected in nearshore western Biscayne Bay since 2003 show a significant relationship between salinity patterns (i.e. mean value, variability) and the seasonal abundance and spatial distribution of SAV. More importantly, these observations support the use of SAV as appropriate indicators of changes in salinity patterns. Nevertheless, it should be noted that the patterns reported were obtained for only three survey years (2003, 2005, 2008), and that an extended effort is needed to fully document interannual variability in SAV abundance and distribution to discern the impacts of the restoration projects and evaluate restoration performance.

9.4.3 Seagrass Communities of Biscayne Bay – Miami-Dade County Department of Environmental Resource Management and South Florida Water Management District

Miami-Dade DERM, in partnership with SFWMD, has conducted a benthic habitat monitoring program in Biscayne Bay since 1985. The monitoring program was initiated with 13 fixed locations throughout the bay, ten of which remain active. The program later expanded to include a rapid survey method that increased the spatial extent of the data collected to all of southern Biscayne Bay. This program"s dataset provides a unique long-term history of the status of SAV in Biscayne Bay.

9.4.3.1 Monitoring

The study area encompasses central to south Biscayne Bay and Card Sound (***Figure 9-59***). Detailed sampling has been conducted at fixed strategic locations on a quarterly basis through 1996, at which time it was decreased to annually with sampling taking place in June. This provides site specific trend-oriented data. In addition, a stratified random sampling has been conducted annually since 1999, providing spatial, status and trend data of the SAV communities (Fourqueran et al., 2002).

9.4.3.2 Results

9.4.3.2.1 North Biscayne Bay–Fixed Stations

S. filiforme is the dominant seagrass in north Biscayne Bay, with *T. testudinum* present in low densities. *H. wrightii* had varying densities. Two stations, at Haulover Inlet (Station 6) and 79th Street Causeway (Station 10) (***Figure 9-59***), experienced a complete loss of seagrass between 1997 and 1999. The SAV at Station 6 was likely affected by nearby periodic maintenance dredging. However, no specific causative factors have been identified for the seagrass losses recorded at Station 10. Both stations currently appear to be undergoing a recovery, with *S. filiforme* and *H. wrightii* returning to Station 6 in 2005 and to Station 10 in 2007.

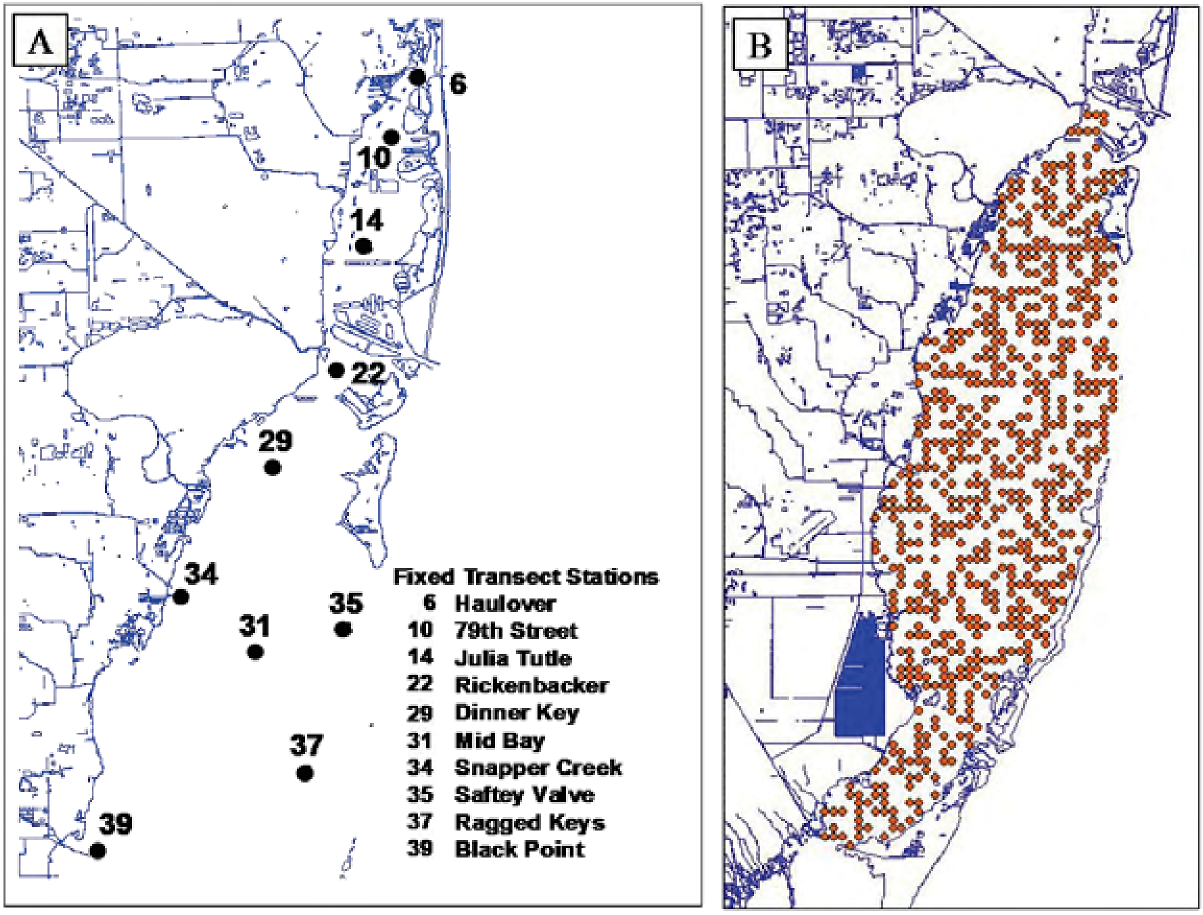


FIGURE 9-59. LOCATION OF THE A) FIXED BENTHIC STATIONS AND B) RANDOM STATIONS

9.4.3.2.2 Central-South Biscayne Bay - Fixed Stations

The central-eastern region of south Biscayne Bay shows a long-term (1985-2008) consistency in monospecific high density *T. testudinum* populations. The transect located east of the mouth of Snapper Creek has shown recent declines in *T. testudinum.* However, data from the polygon containing this transect do not reflect a similar decrease*,* suggesting that this change is not occurring on a larger spatial scale in this region. Two stations located in the northern portion of the study area, Rickenbacker (Station 22) and Dinner Key (Station 29) (***Figure 9-59***), show shifts to *H. wrightii* dominance for a period in the 1990s, and then to *S. filiforme* in the late 1990s to present (***Figure 9-60***). Changes in salinity may have played a role in this shift. At both stations prior to this shift, salinity ranged from 34 to 36 psu. Following this shift, most annual means have been between 32 and 34 psu. The change in salinity regime would likely favor *S. filiforme*, which has a lower optimum salinity than *T. testudinum* (Phillips, 1960). The station at the Black Point channel, dominated historically by *H. wrightii* and *S. filiforme*, has shown a recent shift to *T. testudinum* dominance (***Figure 9-60***) but inferences regarding the relationship between this shift and salinity are not possible due to an absence of appropriate historical salinity data.

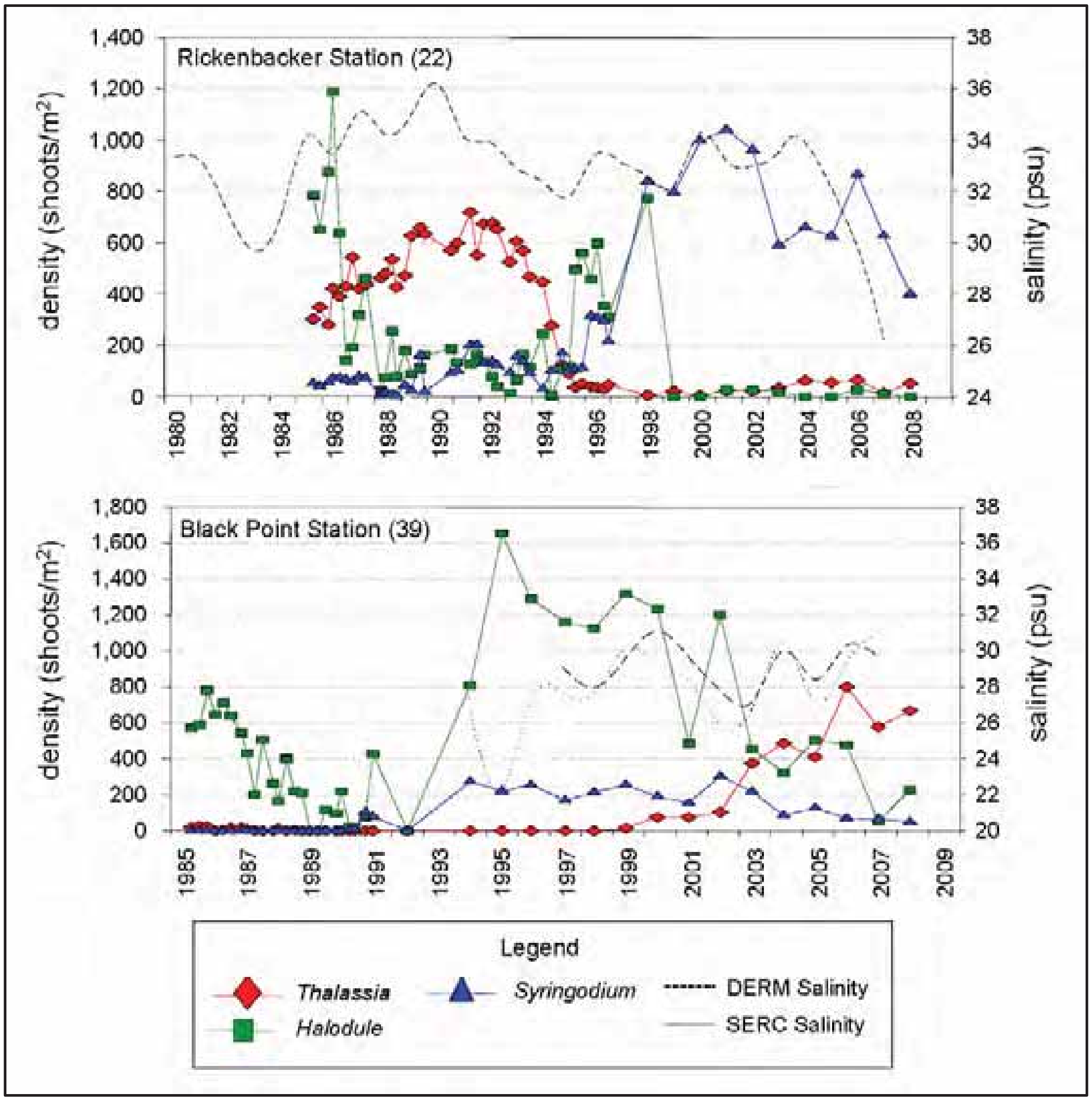


FIGURE 9-60. SHOOT DENSITY OF SEAGRASSES AND SALINITY PATTERNS FROM FIXED MONITORING STATIONS IN BISCAYNE BAY

9.4.3.2.3 Central-South Biscayne Bay and Card Sound - Random Stations

T. testudinum is the dominant seagrass in central-south Biscayne Bay and Card Sound, with score ranges from 1 to 5 (estimated cover of 5 to 100 percent). The highest *T. testudinum* cover is in the northwestern and central-eastern portions of this part of the bay. *H. wrightii* is common though not as abundant within this region, with highest coverage occurring in the northern, western, and extreme southern sections. Scores for *H. wrightii* were commonly 1 or less (cover less than five percent), with a maximum score of 3 (estimated cover of 25 to 50 percent) in one polygon in the extreme south of the area. *S. filiforme* has the most restricted distribution in this area, being found most commonly in the northern and southernmost portions of this region, with a range of scores and cover equivalent to that of *H. wrightii*. *R. maritima* was not identified within the study region.

9.4.3.3 Summary

Patterns of *T. testudinum* cover in south Biscayne Bay follow relationships with salinity regimes, water depth and sediment depth. Shallow depths and consistent tidal exchange characterize the eastern shoal complex known as the Safety Valve, which has the largest expanse of high cover (greater than 75 percent) and low variability of *T. testudinum*. Low percent cover (less than five percent) is common in the hard bottom habitats of the southern mid-bay. Additional areas of lower cover also occur in habitats with greater depth (northern and southern) and/or lower salinity (northern and western). Generally, these conditions favor *H. wrightii* and *S. filiforme* and their distributions reflect this. *H. wrightii* is second to *T. testudinum* in overall presence. Reviews of three-year groupings of data show *H. wrightii* consistently in the northern and southern regions, with low (less than five percent) to moderate cover (25 percent). It also occurs sporadically along the western shore, and in the hard bottom areas at low cover (less than five percent). *S. filiforme* is primarily located in the northern and southern sections, with infrequent records in western nearshore and eastern polygons.

9.4.4 Northeastern Florida Bay Submerged Aquatic Vegetation Monitoring by Miami-Dade County Department of Environmental Resource Management

The SAV monitoring program conducted by Miami-Dade DERM in northeastern Florida Bay began in October 1993 and was expanded east of U.S. Highway 1 in 1997 to include the two southernmost basins of Biscayne Bay, which are Manatee Bay and Barnes Sound. The primary objectives of this program are to identify spatial and temporal trends in SAV patterns, to evaluate differences within and between the study basins, and to identify trends in vegetation patterns across the basins.

Water availability and delivery to Taylor Slough and the C-111 Basin and the effects on northeastern Florida Bay are the focus of regional restoration efforts and water management initiatives. Restoration efforts and water management initiatives, such as Minimum Flows and Levels (MFL) for Florida Bay, have identified the SAV community of this area as a key measure gauging the success of these programs (Hunt et al., 2006).

9.4.4.1 Monitoring

The sampling domain is divided into two regions. The Northern Transition Zone includes Highway Creek, Long Sound, Joe Bay, Alligator Bay, Davis Cove, Trout Cove, Little Madeira Bay and Eagle Key Basin, an area south of Little Madeira Bay. The Eastern Zone includes Manatee Bay, Barnes Sound, Little Blackwater Sound and Blackwater Sound. These basins were selected to reflect effects of point and non-point water sources, watershed runoff, and managed water releases into Taylor Slough.

The evaluation of basin and region-wide vegetation patterns and trends is accomplished through random/rapid sampling methodology based on methodologies used in the USEPA‟s Environmental Monitoring and Assessment Program (EMAP) and the FWC‟s FHAP (***Figure 9-61***). Specific changes in SAV biomass and communities are documented based on intensive sampling at fixed sites that are intended to examine local temporal trends in cover and biomass (***Figure 9-62***). Cover of SAV species and short shoot and blade counts are recorded at one site within each subbasin shown in ***Figure 9-61*** and in May and November at fixed stations shown in ***Figure 9-62***. Estimates of shoot density, leaf area, total standing crop, and below-ground biomass are also determined at the seasonal sites (***Figure 9-62***). Measurements of water quality parameters are also made at each siteincluding dissolved oxygen, salinity, conductivity, temperature, pH and PAR. For a more detailed description of the methodology used, please refer to Avila and Blair (2006, 2008).

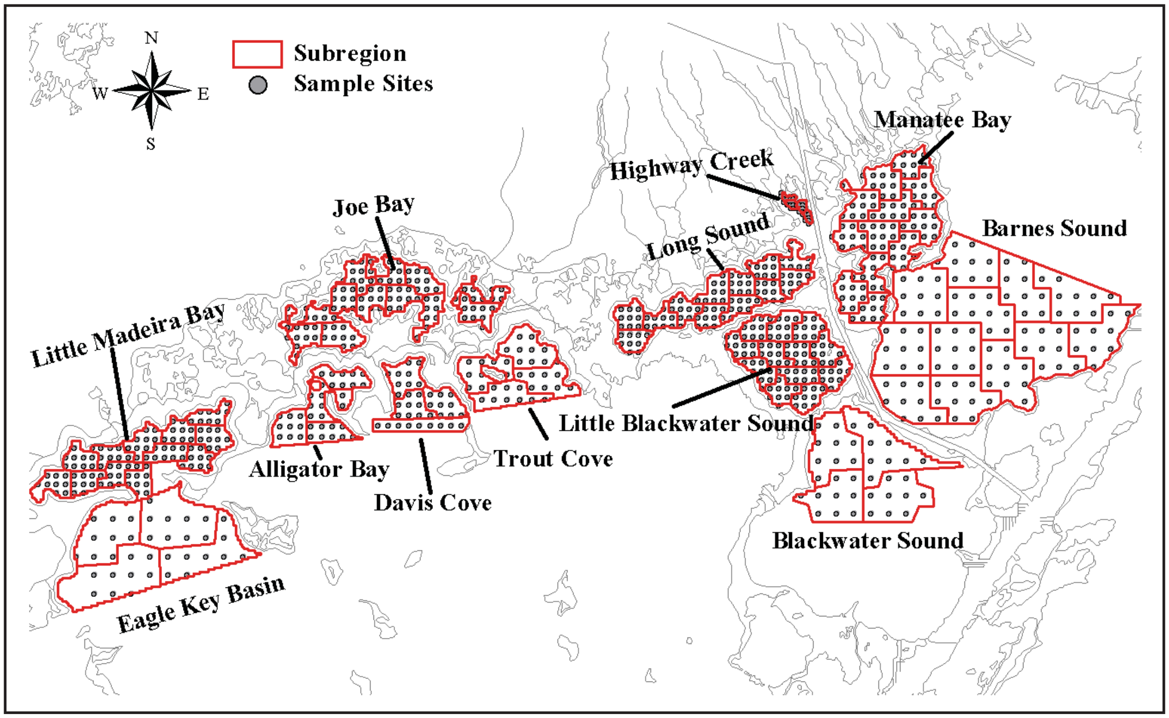


FIGURE 9-61. LEVEL 1 FISH HABITAT ASSESSMENT PROGRAM SAMPLING AREAS IN NORTHEASTERN FLORIDA BAY

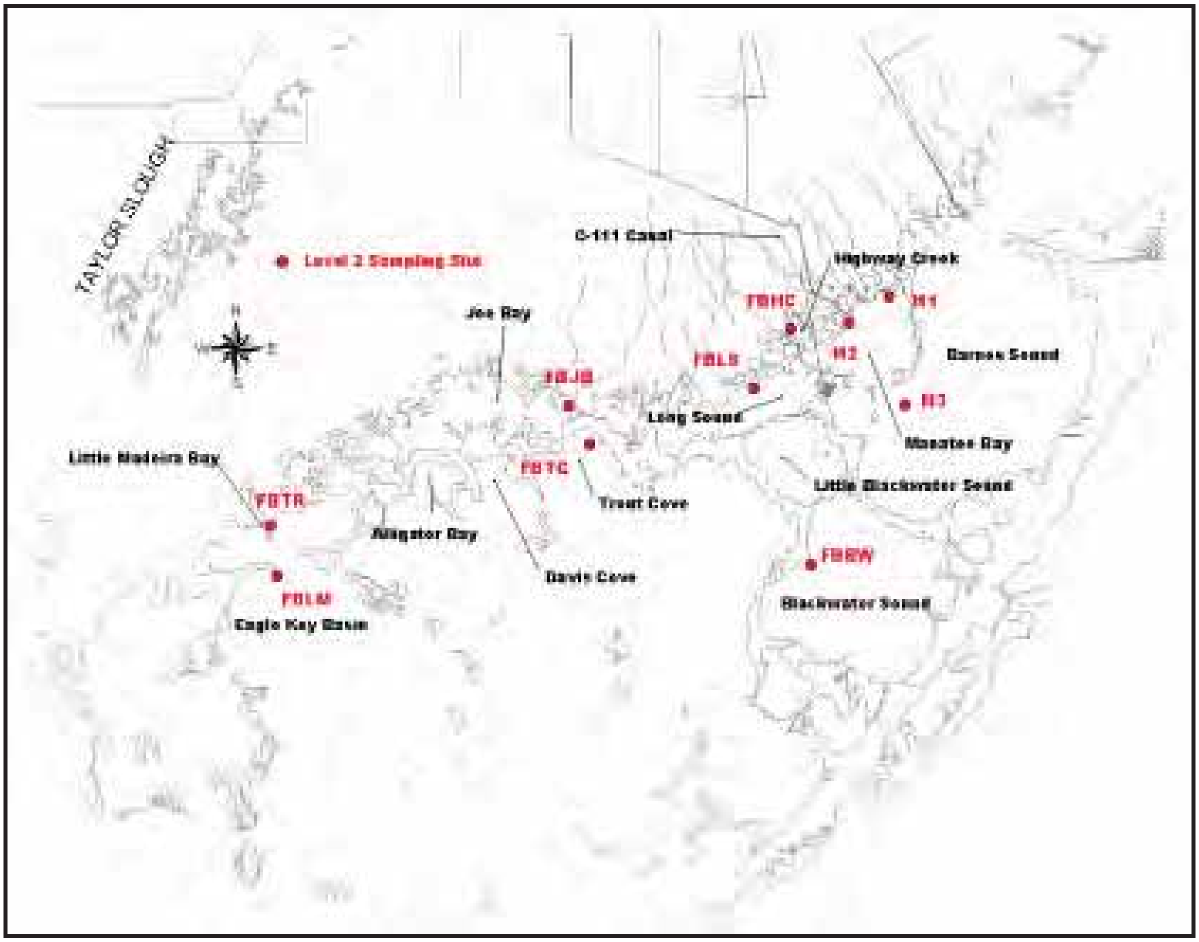


FIGURE 9-62. LEVEL 2 FIXED SAMPLING SITES IN NORTHEASTERN FLORIDA BAY

9.4.4.2 Results

During the period of record of the monitoring program, salinity has shown a general increase over time in northeast Florida Bay. Dissolved oxygen and oxidation-reduction potential vary proportionally to each other, and evidence a decreasing trend with a minimum for both parameters recorded in 2005. Moreover, 2007 may set another minimum for dissolved oxygen in both zones. Oxidation-reduction potential has a strong correlation with hydrogen sulfide concentrations (Eckert, 1993), which create reducing conditions and have been shown to play a role in *T. testudinum* die-offs in Florida Bay (Carlson et al., 1994). Lastly, the extinction coefficient K shows a significant (p less than 0.001) but weak (r^2=0.25) downward trend over time.

The seagrass community evaluation was based on short shoot density. Northeastern Florida Bay is dominated by three seagrass species, *T. testudinum*, *H. wrightii* and *R. maritima*. *Halophila engelmannii*, commonly called star grass, is also present with a limited distribution and abundance. *T. testudinum* is absent from Highway Creek and Joe Bay; however, it is the dominant macrophyte throughout the remainder of the region. *R. maritima* has a limited distribution, maintaining a relatively consistent presence in Joe Bay and Highway Creek (Fourqurean et al., 2002). *R. maritima* is one of the dominant components of the SAV community in the creeks and ponds further upstream (Frezza et al., 2003) and persists in such areas having a dynamic salinity regime.

Localized (i.e. intra-basin) losses of *T. testudinum* were documented following passages of Hurricane Georges in 1998 and Hurricane Irene in 1999, which appear to have been the driving influences of the changes (Bacon et al., 2001; Avila et al., 2003). From May 2004 through May 2005, a prolonged period of elevated salinity occurred in each of the basins monitored (Avila et al., 2005). Concurrent with this period of elevated salinity was the loss of *R. maritima* in Joe Bay and Highway Creek (Avila et al., 2005). The high salinity period ended in summer 2005 due to rainfall from Hurricanes Katrina and Wilma. At approximately the same time, an unprecedented algal bloom occurred in southern Biscayne Bay and Blackwater Sound area (Rudnick et al., 2007; 2008), and did not fully dissipate until 2008. Loss of *T. testudinum* has been documented in Blackwater Sound co-occurring with the storm events and multi-year algal bloom (Rudnick et al., 2008).

9.4.4.2.1 Eastern Zone

Both frequency and shoot density of *T. testudinum* have declined in recent years in the Eastern Zone. In 2006, there was a 34.6 percent decrease in shoot density compared to 2004. Furthermore, three successive years of lower mean shoot density were recorded beginning in 2005, decreasing to period-of-record lows in 2007 (***Figure 9-63***). The 1997 to 2007 data show a generalized trend of increasing salinity (with a drastic decrease during the 2005 hurricane season), increased reducing conditions, and decreasing oxygen concentrations coinciding with decreases in *T. testudinum* density.

The three-year period of 1998 to 2000 was the highest three-year period in terms of shoot density and frequency for *T. testudinum* in the Eastern Zone (***Figure 9-63***). During these three years, salinity was moderate and had relatively low variation. The latter part of 2005, following Hurricane Katrina, showed notable losses of *T. testudinum* in this region, and it was the first of three successive years of period-of-record low shoot densities. In Blackwater Sound and Barnes Sound, for 2005 versus 2007, shoot density decreased 85.9 and 70.1 percent, respectively. However, some recovery is evident by the increase in frequency for Manatee Bay, Barnes Sound and Little Blackwater Sound in 2007. Data from the four sampling sites located in the northern corner of Blackwater Sound show a continued decline in the annual percent frequency of *T. testudinum*, and, for 2007, a region-wide period-of-record low of 35.4 percent was recorded. During the 2004 to 2005 period of elevated salinity, dissolved oxygen and redox potential were depressed from previous years. Following Hurricane Katrina, large areas of the Eastern Zone had dissolved oxygen less than 2 mg/L and redox less than -70 millivolt (mV). To some degree, the *T. testudinum* population of the Eastern Zone may have been effected by the oxygen/sulfide stress dynamic operating at both a chronic low level during the 2004 to 2005 elevated salinity event and an extreme event following Hurricane Katrina. All four of the Eastern Zone basins had period-of-record low *T. testudinum* shoot densities within the time frame of the salinity reaching the respective basin minimums following Hurricane Katrina.

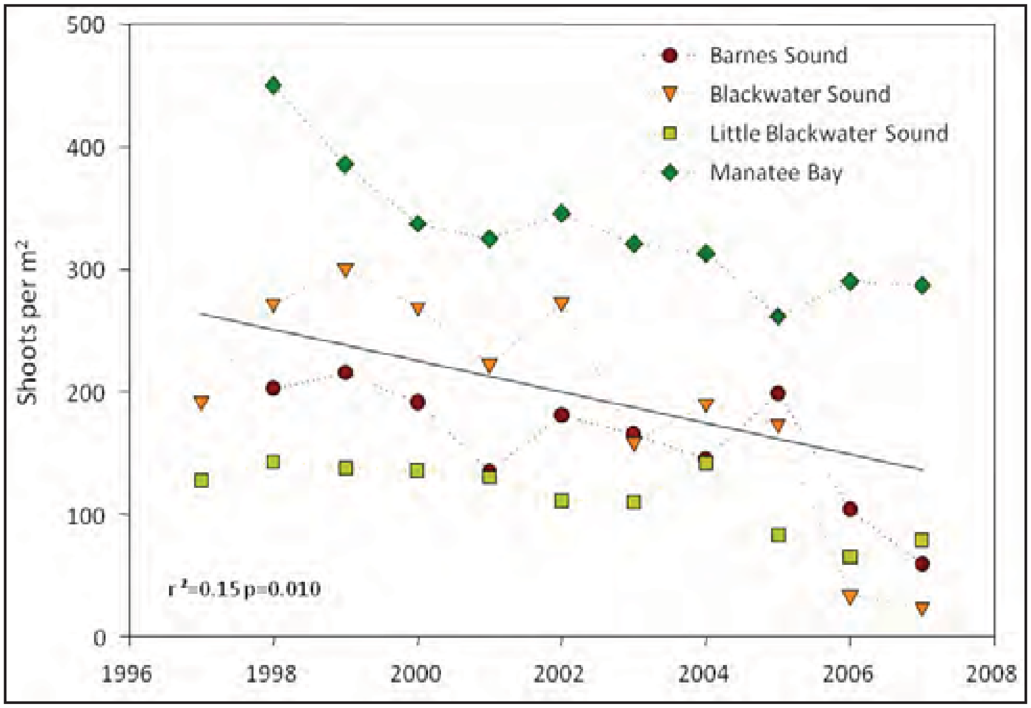


FIGURE 9-63. MEAN ANNUAL SHOOT DENSITY OF *T. TESTUDINUM* IN EASTERN ZONE BASINS WITH THE OVERALL TREND LINE SHOWN IN BLACK

The rapid loss of seagrass started after the bloom event in Manatee Bay, Barnes Sound and Blackwater Sound (Rudnick et al., 2007). During this period, chlorphyll *a* concentrations were measured as high as 18 µg/L in Barnes Sound. At the end of 2007, chlorophyll *a* levels in this region remained 10 to 12 times historic median concentrations. Water clarity was also affected by these events and decreased light values were recorded. For the zone as a whole, 2005 through 2007 light values were notably low and 2006 set the annual record low for all four basins in the Eastern Zone. With the 2007 data analyzed, the zone as whole, as well as some of the individual basins within it, is at period-of-record lows for dissolved oxygen. This is likely the result of some combination of reduced photosynthesis for *T. testudinum* and other SAV species, decomposition of organic matter, and sulfide-induced stress.

Within the Eastern Zone, Little Blackwater Sound had the lowest *T. testudinum* shoot density (***Figure 9-63***) and the highest *H. wrightii* shoot density (***Figure 9-64***) for most years. This basin also experienced the lowest annual salinity and highest annual salinity variation. Within the other three basins, the annual mean metrics for *H. wrightii* exhibit the general trend of opportunistic expansion during periods when *T. testudinum* is reduced. For example, for the first two years of monitoring, Blackwater Sound had high metrics for *H. wrightii* coinciding with relatively lower density of *T. testudinum*. Up to the initiation of the bloom, *T. testudinum* exhibited a slow decline whilst green algae increased.

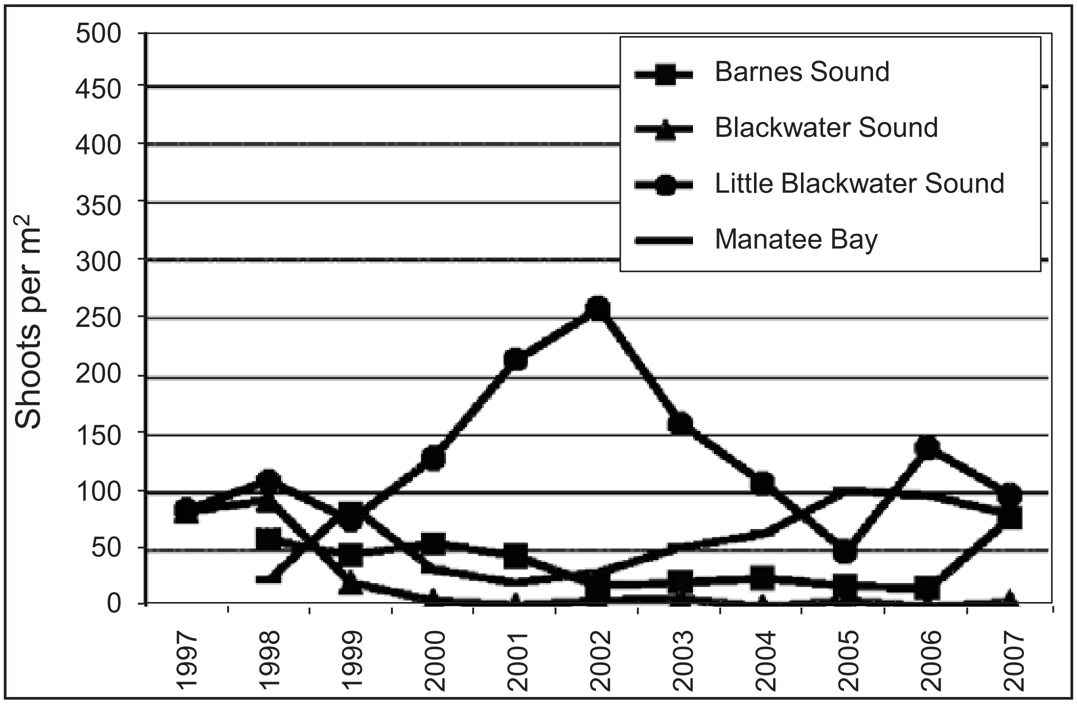


FIGURE 9-64. MEAN ANNUAL SHOOT DENSITY OF *H. WRIGHTII* IN EASTERN ZONE BASINS WITH THE OVERALL TREND LINE SHOWN IN BLACK

H. wrightii in the Eastern Zone showed two consecutive years of low mean shoot density in 2004 and 2005, while during 2006 and 2007, the shoot density was within the range of the previous years (***Figure 9-64***). The frequency of *H. wrightii* showed highest values in 2006 and 2007 since the addition of the Manatee Bay and Barnes Sound basins in 1998. *R. maritima* has not been a consistent or abundant component of the SAV community in the Eastern Zone during the period of record, and it has not been recorded in this zone since 2004. Finally, the seagrass community is showing signs of stabilization and recovery in these basins. *H. wrightii* frequency and density has responded to the displacement of *T. testudinum* throughout the Eastern Zone and, with the exception of Blackwater Sound, the frequency of T. *testudinum* was higher in 2007 than in 2006.

