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

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.

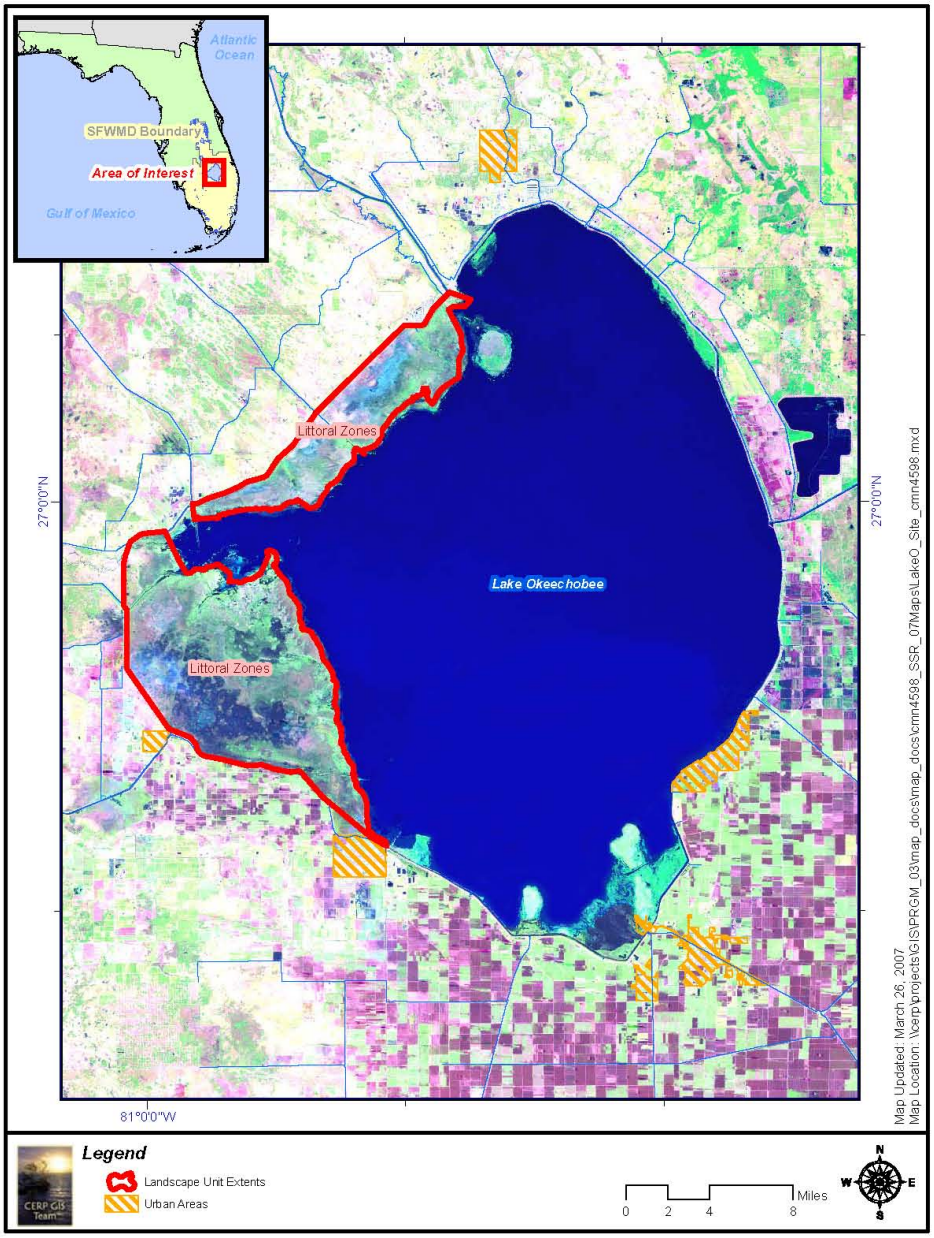


Figure 4-1: Lake Okeechobee

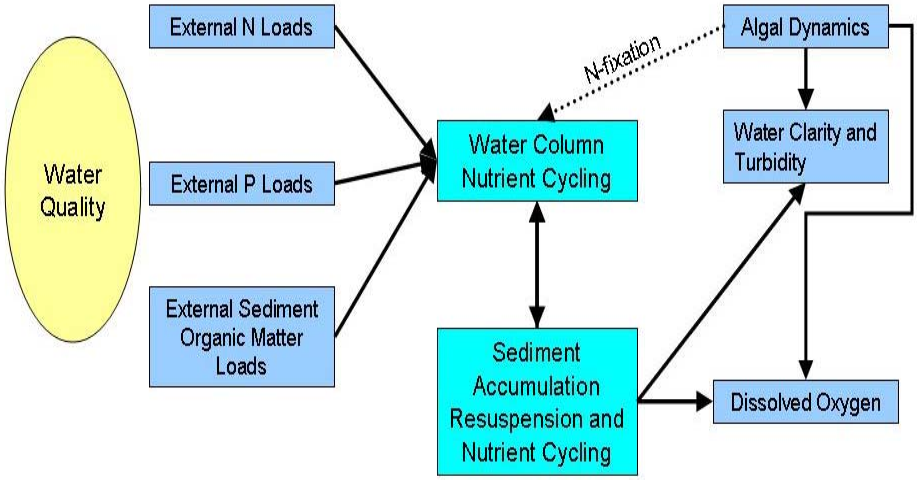
4.2 Lake Okeechobee Hypothesis Cluster–Water Quality

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.



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*

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***).

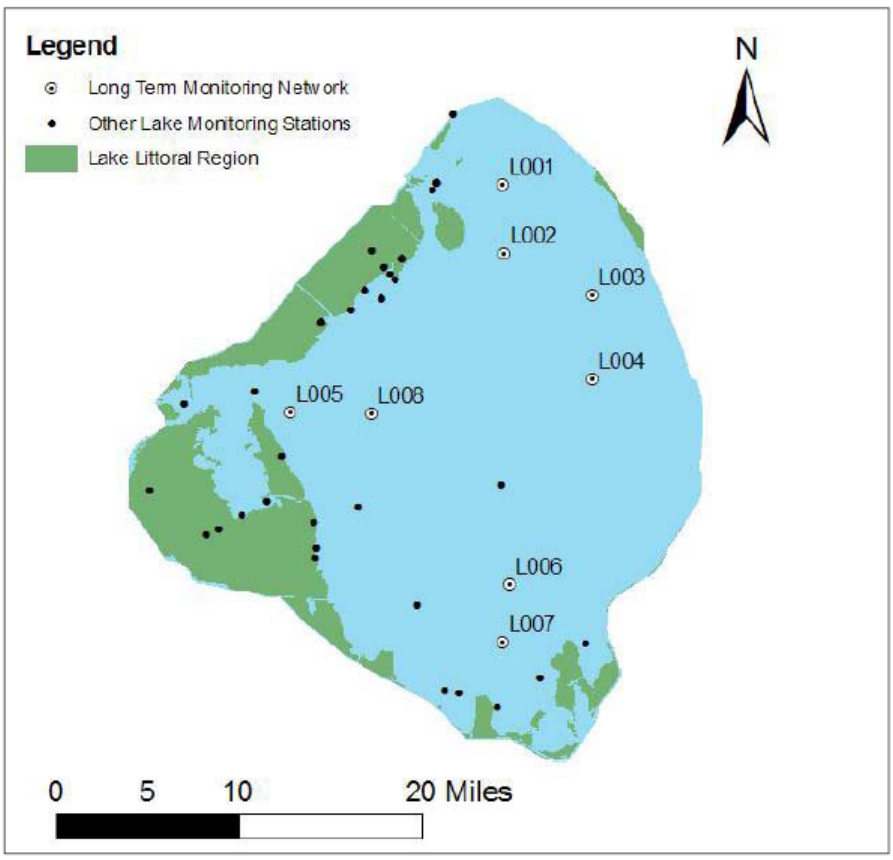
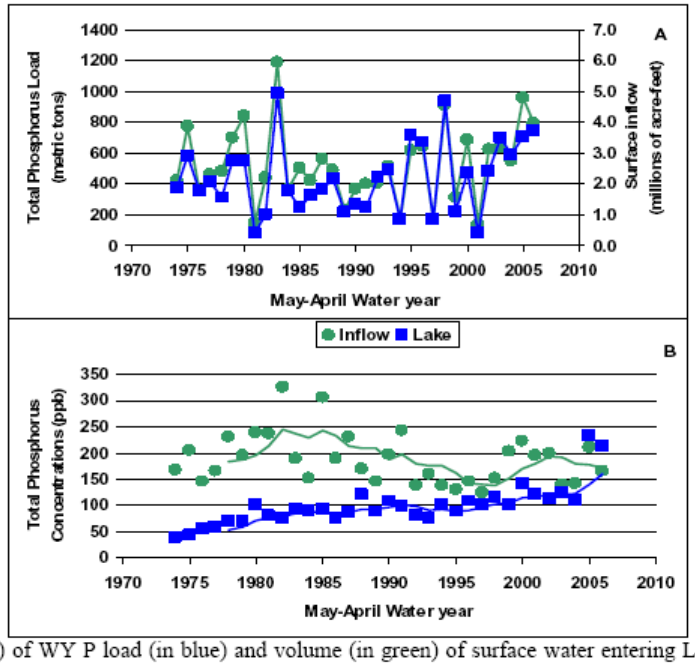
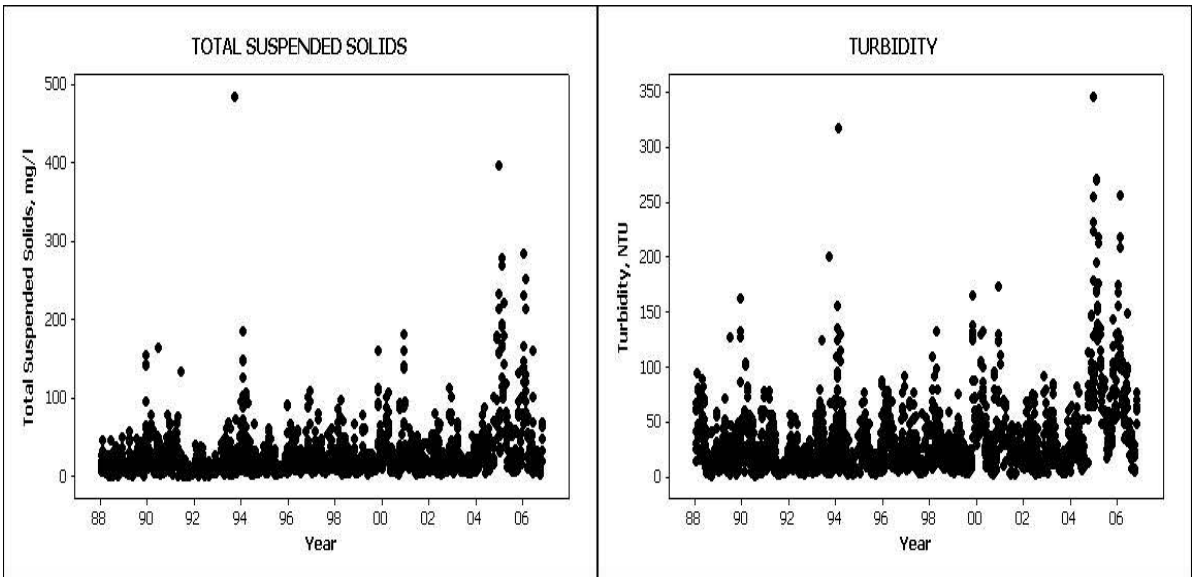


Figure 4-3: Locations of Eight Surface Water Monitoring Stations



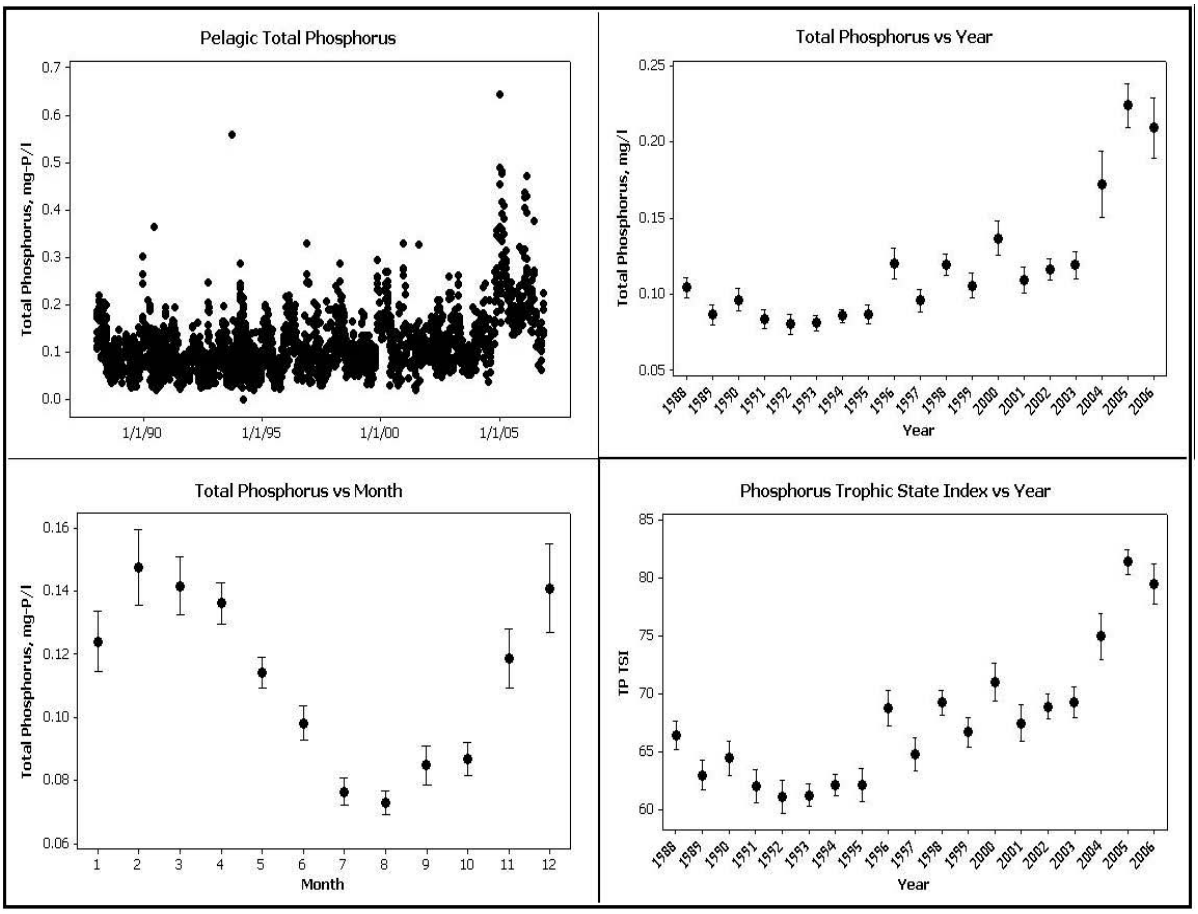
Timelines (top graphic) of WY P load (in blue) and volume (in green) of surface water entering LO, and five-year moving average (bottom graphic) of inflow and in-lake TP concentrations (from 2007 SFER). Note decrease in inflow concentration from 1982 through 1997 and apparent rebound in some years thereafter.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

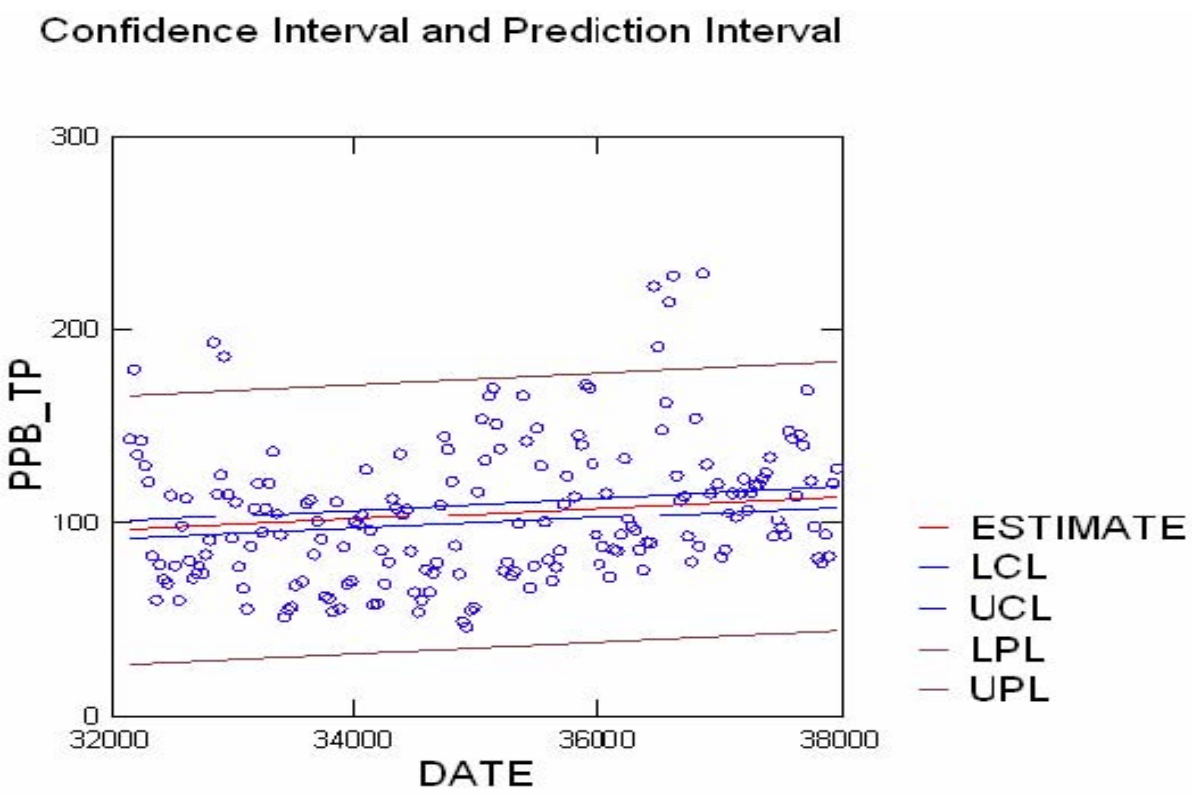
The passage of the 2004 and 2005 hurricanes is readily apparent in the amount of unconsolidated sediment resuspended in the water column (***Figure 4-5***), despite the fact that samples were not taken during, or immediately following, the storms. It may be that the quantity of suspended sediment in 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.



