Chapter 6: Everglades Research and Evaluation

Edited by Fred Sklar and Thomas Dreschel

SUMMARY

The studies and findings discussed in this chapter are presented within four main fields: (1) wildlife ecology, (2) plant ecology, (3) ecosystem ecology, and (4) landscape patterns and ecology. Programs of study are 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, Everglades Forever Act (Section 373.4592, Florida Statutes [F.S.]) mandates, and the Comprehensive Everglades Restoration Plan (CERP). This chapter summarizes Water Year 2016 (WY2016) (May 1, 2015–April 30, 2016) hydrology in the Everglades Protection Area (EPA), followed by an overview of key Everglades studies on wildlife, plants, the ecosystem, and landscapes (**Table 6-1**).

Table 6-1. WY2016 Everglades research findings in relation to operational mandates.

Projects	Findings	Mandates[a]
Hydrology		
Hydrologic Patterns for WY2016	In the Everglades Protection Area, this dry season was the wettest on record since the 1957–1958 dry season. As a result, stages were exceptionally high during the dry season causing deep water stress for tree islands and lack of regular recession rates for wading birds. Similar patterns were observed in the Marl Prairies in the southern Everglades causing unsuitable nesting for the Cape Sable seaside sparrow. At the other end of the spectrum, Florida Bay experienced a Minimum Flows and Levels (MFL) Rule violation due to extremely low cumulative rainfall during the WY2016 wet season.	ROS MFL
Wildlife Ecology		
Wading Bird Nesting Patterns	An estimated 18,637 wading bird nests were initiated in the Everglades. This is 45 percent lower than the decadal average. All focal indicator species (white ibis, wood stork, great egret, snowy egret, and tricolored heron) exhibited reduced nesting effort in 2016 (between 23 and 74 percent below the decadal average).	ROS CERP MFL FEIM
Plant Ecology		
Trees: a Geomorphic Agent Governing Landscape Evolution	Studies of Loxahatchee Impoundment Landscape Assessment (LILA) tree islands highlight how transpiration can govern subsurface moisture dynamics, water–mineral interactions, soil porosity, and hydraulic conductivity. Evidence from constructed tree islands demonstrate the increases in aboveground biomass during early stand development triggers transpiration-driven ion (i.e., nutrient) accumulation in soils.	CERP EFA ROS

Table 6-1. Continued.

Projects	Findings	Mandates[a]
Deploying Tree Saplings in Peat Bags as a Potential Restoration Technique: A Field Study at LILA	Restoration of ghost tree islands may be possible with planting saplings in peat bags because it is low cost and provides about 1.5 feet (0.45 meter) of additional elevation and substrate for initial root growth. Peat bag planting experiments indicate that bags with one sapling, one fertilizer spike, and no additional floatation were most effective.	LTP ROS RS
Status of Everglades National Park (ENP) Mapping	The CERP vegetation mapping program has been focused on Big Cypress National Preserve and the ENP for the last 10 years. Base maps have been mosaicked from photos taken in 2006, 2009, and 2014. All accuracy assessment data were collected within one year of image acquisition. The ENP is 52 percent completed and Big Cypress National Preserve will be mapped starting in fall 2016.	CERP
Ecosystem Ecology		
Cattail Habitat Improvement Project (CHIP)	Soils data collected within CHIP highlight the importance of longer-term studies when evaluating the effect of restoration activities on biogeochemistry. Conclusions of elevated floc TP concentrations were only accurate for the first 3-4 years; 8-10 year post-study initiation, TP values were significantly decreased. The change in vegetation community, a primary driver of the change in nutrient storage, was enabled by the switch to a selective herbicide for cattail control. The CHIP study also highlighted the long-term use of these created openings by wading birds and waterfowl with significantly greater numbers of birds than those recorded in unenriched parts of the landscape.	LTP ROS RS
Florida Bay Water Quality Conditions and Status	The localized drought in South Florida that began in 2014 and continued in 2015 resulted in reduced freshwater flows to the bay. These flows rapidly increased during the very wet dry season beginning in late November 2015. In addition, a Temporary Emergency Deviation to the WCA-3A/3B Water Control Plan beginning on February 12, 2016, resulted in additional water flowing into ENP and likely Florida Bay. The higher nutrients and chlorophyll *a* (Chla) in the central region at the end of WY2016 may be attributable to the increased precipitation and freshwater inflows, the release of nutrients from decaying submerged aquatic vegetation (SAV), or a combination of both.	CERP MFL ROS
Ruppia in the Mangrove Transition Zone	Adult survival and clonal reproduction of new short shoots increases *Ruppia* coverage vegetatively, essential to maintenance of the population. But continued persistence of *Ruppia* in the transition zone appears to depend upon both vegetative production and recruitment of new adults from seed.	CERP MFL ROS
Florida Bay Benthic Vegetation	Within central and western Florida Bay, the elevated salinities and above average temperatures initiated a large seagrass die-off. Future monitoring will be essential to detect and assess any continuing and cascading effects of this die-off. The eastern bay appears much different for two reasons: (1) *Ruppia maritima* has not rebounded from the declines of WY2011–WY2012 and (2) *Thalassia testudinum* continues to increase downstream of many coastal creeks.	CERP MFL ROS
SAV Indicators for Florida Bay 2015–2016	The 2015 seagrass die-off indicates that the gains in the quality of SAV habitat over the past several years are precarious and can be reversed within an annual timescale. The Composite SAV Index indicated that the SAV status for WY2015–2016 regressed in two areas: the Transition Zone and the Central Bay Zone.	CERP MFL ROS

Table 6-1. Continued.

Projects	Findings	Mandates[a]
Landscape		
Tree island Stability and Resilience Assessed through Integrated Sediment Elevation Tables (SETs)	Determining elevation differences between tree islands and their surrounding marsh-slough environment is important to provide a better understanding of how water management decisions affect the ecological integrity of the ridge-slough-tree island ecosystem. These data of the shallow SETs indicate that the active root zones of three hydrologically different tree islands are not producing enough root biomass to compensate for soil oxidation, and even though deposition of material on the ground surface of these islands can be significant, ranging between 2.5 and 6.1 millimeters per year (mm/yr), none of these islands are building elevation.	CERP ROS
S-356 Pump Test: Ecological Monitoring	(1) A fluorescein dye study indicated a surface water movement to the east from culverts east of the One-mile Bridge. West of the One-mile Bridge, flows were towards Shark River Slough. (2) Surface and bottom water measurements of salinity, conducted in Manatee Bay and Barnes Sound showed that following an initial large release from the S-197 structure, subsequent small releases did not appreciably reduce salinity further, though the cumulative effect was to maintain reduced salinity throughout the dry season.	CERP MFL ROS MFL
Interim Findings of the Decomp Physical Model (DPM)	This third DPM flow event demonstrated that (1) experimental sheetflow treatments increased water velocities and sediment transport in sloughs compared to ridges; (2) sustained flow operations of 8 to 10 weeks, rather than multiple pulses, are needed to maximize slough velocities, sediment transport, and sediment redistribution; (3) if just 5 percent of transported slough sediments entered ridges, this could theoretically increase sediment accumulation by 100 percent; and 4) there is a potential shift in algal taxonomy, abundance and biomass with increasing flow rates.	CERP MFL ROS

a. Mandates

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

HYDROLOGY

Rainfall in the Everglades for WY2016 was exceptionally low in the early part of the water year, especially in Florida Bay causing the 5-creek cumulative flow to dip below the 105,000 acre-feet (ac-ft) Minimum Flows and Levels (MFL) Rule threshold by some 27,000 ac-ft. and causing central Florida Bay salinities to go hypersaline. The dry wet season was followed by an exceptional wet dry season beginning in December 2015. Over the entire District, rainfall was nearly one foot (ft) above average from November through May. As a result, for most of the entire Greater Everglades, tree islands experienced deep water stress, wading bird foraging depths were too deep, and there was a lack of regular recession rates necessary to provide prey for wading birds.

WILDLIFE ECOLOGY

An estimated 18,637 wading bird nests were initiated in the water conservation areas (WCAs) and Everglades National Park (ENP) during the 2016 nesting season (December to July). This nesting effort is 45 percent lower than the decadal average (33,781.4 nests) and 35 percent lower than the five-year average (28,750.6 nests). Most species exhibited relatively limited nesting effort in 2016. White ibis (*Eudocimus albus*, 10,545 nests), the most numerous wading bird species nesting in the Everglades, reduced nesting effort by 47 and 42 percent relative to the decadal and five-year averages (**Figure 6-13** later in this report). Numbers of great egret (*Ardea alba*) and wood stork (*Mycteria americana*) nests were down 23 and 59 percent, respectively, from the 10-year average, and down 20 and 61 percent from the 5-year average. The smaller *Egretta* herons, which have exhibited a consistent decline in nest numbers in recent years, had another relatively poor nesting season: nesting effort was reduced 74 percent for the snowy egret (*Egretta thula*) relative to the ten-year average, while tricolored heron (*Egretta tricolor*) nesting was down 74 percent.

PLANT ECOLOGY

Three plant ecology studies are described in this section. (1) The Loxahatchee Impoundment Landscape Assessment (LILA) research facility continues to be a fountainhead of groundbreaking science. This year by comparing the characteristics of groundwater movement, and nutrient and mineral accretion for constructed tree islands of differing substrate, we found that as biomass accretes on the islands, the buildup of ions alters the groundwater geochemical conditions in favor of mineral formation. Hence, trees on tree islands have the ability to build soil by concentrating minerals and contribute to Everglades oligotrophy. (2) Another LILA study is an experimental approach for creating artificial floating peat islands to eventually establish on ghost tree islands. This sapling-in-a-peat-bag approach is simple, low cost, adds about 1.5 ft (0.45 meter [m]) of additional elevation, and provides a substrate for the initial root growth of the planted tree saplings. Results indicate that pond apple (*Annona glabra*) may be the most efficient sapling and the planting approach is best if one sapling, one nutrient spike, and no floatation per bag is implemented. (3) The vegetation mapping status for the seven regions associated with ENP and the Big Cypress National Preserve (BCNP) is described. Vegetation maps provide a mechanism by which managers can track and document the rate and direction of vegetation changes in relation to restoration implementation and climate variability. Base maps have been mosaicked from photos taken in 2006, 2009, and 2014. All accuracy assessment data were collected within one year of image acquisition. The ENP is 52 percent completed and BCNP will be mapped starting in fall 2016.

ECOSYSTEM ECOLOGY

This section is an update on the status of (1) the Cattail Habitat Improvement Project (CHIP), (2) the Florida Bay project and (3) the Tree Island Restoration project. CHIP, designed to provide a better understanding of ways to functionally improve impacted wetlands, found further evidence that burning, combined with herbicide application, provides immediate open water habitat available for submerged aquatic vegetation (SAV) and foraging habitat for wading birds. In these high nutrient areas, regrowth of undesirable vegetation from the seedbank and adjacent community means that repeated herbicide application may be necessary, at least initially, to ensure long-term success. Enriched habitats that are opened by this process are sustainable in the long-term, support significantly more wading birds, and could play a critical role as refugia for foraging birds when hydrological conditions preclude effective foraging elsewhere in the Greater Everglades.

The Florida Bay project reports on four elements this year: (1) water quality, (2) *Ruppia* in the mangrove transition zone, (3) benthic vegetation, and (4) SAV Indicators for 2015–2016. Water quality in Florida Bay (eastern, central, and western regions) has been monitored since 1991 (WY1992). Notable water quality findings for WY2016 include the fact that all three regions of the bay had elevated discrete

salinities during the first half of the water year followed by lower salinities during the second half of the water year. The extended drought in this area which began in WY2014 and stretched into WY2016 abated with the heavy precipitation in November 2015 through January 2016. Chlorophyll a (Chla), total phosphorus (TP), total organic carbon (TOC), and turbidity across the bay were at the long-term median. Dissolved inorganic nitrogen (DIN) was elevated baywide. It will be critical to monitor these water quality parameters especially in the central bay, which experienced the SAV die-off.

As part of Florida Bay research on SAV in the mangrove transition zone, a series of field experiments examined the *Ruppia* community to determine factors that favor recruitment of seedlings to the adult plant population from sexual reproduction, seed germination, seedling, and adult survival, and clonal reproduction of new short shoots. It was found that adult *Ruppia* can tolerate polyhaline environments for a limited time but because seed germination and seedling survival depend on a low salinity phase, the transition from seed to adult represents a "bottleneck" to recruitment of adult plants. Continued persistence of *Ruppia* in the transition zone appears to depend upon both vegetative production and recruitment of new adults from seed. Status indicators for SAV, calculated each year for CERP's Restoration Coordination and Verification Program (RECOVER) and the South Florida Ecosystem Restoration Task Force, showed a decline for some areas in WY2014–WY2017 relative to prior status, likely the result of low rainfall and freshwater input to the bay beginning in 2014.

As part of the Tree Island Restoration project, sediment erosion tables (SETs) and feldspar marker horizons were distributed onto two islands in WCA-3A and one in WCA-3B as tools for evaluating the influence of hydrology on tree island elevation change, subsidence, and accretion. These shallow SET data indicate that the active root zones of three hydrologically different tree islands are not producing enough root biomass to compensate for soil oxidation, and even though deposition of material on the ground surface of these islands can be significant, ranging between 2.5 and 6.1 millimeter per year (mm/yr), none of these islands are building elevation.

LANDSCAPE

Two landscape-scale projects are reported this year. (1) As part of the Modified Water Deliveries to Everglades National Park project, SFWMD in cooperation with ENP, is incrementally modifying deliveries of water to Shark River Slough. This is referred to as the S-356 Pump Test. As part of this test, we utilized high resolution aerial mapping in conjunction with the release of a fluorescent dye to evaluate flow direction and water quality sentinel station locations, and found that surface water movement from culverts east of the One-mile Bridge, moved to the east in step with groundwater seepage. However, west of the One-mile Bridge, flows were towards Shark River Slough. As part of the S-356 Pump Test, we also conducted an evaluation of the downstream salinity changes associated with the opening and closing of the S-197 structure in the coastal habitats of Barnes Sound and Manatee Bay, and found that following an initial large release from the S-197, subsequent small releases did not appreciably reduce salinity further, though the cumulative effect was to maintain reduced salinity throughout the dry season.

(2) The largest and most cutting edge landscape science in the Everglades continues to be the Decomp Physical Model (DPM). The DPM, the second largest adaptive management restoration program in United States history, examines the ecological significance of restoring sheetflow between WCA-3A and WCA-3B. This was the third DPM flow event (November 2015–January 2016), which was enhanced by a state high water emergency order requiring that the 10-culvert DPM structure (the S-152) reopen from February 19–23 and from March 9–May 3. The latter operations provided opportunities to test dry season flow impacts and a larger flow window (17 weeks). Studies included flow distributions, sediment characteristics, biomarkers as tracers, floc characteristics, the use of a dual-use synthetic tracer (DST) and horizontal sediment traps to evaluate sediment transport, nutrient dynamics, water quality changes, algal community responses to flow, and large-scale periphyton changes. This year, DPM is highlighted by a conceptualization of the linkages and feedbacks associated with the biogeochemical responses to flow. Major findings were (1) experimental sheetflow treatments increased water velocities and sediment

transport in sloughs compared to ridges; (2) sustained flow operations of 8 to 10 weeks, rather than multiple pulses, are needed to maximize slough velocities, sediment transport, and sediment redistribution; (3) if just 5 percent of transported slough sediments entered ridges, this could *theoretically* increase sediment accumulation by 100 percent; and (4) there is a potential shift in algal taxonomy, abundance and biomass with increasing flow rates.

HYDROLOGIC PATTERNS FOR WATER YEAR 2016

WATER CONSERVATION AREAS AND NORTHEAST SHARK RIVER SLOUGH HYDROLOGY

Martha K. Nungesser and Fred Sklar

Rainfall in the Everglades for WY2016 was low in the early part of the water year and unseasonably high beginning in December 2015, the usual dry season. Consequently, differences for the year from the basin historical annual averages (**Table 6-2**) were not great: 3.55 to 3.60 inches above average in WCA-1, WCA-2, and WCA-3 and 3.43 inches below average in ENP. Most of the heavy dry season rainfall occurred in eastern ENP, resulting in high stages in Taylor Slough and Shark River Slough. In the EPA, this dry season was the wettest on record since the 1957–1958 dry season. Over the entire District, rainfall was nearly one ft above average from November through May. As a result, stages were exceptionally high during the dry season for many areas in the Everglades, causing issues with the tree islands (deep water) in winter, lack of regular recession rates necessary to provide prey for wading birds, and high water levels that prevented normal nesting by the endangered Cape Sable seaside sparrows.

The skewed rainfall pattern (low in the wet season, high in the dry season) was reflected somewhat in this year's annual average stages (**Table 6-2**). WCAs 1 and 3 and ENP were above average by 0.6, 0.33, and 0.4 ft, respectively, and only WCA-2 was below average by 0.13 ft.

Table 6-2. Average, minimum, and maximum stage in ft National Geodetic Vertical Datum of 1929 (NGVD29) and total annual rainfall in inches for WY2016 in comparison to historic stage and rainfall.[a] (Average depths calculated by subtracting elevation from stage.)

Area	WY2016 Rainfall	Historic Rainfall	WY2016Stage Mean (Minimum, Maximum)	Historic Stage Mean (Minimum, Maximum)	Elevation
WCA-1	55.56	51.96	16.29 (14.38, 17.58)	15.69 (10.0; 18.16)	15.1
WCA-2	55.56	51.96	12.38 (11.35; 14.4)	12.51 (9.33; 15.64)	11.2
WCA-3	54.79	51.24	9.93 (8.65; 11.5)	9.60 (4.78; 12.79)	8.2
ENP	51.05	54.48	6.42 (5.45; 7.16)	6.02 (2.01; 8.08)	5.1

a. Historical averages are based upon varying lengths of records at gauges. See Chapter 2 of this volume for a more detailed description of rain, stage, inflows, outflows, and historic databases.

The following figures of hydropatterns at select gauges highlight the average stage changes in each of the WCAs for the last 2.5 years. The figures relate stages relative to historic averages, flooding tolerances for tree islands, drought tolerances for wetland peat, and recession rates and depths that support foraging and nesting needs of wading birds during the breeding season. The District uses these indices as part of the ecological recommendations to water managers at weekly interagency water operations meetings. Tree island inundation tolerances are considered exceeded when depths on the islands are above 2.0 or 2.5 ft,

depending on the height of the tree islands, for longer than 120 days (Wu et al. 2002). Lower islands are inundated at lower high water levels (e.g., 2 ft versus 2.5 ft). The ground elevations in **Figures 6-1** through **6-8** are used to indicate the threshold for peat conservation. When water levels are more than one foot below ground for more than 30 days, the drought tolerance of peat is considered exceeded according to the criterion for the Everglades' MFL (SFWMD 2014). Peat may be harmed at shallower water levels, however.

The wading bird nesting period ranges from January through May each year. The suitability of habitat is determined by water depths and recession rates and is divided into three categories (poor, moderate, and good) according to recent research on foraging requirements of wading birds in the Everglades (Gawlik 2002, Beerens et al. 2011, 2015, Cook 2014). A green arrow on the hydropattern figures indicates the period of time with good recession rates and depths for wading birds. A yellow arrow indicates water levels that are too shallow or too deep and/or recession rates that are slightly too rapid or too slow. A red arrow indicates poor conditions resulting from poor depths (high or low) and/or unsuitable recession rates (rising or falling too rapidly).

Water Conservation Area 1

The water levels in WCA-1 (Arthur R. Marshall Loxahatchee National Wildlife Refuge) at the start of WY2016 were close to their 21-year average (**Figure 6-1**). Rather than rising with the onset of the wet season, stages continued to decline with the unusually dry wet season, reaching approximately 0.75 ft below average, after which stages rose beginning in July. Only in October did stages reach their long-term average, and from then through the end of the water year, water levels exceeded the 21-year average. Recession rates during the wading bird breeding season (from January through May) were fair to poor in the first couple months, then improved to a long-term good recession rate. The upper tolerance for tree islands was exceeded briefly in the fall and winter, which is not typical of high water levels in South Florida. These brief excursions are not likely to have affected the tree island plant communities. Sensitivity to extended inundation duration depends largely on the species comprising the forest communities on tree islands and probably the time of year that flooding occurs.

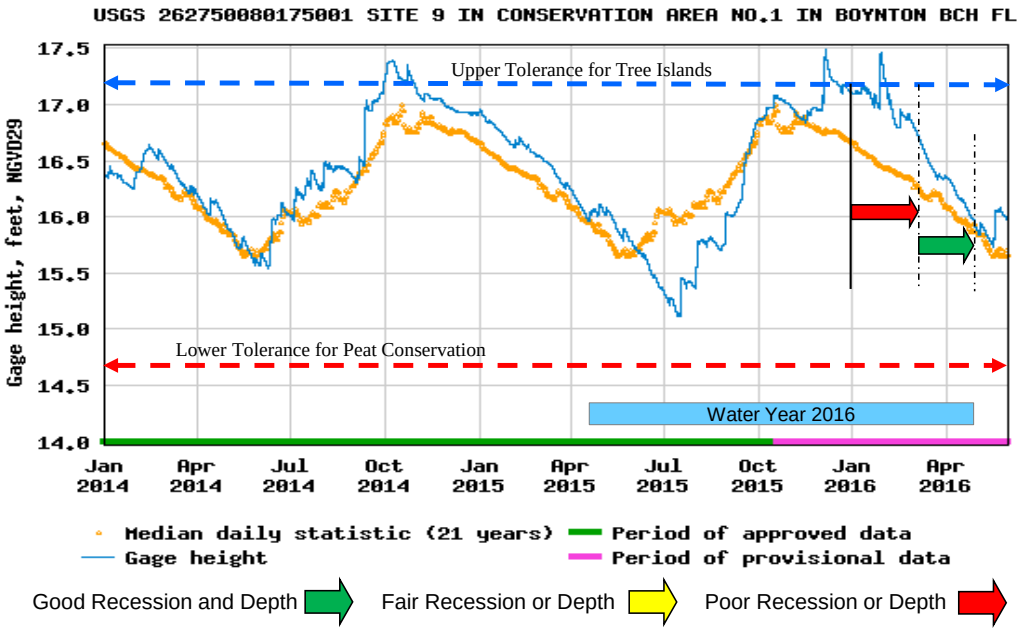


Figure 6-1. Hydrology in WCA-1 in relation to the 21-year median daily stage, as well as indices for tree island flooding, peat conservation, and wading bird foraging. (Note: NGVD29 – National Geodetic Vertical Datum of 1929 and USGS – United States Geological Survey.)

Water Conservation Area 2A and 2B

Similar to the hydropattern in WCA-1, stages during the WY2016 wet season began close to the long-term average. In July, water levels began to rise through October to stages that exceeded the upper flood tolerance for tree islands (**Figure 6-2**), then decreased and rose again in December, remaining above the 21-year median depths for the rest of the water year. In February, water depths exceeded 3 ft briefly, and then continued to exceed the tolerance for tree islands for more than a month in the dry season. This year's pattern was very unusual with high to elevated water levels throughout most of the dry season. Recession rates were good for wading birds in January, followed by very high reversals with deep water and rapid recession rates through March. The water year concluded with good depths and recession rates through the end of April. Typically, WCA-2A provides later season foraging for wading birds as the adjacent areas dry out as water stages recede. This year provided several short periods of suitable good foraging periods, but no extended foraging conditions needed to support nesting.

Because of high water, Florida Fish and Wildlife Conservation Commission (FWC) closed most of WCA-3 and all of WCA-2A to protect wildlife. As a result, scientists were unable to survey the tree islands to determine whether high water in the winter harmed the plant communities.

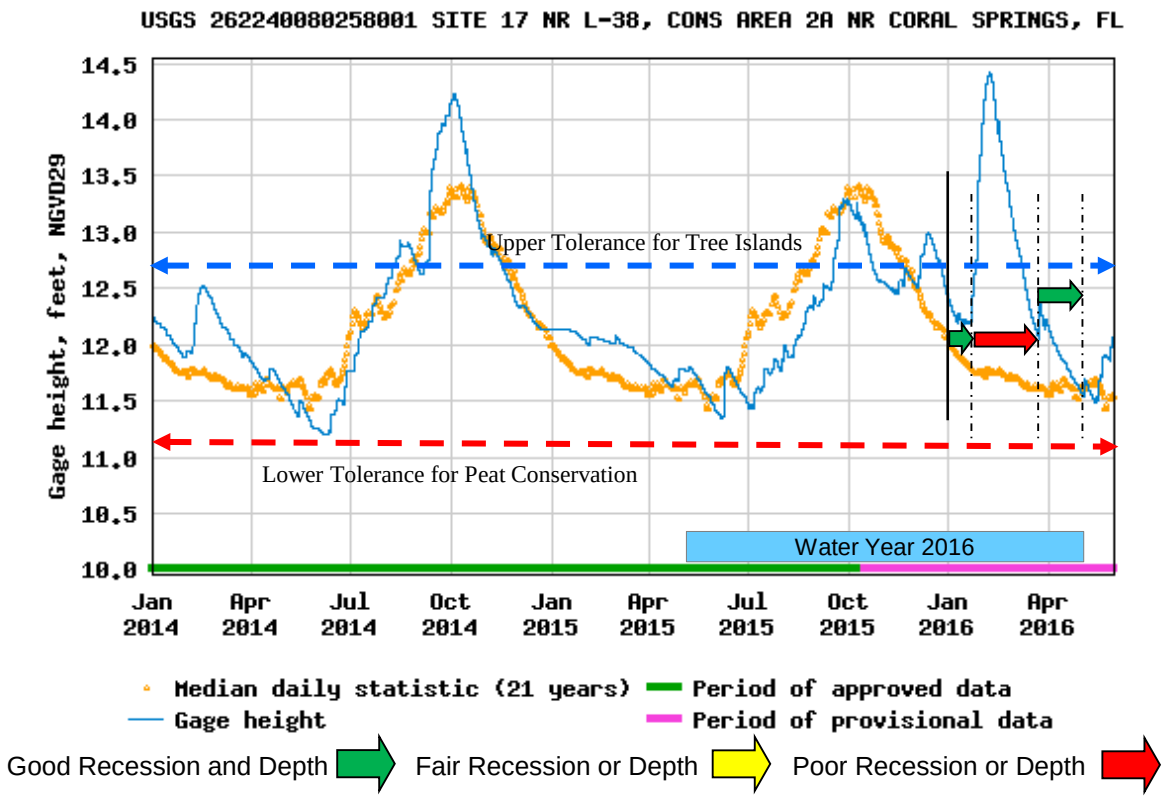


Figure 6-2. Hydrology in WCA-2A in relation to the recent 21-year average, with indices for tree islands, peat conservation, and wading bird foraging.

Unlike the rest of the EPA, WCA-2B tends to be wet and deep most of the year (**Figure 6-3**). However, during WY2016, water depths at gauge 99 fell rapidly to ground level by early July, then by December rose to stages that exceeded the median water levels and remained above this median for the rest of the water year (the red arrow in **Figure 6-3** signifies water levels too deep to support wading birds).

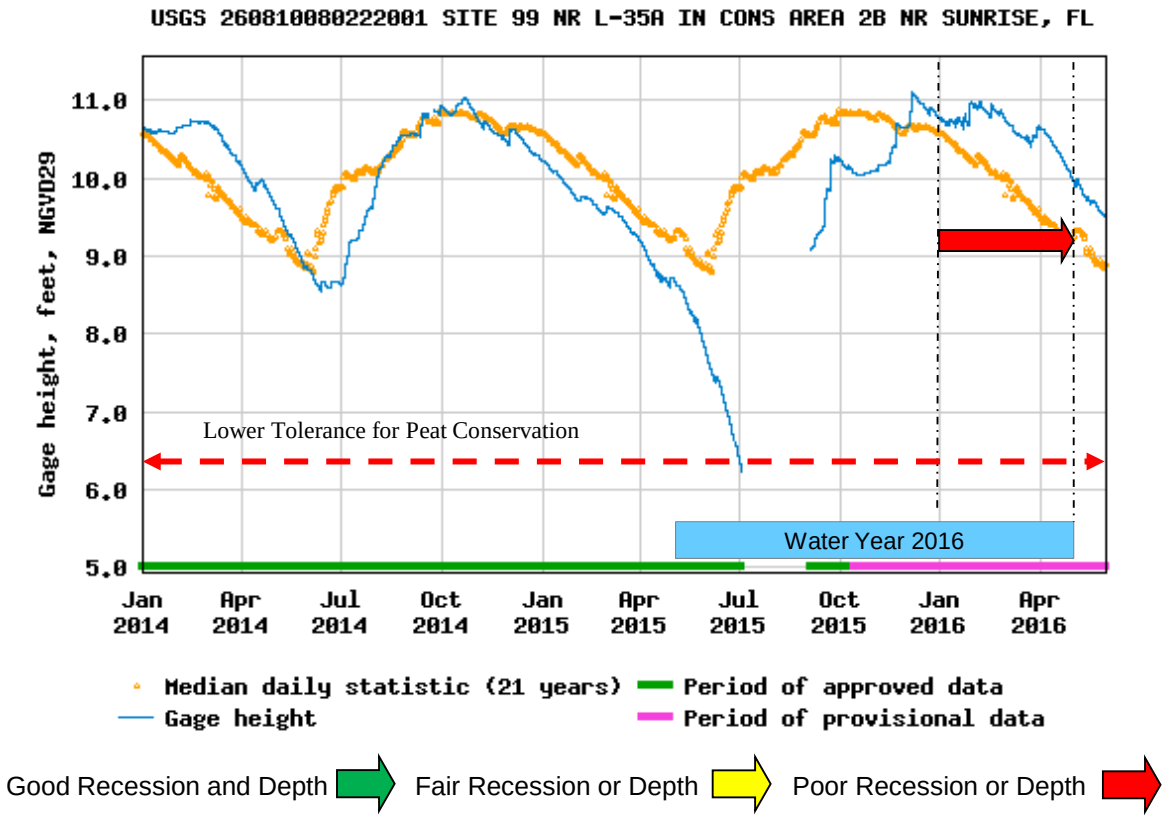


Figure 6-3. Hydrology in WCA-2B (gauge 99) in relation to the recent 21-year average with indices for tree islands, peat conservation, and wading bird foraging.

Water Conservation Area 3A

Water levels in northeastern WCA-3A at gauge 63 during WY2016 were very different from those of the previous two years (**Figure 6-4**). Stages were at ground level at the beginning of May and declined, remaining below ground until very late August, then rose quickly through early October. In December, stages exceeded the median water levels and remained very high through the end of the water year. As in several other regions, tree island tolerance levels were exceeded for several months in the dry season, with recession rates and depths exceeding preferred dry season conditions. January through March experienced large reversals, several periods of fair recession rates, and long periods of poor recession rates and depths. This gauge is near the Alley North colony, one of the most productive for wading birds in the Everglades.

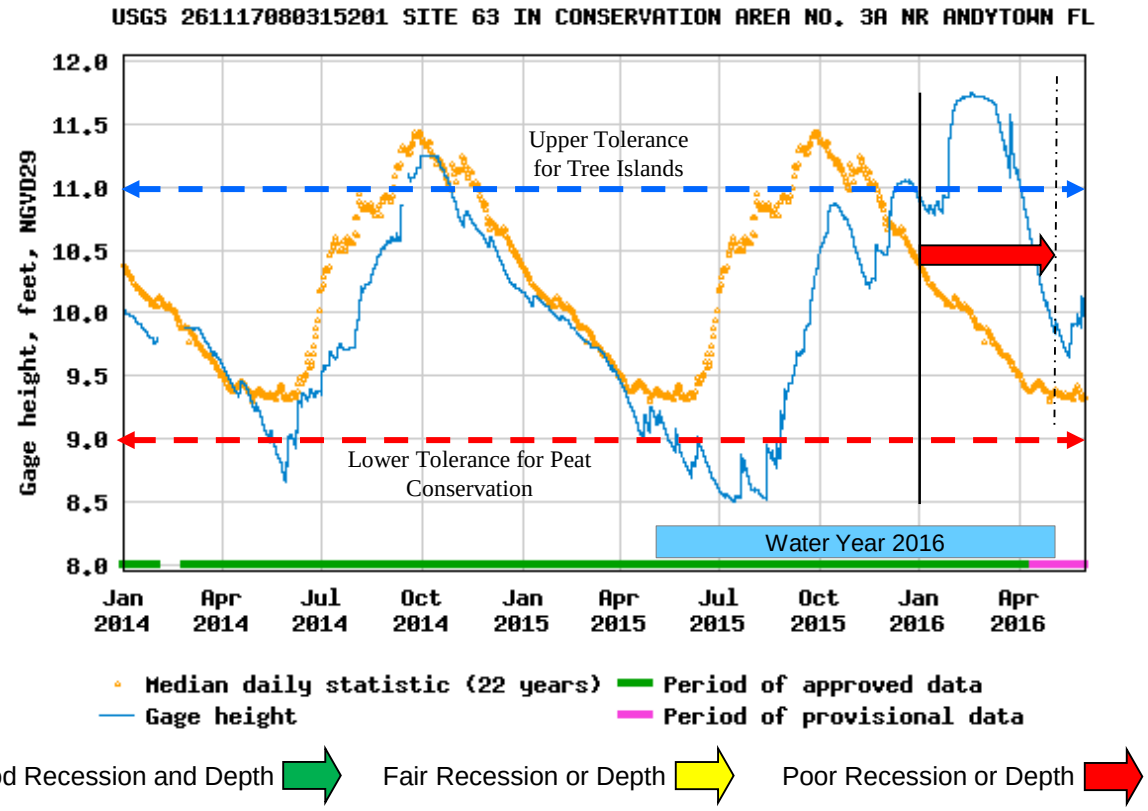


Figure 6-4. Hydrology in northeast WCA-3A (gauge 63) in relation to the recent 22-year average with indices for tree islands, peat conservation, and wading bird foraging.

The hydrologic pattern in central WCA-3A (gauge 64) (**Figure 6-5**) was similar to that at gauge 63 except water depths remained above ground. Water levels rose so that in December, they exceeded the 22-year median depths, peaking in mid-February at more than 2.6 ft deep, exceeding 2.5 ft depths (tree island metric) for eight weeks in the winter. Throughout much of the dry season, water levels were deep and the recession rates and reversals produced poor conditions for wading birds as well as for tree islands. The lower water levels during the previous water year (WY2015) were probably beneficial to the tree island plant communities during WY2016.

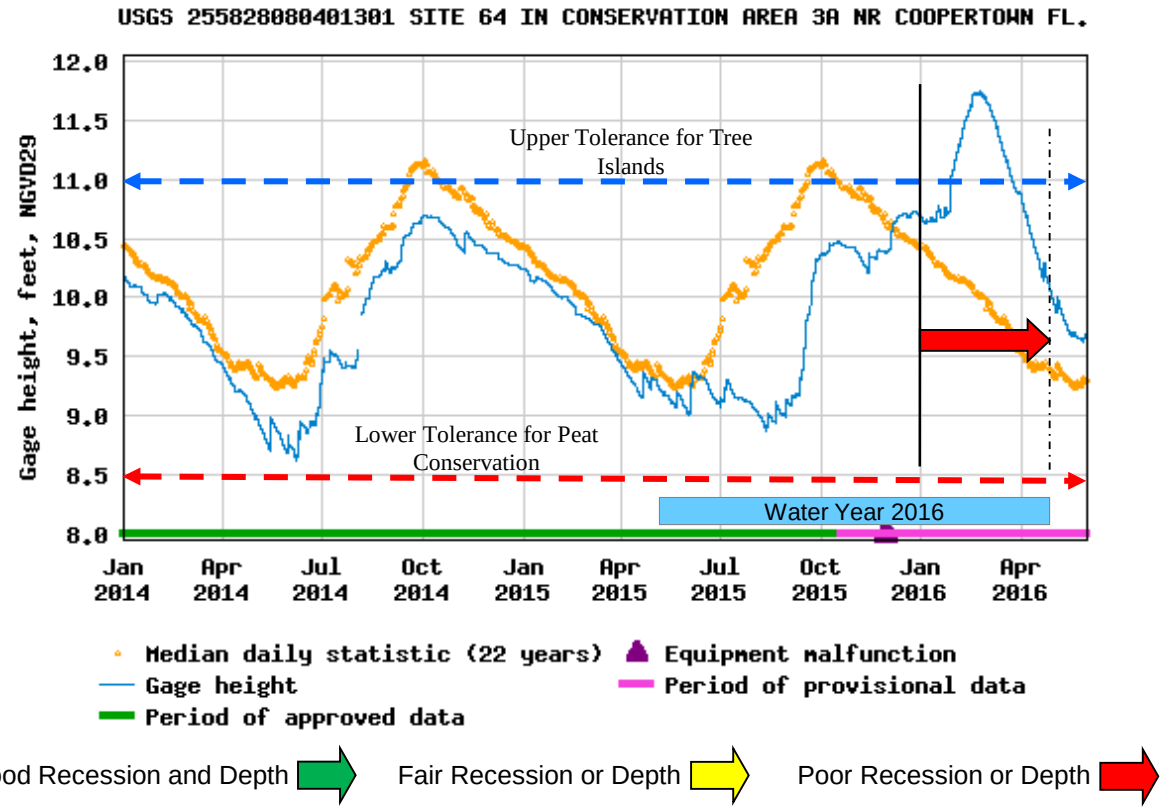


Figure 6-5. Hydrology in central WCA-3A (gauge 64) in relation to the recent 22-year average with indices for tree islands, peat conservation, and wading bird foraging.

Water Conservation Area 3B

As in other parts of the Everglades, water levels declined steadily until mid-August, when water levels began to rise at a rate parallel to the usual ascension rates that begin earlier in the wet season (**Figure 6-6**). In December, stages continued to rise and exceeded the long-term median wet season depths in this unusual dry season. At gauge 71, this dry season reversal continued from December through the end of February, when stages began to decrease again, producing a good recession rate through the end of the water year; however, depths were too high for wading bird foraging. While water was high, it did not exceed depths that are considered stressful for tree islands.

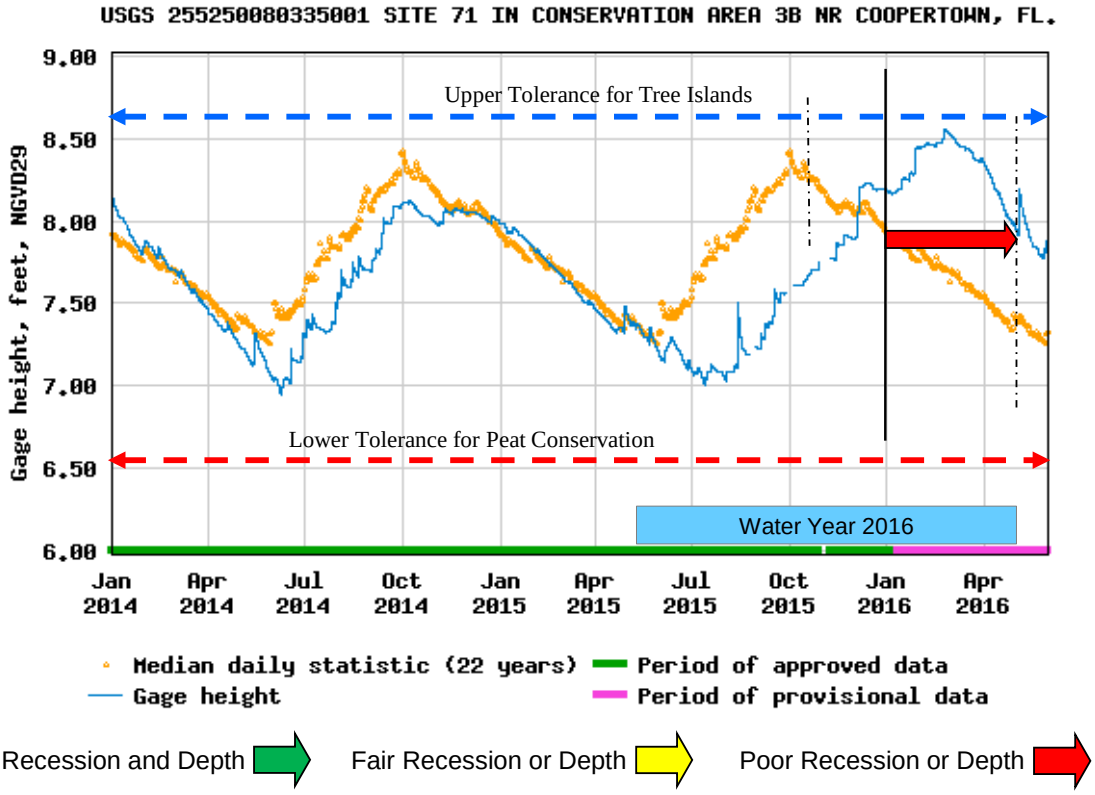


Figure 6-6. Hydrology in central WCA-3B (gauge 71) in relation to the recent 22-year average and indices for tree islands, peat conservation, and wading bird foraging.

Northeast Shark River Slough

At the beginning of WY2016, water levels dropped to stages about 1 ft below ground (**Figure 6-7**). In mid-July, water levels rose to ground levels and continued to rise, exceeding the 32-year median stage in early October. Water remained above ground through the end of the water year, providing high water levels and few good recession rates. Conditions at this gauge were very different this water year relative to previous years, when conditions are usually very dry at the end of the dry season, preventing conditions that favor a good prey base for foraging and support for apple snail breeding.

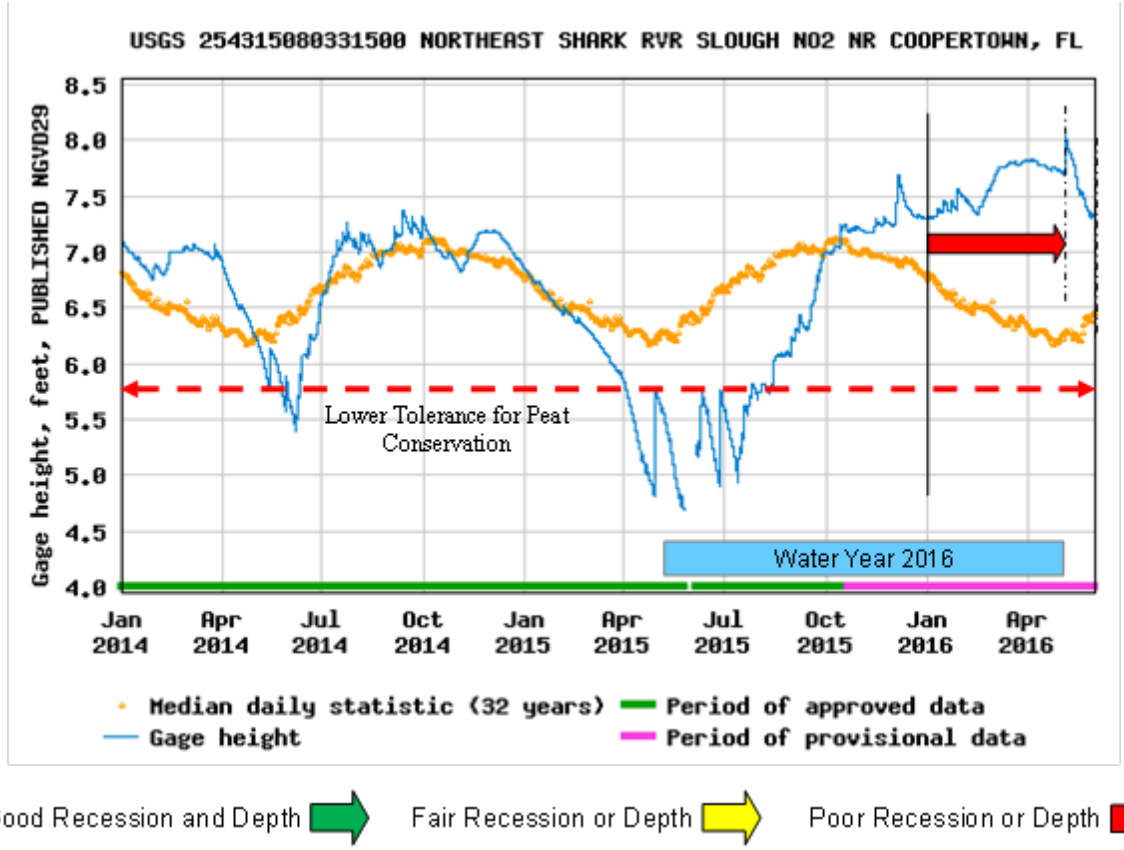


Figure 6-7. Hydrology in Northeast Shark River Slough in relation to the recent 32-year average with indices for tree islands, peat conservation, and wading bird foraging.

EVERGLADES NATIONAL PARK: MARL PRAIRIES

The marl prairies of ENP differ greatly from the peatlands of the WCAs. These are shorter hydroperiod wetlands supporting mixed graminoid and short hydroperiod wet prairie vegetation, and are particularly important for their support of the endangered Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*). Gauge NP-CR3 (**Figure 6-8**) is typical of this habitat.

