Chapter 6: Everglades Research and Evaluation

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SUMMARY

The studies and findings discussed in this chapter of the *2012 South Florida Environmental Report – Volume I* are presented within five main fields: (1) hydrology (2) wildlife ecology, (3) plant ecology, (4) ecosystem ecology, and (5) landscape. Programs of study were based on the short-term operational needs and long-term restoration goals of the South Florida Water Management District (District or SFWMD), including large-scale and regional hydrologic needs in relation to regulation schedules, permitting, the Everglades Forever Act (Section 373.4592, Florida Statutes) mandates, and the Comprehensive Everglades Restoration Plan (CERP). In this year's SFER, Florida Bay science is covered in this Everglades chapter (instead of the coastal ecosystems chapter) to develop a more regional understanding of upstream impacts on Florida Bay and facilitate a more watershed overview and synthesis. Key findings of Everglades research and evaluation during Water Year 2011 (WY2011) (May 1, 2010–April 30, 2011) are summarized in **Table 6-1** at the end of this section.

HYDROLOGY

During WY2011, the hydrologic environment in the Everglades was similar to the WY2009 drought as dry season water depths in most of the freshwater marshes exceeded the lower tolerance for peat conservation. Large areas of Water Conservation Area 3, in particular, became very dry for three to four weeks in May and June 2011, making WY2011 much drier than previous drought conditions. However, there were no peat fires in the Everglades during this rather extreme drydown. WY2012 is expected to be another drought year due to La Nina conditions. If it is, then the District will need to evaluate the possible connection of droughts to climate change and the serious impacts it could have on peat fires, oxidation, and conservation in the Everglades. The hydrology in Florida Bay and the lower regions of Everglades National Park was very different from the upper marshes, in part due to the long lag times associated with the great distances separating the upstream rainfall patterns from the downstream hydrology, the buffering capacity of large bodies of water like Florida Bay, and the rainfall pattern in the southern part of the system. As a result, the salinity was near average and drought effects were not evident in this region until midsummer.

WILDLIFE ECOLOGY

Wading bird nesting was not successful for most species during 2011 (see Appendix 6-1 of this volume). The 2011 nesting season was poor despite the excellent foraging habitat, lack of hydrologic reversals, and smooth and continuous recession rates because the drought was extensive and prolonged. Such drought conditions play an important but poorly understood role in

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the nesting success of wading birds. For this reason, the Loxahatchee Impoundment Landscape Assessment (LILA) facility continues to be an effective "living laboratory" for evaluating fish, crayfish, wading bird, and hydrology interactions. This year, a LILA study to assess drought impacts on wetland faunal community interactions found many predatory fish and few crayfish in the control cells where a drought was not simulated but in the cells rehydrated after the simulated drought, predatory fish were reduced and crayfish were abundant. Droughts seem to reduce populations of these large-bodied fish allowing greater recruitment of crayfish, an important prey component of white ibis. This suggests a strong link between droughts and wading bird super colony formation. It also suggests canals and impounded regions, which can be a source of predatory fishes, may influence wading bird nesting.

Monitoring fish populations in mangrove habitats is part of a concerted effort to understand the interaction of habitat, hydrology, fish, and wading birds. Mangrove fish can have dramatic population swings, but unlike the upper freshwater marsh system, droughts can have more impact on salinity than on water levels. This year, it was discovered that these populations are also susceptible to cold snaps and hydrologic patterns. In WY2010, fish for wading birds were very limited due to cold and high water. In WY2011, many spoonbills left the region despite the rapid rebound of the fish population. It is possible that cues delivered in past years could have stimulated this migration.

PLANT ECOLOGY

Three plant projects discussed in this chapter focus on hydrologic impacts and restoration goals. This year, the District conducted its first evaluations on a new cattail-specific herbicide to better understand the appropriate dosage needed to minimize nontarget damage and maximize cattail removal. Also, the LILA facility continues controlled research efforts on plant, animal, and hydrology interactions. This year, the results of a two-year groundwater study on two types of tree islands with different plant densities indicated that the plants and underlying geologic conditions play a large role in the hydrologic conditions of tree islands and in the concentration of nutrients within tree island soils. In Florida Bay, the status of submerged aquatic vegetation (SAV) and algal communities are monitored to understand the effects of water management on wetland and estuarine ecosystems to support future updates of the Florida Bay Minimum Flows and Levels rule and to assess the effects of the CERP C-111 Spreader Western Project, an effort designed to minimize seepage from Taylor Slough thus providing increased flow to the southeastern Everglades and Florida Bay. An increase in bottom coverage by benthic macroalgae was measured through much of Florida Bay during early WY2011. Increased SAV community diversity, an operational and restoration target, also occurred in WY2011, with the expansion of widgeon grass in nearshore bays and ponds and shoal grass in many Florida Bay basins.

ECOSYSTEM ECOLOGY

Five ecosystem-scale projects are reported in this chapter. First, an update on the Cattail Habitat Improvement Project (CHIP) — a project designed to provide a better understanding of the effect of physical removal of cattail using herbicides — is provided. CHIP assesses whether creating openings in dense cattail areas will sufficiently alter trophic dynamics such that wildlife diversity and abundance increases, and determines to what extent these created open areas' functions compare to the natural Everglades. Openings consistently supported higher nutrients in the soil than the surrounding cattail habitat. Food web analysis demonstrated that this type of active management for slough recovery can change carbon and nutrient cycling from one based on emergent macrophytes and floc to one based on periphyton and SAV, which supports increased prey biomass within enriched and transitionally enriched areas of the Everglades.

The C-111 Spreader Western Project provides the second through fifth ecosystem-scale updates. It was under construction during WY2011 and is expected to be operational in WY2012. Only preliminary results from this monitoring program are presented. Hydrologic, seagrass habitat, and water quality conditions and factors affecting these conditions are discussed, particularly with regard to the availability of nutrients and occurrence of algal blooms in Florida Bay. Results from the FY2011 assessment of potential water quality trade-offs show that good water quality conditions were widespread in Florida Bay, with no major algal blooms in the water column and low phosphorus concentrations. Finally, soil salinity transects indicated that upstream movement of the "white zone" has occurred.

LANDSCAPE

Two science monitoring and analysis projects are gathering data in new and innovative ways on a relatively small scale. Both are focused on helping the District manage the water in a more holistic manner while providing data for systemwide performance measures and therefore are included in the *Landscape* section. The first project is a paleoecological analysis of soil cores taken across a slough-ridge-tree island habitat transect. This analysis used fossil seeds and other proxies to quantify vegetation changes over the last 100 years as a function of water management. The macrofossil data were critical in providing vegetation reconstructions from 1960 to 1970, encompassing the construction of the L-67 canal and levee (which took place during 1960–1963), and for which aerial imagery was lacking. The timing of increased abundance and aerial expansion of water lily and sawgrass into the island interior from the macrofossil record in the early 1960s was consistent with the hypothesis that the initial phase of L-67 construction reduced tree island area by raising water stages and increasing hydroperiods. The macrofossil data also showed that in the dry years of 1989 and 1990, wax myrtle, a flood tolerant woody species, expanded 25 meters into the slough, indicating a positive expansion of tree island area relative to the 1940 island boundary. This result suggests that hydrologic conditions similar to 1989–1990 may provide operational targets for restoring tree island habitat. The second monitoring project is the development of a unique floccometer monitoring platform in southern Water Conservation Area 3A. In support of a better understanding of the mechanisms that create and maintain the ridge-slough microtopography, the floccometer platform is focused on identifying and quantifying in situ, causal relationships that may link regional-scale Everglades driving forces to local floc behavior, particularly to processes that mobilize floc upward into the water column (entrainment) where it would then be subject to downstream transport.

A third project has been completed and is described in this section. This past year marked the milestone publication of Landscapes and Hydrology of the Predrainage Everglades (McVoy et al., 2011). This study was initiated to support hydrologic and ecological restoration of the Everglades within the context of the drainage history. Given the absence not only of predrainage hydrologic time series measurements but also of predrainage ecological studies, it was deemed important to assemble the best possible sources to yield a qualitative and quantitative picture of more natural Everglades conditions.

Table 6-1. WY2011 (May 1, 2010–April 30, 2011) Everglades research findings in relation to operational mandates.[1]

Projects	Findings	Mandates[1]
Hydrology		
Hydrologic Patterns for WY2011	Abnormally low rainfall for most of WY2011 prevented high water damage to tree islands during the wet season and reversals during the dry season nesting period. The drought was less severe at the southern edge of the Everglades and in Florida Bay than in the Water Conservation Areas (WCAs). With hydrologic buffering associated with the volume of the bay and the lagged effect of the previous year's El Nino, salinity was near average and drought effects were not evident in this region until the end of WY2011. Drought created optimum recession rates and excellent foraging conditions throughout the Everglades Protection Area. However, drought conditions, starting in April and ending in July, created extensive loss of foraging habitat and long periods when the lower tolerances of peat conservation were exceeded.	ROS MFL
Wildlife Ecology		
The Role of Drought on Crayfish Population Dynamics	A simulated drought at the Loxahatchee Impoundment Landscape Assessment (LILA) facility achieved a significant, but modest, reduction of predatory fish populations in the early wet season, which would be expected to accompany natural droughts in the Everglades. As predicted, the seasonal reduction in large-bodied fishes appears to have been responsible for a significant and substantial increase in crayfish densities following the drought. Current work is examining the effect of this increase in prey abundance on wading bird foraging responses.	ROS CERP MFL FEIM
Florida Bay Salinity Transition Zone Prey Base	The Audubon monitoring program found that cold temperatures and high water levels in WY2010 led to low biomass and moderate abundance for the wading bird prey base (fish) community across the southern Everglades transition zone. Results for WY2011 are still preliminary, but prey fish abundance appears to have rebounded to very high levels. Combined with excellent recession rates, these conditions would be expected to attract roseate spoonbills to nest and forage across this region (an important restoration indicator). For unknown reasons, however, few spoonbills nested in Florida Bay in WY2011, although numerous juveniles were spotted post-nesting season taking advantage of excellent foraging conditions across the southern Everglades.	CERP MFL ROS
Plant Ecology		
Cattail Control in a Marginally Invaded Sawgrass Marsh	A one-year herbicide evaluation indicates that aerial application of imazamox at 0.28 kilograms acid equivalent per hectare provides excellent control of cattail in marginally-invaded marsh and slough habitat with only minimal damage to desirable emergent macrophytes or submerged aquatic vegetation (SAV). The selectivity of imazamox represents a significant enhancement in herbicidal control of cattail and will likely increase options for various management and restoration scenarios in the WCAs.	LTP
Tree Islands and Hydrology	A study was completed to examine the relationship between aboveground tree biomass, hydrology, and the concentration of nutrients on eight tree islands. Between the first and second year of this study at the LILA facility, the amount of aboveground tree biomass nearly doubled. With this doubling of biomass, a water table depression developed in the center of each island and created a hydraulic divide along the edge of the island. The data suggest that overlying vegetation and underlying geologic conditions play a large role in the hydrologic conditions of tree islands and explains the process responsible for the concentration of nutrients within tree island soils.	CERP EFA ROS
Florida Bay Benthic Vegetation	An increase in bottom coverage by benthic macroalgae was measured through much of Florida Bay during early WY2011. The mechanism of this increase is not certain, but may be related to winter freezes and unusual rains and wetland runoff during the WY2010 winter (during El Nino conditions). Increased SAV community diversity, an operational and restoration target, also occurred in FY2011, with expansion of widgeon grass in nearshore bays and ponds and shoal grass in many Florida Bay basins. Increases in both macroalgae and seagrasses may have resulted from the dissipation of dense phytoplankton blooms in 2005–2008 that reduced light in the water column.	CERP MFL ROS

Table 6-1. Continued.

Projects	Findings	Mandates[1]
Ecosystem Ecology		
Cattail Habitat Improvement	Open plots consistently supported higher nutrients in the soil than the surrounding cattail habitat. Food web diagrams demonstrated that active management has changed carbon and nutrient cycling from emergent macrophytes and floc to periphyton and SAV, which supports increased prey biomass within enriched and transitionally enriched areas of the Everglades.	LTP ROS
Florida Bay Water Quality Conditions	A major Comprehensive Everglades Restoration Plan (CERP) restoration project, the C-111 Spreader Western Project, was under construction during WY2011 and is expected to be operational in WY2012. Long-term patterns of total phosphorus (TP) and chlorophyll *a* point toward the importance of disturbances, with peaks following Hurricane Irene in 1999 and the 2005 storms.	CERP MFL ROS
Central Lakes Region Sediment-water Nutrient Fluxes	Studies on the dynamics of the western boundary of Taylor Slough (the lakes region between Seven Palm Lake and West Lake) indicate that the finding of high nutrient concentrations and chlorophyll *a* in several Everglades National Park lakes may be more related to the lakes' long water residence time and possibly to other nutrient sources such as groundwater.	CERP MFL ROS
Lakes Phytoplankton Study	In the near term, the C-111 Spreader Canal Western Project will likely reduce water flow into the lower basin and model lands, and increase flows and levels toward the west in central Taylor Slough. In the November sampling at the end of the wet season, cyanophytes were generally more prevalent at nearly all sites, and diatoms were reduced, possibly reflecting a shift in nutrient environment, and upstream to downstream transport from the lakes to the open bay.	CERP MFL ROS
Soil Salinity Transects	High soil salinities were measured in the Triangle Lands east of US 1 (up to 14 inch deep strata) within the C-111 Spreader Canal Western Project footprint. These elevated values are indicative of the effects of decades of water withdrawal and blockage to the area through impoundment.	CERP MFL ROS
Landscape		
Areal Losses and Gains in Tree Island Habitat Over the Twentieth Century – Water Quality Conditions	Paleoecological evidence showed tree island 3AS5 in southern Water Conservation Area 3A (WCA-3A) decreased in size from 1960 to 1965 coinciding with the initial L-67 canal and levee construction (1960–1963). The dating method (utilizing the twentieth century atmospheric bomb-14C) enabled high precision (± 2–5 year) estimates of vegetation changes and corroborated aerial imagery. Fossil data additionally indicated that tree island habitat expanded from 1989 to 1990, suggesting operational targets for habitat restoration. However, rapid loss of tree island habitat occurred within one to two years afterward due to consistently high stages and hydroperiods.	CERP EFA ROS MFL
Floccometer – Quantifying Processes that Maintain the Ridge and Slough	The floccometer is a new complex robotic research platform in southern WCA-3A, custom designed to continuously measure the properties and movement of the loose, unconsolidated particulate material, referred to as "floc", present in the water column of soft water sloughs. Understanding the downstream transport of floc will shed light on the ongoing loss of ridge and slough geomorphology.	CERP MFL EFA
Landscapes and Hydrology of the Predrainage Everglades	South Florida Water Management District scientists and the University Press of Florida published this book in April 2011. As a milestone in Everglades science, the book provides a thoroughly documented "base condition" for Everglades restoration. It details the soils, vegetation, geomorphology, and hydrology of the entire Everglades prior to dredging of the first canals in the 1880s.	CERP MFL ROS EFA FEIM

[1]**Mandates**

CERP	Comprehensive Everglades Restoration Plan
EFA	Everglades Forever Act, Section 373.4592, Florida Statutes (F.S.)
FEIM	Florida Everglades Improvement and Management
LTP	Long-Term Plan for Achieving Water Quality Goals in the Everglades Protection Area
MFL	Minimum Flows and Levels, Section 373.042, F.S.; Chapter 40E-8, Florida Administrative Code
ROS	Regulation and Operational Schedules

HYDROLOGIC PATTERNS FOR WATER YEAR 2011

Fred Sklar, Robin Bennett and Christopher Madden

The amount of rain in the Everglades Protection Area (EPA) for Water Year 2011 (WY2011) (May 1, 2010–April 30, 2011) was substantially less than last year and similar to the drought conditions in WY2009. This year (WY2011) rainfall amounts were significantly below average for all regions as shown in **Table 6-2**. In Everglades National Park (Park or ENP), the rainfall was 5.9 inches less (10.7 percent) than the historical average, and 11.2 inches less (18.5 percent) than last year. Water Conservation Area 3 (WCA-3) experienced the most dramatic deviations from last year and from historic averages of any region. The rainfall in WCA-3 was 10.5 inches less (20 percent) than the historical average, and 19.7 inches less (32.5 percent) than last year. In Water Conservation Areas 1 and 2 (WCA-1 and WCA-2, respectively), the rainfall was 4.13 inches less (8 percent) than the historical average and 11.8 inches more (18.1 percent) than last year.

Table 6-2. Average (calculated by subtracting elevation from stage), minimum, and maximum stage [in feet National Geodetic Vertical Datum (ft NGVD)] and total annual rainfall for WY2011 in comparison to historical stage and rainfall. (See Chapter 2 of this volume for a more detailed description of rain, stage, inflows, outflows, and historical databases.)

Area	Rainfall (inches)		Stage (ft NGVD)						Elevation (ft NGVD)
			WY2011			Historical			
	WY2011	Historic	Mean	Minimum	Maximum	Mean	Minimum	Maximum	
Water Conservation Area 1	43.8	51.96	16.03	14.06	17.00	15.63	10.0	18.16	15.1
Water Conservation Area 2	43.8	51.96	12.04	10.74	13.12	12.53	9.33	15.64	11.2
Water Conservation Area 3	40.9	51.37	9.69	8.06	10.69	9.56	4.78	12.79	8.2
Everglades National Park	49.3	55.22	6.20	4.55	6.90	5.99	2.01	8.08	5.1

One would expect from these below average precipitation values that regional water depths would also have been below average. However, average stage data did not show a significant difference from the historic averages (**Table 6-2**) and instead reveals the importance of examining actual time series and the often meaninglessness of arithmetic means. Despite the 10.7 percent lower than historic average rainfall in ENP for WY2011, the average water depth was 1.1 feet, which was 0.2 feet above the historical average. Lower rainfall than last year resulting in greater than average water depth could be due to (1) lag times associated with hydrologic responses to the high stages in WY2010 (a buffering characteristic at the landscape scale), or (2) desiccation resistance of Everglades peat. Both of these features can be seen by examining the WY2011 hydrographs for each region in comparison to WY2010 and WY2009.

This is the first time in the 15 year history of writing this chapter that three water years are included in the discussion of the ecology of the Everglades. The purpose is to highlight the disconcerting return of a second drought in only three years and discuss the ecological implications of a rare hydrograph, especially in light of the fact that the WY2009 drought was a fantastic year for many species of wading birds, WY2010 flooding was a terrible year for most wading birds, but WY2011 drought was not a good year for the nestlings despite similar dry season water depth changes as observed during the WY2009 drought.

The following hydropattern figures highlight the average stage changes in each of the Water Conservation Areas (WCAs) for the last three years in relation to the recent historic averages, flooding tolerances for tree islands, drought tolerances for wetland peat, and recession rates and depths that support both nesting initiation and foraging success by wading birds. These indices were used by the South Florida Water Management District (District or SFWMD) to facilitate weekly operational discussions and decisions. Tree island flooding tolerances are considered exceeded when depths on the islands are greater than one foot for more than 120 days (Wu et al., 2002). Drought tolerances are considered exceeded when water levels are greater than one foot below ground for more than 30 days, i.e., the criteria for Minimum Flows and Levels (MFLs) in the Everglades (SFWMD, 2003). **Figures 6-1** through **6-7** show the ground elevations in the WCAs as being essentially the same as the threshold for peat conservation.

The wading bird nesting period is divided into three simple categories — red, yellow, and green — based upon foraging observations in the Everglades (Gawlik, 2002). A red label indicates poor conditions due to recession rates that are too fast (greater than 0.6 foot per week) or too slow (less than 0.04 foot for more than two weeks). A red label is also given when the average depth change for the week is positive rather than negative. A yellow label indicates fair conditions due to poor foraging depths (i.e., depths greater than 1.5 foot), slow recession rate of 0.04 foot for a week, or rapid recessions between 0.17 and 0.6 foot per week. A green ("good") label is assigned when water depths decrease between 0.05 and 0.16 foot per week and water depths are between 0.1 and 1.5 feet.

The emphasis on wading birds in these hydrographs highlights their importance in terms of government efforts to restore the Everglades. Other organisms strongly impacted by hydrology include crayfish, fish, deer, rodents, and reptiles, as well as the productivity and competitive interactions of all plants and microbial communities. However, their relationships to hydrology has not been as well defined as those for wading birds.

WATER CONSERVATION AREA 1

Right after an exceptionally smooth and steady recession rate from November 2008 until May 2009 in WY2009 (**Figure 6-1**) — a recession rate that fostered record-breaking nesting and foraging for WY2009 — water levels rose about one foot over a two-month period. This is not an extreme rehydration rate, but just enough to bring optimum foraging conditions to an end. At this late stage in the nesting season, the invertivorous (feeding on invertebrates) white ibises (*Eudocimus albus*), the dominant species nesting in Water Conservation Area (WCA-1), were able to weather the reversal by feeding in the Everglades Agricultural Area and urban environments, and very large numbers of nestlings fledged successfully. Water depths in WCA-1 for the WY2011 dry season followed the same smooth and steady recession rates seen in WY2009, producing highly favorable foraging conditions. However, in WY2011 the dry season began at a lower stage than in WY2009 and depths typically were about 0.4 feet lower in WY2011 for a given point in time. As a consequence, the northern and central region of WCA-1, an area that typically supports very large numbers of nesting birds, dried out prior to the start of the nesting season in March–April, and nesting was limited to colonies in the longer hydroperiod southern third of WCA-1.

The water level changes in WCA-1 during the WY2011 dry season and wading bird nesting season were almost a perfect opposite to that in WY2010. Recession rates starting in December 2010 were excellent, and it appeared that the hydrologic pattern would support early nesting by wood storks (*Mycteria americana*), a goal of the Comprehensive Everglades Restoration Plan (CERP). This recession, along with the fact that the WY2010 dry season was too wet to support extensive foraging and was instead conducive to the recruitment of wading bird prey (small-bodied), set the stage again for record-breaking nesting and foraging for WY2011. However, populations of wading birds using WCA-1 in WY2011 were just about average, new rookeries did not form, and nesting success was normal (see Appendix 6-1 of this volume).

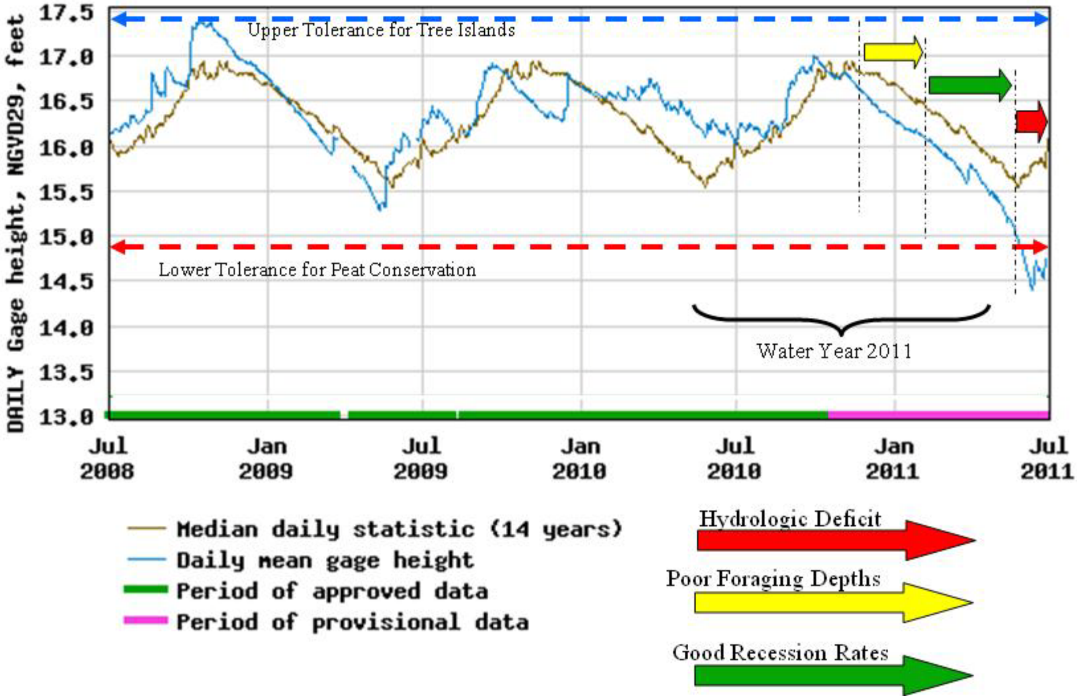

Figure 6-1. Hydrology in Water Conservation Area 1 (WCA-1) in relation to the 14 year median stage [in National Geodetic Vertical Datum of 1929 (NGVD1929)], as well as indices for tree island flooding, peat conservation, and wading bird foraging.

The WCA-1 regulation schedule tends to maintain deeper conditions than the rest of the Greater Everglades. As a result, relatively good nesting and foraging is common in this region during periods of droughts. For WY2011, hydrological conditions in support of wading bird foraging were excellent at the start of the season but the region dried too rapidly leaving limited resources during the middle and tail end of the season.

WATER CONSERVATION AREA 2A AND 2B

It is common for the stage levels during the wet season to exceed the upper flood tolerance for tree islands for 1 to 2 months in Water Conservation Area 2A (WCA-2A) as it did in WY2009 (**Figure 6-2a**) and as it did in the previous three years. Although 1 to 2 months is not considered enough time to cause any long lasting tree island damage (Wu et al., 2002), it is believed that it is also good for tree islands to dry out occasionally (Heisler et al., 2002). This year, water levels during the wet season were 1–2 feet below the upper tolerances for tree islands (**Figure 6-2a**); good for the remaining islands, but a harbinger of poor hydrology to come. Water Conservation Areas 2A and 2B (WCA-2A and WCA-2B, respectively) continue to be the most hydrologically "flashing" regions in the entire EPA. Future efforts to restore WCA-2A tree islands will require a closer examination (i.e., frequency analysis) to see if this kind of hydropattern can enhance the return of woody species to these marshes.

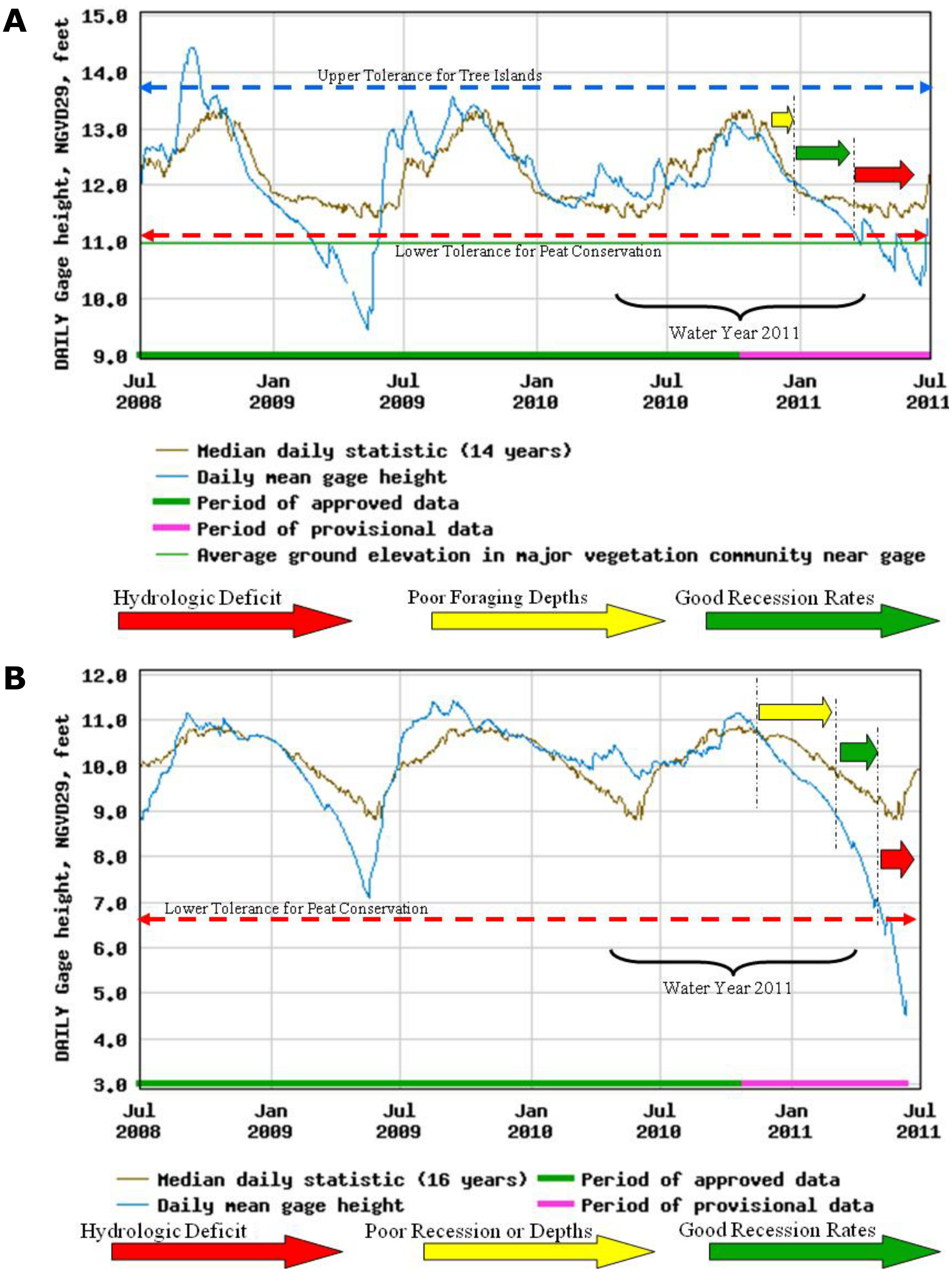

Figure 6-2. Hydrology in (A) Water Conservation Area 2A (WCA-2A) in relation to the recent 14-year average stage and (B) Water Conservation Area 2b (WCA-2B) (gauge 99) in relation to the recent 16-year average stage with indices for tree islands, peat conservation, and wading bird foraging.

In WY2010, WCA-2A did not dry out and good recession rates were short-lived and bracketed by periods of reversals and deep water (**Figure 6-2a**). Some foraging was observed but not as much as previous years. Last year's dry season was probably a period of prey rejuvenation because the previous four water years had periods of complete drydown, which some believe is needed to remove the large predatory fish that limit the crayfish populations that support wading birds (see the *Wildlife Ecology* section in this chapter). In WY2011, WCA-2A had excellent recession rates for most of the entire nesting period and wading bird foraging was extensive until it ended when the lower tolerance for peat conservation was reached in mid-April. Perhaps fish prey did indeed get more abundant during the very wet dry season of WY2010 because foraging was very successful during the WY2011 dry season.

WCA-2B tends to be utilized by wading birds during droughts because it tends to stay deeper for longer periods than the rest of the EPA. This was true in WY2009 when dry season water levels went below ground in WCA-2A and northern Water Conservation Area 3A (WCA-3A), and the wading birds moved to WCA-2B. Unfortunately, in WY2011, when dry season water levels went below ground in WCA-2A and northern WCA-3A, water levels in WCA-2A went below ground shortly thereafter (**Figure 6-2b**) due to the drought, and did not provide an end-of-season foraging area.

WATER CONSERVATION AREA 3A

In the northeastern region of WCA-3A (gauge 63), WY2011 began relatively deep in May and June (**Figure 6-3a**) and was expected to be a very wet, wet season. However, La Nina stabilized (see Chapter 2 of this volume), causing rainfall to drop and water levels to stay below-average for almost the entire water year. This provided perfect wading bird foraging conditions. Recession rates were excellent starting in December (good for wood storks) and optimum depths started to appear around January 2011. Avian scientists recorded extremely large numbers of foraging birds in this important northeastern section of WCA-3A from early January until March 2011, when surface water disappeared. Unfortunately, as a consequence of the dry conditions, birds did not initiate nesting at the Alley North colony where annual nesting during the past decade has frequently exceeded 20,000 nests. Soil moisture during the month of April, May, and June got critically low and posed a muck fire threat to the Alley North colony. Fortunately, no muck fires occurred and rain in July removed the forest fire hazard in the region. Birds responded to the dry conditions at Alley North by nesting at another local colony located to the south of Interstate 75 (I-75), Sixth Bridge colony, where hydroperiods are longer. However, nest numbers were much reduced relative to those typical at Alley North. Of particular interest was the large number of roseate spoonbills (*Ajaja ajaja*) that nested successfully at this colony. This species typically nests in the coastal habitats of Florida Bay and nesting of this magnitude has not previously been detected in the freshwater Everglades.

The WY2010 hydrologic pattern in central WCA-3A (gauge 64) was not very conducive for wading bird foraging. Like most of the Everglades, WY2010 in the central Everglades was probably a good year for prey rejuvenation. Moving into WY2011, water depths were average and did not exceed the upper tolerances for tree islands (**Figure 6-3b**). Good recession rates, supporting maximum foraging behavior and nesting, started in November 2010 and lasted for a full five months, similar to the drought conditions in WY2009. Large flocks of wading birds were observed following the receding drydown fronts in central WCA-3A during both the WY2009 and WY2011 droughts. However, WY2009 was a record year and WY2011 was below average. In WY2011, the region dried down sooner than in WY2009, leaving less available foraging habitat at critical stages of the nesting season. By mid-April, when many birds are close to hatching their eggs, most of the system was already dry. As the prey density data are processed over the next year or so, it will be valuable to see if these differences were due to lack of prey, lack of hydrological buffering capacity in the extant system, or both.

