2008 Science Assessments

Fish and Macroinvertebrates

Wading Birds (Wood Stork, White Ibis and Roseate Spoonbill)

Florida Bay Submerged Aquatic Vegetation

Florida Bay Algal Blooms

Crocodilians (Alligators and Crocodiles)

Oysters

Periphyton-Epiphyton

Juvenile Pink Shrimp

Lake Okeechobee Littoral Zone

Invasive Exotic Plants

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Assessment of Aquatic Fauna In the Everglades

2006

Prepared by

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Introduction

A key goal for Everglades restoration is to ‘get the water right’ with the expectation that other components of the ecosystem will be restored as a result. An implication of ‘getting the water right’ is that operations of the water distribution system lead to water-level fluctuation that reflect historical patterns resulting from rainfall, and that linking surface-water dynamics to rainfall will recapture historical patterns of hydroperiod, including frequency and periodicity of marsh drying. Ecologists agree that frequency and periodicity of drying across the landscape, along with oligotrophic water quality, are key elements to restoring ecosystem function in the Everglades. Thus, assessing performance measures of Everglades management should include rainfall-based targets that adjust expectations for seasonal and inter-annual patterns of regional rainfall. In this assessment, we use a protocol that incorporates **dynamic targets** for performance measures of aquatic consumers that are designed to remove variation resulting from rainfall and focus evaluation on the residual variation resulting from water management choices.

Aquatic fauna are included as indicators of Everglades management and restoration because of their central role in the food web, supporting emblematic Everglades animals such as wading birds and alligators. In Everglades monitoring and assessment, aquatic consumers refers to small fish and crustaceans that are directly consumed by wading birds and juvenile alligators. The linkage of these organisms to water management is well established in the published literature, permitting evaluation of the impact of changing water delivery and quality on their numbers (Fig. 1). Also, their life cycles are generally one year or less, providing relatively rapid responses to changing conditions that can be assessed through standard sampling protocols. Unlike flagship species with high visibility and public support such as wading birds or alligators, aquatic consumers are included in ecosystem assessment solely because of their place in a chain of causality, linking water management and animals of high value to society.

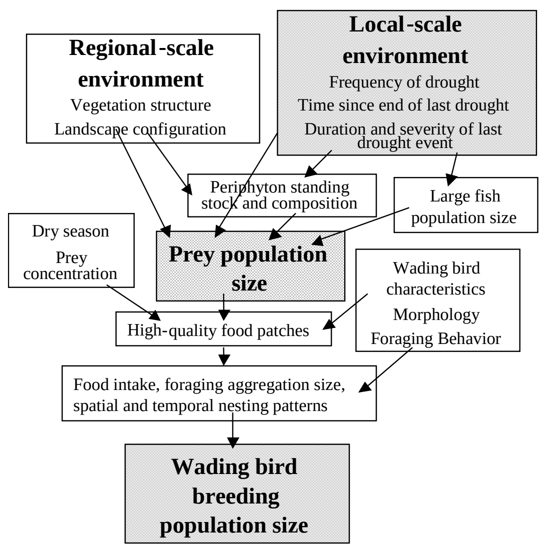

Figure 1. Conceptual model illustrating the linkage of environmental drivers controlled by managers to aquatic consumer performance measures, and their linkage to wading bird population dynamics.

We have a relatively good understanding of the linkage of hydrological dynamics to aquatic fauna, making target setting based on idealized and realized hydrological management feasible. Exactly how regional patterns of aquatic fauna production is linked to nesting success of our apex species (wading birds in this case) is not as well established, but is the target of ongoing research. In this assessment, we ask if hydrological operations are producing the expected spatial and temporal patterns of aquatic consumers given rainfall and desired hydrological variation. Future assessments should identify targets for aquatic consumers tied to wading bird productivity, which in turn will permit identification and resolution of discrepancies, if they exist, between goals for hydrological management and restoration of animals that are highly valued by society.

Aquatic Fauna Performance Measures

We have identified four patterns of population-level responses to marsh drying in wading bird prey species of the Everglades. We believe that these responses represent different life-history strategies for coping with drought stress (DeAngelis et al. 2005) and have selected indicator species to represent groups of species with similar strategies. Three patterns are found in fish and grass shrimp (Trexler et al. 2001; Ruetz et al. 2005; Trexler et al. 2005; DeAngelis et al. 2005). These are: 1) slow recovery following marsh drying, possibly taking years to regain pre-drought density (typical of bluefin killifish *Lucania goodei*, least killifish *Heterandria formosa*, grass shrimp *Palaemonetes paludosus*); 2) maximum density attained soon after drying events and lower densities a year or longer after drying (typical of flagfish *Jordanella floridae* and marsh killifish *Fundulus confluentus*); and 3) a moderate relationship between density and time since drying at a regional site, presumably because of medium-scale movement (10's of kms) from areas that are drying (unique in the Everglades to eastern mosquitofish *Gambusia holbrooki*). A fourth relationship is seen in crayfish and probably differs from fish and grass shrimp parameters because of their ability to burrow and tolerate moderate amounts of marsh drying (Dorn and Trexler 2007). Everglades crayfish (*Procambarus alleni*) display little or no relationship between local time since flooding and density, but regional drying and average water depth over the past 6 months do explain moderate amounts of variability in their density (Dorn and Trexler 2007). Everglades crayfish are more abundant when recent water depths have been shallow or drying is frequent, and slough crayfish (*Procambarus fallax*) are more abundant in deeper water and longer-hydroperiod sites (Dorn and Trexler 2007). We are not currently using slough crayfish as a performance measure because no clear relationship has been identified between their numbers and hydrological parameters; their density may be most strongly affected by biotic interactions indirectly tied to hydrology (Dorn and Trexler, unpublished data). We have selected bluefin killifish, flagfish, eastern mosquitofish, and Everglades crayfish to make assessments because they represent the four life-history strategies and are frequent enough in our samples to provide adequate statistical power to detect effects we believe are important. As a fifth performance measure, we use the summed density of all fish species. This is an index of fish productivity and is positively correlated with time since a site reflooded after the most recent drying event; density is better correlated to hydrological parameters than biomass. A sixth performance measure used is the percent of fish that are non-native. Though the direct impacts of non-native fishes on Everglades ecosystem function are not well understood, their presence conflicts with management criteria for Everglades National Park and there is ongoing concern about their potential impacts on native taxa.

Hydrological Goals used for this Assessment

We used the same hydrological goals for this assessment as employed in the IOP Project Evaluation Report (SFNRC 2005). Those goals were to match the relationship between rainfall and water-depth fluctuation observed in the period between 1993 and 1999. These years included several with very high regional rainfall (1996, 1997), and some with relatively lower levels (1993 and 1998). The high rainfall years may have provided high water levels similar to those found historically and prior to implementation of water drainage programs in the Everglades. Additional hydrological scenarios should be used to construct performance measure targets in future assessments. The Natural System Model (NSM) is a natural choice for such a scenario, as are hydrological models used for evaluation exercises and planning, for example the D13R (USACOE 1999). Any applicable hydrological model can be used, as long as it is run with rainfall data that includes the years being assessed. Some preliminary examination of NSM output run through 2005 and provided on an experimental basis by staff of the South Florida Water Management District indicates that it predicted higher water levels and less frequent marsh drying than simulated in the 1993-1999 goals used in this report. Thus, impacts identified in this report are probably conservative when compared to current thinking about hydrology of the historical Everglades.

Assessment Methods

Overview of Modeling Strategy- We used the years 1993 through 1999 as a baseline to establish phenomenological relationships between water depth measured at our study sites and rainfall from gauges across three regions: Shark River Slough (SRS), Taylor Slough (TSL), and Water Conservation Areas 3A and 3B (WCA3A and WCA3B). We then used these relationships and the observed rainfall in years 2000 through 2006 to project water depths for those years. The resulting projections simulate water depths expected if no change in water management occurred following the baseline period. We used these hydrological projections to forecast performance measures (PM) at each monitoring site. Finally, the PM forecasts were used as targets for comparison to observed values for each PM in order to assess how implementation of new water management operations may have affected aquatic-system function. The following sections give a methodological overview of the modeling process, and present key findings. Our modeling procedure is divided into three different sections to illustrate the steps that we went through to determine our final impact assessments.

Modeling Methodology and Key Findings

Hydrological Models- We used daily rainfall data to derive a statistical relationship between rainfall and surface-water depth at a given long-term monitoring plot in the goal period (November 1,1993 – November 1, 1999). This period was modified from assessments requested by personnel from the South Florida Natural Resources Center and corresponds to a range of relatively dry and wet years based on rainfall records for the southern Everglades region (Fig. 2). We generated several different rainfall parameters corresponding to the cumulative amount of rainfall over a given period of time. To select parameters to predict field water depths we used two criteria: 1) cross-validation predicted residual sums of squares (CVPRESS) and 2) proportion of times we correctly classified observed marsh drying events (classification rate). Marsh drying events are particularly important for this modeling effort because drying (defined here as water depth less than 5 cm) represents a threshold for many aquatic fauna, especially fish. Once our final hydrological model was selected, we used its parameters to predict surface-water depth in the assessment period (January 1, 2000 through December 31, 2006). This simulates surface water depths if water management operations of the 'target setting period' were maintained during the 'assessment period.'

Our models predicted wetter marshes and fewer drying events in many areas south of the Tamiami Trail than were observed during the assessment period; results were mixed for Water Conservation Areas 3A and 3B (Fig. 3). Additionally, we were able to predict the majority of drying events in the goal period, but the same model using rainfall in the assessment period predicted less than half of the drying events observed. These results indicate that water management operations in the assessment period were responsible for the change in surface water when compared to the target period, not differences in rainfall. In the following sections of this report, our assessment of consumer performance measures illustrates the impact of this difference on aquatic consumer density.

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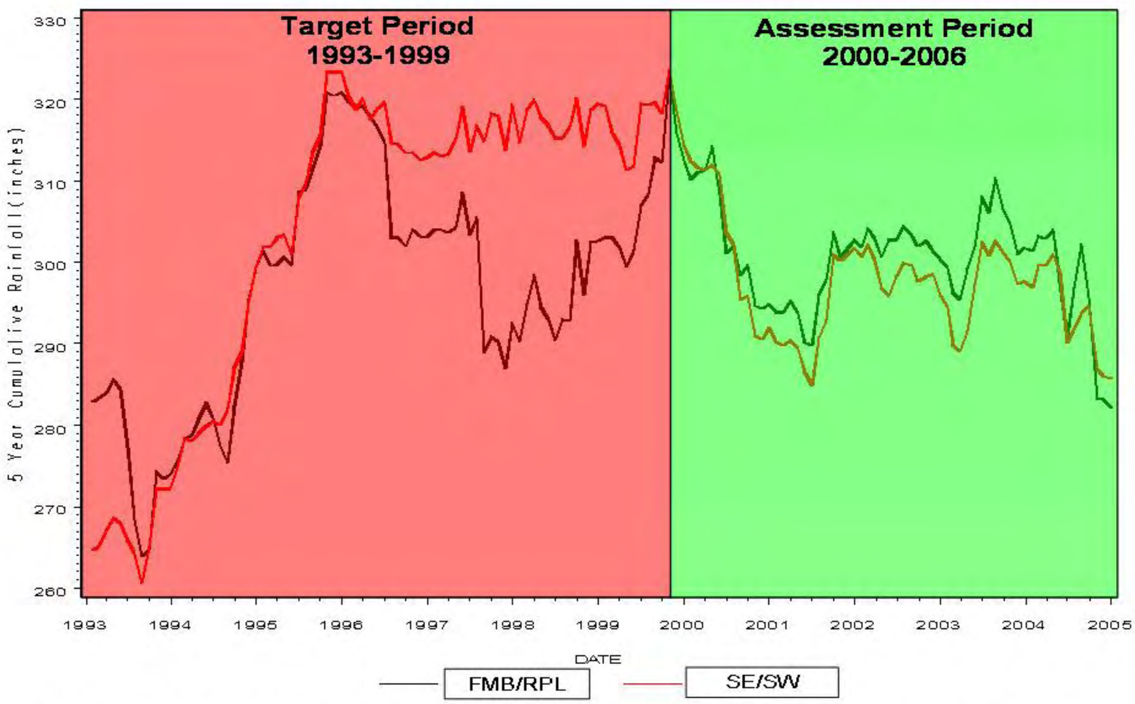

Fig. 2. Time series for the daily 5-year cumulative sum of rainfall (inches) from 1993 to 2006 extract from a time series of running cumulative values starting in 1905 (not shown). The black line corresponds to the average of rainfall record from FMB and RPL, and the red line corresponds to the average of rainfall record from SE and SW. Hydrological goals were set from the time highlighted in red (1993-1999), and assessments were made for the years highlighted in green (2000-2006).

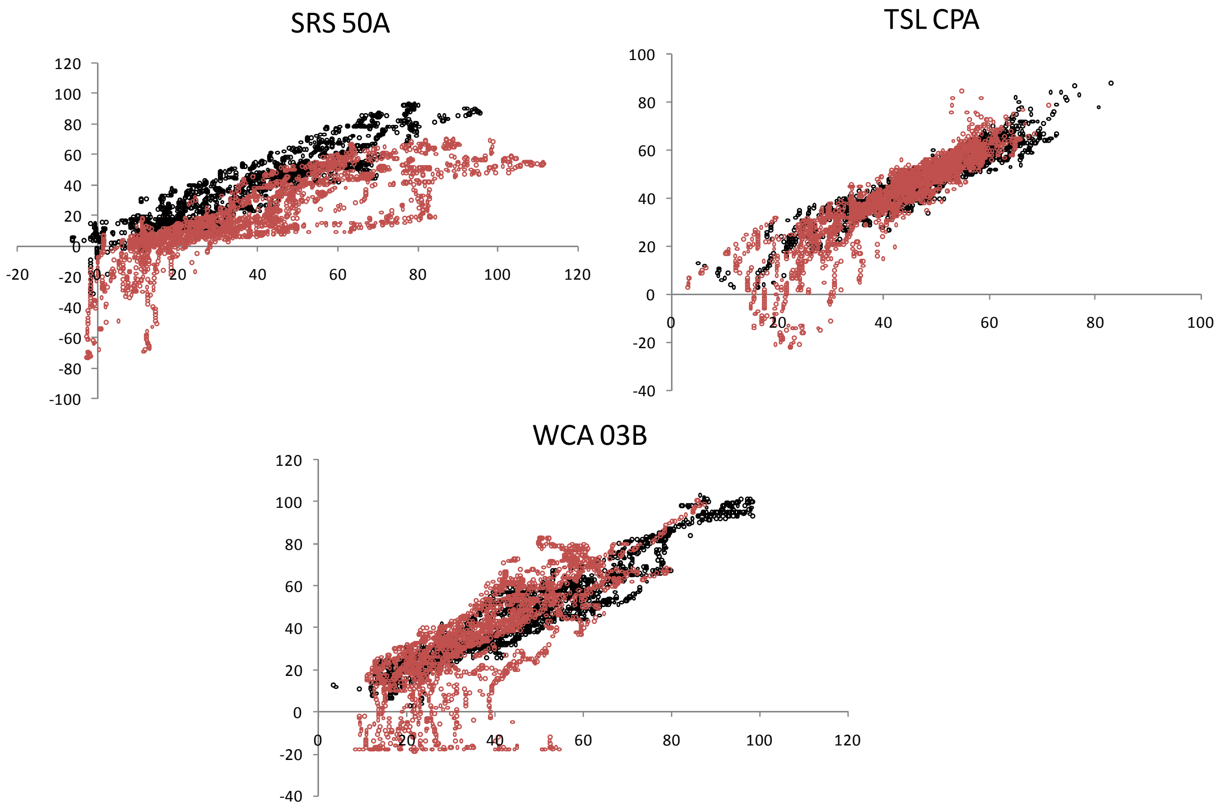

Fig. 3. Hydrological model predictions in the target (black) and assessment (red) periods. Observed depth is plotted on the y-axes and predicted depth is on the x-axes for three exemplary monitoring plots (SRS50A is in Shark River Slough near Shark Valley; TS CPA is in Taylor Slough at Craighead Pond; WCA 03B is in western WCA 3A, south of the L28 Interceptor Canal). Below ground water depths are not well predicted, possibly because of inaccuracies in the observed data.

Ecological Data and Models- Monitoring programs for aquatic consumers focus on small aquatic animals (fish < 8-cm standard length; fish and macroinvertebrates routinely retained on 2-mm mesh sieves) and are conducted in the Everglades by use of a 1-m^2 throw trap (Kushlan 1981; Loftus and Eklund 1994). Several papers support use of this technique based on comparative evaluations with alternative methods that examined bias and efficiency in sampling fishes (Chick et al. 1992; Jordan et al. 1997) and macroinvertebrates (Turner and Trexler 1997; Dorn et al. 2005) in Everglades marshes. Wolski et al. (2004) found little impact of long-term visitation that accompanies throw-trap sampling at fixed sites in the Everglades, further justifying the technique's use for monitoring. A history of PM development and fish monitoring in Everglades National Park is provided in Trexler et al. (2003). Data used for this assessment were obtained from long-term monitoring of the Modified Water Delivery Program (Fig. 4). Future assessments will use data from the Monitoring and Assessment Program of CERP; a brief discussion and assessment using those data for 2005 are given at the end of this report.

We modeled five different performance measures: total fish density (all species of fish summed), eastern mosquitofish (*Gambusia holbrooki*), flagfish (*Jordanella floridae*), and bluefin killifish (*Lucania goodei*), and Everglades crayfish (*Procambarus alleni*). Past work has demonstrated that these fish are representative of the variety of life-history responses to drying events (Trexler et al. 2005; DeAngelis et al. 2005). Flagfish and eastern mosquitofish typically recover quickly from marsh drying, while bluefin killifish recover more slowly (DeAngelis et al. 2005). Additionally the Everglades crayfish has been shown to survive marsh drying conditions and is typical of short-hydroperiod marshes in the southern Everglades (Hendrix and Loftus 2000; Dorn and Trexler, unpublished data). We analyzed these data using hydrological parameters that estimate the time passed since re-flooding from most recent drying event. We define drying as water depth dropping below 5 cm and flooding as when previously low water levels rise above 5 cm. To account for ecological responses driven by hydrology operating at different spatial scales, we created three different hydrological parameters: local days since flooding (LDSF), local days since flooding adjusted for regional drying (ADSF), and regional days since flooding (RDSF). We used linear regression to capture patterns of recovery following marsh flooding and evaluated our models using Akaike's Information Criterion (AIC) to select a preferred model from a hierarchy of models. Our final models generally described the data well, although fit varied across species and regions.

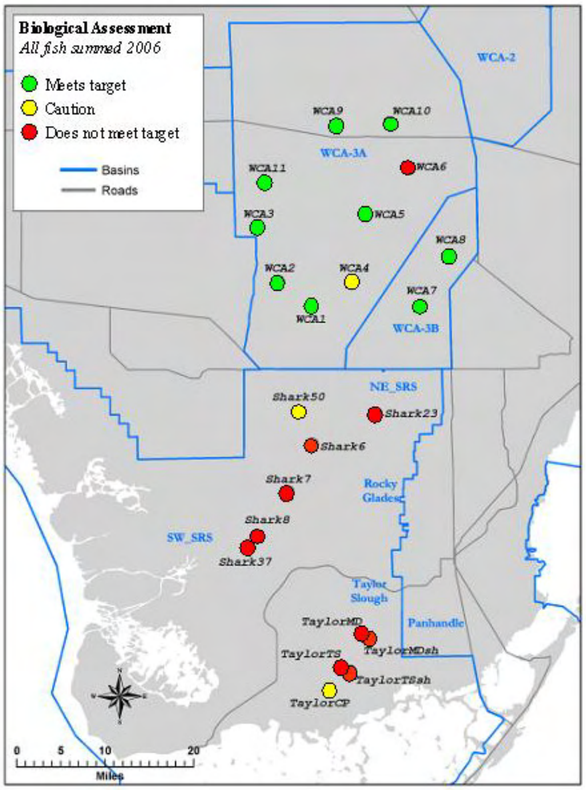

Figure 4. Map of long-term monitoring sites used for this assessment.

Consistent with previous studies, we found that bluefin killifish and total fish typically increased in density following marsh flooding (Fig. 5). In contrast, flagfish and eastern mosquitofish decreased with time following marsh flooding at some sites, though not at the same rate or to the same extent; eastern mosquitofish are almost always much more abundant that flagfish (Fig. 5). Our models were also consistent with published results indicating that Everglades crayfish tend to decrease in density the longer a marsh is inundated. In fact, this species is extremely rare in WCA 3A and 3B, most likely because there are several areas that in those regions that rarely dry. Everglades crayfish could not be assessed in these water conservation areas.

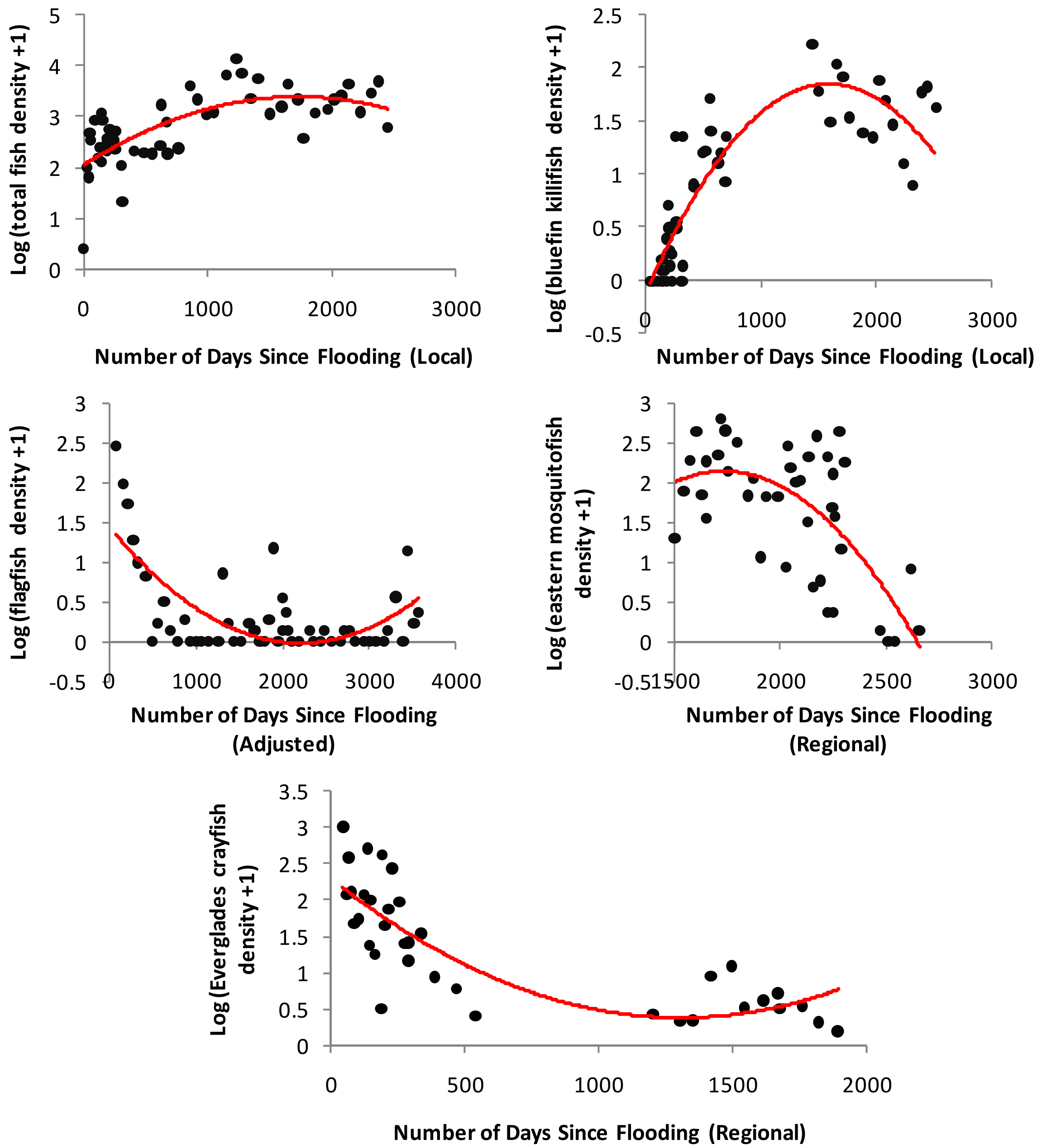

Fig. 5. These graphs are quadratic regressions illustrating our model fits for each fish species at specific sampling sites. Starting in the upper left and preceding right the sites are as follows: SRS 08A, TSL CPA, WCA 02B, WCA 05 B, and TSL MDD.

Water Depth and Ecological Synthesis Models- Using the predicted data from our hydrological model and the parameter estimates from our ecological model, we projected fish densities into the assessment period. This gives us an estimate of aquatic consumer densities if water management were consistent with the goals and targets as defined for this assessment. We found many instances where there was substantial deviation of the observed fish density when compared to predictions by the hydrological goals (Fig. 6). This suggests that the deviation in the relationship between rainfall and water depth in the assessment period, translated to a change in aquatic fauna densities that resulted from water management activities. In the next section we describe how to interpret the results in Figure 4 and summarize the findings in regional assessments.

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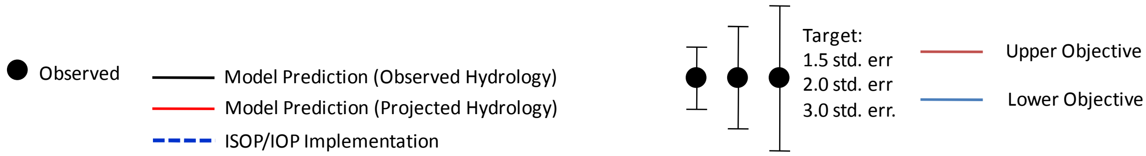
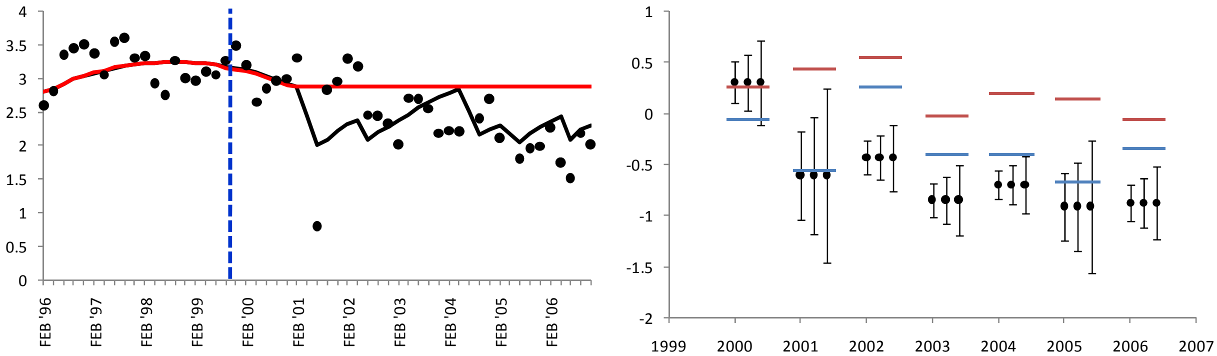
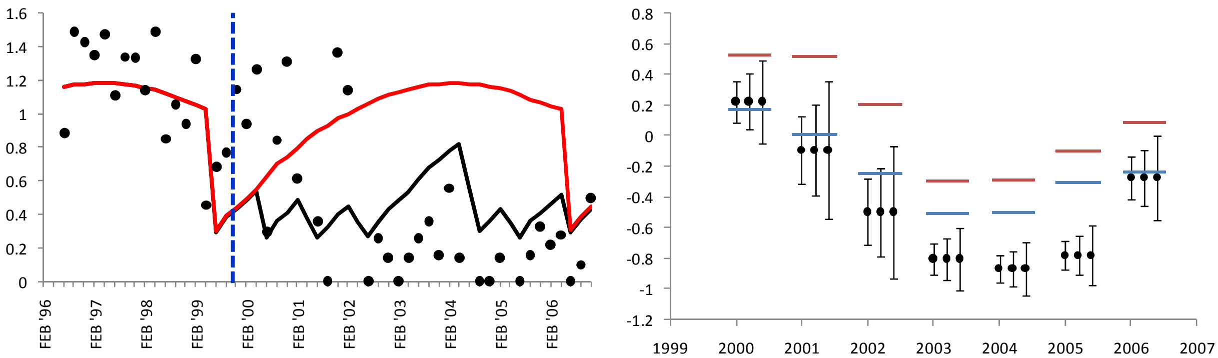


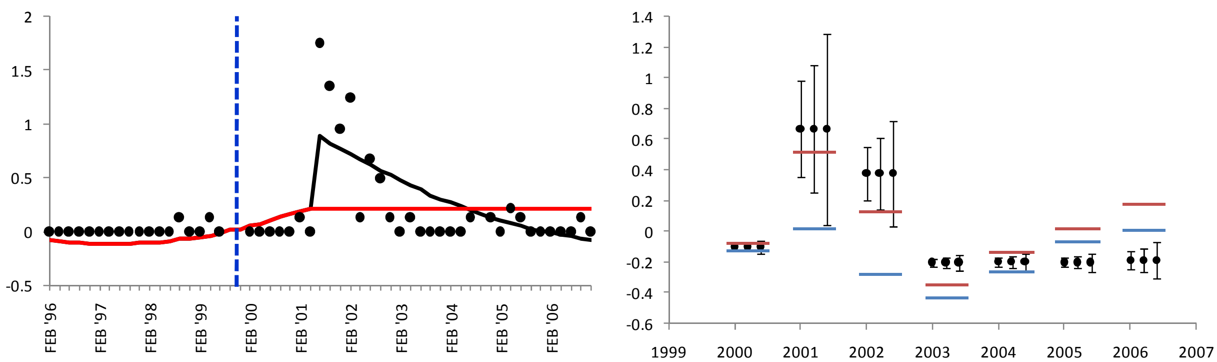
Total Fish



Eastern Mosquitofish



Flagfish



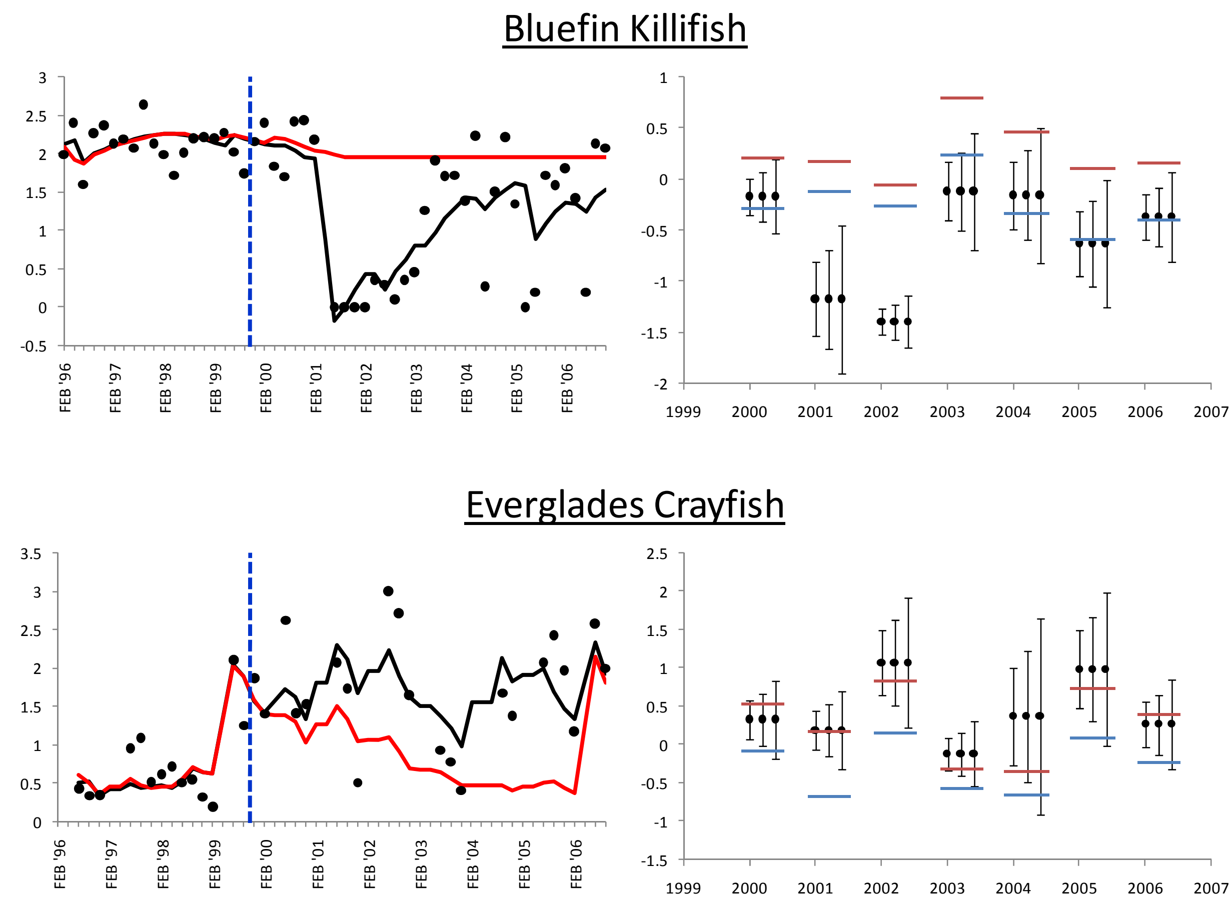


Fig. 6. Illustrations from selected sites of observed time series data and model predictions (left) and the objective limits and targets (right) for five performance measures. There are 64 plots in our database, so the results are aggregated for each performance measure to yield a robust regional assessment using methods discussed in the next section. Upper and lower objective limits are the 95% confidence limits from our ecological targets (this captures uncertainty in fit of our assessment model); the target confidence intervals are derived from deviations for the observed and target data on an annual basis (there are seven groups for the seven annual assessments). The widths of those intervals (1.5, 2.0, and 3.0 standard errors) correspond to different criteria for assessment discussed in the next section. Impacts are evaluated based on overlap of intervals with the upper and lower bounds of uncertainty for fit of the model generating assessment goals **(i.e., if the black bars are outside the blue and red bars, we judge that as a negative impact; if they overlap, we assess based on the amount of overlap)**.

Assessing Impacts to Aquatic Consumers.- In order to assess if management is 'getting the water right', we identified impacts based on deviation between our observed values for each performance measure and goals for hydrological management. We identified two primary sources of uncertainty in this process: uncertainty in the fit of our hydrological and ecological models; and uncertainty in our comparison of sampling data to the targets. To account for uncertainty in our modeling, including systematic (lack-of-fit) and random variability in the models, we estimated an objective interval (mean +/- 2 standard errors) for use as upper and lower limits our targets (blue and red bars in Fig. 4). Assessing the magnitude in deviation between our observed data and ecological targets requires defining an 'impact' based on the magnitude of deviation. We did this by use of estimates of the standard error of deviations between observed and target values calculated on an annual basis (black confidence intervals in Fig. 4). Interpretation of these confidence intervals was based on criteria from Decision Theory used to evaluate time series of data on industrial processes.

We defined two classes of impact: individual years with extreme deviations (type A); and runs of consistent deviations from the ecological targets (types B and C). We followed criteria from Allen et. al (1997) using Shewhart Control Chart Theory and define different criteria for defining an impact:

Type A: one year at least three standard errors above the upper limit of the objective interval, or three standard errors below the lower limit of the objective interval.

Type B: two out of three consecutive years at least two standard errors above the upper limit of the objective interval, or two standard errors below the lower limit of the objective interval.

Type C: four out of five consecutive years with at least 1.5 standard errors above the upper limit of the objective interval, or 1.5 standard errors below the lower limit of the objective interval.

This method ensures that we take into account any lack of fit of the original model to the data when assigning an impact, yielding conservative estimates of impacts that are coded as red stoplights (i.e., we have attempted to minimize misclassifying areas without impacts by setting a high standard to assign red stoplights). In contrast, we assign yellow stoplights more liberally because they are simply indicative of sites deserving additional attention (i.e., we have attempted to minimize misclassifying impacted areas as meeting targets by assigning yellow lights with less rigor; see criteria below).

We selected several monitoring sites to illustrate typical patterns of impact for each PM (Fig. 6). In these time-series graphs (Fig. 6, left panels), we capture hydrological variation well in the target setting (or baseline) period. Following 2000, it is clear that our predictions based on the observed hydrology deviate dramatically from our predictions based on the projected hydrology. The graphs on the right panels of Figure 6 illustrate the objective upper and lower limits of the targets with the three confidence intervals representing our three criteria for assessing impacts. These graphs show that, for the plots reported in this figure, we tended to predict more total fish, eastern mosquitofish and bluefin killifish based on our projected hydrology than were observed. At these sites the patterns were relatively consistent with the mean typically falling outside the objective limits, and several instances where the 3 standard error interval falls below the objective limits. These patterns are typical of Shark River Slough and Taylor Slough. Results for Water Conservation Area 3A are more complex, with impacts depending on the species assessed and location within the landscape. Note that we observed more flagfish and Everglades crayfish than predicted, as expected with drier conditions.

We use assigned stoplights at the regional level (Shark River Slough, Taylor Slough, Water Conservation Area 3A, Water Conservation Area 3B) to communicate the state of aquatic communities in each year beginning in 2000 and ending in 2006 (Table 1). Red stoplights indicate that there is an impact and correspond to Type A, Type B, and Type C impacts. Yellow lights indicate caution and correspond to years where our target is 1.5 standard errors above or below our objective. Finally, green stoplights correspond to years where there is no impact, and the target falls within 1.5 standard errors of the objective. To obtain a regional assessment for each species, we ranked the stoplights: 1=Green, 2=Yellow, and 3=Red, and took the means of the yearly ranks for sites within regions; we rounded to the nearest integer to get a stoplight estimate for each region in each year. Impacts (red lights) were more common in assessments after 2002 because we were able to apply time series criteria with three years of data, and had three ways to detect impacts by 2004. Our ability to assign impacts with confidence increased as more years of data were available, with threshold points after 3 and 5 years because of cumulative information available to interpret findings.

Shark River Slough and Taylor Slough yielded the most striking examples of failure to meet our *a priori* targets (Table 1), with fewer fish and more Everglades crayfish than expected (Table 2). These patterns were most apparent for total fish and bluefin killifish, while eastern mosquitofish tended to yield weaker responses. The only two impacts for flagfish in Shark River Slough indicated that we observed more fish than predicted by our model. In Taylor Slough, our hydrological models described flagfish population dynamics poorly at most study plots, though they were collected, so we were unable to make an assessment. All impacts for Everglades crayfish resulted from observing more specimens than were predicted for the targets (Tables 1 and 2). This suggests that when marshes are drier overall, Everglades crayfish increase their range and abundance. Overall, patterns of impact for bluefin killifish and total fish in WCA 3A were more complex. There were fewer impacts in WCA 3A than in Everglades National Park, and the yearly regional assessments were generally all green (total fish), or a mixture of green, yellow and red, with the current status of bluefin killifish either yellow (WCA 3A) or green (WCA 3B). For flagfish and eastern mosquitofish, most of the impacts resulted because we observed higher density than expected. We believe this resulted from movement of fish from western areas of WCA 3A that dried, and concentrated at sites in the southeast. The current status for either of these species does not indicate an impact. Everglades crayfish is extremely rare in these water conservation areas, so we were unable to make an assessment. Water Conservation Area 3B revealed few deviations from expectations, consistent with its status of isolation from other parts of the ecosystem and limited capacity for impacts from operations (though ground water seepage is a potential mechanism to transfer management impacts to this region from upstream).

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Table 1. Regional stoplight summary of all PMs. Current status refers to 2006.

Performance Measure	2000	2001	2002	2003	2004	2005	Current status
Shark River Slough							
eastern mosquitofish	green	green	yellow	red	red	yellow	yellow
flagfish	green	green	green	yellow	green	green	green
bluefin killifish	green	yellow	yellow	red	yellow	yellow	yellow
total fish	green	green	red	red	red	red	red
Everglades crayfish	green	yellow	green	yellow	red	yellow	yellow
Non-native fishes	yellow	yellow	yellow	yellow	yellow	yellow	yellow
Taylor Slough							
eastern mosquitofish	green	green	green	yellow	yellow	yellow	yellow
flagfish	white	white	white	white	white	white	white
bluefin killifish	green	yellow	red	red	red	red	red
total fish	green	green	yellow	yellow	red	red	red
Everglades crayfish	green	green	green	yellow	red	yellow	yellow
Non-native fishes	yellow	yellow	yellow	yellow	yellow	yellow	yellow
Water Convservation Area 3A							
eastern mosquitofish	green	green	yellow	yellow	yellow	green	green
flagfish	green	green	green	green	green	yellow	yellow
bluefin killifish	green	green	yellow	yellow	yellow	yellow	yellow
total fish	green	green	green	green	green	green	green
Non-native fishes	yellow	yellow	yellow	yellow	yellow	yellow	yellow
Water Convservation							

Area 3B							
eastern mosquitofish	🟢	🟢	🔴	🔴	🔴	🔴	🟡
flagfish	🟢	🟡	🟡	🟡	🟡	🟡	🟡
bluefin killifish	🟢	🔴	🔴	🟡	🟢	🟢	🟢
total fish	🟢	🟢	🟢	🟢	🟢	🟢	🟢
Non-native fishes	🟡	🟡	🟡	🟡	🟡	🟡	🟡

Table 2. Summary of the number of site-level impacts (red stoplights) in a given region as a result of observing fewer animals than expected, more animals than expected, or a mixture of the two. Counts of the number of red stoplights between 2000 and 2006 are listed, reported by the direction of deviations.

Region	Species	More than Expected	Fewer than Expected	Mixture
SRS	Everglades crayfish	16	0	0
SRS	eastern mosquitofish	0	7	0
SRS	flagfish	2	0	0
SRS	bluefin killifish	0	20	1
SRS	total fish	0	24	0
TSL	Everglades crayfish	13	0	0
TSL	eastern mosquitofish	0	10	0
TSL	flagfish	0	0	0
TSL	bluefin killifish	0	22	0
TSL	total fish	0	17	0
WCA3A	eastern mosquitofish	7	0	0
WCA3A	flagfish	3	5	0
WCA3A	bluefin killifish	0	10	0
WCA3A	total fish	0	2	0
WCA3B	eastern mosquitofish	9	0	0
WCA3B	flagfish	6	0	0
WCA3B	bluefin killifish	0	4	0
WCA3B	total fish	0	0	0

Non-native Fishes. In the absence of any ecological data on threshold densities for biological impacts of non-native fishes on aquatic ecosystem function of the Everglades, we used a criterion of relative abundance to assign annual impacts. If non-native taxa comprised at least 2% of all fishes collected in a year at a monitoring site, we assigned a value of caution (yellow) to the site for that year. We also considered evidence of a trend of increasing absolute abundance as a source of concern. Based on unpublished data on Mayan cichlid (*Cichlassoma uropthalmus*) density in the Southern Everglades, we assigned a value of exceeds targets (red) if the summed density of non-native fishes (including Mayan cichlids) exceeded 10%. We also assigned 'exceeds target' if the relative abundance of non-native species exceeded 5% for three or more years in a row.

These targets are arbitrary in the absence of much-needed experimental studies of biotic interactions of these taxa indicating detrimental effects on native taxa or other measures of ecosystem function. At present, one or more non-native fish species can be considered present in all areas of the Everglades, and eradication is not currently possible. For this reason, we set a lower boundary greater than zero, though management criteria for the Everglades National Park would require this. Assessing non-native species requires careful consideration because of known gear bias in fish collections, and impacts of sample size in estimating population parameters (sample size refers to both the number of samples AND the total number of animals collected). Assessments must be made with consistent methods for comparisons, either across space or through time, and emphasize relative differences. For example, minnow-trap sampling in Everglades marshes by placement of traps on the substrate typically yields a higher relative abundance of non-native taxa than throw-trap sampling. Minnow traps are preferable to throw traps to determine ifspecies of non-native fishes are present in an area, but throw traps are preferable to obtain a quantitative measure of their relative abundance in the community at a location (assuming that other conditions are appropriate for throw-trap sampling, such as vegetation cover and water depth). We anticipate much interest in refining this target for future assessments.

We found that non-native fishes were typically between 2 and 4% of the fishes collected by throw trap at all of our monitoring sites over the 7 years of this assessment (Fig. 7). In Shark River Slough, one site produced more than 10% non-native fishes in 2003, and slightly less in 2004. This monitoring site is adjacent to the Shark Valley tram road and near a borrow pit which appears to serve as a reservoir of non-native taxa. However, these two years were also relatively dry at this site and few fish of any species were collected. In fact, the most non-native specimens by far are collected at Rookery Branch, a site near the mangrove zone and close to the headwater creeks of the Shark River. However, this site is generally productive for fishes, so the high numbers of non-native taxa (mostly Mayan cichlids) remains a relatively small proportion of the community. A similar pattern is seen in Taylor Slough, where a short-hydroperiod site on the edge of the main slough harbored the highest frequency of non-native fishes, but the most specimens were collected at the southern end of the slough in Craighead Pond, and at a site near the Madeira Ditch, an artificial permanent water refuge. There were no trends of increasing (or decreasing) frequency or density of non-native taxa at these monitoring sites. We assigned yellow stoplights throughout for non-native taxa because of their persistent low frequency and uncertainty about their impacts.

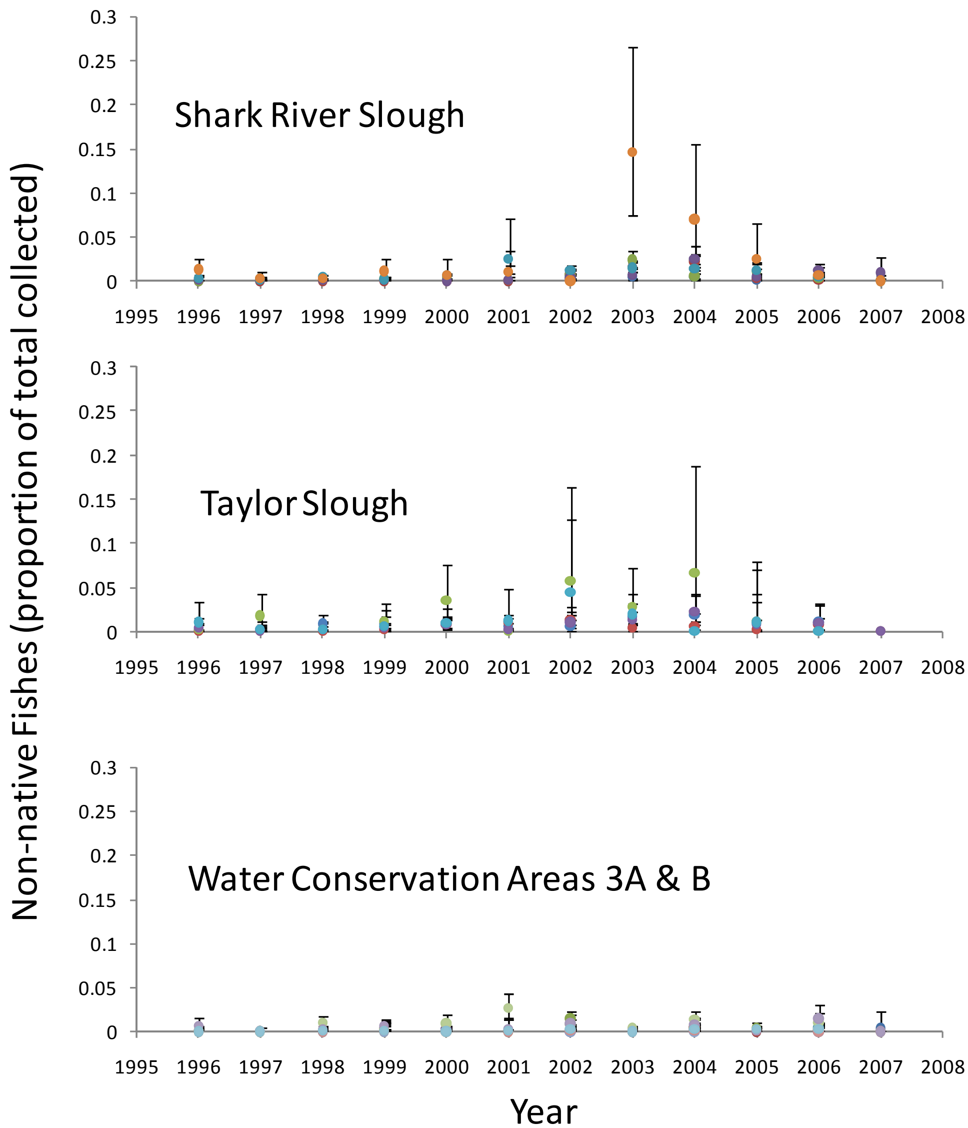

Figure 7. Proportion of non-native fishes collected at each study site in each year, reported separately for regions. Means and 95% confidence intervals were derived by GLM with logit linking function. High values in Shark River Slough and Taylor Slough correspond to site/year combinations when relatively few fish were collected.

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Future Assessments and Lessons

Future assessments should be made system-wide using data collected for CERP-MAP. At present, three system-wide surveys have been completed through wet-season sampling (September through November). Assessments reported here indicate that at least three years of data are needed to implement a robust analysis accounting for trends, and five years is best. An impediment to applying dynamic targets in assessment with the CERP-MAP data is in the lack of landscape-scale hydrological targets for evaluating the monitoring data. Evaluating impacts of hydrological operations requires assessments that account for rainfall patterns, particularly for fishes.

We created a preliminary assessment for 2005 using CERP-MAP data and the performance measure total fish (density of all fish summed). To accomplish this, we used a trial version of the Natural System Model that has been run using rainfall data through the end of 2005. This permitted calculation of the days since last re-flooding parameter for modeling total fish density, in a similar manner used elsewhere in this report. *Note: Use of NSM in this case is on a trial basis only as this version has not undergone full QA/QC.* We assigned red stoplights by observed fish density at least three standard errors beyond the target and yellow when observations were between 2 and 3 standard errors from expected. For these data, we used the relationship between fish density and days since the marsh last re-flooded estimate from Shark River Slough as our target. The observed relationship deviates significantly between Shark River Slough, Taylor Slough, and Water Conservation 3A, probably because of different histories of drying and access to permanently inundated refuges (both of which affect patterns of predation, in addition to direct effects of mortality). We opted to use the Shark River Slough relationship as an ecosystem-wide target because it area has less impact of artificial deep-water refuges compared to Water Conservation Area 3A, but has not experienced repeated slough-wide drying as in Taylor Slough.

Most of our collections deviated markedly from NSM-derived expectations and garnered a red stoplight (Fig. 8). The direction of deviations is of interest, with fewer fish than expected in the south (Shark River Slough and Taylor Slough) and more in the north (WCA-2A and Loxahatchee National Wildlife Refuge); there is a mixture of directions in Water Conservation Area 3A. Some of the results are odd, such as in WCA 3A, where we generally caught more fish than were expected at four out of six sampling points. We will not expend more space evaluating this graph because of the tentative nature of the hydrological model results. However, this illustrates that assessments of aquatic consumers with application of dynamics targets is feasible at the ecosystem scale once appropriate hydrological models are made available.

Additional ecological studies are needed on impacts of non-native fish species to develop targets better linked to ecological impacts, if impacts are documented. Experimental studies are of special importance because impacts cannot be effectively assessed by abundance alone; abundant non-native species could be benign and rare species could act through indirect routes to alter feeding opportunities for wading birds or alligators.

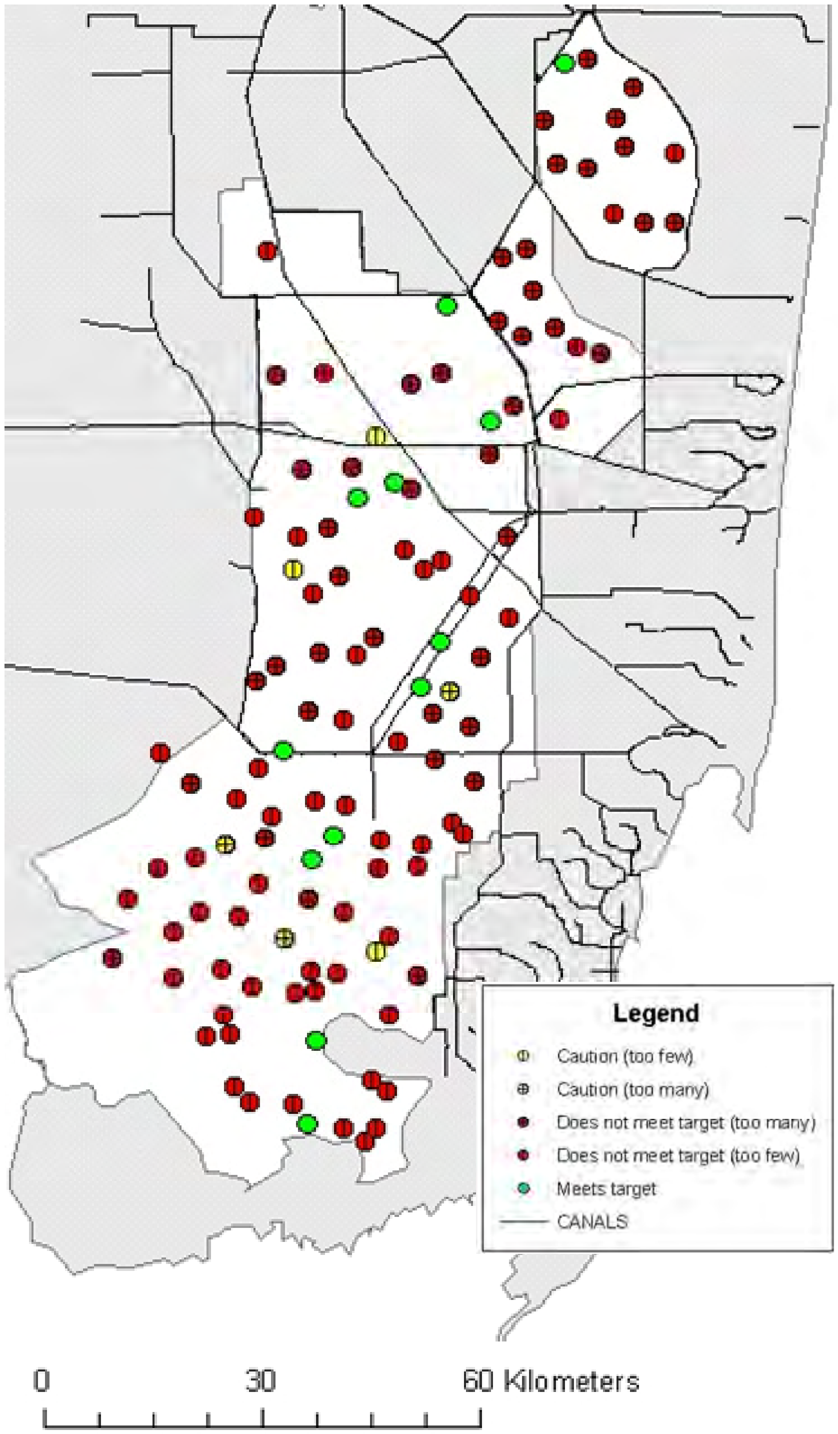

Figure 8. Map of results of assessment using total fish density from 2005 CERP MAP wet-season collections. Targets were derived from an experimental version of NSM used for illustrative purposes.

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SOUTH FLORIDA WADING BIRD REPORT

Volume 13 | Mark I. Cook and Heidi K. Herring, Editors | October 2007

SYSTEM-WIDE SUMMARY

Below average rainfall through the 2006 wet season resulted in the early onset of the 2007 dry season over much of South Florida. Continued below average precipitation led to good recession rates, only minor reversals in stage, but water levels that were generally lower on average than at any time since the drought of 2001. By the time of peak nesting activity in March/April, some areas were too dry for foraging or colony formation.

The estimated number of wading bird nests in South Florida in water year 2007 was 37,623 (excluding Cattle Egrets, which are not dependent on wetlands). This is a 31% decrease relative to last year's successful season, 46% less than the 68,750 nests of 2002, which was the best nesting year on record in South Florida since the 1940s, and 19% less than the average of the last six years. Note that this year's total count is a slightly conservative estimate. Surveys were not conducted this year at J. N. 'Ding' Darling National Wildlife Refuge Complex, which usually adds approx. 1000+ nests to the system-wide total. Also, ground survey coverage for the WCAs was relatively limited this year and may underestimate the total count (see Regional Nesting Reports section).

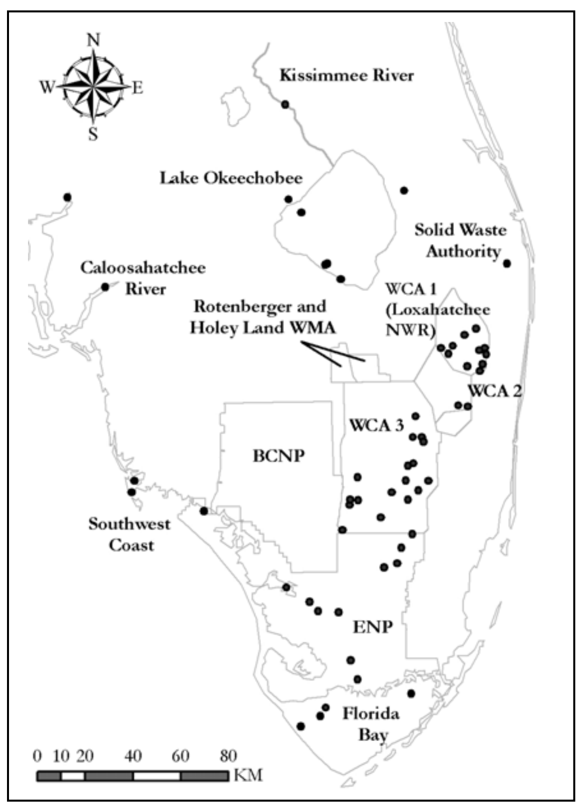
Systematic nest survey coverage has been expanded in recent years to include Lake Okeechobee and the recently restored section of the Kissimmee River floodplain. In 2007, nesting effort in these areas was relatively poor: 774 nests were counted at Lake Okeechobee and only one nest was found on the Kissimmee River floodplain. This is a marked decline from the 11,447 nests found at the two sites in 2006. As with other South Florida wetland systems, both areas were characterized by below average stage during the preceding wet season and below average winter rainfall. Note that the total for these areas is not included in the system-wide total.

INSIDE THIS ISSUE

This year, all species of wading birds experienced reduced nesting effort relative to 2006 but the most extreme declines were for Wood Storks (79%), Tricolored Herons (69%) and Snowy Egrets (96%). Number of Spoonbill nests was 19% below the mean annual average since 1984, and number of White Ibis nests was 16% lower than 2006 but similar to the annual average of the past ten years. In general, 2007 was a poor wading bird breeding season in terms of nesting effort compared to the past ten years and pre-drainage years, but successful relative to the period 1960-1998.

As usual, nesting effort in the Everglades was not uniformly distributed among regions. WCA-3 and WCA-1 supported the most nests (47% and 44% respectively) whereas ENP supported only 9% of nests. This spatial distribution of nesting represents a change from recent years in that this year more nesting occurred in WCA-1 at the expense of nesting in WCA-3. ENP historically supported the largest number of nests in the system and a goal of CERP is to increase the proportion of birds nesting in the traditional estuarine "rookeries" downstream of Shark Slough.

Locations of wading bird colonies with ≥ 50 nests in South Florida, 2007.



Nesting effort in the estuaries has increased over recent years but this year the southern colonies supported only minimum nesting. Another pattern in recent years has been for a large proportion of nests in South Florida to be concentrated in a single large colony (Alley North) located in northeast WCA-3A. This year, Alley North and its adjacent marsh dried prior to breeding and nesting was not initiated at the colony. The loss of this important colony appeared to be offset slightly by increased nesting activity in WCA-1 and by the expansion of two extant colonies proximate to Alley North.

Generally, nesting was not successful for most species. Some of the nest failure can be attributed to the dry condition which led to poor foraging (see Cook and Herring, this issue) and possibly to increased mammalian predation when colonies dried completely. Despite the dry conditions, rain-driven reversal events in March and April also induced moderate nest failure particularly for nests containing eggs or very young chicks. Wood Stork nesting success was particularly poor in 2007. At Paurotis Pond, all nests had failed by late May and at Tamiami West, only about 40 of 90 pairs appeared to fledge young. However, successful nesting was evident at large colonies of both Great Egrets (e.g., Vacation, Cypress City), and White Ibises (6th Bridge, Lox 73, New Colony 4). Given the nest failure in ENP and WCA 3, the number of nests in 2007 may be a liberal measure of overall reproductive success. The relationship between nest numbers and productivity was more direct in 2006 when high nest numbers were accompanied by good nesting success.

Two of four species-groups (White Ibis and Great Egrets) met the numeric nesting targets proposed by the South Florida Ecosystem Restoration Task Force. Two other targets for the Everglades restoration are an increase in the number of nesting wading birds in the coastal Everglades and a shift in the timing of Wood Stork nesting to earlier in the breeding season. The 2007 nesting year did not show an improvement in the timing of Wood Stork nesting or a general shift of colony locations.

Despite the reduced nesting effort and success, Systematic Reconnaissance Flight surveys (SRF) show that large numbers of birds foraged in the Everglades in 2007: the system-wide total abundance was 26% higher than last year and 48% higher than the average of the past five years. Also different from last year was the temporal distribution of foraging birds. In 2006, bird abundance was consistently high from January to June, whereas this year, numbers were elevated until April but declined dramatically thereafter, possibly due to the dry conditions. On the Kissimmee River floodplain the number of foraging wading birds has increased annually since restoration was completed in 2001 but this year it declined dramatically to pre-restoration levels. Extreme low stages on the Kissimmee floodplain and other wetlands precluded foraging for much of the 2007 dry season and birds from these systems were forced to migrate to longer hydroperiod marshes. This exodus may explain the marked increase in the Everglades population.

The annual nesting response of wading birds helps provide a better understanding of how the Everglades ecosystem functions. Recession rates in 2007 were generally classified as 'good' (see Hydrology section) but stages were generally below average and provided unsuitable foraging conditions over large areas of the system, particularly during the later stages of the breeding season. However, the magnitude of the drought and its effect on wading bird reproduction varied considerably by region. Nesting effort and success were greatest in areas where water levels were relatively high at the start of the breeding season, where it declined at appropriate rates, and where it did not dry completely during chick rearing. Little or no nesting occurred in areas that were too shallow prior to nesting. This year's poor nesting effort and reproductive success in the relatively dry marsh of WCA-3A and the switch in nesting effort to the wetter WCA-1 were almost certainly due to differences in hydrologic patterns. However, dry-season hydrologic conditions do not fully explain reproductive patterns. Water depths in WCA-2 and -3 were optimal for foraging early in the breeding season but compared to recent years these important feeding areas supported only limited numbers of wading birds (D Gawlik and M Cook, pers. obs.).

This disconnect between wading bird foraging and hydrology may be related to aquatic prey production. The annual monitoring of aquatic prey during the seasonal dry-down reveals that prey densities were relatively low in WCAs 2 and 3 in 2007 (D Gawlik pers. com.) and, adults that foraged in these areas had low body condition scores (G. Herring pers. com.). Prey production in WCAs 2 and 3 may have been reduced by the extended 2006 dry season during which surface waters fell below ground level for an extended period, potentially killing much of the prey stock for the 2007 breeding season. By contrast, water levels in WCA-1 remained above ground and subsequent prey densities and wading bird reproductive output were relatively high (M. Cook, unpublished data). Thus, wading bird reproduction is likely tied not only to appropriate dry-season hydrologic conditions (a strong recession and shallow water) which increases prey vulnerability but also to the hydrologic conditions of the preceding wet season which affects prey production. This is supported by the observation that the most successful breeding seasons since pre-drainage were associated with high stages during the preceding wet season and with appropriate dry season recession rates/water depths (i.e., 2002, 2004 and 2006). Years without this combination of conditions had much reduced nesting effort. Our long-term nesting data encompasses many years of variable reproductive effort over a range of hydrologic conditions. It may be large enough now that we can more effectively tease apart the ecological factors affecting the timing, distribution and magnitude of nesting.

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HYDROLOGY 2007

The amount of rain in the Everglades Protection Area for water-year 2007 (May 2006 - April 2007) was only 4-5 inches less than last year. However, this was enough to maintain water levels below regulation for all the WCAs for most of the year. The rainfall and associated stage readings for WY2007 are shown in Table 1 below. All three WCAs saw a 14% reduction in historic rainfall amounts. ENP saw only a 5% reduction in historic rainfall amounts.

In WY2007 most of the rain fell during July and August. July totals ranged between 8.1 inches (in WCA-1) and 11.5 inches (in ENP). August totals ranged between 7.7 inches (in ENP) and 8.7 inches (in WCA-1). For the rest of the year rainfall patterns were rather consistently lower than average and the dry season seemed to come a month or two early. October and November rainfall totaled a mere 1.8 inches across WCA-3A. As shown in the following hydrographs, and as might be expected from a below average rainfall water year, the 2006 hydrologic stage conditions were also below average throughout most of the system.

The following hydropattern figures highlight the average stage changes in each of the WCAs for the last two years in relation to the recent historic averages, flooding tolerances for tree islands, drought tolerances for wetland peat, and recession rates and depths that support both nesting initiation and foraging success by wading birds. These indices were used by the District to facilitate weekly operational discussions and decisions. Tree island flooding tolerances are considered exceeded when depths on the islands are greater than 1 foot for more than 120 days. Drought tolerances are considered exceeded when water levels are greater than 1 foot below ground for more than 30 days, i.e., the criteria for Minimum Flows and Levels in the Everglades. Figures 1A through 1G show the ground elevations in the WCAs as being essentially the same as the threshold for peat conservation. The wading bird nesting period is divided into three simple categories (red, yellow, and green) based upon foraging observations in the Everglades. A red label indicates poor conditions due to recession rates that are too fast (greater than 0.6 foot per week) or too slow (less than 0.04 foot for more than two weeks). A red label is also given when the average depth change for the week is positive rather than negative. A yellow label indicates fair conditions due to a slow recession rate of 0.04 foot for a week or a rapid recession between 0.17 foot and 0.6 foot per week. A green/good label is assigned when water depth decreased between 0.05 foot and 0.16 foot per week. Although these labels are not indicative of an appropriate depth for foraging, they have been useful during high water conditions to highlight recession rates that can lead to good foraging depths toward the end of the dry season (i.e., April and May).

WCA-1

The 2007 water-year for WCA-1 started at a relatively low water condition, but then quickly rose to above average conditions and remained above average until October (Figure 1A). After September, rainfall rates declined significantly and stages quickly went below average and stayed below average for the rest of the water year. This was not necessarily bad for WCA-1 because the upper flooding tolerances for tree islands were never reached and recession rates were excellent for most of the dry season. Last water year (2006), there were a number of large-scale reversals in WCA-1 during March and April (Figure 1A), whereas this year, recession rates during the critical wading bird nesting season (January to June) were steady with only a minor reversal observed in April. Water depths became optimum for foraging in central and southern WCA-1 during April and May. Dry season foraging by wading birds in WCA-1 probably slowed significantly in mid-June when water levels increased by 0.5 ft. Just like last year, WCA-1 had the longest duration of good nesting and foraging periods of any region in the EPA. Just like last year, water levels in WCA-1 were below regulation most of the time, upper tolerances levels were never reached, and recession rates were steady.

Table 1. Average, minimum, and maximum stage (ft NGVD) and total annual rainfall (inches) for water-year 2007 in comparison to historic[1] stage and rainfall. Subtract elevation from stage to calculate average depths.

Area	WY2007 Rainfall	Historic Rainfall	WY2007 Stage Mean (min; max)	Historic Stage Mean (min; max)	Elevation
WCA-1	44.94	51.96	15.99 (14.07; 17.08)	15.58 (10.0;18.38)	15.1
WCA-2	44.94	51.96	11.91 (10.42; 13.97)	12.55 (9.33;15.64)	11.2
WCA-3	44.26	51.37	9.61 (8.4; 11.26)	9.54 (4.78;12.79)	8.2
ENP	52.76	55.22	6.15 (5.45; 6.67)	5.98 (2.01;8.08)	5.1

[1] See Chapter 2 of the 2008 South Florida Environmental Report for a more detailed description of rain, stage, inflows, outflows, and historic databases.

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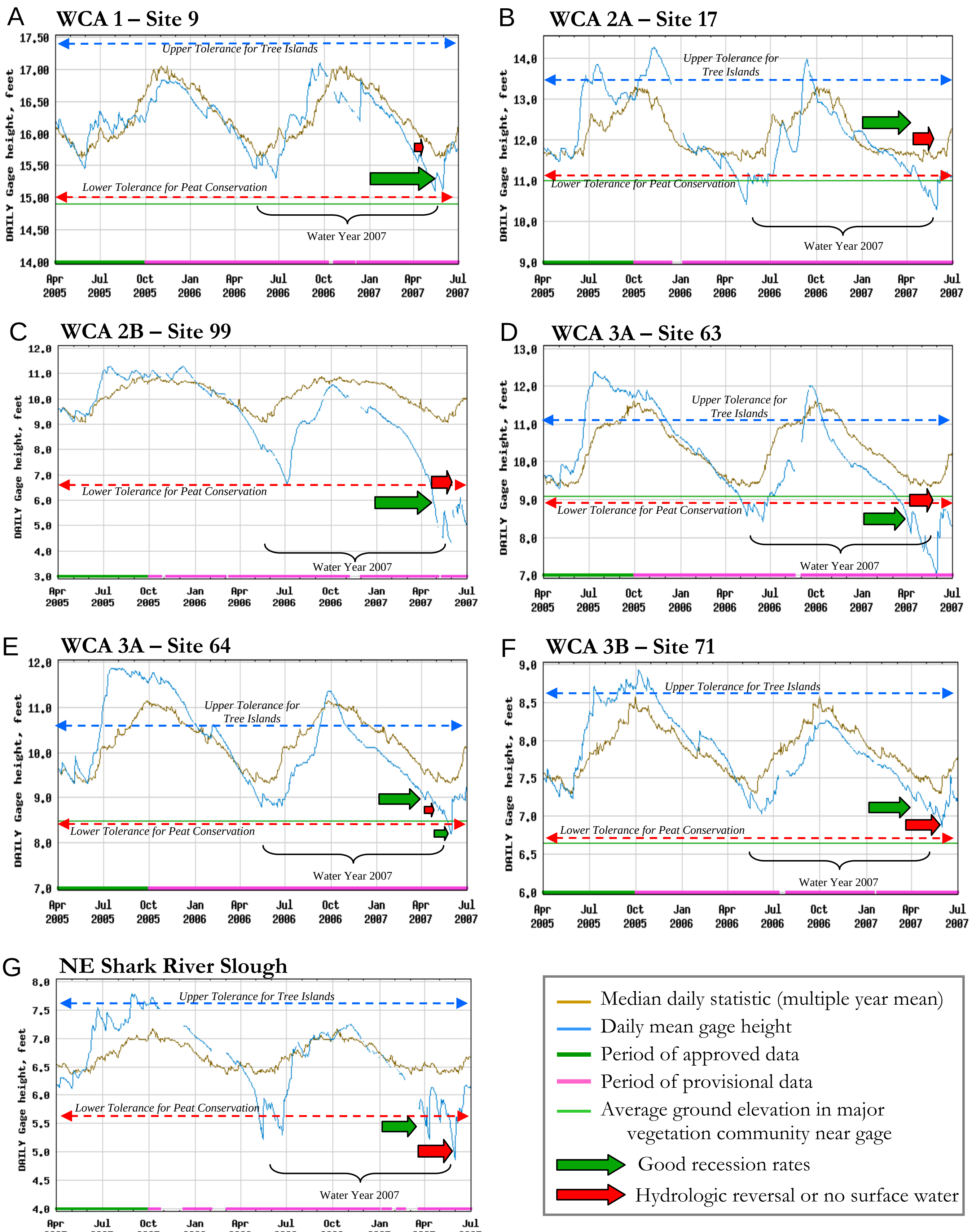
Figure 1. Hydrology in the WCAs and ENP in relation to recent average water depths (A: 9 yr ave, B: 12 yr ave, C: 11 yr ave, D: 12 yr ave, E: 13 yr ave, F: 12 yr ave, G: 24 yr ave) and indices for tree islands, peat conservation, and wading bird foraging depths.



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WCA-2A and 2B

In WCA-2A, the differences between WY2006 and WY2007 were most obvious during the wet seasons (Figure 1B). In June 2005 the wet season began abruptly and was so intensive that it caused this region to exceed the upper flood tolerance for tree islands. In June 2006, WCA-2A was dry and wet season water depths were less than or equal to only 1 foot for most of July and August. For WY2007, only September stage heights were above average, the other 11 months were either average or some one foot below average.

In WCA-2A, the WY2006 and WY2007 dry seasons were very similar. Both had very good recession rates, both had minor recessions and both times the region completely dried out. The difference is that it dried out almost a month sooner in WY2007. The other difference is that in WY2006, WCA-2A exhibited excellent foraging conditions and many flocks of wading birds were observed. This year, although foraging conditions should have been similar, reports of large or many flocks were greatly reduced.

In WCA-2B, there was no hydrologic similarity between WY2006 and WY2007 (Figure 1C). Most of WY2006 was slightly above average, while all of WY2007 was significantly below average. In WY2006 when dry season water levels went below ground in WCA-2A, the wading birds moved to WCA-2B because fortunately, rainfall patterns and recession rates in WCA-2B created a suitable foraging habitat for the displaced wading birds in WCA-2A. This year, 2B and 2A dry season hydroperiods were much more synchronous, and both regions became too dry to support any foraging from May to July. WCA-2B has a history of being the wettest of the WCAs and it was unique to see depths drop some two feet below ground in this region. (Note: More than one foot below ground violates the guidance for Minimum Flows and Levels.)

WCA-3A

The hydrology in the northeastern region of WCA-3A (Gage-63) in WY2007 was very similar to that in WCA-2A (Figure 1D). They both had very much below average stage readings for most of the year, they had the same abrupt September peak, same late beginning of the wet season, good recession rates during the dry season, and an extended dry period when water levels were below ground. (Note: More than one foot below ground violates the guidance for Minimum Flows and Levels.) However, this region dried out to a much greater degree than it did last year, and the combination of a late wet season and extended dry season created an inhospitable environment for wading birds, especially those that frequent the popular Alley North Rookery. Last year this region had good recession rates for the entire nesting season and better foraging conditions (in terms of hydrology) than the previous water year (WY2005). This year, the birds were lucky that their rookery did not burn.

The hydrologic pattern in central WCA-3A (Gage-64) in WY2007 was almost identical to that just shown for the northeast WCA-3A (Figure 1E). However, the hydrograph is shown here in Figure 1E to illustrate the one most significant difference: good foraging hydrology and no violation of the MFL during the dry season. This does not mean, of course, that foraging was indeed good in this area. It is very possible that the shallow depths and short duration of the wet season was sufficient to cause widespread depletion of wading bird prey species.

WCA-3B

Last year, in WY2006, despite good recession rates during the entire nesting season, the water depths in WCA-3B did not go below 0.5 foot (optimum foraging depth) until May 2006, after most nesting behaviors had ceased. This year, in WY2007, reversals occurred in March, April, May and June, making this region marginal for foraging visits by wading birds (Figure 1F). What was said for WCA-3A may also be true here and that is the possibility that the shallow depths and short duration of the wet season was sufficient to cause widespread depletion of wading bird prey species.

Northeast Shark River Slough

The uniqueness of the hydrology and drought in the Everglades during WY2007 is captured by the Northeast Shark River Slough hydrograph (Figure 1G). ENP, like most of southeast Florida did not experience below average rainfall for most of the year. Dry season recession rates were good, for the most part, until April when depths became too low and a series of large reversals caused foraging to probably cease. This trend was similar to that from last year. It was made worse, in all likelihood, by the short duration of the WY2007 wet season.

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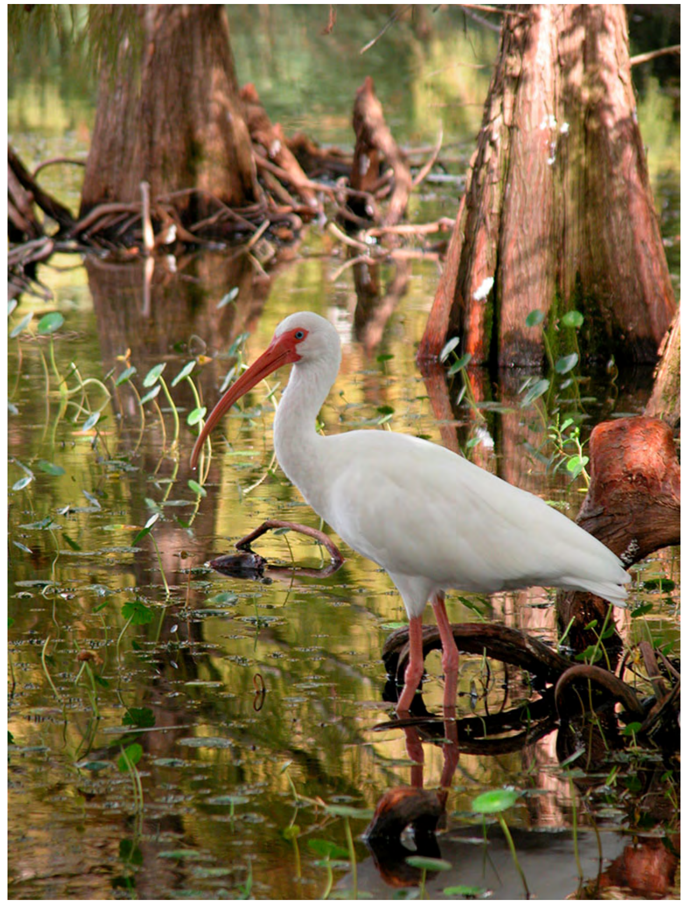
REGIONAL NESTING REPORTS

WATER CONSERVATION AREAS 2 AND 3, AND A.R.M. LOXAHATCHEE NATIONAL WILDLIFE REFUGE



In 2007, the University of Florida team monitored WCAs 2 and 3 and Loxahatchee for nesting by long legged wading birds. We concentrated effort on documenting numbers of Great Egrets, White Ibises, and Wood Storks, and continued our studies of juvenile stork movements and survival.

Methods

We performed 2 types of systematic surveys in 2007: aerial and ground surveys. The primary objective of both kinds of surveys is to systematically encounter and document nesting colonies. On or about the 15th of each month between February and June we performed systematic aerial surveys for colonies, with observers on both sides of a Cessna 172, flight altitude at 800 feet AGL, and east-west oriented flight transects spaced 1.6 nautical miles apart. These conditions have been demonstrated to result in overlapping coverage on successive transects under a variety of weather and visibility conditions, and have been used continuously since 1986. We took aerial photos of larger colonies from directly overhead and from multiple angles, and made detailed counts of the apparently nesting birds showing in these slides via projection. The reported numbers of nest starts are usually "peak" counts, in which the highest count for the season is used as the estimate of nests. The only exceptions to this rule were colonies in which clearly different cohorts were noted in the same colony, in which case the peak counts of the cohorts was summed. In most cases we also modified total aerial counts with information from ground checks.

In the past, we have performed systematic, 100% coverage ground surveys of colonies by airboat in WCAs 1, 2 and 3 once between early April and late May, and were designed to document small colonies or those of dark-colored species that are difficult to detect from aerial surveys. Since 2004, 100% coverage ground surveys were discontinued due to a change in MAP guidelines for monitoring. However, we did perform some systematic ground surveys in WCA-3 that allow for a direct comparison of densities of colonies in certain areas. This was designed to give an index of abundance for small colonies and dark colored species that might be sustainable. In the case of all ground surveys, all tree islands were approached closely enough to flush nesting birds, and nests were either counted directly, or estimated from flushed birds.

As part of an effort to measure nest turnover in colonies, we also estimated nest success in several colonies, by repeatedly recording the contents and fates of marked nests.

Results

Total counts in the WCAs and Loxahatchee NWR

Combining all species at all colonies in LNWR, WCA-2, and WCA-3, we estimated a grand total of 32,032 nests of wading birds (Cattle Egrets, Anhingas and cormorants excluded) were initiated between February and July of 2007. Note that this figure does not include birds nesting at the Tamiami West colony, which we also monitored intensively in ENP.

ABBREVIATIONS

Species: Great Egret (GREG), Snowy Egret (SNEG), Reddish Egret (REEG), Cattle Egret (CAEG), Great Blue Heron (GBHE), Great White Heron (GWHE), Little Blue Heron (LBHE), Tricolored Heron (TRHE), Green Heron (GRHE), Black-crowned Night-Heron (BCNH), Yellow-crowned Night-Heron (YCNH), Roseate Spoonbill (ROSP), Wood Stork (WOST), White Ibis (WHIB), Glossy Ibis (GLIB), Anhinga (ANHI), Double-crested Cormorant (DCCO), Brown Pelican (BRPE), Osprey (OSPR), Bald Eagle (BAEA), small dark herons (SML DRK), and small white herons (SML WHT).

Regions, Agencies, and Miscellaneous: Water Conservation Area (WCA), Everglades National Park (ENP), Wildlife Management Area (WMA), A.R.M. Loxahatchee National Wildlife Refuge (LNWR), Lake Worth Drainage District (LWDD), Solid Waste Authority (SWA), South Florida Water Management District (SFWMD), U.S. Army Corp of Engineers (USACOE), Systematic Reconnaissance Flights (SRF), Comprehensive Everglades Restoration Plan (CERP), and Natural Systems Model (NSM).

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Table 1. Numbers of nests of aquatic birds found in WCAs 2, 3, and Loxahatchee NWR during systematic surveys, January through June of 2007.

Latitude	Longitude	WCA	Colony	GREG	WHIB	WOST	ANHI	GBHE	TRHE	BCNH	SNEG	LBHE	ROSP	SML WT	SML DRK	Colony Total*
N26 31.968	W80 16.572	Lox	New Col 4		7,207			9						1,917	11	9,144
N26 22.330	W80 15.612	Lox	Lox 73	95	1,064		165							882	37	2,078
N26 26.293	W80 23.432	Lox	Lox99	202			7					11		1,540		1,753
N26 27.514	W80 14.419	Lox	New Col 2	505			1					40		355		900
N26 28.103	W80 22.337	Lox		288										54		342
N26 23.937	W80 14.995	Lox		280			26					5		10	18	313
N26 26.129	W80 14.037	Lox										307				307
N26 27.022	W80 15.720	Lox		77			16					111		2		190
N26 27.548	W80 25.401	Lox										159				159
N26 23.532	W80 18.736	Lox		118			36					5		15		138
N26 30.591	W80 19.425	Lox		1								85		2		88
N26 14.601	W80 21.043	2		37			4	3			5			443		488
N26 14.269	W80 18.768	2		31			1	19								50
N26 07.457	W80 32.489	3	6th Bridge	42	10,661			5								10,708
N26 07.445	W80 30.263	3	Cypress City**	652	200		4	19	100		150	100	22	544	125	1,912
N26 00.934	W80 33.763	3		380				14						52		446
N25 57.631	W80 34.324	3		197			4	55			2			37	10	301
N25 52.105	W80 48.398	3		17			4		3		1	41		145	46	253
N25 54.939	W80 37.813	3	Vacation	98				23						78	1	200
N25 53.240	W80 46.255	3									18	102		19		139
N25 46.412	W80 50.233	3	Hidden	15	46						1			72	1	135
N26 01.538	W80 32.350	3	Vulture	82			3	27								109
N25 49.239	W80 40.616	3		79			1	2						17	7	105
N26 12.079	W80 31.724	3	Alley North	36	8			1	10		29		17			101
N26 06.429	W80 29.881	3		91				8								99
N25 55.408	W80 31.115	3		51										21		72
N25 58.456	W80 46.340	3								60		2				62
N25 53.318	W80 48.272	3							1		17	11		30		59
N25 53.362	W80 33.758	3		32			8	10						13		55
N25 57.541	W80 28.739	3		39			13	9							2	50
Total Nests for Colonies > 50				3,445	19,186	0	293	204	114	60	223	979	39	6,248	258	30,756*
Total Nests for Colonies < 50				489	17	0	575	276	29	349	24	35	0	41	16	1,276*
Grand Total				**3,934**	**19,203**	**0**	**868**	**480**	**143**	**409**	**247**	**1,014**	**39**	**6,289**	**274**	**32,032***

* Does not include ANHI
** Small Dark (GLIB) estimated from ground visits; ROSP probably from Alley N.

The size of the nesting aggregation in 2007 in the WCAs and LNWR combined was approximately 80% of the average of similar counts during the past five years, 98% of the average of the past ten years, and 52% of the banner year of 2002. Numbers of Great Egret nests were only 54% the average of the last five years, and 66% of the average of the last ten. In 2007, Wood Stork nests were very much reduced, with no pairs attempting to nest in the WCAs. White Ibis nests were 87% of the average of the last five and 106% the average of the last ten years. Compared with the banner year of 2002, only 60% of the ibis pairs nested in 2007. Snowy Egrets appeared to be nesting in very small numbers according to Table 1, but we believe a large proportion of the unidentified white herons were actually Snowy Egrets.

Generally, nesting was not very successful for most species, with a lot of nest disappearances following water level reversals in late March, and some colonies drying out completely during the nesting period (which we suspect would have led to high predation rates by mammals). In the places that Wood Storks did attempt to nest (Tamiami West, Paurotis Pond) they did not nest successfully. At Paurotis Pond, all nests had failed by late May.

At Tamiami West, approximately 90 pairs did form nests by the end of March, of which approximately 40 appeared to fledge young. Counts of young per nest suggest that approximately 1.37 chicks were brought to a large fledgling stage per successful nest, which would translate into approximately 0.57 young per nest start. Both figures are far below the suggested replacement rates for this species. However, we did see successful nesting at large colonies of both Great Egrets (eg, Vacation, Cypress City), and and White Ibises (6th Bridge, Lox 73, New Colony 4), and large numbers of young ibises were evident in late May at 6th Bridge and Tamiami West.

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EVERGLADES NATIONAL PARK

Mainland Areas February – July 2007

Methods

Aerial colony surveys were conducted monthly (February through July) by 1 or 2 observers using a Cessna 182 fixed-wing aircraft (~22 person hours). Survey dates were: 21, 22 and 23 February, 19, 21 and 23 March, 4 and 23 April, 11, 21, 30 May, 26 June and 20 July. (Note: not all colonies were flown on each date and several colonies were checked via helicopter during other project flights.)

Results

Numbers of colonies and nest numbers within colonies were well below the exceptional 2006 nesting season totals and more in line with the lower counts from previous years. The reduced nesting observed this season was probably due to low water conditions throughout the region. Most colony sites and surrounding areas were already quite dry when checked during other project flights in January as well as during the first colony flight in February. By March, there was little to no surface water seen around most colonies and within many areas of the park. Birds were finally seen incubating on nests in March, however several rain events in April coincided with subsequent colony abandonment. Immediately after the rain events, much of the park appeared to be completely inundated. By the 23 April flight, birds had already abandoned nests and vultures were observed in several colonies.

Overall, nest numbers of all species combined decreased by 68% compared to the 2006 season. A total of 3281 nests within 55 active mainland colonies were surveyed in Everglades National Park. White Ibis were the most abundant of the species surveyed but their nest numbers were down 67% from 2006 numbers. Great Egrets were the second most abundant nesting species but their nest numbers were down 52% compared to the 2006 season. Snowy Egrets and Wood Storks seemed to be the most sensitive to the poor water conditions. Few Snowy Egret nests were seen this season, down 96%, and stork nests were down 70% from 2006 numbers.

Two colonies were still active as of 20 July. White Ibis along with a few Great Egrets were attempting a second nesting at Paurotis Pond. Both ibis and egrets were seen incubating on nests. At Rodgers River Bay, Great Egrets were brooding on approximately 125 new nests. Some small young could be seen. However, now that summer rains have started, it is doubtful that these second nest attempts will be successful. We will continue to monitor the status and outcome of these colonies.

Note: For our final tally of colonies, we combined our counts with additional colonies found during systematic colony searches conducted by University of Florida researchers: Peter Frederick and John Simon.

Table 1. Peak numbers of wading bird nests found in Everglades National Park colonies from February through July 2007.

Mainland colonies only COLONY NAME	Latitude WGS 84	Longitude WGS 84	GREG	WOST	WHIB	SNEG	CAEG	ROSP	TRHE	LBHE	BCNH	LRG WHT	SML WHT	SML DRK	TOTAL
Broad River	25 30.176	-80 58.464	50			30		15							95
Lower Taylor Slough	25 13.618	-80 41.057	10												10
Cuthbert Lake	25 12.560	-80 46.500	100	75											175
East River Rookery	25 16.116	-80 52.071	12												12
Grossmans Ridge West	25 38.176	80 39.166	40							+					40
Madeira Ditches	25 19.390	-80 38.740			20		20								40
NE Grossman A	25 38.810	-80 36.550					60								60
Otter Creek	25 28.068	-80 56.263	120		200			+	+	+					320
Paurotis Pond*	25 16.890	-80 48.180	185	150	410			15	+	+					760
Rodgers River Bay Peninsula*	25 33.400	-81 04.190	105	40											145
Rookery Branch	25 27.814	-80 51.153	125		400	+		+	+	+					525
Tamiami East-2	25 45.561	-80 31.474	8												8
Tamiami West	25 45.447	-80 32.701	60	75	400										535

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Table 1. Cont.

Mainland colonies only COLONY NAME	Latitude WGS 84	Longitude WGS 84	GREG	WOST	WHIB	SNEG	CAEG	ROSP	TRHE	LBHE	BCNH	LRG WHT	SML WHT	SML DRK	TOTAL
UF - L	25 38.000	-80 39.860	69			36									105
UF - COL 1	25 42.451	-80 35.452	38		23										61
UF - T	25 37.850	-80 59.350	30			1									31
UF - COL 12	25 32.718	-80 46.807	13										12	1	26
UF - COL 5	25 31.197	-80 50.678	17										5		22
UF - WP 498	25 29.958	-80 53.977	20										2		22
UF - E	25 40.250	-80 54.620	20										1		21
UF - WP 440	25 37.928	-80 59.342	15		5										20
UF - COL 4	25 32.012	-80 46.773	17										1		18
UF - N	25 37.950	-80 44.440	16												16
UF - WP 497	25 30.677	-80 52.384	16												16
UF - AA	25 36.400	-80 56.060	16												16
UF - W	25 38.000	-81 00.090	14			1									15
UF - M	25 38.170	-80 43.630	11			3									14
UF - COL 10	25 31.290	-80 48.305	9										3		12
UF - COL 11	25 30.100	-80 47.180	7										4		11
UF - WP 438	25 40.492	-80 55.902	5										6		11
UF - Z	25 36.360	-80 56.970	11												11
UF - BB	25 35.550	-80 42.200	9			1									10
UF - Q	25 37.480	-80 55.640	7										3		10
UF - J	25 38.120	-80 56.610	10												10
UF - U	25 38.000	-80 59.380	8			1									9
UF - V	25 37.910	-81 00.060	7												7
UF - Y	25 37.100	-81 01.800	7												7
UF - X	25 37.870	-81 02.550	7												7
UF - K	25 38.350	-80 37.120	7												7
UF - S	25 37.680	-80 58.000	6												6
UF - G	25 38.350	-81 00.200	5												5
UF - WP 496	25 32.274	-80 45.261	4												4
UF - UF1	25 28.793	-80 48.211	2										2		4
UF - I	25 38.720	-80 57.150	3												3
UF - H	25 38.720	-81 00.150	3												3
UF - B	25 41.190	-80 41.540	3												3
UF - R	25 37.350	-80 56.750	3												3
UF - P	25 37.700	-80 47.100	1			1									2
UF - UF4	25 28.793	-80 48.211	2												2
UF - WP 403	25 37.902	-80 46.326	1												1
UF - UF1	25 28.793	-80 48.211	1												1
UF - COL 2	25 34.237	-80 48.865	1												1
UF - WP 408	25 29.532	-80 51.183	1												1
UF - D	25 40.900	-80 52.510	1												1
UF - WP 437	25 40.399	-80 54.837	1												1
TOTAL			1259	340	1458	74	80	30	0	0	0	0	39	1	3281

+ Indicates species present but unable to determine numbers

* includes 2nd nesting attempt

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EVERGLADES NATIONAL PARK

Florida Bay January – July 2007

A formal wading bird aerial nesting survey was not conducted in Florida Bay, however we continue to monitor nesting activity at the large Frank Key colony.

Frank Key
Birds were not seen in February but were already nesting when checked during the 21 March survey. The highest nest numbers were also recorded on that flight date. The colony consisted of about 105 pairs of Great Egrets, 150 pairs of Brown Pelicans and approximately 125 pairs of Double-crested Cormorants. Most birds were already incubating on nests. When checked again on 4 April, many Great Egrets were brooding small young. White Ibis were also seen in the colony but were not nesting. There were also about 5-10 Great White Herons nesting on the island, not all nests were within the central colony.

After several rain events occurred in April, most Great Egrets abandoned their nests with only 20 pairs remaining when checked on 23 April. Approximately 400 White Ibis were seen roosting in the colony but had not set up nests. The pelican and cormorant nest numbers did not change and young were seen in their nests. On 11 May, it appeared that 125 pairs of White Ibis were setting up new nests and some appeared to be incubating. The remaining Great Egret nests were still active, but with very few young seen in nests, most birds seen were adults. When checked again on 30 May, the ibis had abandoned all nests and no adult ibis were seen in the colony. Only a few adult Great Egrets remained. In June, Snowy Egrets were seen roosting within the colony, but didn't attempt to nest at Frank Key this season. Pelicans and cormorants were the only birds that appeared to have a successful nesting season within the Frank Key colony.

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WOOD STORK NESTING AT CORKSCREW SWAMP SANCTUARY

Location: N26° 22.5024 W081° 36.9859

Methods
Corkscrew Sanctuary Staff conducted aerial reconnaissance throughout Southwest Florida from early November through May to record Wood Stork foraging and nesting effort. Flights were conducted two times per week on average using fixed wing aircraft. Initial surveys were conducted at 1000', when foraging and nesting efforts were identified; digital photographs were taken from 1000' and 500'. An 8.2 megapixel Canon EOS 30D body was used in combination with a 70-300mm lens for close-ups and an 18-55mm wide angle lens for landscape images. The close-up lens was equipped with an image stabilizer.

Results
No nesting was initiated. Wood Storks arrived in the Corkscrew watershed as early as October of 2006. Courtship behavior was observed off and on for approximately one week in December, yet no nesting occurred. In April, Wood Storks constructed some nest platforms in the bald cypress areas where nesting typically occurs at Corkscrew, yet no eggs were laid. Research staff documented wood storks foraging in 433 distinct wetlands across Southwest Florida from October 15th through May 17th.

Hydrology
Water levels at the Corkscrew staff gage peaked at 43.56" in mid-September. This is approximately 6.5" above the average wet-season high. A very pronounced dry season followed these high water levels where Corkscrew recorded only 6.54" of rainfall from October 2006 through March 2007, when the mean rainfall is 15 inches.

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Table 1. Wood Stork nesting in Southwest Florida. Nest initiations are nests in which eggs have been laid.

Colony Name	Latitude	Longitude	Date	Stork Nests Initiated	Estimated Number Fledged
Lenore Island (Caloosahatchee West)	26 41.332	-81 49.809	4/10/2007	220	100-150
Peace River	27 01.629	-81 59.478	4/13/2007	63	NA
Morganton	27 02.014	-81 59.241	4/13/2007	18	NA
North Port Charlotte -Myakka River	27 01.962	-82 16.594	4/13/2007	0	0
Corkscrew Swamp	26 22.502	-81 36.985	4/10/2007	0	0
Caloosahatchee East	26 41.795	-81 47.697	3/5/2007	0	0
Collier/Hendry Line	26 22.223	-81 16.363	3/23/2007	0	0
Totals				301	NA

Other 2007 Wood Stork Nesting Colonies in Southwest Florida

An effort was made to document other nesting efforts throughout Southwest Florida monitored by Audubon staff in 2006-07. All sites monitored supported some wading bird nesting of a variety of species, however nesting across all species was clearly well below last year's levels. Only three of the sites monitored had some Wood Stork nesting. These were Lenore Island, (f.k.a. Caloosahatchee West colony), Peace River and Morganton.

Methods

Digital photos of the aerial survey for each colony were projected on a whiteboard and all nests that could be confirmed as Wood Storks were documented as such. At the time this report was compiled other wader species had not been tallied.

Results

Lenore Island was the most productive wood stork nesting site this season. It was monitored and photographed on nine occasions between February and May of 2007. Considerable nest abandonment occurred in April and the total nesting effort at Lenore Island produced an estimated 100-150 fledglings. Wading bird nest abandonment was evident at the Peace River, Morganton and Myakka River sites as numerous large nest structures were guano covered and vultures were observed on the nest platforms at the Myakka River location. For the seven locations monitored during the 2006-07 nesting season there were approximately 301 nest initiations documented. The same seven locations had an estimated 1,540 nest initiations last season.

Estimates of colony nesting effort and productivity can be found in Table 1 above.

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SOLID WASTE AUTHORITY OF PALM BEACH COUNTY ROOKERY

Methods

Typically, Breeding Bird Censuses (BBCs) are conducted from February – July in the SWA Roost by two observers every 8-10 weeks, representing approximately 12 man-hours. During the BBC, all islands from three abandoned shell pits are systematically surveyed from a small boat, and the identified bird species and nest numbers are recorded. Surveys are conducted during the morning hours so as to minimize any burden caused by the presence of observers. However, this year's severe drought restricted boat access into the colony. The peak nest numbers are a compilation of early season boat counts and visual counts from the observation towers.

Location & Study Area

The SWA roost is located on spoil islands in abandoned shell pits that were mined in the early 1960's in Palm Beach County, Florida (N26 46.683 W080 08.533, NAD27). The spoil islands consist of overburden material and range from 5 to 367 m in length, with an average width of 5 m. Islands are separated by 5-6.5 m with vegetation touching among close islands. The borrow pits are flooded with fresh water to a depth of 3 m. Dominant vegetation is Brazilian pepper (*Schinus terebinthifolius*), Australian pine (*Casurina* spp.), and Melaleuca (*Melaleuca quinquenervia*), all non-native species. Local features influencing the roost include: 1) the North County Resource Recovery Facility and landfill and 2) the City of West Palm Beach's Grassy Waters (=Water Catchment Area), a 44 km^2 remnant of the Loxahatchee Slough.

Results

This report presents preliminary data for the 2007 breeding season. Typically, nesting activities have been observed at this colony through September, and these surveys being reported are only through the end of July. Only the peak nest numbers are being reported for each of the bird species (Table 1, next page).

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Table 1. Peak number of wading bird nests in SWA Rookery from February to July 2007*

GREG	SNEG	CAEG	GBHE	LBHE	WOST	WHIB	ANHI	TRHE	Total Nests
53	11	87	0	2	124	676	127	40	1167

*Severe drought restricted boat access; nest numbers are a compilation of boat surveys and tower observations.

The estimated peak number of wading bird nests for the SWA Colony is 1167 which represents about a 19% decrease from the previous 2006 season. Despite the severe drought, there were nests of the following bird species: Great Egrets, Snowy Egrets, Cattle Egrets, Wood Storks, White Ibis, Little Blue Herons, Tricolored Herons, and Anhinga. The Wood Stork nest numbers were less than last year but were yielding 1-2 fledglings (visual observations). It is difficult to draw any real conclusions because of the incomplete data set. It should also be mentioned that there was at least one Roseate Spoonbill nest with fledglings observed from the observation tower.

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ROSEATE SPOONBILL NESTING IN FLORIDA BAY ANNUAL REPORT 2006-2007

Methods

Spoonbill Colony Surveys

Thirty-eight of Florida Bay's keys have been used by Roseate Spoonbills as nesting colonies. These colonies have been divided into five distinct nesting regions (Table 1) based on each colony's primary foraging location (Figure 1, Lorenz et al. 2002). During the 2006-2007 nesting cycle (Nov-May), complete nest counts were performed in all five regions by entering the active colony and thoroughly searching for nests. Nesting success was estimated for the four active regions through mark and re-visit surveys of the most active colony within the region. These surveys entailed marking up to 50 nests shortly after full clutches had been laid and re-visiting the nests on an approximate 7-10d cycle to monitor chick development. Prey fish availability was estimated at six sites (TR, EC and WJ in the Taylor Slough Basin and JB, SB and HC in the C-111 Basin) in the coastal wetlands of northeastern Florida Bay (Figure 1) known to be spoonbill foraging locations for the Northeastern and Central regions. Prey abundance was also estimated at a site located in southern Bear Lake (BL) on Cape Sable where large numbers of spoonbills nesting in the Northwestern region regularly feed. Prey fish were collected monthly from Nov through Apr with a 9m^2 drop trap using the techniques of Lorenz et al. 1997. Prey availability data have not been fully analyzed and the qualitative information presented should be considered preliminary.

Banding Program

The purpose of this banding program is to better understand the movements and dynamics of the state's spoonbill population. We are interested in the location of post breeding dispersers, the possibility of breeder exchanges between Florida Bay and Tampa Bay and state-wide regional movements of the general population. We are hoping to see trends in spoonbills' movements with future banding and resighting efforts. Please refer anyone with information on resighting banded spoonbills to the author or our website (http://www.audubonofflorida.org/who_tavernier_reportspoonbills.html).

In Florida Bay, spoonbill nestlings were banded at 19 of the 24 colonies where spoonbills nested. In Tampa Bay, we banded spoonbills at the largest colony in the region, Richard T. Paul Alafia Bank Bird Sanctuary (Alafia Bank) (Hillsborough Bay), as well as the smaller colony of Washburn Junior (Terra Ceia Bay). Both are mixed colonial waterbird colonies. The 19 colonies in Florida Bay were distributed among five regions: 1 colony in the Northwest, 5 colonies in the Northeast, 6 colonies in the Central, 6 colonies in the Southeast, and 1 colony in Southwest Florida Bay. The northwestern region had 4 active colonies, 3 of which were patrolled heavily by American Crows. In an effort to minimize our impact, banding activities in these colonies were discontinued based on prior observation of intense nest predation by this species.

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Table 1. Number of ROSP nests in Florida Bay Nov 2006-May 2007. An asterisk (*) indicates colony with nesting success surveys (see Table 2).

Sub-region	Colony	2006-07	Summary since 1984		
			Min	Mean	Max
Northwest	Sandy*	100	62	157	250
	Frank	51	0	54	125
	Clive	52	11	27	52
	Palm	15	9	16.25	21
	Oyster	0	0	6.44	45
	Subtotal	218	65	211.55	325
Northeast	Tern*	64	60	109.31	184
	N. Nest	0	0	0.13	1
	S. Nest	26	0	18.59	59
	Porjoe	0	0	29.53	118
	N Park	13	0	19.06	50
	Duck	0	0	2.00	13
	Pass	0	0	0.53	4
	Deer	3	2	2.50	3
	Subtotal	106	101	185.88	333
Cental	Calusa*	21	0	12.43	21
	E. Bob Allen	2	0	14.71	35
	Manatee	0	0	0.00	0
	Jimmie Channel	8	6	20.18	47
	Little Pollock	0	0	2.75	13
	S. Park	3	0	11.24	39
	Little Jimmie	12	12	12	12
	First Mate	1	1	1	1
	Captain	9	9	9	9
	Subtotal	56	15	54.00	96
Southwest	E. Buchanon	0	0	6.53	27
	W. Buchanon	0	0	3.64	9
	Barnes	3	0	0.29	3
	Twin	0	0	1.71	8
	Subtotal	3	0	11.38	35
Southeast	Stake*	13	0	5.07	19
	M. Butternut	1	1	21.71	66
	Bottle	15	0	11.44	40
	Cowpens	0	0	6.13	15
	Cotton	0	0	0.00	0
	West	0	0	3.07	9
	Low	0	0	0.00	0
	Pigeon	1	0	8.87	56
	Crab	8	0	2.29	8
	East	0	0	3.56	13
	Crane	4	2	13.60	27
	E. Butternut	27	4	5.64	27
	Subtotal	69	39	81.57	117
Florida Bay Total		452	429	557.47	880

Nestlings were banded anywhere between 5-20 days of age. We found that a 5 day-old chick was the youngest age we could band due to the small size of their legs. On the youngest chicks, we placed clay on the inner surface of the band to reduce its diameter and thereby stop the band from sliding over the joint. As the chicks age and their legs grow, this soft clay is then displaced, allowing the band to move freely. After approximately 20 days of age, we no longer attempted to band the nestlings due to their extreme mobility. We found that attempting to capture these highly mobile chicks caused unacceptable levels of stress to the chicks and disturbance to the colony. We retrieved nestlings from their nests by climbing the nest trees, or by extending a ladder up to the nest. We then transported the nestlings in five-gallon buckets to a banding station. To keep the birds warm and calm, we lined and covered the buckets with towels.

In Florida Bay, a total of 3 bands were placed on each nestling. A USGS band was placed on the tarsus, and a 2-digit alphanumeric band was placed on the opposite tibia. Florida Bay spoonbills received an additional colored celluloid band, placed above the alphanumeric band, to designate the region in which the bird was banded (blue for NW, white for NE, red for Central, and yellow for SE). Tampa Bay birds were banded with a USGS band and a red alphanumeric band. The Alafia Bank birds were not banded with an additional celluloid band, and the Washburn Junior birds were banded with an additional white celluloid band above the alphanumeric band. At the time of banding, we recorded the age and sibling rank of each chick and the number of siblings or eggs still in the nest.

Frequent visits to the colonies of Florida Bay and Tampa Bay were required in order to band as many nestlings as possible. During these visits, some nestlings were not banded due to the disturbance it caused to neighboring nests with large, mobile chicks. Although it was our goal to band every nestling in Florida Bay, many nests were not banded because they failed before the eggs hatched, the nestlings died before reaching banding age, or it was physically impossible (or too unstable) to reach the nests to retrieve the chicks. In Tampa Bay, we banded large enough chicks during the main nesting cycle, and did not band chicks during the later asynchronous nesting cycle to avoid disturbing the co-nesting White Ibis.

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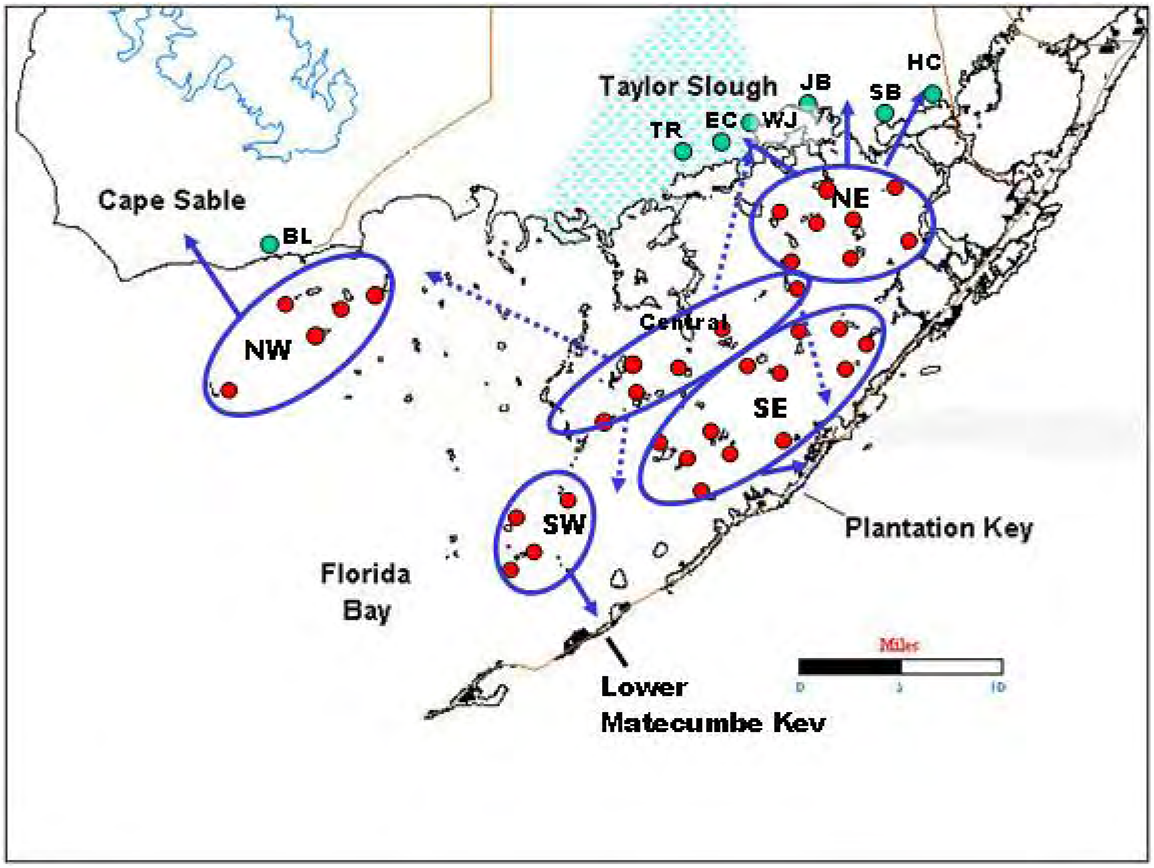
Figure 1. Map of Florida Bay indicating spoonbill colony locations (red circles) and nesting regions (blue circles). Arrows indicate the primary foraging area for each region. The dashed lines from the central region are speculative. Approximate locations of fish sampling sites are represented by green circles.



Spoonbill Monitoring Results

Northwestern Region: Sandy Key

All five colonies in the Northwestern region were surveyed for nesting activity in 2006-07 (Table 1). A total of 218 nests were counted in this region, which is slightly above average for this region compared to the last twenty-three years of survey data. Nesting success surveys were conducted at Sandy Key on Nov 1, 10, 15, 20, Dec 1, 8, 22, 14, 24, Jan 8, Feb 1, and Feb 15. Individual nest attempts were asynchronous compared to this colony's historical nesting record; however, in the last few years, nest attempts have been typically asynchronous. We estimate that the first nest to lay eggs was on Oct 23 while the last nest did not lay eggs until Nov 16. Usually, all nests are initiated within 14 to 21 days of each other. The mean egg laying date was Nov 2, and the mean hatch date was Nov 22. This was, by far, the earliest nesting that has occurred at Sandy Key since hatch records began in 1987. This date is two weeks earlier than the next closest mean hatch date of Dec 5 (1999) and more than 5 weeks earlier than the 1987 to 2006 mean hatch date of Dec 29. The 100 nests counted on Sandy Key were below average (157 nests since 1984). Sixty-one nests were marked for revisitation. Of these, 69% were successful at raising chicks to at least 3 weeks old (the time when they first leave the nest) with the average of 1.66 chicks per nest attempt (c/n; Table 2). The fledging rate was above average (1.27 chicks/attempt since 1984; Table 2) and is considered successful (the standard for being considered a successful nesting is at least 1 chick fledged per nest on average). Total production for Sandy Key was estimated at 166 chicks fledged (slightly higher than last year's 160 chicks fledged).

Table 2. Mean number of chicks per nest attempt. Numbers in parentheses indicate the percentage of successful nest attempts. Success is defined as fledging 1 or more chicks per nest. Second nesting attempts are not included.

			Summary since 1984			
Sub-region	Colony	2006-2007	Min	Mean	Max	% of Yrs Successful
Northwest	Sandy	1.66 (69%)	0.00	1.27	2.5	65%
Northeast	Tern	.96 (54%)	0.00	0.79	2.2	33%
Cental	Calusa	.76 (52%)	0.00	0.81	1.71	30%
Southeast	Stake	.92 (69%)	0.14	0.95	2.09	27%

The results of the colony surveys were supported by results from the banding program. One-hundred and two nestlings from 38 nests were banded at the Sandy Key colony (Table 3). Chicks were banded between Dec 1 and Dec 8. Although 7% of these chicks were found dead before leaving their nest, approximately 67% of the banded chicks were observed post-fledging on the fringes of the colony. Based on band resightings, nesting success was estimated to be 1.79 c/n. All of the chicks had fledged the island by the Feb 15, 2007 survey. One fledgling was resighted at Lake Ingraham (a popular foraging area for birds of Northwest Florida Bay, approximately 9.5 miles NNW of Sandy Key) on January 22; this bird was approximately two months old at the time of the resighting.

A discussion of water levels and prey fish availability at the BL fish collection station is pertinent to understanding why spoonbills nesting in the Northwestern region were successful. Lorenz (2000) estimated that prey fish become concentrated into small pools when water levels on the surrounding wetland drop to about 12.5 cm, thereby making them susceptible to predation by spoonbills and other wading birds. Peak water levels generally occur in late Sep or Oct but in 2006 water levels at BL peaked in Aug and rapidly declined in Sep. By mid-Oct, when spoonbills typically return to Florida Bay for the nesting season, water levels at BL were already below the 12.5 cm mark indicating that prey were already concentrated. This unusual circumstance likely explains the record early nesting date. Water levels continued to drop throughout Nov and Dec, creating an ideal situation for foraging when the chicks hatched in late Nov. Prey concentration data from BL suggests that prey began to concentrate in November and peaked in Dec. By Jan, prey concentrations at BL were depleted even though water levels remained well below 12.5 cm. The ideal water level and prey concentration conditions observed at BL just prior to and for the six weeks following the mean hatch date likely account for the high success rate at Sandy Key.

Northeastern Region: Tern Key

All eight of the spoonbill nesting colonies were surveyed in the Northeastern region of Florida Bay. A total of 106 nests were found in this region, which is well below average, and only slightly higher than the all-time low of 101 nests in the 2002-03 nesting season (Table 1). We counted nests at all eight colonies, however; only four were active during the first nesting cycle (nesting occurred later at an additional colony during what is typically the second nesting cycle). The 106 total nests in the region is the second lowest nesting effort in terms of the number of active nests, but this has occurred twice before in the last 20 years of survey data (the 2002-02 and 2003-04 seasons each had a low total nest count of 106). Spoonbill nest success surveys were conducted at Tern Key on Nov 3, 17, 29, Dec 11, 20, 28, Jan 4, 10, 18, 30, Feb 20, Mar 6, 13, 21, 28, April 4, 18 and May 9. Since the late 1980's, there has been a second nesting cycle at Tern Key, however, this year a second wave of nesting did not occur at the colony. A late-season nesting 'push' did occur at two other colonies in the Northeastern region after the first cycle of nesting was completed (further discussion follows below). At Tern Key, the first egg was laid on Nov 29 and the last nest was initiated on Dec 16 with the mean laying date estimated at Dec 5. The mean hatching date was Dec 25. Unlike Sandy Key, the nesting was somewhat synchronous, with all nests being initiated within 18 days of each other. As has been the trend in recent years, the nesting effort was alarmingly small: only 64 nests compared to almost 200 nests ten years ago and over 500 nests twenty-five years ago. We believe this decline in northeastern Florida Bay is due to water management practices on the foraging grounds. 2006-07 was the second all-time lowest number of nests for this region and is considered alarmingly small. In contrast, Tern Key birds were successful at producing more chicks per nest this season than birds in most other nesting seasons in the last 10 years. On average, each nest attempt produced 0.96 c/n compare to the average of 0.79 c/n since 1984 (Table 2). Of the 64 nests initiated on the island, 48 were marked for revisitation. Of these, 54% were successful at raising chicks to at least 3 weeks old; this is down from last year's remarkable nesting season (63% successful with 1.61 chicks per nest). Total production for the colony was estimated at 61 chicks.

In the northeastern region, 86 nestlings were banded from 37 nests within 5 colonies (Tern, South Nest, North Park, Deer, and Duck Keys; Table 3). Chicks were banded between Jan 3 and April 11. Forty-seven percent of the banded chicks were observed post-fledging but before they abandoned their natal colony for an estimated production of 1.08 c/n, an average well above that estimated by the Tern Key colony surveys. This high production estimate is perhaps bolstered by the South Nest colony, which produced an impressive 1.38 c/n. Although the overall nest effort on South Nest was small, 65% of the nests were successful at raising chicks to at least 3 weeks of age; this high productivity and success rate along with Tern Key's slightly better than average nest success is a hopeful sign that those birds that nest in the Northeastern region, albeit in small numbers, are able to successfully produce young.

In contrast to the early high water peak at BL, the water levels on the northeastern foraging grounds peaked in Sep (as is more typical) and receded much more gradually than at BL. In both the C-111 and Taylor Slough basins, water levels did not reach the 12.5 cm mark until early Dec. Fish concentrations in Taylor Slough peaked just as the first eggs were hatching. Unfortunately, both the C-111 and Taylor Slough basins experienced a reversal in the water level draw down process shortly after eggs began to hatch. The mean hatch date coincides with the peak of this reversal and water levels remained above the prey concentration depth from Dec 16 to Jan 7 in both basins. Fish concentration in Jan was well below peak concentration in both basins. It is clear from these data that most of the spoonbill chicks hatched at an inopportune time. Of the 33 nests that hatched before the mean hatch date, 20 nests failed. All but one of the nests that hatched after Dec 27 succeeded. It appears that the early nesters were subjected to adverse conditions for a longer period and during peak energetic demands of the chicks, thereby explaining the high mortality. In contrast, the late nesters only had to endure a few days of

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Table 3. Number of ROSP banded in Florida Bay Dec 2006-April 2007, and in Tampa Bay, April 2007-May 2007. "Number of ROSP Resighted Alive" indicates the number of birds resighted after the age of 21+ days.

Estuary	Sub-region	Colonies where Roseate Spoonbills were Banded	Number of Nests Banded	Number of Chicks Banded	Number of ROSP Resighted Alive	Number of ROSP Resighted Dead	Number of ROSP where Fate is Unknown
Florida Bay	Northwest	Sandy	38	102	68 (67%)	7 (7%)	27 (26%)
	Northeast	Tern	15	35	15 (43%)	3 (9%)	17 (48%)
		S. Nest	9	20	15 (75%)	0	5 (25%)
		N. Park	5	12	6 (50%)	4 (33%)	2 (17%)
		Deer	5	10	0	0	10 (100%)
		Duck	3	9	4 (44%)	1 (11%)	4 (44%)
	Central	Calusa	13	24	13 (54%)	3 (13%)	8 (33%)
		Jimmie Channel	1	1	1 (100%)	0	0
		E. Bob Allen	2	2	2 (100%)	0	0
		S. Park	3	5	4 (80%)	0	1 (20%)
		Captain	2	4	1 (25%)	1 (25%)	2 (50%)
		L. Jimmie	7	12	6 (50%)	2 (17%)	4 (33%)
	Southeast	Stake	9	14	9 (64%)	0	5 (36%)
		E. Butternut	14	31	25 (81%)	0	6 (19%)
		Pigeon	1	2	1 (50%)	0	1 (50%)
		Crane	2	3	1 (33%)	1 (33%)	1 (33%)
		Bottle	9	20	3 (15%)	2 (10%)	15 (75%)
		Crab	5	9	4 (44%)	4 (44%)	1 (11%)
	Southwest	Barnes	2	4	3 (75%)	0	1 (25%)
		Florida Bay Total	145	319	181 (57%)	28 (9%)	110 (34%)
Tampa Bay		Alafia Bank	73	127	93 (73%)	1 (.8%)	33 (26%)
		Washburn Junior	15	35	24 (69%)	0	11 (31%)
		Tampa Bay Total	88	162	117 (72%)	1 (.6%)	44 (27%)

adverse conditions while chicks were still quite small and their energetic demands relatively low. Shortly after this period, the water level dropped below the prey concentration depth in both basins and, with the exception of two short duration reversals (less than 2d each), water levels remained below 12.5 cm. The Feb fish collections indicated peak concentrations of fish in the C-111 basin and near peak concentrations in Taylor Slough. All combined, these data suggest that the reversal that occurred just prior to the mean hatch date and prolonging into the second week after the mean hatch date resulted in most of the mortality in the northeastern subregion. The draw down reversal did not occur at the BL, suggesting that water management practices that affect the northeastern foraging grounds may have been responsible for chick mortality.

As mentioned above, there was not a second wave of nesting at Tern Key this year, but a later nesting effort did occur at two colonies: Deer and Duck Keys. By mid-February, first nesting attempts in all of the colonies in the Northeastern region had completely finished; this coincided with the initiation of new nests in both Deer and Duck Keys. Nest success surveys were not completed at these colonies, but based on observations and banding at the colonies, the earliest nests were initiated around the second week of February, with the latest chicks hatching out by the last week of March. A total of 13 nests were counted for this second nesting cycle of the Northeastern region (5 nests on Deer Key, 8 nests on Duck Key). It is interesting to note that Deer Key only had 3 nests during the first nesting cycle, and Duck Key had no nests until this second wave of nesting. The small number of nests during the second nesting supports the hypothesis that second nesting is populated by birds that failed to produce young in the primary nesting. It is certain that the birds nesting at Duck Key, and at least a few of the birds at Deer Key, were birds that had failed during the first nesting attempt at other colonies in the area, like Tern Key.

In 2007, the second nesting yielded only two successful nests with an average of 0.31 chicks reaching 21d post-hatching per nest attempt. We estimate that only 4 chicks fledged during the second nesting, based on observations of banded birds. A heavy rainfall event that occurred in early April may have resulted in the complete failure of the Deer Key nests; a nest survey after that event concluded that all adults had abandoned the colony, and that no chicks had survived and fledged on their own.

Southeastern Region: Stake Key

Previous nest success surveys in this region were conducted on Middle Butternut Key. This year, the astonishingly low overall effort of nest production at Middle Butternut (1 nest) instigated us to begin surveying another, more representative colony in this region. We chose Stake Key to replace Middle Butternut Key based on the number of nests on Stake Key at the time when we needed to begin monitoring nests.

All of the 12 Southeastern colonies were surveyed for nesting activity (Table 1). Nest success surveys were conducted at Stake Key on Nov 14, Dec 6, 19, 28, Jan 3, 9, 24, Feb 14, and Mar 2. The first egg was laid on approximately Nov 24, with a mean lay date of Dec 8. The mean hatch date was estimated to be Dec 28. Thirteen nests were initiated on the island; along with last year, which also produced 13 nests, this is the greatest overall nest effort since 1999. On average, each nest attempt produced 0.92 c/n; a marginal success rate. In the Southeastern region, we banded 79 nestlings from 40 nests within 6 colonies (E. Butternut, Stake, Pigeon, Crab, Crane, and Bottle Keys, Table 3). Chicks were banded between Dec 27 and Feb 28. Approximately 9% of these chicks were found dead before leaving their nests, and 54% of the banded chicks were observed post-fledging but before they abandoned their natal colony. Based on the banding effort, the success rate in the Southeastern region was 1.1 c/n, well above the Stake Key survey estimate. This elevated success rate is probably a result of the high number of chicks fledged from the E. Butternut colony (Table 3). Nest surveys were not conducted at E. Butternut, but colony counts of fledged young indicate that overall production for this colony was quite high, contributing to the overall success rate of 1.1 c/n for this region.

The success rate observed through nest surveys is about the same as last year's 0.86 chicks/nest attempt at Middle Butternut Key, and is slightly below the average 0.95 c/n since 1984. Historically, the southeastern colonies focused foraging on the mangrove wetlands on the mainline Florida Keys. Although most of these wetlands were filled by 1972 as part of Keys development boom, we presume (based on anecdotal evidence) that the few remaining Keys wetlands still serve as important foraging grounds for these birds. Since 1972 (when large scale filling of wetlands ended), nesting attempts in the Southeastern region generally faired poorly: 7 of 11 years surveyed were failures (Table 2).

Based on this year's band resight observations, it appears that conditions during the 2006-07 nesting were unusually favorable in the Southeastern region. However, based on previous work (Lorenz et al. 2002) it appears that the quality of the Southeastern region for nesting spoonbills is marginal, at best, thereby explaining the low overall effort. This is in stark contrast to the period prior to the Keys land boom when spoonbills nesting in the Southeastern region successfully fledged young every year with an average production of >2 chicks per nest (Lorenz et al. 2002).

Central Region: Calusa Key

Three new spoonbill nesting colonies were discovered this year in the Central region bringing the number of colonies to nine and the number of nests to 56 (Table 1). Two of the islands on which new nests were found are unnamed according to the Florida Bay Chart #33E, and so we have given them names for the sake of identification during the spoonbill nesting season. They are known as Little Jimmie, based on its proximity to Jimmie Key (~0.75 miles south of Jimmie Key) and First Mate, based on its proximity to Captain Key (~0.65 miles west of Captain Key). Captain Key was the third new spoonbill nesting colony in the Central region for the 2006-07 nesting season.

Nesting success surveys at Calusa Key were conducted on Nov 7, 21, Dec 5, 12, 18, 27, Jan 4, 10, 15, 23, Feb 8, 22, Mar 8 and Mar 29. Twenty-one nests were found on Calusa, which is well above average (12.4 nests since 1984). The first egg was laid on Nov 10, and the last nest initiated on Dec 7, with the mean laying date estimated at Nov 21. The mean hatching date was Dec 10. This nesting effort was much lower than last year's successful season (1.71 chicks per nest attempt) with only 0.76 c/n and only 52% of the nests were successful at raising chicks to at least 3 weeks of age. Total production for the colony was estimated at 16 chicks, and this estimate was confirmed with the observation of 16 fledglings outside the colony (Table 3).

We banded 48 nestlings from 28 nests within 6 colonies (E. Bob Allen, Jimmie, Calusa, South Park, Little Jimmie, and Captain Keys, Table 3) in the Central region. Chicks were banded between Dec 12 and Jan 9. Approximately 56% of the banded chicks were observed post-fledging but before they abandoned their natal colony. The banding effort estimate for production was 0.96 c/n, slightly above the survey estimate.

Significant nesting in the Central region is a relatively new phenomenon, having started in the mid-1980's. As such, little information has been collected on where these birds feed, but the central location suggests that they may opportunistically exploit the primary resources used by the other regions. Spoonbills nesting in the Central region have reasonable access to the entire mosaic of foraging habitats found in the other four regions (Figure 1). This catholic foraging style may cost a little more energetically (longer flights to foraging areas), but the increased likelihood in finding suitable foraging locations may counterbalance the cost. However, if the specific foraging habitats utilized by spoonbills in all of the other four regions become compromised, the spoonbills of the Central region would also be affected deleteriously. If these foraging grounds do not support abundant and concentrated prey, long flights to more productive areas may be too energetically demanding for a spoonbill to make, resulting in lower nest success. Based on flight-line counts and fixed-wing aircraft observations, it appears that the birds from the Central region are flying over the Russell and Black Betsy Keys to the Taylor Slough area to forage. It would appear that this season these flights were perhaps too demanding and foraging habitat was not as productive, resulting in their lower nest success (Table 2).

Southwestern Region: Barnes Keys

All keys in the southwestern region were surveyed multiple times in 2006-07 but only 3 nests were found on Barnes Key (Table 1). This is only the second time since 1984 that spoonbills have nested at Barnes Key. These nests did produce young, and three chicks were observed post 21 days hatching. This is a promising find for the Southwest region, whose historic record high was 153 nests in 1979.

Bay-wide Synthesis

Bay-wide Roseate Spoonbills nest numbers were well below average, indicating a continued downward spiral that began with completion of major water management structures in the early 1980s. Historically, the Northeastern region was the most productive region of the bay (Lorenz et al. 2002). Since 1982, this region has been heavily impacted by major water control structures that lie immediately upstream from the foraging grounds (Lorenz 2000). This year, the success rate at Tern Key was reasonably good and exceeded the 0.79 c/n average since 1984, however, this is still well below the success of 1.4 prior to modern water management. Also, the high degree of nest failure (46%) coinciding with a draw down reversal suggests that water management may have played a role in the overall success rate. Finally, the high success of nests in the Northwestern region and the lack of reversals at BL indicate that conditions should have been good for spoonbills nesting in the Northeastern region in the absence of adverse water management practices.

In all, 319 chicks were banded from 145 nests across Florida Bay. Of these 9% were observed dead either before leaving the nest or outside the colony and 57% were observed alive post-fledging. Outside of their natal colonies, there has been one resighting of a bird banded at Sandy Key in December observed foraging at Lake Ingraham, Everglades National Park, in January.

Comparison to Tampa Bay Nesting Population

We began banding spoonbill nestlings at the Alafia Bank, Tampa Bay, in 2003 as part of a pilot study for the banding program. The goals of this program were two-fold: 1) to determine the movements of spoonbills within the state and the region and 2) to get estimates of nesting success to compare to Florida Bay. Reports of spoonbills producing greater than 2 c/n in Florida Bay were regularly reported throughout Florida Bay as late as the early 1970s. Following the destruction of wetlands in the Keys and water diversion in the northeastern part of Florida Bay, the average dropped below 1 c/n on average. Tampa Bay colonies provided an opportunity to see how productive spoonbills were in another part of the state to assess if this decline was unique to Florida Bay or a more regional response. Answering this question is critical to demonstrating the causal relationships between Everglades management and the observed decline in Florida Bay.

Spoonbills nested in 11 colonies in the greater Tampa Bay area this year. The largest colony in the region is the Alafia Bank in Hillsborough Bay, with 325 pairs in 2007. The colony of Washburn Junior was the second largest with 45 pairs. A total of 393 fledged birds were observed during one survey of the Alafia Bank colony this season.

We concentrated our banding efforts for the Tampa Bay area at the Alafia Bank and Washburn Junior colonies. We banded nestlings on April 12, 13, 20, 25, 26, 30, and May 1. At the Alafia Bank, we banded 127 nestlings from 73 nests (Table 3) during 5 banding sessions (April 13, 25, 26, 30, and May 1). Of the 127 nestlings banded, we resighted 93 (73.2%) of them alive. One bird was observed dead in the colony. Only 33 of the total birds banded have not been resighted at all. Based on our estimation of 1.22 fledged birds/nest (93 resighted nestlings/73 nests), we expect about 397 spoonbills (325 pairs X 1.22 birds/nest) fledged from Alafia Bank. At Washburn Junior, we banded 35 nestlings from 15 nests. Of the 35 nestlings banded, we resighted 24 (68.6%) of them alive in the colony. We do not have any band recoveries for dead birds, and 11 of the total birds banded have not been resighted at all. Based on our estimation of 1.60 fledged birds/nest (24 resighted nestlings/15 nests), we expect about 72 spoonbills (45 pairs X 1.60 birds/nest) fledged from Washburn Junior. Based on the estimates from Alafia Bank and Washburn Junior, we estimate a total of 469 chicks fledged from 370 total nests in two colonies in Tampa Bay. In contrast, Florida Bay fledged virtually the same number of chicks (470) but from 20% more nests than in Tampa Bay. This further indicates the lack of production in the Florida Bay system.

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We banded 164 birds in April 2003, 233 birds in 2004, 105 birds in 2005, and 264 birds in 2006. Since then we have received resight reports for over 170 (22.2%) of those birds. These birds were resighted in Brevard, Collier, Dade, Duval, Flagler, Hendry, Hernando, Hillsborough, Lake, Lee, Manatee, Monroe, Nassau, Palm Beach, Pasco, Pinellas, Polk, Sarasota, St. John's, Taylor, and Wakullah Counties. Banded birds have frequently been observed at Merritt Island, Ding Darling, St. Marks, and Loxahatchee National Wildlife Refuges. Of those resighted birds, 5 birds were observed in Georgia. Over 70 birds have been resighted more than once, with one bird having been resighted 11 times in two locations. Three of the birds that were resighted in Georgia in 2004 and 2005 were resighted in 2006 and 2007 back in the Tampa Bay area. Twenty-two birds have been resighted at the St. Augustine Alligator Farm in the past four years.

Perhaps our most interesting and significant find is a Tampa Bay bird banded in 2003 that is now nesting at Gatorland in Orlando. As of June 11, the bird had hatched out 3 young, and by June 25 two of the nestlings were almost ready to fledge. This is the first documented banded bird reaching reproductive maturity and breeding. Incidentally, this is the first year since the creation of the breeding marsh at Gatorland that a Roseate Spoonbill has nested there, and this banded bird is the only nesting spoonbill at the marsh. It is also interesting to note that this bird had been resighted at the St. Augustine Alligator Farm in 2005.

Of the 205 resightings reported from across the state, 171 (83.4%) were birds banded in Tampa Bay and only 34 (16.6%) were banded in Florida Bay. Florida Bay birds have been resighted as far away as Hillsborough, Lee (Ding Darling), Nassau, Pinellas, and St. Johns Counties (4 out of 5 of the St. Johns County birds were at the Alligator Farm). This further suggests that Florida Bay's productivity is greatly diminished; however, migrations from Florida Bay southward to Cuba and the Yucatan Peninsula cannot be discounted as a cause for the low resightings from Florida Bay.

Clearly, Florida Bay has been, and continues to be, impacted by anthropogenic forces that render production to be less than that of healthy spoonbill nesting areas, including the highly industrialized habitats of Tampa Bay. It is also interesting to note that the rapid growth of spoonbill numbers in Tampa Bay coincides with the rapid decline in spoonbill numbers in Florida Bay since the early 1980s. We will continue to band in both locations using Alafia Bank as a pseudo-control for Florida Bay, as well as a source of information on spoonbill demographics in Florida and the larger Gulf of Mexico and Caribbean geographical regions.

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BIG CYPRESS NATIONAL PRESERVE

Systematic wading bird surveys were not conducted in Big Cypress in 2007.

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HOLEY LAND AND ROTENBERGER WMAS

Systematic wading bird surveys were not conducted this year in Holey Land or Rotenberger WMAs.

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SOUTHWEST COAST

The coastal waterbird nesting season starts toward the end of December in Southwest Florida when a few Ospreys start to sit; then several weeks later Brown Pelicans begin constructing nests. The first wader activity (Great Egret) generally starts mid March but this year they started nesting mid February, at Marco. On the first nest census (4/18), both Great, Snowy and Reddish Egrets had a few nests but the other three small waders (Little Blue Heron, Tricolored Heron, and Cattle Egret), which are usually breeding, were hardly present. After that, wader nesting went down hill, with the desertion of nests not only at Marco but also at Rookery Bay and Chokoloskee Bay. By mid May, there were hardly any waders, let alone any nesting at all three colonies. Usually there is a second wave of wader nesting in June that can be either equal to or slightly smaller than the first wave; this year the second wave started but was smaller than the first. The waders of the second wave, unlike the first, have not deserted and as of the end of July are raising chicks. The above indicates how different wader nesting was and is this year, two years after a severe hurricane.

