ASSESSING FACTORS THAT INFLUENCE LAKE WATER-COLUMN TOTAL PHOSPHORUS VARIABILITY IN FLORIDA'S NUTRIENT ZONES

By

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To my Grandpa

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Bachmann et al. (2012) created six statistically defensible total phosphorus (TP) zones to be considered in establishing numeric nutrient criteria for Florida lakes. However, some of the zones have wide ranges of TP concentrations (e.g., Zone TP3: TP range: 3 µg/L to1857 µg/L). The primary objective of this study was to examine lakes in the 10% and 90% quantiles of TP concentrations for each TP zone (TP1 to TP6) to determine which limnological (physical, chemical and/or biological) factors most influence in-zone TP variability. The 10% and 90% quantiles were chosen because they represent the "oligotrophic" and "eutrophic/hypereutrophic" lakes, respectively, in each TP zone. Of twelve limnological factors analyzed, aquatic macrophyte abundance, true color and surficial hydrologic connectivity seem to be the most important limnological factors affecting TP variability in Florida's nutrient zones. Understanding which limnological factor or factors explain a significant part of TP variability in Florida lakes is critical to developing an appropriate nutrient management strategy for individual Florida lakes.

CHAPTER 1
INTRODUCTION

In the United States, quantification of nutrient concentrations in water bodies (e.g., lakes) is the primary basis for determining progress towards reaching the goals of the Clean Water Act to protect the designated uses of water bodies (USEPA 2000; Reckhow et al. 2005). The U.S. Environmental Protection Agency (USEPA) requires states to either adopt the ambient nutrient criteria that they developed, or establish scientifically defensible numeric nutrient values, i.e., total phosphorus (TP) and total nitrogen (TN) criteria that will protect the designated uses of water bodies (USEPA 2000). In 2010, the Florida Department of Environmental Protection (FDEP) and the USEPA established numeric nutrient criteria for lakes in the state (USEPA 2010). These criteria are used to determine if a lake is nutrient impaired. If a lake violates the criteria, it is placed on the impaired list and additional investigations may be done before it is placed on the verified list. Lakes can also be placed directly on the verified list without additional study. Once a lake is placed on the verified list, total maximum daily loads for nutrients must be established.

Some limnologists in Florida, however, disagreed with the basis on which the numeric nutrient criteria were established. As such, Bachmann et al. (2012) created six total phosphorus (TP) zones (TP1-TP6) (Appendix A) to be considered in establishing alternative numeric nutrient criteria for Florida lakes. The zones were created using the Lake Regions of Florida (Appendix B), which were established through a collaborative project between USEPA, FDEP, and the University of Florida LAKEWATCH Program (Griffith et al. 1997). Griffith et al. (1997) examined how differences in physiography, hydrology, soils, climate, vegetation, and geology affect limnological properties of

Florida lakes. Regional patterns were found to exist and 47 Lake Regions were established, but the USEPA concluded that 47 Lake Regions were too cumbersome for use in regulation and management.

Therefore, Bachman et al. (2012) combined Lake Regions with similar chemical characteristics to create six TP zones in Florida. They accomplished this through an iterative process in which each Lake Region was moved from one trial zone to another until the six TP zones were formed. The distribution of TP concentrations in individual zones was statistically different from the TP distributions in each of the other zones. The six zones presented a foundation for creating quantitative numeric nutrient criteria that acknowledged natural variation in background concentrations of TP primarily as a consequence of differences in regional geology (Bachmann et al. 2012). The zones were presented to FDEP's Environmental Regulatory Commission (ERC), which is charged with approving the adoption of water quality standards (i.e., numeric nutrient criteria) for the State of Florida. The ERC believed that re-establishing numeric nutrient criteria for Florida lakes based on the nutrient zones would be too time consuming and allow the water quality of lakes experiencing nutrient impairment to further deteriorate (Canfield, pers. comm.). The ERC, therefore, passed a rule that required the FDEP to identify the nutrient zone in which a lake on the impaired list resides before the lake is placed on the verified list.

As previously mentioned, TP zones are primarily based on differences in regional geology. However, some TP zones have wide ranges of TP concentrations (e.g., Zone TP3: TP ranges from 3 μg/L to 1,857 μg/L). Based on this in-zone variability, other limnological factors besides regional geology must influence TP concentrations in

Florida's lakes. The primary objective of this study was to examine lakes in the 10% and 90% quantiles of TP concentrations for all six TP zones, and determine which limnological factors (physical, chemical, and biological) most influence TP variability in lakes residing in these quantiles for each of the six TP zones. The 10% and 90% quantiles were chosen for analysis because they represent the nutrient-poor (oligotrophic) and nutrient-rich (hypereutrophic) lakes in each TP zone. Choosing lakes at the extreme ends of the TP range in each zone increased the likelihood that statistical analyses would provide significant results regarding limnological factors that influence TP, as these lakes are likely to be more limnologically different than lakes closer to the mean TP concentrations of each zone. Determining the most influential limnological factors affecting TP concentrations will enable lake managers to protect oligotrophic lakes more effectively and remediate eutrophic/hypereutrophic lakes in need of TP management.

The limnological factors analyzed were placed into physical, chemical, and biological categories. Physical factors were primarily chosen based on the Vollenweider TP loading model (Vollenweider 1968) and included surface area (ha), mean depth (m), as well as the dynamic ratio (Hakanson 1982) (square root of surface area (km^2)/mean depth (m)), surficial hydrologic connectivity, and the Landscape Development Intensity (LDI) index (Brown and Vivas 2005). Chemical factors were chosen based on the regulatory classification system used by USEPA and FDEP to establish numeric nutrient criteria for Florida lakes and included true color (Platinum-Cobalt Units; PCU) and total alkalinity (mg/L as $CaCO_3$). Biological factors were chosen using the theory of alternative stable states, which is based on the concept that shallow, eutrophic lakes exist as either turbid, phytoplankton-dominated or clear, macrophyte-dominated waterbodies (Scheffer 1998; Jeppesen et al. 1998). As such, all biological factors analyzed were indices of aquatic macrophyte abundance or biomass and included percent area covered (PAC), percent volume inhabited (PVI), emergent zone macrophyte biomass (EB; kg/m^2), floating-leaved zone macrophyte biomass (FB; kg/m^2), and submersed zone macrophyte biomass (SB; kg/m^2). A literature review of all twelve limnological factors and their respective relationships with TP in lakes is presented in Chapter 2. Significant differences in multiple limnological factors were believed to exist between lakes in the 10% and 90% quantiles within each zone. It was hypothesized that some of these factors were the main contributors to the TP variability associated with the Bachmann et al. (2012) nutrient zones.

CHAPTER 2
LITERATURE REVIEW

Physical Factors

Physical factors were chosen primarily based on the Vollenweider TP loading model (Vollenweider 1968). The steady state model is: $TP = L/z(\sigma + \rho)$, where TP is the total phosphorus concentration of the lake in µg/L, L is the annual P load per unit area of lake surface ($mg/m^2/yr$), z is the mean depth (m), σ signifies the P sedimentation coefficient per year, and ρ represents the hydraulic flushing rate per year. Physical factors analyzed included surficial hydrologic connectivity, which may affect annual P load per unit area, sedimentation rate, and flushing rate, as well as three morphometric variables: surface area, mean depth, and dynamic ratio. Surface area and mean depth are directly accounted for in Vollenweider (1968). Dynamic ratio (Hakanson 1982) is derived from the surface area and mean depth of individual lakes and has the potential to impact sedimentation rates via sediment resuspension. The Landscape Development Intensity (LDI) index (Brown and Vivas 2005) was also analyzed to determine whether anthropogenic development around lakes and within their watersheds impacts in-lake TP concentrations.

Surficial Hydrologic Connectivity

Connectivity of lakes to other aquatic systems such as wetlands, streams, creeks, rivers, or other lakes via canals (natural or man-made) may be an important physical factor influencing TP variability in individual TP zones. Lakes receiving the majority of their water from surface inflows from wetlands, streams, rivers, or creeks are known as drainage lakes. Lakes receiving the majority of their water from direct precipitation, run-off, and/or groundwater exchange are known as seepage lakes

(Gergel et al. 1999; Saunders et al. 2000). Drainage lakes have been shown to exhibit greater mean TP concentrations relative to seepage lakes (Knowlton and Jones 1997, 2003; Pace and Cole 2002).

Drainage lakes, especially those with surficial outflows, typically have shorter water residence time relative to seepage lakes (Gergel et al. 1999). Lake size, depth, precipitation, evaporation, drainage basin size, soil and rock permeability, and hydraulic conductivity are all known to affect water residence time (Wetzel 1990). Differences in water residence time have implications for TP concentrations (Canfield and Bachmann 1981), depending primarily on the physical (surface area, mean depth, and surficial hydrologic connectivity) and biological (aquatic macrophyte abundance) characteristics of each lake. Drainage lakes are likely to experience greater P sedimentation rates than seepage lakes because P adsorbs to suspended sediments associated with inflowing water (Canfield and Bachmann 1981). These sediments then settle out of the water column effectively reducing in-lake TP concentrations. However, sediment resuspension from wind-driven waves may result in P being released from suspended sediments back into the water column (Bachmann et al. 2000), which may reduce the effectiveness of the sediment-P adsorption mechanism. Further explanations of the relationship that surficial hydrologic connectivity has with individual limnological factors analyzed in this study will be discussed within the section/sub-section of each factor.

Morphometric Factors

Surface area, mean depth (lake volume/surface area), and dynamic ratio were chosen for analysis based on the potential influence they have on in-lake TP concentrations. Differences in surface area and mean depth may be a source of TP variability in the TP zones because lakes with significantly larger surface areas and mean depths (i.e., larger volume) may experience greater dilution effects for a given areal phosphorus load than lakes with smaller surface areas and mean depths (i.e., smaller volume), located in areas with the same edaphic characteristics (i.e., TP zone) (Bachmann et al. 2000). Mean depth was shown to have a significant, negative correlation with TP concentration in a world-wide data set of lakes, whereas no significant correlation was found between TP and surface area in the same data set (Nurnberg 1996). The same study did not find a correlation between either surface area or mean depth and TP in a subset of North American lakes that included Central Ontario lakes on the Precambrian shield, southern Ontario and Quebec lakes, and lakes in sedimentary basins of the eastern United States, including some Florida lakes. Similarly, Bachmann et al. (2012) found the inclusion of mean depth did not substantially increase the amount of the TP variability explained by their model once lake regions were included.

Surface area and mean depth, however, may contribute to the variability in TP concentrations within each Florida TP zone because of the impact they have on sedimentation rates and sediment resuspension (Hakanson 1982; Carrick et al. 1993). Deep lakes (>5 m) have higher P sedimentation rates than do shallow lakes (<5 m) (Canfield and Bachmann 1981). Release of P from sediments into the water column occurs during sediment resuspension or movement of dissolved forms via turbulent wind mixing and diffusion (Syers et al. 1973). Many Florida lakes are shallow (<5 m) in relation to their surface area, which creates the potential for significant sediment resuspension by wave disturbance (Bachmann et al. 2000). Dynamic ratios can be used to determine the frequency and extent of sediment resuspension in lakes (Hakanson

1982). Dynamic ratio is the square root of the surface area (km^2) divided by the mean depth (m) of a lake. Bachmann et al. (2000) found the entire lakebed is subject to wave disturbance, at least some of the time, for lakes with a dynamic ratio > 0.8. They detected statistically significant, positive correlations between dynamic ratio and water column TP concentrations.

Changes in lake levels have the potential to affect the extent and frequency of sediment resuspension because of changes in surface area and mean depth (i.e., dynamic ratio). Individual lakes have exhibited increased water-column TP concentrations when the extent and frequency of lakebed sediment resuspension increases because of lower water levels (Noges et al. 1998; Nagid et al. 2001). These changes can also affect hydrologic connectivity (Hanrahan et al. 2010), water residence time, and sedimentation rates. The influence of lake level changes on other variables (e.g., true color and aquatic macrophyte abundance) is discussed within specific sub-sections. In Florida lakes, water level fluctuations in general are not good predictors of TP concentrations because of inter-lake differences in the effects of lake level changes on TP concentrations (Hoyer et al. 2005). These authors stressed the importance of accounting for the limnology of individual lakes before attributing TP changes to changes in lake level. Based on the results from studies on morphometric variables and TP (Canfield and Bachmann 1981, Nurnberg 1996, Bachmann et al. 2000, Hoyer et al. 2005), it is believed these physical factors have the potential to account for some of the in-zone TP variability associated with the Bachmann et al. (2012) nutrient zones.

Landscape Development Intensity Index

The Landscape Development Intensity (LDI) index is a land-use-based index of human development used to determine the level of potential anthropogenic impacts on watersheds of various waterbodies, including lakes (Brown and Vivas 2005). It is derived from GIS-based land-use data and a development-intensity metric determined by energy use per unit area within a 100-m buffer zone around a waterbody. Land uses and their associated LDI coefficients are presented in Appendix C. Bachmann et al. (2012) looked for correlations between TP and LDI in the lakes used to create the six TP zones. They also ran another correlation, taking regional differences (i.e., Lake Regions of Florida) into account. They found no significant correlation between TP and LDI for either analysis. Similar results were found regarding LDI and water quality in another study in Florida (Fore 2005). These studies did not compare LDI between nutrient-poor and nutrient-rich lakes, which has the potential to show impacts of anthropogenic development on in-lake TP concentration.

Chemical Factors

Chemical factors were chosen based on the regulatory classification system used by the USEPA and FDEP to establish numeric nutrient criteria for Florida lakes (USEPA 2010). Florida lakes were placed into three groups based on long-term geometric mean lake true color (Platinum-Cobalt units; PCU) and total alkalinity (mg/L as $CaCO_3$): $\geq$ 40 PCU, < 40 PCU and $\geq$ 20 mg/L $CaCO_3$, and < 40 PCU and < 20 mg/L $CaCO_3$.

True Color

Florida's lakes exhibit a broad range of true color values because of differences in edaphic factors, hydrologic connectivity, climate, and vegetative communities in the watershed (Shannon and Brezonik 1972; Canfield and Hoyer 1988; Brown et al. 2000; Bachmann et al. 2012). Lakes become colored primarily from allochthonous inputs of dissolved organic carbon (DOC) (e.g., humic and fulvic acids) from the lake's watershed

(Hesslein et al. 1980; McDowell and Likens 1988; Schindler et al. 1992; Wetzel 1992). These inputs are primarily from precipitation run-off of leached and decomposed organic carbon in the watershed. Lakes in watersheds with wetlands commonly have higher loading of DOC, resulting in higher true color values (Gergel et al. 1999; Pace and Cole 2002). Drainage lakes typically have greater watershed/lake area ratios than seepage lakes indicating the greater potential for watershed influence on lake water quality (Gergel et al. 1999). One consequence is increased allochthonous input of DOC in drainage lakes, depending on the edaphic characteristics and extent of hydrologic connectivity in the watershed. Drainage lakes commonly have lower water residence times than seepage lakes, which decreases mineralization of DOC and increases color (Curtis and Adams 1995).

True color of a lake can also be influenced by changes in lake level (Brown et al. 2000) and can vary seasonally and/or annually, based on changes in the amount and timing of precipitation (Pace and Cole 2002). True color is likely an indicator of how changes in the amount of annual precipitation influence TP concentrations in lakes. In some lakes, color is the limiting environmental factor for growth and proliferation of aquatic macrophytes (Bachmann et al. 2002). This limiting effect is especially important with regard to the relationship aquatic macrophytes have with TP, given their ability to sequester inorganic P compounds in their tissues and thus reduce the quantity of biologically available P in the water column.

Color and TP concentrations are positively correlated in lakes from different geographic locations. Nurnberg (1996) found that color was significantly positively correlated to TP in two data sets, North American lakes and a global lakes data set.

Similar results were found for Florida lakes (Canfield et al. 1984; Bachmann et al. 2012). Both color and TP enter lakes in the same allochthonous sources and by the same mechanisms (e.g., decomposed plant materials from surrounding watershed flushing into lake) and, thus, quantitative changes in both parameters can be influenced by similar processes.

Total Alkalinity

Total alkalinity and TP have been shown to be effective indicators of lake productivity (Moyle 1949), although total alkalinity is now regarded as less important as an indicator because of increased understanding of TP-chlorophyll relationships (Dillon and Rigler 1974; Brown et al. 2000). Florida lakes encompass a broad range of alkalinities, primarily because of differences in regional geology, edaphic factors, and hydrology (Shannon and Brezonik 1972; Brenner and Binford 1988; Canfield and Hoyer 1988; Bachmann et al. 2012). Lakes in north Florida and on the ridges in the center of the peninsula are located in sandy soils and are characterized by low dissolved solids, pH, hardness, and alkalinity. Lakes in the broad valleys of peninsular Florida are more likely to receive groundwater that has had contact with carbonate rocks, which may result in higher dissolved solids, pH, hardness, and alkalinity (Canfield and Hoyer 1988). Although most lakes in Florida formed naturally by dissolution of the underlying limestone (Shannon and Brezonik 1972), man-made marl lakes are present, especially in south Florida. Marl lakes are characterized by high bicarbonate alkalinity, calcium concentration and pH, as well as extensive calcium carbonate deposits (Otsuki and Wetzel 1972). It is likely that the $CaCO_3$-P adsorption-precipitation mechanism influences TP concentrations in these lakes, but examination of this relationship in marl lakes was not undertaken.

Although calcareous soils are found throughout the state, a relatively small number of Florida lakes are spring-fed. Most Florida lakes receive the majority of their water directly from rainfall or surface/subsurface run-off (seepage) originating in sandy, non-calcareous soils and are separated from the underlying limestone by thick beds of non-calcareous clays (Shannon and Brezonik 1972; Canfield and Hoyer 1988). This explains why many are softwater systems (Brenner and Binford 1988; Canfield and Hoyer 1988). Nurnberg (1996) found that watersheds of hardwater lakes leach greater amounts of P into their lakes, resulting in higher in-lake TP concentrations in hardwater lakes relative to softwater lakes. Drainage lakes may show higher alkalinity relative to seepage lakes if the input water was in contact with calcareous soils in the watershed and if the watershed of the drainage lake is larger than that of a comparable seepage lake (Schindler 1988).

