Chapter 6: Ecology of the Everglades Protection Area

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SUMMARY

The studies and findings discussed in this chapter of the *2010 South Florida Environmental Report* (SFER) – *Volume I* are presented within four main fields: (1) wildlife ecology, (2) plant ecology, (3) ecosystem ecology, and (4) landscape processes. Programs of study were based on the short-term operational needs and long-term restoration goals of the South Florida Water Management District (SFWMD or District), including large-scale and regional hydrologic needs in relation to regulation schedules, permitting, the Everglades Forever Act [Section 373.4592, Florida Statutes (F.S.)] mandates, and the Comprehensive Everglades Restoration Plan (CERP). **Table 6-1** summarizes elements of the major Everglades research findings during Water Year 2009 (WY2009) (May 1, 2008–April 30, 2009) and highlights these findings in relation to the WY2009 hydrologic pattern and statutory mandates that drive the research.

The hydrologic pattern is immediate and foremost an influence on the restoration of Everglades wading bird populations, but is also critical to long-term ecological trends. In WY2009, the amount of rainfall received was below average, while the pattern of rainfall was close to normal. Water levels rose during the wet season as they have for the last 15 years and October peaks produced water depths ranging from a low of 1.5 feet (ft) in Northeast Shark River Slough to a high of 3.0 ft in the southern region of Water Conservation Area 3A (WCA-3A). Rainfall almost ceased between November and May, but high water levels buffered the system from complete drydown in the Water Conservation Areas (WCAs). Dry season recession rates were optimum for wading bird foraging. There were no reversals anywhere in the system until May 2009 (WY2010), which became the second wettest May in the history of the SFWMD.

WILDLIFE

As a result of the WY2009 recession rates and the previous two years of drought, the 2009 wading bird nesting season was outstanding. The estimated number of wading bird nests in South Florida in 2009 was approximately 80,000. This is the largest nesting effort recorded in the region since the 1940s and represents (1) a 335 percent increase relative to last year's breeding season, (2) a 90 percent increase over the average of the last 10 seasons, and (3) surpasses the previous record year, 2002, by approximately 11,000 nests. The white ibis (*Eudocimus albus*) and the federally endangered wood stork (*Mycteria americana*) (which has generally exhibited very low nesting effort over the past decade), both produced numbers of nests that have not been observed

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since the pre-drainage period. Only the roseate spoonbill (*Ajaja ajaja*) nest numbers in Florida Bay were generally low.

At the Loxahatchee Impoundment Landscape Assessment facility (LILA), where the hydrologic patterns are controlled to study plant and animal processes, the slough communities were found to differ in flowing macrocosms versus non-flowing macrocosms. Spatial and temporal variations in Everglades macroinvertebrates are difficult to understand – yet they may play crucial roles in the development of prey for higher trophic level animals. This year, a summary of the 2000–2008 data indicated that community structure at the species level differs across WCAs and that abundances were lowest in Water Conservation Area 2A (WCA-2A). This pattern is hypothesized to be linked to hydrologic stability.

PLANTS

An investigation into periphyton cyanotoxins has discovered that there are six toxins present in the Everglades study area, but the measured content (per gram) may be relatively low in comparison to other aquatic systems. However, exposure to cyanotoxin concentrations may be significant given the dense periphyton biomass in the Everglades. The implications of these toxins in the Everglades are unknown, but in other systems they can affect larval fish survival and adult fish physiology.

Ecophysiology of tree species on islands with differing hydrologic regimes indicate that stem-water potentials, water-use patterns, sap flow, and leaf-gas exchange can be used to assess plant responses to hydrologic conditions and stress. Trees on highly inundated islands had less sap flow and lower transpiration indicating a possible stress response of entering semi-dormancy.

ECOSYSTEM

The two large-scale experimental manipulations of the cattail (*Typha* spp.) impacted area of WCA-2A that were discussed in the 2008 SFER – Volume I, Chapter 6 [The Accelerated Recovery on Impacted Areas Project (FIRE) and Cattail Habitat Improvement Project (CHIP)], are near completion.

CHIP reports on microbial, vegetation, fish, and wading bird responses to maintaining open habitats within impacted cattail habitat. After two years, these open habitats maintain beneficial microbial communities, support submerged aquatic vegetation (SAV), return to more natural diel oxygen cycles, and are areas of intense wading bird foraging.

FIRE reports on the need to consider water levels, fuel loads, and fire intensity to maximize phosphorus loss from highly enriched habitats. As fire temperatures increase, the ash composition of phosphorus increases and can significantly increase surface water total phosphorus (TP) concentrations and pH over the short-term.

Based upon reflux enclosure experiments in WCA-2A, a simple model of soil and water phosphorus exchanges was used to estimate the time required to reduce the phosphorus-enriched soil to background level concentration (500 mg/kg) proposed for the Everglades by the U. S. Environmental Protection Agency (2007). Six modeled scenarios of three surface water inflow concentrations (10, 20, or 50 μg/L) and two non-labile soil phosphorus rate constants (0.022/yr and 0.01/yr) suggest that 67 to 100-plus years would be required to achieve soil phosphorus concentrations of 500 mg/kg.

LANDSCAPE

This section presents four projects.

The development of the new South Florida Water Depth Assessment Tool builds upon hundreds of real-time water level gauges throughout the region that are managed by several government agencies (e.g., SFWMD, Everglades National Park, U.S. Geological Survey, and Miami-Dade Department of Environmental Resource Management) to provide real-time spatial interpolation and daily animation of water depth and ecological indices (e.g., muck fire potential and recession-inundation indices). The muck fire and recession indices are utilized during the dry season to identify areas that have the potential for serious muck fire ignition and to gain insight by spatially animating daily water depth recession rates to manage levels for better wading bird habitat conditions. The inundation index is utilized during the wet season to ascertain water depth increases in relation to floral and faunal hydrologic requirements.

Peat loss in the Everglades is due to the oxidation of the peat through drainage or drought. Drainage and drought conditions dry out the surface of the peat, leading to microbial oxidation of the organic carbon or fire, which causes the carbon in the peat to be converted to carbon dioxide (CO_2). Estimates of peat loss and carbon emissions over the last 100 years were calculated by comparing historic topographic renderings with current topography used by SFWMD hydrologic models. Subsidence was found to range from a low of 0.01 ft in Everglades National Park (ENP) to a high of 5.5 ft in the Everglades Agricultural Area (EAA). The total CO_2 associated with peat loss for the entire Everglades was estimated at about 3.5 billion metric tons (mt). The total CO_2 emissions numbers were corrected for the datum differences between the two surfaces, which yielded an estimate of 3.4 billion metric tons. These values were calculated with the assumption that all the carbon lost from the Everglades eventually ended up as carbon dioxide.

A carbon mapping project based upon 651 soil samples throughout the WCAs found differences in concentration that mirror the WCAs differing hydrologic regimes and inputs. Notable patterns included low total carbon (TC) values in the northwest region of WCA-3A, due to peat fires, high TC in WCA-1, and a wedge of lower TC in the impacted zone of WCA-2A due to nutrient-stimulated carbon cycling.

Finally, the District continues to lead in remote sensing and photo interpretation. New digital aerial imagery encompassing nearly 2,000 square miles was acquired during April 2009 for the ENP, Southeastern Miami-Dade Coastal Wetlands, and Biscayne Bay Coastal Wetlands for CERP's Restoration Coordination and Verification (RECOVER) vegetation mapping project. This new digital imagery is unique in that it was collected by a new generation frame-based aerial digital camera from Microsoft® called UltraCamX. For this project, the ground pixel resolution is approximately one square foot for the entire project. The digital color intensity range of this data collection is far better than anything collected in the past. In addition, the digital images were captured with enough overlap between adjacent frames so that stereo images can be produced – allowing for the exploration of net assessment tools, such as tree heights and canopy densities.

Table 6-1. Summary of Water Year 2009 (WY2009) (May 1, 2008–April 30, 2009) Everglades research findings in relation to the following operational mandates: Regulation and Operational Schedules (ROS); Comprehensive Everglades Restoration Plan (CERP); Everglades Forever Act (EFA); Long-Term Plan for Achieving Water Quality Goals in the Everglades Protection Area (LTP); Minimum Flows and Levels (MFLs); Florida Everglades Improvement and Management (FEIM); U.S. Environmental Protection Agency (USEPA).

Projects	Findings	Mandates
Hydrologic Pattern	Although rainfall in WY2009 was 10 percent to 30 percent less than the historical average, mean stage differed from the historical average by a much smaller degree (-8 percent to +4 percent). Recession rates near 0.10 foot/week during the dry season were maintained for extended periods of time and were extremely conducive for peat conservation and wading bird foraging. Abnormally high rainfall in May 2009 brought these favorable recession rates and the fledging of wading birds to an abrupt halt.	ROS, MFL
Wildlife		
Wading Bird Nesting Patterns	Approximately 80,000 nests were initiated in South Florida during WY2009, representing a 335 percent increase from WY2008 and the highest recorded number of nests since pre-drainage. High reproductive effort and success can be attributed to a combination of drought and flooded conditions during preceding years and highly favorable hydrologic conditions during nesting.	ROS, CERP, MFL, FIEM
Crayfish Population Dynamics	A comprehensive survey of crayfish populations and other components of the aquatic community in LILA will provide the baseline data for the drought manipulation planned for spring 2010 integrated with the removal of predatory fish. Interesting additional analyses not directly related to the primary hypothesis suggest a shrimp and mollusk preference for flowing and a predatory fish preference for non-flowing conditions among wading birds.	ROS, CERP, MFL, FIEM
Variation in Macroinvertebrate Communities	The macroinvertebrate community structure (i.e., species abundance) differs among regions (WCAs-1–3) and total abundance was lowest in WCA-2A. Invertebrate communities also differed between seasons (i.e., wet and dry). Beetle (*Coleoptera*) and segmented worm *Annelida* abundances were greater during the wet season, while midge (*Chironomidae*) abundance was greater during the dry season. Preferences for wet and dry seasons may be due to the evolution of growth patterns linked to hydrologic cues.	EFA
Plants		
Periphyton Cyanotoxins	Periphyton throughout the Everglades were found to contain cyanotoxins. The most common were a hepatotoxin and two neurotoxins. While observed contents were not great, the potential for significant concentrations exist within the algal mats.	EFA, CERP, LTP
Tree Island Ecophysiology	Physiological parameters indicate that the tree island plant community is sensitive to small seasonal changes in hydrology and that these parameters can be used to assess in-situ instantaneous stress responses to hydrologic conditions. Most of the flood tolerant species showed a significant response to flooding and drydown conditions. This further reinforces the robustness of using physiological parameters to establish hydrologic tolerances.	ROS, CERP, EFA

Table 6-1. Continued.

Projects	Findings	Mandates
Ecosystem		
Cattail Habitat Improvement	Submerged aquatic vegetation, principally the macroalga, *Chara*, became a dominant component of vegetative biomass in open plots. Open plots continued to show greater microbial activity, higher periphyton productivity, greater decomposition rates, and reduced detritivore biomass. In addition, open plots supported higher wading bird foraging than the surrounding cattail (*Typha* spp.) habitat.	LTP, ROS
Accelerated Recovery of Impacted Areas (FIRE)	Average cattail mass loss was twice as high during the first fire than the second fire. Mass, carbon, and nitrogen loss was not different between the highly enriched plot and moderately enriched plot. However, phosphorus loss was highest at the highly enriched plot. Loss was a function of pre-fire mass, nutrient content, water depth, and fire intensity. Fire temperature determined ash chemistry and ash deposition had a direct impact on surface water quality.	LTP
Effect of Phosphorus Loading on Water and Soil Phosphorus Concentration: A Phosphorus Forecast Model	Based upon reflux enclosure experiments in WCA-2A, a simple model of soil and water phosphorus (P) exchanges was used to estimate the time required to reduce the P-enriched soil to background levels. Six modeled scenarios of three surface water inflow concentrations (10, 20, or 50 µg/L) and two non-labile soil P rate constants (0.022/yr and 0.01/yr) suggest that 67 to 100 years would be the minimum range required to achieve soil P concentrations of 500 mg/kg.	LTP
Landscape		
South Florida Water Depth Assessment Tool	Over 100 stage gauges in the Greater Everglades were integrated into a spatial interpolation animation program designed to estimate fine-scale water depth changes on a daily basis. As a result, water resource management can now evaluate real-time spatial trends and environmental risks associated with muck fire potentials and wading bird nesting success.	ROS, MFLs
Peat Loss in the Everglades	Using the late 1800s as a baseline, the estimated average subsidence ranged from a low of 0.01 m (0.1 ft) averaged across all ENP soils (peats and marls) to a high of 1.7 m (5.5 ft) in the EAA. Subsidence in the EAA was greater than previously reported. Total Everglades carbon dioxide emissions associated with peat loss between 1880 and 2005 were estimated to be 3.4 billion metric tons.	ROS, CERP, LTP
Spatial Soil Carbon Distribution	Mapping of total carbon (TC) from a network of 651 soil sampling locations throughout the WCAs was based upon ordinary kriging and Inverse Distance Weighted interpolation for each WCA. Notable patterns included low TC values in the northwest region of WCA-3A, due to peat fires, high TC in WCA-1, and a wedge of lower TC in the impacted zone of WCA-2A due to nutrient-stimulated carbon cycling.	CERP, EFA, LTP
CERP RECOVER Vegetation Mapping Update	New digital UltraCamX aerial photography with a ground resolution of approximately one square foot was acquired April 2009 for the CERP RECOVER vegetation mapping project. These digital images are analyzed via a virtual three-dimensional rendering of overlapping frames. The vegetation map for Northeast Shark River Slough is scheduled to be completed in 2010.	CERP, EFA

HYDROLOGIC PATTERNS FOR WATER YEAR 2009

Fred Sklar

The amount of rain in the Everglades Protection Area (EPA) for Water Year 2009 (WY2009) (May 1, 2008–April 30, 2009) was substantially less than the last water year, a year that brought the Everglades Protection Area (EPA) out of the 2006–2007 droughts. WY2009 rainfall amounts varied from 28.3 inches less than last water year in Everglades National Park (ENP or Park), to 8.1 inches less than in Water Conservation Area (WCA) 1. Most of the difference between the two years can be found during the dry seasons (November through April); in WY2008, rainfall was 5–10 inches above average, but WY2009 had 4–8 inches below-average rainfall. The lack of dry season rainfall in WY2009 was accompanied by a lack of reversals, giving WY2009 another significant distinction from WY2008, especially regarding wading bird foraging behavior and nesting success. The rainfall and associated stage readings for WY2009 are shown in **Table 6-2**. Water Conservation Areas (WCAs) 1 and 2 had a 9 percent decrease from historic rainfall amounts and a 15 percent decrease over WY2008. Similarly, WCA-3 had a 14 percent decrease compared to historic rainfall amounts, but only an 8 percent decrease from WY2008. The largest deviations from the historical average and WY2008 were found in the ENP, where a 31 and 23 percent decreases were recorded, respectively. It is interesting to note the sequence of rain the last three water years; first a severe two-year drought, followed by above-average rainfall (with many reversals during the dry seasons), then, this water year, below-average rainfall with no local reversals. This kind of hydrology may account for the high wading bird nesting success in WY2009 (see the *Wildlife Ecology* section of this chapter). The hydrology of WY2009 for the entire South Florida Water Management District (SFWMD or District) region is presented in Chapter 2 of this volume.

Table 6-2. Average, minimum, and maximum stage [feet National Geodetic Vertical Datum (ft NGVD)] and total annual rainfall (inches) for Water Year 2009 (WY2009) (May 1, 2008–April 30, 2009) in comparison to historic stage and rainfall.[1]

Area	WY2009 Rainfall	Historic Rainfall	WY2009 Stage Mean (min; max)	Historic Stage Mean (min; max)	Elevation
WCA-1	47.4	51.96	16.26 (14.45; 17.38)	15.61 (10.0; 18.16)	15.1
WCA-2	47.4	51.96	12.26 (9.77; 14.46)	12.53 (9.33; 15.64)	11.2
WCA-3	44.3	51.37	10.28 (8.57; 11.99)	9.55 (4.78; 12.79)	8.2
ENP	42.6	55.22	6.29 (4.57; 7.29)	5.98 (2.01; 8.08)	5.1

[1] See Chapter 2 of this volume for a more detailed description of rain, stage, inflows, outflows, and historic databases.

In WY2009, the monthly rainfall seasonality deviated slightly from what is considered typical for this semitropical region. June and July 2008 each had approximately 8 inches of precipitation. Maximum rainfall of 10–12 inches occurred in August 2008 in the EPA. September brought 6 inches while October had 6 inches in WCA-1 and only 4 inches in the southern regions. The start of the dry season in November was abrupt. Monthly rainfall stayed below 0.5 inch for the next four months, but was interrupted shortly in March 2009 with approximately 2 inches of rain and a significant potential for a hydrologic reversal. April, however, had only a 0.5 inch of rain.

What followed in May (and considered part of WY2010) is mentioned here because of its tremendous deviation from the norm and its significance to the fledging of wading birds at the end of the 2008/2009 nesting season. **Table 6-3** summarizes the statistics for rainfall just in May for the last 46–77 years. Notice that May is typically a transition month (from the dry to wet season) and has a very high mean range. Despite this deviation, or maybe because of it, May 2009 had one of the highest precipitation rates of any May on record for the WCAs and tied the overall record for the entire District; these rains came at the critical period when 3,000 to 4,000 wood stork (*Mycteria americana*) chicks were about to fledge.

Table 6-3. Average, minimum, and maximum total monthly rainfall (inches) for the Everglades Protection Area in May 2009 as compared with historic May rainfall.

Area	May 2009 Rainfall	Average May Rainfall	Historic May Rainfall (min; max)	Historic Time Period (n)
WCA-1&2	9.15"	4.60"	(0.38"; 14.60")	1957–2009
WCA-3	9.00"	4.42"	(0.94"; 10.90")	1963–2009
Entire SFWMD	9.04"	4.19"	(1.26"; 9.04")	1932–2009

The hydropattern figures in this section highlight the average stage changes in each of the WCAs for the last two water 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 District 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 belowground 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 categories 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), or when the average depth change for the week is positive rather than negative.

- A yellow label ▬▬➤ indicates fair conditions due to a slow recession rate of 0.04 foot for a week or a rapid recession between 0.17 foot and 0.6 foot per week.

- A green/good label ▬▬➤ is assigned when water depth decreased between 0.05 foot and 0.16 foot per week.

Although these labels are not indicative of an appropriate depth for foraging, they have been useful during high water conditions to highlight recession rates that can lead to good foraging depths toward the end of the dry season (i.e., April and May).

WATER CONSERVATION AREA 1

WY2009 for WCA-1 started as an average year with a little over a 0.5 foot of water on the marsh (**Figure 6-1**) after an above-average dry season with numerous reversals that made foraging in WY2008 difficult. Water depths rose to 2.5 feet (ft) over the next four months, but remained above average for only a month and never posed a flooding problem for tree islands.

Except for a short reversal in March, the recession rates from November through May 2008 were exceptionally smooth and steady, fostering widespread foraging and earlier, more appropriate nesting by wood storks. Optimum foraging depths for wading birds were reached in June 2008 because of good recession rates in April and May. Unfortunately, by July 2008, optimum depths were once again exceeded. For the WY2009 nesting season, optimum foraging depths started in March 2009 and would have lasted a full three months if it were not for the high rainfall during the second half of May. For the fourth year in a row, WCA-1 had the longest duration of good nesting and foraging periods of any region in the EPA.
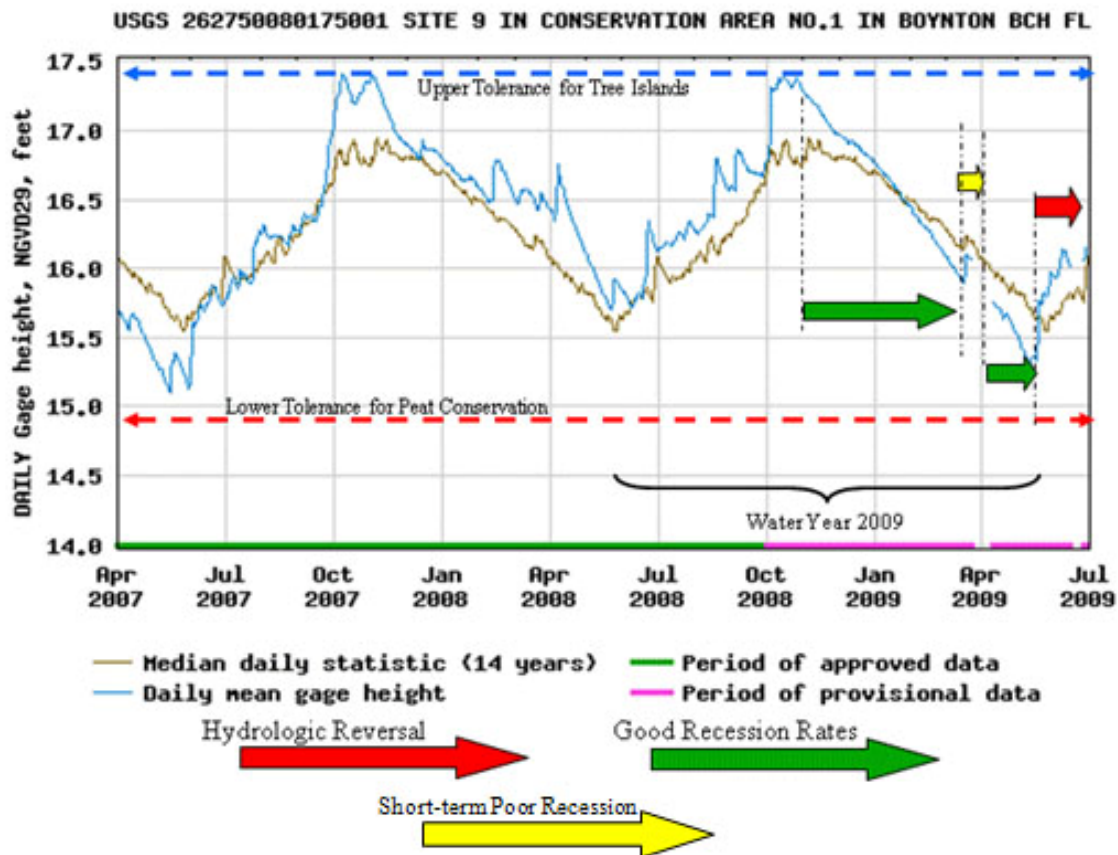

Figure 6-1. Hydrology in Water Conservation Area 1 (WCA-1) in relation to the 14-year median stage, as well as indices for tree island flooding, peat conservation, and wading bird foraging.

WATER CONSERVATION AREAS 2A AND 2B

For the last four years in WCA-2A the stage levels during the wet season have exceeded the upper flood tolerance for tree islands. However, each year it exceed this level for only one or two months, which is not considered enough time to cause any tree island damage (Wu et al., 2002). The few islands that remain in this region are not likely to be impacted by high water levels due to their northwest location and their relatively high elevations. However, future efforts to restore WCA-2A tree islands will require a closer examination (i.e., frequency analysis) of this kind of exceedance. In WCA-2A, the WY2006, WY2007, and now WY2009 dry seasons had very good recession rates, and all three times the region completely dried out. In WY2006, WCA-2A exhibited excellent foraging conditions and many flocks of wading birds were observed. However, WY2009 was more like WY2007, when the hydroperiod was very short and stage was below average for most of the year. As a result, reports of large or many flocks were greatly reduced **(Figure 6-2)**.

WCA-2B has always been used by wading birds during droughts because it tends to stay deeper for longer periods than the rest of the EPA. This was particularly true in WY2006 when dry season water levels went below ground in WCA-2A and the wading birds moved to WCA-2B. In WY2007, the drought was so severe that even WCA-2B became too dry to support any foraging from May through July. The opposite occurred in WY2008 when water levels increased instead of receding during peak months (March, April, and May) and never dropped low enough to support foraging. However, in WY2009, large flocks of wading birds were observed foraging in WCA-2B in April–May 2009 to compensate for the heavy May rainfall and because recession rates were excellent and depths were near optimum **(Figure 6-3)**.
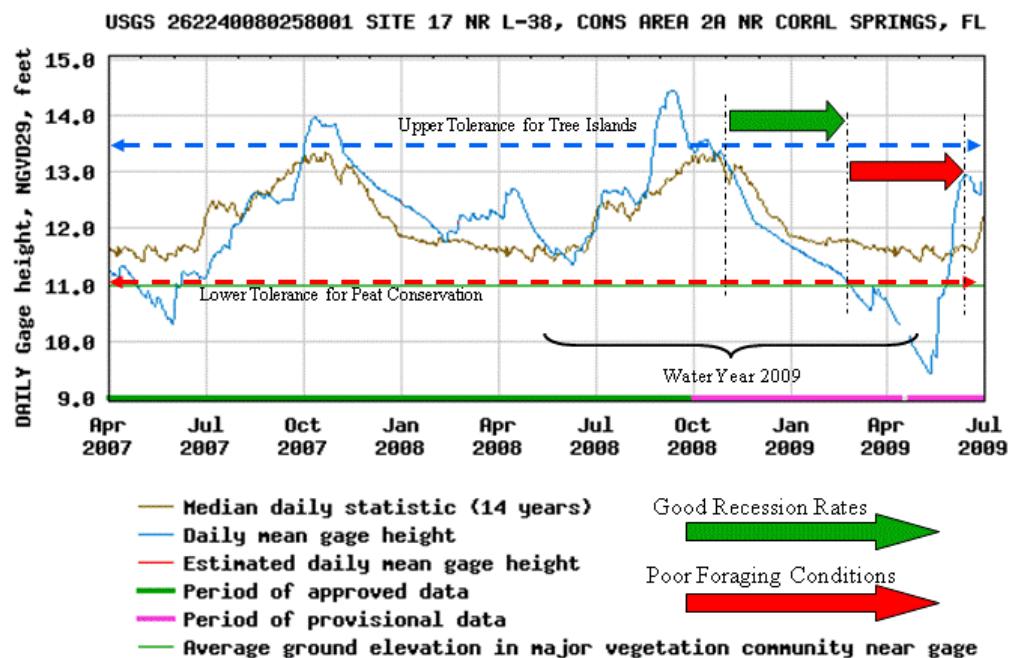

Figure 6-2. Hydrology in Water Conservation Area 2A (WCA-2A) in relation to the recent 14-year average, with indices for tree islands, peat conservation, and wading bird foraging.
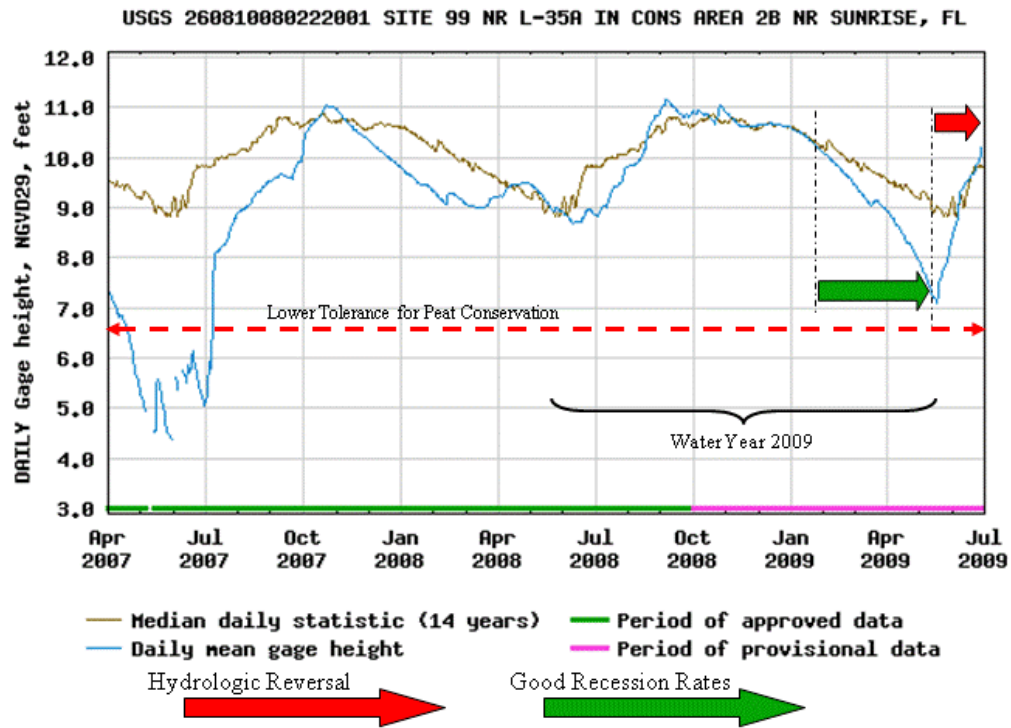
Figure 6-3. Hydrology in Water Conservation Area 2B in relation to the recent
15-year average with indices for tree islands, peat conservation,
and wading bird foraging.

WATER CONSERVATION AREA 3A

In the northeastern region of WCA-3A (gage 63) the WY2007 drought extended well into WY2008 **(Figure 6-4)**. This set-up for WY2009 was not one conducive for foraging wading birds and was made worse by the extensive amount of WY2008 dry season reversals and a very short wet period. However, WY2009 began relatively well in June 2008 with steady rehydration rates and a long wet season with a hydroperiod capable of rejuvenating the prey base for the region's large wading bird rookery (Alley North) where annual nesting during the past decade has frequently exceeded 20,000 nests. Water depths rose to 3.5 ft over the next five months, and remained above the upper tolerance for tree islands in this area for about 120 days, which is the amount of time considered to be stressful and potentially lethal to island vegetation if repeated year after year (Wu et al., 2002). There are few islands left in this area and those that are left are dominated by willow (*Salix americana*), one of the most flood-tolerant species found in the Everglades (Armentano et al., 2002). Therefore, despite the long inundation, there was no immediate danger to islands in this region.
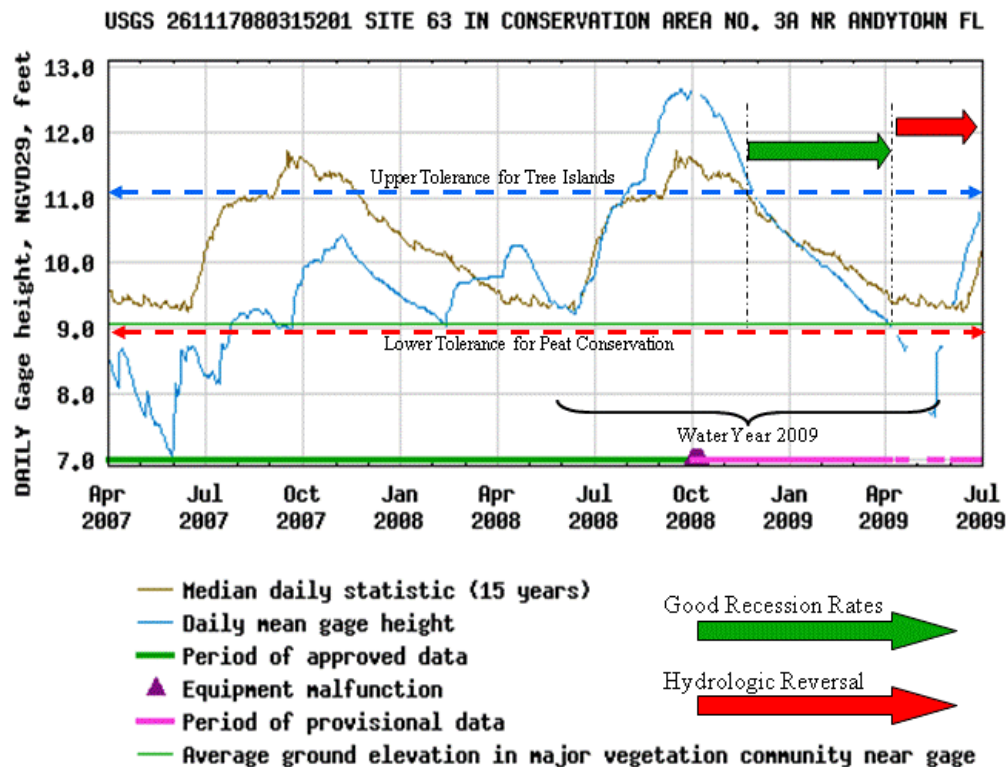

Figure 6-4. Hydrology in northeast Water Conservation Area 3A (WCA-3A) (gage 63) in relation to the recent 15-year average with indices for tree islands, peat conservation, and wading bird foraging.