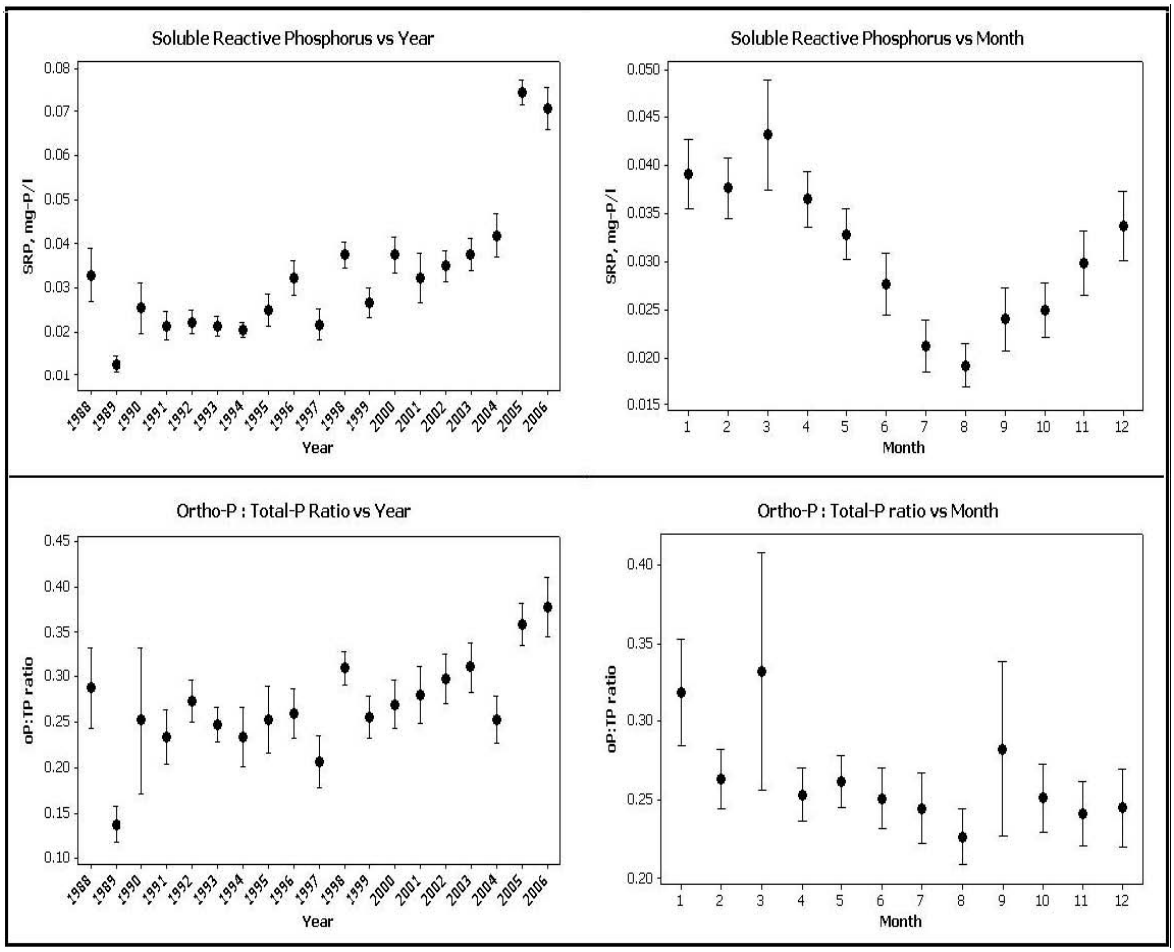
Depicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top right), by month (lower left), and trophic state index for phosphorus by year.

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

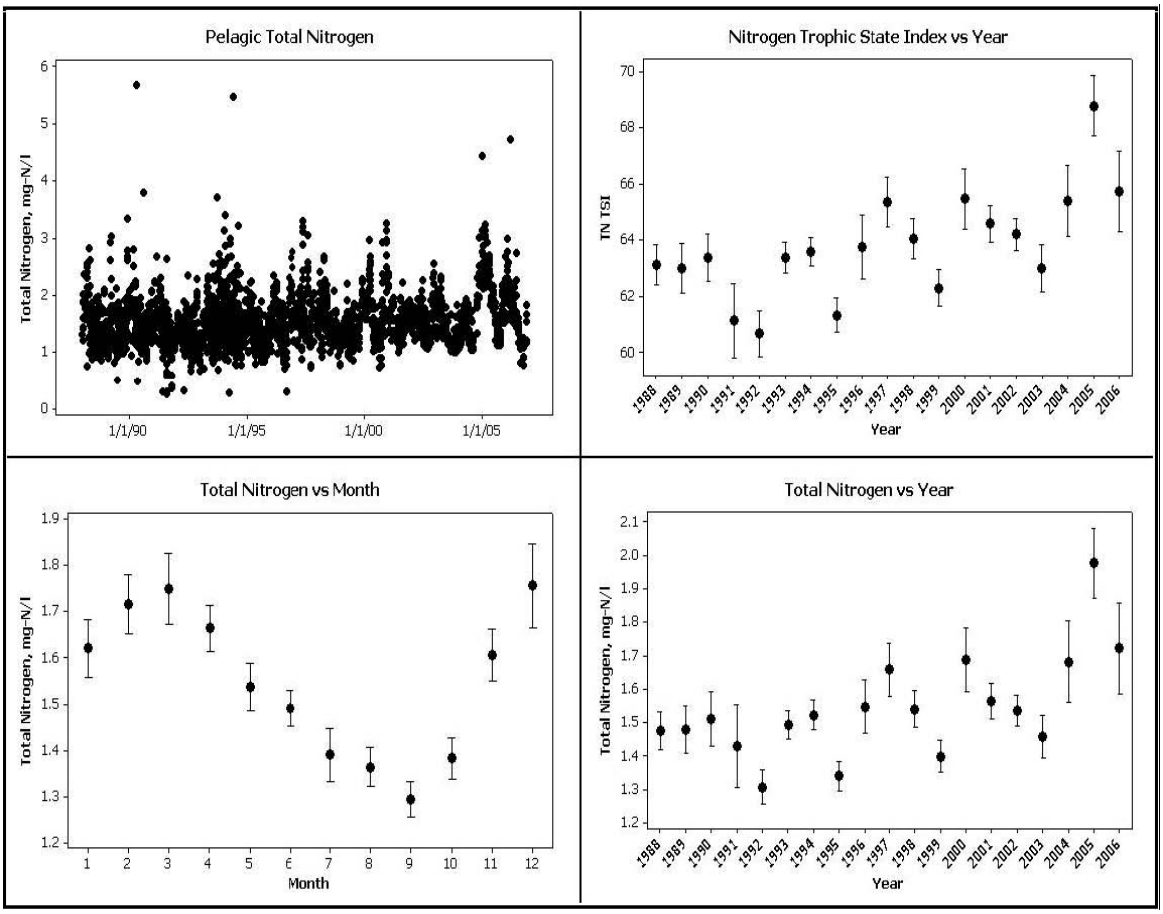
large fraction of the P reported as TP is in the particulate form (TP analyses are performed on unfiltered samples); however, the increases in resuspended sediments were also associated with increases in soluble P (P<0.005). The similar but magnified effect of the 2004 and 2005 hurricanes can be seen 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.



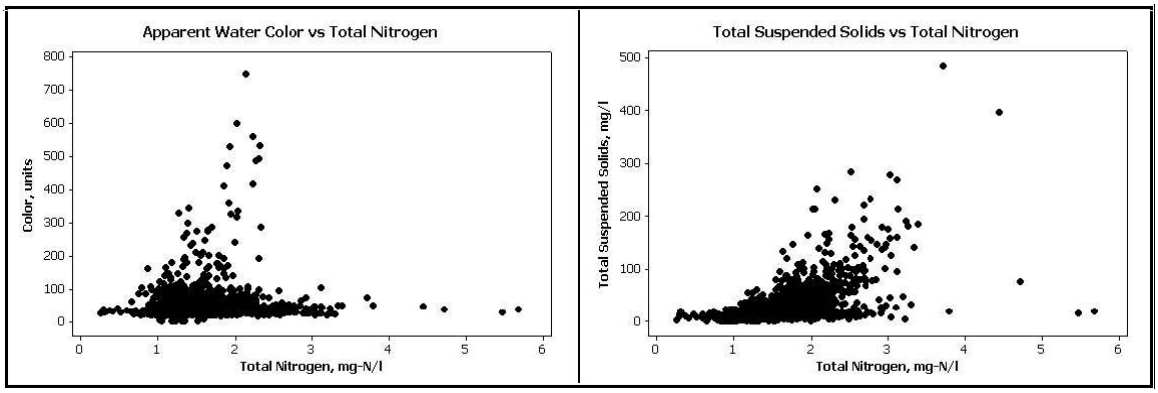
(hurricane-affected years removed) Depicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top right), ortho-P to TP ratios by year (lower left) and by month (lower right).



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.



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.

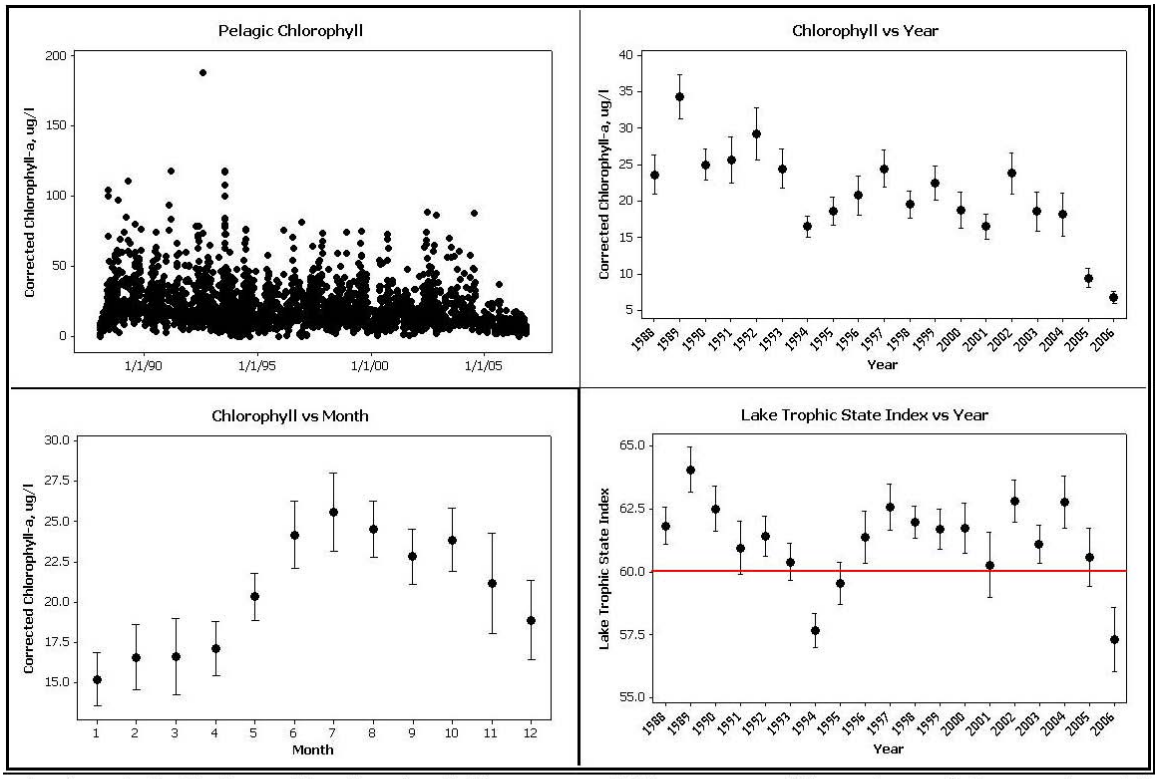


Depicted are individual sampling data pairs. Chlorophyll *a* concentration (not shown) relationship is very similar.

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

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.

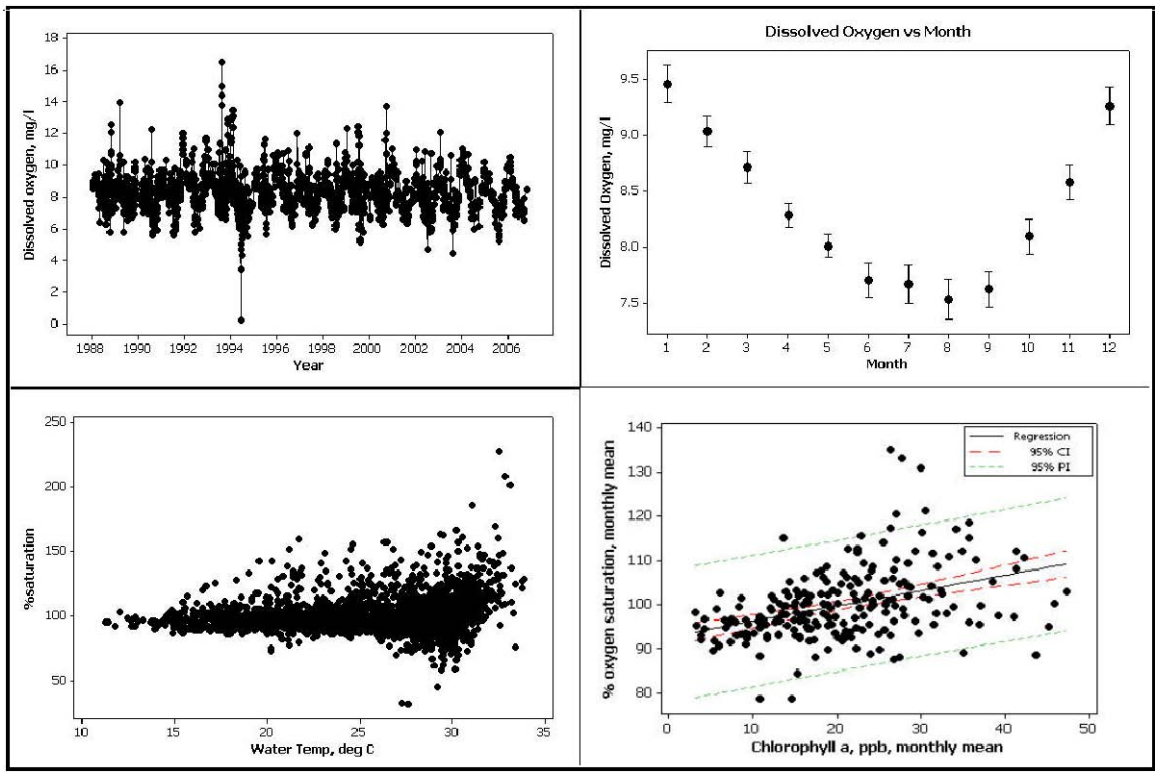


epicted are individual sampling data (top left), means and 95 percent confidence intervals by year (top righ
month (lower left), and lake trophic state index for P by year.

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

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.

Conclusions 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

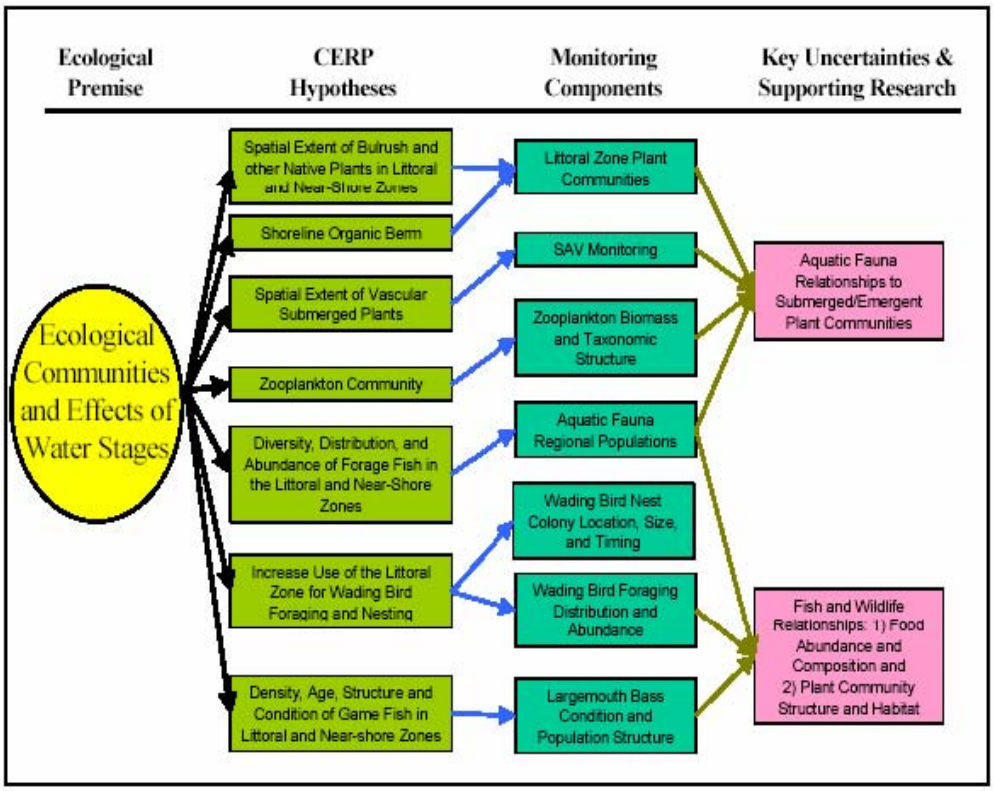
The current LO WSE Operating Schedule is more restrictive than past lake stage regulation schedules in stipulating conditions in which water is released from LO. This has resulted in an increased frequency and duration of undesirably high lake stages, which in turn have adversely affected key lake ecosystem 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

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.