Alkalinity may directly affect in-lake TP concentrations through an adsorption-precipitation mechanism. In lakes with high pH (>8), inorganic orthophosphate (PO_4), a component of TP, has been shown to adsorb to calcite ($CaCO_3$) and precipitate out of the water column (Otsuki and Wetzel 1972; Syers et al. 1973; Murphy et al. 1983; Christophoridis and Fytianos 2006). As phosphates precipitate and settle onto the sediments, measurable TP in the water column decreases. If the sediments are aerobic and not prone to resuspension, the precipitated phosphates remain in the sediments and long-term mean water-column TP concentrations decrease (Syers et al. 1973).

Alkalinity has a weak, positive correlation with TP in Florida lakes (Bachmann et al. 2012). This indicates that most Florida lakes likely do not have the proper pH, total alkalinity, and/or conditions at the sediment-water interface for adsorption-precipitation mechanisms to influence in-lake TP concentrations (Otsuki and Wetzel 1972; Syers et al. 1973; Murphy et al. 1983; Christophoridis and Fytianos 2006). Similar correlation results were found for a set of North American lakes and a set of world-wide lakes (Nurnberg 1996).

Biological Factors

All biological factors investigated in this study were related to aquatic macrophyte abundance or biomass and include PAC, PVI, EB, FB, and SB. Aquatic macrophytes were chosen based on the theory that shallow, nutrient-rich lakes occur in two alternative stable states (Scheffer 1998; Jeppesen et al. 1998). One state is turbid and has low water clarity and high phytoplankton biomass, whereas the other is a clear-water state dominated by aquatic macrophytes. Changes in lake level are considered to be a major factor that influences the mechanisms that drive a switch from one stable state to the other (Scheffer and Jeppesen 1997; Blindlow et al. 1997). Aquatic macrophyte abundance may increase with decreasing water level (Blindlow et al. 1993; Scheffer 1998) or decrease with increasing water level (Havens et al. 2004).

Aquatic macrophytes are known to sequester large quantities of P from the water column, either through direct uptake (Van Donk et al. 1989; Burkholder et al. 1990; Hansson 1990) or by other means such as sediment stabilization (Vermaat et al. 2000), increased particle settling (Brenner et al. 1999; Kufel and Kufel 2000), and co-precipitation of P with calcium at high pH (Murphy et al. 1983). Macrophyte photosynthesis can increase the pH of a lake by uptake of CO_2, which impacts pH-dependent P release at the sediment water interface (Rorslett 1985). Previous studies have analyzed the effects that PAC, PVI and biomass of different growth forms of aquatic macrophytes (i.e., emergent, floating-leaved, and submersed) have on water column TP concentrations (Canfield et al. 1983; Graneli and Solander 1988; Bachmann et al. 2002). Bachmann et al. (2002) found no significant correlation between PAC, PVI, emergent macrophyte biomass, or floating-leaved macrophyte biomass and TP for a subset of Florida lakes. A significant weak, negative correlation was observed between submersed macrophyte biomass and TP, which indicates the greater potential for submersed plants to decrease TP concentrations in lakes relative to emergent and floating-leaved plants.

Emergent and floating-leaved macrophytes commonly possess large, perennial rhizomes, which are used to store carbohydrates, whereas submersed macrophytes typically have finer roots (Graneli and Solander 1988). Emergent and floating-leaved macrophytes primarily obtain their nutrients from sediments, using their robust root structures (Hutchinson 1975). Wetzel (1975) found that submersed plants actively absorb nutrients through their leaves; more recent studies, however, found they absorbed nutrients primarily from the sediments (Carignan and Kalff 1980; Carignan 1982; Graneli and Solander 1988). Submersed macrophytes lack substantial supporting tissues (e.g., cellulose), and thus decompose more rapidly than emergent and floating-leaved macrophytes, upon death (Twilley et al. 1986).

The ability of submersed macrophytes to sequester P from the water column over large temporal scales may not be as great as it is for emergent and floating-leaved macrophytes because submersed plants are more prone to light-limitation from increased dissolved organic carbon and/or phytoplankton concentrations, than are other macrophyte groups (Bachmann et al. 2002). An increase in turbidity may begin a shift from a clear-water state to a turbid state, depending on the extent and duration of light-limitation on submersed plants. Emergent macrophyte biomass is not as susceptible to changes in water transparency because photosynthesis is primarily carried out in structures above or at the water surface.

The distribution and translocation of P in different structures (e.g., shoots, leaves, and roots) of emergent macrophytes has implications regarding inorganic P release into the water column during decomposition. P is translocated to rhizome/root structures as leaves and shoots begin to senesce (Davis and van der Walk 1983; Graneli 1984; Morris and Lajtha 1986; van der Linden 1986). Thus, decomposition of emergent macrophytes contributes less inorganic P compounds to the water column compared to that of submersed macrophytes. P release from living aquatic macrophytes is not viewed as quantitatively important regarding water column TP concentrations (van der Linden 1986; Graneli and Solander 1988).

Losses of P, primarily in the form of inorganic P compounds, from decaying macrophytes are more rapid during the initial stages of decomposition, as a consequence of leaching (Landers 1982; Graneli and Solander 1988). Rates of nutrient leaching display a positive relationship with initial tissue P concentrations (Carpenter 1980), which has implications regarding comparisons of the impacts of aquatic macrophytes on TP between oligotrophic and eutrophic lakes. Nutrient concentrations in the biomass of aquatic macrophytes are typically greater in eutrophic lakes compared to oligotrophic lakes (Graneli and Solander 1988). Various factors impact macrophyte litter decomposition and the amount of P released can vary between lakes of similar TP concentrations because of differences in redox conditions and microbial processes at the sediment-water interface (Graneli and Solander 1988).

Aquatic macrophyte density has been shown to affect flushing and sedimentation rates of organic matter, which has the potential to impact water column TP (Brenner et al. 1999). This also relates to potential differences in the effects of hydrologic connectivity on TP concentrations in lakes with or without substantial aquatic macrophyte biomass. Aquatic macrophytes can disrupt the development of surface waves and reduce water movements within their beds, which can limit the extent and frequency of sediment resuspension in a lake (Jackson and Starrett, 1959; Hamilton and Mitchell, 1996). Dense beds of submersed macrophytes have been found to decrease flow velocity and thus increase water residence time, trap suspended particles and increase sedimentation rates, and absorb inorganic nutrients in drainage lakes of the Upper St. John's River Basin, Florida, effectively reducing the downstream transportation of organic matter and nutrients (Brenner et al. 1999). Oligotrophic lakes typically have slower biomass turnover rates and higher densities of submersed biomass relative to eutrophic lakes, which results in greater sediment oxidation and increased P retention in littoral sediments (Jaynes and Carpenter, 1986). I infer that oligotrophic lakes with extensive macrophyte beds are less subject to extensive sediment resuspension relative to eutrophic lakes that lack similar macrophyte densities, which indicates the compounding effects aquatic macrophytes can have on TP concentrations in lakes.

CHAPTER 3
METHODS

Data Collection

Water chemistry, surface area, and LDI data were obtained directly from the dataset used to create the six TP zones in Bachmann et al. (2012) (n=1,387). A total of 274 lakes, representing all six TP zones (Bachmann et al. 2012) were analyzed in this study. The 274 lakes were selected by analyzing the distributions of TP concentrations for lakes in each TP zone. Lakes whose reported long-term mean TP concentrations placed them in the 10% and 90% quantiles of each zone were selected for analysis. TP1 was not statistically analyzed because the lower 10% and upper 90% quantiles were each represented by only one lake. As mentioned, the 10% and 90% quantiles were chosen because they represented the most nutrient-poor (i.e., oligotrophic/mesotrophic, depending on zone) and most nutrient-rich (i.e., eutrophic/hypereutrophic, depending on zone) lakes, respectively, in each TP zone. Thirty-nine of the 47 Lake Regions of Florida established by Griffith et al. (1997) were represented by this subset of 274 lakes. However, 10 of the 39 Lake Regions were only represented by one lake.

Water chemistry data for most lakes (n=251) were collected by the Florida LAKEWATCH program using methods outlined in Canfield et al. (2002), and includes data from lakes used in creating the Lake Regions of Florida (Griffith et al. 1997). For the remainder of the lakes (n=23), water chemistry data were collected by FDEP. Although LAKEWATCH and FDEP collection and analytical procedures are not identical, comparison studies yielded no statistically significant differences between the water chemistry data obtained by the two procedures (Canfield et al. 2002; Hoyer et al.

2012). Bachmann et al. (2012) used all years of data through May 2009 for their water chemistry data so the cut-off for all other data used in this study (e.g., aquatic macrophyte abundance indices) was December 2009. A total of 11,280 monthly sampling events are covered in this study. The number of monthly samples for individual lakes ranged from 1 to 239, with a mean of 45, median of 22, and standard deviation of 54. Monthly samples are only representative of Florida LAKEWATCH sampled lakes, as such data were not available for the FDEP lakes in the Bachmann et al. (2012) dataset. To obtain mean monthly concentrations for water chemistry variables, the mean monthly value for the sampling stations (which ranged from 1 to 6 depending on lake surface area) was calculated for individual sampling dates. Annual means were then calculated from the monthly means and annual means were averaged to get the long-term mean for each lake.

Physical Factors

Surficial Hydrologic Connectivity

Google Earth©, in conjunction with Google Maps©, was used to determine whether a lake should be classified as a drainage or seepage lake. Starting with Google Earth, each lake in the 10% and 90% quantiles for all six TP zones was located and marked using GPS coordinates provided by Bachmann et al. (2012). Google Earth© software enabled examination of historical imagery, which for most lakes started in the early 1990s. Most lakes had images from at least six dates ranging from the early 1990s to early 2013; however, not all years in that time span were represented and some lakes had more complete records than others. All image dates within the period of record were analyzed for each lake.

If a lake seemed to have surficial hydrologic connectivity, but the image resolution was insufficient to confirm, Google Maps© software was used to help verify presence of surficial hydrologic connectivity. Google Maps© presents maps in a similar format to ArcGIS (Environmental Systems Research Institute, Redlands, California) so surficial connectivity, either natural or man-made, can be viewed without interference from landscape characteristics (e.g., tree canopy cover). Not all surficial inputs and outputs, however, appear in Google Maps©, depending on the timing of data collection. For instance, some creeks have been completely dry for years and did not appear on Google Maps©, which stresses the importance of using both Google Earth© and Google Maps© to accurately classify each lake.

Lakes classified as drainage lakes had inflows and/or outflows in the form of wetlands, creeks, rivers, man-made canals, etc. that were visible in Google Earth© and/or Google Maps© at some point over the period of record, regardless of whether connectivity was lost as a consequence of changes in lake level and/or precipitation over time. Lakes were classified as seepage lakes if they lacked inflows and outflows. Lakes with man-made access canals that were not connected to other lakes, wetlands, rivers, etc. were classified as seepage lakes; however, very few lakes in this study had such man-made access points, so their classification as drainage or seepage lakes likely would not have changed overall results. Seepage lakes are denoted by a zero, whereas drainage likes are denoted by a one in Appendix D.

Morphometric Factors and LDI

Surface areas (ha) for the Florida LAKEWATCH lakes were obtained from Shafer et al. (1986), LAKEWATCH bathymetric maps (2003), or Google Maps©. LAKEWATCH bathymetric maps were constructed using a Trimble Global Positioning System (Trimble

Unit Pro XRS with TSC1 data logger) and a Lowrance depth finder. Map contours (Florida LAKEWATCH 2003) were created using a kriging technique in the Surfer software package (Golden Software, Golden, Colorado). Surface areas for all FDEP lakes (n=23) were obtained with Google Maps©.

Mean depths (z_{mean}) were calculated for lakes (n=47) with available volume (V) and surface area (SA) data using the formula V (m^3)/SA (m^2). Most mean depths (n=41) were calculated directly from surface area and volume outputs from the Surfer software package used in creating the Florida LAKEWATCH bathymetric maps (Florida LAKEWATCH 2003). Surfer provided three volume estimates based on rules used to calculate volume (i.e., Trapezoidal Rule, Simpson's Rule, and Simpson's 3/8 Rule). Volumes calculated by the Trapezoidal Rule were chosen as the standard because they were typically the middle value of the three estimates. Mean depths for the other six lakes (all in Orange County) were obtained directly from Clark et al. (2008), which were also calculated using the formula above. Dynamic ratios (Hakanson 1982) for these 47 lakes were calculated as square root of SA (km^2)/z_{mean} (m). LDI values used in Bachmann et al. (2012) were provided by FDEP and calculated according to the procedures of Brown and Vivas (2005). LDI values for 216 of 274 lakes in this study were obtained from the Bachmann et al. (2012) dataset.

Chemical Factors

Water chemistry data obtained from Bachmann et al. (2012) included mean TP concentrations (μg/L), mean true color (PCU), and mean total alkalinity (mg/L as $CaCO_3$). TP concentrations were determined using procedures of Murphy and Riley (1962), with a persulfate digestion (Menzel and Corwin 1965). True color values (PCU) were determined spectrophotometrically after filtration through a Gelman class A-E filter

(Platinum-Cobalt Modified 2120C Spectrophotometric Method; A.P.H.A. 2005). Florida LAKEWATCH employed a Spectronic 401 single beam spectrophotometer set at 465 nm to determine absorbance values. The following range of standards was made from a certified standard Platinum-Cobalt Solution to generate a linear regression from which sample values were calculated: blank, 5, 10, 15, 20, 25, 30, 40, 50, 75, 100, 125, and 150 PCUs. If results were outside the range of standards, samples were brought into the standard range by dilution and reanalyzed.

Total alkalinity (mg/L as $CaCO_3$) was determined by titration with 0.02 N sulfuric acid (2030 Alkalinity, APHA 2005). All unknown samples were titrated to a pH of 4.7 to determine if samples were low or high alkalinity. If <1 mL of 0.02 N sulfuric acid was used to reach this endpoint, the sample was considered low alkalinity and the appropriate method was applied (Method 2320B: APHA 2005). If >1 mL of titrant was used to reach a pH of 4.7, the sample was classified as high alkalinity and titration continued to pH 4.5 to standardize titrations and avoid interference from silicates, phosphates, and other materials. Detection limits (MDL) based on the method employed ranged from 0.0 to 500.0 mg/L, with a relative accuracy of +/- 1.0 mg/L.

Biological Factors

The five biological factors used in this study were indices of aquatic macrophyte abundance/density and included percent area covered (PAC), percent volume inhabited (PVI), emergent zone macrophyte biomass (EB, in kg/m^2), floating-leaved zone macrophyte biomass (FB, in kg/m^2), and submersed zone macrophyte biomass (SB, in kg/m^2). All five indices were measured by Florida LAKEWATCH professionals once a year during the growing season according to the protocol established by Canfield et al. (1990). Of the 274 total lakes, 70 of the 73 lakes with aquatic macrophyte data had values for all five biological variables (three lakes lacked PAC and PVI data). The 73 lakes were sampled between 1991 and 2009. Forty-eight lakes were sampled once, 15 lakes twice, 2 lakes three times, 2 lakes four times, 3 lakes five times, 1 lake six times, and 2 lakes seven times. For lakes with more than one sampling date, values for all years were averaged.

PAC and PVI for lakes sampled from 1991 to 2005 were estimated according to the methods used by Maceina and Shireman (1980). Four cross-lake transects were run with a Raytheon DE-719 recording fathometer and PAC and PVI were calculated from fathometer charts. For lakes sampled from 2006 to 2009, PAC and PVI were estimated using modified methods of Maceina and Shireman (1980). Methods were modified because of advances in sonar technology. PAC and PVI were measured using bottom sonar transect images recorded with a Lowrance LCX-28c HD sonar and GPS system during the four cross-lake transects. Random samples from all data points for each lake were taken so that lake depth and plant height, if plants were present, could be measured visually using SonarViewer.

Average above-ground wet weights of EB, FB, and SB (all in kg/m^2) were estimated for each plant zone using the criteria of Wetzel (1983). All three variables were sampled along uniformly placed transects (5 to 30, depending on the lake size) around each lake. For each transect, a single, random 0.25-m^2 sample was taken in each plant zone present. The vegetation was placed in nylon mesh bags, spun to remove excess water, and weighed to the nearest 0.1 kg. Average biomass (kg/m^2) for each vegetation zone was calculated by averaging samples from all transects. Epiphytic algae were not removed from each sample and were included in biomass estimates for all three zones.

Data Analyses

All variables were transformed in an attempt to yield a normal distribution. Different transformations were used for different variables. Normality was tested using the Shapiro-Wilk goodness of fit test. Data for mean TP (μg/L), mean true color (PCU), surface area (ha), mean depth (m), and dynamic ratio were LOG_{10} transformed. Total alkalinity (mg/L as $CaCO_3$), LDI, EB (kg/m^2), FB (kg/m^2), and SB (kg/m^2) were square root transformed. PAC and PVI were arcsine transformed because values are percentages ranging from 0% to 100% (i.e., proportions range from 0 to 1). All statistical analyses (i.e., t-test, correlation, Shapiro-Wilk, and Pearson's chi-square test) were performed using the JMP8 statistical package (SAS Institute Inc. 2000). Results were considered significant at $p \leq 0.05$.