The hydrologic pattern in central WCA-3A (gage 64) in WY2008 did not suffer the drought as much as the northeast WCA-3A **(Figure 6-5)**. Although there was not a significant MFL violation, there was a greatly reduced wet-season stage. Water depths did not go above 1 foot until October and never went over 2 ft the entire water year. What should have been a great wading bird foraging environment starting in March was instead disrupted by increasing water levels rather than decreasing water levels for almost the entire nesting (dry) season. This was followed by shallow depths and a short-duration wet season and was probably sufficient to cause widespread depletion of wading bird prey species. In 2009, the lack of foraging and the longer hydroperiod may well translate into a banner prey-base for next year's nesting season (WY2010).
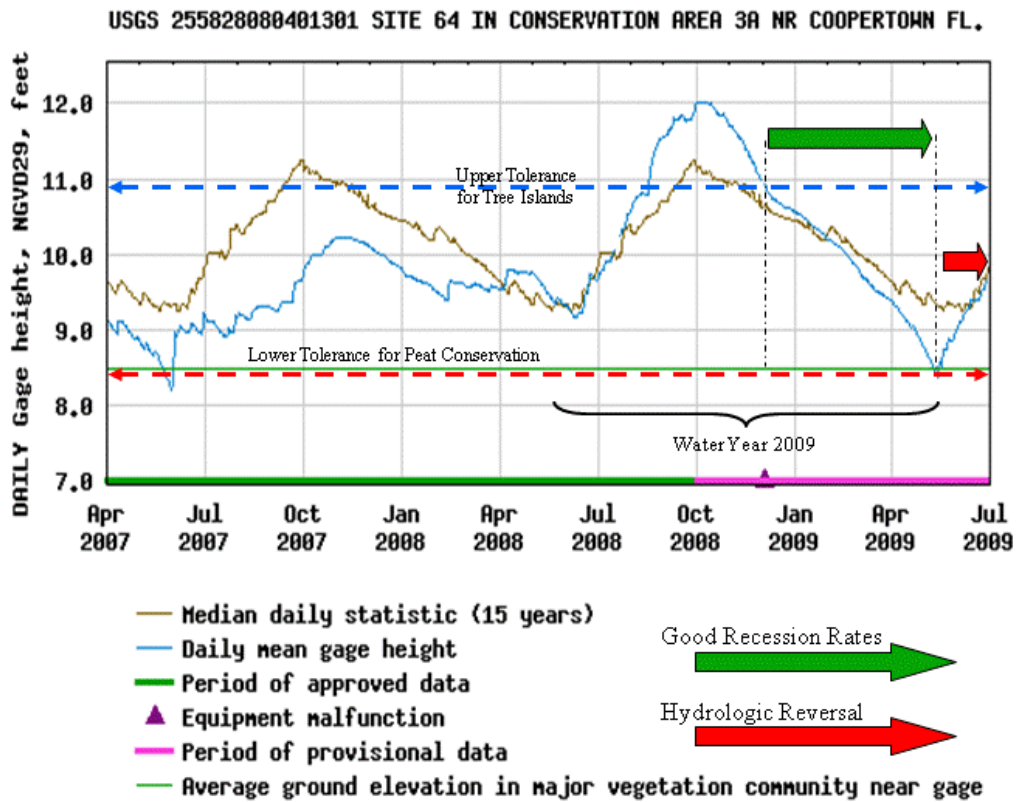
USGS 255828080401301 SITE 64 IN CONSERVATION AREA 3A NR COOPERTOWN FL.

Figure 6-5. Hydrology in central WCA-3A (gage 64) in relation to the recent 15-year average with indices for tree islands, peat conservation, and wading bird foraging.

WATER CONSERVATION AREA 3B

In 2007, the recession rates in WCA-3B were poor at best. In 2008, rising water levels during the wet season followed the average almost exactly and the upper tolerance for tree islands was barely reached. Then, in November 2008, water levels fell at an almost steady perfect 0.10 foot per week until April 2009 when water levels dropped abruptly, followed by a large-scale May reversal. The set-up for optimum March and April 2009 foraging by wading birds could not have been better. Unfortunately, at this time it is not clear if WCA-3B was used extensively because survey flight data for this region were still being compiled when this report was finalized. For WY2006, WY2007, and WY2008, this region's hydrology did not support wading bird nesting or foraging. For WY2009, this region's hydrology should have supported a high amount of foraging, and, in April 2009, wading birds were observed in WCA-3B, but extensive use by large flocks was not documented.
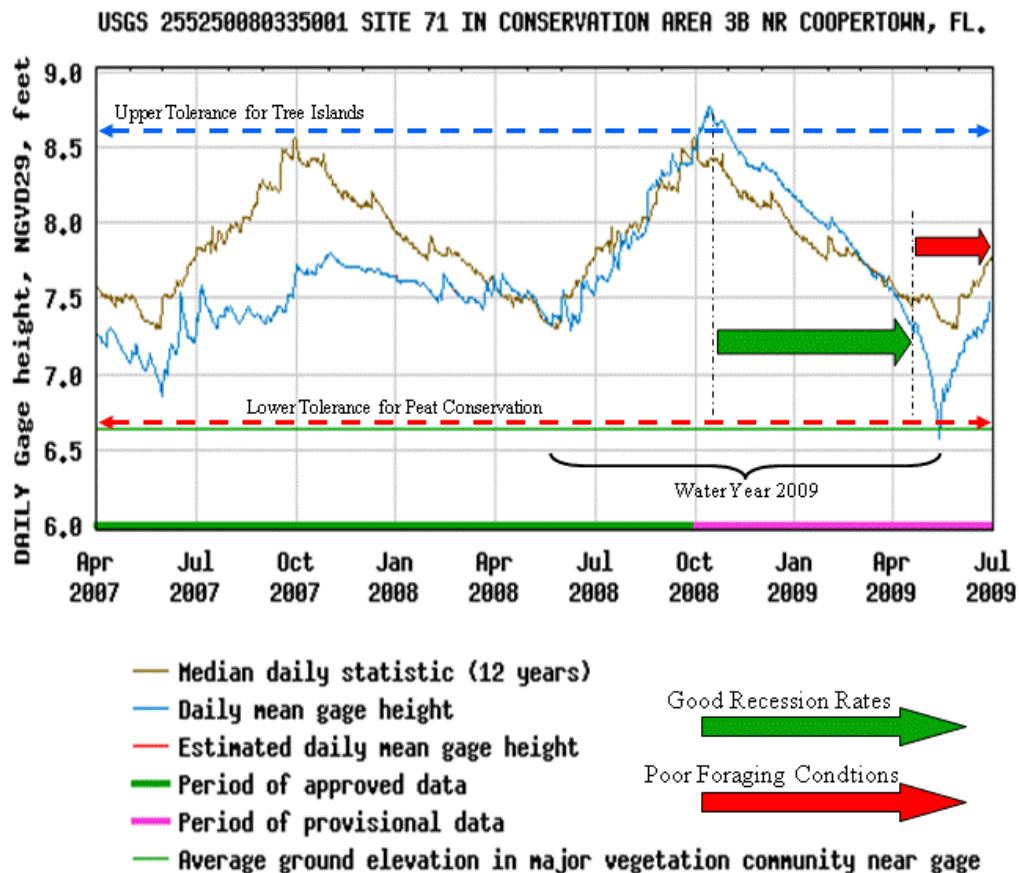
Figure 6-6. Hydrology in central Water Conservation Area 3B (gage 71) in relation to the recent 12-year average and indices for tree islands, peat conservation, and wading bird foraging.

NORTHEAST SHARK RIVER SLOUGH

The uniqueness of the hydrology and drought in the Everglades during WY2007, WY2008, and now WY2009 is captured by the Northeast Shark River Slough hydrograph **(Figure 6-7)**. Four years in a row this region of the ENP experienced violations of the MFLs standard (Sklar et al., 2008). After what was a reasonable return to average wet season conditions in July 2008, water levels began dropping in December and good recession rates for wading bird foraging and nesting remained until April 2009. These good recession rates and water depths ended in April when there was a rapid stage decline and water levels went 1.5 ft below ground. For the peak nesting period, these conditions were unfavorable. High May rainfall created a dramatic beginning to WY2010, increasing in the stage by 2.5 ft over two months (May and June).
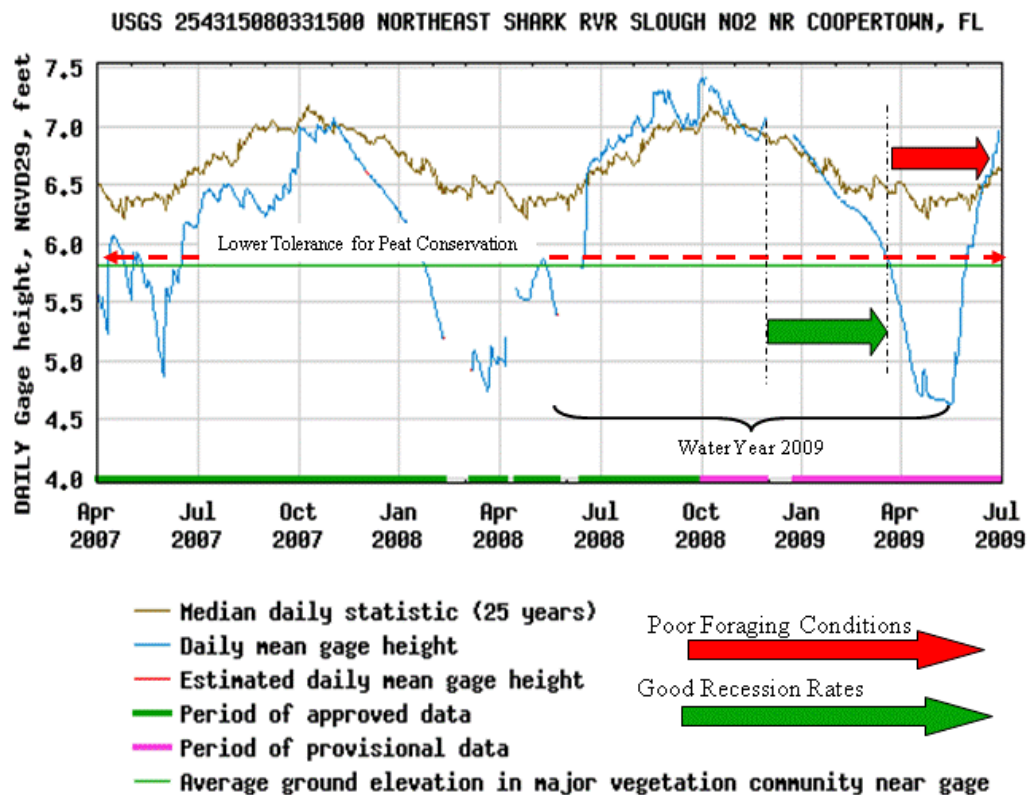

Figure 6-7. Hydrology in Northeast Shark River Slough in relation to the recent 25-year average with indices for tree islands, peat conservation, and wading bird foraging.

WILDLIFE ECOLOGY

Mark Cook, Nate Dorn[2], Robert Shuford and Robert M. Kobza

Previous research has shown that the distribution of wildlife in the Everglades is a function of hydrology, water quality, climate, and biotic interactions. Wildlife studies are conducted in the Everglades by the District, the ENP, U.S. Fish and Wildlife Service, Florida Fish and Wildlife Conservation Commission, and Florida universities. The District focuses largely on the interactions between wading birds, aquatic prey species, and hydrology as part of a long-term goal to restore historical wildlife populations and a short-term goal to prevent further environmental degradation.

This section summarizes wading bird nesting effort and success during the 2009 breeding season (WY2009–early WY2010) and reports on Year 1 of an experimental study at the Loxahatchee Impoundment Landscape Assessment (LILA) facility examining how drought and flow affect aquatic prey populations and, in turn, wading bird foraging (a collaboration with Florida Atlantic University). Also in this section is an updated species list of the macroinvertebrates observed at sentinel sites in the WCAs during wet and dry seasons. Additional wildlife studies were conducted as part of the Cattail Habitat Improvement Project experiment and are reported in the *Landscape Processes* section of this chapter.

WADING BIRD MONITORING

Wading birds are excellent indicators of wetland ecosystem health and have a central role in the Comprehensive Everglades Restoration Plan (CERP). Nesting figures for CERP performance measures are restricted to colonies in the Greater Everglades Region, i.e., the WCAs and the ENP, for the following five species:

- great egret (*Casmerodius albus*)
- snowy egret (*Egretta thula*)
- tricolored heron (*Egretta tricolor*)
- white ibis (*Eudocimus albus*)
- wood stork (*Mycteria americana*)

The timing of breeding, number of nests, and location of nesting colonies within the Everglades are used as CERP targets to evaluate the progress of the Everglades restoration effort. In addition to CERP, wading birds are of special interest to the public and play a prominent role in adaptive protocols, MFLs, and day-to-day operations of the District.

Recovery of pre-drainage (1930–1940) wading bird nesting patterns are evaluated using the following parameters:

1. Increase and maintain the total number of pairs of nesting birds in mainland colonies to a minimum of 4,000 pairs of great egrets, 10,000 to 20,000 combined pairs of snowy egrets and tricolored herons, 10,000 to 25,000 pairs of white ibises, and 1,500 to 3,000 pairs of wood storks.

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[2] Florida Atlantic University, Davie, FL

2. Shift in timing of nesting in mainland colonies to more closely match pre-drainage conditions. Specific recovery objectives would be for wood storks to initiate nesting no later than January in most years and for ibis, egrets, and herons to initiate nesting from February through March in most years.

3. Return of major wood stork, great egret, ibis/small egrets, and heron nesting colonies from the Everglades to the coastal areas and the freshwater ecotone of the mangrove estuary of Florida Bay and the Gulf of Mexico.

4. Reestablishment of historical distribution of wood stork nesting colonies in the Big Cypress Basin and in the region of mainland mangrove forests downstream from the Shark Slough and Taylor Slough basins. Increase the proportion of birds that nest in the southern ridge and slough marsh-mangrove ecotone to greater than 50 percent of the total for the entire Everglades basin.

5. For wood storks, restore productivity for all colonies combined to greater than 1.5 chicks per nest.

6. An interval between exceptional white ibis nesting events, defined as > 70th percentile of annual nest numbers for the period of record.

The information reported in this section represents a compilation of data collected by a variety of institutions which includes Florida Atlantic University (FAU), University of Florida (UF), the ENP, and the Audubon Society. The data counts include all wading bird species (except cattle egret, *Bubulcus ibis*) nesting throughout the South Florida region (for details on colony locations see Cook and Kobza, in prep.). The period covered by this report is the nesting season from January through July 2009 (the last four months of WY2009 plus the first three of WY2010).

The estimated number of wading bird nests in South Florida in 2009 was approximately 80,000. This is the largest nesting effort recorded in South Florida since the 1940s and represents a 335 percent increase relative to last year's breeding season, a 90 percent increase over the average of the last 10 seasons and surpassing the previous record year, 2002, by approximately 11,000 nests. The white ibis and the federally endangered wood stork, which has generally exhibited very low nesting effort over the past decade, both produced numbers of nests that have not been observed since the pre-drainage period. The wood stork produced approximately 6,000 nests in South Florida in 2009, which is a 182 percent increase over the average of the past decade and 1,600 percent greater than 2008. Numbers of white ibis nests were 429 percent higher than last year and 106 percent greater than the average of the past 10 years. Numbers of snowy egret and tricolored heron nests increased relative to the past two years but were lower than the 10-year average (1999–2009). Roseate spoonbill (*Ajaja ajaja*) nest numbers in Florida Bay were generally low and similar to recent years. Conditions in the Greater Everglades were particularly conducive for wading bird reproduction and an estimated record high of 73,096 nests were recorded in this region. For individual species responses in the Everglades, see **Figure 6-8**.
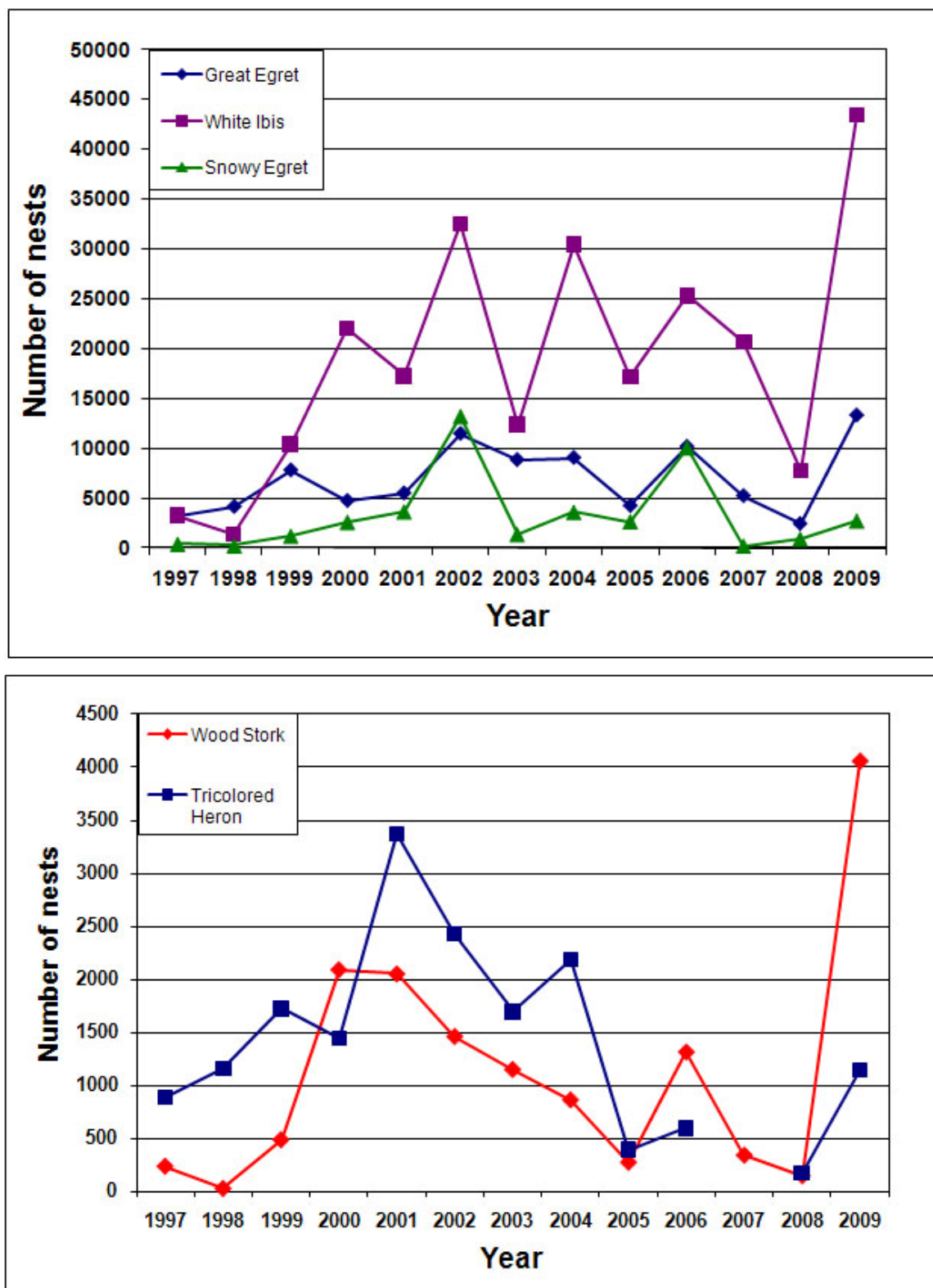
Figure 6-8. Wading bird nesting numbers in the Everglades for individual Comprehensive Everglades Restoration Plan (CERP) species since 1997.

Nesting effort in the Greater Everglades is rarely distributed uniformly among regions. In 2009, WCA-3 supported the most nests (60 percent), with WCA-1 (19 percent) and the ENP supporting a similar proportion of nests (22 percent). This spatial distribution of nests reverses the trend during recent years of an annual increase in the proportion of birds nesting in WCA-1 at the expense of nesting in WCA-3. The ENP historically supported the largest number of nests in the system.

CERP's goal is to restore hydrologic conditions to re-establish aquatic prey densities and concentrations across the landscape that, in turn, will support the return of large, successful wading bird nesting colonies to the Southern Everglades. Nesting effort in the estuaries was low in 2008, but in 2009 the ENP was relatively more attractive to nesting birds (15,432 nests). Indeed, this level of nesting effort has not been observed in the ENP since 1941. Another nesting pattern over the past 10 years has been for a large proportion of nests to be concentrated in a single large colony (Alley North) located in northeast WCA-3A. For the past two breeding seasons, however, Alley North and the adjacent marsh dried out prior to breeding and nesting was not initiated at the colony. But in 2009, Alley North was once again the largest colony in South Florida with 25,102 nests.

Generally, nesting was very successful for most species in 2009. Of 237 white ibis and great egrets nests whose fates were followed in the WCAs, between 63 and 95 percent of nests per colony fledged young. Average numbers of fledglings per nest were high and ranged between 1.7 and 2.4 nestlings per successful nest. However, a District study during WY2009 in WCA-1 revealed that mammalian and alligator predation of white ibis nests can be significant during low water stages, and suggests that hydrologically mediated predation may be an important additional limitation on wading bird reproduction. Wood stork production was exceptionally high with most pairs rearing an average of 2–3 large nestlings to the branchling stage. Unfortunately, the early onset of the rainy season in mid-May flooded key stork foraging areas and many nestlings only a week or two from fledging were abandoned and succumbed to starvation. Despite these losses, thousands of nestlings that hatched relatively early in the season managed to fledge before the rain events and, in general, wood stork nesting was relatively successful.

The annual nesting response of wading birds improves scientific understanding of the Everglades ecosystem functions. Exceptional wading bird nesting events as observed in 2009 are hypothesized to occur one or two years after extensive drought conditions as a consequence of aquatic predator reduction (the 'predator-release hypothesis'; Frederick and Ogden, 2001). In short, drought reduced predatory fish densities (Chick et al., 2004) and allowed crayfish (*Procambarus fallax*) and small fish populations to recruit successfully (Dorn, 2008) and achieve higher biomass in subsequent dry seasons. The 2009 nesting season was preceded by two years of drought (WY2006–WY2007) that potentially reduced predatory fish populations and then by a relatively wet year (2008) that promoted high prey production. Also necessary for successful nesting are the appropriate hydrologic conditions during the current nesting season for the increased prey biomass to become concentrated and available to birds. During the 2009 nesting season, water levels declined at appropriate rates, there were no major water level reversals until very late in the season, and stages were generally ideal for effective wading bird foraging (see the *Hydrologic Patterns for Water Year 2009* section of this chapter). The District is currently testing the 'predator-release hypothesis' at the LILA facility (see the *Testing the Predator-Release Hypothesis: Crayfish Responses to Large Fish Reduction* section of this chapter).

Three of four species (wood storks, great egrets, and white ibis) met the numeric nesting targets proposed by the South Florida Ecosystem Restoration Task Force (**Table 6-4**). Two other performance measures for Everglades restoration are an increase in the number of nesting wading birds in the coastal Everglades and a shift in the timing of wood stork nesting to earlier in the breeding season (Ogden, 1994). Wood storks did initiate breeding earlier in the season (late January) relative to recent years, and there was a slight shift in the distribution of nests towards the coastal zone.

Table 6-4. Numbers of wading bird nests in the WCAs and Everglades National Park (ENP or Park) compared to CERP targets and historical ranges. Target numbers are based on known numbers of nests for each species during the pre-drainage period 1930–1940 as summarized by Ogden (1994).

Species	1998–2000	1999–2001	2000–2002	2001–2003	2002–2004	2003–2005	2004–2006	2005–2007	2006–2008	2007–2009	Target
Great Egret	5,544	5,996	7,276	8,460	9,656	7,829	8,296	6,600	5,933	6,962	4,000
Snowy Egret/ Tricolored Heron	2,788	4,270	8,614	8,088	8,079	4,085	6,410	4,400*	4,067	1,805	10,000–20,000
White Ibis	11,270	16,555	23,983	20,758	24,947	20,993	24,926	21,133	18,033	23,994	10,000–25,000
Wood Stork	863	1,538	1,868	1,596	1,191	742	800	633	633	1,516	1,500–2,500

TESTING THE PREDATOR-RELEASE HYPOTHESIS: CRAYFISH RESPONSES TO LARGE FISH REDUCTION

Hydrology has at least two functional roles in regulating trophic transfer of biomass to wading birds in the Everglades. As Gawlik (2002) points out, appropriate nesting season water depths are crucial for making the sloughs available for foraging. Hydrology plays another requisite function during the prey production phase prior to the nesting season; drying events and flooding extent can affect prey species composition and abundance (Trexler et al., 2005; Dorn and Trexler, 2007; Dorn, 2008). In short, hydropatterns and appropriate seasonal depths together regulate food web architecture and trophic transfer. Resolving the relationship between hydropattern and prey production is therefore fundamental for development of restoration targets.

Patterns of historical wading bird nesting suggest that colonies in the Everglades were enhanced for one to two years following strong droughts (Frederick and Ogden, 2001). Frederick and Ogden (2001) hypothesized that either nutrient release or aquatic predator reduction was responsible for prey enhancement following extensive wetland drying. Ogden et al. (2003) further hypothesized that crayfish might be dynamically linked to drought and pulsed wading bird success, but these hypotheses remain untested.

The current study quantifies the aquatic animal communities in the LILA experimental macrocosms as part of a multi-year investigation of the effects of drying and fish reduction on crayfish population size, densities of other prey, and wading bird foraging response. The study hypothesizes that drying-related reductions of medium-sized predatory fish (Chick et al., 2004) will allow crayfish populations to recruit more successfully (Dorn, 2008) and achieve higher densities by the following dry season which will, in turn, provide better foraging opportunities for birds, especially the white ibis (*Eudocimus albus*) (**Figure 6-9**). Smaller-bodied fishes (e.g., smaller killifishes) may be enhanced for a few years after predator reduction as well, although they are not hypothesized to reach substantial densities in the first year following dry conditions (Trexler et al., 2005). The putative crayfish predators are adult and juvenile sunfish (*Lepomis* spp., *Enneacanthus gloriosus*) as well as similar-sized cichlids (e.g., *Cichlasoma* spp.) and killifish [i.e., 30–100 millimeters (mm) total length] that can feed extensively on juvenile crayfish (< 15 mm total length) and potentially stunt recruitment. Roth et al. (2007) working in lakes, determined that abundant sunfish effectively limit crayfish production by feeding on small juveniles.

The aquatic sampling of the macrocosms within LILA allows researchers to look for effects of flow on aquatic communities since the overarching LILA experimental design contrasts two cells managed with regular flow (M2 and M4) versus two non-flowing cells (M1 and M3).

Methods

In March 2009, 1 square meter (m^2) throw traps were used to quantify the aquatic animal communities in LILA. Fourteen throw trap samples were collected from the eastern half of two macrocosms (one flowing, one non-flowing) in both deep and shallow sloughs. The throw traps were collected after the water had receded from the ridges for two weeks and at depths too deep for extensive wading bird foraging [30–50 centimeters (cm)]. The sampling in March is timed to precede the first bird foraging by a couple weeks. All collected animals were preserved and returned to the lab for processing. Trap nets were also set (i.e., fyke nets) for three consecutive nights in each macrocosm to estimate predatory fish activity-density (catch-per-unit effort). One trap effort was defined as the sum of the average catch for each of three trap types on a given night and nights were treated as replicates in the analysis. Although the replicates are not completely independent (i.e., some fish were probably caught more than once) the catch rates serve as a measurement of the consistency of predatory fish activity-density within a cell.

Densities of crayfish (*Procambarus fallax*), fish (all species combined), shrimp (*Palaemonetes paludosus*) and total molluscs (bivalves plus gastropods) were analyzed with linear models comparing the two macrocosms with traps (n = 14) as the units of observations.
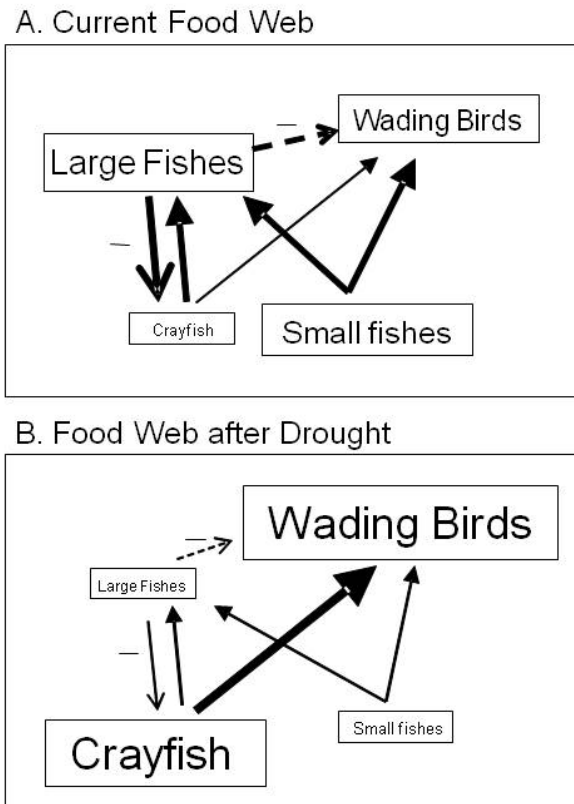

Figure 6-9. The hypothesized food web re-configurations in LILA one to two years after drying and reduction of populations of large-bodied fishes (i.e., sunfish, cichlids). Arrows with solid heads represent energy/matter flow while arrows with open heads indicate direct (solid) and indirect (dashed) effects of large-bodied fishes that are hypothesized to change with the manipulation. Sizes of arrows and boxes indicate the relative sizes of the energy transfers, effects, and populations.

Results and Discussion

The throw-trap density estimates for fish and crayfish were statistically indistinguishable between cells, but shrimp and especially molluscs (including *Sphaeriid* and *Unionid* clams, juvenile *Pomacea paludosa* and planorbid snails) were more abundant in the flowing cell (**Figure 6-10**). The overall crayfish densities (~$1/m^2$) were quite low, but densities were greater in the shallow sloughs than in the deep sloughs (data not shown). For comparison, average densities in six Loxahatchee sloughs normally exceed $4/m^2$ (Dorn and Volin, 2008). The small sunfish densities were nearly equivalent and relatively high (3–$4/m^2$) in both macrocosms. The sunfish assemblages as measured by throw traps were dominated (75 percent) by the dollar sunfish (*Lepomis marginatus*) and the blue-spotted sunfish (*Enneacanthus gloriosus*). Activity-densities of sunfish (**Figure 6-11**) and total predatory species caught in trap nets in a non-flowing cell (M3) were greater than in the flowing cell (M4) (t-tests p < 0.05). These data and the ongoing sampling of all four cells will serve as the baseline values for a manipulation planned for spring 2010, in which the study design is to dry out the deep sloughs and remove fish from two of the four cells.
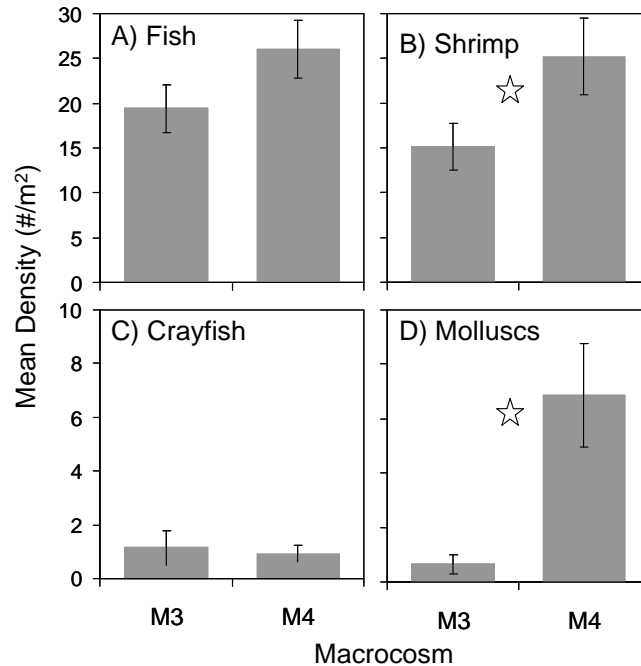
Figure 6-10. Mean throw trap densities [± standard error (S.E.)] of (A) fish, (B) shrimp, (C) crayfish, and (D) molluscs (gastropods and bivalves) in two LILA macrocosms. M4 is managed for flow while M3 is a non-flowing cell. Stars indicate statistically significant differences ($p < 0.05$) between macrocosms.
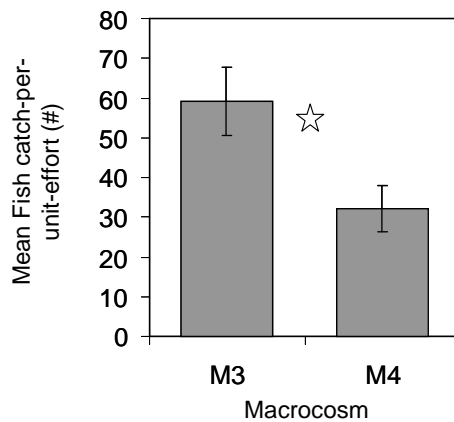
Figure 6-11. Average overnight catch rates of sunfish (mostly *Lepomis* spp. and *Enneacanthus gloriosus*) in M3 and M4 at LILA in March 2009. M4 is the flowing cell. Each macrocosm was trapped for three consecutive nights with a combination of three different trap nets. Four traps of each trap were set each night in each macrocosm and nightly averages for each type were summed. The star indicates that catch rates were statistically different ($p = 0.05$).

SPATIAL AND TEMPORAL VARIATION IN EVERGLADES MACROINVERTEBRATE COMMUNITIES

Invertebrates are ubiquitous throughout the Everglades ecosystem but little is known about their spatial and temporal distributions. While it is assumed that these animals play an important role in nutrient cycling and food web dynamics, basic community structure information is still lacking and general descriptions of their distribution across the Everglades landscapes are rare. This report documents spatial and temporal patterns of macroinvertebrate communities (distribution) in the WCAs and asks the question: Are there significant invertebrate population differences among the WCAs and if so, what might explain it? The answer to these questions are expected to be relevant to both operations and restoration, assuming these animals do play an important role in the wading bird and fish food webs.

Methods

Dip net sampling was conducted at 10 sites in WCA-1, 13 sites in WCA-2A, and 12 sites in WCA-3A during the WY2000–WY2008 wet and dry seasons. Macroinvertebrates were preserved in 10 percent formalin and shipped to the Florida Department of Environmental Protection (FDEP) where taxa were identified to the lowest practical level and enumerated. Invertebrate abundance (as numbers of individuals per sweep), species richness (total number of species), and Shannon-Wiener Diversity Index (Magurran, 1988) were calculated on non-transformed data. Non-metric multidimensional scaling (NMS) analysis coupled with multiple response permutation procedure (MRPP) was conducted on log (x+1) transformed abundance data to detect differences between regions and seasons (McCune and Grace, 2002).