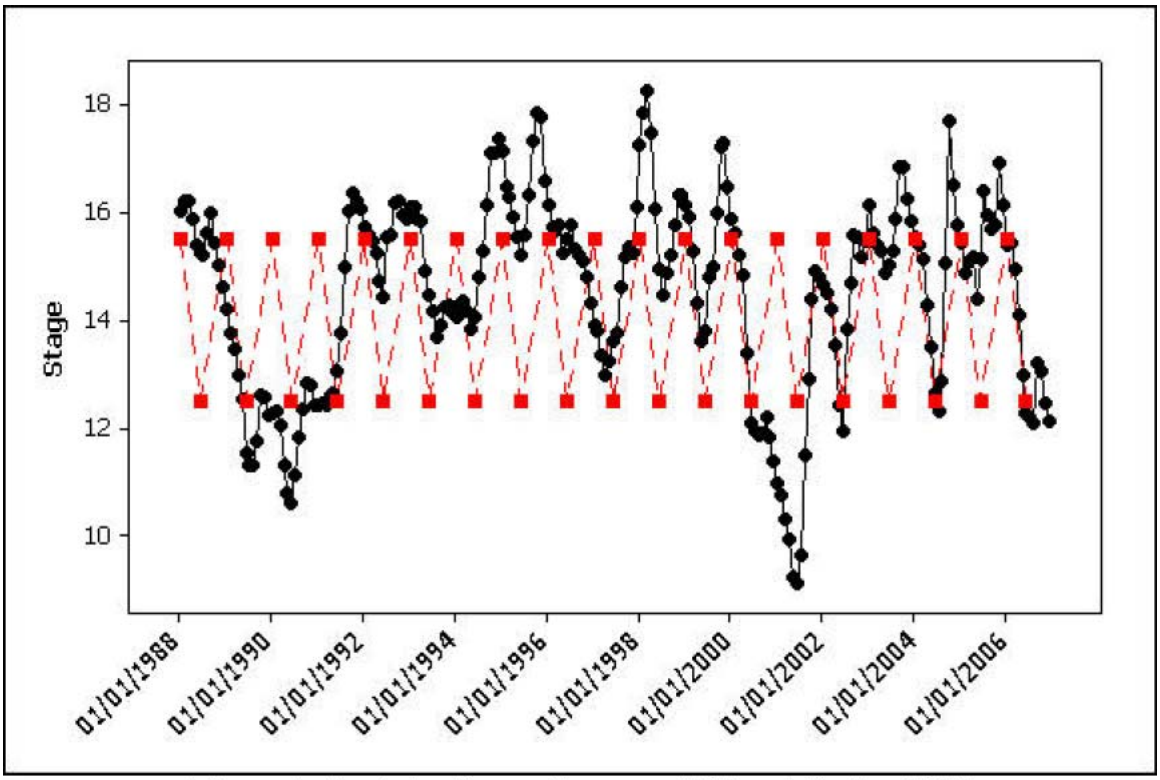


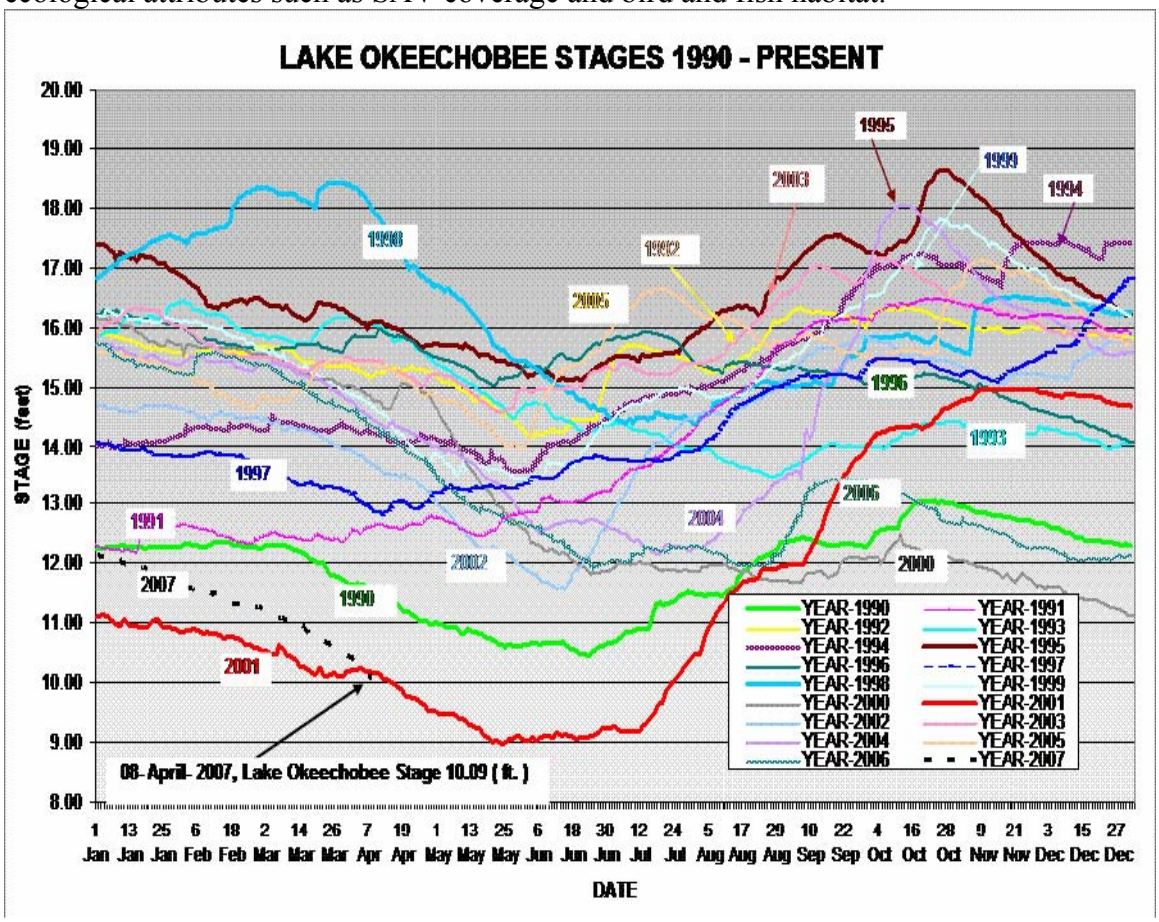
Figure 4-14: Lake Stages Between 1990 and Spring 2007

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.

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).

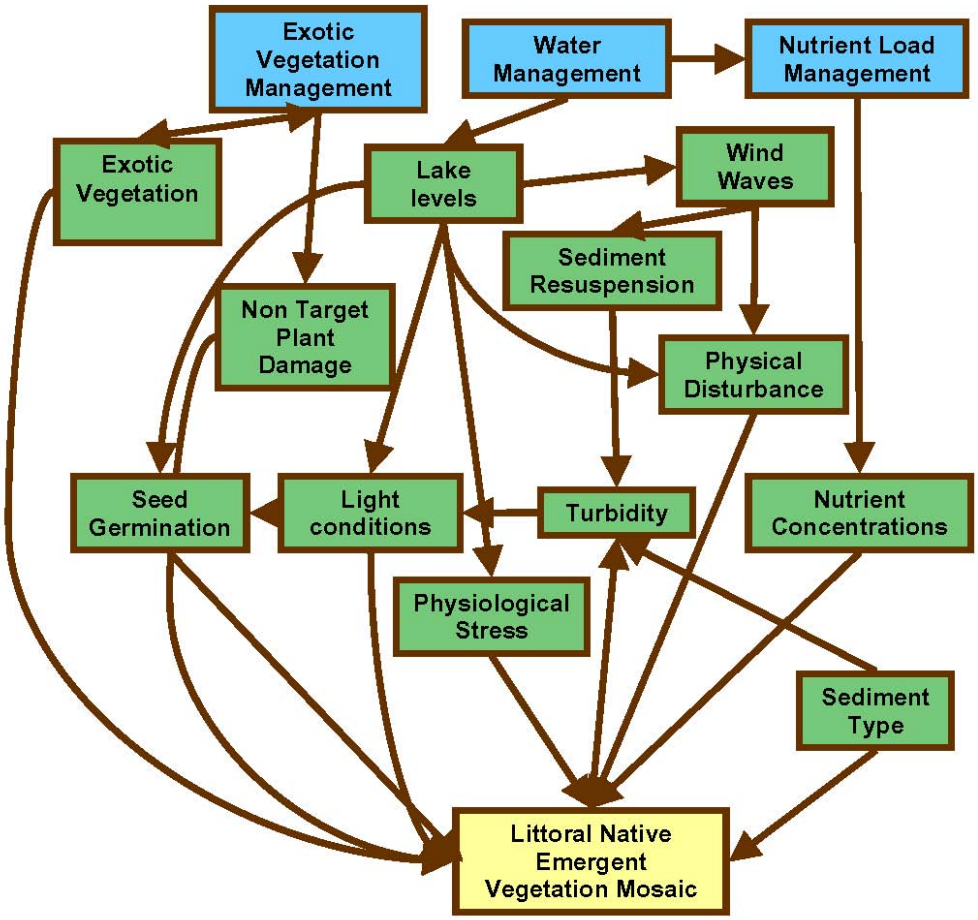


Figure 4-16: Conceptual Model of Stressors and/or Factors That Affect Submerged Littoral Zone Emergent Vegetation

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.

abundance (Havens et al., 2002).

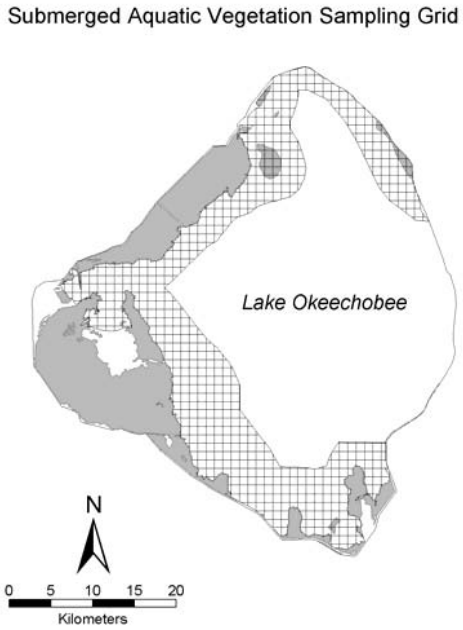


Figure 4-17: Annual SAV Mapping Sampling Grid

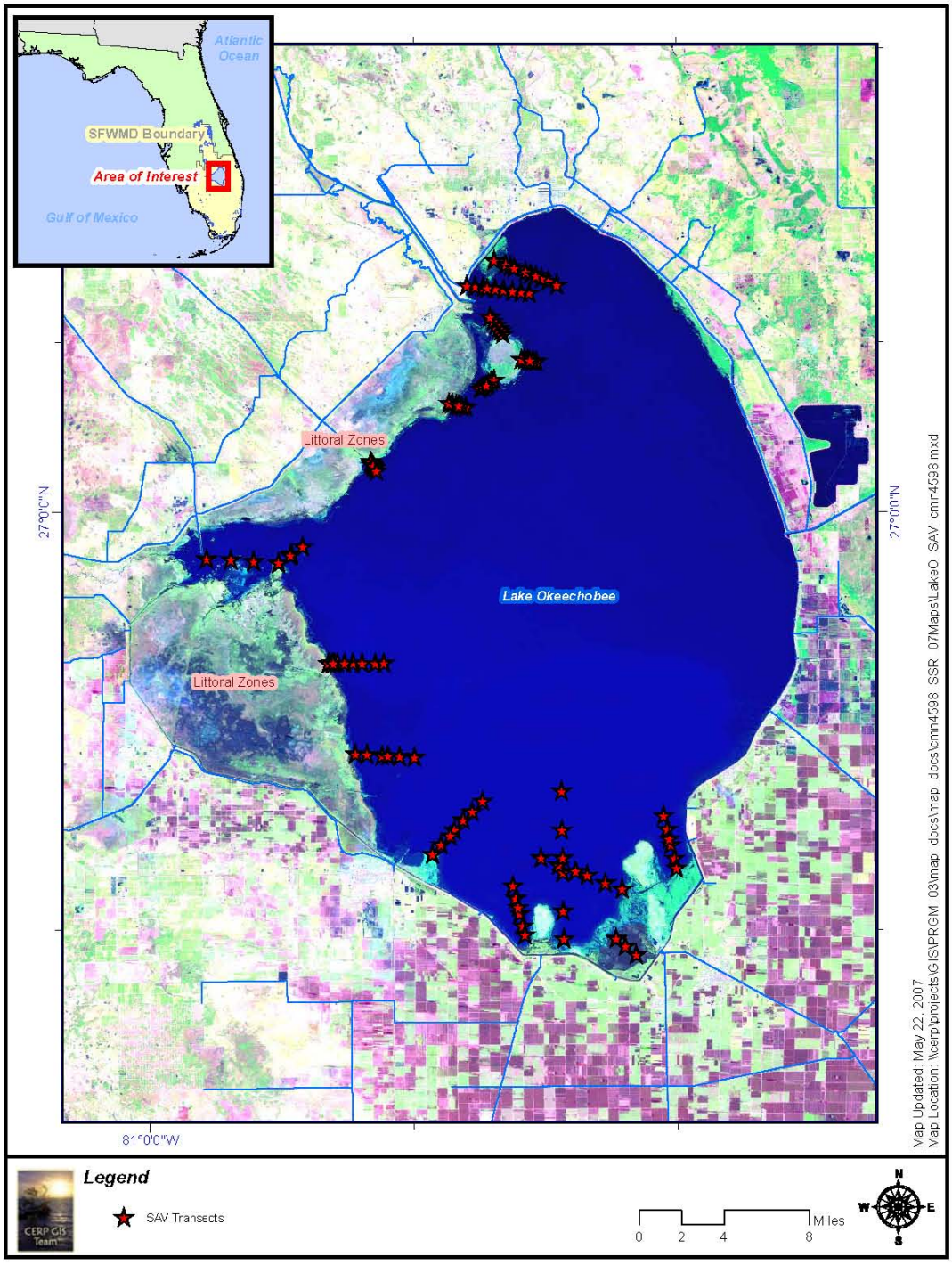


Figure 4-18: Submerged Aquatic Vegetation Transect Locations in Lake Okeechobee

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

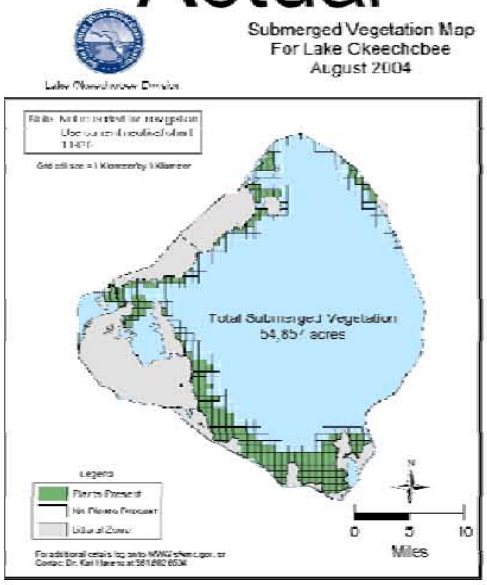
consequence, the annual mapping data lends itself most appropriately to evaluation of longer-term trends and should only be cautiously employed as regards to between-year differences. Conversely, sampling along transects is better suited for identifying and understanding short-term changes (Havens et al. 2002). The two approaches are thus complimentary, and sufficiently define the appropriate timescales as to allow interpretation.

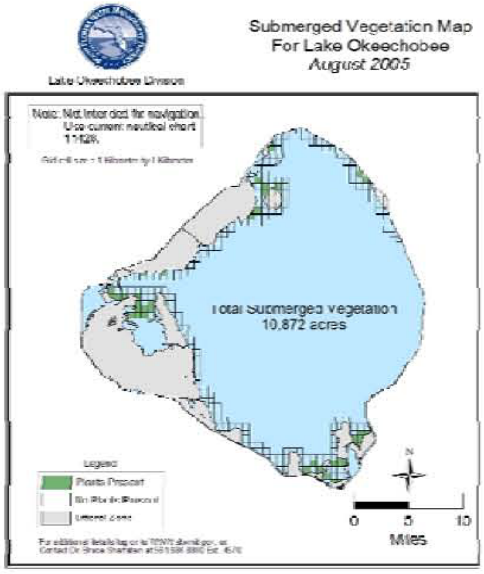
Empirical Modeling An empirical model has been developed that predicts SAV presence or absence distribution based on light penetration to the bottom as a function of water transparency, as indirectly measured by TSS and lake water levels. This model is intended to be used in conjunction with GIS data 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.

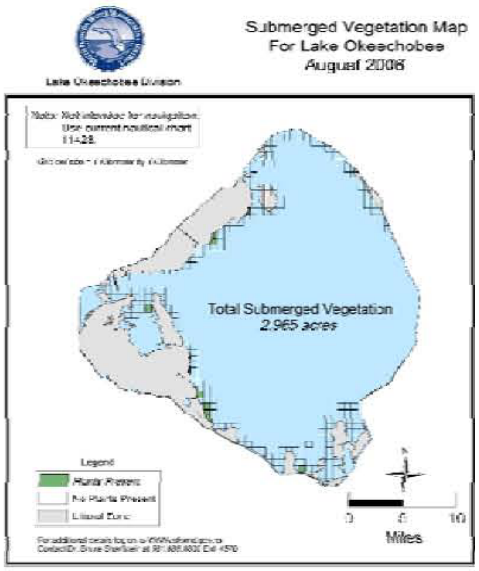
4.4.4 Discussion

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.