For TP1, each quantile was represented by only one lake so statistical analyses were not possible. All variables, however, were compared between the two lakes to determine if there were any limnological differences that may account for the difference in TP concentrations. Within zones TP2 to TP6, the distribution of transformed values for each variable in the 10% and 90% quantiles were compared using a t-test assuming unequal variances to determine if significant differences existed. If significant differences existed, the geometric means for each quantile were calculated to see if the differences detected were limnologically significant. Geometric means were used because they normalize the ranges being averaged, which limits individual values from controlling the weighting, so a given percentage change in any of the values has equal effect on the geometric mean. For LDI specifically, the geometric means were compared to the values in the LDI table (Appendix C) to determine which land use classification they most closely represented. Similar analyses were run comparing drainage lakes and seepage lakes regardless of quantile/zone to determine which variables were significantly different between the drainage lakes and seepage lakes in this study. Pearson's chi-square test was used to compare the proportion of drainage lakes versus seepage lakes in the 10% and 90% quantiles for each zone because connectivity values were analyzed as ordinal variables (i.e., seepage lakes were denoted by zeros and drainage lakes were denoted by ones). Correlation coefficients (r) were determined for each of the eleven factors in relation to TP. The 10% and 90% quantiles for each zone were analyzed separately Non-transformed data for each limnological factor for all lakes are presented in Appendix D (Table D-1).

CHAPTER 4
RESULTS

Physical Factors

Descriptive statistics (N lakes, geometric mean, minimum and maximum) of all physical factors, as well as the proportion of drainage lakes for each quantile of all six TP zones are presented in Table 4-1. T-test results comparing the geometric means for surface area, mean depth, dynamic ratio, and LDI, as well as Pearson's chi-square test results comparing proportions of drainage to seepage lakes between the quantiles of each zone are presented for zones TP2 to TP6 in Table 4-2. Correlation coefficients for all physical factors and TP are presented by quantile for zones TP2 to TP6 in Table 4-3. Correlations could not be run for surficial hydrologic connectivity because it was quantified as an ordinal variable (0 denoted seepage lake, 1 denoted drainage lake). For zone TP1, differences in surface area and surficial hydrologic connectivity seem to be associated with the differences in TP concentrations. The TP1 10% quantile-lake is Owens Lake (Washington County) and the 90% quantile-lake is Trout Pond (Leon County). Owens Lake has a surface area of 35 ha, which is seven times greater than that of Trout Pond (5 ha). Also, Owens Lake was classified as a seepage lake whereas Trout Pond is considered a drainage lake. In terms of surficial hydrologic connectivity, TP3 ($p < 0.0001$) and TP4 ($p = 0.02$) were the only zones with significant differences between quantiles. For both zones, greater proportions of drainage lakes were found in the 90% quantile. In zone TP3, 75% of the 90% quantile lakes (42 of 56) were classified as drainage lakes compared to 35% of the 10% quantile lakes (20 of 58). In zone TP4, 71% of the 90% quantile lakes (34 of 48) were drainage lakes compared to 48% of 10% quantile lakes (23 of 48).

For surface area (ha), only zone TP2 demonstrated a significant difference between the quantiles ($p = 0.02$), with 10% quantile lakes having a larger geometric means (27.5 ha) than 90% quantile lakes (8.5 ha). Significant correlations ($p = 0.04$) were detected between surface area and TP ($r = -0.28$) for the 90% quantile lakes of zone TP3 and the 10% quantile lakes of zone TP5 ($p = 0.03$; $r = 0.80$). Note one correlation coefficient was negative whereas the other was positive. No significant differences in mean depth existed between the 10% quantile lakes and 90% quantile lakes in any TP zone. Correlation analysis also returned no significant correlations between mean depth and TP. Comparisons of dynamic ratios between quantiles only showed significant differences in TP4 ($p = 0.03$). Geometric mean dynamic ratios of the 10% quantile and 90% quantile lakes were 0.2 and 0.5, respectively. A significant negative correlation ($p = 0.02$) was detected between dynamic ratios and TP ($r = -0.66$) in 10% quantile lakes of TP4, but no other significant correlations were detected. Analyses of LDI between quantiles in each zone only returned significant differences in TP3 ($p = 0.01$) and TP4 ($p = 0.01$), but different quantiles had higher geometric means for the respective zones. In TP3, 90% quantile lakes had higher geometric mean LDI than 10% quantile lakes, 4.9 and 3.8, respectively. Conversely, 10% quantile lakes had higher geometric mean LDI than 90% quantile lakes in TP4, 4.5 and 3.2, respectively (see Appendix C for land uses corresponding to LDI coefficients). No correlations between LDI and TP were significant at the selected alpha-level in either quantile for any zone.

Chemical Factors

Descriptive statistics (N lakes, geometric mean, minimum and maximum) of all chemical factors for each quantile of all six TP zones are presented in Table 4-4.

Comparisons of chemical factors between quantiles (i.e., t-tests) are presented by zone in Table 4-5. Correlation coefficients for each chemical factor and TP are presented by quantile for zones TP2 to TP6 (Table 4-6). In terms of mean true color, significant differences between the 10% quantile lakes and 90% quantile lakes were detected for zones TP2 ($p = 0.003$), TP3 ($p < 0.0001$), and TP4 ($p < 0.0001$). For all three zones, the geometric means of true color for the 90% quantile lakes were greater than those of the 10% quantile lakes. In zone TP2, the 90% quantile lakes had a geometric mean of 18 PCU compared to 5 PCU for 10% quantile lakes. In TP3, 90% quantile lakes had a geometric mean of 54 PCU compared to 9 PCU for the 10% quantile lakes. In TP4, 90% quantile lakes had a geometric mean of 63 PCU compared to 15 PCU for the 10% quantile lakes. However, only zone TP3 10% quantile lakes had a significant positive correlation ($p < 0.0001$) between true color and TP ($r = 0.60$). For total alkalinity, only zone TP4 ($p = 0.03$) had significant differences between quantiles, with the geometric mean of the 90% quantile lakes being greater than the 10% quantile lakes (27.8 versus 10.3 mg/L as $CaCO_3$, respectively). No significant correlations between total alkalinity and TP were detected for the quantiles of all 6 TP zones.

Biological Factors

Descriptive statistics (N lakes, geometric mean, minimum and maximum) of all biological factors for each quantile of all six TP zones are presented in Table 4-7. Results comparing all five indices of aquatic macrophyte abundance (i.e., biological factors) between quantiles are presented by zone (Table 4-8). Correlation coefficients and p-values for each of the five biological factors with TP are presented by quantile and zone (Table 4-9). In terms of PAC, TP2 ($p = 0.05$) and TP4 ($p < 0.0001$) were the only zones with significant differences between quantiles, with the 10% quantile lakes in both zones having greater geometric means. Of the quantiles in these two zones, PAC was significantly negatively correlated (p = 0.007) to TP (r = -0.71) in TP4 10% quantile lakes only. Similar results occurred for PVI. TP2 (p = 0.002) and TP4 (p = 0.003) had significant differences between PVI with the 10% quantile lakes in each zone having greater geometric means. PVI was also significantly negatively correlated (p=0.001) to TP (r = -0.78) in TP4 10% quantile lakes only.

The PAC/PVI outlier in the TP3 90% quantile (Little Wilson, Hillsborough County, PAC=100%, PVI=48%) caused the lack of significance between quantiles in zone TP3. Lake Little Wilson has had large amounts of *Hydrilla* sp. since the early 1990s (anecdotal reports), which caused the elevated PAC and PVI relative to other TP3 90% quantile lakes. When removed from analysis, a significant difference between quantiles was observed in PAC (p = 0.005) and PVI (p = 0.05) for TP3. Similar results occurred regarding the correlation between PAC and TP in TP3 90% quantile lakes, but not for PVI and TP. When Little Wilson was removed, the PAC-TP correlation in the TP3 90% quantile became significant (p=0.01) and the negative correlation coefficient changed from -0.75 to -0.99.

For the three aquatic macrophyte zone biomass factors, significant differences were detected between quantiles for EB, but not for FB or SB. Significant differences in EB were observed in zones TP3 (p = 0.03) and TP4 (p = 0.03) with 90% lakes having greater geometric means in both zones. In TP3, 90% quantile lakes had a geometric mean of 4.9 kg/m^2 compared to 1.9 kg/m^2 for the 10% quantile lakes. In TP4, 90% quantile lakes had a geometric mean of 3.7 kg/m^2 compared to 1.7 kg/m^2 for the 10% quantile lakes No significant correlations between EB or FB and TP were found for any quantile of any TP zone. Significant negative correlation ($p = 0.05$) occurred between SB and TP ($r = -0.59$) only in 10% quantile lakes of TP3.

Surficial Hydrologic Connectivity

All of the limnological factors compared between quantiles in each TP zone were also compared between drainage and seepage lakes regardless of quantile and zone. Descriptive statistics (N lakes, geometric mean, minimum and maximum) of all limnological factors for drainage lakes and seepages lakes are presented in Tables 4-10 and 4-11, respectively. T-test results for all limnological factors compared between the two groups are presented in Table 4-12. Correlations for each individual limnological factor with TP were determined for drainage lakes and seepage lakes, respectively (Table 4-13). All lakes and their classifications are presented in Appendix D (Table D-1).

Significant differences were detected between seepage and drainage lakes for both geometric mean TP ($p < 0.0001$) and mean true color ($p < 0.0001$). Mean TP in drainage lakes had a greater geometric mean than in seepage lakes, 43 μg/L and 18 μg/L, respectively. Geometric mean true color was greater in drainage lakes than seepage lakes, 39 versus 12 PCU, respectively. Significant positive correlations between mean true color (PCU) and TP (μg/L) occurred for both drainage ($r = 0.65$; $p < 0.0001$) and seepage lakes ($r = 0.64$; $p < 0.0001$). Significant differences in total alkalinity were not observed between drainage and seepage lakes. However, significant positive correlations between mean total alkalinity and TP were detected for both drainage ($r = 0.35$; $p = 0.003$) and seepage lakes ($r = 0.33$; $p = 0.03$).

Of the three morphometric factors, only surface area ($p < 0.0001$) and dynamic ratio ($p < 0.0001$) were significantly different between drainage and seepage lakes. In terms of surface area, drainage lakes had a geometric mean of 33.7 ha, while seepage lakes had a geometric mean of 5.1 ha. A significant negative correlation ($p = 0.001$) was determined for surface area and TP ($r = -0.28$) in seepage lakes, but no significant correlation existed for the drainage lakes. In terms of dynamic ratio, drainage lakes had a larger geometric mean than seepage lakes, 0.4 and 0.1, respectively. No significant correlation regarding dynamic ratio and TP was determined for either lake type. LDI values between drainage and seepage lakes were also significantly different ($p < 0.0001$). The geometric mean LDI was greater for seepage lakes than drainage lakes, 4.8 and 3.2, respectively (see Appendix C for land uses corresponding to LDI coefficients). The ranges of LDI values for drainage and seepage lakes were quite similar (1.01-8.33 and 1.0-9.13, respectively). A significant positive correlation ($p = 0.02$) was determined between LDI and TP ($r = 0.28$) for seepage lakes, but not for drainage lakes.

Of the five biological factors analyzed, PAC ($p = 0.05$), EB ($p = 0.02$), and FB ($p < 0.0001$) were significantly different between the two lake types. In terms of PAC, drainage lakes had a lower geometric than seepage lakes (16% versus 26%, respectively). PAC and TP were significantly negatively correlated in drainage lakes ($r = -0.42$; $p = 0.005$), but not in seepage lakes. Drainage lakes had greater geometric means for both EB (3.0 kg/m^2 versus 2.0 kg/m^2) and FB (1.1 kg/m^2 versus 0.6 kg/m^2) relative to seepage lakes.

Table 4-1. Descriptive statistics of physical factors for all quantiles.

		Surface Area (ha)				Mean Depth (m)				Dynamic Ratio			
Zone	Quantile	N	Mean[a]	Min[b]	Max[b]	N	Mean[a]	Min[b]	Max[b]	N	Mean[a]	Min[b]	Max[b]
TP1	10%	1	34.5	.	.	0	.	.	.	0	.	.	.
TP1	90%	1	5.1	.	.	0	.	.	.	0	.	.	.
TP2	10%	17	27.5	4.1	490	6	4.8	2.0	8.1	6	0.1	0.03	0.78
TP2	90%	17	8.5	0.8	55	2	3.6	3.3	4.0	2	0.1	0.08	0.23
TP3	10%	58	15.3	0.0*	2024	10	3.5	1.6	6.7	10	0.2	0.06	0.53
TP3	90%	56	7.6	0.0**	725	6	2.8	1.6	7.0	6	0.2	0.04	0.65
TP4	10%	48	13.8	0.5	370	9	2.4	0.9	5.6	9	0.2	0.05	0.53
TP4	90%	48	27.9	0.4	12518	14	2.0	0.8	5.5	14	0.5	0.04	5.24
TP5	10%	7	17.1	0.7	2964	1	.	.	.	0	.	.	.
TP5	90%	7	58.4	6.5	487	0	.	.	.	0	.	.	.
TP6	10%	7	4.0	0.7	46	0	.	.	.	0	.	.	.
TP6	90%	7	1.5	0.2	11	2	.	0.4	1.3	0	.	.	.

Note[a]: Mean indicates geometric mean calculated based on LOG_{10}

Note[b]: Min = minimum of range, Max = maximum of range

* Minimum surface area ~ 0.0000023 ha

**Minimum surface area ~ 0.04 ha

Table 4-1. Continued

		Landscape Development Intensity Index				Surficial Hydrologic Connectivity		
Zone	Quantile	N	Mean[a]	Min[b]	Max[b]	N	% Drainage	% Seepage
TP1	10%	1	1.2	.	.	1	0	100
TP1	90%	1	1.6	.	.	1	100	0
TP2	10%	14	2.6	1.0	7.2	17	35	65
TP2	90%	14	4.0	1.6	8.6	17	47	53
TP3	10%	46	3.8	1.0	8.3	58	35	65
TP3	90%	46	4.9	1.2	9.1	56	75	25
TP4	10%	38	4.5	1.0	8.6	48	48	52
TP4	90%	40	3.2	1.0	8.0	48	71	29
TP5	10%	5	3.5	2.3	6.9	7	43	57
TP5	90%	6	2.5	1.1	6.5	7	57	43
TP6	10%	4	4.6	2.0	7.6	7	0	100
TP6	90%	1	7.4	.	.	7	29	71

Note: Table is horizontal continuation from Table 4-1 on previous page
Note[a]: Mean indicates geometric mean calculated based on LOG_{10}
Note[b]: Min = minimum of range, Max = maximum of range

Table 4-2. Two tailed t-test results for all physical factors by TP zone.

	LOG_{10} Surface Area (ha)		LOG_{10} Mean Depth (m)		LOG_{10} Dynamic Ratio		SQRT LDI[a]		Surficial Hydrologic Connectivity	
Zone	p-value	Greater Mean[b]	p-value	Greater Mean[b]	p-value	Greater Mean[b]	p-value	Greater Mean[b]	p-value*	Higher % of Drainage Lakes
TP2	0.02	10%	0.32	.	0.95	.	0.09	.	0.49	.
TP3	0.10	.	0.44	.	0.80	.	0.01	90%	0.00**	90%
TP4	0.15	.	0.50	.	0.03	90%	0.01	10%	0.02	90%
TP5	0.39	.	.	.	.	.	0.47	.	0.59	.
TP6	0.24	.	.	.	.	.	.	.	0.13	.

Note: Considered significant when $p \leq 0.05$
Note[a]: LDI = Landscape Development Intensity index, SQRT = square root transformation used
Note[b]: Greater mean indicates quantile with greater geometric mean
* Pearson's chi-squared test used
** p-value <0.0001

Table 4-3. Correlation coefficients for each physical factor and total phosphorus and p-values by quantile and zone.

		LOG_{10} Surface Area (ha)		LOG_{10} Mean Depth (m)		LOG_{10} Dynamic Ratio		SQRT LDI[a]	
Zone	Quantile	p-value	r	p-value	r	p-value	r	p-value	r
TP2	10%	0.86	-0.05	0.47	-0.63	0.71	0.38	0.09	0.53
TP2	90%	0.19	0.33	.	.	.	.	0.77	-0.08
TP3	10%	0.11	0.21	0.31	-0.45	0.91	0.06	0.15	-0.22
TP3	90%	0.04	-0.28	0.12	-0.97	0.18	0.95	0.97	0.01
TP4	10%	0.84	-0.03	0.63	0.15	0.02	-0.66	0.83	0.03
TP4	90%	0.86	0.03	0.62	-0.18	0.08	-0.58	0.45	-0.12
TP5	10%	0.03	0.80	.	.	.	.	0.17	-0.97
TP5	90%	0.63	0.23	.	.	.	.	0.08	-0.76
TP6	10%	0.99	0.01	.	.	.	.	0.22	0.89
TP6	90%	0.88	0.07	.	.	.	.	.	.

Note: Correlations considered significant when $p \leq 0.05$
Note[a]: LDI = Landscape Development Intensity Index, SQRT = square root transformation used

Table 4-4. Descriptive statistics of water chemistry factors for all quantiles.

		Total Phosphorus (µg/L)				True Color (PCU)				Total Alkalinity (mg/L as $CaCO_3$)			
Zone	Quantile	N	Mean[a]	Min[b]	Max[b]	N	Mean[a]	Min[b]	Max[b]	N	Mean[a]	Min[b]	Max[b]
TP1	10%	1	2.0	.	.	1	21.8	.	.	1	0.3	.	.
TP1	90%	1	10.6	.	.	1	30.8	.	.	0	.	.	.
TP2	10%	17	3.8	2.0	5.4	14	4.8	1.8	23.0	11	12.6	0.0	47.0
TP2	90%	17	31.4	21.7	56.1	10	17.9	2.0	43.0	11	5.2	0.0	36.8
TP3	10%	58	6.2	2.5	8.1	38	8.8	1.3	86.1	28	31.9	0.0	235.5
TP3	90%	56	86.3	44.8	1857.3	43	54.3	10.7	289.5	24	25.5	0.6	184.0
TP4	10%	48	10.9	6.0	14.0	29	14.5	4.0	61.3	20	10.3	0.0	84.0
TP4	90%	48	142.7	93.9	1448.0	36	63.3	10.5	460.0	24	27.8	0.0	129.5
TP5	10%	7	16.4	8.0	25.5	4	22.8	12.8	46.6	3	37.3	17.3	125.0
TP5	90%	7	378.6	307.5	430.2	6	83.2	18.0	446.5	4	58.9	18.0	206.0
TP6	10%	7	15.7	11.6	19.3	5	21.8	12.2	75.4	2	12.1	1.2	123.0
TP6	90%	7	542.2	360.8	722.9	3	34.5	23.1	48.8	2	33.2	13.0	85.0

Note[a]: Mean indicates geometric mean calculated based on LOG_{10}

Note[b]: Min = minimum of range, Max = maximum of range

Table 4-5. Two-tailed t-test results for all chemical factors by TP zone.