Note: Nest censusing this year was conducted as it was done last year, from a small boat slowly moving around the periphery of the colonies rather than walking through as described below in Location and Methods. As the Marco, Rookery Bay and Smokehouse Key colonies still have much storm debris collapsed in the understory, it is impossible to go through them on foot without causing unacceptable disturbance.

Hydrology
The coastal ponds at Rookery Bay dried down completely this year as did the inland ponds at Corkscrew Swamp Sanctuary. The dry-down at Corkscrew was more prolonged than on the coast because the coastal ponds are influenced by spring high tides. On the coast and inland, the dry-down although severe was not as intense as the only other two dry periods (1989-90 and 2000-01) that have occurred both on the coast and inland over the 24 years of data collection.

Location and Methods
Rookery Bay (RB): N26 01.850, W081 44.716. Two Red Mangrove islands, 0.22 ha in size. Nest census were conducted 6/8, boat, 2 observers, 0.5 hours. All wader nests were on the southern island, as has been the case for the past four years.

Marco Colony (ABC): (named, ABC Islands by State of Florida): N25 57.400, W081 42.216. Three Red Mangrove islands, 2.08 ha in size. Nest census conducted 4/12, two observers, boat, two hours.

Smokehouse Key: (SK): (This colony formerly named Henry Key, now named for the closest body of water) N25 54.850, W081 42.866. One island in Caxambas Pass, 0.8 ha (Red Mangrove; a little terrestrial vegetation on sand ridge in center). Censused 6/23, boat, one hour, two observers.

East River (ER): N25 55.650, W081 26.583. Three Red Mangrove islands, about 0.25 ha in size. Nest census conducted 6/6, canoe, complete coverage, two observers, one hour.

Chokoloskee Bay (CHOK): N25 50.716, W081 24.766. Four Red Mangrove islands, 0.2 ha. This year most of the waders in the area used three of the four islands, boat census, 4/15, two people, one hour.

Note: All of the censuses are conducted during peak nesting and this varies according to species and timing.

Sundown Censusing
For two of the colonies above, birds coming in to roost for the night are censused at sundown. The goal of this project is to get an index of the numbers and species in the area, year round. References below as to the use of the area by the different species are derived from these projects.

Marco Colony (ABCSD)
Censused monthly with two boats and various numbers of volunteers (4-8). Boats were anchored in the two major flyways and species and numbers of birds flying in (and out during the nesting season) one hour before sunset to one half hour after sunset were recorded. This project is ongoing and started in 1979.

Rookery Bay (RBSD)
Censused bi-weekly with one boat, two observers. The boat was anchored so that most of the birds could be observed returning to the roost one hour before sunset to one half hour after sunset. We recorded, species and numbers of birds flying in (and out during the nesting season). This project is ongoing and started in 1977.

Species Accounts
Great Egret
As stated in the introduction, Great Egrets started nesting early at Marco. Nesting had started by the middle of April at Rookery Bay and Chokoloskee. On the first nest census, this species had low numbers of nests. Smokehouse did not have any Great nests but has never had many, nor has East River. As subsequent observations confirmed, something was different and, since censusing did not involve disturbance (see note above), nest censusing was increased as much as possible. These additional censuses showed that many of these first nests were being deserted. See Table 1, for peak numbers of nests.

Examples: At Marco, Great Egret nests dropped from 36 to 10 (4/28) in ten days, only two nests were productive; a much reduced second wave of nesting had also declined from 26 nests on 7/5 to 10 on 7/29.

At Rookery Bay, seven Great Egret nests dropped to one (4/25) in seven days and then that one disappeared; there was no second nesting wave.

Smokehouse Key had no wader nesting (except Reddish Egrets see below) during the first nesting wave; later on 7/3 in the second wave, there were seven Great Egret nests.

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Table 1. Peak Wader Nests Coastal Southwest Florida 2007.

Colony	GBHE	GREG	SNEG	LBHE	TRHE	REEG	CAEG	WHIB	GLIB	Total
Rookery Bay	0	8	9	2	10	0	11	0	0	40
Marco	11	62	33	2	29	6	46	0	3	192
Smokehouse Key	0	12	19	1	15	3	1	165	0	216
East River	0	0	4	2	27	0	0	0	0	33
Chokoloskee Bay	0	64	7	0	0	0	0	0	0	71
Total	11	146	72	7	81	9	58	165	3	552
Mean (24 year)	12	222	287	58	474	5	408	51	40	1557

At Chokoloskee Bay, Great Egret nests dropped from 34 to 20 (4/17) in 17 days; on 7/10 there were 30 nests with small chicks (Note this is a much harder colony for us to visit frequently, therefore we have less data). In summary, this was not much of a year for this species.

Snowy Egret, Little Blue Heron, Tricolored Heron and Cattle Egret
With slight variations, all these species up until now have had similar nesting patterns; therefore I will treat them together. At Marco and Rookery Bay a few nests started early and then were deserted and then later (second wave) about the same number started, some of these were deserted but a few have good sized chicks. At Smokehouse there was no early nesting but there were 19 nests in the second wave. Only one census was done at East River (volunteers) and numbers of nests were lower than usual (Table 1).

Reddish Egret
Although above the annual mean (Table 1), this species had problems similar to the small waders above. They started five nests at Marco and three at Smokehouse but fledged few chicks; possibly one dark and 2 white at Marco; 2 dark at Smokehouse. At present, both colonies have one new active nest; no other attempts at the other colonies.

White Ibis
This species did not attempt to nest at Marco this year; but at Smokehouse they continued the pattern of coming in late in the season and starting nests just about when all of the other waders were almost finished. Smokehouse is the only colony that we monitor in which this species now nests (for this species, coastal nesting in the area has always been limited) and the colony has only been active since 2003. White Ibis started nesting here in 2004 with 3 nests and jumped to 373 nests in 2005, three months before hurricane Wilma. In 2006, with the mangrove destruction from Wilma, the ibis only had 45 nests; interestingly not many adults (65 the most) were recorded nesting. This year 338 adults and 165 nests (Table 1) were the most recorded; as of now a few adults are in high breeding plumage but not building nests. It would appear that there is not enough nesting habitat left in the decaying storm debris.

Glossy Ibis
With only three nests that produced six fledglings at the ABCs not much can be said about this species, except that they are still present. In the sundown censuses the numbers have been very erratic; difficult to guess what is going on.

Once again the natural world has handed us another nesting season that is unlike anything recorded in the 24 years of data collection. Waders (also Pelicans, Cormorants and Anhingas) attempted very few nests and more than half of those that attempted deserted before chicks were showing in the nests. As usual we can make all kinds of assumptions as to the cause (s) but can't come even close to understanding why. All of this seems to indicate that there was not enough food in the area for the birds to be able to start much nesting, let alone sustain it. Food would be the major factor to check, but as there is very little work being done in the area on the fish that are a major part of these birds' diet, it is impossible to draw any conclusions. The following comments serve to illustrate the frustrating aspect of having some data but obviously not the right data.

The drought, although strong, was not as severe as the two periods cited above in hydrology (1989-90 and 2000-01) and, according to the nesting data, in neither of those periods was breeding affected adversely.

Ospreys, whose nesting has about leveled out in the last eight years in the area; again had a good year with 1.23 fledglings per nest. But this species is not directly comparable with waders as they take larger fish and the population of approximately 130 adults in the area, is much smaller.

According to the sundown censusing (description above) there were waders in the area during the nesting period. Great Egrets and White Ibis were actually above the means for both sites and small waders were 43% lower for the same period (this is down but not the lowest that has been recorded in a few normal nesting years).

Least Terns and Black Skimmers (approximate numbers of adults nesting in the area 650 and 700 respectively) had reasonable numbers of nests and have fledged fair numbers of chicks so far. This indicates that there is at least some bait fish around.

Interestingly, the 2007 decline of 66% from the 24 yr. mean in wader nesting was higher than the 2006 drop of 46% after Wilma. The destruction from the hurricane is still obvious at three of the colonies (ABC, RB, SK) and could be affecting nesting, but at the other two colonies (ER, CHOK) which had little damage, the same pattern of poor nesting was recorded. That ought to shoot down the idea that the storm effects had at least a major effect.

Well this could go on but in truth there is not even a decent hint as to what has caused this very unusual nesting season.

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UPCOMING MEETINGS

Waterbird Society Annual Meeting. 30 Oct – 3 Nov 2007. Barcelona, Spain. http://www.wbs2007.org

American Ornithologists' Union, Cooper Ornithological Society, and the Society for Canadian Ornithologists joint meeting. 5-9 Aug 2008. Portland, OR. http://www.pdxbirds08.org/

The Wildlife Society 2008 Annual Conference: 8-12 Nov 2008. Miami, FL. www.wildlife.org/

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WADING BIRD COLONY LOCATION, SIZE, TIMING, AND SUCCESS AT LAKE OKEECHOBEE

Introduction

Lake Okeechobee wading bird populations have been assessed since National Audubon Society wardens began patrolling the area during the early 20th century (David 1994). Systematic aerial surveys began during the early 1970s and then continued annually from 1977–1992 (Zaffke 1984, David 1994, Smith and Collopy 1995). Over the decades, wading bird nest counts ranged from a high of 10,400 in 1974 to a low of 130 nests in 1971 (Ogden 1974, David 1994).

In 2005, Florida Atlantic University renewed wading bird nesting surveys to determine the size and location of wading bird colonies on Lake Okeechobee as part of the CERP Monitoring and Assessment Plan. In 2006, we recorded 11,310 nests. Herein, we report the results of the 2007 surveys and attempt to link results to environmental conditions prevalent during the drought.

Methods

From January through June 2007, two observers surveyed wading bird nests along aerial transects. We flew transects in a Cessna 172 at an altitude of 244 m (800 ft) and a speed of 185 km/hr (100 knots). One transect paralleled the eastern rim of the lake from Eagle Bay Island to the Clewiston Lock. Remaining transects were oriented East-West, spaced at an interval of 3 km (1.6 nm), and traversed the littoral zone. Two observers searched for colonies from each side of the plane. Colonies were defined as any assemblage of ≥ 2 nests that were separated by ≥ 200 m (Erwin et al. 1981, Smith and Collopy 1995). When a colony was located, we lowered to 91 m (300 ft), and the colony was circled several times while we documented species composition and nest count. We also recorded photographs and geographic coordinates with each visit and then mapped colonies to specific stands of vegetation or islands onto 1-m resolution digital orthophotoquarterquadrangles (DOQQs). We calculated intercolony distances using ArcGIS. To maintain consistency with past wading bird reports for Lake Okeechobee (e.g. Zaffke 1984, David 1994a, Smith and Collopy 1995), we counted all birds sighted and categorized them as "nesting" if nests were visible or known assemblages of nests existed for a species. At the largest, most diverse, and accessible colonies, we followed aerial surveys with ground monitoring to improve count accuracy (Frederick et al. 1996).

Nest visits began as soon as colonies of incubating wading birds were observed. Two observers monitored nests along paired 50 X 10-m strip transects within selected wading bird colonies. All nests detected within 5 m of the transect line were marked with orange flagging and assigned a nest number. We visited nests every 6-8 days. We documented a nest as "successful" if at least one young survived to an age where they could branch away from the nest and mature feathers had emerged (Frederick et al. 1992).

Regional rainfall and hydrology data were obtained from the South Florida Water Management District's DBHYDRO data-

base and the National Climatic Data Center. Lake stages and recession rates reported herein were based on average stage readings from four principal gauges located in the pelagic zone at Lake Okeechobee (L001, L005, L006, and LZ40). Lake stages were reported as feet National Geodetic Vertical Datum 1929 (NGVD29). We used the recession rate index from Sklar (2005) to assess the suitability of wading bird foraging conditions. The index was based on weekly changes in lake stage.

Hydrology

From June to December 2006, the Lake Okeechobee region received its lowest, wet-season, rainfall accumulation over the last twelve years, with 4 out of 6 months (JUN, SEP, OCT, NOV) receiving less than half their long-term monthly averages. The South Florida Water Management District reported the recent drought was the third most severe on record. Lake levels at the beginning of 2007 were low and continued to recede toward a historical low by the end of June. Average lake stage was 12.13 ft on January 1, 2007, and steadily receded throughout the breeding season, eventually reaching a low of 8.86 ft on June 30, 2007.

Recession rates suggested that foraging conditions were good to fair throughout the breeding season (Fig. 1). But given low lake levels and a drought that lasted from the previous wet season, conditions left much of the littoral zone waterless and unavailable to foraging birds following last year's recession. Low habitat availability and declining habitat suitability persisted throughout the season, which acted to reduce foraging opportunities and the carrying capacity of Okeechobee for nesting wading birds.

Figure 1. Weekly precipitation totals (in) and average stage levels (feet NGVD 29) for Lake Okeechobee during the 2007 wading bird breeding season. Suitability of wading bird foraging recession rates were depicted in colored arrows. Good foraging conditions (green) existed when average lake stage decreased between 0.05 ft and 0.16 ft per week. Fair foraging conditions (yellow) existed when stage decreased between 0.17 ft and 0.6 ft or decreased only 0.04 ft per week.

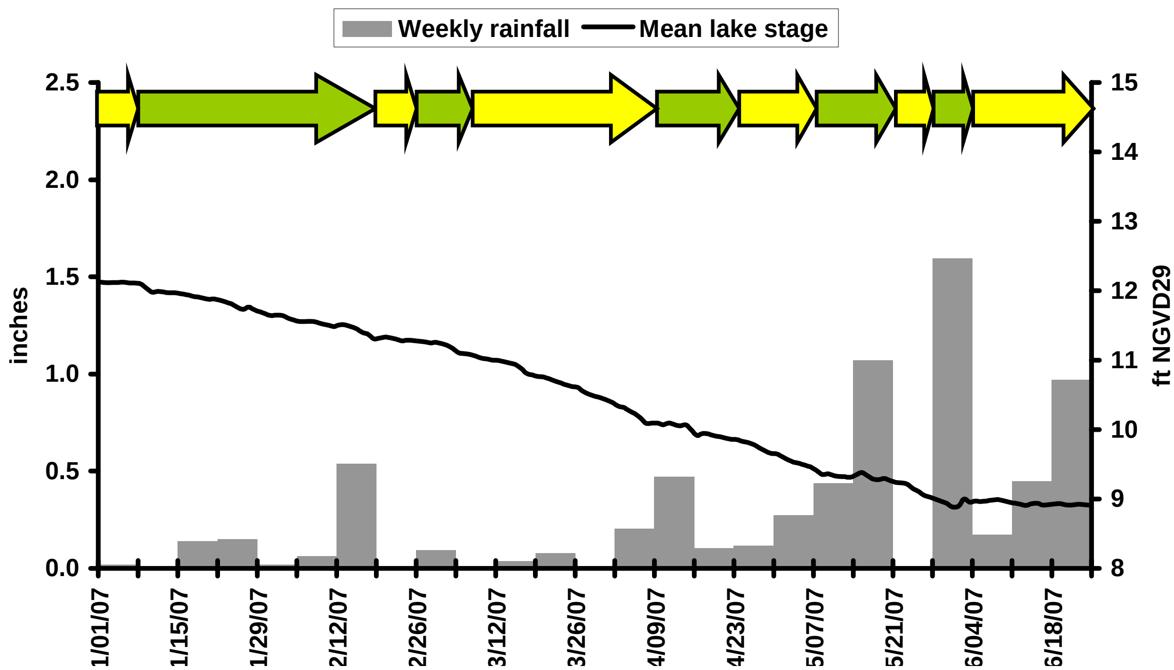

Table 1. Geographic coordinates (ddmmss, NAD 83) and species-specific peak nest efforts in detected colonies during the 2007 breeding season at Lake Okeechobee.

Colony name	ID	Geographic Location		Peak wading bird nesting month [1]	GBHE	GREG	SNEG	TRHE	LBHE	WHIB	GLIB	WOST	ANHI	CAEG
		Latitude	Longitude											
Clewiston Spit [2]	B1	80° 54' 27" W	26° 46' 36" N	APR 2007	'--- [3]	---	485	115	---	---	35	---	---	---
Bird Island	B2	81° 00' 31" W	26° 58' 20" N	JUN 2007	---	---	---	---	---	---	---	---	---	350
Gator Farm	B3	81° 03' 39" W	27° 01' 22" N	MAY 2007	---	7	25	37	14	---	---	12	---	340
Clewiston Channel	B4	80° 53' 53" W	26° 46' 50" N	MAY 2007	---	---	33	---	18	---	3	---	---	---
Little Bear Beach	B6	80° 50' 32" W	26° 43' 17" N	MAY 2007	---	---	70	40	---	---	---	---	---	---
Port Mayaca	B7	80° 34' 28" W	27° 03' 17" N	JUN 2006	3	---	m [4]	m	m	---	---	---	18	640

[1] Does not consider timing of peak CAEG or ANHI nesting

[2] Only colony consistent with 2005 and 2006 locations.

[3] Species undetected during monthly survey effort

[4] Unable to finish counts due to proximity of colony to the Martin County Florida Power and Light power plant (m = missing value).

Results

Locations

We located six wading bird colonies in the Okeechobee area—4 on-lake and 2 off-lake (Fig. 2). The Clewiston Spit colony was the largest colony and the only site perennially occupied from 2005–2007 (Table 1). A smaller colony on another island along the channel was spatially distinct at ca. 800 m ENE from the colony at Clewiston Spit. Bird Island was the second colony we located early in April. Smith and Collopy (1995) reported that Bird Island was occupied from 1989–1992, but we did not detect a colony there until this year. In May, wading birds also nested along the rim canal levee near Little Bear Beach. We also detected two colonies off-lake during foraging wading bird reconnaissance—one on a gator farm near Lakeport, FL and another at the Martin County Florida Power and Light Reservoir.

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Size

Season-wide nest effort for all wading birds peaked at 774 nests (Table 2). Nest effort among Great Blue Herons, Great Egrets, Snowy Egrets, White Ibis, and Glossy Ibis was 550. This estimate is important because the pre-1989 record for Lake Okeechobee includes only those 5 species (David 1994). By comparison, this year's nest effort ranked third lowest on record. Only counts from 1971 and 1981 ranked lower with 130 and 520 nests, respectively.

Figure 2. Map of wading bird colonies found at Lake Okeechobee from January to June 2007.

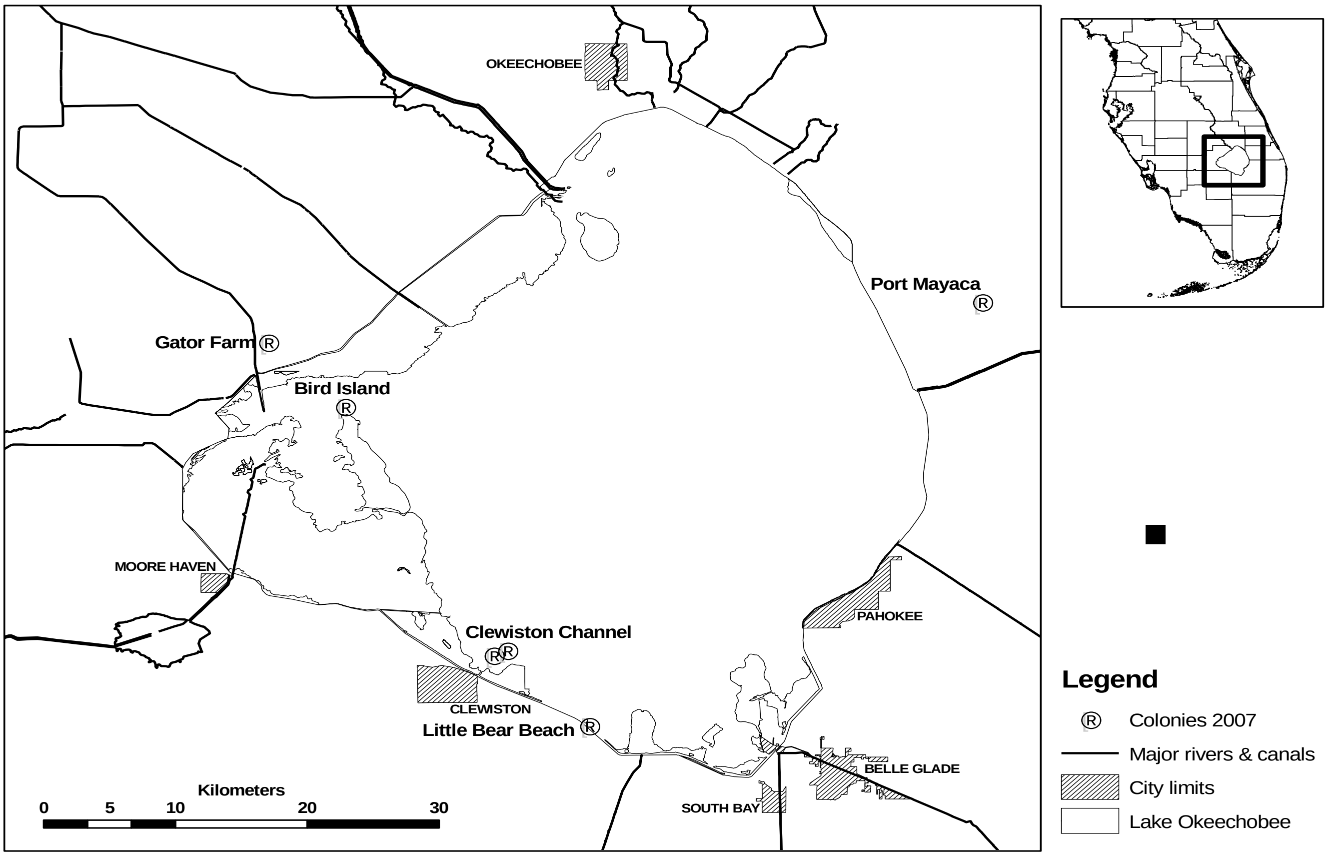

Table 2. Timing and nest effort for species breeding in wading bird colonies during 2007 at Lake Okeechobee. Italics denote species peak nest effort.

Month	ANHI	GREG	SNEG	TRHE	LBHE	GBHE	CAEG	GLIB	WHIB	WOST
January	---	---	---	---	---	---	---	---	---	---[1]
February	---	---	---	---	---	---	---	---	---	---
March	---	---	---	---	---	---	---	---	---	---
April	---	*7*	*543*	*157*	---	---	15	*41*	---	*12*
May	---	7	137	107	*14*	---	886	5	---	11
June	*18*	---	---	---	4	3	*1,260*	---	---	11

[1] Species undetected during monthly survey efforts.

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In 2007, we observed no on-lake nesting among Great Blue Herons, Great Egrets, Little Blue Herons, or White Ibis. Similarly, in 1971, no Great Egrets and no Great Blue Herons were detected, and in 1981, Great Egrets nested but not Great Blue Herons or Glossy Ibis (David 1994). The three seasons shared similar hydrological patterns as well, which were characterized by low lake stages to start the breeding season and below average rainfall during the preceding wet season. These pre-conditions likely contributed to the poor reproductive performance of wading birds at Okeechobee despite a favorable recession throughout the breeding season.

Timing and Success

No wading bird nesting was detected via aerial surveys until April 2007. Although on April 3, 2007, we detected Snowy Egrets and Tricolored Herons carrying nest material near Clewiston Spit and Bird Island via ground surveys. Nest monitoring efforts began on April 11. We found that 43% of nests had full clutches, suggesting that courtship and nest building began during the second or third week of March, which was similar to the timing of small ardeid nest initiation last year. However, colonies appeared to already be under considerable depredation pressure, and by the following week, 38% of nests had failed. During monitoring visits, we observed Boat-tailed Grackles depredating nests, and Fish Crows carrying eggs out of the colony. River otters were also commonly seen in colonies and are documented nest predators of other colonially breeding birds (Verbeek and Morgan 1978, Quinlan 1983).

By May 8, predation pressure, complete recession of water surrounding the island, and an intense storm event a few days prior to the visit combined to trigger wholesale abandonment of the largest colony at Clewiston Spit. The neighboring Clewiston Channel colony did not wholesale abandon with Clewiston Spit and fledged young. Nesting at Little Bear Beach began following abandonment of Clewiston Spit and may have been a re-nesting effort, but was abandoned by June surveys. All told, only 12% of 179 monitored nests fledged young—10 Snowy Egret and 11 Tricolored Heron nests, 4 nests from Bird Island and 17 from Clewiston Channel.

At Bird Island, Cattle Egrets outnumbered small ardeids 8:1 by May 9. Many Tricolored Herons and Snowy Egrets without chicks began to abandon. Glossy Ibis abandoned entirely. On May 15, only 22% of the original 96 wading bird nests remained. Even so, 1 Snowy Egret and 3 Tricolored Heron pairs eventually fledged young by the end of May.

Wood Storks

Most interesting this year was the development of a small Wood Stork colony in cypress trees on an alligator farm about 4 km north of Harney Pond along Highway 721. During aerial reconnaissance, we detected 12 Wood Stork pairs nesting on April 19. Maturity of Wood Stork chicks at the time suggested that storks began nesting between the first and second weeks of March.

Despite getting a late start, the colony fledged 22 young at the end of June. On June 14, plumage condition and movement away from the nest to adjacent branches suggested that chicks were 55-60 days old (Coulter et al. 1999). During our last visit on June 26, we observed only 9 chicks left at the colony and expect that all nestlings eventually fledged following the postflight period of attachment to nest sites (Kahl 1964, Coulter et al. 1999).

Discussion

Wading bird productivity can be limited by access to high quality foraging patches (Powell 1983, 1987, Frederick and Collopy 1989, Kushlan 1989, Gawlik et al. 2004). Yet the exact role that foraging patch dynamics plays in driving wading bird populations is somewhat unclear. Smith and Collopy (1995) found that high nest effort and high nest success were related to falling lake stages. They reasoned that recessions could either concentrate prey into shallower patches, increase access to preferred foraging habitats, or improve foraging efficiency.

Recession rates suggested that foraging conditions were good to fair throughout the breeding season. But given low lake levels and a drought that lasted from the previous wet season, conditions left much of the littoral zone waterless and unavailable to foraging birds following last year's recession. These conditions precluded interaction of hydrology with local floristic and topographic pattern, which is a necessary mechanism for enhancing prey availability across the landscape (Kushlan 1976, Frederick and Collopy 1989a, Gawlik 2002).

During 2007, drought conditions also reduced the availability of suitable nesting and foraging habitat, which acted to reduce carrying capacity of the region for colonially breeding wading birds. To begin, drought conditions reduced the availability of suitable colony locations. Nesting wading birds tend to prefer woody islands surrounded by water for colony sites (Frederick and Collopy 1989b, Smith and Collopy 1995). By January 2007, however, lake levels were low enough that few suitable colony sites remained in the littoral zone where birds traditionally nested. In May, the islands of the Clewiston Spit colony became completely exposed and may have been one of the factors that triggered abandonment.

In complement, drought conditions reduced the availability of suitable wading bird foraging habitat as well. On-lake foraging observations indicated that wading birds were limited to feeding in grass and bulrush beds along the margin of the nearshore and littoral zones to start the season, because water had receded into the near-shore zone. Only the exterior fringes of the littoral zone remained inundated to start the nesting season, and isolated pools of concentrated prey were sparse because patches with suitable water depths were still contiguous with the pelagic zone where fish could disperse into lower densities (Chick and McIvor 1994). These fringes dried-down completely by the middle of May, leaving wading birds only able to forage within the shallow, wide-open-water, nearshore and pelagic zones (Fig. 3).

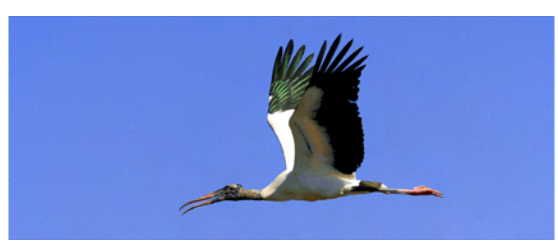

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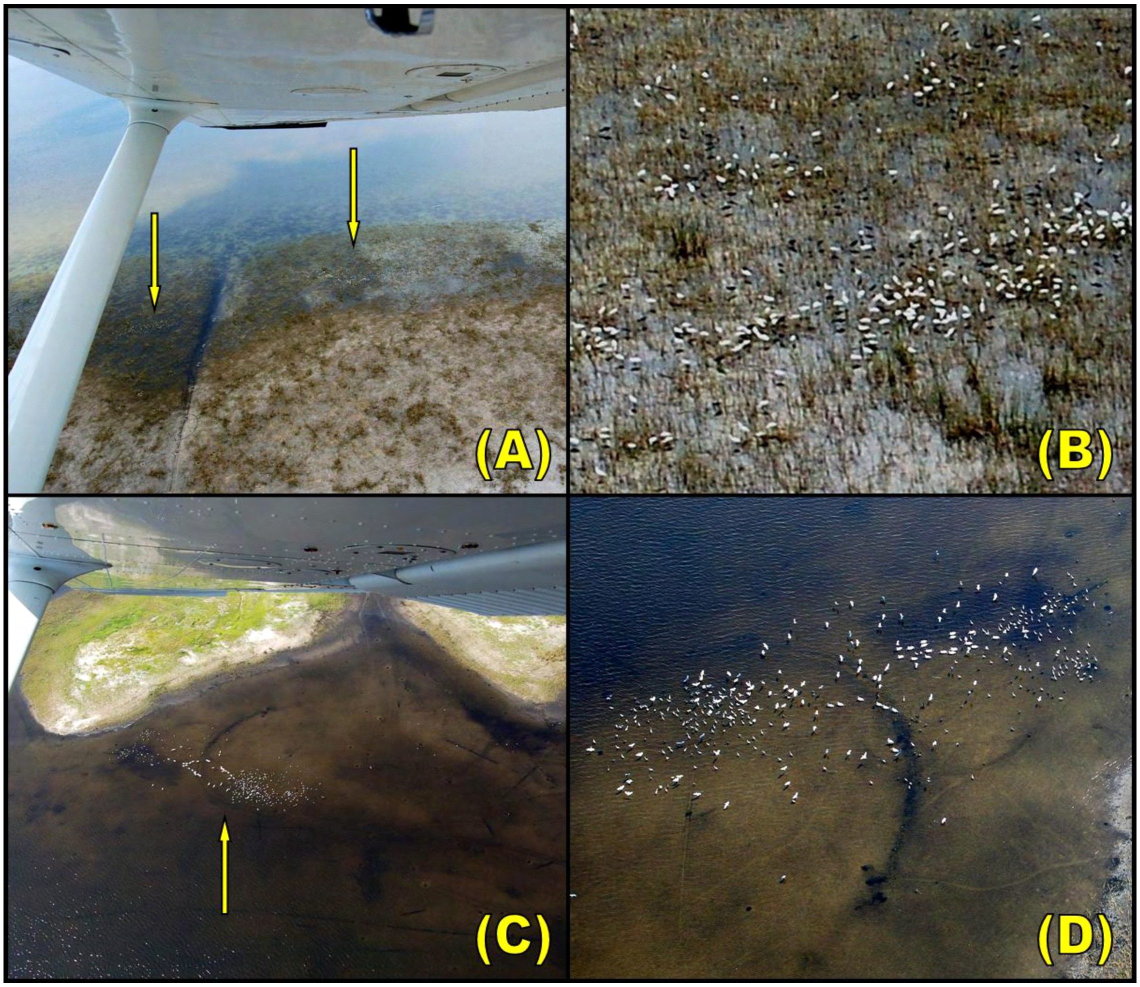
Figure 3. Landscape and zoomed views of foraging wading birds at Lake Okeechobee, FL during the 2007 nesting season. Figs. 3A & 3B depict wading birds foraging in grass beds along littoral zone fringes in February. Figs. 3C & 3D depict wading birds foraging in shallow, wide-open, nearshore areas in May. Notice both foraging areas were still hydrologically connected to the pelagic zone. Yellow arrows mark foraging flock locations in landscape views.



We observed a 93% reduction in nest effort between the 2006 and 2007 breeding seasons and suspect that this year's poor reproductive effort was associated with drought conditions that limited prey and habitat availability. Hydrological conditions between 2007 and other correspondingly low nesting years (i.e. 1971 and 1981) exhibited similar patterns. For each of these years, the region received below average rainfall accumulation and lake stages remained low (< 14.5 ft) during wet season months, which precluded inundation of the littoral zone. Then, a steady recession across the dry season brought lake stages below 11 ft, which left the littoral zone completely waterless and exposed the lake bottom in many near shore areas. The hydrological similarity between these three breeding seasons suggested that persistent drought conditions may negatively affect wading bird reproductive effort in the Okeechobee area.

Even so, we should note that low nest effort has also been linked to high lake stages (> 15 ft NGVD29) at the opposite extreme of the management envelope. David (1994) reported that prolonged high water levels during the late 1970s and early 1980s coincided with declines in wading bird nest effort. And in 1984, the only other year with extreme low wading bird nest effort (< 1,000 nests), lake stages had remained high since August 1982, and breeding season hydrology was characterized by periodic reversals and increasing lake levels. Additional research into the effects of different hydrological scenarios on habitat availability and wading bird reproduction is on-going.

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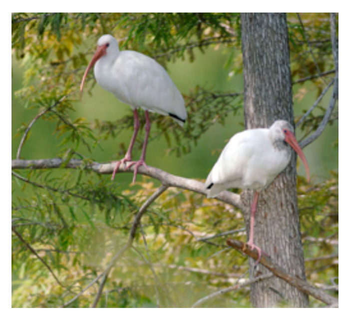

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KISSIMMEE RIVER

Introduction/Background

Prior to its channelization, the Kissimmee River, its 1 – 3 km wide floodplain, and surrounding wetland/upland complex supported substantial numbers of foraging and nesting wading birds (National Audubon Society, 1936 – 1959). Between 1962 and 1971, the Kissimmee River was channelized and its headwater lakes regulated, resulting in the drainage of the majority of its floodplain wetlands and a substantial reduction in the number of wading birds (excluding cattle egrets) using the system (Williams and Melvin, 2005). The Kissimmee River Restoration Project, which was authorized in 1992, seeks to restore ecological integrity to the middle portion of the original river system via 1) reconstruction of the physical form of the river (i.e., canal backfilling, removal of water control structures, and recarving/reconnecting river channels); and 2) reestablishment of historical (pre-channelization) hydrologic (i.e., discharge and stage) characteristics through modifications to regulation schedules of headwater lakes. When completed, the project will restore approximately 104 km^2 of river-floodplain ecosystem, including 70 km of continuous river channel. The restored area is expected to experience seasonal flood pulses and recessions that are favorable for wading bird reproduction. To date, approximately one third of project construction has been completed. All construction is scheduled for completion by the end of 2012; new regulation schedules for headwater lakes will be implemented in 2010. Wading bird responses to the restoration project will be monitored through 2017.

Methods

As part of the Kissimmee River Restoration Project evaluation program, we performed systematic aerial surveys (May 17, Jun 11, Jul 16) to search for wading bird nesting colonies within the floodplain and surrounding wetland/upland complex of the Kissimmee River. Flights dedicated specifically for colony surveys were not conducted from January through April due to a position vacancy. Surveys began at the S65 structure at Lake Kissimmee and proceeded southward to the S65-D structure (Fig. 1). Observers were placed on both sides of a helicopter flying at an altitude of 244 m along east-west transects spaced 2 km apart. Each transect spanned the 100 yr flood line of the river plus an additional 3 km east and west of the flood line. In addition to dedicated flights for colony surveys, nesting colonies were also monitored, when encountered, during separate aerial surveys of foraging wading birds. These surveys (Mar 26, Apr 23, May 21, Jun 18, Jul 23) were flown at a lower altitude (30 m) and were limited to the area within the 100 yr flood line of the river between S65 and S65-D. Once a colony was located, nesting species and the number of active nests were visually estimated by both observers. The number of nests reported for each colony represents the maximum number of nests for each species. Nesting success was not monitored, but two ground surveys (May 8, Jun 29) were conducted at the C-38 colony to obtain more accurate nest counts and determine the presence of less visible dark-colored herons.

Results

One colony containing an estimated 227 nests was observed during the 2007 season, including 226 CAEG and 1 TRHE (Fig. 1). The colony was first encountered by boat on May 8 when birds were either building nests or incubating eggs. The colony was subsequently abandoned sometime between discovery and the May 17 survey flight. Four of five 2006 colonies were absent from this years surveys, but it should be noted that dedicated flights were not conducted this year during the typical peak of nesting activity (Feb-Apr; Table 1).

It is unlikely, however, that any colonies formed and successfully fledged young prior to our May 17 flight given the below-average nesting activity and unfavorable foraging conditions throughout the region (see Kissimmee River Foraging Densities below) and lack of observations during the Mar 26 survey for foraging birds. As in 2006, nearly all nests occurred in a single CAEG colony. The abandonment of this colony in mid-May may have been due in part to the absence of nesting stimuli from native wading bird species that may have lacked sufficient aquatic prey to initiate breeding (Belzer and Lombardi 1989)

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Table 1. Peak numbers of wading bird nesting colonies inside or within 3 km of the Kissimmee River 100 yr flood line between the S65 and S65-D structures. Surveys were conducted Mar-Jun, 2004; Mar-Jun, 2005; Feb-Jun, 2006; and May-Jul 2007.

Latitude	Longitude	Colony Name	Year	ANHI	CAEG	GBHE	GREG	TRHE	Colony Total
81 13.219	27 42.946	42W	2004	-	-	-	-	-	**-**
			2005	-	-	-	-	-	**-**
			2006	-	-	-	8	-	**8**
			2007	-	-	-	-	-	**-**
81 04.466	27 22.853	C38 Caracara Run	2004	-	-	-	-	-	**-**
			2005	-	-	-	-	-	**-**
			2006	-	500	-	-	-	**500**
			2007	-	226	-	-	1	**227**
81 16.527	27 32.088	Cypress West	2004	-	-	-	-	-	**-**
			2005	-	-	-	21	-	**21**
			2006	-	-	-	25		**25**
			2007	-	-	-	-	-	**-**
81 00.380	27 22.620	New Chandler Slough	2004	-	-	-	-	-	**-**
			2005	-	-	-	-	-	**-**
			2006	-	-	-	40	-	**40**
			2007	-	-	-	-	-	**-**
81 04.649	27 21.076	Orange Grove	2004	-	-	-	-	-	**-**
			2005	30	-	5	60	-	**95**
			2006	20	-	4	60		**84**
			2007	-	-	-	-	-	**-**
81 06.442	27 37.791	Pine Island	2004	-	-	-	-	-	**-**
			2005	-	400	-	-	-	**400**
			2006	-	-	-	-	-	-
			2007	-	-	-	-	-	**-**
Total Nests			**2004**	**0**	**0**	**0**	**0**	**0**	**0**
			2005	**30**	**400**	**5**	**81**	**0**	**516**
			2006	**20**	**500**	**4**	**133**	**0**	**657**
			2007	**-**	**226**	**-**	**-**	**1**	**227**

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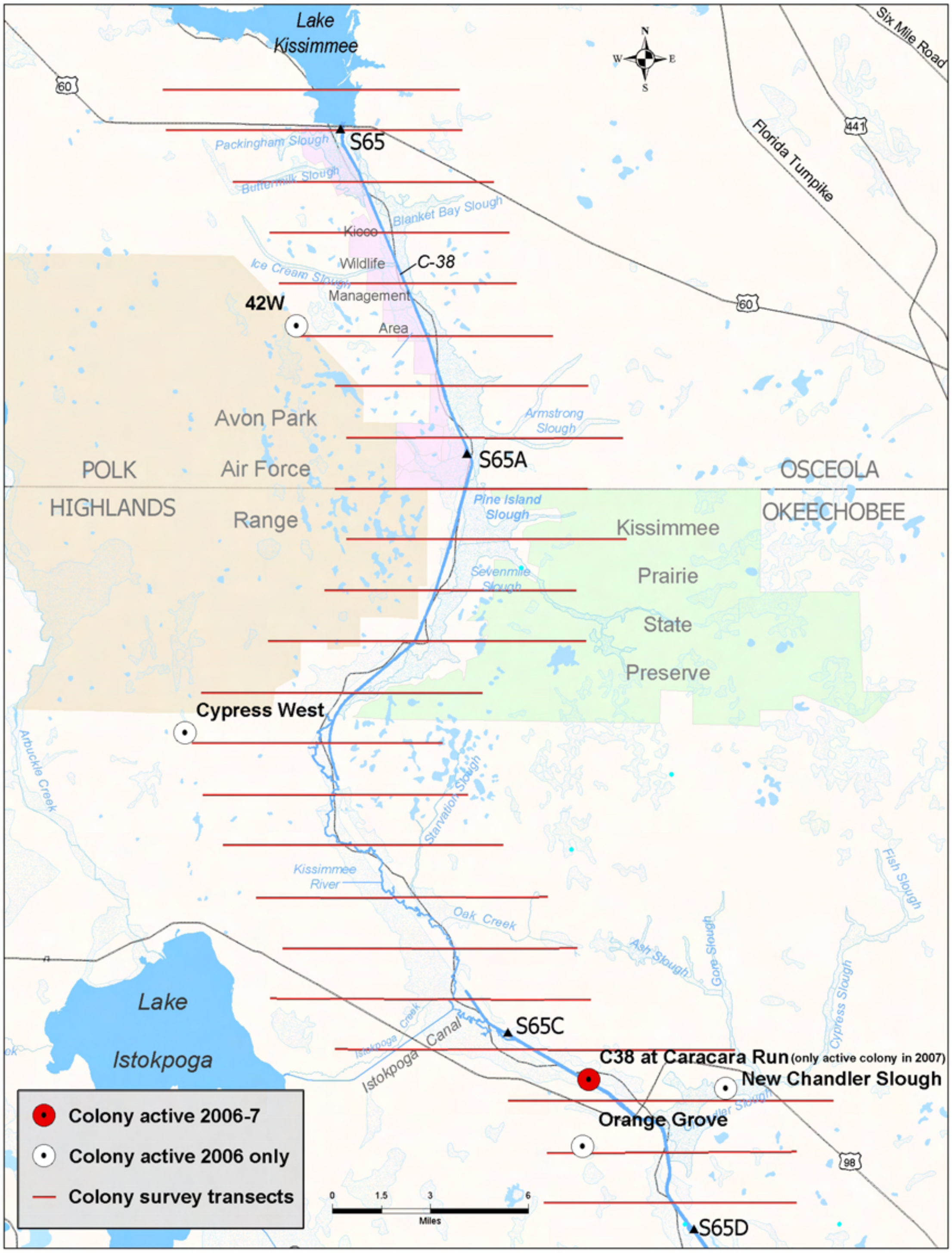
Figure 1. Transect layout and locations of nesting colonies within the Kissimmee River floodplain and surrounding wetland/upland complex during 2006-7.



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REGIONAL WADING BIRD ABUNDANCE

EVERGLADES NATIONAL PARK AREA

Methods

Systematic reconnaissance flights (SRF's) were performed monthly between Dec 2006 and May 2007. Flights were conducted over 3 to 4 consecutive days using a fixed-wing Cessna 182 at an altitude of 60 m. The area covered included the Everglades National Park mainland, the zone east and southeast of the main park entrance, and the southern region of Big Cypress National Preserve. The area was surveyed using transects oriented E to W and separated by 2Km (see Figure 1). Wading birds were counted, identified and geographically located using GPS units. Changes in surface water patterns (hydropatterns) were also recorded. Five categories were used to describe the hydropatterns: DD - absence of surface water and no groundwater visible in solution holes or ponds; WD - absence of surface water but groundwater present in solution holes or ponds; DT - ground surface area mostly dry but small scattered pools of surface water present and groundwater visible in solution holes or ponds; WT - ground surface area mostly wet but small scattered dry areas; and WW - continuous surface water over the area.

Data obtained during each SRF were compiled into a database, which contains the information collected since 1985 to the present. During this period, SRF surveys were not conducted during December 1984, December 1987 and January 1998. Missing data for those months were estimated using years with complete sets of data. From those years, it was calculated the overall percentage of increase or decrease from month to month in order to estimate missing values. In some years, due to personnel constraints, only one observer was used to collect those data. This situation occurred during the surveys of April 1990, May 1990 and from January 1991 to May 1991. Finally, some transects were missing for one observer during April 2004 and May 2005. Densities of birds were estimated using a 2X2 Km grid. The number of birds counted during the SRF inside the 300m width surveyed stripe were extrapolated to the rest of the 4Km^2 cell dividing the number of birds observed by 0.15 for surveys were data from two observers were available. In cases were only data from one observer were available the number of birds inside the 150m strip were extrapolated to the rest of the cell by dividing the birds observed by 0.075.

Results

During this year survey period (December 2006 – May 2007) an increase of thirteen-percent in the abundance of wading birds was observed, for all species combined, in comparison to the previous year (Figure 2). This represents the third consecutive year that an increase in the number of birds was observed since 2005. This year's increase contributed to the overall significant increasing trend observed since 1985 to present, when a linear regression model was used to fit the data ($F=7.112$; $P=0.014$).

Seven of the nine species of birds studied, showed an annual increase in their numbers in relation to those observed in 2006 (see Figure 3). Great White Heron (GWHE) showed an increase of 56%; followed by small dark herons (SMDH) with 34%, Great Blue Herons (GBHE) 31%, Wood Stork (WOST) 20%, White Ibis (WHIB) 19% and small white heron (SMWH) as well as Great Egrets (GREG) with 5% increase for each one. Two species showed a decline in number of birds; those species were Roseate Spoonbill (ROSP) with 7% decreased and Glossy Ibis (GLIB) with 52%.

Figure 3 also shows the annual estimated number of birds by species from 1985 to present. Despite the annual fluctuations observed for each of the different species, a general increase was observed in five of the nine species. Those species are in order of significance; GREG, GBHE, WHIB, SMWH and WOST. Once again, a linear regression model was used to determine the general trend for each species.

This is the fifth consecutive year, since 2003, that GREG showed an increase in the annual estimated number of birds. GBHE also has showed consecutive increases since 2004. Finally, WHIB, SMWH and WOST have been showing increases in their numbers since 2005. Estimates for the number of GLIB and ROSP have declined during the past two years, while SMDH have exhibited an increase during the last three years. Despite the opposite recent trends observed in those species, the overall long term trend since 1985 was basically neutral. Finally, GWHE is the only species that displayed an overall decline; despite increases observed during the last two years.

Although this type of analysis can provide some general ideas of the trends in the number of individuals observed for each species or groups of birds through the years, additional studies and more data analysis will be necessary in order to evaluate the significance of these observations and its relevance to the wading bird populations occurring in Everglades National Park.

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Figure 1. Map of ENP and southern Big Cypress National Preserve with sampling transects and drainage basins.

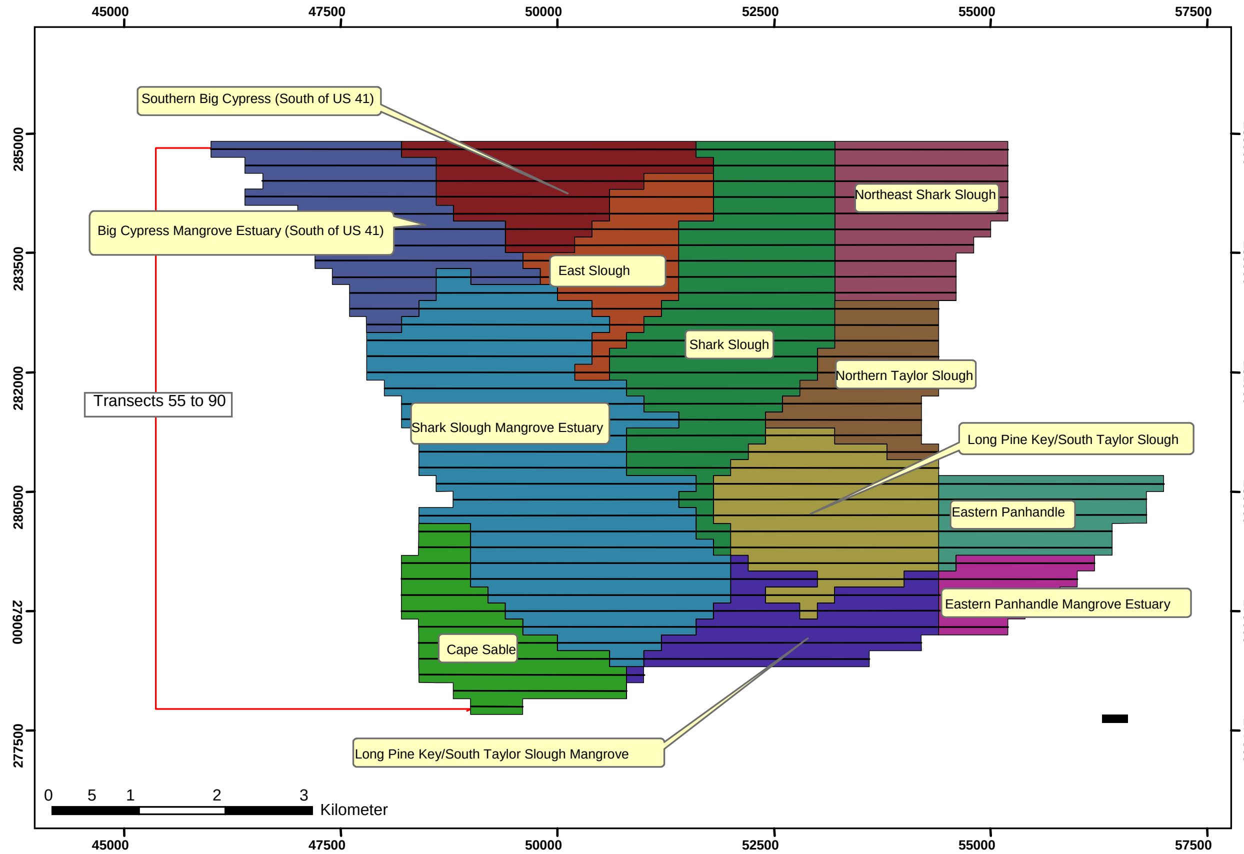

Figure 2. Estimated number of wading birds (all species pooled) observed from the months of Dec-May from 1985 to 2007. Red marks represent years with estimated missing data for one month.

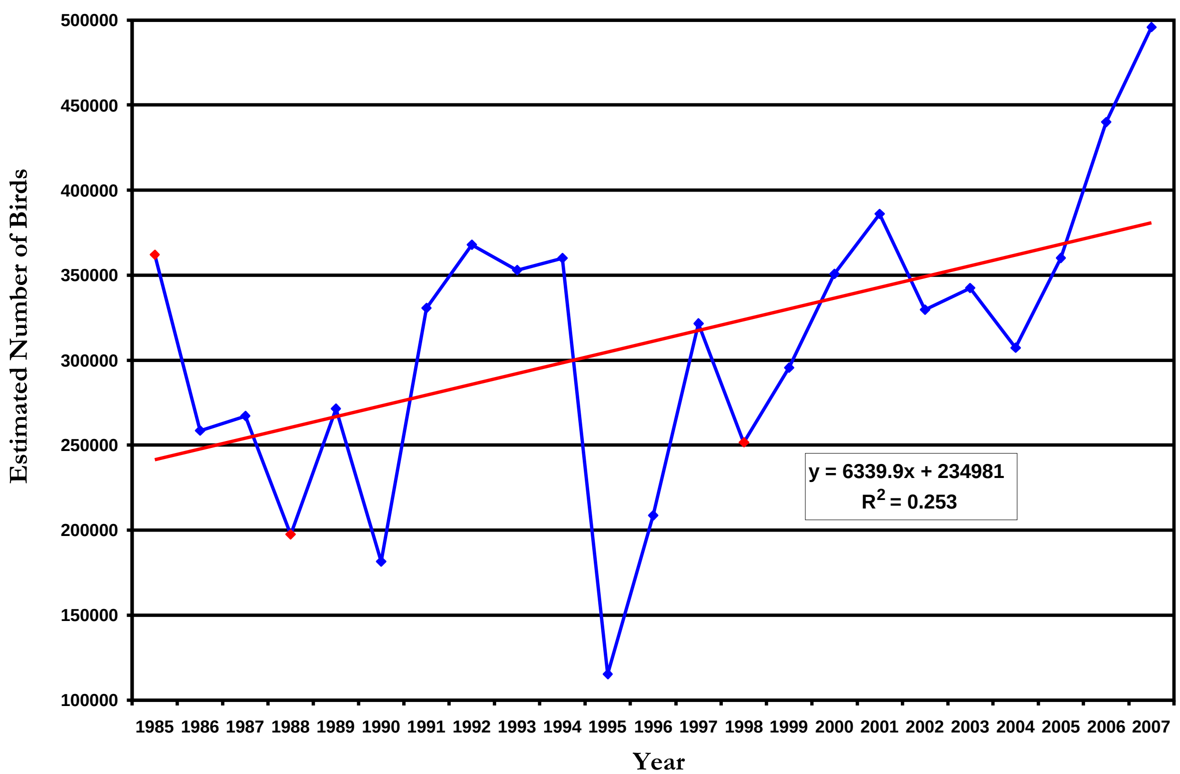

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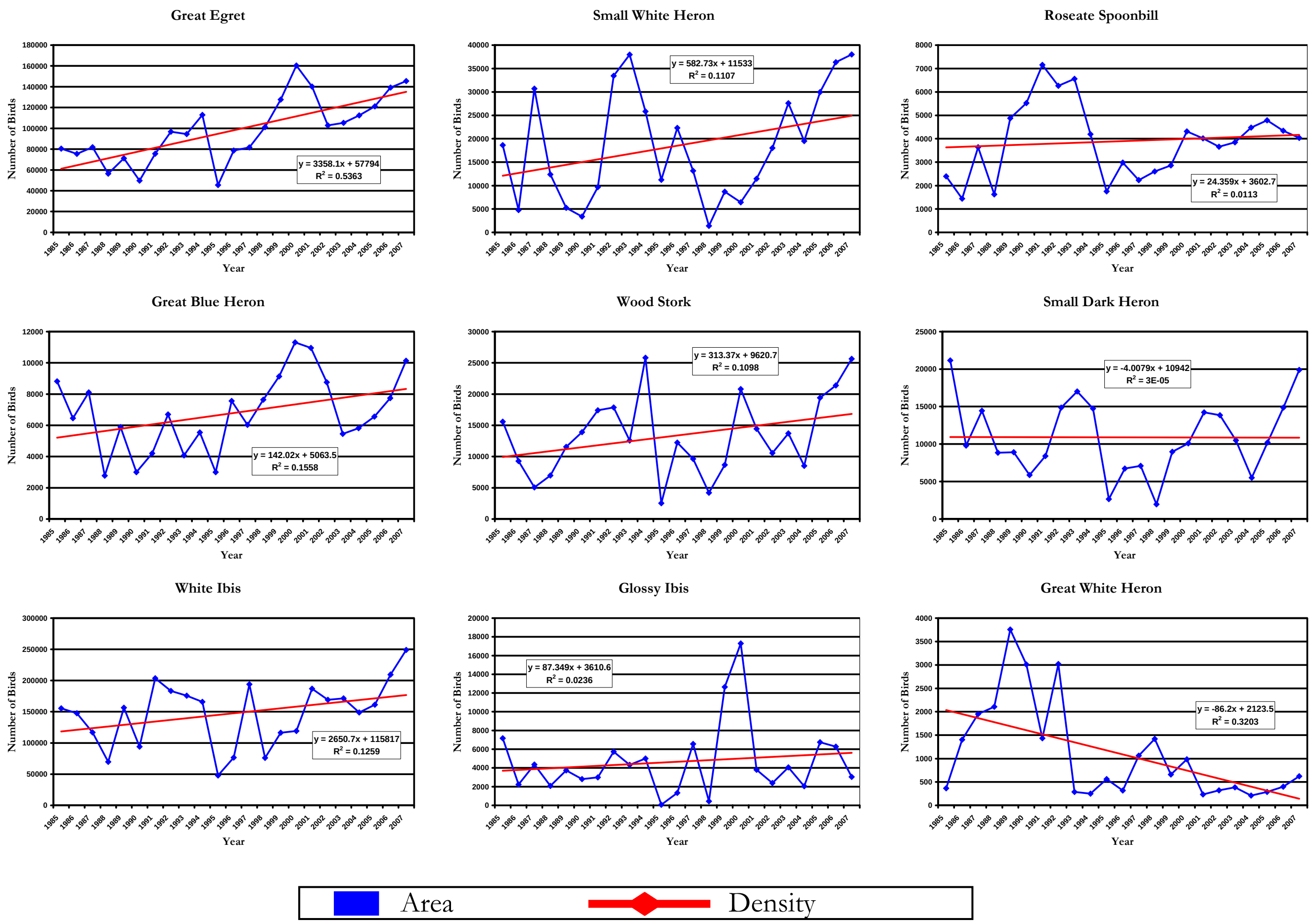
Figure 3. General trends in wading bird populations based on the total number of birds estimated during the surveys performed each year in the Everglades National Park from 1985 to the present.



The maximum density of birds, regardless of the species, occurred this year during the month of December (see Table 1). It was during this month that the highest numbers of GREG, GBHE, SMDH, SMWH and ROSP were observed. Other species such as WHIB, GLIB and WOST reached their peak numbers in February, while GWHE peaked in the month of March. May was the month with the fewest number of birds for all the species combined. It was also in May where the lowest number of birds was observed for all the species but for GLIB, ROSP and GWHE. For those particular species, the lowest concentration of individuals occurred during December, March and April respectively

The most abundant species during the survey period was WHIB representing approximately half of the total number of birds observed followed by GREG (29.3%). These two species combined accounted for almost 80% of the total of birds observed this year. The remaining 20% was composed of the following species SMWH (7.6%), WOST (5.2%), SMDH (4.0%), GBHE (2.0%), ROSP (0.8%), GLIB (0.6%), and GWHE (0.1%).

Table 2 shows the distribution and abundance of wading birds for each of the different drainage basins. Shark Slough (SS) contained the highest number of wading birds (23%), followed by Shark Slough Mangrove Estuary (SSME) with 20%, and East Slough (ES) and Big Cypress Mangrove Estuary (BCME) with 11% each one. These four basins combined, made up 65% of the total number of birds observed during the entire season. In contrast; the basins with the lower number of birds were Northern Taylor Slough (NTS) with less than 1%, Eastern Panhandle Mangrove Estuary (EPME) 1% and Eastern Panhandle with only 2%. A great concentration of birds was observed during December and January at SSME in relation to the other basins. By February, SS became the basin with the greatest number of birds and remained like that until May.

Changes in hydro-patterns and bird distribution observed this season were less pronounced than in the previous year (see Figure 4). The greatest changes in the area covered by the different hydro-patterns took place at the extreme categories. From December to May, the original extent of the area covered by WW was reduced from 28% to 16% (560 Km² reduction), while DD area experienced an increase going from 10% at the beginning of the season to 22% at the end of the season (608 Km² increase). Intermediate categories such as WT and WD showed very slight changes throughout the season. The areal extend for WT decreased from 31% to 28% (148Km²), while WD increased from 13% to 15% (96 Km²). Finally, very small fluctuations occurred in the middle category, DT, with no more that 3% change at the most.

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Table 1. Estimated abundance of wading birds in the Everglades National Park and adjacent areas, Dec 2006- May 2007.

Species	Dec-06	Jan-07	Feb-07	Mar-07	Apr-07	May-07	Total
GREG	36,525	29,251	33,089	24,664	15,929	5,991	145,449
GBHE	3,769	1,721	1,687	1,390	1,200	367	10,134
SMDH	5,924	2,983	3,858	3,359	2,762	1,005	19,891
SMWH	13,070	6,150	7,195	5,457	4,646	1,497	38,015
WHIB	56,977	37,304	60,926	56,764	27,300	9,780	249,051
GLIB	86	240	1,068	986	427	234	3,041
WOST	5,928	4,968	7,260	4,554	2,873	62	25,645
ROSP	1,153	641	802	240	808	394	4,038
GWHE	97	89	129	135	62	111	623
TOTAL	123,529	83,347	116,014	97,549	56,007	19,441	495,887

Table 2. Estimated abundance of wading birds (all species combined) for the different drainage basins in the Everglades National Park, Dec 2006 – May 2007.

Month	SBC	BCME	SS	NESS	ES	SSME	NTS	LPK/ STS	EP	CS	LPK/ STSM	EPME	Total
Dec-06	12,723	17,339	16,477	2,723	11,523	32,042	283	5,310	4,455	12,885	6,523	1,246	123,529
Jan-07	4,686	8,816	16,826	4,243	7,742	23,175	432	1,626	2,468	9,544	2,992	797	83,347
Feb-07	9,323	15,457	22,607	8,711	16,007	20,170	332	1,866	1,290	4,058	13,184	3,009	116,014
Mar-07	5,045	5,801	30,740	12,838	15,464	13,840	34	2,127	873	4,818	5,781	188	97,549
Apr-07	1,208	6,738	24,858	4,558	3,563	4,996	20	3,948	1,051	1,432	3,500	135	56,007
May-07	725	1,103	3,803	1,937	2,023	2,616	35	1,263	664	3,095	2,014	163	19,441
Total	33,710	55,254	115,311	35,010	56,322	96,839	1,136	16,140	10,801	35,832	33,994	5,538	495,887

SBC = Southern Big Cypress (South of US 41)
BCME = Big Cypress Mangrove Estuary (South of US 41)
SS = Shark Slough
NESS = Northeast Shark Slough
ES = East Slough
SSME = Shark Slough Mangrove Estuary
NTS = Northern Taylor Slough
LPK/STS = Long Pine Key / South Taylor Slough
EP = Eastern Panhandle
CS = Cape Sable
LPK/STSM = Long Pine Key / South Taylor Slough Mangrove Estuary
EPME = Eastern Panhandle Mangrove Estuary

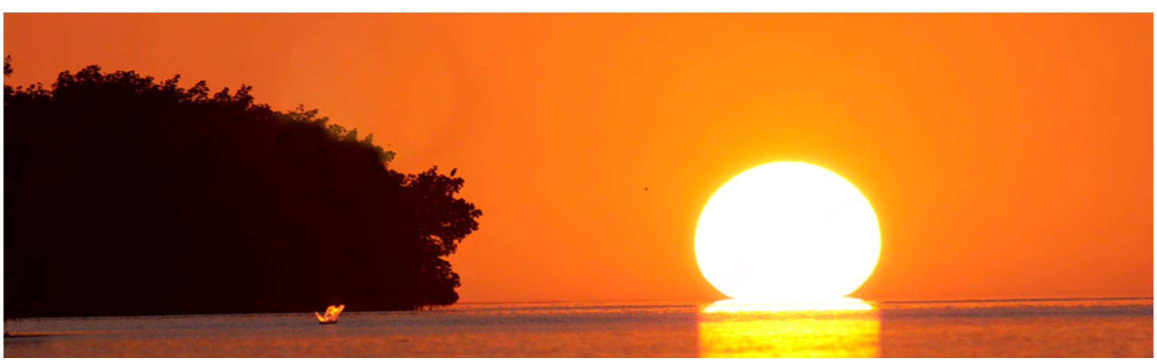
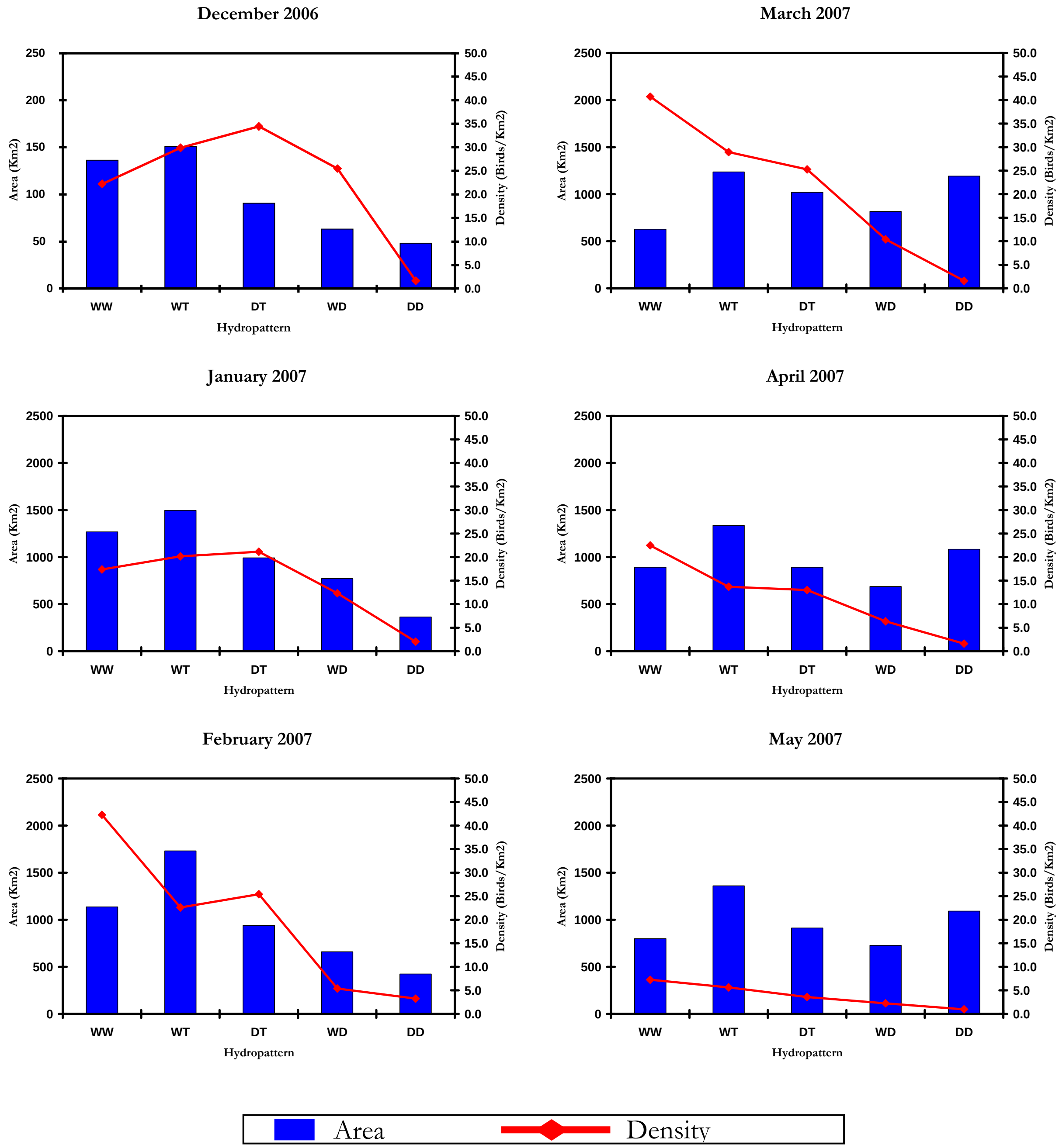


Figure 4. The 2007 areal extent and density of wading birds (all species pooled) in each surface water category.



WW = continuous surface water; WT = mostly wet with scattered dry areas; DT = mostly dry with small scattered pools of water; WD = dry with water only in solution holes; DD = dry surface.

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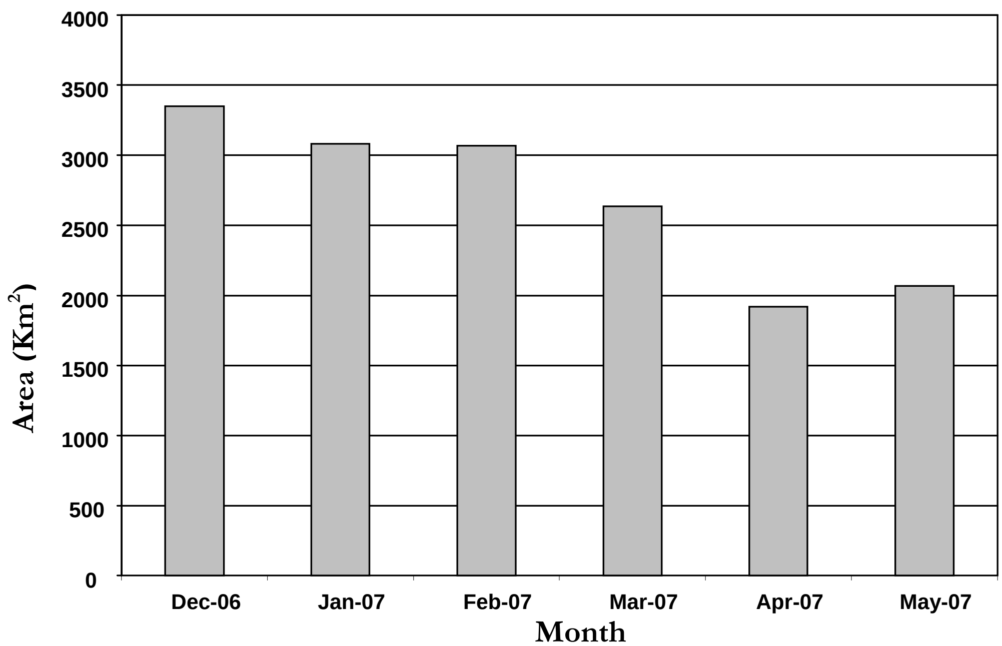
Figure 5. Monthly changes in wading bird areal utilization in the Everglades National Park from Dec-2006 to May-2007



During December and January, the highest densities of birds were mainly located in the DT hydro-pattern. By February and March, as water receded, birds began to concentrate in WW, WT and DT areas respectively. As water depth continued to decrease during the following months, WW, WT and DT areas continued holding the higher densities; however it was obvious that great numbers of birds were leaving the study area. The fact that WW areas are completely covered by water, do not necessarily implies that those areas are to deep for wading birds to forage. Overall, as water recedes, low water levels turned these areas into new territories accessible to foraging birds.

Birds were found foraging in 68% of the study area during the month of December (see Figure 5). This represents the month were birds were more widely distributed. The rainfall deficit observed during this year was the probable cause of this early widespread bird distribution. As water continued to receded, birds began to concentrate. By January and March, birds utilized 63% of the total available area. During March, birds concentrated in an area slightly larger than half of study area, while April was the month with the smallest area utilized. During this month birds were concentrated in only 39% of the total available area. At the end of the season (May), a slight increase in areal utilization (42%) was observed. This increase was probably a result of two major rain events which occurred during the survey.

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WADING BIRD SURVEYS FOR WATER CONSERVATION AREAS AND BIG CYPRESS NATIONAL PRESERVE

Methods

Wading bird surveys were flown with a fixed wing aircraft at an altitude of about 60 meters along parallel transects with 2-km spacing each month from January to July 2006. Wading birds were identified to species when possible, enumerated, their locations recorded, their data entered into a database, and summarized into tables. Densities of each species were separated into 4-km^2 cells and plotted onto maps. Data were recorded using HP720 palm top computers linked to GPS. The data were downloaded into a computer spreadsheet, edited for errors, and compiled using a program written in Dephi programming language.

Results

In the Water Conservation Areas, monthly wading bird relative abundance was generally higher during 2007 than 2006. In the Water Conservation Areas, the maximum relative abundance was observed during April 2007 (108,034; Table 1) and during May 2006 (87,887). In 2007, February, March and April relative abundances were higher than the same months in 2006. The wading bird abundances in June 2006 and July 2006 were higher than the respective months in 2007. During 2007, there were increasingly drought-like conditions from January to April then and an increase in water with the increase in rain during June and July. In the Big Cypress National Preserve, monthly wading bird abundances were slightly higher in 2007 than 2006. The maximum relative abundance was observed during February 2007 (34,407; Table 2) and during February 2006 (32,480). In the Big Cypress National Preserve, February, June and July relative abundances were higher in 2007 than in 2006; March and April wading bird abundances were lower in 2007 than 2006. In the Big Cypress National Preserve, monthly wading bird abundance peaked in February 2007 then declined until April 2007 in response to very dry conditions then increased with the increase in rain in June 2007 and July 2007. Final reports from 1996 to 2006 are currently available.

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Table 1. Wading Bird Estimated Abundances for the Water Conservation Areas, 2007.

Species	January	February	March	April	June	July	Mean
Great Blue Heron	720	680	1093	1320	1173	760	957.67
Great Egret	31527	27800	44274	55474	12327	11527	30488.17
Small Dark Heron	220	380	513	1073	540	220	491.00
Small White Heron	853	633	700	900	600	1000	781.00
White Ibis	40227	69787	48300	38254	6560	5327	34742.50
Glossy Ibis	767	1800	1133	1473	33	53	876.50
Wood Stork	507	947	5287	6860	40	47	2281.33
Great White Heron	1213	1573	1727	2573	1040	0	1354.33
Roseate Spoonbill	187	33	107	107	27	0	76.83
All Species	76220	103634	103134	108034	22340	18933	72049.17

Table 2. Wading Bird Estimated Abundances for Big Cypress National Preserve, 2007.

Species	January	February	March	April	June	July	Mean
Great Blue Heron	240	173	267	140	60	107	164.50
Great Egret	10640	13000	5620	1027	3227	5153	6444.50
Small Dark Heron	33	513	13	33	140	107	139.83
Small White Heron	780	147	453	87	353	527	391.17
White Ibis	17427	18193	2193	940	2613	2527	7315.50
Glossy Ibis	7	287	40	0	27	7	61.33
Wood Stork	1267	1280	907	520	260	27	710.17
Great White Heron	813	807	187	173	300	387	444.50
Roseate Spoonbill	13	7	0	0	0	0	3.33
All Species	31220	34407	9680	2920	6980	8840	15674.50

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KISSIMMEE RIVER FORAGING DENSITIES

Aerial surveys were used to measure the densities of wading birds. Surveys were conducted approximately monthly during the baseline period (pre-restoration; 1996–1998) and have continued after Phase I of the restoration project was completed in 2001. Restoration is expected to bring increased use of the floodplain by long-legged wading birds (excluding Cattle Egrets). Furthermore, mixed species wading bird rookeries are anticipated to regularly form on and near the floodplain and tributary sloughs once abundant food resources and appropriate hydrology have been reestablished.

To investigate densities of wading birds on the floodplain, east-west aerial transects (n = 218) were established at 200 m intervals beginning at the S-65 structure and ending at the S-65D structure (see Figure 1 for structure locations). Each month, transects were randomly selected for counts until a minimum of 15 percent of the 100-year floodplain was surveyed in both the Phase I and unrestored portion of the river/floodplain. Surveys were conducted via helicopter flying at an altitude of 30.5 m and a speed of 130 km/hr. A single observer counted all wading birds and waterfowl within 200 m of one side of the transect line. Because it is not always possible to distinguish Tricolored Herons (*Egretta tricolor*) from adult Little Blue Herons (*E. caerulea*) during aerial surveys (Bancroft et al. 1990), the two are lumped into the category, small dark herons. Likewise, Snowy Egrets (*E. thula*) and immature Little Blue Herons were classified as small white herons (Bancroft et al. 1990). Densities of wading birds were calculated separately for restored and unrestored areas.

Because no quantitative data are available for densities or relative abundances of long-legged wading birds of the pre-channelized Kissimmee River, restoration expectations for responses by wading birds to the KRRP are based on reference data from aerial surveys of a flow-through marsh in Pool B that was built as part of the Kissimmee River Demonstration Project and for floodplain areas along Paradise Run, a portion of the Kissimmee River near Lake Okeechobee that still retains some channel flow and periodic floodplain inundation (Toland 1990; Perrin et al. 1982). The 3.5 km² flow-through marsh was constructed just south of the S65-A tieback levee during 1984–1985 and was manipulated to simulate inundation and overland flow that were typical of the pre-channelized Kissimmee River floodplain (Toth 1991). Based on these reference data, it is expected that annual dry season (December–May) densities of long-legged wading bird (excluding Cattle Egrets) will be ≥ 30.6 birds/km².

Prior to Phase I construction (baseline period), mean annual dry season densities of long-legged wading birds in the Phase I area averaged (± SE) 3.6 ± 0.9 birds/km² in 1997 and 14.3 ± 3.4 birds/km² in 1998. Since completion of Phase I, densities of long-legged wading birds have exceeded the restoration expectation of 30.6 birds/km² each year except 2007, averaging 37.8 ± 15.4 birds/km², 61.7 ± 14.5 birds/km², 59.6 ± 24.4 birds/km², 103.0 ± 31.5 birds/km², and 11.0 ± 2.1 birds/km² in the dry seasons of 2002, 2004, 2005, 2006, and 2007 respectively (2003 data were not collected; Figure 1). Furthermore, the lower limit of the 95 percent confidence interval (95% C.I.) has exceeded the expectation in three of five years. However, this dry season was the first since Phase I completion that the restoration expectation was not met and densities were similar to those observed during the 1998 baseline surveys. This is likely to be an effect of the extreme drought conditions experienced during the 2007 dry season rather than effects of Phase I restoration per se. Most floodplain foraging habitat was completely dry this year and was inundated only during a brief period (Sep 4-16, Hurricane Ernesto) in the wet season prior to an earlier than average fall recession. These conditions may have prevented significant prey base production within abandoned river channels and isolated wetlands and limited prey availability during the winter/spring breeding season (see nesting colony information above). Water levels have not returned to appropriate foraging depths throughout most of the floodplain as of mid-July 2007. Wading bird density remains low, with the exception of Cattle Egrets that continue to occur in significant numbers throughout the floodplain. In areas where water levels are currently returning to appropriate foraging depths with the onset of summer rains, it is likely that prey items are widely dispersed at low densities in newly inundated areas which precludes efficient foraging by wading birds. Anecdotal evidence from June and July survey flights indicates that birds were utilizing adjacent isolated wetlands in greater numbers outside of the floodplain where prey availability may have been greater. Excluding Cattle Egrets, White Ibis was the most common species in all 2007 dry season surveys, with Great Egret, Glossy Ibis, small white heron (Snowy Egret and immature Little Blue Heron), Great Blue Heron, and Little Blue Heron also commonly encountered. Wood Storks were observed only during the December survey.

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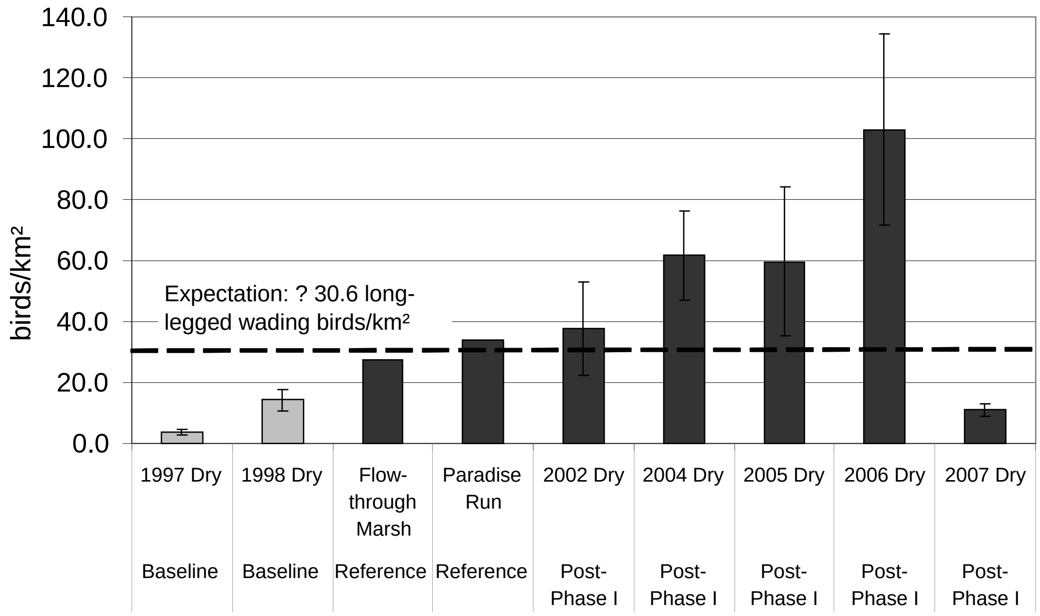
Figure 1. Baseline, reference, and post-Phase I densities (± SE) of long-legged wading birds (excluding cattle egrets) during the dry season (Dec-May) within the 100-year flood line of the Kissimmee River. Baseline densities were measured in the Phase I area prior to restoration. Post-restoration densities were measured beginning approximately 10 months following completion of Phase I.



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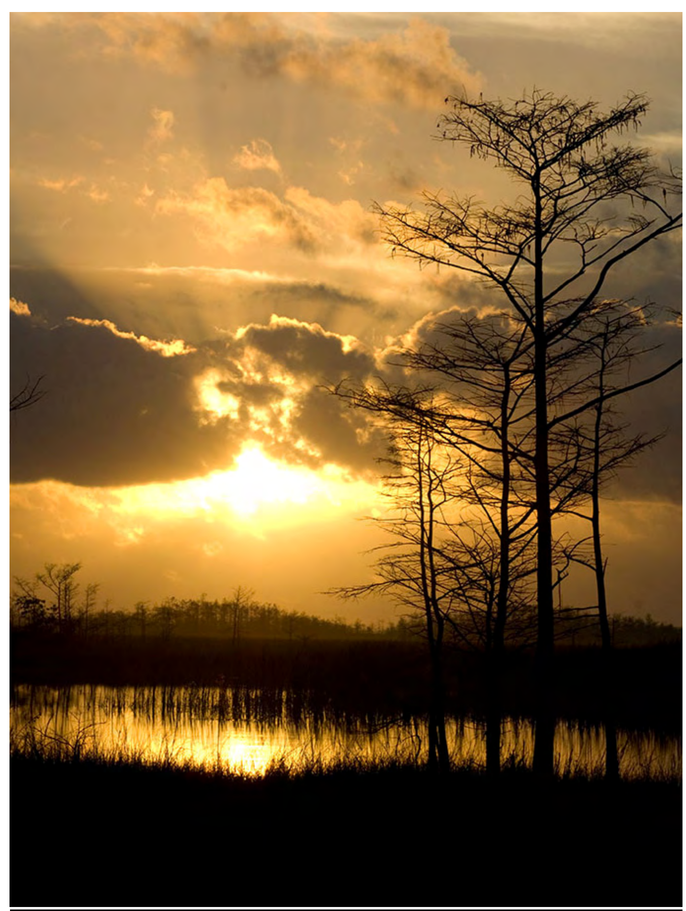

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STATUS OF WADING BIRD RECOVERY

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The purpose of these wading bird status reports is to summarize the annual nesting patterns of wading birds in the context of the wading bird performance measures that have been established for tracking progress by CERP and other ecosystem restoration programs in the south Florida wetlands. These status reports have been produced annually for each volume of the South Florida Wading Bird Report. Up to this point, these annual reports have summarized monitoring data for three parameters of wading bird nesting: numbers of nesting pairs, locations of nesting colonies, and timing of nesting (storks only). The species of wading birds reported in these summaries are Great Egret, Snowy Egret, Tricolored Heron, White Ibis, and Wood Stork. These data have been reported from the three Water Conservation Areas and mainland Everglades National Park only. Following is the results from the 2007 colony surveys, including an updated 3-year running average for numbers of nesting pairs.

Results

Numbers of Pairs: The combined total number of nesting pairs for four species for 2007 (Tricolored Herons are excluded from this total because no ground counts were conducted in ENP, and only partial ground counts were conducted in the WCAs) was 26,411 pairs, divided as follows: 5,193 pairs of Great Egrets, 217 pairs of Snowy Egrets, 20,661 pairs of White Ibis, and 340 pairs of Wood Storks. The 3-year running averages for 2005-2007 for these four species are 6,987 pairs, 4,559 pairs, 21,660 pairs, and 636 pairs, respectively. Comparisons with earlier running averages are shown in the accompanying table.

Colony Locations

Approximately 7.5% of the combined total for these four species nested in the region of the southern Everglades marsh/mangrove ecotone, including the southern mainland mangrove estuary. This southern mainland ecotone/estuary was the location of most large wading bird colonies during the 1930s-1950s, prior to the compartmentalization of the system, and altered flow volumes into the mainland estuaries.

Timing of Nesting

The timing of nesting parameter applies only to Wood Storks. The only stork nesting effort in the Everglades in 2007 was in ENP, including the Tamiami West site. The survey and reporting format in 2007 did not allow for a determination of the timing (month of colony formation) for the four Park colonies.

Discussion

During July 2007, most of the biologists who are now conducting wading bird studies and surveys in the greater Everglades basin met to review and refine the parameters of wading bird nesting that will be used to track the success of the ecosystem restoration programs. An objective for this meeting was to develop a more comprehensive set of wading bird indicators and performance measures, which collectively will better describe the relevant and desired responses by wading birds as the restoration programs are implemented.

The result of this discussion is an expanded list of potential indicators of wading bird responses to ecosystem restoration in south Florida. The following is a list of proposed wading bird indicators for future tracking of wading bird nesting patterns. Some are already in use (for example see above). The newly proposed indicators will be developed and vetted in the coming year.

- *Coastal nesting.* The number and percentage (emphasis on number) of wading birds nesting in the southern coastal zone.
- *Timing of nesting.* For storks, the timing of nesting in the southern coastal zone and Big Cypress basin (there was some agreement that timing by storks will not change for birds nesting in Everglades impoundments).
- A ratio of number of nesting pairs of stork/ibis to number of pairs of Great Egrets in the coastal colonies (or for all colonies; higher proportion of egrets in the WCAs illustrates a pattern that is different from the pre-drainage condition).
- Three year running averages for number of nesting pairs of Great Egrets, Snowy Egrets, White Ibis, Roseate Spoonbills, and Wood Storks.
- *Reproductive success.* Some measure of nesting success (different from number of nest initiations) to track future trends (not a comparison with past patterns).
- *"Super Colony" patterns.* A measure of the interval between events, and the magnitude of each event (measured either for regional numbers or single, large colonies).
- *Tricolored Heron Index.* The possibility of developing an index of nesting effort based on some sampling design.

The goal is to have a common set of wading bird indicators and performance measures applicable at system-wide scales, which can support all restoration planning, assessment and reporting needs and requirements. These comprehensive wading bird performance measures will be used to support, (1) RECOVER's program of CERP assessments, including the System Status Reports and Interim Goals Reports, (2) the reports by the Science Coordination Group to the SFER Task Force on overall progress in restoration, and (3) reports to the public in the form of restoration report cards.

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Table 1. Three year running averages of the number of nesting pairs for the five indicator species in the Everglades

Species	1998-00	1999-01	2000-02	2001-03	2002-04	2003-05	2004-06	2005-07	Target
GREG	5,544	5,996	7,276	8,460	9,656	7,829	8,296		4,000
SNEG/TRHE	2,788	4,270	8,614	8,088	8,079	4,085	6,410		10,000-20,000
WHIB	11,270	16,555	23,983	20,758	24,947	20,993	24,926		10,000-25,000
WOST	863	1,538	1,868	1,596	1,191	742	800		1,500-2,500

*Tricolored Herons are excluded from this total due to incomplete surveys for this species in 2007.

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SPECIAL TOPICS

FOOD AVAILABILITY AND WHITE IBIS REPRODUCTIVE SUCCESS: AN EXPERIMENTAL STUDY

The number of wading bird nests in the Everglades has decreased by approximately 70% since the 1930s (Crozier and Gawlik, 2003) and those individuals that do nest often experience reduced reproductive output. A reduction in prey availability brought about by water management activities is considered the most important factor responsible for these declines. This view is supported by studies showing correlations between hydrologic variables and wading bird reproductive effort and success (e.g., Kushlan et al. 1975, Frederick and Collopy 1989). An observational approach, however, does not verify a causal relationship between hydrology, food supply and breeding success, and understanding the specific mechanisms and pathways responsible for the population declines remain limited. Needed are empirical studies that manipulate food supplies and control for naturally correlated variables that also affect nesting success.

Here we present preliminary results from an experimental study that examines whether food supply limits white ibis nestling growth and survival. The primary objectives were to determine experimentally (1) whether food supply limits White Ibis (*Eudocimus albus*) nesting success, and (2) whether food limitation is a function of hydrologic conditions. The study uses a supplementary feeding experiment in which a group of ibis nestlings were fed with locally collected aquatic prey. The effects of food-supplementation on nestling fitness (growth, survival and physiological responses), nestling behavior, and parental provisioning responses were quantified and compared to a control group. Food was supplemented only during the nestling stage but its effects on offspring fitness and behavior were measured from the early nestling stage into the initial natal dispersal period (i.e., from 5d to 60-80d). The study will be repeated in three breeding seasons with contrasting hydrologic conditions to examine the effects of hydrology on food limitation. We present preliminary data analysis from the first two years of the study. The hypotheses are that (1) the success of chicks from food-supplemented nests should be greater than those of control nests and, 2) the magnitude of the difference between treatments should be greatest during breeding seasons when hydrologic conditions are not conducive to optimal foraging.

The Scientific Details

The study was conducted at tree island colonies in A. R. Marshall Loxahatchee National Wildlife Refuge (hereafter Refuge), between April and June in 2006 and 2007. In 2006 the study colony (New Colony 3: 26º 31' N, 80º 16' W) comprised approximately 5000 White Ibis nests. In 2007 the colony site moved approximately one mile east of its 2006 position (New Colony 4: 26º 32' N, 80º 16' W) and contained about 8000 White Ibis nests.

Nestling behavior and parental food provisioning in supplementary fed and control groups were recorded directly from two raised observation blinds using spotting scopes. Data were collected from 36 and 46 randomly selected nests (2006 and 2007 respectively) situated approx. 50 m from the blinds. Every nest was numbered and visible from at least one blind. Nests with chicks of similar age were matched (to control for possible differences in breeding performance of adults related to hatching date) and assigned to either a supplemented or control group (18 control and 18 supplemented nests in 2006; 24 control and 22 supplemented nests in 2007). Chicks in the supplemented group were hand fed every 1.4 days 10 g of fresh, locally caught fish. This provided sufficient energy to have a potential affect on growth/survival but not so much that the parents would lose their provisioning response. Supplementary feeding began when chicks were six days old and continued until nest departure at about 22d. Growth of all chicks was measured every 3-4 days from age 5d (1-day prior to supplementation) until they could no longer be captured (15-25d). On each occasion, body mass, bill length, right tarsus length and right wing length was measured, and the survival status of each chick was recorded. The District collaborated with FAU to measure physiological parameters (e.g., triglycerides, glycerol, and corticosterone) from blood and fecal samples taken at ages 10d and 20d. Feather samples were also taken at these ages to measure mercury loads. As many nestlings as possible were banded with a BBL aluminum band and a combination of unique color bands to identify individuals after nest departure. At 20d each chick was captured and fitted with a radio transmitter and tracked daily by airboat or helicopter until departure from the Refuge and adjacent areas.

Hydrologic variables and prey density were measured once per week at ten random sloughs within a 5 km radius of the colony (i.e. within the foraging range of the parent ibis). Prey density was quantified using standard methods (1 m^2 throw trap).

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To determine chick diet during the study, bolus samples were taken from a group of surrogate nestlings at roughly weekly intervals and analyzed in the lab. Prey species will be identified to family level or higher.

Growth and survival data were analyzed in relation to treatment (supplemented and control), hatching order (first-hatched and second-hatched), for each breeding season (2006 and 2007). We used a repeated measures mixed model (PROC MIXED) to compare mass growth of nestlings and a logistic regression model to examine chick survival from age 5d to 25d post-hatch. Non-parametric Kruskal-Wallis tests were used for all other analyses.

Results

Nestling Growth

The effect of food supplements on nestling growth varied between the two years. The mass of nestlings in the two treatments just after the start of food supplementation (7d post-hatch) was similar in both 2006 and 2007 (2006: $P>0.05$; 2007: $P>0.05$). Mass growth from 7d to 25d post-hatch was similar between treatments in 2006 ($F_{1, 61} = 0.8$, $P = 0.37$), but in 2007 food supplemented chicks grew larger than control chicks ($F_{1, 59} = 21.3$, $P < 0.001$, Fig. 1). In general, A-chicks grew larger than B-chicks but the non-significant interaction between treatment and hatching order suggests that the relative increase in mass due to supplementation in 2007 was similar for both A- and B-chicks ($F_{1, 59} = 1.46$, $P = 0.23$).

Survival

The role of food supplements on nestling survival also varied between years. In 2006, overall nestling survival was high throughout the colony and supplements had no effect on survival from 5d to 25d post-hatch (supplemented: 84% survived; control: 88% survived; $X_1^2 = 0.21$, $P = 0.64$; Fig. 2). Moreover, there was no difference in survival between first and second hatched chicks ($X_1^2 = 1.76$, $P = 0.18$). In 2007, however, overall survival within the colony was relatively low and food supplementation resulted in a significant increase in nestling survival (supplemented: 82% survived; control: 48% survived; $X_1^2 = 10.59$, $P = 0.001$; Fig. 2). This increase in survival was a function of hatching order, with supplements affecting the survival of second-hatched chicks (supplemented: 64% survival; controls 20% survival) more than that of first-hatched chicks (supplemented: 100%; controls: 82%). In both years, mortality tended to occur at a young age (2006: mean age 6.1d± 1.09 SE days, min: 1d, max: 16d; 2007: mean age 7.2d± 0.66 SE days, min: 1d; max: 16d) but age of mortality was not effected by treatment or hatching order ($P>0.05$). Probability of survival increased once birds reached 20-25d (the crèche period) and the proportion of birds that survived from 25d to dispersal from the colony was extremely high for both treatments in both years (2006: 92% of 39 radio-tracked birds fledged; 2007: 88% of 32 radio-tracked birds fledged). Mean age of dispersal (2006: 59.4 ± 1.3 SE days old, range: 52-66 days; 2007: 60.3 ± 1.3 SE days old, range: 49-74 days) was not affected by treatment or hatching order in either year (all $P > 0.05$). In 2007, the pattern of fledgling dispersal was similar to that in 2006: after leaving the colony fledglings immediately flew out of the Refuge and in most cases were not relocated thereafter.

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Discussion

The primary aim of this study was to test whether prey availability limits White Ibis reproduction during the nesting period. We predicted that if prey availability was a limiting factor then food supplementation would improve nestling fitness. This prediction was upheld in 2007 with a marked increase in nestling growth and survival. By contrast, the prediction was not supported during the particularly successful nesting season of 2006 when nestling growth and survival rates of both treatment groups were high. Taken together these results show that prey availability can limit White Ibis reproduction but that the Everglades still retains the capacity to provide sufficient prey when ecological conditions permit. Moreover, the marked increase in fitness in response to supplements in 2007 and the overall successful nesting of 2006 suggest that nestlings were not unduly affected by other factors thought to limit wading bird breeding such as mercury poisoning or parasitism.

A second objective was to test our understanding of the relationship between nestling fitness and hydrologic conditions. We predicted that the effects of supplementation on nestling fitness would be most marked when conditions were wetter or dryer than are considered optimal for wading bird foraging. To gain a rudimentary understanding of the role of hydrology on nestling production it is necessary to examine at least three contrasting hydrologic years (wet, dry and optimal) and control for potentially confounding factors such as the productivity of prey. As predicted, our results to date show that nestling fitness increased with supplementation during a breeding season with sub-optimal, dry hydrologic conditions (2007) but not when conditions were considered optimal (2006); see Fig. 3 and discussions about hydrology elsewhere in this report. A precise assessment of the importance of food limitation on White Ibis reproduction requires further years of study and our aim is to continue this study during at least one more breeding season when conditions are wetter than average.

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Figure 1. Mean nestling mass (±1 SE) at age categories 7, 10, 13, 16 and 19d post-hatch for A- and B-chicks in food-supplemented and control nests in WY2006 and 2007. Sample sizes are above and below error bars.

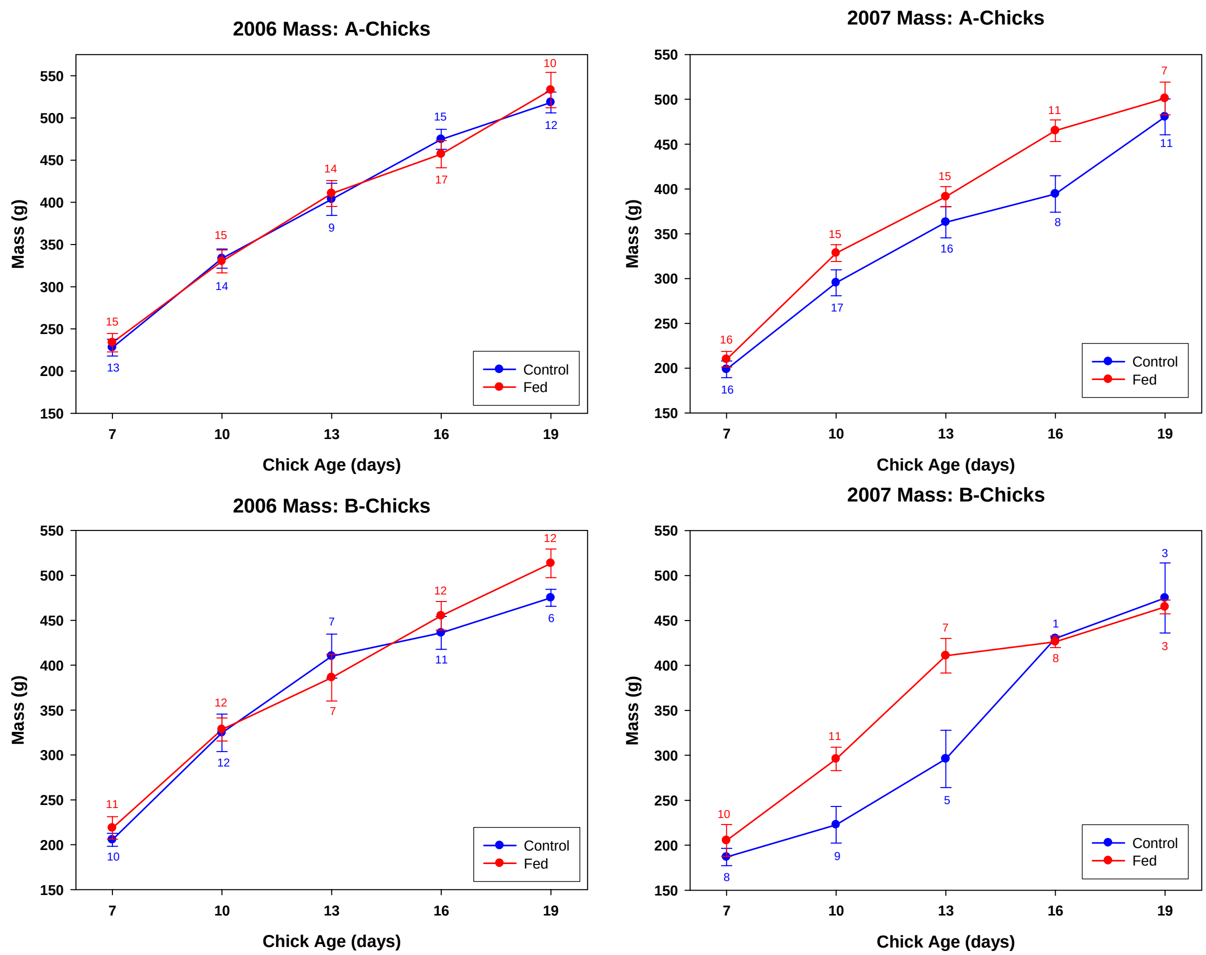

Figure 2. Percentage of nestlings that survived to 25 days old. Sample sizes are shown above data bars.

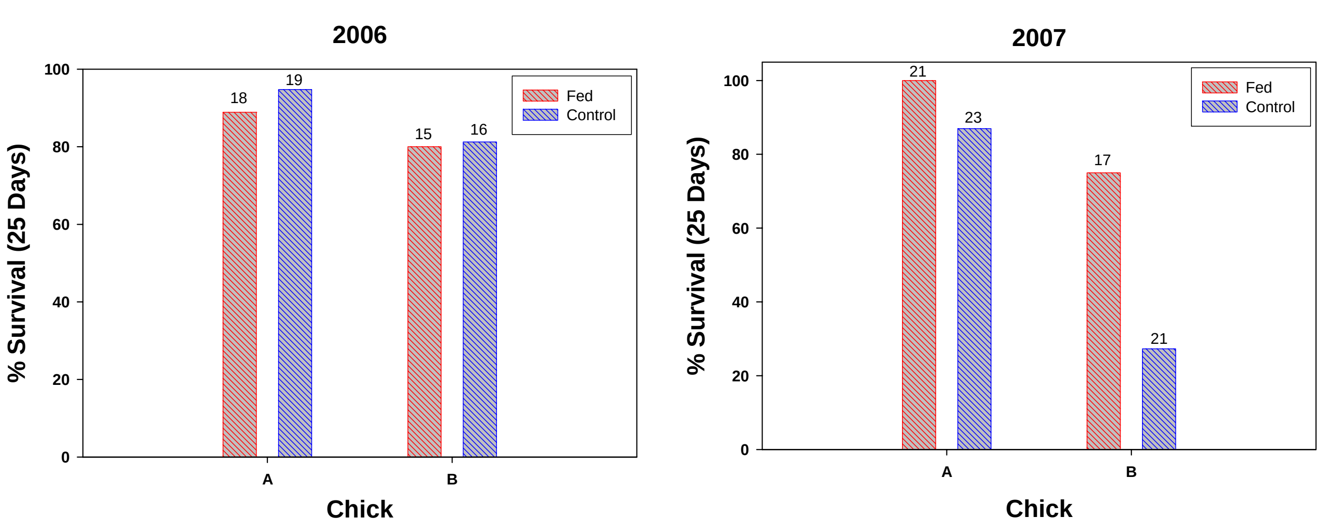

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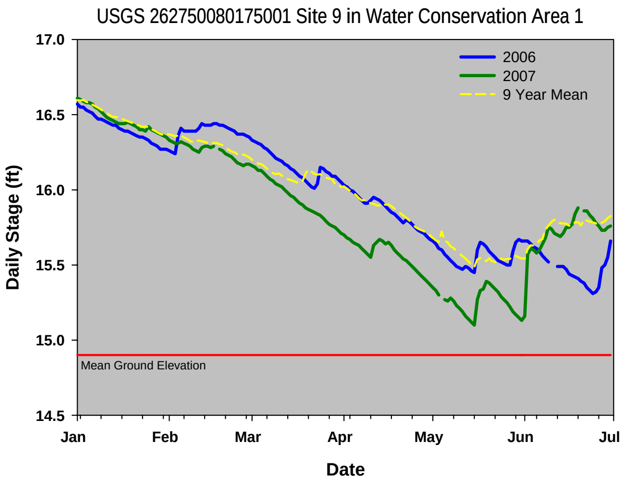
Figure 3. Hydrograph depicting mean water levels in the Refuge during the optimal water year of 2006, the drought year 2007, and the nine year mean.



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SACRED IBIS

Background

Sacred Ibis (*Threskiornis aethiopicus*) were first discovered breeding in the Florida Everglades in 2005 in the Arthur R. Marshall Loxahatchee National Wildlife Refuge, Palm Beach County (Herring et al. 2006). Prior to this, Sacred Ibis have been observed periodically throughout South Florida since the mid 1990s, with occasional breeding confirmed at the Miami Metro Zoo in Miami-Dade County and the Palm Beach Waste Management Facility in Palm Beach County (Herring et al. 2006).

Sacred Ibis are colonial wading birds, native to wetlands throughout Africa (Hancock et al. 1992). However, they have escaped captivity in 12 European countries and the United States and currently breed in the wild in Belgium, France, Italy, the Canary Islands of Spain (Clergeau et al. 2005), the Netherlands (Ottens 2006) and the United States (Herring et al. 2006). Clergeau and Yésou (2006) reviewed the recent population growth and expansion of the escaped Sacred Ibises' range in Western Europe, noting that the species' foraging plasticity, human commensalism, and tolerance of wide ranging environmental conditions increase their chance of successful population establishment and growth.

The potential for successful establishment and population growth of nonindigenous Sacred Ibis is best illustrated by its recent expansion in France. The French nonindigenous Sacred Ibis population stemmed from less than 75 individuals that escaped in the mid 1980s, and now exhibit an exponential population growth rate with over five thousand individuals (Clergeau and Yésou 2006). Sacred Ibis in France have also dispersed hundreds of kilometers from their original site (Clergeau and Yésou 2006) suggesting similar rates of dispersal could occur in Florida if those birds become established.

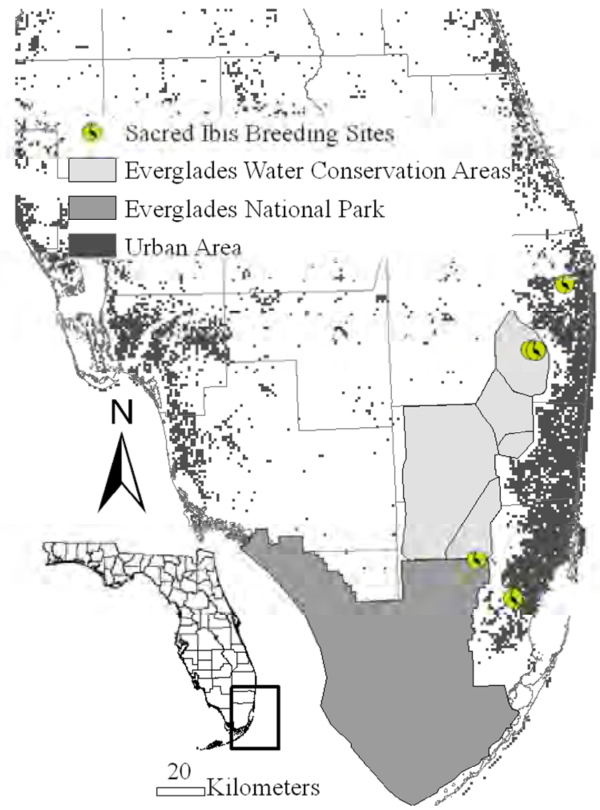
Recent research has shown that Sacred Ibis are effective predators of both eggs and chicks in colonial nesting birds in their native region (Ward and Williams 2006). Sacred Ibis have also been documented destroying Sandwich Tern (*Sterna sandvicensis*), Black Tern (*Chlidonias niger*), and Whiskered Tern (*C. hybridus*) nests in large numbers in areas where they have recently become established (Vaslin 2005; Clergeau and Yésou 2006).

The extent to which nonindigenous populations of Sacred Ibis will predate eggs and chicks of native colonial nesting species has not been determined. However Sacred Ibis in their native range have been known to impact other waterbirds (Williams and Ward 2006). Sacred Ibis were responsible for the predation of 65% of all Cape Cormorant (*Phalacrocorax capensis*) chick predation mortalities on Penguin Island, Lambert Bay, South Africa (Williams and Ward 2006). Williams and Ward (2006) calculated that the total Cape Cormorant losses at the Penguin Island colony due to Sacred Ibis predation were between 10% - 15% of the total annual production. Predation by Sacred Ibis occurred throughout much of the nesting cycle, with ibis targeting eggs and chicks up to five or more weeks old (Williams and Ward 2006).

Nesting

Between 2005 and 2007, Sacred Ibis nesting was documented at 3 wading bird colonies in the Everglades, and at one wading bird colony within the city of Palm Beach (West Palm Solid Waste Management Facility; Fig. 1). Mean number of Sacred Ibis nests per colony was 2.4 ± 0.5 SE, with a mean number of chicks or eggs per nest of 2.3 ± 0.2 SE. Twenty-three adults were observed in wading bird breeding colonies in the Everglades during this same period.

Figure 1. Location of sacred ibis nests between 2005 and 2007 in South Florida.



Potential for establishment

Recent research has shown that Sacred Ibis have a high probability of successful establishment (>70%) in the Florida Everglades (Herring and Gawlik in review). A qualitative assessment of the source population in urban South Florida suggests that it is small enough that eradication is still feasible (Herring and Gawlik 2007). Removing Sacred Ibis from the Everglades without addressing the urban source population will likely only postpone a repeated population expansion by the bird and possible negative interactions with native Everglades wading birds.

Contributors to the South Florida Wading Bird Report are encouraged to be on the lookout for Sacred Ibis, and report their observations in a timely manner to Larry Connor, FWC exotic species database manager, at Larry.Connor@MyFWC.com. Those wishing to discuss control strategies for Sacred Ibis in South Florida, are welcome to contact ENP biologist Skip Snow at skip_snow@nps.gov.

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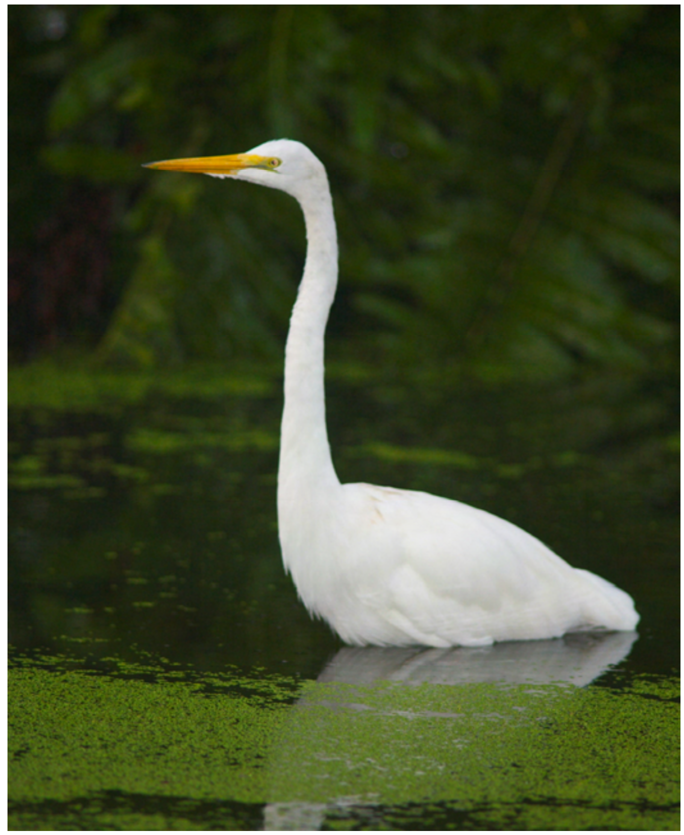

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The Stoplight Restoration Report Card System Applied to Wood Storks, White Ibis and Great Egrets

Wading Bird Data

This section addresses where data for forming the wading bird thresholds and suitability curves was obtained. In addition, this section discusses some of the concerns with data compatibility, and therefore its comparability, between the various sources. Chronicling the data sources used to construct the suitability curves is also important because it will assist those working with the indicators in the future with a "cookbook" for consistent data gathering and analysis. In all cases, the "South Florida Wading Bird Reports" refer to Cook and Call 2005, 2006, 2007, Crozier and Gawlik 2003b, 2004, Gawlik 1997 – 2002b, and Gawlik and Ogden 1996a and b.

<u>Ratio of the Combination of Wood Stork + White Ibis Nests to Great Egret Nests</u>

Data Sources: For the years 1931 through 1989, data were obtained from Ogden 1994, Table 22.2. Numbers of nests in this table are averaged over periods of seven to fourteen years; no numbers are given for individual years. Data from 1990 to 1997, except for 1996, comes from Ogden et. al. 1996 and is broken down by year. Data from 1996 and 1998 to 2006 comes from the annual South Florida Wading Bird reports "Status of Wading Bird Recovery" section and is broken down by year.

Data Concerns: The assumption inherent in this indicator is that the numbers of nesting birds of each species are largely controlled by foraging opportunities. There is some uncertainty about this assumption, since we also know that Great Egret populations were particularly targeted by plume hunters in the early part of the 20th century. It is not clear whether Great Egrets had rebounded to carrying capacity of the marsh by the time of the 1930's, a benchmark era for this indicator. Between the 1930s and the 1960s, Great Egret populations did not appear to change much, and if anything appeared to decrease. This suggests that carrying capacity had been reached by the 1930s. However, the survey data during the 1930s and 1960s is fragmentary and probably not good enough to be conclusive on this point. Therefore there is some uncertainty in using the 1930s ratios.

<u>Month of Initiation of Wood Stork Nesting</u>

Data Sources: We used earliest month reported as our metric. For years 1931-1946 and 1974-1989, data were obtained from Ogden 1994, Table 22.6. Data are reported as averages; they were not broken down by individual year. It was also assumed that Ogden 1994's "Timing of colony formation" in Figure 22.3 was equivalent to month of Wood Stork nest initiation. In an effort to break down nest initiation into finer details, data from Ogden 1994, Figure 22.3, were used for years 1953 to 1989. Years 1962 and 1978 are missing from this data set. For years 1996 through 2006, the earliest date of Wood Stork nest initiation mentioned in the "Status of Wading Bird Recovery" section of the South Florida Wading Bird reports was used. No Wood Stork nest initiation data was recorded for 2003 and 2004 in these reports. No particular concerns were noted about use of this indicator.

<u>Percentage of All Wading Bird Nests That Occur in the Everglades Coastal/Headwaters Ecotone</u>

Data Sources: Data for 1931 through 1946 was obtained from Ogden 1994, Table 22.4. Data for 1974 through 1989 was obtained from Ogden 1994, Table 22.5. According to Ogden, regions 2, 3 and 4 in this table are equivalent to the Coastal/Headwaters Ecotone, while region 5 is considered the Everglades' interior. Percentages for 1996 through 2006 were taken directly from the "Status of Wading Bird Recovery" section of the South Florida Wading Bird reports. In this summary section, only percentages of nests that occurred in the ecotone are given, not individual numbers of nests in the different areas/regions.

Data Concerns: Data from earlier years is difficult to interpret because sometimes only presence/absence is reported. We assumed that the percentage of nests occurring in the ecotone reported in the South Florida Wading Bird reports was in the same manner as the information reported in Ogden 1994, Tables 22.4 and 22.5. It might be possible to increase the data set for analysis from Crozier and Gawlik 2003a if the regions they use could be correlated to the regions used in Ogden 2004.

Interpretation of this indicator must be done carefully since the proportion nesting in the ecotone is strongly affected by numbers nesting in the freshwater areas. Thus a high percentage in the coastal zone could be achieved by having relatively few birds nest in the coastal zone and virtually none in the freshwater. Conversely even very large numbers in the coastal zone might be offset in a proportional sense by equal or greater numbers in the freshwater. With this indicator, the intent of the indicator is clearly that the coastal areas should be proportionally at least as attractive to wading birds as the freshwater areas are.

Interval between exceptional ibis nesting events.

Data Sources: All data, which includes mean number of nesting birds and water conditions covering 1931 through 1946 and 1974 through 1998, comes from Fredrick and Ogden 2001, Table 1. Data from 1998 through 2007 come from South Florida Wading Bird Reports.

Temporal Variance of Data Quantity and Quality

One overarching concern with all the data included in this section is the differences in how and where historical and more recent data have been collected. The data from the 1930's and 1940's is based largely on a review of field notes of Audubon wardens (Ogden 1994). Both the collection and interpretation of these data appear to be somewhat subjective, especially in comparison with the more systematic methods of more modern wading bird estimates. In addition, the areas surveyed are not always clear, and the effort and even methods used are not always or even usually stated in the historical accounts. Therefore, while data from the historic period may be useful in determining some quantitative bounds of the "natural" conditions, its usefulness in trend analysis and setting recovery targets may be questionable depending upon the precision desired in the comparison. While these problems do not preclude use of the historical information, it is clear that interpretation and comparison must therefore be undertaken on a case by case basis.

Suitability Curves

Ratio of the Combination of Wood Stork and White Ibis Nests to Great Egret Nests

Figure 1 shows the ratio of the sum of Wood Stork and White Ibis nests to Great Egret nests from the historic period (1930's and 1940's) until the modern era. During the historic era, the highest ratio of the combination of Wood Stork and White Ibis nests to Great Egret nests was 36 to 1. This trend has fallen off rapidly and in the present era averages approximately 2 for most years. Over the past two decades, there have been as many as five times the number of White Ibis and stork nests to Great Egret nests (Figure 2). Numbers of White Ibis and stork nests were increasing faster than Great Egret nests during 2000 – 2007. The indicator is moving in the desired direction and is projected to be in the green zone in the near future.

Month of Initiation of Wood Stork Nesting

Records from 1931 – 1946 indicate that Wood Storks usually initiated nesting at the latest in December (Ogden 1994). Figure 3 illustrates month of Wood Stork nest initiation from 1953 until present, although ten years are missing from the data sequence (1962, 1978, 1990-1995, 2003-2004).Although the polynomic trend line through the data is not strongly significant, the date of nest initiation may have become slightly earlier over the past decade.

Percentage of All Wading Bird Nests That Occur in the Everglades Coastal/Headwaters Ecotone

The percentage of all wading birds nesting in the ecotone is illustrated in Figure 4, averaged over running 5-year periods where data were available. Percentage estimates are not available for several years during this time period, including 1939, 1941, 1943, 1945, 1947 – 1973.

Interval between years with of exceptionally large White Ibis nesting:

Years with exceptionally large nesting colonies or groups of colonies (= 70th percentile in the period of record) occurred about every 2 years in the 1930s and 1940s in the Everglades (Table 1, Figures 4 and 5).

Thresholds for Wading Bird Stoplight Restoration Report Card

Ratio of White Ibis +Wood Stork nests to Great Egret nests: Green condition - Ratios greater than 30. Yellow conditions – ratio of between 10 and 20 AND improving trend; red condition – ratio of less than ten. (See scoring questions below for finer detail).

Month of Wood Stork nest initiation: Red – running five year Julian date means (of earliest recorded nesting date), corresponding to later than January 30. Yellow – running five year means corresponding to dates between December 31 and January 30 AND improving condition. Green – running five year means corresponding to dates earlier than December 30.

Proportion of nesting population in ecotone: Red condition – Less than 50% of nests in the ecotone. Yellow – between 50 and 70 percent nesting in the ecotone AND increasing trend. Green – greater than 70 percent of nesting in the ecotone.

Interval between exceptional ibis nesting events: Green condition - running 4-year mean should be no greater than 2.8, which is the 1930's mean plus one standard deviation. Yellow conditions prevail if the mean is 2.9 – 5 years. greater than this, but the trend is towards decreasing intervals. Red conditions are mean intervals greater than 5 years.

Table 1. Mean of intervals (in years) between exceptional ibis nesting events in the Everglades expressed for different periods of the record. Exceptional years are defined as 70th percentile of the period of record, or 16,977 nests.

	Mean	s.d.
Period of Record	3.5	7.42
1931 – 1942	1.7	1.21
1986 – 1999	28.0	0
1973 – 2007	5.0	10.15
2000 – 2007	1.2	0.41

Figure 1. Ratio of Wood Stork (WOST) plus White Ibis (WHIB) nests to Great Egret (GREG) nests from 1930s until present.

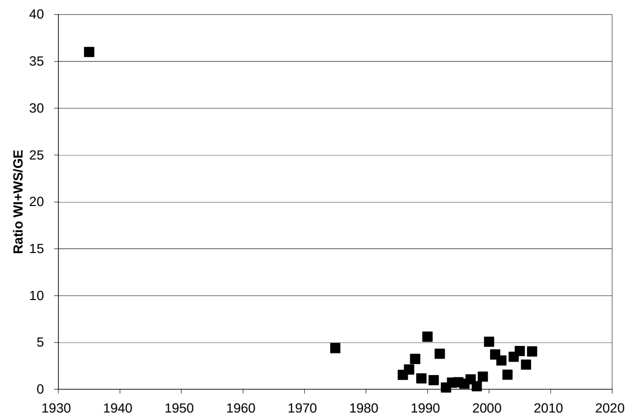

Figure 2. Ratio of stork + ibis nests to Great Egret nests since 1975. Error bars represent interannual variability. The current period seems to show an upturn, but is not close to the 36:1 ratios seen in the 1930's.

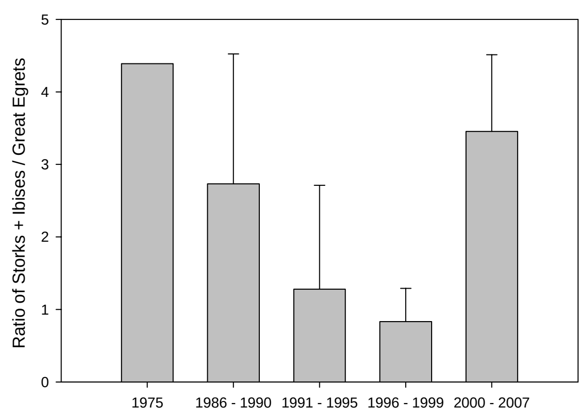

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Figure 3. Month of nest initiation by Wood Storks in the Everglades from 1953 until present. Data for the following years are not available: 1962, 1978, 1990-1995, and 2003-2004.

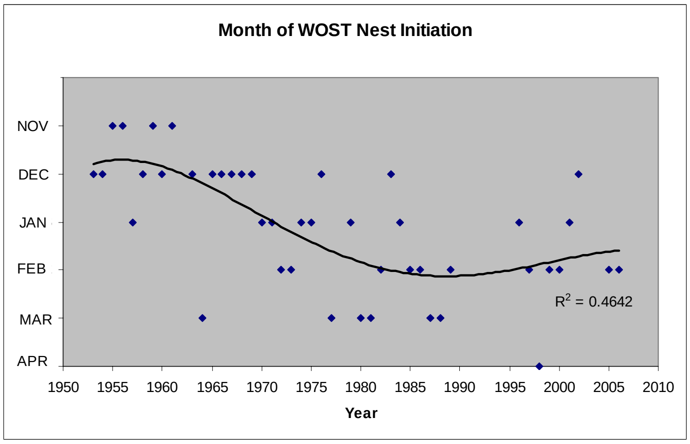


Figure 4. Proportion of all wading bird nests in the Everglades ecosystem occurring in the freshwater/saltwater ecotone from historic era into the present expressed as 5-year running means. Error bars represent interannual variation.

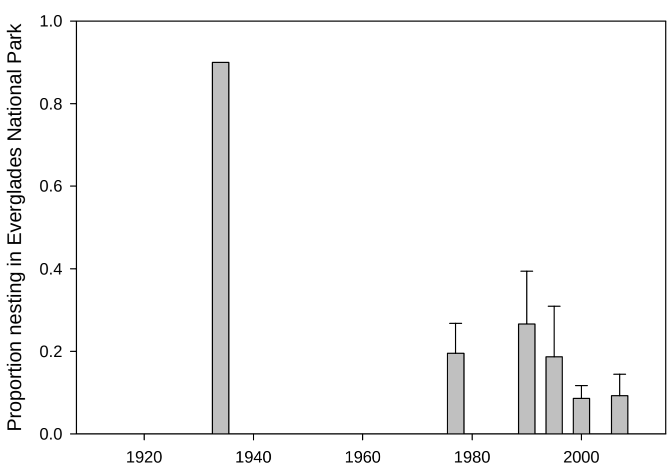
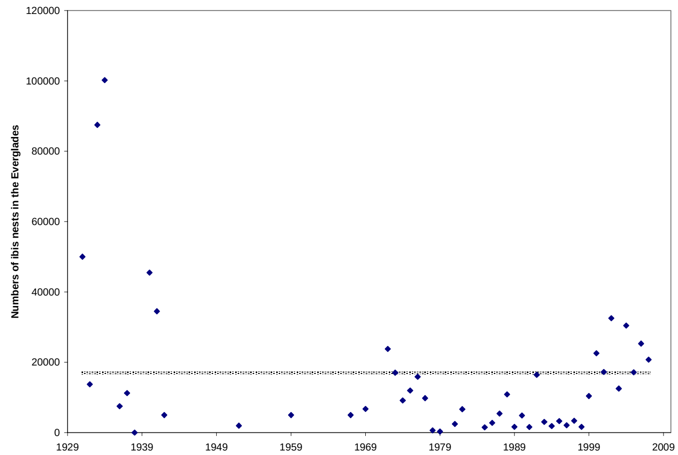


Figure 5. Numbers of White Ibis nests in the Everglades, 1930 – 2007, with horizontal line representing 70th percentile of nesting events in the entire period. This threshold (16,977 nests annually) is used as to identify exceptional nesting years. Note that the data from the period 1942 – 1972 are spotty and probably unreliable.



The Stoplight Restoration Report Card System Applied to Spoonbills

This communication tool is based on MAP performance measures (either by module or system-wide) and is expected to be able to distinguish between responses to restoration and natural patterns. A set of parameters (Table 1) has been developed for each performance measure. Answers are translated as suitability indices identified as stoplight colors with green indicating that targets have been met, yellow indicating that conditions are below the target but within a suitable range of it and red indicating the measure is performing poorly in relation to the target. Two questions are addressed using suitability indices: 1) have we reached the restoration target, or if not, 2) are we making progress toward targets?

Methods for producing suitability curves vary among performance measures. For example, a ten-year running average was used for percentage of years that spoonbills were successful. A five-year running average was used for average annual nest production and nest numbers. Fish community structure changes to a greater percentage of freshwater species only when salinity conditions have been favorable to these species for a two to three year period, therefore this parameter will be reported as an annual metric that covers a three year period. Nesting success will be reported annually because short-term water depth conditions dominate this parameter. By using this suite of performance measures this indicator covers time scales from annual to three, five and ten year cycles.

Calculation of Metrics and Thresholds for the Spoonbill Stoplight Restoration Report Card

Spoonbill nesting success. Lorenz et al., (2002) divided Florida Bay into five regions based on the primary foraging grounds for each of the colonies within each region (Figure 2). They also demonstrated that, under the SDCS operations, the nest productivity and nest number in the northeastern region have experienced a significant decline. The method used to calculate this metric is based on surveys of focal colonies (defined as the two largest colonies within the region) . These surveys entailed marking up to 50 nests shortly after full clutches had been laid and re-visiting the nests on an approximate 7-10d cycle to monitor chick development. The metric is the number of chicks per nest to survive to twenty-one days. After twenty-one days, the chicks become very active and move throughout the colony precluding accurate accounting of individual nest production. Since 2003, chicks have also been leg-banded so that individual chicks can be identified. By resighting these individuals later in the nesting cycle, we are able to use a second method to estimate nest production. Preliminary analysis of this mark-resighting technique generally confirms that the twenty-one day survival is an accurate method to calculate nest production..

This stoplight uses two metrics for nest production. The number of successful nesting years out of ten with success being defined as an average nest production of greater than one chick per nest (c/n) for all nest starts. This metric uses only the northeastern region of the Bay (Figure 2) as this has been demonstrated to be the region most impacted by water management practices (Lorenz et al., 2002). Prior to the establishment of the SDCS, spoonbills nesting in the northeastern region averaged 71% successful years (Lorenz et al., 2002). Stoplight colors were based on this threshold (Table 1, Figure 4).

The second metric of nest production is the five year mean of nest production in the northeastern region. Lorenz et al., (2002) demonstrated that prior to the SDCS annual mean spoonbill production in the northeast region was 1.38c/n and that this dropped to 0.67 post-SDCS. Initially we set this as the target for the stoplight metric where annual production was divided by 1.5 c/n with greater than 67% set as the threshold for a green rating. However, as can be seen in Figure 5, there are no trends in the data with rapid changes occurring from one year to the next. This is due to the interannual differences in hydrologic conditions that affect the ability of spoonbills to capture enough prey to successfully raise young. Simply put, some years are naturally better than others. Taking a multi-year running average smoothes this high variability into more interpretable trends (Figure 5). By examining various time frames from previous data we concluded that by using a five year running average, no single good or bad year out of the five skewed the results into the red or green classification. A single good or bad year in either the two, three or four year running averages could bias the mean, thus resulting in an inaccurate stoplight color.

There are natural background conditions that can result in nest failure that are unrelated to CERP or water management practices. Therefore, we need to control for natural background variation in foraging conditions. We dealt with this problem by using the northwestern region's success rate as control for natural background conditions. While the northeastern region's production declined post SDCS, the northwestern regions production remained relatively high (1.24c/n) even though there was still a great deal of interannual variability. Lorenz and Frezza (2007) concluded that the interannual variation in productivity of the northwestern colonies reflects the natural variation while the variation in the northeast is affected by both this background and by water management practices. Therefore, we propose that the metric used to gage success in the northeastern region be tied to that of the northwestern, i.e., the metric should be calculated by dividing annual northeastern production by that of the northwest thereby resulting in a percentage (Figure 6). The thresholds for stoplight colors are presented in Table 1.

Although this metric solves the problem of natural interannual variation in nesting success, it is also dependant on the continued high rates of success of the northwestern colony. What happens if CERP or other issues begin to negatively affect the success of the northwestern colonies? This would result in the metric receiving higher scores even though there was actually a degradation of the bay for spoonbills. Therefore, stoplight metrics were developed to examine the northwestern regions (explained below in section 2.3.5). If all three of the metrics are yellow or red then the metric for northeastern success should be based on the long term mean production rate of 1.5 c/n for northeastern Florida Bay (Lorenz et al., 2002, Figure 5).

Number of spoonbill nests in Florida Bay. Spoonbill nest counts for Florida Bay have been performed intermittently since 1935 (Powell et al., 1989). Over that period, spoonbills have been recorded nesting on thirty-eight keys throughout the Bay (Figure 2; Lorenz et al., 2002). Spoonbills typically establish nests in Florida Bay in November or December of each year, however, nest initiation has started as early as October and as late as March (Powell et al., 1989, Alvear-Rodriguez, 2001). All known nesting keys are visited every twenty-one days during the nesting season. Our data show that prior to the establishment of the SDCS, the peak number of nests was 1258 in 1978 (Figure 1, Lorenz et al., 2002). For this stoplight, annual nest counts are divided by 1258 to get the annual percentage of the historic peak number of nests (Figure 7) and assigned the stoplight color as per Table 1.

Spoonbill nesting location. This stoplight indicator consists of two metrics: a return to pre-SDCS nest numbers in the northeastern region and return of spoonbills to nesting colonies along the southwest coast of the Everglades in the Shark River Slough and Lostman's Slough estuaries. Powell et al., (1989) reported that in the peak year of 1978 more than half of the 1258 nests were located in the northeast region (688 nests). Following the completion of the SDCS, this number dropped to approximately 100 nests from 2000 to 2007. In 2008 there were a total of 47 nests in the region. For restoration to be considered successful, we should expect a return to nesting numbers to pre-SDCS numbers. This metric is the percentage of 650 nests that occur annually (Figure 8). Similar to nest success and total nests for Florida Bay, the interannual variation can bias individual years and a five year mean was used for this metric (Table 1).

According to Scott (1889), spoonbills "nested in the thousands" along the southwest coast of the Everglades in the Shark River and Lostman's slough estuaries. Restoration of more historic hydrological conditions should promote greater prey abundance and availability in this region, potentially leading to a return of spoonbill nesting in large numbers. In recent years, Everglades National Park has performed aerial wading bird surveys of this area and has documented spoonbill nesting (Pers. Comm, Sonny Bass, Supervisory Wildlife Biologist, Everglades National Park), however accurate surveys of spoonbills nest number can not be performed from aircraft because they tend to nest low in the canopy. Although it is imperative to get a baseline for pre-CERP nesting in this critical region, no funds have been identified to pay for this effort. As a result, no stoplight metrics can be established at the time of this publication.

Prey Community Structure. Spoonbills primarily feed on small demersal fishes found throughout the Everglades system (Allen, 1942, Dumas, 2000). Lorenz et al., (1997) developed a methodology that uniquely sampled fishes in the dwarf mangrove foraging grounds that are the preferred feeding locations for spoonbills nesting in Florida Bay. The sampling design uses a $9m^2$ drop trap at fixed locations at known spoonbill feeding sites. Data collection began in 1990 at four sites. Currently, there are 14 sampling sites associated with Florida Bay's nesting spoonbill population (Figure 2)

Lorenz (1999) documented that these fish respond markedly to changes in water level and salinity and these factors can be altered by water management practices. Lorenz and Serafy (2006) performed a fish community analysis of eight years of these data from six sites. During the eight-year span reported by this study, there were three consecutive years of unusually high rainfall and freshwater flows to the estuary which resulted in low salinity similar those believed to have occurred in the region prior to water management influences. As part of their analysis, Lorenz and Serafy (2006), placed individual species in one of four salinity categories (freshwater, oligohaline, mesohaline or polyhaline) based on the Venice System of Estuarine Classification (Bulger et al., 1993). To accomplish this, the authors used the mean salinity for the thirty days prior to a given collection (based on the findings of Lorenz, 1999) to identify the range of salinities in which each species was found. The median score of each species salinity range was then used to classify the species into one of the four categories. During the period of low salinity and high fish abundance, Lorenz and Serafy (2006) found that more than 40% of the total fish community were freshwater affiliates (Figure 3). Furthermore, they demonstrated that it took two to three years of low salinity for the freshwater populations to respond. Finally, they demonstrated these low salinity communities were much more productive based on both number and biomass of the standing stock (Figure 3). The stoplight for prey abundance will use the percentage of the fish community that was classified by Lorenz and Serafy (2006) as freshwater species as per Table 1. Although the stoplight will be reported on an annual basis, it is integrative for the previous two years as well, i.e., this stoplight measures conditions on a three year time scale.

Table 1. Decision rule targets and scores for forming performance measure/suitability relationships for the Roseate Spoonbill indicator communication tool.