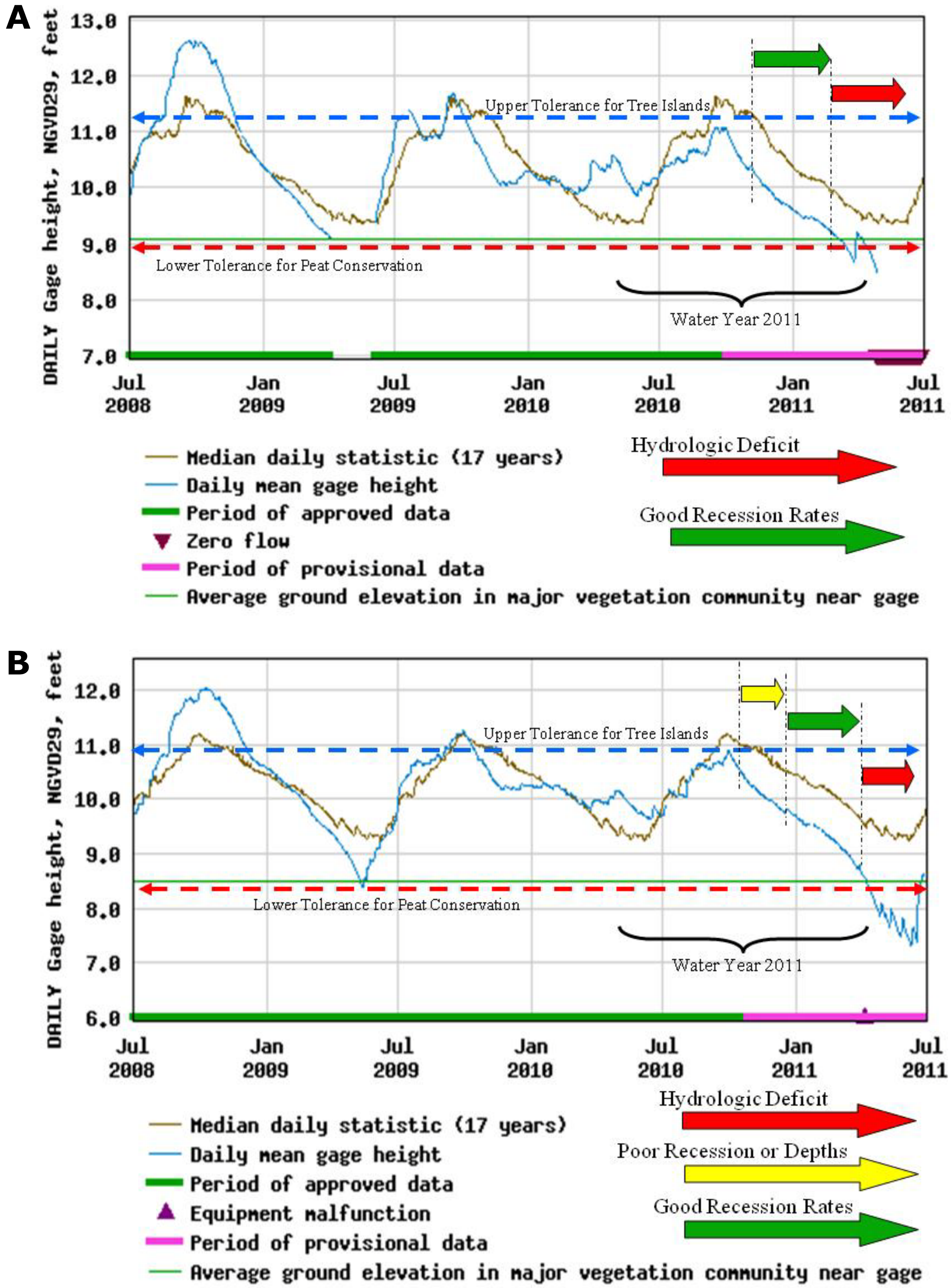

Figure 6-3. Hydrology in (A) northeastern Water Conservation Area 3A (WCA-3A) (gauge 63) and (B) central WCA-3A in relation to the recent 17-year average stage with indices for tree islands, peat conservation, and wading bird foraging.

WATER CONSERVATION AREA 3B

During the WY2009 drought, water levels fell in Water Conservation Area 3B (WCA-3B) at an almost steady, perfect 0.1 foot per week during the dry season. The setup for optimum March and April foraging by wading birds could not have been better. WY2010 was a very different story. Like everywhere else in the EPA, there was an abrupt water level rise in May and June 2009, followed by an almost flat and deep dry season with numerous reversals and a very poor recession rate. WY2011 depths were about half a foot lower than WY2009, but as in WY2009, water levels fell at an almost steady, perfect 0.1 foot per week during the dry season. Foraging in support of nesting was initially outstanding. However, late season drydowns were almost two feet below ground and lasted for months (**Figure 6-4**), and late season foraging was very limited. Again, as the prey density data are processed over the next year or so, it will be valuable to see if these differences between drought years were due to lack of prey or lack of hydrological buffering capacity in the extant system or both.

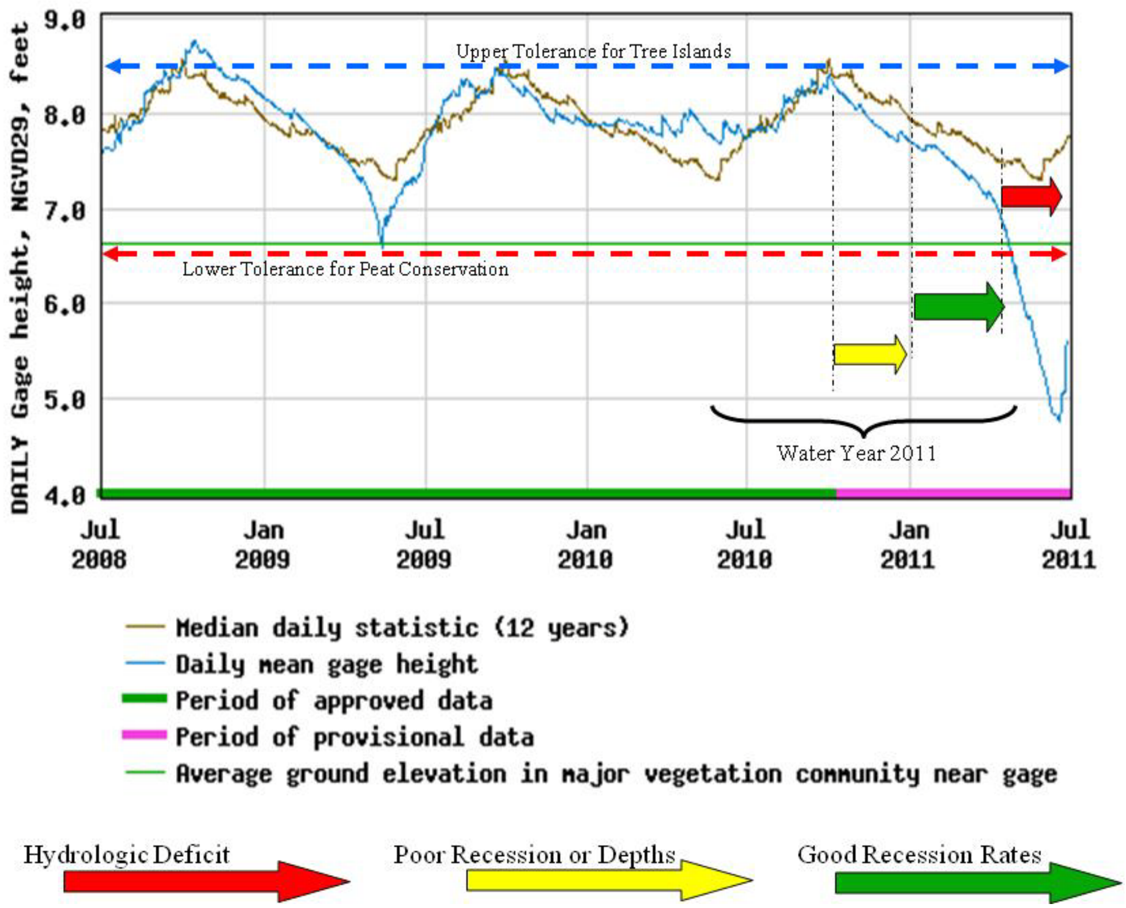

Figure 6-4. Hydrology in central Water Conservation Area 3B (WCA-3B) (gauge 71) in relation to the recent 12-year average stage and indices for tree islands, peat conservation, and wading bird foraging.

NORTHEAST SHARK RIVER SLOUGH

The WY2010 dry season had good recession rates in northeast Shark River Slough for one month (January). The rest of the time, and despite the relatively good water depths, the water levels did not decrease and did not support wading bird foraging. Like the rest of the EPA, this might have been a period of prey rejuvenation due to the lack of predation intensity from foraging wading birds. Although not all the WY2011 data has been processed yet, it appears that the nesting and foraging in this region of ENP was average despite the excellent dry season recession rates and the similar water level changes observed during the WY2009 drought (**Figure 6-5**), when high numbers of wood storks and ibises were found foraging throughout the ENP and nesting success in northeast Shark River Slough was also high.

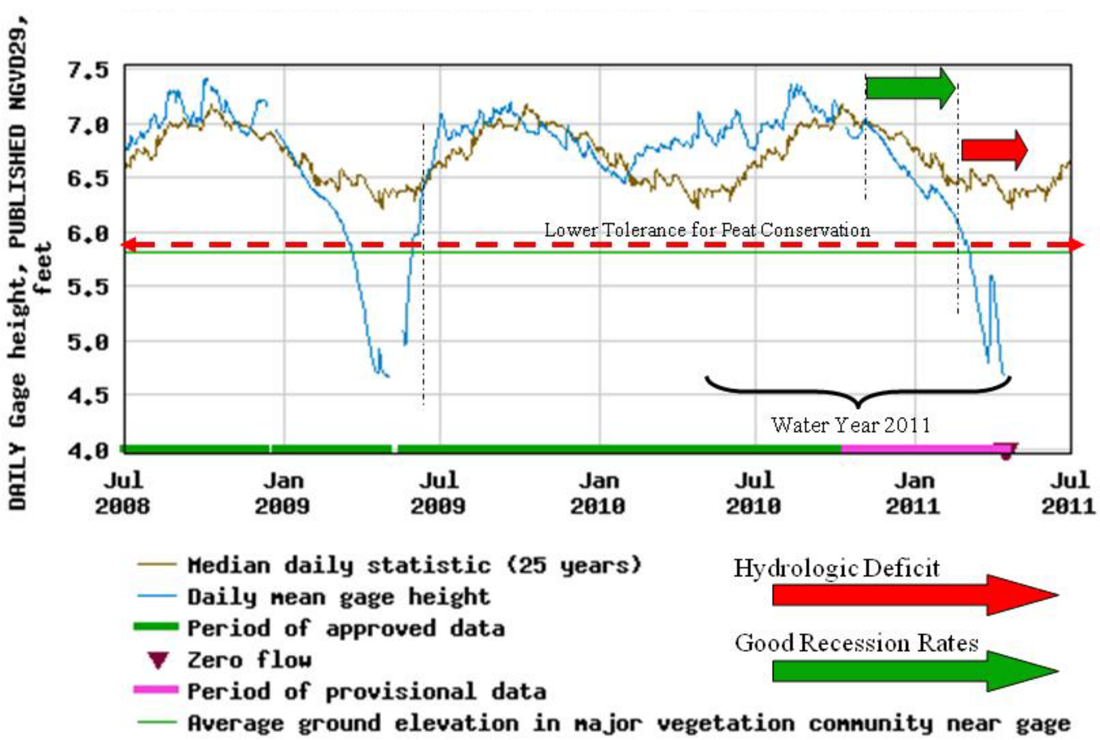

Figure 6-5. Hydrology in northeast Shark River Slough in relation to the recent 25-year average stage with indices for tree islands, peat conservation, and wading bird foraging.

The 2009 and 2011 hydrographs for this section of ENP were extremely similar. Wet season water depths were about 1.2 feet both years, and dry season water levels were below ground for almost the same amount of time. However, wading bird nesting success was significantly different, especially for wood storks. It is likely that nesting was successful in WY2009 and not in WY2011 because WCA-3 dried down too rapidly in WY2011.

FLORIDA BAY

This year was notable for a marked disconnect between the hydrologic conditions of the southern and northern EPA. The south experienced a relatively high degree of buffering, as well as a residual effect from the previous year's El Nino high water conditions, preventing the mangrove transition zone from getting dry and creating near average quantities of freshwater input and near average seasonal salinities for Florida Bay.

ENP basin rainfall was more than 10 percent below average in WY2011. Rainfall onto Florida Bay proper totaled approximately 46 inches, which is just above average (44 inches) for this southernmost portion of the EPA. Significant rain events in September 2010 and January 2011 helped minimize the bay's water budget deficit. Rainfall in September alone made up 25 and 36 percent of the annual total rainfall for the ENP and Florida Bay, respectively, and was most plentiful over the southern part of ENP (with lower accumulations in Shark River Slough), helping this region to better withstand the subsequent drought.

Creek inflows to Florida Bay were near average for the entire WY2011 hydrologic cycle. Five creeks that feed eastern and central Florida Bay (**Figure 6-6**) represent just over 80 percent

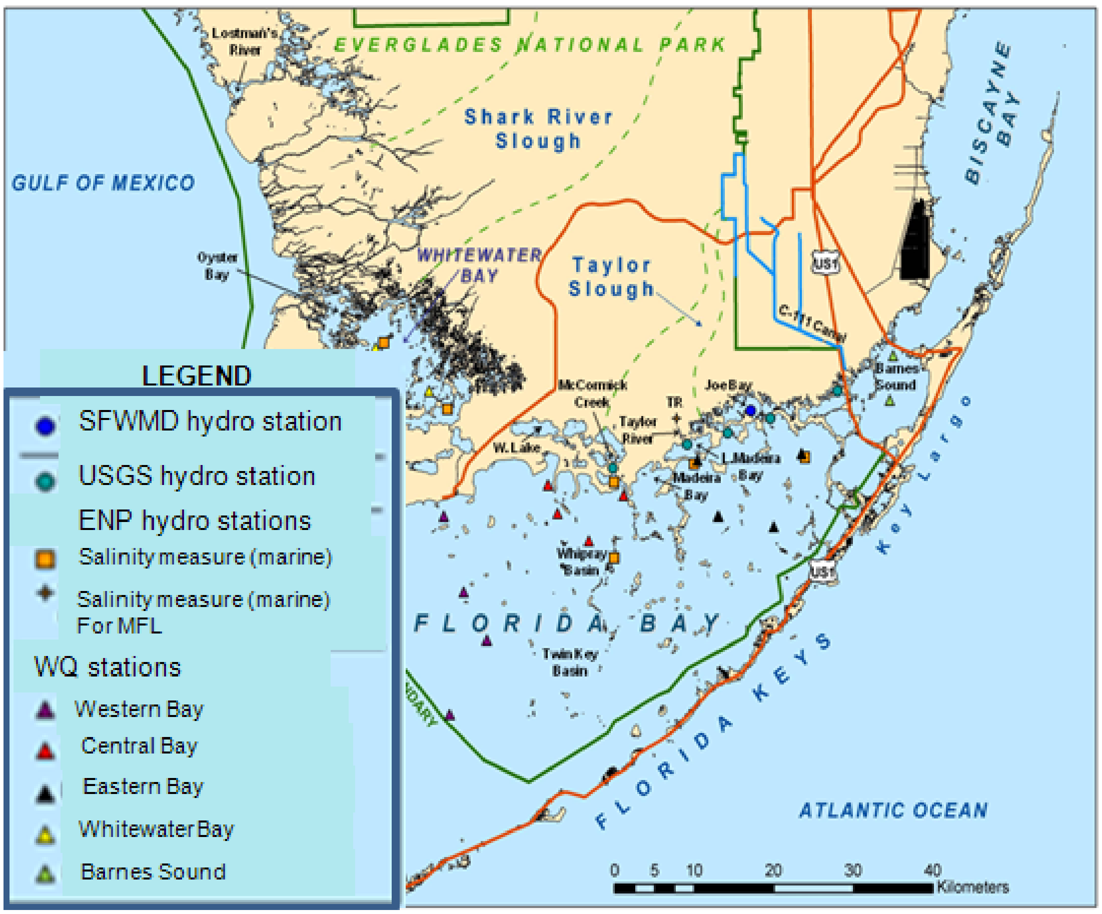

Figure 6-6. Map of Florida Bay and southern Everglades showing areas from Whitewater Bay in the west to Barnes Sound in the east influenced by water management operations. Stations are monitored by the South Florida Water Management District (SFWMD), United States Geological Survey (USGS) and Everglades National Park (ENP).

of total measured flow to the bay. **Figure 6-7** shows that a delayed onset to the WY2011 wet season is reflected in the below average June and July 2010 inflows, and helps explain why annual inflow for these five creeks [227,500 acre-feet (ac-ft) or 28,100 hectare-meter (ha-m)] was approximately 10 percent lower than the long-term annual average [WY 1997–WY2009 average of 251,800 ac-ft (31,100 ha-m) per year]. The 365-day cumulative five creek flow never approached the 105,000 ac-ft (13,000 ha-m) threshold monitored as part of the Florida Bay MFL rule, remaining well above 200,000 ac-ft (24,700 ha-m) for the entire WY2011 [aver-aging 282,600 ac-ft (34,900 ha-m)].

Figure 6-8 shows that the spatial distribution of these creek inflows across the coast was similar to WY2010, with just over 20 percent of the five-creek cumulative flow moving through the two western-most creeks, McCormick Creek and Taylor River. With an explicit goal to move more water down Taylor Slough versus through the C-111 canal, the CERP C-111 Spreader Canal Western Project will assess whether the trajectory of this trend continues moving upward as the project is implemented.

Salinity in Florida Bay is complex and the result of many factors including freshwater flow from the Everglades, precipitation, evaporation, groundwater exchange, exchange with the marine waters from the Gulf of Mexico and Atlantic Ocean, and internal circulation. Because Florida Bay is shallow and its circulation is restricted by mud banks, it is susceptible to rapid and abrupt changes in salinity sometimes creating hypersalinity events that affect the biology and chemistry of the bay. Data are collected continuously at stations in ENP's Marine Monitoring Network, and monthly as part of the District's coastal water quality monitoring network, providing information on spatial and temporal trends in salinity throughout the bay. Salinity for representative ENP Marine Monitoring Network stations (Duck Key and Little Madeira Bay for eastern Florida Bay; Terrapin Bay and Whipray Basin for central Florida Bay) were averaged with monthly grab salinity data collected in the corresponding region for a given sample date.

Similar to the trend for creek inflows in WY2011, Florida Bay salinity was near average throughout WY2011 (**Figure 6-9**, top and middle). Despite a late start to the WY2011 rainy season, extremely wet El Nino conditions from the previous WY2010 dry season maintained relatively low salinity in May and June 2011. Salinity was only briefly above average in WY2011, well within the interannual range of 5–10 practical salinity units (psu) and only approached hypersaline conditions (over 40 psu) in the central bay in April 2011, following a very dry spring.

Low salinity conditions prevailed in the mangrove ecotone through WY2011 as well. The salinity buffer of the WY2010 El Nino event is especially evident in **Figure 6-9** (bottom), showing ENP Taylor River salinity data. It is noteworthy that in calendar year 2010, the seasonal salinity increase did not occur here until June (versus in March, as is more typical) and was an abbreviated event. In addition, despite the severe WY2011 drought, salinity in this area did not increase until April 2011 (again, versus in March). This is important because the ENP-Taylor River platform is the indicator site for the Florida Bay MFL rule that establishes a 30 psu threshold for the 30 day running average salinity. The last two exceedances of this threshold occurred in May 2008 and April 2009. With no salinity exceedances in the 2010 calendar year, an MFL violation (two or more consecutive years with exceedances within a ten-year period) was avoided. Prolonged hot and dry conditions by April 2011, however, led to rapidly rising salinity across the ecotone, showing that the regional drought started impacting the southern Everglades by the end of WY2011.

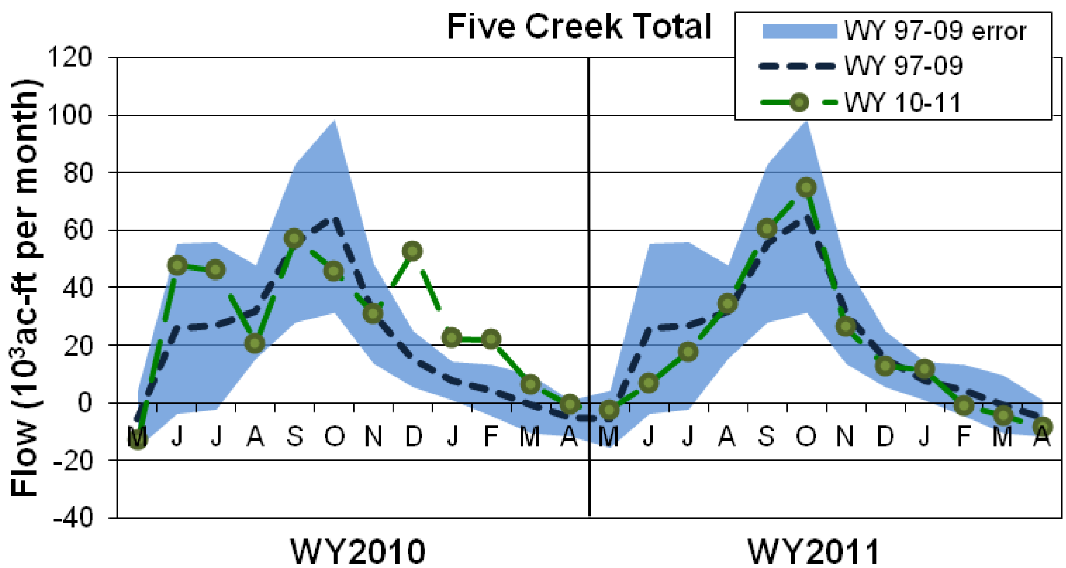

Figure 6-7. Monthly cumulative discharge to Florida Bay through five creeks in WY2010 and WY2011 ("WY 10-11" in the legend), compared to monthly discharge mean ("WY 97-09") and standard deviation envelope ("WY97-09 error") from WY1997–WY2009. Data after September 2009 are provisional, supplied courtesy of the United States Geological Survey.

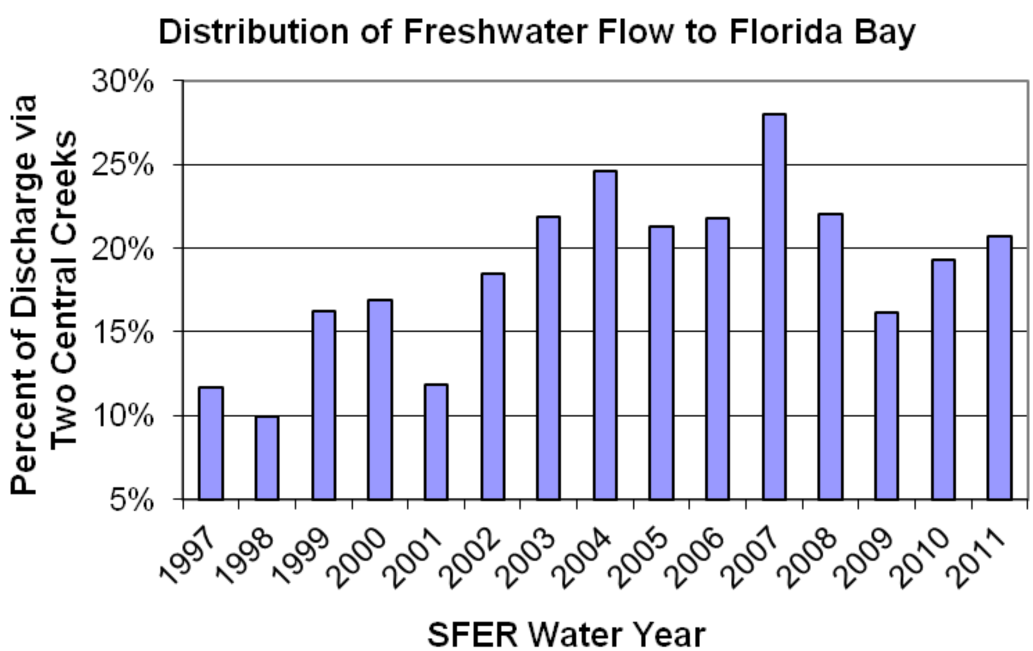

Figure 6-8. Time series ratio of the proportion of annual creek discharge measured in five creeks via the two most western creeks (McCormick Creek plus Taylor River).

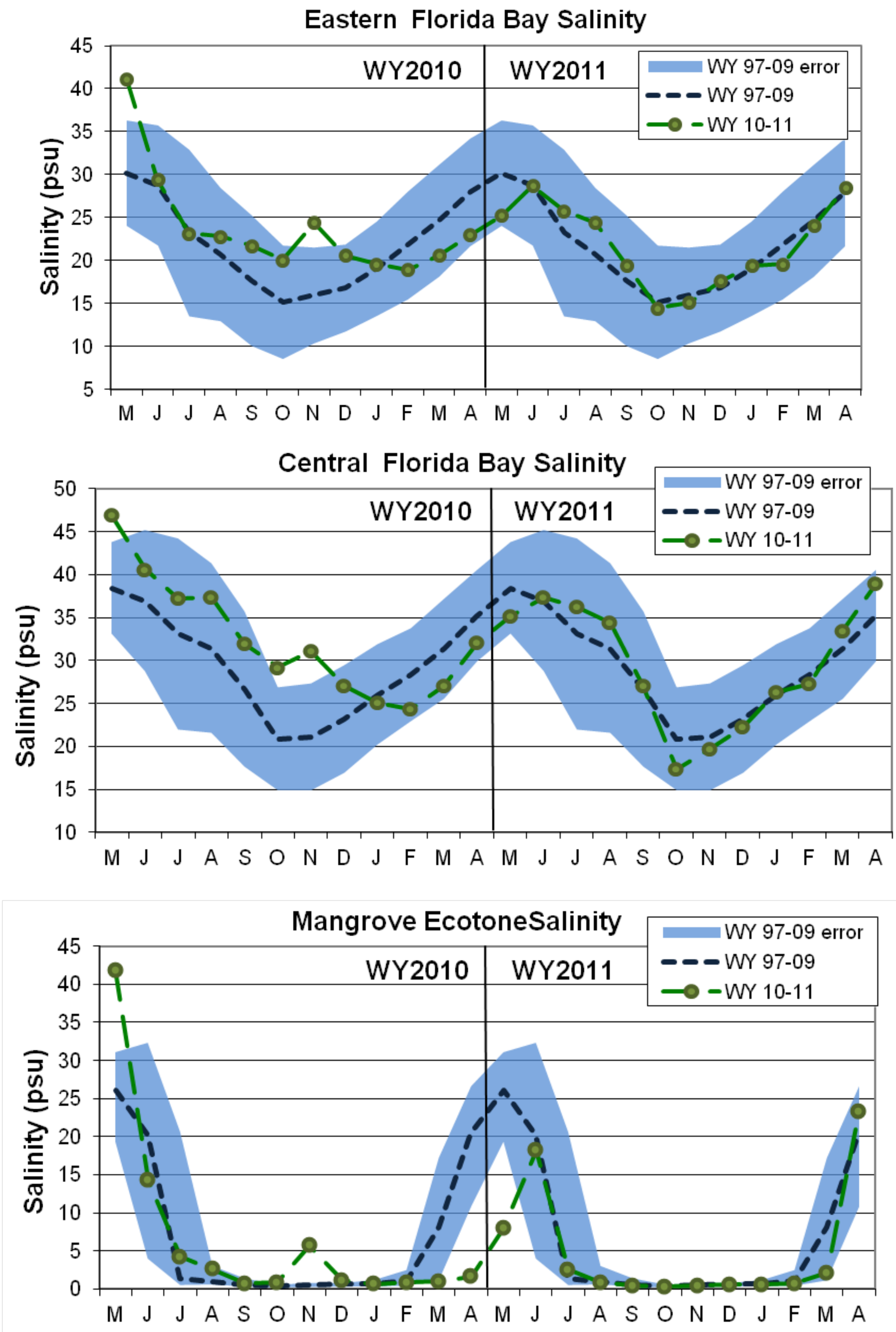

Figure 6-9. Mean monthly salinity values in (top) eastern Florida Bay(top), central Florida Bay (middle), and the mangrove ecotone at the ENP-Taylor River platform (bottom) in WY2010 and WY2011 ("WY 10-11" in the legend), compared to monthly means ("WY 97-09") and standard deviation envelope ("WY 97-09 error") from WY1997–WY2009.

Synoptic Salinity Mapping

Using high speed mapping technology (Madden and Day, 1992), the physico-chemical parameters of Florida Bay are mapped quarterly to determine how upstream conditions affect the distribution of fresh water in the bay. **Figure 6-10** shows that the transition bays still retained fresh water (e.g., Joe Bay) in the middle of the dry season, and the bay proper was marine to hypersaline with salinities in the mid-thirties to low forties. By July, after wet season rains had begun, lower salinities were distributed throughout the bay, ranging from the low twenties to mid-thirties, with the exception of the western bay, which was above marine salinity, coincident with data shown in **Figure 6-9** (top and middle) where a 10-point differential in salinity exists between eastern and central bay salinity monitoring stations. Interestingly, during this period, the transitional bays were not much fresher than the main bay, and much saltier than during the previous dry season, particularly in Joe Bay, Long Sound, and Little Blackwater Sound showing that full hydration of upstream areas had not yet occurred, and giving a view of the sometimes months-long lag between seasonal rains and the freshening of Florida Bay.

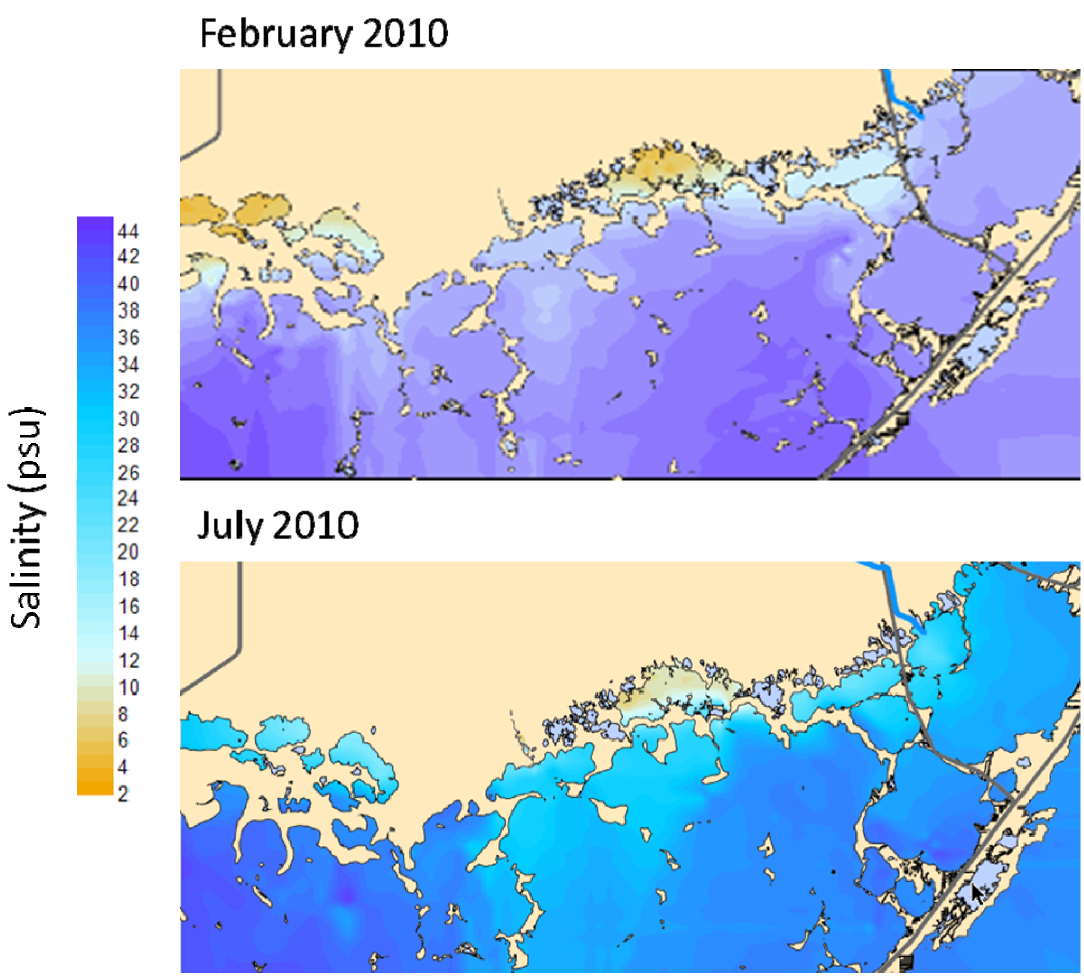

Figure 6-10. Data flow synoptic maps of salinity in Florida Bay and transition zone bays during the February 2010 dry season and the July 2010 wet season.