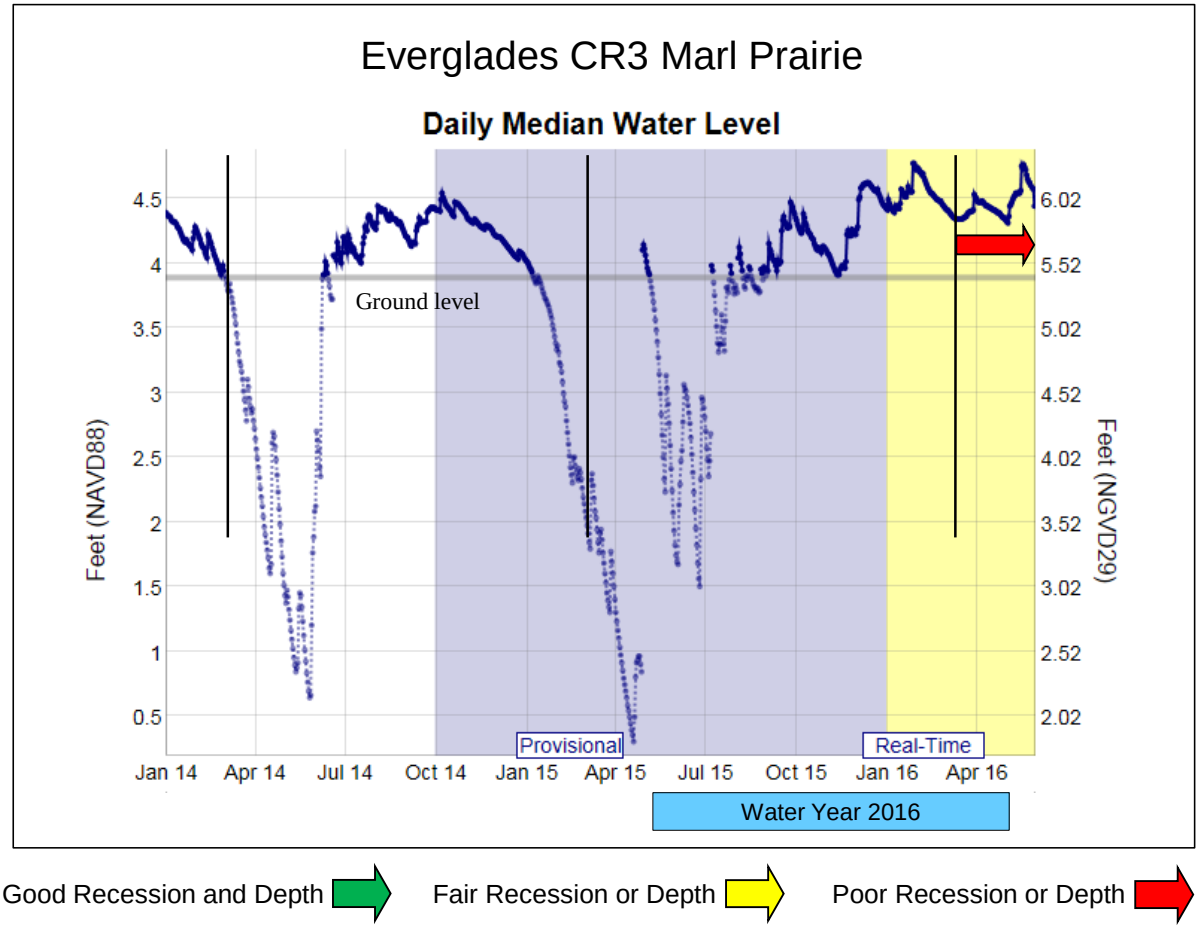


Figure 6-8. Hydrology in the Marl Prairies in ENP from January 1, 2014, through May 31, 2016. The gray horizontal line represents ground level. Water levels above four inches are harmful to the nests of the Cape Sable seaside sparrow.

WY2016 began with very dry conditions. Although it was rising, water in the marl prairie site (NP-CR3) was over 3.5 ft below ground on May 1. Water levels then rose rapidly to ground level, dropped again until late in the wet season, and remained above ground through the rest of the very wet dry season. At this time in the dry season, water is usually below ground, as it was in the prior two years, providing opportunities for the Cape Sable seaside sparrow to breed in nests close to the ground. However, this dry season produced water depths at gauge NP-CR3 that were 0.5 ft to 0.7 ft above ground from January through the end of the water year on April 30, 2016. These depths provided poor breeding and nesting conditions for the sparrows.

Preferred habitat for the endangered Cape Sable seaside sparrow is this short hydroperiod (180–240 days) wet prairie vegetation. They breed from early March through at least the middle of July so long as conditions remain favorable. Water levels need to remain below ground with enough damp areas to provide

aquatic insects for their feeding. Sparrows nest close to the ground in clumps of bunch grasses. Dry conditions extending 60 to 90 days or more are needed to provide adequate time to nest and fledge chicks and when conditions remain favorable, sparrows may produce two to three nests in a year. The extended dry periods in both WY2014 and WY2015 were favorable for the Cape Sable seaside sparrow nesting in the subpopulations monitored by SFWMD and ENP. However, conditions were poor in WY2016 because of high rainfall and the ensuing high water depths.

FLORIDA BAY WATERSHED HYDROLOGY AND SALINITY

Amanda McDonald

The District is part of a multi-agency partnership that maintains a long-term monitoring network throughout the southern Everglades and Florida Bay to provide a continuous record of salinity, rainfall, temperature, water elevation, and flow for the Florida Bay watershed. These data are used in support of District operations, in developing and monitoring CERP performance measures, assessing effectiveness of MFLs, and supporting calibration and verification of hydrologic models.

Methods

Salinity, rainfall, temperature, and flow are monitored at instrumented platforms with salinity, rainfall, and temperature reported every hour and flows measured every 15 minutes. Rainfall and most salinity and temperature platforms are maintained by ENP and are located to allow determination of upstream to downstream effects (**Figure 6-9**). Flow has been measured by the United States Geological Survey (USGS) at five creeks discharging into Florida Bay since WY1997. Rainfall was quantified by averaging daily data across 11 ENP platforms (CP, DK, EPSW, JK, LM, P37, RPL, TB, TC, TR, and WB) and then summing to produce monthly totals for the period of WY1996 to WY2016, which is the period for which all 11 platforms existed. Average daily water temperatures across 8 ENP platforms (DK, GB, JK, LM, TB, TC, TR, and WB) within Florida Bay were also summarized for the corresponding period of WY1996 to WY2016. For the purposes of this assessment, salinity data are truncated to the period of WY2001 to WY2016 to coincide with the installation and operations of S-332D, which was a significant change to the management of the South Dade Conveyance System and water movement towards Florida Bay.

Results

Low rainfall conditions from WY2015 continued into WY2016 with the cumulative rainfall (since the beginning of WY2015) falling below the 25th percentile (period of record [POR] used was WY1996 to WY2014) in August 2014 and below the 10th percentile in June 2015 (**Figure 6-10**). Cumulative rainfall remained below the 10th percentile until the unusually wet December 2015. The low rainfall was accompanied by high water temperatures. WY2016 water temperatures were above the WY1996–WY2014 75th percentile for much of the year (June through December 2015).

The low rainfall contributed to low flows to Florida Bay. The 365-day cumulative flow to Florida Bay from the 5 monitored creeks reached a record low of 77,644 ac-ft in August 2015 (**Figure 6-11**). As a result of low rainfall, low flows, and high temperatures, salinities within Florida Bay remained elevated through summer WY2016. At TR, the sentinel site for the Florida Bay MFL Rule, the 30-day moving average salinity exceeded 30 for the second year in a row. Downstream in Florida Bay, salinities were higher than 50 in central and western Florida Bay (**Figure 6-12**). High rainfall during the dry season starting in November rapidly decreased salinities allowing all areas to end the water year below average.

Relevance to Water Management

Below average rainfall that was lower than the 10th percentile (an 1-in-10 year drought) continued to cause low creek flow to Florida Bay and elevated salinities through summer WY2016. As a result, the 30-day moving average at the Florida Bay MFL compliance site exceeded the 30 30-day moving average salinity threshold twice in consecutive calendar years within a 10-year period (Calendar Year 2008 [CY2008]–CY2009 and CY2014–CY2015), which is a violation of the MFL. A second criterion for the Florida Bay MFL is that the 365-day moving sum of cumulative flow from the 5 monitored creeks cannot go below 105,000 ac-ft. In July 2015, the 365-day moving sum of flow decreased below the 105,000 ac-ft for the first time since the adoption of the MFL. The MFL criteria are thresholds beyond which significant ecological harm is expected to occur. A large scale seagrass die-off that occurred during late summer CY2015 is described later in this chapter.

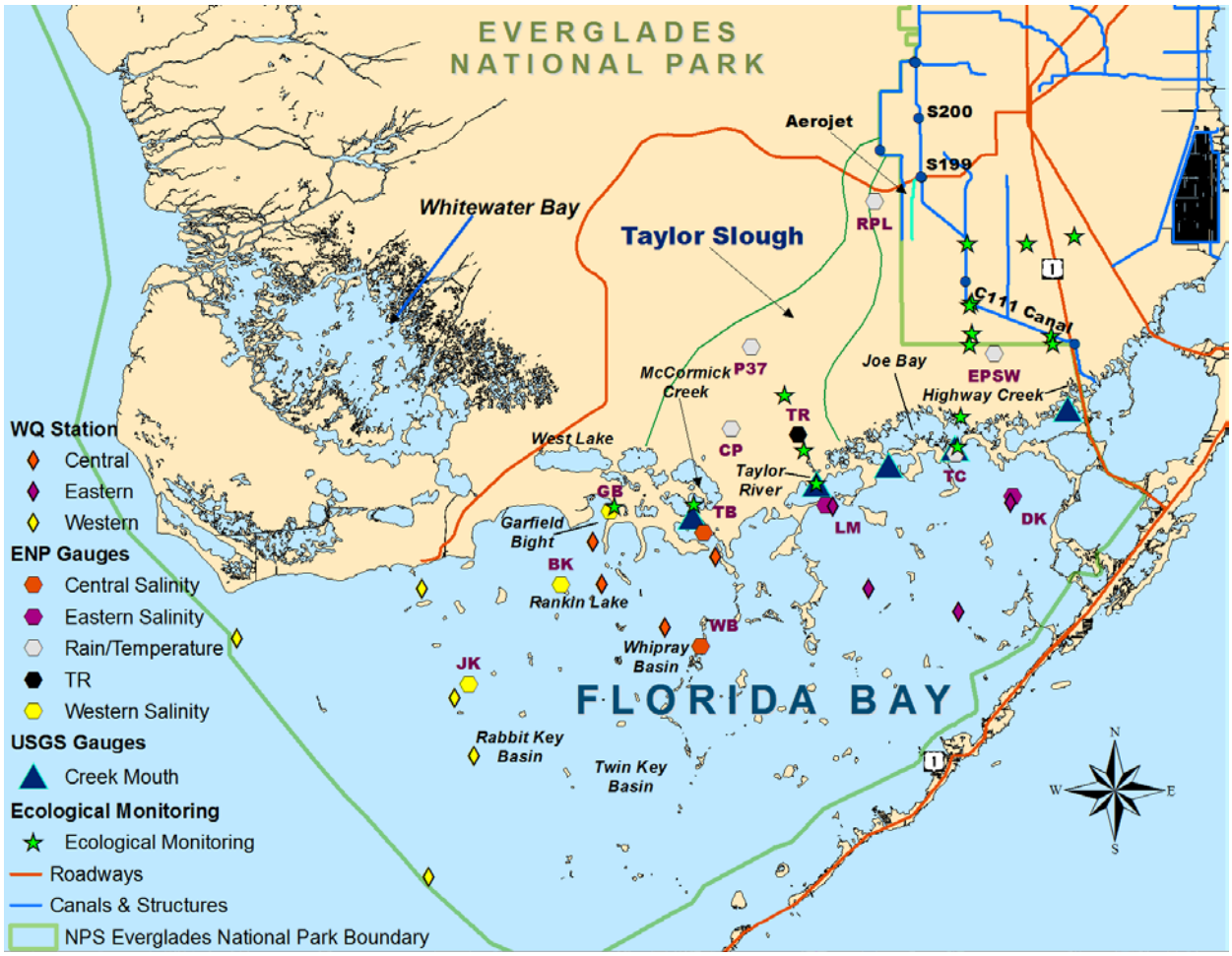


Figure 6-9. Sampling points within Florida Bay and southern Everglades, showing areas influenced by water management operations from Whitewater Bay in the west to Barnes Sound in the east. Environmental parameters are monitored by SFWMD, USGS, and ENP. Purple text identifiers identify the names for the ENP gauges (hexagons).

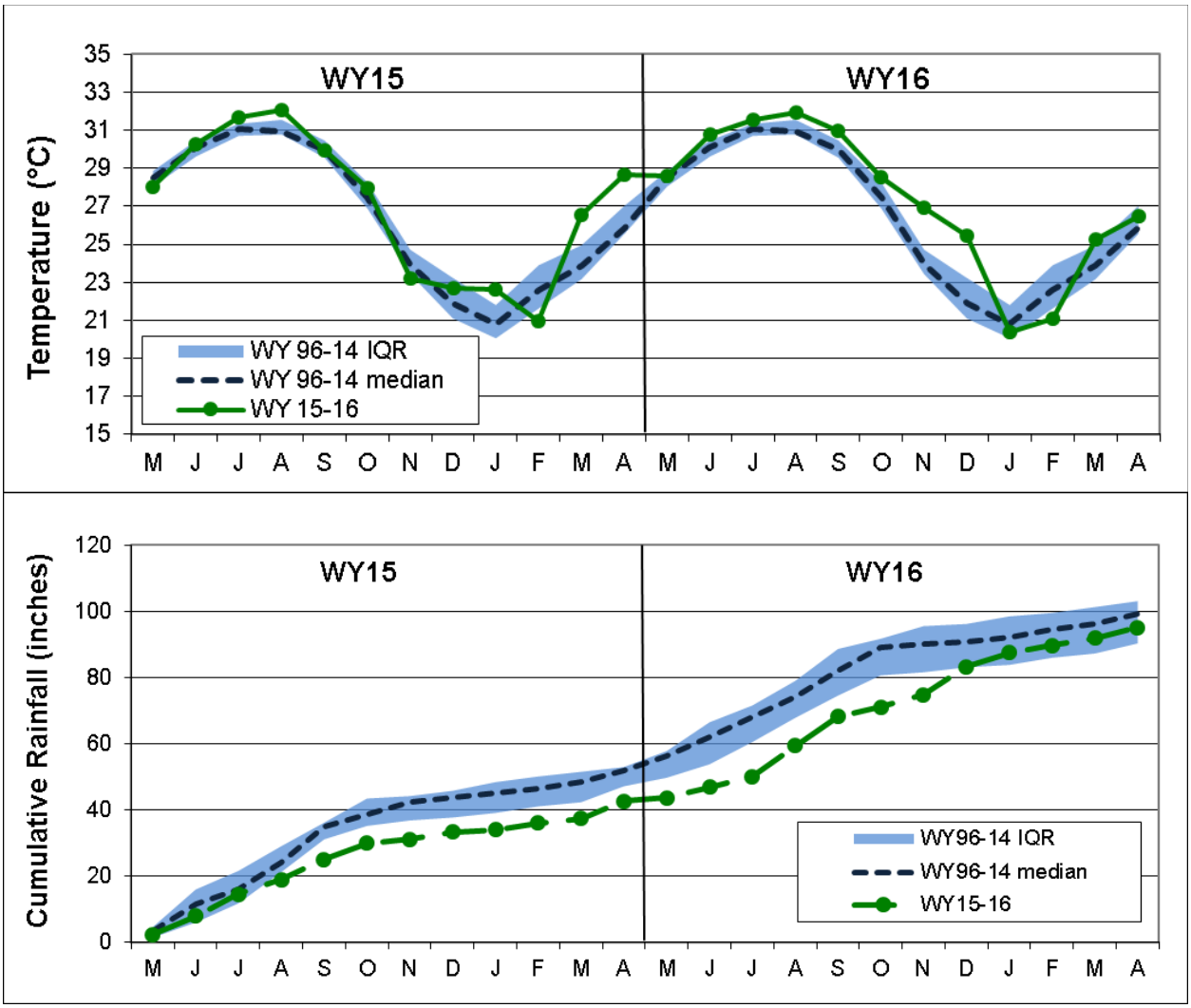


Figure 6-10. Temperature and cumulative rainfall for WY2015 and WY2016 as compared to the 25th, 50th (median) , and 75th percentiles for the time period of WY1996–WY2014. The blue area represents the range between the 25th and 75th percentiles or the interquartile range (IQR). Temperature and rainfall data from 8 and 14, respectively, ENP platforms were averaged to produce a spatial composite for the southern ENP and northern Florida Bay area.

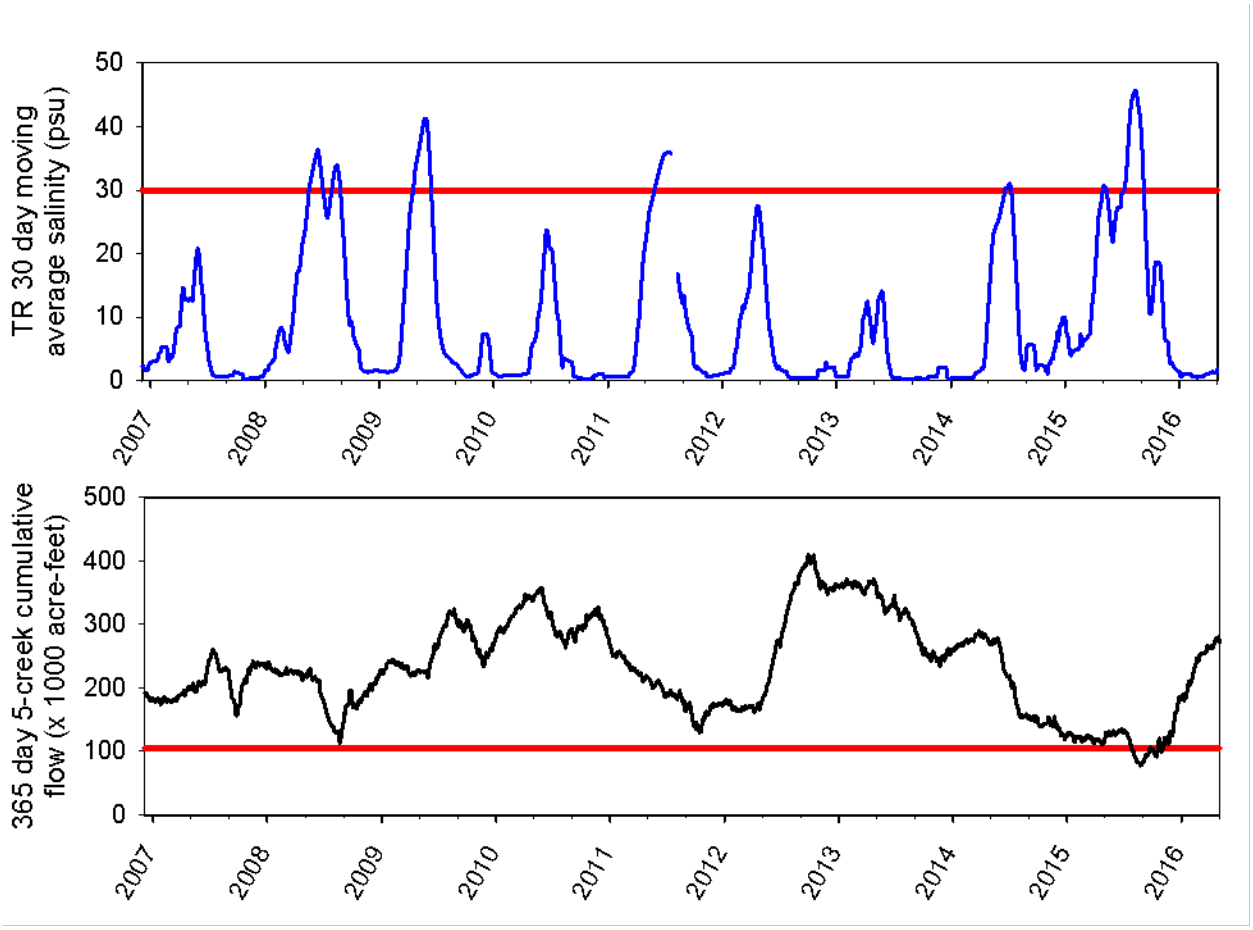


Figure 6-11. The two criteria tracked for the Florida Bay MFL Rule: (1) the 30-day moving average salinity at the ENP-Taylor River platform (top panel) and (2) the 365-day moving sum for flow from the five creeks (bottom panel). The 105,000 ac-ft flow threshold and 30 salinity threshold for the MFL criteria are denoted by red lines on the graphs. Flows below the red line and salinities above the red line constitute exceedances of the MFL. Flow data are provisional for October 2015 forward and supplied courtesy of USGS.

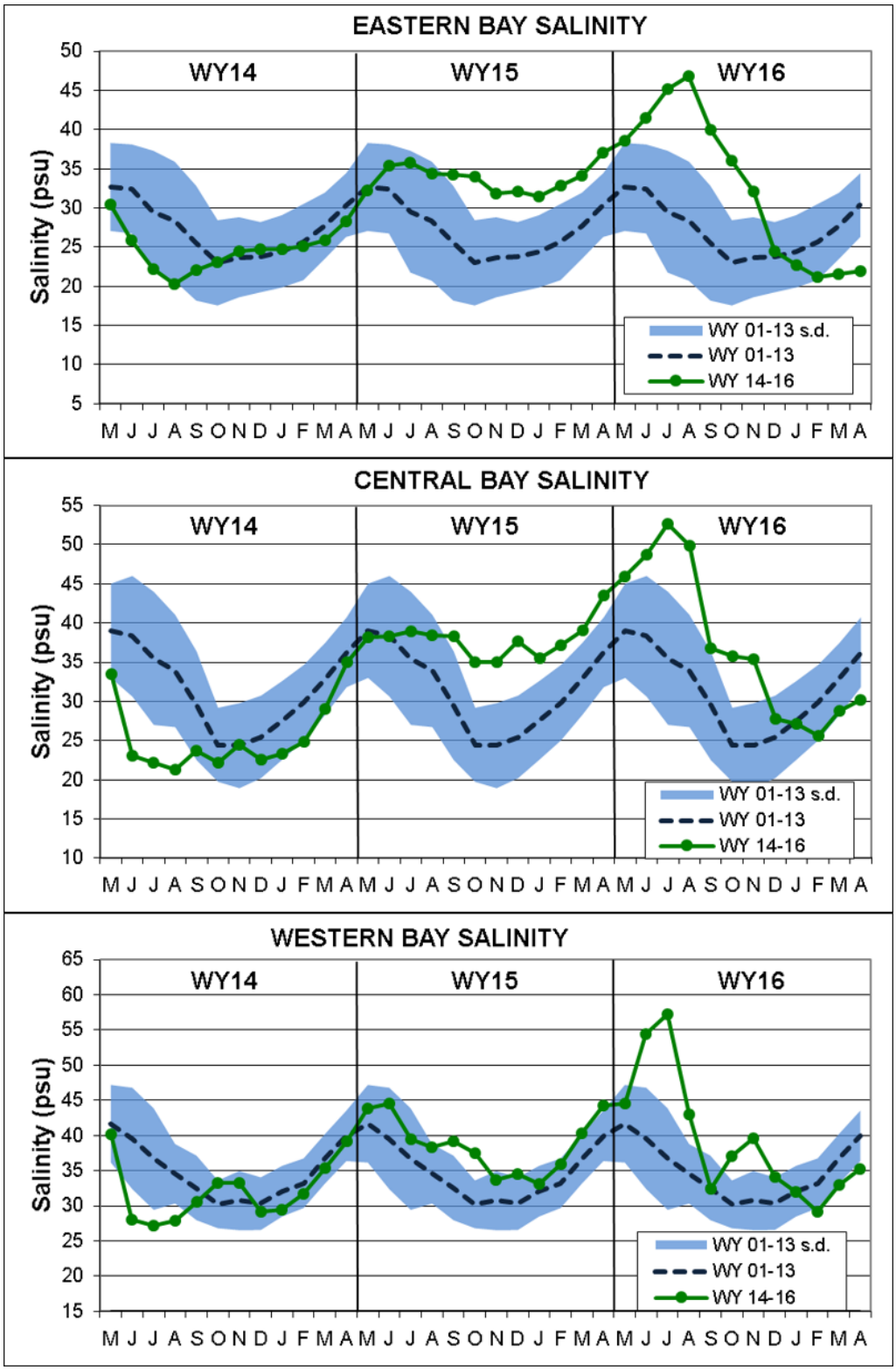


Figure 6-12. Mean monthly salinity values in eastern Florida Bay (top), central Florida Bay (middle), and western Florida Bay (bottom) in WY2014–WY2016 (WY 14-16 in the legend), compared to monthly means (WY01-13) and standard deviation (WY 01-13 s.d.) for WY2001–WY2013. Salinity values are averaged between two continuous monitoring platforms within each region: LM and DK for eastern, TB and WB for central, and GB and JK for western.

WILDLIFE ECOLOGY

Mark I. Cook

The District currently focuses its wildlife research toward gaining a better understanding of the links among hydrology, aquatic prey availability, and wading bird foraging and reproduction. These relationships have been formalized in the Trophic Hypothesis, a conceptual ecological model that provides the underlying scientific framework for District wildlife research (RECOVER 2012; http://141.232.10.32/pm/ssr_2012/ssr_2012_pdfs/2012_ssr_full_web.pdf). This research has improved both the District's capacity to effectively manage the system and ability to predict future restoration scenarios. This utility of the research stems not only from an improved knowledge of how key ecological drivers affect wading bird reproduction, but also from the recasting of these data into practical spatially explicit tools to predict foraging and nesting responses to physical and biological processes in real time and space. This section summarizes wading bird nesting effort and success in the Everglades and Florida Bay during the WY2016 breeding season.

WADING BIRD MONITORING

Large populations of colonially nesting wading birds (order Pelecaniformes: egrets, ibises, herons, spoonbills; and order Ciconiiformes: storks) were a common and defining feature of the predrainage Everglades. Long-term records of their nesting stretch back to the early part of the last century, and some clear reproductive responses to anthropogenic alterations have been established:

- A marked decline in the nesting populations of several species, particularly the tactile foraging species
- A movement of colonies from the over-drained estuarine region to the more ponded interior marshes
- A marked decrease in the frequency of exceptionally large aggregations of nesting white ibises (*Eudocimus albus*)
- Delayed nest initiations of wood storks (*Mycteria Americana*) by a few months (from November–December to February–March), resulting in reduced nestling survival

These responses appear to be consistent with mechanisms that involve foraging and specifically the role that hydrology plays on the production and vulnerability to predation of aquatic prey animals (see Frederick et al. 2009 and references within).

Wading birds are excellent indicators of wetland ecosystem health and have a central role in 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 is evaluated using the two independent sets of measures. RECOVER established three performance measures (http://141.232.10.32/pm/recover/perf_ge.aspx), which include the three-year running average number of nesting pairs of the five key wading bird species, timing of wood stork nesting, and proportion of the wading bird population that nests in the marsh-coastal ecotone (Ogden 1994). In addition, the South Florida Ecosystem Restoration Task Force developed two additional measures for annual stoplight reports: the ratio of visual to tactile wading bird species breeding in the Everglades, and the frequency of exceptionally large white ibis nesting events (Frederick et al. 2009):

- Increase and maintain the total number of pairs of nesting birds in mainland colonies to a minimum of 4,000 pairs of great egrets, 10,000–20,000 combined pairs of snowy egrets and tricolored herons, 10,000–25,000 pairs of white ibises, and 1,500–2,500 pairs of wood storks.
- 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 ibises, egrets, and herons to initiate nesting in February–March in most years.
- Return of major wood stork, great egret, white ibis, and *Egretta* 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.
- Reestablish 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 Greater Everglades.
- For wood storks, restore productivity for all colonies combined to greater than 1.5 chicks per nest.
- Return to an interval between exceptional white ibis nesting events, defined as greater than 70th percentile of annual nest numbers for the POR.

Summary of Nesting Year 2016

The information reported in this section represents a compilation of data collected by the District, University of Florida, ENP, and Audubon of Florida. The population counts include all wading bird species (except Cattle Egret, *Bubulcus ibis*) nesting throughout the Greater Everglades region (the WCAs, ENP and Florida Bay), and the period covered is the nesting season from January–July, 2016. Note that counts do not include estimates of precision due to the challenges of estimating detection bias at the landscape scale, although progress is being made in this respect (e.g., Fredrick et al 2003, Williams et al. 2011). For details on independent sampling methods see Cook (2015).

Nesting Effort

The Greater Everglades is the primary nesting region for wading birds in South Florida, accounting for between 75 and 95 percent of the annual nest count during the past ten years. Wading birds initiated an estimated 18,637 nests in the WCAs and ENP during 2016. This nesting effort is 45 percent lower than the decadal average (33,781.4 nests), 35 percent lower than the five-year average (28,750.6 nests), and

74.5 percent lower than the banner year of 2009 when a record high of 73,096 nests was recorded in the Everglades.

Most species exhibited relatively limited nesting effort in 2016. White ibis (10,545 nests), the most numerous wading bird species nesting in the Everglades, reduced nesting effort by 47 and 42 percent relative to the decadal and five-year averages (**Figure 6-13**). Numbers of great egret and wood stork nests were down 23 and 59 percent, respectively, from the 10-year average, and down 20 and 61 percent from the 5-year average. The smaller *Egretta* herons, which have exhibited a consistent decline in nest numbers in recent years, had another relatively poor nesting season: nesting effort was reduced 74 percent for the snowy egret relative to the 10-year average, while tricolored heron nesting was down 74 percent. These declines in *Egretta* herons do not appear to be due to changes in the distribution of nesting and instead appear to reflect a general reduction in overall nesting effort in the Everglades.

In Florida Bay, roseate spoonbills (*Platalea ajaja*) produced 367 nests, which is very similar to last year's effort (365 nests) and to the decadal average (321.4 nests). However, this effort is poor from a historical perspective being 23 percent lower than the 30-year mean (475 nests) and a considerable drop from the mid-twentieth century when over 1,000 nests per year were common.

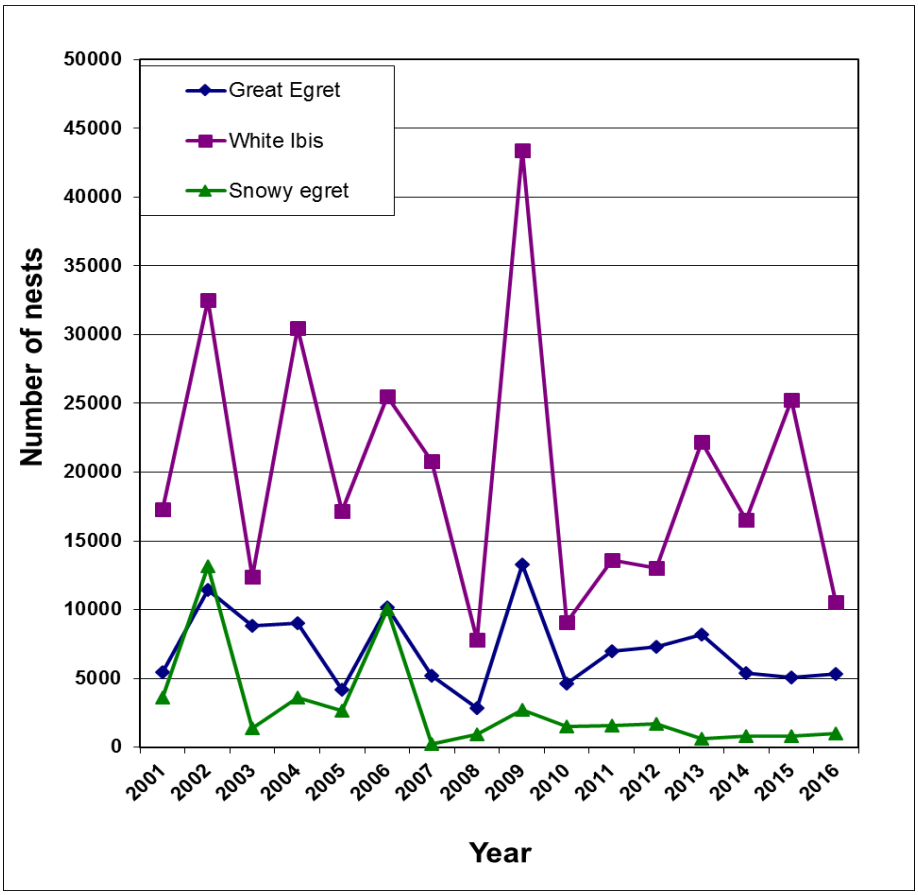
Spatial Distribution

The estuarine region of ENP historically supported about 90 percent of all nesting wading birds in the Greater Everglades, probably because it was the most productive region of the Everglades ecosystem. During the past fifty years, that productivity has declined due to reduced freshwater flows, and the location of nesting has shifted towards inland colonies in the WCAs. An important goal of restoration is to restore the hydrologic conditions that will reestablish prey availability across the southern Everglades landscape that, in turn, will support the return of large successful wading bird colonies to the traditional estuarine rookeries. In 2016, ENP supported 25.6 percent of nests, while WCA-3A and WCA-1 supported 20.5 and 53.9 percent, respectively. This represents a moderate increase in the proportion of birds nesting in ENP relative to the decadal average (22.6 percent) but remains far short of the 50 percent restoration target.

The location of roseate spoonbill nesting colonies within the Florida Bay area has also shifted in recent years. Whereas most nesting has always occurred on small keys within the bay itself, many birds have moved over the past five years to mainland colonies adjacent to the coast (i.e., Madeira Hammock and Paurotis Pond colonies; 41 percent of all nests in 2016). The reason for this move is unclear but might reflect the recent reduction in mammalian predators (raccoons) on the mainland or a reduction in the suitability of habitat for nesting on the keys. Some individuals have deserted Florida Bay entirely. For three of the past four years about 200 pairs have nested at colonies in the central freshwater Everglades such as northern WCA-3A. This number declined a little during 2016, with 72 nests found at inland colonies.

Timing of Nesting

Wood storks have a relatively long reproductive period (about four months), and it is critical they start nesting early in the dry season to ensure nestlings have time to fledge and gain independence prior to the onset of the rainy season in May or June. Wood stork prey (fish) are easy to find and feed upon when concentrated in shallow water during the dry season, but are not available in the wet season (summer) when they redisperse back into deeper marsh waters. Without the dry season supply of highly concentrated prey, parent birds are unable to support their offspring. Wood stork nesting historically started in November or December, but in recent decades nesting initiation has shifted to January to March. The delay is thought to be because of a reduction in the amount and quality of short-hydroperiod wetlands, which provide suitable foraging habitat early in the nesting season. In 2016, wood stork nesting was particularly late, with the first eggs noted on March 22. This start time is later than last year (late February) and 2014 (late January), and falls far short of the CERP target date (December).



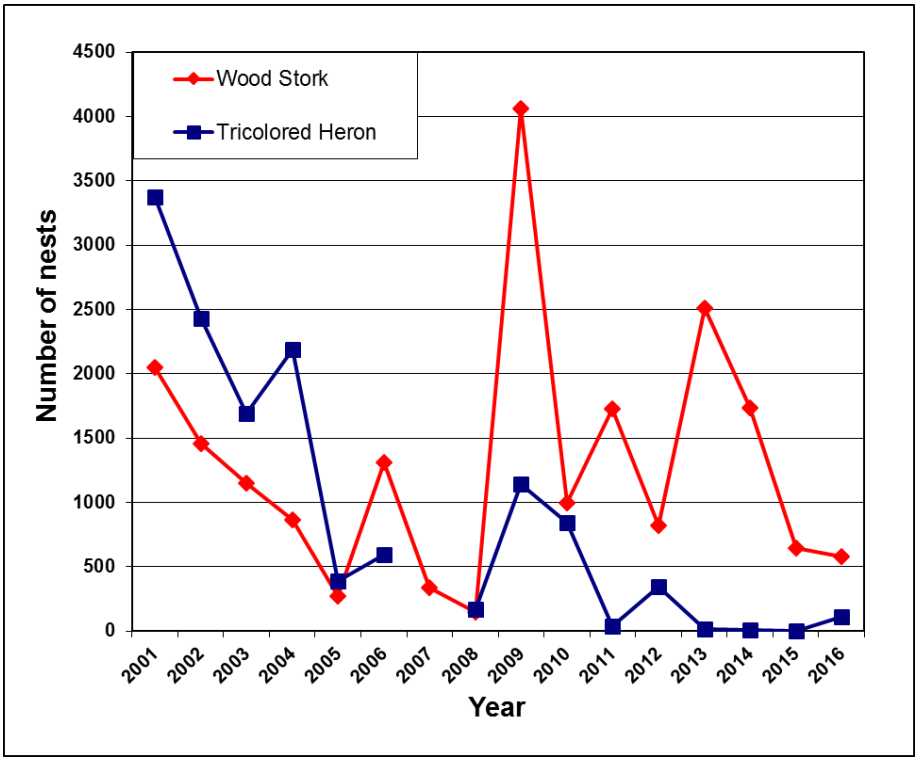


Figure 6-13. Historical wading bird nesting numbers in the Everglades for individual species since 2001.

Roseate spoonbills in Florida Bay appear to be exhibiting a recent shift towards later nesting. For at least seventy years (1936 to 2009) spoonbill nest initiations consistently fell between October 1 and December 31. However, from 2010, nesting began to start increasingly later in the season: from 2010 to 2014 nesting started between January 1 and 10, in 2015 it was January 24, and this year it was February 5, the latest start date ever recorded. Moreover, the timing of laying appears to be getting considerably more asynchronous both within and among colonies. Whereas nest initiations within the bay would historically span a few weeks, lay dates in the past two years have extended from January through April. These changes in the phenology and synchrony of nesting might suggest that the timing of optimal foraging conditions for roseate spoonbills is changing both temporally and spatially within Florida Bay.

Nesting Success

The University of Florida measured nest success (p; the probability of fledging at least one young, the Mayfield method) at five colonies, including three in ENP (Tamiami West, Paurotis Pond, and Cuthbert) and two in WCA-3 (6th Bridge and Alley North). Individual nests of great egret (sample size [n] = 143) and *Egretta* herons (n = 52 at all five colonies), white ibis (n = 44 at Alley North, 6th Bridge and Paurotis) and roseate spoonbill (n = 3 at 6th bridge) were monitored. Nest success was similar among species (from 47 to 59 percent). However, nest success between various colonies ranged widely for each species, for example; great egret nest success was 96 percent (standard deviation [SD] = 0.03) at Cuthbert but only 29 percent (SD = 0.08) at Tamiami West. Reports by biologists at ENP suggest that all wood stork nests failed in 2016.

Role of Hydrology on Nesting Patterns

Wading bird reproductive patterns in South Florida are driven principally by hydrology through its influence on aquatic prey production and vulnerability to predation (Frederick et al. 2009). The 2016 breeding season was preceded by a very late and relatively dry 2015 wet season, which meant that water levels remained below ground for extended periods across large areas of the Everglades (see *Hydrology* section). Such conditions generally limit the production of small fish (Trexler et al. 2005), and reduced fish biomass might account for the observed late nesting and decreased nesting effort by the piscivorous wood stork and great egret in 2016. By contrast, the 2015–2016 winter–spring dry season, the period during which the birds breed, was much wetter than average because of the La Niña conditions. Water levels were generally too deep for wading bird foraging throughout the Everglades until late in the dry season. When water depths and recession rates finally became appropriate for foraging, a rain event in May caused a water level reversal and the dispersal of concentrated prey. This might account for the large proportion of nest abandonments by white ibis in WCA-1 that occurred shortly after the reversal events.

Restoration Targets

White ibises and great egrets met the numeric CERP targets (three-year running averages) but wood storks and snowy egrets/tricolored herons did not (**Table 6-3**). The two other CERP targets (an increase in the proportion of nesting wading birds in the coastal Everglades and a shift in the timing of wood stork nesting to earlier in the breeding season) are discussed above and did not meet the necessary requirements. With respect to the annual stoplight targets, the 2016 white ibis nesting effort did attain the number of nests (> 16,977 nests) required for an exceptional nesting event, and the interval between such events averaged over the past five years is 1.4 years, which is slightly less than the restoration target of 1.6 years recorded from the 1930s. The ratio of tactile to visual foragers was only 1.7, over an order of magnitude lower than the restoration target of 32.

Table 6-3. Numbers of wading bird nests in the WCAs and ENP 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, and which were summarized by Ogden (1994).

Species	2005-2007	2006-2008	2007-2009	2008-2010	2009-2011	2010-2012	2011-2013	2012-2014	2013-2015	2014-2016	Target
Great Egret	6,600	5,869	6,956	6,774	8,303	6,296	7,490	6,961	6,223	5,250	4,000
Snowy Egret/ Tricolored Heron	4,400	3,778	1,723	2,442	2,622	2,008	1,431	1,165	744	902	10,000–20,000
White Ibis	21,133	17,541	23,953	20,081	22,020	11,889	16,282	17,261	21,406	17,512	10,000–25,000
Wood Stork	633	552	1,468	1,736	2,263	1,182	1,686	1,689	1631	988	1,500–2,500

PLANT ECOLOGY

Plant studies form an important basis for evaluating restoration success. This section evaluates the hydrologic interactions between soil, groundwater, marsh, and trees on LILA tree islands, presents the results of a study of a proposed tree planting technique for tree island restoration, and reviews the status of the vegetation mapping of ENP.

TREES: A GEOMORPHIC AGENT GOVERNING LANDSCAPE EVOLUTION[1]

Pamela L. Sullivan[2] and Eric Cline[3]

Trees play a critical role in shaping the earth surface through both chemical and physical weathering. Specifically, they govern subsurface moisture dynamics, and thus water-mineral interactions. This they do simply through root-water uptake to meet transpiration demands but also by altering soil porosity and hydraulic conductivity as their extant root systems push soils and release organic matter into the subsurface (Angers and Caron 1998, Pawlik 2013). Funding for this project was being provided by SFWMD.

In wetland systems it is often small changes in topography (10–100 centimeters [cm]) that support landscape heterogeneity. The landscape patterns of some wetland areas have been attributed to a transpiration-driven ion accumulation mechanism (Eppinga et al. 2008, McCarthy et al. 1993, 1998). Elevated trees are postulated to have elevated transpiration rates compared to lower elevation marsh vegetation. This increased use of water then leads to the movement of water and associated ions. Ions are

[1] Excerpts from Sullivan et al. 2016.

[2] University of Kansas, Lawrence, KS.

[3] Special thanks to Rene' Price, Michael Ross, Susana Stoffella, Jay Sah, and Leonard Scinto from Florida International University and to Thomas Dreschel and Fred Sklar of SFWMD for guidance and advice, and laboratory, field, and editorial support.

then concentrated either through ion exclusion by vegetation or ion cycling through the mineralization of leaf litter and, thus, nutrient gradients develop between high and low elevation adjacent areas.

Over the last decade, evidence from the Everglades suggests transpiration-driven ion accumulation may be one mechanism supporting the presence of tree islands (Wetzel et al. 2005, Ross et al. 2006, Sullivan et al. 2012). This mechanism has been suggested, in part, to explain the high phosphorus (P) concentrations in the groundwater and soil of many Everglades' tree islands (commonly one to three orders of magnitude greater than the adjacent marsh; Gann et al. 2005, Ross et al. 2006). This evidence indicates that trees may create a hydrogeochemical condition under healthy tree islands that maintains the topographic difference between tree islands and the adjacent marsh.

To mechanistically determine biological and physical feedbacks critical to maintaining tree islands and the overall health of the Everglades, a large physical ecosystem model of the Everglades, known as the Loxahatchee Impoundment Landscape Assessment (LILA; Aich et al. 2011), containing eight constructed tree islands was used to conduct the study described herein. The goal of the research presented here was to utilize groundwater geochemistry to examine the evolution of tree island hydrogeochemical function during early forest development; specifically testing two hypotheses:

- Hypothesis 1: Transpiration governs the hydrodynamics of tree islands during early stand development and results in the advective movement of groundwater and accumulation of ions towards the center of the islands.
- Hypothesis 2: The accumulation of biomass during stand development governs ion accumulation and nutrient uptake in a manner consistent with the formation of resource gradients/contrasts between tree islands and the surrounding marsh system.

To test these hypotheses, we collected surface water and groundwater and estimated aboveground biomass over five years on eight tree islands. Transpiration was not directly measured but instead we used aboveground biomass as a proxy for water loss. A third hypothesis was tested as part of this project involving the precipitation of calcite or calcium carbonate ($CaCO_3$), however that analysis is not presented here.

Materials and Methods

Study Area

LILA is located within the Arthur R. Loxahatchee National Wildlife Refuge (LWNR) in Boynton Beach, Florida. LILA consists of four, 8-hectare (ha) macrocosms, each of which contains two tree islands, one constructed predominately of peat and the other constructed of a limestone rubble center overlain by a thin layer of peat. Each island was planted across a topographic gradient with 717 trees of eight species common to the Everglades (Stoffella et al. 2010). Four islands were planted with saplings (one-gallon pot size) in March 2006 (Planting 1); the remaining islands were planted in March 2007 (Planting 2).

Nine wells were installed on each island, three in the center (CENTER), and six on the edges (EDGE), with an average depth of 1.34 m. One additional deep well was installed in the center of each peat tree island with an average depth of 2.00 m (DEEP). All wells had a 50–60 cm screened interval at the bottom. Wells at the same depth are used to interpret horizontal water exchange, while wells at different depths are used to elucidate vertical water exchange.

Distinct wet and dry seasons drive surface water levels in the Everglades, with the majority of rainfall occurring between June and November. On average, 130–160 cm of rainfall occurs annually near the LWNR (Ali et al. 2000). Surface water levels at LILA were managed according to an operational hydrograph that mirrored the seasonal variations in Everglades surface water.

Monitoring and Sampling Design

Groundwater and Surface Water

From October 2007 through April 2012, 28 groundwater wells and 8 surface water sites were sampled biannually in October (wet season) and April–May (dry season; **Figure 6-14**). The measurements serve as a platform to elucidate plant-groundwater-surface water interaction using chloride (Cl^-) and a stable isotope of oxygen ($\delta\ ^{18}O$), and assess the potential development of a nutrient gradient between the marsh and tree island.