1. Northeastern Nesting Success: number of successful nesting attempts (average of >1 chick fledged per nest attempt) out of the previous 10 years in northeastern Florida Bay. Target is 7 out of 10 successful years based on the pre-SDCS average (Lorenz et al., 2002)
   a. 0 – 3 Red
   b. 3 - 6 Yellow
   c. 7 - 10 Green

2 Northeastern Nest Production:
   A. Five year mean of northeastern Florida Bay nest production expressed as a percentage of northwestern Florida Bay nest production. This metric will be used if any of the control metrics for northwestern Florida Bay (number 7 below) are green. In the case of none of the controls being scored green than 2B will be used.
   a. 0 - 33 Red
   b. 33 - 66 Yellow
   c. > 66 Green

   B. Five year mean of the percentage of mean pre-SDCS nest production. Target is 1.5 chicks per nest attempt is based on the mean nest production from 1962 to 1982 (Lorenz et al., 2002). This metric will only be used when all of the northwestern Florida Bay control metrics (number 7 below) are scored as yellow and/or red. In the case of any of the controls being scored a green than 2A will be used.
   a. 0 - 50 Red
   b 50 - 100 Yellow
   c. > 100 Green

3. Nest Number: five year mean of the percentage of pre-SDCS peak nest numbers found throughout Florida Bay. Target is 1250 based on the peak number of nests found in 1978 (Powell et al., 1989).
   a. 0 - 50 Red
   b. 50 - 100 Yellow
   c. > 100 Green

4. Florida Bay Spoonbill Nesting Location: five year mean of the percentage of pre-SDCS peak nest numbers found in northeastern Florida Bay. Target number is 625 based on the peak number of nests found in 1978 (Powell et al., 1989).
   a. 0 - 33 Red
   b. 33 - 66 Yellow
   c. > 66 Green

5. Nesting in Southwestern Everglades Estuaries: No targets or stoplight scores can be set at this time

6. Prey Community Structure: Annual percentage of prey base fish sampling that are classified as freshwater species according to Lorenz and Serafy (2007). Target is that 40% of the total annual catch collected at six sampling sites within the foraging grounds of spoonbills nesting in northeastern Florida Bay (Figure 2: TR, EC, WJ, JB, SB, and HC) are freshwater species using data. Note that this metric is integrative of three years.
   a. 0 - 20 Red
   b. 20 - 40 Yellow
   c. > 40 Green

7. Northwestern Florida Bay Control Metrics:
   A: Five year mean of the percentage of mean post-SDCS nest production in northwestern Florida Bay. Target is 1.24 chicks per nest attempt is based on the mean nest production from 1982-2002 (Lorenz et al., 2002).
   a. 0 - 50 Red
   b 50 - 100 Yellow
   c. > 100 Green

   B. Five year mean of the percentage of post-SDCS mean nest numbers found in northwestern Florida Bay. Target number is 200 based on the number of nests from 1982-2002 (Lorenz et al., 2002).
   a. 0 - 50 Red
   b 50 - 100 Yellow
   c. > 100 Green

C. Number of successful nesting attempts (average of >1 chick fledged per nest attempt) out of the previous 10 years in northwestern Florida Bay. Target is 6 out of 10 successful years based on the post-SDCS average (Lorenz et al., 2002)
   a. 0 – 2 Red
   b. 3 - 5 Yellow
   c. 6 - 10 Green

8. Cumulative Spoonbill Stoplight Metric: the mean of the 6 (or 7 if nesting location on the southwest coast of Florida can be calculated from future efforts) non-baseline stoplights where red is scored 1, yellow is scored 0.5 and red is zero.
   a. 0 - 33 Red
   b. 33 - 66 Yellow
   c. > 66 Green

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Figure 1.

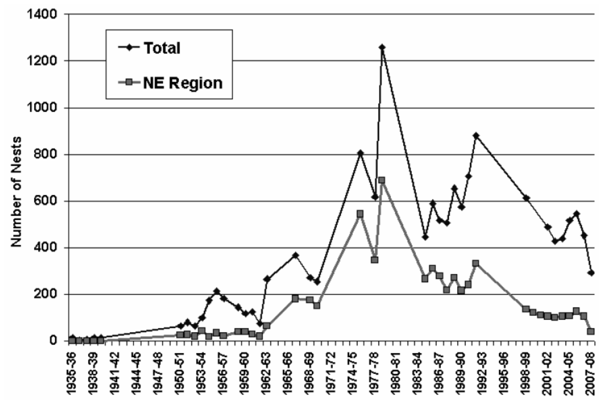

Figure 1. Annual number of roseate spoonbill nests for all of Florida Bay (Total) and for just the northeastern region of the bay from 1935 to 2008.

Figure 2.

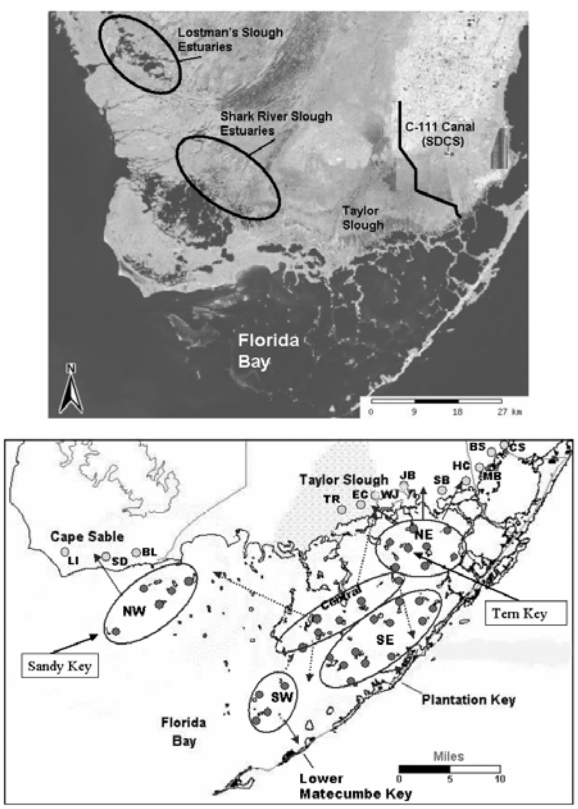

Figure 2. Top : Map of southern Florida indicating the major features discussed. Bottom: Map of Florida Bay indicating all the nesting locations for spoonbills since 1935, the primary foraging areas for five regions of Florida Bay and the fish sampling sites used to evaluate the spoonbill's forage base.

Figure 3.

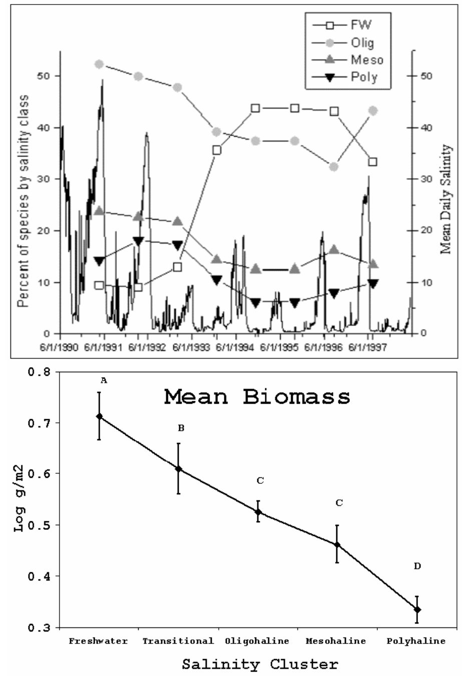

Figure 3. Top: Left Axis : Percent of total species collected annually at the three estuarine fish sampling sites (Figure : TR, JB, HC) by each salinity category as defined by Lorenz and Serafy 2006. Right Axis: Mean daily salinity from the three sites for the period of record. Note that years following a high salinity dry season have lower representation of freshwater species and higher representation of mesohaline and polyhaline species. The figure also indicates that it takes 2 to 3 consecutive years of low salinity for the freshwater species to become the dominate fish category. (Copyright: Hydrobiologia). Bottom: Differences in fish biomass between salinity categories as defined by Lorenz and Serafy (2006) using Non-Metric Multidimensional Scaling from eight years of fish collections at 6 sites. Their results show that samples dominated by lower salinity species have significantly higher biomass than those dominated by higher salinity species. (Copyright: Hydrobiologia).

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Figure 4, 5, 6.

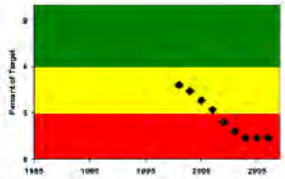

Figure 4. Decadal metric for percent of years nesting was successful. The percentage years out of the previous ten in which spoonbills nesting in northeastern Florida Bay were successful (>1 chick per nest fledged). These data demonstrate the declining number of successful years in spoonbill nesting since 1998. Note that due to data limitations we used the five year average in the figure, however, the ten year mean will be used for the actual stoplight metric.

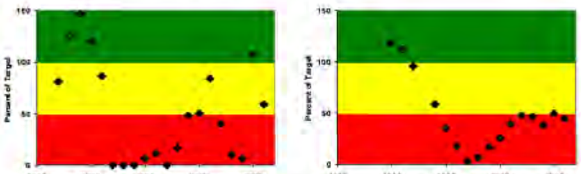

Figure 5. Five year metric used for nest production in northeastern Florida Bay. Left: Percentage of the target production rate of 1.5 chicks per nest fledged in northeastern Florida Bay since the completion of the South Dade Conveyance System (SDCS). The target is based on pre-SDCS nest production data presented by Lorenz et al (2002). Right: The five year running mean of data presented in the figure on the left. Note that due to data limitations the first 3 data points are four year averages, however, the five year mean will be used for the actual stoplight metric. This metric will only be used if the three control metrics for northwestern Florida Bay (Figure) are scored yellow and/or red.

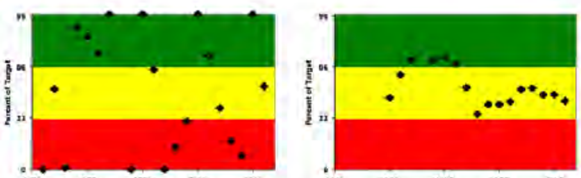

Figure 6. Five year metric used for nest production in northeastern Florida Bay. Left: Northeastern Florida Bay nest production (in chicks fledged per nest attempt) as a percentage of northwestern Florida Bay production since the completion of the South Dade Conveyance System. Right: The five year running mean of data presented in the figure on the right. This metric will be used as the stoplight metric for nest productivity unless the three control metrics for northwestern Florida Bay (Figure) are scored yellow and/or red.

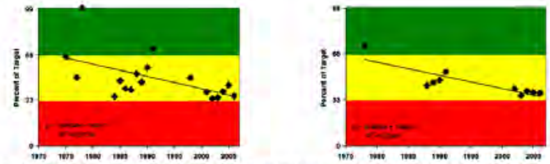

Figure 7. Bay wide nest number metric. Left: Number of nests bay-wide as a percentage of a target of 1250 nests. The target was set based on the maximum number of nests in Florida Bay prior to the completion of the South Dade Conveyance System (SDCS) as reported by Powell et al (1989). . Right: Five year running mean of the data presented to the right. Note that due to data limitations the earliest data point was a mean of only 3 years, however, the five year mean will be used for the actual stoplight metric.

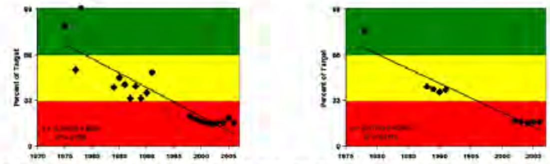

Figure 8. Nest location metric for northeastern Florida Bay. Left: Number of nests in northeastern Florida Bay as a percentage of a target of 625 nests. The target was set based on the maximum number of nests in northeastern Florida Bay prior to the completion of the SDCS as reported by Powell et al (1989). Right: Five year running mean of the data presented to the right. Note that due to data limitations the earliest data point was a mean of only 3 years, however, the five year mean will be used for the actual stoplight metric.

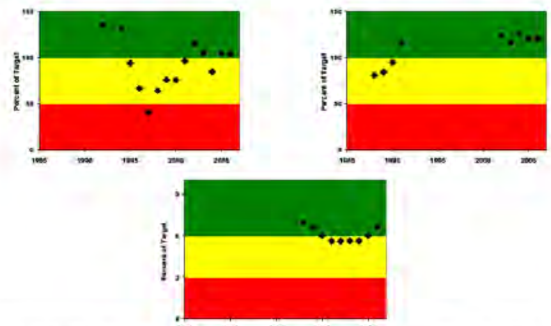

Figure 9. Control metric for using northwestern Florida Bay production as the standard for calculating the stoplight metric in northeastern Florida Bay (Figure). Top Right: Percentage of the target production rate of 1.25 chicks per nest fledged in northwestern Florida Bay since the completion of the SDCS. The target is based on the post-SDCS nest production data presented by Lorenz et al (2002). Top Left: Five year mean of the number of nests in northwestern Florida Bay as a percentage of a target of 200 nests. The target was set based on the average number of nests in northwestern Florida Bay since the completion of the SDCS as reported by Lorenz et al (2002). Bottom: The percentage years out of the previous ten in which spoonbills nesting in northeastern Florida Bay were successful (>1 chick per nest fledged). Note that due to data limitations we used the five year average in the figure, however, the ten year mean will be used for the actual stoplight metric.

Figure 7, 8, 9.

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Southern Estuaries Hypothesis Cluster-Submerged Aquatic Vegetation

Abstract

Seagrasses are the dominant biological communities in the coastal region to be affected by CERP and they provide the majority of the fisheries habitat in this system. The goal of the South Florida Fisheries Habitat Assessment Program (FHAP-SF) is to provide information for the spatial assessment and resolution of inter-annual variability in seagrass communities, and to establish a baseline to monitor responses of seagrass communities to water management alterations associated with CERP activities. FHAP-SF is documenting the status and trends of seagrass distribution, abundance, reproductive, and physiological status (ecoindicators), as well as providing process-oriented data such as photosynthetic quantum yields and epiphyte loads. Resource managers will be able to use these data to address ecosystem-response issues on a real-time basis and to weigh alternative restoration options.

Specific objectives of FHAP-SF are to: (1) develop a basic understanding of the relationships among water quality parameters (e.g. salinity, water clarity, nutrient levels) and seagrass species distribution and abundance in south Florida, (2) provide baseline data in in order to separate anthropogenically induced changes from natural system variation, and (3) assist in verifing model predictions on species and ecosystem-level responses to water quality changes associated with CERP. Results of the 2007 SSR suggest that the methods adopted can detect changes in SAV from pre-CERP conditions when there are sufficient reference data, and that the present trends are consistent with hypothesized causal relationships.

Background Description

Seagrasses (e.g., SAV) are characteristic of shallow coastal waters worldwide; however, few areas contain meadows as extensive as those found in the south Florida region (Fourqurean et al. 2002). SAV communities provide key ecological services, including organic carbon production, nutrient cycling, sediment stabilization, and enhanced biodiversity (Orth et al. 2006). These plants are not only a highly productive base of the food web, but are also a principal habitat for higher trophic levels. Because seagrasses live in close proximity to the land-sea interface, they are subject to physical disturbances and water quality changes associated with human population growth. As perennial plant species, seagrasses integrate net changes in water quality parameters (e.g. salinity, light availability, nutrient levels) which tend to exhibit rapid and wide fluctuations when measured directly. As such, seagrasses serve as biological sentinels of increasing anthropogenic influence in coastal ecosystems (Orth et al. 2006). To a large extent, seagrass abundance determines public perception regarding the health of the coastal waters of Florida (Goerte 1994, Boesch et al. 1995). Thus, the recent changes in the distribution and abundance of seagrasses within south Florida estuaries have been perceived as an especially significant change in the overall ecosystem health. For these reasons, seagrasses have been deemed one of the best indicators of change in the SE module (Fourqurean et al. 1992).

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Submerged Aquatic Vegetation Conceptual Ecological Model

The hypotheses described below are derived from a heuristic conceptual model (***Figure 7-20***) of the factors that influence SAV community structure (e.g., water management, land use and episodic events), and the interaction of SAV with estuarine organisms and the physical environment. CERP implementation will alter the volume, timing, and spatial distribution of freshwater inflow into the SE. SAV field data and concomitant water quality information are being collected to establish baselines (i.e., reference conditions) against which the extent of system change will be measured once CERP is implemented. Analysis of this pre-CERP data is needed to determine the extent of ecosystem change that will be detectible (and how long that might take) once CERP is implemented. At an early stage, it will also reveal systematic problems in the monitoring or analysis and highlight areas where significant improvements can be made.

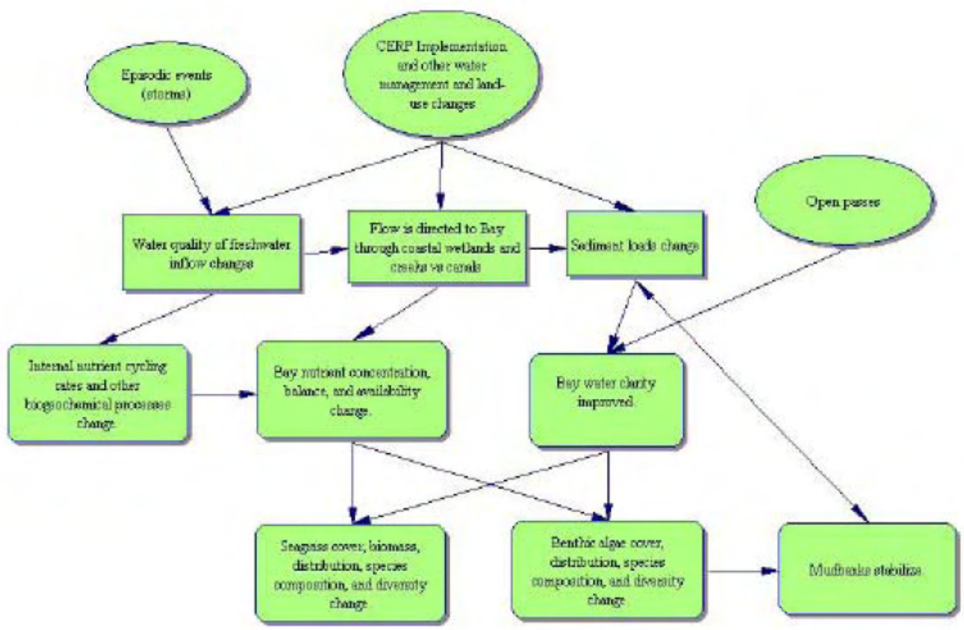

Figure 7-20: SE Module SAV CEM

Major CERP Relevant Hypotheses

• Hypothesis 1: Changes in both salinity and water quality resulting from CERP implementation are expected to result in changes in seagrass cover, biomass, distribution, species composition, and diversity though the combined and interrelated effects of light penetration, epiphyte load, nutrient availability, sediment depth, salinity, temperature, hypoxia/anoxia, sulfide toxicity, and disease.

• Hypothesis 2: Changes related to CERP implementation will include an expansion of areas with *Halodule wrightii* and *Ruppia maritima* cover and a reduction in areas of *Thalassia testudinum* monoculture along the northern third of Florida Bay. Based on

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forecasted changes in hydrology, seagrass density and species composition in the southern two-thirds of Florida Bay and the eastern half of Biscayne Bay are not expected to change.

• Hypothesis 3: Changes in both salinity and water quality resulting from CERP implementation are expected to change benthic algal cover, biomass, distribution, species composition, and diversity though the combined and interrelated effects of light penetration, nutrient availability, salinity, temperature, and changes in seagrass density and species composition.

• Hypothesis 4: Significant changes in benthic algae and seagrass distribution can affect susceptibility of sediments to become resuspended and the stability of mudbanks as well as nutrient availability to other primary producers.

Interim Goals

Submerged aquatic vegetation distribution and abundance are central ecological indicators of ecosystem health in the south Florida region, and as such are PMs throughout the SE domain. However, based on the indicator selection criteria (i.e., predictability [including adequate existing monitoring data], ecosystem restoration effect, ease of recognition and understanding by the intended audience, and manageable total number of indicators), the IGs for SAV in the SE module are currently limited to several locations within Florida Bay. Water management has dramatically altered the natural freshwater flow patterns (quantity, timing, and distribution) to Florida Bay. These changes, including reduced volume of freshwater inflow, are thought to have affected SAV in the Florida Bay ecosystem (McIvor et al. 1994, Durako et al. 2003, Rudnick 2004). The IGs for Florida Bay seagrass are based on an estimate of ecosystem conditions prior to major human interventions. These conditions (i.e., Florida Bay ecosystem history) were determined from paleoecological research and historical accounts (Brewster-Wingaard et al., 2003; Zieman et al., 1999; Cronin et al., 2001).

It is likely that the Florida Bay of the 1970s and early 1980s, with lush *T. testudinum* and clear water, was probably a temporary and atypical condition. Additional ecosystem history research and increased SAV and water quality monitoring will help refine the IGs for SE SAV. Presently, seagrass meadows in northeastern Florida Bay consist primarily of sparse *T. testudinum* communities. Central and western Florida Bay are dominated by sparse *T. testudinum* to dense *T. testudinum* meadows, but *H. wrightii*, and to a lesser extent *Syringodium filiforme* are also common. The occurrence and relative abundance of these community types vary by basin. From an ecological perspective, restoration targets are established that envision a more diverse seagrass community with lower *T. testudinum* density and biomass than during that anomalous period. A diversity of seagrass habitat is expected to be beneficial to many upper trophic level species (Thayer et al. 1999). CERPimplementation should affect SAV in the north shore mangrove zone lakes and coastal embayments (closer to freshwater source) more than offshore areas in the Florida Bay ecosystem. However, central Florida Bay should also be a primary focus area. Spatially explicit SAV restoration targets for the Florida Bay ecosystem are discussed in detail in the Florida Bay and Florida Keys Feasibility Study Draft PMs

(USACE and SFWMD 1999) and to a lesser extent in the Florida Bay and Everglades Mangrove Estuaries CEMs (Rudnick et al. 2005, Davis et al. 2005).

Methods and Analyses

SAV species have been monitored at ten Florida Bay locations since 1995 by the Florida Bay Fisheries Habitat Assessment Program (FHAP-FB). As a component of MAP, the geographic scope of FHAP-FB was expanded in 2005 to a total of 22 locations extending from the Lostman's River to northern Biscayne Bay (***Figure 7-20***), and the program was renamed the FHAPSF. Monitoring stations are determined using a systematic randomsampling design. Each location is divided into approximately 30 tesselated hexagonal grid cells (***Figure 7-21***), and a single station position is randomly chosen from within each grid cell during each monitoring event. Sampling grids were generated using algorithms developed by the EPA's Environmental Monitoring and Assessment Program (EMAP).

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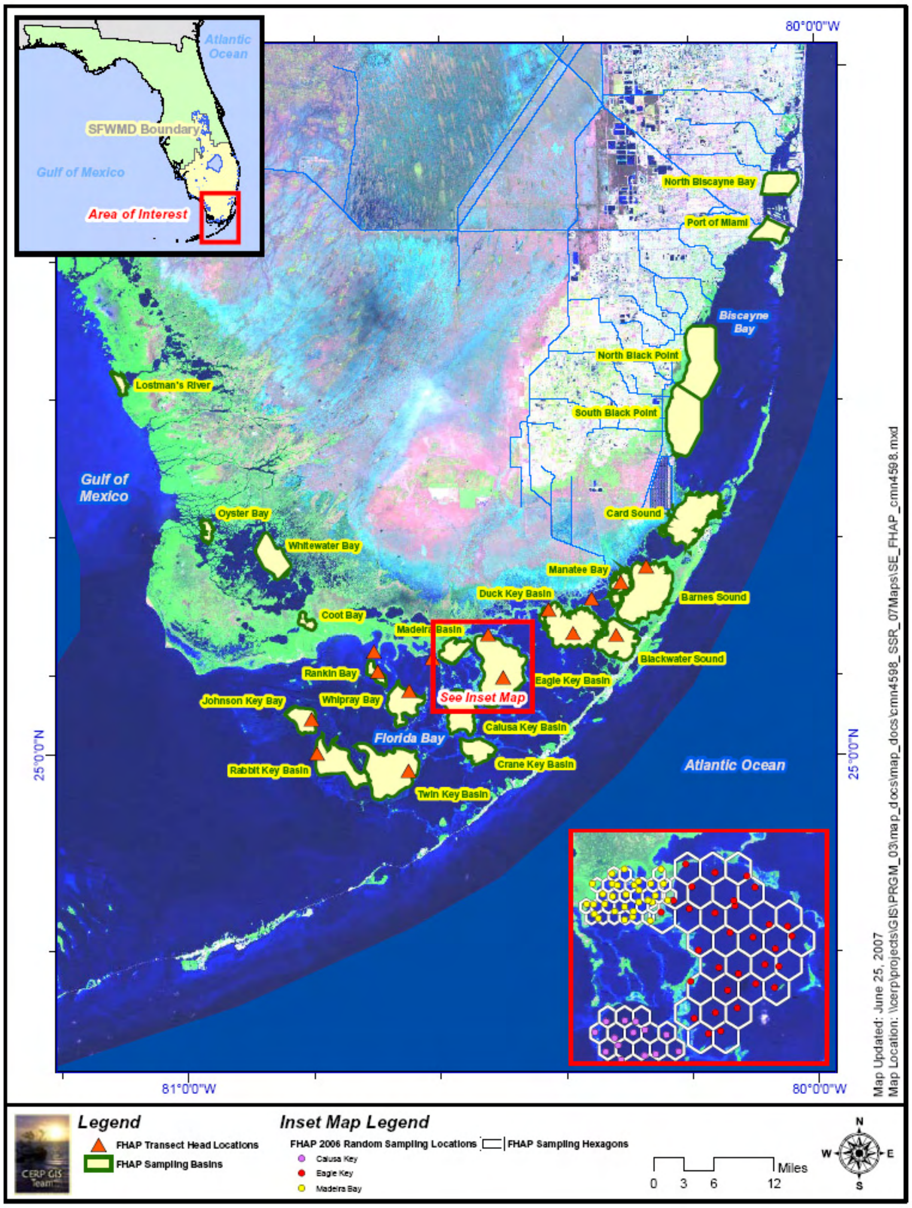

Figure 7-21: Map Depicting the FHAP-SF Sampling Locations and Sampling Design (see Inset Map)

Monitoring is conducted once per year at the end of the dry season (May-June). Salinity stress on seagrasses is typically highest at this time, and this is also the period when the dominant seagrass of the region, *T. testudinum* approaches its maximum leaf biomass, increasing the team's ability to detect changes in cover. Reproductive effort (flowering and fruit development) can also be assessed at this time. SAV community structure at each station is visually quantified using a modified Braun-Blanquet (BB) technique (Fourqurean et al. 2002). A series of 0.25 m^2 quadrats are placed on the bottom at each sampling station. The number of individual BB quadrats examined in each location during each year is provided below in ***Table 7-6.***

	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Lostman's River	-	-	-	-	-	-	-	-	-	-	360	240
Oyster Bay	-	-	-	-	-	-	-	-	-	-	348	240
Whitewater Bay	-	-	-	-	-	-	-	-	-	-	372	232
Coot Bay	-	-	-	-	-	-	-	-	-	-	384	248
Johnson Key Basin	128	124	120	128	124	124	124	112	107	124	336	240
Rabbit Key Basin	136	132	108	112	108	120	112	124	107	108	371	248
Twin Key Basin	128	120	132	124	124	120	120	128	128	120	384	240
Rankin Lake	148	136	144	136	136	120	132	116	132	136	372	240
Whipray Basin	128	120	132	124	116	112	124	124	124	128	372	240
Madeira Bay	124	124	128	132	132	116	128	124	116	132	335	240
Calusa Key Basin	120	120	124	116	116	120	116	116	124	116	360	240
Crane Key Basin	140	132	136	136	136	132	136	140	128	136	408	232
Eagle Key Basin	136	144	148	128	128	136	128	132	144	124	396	248
Duck Key Basin	-	-	-	-	-	-	-	-	-	-	360	232
Blackwater Sound	132	120	128	136	136	120	136	128	124	136	392	232
Manatee Bay	-	-	-	-	-	-	-	-	-	-	360	240
Barnes Sound	-	-	-	-	-	-	-	-	-	-	360	240
Card Sound	-	-	-	-	-	-	-	-	-	-	348	232
South Black Point	-	-	-	-	-	-	-	-	-	-	348	232
North Black Point	-	-	-	-	-	-	-	-	-	-	360	232
Port of Miami	-	-	-	-	-	-	-	-	-	-	336	232
North Biscayne Bay	-	-	-	-	-	-	-	-	-	-	276	208

Table 7-6: Number of Sampling Quadrats Surveyed in Each Basin for Each Year during the Spring Florida Bay Fisheries Habitat (FHAP) Monitoring Event

Species occurring within the quadrats are assigned a cover/abundance value according to the following scale: 0 = absent; 0.1 = solitary with small cover; 0.5 = few with small cover, 1 = numerous but < 5% cover; 2 = any number with 5-25% cover; 3 = any number with 26-50% cover; 4 = any number with 51-75% cover; 5 = any number with 76-100% cover. The average BB score for each species is computed for the quadrats within a site to yield an average BB density estimate for each location. Epiphyte loads are also determined for each site. Most recently PAM fluorometry has been used to estimate quantum yield/photosynthetic efficiency. Concomitant with the SAV sampling, physical data are also collected including at least depth, temperature, salinity, pH, dissolved oxygen, and PAR.

Most Florida Bay seagrass reports to date have qualitatively compared maps of BB estimates per species among different years (e.g., Durako et al. 2002). While these maps are extremely informative, it is felt that a more quantitative procedure would also be required for CERP assessment purposes. However, the distribution of the BB data raised concerns because of the large number of zero observations in the quadrats surveyed, as well as many cases where even the positive values were distributed in a highly non-normal manner (***Figure 7-22***). A procedure known as the delta approach, which has been useful in other contexts where data are positively skewed and zero values predominate (Fletcher et al. 2005), was chosen for testing. The delta approach as applied here involves generating two data sets from the original: the first indicating the species presence (occurrence or frequency proportion of quadrats positive for the species in question), and the second abundance when present (concentration, mean BB abundance per quadrat, when present). The product of frequency and mean abundance values yields an index of relative density. Such a statistic is considered more representative of the data than a mean density estimate calculated in the conventionalway, where the data set has a large number of zeros (Seber 1982). Separate analysis of the two components not only results in more robust estimation and understanding of the variance associated with each but typically reduces variability around the (composite) delta-mean value (Lo et al. 1992). Herein the term "delta-mean" will be used for the average deltadensity values calculated. Note that in the application using nontransformed data, the means calculated are identical to conventional means and only the variance changes. This approach was employed and proved successful in an analysis of spatio-temporal trends in shoreline fish species in southern Biscayne Bay (Serafy et al., 2007). It is also being employed in other MAP monitoring components such as the Juvenile Seatrout Monitoring for similar reasons (the large number of zero observations).

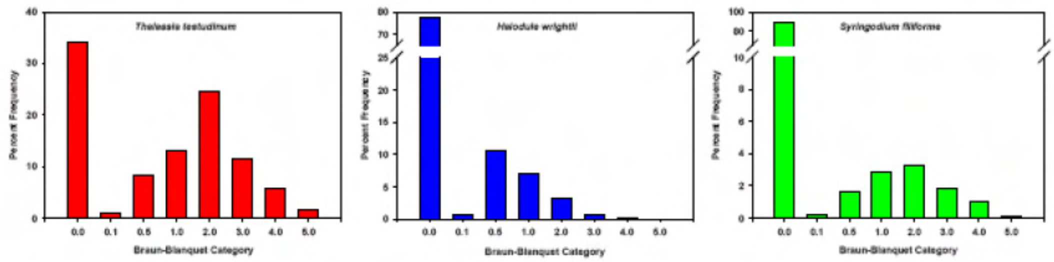

Figure 7-22: Frequency Distribution of BB Estimates for Representative Seagrass Species

Since all data collected thus far reflect pre-CERP conditions, the current assessment is simply the exercise of comparing the available baseline (pre-2006) data to the 2006 data with respectto the summary statistics discussed above. This exercise was confined to two Florida Bay locations, Johnson Key Basin and Blackwater Sound (***Figure 7-22***). These locations have substantially different environmental conditions (e.g., salinity patterns, sediment depth), and the seagrass in these areas have been monitored since 1995. For each of these basins, the 2006 values were compared to the means of the prior observations for that basin with respect to delta-mean and its constituent terms, frequency and concentration, and for predominant SAV taxa. A statistically significant difference occurred when the 2006 values fell outside the 95 percent confidence interval for the means of prior values (1995-2005). While not a power (or sensitivity) analysis in the formal sense, such an exercise is relatively free ofassumptions about the data and yields a quantitative appreciation for the underlying baseline data and its inherent variability (i.e., the context against which CERP induced changes will have to be discerned). In essence, a new tool to explore aspects of the data not usually considered is being added to the assessment toolbox, which will improve the SE module team's ability to detect CERP related changes beyond just comparing the spatial distributionof BB scores at different points in time.

Johnson Key Basin

Brief History–In 1987, extensive areas of *T. testudinum* began dying rapidly in central and western Florida Bay (including Johnson Key Basin). Factors that may have contributed tothe die-off were physiological stressors such as elevated water temperature and prolonged hypersalinity, excessive seagrass biomass leading to increased respiratory demands, hypoxia and sulfide toxity, and disease (Hall et al. 1999). Although *T. testudinum* mortality slowed substantially after several years, seagrass abundance in the central and western bay continued to decline due to an extended period of water column turbidity which began in 1991 and lasted until the late 1990s. Reduced water clarity was caused by resuspended sediments and phytoplankton blooms, most likely associated with the *T. testudinum* die-off (Durako et al. 2007). After water clarity improved, seagrass communities in Johnson Key Basin began to recover.

Results of Analysis–The 2006 Johnson Key Basin SAV community was composed of a variety of taxa, including substantial representation by *T. testudinum*, *H. wrightii* and S. *filiforme*, and occasional macroalgal species. With respect to *T. testudinum*, delta-density significantly differed in 2006 from the baseline condition (***Figure 7-23***), and did so with respect to both its components (frequency and concentration). In contrast, the 2006 *H. wrightii* delta-density was not significantly different from the baseline condition, although it declined substantially as a result of decreasing concentration with no change in occurrence.

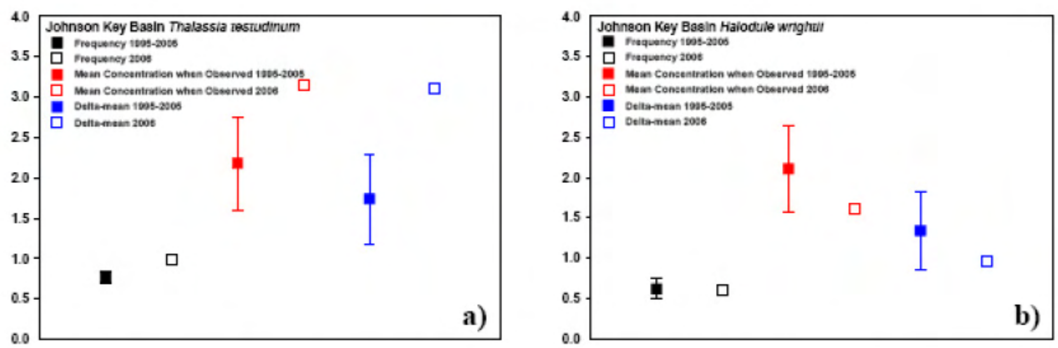

Figure 7-23: Comparison of 2006 spring SAV observations with mean spring observations and 95% confidence intervals from 1995-2005 (n=11) in Johnson Key Basin for (a) Thalassia testudinum and (b) Halodule wrightii
(open symbols indicate 2006 values)

An analysis of longer term trends (***Figure 7-24***) indicates that while *T. testudinum* has been increasing in both concentration and occurrence, the relative contribution of *H. wrightii* peaked approximately five years earlier in 2000.

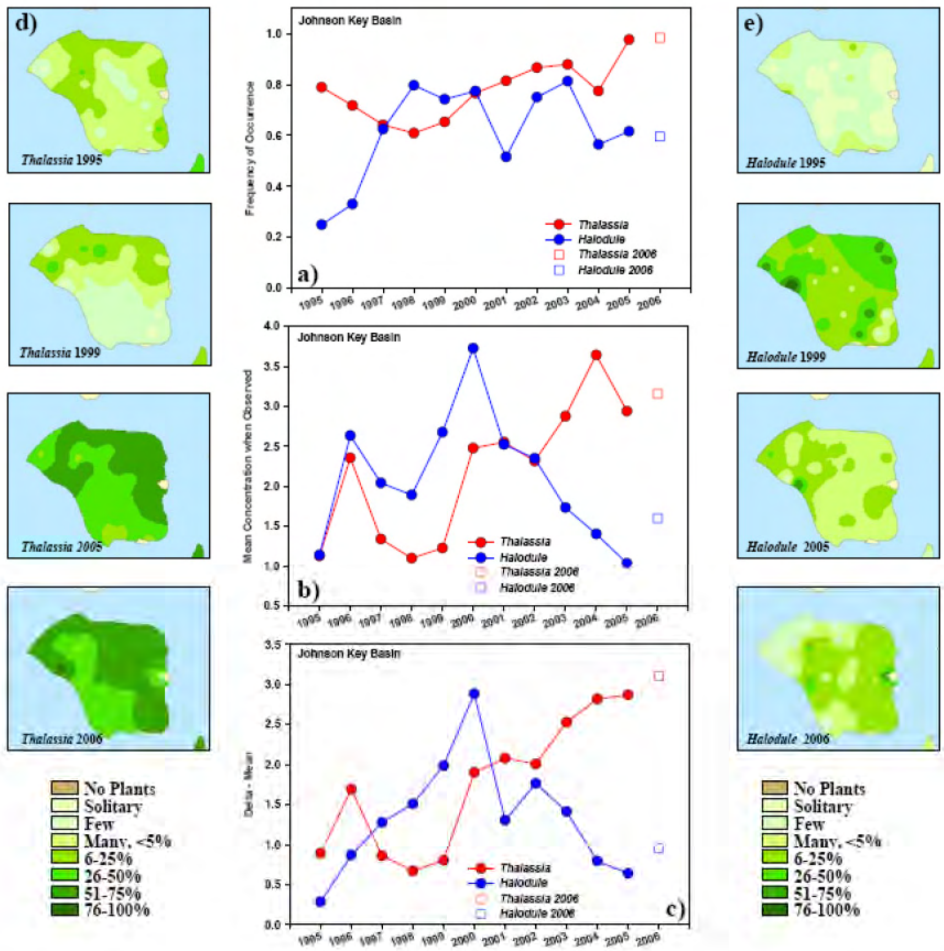

Figure 7-24: (a) Frequency, (b) Concentration, and (c) Delta-density values for *Thalassia testudinum* and *Halodule wrightii* in Johnson Key Basin from spring 1995 to 2006 (open symbols indicate 2006 values). Contour plots illustrate the distribution and abundance of (d) *Thalassia* and (e) *Halodule* in 1995, 1999, 2005, and 2006 in Johnson Key Basin

Comparing these temporal trends to Johnson Key Basin water quality (***Figure 7-25***), it was determined that the water quality trends are consistent with increasing light availability (lowered turbidity and water column chlorophyll *a*), increasing salinity, and decreasing water column nutrients (and one of the team's CERP hypotheses). The decreases in water column nutrients, turbidity, and chlorophyll *a* are also consistent with an overall increase in sediment stability, representing a positive feedback loop (another CERP hypothesis). In any case, there is little question that Johnson Key Basin is continuing to change as fish and invertebrate habitat in conjunction with water quality changes, and is doing so in a direction consistent with the SE module team's general hypotheses.

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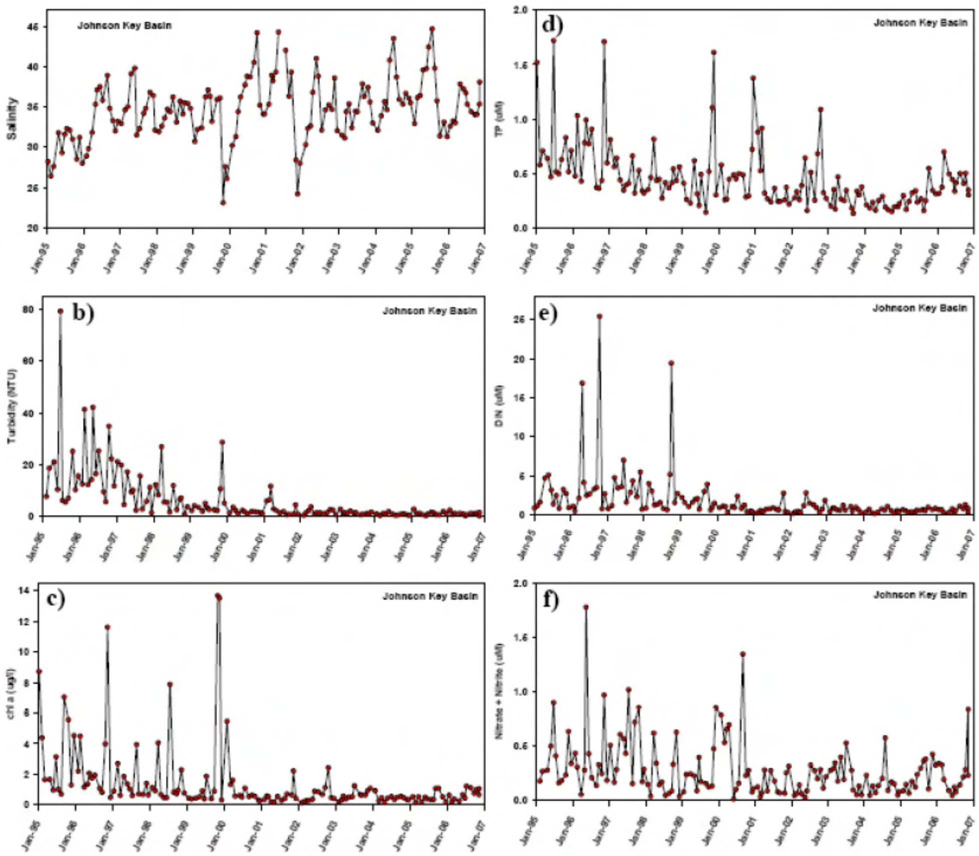
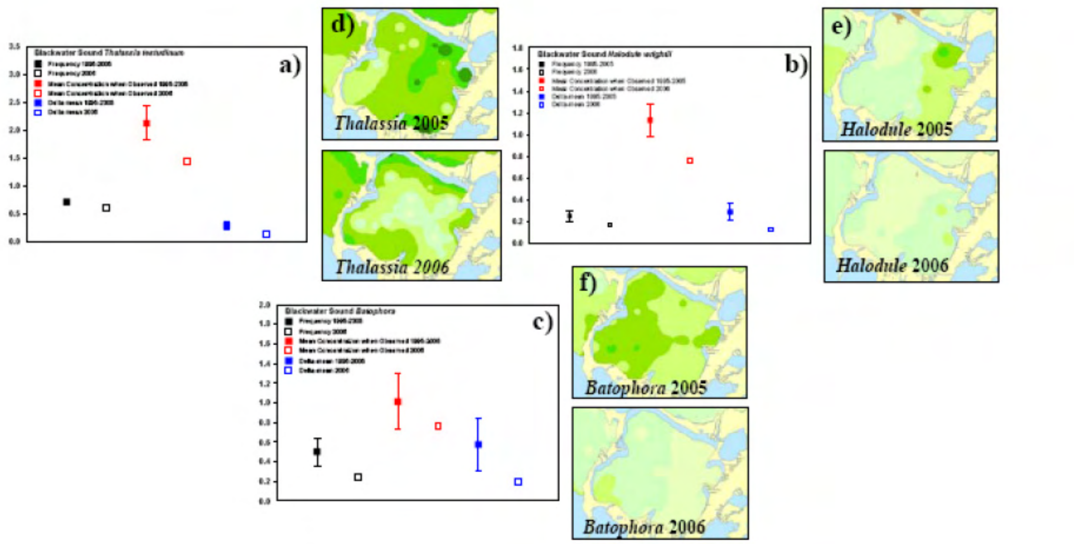


Data provided by the FIU SERC Water Quality Monitoring Network

Blackwater Sound

Brief History-A highly unusual algal bloom has persisted in northeastern Florida Bay and southern Biscayne Bay since fall 2005. Similar algal blooms have been observed in central and western Florida Bay, but never in eastern Florida Bay (Rudnick et al. 2006). Chlorophyll *a* concentrations (an indicator of the amount of algae in the water column) greatly exceeded values recorded during the previous fifteen years of water quality monitoring in this region (SFWMD/FIU Coastal Water Quality Monitoring Program). The algae bloom has been found to be mostly composed of blue-green algae, which are photosynthetic bacteria. Causes of the bloom are not certain, but may be related to at least two factors: 1) disturbance associated with road construction activity along U.S. Highway 1 between the Florida mainland and Key Largo (eighteen mile stretch); and 2) hurricane impacts from August through October 2005 (Hurricanes Katrina, Rita, Wilma). Highway construction has entailed the cutting and mulching of mangrove trees and soil tilling (mixing fresh mulch into the peat soil) and soil stabilization with injection of cement since May 2005. Hurricane disturbances included a large discharge of fresh water and P from the C-111 canal and the impact of high winds, waves, storm surge and abrupt salinity change on plants, soils, sediments, and ground water. The proximity of the blooms to both sides of U.S. Highway 1 (an area where blooms have not been previously recorded) indicates the likelihood that the unique disturbance of road construction is involved as a cause of the bloom (see Rudnick et al. 2006 for complete summary).

Results of Analysis-The 2006 Blackwater Sound SAV community was composed of sparse to moderate *T. testudinum*, sparse *H. wrightii*, and sparse *Syringodium* communities, and occasional macroalgal taxa (e.g., *Batophora*). Results of delta-mean analyses for *T. testudinum*, *H. wrightii*, and *Batophora* are illustrated in ***Figure 7-26***. Delta-density of all three taxa significantly differed in 2006 from the "baseline" condition, primarily as a result of significant declines in concentration.



(open symbols indicate 2006 values). Contour plots illustrate the distribution and abundance of d) *Thalassia*, e) *Halodule*, and f) *Batophora* in 2005 and 2006 in Blackwater Sound

Figure 7-26: Comparison of 2006 spring SAV observations with mean spring observations and 95% confidence intervals from 1995-2005 (n=11) in Blackwater Sound for a) Thalassia testudinum, b) Halodule wrightii, and c) Batophora

These declines are consistent with recent decreases light availability in Blackwater Sound due to substantial increases in both turbidity and chlorophyll *a* levels (***Figure 7-27***).

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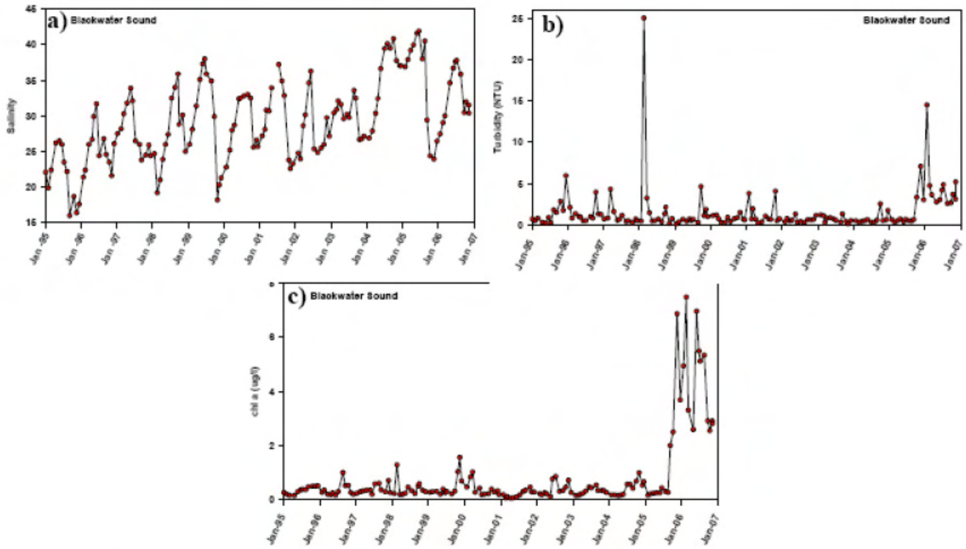

Data provided by the FIU SERC Water Quality Monitoring Network

Figure 7-27: Time-Series Plots in Blackwater Sound for (a) Salinity, (b) Turbidity, and (c) Chlorophyll *a*

7.3.4 Discussion

Results of the 2007 SSR suggest that the methods adopted can detect changes in SAV from pre-CERP conditions when there is sufficient reference data, and that the present trends are consistent with hypothesized causal relationships. Partitioning the relative contribution of the causal factors will require judicious application of the mechanistic SAV model currently being developed for the SE module, as well as some sensitivity analyses. It will also require a considerable time series of data after CERP is implemented. Implicit is the quantitative understanding of the relationship between water management changes and both salinity and water quality. Developing these relationships throughout the SE module will almost certainly depend upon integrated water quality monitoring and modeling (including hydrodynamic and hydrologic). The present analysis suggests that it will require a decade or more of monitoring to obtain an adequate amount of data to detect and interpret ecosystem change related to CERP activities. Fortunately given the present implementation schedule, such a time series will be available if MAP monitoring is sustained as planned. There is a relatively close relationship between the SAV monitoring (and assessment) and the present IGs with respect to the SE but it is far from perfect. In fact the current MAP monitoring will not in itself be sufficient (unless modified and supplemented) to address some of the refined spatial goals discussed above. Explicit targeted transect sampling will be required, but a more troubling concern may be the need for modeling purposes to accurately assess biomass (rather than estimating it indirectly from regression relationships based on limited data). The SE module team has begun to address these concerns by establishing 15 permanent SAV monitoring transects in Florida Bay. These transects are co-located with long-term water quality monitoring stations of the FIU/SERC Coastal Water Quality Monitoring Network, and will be sampled twice each year. Cores for seagrass biomass will be collected in addition to BB cover estimates and seagrass shoot counts. Depending upon model sensitivity, additional

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permanent transects may be required. It is probable that the IGs for SAV in the SE module will change in its next iteration. As a consequence, the MAP sampling will be updated in relatively short (two-four year) contract renewal intervals. The SE module team will have this opportunity, and will clearly need to take full advantage of the opportunity to, improve the match between the processes of model prediction and assessment.

Table 1. Basin-specific targets for Florida Bay seagrass status metrics nominally based on 10-year monitoring program record for each FATHOM basin. The zone within which each basin lies is indicated (NE=northeast, S= southern, W=western, C=central, TR=transition). Numbers in each cell represent the bound between "poor" and "fair" conditions left of the comma, and between "fair" and "good" conditions on the right. Zero and one are the lower and upper bounds for all ranges. Some targets were calculated based on shorter datasets.

FATHOM Basin	Zone	Seagrass Extent (a)	Seagrass Density (b)	Species Dominance (c)	Target Species (d)
5	NE	0.4, 0.6	0.1, 0.3	0.2, 0.5	0.1, 0.3
6	NE	0.4, 0.7	0.1, 0.2	0.2, 0.5	0.1, 0.3
7	TR	0.4, 0.7	0.1, 0.4	0.3, 0.7	0.1, 0.5
8	NE	0.5, 0.7	0.1, 0.3	0.2, 0.6	0.1, 0.3
9	NE	0.4, 0.6	0.1, 0.3	0.2, 0.7	0.1, 0.3
13	TR	0.4, 0.6	0.1, 0.4	0.3, 0.7	0.1, 0.5
14	TR	0.3, 0.6	0.1, 0.4	0.3, 0.5	0.1, 0.5
15	NE	0.4, 0.7	0.1, 0.3	0.2, 0.5	0.1, 0.3
21	S	0.6, 0.8	0.5, 0.7	0.1, 0.4	0.1, 0.4
22	C	0.6, 0.8	0.4, 0.6	0.2, 0.4	0.1, 0.4
24	C	0.6, 0.8	0.4, 0.6	0.2, 0.5	0.1, 0.4
32	S	0.6, 0.7	0.5, 0.7	0.1, 0.5	0.1, 0.4
34	C	0.6, 0.7	0.4, 0.6	0.2, 0.5	0.1, 0.4
37	C	0.6, 0.7	0.4, 0.6	0.2, 0.6	0.1, 0.4
38	W	0.6, 0.8	0.5, 0.8	0.2, 0.5	0.1, 0.3
47	NE	0.3, 0.5	0.1, 0.3	0.2, 0.6	0.1, 0.3

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Table 2. Seagrass indicator metrics and ranges for zones.

Spatial Extent (a)	**Northeast**	**Central**	**Western**	**Southern**	**Transition**
	0.0-0.4	0.0-0.6	0.0-0.6	0.0-0.6	0.0-0.4
	0.4-0.6	0.6-0.8	0.6-0.8	0.6-0.8	0.4-0.6
	0.6-1.0	0.8-1.0	0.8-1.0	0.8-1.0	0.6-1.0

Density (b)	**Northeast**	**Central**	**Western**	**Southern**	**Transition**
	0.0-0.1	0.0-0.6	0.0-0.6	0.0-0.5	0.0-0.1
	0.1-0.6	0.6-0.8	0.6-0.8	0.5-0.7	0.1-0.5
	0.6-1.0	0.8-1.0	0.8-1.0	0.7-1.0	0.5-1.0

Species Dominance (c)	**Northeast**	**Central**	**Western**	**Southern**	**Transition**
	0.0-0.2	0.0-0.2	0.0-0.2	0.0-0.1	0.0-0.1
	0.2-0.5	0.2-0.5	0.2-0.5	0.1-0.4	0.1-0.6
	0.5-1.0	0.5-1.0	0.5-1.0	0.5-1.0	0.6-1.0

Target Species (d)	**Northeast**	**Central**	**Western**	**Southern**	**Transition**
	0.0-0.1	0.0-0.1	0.0-0.1	0.0-0.1	0.0-0.1
	0.1-0.4	0.1-0.4	0.1-0.3	0.1-0.3	0.1-0.5
	0.4-1.0	0.4-1.0	0.3-1.0	0.3-1.0	0.5-1.0

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Table 3. Lookup table of decision gates for Abundance Axis A metrics for current status for Florida Bay, aggregating Extent and Density Indicator at zone scale (e.g. green + yellow = yellow).

Case	Extent Metric (a)	Density Metric (b)	Abundance Axis A
1	🟢	🟢	🟢
2	🟢	🟡	🟡
3	🟡	🟢	🟡
4	🟡	🟡	🟡
5	🟡	🔴	🔴
6	🟢	🔴	🔴
7	🔴	🟢	🔴
8	🔴	🟡	🔴
9	🔴	🔴	🔴

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Table 4. Lookup table of decision gates for Species Axis B metrics for current status for Florida Bay, aggregating Dominance and Target Species Indicator at zone scale (e.g. green + yellow = green).

Case	Dominance Metric (c)	Species Metric (d)	Species Axis B
1	🟢	🟢	🟢
2	🟢	🟡	🟢
3	🟡	🟢	🟢
4	🟡	🟡	🟡
5	🟡	🔴	🟡
6	🟢	🔴	🟡
7	🔴	🟢	🟡
8	🔴	🟡	🟡
9	🔴	🔴	🔴

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Table 5. Lookup table of decision gates for Carrying Capacity Axis C metrics for current status for Florida Bay, aggregating Abundance and Species Axes (A and B) at the zone scale (e.g. green + yellow = green).

Case	**Abundance Axis A**	**Species Axis B**	**Carrying Capacity Axis C**
1			
2			
3			
4			
5			
6			
7			
8			
9			

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2008 Assessment of the Algal Bloom Indicator for the Southern Estuaries (SE)

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Peter Ortner, CIMAS
Davis Rudnick, SFWMD

Background: This assessment on the Algal Bloom Indicator is taken from the RECOVER Comprehensive Everglades Restoration Plan 2007 System Status Report (November 2007).

The Southern Estuaries Algal Bloom Indicator is one of five indicators directly linked to the SE Module. They include; pink shrimp, Florida Bay submerged aquatic vegetation, crocodiles, and roseate spoonbills. In addition, fish, white ibis, and wood storks are indirectly linked to conditions in the southern estuaries.

The SE Indicator includes Florida Bay, the coastal lakes inland from Florida Bay, Biscayne Bay, and estuaries within SW Florida's mangrove zone from Whitewater Bay to Lostmans River (Figure WQ - 2). Altered freshwater inflows have affected circulation, water quality and salinity patterns of the SE, in turn altering the structure and function of these coastal ecosystems. Changes in water quality and salinity and associated loss of dense turtle grass colonies and other submerged aquatic vegetation (SAV) in Florida Bay has created a condition in the bay where sediments and nutrients are regularly disturbed, frequently causing large and dense algal blooms. These blooms in turn often cause further loss of more recently established SAV exacerbating the conditions causing the algal blooms.

The SE and the plants and animals they support reflect the volume, distribution, timing and quality of fresh water flowing into these systems. Past changes to the quality, quantity, timing and distribution of freshwater flow have degraded water quality and altered salinity patterns thus compromising estuarine community composition and function in some areas of the SE.

Chlorophyll *a* (CHLA) was selected as an indicator of water quality because its biomass is an integrator of many of the water quality factors which may be altered by restoration activities. There is concern that the increased freshwater flow due to restoration may result in more frequent, intense and persistent phytoplankton blooms in the SE. The baseline conditions indicate that most of the SE are oligotrophic with median CHLA concentrations of approximately 1 ppb.

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WQ.1 Background Description of Southern Estuaries Water Quality Hypothesis

Water quality in the SE is dependent upon the volume, distribution, and quality of freshwater flowing to the system. The biotic components (e.g., phytoplankton, benthic habitats) of estuaries are sensitive to salinity variability and nutrient loading which may be modified by CERP. Complex interactive mechanisms between water quality and hydrologic drivers as well as internal nutrient cycling will influence CERP effects.

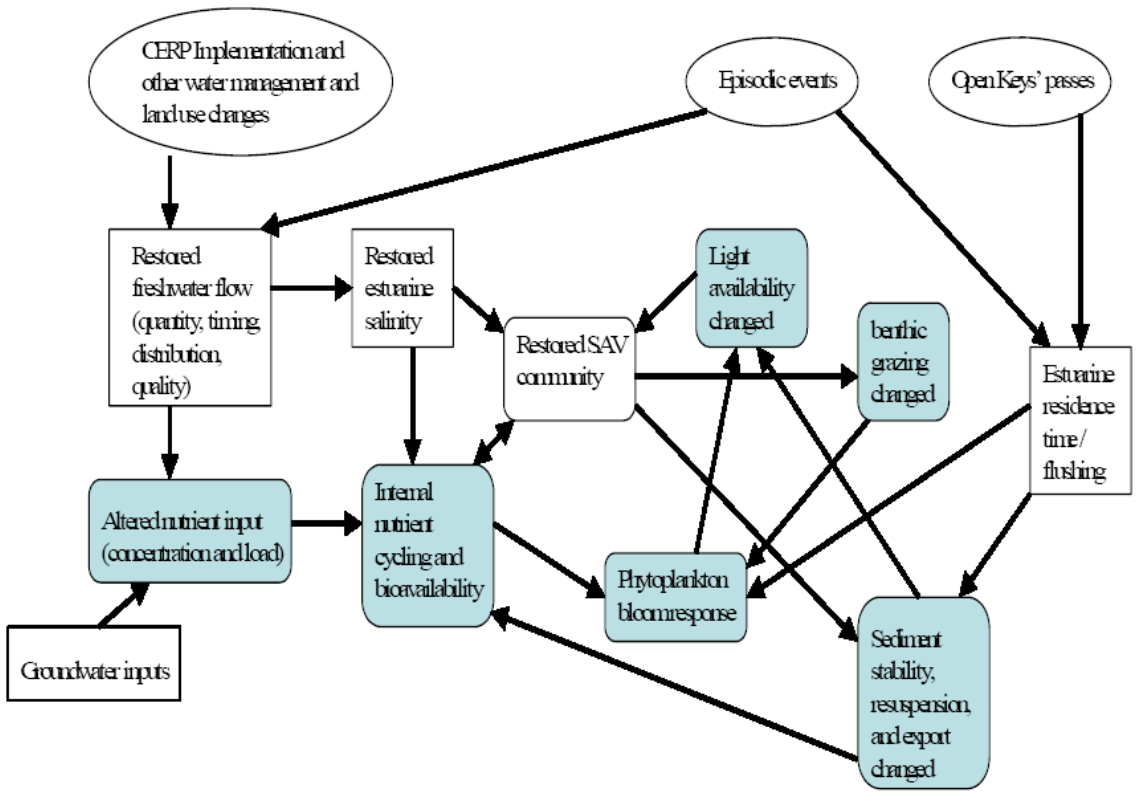

Figure WQ-1: Water Quality Conceptual Ecological Model

Major Relevant CERP Hypotheses

- Through modifications of quantity, quality, timing and distribution of freshwater, CERP implementation will affect dissolved and particulate nutrients delivered to the estuaries and alter estuarine water quality. These modifications will affect primary production and food webs in estuaries. These modifications include:
  1) *changes in the distribution and timing* of nutrient inputs through increased flow via Shark River Slough and diversion of canal flows from 'point source' to more 'diffuse' delivery through coastal wetlands and creeks;
  2) *changes in the quantity* of nutrient inputs to the estuaries through alteration of the mobilization and release of nutrients from developed and agricultural areas, through nutrient uptake in treatment areas, and through changes in nutrient processing and retention in the Everglades;

3) *changes in the* bioavailability of nutrients which depend on both the *quality* of nutrients (e.g., inorganic nutrients and DOM) from the watershed and internal estuary mechanisms (e.g., P limitation of DOM decomposition);

- Internal nutrient cycling rates (e.g., nitrogen fixation and denitrification) and biogeochemical processes, such as phosphate sorption, will change with CERP implementation because of salinity and benthic habitat changes.
- Nutrient accumulation and retention in estuaries is affected by episodic storm events, which can export nutrient rich sediments. CERP implementation will modify benthic habitats and nutrient loading which will affect this export.
- The spatial extent, duration, density, and composition of phytoplankton blooms are controlled by several factors that will be influenced by CERP. These include:
  1) external nutrient loading;
  2) internal nutrient cycling (seagrass productivity/die-off, sediment resuspension);
  3) light availability (e.g., modified by sediment resuspension and CDOM);
  4) water residence time;
  5) biomass of grazers (e.g., zooplankton, benthic filter-feeders).
- Nutrients inputs from groundwater discharges may affect water quality in coastal wetlands and estuaries. CERP implementation will modify these discharges in the coastal zone which will alter nutrient loads to the estuaries.

WQ.2 Data Status/Availability for Water Quality Hypothesis Cluster

Systematic monitoring of water quality at fixed stations in the southern estuaries has been ongoing since late 1989 as part of Florida International University's Southeast Environmental Research Center's (FIU/SERC) Water Quality Monitoring Network. This effort began in Florida Bay and by the mid-1990s had expanded to include the entire southern estuaries domain, including the mangrove transition zone (Table 1). Also, beginning in the mid-1990s NOAA/AOML began monitoring water quality and circulation throughout the southern estuaries via fixed station sampling and continuous synoptic sampling. All of the fixed stations (Figure 1) except those located on the southwest Florida shelf had been sampled monthly by both programs until recent funding shortcomings forced NOAA to cut sampling down to six times per year, hence the decreased sampling effort in 2006 (Table 1).

The continuous synoptic sampling measures sea surface temperature, salinity, chlorophyll *a* fluorescence that can be converted to biomass estimates, beam transmission (λ=660) that can be used to estimate Total Suspended Solids (TSS), and Chromophoric Dissolved Organic Matter (CDOM) fluorescence. These measures can then be used to estimate light attenuation along the underway track which is useful to determine if phytoplankton and/or seagrass growth is light-limited within specific regions of the southern estuaries. At each of the fixed stations samples are collected for chlorophyll *a* biomass and dissolved inorganic nutrients. Additionally, NOAA/AOML samples light attenuation, TSS, DOC, and pH at each station, and FIU/SERC samples TOC, TP, APA, and TN at each station. Recent analyses of water quality in the southern estuaries include: Boyer et al. (1997; 1999) for Florida Bay and mangrove transition zone water quality distributions and trends, Rudnick et al. (1999) for Florida Bay nutrient loading, Kelble et al. (2005) for Florida Bay light attenuation, Kelble et al.

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(2007) for Florida Bay salinity variability, Caccia and Boyer (2005) for Biscayne Bay water quality distributions and Jurado et al. (2007) for bloom dynamics on the southwest Florida shelf. Much of this data is available to the public at www.aoml.noaa.gov/sfp/ and http://serc.fiu.edu/wqmnetwork/SFWMD-CD/index.htm.

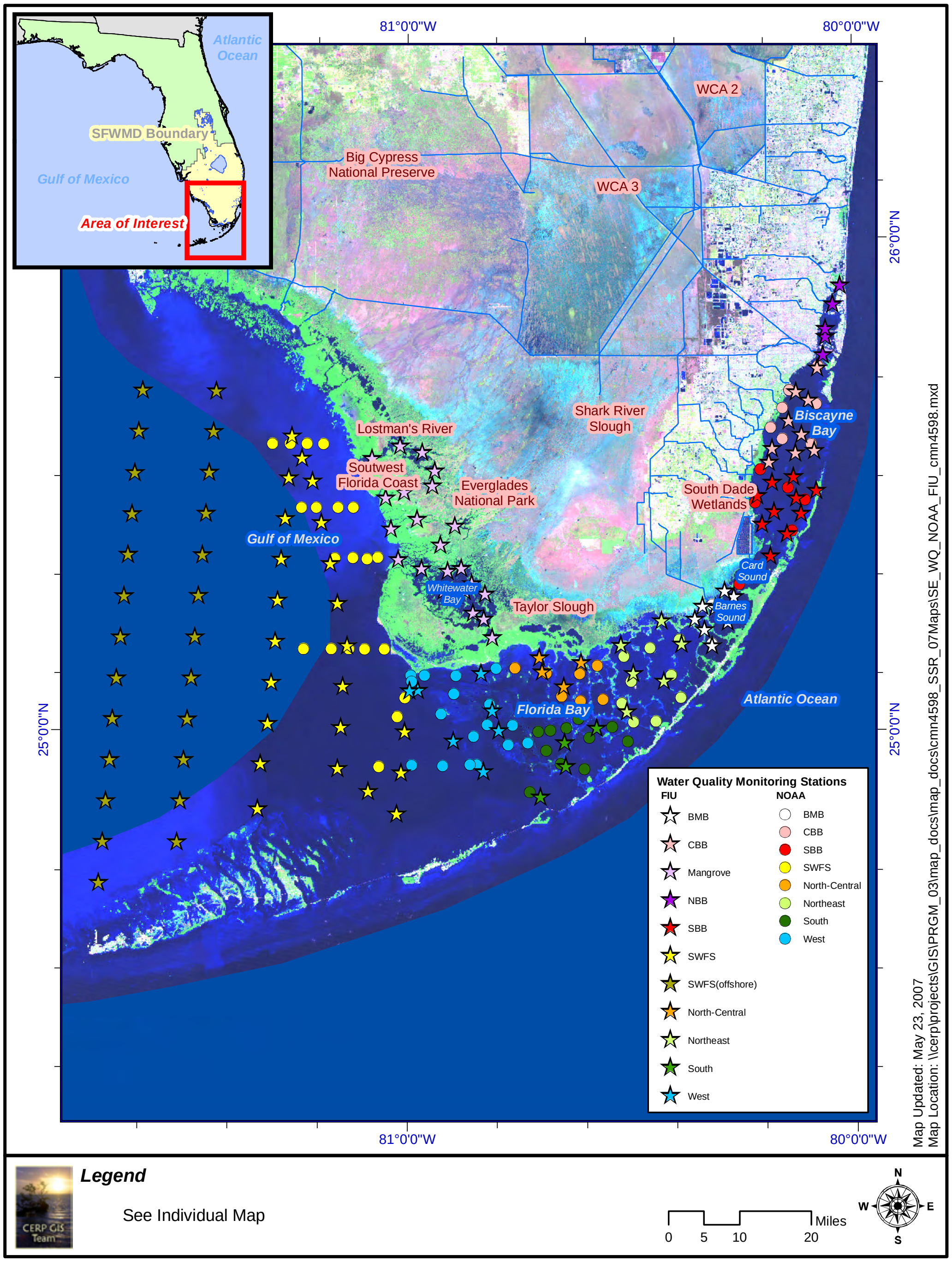

Figure WQ-2: Map depicting NOAA/AOML's and FIU/SERC's fixed water quality sampling stations in the southern estuaries.

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Table WQ-1: Number of fixed station samples for water quality in each sub-region.

	SWFS	MTZ	WFB	SFB	NCFB	NEFB	BMB	SBB	CBB	NBB
1989	-	-	30	20	20	30	40	-	-	-
1990	-	-	90	60	60	90	120	-	-	-
1991	-	-	78	52	52	78	104	-	-	-
1992	-	88	72	48	48	72	96	-	-	-
1993	-	264	72	48	48	72	96	68	32	-
1994	-	264	78	52	52	78	104	204	96	-
1995	72	264	72	48	48	72	96	204	96	-
1996	72	242	86	63	62	89	96	162	103	35
1997	216	286	86	62	58	90	96	132	108	60
1998	216	242	152	106	104	132	96	121	99	55
1999	216	264	182	140	112	152	96	132	108	60
2000	216	264	209	161	120	169	96	132	108	60
2001	216	264	214	164	125	167	96	132	108	60
2002	216	264	215	169	124	171	114	173	139	60
2003	216	264	202	158	118	165	126	201	167	60
2004	186	264	232	180	132	180	132	216	180	60
2005	195	242	222	158	118	162	123	202	174	60
2006	182	264	166	114	89	126	114	174	150	60

WQ.3 Analysis framework to assess water quality in the southern estuaries

Based upon the major relevant CERP water quality hypotheses it was determined that salinity and chlorophyll *a* biomass should be utilized as the primary indicators to assess the status and trends in water quality for the southern estuaries. The hypotheses further state that CERP will affect the rates of external nutrient loading and internal nutrient cycling by several different mechanisms. These rates along with three other factors (light availability, water residence time, and biomass of grazers) which may also be influenced by CERP activities control the magnitude, duration, and spatial extent of phytoplankton blooms for which chlorophyll *a* is a proxy. Moreover, phytoplankton blooms are a major concern to the overall health of the southern estuaries (Rudnick et al. 2005). These blooms decrease light penetration through the water column that can lead to seagrass mortality. Seagrass mortality often results in the release of more nutrients via decomposition and increased sediment resuspension, which in turn stimulates more phytoplankton growth (Rudnick et al. 2005; Zieman et al. 1999). This potential to propagate a negative feedback loop throughout the ecosystem elevates the importance of monitoring water quality and chlorophyll *a*.

The role of nutrient inputs from the Everglades in initiating and perpetuating algal blooms in the southern estuaries is unclear and likely varies throughout the region. Several studies have hypothesized that this is an important factor and that increased freshwater flow with CERP may intensify algal blooms in the southern estuaries (CROGEE 2002; Brand 2002; Jurado et al. 2007). Given this possibility, it is necessary to quantify and understand the baseline conditions for salinity and chlorophyll *a* and be capable of identifying deviations from this baseline which may occur as CERP is implemented. The behavior of water quality variables, particularly salinity and chlorophyll *a*, is distinct throughout individual sub-regions of the southern estuaries due to differences in freshwater runoff patterns (Kelble et al. 2007; Nuttle et al. 2000), circulation (Lee et al. 2006), sediment biogeochemistry (Zhang et al. 2004), nutrient inputs (Rudnick et al. 1999), grazer biomass (Peterson et al. 2006), and phytoplankton species composition (Phlips and Badylak 1996). Therefore, it was necessary to subdivide the southern estuaries module into ten sub-regions (Fig. WQ-2) based upon statistical methodologies (Boyer et al. 1999; Caccia and Boyer 2005) and analysis of circulation patterns (Lee et al. 2006; Lee et al. 2007).

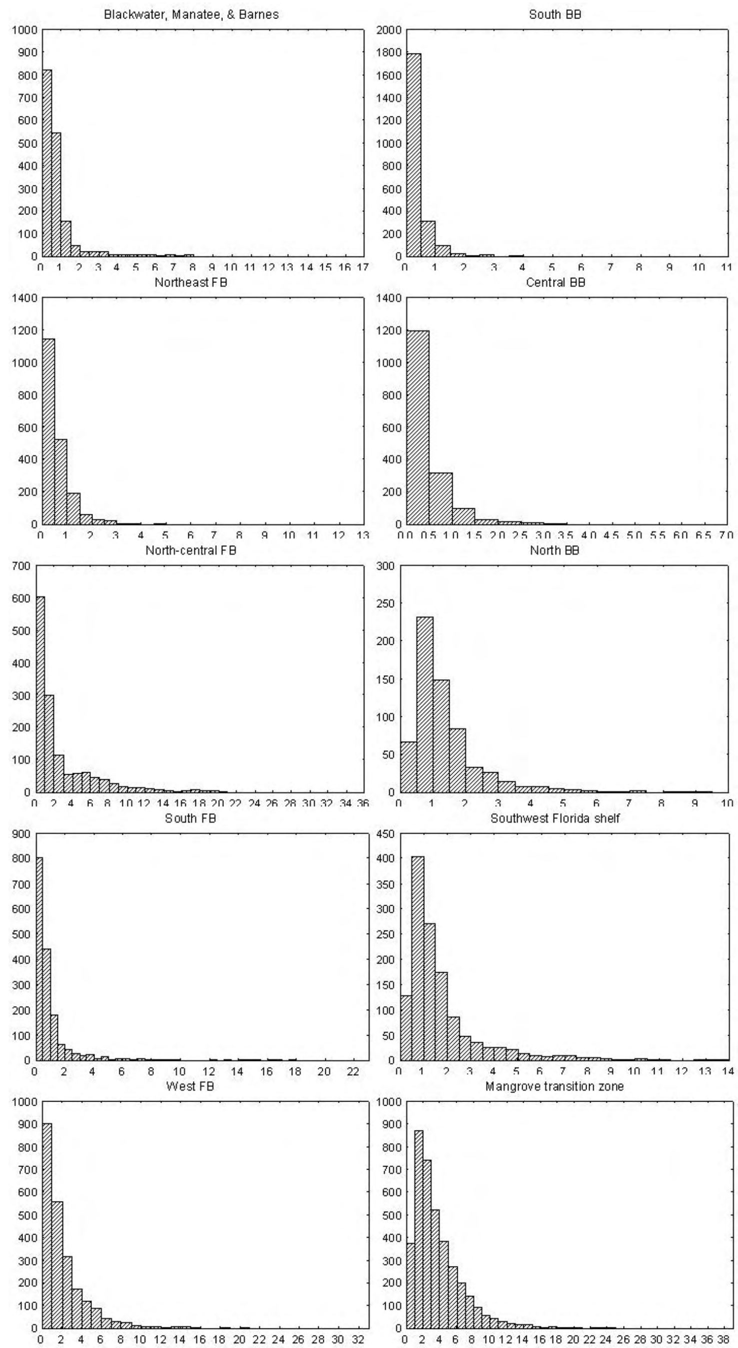

Figure WQ-3: Histograms of chlorophyll *a* (ppb) in each sub-region

The ten subregions are southwest Florida shelf (SWFS), mangrove transition zone (MTZ), west Florida Bay (WFB), north-central Florida Bay (NCFB), south Florida Bay

(SFB), northeast Florida Bay (NEFB), Blackwater, Manatee, and Barnes Sounds (BMB), south Biscayne Bay (SBB), central Biscayne Bay (CBB), and north Biscayne Bay (NBB). The distribution of chlorophyll *a* concentrations was not normal in any of these sub-regions always being heavily weighted towards lower concentrations (Fig. WQ-3). As such, the midpoint of the data was best represented by the median and it was necessary to conduct non-parametric statistical tests to analyze the data. EPA guidelines were applied to establish the reference conditions for chlorophyll *a* concentrations and set criteria for determining what constitutes elevated levels of chlorophyll *a* (EPA 2001). This approach established that a median concentration greater than the reference conditions 75th percentile would be classified as elevated from baseline. Furthermore, Kruskal-Wallis tests were employed to statistically test for differences in chlorophyll *a* between 2006 and all data collected prior to 2006. If any differences were measured, more detailed analyses were undertaken to identify underlying changes in water quality parameters and determine the ultimate cause(s) of the observed change.

WQ.4 Present condition of water quality

The present condition of water quality in the southern estuaries has been the subject of numerous previously mentioned peer-reviewed papers. For consistency when undertaking the bi-annual assessment effort, the current condition of salinity and chlorophyll *a* were examined by a standard easily applied methodology. To examine the distribution of chlorophyll *a* throughout the southern estuaries, the data was divided between months that typically have high salinities (April-September) and those that have low salinities (October-March). This was determined based on analysis of salinity patterns in Florida Bay and Biscayne Bay (Fig. WQ-4). Then, the median for each station during high and low salinity months was calculated and the results were plotted with Surfer (Fig. WQ-5). The highest chlorophyll *a* concentrations are consistently measured along the southwest Florida coast, both in the mangrove transition zone and on the southwest Florida shelf. During low salinity, the elevated chlorophyll *a* water expands further west onto the shelf, further south towards the Keys, and further east along the northern edge of Florida Bay. The SFB, NEFB, BMB, SBB, CBB, and NBB sub-regions had consistently lower chlorophyll *a* concentration for both high and low salinity periods.

The median monthly chlorophyll *a* concentration was calculated in each sub-region and the typical annual cycles of chlorophyll *a* were examined for each sub-region (Fig. WQ-6). As depicted in the contour map there were significant differences in the magnitude of chlorophyll *a* between sub-regions. The three regions of Biscayne Bay displayed similar annual cycles in chlorophyll *a* with elevated concentrations from early summer through the end of the year. However, the NBB sub-region had over double the median chlorophyll *a* for each month compared to the other two sub-regions. There were significant differences in the annual cycles for the five sub-regions of Florida Bay, although they all had higher concentrations in the second half of the year. NCFB displayed the largest degree of variability with a peak in October that was over three times the lower values observed from January through June. SFB had the second largest amount of variability with values in the second half of the year almost double those for the first half of the year. WFB had the highest median values for almost all months with

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all of the median monthly values greater than 1 ppb. BMB and NEFB had the lowest chlorophyll *a* concentrations without much variability. The southwest Florida coast region had significant differences between its two sub-regions. The MTZ had consistently high levels of chlorophyll *a* with a slight seasonal shift of decreased chlorophyll *a* during the second half of the year, which is the opposite of all other sub-regions in the southern estuaries. SWFS had a large degree of seasonal variability with a large peak in median chlorophyll *a* in November. However, this peak may be an artifact of the sampling effort in this sub-region which is undertaken on a quarterly basis. Thus each month has not been sampled each year and the results may be biased by sampling during November only in years with elevated chlorophyll *a* concentrations in this sub-region.

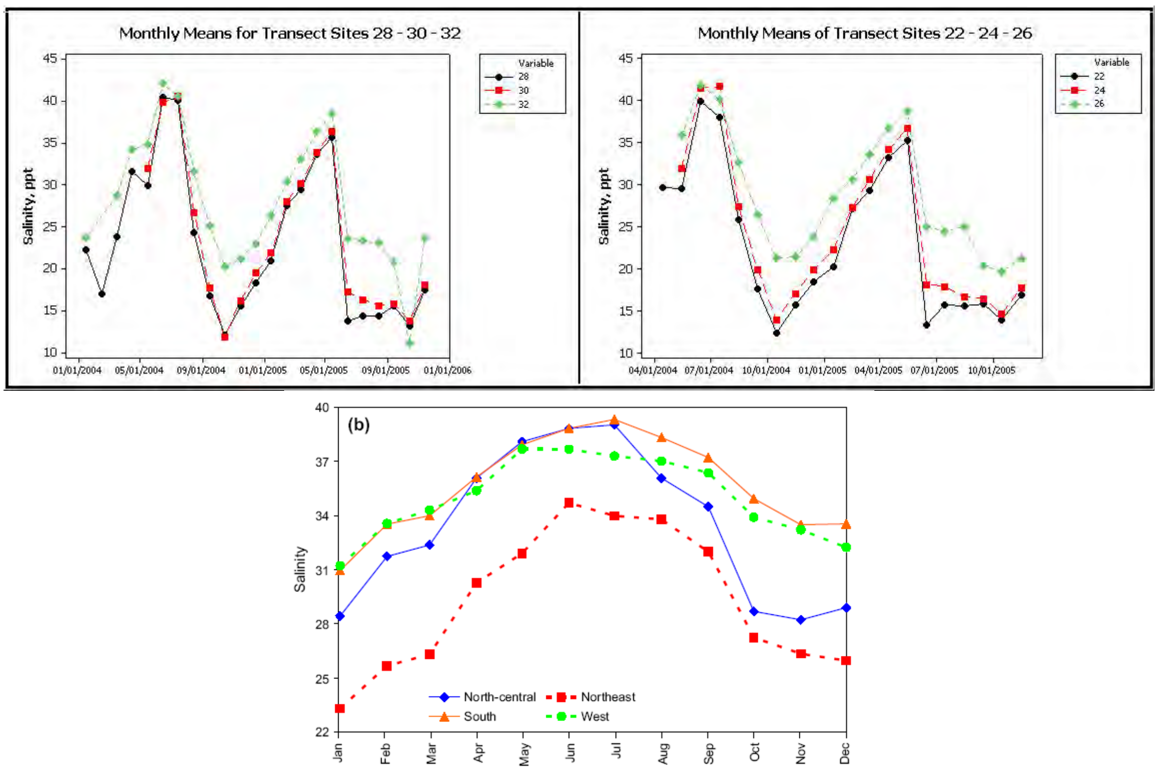

Figure WQ-4. Salinity cycles in Biscayne Bay (top two panels) and Florida Bay (bottom panel)

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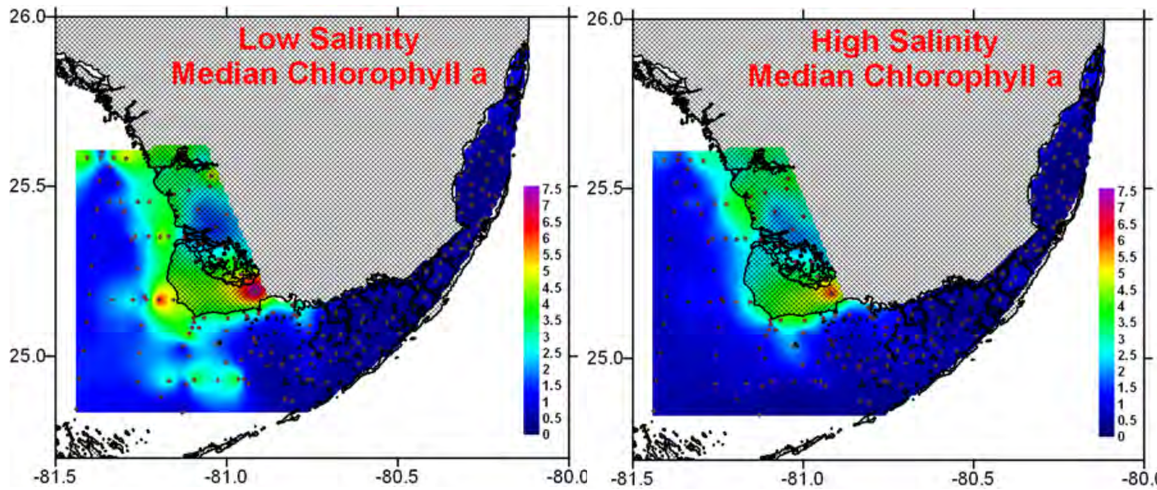

Figure WQ-5: Contour plots of the median chlorophyll *a* distribution in the southern estuaries during low salinity months (October-March) and high salinity months (April-September).

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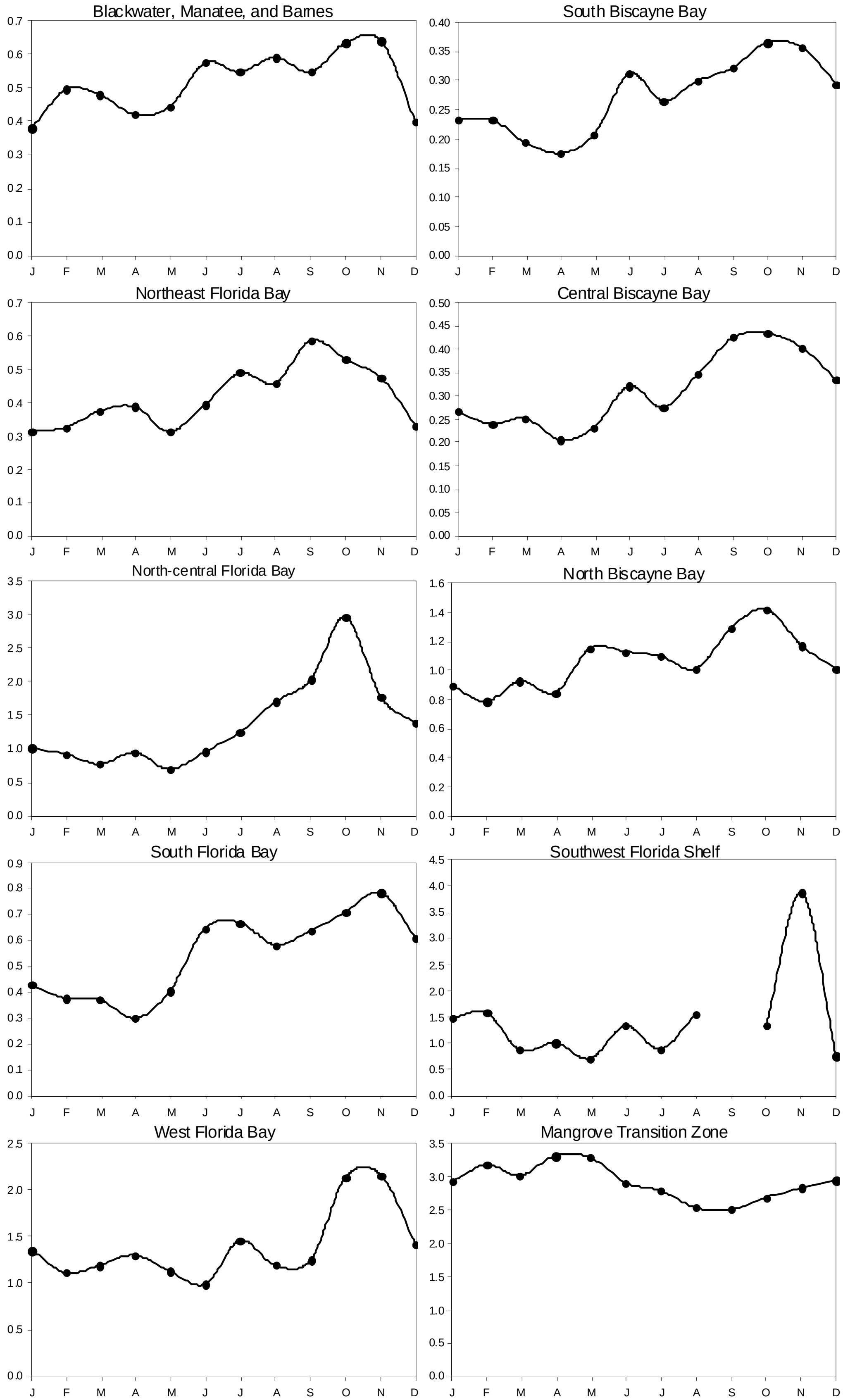

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Figure WQ-6: Annual cycle of median chlorophyll *a* in each sub-region.

WQ.5 Detecting Change

To detect change the data were analyzed with respect to the EPA guidelines outlined above. The median and quartiles were calculated to quantify the reference conditions for the ten sub-regions of the southern estuaries (Table WQ-2). These reference conditions were then used to establish criteria from which the status of chlorophyll *a* and thus water quality in each of the sub-regions can be evaluated on an annual basis. If the annual median chlorophyll *a* concentration is greater than the reference median, but lower than the 75th percentile, the sub-region is marked yellow and if the annual median concentration is greater than the 75th percentile of the reference, the sub-region is marked red. This approach sets low thresholds (almost half of the sub-regions go red at greater than 1 ppb) and regions with higher thresholds like FBNC will still go yellow at slightly over 1 ppb. The only exception is the mangrove transition zone which has significantly higher thresholds. The data is plotted as a series of annual box and whisker plots to provide a visual representation of the analysis and account for the variability in the data. This also allows the criteria to be somewhat malleable, because a significant change in the variability will be observed even if there is not a coincident change in the median (Fig. WQ-7). The box and whisker plots have the median as their centerline, the 95% confidence intervals of the median as the notches in the box, the 25th and 75th percentiles demark the edges of the box and the whiskers extend to the 10th and 90th percentile. Thus, the notches and the boxes can be utilized as a pseudo-test for significant differences between medians.

Table WQ-2. Criteria for evaluating chlorophyll *a*.

Sub-region		Valid N	25th Percentile	Median	75th Percentile
Blackwater, Manatee, Barnes	BMB	1704	0.306	0.526	0.910
Central Biscayne Bay	CBB	1673	0.200	0.313	0.566
Mangrove Transition Zone	MTZ	3803	1.690	2.863	4.903
North Biscayne Bay	NBB	635	0.670	1.048	1.648
North-central Florida Bay	NCFB	1399	0.585	1.216	3.710
Northeast Florida Bay	NEFB	1979	0.254	0.417	0.790
South Biscayne Bay	SBB	2257	0.181	0.264	0.426
South Florida Bay	SFB	1695	0.327	0.533	1.059
Southwest Florida Shelf	SWFS	1297	0.739	1.180	1.976
West Florida Bay	WFB	2304	0.653	1.345	2.845

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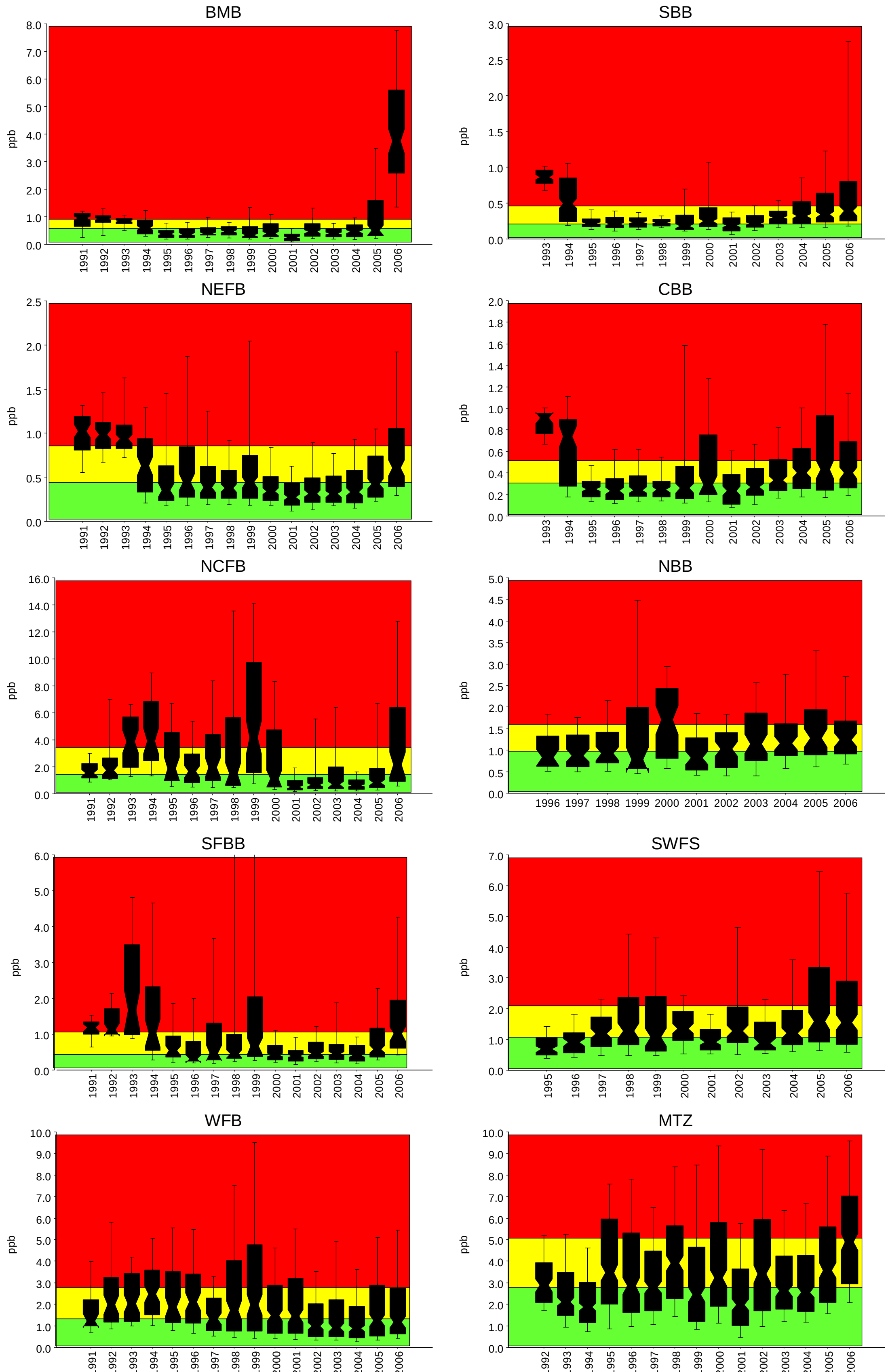

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Figure WQ-7: Box and whisker plots of annual chlorophyll *a* in each sub-region.

From this box and whisker analysis a stoplight map is produced to display the status of chlorophyll *a*/water quality in each sub-region (Fig. WQ-8). The sub-regions which receive yellow ratings may undergo further analysis, if a Kruskal-Wallis test shows there has been a significant change in median chlorophyll *a* concentration. The additional statistical test is conducted, because a random sample will be higher then the median and thus yellow 50% of the time even if no significant change has occurred. The sub-regions which have received red ratings will be further evaluated to determine the cause of degradation in water quality and determine if it was the result of CERP, natural variability, or other anthropogenic activities. The physical environment of the southern estuaries, particularly salinity responds to meteorological events, such as tropical cyclones and El Niño (Fig. WQ-9). Thus, water quality likely responds to these natural events and a change must be shown to be definitively due to CERP.

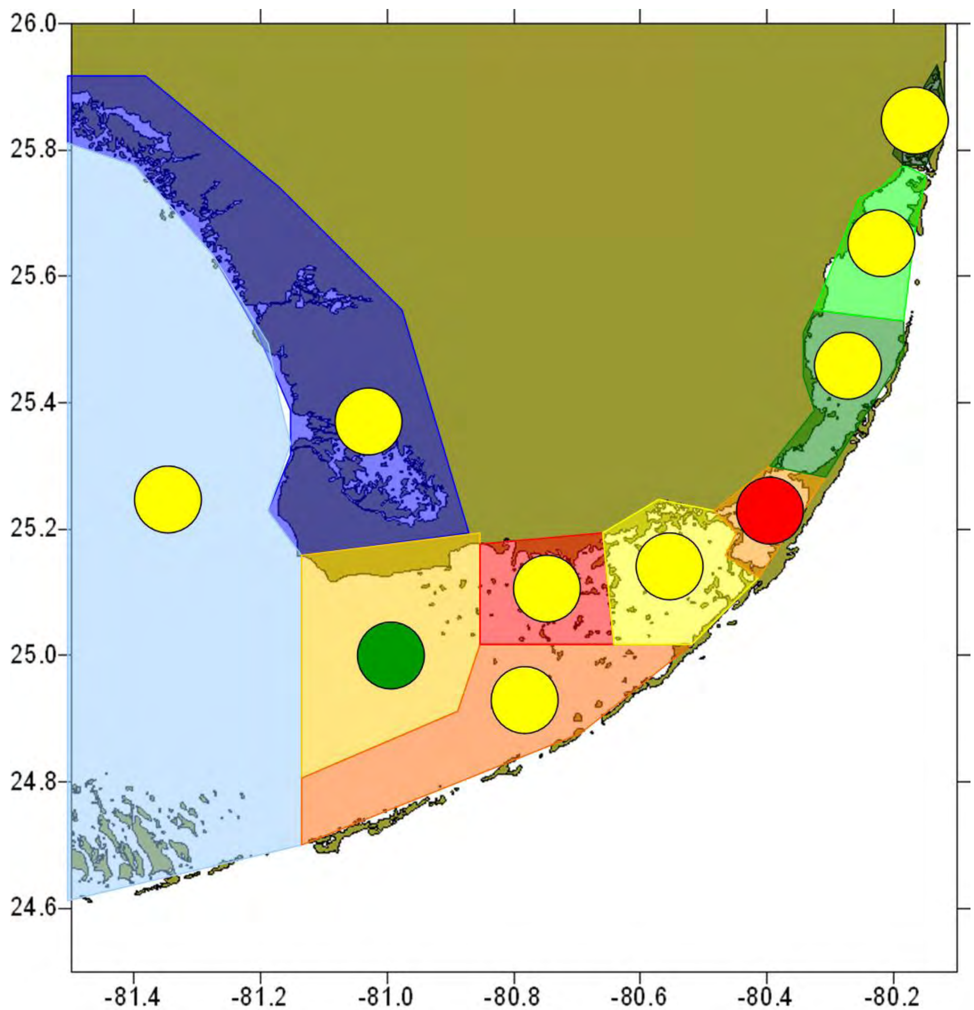

Figure WQ-8: The circle in each sub-region displays the current status of chlorophyll *a*.

The 2006 analysis showed that of the ten sub-regions 1 was green, 8 were yellow, and 1 was red (Fig. WQ-8). Two sub-regions, the MTZ and BMB, had the highest median chlorophyll *a* concentrations of any year on record. Thus, the 8 yellow sub-regions may warrant further investigation and the one red sub-region must undergo further investigation. The red sub-region incorporates Blackwater, Manatee, and Barnes Sounds and the entire 95% confidence interval of the median is located in the red region of the graph, indicating there was a substantial increase in chlorophyll *a* in this sub-region in 2006. This is an area that has been subject to significant disturbances unrelated to CERP over the past two years. In April of 2005 a road construction project began to expand the US Highway 1 in this region. This involved a significant amount of cutting and mulching of mangroves and soil tilling. Also, from August to October 2005 this area was affected by the passing of three hurricanes over the region. In addition to causing a great deal of physical disturbance, there was a large managed release of water that contained elevated levels of phosphorous prior to the first hurricane.

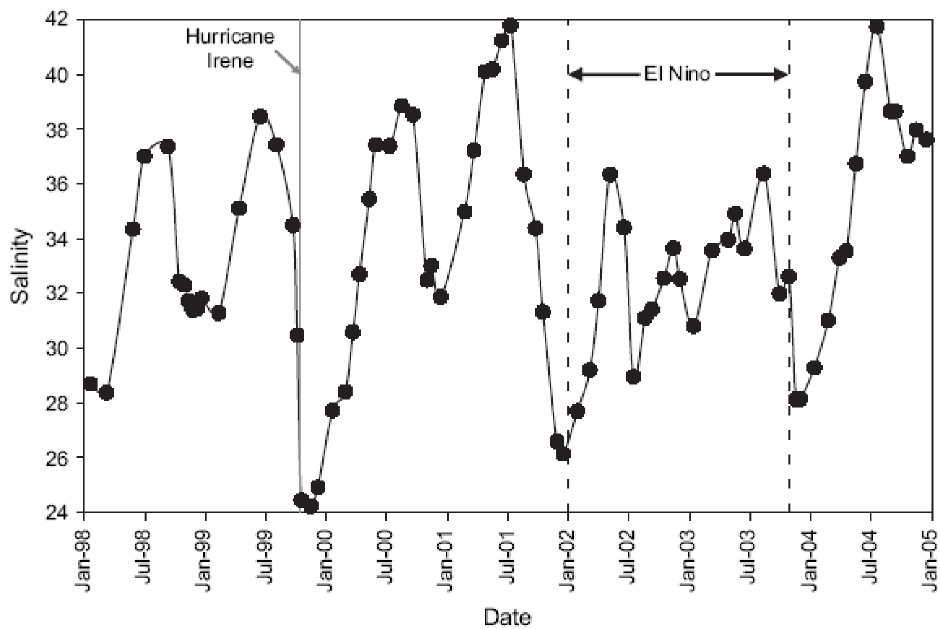

Figure WQ-9: The mean bay-wide salinity of Florida Bay depicts significant deviations due to climactic variation and tropical cyclones.

The result of these activities was the initiation of an atypical algal bloom in this sub-region shortly after October of 2005. Levels of chlorophyll *a* far exceeded previously measured values in this sub-region. Furthermore, the long residence times of this sub-region acted to maintain the bloom's location and helped the bloom to persist throughout 2006. The minimal flushing did not dilute the bloom and its persistence is likely due to the creation of a positive feedback loop, whereby the bloom shades the seagrasses which senesce and decay releasing nutrients and destabilizing the bottom which increases sediment and nutrient resuspension further fueling the bloom. Monitoring results indicate that the bloom was likely initiated by a large increase in total phosphorous prior to the bloom's initiation and total phosphorous has remained elevated throughout the bloom's persistence indicating its importance in fueling the bloom (Fig.

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WQ-10). The bloom is spatially associated with the road construction activities and temporally associated with the impacts of hurricanes. Thus, it is likely that the bloom was the result of these two events occurring coincidentally in the fall of 2005. For more information on this phenomenon and its underlying causes please refer to Rudnick et al. (2007).

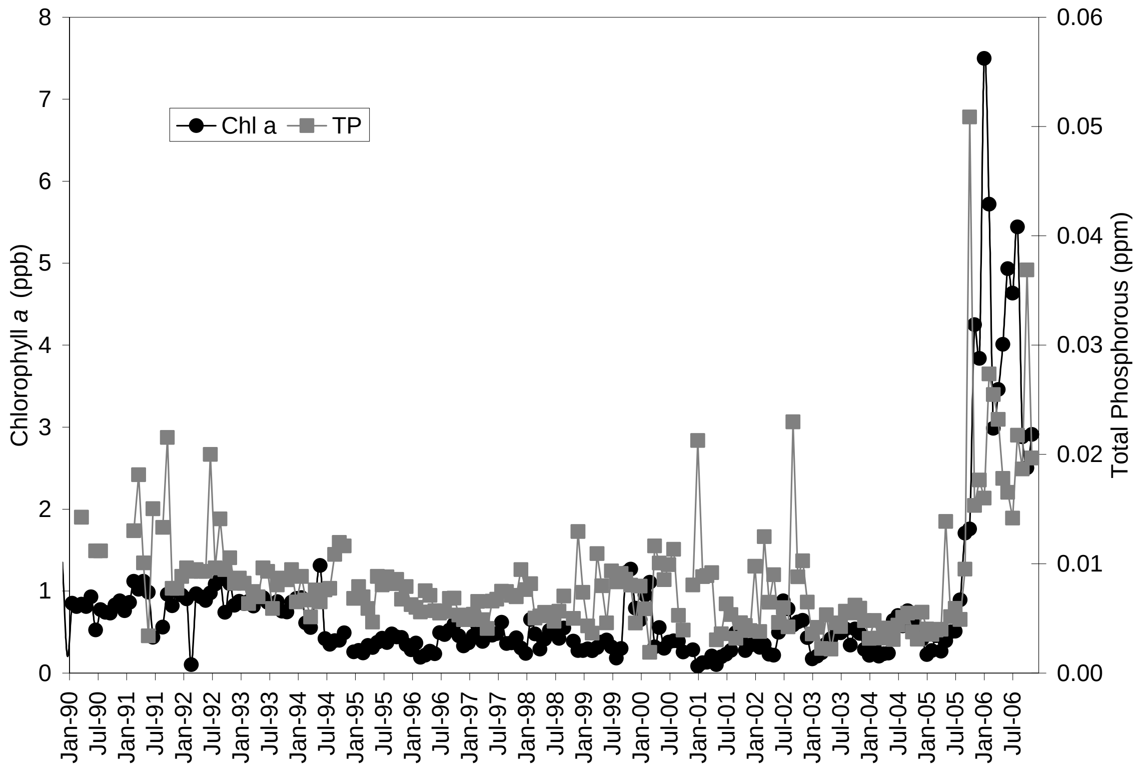

Figure WQ-10: Time series of median chlorophyll and total phosphorous in the BMB sub-region.

WQ.6 Interim Goals

The desired condition is sustained good water quality in Florida Bay, minimizing the magnitude, duration, and spatial extent of algal blooms in the bay such that light penetration is sufficient to sustain healthy and productive seagrass habitat. The interim goal for Florida Bay algal blooms is to prevent any increase in the intensity, duration, or spatial extent of such blooms in Florida Bay or adjacent waters. The proposed assessment along with current monitoring projects is capable of addressing this interim goal in all of the ten sub-regions with the possible exception of the southwest Florida shelf where sampling frequency may not be adequate. The current assessment shows that there has been an increase in algal blooms in one sub-region (Blackwater, Manatee, and Barnes Sounds); however, this increase was not due to CERP, and instead was the result of a combination of hurricanes, managed water releases, and road construction in this sub-region in fall 2005.

The ability to predict water quality and chlorophyll *a* response to CERP is dependent upon the further refinement of the Environmental Fluid Dynamics Code

Model which is being developed as a task of CERP's Florida Bay and Florida Keys Feasibility Study. This model will be used to predict the intensity, duration, and spatial distribution of algal blooms in Florida Bay and the nearshore southwest Florida shelf as CERP is implemented. A similar model may be required for Biscayne Bay. The current monitoring and assessment plans are adequate, except for on the southwest Florida shelf, to detect changes to the intensity, duration, and spatial distribution of algal blooms and assess the accuracy of the model.

WQ.7 Lessons Learned

This approach to assessing water quality has proven to be quite capable of detecting changes as it did in the Blackwater, Manatee, and Barnes Sounds sub-region for 2006. There is precedence for the criteria development and the graphical representations can be easily understood by all audiences. The one weakness is with respect to sampling frequency. It has been recommended by the Advisory Committee on Water Information and the National Water Quality Monitoring Council that water quality be measured monthly to assess the condition of specific estuaries (ACWI and NWQMC 2006). Currently the sampling frequency is not sufficient on the southwest Florida shelf, where sampling is conducted quarterly by both programs. Increasing the sampling frequency in this sub-region is of heightened importance, because CERP is likely to significantly increase freshwater discharge in this sub-region. Furthermore, it is recommended that NOAA/AOML increase its sampling frequency to conduct monthly surveys. This will enable the utilization of their flow-through chlorophyll *a* measurements in further analysis, which would substantially increase the spatial coverage of the assessment.

It is recommended that the salinity section be partitioned out into its own section with a distinct hypothesis cluster and relevant hypotheses. There are specific interim goals for salinity separate from water quality, indicating that it has on occasion been treated as a separate cluster and it should uniformly be treated as such in the future. The desired condition is to reduce the intensity, frequency, duration, and spatial extent of high salinity events, reestablish common mesohaline to oligohaline conditions in mainland nearshore zones, and reduce the frequency and rapidity of salinity fluctuations resulting from pulse releases of fresh water from canals. By altering freshwater flow, CERP will almost certainly affect salinity distributions in the southern estuaries which will in turn result in changes to water quality and all other performance measures. Thus, it is logical and necessary to have a separate salinity hypothesis cluster and performance measure which is assessed annually to ensure we are effectively monitoring this variable and capable of detecting changes which may occur as a result of CERP.

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The Crocodilian Indicator in the Greater Everglades

2006 Assessment Report

UNIVERSITY of FLORIDA

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AMERICAN ALLIGATOR AND CROCODILE

LOCATION	LAST STATUS [a]	CURRENT STATUS [b]	2-YEAR PROSPECTS [c]	CURRENT STATUS [b]	2-YEAR PROSPECTS [c]
American Alligator					
A.R.M. Loxahatchee National Wildlife Refuge				Relative density (component score = 0.83) and body condition (component score = 0.17) combined for a location score of 0.5 and so current conditions do not meet restoration criteria, signifying that this area needs further attention.	A.R.M. Loxahatchee National Wildlife Refuge and management objectives play an important part in determining success here. If conditions remain constant, prognosis for the future will be stable.
Water Conservation Area 2A				Relative density (component score = 0.17) and body condition (component score = 0.5) combined for a location score of 0.34 and so current conditions are below restoration criteria.	With the stable body condition and low relative density of alligators observed here, status will remain substantially below restoration objectives.
Water Conservation Area 3A				Relative density in two of the three locations within WCA 3A is low (northern and southern areas) and higher (yellow) in the central area; body condition scores yellow in the north and central areas, and red in the south. The combined score of both components for the overall area is 0.31, which is well below restoration goals.	This is the only area in which status declined between 2005 and 2006. With the central area of WCA 3A having the highest status (yellow), it can be used a guide for raising the northern and southern areas (both currently red).
Water Conservation Area 3B				Relative density (component score = 0.17) and body condition (component score = 0.5) combined for a location score of 0.34 and so current conditions are below restoration criteria.	With the stable body condition and low relative density of alligators observed here, status will remain substantially below restoration objectives.
Everglades National Park				Relative density in all three locations within Everglades National Park is low. Body condition is higher (yellow) in Shark Slough and estuarine areas, but low (red) in northeast Shark Slough. The combined score of these two components for the overall area, and alligator hole occupancy in the inaccessible areas, is 0.35, which is well below restoration goals.	Everglades National Park management objectives will play a direct role in determining success here. If conditions remain as they currently are, restoration goals will not be met.
Big Cypress National Preserve	insufficient data			Relative density (component score = 0.17) and body condition (component score = 0.5) combined for a location score of 0.34 and so current conditions are below restoration criteria.	Only one year of relative density data has been collected, and body condition has been stable since surveys began in 2004. It is expected that If conditions remain constant, status will remain below restoration objectives.
American Crocodile					
Everglades National Park				Juvenile growth (component score = 0.67) and survival (component score = 0.5) combined for a location score of 0.59 and so current conditions do not meet restoration criteria.	Everglades National Park management objectives will play a direct role in determining success here. If conditions remain constant, prognosis for the future will be stable.
Biscayne Bay Complex				Juvenile growth (component score=0.67) does not meet restoration criteria. There currently is not enough data to calculate a survival component for this area.	Management objectives play an important part in determining success here. If conditions remain constant for growth, prognosis for the future will be stable for this component. Data on survival needs to be collected and figured into the equation.

[a] Data in the Last Status column reflect data prior to calendar year 2006.
[b] Data in the Current Status column reflect data inclusive of calendar year 2006.
[c] The 2-Year Prospect forecast assumes that no large scale hydrological restoration projects are implemented during this time period which would result in significant ecological response of this indicator. The occurrence of significant climatological events during this period may affect the forecast.

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KEY FINDINGS – AMERICAN ALLIGATOR AND CROCODILE

SUMMARY FINDING: On the whole, alligator and crocodile status remained constant during 2006, with only one area (Water Conservation 3A) showing a decline in status compared to previous years. However, the majority of locations show substantial deviations from restoration targets. Status of alligators and crocodiles are expected to improve if hydrologic conditions are restored to more natural patterns.

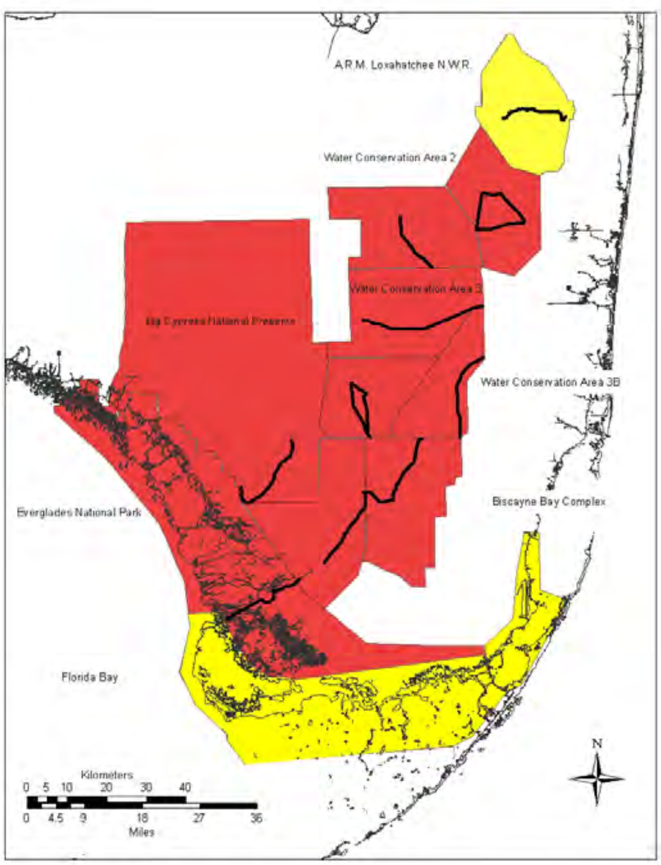

Figure 1. Map of Greater Everglades regions with stoplight ratings by region.

KEY FINDINGS:

1. Alligator overall status at the A.R.M. Loxahatchee National Wildlife Refuge is the highest in South Florida and remains stable.
2. Overall status of alligators throughout the Water Conservation Areas is substantially below restoration targets and requires action in order to meet restoration goals.
3. While body condition of alligators is higher in the southern portion of Everglades National Park (ENP) than in other areas, overall status of alligators throughout ENP is below restoration targets and requires action in order to meet restoration goals.
4. Growth and survival components for crocodiles, while below restoration targets, appear stable at this time and are expected to increase given proper hydrologic conditions through restoration.
5. Restoration of patterns of depth and period of inundation and water flow is essential to improving performance of alligators in interior freshwater wetlands.
6. Restoration of patterns of freshwater flow to estuaries will improve conditions for alligators and crocodiles.
7. Continued monitoring of alligators and crocodiles will provide an indication of ecological responses to ecosystem restoration.

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THE CROCODILIAN INDICATOR IN THE GREATER EVERGLADES

2006 ASSESSMENT REPORT

Frank J. Mazzotti, Rebecca G. Harvey, Kenneth G. Rice, Michael S. Cherkiss, and Brian M. Jeffery

Introduction

Crocodilians (alligators and crocodiles) are the charismatic megafauna of the Everglades. They capture the public's attention and also play central roles in three aspects of Everglades ecology:

1) Alligators and crocodiles are critical in the food web as top predators, influencing abundance and composition of prey (Mazzotti and Brandt 1994).
2) Alligators are ecosystem engineers that create conditions that provide habitat for plants and animals, thereby increasing diversity and productivity of Everglades marshes (Campbell and Mazzotti 2004).
3) Distribution and abundance of crocodilians in estuaries are directly dependent on and immediately responsive to timing, amount, and location of freshwater flow (Dunson and Mazzotti 1989).

Because of these key ecological relationships, monitoring alligators and crocodiles can indicate the overall health of Everglades environments. Status of crocodilian populations relative to hydrologic changes can represent positive or negative trends in restoration.

A system-wide monitoring and assessment plan (MAP) has been developed that describes the monitoring necessary to track ecological responses to Everglades restoration (U.S. Army Corps of Engineers 2004). Included in the MAP are descriptions of selected indicators, how those indicators are linked to key aspects of restoration, and performance measures (monitoring parameters) that are representative of the natural and human systems found in South Florida. The MAP identified crocodilians as one of the indicators, and established the performance measures described in this report.

American alligators (*Alligator mississippiensis*)
Photo: Mike Rochford, University of Florida

American crocodile (*Crocodylus acutus*)
Photo: Wellington Guzman, University of Florida

Crocodilians in South Florida

American Alligator

The American Alligator (*Alligator mississippiensis*) once occupied all wetland habitats in South Florida, from sinkholes and ponds in pinelands to mangrove estuaries during periods of freshwater discharge (Craighead 1968). Alligators are a *keystone species* in the Everglades, meaning they affect nearly all aquatic life in the ecosystem in some way. As top predators, alligators consume a wide variety of prey. They also create trails and holes that provide aquatic refugia for other species during the dry season, and nests that provide elevated areas for turtles, snakes, and plants that are less tolerant of flooding (Enge et al. 2000).

As a result of land development and water management practices in South Florida, alligators are now less numerous than they were historically in prairies, Rocky Glades, and mangrove fringe areas. Canal construction has further altered alligator habitat: unlike alligator holes, canals are not suitable for small alligators, small marsh fish, or foraging wading birds. Restoration of pre-canal hydropatterns and ecological function in the Everglades is underway as part of the Comprehensive Everglades Restoration Plan (CERP, U.S. Army Corps of Engineers 1999). Because of the alligator's ecological importance and sensitivity to hydrology, salinity, habitat, and total system productivity, the species was chosen as an indicator for restoration assessment. The relative density of alligators is expected to increase as hydrologic conditions improve in over-drained marshes and freshwater tributaries. As canals are removed, alligator density in adjacent marshes and use of alligator holes are expected to increase. As hydroperiods and depths approach natural patterns, alligator growth, body condition, and hole occupancy should improve.

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American Crocodile

The American crocodile (*Crocodylus acutus*) is a primarily coastal crocodilian that occurs in parts of Mexico, Central and South America, the Caribbean, and, at the northern extent of its range, in South Florida. This species thrives in healthy estuarine environments and is particularly dependent on natural freshwater deliveries. Habitat loss, due to development supporting a rapidly growing human population in coastal areas, has been the primary factor endangering the crocodile in Florida. Loss of habitat restricted nesting to a small area of northeastern Florida Bay and northern Key Largo by the early 1970s (Kushlan and Mazzotti 1989). After crocodiles were declared endangered in 1975, a crocodile sanctuary in northeastern Florida Bay was established, Crocodile Lake National Wildlife Refuge was created on Key Largo, and Florida Power and Light Company began a long-term management and monitoring program.

Crocodiles are a *flagship species* for southern estuaries, meaning they represent the ecological importance of restoring freshwater flow. Survival of crocodiles has been linked to regional hydrologic conditions, especially rainfall, water level, and salinity (Dunson and Mazzotti 1989) . Alternatives for improving water delivery into South Florida estuaries may change salinities, water levels, and availability of nesting habitat. It is expected that restoration of freshwater flows and salinity regimes will improve conditions for crocodiles. Nesting, growth, and survival of crocodiles can be used to evaluate restoration alternatives and establish criteria for successful restoration efforts in Florida and Biscayne Bay. Crocodiles can also indicate the impacts of freshwater diversion due to coastal development in Miami-Dade, Collier, and Lee Counties.

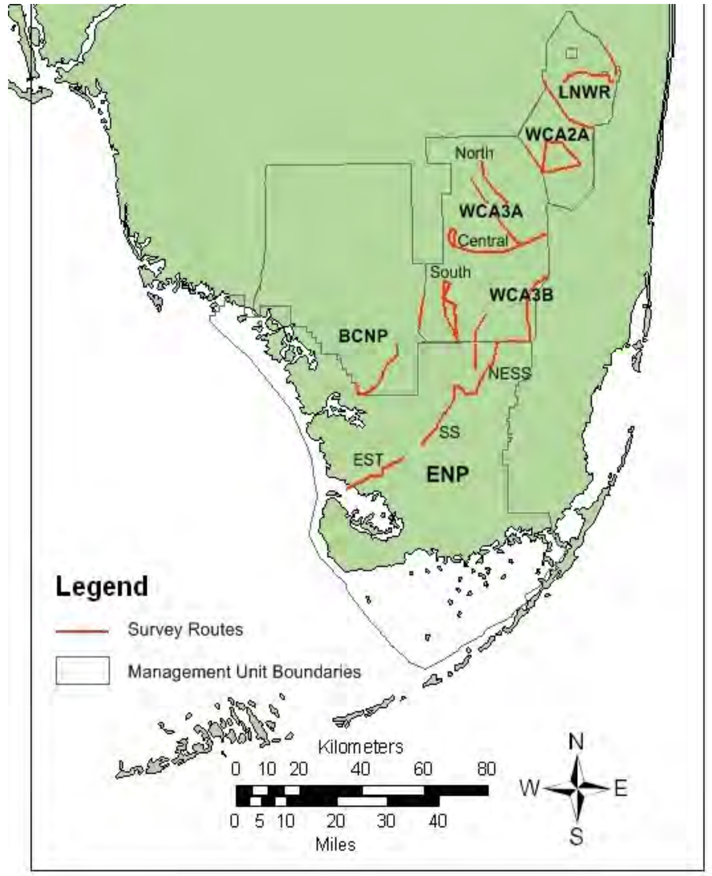

Figure 1. American alligator spotlight survey routes in South Florida, 1999-2006. LNWR = A.R.M. Loxahatchee National Wildlife Refuge, WCA = Water Conservation Area, ENP = Everglades National Park, NESS = Northeast Shark Slough, SS = Shark Slough, EST = Estuarine, BCNP = Big Cypress National Preserve. Source: University of Florida

Study Areas

Alligator monitoring was performed in six management units (two of which were divided into subunits) (Figure 1). Alligator hole occupancy monitoring was only performed in ENP-IA; relative density and body condition were monitored in all other areas.

- Arthur R. Marshall Loxahatchee National Wildlife Refuge (LNWR)
- Water Conservation Area 2A (WCA 2A)
- Water Conservation Area 3A – *three subunits*:
  - North (WCA 3A-North)
  - Central (WCA 3A-Central)
  - South (WCA 3A-South)
- Water Conservation Area 3B (WCA 3B)
- Everglades National Park – *four subunits*:
  - Northeast Shark Slough (ENP-NESS)
  - Shark Slough (ENP-SS)
  - Estuarine (ENP-EST)
  - Inaccessible Areas (ENP-IA; includes areas in Rocky Glades/Southern Marl Prairies and Northeast Shark Slough)
- Big Cypress National Preserve (BCNP)

Crocodile monitoring was performed in two management units (Figure 2):

- Everglades National Park Complex (ENP)
- Biscayne Bay Complex (BBC)

Surveying American alligators by airboat
Photo: Mike Rochford, University of Florida

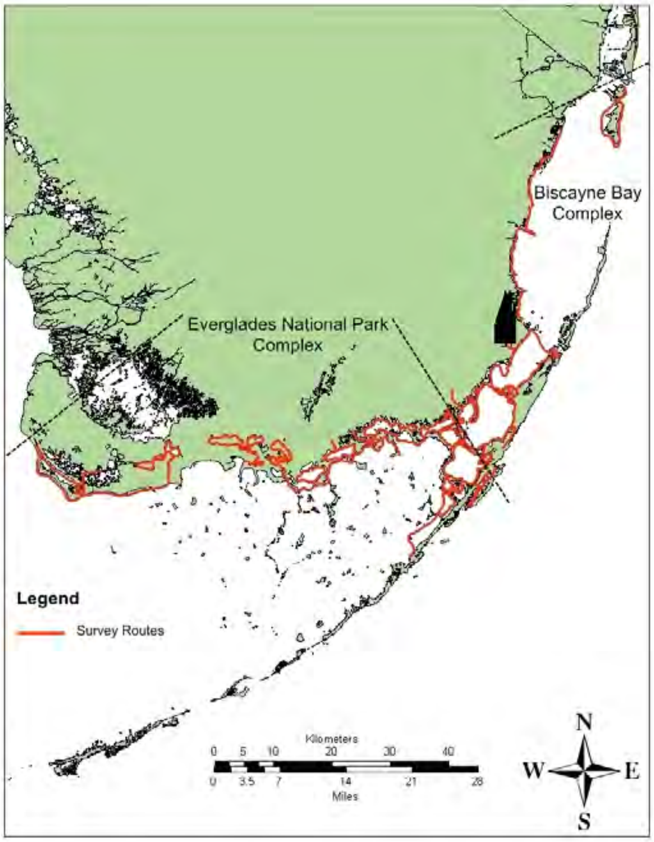

Figure 2. American crocodile spotlight survey routes in South Florida, 2006. Source: University of Florida

Stoplight Restoration Report Card

The stoplight restoration report card translates results for each performance measure into a suitability index representing progress toward meeting restoration targets. For most crocodilian performance measures, targets were established using empirical data from reference sites in the Everglades, except occupancy rate of alligator holes for which the upper target was based on historical information. Targets are presented in the *Methods* sections for each performance measure, below (also see Mazzotti et al. in press, Table 1).

There are generally three components for each performance measure: current status (results from 2006 survey year), the five-year or three-year running average (depending on expected power to detect changes), and the most recent trend (positive, negative, or stable). Alligator hole occupancy, however, has only been monitored since 2005 and thus has only one component (current year percent occupancy).

For each performance measure, the value of each component was compared to the target values to yield a suitability index score (0, 0.5, or 1) with a corresponding color for an easily interpreted "stoplight display:" a value of 0 = red = substantial deviation from restoration targets, 0.5 = yellow = targets have not been reached, and 1 = green = targets have been reached. The most recent trend was determined by regression analyses of data through 2005, as described in each *Methods* section below; stoplight scores were set as 0 = negative trend, 0.5 = no trend, and 1 = positive trend.

Suitability index scores were calculated for each performance measure as the arithmetic mean of the components of the performance measure. Next, a management unit suitability index score was calculated as the arithmetic mean of the performance measures in the given management unit. Calculated index scores were translated to stoplight colors as follows: $0 \leq$ score ≤ 0.4 = red, $0.4 <$ score ≤ 0.8 = yellow, and $0.8 <$ score ≤ 1 = green. A system-wide score was generated for alligators as the geometric mean of all six management unit scores, and a system-wide score for crocodiles was calculated as the geometric mean of the two management unit scores. Finally, a Crocodilian Index Final Score was calculated as the geometric mean of the system-wide alligator and crocodile scores (Appendix 1).

Performance Measures

The stoplight restoration report card includes three performance measures for alligators and two performance measures for crocodiles.

Alligator Performance Measures

- Relative density (number of non-hatchling alligators per kilometer)
- Body condition (length/volume ratio, calculated by Fulton's K)
- Alligator hole occupancy (percent occupied)

Crocodile Performance Measures

- Juvenile growth (centimeters per day total length for crocodiles < 0.75m)
- Hatchling survival (percent monthly fall survival)

These performance measures are hypothesized to be affected by changing hydrologic conditions (depth, duration, timing, spatial extent, water quality, and salinity) (U.S. Army Corps of Engineers, 2004). For crocodiles, nesting effort and success are also important indicators of the status of the population. Although nesting is not yet included in the performance measures for the stoplight score card, we include a discussion of crocodile nesting results (1978-2006) in this report.

American crocodile hatchlings
Photo: Mike Rochford, University of Florida

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American Alligator Monitoring

Alligator Relative Density

Methods

Alligators were counted via spotlight surveys along routes in six management units (Figure 1), following guidelines in the Alligator Survey Network Spotlight Survey Protocol (Rice and Mazzotti 2007, Appendix 1). This report presents results from estuarine transects in ENP-EST and marsh transects in all other management units; surveys in canals were also conducted and are reported elsewhere (Rice and Mazzotti 2007). Surveys were conducted twice in each area in both spring and fall, at least 14 days apart to achieve independent counts (Wood et al. 1985). Alligator locations were recorded using global positioning systems (GPS). Body lengths were estimated in quarter-meter increments, and alligators were placed into the following categories: hatchling (< 0.25 m), juvenile (0.25-1.24 m), subadult (1.25-1.74 m), and adult (≥ 1.75 m). Relative density was calculated by dividing the total number of non-hatchling animals encountered on each survey by the total length (in kilometers) of the survey route.

Three components were used to calculate the stoplight score for relative density: current year status, five-year running mean, and most recent trend. The current status component was defined as mean non-hatchling alligators per kilometer during the spring 2006 survey. Preliminary power analyses demonstrated that we can detect a 5% change in relative density over a five-year period (Rice and Mazzotti 2006). If five years of data were not available, the three-year or four-year mean was used. In BCNP, only one year of data was available because relative density was monitored there for the first time in 2006.

Targets for relative density were developed based on the distribution of relative densities from all spring night surveys conducted on Everglades marsh transects from 1999-2006 (individual replicates of 10 areas over four to eight years; Rice and Mazzotti 2006). This distribution was divided into quartiles; stoplight scores were set as 0=first and second quartiles (density ≤ 1.47 animals/km), 0.5=third quartile (1.47 < density ≤ 2.70 animals/km), and 1=fourth quartile (density > 2.70 animals/km).

Trends in count densities were assessed through 2005 in each management unit. Trends were assessed by loglinear regression of counts of alligators on elapsed time (year) and the quadratic (year + year^2) where appropriate, with mean measured water depth as a covariate.

Results

The average relative density (mean non-hatchling animals per km in spring survey) was much higher in LNWR (6.57, fourth quartile) than in any of the other management units. Density was 2.07 (third quartile) in WCA 3A-Central, and less than 1.47 animals per kilometer (first-second quartiles) in all other areas. The lowest densities were in WCA 3B (0.21) and ENP-SS (0.68) (Figure 3). The five-year running mean followed a similar pattern, with 5.63 animals per km in LNWR (fourth quartile), 2.05 in WCA 3A-Central (third quartile), and values in the first-second quartiles in all other areas. The lowest mean relative density (0.42) was in WCA 3B (Figure 3).

Decreasing trends in total alligator populations were detected in two management units: WCA 3A-North (-0.56 animals/km/year) and ENP-EST (-0.64 animals/km/year). In addition (although not included in the performance measure), decreasing trends in juvenile populations were detected in WCA 2A, WCA 3A-North, and ENP-SS, and a decreasing trend in the adult population was detected in WCA 3A-North. An increasing trend was found for the adult population in LNWR. There was either no trend or insufficient data to detect a trend in all other areas.

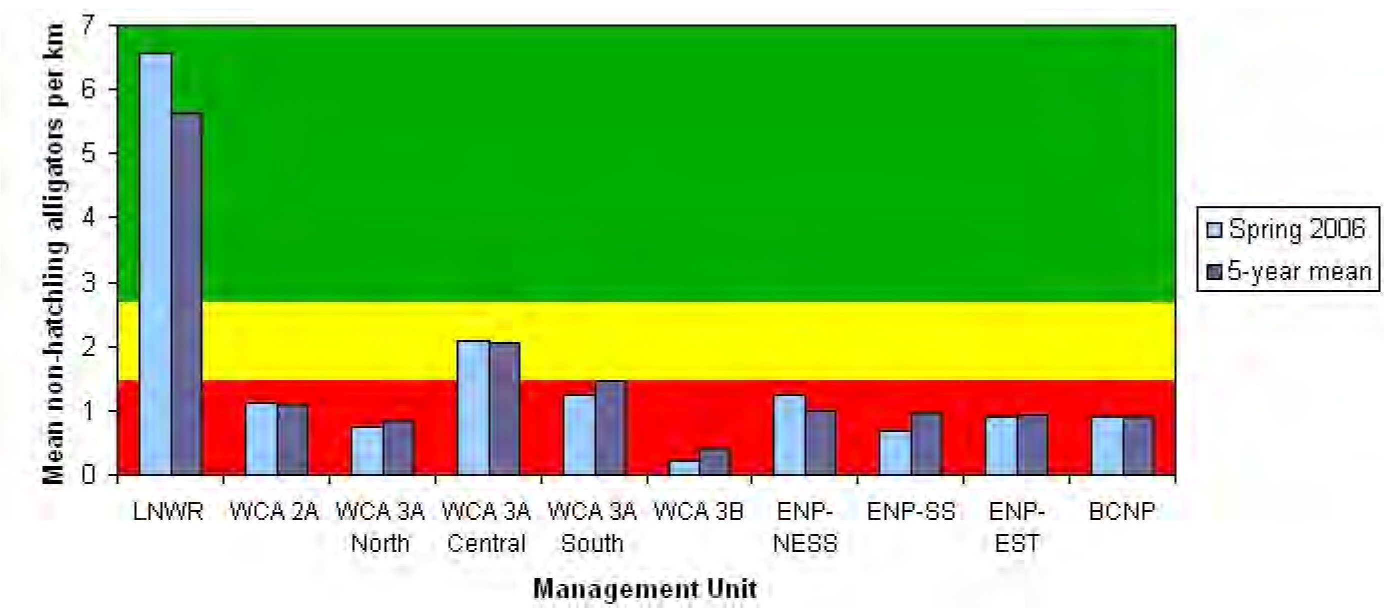

Figure 3. Mean relative density of alligators in Greater Everglades. LNWR = A.R.M. Loxahatchee National Wildlife Refuge, WCA = Water Conservation Area, ENP = Everglades National Park, NESS = Northeast Shark Slough, SS = Shark Slough, EST = Estuarine, BCNP = Big Cypress National Preserve. The background shading refers to the stoplight scores: red = substantial deviation from restoration targets, yellow = targets have not been reached, green = targets have been reached.
Source: University of Florida

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Alligator Body Condition

Body condition (a ratio of body length to body volume) is of interest to researchers because of its potential for assessing how crocodilians are "coping" with their environment (Brandt 1991). Body condition can provide a measure of ecosystem condition and a measure of the quality and accessibility of prey species.

Alligator capture to monitor body condition
Photo: Mike Rochford, University of Florida

Methods

To determine condition of alligator populations, semi-annual capture surveys were performed in the same areas as described for spotlight surveys (Figure 1). A minimum of 15 alligators greater than 1 meter total length were captured by hand, noose or tongs in the fall and spring of each year. Total length (TL), snout-vent length (SVL), head length (HL), tail girth (TG), and weight were measured, sex determined, and any abnormalities noted. To identify recaptures, alligators were marked using Florida Fish and Wildlife Conservation Commission web tags or by clipping scutes (the ridges on alligators' tails). Geographic location, habitat characteristics, and environmental characteristics (air/water temperature, water depth, muck depth, and salinity) were recorded where applicable.

Calculating body condition requires a body length indicator and a volumetric measurement. Head length (HL), snout-vent length (SVL) and total length (TL) are suitable for body length indicators; tail girth (TG), neck girth (NG), chest girth (CG), and weight can all be used as volumetric measurements. In this study, we used a condition factor analysis (Fulton's K; Zweig 2003). Fulton's K uses the ratio of HL/weight and has been evaluated as the best condition index to spatially compare populations of the American alligator (Zweig 2003).

Three components were used to calculate the stoplight score for alligator body condition: current year status, three-year running mean, and most recent trend. The current status indicator was defined as the lowest spring or fall mean condition during the 2006 survey year. A three-year (instead of five-year) running mean was used because expected power should enable trends to be detected in one to three years.

Targets for body condition were developed based on the distribution of body condition (Fulton's K) of all alligators captured and assessed in the Everglades from 1999-2006 (n=1755). This distribution was divided into quartiles; stoplight scores were set as 0=first quartile (Fulton's K ≤ 9.31), 0.5=second and third quartiles (9.31 < Fulton's K ≤ 11.27), and 1=fourth quartile (Fulton's K > 11.27).