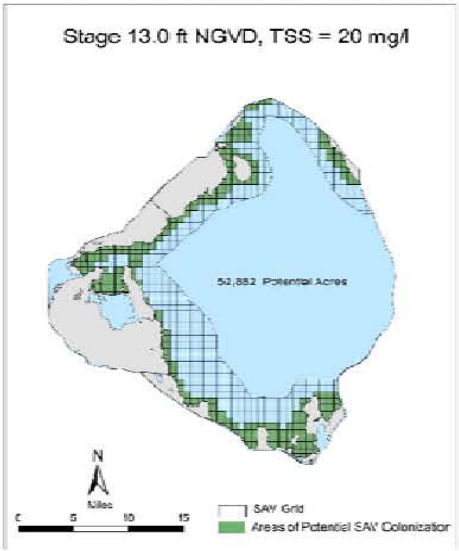
Actual

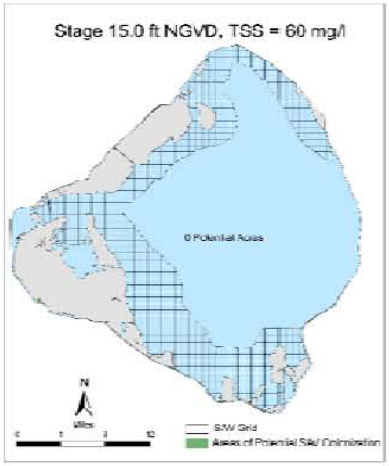






Predicted





2006 Not Yet Available

erged Aquatic Vegetation Biomass versus Model Predicted Biomass

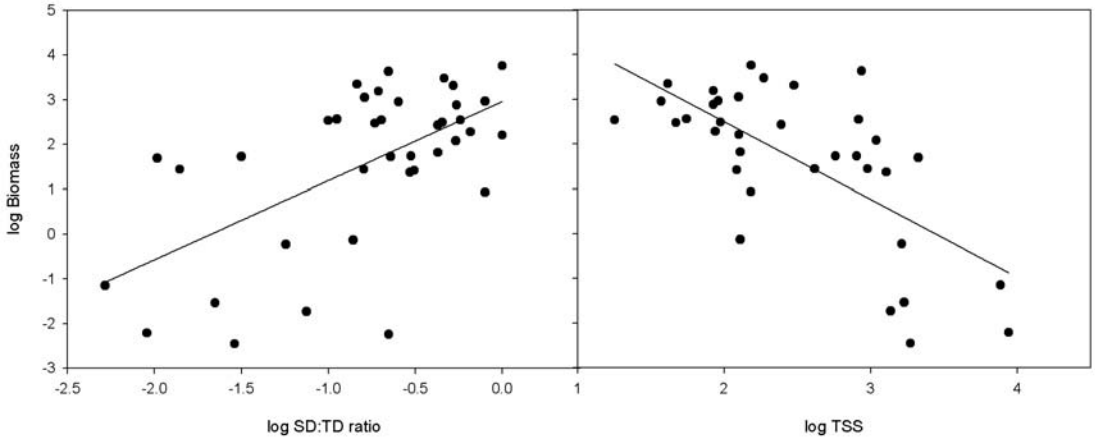
Transect Submerged Aquatic Vegetation Mapping Results The best example of transect mapping's ability to detect substantial short-term perturbations occurred between July 2004 (pre-hurricanes) and October 2004 (post-hurricanes). Average SAV biomass, as measured at 78 quarterly monitoring sites, declined 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

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

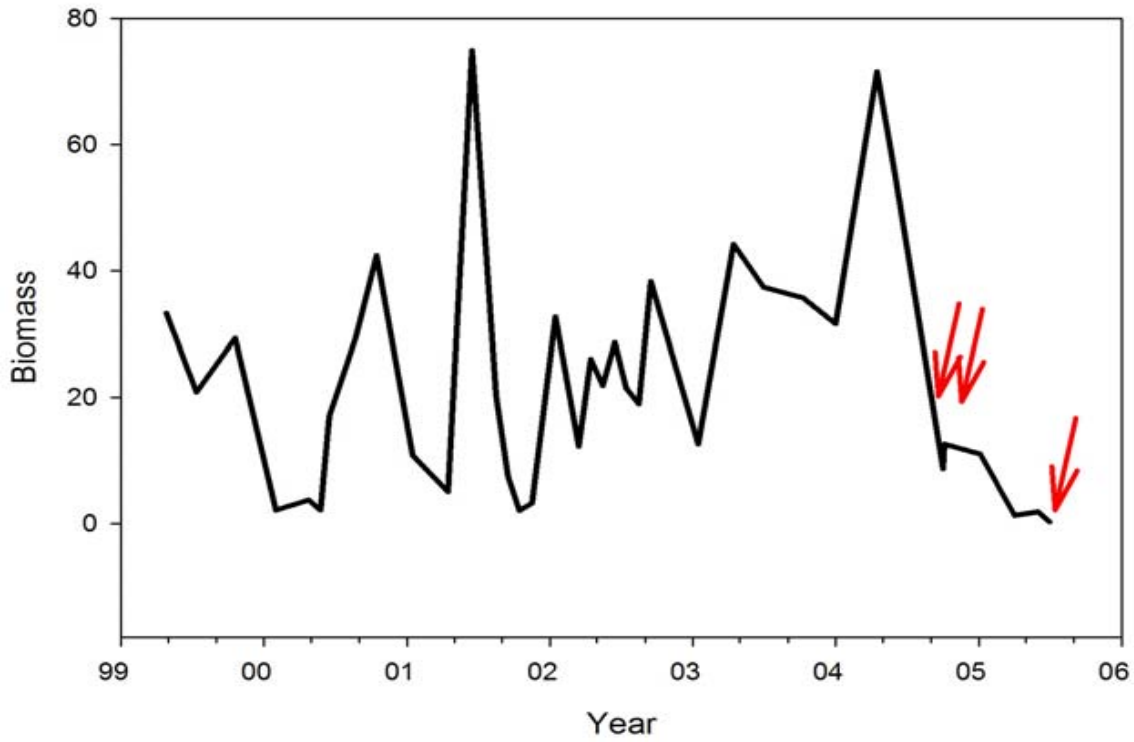


Biomass (g dry wt m^{-2}) as a function of secchi disk depth to total depth ratio (R^2 = 53%), and as function of TSS (mg/l, R^2 = 40%). Relationships are statistically significant (P<0.005). Values shown are logarithms of sampling event means.

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

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.

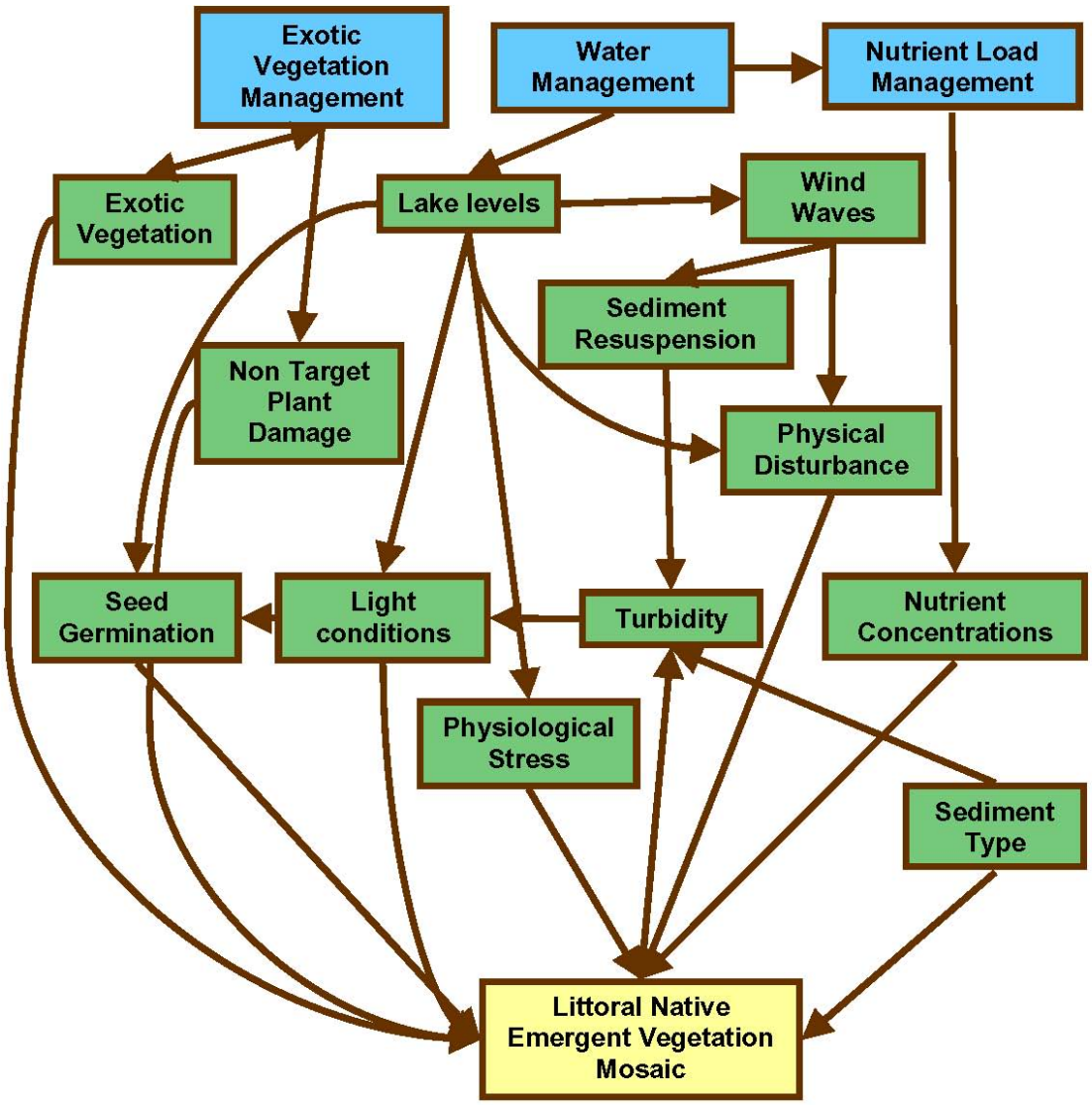


Figure 4-22: Littoral Zone Emergent Vegetation Mosaic Conceptual Ecological Model
Prolonged periods of high lake stage have direct and indirect negative impacts on native shoreline including bulrush (*Scirpus californicus*), SAV and interior marsh vegetation. Native shoreline vegetation is more likely 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

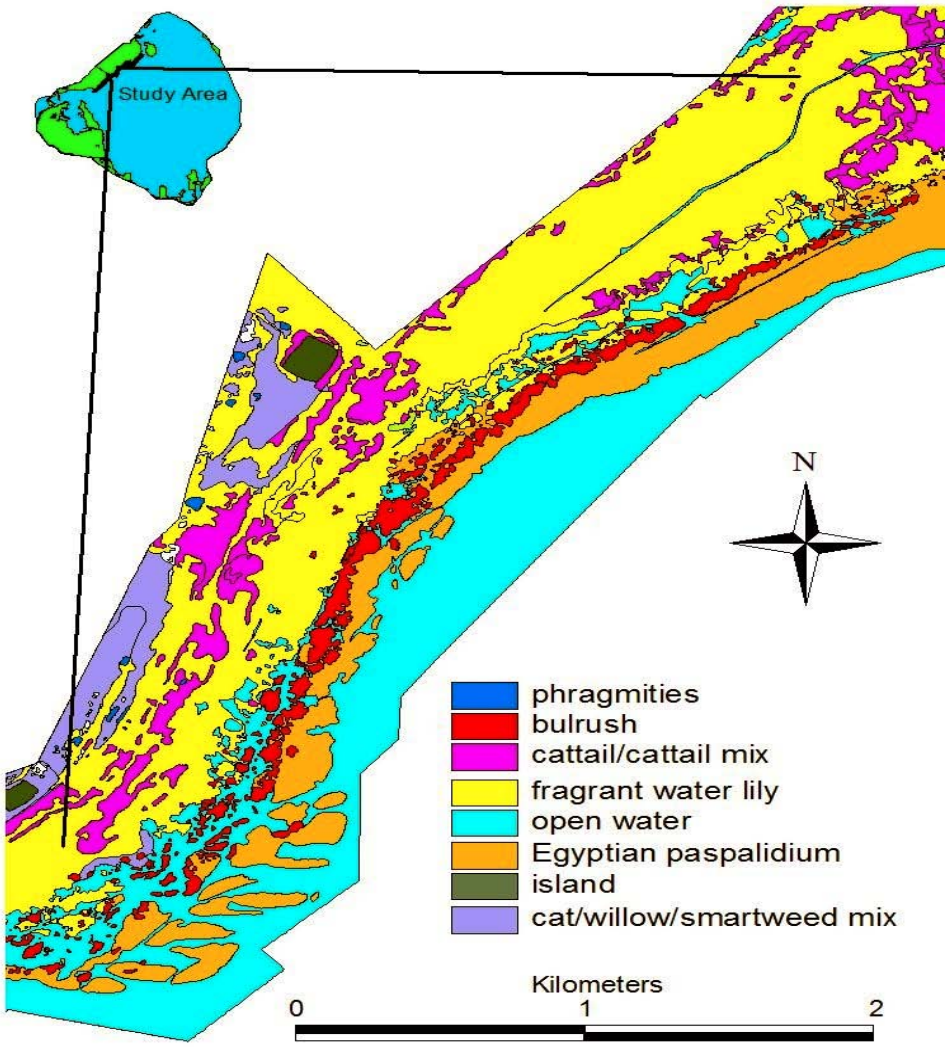
Emergent vegetation maps based on aerial photography for the entire LO marsh and for the western bulrush fringe has been collected since the mid-1990s and comprises a thorough baseline data set. However, the emergent vegetation community in LO, much like SAV, appears to be very dynamic, responding in 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.

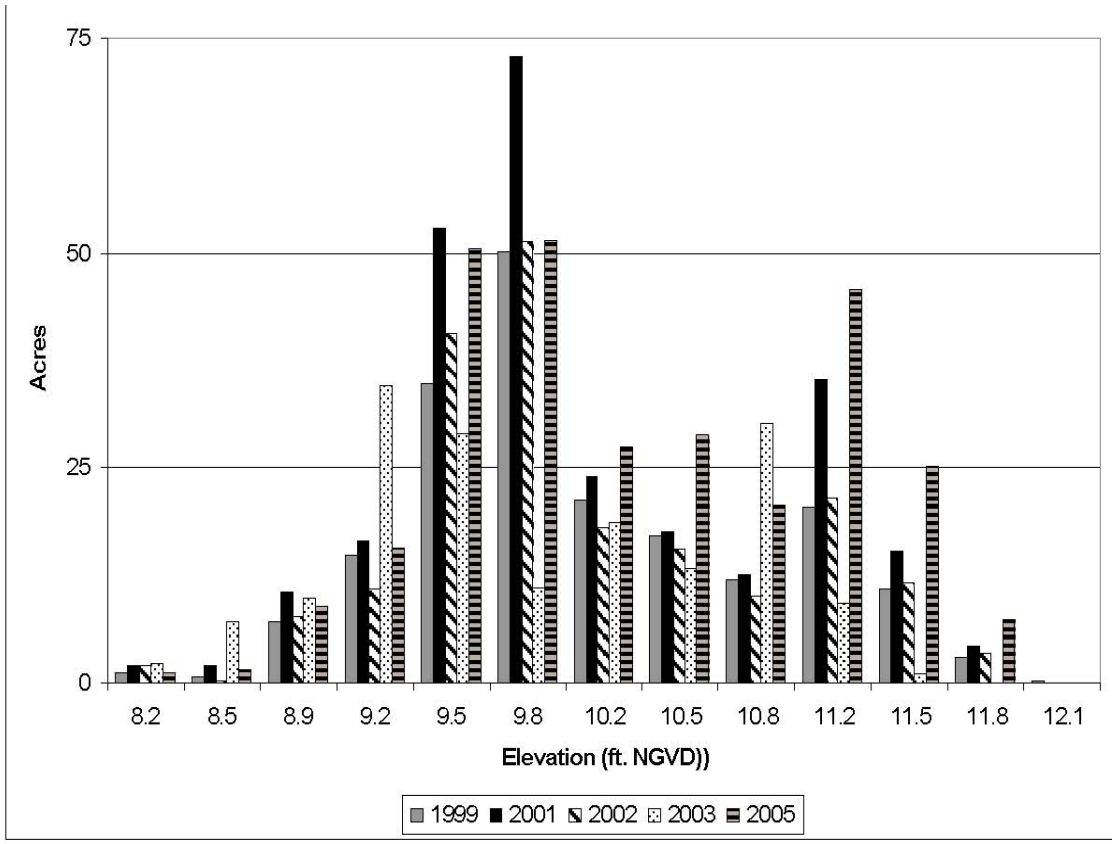
Emergent Aquatic Vegetation–Vegetation Maps A baseline vegetation map describing the distribution and areal coverage of vegetation in LO's marsh was developed in the early 1970s (Pesnell and Brown 1977). A second and more detailed vegetation map was developed in 1996. The most recent GIS map was developed 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



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.



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.