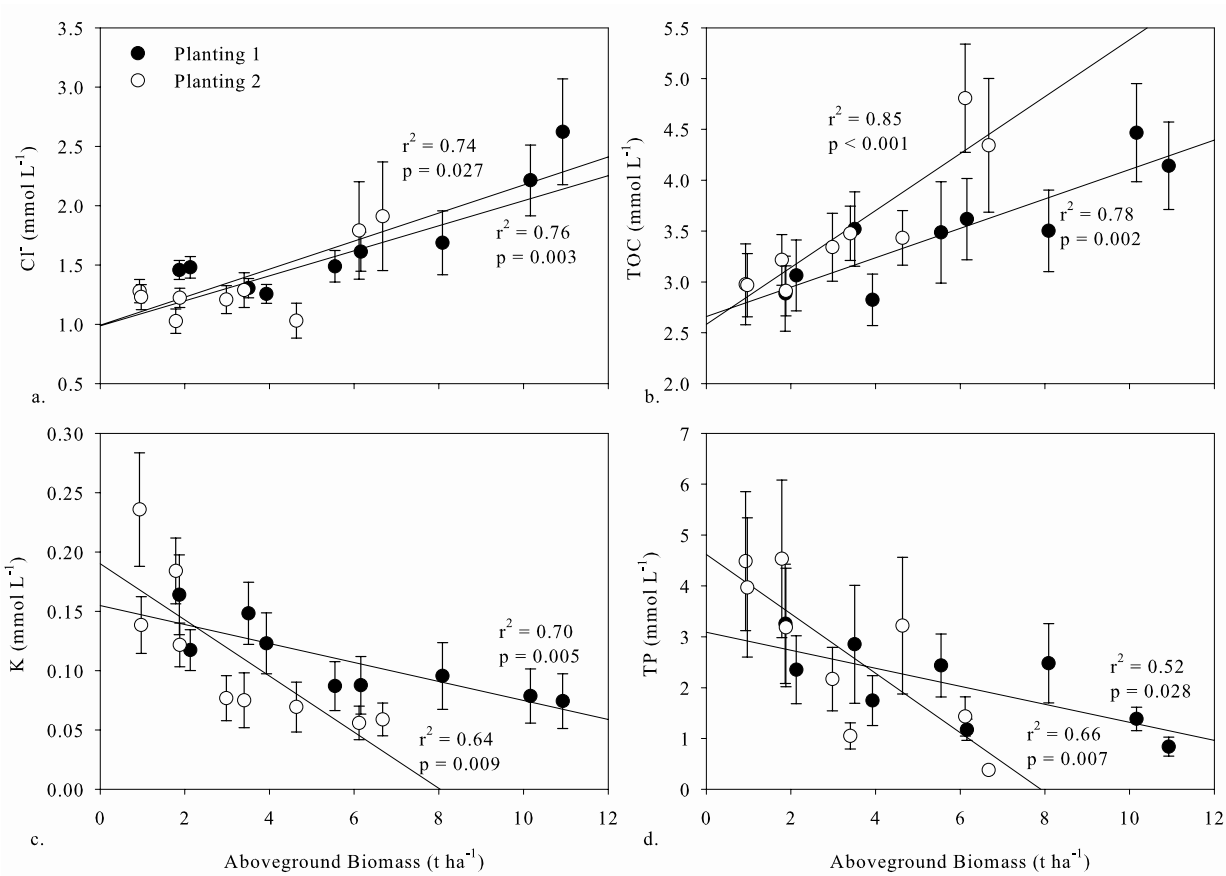


Figure 6-14. The average amount of aboveground biomass compared to the average groundwater water concentrations of (a) chloride, (b) TOC, (c) potassium (K^+), and (d) TP in the Planting-1 (P1, black) and Planting-2 (P2, white) islands.

Isotopes, Nutrients, and Other Chemical Constituents

To determine the effect of plant-groundwater-surface water interaction on the hydrogeochemical condition of tree island groundwater, water isotopes from incoming precipitation were also collected from January 2009 through July 2010. Groundwater and surface water samples were analyzed for potassium (K), soluble reactive phosphorus (SRP), TP, TOC, total alkalinity, anions, and cations. All water samples collected were analyzed for hydrogen and oxygen stable isotopes. Aboveground tree biomass was estimated biannually in March and September from 2007 through 2012.

Results

Water Tracers Indicate Transpiration Governs Tree Island Hydrodynamics

Given that $\delta^{18}O$ and Cl^- are conservative tracers of water, meaning organic and mineral materials have negligible influence on the concentrations of these substances as they move through the environment (Kendall and Caldwell 1998, Schlesinger 1997), they can be used to trace the interaction between precipitation, surface water, and groundwater. At LILA, the $\delta^{18}O$ and Cl^- of the surface water and the groundwater at the edge and center portions of the islands significantly differed with elevated Cl^- concentrations and depleted isotopic values detected in the center. In general this pattern reversed in the deeper groundwater (**Table 6-4**). These measurements further support the conceptual model developed in Sullivan et al. (2012), which demonstrates that in LILA tree islands the flow paths are likely driven by a sustained water table depression.

Table 6-4. The mean and standard error (SE) of Cl- and $\delta^{18}O$ values in tree island groundwater in the CENTER, EDGE, and DEEP and Surface Water. A one-way analysis of variance (ANOVA) and post-hoc Tukey test (α = 0.05) was used to determine significant difference between locations, denoted by the letters (a, b, c). (Note: mmol L-1 should be μmol L^{-1}, which is micromoles per liter.)

Constituents	Units	Statistic	Surface Water		EDGE		DEEP		CENTER		F Values	P-Values
Cl^-	mmol L^{-1}	Mean	0.90	a	1.17	b	1.41	b,c	1.85	c	23.95	<0.001
		SE	*0.04*		*0.04*		*0.17*		*0.11*			
$\delta^{18}O$	(‰)	Mean	2.19	a	0.60	b	0.16	b,c	-0.25	c	80.257	<0.001
		SE	*0.19*		*0.08*		*0.14*		*0.09*			

Establishment of Nutrient Gradients

Over this five year study (2007–2012), aboveground biomass on the islands increased one order of magnitude for the Planting-1 and Planting-2 islands. Concurrent with this increase in biomass was a significant increase in groundwater Cl^- and TOC concentrations while K and TP significantly declined (**Figure 6-14**). We attribute the significant increase in groundwater Cl^- to the transpiration driven ion accumulation phenomena, which we postulate was enhanced over time as a result of the substantial increase in biomass and thus water loss and root ion exclusion by transpiration.

Discussion

Our results are consistent with those suggested in Wetzel et al. (2005), Ross et al. (2006), and Sullivan et al. (2012) where the unifying hypothesis is that transpiration by trees drives inward flow of regional groundwater into tree islands. Specifically we demonstrate that the accumulation of aboveground biomass during early stand development results in the accumulation of ions in groundwater. As biomass accretes on the islands, the build-up of ions alters the groundwater geochemical conditions to favor mineral formation (e.g., $CaCO_3$), the data supporting this analysis (Hypothesis 3) are not presented here. Transpiration driven groundwater uptake and the aboveground biomass accretion at LILA further resulted in the uptake and subsequent storage of nutrients on the islands.

Stand Development Governs Island–Marsh Nutrient Resource Contrast

The trends in groundwater nutrient concentrations at LILA are supported by the accretion of live biomass and the establishment of a functioning forest microbial community. The concurrent increase in

groundwater TOC with biomass can be explained by an increase in both root exudates and microbial activity that is associated with tree growth (Jones et al. 2004). An increase in root depth along with associated exudates and microbes may also explain the doubling of TOC in the groundwater on both Planting-1 and Planting-2 islands following the spring 2011 drought, a trend which was pronounced in the DEEP groundwater.

Given that both K^+ and P are essential for plant growth, it is no surprise that they declined in solution over time as biomass accumulated.

The decline in TP over time likely resulted from the microbial transformation of TP to SRP and the uptake of SRP by overlying trees, and may also be a result of adsorption of P and/or co-precipitation with precipitation of $CaCO_3$, a common phenomenon observed with the formation of $CaCO_3$ in the calcium II (Ca^{2+}) rich water (Dodds 2003). While this phenomenon likely occurs, soil nutrient data from the LILA islands indicated the development of a new organic horizon (0–3 cm deep) that contained significantly higher (1.4–1.8 times) concentrations of TP, total nitrogen (TN), and total carbon compared to underlying soils (3–10 cm deep; Rodriguez et al. 2014). These elevated concentrations were inferred to indicate that plant activity, specifically groundwater uptake by overlying trees, drove nutrient accumulation. Here, we provide direct evidence of transpiration driven groundwater uptake and a concomitant decrease in groundwater nutrient concentrations that support a mechanism for nutrient cycling and the eventual development of a nutrient contrast between tree island and marsh groundwater.

While Sullivan et al. (2014a, b) demonstrated that present day hydrologic fluxes on the island could not account for the mass of extant P found in many tree island soils, data from LILA provide a mechanism for sequestering P that reaches the tree islands by other suspected sources e.g., deposition of dust or bird guano (Irick et al. 2015). If we expand this reasoning, then these mechanisms caused by transpiration-driven ion accumulation on tree islands in the Everglades can act as a P filter helping to maintain this wetland in an oligotrophic state.

Conclusions and Relevance to Water Management

Trees help to govern the geomorphic evolution of subtropical wetlands as their transpiration demands result in the accumulation of ions and nutrients that support greater soil accretion on tree islands as compared with the surrounding marsh. We provide evidence from constructed tree islands that the increase in aboveground biomass during early stand development triggers transpiration-driven ion accumulation in soils. The observed depletion of nutrients (K and P) in the groundwater suggests a connection between the nutrient uptake and storage by the overlying trees and the initial development of a nutrient contrast between the tree islands and marsh. The evidence that the processes of transpiration-driven ion accumulation and nutrient partitioning exist within both the constructed tree islands at LILA and in the Everglades suggests a persuasive motivation for the construction and restoration of tree island forests. Tree islands within the Everglades can act as a P filter helping to maintain this wetland in an oligotrophic state.

DEPLOYING TREE SAPLINGS IN PEAT BAGS AS A POTENTIAL RESTORATION TECHNIQUE: A FIELD STUDY AT LILA

Thomas Dreschel, Eric Cline, and Steven Hill

Introduction

We expanded upon the conceptual approach for creating artificial floating peat islands (pop-ups) to eventually establish on ghost tree islands. This technique was previously described in Dreschel and Cline (2015). The approach taken is low cost, simple, provides about 1.5 ft (0.45 m) of additional elevation, and provides a substrate for the initial root growth of the planted tree saplings.

Everglades tree islands have been heavily impacted by man since the late 1800s through construction, drainage, flooding and fire. Many of the islands, particularly in WCA-2, have lost a significant amount of elevation, leading to the loss of woody species (Aich et al. 2014). These are now termed "ghost" tree islands as their footprint still exists as a sawgrass ridge within the landscape. We believe that recolonization of these ghost islands can occur through plantings and the subsequent seed dispersal by the planted trees. The approach described overcomes the elevation loss by providing the sapling additional substrate in the form of a commercial peat bag. The peat bag has an ability to float for several months during initial deployment, thus allowing the sapling to survive and establish without the potential of inundation during high water.

Tree Island Elevation

Historically, tree islands were typically 2–4 ft (0.6–1.2 m) above slough bottom (McVoy et al. 2011). Ridges were typically 1–2 ft (0.3–0.6 m) above slough bottom (McVoy et al. 2011). Consequently, tree islands should be about 1–2 ft (0.3–0.6 m) above the ridge. Ghost tree islands are typically at about ridge level or about 1–2 ft (0.3–0.6 m) above slough bottom. Thus, we anticipated that an additional 1–3 ft (0.6–0.9 m) would be required to raise the island to an elevation to allow the saplings to establish.

Tree Species Response to Hydrology

Everglades tree island tree species have varying tolerances to flooding for growth and survival (Armentano et al. 2002, Stoffella et al. 2010). Stoffella et al. (2010) found that relative elevation (to the water surface) had varying and profound effects on planted native tree species survival and growth in the constructed tree islands at LILA, due to the specific tolerance of each species to flooding. Higher relative elevation generally led to greater survival.

Methods and Materials

The LILA facility (Aich et al. 2011) was used to conduct the tree island restoration studies. LILA is a large physical model that consists of four 8-ha macrocosms containing the major landscape features of the Everglades. A large ridge along the southern edge of each macrocosm has a boardwalk near the western corner. This ridge was used as analogue ghost tree islands with ready access for study deployment and monitoring.

As in the pilot study (Dreschel and Cline 2015), commercially available peat bags (0.08 cubic meters [m^3] or 3 cubic ft [ft^3]) were used to support one or two tree saplings until they could become established. To do this, the saplings were planted into the bags, which were then floated onto the site, punctured and tethered in place using polyvinyl chloride (PVC) poles. During the first wet season, the bags float, protecting the sapling from complete inundation. During the dry season, when water levels drop below ground, the roots penetrate the substrate beneath the peat bag and allow the permanent establishment of the tree at the site, on top of the added peat, at about 0.45 m (1.5 ft) above the current substrate.

A field study was conducted for two years beginning in October 2013, within two macrocosms at LILA. The effects of added nutrients, the necessity of providing floats and the further effect of crowding on tree establishment and growth were tested. Three species of tree saplings: pond apple (*Annona glabra*), strangler fig (*Ficus aurea*), and red maple (*Acer rubrum*), the treatments/controls, were randomly assigned to one of the 45 sites adjacent to the southwestern boardwalks in Macrocosm 2 and Macrocosm 4, spanning a large ridge. These three species of trees are native to South Florida and found on Everglades tree islands and have varying tolerances to flooding.

Two bag configurations for controls and treatments were randomly placed along the two LILA boardwalks for the study. The peat bags for controls and treatments of the fertilizer study were set on their long edges to provide a taller profile (height; **Figure 6-15**, left) and included the addition of one or two Vigoro® Tree and Shrub fertilizer spikes (Swiss Farms Products Inc., Las Vegas, Nevada, USA) per bag for the treatments (**Figure 6-15**, left). For a description of the nutrients supplied by the spikes within the peat bags, refer to **Table 6-5**. The fertilizer spikes were inserted into punctures made in the top of the bag as far from the potted sapling as was reasonable.

Figure 6-15 Left: High profile pop-up with *Annona* used for the fertilizer study; Middle: double planting of *Acer*, low profile; Right: *Ficus* pop-up with a float constructed from "swimming pool noodles", PVC pipe and fittings, low profile.

Table 6-5. Soil (peat) additions of nitrogen (N), P, and K provided by the Vigoro® Tree and Shrub fertilizer spikes as a function of one (treatment H) or two (treatment F) spikes per treatment bag. A peat bulk density (from specific gravity) of 0.15 grams per cubic centimeter (g/cm^3) (Blondeau 2008) was used to calculate the mass of the nutrient additions for peat bags of 85,000 cubic centimeters (cm^3) and spikes of 127 grams (g). Each bag contained 85,000 cm^3 of peat x 0.15 g/ cm^3 = 12.7 kg.

	Percent in Spike[a]	Mass/Spike (g)	One Spike/Bag Soil Concentration (mg/kg)	Two Spikes/Bag Soil Concentration (mg/kg)
Nitrogen	12.0	15.2	1,197	2,394
Phosphorus	2.2	2.8	220	440
Potassium	5.8	7.4	583	1,166

a. Washington Department of Agriculture 2012.

The pop-ups for the controls and treatments for the low profile study were set on their broad sides to provide greater planting area and single versus two saplings per bag (**Figure 6-15**, middle), plus the use of flotation devices, built from swimming pool "noodles" and PVC pipe (**Figure 6-15**, right). The studies involved 105 trees, 35 each of *Annona*, *Ficus*, and *Acer*.

We hypothesized that fertilizer would help to compensate for flooding and predicted that the ranking of treatment by survival and growth would be: high fertilizer (2-spike; bags with 2 spikes) > low fertilizer (1-spike; bags with 1 spike) > flotation (Float) > high profile control (Control-high) > low profile control (Control-low) > bags with double trees (Double). Measurements of basal diameter (millimeter [mm]) and total height (cm) were taken at deployment, after three months and after one and two years.

LILA Hydrology

The water depths within LILA were controlled according to a hydrograph that mimics the typical stages and hydroperiods of an area within northern WCA-3A (**Figure 6-16**). Stage was monitored by a series of stage gauges at LILA and the data were available on-line from the District's corporate environmental database, DBHYDRO (SFWMD 2013-2015, reported in ft National Geodetic Vertical Datum of 1929 (NGVD29). A sequence of photos demonstrating the changes in water depths experienced by the saplings is presented in **Figure 6-17**.

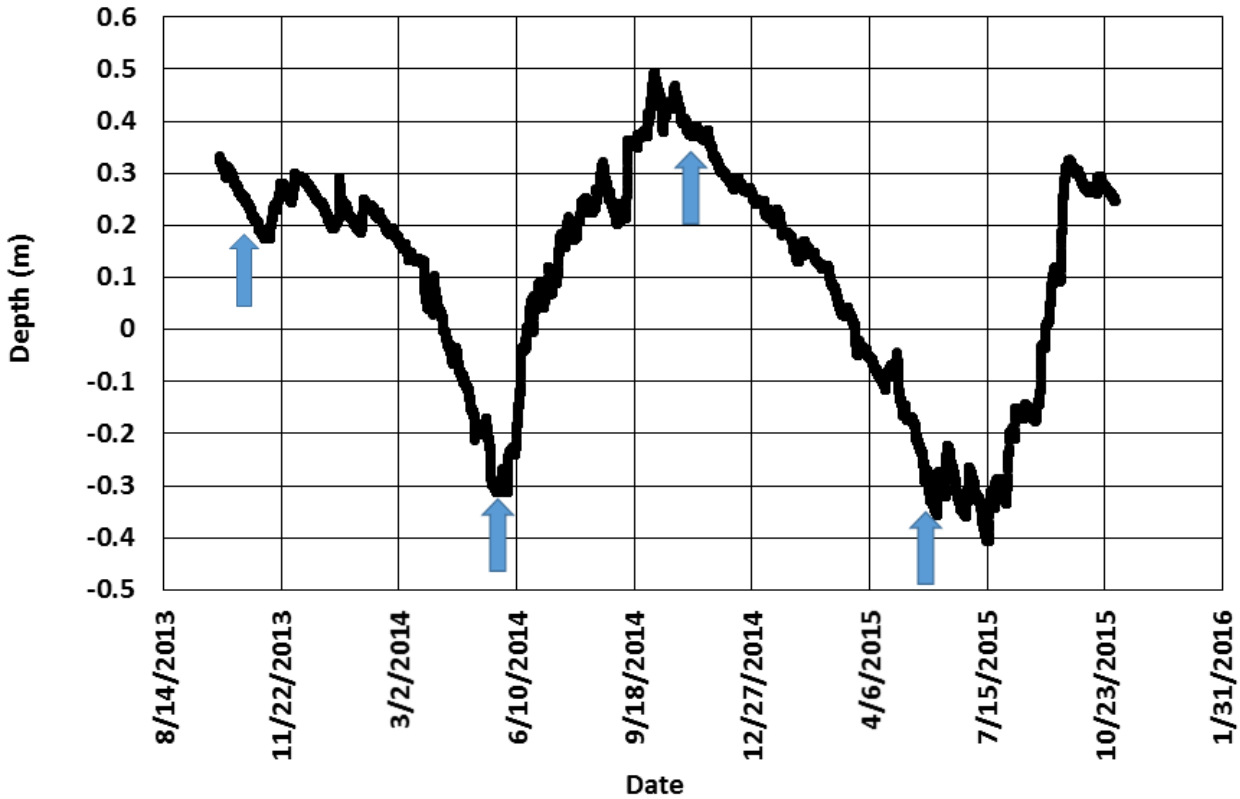


Figure 6-16. The LILA hydrograph (water depth) at the site of the fertilizer treatments and high profile controls (on the ridge) for the study period. The hydrograph for the site of the double and float treatment and low profile controls was nearly identical (SFWMD 2013-2015). The blue arrows indicate the hydrology for the photographs in **Figure 6-17**.

Figure 6-17. Photographic examples of the hydrology experienced by the saplings during the study. Top left: at deployment, late October 2013; Top right: late May 2014; Lower left: November 2014; Lower right: early June 2015. The peat bags floated for the first wet season but were inundated during the succeeding wet seasons. Photos courtesy of SFWMD.

Statistical Analysis

The data were statistically analyzed in SAS (SAS Institute, Inc., Cary, North Carolina, USA) using a mixed model analysis of variance (ANOVA) with the Proc Mixed procedure. The dependent variables, *basal diameter (growth rate)* and *height (growth rate)*, were modeled as a function of the independent variables, *species*, *treatment*, and *time*. *Time* represented the repeated measurements on the plants in the study. The errors (residuals) from the statistical analysis were independent (uncorrelated) over time and thus were modeled with the standard variance components covariance structure. Due to heterogeneous variances between the species, a single set of variance-covariance estimates would not be valid to represent the species effect. Thus, the covariance structures above were utilized to obtain appropriate variance-covariance estimates for each species group.

The double treatment had two plants per peat bag, and measurements were taken on each plant. To utilize the measurements on both plants, data were analyzed in three ways: (1) "One": using measurements of the first plant in a container, (2) "Two": using measurements of the second plant in a peat bag, and (3) "Averaged": averaging the measurements of both plants in a peat bag. A significance level of $\alpha = 0.05$ was used to determine statistical significance for all analyses.

Results

Hydrology

A continuous plot of water depth for the study is presented as **Figure 6-16**. Surface water was at ground level (depth at zero) on the ridges used for the study. The water depths corresponding to the photos in **Figure 6-17** are denoted by arrows within the stage plot. Sample photos of saplings at different water depths are shown in **Figure 6-17** to demonstrate the changes in hydrology experienced by the saplings over the study period. At planting and deployment, the peat bags floated during the duration of the first wet season (2013), approximately 12 weeks. During subsequent wet seasons (2014 and 2015) the bags did not float. During the two years, all but five plants (one *Acer*, four *Ficus*) survived, experiencing water depths ranging from approximately 0.5 m above the ridge (October 2014) to 0.4 m below the surface of the ridge (July 2015). Many of the saplings lost their leaves during high water.

Species Effects

The statistical analysis of the basal diameter data yielded a statistically significant ($F = 25.91$, $df = 2.36$, p-value < 0.0001) species effect for all species interactions. There was a statistically significant height-species ($F = 5.70$, $df = 2.36$, p-value $= 0.0071$) effect for *Annona* versus *Acer* and *Annona* versus *Ficus*. However, these differences were not unexpected, given the differences in sapling size at the initiation of the study. Most of the *Annona* sapling were much greater in height than either the *Ficus* or the *Acer* at deployment. During the study, *Annona* saplings often lost leaves to herbivory (by eastern lubber grasshoppers [*Romalea guttata*]) and many of the *Ficus* saplings showed signs of scale insect infestations.

Treatment Effects

Sapling growth showed a great deal of variability, however, the one-spike fertilized treatment data yielded a statistically significant ($F = 8.48$, $df = 2.36$, p-value $= 0.0010$) treatment effect compared to the controls and compared to the two-spike treatment for basal diameter of all three species. Interactions between species and treatments were not statistically significant (p-values > 0.05).

Throughout the study, *Annona* had greater basal diameter growth and appeared healthier overall, but were also the largest at planting. *Ficus* generally grew well in the fertilized treatments but typically lost leaves and became infested with scale insects during times of high water. *Acer* saplings were the smallest of the three species at planting and many saplings showed growth spurts, but generally lost all leaves during times of high water. Over the course of the two-year study, the surface of the soil in the pots and the

punctures in the top surface of the peat bags provided sites for the colonization of herbaceous plants, ferns, and willows (*Salix caroliniana*).

Discussion

Ranking by growth (height and basal diameter); the fertilized trees generally did the best (the one spike treatment was generally better than two), followed by the controls and the flotation treatment (little difference from controls). The double tree plantings were not statistically different from the controls. The reduction in growth by some saplings receiving the two fertilizer spikes as opposed to those with one spike may be attributable to over-fertilization. Over-fertilization has been suggested as causing poor growth by altering soil chemistry and the soil microbial community, as well as leading to poor root establishment and reduced water absorption (Weinbaum et al., 1992).

Annona showed the greatest basal diameter growth and *Ficus* generally greater height growth. *Ficus* demonstrated greater basal diameter growth than *Acer*. *Ficus* and *Acer* were not significantly different in height growth. The results of the current study suggest that for planting ghost tree islands with saplings in peat bags, the simplest and most effective configuration gives the highest profile, (greater height, less flooding) with one fertilizer spike, and without additional flotation. Since the growth of the saplings in double planted bags was less than the controls, this treatment should not be used.

Removal of the plastics is intended for the future of the current study to determine if the peat continues to be held in place sufficiently by the tree roots to allow the trees to survive after the removal of the pot and plastic bag. Another possible solution would be to utilize biodegradable materials for the tethers (wooden or bamboo poles), pots (peat pots), and for the peat bags (burlap or a similar material) with the intent that the material would maintain integrity and contain the soil and peat until the tree sapling was established in place. A scaling-up this LILA experiment to three or more ghost tree islands in WCA-2A is recommended to estimate the full costs and benefits of our peat-bag-planting design.

Relevance to Water Management

Developing a cost-effective method for planting trees on degraded tree islands is essential to the restoration of these important features of the Everglades. We have demonstrated a potential technique for planting trees in regions of limited access and variable hydrology. This technique would best be employed in peat or organic soils-based regions where some topography has been lost, yet there is a potential to restore forested areas, using degraded Everglades tree islands as an example. The planted trees can provide a seed bank for the recruitment of native species. Restored tree islands will provide habitat and contribute to Everglades oligotrophy. Our observations and measurements indicate that the approach described above is a viable one. Future studies and activities should include other pertinent tree island species and deployment on select ghost tree islands in WCA-2A.

VEGETATION MAPPING OF EVERGLADES NATIONAL PARK AND BIG CYPRESS NATIONAL PRESERVE

Pablo L Ruiz[4] and Theodore N. Schall[5,6]

Introduction

The CERP RECOVER vegetation mapping effort encompasses approximately 9,700 square kilometers (km^2; 2.8 million acres) of public and tribal held lands within the Greater Everglades. To date, approximately 4,420 km^2 (1.1 million acres) of the Greater Everglades have been mapped (**Figure 6-18**). The remaining areas include ENP, portions of BCNP, and several wildlife management areas. Funding for this project is being provided by the United States Army Corps of Engineers (USACE) and the National Park Service (NPS) National Vegetation Inventory Program.

The vegetation mapping of ENP and BCNP is being accomplished through a cooperative partnership between SFWMD, USACE, and the NPS's South Florida/Caribbean Network (SFCN) and National Vegetation Inventory Program. The goal of this cooperative effort is to produce a spatially and thematically accurate, pre-restoration vegetation map of ENP and BCNP that can be used to (1) monitor the spatial extent, pattern, and proportion of plant communities as they respond to management driven hydrologic modifications resulting from CERP implementation and (2) provide baseline vegetation and land cover information to NPS park managers and scientists for use in park management, resource management, research, and monitoring.

This project was initiated in 2006 with a subcontract with Avineon, Inc. to map one-third of BCNP (891 km^2, 220,292 acres). In 2011, SFWMD completed a 406 km^2 (100,378 acres) section of ENP that included the northernmost region of Shark River Slough and the adjacent wet prairie habitats. Upon completion of this initial section, the photo-interpretation portion of the ENP and BCNP mapping effort was transferred to the USACE Jacksonville District office in Jacksonville, Florida; project data collection and field support were delegated to SFCN in South Florida. In 2015, most of the mapping responsibilities associated with this project were transferred to SFCN. This change allowed the project to take advantage of SFCN's institutional knowledge on botany, community ecology, and remote sensing as well as efficiencies afforded by the network's proximity to ENP and BCNP. USACE priorities then shifted towards the completion of Region 1 and assisting the SFCN with the final quality assurance and quality control of mapped products.

Methods

Due to the spatial extent of this project (7,444 km^2; 1.84 million acres), the area was divided into seven regions; Regions 1 through 4 are located in ENP and Regions 5 through 7 are located in BCNP (**Figure 6-18**). Region 1 and Regions 2 through 6 are being mapped by USACE and SFCN, respectively; mapping of Region 7, which fell outside of the CERP footprint, was contracted out by NPS to Avineon, Inc. and completed in 2009.

---

[4] South Florida/Caribbean I&M Network, National Park Service, Palmetto Bay, FL.

[5] United States Army Corps of Engineers, Jacksonville District, Jacksonville, FL.

[6] Special thanks to Helena Giannini, Kevin R. T. Whelan, Michelle Prats, Craig Perry, Michael Foguer, and Joe Ingram of NPS, Cynthia Irvin of USACE and Alejandro Arteaga and Mary-Joe Hernandez of the Geographic Information System Center, Florida International University. Addition thanks to April Patterson from USACE and Karl Brown, Agnes McLean, David Rudnick, and Jed Redwine from NPS.

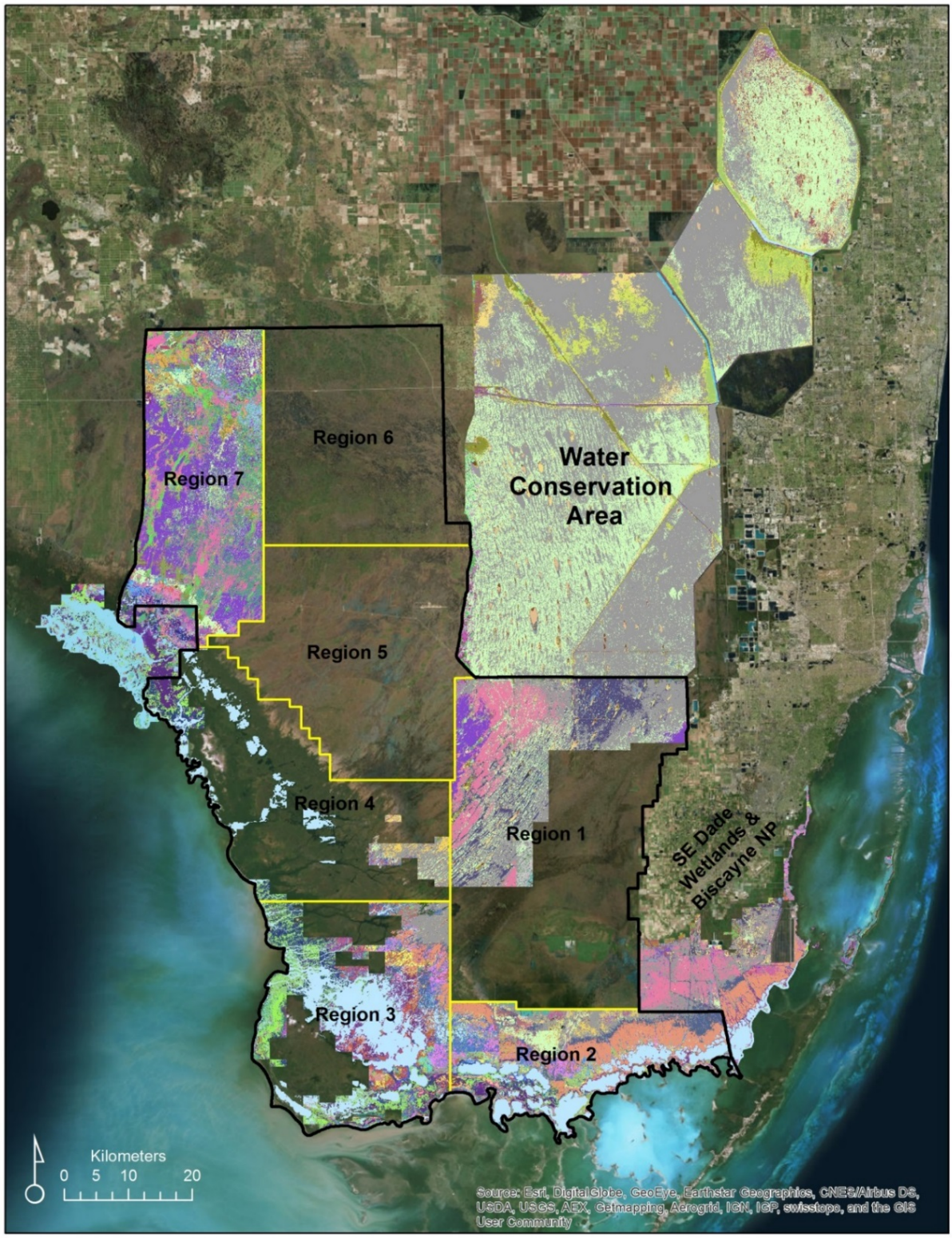


Figure 6-18. Extent of the RECOVER vegetation mapping program showing areas previously mapped and those that are pending. The current mapping effort is focused on Regions 1 through 4 in ENP. Work on Regions 5 and 6 in BCNP will begin when Region 3, in ENP, is complete.

The RECOVER vegetation mapping effort employs a grid-based, ¼-ha (50 m x 50 m cell), mapping approach instead of the more traditional vector method, where vegetation communities are delineated based on observable or perceived boundaries at a set minimum mapping unit. Photo-interpretation of each grid cell is performed by superimposing a two-dimensional ¼-ha grid network, as an ArcGIS shapefile, over digital color-infrared stereoscopic imagery on a photogrammetric workstation. At each grid cell, the dominant vegetation community present is classified using the *Vegetation Classification System for South Florida Natural Areas* (Rutchey et al. 2006). In addition, at each grid cell the photo-interpreters note any evidence of disturbance as either anthropogenic, fire, freeze, or windstorm and categorize the absolute cover of cattail (*Typha* spp.) and any exotic species present as either: Sparse (10%–49%), Dominant (50%–89%), or Monotypic (90%–100%).

The imagery for all four regions of ENP consists of multispectral digital color-infrared (Red, Green, Blue, Near Infrared) aerial imagery at a spatial resolution of 0.30 m, which was acquired in 2009 by SFWMD. In 2014, SFCN contracted Photo Science, Inc. to acquire similar aerial imagery at a spatial resolution of 0.15 m for Regions 5 and 6. The imagery for Region 7 consists of digital scans of 1:24,000 film-based color-infrared (Red, Green, Near Infrared) aerial images acquired in 2006 by Avineon, Inc.

The thematic accuracy of the ENP and BCNP vegetation maps will be assessed using approximately 1,700 randomly selected accuracy assessment grid cells distributed throughout all seven regions. All accuracy assessment data were collected by SFCN (Regions 1 through 6) and a contractor (Region 7) within one-year of image acquisition. Since these data represent the vegetation community found within each grid-cell at the time the imagery was collected, timely accuracy assessment data collection ensured that temporal changes resulting from natural community succession or perturbation events (e.g., fires and windstorms) would not affect the final map accuracy. In addition, to date 5,423 spatially-explicit training point and verification points have been collected to assist in photo-interpretation. The training point data provide species-specific compositional data and structural information while the verification point data provide community type information.

Results and Discussion

To date, approximately 52 percent of ENP has been mapped. The mapping of Region 2 is complete but final quality assurance, quality control, and accuracy assessment is pending. Mapping of Region 1 is approximately 44 percent complete, Region 3 is 72 percent complete, and Region 4 is 18 percent complete. Mapping in Regions 5 and 6 in BCNP will begin once Region 3 is completed in fall 2016. Region 7 in western BCNP, which was mapped by Avineon, Inc., is complete. Distribution of data and map products for this region are pending the completion of a NPS natural resource technical report. The first deliverable for the ENP and BCNP vegetation mapping project (Region 2), is expected in the latter part of 2016, with subsequent deliverables ready each year thereafter through 2019. Additional map products from this project will include landscape-level distribution and cover maps for cattail and exotic species, as well as disturbance maps.

Relevance to Water Management

The marsh and prairie communities of the Greater Everglades are known to respond quickly to natural and management driven hydrologic variation (Nott et al. 1998, Armentano et al. 2006, Zweig and Kitchens 2008, Sah et al. 2013). Due to community resilience and species specific hydrologic tolerances, the rate and direction of vegetation change resulting from the CERP will likely vary spatially throughout the Greater Everglades in response to differences in the historical, management-driven, hydrologic regime of each region, e.g., WCA-3A versus WCA-3B. The CERP RECOVER vegetation maps provide a mechanism by which managers can track and document the rate and direction of these changes. In addition, the vegetation map will allow hydrologist and ecologists to run fine-scale predictive models of potential vegetation change by applying known community hydrologic tolerances to different hydrologic scenarios at an ¼-ha scale.

The model runs will aid in determining which hydrologic regime is best suited for each region within the Greater Everglades.

ECOSYSTEM ECOLOGY

This section describes two projects with foci on more holistic, multidisciplinary objectives. The first project presents the results of the active marsh improvement project focusing on the relationship between nutrients, vegetation, and faunal use. The second ecosystem project is a detailed description of ecological monitoring and research conducted in Florida Bay including water quality (temporal and spatial analyses) and benthic vegetation (monitoring and modeling relative to restoration projects).

CATTAIL HABITAT IMPROVEMENT PROJECT

Susan Newman, Michael Manna, and Mark Cook

Restoration of P impacted regions of the Everglades requires not only a reduction in P loads and concentrations, but also active management efforts to reduce the resilience and resistance inherent to the cattail regime (Hagerthey et al. 2008). The Cattail Habitat Improvement Project (CHIP) is a large-scale in situ study designed to test the ability to rehabilitate habitat in cattail areas by creating open water areas. Using a combination of herbicides and fire, open areas were created in enriched and moderately-enriched areas of WCA-2A in July 2006. In 2009, the first phase of the project was completed, and demonstrated that, relative to controls, created openings maintained higher dissolved oxygen concentrations, submerged aquatic vegetation (SAV)/periphyton dominated the vegetation community, small fish species rather than crayfish dominated the aquatic faunal biomass (Hagerthey et al. 2014), and wading bird foraging was extensive. This study was conducted using broad spectrum (i.e., non-selective) herbicides to maintain the cattail, therefore long-term vegetation succession may be limited. However, a study implemented in 2009 suggested that a recent aquatic registration of a selective herbicide, imazamox (Clearcast™) may be more effective against cattail, but minimally damaging to other plant species (Rodgers and Black 2012). To gain greater understanding of the effectiveness of this herbicide, and facilitate vegetation succession and continued habitat improvement, a strategy using imazamox was implemented in the CHIP plots in 2011.

Methods

CHIP is a large-scale in situ study, using fifteen 6.25-ha plots, initiated to test our ability to rehabilitate cattail areas. The objectives of this second phase are *how the use of a more selective herbicide affects the structure and function of the rehabilitated habitat compared to the natural Everglades* and *whether habitat improvement is sustainable in the long-term*. The original hypotheses, experimental design, and rationale behind this research project were described in Sklar et al. (2007). For all results, sites are delineated based on their location; enriched (E), transitional (T), or reference, (UC) and whether or not they were herbicide/burned, open (O), or controls that were untreated and dominated by cattail (C). Vegetation responses are based on aerial imagery using photographs collected at an oblique angle. Soil nutrients are obtained using a 10-cm diameter thin-walled stainless steel corer, and compositing 3 soils cores per plot. Aerial wading bird surveys were conducted approximately weekly during the dry seasons (November to May; n = 15 to 24 survey weeks per year) of WY2007–WY2015. Each mean was calculated by first summing the number of birds observed each week per plot and dividing the total per plot by the number of observation weeks (range 15 to 24 weeks) to give the average number of birds per week per plot. Given the extremely low numbers of birds in control plots, counts in controls were suspended in 2009. As the plots matured, large numbers of waterfowl were observed, and duck and coots were included in the weekly surveys starting 2012 and 2014, respectively.

Results and Discussion

Vegetation

As we previously documented, the response to the herbicide/fire treatment was the creation of an open water/ SAV dominated ecosystem (**Figure 6-19**).

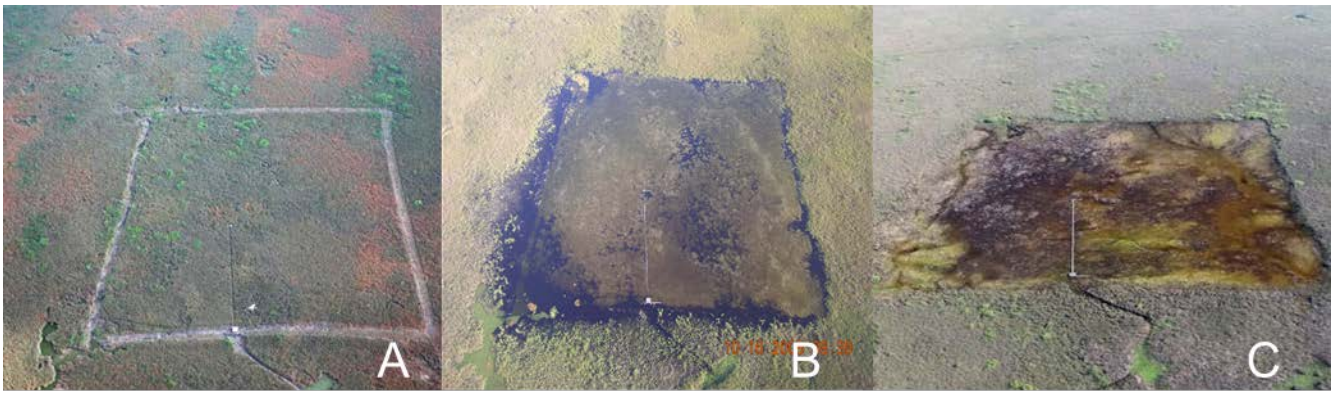


Figure 6-19. Aerial imagery documenting the vegetation community in plot TO1 during the initial phase of CHIP when a combination of glyphosate and imazapyr were used to manage cattail. (A) April 2006, prior to spray; bright green is willow, brownish red is cattail, and blue-green is sawgrass. (B) October 2008, black color is open water and tan/green color is a combination of SAV (primarily *Chara* spp.) and algae. (C) May 2010, greater density of the *Chara*/algal community.

In 2011, the switch from a broad spectrum herbicide to a more selective one provided an opportunity for species that were less sensitive to imazamox to grow and expand within the enriched open and transitional open plots. The most visible change was the initial appearance of *Eleocharis* spp. (spikerush) "circles" (**Figure 6-20A**). In an earlier study of seedbanks along the nutrient gradient in WCA2A, spikerush seeds were only found at a slough site, not within cattail or *Cladium jamaicense* (sawgrass) dominated areas (Van der Valk and Rosburg 1997). Similarly in our own germination studies conducted during 2008–2009, both cattail and sawgrass had significant seed germination, generally higher with the degree of enrichment. By contrast, spikerush seedlings were only present at the unenriched reference sites, and only under saturated conditions; no germination was observed under flooded conditions. We hypothesize that spikerush seeds produced prior to cattail invasion were buried beneath > 10-cm sediments created by cattail and sawgrass than we originally sampled, and current individuals were introduced via waterfowl; as demonstrated by Brochet et al. 2010, and germinated in the open plots when they dried down during the dry season (**Figure 6-20B**). Once established, with cattail competition eliminated, they were able to expand. Spikerush is a clonal species, expansion was initially observed as the expansion of circular stands, and the subsequent colonization of large sections of the plots (**Figures 6-20C** and **6-20D**).

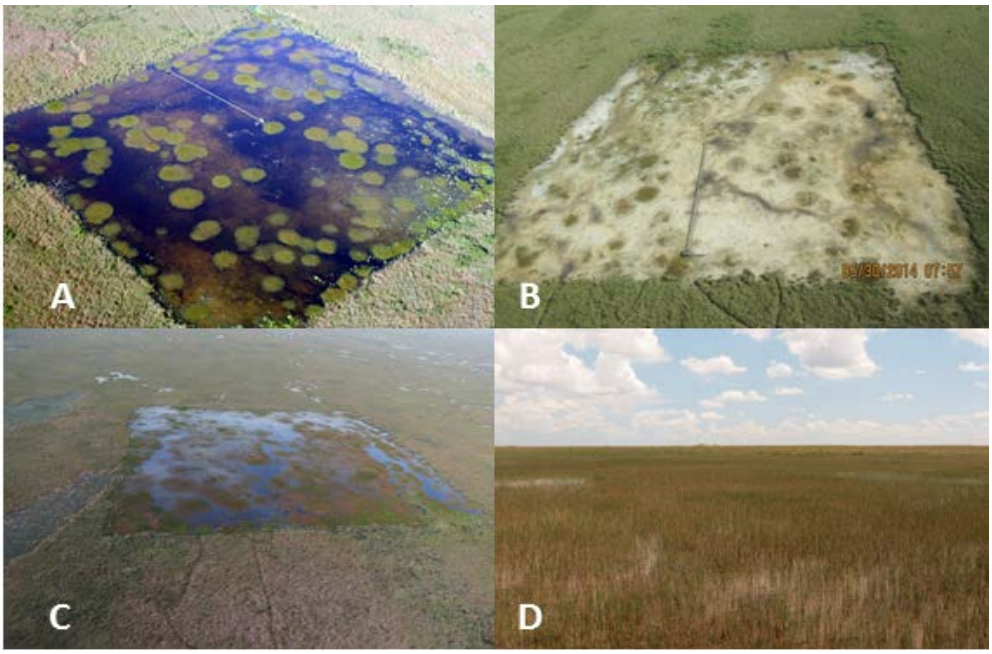


Figure 6-20. Aerially imagery of plot transitional open 1 showing the vegetation community development since the switch to imazamox (April 2011) for periodic control of cattail. Sprays occurred April 2011, January 2013, and December 2014. (A) April 2012: clonal areas of spikerush (*E. cellulosa*), which are the green circles. (B) May 2014: calcareous sediment produced from periphyton/SAV matrix shown as white area as plot dries out and spikerush is the green circles. (C) September 2015: brownish color indicative of seasonal senescence shows spatial distribution of spikerush throughout the plot. (D) September 2015: close up of spikerush area within the plot.