	LOG_{10} True Color (PCU)		SQRT Total Alkalinity (mg/L as $CaCO_3$)[b]	
Zone	p-value	Greater Mean[a]	p-value	Greater Mean[a]
TP2	0.003	90%	0.79	.
TP3	0.00*	90%	0.86	.
TP4	0.00*	90%	0.03	90%
TP5	0.06	.	0.60	.
TP6	0.29	.	0.74	.

Note: Considered significant when $p \leq 0.05$
Note[a]: Greater mean indicates quantile with greater geometric mean
Note[b]: SQRT= square root transformation used
* p-value < 0.0001

Table 4-6. Correlation coefficients for each chemical factor and total phosphorus and p-values by quantile and zone.

		LOG_{10} Mean Color (PCU)		SQRT Total Alkalinity (mg/L as $CaCO_3$)[a]	
Zone	Quantile	p-value	r	p-value	R
TP2	10%	0.23	-0.41	0.73	0.21
TP2	90%	0.29	0.36	0.39	-0.28
TP3	10%	0.00*	0.61	0.35	-0.18
TP3	90%	0.55	0.09	0.84	0.05
TP4	10%	0.93	0.02	0.11	0.31
TP4	90%	0.42	0.13	0.47	0.30
TP5	10%	0.06	0.99	0.97	0.13
TP5	90%	0.72	0.18	0.74	-0.21
TP6	10%	0.78	0.18	.	.
TP6	90%	0.08	0.99	.	.

Note: Correlations considered significant when $p \leq 0.05$
Note[a]: SQRT= square root transformation used

Table 4-7. Descriptive statistics of biological factors for all quantiles.

		Mean Percent Area Covered (%)				Mean Percent Volume Inhabited (%)			
Zone	Quantile	N	Mean[a]	Min[b]	Max[b]	N	Mean[a]	Min[b]	Max[b]
TP1	10%	0	.	.	.	0	.	.	.
TP1	90%	0	.	.	.	0	.	.	.
TP2	10%	9	28.8	9.6	100	9	3.6	1.8	8.5
TP2	90%	6	3.7	0.0	32	6	0.6	0.0	2.2
TP3	10%	14	40.7	0.0	92	14	5.9	0.0	25.5
TP3	90%	7	12.5	4.0	100	7	2.2	0.2	48.0
TP4	10%	13	38.0	16.0	90	13	8.3	1.0	66.8
TP4	90%	19	8.2	0.0	34	19	1.4	0.0	7.0
TP5	10%	1	42.7	.	.	1	26.0	26.0	26.0
TP5	90%	0	.	.	.	0	.	.	.
TP6	10%	0	.	.	.	0	.	.	.
TP6	90%	1	43.4	.	.	1	4.9	.	.

Note: All biological factors are indices of aquatic macrophyte abundance or biomass
Note[a]: Mean indicates geometric mean calculated based on LOG_{10}
Note[b]: Min = minimum of range, Max = maximum of range

Table 4-7. Continued

		Mean Emergent Zone Biomass (kg/m^2)				Mean Floating-leaved Zone Biomass (kg/m^2)				Mean Submersed Zone Biomass (kg/m^2)			
Zone	Quantile	N	Mean[a]	Min[b]	Max[b]	N	Mean[a]	Min[b]	Max[b]	N	Mean[a]	Min[b]	Max[b]
TP1	10%	0	.	.	.	0	.	.	.	0	.	.	.
TP1	90%	0	.	.	.	0	.	.	.	0	.	.	.
TP2	10%	9	2.6	0.0	5.6	9	0.5	0	2.2	9	1.1	0.0	4.1
TP2	90%	6	1.9	0.7	4.3	6	0.7	0	1.9	6	0.3	0.0	3.9
TP3	10%	16	1.9	0.0	7.6	16	0.8	0	2.9	16	1.2	0.2	5.3
TP3	90%	7	4.9	2.7	13.8	7	0.8	0	2.1	7	0.9	0.0	9.6
TP4	10%	13	1.7	0.0	7.6	13	0.8	0	3.5	13	1.8	0.0	4.9
TP4	90%	20	3.7	0.9	6.8	20	2.0	0	5.8	20	0.7	0.0	4.6
TP5	10%	1	3.4	.	.	1	0.5	.	.	1	1.9	.	.
TP5	90%	0	.	.	.	0	.	.	.	0	.	.	.
TP6	10%	0	.	.	.	0	.	.	.	0	.	.	.
TP6	90%	1	3.0	.	.	1	0.5	.	.	1	.	.	.

Note: All biological factors are indices of aquatic macrophyte abundance or biomass
Note: Table is horizontal continuation of Table 4-7 from previous page
Note[a]: Mean indicates geometric mean calculated based on LOG_{10}
Note[b]: Min = minimum of range, Max = maximum of range

Table 4-8. Two-tailed t-test results for all biological factors by TP zone.

	ARCSINE Percent Area Covered		ARCSINE Percent Volume Inhabited		SQRT Emergent Zone Biomass (kg/m^2)[b]		SQRT Floating-Leaved Zone Biomass (kg/m^2)[b]		SQRT Submersed Zone Biomass (kg/m^2)[b]	
Zone	p-value	Greater Mean[a]	p-value	Greater Mean[a]	p-value	Greater Mean[a]	p-value	Greater Mean[a]	p-value	Greater Mean[a]
TP2	0.05	10%	0.002	10%	0.90	.	0.40	.	0.27	.
TP3	0.49	.	0.940	.	0.03	90%	0.99	.	0.79	.
TP4	0.00*	10%	0.003	10%	0.03	90%	0.13	.	0.20	.

Note: All biological factors are indices of aquatic macrophyte abundance or biomass

Note: Considered significant when $p \leq 0.05$

Note: TP1, TP5, and TP6 sample sizes insufficient for t-test

Note[a]: Greater mean indicates quantile with greater geometric mean

Note[b]: SQRT= square root transformation used

*p-value < 0.0001

Table 4-9. Correlation coefficients for each biological factor with total phosphorus and p-values by quantile and zone.

		ARCINE Percent Area Covered		ARCSINE Percent Volume Inhabited		SQRT Emergent Zone Biomass (kg/m^2) [a]		SQRT Floating-leaved Zone Biomass (kg/m^2) [a]		SQRT Submersed Zone Biomass (kg/m^2) [a]	
Zone	Quantile	p-value	r	p-value	r	p-value	r	p-value	r	p-value	r
TP2	10%	0.43	-0.45	0.22	0.63	0.75	-0.20	0.70	0.23	0.92	-0.07
TP2	90%	0.89	0.07	0.23	0.54	0.73	0.16	0.21	0.56	0.53	-0.30
TP3	10%	0.38	-0.31	0.50	-0.24	0.70	0.13	0.40	0.28	0.05	-0.59
TP3	90%	0.74	-0.75	0.86	-0.51	0.13	0.99	0.55	-0.90	0.06	-0.99
TP4	10%	0.007	-0.71	0.001	-0.78	0.96	0.02	0.13	-0.44	0.07	-0.52
TP4	90%	0.44	-0.67	0.43	-0.51	0.43	0.26	0.39	-0.28	0.44	-0.25

Note: All biological factors are indices of aquatic macrophyte abundance

Note: Considered significant when $p \leq 0.05$

Note: TP1, TP5, and TP6 sample sizes insufficient for correlation

Note[a]: SQRT= square root transformation used

Table 4-10. Descriptive statistics for limnological factors in drainage lakes.

Limnological Factor	N	Mean[a]	Min[b]	Max[b]
Total Phosphorus (μg/L)	143	42.6	2.40	1857.3
True Color (PCU)	111	38.5	1.30	460.0
Total Alkalinity (mg/L as $CaCO_3$)	74	15.7	0.00	235.5
Surface Area (ha)	143	33.7	0.03	12518.0
Mean Depth (m)	32	2.6	0.80	6.7
Dynamic Ratio	32	0.4	0.04	5.2
Landscape Development Intensity Index	120	3.2	1.00	8.3
Percent Area Covered	47	0.2	0.00	100.0
Percent Volume Inhabited	47	0.03	0.00	48.0
Emergent Zone Biomass (kg/m^2)	49	3.0	0.50	13.8
Floating-leaved Zone Biomass (kg/m^2)	49	1.1	0.00	5.8
Submersed Zone Biomass (kg/m^2)	49	0.9	0.00	8.7

Note[a]: Mean indicates geometric mean calculated by LOG_{10}
Note[b]: Min = minimum of range, Max = maximum of range

Table 4-11. Descriptive statistics for limnological factors in seepage lakes.

Limnological Factor	N	Mean[a]	Min[b]	Max[b]
Total Phosphorus (μg/L)	131	18.3	2.00	695.4
True Color (PCU)	79	12.3	1.57	172.5
Total Alkalinity (mg/L as $CaCO_3$)	56	26.1	0.00	206.0
Surface Area (ha)	131	5.1	0.00*	215.0
Mean Depth (m)	18	3.3	0.40	8.1
Dynamic Ratio	17	0.1	0.03	0.5
Landscape Development Intensity Index	96	4.8	1.00	9.1
Percent Area Covered	23	0.3	1.00	100.0
Percent Volume Inhabited	23	0.04	0.00	66.8
Emergent Zone Biomass (kg/m^2)	24	2.0	0.00	7.6
Floating-leaved Zone Biomass (kg/m^2)	24	0.6	0.00	3.3
Submersed Zone Biomass (kg/m^2)	24	1.4	0.00	9.6

Note[a]: Mean indicates geometric mean calculated by LOG_{10}
Note[b]: Min = minimum of range, Max = maximum of range
* Minimum surface area ~ 0.0000023 ha

Table 4-12. Two-tailed t-test results comparing drainage lakes and seepage lakes.

Limnological Factor	p-value	Greater Mean[a,b]
LOG_{10} Total Phosphorus (µg/L)	0.00*	D
LOG_{10} True Color (PCU)	0.00*	D
SQRT Total Alkalinity (mg/L as $CaCO_3$)	0.09	.
LOG_{10} Surface Area (ha)	0.00*	D
LOG_{10} Mean Depth(m)	0.22	.
LOG_{10} Dynamic Ratio	0.00*	D
SQRT Landscape Development Intensity Index	0.00*	S
Arcsine Percent Area Covered	0.05	S
Arcsine Percent Volume Inhabited	0.36	.
SQRT Emergent Zone Biomass (kg/m^2)	0.02	D
SQRT Floating-leaved Zone Biomass (kg/m^2)	0.00**	D
SQRT Submersed Zone Biomass (kg/m^2)	0.95	

Note: Considered significant at $p \leq 0.05$

Note[a]: Greater mean indicates lake type with greater geometric mean

Note[b]: D = drainage lakes, S = seepage lakes

* p-value <0.0001

** p-value = 0.0001

Table 4-13. Correlation coefficients and p-values for correlations between each limnological factor and total phosphorus for drainage and seepage lakes.

	Drainage Lakes		Seepage Lakes	
Limnological Factor	p-value	r	p-value	r
LOG_{10} True Color (PCU)	0.00*	0.65	0.00*	0.64
SQRT Total Alkalinity (mg/L as $CaCO_3$)	0.003	0.35	0.03	0.33
LOG_{10} Surface Area (ha)	0.26	-0.10	0.001	-0.28
LOG_{10} Mean Depth(m)	0.05	-0.37	0.10	-0.47
LOG_{10} Dynamic Ratio	0.25	0.22	0.85	-0.06
SQRT Landscape Development Intensity Index	0.27	0.10	0.02	0.28
Arcsine Percent Area Covered	0.01	-0.42	0.25	-0.29
Arcsine Percent Volume Inhabited	0.21	-0.20	0.50	-0.17
SQRT Emergent Zone Biomass (kg/m^2)	0.05	0.30	0.11	0.35
SQRT Floating-leaved Zone Biomass (kg/m^2)	0.13	0.23	0.57	0.13
SQRT Submersed Zone Biomass (kg/m^2)	0.04	-0.31	0.32	-0.22

Note: Considered significant at $p \leq 0.05$
*p-value < 0.0001

CHAPTER 5
DISCUSSION

Of the limnological factors analyzed, surficial hydrologic connectivity, true color, and aquatic macrophyte abundance (i.e., PAC and PVI) are those that most influence TP variability in Florida's nutrient zones. In terms of surficial hydrologic connectivity, differences in TP concentrations between drainage and seepage lakes were statistically significant. Based solely on TP criteria, most drainage lakes in this study would be classified as eutrophic, whereas most seepage lakes would be classified as mesotrophic (Forsberg and Ryding 1980). Drainage lakes also have a geometric mean true color value three times greater than that of seepage lakes. Similar results for TP and true color have been found in other studies comparing drainage and seepage lakes (Knowlton and Jones 1997; Gergel et al. 1999; Pace and Cole 2002; Knowlton and Jones 2003). These results indicate TP concentrations of both drainage and seepage lakes are influenced by allochthonous inputs of suspended or dissolved organic materials; however, water quality in drainage lakes is typically influenced more by watershed factors than is water quality in seepage lakes (Gergel et al. 1999), which indicates the importance of watershed management in remediating drainage lakes that have experienced water quality impairment.

Similar results regarding true color were found, comparing lakes in the 10% and 90% quantiles of certain zones, with the 90% lakes consistently having greater geometric means than 10% lakes. Both analyses of true color (i.e., in-zone quantile and drainage versus seepage lake comparisons) show true color is higher in lakes with greater TP concentrations in lakes residing in areas with similar regional geology (i.e., within a nutrient zone). The lack of significant correlations between true color and TP within (i.e., both quantiles) or among TP zones was surprising due to the significant positive correlations detected in other studies (Nurnberg 1996; Bachmann et al. 2012). However, once the individual scatter-plots from the correlation analysis were analyzed, a few outliers in each quantile of many zones likely caused the lack of consistent correlation between true color and TP.

Aquatic macrophyte abundance was also one of the most important limnological factor influencing TP variability in Florida's nutrient zones. Aquatic macrophyte abundance is believed to influence nutrient concentrations rather than nutrient concentrations controlling aquatic macrophyte abundance (Bachmann et al. 2002). Both analyses (in-zone quantile and drainage versus seepage lake comparisons, respectively) showed lakes with lower TP concentrations (i.e., 10% lakes and seepage lakes) had greater aquatic macrophyte abundance (e.g., PAC) relative to those with higher TP concentrations. These results indicated that many of the nutrient-poor, 10% lakes with substantial amounts of aquatic macrophytes had low TP concentrations relative to those in the same TP zone (i.e., regional geology) because the macrophytes and their attached periphyton were either sequestering P directly from the water column (Van Donk et al. 1989; Burkholder et al. 1990; Hansson 1990), limiting sediment resuspension (Vermaat et al. 2000), enhancing sedimentation rates (Brenner et al. 1999; Kufel and Kufel 2000), or co-precipitating P with calcium at high pH (Murphy et al. 1983), depending on the limnological characteristics of individual lakes. If macrophytes were removed, the lakes would likely show increases in nutrients over time (Canfield et al. 1983) and eventually shift to a turbid, phytoplankton-dominated state. These concepts are the basis for the theory of alternative stable states (Sheffer 1998;

Jeppesen et al. 1998), which has been demonstrated to hold true for shallow, eutrophic lakes around the globe.

Although some of the correlation coefficients for these factors (e.g., true color, PAC, and PVI) were relatively low, they are nonetheless the most important limnological factors regarding TP variability in Florida's nutrient zones. It is important to note many analyses for individual factors (e.g., PAC and PVI) were data-limited. More data must be collected to increase the sample size for these factors within the quantiles of all six TP zones. In terms of TP variability, some lakes had small numbers of water quality samples (i.e., n months and years) taken over time. One extreme outlier TP concentration was found in Snook Pond (Miami-Dade Co.) (TP3 90%; TP = 1857 μg/L), which had three months of water samples from one year comprising its long-term mean TP concentration. The other (Bystre/Hernando Co.) (TP4 90%; TP = 1448 μg/L) had only one month of water chemistry sampling used to calculate its long-term mean TP concentrations. Not only were these lakes an order of magnitude greater than the second highest TP concentrations in their respective zones; they were also an order of magnitude higher than any lake used in creating the TP zones. This situation stresses the importance of establishing a minimum number of sampling events over a designated temporal scale when using quantitative data to create regulatory classifications, especially when those data are long-term means.

Moreover, small sample sizes may also have caused the lack of consistent correlation and/or low correlation coefficients between some limnological factors (e.g., PAC and PVI) and TP within quantiles and across zones. Outliers were common for these factors as well (e.g., true color, PAC, and PVI). In many comparisons, one or two outliers caused the lack of significant differences between the quantiles of individual zones as well as the lack significant correlations between individual factors and TP within quantiles and across zones. Increasing the sample size for these factors (e.g., PAC and PVI) would likely reduce the effects of these outliers on the respective statistical analyses, which could increase the amount of TP variability explained by individual limnological factors.

Differences in sample size as well as magnitude of the differences in TP concentrations between 10% and 90% lakes in individual zones is also important. TP3 and TP4 were the TP zones in which statistically significant results were most common. Not only do these zones have the largest sample sizes, but they also have the largest differences in terms of the range of TP concentrations in the 10% and 90% lakes. Sample size limitations likely caused the lack of significant differences and correlations between lakes in TP5 and TP6 because 90% quantile lakes in these zones had TP concentrations that were an order of magnitude greater than the 10% quantile lakes, but only seven lakes were represented by the respective quantiles in each zone. These zones were also the most data-limited and there were aquatic macrophyte data for only one lake of these two zones. Lack of consistent results in TP2 were likely a consequence of small differences in the magnitude of the TP range of lakes in the 10% and 90% quantiles, which decreased the likelihood that lakes in these quantiles would have statistically detectable differences in the limnological factors.