Results

Regional Analysis

Invertebrate communities in the WCAs were comprised of 229 taxa occupying 125 Families among 17 Orders. Insecta (primarily true flies, Diptera), Amphipoda, Gastropoda, and Oligochaeta accounted for 72 percent of both WCA-1 and WCA-3A and 58 percent of the WCA-2A invertebrate assemblage. Fifty-nine taxa occurred in at least 10 percent of the samples and were used in the NMDS analysis. Total abundance, richness, and diversities were lowest in WCA-2A (**Figure 6-12**) and MRPP indicated highly significant differences in invertebrate communities ($p < 0.00001$, Monte Carlo test of significance) between regions. Varying abundances of biting midges (*Dasyhelea*), beetle larvae (*Derallus, Berosus*), Amphipods (*Hyalella*), grass shrimp (*Palaemonetes*), rams horn snails (*Planorbella*) and multiple non-biting midges (Chironomidae) contributed to the statistically significant differences of the invertebrate community (**Table 6-5**).

Seasonal Analysis

Comparisons of wet and dry season sampling events across all three WCAs revealed temporally distinct communities (**Table 6-6**). Total abundance and diversity were greatest during the wet season. Additionally, worms (Annelida), beetle larvae (Coleoptera), and snails (Gastropoda) were dominant during this period, which is generally marked by elevated water levels. By contrast, the abundances of Amphipods (*Hyalella*), mayfly *(Caenis),* and all Diptera taxa except the soldier fly lavae (*Odontomyia*) were greater during the dry season. Diptera, especially taxa in the Chironomidae family, are the most numerous invertebrate taxa in the Everglades and believed to be an important food source for prey fish (e.g., mosquitofish and topminnows; *Gambusia* spp., *Fundulus* spp., etc.).
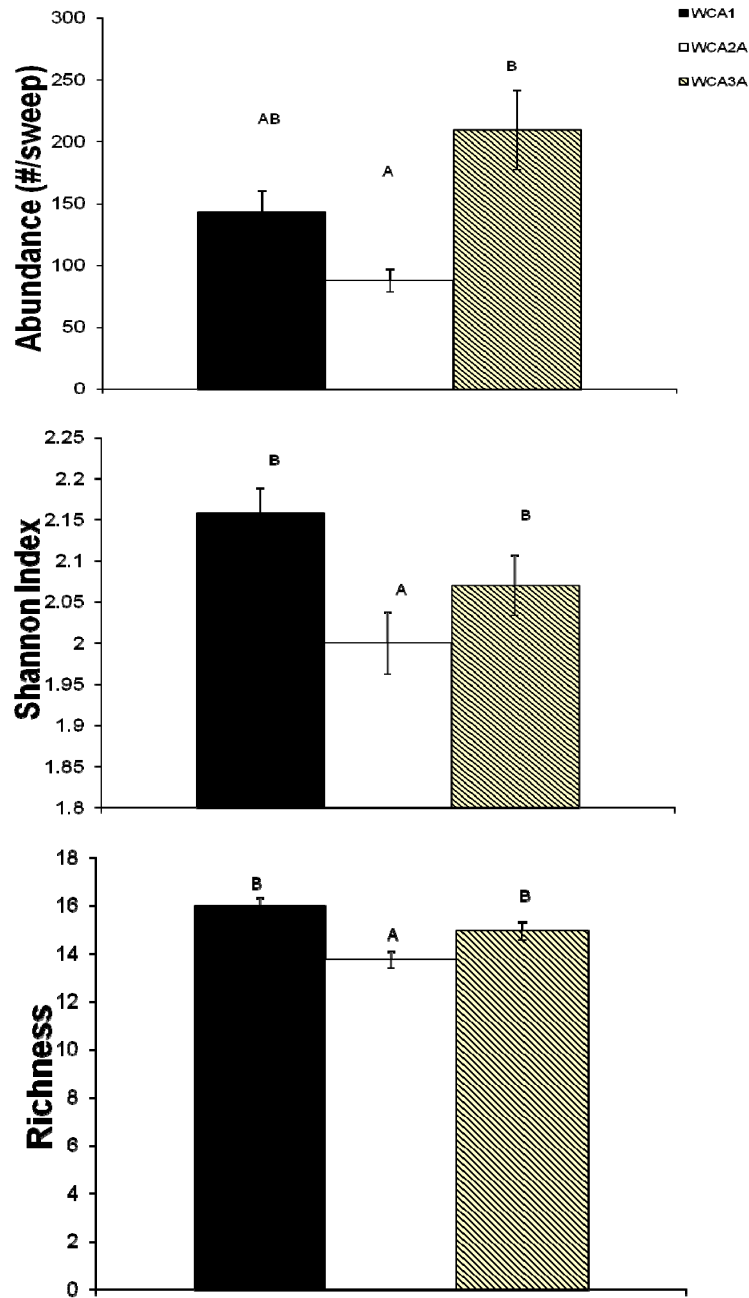
Figure 6-12. Macroinvertebrate abundance, diversity, and richness in WCA-1 (solid bars), WCA-2A (open bars), and WCA-3A (striped bars). Bars are means $\pm$ 1 standard error (S.E.). Means labeled with the same letters are not significantly different ($p > 0.05$).

Table 6-5. Mean and standard error of taxa occurring in at least 10 percent of samples in the three regions (WCA-1, WCA-2A, and WCA-3A) of the Everglades. Mean value is the number of individuals caught per sweep.

TAXON		WCA-1		WCA-2A		WCA-3A	
		Mean	Std Error	Mean	Std Error	Mean	Std Error
	Acariformes						
Arrenuridae	*Arrenurus*	0.57	0.13	0.62	0.14	1.32	0.41
Oribatida	*Oribatei/Oribatida*	0.76	0.28	0.04	0.03	0.47	0.19
	Gastropoda						
Physidae	*Haitia/Physa*	0.34	0.13	0.47	0.12	0.69	0.34
	Physella	0.65	0.2	0.37	0.14	0.93	0.23
Planorbidae	*Micromenetus*	0.12	0.05	0.11	0.04	0.77	0.31
	Planorbella	0.88	0.19	0.48	0.12	0.97	0.22
	Coleotera						
Dytiscidae	*Celina*	0.26	0.09	0.1	0.03	0.29	0.09
	Desmopachria	0.2	0.06	0.22	0.08	0.16	0.06
Hydrophilidae	*Berosus*	0.18	0.08	0.18	0.06	2.61	0.72
	Derallus	0.37	0.18	0.28	0.06	0.96	0.29
Noteridae	*Hydrocanthus*	0.02	0.01	0.44	0.25	0.78	0.25
	Suphisellus	0.06	0.03	0.12	0.04	0.27	0.1
	Diptera						
Ceratopogonidae	*Dasyhelea*	10.21	2.01	11.22	2.22	21.38	3.42
	Palpomyia/bezzia grp.	1.51	0.28	0.5	0.13	11.05	6.36
Stratiomyidae	*Odontomyia*	0.18	0.09	0.43	0.14	0.49	0.11
Tipulidae	*Limonia*	0.45	0.15	0.33	0.1	0.85	0.31
Chironomidae	*Ablabesmyia rhamphe grp.*	0.65	0.28	0.09	0.04	0.39	0.11
	Apedilum	0.55	0.49	0.59	0.18	0.02	0.01
	Beardius	0.5	0.11	0.23	0.08	0.33	0.15
	Chironomus	2.25	0.46	1.58	0.41	18.38	4.76
	Cladopelma	0.14	0.06	0.59	0.15	0.5	0.32
	Cladotanytarsus	4.98	1.77	0.31	0.1	4.56	1.99
	Cladotanytarsus sp. a epler	6.37	1.67	1.89	0.69	3.56	1.09
	Dicrotendipes modestus	1.5	0.55	1.4	0.69	0.3	0.26
	Goeldichironomus holoprasinus	0.74	0.23	0.96	0.46	0.68	0.29
	Kiefferulus	1.05	0.35	0.47	0.14	1.94	0.79
	Labrundinia neopilosella	0.28	0.09	0.15	0.05	0.14	0.05
	Lasia decolorata	3.7	0.76	2.34	0.51	23.74	7.43
	Larsia	2.37	0.65	1.13	0.29	7.32	3.64
	Parachironomus alatus	4.65	1.15	0.72	0.22	1.29	0.38
	Parakiefferiella sp. f epler	10.89	2.41	6.05	1.55	5.98	2.39
	Paramerina	0.58	0.15	0.14	0.05	0.32	0.1
	Paratanytarsus	5.17	1.25	0.29	0.13	1.67	0.62
	Polypedilum	0.32	0.13	0.08	0.03	0.3	0.1
	Polypedilum halterale grp.	0.61	0.31	0.89	0.2	1.91	0.51
	Polypedilum illinoense grp.	1.31	0.33	0.49	0.23	1.76	0.76
	Polypedilum sp. a epler	9.52	2.28	1.68	0.52	2.1	0.68
	Polypedilum trigonum	2.56	0.78	0.58	0.13	7.38	2.77
	Polypedilum tritum	0.9	0.5	0.35	0.1	0.08	0.03

Table 6-5. Continued.

TAXON		WCA-1		WCA-2A		WCA-3A	
		Mean	**Std Error**	**Mean**	**Std Error**	**Mean**	**Std Error**
	Diptera						
Chironomidae	*Pseudochironomus*	0.87	0.32	0.28	0.15	12.21	4.17
	Tanytarsus	1.28	0.42	0.59	0.17	1.82	0.67
	Tanytarsus limneticus	1	0.44	0.78	0.25	0.99	0.4
	Tanytarsus sp. g epler	2.15	0.53	3.53	0.6	5.21	1.1
	Tanytarsus sp. r epler	1.69	0.58	3.89	1.06	2.18	0.59
	Zavreliella marmorata	0.15	0.04	0.05	0.02	1.04	0.24
Baetidae	*Callibaetis*	0.09	0.04	0.42	0.12	0.42	0.15
Caenidae	*Caenis*	2.56	0.54	1.86	0.55	5.88	2.61
	Hemiptera						
Naucoridae	*Pelocoris*	0.13	0.04	0.1	0.03	0.23	0.07
	Odonata						
Coenagrionidae	*Ischnura*	0.57	0.28	0.38	0.18	0.25	0.08
	Trichoptera						
Hydroptilidae	*Oxyethira*	0.73	0.27	0.02	0.01	0.44	0.18
	Amphipoda						
Hyalellidae	*Hyalella azteca*	14.43	1.74	9.98	1.16	14.91	2.5
	Decapoda						
Palaemonidae	*Palaemonetes*	1.28	0.27	1.14	0.29	0.73	0.18
	Oligochaeta						
Naididae	*Bratislavia unidentata*	3.13	0.75	0.77	0.19	2.32	0.61
	Dero digitata complex	4.24	2.23	3.33	0.75	3.61	1.05
	Dero pectinata	9.17	6.07	3.25	0.77	5.54	1.63
	Dero	0.45	0.15	1.03	0.36	1.13	0.44
	Haemonais waldvogeli	0.95	0.38	0.93	0.4	0.66	0.26
	Pristina aequiseta	0.67	0.32	1.45	1.03	0.7	0.2
	Pristina leidyi	1.61	0.84	4.25	3.02	0.8	0.21
	TOTAL	**142.99**	**17.33**	**87.64**	**9.04**	**210.91**	**32.02**

Table 6-6. Dry and wet season means and standard error of invertebrate taxa occurring in at least 10 percent of samples in the three regions (WCA-1, WCA-2A, and WCA-3A) of the Everglades. Mean value is the number of individuals caught per sweep.

TAXON		Dry Season		Wet Season	
		Mean	Std Error	Mean	Std Error
	Acariformes				
Arrenuridae	Arrenurus	0.93	0.29	0.75	0.12
Oribatida	Oribatei/Oribatida	0.5	0.19	0.31	0.12
	Gastropoda				
Physidae	Haitia/Physa	0.85	0.26	0.2	0.06
	Physella	0.6	0.15	0.67	0.16
Planorbidae	Micromenetus	0.09	0.05	0.54	0.19
	Planorbella	0.64	0.14	0.86	0.15
	Coleoptera				
Dytiscidae	Celina	0.11	0.03	0.32	0.08
	Desmopachria	0.06	0.02	0.31	0.07
Hydrophilidae	Berosus	0.34	0.09	1.54	0.45
	Derallus	0.2	0.06	0.84	0.2
Noteridae	Hydrocanthus	0.09	0.04	0.72	0.23
	Suphisellus	0.04	0.02	0.25	0.07
	Diptera				
Ceratopogonidae	Dasyhelea	21.61	2.98	7.97	1.12
	Palpomyia/bezzia grp.	8.72	4.59	0.52	0.1
Stratiomyidae	Odontomyia	0.21	0.07	0.52	0.11
Tipulidae	Limonia	1.15	0.25	0.02	0.01
Chironomidae	Ablabesmyia rhamphe grp.	0.38	0.1	0.34	0.15
	Apedilum	0.72	0.34	0.1	0.03
	Beardius	0.55	0.13	0.17	0.05
	Chironomus	11.69	3.32	3.65	1.03
	Cladopelma	0.83	0.26	0.08	0.02
	Cladotanytarsus	4.32	1.52	2.08	0.9
	Cladotanytarsus sp. a epler	5.46	1.13	2.33	0.76
	Dicrotendipes modestus	2.24	0.67	0.05	0.02
	Goeldichironomus holoprasinus	1.04	0.35	0.59	0.24
	Kiefferulus	2.07	0.6	0.32	0.11
	Labrundinia neopilosella	0.29	0.07	0.1	0.03
	Larsia	6.4	2.64	1.11	0.27
	Larsia decolorata	16.14	5.33	4.47	1.01
	Parachironomus alatus	3.67	0.75	0.72	0.26
	Parakiefferiella sp. f epler	12.81	2.38	2.85	0.85
	Paramerina	0.51	0.12	0.18	0.04
	Paratanytarsus	4.26	0.91	0.43	0.14
	Polypedilum	0.41	0.11	0.07	0.02

Table 6-6. Continued.

TAXON		Dry Season		Wet Season	
		Mean	**Std Error**	**Mean**	**Std Error**
	Diptera				
Chironomidae	*Polypedilum halterale grp.*	1.51	0.33	0.83	0.26
	Polypedilum illinoense grp.	2.24	0.59	0.23	0.11
	Polypedilum sp. a epler	7.86	1.58	0.98	0.24
	Polypedilum trigonum	6.03	2.01	1.2	0.4
	Polypedilum tritum	0.87	0.33	0.04	0.02
	Pseudochironomus	9.31	3.03	0.24	0.13
	Tanytarsus	2.16	0.56	0.38	0.08
	Tanytarsus limneticus	1.22	0.33	0.65	0.26
	Tanytarsus sp. g epler	6.21	0.9	1.49	0.28
	Tanytarsus sp. r epler	4.58	0.91	1	0.34
	Zavreliella marmorata	0.45	0.15	0.37	0.09
	Ephemeroptera				
Baetidae	*Callibaetis*	0.17	0.05	0.44	0.12
Caenidae	*Caenis*	6.12	1.93	1.07	0.2
	Hemiptera				
Naucoridae	*Pelocoris*	0.15	0.05	0.15	0.03
	Odonata				
Coenagrionidae	*Ischnura*	0.29	0.08	0.48	0.19
	Trichoptera				
Hydroptlidae	*Oxyethira*	0.65	0.21	0.13	0.06
	Amphipoda				
Hyalellidae	*Hyalella azteca*	19.13	1.92	7.6	0.98
	Decapoda				
Palaemonidae	*Palaemonetes*	1.46	0.24	0.69	0.16
	Annelida				
Naididae	*Bratislavia unidentata*	1.58	0.34	2.34	0.5
	Dero digitata complex	2.48	0.63	4.73	1.38
	Dero pectinata	2.79	0.7	8.32	3.49
	Dero	0.41	0.16	1.29	0.35
	Haemonais waldvogeli	0.34	0.11	1.28	0.36
	Pristina aequiseta	0.14	0.06	1.68	0.73
	Pristina leidyi	0.31	0.1	4.02	2.11

Discussion

Invertebrate composition in each of three WCAs sampled was similar. However, community differences between regions resulted from significant differences among individual taxa abundances. Macroinvertebrate species were rarely regionally exclusive, which suggests that community differences were influenced by local environmental and ecological factors (e.g., hydroperiod, benthos, and predation, among others).

WCA-3A is located approximately 36 miles southwest of WCA-1, yet these regions had similar total abundance, diversity, and richness. Despite similarities, community differences between WCA-1 and WCA-3A were greatly influenced by the abundances of beetles, biting midges, and multiple taxa of non-biting midges.

By contrast, total abundance, diversity, and richness were consistently lower in WCA-2A (located approximately 12 miles south of WCA-1) than in WCA-1 or WCA-3A (located approximately 24 miles southwest of WCA-2A). Abundances of individual taxa in WCA-2A were rarely greater than that of either WCA-1 or WCA-3A and may be indicative of contrasting management practices in this region.

Temporal differences were primarily influenced by the abundance of Diptera, Coleoptera, and Annelida. Both Coleoptera and Annelida taxa displayed higher abundances during the wet season. This is expected since these taxa require complete inundation for survival, feeding, and development. Diptera and mayfly (*Caenis)* abundances were greater in the dry season. These emergent taxa may modify their aquatic life cycles (i.e., growth rates, emergence timing) in response to environmental cues like changes in water levels or temperature, thus making it possible for these animals to successfully occupy marshes with both long and short hydroperiods. The Decapoda *Palaemonetes,* and Amphipoda *Hyalella* also displayed greater abundances during the dry season. The reason for this pattern is not clear, but it is speculated that as water levels decline during dry season, grazing pressure may be ameliorated as predators are excluded from shallower portions of the marsh.

PLANT ECOLOGY

Brent Bellinger[3], Carlos Coronado, Sharon Ewe[4]
and Scot Hagerthey

The rationale for plant studies is to provide a better understanding of the biological processes that cause vegetation replacement, degradation, and premature death in relation to environmental disturbances such as phosphorus enrichment, hydrology, water quality, fire, wind or temperature change. This section examines the population characteristics of the periphyton and a suite of ecophysiological parameters for tree islands. Previous consolidated reports have focused on how phosphorus enrichment contributes to cattail (*Typha*) expansion and the disappearance of ridge-and-slough communities. Updates for WY2008 (Sklar et al., 2008) were synoptic and descriptive, focusing on (1) climbing fern (*Lygodium* spp.) patterns across hydrologic gradients on tree islands, (2) survivorship and recruitment rates on tree islands, (3) periphyton polysaccharides structure, and (4) tree species characteristics at the LILA facility. Updates this year (WY2009) take a more detailed look at the potential for periphyton to influence food webs, not as a food source, but rather as a toxin producer and a prey control. For WY2009, this section also delves into 12 associated tree physiologic measures to better understand plant function to help guide island recovery and to see which measures are sensitive to water management (islands with contrasting hydrologic regimes are compared).

For the third consecutive year, District scientists are working with contractors to survey tree islands in WCA-3A and 3B for the presence of the invasive Old World climbing fern and other invasive exotic species. This year's survey of approximately 40 tree islands brings the total number of tree islands surveyed by the District to 259 since large-scale surveys began in the year 2005. See Chapter 9 of this volume for details about nonindigenous species monitoring.

PERIPHYTON CYANOTOXINS

In the oligotrophic Everglades, cyanobacteria are a prominent component of periphyton. While it is known that cyanobacteria are capable of producing a variety of secondary metabolites referred to as cyanotoxins, toxin production varies greatly within and among species and is not an obligate process. Stimulation of cyanotoxin production is hypothesized as a response to grazing pressure or as a competitive inhibitor of bacteria, algae, and fungi (Valdor and Aboal, 2007; Berry et al., 2008). In other aquatic systems, the development of cyanobacterial blooms in eutrophic waters may result in the production of cyanotoxins that target the skin, liver, or nerve cells of higher organisms (Hitzfeld et al., 2000). Recently, culture studies have shown that a number of cyanobacterial species isolated from the Everglades produce toxins that are detrimental to fish function and survival (Berry et al., 2007, 2008; Gantar et al., 2008). This section examines whether cyanotoxins are present in Everglades periphyton, with the primary objective being to determine if cyanotoxins need to be considered as an additional regulator of the Everglades food web.

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Methods

Periphyton was collected from oligotrophic regions of WCA-1, WCA-2A, WCA-3A, and the ENP, and from eutrophic sites in WCA-2A. The algal assemblage from WCA-1 is a mix of desmids, diatoms, and cyanobacteria in a loosely consolidated community. Cyanobacteria and diatoms embedded within a thick, cohesive mat were representative of the oligotrophic regions in WCA-2A, WCA-3A, and the ENP. Green algae and cyanobacteria embedded within a gelatinous, slimy matrix were found at the eutrophic sites in WCA-2A. Samples were freeze-dried and shipped to Green Waters Laboratory/Cyano Lab for extraction and analysis of cyanotoxins. Seven toxins were screened: cylindrospermopsin and microcystin which are hepatotoxins (liver); saxitoxin, domoic acid, and anatoxin-a which are neurotoxins (nerve); and debromoaplysiatoxin and lyngbyatoxin-a which are dermatotoxins (skin). Analysis for saxitoxin, domoic acid, cylindrospermopsin, and microcystin was accomplished with an enzyme-linked immunosorbent assay, and anatoxin-a, debromoaplysiatoxin, and lyngbyatoxin-a concentrations were analyzed using high performance liquid chromatography and mass spectrometry. Cyanotoxin content data are expressed as mass of toxin (grams) per mass of periphyton (grams dry weight) (Tolhurst et al., 2005).

Results and Discussion

Six of the seven toxins screened were found in periphyton from the Everglades. The most common toxins were saxitoxin, microcystin, and anatoxin-a; cylindrospermopsin was not observed (**Table 6-7**). Microcystin could be found in periphyton from all regions, while saxitoxin and anatoxin-a were only observed in cohesive cyanobacterial mats from low nutrient, minerotrophic waters. Of note was the observation of small amounts of domoic acid, a toxin typically associated with the marine diatom *Pseudo-nitzschia* spp. (Berry et al., 2007). Recently, this toxin was observed in a new *Nitzschia* species of diatom from a variety of brackish water sites in Vietnam (Kotaki et al., 2004). The two sites in this study where domoic acid was observed in northern WCA-2A are highly minerotrophic as well as eutrophic (Hagerthey et al., unpublished); thus, it is possible that a euryhaline species related to known producers may be present at these sites. Further study is needed to confirm the presence of this toxin and to determine the species of origin. No single toxin was especially abundant (in terms of content); cohesive periphyton from the interior of WCA-2A had the greatest diversity and total content of any periphyton type sampled (**Table 6-7**). However, while the measured contents were lower relative to other aquatic systems (e.g., Velzeboer et al., 2000; Izaguirre et al., 2007), exposure to cyanotoxin concentrations may be equivalent or greater given that periphyton biomass/m^2 can be substantial (**Table 6-7**). Conversely, toxins may represent "qualitative" defenses, thus negating the need for large concentrations (Berry et al., 2008).

Based on the results of this survey, it cannot be ruled out that cyanotoxins may play a potential role in regulating the food web since two of the most common and abundant toxins identified, saxitoxin and anatoxin-a, are neurotoxins that have been shown to affect larval fish survival through paralysis, inhibition of swim bladder inflation, or by impairing embryonic development (Lefebvre et al., 2004).

Table 6-7. Cyanotoxin content for the periphyton samples collected along a longitudinal gradient from the Northern to the Southern Everglades. Cyanotoxin content is expressed as a unitless ratio (mass:mass) derived from nanograms of toxin per gram (ng toxin g^{-1}) periphyton (Tolhurst et al., 2005). Biomass concentrations were obtained from the literature, AFDW is ash free dry weight.

Location	Site/Nutrient Status	Periphyton Type	Biomass (g AFDW · m^{-2})	MC/Nodularin	CYN	STX	DA	ANTX-A	DAT	LA
WCA-1	Interior, oligotrophic	metaphyton	40 - 225[1]	4.5×10^{-8}	-	-	-	-	-	1.3×10^{-6}
WCA-2A	Margin, eutrophic	metaphyton	3 - 53[1]	5.2×10^{-8}	1.4×10^{-8}	-	3.0×10^{-9}	-	-	-
	Interior, mesotrophic	metaphyton		5.6×10^{-7}	-	5.1×10^{-8}	9.0×10^{-9}	-	-	-
	Interior, oligotrophic	metaphyton	1 - 157[1]	-	-	3.1×10^{-8}	-	1.0×10^{-8}	-	-
	Interior, oligotrophic	epiphyton	1 - 120[1]	1.2×10^{-8}	-	4.0×10^{-7}	-	4.0×10^{-8}	1.0×10^{-6}	-
	Interior, oligotrophic	epipelon	20 - 800[1]	-	-	3.1×10^{-8}	-	-	-	-
WCA-3B	Interior, oligotrophic	metaphyton/*Utricularia*	1 - 24[2]	7.5×10^{-8}	-	3.6×10^{-8}	-	3.0×10^{-8}	-	-
ENP	Interior, oligotrophic	metaphyton/*Utricularia*	286 - 3665[3]	-	-	1.0×10^{-7}	-	2.0×10^{-8}	-	-

MC-microcystin

CYN-cylindrospermopsin

STX-saxitoxin

DA-domic acid

ANTX-A-anatoxin-A

DAT-debromoaplysiatoxin

LA-lyngbyatoxin-a

[1] McCormick et al., 1998

[2] Turner et al., 1999

[3] Gottlieb et al., 2006

In addition, saxitoxin, along with domoic acid, has been shown to impact protein expression in the brains of killifish (Salierno et al., 2006). The other most common toxin, microcystin, a heptatotoxin, has been shown to impact phosphoprotein and total protein content in Japanese killifish (*Oryzias latipes* L.) (Mezhoud et al., 2008) and to reduce viability of carp (*Cyprinus carpio* L.) hepatocytes (Li et al., 2007b). Studies on zebrafish (*Danio rerio* Hamilton-Buchanan), a small, tropical freshwater fish used as a model organism in studies, have shown developmental susceptibility when exposed to toxic products from over 20 cyanobacteria isolated from the Everglades (Berry et al., 2007). These results indicate that Everglades fish species of similar physiology and ecology to those studied elsewhere may be negatively impacted by cyanotoxins. Alternatively, invertebrates are common and locally abundant within periphyton mats (Liston and Trexler, 2005), but recent studies indicate that invertebrate herbivores are less susceptible to cyanotoxins (Berry et al., 2008; Thacker et al., 2005).

Significance and Relevance to Water Management

In the oligotrophic Everglades, cyanobacteria are a prominent component of the periphyton inhabiting the slough landscape. Maintaining these communities is essential for the proper functioning of the historic Everglades ecosystem, including the characteristic high periphyton standing stock yet low secondary production (Turner et al., 1999). There are several mitigating factors that may cause this discrepancy between primary and secondary production. Cyanobacteria may be of poor food quality due to nutrient imbalances and lack of essential fatty acids (Demott and Müller-Navarra, 1997).

The exopolysaccharide coating produced by cyanobacteria may act as a physical barrier to feeding (Camacho and Thacker, 2006) and additionally, can become impregnated with calcium carbonate (Sklar et al., 2008). The results here provide another possible factor: the production of cyanotoxins could inhibit consumption by secondary consumers. Understanding the complex biogeochemistry of the periphyton is essential for putting these important producers into the proper context of interactions with other trophic levels in the Greater Everglades ecosystem.

TREE ISLAND ECOPHYSIOLOGY

To define the current condition and determine the future trajectory and health of tree islands within the WCAs as a function of operation and restoration plans, the District is examining the ecophysiology of tree island vegetation to understand vegetation stress responses to hydrologic management. Ecophysiological measures can provide insight into plant function, both on a long- and short-term basis and provide an understanding of the dynamics of tree islands.

The objective of this study was to compare landscape-level changes in plant responses using, as a surrogate, the ecophysiological responses of vegetation across the head (higher elevation) and near-tail habitats for a range of tree islands with contrasting hydrological regimes.

Methods

Four islands of differing hydrology were selected for this study (**Figure 6-13**). In WCA-3A where there is water flow from upstream, island 3AS3 has a shorter hydroperiod relative to 3AS5 (**Figure 6-14**). In WCA-3B, an impounded area that receives no surface flows and is primarily rain-driven, the northern island (3BS2) remains flooded longer than 3BS10 (**Figure 6-14**). One head and near-tail site was selected on each island, providing a total of eight sites for physiological measurements.
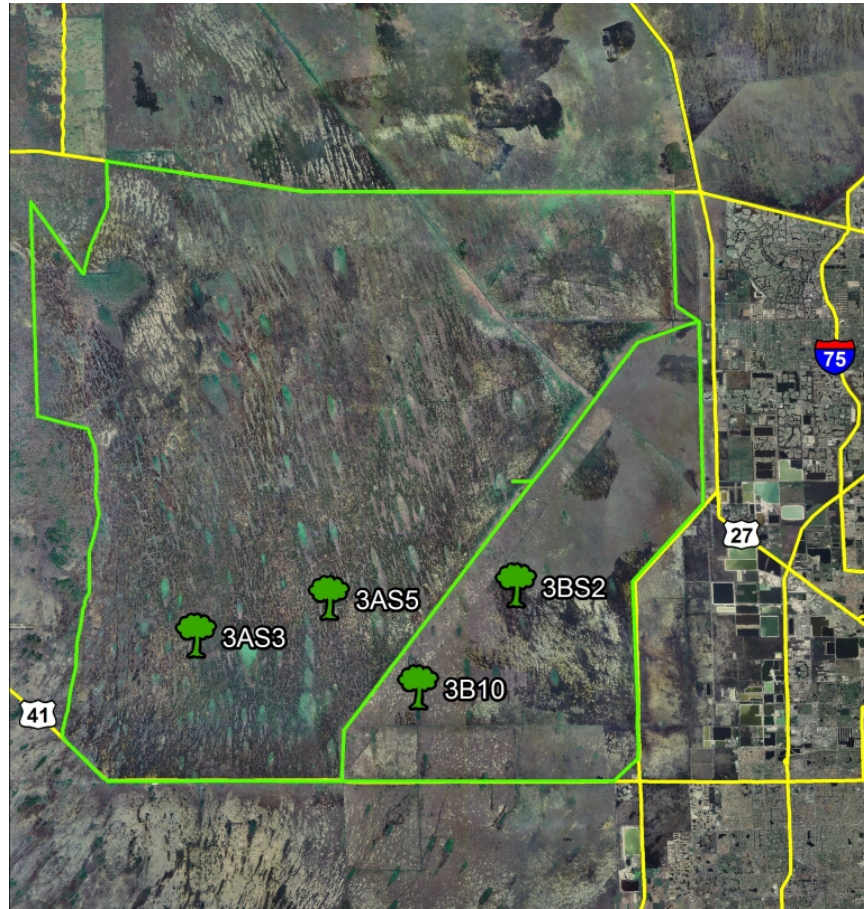
Figure 6-13. Map showing locations of the study islands within
WCA-3A and WCA-3B.

Sampling at these sites was designed to capture plant ecophysiological responses during the highest water levels for the wet season and the lowest water levels during the dry season. Therefore, sampling was conducted during the end of each season.

Ecophysiological parameters measured on individual trees from the head and near-tail of each of the four tree islands were based upon the techniques of Sternberg et al. (1991) and Sternberg and Swart (1987) and include:

- instantaneous leaf gas exchange [e.g., net CO_2 uptake (A), transpiration (E), and leaf internal CO_2 concentration (c_i)]
- leaf carbon isotope ($\delta^{13}C$), and nitrogen ($\delta^{15}N$) isotope analysis
- leaf carbon (C), nitrogen (N_{leaf}), and phosphorus (P_{leaf})
- predawn stem water potentials (Ψ) measured at 3:00 am
- plant water-use patterns (percent GW) as determined from the stable isotopes of water (δD and $\delta^{18}O$); and
- plant sap flow patterns
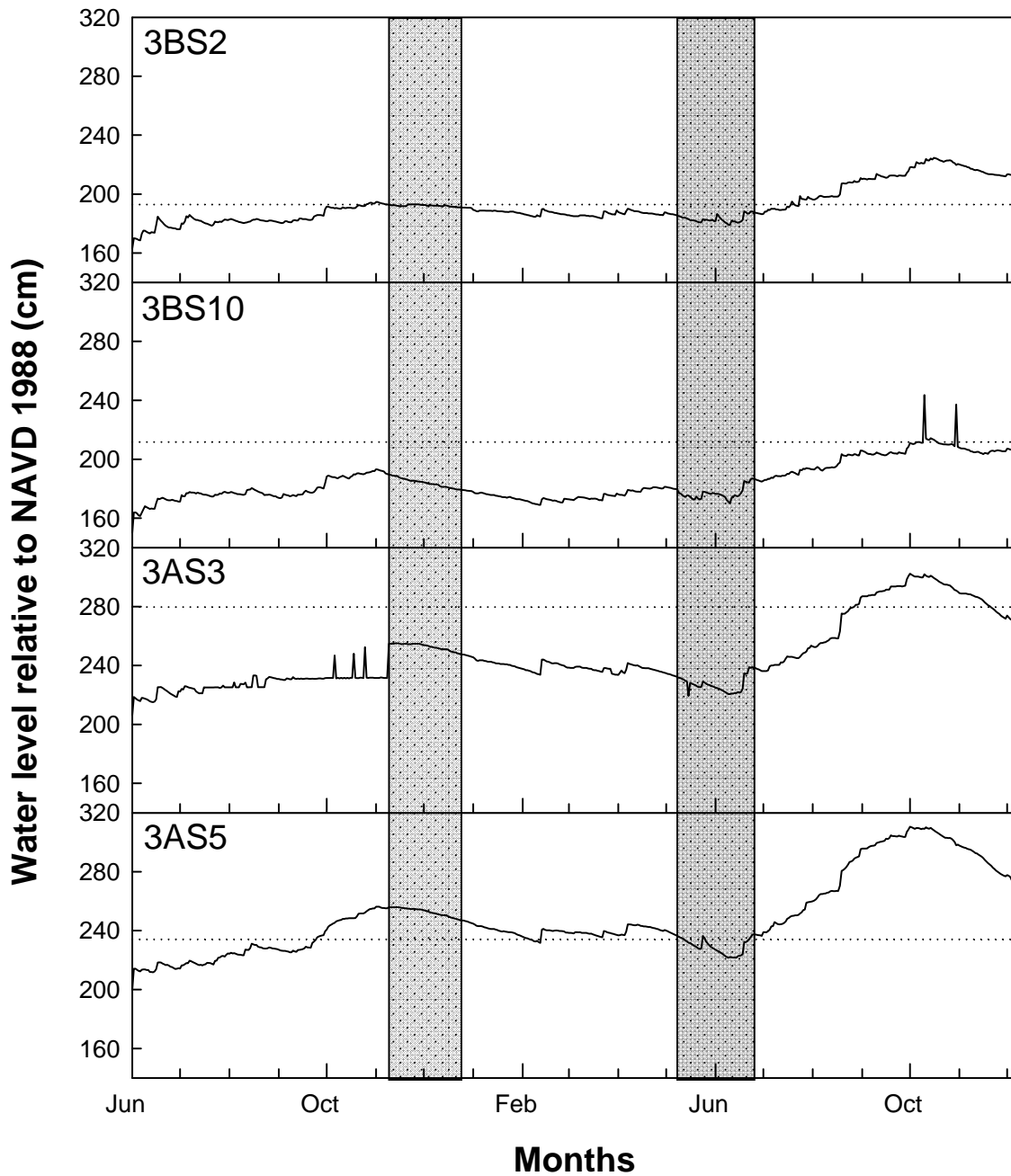
Figure 6-14. Water levels (solid line) relative to ground elevation (North American Vertical Datum 1988) (dotted line) on the heads of the four study islands from June 1, 2007−December 8, 2008. Ecophysiological measurements were carried out in the wet season of 2007 and the dry season of 2008 (grey bars). Islands in WCA-3A (3AS3 and 3AS5) have greater seasonal variations in water levels relative to WCA-3B (3BS2 and 3BS10).