WILDLIFE ECOLOGY

Robin Bennett, Eric Cline, Mark Cook,
Nathan Dorn[2] and Robert M. Kobza[3]

Wading bird survival and nesting success is dependent upon the available food supply. A project conducted in the Loxahatchee Impoundment Landscape Assessment (LILA) facility is described, which is designed to quantify the effect of drought years on crayfish (*Procambarus fallax*) population dynamics, an important food item for white ibis. Additionally, Florida Bay fish production and wading bird success is described.

INCREASED CRAYFISH DENSITY FOLLOWING SIMULATED DROUGHT AND FISH REDUCTION AT LILA

Hydrology has at least two functional roles in regulating the trophic transfer of biomass to wading birds and other mega fauna in the Everglades. As Gawlik (2002) points out, appropriate nesting season water depths are crucial for making the prey in the sloughs available for foraging birds. But hydrology has another function during the prey production phase prior to concentrating prey. Drying events, the time since drying, and flooding extent can affect prey species composition and abundance (Trexler et al., 2005; Dorn and Trexler, 2007; Trexler and Goss, 2009). In short, historical hydropatterns and appropriate seasonal depths together determine food web interactions, and therefore wading bird nesting success. Resolving the relationship between hydropattern and prey production is therefore fundamental for the development of restoration targets and associated hydrologic management objectives.

One of CERP's goals includes the maintenance of large numbers of nesting wading birds and the frequent return of exceptionally large white ibis nesting events (Frederick et al., 2009). Frederick and Ogden (2001) argued that the historical wading bird super colonies typically developed one to two years after strong regional droughts. They hypothesized that aquatic predator reduction and/or nutrient release were responsible for prey enhancement following extensive wetland drying. While they were uncertain how prey might be enhanced by drying, Ogden et al. (2003) later suggested crayfish might be dynamically linked to drought and the formation of exceptionally large nesting events. All the evidence for pulsed production of wading birds to date has been correlative and the understanding of the mechanisms leading to pulsed production has not been generally studied.

Over the past two years, aquatic animal communities in the LILA impoundments have been quantified as part of a multiyear investigation of the effects of drying and predatory fish reduction on crayfish population sizes and the densities of other prey species (*Wildlife Ecology* section of the *2010 South Florida Environmental Report* (SFER) – Volume I, Chapter 6). It was hypothesized that the drying related reductions of predatory fishes (Parkos et al., 2011) will allow crayfish populations to recruit more successfully (Dorn, 2008; Kellogg and Dorn, in review) and achieve higher densities during the following year. In turn, this is hypothesized to provide better foraging opportunities for birds that feed on crayfish, such as the white ibis (Dorn et al., 2011). Smaller-bodied fishes (e.g., smaller killifishes) could also be enhanced after predator reduction (Frederick and Ogden, 2001) although other observations suggest this is not occurring in the Everglades (Trexler et al., 2005).

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[2] Florida Atlantic University, Davie, FL
[3] Contributed as SFWMD staff during the draft SFER production cycle.

The main objective of this study was to experimentally test the predator-release hypothesis (Frederick and Ogden, 2001). The specific aims were to (1) measure aquatic prey (e.g., crayfish and small fishes) densities and predator abundances (i.e., large-bodied fish) through time and in response to the experimental drying in two of the LILA macrocosms that occurred in May 2010 and (2) measure the bird foraging responses to the variability in prey productivity (data not presented here).

Methods

From July 2009 to March 2011, large-bodied predatory fishes (predatory fishes) and small animal densities (small fishes and crayfish) were quantified in all four macrocosms at LILA (See Aich et al., 2011 for details of the LILA facility). During the wet and dry seasons (July and March) of each water year, trap nets were set and recovered for three consecutive nights in the sloughs of each macrocosm to estimate predatory fish activity and density (catch per unit effort). One trap effort was defined as the sum of the average catch for each of the three trap types on a given night and nights were treated as replicates in the analysis. One square meter (m^2) throw traps were also used to quantify the population sizes of smaller fish and crayfish in the LILA sloughs. In the same sample seasons, 15 throw trap samples were collected from each of the wetland macrocosms. Traps were placed in both deep (n=11) and shallow (n=4) slough habitats each season. The sampling was conducted at times when the water was 0–10 centimeters (cm) deep on the ridges but at depths too deep for extensive wading bird foraging in the sloughs (~25–60 cm depending on the slough).

In May 2010, sloughs of two macrocosms (M3 and M4) were dried, and remaining predatory fishes in the deepwater ponds and ditches were removed by a combination of netting and Rotenone. The fish removal served to simulate the depression of predatory fish populations as would be expected during a drought in the Everglades ridge and slough landscape, where deepwater habitats are less prevalent.

Predatory fish catches were analyzed to determine whether our manipulation significantly reduced their abundances in the dried cells and to determine the temporal scale of the reduction. All fish greater than 5 cm long (standard length) were considered large-bodied and predatory in the analysis because they are capable of feeding on young-of-year crayfish. Log-transformed predatory fish abundance (nightly catch) was analyzed with repeated measures of analysis of variance (ANOVA) following a before-after control impact (BACI) design with nested factors in Proc Mixed (SAS 9.2, SAS Institute). In this experiment, two control sites (i.e., wetland macrocosms) and two impacted sites (Underwood and Chapman, 2003) were established. The models included terms to partition variation to macrocosms within treatments, treatments (control or dried designations), periods (water years before versus after experimental drought), and time within each period (July versus March). The focal interaction terms in the BACI design are the treatment x period interaction and the treatment x time(period) interaction; the two terms testing for differences between the treatments that change through time as a function of the impact (i.e., experimental drying). For the predatory fish catches, there was interest in whether catches were reduced in the dried wetlands compared to the controls after the experimental drought.

Crayfish and small-bodied fishes are prey of the predatory fishes and their densities were analyzed using individual throw trap counts as the response variables (i.e., replicates within sites, *sensu* Underwood and Chapman, 2003). Within-macrocosm spatial variation was not modeled explicitly except that a term for the slough was included where the trap was placed (deep versus shallow) because crayfish densities were believed to be generally higher in the shallower and more densely vegetated shallow sloughs. The other terms in the model were similar to those explained in the analyses of the predatory fish catches. Again, there was interest in changes in small fish or crayfish densities in dried wetlands compared to control wetlands from before to after the simulated drought. Small-bodied throw trap counts were square-root transformed before analysis and crayfish counts were analyzed using Proc GENMOD in order to specify the response distribution as a negative binomial (i.e., over dispersed counts).

Results and Discussion

Predatory fish catches were variable between macrocosms and through time (P-values < 0.001) such that catches in the dried macrocosms were reduced in July (treatment x time(period): P < 0.001; **Figure 6-11**), but not for the entire water year after the drought (treatment x period: P = 0.21). In fact, predatory fish were somewhat more abundant in the dried macrocosms by the following March (March contrast: P = 0.02; **Figure 6-11**). In July 2010, after the simulated drought, the predatory fish catches were reduced 42 and 90 percent in the two dried macrocosms relative to the mean of the controls.

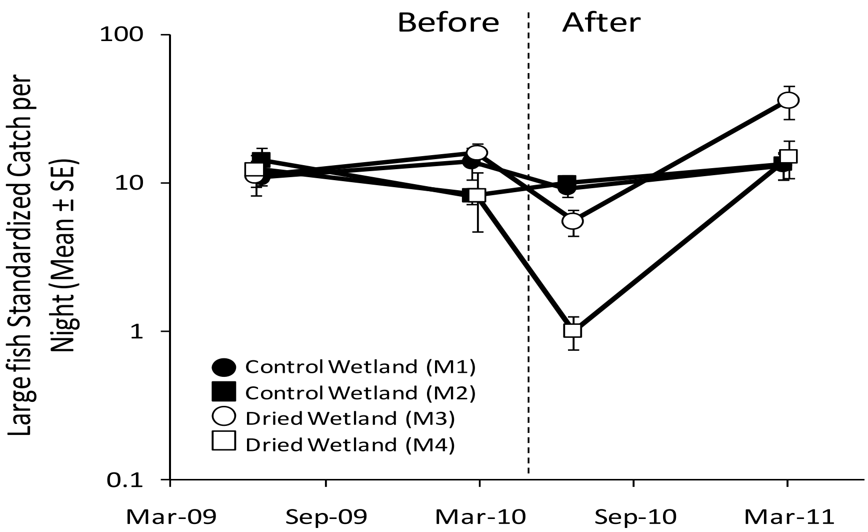

Figure 6-11. Mean ± standard error (SE) standardized nightly catch of large bodied fishes [greater than five centimeters (cm) standard length] in the four wetland macrocosms of the Loxahatchee Impoundment Landscape Assessment (LILA) facility. Fish were captured in three different gear types set in the sloughs and were released each day. SE is a temporal SE based on three consecutive nights of trapping each sample season. The vertical dashed line indicates the time of the experimental drying and fish reduction.

The model results also indicated that the two dried macrocosms differed from one another during the after period [period x macrocosm (treatment): P = 0.05] and predatory fish were more abundant in M3 than in M4 after the simulated drought (P = 0.005; **Figure 6-11**). Other contrasts between macrocosms within treatments were nonsignificant in both periods (P-values > 0.1). These results suggest the experimental removal in M4 was more successful than in M3, but both removals were successful in creating lower large-bodied fish populations.

Small-bodied fish densities (all species combined) varied between macrocosms (P = 0.01), but dried macrocosms did not vary systematically from control macrocosms through time (treatment x period: P = 0.14; treatment x time(period): P = 0.71). The simulated drought and pulsed predatory fish reduction neither stimulated nor reduced densities of small-bodied fish in the LILA macrocosms (**Figure 6-12A**). In contrast, the crayfish density was higher in the dried macrocosms after the simulated drought (**Figure 6-12b**) (treatment*period: P < 0.001,

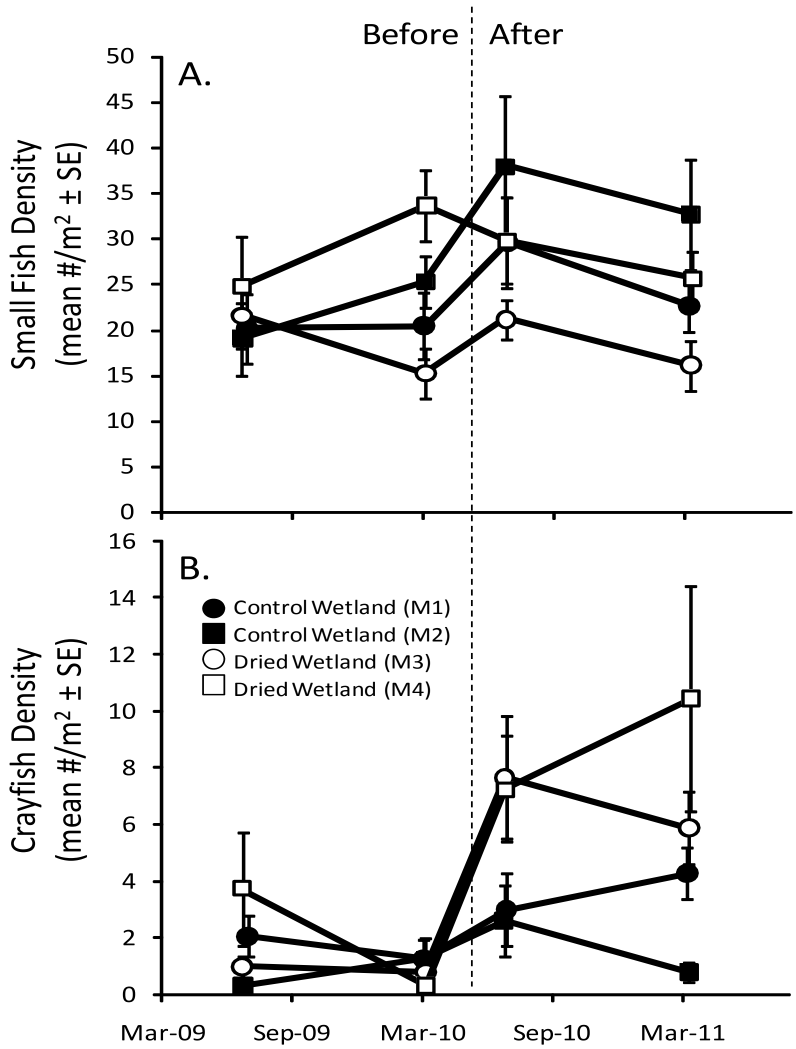

Figure 6-12. Densities ± SE of (A) small-bodied (all species combined) and (B) crayfish in the four LILA wetland macrocosms over two water years. The error bars are standard errors for 15 throw traps taken from each of the macrocosms on each date. The dotted line indicates the timing of the experimental drought when large-bodied fishes were reduced.

treatment x time(period): P = 0.02). Also significant variation was observed among the macrocosms within a treatment after the drought, but primarily in March. The crayfish densities remained the highest in March in M4 (**Figure 6-12B**) and this was the same macrocosm where predatory fishes were most effectively reduced by the simulated drought (**Figure 6-11**). Crayfish densities did not differ between treatments in the year leading up to the drought (contrast: P = 0.93), but were nearly three times higher in the dried macrocosms (μ = 7.8 per m^2) compared to the controls (μ = 2.7 per m^2) in the year after the simulated drought (contrast: P < 0.001).

The results of this study indicate that a significant but modest reduction in predatory fishes in the early wet season was achieved through a simulated drought, as also would be expected during natural droughts in the Everglades (Parkos et al., 2011). The seasonal "pulsed" reduction in predatory fishes appears to have been responsible for a significant and substantial increase in crayfish densities in the water year following the drought.

Relevance to Water Management

Understanding the strong linkages among hydrologic patterns, aquatic prey populations, and wading bird reproductive success is a critical CERP objective and essential for operational management. An important, yet understudied, component of this relationship is the role of antecedent hydrologic conditions on the aquatic food web and its resulting effect on the abundance of key wading bird prey species. This study provides evidence that extreme drought conditions had an important influence on the structure of the aquatic food web by decreasing predatory fish populations. This, in turn, had the predicted effect of increasing production of slough crayfish, a critical prey component of a number of wading bird species, including the most abundant species in the ecosystem, white ibis. These results have two important implications for management. First, periodic droughts appear to be important natural disturbance events in the Everglades ecosystem and may be critical for white ibis super colony formation. Current work at the LILA facility is exploring this idea further by examining wading bird foraging responses to the observed increase in crayfish abundance. The second is that large-bodied fishes appear to compete with wading birds for food resources by feeding on juvenile crayfish and thereby cutting off recruitment of adult crayfish (i.e., bird prey). This may be of particular relevance as it suggests hydrologic conditions that currently promote large-bodied fish populations, such as long hydroperiods in the ponded regions of the WCAs and the deep water of canals adjacent to marshes, may be functioning to reduce wading bird prey populations.

PREY BASE OF THE FLORIDA BAY SALINITY TRANSITION ZONE

Audubon of Florida's Tavernier Science Center has an extensive network of stations (**Figure 6-13**) at which this organization collects continuous hydrologic data (salinity and water level), performs bimonthly submerged aquatic vegetation (SAV) surveys, and samples prey base fishes eight times per year. (Hydrologic and SAV results for WY2011 are addressed in other sections of this chapter.) Water level is an important factor affecting the regional food web, especially with respect to the bay's roseate spoonbill population. Audubon conducts prey base fish sampling across the mangrove ecotone in wading bird foraging areas of southern ENP. The period of record for some stations extends back to WY1991, while others were started in WY2006. Details on drop trap methods used to sample mangrove fish communities can be found in Lorenz et al. (1997); data analysis details can be found in Lorenz (1999) and Lorenz and Serafy (2006). The latest data provided in this chapter were from WY2010, with anecdotal information described from periodic quarterly reports provided by Audubon staff to the District.

WY2010 had unusual hydrologic conditions, dominated by El Nino conditions that resulted in well above average water levels and well below average salinity across the ecotone (2011 SFER – Volume I, Chapter 6, *Hydrologic and Climate Trends* section; Frezza et al., 2011). Based on results in Lorenz (1999) and Lorenz and Serafy (2006), such conditions would generally favor increased density and biomass of prey base fish. While this was the case for a few sites when results were aggregated for the entire WY2010 (**Figure 6-14**), most sites showed results near their respective median values. Biomass and density of prey fish were actually quite low at all sites during the winter months of WY2010, coincident with very high water levels, low salinity, and extreme cold temperatures. The January 2010 cold snap that resulted in major fish kills across South Florida may have altered prey fish species composition at many of these shallow ecotone sites. For instance, densities of normally abundant exotic cichlids (e.g., Mayan cichlid, *Cichlasoma urophthalmus*) dropped to near zero for the remainder of WY2010 at many sites, while other species (Centrarchids) increased (Frezza et al., 2011). The typical community shift one expects as salinity rises, from freshwater-oligohaline species (as defined by Lorenz and Serafy, 2006) early in the water year to mesohaline and polyhaline species later in the water year, was interrupted because of the unusual WY2010 hydrology. Overall for WY2010, oligohaline

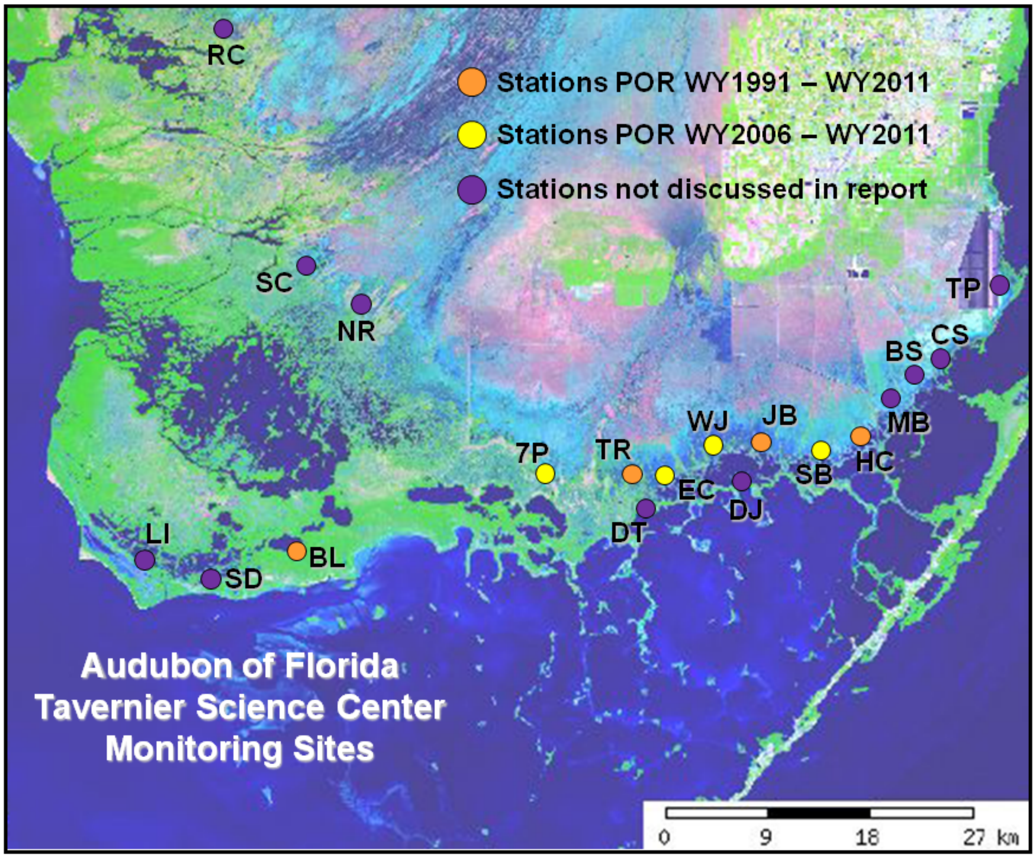

Figure 6-13. Map of monitoring sites in the Audubon Tavernier Science Center network for period of record (POR) WY1991–WY2011 and POR WY2006–WY2011. All sites shown in yellow and orange have continuous hydrologic monitoring (water level and salinity), submerged aquatic vegetation (SAV) surveys, and prey fish sampling (with the exception of BL, which has all but SAV surveys). Data for stations colored purple were not available for this report.

prey fish species, dominated by rainwater killifish (*Lucania parva*) and mosquitofish (*Gambusia affinis*), composed the highest proportion of the catch at all sites (Frezza et al., 2011). These opportunistic species have low biomass compared to other species that were lost in WY2010 (e.g., cichlids), and may thus explain some of the lower biomass results for WY2010 in the bottom panel of **Figure 6-14**. As water levels finally began receding off the mangrove flats habitat in April 2010, very high numbers of prey fish were caught across the mangrove ecotone sites.

The effects of the unusual conditions in WY2010 persisted well into WY2011, with many months of above average water level and below average salinity in the ecotone. Fish samples were still being processed for WY2011 at the time this report was being prepared, but anecdotal results from quarterly status reports suggest numbers of prey fish were well above average for WY2011 (Audubon, 2011). Moreover, the prey fish themselves were larger than normal, perhaps attaining higher biomass with increased SAV cover (see the *Florida Bay Submerged Aquatic Vegetation Community* section within this chapter) and lower salinity conditions (reduced osmotic stress). The fact that prey fish numbers were so high is notable given a prolonged reduction in numbers of exotic cichlids since the WY2010 cold snap. Furthermore, at several

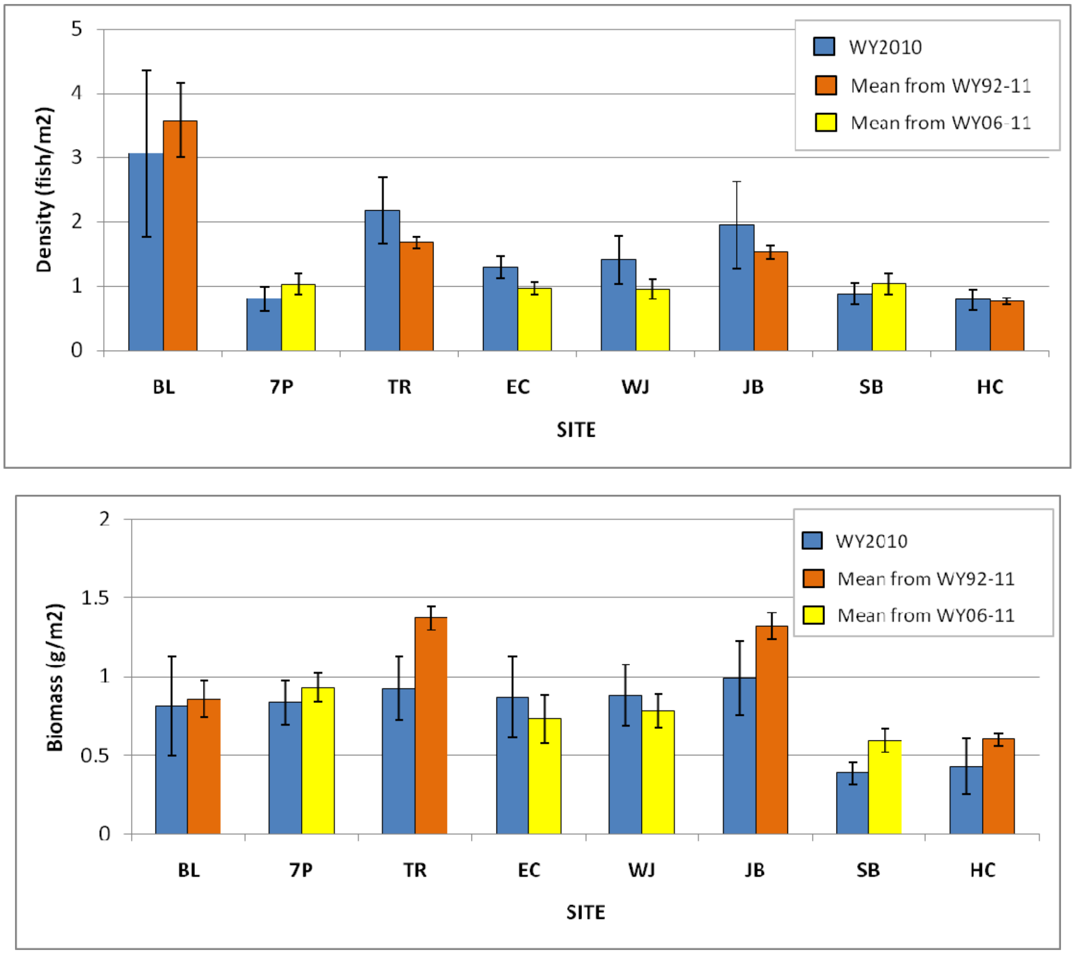

Figure 6-14. Comparison of prey fish density (top) and biomass in grams per square meter (g/m2)(bottom) for WY2010 (latest prey fish data available) versus mean values from long-term sites (designated in orange WY1992–WY2011) and those brought into Audubon's network more recently (in yellow; WY2006–WY2011). Error bars represent standard errors for each site.

sites, many freshwater killifish species (Cyprinodontids) continued to be collected well into the WY2011 dry season, when communities typically transition to mesohaline and polyhaline with increased salinity. It appears that the WY2010 effects were to bolster the freshwater-oligohaline prey fish community in WY2011, resembling expected outcomes under restoration scenarios examined for the C-111 Spreader Canal Western Project.

Relevance to Water Management

The community of small prey base fishes in Florida Bay's salinity transition zone is considered to be an excellent ecological indicator because it changes rapidly in response to changing hydrologic and habitat conditions and is important in the regional food web, especially in sustaining the bay's roseate spoonbill population. Monitoring of this prey base by Audubon is supported by the District in order to assess responses to changing operations and the C-111 Spreader Canal Western Project, and provide information to evaluate and potentially update the Florida Bay MFL Rule.

PLANT ECOLOGY

David Black, Eric Cline, Thomas Dreschel, Christopher Madden, Amanda McDonald, René M. Price[4], LeRoy Rodgers and Pamela Sullivan[4]

Plant studies form an important basis for evaluating restoration success. This section evaluates a selective herbicide that targets cattail, which has invaded enriched areas of the Everglades. The herbicide is designed to eliminate cattail while not damaging other native wetland plant species. In addition, a second project is described. This project characterizes the effect of tree growth and transpiration on the local groundwater hydrology of tree islands by monitoring stage levels in groundwater wells. Thirdly, a description of SAV and macroalgae in Florida Bay is presented.

CATTAIL CONTROL IN MARGINALLY INVADED SAWGRASS MARSH

The proliferation of cattail (*Typha* spp.) in the Everglades is attributed to increased phosphorus (P) levels in the soil and increased water depth and duration of flooding (Newman et al., 1998). Monospecific stands of cattail have replaced the historic sawgrass (*Cladium jamaicense*) marsh ridge and slough landscape over nearly 12,500 hectares (ha) in the Everglades (2011 SFER – Volume I, Chapter 6, *Landscape Processes* section). Everglades restoration has primarily focused on reductions in nutrient concentrations and restoration of hydroperiods, but recent efforts also have investigated a means to actively reduce cattail dominance in severely impacted areas. For example, the Cattail Habitat Improvement Project (CHIP) is investigating methods to rehabilitate cattail-invaded areas using a combination of herbicide and fire (for project details, see the *Ecosystem Ecology* section of this chapter). Most herbicides approved for cattail control in aquatic environments are broad spectrum (nonselective) herbicides and are likely to impact desired emergent macrophytes and some SAV species. In CHIP, a combination of glyphosate and imazapyr effectively eliminated all emergent vegetation in dense cattail stands, creating open water habitat with subsequent colonization by muskgrass (*Chara* spp.), a native macroalga. However, use of these herbicides in less impacted areas, where cattail has yet to displace other species, may be counterproductive due to nontarget impacts to the remnant flora. If herbicidal control of cattail is desired along the leading front of invasion, a more selective herbicide that effectively controls cattail without damaging desirable native species is preferred.

The herbicide imazamox [2-[4,5-dihydro-4-methyl-4-(1-methylethyl)-5-oxo-1H-imidazol-2-yl]-5-(methoxymethyl)-3-pyridinecarboxylic acid] was registered for aquatic use in 2007 under the trade name Clearcast™. Imazamox is a selective, systemic herbicide that kills plants by binding to and inhibiting activity of the acetolactate synthase enzyme, leading to lethal reduction in branched chain amino acid biosynthesis (Shaner et al., 1984). Animals lack acetolactate synthase and obtain branched chain amino acids from their diets so they are not affected by this chemical activity of imazamox. Differences in sensitivity to imazamox between different plants are apparently due to the ability of certain species to metabolically detoxify the herbicide in addition to variations in structure of acetolactate synthase that affect binding and inhibition processes and other poorly-understood factors (Délyea, C. et al. 2011).

Potential injury to animals and nontarget plants is always concerns when applying herbicides in the Everglades. A thorough human health and ecological risk assessment for imazamox

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(SERA, 2010) analyzed available data and supports the concept that imazamox is safe enough for certain uses in natural areas. For example, research in support of United States Environmental Protection Agency registration found that imazamox concentrations must be three orders of magnitude greater than the maximum application rate before toxicity effects are observed in fish [rainbow trout (*Oncorhynchus mykiss*)], aquatic invertebrates (Daphnia), waterfowl [mallard duck (*Anas platyrhynchos*)] and honey bees (Apidae). It was not possible to detect significant toxic effects to mammals, even at high doses. The chemical properties of imazamox indicate that significant bioaccumulation is highly unlikely. No herbicide-related wildlife impacts were noted in the course of this study; however it was not designed to detect toxicity to fauna.

The environmental effect most likely to cause problems would be injury to nontarget plants. The present study addresses this potential problem to some extent and future studies will continue to address this issue. Fortunately, imazamox is neither highly persistant nor very mobile. The half life of imazamox in water ranges from five to 15 days with the length dependent upon water clarity, depth, and available sunlight. Dilution and photolytic breakdown are the primary means of dissipation in water (SERA, 2010). As discussed below, aerial herbicide applications in this study would result in very low concentrations in the water column.

To date, there is limited published data concerning control of freshwater aquatic species using imazamox (Koschnick et al., 2007; Wersal and Madsen, 2007; Mateos-Naranjo, 2009). Clearcast^{TM} is recommended for control of cattail at a rate of 32 to 64 ounces per acre. Expressed as kilograms acid equivalent per hectare (kg ae ha^{-1}), the recommended rates range from 0.28 to 0.56 kg ae ha^{-1}. Initial field evaluations conducted by the District in 2008 indicated that aerial applications of imazamox at a rate of 0.28 kg ae ha^{-1} resulted in substantial control of cattail with little to no damage observed on other emergent macrophytes. These results suggested a potential low rate use of imazamox for selective control of cattail in Everglades marsh habitat. The objectives of this study were to determine the dose response of cattail and other common emergent species to aerially-applied imazamox in a marginally invaded cattail-sawgrass marsh.

Methods

Field test plots were established and treated in fall 2009. The plots were located in WCA-3A, south of Alligator Alley (I-75) in a 75 ha area centered at 26.1386° latitude, -80.5694° longitude. Treatment plots were set up as a randomized complete block. Five experimental blocks were established along a "cattail expansion zone" where cattail was co-dominant in a sawgrass marsh ridge and slough mosaic. Each block was divided into four 0.40 ha treatment plots [40 by 100 meters (m)], which were randomly assigned an imazamox application rate of 0.28, 0.14, 0.07, or 0 (control) kg ae ha^{-1}. These rates align with Clearcast^{TM} rates of 32, 16, 8, and 0 ounces per acre, respectively. Aerial herbicide applications were conducted by helicopter in September 2009. The herbicide was diluted with 187 liters of water per ha, and two adjuvants (0.15 liters < DLZ^{TM} per ha and 0.05 L Nufilm^{TM} per ha) were added to enhance herbicide control and reduce spray drift. The herbicide was applied evenly over each treatment plot with no attempt to avoid application to nontarget species.

Percent cover of emergent plant species and species richness were measured in each treatment plot three weeks prior to and 12 months after treatment. For species richness, two 5-by-90 m belt transects were established along the long axis of each plot, five meters from the plot edges. Presence of all emergent and submerged plant species were recorded in these belt transects. For species cover, six two-by-one meter quadrats were randomly placed in each treatment plot. To minimize effects from surrounding plots, random quadrat coordinates within five meters of the plot boundary were discarded. Quadrats were delineated using a polyvinyl chloride (PVC) pipe frame. Plant species cover was independently estimated by two observers using cover classes (< 1, 1–5, 6–25, 26–50, 51–75, and > 75 percent). Standing dead biomass was not included in species cover estimates, but total necromass was estimated for each quadrat. For each quadrat, the independent cover estimates of each species were averaged. Percent cover estimates for each of the six quadrats (subplots) were then averaged to obtain a cover estimate for each treatment plot replicate (n=5). The change in cover for each species was calculated as the difference between post- and pretreatment percent cover. The proportional change in cover was obtained by dividing the change in cover by the initial cover. The responses of cattail and five common emergent native species to treatments were examined with a one-factor randomized block analysis of variance with multiple comparisons. A log transformation was applied when data were not normally distributed or variances were not homogenous. Where the treatment effect was significant, individual treatments were compared using Tukey's honest significance test. Interaction of block and treatment were tested before examining treatment main effects.