Trends in body condition were assessed through 2005 in

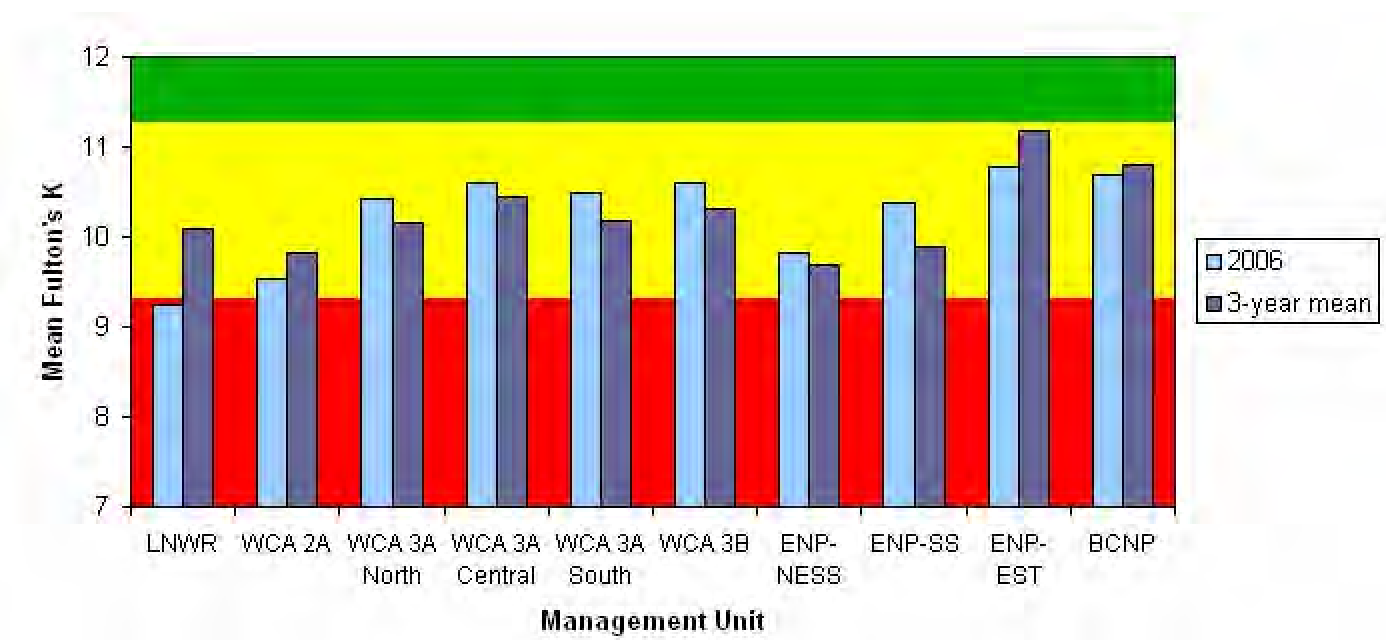

Figure 4. Mean body condition (Fulton's K) of alligators in Greater Everglades. 2006 measure is lowest spring or fall mean. LNWR = A.R.M. Loxahatchee National Wildlife Refuge, WCA = Water Conservation Area, ENP = Everglades National Park, NESS = Northeast Shark Slough, SS = Shark Slough, EST = Estuarine, BCNP = Big Cypress National Preserve. The background shading refers to the stoplight scores: red = substantial deviation from restoration targets, yellow = targets have not been reached, green = targets have been reached. Source: University of Florida

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each management unit by loglinear regression of Fulton's K on elapsed time (year) and the quadratic (year + year^2) where appropriate, with three covariates: season (fall or spring), sex (male or female) and animal length (SVL).

Results

The condition factor of captured alligators (lowest mean spring or fall Fulton's K) was lower (9.25, first quartile) in LNWR than all other management units, where it was in the second-third quartiles; the highest value was 10.77 in ENP-EST. The three-year running mean was in the second-third quartiles in all areas, ranging from 9.70 in ENP-NESS to 11.18 in ENP-EST (Figure 4).

We were able to detect decreasing annual trends in body condition in WCA 3A-South (5.6%), ENP-NESS (3.7%), and LNWR (1.4%). In one area, ENP-EST, we observed an increasing trend of 8% per year. There was either no trend or insufficient data to detect a trend in all other areas.

Females were in better condition than males in four areas ($p_{sex} \leq 0.003$, ENP-SS, LNWR, WCA 3A-Central, WCA 3A-North) but this did not vary between seasons ($p_{sex*season} >$ 0.05). Males were captured more frequently over time in WCA 2A ($p_{sex*year} = 0.0002$). Larger animals were in better condition than smaller animals in five areas ($p_{svl} \leq 0.005$, ENP-SS, LNWR, WCA 2A, WCA 3A-Central, WCA 3A-North). Smaller animals were captured more frequently over time in ENP-SS and WCA 3A-North ($p_{svl*year} \leq 0.09$) and larger animals were captured more frequently over time in WCA 3A-South ($p_{svl*year} < 0.0001$). We observed higher body conditions in spring in ENP-SS, WCA 2A and WCA 3A-North ($p_{season} \leq 0.035$) and in fall in WCA 3B ($p_{sex} = 0.002$).

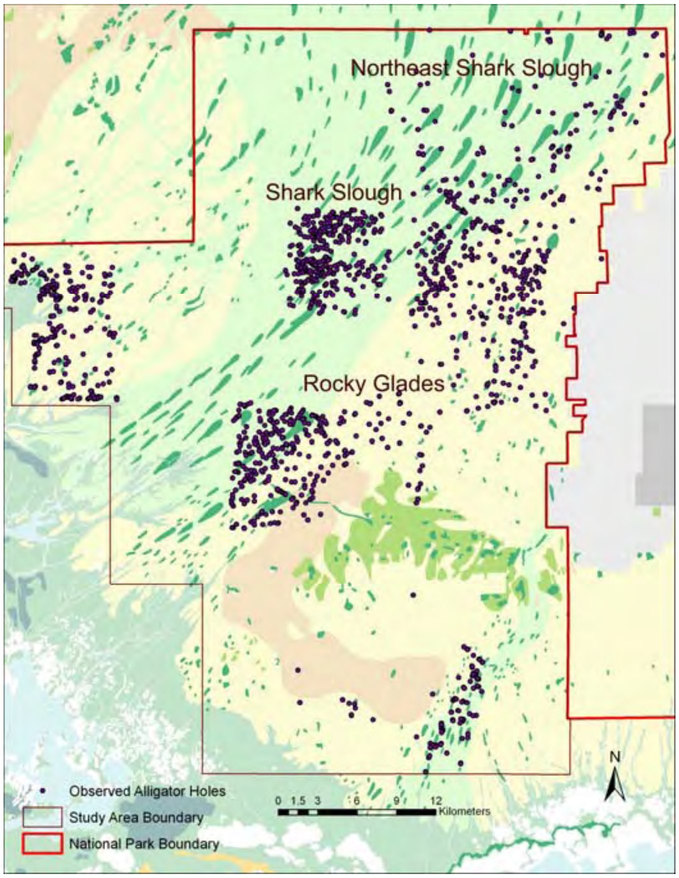

Figure 5. Alligator holes observed in Everglades National Park (ENP) during 2005 and 2006 Standard Reconnaissance Flights (SRFs). (Base-map is Everglades physiographic areas courtesy of ENP.) Source: Rice and Mazzotti, 2007

Alligator Hole Occupancy

Although alligator holes and other dry season refugia have long been recognized as a critical component of the Everglades ecosystem (Craighead 1968, Mazzotti and Brandt 1994), until recently only one alligator hole had been studied in detail (Kushlan 1972). We began to map and characterize alligator holes in parts of the Everglades (Campbell and Mazzotti 2004); however, there is still a lack of data about alligator holes in Shark Slough and the Rocky Glades.

Alligator hole seen from the air in Everglades National Park
Photo: Wellington Guzman, University of Florida

Methods

Surveys for alligator hole occupancy were conducted via Standard Reconnaissance Flights (SRF) in four areas of ENP during five days in May 2006 (May 3 through May 9, 2006). Transects were flown through areas of the Northeast Everglades that had not been visited during a previous accuracy assessment, as well as in an area of Northeast Shark Slough surveyed in April 2005 and area in Shark Slough surveyed in June 2005. Transects were flown at 500-meter east-west intervals. Observers sat on both sides of the helicopter and it was assumed that an observer could identify an alligator hole up to a distance of 250 meters, thus being able to capture all alligator holes within a given area of flown transects. The helicopter flew at an average height of 150 feet above ground, hovering to 50 feet for closer observations. Transects were flown in both the morning and afternoon. When an alligator hole was detected, the pilot navigated from the transect to the observed hole. At each observed alligator hole, the following information was recorded: presence or absence of alligators, size(s) of observed alligator(s), and presence or absence of water in the hole. A GPS location and a photograph were taken of every alligator hole. Holes were considered occupied if the alligator was in the hole or located within a short distance from the hole (e.g., in a trail or basking next to the hole).

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Crocodile capture to monitor growth
Photo: Mark Parry, University of Florida

A single component was used to calculate the stoplight score for hole occupancy: the current year mean proportion of alligator holes (in ENP-IA only) occupied by at least one alligator. As this component was assessed for the first time in 2005, we used additional sources of data to develop targets for stoplight scores. We combined results from our 2005 survey with a study of alligator holes in WCA 3 by Campbell and Mazzotti (2004) and historical information from Craighead (1968), and set values at 0=low (occupancy ≤ 30%), 0.5=medium occupancy (30% < occupancy ≤ 70%), and 1=high (occupancy > 70%). This component is applicable to areas of Northeast Shark Slough, Rocky Glades, and Southern Marl Prairies. These areas are collectively referred to as Inaccessible Areas (ENP-IA) because they are not accessible by airboat and must be monitored by helicopter.

Results

As a result of both 2005 and 2006 SRFs, a total of 1,495 alligator holes in Everglades National Park have now been observed and verified with a GPS location (Figure 5). In 2006, alligators were observed in a total of 269 holes in a surveyed area of 306 km^2. Occupancy ranged from 30% in Shark Slough alligator holes to 72% in the top right corner of ENP in Northeast Shark Slough. It was determined from the surveys that Northeast Shark Slough contained the lowest density of alligator holes (0.5 holes/km^2) while Shark Slough contained the greatest density of alligator holes (7.0 holes/km^2). Not including Shark Slough (which is not part of the inaccessible areas), alligators were observed in 184 holes in Northeast Shark Slough and the Rocky Glades/Southern Marl Prairies (49.9% of observed alligator holes in those areas). This is the value used in the stoplight assessment for ENP-IA. The two-year running mean (2005-2006) is 50.4%, and there is not yet enough data to detect a trend.

Water level appears to influence occupancy of alligator holes. Northeast Shark Slough and the Rocky Glades both had higher occupancy of alligator holes than central Shark Slough, and both were extremely dry at the time of the surveys. With little water in the surrounding marsh, alligator holes were the only refuge from the sun. These conditions may explain the higher occupancy of alligator holes in these areas. In central Shark Slough, on the other hand, holes still contained water, and water was present in some surrounding marsh habitats. Detectability of alligators was not evaluated in 2006 but will be considered in future surveys, because it was generally more difficult to detect an alligator at a hole with deeper water.

American Crocodile Monitoring

Crocodile Juvenile Growth

Methods

Juvenile growth was determined by periodic efforts throughout 2006 to recapture crocodiles that had been marked in previous captures. Stoplight assessments are based on capture areas in ENP (Buttonwood Canal) and Biscayne Bay Complex (BBC; does not include Florida Power & Light's Turkey Point Plant) (Figure 2). Non-hatchling crocodiles (> 50 cm) were captured by hand, tongs, net, or by wire-noose as described by Mazzotti (1983). All crocodiles were weighed and measured for total length (TL) and snout-vent length (SVL). (Head length, tail girth, hind foot length, mass, and other body measurements were recorded occasionally.) Hatchlings were defined as animals < 50 cm in total body length, juveniles were defined as 50–150 cm, sub-adults were defined as 150-175 cm, and animals greater than 175 cm in total body length were classified as adults.

To assess juvenile growth, we measured growth that occurred during the first year of an animal's life, and therefore only analyzed captures of animals less than or equal to 75 cm total length. We defined average growth rate as change in total length between two capture events divided by the number of days between two capture events. Growth was measured in cm/day over the longest period between captures for animals recaptured at least once.

Three components were used to calculate the stoplight score for juvenile growth: current year average growth rate (cm/day for animals ≤75 cm), three-year running mean, and most recent trend. A three-year (instead of five-year) running mean was used because expected power should enable trends to be detected in one to three years. Targets for juvenile growth were developed based on the distribution of growth rate of all crocodiles captured and measured in Everglades National Park and Biscayne Bay from 1978-2006 (n=498; Mazzotti et al.

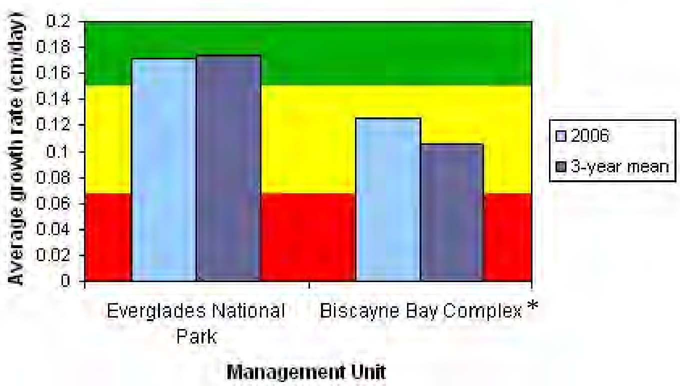

Figure 6. Average growth rate of juvenile (≤75 cm) crocodiles in Greater Everglades. The background shading refers to the stoplight scores: red = substantial deviation from restoration targets, yellow = targets have not been reached, green = targets have been reached. *Growth depicted in this figure does not include hatchlings from the Turkey Point site, for which data were not available. Source: University of Florida

Weighing an American crocodile hatchling
Photo: Mike Rochford, University of Florida

2007). This distribution was divided into quartiles; stoplight scores were set as 0=first quartile (growth ≤ 0.068 cm/day), 0.5=second and third quartiles (0.068 < growth ≤ 0.15 cm/day), and 1=fourth quartile (growth > 0.15 cm/day).

Results

Average growth rate in 2006 was in the fourth quartile (stoplight = 1) in both ENP (0.171 cm/day) and BBC (0.174 cm/day). The three-year running mean was higher in ENP (0.126 cm/day) than in BBC (0.105), and both fell into the second-third quartiles (stoplight = 0.5). The trend stoplight score was 0.5 (no trend) for both management units because there are not yet enough years of data to detect trends (i.e., there is only one three-year running mean, 2004-2006, because data collection started in 2004) (Figure 6).

Crocodile Hatchling Survival

Methods

Hatchling survival was determined by efforts in the fall (August-December, 2006) to recapture hatchling crocodiles (< 50 cm in total body length) that had been captured and marked during the preceding summer. Fall was defined as the critical monitoring period because most hatchlings are born in summer and grow to juvenile size by their first winter. Hatchlings were captured by hand or tongs and marked by removing tail scutes according to a prescribed sequence (Mazzotti 1983). Stoplight assessments are based on capture areas in ENP and BBC (Figure 2), where data on hatchling survival has been collected since 2002.

Three components were used to calculate the stoplight score for hatchling survival: current year survival rate (mean monthly fall survival), five-year running mean, and most recent trend. Targets for hatchling survival were developed by two methods. First, we used the minimum known alive analysis of Mazzotti et al. (2007) to develop a range of possible survival probabilities. Second, we performed multi-state (size class x management unit) capture-recapture survival analyses (Nichols and Kendall 1995) of all captures (n=3981) from 1978-2004 using Program Mark (White and Burnham 1999). The best model of fall hatchling survival included a management unit effect, a period effect (dry years vs. wet years), and a management unit x period interaction. This model had an Akaike weight of 0.96, indicating very strong support (Burnham and Anderson 2002). Targets for stoplight scores were developed by division along the mean estimates of survival from these analyses, with 0 = low survival (<65%), 0.5 = medium survival (65-85%), and 1 = high survival (>85%).

Results

In ENP, mean monthly fall survival in 2006 was 70%, and the five-year running mean was 69%. The trend stoplight score was 0.5 (no trend) because there are not yet enough years of data to detect trends (i.e., there is only one five-year running mean, 2002-2006, because data collection started in 2002). In BBC, no recaptures of hatchlings were made in 2006, so none of the stoplight indicators for hatchling survival could be calculated (Figure 7).

Crocodile Nesting Effort and Success

Nesting is not included in the stoplight performance measures because it responds over a longer time scale than growth and survival (decades vs. years). However, nesting is an important indicator of the status of crocodile populations that has been monitored in South Florida since 1978.

Methods

Monitoring crocodile nests was performed in concert with finding and marking hatchling crocodiles to assess growth and survival. Surveys for nests were conducted from June to August (hatching period), every year from 1978 to 2006. Nests were located from evidence of crocodile activity (tail drags, digging, and scraping); successful nests were determined by presence of one or more hatchlings or hatched shells.

We examined records of crocodiles nesting for numbers, locations, habitat, and fate of nests for the period of 1978-2006. Linear regression models were used for Turkey Point

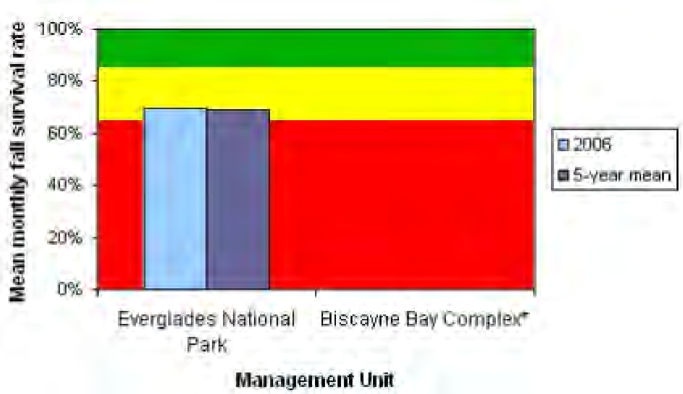

Figure 7. Survival rate (mean monthly fall survival) of hatchling crocodiles in Greater Everglades. The background shading refers to the stoplight scores: red = substantial deviation from restoration targets, yellow = targets have not been reached, green = targets have been reached. *No recaptures of hatchlings in Biscayne Bay Complex in 2006. Source: University of Florida

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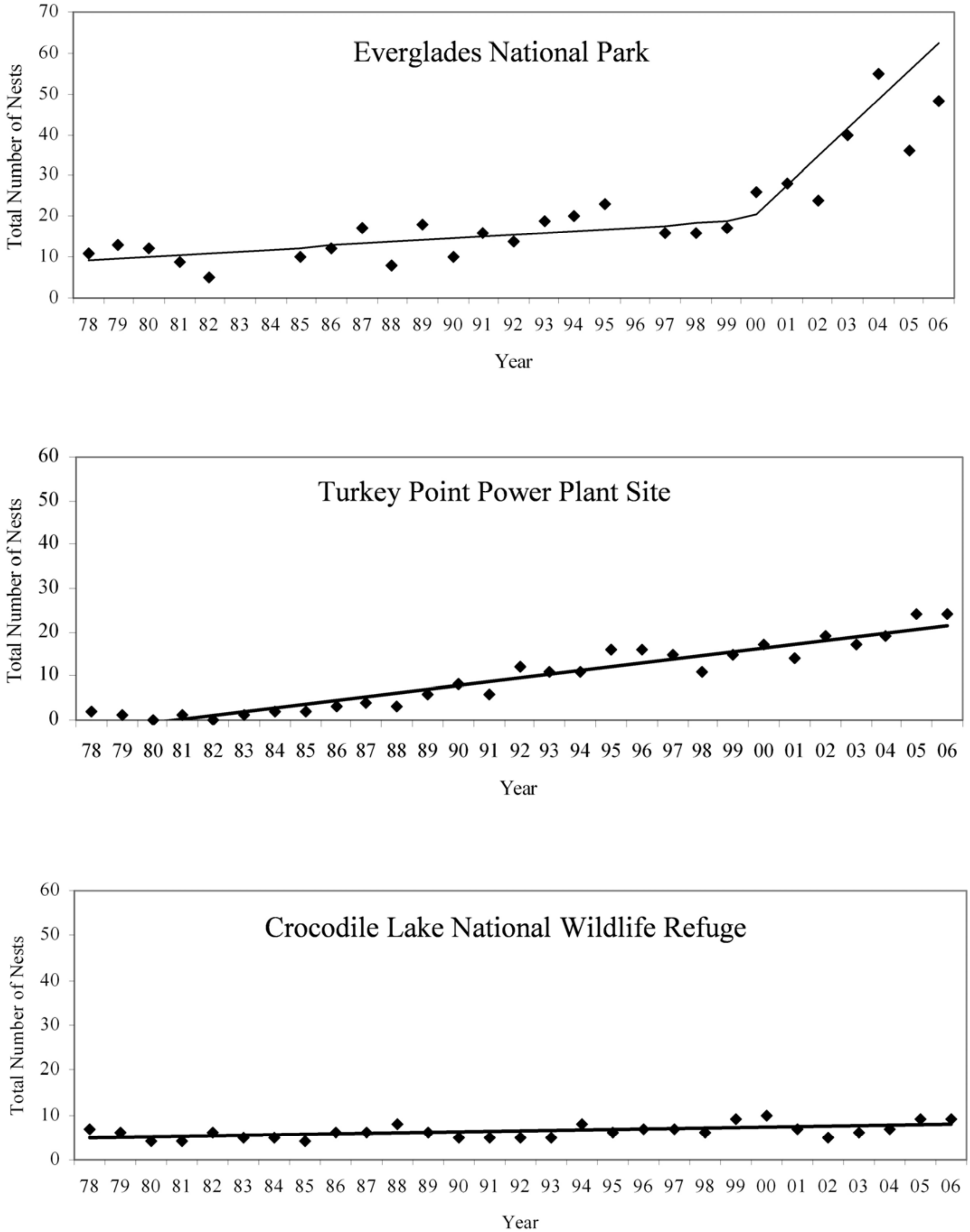

Figure 8. Linear regression for total number of American Crocodile nests found between 1978 and 2006 in the three primary nesting areas (A) Everglades National Park (R2 = 0.6528; p = 0.0001; nests = 523), (B) Turkey Point Power Plant (R2 = 0.920; p = 0.0001; nests = 280) and (C) Crocodile Lake National Wildlife Refuge (R2 = 0.315; p = 0.0015; nests = 183). Source: Rice and Mazzotti, 2007

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(TP) and Crocodile Lake National Wildlife Refuge (CLNWR) nest data. The Gauss-Newton non-linear regression model was employed for ENP.

Results

Fifty-three nests were located in 2006, of which 48 were in Everglades National Park, two were in the Keys (Lower Matecumbe Key, just outside of ENP), and three were in the Biscayne Bay Complex. Of the total 53 nests, 34 (64%) were successful, 17 (32%) were depredated by raccoons, and two (4%) failed for unknown reasons. Thirty-one of the 34 successful nests were in ENP, one was in the Biscayne Bay Complex at Ocean Reef on North Key Largo, and both nests on Lower Matecumbe Key were also successful. The 17 depredated clutches were all located within the boundaries of Everglades National Park, and the two that failed for unknown reasons were both in Biscayne Bay Complex: one at Deering Bay and one at Montgomery Gardens. In addition to the above totals, in 2006, 24 nests were located by Florida Power & Light personnel at TP (also in BBC), and nine nests were found by U.S. Fish & Wildlife Service personnel at CLNWR (also in BBC).

Nine hundred eighty-six crocodile nests were located between 1978 and 2006. Five hundred eighty-nine (71 %) were successful. Turkey Point had the highest rate of nest success at 99% (range 91-100%; N = 276). In ENP, 65% of nests were successful (range 36–100 %; N = 523), and at CLNWR 46% of nests (range 0-100%; N = 183). The number of crocodile nests increased at the TP site, where two nests were discovered in 1978 and 24 were observed in 2006 (Figure 8), all on artificial substrates. The number of nests at CLNWR fluctuated between four and 10 (Figure 8). The number of nests also increased in ENP, from 11 in 1978 to 48 in 2006 (Figure 8). Most of the increase in nesting in ENP occurred on Cape Sable. Nests were also found outside of the three primary nesting areas in or near two Miami-Dade County Parks (eight nests, six successful, 1997–2006), a private residence on Lower Matecumbe Key (six nests, five successful, 2002–2006), and a private resort on northern Key Largo (two successful, 2004-2006).

Crocodile nest on the shoreline
Photo: Michael Cherkiss, University of Florida

American crocodile (*Crocodylus acutus*)
Photo: Mike Rochford, University of Florida

Final Stoplight Scores

Stoplight scores for each management unit and subunit were generated as the arithmetic mean of the component scores, and are presented in Appendix 1. The system-wide alligator index score was calculated as the geometric mean of all six management unit scores, and the system-wide crocodile index score was calculated as the geometric mean of the two management unit scores. Finally, the system-wide crocodilian stoplight score was calculated as the geometric mean of the alligator and crocodile index scores.

- System-wide alligator index score = 0.36 (stoplight = red)
- System-wide crocodile index score = 0.63 (stoplight=yellow)
- System-wide crocodilian stoplight score = 0.47 (stoplight = yellow)

The stoplight scores for both species combined are presented by management unit in Figure 9.

Discussion

On the whole, alligator and crocodile status in the Greater Everglades is substantially below restoration targets (Figure 9). Alligator status is highest at the A.R.M. Loxahatchee National Wildlife Refuge, but still below restoration criteria (yellow); throughout the Water Conservation Areas and Everglades National Park, alligator status is well below restoration targets (red). The low relative density and poor body condition (Figures 3 and 4) of alligators in the Everglades is what we expect in hydrologically altered Everglades ecosystems. Our findings confirm earlier observations that alligators are not doing well in the Everglades (Mazzotti and Brandt 1994).

We hypothesize that alligators do better in areas with less extreme human-caused hydrological alterations, such as the central portion of LNWR. This hypothesis would explain the higher status of alligators in LNWR than in other areas of the Everglades, and suggests that restoration of patterns of depth and period of inundation and water flow would improve performance of alligators in interior freshwater wetlands.

Throughout their range, alligators are typically abundant in coastal wetlands (e.g., Rice and Averitt 1999); thus the low abundance of alligators in Everglades estuaries appears exceptional. Earlier accounts described the oligohaline-freshwater portion of estuaries as important alligator habitat (e.g., Craighead 1968). However, our finding of low relative density of alligators in estuaries (Figure 3) confirms that diminished freshwater flow is a major stressor for Everglades alligators. We expect that restoration of patterns of freshwater flow to estuaries will improve conditions for both alligators and crocodiles.

Unlike American alligators, American crocodiles are successful in South Florida in comparison to other portions of their range (Mazzotti et al. 2007). Growth and survival of crocodiles, however, are below restoration targets (yellow) in both Everglades National Park and Biscayne Bay Complex (Figure 9). Diminished rates of crocodile growth and survival have been related to regional hydrologic patterns (Mazzotti et al. 2007, Rice and Mazzotti 2006). These performance measures for crocodiles appear stable at this time and are expected to increase given proper hydrologic conditions through restoration. Moreover, our ability to monitor growth and survival will improve, as 63% of crocodiles captured in 2006 were recaptures (Rice and Mazzotti 2007). However, differences in current monitoring methods employed at Turkey Point limit comparisons with growth and survival within the BBC and between the BBC and ENP.

The high recapture rate demonstrates the effectiveness of current survey techniques at finding and catching crocodiles, and supports the use of growth and survival as performance measures for Everglades restoration. As body condition can be determined from the same morphometric measurements as growth rates, we recommend that condition also be considered as a performance measure of crocodile responses to ecosystem changes.

As Figure 8 shows, crocodile nesting has increased in South Florida since 1978. More nests were found in each area in 2006 than in previous years, except in ENP where the 2006 count of 48 nests fell short of the record of 55 set in 2004. We attribute this temporary drop in number of nests in part to the impact of two hurricanes in 2005.

Mazzotti (1989) defined optimal nesting habitat for American crocodiles as presence of elevated, well-drained nesting substrate adjacent to relatively deep (> 1 meter), low to intermediate salinity (< 20 ppt) water, protected from effects of wind and wave action, and free from human disturbance. Human-made areas along canal banks (berms) at CLNWR, East Cape Canal in ENP, and the cooling canal system at TP provide nearly ideal nesting conditions. In fact, virtually the entire increase in crocodile nesting in South Florida is due to nesting on artificial substrates in the Cape Sable/Flamingo area of ENP, at CLNWR, and at TP. The rapid increase in nesting in Figure 9A corresponds to the plugging of Buttonwood and East Cape canals in Everglades National Park to reduce saltwater intrusion into interior areas of Whitewater Bay and Cape Sable (Mazzotti and Cherkiss 2003). This finding suggests that restoring salinity patterns in estuaries can have a positive effect on crocodile nesting, leading us to recommend that nesting effort and success should be added to growth and survival as crocodile performance measures.

In 2006, we surveyed more than 292 km of airboat trails and canals for alligators and more than 550 kilometers of shoreline for crocodiles and crocodile nests. We observed 359 crocodiles and captured 161, with a recapture rate of 63% that is unprecedented in crocodilian studies. The crocodile monitoring program is effective at detecting impacts of short-term disturbances that may impact population responses to ecosystem restoration. Using a combination of condition, growth, survival, and nesting of crocodiles allows for monitoring response of crocodile populations at different temporal scales.

Since 1999, we have captured more than 1,700 alligators to monitor body condition. Our current survey program has sufficient power to detect a 5% decrease in the alligator population over five years. We continue to improve alligator survey methods through studies of alligator submergence and detection, which will decrease the amount of time required to detect trends. In 2005-2006, we began monitoring alligator hole occupancy, which is proving to be an excellent performance measure in areas inaccessible to ground-based monitoring.

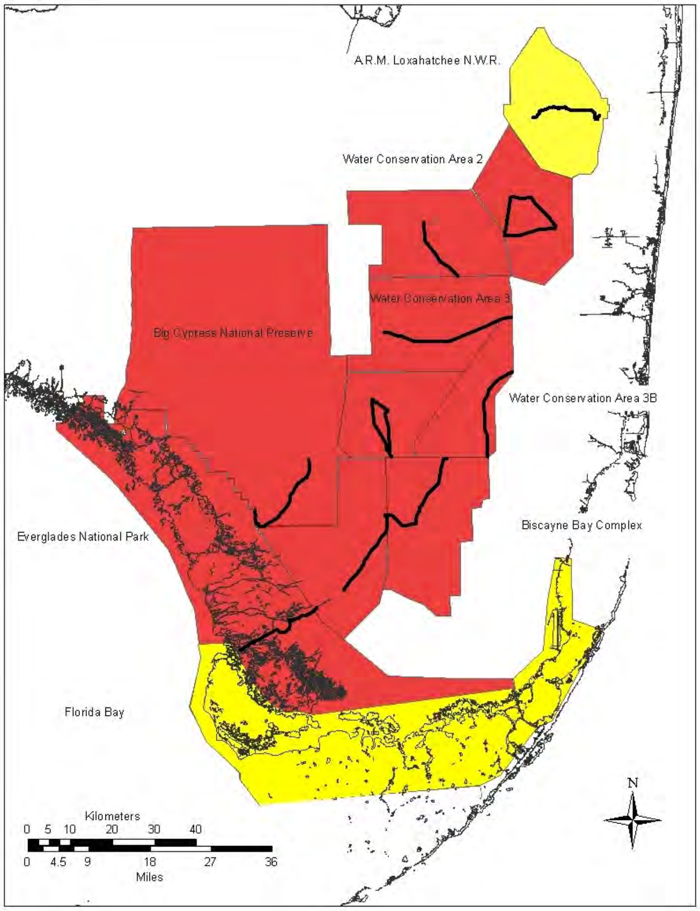

Figure 9. Map of Greater Everglades regions with stoplight ratings by region. Red = substantial deviation from restoration targets, yellow = targets have not been reached. Source: University of Florida

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American alligator hatchlings
Photo: Howard Suzuki, University of Florida

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Appendix 1. 2006 translation of crocodilian performance measures into stoplight display.

Alligators

ARM Loxahatchee National Wildlife Refuge

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	6.57	1		5.63	1		±	0.5		(1+1+0.5)/3=0.83	
Body Condition Fulton's K	9.25	0		10.10	0.5		-	0		(0+0.5+0)/3=0.17	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.83 + 0.17)/2 = 0.5											
Final **ARM Loxahatchee National Wildlife Refuge** Alligator Index Score = 0.5											

Water Conservation Area 2A

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	1.13	0		1.09	0		±	0.5		(0+0+0.5)/3=0.17	
Body Condition Fulton's K	9.53	0.5		9.82	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.17 + 0.5)/2 = 0.34											
Final **Water Conservation Area 2A** Alligator Index Score = 0.34											

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Water Conservation Area 3A North

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	0.75	0		0.85	0		-	0		(0+0+0)/3=0	
Body Condition Fulton's K	10.43	0.5		10.16	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0 + 0.5)/2 = 0.25											
Final **Water Conservation Area 3A North** Alligator Index Score = 0.25											

Water Conservation Area 3A Central

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	2.08	0.5		2.05	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Body Condition Fulton's K	10.59	0.5		10.45	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.5 + 0.5)/2 = 0.5											
Final **Water Conservation Area 3A Central** Alligator Index Score = 0.5											

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Water Conservation Area 3A South

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	1.23	0		1.45	0		±	0.5		(0+0+0.5)/3=0.17	
Body Condition Fulton's K	10.48	0.5		10.17	0.5		-	0		(0.5+0.5+0)/3=0.33	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.17 + 0.33)/2 = 0.25											
Final **Water Conservation Area 3A South** Alligator Index score = 0.25											

Geometric Mean of Water Conservation Area 3A Alligator Index Scores $(0.25 \times 0.5 \times 0.25)^{1/3} = 0.31$	
Final Water Conservation Area 3A Alligator Index score = 0.31	

Water Conservation Area 3B

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	0.21	0		0.42	0		±	0.5		(0+0+0.5)/3=0.17	
Body Condition Fulton's K	10.61	0.5		10.32	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.17 + 0.5)/2 = 0.34											
Final **Water Conservation Area 3B** Alligator Index Score = 0.34											

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Everglades National Park – Northeast Shark Slough

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	1.25	0	🔴	1.00	0	🔴	±	0.5	🟡	(0+0+0.5)/3=0.17	🔴
Body Condition Fulton's K	9.83	0.5	🟡	9.70	0.5	🟡	-	0	🔴	(0.5+0.5+0)/3=0.33	🔴
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.17 + 0.33)/2 = 0.25											
Final **Everglades National Park – Northeast Shark Slough** Alligator Index Score = 0.25										🔴	

Everglades National Park – Shark Slough

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	0.68	0	🔴	0.95	0	🔴	±	0.5	🟡	(0+0+0.5)/3=0.17	🔴
Body Condition Fulton's K	10.37	0.5	🟡	9.89	0.5	🟡	±	0.5	🟡	(0.5+0.5+0.5)/3=0.5	🟡
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.17 + 0.5)/2 = 0.34											
Final **Everglades National Park – Shark Slough** Alligator Index Score = 0.34										🔴	

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Everglades National Park – Estuarine

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	0.90	0		0.92	0		-	0		(0+0+0)/3=0	
Body Condition Fulton's K	10.77	0.5		11.18	0.5		+	1.0		(0.5+0.5+1.0)/3=0.67	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0 + 0.67)/2 = 0.34											
Final **Everglades National Park – Estuarine** Alligator Index Score = 0.34											

Everglades National Park – Inaccessible Areas

Performance Measure	Component 1: Current status			Component 2: 5-year mean			Component 3: Most recent trend			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	N/A										
Body Condition Fulton's K	N/A										
Occupancy Rate %	49.9%	0.5		50.4%	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Final **Everglades National Park – Inaccessible Areas** Alligator Index score = 0.5											

Geometric Mean of Everglades National Park Alligator Index Scores $(0.25 \times 0.34 \times 0.34 \times 0.5)^{1/4} = 0.35$	
Final Everglades National Park Alligator Index score = 0.35	

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Big Cypress National Preserve

Performance Measure	Component 1: Current status			Component 2: 5-year mean*			Component 3: Most recent trend*			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Relative Density (alligators/km)	0.91	0		0.91	0		±	0.5		(0+0+0.5)/3=0.17	
Body Condition Fulton's K	10.69	0.5		10.80	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Occupancy Rate %	N/A										
Mean of Alligator Performance Measure Scores = (0.17 + 0.5)/2 = 0.34											
Final **Big Cypress National Preserve** Alligator Index score = 0.34											

* The mean and trend for relative density in Big Cypress National Preserve are based on only one year's data because monitoring of relative density began in 2006.

Crocodiles

Everglades National Park

Performance Measure	Component 1: Current status			Component 2: 5-year mean*			Component 3: Most recent trend*			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Juvenile Growth (cm/day)	0.171	1		0.126	0.5		±	0.5		(1+0.5+0.5)/3=0.67	
Fall Monthly Hatchling Survival (%)	0.70	0.5		0.69	0.5		±	0.5		(0.5+0.5+0.5)/3=0.5	
Mean of Crocodile Performance Measure Scores = (0.67 + 0.5)/2 = 0.59											
Final **Everglades National Park** Crocodile Index score = 0.59											

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Biscayne Bay Complex

Performance Measure	Component 1: Current status			Component 2: 5-year mean*			Component 3: Most recent trend*			Mean of Component Scores	Performance Measure Stoplight
	Value	Index Score	Stoplight	Value	Index Score	Stoplight	Trend	Index Score	Stoplight		
Juvenile Growth (cm/day)	0.174	1		0.105	0.5		±	0.5		(1+0.5+0.5)/3=0.67	
Fall Monthly Hatchling Survival (%)	Insufficient Data as of 2006.										
Final **Biscayne Bay Complex** Crocodile Index score = 0.67											

Geometric Mean of 6 Alligator Management Unit Scores = $(0.5 \times 0.34 \times 0.31 \times 0.34 \times 0.35 \times 0.34)^{1/6} = 0.36$

System-wide Alligator Index Score = 0.36

Geometric Mean of 2 Crocodile Management Unit Scores = $(0.59 \times 0.67)^{1/2} = 0.63$

System-wide Crocodile Index Score = 0.63

Geometric Mean of Alligator and Crocodile Index Scores = $(0.36 \times 0.63)^{1/2} = 0.47$

System-wide Crocodilian Stoplight Score = 0.47

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Assessment Report for the Oyster Indicator in the Northern Estuaries

2008

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Eastern Oyster *Crassostrea virginica*

LOCATION	LAST STATUS[1]	CURRENT STATUS[2]	2-YEAR PROSPECTS[3]	CURRENT STATUS[2]	2-YEAR PROSPECTS[3]
Eastern Oyster					
Caloosahatchee Estuary	NA			The oysters in the Caloosahatchee Estuary are still being impacted by too much fresh water in summer and too little fresh water in the winter. Too much fresh water impacts reproduction, larval recruitment, survival and growth, while too little fresh water impacts the survival of oysters due to higher disease prevalence and intensity of *Perkinsus marinus* and predation. Current conditions do not meet restoration criteria, signifying that this area needs further attention.	Management objectives for regulating freshwater inflows play an important part in determining oyster success in the Caloosahatchee Estuary. If conditions remain constant, prognosis for the future will be stable. If the hydrological conditions remain the same, we do not expect to see an improvement in oyster responses in this estuary.
St. Lucie Estuary	NA			Insufficient data	Insufficient data
Loxahatchee Estuary	NA			Insufficient data	Insufficient data
Lake Worth Lagoon	NA			Insufficient data	Insufficient data
Lostman's River (Southern Estuaries)	NA			Insufficient data	Insufficient data

Stoplight Color Legend

Red – Substantial deviations from restoration targets creating severe negative condition that merits action.
Yellow – Current situation does not meet restoration targets and merits attention.
Green – Situation is good and restoration goals or trends have been reached. Continuation of management and monitoring effort is essential to maintain and be able to assess "green" status.
Blank - Insufficient data to infer trends.

[1] Data in the last status column reflect data collected prior to calendar year 2000.
[2] Data in the current status column reflect data collected between calendar years 2000 – 2007.
[3] The following assumption is being used for the 2-year prospects column: there will be no changes in the water management from the date of the current status assessment.

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KEY FINDINGS – EASTERN OYSTER

SUMMARY FINDING: On the whole, Eastern oyster status remained constant up to 2007. Given the duration of monitoring of this species, only Caloosahatchee Estuary had sufficient data to infer trends and status of this indicator. Monitoring in other estuaries (St. Lucie Estuary, Loxahatchee Estuary, and Lake Worth Lagoon) are on going, and will yield data to make trend and status assessments in the coming years. Current conditions in the Caloosahatchee Estuary show deviations from restoration targets, therefore restoration actions are merited. Status of oysters is expected to improve if hydrologic conditions are restored to more natural patterns.

KEY FINDINGS:

1. Preliminary results suggest that oyster status in the Caloosahatchee Estuary is the highest in the Northern Estuaries and remains stable. It should be cautioned that insufficient data exists for other estuaries to infer trends and make statistical comparisons.
2. There is too much freshwater inflow into the Caloosahatchee Estuary in the summer months and too little freshwater inflow into the estuary in the winter months, disrupting natural patterns and estuarine conditions. The oysters in the Caloosahatchee Estuary are still being impacted by this unnatural water delivery pattern. Too much fresh water impacts reproduction, larval recruitment, survival and growth while too little fresh water impacts the survival of oysters due to higher disease prevalence and intensity of *Perkinsus marinus* and predation.
3. Overall status of oysters in the Caloosahatchee Estuary is below restoration targets and requires action in order to meet restoration goals.
4. Oyster responses and population in the Caloosahatchee Estuary, while below targets, appear to be stable at this time and are expected to increase given proper hydrologic conditions through restoration.
5. Restoration of natural patterns (less freshwater flows in the summer and more freshwater flows in the winter) along with substrate enhancement (addition of cultch) is essential to improving performance of oysters in the estuaries.
6. Continued monitoring of oysters in the Caloosahatchee and other estuaries will provide an indication of ecological responses to ecosystem restoration and will enable us to distinguish between responses to restoration and natural variation.

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Assessment Report for the Oyster Indicator in the Northern Estuaries 2008

Aswani K Volety, Patricia Sime, Patricia Goodman
and Kimberly Chuirazzi, Editors

Introduction

The Eastern oyster, *Crassostrea virginica*, is a dominant feature of the estuaries in South Florida. Oysters serve as an excellent indicator species for several reasons:

- Salinity and other water quality conditions suitable for oysters also produce optimal conditions for other desirable organisms.
- Oysters filter water and provide habitat, shelter and food for over 300 marine species.
- Crustaceans and fishes that reside in or visit oyster reef communities provide critical prey for larger fish and birds.
- Given the oyster's sedentary nature, it is easy to make cause-and-effect relationships between water quality and oyster health.
- Cause-and-effect relationships between oysters and stressors (water quantity, water quality and sediment loads) have been statistically correlated.
- Oysters are included in the project-level and regional scale modeling, monitoring and assessment efforts.

A system-wide monitoring and assessment plan (MAP) has been developed by the Restoration Coordination and Verification Program (RECOVER) of the Comprehensive Everglades Restoration Project (CERP) that describes the monitoring necessary to track ecological responses to restoration and how responses will be assessed (RECOVER 2004, 2006). Included in the MAP are descriptions of selected indicators, how these indicators are linked to key aspects of restoration, and performance measures that are representative of the natural and human systems found in South Florida. The MAP identified oysters as one of the indicators and established the performance measures described in this report.

Eastern Oyster, Crassostrea virginica

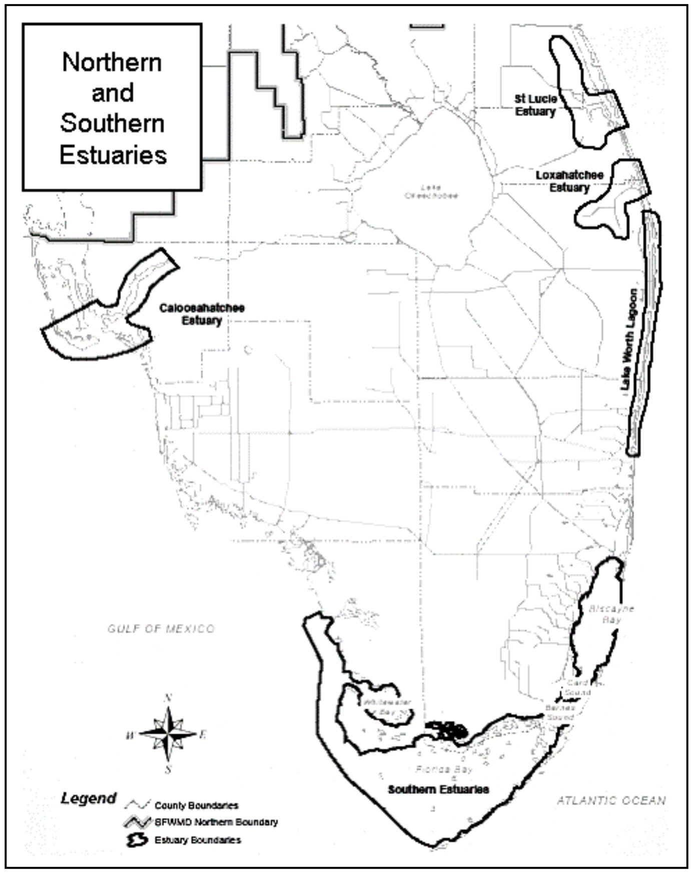

Figure 1. Location of Northern and Southern Estuaries in Florida.

Oysters have also been used as performance measures in many estuarine-linked CERP project plans. The cause-and-effect relationships are described in detail in estuarine conceptual ecological models (Barnes 2005, Sime 2005, Van Armen et al. 2005) and a Total System Conceptual Ecological Model (Ogden et al. 2005) developed by RECOVER. In addition, RECOVER has recommended the oyster be used as an indicator for interim goals (RECOVER 2005).

Oysters in South Florida

Caloosahatchee, Loxahatchee, Lake Worth Lagoon and St. Lucie Estuaries (Figure 1) are collectively referred to as the Northern Estuaries. In these estuaries, oysters have been identified as a "valued ecosystem component" (Chamberlain and Doering 1998a, b). Oysters are natural components of estuaries along the Gulf of Mexico and were abundant in the Northern Estuaries (RECOVER 2007). Currently, MAP oyster monitoring is conducted only in the Northern Estuaries.

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Salinity is an important determinant of the distribution of oysters. Adult oysters normally occur at salinities between 10 and 30 parts per thousand (ppt), but they tolerate a salinity range of 2 to 40 ppt (Gunter and Geyer 1955). Occasional, short pulses of freshwater inflow can greatly benefit oyster populations by reducing predator and parasite impacts (Owen 1953), while excessive freshwater inflows may kill entire populations of oysters (Gunter 1953, Schlesselman 1955, MacKenzie 1977, Volety et al. 2003, Volety and Tolley 2005, Bergquist et al. 2006). Where salinities are between 15 and 20 ppt, populations are dense, reproductive activity is high, predator numbers are low, and spat recruitment and growth rates are high. Quality, quantity, timing and duration of freshwater flows have tremendous effect on oyster health, survival, growth and reproduction, and thus the biological responses of oysters are directly related to freshwater-influenced environmental conditions.

Water management and dredging practices have had a major impact on the historical presence, density and distribution of oysters. Historically, drainage patterns were characterized by gentle, meandering surface water flows through rivers, creeks, sloughs and overland sheet flow through contiguous marshy areas. This natural system absorbed floodwater, promoted ground water recharge, assimilated nutrients and removed suspended materials (ACOE and SFWMD 2002). As South Florida developed, the canal network worked too efficiently and drastically altered the quantity, quality, timing and distribution of fresh water entering the estuaries. Water management practices release significant volumes of fresh water over a short period of time, usually as flood releases, into the estuaries resulting in a sudden drop in salinity. This sudden drop can lead to significant mortality in the oyster population, and decreased growth, reproduction and spat recruitment. Freshwater releases during summer months cause flushing of oyster larvae to downstream locations that are unsuitable habitat. Also, undesirable shifts in the estuarine salinity envelope can result in increased susceptibility to disease. Additionally, flood releases and inland runoff contain numerous contaminants from urban and agricultural development. Inflows are too great in the wet season and too little in the dry season to support a healthy estuary.

The objectives of many CERP projects are focused on reducing these impacts. CERP projects that will restore more natural freshwater inflows into the estuaries will provide beneficial salinity conditions, a reduction in nutrient concentrations and loads, and improved water clarity, which will promote the reestablishment of healthy oyster bars. Healthy oyster bars will benefit other organisms that use this habitat during all or part of their life cycle.

Study Area

The Caloosahatchee Estuary was chosen as a model estuary to examine the impact of watershed alteration on oysters and to develop a stoplight report card for oyster physiologic and ecologic response. Figure 2 shows the oyster sampling sites within the Caloosahatchee Estuary. Oyster monitoring is also being conducted in the other Northern Estuaries and assessments for these will be presented in later assessment reports.

Spotlight Restoration Report Card

CERP projects are expected to moderate the stressors (i.e., freshwater discharges, diminished water quality and habitat loss) and enhance the natural attributes (i.e., oysters) of the Northern Estuaries. This will be accomplished through habitat enhancement, as well as water storage and treatment projects. As various CERP projects are implemented, changes in the hydrology, and thus, the biology of oysters will take place. A stoplight report card system that integrates various responses that are currently being measured as part of a monitoring plan can provide a powerful way to distinguish between restoration changes and natural patterns.

Using oyster responses, we have developed a stoplight report card for the Caloosahatchee Estuary based on CERP performance measures to grade an estuarine system's response to human impacts or restoration conditions. We expect to be able to distinguish between responses to restoration and natural patterns by ~ 2015 after more representative rainfall years (wet, dry and normal). The stoplight report card involves a suitability index score for each organism metric as well as a trend score (- decreasing trend, +/- no change in trend, and + increasing trend). Two questions are addressed using suitability curves: 1) Have we reached the restoration target? and 2) Are we making progress toward targets? Results are translated into a stoplight display showing the status of each component. A final oyster index score is obtained by taking the geometric mean of the components. For the Caloosahatchee Estuary, all the metrics are weighted equally in determining the overall score. In other systems, various responses may be dropped or weighted more or less, as appropriate. Stoplight colors indicate success (green), caution (yellow) or failure (red). In this initial assessment, only the Caloosahatchee Estuary is considered. Other estuaries will be included in future assessments.

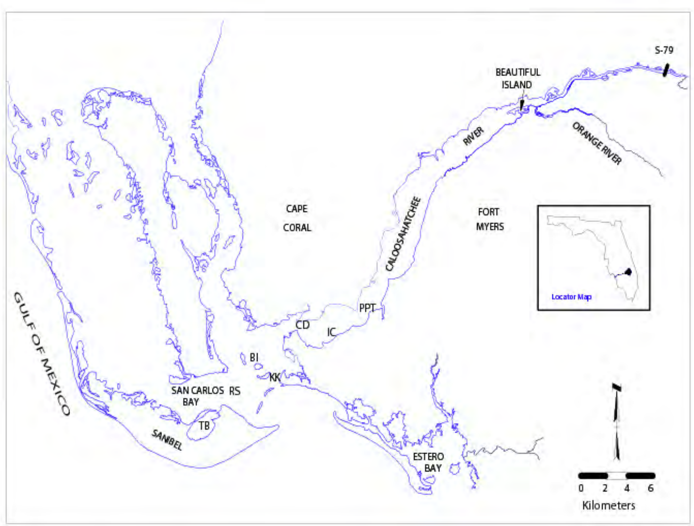

Figure 2. Oyster sampling locations within the Caloosahatchee Estuary. Locations (PPT = Pepper Tree Point, IC = Iona Cove, CD = Cattle Dock, BI = Bird Island and TB = Tarpon Bay) are from upstream to downstream along a salinity gradient.

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Performance Measures

The stoplight restoration report card includes five metrics:

- Density of living oysters
- Condition index
- Gonadal index (reproductive activity)
- Spat (larval) recruitment
- Juvenile growth
- Disease prevalence and intensity

These metrics are correlated with hydrologic conditions including depth, flow, salinity, temperature, dissolved oxygen, season, spatial extent and water quality. Salinity is a critical parameter in estuarine habitats. Targets for oyster performance measures are based on patterns that are considered natural for estuaries along the east and west coast of Florida.

Stoplight scoring criteria for these performance measure metrics are presented in Tables 1a and 1b. A score of 1.0 is the restoration target. All performance measures are averages of 2-5 years data measured during appropriate seasons. The component score (e.g., living density) is the average of the suitability index score plus the trend. Table 2 shows how index ranges are translated into an index score.

Table 1a. Stoplight scoring criteria for suitability index.

Component	Score and Spotlight by Range		
	0	0.5	1
Living Density	0 - 200	>200 - 800	>800 - 4000
Condition Index	0 - 1.5	>1.5 - 3.0	>3.0 - 6.0
Gonodal Index	0 - 1	>1 - 2	>2 - 4
Spat Recruitment	0 - 5	>5 - 20	>20 - 200
Juvenile Growth	0 - 1	>1 - 2.5	>2.5 - 5
Perkinsus marinus Prevalence	>50 - 100	>20 - 50	0 - 20
P. marinus Intensity	>3 - 5	>1 - 3	0 - 1

Table 1b. Stoplight scoring criteria for trend index.

Component	Score and Spotlight by Range		
	0	0.5	1
P. marinus Intensity	>3 - 5	>1 - 3	0 - 1
Trend	- slope	no slope	+ slope

Table 2. Translation for converting suitability or trend index from Tables 1a and b into an index score and stoplight color.

Index Range	Index Score	Stoplight Color
0.0-0.3	0	Red
>0.3-0.6	0.5	Yellow
>0.6-1.0	1.0	Green

Water Quality

Methods

Water quality measurements were taken along with oyster sample collection. Temperature, salinity and dissolved oxygen were measured. Freshwater inflows into the Caloosahatchee Estuary from S-79 Lock and Dam were obtained from the South Florida Water Management District (SFWMD).

Results

As expected, temperatures at the sampling locations in the Caloosahatchee Estuary were higher during the warmer summer – early fall months (April – October) and were lower during the cooler drier months (November – March). In contrast, salinities at the sampling locations were lower during the summer – early fall months (June – October) and higher during the cooler months (November – May; results not shown). There was a significant relationship between flows and salinity at the five sampled locations (Figure 3). The influence of freshwater inflow into the system is more pronounced at the upstream locations compared to the downstream locations.

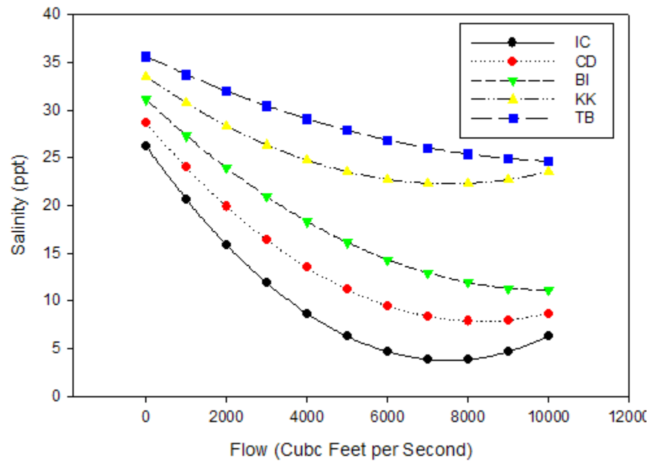


Figure 3. Relationship between freshwater inflow and salinities.

Oyster Density

Methods

Oyster living density, number of living oysters per square meter (oysters/m^2), was measured at the stations shown in Figure 2. Density is measured in the late fall and early spring. This period is the most ideal time for density measurement since oysters have reproduced for the year and spat have settled from the water column. Four 0.25-square meter quadrats were randomly located at the mean low tide height at each reef. The number of living oysters within each quadrat were counted and compared among reefs at various locations.

Results

Salinities are significantly affecting oyster living density. Too much freshwater in the summer months resulting in low salinities reduce the survival or spat and adult oysters at upstream locations. Oyster density ranged between 102 – 2,345 oysters/m^2 at various sampling locations. Mean density for the Caloosahatchee Estuary for the sampling period for

Oyster bar at low tide

which data is available is 765 – 1,795 oysters/m^2. Mean density for all the sampling locations in the estuary was a low of 765 ± 107 (2003) to a high of 1,795 ± 76 oysters/m^2 (2004).

Condition Index

Methods

The physiological condition of an oyster can be measured by its condition index, which is the ratio of meat weight to shell weight (Lucas and Beninger 1985). Although oysters tolerate salinities between 0-42 ppt, growth is maximized at salinities of 14-28 ppt. Slower growth, poor spat production, and excessive valve closure occur at salinities below 14 ppt (Shumway 1996). If an oyster is stressed either by water quality or by disease, it has less energy for growth and reproduction. Consequently, a comparison of oyster condition index among the oyster reefs along the salinity gradient is a good indication of oyster health and the influence of salinity and disease on this health. Oysters from an altered estuary having extreme salinities have significantly lower condition index compared to oysters from an unaltered estuary (Volety and Savarese 2001). Oysters were collected for condition determination monthly between August 1999 and January 2008 at the same time disease prevalence was surveyed.

Results

Annual average oyster condition index ranged between 2.4 and 3.4 (Figure 4). Condition index appears to be related to spawning and salinity conditions. Condition index varied significantly between sampling locations and between sampling months. Condition index in oysters was higher during the December – May period and was lowest in October. Condition index decreased from March – October, a period that coincides with oyster spawning.

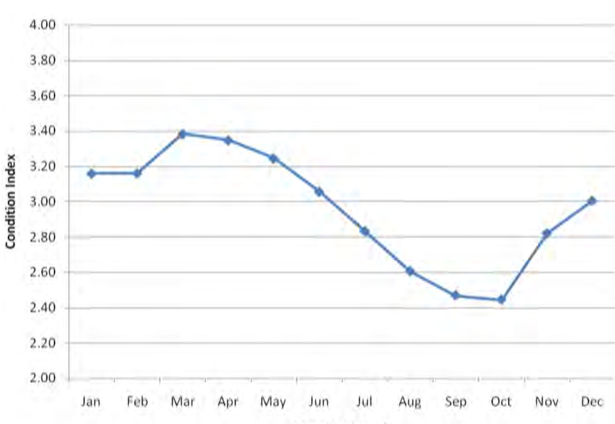
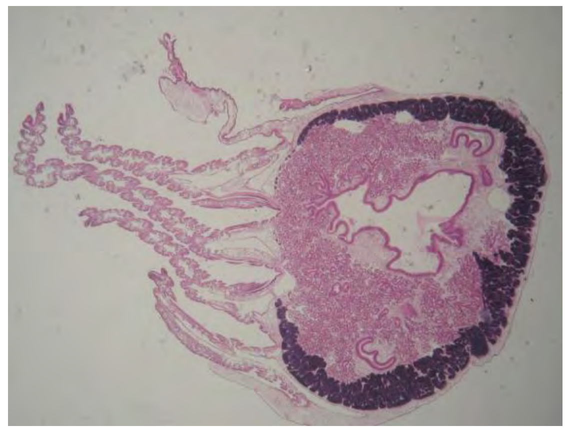


Figure 4. Mean condition index of oysters from all sampling locations in the Caloosahatchee Estuary.

Gonadal Index

Methods

Gonadal index (scale of 0-5) is a measure of the reproductive stage and spawning of oysters. Each month, 10 to 15 samples of oysters were collected from each sampling location between August 1999 and September 2007. Cross-sections of these oysters were made and viewed under a microscope. Gonadal portions of the sections were observed to determine gender and gonadal condition (Volety and Savarese 2001, Volety et al. 2003). The yearly average is used for the index.



Cross-section of an oyster viewed under a microscope used to determine gender and gonadal condition

Results

Salinity may be affecting gonadal condition of oysters, with low salinities detrimental to reproduction. The gonadal index was very cyclical and varied significantly between sampling locations and sampling months. It was higher during April – October, suggesting an active spawning of oysters and was lower during November – March months (Figure 5).

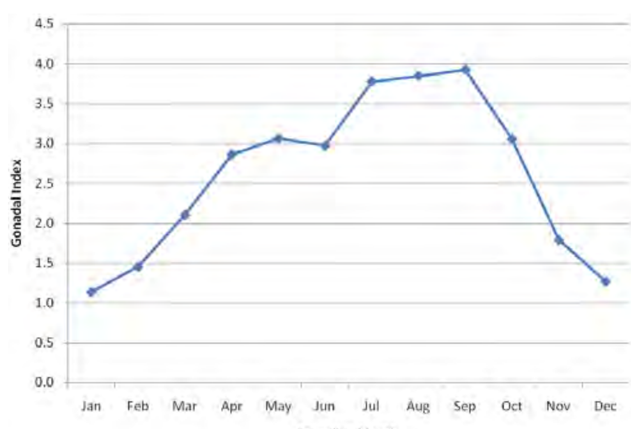


Figure 5. Mean gonadal stage of oysters from all the sampling locations in the Caloosahatchee Estuary.

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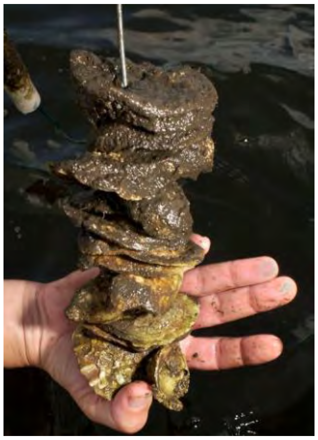
Spat Recruitment

Methods



Shell string used to conduct spat recruitment experiments

Oyster spat recruitment experiments were conducted using old adult oyster shells strung together by a weighted galvanized wire and deployed at sampling locations. A shell string consisting of 12 oyster shells, each 5.0-7.5 cm long, was suspended off the bottom at various sites (Haven and Fritz 1985). Oyster spat settlement was monitored monthly by counting the number of spat settled on the underside of strung shells. Spat settlement is expressed as the number of spat settled per oyster shell per month. Data was collected monthly from each of the sampling locations between August 1999 and January 2008.

Results

Spat recruitment significantly affects oyster spat recruitment. High freshwater inflows and low salinities either result in mortality or flush the larvae to downstream locations where suitable substrate may not be available. Spat recruitment per shell ranged between 2.5 and 25. Spat recruitment of oysters varied significantly between sampling locations and sampling months. Recruitment of spat was higher between April – October, with peak recruitment occurring in August. Little or no spat recruitment was observed between November – March (Figure 6).

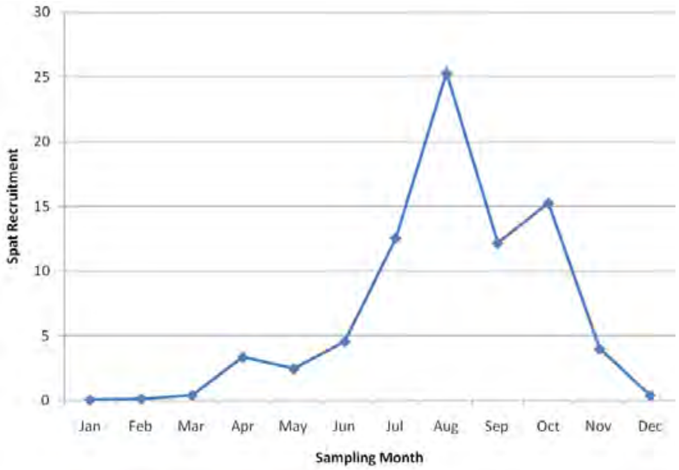


Figure 6. Mean spat recruitment (spat/shell) of oysters from all the sampling locations in the Caloosahatchee Estuary.

Juvenile Oyster Growth

Methods

One to two hundred juvenile oysters (10-20 mm) were deployed at all sampling locations in 0.5-mm closed and open wire mesh bags. Fifty randomly selected oysters were measured to the nearest 0.1 mm every month from each location. Juvenile oysters were placed in wire mesh bags to exclude predation and indicate growth and/or mortality due to water quality.

Results

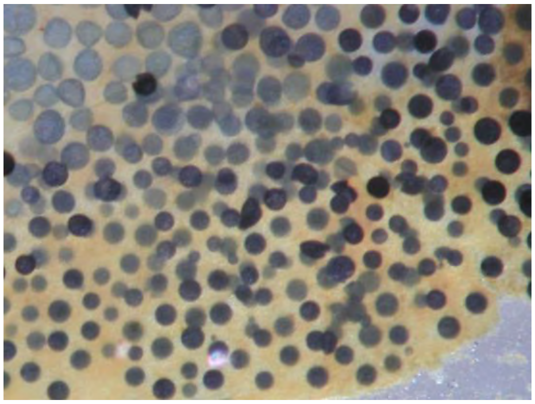
Juvenile oyster growth and survival was poor at the upstream locations given high freshwater inflows and low salinities during the summer months. Juvenile oyster growth (mm/month) and mortality varied widely between sampling locations and sampling months. Significant juvenile mortality was observed when oysters were deployed during the summer months when the salinities are typically low (results not shown). When oysters were deployed in late fall months (October – December), when salinities are higher, higher growth was observed at the upstream locations, which tended to have more estuarine salinities compared to downstream locations where the salinities are marine to hypersaline. Mortality rates were typically 60% - 100% depending on the salinity (results not shown).

Disease Prevalence and Intensity

Methods

Perkinsus marinus, a protozoan parasite, causes disease in oysters. Susceptibility to this disease of oysters along the salinity gradient within the Caloosahatchee Estuary was determined at six locations. A total of 10-15 oysters per location were collected monthly between August 1999 and January 2008.

The presence of *P. marinus* was determined by taking samples of gill and digestive sacs and incubating them for 4-5 days in a solution that will enlarge the *P. marinus* cells allowing for visual identification under a microscope (Ray 1954, Volety et al. 2000, Volety et al. 2003). Prevalence of infection was calculated as percent of infected oysters. The intensity of infection was recorded using a modified Mackin scale (Mackin 1962) in which 0 = no infection, 1 = light, 2 = light-moderate, 3 = moderate, 4 = moderate-heavy, and 5 = heavy.



Perkinsus marinus

Identification and measurement of organisms in oyster reefs

Results

P. marinus infection was significantly affected by freshwater inflows. Low salinities during the summer months and low temperatures during the winter months moderate the infection prevalence and intensities. Mean *P. marinus* prevalence from all the sampling locations and sampling months ranged between 31 and 66% (Figure 7) between sampling months and between 35 and 56% between sampling locations (results not shown). Similarly, *P. marinus* intensity ranged between 0.64 and 1.16 during various sampling months (scale 0-5; Figure 8) and between 0.41 and 1.1 at various sampling locations (results not shown). Disease prevalence and intensity increased with increasing salinity and distance downstream (results not shown). On average, disease prevalence and intensity was higher in January (when salinities tend to be higher) and during August (when temperatures tend to be the highest).

Discussion

Changes in average oyster condition index coincided with the reproductive phase of oysters. As oysters reproduce, gametes are shed resulting in a decrease in body mass and thus a reduced condition index. This trend is reinforced by the gonadal index of oysters as well as spat recruitment. Gonadal index of oysters was higher during the peak spawning months (April – October 8). Larval recruitment was observed at various sampling locations between April – October. Spat recruitment per shell is not limited by larval availability. Juvenile oysters grow faster than adult oysters, thus enabling the determination of growth rates at various locations subjected to various salinities. Given the amount of freshwater inflows into the Caloosahatchee Estuary (0 – >15,000 cfs), growth and survival of oysters was significantly impacted at the extreme end of the salinity range. Disease prevalence and intensity increased with increasing salinity and distance downstream (results not shown). On average, disease prevalence and intensity was higher in January when salinities tend to be higher and during August when temperatures tend to be the highest.

These results are used in the present study to develop an easy to understand Stoplight Report Card System to present the current state of oysters in the Caloosahatchee Estuary. In this case, the results are not used to examine the relationship between various water management practices and interrelationships between oyster responses and other factors that influence them.

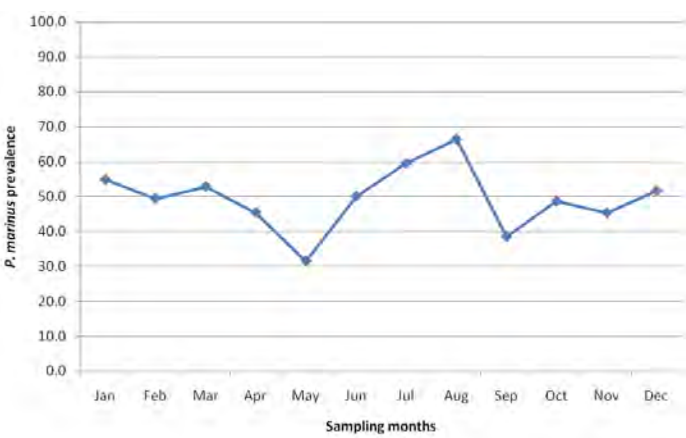


Figure 7. Mean prevalence of *P. marinus* (percent of infected oysters) from all the sampling locations in the Caloosahatchee Estuary.

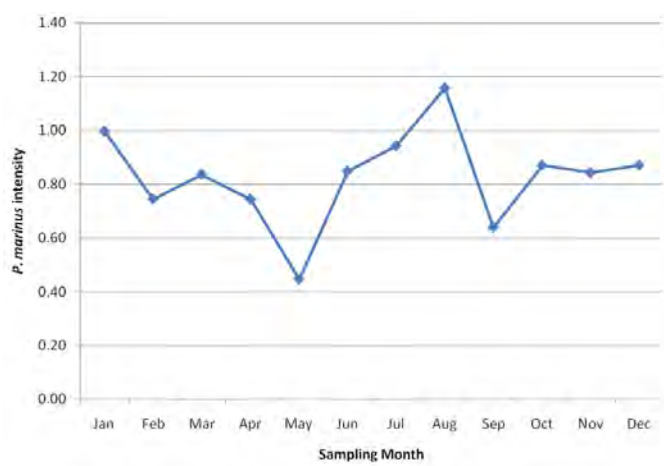


Figure 8. Mean intensity of *P. marinus* in oysters from all the sampling locations in the Caloosahatchee Estuary.

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Final Stoplight Scores

Component stoplight scores and the overall oyster spotlight score based on the available data are presented in Table 3. The component scores were: living density = 0.75, condition index = 0.5, gonadal index = 0.75, spat recruitment = 0.5, juvenile growth = 0.5, *P. marinus* prevalence = 0.25, and *P. marinus* intensity = 0.5. Components yield a combined score for the location of 0.5. The oyster population within the Caloosahatchee Estuary is at a "caution" stage (yellow) indicating current conditions do not meet restoration criteria. This aeaa needs further restoration attention. Management objectives for regulating freshwater inflows play an important part in determining oyster success in the Caloosahatchee Estuary. If conditions remain constant, prognosis for the future will be stable.

Table 3. Component and overall spotlight scores for oysters in the Caloosahatchee Estuary

Component	Parameter Value	Parameter Value Stoplight	Index Score	Trend	Trend Stop light	Trend score	Average Component Score	Stoplight
Living Density (living oysters/m^2)	1029		1	±		0.5	(1+0.5)/2=0.75	
Condition Index	2.96		0.5	±		0.5	(0.5+0.5)/2=0.5	
Gonadal Index	2.61		1	±		0.5	(1+0.5)/2=0.75	
Spat Recruitment (spat/shell)	6.43		0.5	±		0.5	(0+0.5)/2=0.5	
Juvenile Growth (mm/month)	2		0.5	±		0.5	(0.5+0.5)/2=0.5	
P. marinus Prevalence (% of infected oysters)	49.5		0.5	-		0	(0.5+0)/2=0.25	
P. marinus Intensity	0.83		1	-		0	(1+0)/2=0.5	
Geometric mean of oyster component scores $(0.75 \times 0.5 \times 0.75 \times 0.5 \times 0.5 \times 0.25 \times 0.5)^{1/7} = 0.508$								
Final Eastern Oyster Index score = 0.5								

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Aquatic Fauna and Periphyton Production Data Collection
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Periphyton Component

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The utility of periphyton to expose ecological ramifications to restorative or deconstructive change in the Everglades is due its bearing several of the most desirable features of a reliable ecological indicator which include (1) being distributed throughout the system of study, (2) having rapid response to environmental change that is (3) readily quantifiable at several levels of biological organization (individual, species, population and community) with (4) consequences to levels above and below it's placement on the food web (Karr 1999). Reliance on periphyton to indicate environmental change has been well justified by scientific research conducted in the Everglades (McCormick & Stevenson 1998, Gaiser et al., 2006) which adds regional applicability to the existing body of literature in aquatic sciences that has supported the widespread employment of periphyton monitoring in aquatic ecosystem management (Hill et al., 2000; Stevenson 2001). Specifically, we anticipate that patterns of periphyton production, nutrient content and composition among and within the PSUs sampled in this mapping assessment will provide reliable indication of changes driven by hydrology and nutrient enrichment. Alterations in periphyton attributes then cascade through the system to affect higher organisms through changes in food quality, composition and concentration of gasses and nutrients in the water column and ecosystem structure (i.e., soil formation and quality, physical habitat structure).

The following hypotheses were formulated by the RECOVER assessment group using data from descriptive and experimental studies. We list the hypotheses and follow each with a discussion of supporting data and progress in application to this project.

H11b – Lengthened hydroperiods cause an increase in the proportion of floating, calcareous periphyton mat (associated with Utricularia purpurea) but when water depths exceed ~1-2 m, calcareous floating mats are replaced by epiphytic non-calcareous algal communities.

Supporting data: Studies along transects in compartmentalized Everglades wetlands showed hydrologically-driven gradients in periphyton mat structure (Gaiser et al., 2006). Water depths >~1.5 m supported non-calcareous algal communities while sloughs and marl prairies were dominated by thick, highly productive calcareous mats. Other long term studies have shown that calcareous mat productivity is highest in the short-hydroperiod wet prairie (Iwaniec et al., 2006; Ewe et al., 2006) where benthic, sediment-associated mats predominate. At slough sites, *Utricularia*-associated floating mats are less productive but show a distinct seasonality with marked increases in production during the peak of the wet season (Gaiser et al., 2006).

Progress: Although we have yet to incorporate site-specific hydroperiod estimates into our analysis of periphyton distribution patterns, we examined large-scale patterns in the cover of periphyton of different types during the dry and wet season sampling of 2005 and 2006. The cover types are shown in Figure 1 and include calcareous floating mat (associated with *Utricularia purpurea*), calcareous epiphyton (associated with submersed stems of emergent macrophytes), calcareous benthic mat (adhered to the sediment or rock surface), green filamentous algae (non-mat forming) and flocculent detritus (sampled when no other periphyton was available). We found substantial differences in periphyton cover by type as we move from the northern PSUs to the southern part of the system (Figure 2). For instance, sites in Lake Okeechobee, Pal Mar, Holeyland and Loxahatchee National Wildlife Refuge contained little calcareous mat and often had large quantities of filamentous green algae. This is likely a geologically-driven pattern rather than one driven by changes in nutrients or hydrology, per se, because these basins are not underlain by limerock that facilitates precipitation of carbonates. This is reflected in the pH values, which were comparatively low in the northern basins compared to areas further to the south (Figure 3). The continuance of the pattern of increasing calcareous biomass southward through the water conservation areas is likely largely driven by the corresponding decrease in P availability as well as hydroperiod with distance from canal inflows, as these same trends were observed in these basins in earlier transect studies (Gaiser et al., 2006). The increase in floc observed in the oligohaline zone at the very base of our study area reflects the input of particulate organic material from mangroves inhabiting this zone. There were no strong notable seasonal patterns in cover by substrate type, but continued regular sampling should improve our ability to detect such differences if they do exist.

H11c – Nutrient enrichment causes an elevation in periphyton nutrient content, a reduction in the proportion of calcareous floating and epiphytic periphyton mats, and a replacement of native species by non-mat forming filamentous species.

Supporting data: Throughout the system, periphyton has been proven to provide rapid and accurate indication of water quality changes; periphyton responses were critical to establishing the P criterion for freshwater sloughs (McCormick et al., 1996; Gaiser et al., 2004), have been used to indicate rates of coastal salt water encroachment in mangroves (Ross et al., 2001; Gaiser et al., 2004) and for detection of nutrient enrichment in adjacent offshore seagrass beds (Frankovich et al., 2006). Several studies have shown that periphyton not only respond to but also regulate water quality (Thomas et al., 2006; Gaiser et al., 2006) by quickly and efficiently removing excess P from the water column. Gaiser et al. (2005) recommends using periphyton P content as a metric of P enrichment history, rather than water or soil P, because it has been shown repeatedly to provide a much more reliable indication of P load history. This has been adopted in most large-scale monitoring programs in the Everglades (i.e., this study, the EPA REMAP assessment, FCE LTER research).

Progress: We found very strong and temporally consistent spatial patterns in periphyton biomass that showed a general increase from northern to southern PSUs (Figure 4). Periphyton biomass, measured by biovolume, cover, dry and ash-free dry mass all increased to the south, becoming highest in Shark River Slough, the Southern Marl Prairie and Taylor Slough. The opposite pattern was observed for the total phosphorus, organic and chlorophyll a content of the periphyton. Total phosphorus values were highest in the flocculent periphyton of Lake Okeechobee, Pal Mar, Holeyland and Loxahatchee National Wildlife Refuge. They increased again in the oligohaline zone, where P delivery from marine and groundwater is likely (Childers et al., 2006; Price et al., 2006). There were no notable seasonal patterns in these values, although biomass has normally found to be highest in the wet season (Iwaniec et al., 2006). We suspect that continued sampling will reveal this weaker temporal trend.

The spatial patterns in periphyton attributes and total phosphorus content were highly correlated in expected ways. We found a strong and temporally persistent decrease in periphyton cover, biomass and biovolume with increasing total phosphorus content among sites (Figure 5). The organic content increased with total phosphorus availability, which is commonly observed as both a consequence and driver of the concomitant change in periphyton biomass (Gaiser et al., 2006). Notably, however, it is not just a loss of the calcitic matrix that occurs with phosphorus enrichment but a loss of biomass as well (Gaiser et al., 2005), as shown here in the strong negative correlation of ash-free dry periphyton biomass with total phosphorus content. An increase in the chlorophyll a content of that biomass with increased phosphorus availability is expected, as P availability increases productivity of cells in species with high P requirements. We observed a strong positive association of periphyton chlorophyll a with phosphorus, especially in the wet season of 2005 and dry season of 2006. There is a slightly positive correlation between phosphorus availability and water depth, although this trend was only significant in the wet season of 2005.

We also found strong spatial patterns in algal species composition among the PSUs. A total of 229 non-diatom algae and 155 diatom taxa were found in the 2005 survey of 148 sites (Tables 1, 2). Indicator species analysis (Dufrene & Legendre 1997) showed that some species were significantly associated with particular PSUs. Non-metric multidimensional scaling ordination biplots show that subsets of PSU's can be grouped by algal species composition (Figure 6). Communities in Pal Mar, Loxahatchee and Lake Okeechobee differ substantially from other PSU's. The water conservation areas group together on the other side of the biplot. The oligohaline zone is separated from the rest, likely because of the response of the algal community to increased salinity in this area. We found the first gradient in composition to be highly correlated with total phosphorus and negatively correlated with periphyton biomass. Communities associated with calcareous mats were also indicative of low phosphorus quantities, and vise-versa. We found that very productive mats with low TP and organic content were dominated by cyanobacteria while P-enriched communities of low biomass but high organic content were dominated by diatoms (Figure 7).

Because of this strong relationship between composition and total phosphorus availability, we were able to determine total phosphorus optima and tolerances for the most common taxa by weighted-averaging regression. The values reported in Tables 1 and 2 can be used to predict, by weighted averaging regression, the total periphyton P content for sites in which these taxa are found. The prediction power of the model generated with these optima and tolerance values was measured by an $R^2 = 0.54$ and an RMSE of 53. This means that 54% of the variability in species distributions can be explained by phosphorus, and phosphorus predicted at any given site by the species is within 53 µg g-1 of its actual value. Mean TP values indicative of natural conditions vary among wetland PSUs but an error of 53 µg g-1 will be within 10-25% of the mean, suggesting a high predictive power of this model (Gaiser et al., 2006). There were strong trends in the residuals of this model that are important in evaluating the model's accuracy. The trends are related to the gradient reflected in the second axis of the NMDS which is correlated with pH, water and soil depth. Areas of deeper soils are also deeper and peat-forming, causing a reduction in the pH, which is known to be a strong driver of algal community composition. Future models must take this second important gradient into account, indicating that algal-based P inference models will be strongest when created and employed in a regionally-specific manner. The next step in our modeling efforts will be to create such regionally-explicit P-inference models. Basin-specific P-prediction models presented in Gaiser et al. (2006) had much higher predictive power than whole system models, suggesting that model development on a smaller scale is also an appropriate approach in this survey. Conversely, multiple models are more cumbersome so our next steps are to (1) develop a multi-parameter model for the entire system (that explains residual trends driven by large scale variability, above) and (2) develop explicit P-prediction models for each PSU and, by cross-validating across PSUs, determine the optimal scale for a univariate prediction model. We believe that the latter approach will be most powerful, as it will facilitate direct interpretation of eutrophication trends from periphyton species data.

H11a – Shortened hydroperiods cause a reduction in the proportion of diatoms and green algae and an increase in calcareous blue-green algae, possibly reducing food value of periphyton, and affecting overall productivity of the Everglades.

Supporting data: Compositional responses of periphyton to hydrologic change were quantified in field and laboratory studies by Gottlieb et al. (2006 a, b) and Thomas et al. (2006). Gottlieb et al. (2006 a) found marked differences between algal communities in long and short-hydroperiod marshes of Everglades National Park and derived hydrologic optima and tolerances for the most abundant species. Thomas et al. (2006) and Gottlieb et al. (2006) conducted drying and re-wetting experiments to determine the length of time it takes the community to be measurably altered when exposed to an alternative hydroperiod, and found significant change within days to weeks of exposure. Further, studies by Geddes et al. (2003) and Dorn et al. (2006) documented the connection between periphyton composition and consumers showing that the two are connected partly through the periphyton-derived detrital food web and also through nutrient regeneration by the animals.

Progress: We found a decrease in water depth from the water conservation areas to the base of Shark River and Taylor Slough that roughly corresponds to reductions in hydroperiod, although site-specific hydroperiod estimates have not yet been incorporated into this analysis (Figure 3). This decrease in water depth was associated with an increase in periphyton biomass with decreased organic and total phosphorus content (Figures 3-5). Water depth also explained compositional differences among sites with deeper water generally being associated with increased abundance of diatoms and reduced abundances of cyanobacteria (Figure 7) while benthic mats in shallow habitats had low TP and organic content and were dominated by cyanobacteria and reduced abundances of diatoms and green algae (Figure 7). Our next step in this analysis is to incorporate site-specific hydroperiod predictions (generated from EDEN) to quantify hydrologic controls on periphyton trends.

To determine if alterations in periphyton composition affect inferred food quality for invertebrate and fish consumers, we have begun to explore the relationships between periphyton atteributes and fish and macroinvertebrate density. We found reduced numbers of fish and macroinvertebrates with increasing periphyton biomass, which is likely due to the reduced quality of periphyton in short-hydroperiod marshes. These mats are highly calcareous rendering them difficult to graze by many consumers (Geddes et al., 2003). As the organic content of periphyton increased, we found increasing abundances of herbivorous invertebrates (except crayfish) and fish (Figure 8). There were some taxon-specific associations between the herbivore communities and algae, with the abundance of grass shrimp (*Palaemonetes paludosus*) increasing with algae of higher organic content while crayfish abundance was associated with high TP diatom communities (Figure 6). When algae were grouped into higher level taxonomic categories (i.e., cyanophytes, chlorophytes, diatoms) we found even greater associations with the grazer community. For instance, the number of grass shrimp, other grazing macroinvertebrates and fish declined with increasing abundance of filamentous cyanobacteria (Figure 9). This is likely due to the decreased quality of blue-green-dominated mats that not only mechanically deter filter feeders but reduce palatability through the calcium carbonate precipitated on their sheaths and through production of antimicrobial toxins. Conversely, these three consumer groups increased with diatom and green algal abundance probably due to the increased nutritional value of these algal groups. This was our first attempt to link the consumer and periphyton data and we are planning a much more sophisticated analysis at more appropriate spatial scales to better tease apart the underlying consumer-food resource relationships.

2. Communicating the Periphyton Indicator

Periphyton plays a critical role in the food web as a food source and prey refuge. Given its extremely high rates of production (Iwaniec et al., 2006), the vast areas of South Florida marsh covered by periphyton may represent a significant sink for carbon, another important functional role of the Everglades from a global perspective. Taxonomic diversity of microbial organisms that comprise periphyton is higher than most other biotic communities, thereby making a substantial contribution to system biodiversity estimates. Functional consequences of this diversity are unexplored, yet literature would support the contention that it is because of this diversity that algal-microbial communities make such reliable ecological indicators.

2.1. Indicator Performance Measures and Metrics

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Several metrics provide reliable measure of periphyton response to hydrologic and water quality change in this system. They can be broadly grouped into three categories of abundance, quality and community composition. Within these categories at least three measures are recorded within the context of CERP assessment. These include, for abundance, wet biovolume (ml m^{-2}), dry biomass (g m^{-2}) and ash-free dry biomass (g m^{-2}); for quality, organic content (µg dry g^{-1}), chlorophyll a content (µg dry g^{-1}) and total phosphorus content (µg dry g^{-1}); and, for the community category, algal composition and diatom composition (measured using similarity metrics in multi-dimensional ordination space) and substrate affiliation (percent cover by substrate type).

Within each of the categories, all of the parameters respond in the same direction (positive or negative) to changes in hydrologic conditions, including depth, duration, timing, and spatial extent, as well as water quality (Gaiser et al., 2006). The periphyton biomass metrics of wet biovolume, dry biomass and ash-free dry biomass, expressed on a per square meter basis, are correlated with each other all decline with increasing water depth and hydroperiod and with increasing availability of phosphorus (Gaiser et al., 2006; Ewe et al., 2006). The periphyton quality metrics of organic, chlorophyll *a*, and total phosphorus content, expressed per unit dry mass, are correlated with each other and increase with increasing water depth and hydroperiod and with increasing availability of phosphorus (Gaiser et al., 2005, 2006). The communityl metrics are based on compositional similarity to expected community structure, established from collections at reference locations (according to Gaiser et al., 2006). Periphyton cover by substrate type is dealt with in a similar manner, where substrate types are given optima and tolerances along each gradient based on their distribution, and then site water quality predictions based on those optima weighted by relative cover.

2.2. The Stoplight Report Card System applied to Periphyton

The stoplight system for periphyton involves first calibrating the tri-color code by the deviation of values for each metric from an expected baseline condition for each sampling point (Figure 10.). Triplicate samples from principal sampling units (PSU's, randomly selected locations within landscape sampling units, LSU's) visited annually in the mid-wet season are analyzed for each periphyton metric. PSU means are then compared to expected values for background conditions defined for the respective LSU. Background conditions are defined from data collected or inferences made from locations within the LSU that are considered un-impacted by human activities and are not static; that is, ranges of acceptable conditions may change depending on modifications by external drivers not under our control (i.e., climate variability) and advancements in the understanding of the ecosystem. Development of a consistent baseline necessitates long-term data, so we do expect targets to evolve as the duration of monitoring programs grow. However, any changes in baseline expectations will be documented and then hindcast through the stoplight system to re-calibrate former values.

2.3. How We Determine Thresholds for Periphyton Success (Green), Caution (Yellow) or Failure (Red)

Once baseline expectations for each of the 9 variables are established, color codes are assigned to each PSU based on deviation from that expectation. If the value is within one standard error of the mean, it is designated green (natural), within two standard errors is designated yellow (caution) and beyond three standard errors is designated red (altered) (see Figure 10). The PSU is then assigned a color for biomass, quality and community composition. The distribution of color designations can then be mapped by PSU for each of these three performance measures. The final color designation for each LSU is then based on the percentage of yellow and red sites. An LSU is given a final yellow designation if more than 25% of sites are coded yellow or red and a red designation if more than 50% of the sites are red, with these cut-offs being based on variability determined within unimpacted background sites (Gaiser et al., 2006).

Baseline expectations for periphyton TP content, ash-free dry biomass and composition for some LSU's are fairly well-defined and so we provide an example using those data. The expected ranges for these variables for un-impacted conditions for WCA-1A, WCA-2A, WCA-3A, SRS and TS were defined by transect surveys conducted in these areas in 1999 by Gaiser et al., (2006). As stated, green coding was used to define acceptable ranges defined by the mean values of unimpacted sites +/- 1 standard error of that mean, yellow for values between 1-2 standard errors and red for sites departing more than 2 standard errors from the mean. Figure 10 shows how each basin has unique ranges of acceptable values and how each attribute scales differently. Data from 2005 and 2006 CERP Mapping surveys were plotted on these graphs to show the proportion of sites falling in each of the colored regions.

For annual assessments, a map of the distribution of the periphyton TP indicator is displayed (Figure 11) to show within and among-region pattern. Pattern and suspected causes are displayed in the "summary" and "key findings" sections (see stoplight report). Each basin is then assigned a value (again using the green-yellow-red coding) based on the proportion of sites falling into these ranges (explained above). Explanation is then provided for causes of current conditions and prospects for 2 years in the future if water management remains the same.

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Table 1. List of the most common algal taxa in samples from the Fall 2005 survey, with the number of sites present (of 148), their maximum relative abundance, total phosphorus optimum (in µg g^{-1} periphyton dry weight, calculated by weighted averaging) and the location for taxa significantly associated with a particular sampling unit.

Taxon	Freq.	Max	Opt. TP	Location
Achnanthes caledonica	33	10	286	
Achnanthes gracillima	7	6	298	
Achnanthes minutissima	4	1	843	
Amphora coffeaformis	2	9	624	
Amphora sulcata	13	19	319	Holeyland
Anabaena spp.	13	1	565	
Ankistradesmus	2	0	506	
Anomoeoneis sphaerophora f. *costata*	1	0	103	
Aphanocapsa spp.	29	3	255	
Aphanothece spp.	148	63	382	
Bacillaria paxillifer	1	0	478	
Brachysira aponina	1	1	463	
Brachysira brebissonii	23	36	659	Okeechobee
Brachysira neoexilis	120	13	356	
Brachysira neoacuta	6	1	421	
Bulbochaete sp. 1	22	6	422	
Caponea caribbea	2	0	111	
Centritractus spp.	6	1	389	
Characium spp.	4	1	381	
Chroococcidiopsis sp.	44	34	938	Okeechobee
Chroococcus morph large	61	7	262	
Chroococcus morph small	121	10	262	
Closterium spp.	4	0	149	
Coelastrum spp.	8	1	202	
Coelosphaerium spp.	67	8	248	
Coscinodiscus spp.	3	2	258	
Cosmarium commisurale	6	2	289	
Cosmarium contractum	17	2	308	WCA 2A
Cosmarium cf. *depresessum*	6	1	300	WCA 2A
Cosmarium excavatum	3	0	183	
Cosmarium isthmium	2	1	290	
Cosmarium monomazum	2	1	215	
Cosmarium ocellatum	6	1	208	
Cosmarium phaseolus	10	1	554	
Cosmarium pyramidatum	16	4	627	
Cosmarium reniforme	11	2	648	
Cosmarium spp.	76	2	403	
Crucigenia quadrata	2	0	515	
Cyclotella meneghiniana	7	2	731	
Dactylococcopsis sp.	1	0	81	
Desmidium baileyii	29	9	328	
Desmidium schwartzii	16	8	361	
Desmidium sp.	3	6	719	Pal Mar
Diadesmis confervacea	1	5	1759	

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Diploneis oblongella	4	0	255	
Diploneis parma	20	6	918	
Diploneis puella	1	0	1759	
Encyonema evergladianum	126	40	246	
Encyonema sp. 1	40	6	313	
Encyonema sp. 2	76	6	399	
Encyonema sp. 4	5	1	233	
Encyonema silesiacum var. *elegans*	23	11	530	Holeyland
Encyonema sp. 3	6	1	218	
Encyonema spp.	2	1	795	
Encyonopsis microcephala	76	16	417	Pal Mar
Encyonopsis sp. 1	16	10	628	Holeyland
Encyonopsis subminuta	15	4	551	
Euastrum cornubiense	20	1	323	
Euastrum pectinatum	8	0	442	
Euastrum small morph	17	1	326	
Euastrum spp.	8	1	284	
Eunotia camelus	2	0	1260	
Eunotia flexuosa	7	8	1495	
Eunotia incisa	6	1	955	
Eunotia monodon	5	1	569	Pal Mar
Eunotia naegelii	19	7	1073	Okeechobee
Eunotia rabenhorstiana var. elongata	3	0	1105	
Eunotia spp.	11	1	687	
Fischerella spp.	6	4	317	
Fragilaria nana	44	18	551	
Fragilaria spp.	4	3	870	
Fragilaria synegrotesca	105	24	323	Loxahatchee
Fragilaria ulna	3	2	694	
Fragilariforma spp.	19	11	537	Okeechobee
Frustulia rhomboides	2	1	310	
Genicularia elginensis	20	1	195	
Gloeocapsa spp.	1	0	400	
Gloeotaenium spp.	5	1	204	
Gloeocystis spp.	19	3	327	
Gloeothece spp.	138	22	279	
Gomphonema affine	1	1	1759	
Gomphonema auritum	2	0	703	
Gomphonema coronatum	1	1	591	
Gomphonema gracile	7	0	345	
Gomphonema intricatum var. *vibrio*	33	3	452	
Gomphonema maclaughlinii	1	0	661	
Gomphonema parvulum var. *lagenula*	1	0	1000	
Gomphonema pratense	1	0	661	
Gomphonema spp.	4	0	1221	
Gomphonema vibrioides	11	2	202	
Gonatozygon spp.	6	0	290	
Gomphosphaeria spp.	73	7	369	
Hantzschia amphioxys	1	0	1759	
Johannesbaptista sowerbyi	3	1	138	

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Kirchneriella spp.	1	0	434	
Lagynion spp.	4	1	468	
Lemnicola hungarica	1	1	65	
Lyngbya spp.	77	24	250	
Mastogloia lanceolata	5	25	574	
Mastogloia smithii	125	38	268	
Mastogloia smithii var. *lacustris*	1	0	315	
Micrasterias crux-mellitensis	2	0	161	
Micrasterias pinnata	4	1	381	Loxahatchee
Micrasterias spp.	69	9	381	
Microchaete spp.	2	1	627	Pal Mar
Mougeotia large morph	10	3	408	
Mougeotia small morph	16	9	504	
Mougeotia spp.	2	0	133	
Navicula constans	1	1	1759	
Navicula cryptocephala	2	0	535	
Navicula cryptotenella	38	2	332	
Navicula palestinae	2	19	572	
Navicula radiosa	19	1	302	WCA 2A
Navicula radiosafallax	18	1	266	
Navicula spp.	1	5	1759	
Navicula subtilissima	31	1	306	WCA 2A
Nitzschia amphibia	16	9	1049	Okeechobee
Nitzschia nana	6	2	1329	Okeechobee
Nitzschia lacunarum	31	3	585	
Nitzschia palea var. *debilis*	119	19	383	
Nitzschia serpentiraphe	45	3	190	Loxahatchee
Nitzschia spp.	8	7	775	
Oedogonium (large)	16	3	302	
Oedogonium (small)	50	7	429	
Onychonema spp.	6	3	419	Pal Mar
Oocystis spp.	27	2	281	
Oscillatoria spp.	8	4	581	
Palmodictyon spp.	1	0	324	
Pediastrum tetras	3	0	156	
Peridinium spp.	63	2	261	
Pinnularia gibba	5	1	1193	
Pinnularia spp.	15	1	784	
Pinnularia microstauron	4	1	964	
Pleurotaenium minutum var. *attenuatum*	73	6	233	
Pleurotaenium minutum var. *excavatum.*	71	8	302	
Pseudostaurosira brevistriata	4	1	561	
Rhabdoderma linearis	134	16	232	
Rhabdoderma sigmoidea	86	10	210	
Rhopalodia gibba	4	1	206	
Rossithidium lineare	2	2	857	
Scenedesmus acutus	0	0	...	
Scenedesmus armatus	2	1	1162	
Scenedesmus brevistriata	2	3	203	WCA 3AN
Scenedesmus serratus	2	1	158	

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Scenedesmus spp.	6	1	209	
Schizothrix spp.	144	58	233	
Scytonema hofmannii morph 1	34	17	118	WCA 3AN
Scytonema hofmannii morph 2	60	15	200	
Scytonema hofmannii morph 3	116	29	241	
Sellaphora laevissima	6	0	212	
Sellaphora pupula	1	2	1759	
Staurastrum connatum	8	1	483	
Staurastrum cyathipes	2	0	446	
Staurastrum excavatum	64	5	426	
Staurastrum longebrachiatum	15	1	504	Holeyland
Staurastrum ophiurum f. *cambriatum*	3	0	534	
Stauroneis phoenicenteron	4	1	478	
Staurastrum cf. *sonthalium*	2	0	170	
Staurastrum spp.	10	2	321	Okeechobee
Stenopterobia curvula	4	0	527	Pal Mar
Stigeoclonium spp.	2	1	489	
Stipiticoccus spp.	75	3	274	
Teilingia spp.	7	4	416	
Tetraedron caudatum	2	0	671	WCA 2A
Tetraedron pentaedricum	4	1	335	
Tetraedron spp.	22	1	336	
Thalassiosira spp.	3	1	1013	
Triploceras spp.	3	2	165	

Table 2. List of the most common diatom taxa in samples from the Fall 2005 survey, with the number of sites present (of 148), their maximum relative abundance, total phosphorus optimum (in µg g^{-1} periphyton dry weight, calculated by weighted averaging) and the location for taxa significantly associated with a particular sampling unit.

	Freq.	Max	TP Opt.	Location
Achnanthes caledonica L-B	38	12	408	
Achnanthes cf. *minutissima* var. *gracillima* (Meist.) L-B	5	3	242	
Achnanthidium minutissimum (Kütz.) Czar.	5	2	768	
Achnanthidium minutissimum (Kütz.) Czar. morph. 1	8	2	383	
Amphora copulata (Kütz.) Schoen. & Arch.	5	1	341	
Amphora pseudoproteus	1	1	506	
Amphora sulcata Bréb.	16	45	332	Holeyland
Amphipleura pellucida Kütz.	3	1	518	
Bacillaria paxillifer (O. F. Mull.) Hendey	2	1	1243	
Brachysira aponina Kütz.	5	1	215	
Brachysira brebissonii Ross in Hartley	29	64	495	Okeechobee
Brachysira neoexilis Lange-Bertalot	136	45	318	
Brachysira procera Lange-Bertalot & Moser	44	5	270	
Brachysira pseudoexillis Lange-Bertalot & Gerd Moser	68	13	137	
Brachysira vitrea (Grun.) Ross in Hartley	28	2	441	
Caloneis bacillum (Grun.) Cl.	2	4	1597	
Caponea caribbea Podzorski	3	1	420	
Cocconeis placentula var. *euglypta* (Her.) Grun.	4	1	154	
Craticula cuspidata (Kütz.) Mann	2	1	1195	
Cyclotella meneghiniana Kütz.	48	2	400	
Diadesmis confervacea (Kütz.) Grun. in Van Heurck	2	0	1243	
Diploneis oblongella (Naegelii ex. Kütz.) R. Ross	21	1	255	
Diploneis parma Cleve	69	27	842	
Diploneis elliptica (Kütz.) Cl.	3	3	761	
Encyonopsis sp.	25	47	593	Holeyland
Encyonopsis microcephala (Grun) Krammer	83	36	482	Pal Mar
Encyonema sp. 1	2	1	155	
Encyonema evergladianum Krammer	123	68	210	
Encyonema sp. 2	130	53	459	WCA 2A
Encyonema sp. 3	2	1	66	
Encyonema silesiacum var. *elegans*	54	21	520	Okeechobee
Encyonema silesiacum (Bleisch *ex* Rab.) Mann	10	2	592	
Encyonema sp.	11	2	244	
Eunotia camelus Ehr.	12	2	983	Okeechobee
Eunotia sp. 1	14	39	404	
Eunotia flexuosa Bréb. ex. Kütz.	35	11	951	Okeechobee
Eunotia incisa W. Smith ex. Gregory	20	3	770	Pal Mar
Eunotia monodon Ehr.	6	2	698	Pal Mar
Eunotia naegelii Migula	30	27	1192	Okeechobee
Eunotia rabenhorstiana v. *elongata* (Patrick) Metz. & L-B	7	2	697	Holeyland
Fragilaria capucina Desm.	7	1	955	Okeechobee
Fragilaria nanana Lange-Bertalot	46	8	490	
Fragilaria synegrotesca Lange-Bertalot	132	44	332	
Fragilaria ulna [danica complex] (Nitz.) Lange-Bertalot	6	2	744	
Fragilaria ulna (Nitz.) Lange-Bertalot	6	5	698	

Fragilariforma virescens v. *capitata* (Ralfs) Williams & Round	17	1	297	Loxahatchee
Frustulia rhomboides (Her.) de Toni	23	23	505	Okeechobee
Frustulia crassinervia (Bréb.) Lange- Bertalot & Krammer	7	2	448	
Gomphonema affine Kütz.	5	4	1113	
Gomphonema auritum Braun	5	4	1311	
Gomphonema coronatum Ehr.	2	1	480	
Gomphonema gracile Ehrenberg emend. Van Heurck	18	4	613	
Gomphonema intricatum var. *vibrio* Ehr. (Cl.)	106	9	481	
Gomphonema maclaughlinii Reich.	3	1	726	
Gomphonema parvulum var. *lagenula* L.B. & Reich.	5	2	834	
Gomphonema vibrioides Reichardt et Lange-Bertalot	24	2	368	
Mastogloia braunii Grun.	1	2	506	
Mastogloia lanceolata Thwaites ex. W. Sm.	8	6	397	
Mastogloia smithii var. *lacustris* Grunow	19	58	330	
Mastogloia smithii Thwaites ex. W. Sm.	131	88	238	
Navicula angusta Grun.	2	1	636	
Navicula cryptotenella Lange-Bertalot	80	16	481	
Navicula palestinae	1	5	506	
Navicula radiosa Kütz.	65	6	366	
Navicula radiosafallax Lange-Bertalot	21	3	368	
Navicula salinicola Hustedt	1	14	506	
Navicula subtilissima Cl.	52	9	236	
Navicella pusilla (Grun. ex. Schmidt) K.	3	2	294	
Neidium ampliatum (Ehr.) Kramm.	4	0	592	
Nitzschia amphibia var. *amphibia* Grun..	48	17	932	Okeechobee
Nitzschia amphibia var. *frauenfeldii* (Grun.) Lange-Bertalot	2	1	866	
Nitzschia cf. *obtusa* Wm. Sm.	3	4	434	
Nitzschia lacunarum Hustedt in A. Schmidt et al.	3	1	469	
Nitzschia nana Grun. in Van Heurck	17	39	1059	Okeechobee
Nitzschia palea var. *debilis* (Kütz.) Grun.	114	27	248	
Nitzschia palea (Kütz.) W. Sm.	29	6	733	Okeechobee
Nitzschia serpentiraphe Lange-Bertalot	78	31	92	
Pinnularia acrosphaeria W. Sm.	2	5	888	
Pinnularia gibba Ehr.	6	1	296	Okeechobee
Pinnularia gibba Ehr. Morph 2	12	5	1081	
Pinnularia microstauron (Ehr.) Cl.	10	2	762	
Pinnularia stomatophora (Grun.) Cl.	5	1	686	
Pinnularia viridiformis Krammer	13	3	626	Pal Mar
Pseudostaurosira brevistriata (Grun. in V.H.) Williams & Round	4	1	1051	Pal Mar
Rhopalodia gibba (Ehr.) O. Müll.	12	1	891	
Rhopalodia brebissonii Kr.	2	1	838	
Rossithidium lineare (W. Sm.) Round & Bukht.	5	2	591	
Sellaphora laevissima (Kütz.) Mann	18	1	179	
Sellaphora pupula (Kütz.) Mereschk	8	3	1051	
Stenopterobia curvula (W. Sm.) Kr.	7	2	659	
Stauroneis javanica (Grun.) Cl.	2	1	595	
Stauroneis phoenicenteron (Nitz.) Ehr.	13	6	544	
Terpsinoe sp.	1	1	570	

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Figure 1. Photos showing the different types of periphyton mats categorized in this study: A) calcareous floating mat, B) calcareous epiphytic mat, C) calcareous benthic mat and D) filamentous green periphyton.

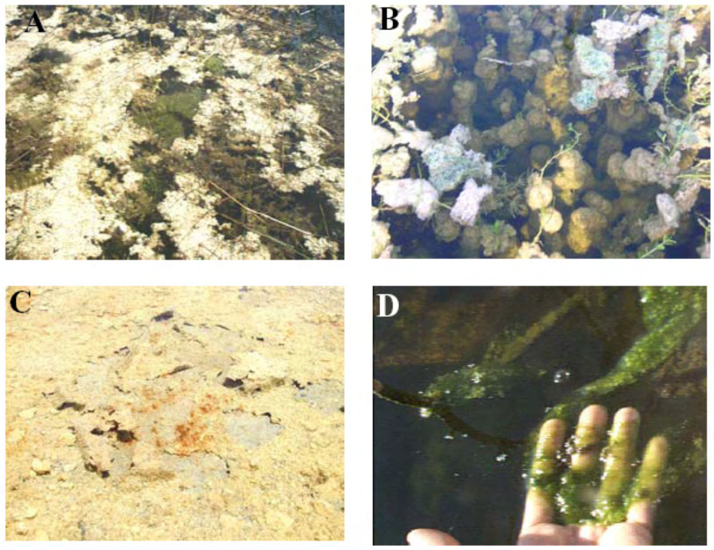

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Figure 2. Patterns in periphyton cover categorized by substrate type among the different localities sampled in the dry and wet seasons of 2005 and 2006.

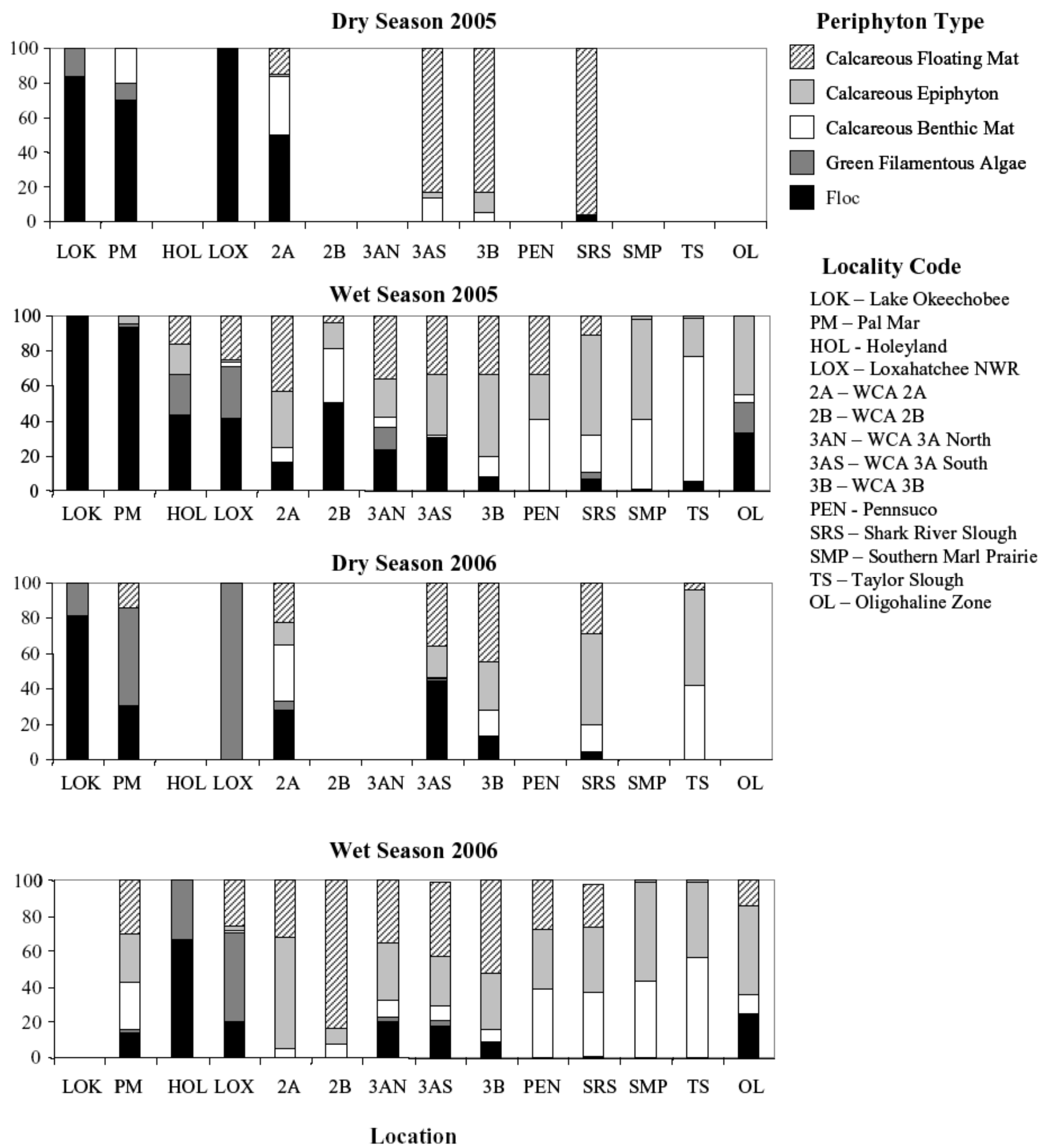

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Figure 3. Patterns in field-measured variables including water depth, pH, soil depth and periphyton volume and cover in the dry and wet seasons of 2005 and 2006. See Figure 1 for locality codes.

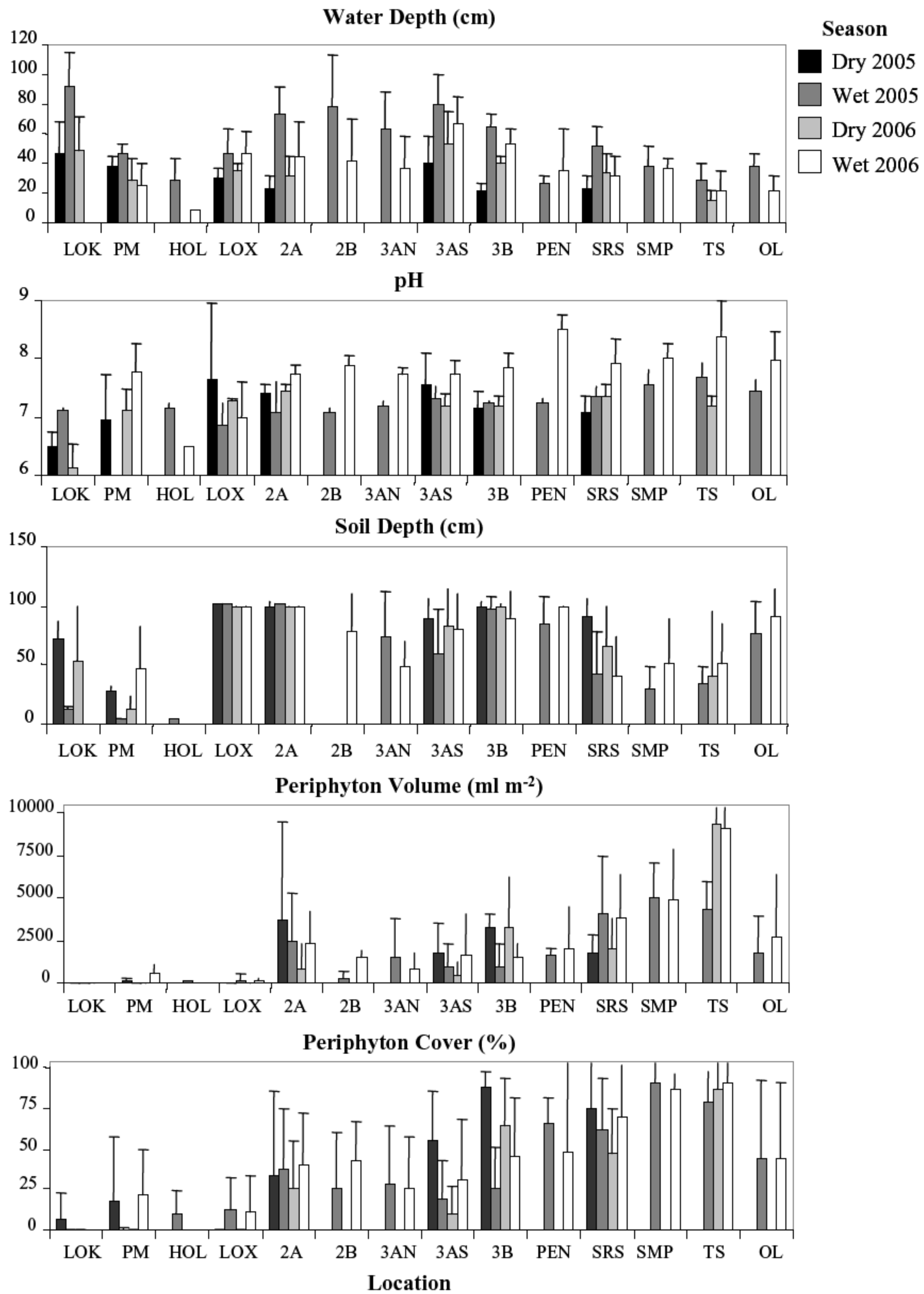

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Figure 4. Patterns in periphyton total phosphorus content, biomass, organic and chlorophyll a content among locations in the dry and wet seaons of 2005 and 2006. See Figure 1 for locality codes.

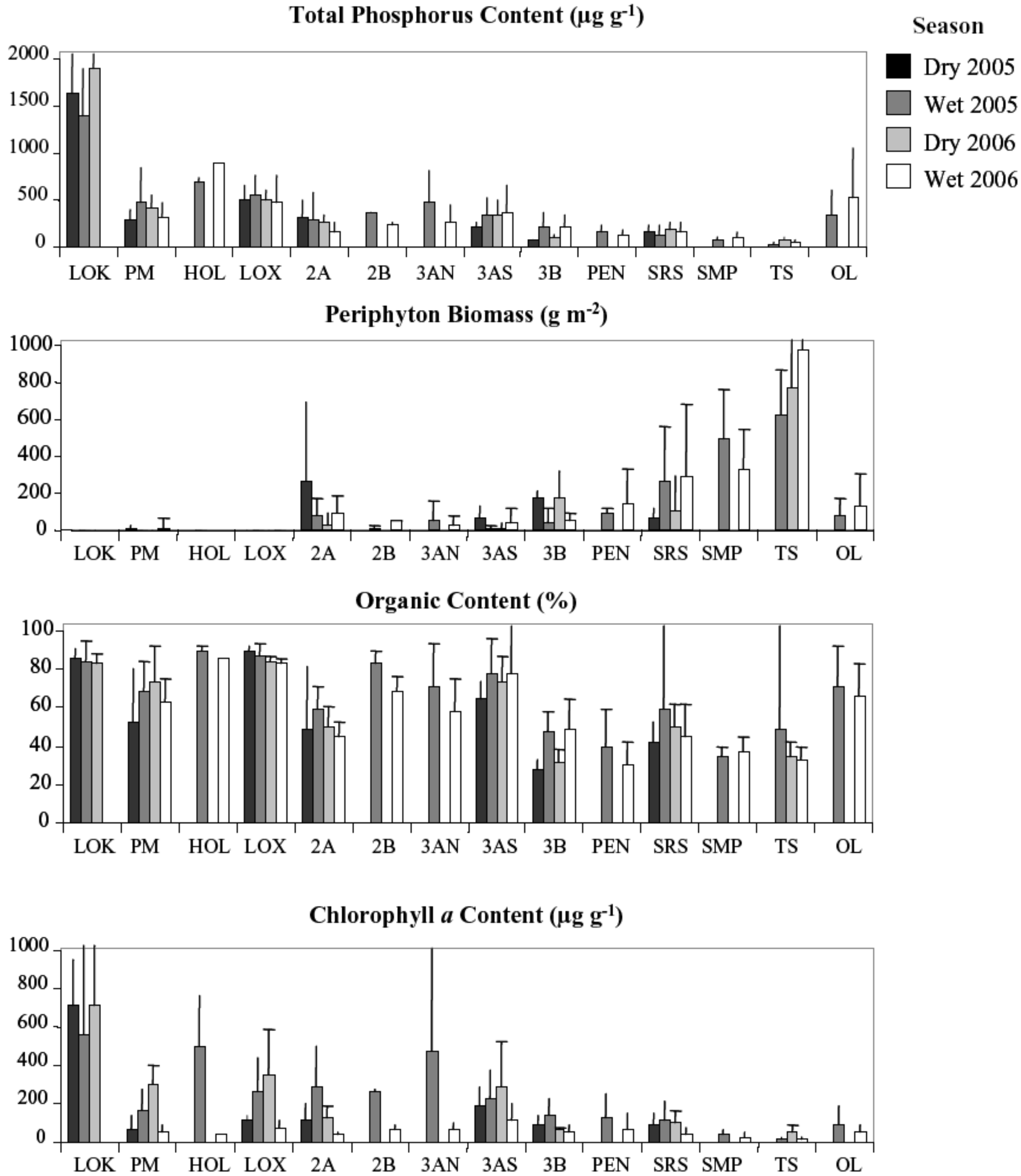

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Figure 5. Relationships of periphyton total phosphorus to cover, biovolume, biomass, water depth and periphyton organic and chlorophyll a content averaged within PSU's during the dry and wet seasons of 2005 and 2006.

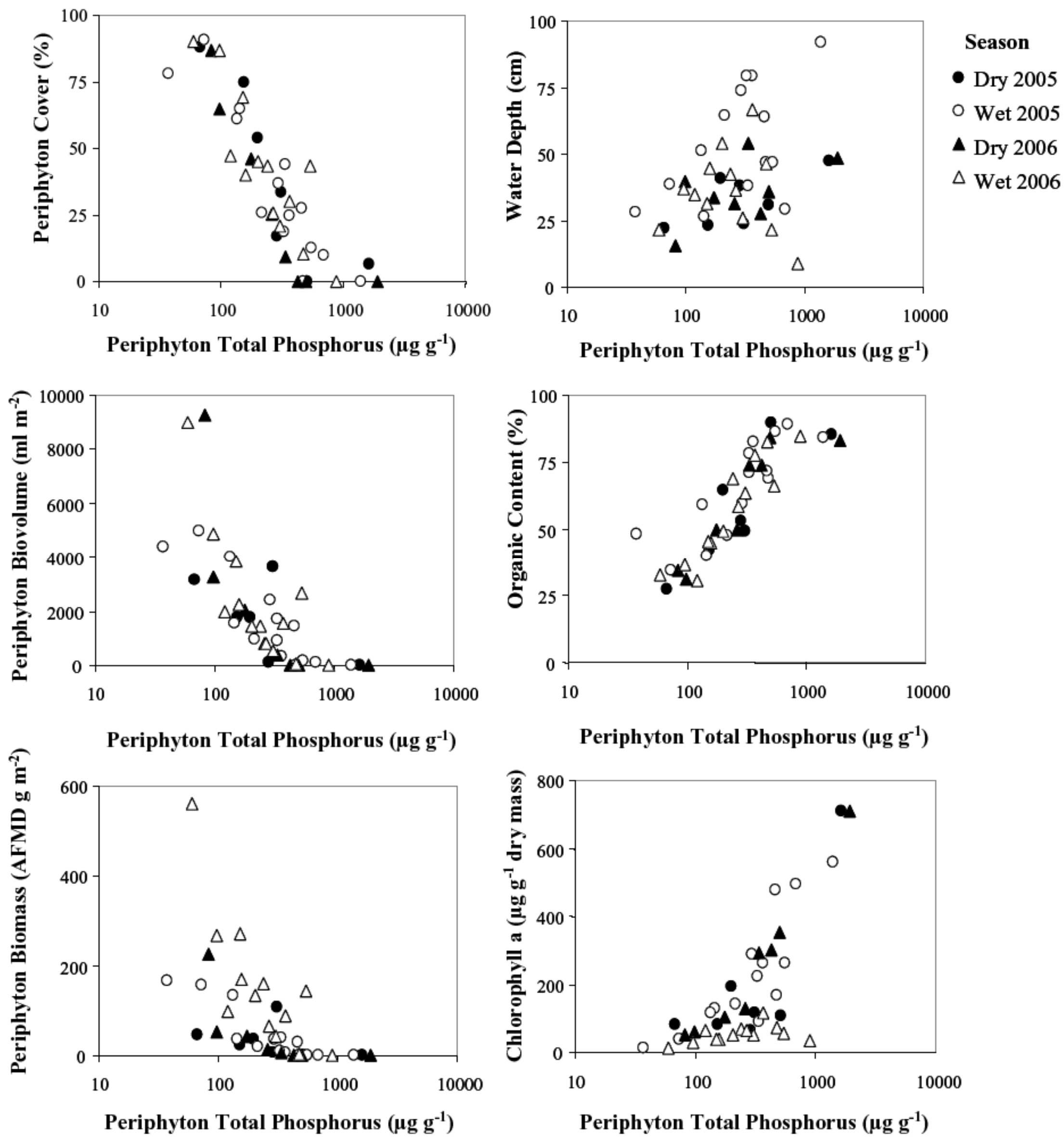

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Figure 6. Non-metric Multidimensional Scaling ordination biplots of based on compositional similarity among all algae and diatoms-only. Site scores are color coded by site and vectors reflect the plane of maximum correlation for variables significantly associated with compositional similarity.

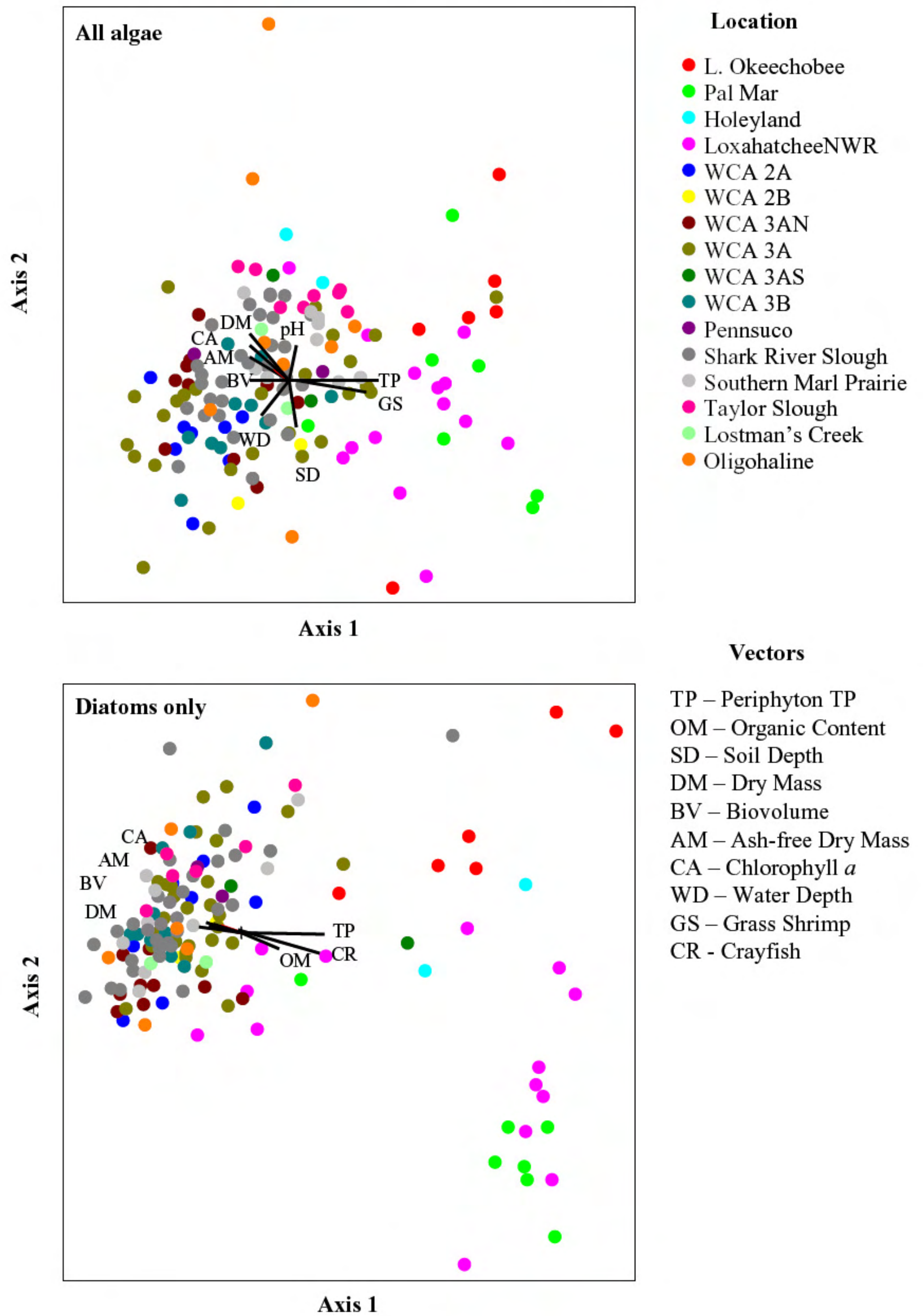

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Figure 7. Relationships of periphyton dry mass, organic content, water depth and total phorphorus content to relative abundance of cyanophytes, chlorophytes and diatoms inhabiting the periphyton.

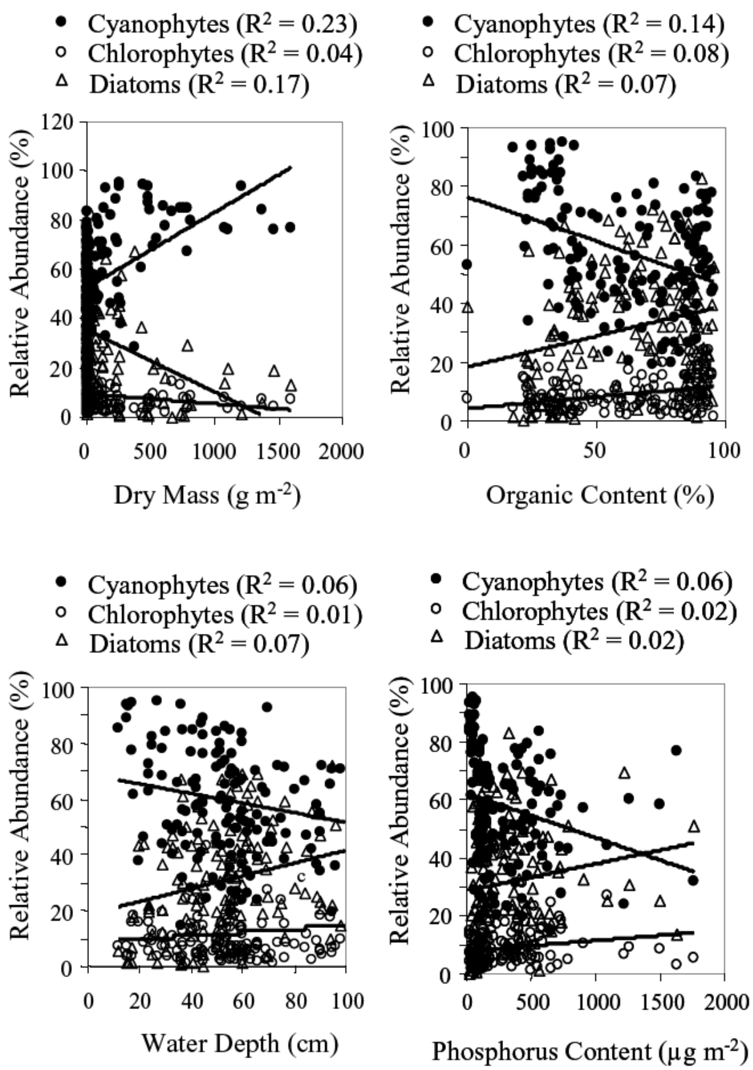

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Figure 8. Relationships of periphyton dry mass and organic content to the abundance of different consumer groups among sites in the Fall 2005 survey.

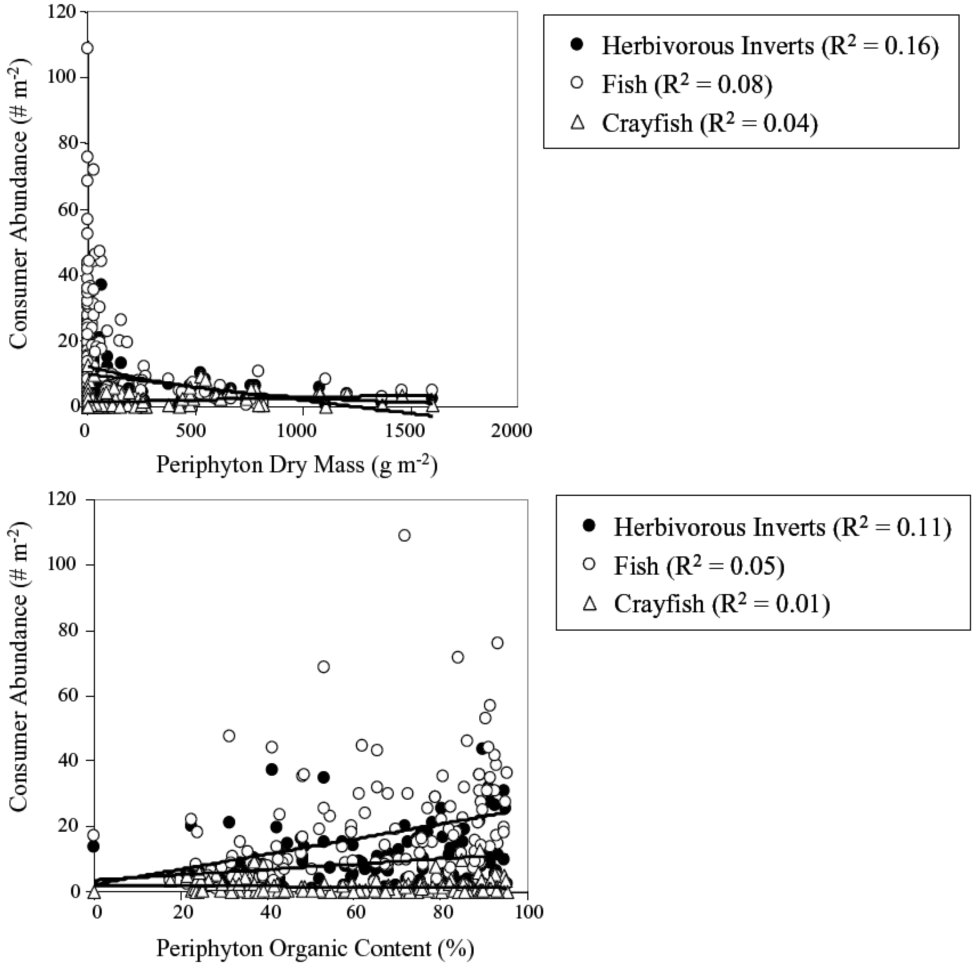

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Figure 9. Relationships of the abundance of different algal groups (filamentous cyanobacteria, desmid algae and diatoms) to the abundance of consumers (grass shrimp, crayfish, herbivorous invertebrates and fish) among sites in the Fall 2005 survey.

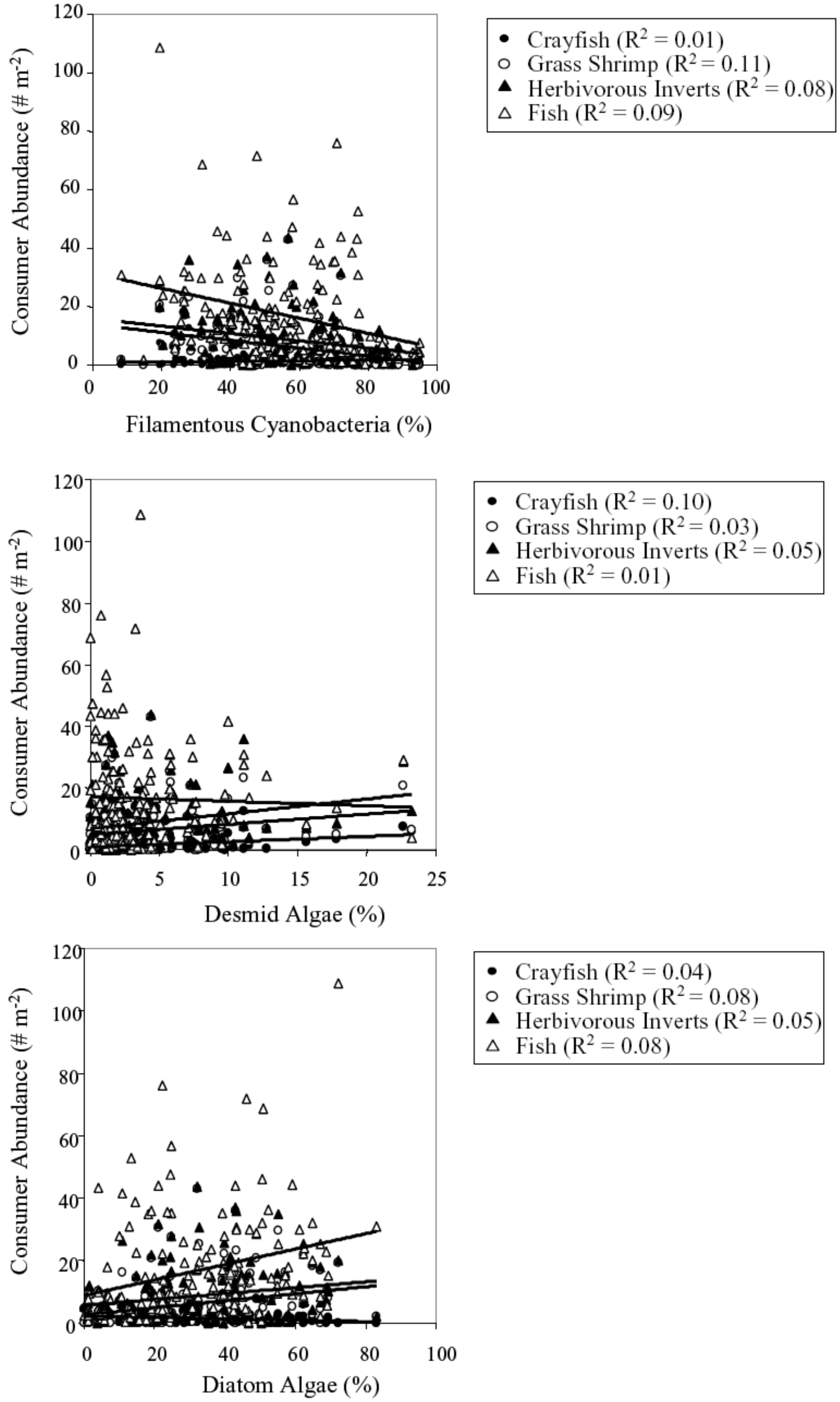

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Figure 10. Periphyton data are summarized using target stoplight colors to illustrate how the performance measures relate to stoplight colors by location.