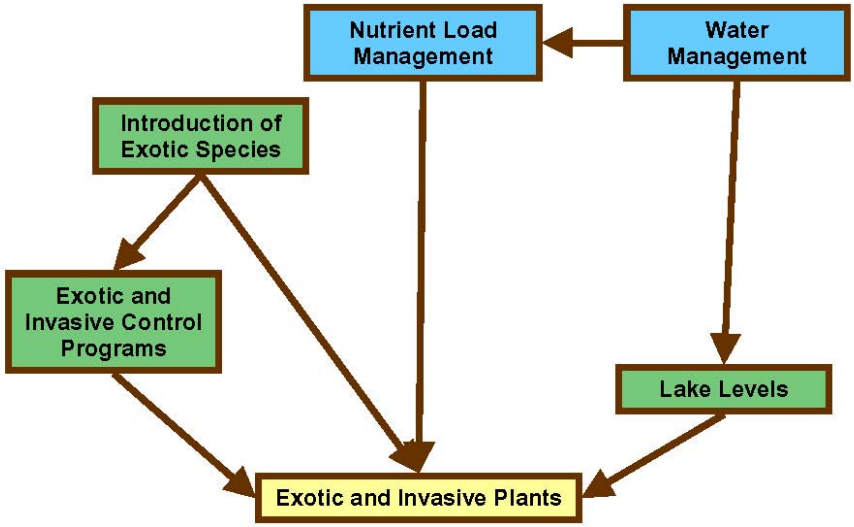


Figure 4-25: Conceptual Model of Stressors and/or Factors That Affect Exotic Vegetation

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

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.

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.

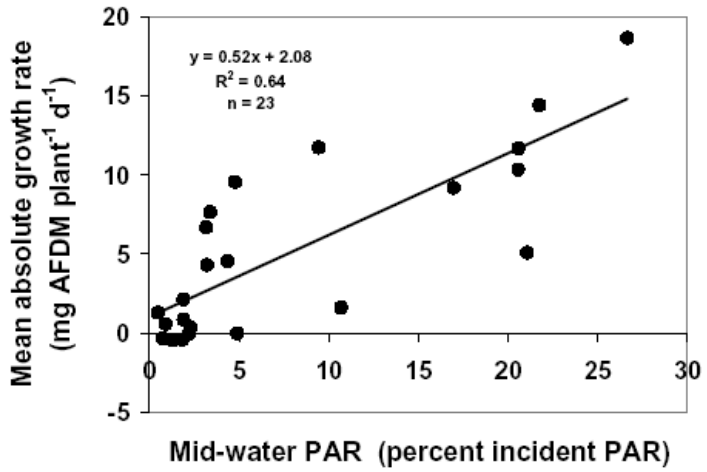


Figure 4-26: Linear Regression of Mid-water PAR Versus the Absolute Growth Rate of Hydrilla From Mesocosm Experiments

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*.

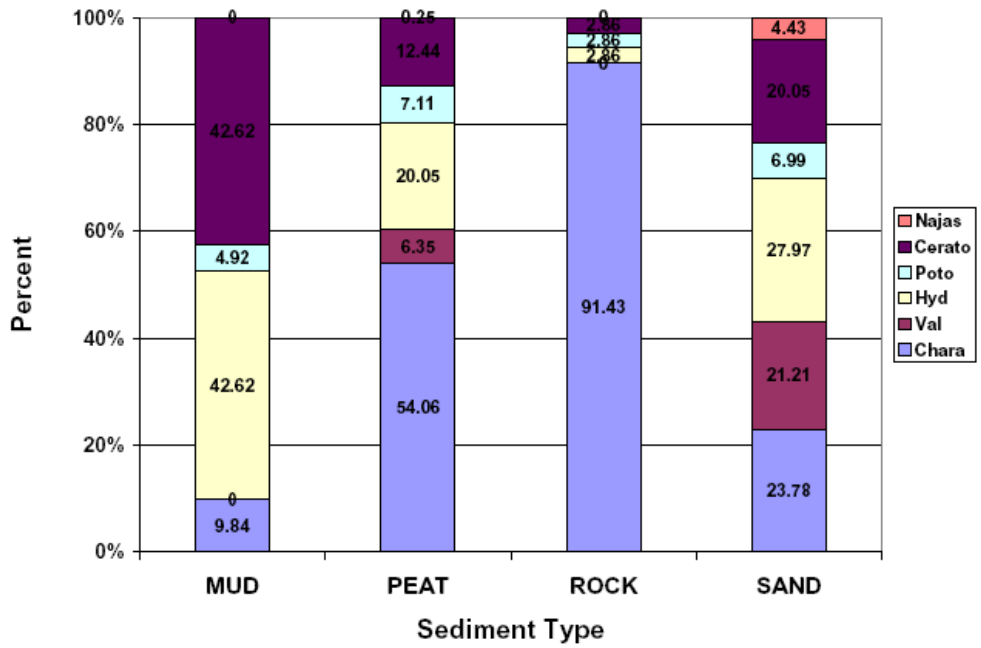


Figure 4-27: Percent of each Submerged Aquatic Vegetation Species As a Function of Sediment Type From the 2001 to 2005 Annual Mapping Data

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.

4.7.2 Background Description

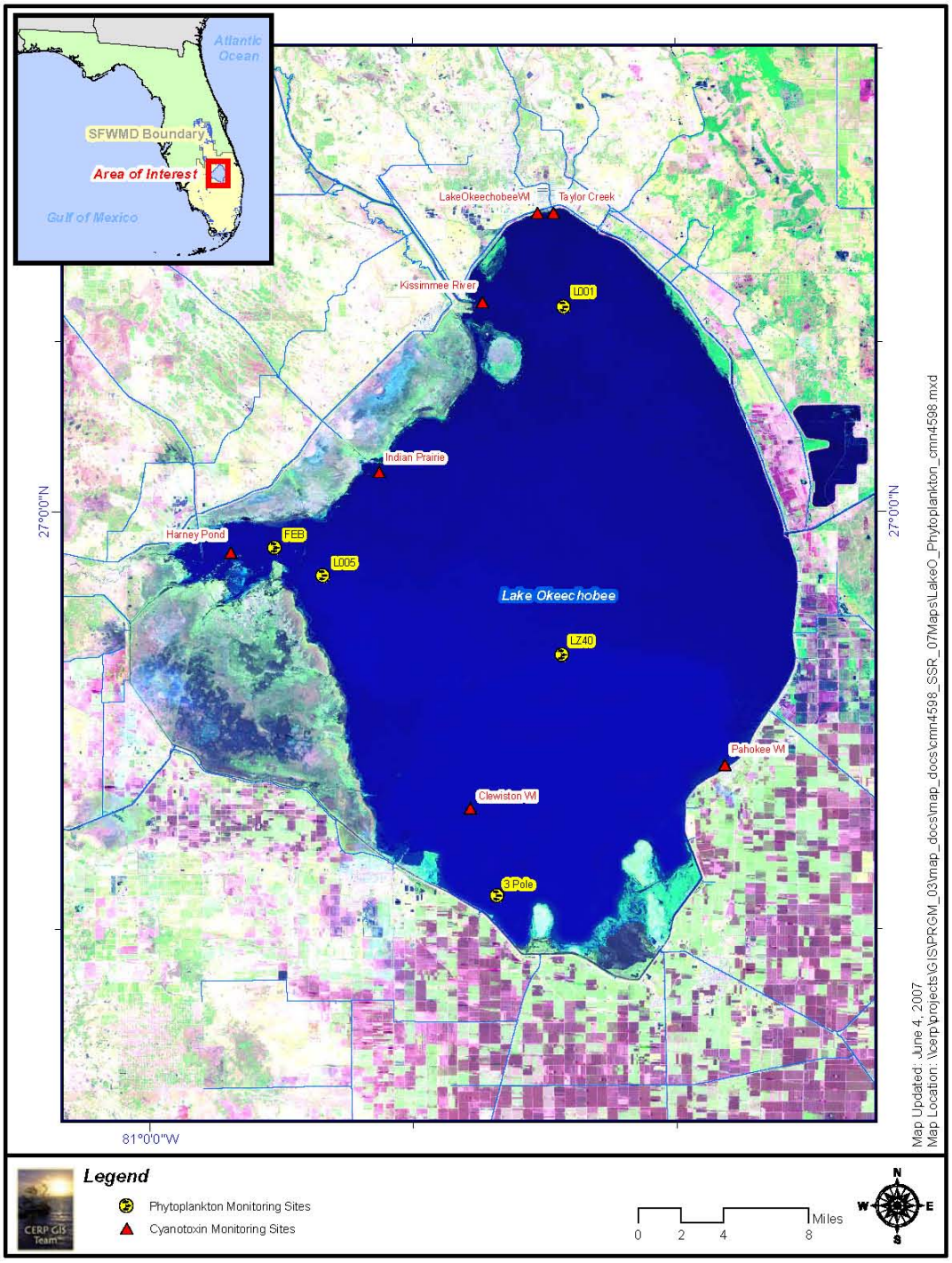
Phytoplankton research has been conducted in the pelagic and nearshore regions of LO since the late 1960s (Joyner 1974). Studies conducted during the 1970s and the early 1980s indicated that phytoplankton assemblages were spatially and temporally heterogeneous (Chichra et al. 1995). Several large surficial 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.



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.

Community Composition 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.

Biomass Determination 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.

Light and Nutrient Bioassays 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.

Diatom to Cyanobacteria Ratio Diatom to cyanobacteria ratios were calculated from the percent total biovolumes for both Divisions.

Cyanotoxin 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 (>

1.3×10^5 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).

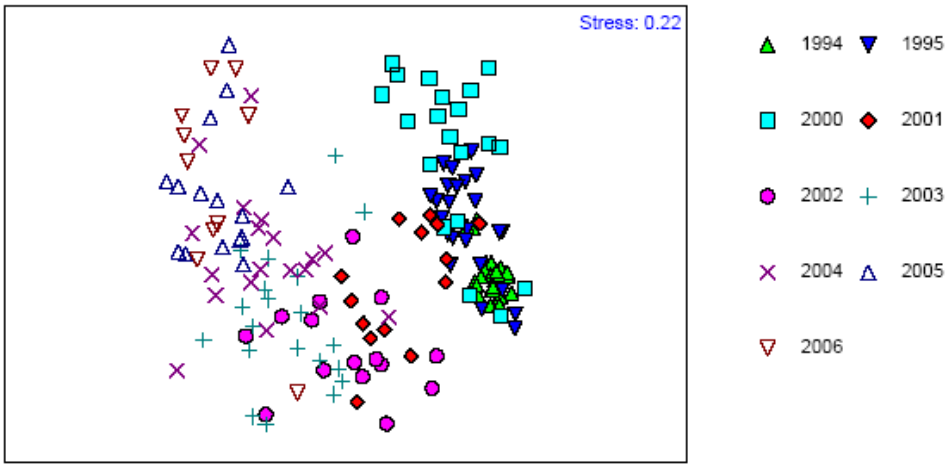
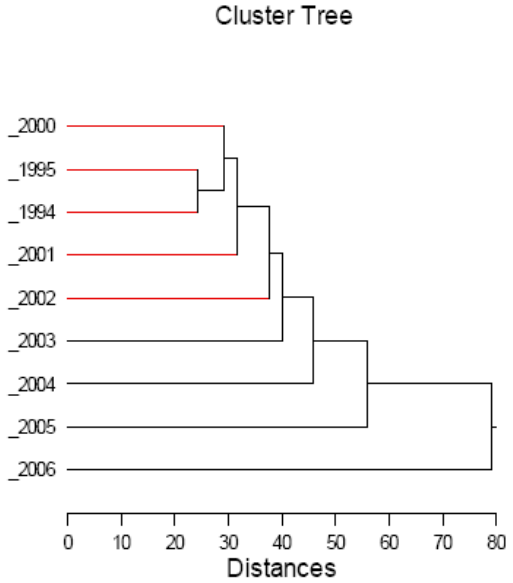


Figure 4-29: Phytoplankton Community Ordination Plot by Year



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

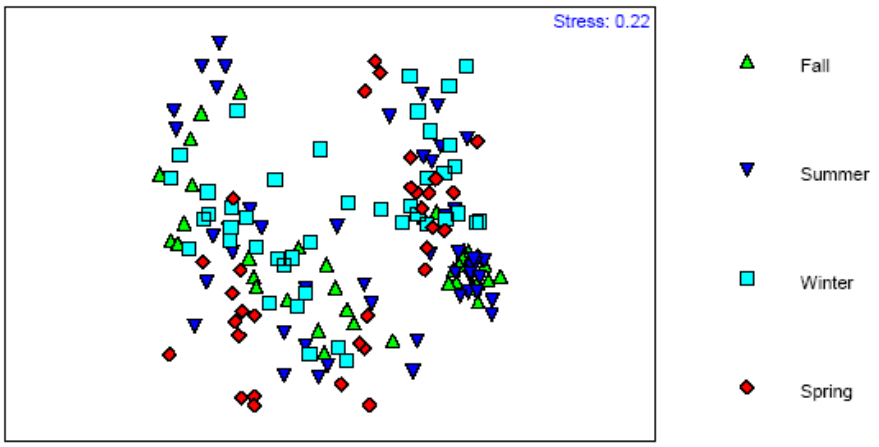


Figure 4-31: Phytoplankton Community Ordination Plot by Season

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.

Biomass Determination 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.

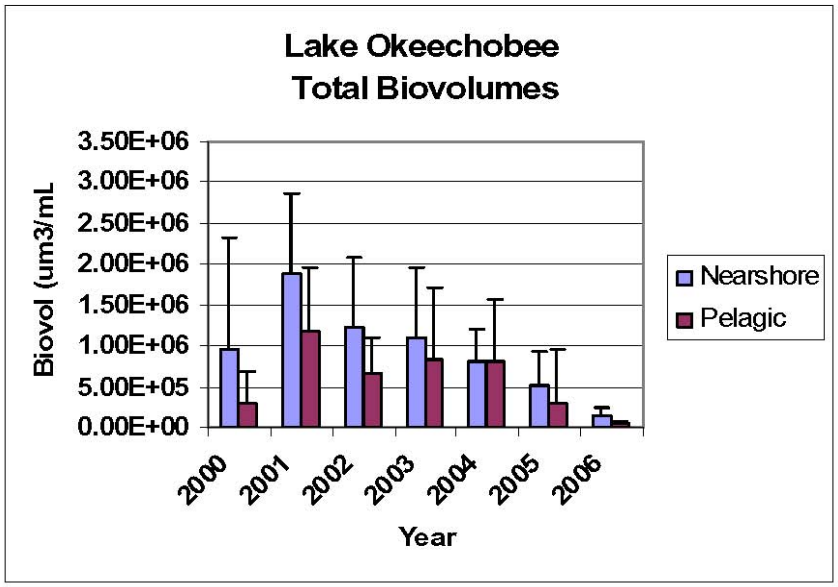


Figure 4-32: Annual Mean Phytoplankton Biomass in Total Biovolumes ± 1 S.D. at Both Nearshore and Pelagic Sites

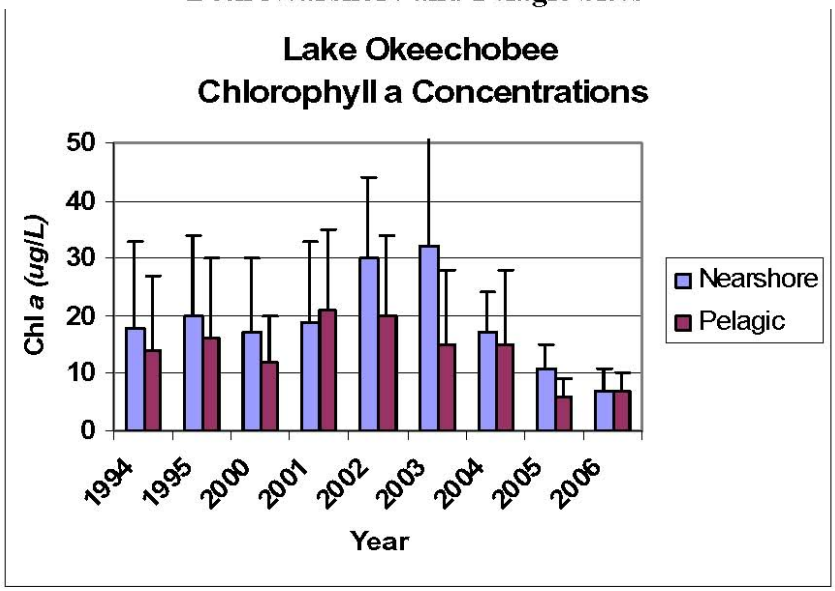


Figure 4-33: Annual Mean Phytoplankton Biomass as Chlorophyll *a* ± 1S.D. at Both Nearshore and Pelagic Sites

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.