Results and Discussion

Prior to herbicide treatments, the species with the highest percent cover (± 95 percent confidence interval), averaged across all plots, included sawgrass (12.5 ± 3.7), southern cattail (*Typha domingensis*; 8.0 ± 3.6), fragrant water lily (*Nymphaea odorata*; 6.3 ± 6.0), pickerelweed (*Pontederia cordata*; 1.5 ± 1.3), bog smartweed (*Polygonum setaceum*; 0.9 ± 0.7), and duck potato (*Sagittaria lancifolia*; 0.3 ± 0.3). Other detected species with mean percent cover less than 0.1 included buttonbush (*Cephalanthus occidentalis*), muskgrass, Gulf Coast spikerush (*Eleocharis cellulosa*), Egyptian panicgrass (*Paspalidium geminatum*), swamp rosemallow (*Hibiscus grandiflorus*), Virginia saltmarsh mallow (*Kosteletzkya virginica*), marsh mermaidweed (*Proserpinaca palustris*), big floatingheart (*Nymphoides aquatica*), maidencane (*Panicum hemitomon*), green arrow arum (*Peltandra virginica*), Carolina willow (*Salix caroliniana*), bladderworts (*Utricularia* spp.), American cupscale (*Sacciolepis striata*), and southern cutgrass (*Leersia hexandra*). At 12 months after treatment, cattail cover decreased 99, 81, and 61 percent in the 0.28, 0.14, and 0.07 kg ae ha^{-1} treatment plots, respectively. Cattail cover increased 52 percent in the control plots. There was a block by treatment interaction ($p < 0.007$) for proportional change in cover of cattail. A significant difference ($p < 0.0001$) in change in cattail cover was detected between the control plots and treatment levels (**Figure 6-15**). Despite a 61 percent reduction in cattail cover in the 0.07 kg ae ha^{-1} treatment plots, most plants continued to maintain viable leaf tissue and exhibited new growth 12 months after treatment. Moderate reductions in cattail cover also occurred in the 0.14 kg ae ha^{-1} plots, although much fewer resprouting plants were observed relative to the 0.07 kg ae ha^{-1} treatment plots. The substantial, but incomplete control of cattail at 0.14 kg ae ha^{-1} suggests that further refinement of a minimum effective rate between 0.28 and 0.14 kg ae ha^{-1} is warranted.

Change in percent cover at 12 months after treatment did not vary significantly between treatment plots for sawgrass, fragrant water lily, pickerelweed, swamp smartweed, and duck potato, suggesting the dominant emergent macrophytes in this study are resistant to imazamox at rates less than or equal to 0.28 kg ae ha^{-1}. **Figure 6-15** shows the proportional change in cover at 12 months after treatment for these common species. Occasional herbicide damage was noted on sawgrass and pickerelweed in the 0.28 kg ae ha^{-1} plots, but the damage appeared to be associated with areas where herbicide spray overlapped. Some minor leaf spotting and yellowing was occasionally observed at 12 months after treatment on sawgrass in the 0.28 kg ae ha^{-1} treatment plots, but new growth from the basal meristem was frequently observed and no evidence of herbicide activity in the new tissue was noted. It is possible that imazamox exerts a growth regulating effect on sawgrass and other plant species at these lower rates, but no such effects were detected in this experiment. The lack of a strong herbicide response for pickerelweed differs with reports of imazamox sensitivity by Koschnick et al. (2007), who report damage to this species at moderate to high dosing rates (150–300 micrograms per liter). This discrepancy is likely explained by differences in treatment rates and application methods (aqueous solutions versus

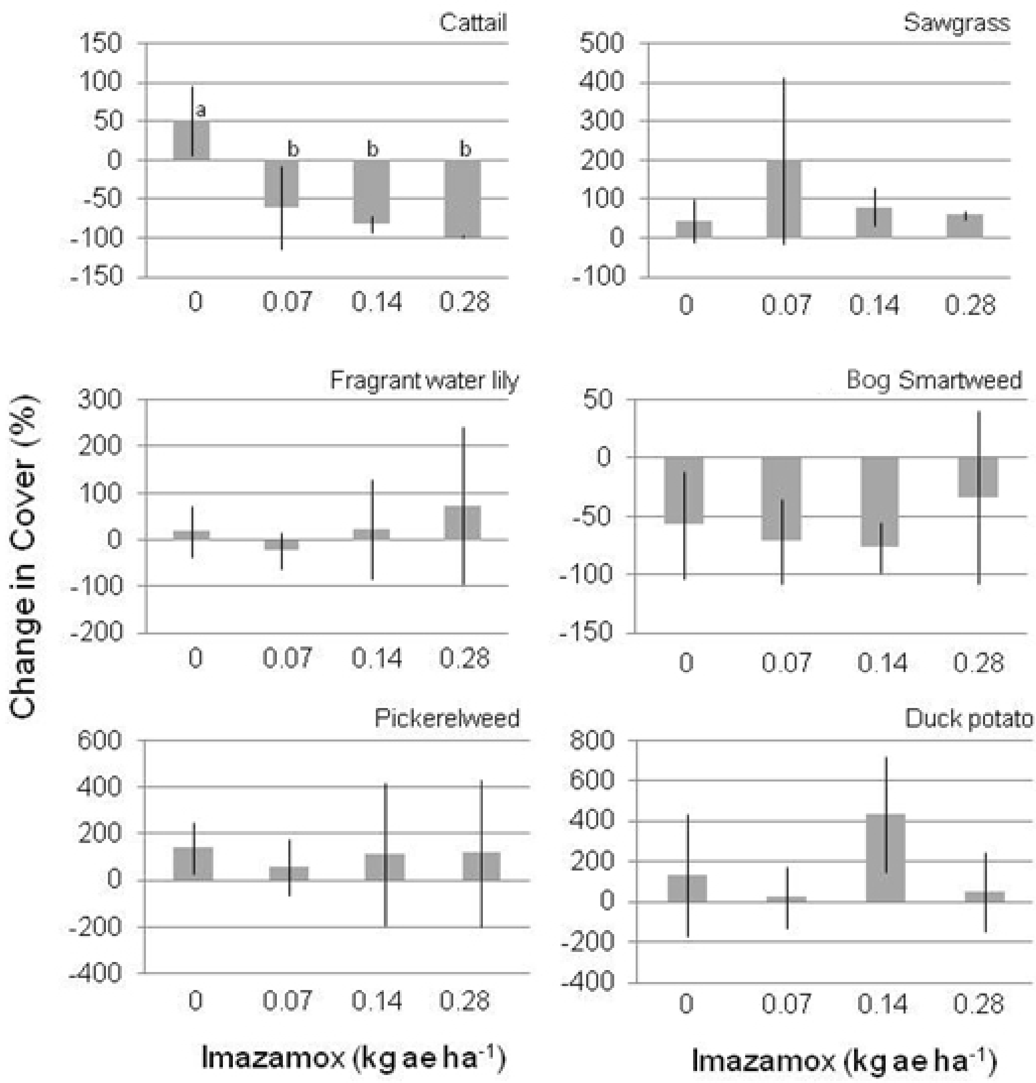


Figure 6-15. Mean proportional change in cover 12 months after treatment for six common macrophytes at four imazamox rates. The vertical lines represent the 95 percent confidence intervals. Negative changes represent decreases in mean cover.

aerial application). In this field study, the moderate cattail canopy and relatively low application volumes (187 liters per ha) may have prevented adequate herbicide contact on pickerelweed and other sparse species. Higher levels of imazamox sensitivity by pickerelweed would likely occur at higher application volumes or more direct application. In addition, the fate of individual plants was not tracked, so there is no confirmation whether the pickerelweed population was initially impacted followed by recruitment of new plants.

While other plant species sampled in treatment plots were found too infrequent to conduct statistical analyses, anecdotal evidence suggested differential sensitivity among species to imazamox at 0.28 and 0.14 kg ae ha^{-1} application rates. At these higher rates, Carolina willow and buttonbush frequently showed evidence of herbicide damage, although plants remained alive and exhibited some new growth with minor expression of herbicide activity. In contrast, Gulf Coast spikerush and maidencane showed no evidence of herbicide injury at 12 months after treatment.

Koschnick et al. (2007) also report imazamox resistance of maidencane. Bladderworts and muskgrass also remained common in the slough portions of the plots. Prior to herbicide treatments, species richness estimates ranged from 9.2 to 10.2 species per 0.09 ha. Species richness estimates were very similar 12 months after treatment, ranging from 9.2 and 10.0 species per 0.09 ha. There was no significant difference in species richness estimates between treatments.

Conclusion

The single aerial application of imazamox at a rate of 0.28 kg ae ha^{-1} provided excellent control of cattail in marginally-invaded marsh ridge and slough habitat with only minimal damage to desirable emergent macrophytes (**Figure 6-16**). The apparent selectivity of imazamox for cattail control is a promising development in ongoing efforts to manage cattail in impacted regions of the Everglades. Specifically, selective control of cattail in marginally-infested marsh ridge and slough mosaic could serve to slow the rate of invasion of cattail. Of course, cattail control in a nutrient enriched and hydrologically altered wetland is merely dealing with the symptom, not the underlying cause. Restoration of the Everglades ultimately relies on the reversal of widespread eutrophication of a formerly oligotrophic landscape. In fact, it is likely cattail will recolonize treated areas without successful reductions in soil P concentrations and restoration of predisturbance hydrologic regimes. Nonetheless, judicious use of imazamox may be an effective tool to reduce cattail dominance in lightly to moderately infested areas. Imazamox may also be a preferred alternative to glyphosate/imazapyr treatments in dense cattail stands, since reduced herbicide sensitivity may result in accelerated colonization of desirable species.

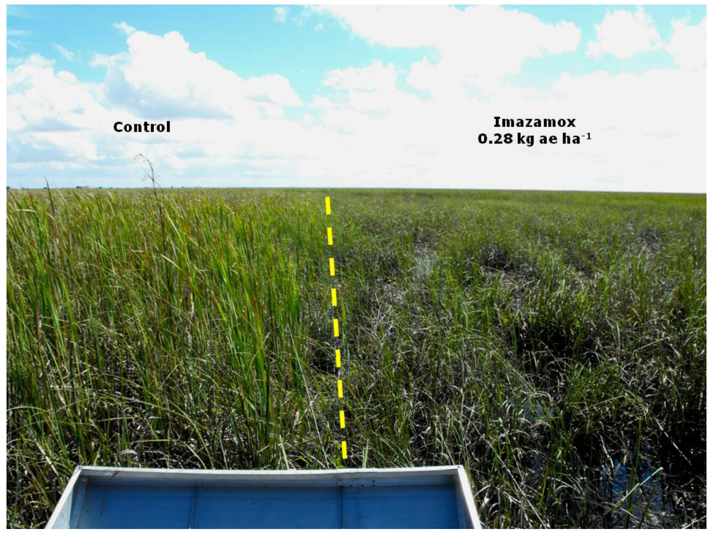


Figure 6-16. Comparison of control plot with the highest imazamox rate [0.28 kilograms acid equivalent per hectare (kg ae/ha)] 12 months after treatment. Cattail is nearly 100 percent controlled in the plot to the right, while sawgrass and other emergent species persist (photo by the SFWMD).

Before large-scale herbicide applications are implemented, additional study is recommended to determine imazamox sensitivity of additional plant species commonly occurring in the Everglades marsh ridge and slough mosaic and to determine in situ imazamox degradation rates and products. The species assemblage observed in this study is typical of the ecosystem, but numerous ubiquitous species were either absent or very sparse in this study (e.g., maidencane). A more complete understanding of imazamox sensitivity among the larger complex of native plant taxa would allow for more informed decision making with regard to plant community impacts and alterations. In addition, information on cattail recolonization rates following imazamox treatments is needed to determine minimum retreatment intervals.

Relevance to Water Management

The accelerated rehabilitation of cattail-impacted areas is a recent focus of scientists and land managers engaged in Everglades restoration. Specifically, active cattail management through combinations of herbicides and prescribed fire are being evaluated as potential restoration tools to shift cattail-dominated marsh to alternative native plant communities (see the 2011 SFER – Volume I, Chapter 6, *Ecosystem Ecology* section). To date, available herbicides for cattail control (e.g., glyphosate and imazapyr) have only been feasible for application in monospecific cattail stands, where impacts to other emergent macrophytes are less of a concern. Findings of this study suggest that imazamox controls cattail at moderate to low aerial application rates with little to no herbicide damage to many ubiquitous emergent macrophytes of the Everglades marsh ridge and slough mosaic. The selectivity of imazamox represents a significant enhancement in herbicidal control of cattail and will likely increase options for herbicide control in other cattail management scenarios such as in marginally infested areas.

TREE ISLANDS AND HYDROLOGY

Variation in groundwater evapotranspiration rates may be one of the largest driving factors in groundwater–surface water interactions and, thus, the formation of landscape patterning across ecosystems of low topographic relief (Eppinga et al., 2008; Wetzel et al., 2005; Rietkerk et al., 2004). Groundwater–surface water interactions strongly influence the chemistry of shallow groundwater and the location and patterns of vegetation in wetlands (Ferone and Devito, 2004; Glaser et al., 1981). A number of researchers have hypothesized that areas with high groundwater evapotranspiration rates lower the water table, creating an inward convective transport of nutrients and ions, which increases the growth rate of biomass, and leads to an accelerated rate of soil accretion. The elevated ion and nutrient concentrations are thought to have positive feedback on local biomass while negatively impacting vegetation at a greater distance by inhibiting access to resources (Rietkerk and van der Koppel, 2008). The main objective of this research was to investigate the hydrodynamics of tree islands in their early stages of development.

Study Area

The LILA facility consists of four eight-ha macrocosms (M1 through M4), which mimic Everglades ridge and slough, and tree island landscape features. Each macrocosm contains two tree islands with different underlying geologic materials, one consisting entirely of peat (peat islands) and one having a limestone rubble core overlaid by a thin layer of peat (limestone islands). Each tree island has nine groundwater wells with an average depth of 1.34 m, the wells on each island are arranged in three transects that extend from each edge and cross the center of the tree island. Over 700 saplings were planted on each tree island consisting of eight woody species common to the Everglades. Tree islands in M1 and M4 were planted in March 2006, while the tree islands in M2 and M3 were planted in March 2007.

Methods

Groundwater levels were monitored and recorded for 28 wells every 15 minutes from July 2007 through July 2009. At least three groundwater wells were monitored on each tree island, forming transects across the islands with one well located in the center of each island and two located on the islands' edges. Surface water stage was monitored at the eastern and western ends of each macrocosm, comprising a total of eight stations. Surface water stages were maintained according to an established hydrograph and surface water levels were recorded every 15 minutes. Daily average groundwater and surface water levels were computed from the 15 minute values. Surface water levels at the center of the two tree islands in each macrocosm were estimated from a linear interpolation between the surface water stages on the eastern and western ends of the macrocosms. Groundwater levels from three wells on each island were normalized to the surface water levels and graphed to characterize the shape of the water table. Daily groundwater evapotranspiration was estimated from the diurnal water table fluctuations using the White method (White, 1932). The White method uses the specific yield of sediments, the sum of the change in storage per day and the net recovery rate of the water table between midnight and 4:00 am to estimate the daily groundwater evapotranspiration rate. Daily groundwater evapotranspiration were averaged based on well location (edge versus center) and tree island type (limestone versus peat). Hydraulic gradients were determined between the surface water and groundwater levels for the center of each of the tree islands. Positive hydraulic gradients indicated groundwater discharging to the surface water while negative values indicated surface water recharging to the groundwater.

Results

The average monthly groundwater evapotranspiration rates on the LILA islands were similar from the first to the second year of the study, but a large increase in biomass, specifically in the center of the islands, may suggest that the ratio of transpiration to evaporation increased. The likely increase in transpiration rates was further supported by the drawdown detected in the water table in the center of the tree islands. The groundwater level in the center of the limestone islands were at times lower than that of the surface water, while peat tree islands always maintained elevated groundwater levels compared to the surface water (**Figure 6-17**).

Discussion

The data suggest overlying vegetation and underlying geologic conditions play a large role in the hydrologic conditions of tree islands. For 2007–2008, groundwater levels indicate that the dominant direction of groundwater flow was from the center of the islands to the edges (**Figure 6-18**). Between the first and second year of the study, the amount of aboveground biomass nearly doubled. With this doubling of biomass, a water table depression developed in the center of all of the islands and created a hydraulic divide along the edge of the island between the surface water and groundwater (**Figure 6-18**). The groundwater drawdown was larger in the limestone islands compared to the peat islands. The presences of the groundwater table depression indicated the tree islands acted as a hydrologic sink, capable of aiding in the concentration of nutrients deposited on the islands overtime. Furthermore, the hydrologic divide between the surface water and groundwater at the edges of the islands indicated a lack of surface water inputs, but does not speak to the interaction between the slough groundwater and tree island groundwater. The fast response in the water table with the growth of juvenile trees was unexpected, but provides insight into the formation and possible reconstruction of tree islands of the Everglades.

Relevance to Water Management

The response of tree island groundwater hydrology to tree growth within several years of planting demonstrates the critical nature of tree islands in maintaining oligotrophy in Everglades marshes. In less than five years, the evapotranspiration of trees on young tree islands is having a measurable impact on the island's ability to accumulate minerals due to the formation of the hydraulic divide. This demonstrates the importance of maintaining a healthy population of woody vegetation on tree islands, which is only possible with the appropriate water depths and durations.

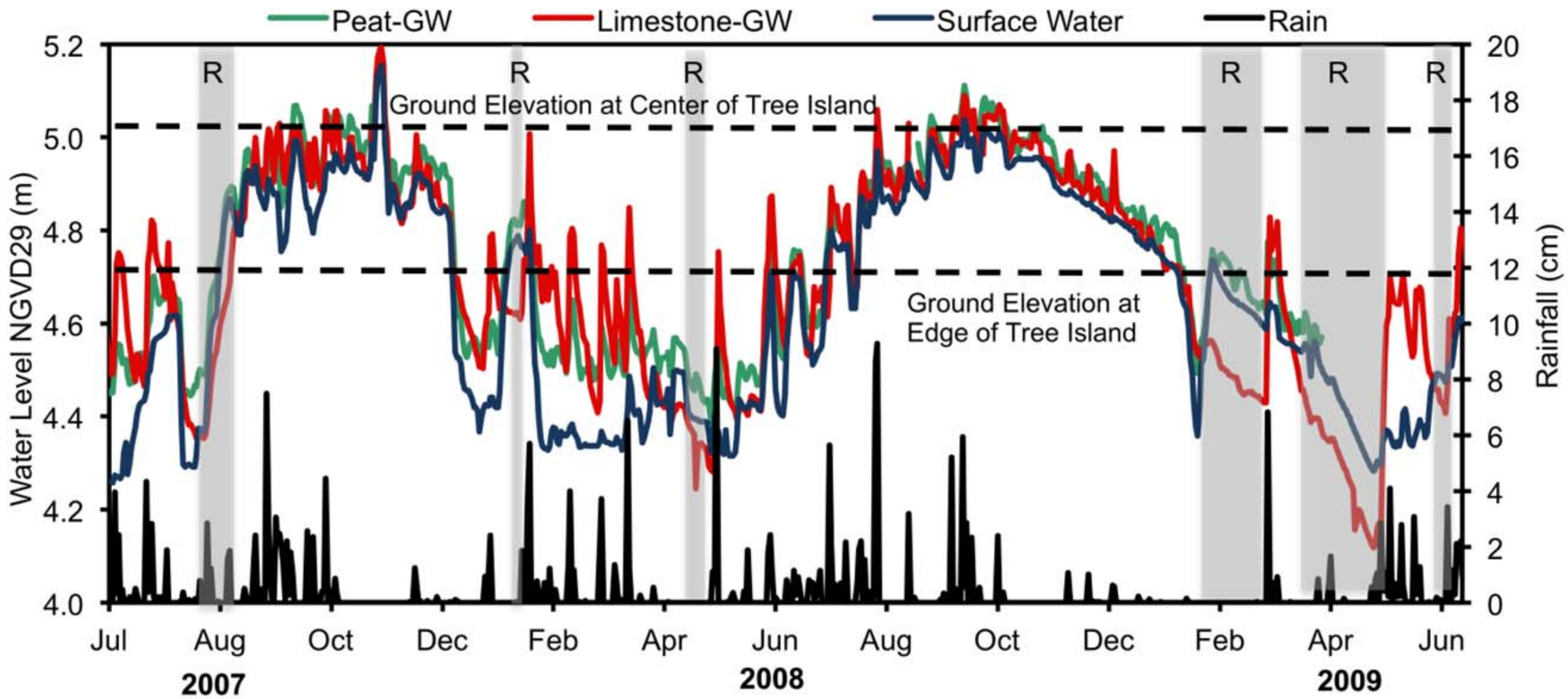


Figure 6-17. Surface water (blue) and groundwater (GW) levels [in meters (m) National Geodetic Vertical Datum of 1929 (NGVD29)] in the center of peat (green) and limestone (red) tree islands. Groundwater levels in the center of the limestone islands were typically lower than the surface water level during dry periods (gray boxes).

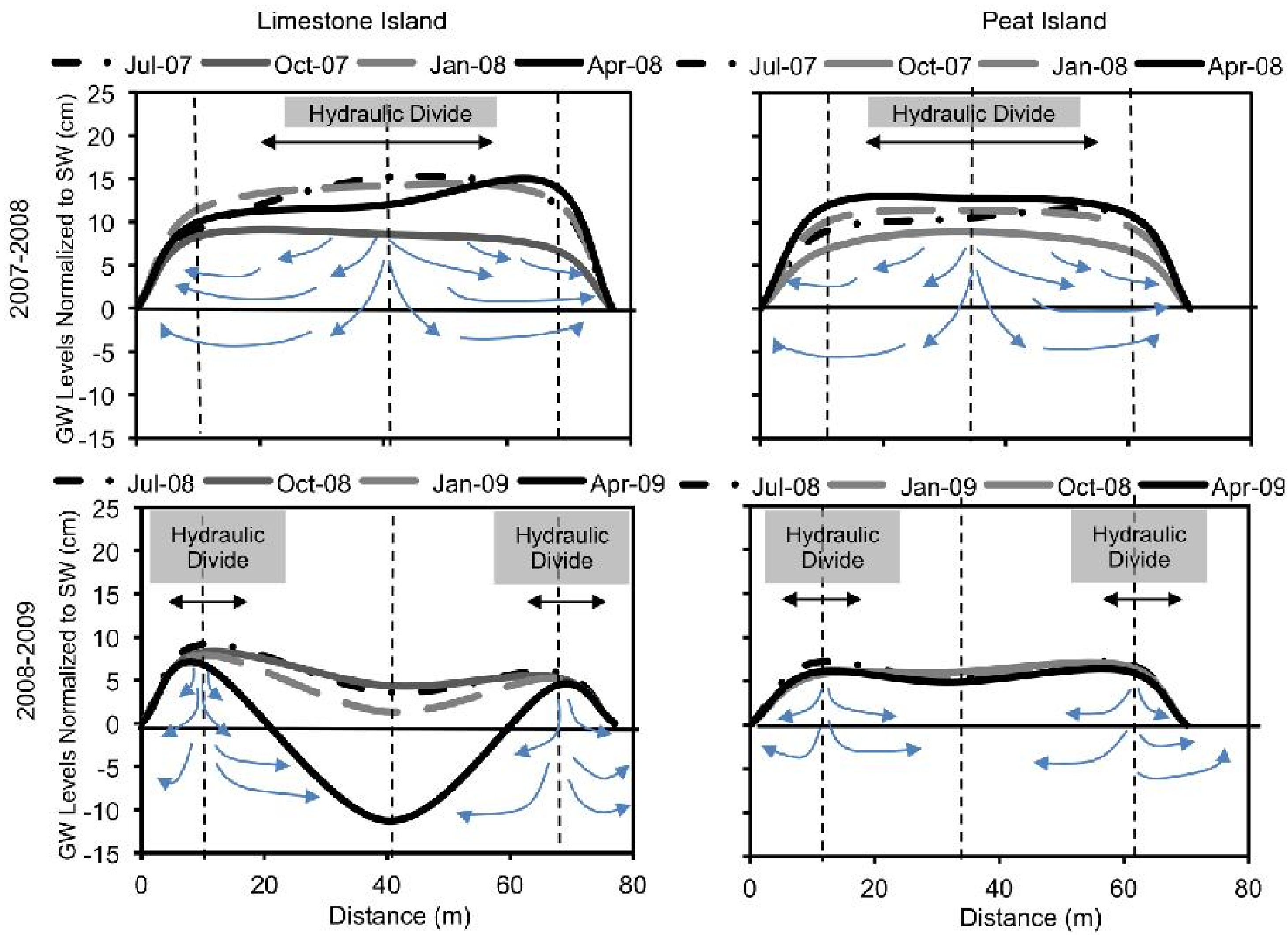


Figure 6-18. The groundwater (GW) table normalized to surface water (SW) levels from July 2007 through April 2009, in limestone (left) and peat (right) tree islands. Positive values indicate that groundwater levels were elevated compared to surface water levels, while negative values indicate groundwater levels were lower than surface water levels.

FLORIDA BAY BENTHIC VEGETATION COMMUNITY

Benthic vegetation, composed of seagrass and benthic macroalgae, provides habitat structure in Florida Bay and associated mangrove zone creeks and ponds. Monitoring and research of benthic vegetation is critical to our understanding of the effects of water management on wetland and estuarine ecosystems and results from these efforts are being applied to a potential update of the Florida Bay MFL rule, which is currently based on the salinity tolerance of widgeon grass (*Ruppia maritima*), and to assessing the effects of the CERP C-111 Spreader Western Project.

Methods

Benthic vegetation is monitored regionally in select locations using a randomized design where several 0.25 m^2 quadrats are assessed for benthic vegetation using indices of percent cover. Three separate monitoring programs cover different areas in Florida Bay. The South Florida Fish Habitat Assessment Program estimates benthic vegetation cover each May using a visual index of bottom occlusion within 17 basins throughout Florida Bay and along the southwest coast. The Miami-Dade Department of Environmental Resource Management estimates benthic vegetation cover quarterly in the nearshore embayments of northeastern Florida Bay using the same visual index as the South Florida Fish Habitat Assessment Program. The Audubon of Florida's Tavernier Science Center estimates benthic vegetation percent cover bimonthly within the mangrove lakes and creeks using a point intercept method. A more complete description of the monitoring programs and the methodologies are presented in the 2011 SFER – Volume I, Chapter 12.

Results

Notably, in WY2011, macroalgal cover increased throughout most of Florida Bay and Whitewater Bay. In general, the percentage of quadrats containing appreciable macroalgae cover (greater than five percent bottom occlusion) increased from about 20 percent in May 2009 to 42 percent in May 2010. The percentage of quadrats with 50 percent or greater bottom occlusion more than doubled in Whitewater Bay, increasing from 17 percent in May 2009 to 48 percent in May 2010. Conversely, the southwest coast (Lostman's River and Oyster Bay) showed a sudden decrease in macroalgal cover with Lostman's River having no quadrats with macroalgal presence in May 2010 (down from 26 percent in 2009) and Oyster Bay having no quadrats with greater than 50 percent macroalgal bottom occlusion (down from 37 percent in 2009).

In Madeira Bay and the Twin Key Basin of Florida Bay, an increase in shoal grass (*Halodule wrightii*) spatial extent and density, a CERP restoration target, had been reported in previous years. This trend was still apparent in these basins, but Whipray Basin also showed an increase in shoal grass density (**Figure 6-19**). Whereas the percentage of quadrats in Whipray Basin containing shoal grass in May 2010 did not vary significantly from May 2008. The percentage of quadrats in 2010 where shoal grass occurred with greater than 25 percent bottom occlusion was about eight times greater than the average from 2005 to 2009 (11.25 percent in 2010 versus the average from 2005 to 2009 of 1.38 percent). It should be noted that this is an area likely to be affected by altered water deliveries due to the CERP C-111 Spreader Canal Western Project. Increases in shoal grass do not seem to have negatively impacted turtle grass (*Thalassia testudinum*) and increased turtle grass frequency and/or density was observed in each of the basins. One of the goals of restoration is to enhance the species richness of seagrasses in the bay and the increase in *Halodule* is viewed favorably.

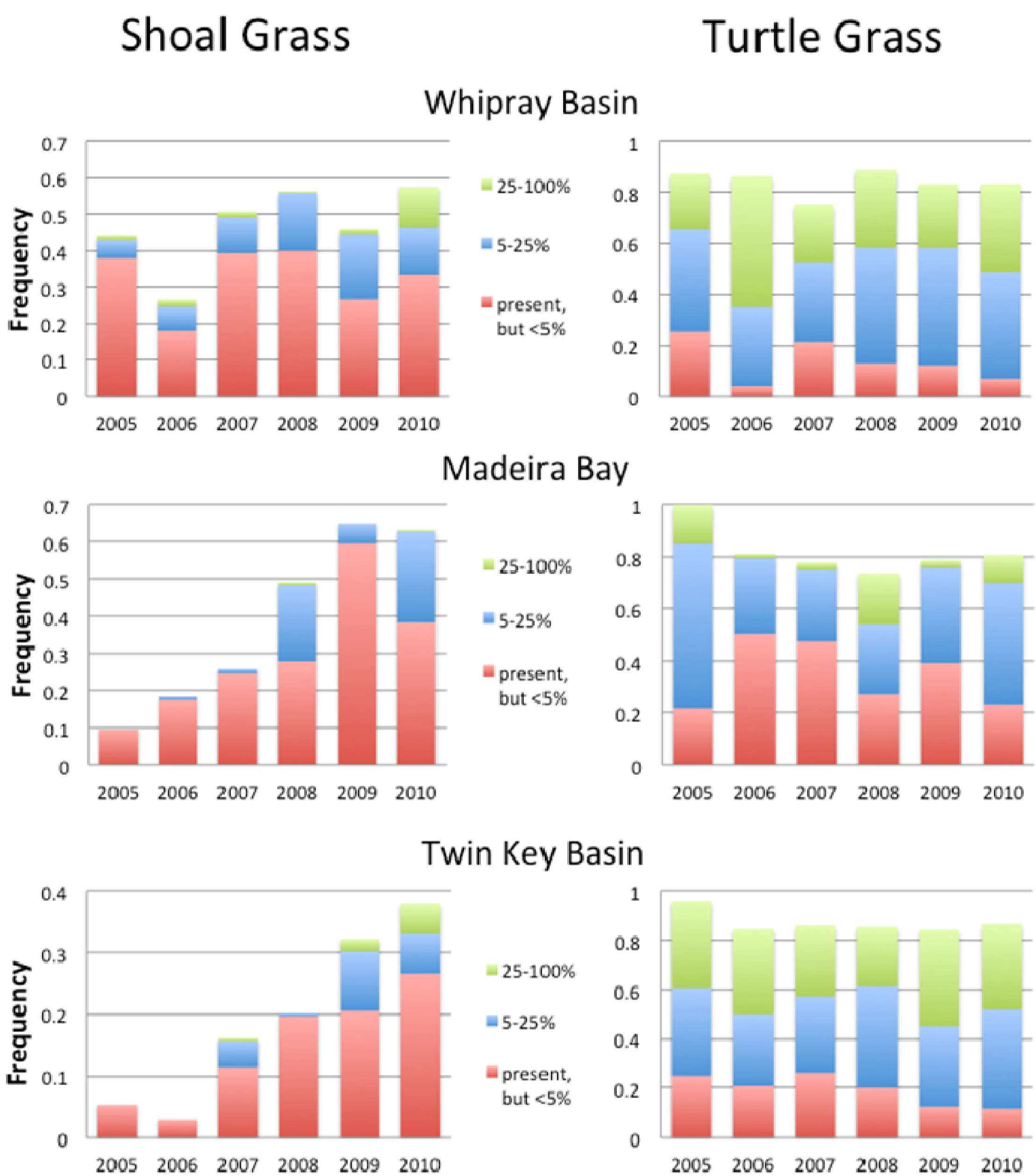


Figure 6-19. Seagrass cover in three central basins of Florida Bay. Percent cover was estimated from bottom occlusion using the Braun-Blanquet cover abundance index (BBCA) and then classes were grouped into three categories: present with less than five percent, 5–25 percent, and 25–100 percent cover. Data are from the Florida Fish and Wildlife Conservation Commission under a cooperative agreement with the SFWMD as part of the Restoration Coordination and Verification Team's (RECOVER's) Monitoring and Assessment Plan (MAP).

In the nearshore areas where data are collected quarterly, the most pronounced trend was a large increase in macroalgal cover in the first quarter of 2011. All the basins monitored by the Miami-Dade Department of Environmental Resource Management except for Little Madeira Bay and Highway Creek showed an increase in macroalgal extent and/or density between the last quarter of 2010 and the first quarter of 2011. The most pronounced increases were in Joe Bay, Alligator Bay, Trout Cove, Long Sound, and the area just outside of Little Madeira Bay. The average macroalgal presence of these basins (defined as the percentage of quadrats that contained macroalgae) increased from 53 percent in October–December 2010 to 88 percent in January–March 2011. Widgeon grass also appeared in 21 percent of quadrats in Joe Bay in the 2011 first quarter sampling.

Mangrove creeks along the northeastern Florida Bay shoreline also saw an increase in benthic vegetation during the first quarter of 2011 (Lorenz, 2011). In particular, widgeon grass cover was the highest ever recorded in an upstream Taylor River site with 52.3 percent cover. The middle part of the Joe Bay transect also saw a high percent cover of widgeon grass (77.7 percent in March 2011), which coincides with the findings from the Miami-Dade Department of Environmental Resource Management for the basinwide sampling during that time. The Barnes Sound transect had an estimated 52.7 percent widgeon grass coverage during the March sampling, which was the highest coverage reported at that transect since 1998.