Physiological parameters were measured on native species representing the dominant canopy cover on the tree islands studied. Nine species were measured including pond apple (*Annona glabra*), Florida holly (*Ilex cassine*), sweet bay (*Magnolia virginiana*), swamp bay (*Persea palustris*), bayberry (*Myrica cerifera*), willow (*Salix caroliniana*), cocoplum (*Chrysobalanus icaco*), and two fern species, leather fern (*Acrostichum danaeifolium*) and swamp fern (*Blechnum serrulatum*). Five individuals of each species were randomly chosen for physiological parameters measurements.

Results and Discussion

Normal wet season water level fluxes, as represented by levels from June through December 2008, ranged from ~40 cm in WCA-3B to ~90 cm in WCA-3A. However, when the experiments were conducted during the wet season of 2007 (**Figure 6-14**), the wet season was drier than normal. This was followed by a wetter-than-normal spring dry season due to periodic rainfall (**Figure 6-14**). Seasonal changes from the start of the wet season in June 2007 to peak water levels in late 2007 were approximately 60–70 percent of seasonal water level fluctuations in an average year (**Figure 6-14**), producing a muted seasonality.

Despite the limited hydrologic variability during the sampling year relative to a normal year, changes between seasons were characterized by differences in instantaneous gas exchange parameters (A, E), Ψ, N_{leaf}, and percent GW. This suggests that key physiological parameters are sensitive to small seasonal variability in hydrology and therefore can be used to assess in-situ instantaneous vegetation responses to slight changes in hydrologic conditions. Leaf E, Ψ, and plant percent GW were higher in the dry season relative to the wet. Leaf dry season E was 47.5 percent higher than wet season values while seasonal shifts in Ψ were observed in all islands except for 3AS3 (**Figure 6-15**). Similarly, CO_2 uptake (A) was higher in the dry season but only for three of the four islands sampled; the only exception to this pattern was 3AS5 (**Figure 6-15**). For 3BS2, 3BS10 and 3AS3, dry season A increased an average of 17.2 percent from the wet season.

Plant sap flow and N_{leaf} showed the opposite pattern, relative to the other parameters, as wet season values were greater relative to the dry season (**Figure 6-15**). The most likely explanation for this whole-plant increase in sap flow is that there were more leaves on the tree in the wet season relative to the dry season (SFWMD, 2003), resulting in overall greater water demand by the plant despite lower water loss per leaf area [i.e., lower transpiration (E) rates]. However, more information on the seasonal ecological characteristics of the canopy is needed to confirm this hypothesis. These data, in conjunction with the Ψ (collected when the plants are in equilibrium with the surrounding soil), indicate that plants across the different islands exhibit a strong demand for water during the period of maximum water levels, inducing a simultaneous aboveground demand for water during a period of increased stress. If plant water uptake rates cannot keep up with water loss through canopy, this can cause xylem cavitation (Lopez et al., 2005) that can potentially result in the death of the stems and trunks.

The increase in N_{leaf} in the wet season relative to the dry season for most islands is consistent with work from Ewe and Sternberg (2003) and Saha et al. (2009) in the ENP. Greater water availability most likely increases the mobilization of soil nitrogen, resulting in greater nitrogen availability for the plants. However, this increase in N_{leaf} was not matched by a seasonal increase in P_{leaf}. These observations indicate a greater availability and uptake of nitrogen, but not for phosphorus. The tree islands measured in this study appear to be phosphorus limited (**Figure 6-15**) as the $N_{leaf}:P_{leaf}$ ratios are in excess of 16:1 (Korselman and Meuleman, 1996); ratios > 16 indicate phosphorus limitation; ratios between 14 to 16 indicate an nitrogen and phosphorus co-limitation, while ratios < 14 are indicative of nitrogen limitation. In the dry season, some species exhibited potential nitrogen limitation but most species appeared to be phosphorus-

limited (**Figure 6-16**). Despite high values of phosphorus within the shallow soils of the tree islands relative to the surrounding marsh (Wetzel et al., 2005, 2009), the plants still reflected limited phosphorus availability relative to nitrogen. There were no relationships between either N_{leaf} or P_{leaf} and $\delta^{13}C$ among the tree islands or between seasons. This finding indicates that foliar nitrogen and phosphorus do not significantly influence the stomatal and gas exchange behavior of the leaves at these sites. As these plants are not severely nutrient-limited, plant responses do not appear to be significantly influenced by these limitations, but rather by changes in hydrology.

Studies have shown both positive (e.g., Li et al., 2007a; Saha et al., 2009), negative (Ma et al., 2007) and even the absence of relationships (Ewe and Sternberg, 2007) between N_{leaf} and $\delta^{13}C$. However, there were positive relationships with N_{leaf} and leaf $\delta^{15}N$ in the dry season and with and P_{leaf} and $\delta^{15}N$ for both seasons. The fractionation of $\delta^{15}N$ is predicted to occur when the limiting nutrient is not readily available (McKee et al., 2002). In the wet season, due to slightly higher N_{leaf} at most of the sites, nitrogen does not appear to be the limiting factor. However, in the dry season, as N_{leaf} levels were slightly lower, indicating lower soil nitrogen, it is likely that the decreased availability of nitrogen would result in the biochemical discrimination of $\delta^{15}N$ uptake by plants on the islands; lowered nitrogen availability in the dry season could also potentially induce an nitrogen limitation and/or a nitrogen–phosphorus co-limitation. The relationship between P_{leaf} and $\delta^{15}N$ is in agreement with the fractionation mechanics proposed. As phosphorus appears to be the limiting nutrient on tree islands in this study, phosphorus availability as reflected in P_{leaf}, is significantly correlated to the $\delta^{15}N$ of the vegetation.
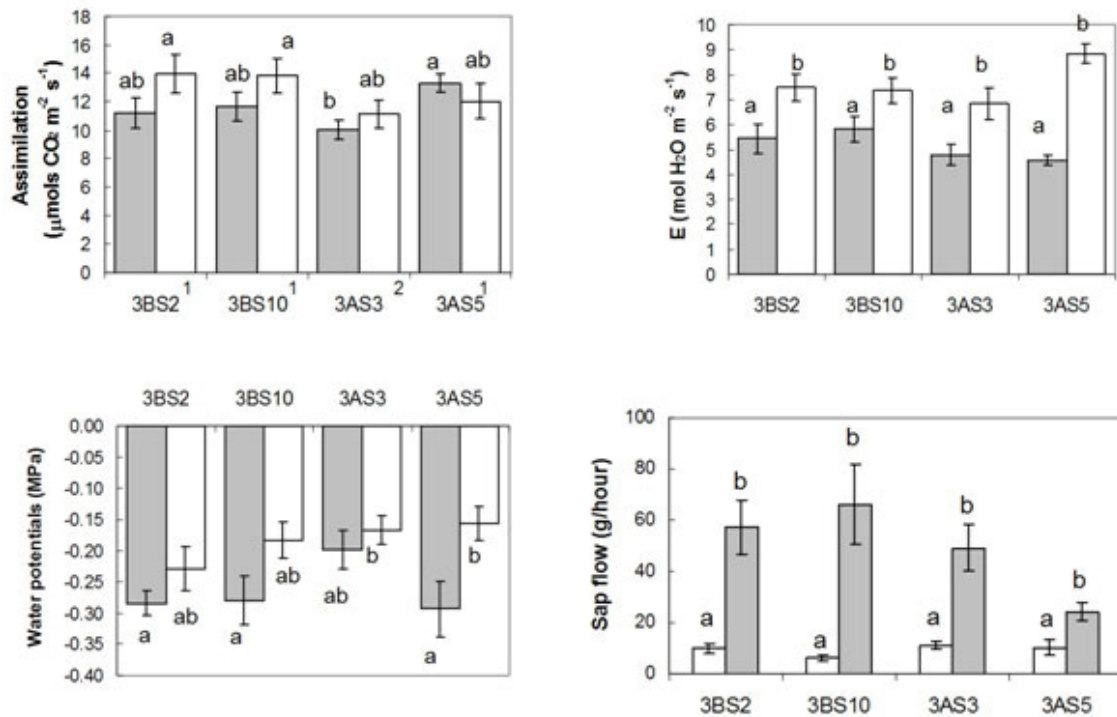

Figure 6-15. Average (± standard error) of carbon uptake, carbon transpiration, predawn water potentials, sap flow, and leaf nitrogen for the wet (▨) and dry (☐) seasons in the four islands. Letters by each bar indicate significance differences among islands while values above each bar indicate by-season interaction differences at p < 0.05 using a post-hoc Tukey honestly significant difference (HSD) test.
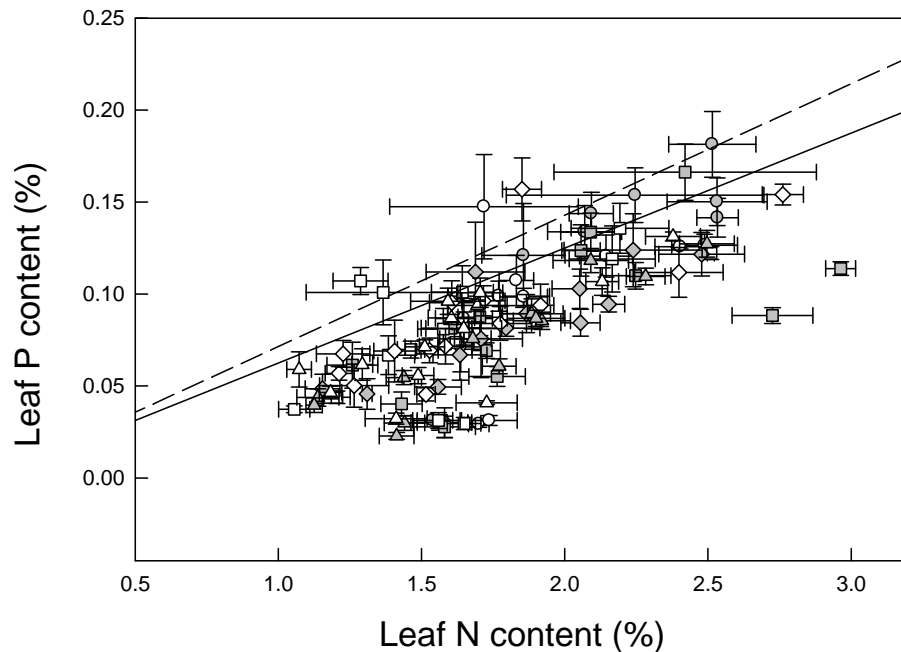
Figure 6-16. Relationship between foliar nitrogen (N) and phosphorus (P) contents for both seasons across all islands. Grey symbols represent the average values (± standard error of the mean) from the wet season while open symbols are for the dry season at 3BS2. Circles are for 3BS2, squares represent 3BS10, diamonds for 3AS3 and triangles for 3AS5. The dotted line indicates a nitrogen:phosphorus ratio of 14:1 while the solid line indicates a ratio of 16:1.

As the near-tails of tree islands are lower in elevation compared to the heads of the islands, it was predicted that these areas would be more susceptible to changes in water availability and flooding. Head to near-tail differences were observed for some of the parameters sampled although the trends were not consistent among the islands. For example, seasonal shifts were observed for E at both locations (heads and near-tails) of 3BS2 and 3AS5 but only at the near-tail of 3AS3; 3BS10 did not show significant seasonal shifts at either the head or near-tail. The N_{leaf} at the heads of the islands was higher than in the near-tail in the dry season but not in the wet season.

Conclusions

The findings from this work show that physiological parameters such as A, E, Ψ, and plant sap flow are sensitive to small seasonal changes in hydrology and can be used to assess in-situ instantaneous vegetation responses to changes in hydrologic conditions. Although most of the species in this experiment were bayhead species that are more flood tolerant relative to the upland hammock tree island species, most of the plants still demonstrated a significant response to flooding and dry-down conditions. This finding further reinforces the robustness of utilizing physiological parameters to obtain a snapshot of plant responses to their conditions despite an atypical hydrologic year.

ECOSYSTEM ECOLOGY

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Robert Shuford and Megan Jacoby

Ecosystem research continues to focus on areas of the Everglades severely impacted by nutrient enrichment. For two years, the focus of reporting has been on the design and implementation of two large-scale manipulations in the cattail zone of WCA-2A: Cattail Habitat Improvement (CHIP) and Accelerated Recovery on Impacted Areas projects (FIRE). In WY2008, CHIP focused on a detailed investigation of microbial communities and wading bird foraging in artificially opened areas, while FIRE focused on the chemistry of cattail and sawgrass (*Cladium jamaicense*) ash, and soil reduction-oxidation reactions (redox) to see if fire can be an effective management tool. In WY2009, CHIP focused upon a comprehensive analysis of how structure and function have been altered in artificially opened areas, while FIRE examined the physical and biological factors affecting cattail mass and nutrient loss after controlled burning. These two studies will allow the District to design a way to restore hydropatterns to WCA-3 and WCA-2A because, combined, these studies examine the costs and benefits of a cattail control program. A STELLA© model, used to evaluate soil-water total phosphorus (TP) exchange for impacted soils as a function of TP inflows, estimates how long it could take for these impacted soils to return to background levels of phosphorus if programs like FIRE and CHIP are not implemented.

CATTAIL HABITAT IMPROVEMENT PROJECT

CHIP is a large-scale in situ study comprising fifteen 6.25 hectacre (ha) plots to test the ability to rehabilitate cattail areas. The two primary objectives are (1) to assess whether creating openings in dense cattail areas will sufficiently alter trophic dynamics such that wildlife diversity and abundance is increased and (2) determine to what extent these created open areas' functions compare 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. Specifically, glyphosate was applied in May 2006 and the plots were burned July 20–July 21, 2006. The hypotheses, experimental design, and rationale behind this research project were described in Sklar et al. (2007) while the detailed project description and methodologies can be found at (www.sfwmd.gov/evergladeswatershed). This year's report (WY2009) focuses on an intensive sampling event that occurred during fall 2008 (late September through early November), just over two years since treatment was initiated. Additional parameters, which were not sampled within this exact sampling window (e.g., wading bird foraging and plant decomposition) are also included to provide a more comprehensive understanding of the functional changes that have been observed. For all results, sites are delineated based on their location; enriched (E), transitional (T), reference (UC) and whether or not they were burned, open (O), or controls (C). Specific hypotheses examined in this project are shown in **Table 6-8**.

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[5] TBE Group, West Palm Beach, FL

[6] Keith and Schnars, Ft. Lauderdale, FL

Table 6-8. Specific hypotheses evaluated for the Cattail Habitat Improvement Project.

PARAMETER	HYPOTHESIS
Structural Changes	
Microbial Community	There will be a change in microbial community composition in the open treatments compared to the emergent macrophyte controls due to differences in the quality of carbon, more submerged aquatic vegetation/periphyton in the open treatments versus more cellulose/lignin in the controls.
Periphyton Community	Increased abundance of periphyton in open versus control plots as dense vegetation is removed and light limitation is eliminated.
Aquatic Prey Community	Prey species richness and diversity will be highest at the moderately enriched treatment and lower at both the highly enriched treatment and reference area.
Functional Changes	
Periphyton	Periphyton productivity will increase in open plots in response to increased light availability. Increased periphyton productivity will result in elevated dissolved oxygen levels in the water column, resulting in increased ecosystem metabolism.
Decomposition	Open plots will have increased microbial respiration resulting in more rapid decomposition of plant material.
Prey Biomass	Herbivorous species (fish) will increase in open plots compared to controls, while detritivorous species (crayfish) will be reduced in open plots. The biomass of prey species will be greater with increasing nutrient enrichment.
Wading Bird Foraging	Wading bird foraging will increase in open plots in response to accessibility to prey species. Foraging will be positively related to nutrient enrichment.

Results

Environmental Conditions During the Project

Compared to previous years, deeper than average water depths were experienced during this sampling period, which likely contributes to differences observed during this event compared to the WY2008 report. The fall 2008 sampling event was heralded by a rapid spike in water column TP concentrations, concomitant with rewetting after the summer dry season (**Figure 6-17**). The TP increase was of the same magnitude as the spike following the initial burning of the plots.
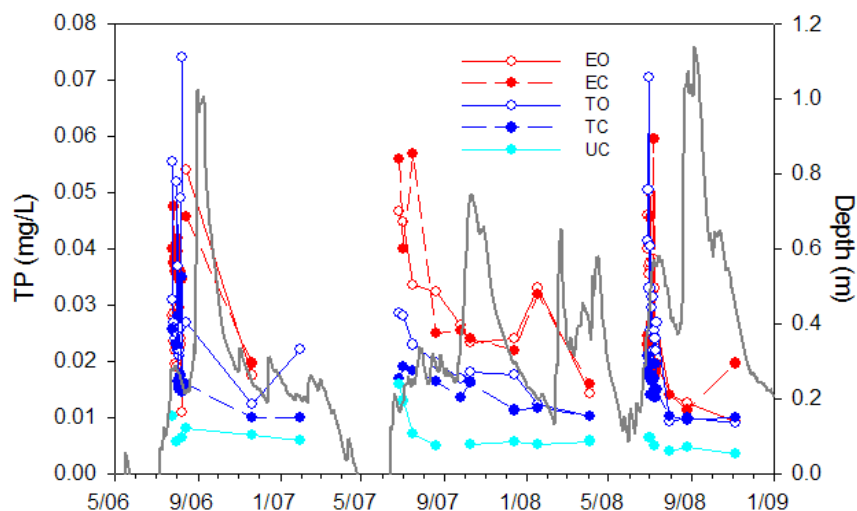
Figure 6-17. Surface water total phosphorus concentrations and water depths (gray line) during the project.

In general, higher water column TP values were associated with higher TP contents in the underlying floc and surficial (0–5 cm) soil, with significant differences between enriched control plots (EC) and enriched open (EO) plots (LSMeans, $p < 0.05$). Floc TP contents (mean ± 1 S.E.) averaged $1,160 \pm 15$, $1,717 \pm 160$, $1,384 \pm 378$, $1,223 \pm 87$, and 199 ± 14 milligrams per kilogram (mg/ kg), in EC, EO, transitional control (TC), transitional open (TO), and reference (UC) plots, respectively. Similarly, in the 0–5 cm soil fraction, TP contents averaged $1,217 \pm 80$, $1,400 \pm 35$, 911 ± 66, 986 ± 40, and 217 ± 26 mg/kg. Open areas also had significant changes in carbon content, with decreased carbon in open plots relative to controls; 449 versus 466 grams of carbon per kilogram (g C/kg) in EO versus EC, and 431 versus 457 g C/kg, in TO versus TC plots, respectively. These changes in floc carbon content are attributable to both changes in structural as well as functional attributes of the plots.

Community Composition

Microbial Community Composition. Periphyton, floc, and sediment were analyzed for microbial biomass and community composition using diagnostic phospholipid fatty acid (PLFA) biomarkers (Newman et al., 2008). Floc, sediments (0–5 cm), and periphyton samples were freeze-dried and shipped to Microbial Insights, Inc., for analysis.

Periphyton was abundant at UC sites and, due to high water conditions, was also abundant at the impacted control (EC and TC) sites. Abundance was lower at the open plots; only site TO2 had sufficient periphyton for collection and analysis. Total PLFA content was greatest at the EC and TC plots with average values of 2,954 and 1,794 nanomoles per gram (nM/g) of periphyton,

respectively. Total PLFA content in periphyton from TO2 was lower than that observed from control plots, but was greater than UC plots (854 versus 572 nM/g periphyton). The statistical analysis of the PLFA samples to date using NMDS and MRPP found that the microbial community was only significantly different between the UC sites and impacted sites ($p < 0.05$).

In both floc and sediments, PLFA content was greatest in open plots relative to controls. Floc biomass was five- to eight-fold greater than found in sediments, but was generally lower than periphyton PLFA biomass (by a factor of 1.5–3). All impacted sites had greater PLFA content relative to UC plots. In the floc, Gram negative bacteria were a greater community component in treatment plots, whereas Gram positive bacteria, sulfate reducing bacteria, and actinomycetes were a greater component of the control plots. Increased abundances of diatom and green algal PLFA in treatment plots is likely a result of increased light availability. UC plots had greater proportions of Gram positive bacteria, sulfate reducing bacteria, and diatoms relative to all other plots. In the sediment of open plots, though slight, greater proportions of fungi and algae were observed. In both floc and sediment, the metabolic status ratio of the Gram negative bacteria was lower in treatment plots relative to controls, indicating improved growth conditions. A reduction in Gram positive bacteria and increases in Gram negative bacteria and fungi reflect the improved oxic conditions, and both of these groups are effective degraders of complex organic matter.

The prokaryotes:eukaryotes ratio decreased in the floc and sediments at all treatment sites, a reflection of increased eukaryotic biomass in algal and fungal groups. The largest increase was observed at the EO sites where the ratio decreased from 38 to 25 in the floc and from 36 to 27 in the sediments. An increase in the abundance of eukaryotes was also observed at transitional treatment plots, but were less marked for floc (ratio decrease from 31 to 29) than in the sediments (35 to 25). Analysis of all floc samples to date show clear separation of microbial community composition between EO and EC sites ($p < 0.05$). Ordination plots also show a clear separation between the composition in TO and TC plots, though differences are only marginally significant ($p = 0.1$). All pair-wise comparisons between impacted and reference sites were significantly different ($p < 0.05$).

Vegetative Composition. The vegetative community in control plots remained similar in 2008 compared to previous years; cattail remained the dominant macrophyte in EC plots, followed by sawgrass, while the dominance was reversed in TC plots. In contrast, with the complete cessation of herbicide treatments, a dramatic increase in the abundance of musk grass (*Chara*) was observed in both EO and TO sites (**Figure 6-18**), likely in response to the elimination of herbicide management as well as continued light penetration following limited emergent macrophyte regrowth.
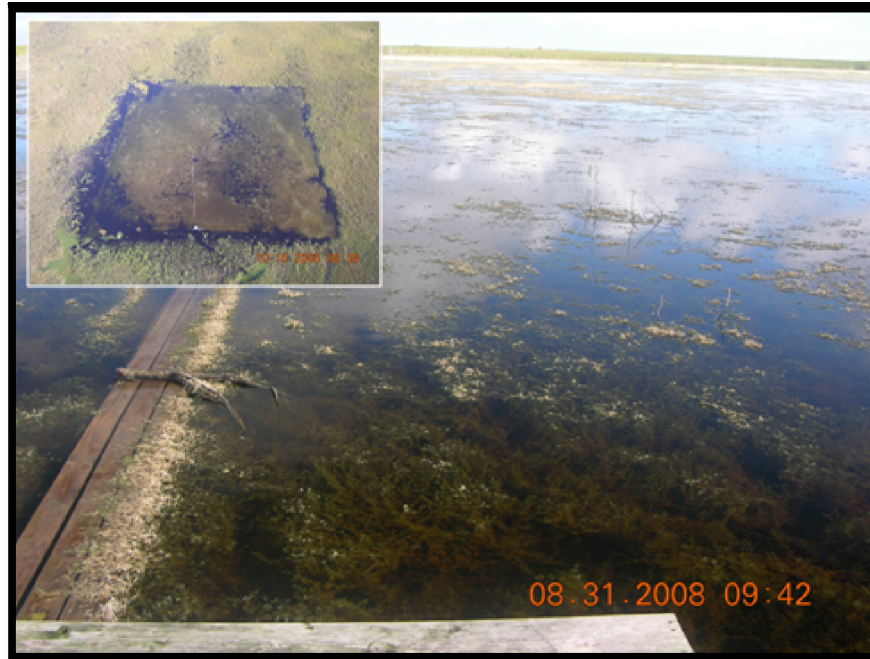
Figure 6-18. Photograph showing the dense coverage of submerged aquatic vegetation (musk grass/*Chara*) in an open plot during the fall 2008 sampling event. Inset shows the spatial extent of *Chara* throughout the entire plot (photo by the SFWMD).

Aquatic Prey Community Composition. Aquatic prey (fish and invertebrates) were sampled in November 2008 using a 1 m^2 bottomless pull trap (Sklar et al., 2008). The samples included 648 individuals across 12 species in the following order of abundance: mosquitofish (*Gambusia holbrooki*), grass shrimp (*Palaemonetes paludosus*), least killifish (*Heterandria formosa*), slough crayfish (*Procambarus fallax*), sailfin molly (*Poecilia latipinna*), bluefin killifish (*Lucania goodei*), Florida flagfish (*Jordanella floridae*), dollar sunfish (*Lepomis marginatus*), Everglades pygmy sunfish (*Elassoma evergladei*), golden topminnow (*Fundulus chrysotus*), redear sunfish (*Lepomis microlophus*), and Mayan cichlid (*Cichlasoma urophthalmus*). Mosquitofish, grass shrimp, least killifish, and crayfish accounted for 87 percent of all aquatic prey.

Diversity scores were calculated and tested for community differences using multivariate techniques in PC-ORD™. Mean species richness (S), and diversity (H) were highest in transitional plots (S = 5.7, H = 1.1), with fewer species found at enriched plots (S = 3.0, H = 0.7) and reference sites (S = 4.3, H = 1.3). To detect compositional differences among treatments, prey data were plotted using NMS ordination, followed by an MRPP. Ordination results indicated that control plots were distinct from open plots and reference areas. Using MRPP, only the comparison between EC and EO indicated significant differences in composition (MRPP, A 0.47, p = 0.02). These differences were very similar to those observed in the October 2007 samples. Indicator species analysis (ISA) revealed that EC plots supported least killifish, mosquitofish, and crayfish (mixed herbivore and detritivore), while EO plots supported bluefin killifish (herbivore). This is similar to observations in 2007 when the ISA results indicated that EC supported crayfish and larger-body fish, while EO supported shrimp and smaller-body fish.

Functional Changes

Microbial Respiration. Microbial respiration of floc samples (n = 3) was determined under both aerobic and anaerobic conditions. Anaerobic and aerobic respiration rates were obtained by incubating samples under either a pure nitrogen (N_2) or pure oxygen (O_2) headspace, respectively, for 24 hours at 25°C, and measuring the subsequent carbon dioxide (CO_2) released. Though not significantly different, aerobic respiration was generally higher in open compared to control plots, with mean CO_2 production rates of 752 ± 418, 472 ± 237, 951 ± 185, 231 ± 231, and 625 ± 314 μg CO_2–C/g/d (grams of carbon per day), for plots EO, EC, TO, TC, and UC, respectively. In contrast, no obvious trend was observed for anaerobic CO_2 production, whose means ranged from 583–1,188 μg CO_2–C/g/d.

Vegetative Biomass and Gross Primary Production. Emergent, floating, submerged aquatic vegetation (SAV), and periphyton were harvested from three 0.25 m^2 quadrates per plot in order to estimate aboveground biomass (g dry weight/m^2). Macrophytes were separated into live and dead material by species or type and dried at 60º C until a constant weight was obtained. Gross primary production estimates were derived from in situ gross photosynthesis estimates of SAV and periphyton using the light–dark bottle method to measure oxygen evolution (McCormick et al., 1998) and for emergent and floating macrophytes using an infrared gas analyzer (LCpro[+]; Dynamax, Inc.) to measure CO_2. Photosynthesis rates were adjusted to grams of carbon per grams of dry weight per day (g C/g dry weight/d) and then multiplied by live biomass to obtain areal gross primary production (g C/m^2/d).

Aboveground live vegetative biomass was significantly greater in EC plots compared to EO plots [**Table 6-9**; analysis of variance (ANOVA) F = 4.62; p < 0.023; Fisher's least significant difference (LSD) p = 0.008] and a similar pattern approached significance between TC plots and TO plots (Fisher's LSD p = 0.075). The significant, less pronounced response between the transition plots was attributed to the high biomass of musk grass (320 g dry weight/m^2) in TO plots. This contrasts from the UC plots which were composed mostly of periphyton.

Total gross primary production was significantly greater for control plots than open plots (**Table 6-9**; ANOVA F = 6.99; p < 0.006; Fisher's LSD p ≤ 0.05). Gross primary production was 10-fold greater for EC than EO and five-fold greater for TC than TO. In control plots, cattail was a greater contributor to gross primary production than sawgrass, accounting for 94 percent and 76 percent of the total for EC and TC, respectively. Gross primary production in EO and TO was regulated by musk grass, whereas UC plots were controlled by periphyton; however, despite the difference in composition, gross primary production was similar among EO, TO, and UC plots.

Table 6-9. Aboveground vegetative live biomass, areal gross primary production, and physicochemical conditions [mean ± standard deviation (S.D.)] for created openings (EO and TO), paired emergent macrophyte controls (EC and TC), and reference plots (UC) measured in October 2008, n = 3.