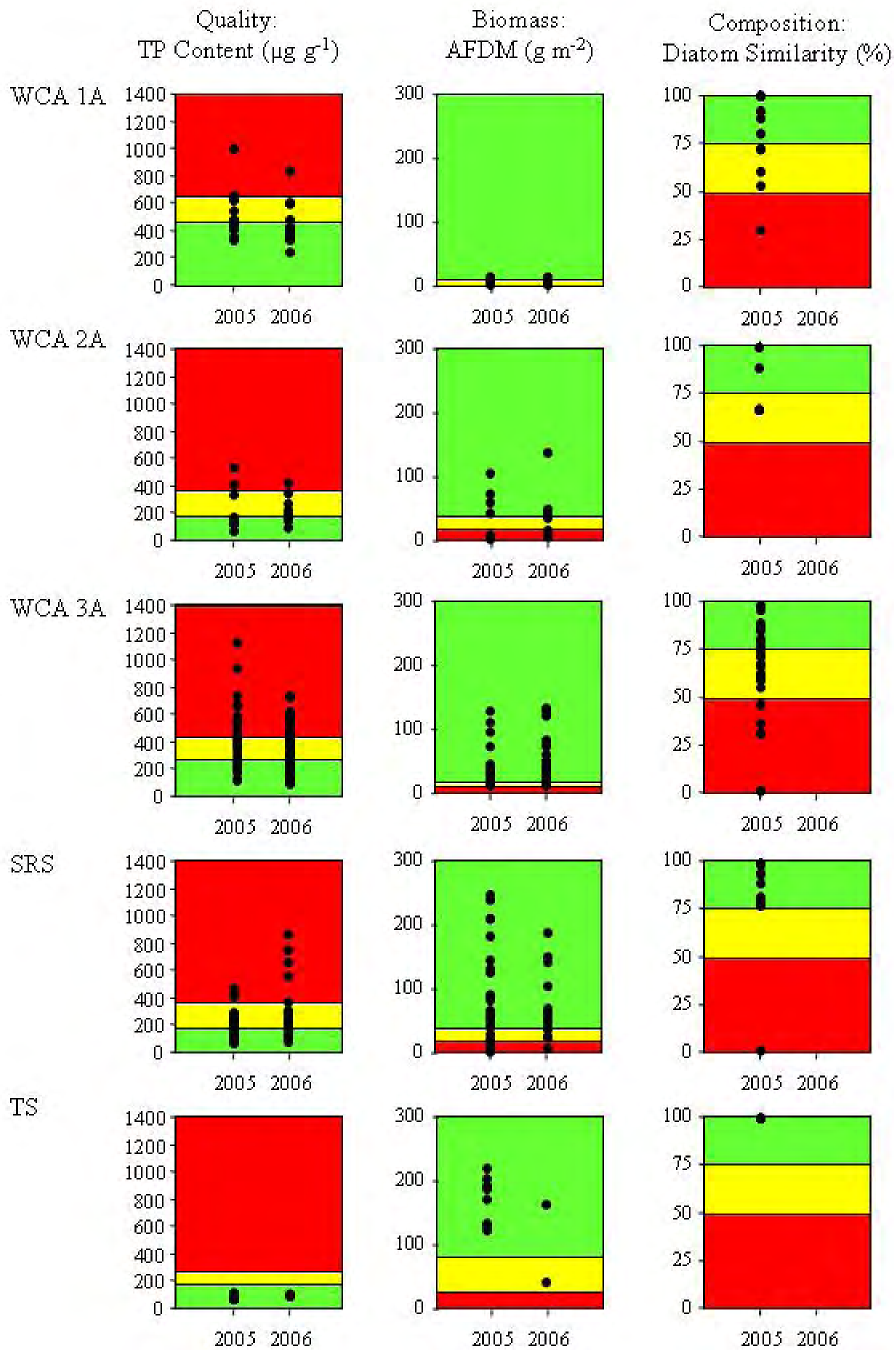

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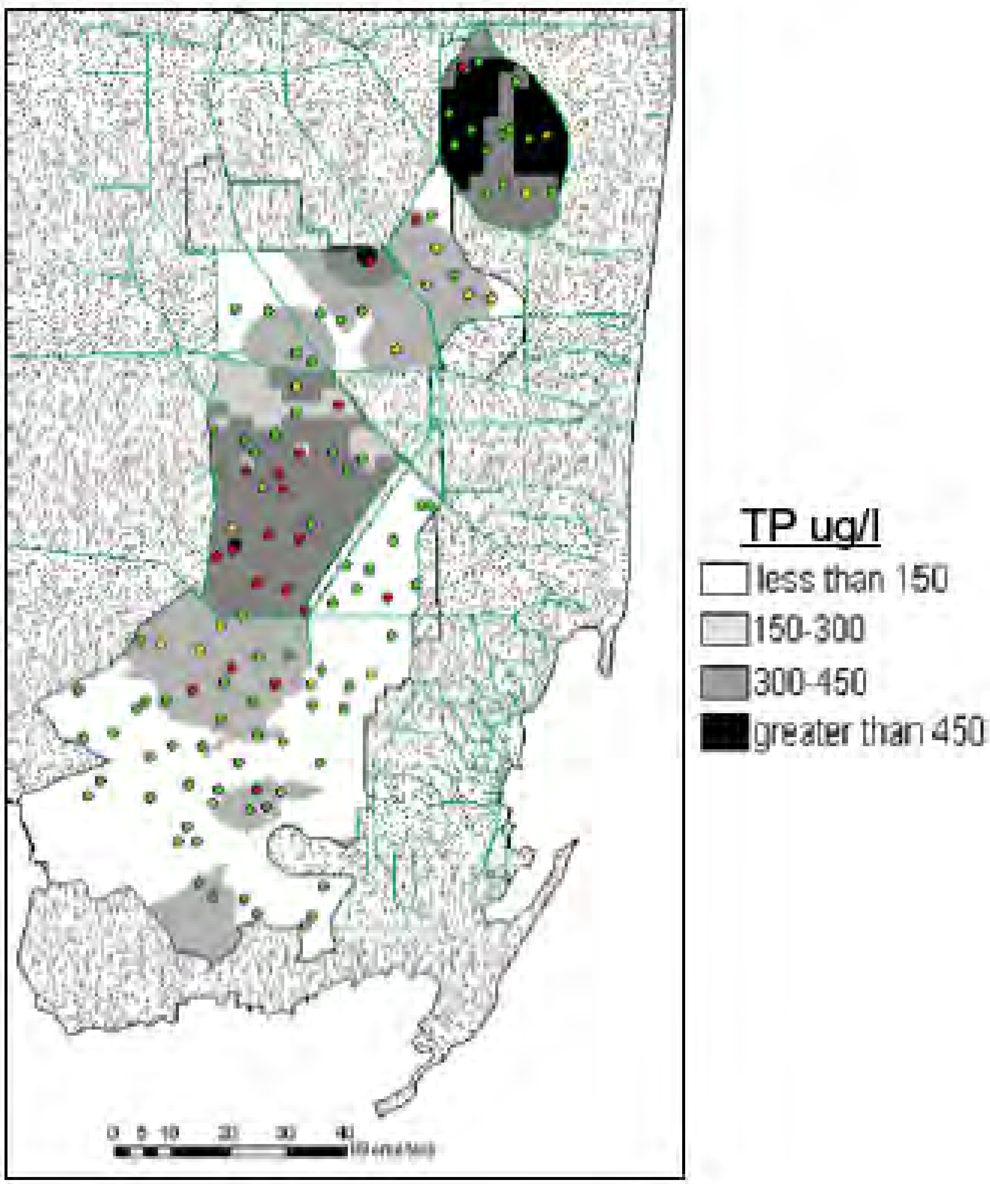
Figure 11. Map showing periphyton performance by area using stoplight coded circles.



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2008 Assessment Update

Pink Shrimp Indicator for the Southern Estuaries (SE)

Supported by
MAP activities 3.2.3.5 and 3.2.4.5

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U.S. Geological Survey
Center for Water and Restoration Studies

Joan A. Browder

NOAA Fisheries Service
Southeast Fisheries Science Center

July 2008

Background

The pink shrimp, *Farfantepenaeus duorarum*, was chosen as a faunal indicator of the ecosystem status of southern estuaries because the species is abundantly represented, wide spread and is economically as well as ecologically important. Furthermore, it has been extensively studied in South Florida with previous work suggesting relationships with salinity, and long data series are available for some locations to help establish targets. South Florida's southernmost estuaries—Biscayne Bay, Florida Bay, and the mangrove estuaries of the lower southwest coast—provide critical nursery habitat for many species of fish and invertebrates that support ecological food webs and fisheries. Salinity patterns influence their habitat quality.

Salinity patterns are established and maintained by the volume and timing of freshwater inputs, controlled not only by weather but also by water management. Previous structural and operational changes in South Florida's water management system have altered salinity patterns and salinity fluctuations in the southern estuaries, deteriorating the quality and spatial extent of nursery habitat for pink shrimp and other species.

For the southern estuaries, ecosystem restoration in the Comprehensive Everglades Restoration Program (CERP) calls for a reduction in the frequency, intensity, duration, and spatial extent of hypersaline events, as well as a reduction in the frequency and intensity of low-salinity pulses. Achieving these restoration objectives is expected to lead to overall increases in densities of juvenile pink shrimp with subsequent benefits to higher trophic levels and the Tortugas fishery, although local abundances may decrease at the lower end of the salinity gradient. A laboratory study on young pink shrimp from Florida Bay indicated a wide salinity tolerance range with a broad optimum in the midrange (Browder et al. 2002). Both extreme high and extreme low salinities depress pink shrimp survival.

Hypotheses in the CERP Southern Estuaries Module that apply to pink shrimp (RECOVER 2007) are summarized as follows: Reestablishing a relatively persistent positive salinity gradient through CERP will increase the area of overlap of favorable salinities with favorable bottom habitat (especially SAV) and shoreline features, thereby increasing the distribution and abundance of species characteristic of estuaries, including pink shrimp. A positive salinity gradient is one in which salinities are lowest nearest to shore. The specific details of optimum habitat will differ by species, however it is inherent in this hypothesis that restoring a broad and positive salinity gradient will optimize the spatial extent of high quality habitat for many characteristic species.

Data Support for the Pink Shrimp Indicator

Under the Monitoring and Assessment Program (MAP) for the Comprehensive Everglades Restoration Program (CERP), the Fish and Invertebrate Assessment Network (FIAN) Project (MAP activities 3.2.3.5 and 3.2.4.5) currently samples pink shrimp and other epibenthic fauna at 19 locations within three regions: Florida Bay, Biscayne Bay, and the southwest mangrove coast estuaries, including Whitewater Bay. Collections are made biannually at the end of the dry (April-May) and wet seasons (September-October). A 1-m^2 throw-trap is used in shallow, open water seagrass habitats to quantify abundance metrics for pink shrimp and other species associated with bottom vegetation. At each monitoring location, sampling occurs within a randomly sited grid of 30 equal-sized, tessellated hexagonal cells. A suite of habitat measurements is collected in conjunction with each faunal sample, including salinity, temperature, water depth, and turbidity, as well as biological habitat features such as taxonomic composition and coverage of bottom vegetation. For each collection, one throw-trap sample is taken at a randomly located site within each grid-cell at each monitoring location, resulting in a total of 570 samples collected each season. This sampling design ensures that sampling encompasses gradients of environment and habitat present in each monitoring location. Additional details of sampling design and sampling method are provided in Robblee and Browder (2007, 2008). Prior use of the throw-trap method in the study domain dates back to 1984, providing considerable historical data to help establish pre-CERP pink shrimp abundance, variability, assessment targets and trends.

Analysis Framework for Assessment Using the Pink Shrimp Indicator

The pink shrimp performance measure compares annual mean spring and fall pink shrimp density and variance estimated in FIAN with the historical period-of-record (Table 1) in six response areas (Figure 1), each encompassing one to two FIAN monitoring locations: Whitewater Bay, Johnson Key Basin (JKB), South-Central Florida Bay, North-Central Florida Bay, Eastern Florida Bay, and South Biscayne Bay (Figure 1). Because of frequency of use, throughout this document these regions are referred to as Whitewater, JKB, South-Central FB, North-Central FB, Eastern FB, and Biscayne. The JKB and Biscayne areas have substantial historical periods-of-record, ≈20 yrs and ≈6 yrs, respectively, but the period-of-record of available historical data consists of 2 or fewer years for the other four response areas; Eastern FB, North-Central FB, South-Central FB and Whitewater. The bases for establishing targets will strengthen with time as the MAP time series, now 3-years long, are lengthened. Available historical throw-trap data have been summarized as spring and fall density and variance estimates for each response area (Table 2); citations for these data are in footnotes to Table 1.

The delta-approach was applied to calculate pink shrimp density to overcome a common problem of faunal sampling, many zeros in the data due to absence of the species of interest in individual samples. Simply put, the delta approach calculates the mean density as the product of occurrence and concentration. Occurrence is the percent of samples in which the species occurs, and concentration is the mean of the samples in which the species occurs. If the concentration data are not normally distributed (which often is the case), the data are transformed (usually log transformed) to better approximate a normal distribution before calculating the concentration mean. Few of the pink shrimp datasets used in the assessment were normally distributed (based on Kolmogorov-Smirnov Distance tests [$p>0.05$]), even after the log transformation, however log transformation strengthened the normality assumption. Of those datasets meeting the normality assumption, three were from JKB, one was from Biscayne, and all were data sets for the fall season, when pink shrimp were most abundant in south Florida estuaries. Log transformation of the concentration data for calculation of delta-density reduced variability, narrowed confidence intervals, and improved detection of differences. The gain from using the delta-density approach was most notable in the fall. Density, rather than occurrence or concentration, was the metric used in the assessment.

The current status of pink shrimp abundance was determined by comparison with quartiles of the distribution of historical spring and fall mean delta-densities over the available period-of-record. Mean pink shrimp densities less than the 25th quartile were scored as zero (poor), values between the 25th and 75th quartile received a score of 0.5 (neutral), and values equal to or exceeding the 75th quartile received a score of 1 (good). Status was determined for spring and fall of 2005, 2006, and 2007. The current assessment relates to 2007. The goal is to maintain, in each region for each season (dry and wet seasons), an annual observed pink shrimp density equivalent to good or a positive trend in density in that season. This goal recognizes that the greatest density attainable differs by assessment region.

Pink Shrimp Indicator Status

In Figures 3 and 4, zones are represented by red, yellow and green, respectively. The quartile density values defining the boundaries between zones are shown in Table 2. These boundaries represent thresholds across which change occurs from poor to neutral, neutral to good, and the converse. In Figures 3 and 4, solid black circles represent mean density values for each MAP year-season and the historical record. Confidence limits (CLs) around the mean density value can be used to determine whether the mean density of the current year is significantly different from that of previous MAP years or the mean of the historical record. Although density was the basis for the assessment, for added perspective, the same statistics (e.g., quartiles, means, confidence limits) for occurrence and concentration also were shown in Figures 3 and 4.

In the six response areas relatively few differences, based on overlap of 95% CL's, in occurrence, concentration or delta-density were observed between FIAN and period-of-record historical pink shrimp mean and variance. Differences were observed primarily in the fall when shrimp were most abundant (Figures 3 and 4). The short historical record in most locations (except JKB and to a lesser extent Biscayne) may in part account for these results. The few years in the period-of record for the other response areas may be too short to represent the weather cycles that affect freshwater runoff and estuarine salinities in South Florida and may not yield reliable thresholds for detecting change in pink shrimp status.

Pink shrimp did well in 2005, with average scores among the six response areas of 0.7 and 0.6, spring and fall, respectively (Table 3). In contrast, 2007 was an extremely poor year with scores among the six response areas averaging only 0.2 and 0.2, spring and fall, respectively. The year 2006 was intermediate, averaging 0.6 and 0.4, spring and fall, respectively. In Johnson Key Basin the fall delta-density of 5.2 shrimp/m^2 (FIAN methods estimated 5.8/m^2) was the fourth lowest in a 20-year historical period-of-record. In 2007 poor pink shrimp status was noted everywhere but Johnson Key Basin in the spring and everywhere but South Biscayne in the fall. Details of assigning scores and associating them with green, yellow, and red stoplights were given in Browder and Robblee (in review).

Reflecting the scores in Table 3, the status of the pink shrimp indicator in 2005, 2006, and 2007 is portrayed in stoplight format in Figure 5 with a time-series of colored circles for each response area. The number of green circles decreases and the number of red circles increases from 2005 to 2007. The geographic distribution of green, yellow, and red circles can be seen in Figures 6 (spring) and 7 (fall).

Interpretation and Evaluation of the Assessment

The basis for the high number of red circles in 2007 needs further investigation with regard to causality. Inshore salinity conditions and/or offshore spawning success may account for the poor showing of pink shrimp in 2007, especially in Florida Bay. Salinity patterns characterizing the six response areas differ, reflecting freshwater inflows and mixing with Gulf of Mexico and Atlantic waters (Figure 8). In South Florida, with distinct dry and wet seasons, it is typical for salinities to increase approaching April and May, and decline approaching September and October. The salinity gradient may facilitate or enhance the movement of postlarval pink shrimp into nearshore nursery grounds, and a positive gradient may be more effective than one that is negative. Hughes (1969a, b) studied movement of postlarval and juvenile pink shrimp in relation to a vertical salinity gradient in a laboratory setting. Although he did not specifically examine the relative effect of positive vs. negative horizontal salinity gradients on pink shrimp movement toward and within estuaries, his results suggest that a negative salinity gradient may not provide the same support. Peak pink shrimp postlarval immigration to western Florida Bay occurs in the fall (Criales et al. 2006). Among the 3 years available for assessment, positive estuarine conditions occurred in the fall of 2005 and 2006 (Figure 5). In contrast, salinities increased through the wet season in JKB and North- and South-Central FB in the fall of 2007, leading to hypersaline conditions in central Florida Bay and a negative salinity gradient. Hypersalinity, which establishes a negative salinity gradient, may have limited pink shrimp immigration into Florida Bay and contributed to the exceptionally low pink shrimp abundance observed in the fall of 2007. A negative salinity gradient in Florida Bay is typical in the spring and may be at least partially responsible for the low abundance of pink shrimp in Florida Bay in the spring relative to the fall.

Weak spawning and postlarval immigration in 2007 relative to 2005 and 2006 is an alternative reason for the low pink shrimp abundance in 2007. With spawning occurring in the vicinity of the Tortugas (Costello and Allen 1966), spawning strength should be best reflected in pink shrimp status in JKB, where immigrating postlarvae first enter the bay. In the absence of other environmental factors such as hypersalinity, pink shrimp status in western Florida Bay should be reflected broadly among the six response areas similar to the pattern observed in 2007. Whenever status is consistent across response areas (e.g., all low or all high), as in 2007, the possibility that offshore spawning strength affects pink shrimp status cannot be discounted. However, the conclusions of Ehrhard and Legault (1999) are relevant to this issue. On the basis of their cohort analysis of fishery catch and effort data, they concluded that environmental factors rather than a spawning stock-recruitment relationship determined year-class-strength.

Lessons Learned

Interpreting pink shrimp status in a response area or among response areas is dependent on the reliability of the historical (reference) condition. Periods-of-record (POR) for historical data sets against which "status" was determined were only 2 years long for four reference areas: Eastern, North-Central, and South-Central FB and Whitewater. Except for JKB, prior-year data are not sufficiently extensive to reliably estimate the 75th percentile of the past distribution of pink shrimp density. The potential of the six response areas to provide a strong basis for assessment of pink shrimp status is evident but more data is needed.

As demonstrated above, assessing several response areas at the same time helps to separate the effect of local conditions from the effect of recruitment success. However, to make such comparisons, it is essential to have strong concurrent time series of data for each response region. The FIAN sampling for MAP is building these time series.

Longer time series of data also are critical to allow assessments to be based on 3-yr running means rather than annual means. Assessments based on annual means are overly sensitive to year-to-year natural variation in rainfall and other climatic conditions. Time series from which to develop thresholds for determining status should be long enough to encompass at least the shorter-term weather cycles. Because of the uniformity of sampling design, MAP data provide a better baseline for assessments of CERP effects than existing historical data, which was collected under various other designs. Early-CERP assessments, based on short time series of MAP data collected before substantive CERP changes are made, should be used to develop robust assessment methods and should be viewed as examples of what could be achieved later with firmer data

The historical data may not be the appropriate basis for establishing thresholds or goals based on quartiles. Natural conditions no longer existed in these estuaries at the time the historical data were collected. Future improvements in the assessment method may include pink shrimp targets developed considering natural conditions but independent of historical data.

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Figure 1. Pink shrimp indicator assessment regions (yellow circles) and the 19 FIAN sampling grid locations (green patches).

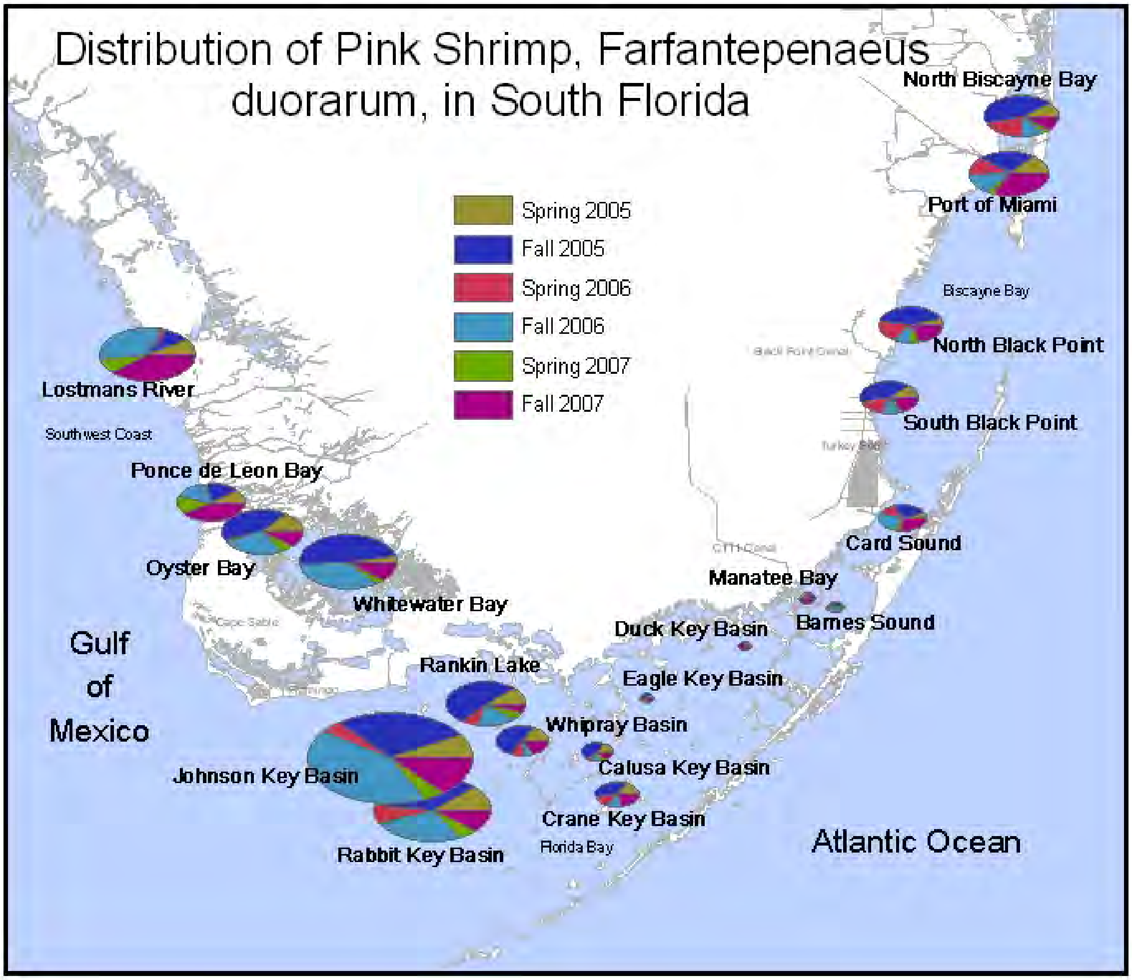

Figure 2. Distribution of the pink shrimp, *Farfantepenaeus duorarum*, distributed among FIAN monitoring locations, Spring 2005 – Fall 2007. The size of the pie represents the sum of pink shrimp for all collections at the monitoring location. The sizes of the slices within each pie represent relative abundance among collections (i.e., season-year).

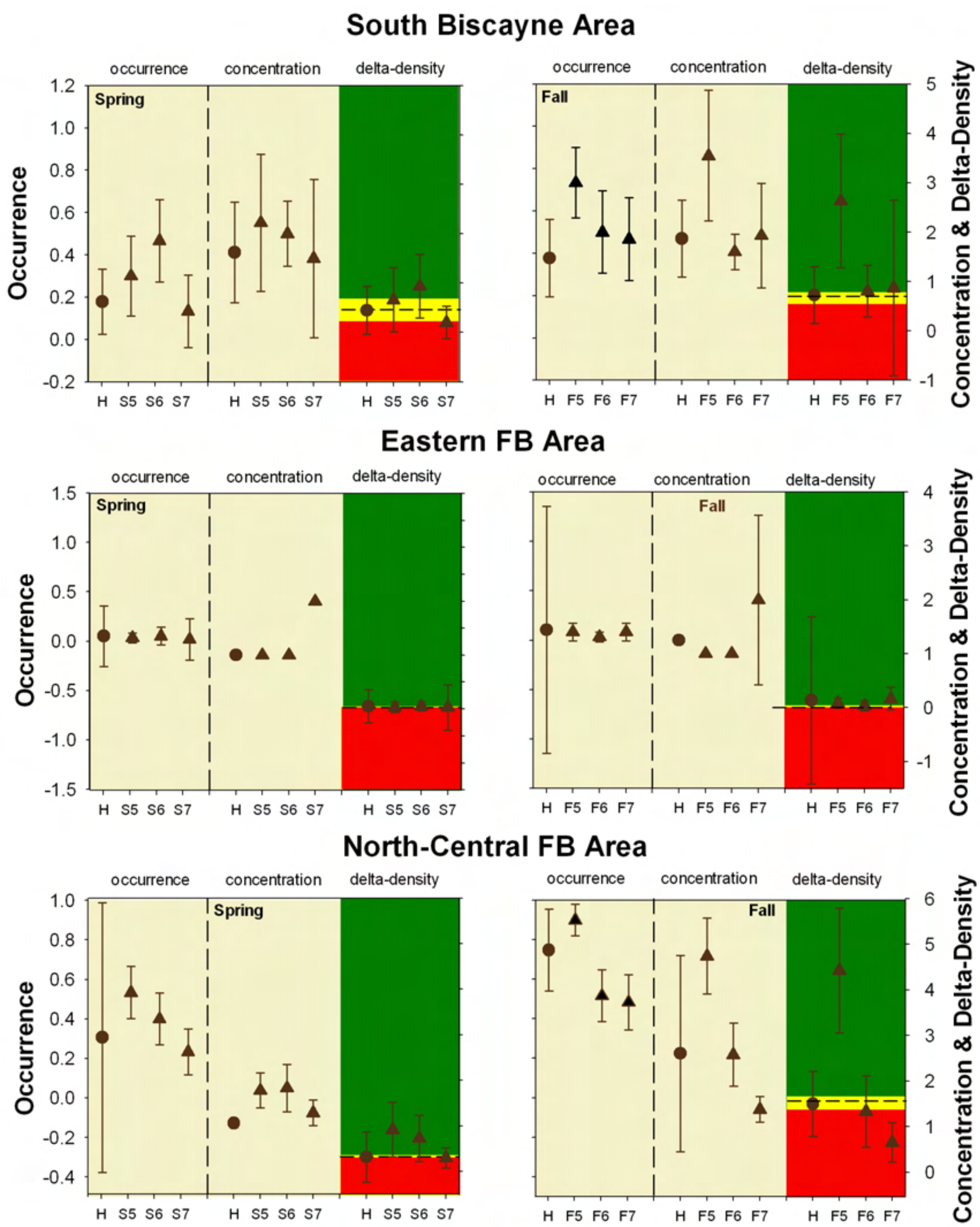

Figure 3. Pink shrimp regional indicator abundance metrics for South Biscayne Bay, Eastern Florida Bay, and North-Central Florida Bay in relation to historical data (H) and indicator targets; Spring 2005-2007 (S5, S6, S7) and Fall 2005-2007 (F5, F6, F7); occurrence (proportion of positive samples), concentration (mean density of positive samples), and delta-density (back-transform of log transformed [ln] concentration x occurrence); indicator targets based on historical data: green > 3rd quartile, yellow >1std and < 3rd quartile, and red < 1st quartile. Solid black symbols indicate means, and vertical bars are 95% confidence intervals.

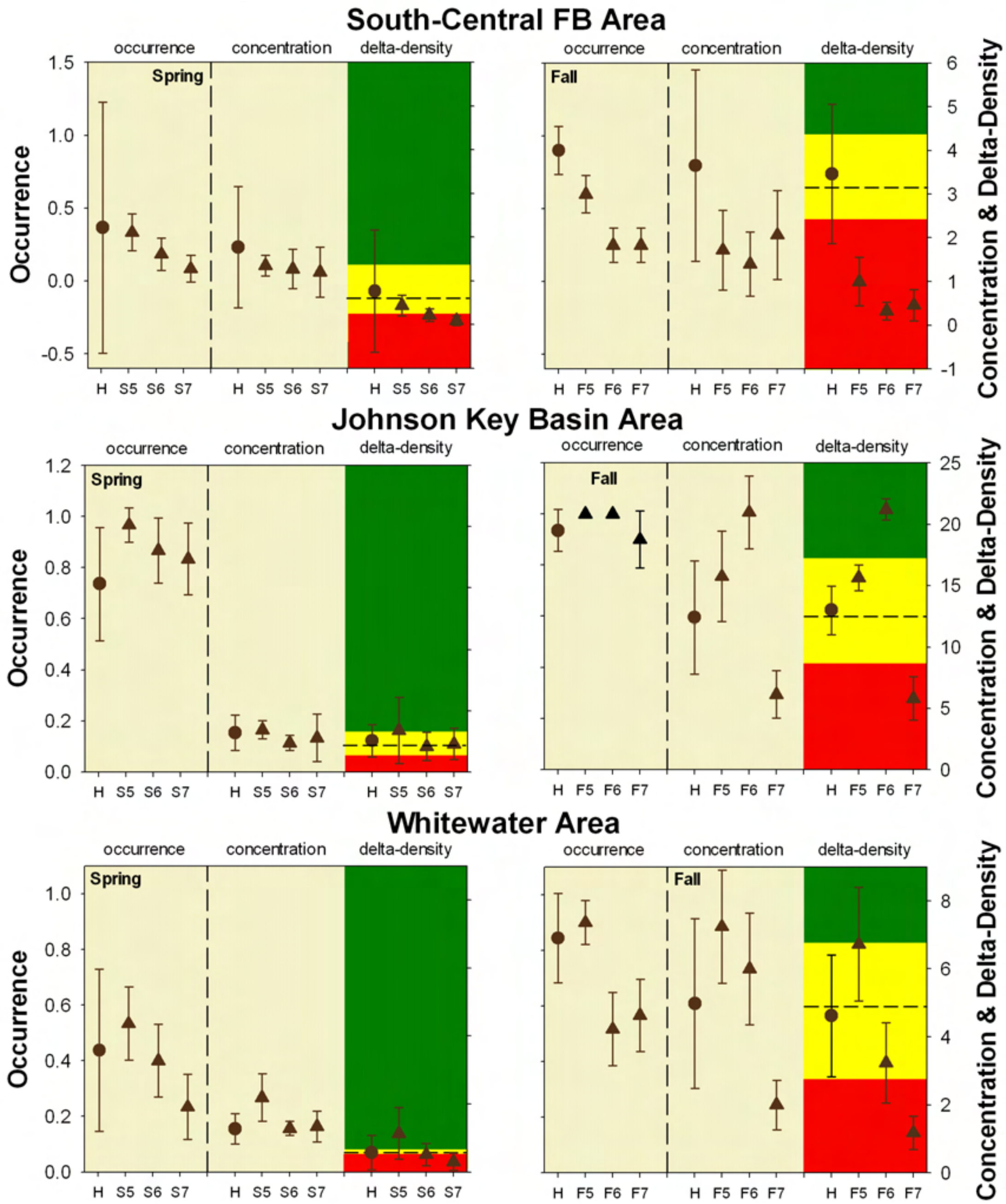

Figure 4. Pink shrimp regional indicator abundance metrics for South-Central Florida Bay, Johnson Key Basin, and the Whitewater Area in relation to historical data (H) and indicator targets; Spring 2005-2007 (S5, S6, S7) and Fall 2005-2007 (F5, F6, F7); occurrence (proportion of positive samples), concentration (mean density of positive samples), and delta-density (back-transform of log transformed [ln] concentration x occurrence); indicator targets based on historical data: green > 3rd quartile, yellow >1std and < 3rd quartile, and red < 1st quartile. Solid black symbols indicate means, and vertical bars are 95% confidence intervals.

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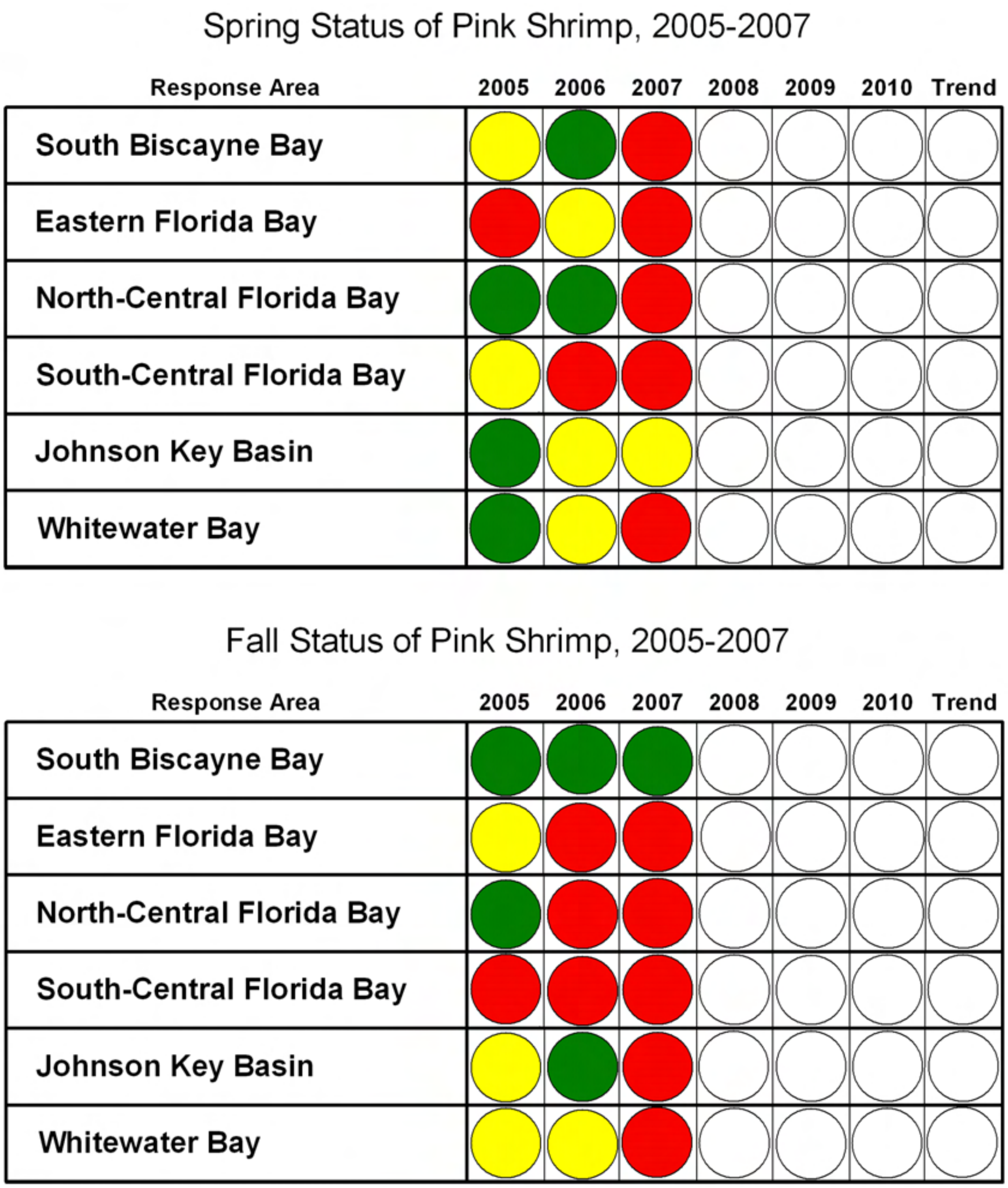

Figure 5. Responses of the pink shrimp indicator, by year and region, spring and fall, based on MAP data (Fish and Invertebrate Assessment Network).

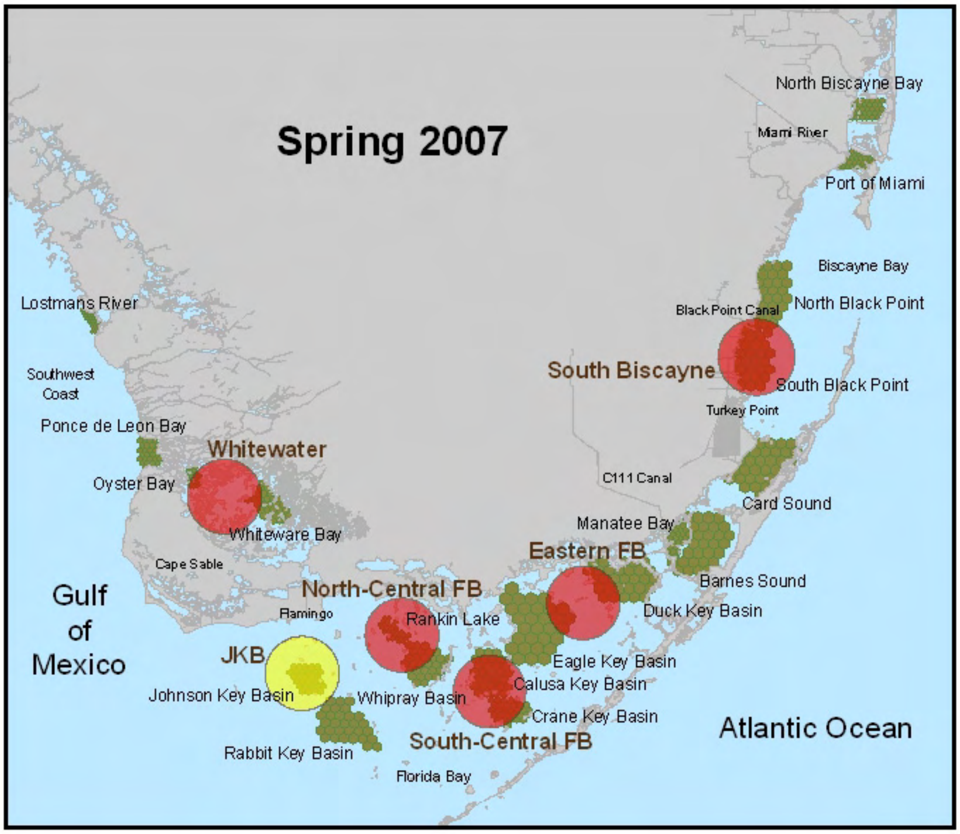

Figure 6. Map of spring 2007 responses of the pink shrimp indicator, by response region.

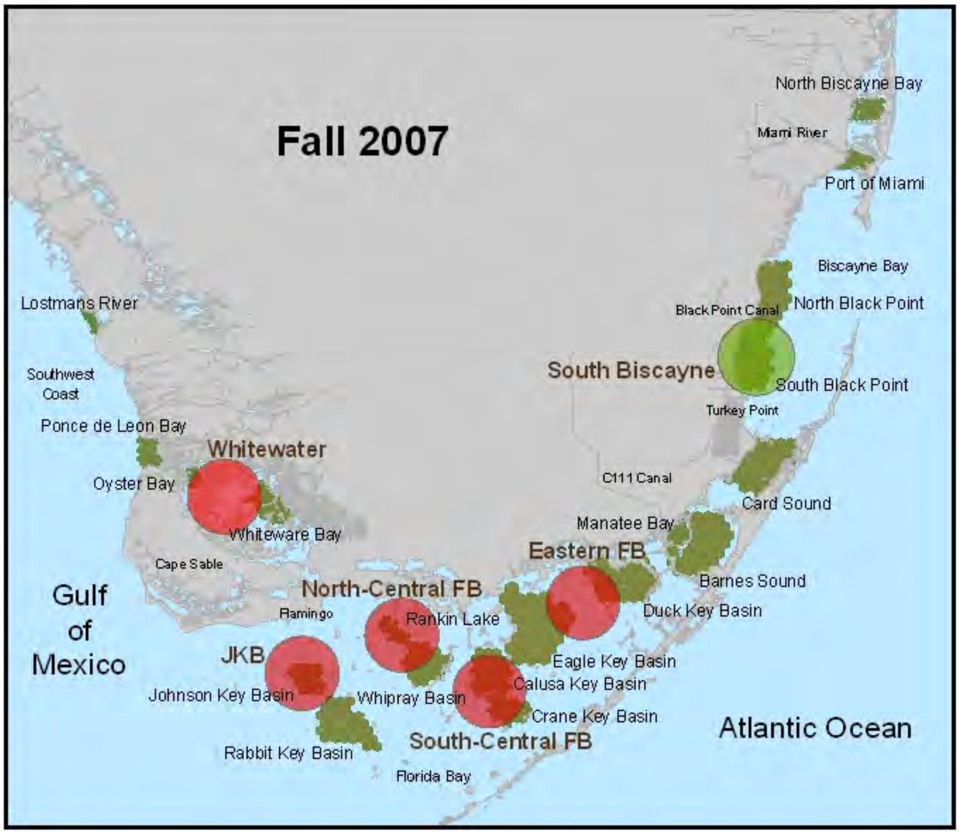

Figure 7. Map of fall 2007 responses of the pink shrimp indicator, by response region.

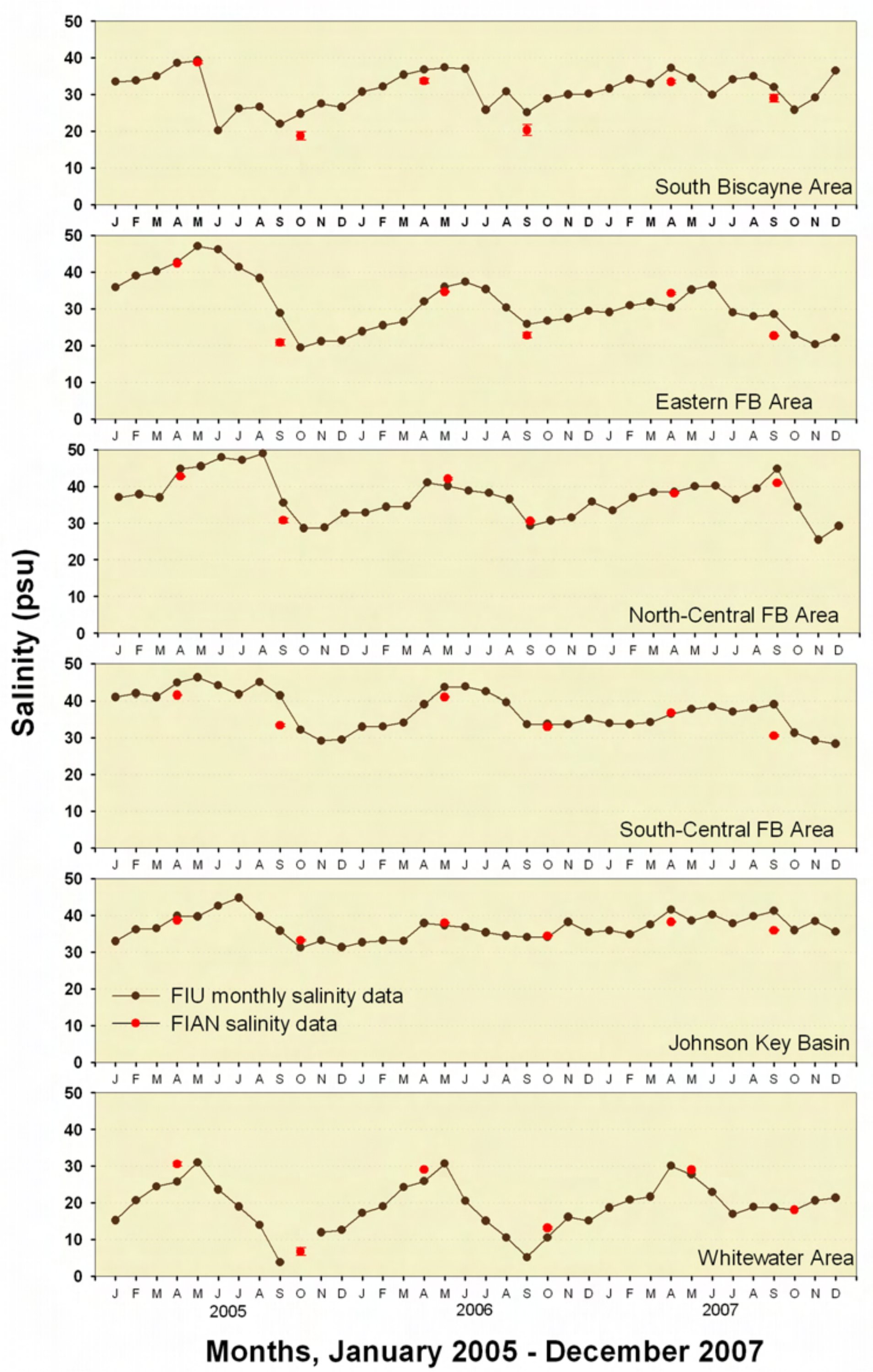

Figure 8. Surface salinity conditions in each pink shrimp response area, 2005 – 2007. Solid black circles are monthly surface salinity measurements from the FIU water quality monitoring network (Boyer and Briceño 2008). Solid red circles are mean and standard error of FIAN salinity measurements, in Practical Salinity Units (psu), from indicated site.

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Table 1. South Florida Fish and Invertebrate Assessment Network (FIAN) monitoring locations used in current assessment and existing historical data (citations to existing data are footnoted).

Location	Existing Data
Biscayne Bay Region	
North Black Point	[1] 2002-2005
South Black Point	[1] 2002-2005
Card Sound	
Barnes Sound	
Manatee Bay	
Florida Bay Region	
Duck Key Basin	[3] 1998-2000
Eagle Key Basin	[5] 1986, [3] 1998-2000
Calusa Key Basin	
Crane Key Basin	
Rankin Lake	[3] 1998-2000
Whipray Basin	[5] 1986, [3] 1998-2000
Johnson Key Basin	[2] 1983-2005
Southwest Coast Region	
Ponce de Leon Bay	[4] 1995-1996
Oyster Bay	[4] 1995-1996
Whitewater Bay	[4] 1995-1996

[1] bi-monthly Oct 2002 – Oct 2005; Browder et al 2005
[2] six-week interval Jul 1983 – Jan 2005, with gaps in the POR; Robblee et al 1991; Robblee unpublished
[3] semi-annually Sept/Oct 1998 – Apr/May 2000; Robblee et al unpublished
[4] monthly Oct 1995 – Oct 1996; Rice 1997
[5] once in Jan/Feb 1986; Robblee et al 1991

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Table 2. Criteria for assessing pink shrimp status.

Spring

Reference Areas	Historical Data						
	median	1st	3rd	mean	95 CL	POR	FIAN Sampling Locations
South Biscayne Bay	0.44	0.21	0.68	0.45	0.49	5 yr	South Black Point
Eastern Florida Bay	0.05	0.04	0.06	0.05	0.31	2 yr	Duck Key Basin, Eagle Key Basin
North-Central FB	0.30	0.30	0.40	0.32	0.55	2 yr	Rankin Lake, Whipray Basin
South-Central FB	0.61	0.23	1.38	0.77	1.40	2 yr	Calusa Key Basin, Crane Key Basin
Johnson Key Basin	2.28	1.45	3.32	2.55	1.31	18 yr	Johnson Key Basin
Whitewater Bay	0.56	0.50	0.63	0.56	0.50	1 yr	Oyster Bay, Whitewater Bay

Fall

Reference Areas	Historical Data						
	median	1st	3rd	mean	95 CL	POR	FIAN Sampling Locations
South Biscayne Bay	0.71	0.52	0.78	0.72	0.58	6 yr	South Black Point
Eastern Florida Bay	0.15	0.08	0.20	0.13	1.55	2 yr	Duck Key Basin, Eagle Key Basin
North-Central FB	1.50	1.42	1.60	1.50	0.71	2 yr	Rankin Lake, Whipray Basin
South-Central FB	3.15	2.43	4.38	3.46	1.59	2 yr	Calusa Key Basin, Crane Key Basin
Johnson Key Basin	12.47	8.62	17.23	12.98	1.95	20 yr	Johnson Key Basin
Whitewater Bay	4.86	2.73	6.75	4.62	1.79	2 yr	Oyster Bay, Whitewater Bay

Table 3. Pink shrimp indicator scores in the six FIAN response areas for Spring 2005 through Fall 2007 collections. Indicator scores: 1 = good, .5 = neutral and 0 = poor; green, yellow and red, respectively.

Pink Shrimp Indicator Scores

Reference Areas	Spring			Fall		
	2005	2006	2007	2005	2006	2007
South Biscayne Bay	0.5	1	0	1	1	1
Eastern Florida Bay	0	0.5	0	0.5	0	0
North-Central FB	1	1	0.5	1	0	0
South-Central FB	0.5	0	0	0	0	0
Johnson Key Basin	1	0.5	0.5	0.5	1	0
Whitewater Bay	1	0.5	0	0.5	0.5	0

4.0 LAKE OKEECHOBEE MODULE

4.1 Brief Description and Background Information for the Lake Okeechobee Module

The recovery of LO is critical to the success of the Everglades restoration plan, as the lake is the heart of the south Florida ecosystem. Failure to realize effective measures to restore LO will adversely affect or delay efforts to restore downstream wetland systems and estuaries that either rely on or are affected by water deliveries from the lake.

LO is a large (1,730 square kilometers [km^2]), and for its size, an extremely shallow (average depth generally <3 meter [m]) freshwater lake located at the center of the interconnected Kissimmee River-LO-Everglades ecosystem in Central and Southern Florida (C&SF) (***Figure 4-1***). On a geologic scale, LO is very young, having originated about 6,000 years ago during the most recent oceanic recession. Under pre-settlement conditions, LO is thought to have been eutrophic (Steinman et al. 2002b) and was considerably deeper than it is today (Aumen 1995). Outflows from the lake were largely restricted to sheetflow to the south and east. A southern marsh comprised the northern headwater of the Florida Everglades, with the lake often supplying water during periods of high lake levels or lake-wide seiches as a result of tropical storms (Gleason 1984). The ability of the lake to provide a large volume of water storage, in concert with the natural storage of wetlands in the upper part of the basin and the relatively slow flow of the historic meanders of the Kissimmee River, allowed for moderation of the effects of wet-dry rainfall cycles on water levels in the sawgrass marshes and prairies of the Everglades to the south (NRC 2005).

Wright (1911) estimated the historic high stage for the lake at approximately 22.5 feet and a low stage of 19 feet. Along the western side of the lake, Heilprin (1887) reported the presence of a substantial sawgrass community, historic observations buttressed by recent research (McVoy et al. 2005). The historic presence of this shoreline community has direct relevance to historic lake stages given that water depth requirements to support a sustained sawgrass community strongly suggests an eight month hydroperiod for the area. Lake stages may have risen above the marsh ground elevation around two feet in the wet season and would fall up to a foot by the end of the dry season (McVoy et al. 2005).

Modern-day LO differs from the historic lake in size, range of water depth and connection with other parts of the regional ecosystem (Steinman et al. 2002a). Connecting LO to the Caloosahatchee River and construction of the St. Lucie Canal in the early 1900s greatly reduced system-wide water storage and sheetflow to the south during drier periods (NRC 2007). Construction of the Herbert Hoover Dike (HHD) around the lake reduced the size of LO's open-water zone by nearly 30 percent, and resulted in a considerable reduction in average water level (Havens and Gawlik 2005). The current littoral zone vegetative community, which consists of emergent, floating and submersed macrophytes, developed in response to post drainage lake stages (Pesnell and Brown 1977); that is, the lowering of water levels due to levee systems and control structures in both the Everglades and in the lake over the past 100 years (Richardson and Harris 1995). Perhaps more importantly, the dike also hydrologically disconnected the surrounding marshes from LO's historical littoral zone (Aumen 1995, Havens and Gawlik 2005), especially along the northwest side of the lake. This effectively reduced the extent of the littoral zone and disrupted both the ecologic and hydrologic connectivity to the Indian Prairie marsh system, which has been described as historically being one of the largest marshes in the Kissimmee River, Lake Istokpoga-Indian Prairie and Fisheating Creek basin complex.

During the last century and until relatively recently, when aggressive efforts to improve quality of runoff have been undertaken, LO was the recipient of increasingly excessive inputs of nutrients primarily from agricultural activities in the watershed (Flaig and Havens 1995, Havens et al., 1996). The sustained influx of these nutrients has resulted in dramatic undesirable changes in water quality. In the open water or pelagic region of LO, large algal blooms have occurred. Vast quantities of soft organic, nutrient-laden sediments have accumulated which are easily resuspended in the shallow lake by even moderate winds (Maceina and Soballe 1991). This has caused LO to become both increasingly turbid and has served to exacerbate water-column nutrient concentrations via release of those nutrients present in the resuspended sediment.

Despite an onerous series of human impacts, LO continues to be a vital aquatic resource of south Florida, with irreplaceable natural and societal values. LO is one of North America's most unique and economically valuable natural resources. LO's location and size has resulted in its being expected to support the demands of a variety of user groups that range from supplying potable water for several cities on the lake's perimeter, supplying water to recharge surface water wells in Florida's densely populated southeast coast and supporting commercial and recreational fisheries important to local economies. Unfortunately, commercial fisheries were suspended following the 2004 and 2005 hurricanes due to reduction in fish stocks and the attendant effect on profitability. The annual combined recreational and commercial asset value of LO has been estimated to be in excess of 180 million dollars (Bell 1987, adjusted to 2007 dollars).

As a consequence of being a key resource relied upon by both agricultural and urban concerns, as well as its ecological effect on all south Florida including the Everglades, Florida Bay and the other estuaries, the importance of LO's health cannot be overstated. The quality of LO's water and habitat has been influenced by a number of factors. During the period from the early 1970s through the 1980s, LO's phosphorus (P) concentration doubled (Havens and James 2005). Large frequent blue-green algae blooms in the late 1980s prompted concerns that LO was becoming hypereutrophic, and fueled fears in the press of an impending collapse.

As a result of the varied and widely-held concerns, CEMs were developed for LO to provide a science-based path forward toward restoration (SFWMD 2006). These models succinctly depict the interrelationships that exist between water level and nutrient condition, and those key flora and faunal communities that respond to or are affected by them. The models account for LO's three sub-regions that are functionally dissimilar, and as a consequence may respond to changes in water level and/or water quality quite

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differently, namely: a littoral marsh, a nearshore region and an open water region (***Figure 4-1***). The models also reflect LO's present spatial extent, rather than the larger historical boundaries.

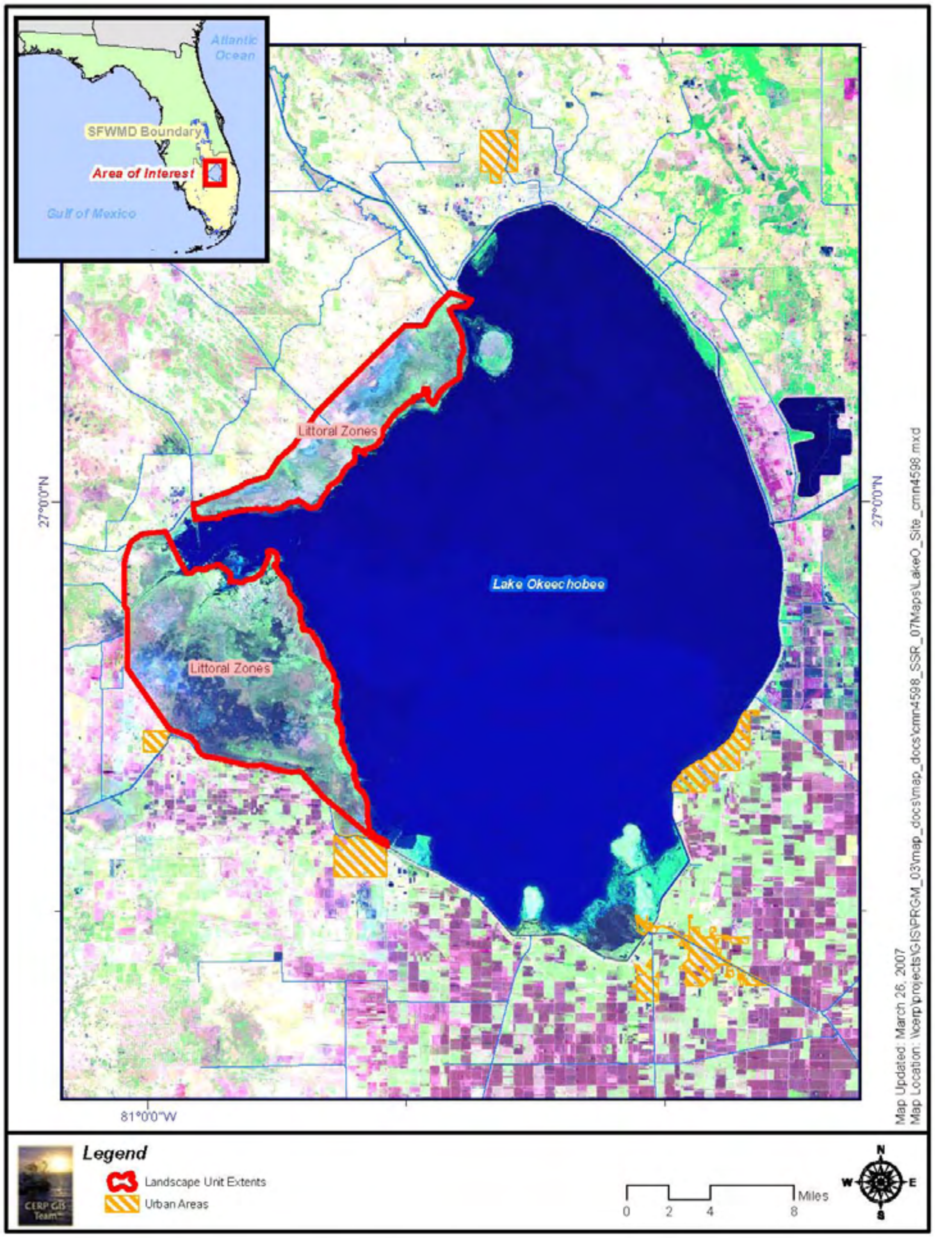

Figure 4-1: Lake Okeechobee

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4.2 Lake Okeechobee Hypothesis Cluster–Water Quality

4.2.1 Abstract

An evaluation of the water quality condition of LO was performed utilizing data from pelagic water quality monitoring stations 1988 to present. These stations characterize the vast majority of the volume of water within the lake and thus stand as the harbingers of change within this system as restoration efforts are successfully implemented. Nutrients, and in particular P remain elevated. Resuspension of sediment, which has become a large reservoir of potentially available P and nitrogen (N), is a major factor driving nutrient concentration. Increases in N concentration are correlated to increases in water color and chlorophyll *a*. P and total suspended solids (TSS)/turbidity measures signal slowly increasing trends, chlorophyll *a sh*ows a decreasing trend and N evinces no detectable trend. This suggests that diminished water clarity has served to damp algal blooms. The ability to detect beneficial change afforded by restoration efforts is high, since (1) current trends though small are in the wrong direction, and (2) major versus incremental changes in nutrient status are required. Implementation of CERP projects is expected to result in improved water quality attributes, such as reduced water column N and P concentrations, reduced TSS and chlorophyll *a* concentrations. The success of Everglades restoration hinges to a significant degree on the realization of effective measures to address and improve LO's water quality.

4.2.2 Background Description

The importance of water quality in LO in its role in the restoration of the south Florida landscape cannot be overstated. LO is the primary source of water for restoration in the southern half of the system, and attaining reduced nutrient levels in the lake is a critical and an essential requirement toward enabling treatment facilities south of the lake to attain the treatment efficiencies required to supply the southern system with water not exceeding ten parts per billion (ppb) P in concentration. Realizing the infrastructure necessary to improve the quality of water entering and leaving LO, as well as possessing the physical ability and flexibility required to better control lake stage (which affects water quality) is repeatedly identified as a prerequisite for restoration of both the Everglades and south Florida's estuarine ecosystems. Key water quality characteristics of concern for LO are the concentration of the nutrient P and the water column ratio of N to P, algal bloom frequency and composition, turbidity, sedimentation rates, sediment resuspension and cycling of nutrients sequestered in the bottom sediments (***Figure 4-2***). Of overarching concern is increasing P nutrient concentrations in LO which over the last 40 years have nearly doubled, and in consequence have been associated with periodically large algal blooms. Large blooms are a concern because of toxins that can kill fish, in addition to affecting taste and odor of drinking water.

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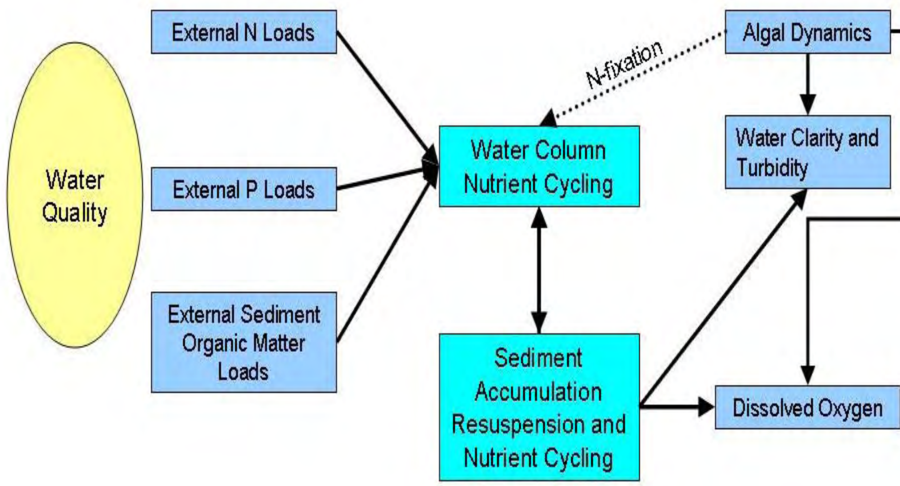

Dotted lines denote important feedback loop. Undesirable algal species can fix atmospheric N; sediment cycling of P exacerbates basin P loads.

Excessive loads of P to LO originate from agricultural and urban activities that dominate land use in the watershed. Total P (TP) loading now averages 714 metric tons per year (mt/yr) averaged over WY2002–WY2006. This loading is more than five times higher than the total maximum daily load (TMDL) of 140 mt/yr (five-year rolling average) considered necessary to achieve the target in-lake total phosphorus (TP) goal of 40 ppb (FDEP 2001, Havens and Walker 2002). The loadings from WY2006 were 795 mt of P which included the influence of Hurricane Wilma; 237 mt of this load originated from the Kissimmee River. This is lower than the previous year, WY2005 (960 mt), which included impacts from hurricanes Charley, Frances and Jeanne. However, the total flow to LO was greater in WY2006 than WY2005. This is attributed to the wetter spring and summer season in WY2006 compared to WY2005, which offset the flows from the September 2004 hurricanes. The lower load in WY2006 is attributable to lower TP inflow concentration.

Reducing nutrient loading to LO from its surrounding basin is only part of the process leading to an ecologically healthier system. As a result of excessive nutrient loading, primarily over the past 60 years (Brezonik and Engstrom 1998), over 30,000 tons of P is sequestered in LO's sediment (Reddy et al., 1995). Since LO is relatively shallow compared to its surface area, these sediments can easily be resuspended and through equilibrium processes, release P into the water column. The release of P into the water column through resuspension is of particular concern during hurricanes when massive disturbance of the shallow sediment can result in large spikes in post-hurricane water column P concentration.

Internal P loading also is of concern, as diffusive soluble reactive phosphorus (SRP) release from the sediments to the overlying water column is significant (Fisher et al., 2005). Sediment P assimilative capacity appears to be diminishing, thus contributing to increases in P concentration in the water column, despite an overall reduction in external P loading since the 1980s (SFWMD 2002, Havens and James 2005). Clearly, understanding the role that sediments play in lake restoration is paramount in developing restorative measures to reduce in-lake P concentrations. The current known set of feasible options to reduce P concentration includes dredging, chemical treatment and simply allowing natural processes to proceed. The latter is the currently selected course of action (Blasland, Bouck and Lee Inc. 2003). Expectations are that if loads can be reduced in the near-term, an acceptable degree of nutrient reduction and hence ecological recovery may be realized within the ensuing decades (Havens and James 2005). The current estimate for the no action horizon for positive impacts, without LO remediation is 70 years.

The presence of an easily resuspended organic mud on the bottom of the central area of LO presents additional water quality concerns. Frequently elevated suspended solids in the water column reduce light penetration. When conditions are favorable for transport of these sediments to the nearshore zone, which happens when lake levels are high, corresponding negative impact on plants may result, which in turn may affect those organisms that utilize the plant communities as a food source or for habitat. The basis of life in any system is conversion of the sun's energy to biomass that may then be utilized by all the subsequent trophic levels up the entire food chain. Lake turbidity prevents light penetration resulting in little photosynthetic activity except in the shallower areas or where plants have succeeded in stabilizing the sediment. If pelagic zone turbidity remediation occurred without nutrient remediation, severe algal blooms might result. LO's pelagic food chain is currently dominated by heterotrophic bacteria, indicating a switch in carbon source could have potential far reaching effects. Ultimately, ecological improvements in LO are dependent on reduction in nutrient loads and allowing lake sediment stability to improve through natural processes (e.g., compaction).

4.2.3 Methods and Analysis

A multitude of investigative studies and long-term monitoring efforts are underway, both to examine processes occurring in LO and within the lake's watershed. Details of these efforts may be found in the most recent version of the 2007 South Florida Environmental Report (SFER) (SFER-www.sfwmd.gov). Water quality data for LO is stored and is available in the SFWMD's "dbHydro" database. A subset of the available data was used for this report, namely grab sample data from the eight long-term monitoring (in-lake) stations (***Figure 4-3***). These sites have been identified by the State of Florida Department of Environmental Protection (DEP) to track progress in achieving the imposed TMDL which seeks to reduce P loads entering LO with the goal of reducing in-lake P concentration. Correlations between water quality data were determined on ranked data using the nonparametric Spearman rank method. Water quality trends were evaluated utilizing the nonparametric Seasonal Kendall Trend test, using the twelve months as individual seasons. Where trends were significant ($p \leq 0.05$), rates of trend were estimated using Sen's Slope technique on the series of monthly averages of the eight sites combined.

Figure 4-3: Locations of Eight Surface Water Monitoring Stations

4.2.4 Discussion

***A thorough discussion of water quality issues surrounding LO can be found in the*

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SFER for 2006 and 2007 (available online at www.sfwmd.gov) and the 2008 SFER (draft available September 2007).

There has been a tremendous amount of concern expressed regarding the accelerated eutrophication of LO, with the principal focus being rising P concentrations within the lake since the 1970s when SFWMD began monitoring water quality. The P load entering LO is closely related to the volume of water flowing into the lake from its tributaries (***Figure 4-4***). Numerous control efforts are underway or already have been instituted in the LO watershed to capture a percentage of nutrients which would otherwise enter the lake, further fueling worries regarding accelerated eutrophication of the lake (LO Protection Plan 2007). However, the concentration of P in the water column is merely part of the issue; a vast reservoir of P is sequestered in LO's sediments. The decreasing trend in assimilative capacity of the sediments to bind P suggests the counterintuitive response that once CERP watershed projects are completed and inflow P concentrations and loads to the lake decline, there may be a decades-long lag before in-lake concentrations similarly begin to decline (***Figure 4-4***).

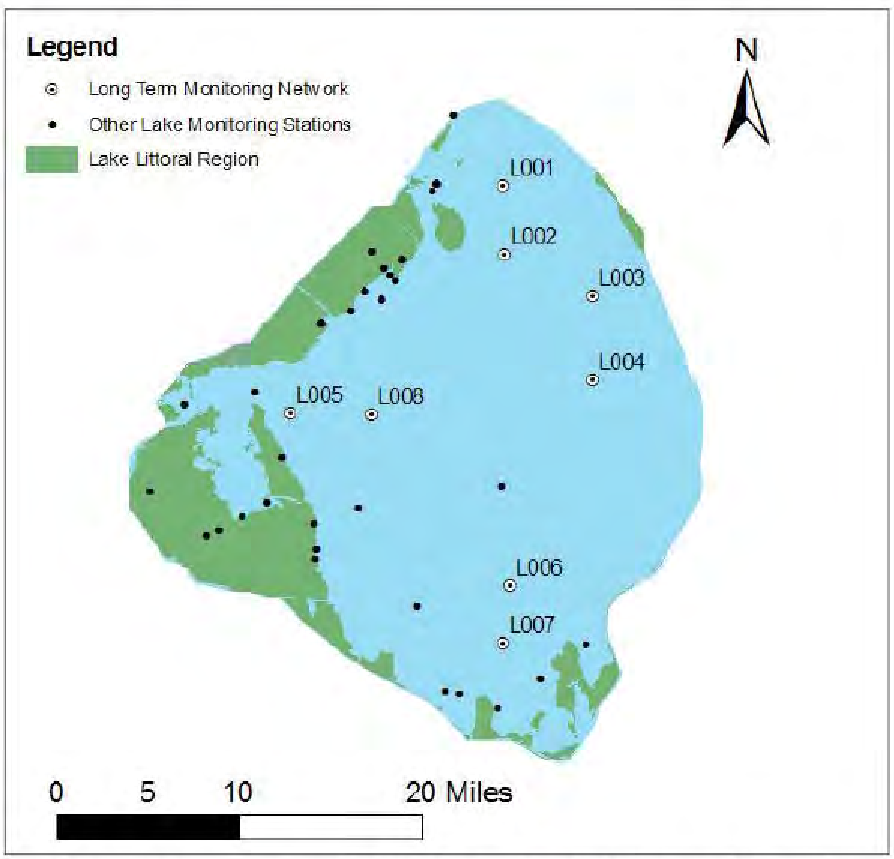

Figure 4-3: Locations of Eight Surface Water Monitoring Stations

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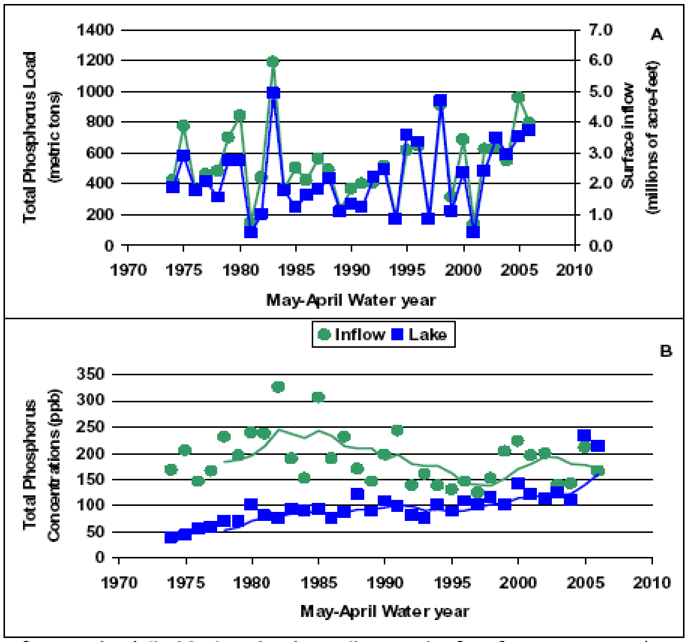
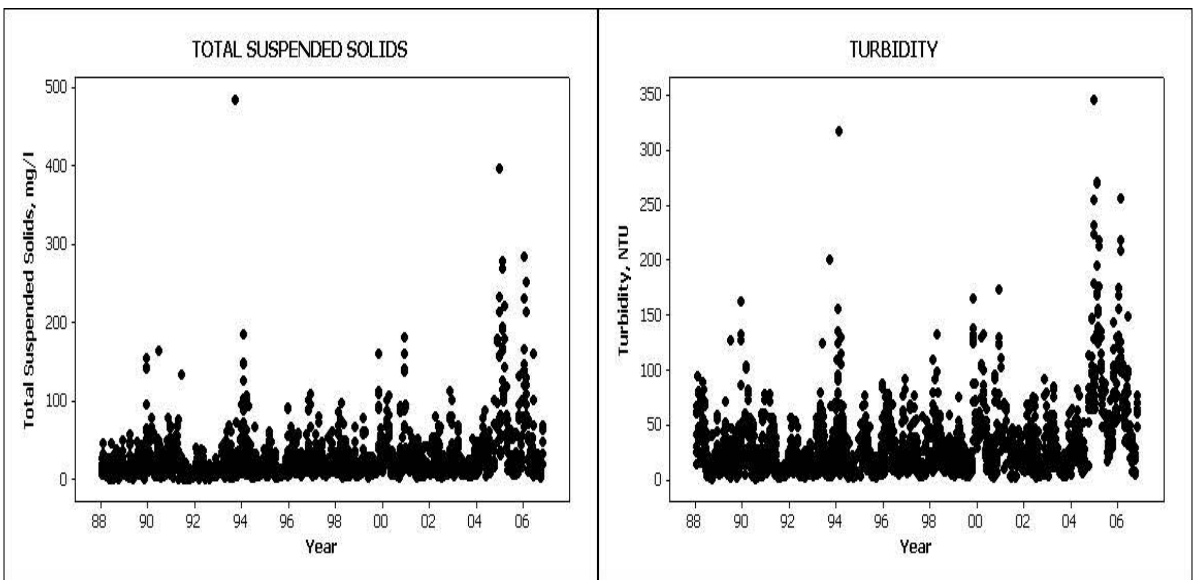


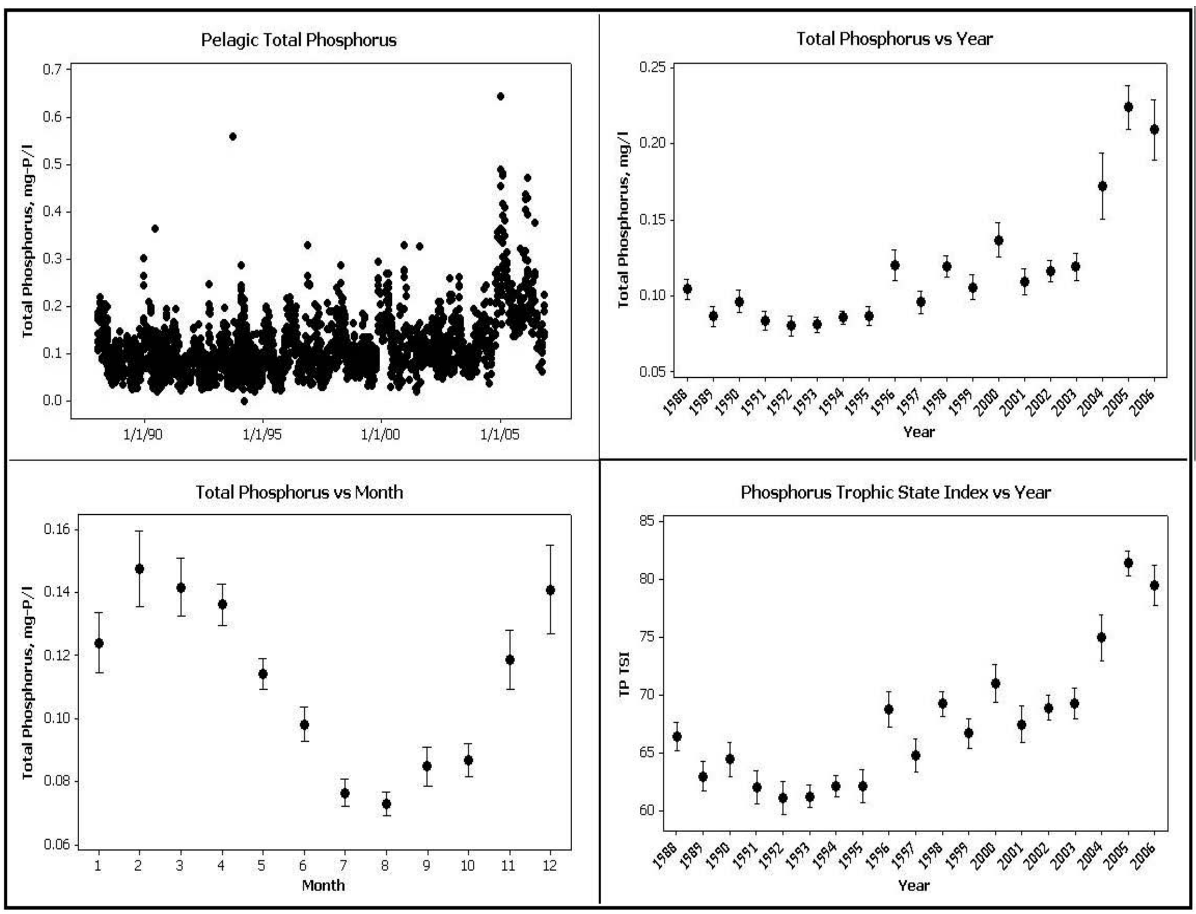
Timelines (top graphic) of WY P load (in blue) and volume (in green) of surface water entering LO, and five-year moving average (bottom graphic) of inflow and in-lake TP concentrations (from 2007 SFER). Note decrease in inflow concentration from 1982 through 1997 and apparent rebound in some years thereafter. Internal concentrations 2005-2006 were for the first time observed higher than inflow concentrations.



Data does not reflect conditions occurring during or immediately after hurricane impact

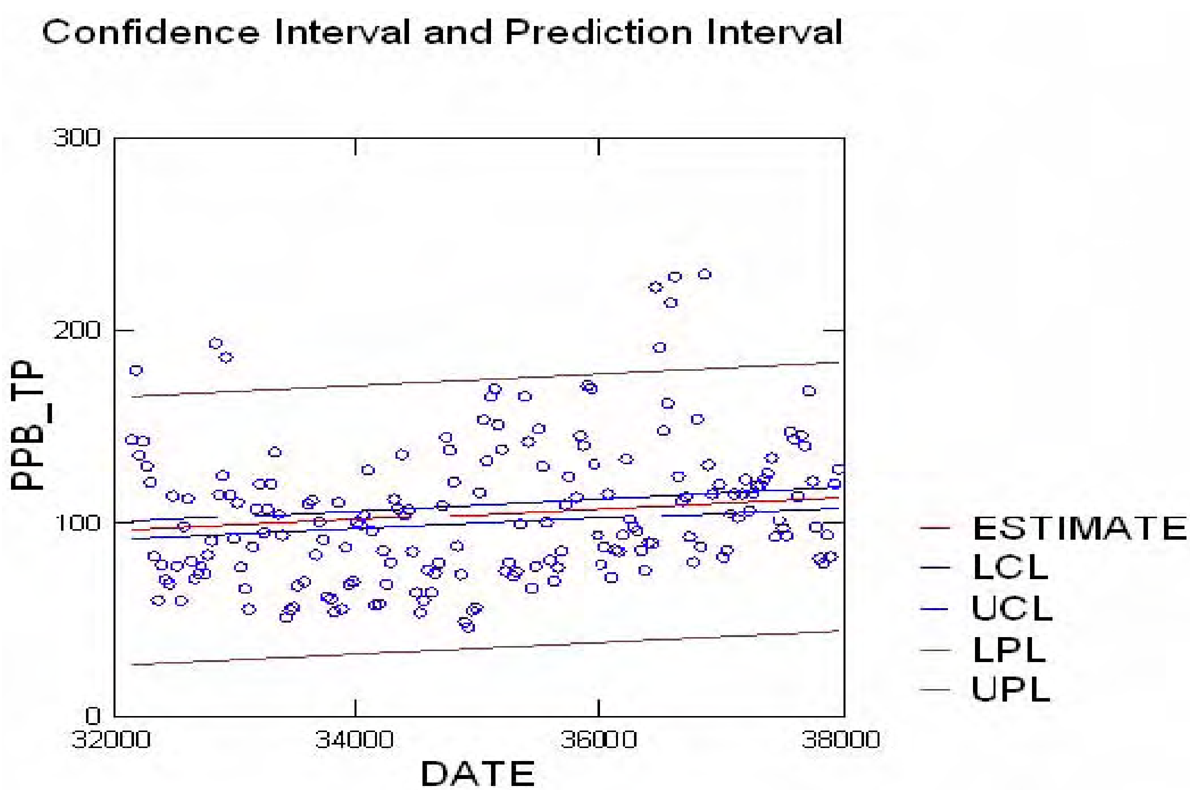
Figure 4-5: Mean Pelagic Total Suspended Solids and Turbidity Versus Year

The passage of the 2004 and 2005 hurricanes is readily apparent in the amount of unconsolidated sediment resuspended in the water column (***Figure 4-5***), despite the fact that samples were not taken during, or immediately following, the storms. It may be that the quantity of suspended sediment in the water column during the storms was orders of magnitude greater than that depicted (***Figure 4-5***). Spikes in concentration present in non-hurricane years (i.e., 1990, 1994, and 2000-01) serve to convey the ease with which the sediments in this large shallow lake are affected by wind-induced waves and currents (Maceina 1990). Both TSS and to a greater extent turbidity are correlated ($P<0.01$) to mean daily wind speed. The periods of non-hurricane attributable to increases in sediment resuspension coincided with lowered lake stage in 1990 and to a lesser extent in 1994, but stage and measures of resuspension of bottom sediment are not significantly correlated. There is a slight but significant ($P=0.001$) upward trend in TSS which remains even when the 2004 through 2006 hurricane influenced data is removed. A similar trend ($P=0.02$) is apparent in turbidity.



Depicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top right), by month (lower left), and trophic state index for phosphorus by year.

P concentration in LO (***Figure 4-6***) is correlated ($P<0.005$) to TSS and turbidity, which corresponds with the resuspension of the P otherwise sequestered in the sediments. A large fraction of the P reported as TP is in the particulate form (TP analyses are performed on unfiltered samples); however, the increases in resuspended sediments were also associated with increases in soluble P (P<0.005). The similar but magnified effect of the 2004 and 2005 hurricanes can be seen in the marked upward jump in P concentration seen in both of those years. The failure of the 2006 concentration to return to pre-2004 levels is of concern, and is explained by the sediments being less consolidated than they were pre-hurricanes Frances and Jeanne in 2004. There is a clear seasonal pattern which is a product of higher wind velocities typically occurring during the winter and spring. The resultant wave action resuspends sediments which in turn result in elevated TP concentrations. Removing hurricane influenced data (i.e., 2004 through 2006) does not remove the significant (P<0.001) upward trend in TP concentration at a rate of two to three ppb/yr (***Figure 4-7***). The P trophic state index, computed using the formula specified in the Florida Administrative Code (F.A.C.) rule 62-302 evidences a consistent upward trend, which is troubling since higher values indicate worsening trophic conditions.



(hurricane-affected years removed) Depicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top right), ortho-P to TP ratios by year (lower left) and by month (lower right).

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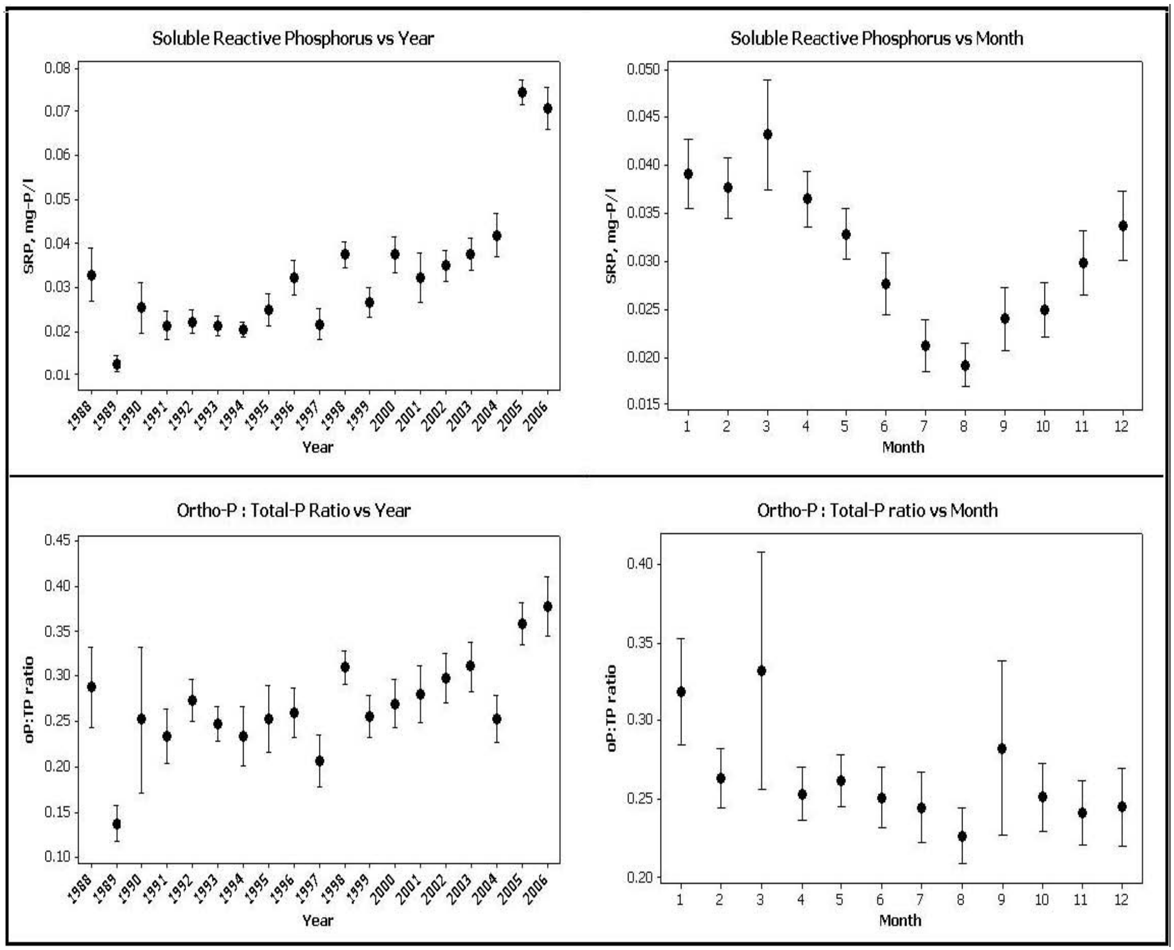

The pattern over time exhibited by soluble reactive P (SRP or ortho-P; ***Figure 4-8***) mimic those observed for TP. Both the ortho-P concentration as well as the ratio of ortho to TP exhibit significant ($P<0.001$) increasing trends. The increasing trend in the ratio of soluble to TP indicates that not only is more P present in the water, but more of it is in the more bioavailable form.

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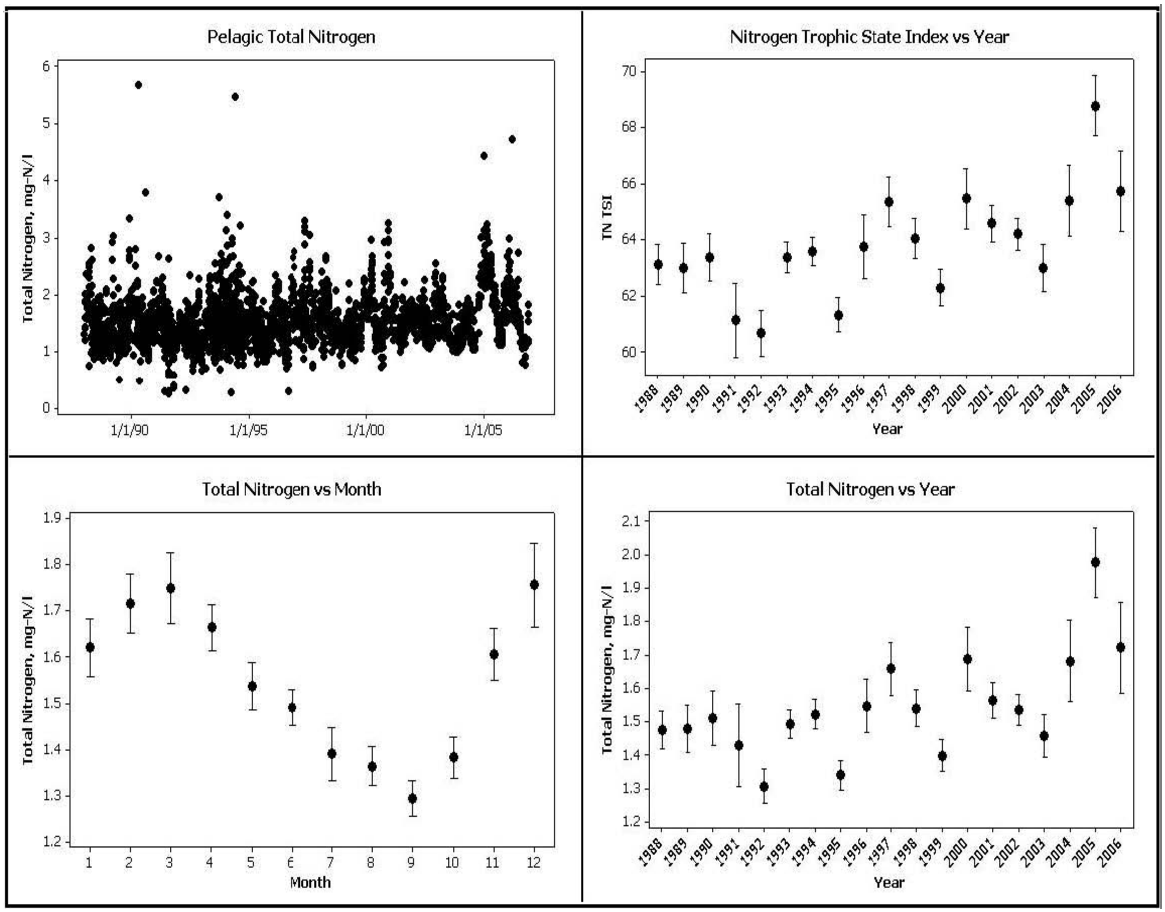
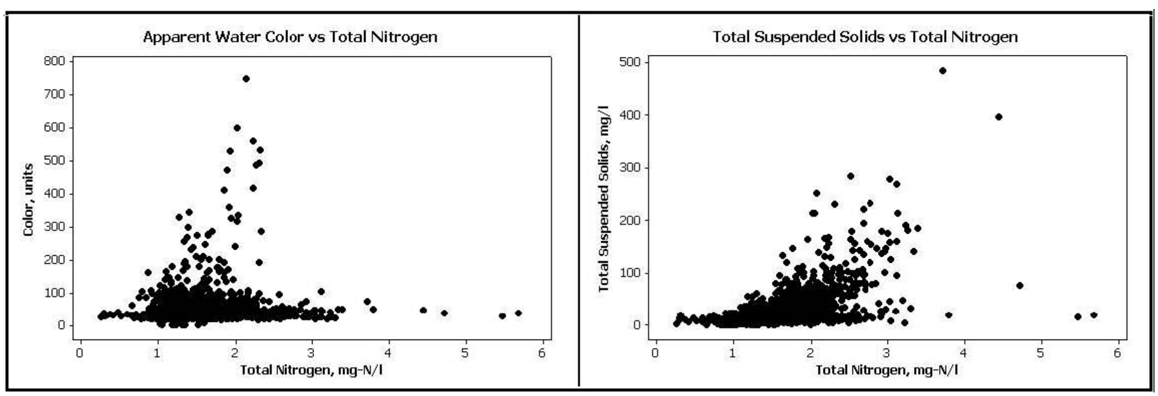


Depicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top right), month (lower left) and trophic state index for N by year.



Depicted are individual sampling data pairs. Chlorophyll *a* concentration (not shown) relationship is very similar.

Figure 4-10: Pelagic Total Nitrogen Concentration Versus Apparent Color and Total Suspended Solids, January 1988–September 2006

Total nitrogen (TN) concentrations are correlated with TP concentrations ($P<0.005$), despite no significant trend being apparent in TN concentration when 2004-2006 is

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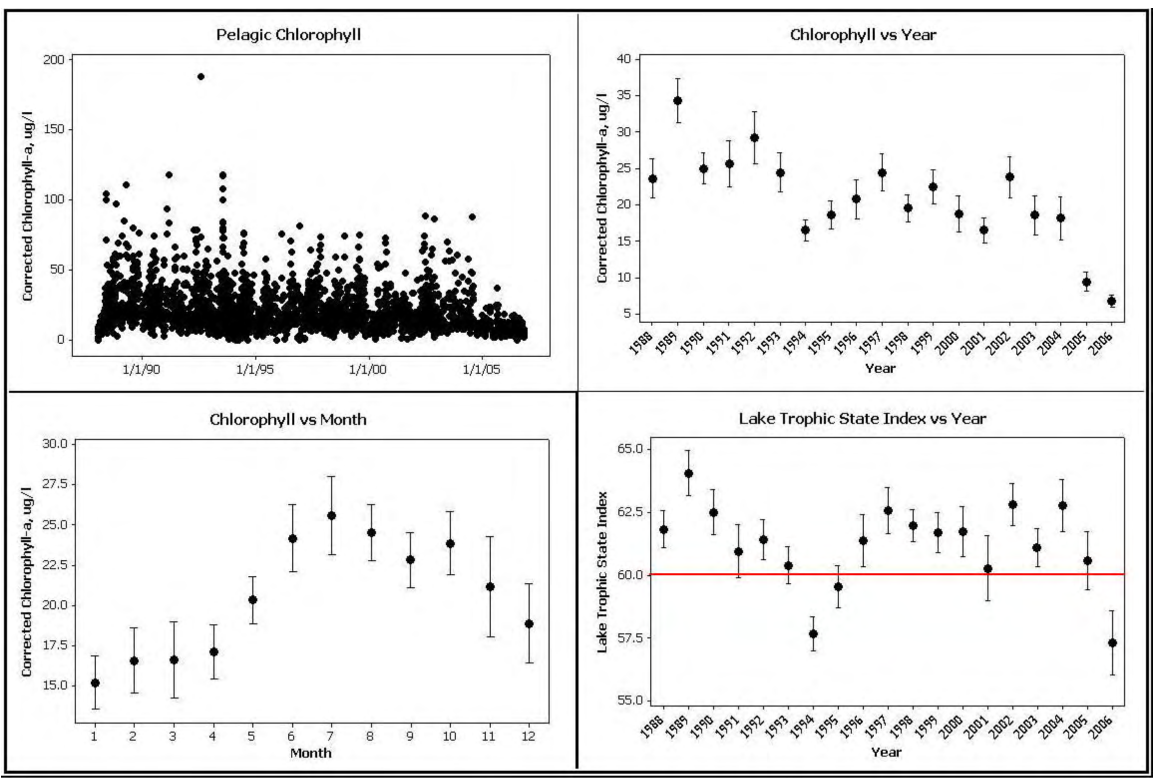
removed from the analysis; events where TP are high are often accompanied by higher TN. The general seasonal and annual patterns in N and P concentration (***Figure 4-9*** and ***Figure 4-6***) suggest that similar or related factors that affect and drive the P regime similarly affect the N regime. TN (N=2517) is also correlated to TSS (ρ=0.603, P<0.001), and to a lesser extent apparent color (ρ=0.186, P<0.001) and inversely to chlorophyll *a* (ρ=-0.068, P=0.04)–the latter being easily interpreted insofar as high suspended matter and/or color equate with reductions in light penetration. However, only three of the 25 observed TN values above three milligrams per liter (mg/l) were not accompanied by measures of either high suspended solids (most probable cause), apparent color, chlorophyll *a* or some combination thereof (***Figure 4-10***). There is no significant trend apparent in water color.



epicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top righ
month (lower left), and lake trophic state index for P by year.

Figure 4-11: Pelagic Chlorophyll *a* Concentration, January 1988–September 2006
Depicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top right), by month (lower left), and lake trophic state index for P by year.

A slight downward trend in chlorophyll *a* concentration remains significant (P=0.001) when both the 2004-2006 data as well as the 1988-1989 data are removed (to test whether the trend may be a mathematical artifact arising from early-year documentation of very high chlorophyll *a* concentrations not reproduced in later years, and by the lower chlorophyll *a* concentration following the hurricanes) (***Figure 4-11***). Possible explanations include the decrease in light penetration (as evidenced by trends in TSS and

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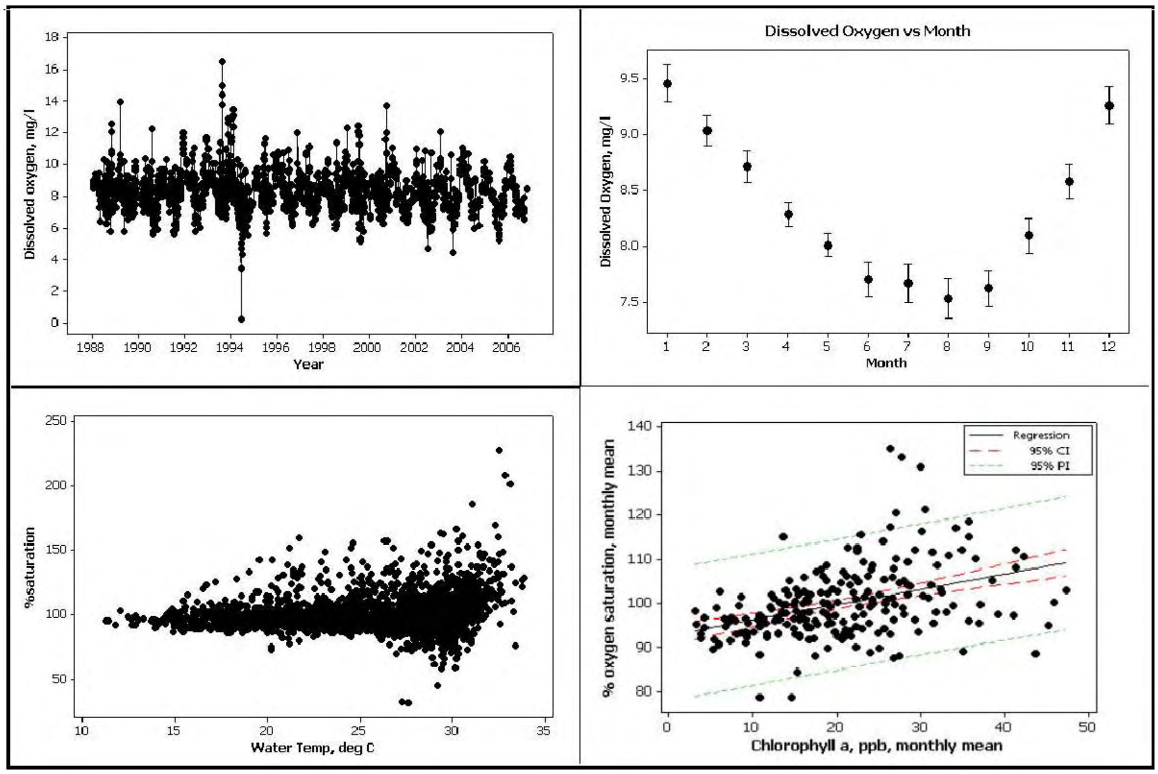
turbidity) which offset the increased availability of P nutrient, and the general lack of trend in TN. Disregarding the overarching role of light penetration in bloom formation, bioassays have indicated that N was the most frequent limiting nutrient controlling phytoplankton growth (Phlips et al. 1997) which is not surprising given the ubiquity of P availability. East and Sharfstein (2006) reported that light limitation was the dominant factor approximately 60 percent of the time, with N or co-limitation by N and P dominating the remainder of the time. The within-year seasonal trend is not surprising in that warmer summer days were typically less windy than winter months, thus improving light penetration and resulting in more algae (i.e., higher chlorophyll *a*). The Lake Trophic State Index was calculated based on nutrients and chlorophyll *a* concentrations as referenced in the F.A.C. Values for the lake index above 60 units denote impairment per Florida's Impaired Water Rule (F.A.C. rule 62-303). All but three of the years evaluated exceeded this threshold.



Dissolved oxygen (DO) regime ranged from 6.5 to 10.2 mg/l for 90 percent of the observations during this period of time. A clear seasonal pattern is present (upper right). Percent DO saturation versus water temperature (lower left) showing increases in variability and probability of super-saturation with warmer temperatures. Mean monthly DO saturation versus mean monthly chlorophyll *a* concentration showed overall increasing probability of super-saturation with increasing chlorophyll.

Figure 4-12: Pelagic Dissolved Oxygen Concentration, 1988 thru 2006 (upper left)

Overall, the DO regime in LO is fairly stable, with 90 percent of the observed values falling between 6.5 and 10.3 mg/l. DO concentration is inversely correlated (P<0.001) with water temperature (i.e., colder water can dissolve more oxygen) as depicted in the seasonal pattern (top right panel of ***Figure 4-12***). DO saturation is positively correlated with water temperature ($P<0.001$) such that warm water conditions are more likely to achieve a saturated or higher condition. In addition, the variability of DO saturation is increased at higher temperatures (***Figure 4-12***). The increasing variability and the greater likelihood of achieving saturation are both explainable by the presence of an increasingly dynamic algal community as temperatures rise, alternately generating or consuming oxygen as blooms wax or wane. DO concentration and saturation are correlated with chlorophyll *a* ($P<0.001$), and examination of the 179 observations taken during the day where DO exceeded ten mg/l evidenced a corresponding mean and median chlorophyll *a* concentration of 30 and 25 micrograms per liter (ug/l), respectively, and are values indicative of bloom conditions. Algal blooms which produce oxygen during daylight will also consume oxygen at night, which can result in oxygen crashes (and in severe cases, fish kills) at night; such diurnal occurrences have been documented in the littoral zone, but significant oxygen swings have not been observed in the pelagic zone.

<u>Conclusions</u> Water quality in LO is highly variable, and efforts to improve conditions which only make small improvements will be undetectable against the backdrop of year-to-year and intra-year change. Concerted efforts to reduce nutrient loads entering LO will be tempered by the presence and availability of nutrients currently present in the sediments, and especially during the relatively frequent events when wave energy is sufficient to mix those sediments back into the water column; however, dramatically reducing nutrient loads to the lake may permit various benthic processes to mediate the sediment nutrient reservoir and allow near term improvements in condition to occur (Jeppesen et al. 2003). Even if internal loading does delay full recovery, observations of shallow eutrophic systems elsewhere around the world where internal loading was considered a significant factor (Jeppesen et al. 2005) indicate the possibility of establishment of a new P equilibrium and measurable improvement in as little as ten to 15 years. However, many of those familiar with LO consider this forecast overly optimistic and recovery estimates without internal load remediation have been predicted to be on the order of 50 to 70 years (Blasland, Bouck and Lee Inc. 2003). Nevertheless, improved water quality conditions in terms of reduced nutrient, TSS and chlorophyll *a* concentrations, coupled with maintenance of appropriate water levels as a result of CERP project completion could result in immediate benefits in the nearshore and littoral zones; zones where most of LO's ecological functions occur and societal values originate (James and Havens 2005). Regardless of how recovery may in fact proceed, it is clear that realizing the benefits to better manage LO and its basin will require patience. Long-term effective measures will produce benefits, but detecting these changes will require unabated commitment to monitoring that produce quality datasets extending to 2050 and perhaps beyond. The current gradually worsening condition depicted in the pelagic zone data nevertheless holds promise as a mechanism to detect near-term positive changes as a result of restoration initiatives, by removing or reversing these trends.

4.3 Lake Okeechobee Hypothesis Cluster–Stage

4.3.1 Abstract

The current LO WSE Operating Schedule is more restrictive than past lake stage regulation schedules in stipulating conditions in which water is released from LO. This has resulted in an increased frequency and duration of undesirably high lake stages, which in turn have adversely affected key lake ecosystem features such as appropriate coverage of submerged aquatic vegetation (SAV). Revisions to the Lake Okeechobee Regulation Schedule (LORS) are currently being evaluated by Corps and SFWMD, as part of a cooperative effort including the U.S. Fish and Wildlife Service (USFWS), Florida Fish and Wildlife Conservation Commission (FFWCC), City of Sanibel, and Martin and Lee counties. It is anticipated that the new interim LORS, if approved by the Corps, is expected to sustain a higher frequency of lower lake stages. This interim schedule will promote the health of LO and aid in its recovery as the CERP, Acceler8, Fast Track, and other projects come online between 2010 and 2015, at which time a more permanent schedule will be implemented.

4.3.2 Background and Description

Water level in LO is a primary factor (***Figure 4-13***) affecting both the aquatic vegetation and the community of animals that utilize these plants for habitat and sustenance (Johnson et al., 2007). Since implementation of the current WSE, LO has experienced an increased frequency of high lake stages (***Figure 4-14***). Extreme high or low lake levels of any duration, or moderately high or low lake levels of prolonged duration greater than six months can cause significant harm to the ecosystem (Havens and Gawlik 2005). Extreme high stage facilitates the inflow of turbid, nutrient-rich pelagic water into the littoral and nearshore zones. Movement of this P-rich pelagic water into the nearshore region can promote algal blooms, and also is detrimental to emergent and SAV growth and biomass by increased water column depth, turbidity and wave energy. Increased wave energy can cause increased uprooting of vegetation, especially during high-wind and tropical storm events. Increased wave energy has direct negative impacts on emergent vegetation, such as bulrush, in the nearshore zone, and encourages the formation of a nearshore organic berm that can block fish migration into and out of the marsh. Increased water column depth and turbidity also results in poor water column light penetration (Havens 2004b). High stage may result in loss of habitat for fish, birds and other aquatic fauna as a consequence of reduced extent and quality of SAV and emergent plants.

Conversely, extreme low lake stage results in the desiccation of the western littoral marsh, which promotes the spread of exotic vegetation such as torpedo grass and melaleuca. When the marsh becomes dry, fish and wading birds are negatively impacted due to habitat loss. The federally protected and endangered Everglades snail kite also loses critical habitat and their primary food source, the Florida apple snail. Nearshore areas which can support high SAV biomass also can dry out under extreme low lake stage, thus resulting in replacement of SAV with emergent or terrestrial plants, and loss of habitat for fish, birds, alligators and other aquatic fauna. Conversely, extreme low lake stages can encourage the occurrence of brush fires that may help to control invasive and terrestrial taxa, such as cattail and torpedograss, which can quickly become a nuisance when covering large areas of the marsh or shallow nearshore areas. Low lake stages can also permit the oxidation of organic muck sediments exposing the underlying native seed bank and stabilizing material that might otherwise become resuspended where it would

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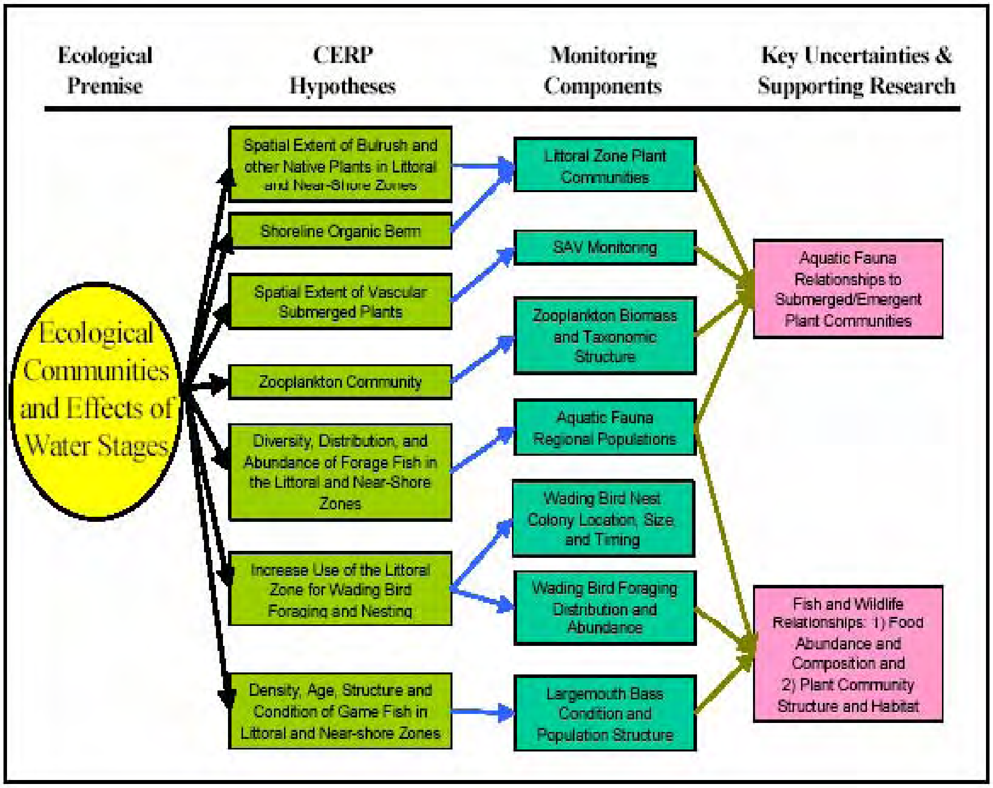
increase turbidity and reduce light penetration.



Higher lake stages can increase nearshore wave energy and drown shallow marshes. Although higher water levels have been shown to result in decreased nutrients and TSS, and increased secchi depth in the deeper offshore region, greater depths conversely result in higher nutrients, chlorophyll *a*, and TSS, and decreased secchi depth in the nearshore zone (James and Havens 2005); however, the decrease in offshore TSS and nutrients under high lake stage is small relative to the nearshore decrease in the same parameters under lower lake stages. The importance of these relative reductions in the nearshore versus pelagic areas is further magnified from the standpoint of ecological benefit; the nearshore zone is far more important to the ecology of LO than that of the pelagic zone. Higher stages have been shown to result in decreased water clarity which in turn limits the depth at which SAV can effectively establish (Havens 2003). In areas where SAV does occur, the SAV serves to stabilize sediments and to compete for available nutrients resulting in reduced chlorophyll *a* and TSS, and increased water clarity allowing for increased SAV cover (Havens 2003). Decreases in chlorophyll *a* concentration have been correlated with increases in SAV and epiphyton biomass on a seasonal basis (Phlips et al. 1993). Higher lake stages have also been shown to result in higher P and chlorophyll *a* concentrations in the nearshore during the summer, and that as lake stage increased the importance of wind in explaining nearshore P concentration also increased (Maceina 1992). Increased turbulence as a consequence of increased stage results in elevated P concentrations and turbidity in the nearshore zone, which can damage existing SAV communities and stimulate cyanobacterial blooms. It is important to note that these nearshore vegetated zones are where most of the beneficial ecosystem functions occur. Increased wave energy has direct negative impacts on emergent vegetation, such as bulrush, in the nearshore zone, and encourages the formation of a nearshore organic berm that can block fish migration into and out of the marsh. As a consequence of its effects on SAV and emergent plants, prolonged high lake stages may result in loss of habitat for fish, birds and other aquatic fauna.

A certain degree of natural variation in lake stage has been shown to benefit the plant and animal communities in LO (Havens et al. 2001, 2002, 2005; Havens 2003). Declining water levels in late winter and early spring benefit wading birds by concentrating prey resources in the littoral zone where those birds forage (Smith et al. 1995). Water levels near 12.5 feet benefit SAV and emergent vegetation such as bulrush by providing optimal light levels for photosynthesis in the summer months (Havens et al. 2004). Variation in the prescribed lake stage range results in annual flooding and drying of upland areas of the littoral zone, which favors development of a diverse emergent plant community (Richardson et al. 1995). This beneficial variation has been defined as avoiding extreme high water levels (stage greater than [>] 17 feet and stage > 15 feet for more than 12 consecutive months) and extreme low water levels (stage < 11 feet and stage < 12 feet for more than 12 consecutive months), increasing the frequency of spring recessions (yearly stage decline from near 15.5 feet in January to near 12.5 feet in June, with no reversal > 0.5 feet). Although reduction in extreme high and low lake stages is an important goal, one extreme low stage event once per decade is currently believed beneficial to oxidize muck sediment and facilitate germination of the bulrush seed bank.

4.3.3 Methods and Analysis

Lake stage is a major driving stressor, and stage directly or indirectly affects the physical and biological quality of LO. Data regarding lake stage is maintained in the dbHydro database.

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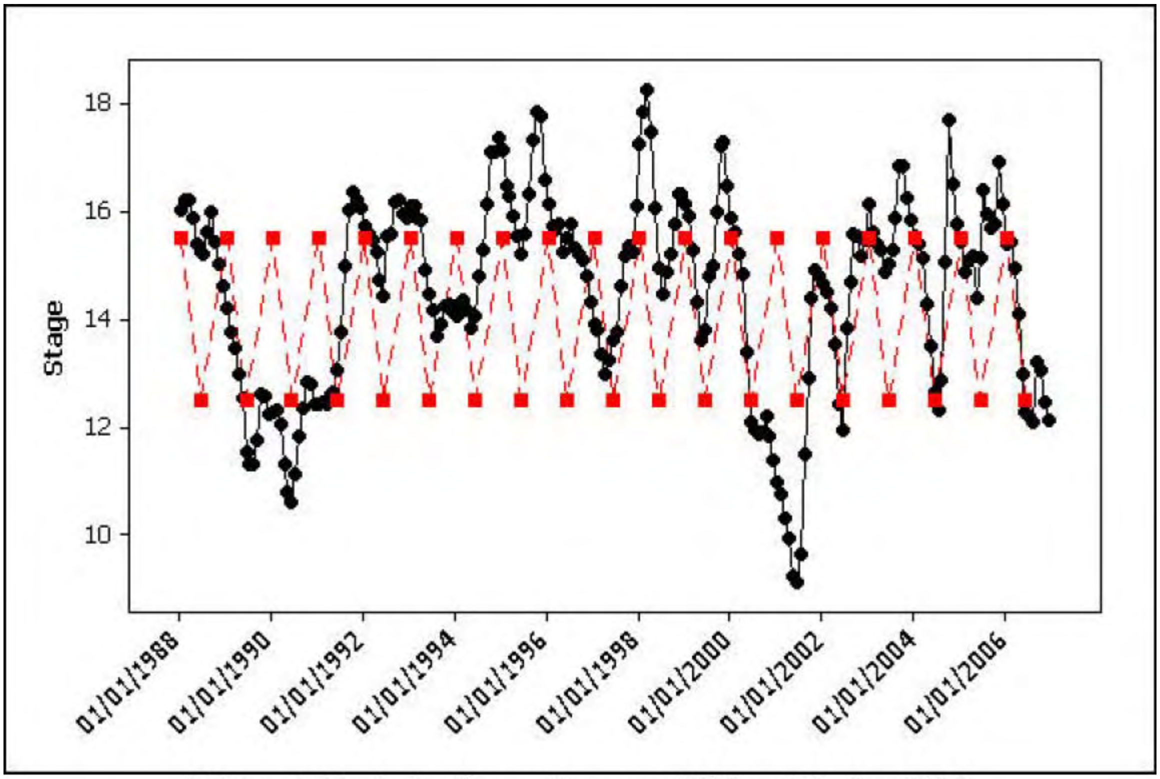

Figure 4-14: Lake Stages Between 1990 and Spring 2007

4.3.4 Results and Discussion

Efforts are underway, via the Lake Okeechobee Regulation Schedule Study (LORSS), to optimize LO's operating schedule within existing structural constraints to meet the diverse requirements of the lake, its receiving waters, and its users (***Mean monthly*** stage data (in black)

in feet above mean sea level, 1988 through 2006. Desired recession rates from January high of 15.5 to June low of 12.5 (in red) provided as reference to illustrate extent of deviation from ideal.

Figure 4-15). The goal is to bridge the gap until the CERP, Acceler8 and Fast Track projects begin implementation in 2010. Approval of a revised regulation schedule at this time is on hold temporarily, and contingent upon acceptance by all stakeholders. A revised regulation schedule will be supported by a Supplemental Environmental Impact Statement (SEIS) and selection of the Tentatively Selected Plan (TSP) is based on overall system wide benefits. The benefits are evaluated for the following areas: the Caloosahatchee and St. Lucie River estuaries, Everglades, WCAs and water supply, the Lake Okeechobee Service Area (LOSA), Lower East Coast Service Area (LECSA), snail kite habitat, HHD integrity and navigation impacts. All of the alternative regulation schedules developed to date were evaluated against PMs that were developed as part of CERP and the RECOVER program. Each evaluated alternative regulation schedule includes temporary forward pumps as a water supply component in the event of extreme low lake stages (< 10.1 NGVD) similar to those that arose during the 2000-2001 drought. Lake stage <10.2 NGVD precludes the release of water from the south end of LO to the south via gravity, so at this stage and below, the temporary forward pumps will be used to augment water supply for agricultural and irrigation purposes.

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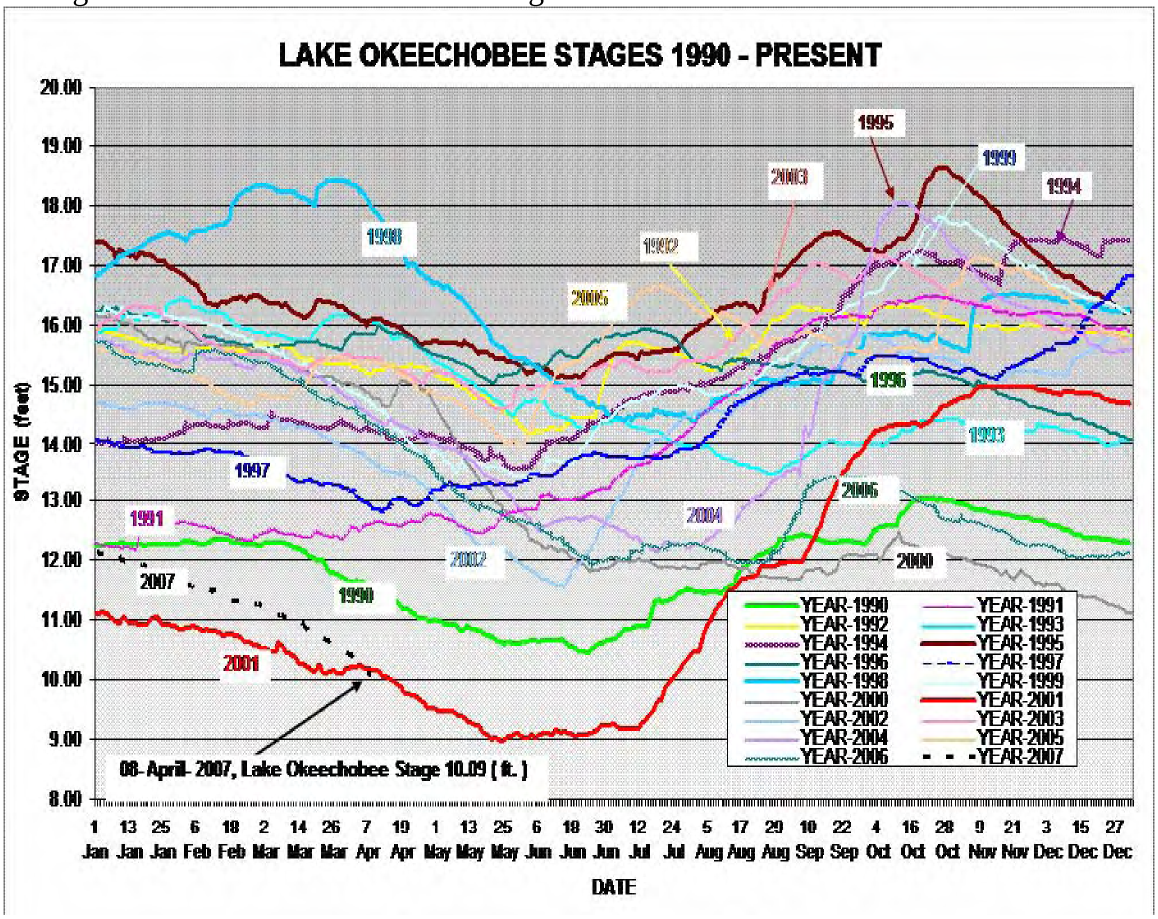
Once a preferred TSP has been selected, the Corps will hold public meetings throughout south Florida following the release of the Draft SEIS. A revised Water Control Plan (WCP) will also be released for a public review period. Once the interim schedule is implemented, efforts will be underway immediately to incorporate the CERP Band 1, Acceler8, permanent pumps and any additional storage projects into a new schedule. It is anticipated that the selected TSP will result in a new lake regulation schedule that will result in the lake being generally shallower and with less extreme lake stage fluctuation than has occurred in the past decade. This new LORS is anticipated to minimize impacts to overall system-wide benefits, such as water quality and quantity, navigation and ecological attributes such as SAV coverage and bird and fish habitat.



Mean monthly stage data (in black) in feet above mean sea level, 1988 through 2006. Desired recession rates from January high of 15.5 to June low of 12.5 (in red) provided as reference to illustrate extent of deviation from ideal.

4.4 Lake Okeechobee Hypothesis Cluster–Submerged Aquatic Vegetation

4.4.1 Abstract

SAV and its relationship to the health of LO is assessed by periodically sampling plant biomass and species composition along strategically located fixed transects, and by large-scale mapping of species specific vegetative coverage. Plant community structure can be successfully related to pertinent stressors, in particular to lake stage and factors that affect water clarity and light penetration. As CERP projects and other complimentary efforts improve conditions within LO, detectable trends of expansion in SAV areal coverage and increased biomass are expected. Except perhaps for the impacts of major physical perturbations such as hurricanes, the probability for successful utilization of this assessment tool is high.

A key objective of this long-term SAV monitoring is to understand changes in the SAV community in LO as they relate to changes in water level and transparency. More specifically, it is to provide data to evaluate the relationship of physio-chemical factors (e.g., nutrient concentrations, light availability) to the spatial and temporal dynamics of SAV biomass and species assemblages within the community. Changes in the spatial and temporal extent of SAV are key PMs that will be available for use in CERP-related modeling and evaluation efforts. Data generated from SAV monitoring are mapped and analyzed annually and multi-annually for long-term trend analysis to determine if the distribution and abundance is improving as a result of CERP implementation.

4.4.2 Background and Description

SAV plays a key role in shallow lakes, providing diverse spawning and foraging habitat for fish and provides an important food and habitat resource for wading birds and other wildlife (Havens and Gawlik, 2005) (***Figure 4-16***). SAV can also directly affect water quality attributes such as nutrient concentrations, water column transparency and phytoplankton biomass through a number of processes. Increased transparency and reduced turbidity often result in SAV beds due to the stabilization of the bottom sediment by roots and by reduction of current velocities and shearing stress to sediment surfaces, and as such constitutes an effective positive feedback loop that both benefits existing SAV as well as promotes their expansion (Koch 1996, Sand-Jensen and Mebus 1996, Bartleson and Rodusky *in prep*). Uptake of nutrients by SAV and associated epiphytes (attached algae) might be an important process in LO in areas where SAV is abundant, as a large colonizable SAV surface can result in abundant periphyton (Steinman et al., 1997; Rodusky et al. 2001). When periphyton is abundant and photosynthesis is intense, pH may sufficiently be elevated such that co-precipitation of P with calcium occurs (Murphy et al. 1983, Dennison et al. 1993, Scheffer 1998, Vermaat et al. 2000) and nutrients being removed from the water column that might otherwise be available to phytoplankton. Lakes with dense SAV, clear water and low phytoplankton biomass can switch to an alternative state of highly turbid water and increased severity of algal blooms if the SAV and associated epiphytes are lost (Scheffer 1989, 1998). Some lakes, including LO, have shallow areas where abundant SAV and clear water can exist adjacent to deeper areas with no SAV and turbid water (Phlips et al. 1993, Scheffer et al. 1994, Havens et al. 2004, James and Havens 2005). While the maintenance of alternative steady states is viewed as being a positive feedback loop, lake level, periodic wind-driven high turbidity and major physical perturbations such as hurricanes act as external forcing functions to drive changes from one state to the other; thus, the nearshore zone switches between a SAV/clear water state when water levels and turbidity are low to a phytoplankton/turbid water state when there are periods of prolonged high water levels with accompanying sediment resuspension and possible physical disruption of the plant community by wind driven waves and seiches (Havens et al. 2001, Havens 2003, Havens et al. 2004, James and Havens 2005).

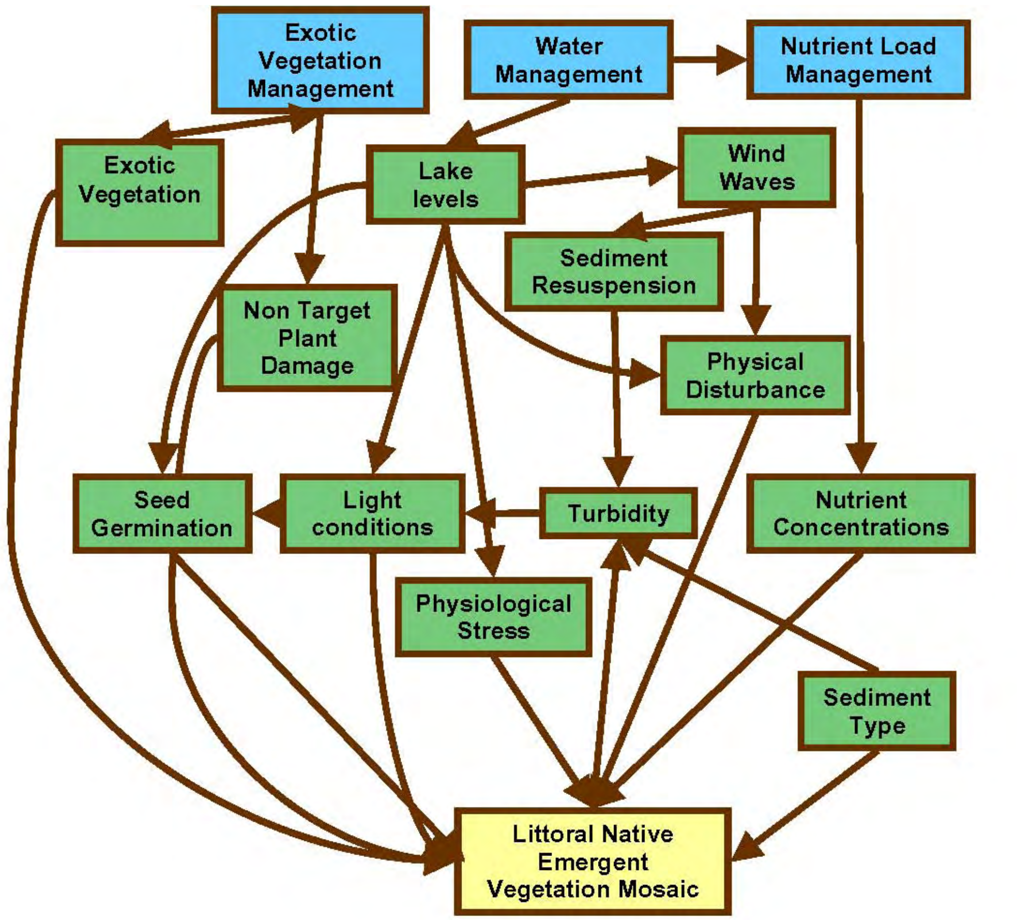


Figure 4-16: Conceptual Model of Stressors and/or Factors That Affect Submerged Littoral Zone Emergent Vegetation

Overall Goal CERP RECOVER targets currently specify an annual standing stock of 49,420 acres (200 km^2) of total SAV, with at least 50 percent composed of vascular native species. Under existing watershed uses and lake management activities, the spatial extent and abundance of SAV varies widely from year-to-year.

4.4.3 Methods and Analysis

A SAV monitoring program has been in place in LO since the spring of 1999 and encompasses data collected over a wide range of hydrological and environmental conditions. A change in collection methodology, however, allows a comparison only of the data collected since the summer of 2000. Additionally, historical SAV biomass and distribution data exists from transect studies conducted in the late 1980s and early 1990s (Zimba et al. 1995) that can be used to compare to the current SAV distribution and abundance.

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

Annual Mapping The total spatial extent, species distribution, and density of SAV are determined by an intensive sampling program (***Figure 4-17***) that is carried out at the end of the peak SAV growing season, i.e., August through September (Havens et al. 2002). Rather than sampling random locations, the entire nearshore area is evaluated at a spatial scale sufficient to detect significant changes. GIS coverage of LO's surface is overlaid onto a rectangular grid of 1,000 x 1,000 m cells in the GIS program ARC/INFO. GIS coverage of the littoral zone is laid onto the map, and common cells are clipped from the final coverage, as is the deeper central pelagic region. This results in a nearshore grid of approximately 750 sampling sites. Coordinates for the grid cell center-points are loaded into Trimble Pathfinder global positioning system (GPS) units (differentially corrected) for use in navigating to the sampling sites. A simple program is set up in each data logger so that users can enter information regarding water depth, Secchi depth (a measure of water transparency), sediment type, presence versus absence of vegetation by species and a qualitative estimate of overall plant biomass (sparse, moderate, dense). Field data are downloaded from the GPS logger into ARC/INFO, where maps are developed for each of the measured attributes and spatial extents for each dominant plant species are calculated in acres. This sampling effort provides information on the total number of acres of plants that the lake gained (or lost) under the prevailing hydrologic conditions of a given growth cycle year but these data should be used in the context of a coarse temporal scale trend analysis, due to annual growth season fluctuations that might result in months other than August-September containing peak SAV Transect Monitoring In order to obtain relatively rapid quantitative estimates of plant species biomass, sampling is conducted at up to 78 sites located along 16 transects in areas of LO that support submerged plants (***Figure 4-18***). The sites represent a subset of sites that were sampled in the LO Ecosystem Study (Zimba et al. 1995) in the late 1980s and early 1990s. This allows for a comparison of historical data. Sampling frequency varies from quarterly to monthly depending on how dynamic anticipated changes are expected to be in the plant population (for example, more frequent sampling is done during periods of recovery from hurricanes) and has been conducted monthly since the fall of 2004. Samples are collected at sites along each transect, starting at the shoreline and progressing lakeward until a site is reached that has no plants. Plant sampling is accomplished using a tool constructed of two standard garden rakes bolted together at mid-point to create a tong-like device (Rodusky et al. 2005). The degree of opening is constrained by placing a chain between the two handles so three replicate samplings with the device remove ~1 square meter (m^2) of bottom cover. The harvested material is sorted by species, stripped of epiphyton and dried to a constant weight. This sampling effort provides information on plant responses and relative plant distribution and density to changing water levels on a short time scale, than that for the annual SAV mapping, and can be used as input to real-time operations.

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abundance (Havens et al., 2002).