9.4.4.2.2 Northern Transition Zone

Both frequency and shoot density of *T. testudinum* have declined in recent years. In 2006, frequency decreased 8.2 percent compared to 2004. Shoot density decreased by 19.7 percent from 2004 to 2006. Beginning in 2005, three successive years of lower mean shoot density have been recorded, decreasing to period-of-record lows in 2007 for almost all the Northern Transition Zone sub-basins (***Figure 9-65***).

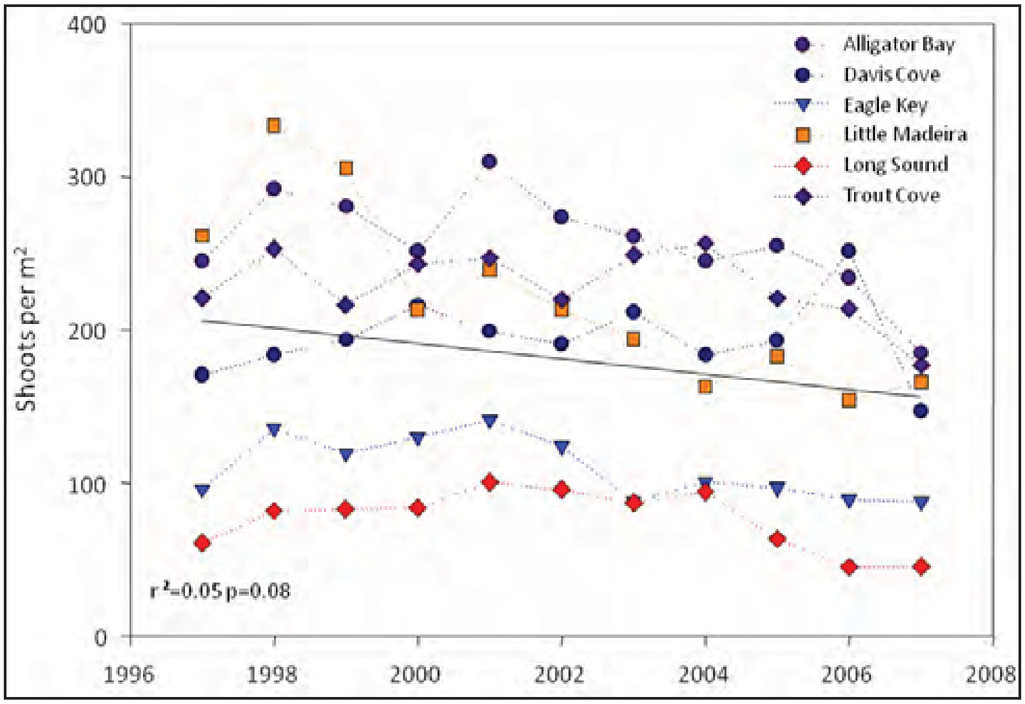


FIGURE 9-65. MEAN ANNUAL SHOOT DENSITY OF *T. TESTUDINUM* IN NORTHERN TRANSITION ZONE BASINS WITH THE OVERALL TREND LINE SHOWN IN BLACK

T. testudinum decreased 20 percent within the Northern Transition Zone when comparing the first three years of the program to the three most recent years. From 1997 to 1999, Little Madeira Bay had the highest *T. testudinum* density of the basins within the zone. However, in 2000 the basin experienced a loss of *T. testudinum*, and has remained at a lower density through subsequent years (***Figure 9-65***). The three open coves of Duck Key Basin, Alligator Bay, Davis Cove and Trout Cove, as well as the large open Eagle Key basin, have maintained a *T. testudinum* frequency near 100 percent for the period of record, and, in terms of shoot density, all four basins appear to be within historical ranges. In the downstream basins dominated by *T. testudinum*, the overall long-term changes in density appear to be driven primarily by the loss of a large area of high shoot density within Little Madeira Bay in late 1999 (Bacon et al., 2001; Avila et al., 2003) as well as some of the more recent, although less dramatic, declines in other basins (e.g., Long Sound).

For *H. wrightii*, Little Madeira Bay and Long Sound have had the highest density and frequency among the six basins in the Northern Transition Zone (***Figure 9-66***). Joe Bay and Highway Creek are evaluated separately due to the absence of *T. Testudinum*. No apparent trend was observed with the metrics measured for this seagrass in these basins, with the exception of Long Sound and Little Madeira Bay, which experienced increases in *H. wrightii* following declines in *T. testudinum*. The other four basins show a slight decline in frequency.

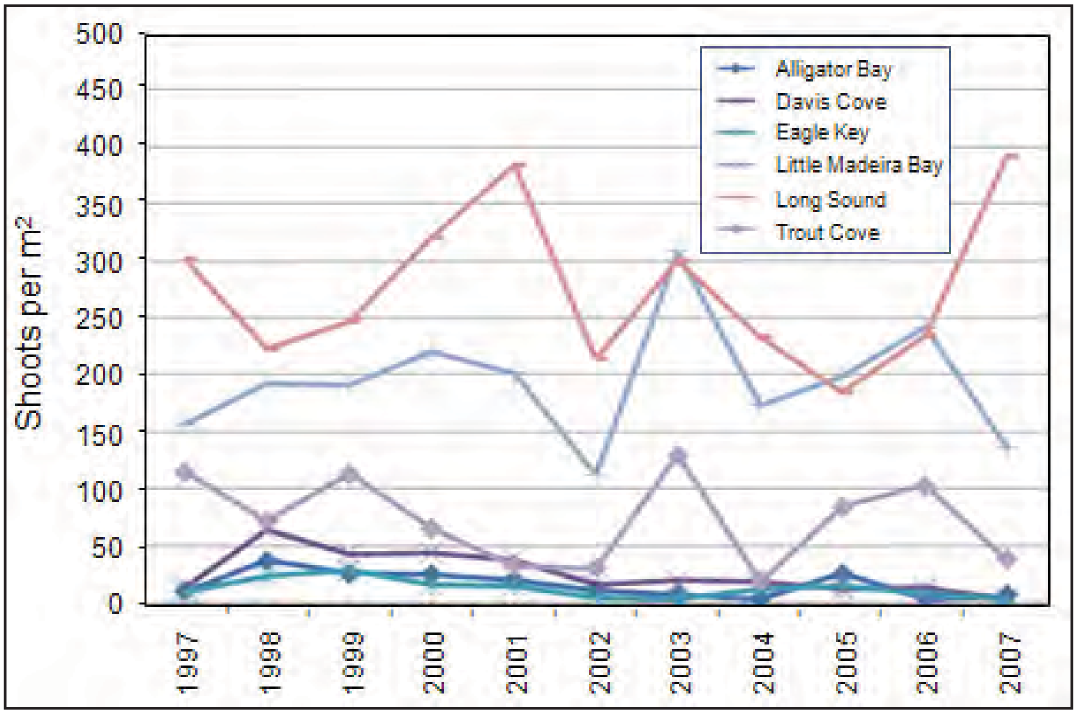


FIGURE 9-66. MEAN ANNUAL SHOOT DENSITY OF *H. WRIGHTII* IN THE NORTHERN TRANSISTION ZONE BASINS AND OVERALL TREND LINE (IN BLACK)

In Joe Bay and Highway Creek, which do not have *T. testudinum* and are primary habitat for *R. maritima*, peaks in *H. wrightii* shoot density have coincided with periods of elevated salinity. Two such periods have occurred within the period of record: 1999 to 2000 and 2004 to 2005. During the latter period of elevated salinity, both basins had a complete loss of *R. maritima* and this species has not been recorded in these basins since. Mean salinities in 2006 and 2007 for both basins are below 20 psu; however, reviews of quarterly data show dry season peaks above 31 psu and four to five month decreases to wet season lows of approximately 1 psu. Concurrently, *H. wrightii* density has varied (***Figure 9-66***), but frequency has increased approximately 60 percent, and is now the only seagrass species in these basins.

The Northern Transistion Zone maintained a consistent presence of *R. maritima* into 2005, but density and frequency of *R. maritima* decreased during 2005, showing period-of-record low values. In 2006, *R. maritima* showed a minimal increase in these metrics.

For a more detailed description of the results of this monitoring program, please refer to Avila and Blair (2006, 2008).

9.4.5 South Florida Fisheries Habitat Assessment Program in Florida Bay

The Florida Bay FHAP has provided spatially explicit data on the distribution, abundance and species composition of Florida Bay SAV since 1995. As a component of the MAP, the geographic scope of the Florida Bay FHAP was expanded in 2005 to include additional locations in the Southern Coastal Systems ranging from Lostman‟s River to southern Biscayne Bay and the program was renamed South Florida FHAP. The goal of South Florida FHAP is to provide information for spatial assessment and resolution of inter-annual variability in seagrass communities, and to establish a baseline to monitor responses of seagrass communities to water management alterations associated with restoration activities. 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 would be able to use these data to address ecosystem response issues on a near real-time basis and to weigh alternative restoration options.

9.4.5.1 Monitoring

South Florida FHAP sampling is conducted annually, at the end of the dry season (May-June) when salinity stress on seagrasses is typically highest. Sampling areas are shown as fuchsia in ***Figure 9-56.*** SAV is visually quantified at each station. When *T. testudinum* is present, short shoots are collected to determine leaf epiphyte biomass and seagrass morphometric data. Photosynthetic efficiencies are also measured for *T. Testudinum*. A more intensive sampling effort is conducted twice annually (May-June at the end of the dry season, and October-November at the end of the wet season) at 15 permanent transects in Florida Bay, which are represented by burgundy triangles in ***Figure 9-56***. Cover, above- and below-ground seagrass biomass, short shoot density, and leaf morphometrics are measured.

9.4.5.2 Results

9.4.5.2.1 Seagrass Community Structure - Spring 2008

Total seagrass is a measure of segrass distribution, which includes the infrequently observed seagrass species *Halophila decipiens* (paddle grass), *H. engelmanii* and *R. maritima*. ***Figure 9-67*** shows percent cover of total seagrass and dominant species within the Southern Coastal Systems. Results for North Biscayne Bay and Port of Miami are from spring 2007; results for the remaining areas are from spring 2008. With the exception of the southwest Florida locations, seagrasses were abundant throughout the study area (***Figure 9-67a***). *T. testudinum* was the dominant species in Florida Bay and southern Biscayne Bay (***Figure 9-67b***); however, *H. wrightii* was also consistently present in low densities (***Figure 9-67c***). *S. filiforme* occurred primarily in western Florida Bay, where it was locally abundant (***Figure 9-67d***). The northern Biscayne Bay seagrass community was dominated by *S. filiforme*, but *H. wrightii* and *T. testudinum* also occurred throughout the region in sparse to moderate densities. In contrast to most of Florida and Biscayne Bays, seagrass cover was sparse in all southwest Florida sampling locations. However, *Chara*, a macroalgal species that occurs in brackish water habitats, was found in relatively high densities throughout Coot Bay.

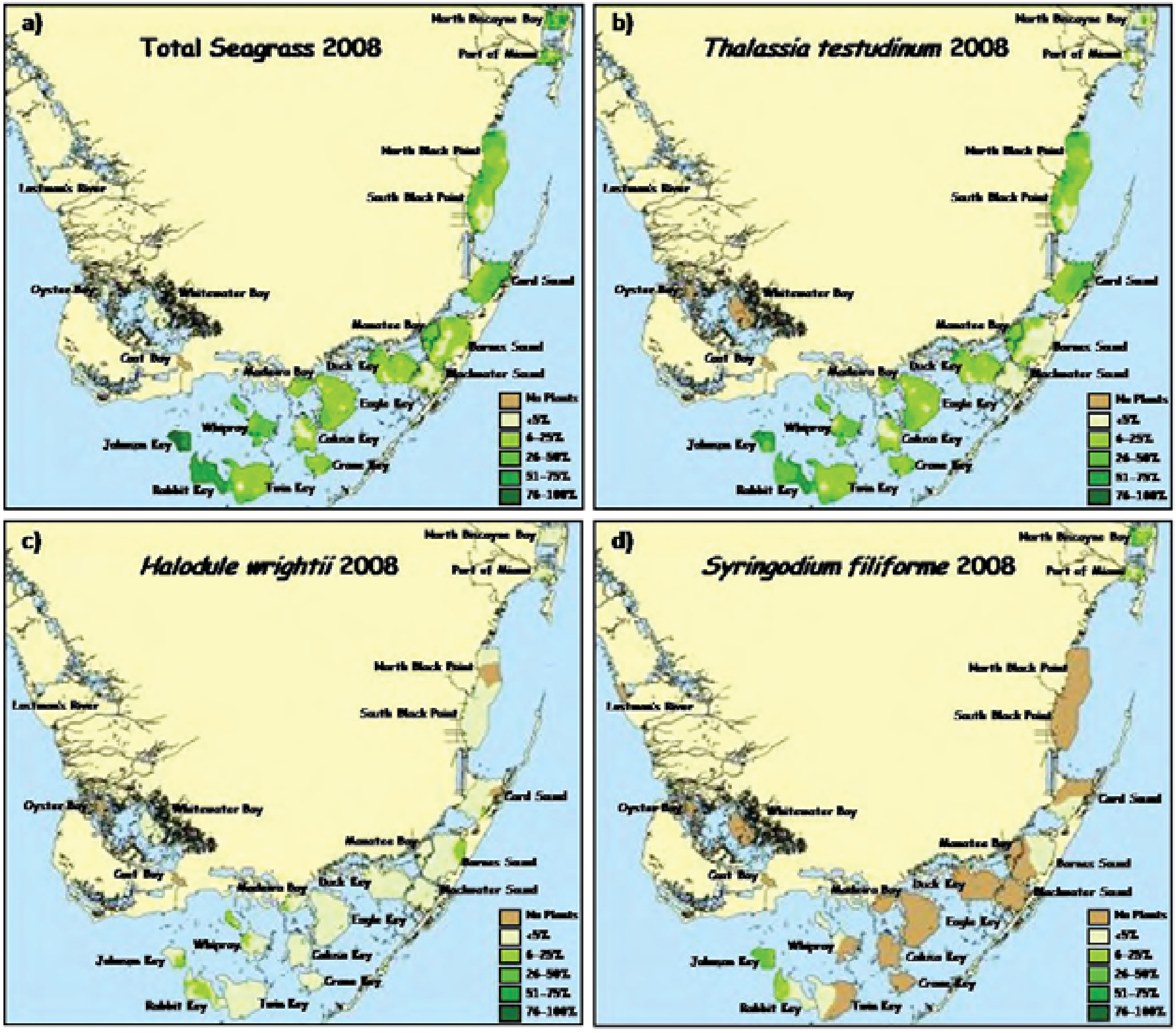


FIGURE 9-67. PERCENT COVER OF A) TOTAL SEAGRASS, B) *T. TESTUDINUM*, C) *H. WRIGHTII* AND D) *S. FILIFORME* IN SPRING 2008

9.4.5.2.2 Seagrass Trends 1995 - 2008

Evaluating trends in seagrass community structure over short time periods (e.g., 2005 to 2008) is problematic due to inherent variability in seagrass distribution and abundance between years. However, regression analyses of longer-term data (1995 to 2008) revealed significant trends in seagrass density within several Florida Bay sampling locations that corresponded to changing water quality conditions, as well as predicted species successional patterns. These trends were most apparent in western Florida Bay (***Figure 9-68***), which was subject to a widespread die-off of *T. testudinum* in the late 1980s, followed by a decline in light availability from the early to mid-1990s due to algal blooms and resuspended sediments.

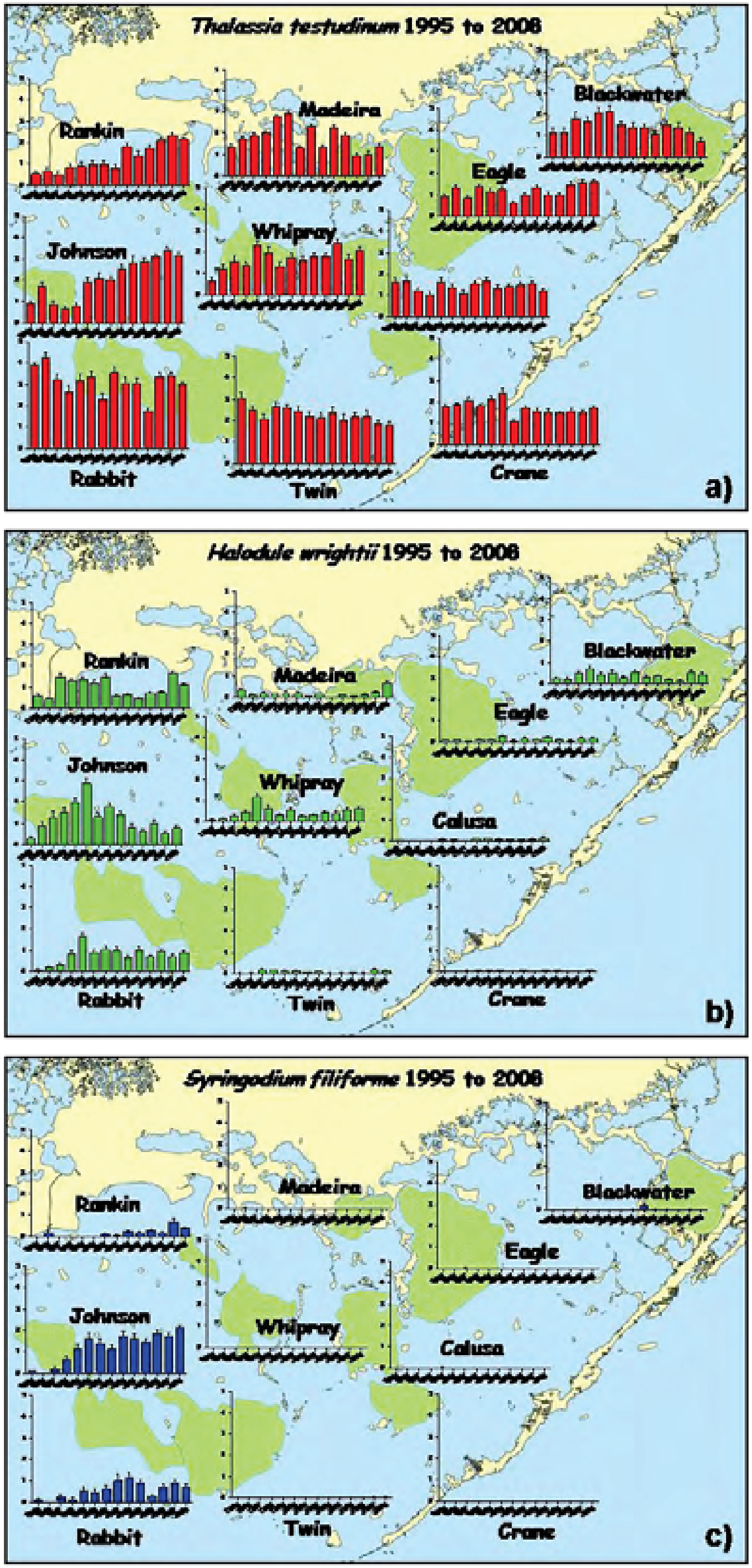


FIGURE 9-68. MEAN DENSITIES ± STANDARD ERROR (SE) OF A) *T. TESTUDINUM*, B) *H. WRIGHTII* AND C) *S. FILIFORME* FROM SPRING 1995 TO SPRING 2008

The seagrass dynamics in Johnson Key Basin illustrate a representative sequence of events in western Florida Bay (***Figure 9-69a***, ***b***). When sampling was initiated in spring 1995, cover of all seagrass species was sparse and water clarity was poor. Over time, water clarity improved and the frequencies and densities of all seagrass species increased, but temporal patterns were quite different among species. *H. wrightii*, the fastest growing seagrass species, showed the most rapid response. Eventually, the abundance of the longer lived species, *T. testudinum* and *S. filiforme*, also began to increase. *H. wrightii* density peaked in 2000, but has declined since then, exhibiting a significant cubic trend. Densities of *T. testudinum* and *S. filiforme* have continued to increase, and are exhibiting significant positive linear trends. The changes in species abundance in Johnson Key Basin are following a secondary successional pattern typical for subtropical seagrass systems, and have resulted in a mixed species community dominated by *T. testudinum*. Comparing these temporal trends to Johnson Key Basin water quality data (***Figure 9-69c***, ***d***) suggests that seagrasses are responding to increasing light availability (lowered turbidity and water column chlorophyll *a*).

More detailed information regarding the species composition, distribution, and abundance of seagrass and macro-algal species as well as other South Florida FHAP performance measures, study design, and methodology is available in the 2006, 2007 and 2008 South Florida FHAP Annual Reports (Hall and Durako, 2006; 2007; 2008).

9.4.6 Conclusions

The ability to statistically detect change from baseline conditions is a crucial component of the MAP program. Results to date suggest that changes in seagrass species composition, distribution and abundance in the Southern Coastal Systems can be detected, and that these trends correlate with available water quality information. However, both species succession, which reflects life history characteristics, and the intra- and inter-annual variability in seagrass community structure observed from 1995 to 2008 confirm that long time series are required to accurately interpret ecosystem changes that can be related to restoration. Given the present implementation schedule it is likely that such a time series can be generated, if MAP monitoring is sustained as planned.

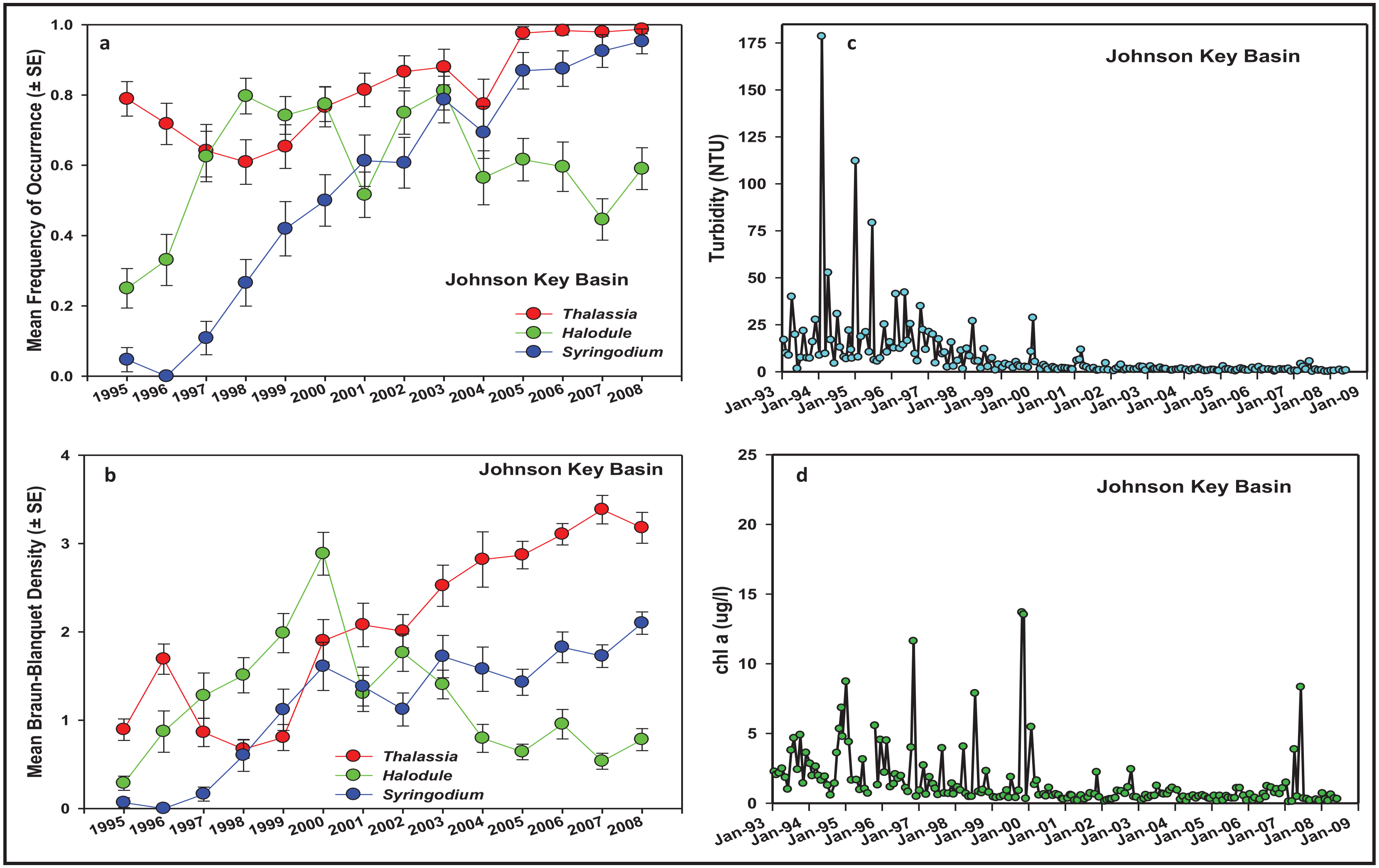


FIGURE 9-69. A) MEAN FREQUENCIES (± SE) AND B) MEAN DENSITIES (± SE) OF *T. TESTUDINUM*, *H. WRIGHTII* AND *S. FILIFORME* FROM SPRING 1995 TO SPRING 2008, AND TIME SERIES PLOTS FOR C) TURBIDITY AND D) CHLOROPHYLL *A* FROM JANUARY 1993 TO JUNE 2008 IN JOHNSON KEY BASIN

9.5 NEARSHORE FAUNAL COMMUNITIES HYPOTHESIS CLUSTER

9.5.1 Introduction and Background

Patterns of variation in Southern Coastal Systems nearshore fish and epifauna reflect processes that occur within, upstream and offshore of Biscayne and Florida Bays and the southwestern Florida coast. As the most downstream component of the Everglades ecosystem, the Southern Coastal Systems integrates multiple upstream changes, which ultimately impact the abundance and diversity of its fish and invertebrate communities. These fish and invertebrate communities are of critical importance in coastal food webs and form the basis of several commercial and recreational fishes. Faunal response to restoration is dependent on the magnitude and spatial extent of alteration in several factors that include physical (salinity, turbidity), primary producer (seagrass and phytoplankton) and secondary producer (zooplankton and epibenthic invertebrates) changes (***Figure 9-70***).

Implementation of restoration projects is expected to improve salinity patterns in space and time, creating a salinity regime that is more characteristic of estuaries. As a result, faunal expected to increase and become more characteristic of south Florida estuaries densities and species diversity within these areas are (***Figure 9-70***).

The faunal projects described in this section provide fishery-independent assessments of the early life stages of several important coastal fish populations in the Southern Coastal Systems. Thus, the data provide indicators of the potential effect of restoration on the coastal fishery populations in south Florida, one of south Florida‟s primary economic resources. Moreover, the majority of the public will likely perceive the success of restoration based upon its impact on the Southern Coastal Systems, especially fishery resources dependent upon the habitat within this region. This contention of public perception of success is not arbitrary, but based upon the number of visitors to the Southern Coastal Systems coastal waters to interact with the fauna via fishing, snorkeling and diving, far outnumbering the number of visitors to the wetlands in and adjacent to Everglades National Park. Given their ecological, economic and social importance, it is critical that the faunal components of the Southern Coastal Systems be an integral part of the adaptive management decision-making framework.

Performance measures developed for Southern Coastal Systems are 1) fish and 2) juvenile pink shrimp and associated epifauna. Documentation for these measures can be found at www.evergladesplan.org/pm/recover/perf_se.aspx. In addition, an interim goal has been developed for juvenile pink shrimp; documentation for which can be found at www.evergladesplan.org/pm/recover/recover_docs/igit/igit_mar_2005_report/ig_4-3_seshrimp.pdf.

Several fish and epifauna monitoring efforts are underway in the Southern Coastal Systems. These include studies carried out by NOAA in Biscayne Bay on mangrove fish and alongshore epifaunal communities. NOAA monitors juvenile sportfish in Florida Bay and both NOAA and USGS assess seagrass fish and invertebrates in the Southern Coastal Systems. The locations of these monitoring efforts are shown in ***Figure 9-71***. This figure shows oyster monitoring sites. Oysters are discussed in Section 9.6 Oyster Hypothesis Cluster.