Relevance to Water Management

The bay-wide increase in macroalgae may be related to the higher than normal total nitrogen (TN) reported in the *Florida Bay Water Quality Conditions* section of this chapter or to the wetter than normal dry season of WY2010. This trend should be watched carefully since macroalgae can be an indicator of elevated nutrient concentrations in the water column. Model simulations of Florida Bay showed that increased nutrients could trigger a water column response in the form of phytoplankton blooms, which was validated by an actual bloom in Florida Bay in 2005–2008, as presented in the 2010 SFER – Volume I, Chapter 12. The recent macroalgal response observed in WY2010 may be a secondary response by slower growing macrophytes to elevated TN recycled from the earlier phytoplankton bloom. The shoal grass increases, a desired restoration target within Florida Bay, may be associated with lower salinities as a result of the wetter than normal dry season of WY2010 and could persist if the operation of the C-111 Spreader Canal Western Project redistributes more water to the central bay region. In the mangrove creeks, the increase in widgeon grass coverage could be due to enhanced propagation due to the cold temperatures experienced during the December–January period (Koch and Dawes, 1991) and to the low salinities that persisted into March.

SUBMERGED PROCESSES AFFECTING AQUATIC VEGETATION IN THE TRANSITION ZONE

Restoration goals for the CERP include the expansion of widgeon grass beds in the northern Florida Bay ecotone boundary and the northern bay proper. These beds provide habitat and a high quality food source for fish and other fauna. Historically, this area had relatively high widgeon grass abundance, but with a dynamic pattern along the Joe Bay (northeastern Florida Bay) transect with episodes of denuded or low density bottom that shifted between macroalgae and rooted macrophytes.

Methods

A monitoring effort was initiated to establish baseline conditions in the transition zone. Seagrass and macroalgae cover are measured in transects in the wetland ecotone along with environmental conditions including salinity, temperature, and meristem oxygen. In an effort to gain a better understanding of the mechanisms that control the abundance of widgeon grass, regrowth experiments were conducted in experimental plots at sites established in Joe Bay directly south of the C-111 canal. SAV density was manipulated by clearing selected plots and measuring the rate of re-recruitment into the plots (Koch, 2011). Although total abundance of widgeon grass was low during the sampling, important processes were identified that appear to control its development and growth.

Results

Three dominant species of SAV in Joe Bay, widgeon grass, another submerged vascular plant, *Najas marina*, and the macroalgal species *Chara hornemannii* can maintain percent cover when salinities increase to mesohaline conditions. However, when salinities drop precipitously (to approximately 6 psu per hour), *N. Marina* and *C. hornemannii* decline. Widgeon grass appears to be more resilient and can withstand abrupt freshening better than the macroalgae species. However, *Chara* and *Najas* can rapidly recover after their biomass loss, allowing them to be competitively dominant over widgeon grass when lower salinity conditions return. Removal of all biomass from plots allowed widgeon grass to be more competitive but with a lag. The mechanism for this needs further examination. Thus, with extremely variable salinities that change abruptly, the species with rapid recruitment rates are favored, which in this case are *Chara* and *Najas*.

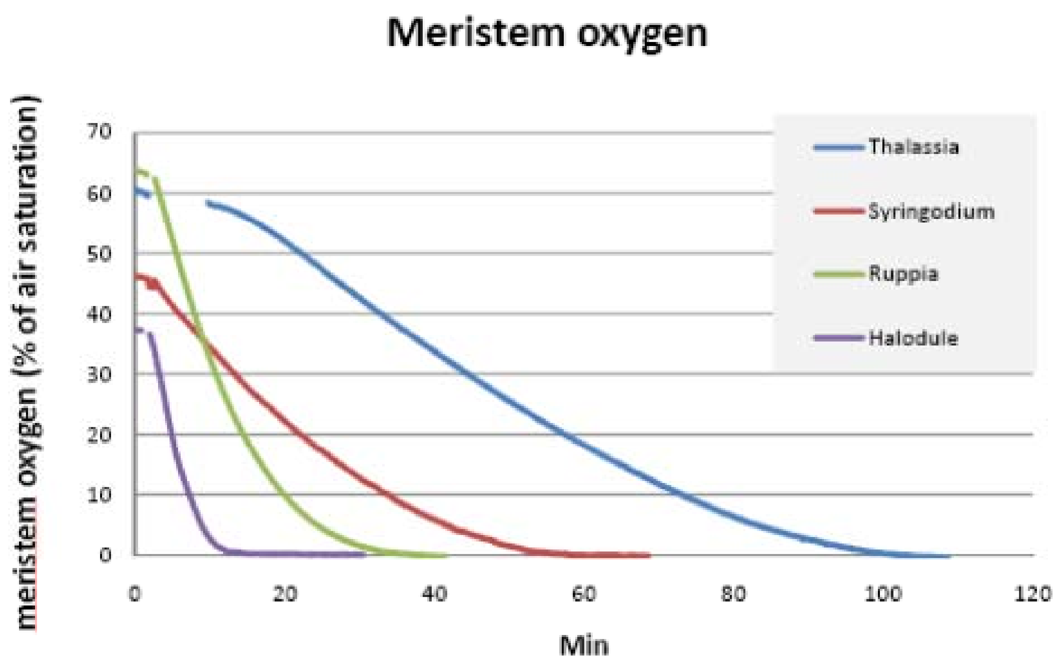
Based on preliminary work, turtle grass has shown a much higher capacity to store oxygen and a slower loss rate of oxygen (**Figure 6-20**), which would enable it to survive better during periods of hypoxia relative to other species.

Submerged Aquatic Vegetation Physiology

Major seagrass mortality events have been potentially linked to hypoxia and an interaction with sulfide exposure. However, seagrasses in Florida Bay have varying capacities to store oxygen in their meristems overnight during the period of maximum hypoxia (Koch, 2011). This adaptation would certainly affect a species' ability to sustain aerobic respiration and keep sulfides in the porewater away from meristematic tissues.

Relevance to Water Management

Knowledge of seagrass characteristics and environmental requirements is essential to understanding the restoration actions needed to maintain and expand healthy, high quality habitat. Restoration targets for seagrass include both expanded coverage of seagrass plants and a diverse mixture of species more resilient in withstanding environmental change. Data from research on salinity level and variability effects on seagrass have shown that low salinities in the wet season with slow seasonal salinity variability in the dry season is optimal for CERP-targeted expansion of desired widgeon grass habitat. Widgeon and shoal grasses are identified as most sensitive to low oxygen conditions, giving a lower boundary to the degree to which oxygen may be allowed to fall in periods of blooms or high organic production.



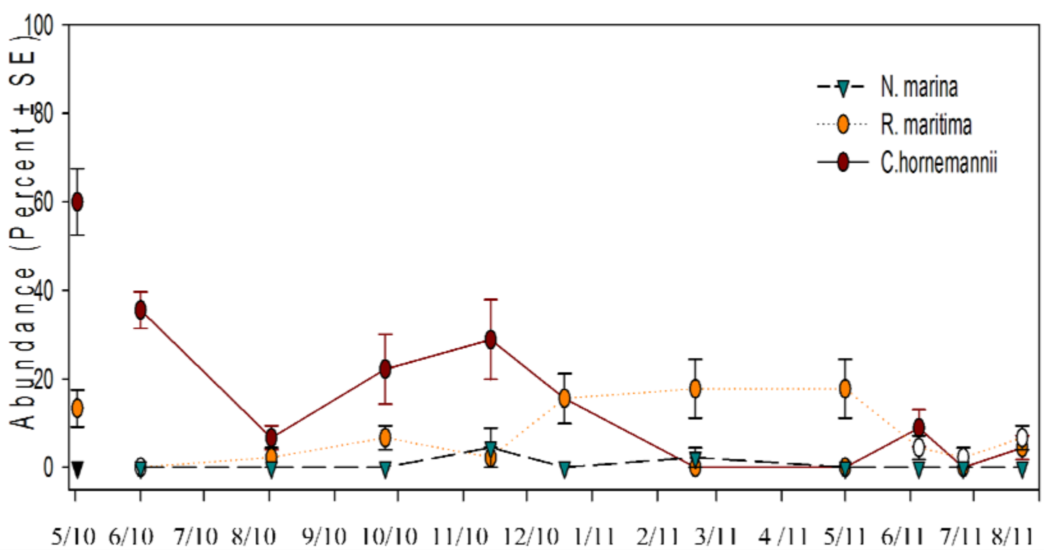


Figure 6-20. Meristem oxygen levels in four rooted aquatic plants [turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), widgeon grass (*Ruppia maritima*), and shoal grass (*Halodule wrightii*)] as a function of time (minutes) in anoxic water (top). Average species abundances after regrowth in cleared plots (percent cover ± SE, n=5) of *Najas marina*, widgeon grass, and the macroalgae *Chara homemanni* from May 2, 2010 to July 24, 2011. Single points on May 2, 2010 were initial abundances prior to vegetation removal.

ECOSYSTEM ECOLOGY

Christopher Madden, Susan Newman, Scot Hagerthey[3], Mark Cook, David Rudnick, Stephen Kelly, Robert M. Kobza[3], Michael Manna, Kristin Seitz and Robert Shuford

This section focuses on two regions: (1) areas of WCA-2A severely impacted by nutrient enrichment and (2) coastal communities of Florida Bay expected to be affected by the C-111 Spreader Western Project. For WCA-2A, results of maintaining openings in dense cattail stands for ecosystem restoration are presented. The C-111 Spreader Western Project is s major CERP restoration project that was under construction during WY2011 and is expected to be operational in WY2012. Reporting on areas of the Florida Bay coast expected to be affected by this project includes (1) Florida Bay water quality, (2) sediment-water nutrient fluxes in the Central Lakes region, (3) phytoplankton surveys, and (4) "white-zone" soil salinity.

CATTAIL HABITAT IMPROVEMENT PROJECT

Restoration of phosphorus (P) impacted regions of the Everglades requires not only a reduction in P loads and concentrations, but also active management efforts to reduce the ability of the cattail regime to absorb disturbances and maintain itself, that is, the inherent resilience of the cattail regime (Hagerthey et al., 2008). The Cattail Habitat Improvement Project (CHIP) was a large-scale in situ study comprised of fifteen 6.25 ha plots to test the ability to rehabilitate cattail areas by creating an alternative SAV regime. The two primary objectives were to (1) assess whether creating openings in dense cattail areas would sufficiently alter trophic dynamics such that wildlife diversity and abundance is increased and (2) determine to what extent these created open areas' functions compared to the natural Everglades. Using a combination of herbicides and fire, open areas were created in enriched and moderately-enriched areas of WCA-2A in July 2006. The numerous hypotheses, experimental design, and rationale behind this research project were described in the *Ecosystem Ecology* sections of the 2007–2011 SFERs – Volume I, Chapter 6, and detailed project description and methodologies can be found at www.sfwmd.gov/evergladeswatershed. This year's report evaluated aboveground carbon (C) and nutrient [nitrogen (N) and phosphorus (P)] storage and cycling during the final sampling events in WY2010 to assess whether we altered trophic dynamics within the created regime. Wading bird surveys associated with this water year occurred from November 2009–April 2010. For all results, sites are delineated based on their location; enriched, transitional, reference, and whether or not they were burned, open, or controls.

Methods

The open plots were created using a combination of herbicides and burning, and subsequently maintained with herbicide application as necessary. Specifically, glyphosate was applied in May 2006 and the plots were burned on July 20 and 21, 2006. It was determined that glyphosate alone was not sufficient to control cattail regrowth; thus, an additional spray was conducted in August 2006, using a combination of glyphosate and imazapyr. Some vegetative strips remained and a partial re-spray occurred in September 2006. Two additional sprays to maintain the openings occurred in March and November 2007.

Soil and vegetation were sampled from three locations within each plot and composited. Soil and floc samples were collected using 10 cm, thin walled, stainless steel, coring tubes. Vegetative biomass was obtained from 0.25 m^2 quadrats. Aquatic fauna (fish and invertebrates) were sampled from a single location within each plot using a 1 m^2 bottomless pull trap (for details see the 2009 SFER – Volume I, Chapter 6, *Ecosystem Ecology* section). Macroinvertebrates were sampled within a portable 0.25 m^2 enclosure using 0.035 micrometer mesh dip nets. Soil, vegetation, and aquatic fauna were sampled in August and October 2009. Nutrient contents were analyzed using standard methods by DB Environmental, Inc. (soil, vegetation, fish, and macroinvertebrates) and microanalytical methods by the Analytical Chemistry Laboratory at the University of Georgia (small invertebrates). Aerial wading bird surveys were conducted approximately weekly over a 26 week period (n = 19 survey weeks) from early November 2009 to late April 2010.

Results and Discussion

The ultimate objective of the CHIP was to accelerate the ecological rehabilitation of the P-enriched, Everglades cattail marsh. Our approach was to initiate a regime shift using herbicides and fire to create and sustain a SAV and/or periphyton dominated habitat. We hypothesized that by switching primary production to the water column, compared to emergent macrophyte biomass, we would (1) alter biogeochemical cycling among the C dominant pools changing food quality by reducing the amount of structural C (i.e., cellulose and lignin) and (2) increase the biomass of herbivorous species (fish) in open plots compared to controls and decrease detritivorous species (crayfish) in open plots.

Standing stocks and nutrient storage of aboveground vegetation, floc, soil, and periphyton clearly reflect that the C dominant pools changed significantly (**Table 6-3**). As expected in SAV versus emergent macrophyte dominated systems, aboveground biomass and nutrient storage were significantly decreased in the open plots. On average, open water plots contained only 14–55 percent and 12–27 percent of the live and dead, respectively, aboveground mass and nutrients observed in the control plots. Storage of C, N, and P in the surficial soils was elevated in open plots compared to control plots, however as noted previously, these were similar to those observed in January 2007 (2011 SFER – Volume I, Chapter 6, *Ecosystem Ecology* section). Because contents remained similar over time, this suggests, following treatment implementation, the majority of C and nutrients transferred from aboveground macrophytes to surficial soils were generally stable. In contrast, floc had significantly lower C storage in open relative to control plots. This reflected the switch to less structural and more labile floc sources (SAV and periphyton), which are also more spatially patchy. Periphyton mass was three to five times greater in open compared to control plots, resulting in concomitantly three to five times increased storage of C, N, and P. These changes in C and nutrient storage, and changes in habitat (e.g., switch to periphyton with increased oxygen levels) in open plots were reflected in the aquatic faunal community. Faunal biomass and nutrient storage were significantly greater in open compared to control plots. This was primarily due to increased fish biomass, thus supporting our initial hypothesis. Though no concomitant decrease in crayfish biomass was observed [open plots had 1.40 ± 0.85 grams dry weight per square meter (g dw/m^2); control plots had 1.38 ± 0.59 g dw/m^2], densities were significantly different. Average crayfish densities in control plots were 10 ± 4 compared to open plot densities, which were 2 ± 3.

As expected, the openings in the cattails resulted in a dramatic increase in wading bird (Ciconiiformes) usage. Relative to control plots, the open plots attracted approximately two orders of magnitude more foraging individuals and over double the number of species (**Table 6-4**). The openings in the enriched region supported the greatest number of birds, almost double that of the openings in the transitional region, but species diversity and the number of weeks birds were present in the openings differed little between regions.

Table 6-3. Mass and nutrient storage [mean ± standard deviation (SD)] as grams per square meter (g/m^2) determined in enriched and transitional control plots and enriched and transitional created openings in WY2010, four years following treatment implementation. Arrows indicate direction of change of open plots relative to control plots when analysis of variance (ANOVA) indicated active management caused a significant change. Tests for significance were evaluated at α = 0.05.

	Enriched Control (g/m^2)			Enriched Open (g/m^2)			Transitional Control (g/m^2)			Transitional Open (g/m^2)				ANOVA Results for Treatment	
Parameter	**Mean**		**SD**	**Mean**		**SD**	**Mean**		**SD**	**Mean**		**SD**		**F**	**p**
Aboveground – Live															
Mass	715	±	144	298	±	225	687	±	208	159	±	94	↓	54.71	<0.001
Phosphorus	0.572	±	0.104	0.260	±	0.131	0.278	±	0.085	0.091	±	0.018	↓	74.05	<0.001
Nitrogen	5.43	±	0.94	3.00	±	1.75	4.53	±	1.34	1.62	±	0.70	↓	37.p6	<0.001
Carbon	317	±	72	78	±	52	328	±	101	46	±	18	↓	77.2	<0.001
Aboveground – Dead															
Mass	1,702	±	584	202	±	130	1,692	±	675	227	±	123	↓	48.74	<0.001
Phosphorus	0.422	±	0.152	0.079	±	0.053	0.295	±	0.137	0.079	±	0.048	↓	22.31	0.003
Nitrogen	9.11	±	2.99	1.41	±	0.86	9.78	±	5.10	2.02	±	1.25	↓	28.68	0.002
Carbon	797	±	292	92	±	61	823	±	334	106	±	58	↓	47.32	<0.001
Floc															
Mass	412	±	171	168	±	193	464	±	119	276	±	198		4.66	0.074
Phosphorus	0.50	±	0.17	0.29	±	0.36	0.46	±	0.13	0.42	±	0.44		0.51	0.501
Nitrogen	10.96	±	4.43	5.58	±	6.51	12.59	±	2.54	8.75	±	7.43		2.07	0.201
Carbon	188	±	76	73	±	83	206	±	53	112	±	78	↓	6.04	0.049
Soil (0–5 centimeters)															
Mass	3470	±	309	4342	±	190	3562	±	267	3789	±	449	↑	16.75	0.006
Phosphorus	4.03	±	0.12	6.02	±	0.69	3.24	±	0.37	3.69	±	0.40	↑	65.2	<0.001
Nitrogen	94.74	±	8.90	130.73	±	6.87	96.73	±	1.92	102.61	±	9.10	↑	28.41	0.002
Carbon	1594	±	147	1934	±	101	1591	±	101	1647	±	164	↑	16.92	0.006
Periphyton															
Mass	7.82	±	11.90	37.20	±	31.35	18.75	±	17.37	60.25	±	29.03	↑	10.05	0.019
Phosphorus	0.013	±	0.021	0.067	±	0.056	0.016	±	0.014	0.053	±	0.038	↑	7.17	0.037
Nitrogen	0.27	±	0.41	1.14	±	0.95	0.52	±	0.47	1.52	±	1.00	↑	6.82	0.04
Carbon	3.31	±	5.02	14.61	±	12.27	7.81	±	7.13	22.02	±	13.08	↑	7.09	0.037
Aquatic Fauna															
Mass	2.50	±	0.42	3.67	±	0.67	1.87	±	0.87	3.80	±	0.23	↑	19.99	0.004
Phosphorus	0.035	±	0.011	0.083	±	0.016	0.031	±	0.018	0.066	±	0.021	↑	56.31	<0.001
Nitrogen	0.23	±	0.04	0.38	±	0.06	0.17	±	0.09	0.37	±	0.01	↑	25.53	0.002
Carbon	1.00	±	0.16	1.63	±	0.31	0.74	±	0.39	1.65	±	0.04	↑	26.47	0.002

Table 6-4. A comparison of wading bird usage of control and open plots at enriched and transitional regions during the WY2010 dry season.

	Enriched		Transitional	
	Control	Open	Control	Open
Mean birds per treatment per week ± standard deviation (all species pooled)	0.5 ± 1.4	35.5 ± 36.3	0.2 ± 0.6	21.4 ± 25.9
Total number of birds observed per treatment	26	2,024	9	1,218
Total number of species per treatment	4	9	4	10
Number of weeks (n = 19) that a minimum of one bird was observed	6	17	5	18

When all treatments were pooled, 3,277 birds were observed representing all ten species of the northern Everglades wading bird guild. White ibis was the dominant species (comprising 49 percent of the total), little blue heron (*Egretta caerulea*), glossy ibis (*Plegadis falcinellus*) and snowy egret (*E. thula*) were moderately abundant (together encompassing 40 percent of the total), while great egret (*Ardea alba*), tricolored heron (*E. tricolor*), great blue heron (*A. herodias*), roseate spoonbill, wood stork and black-crowned night-heron (*Nycticorax nycticorax*) each comprised a relatively small proportion of the total.

The next step in our analysis is to provide direct evidence of the connection between altered nutrient cycling and the food web using stoichiometric relationships (Sterner and Elser, 2002). The reduction of complex ecological dynamics into simple mass balance relationships (e.g., C:N:P ratios) has been used successfully to increase our understanding of a multitude of linkages, including trophic interactions (Elser et al., 1998) and the role of species in ecosystems (Vanni et al., 2002). The elemental composition of the consumers is a proxy representing the nutrient needs (i.e., demand) while the proportional elemental composition of the food represents the supply (Sterner and Elser, 2002).

Using enriched open and control plots as an example, this supply and demand relationship was examined and the food web structure was determined based on functional group stoichiometry and differed between open and control plots (**Figure 6-21a**). Within control plots, food webs were regulated by detrital (floc) dynamics, with resource supply to primary consumers controlled by emergent macrophyte production. The high cellulose and lignin content of emergent macrophytes (primarily cattail and sawgrass) resulted in a high standing stock of live and dead material with high C:N:P ratios.

Decomposition resulted in a high quantity but low quality and recalcitrant (C:N:P > 933:48:1) food resource (**Figure 6-21a**). In contrast, in enriched open plots, although resource supply to primary consumers was also through detrital pathways, the substantially greater contribution of more labile aquatic (SAV and periphyton) production improved resource quality (**Figure 6-21b**). Although the quantity of resources were less than enriched control plots, the quality for enriched open plots was greater as evident by the lower C:N:P ratios for floc (666:44:1), periphyton (693:50:1), and SAV (603:28:1).

In control plots, energy flux and nutrient transfer to consumers was primarily through omnivory by invertebrates (C:N:P = 70:17:1) with a standing stock of 0.61 ± 0.25 grams organic C per m^2 (**Figure 6-21a**). However, in open plots, energy flux and nutrient transfer were primarily through herbivorous and omnivorous fish (**Figure 6-21a**), with C:N:P ratios of 62:11:1 and 33:7:1, respectively. Therefore, relative to the control plots, the quality, quantity, and diversity of primary consumers were greater. In addition, because the active management strategy created a physical opening, these improved quality prey are also readily accessible for consumption by wading birds. The next step in this analysis is to assess the relative degree of stoichiometric balance and imbalance of food sources and consumers at transitional and unenriched sites.

Relevance to Water Management

In previous SFERs, individual parameter responses within each plot were documented (e.g., dissolved oxygen concentrations and percent vegetative cover, which showed extensive use by wading birds). This year, results show that these created openings not only support wading bird foraging because of the created open environment, but also confirm the hypothesis that the quality of food in the actively managed plots is greater than adjacent cattail dominated plots. Summarizing relationships through mass balances of C:N:P further showed how nutrient enrichment and C quality influences the food web, which will be an important attribute when assessing how well these created openings functions differ, or are comparable to, the naturally oligotrophic Everglades, where high C:N:P values (i.e., P-limited) result in low growth rates and affect reproductive success. As noted in previous reports, the creation of these open areas relatively near inflow structures provide increased opportunities for foraging wading birds, including earlier foraging (due to earlier hydration) and foraging during hydrologic reversals (due to higher elevation), as well as greater prey biomass with greater nutritional value. Therefore, while not recreating the natural oligotrophic condition, these created openings provide significant environmental benefits in regions of the Everglades that currently have limited ecological value.

FLORIDA BAY WATER QUALITY CONDITIONS

A primary objective of the C-111 Spreader Western Project is to minimize seepage from Taylor Slough toward the C-111 canal via installation of a seepage barrier on the eastern border of Taylor Slough. The intended result is that more natural patterns of flow distribution and timing will facilitate and sustain more natural ecosystem structure and function in the southeastern Everglades and Florida Bay. These hydrologic modifications are not expected to alter water quality characteristics, but any degradation of water quality could constrain restoration. This section presents water quality conditions and factors affecting these conditions, particularly with regard to the availability of nutrients and occurrence of algal blooms in Florida Bay.

Water quality in Florida Bay and the other southern coastal systems has been monitored since 1991 (WY1992) to ensure that District operations and projects protect and restore the ecosystem to the extent possible. CERP performance measures focus on chlorophyll *a* concentration, an indicator of algal blooms, as well as the nutrient inputs that initiate and sustain blooms. The performance measures call for no increase in the magnitude, duration, or spatial extent of blooms. As in last year's SFER, water quality in Florida Bay is reported as well as the water quality of the coastal systems to the west (Whitewater Bay) and east (Barnes Sound in Biscayne Bay). These estuarine regions have distinct water quality and ecological characteristics, in part a consequence of the differential inputs of water and associated materials from the Everglades from Shark Slough flowing into Whitewater Bay to the west and the C-111 canal flowing into Barnes Sound to the east. Changing operations associated with continuing implementation of the C-111 South Dade Project, Modified Water Deliveries to ENP (especially Tamiami Trail modifications), and the C-111 Spreader Canal Western Project will change freshwater flow patterns and may also change water quality patterns. In this section, these patterns are summarized by comparing WY2010 and WY2011 water quality to the temporal median of the monthly spatial means and the interquartile range for the entire period of record.

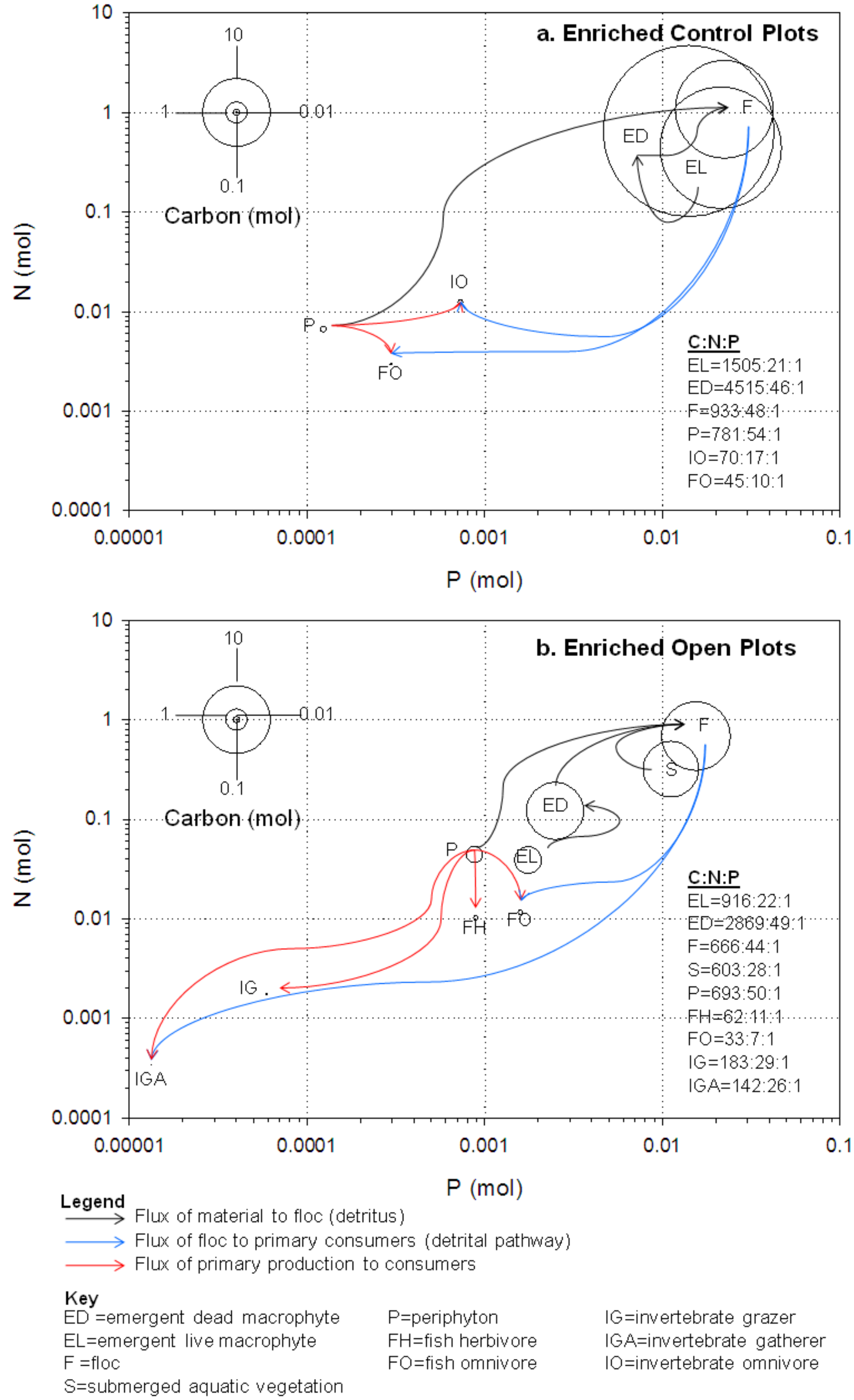


Figure 6-21. Carbon (C) pools, nitrogen (N) and phosphorus (P) ratios, and trophic linkages for (a) enriched control and (b) enriched open plots. The amount of C is expressed by the size of the bubble scaled according to the concentric circles in the upper left-hand corner of each panel. Values are the mass (mol) of N, P, and organic C per square meter. Secondary consumer lines were not connected to maintain clarity.

Algal blooms did not occur in estuarine waters from Whitewater Bay to Barnes Sound during WY2011. While chlorophyll *a* concentrations were greater than the long-term (WY1992–WY2009) median during the wet season in eastern Florida Bay (**Figure 6-22a**) and Barnes Sound (**Figure 6-23b**), absolute concentrations were very low (well below the RECOVER water quality performance measure upper range of 2 micrograms per liter). During the wet season, western Florida Bay had higher chlorophyll *a* concentrations that exceeded the 75^{th} interquartile range and were near the upper range for this region's performance measure (**Figure 6-22c**). In all regions, dry season chlorophyll *a* concentrations were generally below or near the long-term median (**Figures 6-22** and **6-23**). Notably, the central part of the bay, where Florida Bay's algal blooms have been most frequent and intense, lacked the typical October chlorophyll *a* peak and had extremely low concentrations throughout the year (**Figure 6-23b**). Operations have been conducted so as to increase flows through Taylor Slough and toward the central bay relative to flows through the C-111 canal, and with near average flows (see **Figures 6-7** and **6-8** in the *Hydrologic Patterns for Water Year 2011* section in this chapter), there does not appear to have been a water quality "tradeoff" during WY2011.

Total phosphorus (TP) concentrations in all regions, except the western region, continued the trend of being less than the long-term median and often less than or equal to the 25 percent of the interquartile range for the entire water year (**Figures 6-24** and **6-25**). Interestingly, in nearly all areas except Barnes Sound, TN had the opposite trend of TP, with TN being greater than the long-term median and often greater than the 75 percent interquartile range for the entire water year (data not shown). The reasons for these trends are unclear but not unprecedented in the region. Abbott et al. (2005) evaluated long-term (1991–2003) water quality data at 13 sites throughout the Biscayne Bay watershed and found that TN concentrations were generally increasing and TP concentrations were declining over this period. Annual average TP and chlorophyll *a* concentrations in all regions, except the western region, which showed a slight increase in the last two water years, have been declining since about WY2006–WY2007 (**Figure 6-26**), which included the disturbance of three hurricanes in 2005. Long-term patterns of TP and chlorophyll *a* point toward the importance of such disturbances, with peaks following Hurricane Irene in 1999 and the 2005 storms.

CENTRAL LAKES REGION SEDIMENT-WATER NUTRIENT FLUXES

As reported in the 2011 SFER – Volume I, Chapter 12, the District undertook studies on the dynamics of the western boundary of Taylor Slough (the lakes region between Seven Palm Lake and West Lake), which is a little studied area that will be critical to evaluation of CERP restoration projects, especially the C-111 Spreader Canal Western Project. An objective of the lakes study program is to quantify current (pre-restoration) rates of benthic nutrient and metabolic gas fluxes from sediment cores taken from Seven Palm Lake, Middle Lake or Munroe Lake, Terrapin Bay, West Lake, Long Lake, and Garfield Bight. It is important to understand the water column and sediment processes that will be affected by increased freshwater input to the transition zone because these processes have the potential to release nutrients to the overlying waters where downstream transport to the bay may cause algal blooms. Briefly, intact sediment cores and bottom water were collected and incubated in a temperature controlled incubator in the lab under dark and light conditions for about 4 hours each. At seven time points, three in the dark, one at the dark/light transition, and three in the light, samples were collected for various nutrient and gas samples. Sediment-water exchange rates were calculated from the slope of the change of the chemical constituent concentrations in the overlying water. Results from the first of two studies (Owens and Cornwell, 2011) indicate sediment recycling of inorganic N results in relatively large effluxes of ammonium (**Figure 6-27**). In most cases denitrification was relatively small.