Temporal variability in precipitation may also influence the consistency of correlation results. In Florida, precipitation patterns are caused by the Atlantic Multidecadal Oscillation (AMO), which operates on a ~20 year cycle (Enfield et al.

2001). The State of Florida has been in the midst of a drought for the last couple decades (positive AMO), which is when most of the data for the lakes in this study were collected. The decrease in precipitation has caused many lakes to experience changes in lake level and true color. Changes in lake level affect TP concentrations in Florida lakes (Hoyer et al. 2005); however, inconsistencies in the direction (i.e., increasing or decreasing) and magnitude of change were observed from lake to lake. Changes in lake level affect surficial hydrologic connectivity as inflows and/or outflows lack the quantity of water needed to maintain connectivity to lakes during drought conditions. Disruption of surficial hydrologic connectivity may contribute to inconsistent significant correlations and/or low correlation coefficients observed between true color and TP, especially in the 90% quantiles, as they were composed primarily of drainage lakes in most zones (e.g., TP3 and TP4). Lake level changes have also been shown to affect aquatic macrophyte abundance (Blindlow et al. 1993; Scheffer 1998; Havens et al. 2004).

Time lags between changes in precipitation and changes in nutrient concentrations are likely (Bigham 2012). These time lags may influence results in this study because of variation in the number of water samples collected and temporal inconsistencies in sampling frequency between lakes. Some lakes were sampled consistently once a month for 20 years, whereas others were sampled sporadically, with sampling gaps of many years in some cases. The issue of sampling frequency (Knowlton and Jones 2006) is complicated by the fact that lakes are dynamic systems and change over various time scales (e.g., day to day, month to month, year to year). It is important to note that all samples were not collected on the same day, and that all factors were not represented by the same number of samples. This lack of concurrent sampling is potentially exacerbated by the use of long-term means, especially for water chemistry samples. Long-term means remove variability in the estimates. Loss of this variability in TP for individual lakes makes it more difficult to determine statistically which limnological factors caused the observed TP variability in Florida’s nutrient zones.

Significant differences in variables between 10% and 90% quantile lakes, as well as seepage and drainage lakes, illustrate the limnological differences between lakes in these groups (i.e., lower TP versus higher TP, respectively). The effect that each of these variables had on TP has been documented in other studies and similar effects likely occurred in the lakes analyzed in this study. Variability in limnological factors from lake to lake was believed to be the most significant contributor to the lack of consistent results from zone to zone. In other words, different combinations of physical, chemical, and biological factors exist in each lake. These different combinations result in different phosphorus dynamics within each lake, i.e., lakes are “individuals.” No single limnological factor has an overriding influence on TP variability in Florida’s nutrient zones and combinations of factors must be considered if large proportions of TP variability are to be explained.

CHAPTER 6
RECOMMENDATIONS AND CONCLUSIONS

Bachmann et al. (2012) set out to create an alternative method for establishing numeric nutrient criteria for lakes throughout Florida. Although nutrient zones explained substantial amounts of TP (and TN) variability, they were initially viewed by FDEP as too variable to achieve their intended purpose (Canfield, pers. comm.). Thus, the ERC mandated that nutrient zones must be examined before a lake can be moved from the impaired list to the verified list. This study demonstrated that limnological variables account for some in-zone nutrient variability and showed the inherent difficulties of attempting to combine large numbers of experimental units (i.e., lakes) into a few regulatory classifications, e.g., nutrient zones or the USEPA/FDEP regulatory classification system.

Surficial hydrologic connectivity, true color, and aquatic macrophyte abundance were found to be the most influential limnological factors regarding TP variability in Florida's nutrient zones. Based on these findings, it is important that these factors be evaluated when attempting to verify impairment, according to the law ERC set in place. True color, which is already included in the regulatory classification system used by the USEPA and FDEP, should be included as an indicator, as it can be used to infer various limnological characteristics of individual lakes, including surficial hydrologic connectivity, watershed influence, and precipitation cycles, i.e., wet or dry cycles dictated by the AMO. Variability in TP can be explained by true color in the nutrient zones as well as lake type (drainage versus seepage). Nevertheless, differences can exist regarding the magnitude that both variables can have and the impacts that changes in allochthonous inputs have on the water quality of individual lakes, depending on their hydrologic type.

Aquatic macrophyte abundance may be the most important factor to examine when determining where to place individual lakes within a regulatory classification system. Aquatic macrophytes affect in-lake TP concentrations in many ways (i.e., direct sequestration, increasing sedimentation, decreasing sediment resuspension), which indicates their utility in minimizing the effects of increased TP loading into lakes. Canfield et al. (1983) suggested that TP contained in the tissues of aquatic macrophytes must be included when determining TP concentrations for individual lakes. The theory of alternative stable states (Scheffer 1998; Jeppesen et al. 1998) reinforces this suggestion and the importance of aquatic macrophytes with regard to nutrient management in general. Advancements in technology have enabled lake managers/researchers to assess and monitor aquatic macrophytes in ways that are inexpensive, quick, and accurate. More consistent monitoring of aquatic macrophyte abundance in a larger number of lakes is critical to understanding TP variability, especially in lakes throughout Florida.

This study would not have been possible without the robust, long-term dataset Florida LAKEWATCH obtained through cooperation with citizen scientists. It stresses the importance of consistent monitoring, which may be considered financially limiting and unimportant relative to other research and/or management projects. The use of citizen scientists is an excellent way to collect accurate data that can be used to answer broad ecological questions (Ecological Society of America 2012), especially during times when financial support from federal and state sources is decreasing. Studies have shown that no statistically detectable differences exist between data collected by Florida LAKEWATCH volunteers and data collected by professionals (Canfield et al. 2002, Hoyer et al. 2012), which further argues for research organizations to utilize properly trained citizen scientists.

Identifying the limnological factor or factors that explain the majority of TP variability in Florida lakes is critical to verifying impairment of individual lakes when nutrient zones do not explain the numeric nutrient criteria violation. If the most influential factors are not considered during this process, lakes may be incorrectly verified as impaired when they are not. It is critical that these errors be avoided so that only Florida lakes truly experiencing nutrient impairment are remediated, especially when financial resources are limited. Numeric nutrient criteria and effective limnological assessments are critical to ensuring Florida's lakes are managed to meet their designated use/uses, especially when the goal of lake managers is to protect oligotrophic lakes and remediate eutrophic lakes.

APPENDIX A
MAP OF TOTAL PHOSPHORUS ZONES OF FLORIDA

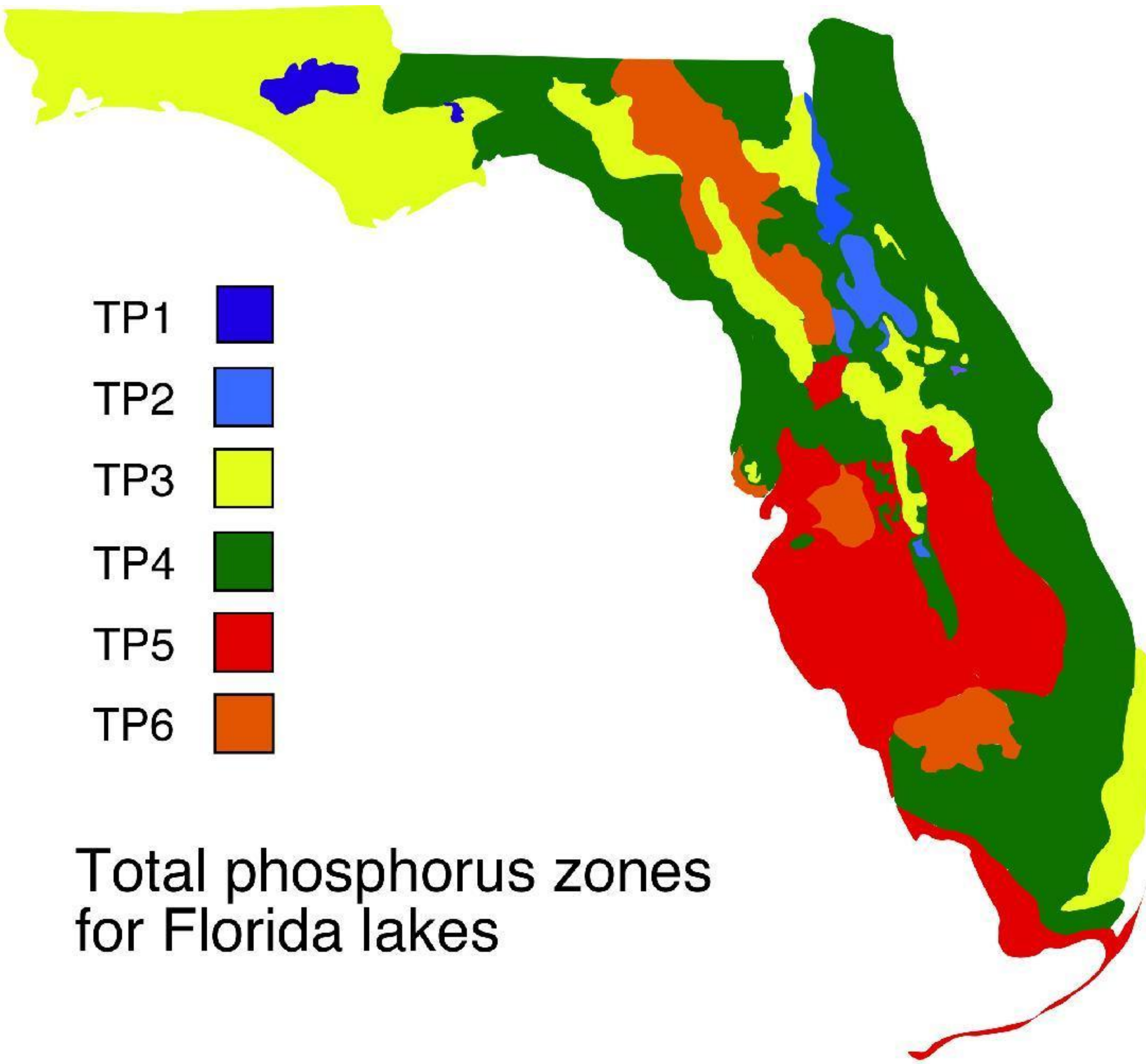

Figure A-1. Map of total phosphorus zones for Florida lakes (Bachmann et al. 2012).

APPENDIX B
LAKE REGIONS OF FLORIDA

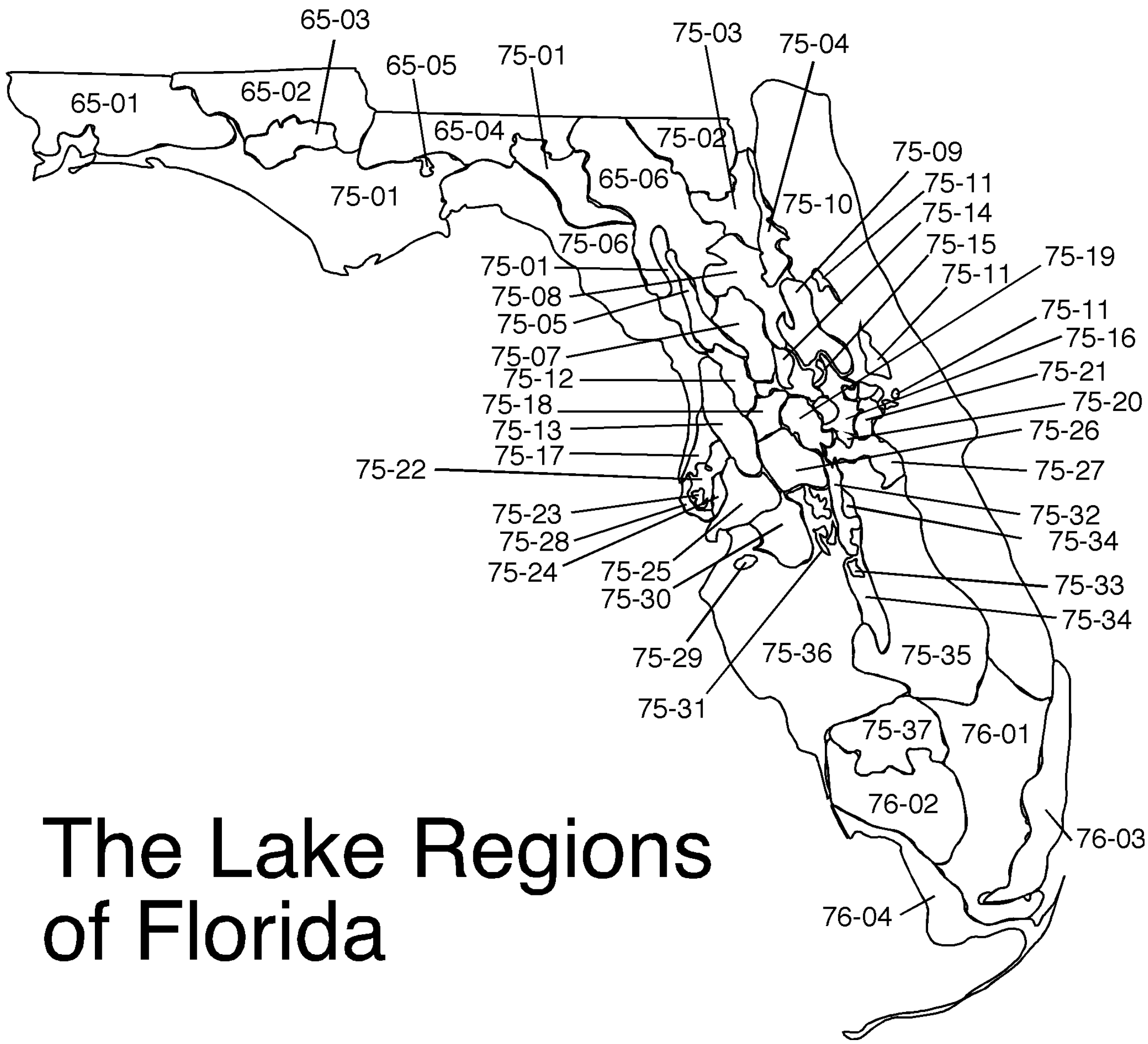

Figure B-1. Map showing USEPA’s Lake Regions of Florida (Griffith et al. 1997).

APPENDIX C
LANDSCAPE DEVELOPMENT INTENSITY COEFFIECIENTS

Table C-1. Land use classifications and corresponding Landscape Development Intensity coefficients (Brown and Vivas 2005).

TABLE II

Land use classification, nonrenewable empower density, and resulting LDI coefficients

Land use	Nonrenewable empower density (E14 sej/ha/yr)	Ln Nonrenewable empower density	LDI coefficients[a]
Natural system	0.00		1.00
Natural open water	0.00		1.00
Pine plantation	5.10	1.63	1.58
Recreational / open space – low-intensity	6.55	1.88	1.83
Woodland pasture (with livestock)	8.00	2.08	2.02
Improved pasture (without livestock)	17.20	2.84	2.77
Improved pasture – low-intensity (with livestock)	33.31	3.51	3.41
Citrus	44.00	3.78	3.68
Improved pasture – high-intensity (with livestock)	46.74	3.84	3.74
Row crops	107.13	4.67	4.54
Single family residential – low-density	1077.00	6.98	6.9
Recreational / open space – high-intensity	1230.00	7.11	6.92
Agriculture – high intensity	1349.20	7.21	7.00
Single family residential – medium density)	2175.00	7.68	7.47
Single family residential – high density	2371.80	7.77	7.55
Mobile home (medium density)	2748.00	7.92	7.70
Highway (2 lane)	3080.00	8.03	7.81
Low-intensity commercial	3758.00	8.23	8.00
Institutional	4042.20	8.30	8.07
Highway (4 lane)	5020.00	8.52	8.28
Mobile home (high density)	5087.00	8.53	8.29
Industrial	5210.60	8.56	8.32
Multi-family residential (low rise)	7391.50	8.91	8.66
High-intensity commercial	12 661.00	9.45	9.18
Multi-family residential (high rise)	12 825.00	9.46	9.19
Central business district (average 2 stories)	16 150.30	9.69	9.42
Central business district (average 4 stories)	29 401.30	10.29	10.00

[a]The LDI coefficient is calculated as the normalized (on a scale of 1.0 to 10.0) natural log of the empower densities.

APPENDIX D
ALL DATA

Table D-1. Non-transformed data for all physical factors for each lake in this study. For surficial hydrologic connectivity, 0 denotes seepage lakes and 1 denotes drainage lakes.