Biogeochemical Response

At the beginning of the study there was a significant increase in the TP content of the floc (**Figure 6-21**), attributable to the mass load of detritus to the sediment from the vegetation that was below the water surface upon burning. The increase was greater in the enriched open versus transitional open plots due to higher original nutrient content in that area. However after 3 to 4 years, the TP content of the floc in the treated plots is similar to that in the controls. We hypothesize that over time the original floc will decompose and newly generated floc will dominate. The organic and inorganic material will be produced by the novel vegetation community; SAV, algae, and *Eleocharis* spp. TP in algae and SAV are primarily derived from water column sources, likely to have lower TP than those that remove TP from the sediments. In addition, significant accumulation of the more calcium rich sediments in the open plots, may help cap the soil and reduce the flux of legacy TP.

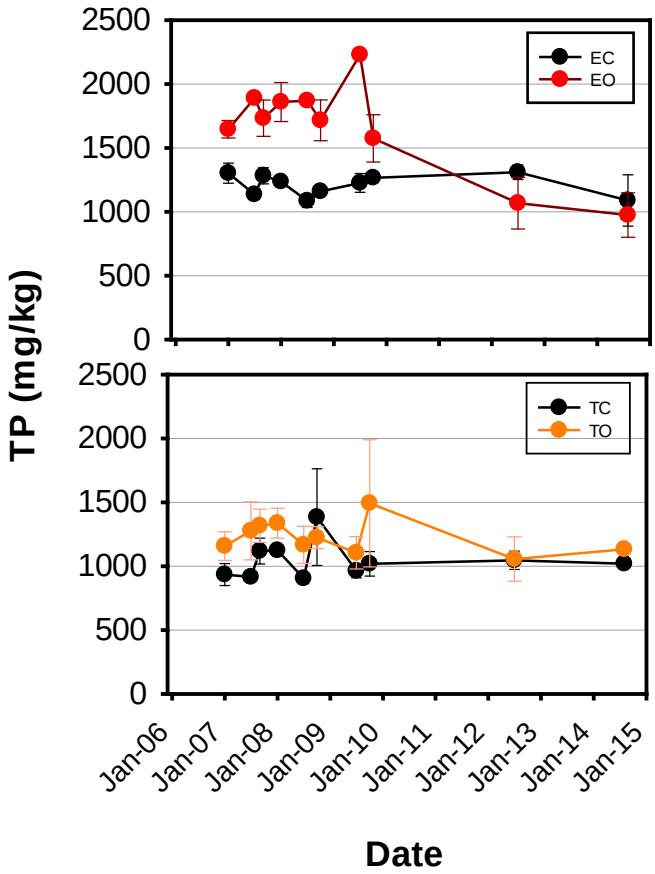


Figure 6-21. Change in the TP content of the surficial floc in control (cattail) and created open lots since the initiation of the project. Sites enriched cattail (EC) and enriched open (EO) are in more enriched sections of WCA-2A compared to sites transitional cattail (TC) and transitional open (TO).

Faunal Response

We previously demonstrated that this management approach to create sloughs produced habitat that was highly attractive to foraging wading birds (Sklar et al 2008). We can now confirm that the quality of the habitat was sustainable in the longer-term (**Figure 6-22**). There was little overall difference in wading bird use between open plots in the transitional and enriched regions, which on average support significantly higher wading bird numbers than the unenriched Everglades. What is not evident from these graphs is the temporal component of bird use within years. Birds in the eutrophic regions foraged in large numbers for many weeks while in the oligotrophic region peak feeding was limited to a week or two. This occurred because aquatic prey production is considerably greater in the eutrophic open CHIP plots than in the oligotrophic reference region (Hagerthey et al. 2014). Wading bird foraging in CHIP is not constrained by the typical mechanisms that drive prey availability in the Everglades such as water level recessions and the extremely shallow conditions that function to concentrate limited prey resources (e.g., Frederick et al. 2009). As such, birds in CHIP foraged effectively over a greater range of hydrological conditions, including during water level reversal events and deeper conditions when concentrated prey redisperse into the marsh and often lead to abandonment of nesting colonies. CHIP, therefore, plays a critical role as a refugia for foraging birds when hydrologic conditions preclude effective foraging elsewhere in the ecosystem.

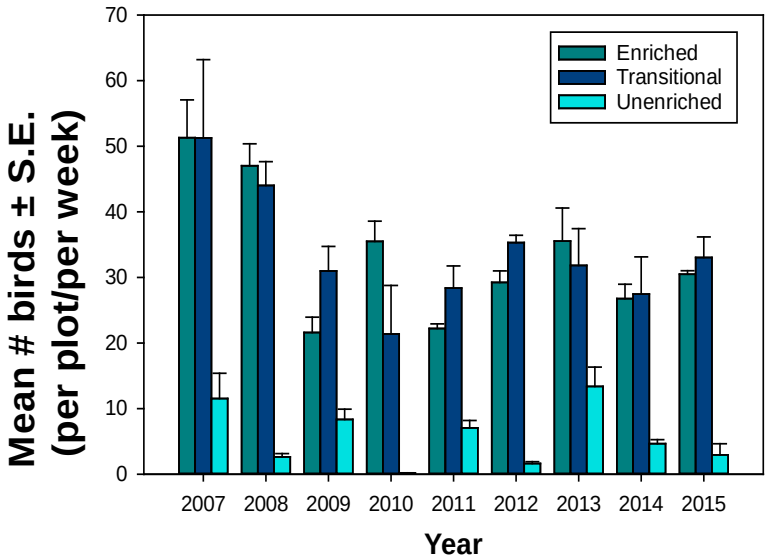


Figure 6-22. Mean number of wading birds per plot per week since the initiation of CHIP. Only open (treated) and reference plots are shown. Due to extremely low numbers of wading birds in control plots, monitoring of those was discontinued after 4 years (data now shown).

The open plots also provided excellent foraging conditions for dabbling ducks (*Anas sp.*), primarily blue-winged teal (*Anas discors*; **Figure 6-22A),** and coots (*Fulica* spp.; **Figure 6-22B**). The dabbling ducks showed a clear preference for the different regions with enriched open > transitional open > reference plots. By contrast, the coots did not show any preference between enriched and transitional sites, but similar to ducks, were rarely observed in the unenriched Everglades.

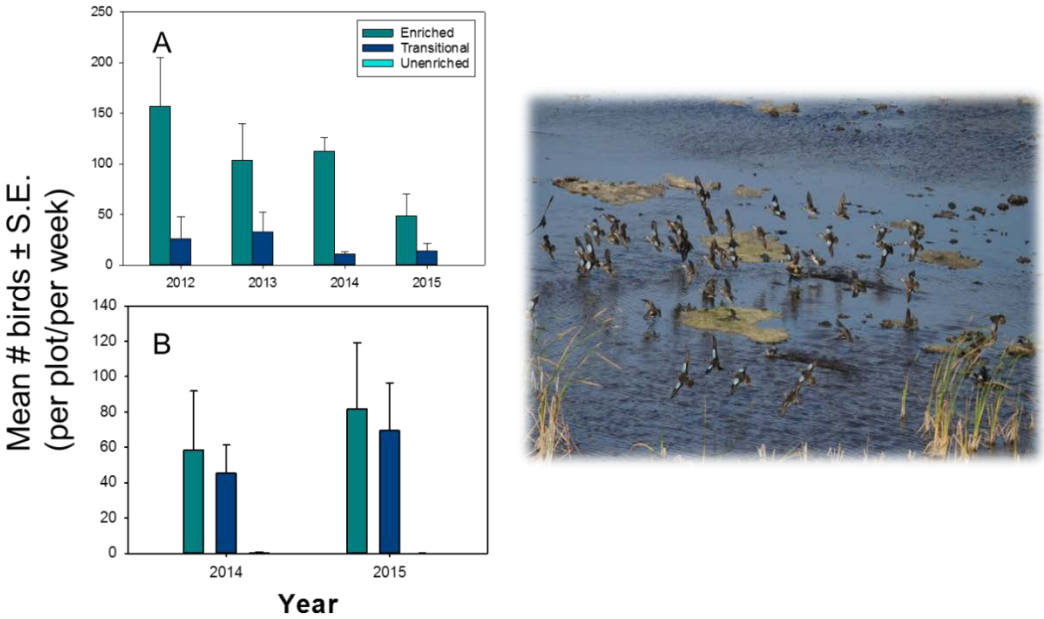


Figure 6-22. Mean number of birds per plot per week ± standard error (S.E.) in the open and reference plots of CHIP: (A) ducks and (B) coots. Photograph at right shows the abundant duck usage of an open water plot.

Relevance to Water Management

While nutrients loads to the Everglades have been reduced, the P-enriched downstream ecosystem is resilient and resists change. In previous SFERs, the results from the CHIP study demonstrated that, by managing the vegetation, we were able to create alternate regimes, which altered biogeochemical cycling and provided foraging areas for wading birds in areas previously inaccessible. However, the herbicides used were broad spectrum, thus desirable vegetation, in addition to cattail, were damaged or killed when areas were resprayed. By switching to a more selective herbicide, we were able to improve upon the original design. Vegetation imagery indicates the relatively rapid succession to a more diverse and desirable community, which in turn may change how TP accumulates in the soils. These open water habitats are proving to provide a sustainable foraging habitat for wading birds, an extremely important refuge when conditions are deeper elsewhere, or when a reversal occurs.

THE FLORIDA BAY PROJECT: WATER QUALITY CONDITIONS AND STATUS

Stephen P. Kelly

Water quality in Florida Bay (eastern, central, and western regions) has been monitored since 1991 (WY1992) to ensure that District operations and projects protect and restore the coastal ecosystem to the extent possible. CERP performance measures focus on Chla concentration, an indicator of algal biomass, as well as the nutrient inputs that initiate and sustain blooms. Operational changes to the South Dade Conveyance System, including implementation of the C-111 South Dade Project, Modified Water Deliveries to ENP (especially Tamiami Trail modifications), and the C-111 Spreader Canal Western Project, which became operational during WY2013, will further change freshwater flow patterns and may alter downstream water quality. Presented here is a brief update of ecologically important parameters and how they have changed in WY2016. For a complete description of water quality assessments in Florida Bay, refer to Chapter 6 of the *2015 South Florida Environmental Report (SFER) – Volume I* (Sklar and Dreschel 2015).

Water samples and physical parameters (temperature, salinity, conductivity, pH, and dissolved oxygen) are collected every other month at all sites. Samples are collected at 0.5 m below the surface and processed according to the SFWMD *Field Sampling Quality Manual* (SFWMD 2015b) following Florida Department of Environmental Protection (FDEP) protocols. Physical parameters are collected with a calibrated multi-parameter water quality sonde following SFWMD protocols. Samples are processed on site, stored on ice, and shipped overnight to the SFWMD Analytical Lab in West Palm Beach for analysis according to the SFWMD *Chemistry Laboratory Quality Manual* (SFWMD 2015a) and following FDEP protocols. All sample results are quality assured before being uploaded to DBHYDRO.

There are many factors that influence the nutrient and Chla concentrations and variation in the bay including inflows, storm and wind events, circulation patterns, nutrient recycling, and even construction events. It is not possible to attribute any one of these factors as the most important driver of current conditions, but rather it is likely a synergistic effect of some or all of them (Chapter 12 [Abbot et al. 2007] and Appendix 12-3 [Rudnick et al. 2007] of the 2007 SFER – Volume I). Additionally, starting in July 2015, a large die-off of SAV, primarily turtle grass (*Thallasia testudinum*), occurred in the central region of the bay encompassing approximately 40,000 acres. The die-off is hypothesized to have been caused by a localized drought in South Florida that began in 2014 and continued in 2015. This resulted in reduced fresh water flows to the bay and, along with extreme temperatures, salinities, and dissolved oxygen levels, likely triggered the die-off. The large quantity of dead seagrass leaves as well as the decaying root systems

will likely release nutrients into the water column when temperatures begin to rise in August 2016 (WY2017) and could trigger an algal bloom.

Annual averages of all parameters analyzed—Chla, TP, TN, TOC, dissolved inorganic nitrogen (DIN), and turbidity—continue to be stable during the last seven water years, indicated by little to no interannual concentration changes, and were statistically similar between WY2015 and WY2016 with a few exceptions (**Table 6-6**). The higher nutrients and Chla in the central region may be attributable to the increased precipitation leading to increased freshwater inflows from November 2015 through January 2016 (**Figure 6-11**), the release of nutrients from decaying SAV, or a combination of both. The increase in nitrogen (N) is due exclusively to ammonium (NH_4) at all four sites representing this area in January, February, and April 2016.

Table 6-6. Statistically significant differences ($p < 0.05$) between annual averages of WY2015 and WY2016. Numbers in bold are higher.

Region	Parameter[a]	WY2015	WY2016
Eastern	TN (µM/L)	50.8 ± 6.7	**61.9 ± 9.9**
	DIN (µM/L)	5.14 ± 2.98	**10.8 ± 8.6**
	Turbidity (NTU)	2.53 ± 1.2	**5.11 ± 5.98**
Central	TP (µM/L)	0.22 ± 0.07	**0.41 ± 0.18**
	TN (µM/L)	81 ± 21	**106 ± 34**
	DIN (µM/L)	2.85 ± 1.00	**16.8 ± 21.4**
	Chla (µg/L)	0.53 ± 0.38	**1.88 ± 1.31**
Western	TN (µM/L)	25.5± 6.6	**35.3 ± 21.8**

a. Key to units: µg/L – micrograms per liter, µM/L – micromoles per liter, and NTU – nephelometric turbidity units.

Results in WY2016 were compared to the POR and the 25th and 75th percent quartiles. The POR for all analytes with the exception of TN is WY1992 to WY2013. The POR for TN was reduced to WY2009 to WY2013 due to a possible error in the longer-term data set. All analysis were performed at the Districts' in-house analytical lab starting in January 2009. At that time, there was an approximately 30 percent increase in the TN that was not immediately noticed as the concentrations were within the long-term range of TN. No other analytes were affected and, until an intercalibration of the analytical TN methods can be completed to document if this is an analytical error, the POR was reduced. No conclusions or interpretations were affected by this change.

The WY2016 results were mixed. In the east, TP was lower than the 25 percent quartile while TN was greater than the 75 percent quartile. Other results in the east were also mixed with Chla, DIN, and turbidity less than the 25 percent quartile and discrete salinity collected at the time of sample collection was above the 75 percent quartile during the first half of the water year. DIN rose to more than the 75 percent quartile and TOC and salinity fell to less than the 25 percent quartile during the second half of the water year (**Figures 6-23** through **6-25**, and **Table 6-6**).

In the central region, TN was greater than the 75 percent quartile and turbidity was less than the 25 percent quartile the entire water year. Discrete salinity and TOC were above the 75 percent quartile and Chla and TP were less than the 25 percent quartile during the first half of the water year. During the second half of the water year, salinity fell to less than the 25 percent quartile while DIN rose to well above the 75 percent quartile. The end of the water year showed elevated Chla, TP, and TOC as well (**Figures 6-23** through **6-25**, and **Table 6-6**).

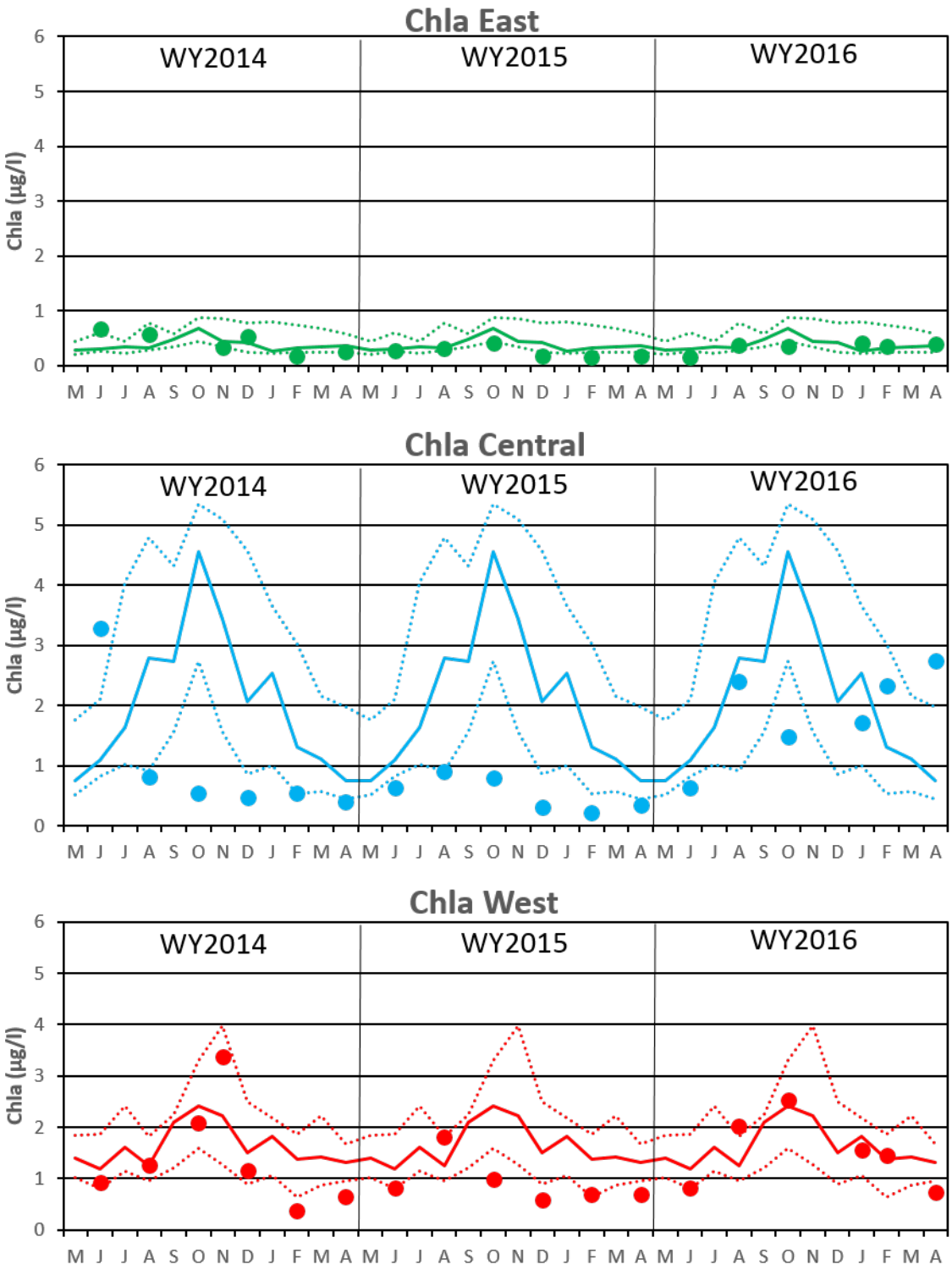


Figure 6-23. Mean Chla concentrations in micrograms per liter (µg/L) in the three regions of the bay studied during WY2014–WY2016 (solid symbols) compared to the monthly median and the interquartile range for the entire POR, WY1992–WY2013 (solid and dotted lines).

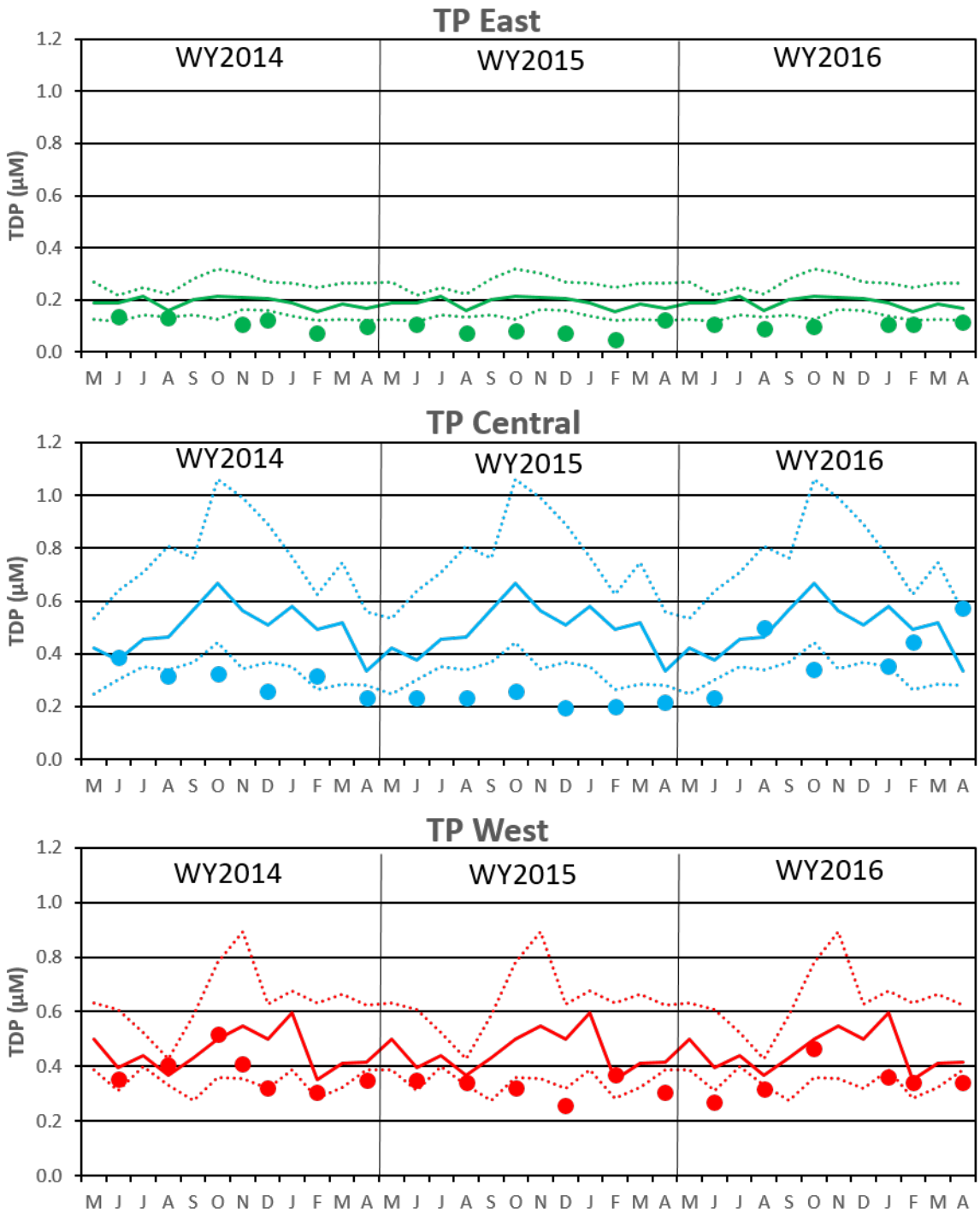


Figure 6-24. Mean TP concentrations in micromoles per liter (µM/L) in the three regions of the bay studied during WY2014–WY2016 (solid symbols) compared to the monthly median and the interquartile range for the entire POR, WY1992–WY2013 (solid and dotted lines). [Note: 1.0 µM/L = 31 µg/L = 31 parts per billion.]

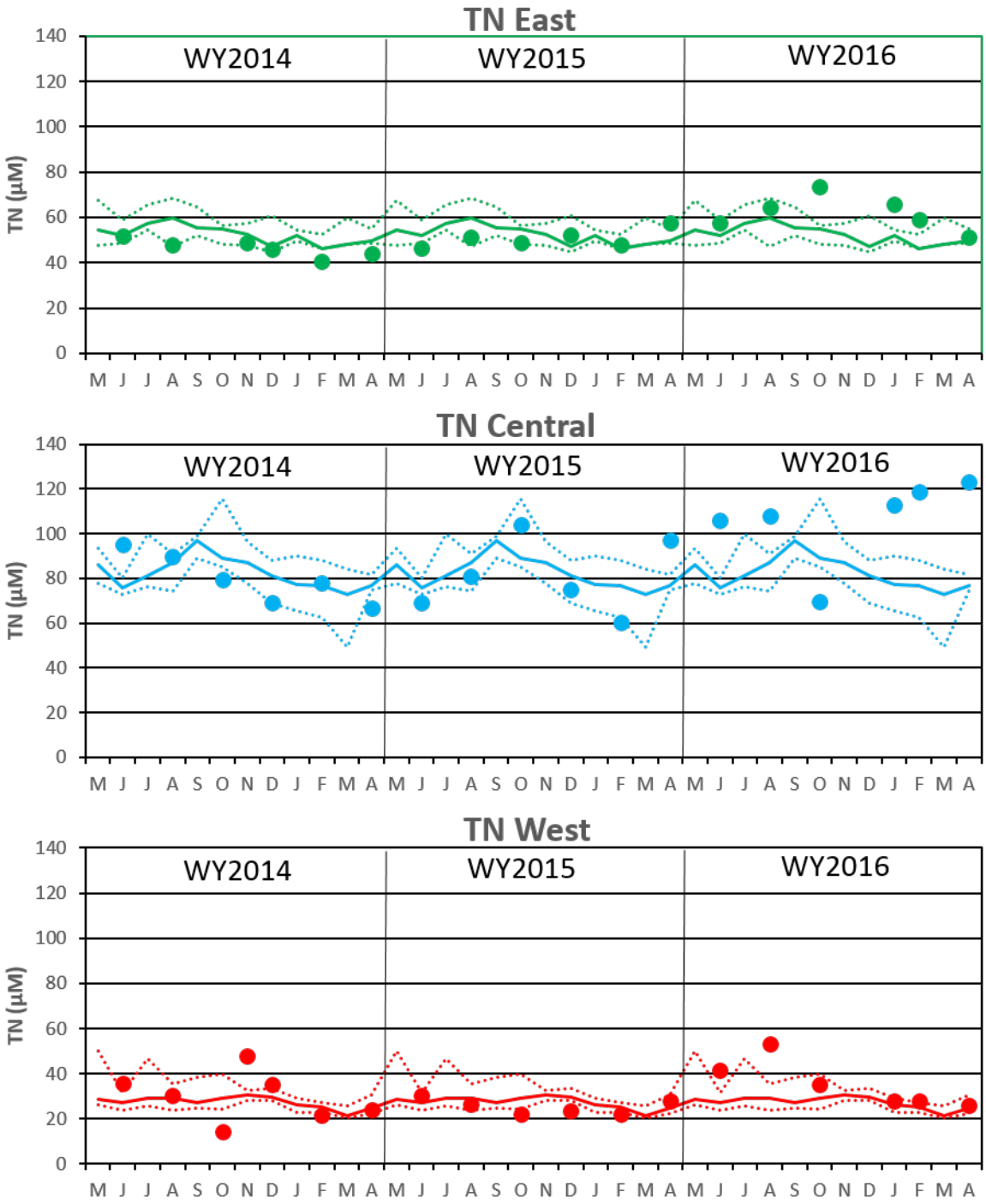


Figure 6-25. Mean TN concentrations in micromoles per liter (µm/L; µM) in the three regions of the bay studied during WY2014–WY2016 (solid symbols) compared to the monthly median and the interquartile range for the revised POR, WY2009–WY2013 (solid and dotted lines). [Note: 100 µM/L = 1.4 micrograms per liter = 1.4 parts per million.]

Table 6-6. Deviation of WY2016 from the POR (WY1992–WY2015) averages unless otherwise noted. Numbers in bold italics are statistically different ($p < 0.05$).[a]

Region	TN[b] (µM/L)	DIN (µM/L)	TP (µM/L)	Chla (µg/L)	TOC (µM/L)	Salinity	Turbidity (NTU)
Eastern	***121 %***	129 %	***45 %***	71 %	87 %	***117 %***	61 %
Central	***134 %***	***266 %***	73 %	77 %	***83 %***	***114 %***	***28 %***
Western	***126 %***	99 %	***67 %***	84 %	***80 %***	102 %	44 %

a. Key to units: µg/L – micrograms per liter, µM/L – micromoles per liter, and NTU – nephelometric turbidity units.
b. TN POR WY2009–WY2013.

To the west, TP, TOC, Chla (with the exception of August), and turbidity were equal to the median or close to the 25 percent quartile while TN and DIN were equal to the median or close to the 75 percent quartile during the entire water year. Discrete salinity was above the 75 percent quartile during the first half of the water year, falling to the 25 percent quartile during the second half of the water year (**Figures 6-23** through **6-25**, and **Table 6-6**).

The following figures are the monthly spatial averages of each parameter in the three regions of the bay compared to the temporal median of the monthly spatial means and the interquartile range for the entire POR (WY1992–WY2013) with the exception of TN as mentioned above.

In summary, all three regions of the bay had elevated discrete salinities during the first half of the water year followed by lower salinities during the second half of the water year. The extended drought in this area, which began in WY2014 and stretched into WY2016, finally abated with the heavy precipitation in November 2015 through January 2016 (**Figure 6-10**). Chla, TP, and turbidity across the bay were at the long-term median or close to the 25 percent quartile with the exceptions noted above. TOC in the east and west were at the long-term median or close to the 25 percent quartile while the central region had mixed results. DIN was elevated baywide at some times during WY2016. It will be important to monitor these water quality parameters for changing nutrient conditions that may lead to algal blooms, especially in the central bay, which experienced the SAV die-off.

Relevance to Water Management

As mentioned earlier, there are many factors that influence the nutrient and Chla concentrations and variation in the bay including inflows. These inflows can be caused by precipitation and also by water management changes upstream of the bay. The localized drought in South Florida that began in 2014 and continued in 2015 resulted in reduced freshwater flows to the bay (**Figure 6-11**). These flows rapidly increased during the very wet dry season beginning in late November 2015. In addition, a temporary emergency deviation to the WCA-3A and WCA-3B water control plan beginning on February 12, 2016, resulted in additional water flowing into ENP and likely Florida Bay. The higher nutrients and Chla in the central region at the end of WY2016 may be attributable to the increased precipitation and freshwater inflows, the release of nutrients from decaying SAV, or a combination of both.

THE FLORIDA BAY PROJECT: *RUPPIA* IN THE MANGROVE TRANSITION ZONE

Christopher Madden and Theresa Strazisar[7]

The maintenance of healthy *Ruppia maritima* (widgeon grass) habitat at the transition zone is critical to ecosystem processes and to wetland fauna including waterfowl. The species is a valued ecosystem component of the Florida Bay transition zone and is a sentinel indicator for the Florida Bay MFL assessment because it requires low salinities during at least part of the year. The MFL regulatory framework is intended to prevent significant harm to the ecosystem by requiring a minimum amount of freshwater flow to the Florida Bay estuary.

Methods

A series of field transplant experiments examined the *Ruppia* community to determine factors that favor recruitment of seedlings to the adult plant population from sexual reproduction, seed germination, seedling and adult survival, and clonal reproduction of new short shoots. At five sites in the transition zone (three eastern and two western) we examined site-specific salinity, temperature, light, and nutrient conditions relative to seedling and adult survival in CY2013–CY2014 (**Figure 6-26**). At each site, five *Ruppia* adults and 25 seedlings were transplanted from the wild to containers in situ. To determine effect of interspecific competition, the plants were replanted either in the presence or absence of the dominant co-occurring SAV at the site (n = 8 and n = 4, respectively). We also examined effects of sediment phosphorus limitation in the eastern ecotone sites where sediment P is 2 to 3 times lower than the western ecotone and apparently limiting to growth (Strazisar et al. 2013). Sediment N is high relative to P and not assessed for limitation. Timed release fertilizer (Hi-Yield® superphosphate) was added to sediments of an additional set of adult and seedling transplants and enhanced *Ruppia* leaf tissue P (1,300 milligrams per kilogram [mg/kg]) to within the range in the western concentrations (840 to 1,615 mg/kg) and reduced carbon (C):P and N:P ratios to 648 and 26, respectively, below maxima in the west (778 and 31, respectively). In CY2014, recruitment was further examined during a large, meadow-forming event covering several hectares in Long Lake that is representative of many meadows in the western chain of lakes between April and August (**Figure 6-26**).

[7] Department of Biology, Florida Atlantic University, Boca Raton, FL.

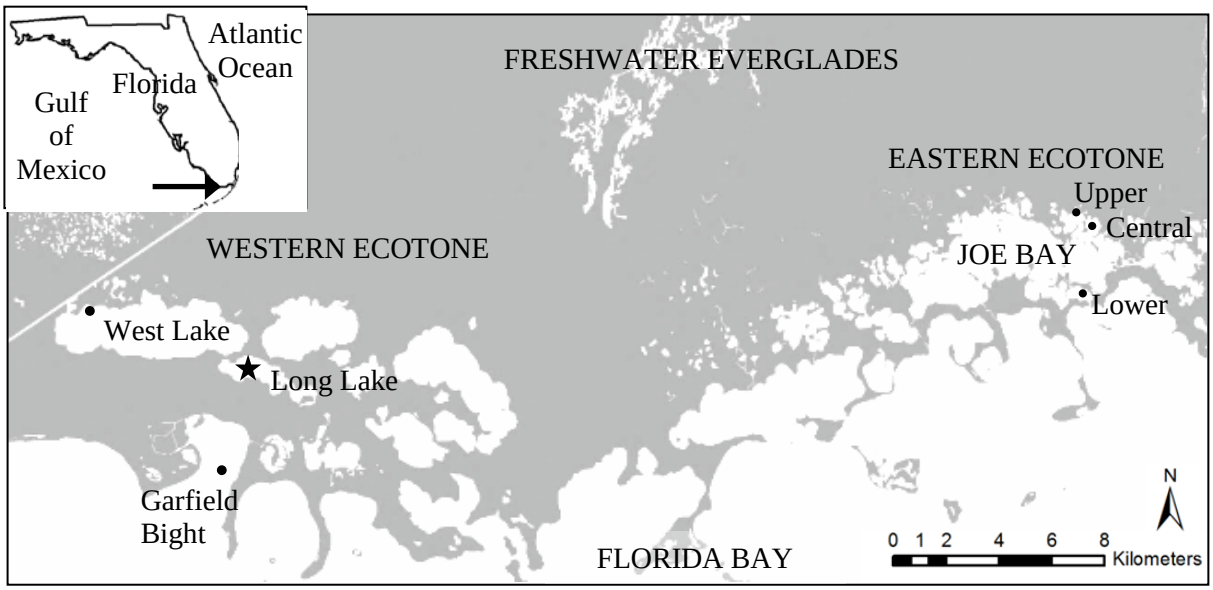


Figure 6-26. Field experiments examined *Ruppia maritima* seedling and adult survival and clonal reproduction of new short shoots across a range of environmental conditions (salinity, temperature, light, nutrients, and other SAV) at five sites across the Everglades-Florida Bay transition zone in CY2013–CY2014 (dots). *Ruppia maritima* recruitment was further examined in a large reproductive meadow in the western transition zone at a site in Long Lake between April and August 2014 (star).

Results

By experimentally increasing sediment P at the eastern sites through fertilizer addition, seedling survival was increased by a factor of 2 to 5 ($p < 0.001–0.04$), reaching 19 surviving seedlings while the density of short shoots nearly doubled at most sites through vegetative reproduction ($p < 0.05$), although sexual reproduction was minimal.

In the West Lake site, sediment P was 2 to 3 times greater than in the east and likely not the limiting resource. Salinity stress was also low there and varied little from 9 there during the year-long study, while variability was extreme in the downstream western site, Garfield Bight, ranging from 7.4 to over 55. In the eastern sites, salinity was highly variable, spanning two orders of magnitude at every site in every season. High salinity variability has been shown to be limiting to productivity due to osmotic stress (Strazisar et al. 2015).

In West Lake, light at canopy height was the primary factor limiting to growth (< 7,000 lumens per square meter [m^2]), as salinities were stable and nutrients were not limiting. Sight-specific light levels were consistently lower at this site than the eastern sites, by as much as an order of magnitude (10,000–53,000 lumens/m^2). Adult survival in the west was generally lower (< 3 of 5 initial individuals) than the east, where up to 5 living adults were observed per site. Vegetative reproduction could only maintain or slightly increase short shoot density in the west (16–30 shoots), while up to 3 times more short shoots were produced vegetatively at eastern sites (≤ 92). When the macroalga *Chara hornemannii* was present in West Lake, light at the canopy dropped to 5 percent surface irradiance, both seedling and adult survival declined to < 1 living individual, on average, and shoot density to ~10, half the initial 20 shoots. However, when the water column cleared and light was not limiting in the western sites, *Ruppia* density increased by an order of magnitude, forming large meadows throughout the lakes, a phenomenon that did not occur in the east.

In CY2014, one of these large meadows in Long Lake was studied to examine the process of seed recruitment. Inspection of the sediments revealed a large seed bank with twice as many viable seeds as just outside the meadow (**Figure 6-27**). Thirty-five percent of seed germinations successfully developed into seedlings that created the dense meadow. During the growing season, all meadow vegetation, including new seedlings, gradually died off and by August, there was no standing vegetation. However, viable seeds in the seedbank quadrupled during the growing season (**Figure 6-27**) indicating that the process of relatively brief meadow formation and sexual reproduction is likely to be important in maintaining the population in the west via seed bank replenishment, providing a base for recruitment in subsequent growing seasons. These meadows may represent hotspots of meadow formation where the seedbank is self-sustaining through rapid adult growth during light-sufficient periods, followed by high seed production.

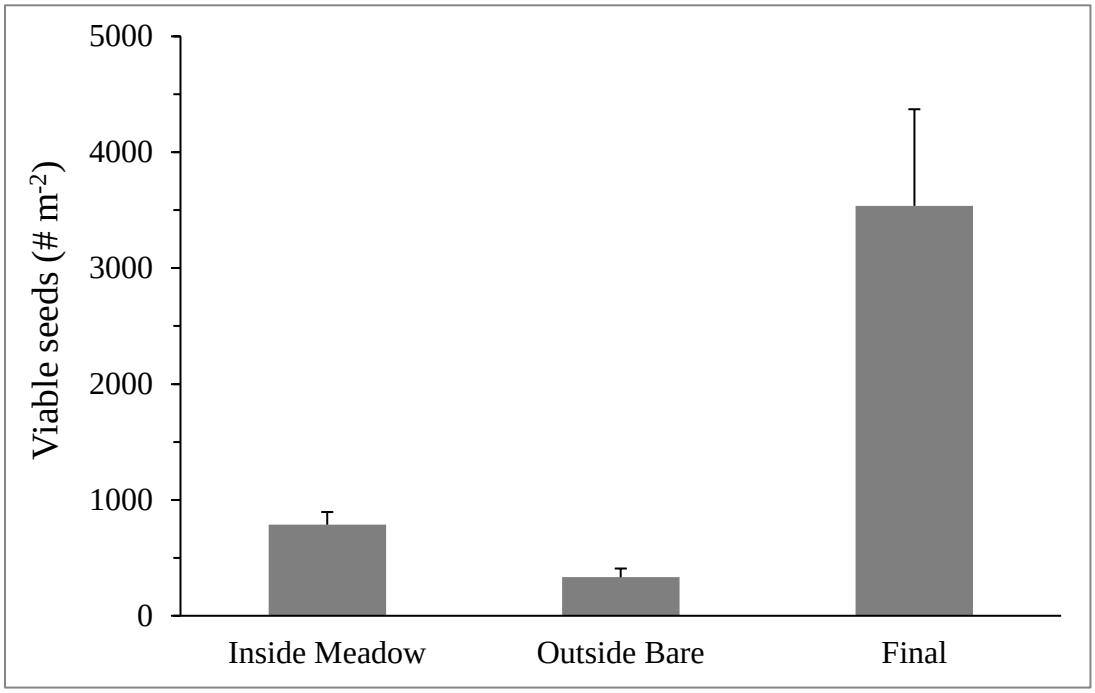


Figure 6-27. Viable *Ruppia maritima* seed densities in the sediment seed bank throughout a large-scale reproductive event in the western Everglades-Florida Bay ecotone in CY2014 (mean ± standard error; n = 20). Sediment seed bank densities were sampled inside a large reproductive *R. maritima* meadow (Inside Meadow) and outside the meadow where vegetation was absent (Outside Bare) on April 3, 2014. Following dieback of vegetation on August 5, 2014, the sediment seed bank was resampled to determine final viable seed bank densities (Final) following the reproductive event.

Relevance to Water Management

It is important to know the salinity tolerance of each growth stage of *Ruppia* so that management of freshwater flows can be optimized to enhance the survival of the population. Resiliency of the *Ruppia* community via the seedbank enables continuation of the population following disruptive events. Adult *Ruppia* can tolerate polyhaline environments for a limited time but because seed germination and seedling survival depends on a low salinity phase, the transition from seed to adult represents a “bottleneck” to recruitment of adult plants. Adult survival and clonal reproduction of new short shoots increases *Ruppia* coverage vegetatively, essential to maintenance of the population. But continued persistence of *Ruppia* in the transition zone appears to depend upon both vegetative production and recruitment of new adults from seed.

FLORIDA BAY PROJECT: FLORIDA BAY BENTHIC VEGETATION

Joseph Stachelek and Amanda McDonald

Benthic vegetation, composed of seagrass and benthic macroalgae, provides habitat structure in Florida Bay and its associated creeks, ponds, swamp forests, and marshes of the mangrove transition zone. Monitoring and research of benthic vegetation is critical to understanding the effects of water management and restoration on wetland and estuarine ecosystems. Results from these efforts are used in assessing the effectiveness of the current Florida Bay MFL rule, which is based on the salinity tolerance of widgeon grass (*Ruppia maritima*). These surveys are used to provide ecosystem status updates for RECOVER, assessments of District operations, and calibrate and verify the Florida Bay Seagrass Community Model (SEACOM).

Methods

Benthic vegetation is monitored regionally in select locations using a randomized design where several 0.25-m^2 quadrats are assessed for benthic vegetation using indices of percent cover. Three separate monitoring programs cover different areas in Florida Bay. The South Florida Fish Habitat Assessment Program (FHAP) and the Miami-Dade County Department of Environment Resources Management (DERM) provide estimates of benthic vegetation cover using a visual index of bottom occlusion at random sites within each basin. FHAP monitors 17 basins throughout Florida Bay and along the southwestern coast every May while DERM monitors the nearshore embayments of northeastern Florida Bay quarterly. In addition, both programs visit fixed sampling locations twice a year, once in the spring and once in the fall. Audubon of Florida monitors SAV using a point-intercept method every other month at sites along nine transects extending from the freshwater marshes of the southern Everglades to Florida Bay. A more complete description of the monitoring programs and the methodologies are presented in Chapter 12 of the 2011 SFER – Volume I (Alleman 2011).

Results

Salinity and water level dynamics during WY2016 in the upper transition zone were very unusual. Extremely high salinities relative to long-term records were observed in the wet season while extremely low salinities were recorded in the dry season (see the *Florida Bay Hydrology* subsection in the *Hydrologic Patterns for Water Year 2016* section).

Two notable trends in SAV cover carried over into WY2016 from previous water years. First, cover of *Ruppia maritima* still did not rebound from the dramatic declines observed following record high cover in WY2011 and WY2012. In fact, *R. maritima* cover during the past two water years (2015 and 2016) were at their lowest level since monitoring began in 1996. In Taylor River, these decreases were tracked by the cover of other saltwater intolerant species such as *Chara hornemanni*. However, along the eastern transects of the monitoring program (Joe Bay and Highway Creek), cover of *C. hornemanni* remained relatively stable and even increased at some sites. Second, cover of *Thalassia testudinum* continues to increase at the downstream end of the many monitoring transects. Increases were particularly evident at Taylor River and Highway Creek. Average cover at Taylor River equaled the record high for this transect set in WY2015 (20 percent).

During summer 2015, within central and western Florida Bay, the elevated salinities and above average temperatures initiated a seagrass die-off reminiscent of the die-off event of the late-1980s (Hall and Durako 2016). The May 2015 sampling event found normal conditions within Florida Bay, but a reconnaissance trip in late August 2015 revealed large areas of dead seagrass within Garfield Bight, Rankin Lake, and Johnson Key Basin (**Figure 6-28**). The fixed sampling locations within Garfield Bight and Rankin Lake

showed a large decrease in *T. testudinum* cover in October 2015 (**Figure 6-29**). An extra mapping effort was undertaken in October 2015 with the FHAP team revisiting the May 2015 random sampling locations in Rankin Lake, Whipray Basin, Johnson Key Basin, and Rabbit Key Basin. These data were not yet available for the writing of this report. In addition, aerial photographs of the die-off area were acquired through a partnership between FWC, ENP, and the District in April 2016. These photos are being processed and analyzed by FWC.

Figure 6-28. Path for August 2015 reconnaissance trip in central and western Florida Bay after reports of seagrass die-off. Green circles represent areas where divers did not note any apparent die-off and red circles represent areas where die-off patches of *Thalassia testudinum* were noted.