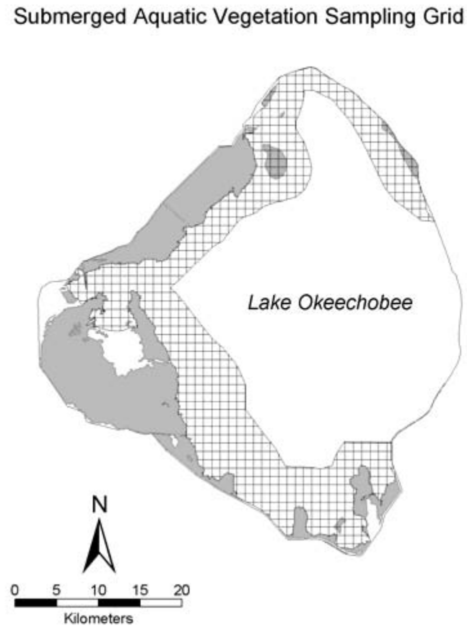

Figure 4-17: Annual SAV Mapping Sampling Grid

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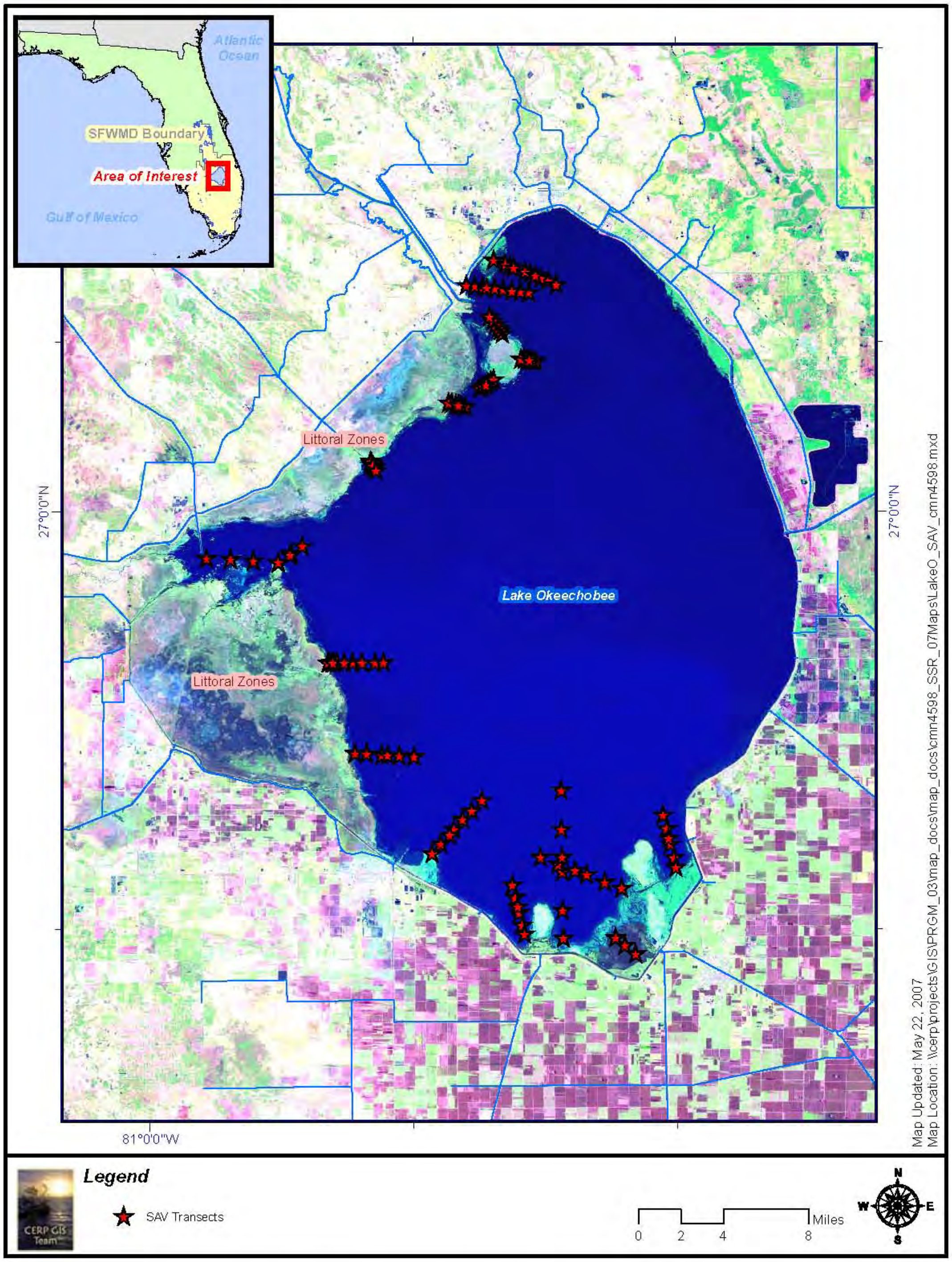

Figure 4-18: Submerged Aquatic Vegetation Transect Locations in Lake Okeechobee

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

Empirical Modeling An empirical model has been developed that predicts SAV presence or absence distribution based on light penetration to the bottom as a function of water transparency, as indirectly measured by TSS and lake water levels. This model is intended to be used in conjunction with GIS data layers such as bathymetry and SAV sampling sites to predict areas within LO that are likely locations for SAV colonization when favorable water depth, light penetration, and turbidity conditions occur. Future versions of the model will include attributes such as sediment type, seed bank viability and water quality variables. At the current stage of sophistication, the model only predicts areas containing a favorable light regime for SAV growth, and is not intended to predict finite growth areas. While results indicate conditions where SAV cannot occur (constraints), they do not indicate clearly whether or not SAV will attain high biomass under otherwise presumably favorable conditions.

4.4.4 Discussion

Annual Submerged Aquatic Vegetation Mapping Results SAV in August 2006 covered 2,965 acres of LO (***Figure 4-19***). This compares with the highest total coverage of 54,857 acres in late summer 2004. The large decrease in SAV from 2004-2006 likely was a response to poor light conditions, physical disturbance (wind and wave-induced uprooting) and high water levels caused by four hurricanes, three in 2004 and one in 2005. Lake stage increased on average by approximately 1.8 m after hurricanes Frances and Jeanne. It appears that the hurricanes stirred up fine sediments in LO resulting in a long exposure to very turbid, deep water column with very poor light penetration. After the hurricanes it was common to have Secchi disk readings of less than 30 centimeter (cm). The acreages of dominant plants in 2006, as compared to 2005 are as follows: *Vallisneria*-750 acres, compared to 494 acres in 2005; *Hydrilla*–0 acres, compared to 7,166 acres in 2005; *Potamogeton*–0 acres, compared to 494 acres in 2005; *Ceratophyllum*–495 acres, compared to 7,166 acres in 2005, and *Chara*–2,470 acres, compared to 247 acres in 2005. With regard to the latter, *Chara* is a macro-alga rather than a true vascular plant and is considered a pioneer species in LO, hence its relatively extensive coverage in 2006.

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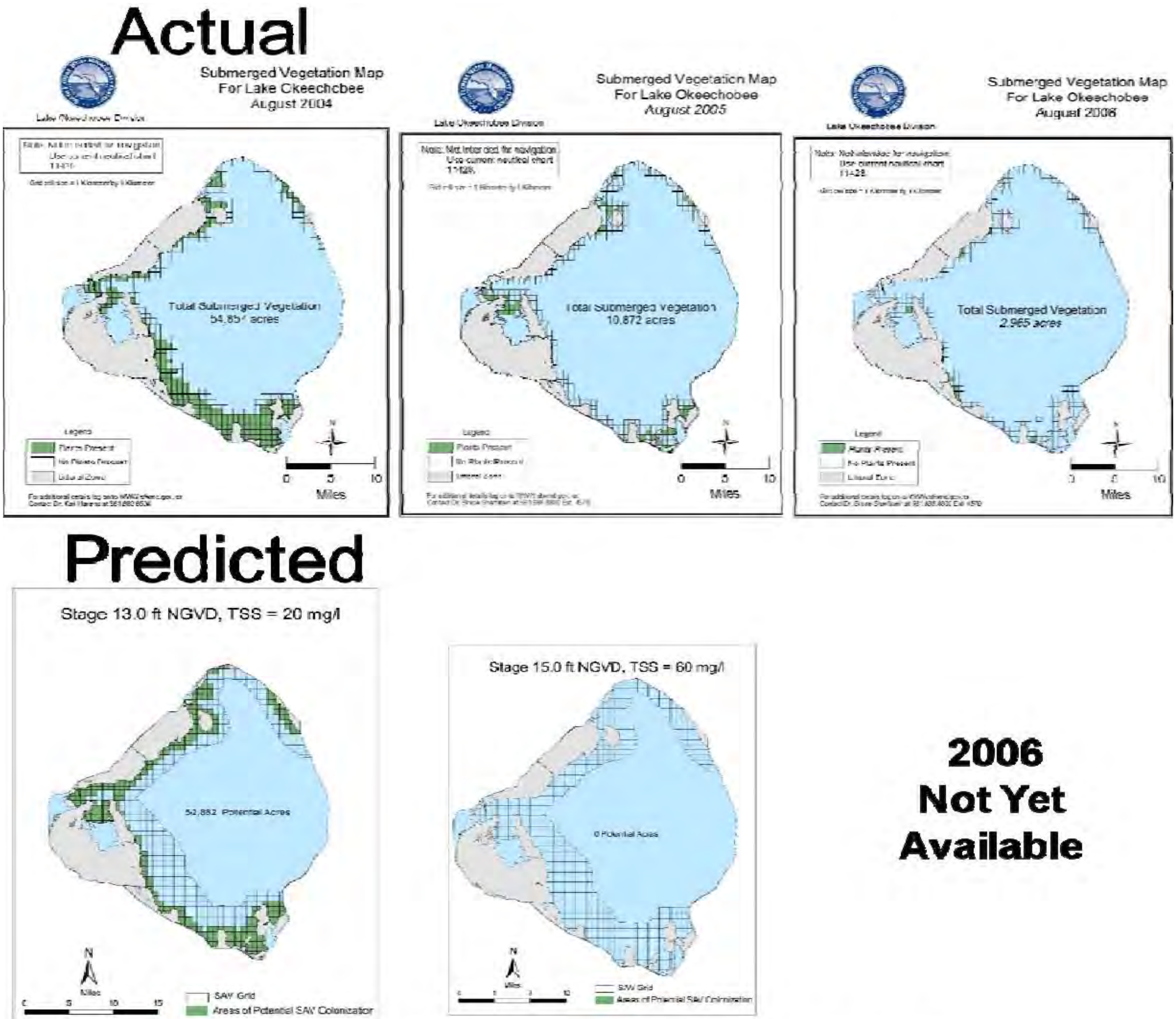

rged Aquatic Vegetation Biomass versus Model Predicted Biomass

Transect Submerged Aquatic Vegetation Mapping Results The best example of transect mapping's ability to detect substantial short-term perturbations occurred between July 2004 (pre-hurricanes) and October 2004 (post-hurricanes). Average SAV biomass, as measured at 78 quarterly monitoring sites, declined during this quarter from 32.3 (± 49.9 SD) to 4.7 (± 9.4 SD) g dry weight per square meter (g dry wt m^{-2}), probably as a result of increased TSS and decreased light penetration as reflected in Secchi to total depth ratios (***Figure 4-20***) brought about by direct wind, wave, seiche, and lake stage impacts. However, from January 2005 to June 2006, SAV biomass continued to decline from 4.46 g dry wt m^{-2} to less than 0.04 g dry wt m^{-2} (***Figure 4-21***). Although declines over the winter period are expected due to seasonal conditions such as lower temperatures, increased turbidity, and shorter photoperiod, the significant declines observed are primarily a result of long-term light deprivation related to water quality and lake stage effects.

Analysis The interplay between SAV biomass, lake stage, and water transparency is complex. Prolonged periods of high stage and poor water column light regime may greatly diminish the spatial extent of native vascular submerged plants. During years of lower lake stages, spatial extent of vascular and non-vascular SAV combined can exceed 50,000 acres. Once significant SAV communities become well established, higher lake stages can occur with little loss of plants unless the higher stages are sustained across many months. In the case where SAV communities have been completely lost, moderate stages will not typically facilitate their return; instead, very low stages may be required to re-establish successful and resilient communities of plants. In years of recovery from high water stress, much of the SAV community may be comprised of pioneer species, such as the non-vascular macro-algae *Chara*, which may provide limited habitat or water quality benefits as compared to the vascular species *Vallisneria*, *Potamogeton*, and *Najas*, but may promote the clear water state needed for colonization by the slower growing, higher light requiring vascular species. By reducing the frequency of extreme high and low water levels and increasing the frequency of spring recessions through CERP implementation, beneficial water quality and habitat conditions should be created that promote an increase in the spatial extent and density of native vascular submerged plants.

Light penetration defines the area capable of supporting dense SAV. Within a given water clarity regime, higher lake stages equate with decreasing light energy at depth. In addition, higher stages effectively connect the pelagic and nearshore zones resulting in increased turbidity which further exacerbates light availability (James and Havens 2005). Lower lake stages decrease the depth of water that light must penetrate to sustain photosynthetic activity; thus plants can survive despite conditions (e.g., wind and waves) that might decrease water clarity. Thus, lower lake stages and improved water quality conditions (e.g., reduced TSS) as a result of CERP implementation projects may result in larger areas of the lake bottom receiving adequate light to support growth. However, stage alone does not explain water clarity as stage is not directly correlated with either TSS or secchi depth measures of clarity.

Previous studies have shown that the biomass of submerged plants is negatively correlated with water depth and positively correlated with water transparency (Hopson

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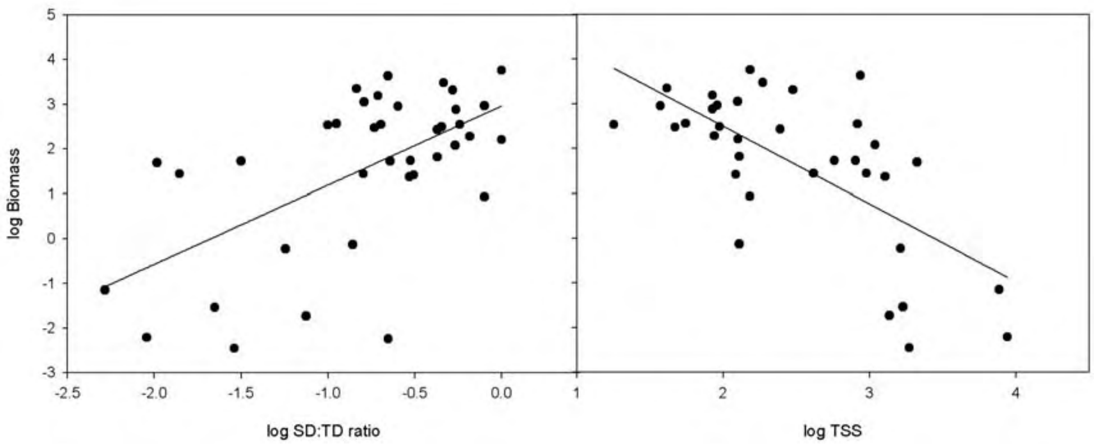
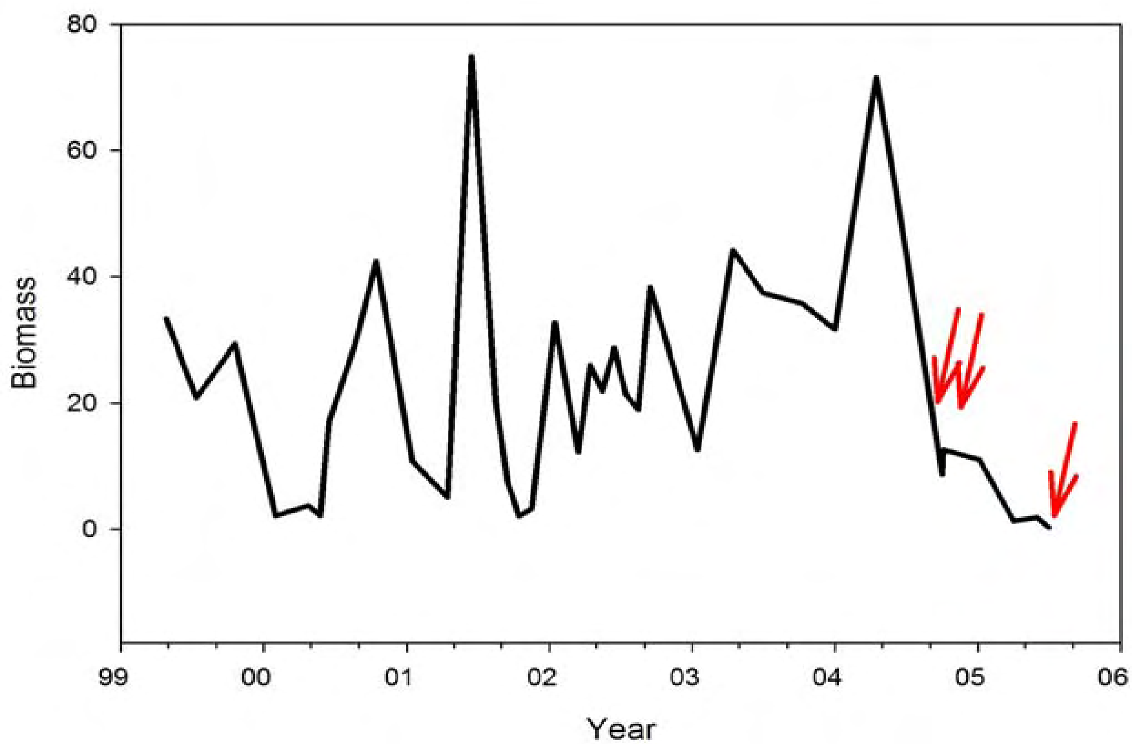
and Zimba 1993, Steinman et al. 1997). Analyses of recent data (***Figure 4-20***) substantiate the significant relationships between SAV biomass and water clarity.



Biomass (g dry wt m^{-2}) as a function of secchi disk depth to total depth ratio (R^2 = 53%), and as function of TSS (mg/l, R^2 = 40%). Relationships are statistically significant (P<0.005). Values shown are logarithms of sampling event means.

Figure 4-20: Biomass as a Function of Secchi Disk Depth to Total Depth Ratio, and as A Function of Total Suspended Solids

Wind and wave events increase turbidity, but large storms and hurricanes can result in large scale destruction of SAV by direct physical tearing and uprooting of plants. Strong currents can be generated that run parallel to the shore (Havens et al. 2001), and coupled with large wind-driven waves, can uproot submerged plants. Chimney (2005) reported large north to south seiches during Hurricanes Frances and Jeanne in September of 2004. Similarly, Hurricane Wilma caused large east to west seiches in October of 2005. These seiches deposited large quantities of aquatic plants along LO's shore. Although monthly transect sampling data suggested that the SAV community had been severely affected by all three hurricane events, a direct cause/effect relationship could not be determined because of the delay in sampling relative to the passage of the storms. However, observational and monitoring data collected within two weeks of the passage of Hurricanes Jeanne and Wilma indicated that a rapid decline in SAV density and distribution had occurred. Although this phenomenon would occur sporadically and is independent of CERP effects, it has potential major consequences for the ecological health of LO. SAV coverage of 54,875 acres in late summer 2004 was reduced to 10,872 acres in late summer 2005 and further reduced to 2,965 acres in late summer 2006 as a result of direct and indirect hurricane effects as indicated by the annual mapping surveys. It is as yet unclear what the timing and pattern of SAV recovery from such extreme forcing events will be.



Present Conditions and Trends Except for anecdotal information, the only quantitative SAV data available for LO prior to 1999 is the work of Zimba et al. (1995) during a 1989-1991 study. The current SAV transect sites similarly located those that were sampled in that study. This allows for some degree of comparison and use of the historical data in the assessment of baseline conditions. However, the last six years of mapping and transect data indicate that LO's SAV community is extremely dynamic and highly sensitive to environmental perturbations as demonstrated by the nearly five-fold change in areal coverage that has been observed between 2000 and 2005. Consequently, the concept of an appropriate baseline may be better expressed as the degree of variability reflected in the 2000-2005 data while CERP targets may need to reflect both an acceptable areal distribution and species composition for SAV (as expressed in the revised RECOVER LO Vegetation Mosaic PM) and a persistence or duration goal (for example, inter-annual variability in areal coverage and species composition around the target goal of 49,000 acres of not more than 15 percent). It also may be necessary to exclude data from periods of major physical perturbations and recovery when performing trend analysis to identify the impacts of CERP and associated projects.

Previous studies of SAV in LO identified water depth and transparency as key environmental variables (Steinman et al. 1997, Havens et al. 2002). Using transect data from the first three years of the monitoring effort, Havens (2003) determined that water depth and the concentration of TSS most strongly correlated with SAV biomass. Results also indicated that if water depths in the shoreline region of LO could be maintained at <2 m and TSS held below 20-30 mg/l, there might be favorable conditions for more widespread SAV growth. This, in turn, might lead to water quality improvements.

Future Developments

- For the foreseeable future, periodic transect and annual mapping data will continue to be collected.
- A number of mesocosm experiments designed to provide inputs for model development are being conducted on subjects including minimum light requirements for individual species growth (e.g. *Hydrilla, Potamogeton, Vallisneria*), seed germination requirements, and species succession and data analysis and manuscript preparation is currently underway.
- Flow data collected along a transect through both a *Hydrilla* and *Vallisneria* bed in 2005 is currently being analyzed and a draft manuscript currently in SFWMD internal review has been prepared The results will be used in the LO hydrodynamic model. Using these field data will enable a better calibration of the SAV component in the model to better assess the effect of SAV growth on alteration of flow patterns in the nearshore region of LO.
- Mesocosm studies are currently underway to understand the dynamics of interspecific interactions and succession of the most common SAV species in LO; triggered by transect and mapping data that indicates a progression from non-vascular to vascular plants and from mono-specific to multi-specific beds.
- A three year field study is underway to relate fish, macrovertebrate and amphibian abundance and species composition to SAV (and emergent) species composition and distribution in anticipation of being able to derive meaningful values for these key ecosystem components based on regular measurements of plant density alone. This study has been significantly delayed due to drought induced extreme low lake levels which are anticipated to persist at least through the summer of 2008, barring the occurrence of a number of major storm events focused over LO and it's watershed between now and then.
- Given the recent increase in frequency of tropical storm passage near LO, development of a pre- and post- wind/wave driven sampling program is important to better capture SAV responses to episodic wind and wave events.

4.5 Lake Okeechobee Hypothesis Cluster-Littoral Zone Vegetation

4.5.1 Abstract

The emergent vegetation provides a variety of benefits, but can be detrimentally impacted

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by high lake stages especially when coupled with large wind-driven waves. Higher stage conditions provide a pathway for elevated nutrients which can disrupt the normal plant community composition, and allow floating exotic vegetation to invade inshore areas. Low water conditions allow seed banks that might otherwise not germinate to do so, and increase the likelihood of fire which helps maintain balance in the plant community. Aerial photography is successfully used to monitor the littoral zone. Lower overall lake stages as a result of CERP project implementation and a new lake regulation schedule is anticipated to improve ecological conditions for emergent plants by improving system-wide water storage which will reduce the frequency of extreme high and low lake levels.

4.5.2 Introduction and Background

The littoral zone emergent vegetation is a diverse mosaic of native and exotic plants covering an area larger than 400 km^2. It provides nesting habitat and food resources for economically important fish populations, wading birds, alligator, and the endangered Everglades snail kites. Along the shoreline, emergent plants also help stabilize sediments, support attached algae that help to remove P from the water, and provide a substrate for macro-invertebrates, an important food resource for fish. Dense stands of emergent plants protect submerged plant beds by reducing their exposure to waves. The distribution and abundance of emergent plants are strongly influenced by hydroperiod nutrient inputs and exotic vegetation. The conceptual model below (***Figure 4-22***) summarizes environmental interactions that are known to affect emergent vegetation density, aerial distribution and species composition in the littoral zone of LO.

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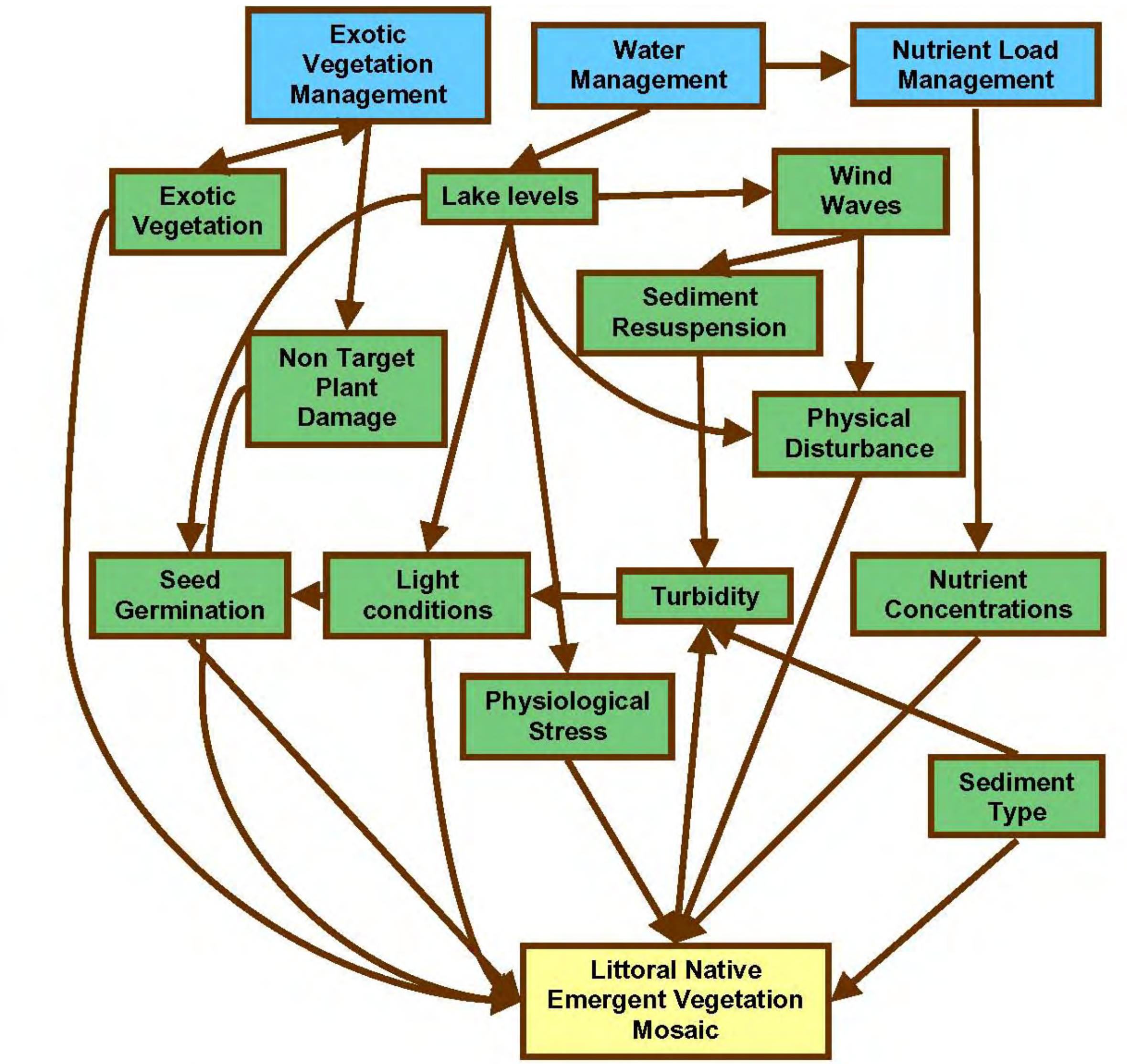

Figure 4-22: Littoral Zone Emergent Vegetation Mosaic Conceptual Ecological Model

Prolonged periods of high lake stage have direct and indirect negative impacts on native shoreline including bulrush (*Scirpus californicus*), SAV and interior marsh vegetation. Native shoreline vegetation is more likely to be uprooted by wind driven waves when lake stage is high. Following a reduction in the spatial extent of rooted macrophytes, turbidity will increase, light availability will be reduced and plant growth will be inhibited due to poor water quality conditions. Additionally, the transport of pelagic water (TP > 100 parts per million [ppm]) into interior regions of the marsh where TP concentrations are often less than 15 ppm occurs mostly at higher lake stages (e.g > 14ft m.s.l.). An increase in nutrients in the interior marsh will result in the loss of desirable vegetation such as spikerush and the expansion of cattail and other less desirable vegetation.

Physically, rooted macrophytes help stabilize bottom sediments thereby reducing sediment resuspension during wind/wave events. During the late 1990s shoreline vegetation was commonly exposed to inundation depths > 2 m. This resulted in the uprooting and elimination of thousands of acres of emergent macrophytes. The loss of shoreline vegetation also was accompanied by an increase in turbidity and a decrease in light availability. The negative feed back loop associated with high lake stage, decreases in the spatial coverage of rooted macrophytes, and declines in water quality will further inhibit the growth of desirable rooted vegetation.

Depth and duration of flooding are also important in determining the distribution of emergent macrophytes. In deep water, emergent species may not have enough leaf area above the surface of the water to obtain the oxygen needed for respiration and/or the carbon dioxide needed for photosynthesis. Reduced oxygen uptake through the leaves can lead to inadequate supplies of oxygen to the roots and rhizomes and eventually lead to plant death. Thus, high lake stage creates physiological stress in rooted emergent macrophytes that can result in plant death if the water depth exceeds a plants flood tolerance (Van der Valk 1994).

Seeds of a number of desirable emergent species (for example bulrush) will not germinate under flood conditions. Therefore, in the absence of draw downs, recruitment of new plants from the seed bank will not occur. Prolonged high lake stage inhibits/prevents the germination of many desirable plant species in the marsh (Williges and Harris 1995). Without recruitment of new plants from the seed bank, the expansion and persistence of desirable marsh vegetation will occur only from vegetative reproduction.

Additionally, floating exotic vegetation can have a negative impact on bulrush and other native plants which is further exacerbated under high lake stage conditions. High lake stage enhances the wind driven transport of floating exotics (water hyacinth and water lettuce) from previously isolated locations (interior areas of Torry and Kreamer Islands) and from the watershed into open shoreline regions of the marsh. These exotics, especially water hyacinth, commonly form large floating mats that exceed 50 m in length. These mats can cause extensive physical damage through uprooting and/or breaking emergent plant stems (e.g., bulrush) as they are pushed around LO by wind and waves.

Exotic and invasive species including torpedograss, *Melaleuca* and cattail grow well in exposed moist soil environments and shallow water habitats. These species commonly form dense monodominant communities that out compete and displace native plant communities, due in part to the absence of their native biocontrol organisms that prevent the exotic plants from becoming invasive weeds in their original range. Although low water conditions favor the growth of many non-desirable species, it also promotes seed germination of desirable native plants and allows for natural and controlled fires which can be effectively used with other management tools to control exotic and invasive species. Periodic low water events occurring with a frequency of approximately once per decade are postulated to provide an appropriate balance between the positive and negative effects of low water events.

The occurrence of low water events accelerates the spread of exotic and nuisance invasive vegetation such as torpedograss, *Melaleuca* and cattail. However, low water events also will stimulate the germination of desirable native vegetation (e.g., spikerush, beakrush and bulrush) and encourage the occurrence of fire which may help control non-desirable exotic and invasive species.

Operating LO at lower overall lake stages and providing periodic recession events will reverse these trends and encourage the expansion of desirable native emergent vegetation.

4.5.3 Methods and Analysis

Emergent vegetation maps based on aerial photography for the entire LO marsh and for the western bulrush fringe has been collected since the mid-1990s and comprises a thorough baseline data set. However, the emergent vegetation community in LO, much like SAV, appears to be very dynamic, responding in a relatively short timeframe to changes in water depth, physical perturbations such as hurricanes and exotic and invasive control operations. Additional research into herbicide treatment effects, seed germination and viability and hydrologic impacts on the recruitment of bulrush and torpedo grass are also being pursued to better understand the changes documented by ongoing mapping activities.

4.5.4 Results and Discussion

Bulrush Giant bulrush (*S. californicus*) stands, located in LO at lake-bed elevations of 10 to 10.5 feet (3 to 3.2 m) NGVD, appeared to suffer damage when exposed to prolonged periods of deep flooding. These bulrush stands provide important fish and wildlife habitat. They also dampen wave energy and stabilize bottom sediments; thus, reducing turbidity and protecting desirable submersed vegetation behind the bulrush barrier. The concern is that excessive inundation of these stands due to prolonged occurrence of high stage levels might cause their failure. Loss of the protective bulrush stands might cause a cascade of events leading to loss of other native vegetation and degradation of water quality and wildlife habitat. An evaluation of the influence of water depth on the persistence of giant bulrush was conducted to support prudent management of LO and minimize adverse effects of stage level manipulation.

The results of this study indicate that undisturbed bulrush can persist at a water depth of three feet or less (lake stage of 13–13.5 ft, or 3.9–4.1 m NGVD); however, prolonged periods of water depths greater than three feet (0.9 m) may cause bulrush stands to fail. Disturbances such as herbivory or strong winds appear to reduce the ability of giant bulrush to persist at the three feet (0.9 m) inundation. Based on data collected from this study, inundation of bulrush stands should be maintained at less than three feet (0.9 m) to minimize adverse effects of stage level manipulation on the persistence of giant bulrush.

Emergent Aquatic Vegetation–Vegetation Maps A baseline vegetation map describing the distribution and areal coverage of vegetation in LO's marsh was developed in the early 1970s (Pesnell and Brown 1977). A second and more detailed vegetation map was developed in 1996. The most recent GIS map was developed in 2005 using color infrared aerial photography collected in 2003 (***Figure 4-23***). Analysis of these maps indicates that there have been a number of changes in the littoral

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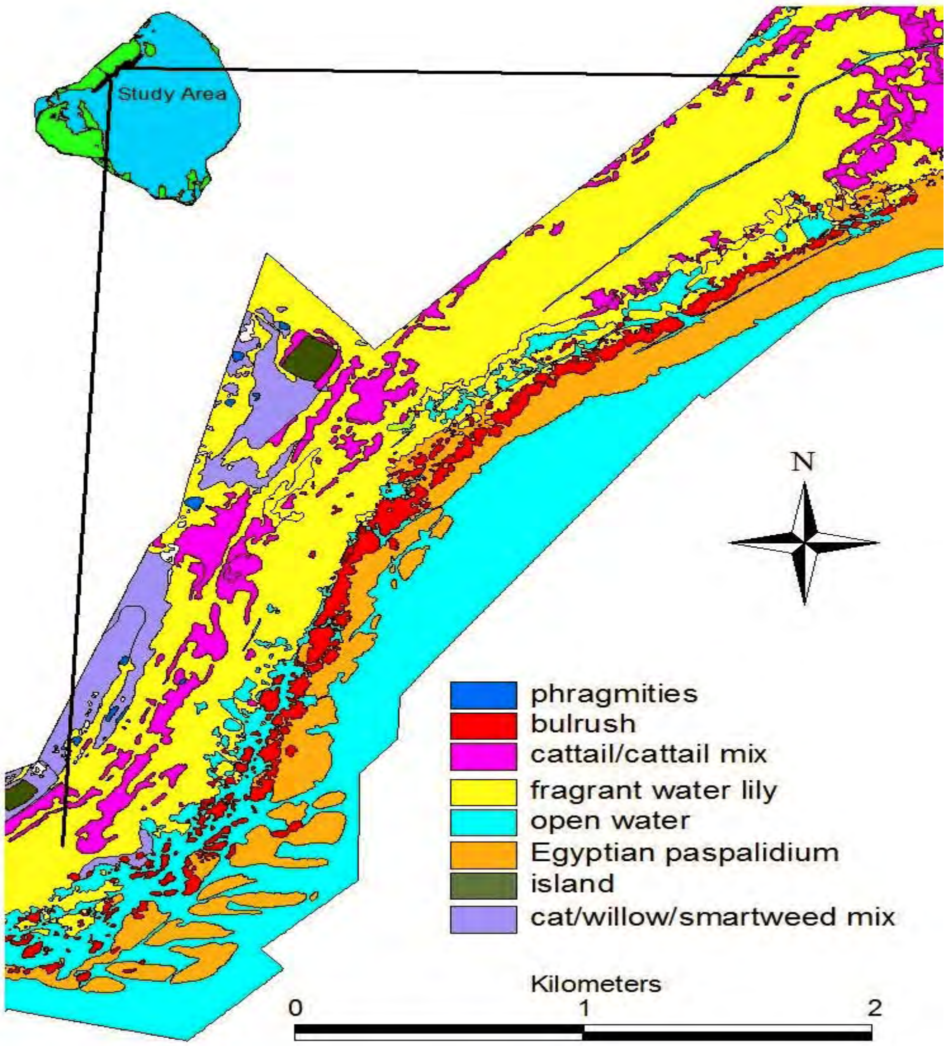
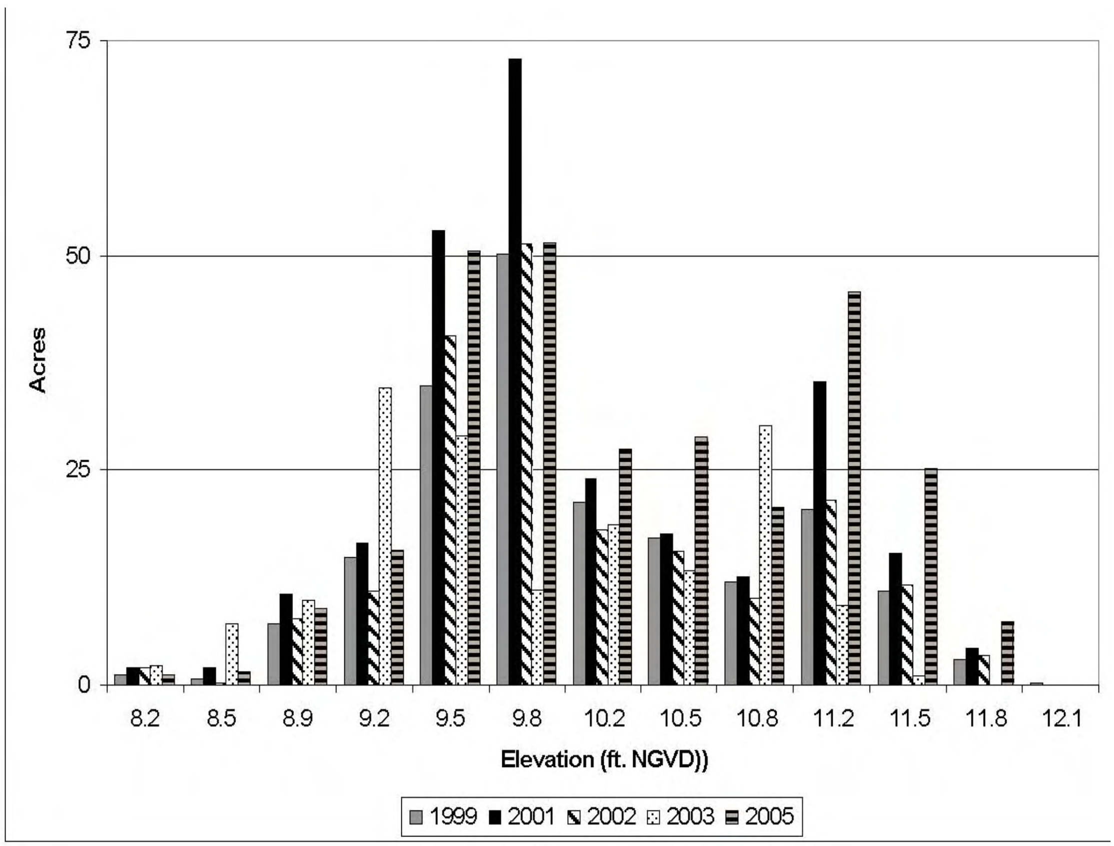


In the 1970s, cattail was located primarily along the lakeward edge of the marsh and covered less than 20,000 acres (8,094 hectares). The dominant emergent vegetation in the interior marsh included beakrush (*Rhynchospora baldwinii*), spikerush (*Eleocharis cellulosa*), mixed grasses and cord grass (*Spartina bakeri*). By 1996, cattail coverage increased to nearly 25,000 acres (10,117 hectares) and was established in some areas of Moonshine Bay. In the upper elevation regions of the interior marsh (shorter hydroperiod) the exotic species torpedograss (*Panicum repens*) displaced more than 13,000 acres (5,261 hectares) of beakrush and spikerush. In regions with longer hydroperiods (e.g., Moonshine Bay), the coverage of fragrant water lily increased to greater than 8,000 acres (3,237 hectares). In 2003 cattail coverage decreased to 23,840 acres (9,648 hectares). The reduction is attributed to large-scale fires and the record drought of 2001 and 2002. Although the total acreage of cattails decreased, the distribution of cattail increased in Moonshine Bay. At elevations generally greater than 13.5 feet (4.1 m) NGVD, torpedograss coverage increased to greater than 17,000 acres (6,880 hectares) despite the treatment of 10,000 acres (4,047 hectares) of torpedograss with herbicide in 2000 to 2002. The distribution of fragrant water lily increased to nearly 11,000 acres (4,452 hectares). Although fragrant water lily is a native, excessive growth of this plant may not be desirable because large amounts of detrital material can accumulate in dense lily beds.

The distribution of bulrush along the northwest marsh edge has been monitored closely since 1999 (***Figure 4-24***). Bulrush coverage varied from 194 acres (78 hectares) in 1999, 266 acres (108 hectares) in 2001, 193 acres (78 hectares) in 2002, 167 acres (68 hectares) in 2003 to 285 acres (116 hectares) in 2005. The increase in bulrush coverage in 2001 occurred in conjunction with a large reduction in lake stage during the drought. The reductions in bulrush coverage that occurred after 2001 occurred in conjunction with prolonged exposure to extreme dry conditions (sediments exposed > four months) followed by exposure to excessive flooding depths that exceeded two meters.



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4.6 Lake Okeechobee Hypothesis Cluster-Exotic Plants

4.6.1 Abstract

Control of exotic invasive plants is an important aspect of the successful restoration of LO. An overview of the current status of ongoing efforts is presented. It is anticipated that CERP project completion will result in a reduction of extreme low lake level events, thereby reducing opportunities for rapidly-spreading invasive exotic plants such as torpedograss (*Panicum repens*) to increase littoral zone areal coverage. Conversely, CERP projects are anticipated to contribute to reduced nearshore TSS concentrations, thereby reducing the competitive advantage exotic SAV taxa such as *Hydrilla* has demonstrated in low water column light regimes.

4.6.2 Background Description

Invasive exotic plants cause significant ecological harm by displacing native vegetation, upon which native fish and wildlife depend for food and shelter (***Figure 4-25***). LO contains approximately 100,000 of acres of littoral zone with herbaceous marshes, other emergent wetlands and numerous islands. More than 80 non-native plant species have been identified in LO. Of these, eight are considered serious, invasive, and/or potentially threatening to the LO ecosystem. Despite intensive control programs, dedicated funding and continual monitoring, some species have proven difficult to control. During fiscal year 2006, SFWMD expended $164,000 on controlling Brazilian pepper (*Schinus terebinthifolius*), $282,000 on Melaleuca (*Melaleuca quinquenervia*), and $816,000 on torpedo grass (*P. repens*) in LO.

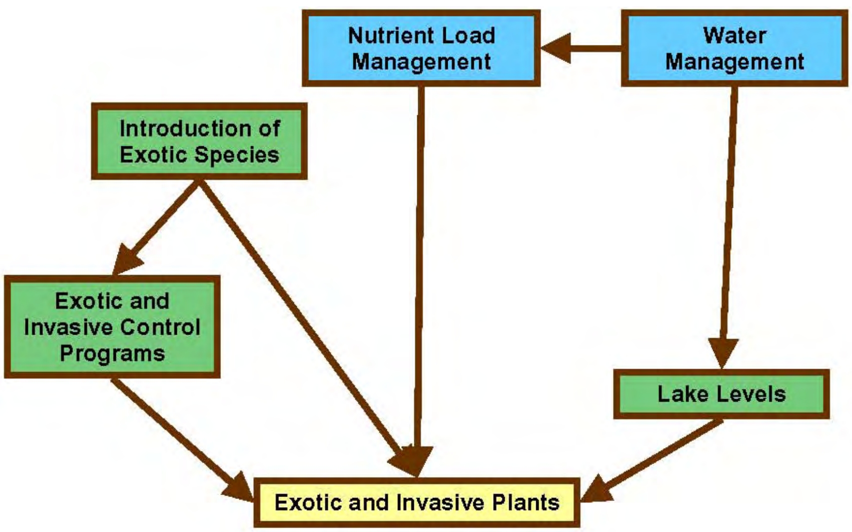

Figure 4-25: Conceptual Model of Stressors and/or Factors That Affect Exotic Vegetation

Floating aquatic plants, such as water hyacinth *(Eichhornia crassipes)* and water lettuce *(Pistia stratiotes)* are managed by a Corps program started in the 1920s. The goal of the program is to keep plants at the prescribed maintenance level (Chapter 369.22, Florida Statutes). In the past 15 years, LO averaged about 240 acres of combined hyacinth and lettuce, with an average of over 5,000 acres being treated each year. Without continued control, water hyacinth and lettuce would quickly expand and cover large areas, reducing light penetration into the water column. Reduced light penetration could result in shading of native SAV and areas of low DO below the canopy of these exotics. If DO concentrations are reduced, fish habitat might be reduced or lost, depending on the severity of the DO reduction. In addition, these floating aquatics tend to form large wind driven mats or tussocks which can mow down and uproot desirable emergent vegetation

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like bulrush and other species.

Alligator weed *(Alternanthera philoxeroides)* has been successfully controlled since the 1960s. Three insects, the alligatorweed flea beetle (*Agasicles hygrophila*), alligatorweed thrips (*Amynothrips andersoni*) and alligatorweed stem borer (*Vogtia/Arcola malloi)* currently keep populations of alligator weed at low levels in LO. Barring any negative impacts to the biocontrol agents, alligator weed is not expected to cause any measurable impacts in the near future, and serves as an example of what successful biocontrol programs can accomplish.

West Indian marsh grass (*Hymenachne amplexicaulis*) is a perennial, stout semi-aquatic grass native to Central and South America. Invading tropical seasonally wet waterways, wetlands and drainage systems, it impedes flood protection and water management. It has overwhelmed riparian systems in many locations worldwide. In LO, it is increasing its range, particularly in Fisheating Bay. Upstream of LO, in Fisheating Creek, West Indian marsh grass has established dense populations along the edge of the creek and in the cypress forest understory. To date, very little control of West Indian marsh grass has occurred in LO, and estimates of its population already range to 100 acres (Mike Bodle, SFWMD, personal communication). SFWMD initiated a herbicide control program for this species in 2005 within the DEP aquatic plant control program.

Torpedograss (*P. repens*) has been the target of extensive control in LO's western marsh. By 1996, torpedograss had displaced more than 16,000 acres of native plants and shallow open water habitat. Torpedograss can tolerate periods of deep flooding but spreads most rapidly on moist soil or when exposed to shallow water column depths. During the 2000-01 drought, the areal coverage of torpedograss increased to greater than 20,000 acres. Despite widespread aerial treatments in 2002 through 2005, large areas remain affected. Since 2000, nearly 25,000 acres of torpedo grass were treated with some areas requiring one application while more stubborn infestations required repeated application; yet despite these efforts about 7,400 acres still remain. Torpedograss coverage was estimated in 2006 to comprise approximately eight percent of the total marsh area in LO. Recent data collected by the SFWMD staff indicates that DO concentrations can be significantly lower and more diurnally variable in torpedograss compared to native spikerush (*Eleocharis cellulosa*) habitat (Rodusky personal communication). These data suggest that torpedograss may not be as suitable as spikerush as fish habitat. Fish, periphyton, zooplankton and macroinvertebrate data recently collected in both of these habitats are being compared to gain some insight as to the food web structure for higher trophic level organisms that utilize both habitats for food and as a refuge.
Non-indigenous plant species considered a priority in the LO module are listed in ***Table 4-1***. Recently, the first population of Old World climbing fern (*Lygodium microphyllum*) was reported in the western marsh in July 2006. This exotic may be added to future priority lists if control efforts are not undertaken in the near future.

4.6.3 Methods and Analysis

Stated hypotheses are evaluated using the most recent data available.

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Table 4-1: Status and Prognosis Table for Priority Invasive Plant Species					
PERFOMANCE MEASURE	LAST STATUS[a]	CURRENT STATUS[b]	2-YEAR PROSPECTS[c]	CURRENT STATUS[b]	2-YEAR PROSPECTS[c]
Submerged Aquatic Vegetation Areal Coverage NEARSHORE REGION				Submerged aquatic vegetation (SAV) coverage, especially vascular plant coverage, decreased dramatically since the fall of 2004. This decline in areal coverage was caused by physical disturbance (uprooting) from three hurricanes (Frances, Jeanne and Wilma) followed by prolonged water column turbidity. *Chara* spp. coverage dramatically increased during 2007, covering approximately 27,700 acres. However, vascular plants accounted for only approximately 500 total acres.	Unknown. Most of the nearshore region known to contain SAV over the past decade has been dry for approximately the past 9-12 months. Seed-bank viability in these areas is unknown. The SAV response to reflooding upon the return to average lake stages is, therefore, uncertain at this time.

4.6.4 Results and Discussion Hypothesis-Under physical conditions that results in low light levels, the exotic SAV species
Hydrilla (*Hydrilla verticillata*) may have a competitive advantage over more desirable native SAV species.

Rationale Mesocosm experiments conducted under natural light indicate that *Hydrilla* has a lower light requirement (***Figure 4-26***) than both *Vallisneria* and *Chara*, the major SAV species tested from LO to date (Grimshaw and Sharfstein in preparation). The minimum light requirements for *Hydrilla*, *Vallisneria* and *Chara* are 1.8, 4.1, and 4.7 percent of incident photosynthetically active radiation (PAR), respectively (Grimshaw et al. 2002, 2005).

Hydrilla has been in LO for about 20 years, but was not a consistent problem. Its acreage varies annually with water clarity, wind, wave action, water level and substrate conditions. In some years *Hydrilla* has expanded rapidly to cover thousands of acres and required mechanical harvesting to open up boat trails. Wave and wind from hurricanes resulted in prolonged periods of elevated turbidity and the corresponding reduction in light availability cannot fully account for the observed reduction in *Hydrilla* populations for the past several years, primarily because *Hydrilla* is very low light tolerant. Arguably, their decrease may largely be because of physical disruption during the storms. Recent Hurricanes Irene (1999), Frances (2004), Jeanne (2004) and Wilma (2005) impacted LO to the extent that virtually no *Hydrilla* was detectable until summer 2006. However, the exponential growth rate of the plant, maintenance of stages favorable to its spread and a few consecutive years free from hurricanes could permit *Hydrilla* to spread rapidly and become a major concern.

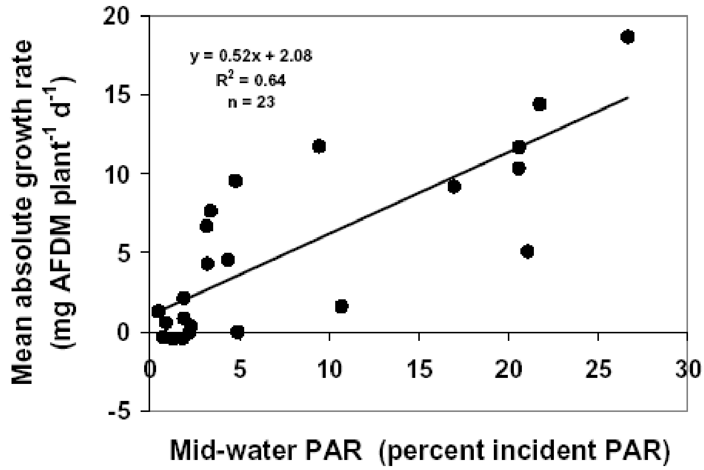

Figure 4-26: Linear Regression of Mid-water PAR Versus the Absolute Growth Rate of Hydrilla From Mesocosm Experiments

Hypothesis-Changes in the extent of mud sediments in the pelagic-littoral fringe zone of LO, resulting from changes in runoff and nutrient loading, influence the potential area available for colonization by desirable SAV.

In LO, SAV colonizes peat and sand sediment but does not grow as well in mud sediments. Changes in runoff and nutrient loading are expected to reduce area of the lake experiencing high turbidity, thereby increasing the area potentially available for colonization by SAV.

Results to date seem to contradict this statement (***Figure 4-27***). *Hydrilla* and *Ceratophyllum* will colonize mud sediments if they are in an area where sufficient light is reaching the bottom; however, the more desirable native *Vallisneria* appears to prefer sand and peat substrates. The major colonizer of rock substrate is the non-vascular macroalga *Chara*.

Overall, coverage of exotic emergents such as torpedograss and SAV such as *Hydrilla* are anticipated to be reduced after CERP projects are on-line. Reduced frequency of extreme low lake levels favored by torpedograss may result in less littoral area coverage and less effort required to maintain this limited coverage. In the nearshore region of LO, improved light penetration into the water column after CERP project implementation may enhance the ability of native SAV taxa such as *Vallisneria* to outcompete exotics such as *Hydrilla*.

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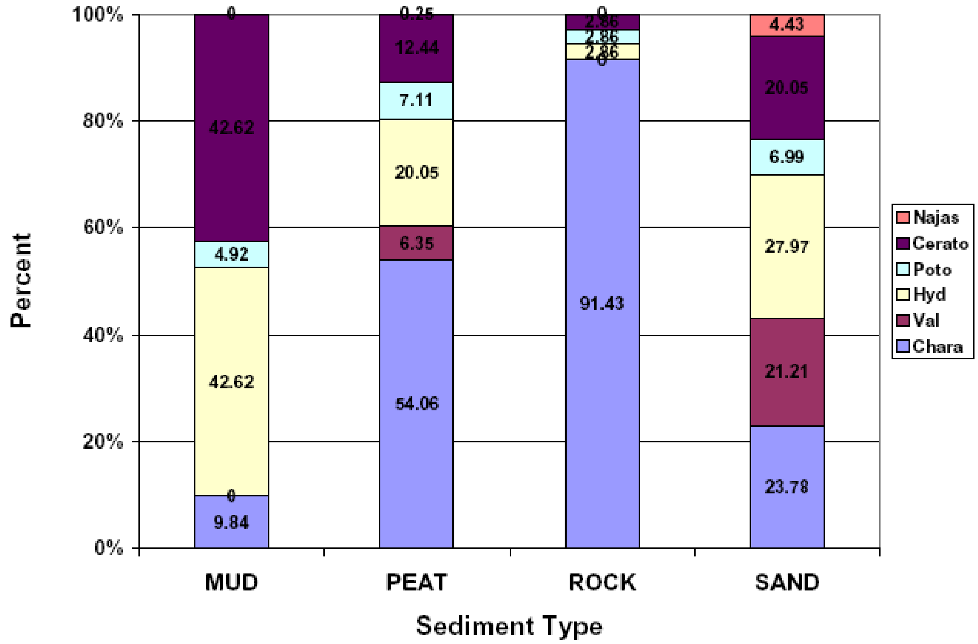

Figure 4-27: Percent of each Submerged Aquatic Vegetation Species As a Function of Sediment Type From the 2001 to 2005 Annual Mapping Data

4.7 Lake Okeechobee Hypothesis Cluster-Phytoplankton Dynamics

4.7.1 Abstract

Phytoplankton monitoring is an important component of LO research. LO is designated as a P Class I drinking water source by the DEP. Approximately 60,000 people rely on LO as their primary source of potable water. Periodic large-scale surficial blooms since the 1980s, the last of which occurred during the summer of 2005, has elevated concern regarding cyanotoxins and potential adverse health effects for wildlife, livestock and humans. In addition, phytoplankton is one of the primary producers at the base of the pelagic and nearshore food webs and as such is an important food source for numerous organisms. Data collected as part of a long-term monitoring program indicates that phytoplankton community has been shifting from one dominated by diatoms in the 1970s to a community dominated by cyanobacteria since the 1990s (Havens et al. 1996). Most of the phytoplankton taxa are not readily grazed by zooplankton, suggesting that energy transfer from phytoplankton up the food web to the higher trophic levels may be less important than along microbial pathways. Continued excessive nutrient loading from the watershed, fluctuating climactic events ranging between excessively dry and wet years and the passage of three hurricanes during 2004-2005 may be factors which are influencing changes in the phytoplankton community. The long-term data set will be useful to establish pre-CERP implementation conditions and to assess if CERP projects contribute to the restoration of a more diverse, heterogeneous phytoplankton assemblage that is dominated or co-dominated by diatoms rather than cyanobacteria.

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4.7.2 Background Description

Phytoplankton research has been conducted in the pelagic and nearshore regions of LO since the late 1960s (Joyner 1974). Studies conducted during the 1970s and the early 1980s indicated that phytoplankton assemblages were spatially and temporally heterogeneous (Chichra et al. 1995). Several large surficial blooms during the mid 1980s were interpreted as a shift in the phytoplankton community to that of an increasingly eutrophic lake. Havens et al. (1995) found that bloom frequencies, defined as chlorophyll *a* concentrations >40 ppb, increased during the 1980s and were positively correlated with water temperature but inversely correlated with total and soluble N and P, and wind velocity. Maceina (1993) identified a positive relationship between lake stage and chlorophyll *a* concentrations in the littoral and nearshore regions of LO. Maceina (1993) hypothesized that a higher lake regulation schedule, implemented in 1978 for water supply, resulted in greater movement of nutrient-rich pelagic water into the nearshore and littoral regions of LO, thus stimulating increase phytoplankton biomass. Phlips et al. (1994) found spatial variability in LO phytoplankton biomass (as chlorophyll *a*) over a 17-year period, and suggested that the phytoplankton were light-limited in the central pelagic region and nutrient-limited in the less turbid nearshore region.

Phytoplankton studies conducted as part of the Lake Okeechobee Ecosystem Study (LOES) from 1988 to 1990 indicated that the phytoplankton community was dominated by cyanobacteria at 14 of the 21 study sites. Diatoms dominated at the six remaining sites, and co-dominated with cyanobacteria at the remaining site (Chichra et al. 1995). During this same period, spatial and temporal changes in phytoplankton biomass were related to variability in nutrients, lake stage, light availability and wind speed (Phlips et al. 1995). Smith et al. (1995) suggested that a significant decline in the TN to TP ratio (TN:TP) in LO from the early 1970s to the 1990s coincided with increased planktonic N limitation and suggested that this may represent an increased potential for blooms of N-fixing cyanobacteria. Bioassays conducted as part of LOES indicated that phytoplankton were generally N-limited, though other factors such as light were also posed as co-limiting factors in roughly a third of the assays (Aldridge et al. 1995). In a subsequent bioassay study conducted from 1997 to 2000, light was the most common limiting factor followed by N and
N:P co-limitation (East and Sharfstein 2006). During this period, the phytoplankton in LO were either light-limited or nutrient-limited, with limitation being determined seemingly as a function of irradiance-related parameters. Light limitation was more prevalent during the windier, more turbulent winter months and nutrient limitation was more dominant during the summer.

In response to a legislative mandate to restore LO, a phytoplankton monitoring program in the pelagic and nearshore regions of LO was initiated in 1994 and is currently conducted on a quarterly basis (East and Sharfstein 2006). As part of this research, phytoplankton abundance (as chlorophyll *a* and biovolume) and community composition are determined at five sites (***Figure 4-28***). Since 1997, photosynthesis-irradiance (P-I) curves have been generated using *in situ* water from these five sites, to determine whether light or nutrients are limiting phytoplanktonic photosynthetic activity. The P-I data suggest that low lake stage is highly correlated with photosynthetic parameters, suggesting an ecological heterogeneity, while under high lake stage, the parameters do not vary, suggesting that phytoplankton are ecologically homogeneous among sites (Maki et al. 2004). Additionally, cyanotoxin monitoring commenced in 2004 and is currently being conducted on a monthly basis at seven sites near major inflows or municipal water intake structures.

Overall Goal CERP RECOVER targets are to reduce the dominance of cyanobacteria and increase diatoms such that the diatom to cyanbacteria ratio becomes greater than 1.5:1. A decrease in P inputs to LO as part of CERP implementation and basin control efforts is expected to increase the TN:TP ratio to above 22:1, and decrease cyanobacterial bloom frequency and bloom composition, with cyanobacteria comprising < 50 percent of bloom composition. There currently are no biovolume or abundance targets set forth for phytoplankton as part of CERP.

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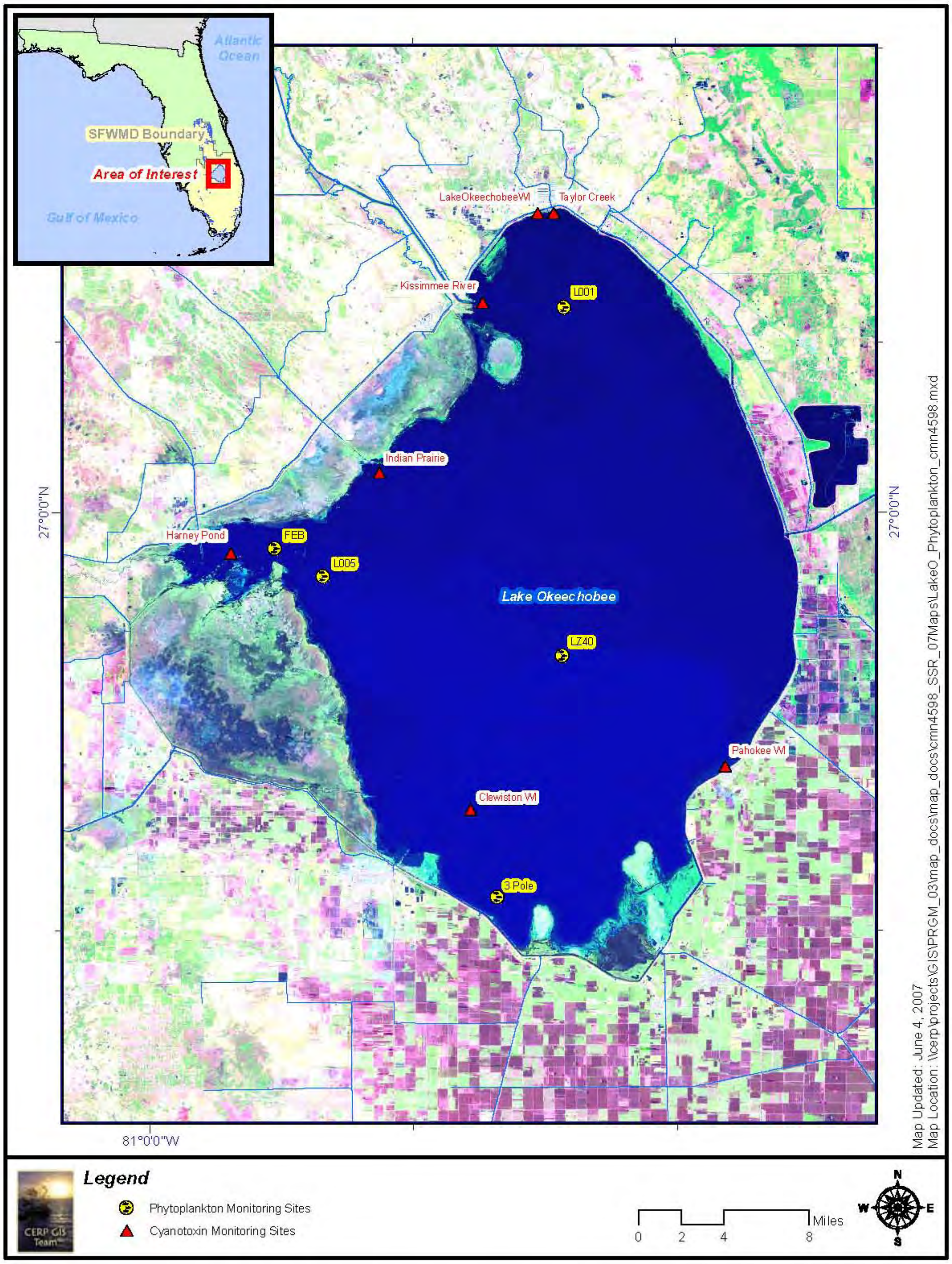


The unlabelled nearshore site was excluded from some of the analyses.

Figure 4-28: Phytoplankton (Star Shape) and Cyanotoxin (Sun Shape) Monitoring Sites on Lake Okeechobee

4.7.3 Methods and Analysis

Phytoplankton monitoring consists of collecting water from all but the bottom 0.5 m of the water column with an integrated sampling tube, on a quarterly basis as described in East and Sharfstein (2006). Water is collected for biomass determination (as chlorophyll *a*), community taxonomic composition and for laboratory bioassays which are used to determine whether light or nutrients are potentially limiting phytoplankton growth (East and Sharfstein, 2006). Additionally, P-I curves were generated using additional water samples. These P-I curves were used to evaluate how photosynthetic characteristics varied among sites located in ecologically distinct regions of LO (Phlips et al. 1993, Maki et al. 2004). Physical water quality data are also collected from a sonde and datalogger unit, while water chemistry is measured either by sonde or from surficial water grab samples.

<u>Community Composition</u> Samples collected for phytoplankton community composition have been enumerated to species or lowest practicable level, and are reported as biovolumes (μm^3/mL). Two contract taxonomists have conducted phytoplankton identifications since 1994. Both contractors identified samples 1994-1996, and the current contractor has been identifying samples since 2000. Sampling frequency was reduced to quarterly in 2003 and has remained at that frequency. Therefore, all community composition analyses consist of quarterly samples collected from 1994-1995 and 2000 to present, with samples collected at four of the five sites (two nearshore, two pelagic). The fifth site was not included in these analyses because it was not monitored during 1994-1995. Differences in replication exist between the 1994-95 and the 2000-06 datasets. Ordinations should be considered exploratory at this time.

<u>Biomass Determination</u> Chlorophyll *a* concentrations are determined spectrophotometrically following grinding of filtered water samples followed by extraction of the pigments in 90 percent acetone, following standard methods (APHA 1995). Total biovolumes are only reported in this section for 2000-2006 because these data have not yet been separated by taxonomist for 1994-1995. Therefore, only data for the current taxonomist were used. Data reported in this section are from the same nearshore and pelagic sites previously described.

<u>Light and Nutrient Bioassays</u> Bioassays were conducted to measure phytoplankton photosynthesis and determine whether light or nutrient concentrations limited photosynthetic activity. These bioassays were conducted using five light levels as outlined in Maki et al. (2004). The bioassays were run within 24 hours of sample collection, at one of 20 irradiance levels ranging from 0 to ~ 1000 µmol photons m2 s-1 at ambient lake temperature, for one hour.

<u>Diatom to Cyanobacteria Ratio</u> Diatom to cyanobacteria ratios were calculated from the percent total biovolumes for both Divisions.

<u>Cyanotoxin</u> Cyanotoxin samples were collected from surface water grabs near three municipal water intake structures and four inflows to LO. All sites were located at nearshore locations in the north, west, and eastern part of LO (***Figure 4-28***). Microcystin, anatoxin and cylindrospermopsin concentrations were determined at a contract

laboratory.

Results

Community Composition This data set comprises quarterly data from one contractor. Split samples collected between 1994 and 1996 and analyzed by both taxonomists suggests that there are significant differences among both primarily in several of the cyanobacteria taxon identifications, as evaluated by the Analysis of Similarity (Anosim) test (ANOSIM, Global R = 0.73, p=0.001). Since one taxonomist identified all samples from 1996 through 1999, and identified a smaller set of samples, these samples have been excluded from this analysis.

Nonmetric multidimensional scaling (NMDS) ordination analysis of the community data suggests that year and then season were the two most significant factors. Among years (all sites combined), displayed the clearest separation between the phytoplankton communities (R<0.83, p=0.001, ***Figure 4-29***). The group patterns suggest that while there was some differences in the community structure between 1994 and 1995 (R=0.52, p=0.001), there was relatively clear differences among the groups (R>0.7, p=0.001) between 1994, 1995 and each year from 2000-06. Some of the most significant (R>0.9, p=0.001) among-years differences occurred between 2000 and each of the subsequent years. During 2001 a lake recession and prolonged drought occurred and lake stage decreased by May of that year to roughly 1.7 m below the long-term seasonal average, which was, at that time, the lowest ever-recorded lake stage (since exceeded in June 2007). There was little difference between the communities in 2005 and 2006 (R=0.13, p<0.10).

The within-year phytoplankton communities had mean similarity percentages that ranged from a high of 45 percent (1994) and were <28 percent for each year between 2000 and 2006. These values represent mean taxon contribution to the community structural similarity among samples for each year and suggest that there was substantial within-year variability during 2000–06. Mean dissimilarity percentages between each yearly comparison ranged between 67 percent (1994 and 1995) and 98 percent (1994 and 2006). This among-year variability is illustrated in dendritic form in ***Figure 4-30***. The taxa that were most dissimilar among 1994 and 2006 were the cyanobacteria taxa *Lyngbya limnetica, Lyngbya contorta, Anabaena flos-aquae* and *Anabaena cicinalis*. All of these taxa were abundant in 1994 (> 1.3 x 10^5 µm^3/mL) and with the exception of *Anabaena circinalis* (107 µm^3/mL), were not identified in the 2006 samples. The same taxa had the biggest contribution to the differences between the 1995 and 2006 assemblages, where the dissimilarity percentage between the two years was nearly as high (97 percent) as it was between 1994 and 2006.

In general, cyanobacteria taxa comprised three or four of the top five taxon that contributed most to the relative dissimilarity. Diatoms were much less important, although they did comprise one to two taxa which made significant contributions to the among-groups dissimilarity values. In these cases, it was diatom taxa which were found primarily in 2001– 2006. Among the 2000–2006 group comparisons, there existed a general mix of three diatom and one or two cyanobacteria taxa which contributed most significantly to the among-years dissimilarity values. These results suggest that the phytoplankton assemblage experienced an increase in diatom importance and variability after 2000. However, community variability also may have been due in part to variability in sample identifications (e.g., taxonomic drift).

Separation among the community on a seasonal basis was less clear ($R<0.44$, $p=0.001$, ***Figure 4-31***). The largest separation, which was fairly significant, was between winter and fall ($R=0.66$, $p=0.001$) and the smallest separation, which was marginal, was between winter and spring ($R=0.37$, $p=0.001$). The same taxonomic pattern described for the among-year comparisons was observed for this data set, though there were typically one or two cyanobacteria taxa which contributed to the among-season differences than was observed in the among-years communities. It should be noted that the stress value associated with both the two dimensional among-years and among-seasons plots was sufficiently high to caution their use for anything beyond examination of general trends (per guidelines presented in Clarke and Warwick 2001).

There was very little difference among sites ($R=0.11$, $p<0.01$), whether examined on an among-years or among-seasons basis. The largest separation was between the communities at sites L005 and LZ40, but the amount of separation ($R=0.32$, $p=0.001$) was marginal. These comparisons suggest that temporal factors were more important than geographic location in influencing the community structure and that overall, there was little discernable separation and difference in the phytoplankton community structure among each site. Separating the data into years of lower lake stages (e.g., years <14 ft msl) and higher lake stages (e.g., years >16 msl) may have yielded better separation among sites, as photosynthetic behavior was shown to homogenous among sites during higher lake stages and heterogeneous under lower lake stages (Maki et al. 2004).

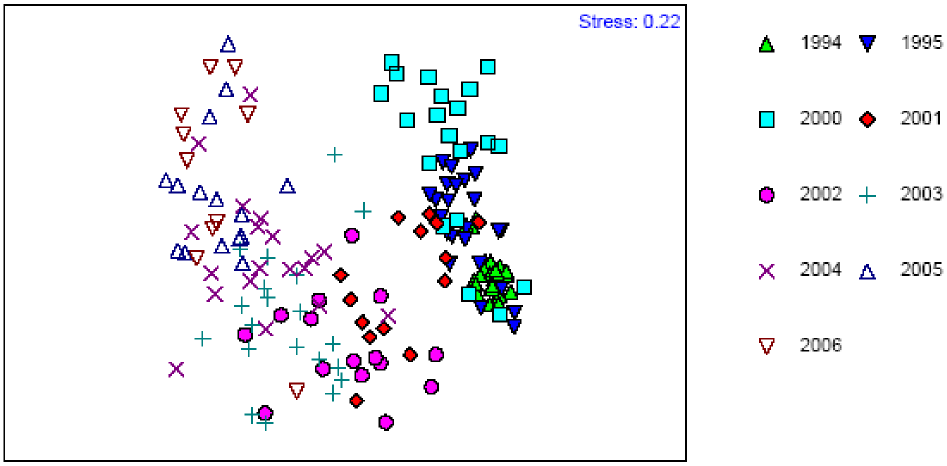
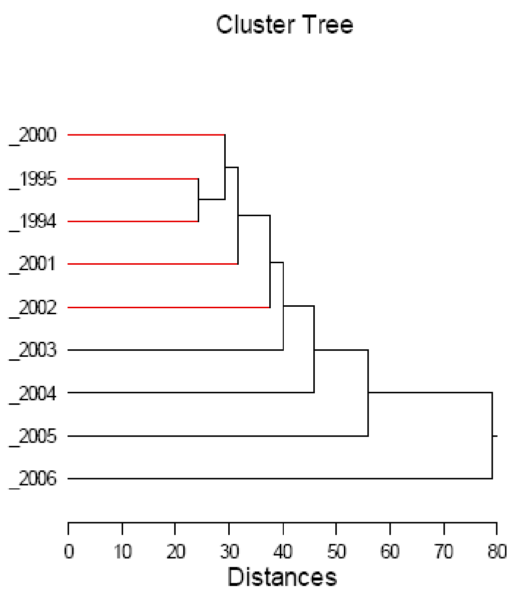


Figure 4-29: Phytoplankton Community Ordination Plot by Year



The distances from mean Bray-Curtis similarity coefficients on square root transformed biovolume data.

Figure 4-30: Dendrogram Representation of Among-years Differences in the Phytoplankton Communities

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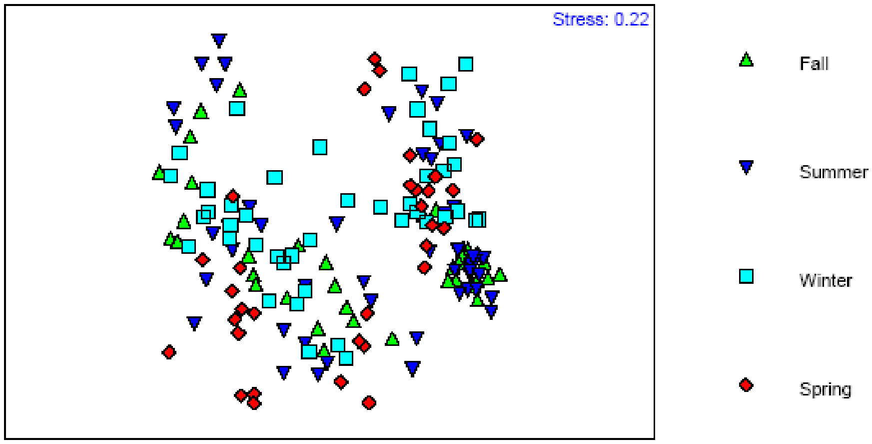

Figure 4-31: Phytoplankton Community Ordination Plot by Season

Attempts to correlate 12 years of water quality data to phytoplankton community structure in LO were not conclusive. Stepwise addition of water quality variables suggested a positive but weak relationship (Spearman ρ=0.284) between a combination of Secchi disc depth to total depth (SD:TD), TSS, pH, mean wind speed, lake stage and the phytoplankton community composition. Similarly weak positive correlations between combinations of subsets of these variables also were observed.

<u>Biomass Determination</u> Biomass defined as mean annual total biovolumes were variable among the nearshore and pelagic sites, and appears to be similar among site types for most years (***Figure 4-32***). Mean annual biovolumes appear to be significantly lower in 2006 relative to the other years and may be related to extremely low light levels in the water column since the passage of Hurricanes Frances and Jeanne in 2004 and Wilma in 2005. Mean annual biovolumes varied between 48,000 μm^3/mL in 2006 (pelagic sites) to 1,900,000 μm^3/mL in 2001 (nearshore sites).

Biomass as mean annual chlorophyll *a* concentrations were less variable and very similar among site types for all years (***Figure 4-33***). Mean annual chlorophyll *a* concentrations were generally between 10 and 20 µg/L. Algal bloom frequency, as previously defined (Havens et al. 1995) was infrequent during this period. Blooms were observed on average once a year (from quarterly samples) at either of the nearshore and pelagic sites. A large surficial bloom was observed in August, 2005, but these blooms occurred between the summer and fall quarterly sampling events.

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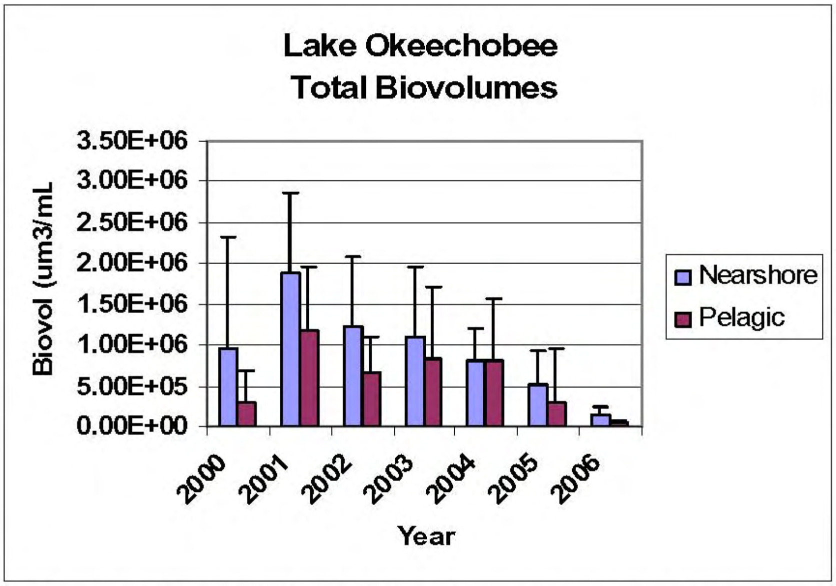

Figure 4-32: Annual Mean Phytoplankton Biomass in Total Biovolumes ± 1 S.D. at Both Nearshore and Pelagic Sites

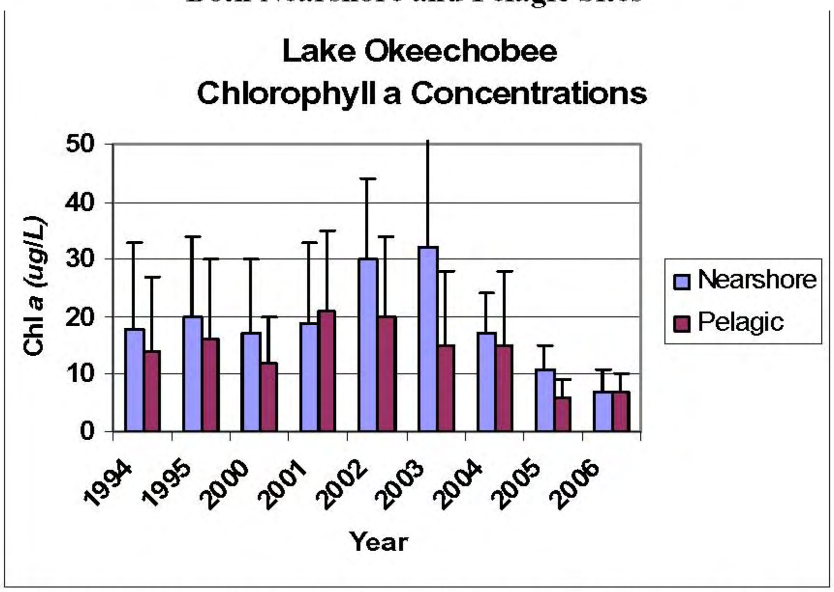

Figure 4-33: Annual Mean Phytoplankton Biomass as Chlorophyll *a* ± 1S.D. at Both Nearshore and Pelagic Sites

Light and Nutrient Bioassays Long-term (1997-2000) bioassay results indicate that light limited phytoplankton growth approximately 60 percent of the time, while N limited growth the remaining time (East and Sharfstein, 2006). While these bioassays continue to be conducted on a quarterly basis, the data analysis is on-going.

Diatom to Cyanobacterial Ratio Diatom to cyanobacteria ratios have been < 1:1 since the mid 1990s (***Figure 4-34***). Since 2003, it appears that ratios have been increasing, at both the nearshore and pelagic sites, such that they have exceeded the PM since 2004. Since 2004, the diatom *genera Fragilaria, Aulacoseria* and *Cyclotella* have become increasingly important in both among-years biovolumes and how frequently they are found in each sample. This achievement of the PM target during a period of time when LO is in notably poor condition as a result of the hurricanes and high water levels, brings into question the validity of the PM. It may be that what is being measured is resuspended meroplankton rather than the diatom and cyanobacterial assemblage. Nevertheless, meeting restoration targets without restoration during extremely poor conditions suggests that either the PM should be modified to specify that the diatom species in question are typical pelagic organisms, or alternatively dropped altogether.

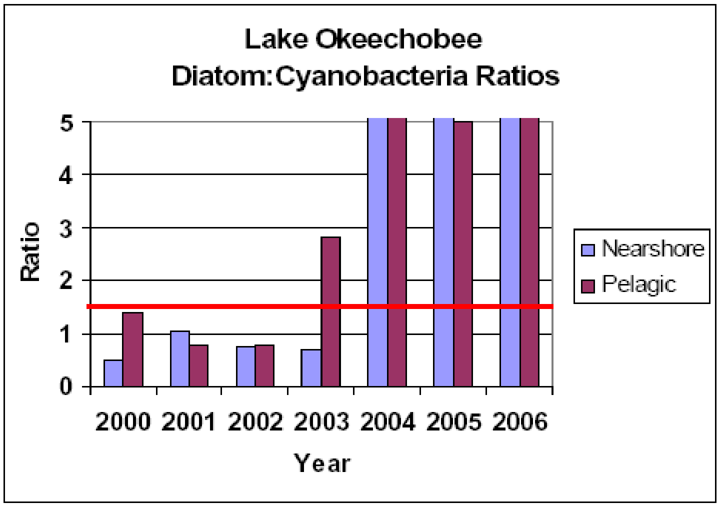

Figure 4-34: Annual Mean Diatom to Cyanobacteria Ratio at Both Nearshore and Pelagic Sites

Cyanotoxin Mean microcystin concentrations are generally low (e.g., <5µg/L), but were considerably higher during late summer (August-October) in 2005 (***Figure 4-35***).

4.7.4 Discussion

Phytoplankton has received considerable study on LO and past research has suggested that lake stage, nutrients, and light availability affect the phytoplankton (Philps et al. 1993, 1995). There also has been a shift in dominance from a diatom-dominated to a cyanobacteria-dominated assemblage that has coincided with cultural eutrophication of LO (Havens et al. 1996). Blooms have become more frequent since the 1980s (Havens et al. 1995) and conditions have become increasingly favorable for blooms to be comprised primarily of N-fixing cyanobacteria as the TN:TP ratio has declined (Smith et al. 1995).

The variability in the 1994-95 and 2000-2006 community composition data suggest changes that may be reflective of the dynamic climatic events experienced by LO over the past decade. Lake stage has fluctuated between a historical high of 18.5 feet msl during an extremely wet 1995 and a historical low of 8.97 feet msl following a lake recession and prolonged drought in 2001. The current prolonged drought has resulted in a minimum lake stage of 8.84 msl, recorded during June, 2007. Additionally, three hurricanes passed very near LO between 2004 and 2005, and turbidity levels became

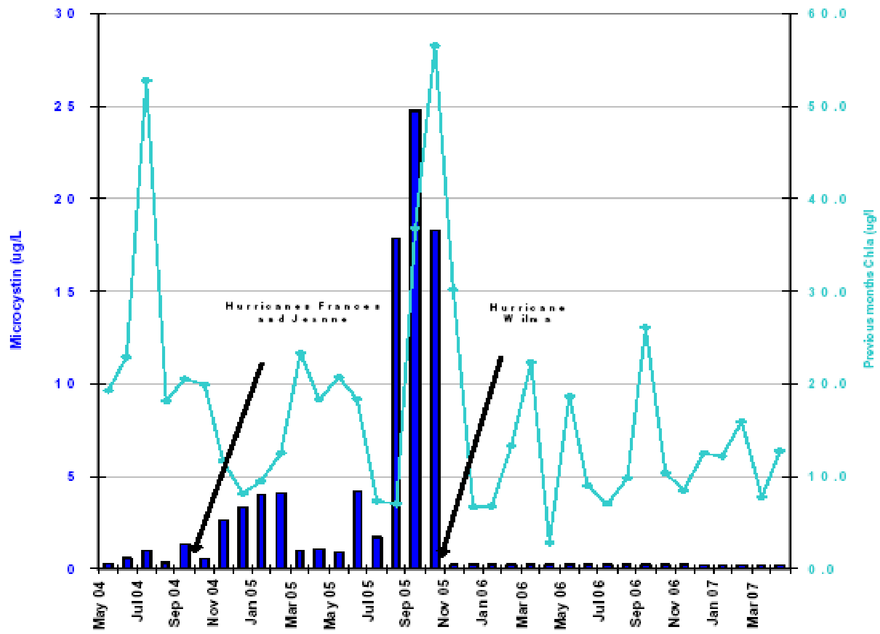

Figure 4-35: Mean Microcystin Concentrations (µg/L)

extremely high (e.g., over 100 ppm TSS) for up to six months. During 1994 and 1995, taxa which contributed most significantly to within-year similarity values were predominantly the cyanobacteria genus *Lyngbya*, *Anabaena*, *Oscillatoria* along with the diatom *Melosira* and the cryptomonad genera *Cryptomonas* and *Rodomonas*. While the cyanobacteria taxa *Lyngbya* and *Oscillatoria* continued to play the most significant part in within-group similarity for 20002002, these similarities have been increasingly influenced by diatom genera such as *Fragilaria*, *Aulacoseria* and *Cyclotella*. Since 2004, at least four of the top five most similar within-year taxa for each have been diatoms, suggesting that they are more consistently being found in samples. Several of these taxa, such as *Thalassiosira proschkinae*, and species of the genera *Aulacoseria*, *Cyclotella and Stephanodiscus* are nutrient tolerant and indicative of eutrophic or hypereutrophic conditions (Yang et al. 2005).

Based on past research, it was surprising that there was very little difference among sites, whether evaluated on a yearly or seasonal (quarterly) basis. This contrasts with the oft-utilized heterogeneous characterizations of LO and subsequent development of the ecological zones concept delineating pelagic and nearshore regions of LO (Phlips et al. 1993), including phytoplankton spatial heterogeneity (Aldridge et al. 1995). The lack of significant separation among sites may be due to how the data were aggregated and it may be that examination at a finer scale would reveal differences among sites. Alternatively, these results suggest that phytoplankton may not be as spatially heterogeneous as they were during previous studies.

The weak correlations between water quality variables and community composition suggest that relationships are complex and community structure is likely dependent on dynamically varying individual or composites of water quality factors at different times and under different conditions. This may also suggest that unmeasured variables play an unexpected role, or that the frequency at which variables were measured was insufficient to define their sway on phytoplankton community structure (e.g., inability to appropriately lag data due to the coarseness of its periodicity). However, since phytoplankton generation times are often on the order of one day or less, the discontinuity between water quality data—most of which were collected the same day as phytoplankton samples—and which included light and nutrient concentration measurements was surprising. Still, it is anticipated that CERP projects will result in reduced nearshore and pelagic zone nutrient concentrations, which may facilitate a shift back to consistent diatom dominance, less frequent algal blooms and blooms comprised of a smaller portion of cyanobacteria than has been observed over the past 20 years.

4.8 Lake Okeechobee Hypothesis Cluster–Fish

4.8.1 Abstract

Decreases in habitat and disruption of the food chain have resulted in decreases in the number and size of fish, as well as a shift to less desirable species. Efforts to resolve problems in water quality and volume (stage) will be reflected as improvements in fish habitat and quality and quantity of fish.

4.8.2 Background Description

Biological integrity of a system may be defined as "maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region" (Karr and Dudley 1981). Fish, besides being the most visible and sought after commodity in most water bodies, are at the trophic pinnacle among aquatic organisms, integrating the effects of both water management and basin development. Fish require a viable foodweb (thus reflecting the status and health of the invertebrate community) and require suitable habitat to avoid predation and ensure reproductive success (thus reflecting the status and health of aquatic vegetation). A CEM (***Figure 4-36***) has been developed which relates the various stressors and drivers in LO to reponses in the fish community.

Fish have been used for many years to indicate whether waters are clean or polluted, doing better or getting worse. LO has supported valuable commercial and recreational fisheries estimated at times in the hundreds of millions of dollars. Among important species taken from LO are white catfish (*Ameiurus catus*), bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), black crappie (*Pomaxis nigromaculatus*), and readear sunfish (*L. microlophus*).

4.8.3 Methods and Analysis

Fish populations in the littoral edge, interior marsh, and open water areas of LO were sampled to assess relative abundance, and acquire statistics for evaluation of length

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frequency and length/weight relationship determination. Fish populations in open water areas were sampled utilizing a trawl methodology at previously established sites and according to procedures from a previous study conducted in LO from 1987 to 1991 (Bull et al. 1995). Fish populations in the littoral edge and interior marsh were sampled utilizing electrofishing techniques at previously established areas and according to procedures developed for an ongoing evaluation of the largemouth bass (*Micropterus salmoides*) population in LO (Havens et al. 2005). Methods used allowed comparison to previous FWC surveys, some of which date back to the early 1990s. Locations of the sampling sites in 2005 are shown in ***Figure 4-37***. Individual fish were identified, weighed and measured for length.

Future analyses to assess fish health in LO may rely more heavily on the 30+ years of existing creel survey data. A full analysis of this data will be pursued to acquire longer-term baseline information on the sport fishery in LO. These analyses may also improve current understanding of fish dynamics over time as a function of stage, SAV and other factors that may be related to fish health and abundance.

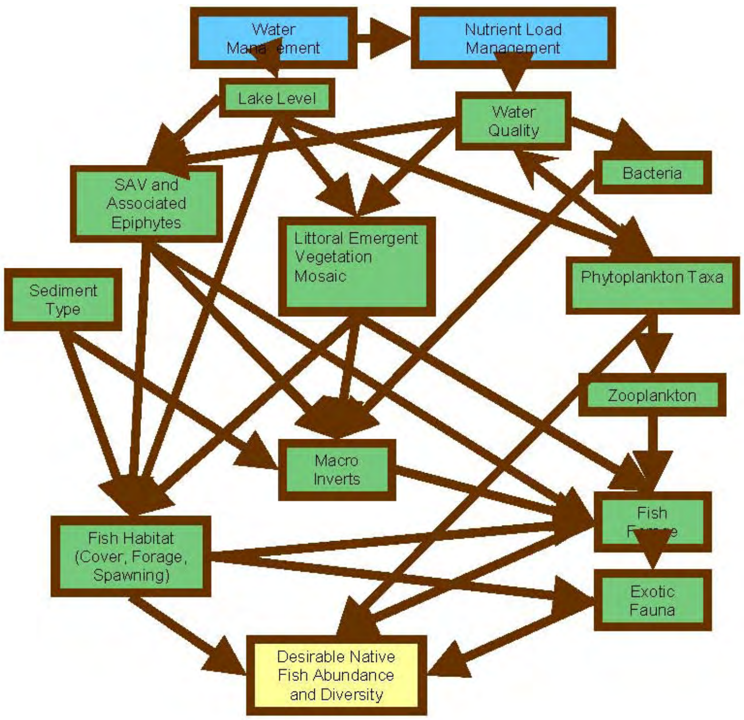

Figure 4-36: Conceptual Ecological Model Diagram Depicting the Factors Affecting Fish Populations

4.8.4 Discussion

A previous multi-year study (Bull et al. 1995) identified threadfin shad as the most abundant species sampled (***Table 4-2***) and black crappie as most abundant in terms of biomass ***Table*** 4-3**.** This relationship reflects the predator-prey relationship between the two species, namely that adult black crappie feed almost exclusively on threadfin shad. Although, threadfin shad remained a significant fraction of relative abundance in 2005 and 2006, overall counts of individual fish had dropped a hundred-fold. Since threadfin shad feed primarily on microscopic plant and animal life, phytoplankton and zooplankton, the reduction in their number can be arguably attributed to some combination of the 2004-2005 hurricanes, increased turbidities and stage which effectively reduced the shad's food source and thus their population. As a consequence, black crappie biomass was reduced to around one percent of the overall fish population assemblage.

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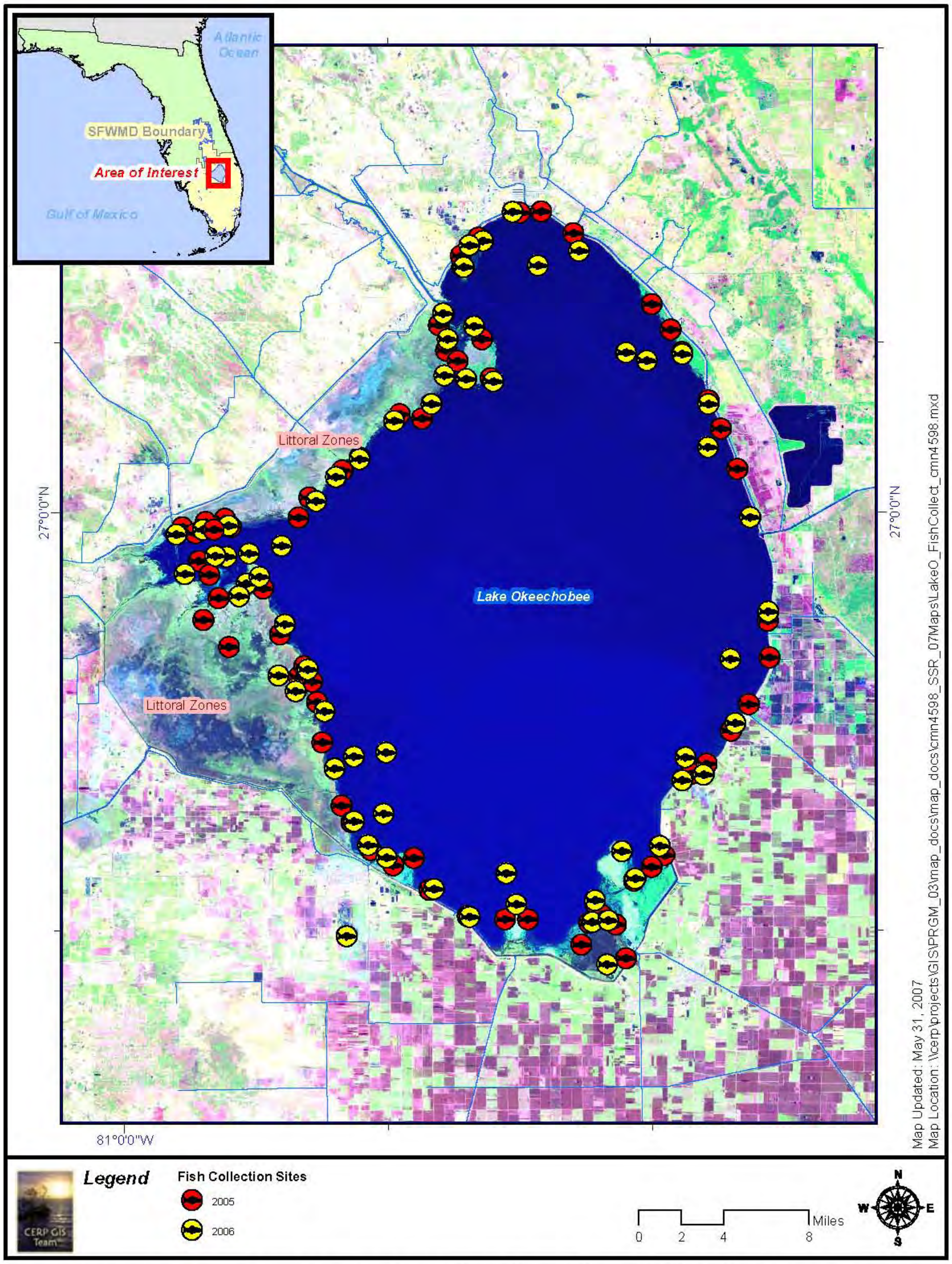
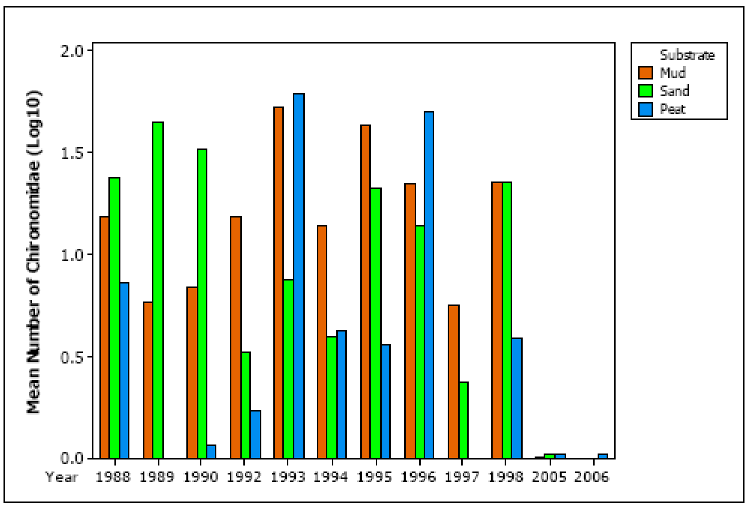


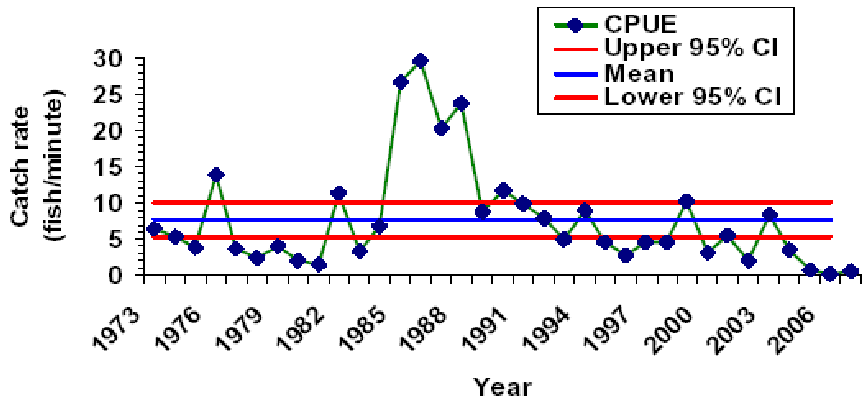
Figure 4-37: Sample Site Locations Used for 2005 Fish Collections

An additional important driving factor affecting LO's fish population density and structure is the reduction in numbers of the chironomidae macroinvertebrates (***Figure 4-38***). Chironomid larvae comprise the primary food source of juvenile black crappie and the decline in the former is another causative factor, along with the decline in threadfin shad, explaining the decline of the latter. Bluegill, also known as bream or brim, feed on very small fish and invertebrates. Bluegill abundance decreased in comparison to the 1987-91 data in 2005 and 2006 by 94 and 92 percent, respectively, which mirrors the decline in invertebrates as their direct prey and that of many of the smaller fish upon which they feed. However, concern regarding these precipitous declines in crappie population must be tempered by observing that 1986-90 was an unusually productive period (***Figure 4-39***), and that accordingly the 1987-91 dataset was biased high. Although the 2005-07 timeframe denotes the lowest catch rate on record, other periods of time have been similarly poor. Nevertheless, the preceding clearly illustrates the intertwined relationships among all the lake health attributes (e.g., SAV, water quality, lake stage, macroinvertebrates, and demonstrates the necessity to assess and manage LO from the widest holistic perspective of balanced ecosystem function).



Average number of individuals (log10+1) of the order chironomidae sampled in each replicate sample by year by substrate type. Chironomidae are the main foodstuff for juvenile black crappie.

Figure 4-38: Average Number of Individuals of the Order Chironomidae Sampled in Each Replicate Sample by Year by Substrate Type



(courtesy FFWCC)

Figure 4-39: Catch Rate of Black Crappie 1973–2007 Collected With Ten Meter Otter Trawl from Lake Okeechobee

A decline in other important species in LO is also apparent (***Figure 4-40*** and ***Figure 4-41***). In comparison to the fish community structure of 1987-91, coarser fish appear to be becoming more dominant at the expense of more desirable species. Relative counts of Florida gar have risen approximately one percent in the older data to 16 and 11 percent of the total in 2005 and 2006, respectively. Florida gar, whose roe is toxic to mammals and birds, is a very tolerant species that can breathe by gills or a lung-like air bladder and is protected by scales that make respectable arrowheads. By weight, Florida gar are currently the most prevalent species, whereas in the previous dataset black crappie, a desirable recreational fish, was dominant.

Black bass fishing is the most popular type of fishing in the United States, with 44 percent of all freshwater anglers considering themselves to be bass anglers, and Florida ranks second behind Texas in number of bass anglers and number of bass fishing trips (USFWS 1996). LO is famed for its year-round bass fishing, and has yielded a large number of trophy bass. Although the data (***Figure 4-40*** and ***Figure 4-41***) indicates that relative counts and weight of largemouth bass have remained stable, there is increasing concern that the last few years' spawning cycles have not been fully realized, although the exact causes remain uncertain. Several possible reasons exist, among them are: 1) the bass successfully spawned but the juveniles did not survive due to a lack of food, 2) juveniles did not survive due to lack of SAV and could not avoid predation, or 3) simply the bass did not spawn. Since largemouth bass are such an important LO species portending ecological as well as financial consequence to recreational fisheries and their professional guides, the success of subsequent spawning cycles remains a concern. Black bass catch rate has shown precipitous declines which appear associated with extreme low lake stages (***Figure 4-42***); however, catch rate is a consequence of a complex set of factors among which are reproductive success and prey availability.

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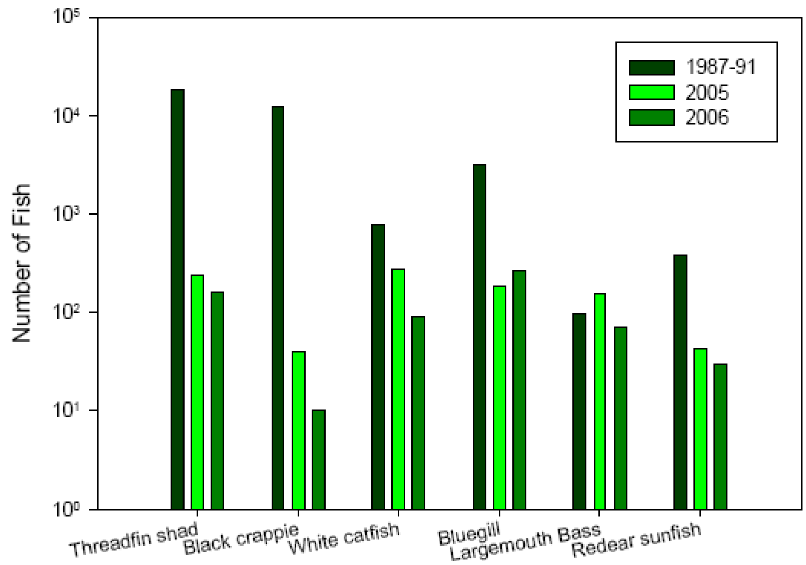


Figure 4-40: Comparative Abundance ($\log_{10}$) of Recent Sampling Efforts (2005 and

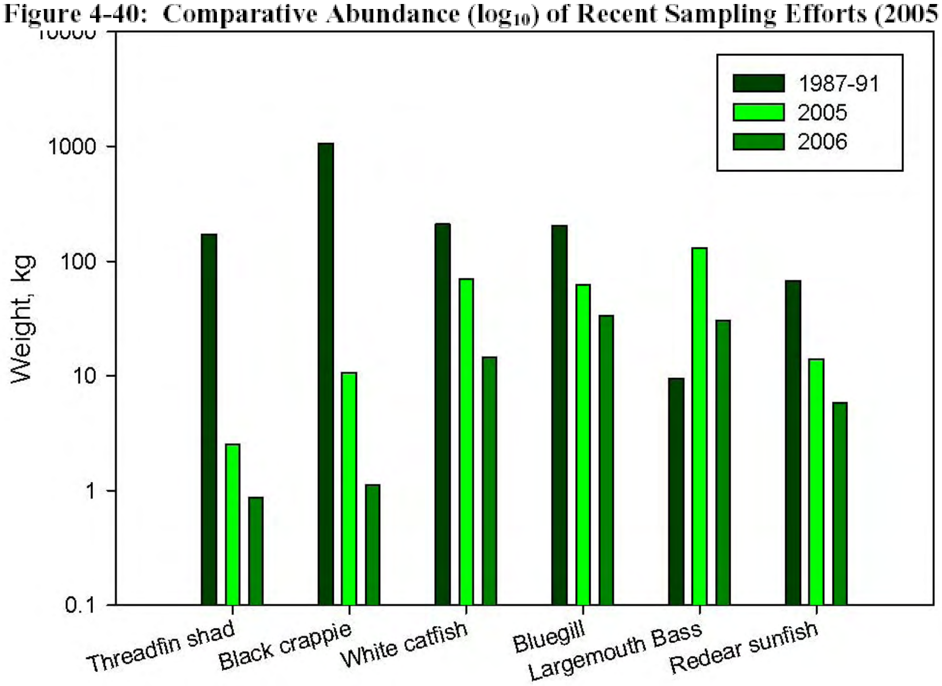
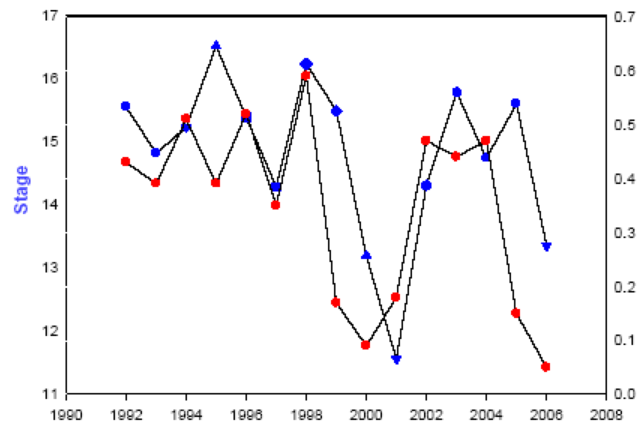


Figure 4-41: Comparative Weights in Kilograms ($\log_{10}$) of Recent Sampling Efforts (2005 and 2006) to Historic Data (1987-91) of Selected Important Fish Species



Variables are significantly correlated (Spearman rho=0.568, p=0.034) when catch rate is lagged one year. (Catch rate data courtesy FFWCC)

Figure 4-42: Largemouth Bass Catch Rate (Electrofishing, Fish/Minute) as Function of Stage (Feet MSL) by Year

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Conclusion The abundance and diversity of the native fish population in LO is dependent on a variety of interwoven factors. Most ostensible among these factors being SAV as habitat and the macroinvertebrate community as a basis of the foodweb. Water management and basin management directly affect lake stage and lake water quality, respectively, with consequence to SAV, algal blooms, sediment integrity, and so forth. The fish population evidences considerable variation year to year, with periodic very good years interspersed with years less so. Clear linkages between changes in the lake and changes in the fish community are not readily deducible from current datasets, presumably because ecological condition in the lake has evidenced diminished quality since fish monitoring started resulting in times when multiple stressors align to adversely affect the population. Recent downturns in fish community health are worrisome, but additional data will be necessary to determine whether these are cyclic events or an actual concern.

4.9 Lake Okeechobee Hypothesis Cluster Macro-invertebrates

4.9.1 Abstract

Macroinvertebrates in the pelagic zone of LO have been intermittently monitored since 1969 (Warren et al. 1995) and are currently being monitored at 18 synoptic sites located in the peat, mud and sand sediments of LO. These are the same sites in which monitoring occurred during 1969-1970, and 1987-1996 and these macroinvertebrate communities will provide pre-CERP implementation baseline data, which can be compared to post-CERP project completion community data. A five-year study assessing macroinvertebrate assemblages in three SAV and two emergent vegetation communities is also currently being conducted.

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After CERP projects are on-line and nutrient and sediment loads to LO are reduced, the dominance by taxa tolerant to organic loading is anticipated to decrease, while numbers of intolerant taxa, species diversity, species richness and evenness of distribution are expected to increase.

4.9.2 Background Description

Macroinvertebrates have been used for the past century as indicators of water and habitat quality in lakes. Freshwater invertebrate communities are extremely sensitive to existing water quality conditions, and reflect a lakes trophic status because they are unable to escape perturbations (Warren et al. 2007) (***Figure 4-43***). Species composition, absolute abundance, relative abundance, diversity, species richness and evenness are metrics commonly used to evaluate the ecological condition in lakes. As the eutrophication process progresses, macroinvertebrate species richness and diversity are reduced, while the community composition shifts to one dominated by pollution-tolerant taxa. As a lake becomes increasingly eutrophic, macroinvertebrates which require higher levels of DO, such as many mussels (Pelecypoda), mayflies (Ephemeroptera), caddisflies (Trichoptera), dragonflies (Anisoptera), and damselflies (Zygoptera), are eliminated. The invertebrate communities then become dominated by groups of species physiologically adapted to withstand high degrees of organic loading and extended periods of low (<4.0 ppm) DO (Brinkhurst 1974, Warren 2007). If LO becomes hypereutrophic, all but the most tolerant segmented worm (Oligochaeta) species may be eliminated (Brinkhurst 1974, Wetzel 1983). Since macroinvertebrates are an important component of freshwater food webs, elimination of most of the macroinvertebrate taxa could have severe negative impacts on fish and other higher-trophic level organisms which utilize macroinvertebrates as a food source.

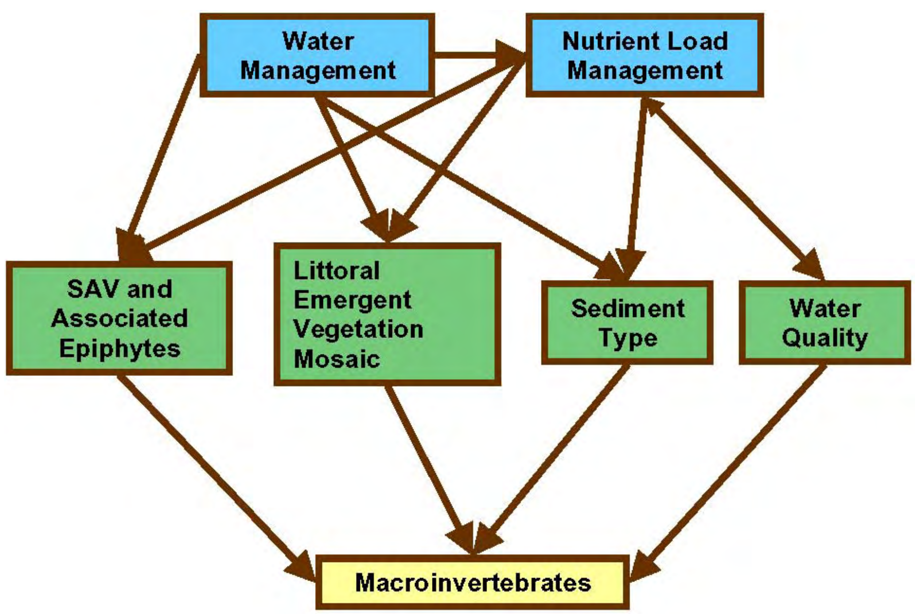

Figure 4-43: Macroinvertebrate Conceptual Ecological Model

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4.9.3 Methods and Analysis

As a component of the MAP, pelagic zone and SAV/emergent vegetation macroinvertebrate communities are currently being monitored to establish pre-CERP implementation baseline conditions. The pelagic zone macroinvertebrate monitoring results also will be compared to those collected during a 1987-1996 ecosystem study (Warren et al. 1995), while the SAV/emergent vegetation monitoring results will be compared to those collected during 1986-87 (Rudolph and Strom 1990), thereby enhancing the pre-CERP implementation baseline data.

The LO synoptic pelagic zone monitoring is currently being conducted by the FWC, and will be compared to existing macroinvertebrate data from 1987 thru 1996. These benthic invertebrate community samples were collected from six sites within each of three aerially dominant habitat zones (mud, sand, peat) twice annually (***Figure 4-44***) using a petite ponar dredge, yielding a total of 54 samples per collection (see Warren et al. 2007 for complete details). Community structure metrics include taxonomic composition, taxa richness, absolute abundance, relative abundance, diversity (Shannon's equation, as per Krebs 1999), and evenness (as per Pielou 1977).

The SAV/emergent vegetation monitoring is being conducted bi-annually, in triplicate SAV (*Hydrilla*, *Potamogeton*, *Vallisneria*) and emergent (*Scirpus*, *Typha*) sites located along the north, west and southern nearshore region of LO (***Figure 4-44***). This monitoring, as part of a larger trophic study involving fish, is being conducted by Malcolm-Pirnie, Inc., and is scheduled to continue through 2010. Results of the first SAV/emergent vegetation macroinvertebrate sampling event, which was conducted in October 2006, are still being analyzed.

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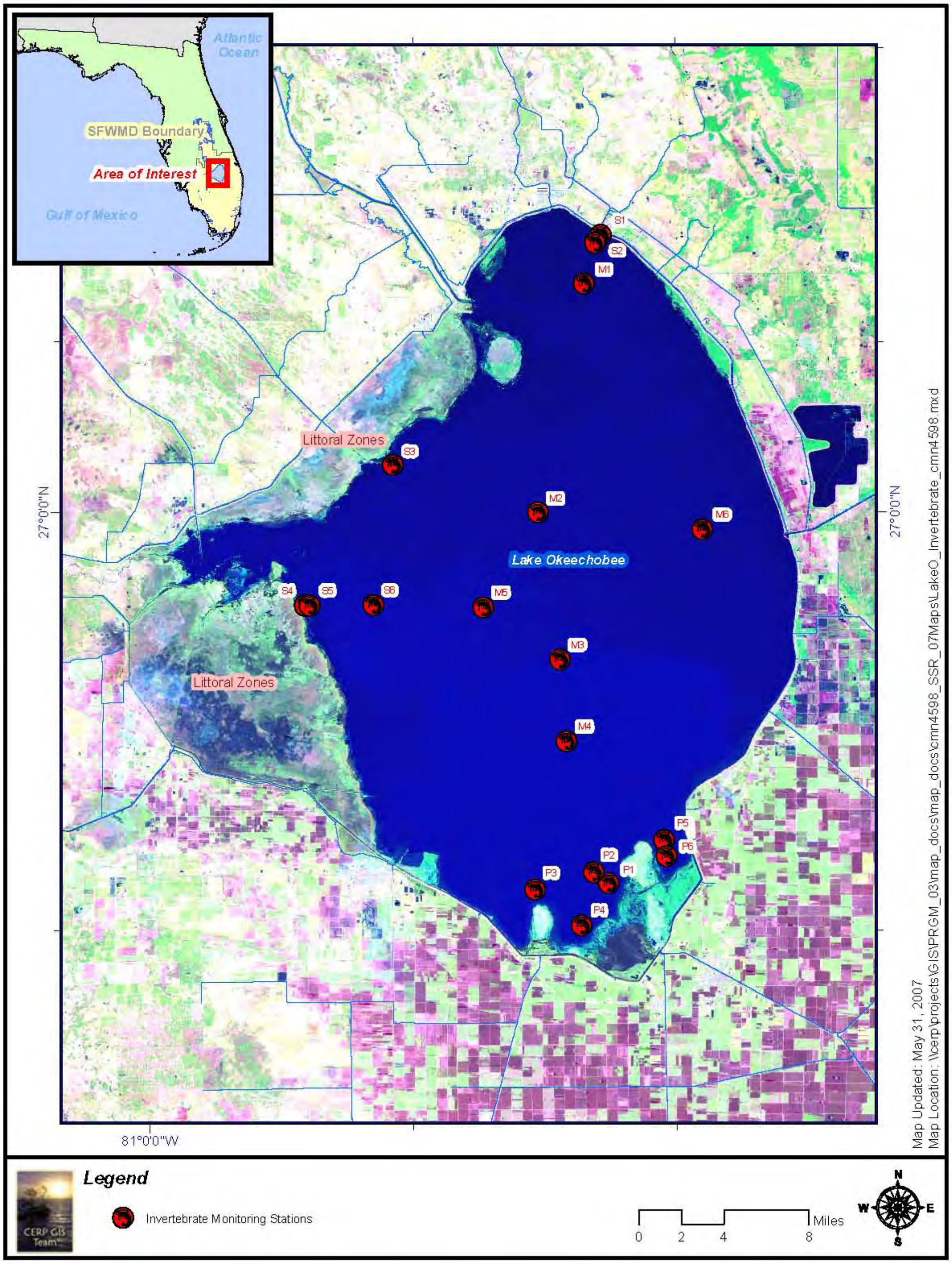

Figure 4-44: Florida Fish and Wildlife Conservation Commission Sublittoral Zone Benthic Invertebrate Sampling Sites in Lake Okeechobee

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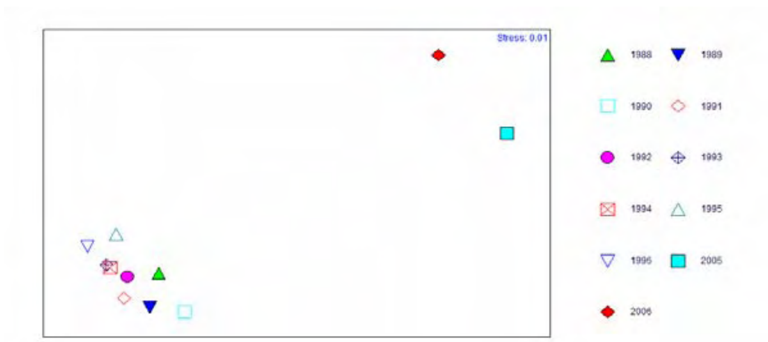
4.9.4 Discussion

Overall Lake Community Results from macroinvertebrate investigations conducted in the 1980s suggested that the communities reflected eutrophic conditions in LO's nearshore SAV and emergent vegetation beds (Rudolph and Strom 1990) and the pelagic zone (Warren et al. 1995).

Results from the recent pelagic zone data indicated that a total of 48 individual aquatic invertebrate taxa representing 20 major taxonomic groups were collected from LO during the two sampling events (August 2005 and February 2006) conducted in study year one. Oligochaeta (segmented worms) numerically predominated the mud and sand habitat zones during both sampling events, and accounted for 64.3 percent (lakewide mean = 2,147 individuals m^{-2}) of the total number of organisms collected during the study year. Most abundant among the Oligochaeta were the tubificids *Limnodrilus hoffmeisteri* (974 m^{-2}, 29.2%) and *Haber speciosus* (580 m^{-2}, 17.3%).

Aquatic Acari (water mites) numerically predominated the peat zone of the southern lake region and accounted for 11.9 percent of all organisms collected (lakewide mean = 397 m^{-2}). The introduced Asian clam *Corbicula fluminea* was the fourth-most abundant taxon and accounted for 9.4 percent of all organisms collected (314 m^{-2}). No other individual taxon accounted for more than five percent of the total organisms. Chironomidae (non-biting midges), which usually account for a large percentage of the benthic fauna in the open water zones of lakes, accounted for only 2.8 percent of the total organisms collected from the sublittoral zone. The tubicolous detritivore *Chironomus crassicaudatus*, which has accounted for a substantial percentage of the sublittoral zone benthos in past collections (Warren et al. 1995), was present with a mean density of only one m^{-2} (<0.1% of total organisms). Other taxa notably important in past collections, but absent or present in low numbers in the 2005-06 collections, included the gastropods (snails) *Viviparus georgianus* and *Melanoides* sp., the amphipod crustaceans *Gammarus tigrinus* and *Hyalella azteca*, the isopod crustaceans *Cyathura polita* and *Cassidinidea ovalis*, and the chironomids *Cladotanytarsus* sp. and *Polypedilum halterale* (Warren 1995).



The closeness of points to one another reflects the similarity of the macroinvertebrate assemblage, in this case from year to year.

Figure 4-45: Multidimensional Scaling Ordination of Combined Annual

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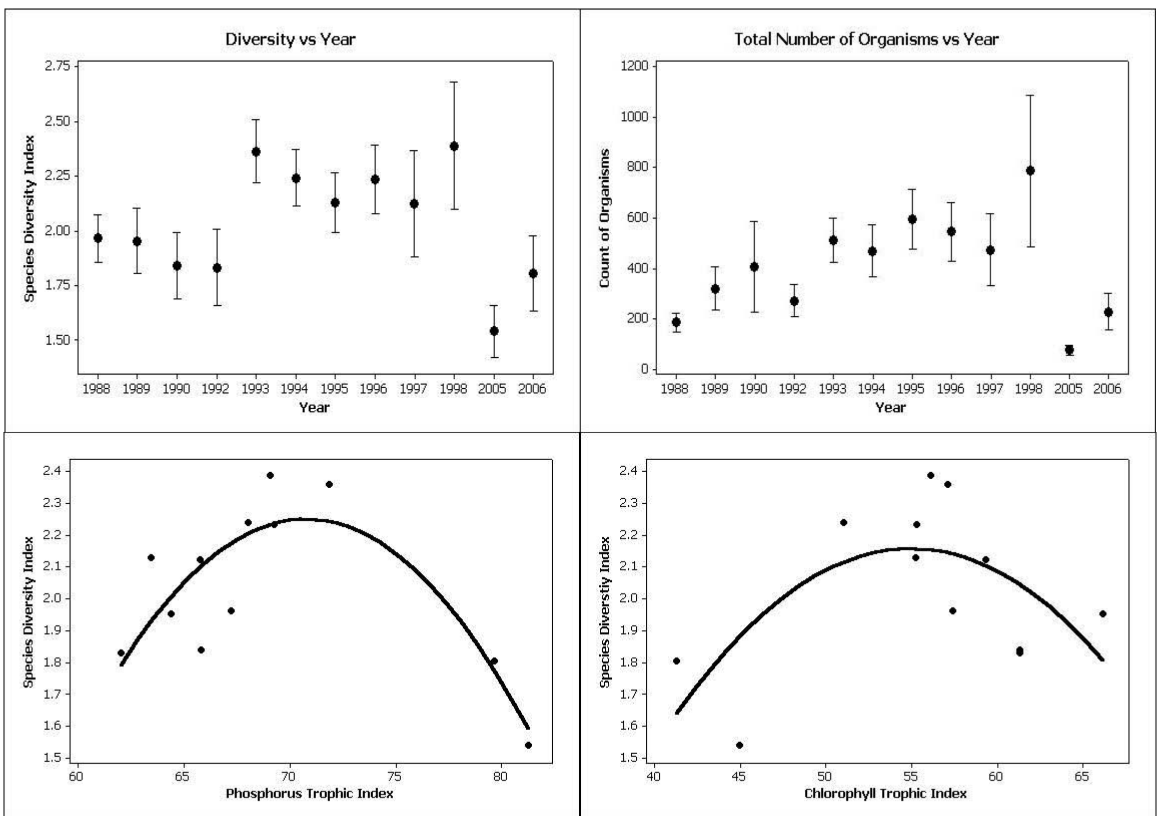
Macroinvertebrate Community Structure for Each Year Samples Were Collected

It is clear that a distinct change in benthos community structure has occurred between 1996 and 2005 (***Figure 4-45***). Lower species diversity was evidenced in LO from 1988-1992, with a period of higher diversity observed during 1993-1996, followed by a large reduction in diversity occurring sometime between 1996 and 2005. Some factor likely associated with the hurricanes of 2004 and 2005 (e.g., perhaps excessive and prolonged high turbidity) may have resulted in decreased diversity and total number of organisms. This pattern is also apparent in the total number of organisms (***Figure 4-46***). As an alternative explanation, the unusually extreme low lake stage experienced during the 2001 dry season may have led to conditions which had negative impacts on the macroinvertebrate community. However, the DO regime did not reflect that lowered stages corresponded with low DO concentrations. Further analysis was unable to attribute reductions in diversity related to the 2001 dry season event to any stage or water level effects.

A small recovery in diversity and total number of organisms appears to have occurred in 2006 in comparison to 2005, which seems to substantiate a recent causative event and lends credence to the 2004-2005 hurricane hypothesis (***Figure 4-46***). However, these relationships also extend to the nutrient regime (***Figure 4-47***), so no absolute causative agent can be assumed.



Significant relationships exist between mean annual diversity and trophic state indices for P (R^2 = 72%, P = 0.003), and to a lesser extent chlorophyll *a* (R^2 = 43%, P = 0.079).

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Figure 4-47: Classic Species Diversity Index Response to Increasing Eutrophication

Habitat and Seasonal Influences Bottom substrate type is the primary determinant of invertebrate community structure in the LO sublittoral zone. Four primary benthic habitat regions characterize the sublittoral: mud, sand, peat, and limestone bedrock (Reddy 1993) (***Figure 4-48***). The mud region is distinguished by deep, fine-particle sized organic sediments that occupy the central and north-central areas of LO. The mud region accounts for more than 50 percent of the total bottom surface area of the sublittoral zone. The sand region zone is located at the periphery of the sublittoral zone in the northeastern, northern, and northwestern lake areas and, in the western lake area, extends lakeward for several miles along the entire length of Observation Shoal. The peat habitat region is located in the southern quarter of LO and is characterized by areas of both fine and coarse peat. The fourth habitat type is a limestone bedrock reef that separates the peat habitat region from the mud habitat region. For the purposes of this study, only the mud, sand, and peat habitat regions were sampled. The limestone reef was not sampled because of difficulties obtaining legitimate quantitative samples with the petite ponar dredge from the hard limestone substrate. A two-way ANOSIM (Clarke et al. 1993) suggests that separation among the macroinvertebrate communities was more significantly associated with the sediment type (Global R=0.64, $p<0.01$) than with variability among the summer sampling seasons. Mud and peat-associated macroinvertebrate communities had the clearest separation (R=0.83), while separation among the sand and mud communities was the less defined (R=0.48). The distribution of community types as a function of substrate type is depicted in ***Figure 4-49***.

The global year-to-year differences, and in particular the differences in community structure between the 2005 and 2006 sample collection years is borne out by a similar pattern when each of the substrate types are examined independently (***Figure 4-50***). This indicates that the change that occurred between 1996 and 2005 affected all substrate types in a similar fashion. Examination of the non-2005/2006 sampling data (***Figure 4-51***) indicates a somewhat orderly progression of sample sets across the period of record in mud and peat; both of these communities are susceptible to oxygen stress since both substrate types are typically reducing environments. This may indicate that benthic oxygen stress may be increasing.

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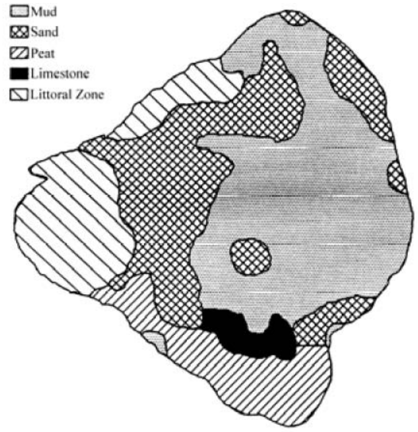
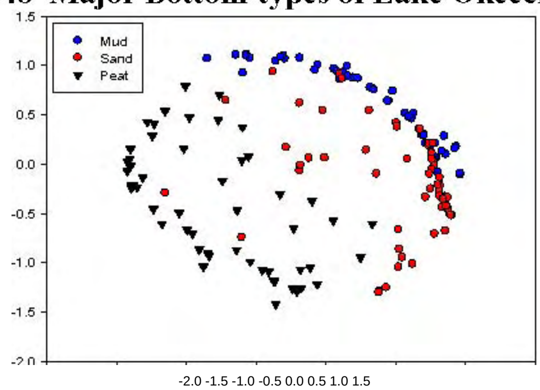


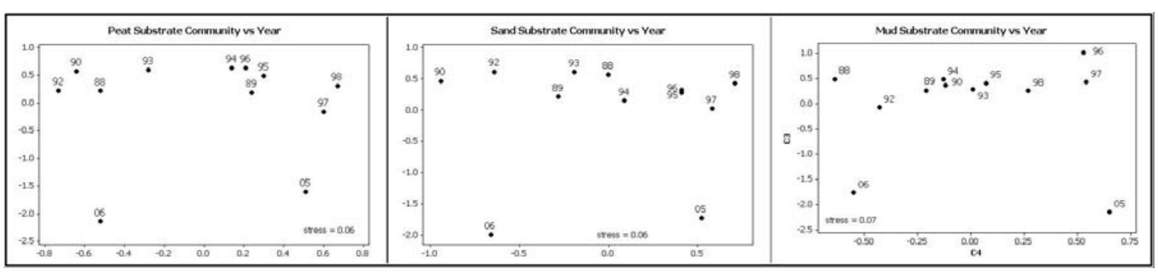
(Modified from Reddy 1993)

Figure 4-48 Major Bottom-types of Lake Okeechobee, FL



Multidimensional scaling ordination of species community structure as a function of substrate type, based on 162 replicate samples (6 sites of 3 replicates in each of 3 substrate types) collected in August 2005, February 2006 and August 2006 combined. Mud and peat benthic communities are the most dissimilar with sand communities intermediate.

Figure 4-49: Multidimensional Scaling Ordination of Species Community Structure as a Function of Substrate Type



Community structure in 2005 and 2006 differs substantially from all other years in all three major substrate types.

Figure 4-50: Multidimensional Scaling Ordination of Species Community Structure as A Function of Ranked Mean Species by Each Substrate Type by Each Year

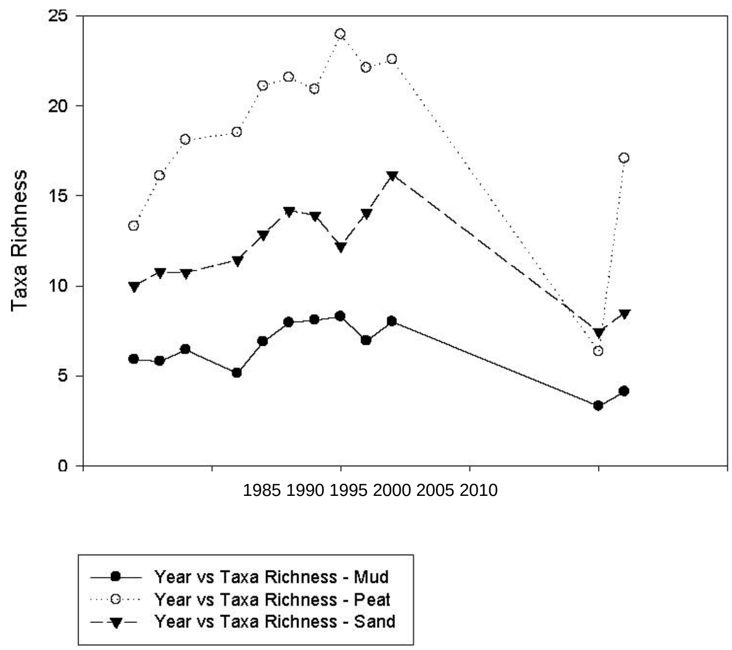


Figure 4-51: Species Richness Mean as a Function of Sample Year and Substrate Type

Results from previous studies have shown that benthic invertebrate communities of the LO mud zone have displayed the lowest species richness and diversity of all sublittoral habitat type communities sampled (Warren et al.1995). The current study reflects these results. Mud sediment-associated community species richness were typically about half the species richness means from the other substrate types sampled except for the peat region in February 2006 which evidenced a sharp rebound from the 2005 low (***Figure 4-51***). Mud region species diversity was also lowest among habitat types in both sampling seasons, while mean values of evenness were nearly equivalent across all habitats and sample dates. Three individual segmented worm taxa (*Limnodrilus hoffmeisteri*, *Ilyodrilus templetoni*, and *Stephensoniana trivandrana*) numerically dominated the mud region and accounted for 77 percent of the total abundance. No other individual taxon accounted for more than four percent of the total abundance in the mud region. Mud region communities exhibited little seasonal variation in taxonomic composition.

Segmented worms also dominated the benthos of the sand habitat region. *Limnodrilus hoffmeisteri*, *Ilyodrilus templetoni*, and *Stephensoniana trivandrana* together accounted for 67 percent of all sand habitat organisms collected. The Asian clam *Corbicula fluminea* was the third most abundant invertebrate collected from the sand habitat, occurring with a mean density of 578 m^{-2} and accounting for ten percent of all sand organisms. No other individual taxon accounted for more than five percent of sand total speciation. There were no substantial differences between seasonal means of species richness, evenness, or diversity within the sand habitat region.

The taxonomic composition of the peat habitat region invertebrate community differed substantially from the taxonomic compositions of mud and sand region communities, however the pattern of extreme dominance by a few number of taxa exhibited in mud and sand-inhabiting communities was also reflected in the peat region. Aquatic Acari were, by far, the most abundant benthic invertebrates collected from peat, occurring with a

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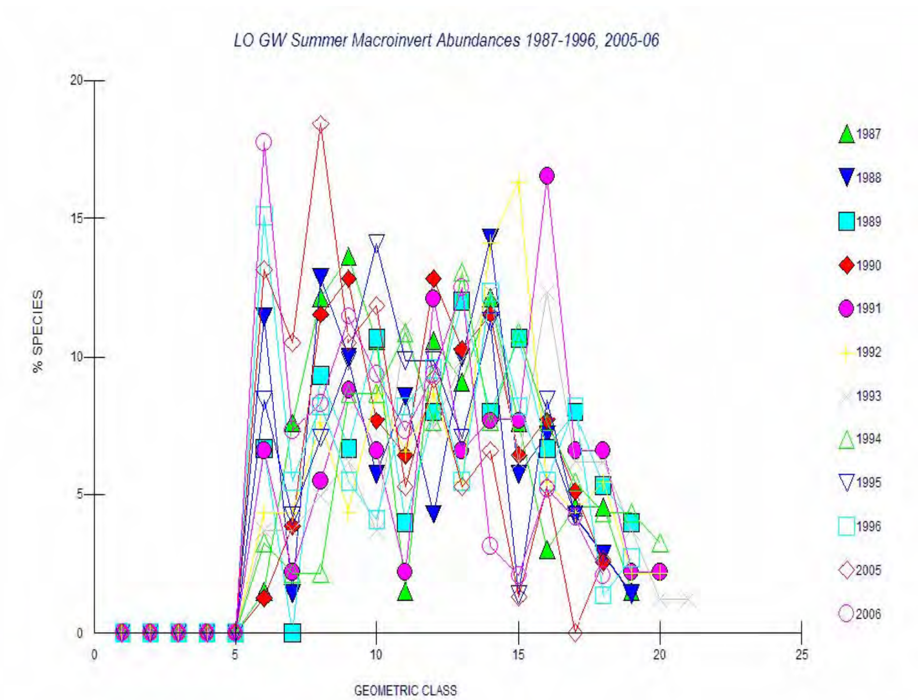
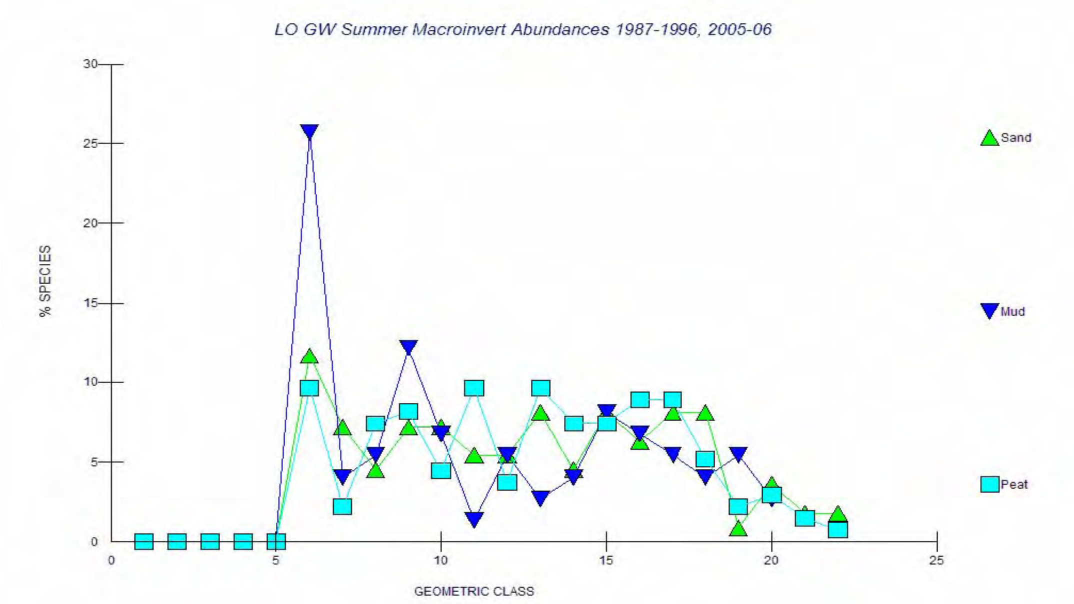
mean density of 1,104 m^{-2} and accounting for 42 percent of all peat organisms. Other important dominants included *Corbicula fluminea* (364 m^{-2}, 14 %), Nematoda (359 m^{-2}, 14 %), the amphipod *Gammarus* nr. *tigrinus* (204 m^{-2}, 8 %), and the segmented worm *Stephensoniana trivandrana* (165 m^{-2}, 6 %).