County	Lake	Lake Region	TP Zone	Quantile	Surficial Hydrologic Connectivity	Surface Area (ha)	Mean Depth (m)	Dynamic Ratio	LDI
Washington	Owens	65-03	1	10	0	34.5	.	.	1.2
Leon	Trout Pond	65-05	1	90	1	5.1	.	.	1.61
Putnam	Mariner	75-04	2	10	0	55.0	.	.	.
Clay	Magnolia	75-04	2	10	1	78.5	.	.	1.87
Marion	Shoesole	75-09	2	10	0	11.0	.	.	3.74
Putnam	Key pond	75-04	2	10	0	4.3	.	.	1.39
Marion	Mary	75-09	2	10	1	63.0	.	.	1.57
Putnam	Barco	75-04	2	10	0	13.0	.	.	1
Lake	Sellers	75-09	2	10	1	203.0	3.0	0.48	1.77
Highlands	Denton	75-33	2	10	0	28.0	7.1	0.07	5.26
Highlands	McCoy	75-33	2	10	0	16.0	7.6	0.05	3.67
Putnam	Long Pond	75-04	2	10	0	14.0	.	.	2.56
Clay	Sheelar	75-04	2	10	0	7.0	8.1	0.03	1.16
Clay	Lowry	75-04	2	10	1	490.0	.	.	1.3
Highlands	Byrd	75-33	2	10	1	25.0	4.5	0.11	6.43
Highlands	Lynn	75-33	2	10	0	7.3	.	.	5.73
Highlands	Isis	75-33	2	10	0	21.0	.	.	7.18
Putnam	Johntry Pond	75-11	2	10	0	4.1	.	.	.
Putnam	Cowpen	75-04	2	10	1	256.0	2.0	0.78	.
Lake	Sunshine	75-14	2	90	0	7.0	.	.	8.61
Putnam	Faye	75-04	2	90	1	1.8	.	.	6.43
Lake	Hermosa	75-15	2	90	0	11.6	4.0	0.08	4.43
Highlands	Lachard	75-33	2	90	0	6.0	.	.	3.37

Putnam	East	75-04	2	90	1	10.0	.	.	3.1
Clay	Hillcrest	75-04	2	90	0	4.2	.	.	3.31
Clay	West Smith	75-04	2	90	1	32.6	.	.	2.08
Lake	Dixie	75-14	2	90	0	0.8	.	.	7.31
Clay	Margie	75-04	2	90	0	19.8	.	.	4.83
Putnam	Twin West	75-04	2	90	1	45.0	.	.	.
Lake	Gracie	75-15	2	90	0	8.9	.	.	5.67
Clay	Pebble	75-04	2	90	0	2.1	.	.	.
Bradford	Bedford	75-04	2	90	1	44.0	.	.	2.19
Marion	Echo	75-09	2	90	0	2.0	.	.	1.55
Lake	Bear 2	75-09	2	90	1	4.1	.	.	.
Lake	Bay	75-09	2	90	1	19.6	.	.	2.99
Highlands	Little Jackson	75-33	2	90	1	55.0	3.3	0.23	7.07
Putnam	Clear	75-11	3	10	1	50.0	.	.	4.32
Miami-Dade	Lago Luna	76-03	3	10	0	39.9	.	.	8.26
Miami-Dade	Crossings	76-03	3	10	0	1.9	.	.	.
Bay	Marin	75-01	3	10	0	7.2	.	.	.
Miami-Dade	Lago Sol	76-03	3	10	0	33.9	.	.	7.39
Hillsborough	Alice	75-23	3	10	1	38.0	.	.	4.49
Miami-Dade	Pavillion 12	76-03	3	10	0	0.0	.	.	5.68
Holmes	Cassidy	65-02	3	10	1	134.0	6.7	0.17	1.04
Miami-Dade	Bonita 1	76-03	3	10	0	12.1	.	.	8.01
Miami-Dade	Devon Aire	76-03	3	10	0	4.7	.	.	.
Santa Rosa	Ski Watch	75-01	3	10	0	14.0	.	.	3.56
Okaloosa	Noname	65-01	3	10	1	7.8	.	.	3.36
Miami-Dade	E	76-03	3	10	0	37.0	5.2	0.12	.
Broward	Sterling	76-03	3	10	0	4.0	.	.	5.76
Polk	Blue	75-32	3	10	0	8.8	.	.	4.71
Volusia	Sawyer	65-05	3	10	0	31.8	.	.	3.14
Putnam	Broward	75-11	3	10	0	191.0	3.4	0.40	4.93
Miami-Dade	Esplanade	76-03	3	10	0	3.3	.	.	.

Miami-Dade	Colonial	76-03	3	10	0	1.0	.	.	7.18
Okaloosa	Roberts	65-01	3	10	1	6.6	.	.	1.45
Walton	Campbell	75-01	3	10	1	44.5	.	.	1.35
Walton	Wilson	65-01	3	10	1	12.4	.	.	1.95
Miami-Dade	GMSC	76-03	3	10	0	80.8	.	.	.
Putnam	Como	75-11	3	10	1	99.0	2.1	0.48	4.73
Volusia	Winnemissett	75-11	3	10	1	74.0	3.4	0.25	4.46
Orange	Johio	75-16	3	10	0	11.0	.	.	5.59
Volusia	Broken Arrow	75-11	3	10	0	4.2	.	.	6.88
Miami-Dade	Pineland	76-03	3	10	0	0.9	.	.	.
Alachua	Watermelon Pond	75-05	3	10	0	215.0	.	.	.
Escambia	Blue Angel	75-01	3	10	0	2.2	.	.	1.3
Walton	Western	75-01	3	10	1	69.2	1.6	0.53	1.95
Broward	Windermere	75-33	3	10	0	5.6	.	.	6.66
Miami-Dade	Lindgren	75-33	3	10	0	13.5	.	.	.
Miami-Dade	Chara	76-03	3	10	0	2.8	.	.	7.19
Hillsborough	Glass	75-15	3	10	0	7.0	.	.	3.99
Broward	Desoto	76-03	3	10	0	26.5	.	.	6.44
Lake	Arthur	75-19	3	10	0	51.0	.	.	2.3
Washington	Becton pd	65-02	3	10	0	26.7	.	.	1.68
Putnam	Banana	75-11	3	10	0	77.8	.	.	4.73
Polk	Parks	75-32	3	10	1	40.0	.	.	3.35
Hillsborough	Mound	75-23	3	10	1	32.0	.	.	3.08
Pasco	Treasure	75-24	3	10	0	4.8	.	.	4.93
Seminole	Emma	75-11	3	10	0	20.0	.	.	4.15
Miami-Dade	Royal	76-03	3	10	1	1.0	.	.	.
Miami-Dade	Oakland	76-03	3	10	0	3.7	3.1	0.06	.
Walton	Camp Creek	75-01	3	10	1	23.4	.	.	2.75
Walton	Stanley	65-02	3	10	1	40.0	.	.	5.43
Walton	Morris	75-01	3	10	1	31.7	2.9	0.19	1.37

Orange	Bessie	75-20	3	10	0	63.0	4.1	0.20	6.6
Lake	Desire	75-32	3	10	0	13.0	.	.	5.24
Lake	Grassy	75-19	3	10	0	20.4	.	.	3.01
Polk	Aurora	75-19	3	10	0	71.0	.	.	3.41
Bay	Deer Point	75-01	3	10	1	2024.0	.	.	.
Lake	Kirkland	75-19	3	10	1	115.0	.	.	3.79
Lake	Arlene	75-19	3	10	0	14.1	.	.	4.2
Orange	Down	75-20	3	10	1	483.0	6.0	0.37	5.66
Washington	Pate	65-02	3	10	1	155.8	.	.	1.86
Hillsborough	Grace	75-23	3	10	0	6.2	.	.	4.28
Hillsborough	Silver	75-24	3	90	0	7.0	.	.	7.69
Seminole	Yvonne-2	75-16	3	90	0	1.3	.	.	7.3
Pasco	Little Black	75-24	3	90	1	3.2	.	.	4.4
Hillsborough	Virginia	75-24	3	90	1	9.0	.	.	3.91
Volusia	Marie	75-11	3	90	0	6.0	.	.	7.12
Hillsborough	Jeanette	75-24	3	90	0	0.6	.	.	8.26
Orange	Olivia	75-16	3	90	1	35.0	5.1	0.12	5.8
Broward	Delevoe	76-03	3	90	0	8.0	7.0	0.04	.
Osceola	Fish	75-27	3	90	1	725.0	.	.	4.44
Citrus	Hampton	75-12	3	90	1	11.5	.	.	4.22
Volusia	Macy	75-11	3	90	1	8.0	.	.	6.46
Palm Beach	Santee	76-03	3	90	0	8.7	.	.	4.42
Hillsborough	Little Wilson	75-24	3	90	1	2.5	.	.	6.16
Hillsborough	Mid	75-24	3	90	0	0.7	.	.	7.76
Seminole	Searcy	75-11	3	90	1	4.0	.	.	2.75
Orange	Lotta	75-16	3	90	1	14.0	.	.	.
Hillsborough	Morley	75-24	3	90	0	1.7	.	.	8.31
Polk	Ida	75-32	3	90	1	34.0	.	.	3.08
Orange	Horseshoe	75-16	3	90	1	5.0	.	.	6.18
Seminole	East	75-21	3	90	1	2.8	.	.	4.64
Hillsborough	Morris	75-24	3	90	0	1.0	.	.	8.42

Hillsborough	Forest	75-24	3	90	1	0.3	.	.	.
Osceola	Center	75-27	3	90	1	120.0	1.7	0.65	3.53
Broward	Bonnet Slough	76-03	3	90	0	0.4	.	.	.
Orange	Mud	75-27	3	90	1	94.3	.	.	1.17
Palm Beach	Osborne Thompson	76-03	3	90	1	144.0	.	.	5.9
Jackson	Pond	65-02	3	90	1	14.5	.	.	.
Seminole	Bath	75-11	3	90	1	5.6	.	.	6.88
Orange	Lawne	75-16	3	90	1	63.0	2.2	0.36	4.71
Seminole	Lotus	75-16	3	90	1	46.0	.	.	4.72
Walton	Oyster	75-01	3	90	1	8.9	1.6	0.18	3.68
Wakulla	Otter	75-01	3	90	1	54.0	.	.	1.15
Hillsborough	North Pond	75-24	3	90	1	1.3	.	.	.
Seminole	Dot	75-11	3	90	0	1.6	.	.	7.54
Seminole	Amory	75-11	3	90	1	2.8	.	.	7.17
Hillsborough	Heather	75-24	3	90	1	14.2	.	.	6.89
Seminole	Minnie	75-11	3	90	1	1.0	.	.	5.8
Orange	Warren	75-27	3	90	1	44.3	.	.	1.63
Palm Beach	Clarke	76-03	3	90	1	19.3	.	.	7.68
Orange	Orlando	75-16	3	90	1	70.7	2.2	0.38	6.55
Lake	Horseshoe	75-16	3	90	1	13.1	.	.	.
Orange	Sandy	75-20	3	90	0	10.7	.	.	9.13
Orange	Marshall	75-16	3	90	1	22.0	.	.	4.57
Bradford	Rowell	75-03	3	90	1	147.0	.	.	1.35
Palm Beach	Ida	76-03	3	90	1	42.7	.	.	6.64
Hillsborough	Buck	75-23	3	90	1	12.8	.	.	3.15
Seminole	Alma	75-11	3	90	1	8.0	.	.	5.86
Broward	Alex	76-03	3	90	1	5.0	.	.	.
Orange	Floy	75-20	3	90	0	1.6	.	.	7.27
Palm Beach	Julie	76-03	3	90	1	5.6	.	.	.
Leon	Munson	75-01	3	90	1	103.0	.	.	1.97

Okaloosa	Kell-Aire	75-01	3	90	1	3.6	.	.	8.33
Okaloosa	Coleman	75-01	3	90	1	1.8	.	.	7.94
Walton	Stewart	75-01	3	90	0	6.4	.	.	6.42
Leon	Eightmile Pond	75-01	3	90	1	1.2	.	.	3.47
Miami-Dade	Snook Pond	76-03	3	90	1	0.0	.	.	.
Liberty	Mystic	65-04	4	10	0	45.0	.	.	7.15
Leon	Erie	65-04	4	10	0	21.0	0.9	0.53	3.02
Orange	Price	75-10	4	10	0	34.0	4.1	0.14	4.97
Putnam	Davis Lake	75-10	4	10	0	4.4	.	.	3.32
Polk	Otter	75-34	4	10	0	2.5	.	.	2.72
Dixie	Governor Hill	75-06	4	10	0	63.0	.	.	.
Highlands	Mills Pond	75-34	4	10	1	1.7	.	.	.
Flagler	Rodgers	75-10	4	10	1	3.9	.	.	.
Highlands	Hill	75-34	4	10	1	30.0	2.6	0.21	4.34
Polk	Dexter	75-31	4	10	1	70.0	2.2	0.38	4.6
Orange	Conway North	75-21	4	10	1	274.0	.	.	7.03
Orange	Pickett	75-10	4	10	1	339.3	5.6	0.33	2.7
St Lucie	Patricia	75-10	4	10	0	0.6	.	.	7.59
Clay	Ryan	75-10	4	10	1	1.8	.	.	.
Lake	Haines	75-08	4	10	0	3.4	.	.	.
Leon	Orchard Pond	75-32	4	10	1	85.0	.	.	1.22
St Lucie	Bel Air	75-10	4	10	0	2.6	.	.	6.83
St Lucie	Jean	75-10	4	10	0	2.4	.	.	.
St Lucie	Jeffery	75-10	4	10	0	1.2	.	.	7.37
Polk	Lucerne	75-31	4	10	0	17.0	.	.	7.32
Polk	Patrick	75-34	4	10	1	159.0	.	.	2.45
Orange	Georgia	75-21	4	10	0	33.0	3.4	0.17	6.52
Highlands	Apthorpe	75-34	4	10	1	89.0	.	.	2.5
Seminole	Crystal Bowl	75-21	4	10	0	2.9	.	.	.
Orange	Little Conway	75-21	4	10	1	370.0	.	.	6.99
Duval	Marietta	75-10	4	10	0	12.0	.	.	7.03

Lake	Purdum Pond	75-08	4	10	0	10.9	.	.	4.91
Seminole	Baptismal	75-21	4	10	1	4.9	.	.	7.59
Seminole	Noname	75-21	4	10	0	0.9	.	.	6.53
Flagler	Gore	75-10	4	10	1	34.0	.	.	1.54
Seminole	Deep	75-21	4	10	1	17.0	.	.	6.31
St Lucie	David	75-10	4	10	0	1.1	1.1	0.10	7.4
Hillsborough	Place	75-22	4	10	1	7.3	.	.	6.23
Orange	Frederica	75-21	4	10	1	29.0	.	.	5.82
Union	Palestine	75-02	4	10	1	369.0	.	.	1.01
St Lucie	Phyllis	75-10	4	10	0	0.5	.	.	7.59
Polk	Martha	75-31	4	10	0	35.2	.	.	7.29
Leon	Carr	75-19	4	10	1	254.0	.	.	1.51
St Lucie	Margaret	75-10	4	10	0	1.2	2.0	0.05	7.37
Orange	Sunset	75-21	4	10	0	12.6	.	.	8.6
Nassau	East Nassau	75-10	4	10	1	3.8	.	.	.
Hernando	Tooke	75-17	4	10	0	31.3	.	.	2.98
Seminole	Mills	75-10	4	10	1	94.0	.	.	.
Putnam	Noname	75-10	4	10	1	7.7	.	.	2.17
Seminole	Florence	75-21	4	10	0	13.0	3.5	0.10	7.35
Seminole	Hodge	75-21	4	10	0	6.7	.	.	.
Lake	Idlewild	75-08	4	10	1	10.0	.	.	4.3
Highlands	Lelia	75-34	4	10	1	67.0	.	.	3.45
Orange	Gear	75-21	4	90	0	4.0	.	.	7.95
Leon	Harriman Pond	65-04	4	90	0	0.6	.	.	.
Putnam	Crescent	75-10	4	90	1	6514.0	.	.	2.24
Alachua	Calf Pond	75-08	4	90	1	5.7	.	.	6.4
Leon	Killarney	65-04	4	90	1	32.4	.	.	7.8
Citrus	North	75-13	4	90	0	0.6	.	.	2.37
Highlands	Charlotte	75-34	4	90	1	127.0	3.2	0.35	3.81
Volusia	Ashby	75-10	4	90	1	417.0	.	.	1.35
Highlands	Glenada	75-34	4	90	1	72.0	4.1	0.21	4.69

Leon	McCord Pond	65-04	4	90	1	1.2	.	.	.
Orange	Apopka	75-08	4	90	1	12518.0	2.1	5.24	2.57
Highlands	August	75-34	4	90	1	19.9	.	.	7.28
Orange	Lawsona	75-21	4	90	0	3.0	.	.	7.53
Orange	Como	75-21	4	90	0	2.0	.	.	7.55
Orange	Walker	75-21	4	90	1	1.6	.	.	.
Orange	Wade	75-21	4	90	0	1.0	.	.	6.86
Indian River	Blue Cypress	75-10	4	90	1	2759.0	2.4	2.16	1.22
Polk	Smart	75-31	4	90	1	111.0	.	.	3.24
Gadsden	Tallavana	65-04	4	90	1	58.8	0.8	0.94	2.16
Leon	Susan	65-04	4	90	1	1.8	.	.	.
Putnam	Redwater	75-08	4	90	1	115.0	.	.	3.67
Putnam	Ross	75-10	4	90	1	33.6	1.9	0.30	1.95
Alachua	Wauberg	75-08	4	90	1	150.0	2.5	0.50	1.75
Lake	Beauclaire	75-08	4	90	1	407.0	1.5	1.38	4.77
Marion	Redwater Lake	75-08	4	90	1	72.0	.	.	1.01
Orange	Davis	75-21	4	90	0	7.0	1.4	0.19	7.23
Highlands	Huckleberry	75-34	4	90	1	48.0	.	.	3.59
Indian River	Stick Marsh	75-10	4	90	1	963.0	2.2	1.40	1.95
Leon	Goose Pond	65-04	4	90	1	0.4	.	.	.
Indian River	Farm 13	75-10	4	90	1	2744.4	.	.	1.95
Seminole	Forest	75-10	4	90	0	17.9	.	.	3.89
Citrus	Park	75-13	4	90	0	2.5	.	.	5.14
Polk	Ring	75-31	4	90	0	1.0	.	.	7.58
Hernando	Harris Pond	75-13	4	90	0	0.5	.	.	.
Seminole	Jesup	75-10	4	90	1	4051.0	.	.	1.43
Alachua	Newnan	75-08	4	90	1	3006.0	.	.	1.71
Lake	Akron	75-10	4	90	1	81.0	.	.	1.4
Flagler	Dead	75-10	4	90	1	162.0	.	.	1.26
St Lucie	Mile	75-10	4	90	0	10.0	.	.	1.54
Taylor	Jaycee Pond	75-06	4	90	0	0.4	.	.	.