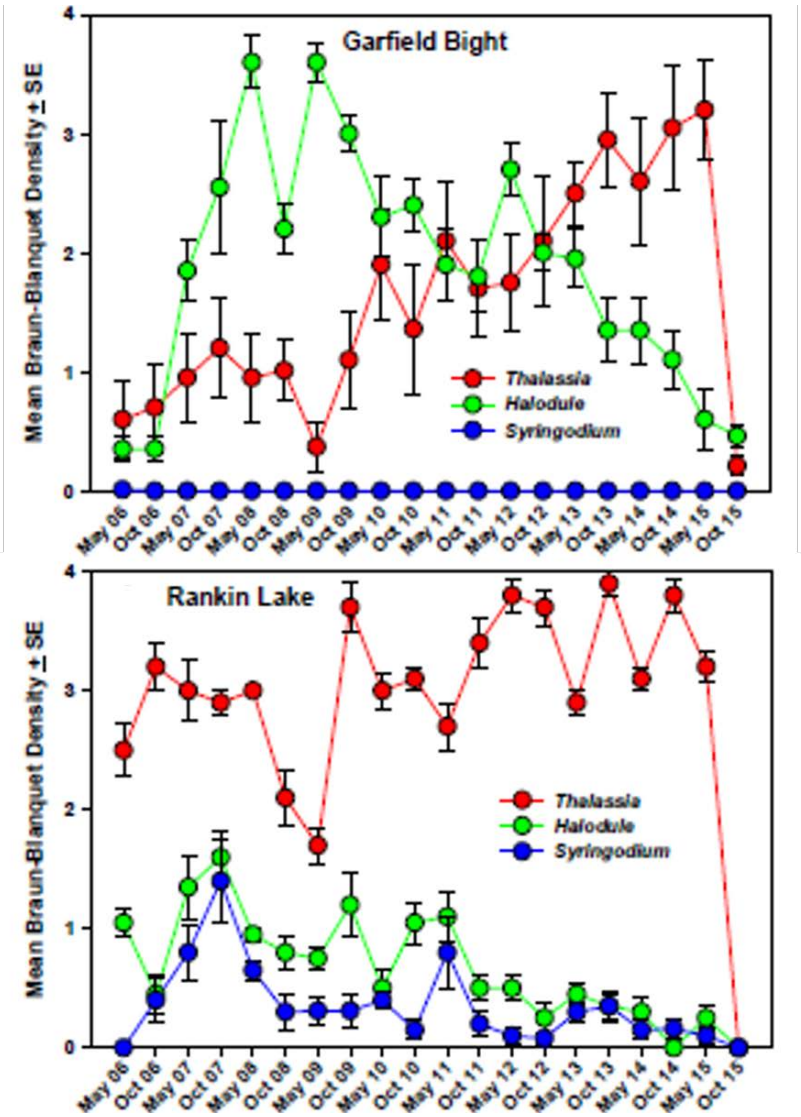


Figure 6-29. Mean Braun-Blanquet Density data (± standard error) for the 3 main species of seagrass in central Florida Bay (*Thalassia testudinum*, *Halodule wrightii*, and *Syringodium filiforme*) at the fixed sampling locations within Garfield Bight and Rankin Lake. Both areas are within the region of the recent (2015) seagrass die-off and experienced a significant decrease in *T. testudinum* (the red line). Excerpted from Hall and Durako (2016).

Relevance to Water Management

SAV dynamics in the transition zone were similar to previous water years and the continued low *Ruppia maritima* cover was expected as a result of the high salinities and the violation of the Florida Bay MFL criterion. Within Florida Bay, the seagrass die-off will have multi-year impacts as the community attempts to recover. Future monitoring will be essential to detect and assess any continuing and cascading effects of the drought and the record high salinities of WY2016 on the benthic community and water quality.

FLORIDA BAY PROJECT: 2015–2016 SAV INDICATORS FOR FLORIDA BAY

Christopher Madden and Amanda McDonald

Introduction and Methods

Status indicators for SAV are calculated each year and color-coded relative to a desired reference condition to summarize the status and trends of benthic vegetation in Florida Bay. Overall the SAV indicators showed a decline for some areas in WY2015 and WY2016 relative to prior years, likely the result of low rainfall and low freshwater input to the bay beginning in 2014. A severe drought in summer 2016 further reduced the indicator status for WY2017 as SAV die-off occurred in substantial portions of the central and western bay.

The indicators for SAV combine an Abundance Index based on underlying measures of areal extent and density, and a Species Index based on measures of species diversity and the presence of desired species. These values are calculated on a basin scale and spatially averaged across representative zones within the bay: Northeast, Transition, Central, Southern, and Western. The Abundance and Species indices are combined to generate an overall Composite score per zone.

Results

Abundance

The Abundance Index remained in the good (green) range for the Northeast Zone during WY2015, WY2016, and WY2017 (**Table 6-7**). The underlying spatial extent component of this score was in the good range for all basins in the Northeast Zone. However, due to deterioration in the underlying density score for several basins, there were notable declines in Index A scores at the basin level: Duck Key declined to fair (yellow) in WY2015 then rebounded in WY2016–WY2017; Barnes Sound declined to fair in WY2016 and WY2017. Eagle Key Basin declined to fair in all three years. For the Transition Zone, the Abundance Index continued fair in WY2015–WY2017 after declining from good a decade ago. All basins had spatial extent scores of good except Highway Creek, which declined dramatically from good to poor (red) in WY2016, rebounding to fair in WY2017. The density component remained at fair for most basins and good for Little Madeira Bay.

The Abundance Index in the Central Zone remained in the fair range for WY2015–WY2017, since it improved from poor in WY2008. Spatial extent was good in all basins until WY2017 when die-off affected Rankin Lake, depleting SAV abundance. Density declined to poor in Rankin, remained poor in Madeira, and was fair in most other basins, reducing the overall Abundance Index score for the Central Zone to fair. The Southern Zone continued to reflect a poor rating in the Abundance Index in WY2015–WY2017. Despite high scores for the underlying spatial extent metric in all southern basins, Abundance Index scores were reduced by underlying density component scores in Twin (poor) and Crane Key (fair). The Western Zone Abundance Index A improved to good in WY2016 after three years in the fair range but declined again to fair in WY2017 due to die-off in Johnson reducing the extent and density scores to fair in WY2017.

Species

The aggregate Species Index in the Northeast Zone fell to fair for WY2015–WY2017 (**Table 6-7**), with scores of fair (Barnes, Blackwater, and little Blackwater) or poor (Duck, Manatee, and Eagle Key) in most basins for the underlying species dominance score, meaning that mixed communities are not well-established. The underlying target species score averaged in the low good range for the Northeast overall with fair scores for Blackwater, Manatee, Duck, and Eagle Key and good for Little Blackwater and Barnes

Sound. The Species Index in the Transition Zone also declined to fair for WY2015–WY2017 as salinity conditions deteriorated in this region in recent years. The underlying species dominance component was almost universally poor in all zones from WY2015 through WY2017. The target species component for the zone remained at good although Long Sound dropped to fair, and Davis Cove to poor while Alligator improved to fair. Joe Bay and Little Madeira remained good for all years.

Table 6-7. Florida Bay SAV Indicator Status for WY2013-WY2017.

Performance Measure	Water Year				
	2013	2014	2015	2016	2017
Northeast Zone					
Abundance	🟢	🟢	🟢	🟢	🟢
Target Species	🟢	🟢	🟡	🟡	🟡
Transition Zone					
Abundance	🟡	🟡	🟡	🟡	🟡
Target Species	🟡	🟢	🟡	🟡	🟡
Central Zone					
Abundance	🟡	🟡	🟡	🟡	🟡
Target Species	🟢	🟢	🟡	🟡	🟡
Southern Zone					
Abundance	🔴	🔴	🔴	🔴	🔴
Target Species	🟡	🟡	🟡	🟡	🟡
Western Zone					
Abundance	🟡	🟡	🟡	🟢	🟡
Target Species	🟢	🟢	🟢	🟢	🟢

Species Index fell to fair for the Central Zone in WY2015 through WY2016 then improved to good in WY2017 possibly as a result of die-off reducing the presence of *Thalassia* relative to *Halodule* and the target species component remained good for all years, showing evidence of a mixed community even in some basins affected by die-off. Index B remained in the fair range in the Southern Zone since WY2009 after several years in the poor range. The species dominance component fell to poor in WY2016 and WY2017 due to a buildup of *Thalassia* in Twin Key and despite improvement to fair for Crane Key in both years. The target species component fell to poor in the Southern zone in WY2016 and WY2017.

Overall

The Composite Index combines both the Abundance Index and Species Index scores and gives a color-coded summary of overall status for SAV in each zone (**Figure 6-31**). The Composite Index shows that relative to WY2013-WY2014 (**Figure 6-31A**), the overall status of SAV in WY2015–WY2017 declined in the Transition, Central Bay, and Western zones (**Figure 6-31B**) as SAV was impacted by poor conditions in WY2015–WY2016 and die-off in WY2016–WY2017. The affected zones are the most sensitive of the entire bay to low freshwater input and elevated salinity.

Relevance to Water Management

The gains in the quality of SAV habitat over the past several years are precarious and can be reversed within an annual timescale. The steady rebound of the SAV community from losses due to a severe algal bloom in the eastern bay in WY2005–WY2008 and a seagrass die-off in WY1987 were reflected in improving scores in the late 2000s and early 2010s. The relatively wet years of WY2012 and WY2013 resulted in lower salinities, favorable conditions, and further improved SAV status in Florida Bay. The dry years that followed in WY2014 and WY2015 are characterized here by a decline in SAV status indicators. The extreme drought of WY2015–WY2016 will continued this negative trend with severe die-off the result. Water management initiatives such as the C-111 Spreader Canal Western Project and the Florida Bay Restoration Project are designed to reduce the impacts of high salinity by retaining more water in Taylor Slough and supplying more fresh water to central Florida Bay.

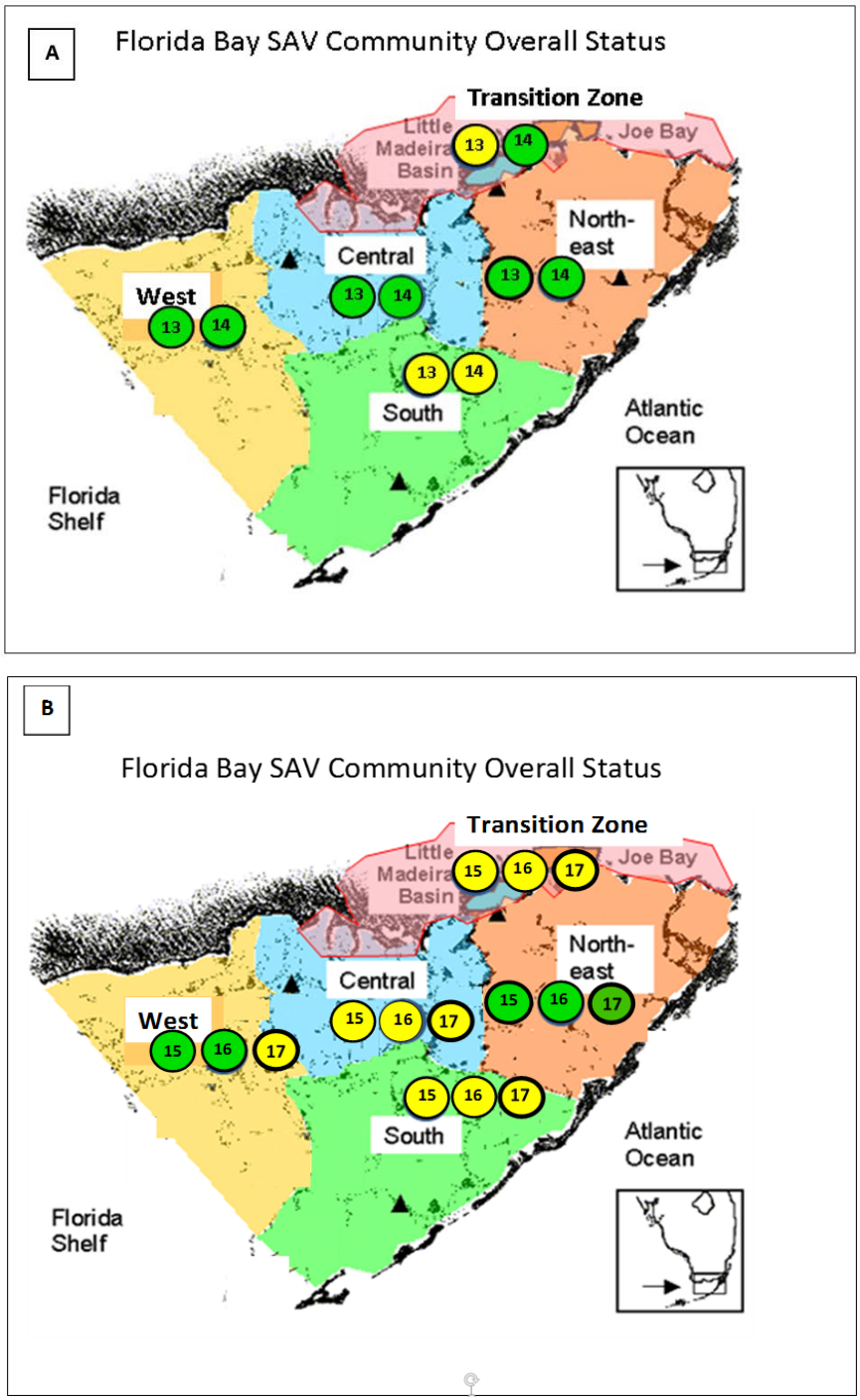


Figure 6-31. SAV status indicators during (A) WY2013–WY2014 and (B) WY2015–WY2017. The overall indicator for each year combines Abundance and Species indices into a Composite Index represented by fair (yellow) or good (green) status. Indicators declined during WY2015–WY2017, at least initially due to higher salinity, and then the occurrence of die-off in the Central and Western zones.

TREE ISLAND STABILITY AND RESILIENCE ASSESSED THROUGH INTEGRATED SEDIMENT ELEVATION TABLES

Carlos Coronado-Molina, Fabiola Santamaria[8], and Michelle Blaha[8]

Restoration of degraded tree islands and protection of intact tree islands are among the goals for restoring the Everglades ridge and slough ecosystem (Sklar et al. 2005). Current restoration plans predict dramatic changes in depth patterns over portions of the ridge and slough ecosystem, including a large number of tree islands (Sklar and van der Valk 2002). Information about soil elevation change and topographic differences across the tree island-ridge and slough ecosystem are needed to estimate the effects of proposed hydrologic changes on tree island ecological function. Predicting the effects of changes in water depth and hydroperiod on soil elevation change on tree islands cannot be accomplished until the spatial and temporal patterns of soil formation on tree islands are better known. However, extreme events such as droughts can lead to oxidation of peat material and allow muck fires, both of which reduce soil surface elevation in wetlands. In contrast, long hydroperiods can lead to low belowground production, a process that also promotes reduction in soil elevation. Thus, knowing the relationship between the soil elevation change and soil accretion is critical for planning hydrological restoration within the tree island ridge and slough landscape of the central Everglades.

The main objective of this study is to test hypotheses regarding abiotic and biotic controls on wetland sediment surface elevation among three plant communities, including the head, near tail and tree island/marsh transition. Specific tree island hypotheses include (1) wetland sediment surface elevation is more variable in the tree island/marsh transition than on the head and near tail communities; (2) surface water fluctuation is a critical driver of sediment elevation in upstream freshwater wetlands; and (3) disturbance events, including fire, drought, and extended period of inundation, cause loss of soil surface elevation, which may continue for decades.

Study Site

In 2009, soil elevation and accretion monitoring was initiated in hydrologically contrasting tree islands, including head, ridge/transition, and near tail communities, located in WCA-3, where three tree islands were chosen to study the effect of changes in hydrology on soil elevation (**Figure 6-32**). Two tree islands located in WCA-3A and one tree island located in WCA-3B were chosen for this study because they provide contrasting flood regimes, from short to long hydroperiods and contrasting soil properties from nutrient-rich soils to nutrient-poor soils. Tree Island 3AS2, a tropical hammock island that experiences a short hydroperiod (about one month of inundation per year) with a defined elevated area at its northern end, is dominated by cocoplum (*Chrysobalanus icaco*) on the head and by sweet bay (*Magnolia virginiana*) on the near tail and tail. In contrast, Tree Island 3AS5 is a flooded island with an extended hydroperiod (> 9 months inundation per year) at both the head and near tail and is dominated by adult individuals of willow (*Salix caroliniana*) throughout the entire tree island. Tree Island 3BS2 is an island with a long hydroperiod with 6 to 9 months of inundation per year at both head and near tail. This island is dominated by pond apple (*Annona glabra*) and willow (*S. caroliniana*).

[8] Scheda Ecological Associates, Inc., West Palm Beach, FL.

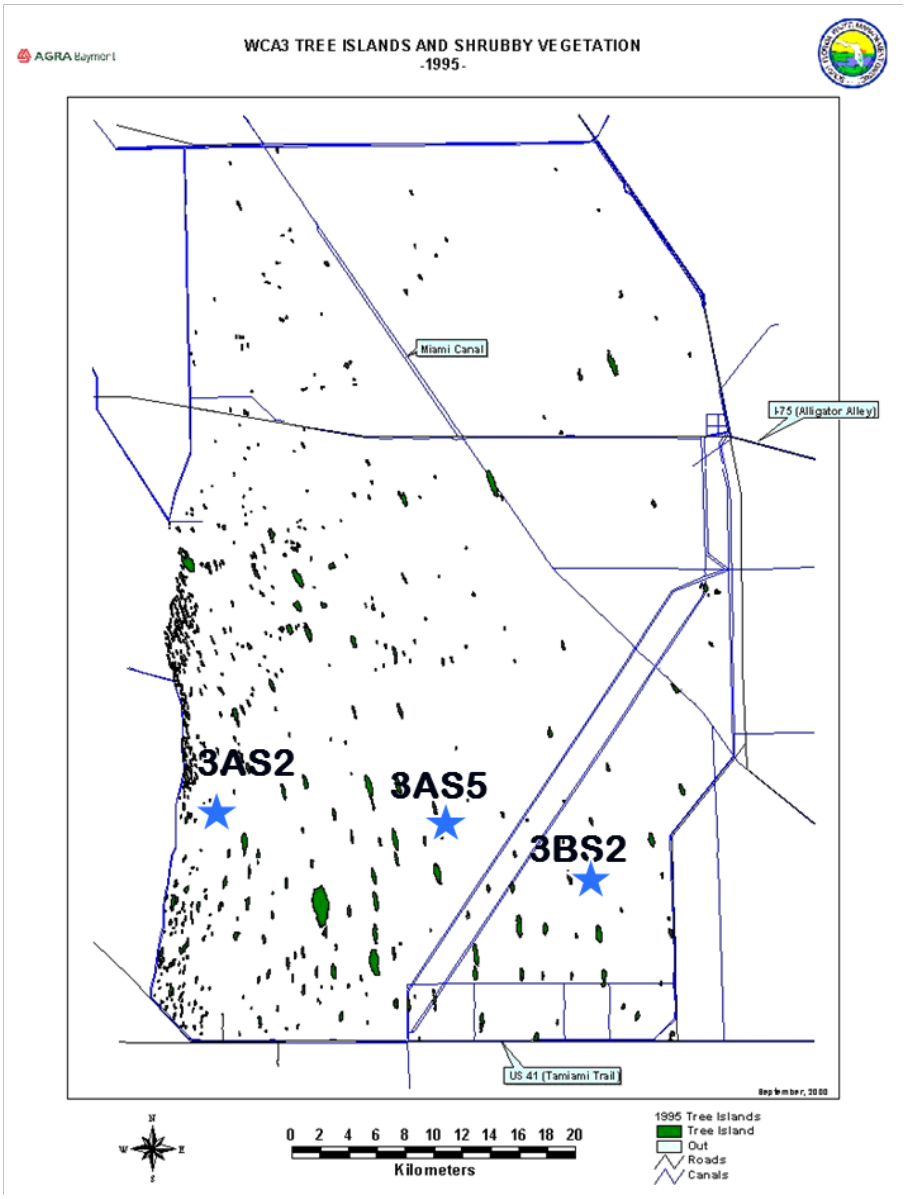


Figure 6-32. Study sites in WCA3 include islands 3AS2, 3AS5 and 3BS2, which represent examples of tree islands with contrasting hydrology

Field Methods

Monitoring soil accretion and elevation changes consists of employing standard methods that involve sediment erosion tables (SETs; Cahoon et al. 2002a, 2003, Whelan et al. 2007) and feldspar markers to assess changes in current elevations, accretion, and shallow subsidence. For this study, shallow rod SETs (0.40 m) were installed, to account for processes that occur within the root zone (Cahoon et al. 2002b). When used together, shallow SETs and feldspar marker horizons provide information on belowground processes that also influence sediment elevation within the root zone. Marker horizons and shallow rod SETs were established starting in 2009 to measure changes in the surface elevation of the wetlands over time. The portable portion of the SET consists of a vertical arm, a horizontal arm, a flat plate, and nine pins. During sampling, the portable SET is inserted into the support pipe in the desired direction and leveled. The pins are lowered through the table until they contact the sediment surface and then fastened so they

cannot move. Once the pins are placed, the distance from the top of the table to the top of each pin is measured in millimeters.

Results and Discussion

Accretion, the build-up of material on the forest floor, is largely determined by the hydrologic and geomorphic conditions of the area under study. In this sense, the rate at which accretion occurs is a function of the combination of inputs of both inorganic and organic material to the soil. Elevation change, which is determined relative to a subsurface datum, is affected by accretion and subsurface processes occurring below the marker horizon. Thus, the continued existence of tree islands largely depends on the ability to accrete material such that vertical elevation gain is equal or greater than the surrounding marsh. This study presents the results of soil accretion and elevation change measured on tree islands located in WCA-3A and WCA-3B. Over the six years of the measurement period, average vertical accretion rate within the three plants community at Tree Island 3AS2 was 2.5 ± 1.12, 3.8 ± 1.21, and 4.6 ± 1.14 mm/yr at the head, near tail, and transition marsh sites, respectively (**Figure 6-33**). Similarly, average vertical accretion rate within the three plants community at Tree Island 3AS5 was 6.1 ± 3.3, 3.8 ± 1.7, and 4.4 ± 1.14 mm/yr at the head, near tail, and transition marsh sites, respectively (**Figure 6-34**). At Tree Island 3BS2, average vertical accretion was 3.9 ± 1.2, 5.9 ± 2.2, and 3.1 ± 1.14 mm/yr at the head, near tail, and transition marsh sites, respectively (**Figure 6-35**).

Soil elevation change within the three plants community at Tree Island 3AS2 was -1.7 ± 0.52, 3.7 ± 1.3, and -0.86 ± 0.54 mm/yr at the head, near tail, and transition marsh sites, respectively (**Figure 6-33**). Similarly, average soil elevation change within the three plants community at Tree Island 3AS5 was -0.1 ± 0.1, -2.8 ± 1.2, and -3.7 ± 1.14 mm/yr at the head, near tail, and transition marsh sites, respectively (**Figure 6-34**). At Tree Island 3BS2, average elevation change was 2.1 ± 0.9, -1.7 ± 0.8, and -6.1 ± 3.4 mm/yr at the head, near tail, and transition marsh sites, respectively (**Figure 6-35**). These rates are similar to those reported by Erwin et al. (2006) who measured similar parameters in *Spartina* marshes along the Atlantic Coast.

Elevation change rates at the study sites have been very small and in most sites there has been a loss in soil elevation, particularly on tree islands where high water depth and long period of inundation characterized the plant communities, such as near tail and transition marsh sites. For instance, at 3AS5, a tree island located within the impounded region of 3AS3, all three plant communities showed a significant decrease in soil elevation despite the fact that soil accretion (depth above the feldspar marker horizon) increased over the study period. In contrast at tree island 3AS2, where hydrologic conditions are much drier than at 3AS5, elevation change was negligible particularly in the head zone, the driest region of the tree island. Similarly, extremely long hydroperiods (i.e., more than 10 months of continuous inundation) caused a loss of elevation with a high degree of shallow subsidence, 6.2 to 8.1 mm/yr, at 3BS2 and 3AS5, respectively. In contrast, tree island 3AS2 located within a drier region in 3AS3 showed lower soil elevation loss. This pattern suggests that elevation change and subsurface processes within the root zone, including decreased root production and/or more rapid root decomposition, are affected by extreme hydrology, either very wet or very dry.

Relevance to Water Management

Based on this preliminarily analysis, extreme hydrology, either high water levels along with extended days of inundation or very dry conditions, do not promote soil formation. Determining the optimum hydrologic conditions that promote elevation differences between tree islands and their surrounding marsh-slough environment is critical to provide a better understanding of how water management decisions can be made to maintain and restore the ecological integrity of the ridge and slough-tree island ecosystem in the central Everglades.

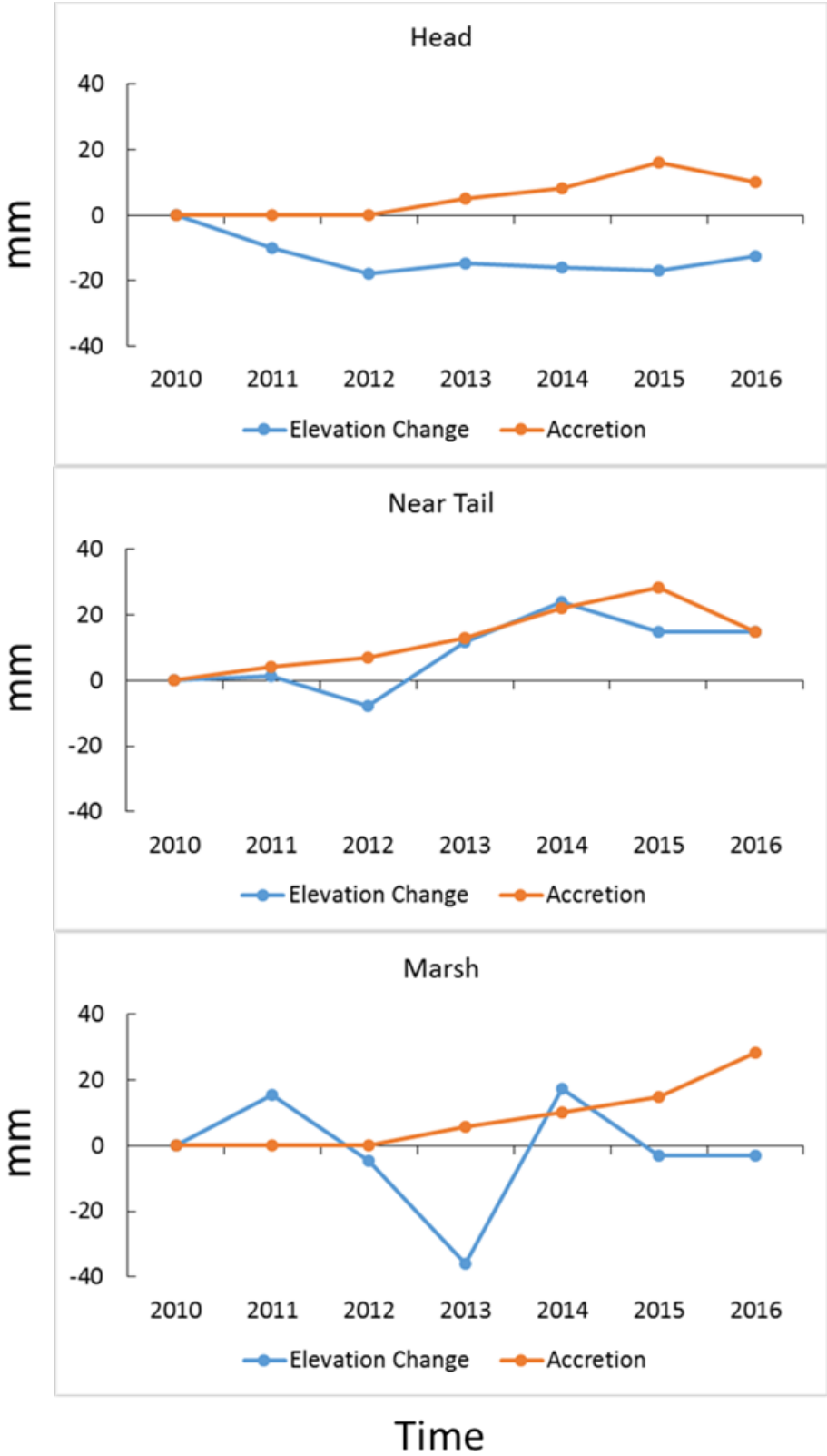


Figure 6-33. Root-zone soil elevation change and soil accretion at Tree Island 3AS2.

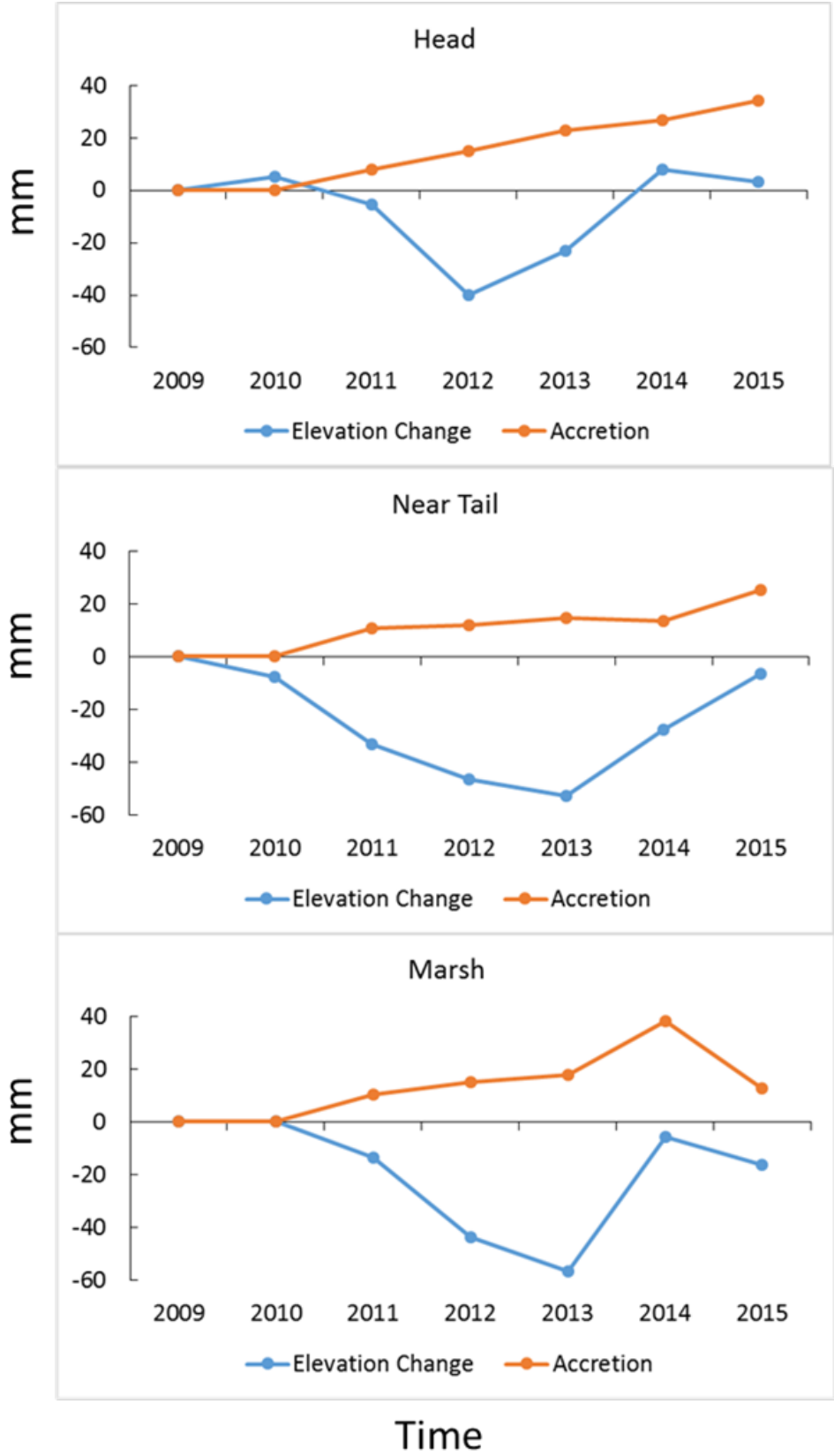


Figure 6-34. Root zone soil elevation change and soil accretion at Tree Island 3AS5.

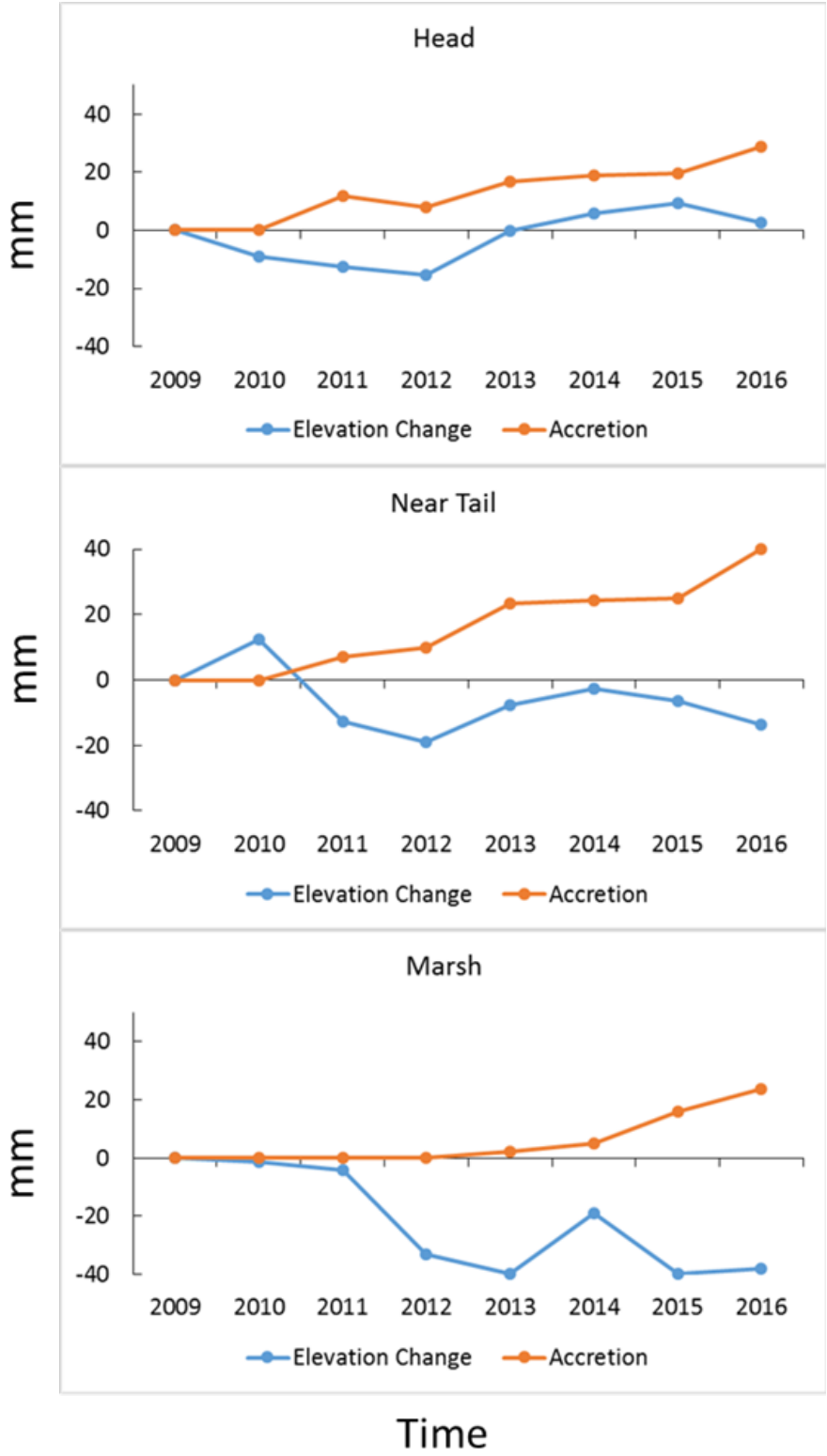


Figure 6-35. Root zone soil elevation change and soil accretion at Tree Island 3BS2.

LANDSCAPE ECOLOGY

This section examines: (1) the remote sensing technique being developed to track flow from the S-356 Pump Test, (2) the salinity measurements and salinity mapping results from the freshwater releases into the Florida Bay region, and (3) the year three results of the Decomp Physical Model (DPM).

S-356 PUMP TEST: ECOLOGICAL MONITORING OF SHARK RIVER SLOUGH

Christa Zweig and Sue Newman

Introduction

The S-356 Pump Test is an incremental approach USACE is undertaking to develop the final operating plan for the Modified Water Deliveries to ENP (MWD) and C-111 South Dade projects. The first increment of the test evaluates the raising of the G-3273 constraint of 6.8 feet NGVD29 and holding the L-29 canal stage at 7.5 ft (2.3 m) to enable increased water deliveries to ENP. The second increment will evaluate raising the L-29 canal stage constraint. The final increment will incorporate information learned from the first two increments and will involve modeling to develop the final operating plan. As a part of the ecological monitoring, SFWMD is responsible for the collection of remote sensing imagery, some field assessment data, and dye studies to determine water flow direction.

Dye Study

The SFWMD dye study was used to establish vegetation transect orientation and width by tracking flows from the L-29 canal into ENP at a canal stage of 7.5 ft (2.3 m). During Project Year 1 (CY2015–CY2016), a dye delivery protocol and concentration worksheet were developed, a contractor was hired, and four Fluorescein dye injections were performed along the L-29 canal (October 9, 2015, October 15, 2015, and two injections on November 13, 2015). Dye was injected according to the protocol and concentration worksheet to keep concentrations of the Fluorescein dye below 10 milligrams per liter (mg/L), as stated in the ENP Wilderness permit. The dye was injected into the downstream pool of culverts into ENP from the L-29 canal (**Figure 6-36**).

Figure 6-36. Photo of injection of Fluorescein dye into culverts along the L-29 canal.

The area was flown with the District helicopter and remote sensing equipment (box-on-aircraft or BOA) the day after injection and the BOA collected photography and global positioning satellite (GPS) points of all dye locations (**Figure 6-37**). This provided a relative water velocity and direction of flow along the northeastern section of ENP. The information was provided to ENP, Florida International University, and SFWMD within a week of collection.

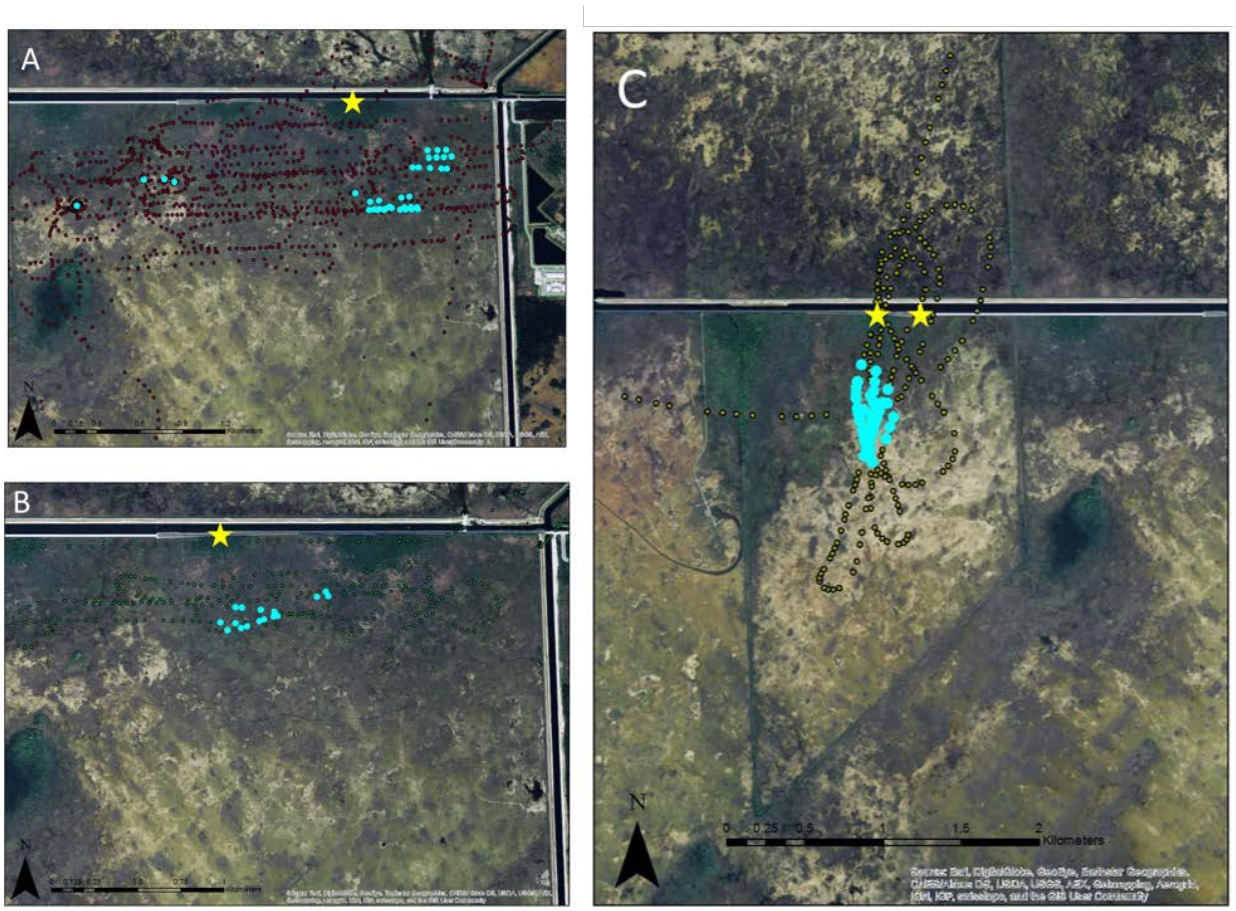


Figure 6-37. Locations of dye in ENP after one day. (A) October 9, 2015, dye drop; photo includes all photo sampling points (red dots), points where dye was detected (blue dots), and dye injection site (yellow star). (B) October 15, 2015, dye drop; photo includes all photo sampling points (red dots), points where dye was detected (blue dots), and dye injection site (yellow star). (C) November 13, 2015, dye drop; photo includes all photo sampling points (yellow dots), points where dye was detected (blue dots), and dye injection site (yellow star).

High Resolution Aerial Mapping for Vegetation Community Composition Record and Change Analysis: Proof of Concept

The BOA used for this proof of concept project was specifically designed for use on the SFWMD 407 helicopter. The BOA has two Cannon EOS REBEL SL1 cameras with Electro-Focus 40-mm 1:2.8 Stepper Motor lenses, one for visible spectrum and the other for near-infrared use. Each cameras has an associated GPS/inertial measurement unit (IMU) unit recording all positioning data required for image processing, which is then logged onto Linux onboard computers. After each flight missions all photographs and GPS/IMU data are downloaded from the onboard computers via an external hard drive and then uploaded onto a SFWMD server for processing and storage.

Trials throughout the proof of concept phase led to important findings that have made use of the BOA more dependable and efficient. The vegetation transects will be monitored twice over the five-year period for changes in vegetation. The data gathered from this remote sensing effort will be used in investigating the effects of the MWD on vegetation communities in this area of ENP. SFWMD has completed a proof of concept stage, which has resulted in high quality aerial imagery taken from the BOA technology.

BOA Lessons Learned

Various BOA maintenance tasks are essential for correct preparation for any flight missions. With these lessons learned during the proof of concept phase, it is anticipated that all flight missions moving forward will be relatively free of error.

- The BOA requires routine maintenance before and after data acquisition flights:
  - The longer the GPS/IMU units have to acquire the appropriate satellites the better, but it is recommended that the unit be turned on for as long as possible the day before the data acquisition flight takes place.
  - After each flight, it is essential to check to ensure that the data log, which records each individual photos geographic and spatial location, is correctly labeled.
- For the day of the data acquisition flight:
  - Make a decision on weather conditions.
  - Schedule back up days.
  - All wires must be traced from source to source before start up to avoid any issues with the system.
  - Full equipment check.

Post-processing and Flight Planning Lessons Learned

The aerial photographs taken by the BOA are processed using Agisoft Photoscan Pro to mosaic the photographic data of interest into one georeferenced tagged image file (TIF). Flight plans must allow for complete coverage to create a successful mosaic. Initial flight missions during the proof of concept phase did not result in completed coverage. An extensive resource review and consultation with Agisoft software support led to the decision that increased photo overlap and orientation of photographs was needed to ensure complete coverage. Agisoft typically suggests 80 percent forward overlap and 60 percent side overlap, but cautions that complex areas may require even higher overlap for successful processing. The vegetation communities in question fall under the complex area caveat.

Post-processing is a multi-step process, which includes photo alignment, dense point cloud building, three-dimensional mesh creation, orthophotograph creation and, lastly, TIF export. When issues with photo alignment arise, there are options within each step of processing to improve the models created from the original data collected. One such example of improved coverage can be seen below (**Figure 6-38**). These were photos taken with insufficient photo overlap to process using the traditional Agisoft settings but additional coverage was gained using various techniques. This proof of concept has allowed us to refine our techniques and caused us to redesign the flight paths and areas that can be monitored for vegetation change.

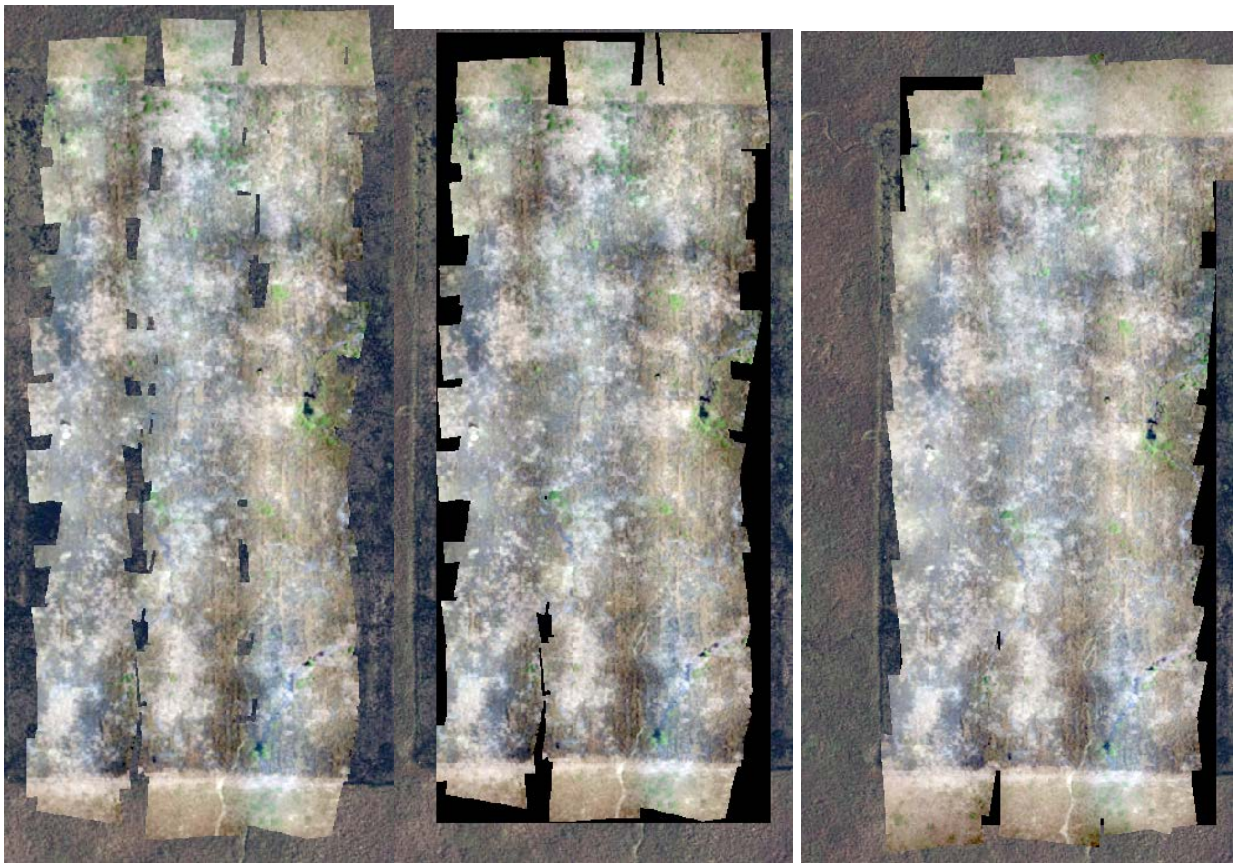


Figure 6-38. The first photo is the original after the initial attempt at mosaicking indicating insufficient photo overlap. The next two photos use different overlap techniques to fill in holes in the mosaic.