Mean taxa richness in the peat region winter community (Feb. 2006) was less than half of the corresponding summer value. A remarkable result from the 2006 winter sampling event was that no Chironomidae were present in any peat region samples (see LO Fish chapter for further discussion). Chironomidae accounted for 26 percent of all organisms collected from the peat region during the 1987–1991 sampling period (Warren et al. 1995).



The plots are of the number of taxa represented by only one individual in the sample (class 1 on the x-axis), two to three individuals (class 2), four to seven (class 3), eight to15 (class 4), and so forth.



In systems where a measure of balance exists between numbers of rare and common taxa, a geometric abundance class plot portrays a smooth curve. Where few rare taxa are represented, the higher geometric abundance classes are more strongly represented. Plots of the lake data (***Figure 4-52*** and ***Figure 4-53***) indicate that rare taxa did not occur in LO during the 2005-2006 sampling events. Gray and Pearson (1982) have suggested that taxa in the three to five abundance classes are most sensitive to pollution-induced changes (a way to select indicator taxa). Class 3 through 5 were also absent from LO.

The taxonomic composition, species richness, evenness of distribution, and diversity of LO sublittoral zone benthic invertebrate communities during the two sample periods of the 200506 study year were overall indicative of very poor water quality. Extreme dominance of the mud and sand habitat regions by three species of pollution tolerant Oligochaeta represents a pattern echoing the findings of Warren et al. (1995). The 2005-06 absence of many taxa that were present during the 1987-91 study period may signal poorer habitat conditions in the sublittoral zone. However, complete analyses of the additional data acquired during the remaining two years of the present study are required for a comprehensive evaluation of lake status.

Expectations are that as LO nutrient levels decline, in part due to CERP implementation and in part due to the various complimentary efforts to control P runoff, the extreme numerical dominance of pelagic zone invertebrate communities by segmented worms (Tubificidae) as previously documented (Warren et al. 1995) should be supplanted by a more diverse, balanced and sundry community. Increases in the relative abundance of less pollution-tolerant taxa (e.g., snails, crustaceans, mayflies, caddisflies) will signal a return to a less eutrophic, pollution-tolerant and more natural condition.
Conclusion The macroinvertebrate community in LO has continued to reflect the eutrophic conditions that preceded initiation of benthic sampling. The range of variation in the lake's macroinvertebrate community has continued to swing somewhere between moderate and poor. The macroinvertebrate community monitoring reflects that variability, but also continues to be dominated by pollution-tolerant taxa, reflecting the

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poor water quality in LO. Against this backdrop, there exists a high probability that improving water quality conditions within LO will result in a demonstrably improving benthic community. Such improvements in the macroinvertebrate community will foment positive effects on fisheries and the ecological health of LO as a whole.

4.10 Lake Okeechobee Module Conclusion

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

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

A number of CERP and related non-CERP projects are currently underway in the watershed, to reduce nutrient influxes to LO and to improve lake hydrology. These projects consist of aquifer storage recovery wells, stormwater treatment areas (STAs) and water storage reservoirs. Additional projects such as dredging and chemical inactivation of the sediments are being contemplated to reduce LO's internal nutrient load as a means of accelerating ecological improvements to the system, since internal loading may delay the ecological restoration of LO on the order of many decades. In any discussion of LO restoration, it must be kept clearly in mind that due to LO's central location in the south Florida aquatic ecosystem, failure to resolve its nutrient and hydrologic problems jeopardizes the restoration of the NE and the entire southern half of the Everglades ecosystem.

Currently, routine monitoring is in place for lake water quality and hydrology, submerged

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and emergent aquatic vegetation, fish, macroinvertebrates and phytoplankton. In addition, the littoral portion of LO is included in system level monitoring for wading birds and their prey species, and for the Everglades snail kite. A number of other research studies are ongoing which aim to elucidate key ecological relationships between various ecosystem components. At present, research and monitoring efforts on LO are probably sufficient to detect significant changes expected to be brought about by restoration activities. However, only a small proportion of these monitoring and research activities are funded by CERP, while the balance are either funded through other mandated or non-mandated SFWMD programs or are done for permit compliance. In either case, the availability of this necessary data is outside the direct control of RECOVER.

Lessons Learned The single most important lesson learned in assembling the LO SSR is the wide range of variability encountered in each parameter monitored. While some of this variability can be clearly associated with known natural and man made major physical perturbations, much of it cannot. As such, it is becoming increasingly clear that extensive, long duration monitoring will be required to clearly identify the impacts of restoration activities within the noise of normal environmental variability. Even with CERP watershed projects in place, there will continue to be good years and bad years and the ability to detect system wide improvement may depend on the ability to identify changes in the relative frequency or magnitude of these good and bad years.

A corollary to this lesson may be that certain monitoring parameters, such as SAV and macroinvertebrates, may prove more responsive to environmental restoration than others, and it will be these parameters in which long-term efforts need to be concentrated. Similarly, as ongoing research continues to elucidate relationships between key environmental components, it may become possible to monitor fewer parameters without sacrificing assurances that the entire ecosystem is benefiting from restoration activities.

4.11 Status of Monitoring in the Lake Okeechobee Module

The following table provides an abbreviated status of monitoring in the LO Module. The table includes a list of monitoring components, links them to the associated hypothesis cluster(s) and PMs, and provides a brief description of the monitoring itself as well as its status. The table is not meant to be exhaustively comprehensive and represents the most current information to date when the table was developed.

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Table 4-4: Status of monitoring in the LO Module

Lake Okeechobee Hypothesis Cluster	Related Monitoring Components	Performance Measure	Description of Monitoring/Research	Status of Monitoring and Research
SAV	stage, WQ	Lake Okeechobee Vegetative Matrix	Monthly/quarterly biomass transect monitoring Annual 1km2 grid cell presence/absence abundance estimation	Ongoing since 1999 Ongoing since summer 2000
Littoral Zone Emergent Vegetation	stage, WQ	Lake Okeechobee Vegetative Matrix	Scirpus (bulrush) stem count density monitoring at nearshore sites on a quarterly basis.	Ongoing since June 2005
Phytoplankton	WQ	Lake Okeechobee Vegetative Matrix	Samples collected from 4 long-term (>12 years) monitoring sites on a quarterly basis, taxonomic identification in progress. Also collect samples for taxonomic identification at 7 nearshore sites and 2 near municipal water intakes	Ongoing since 1994
Exotic Vegetation	stage, WQ	Lake Okeechobee Stage	various treatment methods are currently being evaluated for Panicum repens (torpedo grass) and Typha (cattail)	Ongoing since 1994
Macroinver-tebrates	WQ, sediment quality, SAV	Lake Okeechobee Macroinvertebrates	Bi-annual benthic monitoring is in progress at 18 pelagic and nearshore sites (6 in mud, 6 in peat and 6 in sand sediments	Three year monitoring project commenced in 2005
Fish	Pelagic Zone Fish SAV, Littoral Zone Emergent Vegetation, Stage, WQ	Lake Okeechobee Fish	Synoptic pelagic and nearshore fish communities are being monitored at long-term FFWCC sites. Sampling occurs annually (summer) Abundance and distribution of pelagic and littoral fish communities are being evaluated with regard to vegetation type, density and water depth. Sampling occurs annually.	Three year monitoring for annual sampling commenced in 2005. Five year monitoring for SAV and emergent – associated fish communities commenced in 2006
N/A	Stage, Littoral and Emergent Vegetation	Lake Okeechobee Amphibians and Reptiles	Abundance and distribution is being evaluated with regard to vegetation type, density, seasonality and water depth. Sampling occurs 2 X per year.	Five year monitoring for emergent –associated communities commenced in 2006

4.12 Communicating the Lake Okeechobee Indicators

Indicator Metrics
The restoration goal for littoral zone emergent vegetation and near-shore region SAV is primarily focused on spatial extent; though for SAV, the ratio of vascular to non-vascular plants is also an important metric (RECOVER 2007a). *S*patial coverage targets are evaluated by comparison to anecdotal or empirically-measured best conditions from the recent past (e.g., Havens et al., 2002). That is, restoration targets are pragmatically based on best observed ecological conditions which have been documented in the littoral zone/near-shore region of a highly managed and physically altered lake ecosystem, not on pre-drainage or pre- dike conditions.

Bulrush – In general, targets for emergent vegetation in the littoral zone of Lake Okeechobee are non-numeric and challenging to enumerate given the limited information available on the emergent vegetation community as a whole (Doren, 2006). The restoration goal for spatial extent of bulrush identifies the desire for a more continuous and thicker band of bulrush located along the western edge of the lake (length of approximately 50 kilometers). Although the current RECOVER PM for bulrush (RECOVER, 2007a) does not define an explicit target, it is probable that the maximum areal coverage of bulrush, as reported by Pesnel and Brown (1977), could be re-established given successful restoration of the lake's quality of water and sustained ability to appropriately manage lake stage. At present, a hydrologic surrogate for bulrush suitability is used as the indicator metric for assessing Lake Okeechobee-wide health of bulrush (sensu Doren, 2006).

SAV – When conditions are favorable, SAV can occupy more than 40,000 acres in Lake Okeechobee, but coverage can be reduced to near zero when conditions are poor (e.g., Havens et al., 2004). Ideally, the target for SAV is to have an average annual coverage at the end of each growing season of 40,000 acres or more, where at least half this acreage is comprised of desirable vascular species. While this metric presently focuses on areal coverage, the addition of a temporal component also would be beneficial (see Discussion below).

The Stoplight Restoration Report Card System Applied to Lake Okeechobee

Bulrush – The influence of water depth on the persistence of giant bulrush was studied to examine how to minimize impacts of stage level manipulation on long-term bulrush survival. Currently experiments are being conducted to identify the specific effects on growth, vegetation propagation and seed bank germination of various hydroperiod regimes and water transparencies (James and Zhang, 2008). These data will help refine our understanding of bulrush growth dynamics as they relate to lake stage and water quality, the two parameters most likely to be affected by Lake Okeechobee restoration efforts. Recent evidence also suggests that the physical effects of tussocks of free floating aquatic vegetation (e.g., Eichornia (water hyacinth) and Pistia (water lettuce)) exerting wind and wave-driven pressure against bulrush stands, as well as non-targeted spray damage from treating such vegetation in bulrush stands may have substantial and long-lasting effects on the bulrush in Lake Okeechobee (James and Zhang, 2008). Nevertheless, our current understanding is that undisturbed bulrush persists when water depths are below of 0.9 m (lake stage of 3.9 – 4.1 m NGVD), but prolonged periods of high-water inundation (e.g. water depths above 4.3 to 4.6 m depending on duration), or extended periods of dry conditions (lake stage less than 3.0 m NGVD and duration greater than four months) may cause bulrush stands to decrease in areal coverage, especially since bulrush is more susceptible to disturbances such as herbivory or strong winds (Zhang et al., 2007). A suitability index was developed to relate hydrological condition to bulrush health.

SAV –The ability to satisfy a spatial coverage target for SAV is determined by inter-dependent environmental stressors (e.g., lake stage and water transparency which combine to determine light availability in the water column). Thus an assessment of the health of SAV in Lake Okeechobee needs to be interpreted in the context of how much SAV exists in the lake relative to model projections of suitable SAV habitat.

A model has been developed that predicts potential SAV habitat availability for a given year, based on multiple years of monitoring data. SAV habitat availability is evaluated as a function of water transparency, which is indirectly measured by total suspended solids, and lake water levels (Zhang et al., 2007). Using bathymetry information, this model is applied to the SAV spatial sampling grid with GIS, and predicts areas within the near-shore region of Lake Okeechobee that are suitable SAV colonization habitats when favorable water depth, light penetration, and turbidity conditions occur.

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Combining metrics – Further refinement of the SAV habitat suitability model (see Discussion below) and results from the ongoing bulrush research described above will be valuable for further refinement of the individual SAV and bulrush indicators. Future efforts will focus on development of a combined index as many factors which affect health of bulrush and SAV are the same. Environmental conditions such as light availability may have dominant effects on both bulrush and SAV. For example, lower stages result in higher light availability which may be favorable for both plant indicators (albeit relationships of underwater light to growth may be more complex for bulrush because of its emergent growth habit). Additionally, water depth is a common environmental factor for the two indicators, although ideal water depths for these two metrics may not be the same. Preliminary results from recent bulrush studies suggest that bulrush expansion are enhanced by a range of lake stages that are low enough that inshore water levels in the nearshore region become too shallow to support vascular SAV habitat (i.e., the extent of available habitat for SAV colonization in the inshore portion of the nearshore region is reduced; e.g., Havens et al., 2004). The relationship between Lake Okeechobee stage and exposure of the littoral and nearshore zone can be viewed at: http://my.sfwmd.gov/gisapps/losac/sfwmd.asp.

Components of the Lake Okeechobee Stoplight Restoration Report Card

Bulrush – Lake-stage conditions in the Lake Okeechobee nearshore zone are applied for the bulrush indicator.

SAV – Three components are involved in the SAV indicator: (1) the annual areal distribution of SAV in the nearshore region of Lake Okeechobee; (2) the environmental conditions recorded during the annual SAV mapping effort; and (3) the SAV suitability model.

Scoring and Thresholds for the Lake Okeechobee Stoplight Restoration Report Card

Bulrush – The first step for scoring the bulrush indicator involves application of a suitability index for bulrush in the littoral zone of Lake Okeechobee, based on monitoring and research information (Doren, 2006) that helps to delineate between poor, acceptable, and optimal conditions for suitable bulrush establishment and survival as a function of lake stage for the present year (Table 2). The second step involves examining the suitability score for the present year along with suitability scored from the two prior years – a single-year snapshot of the status of bulrush is insufficient to characterize the spatio-temporal health of bulrush in Lake Okeechobee. Three consecutive years of performance scores (a time period identified based on our best professional judgment) are then assembled in sequence which is then compared to an interpretative matrix to derive an overall prediction of bulrush suitability. While the three years combine to influence the overall conclusion, the current year’s status is afforded a slightly larger influence over that of other years as the following year is more strongly influenced by the current year than prior years.

SAV – The first step for scoring the status of SAV in Lake Okeechobee involves calculating the acreage of total SAV, and the percentage of total SAV acreage comprised

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of Chara spp. The second step involves examining the Lake Okeechobee SAV suitability model performance for a given year determined by comparison of actual total SAV acres with modeled total SAV suitable habitat acres, expressed as a percentage of modeled SAV suitable habitat acres.

The final assessment step combines the scoring results from the SAV acreage and SAV modeling performance components. When actual SAV acreage has attained the coverage goal for a given year, the SAV acreage component drives the overall score of the final conclusion for that year. In essence, when SAV conditions are good and consistently attain the ultimate restoration coverage targets, results from the SAV habitat model results are not incorporated into the annual overall score (i.e., they were discounted). Likewise, when the actual SAV acreage is poor, the SAV acreage component drives the overall score of the final conclusion for that year. Only, when actual SAV acreage is moderate, does the SAV modeling performance influences the overall score. When a moderate actual SAV acreage is below that predicted by the SAV suitability model, the overall score is poor (red), suggesting that the SAV community is in worse than expected shape. When a moderate actual SAV acreage is better than predicted by the SAV suitability model, the overall score is good (green), suggesting that SAV community is in better shape than expected.

This metric has been applied to several years of SAV data. Overall SAV status in 2002 would be considered to be green (green in acreage and green in model performance); SAV in 2003 yellow (yellow and yellow, respectively); SAV in 2004 and 2005 green (yellow and green, respectively).

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4.13 Acknowledgements

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4.13.1 Appendix 4A–MAP Metadata

All maps appearing in this document meet the standards and guidelines as defined in the CERP GIS SOP Manual. These maps are NOT to be used as Stand Alone Documents. To utilize a map as a stand alone hand out, please contact the map creator for additional map elements.

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Lake Okeechobee Landscape Units Boundaries Map Author: Laura Biddison, CERP GIS Map Technician Map Created: March 26, 2007 Map Location: \\cerp\projects\GIS\PRGM_03\map_docs\cmn4598_SSR_07Maps\LakeO_Site_cmn4598. mxd Base Imagery: Land Sat Imagery 2004 Datasets used: Landscape Unit Extents-\\cerp\projects\GIS\PRGM_03\spatial\shp\cmn4598\LSU_Outline.shp SFWMD Canals–CERP SDE; HYSUR_CANAL_CRL_SFWMD Urban Areas–CERP SDE; GISLIB.BDDEP_MUNICIPAL_BOUNDARY CERP SDE; GISLIB.MIAMIDADE_BDJUR_MUNICIPAL

Lake Okeechobee Fish Collection Stations Map Author: Laura Biddison, CERP GIS Map Technician Map Updated: May 31, 2007 Map Location: \\cerp\projects\GIS\PRGM_03\map_docs\map_docs\cmn4598_SSR_07Maps\LakeO_Fis hCollect_cmn4598.mxd Base Imagery: Land Sat Imagery 2004 Datasets used: Fish Collection Sites-CERP SDE; GISLIB.RECOVER_MONITORING; LO_FISH_COLLECT_05_06 SFWMD Canals–CERP SDE; HYSUR_CANAL_CRL_SFWMD

Lake Okeechobee Invertebrate Monitoring Stations Map Author: Laura Biddison, CERP GIS Map Technician Map Updated: May 31, 2007 Map Location: \\cerp\projects\GIS\PRGM_03\map_docs\cmn4598_SSR_07Maps\LakeO_Invertebrate_c mn4598.mxd Base Imagery: Land Sat Imagery 2004 Datasets used: Invertebrate Monitoring Stations–CERP SDE; GISLIB.RECOVER_MONITORING; LO_INVERTEBRATE SFWMD Canals–CERP SDE; HYSUR_CANAL_CRL_SFWMD

Lake Okeechobee Phytoplankon and Cyanotoxin Monitoring Stations. Map Author: Laura Biddison, CERP GIS Map Technician Map Updated: June 4, 2007 Map Location: \\cerp\projects\GIS\PRGM_03\map_docs\cmn4598_SSR_07Maps\LakeO_Phytoplankton _cmn4598.mxd Base Imagery: Land Sat Imagery 2004 Datasets used: Phytoplankton Monitoring Sites–CERP SDE; GISLIB.RECOVER_MONITORING;

LO_PHYTOPLANKTON Cyanotoxin Monitoring Sites-CERP SDE; GISLIB.RECOVER_MONITORING; LO_CYANOTOXIN SFWMD Canals–CERP SDE; HYSUR_CANAL_CRL_SFWMD

Lake Okeechobee SAV Monitoring Stations Map Author: Laura Biddison, CERP GIS Map Technician Map Updated: May 22, 2007 Map Location: \\cerp\projects\GIS\PRGM_03\map_docs\cmn4598_SSR_07Maps\LakeO_SAV_cmn459 8.mxd Base Imagery: Land Sat Imagery 2004 Datasets used: SAV Transects–CERP SDE; GISLIB.RECOVER_MONITORING; LO_SAV_Transect_Rodusky SFWMD Canals–CERP SDE; HYSUR_CANAL_CRL_SFWMD

Chapter 9: The Status of Nonindigenous Species in the South Florida Environment

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SUMMARY

Successful restoration of the South Florida ecosystem, which includes only vestiges of a once vast Everglades, hinges on the ability to reverse the environmental degradation chiefly caused by human activities over the last 100+ years and to prevent further degradation. While efforts of the Comprehensive Everglades Restoration Plan (CERP) and Restoration Coordination and Verification (RECOVER) programs have made it clear that restoration involves numerous factors (e.g., water quantity, water quality, and abundance of flora and fauna), the potential impact of invasive species has emerged as a high priority for CERP planning. Invasion of South Florida's natural habitats by nonindigenous (non-native or exotic) plant and animal species has significantly changed the ecosystem, particularly by displacing native species.

In support of the collective activities of the many agencies involved in Everglades restoration and CERP, this chapter reviews the broad issues involving nonindigenous species in South Florida and their relationship to restoration, management, planning, organization, and funding. This chapter also provides an overview of nonindigenous species using an "all-taxa" format for understanding and presenting an inclusive picture of the magnitude of the far-reaching invasive species threats that exist in South Florida. While detailed information on many nonindigenous species is still unknown, this document provides a complete listing with annotations for those species considered serious threats to Everglades restoration. The species are presented using the

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[3] U.S. Fish and Wildlife Service, A.R.M. Loxahatchee National Wildlife Refuge, Boynton Beach, FL
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RECOVER and Science Coordination Group (SCG) Modules for Everglades restoration. Species impacts are also discussed by region, as available. Supporting background information, including management tools used to control invasive exotic species in South Florida, is presented in the 2006 South Florida Environmental Report – Volume I, Chapter 9. Numerous groups and agencies are involved with nonindigenous species management. A summary of these agencies and their corresponding tasks and responsibilities as they pertain to nonindigenous species can be found on the Environmental Law Institute website in a report entitled *Filling the Gaps: Ten Strategies to Strengthen Invasive Species Management in Florida*.

In addition to providing a comprehensive look at nonindigenous species across taxa, this document takes an important step toward trying to determine what, if any, control or management has been initiated for targeted species. This progress assessment technique has been established along with the development of the SCG systemwide ecological indicators for invasive plants through coordination among the SCG, the Noxious Exotic Weed Task Team (NEWTT), and the Florida Invasive Animal Task Team (FIATT) of the South Florida Ecosystem Restoration Task Force (SFERTF). Continued collaboration is expected to put in place a coherent and integrated method for evaluating progress on controlling invasive plants. It is anticipated that a parallel system for exotic animals will be developed within the next two to three years.

This chapter covers the entire Central and Southern Florida Restudy area, which encompasses approximately 18,000 square miles (sq mi) from Orlando to the Florida Reef Tract with at least 11 major physiographic provinces:

Everglades	Biscayne Bay	Florida Keys
Big Cypress	Florida Reef Tract	Immokalee Rise
Lake Okeechobee	Near-shore coastal waters	Kissimmee River Valley
Florida Bay	Atlantic Coastal Ridge	

The Kissimmee River, Lake Okeechobee, and the Everglades are the dominant watersheds, connecting a mosaic of wetlands, uplands, coastal areas, and marine areas. This area includes all or part of 16 counties: Monroe, Miami-Dade, Broward, Collier, Palm Beach, Hendry, Martin, St. Lucie, Glades, Lee, Charlotte, Highlands, Okeechobee, Osceola, Orange, and Polk.

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NONINDIGENOUS SPECIES AND EVERGLADES RESTORATION

Control of invasive non-native species is an important issue for the overall ecological health of South Florida's public conservation lands. The importance of this issue in the Everglades Protection Area (EPA) is demonstrated by the great number of plans, reports, statements, and papers written by numerous committees, state and federal agencies, public and private universities, state and federal task forces, and various other organizations. Most of these documents support an "all-taxa" approach. The consensus of these parties is that control and management of invasive nonindigenous species is a critical component of ecosystem restoration in South Florida.

The topic of invasive species has been identified as an issue since the beginning of the Everglades restoration initiative. Several organized efforts and mandates have highlighted the problems associated with exotic species in the Everglades region. Control and management of invasive nonindigenous species are in the priorities established by the SFERTF in 1993. One of the tasks in the 1993 charter for the former Management Subgroup (December 16, 1993) was to develop a restoration strategy that addressed the spread of invasive exotic plants and animals.

The U.S. Fish and Wildlife Service (USFWS) was designated as the lead agency for this strategy and submitted a brief report (Carroll, 1994). Among issues highlighted in the report are:

1. A limited number of species are designated as "nuisance" species and can be prohibited by law.
2. Current screening processes are deficient.
3. Responsibilities remain vague.
4. There is a general lack of awareness and knowledge of the harmful impacts of invasive species.
5. An urgent need exists for statewide coordination and cooperation to eliminate exotic species.

The USFWS report indicated the greatest obstacle to combating invasive non-native species is the lack of sufficient funding and manpower.

The South Florida Ecosystem Restoration Working Group's (SFERWG) first Annual Report in 1994 addressed all invasive nonindigenous plant and animal species. The overall objectives stated were to (1) halt or reverse the spread of invasive species already widespread in the environment; (2) eradicate invasive species that are still locally contained; and (3) prevent the introduction of new invasive species to the South Florida environment. The 1994 Everglades Forever Act (EFA) requires the District to establish a program to monitor invasive species populations and to coordinate with other federal, state, and local governmental agencies to manage exotic pest plants, with an emphasis in the EPA. This work is ongoing through various interagency working groups.

One such group (the Everglades Cooperative Invasive Species Management Area, or CISMA) is working to improve coordination, control, and management of invasive species through the designation of an Everglades invasive species management area. The group is modeled after very successful partnerships in western states known as Cooperative Weed Management Areas (IWCC, 2005). Representatives from the USNPS, USFWS, South Florida Water Management District (SFWMD or District), Florida Fish and Wildlife Conservation

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Commission (FWC, formerly the Florida Game and Fresh Water Fish Commission), Florida Department of Environmental Protection (FLDEP), Florida Department of Transportation (FLDOT), Florida Power & Light (FP&L), U.S. Army Corps of Engineers (USACE), the Seminole Indian Tribe of Florida, and the Miccosukee Tribe of Indians of Florida have met several times to develop a Memorandum of Understanding, which will be distributed for signature this year. Additionally, the group has worked to enhance the District's treatment database (WEEDAR) into a multi-agency system to track invasive species treatment throughout the region. This will allow participating agencies to store, compile, and analyze treatment data from all agencies. Other activities will involve developing an expert's directory, coordinating control and monitoring activities through a region-wide strategy, developing early detection and rapid response programs, and identifying research priorities. To facilitate coordination on these activities, the District and the U.S. Department of the Interior (USDOI) co-sponsored the 4th Annual Everglades Invasive Species Summit in July 2007. Land managers from each entity provided operational updates on their invasive plant and/or animal control programs, shared lessons learned, and participated in workshops intended to improve coordination and identify needs and gaps. During the meeting, participants developed a framework for a multi-agency program, which included a reporting system, identified experts for taxonomic confirmation, risk assessment tools, and rapid-response teams to eradicate new populations.

Reinforcing all efforts is the SFERTF Scientific Information Needs Report (SSG, 1996), which contains a region-wide chapter on harmful invasive non-native species. An overall regional objective for restoration is to develop control methods for nonindigenous species at entry, distribution, and landscape levels. The specific objectives are to halt and reverse the spread of established invasive nonindigenous species and to prevent invasions by new nonindigenous species. The major issues in South Florida are inadequate funding for scientific investigations to develop effective controls, lack of funding to apply control methods to problem species, and delays and lack of consistency in responses to these new problems. Most resources for nonindigenous animals have focused on agricultural pests, with little investigation of species that threaten natural areas. Accelerated study of control technologies and the basic biology and ecology of invasive nonindigenous species are needed to answer the following priority questions: (1) How will water management alterations affect introduced plants and animals? (2) What are the principal controls on expansion of a species? (3) What are the impacts of invasive nonindigenous species on native species and ecosystems? (4) What makes a natural area susceptible to invasion? and (5) What are the most effective screening and risk assessment technologies to help focus on the greatest potential problems? Overall, the major issue is the lack of meaningful information concerning the effects of invasive nonindigenous species in South Florida.

The Comprehensive Review Study Final Feasibility Report and Programmatic Environmental Impact Study (USACE and SFWMD, 1999) addresses the presence of non-native animals as one of several factors that preclude serious consideration of achieving true restoration of the natural system, one in which nonindigenous species are not present. The report discusses how removal of canals and levees, which act as deepwater refuges for non-native fish and as conduits into interior marshes for other species, may help to control invasive species by slowing further movement into relatively pristine areas. On the other hand, restoration of lower salinity levels in Florida Bay might result in increases of reproductively viable populations of nonindigenous fishes, such as the Mayan cichlid, in the freshwater transition zone. These unintended negative consequences of the restoration effort must be addressed during the detailed design.

The USFWS Coordination Act Report for the Comprehensive Everglades Restoration Plan (CERP) also considers control and management of non-native species as a critical aspect of

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ecosystem restoration in South Florida. The report discusses the effects of the present canal and levee system and of the preferred alternative of this system on the distribution of nonindigenous animals. Some components of CERP involve construction of canals and reservoirs, which could provide additional conduits from points of introduction into the Everglades for organisms such as fish, amphibians, and snails. Other components involve removal or partial removal of canals, processes that should reduce the spread of non-native fishes. Removal of levees, which act as artificial terrestrial corridors into the wetland landscape, should reduce the spread of species such as the fire ant (*Solenopsis invicta*) and Burmese python (*Python molurus bivittatus*). The USDOI also recommended establishment of the FIATT to work on the issue as part of CERP. For the planned Water Preserve Areas and flow-ways, it was recommended that an aggressive plan be developed for the perpetual removal of invasive nonindigenous plants and animals. It was also recommended that existing control measures should be accelerated, techniques that are more effective should be developed, and regulations should be revised and better enforced to prevent additional introductions of exotic species (FGFWFC, 1999). USACE and SFWMD (1999) responded that in CERP this recommendation [team] should be presented to the SFERTF.

Several other plans and reports also include invasive nonindigenous species. The Coordination Act Reports (FGFWFC, 1999) from the FWC emphasize that the extent of the canal system's role in the spread of non-native fishes into natural marshes — as opposed to the fish remaining primarily in the disturbed areas — is debatable. The draft report, A New Look at Agriculture in Florida (Evans, 1999), discusses the introduction of non-native pests and diseases as a serious obstacle to sustainable agriculture and addresses the importance of exclusion and control strategies. The South Florida Multi-Species Recovery Plan (USFWS, 1999) identifies non-native animal control as a restoration need for two-thirds of the ecological communities and the individual species covered in the plan. In addition, the South Florida Regional Planning Council's 1991 and 1995 regional plans for South Florida list the removal of nonindigenous plants and animals and discouragement of introductions as regional policies (SFRPC, 1991; 1995).

In 2002, USACE authorized the Melaleuca Eradication and Other Exotic Plants project. This project was listed in the Central and Southern Florida Comprehensive Review Study (Restudy) as an "other project element," but funding was not initially authorized for it under CERP in the 1999 Water Resources Development Act. The 2002 authorization assigned the project's four major components at an estimated cost of $5.5 million for the USACE. These components include the following:

1. A cost-share agreement with the University of Florida for the design and construction of a new facility for biocontrol in Ft. Pierce, Florida. This facility was designed and constructed by the University of Florida without federal cost-sharing participation. An additional facility was designed and constructed by USACE at Davie, Florida with USDOI and SFWMD funding.

2. A cost-share agreement with the Florida Department of Agriculture and Consumer Service (FDACS) for the design and construction of the upgrade and renovations for the existing biocontrol facility in Gainesville, Florida. This component was not pursued due to funding constraints.

3. A cost-share agreement with the SFWMD for the "controlled release" of biological agents. In July 2004, a CERP Design Agreement amendment was approved by the District and USACE to proceed with development of this cost-share project. A final draft of the Project Management Plan (PMP) for this project was completed in January 2005. Work began on the Project Implementation Report (PIR) in July 2005. The PIR will seek to determine the best method to fund the rearing, release, and monitoring of approved biocontrol agents. It is anticipated that the project will benefit melaleuca (*Melaleuca quinquenervia*), Old World climbing fern (*Lygodium*

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microphyllum), Brazilian pepper (*Schinus terebinthifolius*) and Australian pine (*Casuarina equisetifolia*) biocontrol projects. The PIR is scheduled for completion in 2008, with the first appropriation expected in FY2010. Implementation of the project is anticipated to span 17 years with a federal cost of about $5.5 million.

4. The Special Reconnaissance Report on invasive species to determine federal interest and future federal involvement in invasive species projects in South Florida was completed in December 2005. This report incorporates the NEWTT's "Weeds Won't Wait" strategy and recommends federal involvement in developing a comprehensive plan for management of invasive species in South Florida in collaboration with other federal, state, and local agencies. A Project Delivery Team is being assembled to develop the Program Management Plan for the Invasive Species Master Plan to implement the recommendations from the report.

In a separate but complementary program, the FDEP also administers funding for invasive upland plant control efforts in Florida through regional working groups. The Upland Invasive Plant Management Program was established within the FDEP in 1997. To implement a statewide program, the FDEP formed Regional Invasive Plant Working Groups. This program funds individual non-native plant control projects on public conservation lands throughout the state based upon the working groups' recommendations. The FDEP melds these regional priorities into an integrated process that provides the needed support infrastructure (e.g., control method development, research results, oversight, and funding) to conduct an efficient and cost-effective statewide control program. Program funding is provided through the Invasive Plant Management Trust Fund, as set forth in Section 369.252(4), Florida Statutes (F.S.). Additionally, DEP provides leadership to Florida's Invasive Species Working Group (ISWG). The ISWG is an interagency group comprised of federal, state, local government agencies and other interested parties. It strives to coordinate invasive species activities and provide policy direction within state government.

Public awareness of invasive species and their impacts to Florida's natural resources is an important component of successful invasive species prevention and management efforts. Promoting behavioral changes of individuals and industries can help curtail the introduction of potentially invasive non-native species. A 2006 FWC-funded invasive species awareness study found that roughly 50 percent of Floridians have some knowledge of invasive species issues and most strongly agree that invasive species represent a significant threat to Florida's natural resources and human welfare.

State and federal agencies involved in natural resource protection have a variety of programs to educate the public and industries. These agencies regularly produce and distribute at outreach events printed media such as weed identification cards and flyers. For instance, the FWC collaborated with other agencies to publish an eight-page insert on invasive species in a 2006 Sunday edition of the *Orlando Sentinel*. The insert reached approximately 600,000 readers. A South Florida edition is planned for publication in the *Miami Herald* in February of 2008. Figure 9-1 depicts a sign produced by the District and National Park Service as part of outreach efforts pertaining to animal releases on canal and levee right-of-way.

The ISWG web site at http://iswgfla.org/ includes news, education, and other resources promoting public awareness. Likewise, other state and federal agencies have continually expanded invasive species educational content on their websites and improved cross-agency website linking to further facilitate access to invasive species information.

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Despite these education and outreach programs, the FWC survey suggests that more efforts are needed to raise invasive species awareness among Floridians. Additional funding and improved interagency coordination are both needed to adequately reach the growing and often transient Florida population. The Statewide Invasive Species Strategic Plan for Florida called on the ISWG to make recommendations for a coordinated public awareness campaign. Consequently, the ISWG established a public education sub-working group composed of communications professionals from member agencies charged with providing specific recommendations for implementing a public awareness campaign. The *Miami Herald* newspaper insert mentioned above is a result of this sub-working group. The sub-working group is also cooperating with a new interagency invasive species awareness effort being coordinated by the FWC.

Figure 9-1. Sign posted throughout the southern part of the District as part of a public awareness campaign.

BIOLOGICAL MONITORING FOR NONINDIGENOUS SPECIES IN SOUTH FLORIDA

Monitoring programs are important in establishing the extent of a problematic species and can offer valuable spatial information for ecological purposes, control purposes and benchmarks once operational control programs begin. Similarly, long-term, repeatable monitoring is key to answering questions related to the impacts of invasive species. The general occurrence of most invasive nonindigenous plants in South Florida are fairly well understood (Wunderlin et al., 1995; FLEPPC, 2005), although detailed information on distributions and expansion rates are lacking. Agency-sponsored programs are in place that track the regional distribution of certain target exotic plant species, yet spatial data for most other invasive taxa in natural areas is lacking or not readily accessible. The FWC maintains a county-level database for reptiles, amphibians, birds, and terrestrial mammals at (http://www.myfwc.com/critters/exotics/exotics.asp). FWC biologists compiled these data from both published and unpublished sources. The U.S. Geological Survey (USGS) maintains an extensive database for nonindigenous aquatic species by watershed (P. Fuller, personal communication). This report makes extensive use of these valuable resources, but it is difficult to glean information about species population dynamics without more detailed location and/or historical spatial data.

The distributions of several animal species are tracked at a higher level of detail in South Florida, but not in a consistent cross-taxa manner and not by any single agency. For instance, varying agencies track detailed distributions of Burmese python (*Python molurus bivittatus*), lobate lac scale (*Paratachardina lobata lobata*), and Mexican bromeliad weevil (*Metamasius callizona*). While these single-species monitoring programs do successfully track individual species, the state has no coordinated database that spans taxa. Moreover, obstacles to monitoring invasive animals are considered in part, "the nature of the beast," as tracking mobile organisms is inherently more difficult than documenting the occurrences of plants.

Remote sensing (RS) technologies have been applied to operational invasive species programs to date with only limited success. RS technologies useful for mapping generalized plant communities cannot accurately identify small incipient plant populations, and are often unable to provide precise spatial coordinates of exotic species presence, both critical needs for invasive plant managers. Additionally, RS technologies cannot yet consistently detect target plants growing under and among the canopy of other plants; researchers must spend considerable time and energy ground-truthing data gained from aerial photos and satellite images. Agency-sponsored invasive plant control operations are ongoing throughout Florida, and the coverage of the target invasive plants changes constantly. Given time and budgetary constraints, resource managers often opt to kill the target species and map treatment sites rather than create detailed coverage maps prior to beginning a treatment program. Therefore, RS technologies are acknowledged as successful for mapping large invasive plant monocultures, but the usefulness of resulting data to on-the-ground resource managers tasked with controlling species is limited.

The Everglades Forever Act (EFA) requires the SFWMD to conduct surveys to measure the extent of exotic plants in the Everglades Protection Area (EPA). Systematic Reconnaissance Flight (SRF) surveys were initiated to give operational resource managers a tool to quickly and affordably assess target plant populations and gauge successes or failures. The SRF method is widely used in tracking wildlife (Russell et al., 2001; Dalrymple, 2001; Mauro et al., 1998). It involves flying at a fixed height and speed across a study area on a predetermined transect while observers count targets (plants or animals) in a strip of land on either side of the aircraft.

The U.S. Forest Service (USFS) conducted the initial survey for melaleuca in South Florida in 1980 (Cost and Craver, 1980). This survey was initiated in order to estimate forested and non-forested land cover in the area south of Lake Okeechobee. The data derived from this survey was valuable in documenting the problems associated with melaleuca in the Everglades and helped to legitimize melaleuca spread as an issue in the state of Florida.

In the early 1990s, the SFWMD and the National Park Service (NPS) began conducting independent, parallel SRF surveys for exotic plants in the region. The District surveys covered the entire peninsula south of the north rim of Lake Okeechobee (8 million acres). The transects, modeled after the USFS 1980 survey, were spaced at 2.5-mile intervals east and west across the state. The NPS surveys focused on national park lands in the region. NPS transects were finer (at 1-km intervals), and observers deviated from the transect when exotic plant populations were encountered. Both surveys recorded plant species and density classifications. In 1999, the District and the NPS began to conduct the biannual surveys collaboratively. The surveys are now nested, with the District survey using 4-km transects and the NPS using 1-km transects; the transects overlap on federal lands (Ferriter and Pernas, 2005).

The SFWMD conducts surveys of the EPA biannually as required by the Everglades Forever Act, but has expanded the scope of the survey in recent years to include the entire District (2005) and the entire range of several key species (2006). Due to its geographical extent (almost 20 million acres) and the fact that the survey is only flown in the winter months to optimize plant detection, the survey has been compartmentalized. Portions of the state are flown each year in an alternating regional design to allow for complete coverage of the study area. Past survey results (1993 through 2005) are available for viewing at http://maps.google.com/ and able to be downloaded in shapefile format at http://tame.ifas.ufl.edu/ (Ferriter and Pernas, 2005). Results from the most recent surveys (2006 through 2007) and acreage estimates for priority species are provided in this document and shapefiles of the 2006/2007 data will be available on the website in August 2007.

The 2007 SRF survey aimed to cover the entire range of melaleuca in Florida as part of the TAME Melaleuca project (**Table 9-1**). Survey teams flew east-west transects up the peninsula to the area just south of Gainesville. It is generally considered that this expanded study area includes the entire range of melaleuca, Old World climbing fern, Brazilian pepper, and Australian pine in Florida. Distribution of these four species is depicted in **Figures 9-2** through **9-5**. This study area was expanded even further and mapped for the occurrence of cogongrass (**Figure 9-6**). Occurrences of melaleuca, Old World climbing fern, and Australian pine did not continue northward throughout the expanded study area. However, occurrences of Brazilian pepper were recorded along the east coast of Florida throughout the expanded survey area, indicating that its range extends northward in coastal areas of the state as does that of cogongrass.

Table 9-1. Nonindigenous plant acreage estimates based on results of 2007 SRF survey. Note that survey area includes the Florida peninsula south of Gainesville, but acreage estimes are for the District only.

SPECIES	ACRES
Melaleuca *(Melaleuca quinquenervia)*	273,014
Old World climbing fern *(Lygodium microphyllum)*	159,220
Brazilian pepper *(Schinus terebinthifolius)*	695,202
Australian pine *(Casuarina equisetifolia)*	207,197
Cogongrass (*Imperata cylindrica*)	6,897

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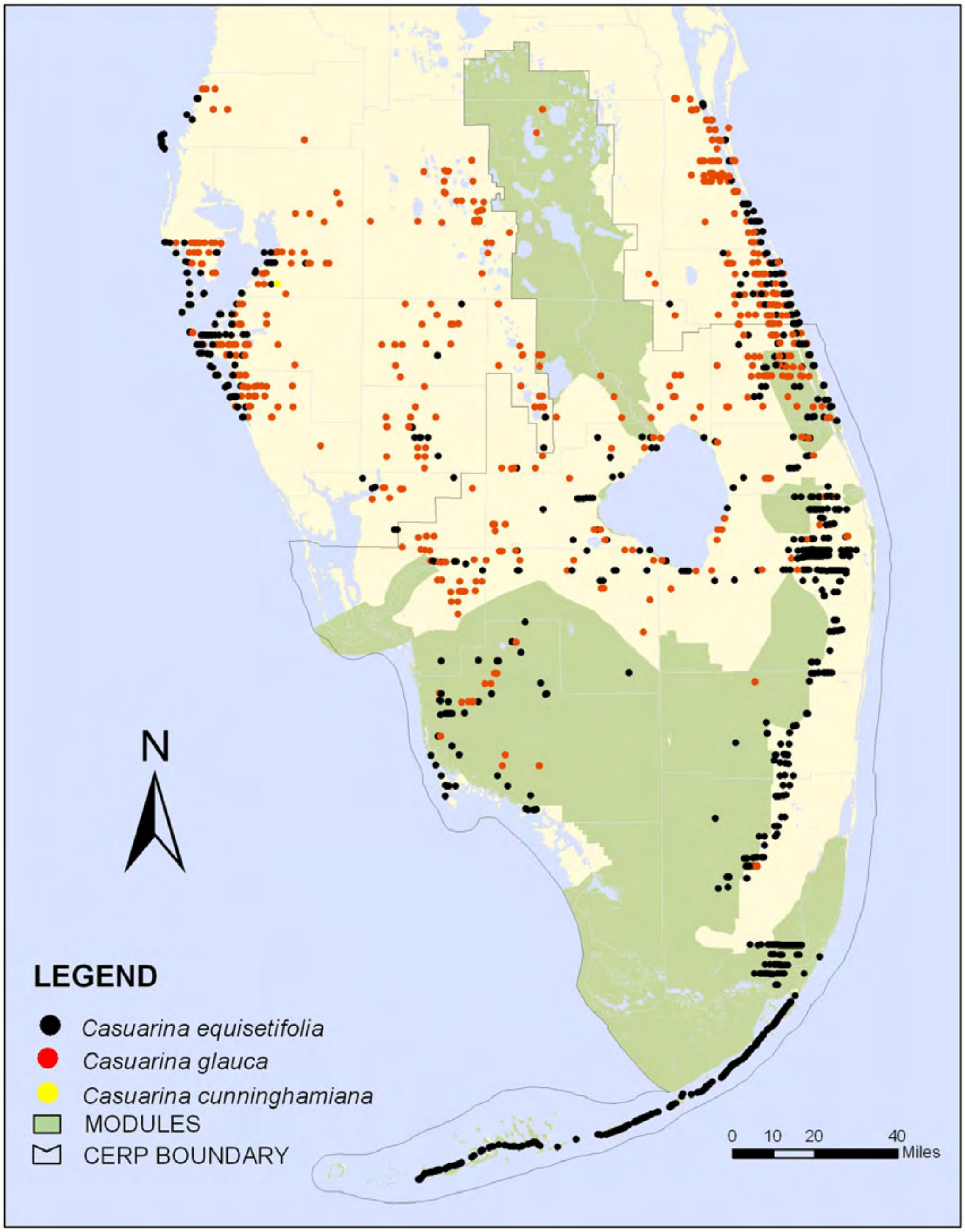

Figure 9-2. Distribution of Australian pine *(Casuarina* spp.*)* across South Florida (2007).

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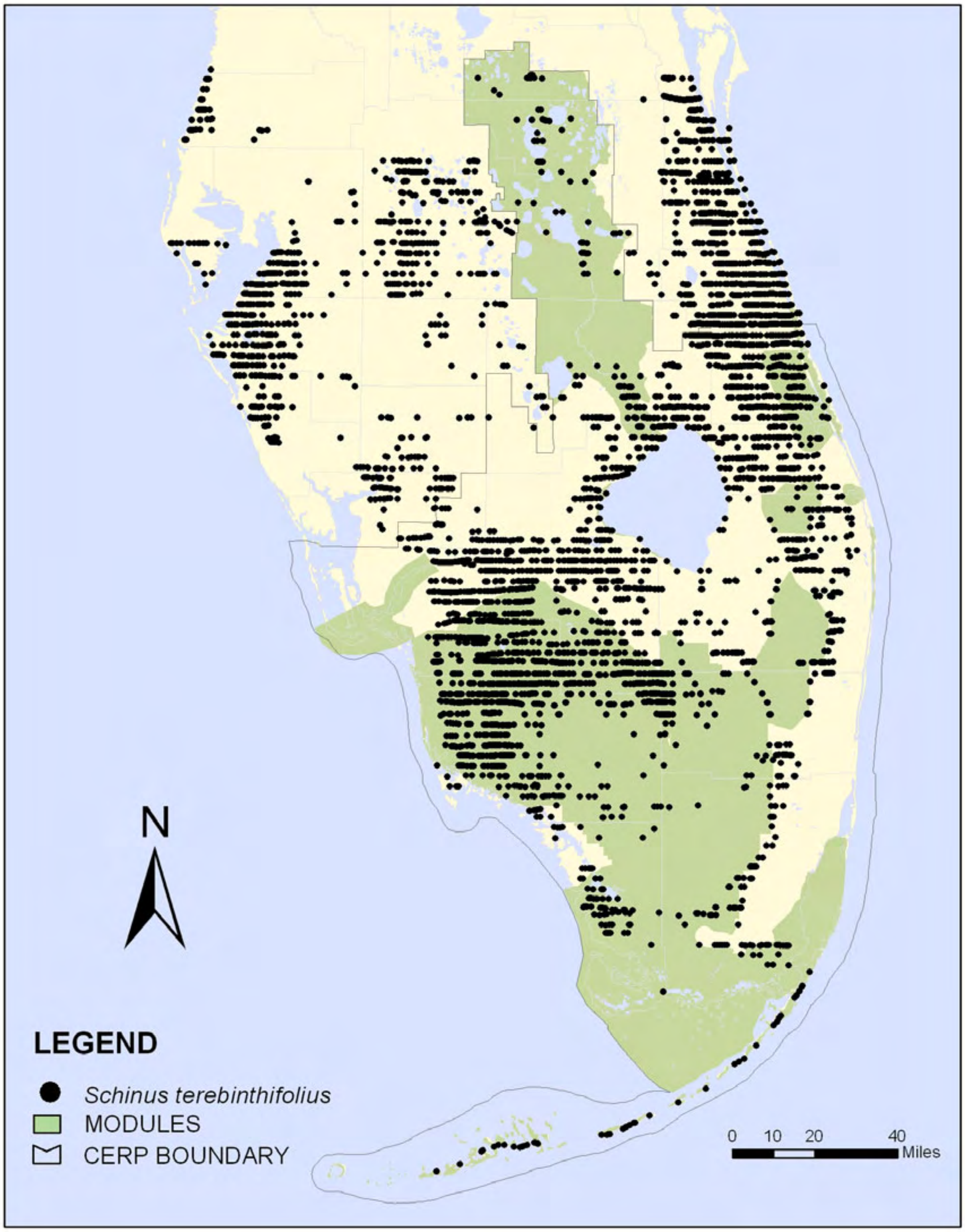

Figure 9-3. Distribution of Brazilian pepper (*Schinus terebinthifolius*) across South Florida (2007).

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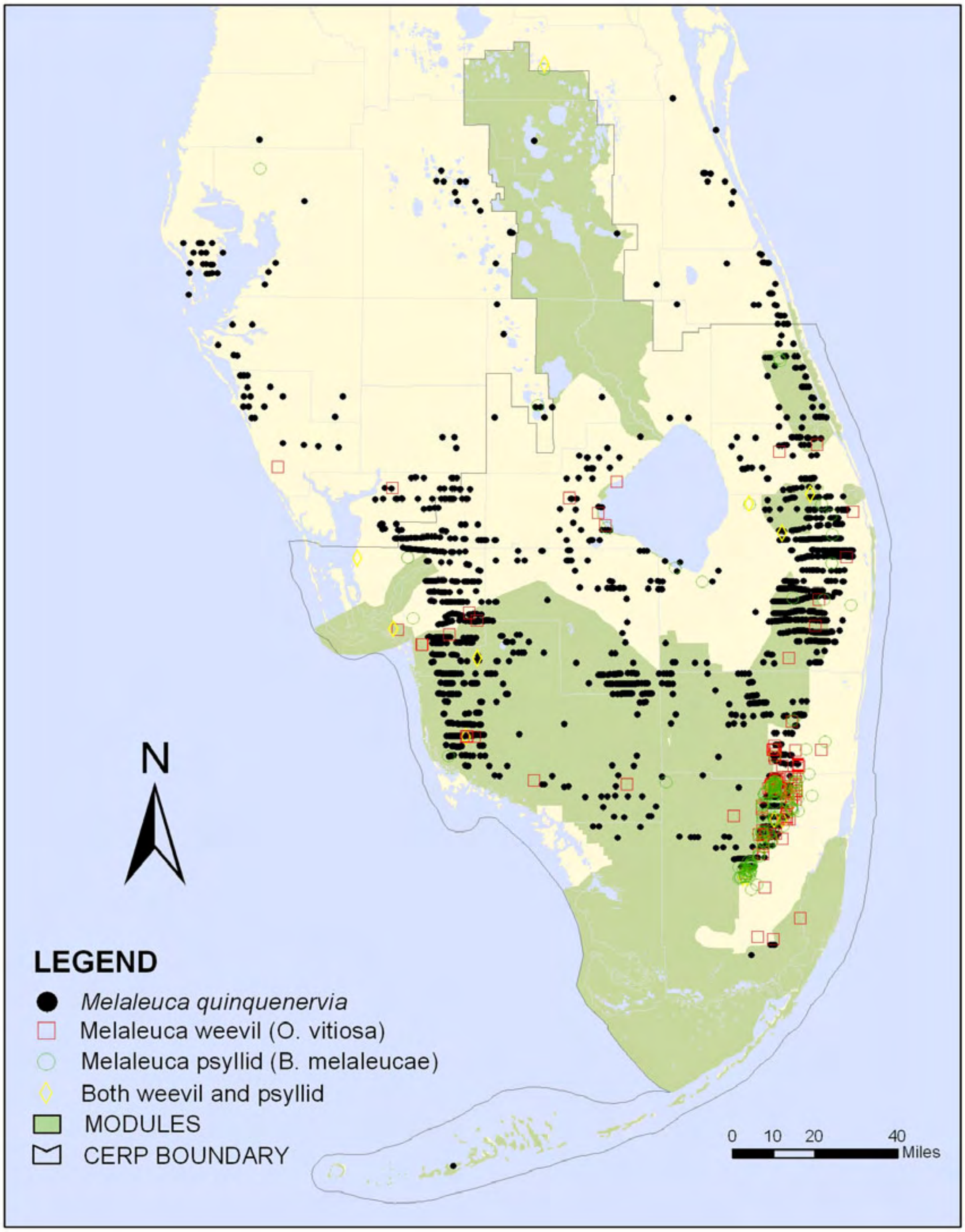

Figure 9-4. Distribution of melaleuca *(Melaleuca quinquenervia)* across South Florida (2007) and sites of original biocontrol agent releases since 1997. (Release site data courtesy of P. Pratt, USDA-ARS.)

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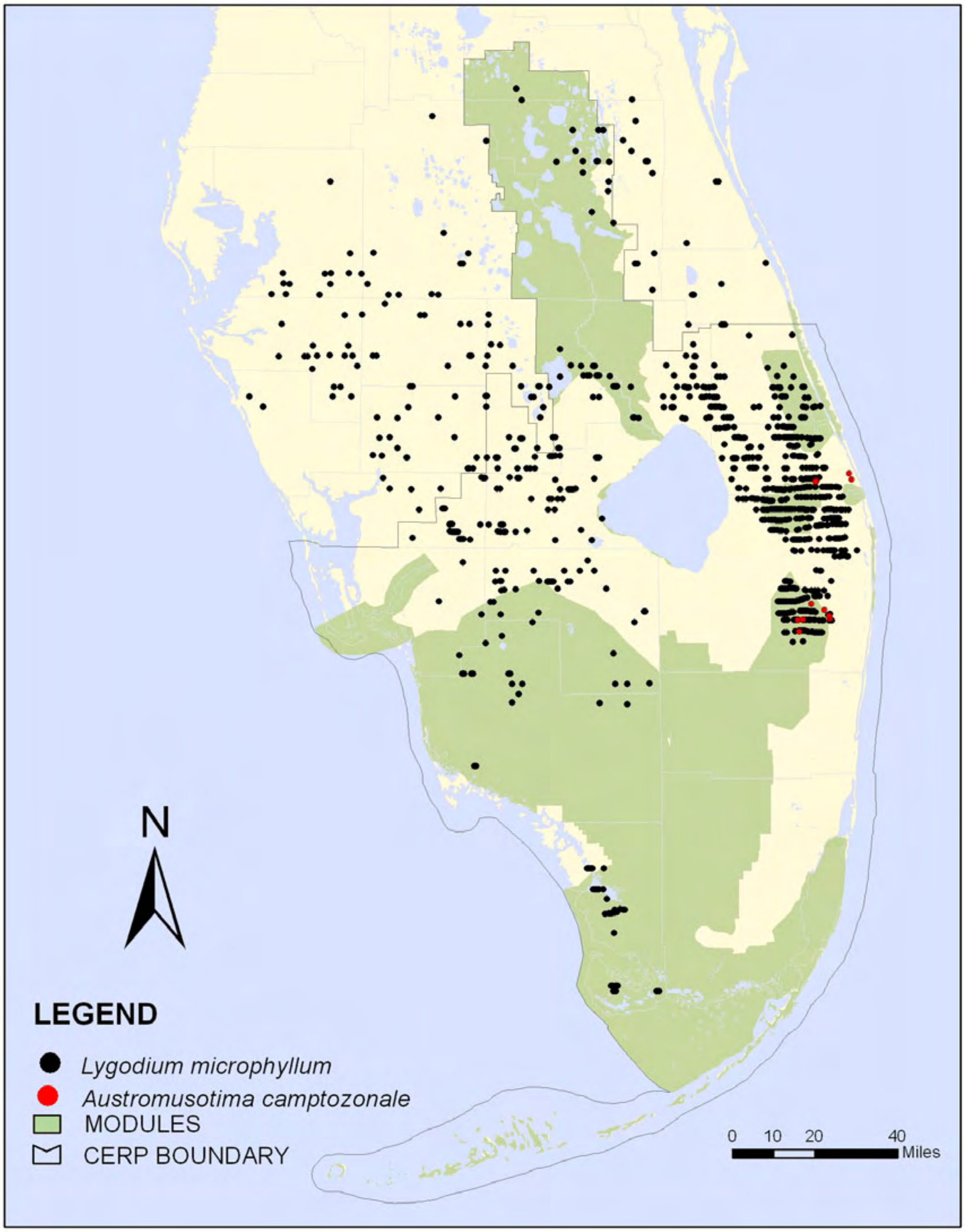

Figure 9-5. Distribution of Old World climbing fern *(Lygodium microphyllum)* across South Florida (2007) and biocontrol (*Austromusotima camptozonale*) release sites from 2006.
(Release site data courtesy R. Pemberton, USDA-ARS).

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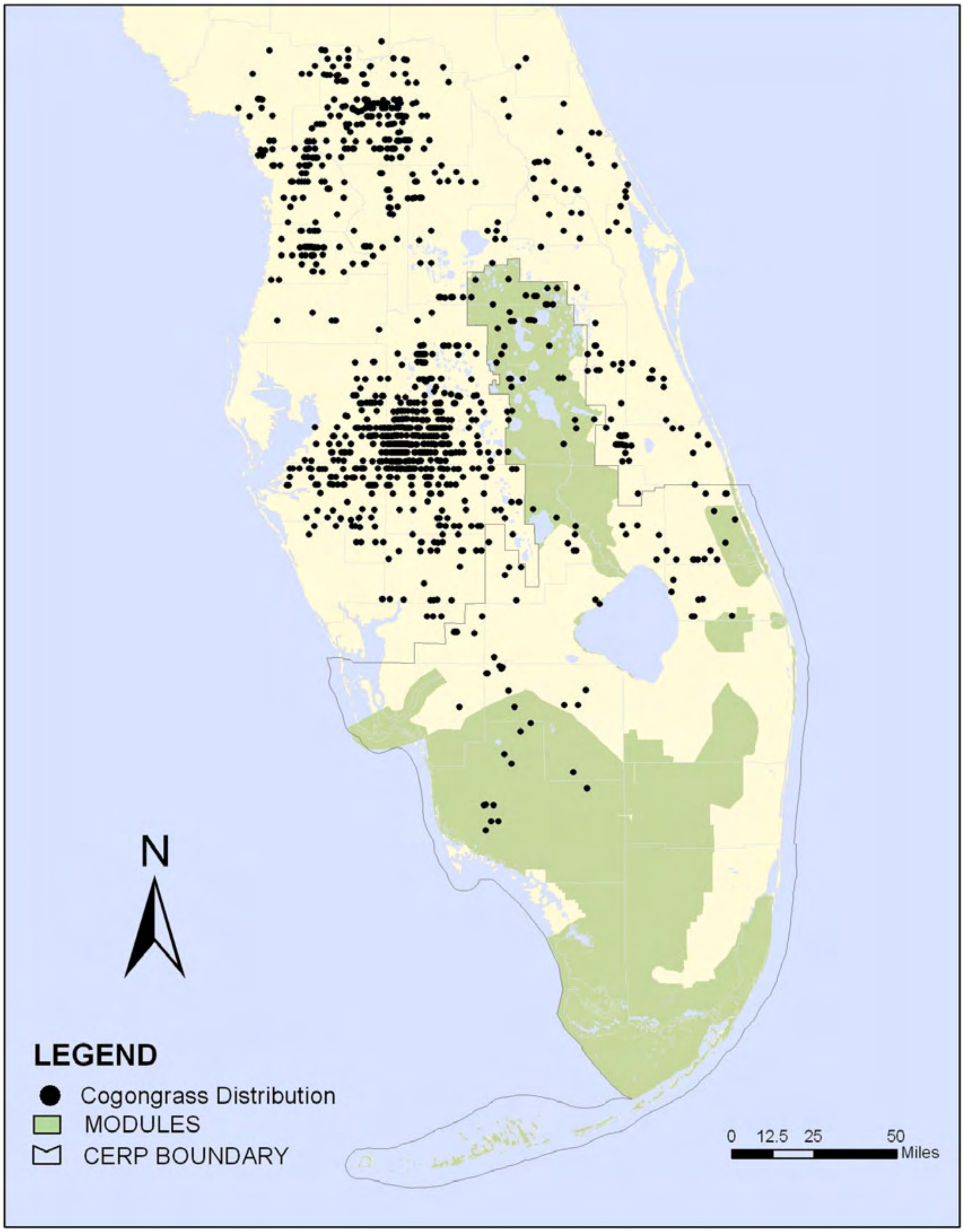

Figure 9-6. Distribution of cogongrass (*Imperata cylindrica)* across South Florida (2007).

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AN ASSESSMENT OF NONINDIGENOUS SPECIES IN SOUTH FLORIDA

Significant scientific evidence and research reveals that invasive exotic plants are degrading and damaging natural ecosystems in South Florida (see Doren and Ferriter, 2001). These species cause significant ecological harm by crowding out and displacing native vegetation upon which native fish and wildlife depend for food and shelter. Other negative impacts of invasive species can include the (1) alteration of soil types and soil and water chemistry, (2) alteration of ecosystem functions such as carbon sequestration and nutrient cycling, (3) attenuation of gene pools and genetic diversity, (4) reduction of native species diversity, and (5) alteration of community composition. Most exotic plants provide little or no habitat value for native wildlife, yet they can change in hydrology and soil composition, degrade water quality, and decrease the biodiversity of an entire ecosystem. The distribution, magnitude, and impacts of exotic animals in South Florida are poorly understood. If the Everglades is to be restored and preserved, and if South Florida's natural environments are to remain intact, then the problem of invasive plant and animal species must be addressed comprehensively and with sufficient resources.

Sixteen different federal and state agencies, numerous local agencies, and two Indian tribes are involved in Everglades restoration and, thus, in one or more activities related to the management, regulation, control, interdiction, and prevention of invasive exotic species in Florida. Collectively, these agencies have management authority for more than 13.7 million acres (about 21,500 sq mi) of Florida's natural lands. Individual agencies have identified 32 of the 66 priority plant species named in *Weeds Won't Wait* as particularly serious and specifically targeted for control (Doren and Ferriter, 2001). Nevertheless, the process of documenting problems associated with exotic animal species in South Florida began only recently (Goodyear, 2000; A. Roybal, USFWS, personal communication).

The many agencies supporting CERP and the broader restoration efforts coordinated by the SFERTF target invasive species as a serious threat to the Everglades Restoration Initiative and restoration program goals. This is the first report to use an all-taxa approach to identify nonindigenous species by region and organize these species spatially, thus launching the process of prioritizing species in terms of threat posed to Everglades restoration.

This report organizes nonindigenous species data using the terms, geographical references, and structure developed by Restoration Coordination and Verification (RECOVER) — an arm of CERP responsible for linking science and the tools of science to a set of systemwide planning, evaluation and assessment tasks (**Figure 9-7**). The Science Coordination Group (SCG) 2005 Recommendations for Interim Goals and Interim Targets for CERP also are considered. In addition, RECOVER has identified invasive species as "drivers" and "stressors" in the conceptual ecological models (CEMs). The CEMs include Florida Bay, Everglades Ridge and Slough, Southern Marl Prairies, Greater Everglades, Everglades Mangrove Estuaries, Big Cypress Regional, Lake Okeechobee, and Loxahatchee Watershed (at http://www.evergladesplan.org/pm/recover/recover.cfm). CEMs and the performance measures and ecological indicators derived from them serve as the basis for adaptive management activities and the development of "Vital Signs" (systemwide ecological indicators) for Everglades restoration by the SFERTF. Additional information on CERP and RECOVER is presented in Chapters 7A and 7B of this volume, respectively.

Information in this chapter is organized according to these established formats to maintain consistency among the many different agencies and personnel working on Everglades restoration projects. Nonindigenous species are presented by occurrence within eight geographic divisions, or modules, related to the South Florida restoration programs:

- Florida Keys
- Florida Bay and the Southern Estuaries
- Greater Everglades
- Big Cypress
- Lake Okeechobee
- Northern Estuaries – East
- Northern Estuaries – West (Caloosahatchee Estuary)
- Kissimmee River Basin

The plant and animal species lists for each module presented in **Tables 9-2** and **9-4** through **Table 9-11** were compiled from the FWC exotic animal occurrence data, USGS watershed data, the Exotic Animal Report (Goodyear, 2000), Florida Exotic Pest Plant Council data (www.fleppc.org), peer review from NEWTT and FIATT members, and interviews with land managers. Within the geographic areas, animal species are divided by broad taxonomic groups — amphibians, reptiles, birds, mammals, fish, and invertebrates. In addition, the animal table indicates whether a species is widely or locally distributed (i.e., occurring in all modules or all but one module, or in only one module). This distribution information indicates the scope of the problem and, in the future, may help agencies to prioritize animal species for control and management in the region.

Due to limited availability of animal distribution data, lists in **Table 9-2** may not be comprehensive or entirely accurate. For instance, some nonindigenous species listed for a module may actually occur outside of the module noted in **Table 9-2**, because the listing relies on incomplete county data as the most specific location data available. The lists have been developed and refined through peer review by taxonomic experts and land managers to reflect regional considerations (such as coastal versus inland habitats), but should be used with the knowledge that animal distribution data — especially across taxa — is deficient in Florida.

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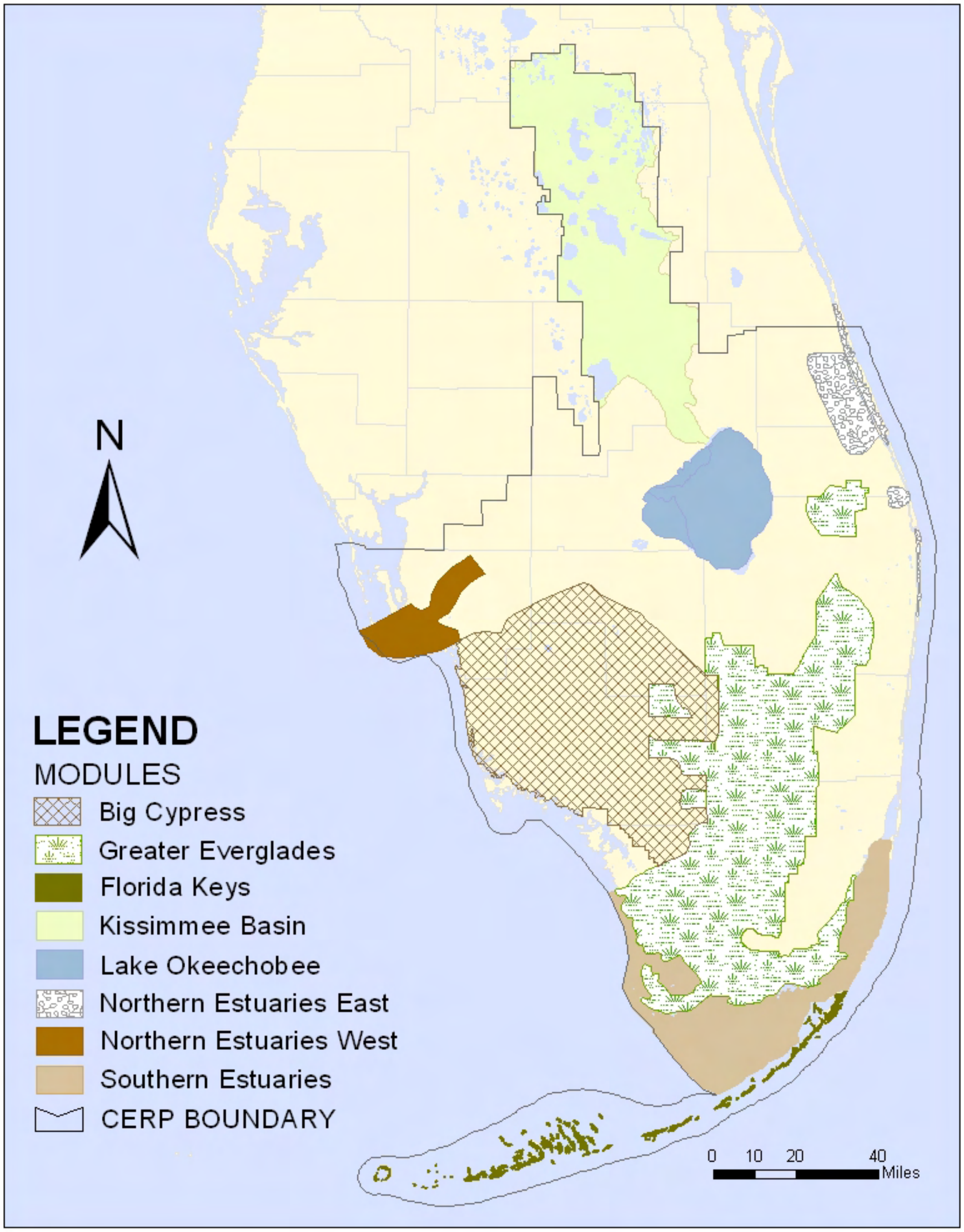

Figure 9-7. The nonindigenous species information in this report is organized using the terms, geographical references, and structure developed by Restoration Coordination and Verification (RECOVER).

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Table 9-2. Summary of South Florida's nonindigenous animal species by RECOVER module.[11]

		KY	SE	GE	BC	NW	NE	LO	KR
Amphibians									
Bufo marinus	Giant toad	x	x	x	x	x	x	x	x
Eleutherodactylus planirostris	Greenhouse frog	x	x	x	x		x	x	x
Osteopilus septentrionallis	**Cuban treefrog**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
Eleutherodactylus coqui	Coqui	x	x	x				x	
Reptiles									
Agama agama	African redhead agama	x	x	x	x		x	x	
Ameiva ameiva	Giant ameiva		x	x			x	x	
Anolis chlorocyanus	Hispaniolan green anole		x	x			x	x	
Anolis cristatellus cristatellus	Puerto Rican crested anole		x	x					
Anolis cybotes	Largehead anole		x	x			x	x	
Anolis distichus	Bark anole	x	x	x	x	x	x	x	
Anolis equestris equestris	Knight anole	x	x	x	x	x	x	x	x
Anolis extremus	Barbados anole					x			
Anolis garmani	Jamaican giant anole		x	x		x	x	x	
Anolis porcatus	Cuban green anole		x	x					
Anolis sagrei	**Brown anole**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
Basiliscus vittatus	Brown basilisk		x	x	x		x	x	
Boa constrictor	Common boa			x	x				
*BOIGA IRREGULARIS**	BROWN TREE SNAKE								
Caiman crocodiles	Common caiman			x				x	
Calotes mystaceus	Indochinese tree agama							x	x

Table Key		
KY = Keys	NW = Northern Estuaries West	Green — Found in one module
SE = Southern Estuaries	NE = Northern Estuaries East	Orange — Found in all modules
GE = Greater Everglades	LO = Lake Okeechobee	Blue — Found in all but one module
BC = Big Cypress	KR = Kissimmee River	

***** Species designated for Early Detection and Rapid Response
Species entries in bold indicate they are discussed in Modules
Species entries in red indicate new additions to the Report
SPECIES ENTRIES IN CAPITAL LETTERS NOT PRESENT IN SOUTH FLORIDA BUT REPRESENT EXTREME RISK (FIATT, 2007)

Table Summary		
Found in 1 Module	Found in All Modules	Found in All but 1 Module
0 amphibians	2 amphibians	1 amphibian
6 reptiles	6 reptiles	5 reptiles
3 birds	4 birds	2 birds
5 mammals	6 mammals	0 mammals
18 fish	0 fish	1 fish
42 invertebrates	5 invertebrates	0 invertebrates

[11]Due to limited availability of animal distribution data, species lists presented in table are not comprehensive, but are considered representative of the species found within the modules.

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Table 9-2. Continued.

		KY	SE	GE	BC	NW	NE	LO	KR
Reptiles *(continued)*									
Calotes versicolor	Oriental garden lizard						x		
Chamaeleo calyptratus	Veiled chameleon					x			
Cnemidophorus lemniscatus	Rainbow lizard		x	x					
Cnemidophorus motaguae	Giant whiptail		x	x					
Cosymbotus platyurus	Asian flattail house gecko		x	x		x			
Ctenosaura pectinata	**Mexican spinytail iguana**		**x**	**x**					
Ctenosaura similis	**Black spinytail iguana**		**x**	**x**	**x**	**x**			
Eunectes notaeus	**Yellow anaconda**				**x**				
Gekko gecko	Tokay gecko	x	x	x	x	x			
Gonatodes albogularis fuscus	Yellowhead gecko	x	x	x	x		x		
Hemidactylus frenatus	Common house gecko	x	x	x	x	x	x	x	
Hemidactylus garnotii	Indo-pacific gecko	x	x	x	x	x	x	x	x
Hemidactylus mabouia	Tropical house gecko	x	x	x	x	x	x	x	x
Hemidactylus turcicus	Mediterranean gecko	x	x	x	x	x	x	x	x
Iguana iguana	**Green iguana**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	
Leiocephalus carinatus armouri	**Northern curlytail lizard**	**x**	**x**	**x**	**x**		**x**	**x**	**x**
Leiocephalus personatus scalaris	Green-legged curlytail lizard			x					
Leiocephalus schreibersii schreibersii	Red-sided curlytail lizard		x	x					
Leiolepis belliana belliana	Butterfly lizard	x	x	x	x		x	x	x
Mabuya multifasciata	Many-lined Grass Skink		x	x					
Phelsuma madagascariensis grandis	Giant day gecko	x	x	x	x	x			
Phrynosoma cornutum	Texas horned lizard		x	x			x	x	x
Python molurus bivittatus	**Burmese python**	**x**		**x**	**x**				
Ramphotyphlops braminus	Brahminy blind snake	x	x	x	x	x	x	x	x
Sphaerodactylus argus argus	Ocellated gecko	x	x	x	x				
Sphaerodactylus elegans elegans	Ashy gecko	x	x	x	x				
Tarentola annularis	White-spotted wall gecko		x	x		x			
Tarentola mauritanica	Moorish wall gecko		x	x		x			
Trachemys scripta elegans	Red-eared slider	x	x	x	x	x			x
Varanus niloticus	**Nile monitor**		**x**	**x**	**x**	**x**			**x**
Varanus salvator	Water monitor			x					

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Table 9-2. Continued.

		KY	SE	GE	BC	NW	NE	LO	KR
Birds									
Acridotheres tristis	Common myna	x		x			x	x	
Brotogeris chiriri	Yellow-chevroned parakeet	x				x			x
Cairina moschata	Muscovy duck	x		x	x	x	x	x	
Columba livia	Rock dove	x	x	x	x	x	x	x	x
Myiopsitta monachus	Monk parakeet	x		x	x	x	x	x	x
Nandayus nenday	Black-hooded parakeet					x			
Passer domesticus	House sparrow	x	x	x	x	x	x	x	x
Porphyrio porphyrio	**Purple swamphen***			**x**					
Streptopelia decaocta	Eurasian collared-dove	x	x	x	x	x	x	x	x
Sturnus vulgaris	**European starling**	**x**		**x**	**x**	**x**	**x**	**x**	**x**
Threskironis aethiopicus	**Sacred ibis***			**x**					
Zenaida asiatica	**White-winged dove**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
Mammals									
Canis familiaris	Feral dog	x	x	x	x	x	x	x	x
Capra hircus	Feral goat								x
Chlorocebus aethiops	Vervet monkey			x					
Cricetomys gambianus*	**Gambian pouch rat**	**x**							
Felis catus	**Feral cat**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
Lepus californicus	Black-tailed jackrabbit		x	x			x	x	
Macaca mulatta	Rhesus monkey		x	x					
Molossus molossus tropidorhynchus	Pallas's mastiff bat	x	x	x					
Mus musculus	House mouse	x	x	x	x	x	x	x	x
Mustela putorius	Ferret								x
Nasua narica	White-nosed coati		x	x			x	x	x
Rattus norvegicus	Norway rat	x	x	x	x	x	x	x	x
Rattus rattus	Black rat	x	x	x	x	x	x	x	x
Saimiri sciureus	Squirrel monkey		x	x	x				x
Sciurus aureogaster	**Mexican red-bellied squirrel**		**x**						
Sus scrofa	**Feral pig**			**x**	**x**	**x**	**x**	**x**	**x**
Vulpes vulpes	Red fox	x	x	x	x	x	x	x	x

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Table 9-2. Continued.[12]

		KY	SE	GE	BC	NW	NE	LO	KR
Fishes									
Astronotus ocellatus	**Oscar**		**x**	**x**	**x**			**x**	**x**
Belonesox belizanus	Pike killifish	x	x	x	x				
Callichthys callichthys	Cascarudo								x
Channa marulius	Bullseye snakehead		x						
Chitala ornata	Clown knife		x						
Cichla ocellaris	**Butterfly peacock cichlid**		**x**	**x**					
Cichlasoma bimaculatum	**Black acara**		**x**	**x**	**x**		**x**	**x**	**x**
Cichlasoma citrinellum	Midas cichlid			x					
Cichlasoma managuense	Jaguar guapote			x	x				
Cichlasoma festae	**Guayas cichlid**					**x**			
Cichlasoma octofasciatum	Jack Dempsey						x		
Cichlasoma salvini	Yellowbelly cichlid			x					
Cichlasoma urophthalmus	**Mayan cichlid**		**x**	**x**	**x**	**x**	**x**	**x**	
Clarias batrachus	**Walking catfish**		**x**	**x**	**x**	**x**	**x**	**x**	**x**
Dorosoma petenense	Threadfin shad								x
Geophagus surinamensis	Redstriped eartheater			x					
Hemichromis letourneuxi	**African jewelfish**	**x**		**x**	**x**	**x**			**x**
Heros severus	Banded cichlid			x					
Hoplosternum littorale	**Brown hoplo**			**x**	**x**	**x**	**x**	**x**	**x**
Hypostomus plecostomus	Suckermouth catfish			x					
Macrognathus siamensis	**Spotfinned spiny eel**			**x**					
Monopterus albus	**Asian swamp eel**			**x**					
Oreochromis aureus	**Blue tilapia**		**x**	**x**	**x**		**x**	**x**	**x**
Oreochromis mossambicus	**Mozambique tilapia**			**x**		**x**	**x**		
Pterygoplichthys disjunctivus	**Vermiculated sailfin catfish**								**x**
Pterygoplichthys multiradiatus	**Orinoco sailfin catfish**			**x**				**x**	**x**
Sarotherodon melanotheron melanotheron	Blackchin tilapia						x		
Tilapia mariae	**Spotted tilapia**		**x**	**x**	**x**	**x**	**x**		
Tilapia zillii	Redbelly tilapia						x		
Xiphophorus hellerii	Green swordtail						x		
Xiphophorus maculatus	Southern platyfish						x		
Xiphophorus variatus	Variable platyfish						x		

[12] This list contains only established records of nonindigenous fish according to the USGS definition (reproducing and overwintering population). Comprehensive exotic fish lists were reviewed by USGS experts (Bill Loftus and Pam Fuller), and FWC experts (Shafland, 1996) with unique knowledge of the subject. The FWC lab uses a more conservative listing of established fishes (permanent populations so widespread no elimination is possible). The USGS listing was chosen primarily because it provides an indication of species present and capable of expansion in the future. However, any FWC occurrences not listed by USGS are included here. There were some differences between USGS listings, so Loftus occurrences were authoritative for KY, SE, GE and BC; Fuller for NW, NE, LO and KR; as agreed by those reviewers.

Table 9-2. Continued.

		KY	SE	GE	BC	NW	NE	LO	KR
Invertebrates									
Aedes albopictus	Asian tiger mosquito	x	x	x	x	x	x	x	x
Aethina tumida	Small hive beetle						x		
*AGRILUS PLANIPENNIS**	EMERADL ASH BORER								
Amblyomma auricularium	Reptilian tick				x				
Amblyomma chabaudi	Madagascan tortoise tick			x					
Amblyomma exornatum	Monitor lizard tick			x		x			
Amblyomma fimbriatum	Reptilian tick					x			x
Amblyomma flavomaculatum	Yellow-spotted monitor lizard tick			x		x			
Amblyomma helvolum	Reptilian tick				x				
Amblyomma humerale	Reptilian tick			x					
Amblyomma latum	Snake tick			x		x			x
Amblyomma marmoreum	African tortoise tick			x	x	x			
Amblyomma nodosum	Reptilian tick			x					
Amblyomma nuttalli	Small reptile tick			x		x			
Amblyomma sabanerae	Neotropical tortoise tick			x	x				
Amblyomma varanense	Asian monitor lizard tick			x					
Apis mellifera scutellata	African bee			x					
Aulacaspis yasumatsui	Armored scale insect			x					
Balanus reticulatus	Barnacle		x						
Balanus trigonus	Barnacle		x			x	x		
Blattella asahinai	Asian cockroach	x		x			x		
Cactoblastis cactorum	**Cactus moth**	**x**	**x**				**x**		
Callinectes bocourti	Bocourt swimming crab		x						
Cepolis varians	Caribbean land snail		x						
Ceroplastes rusc	Fig wax scale			x		x			
Chaetanophotrips leeuwenia	Thrips			x					
Charybdis helleri	Indian Ocean portunid crab						x		
Chelymorpha cribraria	Tortoise beetle		x	x					
Cipangopaludina japonica	Japanese mysterysnail								x
Cittarium pica	West Indian trochid	x							
Corbicula fluminea	**Asian clam**		**x**	**x**		**x**		**x**	**x**
Craspedacusta sowerbyii	Freshwater jellyfish		x	x					x
Crocothemis servilia	Scarlet skimmer			x		x		x	x
Cryptosula pallasiana	Bryozoan						x		
Cuthona perca	Lake Merritt cuthona		x						
Daphnia lumholtzi	Water flea		x	x				x	x
*DREISSENA POLYMORPHA**	ZEBRA MUSSEL								
Erythemis plebeja	Black pond hawk			x					
Eupristina masoni	Wasp			x					

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Table 9-2. Continued.

		KY	SE	GE	BC	NW	NE	LO	KR
Invertebrates *(continued)*									
Glossodoris sedna	Marine nudibranch	x	x						
Haliplanella luciae	Sea anemone		x			x			
Hyalomma aegyptiujm	Reptilian tick			x					
Iridomyrmex humilis	Argentine ant	x	x	x	x	x	x	x	x
Litopenaeus stylirostris	Pacific white shrimp	x							
Litopenaeus vannamei	Pacific white shrimp	x							
Littorina littorea	Common periwinkle	x	x						
Lyrodus mediolobatus	Indo-Pacific shipworm						x		
Marisa cornuarietis	Giant Rams-horn snail		x	x		x			
Melanoides tuberculatus	Red-rim melania		x	x	x				
Metamasius callizona	**Mexican bromeliad weevil**			**x**	**x**	**x**	**x**		
Micrathyria aequalis	Spottedtailed skimmer			x					
Micrathyria didyma	Three-striped skimmer			x					
Monomorium pharaonis	Pharaoh ant	x	x	x	x	x	x	x	x
Myllocerus undatus	**Sri Lanka Mimic Weevil**						**x**		
Mytella charruana	**Charru mussel**						**x**		
Oceanaspidiotus araucariae	Scale			x					
Ozamia lucidalis	Moth	x							
Parapristina varticillata	Wasp			x					
Paratachardina lobata	**Lobate lac scale**	**x**		**x**	**x**	**x**	**x**		
Paratrechina longicornis	Crazy ant	x	x	x	x	x	x	x	x
Perna viridis	**Green mussel**				**x**	**x**	**x**		
Phyllorhiza punctata	Spotted jellyfish						x		
Pinctada margaritifera	Black-lipped pearl oyster						x		
Pomacea bridgesii	Spiketop applesnail		x	x	x				
Pomacea insularum	**Island applesnail (= Channeled applesnail)**			**x**			**x**	**x**	**x**
RAOIELLA INDICA*	RED PALM MITE								
Retithrips syriacus	Thrips			x					
Solenopsis invicta	Imported fire ant	x	x	x	x	x	x	x	x
Sphaeroma terebrans	Wood-boring isopod		x			x			
Sphaeroma walkeri	Fouling isopod		x				x		
Styela plicata	Sea squirt						x		
Sundanella sibogae	Bryozoan						x		
Technomyrmex albipes	White-footed ant			x	x		x		
Tridacna crocea	Giant clam		x						
Tridacna maxima	Giant clam		x						
Truncatella subcylindrica	Snail	x	x	x					
Victorella pavida	Bryozoan						x		
Wasmannia auropunctata	Little fire ant			x					
Watersipora subovoidea	Bryozoan						x		
XYLEBORUS GLABRATUS*	REDBAY AMBROSIA BEETLE								
Zachrysia provisoria	Cuban garden snail	x							

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EXOTIC PLANT INDICATORS

The SFERTF directed the SCG to develop a suite of ecological indicators to help determine whether CERP restoration is being achieved. This suite is intended to reflect systemwide ecological indicators and restoration compatibility indicators for "built system" projects. The ecological indicators are to incorporate important "cross-scale features" of the Everglades, including biogeographic regions (see module names in **Figure 9-7**), vegetation mosaic and exotic interactions, landscape characteristics, and numerous physical and biological properties.

The indicator for invasive exotic plants is not similar in nature or context to other RECOVER indicators because nonindigenous species are inherently ill-suited to indicate ecological function, process, or structure, especially in the context of restoration. In addition, measurements of their biological "performance" do not reflect how they may or may not affect restoration. While the spread of nonindigenous plants may change ecological function and structure, it does not necessarily indicate anything of the overall ecological condition (or restoration) except as it pertains to the level of invasion and resultant adverse impacts to the ecosystem. However, restoration efforts could fail without active control and management of nonindigenous species, because these species have the capacity to drastically alter the natural environment (Mack et al., 2000). Therefore, the invasive exotic plant indicator is being developed to allow regular reporting on the status, progress, and outlook of nonindigenous plants in the context of the South Florida ecosystem restoration initiative.

It is important to note that this assessment only synthesizes existing sources of information to allow evaluation of and reporting on the status of invasive plant species. This constraint underlies the design and application of indicator questions; pilot indicators cannot be used to answer questions outside of available parameters. Each module — and each priority species within each module — are assessed based on six parameters:

1. Number of different invasive exotic plant species present.
2. Number, abundance, and frequency of new exotic plant species in the ecosystem.
3. Number and abundance of extant invasive exotic plant species found in new locations.
4. Location and density of invasive exotic plants, particularly in relation to native plant communities.
5. Rate of invasive exotic plant spread, especially in relation to restoration activities (e.g. removal of canals or levees).
6. Effectiveness of control actions/programs for invasive exotic plants, generally measured as a decrease in spatial extent of a species.

The individual responses are collated into a single response in the "stoplight" tables found within each module. While the development of an assessment/monitoring program specifically designed for this purpose would be ideal, the exotic plant indicator is currently constrained to using existing monitoring/research programs that collect information on nonindigenous plants.

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MODULES OVERVIEW

For each of eight modules, this report includes a narrative of relevant nonindigenous species issues. Priority plant species are presented in an indicator-based stoplight table (in which a red "stoplight" indicates a severe negative condition). Pilot exotic plant indicator tables are also provided to demonstrate the use of the indicator tool in gauging progress in overall agency-sponsored invasive plant control efforts as related to the restoration initiatives. In **Table 9-3**, the District's Fiscal Year (FY) 2007 expenditures on nonindigenous plant control are summarized by module. The District spent over $23.8 million in FY2007 for overall invasive plant prevention, control, and management in South Florida. Distribution of the five species for which systemwide control efforts are under way is presented in **Figures 9-2** through **9-6**.

Table 9-3. Summary of invasive plant species control expenditures by module by the District in FY2007.

	Lake Okeechobee	Kissimmee	Big Cypress	Greater Everglades	Northern Estuaries East	Northern Estuaries West	Systemwide Biological Control
Australian Pine *(Casuarina equisetifolia)*	--	$742	$584	$307,720	--	--	$20,000
Brazilian Pepper *(Schinus terebinthifolius)*	$279,328	$88,664	$31,031	$591,062	$386,119	--	$49,000
Shoebutton Ardisia *(Ardisia elliptica)*	--	--	--	$222,619		--	--
Old World Climbing Fern *(Lygodium microphyllum)*	--	$254,164	--	$955,015	$125,658	--	$150,000
Melaleuca *(Melaleuca quinquenervia)*	$362,235	--	$159,999	$4,723,739	--	--	$150,000
Torpedograss *(Panicum repens)*	$2,658,657	100,633	--	$1,484	--	--	--
Cogongrass (*Imperata cylindrica*)	--	--	--	--	$6,576	--	--

While overall animal taxa lists have been provided for each module (**Table 9-2**) and certain animal species are discussed as priorities in the individual modules, no attempt is made to "score" animal taxa as part of an indicator. It should be noted that the table does not imply that the individual species are expanding or negatively influencing the respective modules. This table, representing nonindigenous species of interest in a geographic framework, provides a baseline list of organisms that occur in the modules and have the potential to impact restoration efforts.

Priority animal species are discussed in modules where agency efforts to deal with the individual species are ongoing, where evidence suggests that these species are causing negative impacts, or to highlight the need for resources or early detection and rapid response efforts. While most agencies strive to use scientific data to support the management of these priority species, these data are often unavailable. Consequently, agency managers must use their best judgment in initiating control programs for these animal species.

Figure 9-8. Giant toad (*Bufo marinus*) (Photo by Craig G. Morley, Global Invasive Species Database).

It is important to note that certain nonindigenous animal species occur in almost every module. These species (32 total) include the giant toad (*Bufo marinus*, **Figure 9-8**), Cuban brown treefrog (*Osteopilus septentrionallis*), green iguana (*Iguana iguana*), monk parakeet (*Myiopsitta monachus*), and feral dog (*Canis familiaris*). Not all of these species are described in detail because they cannot all be adequately covered in this chapter. Omitting specific mention of some of these species in module narratives does not imply that the species are not problematic, or that they should not be controlled. On the contrary, work is urgently needed to establish distribution and biological data for these organisms, given their ubiquitous nature in South Florida. For additional information on those organisms not discussed in detail herein, readers may refer to extension documents put out by the University of Florida, or visit the links listed on this University of Florida extension site at http://pcb2441.ifas.ufl.edu/list%20of%20species.htm (as of November 6, 2007).

Many nonindigenous plant species, too, are problematic in multiple modules, though their biology, ecological impact, and the control efforts put forth against them may be described in detail in only one module.

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FLORIDA KEYS MODULE

The Florida Keys Module was created as a separate module because it is a unique and important ecological unit that is part of the South Florida environment, but it was not included in the scope of CERP. Unlike virtually every other habitat in Florida, the invadable land area is relatively small in the Florida Keys. This allows land managers to prioritize species effectively and deal systematically with relatively small parcels. Through the well-coordinated Florida Keys Invasive Exotics Task Force, a list of priority animal and plant species has been developed. The updated priority animal species list is expected to be complete by 2008 and will include a ranking of priority animals along with suggested eradication methods. Land managers are currently inventorying all the land within this module, documenting the presence of priority plant species on both public and private holdings. The maps resulting from this effort are expected to be finished by the close of 2007 (A. Higgins, The Nature Conservancy [TNC], personal communication). Virtually all listed conservation lands are considered to be under maintenance control for target plant species, and other public lands are being addressed. As work to assess, prioritize, and control nonindigenous animals in the Florida Keys has begun, this module is the best organized for an all-taxa approach to management and control of invasive plant and animal species and is likely to serve as a model for other regions in South Florida.

Nonindigenous Plants

Although public lands in the Florida Keys are well maintained, land managers report that populations of some species (e.g., seaside mahoe, and half-flower) are decreasing on public lands but increasing on private lands because of continued horticultural landscape use. Although latherleaf (*Colubrina asiatica)* appears to be decreasing on public lands as a result of systematic control efforts, challenges in detecting this sprawling coastal shrub species make it difficult to determine whether populations are decreasing overall in the Florida Keys. In the past, localized problems developed with sickle bush (*Dichrostachys cinerea)* and laurel fig (*Ficus microcarpa,* **Figure 9-9**). However, both were targets of coordinated control measures that resulted in their eradication. Both species are still actively searched for, but neither inhabits the Keys at this time.

Figure 9-9. Laurel fig *(Ficus microcarpa)* (Photo by Vic Ramey, Univ. Florida).

Other priority species such as sapodilla (*Manikara zapota)* are problematic in localized areas, especially hardwood hammocks and old homesteads. Species such as leadtree (*Leucaena leucocephala)* and umbrella tree (*Schefflera actinophylla)* are increasing chiefly along roadsides and in disturbed sites (**Figure 9-10**). Resource managers in the Keys note that leadtree is particularly difficult to control with herbicides. Priority plant species are listed in **Table 9-4**.

Figure 9-10. Leadtree *(Leucana leucocephala)* (Photo by Ann Murray, Univ. Florida).

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Table 9-4. Stoplight table for priority plant species in the Florida Keys Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
FLORIDA KEYS MODULE (Results in this row reflect module-level questions, not species-level questions)	Y	Restoration efforts under way for several years; much progress made on most species; still some use of invasive species in private landscapes	Y	Significant control program for several years; progress on many species evident, continued monitoring and control needed to prevent serious reinvasions of species still threatening this region and new species	Y G
Australian Pine (*Casuarina* spp.)	Y G	Effective program in place and Australian pine not currently a problem in natural areas of Keys, decreasing on private	Y G	Chemical control effective with most natural areas clear or clearable with modest effort; biocontrol research under way	G
Latherleaf (*Colubrina asiatica*)	Y	Little known about spread throughout region; actively removed in coordinated manner	Y	Removal needed constantly, but coordinated control programs expected to keep populations at easily maintained levels	Y G
Sickle Bush (*Dichrostachys cinerea*)	Y	Actively searched for but effectively removed from module	G	Actively searched for but effectively removed from module	G
Laurel Fig (*Ficus microcarpa*)	Y	Actively searched for but effectively removed from module	G	Actively searched for but effectively removed from module	G
Leadtree (*Leucana leucocephala)*		Not new to module but considered new priority; controlled on public lands; increasing on private; prolific seedbank; resistant to chemicals	Y R	Control efforts increasing; control techniques being perfected	Y
Sapodilla (*Manilkara zapota*)	Y	Know little about spread throughout region; actively removed in coordinated manner	Y R	Localized problem; difficult to detect, may become serious pest in areas where other exotics controlled; invades natural forests; difficult to control	Y R
Half Flower (*Scaevola taccada*)	Y	Fairly easy to detect; actively removed from public land in coordinated manner; still popular for landscape on private land	Y R	Seeds float, long-term management difficult; biocontrol probably not option given closely related native *Scaevola* species	Y R
Brazilian Pepper (*Schinus terebinthifolius*)	Y	Invades most habitats, very destructive; chemical control ineffective in systemwide spread; local control programs proving effective in Keys	Y	Control programs effective in the Keys, with most populations limited; new biocontrol agents under study for future release in 2007-2008	Y
Seaside Mahoe (*Thespesia populnea*)		Not new to module, new to table; removed from public land in coordinated manner; still popular on private land; spreads easily;	Y	Active control program maintains populations; requires constant effort	Y

Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources, the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain green status.

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Nonindigenous Animals

In addition to the problems associated with nonindigenous plant species, this module also has several priority nonindigenous animals which threaten ecosystem function in the Florida Keys.

Cactoblastis

Cactoblastis cactorum is a South American moth whose larvae feed exclusively on species of prickly pear cactus (*Opuntia* spp.) (**Figure 9-11**). The moth was first discovered in North America on Big Pine Key in 1989. It was most likely introduced to Florida accidentally through the horticulture trade. Distribution of this species now occurs along the Atlantic coast to Charleston, South Carolina, and westward along the Gulf Coast to Dauphin Island, Alabama. The cactus moth is attacking and destroying native species of prickly pear and represents a substantial threat to the southwestern U.S. and Mexico, areas that are rich in cactus diversity and have substantial industries dependent on prickly pear cacti.

Figure 9-11. *Cactoblastis cactorum* larvae on *Opuntia* (Photo by Stephen Davis, USDA-APHIS).

In the Florida Keys, this moth threatens the endemic and endangered *O. corallicola* and other native prickly pear cacti, as well as populations of ornamental species. The U.S. Department of Agriculture's Agricultural Research Service (USDA-ARS) has conducted work to track the abundance and location of the moth with development of a female, sex pheromone-baited trap (**Figure 9-12**). USDA-ARS has also developed a Sterile Insect Technique (SIT) program as a control/exclusion strategy for this moth (S. Hight, ARS, personal communication). The SIT validation study continued for a second year at sites along the Florida panhandle and southern Alabama. Year-long sanitation efforts (removal of infested pads and cactus moth eggsticks, larvae, and pupae) reduced the densities of invading moths, but did not keep the moth population from rebounding. Combining sanitation with sterile insect releases, however, did substantially reduce the population of wild cactus moths. Sterile insects released in the wild were shown to be highly competitive against wild moths. Continued release and evaluation of sterile cactus moths at SIT validation sites is planned through 2007.

Figure 9-12. *C. cactorum* eggsticks (Photo by Ignacio Baez, USDA-ARS).

Although laboratory tests of insecticides show positive results for controlling the cactus moth, widespread use of pesticides may not be suitable for the Florida Keys due to the occurrence of rare and endangered *Lepidoptera* (e.g., Schaus swallowtail, Florida leaf-wing, and Bartram's scrub-hairstreak; M. Barett, USFWS, personal communication). Until effective control methods are developed, land managers in the Florida Keys are monitoring *Opuntia* spp. populations and manually removing impacted cactus pads. Fortunately, since the original infestation in early 2000, cactus moth outbreaks have occurred less frequently.

Gambian Pouch Rat

Gambian pouch rats (*Cricetomys gambianus*), native to Africa, were bred in captivity on Grassy Key. It is believed eight rats escaped between 1999 and 2002 and established a reproducing population. Gambian rats weigh an average of three pounds and measure

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20–35 inches from head to tail, which is much larger than native species, including the Key Largo wood rat, the cotton rat, and silver rice rat. Its large size makes this species popular in the exotic pet trade, although the Food and Drug Administration has banned their transport and sale because they are a carrier of monkey pox.

These rodents primarily eat fruits and grains, but are also known to eat invertebrates (Novak and Paradiso, 1991). Gambian rats are concentrated in the vicinity of dwellings near the initial release site on Grassy Key, although there has been dispersal to the adjacent Crawl Key. The population relies on refuse, pet food, and water from homeowners. Scientists are concerned this species is poised to move from Grassy Key onto adjacent keys, and then to Florida's mainland.

In February 2006, a pilot eradication project was initiated on Crawl Key where Gambian rat photographs were recorded in 2005. In June 2006, USDA-APHIS WS deployed 94 bait stations. Supplemental trapping was done to obtain rats for radio telemetry. It was determined that the combined effects of the eradication effort, along with impacts from Hurricane Wilma, eliminated this sub-population. Using previous trapping and radio telemetry, a bait-station grid was established for Grassy Key using a 40-meter grid in the "core" area. On the periphery, bait stations were placed 50 meters apart. Lot owners in the affected areas were contacted to seek access to their property for placing bait stations (**Figure 9-13**).

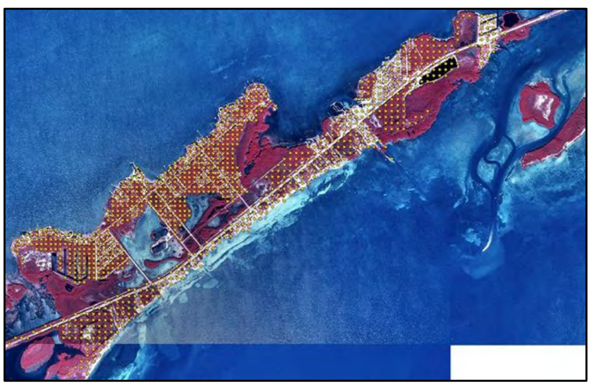

Figure 9-13. Bait station grid (left) and property owner permission status of Gambian pouch rat project (Figures by Scott Hardin, FWC).

From January to May, 2007, 1,000 bait stations were placed throughout Grassy Key hammock and residential areas. In March 2007, 20 Gambian pouched rats were trapped for the USDA APHIS National Wildlife Research Center (NWRC) for studies of more effective attractants and third generation rodenticides (**Figure 9-14**). As an indication that the pouched rat population was recovering from the impacts of Hurricane Wilma in late 2005, three of 11 females transported to the NWRC lab in Fort Collins, Colorado, had litters either in flight or shortly after arrival.

Figure 9-14. Gambian pouch rat to be transported to NWRC lab (Photo by Scott Hardin, FWC).

On May 21, the eradication effort commenced with the pre-baiting of roughly 600 stations around the periphery of the core area; actual toxic bait was removed from all stations by June 15. A high level of public awareness throughout the project resulted in no adverse public reactions during the active eradication phase. Two dead Gambian pouched rats were observed by residents, and several dead black rats were observed by WS staff.

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Intensive surveys using remote cameras and trapping will be conducted in July and September, 2007, to detect and eliminate any surviving Gambian pouch rats.

Funding for the project is provided by the Wildlife Foundation of Florida, Inc.; FWC; USDA-APHIS, Wildlife Services and National Wildlife Research Center; U.S. Fish and Wildlife Service, National Wildlife Refuges and Partners in Wildlife; and the District.

Green Iguana

Green iguanas (*Iguana iguana*) are native to Central and South America and some Caribbean islands, but have become well established in South Florida (Meshaka et al., 2004) (**Figure 9-15**). The range of the green iguana appears to be expanding in South Florida, having been initially limited to Dade County in 1966 and later expanding to Broward, Lee, Monroe, Palm Beach, Highlands, Hillsborough, Alachua, Indian River, Collier, Martin and St. Lucie counties. Breeding populations are established in seven of these counties (Meshaka et al., 2004).

Figure 9-15. Green iguana (*Iguana iguana*) (Photo by Stacey Sekscienski).

Green iguanas are popular in the pet trade and frequently escape or are released, although it is illegal to release iguanas and other non-native wildlife in Florida per Chapter 39-4.005, Florida Administrative Code (F.A.C.). They are generally found in suburban areas (especially with canals), agricultural areas, and hammock communities where they bask in open areas including sidewalks, docks, mowed regions, and exposed branches of trees. This long-lived species produces clutches of up to 49 eggs (Meshaka et al., 2004) and quickly reaches sexual maturity (males in 20 months, females in 31 months) (Smith et al., in press). Both traits have greatly contributed to its colonization success. High densities (up to 626 iguanas/km^2 have been reported for managed natural areas in South Florida (Smith et al., 2006; Smith et al., 2007; Smith et al., in press).

Adult green iguanas are generally herbivorous, feeding on foliage, flowers, and fruit, though they occasionally eat invertebrates. Iguanas consume both native and ornamental plant species in South Florida, and have also been found to prey on tree snails, especially *Drymaeus multilineatus* in Key Biscayne (Townsend, 2005). In the Florida Keys, iguana feeding could have serious implications for populations of other snail species, such as the stock island tree snail (*Orthalicus reses*), federally designated as a threatened species, and the Florida tree snail (*Liguus fasciatus*), a state-listed species of special concern.

In addition to eating valuable native and landscape plants, droppings of green iguanas are unsightly and unhygienic and a possible source of *salmonella* bacteria. Green iguanas weaken canals and levees with their extensive burrowing (see the *Greater Everglades Module* section, page 9-49), creating a maintenance liability to surface water infrastructure. They can contribute to weed seed dispersal through ingested seed and provide potential collision hazards on airport runways (Smith et al., in press). Furthermore, adult green iguanas are powerful animals that can bite and scratch, and aggressively whip with their tail (Smith et al., 2006). Although green iguanas normally avoid people, they will defend themselves if threatened, with males becoming more aggressive during mating season.

Green iguanas are a protected species in their native range because they are economically valued there and are often rare due to over-collection for the pet trade (at

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http://www.cites.org/eng/app/appendices.shtml). There are currently no agency-sponsored, coordinated control efforts for the nonindigenous green iguana in South Florida (including the Keys), though small-scale removal projects are in place (e.g., through a "Parknership" collaboration with the USDA-WS and Florida Park Service). Future controls likely will be implemented, however, given the region's expanding green iguana populations, impacts to water management operations and potential impacts of this nonindigenous species on native species such as the Florida burrowing owl (Makie et al., 2005; Smith et al., in press).

Feral Cat

FWC estimates populations of feral cats (*Felis catus*) to be between 6.3 and 9.6 million in the state of Florida (at http://www.floridaconservation.org). Worldwide, feral cats feed heavily on small birds, reptiles, and mammals, and have led to the extinction of numerous species. Feral cats (**Figure 9-16**) also spread diseases and parasites. In Florida, feral cats are known to prey upon the green sea turtle, roseate tern, least tern, Florida scrub-jay, Choctawhatchee beach mouse, Anastasia Island beach mouse, Key Largo cotton mouse, Southeastern beach mouse, Perdido Key beach mouse, Key Largo woodrat, Lower Keys marsh rabbit, and rice rat — all federally listed species.

Figure 9-16. Feral cat (Photo by Rex Williams, Chatham Island Taiko Trust, Global Invasive Species Database).

Although feral cats are problematic in all Modules, they are recently of particular concern in the Florida Keys. They have contributed to a 50 percent decline in populations of Hugh Hefner's rabbits (*Sylvilagus palustris hefneri*, an endangered subspecies of marsh rabbit named for Hefner's contributions to their research) on Big Pine Key (CNN.com, accessed May 20, 2007). Numerous trap-neuter-release programs that have been in place on the Keys and throughout South Florida for many years have proven ineffective. Consequently, wildlife officials began trapping the animals in May 2007, with the intent of removing and transporting them to animal shelters. Because escaped and abandoned cats continuously supplement feral cat populations, increased public awareness is needed to ultimately decrease populations of feral cats in the Keys and throughout South Florida.

Figure 9-17. Park biologist Jim Duquesnel, Joanne Potts and Clay DeGayner (left to right) (Photo by Britta Muizenieks, USFWS).

Burmese python

On Friday, April 13, 2007, graduate student Joann Potts and volunteer Clay DeGayner discovered the invasive Burmese python (described in detail in the *Greater Everglades Module*, page 9-47) inhabiting the Keys (**Figure 9-17**). This alarming find on Key Largo was compounded by the discovery of two woodrats, a federally listed endangered species, in the digestive tract of the captured python (J. Duquesnel, FDEP). This validates the concern that these invasive snakes pose an immediate threat to the ecological health and function of South Florida's ecosystem. It is unlikely that this was the only individual living in the Keys. Monitoring efforts will estimate python populations in the Keys and an eradication plan will follow.

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FLORIDA BAY AND SOUTHERN ESTUARIES MODULE

The Florida Bay and Southern Estuaries Module is made up of the coastal estuaries, coastal mangroves and islands of the southern Everglades. It is bordered by the Florida Keys Module to the southeast and the Greater Everglades Module to the north. This Module is a gradual transition between freshwater flowing from the mainland Everglades, and the open ocean. Nonindigenous species management in this region focuses on Florida Bay, the Bay's keys, coastal areas of Everglades National Park (ENP or Park), and the islands and mainland of Biscayne National Park. Control operations have been ongoing since the 1980s.

Nonindigenous Plants

The ecological effects of latherleaf have been most prevalent in this region (Jones, 1997). Latherleaf, first noted as naturalized in the module by Small (1933), is now well established and distributed throughout the coastal areas of the ENP and Biscayne National Park. This species occurs from the Ten Thousand Islands south to Cape Sable along the Gulf Coast and east along the northern fringe of Florida Bay to the Florida Keys.

Latherleaf invades coastal ridges just above the mean high-tide line (Russell et al., 1982), tropical hammocks, buttonwood and mangrove forests, and tidal marshes (Schultz, 1992). It also forms thickets on disturbed coastal roadsides. Latherleaf can invade disturbed and undisturbed forest sites (Olmsted et al., 1981; Jones, 1996), forming thick mats of entangled stems up to several feet deep, and growing over and shading out vegetation including trees (Langeland, 1990; Jones, 1996). This species is of particular concern in Florida's coastal hammocks, where it threatens a number of rare habitats and native plants, such as Florida thatch palm, Keys thatch palm, wild cinnamon, manchineel, cacti, bromeliads, and orchids (Jones, 1996).

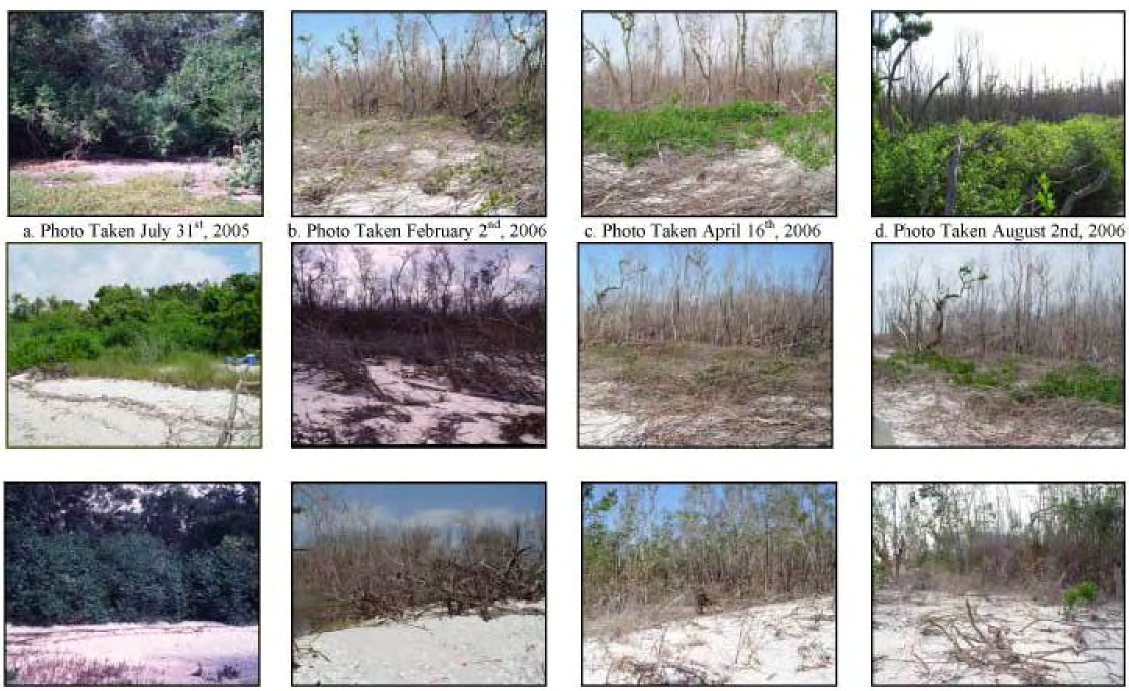

Figure 9-18. Vegetation communities in the lower Ten Thousand Islands were severely damaged first by Hurricane Rita and then by Hurricane Wilma. This time-series of photos demonstrates the ability of Latherleaf (*Colubrina asiatica*) to rebound following a major storm event relative to native species. Turkey Key (top), Wood Key (center), and Plover Key (bottom) are shown. Photos in each column were taken on the same date (Photos by Tony Pernas, NPS).

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Latherleaf is actively managed in the ENP and Biscayne National Park, although there are increased concerns about this species in the Southern Estuaries and its movement into the natural reserves of north Key Largo. Due to difficulties in early detection of this intertwined scandent shrub, resource managers are unable to accurately estimate the distribution of latherleaf in the region, complicating systematic control operations.

Land Managers have long speculated that the success of latherleaf in South Florida is the result of latherleaf's having high seed germination success, a long lived seed bank, and possible allelopathy. The NPS contracted with the UF to study the seed ecology and allelopathy of latherleaf. The study (McCormick and Langeland, 2007) concluded that latherleaf seeds have very low germination success and that seed viability is typically less than one year. However, field observations during the study showed strong evidence that latherleaf is more resilient than native species following severe hurricanes. It is the first species to flush with growth following storm events and is then able to thrive due to removal of canopy and the influx of light, water, and nutrients (**Figure 9-18**). Latherleaf in its native range is well adapted to regular cyclonic activity. In South Florida, latherleaf seeds are moved by ocean currents; flooding events such as storm surge move seeds inland.

The NPS, SFWMD and Miami-Dade County have been working together on invasive plant control through the South Biscayne Bay Exotic Plant Working Group. A primary focus of the group is Australian pine (*Casuarina* spp.), represented by three unique species in South Florida. In Biscayne National Park, this species is considered to be under maintenance control. This year, the District began initial treatment of approximately 80 acres within the Biscayne Bay Coastal Wetlands (BBCW) project area bordering the entrance to Biscayne National Park (**Figure 9-19**). While primarily targeting Australian pine and Brazilian pepper, crews also discovered several pockets of *Lygodium*, which they quickly treated. As acquisitions of adjacent properties for the Acceler8 BBCW project continue, these areas will undergo the incremental process of controlling invasive species and the seed banks they generate. However, there is a constant floating seed source from surrounding areas of the coastal mainland and islands to the south, making long-term control impossible without a continuous, active treatment program.

Figure 9-19. Treated *Casuarina* spp. (Photo by Jason Smith, SFWMD).

Biological control research is actively being pursued for Australian pine. However, the program may face limitations resulting from conflicts with agricultural interests. Australian pine is frequently planted as an ornamental or for wind protection around citrus groves. This conflict of interest between those planting Australian pine and those trying to control it has led researchers to target seed-feeding agents that leave the adult plants intact while preventing them from reproducing (G. Wheeler, USDA-ARS, personal communication). This program is in the early stages; the majority of work currently entails field explorations for potential seed-feeding biocontrol candidates in the plant's native range. Only one species of *Casuarina* in Florida reproduces solely by seed, so seed-feeding insects are not projected to have a large impact on the remaining two species.

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Figure 9-20. Crocodile nest on *Casuarina*-impacted island in northeastern Florida Bay (Photo by Tony Pernas, NPS).

Australian pine is of special concern in the Southern Estuaries because it threatens the habitat of the endangered crocodile (*Crocodylus acutus*) and nesting sea turtles. Australian pine's shallow root system has been observed to interfere with both sea turtle nests on beaches and crocodile nests in northeastern Florida Bay (**Figure 9-20**).

Other problematic species in the southern coastal estuaries include half-flower (*Scaevola taccada*) and seaside mahoe (*Thespesia populnea*). Like Australian pine, the seeds of these species float, and there is constant seed pressure from surrounding natural areas and ornamental plantings in coastal urban communities, making perpetual control necessary. The sapodilla tree (*Manilkara zapota*) is interspersed with tropical hardwood communities throughout some coastal islands, making on-the-ground control tedious as herbicide applicators are forced to canvass the forested area on foot looking for the nonindigenous tree among native tree species (**Figure 9-21**).

Figure 9-21. Sapodilla (*Manilkara zapota)* fruit, and interspersed along the southern coastline (Photos by Ann Murray, Univ. Florida and Tony Pernas, NPS).

The priority plant species for the Florida Bay and Southern Estuaries Module are listed in **Table 9-5**.

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Table 9-5. Stoplight table for priority plant species in the Southern Estuaries Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
FLORIDA BAY & SOUTHERN ESTUARIES MODULE (Results in row reflect module-level questions, not species-level questions)	Y	Control programs under way for many years, achieve significant control; however, many species invaded in recent years and their possible effects unclear; most Florida Bay not included in any monitoring program for invasive plants	Y	Some species, e.g. Latherleaf, have been serious invaders of rare habitats along the southern coast of the Park; other new species simply off the radar as far as inclusion in a systematic control or monitoring program and are serious unknowns	Y R
Australian Pine (*Casuarina* spp.)	Y	Effective control program in place in southern and western coastal areas of Park; surrounding seed sources make continuous long-term management necessary in these areas; impacts endangered species	Y	Chemical control effective and most coastal habitats clear but ongoing control still needed in coastal areas due to (floating) seed pressure from other areas; biocontrol research under way	Y G
Latherleaf (*Colubrina asiatica*)	Y R	Spread of latherleaf documented for over a decade; overall, distribution and impacts in coastal habitats increasing; difficult to detect remotely,; especially problematic to rare coastal habitats; not part of systematic monitoring program	Y R	Spreading north along Park's west coast, east along Florida Bay, and south into Keys; poses serious threat to natural areas of north Key Largo; herbicidal control logistically challenging; seed viability poorly understood; no biocontrol programs under way	R
Old World Climbing Fern (*Lygodium microphyllum*)		Small population discovered in module in 2006; treatments made, although likelymore widespread than previously thought	Y	Careful monitoring key to successful control; populations currently small, but can spread extremely rapidly in undistrurbed and remote areas; biocontrol releases made	Y
Sapodilla (*Manilkara zapota*)	Y	Scattered throughout coastal hardwood habitats; difficult to detect remotely; not included in Indicator systematic monitoring program	Y	Because intermixed in native tropical hardwood communities, detection and control difficult and logistically challenging; likely spread by animals; no biocontrol program under way	Y R
Half Flower (*Scaevola taccada*)	Y	Limited to coastal habitats; easy to detect but not part of Indicator systematic monitoring program	Y	Effectively controlled along beaches in most locations, but surrounding seed sources from ornamental plantings make long-term control problematic; no biocontrol program under way; Prospects poor, given native *Scaevola* species	Y R
Brazilian Pepper (*Schinus terebinthifolius*)	Y	Invades most habitats, including coastal communities, and very destructive; chemical control ineffective in reducing ecosystemwide spread so far; however, localized control programs are proving effective	Y	Control programs in southern Park areas effective in reducing local populations; most populations limited so far in this region but coastal mangroves still threatened; new biocontrol agents under study, releases 2007/2008	Y
Seaside Mahoe (*Thespesia populnea*)	Y	Invades coastal habitats and forms dense monocultures; not part of systematic monitoring program	Y	Control ongoing in Elliot Key and scattered locales in Florida Bay; surrounding seed sources from wild populations and ornamental plantings; floating seeds spread into natural areas with high tide, make long-term control difficult	Y

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Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources, the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain green status.

Nonindigenous Animals

In addition to well documented problems associated with nonindigenous coastal plant species (**Table 9-5**), the Florida Bay and Southern Estuaries Module also has several priority nonindigenous animals, highlighted in this chapter because recent evidence indicates that populations are expanding and may be impacting ecologically sensitive areas in this region.

Mexican Red-bellied Squirrel

The Mexican red-bellied squirrel (*Sciurus aureogaster*, **Figure 9-22**) is native to southern Mexico (reviewed in Koprowski et al. in review). Two pairs of squirrels were purposefully introduced from eastern Mexico to Elliott Key in 1938. They quickly established a breeding population on the island and were widespread by the 1960s. The species has also been reported on two adjacent islands, Adams Key and Sand Key.

Firgure 9-22. Mexican red-bellied squirrel (*S. qureogaster*) (Photo by NPS).

Hurricane Andrew (1992) resulted in losses of island forests (Ogden, 1992; Davis et al., 1994). Many mammal species survived the storm on mainland Miami-Dade County (Ogden, 1992; Davis et al., 1994), but the island populations of red-bellied squirrels were thought to have been extirpated on Elliott, Adams, and Sand keys (Koprowski et al.,, in review). Recent sightings and conspicuous nests in large trees on Elliott Key suggest that this species survived the hurricane and is increasing in number (T. Pernas, NPS, personal communication).

The Mexican red-bellied squirrel breeds year-round. They are opportunistic feeders (J. Koprowski, University of Arizona, personal communication) with a diet that includes the fruits of many native species including sea grape (*Coccoloba uvifera*), mastic (*Mastichodendron foetidissimum*), gumbo limbo (*Bursera simaruba*), Keys thatch palm (*Thrinax morrissii*), Florida thatch palm (*Thrinax radiata*), and most notably, the endangered Sargent's buccaneer palm (*Pseudophoenix sargentii*). They also feed on eggs and invertebrates, and pre-Andrew NPS assessments of the squirrel on Elliott Key suggested that they feed on the declining liguus tree snail (*Liguus fasciatus)* (Tilmant, 1980).

The potential and actual impacts of this exotic species on Florida Bay and the Southern Estuaries are poorly understood, although introduced populations of other squirrels in Europe and the western U.S. are known to have detrimental impacts (Steele and Koprowski, 2001). An NPS ranger intercepted a swimming squirrel near Old Rhodes Key (Layne, 1997), suggesting that this species could spread throughout the Southern Estuaries and into the Florida Keys, where endangered rodent species (e.g. the Key Largo woodrat, *Neotoma floridana smalli,* and the Key Largo cotton mouse, *Peromyscus gossypinus allapaticola*) would be vulnerable to competition.

This invasive potential of the Mexican red-bellied squirrel, coupled with the conspicuous number of individuals and increased abundance of nests on Elliott Key, suggests that this species warrants further investigation. In response to this threat, the NPS has begun development of a Rapid Assessment of the Mexican Red-bellied squirrel at Biscayne National Park with the University of Arizona. This work will use nest surveys, live trapping, and radio telemetry to document the status of this nonindigenous squirrel on Elliott, Sand, and Adams, and Old Rhodes Keys. Population surveys of Elliott Key conducted from 2005 through 2007 identified over 200 squirrel nests (**Figure 9-23**). Of concern was the observation of this squirrel on Old Rhodes Key, just a few hundred yards from Key Largo, indicating that the squirrel has managed to cross water barriers and it is plausible that the species can reach Key Largo in the future.

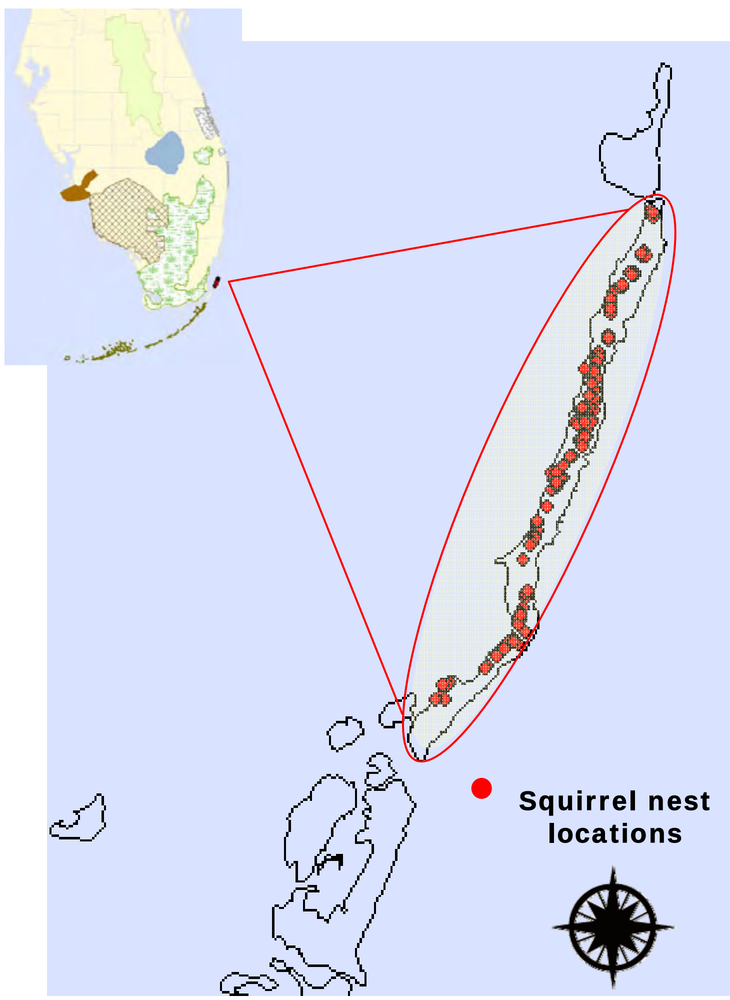

Figure 9-23. Mexican red-bellied squirrel (*Sciurus aureogaster*) population surveys by the NPS and Univ. of Arizona, 2005–2007 (data from NPS and Univ. Arizona).

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Mayan Cichlid

The Florida population of the Mayan cichlid (*Cichlasoma urophthalmus*) was first recorded in 1983 in Snook Creek, a tributary of Joe Bay in northeastern Florida Bay (Loftus, 1987). Although the source of this introduction is unknown, scientists suspect one or more accidental or purposeful aquarium releases (Loftus and Kushlan, 1987). The Mayan cichlid is native to the Atlantic slope waters of southeastern Mexico and Central America. It thrives under a wide range of environmental conditions, exhibiting a tolerance to brackish and marine conditions (**Figure 9-24**). Since its discovery in Florida Bay in the early 1980s, this species has expanded its range; it is common throughout the District canal system, freshwater wetlands, and estuarine mangrove swamps of the Southern Estuaries. The Mayan cichlid is an established, introduced species (Loftus, 1987), which is unlikely to be eradicated.

Figure 9-24. Mayan cichlid (*Cichlasoma urophthalmus*) (Photo by Paul Shafland, FWC).

The Mayan cichlid has a varied diet, preying on small fishes and aquatic invertebrates. Given its broad salinity tolerance and aggressive nature, it is likely to continue to impact the Florida Bay and the Southern Estuaries, expanding its range in southern Florida (Loftus, 1987). Analysis of recent data from mangrove areas along northern Florida Bay showed that densities of native species varied inversely with densities of Mayan cichlids (Trexler et al., 2000). Potential impacts of this species could include altering native fish community structure through direct interaction, breeding ground competition, and the predation of juveniles (Shafland, 1996).

Cuban Treefrog

The Cuban treefrog (*Osteopilus septentrionalis*) is native to Cuba, the Cayman Islands, and the Bahamas. It was introduced to the Keys as early as 1920, likely as a ship stowaway. It has since invaded the mainland, its invaded range now stretching to the panhandle and the Georgia coast. This species infests a wide range of habitats including pine forests, hardwood hammocks, swamps, homes and buildings, and gardens. Cuban treefrogs vary in color from yellow to green to dark brown, but are frequently a dull or light brown. They are discernable from native treefrogs by distinctive warts, larger eyes and larger size. This species feeds upon snails, millipedes, spiders, a vast array of insects, lizards, and native frogs (**Figure 9-25**). Their propensity to compete with and/or prey upon native frogs has resulted in Cuban treefrogs becoming the most common frog species in Florida. In addition, Cuban treefrogs are a nuisance to plumbing infrastructure and yard aesthetics and can cause power outages due to short circuits. They also exude a sticky secretion that is irritating to the mucous membranes of people. This species spreads rapidly both with prolific reproduction and with frequent hitchhiking on automobiles, boats, and landscape plants. The University of Florida maintains a database logging the spread of this species, although its impacts are still not fully understood.

Figure 9-25. Cuban treefrog feeding on native green treefrog (Photo by Brent Anderson, Univ. Florida IFAS Extension UW259).

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GREATER EVERGLADES MODULE

The Greater Everglades Module is made up of a mosaic of historically interconnected wetlands. It includes the Water Conservation Areas, Everglades National Park, the Loxahatchee National Wildlife Refuge and the J.W. Corbett/Pal Mar Wildlife Management Area.

Nonindigenous Plants

Melaleuca

Before organized state and federal nonindigenous plant control operations were initiated in 1990, melaleuca (*Melaleuca quinquenervia*) was widely distributed throughout the Water Conservation Areas (WCAs), the ENP and Arthur R. Marshall Loxahatchee National Wildlife Refuge (Refuge).

Figure 9-26. Controlling melaleuca (Photo by Albert Mayfield, FDACS).

Overall, agency efforts to control melaleuca are succeeding in containing and reducing its spread in the Greater Everglades. Melaleuca has been systematically cleared from WCA-2A, 3A, and 3B; these areas are now under maintenance control. Melaleuca populations in the northeasternmost area of the ENP are also decreasing. Operational work is currently focused on methodically treating the remaining 7,000 gross infested acres (**Figure 9-26**). Unfortunately, melaleuca populations in northernmost sections of the Greater Everglades Module are increasing, and control operations do not appear to have been systematic in approach. Areas of the Refuge and Corbett Wildlife Management Area that had light to medium levels of melaleuca in the early 1990s are now dominated by large, dense stands. With technical and fiscal support from the District and Florida DEP, the Refuge has recently seen results from its efforts to control melaleuca. Many acres of infested lands in the southern Refuge have been treated (**Figure 9-27**), and efforts to control northern Refuge infestations are underway (G. Martin, USFWS, personal communication). See the *Big Cypress Module* (page 9-54) for information on the biological control program of melaleuca.

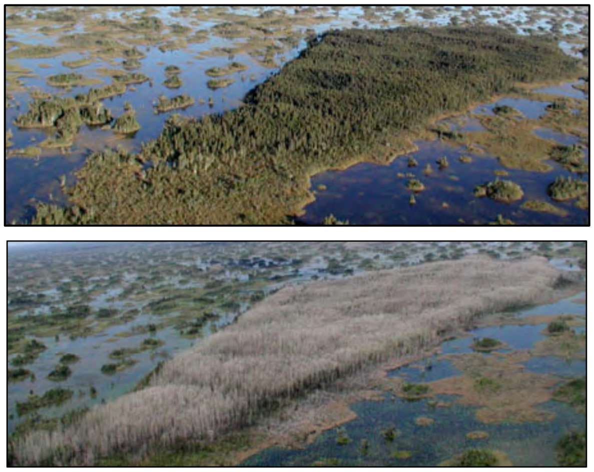

Figure 9-27. Pre (above) and post views following aerial melaleuca treatments in the Refuge (Photos by SFWMD).

Old World Climbing Fern

Perhaps no other individual plant species poses a greater threat to the Everglades than Old World climbing fern (*Lygodium* spp.). As depicted in **Figure 9-28**, this highly invasive vining fern smothers native vegetation, severely compromising plant species composition, destroying tree island canopy cover, and dominating understory communities, which are all cited as key parameters in measuring Everglades restoration success. When surveys for the species began in the early 1990s, Old World climbing fern occurred on limited tree islands

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in the northern quarter of the Refuge (Ferriter and Pernas, 2006). Today, it dominates Refuge tree islands, and now occurs, at various levels of density, in virtually every habitat in the Greater Everglades Module (Ferriter, 2001).

Figure 9-28. Old World climbing fern (Photo by Peggy Greb, USDA-ARS).

ENP staff first discovered hundreds of acres of Old World climbing fern on the Park's western edge in 1999; by 2000 it had spread to thousands of acres (T. Pernas, NPS, personal communication), and District field biologists observed small strands in WCA-3 beginning in 2001 (M. Korvela, SFWMD, personal communication). This species could potentially overtake most of the southern peninsula of Florida (Lott et al., 2003; Volin et al., 2004). Based on the documented impacts of this species in the Refuge (Brandt and Black, 2001) and the Park, the District initiated a detailed ground-based tree island survey to estimate the extent to which Old World climbing fern occurs in the WCAs. The District has conducted biannual SRF surveys documenting the rapid spread of this species since 1993 and is conducting ongoing operational and field research to effectively control the species and determine environmental factors that affect its growth and spread. (Stocker et al., 1997; Gann et al., 1999; Ferriter, 2001; Langeland and Link, 2006).

Due to the remoteness of the Old World climbing fern populations in the Park, Park staff is limited to using helicopters to conduct aerial treatments, evaluate non-target damage, and assess the effectiveness of these treatments. District contract crews treat this species as they encounter it on tree islands throughout the Everglades. Over the last year, District and FWC contractors have conducted intensive ground-based tree island surveys in the WCAs to locate remote, incipient Old World climbing fern populations. Based on preliminary results from a random survey of 80 tree islands, roughly 9 percent of the tree islands surveyed had at least one Old World climbing fern infestation. The occurrence of infestations did not correlate with site conditions such as island size, island elevation, or species richness, suggesting that most islands are susceptible to invasion by this plant. The District is entering into an operational phase of tree island surveys, which increases survey frequency and improves coordination between surveyors and vegetation management contractors. Once field biologists discover populations, the coordinates and infestation characteristics are transferred to the District's Vegetation Management Division, which then dispatches control contractors.

Figure 9-29. Pre (above) and post views showing effects of aerial herbicide treatment on *Lygodium* infested tree islands in the Refuge (Photo by L. Rodgers, SFWMD).

The USFWS has had resource management responsibilities for WCA-1 since 1951 when it was designated as the Arthur R. Marshall Loxahatchee National Wildlife Refuge. Since 2002, the Refuge has worked to implement an integrated plan for the control of its worst invasive plants — Old World climbing fern, melaleuca, Brazilian pepper,

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and Australian pine. The Refuge plan utilized a containment strategy — working from the less-infested areas in the southern Refuge towards the dense infestations in the north. By 2005, it was clear that the Refuge had insufficient resources to implement the containment strategy quickly enough to curb the explosive spread of Old World climbing fern that was infesting nearly all tree islands in the northern half.