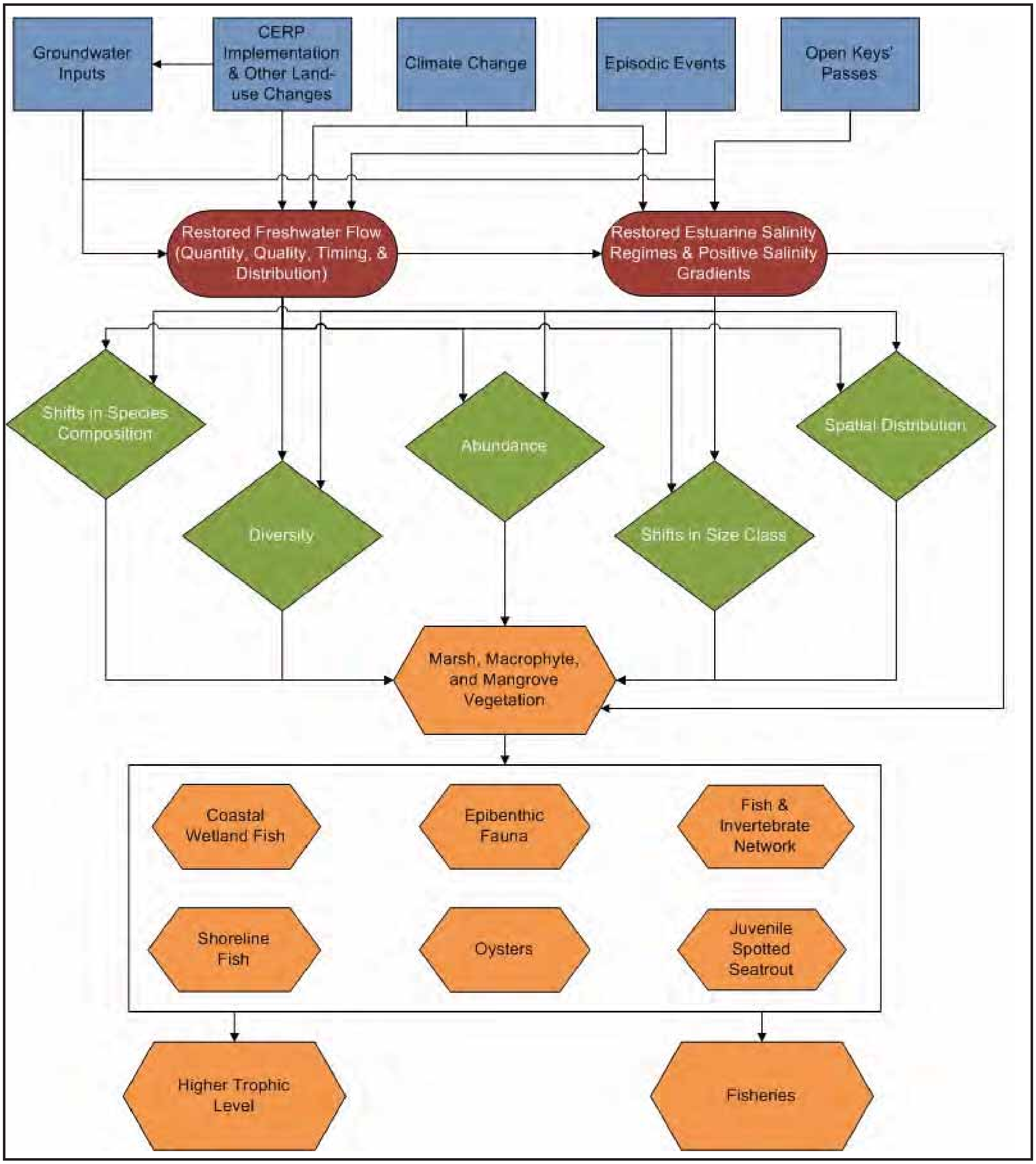


FIGURE 9-70. CONCEPTUAL ECOLOGICAL MODEL FOR NEARSHORE FAUNAL COMMUNITIES FOR THE SOUTHERN COASTAL SYSTEMS

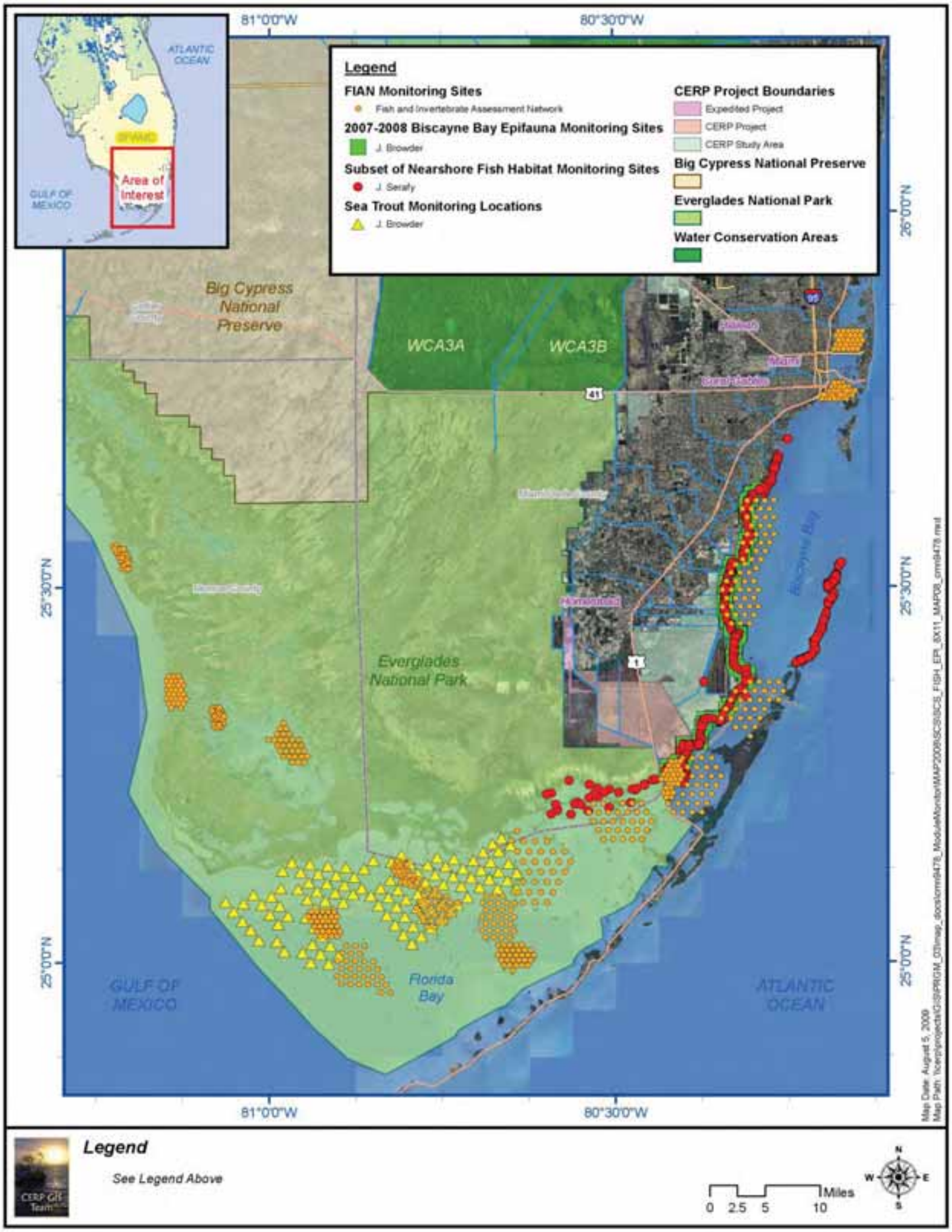


FIGURE 9-71. MULTI-SPECIES NEARSHORE FISH AND EPIFAUNAL MONITORING NETWORK ACROSS THE SOUTHERN COASTAL SYSTEMS DOMAIN

9.5.2 Biscayne Bay Mangrove Fish

9.5.2.1 Monitoring

A visual survey conducted in Biscayne Bay quantifies juveniles of larger fish species not adequately sampled by other efforts, including some important sport and commercial fish (e.g., snapper, grunt and snook) that are found in or near the mangrove zone of Biscayne Bay. Restoration-related impacts on coastal systems are likely to be the strongest and most easily discerned along the mangrove-lined shores of south Florida‟s mainland. Southern Biscayne Bay‟s shoreline fish community has been monitored visually since 1998 (***Figure 9-72***). Methods and results regarding the first two years of monitoring are detailed in Serafy et al. (2003). Since then, monitoring has expanded spatially and sampling intensity has increased to 1) determine pre-restoration variation in selected fish community indices (i.e. fish taxonomic richness and dominance) and taxon-specific abundance metrics such as occurrence and density of individual taxa; and 2) examine relationships between nearshore salinity regimes and the composition and structure of the shoreline fish community. Sampling is conducted twice annually, during the wet (July to September) and the dry (January to March) seasons. To date, the survey spans 21 consecutive wet-dry seasons (10.5 years) along four shoreline segments. Collected since 2005 are data associated with 20 to 50 belt transects (samples) per annum for each shoreline segment. Monitoring has focused on the western and eastern margins of southern Biscayne Bay, Card Sound and Barnes Sound (***Figure 9-72***).

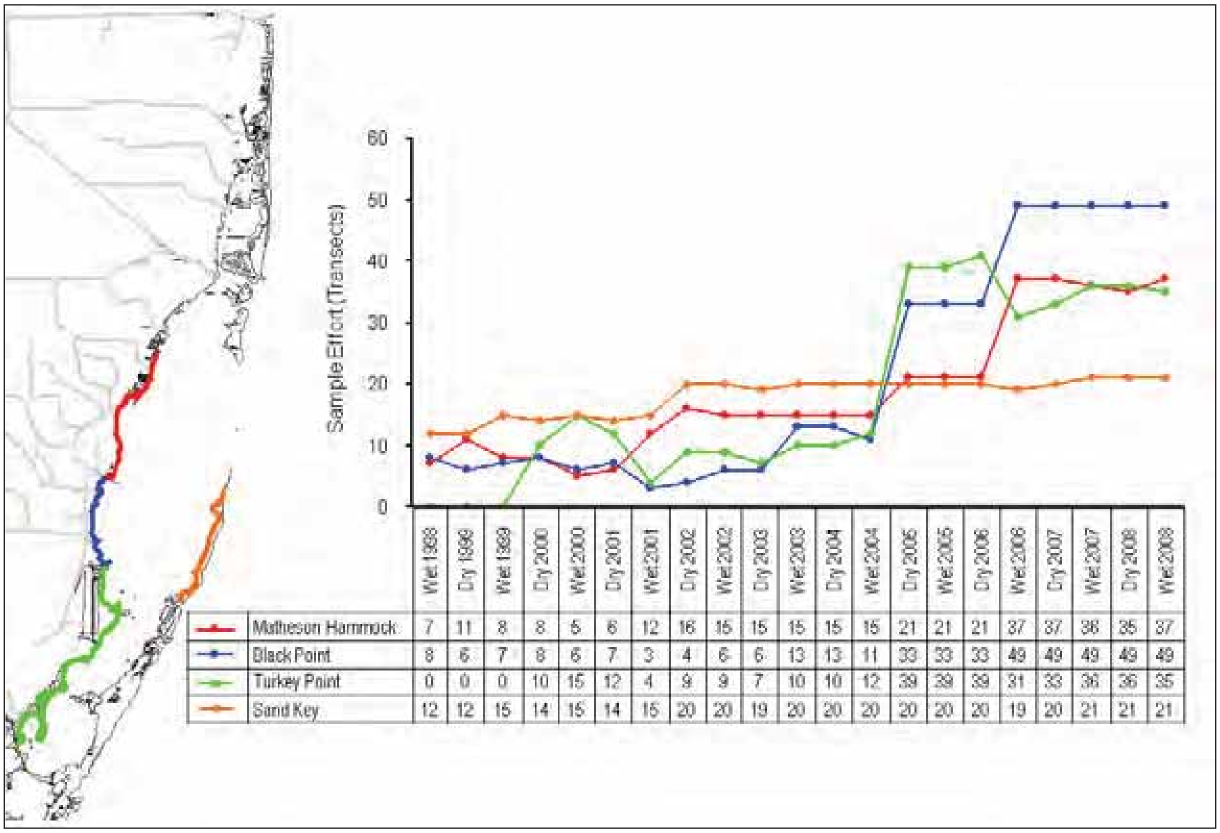


FIGURE 9-72. SPATIAL AND TEMPORAL EXTENT OF SAMPLING EFFORT ASSOCIATED WITH BISCAYNE BAY MANGROVE FISH ASSESSMENT WITH VALUES REPRESENTING NUMBERS OF VISUAL TRANSECTS

9.5.2.2 Results

9.5.2.2.1 Community Indices

Figure 9-73 presents the taxonomic richness time series for mangrove fish along each shoreline segment (per color-coding of ***Figure 9-72***) monitored in Biscayne Bay. Taxonomic richness (i.e., the number of species) is significantly lower ($p<0.05$) along the western shoreline (red, blue, and green) in comparison to the eastern shoreline Sand Key transect (orange). Note that the taxonomic richness of mangrove fish are based on a size class of greater than two centimeters, precluding direct comparison to the epifaunal studies (next section). For comparison, a summary of mangrove fish surveys is available in Faunce and Serafy (2006). Power analyses (Serafy et al., 2009) indicate allocation of sampling effort is adequate to detect a 20 percent change in species richness (alpha=0.05, beta=0.8) on an annual basis. The error bars in the figure represent 95 percent confidence intervals. Linear regression analyses indicate the community index has been relatively stable for all shoreline segments over the period of record.

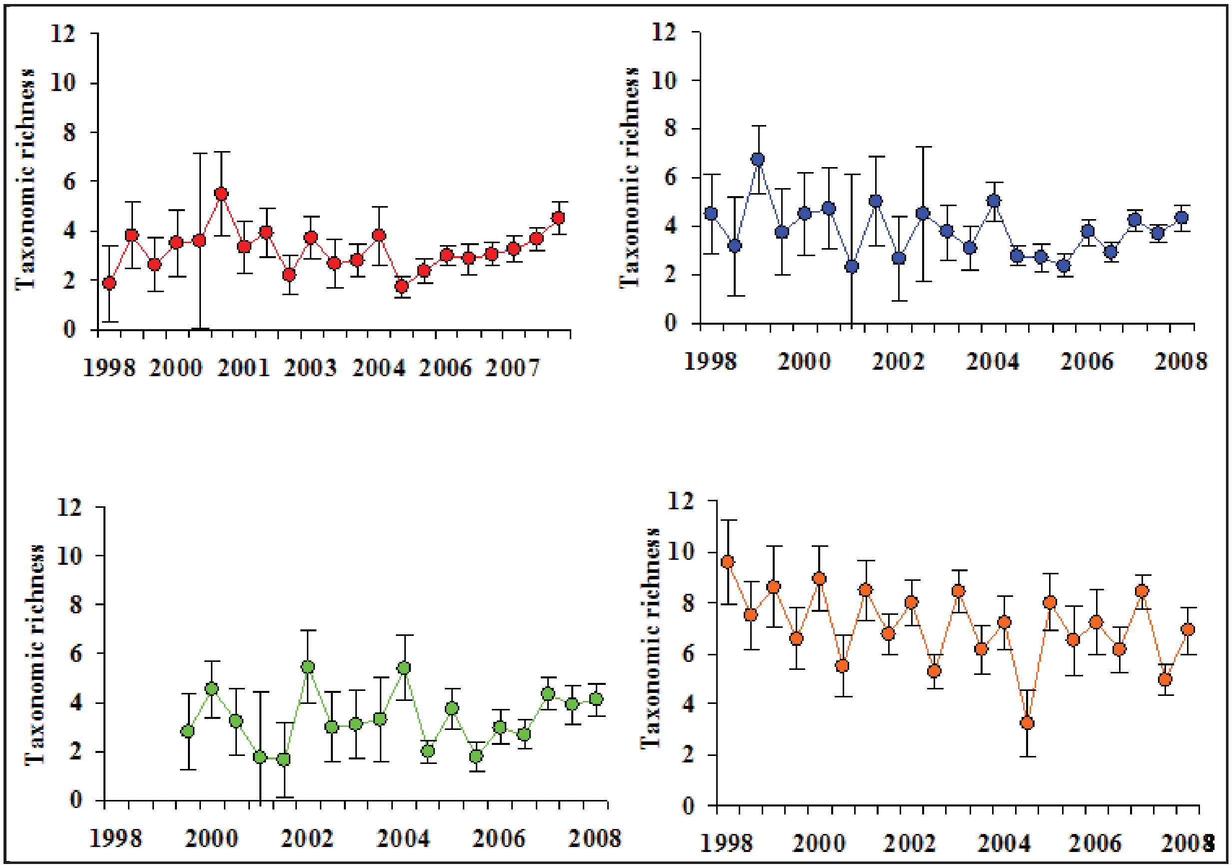


FIGURE 9-73. TIME SERIES PLOTS PROVIDING MEAN FISH TAXONOMIC RICHNESS ALONG FOUR NEARSHORE SEGMENTS OF THE MONITORING DOMAIN

Note: Color-coding refers to convention presented in ***Figure 9-72***

9.5.2.2.2 Taxon-Specific Abundances

Typically, taxon-specific catch-per-unit-effort and density values are dominated by zero values, which compromise the application of standard parametric statistical procedures. In these cases, analysis of the frequency of occurrence and concentration (i.e. density when present) of individual taxa is powerful from both statistical and ecological standpoints. Two examples of the type of time series data that has been developed for several fish taxa are presented in ***Figure 9-74.*** These plots track temporal variation in gray snapper (*Lutjanus griseus*) and yellowfin mojarra (*Gerres cincereus*) abundance along the mainland stratum of all segments combined over the 10.5-year period of record. Linear regression trend analyses indicate that abundance levels of both species have been relatively stable over the period of record. Power analyses indicate that the current sampling design is sufficient to detect a change of 40 percent or more (alpha=0.05, beta=0.8) in the occurrences of both species on a season-by-season basis. Detecting change (20%, alpha=0.05, beta=0.8) in the concentrations of both species, however, requires pooling of samples into five-year time intervals.

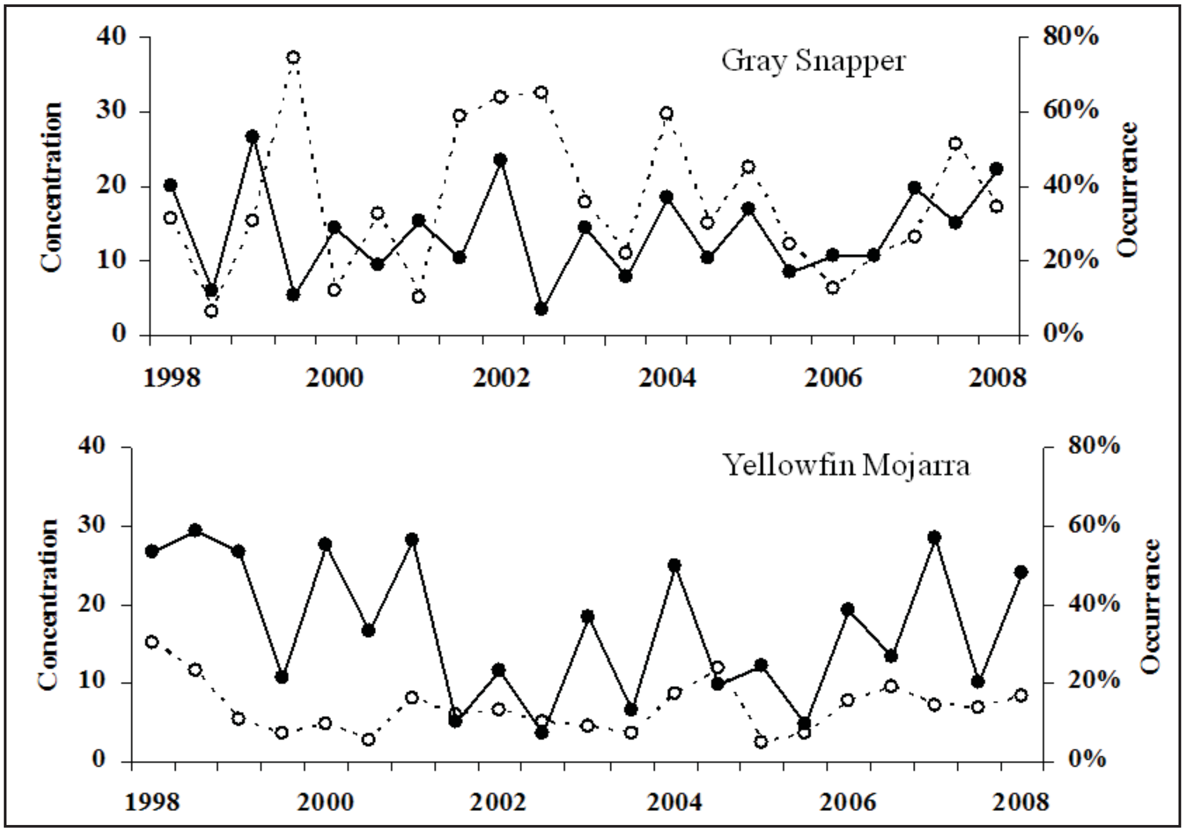


FIGURE 9-74. GRAY SNAPPER AND YELLOWFIN MOJARRA ABUNDANCE TIME SERIES ALONG THE MAINLAND STRATUM WITH OCCURRENCE (CLOSED CIRCLES) AND CONCENTRATION (OPEN CIRCLES)

9.5.2.2.3 Habitat Suitability Models

Logistic regression techniques were utilized to draw inferences about possible impacts of future salinity changes on habitat suitability for individual fish taxa. ***Figure 9-75*** presents results for gray snapper and yellowfin mojarra. Data analyzed were restricted to the wet season at sites along the mainland stratum. The same response surfaces are depicted in upper three-dimensional plots and lower contour plots. The color scale in contour plots ranges from low (black) to high (red) where warmer colors indicate higher occurrence values. Until restoration-related changes to salinity fields are implemented, statistical models of this type, coupled with appropriate laboratory studies, offer insights into how a range of different salinity regime scenarios may affect the suitability of shoreline habitats for resident and transient fishes. Depth appears to be an important factor given that as with all field observations there are confounding effects, which can be statistically removed (Serrano et al. 2010).

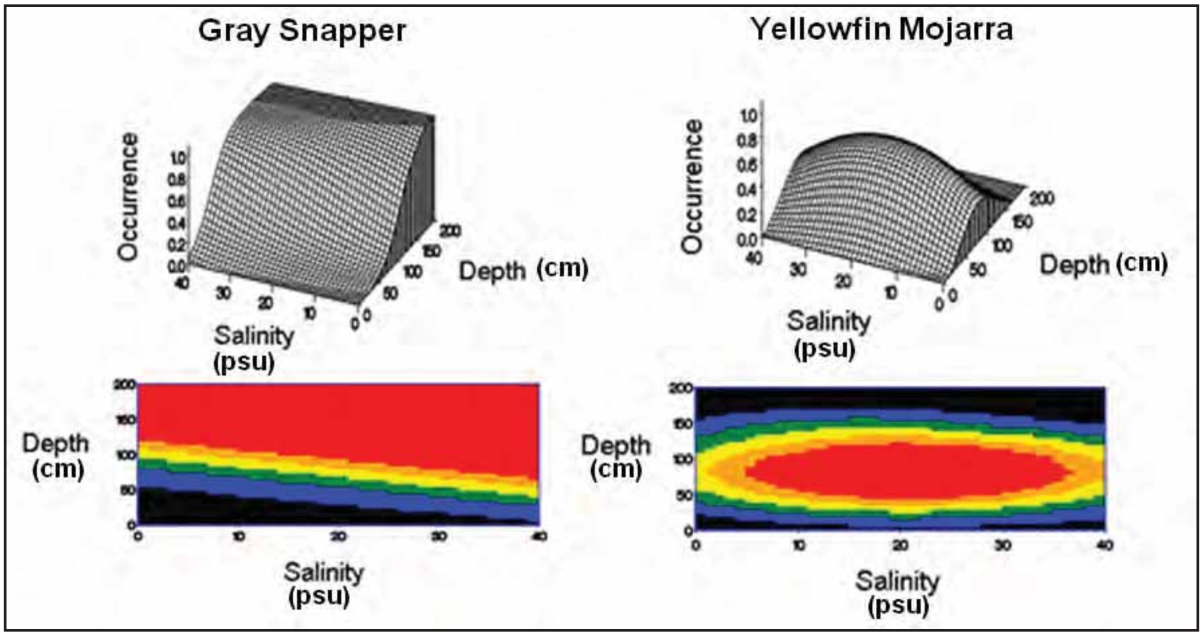


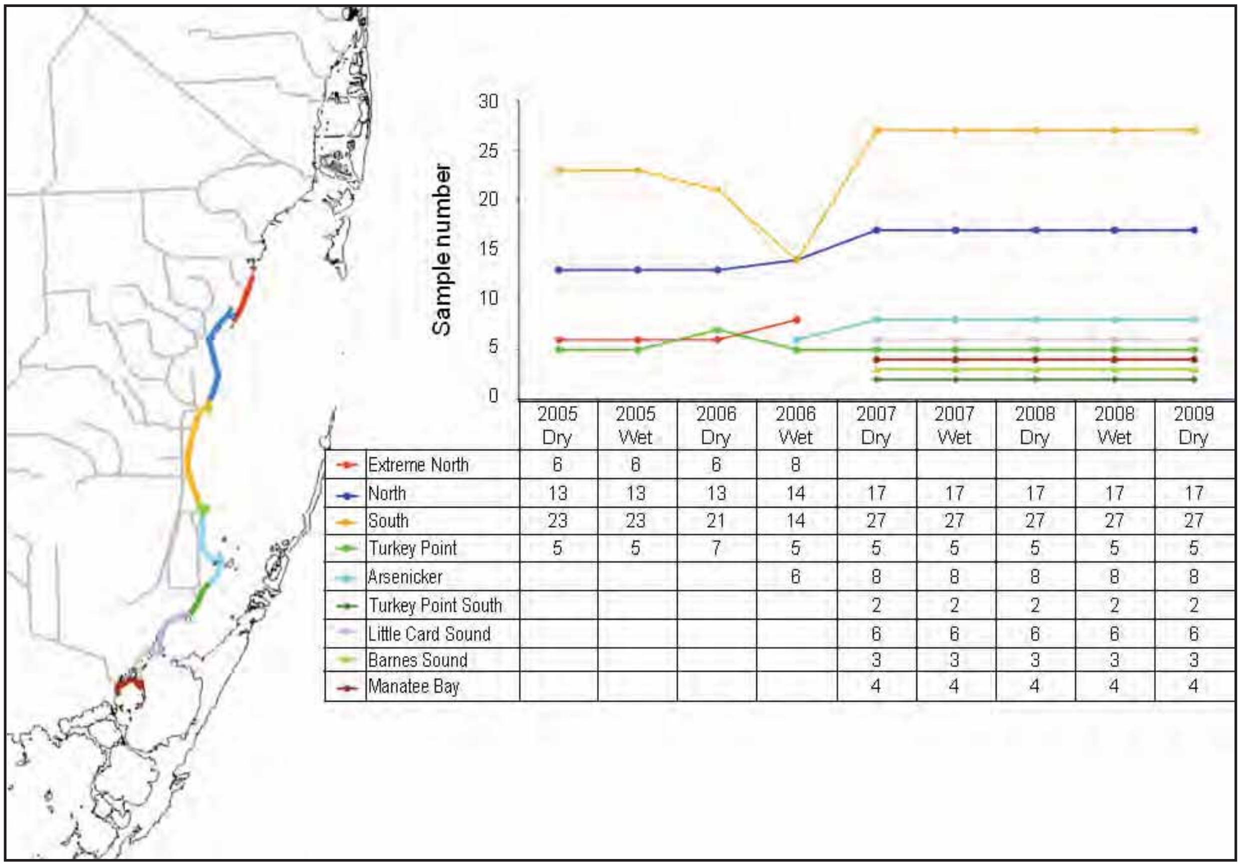
FIGURE 9-75. DEPICTION OF LOGISTIC REGRESSION RESULTS PERFORMED TO ASSESS HABITAT SUITABILITY RELATIONSHIPS FOR GRAY SNAPPER AND YELLOWFIN MOJARRA

9.5.3 Biscayne Bay Alongshore Epifauna Communities

9.5.3.1 Monitoring

The Biscayne Bay alongshore epifaunal communities monitoring focuses on faunal assemblages as a whole and a suite of the more common species in the shallow open water along the shoreline of south Biscayne Bay, which is expected to be the part of the bay detectably affected by restoration. Requirements for restoration of this community are thought to be a positive salinity gradient distributed broadly along the shoreline, a zone of salinity less than 20 psu throughout the year, and reduced frequency of short-term salinity fluctuations caused by the operation of canal control structures.

The sampling area is the open water immediately off the western shoreline of Biscayne Bay, from Shoal Point through Manatee Bay. Four years of data, 2005 through 2008, are available for the area from Shoal Point to Turkey Point, and two years of data, 2007 through 2008, are available for the area from Turkey Point through Manatee Bay (***Figure 9-76***). Two sets of samples are collected each year, one in the dry season (January-March) and one in the wet season (July-September). In 2005 and 2006, 47 sites were sampled and, in 2007 and 2008, 72 sites were sampled. Each site is located in close proximity to a shoreline fish visual survey site (Serafy et al., 2009) in order to facilitate analyses of interactions between shoreline fishes and the alongshore epifauna.



	2005 Dry	2005 Wet	2006 Dry	2006 Wet	2007 Dry	2007 Wet	2008 Dry	2008 Wet	2009 Dry
Extreme North	6	6	6	8					
North	13	13	13	14	17	17	17	17	17
South	23	23	21	14	27	27	27	27	27
Turkey Point	5	5	7	5	5	5	5	5	5
Arsenicker				6	8	8	8	8	8
Turkey Point South					2	2	2	2	2
Little Card Sound					6	6	6	6	6
Barnes Sound					3	3	3	3	3
Manatee Bay					4	4	4	4	4

FIGURE 9-76. SAMPLING AREA AND NUMBER OF SAMPLES BY AREA (DOWN) AND COLLECTION PERIODS (ACROSS) FOR ALONGSHORE EPIFAUNA COMMUNITIES MONITORING IN BISCAYNE BAY

Each species collected has been tentatively classified as estuarine or marine based on local references (Gilmore, 1995; Lorenz and Serafy, 2006) and the online database FishBase available at www.fishbase.org. Classifications according to halo-habitat will be refined and expanded based on new knowledge gained in this project. Analytical approaches applied to better understand relationships of individual species with salinity include: 1) faunal density-weighted salinity measured at sites where the species has been captured, as compared to the average measured salinity across all the sites; 2) multidimensional scaling (MDS) plots that portray the relative similarity of spatial distributions among species; and 3) curves of species abundance relative to salinity produced by statistical models. See Browder et al. (2009) for a detailed discussion of monitoring methods and analytical approaches.

9.5.3.2 Results

The eight collections of epifauna communities alongshore of Biscayne Bay consisted of 50 identified fish species, six identified crab species, and pink and caridean shrimp. The average number of fish per collection at all sites during 2007 and 2008 was 30 in the dry season and 53 in the wet season. In the 2007 and 2008 collections, the average number of fish caught per site at sites 1 through 47 was almost twice that at sites 48 through 72 (61 versus 32 over all four collections). The most abundant species collected was the rainwater killifish (*Lucania parva*) and the most abundant crab was the blue crab (*Callinectes sapidus*). Caridean shrimp were substantially more abundant than penaeid shrimp, but the pink shrimp (*Farfantepenaeus duorarum*) was well distributed across sites. Initial halo-habitat classifications were assigned to the species captured over the four-year period and can be seen in Browder et al. (2009).

Figure 9-77 is based on data from sites 1 through 47 from up to eight collections from 2005 through 2008. It shows the mean and 95 percent confidence limits of standardized density-weighted salinity for each species over all collections in which the species occurred. Density-weighted salinity is determined by summing, for all individuals of a given species, the salinity at which each individual is found and dividing by the total number of individuals of that species. Standardization was by subtracting from each density-weighted salinity, the average salinity across sites for that collection, which centers results at zero. Negative density-weighted salinity values indicate that species abundance was centered at less than the average salinity of the sites, whereas positive values indicate that species abundance was centered at greater than the average salinity of the sites. Means and confidence limits are based on two to eight collections. Only the species that occurred in at least two of the eight collections appear in ***Figure 9-77***. Species with exceptionally high confidence intervals (>10 psu) were also excluded (lined sole, bluestriped grunt, longsnout seahorse, flagfin mojarra, highfin blenny, and mud crab sp.). Their high CI might have been due to only two or three individuals being caught--and those at sites of different salinity. The average salinity across sites for each collection was 33.7, 27.5, 26.1, and 25.3 for 2005-2008 dry season collections and 21.9, 21.1, 20.4, and 23.6 for 2005-2008 wet season collections. The mean salinity across all eight collections was 25, however mean average site salinity may differ from 25 for species that appeared in less than eight collections.

The right panel of ***Figure 9-77*** shows, by taxa, the mean salinity across only the collections in which the species occurred. The center line in the right panel is set at 25 to better indicate the extent to which the mean average site salinity differed from 25 for a given species. There is no bar (just an empty space) for those whose mean average site salinity approximated 25. To relate each species in the graph to a given salinity, mentally substitute the mean site salinity, either 25 or the other salinity indicated for the species, for the zero on the main figure. The weighted density of most species occurred within the salinity range from 17 to 31. This puts most of them in the polyhaline (18-30) category of halohabitat. Exceptions may be the species whose mean site salinity bars extend substantially from the mid line of 25. Those with mean average site salinity substantially greater than 25 were spotted whiff, dusky pipefish, and lesser blue crab. Those with a mean average site salinity substantially below 25 were crested goby and swimming crab sp.