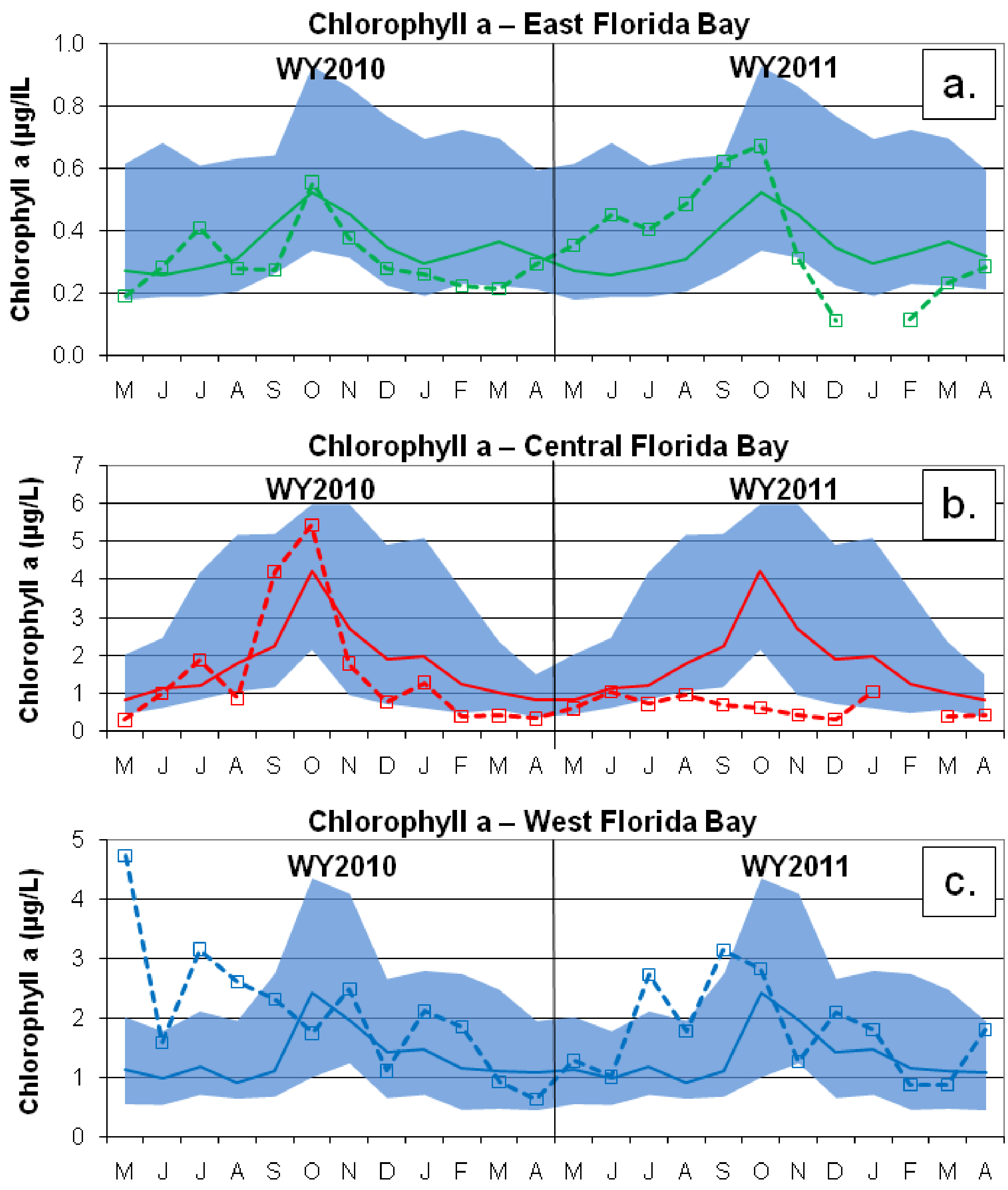


Figure 6-22. Monthly chlorophyll *a* concentrations [in micrograms per liter (µg/L)] in the three regions of Florida Bay during WY2010 and WY2011 (dashed line with open symbols) compared to median and interquartile range of monthly means from WY1992–WY2009 (solid line and blue shading).

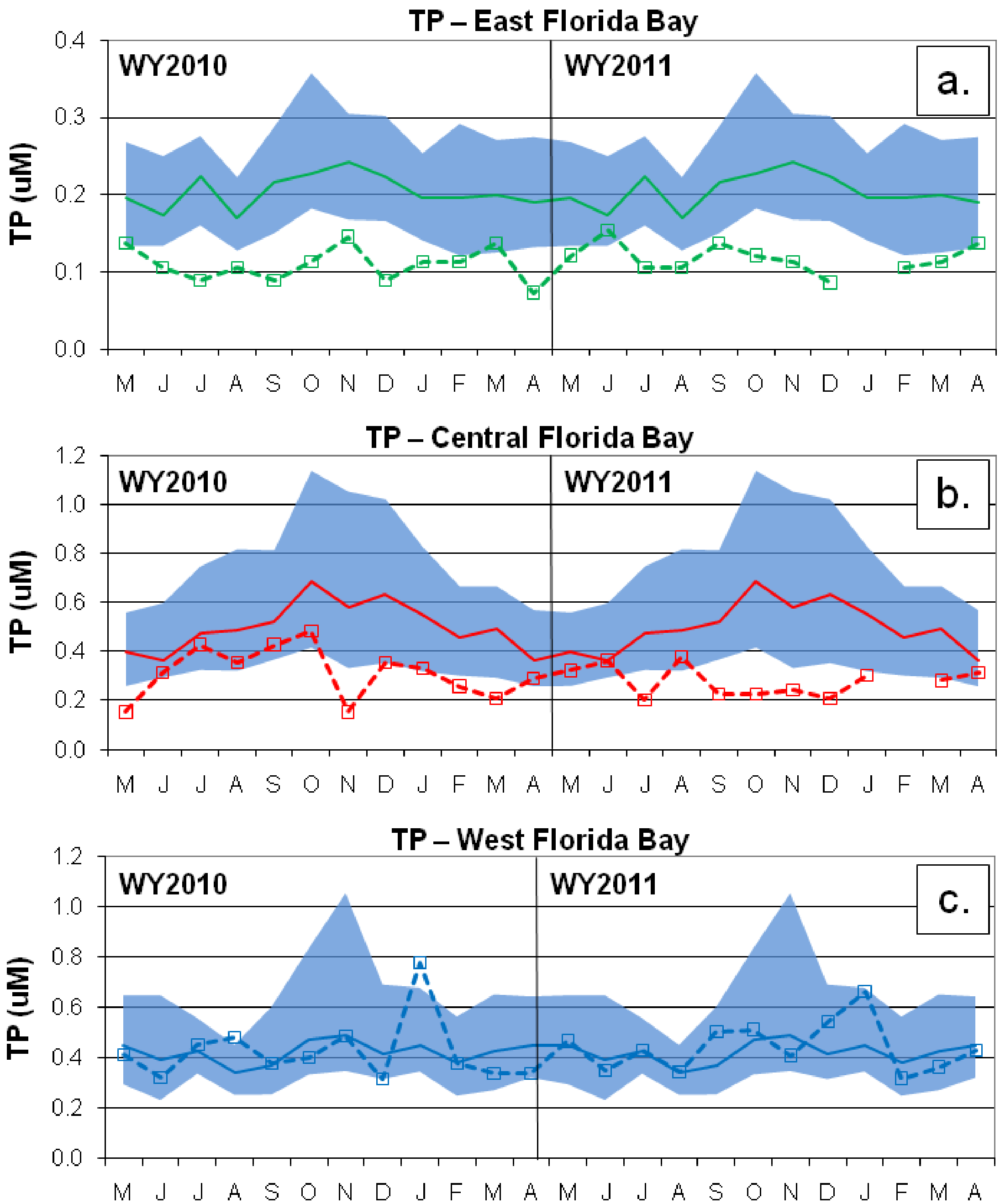


Figure 6-23. Monthly total phosphorus (TP) concentrations (1 µM = 31 µg/L) in the three regions of Florida Bay during WY2010 and WY2011 (dashed line with open symbols) compared to median and interquartile range of monthly means from WY1992–WY2009 (solid line and blue shading).

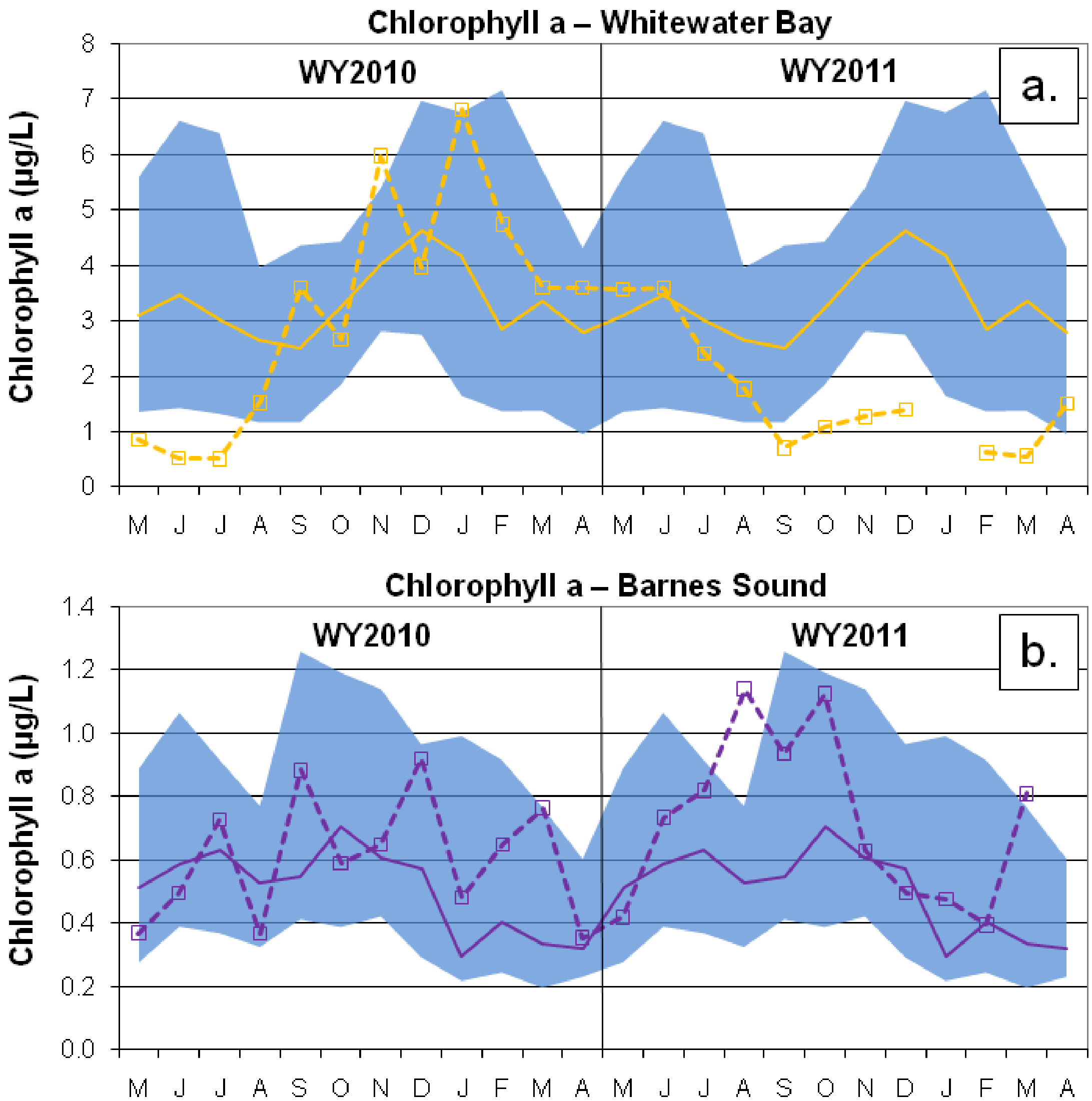


Figure 6-24. Monthly chlorophyll *a* concentrations in Barnes Sound and Whitewater Bay during WY2010 and WY2011 (dashed line with open symbols) compared to median (solid line) and interquartile range (blue shading) of monthly means from WY1992–WY2009.

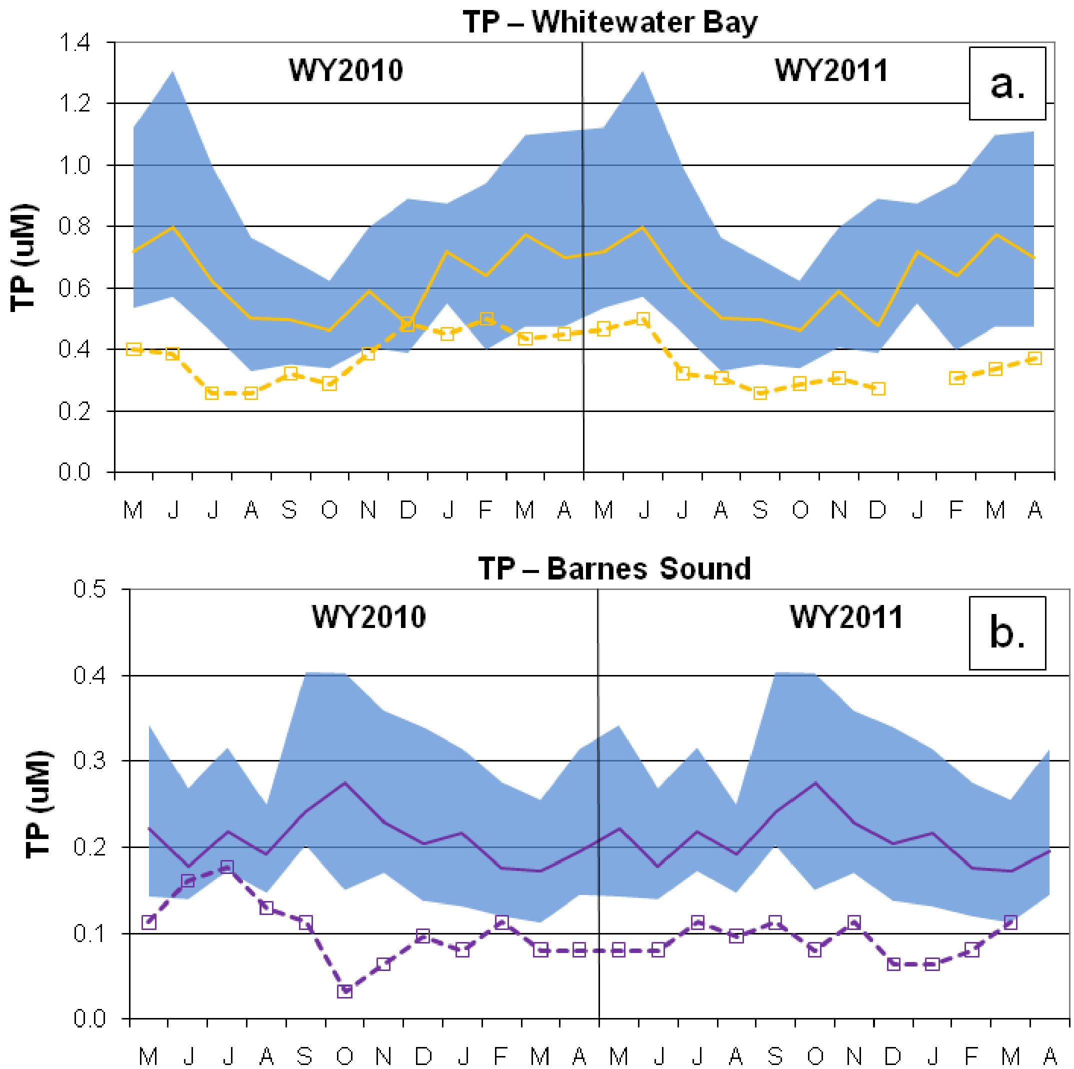


Figure 6-25. Monthly TP concentrations (1 µM = 31 µg/L) in Barnes Sound and Whitewater Bay during WY2010 and WY2011 (dashed line with open symbols) compared to median (solid line) and interquartile (blue shading) of monthly means from WY1992–WY2009.

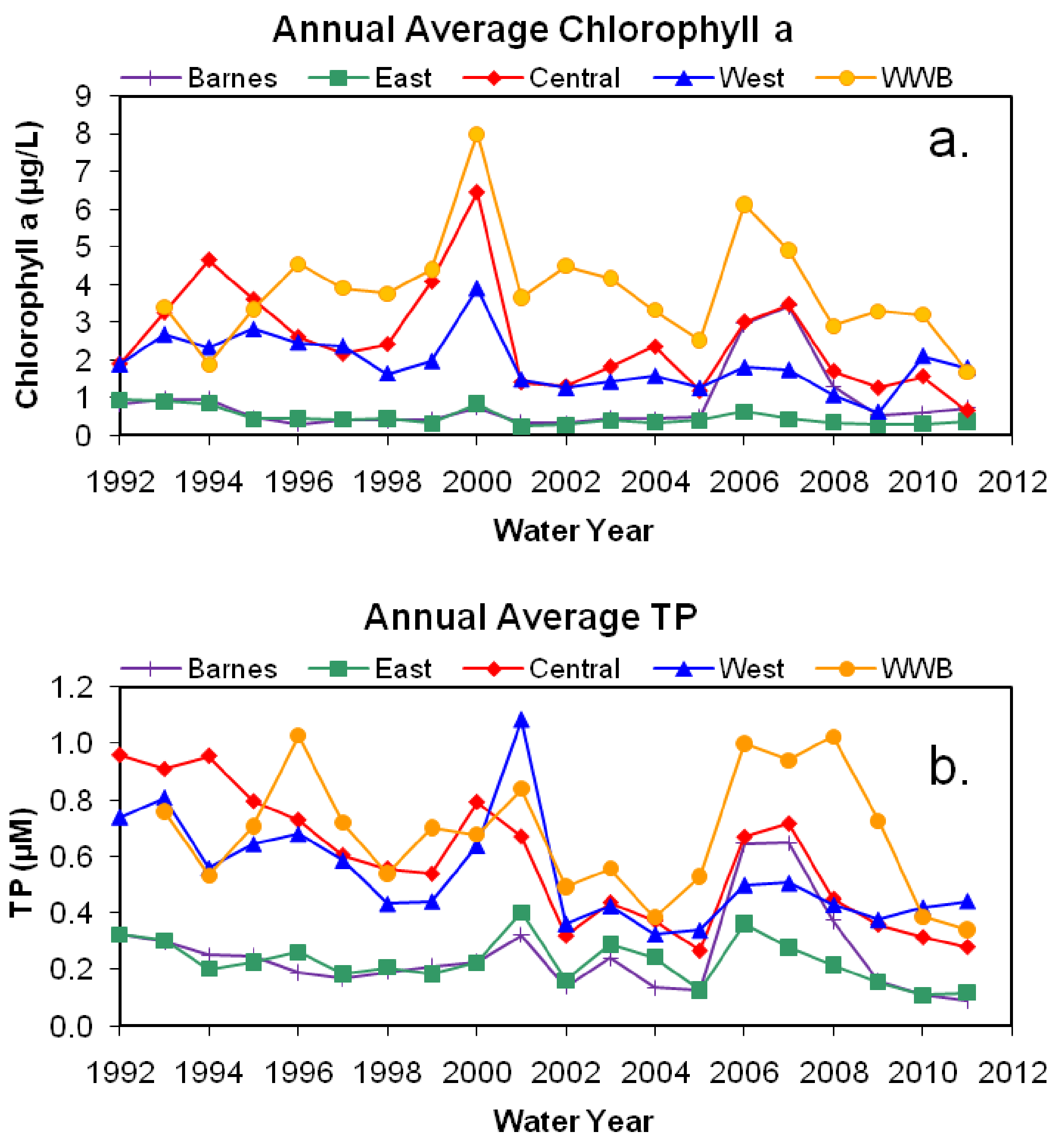


Figure 6-26. Annual average chlorophyll *a* and TP (1 µM = 31 µg/L) concentrations in five Florida Bay regions for the entire period of record. Linear regression analysis of TP versus water year results (slope, r^2): Barnes Sound (Barnes: 0.001, 0.00), Eastern Florida Bay (East: -0.005, 0.13), Central Florida Bay (Central: -0.030, 0.61*), Western Florida Bay (West: -0.017, 0.30*), and Whitewater Bay (WWB: -0.006, 0.02). * Indicates slope is statistically different from zero at $p<0.05$.

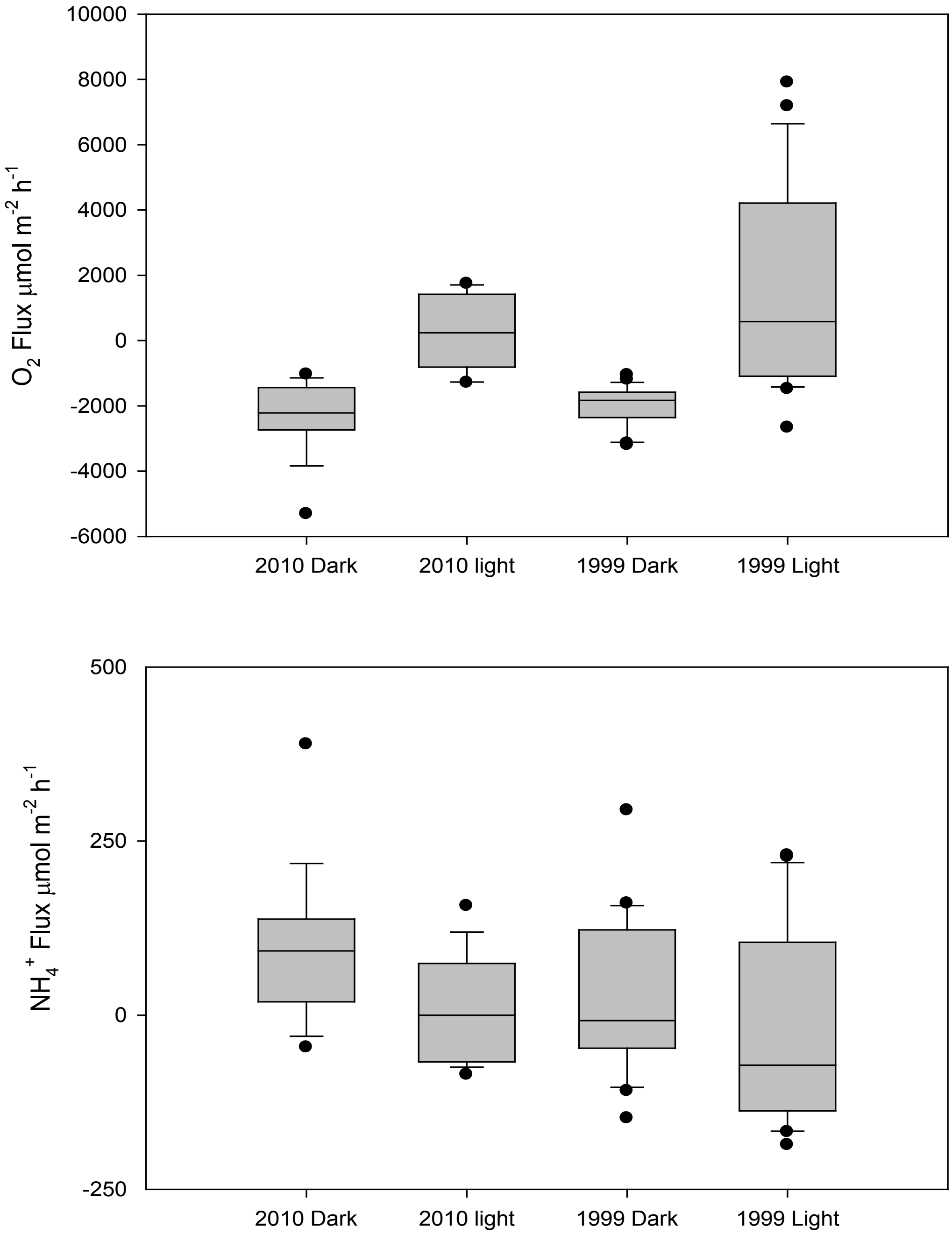


Figure 6–27. Oxygen (O_2) and ammonium (NH_4) fluxes [in micro moles per square meter per hour (µmol m^{-2} h^{-1})] in July 2010 at lake and estuary sites, as well as nine sites in Florida Bay (Kemp and Cornwell, 2001).

The effects of benthic microalgae were evident throughout these northern Florida Bay and southern Everglades ecosystems, with high rates of photosynthesis and strong retention of N under illuminated conditions. Losses of fixed N as N_2-N were relatively low, likely reflecting (1) the dominance of autotrophic N uptake and (2) the relative inefficiency of coupled nitrification-denitrification in sulfidic sediments with high rates of oxygen uptake. Net fluxes of both TP and TN were highly variable, with positive and negative fluxes. Results from this study of ENP saline lakes are similar to benthic flux results from nine sites in Florida Bay in 1999 (**Figure 6-28**). This implies that the finding of high nutrient concentrations and chlorophyll *a* in several ENP lakes may be more related to the lakes' long water residence time and possibly to other nutrient sources (such as groundwater).

LAKES PHYTOPLANKTON STUDY

In the near term, the C-111 Spreader Canal Western Project will likely reduce water flow into the lower C-111 Basin and Model Lands, and increase flows and levels toward the west in central Taylor Slough. The effect these hydrologic changes will have on the biology downstream in Florida Bay is the subject of a series of biogeochemical measurements during wet and dry seasons in WY2010–2011. In addition to changing hydrology, nutrient forms and transport, light availability, salinity, organic matter, and trophic relationships will be potentially altered, particularly in the mangrove ecotone and areas of northern Florida Bay that are susceptible to algal blooms. The status and composition of the phytoplankton is important to habitat and food quality. Blooms of cyanobacteria have been shown to be associated with high organic:inorganic N composition. Outbreaks of diatoms have been associated with high inorganic-to-organic N ratios. Heterotrophic bacteria abundance is associated with high organic:inorganic P (Glibert et al., 2004). Differing nutrient composition ratios are a consequence of varying source inputs in different regions of Florida Bay. In developing baseline data in the C-111 Spreader Canal Western Project footprint, several ecotone lakes are being studied to determine phytoplankton assemblages currently present and the nutrient environment supporting them. The information will be used to develop predictions of how the phytoplankton species composition may change in response to altered nutrient and hydrologic conditions and how this may affect downstream waters in Florida Bay. Water containing natural populations of phytoplankton from several sites in the lakes and Florida Bay were incubated for 8-16 hours using differential N:P ratios and different N substrates including urea, ammonium, nitrate, and mixed dissolved organic matter acquired from the C-111 marsh basin. Incubations were performed in the spring dry season and in the late summer rainy season.

Results

Pigment ratios were determined to assess the dominant phytoplankton species present. The species response to the presence of various ratios and species of nutrient substrates was assessed by bioassay incubations. The nutrient addition experiments (Glibert, 2011) show that the relative proportion of cyanobacteria, a species of low food quality (zeaxanthin:chlorophyll *a*) was highest at those sites with a high concentration of dissolved organic N and a high molar ratio of dissolved organic P:dissolved inorganic P. The relative proportion of diatoms, a high food quality species (fucoxanthin:chlorophyll *a*) was highest at those sites with the highest concentrations of urea. Cryptophytes (alloxanthin:chlorophyll *a*) were highest at those sites with the highest concentration of ammonia. Dinoflagellates (peridinin:chlorophyll *a*) were highest when dissolved organic P:dissolved inorganic P was lowest.

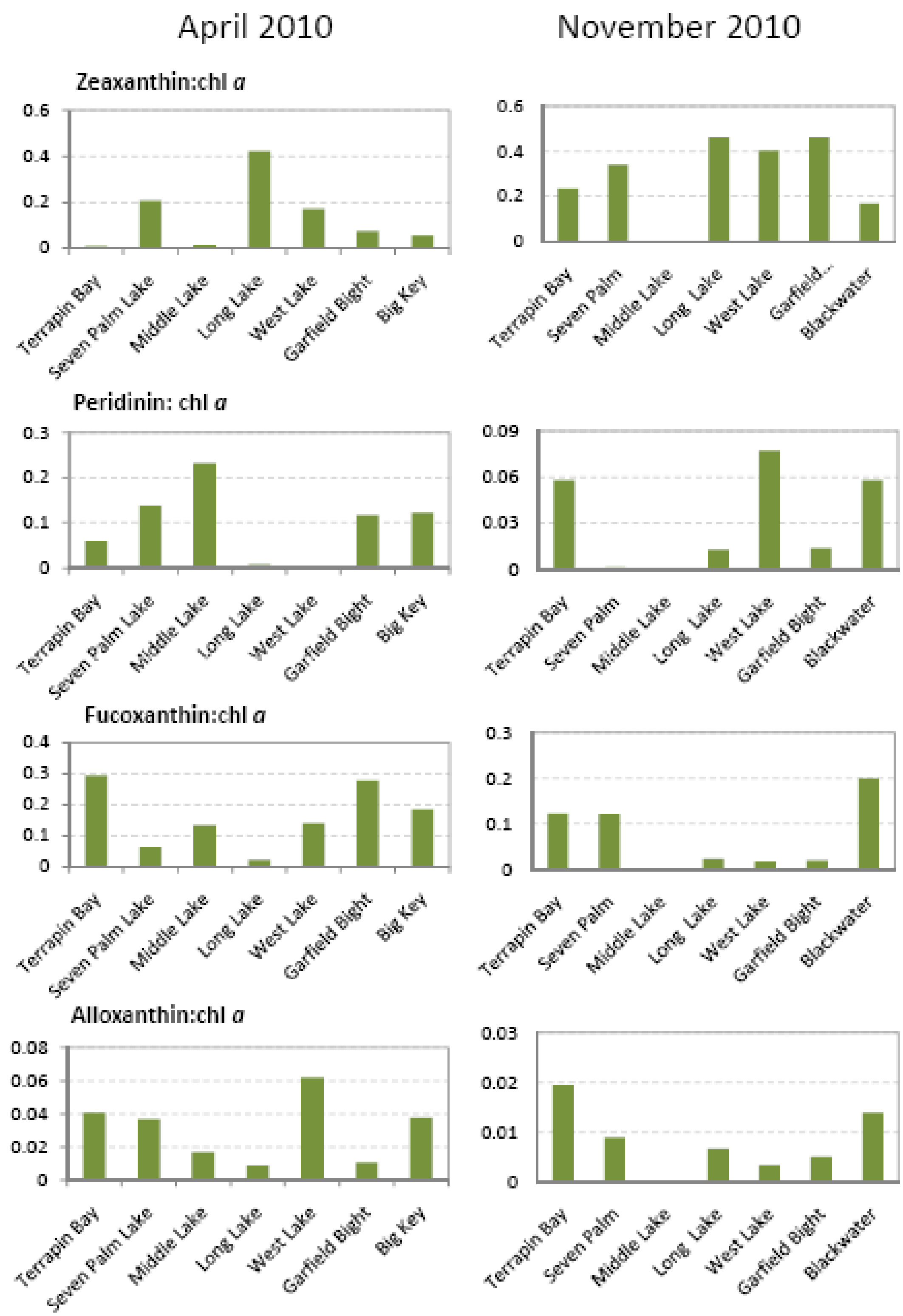


Figure 6–28. Pigment ratios at sites in transitional bays in the ecotone (Terrapin, Seven Palm, Middle, Long, and West lakes) and downstream Florida Bay (Garfield Bight, Big Key, and Blackwater Sound) during dry (April) and wet (November) season sampling. [Note: chl *a* = chlorophyll a.]

Results show that there is significant seasonal variability in species assemblage at many sites with a desirable mix of diatoms and dinoflagellates predominant in the downstream sites of Terrapin Bay, Big Key, and Garfield Bight in the dry season (**Figure 6-28)**. The upstream sites at Seven Palm Lake, West Lake, and Long Lake reflected higher dominance of cyanophytes. In the November sampling at the end of the wet season, cyanophytes were generally more prevalent at nearly all sites, and diatoms were reduced, possibly reflecting a shift in nutrient environment and upstream to downstream transport from the lakes to the open bay.

Relevance to Water Management

The availability of nutrients and nutrient species composition are considered to be major factors controlling the phytoplankton community and the occurrence of blooms in Florida Bay waters. Consequently, the process of phytoplankton nutrient uptake affects nutrient dynamics and ecological processes and characteristics (especially light availability for SAV) in the estuary. Results support the conceptual understanding of plankton dynamics in Florida Bay that has been developed over the past several years — that the phytoplankton community composition is highly related to nutrient form, ratios, and loads. The dominant primary producer in the transitional basins, cyanobacteria, is able to outcompete other phytoplankton classes when P availability is comparatively low, which is the nominal condition in western and central Florida Bay and the associated ecotone. Western Florida Bay has not been characterized by any significant blooms during the study period and biomass response to nutrient stimulation in the western bay was low. The central bay region was more variable, alternating between periods of N and P limitation and between stimulation by inorganic and organic nutrient forms. In the eastern bay, P limitation was evident and if additions of P were to occur, it would likely initiate severe bloom conditions as occurred there in 2005–2008.

SOIL SALINITY TRANSECTS

In a series of baseline measurements in the C-111 Spreader Canal Western Project footprint, north-south transects were established to determine the dynamics of soil salinity in eastern, central, and western sites where hydrology is expected to be altered by the project. The area includes the “white zone” landscape where decreased freshwater input over decades has enabled the encroachment of higher salinity water inland, creating a low quality wetland habitat of increased soil salinity and reduced productivity. The region is named for its higher reflectance on remote sensing imagery due to low plant density. These data represent the first systematic quantitative monitoring of current and potentially changing soil salinity in the region (Troxler, 2010).

Results

Results of the first measurements in the dry season indicate a significant gradient, with higher salinities in the eastern and lower in the western transects **(Figure 6-29**). In Taylor Slough, salinities increased from north to south, and generally with lower depth in the soil. Western Taylor Slough sites showed virtually no soil salinity. The Triangle Land sites, most severely impacted by decades of freshwater diversion away from the area, showed highest salinities, monotonically increasing with depth. Interestingly, the salinity decreased slightly in the downstream direction, likely a result of the interaction of evaporation rate and the degree of incursion of high salinity water inland.