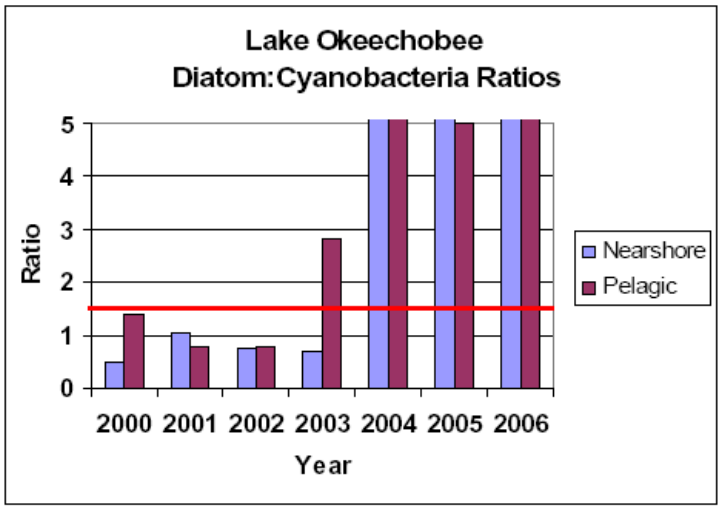


Figure 4-34: Annual Mean Diatom to Cyanobacteria Ratio at Both Nearshore and Pelagic Sites

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

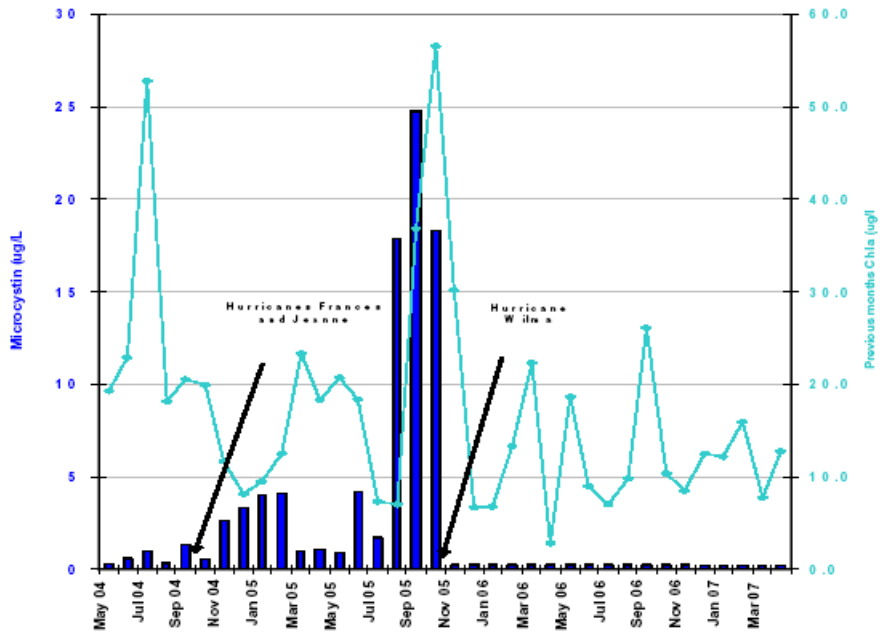


Figure 4-35: Mean Microcystin Concentrations (µg/L)

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

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.

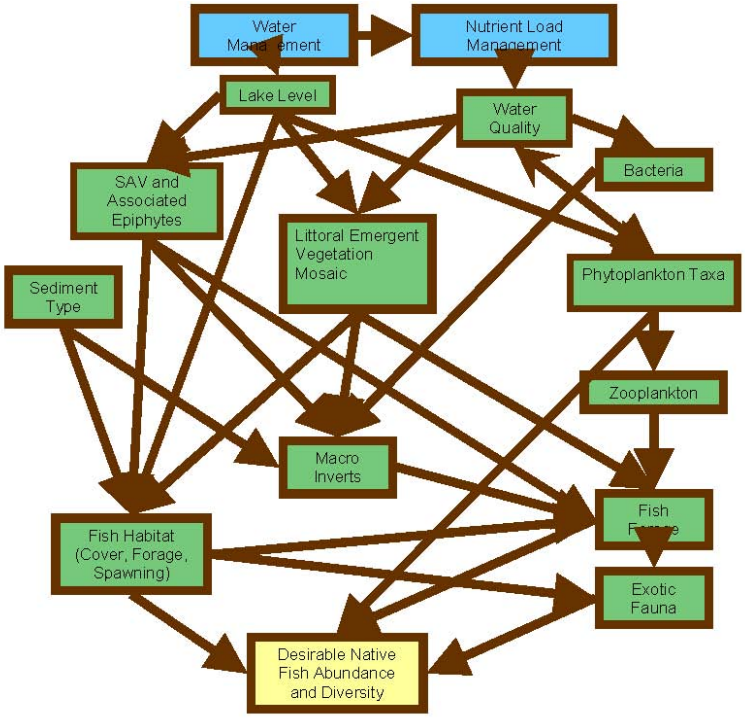


Figure 4-36: Conceptual Ecological Model Diagram Depicting the Factors Affecting Fish Populations

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.

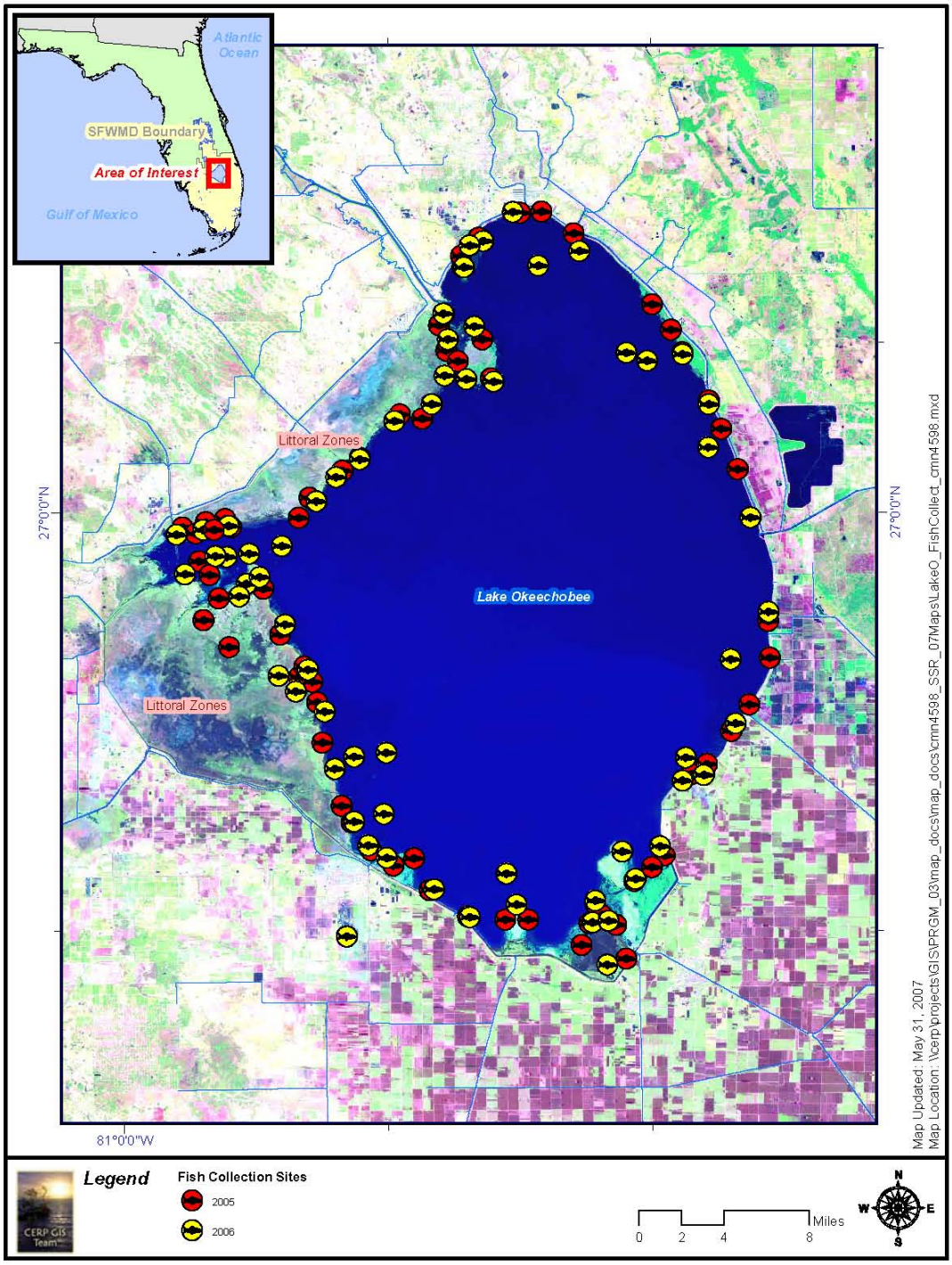
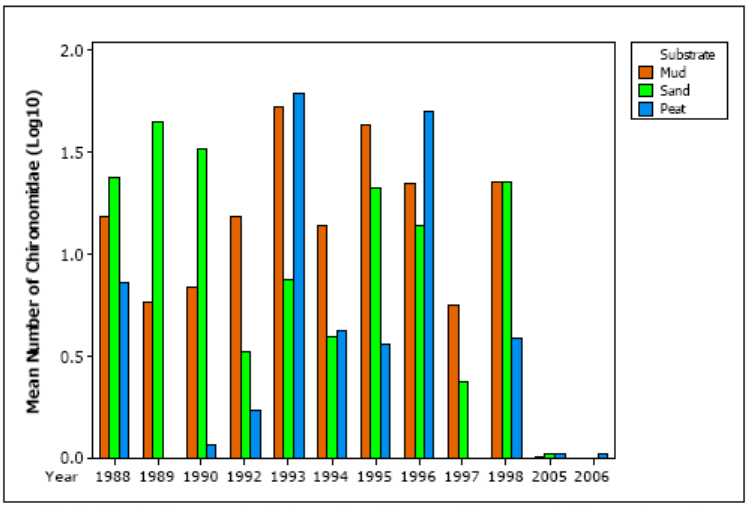


Figure 4-37: Sample Site Locations Used for 2005 Fish Collections

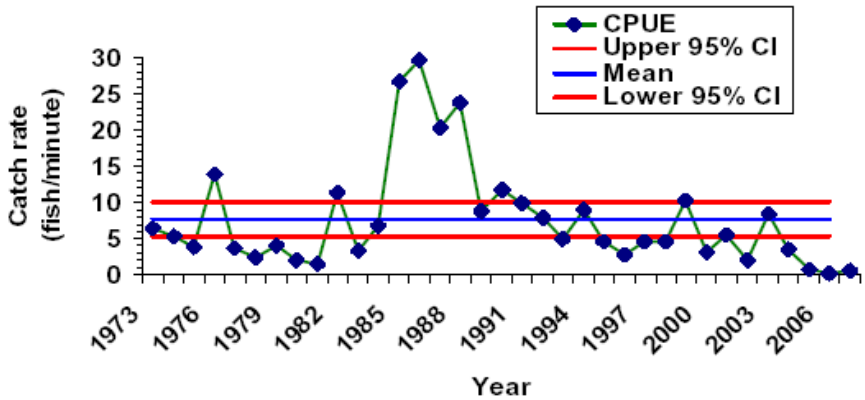
An additional important driving factor affecting LO's fish population density and structure is the reduction in numbers of the chironomidae macroinvertebrates (***Figure 4-38***). Chironomid larvae comprise the primary food source of juvenile black crappie and the decline in the former is another causative 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.

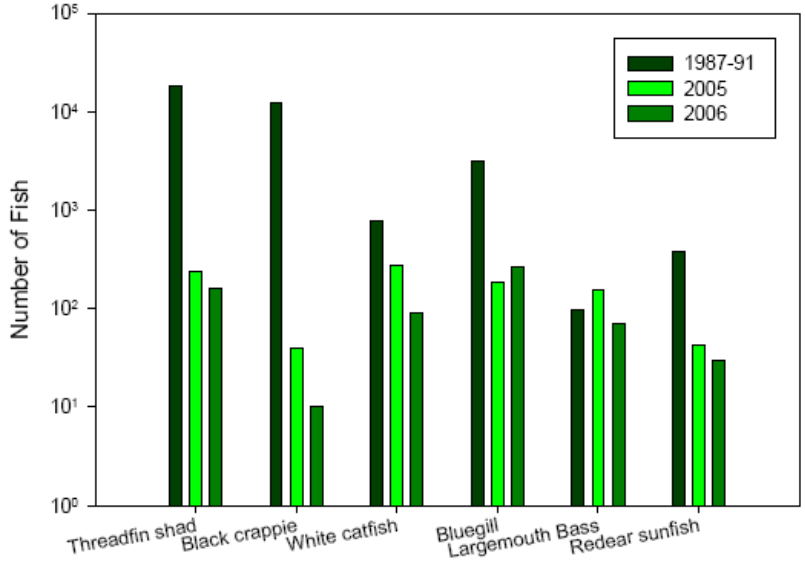


Figure 4-40: Comparative Abundance (log_{10}) of Recent Sampling Efforts (2005 and

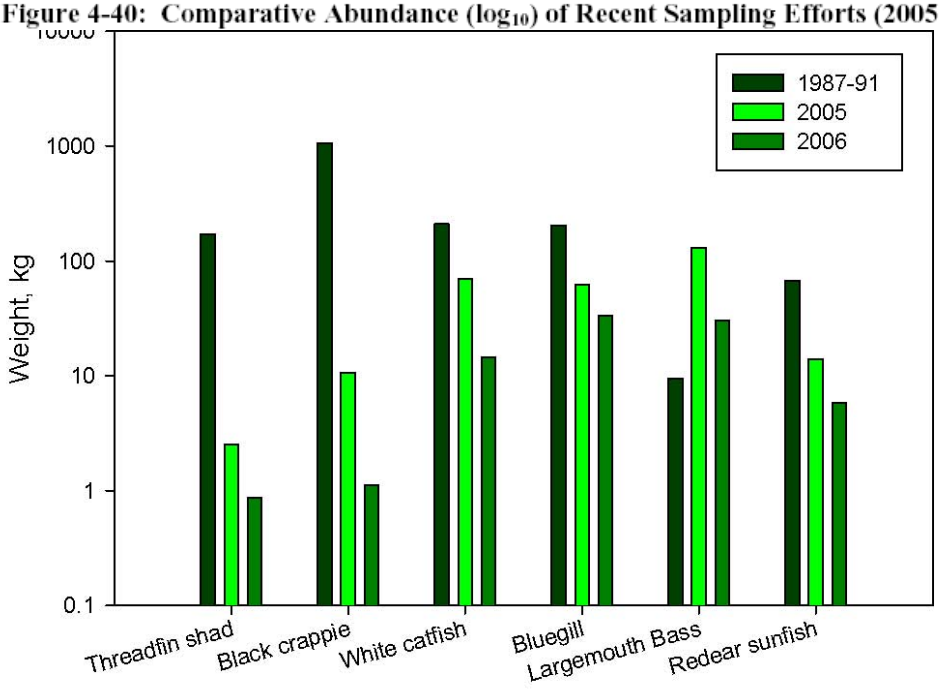
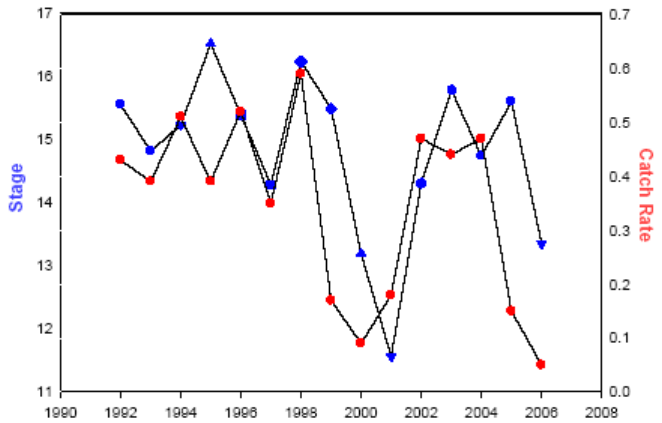


Figure 4-41: Comparative Weights in Kilograms (log_{10}) of Recent Sampling Efforts (2005 and 2006) to Historic Data (1987-91) of Selected Important Fish Species



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.

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.

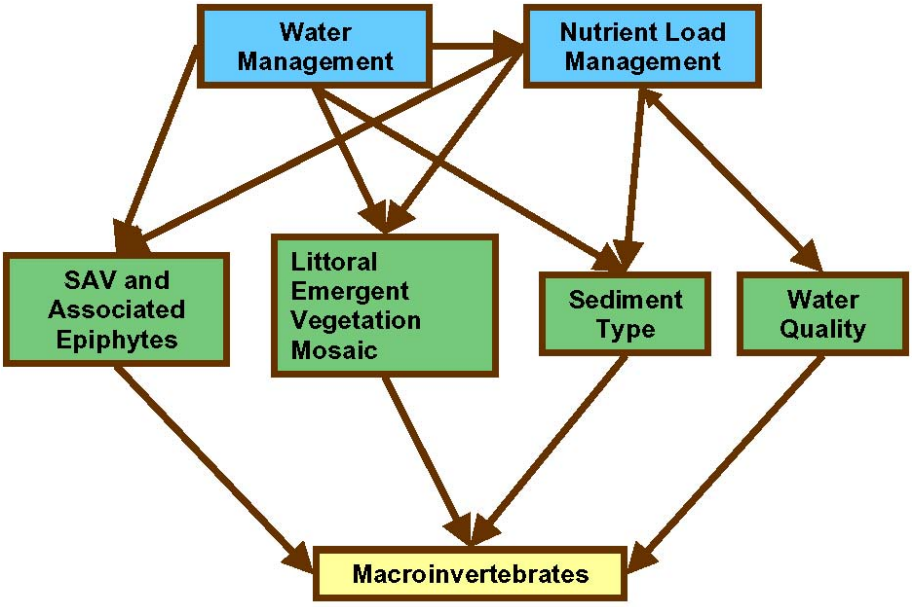


Figure 4-43: Macroinvertebrate Conceptual Ecological Model

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.