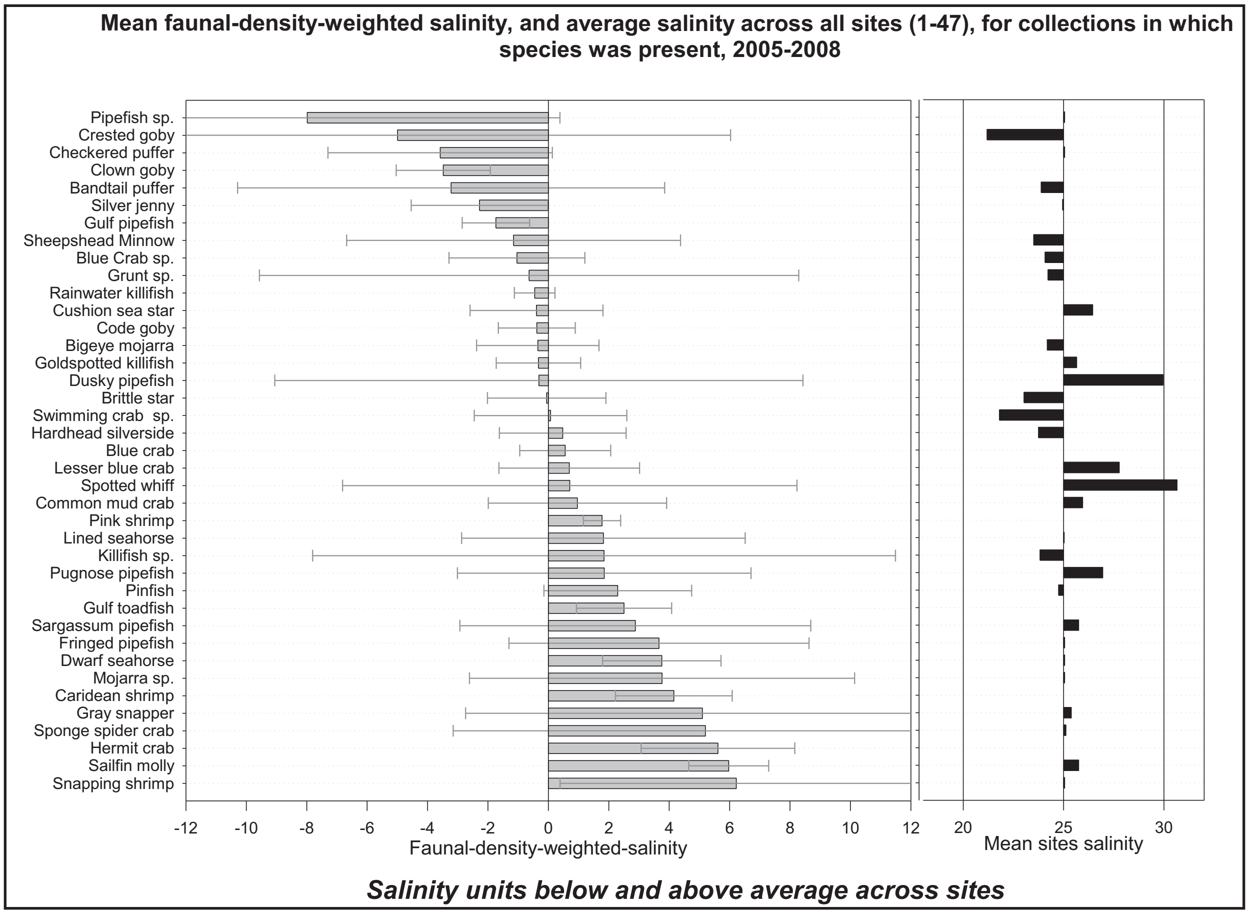
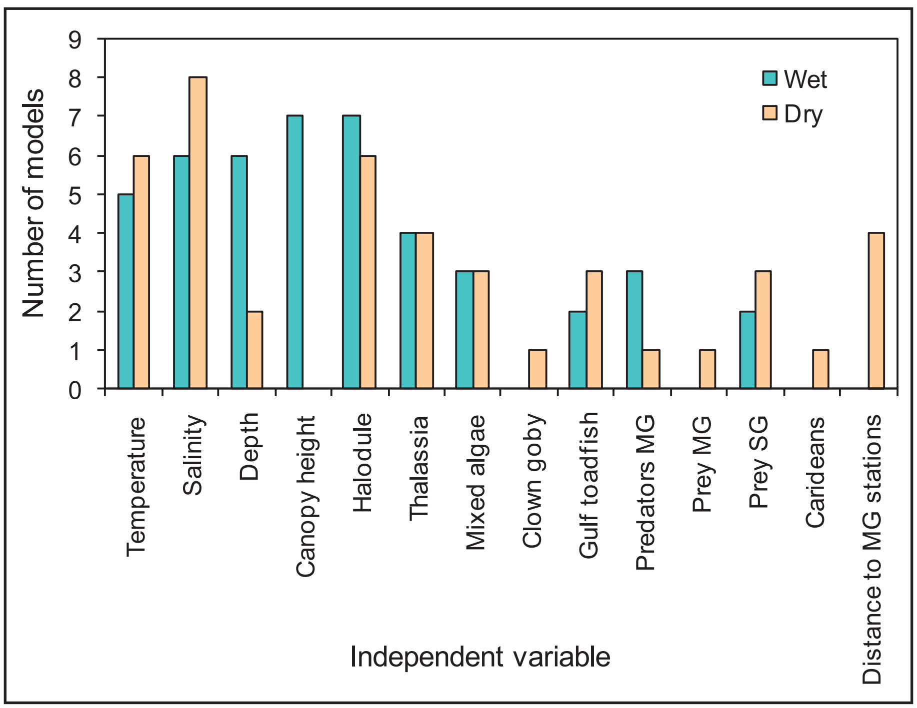


FIGURE 9-77. MEAN AND 95% CONFIDENCE LIMITS ACROSS ALL COLLECTIONS, WET AND DRY SEASON, OF THE STANDARDIZED FAUNAL DENSITY-WEIGHTED SALINITY FOR EACH TAXA AND COLLECTION FROM SITES 1 TO 47 FOR 2005 THROUGH 2008

Note: Species are ordered by their density-weighted salinity, from highest to lowest

Faunal abundance metrics were more strongly related to salinity in the dry season than in the wet season in simple linear regressions (of 5-psu salinity binned data) for nine taxa: pink shrimp, caridean shrimp, blue crab, goldspotted killifish (*Floridichthys carpio*), code goby (*Gobiosoma robustum*), rainwater killifish, clown goby (*Microgobius gulosus*), Gulf toadfish (*Opsanus beta*), and Gulf pipefish (*Syngnathus scovelli*) (Browder et al., 2009). Caridean shrimp density displayed a significant increasingly positive relationship with salinity to the limit of the range, 35 to 40 psu, in both dry and wet seasons. Dry season pink shrimp density had a significant parabolic relationship with salinity (maximum at 30 to 35 psu). Dry season blue crab and Gulf toadfish densities had significant positive linear relationships with salinity throughout the available salinity range. Dry season rainwater killifish density had a significant negative linear relationship with salinity throughout the range.

Salinity was an important variable in multiple regression models describing faunal density during both seasons, which was significant ($p \leq 0.1$) in dry season models for eight out of nine species and in wet season models for six out of ten species (Browder et al., 2009). In the dry season, four species (blue crab, caridean shrimp, pink shrimp and Gulf pipefish) were positively correlated with salinity, and three species (clown goby, rainwater killifish and goldspotted killifish) were negatively correlated with salinity. In the wet season, four species (blue crab, clown goby, rainwater killifish and goldspotted killifish) were negatively correlated with salinity, and two species (caridean shrimp and Gulf toadfish) were positively correlated with salinity. Salinity was a significant variable in explaining fish species richness in both seasons. Species richness was negatively correlated with salinity in the dry season and positively correlated with salinity in the wet season (Browder et al. 2009). Clown goby, gulf toadfish, and carideans were included as independent variables because of their hypothesized (and in some cases, statistically supported) interactions (as prey, predator, or competitor) with the dependent variable. Other independent variables such as water depth, canopy height, *Halodule* cover and mangrove fish contributed significantly to predicted epifaunal density.



Note: MG denotes mangrove, SG denotes seagrass.

FIGURE 9-78. NUMBER OF MULTIPLE REGRESSION MODELS PREDICTING SPECIES DENSITY IN WHICH THE INDICATED INDEPENDENT VARIABLE WAS A SIGNIFICANT (P≤0.10) COMPONENT FOR WET SEASON (TEN MODELS IN TOTAL) AND DRY SEASON (NINE MODELS IN TOTAL)

9.5.3.3 Summary

One focus of this monitoring is on distinguishing members of the community that have an affinity or reliance on freshwater inflow (i.e., estuarine species) so that these species can be followed separately from purely marine species in documenting changes in species richness after restoration is fully implemented. Building the number of samples that can be used to identify species halo-habitat is expected to increase the reliability of designations arising from this project.

Analytical results from the first four years of sampling were consistent with expected relationships between faunal distributions and salinity could be found in data acquired from sampling the shallow water shoreline epifauna. Results also supported prospects for finding ecological relationships between mangrove fishes and the epifauna.

Faunal density-weighted salinities may help to screen species for tendencies toward estuarine or marine habitat; however, the confidence limits on most species extended on both sides of the line of neutrality. More collections would help to reduce the confidence intervals and ensure the reliability of these results, making them more useful in refining the classification of species by halo-habitat.

The combination of general additive modeling and multiple regression modeling with splines produced stronger relationships with salinity for more species than the salinity-bin modeling. Inclusion of the other explaining variables may have helped to better define the relationships with salinity. Examining potential relationships between predator and prey, and between epibenthic prey and benthic habitat may be the key to understanding ecological relationships with salinity for all three trophic groups, because the relationship with salinity for any given species may be indirect rather than direct. Identifying interrelationships between shoreline fishes, epifauna and benthic habitat added another dimension to ecological characterization of the pre-restoration status of the shoreline epifauna in south Biscayne Bay.

These results indicate progress in:

1) development of a baseline for characterizing restoration effects in shallow open-water habitat along the south Biscayne Bay shoreline,

2) classification of species based on halo-habitat,

3) determination of relationships of faunal abundance with salinity and other environmental and habitat metrics, and

4) relating predator species in the mangrove fringe to potential prey species in the shallow open waters alongshore.

Browder et al. (2009) has made progress in developing a performance metric that assesses restoration progress on the basis of geographic nearness of sites with similar species composition and the development of an overall framework for conducting assessments.

9.5.4 Seagrass Fish and Invertebrate Assessment Network

9.5.4.1 Monitoring

The Seagrass Fish and Invertebrate Network (FIAN) combines systemwide faunal response by sampling across Biscayne and Florida Bays, and the southwest Florida coast. The project is tightly linked to the South Florida Seagrass Assessment Network conducted by the FWC locations to allow integration of the data of these complementary projects. The value of the FIAN data is further reinforced by the historic long-term (1984-1991, 1994-2007) sampling in Johnson Key Basin.

The FIAN network is characterizing the pre-restoration seagrass-associated fish, crab and shrimp communities twice annually at the end of the dry season (April/May) and at the end of the wet season (September/October). A 30-cell sampling grid defines each of the 19 monitoring locations shown in ***Figure 9-79***. Environmental measurements that include salinity, temperature, turbidity, and water and sediment depth are also recorded. . Also, cover, abundance and canopy height is estimated for seagrasses and algae. More details on this monitoring can be found in Browder et al. (2009).

Species collected include the pink shrimp, which serve as one of several biological indicator species for assessing the response of south Florida‟s southern estuaries to upstream changes in hydrology expected from Everglades restoration (RECOVER, 2005, 2006; Browder and Robblee, 2009). These estuaries provide critical nursery habitat for juvenile pink shrimp, a commercially and recreationally important species playing a pivotal ecological role in the estuarine food web of south Florida. Open yellow circles in ***Figure 9-79*** indicate monitoring locations aggregated as assessment areas for the pink shrimp performance measure; however, at this time only South Biscayne Bay and Johnson Key Basin have sufficiently robust historical datasets necessary to provide reliable comparisons to MAP data (i.e., baseline). MAP data will provide the baseline when restoration is fully implemented.

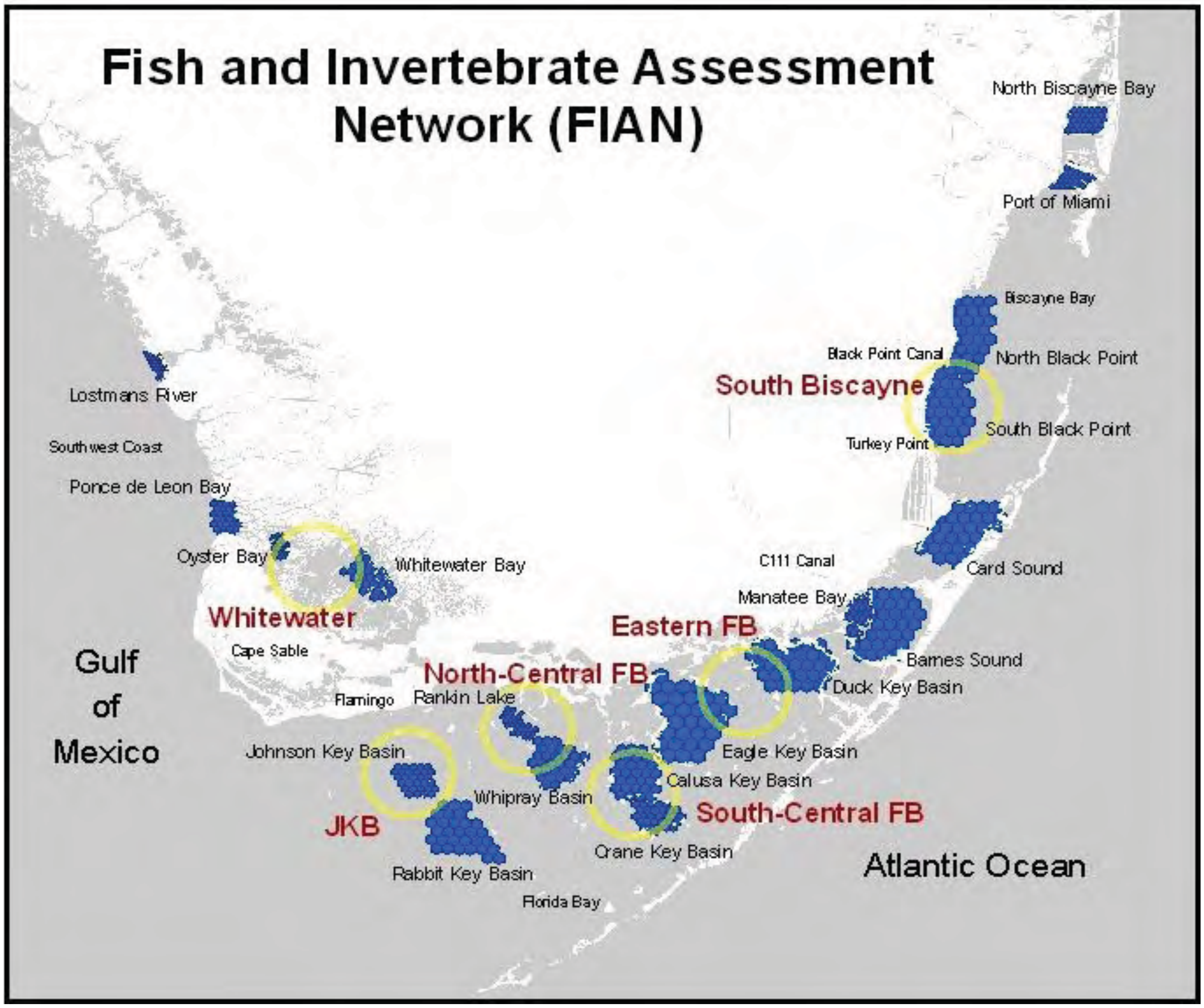


FIGURE 9-79. FISH AND INVERTEBRATE ASSESSMENT NETWORK MONITORING SITES

9.5.4.2 Results

Relative abundance of pink shrimp over the four years (2005-2009) of the Seagrass FIAN is shown in ***Figure 9-80***. The size of each pie graph represents the sum of average dry and wet season shrimp density. Shrimp density ranges from a maximum of 7.75 per square meter at Johnson Key Basin and a minimum of 0.06/ square meters at Duck Key Basin.

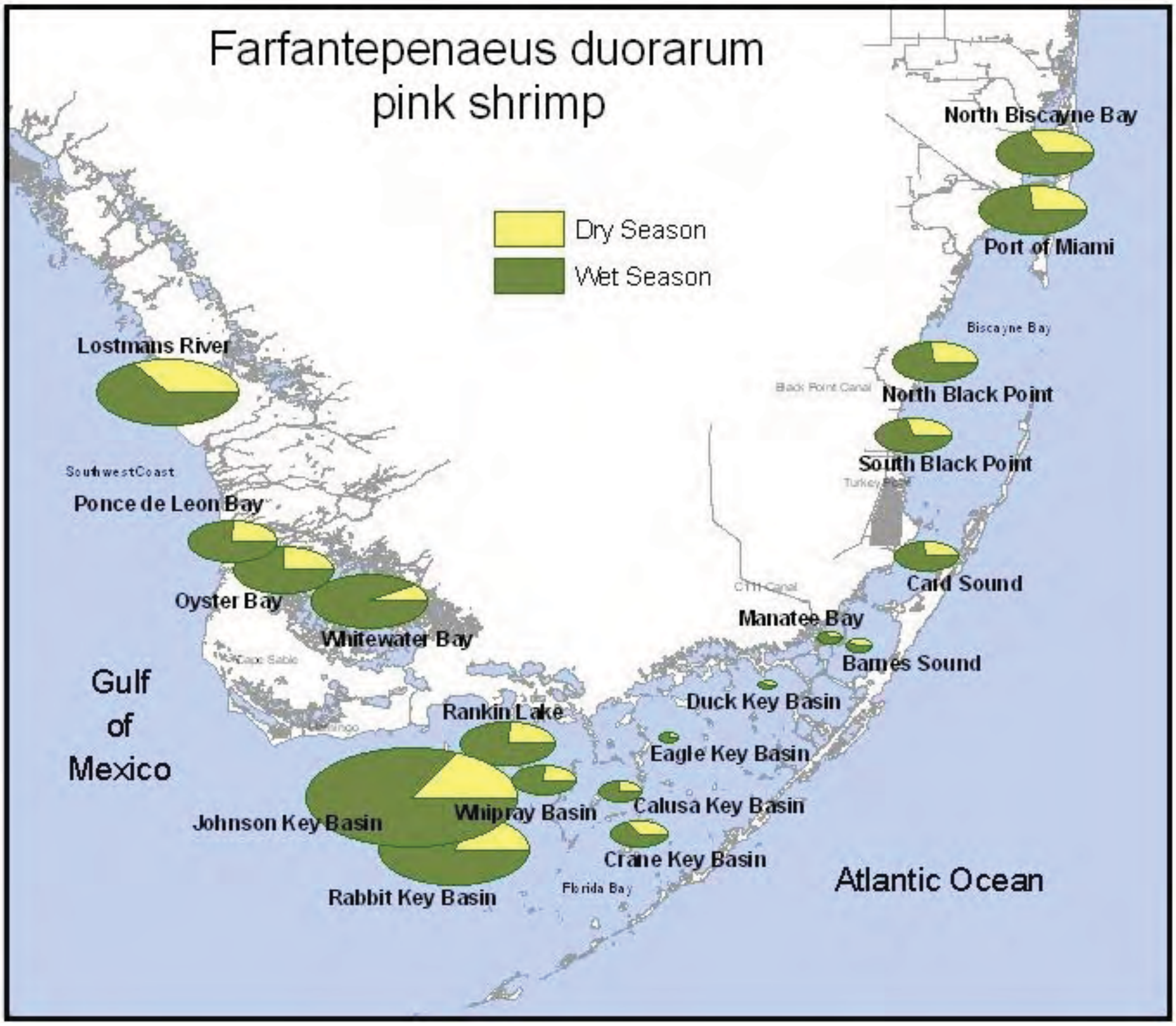


FIGURE 9-80. RELATIVE ABUNDANCE OF THE PINK SHRIMP 2005-2009

The relationship between pink shrimp and salinity suggest that water management affects inshore pink shrimp abundance. Laboratory trials with growth and survival of small juvenile pink shrimp from western Florida Bay were significantly related to salinity (Browder et al., 2002). Indices of pink shrimp abundance based on Tortugas fisheries data were significantly related to indices of freshwater flow from the Everglades (Browder, 1985; Sheridan, 1996). Meta-analyses of prominent fauna in Florida Bay found that pink shrimp were more closely correlated with salinity and seagrass than as many as 19 other species examined (Johnson et al., 2002a; 2002b; 2005). Based on the historical record from western Florida Bay, mean fall (September/October) densities of juvenile pink shrimp were significantly negatively correlated with salinity over the range of 28 to 45 psu (***Figure 9-81***). The FIAN investigators are examining data from the southwest coast to expand these relationships and understanding into the lower salinity range, i.e, below 28 psu, to be addressed in the next SSR.

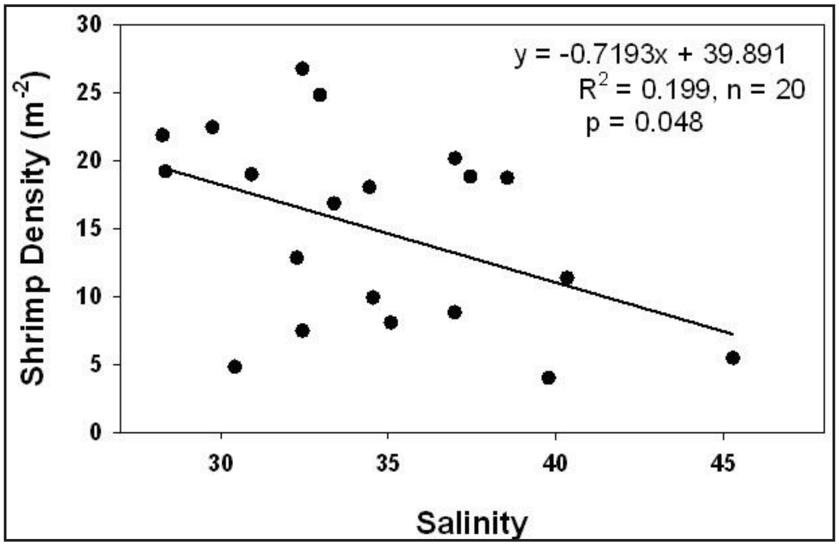


FIGURE 9-81. FALL MEAN JUVENILE PINK SHRIMP DENSITY PER SQUARE METER IN JOHNSON KEY BASIN IN RELATION TO SALINITY

Six assessment areas, each encompassing one or more FIAN monitoring locations, provide the spatial context for evaluating the status of pink shrimp in south Florida‟s nearshore waters (***Figure 9-79***). At present, annual assessment consists of comparison of spring and fall mean shrimp density in relation to available historical data for each area. Thresholds for scoring are based on quartiles of the distribution of available historical data from each assessment area. Values less than the first quartile are scored as 0 (poor response), values from the first to the third quartile are scored as 0.5 (neutral response), and values greater than the third quartile are scored as 1 (positive response). It is important to note that these scores are relative to conditions observed previously, and are not an indication of status being similar to either pre-development or expected post-restoration condition. Accordingly, these scores are only informative of current trends and current variation, and should not be interpreted otherwise.

Long-term historical datasets of juvenile pink shrimp density are available for Johnson Key Basin (n=20) for much of the period 1984 through 2007 and for South Biscayne Bay, i.e., Black Point to Turkey Point, (n=6) for the period 2002 through 2007 (Robblee and Browder, 2007). Historical data are limited to two to three-year periods of record for the other assessments areas and not considered sufficiently representative of inter-annual variation for reliable comparisons with FIAN estimates of pink shrimp density. The FIAN project is extending the pre-restoration baseline dataset for the 19 monitoring locations and the six assessment areas.

Data that are highly skewed and contain many zeros may complicate statistical analyses. The proportion of zeros and skewness in the pink shrimp data reflect the underlying distribution (homogeneous to heterogeneous) and ultimately affects the precision of pink shrimp density estimates. In ***Figure 9-82***, historical and FIAN data for the Johnson Key Basin and south Biscayne assessment areas were used to calculate occurrence and concentration of pink shrimp. The long-term historical record, represented by "H" in ***Figure 9-82***, is indicated with a solid circle plus or minus the 95 percent confidence interval for occurrence (portion of samples

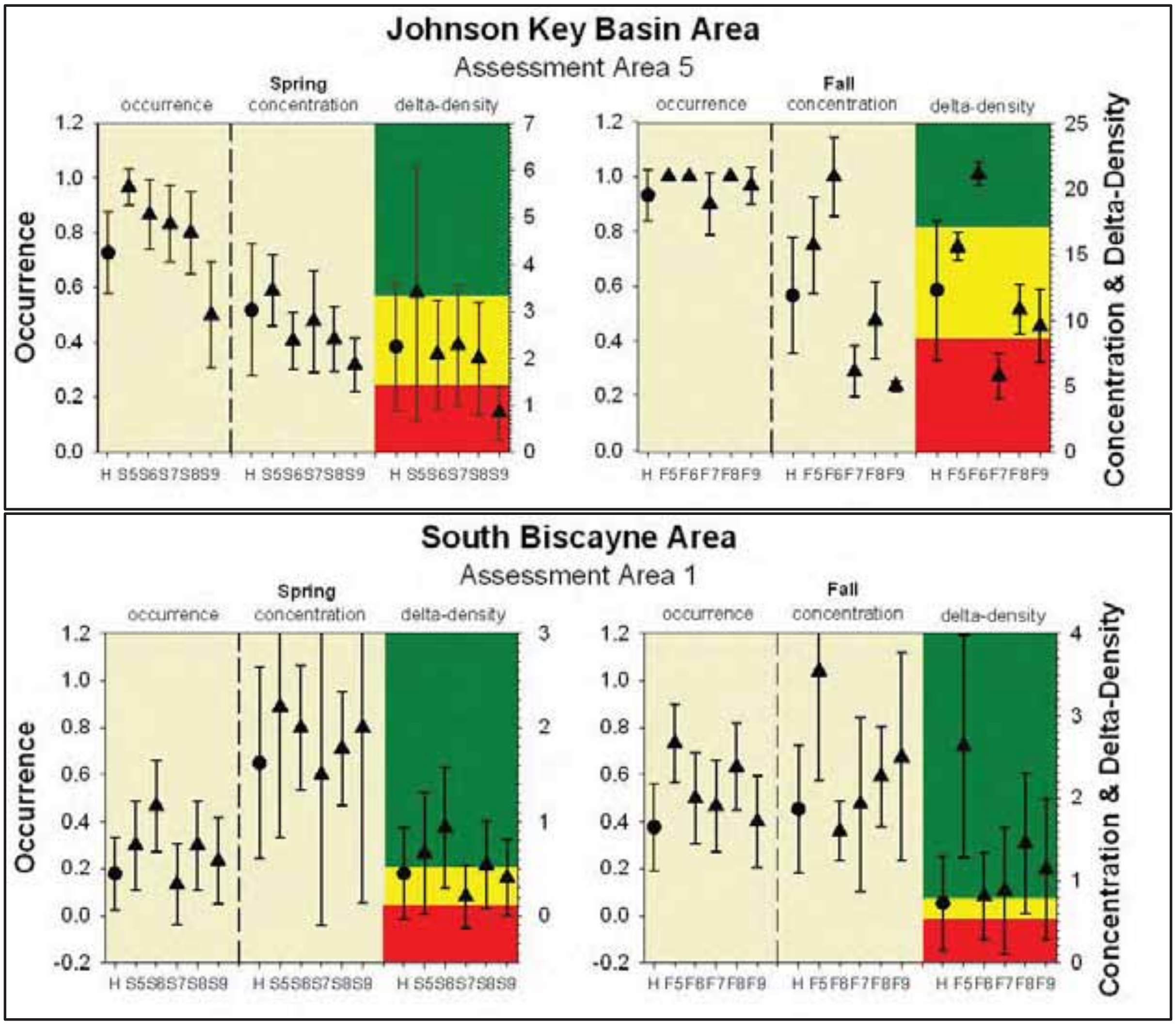


FIGURE 9-82. OCCURRENCE, CONCENTRATION AND DELTA DENSITY OF PINK SHRIMP 2005-2008 COMPARED WITH HISTORICAL RECORDS OF VARIATION IN ABUNDANCE FOR JOHNSON KEY BASIN AND SOUTH BISCAYNE BAY

positive for pink shrimp), concentration (in transformed density of shrimp among positive samples only), and delta density (occurrence x concentration). A solid triangle plus or minus the 95 percent confidence interval represents FIAN abundance estimates of pink shrimp for the dry season (S5-S8 on the x-axis) and wet season (F5-F8 on the x-axis). The status of pink shrimp is indicated by the overlap of delta density with the distribution of the historical record. Red indicates the status less than the first quartile, which is poor. Yellow indicates the first through third quartile, which is neutral. Green is greater than the third quartile, which is positive. The product of occurrence and concentration is delta density. When the concentration data is natural log-transformed, delta density is considered to be more representative of the data than a mean density estimate calculated with zero values included (Pennington 1983, Pennington 1986). When the concentration data is not natural log-transformed, then delta density exactly equals density with the zeros included. However, there is a better estimate of the confidence interval and there is a separate confidence interval for occurrence and for concentration.

Relative to available historical data, pink shrimp did well in 2005, with average scores among the six assessment areas of 0.75 for the dry season and 0.67 for the wet season. In contrast, 2007 was an extremely poor year with scores among the six response areas averaging only 0.33 for both seasons (***Table 9-17***). On average, 2006 and 2008 were intermediate years. For the five years of FIAN seasonal scoring, mean dry and wet season densities are highest in 2005. By comparison 2006 thru 2008 were poorer years, with the poorest performance overall in the fall of 2009. There exists a significant (p=0.02, Kendall tau) five-year overall downward trend in the wet season, of particular concern because the wet season is typically the period of peak shrimp density on the nursery grounds. Scoring among assessment areas is highly variable suggesting the importance of differences in salinity, habitat and post-larval accessibility among them.

TABLE 9-17. STATUS OF THE PINK SHRIMP 2005-2008 RELATIVE TO AVAILABLE HISTORICAL DATA

Assessment Areas	Spring (Dry Season)						Fall (Wet Season)					
	2005	2006	2007	2008	2009	mean	2005	2006	2007	2008	2009	mean
South Biscayne	1	1	0.5	1	0.5	0.80	1	1	1	1	1	1.00
Eastern FB*	0	0.5	0	0	1	0.30	1	0	1	0.5	0	0.50
North-Central FB*	1	1	0.5	0.5	0	0.60	1	0	0	0	0	0.20
South-Central FB*	0.5	0	0	0.5	0	0.20	0	0	0	0	0	0.00
JKB	1	0.5	0.5	0.5	0	0.50	0.5	1	0	0.5	0	0.40
Whitewater*	1	0	0.5	1	1	0.70	0.5	0.5	0	0	0	0.20
mean	0.75	0.50	0.33	0.58	0.42	0.53	0.67	0.42	0.33	0.33	0.17	0.47

NOTE: Asterisk denotes limited time series; see text

Scoring the annual status of the pink shrimp and interpreting patterns in this indicator show great promise for further understanding of nursery processes in the Southern Coastal Systems and for assessing responses to Everglades restoration. However, because historical data are limited for those assessment areas indicated with an asterisk in ***Table 9-17***, caution should be exercised in their interpretation until FIAN extends the pre-restoration time series. For further details, see the annual reports submitted by Robblee and Browder (2006, 2007, 2008, 2009).

9.5.5 Florida Bay Juvenile Sportfish

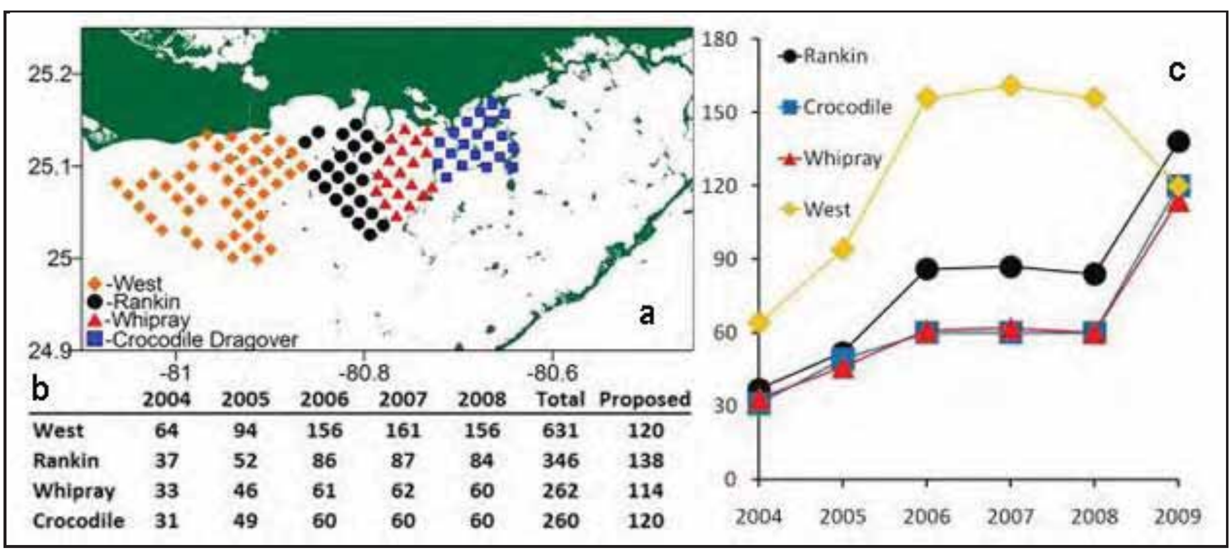
9.5.5.1 Monitoring

The Florida Bay juvenile sportfish monitoring focuses on important sportfish species, including spotted seatrout (*Cynoscion nebulosus*), where visual surveys are impractical due to highly variable water clarity. Thus, this and the Biscayne Bay mangrove fish sampling fulfill similar roles in their respective systems. Juvenile sportfish data collection in Florida Bay began in the mid-1980s, before the seagrass die-off occurred in Florida Bay.

Spotted seatrout by and large spend their entire life history within Florida Bay and the distribution of juvenile spotted seatrout has been observed to vary in response to salinity conditions (Thayer et al., 1999), making them an ideal indicator to assess Florida Bay‟s response to water management changes that will occur as restoration is implemented. Monitoring of juvenile spotted seatrout employs a stratified random sampling design focused on the northern half of western and central Florida Bay (***Figure 9-83***). Sampling takes place from June to November to coincide with the peak of spotted seatrout spawning in Florida Bay. Historically, this sampling was distributed by subregional area; however, two recently completed power analyses produced complementary results suggesting the sampling effort should be redistributed. Future sampling would distribute samples more equally amongst subregions and increase overall sampling effort to enable the capability of detecting at least 20 percent changes (alpha=0.05; beta=0.80) in a five-year time block. For more details see Kelble et al. (2009).

9.5.5.2 Results

Juvenile sportfish results obtained thus far display a wide degree of temporal and spatial variability (***Figure 9-85***). Data are by year during the MAP study (2004-2008) and grouped as one category for all pre-MAP data from 1984 to1985 and 1994. The west subregion appears to support the largest juvenile spotted seatrout population. The frequency of occurrence in the west consistently exceeded 11 percent each year and was 25.2 percent for the entire MAP study (2004-2008). Whipray had the next highest frequency of occurrence (22.4%), followed by Rankin (13.2%), and Crocodile Dragover (4.2%). Overall, 2006 had the highest frequency of occurrence and density of juvenile spotted seatrout; whereas 2008 had the lowest with no juvenile spotted seatrout observed in Crocodile Dragover or Rankin and only one observed in Whipray. Interestingly, 2008 had the highest mean salinities for all subregions (***Figure 9-85***) and the lowest populations of juvenile spotted seatrout in all subregions except the west, strongly suggesting an inverse relationship exists at least at high salinities.