Species	Treatment				
	Enriched Control (EC)	Enriched Open (EO)	Transitional Control (TC)	Transitional Open (TO)	Reference (UC)
Live Vegetative Biomass (g dry weight/m^2)					
Typha spp.	696 ± 540	-	264 ± 171	-	-
Cladium jamaicense	173 ± 81	-	495 ± 461	-	-
Eleocharis spp.	-	-	-	-	17 ± 2.0
Nymphaea odorata	-	-	4.2 ± 6.3	-	5.8 ± 10
Utricularia foliosa	-	-	6.7 ± 12	-	-
Utricularia purpurea	-	-	-	-	0.2 ± 0.2
Chara spp.	-	103 ± 176	0.1 ± 0.1	315 ± 166	17 ± 13
Epipelon	-	-	-	-	17 ± 6.9
Epiphyton	5.7 ± 7.7	-	19 ± 13	5.5 ± 8.3	160 ± 62
Metaphyton	-	-	-	-	17 ± 24
Total	**874 ± 479**	**103 ± 176**	**789 ± 343**	**320 ± 171**	**234 ± 93**
Gross Primary Production (g C/m^2/d)					
Typha spp.	7.8 ± 5.1	-	7.1 ± 4.6	-	-
Cladium jamaicense	0.5 ± 0.4	-	1.8 ± 1.7	-	-
Eleocharis spp.	-	-	-	-	0.5 ± 0.1
Nymphaea odorata	-	-	0.4 ± 0.6	-	0.1 ± 0.2
Utricularia foliosa	-	-	<0.1	-	-
Utricularia purpurea	-	-	-	-	<0.1
Chara spp.	-	0.8 ± 1.3	<0.1	1.8 ± 0.9	0.3 ± 0.2
Epipelon	-	-	-	-	<0.1
Epiphyton	0.4 ± 0.5	-	<0.1	<0.1	0.31 ± 0.43
Metaphyton	-	-	-	-	<0.1
Total	**8.3 ± 4.7**	**0.8 ± 1.3**	**9.4 ± 3.6**	**1.8 ± 0.9**	**1.2 ± 0.3**
Physicochemical Environment					
Mean DO (mg/L)	1.6 ± 0.4	5.2 ± 0.3	1.8 ± 0.4	7.5 ± 1.3	8.8 ± 0.3
Maximum DO (mg/L)	2.2 ± 0.7	7.7 ± 0.9	3.2 ± 0.5	9.7 ± 3.2	10.1 ± 0.1
Minimum DO (mg/L)	1.0 ± 0.2	3.1 ± 0.1	1.0 ± 0.3	5.9 ± 0.8	7.6 ± 0.6
Mean Temperature (ºC)	19.1 ± 0.2	19.7 ± 0.1	19.7 ± 0.2	19.1 ± 1.4	20.2 ± 0.1
Maximum Temperature (ºC)	20.0 ± 0.4	22.2 ± 0.2	20.5 ± 0.3	20.9 ± 1.9	21.9 ± 0.2
Minimum Temperature (ºC)	18.3 ± 0.3	17.5 ± 0.1	19.0 ± 0.2	17.6 ± 0.8	18.8 ± 0.1
Mean pH	7.1 ± 0.1	7.5 ± 0.1	7.2 ± 0.1	7.6 ± 0.3	7.9 ± 0.2
Maximum pH	7.2 ± 0.1	7.7 ± 0.1	7.2 ± 0.1	7.9 ± 0.5	8.1 ± 0.2
Minimum pH	7.1 ± 0.1	7.4 ± 0.1	7.2 ± 0.1	7.5 ± 0.2	7.8 ± 0.2

Eleocharis spp. – spikerush
Nymphaea odorata – fragrant water lily
Utricularia foliosa – leafy bladderwort
Utricularia purpurea – eastern purple bladderwort
DO – dissolved oxygen

Physiochemical Environment. Dissolved oxygen (DO), temperature, and pH of the water column measured mid-depth was used to assess whether ecosystem function differed between treatments and among regions since the physiochemical environment strongly influences both biogeochemical reactions and the distribution of organisms. These defining parameters were measured at 30 minutes intervals using Hydrolab™ sondes deployed in each plot October 17–31, 2008. Significantly greater mean (ANOVA F = 94.4; p < 0.001; Fisher's LSD p ≤ 0.05), maximum (ANOVA F = 138.9; p < 0.001; Fisher's LSD p ≤ 0.05), and minimum (ANOVA F = 26.1; p < 0.001; Fisher's LSD p ≤ 0.05) DO levels were observed for the musk grass-dominated open (EO and TO) plots compared to emergent macrophyte-dominated control (EC and TC) plots. These differences were also reflected in the diel patterns (**Figure 6-19**). Mean DO for EO and TO were above 5.0 milligrams per liter (mg/L) but were slightly lower than the UC plots. DO for the control plots average less than 2.0 mg/L. Mean temperature did not differ significantly (ANOVA F = 2.3; p = 0.1281) between open and control plots; however, temperature varied more within open than control plots as evidenced by lower minimum and higher maximum temperatures (**Table 6-9**). Compared to the control plots, the open plots had significantly higher values of mean pH (ANOVA F = 12.6; p = 0.0010; Fisher's LSD p ≤ 0.05), maximum pH (ANOVA F = 10.6; p = 0.0019; Fisher's LSD p ≤ 0.05), and minimum pH (ANOVA F = 11.2; p = 0.0015; Fisher's LSD p ≤ 0.05). Although pH varied more in the open plots, pH in all plots remained circumneutral and were lower than UC conditions (**Table 6-9**). These results suggest that structural changes resulting from the removal of emergent macrophytes significantly altered the physicochemical environment likely caused by the increase in SAV photosynthesis.
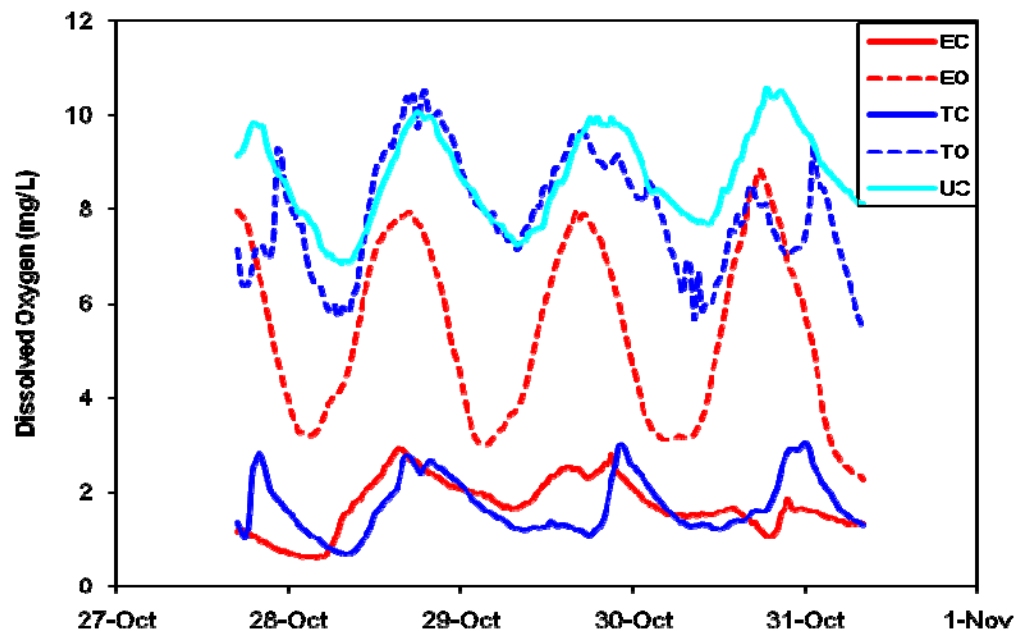

Figure 6-19. Mean dissolved oxygen diel profiles for the created openings (EO and TO), paired emergent macrophyte controls (EC and TC), and reference plots (UC).

Decomposition. Decomposition is a key controlling factor in peat accumulation. Both above- and belowground decomposition rates were estimated using litter bags containing standing dead cattail shoots and roots, respectively. Aboveground bags were deployed in December 2006 and belowground bags a year later in January 2008. Aboveground samples were collected every six months, treatment effects were first observed after one year, and continued through the following 18-month sampling, with open plots showing greater mass loss than controls (**Figure 6-20**). After 24 months, litter bags from EC and TC plots averaged 22 and 34 percent of their initial mass, respectively. In contrast, only 8 and 11 percent of initial mass remained in TO and EO plots. However, treatment effects were no longer significantly different, likely due to the low quantities of material remaining and their associated higher variability. At the UC sites, decomposition was relatively slow with 74 percent of material remaining after 24 months. This was probably due to continued low phosphorus conditions combined with low nutrient contents of the plant material.

Unlike aboveground litter, roots did not show significant treatment or nutrient effects on their mass remaining during the initial 12 months of burial in the plots, with 47 to 59 percent of mass remaining (**Figure 6-20**).
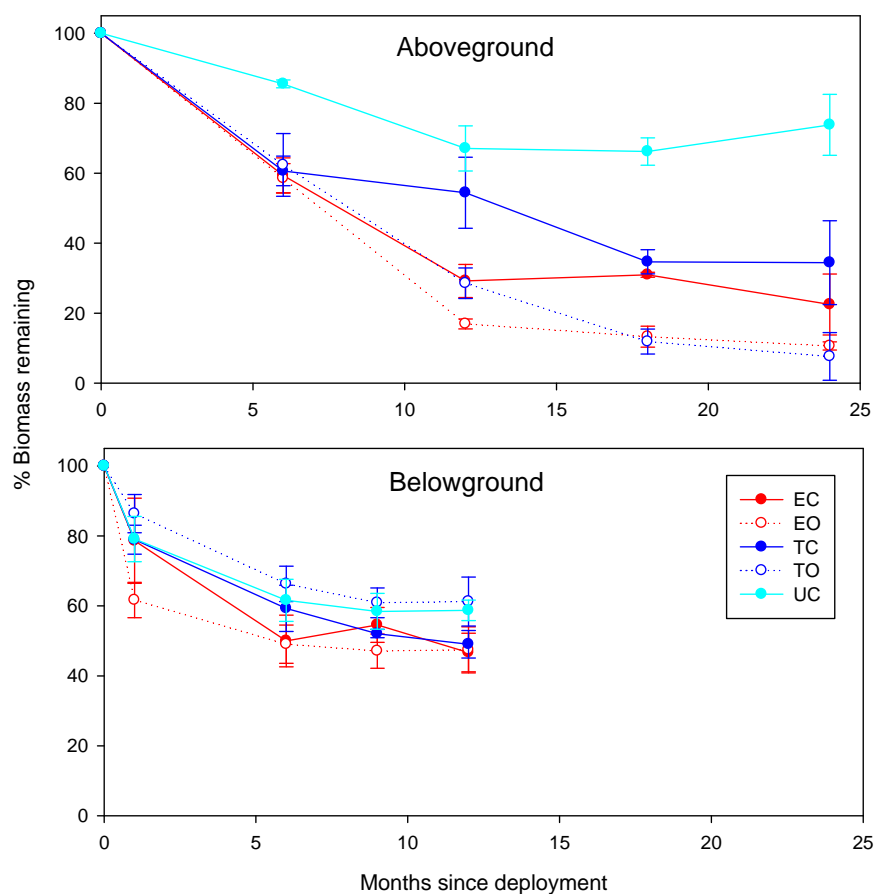
Figure 6-20. Percent mass remaining in litter bags collected from the created openings (EO and TO), paired emergent macrophyte controls (EC and TC), and reference plots (UC). Aboveground litter bags contained cattail (*Typha* sp.) litter, deployed at the soil surface, and belowground bags containing cattail roots which were buried in the soil.

Prey Biomass. Prey responses were examined using one-way ANOVA models with biomass of fish and crayfish as the response variable and site (EO, EC, TO, TC) as the predictor variable. High variability precluded significant differences in herbivore (fish) ($F_{3,11} = 0.26$, $p = 0.85$) or detritivore (crayfish) ($F_{3,11} = 2.34$, $p = 0.15$) biomass between sites, matching results from October 2007. Thus, contrary to predictions, neither treatment nor enrichment affected prey biomass. However, as noted previously, significant species differences were observed.
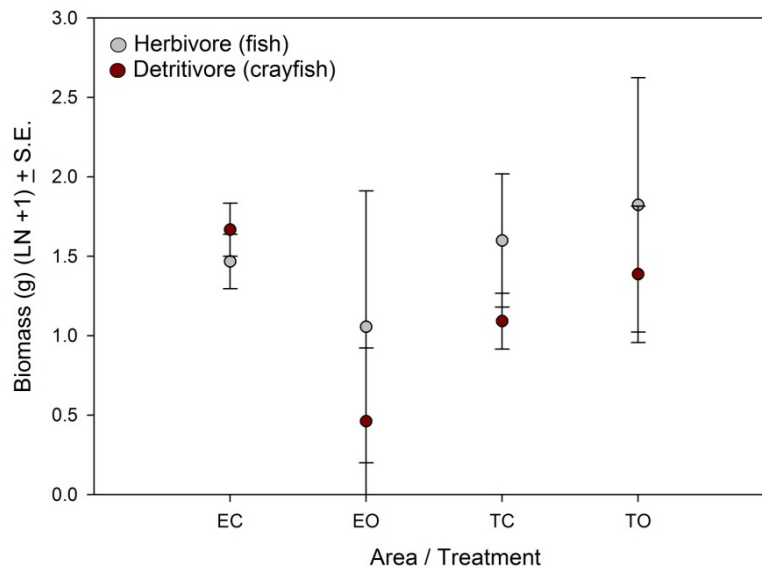
Figure 6-21. Prey biomass collected from the created openings (EO and TO) and paired emergent macrophyte controls (EC and TC) in November 2008.

Wading Bird Foraging. Wading birds were surveyed weekly by helicopter from November 4, 2008–March 27, 2009. A total of 3,664 individuals of 11 species were counted in decreasing order of abundance: white ibis, glossy ibis (*Plegadis falcinellus*), little blue heron (*Egretta caerulea*), tricolored heron (*Egretta tricolor*), snowy egret, great egret, great blue heron (*Ardea herodias*), black-crowned night-heron (*Nycticorax nycticorax*), wood stork, roseate spoonbill, and limpkin (*Aramus guarauna*). White ibis accounted for almost half the observed birds.

Water depths were suitable for wading bird foraging (5–25 cm) in enriched and transitional plots from early December 2008 through January 2009. During this period, considerably more birds were observed in open plots than control plots (all species combined), and average densities in the openings were generally high, often averaging over 50 birds per plot (**Figure 6-22**). Foraging densities in the 2008/2009 wading bird nesting season were generally less than the previous season, which often exceeded 100 birds per plot, peaking at greater than 200 birds per plot. Despite the lower densities, the current results mirror those from the 2007/2008 season, and agree with the prediction that created openings would attract greater wading bird foraging.

Observations also provided some support for the prediction that foraging would be positively related to nutrient enrichment. During both the 2007/2008 and 2008/2009 seasons, considerably more birds were observed in transitional and enriched plots than in reference plots; however, during 2008/2009 there was a tendency for more foraging in the transitional plots than enriched plots. This suggests that foraging is positively related to nutrient enrichment at a relatively broad scale, i.e., between enriched and unenriched areas, but that other unknown factors may be

affecting foraging within the enriched region. The significance of this pattern is unclear at this point, particularly given that prey biomass was similar in transitional and enriched plots.

The total period that birds spent foraging in open plots in 2008/2009 (11 weeks) was similar to that of 2007/2008 (12 weeks). The temporal availability of foraging in the open plots appears to be considerably longer than that in the natural system where large foraging flocks at any one place rarely remain for more than a week (Cook, personal observation).
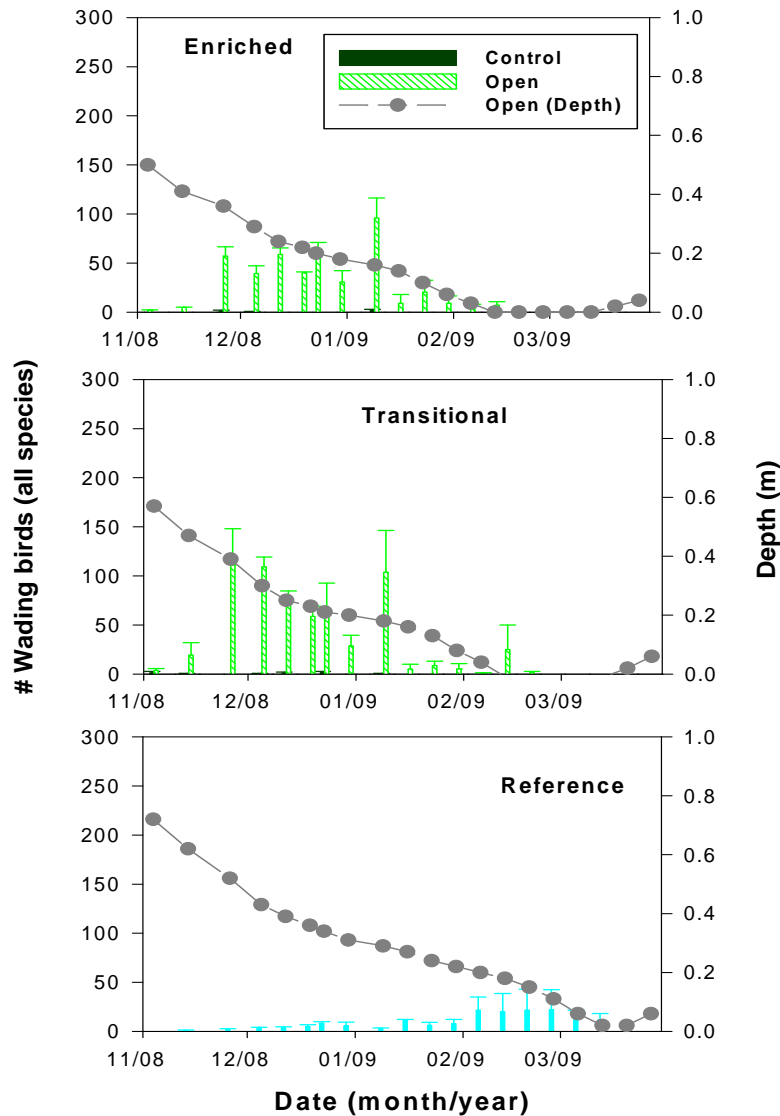
Figure 6-22. Wading bird numbers and water depths in Cattail Habitat Improvement Project plots from November 2008 through March 2009 (mean ± 1 S.E.).

Conclusions

Intensive sampling shows that two years after creating openings within densely vegetated areas, significantly beneficial ecological changes, i.e., habitat improvement, to the structure and function of these areas have been sustained. Specifically:

- PLFA analyses of the floc indicated significant changes to the microbial community structure, with greater overall biomass of microorganisms, especially eukaryotic organisms, in the open plots.

- The changes in microbial communities and metabolism in the created open areas, in addition to the appearance of SAV, increased the oxygenation of the plots, and fostered the sustained return to diel O_2 cycles, driven by high net primary productivity. This switch likely attributed to the increased carbon cycling, increased decomposition and reduced carbon content observed in open plots.

- Wading bird foraging was higher in open plots compared to controls, and of longer duration than that observed in the natural system.

In summary, the sustained elimination of emergent macrophytes decreased the total areal biomass and gross primary productivity compared to control plots. As a result, ecosystem function was fundamentally altered by shifting photosynthesis from a dominance by terrestrial type vegetation (i.e., emergent macrophyte) to aquatic vegetation (SAV and periphyton). This shift caused dramatic changes in aquatic carbon cycling and oxygen dynamics with important consequences for ecosystem processes like food web dynamics, nutrient cycling, and decomposition. Continued analysis of data collected over the remaining year of the project will provide further evidence of the temporal trends and longer-term sustainability of these responses. Additionally, nutrient budgets that allow for a more detailed analysis of carbon cycling still are under development.

ACCELERATED RECOVERY OF IMPACTED AREAS: FIRE IMPACTS ON MASS AND NUTRIENT LOSS AND ASH CHEMISTRY

The FIRE Project is a research component of the Long-Term Plan designed to evaluate prescribed fires as a management tool for accelerating the recovery of phosphorus-impacted areas of WCA-2A. Fire shaped the historic Everglades landscape, possibly playing a role in the creation of the ridge-and-slough topography and sawgrass-dominated areas (Loveless, 1959; Wade et al., 1980; Gunderson, 1994). Nutrient loading to the area resulted in a species shift from sawgrass-dominated communities to cattail monocultures. This project employs a Before-After-Control-Impact-Paired Series (BACIPS) design (Sklar et al., 2008) to evaluate the impact of fire under different nutrient enrichment conditions. The objectives of this project are to:

- Understand fire impacts on the structure, function, and ecological processes in soil, water, and vegetation in the Everglades.

- Assess the use of repeated prescribed fires to accelerate ecosystem recovery from nutrient enrichment as demonstrated by a species shift from cattail-dominated communities to more historic Everglades plant communities.

The project has three overall hypotheses:

- Fire will stress cattail, with the degree of stress depending on burn timing and environmental conditions.

- Frequent multiple fires will accelerate the decline of cattail communities through repeated disturbance and enhance recovery of native species adapted to fire, such as sawgrass.

- Multiple fires will decrease soil phosphorus availability by burying peat with a high phosphorus content under detritus with a low phosphorus content.

Because field sampling is ongoing, the 2010 SFER focuses on fire's effects on mass (live and dead leaves) and nutrient loss.

Project Milestones

The project was initiated in 2005 and will conclude in WY2010. Six plots were established in three areas of WCA-2A using a modified BACIPS design. Two plots were established in each of three areas: highly enriched, moderately enriched, and unenriched. Two fires were implemented at each of two plots, one in the highly enriched area and one in the moderately enriched area, with several samplings of each plot before and after burning. The other plot in the area served as a control, and the plots in the unenriched area served as a reference. Approximately 20 sub-studies were conducted to capture the impact of accelerated and natural recovery on major ecosystem processes including water, soil, periphyton, and vegetation nutrient and biomass dynamics. The first set of fires was implemented in 2006 and the second set of fires was conducted in summer 2008 (about two years apart). While the project largely focuses on field sampling, the team has been conducting preliminary data analysis to improve experimental designs of large-scale ecosystem studies. The project illustrated various non-replicated designs including Before/After (BA), Control/Impact (CI), Reference/Impact, BACI, and BACIPS designs and multiple temporal sampling schemes to account for the spatial and temporal structure of the ecosystem (Miao et al., 2009).

In addition to focusing on experimental design, the research team also published several papers on specific ecosystem processes. For instance, fire impacts on seed bank dynamics were examined and it was found that cattail seed germination was negatively affected by fire regardless of the timing of fire, while sawgrass seed germination was not significantly affected (Miao and

Zou, 2009). Dissolved inorganic carbon (DIC) in surface water after the first fire at the highly enriched plot analyzed and ash addition to surface water increased pH, decreasing the CO_2 portion of DIC in the short term, and periphyton sequestered CO_2 in the longer-term (Gu et al., 2008). Further, ash chemistry has been shown to be a function of fire temperature, plant species, and habitat nutrient enrichment (Qian et al., 2009b). In addition, soil redox potential was strongly controlled by water depth, with redox potential at approximately -200 millivolts (mV) when water depth was $\geq$ 5 cm above peat surface and exponentially increasing to +600 mV as the water table fell below the peat surface (Thomas et al., 2009). Soil redox potential was also higher in the moderately enriched area than either the highly enriched or unenriched areas, but was not different between cattail- and sawgrass-dominated plant communities.

Current Findings

Mass and Nutrient Loss

Fires in nutrient enriched areas of the Everglades resulted in 735 g/m^2 of aboveground mass loss (**Table 6-10**), which means 44 percent of pre-fire mass was consumed by fire. Pre-fire mass was higher before the first fire at both plots than before the second fire (p = 0.03). Also, the proportion of dead leaves to live leaves was higher before the first set of fires than the second set of fires, indicating the fuel load was greater (**Figure 6-23**). The amount of mass loss through burning was over twice as high as measured by both g/m^2 (p < 0.01) and percent of pre-fire mass (p = 0.05) during the first set of fires than the second set of fires. However, there was no significant difference in mass loss between the highly enriched and moderately enriched plots. Carbon loss followed very similar patterns to mass loss.

Table 6-10. Mass and nutrient loss from multiple fires at two wetland plots with different levels of nutrient enrichment. To calculate the averages, the total number of samples were averaged for each grouping.*

	Pre-fire mass (g/m²)	Mass Loss (g/m²)	Carbon Loss (g/m²)	Nitrogen Loss (g/m²)	Phosphorus Loss (mg/m²)	Water Depth (cm)
Overall	**1,681**	**735**	**351**	**7**	**366**	**29**
Nutrient Enrichment						
Highly enriched	1,778	952	449	11	641	27
Moderately enriched	1,609	573	252	5	159	27
Fire Frequency-Enrichment						
Highly enriched						
1st Fire	2,143	1,294	585	11	673	12
2nd Fire	1,412	610	379	10	608	43
Moderately enriched						
1st Fire	1,911	926	424	10	146	22
2nd Fire	1,508	456	195	3	164	32

*For the overall group, n = 14 sampling events; for the highly enriched group, n = 6; for the moderately enriched group, n = 8; for the highly enriched 1st and 2nd fires, n = 3 for both; for the moderately enriched 1st fire, n = 2, and for the moderately enriched 2nd fire, n = 6.
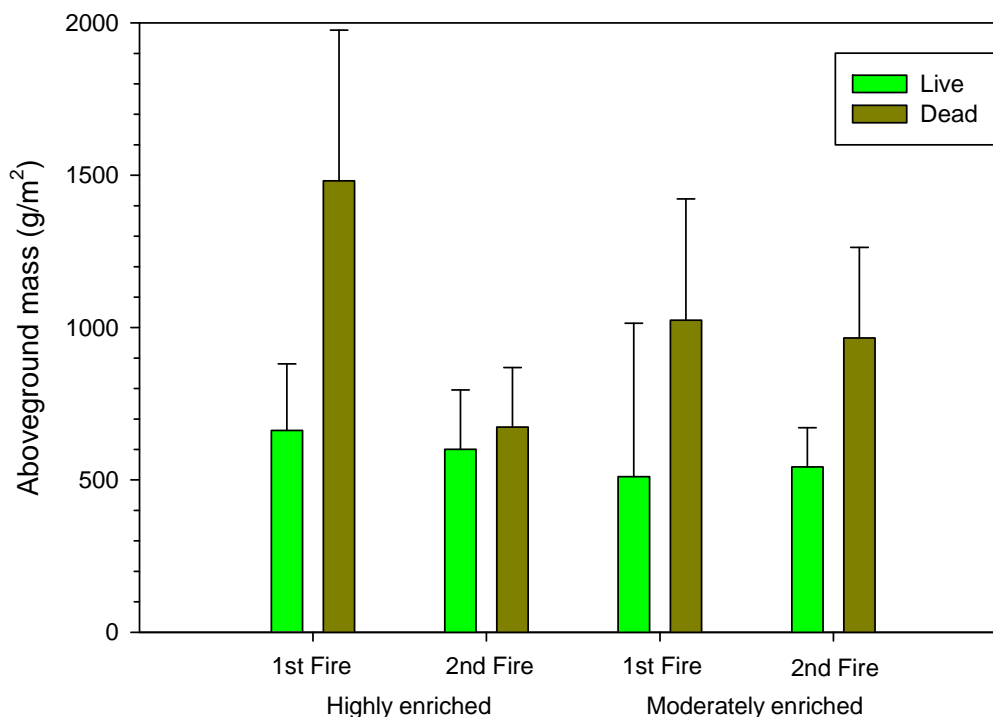
Figure 6-23. Aboveground live and dead leaf mass before each fire.
Error bars are one S.D.

Nitrogen loss on average was 7 g N/m^2 during a fire (**Table 6-10**) accounting for 52 percent of pre-fire nitrogen content. Pre-fire nitrogen content was higher before the first fire than the second fire (p = 0.01) and the loss was over twice as high during the first set of fires than the second set of fires. Nitrogen loss differences were significant for both g/m^2 (p < 0.01) and percent loss (p = 0.05). However, nitrogen loss was not different between plots.

Phosphorus loss on average was 366 mg P/m^2 during fire, accounting for 45 percent of pre-fire phosphorus content. Phosphorus patterns differed from mass, carbon, and nitrogen in that pre-fire phosphorus content and phosphorus loss were higher at the highly enriched plot than the moderately enriched plot (p < 0.01), but were not different between fires. Phosphorus loss was more than four times higher at the highly enriched plot than the moderately enriched plot in g/m^2 (p < 0.001) (**Table 6-10**), but the percent of phosphorus loss was not different between plots. This was a factor of higher phosphorus concentrations in live and dead leaves at the highly enriched plot than at the moderately enriched plot with similar levels of mass loss between the plots.

Factors Affecting Mass and Nutrient Loss: Pre-fire Mass and Water Depths

Pre-fire mass and nutrient content were compared to mass and nutrient loss. Live leaf mass was similar between plots and fires (**Figure 6-23**). Therefore, higher pre-fire mass was the result of higher litter, which has a greater potential for combustion than live leaves. Pre-fire mass was a significant determining factor of mass loss during fire (**Figure 6-24**), accounting for almost 65 percent of loss variability. Carbon loss was also significantly related to pre-fire mass, but nitrogen and phosphorus loss was not.
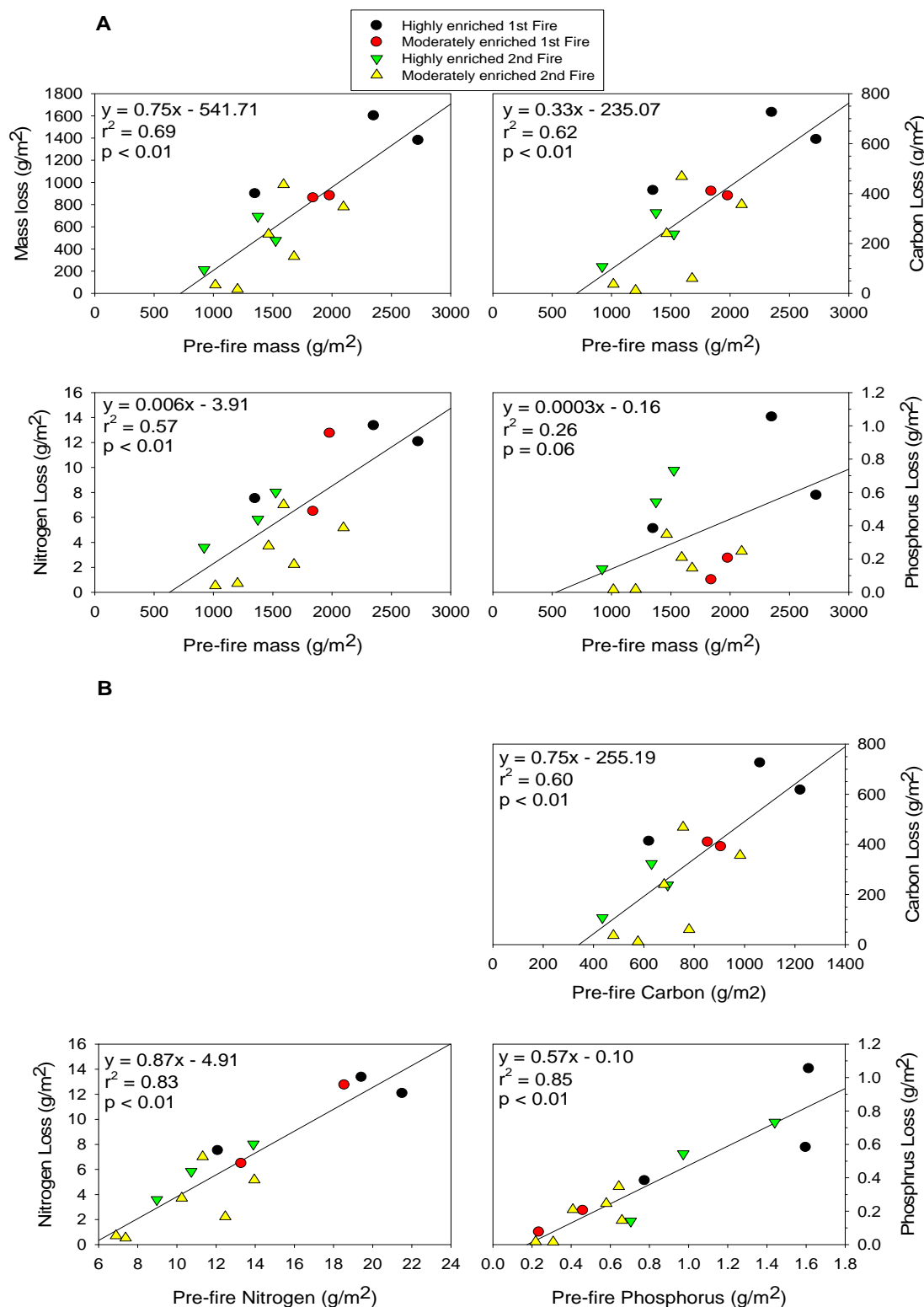
Figure 6-24. Aboveground vegetation mass and nutrient loss in relation to pre-fire mass and nutrient content for a highly enriched and a moderately enriched plot for multiple fires. (A) Relationship of mass and nutrient loss to pre-fire mass. (B) Relationship of nutrient loss to pre-fire nutrient content.

Instead, phosphorus loss was significantly related to pre-fire phosphorus content (**Figure 6-24**), which was a function of leaf concentration. As there was less mass during the second set of fires, carbon and nitrogen loss were less, while phosphorus loss was the same between fires, but less at the moderately enriched plot than the highly enriched plot for both fires.

Water depth also played an important role in determining mass loss (**Figure 6-25**) and explained 27 percent of mass loss variability. When water levels were low, more mass was exposed to flames allowing for greater combustion of leaves and, potentially, a hotter fire. During the first set of fires, water depths were between 10 and 22 cm above the peat surface and combustion averaged 54 percent of total aboveground mass. However, during the second set of fires, water depths were between 25 and 46 cm above the peat surface, and combustion averaged 31 percent of total aboveground mass.
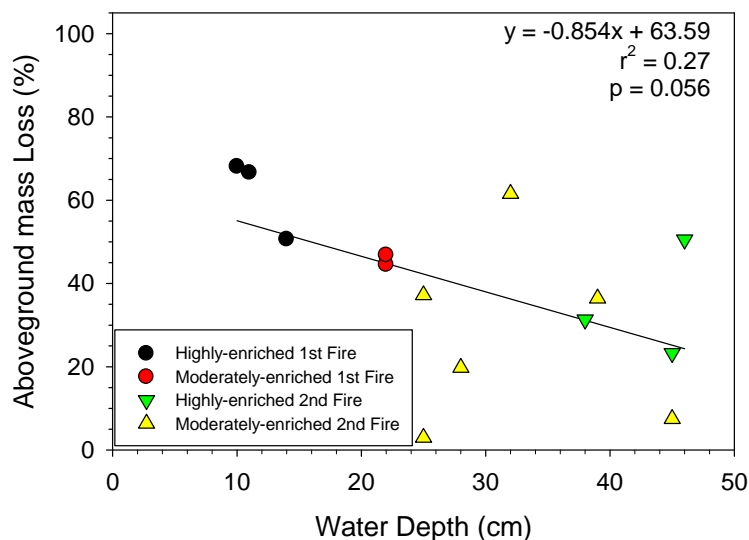
Figure 6-25. Mass loss for two sets of fires at a highly enriched and moderately enriched plot with different water depths.

Factors Affecting Mass and Nutrient Loss: Fire Temperature

To explore the effect of fire temperature on mass and nutrient loss, a laboratory simulation was conducted. Cattail and sawgrass live and dead leaves from the different plots were combusted at a range of temperatures. Relationships between fire temperature and nutrient and mass loss as well as ash chemistry were then developed and compared to field results.

Cattail and sawgrass ash total nitrogen (TN) and total carbon (TC) concentrations decreased with increasing temperature, while total phosphorus (TP) showed complex variations with temperatures from 150–550°C. Almost 99 percent of TN and TC were lost between 450 and 550°C, while 30–70 percent of TN and 60–90 percent of TC losses occurred between 150–350°C (Qian et al., 2009b). Field data showed greater variability than lab data, but followed a similar pattern.

As there were deviations from a linear relationship between percent mass and nutrient loss and temperature, nonlinear models were created to determine mass, carbon, and nitrogen loss based on fire temperature (**Table 6-11**) (Qian et al., 2009b). These equations predict that complete volatilization of carbon and nitrogen will occur with temperatures greater than 490°C.

Table 6-11. Models used to predict mass and nutrient loss (Y) as a function of temperature (X) during fire. Cattail and sawgrass (*Cladium jamaicense*) data were combined when developing these models. (Adapted from Qian et al., 2009b).

	Parameter				
	Model	**a**	**b**	**r^2**	**p**
Mass loss	$Y = a * X^b$	0.07	1.16	0.84	< 0.01
TN loss	$Y = a * X^b$	0.01	1.46	0.83	< 0.01
TC loss	$Y = a * X^b$	0.11	1.10	0.86	< 0.01

The laboratory results of cattail and sawgrass leaf combustion were similar to field results. During combustion, the loss of TN and TC had a significant linear relationship with mass loss ($p < 0.01$ for all correlations) (Qian et al., 2009b). The slopes were near 1 indicating the TN and TC loss occurred primarily by volatilization during combustion.