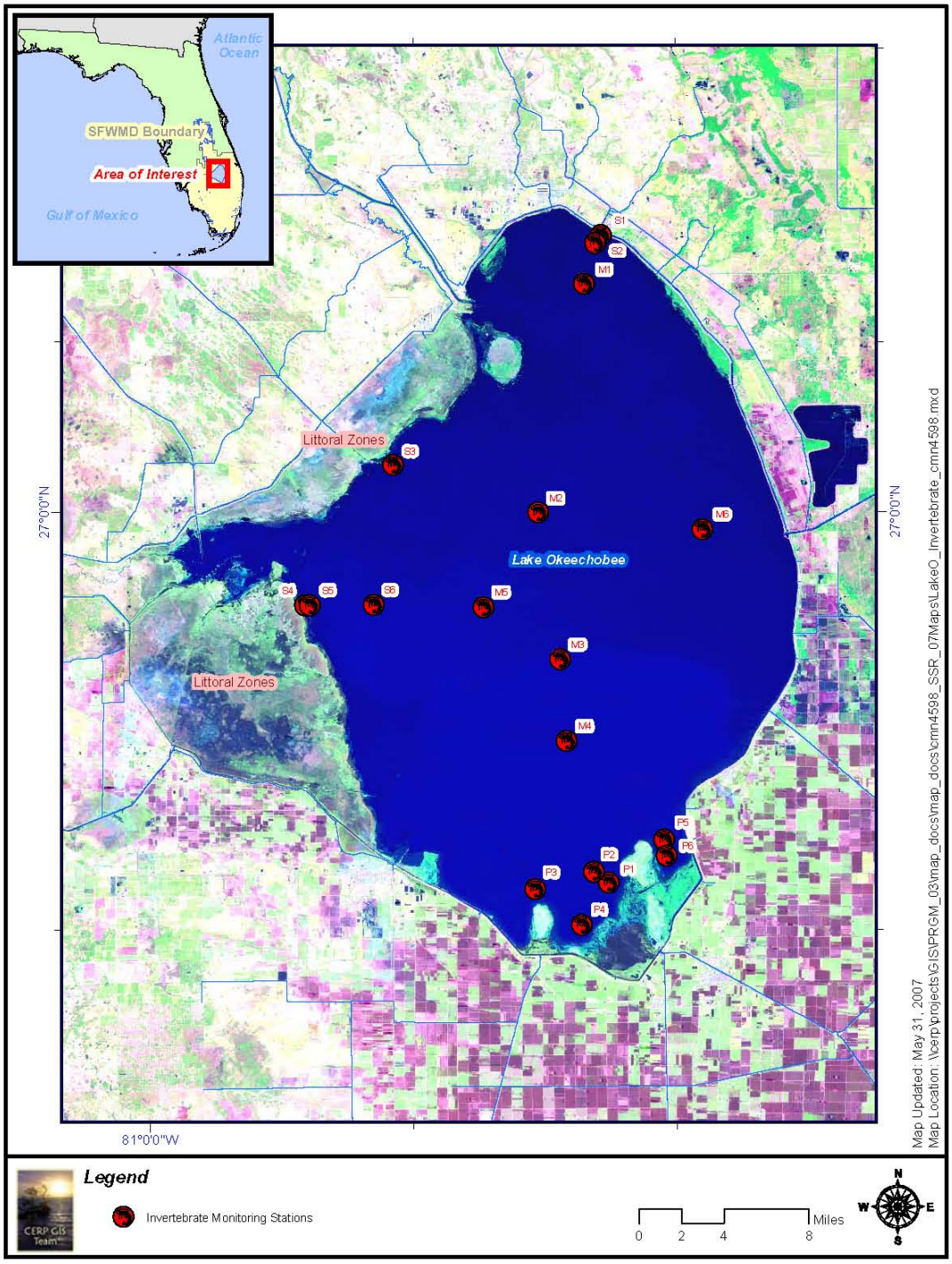


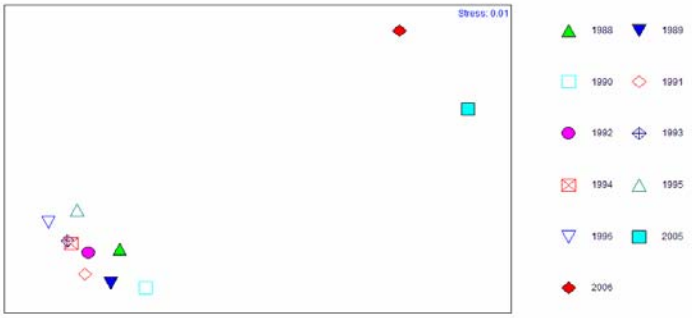
Figure 4-44: Florida Fish and Wildlife Conservation Commission Sublittoral Zone Benthic Invertebrate Sampling Sites in Lake Okeechobee

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).



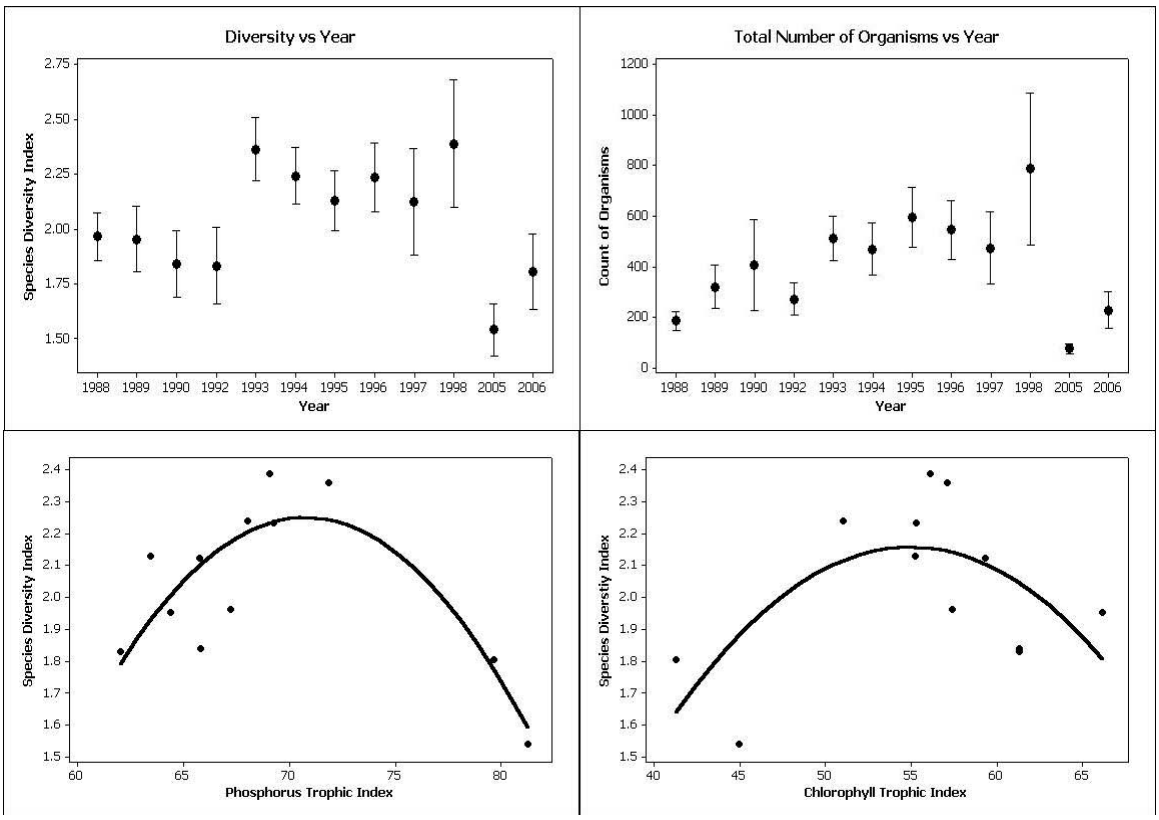
The closeness of points to one another reflects the similarity of the macroinvertebrate assemblage, in this case from year to year.

Figure 4-45: Multidimensional Scaling Ordination of Combined Annual

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.

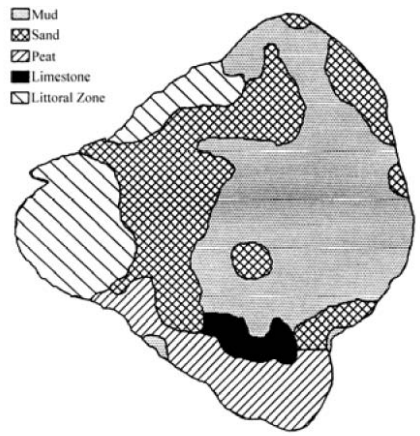


Significant relationships exist between mean annual diversity and trophic state indices for P (R^2 = 72%, P = 0.003), and to a lesser extent chlorophyll *a* (R^2 = 43%, P = 0.079).

Figure 4-47: Classic Species Diversity Index Response to Increasing Eutrophication

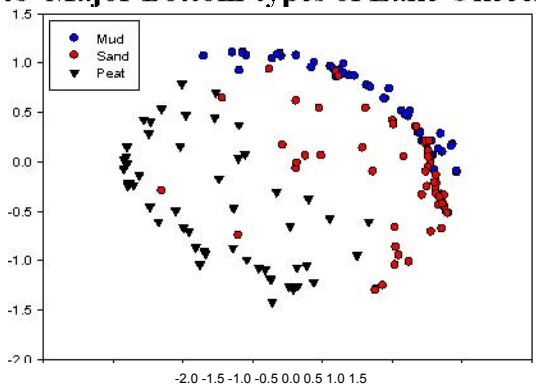
Habitat and Seasonal Influences Bottom substrate type is the primary determinant of invertebrate community structure in the LO sublittoral zone. Four primary benthic habitat regions characterize the sublittoral: mud, sand, peat, and limestone bedrock (Reddy 1993) (***Figure 4-48***). The mud region is 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.



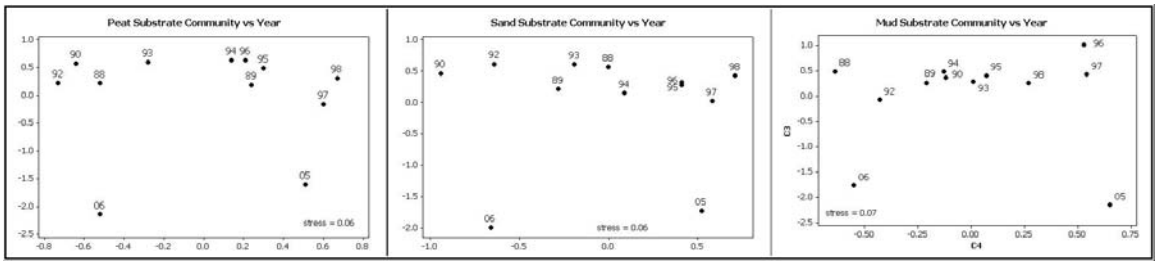
(Modified from Reddy 1993)

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

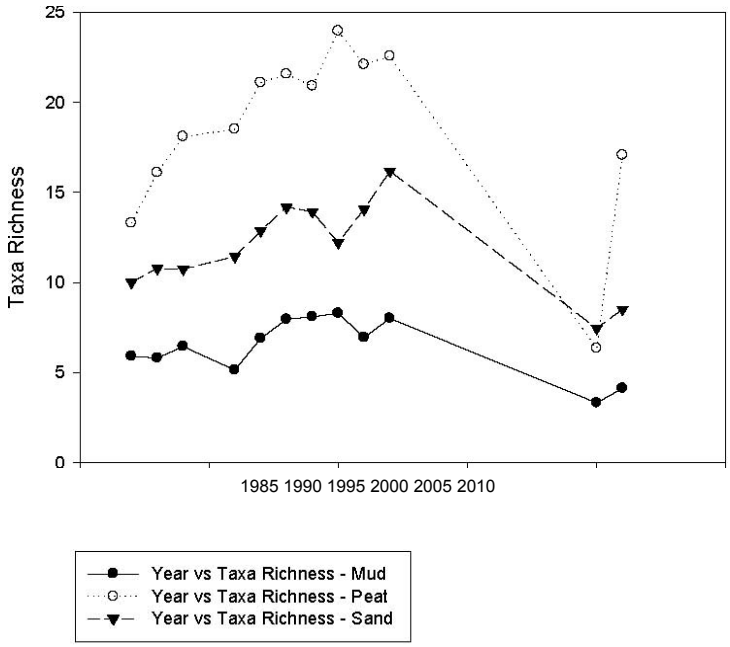


Figure 4-51: Species Richness Mean as a Function of Sample Year and Substrate Type

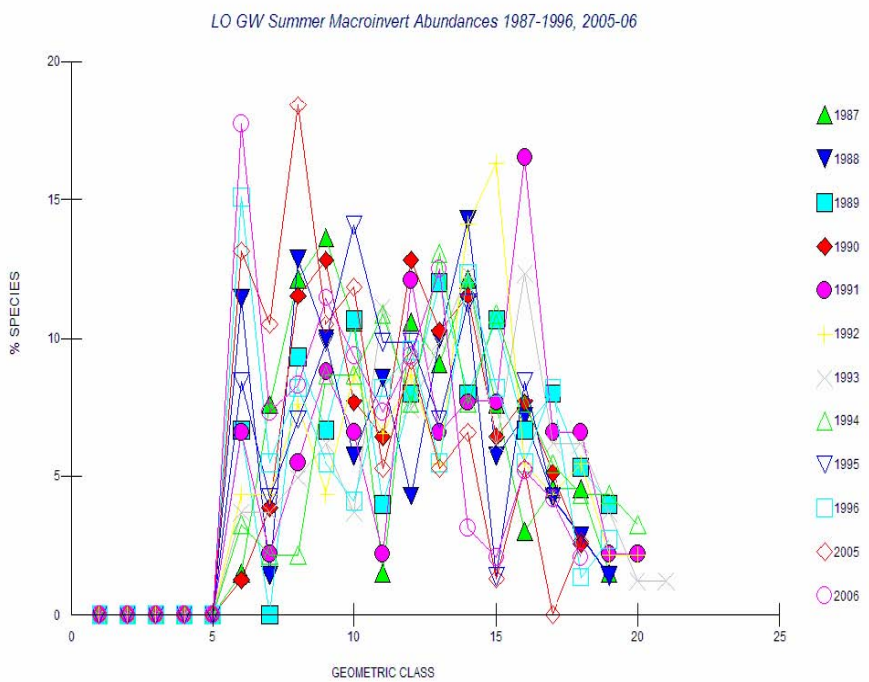
Results from previous studies have shown that benthic invertebrate communities of the LO mud zone have displayed the lowest species richness and diversity of all sublittoral habitat type communities sampled (Warren et al.1995). The current study reflects these results. Mud sediment-associated community 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.

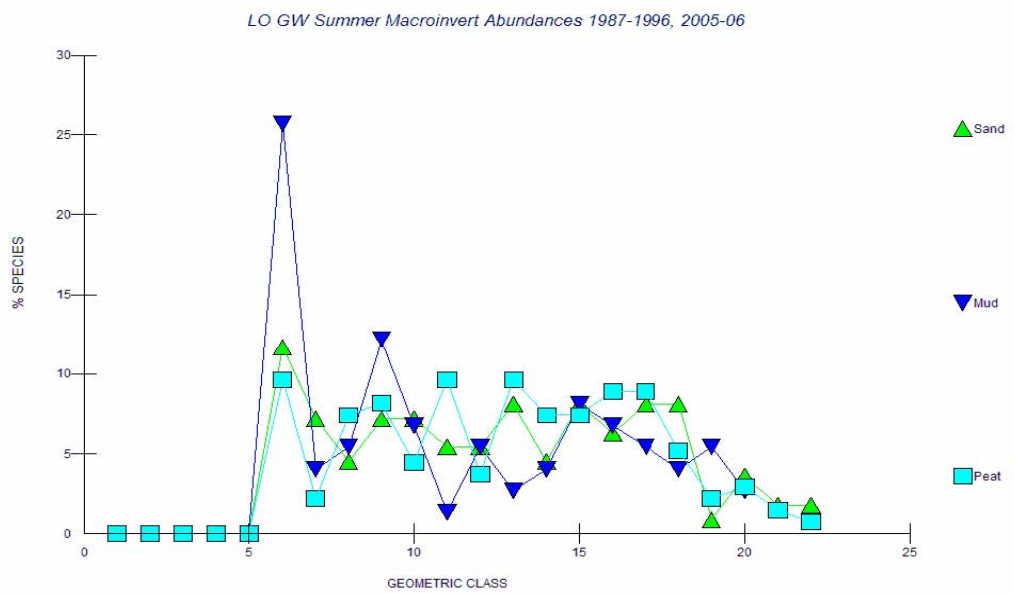
The taxonomic composition of the peat habitat region invertebrate community differed substantially from the taxonomic compositions of mud and sand region communities, however the pattern of extreme dominance by a few number of taxa exhibited in mud and sand-inhabiting communities was also reflected in the peat region. Aquatic Acari were, by far, the most abundant benthic invertebrates collected from peat, occurring with a

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

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

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.

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.

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

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).

4.13 References

Aldridge, F.J., Phlips, E.J., Schelske, C.L. 1995. The use of nutrient enrichment bioassays to test for spatial and temporal distribution of limiting factors affecting phytoplankton dynamics in Lake Okeechobee, Florida. Arch. Hydrobiol. Beih 45: 177-190.

Aumen, N.G. 1995. The history of human impacts, lake management, and limnological research on Lake Okeechobee, Florida (USA). Arch. Hydrobiol. Beih. 45: 1-16.

Bell, F.W. 1987. The economic impact and valuation of the recreational and commercial fishing industries of Lake Okeechobee. Florida Game and Fresh Water Fish Commission, Florida Department of Environmental Regulation, Tallahassee, FL.

Blasland Bouck and Lee Inc. 2003. Lake Okeechobee Sediment Management Feasibility Study. Boca Raton.

Brezonik, P. L., and D. R. Engstrom. 1998. Modern and historic accumulation rates of phosphorus in Lake Okeechobee, Florida. J. Paleolimnol. 20: 31-46.

Brinkhurst, R.C. 1974. The benthos of lakes. Macmillan Press, London.

Brundin, L. 1949. Chironomiden und andere Bodentiere der sudschwedischen

Urgebirgsseen. Ein Beitrag zur Kenntnis der bodenfaunustischen Charakterzuge schwedischer oligotropher Seen. Report of the Institute of Freshwater Research, Drottningsholm 30:1-914.

Bull, L.A., D.D. Fox, W. Brown, L.J. Davis, S.J. Miller, and J.G. Wullschleger. 1995. Fish distribution in limnetic areas of Lake Okeechobee, Florida. Arch. Hydrobiol., Beih. 45: 333-342.