	2004	2005	2006	2007	2008	Total	Proposed
West	64	94	156	161	156	631	120
Rankin	37	52	86	87	84	346	138
Whipray	33	46	61	62	60	262	114
Crocodile	31	49	60	60	60	260	120

FIGURE 9-83. SPATIAL AND TEMPORAL DISTRIBUTION OF JUVENILE SPOTTED SEATROUT

Note: a) location of all potential sampling locations
b) number of stations sampled in each subregion each year, including the total number of samples collected thus far and the proposed number of samples for this year
c) temporal sampling distribution in each subregion and the proposed 2009 sampling effort

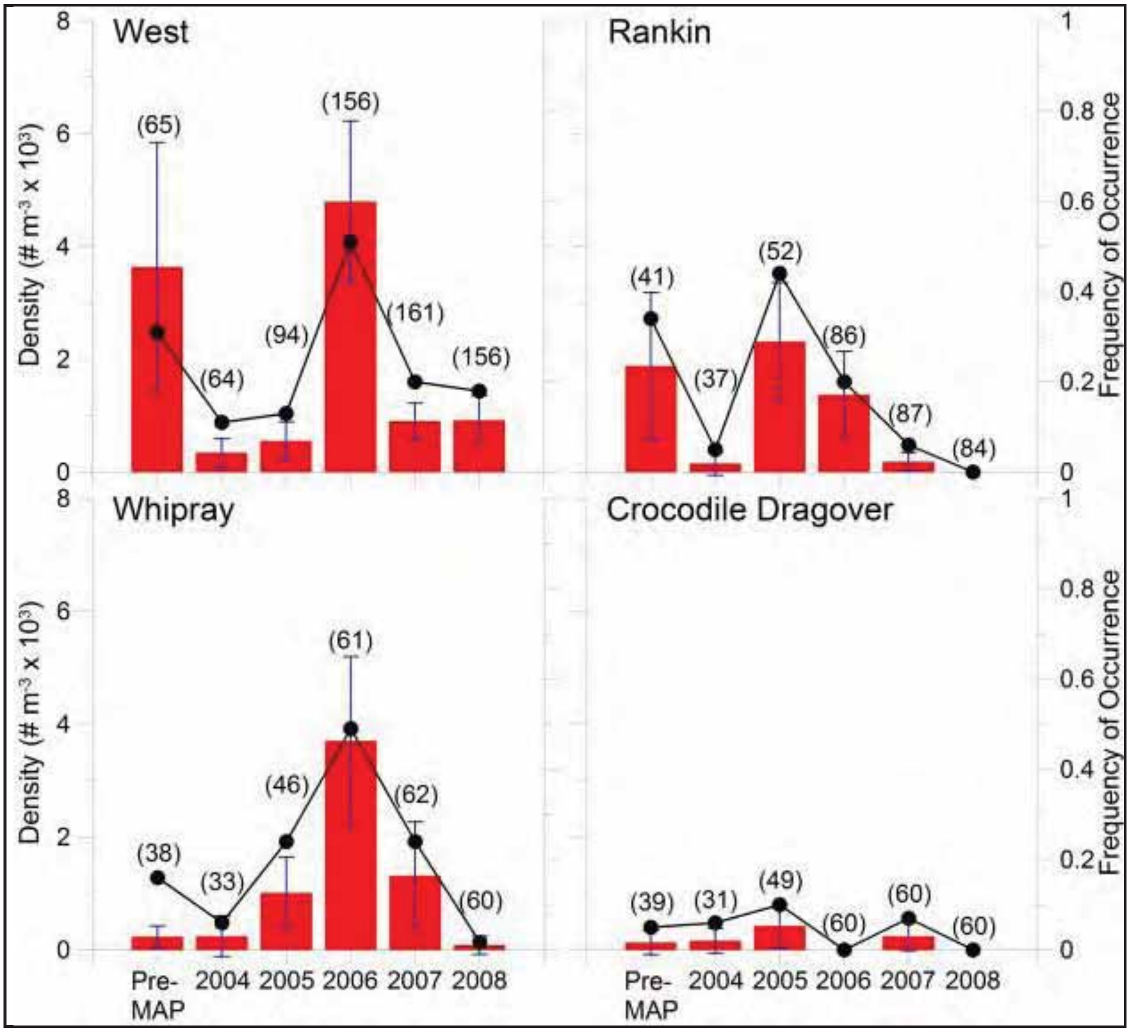


FIGURE 9-84. JUVENILE SPOTTED SEATROUT DENSITY (RED BARS) AND (BLACK DOTS) IN EACH OF THE FOUR SUBREGIONS OF FLORIDA BAY

Note: Number of samples are shown in parentheses

Statistical analyses and a general linear model further analyzed the relationship between juvenile spotted seatrout and salinity. The general linear model examined the baywide distribution of juvenile spotted seatrout and seagrass biomass, temperature and salinity as independent variables delineated by quartiles. The best general linear model incorporated all three variables, including salinity for which an inverse relationship emerged (***Figure 9-85***). A significant inverse linear relationship between salinity and juvenile spotted seatrout frequency of occurrence was observed in the field data for all of the subregions, except the west, where there was no significant regression (***Figure 9-85***). These results suggest that within the central area of Florida Bay the juvenile spotted seatrout population is inversely related to salinity, but this may not be the case in the west region, perhaps due to varying contributions by multiple spawning grounds, thereby confounding salinity effects. Based on these results, if restoration effectively mitigates hypersalinity in central Florida Bay, this improvement is likely to increase the population of juvenile spotted seatrout. See the 2009 annual report submitted by Kelble et al. (2009) for a broader and more detailed description of results of Florida Bay juvenile sportfish monitoring.

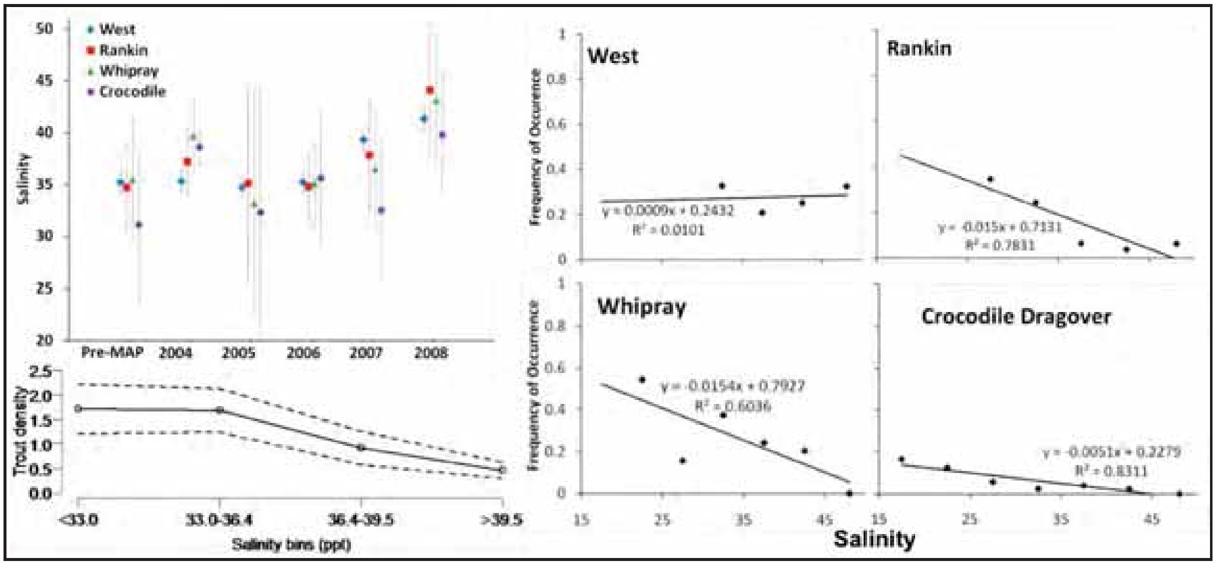


FIGURE 9-85. GRAPHS ILLUSTRATING THE DEPENDENCE OF JUVENILE SPOTTED SEATROUT ON SALINITY

Note: a) mean and standard deviation for salinity for each year during the study period in each subregion,
b) relationship between salinity and density of juvenile spotted seatrout calculated by the general linear model
c) linear relationship between salinity and frequency of occurrence for juvenile spotted seatrout in each subregion

9.5.5.3 Summary

Deciphering the effects that restoration may have on the distribution and abundance of fish and invertebrates in the Southern Coastal Systems will prove crucial to provide the science-based guidance to ensure and maximize benefit. Accordingly, concurrent and complementary faunal monitoring efforts are providing data relevant to assessing the suite of nearshore community hypotheses. The spatio-temporal overlap of the faunal sampling efforts allow characterization of species assemblages across trophic levels, their variation in space and time, and relationships with habitat.

Power analyses performed for these assessments have shown the importance of having an uninterrupted time series for more than five years before and after implementation of a restoration project for optimal detection of faunal responses to restoration impacts, such as altered salinity regime, on the system (Serafy et al., 2009). This suggests that interruption of these time series when projects are being undertaken that are likely to affect this ecosystem in the next five years, such as the Biscayne Bay Coastal Wetlands, C-111 Spreader Canal, and Modwaters, would greatly reduce the ability to evaluate the effect of these projects on Southern Coastal System fauna and thus increase the chance that the public perceives them as unsuccessful. In conclusion, performance measures based on monitoring the rich resources of the Southern Coastal Systems, at the downstream end of the water management system are needed to determine restoration success.

9.6 OYSTER HYPOTHESIS CLUSTER

9.6.1 Introduction and Background

Water management practices in south Florida have significantly altered patterns of water delivery and quality to the Southern Coastal Systems resulting in altered salinity regimes in those coastal waters. These salinity changes have reduced or eliminated many eastern oyster (*Crassostrea virginica*) reef areas and have impacted both the timing and extent of oyster reproduction (Berrigan et al., 1991). The diverse floral and faunal community associated with oyster reefs has been correspondingly impacted. Restoration of more natural freshwater inflows to estuaries and the corresponding reestablishment of mesohaline and polyhaline salinities resulting from restoration should provide appropriate habitat conditions to restore healthy oyster beds and associated communities in the Southern Coastal Systems (***Figure 9-86***).

9.6.2 Monitoring

Oyster monitoring in the Southern Coastal Systems has been conducted in Biscayne Bay and in the lower southwest coast estuaries from Whitewater Bay to the Ten Thousand Islands region (***Figure 9-87***), with the goal of documenting changes in oyster health, recruitment and reef development. Monitoring typically examines eight aspects of oyster ecology including 1) spatial and size distribution patterns of adult oysters, 2) physiological condition as measured by condition index, 3) reproduction, 4) recruitment, 5) juvenile growth and survival, 6) prevalence and intensity of oyster diseases, 7) oyster living density, and 8) oyster reef distribution. Detailed descriptions of methods can be found in Arnold et al. (2008), Volety et al. (2008) and Volety and Savarese (2001).

9.6.2.1 Biscayne Bay

Oyster community condition was evaluated at three sites in Biscayne Bay by the Florida Wildlife Research Institute from 2005 through 2007. At the end of 2007, it was determined, due to the paucity of oysters in the study area, sufficient baseline information had been collected and monitoring was terminated until restoration begins. Further details on this monitoring can be found in Arnold et al. (2008).

9.6.2.2 Everglades National Park

Oysters were monitored along western Everglades National Park during a study conducted from 2006 to 2008 by Florida Gulf Coast University. Oyster status was examined in Chatham, Lostman‟s and Broad Rivers, and in Whitewater, Oyster and Ponce de Leon Bays (***Figure 9-87***). Lostman‟s estuary was monitored thoroughly for oyster measures of oyster physiology and ecology. In the other western Everglades National Park estuaries, only the mapping of oyster reefs occurred. Details can be found in Volety et al. (2008).

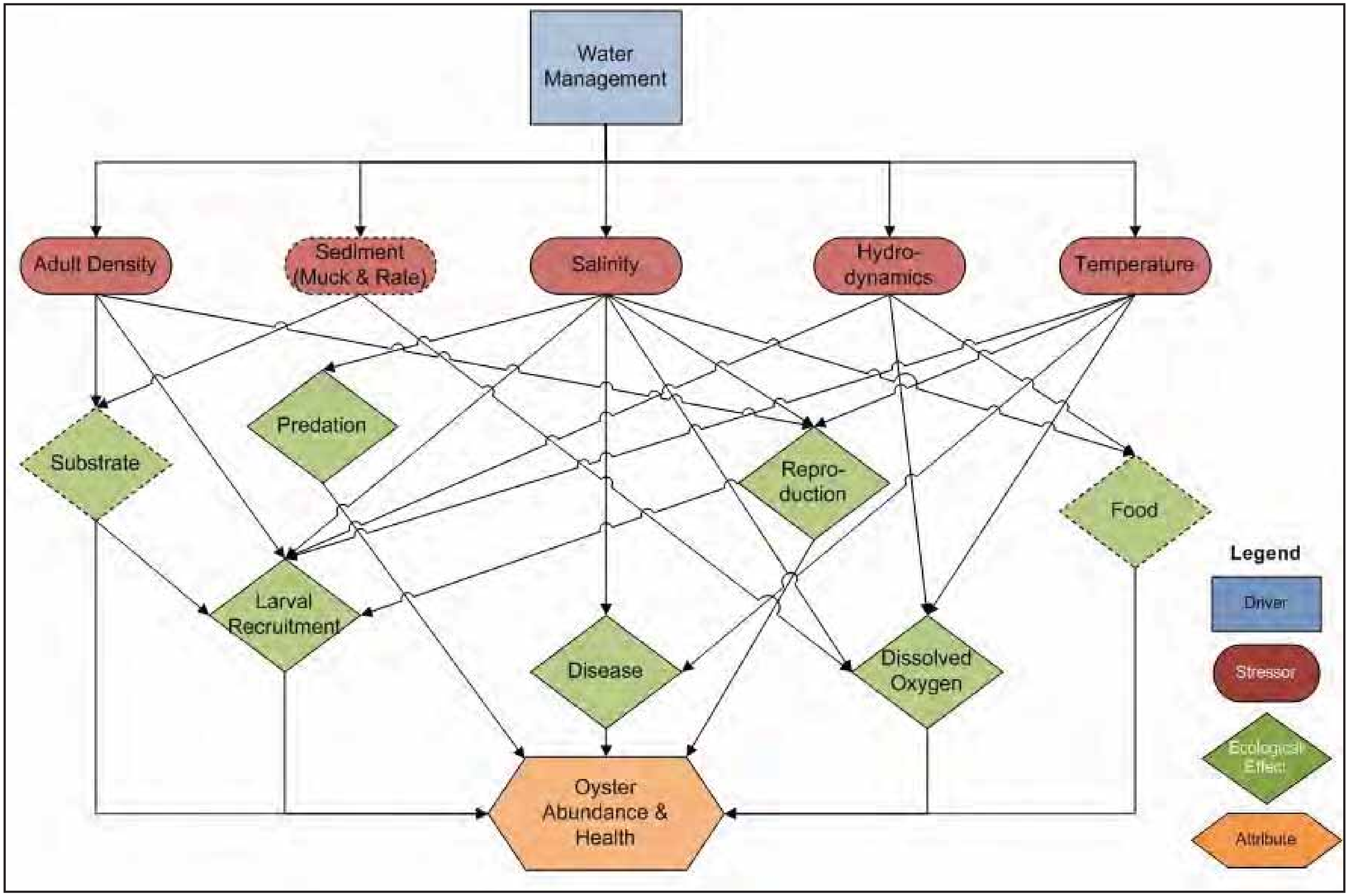


FIGURE 9-86. CONCEPTUAL ECOLOGICAL MODEL FOR OYSTERS IN SOUTHERN COASTAL SYSTEMS

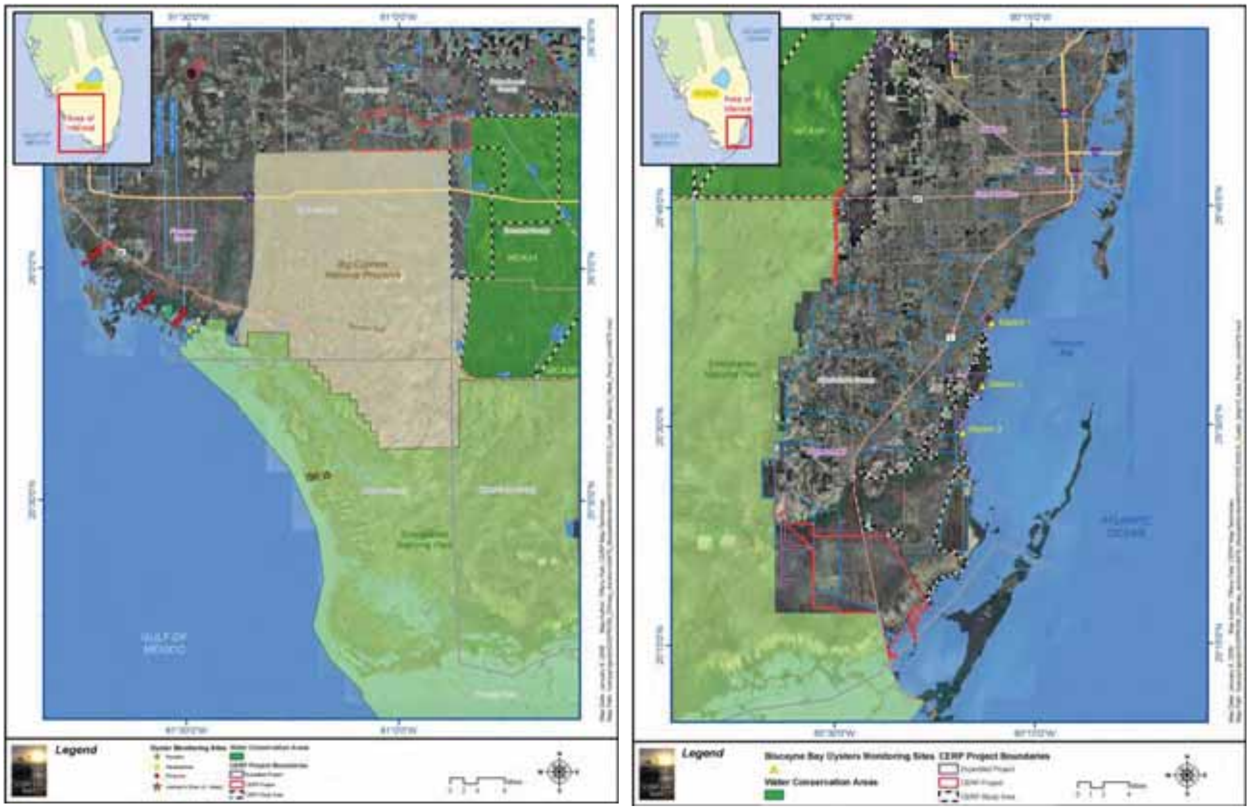


FIGURE 9-87. OYSTER SAMPLING LOCATIONS IN THE SOUTHERN COASTAL SYSTEMS

9.6.2.3 Ten Thousand Islands

Oyster communities were examined in three estuaries, Blackwater River, Faka Union Bay and Henderson Creek, in the Ten Thousand Island area (***Figure 9-87***) during a Florida Gulf Coast University study conducted from 1999 to 2001. In two subsequent studies, oyster health and habitat quality was evaluated in Fakahatchee and Pumpkin Bays (Savarese, et al., 2004, Savarese and Volety 2008).

9.6.3 Results

9.6.3.1 Biscayne Bay

9.6.3.1.1 Distribution Patterns of Adult Oysters

Biscayne Bay currently presents poor conditions for oyster reef development and only one substantial population was documented (on the seawall at the Deering Estate). Oysters associated with reef structure were absent throughout the three-year study period, although they were found attached to red mangrove prop roots along the shoreline. Oysters living on mangrove roots were not quantified due to sampling protocol incompatability, although their presence should provide a reproductive source once the salinity regime improves.

9.6.3.1.2 Physiological Condition and Reproductive Development

Due to the small number of samples, it was not possible to reliably discern reproductive development or condition index in oysters collected from Biscayne Bay.

9.6.3.1.3 Spat recruitment

Oyster spat recruitment monitoring was conducted throughout the 2005 to 2007 monitoring period. Little to no larval oyster recruitment was recorded in Biscayne Bay during the study period. However, the oyster spat that were documented occurred most commonly in late fall. Peak abundances were very low with a maximum of 0.18 ($\pm$ 0.74) occurring in November 2007. Although the substrate in Biscayne Bay along the western shore is generally soft sediment, there are significant deposits of old oyster shell at the mouths of primary creek beds where spat might attach.

9.6.3.1.4 Juvenile growth

Juvenile oysters were placed in closed and open sided cages at each station in September 2005, July 2006 and May 2007. Growth results were variable. Mean growth was approximately 4.2, 2.7 and 2.1 millimeter (mm) per month for 2005, 2006 and 2007, respectively. Growth was greater for oysters planted on the outside of cages in 2005 compared to individuals planted inside cages, but less in 2006 and 2007.

9.6.3.1.5 Disease

Live adult oysters were collected in four separate months in 2005, not at all in 2006, and quarterly in 2007 for determination of the prevalence and intensity of the oyster diseases *Perkinsus marinus* (dermo) and *Haplosporidium nelsoni* (MSX). The mean infection level remained below 1.0 throughout the sampling period, indicating that sampled oysters were only slightly infected with the parasite. The prevalence of dermo was relatively low with the highest level of dermo during one month being a 40 percent infection rate. None of the oysters collected from the Biscayne Bay study area were infected with MSX.

9.6.3.1.6 Live Oyster Survey

During fall 2008, Biscayne National Park, under contract with the SFWMD, examined a subset of sites for the presence of live oysters that were previously identified as locations where oyster shell had been reported in previous studies. Live oysters were found at 12 of the 15 sites, but density was sparse except at one site where they were classified as abundant on mangrove roots and abandoned traps. For further details see Bellmund et al. (2009).

9.6.3.2 Everglades National Park

9.6.3.2.1 Distribution Patterns of Adult Oysters

Surveys conducted along the western margin of Everglades National Park in 2007 revealed oyster reefs throughout the Chatham River Estuary, though reefs were more frequently documented at the bay‟s mouth. Reefs in Lostman‟s and Broad Rivers are located almost exclusively at the mouths. This difference in reef spatial distribution (reefs found further upstream in Chatham and not similarly in Lostman‟s and Broad Rivers) suggests that a comparatively more favorable salinity regime exists in upstream Chatham River Estuary. Interestingly, oysters are absent from mangrove prop roots within the interior bays of Lostman‟s and Broad, also suggesting an inappropriate salinity stability regime, rather than substrate limitation, as the controlling factor of oyster distribution. Reef mapping in Whitewater, Oyster and Ponce de Leon Bays revealed intertidal oyster reefs were mostly absent, with only three intertidal reefs identified. Oysters were commonly found on red mangrove prop roots within Whitewater Bay, similar to what was found in Biscayne Bay. It should be noted that although subtidal oyster reefs were not detected within these bays, the survey methods employed were not exhaustive and their existence cannot be ruled out. Density of living oysters measured in Lostman‟s River during the dry season in 2007 was higher in the middle of the estuary than that at upstream and downstream locations.

Physiological Condition and Reproductive Development

Condition index of oysters generally increased from upstream locations to downstream locations, a trend that is observed in other southwest Florida estuaries. Condition index of oysters ranged from 1.51 to 3.6 at various locations in November and increased to a range of 2.79 to 5.39 in May. This was followed by a sharp decrease in condition index ranging from 0.98 to 1.7, possibly due to spawning activity as shedding of tissue mass in the form of gametes will result in lower condition index. Oysters in southwest Florida estuaries typically accumulate energy reserves in the cooler winter months and spawn during March through May. This trend of decreased condition index occurring in or around May is observed in other southwest Florida estuaries (Volety et al., 2008).

9.6.3.2.2 Spat Recruitment

Spat recruitment was first observed on the shell strings between May and July. Mean spat per shell in July at various locations ranged between 0.57 and 1.21, a value that is low compared to the Estero Bay or Caloosahatchee River Estuaries. The 2007-2008 water year was one of the driest years on record, resulting in higher salinities than in years with normal rainfall. The observed low spat recruitment may have been due to osmoregulation related stress as well as due to increased predation as oyster predators are more prevalent in higher salinities.

9.6.3.2.3 Juvenile Growth

Two hundred juvenile oysters were deployed in wire mesh bags at three locations in Lostman‟s River. Juvenile oysters showed excellent growth and survival at all locations. Mean survival of juvenile oysters ranged between 36 and 68 percent. It should be noted that since these juvenile oysters are deployed in wire-mesh cages, mortality is likely water quality related because most predation-related mortality is excluded.

9.6.3.2.4 Disease

Fifteen oysters were collected bi-monthly at three sites situated along the axis of the Lostman‟s River beginning in November 2007 to determine prevalence and intensity of dermo infections. Condition index was also determined. Dermo prevalence at the three stations ranged between 93 and 100 percent in November and decreased to between 53 and 73 percent by January 2008. Incidence of dermo increased from 67 to 100 percent at various sites by July 2008, possibly due to higher temperatures and salinities. Disease intensity ranged from 2.00 to 3.27 on a scale of 0 to 5 in November 2007 and decreased to between 0.8 and 1.73 in January 2008. From March 2008 to the end of the study, disease intensity was below 1.5.

9.6.3.3 Ten Thousand Islands

9.6.3.3.1 Distribution Patterns of Adult Oysters

In Faka Union Bay, a system that receives excessive freshwater volumes in pulsed releases during the rainy season, oyster reefs are relatively scarce in number and small in aerial extent when compared to the other two estuaries. In Faka Union Bay the distribution of reefs, the regions of maximum living density, and the foci of maximum oyster productivity are displaced seaward relative to the estuaries of Henderson Creek, Blackwater River, Pumpkin Bay, and Fakahatchee Bay. Except for the outermost region, Faka Union Bay had the lowest living density of the three estuaries. Henderson Creek, which receives greater nutrient input and pulses of fresh water due to water quality and a simplistic weir design, respectively, had higher living oyster densities, but the population was skewed heavily toward smaller individuals, suggesting juvenile mortality as a probable causative agent. Blackwater River and Pumpkin Bay exhibit greater productivity further upstream in the watersheds, presumably a result of changes in the availability and delivery of freshwater attributable to the Faka Union canal operation under pre-restored condition. Faka Union Bay has relict oyster reefs - now essentially occurring as dead oyster substrate - within the lower reaches of the original Faka Union River. Mortality is presumably a consequence of the extreme freshets caused by Faka Union canal operation.

9.6.3.3.2 Physiological Condition and Reproductive Development

Growth rates and condition index of oysters were consistently higher at higher salinity locations in the Ten Thousand Islands estuaries. Faka Union Bay had the lowest condition indices of the three estuaries and exhibits a greater variance in biomass, indicating a wider age distribution. Henderson Creek oyster populations have higher mean productivities than the other estuaries studied. Productivity in Faka Union Bay and Henderson Creek decreased from June to February, but the opposite was true in Blackwater River, suggesting that excess fresh water during the wet season in the more altered systems may be the cause. In the Savarese et al. (2003) study, oyster condition in Faka Union Bay was found to be significantly lower than in relatively pristine Estero Bay. Oysters from the most upstream site in the Faka Union Bay had a significantly higher condition index when compared to those in the middle and downstream sites, while the opposite was observed in Estero Bay.

9.6.3.3.3 Spat Recruitment

Spat recruitment was highest in Henderson Creek, followed by Blackwater River and Faka Union Bay. Lack of significant adult populations, combined with intense salinity stress during the wet season, may have contributed to the low recruitment rates in Faka Union Bay. Recruitment was low in the upstream end of all three areas, suggesting that salinity may kill juvenile oysters in these areas. As was found in Biscayne Bay, red mangrove prop roots provide a firm substrate for oyster recruitment, which is utilized throughout the Ten Thousand Islands region.

9.6.3.3.4 Juvenile Growth

The juvenile oysters deployed in wire mesh bags in upstream areas of Faka Union Bay and Henderson Creek experienced severe mortality due to freshwater inflow in late summer months. This was particularly severe in Faka Union Bay where over 90 percent mortality was recorded in the first month after deployment. Those juvenile oysters that survived grew at all locations in the three estuaries, with significant differences in juvenile growth observed among locations, among estuaries and among sampling months. Surprisingly, the highest growth rate was observed at the upstream locations, which likely occurred during the dry season when salinities were moderate.

9.6.3.3.5 Disease

Mean infections in oysters from all locations in all the estuaries were low. Near the downstream end of these estuaries, which has more natural flow patterns, small-sized oysters with few older, large individuals were prevalent. This may be a result of a greater occurrence of dermo and/or predation, which was not studied, within environments of higher salinity. Higher salinity and temperature increases dermo prevalence in adult oysters. Oyster disease was more prevalent in Blackwater River, an estuary that has consistently higher salinity than Faka Union Bay and Henderson Creek. In the follow-up study (Savarese et al., 2003), disease intensity in Faka Union Bay was found to be significantly higher than in Estero Bay, a less impacted estuary, but disease prevalence was similar in the two estuaries. Temperatures and salinities in southwest Florida estuaries are acting antagonistically in keeping dermo infections low and masking obvious trends due to temperature and salinity.

Figure 9-88, Figure 9-89 and ***Figure 9-90*** shows infection intensity of *P. marinus* in oysters from the Faka Union estuary. Ten to twenty oysters were collected from each sampling location monthly and *P. marinus* infection intensity assessed according to Ray (1954). Sampling locations 1 through 3 are from upstream to downstream.