Given the close relationship between laboratory and field loss rates, the temperature prediction models developed using the laboratory simulation were applied to the field loss rates. For each fire at each site, the mass, nitrogen, and carbon loss percentages were entered into the model equations and the results were averaged to obtain an averaged predicted temperature for each fire. Predicted fire temperatures ranged from 174°C to 324°C and were similar to field temperatures measured during the fires.

Factors Affecting Ash Chemistry: Fire Temperature

The temperature of the fire affected ash nutrient concentration and pH. As temperature increased, carbon decreased in concentration regardless of species or status of the leaves, and almost completely volatilized at temperatures around 450°C (**Figure 6-26**). Nitrogen concentration in ash increased up to 350°C with live leaves concentrating more nitrogen than dead leaves. Nitrogen then began to volatilize and almost all of it was volatilized at 450°C. Having a much higher volatilization threshold, phosphorus increased in concentration with increasing temperature. The concentrations of phosphorus in ash produced at higher temperatures were greater for live leaves than dead leaves for both species. The pH of ash also increased with increasing temperature (**Figure 6-27**). The increase in pH can be attributed to base cation release during combustion (Qian et al., 2009) and may have important ecological impacts. Ash deposition into surface water caused TP concentrations to increase between 250 to 500 percent of pre-fire concentrations and pH to spike from 7.0 to as high as 8.4. However, these water quality changes lasted less than two weeks.
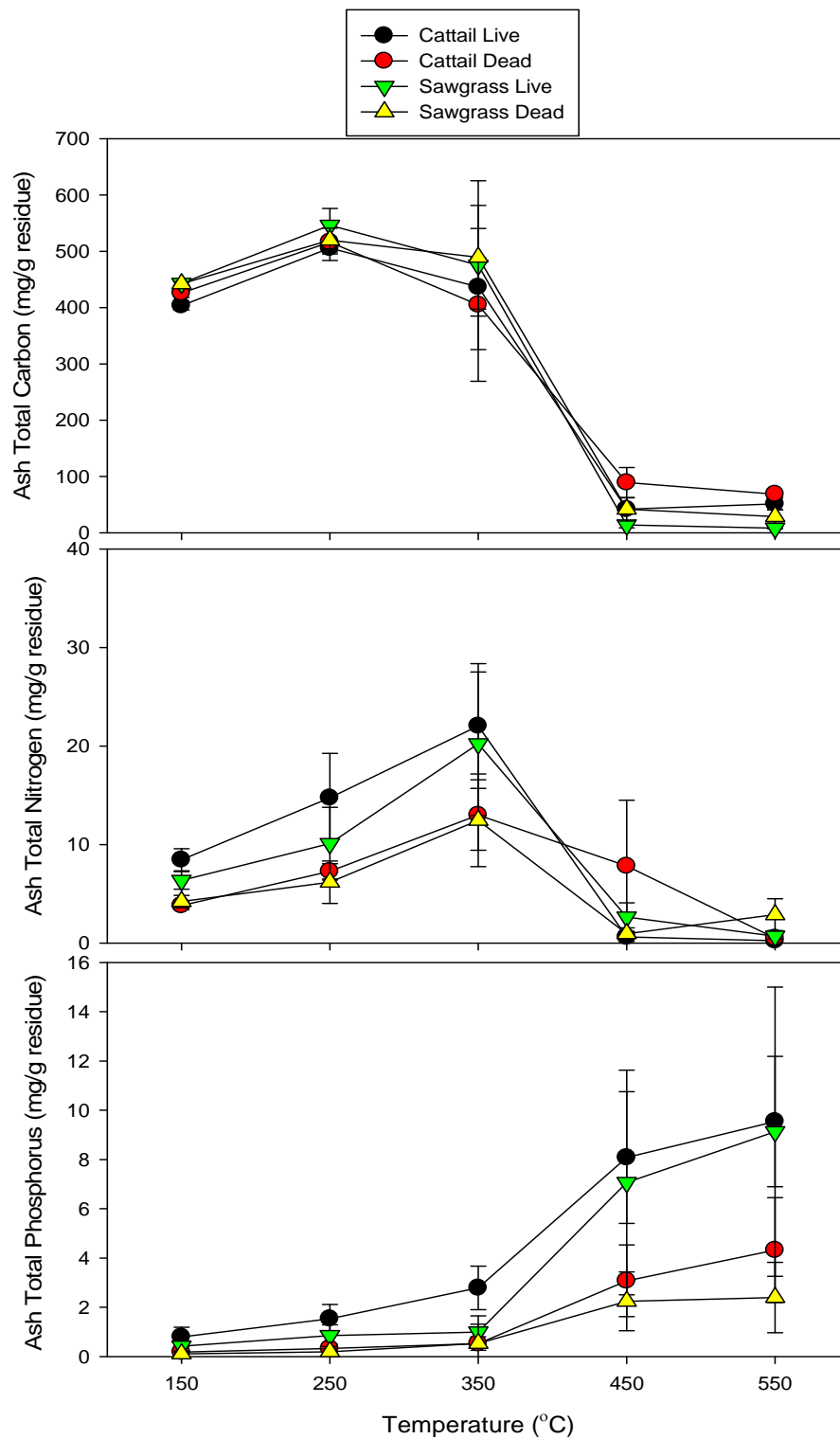
Figure 6-26. Ash nutrient concentration at different temperatures for live and dead leaves of cattail and sawgrass from the highly enriched and moderately enriched plots. Error bars are one standard deviation. (Modified from Qian et al., 2009a.)
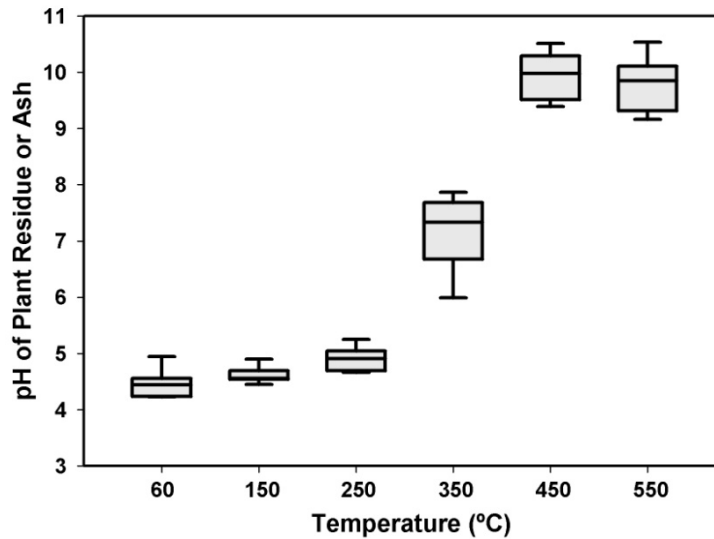
Figure 6-27. Ash pH at different fire temperatures. Each box included data for live and dead leaves of cattail and sawgrass plants from the different nutrient enrichment areas. The solid lines within the box represent medians; the lower and upper whiskers are 10th and 90th percentiles, respectively; and the lower and upper boundaries of the boxes are 25th and 75th percentiles, respectively. (Adapted from Qian et al., 2009a.)

Conclusions and Management Implications

The present study demonstrated that fire acted as a mass and nutrient removal pump for wetlands with almost 40 percent of aboveground mass and nutrients removed by fire. Mass, carbon, and nitrogen loss were over twice as high during the initial set of fires than the second set of fires, but was not different between the two plots, while phosphorus loss was 3.5 times higher at the highly enriched plot than the moderately enriched plot, but was not significantly different between fires.

Mass and nutrient loss were functions of pre-fire mass and nutrient contents as well as water levels and fire temperature. With a two-year fire interval, dead litter, the major source of fuel, was in greater quantity during the initial set of fires than the second set of fires, and therefore a greater percentage of mass was available for combustion during the first fires. Nitrogen loss was a function of pre-fire mass, and therefore was higher during the first set of fires, but was not different between plots. Phosphorus loss was a function of leaf concentrations, and therefore was higher at the highly enriched plot than at the moderately enriched plot, but was not different between fires.

Fire temperature determined ash chemistry. As temperature increased, both carbon and nitrogen were volatilized in greater quantity, producing low concentrations in the resulting ash. Phosphorus, on the other hand, increased in concentration with increasing temperature, as did ash pH. Ash deposition has been noted to significantly increase surface TP concentration and pH, but the effect was short-lived. Therefore, fire intensity during prescribed burns should be considered to meet management goals. A low-intensity fire will result in ash with low phosphorus concentration, while a high-intensity fire will produce ash with a high phosphorus concentration and high pH.

For long-term application of fire as a tool for reducing nutrients, fuel load and water levels are key considerations for maximizing nutrient loss. Conducting fires when fuel loads are high and water levels are around 10 cm above the peat surface will result in a hotter fire and increased nutrient loss. However, phosphorus loss is also a function of the nutrient gradient and phosphorus loss will be greatest in the highly enriched areas, as the leaves in these areas have higher phosphorus concentrations. Therefore, to maximize phosphorus loss, fires will be most effective in the highly enriched area when water levels are around 10 cm.

AN ASSESSMENT OF THE EFFECT OF PHOSPHORUS LOADING ON SOIL PHOSPHORUS CONCENTRATION

Loading of phosphorus throughout the Everglades has been shown to cause ecological impacts, as well as increases in soil phosphorus. Elevated soil phosphorus, especially in the vicinity of nutrient-rich inflows, may persist long after reductions in external loading. These enriched regions are capable of promoting prolific growth of nutrient-adapted communities such as cattail, as well as serving as a source of phosphorus to the water column (Miao and DeBusk, 1999; Fisher and Reddy, 2001).

The magnitude of phosphorus release to the water column, and thus the time required to restore soil phosphorus to pre-impact conditions, depends primarily on two factors: surface water phosphorus loading and the size of labile and non-labile soil phosphorus pools. A model of soil and water column phosphorus exchanges was developed by DB Environmental Labs, Inc., to estimate the time required to reduce the phosphorus-enriched soil within an impacted region in Water Conservation Area-2A (WCA-2A) to a background concentration of 500 mg/kg.

Modeling Approach

A conceptual model was developed that included three storage compartments and four flow paths (**Figure 6-28**).

Phosphorus was hypothesized to exist in three main pools:

- surface water
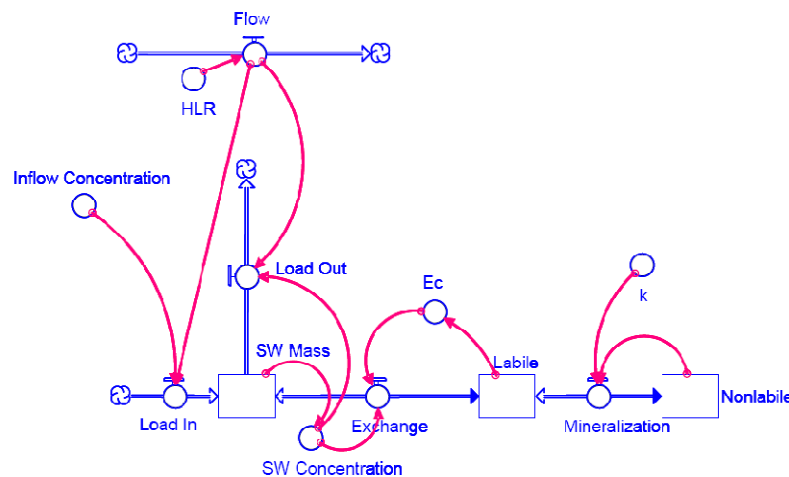- labile soil and biomass pool
- non-labile soil pool

Figure 6-28. Stock-and-flow model of pools and exchanges of phosphorus between soil and overlying water. (Note: E_c = the variable exchange coefficient; HLR = hydraulic loading rate; SW = surface water; k = non-labile rate constant.)

A dynamic model of phosphorus transfers was developed to estimate temporal changes in these three pools sizes, as well as predict downstream surface water phosphorus concentration. One expected outcome of this modeling effort was an estimate of the time required to reach a specific soil phosphorus goal, such as the 500 mg/kg soil goal proposed in 2007 for the Everglades by the U. S. Environmental Protection Agency (2007).

Field data were collected from two sets of enclosures over four wet seasons at the DB Environmental Reflux site in the impacted zone of WCA-2A (Community Watershed Fund and DB Environmental, Inc., 2009). This dataset provided phosphorus loading and the net phosphorus release relationships used to characterize the relationship between surface water phosphorus concentration and soil/biomass phosphorus release.

Inflow and outflow water quality samples were collected at selected times from 2005–2008. The difference between outflow and inflow phosphorus was used to calculate soil phosphorus release (reflux), while inflow phosphorus concentration and hydraulic loading rate were used to calculate phosphorus loading to the enclosures. Wet season averages of TP concentration and reflux were used to develop a relationship between inflow TP concentration and reflux in Equation 1:

$$Reflux\ (mg\ P/m^2/day) = -0.06 * inflow\ TP\ concentration\ (\mu g\ P/L) + 3.88\ (r = 0.91)\ (1)$$

This relationship implies that for every 10 µg phosphorus per liter (P/L) decrease in inflow phosphorus concentration, there is a 0.60 mg/m^2/day (0.22 g/m^2/yr) increase in biomass/soil phosphorus release.

This relationship constituted the principal soil-water column exchange mechanism for a simple stock-and-flow model of phosphorus pools and exchanges (**Figure 6-29**). The model was developed in STELLA®, version 9.1.1 (ISEE Systems). Boxes within the model represent stocks, or mass of phosphorus in specific pools.

Overall, the model depicts two soil phosphorus pools (labile and non-labile), a surface water pool, and inflow and outflow of phosphorus in the water column. Calculation of water column concentration was accomplished by including a hydrologic sub-model, called "Flow".
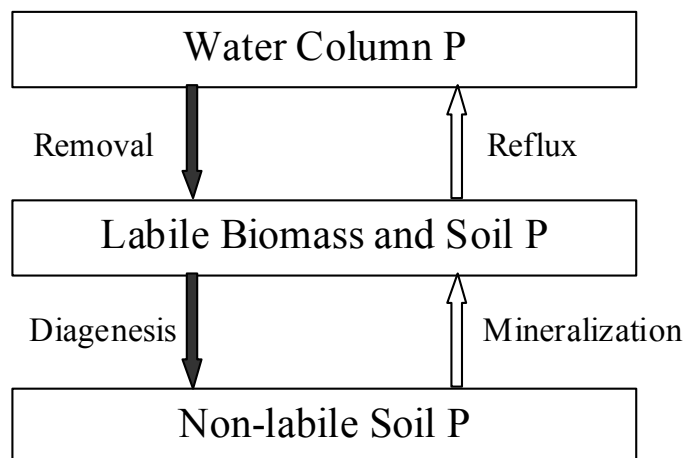

Figure 6-29. Schematic diagram of significant phosphorus (P) storages and flow paths. The downward-pointing arrows are the prevalent phosphorus flow paths during high phosphorus loading whereas the upward-pointing arrows are the prevalent phosphorus flow paths during low phosphorus loading, such as what is occurring now after Stormwater Treatment Area implementation.

Exchange. The model calculated the exchange (i.e., reflux) of phosphorus between labile soil phosphorus and the water column and was primarily driven by the relationship found between water column concentration and soil phosphorus reflux shown in Equation 1 at the Reflux site. Theoretically, flux magnitude should also be dependent on the size of the labile soil phosphorus pool, which is expected to decline with low phosphorus loads. However, the absolute effect of labile pool size on flux is unknown, since only a single value of the labile pool was measured during this experiment.

Mineralization. This was the net conversion of non-labile to labile phosphorus in the soil resulting from microbial and geochemical processes. "Parameterizing" the mineralization rate was challenging, and represented the largest uncertainty in the model. Determining the net of all processes leading to movement of phosphorus between non-labile to labile soil pools is complicated by the complex and rapid transfer of phosphorus among microbial, plant, detrital, mineral, and other phosphorus-containing soil fractions, especially when each of these fractions may contain phosphorus compounds of varying lability (Wetzel, 1999).

Model Assumptions and Limitations

- Model output applies to only the impacted area of WCA-2A.
- The depth of the excessive soil phosphorus enrichment does not exceed 30 cm.
- The water column depth is constant at 50 cm.
- There exists a discrete cut-off in the labile and non-labile soil phosphorus pools. In the inorganic fractionation, non-labile phosphorus consists of the calcium/magnesium-bound and residue phosphorus; non-labile phosphorus in the organic phosphorus fractionation is comprised of the humic acid- and fulvic acid-associated phosphorus, and residue phosphorus.
- Since the relative contributions by biota, litter, and soil to the standing water phosphorus pool are unknown, all these components were lumped into one compartment entitled "Labile Biomass and Soil Phosphorus."
- Only the net effect of organic matter breakdown is depicted in the transfer of phosphorus from non-labile to labile phosphorus.
- The rate of depletion of the labile phosphorus/biomass pool depends upon the ratio of the mass of phosphorus remaining after each time step to the initial phosphorus amount.

Model Validation

Data collected from a different set of enclosures at the Reflux site, separate from the six used for calibrating the model, served to validate the model (Community Watershed Fund and DB Environmental, Inc., 2009). These three independent enclosures were started a year after the six enclosures used for calibration, and received pre-treated water from 2006–2008. Because of the high hydraulic loadings in 2007, which were double those used to calibrate the model, data collected during that year were excluded in the validation.

Model goodness of fit was judged by comparing actual observed outflow concentration to that predicted by the model. Weekly values of inflow TP concentrations and hydraulic loading rates were used to generate each predicted outflow concentration. A constant value of non-labile phosphorus mineralization of 4.38×10^{-4} per week, or 2.28 percent per year, was used for all simulations. The model ran for 26 weeks to achieve a steady-state value of outflow concentration. The percent root mean square error was 29 percent.

Model Estimate of Soil Phosphorus Reduction and Overlying Water Concentrations

Results of six model scenarios were compared over a time span of 100 years (**Figure 6-30**). The effects of three different surface water inflow concentrations were compared: 10, 20, or 50 micrograms per liter (µg/L). The latter two concentrations were within the range of the wet-season mean concentrations observed during the 2005–2008 Reflux study period, whereas the 10 µg/L represents the targeted criterion concentration (SFWMD, 2006). For each inflow concentration, two values for the non-labile rate constant (k) were examined. The first rate constant value, 0.0228/yr, is the average potential net refractory mineralization rate constant for surface soils in six northern wetland types (Bridgham et al., 1998). The second rate constant value, 0.01/yr, is a more conservative turnover rate for the non-labile phosphorus pool. It was an attempt to account for the replenishment of the non-labile phosphorus pool by the decomposition of future organic matter produced in WCA-2A. The hydraulic loading rate was held constant at 0.693 meters per week. Soil phosphorus data from the reflux site was used as the initial model condition, and this corresponded to 1,248 mg/kg for the non-labile pool and 449 mg/kg for the labile pool.

In general, phosphorus reflux rates (as indicated by the outflow TP concentrations) were initially high and decreased rapidly through time for the 10 and 20 µg/L inflow TP concentration simulations (**Figure 6-30**) as labile organic pools are depleted, leaving a progressively increasing proportion of recalcitrant organic matter to be mineralized. Although the non-labile pools had very low turnover rates, the pool sizes were large and contributed to a substantial percentage of the observed reflux over time.
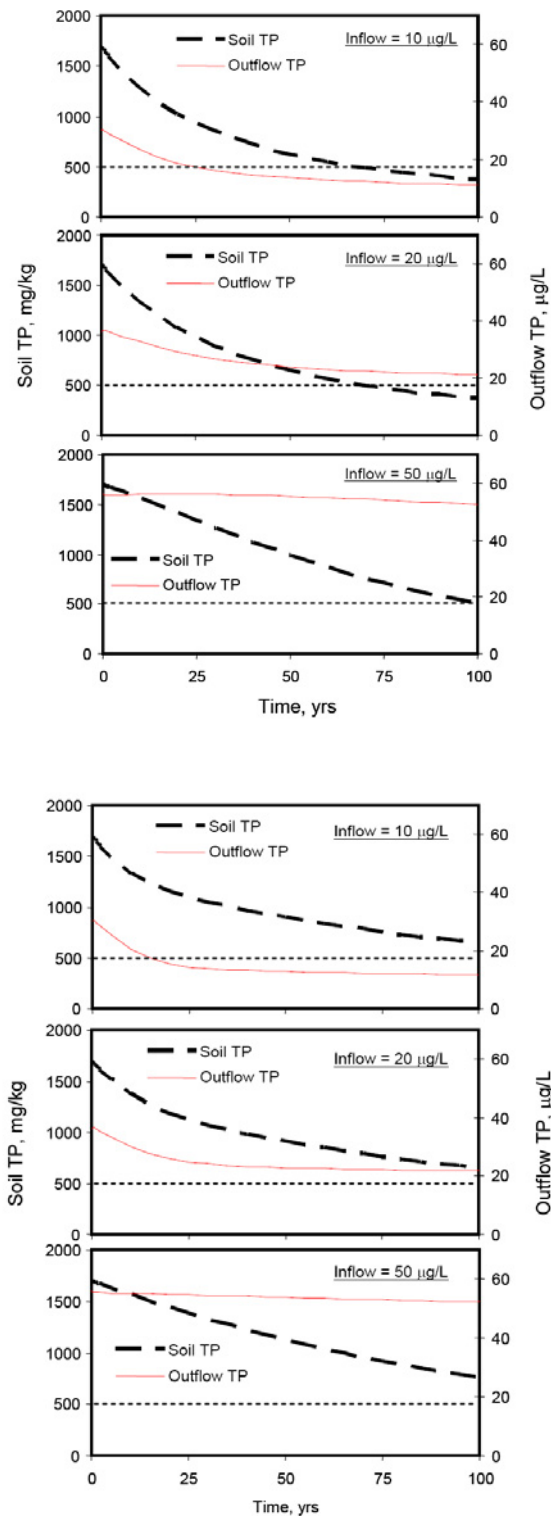
Figure 6-30. Model prediction of long-term changes in soil total phosphorus (TP) and overlying water TP for three inflow TP concentrations [10, 20, and 50 micrograms per liter (µg/L)] at a mineralization rate (k) of 0.0228/yr (top) and 0.01/yr (bottom).

Depending on the rate constant and inflow TP concentrations, the time to reach 500 mg/kg ranged from 67 to greater than 100 years. Over the long term, the rate of reduction in soil phosphorus depends more on selection of the non-labile rate constant and less on the inflow TP concentration, at least within the range of values examined here. For a mineralization rate constant of 0.0228/yr, based on the average of six northern Minnesota wetlands (Bridgham et al., 1998), soil TP concentration tends towards 370–500 mg/kg after 100 years, depending on inflow TP concentration (**Figure 6-30**). Increasing inflow TP concentration lengthens the time to reach lower soil phosphorus values. The time to reach 500 mg/kg was 67 years for 10 µg/L inflow TP, 69 years for 20 µg/L, and 101 years for 50 µg/L inflow. For a mineralization rate (0.01/yr) that is 56 percent lower, the soil TP content did not reach 500 mg/kg within 100 years for any of the three inflow TP concentrations (**Figure 6-30**).

Implications Beyond the Impacted Zone of WCA-2A

Since the data used to calibrate and validate the model originated within a very small area of the impacted zone of WCA-2A, the extent of the modeling results to other, less impacted, areas within WCA-2A is largely unknown. Investigators have documented a widespread increase in low-level soil phosphorus enrichment during periods of high (Reddy et al., 1998; DeBusk et al., 2001) and low (Grunwald et al., 2008) phosphorus loadings. The implication is that down-gradient transport of released phosphorus from the enriched zone is likely to be reincorporated into the biomass/soil complex of the less-enriched areas of WCA-2A. Such an occurrence would have the practical effect of prolonging the time to depurate the entire conservation area to soil levels of 500 mg/kg and lower.

A major uncertainty in the model output is the rate of mineralization from non-labile to labile soil phosphorus pools. Since this rate exerts a major influence over the time required to reduce soil phosphorus to background levels, future efforts should focus on refining this parameter. Although attempts have been made to acquire site-specific mineralization values for soils within WCA-2A (Chua-Oña, 2000), the implementation of a long-term, well-defined study to better characterize the mineralization rate would benefit future modeling.

LANDSCAPE PROCESSES

Ken Rutchey, Jason Godin, Thomas W. Dreschel, Susan
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DEVELOPMENT AND IMPLEMENTATION OF THE SOUTH FLORIDA WATER DEPTH ASSESSMENT TOOL

Persistent drought conditions over the last few years have made it critical for water supply purposes to understand present (real-time) and historical hydrologic conditions in the South Florida environment from a spatial perspective. The South Florida Water Depth Assessment Tool (SFWDAT) was developed by the District as a management tool to illustrate the "state of the system." The two primary purposes of the SFWDAT are: (1) to provide a greater understanding of the spatial and temporal dynamics of water depth across entire ecosystems (e.g., wetlands, lakes, groundwater); and (2) to provide easy access to real-time water depth data and water depth-based indices for use by scientists, managers, and stakeholders.

Hundreds of real-time water level gauges throughout the region are managed by several government agencies [e.g., the SFWMD, the ENP, U.S. Geologic Survey (USGS), and the Miami-Dade Department of Environmental Resource Management]. The SFWDAT couples these water level gauges to produce spatially contiguous estimates of mean daily surface water elevations calibrated to ± 0.1 foot. Water depth is then calculated by subtracting known ground elevations (or topography) from these surface water elevations. Ground elevation data for the Greater Everglades was compiled from several different sources, with the vast majority (97 percent) based on the USGS High Accuracy Elevation Dataset. This ground elevation data has an estimated accuracy of ± 0.5 foot. The primary outputs of the SFWDAT are animated (Internet browser based one-year retrospective) and static interactive (Google Earth) spatial pictures of water depth and depth-related indices over the entire Greater Everglades landscape (**Figure 6-31**).

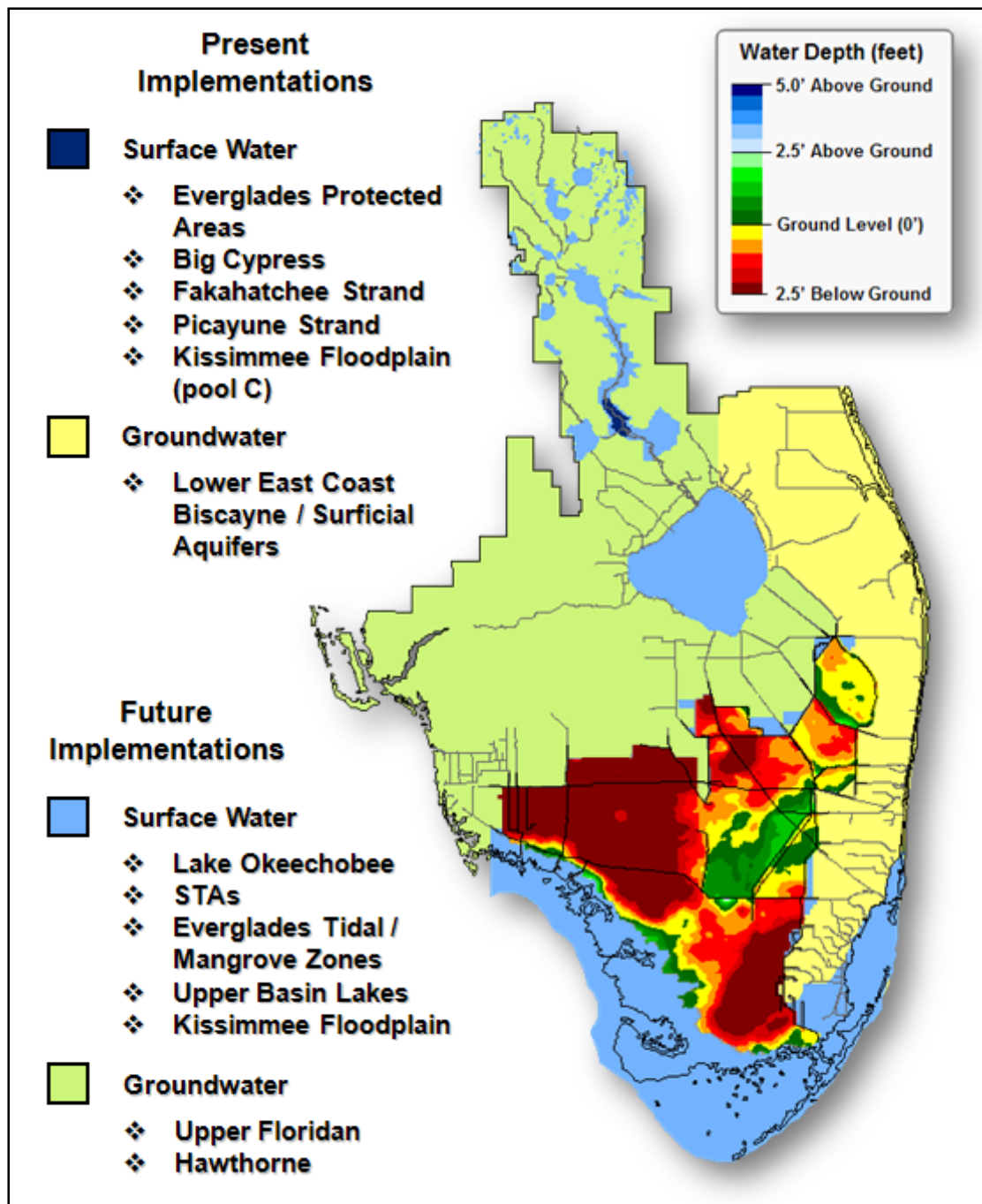---

[7] Photo Science, Inc., St. Petersburg, FL

Figure 6-31. Extent of the South Florida Water Depth Assessment Tool implementation in the Greater Everglades, Big Cypress Preserve, Fakahatchee Strand, and Picayune Strand. Multicolored water depth classes for May 14, 2009, shown, and long-term and short-term surface water and groundwater implementation horizons.

The estimated water depths are used to calculate sequences that depict the spatial and temporal conditions for two environmental indices used by water resource managers. These two indices include (1) a soil-based muck fire index (**Figure 6-32,** a), and (2) a recession-inundation index (**Figure 6-32,** b). The recession portion of the index is particularly useful during the dry season (November through May) to define optimum areas for wading bird foraging and nesting success. The inundation portion of the index is utilized during the wet season (June through October) to ascertain water depth increases in relation to floral and faunal hydrologic requirements.
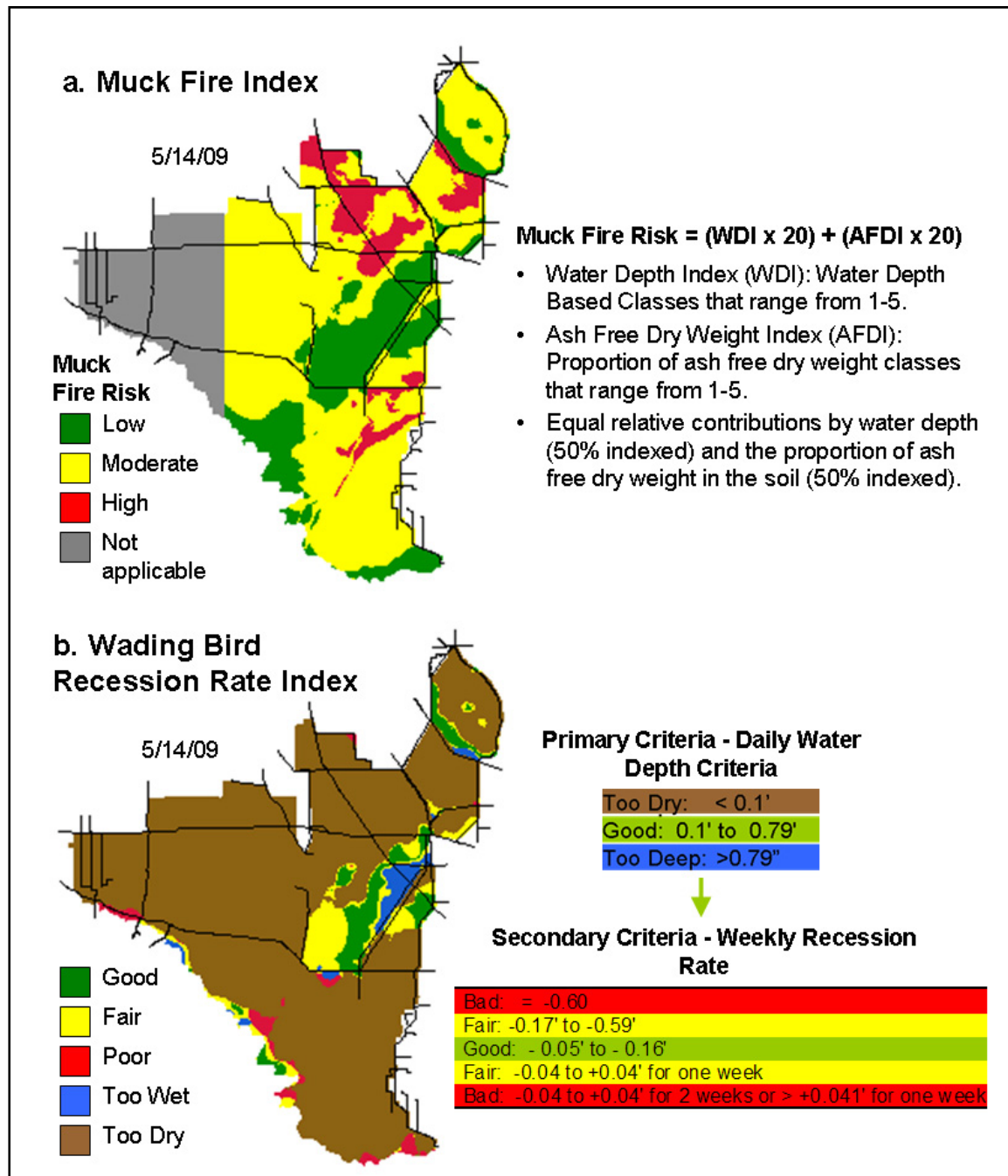

Figure 6-32. One-day snapshots and criteria for the (a) muck-fire index and (b) recession-inundation rate index.