Relevance to Water Management

The dye study can be used to confirm hydrologic models and budgets of ENP and how water will move with higher stages in the L-29 canal and with the S-356 pump running. The vegetation study will allow the determination of the effects of more water being routed through Shark River Slough and can be directly applied to decisions made related to Appendix A of the 1995 Everglades Settlement Agreement (Consent Decree, 847 F. Supp. 1567), which sets limits for total phosphorus concentrations entering ENP.

S-356 PUMP TEST: ECOLOGICAL MONITORING DOWNSTREAM OF THE S-197 STRUCTURE

Stephen Kelly, Christopher Madden, Joseph Stachelek, Fabiola Santamaria[9], Michelle Blaha[8], and Amanda McDonald

Introduction

As part of a cooperative agreement between ENP and SFWMD, benthic salinity and temperature monitoring probes were installed downstream of the S-197 structure to supplement the ability of the existing network to capture the eco-hydrologic response to an opening of S-197. Salinity and temperature readings were taken every 30 minutes by the probes and the data downloaded every three months. The probes were distributed along a salinity gradient from S-197 to the saline waters of Barnes Sound. An event-driven (S-197 opening) deployment of a high-resolution surface water quality mapping unit known as Dataflow was implemented when the S-197 was operated as part of the S-356 Pump Test and the structure remained open for more than three days. A spatial analysis (i.e., kriging) of the Dataflow output was used to estimate freshwater plume dynamics in Barnes Sound and evaluate the potential for ecological impacts, both positive and negative.

Probe Deployment

Three salinity and temperature probes (YSI 600 Multi-Parameter Water Quality Sondes) were deployed in Manatee Bay on December 16, 2015. Locations of these deployments as well as the District's permanently deployed station in Manatee Bay (MBTS) are shown in **Table 6-7** and **Figure 6-39**. Site Man 1 is closest to the mouth of the C-111 canal, Man 2 is southeast of Man 1 and Man 3 is south of Man 2, furthest from the mouth of the C-111 canal, and closest to the entrance to Barnes Sound. Probes were calibrated prior to deployment and set to log every 15 to 30 minutes. Deployment was accomplished by attaching the sonde to a cinderblock with tie-wraps, and slowly lowering it to the bottom with a line attached to a float, used to relocate them during subsequent visits.

At the time of deployment, and at all subsequent deployments, surface and bottom salinity and temperature measurements were made. This information was used to determine if there is any stratification in the water column, as well as providing an independent check of the probe calibration. This is especially important after the probes have been deployed for an extended period, in order to check for calibration drift. This information is used to verify that the probes are functioning properly.

In addition to the three sondes deployed on the bottom, an additional sonde (MBTS) was permanently deployed at the mouth of Manatee Bay (**Figure 6-39**). The MBTS sonde is deployed at 0.5-m depth and thus represents the surface water conditions.

Table 6-7. Locations of the sondes deployed in Manatee Bay.

Site	Latitude	Longitude
Man 1	25.25665 °N	80.42611 °W
Man 2	25.25048 °N	80.42626 °W
Man 3	25.24038 °N	80.42807 °W

[9] Scheda Ecological Associates, Inc., West Palm Beach, FL.

Figure 6-39. Location of sondes in Manatee Bay and the District's permanent site (MBTS).

Dataflow Mapping

In addition to sonde deployment, if S-197 is operated as part of the pump text and the structure remains open for more than 3 days, a Dataflow mapping trip in Manatee Bay and Barnes Sound was conducted. The Dataflow system is a portable, self-contained flow-through instrument containing sensors for continuous sample acquisition and high speed real-time in vivo measurement of a suite of water quality parameters. The Dataflow system is mounted on a small boat that acts as a sampling platform. While underway, the system captures bay water via a transom-mounted ram. This water is passed through a debubbler before entering the Dataflow sensor suite (the C-6), and ultimately returning into the bay. A digital data logging device records data values while simultaneously recording the GPS location such that these water quality parameters can be spatially mapped and subject to spatial pattern analysis. Five mapping trips were conducted during this reporting period: December 1, 2015, December 16, 2015, January 12, 2016, March 8, 2016, and April 5, 2016.

S-197 Openings

The gates at S-197 were open multiple times during December 2015 through April 2016 (**Table 6-8**, **Figure 6-40**). However the first opening on December 4, 2015 was not part of the S-356 Pump Test. This opening was in response to a rain event to prevent flooding of the agricultural and residential areas upstream

of the C-111 canal. All subsequent openings have been part of the pump test, and sondes have been deployed continuously since December 16, 2015.

Table 6-8. Dates, durations, and discharges from the S-197 structure during the period December 2015–April 2016. (Note: cfs – cubic feet per second.)

Date Open	Date Close	Duration (days)	Maximum Discharge (cfs)	Total Discharge (cfs)
12/4/2015	12/19/2015	15	2,025	3,096,470
1/28/2016	2/10/2016	13	111	200,218
2/23/2016	3/3/2016	9	221	237,869
3/17/2016	3/28/2016	11	265	313,789
3/31/2016	4/12/2016	12	58	89,238
4/16/2016	4/21/2016	5	188	97,775

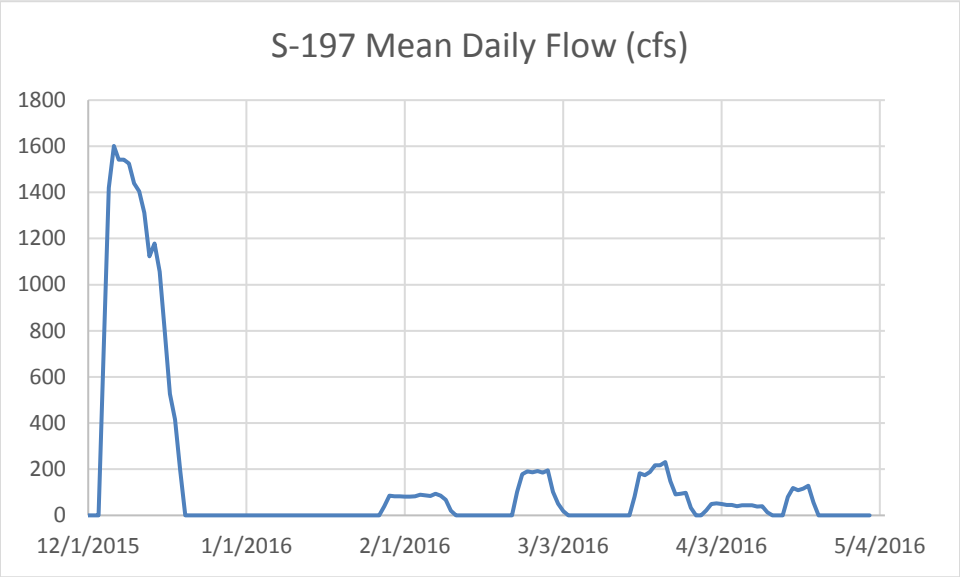


Figure 6-40. Mean daily discharge from the S-197 structure in cubic feet per second (cfs).

Results

Instantaneous salinity and temperature from the MBTS site is shown in **Figures 6-41** and **6-42**. The boxes in this and all subsequent figures represent the times when S-197 was open. The red box indicates a non-pump test opening while the green boxes are associated with the pump test. The large releases in December 2015 can be clearly seen in the salinity figure, and even the subsequent smaller releases are lowering the salinity in the surface waters.

The effects of water releases on the temperature signal is difficult to interpret as the seasonal signal cannot be teased apart from the releases. However, the smaller releases associated with the pump test all exhibited a drop in temperatures during the opening (**Figure 6-42**).

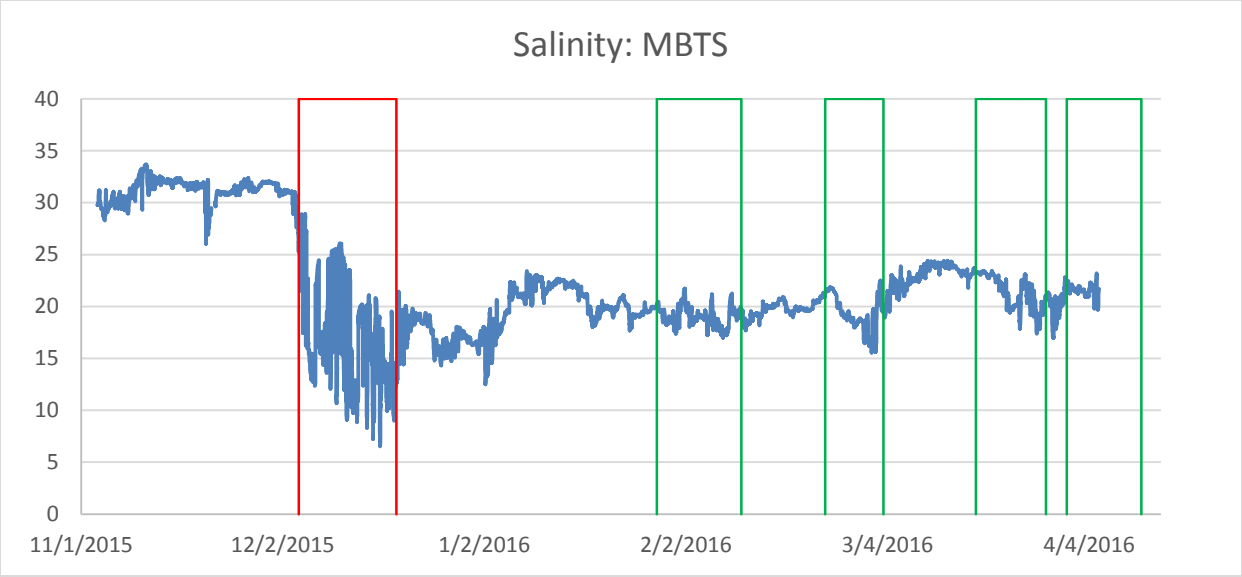


Figure 6-41. Instantaneous salinity of the surface water from the MBTS site.

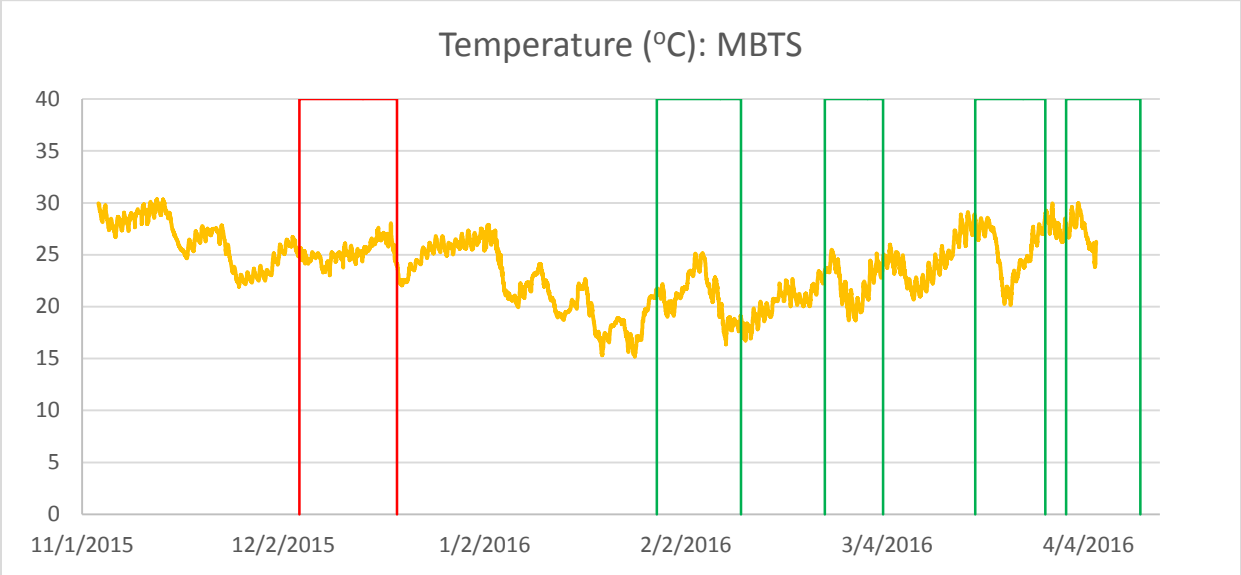


Figure 6-42. Instantaneous temperature in the surface water from the MBTS site.

Surface and bottom water measurements of salinity (**Table 6-9**) showed stratification in December 2015 at two of three sites (Man 3 not measured) and again at Man 1 in January 2016. The impacts of fresh water on the benthos may be mitigated somewhat by stratification, which allows higher salinity water on the bottom to buffer the osmotic shock on biota as the water masses gradually mix, leading to a mesohaline regime. Freshwater discharges associated with opening of S-197 are having the effect of lowering the salinity in the bottom waters most times. Another effect, especially at the Man 1 sites, is an increase in the variability of the salinity. Man 1 is closest to the mouth of the C-111 canal and this could explain this increase in the variability (**Figures 6-43** and **6-44**).

Table 6-9. Temperature and salinity measurements made during the S-197 gate openings.

Site	Date	Temperature (ºC)		Salinity	
		Surface	Bottom	Surface	Bottom
Man 1	12/16/2015	27.08	27.76	3.79	5.92
Man 2	12/16/2015	26.60	27.66	4.53	21.76
Man 3	12/16/2015	NA[a]	NA	NA	NA
Man 1	1/13/2016	18.25	18.37	18.04	19.80
Man 2	1/13/2016	19.42	19.46	22.24	22.32
Man 3	1/13/2016	18.87	19.00	22.00	22.20
Man 1	3/8/2016	22.15	22.22	21.72	21.72
Man 2	3/8/2016	22.1	22.08	21.49	21.5
Man 3	3/8/2016	22.23	22.22	22.14	22.14

a. NA – not applicable.

Figure 6-43. Instantaneous salinity in the bottom waters at the three sonde sites in Manatee Bay. The boxes represent the times when the S-197 structure was open. The red box indicates a non-pump test opening while the green boxes are associated with the pump test.

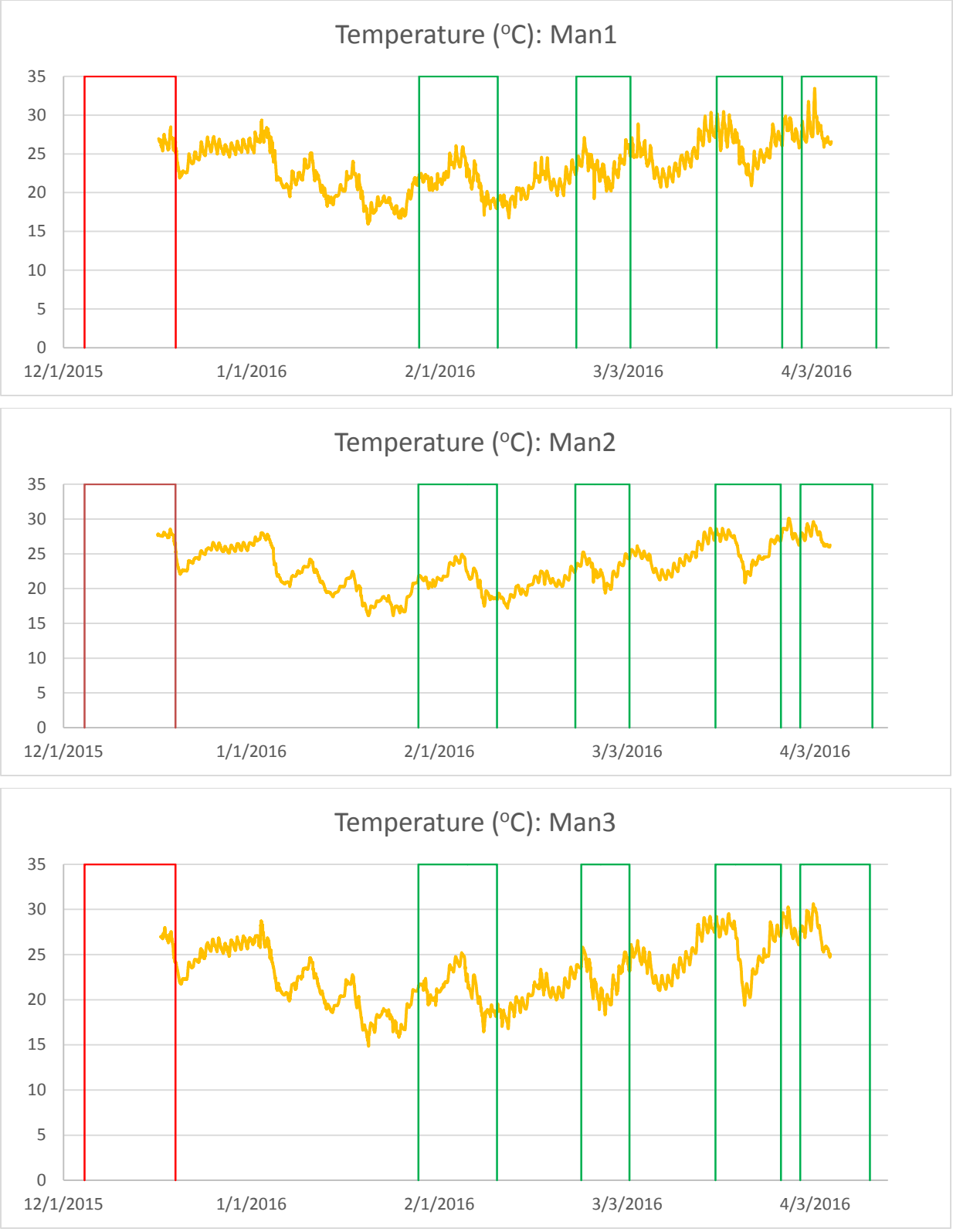


Figure 6-44. Instantaneous temperature in the bottom waters at the three sonde sites in Manatee Bay. The boxes represent the times when the S-197 structure was open. The red box indicates a non-pump test opening while the green boxes are associated with the pump test.

Surface water salinity mapping is shown in **Figure 6-45**. During the five water release events from the S-197 structure on the C-111 canal between December 2015 and March 31, 2016, salinity decreased downstream in both Manatee Bay and Barnes Sound. The horizontal impact of the releases extended into Barnes Sound reducing salinities throughout. The initial salinity level in both basins was 30 to 35 and declined to 20 to 25 during and following releases. The salinity map sequence (**Figure 6-45**) shows that, following an initial large release in early December, subsequent small releases did not appreciably reduce salinity further, though the cumulative effect was to maintain reduced salinity throughout the dry season to date. It is unknown how much the direct precipitation from greater than average rainfall during this El Niño season contributed to salinity reduction. Most recently, higher salinity water is reentering Barnes Sound from Card Sound.

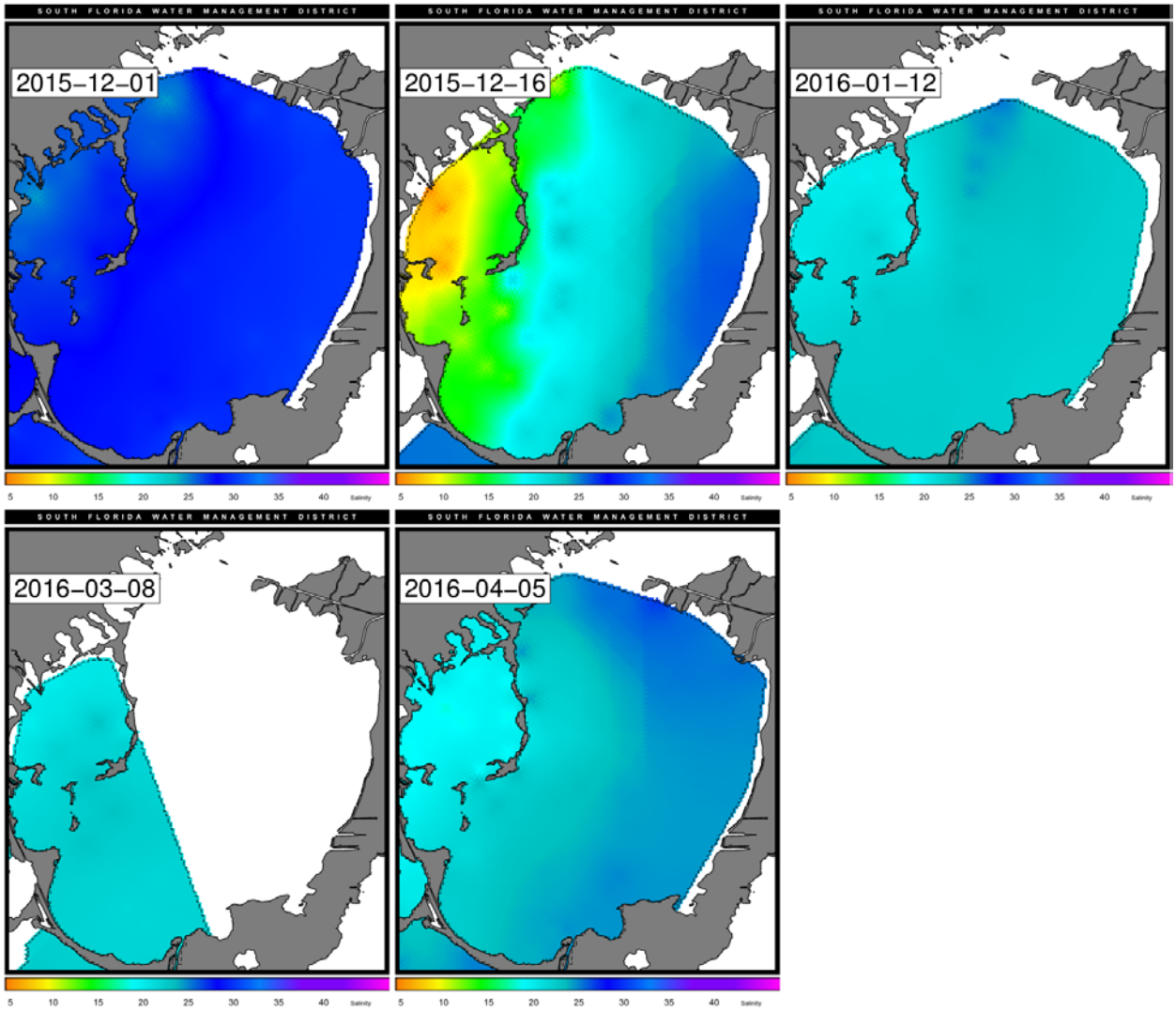


Figure 6-45. Salinity distribution in Manatee Bay and Barnes Sound.

Although no benthic assessments were conducted for this study, it can be reported from visual observations that bottom vegetation seems unaffected by the releases, with *Thalassia* and *Halodule* continuing to maintain moderate bottom cover. No evidence of fish kills or high Chla concentrations were observed and the waters in both basins remained mostly clear throughout the study period.

Relevance to Water Management

The S-197 gate openings provided an opportunity to characterize the effects of the freshwater pulses on Florida Bay. Temperature and salinity measured at specific points using sondes show changes over time, critical to evaluating long-term impacts. These parameters measured spatially using the Dataflow show the spatial extend of changes due to these pulses. These data are critical in predicting future changes from the MWD project.

EXPERIMENTAL EFFECTS OF SHEETFLOW ON SEDIMENT TRANSPORT, WATER QUALITY, AND PERIPHYTON DYNAMICS IN THE RIDGE AND SLOUGH – INTERIM FINDINGS OF THE DECOMP PHYSICAL MODEL

Colin Saunders[10]

Introduction and Background

Driven by the natural, gradual slope of water across the landscape, sheetflow was a characteristic feature of the pre-drainage Everglades. It is believed that historically velocities were sufficient to entrain and redistribute highly organic sediments from water lily sloughs into sawgrass ridges, building the characteristic topography and landscape patterning. The amount of flow needed to trigger such mechanisms, however, remains a critical uncertainty for the CERP projects that aim to reconnect and rehydrate the system. Reduced vegetative resistance and lower topographies of sloughs both generated and maintained higher flow velocities and erosive conditions, reinforcing and stabilizing the landscape. In over-drained areas like WCA-3B, however, the widespread expansion of sawgrass ridges may hinder our ability to restore the system by simply adding flow. Addressing this hydro-ecological uncertainty is a primary objective of the Decomp Physical Model (DPM), a landscape-scale, adaptive management experiment to quantify the benefits of sheetflow in restoring the ridge and slough landscape. Located in the "pocket" between the L-67A and L-67C canal/levee features (**Figure 6-46**), the project utilizes a culvert structure (S-152) to deliver experimental flows into the pocket to evaluate hydrologic, sediment, and biogeochemical responses to flow. Funding for this project is provided by USACE and SFWMD.

This year we report on results from three flow events in DPM studying the relationship between velocity, sediment transport and chemical properties of suspended sediments and benthic flocculent sediments (floc). Previous results from the first two flow events demonstrated that experimental sheetflow treatments increased water velocities and sediment transport in sloughs compared to ridges (see Sklar and Dreschel 2015, Saunders et al. 2016). Flow pulses, generated by initial opening of the S-152 structure, provided a 10-fold increase in suspended sediments and relatively higher velocities than sustained flows but lasted only a few hours. The first two flow events also suggested that despite relatively constant culvert discharges over 8 to 10 weeks, slough velocities and sediment transport tended to increase with sustained flow. Finally, preliminary evidence also suggested high flows were associated with changes in the algal species contributing to the organic matter in sediments and floc, particularly in sloughs.

This year, we present a new and growing body of evidence from DPM that sheetflow both impacts and is shaped by biogeochemical changes in slough periphyton and algal communities. To complement the physical response variables from previous flow events, this year a pilot study was initiated to characterize

[10] Specific investigations, including laboratory and field studies, data collection and analysis, and writing were contributed by Christopher Hansen, Carlos Coronado-Molina, Michael Manna, Susan Newman, Fred Sklar, Erik Tate-Boldt, and Christa Zweig of SFWMD; Laurel Larsen from the University of California, Berkeley; Jay Choi, Mark Dickman, Judson Harvey, and Barry Rosen of USGS; Rudolf Jaffé and Peter Regier of Florida International University; and Michelle Blaha and Fabiola Santamaria of Scheda Ecological Associates, Inc.

periphyton productivity, biomass, and species responses to flow. We also hypothesized that accelerated flows increase local TP loads, attenuating with distance from inflow. Because the native periphyton community is P limited, we hypothesize that increased loads will increase periphyton biomass and productivity due to their sensitivity to P enrichment (McCormick et al. 1996, 2002, Gaiser et al. 2004, 2008, Hagerthey et al. 2011), and if their threshold P load is exceeded, there will be a loss of native periphyton species. Thus flow was expected to alter algal communities of periphyton. Additionally, during the third flow event, we focused high-resolution aerial imagery to capture relatively rapid, large-scale changes in the periphyton community that had been visually observed in previous years.

Study Design and Methods

The DPM study uses a before-after-control-impact (BACI) experimental design, with additional sites for spatial and synoptic surveys (**Figure 6-46**), to evaluate hydrological and ecological responses to flow. Ten gated culverts are used to move water from the L-67A canal into an area known as "the pocket" between the L-67A and L-67C canals. A 3,000-foot gap and three backfilling treatments were constructed in the L-67C canal/levee feature. Results of the latter portion of the study will be presented in forthcoming SFER reports. The culverts were designed to generate velocities of at least 2 centimeters per second (cm/s), the expected threshold velocity to entrain sediments. The study includes three baseline years and three flow events, which started in the fall 2013 (November–December, WY2014), 2014 (November–January, WY2015), and 2015 (November–January, WY2016). On November 16, 17, and 19, flow was started, stopped, and restarted, respectively, to test effects of short duration pulses. Further, in WY2016–WY2017, a high water emergency order required reopening the S-152 (February 19–23, 2016, and March 9–May 3, 2016). The latter operations provided opportunities to test dry season flow impacts and a larger flow window (17 weeks total).

Hydrology and Sediments

Prior to the first S-152 flow event in 2013, USGS installed instrumentation to compute continuous records of headwater and tail water stages at the S-152 to calibrate measured flows through S-152 (Mark Dickman, personnel communication, September 15, 2015). A Sontek Argonaut SW acoustic Doppler velocimeter (ADV) was deployed inside of the downstream end of culvert pipe number 2, as counted from the north end of the structure, and a second SW ADVM was installed inside of the downstream end of pipe number 6. To develop discharge ratings, 10 discharge measurements were made on the downstream end of each culvert using a Sontek FlowTracker according to the point velocity area method outlined in the Sontek FlowTracker Technical Manual (Sontek YSI Inc. 2007). The USGS also measured continuous water temperature and specific conductance on the headwater and tailwater sides of S-152, EDEN 8, and both sides of Site 69, for the purpose of determining the source of water flowing through the S-152 structure.

To quantify water flow at the eight sentinel sites in the pocket, water velocity was measured using ADV. At each site, flow speed and direction was measured at a fixed depth in the water column. The ADV approach measured flow velocity to a resolution of 0.01 cm/s with an accuracy of 1 percent of measured velocity (SonTek 2001). Operating procedures and data filtering followed those described in Harvey et al. (2009), Riscassi and Schaffranek (2002), and SonTek (2001). Water velocity measurements were also made at sites throughout the study area using a SonTek Handheld FlowTracker-ADV®. These provided additional information on the velocities at sediment trap locations (below) or locations selected to provide more information about spatial variation in flows. FlowTrackers were configured to measure velocities over a 120-second sampling period.

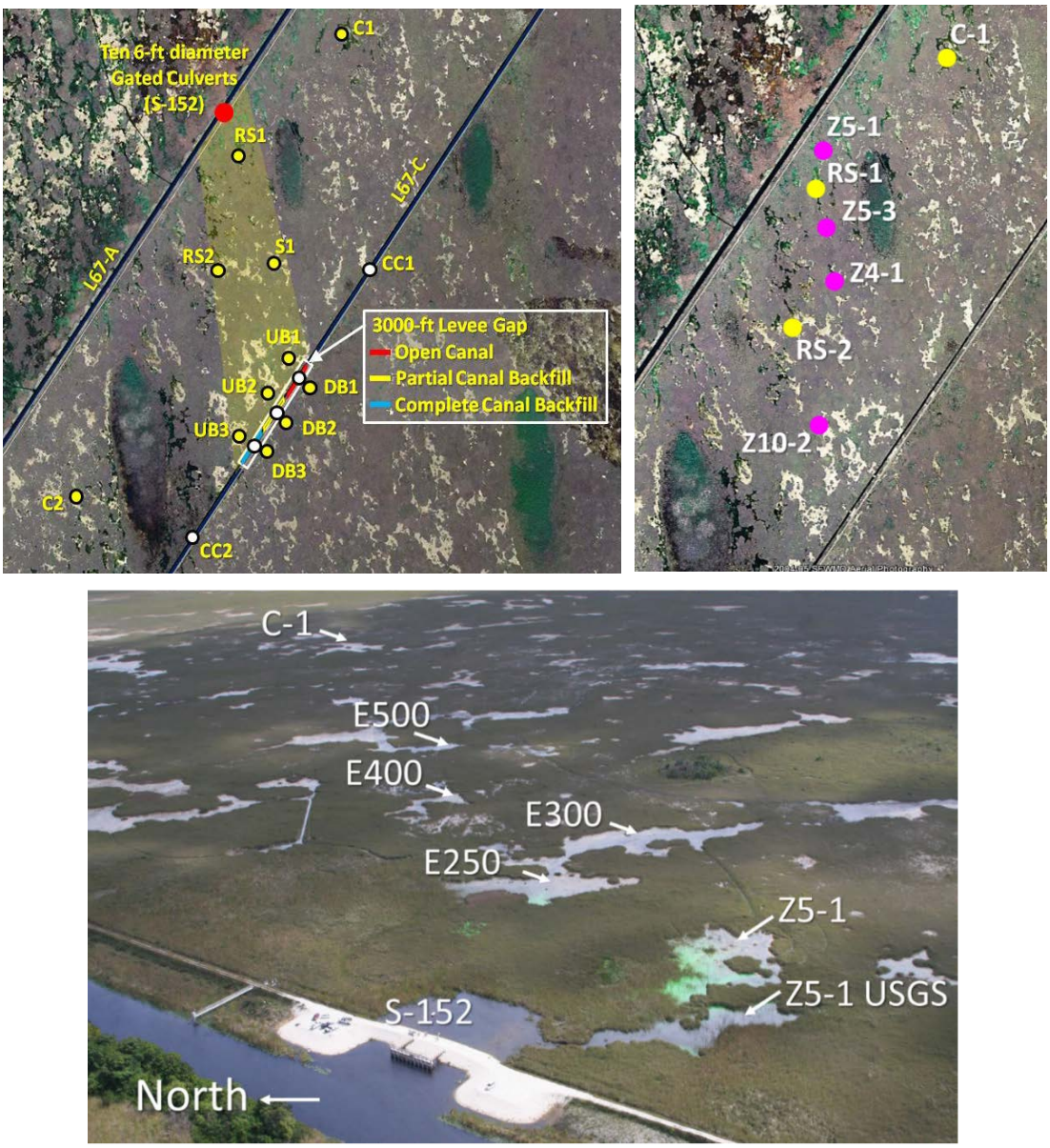


Figure 6-46. Top left: DPM study located in "the pocket" between the L-67A and L-67C canal/levees, showing 11 marsh and 5 canal sites. Site RS1 includes an upstream and downstream boardwalk, RS1U and RS1D, respectively. Top Right: additional sites (pink) added to analyze changes along the north-to-south flow path in a subset of response variables. Bottom: location of east transect sites (E250, 300, 400, and 500) and two Z5-1 sites for additional water quality and flow monitoring. (Note: green-colored water, especially at Z5-1, is a Fluorescein dye used to track flow.)

Sediment transport was measured using horizontal traps (Phillips et al. 2000) deployed in ridges and sloughs at RS1, RS2, and C1. Traps were deployed at three-week intervals in the months of November, December, and January and approximately six-week intervals after that. Deployments were stopped when slough water depths fell below 20 cm, typically April or May of each water year. As median sediment particle size in Everglades wetlands is less than 100 microns (µm) (Noe et al. 2007) only the fluxes of small particles (< 1 mm) are reported here. Samples were collecting using settling funnels and dried at 80 degrees Celsius (º C) for weight determination. Marsh loading rates per frontal area were estimated based on weight, inlet tube area, and deployment time. With each trap deployment, the height of benthic sediments, or floc,

was measured using a meter stick, and mid-water column velocity was measured using a Sontek hand-held FlowTracker, averaged over 2-minute intervals.

As described in last year's SFER (Saunders et al. 2016), molecular organic biomarkers were used to determine changes in the source of organic matter in benthic sediments (floc) and the water column. Based on concentrations of different types of lipid compounds, these proxies have been used in Everglades communities to distinguish organic matter derived from sawgrass and slough habitats, as well as different algal sources (Jaffé et al. 2001, Saunders et al. 2016). Sediment biomarker analyses were conducted on benthic floc and advected sediments (when sufficient sample was collected) from horizontal traps. In the first flow event, a biomarker specific to green algae, Botryococcenes (Gao et al. 2007), indicated that green algal sources of floc increased in high flow areas. Findings from the second flow event, including Botryococcenes and the biomarker C20 Highly Branched Isoprenoids typical of cyanobacterial-dominated metaphyton, are presented here.

A dual signature tracer (DST), composed of fluorescent, paramagnetic material, was used to determine the spatial extent of particle movement through and across the boundaries of ridge and slough habitats. The DST was hydraulically matched to the size and settling velocity of Everglades floc (PARTRAC 2008). Prior to opening the S-152 structure, 25 kilograms (kg) of DST was frozen and then deployed in slough habitat (2010 and 2013), a slough-ridge ecotone (2014), and ridge habitat (2015) at the high flow site (RS1) and the control (C1). To measure spatial movement, 20 to 24 magnets (11-gauss) were placed radially around the deployment location at high flow and low flow sites and retrieved the week after flow started. DST collected on each magnet was then dried and weighed. Due to inadvertent capture of organic matter, DST weight could only be reasonably measured to within 0.005 gram; therefore, for samples weighing less than 0.005 gram, DST abundance was approximated by visually counting particles under an ultraviolet lamp, the final count based on the average of two independent observations per sample.

Water Quality and Nutrient Dynamics

Water column samples were collected monthly at DPM marsh sentinel sites, July–March, during the 2011–2016 sampling periods. Samples were collected mid-water column, acidified, and analyzed for TP following standard District protocols. Other parameters were also analyzed, including specific conductivity and total calcium, using standard District protocols. S-152 and S-151 samples were collected weekly. For each water quality parameter, hierarchical cluster analysis (Wards method) was performed using JMP software (SAS corporation) to elucidate spatial grouping among sites by month. Generally, the first break was used to identify primary site groupings.

In last year's SFER (Saunders et al. 2016) we examined how flow affected water quality along the preferential, eastern flow path from S-152. Sites ranged 250 to 870 m from inflow, and flow effects were most evident within 500 m. This year, we continued the examination of this flow path, collecting samples during the start of two flow releases at sites 250, 300, 400, and 500 m from S-152. Concurrent flow (Sontek FlowTracker®) measurements were collected along with water samples, and water level (Onset HOBO® water depth data logger) and specific conductivity (Hydrolab®) were logged at 2- and 30-minute intervals, respectively.

Algal Processes

Changes in algal community, biomass, and productivity were monitored in open water sloughs along the flow gradient at 250, 300, 400, and 500 m from S-152. Artificial substrates consisting of 7.5-cm long x 0.12-cm diameter acrylic dowels and 0.75-cm^2 acrylic plates were deployed at each site. Periphyton colonization was examined monthly from January through May 2016, spanning the two dry season openings in February and May (see *Study Design* above). The May 25 sampling date occurred during non-flowing conditions. Each sample represents the accumulation of algae and cyanobacteria on an acrylic plate over three-weeks prior to sampling. Biomass and dissolved oxygen measurements were performed using the

dowels, while the plates were used for species determination. Productivity was measured using the SFWMD standard operating procedure for determining periphyton primary production (SFWMD 2015c). Productivity is calculated as a function of photosynthetic active radiation (PAR); however, PAR measurements were not taken during the January sampling and these were excluded from analysis. Subsamples of known volume (determined gravimetrically) were enumerated at 400x microscopically and calculated as number of cells per unit area of acrylic plate. Statistical tests included ANOVA followed by Tukey honest significant difference multiple comparison tests using JMP software (SAS corporation).

Because changes in periphyton cover were observed in high flowing sloughs during previous years, in Flow Event 3, we photo-documented periphyton loss in the RS1 slough using the BOA imaging system (Sklar and Dreschel 2014) at 900 ft altitude, resulting in an image with 3.7 cm per pixel resolution. We also measured floc depths along a transect across RS1 over time, during sediment trap deployments, to document concomitant changes in floc along with the changes in periphyton.

Hydrology and Sediment Characteristics

S-152 Discharges and Landscape Flow Velocities

Measured discharges through S-152 during the WY2014, WY2015, and WY2016 flow events are shown in **Figures 6-47** and **6-48**. The combined discharge capacity of approximately 250 cubic feet per second (cfs) through the S-152 structure generated water velocities as high as 4 to 5 cm/s.

Water released during the WY2014 flow event spread radially from the S-152 across the north and eastern portions of the study area (Sklar and Dreschel 2015). Whereas water levels, flow speeds, and flow directions reached steady conditions within hours at stations within a few hundred meters from the S-152, it took three days for steady conditions to be reached 3 kilometers away. The maximum rise in water depths caused by the flow event was 11 cm (RS1), 6 cm (RS2), 5 cm (S1), 3.5 cm (UB2), and 2.5 cm (DB2). Flow speeds increased from background values (< 1 cm/s) reaching speeds up to 4.5 cm/s at RS1 (**Figure 6-47**). The highest flow speeds were considerably less at RS2 (2 cm/s), S1 (1 cm/s), UB1-3 (1 cm/s), and DB1-3 (3 cm/s).

In the WY2015 flow event, the rise in water depth was 12 cm (RS1), 5.5 cm (RS2), 5 cm (S1), 3 cm (UB2), and 4 cm (DB2). Flow speeds increased by 3.0 cm/s at RS1, 1.2 cm/s at RS2, 1.1 cm/s at S1, 0.3 cm/s at UB sites, and 0.7 cm/s at DB1 and DB2. Thus the effects of the S-152 flows on water depth and flow velocity were concentrated near the S-152 and those effects decayed substantially with distance away. Flow speeds at control site C2 remained below 1 cm/s during the 2014 S-152 flow event.

In the WY2016 flow event, we installed a new monitoring station, Z51_USGS upstream of RS1U, and our monitoring efforts were more focused on the transect between Z51_USGS to RS1D. The maximum rise in water depths caused by the flow event was 20 cm at Z51_USGS and 10 cm at RS1D. Flow speeds increased by 4 to 5 cm/s and 13 cm/s at Z51_USGS ridge and slough, respectively. At RS1D, the flow speed at the slough increased to 6 cm/s (**Figure 6-48**), but the ridge side increased by only 3 to 4 cm/s.

Flow speeds in sloughs generally were 25 to 40 percent faster than in ridges, due to reduced biomass and flow resistance in sloughs. Also, flow directions in sloughs tended to align with slough direction and while flow on ridges was somewhat misaligned with sloughs (with direction indicating that flow jumps across ridges between sloughs), these patterns in flow speed and direction through sloughs and ridges corroborate previous observations in the Everglades ridge and slough landscape (Harvey et al. 2009). In the WY2015 and WY2016 flow events, it was possible to monitor the extended steady state flow condition, because we had a much longer period of the high flow condition than with the WY2014 experiment. From both WY2015 and WY2016 flow events, there was a gradual increase in flow speed at RS1D between November and December, followed by a decrease in the flow speed in January (**Figures 6-47** and **6-48**). The decrease can be caused by the gradual decrease in the water surface slope, the main driver of increasing the flow speed, due to the filling up the DPM study area between the L-67A and L-67C levee/canal system.

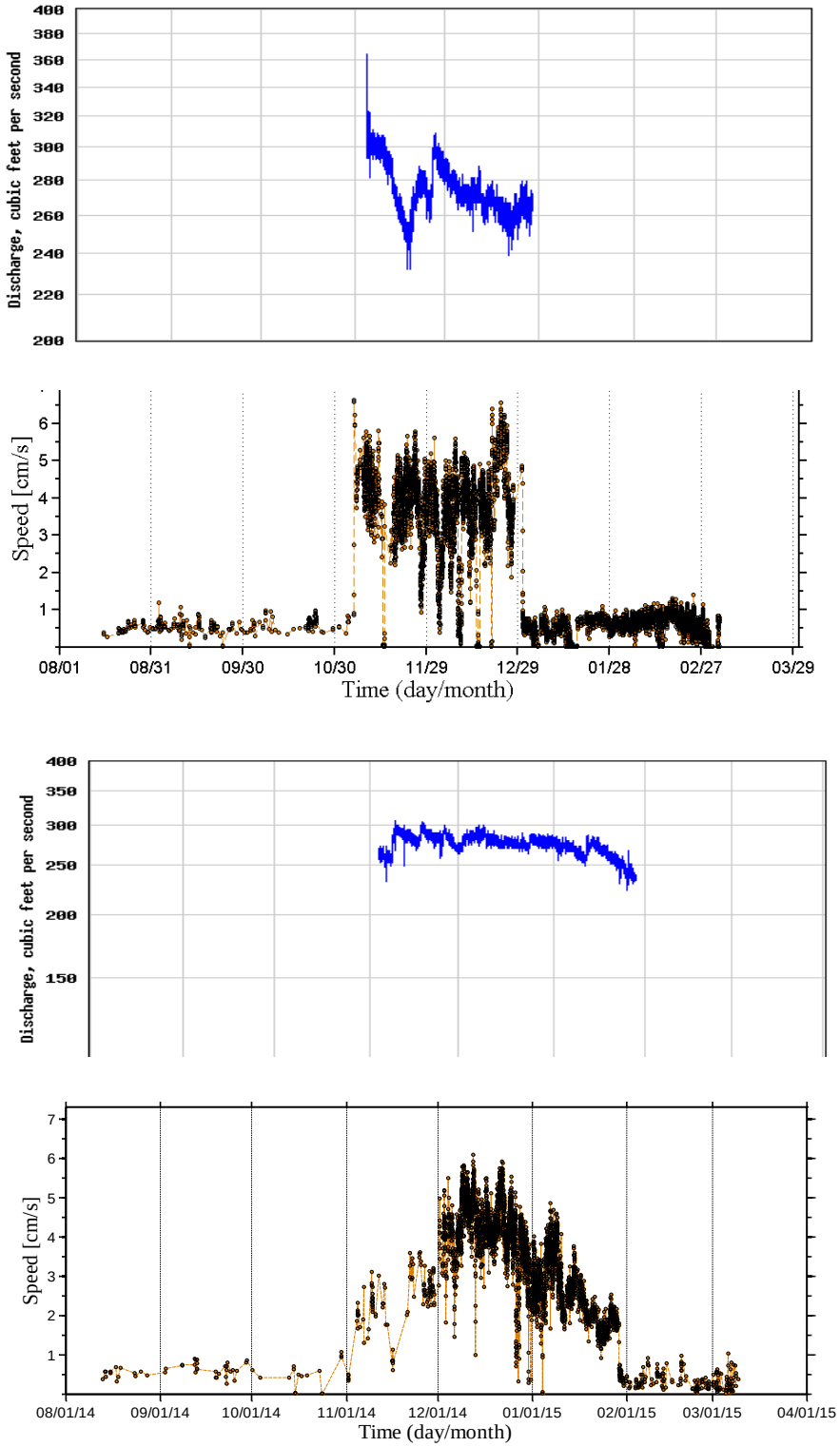


Figure 6-47. Discharge measurements at the outlet, or east end, of S-152 and continuously sampled burst average ADV flow speed recorded at RS1DS from WY2014 (August 1, 2013–March 31, 2014) (top two graphs) and from WY2015 (August 1, 2014–March 31, 2015) (bottom two graphs).