Alachua	Hidden	75-08	4	90	1	4.0	5.5	0.04	2.96
Alachua	Bivans Arm	75-08	4	90	1	76.0	.	.	4.4
Highlands	Wolf	75-34	4	90	1	49.0	0.8	0.83	3.44
Alachua	Johnson Pond	75-08	4	90	1	2.7	.	.	2.96
Lake	Trout	75-08	4	90	1	44.0	2.4	0.27	5.13
Polk	Conine	75-31	4	90	1	96.0	.	.	5.06
Leon	Spring Pond	65-04	4	90	0	2.0	1.6	0.09	.
Hernando	Bystre	75-13	4	90	1	125.3	.	.	3.12
Lee	Donut	75-36	5	10	0	1.0	.	.	.
Sarasota	Loon	75-36	5	10	0	0.7	.	.	.
Osceola	Laurel	75-24	5	10	1	4.4	.	.	6.87
Hillsborough	St. Clair	75-36	5	10	0	22.8	.	.	2.28
Lee	Little Murex	75-36	5	10	0	6.3	.	.	5.74
Polk	Lowrey	75-36	5	10	1	355.8	.	.	2.49
Polk	Weohyakapka	75-35	5	10	1	2964.0	2.0	.	2.3
Lee	Murex	75-36	5	90	0	6.7	.	.	6.51
Hillsborough	Grady	75-36	5	90	1	57.0	.	.	3.57
Polk	Livingston	75-36	5	90	1	487.0	.	.	1.06
Hillsborough	Thonotosassa	75-25	5	90	1	331.0	.	.	4.01
Hillsborough	Fantasia	75-36	5	90	0	16.3	.	.	2.18
Lee	Dunes	75-36	5	90	0	6.5	.	.	.
Sarasota	Upper Myakka	75-36	5	90	1	354.7	.	.	1.07
Pinellas	Skipper	75-28	6	10	0	2.0	.	.	.
Columbia	Jeffery	65-06	6	10	0	46.0	.	.	2.03
Pinellas	Hewitt	75-28	6	10	0	1.5	.	.	.
Pinellas	Harbor	75-28	6	10	0	15.0	.	.	5.38
Pinellas	Fusion	75-28	6	10	0	0.7	.	.	.
Pinellas	Spring	75-28	6	10	0	6.6	.	.	5.62
Alachua	Meta	65-06	6	10	0	2.0	.	.	7.55
Polk	Little Bass	75-30	6	90	0	2.0	0.4	.	.
Pinellas	Flamingo	75-28	6	90	0	0.7	.	.	.

Alachua	Alice	65-06	6	90	1	11.0	1.3	.	.
Pinellas	Egret	75-28	6	90	0	0.2	.	.	.
Pinellas	Heron	75-28	6	90	0	0.6	.	.	.
Pinellas	Placido	75-28	6	90	0	5.3	.	.	.
Alachua	Gwynn Oaks	65-06	6	90	1	2.8	.	.	7.38

Table D-2. Non-transformed data for all chemical factors for all lakes.

County	Lake	Lake Region	TP Zone	Quantile	Mean Total Phosphorus (µg/L)	Mean Total Alkalinity (mg/L as $CaCO_3$)	Mean True Color (PCU)
Washington	Owens	65-03	1	10	2.0	0.3	21.8
Leon	Trout Pond	65-05	1	90	10.6	.	30.8
Putnam	Mariner	75-04	2	10	2.0	.	.
Clay	Magnolia	75-04	2	10	2.4	.	23.0
Marion	Shoesole	75-09	2	10	2.6	.	.
Putnam	Key pond	75-04	2	10	2.7	.	21.1
Marion	Mary	75-09	2	10	2.9	.	.
Putnam	Barco	75-04	2	10	3.3	0.0	1.9
Lake	Sellers	75-09	2	10	3.7	0.0	2.6
Highlands	Denton	75-33	2	10	3.8	41.0	2.1
Highlands	McCoy	75-33	2	10	4.1	22.0	3.4
Putnam	Long Pond	75-04	2	10	4.5	0.0	1.8
Clay	Sheelar	75-04	2	10	4.6	0.4	2.4
Clay	Lowry	75-04	2	10	4.8	.	12.2
Highlands	Byrd	75-33	2	10	5.0	14.7	8.8
Highlands	Lynn	75-33	2	10	5.3	47.0	3.2
Highlands	Isis	75-33	2	10	5.3	16.0	2.5
Putnam	Johntry Pond	75-11	2	10	5.3	0.0	15.0
Putnam	Cowpen	75-04	2	10	5.4	0.0	3.4
Lake	Sunshine	75-14	2	90	21.7	.	.
Putnam	Faye	75-04	2	90	22.7	2.4	33.5
Lake	Hermosa	75-15	2	90	22.7	28.0	12.0
Highlands	Lachard	75-33	2	90	23.0	36.8	.
Putnam	East	75-04	2	90	24.0	.	.
Clay	Hillcrest	75-04	2	90	24.3	7.2	.
Clay	West Smith	75-04	2	90	27.7	.	2.0
Lake	Dixie	75-14	2	90	31.8	.	.

Clay	Margie	75-04	2	90	33.5	0.0	9.3
Putnam	Twin West	75-04	2	90	35.0	2.2	.
Lake	Gracie	75-15	2	90	35.2	33.0	25.5
Clay	Pebble	75-04	2	90	36.6	0.7	35.6
Bradford	Bedford	75-04	2	90	36.9	3.0	13.0
Marion	Echo	75-09	2	90	37.1	.	.
Lake	Bear 2	75-09	2	90	37.7	0.4	43.0
Lake	Bay	75-09	2	90	48.3	.	33.0
Highlands	Little Jackson	75-33	2	90	56.1	12.7	26.3
Putnam	Clear	75-11	3	10	2.5	.	.
Miami-Dade	Lago Luna	76-03	3	10	3.1	109.0	2.0
Miami-Dade	Crossings	76-03	3	10	3.3	184.0	6.0
Bay	Marin	75-01	3	10	3.3	.	2.1
Miami-Dade	Lago Sol	76-03	3	10	4.1	113.0	2.0
Hillsborough	Alice	75-23	3	10	4.2	0.0	3.5
Miami-Dade	Pavillion 12	76-03	3	10	4.3	93.0	.
Holmes	Cassidy	65-02	3	10	4.4	0.0	1.3
Miami-Dade	Bonita 1	76-03	3	10	4.5	83.0	.
Miami-Dade	Devon Aire	76-03	3	10	4.7	.	.
Santa Rosa	Ski Watch	75-01	3	10	4.7	0.0	1.6
Okaloosa	Noname	65-01	3	10	4.7	.	29.9
Miami-Dade	E	76-03	3	10	5.0	96.6	4.2
Broward	Sterling	76-03	3	10	5.2	110.7	.
Polk	Blue	75-32	3	10	5.3	.	.
Volusia	Sawyer	65-05	3	10	5.3	0.0	.
Putnam	Broward	75-11	3	10	5.7	0.5	6.6
Miami-Dade	Esplanade	76-03	3	10	5.7	.	5.6
Miami-Dade	Colonial	76-03	3	10	5.8	73.0	3.1
Okaloosa	Roberts	65-01	3	10	5.9	.	10.5
Walton	Campbell	75-01	3	10	6.3	.	20.7
Walton	Wilson	65-01	3	10	6.3	7.8	.

Miami-Dade	GMSC	76-03	3	10	6.4	.	10.5
Putnam	Como	75-11	3	10	6.4	0.0	4.7
Volusia	Winnemissett	75-11	3	10	6.5	8.6	8.4
Orange	Johio	75-16	3	10	6.5	.	7.9
Volusia	Broken Arrow	75-11	3	10	6.7	.	16.9
Miami-Dade	Pineland	76-03	3	10	6.8	.	5.0
Alachua	Watermelon Pond	75-05	3	10	6.9	.	.
Escambia	Blue Angel	75-01	3	10	6.9	.	12.5
Walton	Western	75-01	3	10	7.0	24.3	54.4
Broward	Windermere	75-33	3	10	7.0	.	4.4
Miami-Dade	Lindgren	75-33	3	10	7.0	135.0	.
Miami-Dade	Chara	76-03	3	10	7.0	.	4.3
Hillsborough	Glass	75-15	3	10	7.1	0.0	.
Broward	Desoto	76-03	3	10	7.2	.	6.7
Lake	Arthur	75-19	3	10	7.3	1.8	.
Washington	Becton pd	65-02	3	10	7.3	.	77.1
Putnam	Banana	75-11	3	10	7.4	.	.
Polk	Parks	75-32	3	10	7.4	.	14.5
Hillsborough	Mound	75-23	3	10	7.4	.	.
Pasco	Treasure	75-24	3	10	7.4	49.0	17.1
Seminole	Emma	75-11	3	10	7.4	.	.
Miami-Dade	Royal	76-03	3	10	7.7	235.5	8.0
Miami-Dade	Oakland	76-03	3	10	7.8	86.0	5.0
Walton	Camp Creek	75-01	3	10	7.8	.	66.8
Walton	Stanley	65-02	3	10	7.8	.	.
Walton	Morris	75-01	3	10	7.9	.	86.1
Orange	Bessie	75-20	3	10	8.0	38.8	7.2
Lake	Desire	75-32	3	10	8.0	47.0	11.0
Lake	Grassy	75-19	3	10	8.0	.	.
Polk	Aurora	75-19	3	10	8.0	.	.

Bay	Deer Point	75-01	3	10	8.0	37.5	16.0
Lake	Kirkland	75-19	3	10	8.0	.	11.5
Lake	Arlene	75-19	3	10	8.0	21.0	.
Orange	Down	75-20	3	10	8.1	.	9.0
Washington	Pate	65-02	3	10	8.1	0.3	42.5
Hillsborough	Grace	75-23	3	10	8.1	.	.
Hillsborough	Silver	75-24	3	90	44.8	49.2	.
Seminole	Yvonne-2	75-16	3	90	44.9	.	.
Pasco	Little Black	75-24	3	90	45.2	.	81.6
Hillsborough	Virginia	75-24	3	90	45.6	17.0	41.9
Volusia	Marie	75-11	3	90	47.2	.	.
Hillsborough	Jeanette	75-24	3	90	49.4	.	20.0
Orange	Olivia	75-16	3	90	49.7	49.0	18.0
Broward	Delevoe	76-03	3	90	51.4	63.3	10.7
Osceola	Fish	75-27	3	90	51.6	37.0	41.9
Citrus	Hampton	75-12	3	90	53.8	46.2	155.5
Volusia	Macy	75-11	3	90	55.0	20.5	200.0
Palm Beach	Santee	76-03	3	90	56.7	.	.
Hillsborough	Little Wilson	75-24	3	90	57.4	51.0	51.0
Hillsborough	Mid	75-24	3	90	57.6	25.3	38.0
Seminole	Searcy	75-11	3	90	57.7	.	64.7
Orange	Lotta	75-16	3	90	58.3	.	.
Hillsborough	Morley	75-24	3	90	58.7	.	36.0
Polk	Ida	75-32	3	90	59.1	.	.
Orange	Horseshoe	75-16	3	90	59.1	.	.
Seminole	East	75-21	3	90	59.8	.	45.0
Hillsborough	Morris	75-24	3	90	60.2	.	.
Hillsborough	Forest	75-24	3	90	60.5	77.0	21.2
Osceola	Center	75-27	3	90	60.6	1.2	289.5
Broward	Bonnet Slough	76-03	3	90	61.8	184.0	24.1
Orange	Mud	75-27	3	90	61.9	.	157.0

Palm Beach	Osborne	76-03	3	90	62.0	.	35.8
Jackson	Thompson Pond	65-02	3	90	62.6	0.6	112.5
Seminole	Bath	75-11	3	90	62.9	.	47.8
Orange	Lawne	75-16	3	90	63.7	.	.
Seminole	Lotus	75-16	3	90	64.7	.	54.1
Walton	Oyster	75-01	3	90	66.1	24.0	176.7
Wakulla	Otter	75-01	3	90	66.5	.	250.7
Hillsborough	North Pond	75-24	3	90	70.4	.	23.6
Seminole	Dot	75-11	3	90	71.6	.	35.0
Seminole	Amory	75-11	3	90	75.1	.	84.3
Hillsborough	Heather	75-24	3	90	75.5	.	85.8
Seminole	Minnie	75-11	3	90	76.4	.	207.9
Orange	Warren	75-27	3	90	78.3	.	41.2
Palm Beach	Clarke	76-03	3	90	78.5	.	68.8
Orange	Orlando	75-16	3	90	83.7	.	.
Lake	Horseshoe	75-16	3	90	85.8	6.0	57.0
Orange	Sandy	75-20	3	90	85.8	.	16.7
Orange	Marshall	75-16	3	90	89.0	39.0	36.3
Bradford	Rowell	75-03	3	90	98.8	14.0	20.0
Palm Beach	Ida	76-03	3	90	107.6	150.0	53.8
Hillsborough	Buck	75-23	3	90	131.6	6.0	.
Seminole	Alma	75-11	3	90	169.8	13.3	.
Broward	Alex	76-03	3	90	231.3	99.0	23.0
Orange	Floy	75-20	3	90	246.3	.	.
Palm Beach	Julie	76-03	3	90	251.0	.	53.0
Leon	Munson	75-01	3	90	275.2	.	42.5
Okaloosa	Kell-Aire	75-01	3	90	296.0	35.0	58.0
Okaloosa	Coleman	75-01	3	90	364.1	42.0	66.0
Walton	Stewart	75-01	3	90	417.7	.	32.0
Leon	Eightmile Pond	75-01	3	90	430.6	38.7	67.2

Miami-Dade	Snook Pond	76-03	3	90	1857.3	.	184.0
Liberty	Mystic	65-04	4	10	6.0	4.0	4.6
Leon	Erie	65-04	4	10	6.8	0.0	.
Orange	Price	75-10	4	10	7.8	6.1	.
Putnam	Davis Lake	75-10	4	10	8.2	.	61.3
Polk	Otter	75-34	4	10	8.7	.	.
Dixie	Governor Hill	75-06	4	10	9.0	.	.
Highlands	Mills Pond	75-34	4	10	9.4	.	22.5
Flagler	Rodgers	75-10	4	10	9.5	.	34.3
Highlands	Hill	75-34	4	10	9.8	0.3	20.0
Polk	Dexter	75-31	4	10	9.8	33.0	9.6
Orange	Conway North	75-21	4	10	9.9	.	.
Orange	Pickett	75-10	4	10	10.0	2.9	37.3
St Lucie	Patricia	75-10	4	10	10.0	.	.
Clay	Ryan	75-10	4	10	10.0	.	.
Lake	Haines	75-08	4	10	10.2	3.4	.
Leon	Orchard Pond	75-32	4	10	10.3	.	9.0
St Lucie	Bel Air	75-10	4	10	10.5	.	17.9
St Lucie	Jean	75-10	4	10	10.6	.	.
St Lucie	Jeffery	75-10	4	10	10.6	.	13.6
Polk	Lucerne	75-31	4	10	10.7	.	.
Polk	Patrick	75-34	4	10	10.8	.	.
Orange	Georgia	75-21	4	10	10.8	.	16.4
Highlands	Apthorpe	75-34	4	10	10.8	11.5	26.1
Seminole	Crystal Bowl	75-21	4	10	11.0	.	4.9
Orange	Little Conway	75-21	4	10	11.2	34.0	7.6
Duval	Marietta	75-10	4	10	11.3	32.0	8.3
Lake	Purdum Pond	75-08	4	10	11.3	84.0	.
Seminole	Baptismal	75-21	4	10	11.3	23.0	.
Seminole	Noname	75-21	4	10	11.5	.	.
Flagler	Gore	75-10	4	10	11.6	.	.

Seminole	Deep	75-21	4	10	11.6	8.4	12.6
St Lucie	David	75-10	4	10	12.0	.	16.5
Hillsborough	Place	75-22	4	10	12.2	42.5	.
Orange	Frederica	75-21	4	10	12.3	26.0	10.0
Union	Palestine	75-02	4	10	12.3	.	4.0
St Lucie	Phyllis	75-10	4	10	12.6	.	.
Polk	Martha	75-31	4	10	12.6	57.0	8.8
Leon	Carr	75-19	4	10	12.7	.	17.0
St Lucie	Margaret	75-10	4	10	13.3	.	14.2
Orange	Sunset	75-21	4	10	13.3	.	15.0
Nassau	East Nassau	75-10	4	10	13.3	13.7	.
Hernando	Tooke	75-17	4	10	13.5	3.3	15.9
Seminole	Mills	75-10	4	10	13.5	1.2	23.0
Putnam	Noname	75-10	4	10	13.6	.	28.7
Seminole	Florence	75-21	4	10	13.8	9.2	9.3
Seminole	Hodge	75-21	4	10	13.9	.	33.3
Lake	Idlewild	75-08	4	10	13.9	.	.
Highlands	Lelia	75-34	4	10	14.0	.	12.3
Orange	Gear	75-21	4	90	93.9	.	19.0
Leon	Harriman Pond	65-04	4	90	94.7	.	19.0
Putnam	Crescent	75-10	4	90	96.2	.	48.0
Alachua	Calf Pond	75-08	4	90	96.6	65.0	46.5
Leon	Killarney	65-04	4	90	97.8	13.0	11.7
Citrus	North	75-13	4	90	99.0	98.0	10.5
Highlands	Charlotte	75-34	4	90	99.5	0.0	82.8
Volusia	Ashby	75-10	4	90	99.7	.	149.3
Highlands	Glenada	75-34	4	90	102.2	38.7	42.7
Leon	McCord Pond	65-04	4	90	102.7	.	20.0
Orange	Apopka	75-08	4	90	107.8	86.0	.
Highlands	August	75-34	4	90	108.6	6.8	123.7
Orange	Lawsona	75-21	4	90	109.1	.	.