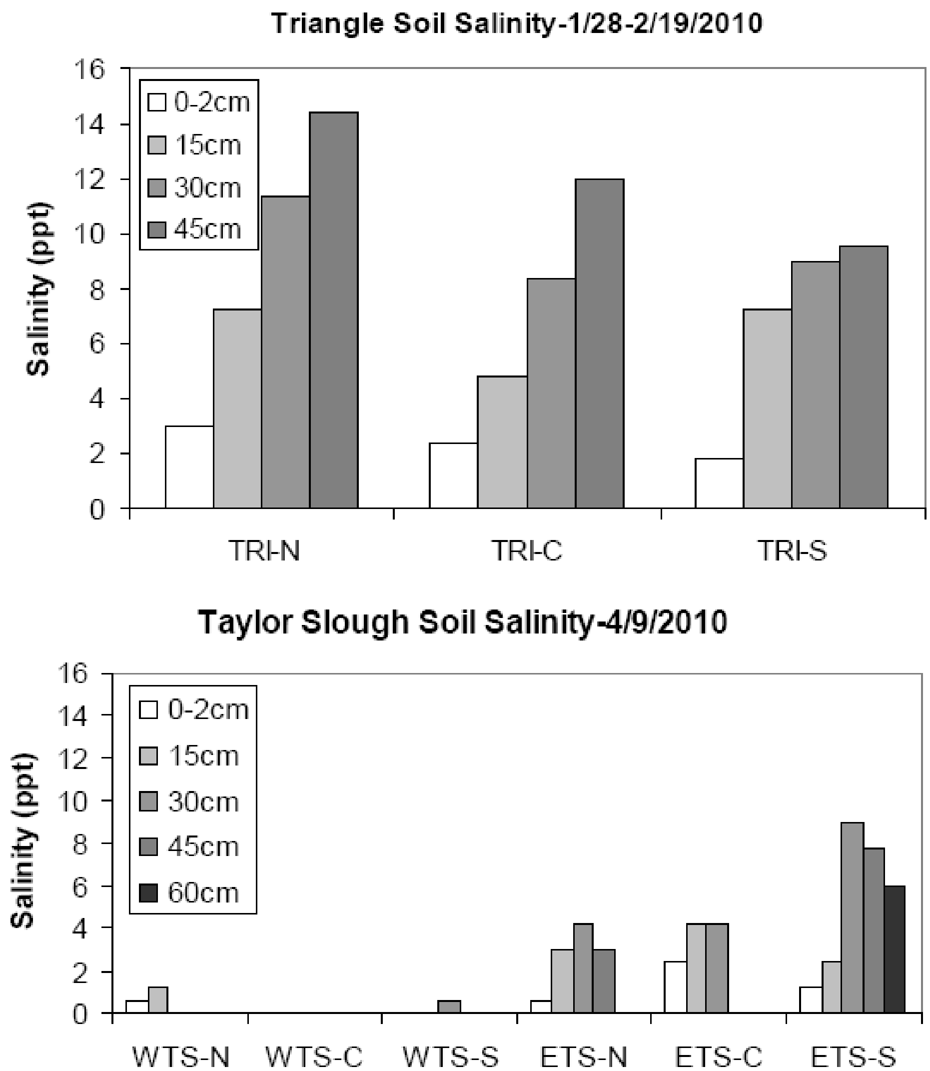


Figure 6-29. Soil salinities [in parts per thousand (ppt); 1 ppt = 1 practical salinity unit (psu)] in the Triangle Lands east of US 1 (WTS-N and WTS-C sites) and western and eastern Taylor Slough (ETS-N, ETS-C, and ETS-S sites) with depth. Salinities reflect an increasing gradient from west to east in Taylor Slough continuing to the Triangle Lands. In Taylor Slough, salinities also increase to the south. In the Triangle Lands, salinities slightly decrease along a gradient from

Relevance to Water Management

The high soil salinities in the Triangle Lands east of US 1 (up to 14 inch deep strata) are indicative of the effects of decades of water withdrawal and blockage to the area through impoundment. The white zone of poor quality wetland habitat resulting from these historical hydrological changes represents a degraded condition that has been targeted for restoration in phase 2 of the C-111 Spreader Canal Project, with a specific goal of reducing soil salinity. Salinity in Taylor Slough soils shows a maximum of about nine in lower soil strata and two or less in the surface stratum. The importance of lower salinities in deeper strata is that the plant root zone extends to these depths where salinity levels control productivity, viability, and species composition.

LANDSCAPE

Jim Brock[5], Christopher McVoy[3], Mark Oates[6],
Fabiola Santamaria[7], Colin J. Saunders, Ted Schall[8]
and Erika Wunderlich[9]

Tree island habitat change analysis over the past century was conducted using vegetation and chemical proxies. In addition, the development and implementation of a means for determining particle movement for the preservation and restoration of the Everglades ridge and slough landscape is described as well as a newly published book on the predrainage Everglades.

AREAL LOSSES AND GAINS IN TREE ISLAND HABITAT OVER THE TWENTIETH CENTURY – INFERENCES FROM COMBINED PALEOECOLOGICAL AND IMAGERY ANALYSES

Though tree islands account for less than 10 percent of the Everglades landscape, they provide critical habitat needed to maintain floral and faunal biodiversity and also significantly affect hydrological and biogeochemical processes of the ecosystem (Troxler Gann and Childers 2006; Givnish, 2008; Wetzel et al., 2008; Ross and Sah, 2011). Over the twentieth century, impacts of water management activities roughly halved the number and total area of tree islands (Patterson and Finck, 1999; Brandt et al., 2000). A key CERP goal includes restoration of the predrainage Everglades ridge-slough-tree island landscape. Achieving this goal requires quantifying how management- and naturally-driven forces (e.g., hydrology, disturbance, or internal ecological feedbacks) impact tree island structure and functioning (e.g., primary production, plant community diversity, and soil accretion), which are fundamental elements of tree island health.

Several lines of field data have demonstrated that extreme hydrologic patterns have degraded tree island forest composition (Sklar and van der Valk, 2002; Rutchey et al., 2008; Wetzel et al., 2008), but most information does not extend beyond the last 20 years (Wetzel et al., 2008). Two additional lines of evidence — paleoecological analyses of soil cores and aerial imagery analysis — extend our timeline of known Everglades tree island and ridge and slough landscape dynamics (Willard et al., 2006; Bernhardt and Willard, 2009). With these advances, however, there remain important questions and uncertainties about tree island dynamics. Because the majority of woody species are pollinated by insects, the pollen record is often sparse compared to soil cores from herbaceous marsh species (Willard et al., 2006). Additional paleoecological methods including macrofossil analyses (mainly seeds) can potentially provide information about the dynamics of woody species on tree islands. Macrofossils record more localized organic matter deposition (5–10 m resolution; Saunders et al., 2006, 2008) and thus have the potential to produce spatially explicit reconstructions of tree island areal loss or gain, and the hydrologic conditions underlying those changes. Moreover, because fossil seeds can be individually dated using accelerated mass spectroscopy (AMS) methods, with an age resolution of ± 2–5 years (Donders et al., 2004), they provide more precise temporal resolution compared to other radiometric dating methods. These methods thus enable us to assess long-term effects of water management activities and natural climate driven hydrologic variability on tree island structure and function.

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Recently, spatially explicit soil core transects, distributed along vegetation boundaries have characterized the stability of ridge and slough habitats (Bernhardt et al., 2009) and ridge-slough ecotone shifts (Saunders et al., 2008, and in prep). In this study, paleoecological analyses of soil cores taken along slough-ridge-tree island transects in WCA-3 paired with aerial imagery analysis were used to resolve past rates of tree island habitat change from predrainage period (pre-1900) to present. Objectives of this study are to (1) quantify the timing and spatial extent of vegetation change and correspondence with past hydrologic variation, including water management activities; and (2) test the degree of consistency between two methods — macrofossil and aerial imagery-based inferences — in reconstructing vegetation changes. Specifically, the study aims to address the hypothesis that tree island habitat changes in the twentieth century primarily coincide with and reflect impacts of the construction (1960–1963) and subsequent enlargement (1970–1973) of the L-67 canal and levee, which effectively impounded southern WCA-3A (Light and Dineen, 1994).

The study site is tree island 3AS5 in WCA-3 (**Figure 6-30**). Report results are combined paleoecological and imagery analyses conducted on tree island 3AS5 in southeastern WCA-3A, located upstream of the L-67 canal and levee and consequently exposed to prolonged hydroperiods and high water levels. Subsequent reporting will include results from island 3BS2 in WCA-3B (downstream of the L-67C), subjected to shorter hydroperiods and lower water levels.

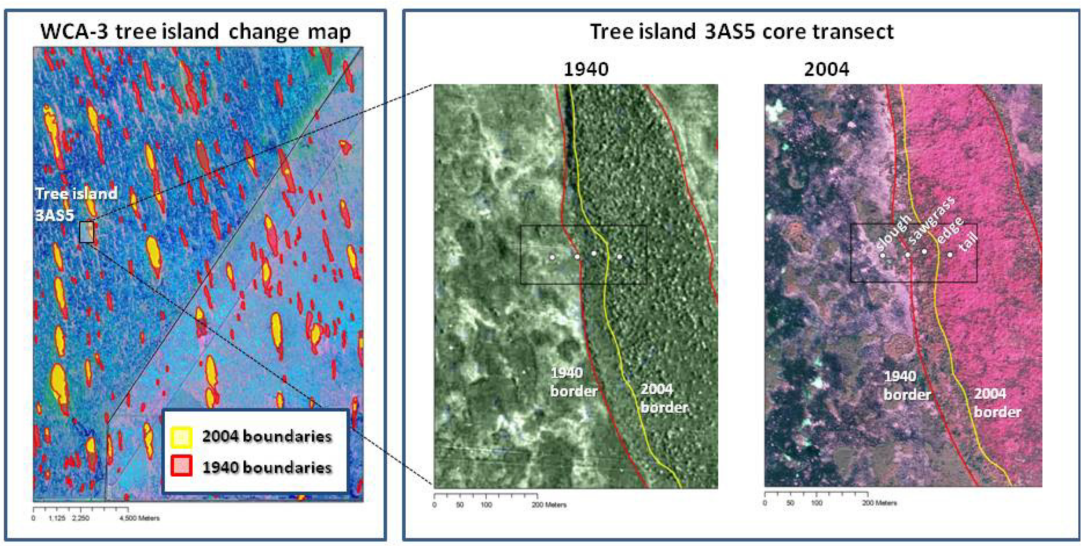

Figure 6-30. Map of the study site showing tree island 3AS5 and aerial imagery of core transect, spanning slough-sawgrass-edge-tail habitat in 1940 and 2004.

Methods

Four soil cores (10-cm diameter, 60-cm length) were collected along a 75 m slough-to-tree island transect at island 3AS5 **(Figure 6-30)**. Cores were extruded in the lab at 1 cm intervals, radiometrically dated, and analyzed for macrofossils. Macrofossil analyses were conducted on particles greater than one millimeter (mm) and greater than 500 micrometers, which contained identifiable (under 30 times magnification) seeds, seed fragments, and other identifiable macrofossils (characteristic plant leaf or root tissues), based on characteristics of seeds and fragments from standards collected from live plants and surficial litter samples. Seeds were carefully recovered from selected soil samples, extracted with forceps, and cleaned with deionized water to reduce contaminating carbon sources. Depths with high seed abundance were selected for 14C-AMS dating. In this report, a subset of the macrofossil data is presented to indicate the primary hydrologic or ecological conditions: these include deep-water sloughs (water lily tissue), intermediate hydroperiods (sawgrass leaf sclereids), open-canopy marsh habitats within tree islands (seeds of broadleaf marsh species), and a woody-species indicator (wax myrtle, *Myrica cerifera*, leaf fragments). Other macrofossil data collected (including seeds of sawgrass, water lily, banana lily, and wet prairie sedges; fern tissue fragments; and charophyte spores) corroborate findings presented in this report. Age models were generated based on 14C AMS dating of fossil seeds (Turetsky et al., 2004; Donders et al., 2004) and of bulk soil, when seeds were scarce. Fossils were initially dated by calibrating 14C concentrations (F14C, units: percent Modern Carbon, or pMC) against the twentieth century atmospheric bomb-14C curve using function Sequence in Oxcal v.4.1 (Bronk Ramsey, 2008, 2009). Radiocarbon ages from premodern (pre-1900) depths were calibrated individually using INTCAL04 radiocarbon age calibration (Talma and Vogel, 1993; Reimer et al., 2004). A locally weighted regression function in the Splus statistical software was then used to generate an age model and 95 percent confidence interval for each core (Lynch et al., 2002). Standard acid/alkali pretreatments of soil and seeds were performed prior to analysis. All 14C analyses were performed by Beta Analytic, Inc., Miami, FL, and National Ocean Sciences AMS Facility at Woods Hole Oceanographic Institution in Woods Hole, Massachusetts.

Vegetation mapping was carried out for a 50 m rectangular buffer zone around a four soil core transect located along the western edge of 3AS5 (**Figure 6-30**). The mapping was accomplished using manual, stereo image interpretation techniques aided by modern field sampling. The vegetation mapping process began with the identification of the vegetation depicted in the most modern imagery and progressed backwards in time by decade. Changes were made to each decade's vegetation data (with exception to the most modern) if the imagery demonstrated a significant change from the next most modern vegetation map. Vegetation cover was delineated and labeled according to the Vegetation Classification for South Florida Natural Areas (Rutchey et al., 2006) to a level of specificity that could be adequately determined from each specific set of imagery (i.e., more specific vegetation information could be determined from the color infrared and higher resolution imagery than could be determined from the courser resolution and panchromatic imagery). Multiple classes were identified for a single polygon whenever the image patterning suggested the presence of mixed classes and those mixed classes could be adequately identified.

Results

High-precision 14C AMS dating utilizing the twentieth century atmospheric F14C curve (Turetsky et al., 2004) here presents a novel application for Everglades soils. Soil macrofossil F14C values exhibited down core patterns consistent with the twentieth century atmospheric F14C curve (Hua and Barbetti, 2004), exemplified by the slough and tail cores in **Figure 6-31**. Over soil depths capturing the twentieth century period, macrofossil F14C values were precise enough that, when matched against the atmospheric F14C curve, the resulting age uncertainty was, on average, ± 2 years (with a 95 percent confidence interval) for a given fossil sample in the tail core and ± 11 years for samples from the slough core. The largest variation occurred in pre-1950 depths in all cores.

Macrofossil-based vegetation changes are presented over the 1925–2008 period, which contained the best soil age resolution to allow comparisons with water stage and rainfall records and aerial imagery. To facilitate comparisons with aerial imagery data, macrofossil data are presented spatially, as smoothed contour plots versus time and transect distance (**Figure 6-32**). Transect distance is hereafter expressed in meters relative to the 1940 tree island boundary, with positive (negative) values representing locations interior to (outside of) the 1940 boundary.

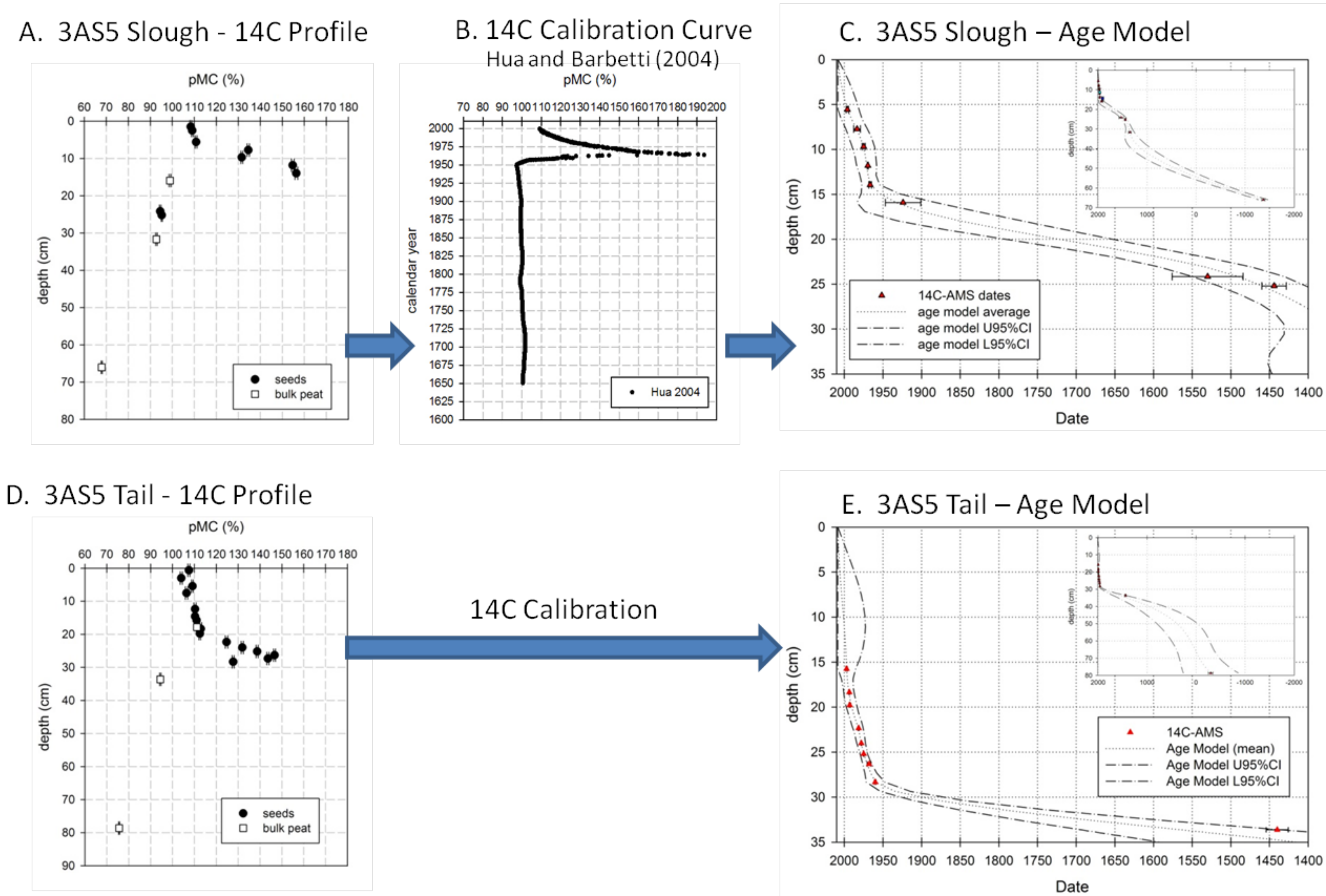

Figure 6-31. Profiles of fossil seed (filled diamonds) and bulk peat (open diamonds) F14C [percent of modern carbon (pMC)] from two representative cores (A and D are slough tail sites with mean ± 2 SE) at 3AS5. Atmospheric-F14C curve in B is used to date the individual samples (red triangles, mean ± 2 SE), and individual dates are used to generate age models (dotted line=mean; dashed lines=95 percent confidence interval) for the entirety of the core (shown for slough D and tail E cores). Inset graphs show age models for entire cores (0–80 centimeters).

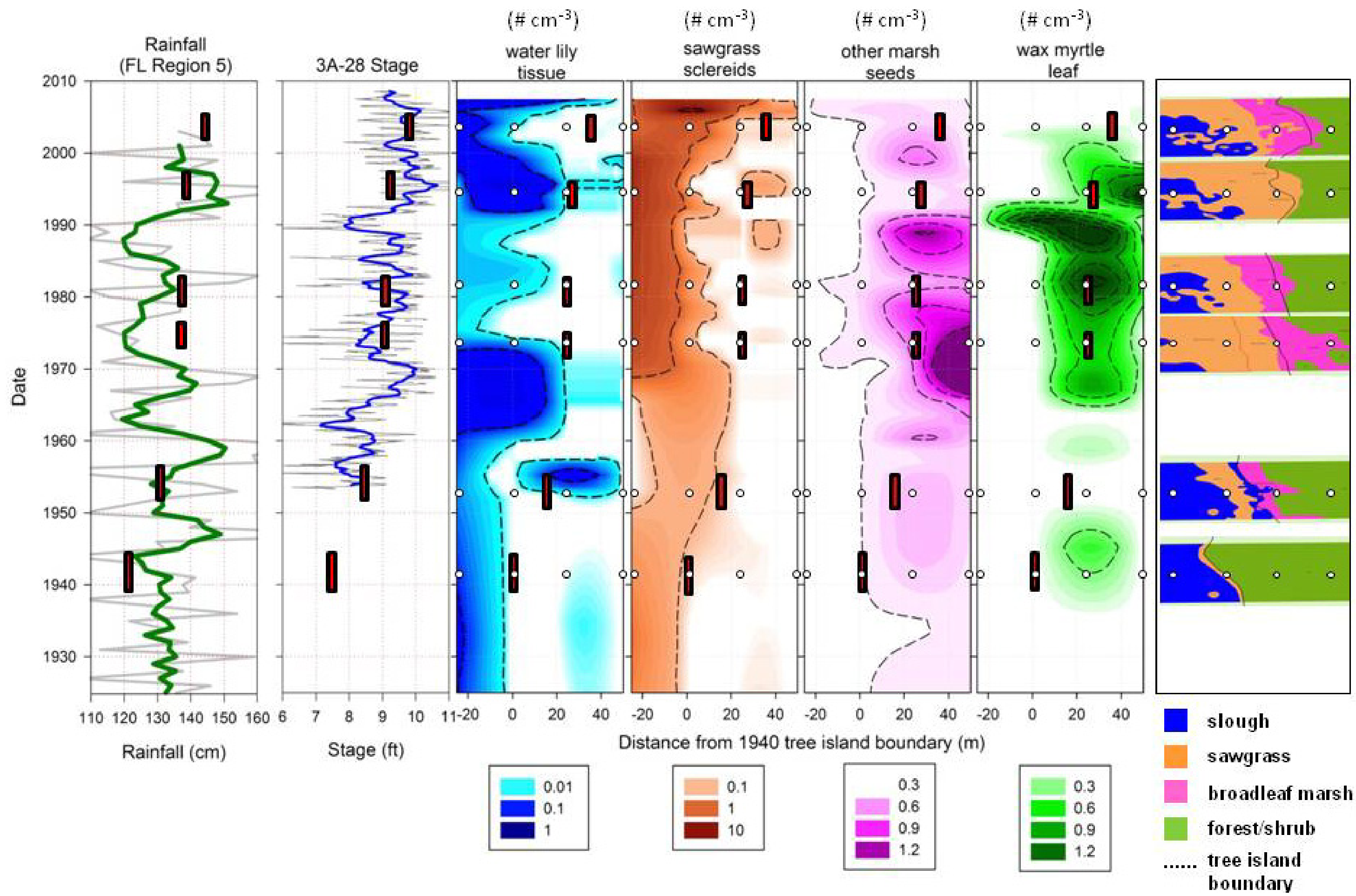

Figure 6-32. Florida Region 5 rainfall (cm, annual and five-point smooth), WCA-3A stage from gauge 3A-28 [feet North American Vertical Datum of 1988 (NAVD88)], and tree island vegetation from 1925 to 2008. Macrofossil-based vegetation reconstruction is from macrofossil abundance [number of plant fragments or seeds per cubic centimeter (# cm^{-3})] in soil cores (superimposed as white symbols) located along a slough-ridge-tree island transect. Transect distance is represented as meters inside (positive values) and outside of (negative values) the 1940 island boundary inferred from aerial imagery (far right column). Vegetation classes from aerial imagery represent combined vegetation types to facilitate comparisons with macrofossil data. Tree island boundaries, estimated from aerial imagery, are shown as brown rectangles superimposed on macrofossil graphs.

From 1925–2008, the primary vegetation changes inferred from soil macrofossils included (1) a general reduction in island area indicated by interior expansion of water lily and sawgrass across the entire 75 m transect; (2) a mid-twentieth century rise and loss of broadleaf marsh species in the tree island interior; and (3) areal expansion (late 1980s) and rapid reduction (early 1990s–present) of woody species (wax myrtle) macrofossils. From circa 1925–1945, water lily and sawgrass macrofossils were confined to locations outside the 1940 island boundary (0–25 m). From 1945–1960, sawgrass shifted east into the island (circa -20 to +20 m), and from 1950–1955 an isolated water lily patch appeared in the island interior (centered at +25 m), corroborated by 1953 aerial imagery. From 1960 to 1970, water lily macrofossils expanded in spatial extent, spanning distances of -25 m to +20 m across the 1940 boundary. At the same time, broadleaf marsh species increased in abundance, but this occurred primarily inside the island interior (+25 to +50 m). The latter result was also corroborated by 1973 imagery, showing widespread broadleaf marsh habitat in the island interior.

From 1970 to 1990, peak sawgrass fossil concentrations shifted to areas outside the 1940 boundary and into the slough (-25 m), but sawgrass remained abundant at +15 m. During this period, water lily concentrations were also highest at locations farthest from the island (-25 m) but varied in spatial extent from -15 m (circa 1972) to +15 m (circa 1983) relative to the 1940 boundary. Broadleaf marsh species remained in highest concentrations +25–50 m inside the 1940 island boundary, but decreased in concentration during the 1980–1985 period. Around 1980, wax myrtle, a flood tolerant woody species, increased in abundance at +25 m. By 1990, wax myrtle macrofossils reached maximum concentrations and spanned a +50 m distance, ranging from around -20 m to +30 m. Vegetation changes in the late 1980s were sharply reversed after 1990: water lily and sawgrass macrofossils increased in concentration and spatial extent. During the mid-1990s, both were present across the entire transect. Wax myrtle fossils reduced in concentration to near below detectable levels, and both macrofossil and imagery (1995) indicated a reduction in broadleaf marsh in the island interior. From 2000 to 2008, the trend of the 1990s continued: water lily and sawgrass fossils became the dominant species across the entire 75 m transect.

Discussion and Relevance to Water Management

The combined approach of reconstructing tree island vegetation using both macrofossil and aerial imagery was useful for quantifying the rate and extent of tree island habitat loss due to twentieth century water management activities. Both approaches yielded strong agreement on the timing and spatial extent of vegetation changes. These included subtle vegetation changes such as the localized rise in water lily abundance interior to the island (+25 m) in the mid-1950s and increased abundance of broadleaf marsh species in interior portions of the island (+50 m) in the early 1970s. The macrofossil data, however, were critical in providing vegetation reconstructions for the 1960–1970 period, encompassing the construction of the L-67 canal and levee (1960–1963), and for which aerial imagery is lacking. The timing of increased abundance and aerial expansion of water lily and sawgrass into the island interior (>+25 m) from the macrofossil record in the early 1960s is consistent with the hypothesis that the initial phase of L-67 construction reduced tree island area by raising water stages and increasing hydroperiod within the islands. This finding confirms similar findings from previous tree island paleoecological data (Willard et al., 2006); however, 14C dating of the macrofossils (rather than bulk soil) and the spatial design of soil cores in this study adds greater precision on the timing and extent of areal loss (approximately 25 m loss within a few years) that is likely associated with L-67 construction. Additionally, while both macrofossils and imagery indicate an increase in broadleaf marsh species in the island interior (+50 m) by 1973, the macrofossil data confirm that this (likely) disturbance also originated during L-67 construction.

While imagery data are not available for the 1980s, the macrofossil data importantly show that in the dry years of 1989 and 1990, wax myrtle, a flood tolerant woody species, expanded 25 m westward into the slough, indicating a positive expansion of tree island area relative to the 1940 island boundary. This result suggests that hydrologic conditions similar to 1989–1990 may provide operational targets for restoring tree island habitat. However, the 1989–1990 hydrologic conditions did not produce a resilient tree island habitat: a rapid and substantial loss of wax myrtle and rise in water lily sloughs occurred throughout the 1990s, characterized by consistently wet hydrologic conditions.

This pilot study shows that spatially explicit paleoecological methods can be used to measure the rate and spatial extent of tree island habitat changes in relation to water management and natural climate variation, and this method is corroborated by aerial imagery. While the study offers insights about hydrologic targets appropriate for tree island restoration, important caveats remain. In this study, poor preservation and reduced soil age resolution exists in the early to pre-twentieth century depths, requiring future studies to apply finer depth sampling (0.5 cm or less, instead of 1 cm). In addition, the results shown here could be influenced by highly localized vegetation responses; thus, replicate cores are needed to provide a more complete picture of the spatial variation at any one site. At this time, no ideal macrofossil representing tree island hardwood hammock habitat has been recovered and identified. While some seeds have been found in early twentieth century and predrainage depths within the interior tree island core (data not shown), these require further identification. Recent advances in molecular organic biomarkers for tree island woody species biomass offer some promising and complementary approaches in identifying past tree island boundary shifts (R. Jaffé, Florida International University, unpublished data).

PRESERVING THE RIDGE AND SLOUGH LANDSCAPE: FLOCCOMETER TRANSPORT STUDIES

The ridge and slough landscape encompasses most of the presently remaining Everglades. This peat-based, highly directional, and "patterned" wetland was named in 1915 for its arrangement of elongated sawgrass ridges alternating with interconnected water lily sloughs. Though both were underlain by peat, ridges historically were approximately 1.5 foot (45 cm) higher than sloughs. Elevated tree islands (60–120 cm, 2–4 feet above slough bottom) formed the third element of the original ridge and slough geomorphology. The landscape patterning — streamlined and matched to the regional topography — has led many observers to hypothesize that surface water flow formed and/or maintained the patterning. The losses of ridge and slough patterning (both horizontal and vertical) observed during the last half century of impounded conditions (Bernhardt and Willard, 2009; Larsen et al., 2011; Nungesser in press; Watts et al., 2010) and the associated reductions in water flow may be an additional indication that flow and/or sediment transport were necessary geomorphological drivers. We describe here a new monitoring platform designed to investigate the driving forces and processes influencing landscape pattern in the Everglades.

A mechanistic understanding of the processes linking water flow, carbon cycling, and ridge and slough geomorphology is needed to provide an informed basis for planning and implementing restoration. The key element is likely to be the process or combination of processes that maintained the elevation differential between sloughs and ridges, specifically the processes that prevented slough infill. Predrainage water flow, while frequently reported to have been visible (McVoy et al., 2011), was almost certainly too slow to physically erode fibrous peat. If carbon export out of sloughs was responsible for maintaining slough elevations, then it was most likely associated with transport of the near-neutrally buoyant, unconsolidated particulate flocculent organic material still found in the remaining soft water portions of the Everglades. This material typically occurs in sloughs as a loose, 0–30 cm thick layer of coarse-grained detritus or gyttja at the bottom of the water column, and is here referred to as "floc."

The floccometer platform is focused on identifying and quantifying, in situ, causal relationships between regional-scale Everglades driving forces and local floc behavior, particularly processes that can mobilize floc upward into the water column (entrainment) where it would then be subject to downstream transport. Entrainment by shear stress is one possible process, but floc buoyancy suggests that other processes not dependant on exceeding threshold velocities may also be important. Possibilities include thermal inversions and photosynthesis-induced buoyancy changes (gas production). The floccometer platform was designed to gather two sets of continuous, long-term (multiple wet and dry seasons) time series measurements: (1) regional-scale Everglades driving forces and (2) localized floc responses to these forces. The floc-related time series variables include precise elevation of the floc-water interface; water column and floc layer profiles of dissolved oxygen, pH, and conductivity; and elevation of the peat surface underneath the bottom of the floc layer. Subsurface video clips of particle movement are also captured at regular intervals. The time series of driving forces include meteorological data (wind speed and direction, air temperature, rainfall, atmospheric pressure, and solar radiation); vertical profiles of the water column and floc layer temperature, light penetration, and water velocity; and water stage, depth, and slope.

Methods

The floccometer design consists of (1) a staging platform supporting control and logging electronics, a meteorological station, two acoustic Doppler current profilers, a stage gauge, a webcam, and remote Internet connectivity; (2) a waterproof instrument "pod" (multiparameter sonde) with downward projecting sensor probes; (3) a supporting truss with a computer-controlled drive to vertically position the instrument pod; (4) a set of linear rails and computer-controlled drive to horizontally position the truss and pod in the slough anywhere between 0 and 10 m away from the platform (**Figure 6-33**); and (5) a "garage" where the truss and pod "rest" when not actively sampling. The platform is located near the east side of a slough approximately 2.1 kilometers north of the S-12C structure.

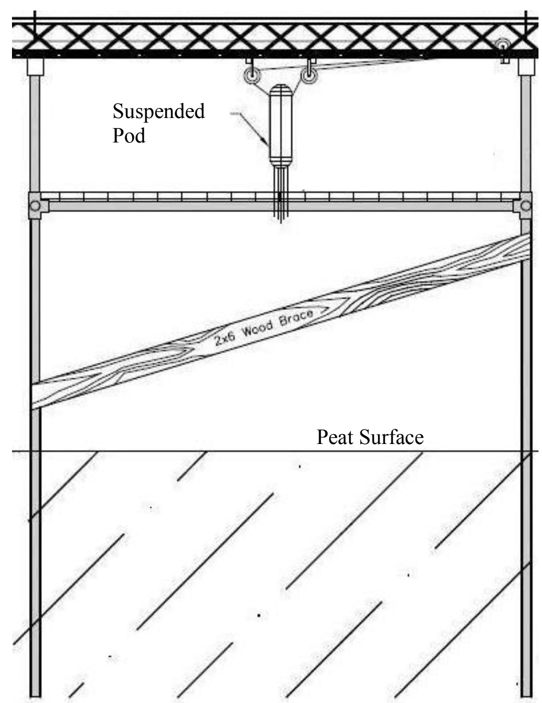

Figure 6-33. Diagram of the Floccometer 101 instrument pod and supporting rail structure.