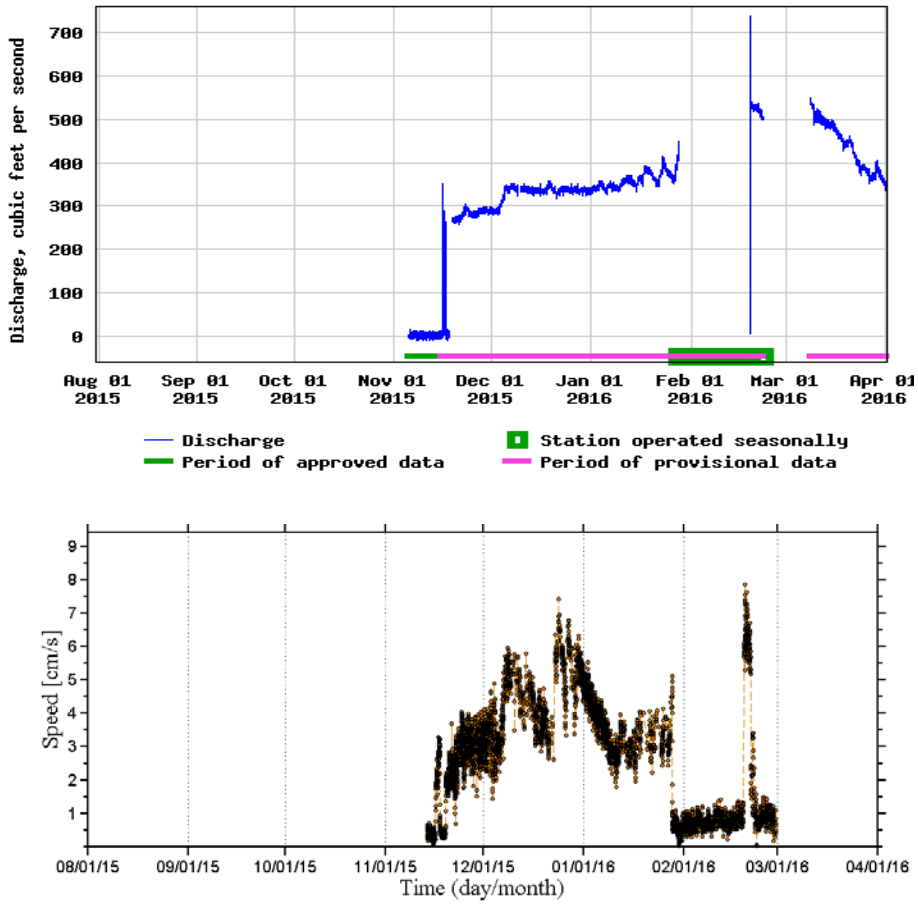


Figure 6-48. Discharge measurements at the outlet, or east end, of S-152 during the WY2016 experiment (National Water Information System, USGS, 2015) (top) and continuously sampled burst average ADV flow speed recorded at RS1DS site during WY2016 (August 1, 2015–March 31, 2016) (bottom). Processing of ADV data after March 31, 2016 from RS1DS is ongoing and therefore omitted. The large variation in S-152 discharge in February indicates high variability associated with opening the structure; however, all S-152 data shown should be considered provisional at this time.

Sediment Properties

Biomarkers of Advected Sediments and Floc. At sites near the S-152 structure, the algal sources of floc organic matter changed within the first flow event. While this is driven by local algal community changes (see the *Algal Changes* section below), the physical transport of organic matter also likely contributes to some of this change. During and shortly after the first flow event, concentrations of Botryococcenes (a green algal marker) in floc increased from very low concentrations—< 0.1 micrograms per grams dry weight (µg/gdw)—to 15 and 22 µg/gdw at sites 150- and 350-m, respectively, from S-152 (**Figure 6-49**). Concomitantly, concentrations of C20 Highly Branched Isoprenoids, indicative of cyanobacteria (abundant in metaphyton), showed decreased concentrations at the same sites. For both compounds, advected sediments (from horizontal traps) contained concentrations similar to those in the floc. The similarity of biomarker concentrations in both floc and advected sediment, and the magnitude of sediment transport (see the *Sediment Transport* section below), suggest sediment movement downstream likely contributes to the chemical changes in floc.

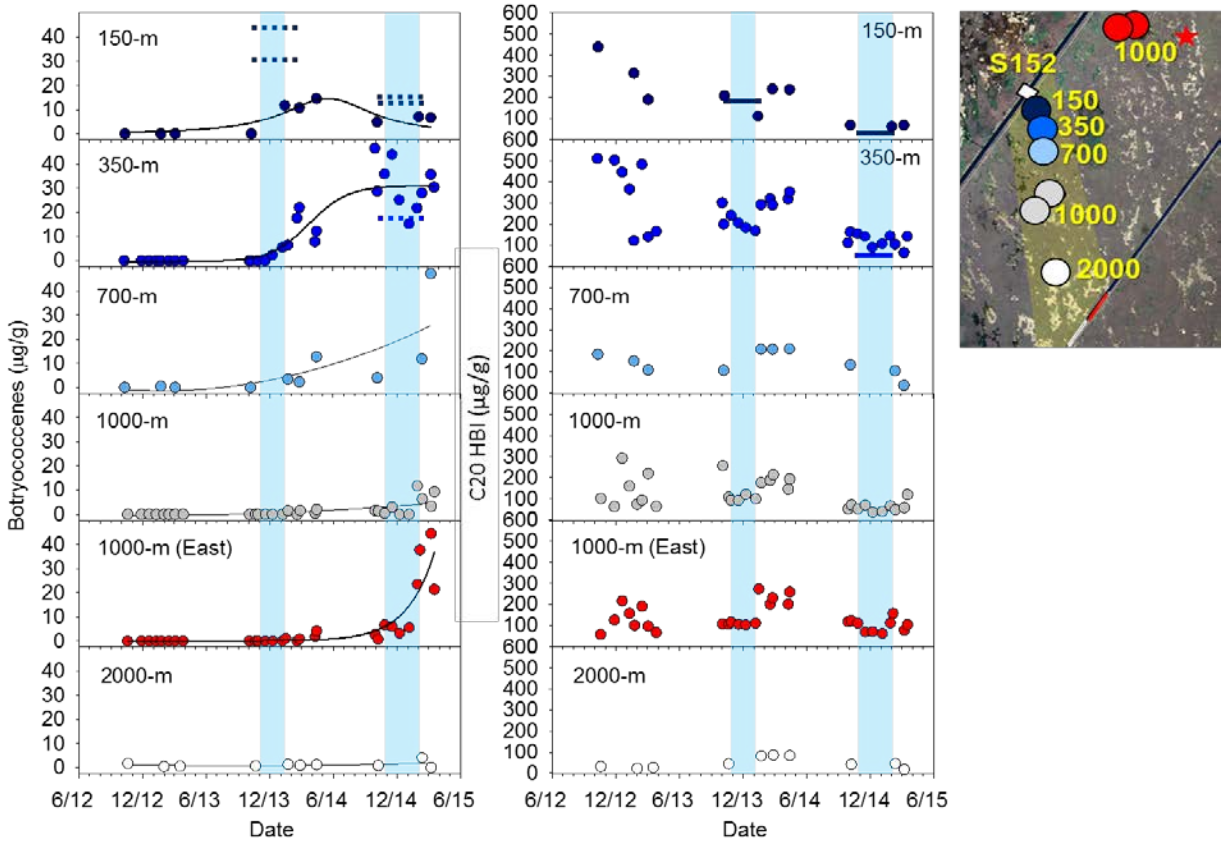


Figure 6-49. Changes in Botryococcenes (left) and C20 Highly Branched Isoprenoids (C20HBI, right) in slough floc collected at sites along the north-south flow path and at control site (east). Blue shading indicates timing of high flow events. Dashed lines represent concentrations of advected sediment collected in horizontal traps.

Although slough velocities of ≥ 1 cm/s were limited to ~500 m from the S-152, biomarker results from the second flow event indicate biogeochemical changes in floc are occurring over a larger and expanding area. Shortly after the second flow event, Botryococcene concentrations increased at sites 700-m south and 1,000-m east of the S-152. Thus, biomarkers are useful indicators of slower, but larger-scale, responses of sediment biogeochemistry to sheetflow.

Sediment Transport

Modeling studies suggest particle transport is an essential mechanism for the development and maintenance of the Everglades ridge and slough landscape by redistributing entrained sediments (Larsen and Harvey 2011). We expected that higher-velocity sloughs would demonstrate greater sediment transport than ridges. Horizontal traps showed that high flow significantly increased slough sediment transport at RS1 (**Figure 6-50**; BACI time x treatment, t = -3.54, n = 40, p = 0.0010). Transport during the high flow period was 5.4-fold higher (5.9 ± 0.7 standard error [SE] milligrams per square centimeter per day [mg/cm^2/d]) than during the baseline period (1.1 ± 0.9SE). At the control site, transport declined since 2011, likely reflecting recovery from a fire in 2011; thus, flow effects are likely underestimated by the BACI analysis. Comparing average transport measured during flow versus October (pre-flow) of each flow year, transport increased 12- to 15-fold above pre-flow values. Ridge transport also increased under high flow (t = -2.7, n = 37, p = 0.0366), but by a smaller proportion (58 percent) relative to the baseline. On a ground area basis, slough sediment transport at RS1 ranged from 3,600 to 29,000 grams per square meter per year

(g/m^2/yr) under high flow, approximately 10 to 100 times the magnitude of both floc standing stocks (~100 to 300 grams per square meter (g/m) and sediment accumulation (~160 to 250 g/m^2/yr); Craft and Richardson 1998). If just 5 percent of transported slough sediments entered ridges, this could theoretically increase sediment accumulation by 100 to 1,000 percent. These findings suggest flow-mediated transport could be critical factor affecting differential sediment accumulation in ridges and sloughs.

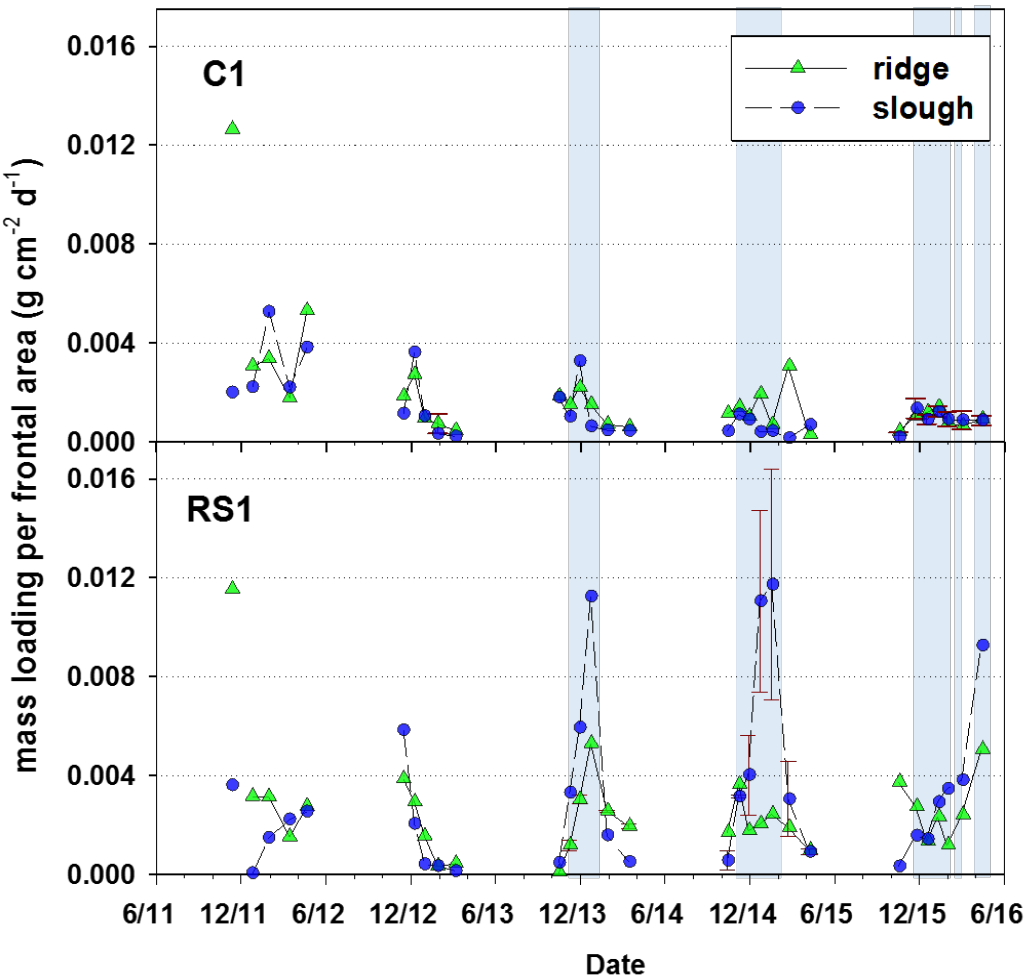


Figure 6-50. Sediment transport estimated from horizontal trap deployments in ridge and slough habitats at the control (C1) and high flow (RS1) sites. Blue shading indicates the timing of flow events (S-152 operations). Note that in WY2016, the S-152 was operated in February and March–April, beyond the normal operating window (November–January), due to a high water emergency. Error bars indicate the minimum and maximum values when duplicate traps were deployed. The study area burned in June 2011.

In all three flow events, slough sediment transport increased as flow continued, despite relatively constant S-152 discharges each time (**Figures 6-50**, **6-47**, and **6-48**). In the latter two flow events, floc height in the RS1 slough also decreased over the duration of flow, the reduction most evident 8 to 10 weeks after flow started (**Figure 6-51**). Two weeks after flow started in November 2015 (WY2016), floc at RS1 slough approximately doubled in height, but also varied spatially, in a sinusoidal pattern across the slough (**Figure 6-51**). This likely reflects the sinking and disintegration of floating periphyton mats into the floc layer; a process visually observed within days or weeks during all three flow events (see the *Algal Changes*

section below). Based on the gradual decrease in floc height during flow events 2 (WY2015) and 3 (WY2016), which was 10 to 15 cm per 8 weeks, the estimated turnover time of floc was approximately 70 to 80 days. This rate is consistent with, if slightly longer, than floc turnover estimated from in situ incubations (turnover time = 30 to 60 days; Wood 2005).

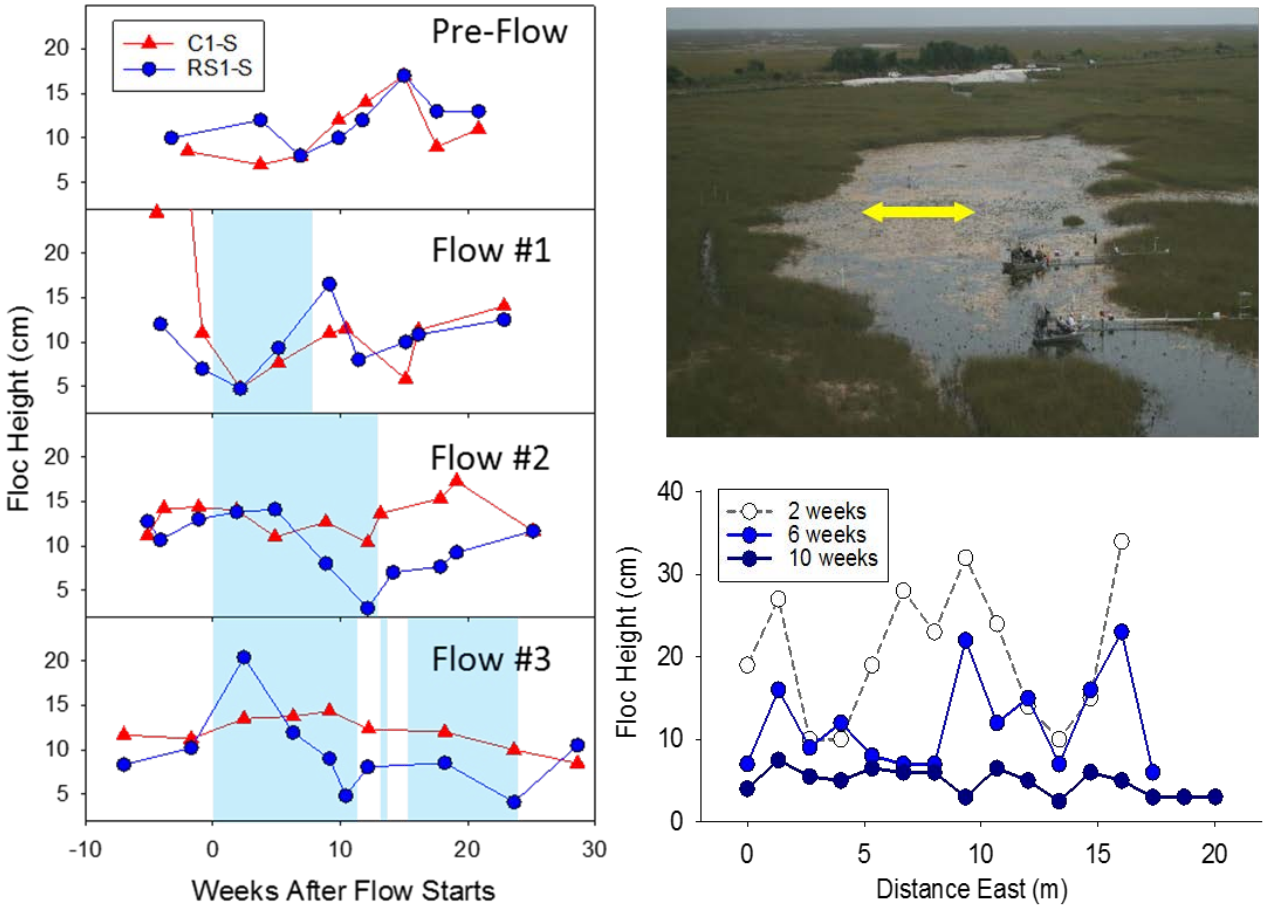


Figure 6-51. Left panels: floc height in the RS1 (blue circles) and C1 (red triangles) sloughs versus number of weeks after flow starts, or for the baseline period, number of weeks after November 1. Blue shading indicates timing of high flow events. Upper right: approximate location of floc transect (yellow arrow) at RS1. Lower right: floc height as a function of distance across the RS1 slough, at 2, 6, and 10 weeks after the start of flow on November 16, 2015.

Using a DST, we tested the hypothesis that increased sheetflow would entrain and transport sediments in slough habitats, where velocities are highest, and deposit sediments in sawgrass ridges. DST was entrained under high flows created by the S-152 structure. DST travelled mainly south at the high flow site (**Figure 6-52**). Due to low DST mass captured at 3 and 6 m from the deployment plot, count data are used to compare particle movement at those distances. The DST count and mass data show more particulate movement in the sloughs, while in the ridge DST did not move beyond 3 m (**Figure 6-52**). These findings support the hypothesis that sediment movement is less impeded through sloughs while ridge movement is reduced due to vegetative resistance and decreased velocities. Examinations of DST deployments just upstream of ridges showed sediment moved at most 10 m into the ridge from the slough under high flow (data not shown). These results confirm the importance of sheetflow in redistributing sediment from sloughs to ridges, a critical mechanism for rebuilding topography and patterning of the landscape.

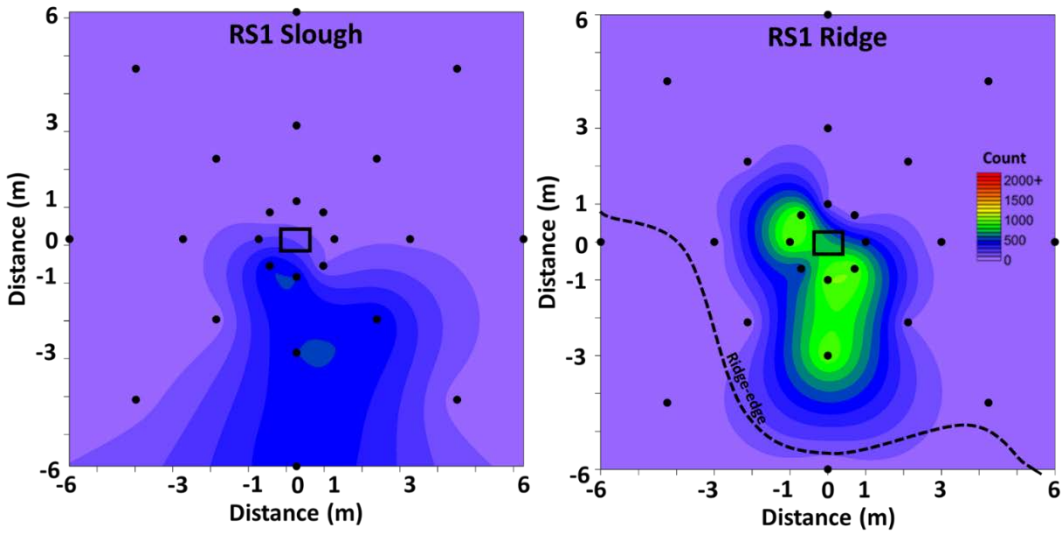


Figure 6-52. Spatial distribution of DST collected from deployments in the slough (WY2014, left) and ridge (WY2016, right) at the RS1 site. More particulate movement was found in the slough, while in the ridges, the DST does not move past 3 m.

Nutrient Dynamics

Monthly Water Column Sampling

Hydrologic changes generated by both the S-152 flows and L-67C levee gap were key drivers of both spatial and temporal patterns in water quality. In pre-flow months, flow vectors in the pocket oriented primarily toward the gap, such that the southern portion of the study area flowed east or northeast (toward the gap) and the northern portions nearer the gap flowed south to southeast (**Figure 6-53**). Thus the southern and northern areas likely received different water sources, explaining why southern (UB/DB, and C2) and northern sites (Z5-1, RS1, RS2, and C1) clustered separately during pre-flow months (**Figures 6-53** and **6-54**). Conductivity and TP showed that high flows spread water relatively evenly across the study area after 1 to 2 months (**Figure 6-54**). This reflects greater overall site-to-site connectivity associated with elevated velocities. During high flow, water moved from S-152 preferentially eastward toward the L-67C canal, then southwest down the canal. As a result, UB1-2 and DB1-2 tended to resemble S-152 source water faster than UB/DB3 (**Figure 6-54**).

The effects of flow were more complex for some nutrients. For example, over the duration of flow, southernmost sites (C2, UB3, and DB3) significantly clustered based on similar calcium (Ca) concentrations. Ca in fact increased with flow duration at these sites, leading us to hypothesize that flow may generate local Ca sources around this portion of the study area, though the mechanism remains unclear. Water TP variation also suggested dependence on localized processes. While most sites displayed low TP ($\leq$ 0.006 mg/L) as flow continued, near inflow sites Z5-1 and RS1 grouped together with higher TP. Despite the rapid velocities at these sites transporting P downstream to RS2, C1, and UB/DB sites, the limited extent of high TP reflects the rapid uptake and immobilization of P. This is not unexpected, as P is the most limiting nutrient of primary production in ridges and sloughs (Noe et al. 2001).

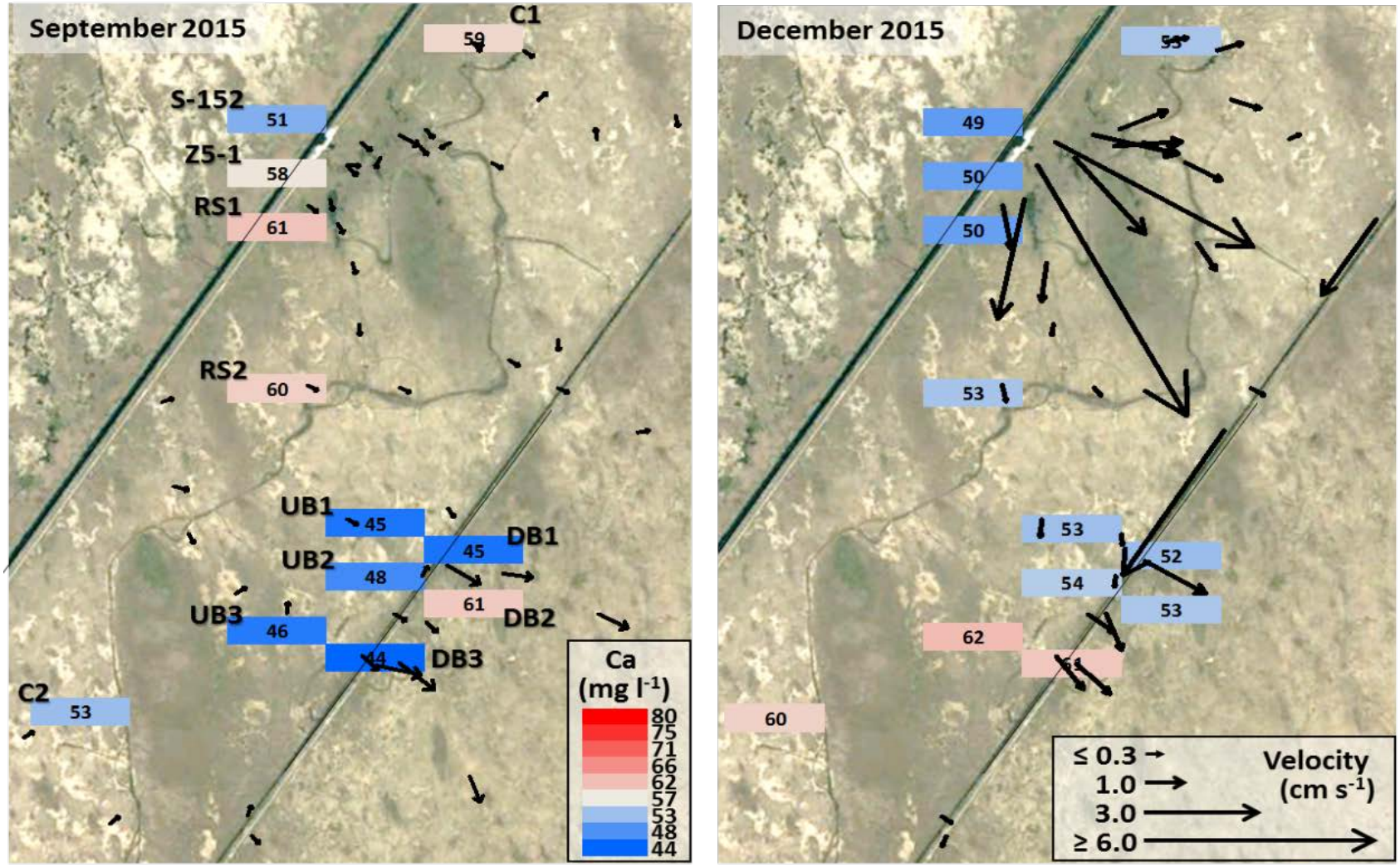


Figure 6-53. Spatial variation in velocities and Ca concentrations during pre-flow (left) and high-flow (right) periods. Dates indicate the sampling date for Ca. Pre-flow velocities represent average velocities from August to October (2012–2015) and high-flow velocities are averages of all three flow events. Velocities in the L-67C canal are based on visual observations from dye studies but consistent with in situ measured velocities provided in the 2016 SFER (Saunders et al. 2016).

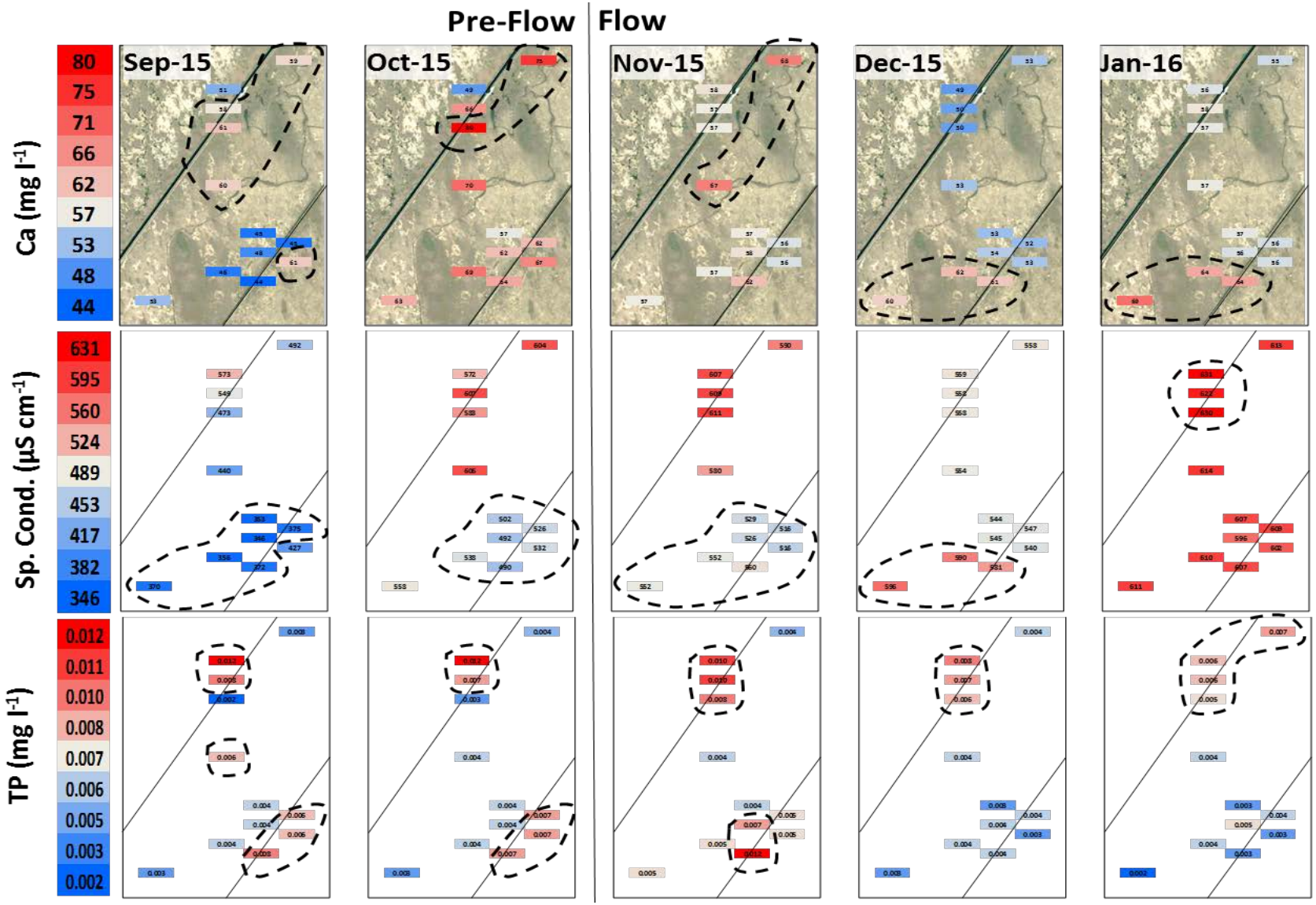


Figure 6-54. Monthly Ca (top panel), conductivity (middle), and TP (bottom) from September 2015 through January 2016. For each month, dashed lines indicate sites that group together based on cluster analysis.

Water Column Nutrients and Velocities during Pulses along the Eastern Flow Path

Along the preferential, eastern flow path, water levels increased and decreased concomitantly with the opening and closing of the S-152 structure (**Figure 6-55**). Z5-1, initially at the same level as E250, rose ~6 cm higher, suggesting a slight head build-up prior to discharge to the south. Similarly, while initial water depths at E400 were a few cm deeper than E250, E400 was shallower than E250 during flowing conditions. As expected, a velocity gradient was created along the flow path. The highest velocities measured during sampling events on the first few days of this year's pulses were 15, 13, 5, 4 to 3 cm/s at sites Z5-1, E250, E300, E400, and E500, respectively.

The lag of water movement across the landscape, and its associated effect on water chemistry, was evident from the sequential increase in specific conductivity with increasing distance from inflow (**Figure 6-56**). The highest conductivity, 687 microsiemens per centimeter (µS/cm) was observed in the canal during the second pulse. With the exception of E250, which had the lowest peak conductivity (maximum = 630 µS/cm), conductivity values exhibited a slight decrease with distance, showing the dilution effect of marsh water on the S-152 input. The inflow water had a distinct diel conductivity pattern, with maxima occurring around midnight. This pattern, though muted, was also observed in the marsh sites.

Similar to previous years, there was an increase in surface water TP concentrations as flow influenced local site conditions (**Figure 6-55**). While a single value of 35 micrograms per liter (µg/L) TP was recorded at one location (1 out of 5 subsamples within site E250, data not shown), in general, TP concentrations doubled from 6 µg/L to 13 µg/L at sites closest to inflow and experiencing the highest velocities (Z5-1 and E250) within 1 hour of the first opening the S-152. These concentrations decreased to 10 µg/L within 2 hours, but did not decline to initial conditions until flow ceased. Upon the second opening, TP values were elevated to 10 to 12 µg/L, again for a limited 2 hour-time period, before decreasing to background levels. Sites downstream also saw an increase in TP levels, and as expected, based on lower observed velocities and delayed changes in conductivity and water depth, this lagged behind changes observed upstream. TP concentrations at sites 300 to 500 m from S-152 did not increase above 10 µg/L during our sampling timeframe. It is unknown whether TP concentrations increased at E500 in response to the first flow pulse, because an elevated concentration was measured during the following stagnant period.

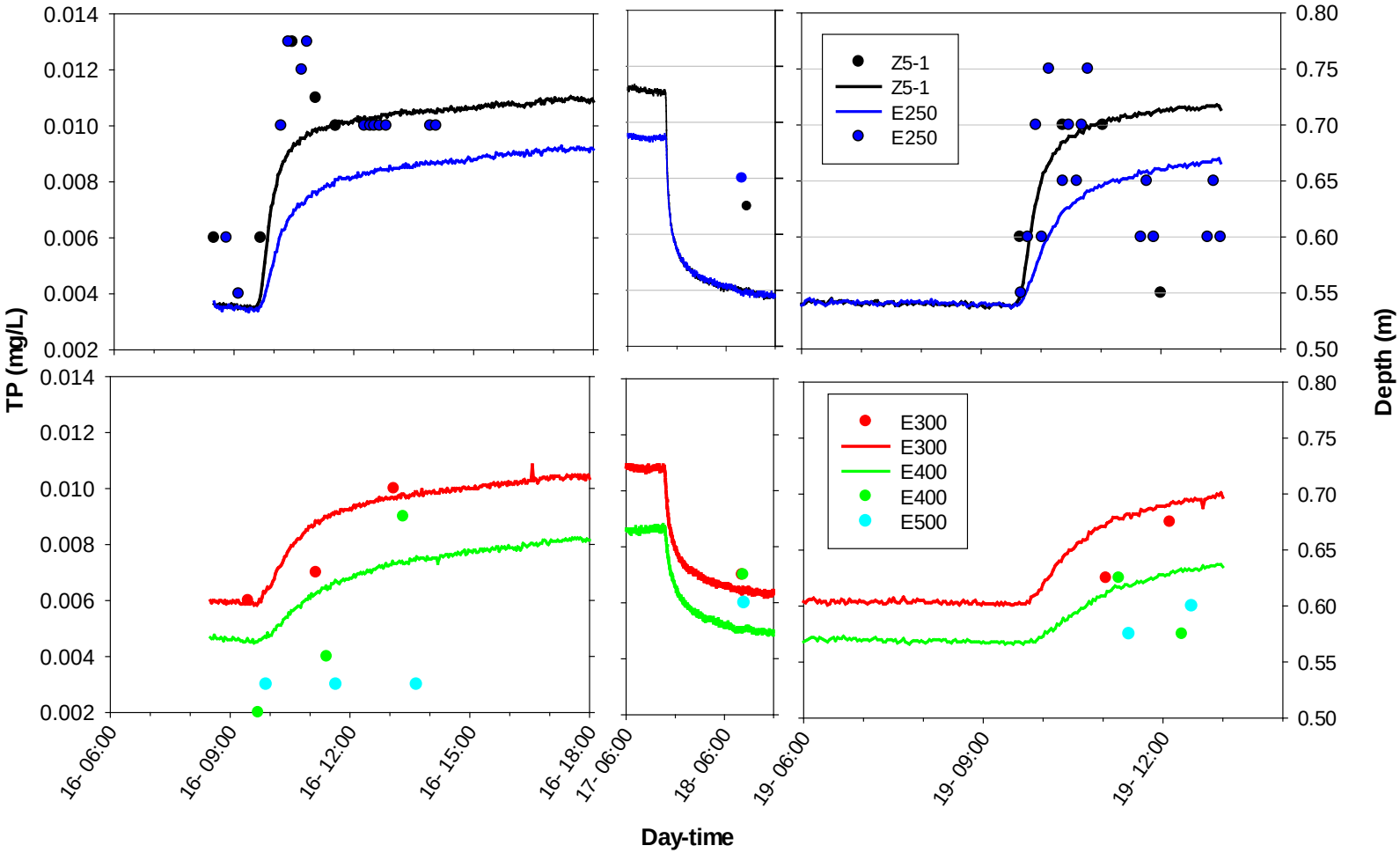


Figure 6-55. Water depths and surface water TP concentrations recorded at Z5-1 and at sites ranging 250 to 500 m along an eastern transect from S-152, immediately before and during the WY2016 pulse flow events. S-152 opened and closed on November 16, 2015, at 9:30 a.m. and November 17, 2015, at 3:20 pm, respectively, and fully reopened November 19, 2015, at 9:49 am. There was no water level recorder at E500. See **Figure 6-46** for site locations.

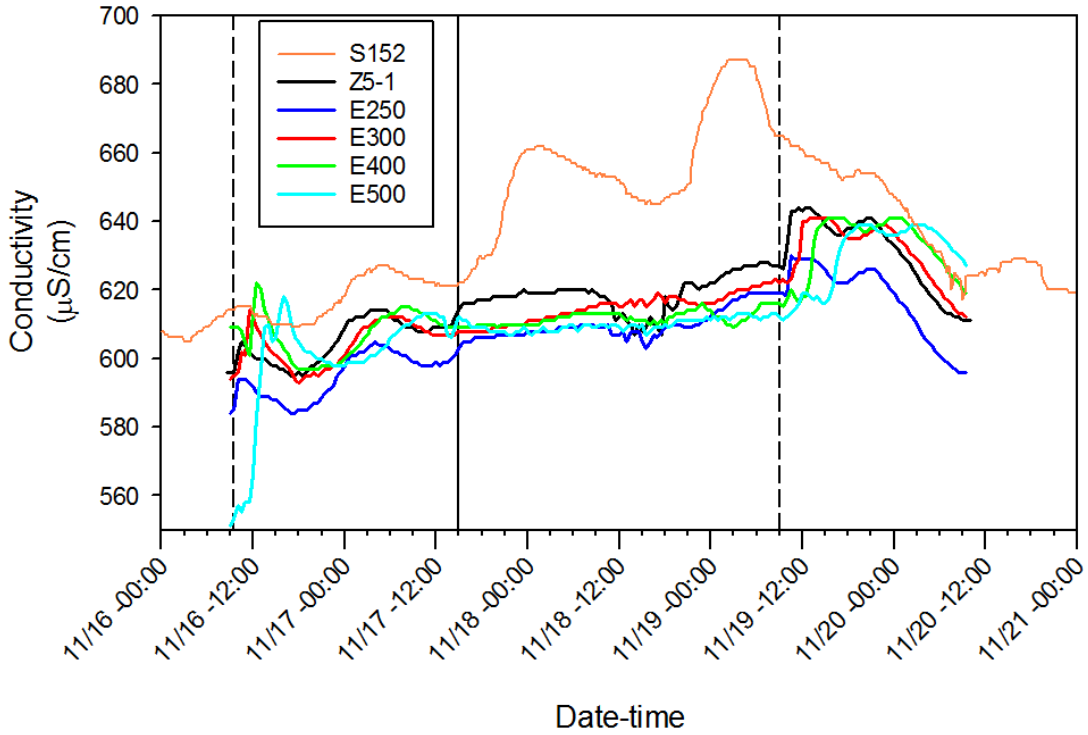


Figure 6-56. Specific conductivity recorded at S-152, Z5-1, and at sites 250 to 500 m along an eastern transect from S-152, immediately before and during the initial WY2016 flow event. Vertical dashed lines indicate S-152 opening, vertical solid line indicates S-152 closure. See **Figure 6-46** for site locations.

Algal Changes

Algal Community Responses to Flow

Accelerated flows at the sites closest to the S-152 may have increased local TP loads and in turn changed algal communities in sloughs. Changes in algal taxonomy were evaluated at the eastern transect sites (**Figures 6-55** and **6-56**) during periods of varying S-152 discharges (**Figure 6-57**). Across all sites, diatoms dominated, comprising 73 percent of organisms found, followed by Cyanobacteria, green algae, Chrysophytes, and other (sum of Dinoflagellates, Euglenoids, and Cryptophytes). Diatoms had the highest abundance (number per square millimeter [mm^2]) of all but one sample and were greatest at the E300 site. With the exception of E500, diatoms peaked in April, during the highest flows, before declining in late May, the only colonization period with no S-152 discharges. All sites had less colonization in February (collected March 2), the phenomenon most pronounced at E250 and E300. Overall, these observations suggest reduced flow contributed to lower algal abundance. This supports findings described in the SFER 2016 (Saunders et al. 2016) of high flows associated with higher algal species richness, and additionally higher green algae abundance. Similarly, the greatest abundance of green algae occurred at E250 (**Figure 6-58**), peaking during the March (highest S-152 discharges) colonization period. At E400 and E500, green algae peaked in the April colonization period.

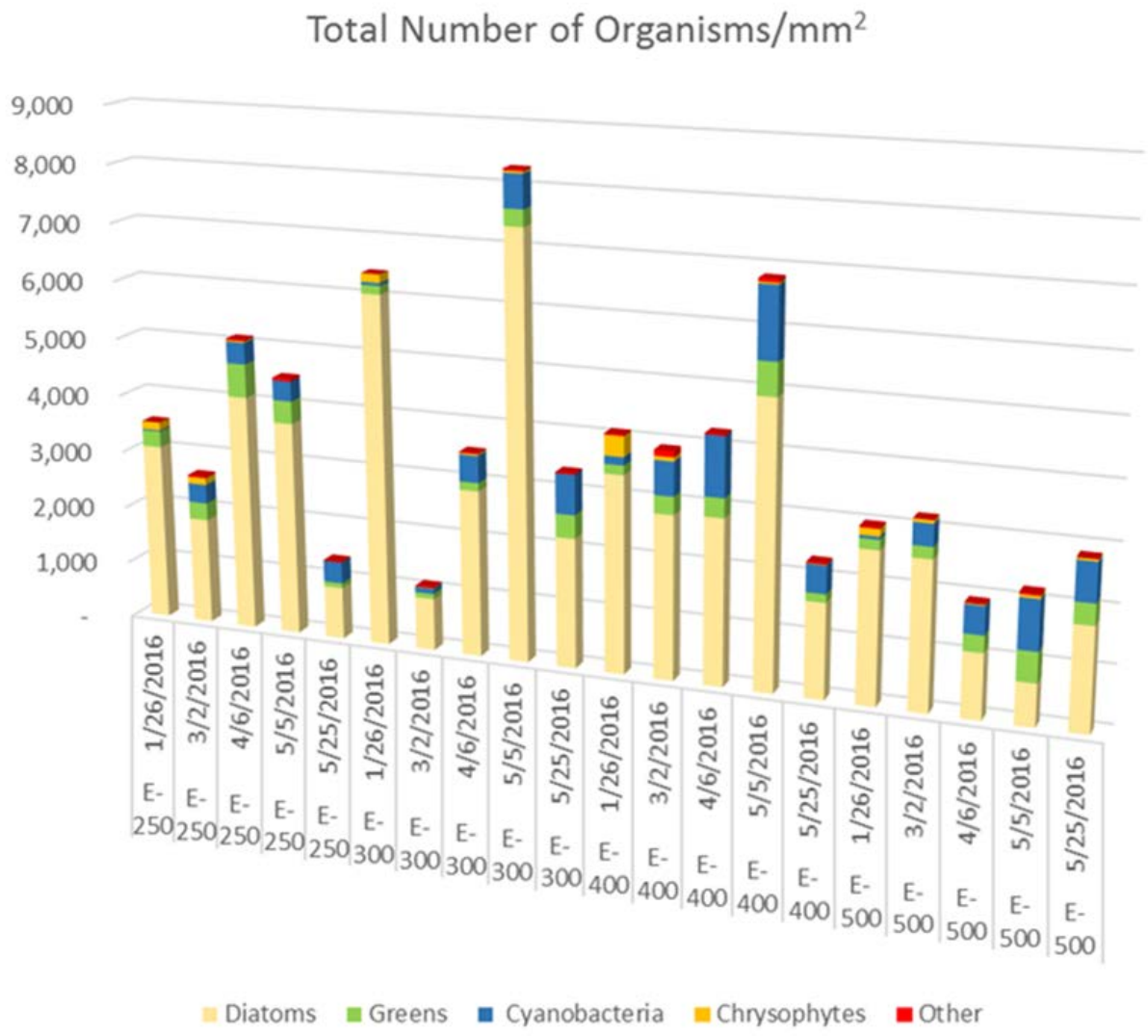


Figure 6-57. Number of each taxonomic group per surface area across site and time. Associated dates represent the collection date following a 3-week colonization period.