The SFWDAT strings together these indices and the water depth information to produce animations that show systemwide dynamics over time, viewable using any web-browser. This tool also uses Google Earth as a platform that allows users to view water depth and its dependent indices in an easy to use and interactive environment. It presently provides assessments for most of Greater Everglades natural areas (e.g., WCAs, Holey Land Wildlife Management Area, Rotenberger Wildlife Management Area, non-tidal portions of the ENP, Fakahatchee Strand Preserve State Park, Picayune Strand, and non-tidal portions of Big Cypress National Park). The completed SFWDAT has been used in the following general areas:

- Status and trends reporting of water levels
- District-wide drought management
- District-wide operations
- Assessment of alternative regulation schedules for WCA-2A
- Spatial projection of model products including position analyses
- Ecological data analyses

ESTIMATING EVERGLADES PEAT AND CARBON LOSS

The historic Everglades was a broad, shallow, flowing freshwater marsh system that originated at the southern edge of Lake Okeechobee and ended in Florida Bay, covering most of southern Florida. Interest in converting this marsh to agriculture during the 19th century was the driver for human intervention (Parker et al., 1955; Blake, 1980; Davis et al., 1994; Douglas, 1997). Drainage, farming, and other activities have dramatically changed the natural ecosystem of the Everglades Agricultural Area (EAA) over the past 130 years, reducing the Everglades marsh area by about one-fourth and impacting the quantity and quality of water to the remaining natural system (Whitney, 1994; McCally, 1999; Godfrey and Catton, 2006).

Mathematical models have been developed and used by the U.S. Army Corps of Engineers (USACE) and the SFWMD to predict surface water responses to changes in water management. Digital elevation surfaces used in the models have been produced by the SFWMD and the USACE (McVoy et al., 2009; Holt et al., 2006), which also represent estimated soil surfaces during the historic and recent time periods.

These surfaces were used to determine the amount of peat lost since the 1850s. In this exercise, all the carbon lost as peat subsidence was assumed to be converted to atmospheric CO_2 and that the surface projections are consistent, relative to mean sea level elevation.

Methods

Data Sources

The datasets that were utilized for this spatial analysis were: (1) peat depth points from land surveys conducted in the early 1900s within the footprint of the current EAA (McVoy et al., 2009), projected onto a surface using kriging; (2) the historic surface topographic raster grid with a 100-ft resolution (version 5.5) currently used in the Natural Systems Regional Simulation Model (NSRSM) produced by the SFWMD in 2007; and (3) a Surface Elevation Model of the current Everglades assembled by the USACE. The pre-drainage surfaces in the NSRSM were estimated by extracting data from early maps and historic accounts from surveys and narratives from expeditions through the Everglades during the 1800s and early 1900s (Mc Voy et al., 2009). The recent surface elevation raster grid was generated from various sources (Holt et al., 2006).

Procedure

Using the map calculator function of the spatial analysis tool within the ArcGIS® software (ESRI®, 2004) and by applying the raster layer subtraction technique for the historic and current peat surfaces (**Figure 6-33**), the volume of peat loss for the EAA, the WCAs, and the ENP was determined. Carbon dioxide flux grams per square meter per hour ($g/m^2/hr$), the total mass of CO_2 lost and the average peat subsidence were calculated for each region using these volumes, and values from Snyder (1994) for bulk density and percent carbon for Pahokee soils (average bulk density of 0.26 grams per cubic centimeter (g/cm^3) and a carbon content of 51.8 percent).

Peat surface maps were used to estimate the amount of peat present today and more than 100 years ago (**Figure 6-33**). The soil loss, estimated by taking the GIS-derived difference between the two surfaces, was then converted to carbon and CO_2 using the data for physical characteristics of peat from the literature (Snyder, 1994; Corstanje et al., 2006). The surfaces used for this activity were derived from a smoothed historic elevation surface of the NSRSM model produced by the SFWMD and an unsmoothed surface of the recent topography obtained from the USACE (Holt et al., 2006). An additional historic surface for the EAA was produced from early land surveys (circa 1915) during which the peat depth was measured at specific township points (McVoy et al., 2009). A map of these points was produced and kriging performed to create a surface. This surface was likely slightly lower than the original surface because the measurements were limited by the use of a 10-foot rod, and were taken about a quarter century after the initiation of drainage. Thus, two sets of values were produced for the EAA during this exercise, one from the land surveys and one from the NSRSM as the historic surfaces.

The Holt et al. (2006) raster grid data were in North American Vertical Datum 1988 (NAVD88) and the NSRSM surface was in National Geodetic Vertical Datum 1929. An average offset between the two datums for each area was calculated. This offset was identified by averaging the offset indicated by applying the CORPSCON program (USACE, 2004) to a set of points on a regular 2 by 2 miles (3.22 km) grid, all assumed to be at elevation 10 ft NAVD88. The small standard deviation, 0.05 foot, suggested that a single offset for each area was an appropriate approximation (**Figure 6-33**). The offsets ranged from 1.39 ft for the EAA to 1.53 ft for WCA-2B. These offsets were used to calculate a volume correction factor by multiplying the average offset by the area of that region. The corrected volume of peat loss was then used for the final calculations of carbon loss and subsidence.
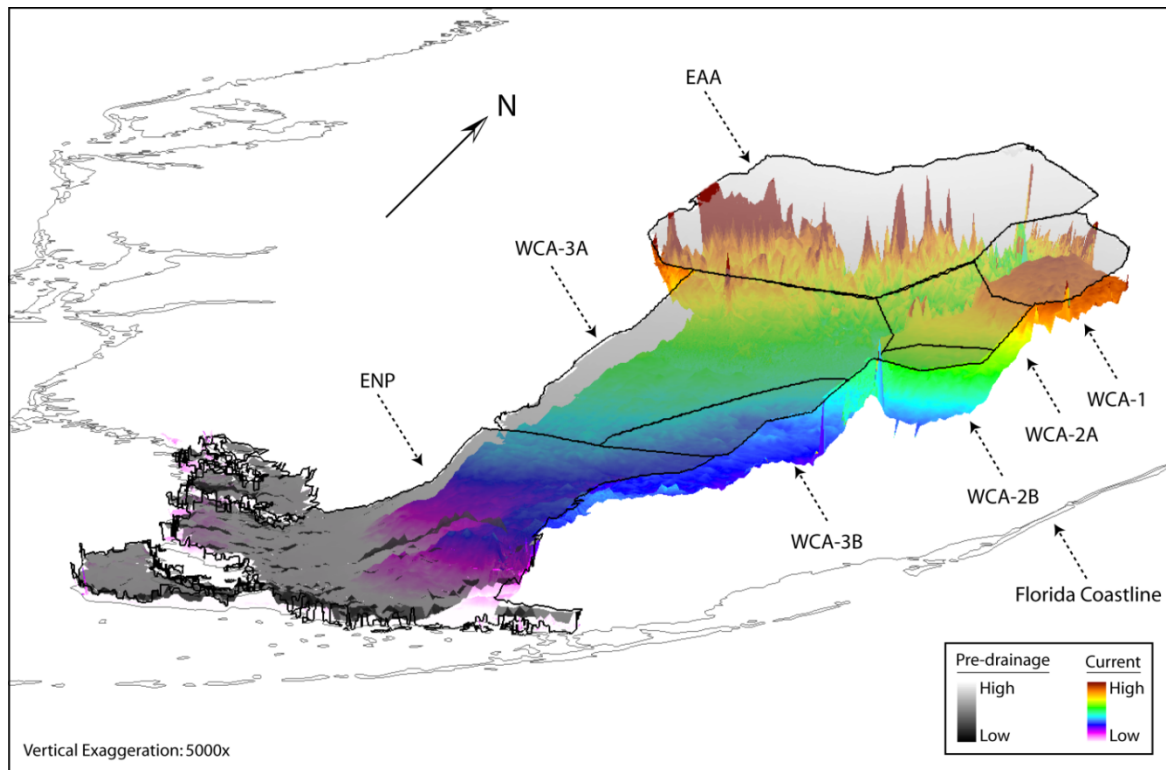
Figure 6-33. Visualization of the contours of the peat surfaces used in the peat loss and carbon flux calculations for the Greater Everglades. The upper grey-scaled surface represents the pre-drainage peat surface (Natural Systems Regional Simulation Model (NSRSM) and the lower color-scaled surface the current condition (USACE). The vertical difference is exaggerated 5,000 times; as an example, the distance between the pre-drainage surface and the current surface of the WCAs is approximately 0.91 meter to 1.52 meters (3–5 ft).

Results

The pre-drainage depth of the peat was greatest in the northern region of the EAA, surrounding Lake Okeechobee (SFWMD, 2007). Peat thickness gradually decreased towards the south. By the 21st century, it was well established that significant subsidence had taken place within the entire Everglades, with an average subsidence of 1.2 m (3.94 ft) to 1.5 m (4.92 ft) across the EAA (Snyder, 2005). These reported values are slightly lower than the findings of 1.7 m (5.5 ft) (**Table 6-12**). Subsidence in the other regions ranged from 0.01 meter (0.1 foot) in the ENP to 0.9 meter (3.0 ft) in WCA-2B.

The difference in peat volume from pre-drainage to current times, calculated using these geospatial techniques, yielded a total of about seven billion cubic meters of peat loss (**Table 6-12**). To convert this value to the amount of carbon lost to the atmosphere, values for specific gravity and percent carbon for Pahokee peat from Snyder (1994) were used, which translated to almost three and a half billion metric tons of CO_2 released between the early 1900s and today (**Table 6-12**).

The average CO_2 flux was found to range from 0.02 g/m^2/hr to 1.1 g/m^2/hr in the ENP and the EAA, respectively (**Table 6-12**).

Table 6-12. Calculations of carbon loss based on average emission and in bulk due to soil (peat) subsidence in the Everglades.

Source	Cubic meters of peat volume lost (ft^3)	WCAs: Grams CO_2 lost per square meter per hour[†]	Total metric tons of CO_2 lost[†]	Average Subsidence in meters (ft) [±]
WCA-1	2.2×10^8 m^3 (7.9×10^9)	0.18	1.1×10^8	0.4 m (1.3 ft)
WCA-2A	2.1×10^8 m^3 (7.9×10^9)	0.23	1.1×10^8	0.5 m (1.7 ft)
WCA-2B	1.1×10^8 m^3 (3.6×10^9)	0.41	4.9×10^7	0.9 m (3.0 ft)
WCA-3A	1.3×10^9 m^3 (4.6×10^{10})	0.30	6.2×10^8	0.6 m (2.1 ft)
WCA-3B	2.5×10^8 m^3 (9.0×10^9)	0.30	1.2×10^8	0.6m (2.2 ft)
ENP	1.2×10^8 m^3 (4.4×10^9)	0.02	6.1×10^7	0.01 m (0.1 ft)
EAA*	4.5×10^9 m^3 (1.62×10^{11}) and 4.9×10^9 m^3 (1.7×10^{11})	1.1 and 0.9	2.2×10^9 and 2.3×10^9	1.62 m (5.31 ft) and 1.7 m (5.5 ft)
Total	**7.0×10^9 m^3 ($2.5\ 10^{11}$)**		**3.4×10^9**	

*Two values were produced for the EAA as two pre-drainage surfaces were utilized. The first surface was constructed directly from soil surveys covering the late 1800s to the early 1900s and the second was taken from the NSRSM (see text).

[†]Using data from Snyder (1994) for bulk density and carbon content.

[±]From the m^3 of peat volume lost and the area of the region.

Discussion

The average CO_2 flux ranged from 0.02 g/m^2/hr for the ENP to 1.1 g/m^2/hr for the EAA. These findings are in good agreement with the measurements of Gesch et al. (2007) which ranged from 0.40 g/m^2/hr to 2.67 g/m^2/hr during their short-term, controlled chamber studies from tilled and untilled EAA soils.

These estimated peat surfaces provide an estimate of the degree of soil loss that occurred during the period following the construction of the early drainage canals, and the resulting impact to the global carbon cycle. For nearly a century and a quarter, weather patterns, drainage, and impoundment have resulted in the loss of peat from the Greater Everglades. Given that native vegetation should respond to inundation similar to the way it did in the historic landscape, the restoration of hydroperiod within these areas should result in the maintenance of robust marsh species growth (Whitney et al., 2004) and peat deposition. This growth, in turn, would lead to additional carbon storage and could result in the region increasing its value as a carbon sink, thus providing an important ecosystem service.

The Union of Concerned Scientists (2007) report that the annual CO_2 emission total from fossil fuels in the U.S. is about 6 billion metric tons (**Table 6-13**). For Florida, the USEPA (2007) reports the total as about 256 million metric tons (**Table 6-14**). Keeping in mind that the highest

peat loss occurred during the canal drainage period (1880s–1940s), on a rough, average annual basis, the carbon emissions from the Everglades were estimated to have been equivalent to about 13 percent of the current Florida fossil fuel carbon emissions (or about a quarter of that from transportation and electric power generation in Florida) and about 0.6 percent of the annual national fossil fuel emissions.

Table 6-13. Annual CO_2 emissions for China and the U.S. (2006).*

Country	Total Emissions	Per Capita Emissions
	(Million metric tons of CO_2)	(Tons/capita)
China	6,017.69	4.58
United States	5,902.75	19.78

*Source: http://www.ucsusa.org/global_warming/science_and_impacts/science/each-countrys-share-of-co2.html

Table 6-14. Annual CO_2 emissions for Florida (2007). Carbon dioxide emissions from fossil fuel combustion in million metric tons CO_2.**

Florida (total)	256.27
Commercial	4.33
Industrial	12.34
Residential	1.76
Transportation	114.73
Electric Power	123.10

**Source: http://www.epa.gov/climatechange/emissions/downloads/CO2FFC_2007.pdf

SPATIAL SOIL CARBON DISTRIBUTION

Despite a large variability, the general conclusion of the scientific community is that Everglades peatlands act as a large repository for carbon sequestration (Reddy and DeLaune, 2008). However, through a combination of over-drainage, and in more severe cases, muck fires, carbon has been lost from the ecosystem. In addition, with the uncertainty associated with changes in climatic conditions, whether less carbon storage (due to higher temperatures and drier conditions), or more carbon sequestration will occur (as a result of wetter conditions) is unknown. A key step in understanding how the system has reacted to past changes, and therefore may be influenced by future events, is to delineate the current spatial extent of carbon in the ecosystem. Of equal importance is understanding the error associated with these estimates.

In this study, data from the spatially extensive 2003 soil mapping effort (Newman et al., in prep) were used to develop spatial maps of carbon distribution in surficial soils of the WCAs. Previous studies have examined nutrient distributions within individual WCAs (Grunwald et al., 2004; Bruland et al., 2006; Corstanje et al., 2006; Bruland et al., 2007; Rivero et al., 2007; Marchant et al., accepted). Given that the ecosystem is connected via a network of canals and inflow structures, an important study component was to evaluate how those hydrologic connections may have influenced the spatial distribution of carbon. The 0–10 cm surface peat fraction was used in this study because it aggregates changes that occurred in the last 10–100 years of anthropogenic influence on the system.

Methods

This mapping of total carbon (TC) used soil data collected from 651 sampling sites in WCA-1, WCA-2, and WCA-3 (**Figure 6-34**). Data were from the joint University of Florida/SFWMD 2003 soil mapping effort of the South Florida ecosystem. Surface interpolation was performed using ESRI® ArcGIS® 9.2 software, including the Geostatistical Analyst Extension. Kriging was used to spatially interpolate the data and quantify the spatial autocorrelation among measured points. Output includes prediction errors and standard error maps that can be used to evaluate the confidence of the kriged prediction surface. Deterministic interpolation techniques were used to create surfaces from measured points, based on either the extent of similarity (Inverse Distance Weighted, IDW) or the degree of smoothing (Radial Basis Functions, RBF). Root means square error (RMSE) was then used to assess the fit of the surface to the data.

Initially, IDW was run on the entire data sample set (**Figure 6-35**, panel A). The RMSE was high (61.7) and further examination demonstrated the data were highly clustered and, therefore, not a good dataset for IDW. The RMSE for RBF (59.56) was similar to IDW (**Figure 6-35**, panel B), and likewise not satisfactory. Additionally, the data were not normally distributed, and therefore not appropriate for kriging (**Figure 6-35**, panel C).
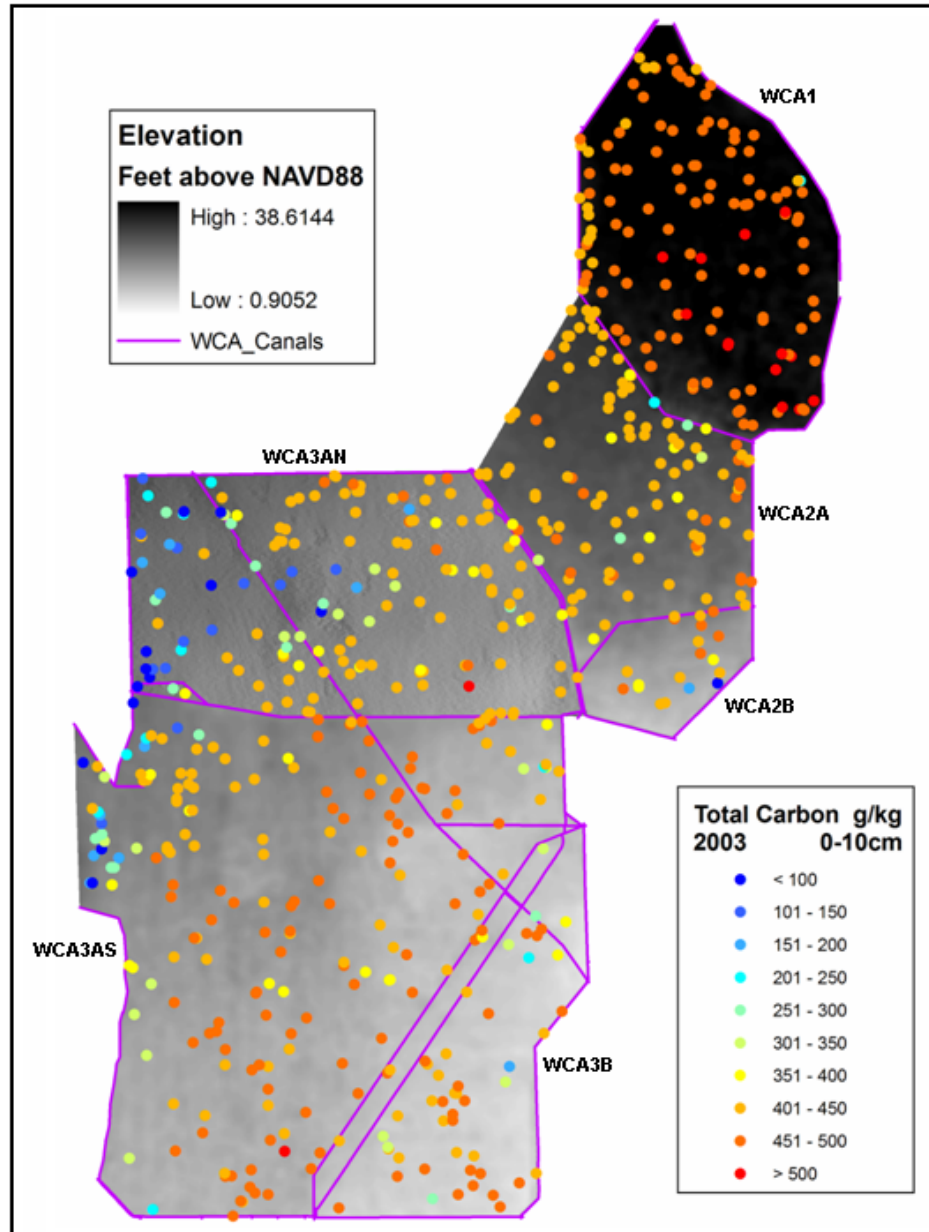
Figure 6-34. Location and total carbon values in grams per kilogram of soil samples collected as part the 2003 Everglades soil mapping effort.
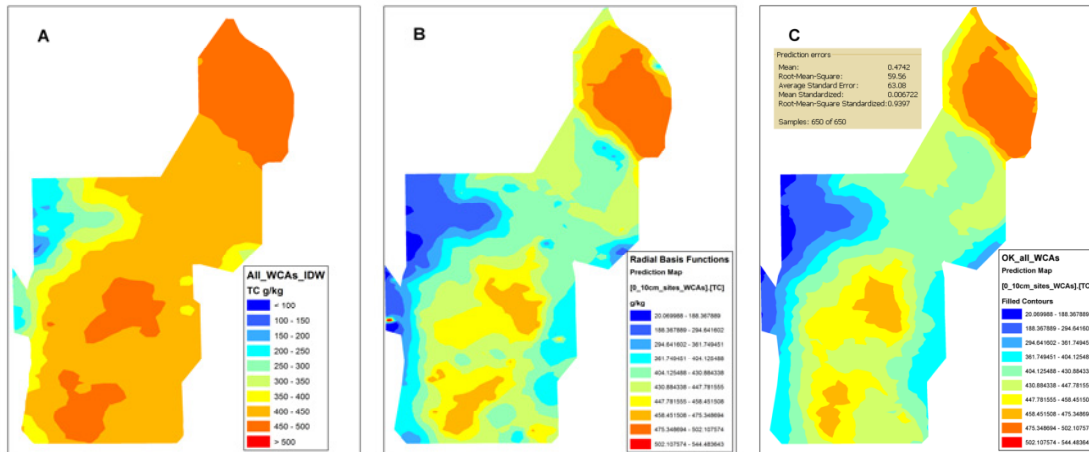
Figure 6-35. Comparison of total carbon surfaces created using different spatial interpolations; (A) inverse distance weighted, (B) radial basis function, and (C) ordinary kriging.

What was apparent from this analysis was that the compartmentalization of the WCAs created discontinuities in elevation and water flow, resulting in boundaries of separation in soil chemistry. This contributed to the high clustering and non-Gaussian structure of the dataset. An alternative approach was to subdivide the data into smaller regions and re-evaluate for appropriate method(s) of interpolation in each region.

The sample dataset was divided into six areas: WCA-1 (n=131 samples), WCA-2A (n=111), WCA-2B (n=19), WCA-3AN (n=147), WCA-3AS (n=189), and WCA-3B (n=54). These data were then used to create an ordinary kriged prediction surface for each of the WCAs. In addition, each WCA was interpolated by IDW and clustering was determined by Nearest Neighbor Distance (NND). This provided a baseline for comparison with surfaces created with kriging. For WCA-2B, the data were too sparse and non-Gaussian to use kriging. RBF (completely regularized spline and parameter optimization) was used to interpolate the prediction surface. The RMSE (27.75) was reasonable. The prediction errors and standard error map values for the individual WCAs were better than the errors reported for the data treated as a single area. There was also an improvement compared with the IDW RMSE of each individual area. Each of the six prediction maps was reclassed to increments of 25 grams of carbon per kg (g C/kg) and output to filled contour shapefiles. The six shapefiles were combined together into one TC surface map for the WCAs and standard error maps were produced for the five areas (not WCA-2B) and combined (**Figure 6-36**).

Results

The spatial interpolations highlight the higher TC contents in WCA-1 compared to the rest of the WCAs (**Figure 6-36**). Other notable patterns include the extremely low carbon values in the northwest region of WCA-3A. These are likely due to soil loss following a muck fire in 1999, as well as earlier burns (Bruland et al., 2006). The lower TC wedge immediately downstream of the S-10 structures between WCA-1 and WCA-2A is associated with increased carbon cycling due to high nutrient enrichment. The carbon content of the soil tends to follow the peat depths (Scheidt and Kalla, 2007). However, the storage of carbon in this system is greater than that suggested by surficial data shown in **Figure 6-36**, for example, core collected by the USGS in WCA-1 show organic carbon contents increase in deeper peats (Orem et al., unpublished data).
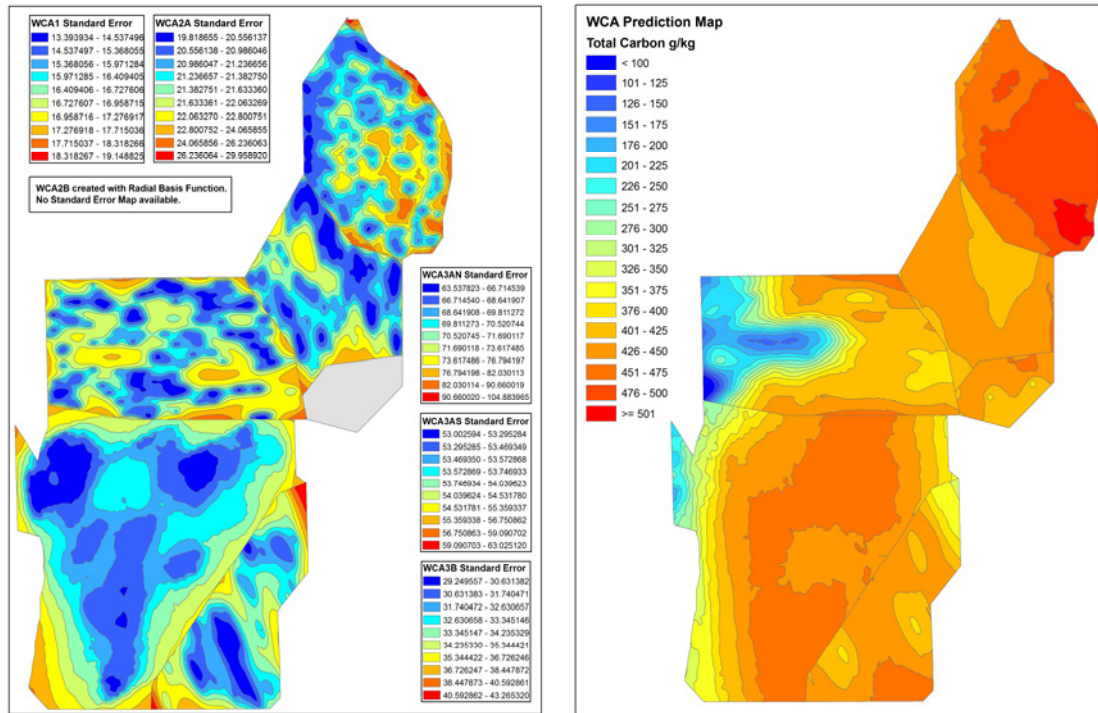
Figure 6-36. Predicted surface for TC distribution in 0-10 cm soil depths of Everglades soils (right), and associated error of the predicted surfaces (left).

Conclusions

Anthropogenic alterations of the Everglades ecosystem have had a dramatic effect on carbon storage in this peatland. In unenriched (baseline) regions, peat accumulation rates average < 1–2.5 millimeters per year (Bernhardt et al., 2004; Craft and Richardson, 1998; Reddy et al., 1993), thus factors that exacerbate carbon loss will have long-term consequences on the landscape. Managing water to prevent over-drainage and limit soil oxidation is essential to maintain these organic soils. However, there are also tradeoffs that need to be considered. If the water used to keep the soils hydrated is high in phosphorus, then carbon loss may be accelerated (for example, in WCA-2A). Thus, continued efforts to reduce external nutrient inputs and redistribute water throughout the system are essential if natural peat accumulation is to be profitably restored.

ACQUISITION OF NEW AIRBORNE DIGITAL AERIAL IMAGERY FOR THE RECOVER VEGETATION MAPPING PROJECT

New digital aerial imagery encompassing nearly 2,000 square miles was acquired during April 2009 for the ENP, Southeastern Miami-Dade Coastal Wetlands, and Biscayne Bay Coastal Wetlands for the CERP Restoration Coordination and Verification (RECOVER) vegetation mapping project (**Figure 6-37**). Digital aerial photography for other areas (361 square miles) not part of the project were also acquired under the same contract in order to take advantage of economies of scale and the resulting cost savings. These other areas included LILA, Everglades Stormwater Treatment Areas (STAs), northern WCA-2A, Lake Istokpoga, Lake Josephine, the Kissimmee Chain of Lakes, and the Kissimmee River floodplain.

This new digital imagery is unique in that it was collected by a new generation frame-based aerial digital camera from Microsoft called UltraCamX. For this project, the ground pixel resolution is approximately one square foot for the entire project, with the exception of LILA, which was collected at a one-quarter square foot resolution. The digital color intensity range of this data collection is far better than anything collected in the past due to the advanced data collection technology that has been built into the new camera system. The combination of these two factors (improved color intensity range and ground resolution) makes this dataset superior to any products utilized to date for conducting vegetation mapping in the Greater Everglades (**Figure 6-38**).

In addition, the digital images were captured with enough overlap between adjacent frames so that stereo images could be produced utilizing state-of-the-art mapping technology. This technology can be used to create a vegetation database for 4,200 square miles of Everglades wetlands. Customized geo-visualization tools have been developed which allow a trained photointerpreter to view, roam, interpret, and store data throughout a virtual three-dimensional (3-D) Everglades landscape rendered on the computer workstation. The 3-D view is created by projecting a stereo pair of digital aerial images onto a single plane in front of the computer screen. The computer operator is able to interpret the stereo projected imagery as a 3-D scene by wearing a specialized set of polarized glasses that allows one image of the stereo pair to be viewed by each eye. All stereo imagery is in the uncompressed Tagged Image File Format (TIFF) and is referenced to the State Plane Coordinate System, Florida East Zone 0901, and North American Datum of 1983/High Accuracy Reference Network adjustment. Vertical information is referenced to NAVD88.