Chichra, M.F., Badylak, S., Henderson, N., Rueter, B.H., Phlips, E.J. 1995. Phytoplankton community structure in the open water zone of a shallow subtropical lake (Lake Okeechobee, Florida, USA). Arch. Hydrobiol. Beih 45: 157-176.

Chimney, M. J. 2005. Surface seiche and wind set-up on Lake Okeechobee (Florida, USA) during hurricanes Frances and Jeanne. Lake and Reserv. Manage. 21(4): 465-473

Clarke, K.R., Warwick, R.M. 2001. Change in marine communities: an approach to statistical analysis and interpretation, 2nd ed. PRIMER-E, Plymouth, UK.

Dennison, W.C., R.J. Orth, K.A. Moore, J.C. Stevenson, V. Carter, S. Kollar, P.W. Bergstrom and R.A. Batiuk. 1993. Assessing Water Quality with Submersed Aquatic Vegetation. BioScience, 43: 86-94.

East, T.L., Sharfstein, B. 2006. Development of a decision tree model for the prediction of the limitation potential of phytoplankton in Lake Okeechobee, Florida, USA. Arch. Hydrobiol. 165(1): 127-144.

Engstrom, D.R., Schottler, S.P., Leavitt, P.R., Havens, K.E. 2006. A reevaluation of the cultural eutrophication of Lake Okeechobee using multiproxy sediment records. Ecol. Appl. 16(3): 1194-1206.

Florida Department of Environmental Protection. 2001. Total Maximum Daily Load for Total Phosphorus Lake Okeechobee, Florida. Talahassee, FL. 47pp.

Flaig, E.G., and K.E. Havens. 1995. Fate of phosphorus in the Lake Okeechobee ecosystem
I. Land use and nutrient loading. Archiv fur Hydrobiologie Monographishe Beitrage 107: 1-24.

Florida Game and Fresh Water Fish Commission. 1991. Lake Okeechobee-Kissimmee River-Everglades resource evaluation project. Federal Wallop-Breaux completion report (F-52-5). 328 pp.

Gleason, P.J. 1984. Environments of South Florida Present and Past II. Miami Geological Society, Coral Gables, FL.

Grimshaw, H.J. and B. Sharfstein (in preparation).

Grimshaw, H.J., K. Havens, B. Sharfstein, A. Steinman, D. Anson, T. East, R.P. Maki, A. Rodusky, and K-R. Jin. 2002. The effects of shading on morphometric and meristic characteristics of Wild Celery, Vallisneria americana MICHX., transplants from Lake Okeechobee, Florida. Archiv fur Hydrobiologie, 155: 65-81.

Grimshaw, H.J., B. Sharfstein, and T. East. 2005. The effects of shading on Chara zeylanica KLEIN ex WILD. and associated epiphytes. Archiv fur Hydrobiologie,

162: 253-266.

Havens, K.E., Hanlon, C., James, R.T. 1995. Historical trends in the Lake Okeechobee ecosystem V. Algal blooms. Arch. Hydrobiol. Suppl. 107 (1): 89-100.

Havens, K.E., Aumen, N.G., James, R.T., Smith, V.H. 1996. Rapid ecological changes in a large subtropical lake undergoing cultural eutrophication. Ambio 25: 150-155.

Havens, K.E., N.G. Aumen, R.T. James and V.H. Smith. 1996. Rapid ecological changes in a large subtropical lake undergoing cultural eutrophication. Ambio 25: 150-155.

Havens, K. E., K.-R. Jin, A. J. Rodusky, B. Sharfstein, M. A. Brady, T. L. East, N. Iricanin,
R. T. James, M. C. Harwell, and A. D. Steinman. 2001. Hurricane effects on a shallow lake ecosystem and its response to a controlled manipulation of water level. The Scientific World 1: 44-70.

Havens, K.E., J.R. Beaver, T.L. East, A.J. Rodusky, B. Sharfstein, A. St. Amand and A.D. Steinman. 2001. Nutrient effects on producers and consumers in the littoral plankton and periphyton of a subtropical lake. Archiv fur Hydrobiol., 152: 177-201.

Havens, K.E., and W.W. Walker. 2002. Development of a total phosphorus concentration goal in the TMDL process for Lake Okeechobee, Florida (USA). Lake Reservoir Manag.
18: 227-238.

Havens, K.E., M.C. Harwell, M.A. Brady, B. Sharfstein, T.L. East, A.J. Rodusky, D. Anson and R.P. Maki. 2002. Large-scale mapping and predictive modeling of submerged aquatic vegetation in a shallow eutrophic lake. The Scientific World. 2; 949-965.

Havens, K.E. 2003. Submerged aquatic vegetation correlations with depth and light attenuating materials in a shallow subtropical lake. Hydrobiologia, 493: 173-186.

Havens, K.E., B. Sharfstein, M.A. Brady, T.L. East, M.C. Harwell, R.P. Maki and A.J. Rodusky. 2004. Recovery of submerged plants from high water stress in a large subtropical lake in Florida, USA. Aquatic Botany, 78: 67-82.

Havens, K. E., D. Fox, S. Gornak, and C. Hanlon. 2005. Aquatic vegetation and largemouth bass population responses to water-level variations in Lake Okeechobee, Florida (USA). Hydrobiologia 539: 225-237.

Havens, K.E., and D.E. Gawlick. 2005. Lake Okeechobee conceptual model. Wetlands , 25(4): 908-925.

Havens, K. E., and R. T. James. 2005. The phosphorus mass balance of Lake Okeechobee, Florida: implications for eutrophication management. Lake Reservoir Manag. 21: 139148.

Heilprin, A. 1887. Explorations on the west coast of Florida and in the Okeechobee Wilderness. Transactions of the Wagner Free Inst. of Science of Philadelphia 1: 365-506
+ 21 plates.

Hopson, M.S. and P.V. Zimba. 1993. Temporal variation in the biomass of submersed

macrophytes in Lake Okeechobee, Florida. J. Aquat. Plant Manage. 31: 76-81.

James, R.T. and K.E. Havens. 2005. Outcomes of Extreme Water Levels on Water Quality of Offshore and Nearshore Regions in a Large Shallow Subtropical Lake. Archiv fur Hydrobiologie, 163: 225-239.

Janus, L.L., D.M. Soballe, and B.L. Jones. 1990. Nutrient budget analyses and phosphorus loading goal for Lake Okeechobee, Florida. Verh. Internat. Verein. Limnol. 24:538-546.

Jeppesen, E., J.P. Jensen, M. Sondergard, K.S. Hansen, P.H. Moller, H.U. Rasmussen, V. Norby, and S.E. Larsen. 2003. Does resuspension prevent a shift to a clear state in shallow lakes during reoligotrophication? Limno. Oceanogr., 48(5), 1913-1919.

Jeppesen, E., M. Sondergaard, J.P. Jensen, K.E. Havens, O. Anneville, L. Carvalho, M.F. Coveney, R. Deneke, M.T. Dokull, B. Foy, D. Gerdeaux, S.E. Hampton, S. Hilt, K. Kangur, J. Kohler, E.H.H.R. Lammens, T.L. Lauridsen, M. Manca, M.R. Miracle, B. Boss, P. Noges, G. Persson, G. Phillips, R. Portielje, S. Romo, C.L. Schelske, D. Straile,
I. Tatrai, E. Willen, and M. Winder. 2005. Lake responses to reduced nutrient loading– an analysis of contemporary long-term data from 35 case studies. Freshwater Biology (2005), No. 50, pp. 1747-1771.

Johnson, K.G., M.S. Allen, and K. E. Havens. 2007. A review of littoral vegetation, fisheries, and wildife responses to hydrologic variation at Lake Okeechobee. Wetlands, Vol. 27, No. 1, pp. 110-126.

Jonasson, P.M. 1969. Bottom fauna and eutrophication. In: Eutrophication: causes, consequences, correctives. Pages 274-305. Nat. Acad. Sci., Washington D.C.

Joyner, B.F. 1974. Chemical and biological conditions of Lake Okeechobee, Florida, 1969
72. United States Geological Survey, Report of Investigations No 71: 94pp.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspectives on water quality goals. Environmental Management 5: 55-68

Krebs, C.J. 1999. Ecological methodology, second edition. Addison Wesley Longman, New York.

Maki, R.P., Sharfstein, B., East, T.L., and Rodusky, A.J. 2004. Phytoplankton photosynthesis-irradience relationships in a large, managed, eutrophic subtropical lake: The influence of lake stage on ecological homogeneity. Arch. Hydrobiol. 161(2): 159180.

Maceina, M.J. 1990. Wind-related limnological variation in Lake Okeechobee, Florida. Lake and Reservoir Management, 6(1): 93-100.

Maceina, M.J. and D.M. Soballe. 1991. Wind-related limnological variation in Lake Okeechobee, Florida. Lake and Reservoir Management, 6: 93-100.

Maceina, M.J. 1993. Summer fluctuations in planktonic chlorophyll a concentrations in Lake Okeechobee, Florida: The influence of lake levels. Lake and Res. Manage. 8(1): 1-11.

McVoy, C., W. P. Said, J. Obeysekera, and J. VanArman. 2005. Pre-Drainage Everglades Landscapes and Hydrology. Peer-reviewed draft, SFWMD, West Palm Beach, FL.

Murphy, T., K. Hall and I. Yesaki, 1983. Co-precipitation of phosphate and calcite in a naturally eutrophic lake. Limnol. Oceanogr. 28: 58-67.

National Research Council (NRC). 2005. Re-engineering water storage in the Everglades – risks and opportunities. National Academies Press; Washington, D.C.

National Resources Council (NRC). 2007. Progress toward restoring the Everglades:The first biennial review. 236pp.

Pesnell, G.L. and R.T. Brown. 1977. The major plant communities of Lake Okeechobee, Florida, and their associated inundation characteristics as determined by gradient analysis. Technical Publication 77-1. South Florida Water Management District, West Palm Beach, Florida.

Phlips, E.J., F.J. Aldridge, P. Hansen, P.V. Zimba, J. Ihnat, M. Conroy, and P.R. Ritter. 1993. Spatial and temporal varability of trophic state paramters in a shallow, subtropical lake (Lake Okeechobee, Florida, USA). Arch. Hydrobiol. 128: 437-458.

Phlips, E.J., P.V. Zimba, M.S. Hopson, and T.L. Crisman. 1993. Dynamics of the plankton community in submerged plant dominated regions of Lake Okeechobee, Florida, USA. Verh. Internat. Verein. Limnol., 25, 423-426.

Phlips, E.J., F.J. Aldridge, Hanlon, C. 1994. Review of a seventeen-year chlorophyll record as it pertains to the trophic status of Lake Okeechobee, Florida, USA. Water Res. Bull. 30(2): 229-238.

Phlips, E.J., Aldridge, F.J., Hanlon, C. 1995. Potential factors limiting phytoplankton biomass in a shallow subtropical lake (Lake Okeechobee, Florida, USA). Arch. Hydrobiol. Beih
45: 117-136.

Phlips, E.J., F.J. Aldridge, P. Hansen. 1995. Patterns of water chemistry, physical and biological parameters in a shallow subtropical lake (Lake Okeechobee, Florida, USA). Arch. Hydrobiol., Spec. Issue Adv. Limnol. 45: 117-135.

Phlips, E.J., M. Cichra, K. Havens, C. Hanlon, S. Badylak, B. Rueter, M. Randall, and P. Hansen. 1997. Relationships between phytoplankton dynamics and the availability of light and nutrients in a shallow sub-tropical lake. Journal of Plankton Research, Vol. 19, No. 3, 319-342.

Pielou, E.C. 1977. Ecological diversity. John Wiley and Sons, New York.

Reddy, K.R. 1993. Lake Okeechobee phosphorus dynamics study: physico-chemical processes in the sediments, Vo. 2. University of Florida Soil Sciences Dept., Final Report to South Florida Water Management District.

Reddy, K.R., Y.P. Sheng, and B.L. Jones. 1995. Lake Okeechobee Phosphorus Dynamics Study. West Palm Beach, FL

Richardson, J.R., and T.T. Harris. 1995. Vegetation mapping and change detection in the

Lake Okeechobee marsh ecosystem. Arch. Hydrobiol Beih. 45: 17-39.

Richardson, J.R. and T.T. Harris. 1995. Vegetation mapping and change detection in the Lake Okeechobee marsh ecosystem. Pp. 17-39 in: Arch. Hydrobiol. Spec. Issues Advances in Limnology, Ecological studies on the littoral and pelagic systems of Lake Okeechobee, Florida (USA).

Richardson, J.R., T.T. Harris, and K.A. Williges. 1995. Vegetation correlations with various environmental parameters in the Lake Okeechobee marsh ecosystem. Arch. Hydrobiol. Beih. 45: 41-46.

Rodusky, A.J., B. Sharfstein, T.L. East, R.P. Maki. 2005. Comparison of Three Methods to Collect Submerged Aquatic Vegetation in a Shallow Lake. Environmental Monitoring and Assessment, 110: 89-97.

Scheffer, M. 1989. Alternative stable states in eutrophic shallow freshwater systems: a minimal model. Hydrobiological Bulletin 23: 73-85.

Scheffer, M., M. Van den Berg, A. Breukelaar, C. Breukers, H. Coops, R. Doef, and M.L. Meijer. 1994. Vegetated areas with clear water in turbid shallow lakes. Aquat. Bot. 49: 193-196.

Scheffer, M. 1998. Ecology of Shallow Lakes. Chapman and Hall, New York.

Smith, J.P., J.R. Richardson and M.W. Callopy. 1995. Foraging habitat selection among wading birds (Ciconiiformes) at Lake Okeechobee, Florida, in relation to hydrology and vegetative cover. Arch. Hydrobiol., Spec. Issue Adv. Limnol. 45: 247-285.

South Florida Water Management District and Florida Department of Environmental Protection. 2007. 2007 South Florida Environmental Report. Online at https://my.sfwmd.gov/portal/page?_pageid=2714,14424181,2714_14424212&_dad=port al&_schema=PORTAL

South Florida Water Management District. 2006. Lake Okeechobee Conceptual Ecological Model (DRAFT UPDATE 12/12/2006).

South Florida Water Management District, Florida Department of Environmental Protection, and Florida Department of Agriculture and Consumer Services. 2007. Lake Okeechobee Protection Plan Evaluation.

Steinman, A.D., R.H. Meeker, A.J. Rodusky, W.P. Davis, and S.J. Hwang. 1997. Ecological properties of charophytes in a large subtropical lake. J.N. Am. Benthol. Soc. 16: 781-793.

Steinman, A.D., K.E. Havens, H.J. Carrick and R. VanZee. 2002a. The past, present, and future hydrology and ecology of Lake Okeechobee and its watershed. In: Porter, J. and K. Porter (Eds.), The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: an Ecosystem Sourcebook. CRC Press, Boca Raton, FL, pp. 19-37.

Steinman, A.D., K.E. Havens and L. Hornung. 2002b. The managed recession of Lake Okeechobee, Florida: Integrating science and natural resource management. Cons. Ecol. 16(2): 1-16.

Systat Software, Inc. 2002. Systat 10.2. Richmond, CA.

Thienemann, A. 1925. Die Binnengewasser Mitteleuropas. Eine limnlogische Einfuhrung. Die Binnengewasser I:1.255.

US Fish and Wildlife Service. 1996. Black bass fishing in the U.S. Addendum to the 1996 National Survey of Fishing, Hunting and Wildlife-Associated Recreation. Report 96-3, Washington, D.C.

Vermaat, J.E., L. Santamaria, and P.J. Roos. 2000. Waterflow across and sediment trapping in submerged macrophyte beds of contrasting growth form. Arch. Hydrobiol. 148: 549562.

Warren, G.L. 1991. Invertebrate communities of Lake Okeechobee. In: Florida Game and Fresh Water Fish Commission, Lake Okeechobee–Kissimmee River–Everglades resource evaluation. Federal Wallop–Breaux completion report, project no. F-52.

Warren, G.L., M.J. Vogel, and D.D. Fox. 1995. Trophic and distributional dynamics of Lake Okeechobee sublittoral benthic invertebrate communities. Arch. Hydrobiol. Beih. 45:317-332.

Warren, G.L., T.J. Ferring, and D.A. Hohlt. 2006. Sublittoral zone benthic invertebrate community structure as an indicator of nutrient influence and overall health of the Lake Okeechobee ecosystem. Year One Annual Report, Contract No. CP051040, Task 5.1 Deliverable, June 15, 2006, FWRI File Code: F2551-05-A1, Aquatic Invertebrate Resources Unit, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, Dept. of Fisheries and Aquatic Sciences, University of Florida, Gainesville, Florida.

Wetzel, R.G. 1983. Limnology, second ed. Saunders College Publ., New York, NY.

Wiederholm, T. 1980. Use of benthos in lake monitoring. Journ. Water. Poll. Control Fed. 52:537-547.

Wright, J. O. 1911. Report on the drainage of the Everglades of Florida. Pp. 140-180 in: Everglades of Florida - Acts, reports, and other papers, state and national, relating to the Everglades of the State of Florida and their reclamation. Senate Document Number 89, 62nd Congress, 1st Session. Reprinted in South Florida in Peril, A Study in Bureaucratic Self-Deception. Florida Classics Library; Port Salerno, Florida.

Xiang-Dong YANG, Xu-Hui DONG, Guang GAO, Hong-Xi PAN, Jing-Lu WU (2005) Relationship Between Surface Sediment Diatoms and Summer Water Quality in Shallow Lakes of the Middle and Lower Reaches of the Yangtze River Journal of Integrative Plant Biology 47 (2), 153–164.

Zimba, P.V., M.S. Hopson, J.P. Smith, D.E. Colle, and J.V. Shireman. 1995. Chemical composition and distribution of submersed aquatic vegetation in Lake Okeechobee, Florida (1989-1991). Arch. Hydrobiol. Beih. 45: 241-246.

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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_cmn 459 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