The site was chosen to represent sloughs in southern WCA-3A, an area with water depths similar to predrainage water depths. The specific slough was chosen for (1) its history of relatively minimal airboat traffic, (2) its similarity to surrounding sloughs, (3) as a location likely to experience sheet flow toward the S-12 structures, and (4) its alignment between the S-12C headwater stage gauge and the 3A-28 (Site 65) stage gauge. The 3A-28 gauge has been in place since 1953 and is the southernmost of the three gauges used in WCA-3A regulation schedules.

The floccometer is fully solar powered and Internet connected. Vertical and horizontal motion is under autonomous, on-site computer control with remote override capability. Sampling is temporally regular with event-driven increases in sampling intensity. Vertical sampling is regularly spaced with modifications based on the sensed elevations of the water surface, the floc layer surface, and the peat surface. All three elevations are measured to very high resolution (≤ 0.3 mm) to allow sensitive change detection and to detect floc movement. Horizontal positioning along the 10 m long linear rails provides multiple spatial replications within the slough of the vertical sampling profiles. Robotic control of both vertical and horizontal positions of the floccometer sensor pod allows the sensor probes to be removed from the water, and to be automatically recalibrated between sampling, solving many of the biofouling and drift problems that have typically challenged long-term field studies in aquatic systems.

All pod sensor probes have been designed to minimize disruption of the water column and the floc layer. All probes are long (20 cm), narrow (< 6 mm diameter), and smooth (stainless steel, aluminum, or carbon fiber). The probes have been either specially designed and fabricated for this study or adapted from commercially available probes. Precise computer control keeps the vertical motion as slow as necessary to avoid disturbance. Floc detection is by beam interruption of an audio-modulated infrared beam. Peat surface detection is by weighted plunger, magnet, and Hall effect sensor. Dissolved oxygen is measured using optical fluorescence (Ocean Optics, "Neofox") and pH is measured by ISFET technology. A waterproof borescope and pod-housed webcam capture floc movement. All signal conditioning electronics is housed inside the pod, which wirelessly transmits to logging electronics on the platform. Additional probes may be added in the future (e.g., oxidation-reduction potential).

Proximity to the S-12 outflows measures an important driver of water and floc behavior at the floccometer site. As the S-12 structures open and close, southern WCA-3A alternates between nonflowing, level pool conditions (**Figure 6-34**) and sloped, flowing conditions. These measurements allow differentiation of floc behavior with and without flow.

A publicly accessible webcam allows remote monitoring of the floccometer and slough. Once daily capture of slough snapshots is automated, the library will be available for vegetative analysis, for example, leaf area index of water lily, changes in wet prairie species abundance, and possibly quantification of apple snail (*Pomacea paludosa*) egg masses. In addition to its research goals, the floccometer project will contribute to public education and outreach by providing a real-time, multivariable "window on the Everglades."

Preliminary Results

Progress and proof-of-concept to date at the field platform includes stage data (**Figure 6-34**), webcam video, and meteorological data transmitted continuously from the platform for one year. This successful performance included an eight-month period during which the platform collected data during both wet season and drought condition water levels, with zero maintenance. **Figure 6-34** illustrates use of floccometer stage data to identify the level pool period during the wet and early dry season, as well as later deviations from level pool. The stage separation between Site 65 (3A-28 gauge) and the floccometer seen in May and June revealed that the 3A-28 gauge actually measures ridge rather than slough hydrology, which was later confirmed with field transects. Progress to date of the positioning robotics and sensing instrumentation includes the following:

- Vertical position resolution of 0.25 mm with 1-in-10,000 repeatability
- Floc layer and peat surface detection with 0.25 mm resolution and high repeatability
- Automated head calibration to maintain vertical datum
- Slow speed control to allow probe movement without floc layer disruption

- Horizontal position control to within 2 mm
- Timed and event-driven borescope video clip capture
- Preliminary indications of interaction between light penetration and floc layer elevation

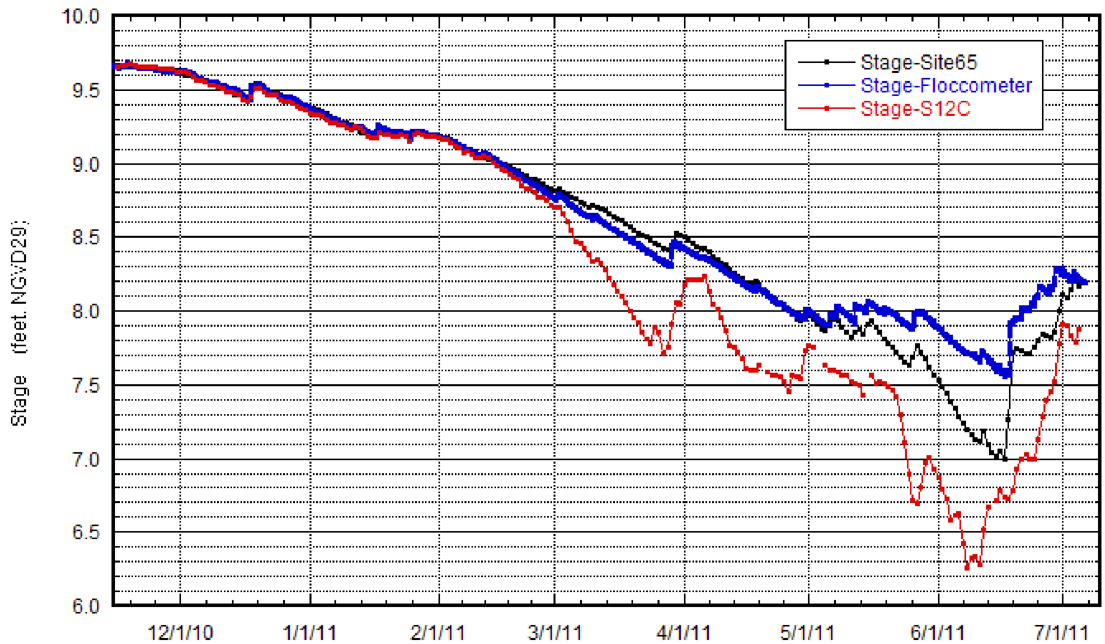

Figure 6-34. Stage data from the floccometer, gauge 3A-28 (Site 65), and the headwater gauge at the S-12C structure. Level pool conditions (all three gauges equal), which are reflective of impoundment, occurred through the end of February. Deviation of Site 65 and floccometer gauges in May–June reflects below ground stage at Site 65, which is located on edge of a sawgrass ridge.

The very high vertical precision achieved in detecting the position of the top of the floc layer, and the robustness of the detection methodology has made it possible to apply state-of-the-art statistical time series tools (e.g.,; Little and Jones, 2011; Little et al., 2011). For appropriate data sets such as that obtained from this study, these tools can distinguish between modal or stepped processes and continuously varying processes (**Figure 6-35**). In the Everglades, this corresponds to being able to distinguish between noncohesive floc layers typical of less disturbed, soft water sites and cohesive flocs such as those found in the presence of high alkalinity water.

Relevance to Water Management

Operation of the floccometer will produce unique types and durations of Everglades ridge and slough landscape data, including floc thickness, peat elevation, hourly dissolved oxygen and thermal profiles, and net community carbon balance within the water column and floc layers. While each dataset will itself be significant, the greatest significance will come from having simultaneous time series data for both driving force parameters and for floc response parameters. As the floccometer platform continues to measure during wet and dry seasons and during both flowing and nonflowing periods, it will help guide both operations and restoration planning by quantifying the field conditions needed for preservation of sloughs through downstream carbon export.

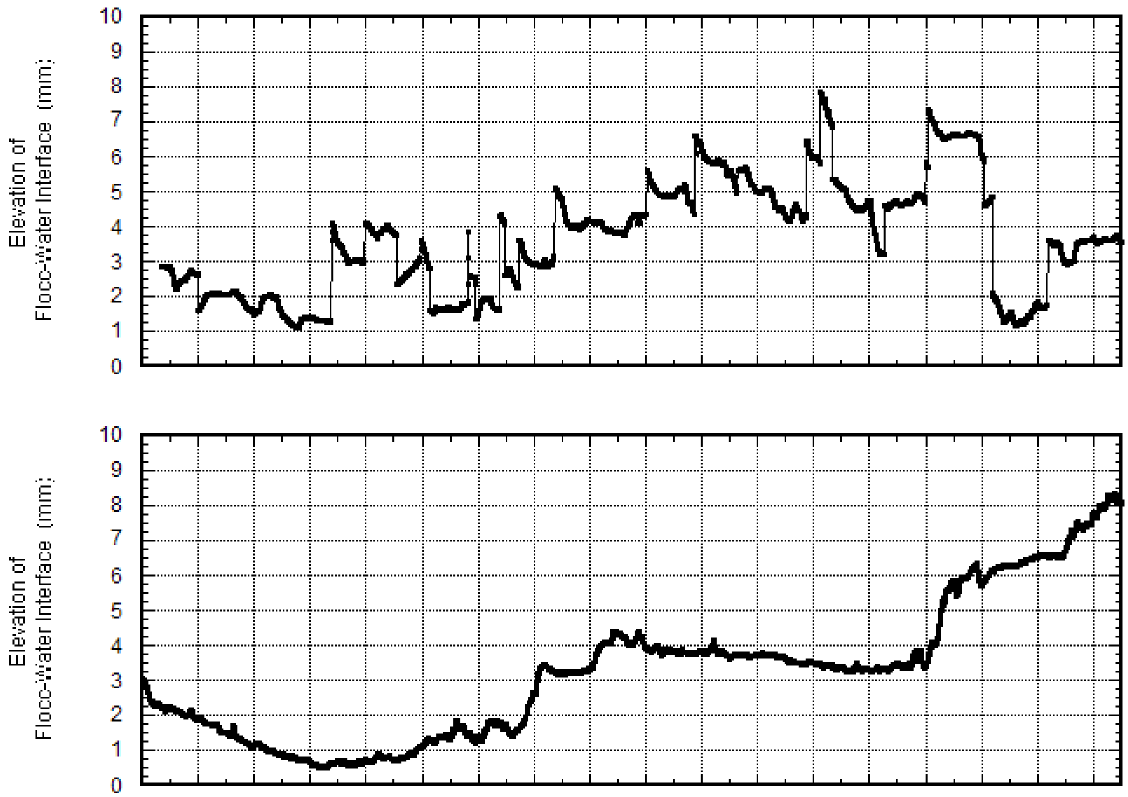

Figure 6-35. Characterization of two different floc types using high frequency (90 second sampling interval) measurement of floc elevation [in millimeters (mm)] using the floccometer. Data were collected during different time periods, but both time series are 36 hours long. The top graph is for a cohesive floc collected in WCA-3B. The bottom graph is for a noncohesive floc collected from the Floccometer 101 site. This floc is typical for sloughs of the central and southern portion of WCA-3A. Note the multitude of instantaneous elevation "jumps" or steps in the cohesive floc and absence of steps in the noncohesive floc.

LANDSCAPES AND HYDROLOGY OF THE PREDRAINAGE EVERGLADES PUBLICATION

In May 2011, the book entitled Landscapes and Hydrology of the Predrainage Everglades (McVoy et al., 2011) was published. This book, the product of over a decade of research, synthesizes more than 900 historical sources to characterize the historic (circa 1850s) Everglades. By combining numerous firsthand historical observations with modern Geographic Information System (GIS) mapping tools, the authors were able to develop a detailed characterization of the predevelopment Everglades as a whole. The specific objectives of this study were to do the following:

- Characterize and map the vegetation, soils, regional topography, microtopography, and hydrology of the predrainage landscapes of the Everglades.
- Quantitatively estimate predrainage water depths and hydroperiods for each landscape.

- Characterize and map the landscapes bordering the Everglades sufficiently to understand their individual relationship to overall Everglades hydrology (**Figure 6-36**, left).
- Map predrainage patterns of Everglades water flow (**Figure 6-36**, right).
- Document changes that occurred within the Everglades between the time of first drainage (1880s) and the time of the first systemwide, scientific mapping (1940s).
- Make available to the reader as much as possible of the historic data used to achieve the previous objectives (e.g., photographs, maps, and essential portions of historical accounts).

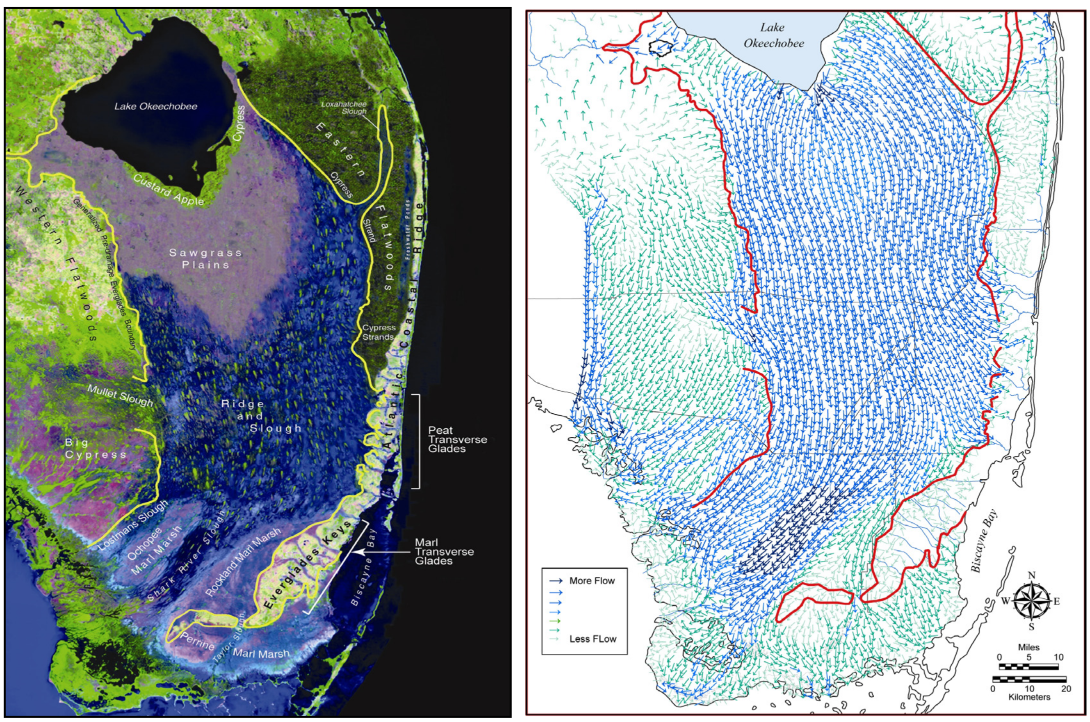

Figure 6-36. The landscapes (left) and hydrology (right) of the predrainage Everglades.

This study identified flows of water into and out of the Everglades as well as patterns of internal flows. Understanding of flows, combined with definition of water depths, captured predrainage hydrologic functioning of this wetland. Overall, this study's reconstruction of predevelopment soils, hydrology, geomorphology, and vegetation provides validation tools for regional models, a yardstick for evaluating current conditions, and reference conditions to guide restoration. Tracing of past changes caused by man-made drainage and impoundment provide guidance regarding likely Everglades trajectories under future management scenarios.

LITERATURE CITED

Aich, S., T.W. Dreschel, E.A. Cline and F.H. Sklar. 2011. The development of a Geographic Information System (GIS) to document research in an Everglades physical model. *Environmental Science and Engineering*, 5(3):289-302

Audubon. 2011. Fourth Quarter Status Report for Water Year 2011 (covering conditions from February 1, 2011–April 30, 2011). Submitted by Audubon of Florida Tavernier Science Center, Tavernier, FL under Contract No. 4600001851, South Florida Water Management District, West Palm Beach, FL.

Bernhardt, C.E. and D.A. Willard. 2009. Response of the Everglades ridge and slough landscape to climate variability and twentieth century water management. *Ecological Applications,* 19(7):1723-1738.

Brandt, L.A., K.M. Portier and and W.M. Kitchens. 2000. Patterns of change in tree islands in Arthur R. Marshall Loxahatchee National Wildlife Refuge from 1950 to 1991. *Wetlands,* 20(1):1-14.

Bronk Ramsey, C. 2009. Dealing with outliers and offsets in radiocarbon dating. *Radiocarbon*, 51:1023-1045.

Bronk Ramsey, C. 2008. Deposition models for chronological records. *Quaternary Science Reviews*, 27(1-2):42-60.

Délyea, C, F. Pernina, and L. Scarabelb 2011. Evolution and diversity of the mechanisms endowing resistance to herbicidesinhibiting acetolactate-synthase (ALS) in corn poppy (*Papaver rhoeas* L.). *Plant Science,* 180: 333–342.

Donders, T.H., F. Wagner, K. Van Der Borg, A.F.M. De Jong and H. Visscher. 2004. A novel approach for developing high-resolution sub-fossil peat chronologies with C-14 dating. *Radiocarbon,* 46(1):455-463.

Dorn, N.J. 2008. Colonization and reproduction of large macroinvertebrates are enhanced by drought-related fish reductions. *Hydrobiologia,* 605:209-218.

Dorn, N.J. and J.C. Trexler. 2007. Crayfish assemblage shifts in a large drought-prone wetland: the roles of hydrology and competition. *Freshwater Biology,* 52:2399-2411.

Dorn, N.J., M.I. Cook, G. Herring, R.A. Boyle, J. Nelson and D.E. Gawlik. 2011. Aquatic prey switching and urban foraging by the white ibis *Eudocimus albus* are determined by wetland hydrological conditions. *Ibis,* 153:323-335.

Elser, J.J., T.H. Chrzanowski, R.W. Sterner and K.H. Mills. 1998. Stoichiometric constraints on food-web dynamics: A whole-lake experiment on the Canadian shield. *Ecosystems,* 1:120-136.

Eppinga, M.B., M. Rietkerk, W. Borren, E.D. Lapshina, W. Bleuten, and M.J. Wassen. 2008. Regular surface patterning of peatlands: Confronting theory with field data. *Ecosystems,* 11:520-536.

Ferone, J.M. and K.J. Devito. 2004. Shallow groundwater-surface water interactions in pond-peatland complexes along Boreal Plains topographic gradient. *Journal of Hydrology,* 292:75-95.

Frederick, P.C. and J.C. Ogden. 2001. Pulsed breeding of long-legged wading birds and the importance of infrequent severe drought conditions in the Florida Everglades. *Wetlands*, 21: 484-491.

Frederick, P.C., D.E. Gawlik, J.C. Ogden, M.I. Cook and M. Lusk. 2009. The white ibis and wood stork as indicators for restoration of the everglades ecosystem. *Ecological Indicators,* 9S:S83–S95.

Frezza, P.E., M. Robinson and J.J. Lorenz. 2011. Monitoring Ecosystem Response to Water Management Practices in the Everglades Mangrove Zone: C-111 and Taylor Slough Basins. 2009-2010 Annual Report Submitted under Contract No. 4600001851 to South Florida Water Management District, West Palm Beach, FL.

Gawlik, D.E. 2002. The effects of prey availability on the numerical response of wading birds. *Ecological Monographs,* 72:329-346.

Givnish, T.J., J.C. Volin, V.D. Owen, V.C. Volin, J.D. Muss and P.H. Glaser. 2008. Vegetation differentiation in the patterned landscape of the central Everglades: importance of local and landscape drivers. *Global Ecology and Biogeography,* 17(3):384-402.

Glaser, P.H., G.A. Wheeler, E. Gorham and H.E. Wright, Jr. 1981. The patterned mires of the Red Lake peatland, Northern Minnesota: Vegetation, water chemistry, and landforms. *Journal of Ecology,* 69(2):575-599.

Glibert, P. 2011. Phytoplankton Processes in the Lakes Region and Downstream Waters in Central Florida Bay. Report to South Florida Water Management District, West Palm Beach, FL.

Glibert, P.M., C.A. Heil, D. Hollander, M. Revilla, A. Hoare, J. Alexander and S. Murasko. 2004. Evidence for dissolved organic nitrogen and phosphorus uptake during a cyanobacterial bloom in Florida Bay. *Marine Ecology Progress Series,* 280:73-83.

Hagerthey, S.E., S. Newman, K. Rutchey, E.P. Smith and J. Godin. 2008. Multiple regime shifts in a subtropical peatland: Establishing community specific thresholds to eutrophication. *Ecological Monographs,* 78(4):547-565.

Heisler, L., D.T. Towles, L.A. Brandt and R.T. Pace. 2002. Tree island vegetation and water management in the central Everglades. Pages 283-310 in: F. H. Sklar and A. van der Valk (eds), *Tree islands of the Everglades*, Kluwer Academic, Dordrecht, The Netherlands.

Hua, Q. and M. Barbetti. 2004. Review of tropospheric bomb radiocarbon data for carbon cycle modelling and age calibration purposes. *Radiocarbon*, 46:1273-1298.

Kellogg, C.M. and N.J. Dorn. In review. Consumptive effects of fish reduce wetland crayfish recruitment and drive species turnover. *Oecologia*.

Kemp W.M. and J.C. Cornwell. 2001. Role of benthic communities in the cycling and balance on nitrogen in Florida Bay. Final Report. University of Maryland Center for Environmental Science, Cambridge, MD.

Koch, M.S. 2011. Florida Bay Seagrass Studies Year 3 Progress Report. Submitted under Contract No. 4600001483, to South Florida Water Management District, West Palm Beach, FL.

Koch, E.W. and C.J. Dawes. 1991. Ecotypic differentiation in populations of *Ruppia maritima* L. germinated from seeds and cultured under algae-free laboratory conditions. *Journal of Experimental Marine Biology and Ecology*, 152:145-159.

Koschnick, T.J., M.D. Netherland and W.T. Haller 2007. Effects of three ALS-inhibitors on five emergent native plant species in Florida. *Journal of Aquatic Plant Management,* 45:47-51.

Larsen, L., N. Aumen, C. Bernhardt, V. Engel, T. Givnish, S. Hagerthey, J. Harvey, L. Leonard, P. McCormick, C. McVoy, G. Noe, M. Nungesser, K. Rutchey, F. Sklar, T. Troxler, J. Volin and D. Willard. 2011. Recent and historic drivers of landscape change in the Everglades ridge, slough, and tree island mosaic. *Critical Reviews in Environmental Science and Technology,* 41:344-381.

Light, S.S. and J.W. Dineen. 1994. Water control in the Everglades: A historical perspective. Pages 47–84 in: S.M. Davis and J.C. Ogden (eds), *Everglades: The Ecosystem and Its Restoration*, St. Lucie Press, Delray Beach, FL.

Little, M.A. and N.S. Jones. 2011. Generalized methods and solvers for noise removal from piecewise constant signals: Part I – Background theory. *Proceedings of the Royal Society*, online ahead of print, doi: 10.1098/rspa.2010.0671.

Little, M.A., B.C. Steel, F. Bai, Y. Sowa, T. Bilyard, D.M. Mueller, R.M. Berry and N.S. Jones. 2011. Steps and bumps: Precision extraction of discrete states of molecular machines. *Biophysical Journal,* 101(2):477-485.

Lorenz, J.J. 1999. The response of fishes to physicochemical changes in the mangroves of northeast Florida Bay. *Estuaries,* 22(2B): 500-517.

Lorenz, J. 2011. Status Report for Monitoring Hydrology, Aquatic Vegetation, and Prey-Base Fauna in the Southern Everglades Mangrove Transition Zone. Report under Contract 4600001851 to South Florida Water Management District, West Palm Beach, FL.

Lorenz, J.J., C.C. McIvor, G.V.N. Powell and P.C. Frederick. 1997. A drop net and removable walkway used to quantitatively sample fishes over wetland surfaces in the dwarf mangroves of the southern Everglades. *Wetlands*, 17(3):346–359.

Lorenz, J.J. and J.E. Serafy. 2006. Subtropical wetland fish assemblages and changing salinity regimes: Implications for Everglades restoration. *Hydrobiologia*, 569:401–422.

Lynch, J.A., J.S. Clark, N.H. Bigelow, M.E. Edwards and B.P. Finney. 2002. Geographic and temporal variations in fire history in boreal ecosystems of Alaska. *Journal of Geophysical Research-Atmospheres,* 108(D1).

Madden, C.J. and J. Day. 1992. An instrument system for high speed mapping of chlorophyll and physico-chemical parameters. *Estuaries,* 15(3):421-427.

Mateos-Naranjo, E., S. Redondo-Gomez, L. Cox, J. Cornejo and M.E. Figueroa 2009. Effectiveness of glyphosate and imazamox on the control of the invasive cordgrass *Spartina densiflora*. *Ecotoxicology and Environmental Safety*, 72(6):1694-1700.

McVoy, C.W., W.P. Said, J. Obeysekera, J. VanArman and T. Dreschel. 2011. *Landscapes and Hydrology of the Predrainage Everglades*. University Press of Florida, Gainesville, FL.

Newman, S, J. Schuette, J.B. Grace, K. Rutchey, T. Fontaine, K.R. Reddy and M. Pietrucha. 1998. Factors influencing cattail abundance in the northern Everglades. *Aquatic Botany,* 60(3):265-280.

Nungesser, M.K. In press. Reading the landscape: Temporal and spatial changes in a patterned peatland. *Wetlands Ecology and Management*.

Ogden, J.C., S.M. Davis and L. Brandt. 2003. Science Strategy for a Regional Ecosystem Monitoring and Assessment Program: The Florida Everglades example. Pages 135–163 in: D.E. Busch and J.C. Trexler (eds), *Monitoring Ecosytems*, Island Press, Washington, D.C.

Owens M.S. and J.C. Cornwell. 2011, Measurement of Sediment-Water Nutrient Fluxes in Everglades National Park Saline Lakes. Final Report. Chesapeake Biogeochemical Associates, Sharptown, MD.

Parkos, J.J., C.R. Ruetz and J.C. Trexler. 2011. Disturbance regime and limits on benefits of refuge use for fishes in a fluctuating hydroscape. *Oikos,* doi:10.1111/j.1600-0706.2011.19178.x

Patterson, K. and R. Finck. 1999. Tree Islands of the WCA-3 Aerial Photointerpretation and Trend Analysis Project Summary Report. Report by Geonex Corporation, St. Petersburg, FL, submitted to South Florida Water Management District, West Palm Beach, FL.

Reimer, P.J., M.G.L. Baillie, E. Bard, A. Bayliss, J.W. Beck, C.J.H. Bertrand, P.G. Blackwell, C.E. Buck, G.S. Burr, K.B. Cutler, P.E. Damon, R.L. Edwards, R.G. Fairbanks, M. Friedrich, T.P. Guilderson, A.G. Hogg, K.A. Hughen, B. Kromer, G. McCormac, S. Manning, C.B. Ramsey, R.W. Reimer, S. Remmele, J.R. Southon, M. Stuiver, S. Talamo, F.W. Taylor, J. van der Plicht and C.E. Weyhenmeyer. 2004. IntCal04 terrestrial radiocarbon age calibration, 0–26 cal kyr BP. *Radiocarbon,* 46(3):1029-1058.

Rietkerk, M., and J. van der Koppel. 2008. Regular pattern formation in real ecosystems. *Trends in Ecology and Evolution,* 23(3):169-176.

Rietkerk, M., S.C. Dekker, M.J. Wassen, A.W.M. Verkroost and M.F.P. Bierkens. 2004. A putative mechanism for bog patterning. *The American Naturalist,* 163(5):699-708.

Ross, M. and J. Sah. 2011. Forest resource islands in a sub-tropical marsh: Soil-site relationships in Everglades hardwood hammocks. *Ecosystems*, 14: 632-645.

Rutchey, K., T.N. Schall, R.F.Doren, A. Atkinson, M.S. Ross, D.T. Jones, M. Madden, L. Vilchek, K.A. Bradley and J.R. Snyder. 2006. Vegetation Classification for South Florida Natural Areas. Open-File Report 1240, United States Geological Survey, Washington, D.C.

Rutchey, K., T. Schall and F. Sklar. 2008. Development of vegetation maps for assessing Everglades restoration progress. *Wetlands*, 28(3):806-816.

Saunders, C.J., M. Gao, J.A. Lynch, R. Jaffe and D.L. Childers. 2006. Using soil profiles of seeds and molecular markers as proxies for sawgrass and wet prairie slough vegetation in Shark Slough, Everglades National Park. *Hydrobiologia,* 569:475-492.

Saunders, C.J.., R. Jaffe, M. Gao, W. Anderson, J.A. Lynch and D. Childers. 2008. Decadal to millennial dynamics of ridge-and-slough wetlands in Shark Slough, Everglades National Park: Integrating paleoecological data and simulation modeling. National Park Service, Miami, FL.

Saunders C.J., C. Moses, J.A. Lynch, C. Craft, D.L. Childers and A.L. Bubp. In prep. Millennial-scale dynamics of Everglades ridge-and-slough wetlands in historic Shark River Slough and influences of long-term climate drivers.

SERA. 2010. Imazamox Human Health and Ecological Risk Assessment. Final report submitted by Syracuse Environmental Research Associates, Inc. under contract AG-3187-C-06-0010 to the United States Department of Agriculture Forest Service, Washington, D.C. Accessed at http://www.fs.fed.us/foresthealth/pesticide/pdfs/052-24-02a_Imazamox.pdf.

SFWMD. 2003. Chapter 6: Ecology of the Everglades Protection Area. In: *2003 Everglades Consolidated Report*. South Florida Water Management District, West Palm Beach, FL.

Shaner, D.L., P.C. Anderson and M.A. Stidham. 1984. Imidazolinones potent inhibitors of acetohydroxyacid synthase. *Plant Physiololgy,* 76:545-546.

Sklar, F.H. and A. van der Valk. 2002. *Tree Islands of the Everglades*. Springer, New York, NY.

Sterner, R.W. and J.J. Elser. 2002. *Ecological Stoichiometry: The Biology of the Elements from Molecules to the Biospehere*. Princeton University Press, Princeton, New Jersey.

Talma, A.S. and J.C. Vogel. 1993. A simplified approach to calibrating 14C dates. *Radiocarbon,* 35(2):317-322.

Trexler, J.C. and C. Goss. 2009. Aquatic fauna as indicators for Everglades restoration: Applying dynamic targets in assessments. *Ecological Indicators,* 9S:S2-S16.

Trexler, J.C., W.F. Loftus and S. Perry. 2005. Disturbance frequency and community structure in a twenty-five year intervention study. *Oecologia,* 145:140-152.

Troxler Gann, T. and D.L. Childers. 2006. Relationships between hydrology and soils describe vegetation patterns in seasonally flooded tree islands of the southern Everglades, Florida. *Plant and Soil,* 279(1-2):271-286.

Troxler,T. 2010. Ecological monitoring of Southern Everglades, the mangrove transition zone and the "White Zone" interactions with Florida Bay, Final Report for Year 1 of Contract No. 4600002149, South Florida Water Management District, West Palm Beach FL.

Turetsky, M.R., S.W. Manning and R.K. Wieder. 2004. Dating recent peat deposits. *Wetlands,* 24(2):324-356.

Underwood, A.J. and M.G. Chapman. 2003. Power, precaution, type II error and sampling design in assessment of environmental impacts. *Journal of Experimental Marine Biology and Ecology,* 296:49-70.

Vanni, M.J., A.S. Flecker, J.M. Hood and J.L. Headworth. 2002. Stoichiometry of nutrient recycling by vertebrates in a tropical stream: linking species identity and ecosystem processes. *Ecology Letters,* 5:285-293.

Watts, D.L., M. J. Cohen, J.B. Heffernan and T.Z. Osborne. 2010. Hydrologic modification and the loss of self-organized patterning in the ridge slough mosaic of the Everglades. *Ecosystems,* 13(6):813-827.

Wersal, R.M. and J.D. Madsen 2007. Comparison of imazapyr and imazamox for control of parrotfeather (*Myriophyllum aquaticum* (Vell.) Verdc.). *Journal of Aquatic Plant Management,* 45:132-136.

Wetzel, P.R., T. Pinion, D.T. Towles and L. Heisler. 2008. Landscape analysis of tree island head vegetation in Water Conservation Area 3, Florida Everglades. *Wetlands,* 28(2):276-289.

Wetzel, P.R., A.G. van der Valk, S. Newman, D.E. Gawlik, T.T. Gann, C.A. Coronado-Moliana, D.L. Childers and F.H. Sklar. 2005. Maintaining tree islands in the Florida Everglades: nutrient redistribution is the key. *Frontiers in Ecology and the Environment*, 3(7):370-376.

White, W. N. 1932. A method of estimating ground-water supplies based on discharge by plants and evaporations from soils: results of investigations in Escalante Valley, Utah. United States Geological Survey, Washington, D.C. Water Supplement Paper 658A.

Willard, D.A., C.E. Bernhardt, C.W. Holmes, B. Landacre and M. Marot. 2006. Response of everglades tree islands to environmental change. *Ecological Monographs,* 76(4):565-583.

Wu, Y., K. Rutchey, W. Guan, L. Vilchek and F.H. Sklar. 2002. Spatial simulations of tree islands for Everglades restoration. Pages 469–498 in: F. Sklar and A. van der Valk (eds), *Tree Islands of the Everglades*, Kluwer Academic Publishers, Boston, MA.