The vegetation mapping project is on schedule and a new 2004 aerial photo-based vegetation map of WCA-3 will was expected to be complete by the end of 2009. These data provide an update to the earlier map done for this impoundment in 1995 (Rutchey et al., 2005). The next area scheduled for mapping is Northeast Shark River Slough (**Figure 6-39**).
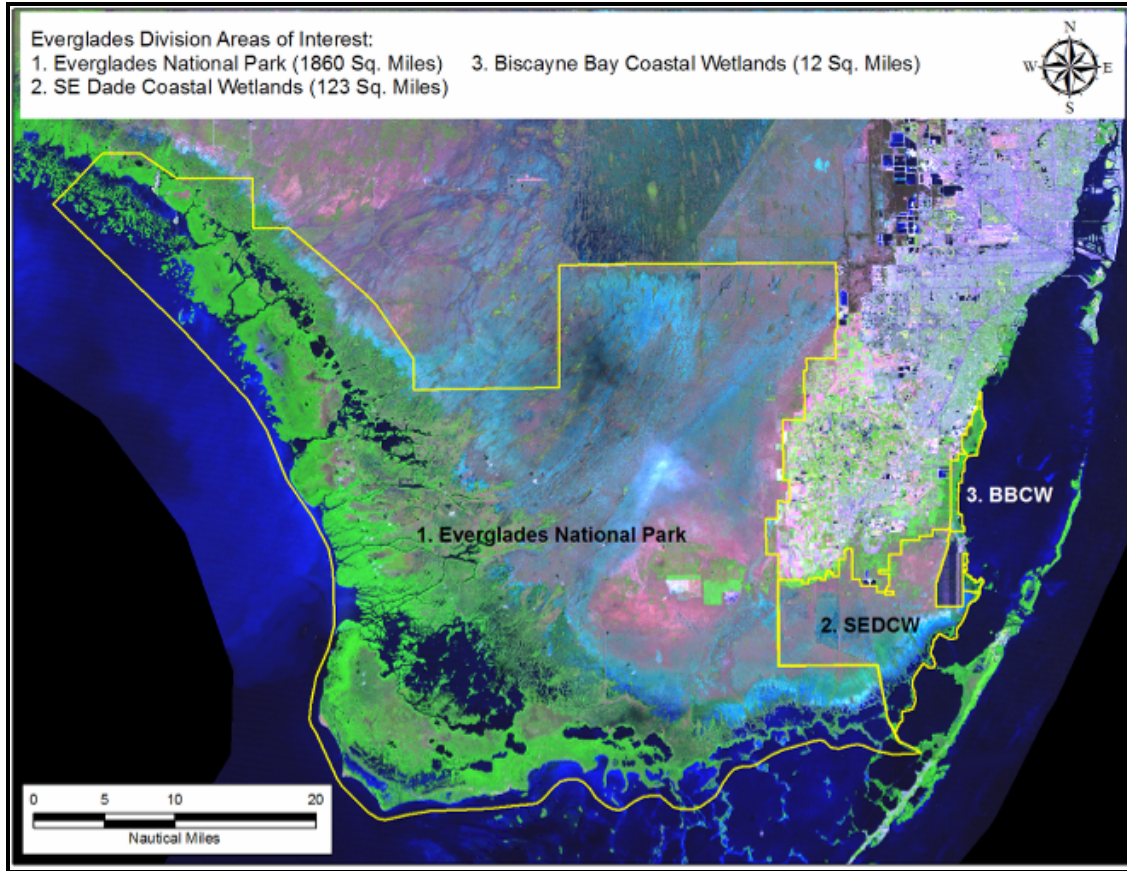
Figure 6-37. Digital aerial photography acquisition in 2009 includes Everglades National Park, Southeastern Miami-Dade Coastal Wetlands (SEDCW), and the Biscayne Bay Coastal Wetlands (BBCW).
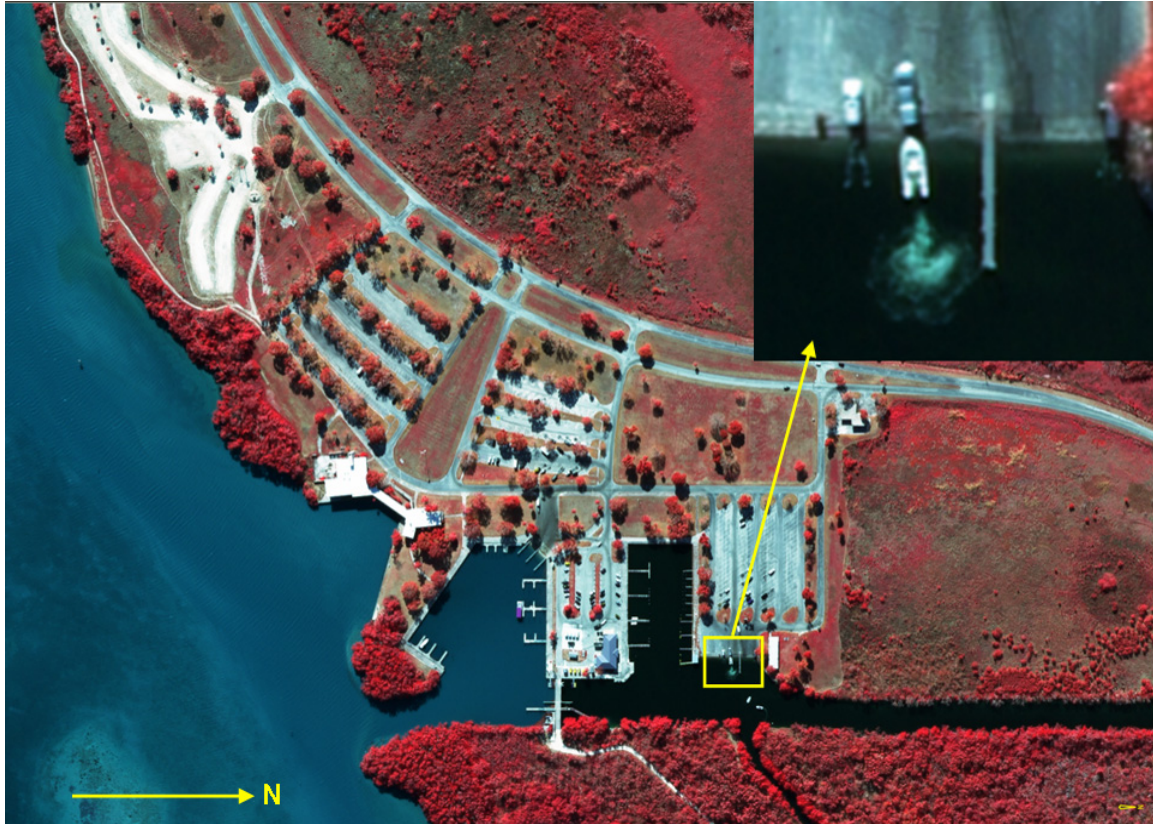
Figure 6-38. An example of a color infra-red composite of digital aerial photography located at the Flamingo Marina in Everglades National Park. The enlargement shows a boat being loaded onto a trailer – notice the propeller wash behind the boat. Actual data is viewable in three dimensions and has a vertical perspective.
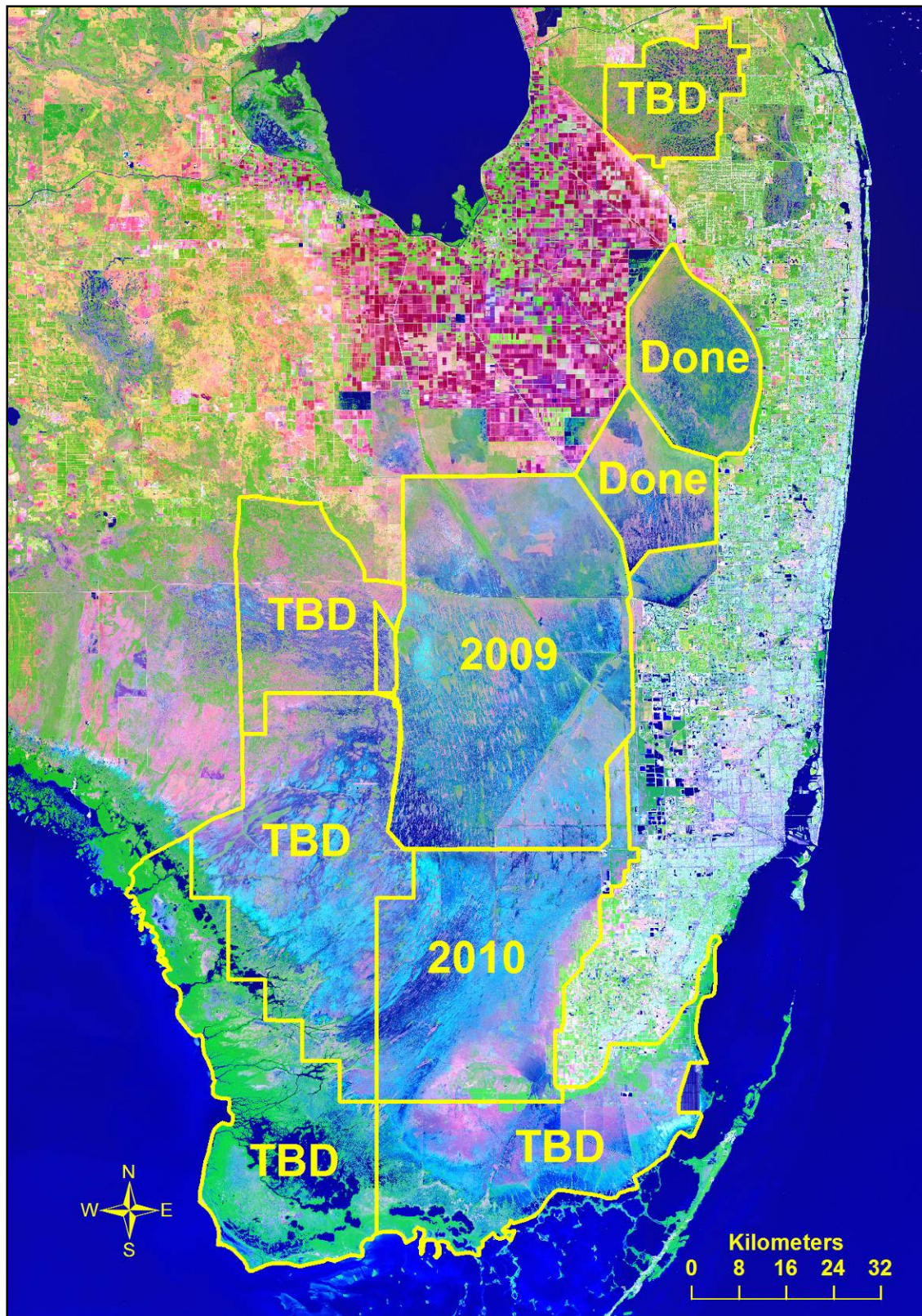
Figure 6-39. Vegetation mapping areas that have been completed, are projected to be completed, and to be decided (TBD).

CONCLUSIONS

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Data and preliminary findings from 12 projects were presented in Chapter 6 of the 2010 SFER. Findings were presented across trophic scales spanning population to landscape-level and highlighted the broad requirements of water management decision making. Multi-scale data such as these are critical because almost any water management decision can produce scientific questions or trade-off issues at the system level. As the Everglades system continues to evolve due to a legacy of drainage, storage, agriculture, development, hurricanes, and exotics, restoring the region to historical reference points becomes a highly dynamic process. Scientific research is at the core of these efforts, providing the basis and methods needed to achieve the long-term restoration goals while balancing the day-to-day water management needs of agriculture, residents, and the environment.

The wading bird response in WY2009 was an example of the District's potential to make effective, positive changes to the Everglades landscape. The large nesting populations of great egrets, ibis, and wood storks were a testament to the resilience of the prey base and the need for large, inter-annual hydrologic deviations. Although wading birds may be considered one of the first responders to hydrologic change and water management, these data point again to the significance of multi-year processes and cumulative impacts. Water managers are now learning ways to restore historic buffering capacities by actively preventing hydrologic reversals and trying to maintain more natural recession rates during the dry season such that:

- Recessions rates, droughts, and their management created the largest nesting effort recorded in the Everglades since the 1940s. The 80,000 nests surpasses the previous record year, 2002, by 11,000 nests. However, recovery was not complete since snowy egrets and tricolored herons were 90 percent short of their restoration targets and because large numbers of wading birds have not yet returned to the ENP.

This SFER indicates that more wildlife studies are needed. In order to continue the restoration momentum demonstrated by the wading bird nesting season of 2008-2009, the role of crayfish, other invertebrates, cyanotoxin production, exotics, and habitat loss in relation to water management all need further study.

Large scientific uncertainties also still remain in association with the restoration of impacted areas, tree islands, and sloughs. Studies associated with the physiological response of trees on islands in WCAs 3A and 3B indicated that all species quickly respond to water level changes, even the most flood-tolerant species. These data provide useful information for water level management, but more study will be needed to determine the comprehensive methods needed to restore these habitats.

Despite the finding that transport of TP downstream of the impacted zone in WCA-2A will likely enrich sites that are currently un-enriched, three approaches (a synthesis of multi-year phosphorus additions to enclosures in WCA-2A, an innovative ecosystem-scale manipulation, and a simple process-based model) concluded that:

- The time needed (without interventions) to reach the background level of 500 TP mg/kg in soils from the impacted zone ($>$1,500 TP mg/kg) will be 67 years for 10 μg/L of TP inflow and 101 years for 50 μg/L of TP inflow.

Finally, a new level of mapping sophistication was presented this year. Now the SFWMD, in cooperation with the USGS, is able to evaluate detailed, near real-time water depths across a heterogeneous landscape and determine if particular regions are more or less at risk from weather

patterns and water management decisions. This tool, combined with plant and wildlife studies, has already improved the District's ability to integrate biological and ecological knowledge into weekly operational recommendations. Areas of foraging and nesting are now better protected, while regions that tend to dry out are less susceptible to peat fires. The District also entered a new phase of vegetation mapping in WY2009 and has invested in photogrammetric workstations with resolution once only available to the military such that:

- Digital aerial photography has a resolution of one square foot and, combined with three-dimensional rendering software, provides the ability to produce accurate assessments of vegetation change, cattail retreat, and tree island canopy height and thickness.

LITERATURE CITED

Armentano, T.V., D.T. Jones, M.S. Ross and B.W. Gamble. 2002. Vegetation Pattern and Process in Tree Islands of the Southern Everglades and Adjacent Areas. Pp 225-281. In: Tree Islands of the Everglades. van der Valk, A. and F. Sklar, eds. Kluwer Academic Publishers, Dordrecht.

Bernhardt, C.E., D.A. Willard, M. Marot and C.W. Holmes. 2004. Anthropogenic and Natural Variation in Ridge and Slough Pollen Assemblages. USGS Open File Report 2004-1448.

Berry, J.P., M. Gantar, P.D.L. Gibbs and M.C. Schmale. 2007. The Zebrafish (*Danio rerio*) Embryo as a Model System for Identification and Characterization of Developmental Toxins from Marine and Freshwater Microalgae. *Comparative Biochemistry and Physiology*, Part C, 145:61-72.

Berry, J.P., M. Gantar, M.H. Perez, G. Berry and F.G. Noriega. 2008. Cyanobacterial Toxins as Allelochemicals with Potential Applications as Algaecides, Herbicides and Insecticides. *Marine Drugs*, 6:117-146.

Blake, N.M. 1980. Land into Water – Water into Land: A History of Water Management in Florida. University Press of Florida, Gainesville, FL.

Bridgham, S.C., K. Updegraff and J. Pastor. 1998. Carbon, Nitrogen and Phosphorus Mineralization in Northern Wetlands. *Ecology*, 1545-1561.

Bruland, G.L., S. Grunwald, T.Z. Osborne, K.R. Reddy and S. Newman. 2006. Spatial Distribution of Soil Properties in Water Conservation Area 3 of the Everglades. *Soil Science Society of America Journal*, 70:1662-1676.

Bruland, G.L., T.Z. Osborne, K. Reddy, S. Grunwald, S. Newman and W.F. DeBusk. 2007. Recent Changes in Soil Total Phosphorus in the Everglades: Water Conservation Area 3. *Environmental Monitoring and Assessment*, 129:379-395.

Camacho, F.A. and R.W. Thacker. 2006. Amphipod Herbivory on the Freshwater Cyanobacterium *Lyngbya wollei*: Chemical Stimulants and Morphological Defenses. *Limnology and Oceanography*, 51:1870-1875.

Chick, J.H., C.R. Ruetz and J.C. Trexler. 2004. Spatial Scale and Abundance Patterns of Large Fish Communities in Freshwater Marshes of the Florida Everglades. *Wetlands*, 24: 652-664.

Chua-Oña, T.C. 2000. Mineralization of Organic Phosphorus in a Subtropical Freshwater Wetland. Ph.D. dissertation. University of Florida, Gainesville, FL. 147 pages.

Community Watershed Fund and D. B. Environmental Inc. 2009. Enhancing Sediment Phosphorus Storage in Impacted Regions of the Everglades Protection Area. Prepared for Florida Department of Environmental Protection, Everglades Agricultural Area–Environmental Protection, and South Florida Water Management District. West Palm Beach, FL.

Corstanje, R., S. Grunwald, K.R. Reddy, T.Z. Osborne and S. Newman. 2006. Assessment of the Spatial Distribution of Soil Properties in a Northern Everglades Marsh. In: *Journal of Environmental Quality*, 35:938-949. Technical Reports: Wetlands and Aquatic Processes.

Craft, C.B. and C.J. Richardson. 1998. Recent and Long-term Organic Soil Accretion and Nutrient Accumulation in the Everglades. *Soil Science Society of America Journal*, 62: 834-843.

Davis, S.M., J.C. Ogden and W.A. Park. Eds. 1994. Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, FL.

DeBusk, W.F., S. Newman and K.R. Reddy. 2001. Spatio-temporal Patterns of Soil Phosphorus Enrichment in Everglades Water Conservation Area 2A. *Journal of Environmental Quality*, 30:1438-1446.

Demott, W.R. and D.C. Müller-Navarra. 1997. The Importance of Highly Unsaturated Fatty Acids in Zooplankton Nutrition: Evidence From Experiments with *Daphnia*, a Cyanobacterium and Lipid Emulsions. *Freshwater Biology*, 38:649-664.

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. 2008. Colonization and Reproduction of Large Macroinvertebrates are Enhanced by Drought-related Fish Reductions. *Hydrobiologia*, 605: 209-218.

Dorn, N.J. and J.C. Volin. 2008. Crayfish Population Dynamics: Hydrological Influences. Annual Report to the South Florida Water Management District. RECOVER Contract 9500005560. South Florida Water Management District, West Palm Beach, FL.

Douglas, M.S. 1997. The Everglades: River of Grass. Pineapple Press, Sarasota, FL.

ESRI. 2004. ArcGIS 9: Using ArcGIS® Spatial Analyst. ESRI, Inc., Redlands, CA.

Ewe, S.M.L. and L. da S.L. Sternberg. 2003. Seasonal Gas Exchange Characteristics of *Schinus terebinthifolius* in a Native and Disturbed Upland Community in Everglades National Park. *Forest Ecology and Management*, 179:27-36.

Ewe, S.M.L. and L. da S.L. Sternberg. 2007. Water Utilization Patterns of the Invasive Exotic *Schinus terebinthifolius* Contrasted Against Native Species in Two Coastal Plant Communities. *Journal of Coastal Research*, 23:255-264.

Fisher, M.M. and K.R. Reddy. 2001. Phosphorus Flux from Wetland Soils Affected by Long-term Nutrient Loading. *Journal of Environmental Quality*, 30:261-271.

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.

Gantar, M., J.P. Berry, S. Thomas, M. Wang, R. Perez, and K.S. Rein. 2008. Allelopathic Activity Among Cyanobacteria and Microalgae Isolated from Florida Freshwater Habitats. *FEMS Microbiology Ecology*, 64:55-64.

Gawlik, D.E. 2002. The Effects of Prey Availability on the Numerical Response of Wading Birds. *Ecological Monographs*, 72: 329-346.

Gesch, R.W., D.C. Reicosky, R.A. Gilbert and D.R. Morris. 2007. Influence of Tillage and Plant Residue Management on Respiration of a Florida Everglades Histosol. *Soil Tillage Research*, 92: 156-166.

Godfrey, M.C. and T. Catton. 2006. River of Interests: Water Management in South Florida and the Everglades, 1948-2000. U.S. Army Corps of Engineers, Comprehensive Everglades Restoration Program, Jacksonville, FL.

Grunwald, S., K.R. Reddy, S. Newman and W.F. DeBusk. 2004. Spatial Variability, Distribution and Uncertainty Assessment of Soil Phosphorus in a South Florida Wetland. *Environmetrics*, 15:1-15.

Grunwald, S., T.Z. Osborne and K.R. Reddy. 2008. Temporal Trajectories of Phosphorus and Pedo-patterns Mapped in Water Conservation Area 2, Everglades, Florida, USA. *Geoderma*, 146:1-13.

Gu, B., S.L. Miao, C. Edelstein and T. Dreschel. 2008. Effects of a Prescribed Fire on Dissolved Inorganic Carbon Dynamics in a Nutrient-enriched Everglades Wetland. *Fundamental and Applied Limnology*, 171:263-272.

Gunderson, L.H. 1994. Vegetation of the Everglades: Determinants of Community Composition. Pp. 323-340. In: S.M. Davis and J.C. Ogden, eds. Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Boca Raton, FL.

Hitzfeld, B.C., S.J. Hoger and D.R. Dietrich. 2000. Cyanobacterial Toxins: Removal During Drinking Water Treatment, and Human Risk Assessment. *Environmental Health Perspectives*, 108:113-122.

Holt, P.R., R.J. Sutton and D. Vogler. 2006. South Florida Digital Elevation Model, Version 1.1 U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL.

Izaguirre, G., A.D. Jungblut and B.A. Neilan. 2007. Benthic Cyanobacteria (*Oscillatoriaceae*) that Produce Microcystin-LR, Isolated from Four Reservoirs in Southern California. *Water Research*, 41:492-498.

Koerselman, W. and A.F.M. Meuleman. 1996. The Vegetation N:P Ratio: A New Tool to Detect the Nature of Nutrient Limitation. *Journal of Applied Ecology*, 33:1441-1450.

Kotaki, Y., Lundholm N., Onodera H., Kobayashi K., Bajarias, F.F.A., Furio E.F., Iwataki M., Fukuyo Y. and Kodama M. 2004. Wide Distribution Of *Nitzschia Navis-Varingica*, A New Domoic Acid-Producing Benthic Diatom Found in Vietnam. *Fisheries Science*, 70:28-32.

Lefebvre, K.A., Trainer V.L. and Scholz N.L. 2004. Morphological Abnormalities and Sensorimotor Deficits in Larval Fish Exposed to Dissolved Saxitoxin. *Aquatic Toxicology*, 66:159-170.

Li, SG, M. Tsujimura, A. Sugimoto, G. Davaa, D. Oyunbaatar and M. Sugita. 2007. Temporal Variation of Delta C-13 of Larch Leaves from a Montane Boreal Forest in Mongolia. *Trees-Structure and Function*, 21: 479–490.

Li, X.-Y., J. Wang, J.-B. Liang and Y.-D. Liu. 2007. Toxicity of Microcystins in the Isolated Hepatocytes of Common Carp (*Cyprinus carpio L.*). *Ecotoxicology and Environmental Safety*, 67:447-451.

Liston, S.E. and J.C. Trexler. 2005. Spatiotemporal Patterns in Community Structure of Macroinvertebrates Inhabiting Calcareous Periphyton Mats. *Journal of the North American Benthological Society*, 24:832-844.

Lopez, O.R., T.A. Kursar, H. Cochard and M.T. Tyree. 2005. Interspecific Variation in Xylem Vulnerability to Caviation Among Tropical Tree and Shrub Species. *Tree Physiology*, 25:1553-1562.

Loveless, C.M. 1959. A Study of the Vegetation in the Florida Everglades. *Ecology*, 40:1-9.

Ma, J.Y., K. Chen, D.S. Xia, G. Wang and F.H. Chen. 2007. Variation in Foliar Stable Carbon Isotope Among Populations of a Desert Plant, *Reaumuria soongorica* (Pall.) Maxim. in Different Environments. *Journal of Arid Environments*, 69:365-374.

Magurran, A.E. 1988. Ecological Diversity and Its Measurement. Princeton University Press, Princeton, NJ.

Marchant, B.P., S. Newman, R. Corstanje, K.R. Reddy, T.Z. Osborne and R.M. Lark. Accepted. Spatial Monitoring of a Non-stationary Soil Property: Phosphorus in a Florida Water Conservation Area. *European Journal of Soil Science*, 60:994-1004.

McCally, D. 1999. The Everglades: An Environmental History. University Press of Florida, Gainesville, FL.

McCormick, P.V., R.B.E. Shuford, J.G. Backus and W.C. Kennedy. 1998. Spatial and Seasonal Patterns of Periphyton Biomass and Productivity in the Northern Everglades, Florida, U.S.A. *Hydrobiologia*, 362:185-208.

McCune, B., J.B. Grace and D.L. Urban. 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, OR.

McKee, K.L., I.C. Feller, M. Popp and W. Wanek. 2002. Mangrove Isotopic (δ^{15}N and δ^{13}C) Fractionation Across a Nitrogen versus Phosphorus Limitation Gradient. *Ecology*, 83:1065-1075.

McVoy, C.W., W. Park Said, J. Obeysekera, J. Vanarman and T. Dreschel. 2009. In Press. Landscapes and Hydrology of the Pre-drainage Everglades. University Press of Florida, Gainesville, FL.

Mezhoud, K., D. Praseuth, S. Puiseux-Dao, J.C. Francois, C. Bernard and M. Edery. 2008. Global Quantitative Analysis of Protein Expression and Phosphorylation Status in the Liver of the Medaka Fish (*Oryzias latipes*) Exposed to Microcystin-LR I. Balneation Study. *Aquatic Toxicology*, 86:166-175.

Miao, S.L. and S. Carstenn. 2006. A New Direction for Large-scale Experimental Design and Analysis. *Frontiers in Ecology*, 4:227.

Miao, S.L. and W.F. DeBusk. 1999. Effects of Phosphorus Enrichment on Structure and Function of Sawgrass and Cattail Communities in the Everglades, pp. 274-299. In: Reddy, K.R., G.A. O'Connor, and C.L. Schelske. eds. Phosphorus Biogeochemistry in Subtropical Ecosystems. Lewis Publishers, Boca Raton, FL.

Miao, S.L. and C.B. Zou. 2009. Seasonal Variation in Seed Bank Composition and its Interaction with Nutrient Enrichment in the Everglades Wetlands. *Aquatic Botany*, 90:157-164.

Miao, S.L., S. Carstenn and M. Nungesser. 2009. Real World Ecology: Large-Scale and Long-Term Case Studies and Methods. Springer Science, New York, NY.

Newman, S., T.Z. Osborne, K. Rutchey, K.R. Reddy and S.E. Hagerthey. In prep. Anthropogenic Influences Drive Landscape Evolution: Multiple Regime Shifts in the Everglades.

Newman, S., S. Miao., S. Hagerthey, M. Cook, R.M. Price, T. Troxler, C. Thomas, C. Coronado, B. Gu, P. Sullivan, Y. Qian, E. Sindhoj, B. Bellinger, M. Kobza, K. Wheeler, M. Manna, R. Shuford, M. Jacoby, F. Dierberg and T. DeBusk. 2008. Ecosystem Ecology. Chapter 6: Ecology of the Everglades Protection Area. Sklar, T. Dreschel and K. Warren, eds. In: *2008 South Florida Environmental Report – Volume I*. South Florida Water Management District, West Palm Beach, FL.

Oehl, F., A. Oberson, S. Sinaj and E. Frossard. 2001. Organic Phosphorus Mineralization Studies using Isotopic Dilution Techniques. *Soil Science Society of America Journal*, 65:780-787.

Ogden, J.C. 1994. Status of Wading Bird Recovery. D.E. Gawlik, ed. In: *South Florida Wading Bird Report – Volume 3*, South Florida Water Management District, West Palm Beach, FL.

Ogden, J.C., S.M. Davis and L. Brandt. 2003. Science Strategy for a Regional Ecosystem Monitoring and Assessment Program: The Florida Everglades Example. Pp. 135-163 In: Monitoring Ecosytems. D.E. Busch and J.C. Trexler, eds. Island Press, Washington, D.C.

Parker, G.G., G.E. Ferguson and S.K. Love, et al. 1955. Water Resources of Southeastern Florida. Geological Survey Water-Supply Paper 1255, United States Government Printing Office, Washington, D.C.

Qian, Y., S.L. Miao, B. Gu and Y.C. Li. 2009a. Effects of Burn Temperature on Ash Nutrient Forms and Availability from Cattail (*Typha domingensis*) and Sawgrass (*Cladium jamaicense*) in the Florida Everglades. *Journal of Environmental Quality*, 38:1-15.

Qian, Y., S.L. Miao, B. Gu and Y.C. Li. 2009b. Estimation of Post-fire Nutrient Loss in the Florida Everglades. *Journal of Environmental Quality*, 38:1812-1820.

Reddy, K.R. and R.D. DeLaune. 2008. Biogeochemistry of Wetlands. CRC Press, Boca Raton, FL.

Reddy, K.R., R.D. DeLaune, W.F. DeBusk and M.S. Koch. 1993. Long-term Nutrient Accumulation Rates in the Everglades. *Soil Science Society of America Journal*, 57: 1147-1155.

Reddy, K.R., Y. Wang, W.F. DeBusk, M.M. Fisher and S. Newman. 1998. Forms of Soil Phosphorus in Selected Hydrologic Units of the Florida Everglades. *Soil Science Society of America Journal*, 62:1134-1147.

Reddy, K.R., G.A. O'Connor and C.L. Schelske, eds. 1999. Phosphorus Biogeochemistry in Subtropical Ecosystems. CRC Press, Boca Raton, FL.

Rivero, R.G., S. Grunwald, T.Z. Osborne, K.R. Reddy and S. Newman. 2007. Characterization of the Spatial Distribution of Soil Properties in Water Conservation Area 2A, Everglades, Florida. *Soil Science*, 172:149-166.

Roth, B.M., J.C. Tetzlaff, M.L. Alexander and J.F. Kitchell. 2007. Reciprocal Relationships Between Exotic Rusty Crayfish, Macrophytes and *Lepomis* Species in Northern Wisconsin Lakes. *Ecosystems*, 10: 74-85.

Rutchely, K., L. Vilchek, and M. Love, 2005. Development of a Vegetation Map for Water Conservation Area 3. Technical Publication ERA #421, West Palm Beach, FL, South Florida Water Management District.

Saha, A.K., L. da S.L. Sternberg and F. Miralles-Wilhelm. 2009. Linking Water Sources with Foliar Nutrient Status in Upland Plant Communities in the Everglades National Park, USA. *Ecohydrology*, 2:42-54.

Salierno, J.D., N.S. Snyder, A.Z. Murphy, M. Poli, S. Hall, D. Baden and A.S. Kane. 2006. Harmful Algal Bloom Toxins Alter c-Fos Protein Expression in the Brain of Killifish, *Fundulus heteroclitus*. *Aquatic Toxicology*, 78:350-357.

Scheidt, D.J. and P.I. Kalla. 2007. Everglades Ecosystem Assessment: Water Management and Quality, Eutrophication, Mercury Contamination, Soils and Habitat: Monitoring for Adaptive Management: A R-EMAP Status Report. EPA 904-R-07-001, USEPA, Region IV, Athens, GA.

Sklar, F.H., E. Cline, M. Cook, W.T. Cooper, C. Coronado, C. Edelstein, M. Ferree, H. C. Fitz, M.A. Furedi, P.B. Garrett, D. Gawlik, B. Gu, S.E. Hagerthey, R.M. Kobza, S. Miao, S. Newman, W.H. Orem, J. Palmer, K. Rutchey, E. Sindhoj, C. Thomas, J. Volin and N. Wang. 2007. Chapter 6: Ecology of the Everglades Protection Area. In: *2007 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Sklar, F.H., E. Cline, M. Cook, T. Dreschel, C. Coronado, D.E. Gawlik, B. Gu, S. Hagerthey, S. Lantz, C. McVoy, S. Miao, S. Newman, J. Richards, K. Rutchey, C. Saunders, L. Scinto, R. Shuford and C. Thomas. 2008. Chapter 6: Ecology of the Everglades Protection Area. In: *2008 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Sklar, F.H., E. Cline, M. Cook, T. Dreschel, C. Coronado, D.E. Gawlik, B. Gu, S. Hagerthey, S. Lantz, C. McVoy, S. Miao, S. Newman, J. Richards, K. Rutchey, R. Shuford and C. Thomas. 2009. Chapter 6: Ecology of the Everglades Protection Area. In: *2009 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Snyder, G.H. 1994. Chapter 3: Soils of the EAA. In: Everglades Agricultural Area (EAA), Water, Soil, Crop, and Environmental Management. A.B. Bottcher and F.T. Izuno, eds. University Press of Florida, Gainesville, FL.

Snyder, G.H. 2005. Everglades Agricultural Area Soil Subsidence and Land Use Projections, Pp. 44-51. In the 64th Proceedings of the Soil and Crop Science Society of Florida, Gainesville, FL.

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

SFWMD. 2007. Quick Facts On…Natural System Regional Simulation Model (NSRSM): Simulating South Florida's Pre-Development Hydrology. Splash! South Florida Water Management District, West Palm Beach, FL.

Sternberg, L., N. Ish-Shalom-Gordon, M. Ross, J. O'Brien. 1991. Water Relations of Coastal Plant Communities Near the Ocean/Freshwater Boundary. *Oecologia*, 88:305-310.

Sternberg, L. and P. Swart. 1987. Utilization of Freshwater and Ocean Water by Coastal Plants of Southern Florida. *Ecology*, 68:1898-1905.

Thacker, R.W., A.M. McLeod and S.W. McLeod. 2005. Herbivore-induced Saxitoxin Production in the Freshwater Cyanobacterium *Lyngbya wollei*. *Archiv fur Hydrobiologie*, 159:415-425.

Thomas, C.R., S.L. Miao and E. Sindhøj. 2009. In press. Environmental Factors Affecting Temporal and Spatial Patterns of Soil Redox Potential in Florida Everglades Wetlands. *Wetlands*.

Tolhurst, T.J., A.J. Underwood, R.G. Perkins and M.G. Chapman. 2005. Content Versus Concentration: Effects of Units on Measuring the Biogeochemical Properties of Soft Sediments. *Estuarine, Coastal and Shelf Science*, 63:665-673.

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.

Turner, A.M., J.C. Trexler, C.F. Jordan, S.J. Slack, P. Geddes, J.H. Chick and W.F. Loftus. 1999. Targeting Ecosystem Features for Conservation: Standing Crops in the Florida Everglades. *Conservation Biology*, 13:898-911.

Union of Concerned Scientists. 2007. Annual Global Carbon Dioxide Emissions. http://www.ucsusa.org/global_warming/science_and_impacts/science/each-countrys-share-of-co2.html

USEPA. 2007. Proposed Total Maximum Daily Load (TMDL) for Dissolved Oxygen and Nutrients in the Everglades (WBIDs 3252A, 3252B, 3252C, 3260, 3263, 3265B). Prepared by USEPA Region 4, Atlanta, Georgia.

USEPA. 2007. Annual Carbon Dioxide Inventory. http://www.epa.gov/climatechange/emissions/downloads/CO2FFC_2007.pdf

Valdor, R. and M. Aboal. 2007. Effects of Living Cyanobacteria, Cyanobacterial Extracts and Pure Microcystins on Growth and Ultrastructure of Microalgae and Bacteria. *Toxicon*, 49: 769-779.

Velzeboer, R.M.A., P.D. Baker, J. Rositano, T. Heresztyn, G.A. Codd and S.L. Ragget. 2000. Geographical Patterns of Occurrence and Composition of Saxitoxins in the Cyanobacterial Genus *Anabaena* (*Nostocales*, *Cyanophyta*) in Australia. *Phycologia*, 39: 395-407.

Wade, D.D., J. Ewel and R. Hofstetter. 1980. Fire in South Florida Ecosystems. General Technical Report SE-17. Southeastern Forest Experiment Station, U.S. Forest Service, U.S. Department of Agriculture, Asheville, NC.

Wetzel, R.G. 1999. Organic Phosphorus Mineralization in Soils and Sediments, Pp. 225-245. In: Phosphorus Biogeochemistry in Florida Ecosystems. K.R. Reddy, ed. Lewis Publishers, Boca Raton, FL.

Wetzel, P.R., A.G. van der Valk, S. Newman, D.E. Gawlik, T. Troxler-Gann, C.A. Coronado-Molina, 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*, 3:370-376.

Wetzel, P.R., A.G. van der Valk, S. Newman, D.E. Gawlik, C.A. Coronado-Molina, T. Troxler-Gann, D.L. Childers, W.H. Orem and F.H. Sklar. 2009. Heterogeneity of Phosphorus Distribution in a Patterned Landscape, the Florida Everglades. *Plant Ecology*, 200:83-90.

Whitney, E., D.B. Means and A. Rudlow. 2004. Priceless Florida: Natural Ecosystems and Native Species. Pineapple Press, Sarasota, FL.

Whitney, G.G. 1994. From Coastal Wilderness to Fruited Plain: A History of Environmental Change in Temperate North America from 1500 to the Present. Cambridge University Press, Cambridge, UK.

Wu, Y., K. Rutchey, W. Guan, L. Vilchek and F.H. Sklar. 2002. Spatial Simulations of Tree Islands for Everglades Restoration. pp 469-498. In: Tree Islands of the Everglades. F. Sklar and A. van der Valk, eds. Kluwer Academic Publishers, Boston, MA.