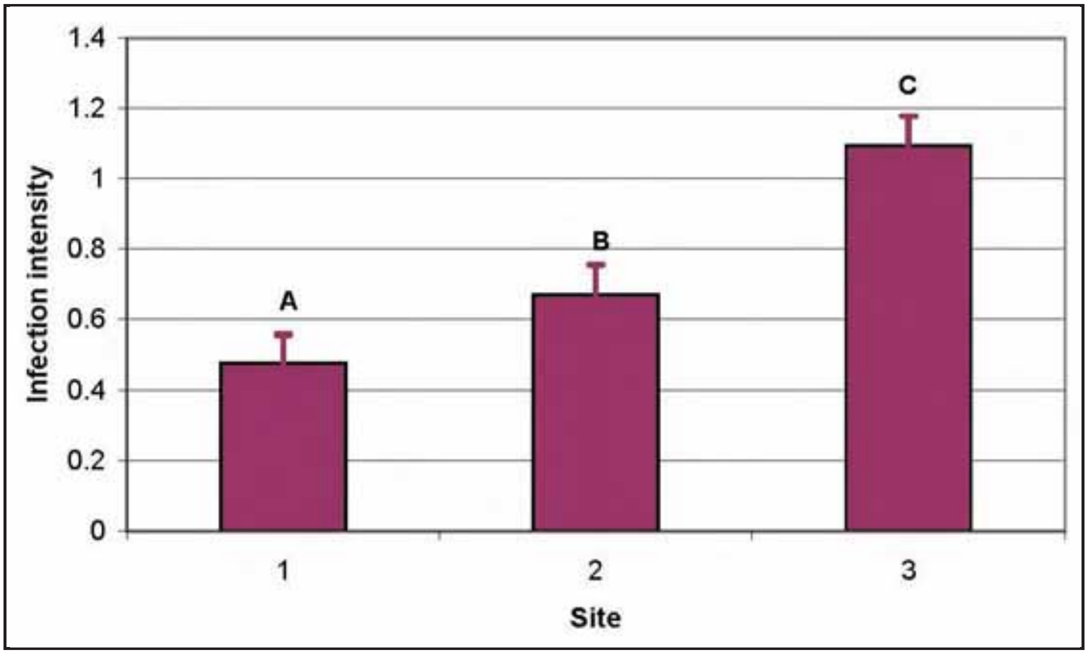


FIGURE 9-88. OVERALL INFECTION INTENSITY OF *P. MARINUS* IN OYSTERS (N=208-219) FROM EACH SITE IN THE FAKA UNION BAY ESTUARY

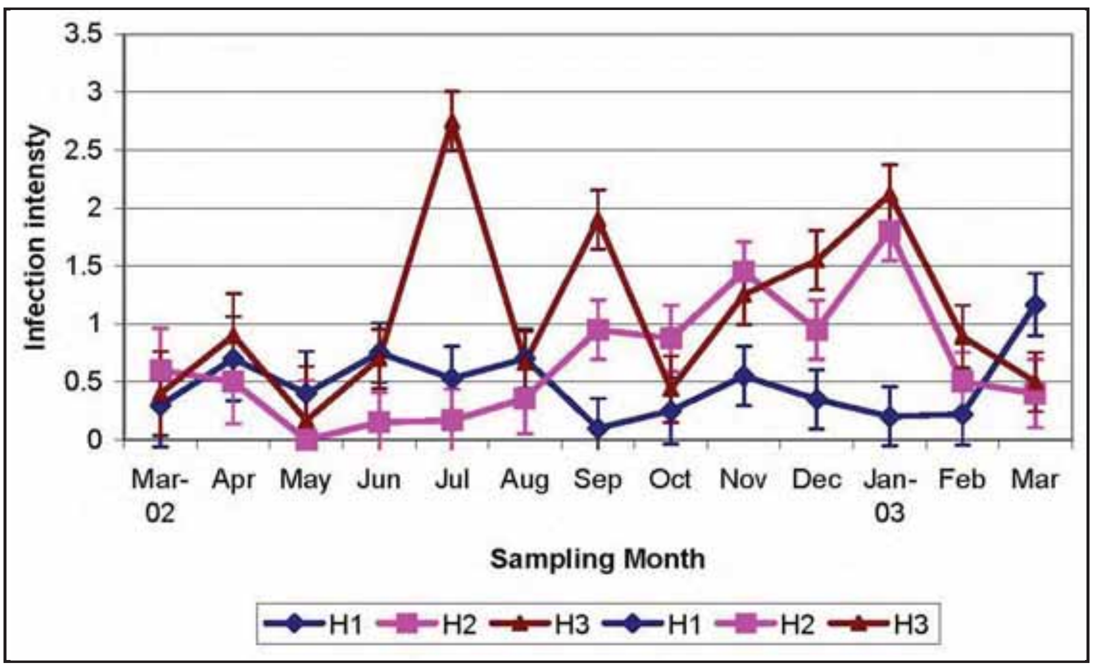


FIGURE 9-89. MONTHLY INFECTION INTENSITY OF *P. MARINUS* IN OYSTERS FROM EACH SITE IN FAKA UNION BAY ESTUARY

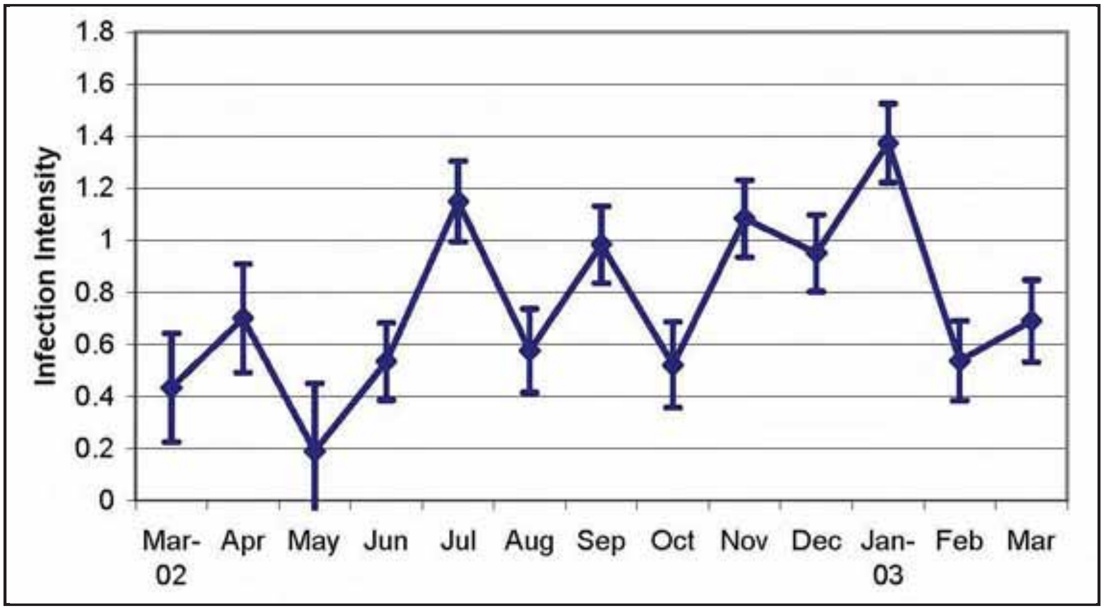


FIGURE 9-90. MONTHLY INFECTION IN INTENSITY OF *P. MARINUS* IN OYSTERS FROM FAKA UNION BAY ESTUARY FOR ALL SIGHTS COMBINED

9.6.4 Summary

Monitoring results revealed a scarcity of living oyster reefs in Biscayne Bay and some areas along the southwest Florida coast; however, evidence exists that this is a recent phenomenon. Meeder et al. (1999) found that oyster reefs were at one time common in the myriad of creeks along the western shore of central and south Biscayne Bay. Volety and Savarese (2001) noted that relic reefs in the upper reaches of Faka Union Bay range in age from 34 to 156 years, suggesting that active growth existed prior to the construction and operation of the Faka Union Canal. Savarese et al. (2004) documented relic oyster reefs approximately one meter in thickness along the entire estuarine axis of Blackwater, Pumpkin, Faka Union and Fakahatchee Bays using sediment cores. These studies show conditions were previously suitable for this keystone species in the recent past where living oyster reefs are no longer found.

Restoration projects such as the Biscayne Bay Coastal Wetlands and Picayune Strand Restoration Projects endeavor to reestablish salinity regimes favorable to reef development and productivity. Volety and Savarese (2001) noted that restored salinity patterns in mid-Faka Union Bay might emulate the pattern observed in the relatively undisturbed and productive oyster reef area of mid-Blackwater River. Geomorphologic characteristics indicate that mid-Faka Union Bay presents the opportunity for significant oyster expansion should restoration goals be realized.

The Biscayne National Park live oyster survey found live oysters on prop roots and on other elevated structure, and not on the bottom where the historic oyster shell beds would have offered suitable substrate. This suggests that the freshwater lens that forms in the upper layer of water column creates conditions sufficiently acceptable to sustain live oyster. Unverified anecdotal reports that oysters may also be present on mangrove prop roots in Madeira Bay and Faka Union Bay hints that the Biscayne Bay findings may not be unique, but perhaps indicative of widespread occurrence of a few oysters scattered throughout the Southern Coastal System mangrove fringe wherever conditions are sufficient to permit their survival. It follows that where salinity regimes are sufficiently restored, these scattered live oysters could provide the spat necessary to repopulate the newly formed estuarine habitat.

A HSI model is currently being developed and validated for oysters in the Caloosahatchee River (Volety et al., 2005). It is anticipated that this model will be adapted for application to the oyster populations in the Southern Coastal Systems, which should provide valuable information regarding restoration progress toward establishing a sustainable estuarine environment. Additional monitoring data is required to evaluate trends in oyster communities and to determine relationships between oyster parameters and their ecological drivers, such as natural and anthropogenic changes to salinity. Presently, the depth of data on oysters in the Southern Coastal Systems is not sufficient to validate the HSI for these areas.

9.7 PREDATOR-PREY INTERACTIONS OF WADING BIRDS AND AQUATIC FAUNA FORAGE BASE HYPOTHESIS CLUSTER

9.7.1 Introduction and Background

As with the Greater Everglades, the success of wading bird foraging in Picayune Strand is influenced by wet season population densities and dry season concentrations of marsh fishes, crayfishes, and other aquatic prey organisms (***Figure 9-91***). Wet season density and size structure of aquatic prey organisms are directly related to area of surface water, the time since the last dry down, and the length of time the wetlands were dry. The concentration of aquatic prey organisms into high density patches where wading birds can feed effectively is controlled by the rate of dry season water level recession interacting with local topography and habitat heterogeneity. In addition to wading birds, roseate spoonbills forage and nest within the Southern Coastal Systems. This species is discussed in the Greater Everglades Module in Section 5.5.5 Roseate Spoonbill Nesting and Estuarine Predator-Prey Interactions.

Prior to drainage, Picayune Strand was primarily a wetland ecosystem, with a variety of short to long hydroperiod plant communities that were ideal for the production of large numbers of small fish, amphibians and aquatic macroinvertebrates. This forage base was in turn able to support a variety of predators, including white ibis and wood stork. With the exception of a few small areas where organic soil loss has lowered the ground surface to a new equilibrium with the current water table, the drainage canals have eliminated the presence of standing water over most of Picayune Strand for more than short periods after heavy rainfall events. Without extended periods of inundation, aquatic fauna populations cannot survive and increase density and biomass over the summer wet season to form the abundant prey base for wading birds as occurred prior to drainage.

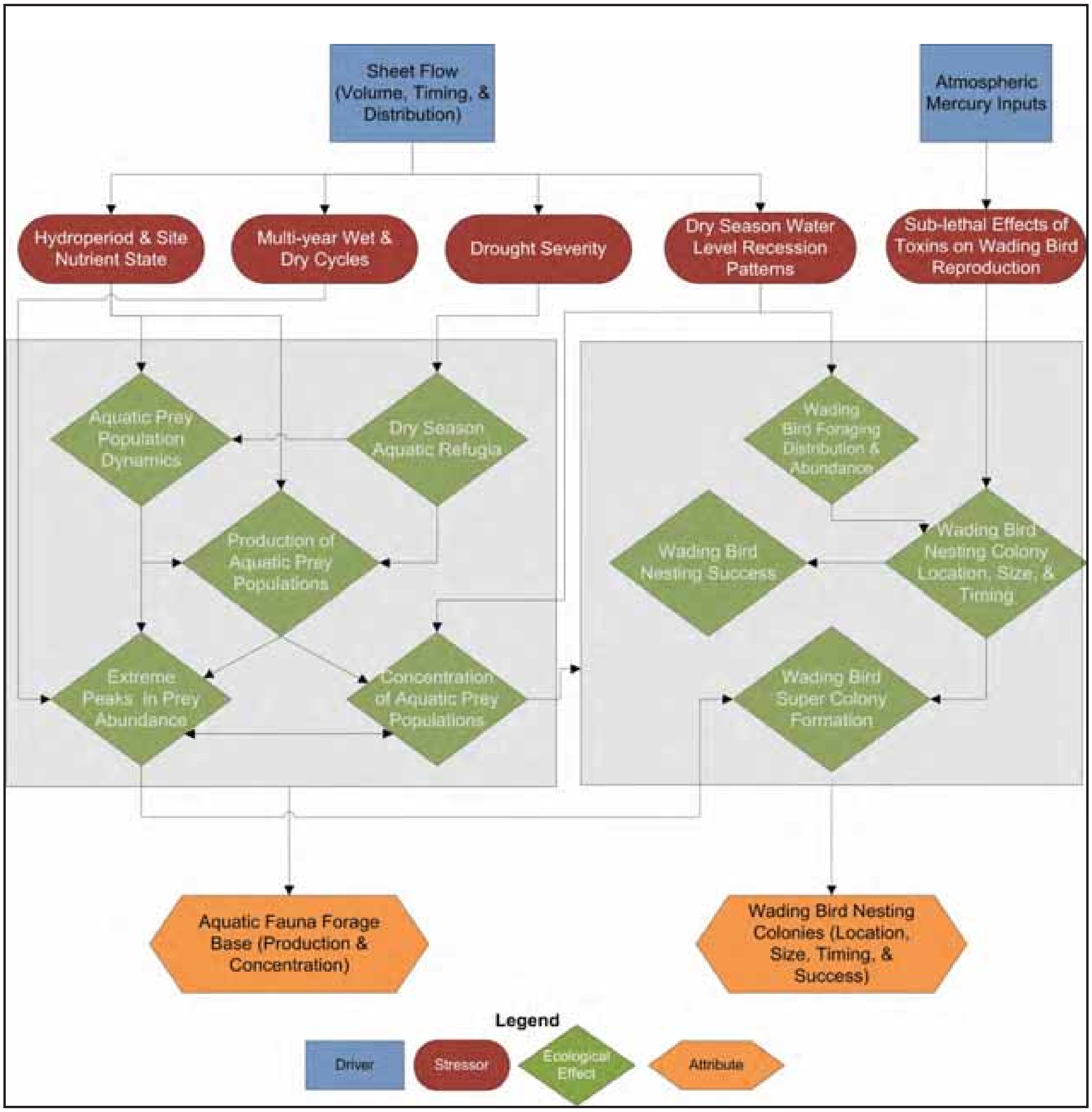


FIGURE 9-91. CONCEPTUAL ECOLOGICAL MODEL FOR PREDATOR-PREY INTERACTIONS OF WADING BIRDS AND AQUATIC FAUNA FORAGE BASE IN SOUTHERN COASTAL SYSTEMS

Restoration of the natural water surface coverage and duration resulting from the Picayune Strand Restoration Project would once again provide habitat for large numbers of small fish, amphibians and aquatic macroinvertebrates. Picayune Strand is fortunate in that it is surrounded by large areas that have not been as severely drained, which would be sources of fish, amphibians and aquatic macroinvertebrates that have been mostly absent from the area for many decades. All of Picayune Strand is less than five miles from an existing population of these organisms, which are capable of moving into all parts of Picayune Strand within one or two years once its normal hydrologic regime is restored. It is expected that following restoration, aquatic fauna would reestablish something approximating their pre-drainage populations within a year or two and wading birds would quickly begin to take advantage of this restored forage base, which would contribute to nesting success of colonies in the area.

As the Picayune Strand Restoration Project is implemented, the composition and distribution of wetlands within and adjacent to Picayune Strand would likely change. Monitoring wading birds would provide information on species numbers and distribution throughout the project area, and on adjacent affected lands. The pre-construction biological database can later be used to evaluate the success of the restoration and provide insight on the effects of other hydrologic restoration efforts in south Florida.

Three Greater Everglades performance measures pertain to this hypothesis cluster: 1) Wetland Trophic Relationships - Regional Populations of Fishes, Crayfish, Grass Shrimp and Amphibians, 2) Wetland Trophic Relationships – Wading Bird Foraging Patterns on Overdrained Wetlands, and 3) Wetland Trophic Relationships – Wading Bird Nesting Patterns. These performance measures are available online at www.evergladesplan.org/pm/recover/perf_ge.aspx. Two interim goals relate to this hypothesis cluster have been developed as well: 1) Indicator 3.9 - Aquatic Fauna Regional Populations in Everglades Wetlands and 2) Indicator 3.11 - System-wide Wading Bird Nesting Patterns. These can be found online at www.evergladesplan.org/pm/recover/ igit_subteam.aspx.

9.7.2 Monitoring

Wading bird surveys are being conducted in the Picayune Strand area to:

1) obtain seasonal estimates of wading bird species abundance and composition within the area affected by the project before restoration,
2) determine the spatial distribution of wading birds relative to habitat type before restoration, and
3) provide information on project effects on adjacent national interest lands, including the Florida Panther and Ten Thousand Islands National Wildlife Refuges, and Fakahatchee Strand Preserve State Park.

The study area extends from the outer Ten Thousand Islands, north to the Florida Panther National Wildlife Refuge, and from State Road 29 in the east to County Road 951 in the west (***Figure 9-92***). Ecological communities found within the study area include cypress swamp, scrub cypress, cabbage palm flatwoods, south Florida flatwoods, wet prairie, freshwater marsh, mangrove swamp, and mudflats. Coastal areas consist of hundreds of mangrove-dominated islands and associated shallow water estuarine bays.

Aerial surveys are flown twice monthly along transects spaced approximately two kilometers (km) apart and oriented in an east-west direction (***Figure 9-92***). Transects are aligned with the center of a 2-kilometer by 2-kilometer grid cell established throughout much of south Florida to assess and model restoration, and correspond to those currently flown in Big Cypress National Preserve and Everglades National Park. Individual species are recorded as possible, but otherwise are assigned to groups (e.g., white herons, dark herons). In addition, estimates of hydrologic condition are recorded for each 2-square kilometer (km^2) cell.

9.7.3 Results

Aerial wading bird surveys in the WCAs, Big Cypress National Preserve, and Everglades National Park are typically conducted from January to April, and June and July. The Picayune Strand Restoration Project surveys indicate that the highest number of wading birds counted at roost sites typically occur outside of that period, peaking in November for the upland areas and September for the partial coastal surveys. Both complete and partial surveys substantiate the strong seasonal variation in wading bird abundance in southwest Florida. More information on these wading bird surveys can be found in Doyle et al. (2008).

Rising tide did not appear to cause wading birds foraging in the estuarine fringe to move inland, but apparently caused them to move into the adjacent mangrove forest, making them difficult to observe. Foraging resumed when lower tides returned. Similar behavior has been observed in the Ten Thousand Islands National Wildlife Refuge. Thus, the most effective means of surveying wading birds in coastal areas appears to be to conduct surveys under low tide conditions.

Much of the Picayune Strand Restoration Project area is forested, where the likelihood of detection is reduced; however, by maintaining a consistent and reproducible monitoring approach, the ability to detect relative change before and after restoration should be possible. However, the absence of accurate quantification of detection probabilities across habitat types (e.g. forested versus marsh) will make such comparisons problematical.

9.7.4 Summary

With only three years of data currently available, statements regarding long-term recovery are not yet possible. Since the Picayune Strand restoration effort is moving forward, and making measurable ecological improvements, it should be possible to relate these to a beneficial response in the bird population dynamics.

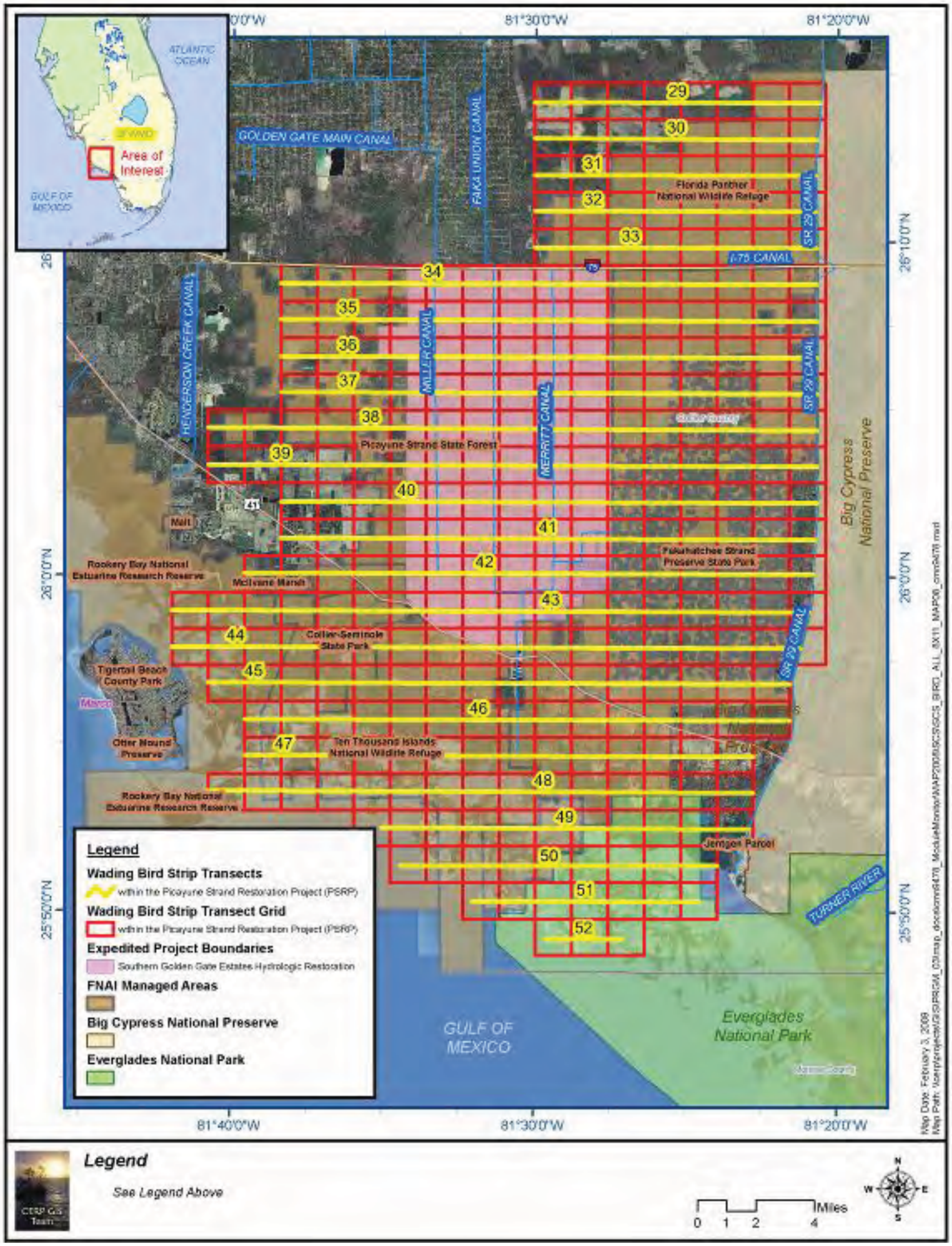


FIGURE 9-92. WADING BIRD MONITORING TRANSECTS WITHIN AND AROUND PICAYUNE STRAND

9.8 NATIVE VEGETATION MOSAIC HYPOTHESIS CLUSTER

9.8.1 Introduction and Background

The vegetation of Picayune Strand and Biscayne Bay coastal wetlands can be described broadly in terms of visually and ecologically distinct landscape units or mosaics. The composition and distribution of plant communities along elevation gradients are determined by patterns of hydroperiod, water depth, salinity, nutrient dynamics and fire patterns (***Figure 9-93***). Vegetation patterns in the Southern Coastal Systems wetlands would shift to a more freshwater wetland plant community when more natural hydrologic conditions with longer hydroperiods are restored. Through modifications of quantity, quality, timing and distribution of fresh water, exotic and other undesirable species would gradually give way to appropriate native vegetative communities.

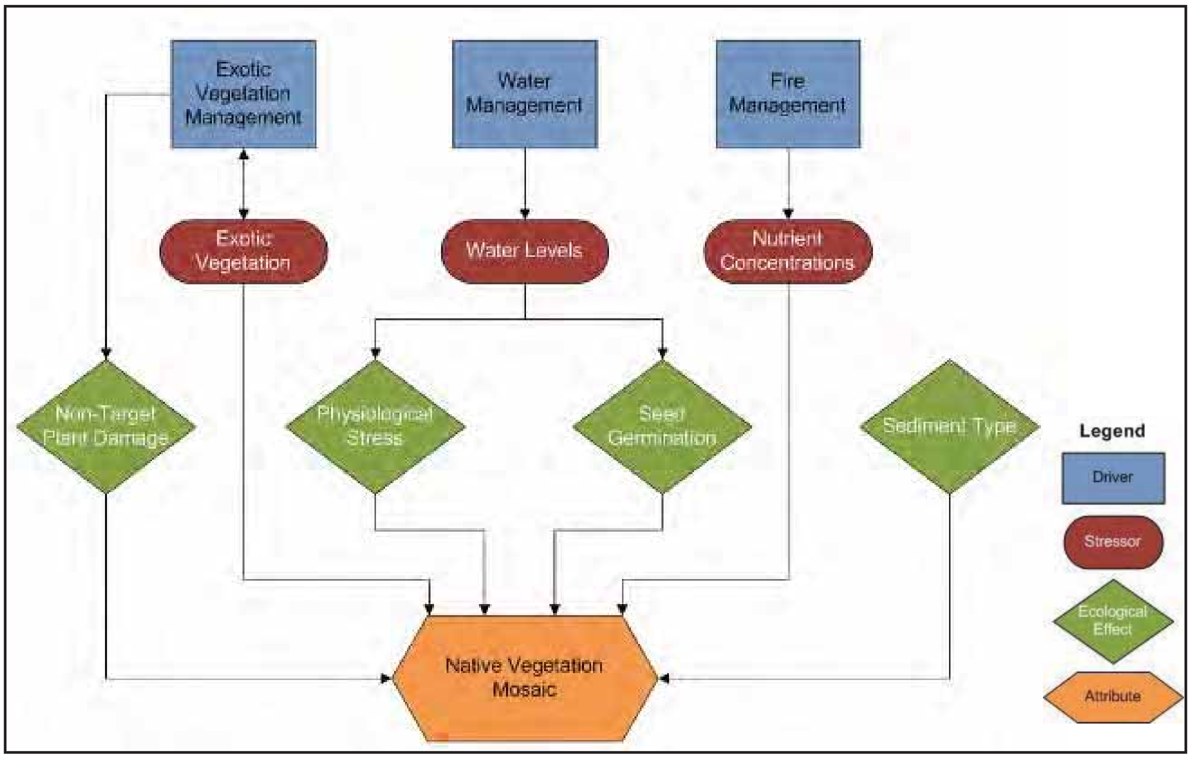


FIGURE 9-93. CONCEPTUAL ECOLOGICAL MODEL FOR THE NATIVE VEGETATION MOSAICS OF BISCAYNE BAY COASTAL WETLANDS AND PICAYUNE STRAND

9.8.2 Biscayne Bay Coastal Wetlands

Coastal Biscayne Bay is dominated by mangrove forest that currently stretches inland from the shore over a mile in places. Freshwater wetlands situated inland from the mangroves that were once graminoid marsh are now composed of forested wetlands and shrub/scrub wetlands and are generally infested with non-native invasive species, such as Brazilian pepper (*Schinus terebinthifolia*), Australian pine (*Casuarina* spp.), and shoe-button ardesia (*Ardisia elliptica*).

The earliest aerial photographs and descriptions of Biscayne Bay coastal wetlands indicate that freshwater graminoid marshes continued almost to the edge of the bay. Transverse glades allowed flow from the Everglades to cross the intervening limestone ridge and spread into the bay through numerous creeks. This slow moving water coupled to the natural storage provided by the Everglades sustained flows sufficient to allow sawgrass to extend to the edge of the bay where oysters were common at creek mouths. As the Everglades stage was lowered, surface and groundwater flows decreased. Conveyance canals were typically constructed in the transverse glades due to their lower elevation, which facilitated drainage, and replaced the sustained, broad flow paths that the transverse glades afforded with point source discharge. The canals and water control structures also replaced natural timing of flow with flood control to protect an increasingly developed coastal region. Parker et al. (1955) and Parker (1974) documented that the water table throughout Miami-Dade County has dropped between four to six feet since the turn of the century. The construction of the L-31E protection levee and canal complex completely eliminated sheet flow to the bay in the mid-1960s.

One of the consequences of the hydrologic change in southern coastal Biscayne Bay, referred to as Model Lands, is an area of low productivity that can be seen on aerial photographs and is referred to as the “white zone.” This zone has spread inland and increased in spatial extent over the last 60 years. Some have speculated that the inland movement of the white zone might be due to sea level rise; however, it reaches further inland on the eastern side of U.S. Highway 1 than the western side, suggesting that its spread is not due to sea level rise alone. Following restoration of more natural flow to Biscayne Bay, the boundary of the white zone in Biscayne Bay wetlands is expected to move seaward and shrink in size.

Also, in coastal Biscayne Bay, the marl prairie community has been severely degraded and reduced in size by invasions of exotic vegetation, encroachment of upland mangroves, and replacement of spikerush (*Eleocharis* spp.) by sawgrass (*Cladium jamaicense*). Grasses and sedges tolerant of wet conditions are expected to become a larger component of the marl prairie vegetation in the Biscayne Bay coastal wetlands as more natural flow to Biscayne Bay is restored.

9.8.2.1 Monitoring

Results from a mapping effort conducted by the USFWS in 2005 were used to describe current vegetation conditions in the Biscayne Bay coastal wetlands. This map covers 17,875 acres of the preliminary Biscayne Bay Coastal Wetlands Project area between Shoal Point and Turkey Point. Distinct vegetation types and/or communities were delineated from aerial photographs provided to the USFWS by Miami-Dade DERM. The map was groundtruthed by helicopter and automobile.

9.8.2.2 Results

A total of 28 vegetation classes were used to produce the map in ***Figure 9-94***. Of these, 13 are mixed vegetation classes that were required due to the complexity of the vegetation in the project area. ***Table 9-18*** provides the number of acres, number of polygons, average sizes of polygons, and percent of total acreage for each vegetation class delineated in the study area. Not surprisingly, mangroves are the dominant vegetation type east of the L-31E levee; they occasionally occur just west of the levee. Coastal band mangroves cover 2,737 acres and dwarf dense mangroves cover 1,258 acres; these coverages represent approximately 15 and 7 percent of the total project area, respectively. Dwarf sparse mangroves are much less common, covering only 336 acres of the project area. Land coverage west of L-31E is dominated by the filled-developed-disturbed category, which comprises 4,251 acres or about 24 percent of the total project area. Agriculture is the primary land use that defines this category. Three vegetation classes have roughly equal representation west of L-31E. These three include native forested or shrub wetland/Brazilian pepper mix (1,901 acres), sawgrass (1,471 acres), and native forested wetland or shrub/Australian pine mix (1,315 acres), which account for 11, 8 and 7 percent of the total project area, respectively. It is important to note that nine of the 28 vegetation classes include non-native species, with the non-native species usually dominating the native species in these areas. Combined, these non-native classes cover 5,177 acres, or 29 percent, of the total project area. Note that cattail (*Typha spp.*) is not included as a non-native species, even though it is usually present in south Florida only when water quality has been unnaturally altered.

The high degree of complexity and mixing of vegetation classes in the project area cannot be overemphasized. Undisturbed natural areas usually reach relative equilibrium over time and typically have well defined boundaries or transition areas between the various vegetation communities. Anthropogenic changes cause much mixing of vegetation types, partly due to infestation by invasive non-native vegetation, and the number of possible vegetation classes is higher. Due to significant anthropogenic effects, the Biscayne Bay coastal wetlands are very complex, resulting in a relatively large number of community types especially when invasive non-native species are present. For example, a total of 1,050 vegetation polygons were delineated within the 17,875 acres included in this mapping effort, resulting in an average polygon size of 17 acres. Significant mixing of vegetation types occurs throughout the project area. Of particular note is the mixing of native forested or shrub wetland and Brazilian pepper (1,901 acres), native forested or shrub wetland and Australian pine (1,315 acres), native forested or shrub wetland and other exotics (528 acres), and Australian pine and sawgrass (521 acres).

9.8.2.3 Summary

From the description above, it is apparent that the Biscayne Bay coastal wetlands contain a broad and diverse representation of wetland habitat types common in the south Florida ecosystem. It is also apparent that the project area contains a very complex mix of vegetation types, especially west of the L-31E Levee. A significant degree of this complexity is likely unnatural and has resulted from anthropogenic actions, primarily drainage to facilitate agricultural and urban development. The prevalence of dense stands of non-native forests, particularly Brazilian pepper and various mixes of non-native and native vegetation, underscores this point. The Biscayne Bay Coastal Wetlands Project objectives of reestablishing a more natural overland flow pattern by diverting water from conveyance canals into the coastal wetlands should serve to restore and enhance freshwater and estuarine wetlands in the project area, thus increasing the function of these habitats for fish and wildlife. Reestablishing connectivity between adjacent subbasins, which is another project objective, should aid in restoring a more natural broad overland flow across these wetlands.