Recognizing that the biological integrity of the Refuge was quickly being lost to these aggressive weeds, the Refuge, the District and FDEP came together in early 2007 to implement an accelerated invasive plant management plan. Utilizing financial resources from the FDEP, and technical and logistical resources from the District and Refuge staff, this collaborative effort aims to complete a first-pass treatment of all invasive plant infestations within the Refuge by September 2008. Invasive plant managers from each of the three agencies drafted the plan in early March 2007, and began implementing the work later that month. Dense infestations of invasive plants were treated aerially (primarily melaleuca and Old World climbing fern, **Figures 9-27** and **9-29**), followed by the deployment of ground crews to treat small or sparse infestations where non-target damage from aerial treatments would be unacceptable. All aerial treatments of dense infestations were completed on May 20, 2007, with an estimated 11,800 combined acres of melaleuca and Old World climbing fern treated. Ground-based efforts were initiated in mid-March 2007, but low water levels prevented access to the Refuge interior by early April. Surface water levels returned to navigable levels in mid-July. Roughly 7,120 acres of the Refuge have been canvassed by ground applicators.

The success of the plan is dependent upon future resource allocations to follow-up treatments. Given the scale of the problem in the Refuge, invasive-plant managers estimate that an annual allocation of $3 million for the next five years will be necessary to bring the worst weeds within the Refuge under reasonable levels of control.

In addition to the efforts outlined above, several ongoing research initiatives are underway at the Refuge. These include (1) determining the effects of fire as a post-treatment strategy on tree islands, (2) assessing post-fire recruitment of Old World climbing fern, and (3) monitoring the effects of repeated aerial herbicide applications on *Lygodium microphyllum* and native vegetation (B. Miller, USFWS, personal communication).

Figure 9-30. *A. camptozonale* (Photo by Christine Bennett, USDA-ARS).

Land managers statewide agree that biocontrol may be the key to effective long-term regional management of Old World climbing fern. There are only two agents currently permitted for release: the pyralid moth, *Austromusotima camptozonale*, (**Figure 9-30**) and the leaf-gall mite (*Floracarus perrepae*). During 2005, 12,000 adult *A. camptozonale* moths were released in South Florida, but these failed to establish. In 2006, on the supposition that *A. camptozonale* caterpillars would be a more resilient life stage for transportation and release, a total of 16,000 caterpillars were released at the same sites (see **Figure 9-5**). Early monitoring indicated that this release method held promise, as the caterpillars had survived and reproduced at half of the release sites (R. Pemberton, USDA-ARS, personal communication). Approval of the federal release permit for *F. perrepae* was issued in 2007, with initial releases planned for early 2008.

A third agent, another species of pyralid moth (*Neomusotima conspurcatalis*), was approved for release by the Technical Advisory Group for Biological Control of Weeds, and researchers are

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awaiting issuance of a federal release permit from USDA-APHIS-Plant Protection Quarantine (APHIS-PPQ) (R. Pemberton, USDA-ARS, personal communication). In addition to the agents mentioned above, numerous other insects are being studied both in the field abroad and in the laboratory for their biology and host specificity. These include the sawfly, *Neostrombocerus albicomus*, the noctuid moth, *Callopistria* spp., the pyralid moth, *Lygomusotima stria*, the flea beetle, *Manobia* spp., and the stem-boring moths, *Siamusotima aranea*, *Ambia* spp. “S”, and *Ambia* spp. “H”.

Brazilian pepper

Brazilian pepper (*Schinus terebinthifolius)* is common on levees and tree islands throughout the Greater Everglades. Unlike melaleuca, operational control for this species is not systematic in approach, with the exception of the ENP’s “Hole in the Donut” (HID) Project, where impenetrable monocultures of Brazilian pepper are controlled through the complete removal of previously farmed and rock-plowed substrate. This intensive process results in recolonization by native wetland vegetation to the exclusion of Brazilian pepper. In contrast, vast areas of the western coastal mangroves and marshes of the Park are being dominated by Brazilian pepper, and resource managers face almost insurmountable obstacles in treating these populations due to the breadth and remoteness of the sites. This underscores the need for effective biological controls for this species.

ENP staff observed large areas of dead or dying Brazilian pepper along the western edge of the Park after Hurricanes Katrina/Wilma in late 2005 (**Figure 9-31**). Although it was thought that this Brazilian pepper mortality might have resulted from increased salinity caused by storm surge, soil samples taken in the area revealed no significant differences in salinity levels in areas where the Brazilian pepper had died (T. Pernas, NPS, personal communication). The Park staff continues to monitor this area.

Figure 9-31. Dead Brazilian pepper along western edge of the ENP following 2005 hurricanes (Photo by Tony Pernas, NPS).

There are two haplotypes of Brazilian pepper found in Florida, with extensive hybridization having occurred between the two (Williams et al., 2005). This further complicates the task of identifying suitable biocontrol agents because those agents (with suitable host specificity) that attack one haplotype are unlikely to attack the other, nor the hybrids of the two (G. Wheeler, USDA-ARS, personal communication). Extensive field explorations conducted in Argentina and Brazil have resulted in the identification of multiple potential agents. Two species (*Pseudophilothrips ichini* and *Heteroperreyia hubrichi)* have undergone extensive testing. In May 2007, TAG recommended the release of the thrips *P. ichini*. The University of Florida will prepare the Environmental and Biological Assessments in June 2007. Additional promising insects (some naturally occurring in Florida) are currently being tested for host specificity and effectiveness. Expanded field explorations are also planned for Brazil in the near future, pending the acquisition of collecting permits.

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Figure 9-32. Australian pine (Photo by Amy Ferriter, Boise State University).

Australian pine

Australian pine (*Casuarina* spp.) grows quickly; is salt tolerant; fixes nitrogen; readily colonizes rocky coasts, dunes, sandbars, islands; and invades far-inland, moist habitats (Morton, 1980) (**Figure 9-32**). It forms dense forests, eventually excluding other plant species. Efforts to control Australian pine in the Greater Everglades are ongoing, but are not yet systematic in approach. This species is still common along District levee berms, in the District's southern saline glades (C-111 basin), and Biscayne National Park. In the northeastern portion of the ENP there no longer exist large, dense stands of Australian pine. Treatment efforts are focused on removing the remaining scattered stands with most areas now at maintenance levels. The coastal mainland and coastal islands are routinely colonized by Australian pine — but are also under maintenance control. The largest remaining populations found in the ENP exist in the saline glades in the southern region. Systematic treatment efforts have not yet been conducted in this area. The seeds are windblown, carried by birds, and probably drift throughout the Everglades via canals.

Australian pine threatens key habitat for the endangered Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis)*, which needs the short-hydroperiod marl prairies of the southeastern Everglades to nest. To restore sparrow nesting habitat invaded by Australian pine, the ENP and USACE began a ground-based, systematic program along the eastern edge of the Park that is still ongoing. Australian pine in this region is currently at maintenance levels.

Shoebutton ardisia

Figure 9-33. Shoebutton ardisia *(Ardisia elliptica)* (Photo by Amy Ferriter, Boise State Univ.).

Shoebutton ardisia (*Ardisia elliptica*) is a shade-loving shrub that was originally reported in the HID (**Figure 9-33**). It spread into adjacent tropical hardwood hammocks in the Long Pine Key area of the Park (Seavey and Seavey, 1994) and was observed in the Flamingo Bay area in 1995 (Doren and Jones, 1997). Large monotypic stands of this species now occur on District lands adjacent to the Park. Sporadic District and NPS control operations are ongoing for this species, but recent field observations by District contract crews (M. Blankenship, Applied Aquatics, personal communication) indicate that this plant is invading the understory of many tree islands and bayheads in WCA-3. If this species continues to spread in the WCAs, it will threaten the integrity of tree island plant communities. Shoebutton ardisia prefers wetlands and in other areas of the Greater Everglades, it forms dense, monotypic stands that completely exclude understory vegetation. Early detection on tree islands and bayheads will be extremely challenging, as this species is difficult to detect remotely, and a closely related native, marlberry (*Ardisia escallonioides*), has a very similar form. While birds are the principal dispersers of the seed, raccoons and opossums also eat the fruit and disperse seeds (Miami-Dade County, 2002). The priority plant species for the Greater Everglades Module are listed in **Table 9-6**.

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Table 9-6. Stoplight table for priority plant species in the Greater Everglades Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
GREATER EVERGLADES MODULE (Results in this row reflect module-level questions, not species-level questions)	Y	Old World climbing fern and Brazilian pepper still widespread, serious threats; continued rapid spread of these two species with little results from control efforts; still several other species present with little or no control effort or efficacy	Y	Good control of melaleuca and Australian pine; biocontrol for melaleuca effective; first biocontrol releases for Old World climbing fern, new biocontrol for Brazilian pepper soon; other species still localized, no new serious invaders detected	Y
Shoebutton Ardisia (*Ardisia elliptica*)	Y R	Was localized problem in Park but now infests tree islands and bayheads throughout WCAs; difficult to detect and not part of systematic monitoring program	Y R	No significant control program, no biocontrol effort; now found in WCA tree islands and bay-heads, posing a serious threat; difficult to monitor remotely; resembles native species, detection and control difficult	R
Australian Pine (*Casuarina* spp.)	Y	Still common in northeast portions of Park, on District canal banks and throughout South Dade Wetlands	Y	Chemical control effective; most natural areas clear with exception of northeast part of Park and South Dade Wetlands where significant control still needed; biocontrol research under way	Y
Old World Climbing Fern (*Lygodium microphyllum*)	R	Serious invader, rapidly spreading; invades most habitats and very destructive; long-term management difficult given variety of habitats it infests	R	No effective control yet, but biocontrol release made with additional release expected in 2007; chemical control studies continuing	R Y
Melaleuca (*Melaleuca quinquenervia*)	Y	Large portions of module under maintenance control and biocontrols showing promising results; however, some areas in east Everglades, Refuge, and Corbett WMA still need significant work	Y	Chemical control effective on most public lands; biocontrol agents effective and additional spread of existing agents and new agents expected in 2007 and 2008	Y G
Brazilian Pepper (*Schinus terebinthifolius*)	R	Rapidly spreading; invades most habitats, very destructive; chemical control ineffective in reducing overall spread; the Park (particularly mangroves) seriously impacted; no coordinated control program	R	No effective regionwide controls yet; chemical control programs effective in limited areas where significant resources can be applied; new biocontrol agents under study for possible release in 2007–2008	R

Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources, the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain green status.

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Nonindigenous Animals

In addition to the priority plant species listed in **Table 9-7**, many nonindigenous animal species occur in the Greater Everglades Module. The priority animal species discussed below have raised special concerns among agency scientists in the region and have the potential to impact Everglades restoration initiatives.

Lobate Lac Scale

The lobate lac scale insect (*Paratachardina lobata)* native to India and Sri Lanka and was first discovered in 1999, on ornamental hibiscus (*Hibiscus rosa-sinensis*) in Davie, Florida The scale began spreading at an alarming rate, with new populations reported with increasing frequency throughout urban and natural areas. Host species include many different ornamental shrubs and trees, including fruit trees, and it is known to occur on over 40 native plant species. Some plant families, notably *Fabaceae* (peas and beans), *Myrtaceae* (myrtles), and *Moraceae* (mulberry) seem to have many species that are especially susceptible to the scale. Field observations in the Greater Everglades indicate that the insect occurs on many native plants, and certain native species appear to be highly susceptible, such as the wax myrtle (*Myrica cerifera*), cocoplum (*Chrysobalanus icaco*), buttonwood (*Conocarpus erectus*), strangler fig (*Ficus aurea*), myrsine (*Myrsine guianensis*), red bay (*Persea borbonia*), and wild coffee (*Psychotria nervosa*) (**Figure 9-34**).

Figure 9-34. Lobate lac scale (*Paratachardina* spp.*)* (Photo by F.W. Howard, Univ. Florida).

This insect is already seriously affecting native tree islands; aerial surveys indicate that large specimens and populations of wax myrtle and cocoplum have been killed by this insect in areas within the Everglades. Recent observations indicate this species is decreasing across South Florida (P. Pratt, USDA-ARS, personal communication). However, the importance of healthy tree islands in Everglades restoration, the value of canopy cover for wading bird nesting, and the propensity of some exotic plants to rapidly colonize disturbed sites (such as areas of canopy dieback), all warrant research to understand the distribution of this invasive species and steps to contain its spread.

No available insecticides are labeled for use in wetland areas, and selective control of this species with pesticides would be difficult, if not impossible. In addition, using pesticides in sensitive natural areas may have secondary effects, especially on native insect populations. Consequently, biological control agents are seen as the only option for controlling this species.

The USDA-ARS and the University of Florida have carried out extensive overseas searches for natural enemies of lobate lac scale. After several years of searching its native range, the USDA-ARS found populations of the scale in southern India in August 2005 (R. Pemberton, USDA-ARS, personal communication). Multiple Indian specimens (*Paratachardina lobata*) were shipped to the quarantine facility in Davie in order to develop biological control agents. Though parasitoids reared from the Indian material readily attacked Florida lobate lac scales, they failed to reproduce. Taxonomic analyses were recently conducted to determine the cause of this problem. Results demonstrate that the invasive scale in South Florida is not *P. lobata* and is a new species — also invasive in the Bahamas and Christmas Island. The USDA-ARS is currently

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determining the origin of this new species (which appears to be Indonesia), and will begin developing biocontrol agents soon. Despite this progress, it will be many years before a safe, effective biological control for lobate lac scale is available in Florida. (R. Pemberton, USDA, personal communication).

Figure 9-35. Tropical almond leaf fed upon by *M. undatus* (Photos by Jeffry Lotz and Susan Halbert).

Sri Lanka Mimic Weevil

Weevils collected from numerous east coast South Florida locations extending from Homestead to Boca Raton were recently identified as Sri Lanka mimic weevil (*Myllocerus undatus*), a native of Sri Lanka and new to the Western Hemisphere (**Figure 9-35**). This weevil has an extremely broad host range; thus far it has been shown to attack 68 different plant species occurring in Florida (M. Thomas, FLDACS-DPI). This fact makes *M. undatus* a particularly frightening invader in South Florida. Unfortunately, very little is known about this weevil in its native range, and so control efforts are likely to prove difficult. A list of species known to be impacted by this insect can be found at www.doacs.state.fl.us/pi/enpp/ento/weevil-pest-alert.html.

Burmese Python

The Burmese python (*Python molurus bivittatus*), a native to Southeast Asia, can reach a length greater than 20 feet. This long-lived (15–25 years) python is a behavioral, habitat, and dietary generalist, capable of producing large clutches of eggs (8–107). The python's diet in the Everglades includes alligator, raccoon, rabbit, muskrat, squirrel, opossum, cotton rat, black rat, cat, house wren, pied-billed grebe, white ibis, and limpkin. As the Burmese python is known to eat birds and is known to frequent wading bird colonies in their native range, the proximity of python sightings to the Paurotis Pond and Tamiami West wood stork rookeries is troubling**.**

Observations of pythons exist primarily from three locations in the ENP: (1) along the Main Park Road in the saline and freshwater glades and mangroves between Pay-hay-okee and Flamingo, (2) in the greater Long Pine Key area (including Hole-in-the-Donut), and (3) in the greater Shark Valley area along the Tamiami Trail (including L-67 Ext.). The pythons have also been repeatedly observed on the eastern Park boundary, along canal levees, in the remote mangrove backcountry, and in Big Cypress National Preserve. In recent years (2003–2007), individuals of all size classes have been seen with increasing regularity in and around the ENP. The measured total length for snakes recovered ranges from 2 to 14 feet, including five hatchling-sized animals recovered in the summer 2004 and two hatchlings in 2005. Clutches of eggs (both fertilized and already hatched) have been discovered since 2006.

The non-native Burmese python populations are continuing to expand at an alarming rate in the Greater Everglades, as documented in previous SFERs (**Figure 9-36**). In 2006, approximately 170 pythons were removed from the ENP and surrounding areas, representing a twofold increase from 2005. As of October 2007, 201 pythons were removed.

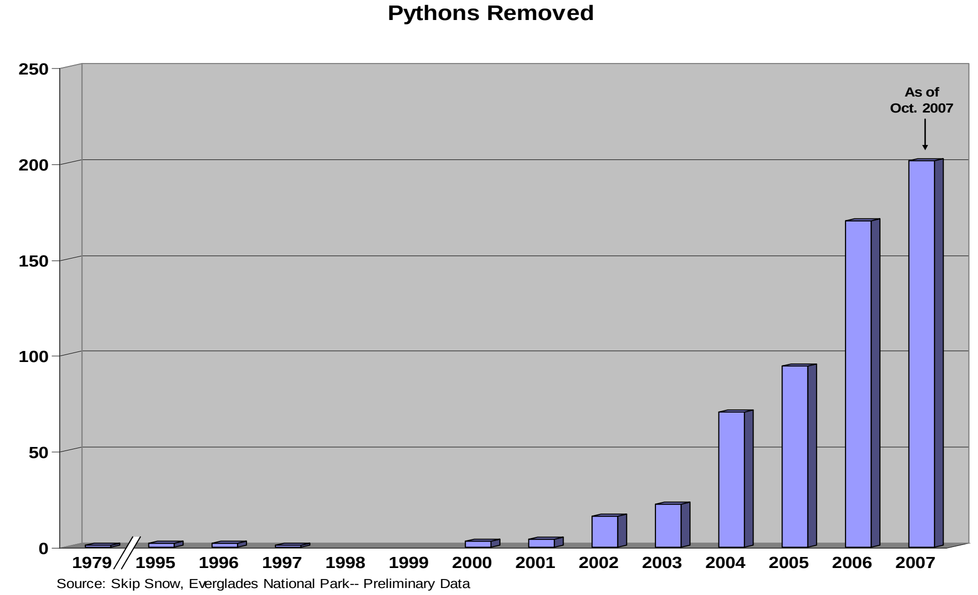

Figure 9-36. Number of Burmese pythons (*Python molurus bivittatus*) removed from the Greater Everglades region between 1979 to October 2007 (unpublished data courtesy of Skip Snow, NPS).

Burmese pythons present a potentially significant threat to the successful ecological restoration of the Greater Everglades. Established and breeding in South Florida, the populations have the clear potential to occupy the entire footprint of CERP, adversely impacting valued resources across the landscape. Observations of Burmese python/ American alligator conflicts are troubling, particularly because the alligator is widely considered a top predator in the Greater Everglades region (**Figure 9-37**).

The pathway of invasion for the Burmese python is through the pet industry; pythons are still commonly sold in pet stores. Roughly 6,000 Burmese pythons were imported through the Port of Miami between 2003 and 2005. In an attempt to "cork the bottle," the SFWMD Governing Board petitioned the USFWS to list the Burmese python as an injurious species under the Lacey Act (42 U.S.C. § 18). The USFWS regulates international wildlife trade and addresses threats to native wildlife resources. A 1981 amendment to the Lacey Act allows for the regulation of importation or interstate commerce of animals that have been determined to be injurious to human beings or to wildlife resources of the U.S. To date, no decision on this request is made.

At the state level, however, the 2007 Florida Legislature passed Senate Bill 2766 which increases regulations for the capture, possession, transportation, or exhibition of "reptiles of concern." The revised regulations increase the penalties for releasing pythons, anacondas or other nonnative reptiles into the wild. The bill also authorizes the FWC to require annual registration fees for owners of listed reptiles, thereby limiting "impulse buys" that often lead to unlawful

Figure 9-37. Alligator consuming Burmese python in the ENP (Photo by Lori Oberhofer, ENP).

releases when large snakes become difficult to care for. In addition, the measure also increases the $1,000 bond required to 'exhibit' reptiles or certain wildlife to $10,000.

A multi-agency workshop convened in March 2007 to discuss the current Florida distribution, reproductive biology, and ecological impacts of the Burmese python. Scientists and regulators from a number of state and federal agencies (FWC, SFWMD, USFWS, USGS, USNPS) discussed the next steps necessary, on technical and policy fronts, to manage this invasive animal. The USFWS, USGS and NPS agreed to move forward with a risk assessment to determine the possible ecological impacts and potential expansion range of the Burmese python. This assessment and an economic impact analysis are necessary for USFWS to complete their review of the Burmese python under the Lacey Act. The workshop attendees also agreed to work collaboratively toward other priorities including development and implementation of capture technologies, improvements in communication (e.g. python listserve), improvements in funding through cross-cut budget initiatives, and identification of public education programs.

Island Applesnail (previously Channeled Applesnail)

Recent (2005) field observations by the Florida International University and ENP scientists indicate that other species such as the island applesnail (previously channeled applesnail, see the *Kissimmee Basin Module* section, page 9-86 for species-specific information) are present in the Greater Everglades Module. These snails and their egg masses were found in an old borrow canal within the northern boundary of Everglades National Park just east of the entrance to Shark Valley (S. Snow, ENP, personal communication). Surveys for this nonindigenous species continue in neighboring waterways as well as adjacent freshwater marshes, and work is beginning to explore available control strategies (S. Snow, ENP, personal communication).

Green Iguana

The green iguana (*Iguana iguana*) (see the *Florida Keys Module* section, page 9-31, for species-specific information) is a widespread nonindigenous reptile species in Southern Florida. District field observations of large groups of this species have increased dramatically in recent years and many canals and levees in and around the Greater Everglades are now peppered with green iguana burrows. This extensive burrowing presents a maintenance liability to surface water infrastructure important to the Everglades restoration effort. Waterways and water structures with notably high numbers of green iguanas include the C-7, C-11 and C-1 West canals. Iguanas burrow into canal banks, leading to bank instability and bank erosion. District and NPS biologists have completed preliminary surveys of burrow characteristics to evaluate their impact on bank stability (**Figure 9-38**). Burrows measured at the S-13 structure in Broward County tended to extend horizontally into the banks, ranging from 0.3 to 2.4 meters deep and generally from 10 to 20 centimeters in diameter. Recent evaluations demonstrate that moderate densities of green iguanas have definite economic impacts on bank integrity and maintenance costs (Sementelli et al., in review).

Figure 9-38. Green iguana burrows (Photo by SFWMD).

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Sacred Ibis

The sacred ibis (*Threskironis aethiopicus*), a large, long-legged wading bird native to parts of Africa and Iraq, escaped captivity and became a serious pest in parts of Europe, and is considered a major threat to European tern colonies. The physical appearance of the sacred ibis is similar to the native and federally threatened wood stork (*Mycteria Americana)*. Overall, coloration is white with black plumes composing the tail. During flight, scarlet patches are noticeable under the wings near their base and on the sides of the breast. The head and neck are bare, scaly and gray in color. The bill is curved and is similar to native white, glossy and scarlet ibis. This nonnative ibis is much larger than any other native ibis, but slightly smaller than the protected wood stork.

The sacred ibis prefers marshes, moist soil wetlands, flooded agricultural fields, coastal estuaries, and lagoons. It shares communal roosting and nesting areas with native wading and water birds, and has life cycle requirements similar to those of egrets, herons, and wood storks in Florida (Rodgers et al., 1996). The diet consists primarily of mollusks, frogs, and aquatic insects, but this species has been reported to prey upon the eggs and young of other wading birds.

Although not confirmed, it is believed that populations in South Florida came from a breeding population that escaped the Miami Metrozoo following Hurricane Andrew in August 1992. This species appears well-suited to Everglades habitats including the WCAs and surrounding agricultural lands. State and federal agencies view this nonindigenous species as a potential threat to native water bird populations. The sacred ibis could impact native wading and water bird populations due to its opportunistic feeding nature, and the bird may compete with native wading birds for food and nesting space.

Figure 9-39. Adult sacred ibis and chick observed in Loxahatchee NWR (Photo by Garth Herring, FAU).

District biologists observed six to eight individuals nesting in the southern Refuge interior during the 2005 wading bird nesting season. In May 2006, sacred ibis were reported nesting among active wading bird colonies in the Refuge (W. Calvert, USFWS, personal communication, 2006). A rapid-response control measure was initiated by the USFWS Region 4 Invasive Species Strike Team following a 2006 District report of a single nesting pair located in an active wading bird rookery. Both individuals were dispatched. Since treatment, no additional sacred ibis have been observed at this colony.

During the 2007 wading bird breeding season, Florida Atlantic University researcher Garth Herring observed three sacred ibis nests (**Figure 9-39**) in an active wading bird rookery in the Refuge. At least three nesting adults were observed, though biologists were unable to dispatch them due to accessibility issues. Two nests hatched chicks (one and two chicks respectively), and the third had a clutch of three eggs. The three chicks were collected, and the nest with the eggs was destroyed (G. Herring, FAU, personal communication).

An adult sacred ibis was seen foraging near the Solid Waste Authority North County Landfill along the Florida Turnpike (Sarah Barrett, Palm Beach County ERM, personal communication), though it is unknown where the adults at the Refuge were foraging. Preliminary assessment of the three collected chicks suggests that their diet was most likely from waste management facilities, with unidentified meat comprising over 30 percent. To a lesser extent, crayfish and other invertebrates were also eaten. Over 25 percent of the chicks' diet consisted of non-food items: glass, metal pieces, and plastic. Most importantly, all sacred ibis chicks appeared to be in excellent condition.

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Purple Swamphen

The purple swamphen (*Porphyrio porphyrio*) is a rail native to Australia, Europe, Africa, and Asia. This species is very similar in coloration to the native purple gallinule (*Porphyrula martinica)* but is much larger, approximately the size of a domestic chicken. The species has huge feet, pinkish legs and a characteristic bright red bill and red frontal shield that extend onto the crown. They may have escaped from Miami Metrozoo after Hurricane Andrew in 1992 or from avicultural hobbyists (Pranty et al., 2000). Little is known about purple swamphens in Florida; most information comes from overseas research. By nature, purple swamphens are communal. Multiple females share incubation and parental nurturing duties. Often more than one female lays eggs (3–6 each) in one nest. Purple swamphens feed on shoots and reeds, invertebrates, and small mollusks. However, they have also been reported to feed on the eggs and young of waterfowl.

The original South Florida purple swamphen population is believed to have established in Pembroke Pines in 1996 (S. Hardin, FWC, personal communication). This population has been reported on varied bird-watching web sites, including the Broward County Audubon Society. In recent years, purple swamphens have been sighted in WCAs and adjacent to the Greater Everglades Module in STA-1 West (STA-1W), STA-1E, STA-5, and STA 3/4 (**Figure 9-40**). A single bird was reported in Orlando following the active 2005 hurricane season (S. Hardin, FWC, personal communication) but is not believed to have survived. Efforts to locate swamphens in Loxahatchee Refuge and South Florida State Parks have not been successful (E. Donlan, SFWMD and H.T. Smith, personal communications).

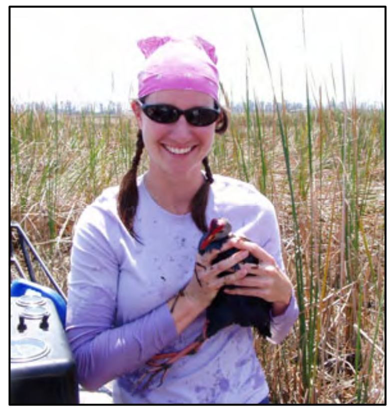

Figure 9-40. Ellen Donlan (SFWMD) and purple swamphen (Photo by SFWMD).

The purple swamphen seems to prefer the edges of manmade ponds, lakes, or impoundments, including storm treatment areas (STAs), and often uses levees and dikes for feeding and travel to, from, and within the STAs. Large concentrations of the purple swamphen could impact native water birds through competition for food and space and through direct predation. The consensus among land management agencies in Florida is that this species could be effectively controlled and possibly eradicated as part of an Early Detection and Rapid Response Program, pending appropriate funding and expeditious implementation of a management and control program. Consequently, a removal program is underway. Most state and federal agencies view this non-native bird species as a potential threat to native water bird populations. Control of purple swamphens in the Refuge is coordinated through the USFWS Region 4 Invasive Species Strike Team; no swamphens have been observed in recent inspections. The FWC has conducted a survey to document the absence/presence of this species on Florida's conservation lands, also producing a combination identification/fact sheet as a component of the initial survey package. Removal efforts have since begun, resulting in the removal of 600 individuals thus far.

Purple swamphens are under consideration for addition to the Migratory Bird Treaty Act (MBTA) since they are native to American Samoa where there is a concern for protecting them. The MBTA does not have a history of making geographic distinctions and subsequently provides protection to a species throughout all of the holdings and interests of the U.S., including trusts, territories, etc. This federal protection has yet to become effective. USFWS staff, aware that this species is not native to North America, is evaluating the need for geographic distinctions in these types of cases. The USFWS currently recommends elimination of as many birds as possible in Florida before any implementation of MBTA protections.

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Swamp Eel

During the late 1990s, three reproducing populations of non-native Asian swamp eel (*Monopterus albus*, **Figure 9-41**) were discovered in Florida: North Miami canals, canal networks near Homestead adjacent to ENP, and in water bodies near Tampa (Fuller et al., 1999; L.G. Nico, USGS, personal communication). It is believed that wild populations in Florida originated as escapes or releases associated with aquaculture, the pet trade, or live food markets.

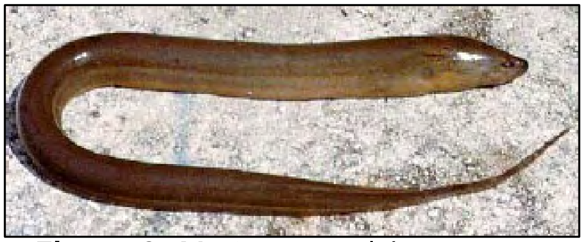

Figure 9-41. Swamp eel (*Monopterus albus)*(Photo Jim Williams USGS).

These fish are now in District canals in Miami-Dade County. Swamp eels have certain characteristics that concern scientists, setting them apart from most other nonindigenous fish species documented in the Greater Everglades Module. The diverse wetland habitats of the Greater Everglades are ideal for the species. Swamp eels are versatile animals, capable of living in extremely shallow water, traveling over land when necessary, and burrowing into mud to survive periods of drought. The eels, which can grow to more than 3 feet in length, are predators that feed on invertebrates, frogs, and other fishes. Although swamp eels are not yet known to have spread from canal systems into the interior of the Everglades, their proximity to restoration efforts is a concern.

Figure 9-42. Electrofishing for swamp eels (Photo John Galvez, USFWS).

Since the discovery of nonnative eels in Florida, USGS scientists have studied aspects of swamp eel biology, including changes in distribution and abundance, diet and reproduction, genetics, environmental tolerances, and ecological effects. Given the abundance and wide distribution of swamp eels in Florida's canals, elimination is probably impossible; however, various control methods are currently under investigation. The USFWS conducted a swamp eel removal project utilizing electrofishing techniques in 2006 (**Figure 9-42**). In addition to the Asian swamp eel, the project also focused on removing exotic spotfinned spiny eels *(Macrognathus siamensis*, aka peacock eels) which occur at bottom depths in slower moving water than Asian swamp eels. The project was conducted on C-111 and C-113 canals and resulted in an average 53 percent efficiency with the removal of 905 Asian swamp eels and 82 peacock eels (J. Galvez, USFWS, personal communication). This project continues during the summer of 2007.

Other Nonindigenous Fish

At least 32 nonindigenous fish species have become established in South Florida through anthropogenic introductions (**Table 9-2**), and many species are now abundant within the canal system that surrounds and dissects the Greater Everglades (USGS, 2004). Nonindigenous fish are often detrimental to their host communities (Ogutu-Ohwayo, 1993; Clavero and García-Berthou, 2005) and have the potential to significantly impact aquatic communities of the Everglades. This concern led CERP to set nonindigenous fish population levels in the EPA as an ecological performance measure (RECOVER, 2003).

Most nonindigenous fish in South Florida are tropical in origin, and their populations are believed to be regulated by annual minimum temperatures, which restrict their range to tropically

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warm or deep-water refugia (Trexler et al., 2000). Scientific consensus suggests that thermal constraints, and the difficulty associated with migrating within the ridge-and-slough landscape, limit their distribution to within approximately 1 km of canals. As such, their impact on the marsh communities to date is considered minimal (Shafland, 1996). A number of nonindigenous fish species have been recorded in low relative abundance within certain marshes of the Greater Everglades (e.g. Chick et al., 2004; Kobza et al., 2004; Dunker, 2003; Trexler et al., 2000), but no extensive, long-term systematic surveys have specifically targeted nonindigenous fish, and the sampling methods employed to date have biases that potentially under-sample nonindigenous fish (Loftus, 1987). These findings indicate that the distribution, abundance, and species diversity of nonindigenous fish in the Greater Everglades may be considerably underestimated, and that little is understood about nonindigenous fish species and their impacts in the marsh.

The District investigated nonindigenous fish diversity in WCA-3A and examined whether these species are established in the marsh or restricted in distribution by proximity to a canal during a study in 2005. To determine establishment, their relative abundance was evaluated in relation to distance from the L-67A canal. A species was considered established if its relative abundance beyond 1 km of the canal was greater than or equal to that within 1 km. The nonnative fish captured in this study included three species of cichlid and a catfish. These species were an important component of the marsh fish community, accounting for 16 percent of the species count, 5 percent of the total biomass, but less than one percent of the total fish count.

The black acara (*Cichlasoma bimaculatum*) was found 3 to 4 km from the canal, suggesting it is established in the marsh. This species was caught only 3º C above its stated minimum lethal temperature (P. Shafland, personal communication). The Mayan cichlid (*C. urophthalmus*) was the eighth most abundant fish of the marsh fish community in terms of biomass. Mayan cichlids were distributed equally among the three distance categories, juveniles were captured 3 to 4 km from the canal, and it is likely that this species is established in the marsh. It was captured up to 2º C above its stated minimum lethal temperature (P. Shafland, personal communication).

A juvenile brown hoplo (*Hoplosternum littorale*) was captured 2 to 3 km from the canal. While a single individual (**Figure 9-43**) reveals little about possible establishment, its capture 2 to 3 km from the canal and observations of bubble nests in other areas of WCA-3A suggest that this species is established and warrants further investigation. A single juvenile spotted tilapia (*Tilapia mariae*) was captured within 1 km of the canal. This species is widespread in South Florida (Fuller et al., 1999), but its establishment outside of the canals, lakes, and ponds surrounding WCA-2A is unknown.

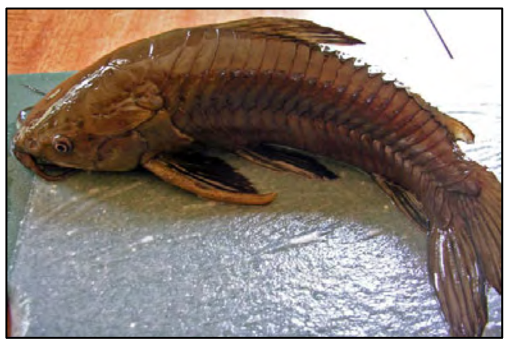

Figure 9-43. Brown hoplo (*Hoplosternum littorale*) (Photo by Joe Guthrie, courtesy Archbold Biological Station).

Although this survey was unable to statistically determine establishment for these nonindigenous fish species, it suggests that at least two species are established in the interior of the Central Everglades. A similar study examining the community structure of fishes and invertebrates along transects originating at canals in the central and southern Everglades did not report nonindigenous fishes (Rehage and Trexler, 2006). However, localized canal effects attributable to nutrient enrichment were found, and those authors call for further study of predatory fish movements within canals and their impacts. Future studies are needed to examine ecological factors affecting distribution of nonindigenous species and to reevaluate species-specific physiological tolerances to seasonal minimum temperature.

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BIG CYPRESS MODULE

The Big Cypress Module is made up of Big Cypress National Preserve (BCNP) to the east, a patchwork of public and private lands to the west, and tribal lands to the north.

Nonindigenous Plants

The Seminole Tribe of Florida has completed an Invasive Species Management Plan, along with a Tribal Invasive Species database for internal tracking of invasive species populations. The Invasive Species Management Plan has aided in prioritizing target species such as melaleuca, Brazilian Pepper, *Lygodium*, and Tropical Soda Apple for treatment. The Seminole Tribe of Florida has also funded invasive species research on *Lygodium*. Melaleuca is effectively controlled on most public lands, but appears to be spreading on private lands. The USDA-sponsored Melaleuca Biological Control Program is a particularly important component of the overall melaleuca management strategy in this module because some of the first releases were made here, and the biocontrol insects are having marked impacts to the melaleuca in this area.

The first melaleuca biocontrol agent, a melaleuca weevil (*Oxyops vitiosa*), was introduced in 1997 and subsequently established on melaleuca throughout the region **Figure 9-44**. Recent studies by USDA entomologists have determined that weevil attacks suppress reproduction by 80 percent. The few trees that do reproduce have smaller flowers that contain fewer seeds. The second agent, the melaleuca psyllid (*Boreioglycaspis melaleucae*), was released in 2002. USDA entomologists have determined that psyllid feeding on melaleuca seedlings results in 60 percent mortality in less than a year. This type of feeding accelerates the defoliation caused by the weevil and further weakens melaleuca trees. The combined efforts of these two biological control agents have resulted in thinning of the melaleuca canopy in many areas (**Figure 9-45)**, which allows more sunlight to reach the forest floor. As a result, native species are beginning to return to some melaleuca-dominated habitats and are able to compete with the exotic tree. To facilitate the distribution of these biological control agents, state and federally supported collection and redistribution efforts have resulted in the release of over 1.9 million insects at 319 locations across 15 counties in South Florida (**Figure 9-4**). A coordinated strategy concentrated insect releases in environmentally sensitive restoration sites or melaleuca-dominated areas that were not currently slated for herbicide treatments. This approach aims to use biological control agents to reduce re-invasion of managed sites and halt continued melaleuca spread in untreated sites. The effects of these two biocontrol agents are most apparent in the Big Cypress Module and will be important in the long-term control of this tree given the large percentage of melaleuca that remains on unmanaged private lands. Statewide, *O. vitiosa* and *B. melaleucae* have dispersed from their original release sites by 35 and 60 percent, respectively; statewide foliage destruction ratings are estimated at ~30 percent for both species, though this number varies by site (P. Pratt, USDA-ARS, unpublished data).

Figure 9-44. Melaleuca weevil (Photo by Stephen Ausmus, USDA-ARS).

Figure 9-45. Melaleuca biocontrol weevil damage (top branch) (Photo by Peggy Greb, USDA-ARS).

The bud-gall fly, *Fergusonina turneri*, (and its obligate mutualistic nematode *Fergusobia quinquenerviae*) was the third insect species to be distributed against melaleuca. The USDA Animal and Plant Health Inspection Service (USDA-APHIS) issued a permit for the release of

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F. turneri (+ F. quinquenerviae), and releases were made at six sites in South Florida in 2005. The original releases were not successful, though releases made in winter 2007 have resulted in the preliminary establishment of these mutualistic species. It will be necessary for these biocontrol agents to make it through the 2007 hurricane season before this release can be considered completely successful. Additional releases are planned for the near future in order to expand their distribution in South Florida (P. Pratt, USDA-ARS, personal communication). In addition to the above-mentioned biocontrol agents that have already been released, the melaleuca biocontrol program will soon be strengthened by the addition of the gall midge, *Laphlodiplosis trifida*, and the weevil, *Haplomyx multicolor*. The petition for release was submitted to the Technical Advisory Group in May 2007. *H. multicolor* is in quarantine with rearing techniques currently being perfected for this species.

Old World climbing fern, as in the Greater Everglades Module, poses a serious threat to restoration initiatives in this module. The District launched the first operational control program for this species at the Corkscrew Regional Ecosystem Watershed property in 1999. District land managers are effectively controlling this species on District lands in the Big Cypress Module, but constant vigilance is necessary as new populations are continuously found. BCNP employs a "find and treat" contractor devoted to scouting for incipient populations of Old World climbing fern. This is a responsible strategy given the potential for this species to dominate many different habitats over large areas of the Preserve. A closely related nonindigenous species, Japanese climbing fern (*Lygodium japonicum)*, was recently found and controlled in the BCNP (J. Sadle, NPS, personal communication) (**Figure 9-46**). This species was previously thought to occur mostly north of Lake Okeechobee, and its possible invasion into southern Florida is of concern.

Figure 9-46. Japanese climbing fern (Photo by Chris Evans, River to River CWMA).

The floating aquatic fern, giant salvinia (*Salvinia molesta*) is a nonindigenous plant species of great concern in this module. It was first reported in Naples (1999) in the Airport Road Canal and later in the Golden Gate Canal (2004). This species is a notorious weed in other parts of the world. It quickly forms thick mats on top of the water and prevents light penetration of the water column, shading out native vegetation and degrading habitat for fish and wildlife. Given the threat this species poses to the aquatic and wetland areas of the state, the District initiated a program to treat and maintain this outbreak of giant salvinia in the hopes of containment. The USDA is also studying a biological control agent, the Salvinia weevil (*Cyrtobagous salviniae*) that was introduced (the source of this introduction is unknown) and has been heavily attacking giant salvinia in the Naples area. So far, the control programs including the biocontrol effort seem to be quite effective in South Florida, partly because the Salvinia weevil is a tropical species.

Figure 9-47. Crested floating heart (*Nymphoides cristata*) (Photo by NPS).

Crested floating heart, *Nympmhoides cristata* (**Figure 9-47**) is an aquatic exotic species of Asian origin that escaped ornamental usage in 1996 and invaded south and central Florida. The majority of this plant's biomass is beneath the water surface. Numerous control efforts have been initiated against this species. However, it has proven difficult to control because treated leaves die back but are able to regenerate from stems in the substrate. Priority plant species for the Big Cypress Module are in **Table 9-7**.

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Table 9-7. Stoplight table for priority plant species in the Big Cypress Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
BIG CYPRESS MODULE (Results in this row reflect module-level questions, not species-level questions)	R	Exotic populations decreasing significantly on publicly owned areas; occasional reductions on privately held areas	R Y	Good control of melaleuca and Australian pine ; first biocontrol releases for Old World climbing fern; new biocontrol for Brazilian pepper under study; other species still localized, but one new and potentially serious invader documented by NPS	Y
Australian Pine (*Casuarina* spp.)	Y G	Remnant populations exist along canals and a few natural sites, but decreasing overall	Y G	Chemical control effective; most natural areas clear or clearable with modest effort; biocontrol research under way	G
Air Potato (*Dioscorea bulbifera*)	Y	Not in Indicator systematic monitoring program; mostly occurs in developed areas	Y	No coordinated control programs in the module; biocontrol effort under way	Y
Cogon Grass (*Imperata cylindrica*)	Y	Mainly distributed along roadsides and levees; not part of a systematic monitoring program; currently not severe	Y	Treated as encountered in BCNP; no significant coordinated control efforts; no biocontrol effort under way; potential serious invader	Y R
Old World Climbing Fern (*Lygodium microphyllum*)	R Y	Serious invader, rapid spread; invades most habitats and very destructive; chemical control so far effective due to localized populations but spreads rapidly	Y	Module-wide controls not coordinated; biocontrol release made with additional release expected 2007; chemical control studies continuing	Y G
Japanese Climbing Fern (*Lygodium japonicum*)	Y R	Southernmost extent of species so far; little is known about its impacts in the module	Y	Populations have been controlled in the module so far; however, distribution and spread are unknown and no biological control is program under way	Y R
Melaleuca (*Melaleuca quinquenervia*)	Y G	Coordinated efforts to control species but is still abundant on private lands; biocontrol agents reducing cover, spread throughout module	Y G	Chemical control effective on most public lands; biocontrol agents effective and additional spread of existing agents and new agents expected in 2007-2008	G
Crested Floating Heart (*Nymphoides cristata*)		Not new to module but new to table; difficult to control; not part of systematic monitoring program	Y R	Potential to spread widely; past and current control efforts not successful	R
Downy Rose-myrtle (*Rhodomyrtus tomentosa*)	Y	Localized in coastal uplands; not included in Indicator systematic monitoring program	Y	No fully coordinated control efforts in module; no biological control programs under way	Y R
Giant Salvinia (*Salvinia molesta*)	Y	Seems to be under control in module; not included in Indicator systematic monitoring program	Y G	Serious aquatic weed in many parts of the world and southern US; module populations do not present a serious threat now due to active control efforts	Y G
Brazilian Pepper (*Schinus terebinthifolius*)	Y	Serious invader with rapid spread; invades most habitats and very destructive; chemical control ineffective in reducing module-wide spread; local control programs effective where resources available	Y	BCNP control program effective; many populations slated for control; new biocontrol agents under study for future release in 2007-2008	Y G
Tropical Soda Apple (*Solanum viarum*)	Y	Little known about spread or distribution; not present in stable, natural areas; not included in Indicator systematic monitoring program	Y	Controlled when encountered in BCNP; distribution poorly understood; introduced in contaminated sod; biological control program under way	Y

Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain green status.

Nonindigenous Animals

In addition to the priority plant species listed above, several nonindigenous animal species are considered priorities in the Big Cypress Module. Recent studies have collected several new records of nonindigenous fish for this region, and also indicate range expansions of several species northward from Everglades National Park. The African jewelfish (*Hemichromis letourneauxi*, **Figure 9-48**) is a new record for the Big Cypress area and is expanding its range northward after becoming abundant in solution holes of the Rocky Glades in southern Miami-Dade County. This species displays several traits that make it successful, including being extremely aggressive, saltwater-tolerant and guarding young from predation. The walking catfish (*Clarias batrachus*) is probably the best-known exotic fish in South Florida since it established in the 1980s and sparked a heated debate about the impact of exotic species. Adaptations that make this fish successful include the ability to emerge from water and move short distances across land, resistance to deoxygenated water, a cosmopolitan diet and the ability to produce many young.

Figure 9-48. African jewelfish (*Hemichromis letourneauxi)* (Photo by Noel Burkhead, USGS).

Feral Hogs

Feral hogs (*Sus scrofa)* are reported in all 67 counties of Florida and are extremely common in the Big Cypress Module. They were first introduced by the Spanish over 400 years ago (Frankenberger and Belden, 1976). Sporadic introductions of new populations have occurred over time by sportsmen (Tiebout, 1983). Florida's feral hogs consist of feral domestic hogs or hybrids of domestic hogs and wild boars, which readily interbreed (Johnson et al., 1982; Whitaker, 1988).

Feral hogs (**Figure 9-49**) are omnivorous and their diet varies seasonally. These hogs consume a variety of vegetation, invertebrates, insects, reptiles, frogs, bird eggs, rodents, small mammals, and carrion (Lowery, 1974; Bratton et al., 1982; Laycock, 1966; Baber and Coblentz, 1986; Gingerich, 1994). Although feral hogs are common throughout the Big Cypress Module, the greatest population numbers are found in pine flatwood savanna communities with an open canopy of slash pine (*Pinus elliotti* var. *densa*), an understory of palmetto (*Serenoa repens*), and a diverse ground cover of grasses, sedges, and forbs.

Figure 9-49. Feral hog (*Sus scrofa*) (Photo by Jim Mitchell, Global Invasive Species Database).

The composition and structure of major plant communities is a performance measure developed as a basis for monitoring Big Cypress within the context of RECOVER. The impacts from feral hogs in the Big Cypress Module (and Florida) are not well documented. However, it is widely known that hogs damage plant communities through rooting, compete with native wildlife species for forage, and host diseases and parasites communicable to humans, livestock, and wildlife (Laycock, 1984; Gingerich, 1994; Engeman et al., 2003, 2004a, 2004b).

Hogs use their snouts to uproot large areas of soil in search of edible plants, nuts, and acorns. In so doing, they damage natural plant communities, leaving large disturbed areas of bare ground (Engeman et al., 2003, 2004a, 2004b). These "plowed" areas impact water quality and interrupt native vegetation succession, facilitating the establishment and spread of exotic plants (Duever et al., 1986; Layne, 1984; Belden and Pelton, 1975; Laycock, 1984). This widespread activity is undoubtedly resulting in plant community alterations in this region. In addition to the direct physical impacts of rooting, feral hogs are also known to carry many diseases and parasites including pseudorabies (which is fatal in panthers; Gingerich, 1994), hog cholera, brucellosis, tuberculosis, salmonellosis, anthrax, ticks, fleas, lice, and various flukes and worms.

A recent damage estimate was conducted for feral swine impacts on Savannas Preserve State Park (see *Northern Estuaries – East Module* page 9-72), based on the monetary amounts wetland regulators have allowed to be spent in mitigation attempts to replace lost wetland resources. Even though the damage estimate was believed conservative by not taking all feral hog impacts into account, the benefit-cost ratio demonstrated that the benefits of feral hog removal are very high compared to the costs of control (Engeman et al., 2004a, 2004b).

Although the ecological impacts caused by this species in Florida are apparent (Engeman 2003, 2004a, 2004b), proposals for feral hog eradication are controversial since they are a valued game species (Baber and Coblentz, 1987; Laycock, 1984). Feral hogs are viewed as a source of income, recreational opportunity, and food (Belden, 1990) throughout Florida. Complicating the issue further, the endangered panther preys on feral hogs (Maehr et al., 1990) and it has been argued that feral hogs are important to the survival of this endangered species in Florida.

Mexican Bromeliad Weevil

The Mexican bromeliad weevil (*Metamasius callizona*, **Figure 9-50**) was originally introduced to Florida via a shipment of bromeliads imported from Mexico. It was first detected in 1989, and is now found in 22 counties in South Florida (Frank and Thomas, 1994, H. Frank, University of Florida, personal communication). The weevil is now attacking epiphytes in Big Cypress National Preserve, Florida Panther National Wildlife Refuge, and Fakahatchee Strand Preserve State Park.

Figure 9-50. Mexican bromeliad weevil (*Metamasius callizona*) (Photo by Sean McCann, Univ. Florida).

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The weevil attacks native bromeliad species, including 10 that are state-listed as threatened and endangered, and one endemic species. Two of these bromeliad species were listed due to damage done to their populations by the weevil (F.A.C., 2000). While adult weevils eat the leaves of bromeliads, weevil larvae cause the most damage as they bore deep into the growing tissue of a plant. The plant eventually dies and falls to the ground. Weevils can eventually destroy entire populations of a species. Bromeliads are important to many native taxa. Capturing water between leaf axils, bromeliads are a source of water and protection for many native insect, worm, frog, snake, and salamander species. In addition, this region of Florida is known for its rich epiphytic plant life. Fakahatchee Strand State Preserve was acquired by the state of Florida in 1972 to protect its unusual collection of rare plants, including rare bromeliads.

Figure 9-51. Drs. Howard Frank and Ron Cave release biocontrol flies against *M. callizona* (Photo by Robin Koestoyo, IFAS).

Pesticides are used to effectively keep these weevils in check in cultivated bromeliads, but the use of insecticides is not feasible in natural areas due to the epiphytic nature of wild bromeliads and the potential for impacting native insects. The University of Florida is working to track the spread of this insect and develop biological controls for the weevil. A potential biocontrol agent (the Honduran fly *Lixadmontia franki*) has been reared and tested for host specificity at the university's quarantine facility in Fort Pierce, Florida. Applications for its release permit were filed with APHIS-PPQ in December, 2006 and approved in May 2007. The first releases were made May 29, 2007, at Lake Rogers Park in Hillsborough County (**Figure 9-51**). Additional releases were made on the Loxahatchee National Wildlife Refuge on July 20, 2007, and October 12, 2007, and on August 29, 2007, at Big Cypress National Preserve. Baited traps will be put out over the course of the following months to determine whether the second generation of flies can find and parasitize the weevil (J. Frank, University of Florida, personal communication). In the meantime, additional field explorations have been and will continue to be conducted in Central America in search of supplementary biocontrol agents. Given the mounting obstacles in managing this pest with traditional chemical control methods, biological controls hold the only hope in controlling this species in Florida's wildlands.

Yellow Anaconda

The yellow anaconda, *Eunectes notaeus*, is a large snake native to South America that is almost always found near water (**Figure 9-52**). This heavily built animal can exceed 15 feet in length. It is yellow with uniform black oval markings down its body. Females are larger than males and give birth to live young (usually 8 to 30) after five months of gestation. This species was first discovered in South Florida in January 2007, likely introduced via the pet trade. Yellow anacondas feed primarily on small animals including heron, egrets, rodents, fish, and ducks. This diet makes their presence in the Everglades region particularly worrisome.

Figure 9-52. Yellow anaconda (*Eunectes notaeus*) (Photo by NPS).

Northern Estuaries – West Module

Figure 9-53. Invasive Species Strike Team and trailer (Photo by Bill Thomas, USFWS).

The Northern Estuaries – West Module is made up of the coastal estuaries of the west coast. It includes the Caloosahatchee estuary and the coastal communities and islands.

The Region 4 Invasive Species Strike Team is a two-person team formed by the USFWS to coordinate invasive exotic plant and animal management activities in South Florida. While its coverage includes all Florida National Wildlife Refuges, the team is based out of this module at the "Ding" Darling National Wildlife Refuge. The team conducts rapid response eradication efforts of invasive species and coordinates efforts with land managers across Florida and the southeast U.S. (**Figure 9-53**).

Nonindigenous Plants

Figure 9-54. Fallen Australian pine trees cause extensive structural damage (Photo by SFWMD).

A large portion of the invasive plant control operations in the coastal Caloosahatchee Estuary are carried out by local governments such as Lee County and the City of Sanibel. A town-sponsored program eliminated melaleuca from Sanibel Island in the 1980s. There is currently an Island Partnership focusing on Australian pine, Brazilian pepper, java plum, earleaf acacia, and *Sanseveria*. The USFWS provided $1.1 million for exotic species control on Partner lands, regionally, extending through 2007, possibly continuing into 2008 to be fully completed. Work to control Brazilian pepper is ongoing, with several mechanical removal projects under way. Efforts to control well-established Australian pine on the coastal islands of the estuary originally met with public resistance. That changed on August 13, 2004 when Hurricane Charley impacted Sanibel and Captiva islands. Many of the large Australian pine trees toppled and barricaded access to the islands for post-storm relief efforts (**Figure 9-54**). The tall trees also snapped power lines and were responsible for extensive structural damage (R. Loflin, City of Sanibel, personal communication; Ferriter et al., 2005). In light of the problems encountered as the result of the hurricane, city leaders now embrace the effort to control Australian pine on these coastal islands and other City-owned conservation lands (**Figure 9-55**). Federal Emergency Management Agency (FEMA) funding made broad scale control of this species possible. While Australian pine is at maintenance levels on most public, city, and conservation lands, it can still be found on private lands. The City of Sanibel strongly encourages private property owners to remove Australian pine, but at this time, there is no mandatory removal ordinance.

Figure 9-55. Mechanical control of Australian pine (Photo by Bill Thomas, USFWS).

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Climbing cassia (*Senna pendula*) and seaside mahoe (*Thespesia populnea*, **Figure 9-56**) are new additions to the priority plant list for this module. Climbing cassia is encroaching roadsides of I-75 and beginning to appear on Sanibel conservation lands; in the City of Sanibel, on the "Ding" Darling Refuge and Sanibel Captiva Conservation Foundation. Invasive plant contractors in this module are encountering seaside mahoe with regularity on Sanibel and satellite coastal island refuges.

Figure 9-56. Seaside mahoe (*Thespesia populnea*) (Photo by Amy Richard, Univ. Florida).

In addition to these species, several grasses were cited by land managers as problematic in the Caloosahatchee Estuary. Guinea grass (*Panicum maximum*), cogongrass (*Imperata cylindrica*), Burma reed (*Neyraudia reynaudiana*), itch grass (*Rottboellia cochinchinensis*), West Indian marsh grass (*Hymenachne amplexicaulis*), and para grass (*Urochloa mutica*) were cited as spreading and difficult to control, particularly in areas such as dredged spoil along the Caloosahatchee River. They are a management challenge because they occur in wetland areas, and the biology of these species is not sufficiently understood to effectively manage them in wetland areas (see also the *Lake Okeechobee Module* section).

Figure 9-57. Guinea grass (*Panicum maximum*) (Photo by Vic Ramey, Univ. Florida).

Guinea grass (**Figure 9-57**) has been successfully controlled on "Ding" Darling Refuge. However, on Sanibel, it is being spread through routine mowing operations that use heavy equipment contaminated with the plant. Contracting mowing operations to private companies is further accelerating the problem, as well as sharing equipment among Island Partner groups; thus moving seeds from one conservation parcel to another.

The priority plant species for the Northern Estuaries Module – West Coast are listed in **Table 9-8**.

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Table 9-8. Stoplight table for priority plant species in the Northern Estuaries – West Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
NORTHERN ESTUARIES – WEST MODULE (Results in row reflect module-level questions, not species-level questions)	R Y	Much progress made with melaleuca, Brazilian pepper, Australian pine; other species gaining foothold and most not included in any Indicator monitoring program; little known about large majority of invaders and not able to assess their status in an objective or repetitive way to study trends	R Y	Some control of melaleuca; first biocontrol releases for Old World climbing fern; new biocontrol for Brazilian pepper under study; other species still localized but numerous; potentially serious invaders exist for which little is known about biology or spread	Y R
Australian Pine (*Casuarina* spp.)	Y G	Populations exist along roadsides, canals, around agricultural fields ,and a few natural sites, removal programs in place, considered effective	Y G	Chemical control effective, many natural areas clear or clearable with modest effort; biocontrol research under way	G
Air Potato (*Dioscorea bulbifera*)	Y	Little known about spread/dist; not included in Indicator systematic monitoring program	Y	Control efforts not coordinated; biocontrol effort under way	Y R
West Indian Marsh Grass (*Hymenachne amplexicaulis*)	Y	Distributed in wet areas; not included in Indicator systematic monitoring program	Y	Species problematic because it is difficult to control with herbicides in wetlands; no biocontrol	Y R
Cogon Grass (*Imperata cylindrica*)	Y	Little known about spread/dist; not included in Indicator systematic monitoring program	Y	Species problematic because it is difficult to control with herbicides; no biocontrol effort under way	Y R
Old World Climbing Fern (*Lygodium microphyllum*)	R	Serious invader; rapid spread throughout module; invades most habitats; very destructive	R	No significant effective controls; biocontrol release made; more expected in 2007; chemical control studies continuing	R Y
Melaleuca (*Melaleuca quinquenervia*)	Y G	Still abundant on private lands but biocontrol reducing cover and spread, reduction of dense monocultures attributed to land clearing (i.e. development)	Y G	Chemical control effective on most public lands; biocontrol agents reducing rate of spread; new agents expected 2007/2008; continuous effort required	Y G
Burma Reed (*Neyraudia reynaudiana*)	Y	Little know about spread or distribution in the module; not included in Indicator systematic monitoring program	Y	Species problematic because difficult to control with herbicides; no biocontrol effort under way	Y R
Guinea Grass (*Panicum maximum*)	Y	Little known about distribution; spread accelerated by mowing; not included in indicator systematic monitoring program	Y	Species problematic because difficult to control with herbicides; no biocontrol effort under way	Y R
Itch Grass (*Rottboellia cochinchinensis*)	Y	Spreading in wetland areas; not included in Indicator systematic monitoring program	Y	Difficult to control with herbicides in wetlands; in tropical America, a serious invader often leading to land abandonment	Y R
Half-flower (*Scaevola taccada*)	Y	Coastal species; spreading but easy to detect; not included in Indicator systematic monitoring program	Y	Control efforts effective where implemented; seed source from surrounding ornamental plantings makes long-term control necessary; biocontrol prospects limited due to native *Scaevola*	Y R
Brazilian Pepper (*Schinus terebinthifolius*)	Y	Serious invader with rapid spread throughout module; invades most habitats and is very destructive; local control programs are proving effective where resources available	R Y	Control programs in module effective in natural areas where management programs under way; new biocontrol agents under study for future release; spreads easily so constant control needed	R Y

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Table 9-8. Continued

Climbing cassia (*Senna pendula*)		New to priority plant list but not to module; covers roadsides and increasing on conservation areas; not included in Indicator systematic monitoring program	Y	Populations increasing throughout module; potential to spread rapidly; no coordinated control efforts	Y R
Seaside Mahoe (*Thespesia populnea*)		New to priority plant list but not module; increasing on Sanibel and coastal island refuges; not in Indicator monitoring program	Y	Populations increasing here; potential to spread rapidly; no coordinated control efforts	Y R
Para grass (*Urochloa mutica*)	Y	Distributed in wetland and disturbed areas, un-maintained canal and roadside ditches; not included in Indicator systematic monitoring program	Y	No coordinated control efforts in place for the module; no biocontrol effort under way although local populations can be eliminated	Y R

Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources, the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain green status.

Nonindigenous Animals

Figure 9-58. Guayas cichlid (Photo by Ernst Sosnas, AquaNet).

This area has experienced coastal and inland development pressure and also receives freshwater releases from Lake Okeechobee. While marine fisheries monitoring appears to be adequate, additional freshwater fish monitoring may be necessary in this region to quickly detect new introductions and impacts. In January 2007, a new species of nonindigenous fish, Guayas cichlid (*Cichlasoma festae*, **Figure 9-58**), was discovered in a freshwater lake in a subdivision in Lee County (USGS-NAS Alert, January 2007). The freshwater lake is connected directly to the Caloosahatchee River, raising fear among agencies that this species has the potential to become established in the Caloosahatchee drainage system and associated estuaries. Cichlids, in general, have the ability to tolerate a wide range of water salinities.

The Mozambique tilapia (*Oreochromis mossambicus*) is a well-known food fish, has been aquacultured extensively, is an aquarium-trade species, and has become a popular sport fish. Some successful adaptations include tolerance to low oxygen, a non-specific diet and the ability to modify breeding behavior. The spotted tilapia (*Tilapia mariae*) has a broad tolerance to salt water and shows biparental protection of young which may contribute to its success. In fact, this

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species was so successful that it served as the primary justification for the release of the exotic peacock cichlid (*Cichla ocellaris*) to act as a control in Miami-Dade County. In addition to the fish species listed above, several animal species are considered priorities in the Northern Estuaries – West Module and could seriously impact this coastal ecosystem.

Monitor Lizard

The African Nile monitor lizard (*Varanus niloticus*) was first noted in Cape Coral in 1990 and has rapidly colonized the region. The source of the Cape Coral population is undocumented, but researchers believe that several monitor lizards were either intentionally or accidentally introduced. This agile climber and swimmer has since dispersed to nearby islands such as Pine Island (G.S. Player, FWS, personnel communication), Sanibel Island (Brad Smith, SCCF, personnel communication) and the mainland, and has recently been observed in the sawgrass prairies in extreme southern Miami-Dade County (K. Krysko, Florida Museum of Natural History, personal communication; **Figure 9-59**). A number of individuals have been observed in a lake north of Orlando, and also along a canal in Palm Beach County, indicating that additional populations may be established around the state (T. Campbell, Univ. Tampa, personal communication).

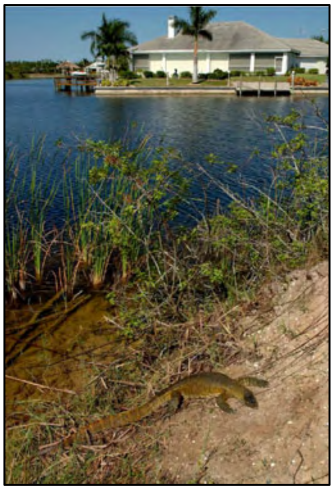

Figure 9-59. Nile monitor lizard (Photo by Todd Campbell, Univ. Tampa).

The median size for an adult male is 5 feet, but they can reach lengths of more than 7 feet (Faust, 2001). Although this large reptile species is an ill-suited pet, it is a popular novelty in the exotic pet trade. The rapidly expanding Southwestern Florida Nile monitor lizard population is of concern for several reasons. Cape Coral is situated between Matlacha Pass and the Caloosahatchee River. It has more than 400 miles of canals and is fringed with ecologically important mangrove communities, tidal creeks, and marshes of the Charlotte Harbor State Buffer Preserve and the Matlacha Pass State Aquatic Preserve. These habitats have proven to be ideal for this reptile, which is poised to become a top predator. In its native range, the Nile monitor lizard preys or scavenges on a variety of snails, clams, oysters, crabs, fishes, birds, eggs, and small mammals. Amphibians and reptiles, and the eggs of both, comprise a significant portion of their diets, and as a result, the impacts on native amphibians and reptiles may be significant.

Researchers fear that it is only a matter of time before the species begins to breed in other estuarine and freshwater swamps, marsh edges, riverbanks, canals, and lakes, which are all suitable habitats (Enge et al., 2004). In response to the threats associated with this species in southwest Florida and beyond, the University of Tampa initiated an aggressive trapping program on Cape Coral (**Figure 9-60**). Unfortunately, funding for this program ran out in 2005, with only one part-time trapper currently employed. The Cape Coral population is now estimated at well over 1,000 individuals of various size classes, and is increasing. Cape Coral has the largest population of burrowing owls in Florida, and a Nile monitor lizard was once observed killing a young owl. Monitors could also impact populations of other listed species in this region (Enge et al., 2004).

Figure 9-60. Researcher Todd Campbell with Nile monitor (Photo by T. Campbell, Univ. Tampa).

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One of the biggest concerns is an impact to birds on the "Ding" Darling Refuge on Sanibel Island, one of the most important bird sanctuaries in the state. Given the lack of funding to eradicate Nile monitors from Cape Coral and the surrounding area, land managers are trying to obtain funding to at least keep them off Sanibel Island (T. Campbell, Univ. Tampa, personal communication). A few sightings were regularly reported during 2005/2006 from the Sanibel Bayous, an exclusive community located due west of the "Ding" Darling Refuge and other Partner conservation lands. A flier was produced and distributed with contact information to report Sanibel sightings in an effort to rapidly respond and remove the animal(s). One individual animal was harvested 'under the radar' on Sanibel (Brad Smith, SCCF, personnel communication) since 2005, and a second individual was spotted on the island in June 2007, indicating there might be an established population.

Associated research at the University of Tampa and the University of Florida aims to understand the basic biology — feeding habits, activity patterns, and reproductive cycle — of the species, information that is critical to developing an effective management plan for this reptile, which appears to be approaching an exponential rate of expansion in Southwest Florida.

Black Spiny-Tailed Iguana

The black spiny-tailed iguana (*Ctenosaura similis*) (**Figure 9-61**) and Mexican spiny-tailed iguana (*C. pectinata*) are large, primarily herbivorous reptiles that are established in South Florida. The spiny-tails have a more aggressive nature than green iguanas (*I. iguana*) and, although also introduced by the pet trade, are much less suitable as pets than the green iguana.

Adult spiny-tailed iguanas reach 4 feet in length and feed primarily on leaves, fruit and flowers, but occasionally eat insects, small animals, bird eggs, and hatchling sea turtles. Juveniles are more carnivorous than adults.

Figure 9-61. Black spiny-tailed iguana (*Ctenosaura similis*) (Photo by Ellen Donlan, SFWMD).

Black spiny-tailed iguanas were introduced to the Northern Estuaries – West Module in the mid-1970s. They now occur on Gasparilla Island, Cape Haze, Gulf Cove, Cayo Costa, Keewaydin Island, and Little Marco Island and on the mainland at Placida (Krysko et al., 2003). On the east coast, they occur in Key Biscayne and elsewhere in Miami-Dade and Broward counties (Townsend, 2003). This species endangers the threatened least tern (*Sterna antillarum*), Wilson's plovers (*Charadrius wilsonia*), and snowy plovers (*C. alexandrinus*) and could impact nesting loggerhead sea turtles (*Caretta caretta*) (Krysko et al., 2003). Spiny-tailed iguanas could also contribute to burrowing owl impacts (see the *Monitor Lizard* section in this module) if they spread to Cape Coral. They would likely compete for burrows and could prey on nestlings (Krysko et al., 2003).

In addition to impacts to native species, the reptiles actively dig extensive burrows along and under cement walls, seawalls, or pavement and, most troubling, in the dunes along beaches. These burrows can weaken natural dunes and lead to structural erosion, undermining, and collapse of manmade features. Their droppings are possible sources of salmonella contamination as are their bites. When cornered, spiny-tailed iguana bites and claws can cause serious lacerations, and tail slaps can deliver powerful blows.

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Native predators control young iguanas to some degree. Raccoons dig up nests while raptors, alligators, wading birds and snakes may possibly take immature iguanas. However, once mature, few Florida animals serve as natural enemies, unlike the large cats and snakes resident in the iguanas' native range.

Mature black spiny-tailed iguanas are faster than green iguanas making noose capture techniques difficult. Snares, trapping and hunting may be effective control methods but are subject to state and local regulations. One of the most troublesome aspects of iguana control in the area is how to dispose of the dead animals. Chapter 39-4.005, F.A.C., prohibits non-native animal releases, but the animals can be sold or given to pet stores, often exacerbating the problem.

In response to this threat in the module's coastal communities, Lee County commissioners recently voted unanimously to devote $180,000 toward the extermination of an estimated 20,000 iguanas purported to infest Boca Grande (at www.News-press.com, accessed April 2007). Lee County has also developed a brochure to educate tourists and residents about discouraging iguanas and effecting their breeding habits. The brochure, "Do Not Feed the Iguanas", shows photographs and facts about iguanas, including ways to stress them enough to reduce their population.

Green Mussel

The green mussel (*Perna viridis*) was first discovered in 1999 by maintenance divers inspecting a jammed intake valve at the Big Bend power plant in Tampa Bay, Florida. Larvae-infested commercial ballast water releases are believed to have been the source of this introduction. A native to the Indo-Pacific region, this species is now well-established in Tampa Bay, fouling bridges, piers, buoys, and decimating oyster beds (**Figure 9-62**).

Figure 9-62. Green mussel (*Perna viridis*) (Photo by Patrick Baker, Univ. Florida).

From Tampa Bay, currents dispersed green mussel larvae south along the Gulf Coast to Boca Grande outside of Charlotte Harbor (Benson et al., 2001), and the mussel now occurs as far south as Naples (Fajans and Baker, 2004). In 2002, green mussels were confirmed in Pensacola Bay in the Florida Panhandle, in the Ten Thousand Island region, southwest Florida, and along the Northeast Florida coast stretching from Daytona Beach to the Georgia-Florida border. It is believed that these populations resulted from either adults being transported on vessel hulls or larvae present in contaminated ballast water (available at www.greenmussel.ifas.ufl.edu). The 1–2 year prognosis is bleak, as experts believe that this invasive species will continue to spread throughout Florida's waters.

Prior to 2002, the species was believed to be confined to manmade structures. However, recent surveys show that green mussels are establishing in a wider variety of habitats (Baker, 2003). Of particular concern is the evidence that green mussels are becoming abundant on eastern oyster (*Crassostrea virginica*) beds (Baker and Benson, 2002). Densities can be very high in these areas (**Figure 9-63**), and this nonindigenous species is replacing the biomass formerly produced by oysters. Baker (2003) found that the oyster reef matrix and structure remain, but over 90 percent of adult oysters are recently dead (shells still articulated by the ligament).

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Figure 9-63. Green mussel infestation in Tampa Bay (Photo by Marc Blouin, USGS).

Several factors make this species a threat to the Caloosahatchee Estuary. It disperses easily, grows fast, and reproduces quickly. Fajans and Baker (2004) found high densities of approximately 4,000 individuals per square meter in Tampa Bay. The green mussel appears to have a lack of local predators and high tolerance of environmental conditions. Researchers expect the mussel population to expand in Gulf Coast and Atlantic habitats until it reaches its thermal limits. Unfortunately, there is little that can be done if green mussels overtake the oyster beds of the Caloosahatchee Estuary. Non-native marine invertebrates are challenging to manage. Intensive mechanical and chemical (continuous high-level chlorination) control is possible in closed systems such as power plants, but these methods are not feasible in a natural ecosystem, making selective control and eradication of this species in oyster beds virtually impossible.

Healthy oyster beds are a key ecological performance measure in restoration efforts, but to date the invasion of this nonindigenous invertebrate has not been considered in restoration models. Important work is under way by the University of Florida and the USGS to understand the spread and environmental impacts of this species in coastal ecosystems.

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NORTHERN ESTUARIES – EAST MODULE

The Northern Estuaries – East Module is made up of a strip of coastal estuaries along the eastern coast of South Florida. Priority species for this region mainly include coastal species. The majority of the work is done by the FDEP, local governments, and volunteer groups.

Nonindigenous Plants

The construction and maintenance of the Intracoastal Waterway channel and barrier island inlets resulted in the formation of a chain of spoil islands in this area. These islands, formed by the deposition of the dredged material (spoil), generally parallel the channel alignment. They are often dominated by exotic vegetation, such as Australian pine and Brazilian pepper. Australian pine was most likely planted on these islands in an effort to stabilize them. The other coastal systems in this module are also highly prone to invasion by Australian pine and Brazilian pepper. East coast populations of mangroves are near their northernmost range in this module and are impacted by periodic freezes. Because damaged mangrove communities reestablish slowly, they can be replaced by these faster-growing exotic species.

Figure 9-64. Air potato *(Dioscorea bulbifera)* (Photo by James Miller, USDA Forest Service).

Mangroves stabilize shorelines by trapping sand in their roots, providing homes to countless birds and fish, and providing the food base for almost every species living in the estuaries. Agency control efforts spearheaded by the FDEP are ongoing to restore mangrove, salt marsh, and upland habitat along the shoreline; a coalition of volunteer groups is active in working to remove Brazilian pepper and replant native shoreline vegetation. Several other species are considered priorities in this module. Torpedograss (*Panicum repens*), is becoming a major problem in low-lying areas in the module's floodplains. At Savannas Preserve and areas along the St. Lucie River, torpedograss is spreading quickly, but little is being done to manage this species. Shoebutton ardisia (*Ardisia elliptica*) is a major understory problem in many areas around the North Fork and in wetland areas along or adjacent to the Indian River. Air potato (*Dioscorea bulbifera*) is a continual problem in several areas of the module, and the plant is persistent in treated areas (**Figure 9-64**). A biological control program has been initiated against air potato, fortunately, with numerous promising species resulting from field explorations for potential candidates (R. Pemberton, USDA-ARS). Tropical soda apple (*Solanum viarum*) is found throughout improved and unimproved pastures within this module.

Downy rose myrtle (*Rhodomyrtus tomentosa*, **Figure 9-65**) is a landscape shrub of Asian origin that now occurs throughout South Florida, overtaking native pinelands' understory. This fast-growing shrub spreads more prolifically than shoebutton ardisia and other nonindigenous plant species currently of concern. Consequently, this species was added to the priority plant list in 2007. Little is known about its biology and it is challenging to control. Recent herbicide trials using Vanquish show promise since the chemical is effective and demonstrates reasonable selectivity in flatwoods.

Figure 9-65. Downy rose myrtle (Photo by Amy Richard, Univ. Florida).

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In addition to the plants discussed above, the occurrence of a nonindigenous marine plant (an alga) in the region's coastal areas concerns many scientists and managers (**Figure 9-66**). In 2001, an invasive non-native macroalga was identified growing on underwater reefs located off the coast in Palm Beach County. *Caulerpa brachypus*, a commonly sold marine aquarium plant native to Pacific waters, has now been found as far north as Fort Pierce and it is likely that it will continue to spread north and south from Palm Beach County. Because this species has not been carefully monitored, its actual distribution has not been determined. Anecdotal information gathered from dive operators and fisherman have reported that the species is now becoming so thick it is forcing fish and lobster away from reefs. Scientists have speculated that besides forming a dense canopy or blanket over a coral reef and killing it, the macroalga is reducing the food source for many fish species.

Current thinking within the scientific community suggests that excess nutrients, particularly nitrogen from septic seepage and offshore outfalls, may be responsible for the rapid colonization of Palm Beach County's underwater reefs by *Caulerpa brachypus* and two other native macroalga species. Studies by Harbor Branch Oceanographic Institution personnel are under way to determine if excess nutrients are fueling macroalgae blooms along South Florida's coastline. This is a potentially serious problem for the reefs along the Florida Keys as nutrient run-off from the keys has already been documented as a problem for the reefs (Lapointe & Clark, 1992; Leichter et al., 2003).

Figure 9-66. Caulerpa (*Caulerpa brachypus*) (Photo by FDEP).

Since 1984, a related nonindigenous species, *C. taxifolia* (**Figure 9-67**) has invaded broad areas of the Mediterranean and is documented in a San Diego, California lagoon and in the harbor of Sydney, Australia. In California, a $6 million chlorine treatment controlled an infestation in 2000. To date, this species affects thousands of acres of Mediterranean reefs causing at least $1 billion in damages. Also, internal toxins of *C. taxifolia* have been found to repel herbivory as well as inhibit the proliferation of several species of phytoplankton. At this time, it is unclear whether *C brachypus* will have the same impacts (Lemée et al., 1997) in South Florida's marine systems, but given the potential of this plant species to spread in coastal environments, it is clear that if it does become established, it will impede key restoration performance indicators such as healthy native submersed aquatic vegetation communities, fish communities, oyster beds, and healthy near-shore reefs.

Figure 9-67. *Caulerpa taxifolia* (Photo by Rachel Woodfield, Merkel and Associates, Inc.).

In response to these macroalgae blooms along the coast, the Florida Harmful Algal Bloom Task Force was created by the Florida legislature in 1999 to review information, prioritize research needs, and recommend plans to predict, mitigate, and control harmful algal blooms. Panel members include representatives from the FDEP, FWC, St. Johns River Water Management District, Harbor Branch Oceanographic Institution, National Undersea Research Center, Smithsonian Institution, and the Indian River Lagoon Estuary Program.

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Figure 9-68. Feathered water fern (*Azolla pinnata*) plant and infestation (Photo by Mike Bodle, SFWMD).

In June 2007, feathered water fern, *Azolla pinnata*, was identified in the South Indian River Water Control District drainage canal system in unincorporated Jupiter, Florida. (**Figure 9-68**). Weir outfalls have undoubtedly released the plant into the drainage canal of the adjacent Florida Turnpike. This constitutes the plant's first Florida report, and the only previous North American report is from North Carolina (available at http://plants.usda.gov/java/profile?symbol=AZPI). This Old World native is listed as a Federal Noxious Weed, and the FDEP and the District are mounting containment treatments to try to restrict the population from wider establishment.

The priority plant species for the Northern Estuaries – East Module are in **Table 9-9**.

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Table 9-9. Priority plant species in the Northern Estuaries – East Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
NORTHERN ESTUARIES MODULE – EAST COAST (Results in row reflect module-level questions, not species-level questions)	Y	Much progress made with melaleuca, Brazilian pepper and Australian pine, other species increasing, most not included in Indicator monitoring programs; little known about majority of invaders; unable to assess status in repetitive way to determine trends	Y	Good control of melaleuca, Brazilian pepper, and Australian pine; first biocontrol releases for Old World climbing fern; Brazilian pepper biocontrol under study; other species still localized but numerous, potentially serious invaders exist for which little is known about biology or spread	Y R
Shoebutton Ardisia (*Ardisia elliptica*)	Y R	May be entering exponential spread phase; moving into floodplain communities and dominating understory; difficult to monitor, especially remotely	Y R	No coordinated, significant control efforts or biocontrol efforts underway	R
Feathered Water Fern (*Azolla pinnata*)		New to module; early eradication and containment programs in place	Y	Problematic species in other parts of world but rapid response efforts enacted	Y G
Australian Pine (*Casuarina* spp.)	Y G	Remnant populations exist along canals and a few natural sites, removal program in place and effective	Y G	Chemical control effective, most natural areas clear or clearable with modest effort; biocontrol research under way	G
Caulerpa (*Caulerpa brachypus*)	Y	Little known about spread or distribution; not included in Indicator systematic monitoring program	Y	Potential to eliminate most species on hard bottom coastal areas; no significant control efforts under way	Y R
Air Potato (*Dioscorea bulbifera*)	Y	Little known about spread or distribution; known populations increasing despite some control efforts; not included in Indicator systematic monitoring program	Y	Control programs in the module have limited success in natural areas; biocontrol effort under way	Y R
Old World Climbing Fern (*Lygodium microphyllum*)	R	Serious invader, rapidly spreading despite control efforts; invades most habitats; very destructive	R	No effective module-wide control programs; biocontrol release made, additional release expected in 2007; chemical control studies continuing	R
Melaleuca (*Melaleuca quinquenervia*)	Y G	Decreasing or static on public lands; increasing on private; biocontrol agents slowly establishing in this module	Y	Chemical control effective on public lands; biocontrol agents effective; and new agents expected in 2007/2008	Y
Torpedograss (*Panicum repens*)	Y	Little known about spread or distribution but increasing in many natural areas; not included in Indicator systematic monitoring programs	Y R	No coordinated control efforts in place; no biocontrol efforts underway	Y R
Downy Rose Myrtle (*Rhodomyrtus tomentosa*)		Not new to module, but new to table; moving into floodplain communities and dominating understory; difficult to monitor, especially remotely	Y R	No coordinated, significant control efforts or biocontrol efforts underway	R
Brazilian Pepper (*Schinus terebinthifolius*)	Y	Serious invader still spreading; chemical control ineffective in reducing systemwide spread; local control programs effective where resources available	Y	Control programs in the module effective on public lands; new biocontrol agents under study for future release in 2007-2008	Y G
Tropical Soda Apple (*Solanum viarum*)		Not new to module but new to table; increasing on private lands despite minor control efforts	Y	Control efforts limited, although local populations can be eliminated; additional biocontrol agents to be released in 2007	Y

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Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources, the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain Green status.

Nonindigenous Animals

In addition to the plant species listed in **Table 9-9**, several nonindigenous animal species are considered a priority for the Northern Estuaries – East Module. Several of these species are discussed in other modules and are of special concern to the east coast estuaries. The green mussel (see the *Northern Estuaries – West Module* section, page 9-66) was recently found on the eastern coast of Florida and threatens to decimate oyster beds in this area. The Mexican bromeliad weevil (see the *Big Cypress Module* section, page 9-58) is impacting the inland areas of this region, killing bromeliads in the Savannas State Preserve in St. Lucie County. In addition, several nonindigenous fish species such as the brown hoplo, Mayan cichlid, walking catfish, sailfin catfish and the island applesnail (= channeled applesnail) have all been found in or near the District's C-24 canal, as well as numerous exotic reptiles and amphibians (Meshaka and Smith 2005, Cress et al., 2007).

Feral Hog

Feral hogs (*Sus scrofa*, see *Big Cypress Module* page 9-57 for species specific information*)* were first introduced by the Spanish over 400 years ago and now occur throughout Florida. Economic impacts for feral swine damage were conducted in wetlands in Savannas Preserve State Park in this module in 2003 and again in 2004 following a one-year implementation of a swine control program (Engeman et al., 2004a, 2004b). Values used for the swine damage were based on the monetary amounts wetland regulators allowed to be spent on mitigation attempts to replace lost wetland resources. In 2003, the area of natural habitat damaged by feral hogs was given a monetary value (admitted to being a conservative estimate for not taking all feral hog impacts into account). The damage to the study area was re-estimated in January 2004, after swine removal. Damage was significantly reduced from 2003 to 2004, with 31 percent of sampling transects showing damage in January 2004, versus 92 percent in January 2003. Similarly, the total area and subsequent value of swine damage had also decreased dramatically in 2004. The benefit-cost ratio of the damage reduction against feral hog control costs was conservatively estimated at $480–$1,562, demonstrating that the benefits of swine removal are very high relative to the costs of control. Feral hog damages to pine flatwoods also have been evaluated in three Florida State Parks located in this module (Savannas Preserve, Jonathan Dickinson and Atlantic Ridge) (Engeman et al., 2003) Intensive hog removal at one park resulted in the lowest level of habitat damage (1.3 percent).

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Northern Curlytail Lizard

The Northern Curlytail Lizard, *Leiocephalus carinatus armouri*, is endemic to the Little Bahama Bank. It was first introduced to Florida by the intentional release of 20 pairs on the island of Palm Beach in the 1940's, possibly to rid sugarcane fields of pests. This species is also popular in the pet trade, which has resulted in additional releases and escapes. Its range is contiguous for 90 km along the Atlantic Coast from Martin County to Broward County (Meshaka et al., 2005). Its rate of expansion on the Florida mainland was 2.4 km/year over a 34 year period (Smith and Engeman, 2004). Another subspecies (*L. carinuatus virescens*) occurred in Miami prior to 1940 but died out shortly afterwards. A third subspecies (*L. carinatus coryi*) was found on Virginia Key and Key Biscayne in Dade County, but its present status is unknown.

Figure 9-69. Northern curlytail lizard (*Leiocephalus carinatus armouri*) (Photo by Elizabeth Golden, DEP-Florida Park Service).

The northern curlytail lizard is found in mostly terrestrial habitats (Smith and Engelam, 2004) (**Figure 9-69**). It climbs well and prefers areas with ground rubble. Males may reach a length of 28 cm and be gray to tan, with light stripes on the nape and back. The dark-banded tail is held curved above the back. These lizards reach sexual maturity within one year and lay clutches of approximately four large eggs over a four or five month period (Meshaka et al., 2006). Their fast growth to maturity and their staggered generations contributed to the colonization success of this species (Meshaka et al., 2006).

Northern curlytails feed primarily on insects, but have been observed feeding on anoles (Smith and Engeman, 2004). Various falcons, hawks, a little blue heron, domestic and feral cats, black racers, and other animals have been witnessed feeding upon these lizards. Although competition between northern curlytails and native species has not yet been documented, populations of the exotic brown anole (*Anolis sagrei*) have been shown to decrease where they overlap with the current range of northern curlytails. It is reasonable to speculate that native lizards have been or will be impacted by northern curlytails within their expanding range (Smith and Engeman, 2004). Further study of this lizard and its interactions with native species is warranted.

Charru Mussel

The charru mussel *(Mytella charruana)* is native to Central and South America. It was first reported in Florida in 1986 when large numbers were found in power plant intake pipes on the St. Johns River. The mussel failed to become further established in the Jacksonville area, and most likely died off in the winter of 1987. (Boudreaux and Walters, 2006). The charru mussel was found in the Mosquito Lagoon Basin of the Indian River Lagoon in 2004 (Boudreaux and Walters, 2006). Since this report, many more charru mussels have been identified, and their numbers appear to be increasing, prompting the University of Central Florida to begin lagoon-wide surveys in 2006 to determine the distribution of the charru mussel in this module. As of Spring 2006, nearly 600 individuals had been collected from the Mosquito Lagoon portion of the Indian River Lagoon system. Like the green mussel (*Perna viridis*) described in the Northern Estuaries – West Module, this species threatens to compete with native mussels, oysters, and other organisms for food and colonizable substrate.

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LAKE OKEECHOBEE MODULE

Lake Okeechobee is a 450,000-acre lake with an average depth of only 9 feet. It also contains approximately 100,000 acres of littoral zone with herbaceous marshes, other emergent wetlands, and numerous islands. More than 80 non-native plant species have been identified in the Lake Okeechobee Module. Of these, 10 have been or are considered serious, invasive, and potentially threatening to the Lake Okeechobee ecosystem. The lake is a highly regulated and managed system that has serious nutrient enrichment problems (Havens et al., 1996). Fortunately, the majority of invasive plant species of concern in the lake have dedicated funding and effective control programs in place. Still, however, some species have proven difficult to control. The current status of invasive species, although improving in many areas, is not optimal. The lake has an interagency group led by representatives from the FDEP, FWC, SFWMD, and USACE. This group meets every second month to discuss the state of invasive plants and control activities on the lake. The purpose of this group is to coordinate treatments, prioritize activities, and recommend actions for the lake. There are also more than 100 non-native animal species in and around the lake, and there is currently little understanding of their impacts to native species or the ecosystem. No control programs are presently in place to address exotic animal invaders.

Nonindigenous Plants

Floating aquatic plants, such as water hyacinth *(Eichhornia crassipes)* and water lettuce *(Pistia stratiotes)* (**Figure 9-70)** are currently managed by the USACE. The USACE program started in the 1920s with mechanical removal of hyacinth, and it continues today principally with chemical and biocontrol methods. The goal of the program is to keep the plants at a maintenance level as stated under Chapter 369.22, F.S. While hurricanes helped keep infestations low in 2005 through 2006, near-record low water levels kept populations down in 2007. In the past 16 years, the lake has averaged 240 acres (combined) of hyacinth and lettuce, with an average 5,000 acres treated each year. Without continued control of these plants, however, they would quickly expand and have severe environmental impacts on the lake. Even with the current control programs in place, damage to natives occasionally occurs with their displacement and accidental treatment during control. For this reason, and because herbicide treatments control hyacinth quickly but not permanently, well-dispersing biocontrol agents capable of building large populations rapidly are needed. Currently, one potential biocontrol agent is in quarantine, with additional agents from South America set to be studied shortly (P. Tipping, USDA-ARS).

Figure 9-70. Water lettuce (*Pistia stratiotes)* (Photo by Kenneth Langeland, Univ. Florida).

Hydrilla (*Hydrilla verticillata*) has been in Lake Okeechobee for 20 years, but has not been a consistent problem. Its acreage varies annually with water clarity, wind, wave action, water level, and substrate conditions. In some years, hydrilla expanded rapidly to cover thousands of acres and required mechanical harvesting to open up boat trails. Wave and wind from hurricanes are partially responsible for keeping populations of hydrilla low. In 2007, water levels nearing record lows were responsible for keeping infestations small (M. Bodle, SFWMD, personal communication). However, hydrilla's exponential growth rate and new water regulation schedules could allow this plant to be a major concern in the future. Both the USDA-ARS and the

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University of Florida are currently undergoing extensive field explorations in search of more effective biocontrol agents.

Alligator weed *(Alternanthera philoxeroides)* has not been problematic since the 1960s due to successful biocontrol. Presently, three insects: alligatorweed flea beetle (*Agasicles hygrophila*), alligatorweed thrips (*Amynothrips andersoni*), and alligatorweed stem borer (*Vogtia/Arcola malloi)* are all present on the lake and keep populations of alligator weed at low levels. Thousands of acres of alligator weed were treated annually by chemical and mechanical means prior to the introduction of the biocontrols. Barring any situation that would negatively impact the biocontrol agents, alligator weed is not expected to cause any measurable impacts in the near future, and serves as a good example of what successful biocontrol programs can accomplish.

Figure 9-71. Dense population of Brazilian pepper (*Schinus terebinthifolius*) (Photo by Amy Ferriter, Boise State University).

Extensive control programs from 1993 to 2007 have brought three species of exotic trees under virtually complete control in Lake Okeechobee. The most environmentally threatening of these was melaleuca, which had developed significant coverage in the lake's 100,000 acres of emergent marsh. By 1993, large monospecific heads were common, and outlier seedlings were rapidly expanding the tree's coverage. Control efforts, ultimately costing $10 million, have now brought melaleuca under "maintenance control." The release and establishment of the melaleuca biocontrol agents throughout the South Florida region are showing significant effects on large areas of melaleuca.

Two other exotic trees, Australian pine and Brazilian pepper (**Figure 9-71**), had originally established sizeable populations on artificially elevated sites in the lake's watershed including spoil deposits and the lake's levees. In the 1995–2007 timeframe, these trees have essentially been eliminated, primarily through the efforts of the USACE and the District. However, ongoing control and maintenance programs are needed to retain maintenance control levels since no biological controls have yet been released in Florida for the control of either of these two species (although the release of the Brazilian pepper thrips is forthcoming).

West Indian marsh grass (*Hymenachne amplexicaulis*) is a perennial, stout semi-aquatic grass native to Central and South America (**Figure 9-72**). Invading tropical seasonally wet waterways, wetlands, and drainage systems, it impedes flood protection and water management and has overwhelmed riparian systems worldwide. In Lake Okeechobee, it is increasing its range, particularly in Fisheating Bay. Upstream of the lake, in Fisheating Creek, *H. amplexicaulis* has established dense populations along the edge of the creek and in the cypress forest understory. Reproduction occurs by seed germination on moist soils and by aquatic transport of rhizome segments. To date, very little control of West Indian marsh grass has occurred in the lake, and estimates of its population already range to 100 acres (M. Bodle, SFWMD, personal

Figure 9-72. West Indian marsh grass (*Hymenachne amplexicaulis*) (Photo by Univ. Florida IFAS Extension IN491).

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communication). The District initiated an herbicide control program for this species in 2005 within the FDEP aquatic plant control program.

Figure 9-73. Selective control efforts are being used to control torpedograss (Photo by Ann Murray, Univ. Florida).

Torpedograss (*Panicum repens*, **Figure 9-73**) had invaded more than 16,000 acres by 1996. Subsequently, its spread was exacerbated by the lake's record low water level in April 2001. It is estimated that the plant expanded its range to more than 25,000 acres by 2002 (M. Bodle, SFWMD, personal communication). Torpedograss tolerates deep flooding without significant growth or expansion but may spread rapidly and broadly when waters recede. Spread is apparently by vegetative means; floating plant sections serve as propagules, and rhizomes spread broadly from sites of initial establishment. No fertile torpedograss seed production has been found in Lake Okeechobee. Torpedograss has been the target of extensive control in the lake's 100,000-acre western marsh since 1999. More than 29,000 acres of torpedograss were aerially treated in Lake Okeechobee from 2002 through 2007, though some of this acreage consists of infestations treated multiple times. (Treatment effectiveness varies from site to site due to uncontrollable variations in environmental conditions.) Large areas remain to be treated by both aerial and surface applications, however, because funding for the control of this invasive plant often falls short of management program needs. The District continues to treat torpedograss in the lake whenever possible, and wintertime trials show promise for selective treatments that will kill torpedograss and spare dormant native species.

Indian rosewood (*Dalbergia sissoo*) is an invasive tree originally introduced to the Lake Okeechobee Module as an ornamental shade tree at campgrounds and boat ramps (**Figure 9-74**). It has since become a nuisance plant. An intensive chemical and mechanical control program was initiated against this species by the District, and in 2007, the program reached maintenance levels where monitoring and treatment of seedlings are sufficient to keep this plant's population in check.

Figure 9-74. Indian rosewood (*Dalbergia sissoo*) (Photo by Jeff Hutchison, courtesy Archbold Biological Station).

In late July 2006, the first population of Old World climbing fern was reported along the north shore of the lake. This sighting was never successfully confirmed, however. State and federal agencies are actively searching for this species and will enact rapid response tactics if the plant should be discovered. If the species is confirmed present in this module, it will be added to the priority plant list for Lake Okeechobee.

Nonindigenous plant species considered a priority in the Lake Okeechobee Module are listed in **Table 9-10.**

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Table 9-10. Stoplight table for priority plant species in the Lake Okeechobee Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
LAKE OKEECHOBEE MODULE (Results in row reflect module-level questions, not species-level questions)	Y	Restoration efforts under way for while, much progress made; however, several serious species occur in module and continued disturbance of littoral zone may increase chances of new invasions	Y	Module has had large control program under way for many years; progress on many species evident, but continued monitoring and control efforts needed to prevent serious reinvasions of the many species threatening region	Y
Alligator Weed (*Alternanthera philoxeroides*)	G	Effective biocontrol program underway for many years; control programs achieved complete control in most areas	G	Biocontrol and monitoring programs in place and achieving good results	G
Australian Pine (*Casuarina* spp.)	G	Effective removal program in place, not currently a serious problem in this module	G	Chemical control effective; natural areas clear with modest effort; biocontrol research under way	G
Indian rosewood (*Dalbergia sissoo*)		Not new to module but recent addition to priority plant table. Large efforts recently brought population under control	G	Recent control efforts brought population to maintenance levels; only modest effort needed in future to control new seedlings	G
Water Hyacinth (*Eichhornia crassipes*)	Y G	Control programs under way for years; maintenance control goals currently met due to record lows of Lake.	G	Ongoing control and monitoring programs in place; increases in water levels could trigger massive regrowth from seedbank	G Y
Hydrilla (*Hydrilla verticillata*)	Y G	Control programs in place, not necessary in recent years; hurricanes, hydrologic conditions, flocculent substrate prohibit widespread expansion	Y G	Effective control and monitoring programs in place and have been achieving good results; increases in water levels could trigger massive regrowth from seedbank	G Y
West Indian Marsh Grass (*Hymenachne amplexicaulis*)	Y	Little known about spread or distribution throughout system; not included in Indicator systematic monitoring program	Y	Increases in spread/distribution may be occurring; may become serious pest in areas where other exotics have been controlled	Y R
Melaleuca (*Melaleuca quinquenervia*)	G	Effective chemical control program under way for several years with excellent efficacy	G	Chemical and biocontrol effective; spread of agents, new agents expected in 2007/2008	G
Torpedograss (*Panicum repens*)	Y	Impacts at least 20,000 acres of wetlands; static; not included in Indicator systematic monitoring program	Y	Control efforts underway but frequently under-funded; lake management, drawdowns may increase spread despite program	Y R
Water Lettuce (*Pistia stratiotes*)	Y G	Control programs underway for years; maintenance control goals currently met due to record lows of Lake.	G	Ongoing control and monitoring programs in place; increases in water levels could trigger massive regrowth from seedbank	G
Brazilian Pepper (*Schinus terebinthifolius*)		Not new to module but recent addition to priority plant table; effective removal program in place, not currently a serious problem in this module	G	Chemical control effective; natural areas clear with modest effort; biocontrol research underway, new releases 2007/2008	G

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Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources, the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain green status.

Nonindigenous Animals

In addition to the plant species listed in **Table 9-10**, several nonindigenous animal species are considered a priority for the Lake Okeechobee Module. Due to the aquatic nature of this module, fishes are the majority of the problematic nonindigenous animal species within the lake. Besides nonindigenous fish, a variety of non-native reptiles, mammals, and birds inhabit marshes and levees of Lake Okeechobee.

Sailfin Catfish

Since the early 1990s, the Orinoco sailfin catfish (*Pterygoplichthys multiradiatus*) has been observed in the lake (**Figure 9-75**). These numbers are increasing as evidenced by FWC electroshocking surveys and anecdotal evidence from commercial fishermen in the lake that have seen dramatic increases in their catches since the mid-1990s. This fish is suspected to have been introduced by aquarist releases into canals and other water bodies (Hoover et al., 2004). These fish appear to reproduce easily in South Florida and have spread into Lake Okeechobee and throughout the region via the District's extensive canal system. Numerous burrows are found on the lake and the surrounding canal banks, dikes, and levees. Environmental impacts of the sailfin catfish are potentially significant and include displacement of native fishes, mortality of shorebirds, disruption of aquatic food webs, and shoreline erosion (Hoover et al., 2004). In Florida, Orinoco sailfin catfish tunneling is believed to damage canals and levees and result in increased siltation (Hill, 2002; King, 2004).

Figure 9-75. Orinoco sailfin catfish *Pterygoplichthys multiradiata*) (Photo by Leo G. Nico, US Geological Survey).

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Other Nonindigenous Fishes

In addition to the sailfin catfish, there are other fish species of concern in Lake Okeechobee, and these species could have a direct or cumulative impact on the lake ecosystem. Populations of oscar (*Astronotus ocellatu*, **Figure 9-76**), Mayan cichlid, and blue tilapia (*Oreochromis aureus*) have all also increased in the lake. Not enough is known about population dynamics, reproduction, feeding habits, and biology of these species in the lake to determine what impacts they may be having. Largemouth bass (*Micropterus salmoides*) and black crappie (*Pomoxis nigromaculatus*) populations are decreasing on the lake, and their recruitment has been poor for several years (FWC, personal correspondence). Agency fishery biologists have linked extreme fluctuations of Lake Okeechobee water levels and resultant reduced and degraded habitat as having a negative impact on the bass and crappie populations. However, no links between invasive fishes and the declining habitat and falling native fish populations have been studied to date.

Figure 9-76. Oscar (*Astronotus ocellatu*) (Photo by Mac Kobza, SFWMD).

Other Nonindigenous Animals

In addition to nonindigenous fish, Lake Okeechobee has documented populations of many other nonindigenous animals including feral hogs (see *Big Cypress Module* section), green iguanas (see *Florida Keys* and *Greater Everglades Modules* sections), brown anoles, Cuban treefrog, and island applesnails (= channeled applesnails, see *Greater Everglades Modules* and *Kissimmee* sections). Any of these species could have negative impacts on the lake. Feral hogs are omnivores noted for foraging on roots of native trees and impacting native birds. Populations of brown anoles (*Anolis sagrei*, **Figure 9-77**) and Cuban treefrogs (*Osteopilus septentrionalis*) have increased around the lake, and the island applesnail has been documented in Lake Okeechobee. The purple swamphen (see the *Greater Everglades Module* section, page 9-51, for species-specific information) was observed in the marshes around Torry Island during 2005 and 2006. Though it has not been observed in this module recently, the purple swamphen could be a species of concern to the native marsh and wading birds, as it has been noted in other locations to forage on other birds' eggs and on baby birds, including ducklings. Not enough is known about the population dynamics, reproduction, feeding habits, or biology of any of these nonindigenous animal species to make evaluations of their current and future potential impacts to the Lake Okeechobee region.

Figure 9-77. Brown anole (*Anolis sagrei*) (Photo A. Paterson, Williams Baptist College).

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KISSIMMEE BASIN MODULE

The Kissimmee Basin Module includes a diverse group of wetland, aquatic and lake systems. Current initiatives in the Module include the Kissimmee River Restoration Project, Kissimmee River Headwaters Revitilization Project and the Kissimmee Chain of Lakes Long-Term Management Plan.

Nonindigenous Plants

Water hyacinth and water lettuce are the most pervasive nonindigenous aquatic plants in the Kissimmee Basin Module. The District manages these species in the Kissimmee Chain of Lakes (KCOL) and in the Kissimmee River/C-38 portion of the system. Water hyacinth and water lettuce coverage in the KCOL increased significantly during 2006 due to the flushing of plants from adjoining watersheds during fall hurricanes and heavy spring rains, but active control programs are currently keeping these populations static. Increased flow in restored portions of the river provides less suitable conditions for these species, and populations of these floating plants are reduced in about 14 miles of the restored sections of the Kissimmee River channel. However, new open water habitat created by restoration efforts on the re-flooded floodplain seem to provide suitable areas for growth of water hyacinth and water lettuce, at least temporarily.

Hydrilla continues to be a priority nonindigenous aquatic plant species in the lakes of the Kissimmee basin. Hydrilla infestations have covered approximately 52,500 acres in lakes Tohopekaliga, Cypress, Hatchineha, Kissimmee, and Istokpoga and account for more than half of the hydrilla in all of Florida's public waterways. As a result of management efforts and effects of recent hurricanes, including uprooting by winds and persistent turbidity that limits regrowth, hydrilla in the KCOL covered only 6,500 acres during the 2006 and 2007 seasons (M. Bodle, SFWMD personal communication). These are the lowest levels in the last five years. New open water habitat created by restoration efforts on the reflooded floodplain of the Kissimmee River has provided new areas for hydrilla growth. To date, these sites have been flooded only seasonally, so hydrilla's impacts appear to be negligible at this time.

During the past several years, the District has increased herbicide applications to control the potential source of floating plants in the adjacent river channel and downstream canal (C-38). As native wetland plant communities reestablish, the amount of open water and associated coverage of floating exotic plants is expected to decrease. However, given the magnitude of recent required control efforts, it is expected that extensive herbicide treatments of water hyacinth and water lettuce on the reflooded floodplain will be needed for several more years. There is a similar concern for increased coverage of water hyacinth in isolated wetlands within the boundaries of the adjacent Kissimmee Prairie Preserve. Another mat-forming species, Cuban bulrush (*Scirpus cubensis*, **Figure 9-78**), is periodically spot-treated in both the lakes and river/canal system. This species has been eliminated from the restored sections of river channel with restored flow.

Figure 9-78. Cuban bulrush (*Scirpus cubensis*) (Photo by Kerry Dressler, Univ. Florida).

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Although torpedograss and para grass have colonized the backfilled canal and locations where former spoil mounds have been degraded within the Kissimmee River restoration project area, existing growths of these species do not appear to be impacting the recovery of wetland communities on these highly disturbed areas. Both of these species are found on the spoil mounds within the remaining channelized river, and torpedograss is reportedly spreading in disturbed seasonal wetlands on and adjacent to the Lake Wales Ridge. There are currently no active/coordinated control programs in place for these species in the Kissimmee Basin Module. Localized patches (totaling hundreds of acres) of West Indian marsh grass (*Hymenachne amplexicaulis*) have been found on the floodplain in the northern end of the restoration project area but were successfully treated.

Restoration of former wetland communities on the Kissimmee River floodplain appears to be most severely threatened by the establishment and continuing spread of limpograss (*Hemarthria altissima*). Limpograss is an introduced forage grass that has invaded the floodplain from adjacent upland pastures and is thriving in the hydrologic regimes provided by the restoration project (**Figure 9-79**). It presently forms monospecific stands covering approximately 2,000 acres of the east-central portion of the reflooded floodplain and is spreading to the north and west. Initial limpograss chemical control test plots were established in the Kissimmee River floodplain in 2006 to help define best management practices. Although no active control efforts have take place thus far, funding is available from the FDEP for future operation control work. The first coordinated chemical control effort occurred in June of 2007.

Figure 9-79. Limpograss *(Hemarthria altissima)* has invaded the Kissimmee floodplain from adjacent pastures (Photo by B. Cook, DPI & F Australia).

Chinese tallow (*Sapium sebiferum*) is a serious invader of wetlands in this region (**Figure 9-80**). Dense stands are able to develop rapidly because wildlife transport abundant seeds quickly and over long distances. Shallow marshes, lake edges, swales, and riparian sites develop dense impenetrable monocultures. No biocontrol is currently available, though field explorations for suitable biocontrol agents have recently commenced. This species has been of agricultural importance in China for a very long time. Consequently, pests (potential biocontrol agents) have been thoroughly documented, making the agent selection process more efficient.

Figure 9-80. Chinese tallow *(Sapium sebiferum)* (Photo by Cheryl McCormick, Univ. Florida).

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There are already numerous species showing promise as excellent biocontrol agents. Chemical control is readily achieved against Chinese tallow, but no systematic control has begun.

Figure 9-81. Cogon grass (*Imperata cylindrica*) (Photo by Wilson Faircloth, USDA-ARS).

Archbold Biological Station staff indicated that natal grass (*Rhynchelytrum repens*) and cogon grass (*Imperata cylindrica*) are continuing to spread throughout the region, particularly in disturbed upland habitats (**Figure 9-81)**. Cogon grass is presently the exotic species of greatest concern on Kissimmee Prairie Preserve, where it is increasing on leased cattle pastures and along roads. Cogon grass is also commonly found on the spoil mounds of channelized river.

Old World climbing fern is the primary nonindigenous plant species of concern in riparian and upland habitats in the Kissimmee valley. Control efforts on the Kissimmee River floodplain have involved aerial and ground treatments, and have been successful in reducing cover density of Old World climbing fern on a localized scale. This includes the *Lygodium* within the mesophytic shrub community in the lower portion of the restoration project area, where regrowth following several annual aerial herbicides applications appears to have been inhibited by prolonged inundation. Similarly, because of intensive control efforts, cover of Old World climbing fern has decreased on the Avon Park Air Force Range. The reduction/thinning of tree and shrub canopy by the 2004 hurricanes increased the visibility of lygodium cover during aerial surveys and facilitated more thorough treatments of observed distributions of this species in the Kissimmee basin. Still, this plant currently occurs in multiple habitats with varying land ownership (public and private). Consequently, control efforts have been difficult to coordinate, leading to its present rate of spread.

Though not as widely distributed as Old World climbing fern, a Japanese climbing fern (*L. japonicum*) population has spread from the lower end of Pool D into Pool E of the channelized Kissimmee River. Japanese climbing fern has also been found on Avon Park Air Force Range, where staff has expressed concern about the effectiveness of available herbicides for this species.

Tropical soda apple (*Solanum viarum*) is another pervasive exotic species of concern in the pastures of the Kissimmee valley (**Figure 9-82**). Cover of this species is reportedly increasing on private lands neighboring Avon Park Air Force Range. Chemical and mechanical control efforts put forth against this species have had limited effect. The biocontrol program has resulted in the release of one agent to date (*Gratiana boliviana*) with three additional species expected to be released by late summer 2007 by FLDACS-DPI. Other exotic plants that have been locally treated in the module include strawberry guava (*Psidium littorale*), caesarweed (*Urena lobata*), and star grass (*Cynodon nlemfuensis*).

Figure 9-82. Soda apple *(Solanum viarum)* (Photo by J. Jeffrey Mullahey, Univ. Florida).

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Additional exotic vines of concern in upland tree and/or shrub habitats in the valley include air potato (*Dioscorea bulbifera*), rosary pea (*Abrus precatorius*), and flame vine (*Pyrostegia venusta*), which have been observed by staff at Archbold Biological Station to spread aggressively after initial establishment. Herbicide treatments have decreased the population of air potato in Pools D and E of the channelized river. However, this species is reportedly spreading along the Lake Wales Ridge. An active biological control program against air potato is in the stages of field exploration, with numerous promising species resulting from these efforts (R. Pemberton, USDA-ARS, personal communication).

The somewhat scattered Brazilian pepper and melaleuca infestations are generally targeted for control by the module's natural resource managers. Brazilian pepper has been largely eliminated by inundation within the reflooded portion of the Kissimmee River floodplain, and melaleuca appears to be decreasing due to control efforts by Highlands County and local lakeshore development activities.

Figure 9-83. Wright's nutrush (*Scleria lacustris*) (Photo by Vic Ramey, Univ. Florida).

Wright's nutrush (*Scleria lacustris*) is a sedge that was first reported in Florida in 1988 (**Figure 9-83**). Freshwater marshes and lake shorelines with seasonal water fluctuations are highly susceptible to invasion by this plant, which disperses its nutlets via birds, airboats, and water transport through drainage systems. Although this plant is not new to the Kissimmee Basin Module, recent increases in Wright's nutrush populations warrant its addition to the priority plant list. This plant currently occurs in multiple habitats with varying land ownership (public and private). Consequently, control efforts against Wright's nutrush have been disjointed and difficult to coordinate, leading to its present rate of spread.

Nonindigenous plant species considered a priority in the Kissimmee Basin Module are listed in **Table 9-11**.

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Table 9-11. Stoplight table for priority plant species in the Kissimmee Basin Module.

	2006 STATUS	2007 STATUS		1-2 YEAR PROGNOSIS	
KISSIMMEE MODULE (Results in this row reflect only module-level questions, not species-level questions)	Y R	Many very serious nonindigenous species occur in this region for which little is known about how invasive they may become; restoration efforts underway in this module for many years, much progress made; new programs started	Y R	Many of the species occur only in this region and little is known about their biology, yet some are very serious weeds in other parts of world; rehydrated wetlands providing new habitat for aquatic species including hydrilla; many new control programs started	Y R
Water Hyacinth (*Eichhornia crassipes*)	Y G	Significant control efforts underway for many years; control programs achieving good results	Y G	Systematic control and monitoring programs in place and achieving good results	G
Limpograss (*Hemarthria altissima*)	Y R	Little known about spread or distribution; increasing in scope; included in FDEP aquatic plant surveys; new chemical program initiated	Y R	No biocontrol effort underway; new funding and chemical control program may be bring populations to maintenance level	Y R
Hydrilla (*Hydrilla verticillata*)	Y G	Limited control efforts and biocontrol programs under way for many years; control programs have mixed results; storms and water levels currently having most impact	Y G	Systematic control and monitoring programs in place and achieving good results; recent herbicide resistance creating new control problems along with increased habitat on rehydrated floodplain	G Y
West Indian Marsh Grass (*Hymenachne amplexicaulis*)	Y	Little known about spread or distribution throughout the system; included in FDEP aquatic plant surveys; control efforts increasing	Y G	Control efforts in this module good and increasing; most populations in natural areas under reasonable control	Y G
Cogon Grass (*Imperata cylindrica*)	Y	Little known about spread or distribution; not in Indicator systematic monitoring program	Y	Controlled to varying degrees on public lands in this module; no biocontrol effort under way	Y R
Old World Climbing Fern (*Lygodium microphyllum*)	R	Serious invader with rapid spread throughout module; invades most habitats and very destructive; active biocontrol program but current agent effectiveness not yet seen	R	Chemical control has brought populations to maintenance levels on public land; biocontrol releases made, more expected in 2007; chemical studies continuing	R Y
Japanese Climbing Fern (*Lygodium japonicum*)	Y R	Controlled thus far, but little known about potential impacts in module	Y R	Populations controlled so far; however, distribution and spread unknown; no biocontrol program underway	Y R
Melaleuca (*Melaleuca quinquenervia*)	Y G	Still abundant on private lands but biocontrol reducing cover and spread	Y G	Chemical control effective on most public lands; biocontrol agents effective; additional spread & introductions expected in 2006	G
Torpedograss (*Panicum repens*)	Y	Little known about spread or distribution but believed to be increasing; included in FDEP aquatic plant surveys	Y R	No significant control efforts or effectiveness; no biocontrol effort under way although local populations can be eliminated	Y R
Water Lettuce (*Pistia stratiotes*)	Y G	Significant control efforts and biocontrol programs underway for several years; control programs achieving good results; included in FDEP aquatic plant surveys	Y G	Systematic control and monitoring programs in place and achieving good results	G
Chinese Tallow (*Sapium sebiferum*)	Y	Distributed along many lake edges in Kissimmee Chain of Lakes; not included in Indicator systematic monitoring program; populations increasing	Y R	No significant control efforts or effectiveness; no biocontrol effort underway although local populations can be eliminated	Y R

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Table 9-11. Continued.

Brazilian Pepper (*Schinus terebinthifolius*)	Y	Serious invader, invades most habitats, very destructive; chemical control ineffective in reducing systemwide spread so far; however, local control programs proving effective where resources available	Y	Control programs effective in natural areas where management programs under way; new biocontrol agents under study for future release	Y G
Wright's Nutrush (Scleria lacustris)		Not new to module but recent addition to priority plant table; not currently serious problem but uncoordinated control efforts leave this plant free for future expansion	Y R	Without coordinated control efforts in near future, population will continue to expand unabated	R
Tropical Soda Apple (*Solanum viarum*)	Y	Little known about spread or distribution; biological control agents released with more on way; not included in Indicator systematic monitoring program	Y	Control efforts limited, although local populations can be eliminated; additional biocontrol agents to be released in 2007	Y G

Red = Severe Negative Condition, or one is expected in near future, with out-of-control situation that merits serious attention.

Yellow/Red = Problem was previously localized or not too severe but is or appears to be progressing toward a Severe Negative Condition generally due to inaction. Without attention and resources, the situation may develop or become red.

Red/Yellow = Currently a Negative Condition but there are reasonable control efforts underway. However, without continued or improved efforts this species may revert to a severe situation or become a future serious invader and revert to yellow/red or red.

Yellow = Situation is improving due to reasonable control program and either is stable or moving toward stabilizing, or the species is still very localized but is expected to spread if sufficient resources or actions are not continued or provided. The situation could still reverse.

Green/Yellow = Situation is generally good and under control but still needs regular, even if low-level, attention to continue progress to yellow/green or green.

Yellow/Green = Significant progress is being made and situation is moving toward good maintenance control and is expected to continue improving as long as resources are maintained.

Green = Situation is under control and has remained under control for several years, particularly where biocontrol is found to be effective. Where chemical maintenance control is in place, continuation of control efforts is essential to maintain green status.

Nonindigenous Animals

Several nonindigenous animal species are considered priorities for the Kissimmee Module. The feral hog is the most ubiquitous exotic animal of concern for potential impacts to natural habitats in the Kissimmee valley (see the *Big Cypress Module* section). Although the current population of feral hogs within the Avon Park Air Force Range is reportedly lower than previous years, the population is apparently increasing on Kissimmee Prairie Preserve and is of major concern for impacts to the dry prairie habitat. Current levels of hunting and trapping have not had any significant effect on feral hog populations despite the lack of a daily limit in most regions, so an increase in the length of the hunting season has been proposed to attempt to reduce the abundance of this species.

Similarly, although the population of Asian clam (*Corbicula fluminea*) has increased in the section of Kissimmee River channel with restored flow, its potential threat to reestablishment of native invertebrate fauna has not been determined. Avon Park staff has expressed concern about potential impacts of the broadly distributed populations of walking catfish (*Clarias batrachus*) in aquatic habitats, and Kissimmee Prairie staff is alarmed about increasing populations of European starlings (*Sturnus vulgaris*). White winged doves (*Zenaida asiatica*) appear to be locally common in at least Highlands County and have been observed roosting in large numbers in upland habitats adjacent to the Kissimmee River. Nile monitors, too, are appearing in this module. Ryan Higgins (SFWMD) has repeatedly seen a greenish-gold spotted 4-foot lizard on the banks of Shingle Creek, upstream of Lake Tohopekaliga in Osceola County. Numerous reports have also come from local residents in recent years.

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Fishes

Extensive fish sampling was conducted throughout this module to provide current records about nonindigenous fish distribution in the Kissimmee River and floodplains. The brown hoplo (*Hoplosternum littorale*) is an armored catfish that occurs in abundance within the river and some floodplain pools. This species has achieved a nearly cosmopolitan distribution throughout the fresh and saltwater habitats of mid- to southern Florida. It is both an aquarium and food fish, often released and harvested as a cultural food source. The vermiculated sailfin catfish (*Pterygoplichthys disjunctivus*) and Orinoco sailfin catfish (*Pterygoplichthys multiradiatus)* are also common in this module. These are very popular aquarium fish, commonly called “algae eaters.” They are some of the most resilient exotic species in Florida. Although little is known about their habitat preferences, thick scales, venomous spines, and the abilities to breathe air and use teeth to scrape algae for nutrition make them adaptive and problematic. The Kissimmee River represents the northern range limit for many exotic tropical fishes.

Island Apple Snail

Recent taxonomic work (Tim Collins, Florida International University) indicates that the nonindigenous species previously known as the channeled apple snail (*Pomacea canaliculata*) was incorrectly named and is in all actuality the island apple snail (*Pomacea insularum*). The biology, distribution, and impact of this species remain the same; only the taxonomy has changed.

The island apple snail is a large (up to 10 cm) South American freshwater mollusk established in California, Texas, and Florida through the aquarium trade (**Figure 9-83**). This species has been nominated as one of the “100 World’s Worst Invaders”. Since its establishment in Southeast Asia and Hawaii in the 1980s, it has become the number one rice and taro pest, causing large economic losses. It has also been implicated in the decline of native apple snails in Southeast Asia. Likely impacts in Florida include destruction of native aquatic vegetation and serious habitat modification in addition to competition with native aquatic fauna. The continued spread of the island applesnail may be a problem for the endangered everglades kite, in particular, if it outcompetes the native applesnail, *P. paludosa*, which is the primary food of the everglades kite. The snail serves as a vector for disease and parasites. Spread has commonly occurred as intentional introductions to wetlands, as discards from aquaria or, as reported in Asia, as releases to establish a food crop.

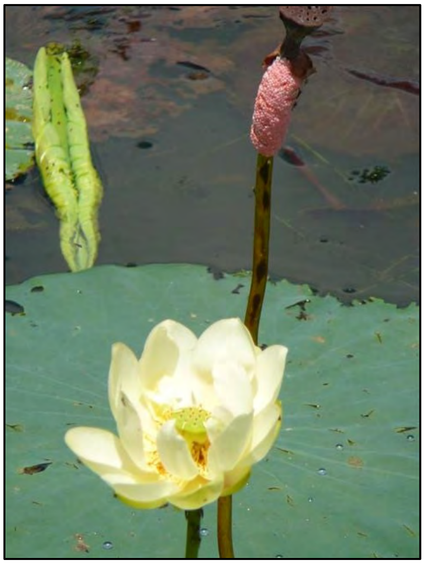

Figure 9-83. Island apple snail egg mass (*Pomacea canaliculata*) (Photo by SFWMD).

In the KCOL, the island apple snail is now common in northern Lake Tohopekaliga and particularly in the lake’s northeastern Gobblett’s Cove. The USFWS has contracted for snail populations to be monitored in the future, although little work has been done to outline a control strategy for this nonindigenous species. Studies conducted to date by the University of Florida suggest that any molluscicide that will be toxic to the island applesnail will also be toxic to the native applesnail. The only possibility for differential control between the two snails would be to apply toxicants directly to the easily recognized bright pink exotic apple snail eggs (W. Haller, Univ. Florida, personal communication), which a District employee is currently doing.

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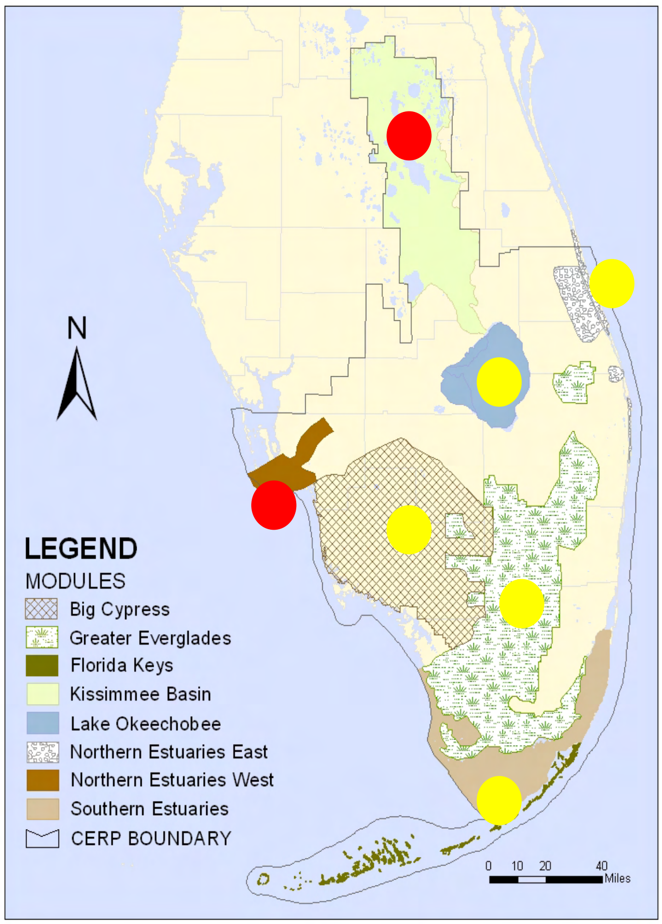
MODULE SUMMARY

For nonindiginouse plants, most modules have some level of control program for high priority species and are showing progress with commonly known and wide spread species such as melaleuca, particularly on public lands (Figure 9-84). Biocontrol efforts are proving successful against Melaleuca, and many other biocontrol agents are being released against other species. However, even Brazilian pepper and Old World climbing fern continue to be serious invaders in many modules, and several new and recently introduced species are being identified in many modules with little information in existence on distribution or control methods.



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

Yellow – Current situation does not meet restoration targets and merits attention
.

Green – Situation is good and restoration goals or trends have been reached. Continuation of management and monitoring effort is essential to maintain and be able to assess “green” status.

Figure 9-84. Overall module status in relation to exotic plants.

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All of the modules have significant invasive exotic plant problems that are documented to be affecting natural areas and altering natural habitats and processes. Monitoring programs to assess the trends in invasive exotic plants only cover the entire restoration area for 6 high priority species, and monitoring that would identify new species or new distributions for existing species only covers portions of the Greater Everglades module; the other modules are not being monitored.

Key Recommendations

1. Existing monitoring programs need to be expanded to cover remaining modules in order to be able to determine where and when new species arrive (then establish) and assess success of control programs in these areas.
2. In order to get ahead of the exotic plant invasion rate, control programs (chemical and biological) need to expansion; the many agencies undertaking these programs need to develop formal strategic agreements regarding implementation and fiscal planning.
3. Effective preemptive monitoring is required at ports of entry to identify and assess new species and their invasion potential, and detect these species prior to their establishment in natural areas.
4. Risk assessments tools need to be formally accepted by the agencies and used to assess the invasion potential of the many exotic plant species in order to help prioritize resources and control programs.

Exotic animal trends by module differ from those of nonindigenous plants in that module-wide control efforts rarely exist. As stated throughout this document, the ubiquitous nature of animals makes large-scale monitoring and control efforts extremely difficult. The lack of baseline monitoring data for many nonindigenous animals makes tracking progress impossible. Still, select control efforts against some species have been aggressive and appear to be keeping them in check, e.g. the cactus moth (*Cactoblastis cactorum*) and purple swamphen (*Porphyrio porphyrio*). Key recommendations for exotic animal management are more basic than those listed for nonindigenous plants. It is crucial that consistent monitoring programs and risk assessment tools first be developed for nonindigenous animals.

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SUMMARY OF NEEDS AND GAPS

The elements of a comprehensive nonindigenous plant management program — legislation, coordination, planning, research, education, training, and funding — have been in place in Florida for many years. The majority of plants identified in this document as priority species are all being controlled on public lands by local, state, or federal agencies. Unfortunately, the same cannot be said for animals, and there are hundreds of nonindigenous organisms in South Florida with unknown distributions and invasive potentials. The threat of nonindigenous animals is becoming an important ecological and restoration issue for many agencies in Florida, and certain species are beginning to be addressed. Funding and coordination for a comprehensive nonindigenous animal management plan for Florida are needed. There is also a need to set priorities for animal management in South Florida; this task is being undertaken on two fronts. First, the Everglades Cooperative Invasive Species Management Area (CISMA) is developing a proposal to evaluate the use of existing risk assessment tools for South Florida's nonindigenous animal species. Second, the South Florida Water Management District is developing a comprehensive literature review for those plant and animal species identified as most critical for restoration. The sheer number of nonindigenous animals is overwhelming, and agencies charged with managing natural systems have a responsibility to understand the distribution and impacts of these species and either initiate control operations or accept their occurrence and consequences in natural areas.

Resource managers charged with controlling nonindigenous plants in Florida have recognized for almost a decade that single-species management is not effective. The control of one plant species often leads to reinvasion by another nonindigenous plant. Similarly, the time has come to consider that single-taxa management is not an effective long-term strategy. Melaleuca serves as a preferred host for lobate lac scale. The remaining large populations of melaleuca in South Florida harbor large populations of lobate lac scale, effectively serving as a reservoir for this nonindigenous insect species. An integrated management approach is needed for these species where interactions between and among nonindigenous species are a factor. It is also important for agencies to consider ways in which the public can be encouraged to identify, monitor and manage nonindigenous plant and animal species on privately held lands.

Given the documented impacts of nonindigenous organisms in South Florida, scientists are obliged to begin to factor these species into restoration models, and research is needed to understand the distribution, biology, and impacts of these nonindigenous organisms (**Table 9-12**). Controlling and managing nonindigenous organisms in an all-taxa approach is a nascent idea, even among ecologists, but it is sure to emerge as an important field of science given global trade and the virtual "open barn" situation. Organisms will continue to arrive and will continue to establish breeding populations in new environments, including South Florida. The abundance of nonindigenous plants in the region may be accelerating this process, as animals are arriving not only without their natural enemies but also into a hospitable environment that includes plant species from their native range. It may be no coincidence that the Burmese python is common along canal levees covered with Burma reed.

Irrespective of taxa, the process an invasive species goes through from introduction to establishment to invasion to ecosystem engineer is complex, involves many environmental factors, and may take many decades to complete. Relatively few exotic species become invasive in *de novo* environments, but a very few species can wreak major economic and ecologic havoc. Species that appear benign for many years or even decades can suddenly spread rapidly following events such as flood, fire, drought, hurricane, long-term commercial availability, or other factors. Resource managers must recognize these species during the early incipient phase in order to maximize available operational resources. As part of this effort, there is a need to establish an "applied monitoring" program and a project tracking system for nonindigenous plant and animal species before their introduction (to try to prevent introduction or to be better prepared for eradication efforts).

Species like the purple swamphen in the Greater Everglades and the Gambian pouch rat in the Keys illustrate the need for state and federal agencies to act quickly to contain and attempt to eradicate animals that have the potential to become widespread and difficult to control. Recent additions to non-native wildlife rules (now housed in the new Chapter 68-5 under Title 68) increase the scope of existing rules (limiting the trade of the red-eared slider for example). However, many more restrictions are called for to adequately curb the purposeful and accidental release of non-indigenous animals into the South Florida environment. While it is acknowledged that definitive research is lacking to support the immediate management of these particular species, it is widely accepted in the invasive species literature that catching a species in its incipient phase is advantageous, even where research may be inadequate or lacking. This is one of the most important reasons to develop a biological risk assessment "tool box" for exotic species in order to help discern which species are most likely to become invasive both prior to introduction and during the earliest phases of their establishment when eradication is feasible.

The use of an early detection and rapid response (EDRR) program increases the likelihood that invasions will be controlled while the species is still localized and population levels are so low that eradication is possible (National Invasive Species Council, 2003). Once populations of an invasive species are widely established, eradication becomes virtually impossible and perpetual control is the only option. In addition, implementing EDRR programs is typically much less expensive than a long-term invasive species management program. Given the risks associated with waiting for research and long-term monitoring to "catch up," some agencies have opted to initiate control programs concurrently with biological or ecological research programs. Biological risk assessments are being developed (particularly for plants) to allow agencies to determine which species are most likely to become problems. Many states struggle with how to implement an EDRR approach because awareness and funding often lag, preventing a real "rapid" response. For South Florida, groups such as NEWTT and FIATT are attempting to initiate EDRR efforts. Species chosen by FIATT as EDRR candidates are noted in **Table 9-2** and include organisms such as the red palm mite (*Raoiella indica*) and redbay ambrosia beetle (*Xyleborus glabratus*), both of which do not currently occur in South Florida but present extreme risks if they establish.

The District's Strategic Plan provides the agency and the public it serves with a blueprint for meeting the challenges of balancing the needs of the natural environment with the demands of Florida's growing population and important agricultural industries. Control of nonindigenous species are cited as important strategies and success indicators in the District Strategic Plan. Exotic species treatment is specifically listed as a deliverable in five of the 11 overall Strategic Plan Goals. Successful management of these species is also tangentially key to many of the other Strategic Plan Goals as nonindigenous species impact everything from evaluating Environmental Resource permits to operating Stormwater Treatment Areas to restoring natural fire regimes.

Priority plant species are listed within each Module summary in this Chapter. Animal species have not been prioritized in a similar manner. Given differing agency priorities and responsibilities, a definitive "priority animal list" may be years from being developed and accepted by resource mananegment agencies in Florida. Given the District's mission, the following list is a summary of animal species which threaten the success the District's Strategic Plan Goals.. These animal species are presented with a "District-centric"justification for listing, and it should be noted that priorities may differ for other agencies, depending on regional factors and agency priorities and goals.

District Priority Animal Species

1. Burmese python (*Python molarus bivittatus*)
    - As a top predator, threatens to disrupt entire food chain and ecosystem function within the Everglades
2. Feral hog (*Sus scrofa*)
    - Disrupts both plant and animal communities
    - Rooting behavior alters land management, increasing soil disruption and erosion
3. Bromeliad weevil (*Metamasius callizona*)
    - Directly threatens native bromeliad populations, many of which are threatened.
    - Removal of native epiphytes disrupts ecosystem function
4. Lac scale (*Paratachardina lobata* or different species, this is being studied by USDA-ARS)
    - Attacks numerous native tree and shrub species, threatening District Everglades tree island restoration
5. Green iguana (*Iguana iguana*)
    - Burrowing undermines and weakens infrastructure of canal banks, threatening District operation and maintenance infrastructure.
6. Purple Swamphen (*Porphyrio porphyrio*)
    - Disrupts wading bird communities, potentially impacting District Everglades restoration
7. Swamp eel (*Monopterus albus*)
    - Predators, may impact animal communities
    - Mainly in canal system, proximity to restoration efforts is a concern.
8. Island applesnail (*Pomacea insularum*)
    - Disrupts wetland communities
    - Threatens ecology of Everglades system
9. Sailfin catfish (*Pterygoplichthys spp.*)
    - May alter/decrease aquatic community function
    - Burrows into canals and levees, potentially impacting infrastructure
10. Monitor lizard (*Varanus niloticus*)
    - Predator, impacts bird, amphibian and reptile populations.
    - Potential for spread and to impact restotation activities

An overarching theme in this document is describing the alarming extent and impacts of some exotic species infestations and stating the need for increased control efforts. While these observations are entirely true and warrant more attention, it should be noted that past control

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efforts against certain nonindigenous species have proven successful and demonstrate that control of some species is possible. For instance, melaleuca is now under maintenance control on Lake Okeechobee and in several WCAs. It is tempting to assume that when CERP restoration goals are achieved, results will include a reduced need to control nonindigenous plants and animals. Although it is true that the spread of some invasive species may be reduced in some locations by increasing hydroperiods (e.g. Brazilian pepper), there has been little or no research to determine what effects long-range hydrologic changes or nutrient reductions or alterations will have on nonindigenous species throughout the system. Nutrient enrichment studies have evaluated changes to native flora but have virtually excluded the study of invasive species. The Mexican bromeliad weevil, lobate lac scale, Old World climbing fern, and Brazilian pepper have successfully invaded areas with few apparent human alterations, including the mangrove zones of Southwest Florida and remote areas of Big Cypress and ENP. A more comprehensive approach must be taken when looking at the long-term restoration process with regard to the nonindigenous species composition response. It is also necessary to stress to the public and policy makers that nonindigenous species will always require some level of maintenance and that new introductions and expected arrivals (such as the red palm mite) must be recognized and prevented early in order to avoid future costs.

Public awareness of invasive species and their impacts to Florida's natural resources is an important component of a successful prevention and management program. Promoting behavioral changes of individuals and industries can help curtail the introduction of potentially invasive species. The Florida Exotic Pest Plant Council was successful in working with the Florida Nursery Growers and Landscape Association to discourage the use and sale of known invasive plant species, and in 2006, the Lowe's chain of home-improvement stores agreed not to sell certain invasive plants in their Florida stores.

Table 9-12. Top five research gaps as identified by a consensus of the managers and scientists involved in South Florida invasive species management and control for South Florida restoration.

1. Develop control strategies and techniques, including control and monitoring methods and approaches, to control and manage invasive exotic, aquatic animals
2. Develop control strategies and techniques including control and monitoring methods and approaches to prevent the transfer and spread of invasive exotic organisms between wetland waterbodies in the Everglades via water management structures or operations. Also develop a detailed review and synthesize information on existing technologies and strategies used in other areas.
3. Identify research gaps for high priority species in order to develop better information about biology of different organisms and development of practical management practices.
4. Develop a biological risk assessment tool for helping prioritize new animal species for control, and management (Contact: J.T. Hillary. C.) • Includes fluctuating populations of animals over time • "Filters/Risk Assessments" for prioritizing species for control and management o Literature review of animal groups, lifeforms, species as to patterns of invasiveness ▪ Prediction models for determining invasiveness ▪ "Coarse" assessments • Evaluate "New Zealand" risk assessment tool for invasive animals • Begin with fish and reptiles
5. Development a integrated strategy and conceptual approach to guide the development of monitoring programs for individual animal groups, life forms or species to better coordinate and integrate monitoring data and sampling approaches.

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