Seasonal temperature changes likely account for some changes in algal community. For instance, Chrysophytes, which prefer cold water, were most abundant in January (**Figure 6-57**). Green algae generally increased from January through April at all sites except E250, suggesting some temperature effect (**Figure 6-58**). Cyanobacteria, a thermophilic community, increased in abundance from January through May. The overall greatest number of cyanobacteria occurred at E400, which peaked during the April colonization period for both E400 and E500. The decrease in late May could reflect a response to reduced flow, which is contrary to the expected change based on season. E250 had fewer cyanobacteria compared to the other sites, and after the January colonization period, was the only site that did not show a seasonal response.

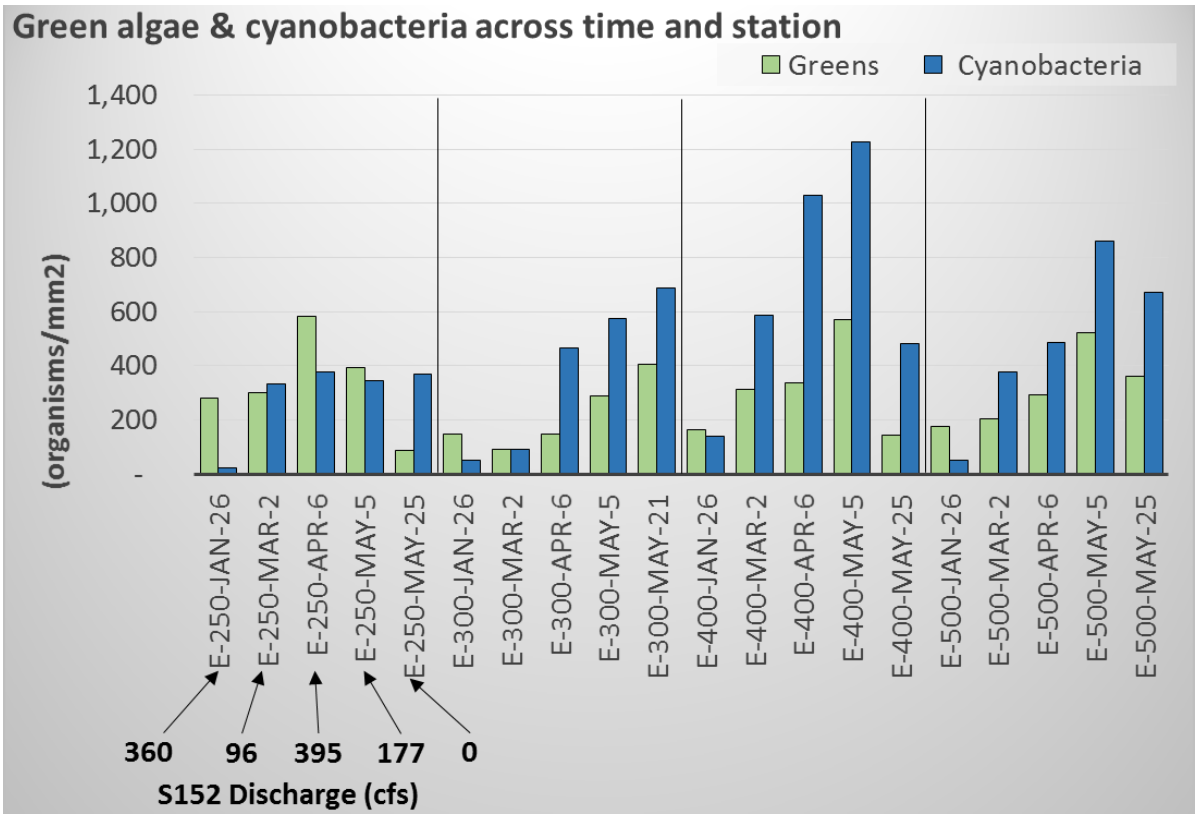


Figure 6-58. Trends in green algae and cyanobacteria temporally and spatially near S-152. Number of each taxonomic group per surface area across site and time. Discharge represents the mean daily average discharge from S-152 for each colonization period. (Note: mm2 – mm^2.)

Flow Impacts on Algal Biogeochemistry

We hypothesized that periphyton biomass and productivity would increase in response to higher flow and in turn higher P loading. Periphyton biomass tended to be higher at sites nearest the S-152 and during periods of higher water flow, however the monthly variability suggests the influence of other factors such as temperature. Samples from both March and April had higher biomasses at E250 compared to E500 ($p < 0.0001$ and $p = 0.0082$) (**Figure 6-59**). In contrast, while the May 5 and May 25 sampling dates showed significant differences among sites ($p = 0.0029$ and $p = 0.0338$), neither showed a clear, monotonic relationship with distance from S-152. One caveat is that site E400's low biomass in the May 25, 2016, sampling date was likely due to shading of dowels and plates by the submerged aquatic plant *Utricularia foliosa*.

Pre-flow productivity measurements on March 2, 2016, had significant differences in gross primary production (GPP) between sites ($p < 0.001$); however, sites nearest to S-152 did not always exhibit the highest GPP (**Figure 6-60**). On April 6, GPP showed a significant ($p < 0.0015$) effect of distance, site E250 being lower than E300 and E500. On May 5, GPP was significantly different among sites ($p < 0.0001$), but E250 and E300 had higher GPP than E400 and E500. On May 25, 2016, post-flow, there were no significant differences in GPP among sites ($p = 0.2922$).

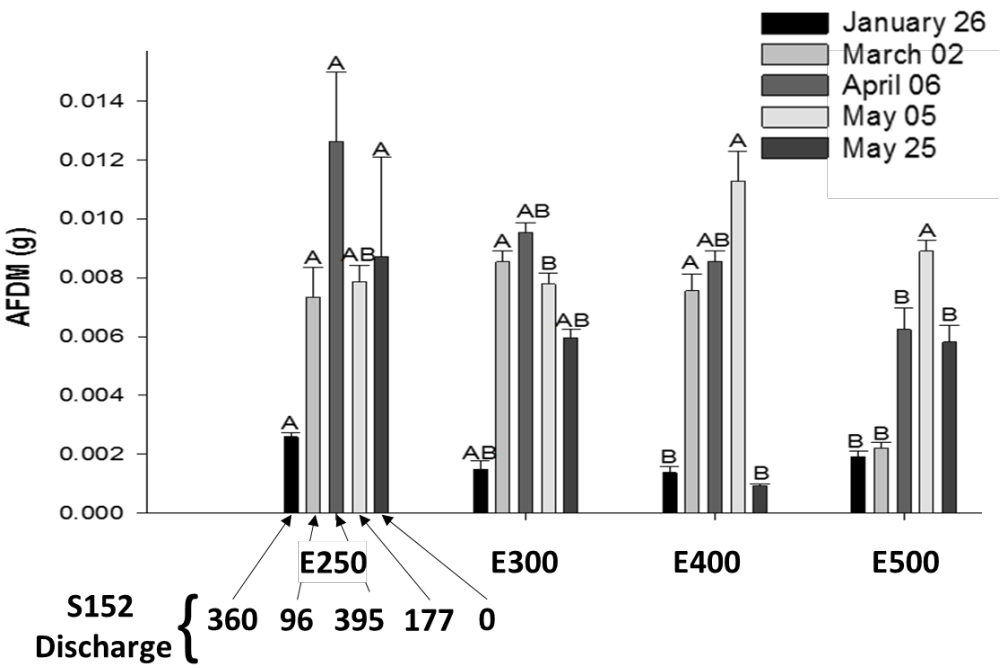


Figure 6-59. Periphyton biomass on artificial dowel substrates following 3-week incubations at each site. Dates represent collection dates. Error bars represent standard error. Discharge (cfs) represents the mean daily average discharge from S-152 for each incubation period. Letters indicate significant difference between sites within a month.

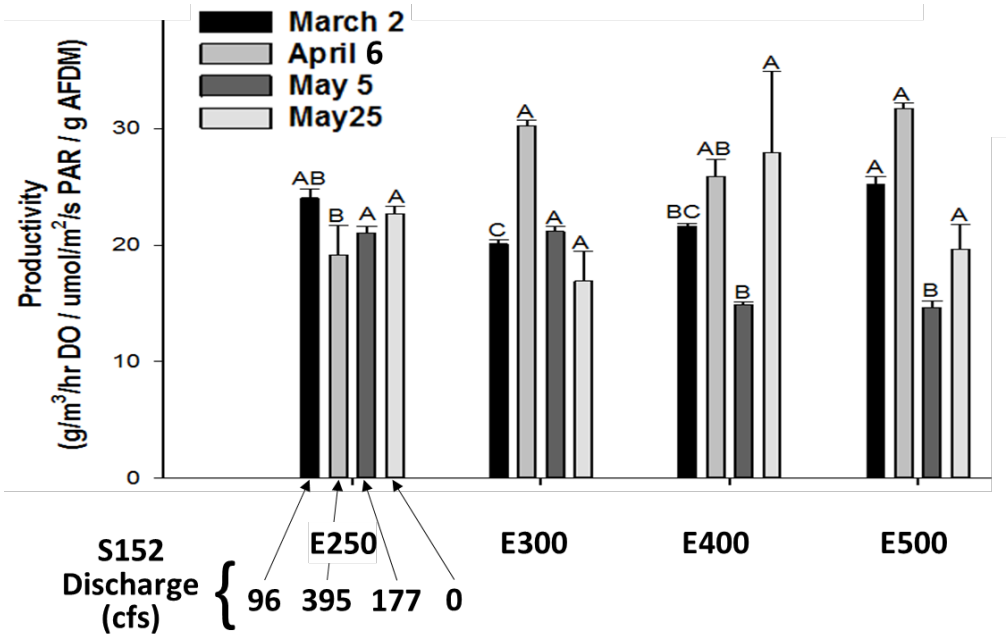


Figure 6-60. Periphyton productivity expressed as GPP. Error bars represent standard error. Letters indicate significant difference between sites within a month. (Note: g/m^3/hr DO – grams per cubic meter per hour dissolved oxygen; µmol/m^2/s PAR – micromole per square meter per second photosynthetically active radiation; and g AFDM – grams ash free dry mass.)

This study was a first attempt at linking flow effects on both biogeochemical processes and taxonomic shifts in algae. The hypothesis that periphyton biomass and productivity would increase with flow was supported during some but not all deployments from January through May 2016. Mostly, biomass attenuated with increased distance from S-152 (i.e., lower velocities) or was reduced under low or non-flowing conditions. Higher biomass with higher flow also appears consistent with greater algal species abundance, particularly green algae, under high flow. Temperature or other seasonal variation may also play a role and potentially interact with flow. Productivity results, however, generally showed greater variation, and both positive and negative responses to flow. Though preliminary, the results suggest that even under low water TP conditions, P loading due to high velocities may be important in governing algal community type and biomass, and the production and cycling of organic matter and P.

Large-scale Changes in Periphyton

The loss of periphyton in sloughs with elevated velocities, such as RS1, had been visually observed during previous flow events. Just prior to the third flow event (WY2016), high resolution imagery showed that floating metaphyton (white material in **Figure 6-61**, left) covered virtually the entire RS1 slough. Sinking periphyton (tan/orange material) was observed within 4 days after flow began (**Figure 6-61**, right). In the pre-flow condition, open water areas (evident as black lines in the imagery) represent likely alligator or other wildlife trails through the slough. The addition of flow caused these trails to widen or the periphyton in those areas to sink. We hypothesize that such alligator and wildlife trails may provide a path of least resistance through a slough and serve as starting points for larger flow paths. Based on the imagery, over 90 percent of the periphyton at RS1 had settled or was in the process of sinking with 4 days of flow. Settling of periphyton is consistent with the initial increase in floc height at RS1 (**Figure 6-51**). Moreover, areas of minimal floc across the transect are apparently in line with the initial alligator or wildlife trails. The sinking and eventual near disappearance of both metaphyton and floc (**Figure 6-51**) would reduce vegetative (live and dead) resistance to flow, explaining why velocities and sediment transport tended to increase with flow duration at RS1 (**Figure 6-50**).

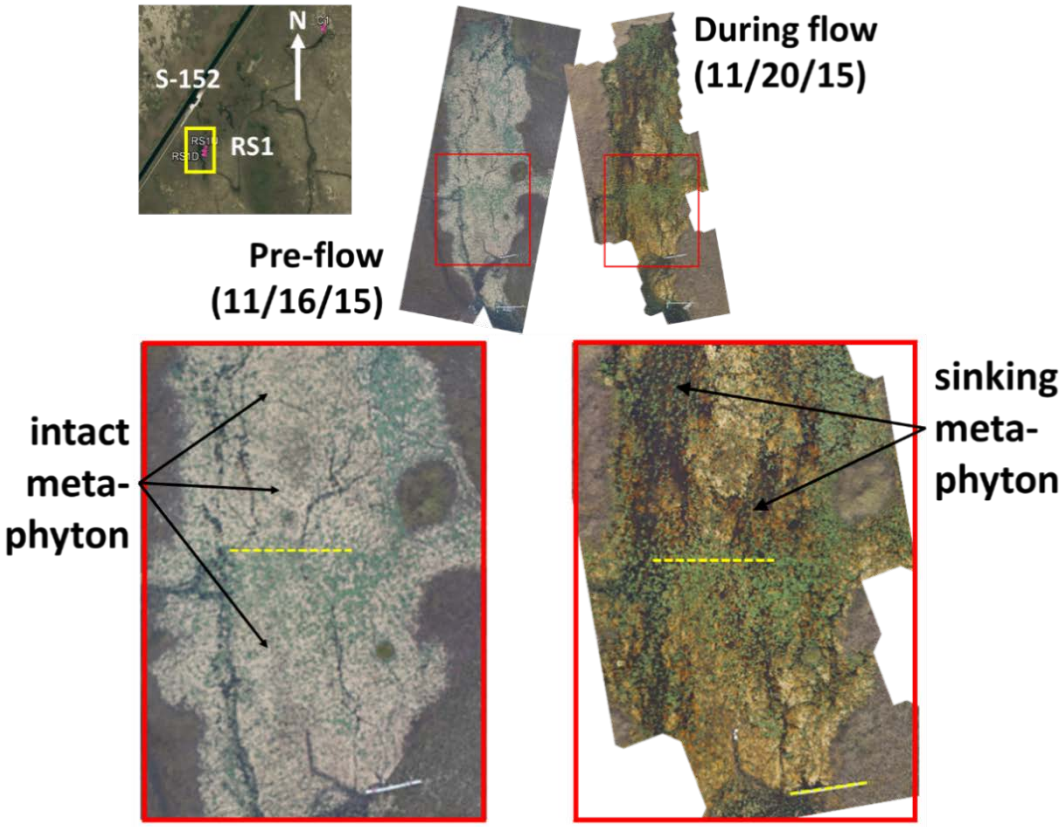


Figure 6-61. High-resolution imagery of RS1 slough during pre-flow conditions (left) and 4 days after flow was initiated (right). The yellow dashed line indicates approximate location of floc height measurements taken along a transect (see **Figure 6-51**). Line in bottom right portion of the images is the RS1u boardwalk (12-m length).

Synthesis

Findings from the three flow events together suggest that sediment dynamics in the ridge and slough are complex, involving the interplay of biological responses to flow as well as physical impacts. Using findings presented here and in previous reports, we present a conceptual model (**Figure 6-62**) to synthesize the sequence by which these feedbacks occur, their ramifications for restoration, and to provide insights about landscape-scale and longer-term responses to flow. The model also highlights new information needed to explain processes that help predict the pace and scale of benefits from sheetflow restoration.

In areas within 500 m of S-152 inflows (**Figure 6-62**, top), the most immediate responses to high flow included increased slough velocities of ~3 cm/s within hours, and sinking of most slough metaphyton community within days, exemplified at RS1. Though the exact mechanism is unknown, we hypothesize periphyton loss reflects excess loading of P with flow, previously shown to lead to mat disintegration (Gaiser et al. 2006), though in non-flowing conditions. Metaphyton collapse leads to initial, rapid floc accumulation, but metaphyton does not recover. We hypothesize that floc production is reduced with the loss of metaphyton. Possibly as a result to reduced metaphyton, green algae production increases, confirmed by biomarker and taxonomic analyses of periphyton, advected sediments, and benthic floc. While our pilot algal study suggested flow may stimulate biomass and productivity, floc measurements suggest this stimulation is insufficient to maintain original pre-flow floc levels. Other measurements show floc becomes more erodible after sustained flows, likely reflecting a change in organic matter source influenced to some degree by green algal sources. As green algal-dominated sloughs have been shown to have higher aquatic respiration than ones dominated by cyanobacterial metaphyton (Hagerthey et al. 2010), increased green

algal floc suggests flow may alter sediment decomposition (and ultimately accumulation) rates. The combination of reduced floc production, increased floc erodibility, and potentially more labile floc likely contribute, to varying degrees, to reduced floc stocks over 8 to 10 weeks of flow. Finally, since slough velocities increase with reduced biomass (live + dead), the combined loss of metaphyton, along with floc, provide a positive feedback increasing slough velocities, which typically peaked after 2 months of flow. This positive feedback would then amplify the physical and biological responses described above.

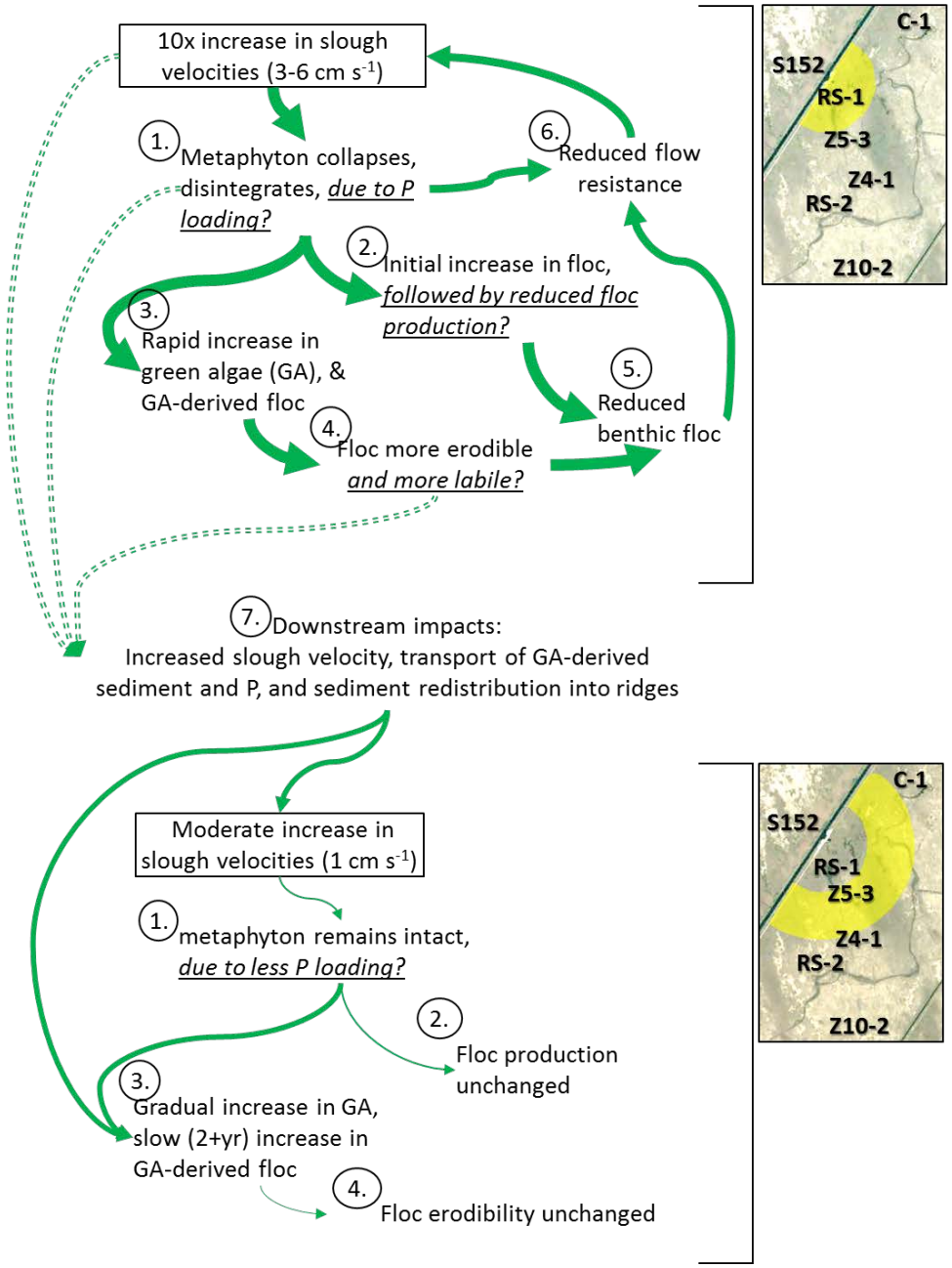


Figure 6-62. Conceptual model of flow and biogeochemical processes in the ridge and slough, based on observations within 500-m of the S-152 (top), and 500–1,000 m from the S-152 (bottom). Remaining uncertainties (underlined and italicized) are presented as hypotheses about underlying mechanisms. Dashed lines indicate processes that affect areas downstream. Arrow thickness is related qualitatively to the magnitude of observed changes to flow. (Note cm s^{-1} – cm/s.)

These processes likely have downstream impacts (**Figure 6-62**, bottom). At sites 500–1,000 m from S-152, slough velocities increased but slightly. Increased sediment transport (presumably with associated P) was not readily detectable, suggesting sediment was sequestered in ridges or sloughs nearer the 500-m boundary. We hypothesize that P loading was insufficient to disrupt slough metaphyton. At sites 700-m (Z5-3) to 1000-m (C1) from S-152, green algal abundance did increase in floc, particularly after the second flow. This lagged response indicates there is some transport of green algae-derived organic matter and/or algal community changes from P loading. Should these biogeochemical changes of algae and floc accumulate, then we hypothesize the striking changes observed within 500-m of S-152 (i.e., higher velocities and sediment transport, metaphyton collapse, increased green algae, and reduced floc) will eventually happen farther away. Ultimately, the pace by which these sheetflow impacts "spread" across the greater landscape remains a key unknown. Repeated flow events over multiple years will ultimately determine whether such sheetflow benefits are spreading at a snail's pace (e.g., a few meters per decade) or something more substantial for restoration (100s–1,000s of meters per year). The former would indicate the need for active management approaches.

Relevance to Water Management

The third DPM flow event continued to support previous findings that sustained flow operations of 8 to 10 weeks, rather than multiple pulses, are needed to maximize slough velocities, sediment transport and sediment redistribution, critical steps for landscape restoration. The pulse study indicated that successive pulse events do not have large-scale or lasting effects. As described in the conceptual model above, the advantage of maintaining continuous sheetflow (as opposed to pulses) is that structural changes to sloughs (loss of periphyton) may then lead to changes in the biological and physical properties of floc (i.e., more erodible, possibly more labile sources), which further accelerate flow and sediment redistribution, leading to restoration of topography and pattern.

The effectiveness of S-152 in restoring large areas still remains a key unknown. At this time, our results suggest three potential trajectories for restoration within the DPM study area: (1) sheetflow generated by S-152 will only restore small areas (500-m radius), therefore active management is needed; (2) sheetflow impacts may eventually "spread" across the landscape, likely involving feedbacks between sheetflow and biogeochemical responses; or (3) some combination of 1 and 2. Whether high velocities can be extended beyond 500-m appears to be linked to biogeochemical responses of slough SAV and periphyton to both water column TP and velocity (P loading).

Understanding the responses of algal communities to low-level P loading may help guide S-152 operations by elucidating discharge and inflow TP conditions that maximize ecological benefits. The pilot study initiated in the third flow event was a first attempt at evaluating the ecological responses to flow and P load under low TP conditions. Going forward, we will prioritize evaluating algal responses to P load, particularly at distances where high velocities and sediment transport currently taper off (~500-m from S-152). Across the DPM study area, water TP has remained low, including at sites close to the inflow ($\leq$ 10 mg/L); however, it is unclear at this time whether subtle changes in TP (by ± 2–4 mg/L) versus velocity changes drive the fundamental changes in slough algae, as at RS1. Flow impacts on long-term sediment TP dynamics will also be presented in future reporting as increased green algae dominance and P loading to soils could potentially promote cattail growth (Hagerthey et al. 2008), although such changes have not been observed in this study. Taken together, this information may be useful for evaluating sediment and nutrient spiraling in other water management efforts, including monitoring for the Everglades Stormwater Treatment Areas and the Central Everglades Planning Project.

LITERATURE CITED

Abbot, L., T. Barnes, R. Bennett, R. Chamberlain, T. Coley, T. Conboy, C. Conrad, K. Cunniff, P. Doering, M. Gostel, D. Haunert, K. Haunert, M. Hedegepeth, G. Hu, M. Hunt, S. Kelly, C. Madden, A. McDonald, R. Robbins, D. Rudnick, E. Slornick, P. Walker and Y. Wan. 2007. Chapter 12: Management and Restoration of Coastal Ecosystems. In: *2007 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Aich S., T.W Dreschel, E.A Cline and F.H. Sklar. 2011. The development of a geographic information system (GIS) to document research in an Everglades Physical Model. Journal of Environmental Science and Engineering 5:289-302.

Aich, S., S.M.L Ewe, B. Gu and T.W. Dreschel. 2014. An evaluation of peat loss from an Everglades tree island, Florida, USA. *Mires and Peat* 14:(2). (Available online at http://www.mires-and-peat.net/pages/volumes/map14/map1402.php).

Ali A., W. Abtew, S. Van Horn and N. Khanal. 2000. Temporal and spatial characterization of rainfall over Central and South Florida. Journal of the American Water Resources Association 36(4):833-848.

Alleman, R. 2011. Chapter 12: Coastal Ecosystems. In: *2011 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Angers, D.A. and J. Caron. 1998. Plant-induced changes in soil structure: processes and feedbacks. *Biogeochemsitry* 42:55-72.

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. Pages 225–281 in: F.H. Sklar and A.G. van der Valk (eds.), *Tree Islands of the Everglades*, Kluwer Academic Publishers, Dordrecht, Netherlands.

Armentano, T.V., J.P. Sah, M.S. Ross, D.T. Jones, H.C. Cooley and C.S. Smith. 2006. Rapid responses of vegetation to hydrological changes in Taylor Slough, Everglades National Park, Florida, USA. *Hydrobiologia* 569:293-309.

Beerens, J.M., D.E. Gawlik, G. Herring and M.I. Cook. 2011. Dynamic habitat selection by two wading bird species with divergent foraging strategies in a seasonally fluctuating wetland. *Auk* 128:651-662.

Beerens J.M., E.G. Noonburg and D.E. Gawlik. 2015. Linking dynamic habitat selection with wading bird foraging distributions across resource gradients. *PLoS ONE* 10(6): e0128182. doi:10.1371/journal.pone.0128182.

Blondeau, E. 2008. *Peat Moss Material Safety Data Sheet*. Lambert Peat Moss, Rivière-Ouelle, Qc, Canada, October 19, 2008

Brochet, A.L., M. Guillemain, H. Fritz, M. Gauthier-Clerc and A.J. Green. 2010. Plant dispersal by teal (*Anas crecca*) in the Camargue: Duck guts are more important than their feet. *Freshwater Biology* 55: 1262-1273.

Cahoon, D.R., J.C. Lynch, P. Hensel, R. Boumans, B.C. Perez, B. Segura and J.W. Day. 2002a. High-precision measurements of wetland sediment elevation: I. Recent improvements to the sedimentation-erosion table. *Journal of Sedimentary Research* 72:730-733.

Cahoon, D. R., J. C. Lynch, B. C. Perez, B. Segura, R. D. Holland, C. Stelly, G. Stephenson and P. Hensel. 2002b. High-precision measurements of wetland sediment elevation: II. The rod surface elevation table. *Journal of Sedimentary Research* 72:734-739.

Cahoon, D.R., P. Hensel, J. Rybczyk, K.L. McKee, C.E. Proffitt and B.C. Perez. 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology* 91:1093-1105.

Cook, M.I. (ed.). 2015. *South Florida Wading Bird Report, Volume 20*. South Florida Water Management District, West Palm Beach, FL.

Cook, M.I., E.M. Call, R.M. Kobza, S.D. Hill and C.J. Saunders. 2014. Seasonal movements of crayfish in a fluctuating wetland: Implications for restoring wading bird populations. *Freshwater Biology* 59:1608-1621.

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.

Dodds W.K. 2003. The role of periphyton in phosphorus retention in shallow freshwater aquatic systems. *Journal of Phycology* 39(5):840-849.

Dreschel, T.W. and E.A. Cline. 2015. Deploying Tree Saplings in Peat Bags as a Potential Restoration Technique: A Pilot Study. Pages 32–39 in: F.H. Sklar and T.W. Dreschel (eds.), Chapter 6, Everglades Research and Evaluation, Plant Ecology, *South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL

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

Erwin, R.M., D.R. Cahoon, D.J. Prosser, G.M. Sanders and P. Hensel. 2006. Surface elevation dynamics in vegetated *Spartina* marshes versus unvegetated tidal ponds along the mid-Atlantic Coast, USA, with implications to waterbirds. *Estuaries and Coasts* 29(1):96-106.

Frederick P.C., B. Hylton, J.A. Heath and M. Ruane. 2003. Accuracy and variation in estimates of large numbers of birds by individual observers using an aerial survey simulator. *Journal of Field Ornithology* 74:281-287.

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

Gaiser, E.E., L.J. Scinto, J.H. Richards, K. Jayachandran, D.L Childers, J.C. Trexler and R.D. Jones. 2004. Phosphorus in periphyton mats provides the best metric for detecting low-level p enrichment in an oligotrophic wetland. *Water Research* 38:507-516.

Gaiser E.E., D.L. Childers, R.D. Jones, J.H. Richards, L.J. Scinto and J.C. Trexler. 2006. Periphyton responses to eutrophication in the Florida Everglades: Cross-system patterns of structural and compositional change. *Limnology and Oceanography* 51:617-630.

Gaiser, E.E., J.H. Richards, J.C., Trexler, R.F. Doren, P.V. McCormick and S. Newman. 2008. Comment on "Estimating ecological thresholds for phosphorus in the Everglades". E*nvironmental Science & Technology* 42:6770-6771.

Gao, M., B.R.T. Simoneit, M. Gantar and R. Jaffe. 2007. Occurrence and distribution of novel botryococcene hydrocarbons in freshwater wetlands of the Florida Everglades. *Chemosphere* 70:224-236.

Gann T., D.L. Childers and D.N. Randeaau. 2005. Ecosystem structure, nutrient dynamics, and hydrologic relationships in tree islands of the southern Everglades, Florida, USA. *Forest Ecology and Management* 214:11-27.

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

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

Hagerthey, S.E., J.J. Cole and D. Kilbane. 2010. Aquatic metabolism in the Everglades: The dominance of net heterotrophy. *Limnology and Oceanography* 55:653-666.

Hagerthey, S.E., B.J. Bellinger, K. Wheeler, M. Gantard and E. Gaiser. 2011. Everglades periphyton: A biogeochemical perspective. *Critical Reviews in Environmental Science and Technology* 41:309-343.

Hagerthey, S.E., M.I. Cook, R.M. Kobza and S. Newman. 2014. Aquatic faunal responses to an induced regime shift in the phosphorus impacted Everglades. *Freshwater Biology* 59:1389-1405.

Hall, M.O. and M.J. Durako. 2016. *South Florida Fisheries Habitat Assessment Program: Annual Report for FY2015*. Submitted under Agreement 4600002556 to the South Florida Water Management District, West Palm Beach, FL. June 28, 2016.

Harvey, J.W., R.W. Schaffranek, G.B. Noe, L.G. Larsen, D.J., Nowacki and B.L. O'Connor. 2009. Hydroecological factors governing surface water flow on a low-gradient floodplain. *Water Resources Research* 45, W03421, doi: 10.1029/2008WR007129, 2009.

Irick D.L., B. Gu, Y.C. Li, P.W. Inglett, P.C. Frederick, M.S. Ross, A.L. Wright and S.M.L. Ewe. 2015. Wading bird guano enrichment of soil nutrients in tree islands of the Florida Everglades. *Science of the Total Environment* 532:40-47.

Jaffé, R., R. Mead, M.E. Hernandez, M.C. Peralba and O.A. DiGuida. 2001. Origin and transport of sedimentary organic matter in two subtropical estuaries: A comparative, biomarker-based study, *Organic Geochemistry* 32:507-526.

Jones D.L., A. Hodge and Y. Kuzyakov. 2004. Plant and mycorrhizal regulation of rhizodeposition. *New Phytologist* 163:459-480.

Kendall C. and E.A. Caldwell. 1998. Fundamentals of isotope geochemistry. Pages 51-86 in: C. Kendall and J.J. McDonnell (eds.), *Isotope Tracers in Catchment Hydrology*, Elsevier, Amsterdam.

Larsen, L.G. and J.W. Harvey. 2011. How vegetation and sediment transport feedbacks drive landscape change in the Everglades and wetlands worldwide. *American Naturalist* 176:E66-E79.

McCarthy T.S., W.N. Ellery and K. Ellery. 1993. Vegetation-induced, subsurface precipitation of carbonate as an aggradational process in permanent swamps of Okavango (delta) fan, Botswana. *Chemical Geology* 107:111-113.

McCarthy T.S., W.N. Ellery and J.M. Danergfield. 1998. The role of biota in the initiation and growth of islands on the floodplain of the Okavanga Alluvial Fan, Bostwana. *Earth Surface Processes and Landforms* 23:281-316.

McCormick, P.V., P.S. Rawlik, K. Lurding, E.P. Smith and F.H. Sklar. 1996. Periphyton-water quality relationships along a nutrient gradient in the northern Florida Everglades. *Journal of the North American Benthological Society* 15:433-449.

McCormick, P.V., S. Newman, S. Miao, D.E. Gawlik, D. Marley, K.R. Reddy and T.D. Fontaine. 2002. Effects of anthropogenic phosphorus inputs on the Everglades. Pages 83-126 in J.W. Porter and K.G. Porter (eds.), *The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook*. CRC Press, Boca Raton, FL.

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

Nott, M.P., O.L. Bass, Jr., D.M. Fleming, S.E. Killeffer, L. Manne, J.L. Curnutt, T.M. Brooks, R. Powell and S.L. Pimman. 1998. Water levels, rapid vegetational changes, and the endangered Cape Sable seaside sparrow. *Animal Conservation* 1:23-32.

Noe, G.B., D.L. Childers and R.D. Jones. 2001. Phosphorus Biogeochemistry and the Impact of Phosphorus Enrichment: Why Is the Everglades So Unique? *Ecosystems 4* 603-624.

Noe, G.B., J.W. Harvey and J.E. Saiers. 2007. Characterization of suspended particles in Everglades wetlands. *Limnology and Oceanography* 52:1166-1178.

Ogden, J.C. 1994. A Comparison of Wading Bird Nesting Dynamics, 1931–1946 and 1974–1989 as an Indication of Changes in Ecosystem Conditions in the Southern Everglades. Pages 533–570 in: S. Davis and J.C. Ogden (eds.), *Everglades: The Ecosystem and Its Restoration*, St. Lucie Press, Delray Beach, FL.

PARTRAC. 2008. *Quantifying Floc Transport in Vegetated and Non-vegetated Habitats in the Florida Everglades*. Glasgow, Scotland, UK.

Pawlik, L. 2013. The role of trees in the geomorphic system of forested hillslopes—a review. Earth Sci Rev 126:250–265

Phillips, J.M., M.A. Russell and D.E. Walling. 2000. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments, Hydrological Processes 14, 2589-2602.

RECOVER. 2012. *2012 System Status Report Interim Update*. Restoration Coordination and Verification, c/o United States Army Corps of Engineers, Jacksonville, FL; and South Florida Water Management District, West Palm Beach, FL. December 2012.

Riscassi, A. L. and R. W. Schaffranek. 2002. *Flow Velocity, Water Temperature, and Conductivity in Shark River Slough, Everglades National Park, Florida, July 1999–August 2001*. Open-File Report 02-159, United States Geological Survey, Washington, DC.

Rodgers, L. and D. Black. 2012. Effects of aerially-applied imazamox on southern cattail and non-target emergent vegetation in a eutrophic sawgrass marsh. *Journal of Aquatic Plant Management* 50:125-129.

Rodriguez A.F., A. Serna and L.J. Scinto. 2014. Soil accretion influenced by elevation, tree density, and substrate on reconstructed tree islands. Soil Science Society of America Journal 78:2090-2099.

Ross M.S., S. Mitchell-Brucker, J.P. Sah, S. Stothoff, P.L. Ruiz, D.L. Reed, K. Jayachandran and C.L. Coultas. 2006. Interaction of hydrology and nutrient limitation in ridge and slough landscape of southern Florida. *Hydrobiologia* 569:37-59.

Rudnick, D., C. Madden, S. Kelly, R. Bennett and K. Cunniff. 2007. Appendix 12-3: Report on Algae Blooms in Eastern Florida Bay and Southern Biscayne Bay. In: *2007 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Rutchey, K., T.N. Schall, R.F. Doren, A. Atkinson, M.S. Ross, D.T. Jones, M. Madden, L. Vilchek, K.A. Bradley, J.R. Snyder, J.N. Burch, T. Pernas, B. Witcher, M. Pyne, R. White, T.J. Smith III, J. Sadle, C.S. Smith, M.E. Patterson and G.D. Gann. 2006. *Vegetation Classification for South Florida Natural Areas*. Open-File Report 2006-1240, United States Geological Survey, Washington DC.

Sah, J.P., M.S. Ross, S. Saha, P. Minchin and J. Sadle. 2013. Trajectories of vegetation response to water management in Taylor Slough, Everglades National Park, Florida. *Wetlands* DOI: 10.1007/s13157-013-0390-4.

Saunders, C.J., S. Newman, E. Tate-Boldt, C. Hansen , J. Harvey, L. Larsen, B. Buskirk, J. Lewis, J. Gomez-Velez, A. Swartz, R. Jaffé, B. Rosen, J. Choi, E. Cline, C. Coronado-Molina, C. Zweig and F. H. Sklar. 2016. Appendix 6-1: Interim Findings of the Decomp Physical Model. In *2016 South Florida Environmental Report – Volume I*. South Florida Water Management District, West Palm Beach, FL.

Schlesinger, W. 1997. *Biogeochemistry, An Analysis of Global Change*. Academic Press, San Diego, CA.

SFWMD. 2013-2015. DBHydro Hydrologic and Chemistry Data Base: http://my.sfwmd.gov/dbhydroplsql/show_dbkey_info.main_menu. Accessed November 9, 2015.

SFWMD. 2014. *Rules of the South Florida Water Management District, Minimum Flows and Levels*. Chapter 40E-8, Florida Administrative Code, South Florida Water Management District, West Palm Beach, FL.

SFWMD. 2015a. *Chemistry Laboratory Quality Manual*. SFWMD-LAB-QM-2015-01, South Florida Water Management District, West Palm Beach, FL. Effective February 6, 2015.

SFWMD. 2015b. *Field Sampling Quality Manual*. SFWMD-FIELD-QM-001-08.2, South Florida Water Management District, West Palm Beach, FL. Effective January 15, 2015.

SFWMD. 2015c. *Standard Operating Procedure for Measuring Periphyton Primary Production*. SFWMD-EVER-SOP-DPM-Periphyton Primary Production-01, South Florida Water Management District, West Palm Beach, FL.

Sklar, F.H. and T. Dreschel. 2014. Chapter 6: Everglades Research and Evaluation. In *2014 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Sklar, F.H. and T. Dreschel. 2015. Chapter 6: Everglades Research and Evaluation. In *2015 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

Sklar, F.H. and A. van der Valk. 2002. *Tree Islands of the Everglades*. Kluwer Academic Publishers, Dordrecht, Netherlands.

Sklar, F.H., M.J. Chimney, S. Newman, P. McCormick, D. Gawlik, S. Miao, C. McVoy, W. Said, J. Newman, C. Coronado, G. Crozier, M. Korvela and K. Rutchey. 2005. The scientific and political underpinnings of Everglades restoration. *Frontiers in Ecology and the Environment* 3(3):161-169.

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

SonTek. 2001. *SonTek ADV Acoustic Doppler Velocimeter Technical Documentation*. SonTek, San Diego, CA.

Sontek YSI Inc. 2007. *Flowtracker Handheld ADV Technical Manual*. Sontek YSI Inc., San Diego, CA.

Stoffella, S.L. M.S. Ross, J.P. Sah, R.M. Price, P.L. Sullivan, E.A. Cline and L.J. Scinto. 2010. Survival and growth responses of eight Everglades tree species along an experimental hydrological gradient on two tree island types. *Applied Vegetation Science* 13:439-449, DOI: 10.1111/j.1654-109X.2010.01081.

Strazisar, T., M.S. Koch, E. Dutra and C.J. Madden. 2013. Ruppia maritima L. seed bank viability at the Everglades-Florida Bay Ecotone. Aquatic Botany 111: 26-24.

Strazisar, T., M.S. Koch and C. Madden. 2015. Seagrass (*Ruppia maritima* L.) Life history transitions in response to salinity dynamics along the Everglades-Florida Bay ecotone. *Estuaries and Coasts* 38(1):337-352, doi:10.1007/s12237-014-9807-4.

Sullivan P.L., R.M. Price, F. Miralles-Wilhelm, M.S. Ross, L.J. Scinto, T.W. Dreschel, F. Sklar and E. Cline. 2012. The role of recharge and evapotranspiration as hydraulic drivers of ion concentrations

in shallow groundwater on Everglades tree islands, Florida (USA). Hydrological Processes, doi:10.1005/hyp. 9575.

Sullivan P.L., V. Engel, M.S. Ross and R.M. Price. 2014a. The influence of vegetation on the hydrodynamics and geomorphology of tree islands in Everglades National Park (Florida, United States). *Ecohydrology* doi:10.1002/eco. 1394.

Sullivan P.L., R.M. Price, J.L. Schedlbauer, A. Saha and E.E. Gaiser. 2014b. The influence of hydrologic restoration on groundwater–surface water interactions in a karst wetland, The Everglades (FL, USA). *Wetlands* 34:23-35.

Sullivan P., R.M. Price, M.S. Ross, S.L. Stoffella, J.P. Sah, L.J. Scinto, E. Cline, T.W. Dreschel and F.H. Sklar. 2016. Trees: A powerful geomorphic agent governing the landscape evolution of a subtropical wetland. *Biogeochemistry* doi 10.1007/s10533-016-0213-9.

Trexler, J.C., W.F. Loftus and S. Perry. 2005. Disturbance frequency and community structure in a twenty-five year intervention study. *Oecologia* 145(1):140-152.

Van der Valk, A. G. and T. R. Rosburg. 1997. Seed bank composition along the phosphorus gradients in the northern Florida Everglades. *Wetlands* 17:228-236.

Variano, E.A., D.T. Ho, V.C. Engel, P.J. Schmieder and M.C. Reid. 2009. Flow and mixing dynamics in a patterned wetland: Kilometer-scale tracer releases in the Everglades. *Water Resources Research* 45, doi:10.1029/2008WR007216.

Washington State Department of Agriculture. 2012. *Vigoro Tree & Shrub Fertilizer 12-5-7, Selected Fertilizer Products, Fertilizer Product Information, Guaranteed Analysis*. WSDA Home, Pesticide & Fertilizer Home, Commercial Fertilizer Home. Available online at http://agr.wa.gov/pestFert/Fertilizers/FertDB/prodinfo.aspx?pname=3598.

Weinbaum, S.A., R.S.Johnson and T.M. DeJong. 1992. Causes and consequences of overfertilization in orchards. *HortTechnology* 2:112-121.

Wetzel P.R., A.G. van der Valk, S. Newman, D.E. Gawlik, T.T. 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.

Whelan, K.R., T.J. Smith III, D.R. Cahoon, J.C. Lynch and G.H. Anderson. 2007. Groundwater control of mangrove surface elevation: Shrink and swell varies with soil depth. *Estuaries* 28(6):833-843.

Williams, K.A., P.C. Frederick and J.D. Nichols. 2011. Use of the superpopulation approach to estimate breeding population size: An example in asynchronously breeding birds. *Ecology* 92:821-828.

Wood, A.D. 2005. Dynamics of Detrital Particulate Organic Material in the Ridge and Slough Landscape of the Everglades. M.S. thesis, Florida International University, Miami, FL.

Wu, Y.K. Rutchey, W. Guan, L. Vilchek and F.H. Sklar. 2002. Spatial Simulations of Tree Islands for Everglades Restoration. Pages 469-498 in: F. Sklar and A. van der Valk (eds.), *Tree Islands of the Everglades*, Kluwer Academic Publishers, Dordrecht, Netherlands.

Zweig, C.L. and W.M. Kitchens. 2008. Effects of landscape gradients on wetland vegetation communities: Information for large-scale restoration. *Wetlands* 28:1086-1096.