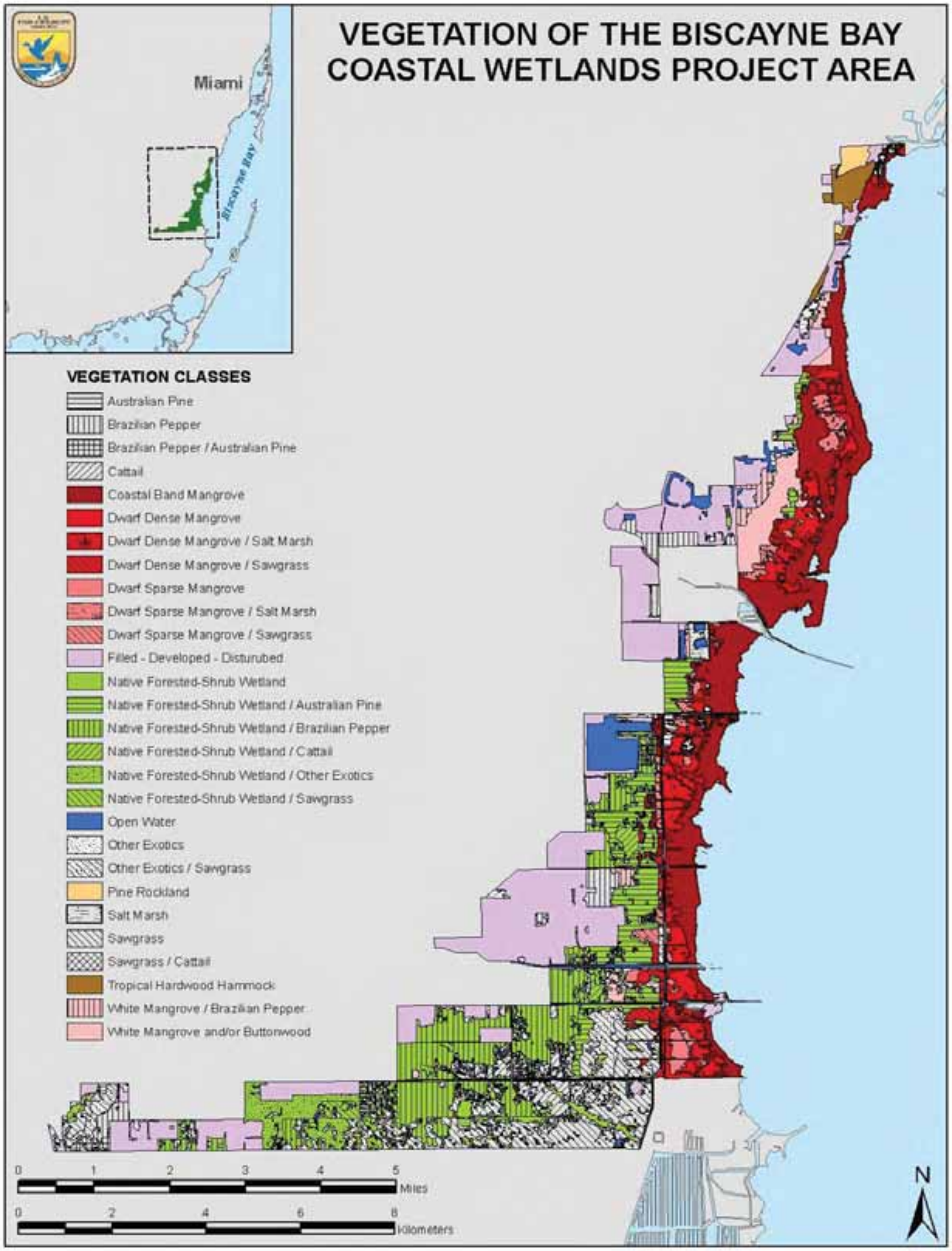


FIGURE 9-94. VEGETATION COVERAGE IN THE BISCAYNE BAY COASTAL WETLAND AREA BETWEEN SHOAL POINT AND TURKEY POINT

TABLE 9-18. SUMMARY OF 28 VEGETATION/LAND USE CATEGORIES IN THE BISCAYNE BAY COASTAL WETLAND AREA BETWEEN SHOAL POINT AND TURKEY POINT

Vegetation Class	Acres	Number of Polygons	Mean Polygon Size (acres)	Percent of Total Acres
1. Coastal band mangrove	2737.0	82	33.4	15.3
2. Dwarf dense mangrove	1257.5	115	10.9	7.0
3. Dwarf sparse mangrove	336.2	53	6.3	1.8
4. Salt marsh	118.5	24	4.9	0.7
5. White mangrove and/or buttonwood	507.2	30	16.9	2.8
6. Native forested-shrub wetland	196.4	66	3.0	1.1
7. Brazilian pepper	467.7	31	15.1	2.6
8. Australian pine	158.9	28	5.7	0.9
9. Other exotics	105.6	19	5.6	0.6
10. Sawgrass	1470.6	114	12.9	8.2
11. Cattail	363.3	109	3.3	2.0
12. Pine rockland	87.4	3	29.1	0.5
13. Tropical hardwood hammock	184.0	5	36.8	1.0
14. Filled-developed-disturbed	4250.7	63	67.5	23.8
15. Open water	574.2	60	9.6	3.2
2/4. Dwarf dense mangrove / salt marsh	33.2	4	8.3	0.2
2/10. Dwarf dense mangrove / sawgrass	126.6	11	11.5	0.7
3/4. Dwarf sparse mangrove / salt marsh	23.9	6	4.0	0.1
3/10. Dwarf sparse mangrove / sawgrass	99.0	18	5.5	0.6
5/7. White mangrove / Brazilian pepper	128.7	5	25.7	0.7
6/7. Native forested-shrub wetland / Brazilian pepper	1901.3	33	57.6	10.6
6/8. Native forested-shrub wetland / Australian pine	1314.9	59	22.3	7.3
6/9. Native forested-shrub wetland / other exotics	528.1	28	18.9	3.0
6/10. Native forested-shrub wetland / sawgrass	152.6	17	9.0	0.9
6/11. Native forested-shrub wetland / cattail	104.8	7	15.0	0.6
7/8. Brazilian pepper / Australian pine	49.7	6	8.3	0.3
8/10. Australian pine / sawgrass	521.8	32	16.3	2.9
10/11. Sawgrass / cattail	71.7	21	3.4	0.4

Of particular importance for restoring the wetlands will be to reestablish hydroperiods and water levels that are suitable for native freshwater marshes. Hydropatterns appropriate for native marshes are not conducive to the survival and proliferation of the seedlings of many nuisance non-native species, such as Brazilian pepper and Australian pine. However, it should be noted that restoration of more natural hydrology alone should not suffice to eradicate established stands of Brazilian pepper. It may be necessary to physically remove this vegetation. It is also anticipated that a return of a more natural fire regime would be required to maintain the desired vegetation communities, which would likely require the implementation of a fire management program.

Restoring natural vegetation communities in the project area would result in the restoration and enhancement of the area‟s fish and wildlife resources. Wetland habitat types found in the project area, particularly freshwater graminoid marsh and mangrove wetlands, are extremely important in terms of supplying plant biomass to the food web of Biscayne Bay. Both serve as vital refuge and nursery grounds for a wide variety of freshwater and estuarine fish and invertebrate species, as well as foraging grounds for a variety of wading birds and reptiles, including the federally threatened American crocodile (*Crocodylus acutus*). Specifically, freshwater and oligohaline graminoid marshes foster the production of large numbers of mosquito fish (*Gambusia affinis*), sailfin mollies (*Poecilia latipinna*), and other small fishes that serve as the prey base for many of the region‟s wading birds and higher trophic level predators. Current water management practices have resulted in altered hydroperiod and water levels in these marshes, which has reduced prey fish production and impacts breeding capabilities of wading birds. Restoring more natural salinities and hydropatterns to these marshes, as planned in the Biscayne Bay Coastal Wetland Project, should serve to increase prey fish production and further enhance these habitats for wading birds and other fish and wildlife resources.

9.8.3 Picayune Strand

Picayune Strand is made up of a background matrix of marl-forming wet prairie, or short hydroperiod marsh, interspersed with distinct patches of swamp hardwood forest, cypress strands, cypress domes, bay heads, willow ponds and tropical hardwood hammocks. In the Picayune Strand area, historically and today, the cypress strands and swamp forests are found in elongated depressions and the bay heads and cypress domes are found in circular depressions. Bald cypress and temperate swamp species are associated with the strands and domes. Temperate hardwood forest species make up the major species components of the hardwood hammocks. Today, marl prairie vegetation is degraded in the Picayune Strand area with a shift toward cabbage palm forests.

Prior to development, Picayune Strand was characterized by seasonal flooding and slow moving overland sheet flow that supported a variety of plant and animal communities in uplands and freshwater wetlands and in its downstream brackish wetlands and estuaries. Channelization of water flows resulted in elimination of sheet flow and lowered water tables. Picayune Strand was crisscrossed by a network of east-west roads every quarter mile that were connected by north-south roads approximately every mile. The road network further impeded natural sheet flow, and has resulted in the fragmentation of an extensive block of contiguous natural lands. The drained conditions have resulted in widespread and more intense wildfires than occurred under pre-drainage conditions. These fires accelerated a shift in vegetation from wetlands to upland communities dominated by fire tolerant species such as cabbage palm (*Sabal palmetto*) and exotics such as Brazilian pepper (*Schinus terebinthifolius*). In addition, these impacts extended a mile or more into other conservation areas, including the Fakahatchee Strand Preserve State Park.

Plugging and filling of the Faka Union Canal system and removal of most of the roads would restore sheet flow across the landscape, reestablish natural flowways, and bring back groundwater levels to near pre-drainage conditions within Picayune Strand and much of the surrounding public lands (USACE and SFWMD, 2004). Cabbage palm invasion should decrease or halt and areas presently shifting toward palm forest would gradually experience an increase in cypress seedlings and saplings. Areas where cabbage palm are removed and/or treated would be restored in the long-term. In addition, grasses and sedges tolerant of wet conditions would become a larger component of the marl prairie vegetation. A combination of restored hydrology, more natural fire regime, and appropriate exotic vegetation control can be expected to reestablish the pre-drainage character of Picayune Strand plant communities (USACE and SFWMD, 2004). Restoration of hydrology, as well as exotic vegetation control, has begun in Picayune Strand.

By 2007, 65 miles of road east of Merritt Canal had been removed, and the north-south portion of Prairie Canal was filled. The plugging of this canal eliminated the rapid loss of water along most of its seven-mile length by stopping quick flows and reestablishing sheet flow in the area during high rainfall periods. Plugging this canal has also improved groundwater levels within the western portion of Fakahatchee Strand Preserve State Park.

For a more complete discussion of Picayune Strand vegetation prior to and following drainage, see Appendix 7A-2 of the 2008 and 2010 South Florida Environmental Reports (Chuirazzi and Duever, 2008; Chuirazzi et al., in prep). This document can be found online at www.sfwmd.gov/sfer.

9.8.3.1 Monitoring

Extensive hydrological monitoring has been conducted in Picayune Strand and surrounding areas ranging from staff gauges and shallow manually read peizometers to the deep wells installed by the SFWMD in 2003 (*Figure 9-95* and ***Figure 9-96***).

To detect and track changes in vegetation resulting from the plugging and filling of Prairie Canal and road removal, three east-west vegetation transects were established in the upper, middle and lower portions of the seven-mile long north-south portion of the canal in spring 2004 (***Figure 9-97)***. The original baseline monitoring was conducted during spring 2004, and all post-construction monitoring began in spring 2008. Data collected in the northernmost areas have potentially been affected by a lengthened hydroperiod during the rainy seasons of 2005, 2006 and 2007, while the southern portion would have only experienced one rainy season in 2007, which was a drought year.

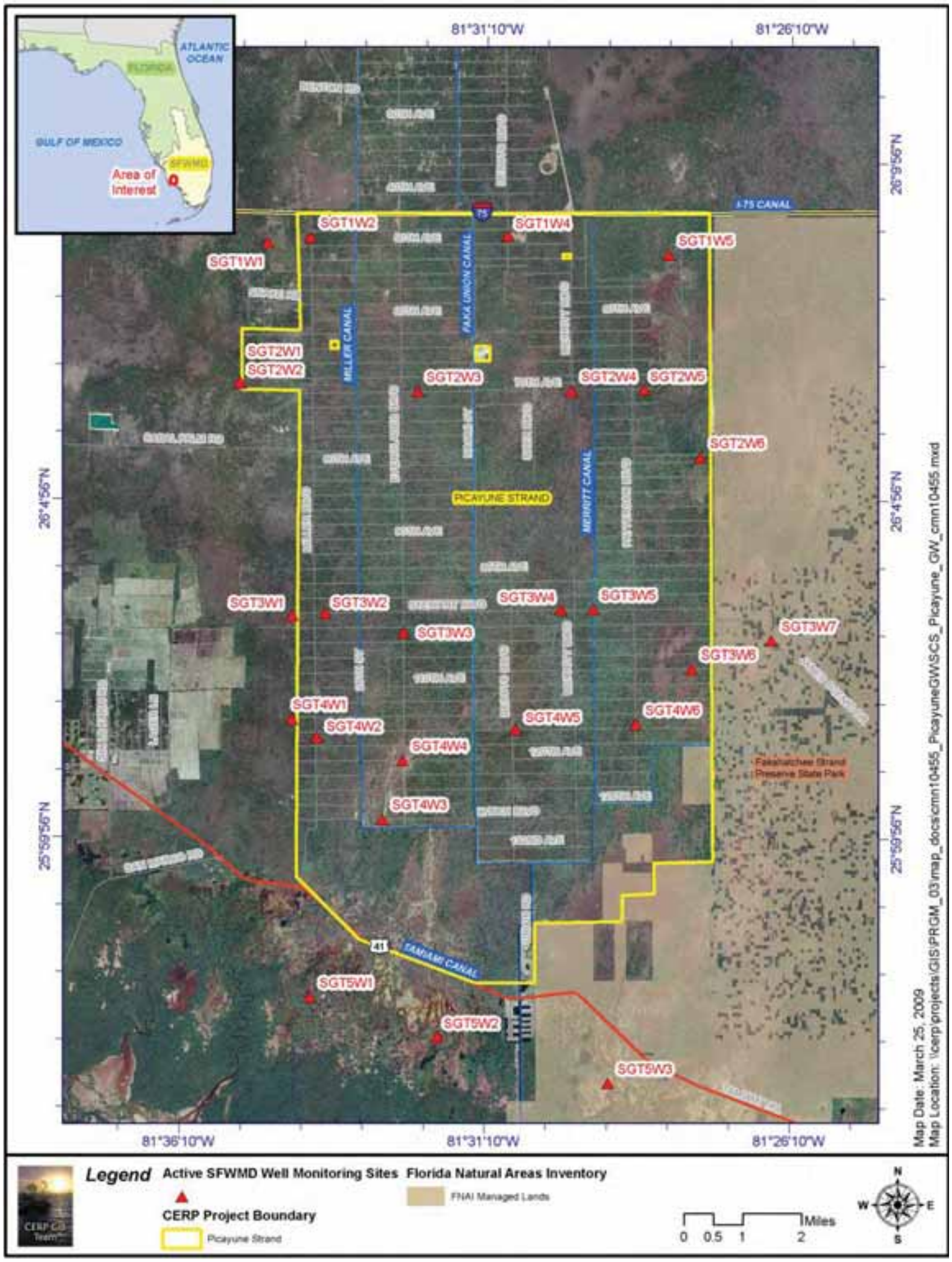


FIGURE 9-95. HYDROLOGIC MONITORING SITES WITHIN PICAYUNE STRAND

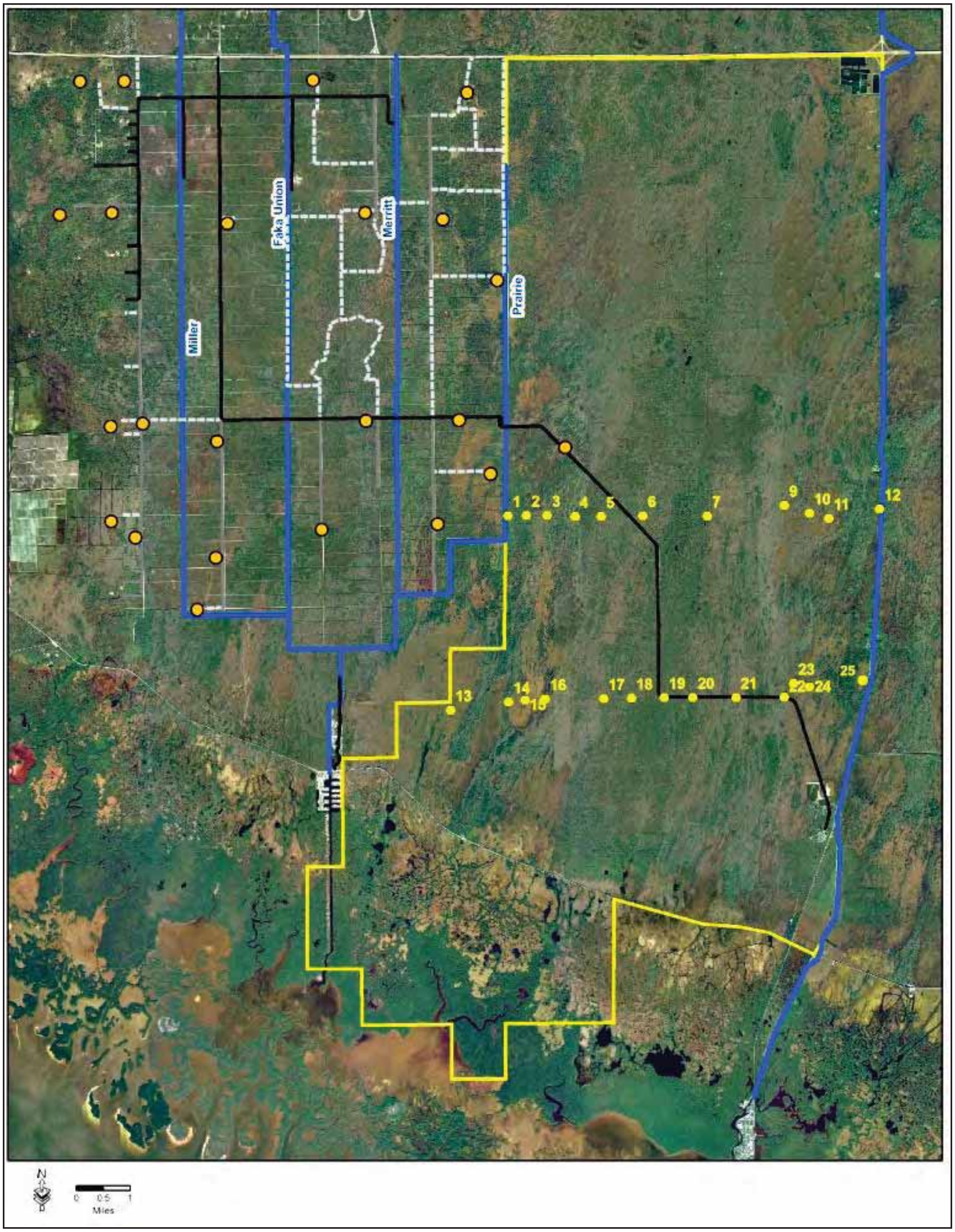


FIGURE 9-96. YELLOW DOTS INDICATE GROUNDWATER WELLS WITHIN FAKAHATCHEE STRAND PRESERVE STATE PARK

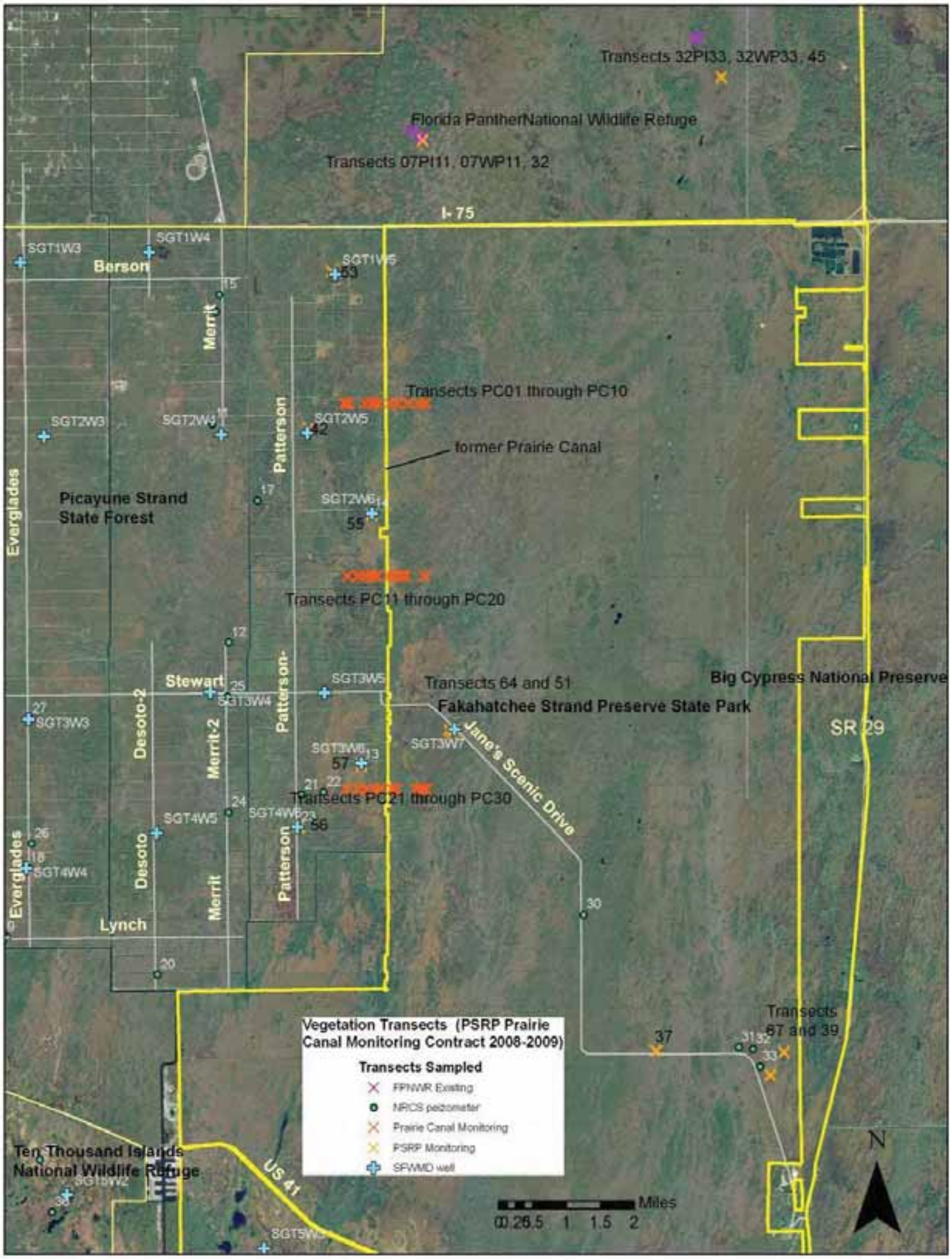


FIGURE 9-97. EAST-WEST VEGETATION TRANSECTS IN PICAYUNE STRAND

A total of 12 distinct habitats, including altered habitat types, were studied (***Table 9-19***). Dominant habitats studied in restoration plots included prairie (9 transects), hydric pine flatwoods (10 transects), and mesic pine flatwoods (2 transects). However, when all varieties of cypress habitat types are combined, cypress is the dominant habitat type (13 transects). Control plots focused on these dominant habitats with the exception of mesic pine flatwoods, which is not abundant in Fakahatchee Strand Preserve State Park. Fire interval and wind damage along these transects was measured as they probably had a larger effect on changes in vegetation from 2004 to 2008 than hydrology.

TABLE 9-19. TRANSECTS BY HABITAT WITH STUDY TYPE INCLUDED

Habitat	Number of Transects	Transect Names
Control Sites		
Cypress strand	**3**	32, 37, 51
Cypress w/graminoid understory	**1**	45
Wet prairie	**4**	07WP11, 32WP33, 39, 64
Hydric pine flatwoods	**3**	07PI11, 32WI33, 67
Restoration Sites		
Cypress strand	**6**	42, PC12, PC20, PC26, PC27, PC28
Cypress w/graminoid understory	**5**	PC11, PC13, PC25, PC29, PC30
Cypress and hardwood slough	**2**	PC02, PC15
Wet prairie	**9**	57, PC06,PC07, PC14, PC16,PC17, PC21, PC22, PC24
Hydric hammock	**1**	56
Hydric pine flatwoods	10	55, PC03, PC04, PC05, PC08, PC09, PC10, PC18, PC19, PC23
Mesic pine flatwoods	2	53, PC01
Total transects =	46	

9.8.3.2 Results

9.8.3.2.1 Hydrology

Based on 20 years of monitoring water levels in the adjacent Fakahatchee Strand Preserve State Park, the effects from the Prairie Canal plugging have extended from one to three miles into Fakahatchee Strand during wet and dry periods, respectively. ***Figure 9-98*** shows water levels along the north transect in Fakahatchee Strand Preserve State Park for 2003 and 2009; this transect is made up of wells 1 through 12 in ***Figure 9-96***. ***Figure 9-99*** shows water levels along the south Fakahatchee transect for 2003 and 2006; this transect is made up of wells 13 through 25 in ***Figure 9-96***. Overall water levels were lower in 2009 than in 2003 due to an extended period of drought. In all four of the graphs, relative water levels increase as distance increases between the well and the canal increases. However, along the north transect, water level decreased much more sharply the closer the well was to the canal in 2003 than in 2009 (***Figure 9-98***). This reflects the decrease in drawdown since Prairie Canal was filled. Since water still drains into the Faka Union Canal, drawdown has not decreased along the south transect between 2003 and 2009 (***Figure 9-99***).

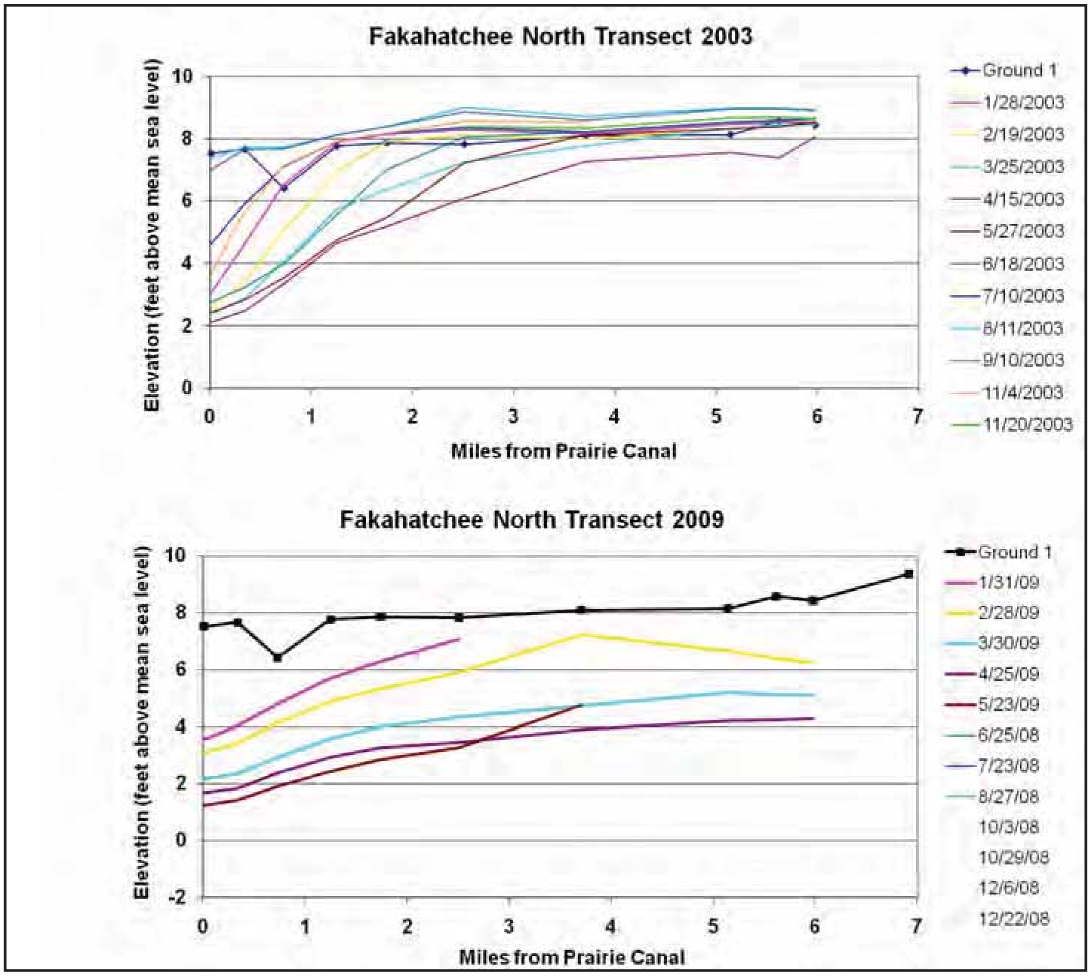


FIGURE 9-98. WATER LEVELS AT INCREASING MILES FROM PRAIRIE CANAL ALONG THE FAKAHATCHEE NORTH TRANSECT IN 2003 AND 2009

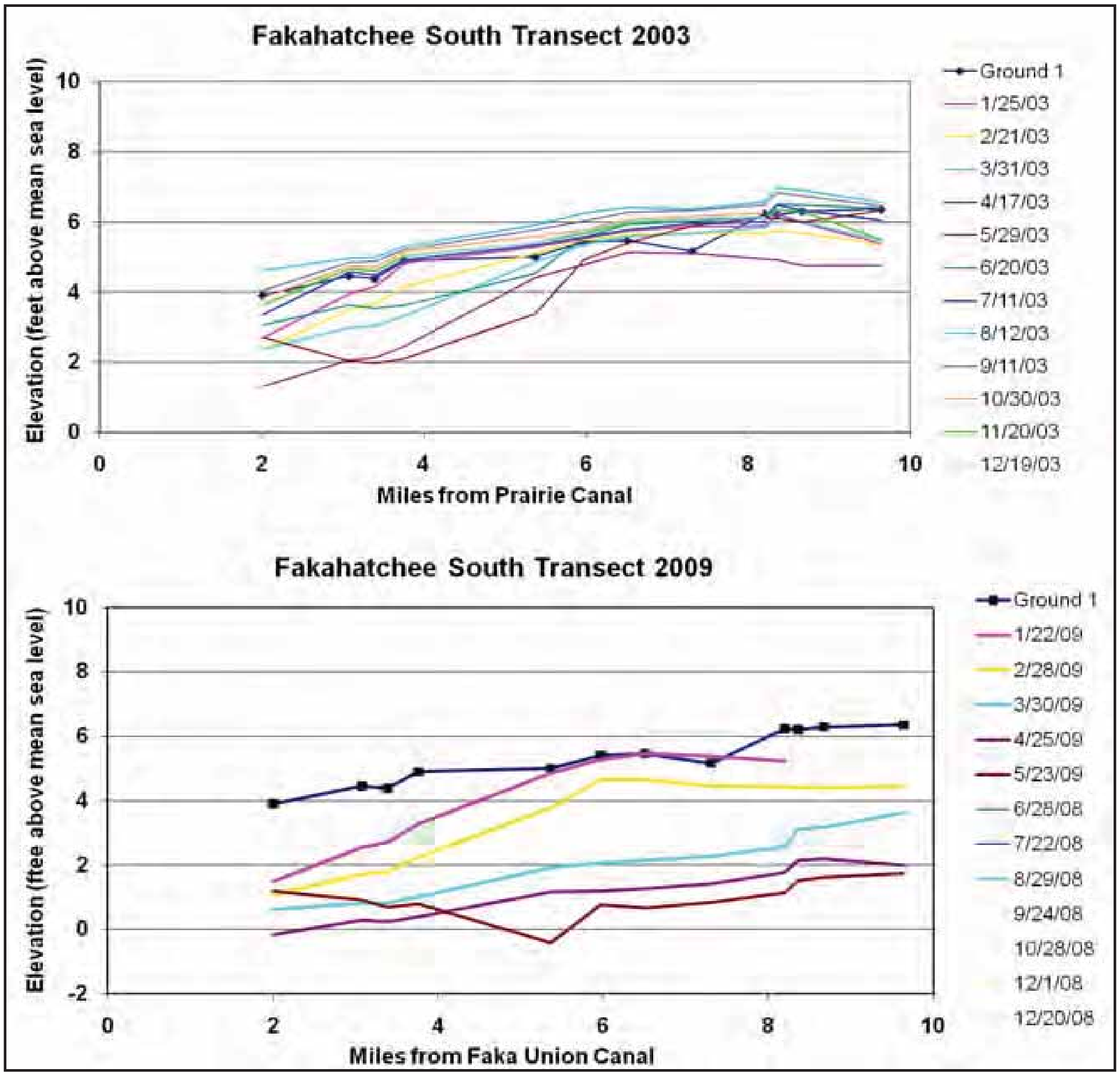


FIGURE 9-99. WATER LEVELS AT INCREASING MILES FROM FAKA UNION CANAL ALONG THE FAKAHATCHEE SOUTH TRANSECT IN 2003 AND 2009

The area hydrologically benefited by plugging and filling the Prairie Canal includes virtually all of the Picayune Strand Restoration Area to the east of Patterson Boulevard. These effects are increasingly greater as one gets closer to the canal. ***Figure 9-100*** shows the water levels at the wells along transect 2, which are the wells beginning with SGT2 in *Figure 9-95*, and ***Figure 9-101*** shows the water levels at wells along transect 3, which are the wells beginning with SGT3 in *Figure 9-95*. These plots show reduced water level drawdowns in the Prairie Canal vicinity since it was filled compared to those near the Merritt Canal, which has not been filled, and halfway between the two canals.

For more information on Picayune Strand hydrology see Barry and Saha (2008) and the Picayune Strand Restoration Project Baseline Reports, which are appendices in the 2008 and 2010 South

Florida Environmental Reports (Chuirazzi and Duever, 2008; Chuirazzi et al., in prep; www.sfwmd.gov/sfer).

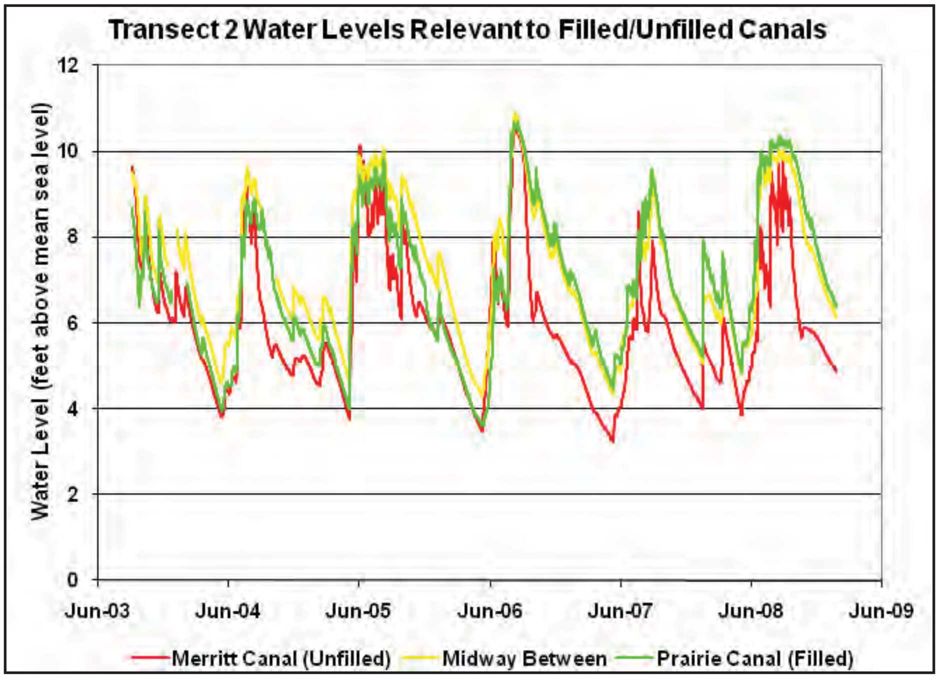


FIGURE 9-100. TRANSECT 2 WATER LEVELS RELATIVE TO FILLED AND UNFILLED CANALS

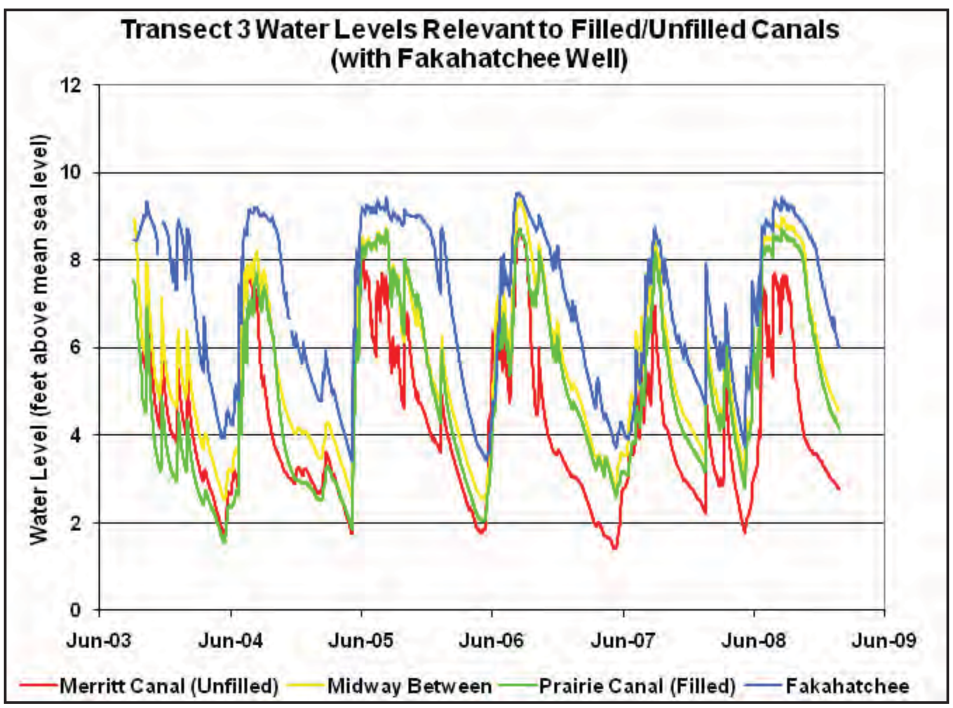


FIGURE 9-101. TRANSECT 3 AND FAKAHATCHEE WELL WATER LEVELS RELATIVE TO FILLED AND UNFILLED CANALS

9.8.3.2.2 Fire Interval

As mentioned above, fire may have a much more immediate and substantial effects on vegetation transect data than hydrological restoration in the short term. ***Table 9-20*** summarizes fire intervals for the control and restoration habitats (Barry and Saha, 2008).

TABLE 9-20. SUMMARY OF FIRE INTERVAL BY HABITAT TYPE WITHIN CONTROL AND RESTORATION SITES

Habitat	Number of Transects							
	Time Since Fire 2004/2005			Time Since Fire 2008				Burned Between Events
	≥1 year	1-7 years	>7 years	≥1 year	1-7 years	>7 years	Unknown	
Control								
Cypress strand			3			3		1
Cypress with graminoid understory		1					1	0
Wet prairie		3	1		3		1	2
Hydric pine flatwoods		2	1		1	1	1	0
Total control transects	**0**	**6**	**5**	**0**	**4**	**4**	**3**	**3**
Restoration								
Cypress strand		1	5			6		0
Cypress w/graminoid understory		3	2		3	2		1
Cypress and hardwood slough			2		1	1		1
Wet prairie		7	2	3	5	1		8
Hydric hammock		1			1			1
Hydric pine flatwoods	1	9		5	5			9
Mesic pine flatwoods			2		1	1		1
Total restoration transects	**1**	**21**	**13**	**8**	**16**	**11**	**0**	**21**

In 2004 or 2005, prior to restoration activities, the control sites had no transects burned within one year of monitoring. Six transects, all in cypress with graminoid understory, wet prairie and hydric pine flatwoods, had burned within the previous one to seven years. Five transects had not burned in over seven years. Most of these were in cypress strand habitat, which rarely burns.

In 2008, of the sites for which information was available, no sites had burned within one year, four sites had burned within the last one to seven years, and four sites had not burned for over seven years. Information was unavailable for three transects. All transects within wet prairie and all but one of within hydric pine flatwoods burned within the last one to seven years. None of the cypress strand sites had burned in the last seven years. Three of the sites had burned between sampling events.

Only one transect in the restoration sites had burned within a year prior to the baseline monitoring period (2004-2005). This was within hydric pine flatwoods habitat. Most transects (21) had burned within one to seven years of baseline monitoring. These transects are within all habitats except for cypress with hardwood slough and mesic pine flatwoods. Only one cypress strand transect burned within one to seven years prior to baseline monitoring. Thirteen sites, the majority of which were cypress or former cypress habitats, had not burned for over seven years. Some transects in wet prairie and both in mesic pine flatwoods habitats had also not burned within seven years. Fire had been suppressed for a long time in these sites.

Fires occurred in eight restoration transects within a year prior to the first post-construction monitoring event in 2008. All of these fires occurred in wet prairie or hydric pine flatwoods habitats. Again, the majority of sites (16) had burned within one to seven years prior to the post-construction monitoring. These transects fell within all habitats except for cypress strand. All transects within cypress strand, cypress with hardwood slough, and mesic pine flatwoods had not burned in over seven years. Two out of the five transects within the cypress with graminoid understory habitat and one transect in wet prairie habitat had also not burned in over seven years. Overall, 11 transects had not burned in over seven years. Fire occurred between the two monitoring events at 21 transects. The effects of fire on vegetation are discussed under the Vegetation subsection below.

9.8.3.2.3 Wind Damage

The 75-mile wide eye of Hurricane Wilma passed directly over the study area with 120-miles per hour sustained winds on October 24, 2005. Though the strength of gusts likely varied from transect to transect, sustained winds were likely similar at all transect locations due to the size of the eye, which covered the entire study area. Other storms such as hurricanes Charley, Frances and Jeanne may have affected some of the transects to a lesser degree since initial sampling in 2004. Storms Katrina and Rita prior to Wilma may also have had some minimal effects. As discussed above with fire, it is important to note that these storms may have a more immediate and substantial effect than hydrological restoration for the short term. Wind damage to each community is discussed in the following section.

9.8.3.2.4 Vegetation

Since restoration of hydrology in Picayune Strand should favor healthy cypress communities, density of pond cypress (*Taxodium ascendens*) was determined and compared between years along transects. Pond cypress density in the canopy of all cypress dominated restoration sites decreased. Some of this change can be directly attributed to wind damage, while no mortality from wind was observed in control sites. Fire probably contributed to the decline. Pond cypress grew in diameter more in control versus restoration transects for most sites. Pond cypress density in the subcanopy decreased in both control and restoration sites. Although causes of mortality were not obvious enough in the field to definitively attribute to Hurricane Wilma, it is likely a contributor.

Restoration of Picayune Strand requires a reduction in cabbage palms. At this time, very little mortality in canopy cabbage palms were observed in control and restoration transects, though increases were common. A substantial increase in groundcover and seedling cabbage palms was consistently observed in the cypress dominated communities within the control sites. The low water levels and short hydroperiod due to the drought in 2007 may have allowed cabbage palm recruitment in these areas. An increase was observed in cypress dominated restoration areas. It is likely that the drought enabled the seedlings to establish and persist. The one exception was transect PC25, which is made of cypress with graminoid understory and is located just outside the footprint of the former Prairie Canal (***Figure 9-97***). This transect showed a substantial die-off in young cabbage palms. Seedlings decreased from 494 in 2004 to 10 in 2008. Groundcover palms decreased from 180 in 2004 to 89 in 2008. There were many hydrological indicators absent in 2004, such as obvious lichen lines and algal mats, suggesting that flooding may have indeed reduced lower strata cabbage palms. On the other side of the filled canal, Brazilian pepper declined in transect PC26, which is made up of cypress strand habitat. Coverage in this transect was 88 percent Brazilian pepper cover in 2004, but was reduced to less than one percent cover. It appears that on both sides of the former canal, hydrology and, therefore, vegetation is substantially affected by restoration.

The direction of change in species richness, species diversity and percent cover of common and wetland indicator species were examined across sampling years (2004 and 2008 for restoration and 2005 and 2008 for control transects). If the effects of restoration are over and above the effects of drought, the comparison across years would show stronger restoration effects. Control sites on the other hand, would not be expected to show any effect of restoration. Any potential observed results here may be attributable to other factors such as fire, drought and random effects.

Comparisons in parameters of species composition were made for transects with comparable fire histories to discern the effects of restoration and hydrology. For control sites, species richness across transects increased in 2008 as compared to 2005 (*Figure 9-102*) while species diversity did not change significantly. The change in species richness is probably a result of excessive flooding due to Hurricane Wilma followed by the 2006 to 2007 drought, which allowed additional species to become established in cypress habitats.

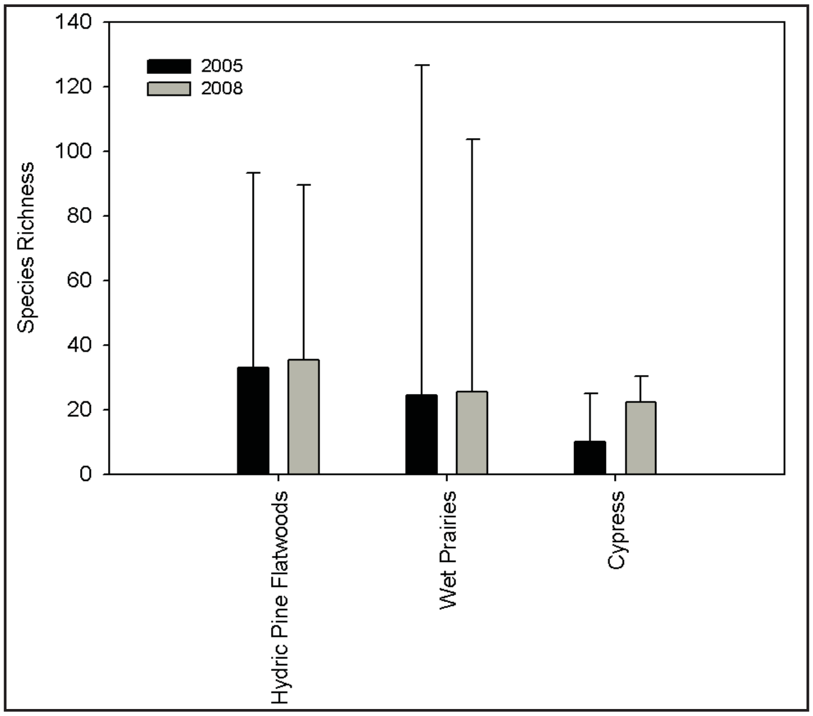


FIGURE 9-102. SPECIES RICHNESS IN CONTROL TRANSECTS WITH COMPARABLE FIRE HISTORIES

Within restoration sites, analyses suggest an increase in species richness from 2004 to 2008 in all vegetation types along restoration transects with different fire histories. (***Figure 9-103***). Along transects with comparable fire histories, all vegetation types except mesic pine flatwoods showed an increase in species richness from 2004 to 2008. This suggests that regardless of fire history, the Picayune Strand restoration efforts thus far have increased species richness, which is most likely due to hydrologic improvements (*i.e.*, the increase in ground water stage).

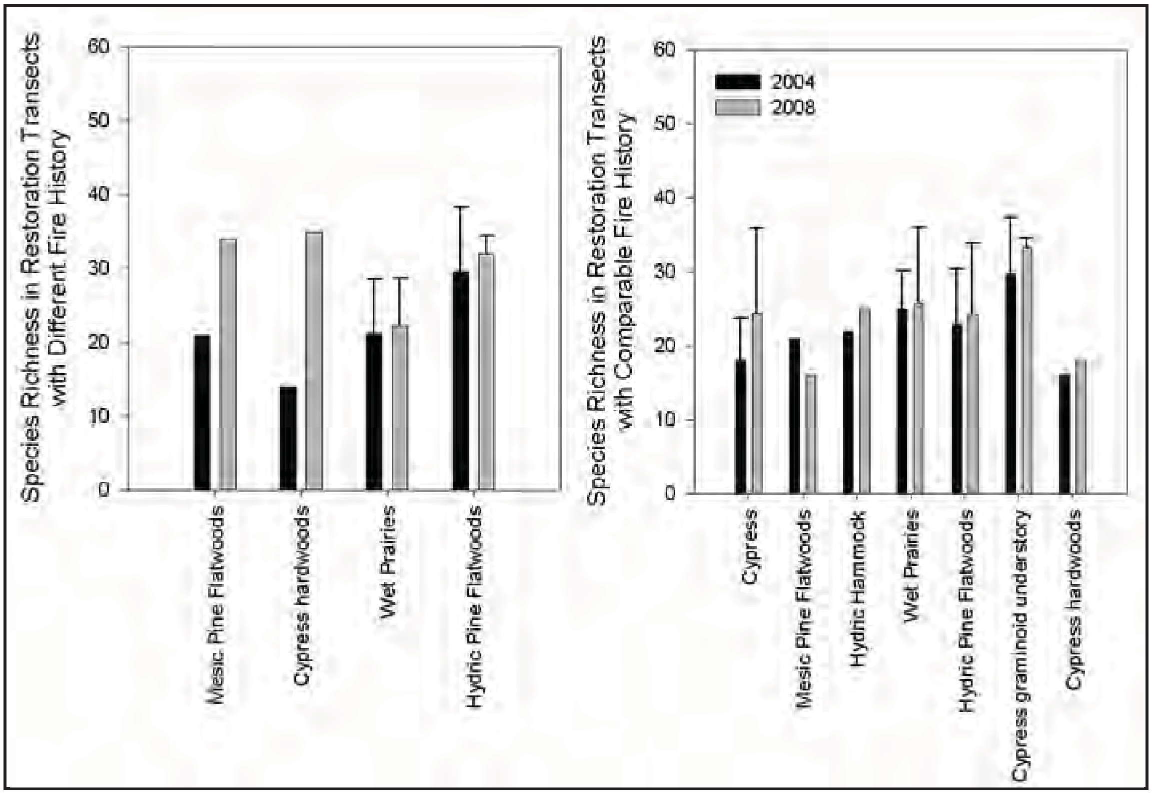


FIGURE 9-103. SPECIES RICHNESS IN RESTORATION TRANSECTS WITH DIFFERENT (LEFT) AND COMPARABLE (RIGHT) FIRE HISTORIES

In conjunction with patterns of species richness and diversity, the quality of species change was assessed by comparing wetland affinity indices across years before and after restoration (***Figure 9-104***). No significant changes occurred in the quality of species assembly before and after restoration for all the habitat types except for wet prairie. For this habitat, the quality of species assemblage improved after restoration in transects with comparable fire history. Transects with differing fire histories were not compared for wetland affinity. This result illuminates that species richness and diversity are not the only measures useful for assessing effects of restoration and climate on species composition, but also quality of species in relation to their indicator status is useful.

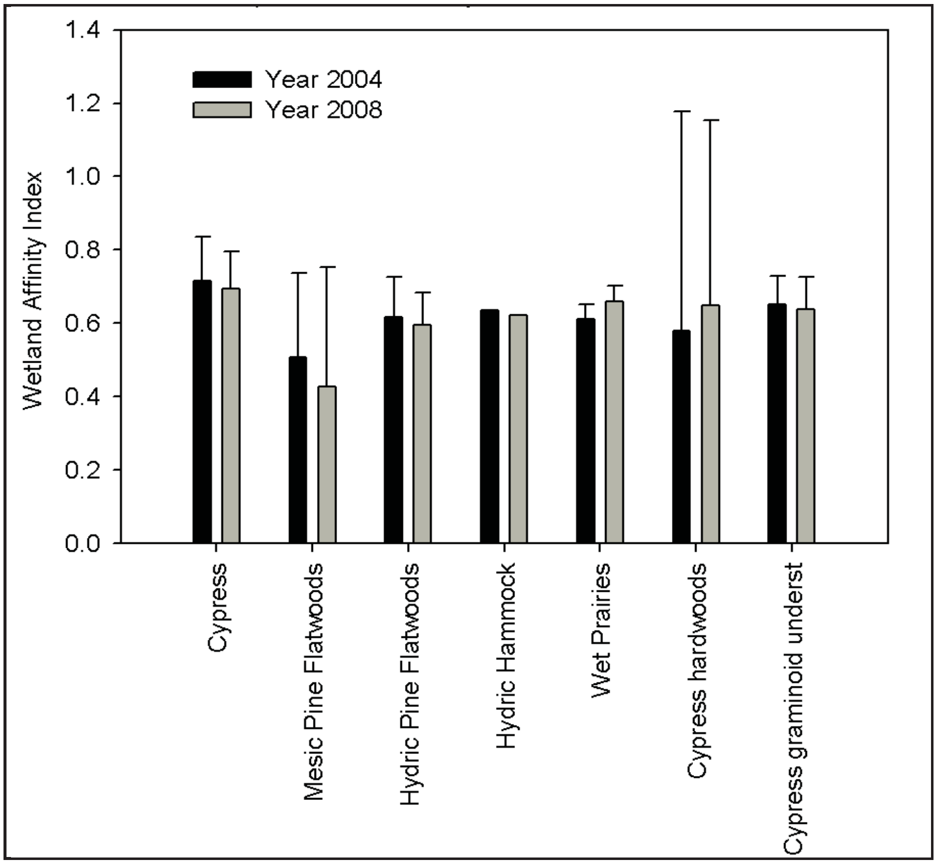


FIGURE 9-104. WETLAND AFFINITY INDEX IN RESTORATION TRANSECTS WITH COMPARABLE FIRE HISTORY

Certain low coverage, high frequency, diminutive forb species typical of the Florida Panther National Wildlife Refuge (Barry and Woodmansee, 2006) are lacking in the restoration sites. Specifically, smallfruit primrose willow (*Ludwigia microcarpa*) seemed to be disproportionately lacking in drained hydric flatwoods and wet prairies of Picayune Strand. Curtiss‟ primrose willow (*Ludwigia curtissii*) was included with these data because, though less frequent at both sites, it behaves similarly ecologically. Both species of mermaidweed (*Proserpinaca palustris*, *P. pectinata*) and both species of hornpod (*Mitreola petiolata*, *M. sessilifolia*) follow a similar pattern in these habitats, and in fact were nearly absent in restoration site wet prairies and hydric pine flatwoods in 2005. These species, typical of lower strata in graminoid dominated short hydroperiod wetlands, appear to have disappeared from these drained areas and may be useful indicators to use during restoration monitoring in this area.

Figure **9-105** shows results of indicator species frequency tests for these habitats at control sites. Smallfruit primrose willow decreased in wet prairie but increased in hydric pine flatwoods and cypress habitats. Hornpod increased in wet prairie and hydric pine flatwoods. Mermaidweed increased in cypress habitats but decreased in the other habitats. A decrease in these species in hydric pine flatwoods or wet prairie habitats may suggest influence of drought. An increase of these species in cypress may also suggest this as they are species more typical of shorter hydroperiod areas such as the other habitats mentioned. Thus, most of the changes observed suggest influence of drought except for the increase in smallfruit primrose willow in hydric pine flatwoods and the decrease in hornpod.

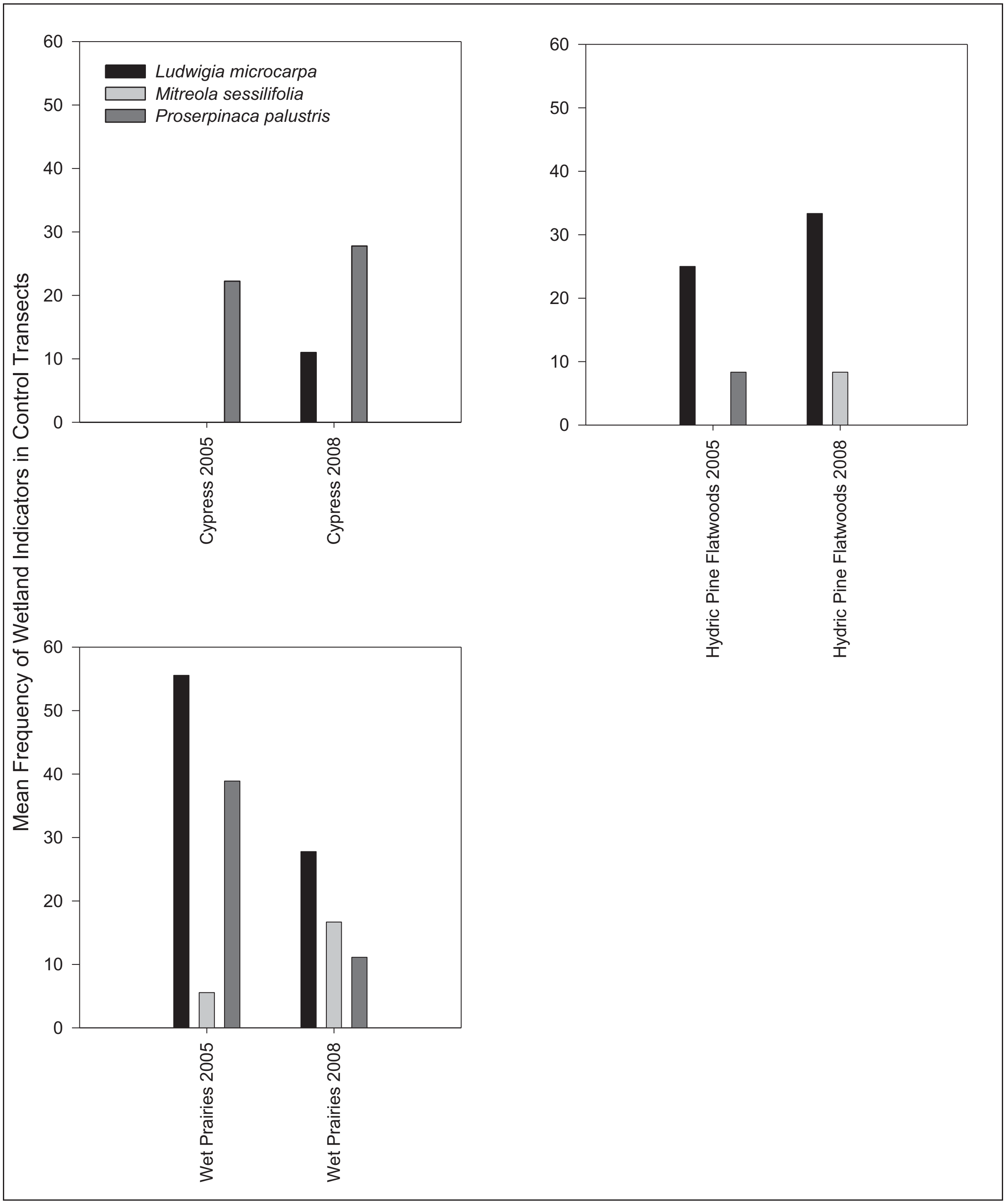


FIGURE 9-105. FREQUENCY OF WETLAND INDICATOR SPECIES IN CONTROL TRANSECTS

At restoration transects, smallfruit primrose willow significantly decreased in percent frequency across years from 2004 to 2008 in cypress with graminoid understory habitat, while a insignificant decrease was noticed in hydric pine flatwoods (***Figure 9-106).*** An increase was actually observed in wet prairie and cypress/hardwood slough habitats. As mentioned above, the increase in these indicator species in the deeper cypress habitats may suggest influence of drought. However, because these sites are so drained, the hydroperiod may actually just now be suitable for these species with an increased hydroperiod. The decrease of these indicators suggests influence of drought.

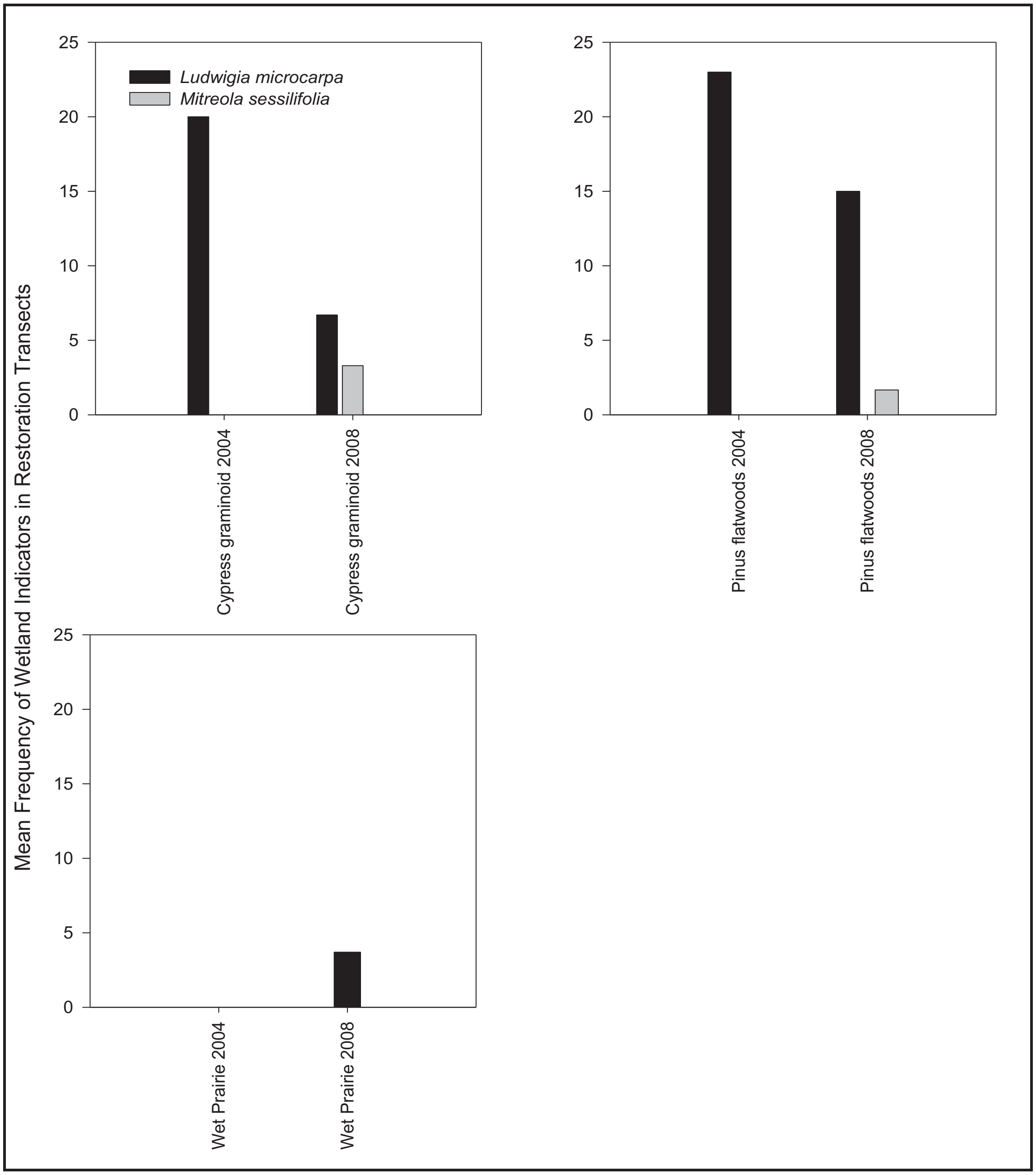


FIGURE 9-106. FREQUENCY OF WETLAND INDICATOR SPECIES IN RESTORATION TRANSECTS

No major decline in the frequency and percent cover of common or dominant species has been recorded, yet an increase in species richness across the habitats has been. This leads to increase in plant cover and biomass across years, which is probably a result of an increase in availability of niches for plants to establish as a result of drought.

For more information on vegetation monitoring in Picayune Strand, see Appendix 7A-2 of the 2010 South Florida Environmental Report (Chuirazzi and Duever, in prep.) and in the first post-restoration monitoring report for Picayune Strand (Barry and Saha, 2008). The links will be provided once these documents are posted.

9.8.3.3 Summary

The filling of the Prairie Canal and the removal of all roads east of Merritt Canal has altered hydrology in the areas adjacent to the canal, especially to the east of the canal. Water levels have increased in western Fakahatchee Strand Preserve State Park, which is adjacent to Picayune Strand. Benefits diminish west of the Prairie Canal. The area west of Patterson Boulevard would probably continue to be severely impacted until Merritt Canal is plugged. Also, since the east-west portion of Prairie Canal below the restored upper seven miles is still open and draining into Merritt Canal, it is likely that the water table in the lower one-to-three miles of the area between Patterson Boulevard and the north-south portion of the filled Prairie Canal is still being impacted.

Most of the changes between the 2004 and 2008 vegetation are most likely caused by drought, wind damage and fire, but some indications of response to the plugging of Prairie Canal and removal of roads east of the Merrit Canal can be seen. A large die off of very young cabbage palms occurred in transect PC25 and Brazilian pepper declined from 88 percent Brazilian pepper cover to less than one percent cover in transect PC26 following restoration activities. It appears that on both sides of the former canal in this area, hydrology and therefore vegetation is substantially affected by restoration. As restoration progresses, an increase in cypress densities and a decrease in cabbage palm and Brazilian pepper densities is expected. The wetland affinity index should increase and the frequency of wetland indicator species should increase as well.

9.9 CONCLUSIONS

The 2009 Southern Coastal Systems contribution to the SSR has made significant progress toward integrating the data and analyses obtained through the MAP program by uniting pieces of the multi-agency effort together for the first time. Nevertheless, there continue to be concerns regarding sustainability of programs necessary to make a science-based adaptive management program a success. Efforts to optimize monitoring need to ensure that data streams remain sufficiently robust to allow for reliable decision-making as restoration unfolds.

Florida‟s Southern Coastal Systems are of enormous ecological value that are operating at less than their full potential due to anthropogenic changes to freshwater flow that began more than a century ago. Today, nearshore salinity regimes of northeastern Florida and Biscayne Bays are destabilized, or otherwise altered, by watershed development, shoreline modification, and especially the operation of a canal network. The canal system has profoundly affected the volume, distribution and quality of fresh water flowing into the Southern Coastal Systems, resulting in reduced estuarine diversity and production. Restoring more natural salinity regimes to nearshore environments would have positive consequences for the region‟s flora and fauna, particularly its economically important fisheries.

A major objective of the Southern Coastal Systems monitoring efforts summarized in this report is to establish present baseline levels and/or patterns (pre-CERP conditions) of a selected set of biotic and abiotic measurements and then track changes in them as project-related modifications ensue. Results presented for the Southern Coastal Systems indicate that the approaches and methods used for assessing water quality, salinity regimes, SAV and nursery function are appropriate and efficient relative to the size, complexity and value of the systems in question. In addition, due to differences in scale, spatiotemporal overlap, and coordination of sampling design and analyses, the individual projects are highly complementary. Therefore, data obtained from Southern Coastal Systems sampling would prove critical, not only in an assessment and monitoring context, but also for advancing the understanding of how subtropical ecosystems respond to incremental reductions in anthropogenic stress. Certainly, the ability to interpret monitoring results to adaptively manage and guide restoration efforts would be directly proportional to this understanding, which hinges on long-term, uninterrupted time series combined with focused short-term research.

The lack of quantitative spatially-explicit historical data, and the many uncertainties regarding the timing, magnitude and form of restoration responses, requires that several physical, chemical and ecological community components be measured and modeled in multiple ways. Without question, reducing monitoring effort would diminish the ability to distinguish restoration-impacts from the backdrop of natural variation, and to make realistic assessments and predictions of system status that are fundamental to AM.

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