Orange	Como	75-21	4	90	112.5	.	.
Orange	Walker	75-21	4	90	112.7	.	.
Orange	Wade	75-21	4	90	113.1	.	.
Indian River	Blue Cypress	75-10	4	90	115.3	.	182.3
Polk	Smart	75-31	4	90	116.3	.	.
Gadsden	Tallavana	65-04	4	90	117.7	.	27.1
Leon	Susan	65-04	4	90	118.7	3.4	24.5
Putnam	Redwater	75-08	4	90	120.9	1.6	224.7
Putnam	Ross	75-10	4	90	122.5	0.0	401.0
Alachua	Wauberg	75-08	4	90	126.9	22.5	47.1
Lake	Beauclaire	75-08	4	90	126.9	.	54.1
Marion	Redwater Lake	75-08	4	90	127.3	.	460.0
Orange	Davis	75-21	4	90	134.9	.	13.8
Highlands	Huckleberry	75-34	4	90	140.6	9.6	65.4
Indian River	Stick Marsh	75-10	4	90	141.9	129.5	85.1
Leon	Goose Pond	65-04	4	90	143.6	46.0	.
Indian River	Farm 13	75-10	4	90	144.0	126.5	93.2
Seminole	Forest	75-10	4	90	149.7	.	172.5
Citrus	Park	75-13	4	90	150.5	87.0	19.9
Polk	Ring	75-31	4	90	152.3	.	104.0
Hernando	Harris Pond	75-13	4	90	154.6	66.8	21.4
Seminole	Jesup	75-10	4	90	157.8	97.0	57.1
Alachua	Newnan	75-08	4	90	159.2	8.8	146.4
Lake	Akron	75-10	4	90	168.9	.	.
Flagler	Dead	75-10	4	90	174.7	41.0	142.5
St Lucie	Mile	75-10	4	90	176.5	23.0	152.5
Taylor	Jaycee Pond	75-06	4	90	180.0	.	.
Alachua	Hidden	75-08	4	90	183.2	45.0	68.4
Alachua	Bivans Arm	75-08	4	90	189.0	74.5	30.5
Highlands	Wolf	75-34	4	90	193.2	2.2	235.0
Alachua	Johnson Pond	75-08	4	90	211.4	.	.

Lake	Trout	75-08	4	90	241.3	.	199.7
Polk	Conine	75-31	4	90	260.3	.	.
Leon	Spring Pond	65-04	4	90	338.3	.	.
Hernando	Bystre	75-13	4	90	1448.0	.	44.0
Lee	Donut	75-36	5	10	8.0	.	.
Sarasota	Loon	75-36	5	10	8.0	.	.
Osceola	Laurel	75-24	5	10	17.7	24.0	.
Hillsborough	St. Clair	75-36	5	10	21.0	.	14.0
Lee	Little Murex	75-36	5	10	22.2	125.0	12.8
Polk	Lowrey	75-36	5	10	24.1	.	32.4
Polk	Weohyakapka	75-35	5	10	25.5	17.3	46.6
Lee	Murex	75-36	5	90	307.5	206.0	18.0
Hillsborough	Grady	75-36	5	90	312.7	18.0	145.1
Polk	Livingston	75-36	5	90	385.9	.	446.5
Hillsborough	Thonotosassa	75-25	5	90	389.0	.	.
Hillsborough	Fantasia	75-36	5	90	418.5	83.0	28.0
Lee	Dunes	75-36	5	90	428.9	.	50.8
Sarasota	Upper Myakka	75-36	5	90	430.2	39.2	200.0
Pinellas	Skipper	75-28	6	10	11.6	.	15.5
Columbia	Jeffery	65-06	6	10	15.5	1.2	75.4
Pinellas	Hewitt	75-28	6	10	15.8	.	12.2
Pinellas	Harbor	75-28	6	10	15.9	.	.
Pinellas	Fusion	75-28	6	10	16.2	.	14.0
Pinellas	Spring	75-28	6	10	16.5	.	24.5
Alachua	Meta	65-06	6	10	19.3	123.0	.
Polk	Little Bass	75-30	6	90	360.8	.	23.1
Pinellas	Flamingo	75-28	6	90	448.6	.	.
Alachua	Alice	65-06	6	90	489.5	85.0	36.5
Pinellas	Egret	75-28	6	90	549.2	.	48.8
Pinellas	Heron	75-28	6	90	630.1	.	.
Pinellas	Placido	75-28	6	90	695.4	.	.

Alachua	Gwynn Oaks	65-06	6	90	722.9	13.0	.

Table D-3. Non-transformed data for all biological factors for all lakes.

County	Lake	Lake Region	TP Zones	Quantile	Mean Percent Area Covered (%)	Mean Percent Volume Inhabited (%)	Mean Emergent Zone Biomass (kg/m^2)	Mean Floating-leaved Zone Biomass (kg/m^2)	Mean Submersed Zone Biomass (kg/m^2)
Washington	Owens	65-03	1	10	.	.	.	.	.
Leon	Trout Pond	65-05	1	90	.	.	.	.	.
Putnam	Mariner	75-04	2	10	.	.	.	.	.
Clay	Magnolia	75-04	2	10	.	.	.	.	.
Marion	Shoesole	75-09	2	10	.	.	.	.	.
Putnam	Key pond	75-04	2	10	.	.	.	.	.
Marion	Mary	75-09	2	10	28.0	2.0	1.0	0.1	0.6
Putnam	Barco	75-04	2	10	100.0	6.4	5.6	0.2	2.2
Lake	Sellers	75-09	2	10	46.5	3.7	3.1	2.2	1.4
Highlands	Denton	75-33	2	10	13.0	2.0	2.5	0.0	1.2
Highlands	McCoy	75-33	2	10	27.0	2.5	4.1	0.5	2.7
Putnam	Long Pond	75-04	2	10	26.0	4.0	0.0	0.0	0.1
Clay	Sheelar	75-04	2	10	9.6	1.8	1.4	0.0	0.0
Clay	Lowry	75-04	2	10	.	.	.	.	.
Highlands	Byrd	75-33	2	10	21.0	8.5	3.7	0.5	4.1
Highlands	Lynn	75-33	2	10	.	.	.	.	.
Highlands	Isis	75-33	2	10	.	.	.	.	.
Putnam	Johntry Pond	75-11	2	10	.	.	.	.	.
Putnam	Cowpen	75-04	2	10	57.0	6.8	2.1	1.3	0.9
Lake	Sunshine	75-14	2	90	.	.	.	.	.
Putnam	Faye	75-04	2	90	.	.	.	.	.
Lake	Hermosa	75-15	2	90	1.0	0.0	2.0	0.0	0.6
Highlands	Lachard	75-33	2	90	.	.	.	.	.
Putnam	East	75-04	2	90	.	.	.	.	.

Clay	Hillcrest	75-04	2	90	.	.	.	.	.
Clay	West Smith	75-04	2	90	.	.	.	.	.
Lake	Dixie	75-14	2	90	.	.	.	.	.
Clay	Margie	75-04	2	90	.	.	.	.	.
Putnam	Twin West	75-04	2	90	.	.	.	.	.
Lake	Gracie	75-15	2	90	32.0	0.8	3.6	1.9	3.9
Clay	Pebble	75-04	2	90	.	.	.	.	.
Bradford	Bedford	75-04	2	90	1.9	0.1	2.3	1.0	0.0
Marion	Echo	75-09	2	90	.	.	.	.	.
Lake	Bear 2	75-09	2	90	0.0	0.0	0.7	0.2	0.2
Lake	Bay	75-09	2	90	1.0	0.0	1.1	0.4	0.1
Highlands	Little Jackson	75-33	2	90	11.0	2.2	4.3	1.4	0.1
Putnam	Clear	75-11	3	10	.	.	.	.	.
Miami-Dade	Lago Luna	76-03	3	10	.	.	.	.	.
Miami-Dade	Crossings	76-03	3	10	.	.	.	.	.
Bay	Marin	75-01	3	10	.	.	.	.	.
Miami-Dade	Lago Sol	76-03	3	10	.	.	.	.	.
Hillsborough	Alice	75-23	3	10	92.0	25.5	1.3	1.0	2.5
Miami-Dade	Pavillion 12	76-03	3	10	.	.	.	.	.
Holmes	Cassidy	65-02	3	10	12.0	1.0	6.7	0.1	5.3
Miami-Dade	Bonita 1	76-03	3	10	.	.	.	.	.
Miami-Dade	Devon Aire	76-03	3	10	.	.	.	.	.
Santa Rosa	Ski Watch	75-01	3	10	.	.	.	.	.
Okaloosa	Noname	65-01	3	10	.	.	.	.	.
Miami-Dade	E	76-03	3	10	46.0	6.1	0.0	0.0	2.6
Broward	Sterling	76-03	3	10	.	.	.	.	.
Polk	Blue	75-32	3	10	.	.	.	.	.
Volusia	Sawyer	65-05	3	10	.	.	.	.	.
Putnam	Broward	75-11	3	10	53.0	6.4	1.7	0.4	1.8
Miami-Dade	Esplanade	76-03	3	10	.	.	.	.	.
Miami-Dade	Colonial	76-03	3	10	.	.	.	.	.

Okaloosa	Roberts	65-01	3	10	.	.	.	.	.
Walton	Campbell	75-01	3	10	.	.	.	.	.
Walton	Wilson	65-01	3	10	.	.	.	.	.
Miami-Dade	GMSC	76-03	3	10	.	.	.	.	.
Putnam	Como	75-11	3	10	22.0	1.7	3.2	2.7	1.5
Volusia	Winnemissett	75-11	3	10	52.0	3.8	0.5	1.1	1.2
Orange	Johio	75-16	3	10	.	.	.	.	.
Volusia	Broken Arrow	75-11	3	10	.	.	.	.	.
Miami-Dade	Pineland	76-03	3	10	.	.	.	.	.
Alachua	Watermelon Pond	75-05	3	10	.	.	.	.	.
Escambia	Blue Angel	75-01	3	10	.	.	.	.	.
Walton	Western	75-01	3	10	0.0	0.0	2.9	0.0	0.2
Broward	Windermere	75-33	3	10	.	.	.	.	.
Miami-Dade	Lindgren	75-33	3	10	.	.	.	.	.
Miami-Dade	Chara	76-03	3	10	.	.	.	.	.
Hillsborough	Glass	75-15	3	10	.	.	.	.	.
Broward	Desoto	76-03	3	10	.	.	.	.	.
Lake	Arthur	75-19	3	10	.	.	.	.	.
Washington	Becton pd	65-02	3	10	.	.	.	.	.
Putnam	Banana	75-11	3	10	26.0	7.1	2.9	0.4	0.5
Polk	Parks	75-32	3	10	.	.	.	.	.
Hillsborough	Mound	75-23	3	10	52.0	5.9	7.6	2.6	2.9
Pasco	Treasure	75-24	3	10	.	.	.	.	.
Seminole	Emma	75-11	3	10	.	.	.	.	.
Miami-Dade	Royal	76-03	3	10	.	.	.	.	.
Miami-Dade	Oakland	76-03	3	10	70.0	10.0	0.2	0.0	2.0
Walton	Camp Creek	75-01	3	10	.	.	3.2	0.7	1.2
Walton	Stanley	65-02	3	10	.	.	.	.	.
Walton	Morris	75-01	3	10	22.0	5.7	0.7	0.8	0.2
Orange	Bessie	75-20	3	10	65.3	7.1	0.5	0.2	0.3

Lake	Desire	75-32	3	10	.	.	.	.	.
Lake	Grassy	75-19	3	10	.	.	.	.	.
Polk	Aurora	75-19	3	10	.	.	.	.	.
Bay	Deer Point	75-01	3	10	49.0	15.6	5.2	1.5	3.8
Lake	Kirkland	75-19	3	10	0.0	0.0	1.5	0.5	0.9
Lake	Arlene	75-19	3	10	.	.	.	.	.
Orange	Down	75-20	3	10	.	.	6.8	2.9	2.0
Washington	Pate	65-02	3	10	.	.	.	.	.
Hillsborough	Grace	75-23	3	10	.	.	.	.	.
Hillsborough	Silver	75-24	3	90	.	.	.	.	.
Seminole	Yvonne-2	75-16	3	90	.	.	.	.	.
Pasco	Little Black	75-24	3	90	.	.	.	.	.
Hillsborough	Virginia	75-24	3	90	.	.	.	.	.
Volusia	Marie	75-11	3	90	.	.	.	.	.
Hillsborough	Jeanette	75-24	3	90	.	.	.	.	.
Orange	Olivia	75-16	3	90	.	.	.	.	.
Broward	Delevoe	76-03	3	90	12.0	0.2	4.8	0.6	9.6
Osceola	Fish	75-27	3	90	16.0	3.8	4.5	0.2	6.7
Citrus	Hampton	75-12	3	90	14.0	4.8	2.7	2.1	0.4
Volusia	Macy	75-11	3	90	.	.	.	.	.
Palm Beach	Santee	76-03	3	90	.	.	.	.	.
Hillsborough	Little Wilson	75-24	3	90	100.0	48.0	3.7	1.7	8.7
Hillsborough	Mid	75-24	3	90	.	.	.	.	.
Seminole	Searcy	75-11	3	90	.	.	.	.	.
Orange	Lotta	75-16	3	90	.	.	.	.	.
Hillsborough	Morley	75-24	3	90	.	.	.	.	.
Polk	Ida	75-32	3	90	.	.	.	.	.
Orange	Horseshoe	75-16	3	90	.	.	.	.	.
Seminole	East	75-21	3	90	.	.	.	.	.
Hillsborough	Morris	75-24	3	90	.	.	.	.	.
Hillsborough	Forest	75-24	3	90	.	.	.	.	.

Osceola	Center	75-27	3	90	7.0	1.3	5.4	1.2	0.1
Broward	Bonnet Slough	76-03	3	90	.	.	.	.	.
Orange	Mud	75-27	3	90	.	.	.	.	.
Palm Beach	Osborne	76-03	3	90	.	.	.	.	.
	Thompson								
Jackson	Pond	65-02	3	90	.	.	.	.	.
Seminole	Bath	75-11	3	90	.	.	.	.	.
Orange	Lawne	75-16	3	90	4.0	1.2	4.3	0.0	0.0
Seminole	Lotus	75-16	3	90	.	.	.	.	.
Walton	Oyster	75-01	3	90	6.5	1.1	13.8	0.5	0.1
Wakulla	Otter	75-01	3	90	.	.	.	.	.
Hillsborough	North Pond	75-24	3	90	.	.	.	.	.
Seminole	Dot	75-11	3	90	.	.	.	.	.
Seminole	Amory	75-11	3	90	.	.	.	.	.
Hillsborough	Heather	75-24	3	90	.	.	.	.	.
Seminole	Minnie	75-11	3	90	.	.	.	.	.
Orange	Warren	75-27	3	90	.	.	.	.	.
Palm Beach	Clarke	76-03	3	90	.	.	.	.	.
Orange	Orlando	75-16	3	90	.	.	.	.	.
Lake	Horseshoe	75-16	3	90	.	.	.	.	.
Orange	Sandy	75-20	3	90	.	.	.	.	.
Orange	Marshall	75-16	3	90	.	.	.	.	.
Bradford	Rowell	75-03	3	90	.	.	.	.	.
Palm Beach	Ida	76-03	3	90	.	.	.	.	.
Hillsborough	Buck	75-23	3	90	.	.	.	.	.
Seminole	Alma	75-11	3	90	.	.	.	.	.
Broward	Alex	76-03	3	90	.	.	.	.	.
Orange	Floy	75-20	3	90	.	.	.	.	.
Palm Beach	Julie	76-03	3	90	.	.	.	.	.
Leon	Munson	75-01	3	90	.	.	.	.	.
Okaloosa	Kell-Aire	75-01	3	90	.	.	.	.	.

Okaloosa	Coleman	75-01	3	90	.	.	.	.	.
Walton	Stewart	75-01	3	90	.	.	.	.	.
Leon	Eightmile Pond	75-01	3	90	.	.	.	.	.
Miami-Dade	Snook Pond	76-03	3	90	.	.	.	.	.
Liberty	Mystic	65-04	4	10	.	.	.	.	.
Leon	Erie	65-04	4	10	86.7	66.8	3.0	3.3	2.9
Orange	Price	75-10	4	10	.	.	.	.	.
Putnam	Davis Lake	75-10	4	10	.	.	.	.	.
Polk	Otter	75-34	4	10	.	.	.	.	.
Dixie	Governor Hill	75-06	4	10	.	.	.	.	.
Highlands	Mills Pond	75-34	4	10	.	.	.	.	.
Flagler	Rodgers	75-10	4	10	.	.	.	.	.
Highlands	Hill	75-34	4	10	29.0	5.5	0.5	0.7	2.0
Polk	Dexter	75-31	4	10	73.9	27.3	4.6	3.5	4.7
Orange	Conway North	75-21	4	10	39.8	5.3	2.0	0.5	1.0
Orange	Pickett	75-10	4	10	.	.	.	.	.
St Lucie	Patricia	75-10	4	10	.	.	.	.	.
Clay	Ryan	75-10	4	10	.	.	.	.	.
Lake	Haines	75-08	4	10	.	.	.	.	.
Leon	Orchard Pond	75-32	4	10	.	.	.	.	.
St Lucie	Bel Air	75-10	4	10	.	.	.	.	.
St Lucie	Jean	75-10	4	10	32.0	5.8	0.3	0.0	1.8
St Lucie	Jeffery	75-10	4	10	64.0	18.9	0.0	0.0	0.0
Polk	Lucerne	75-31	4	10	.	.	.	.	.
Polk	Patrick	75-34	4	10	.	.	.	.	.
Orange	Georgia	75-21	4	10	90.0	32.6	3.9	2.4	4.9
Highlands	Apthorpe	75-34	4	10	34.0	3.8	6.1	0.7	0.9
Seminole	Crystal Bowl	75-21	4	10	.	.	.	.	.
Orange	Little Conway	75-21	4	10	48.0	10.1	1.5	0.3	2.1
Duval	Marietta	75-10	4	10	.	.	.	.	.
Lake	Purdum Pond	75-08	4	10	.	.	.	.	.

Seminole	Baptismal	75-21	4	10	.	.	.	.	.
Seminole	Noname	75-21	4	10	.	.	.	.	.
Flagler	Gore	75-10	4	10	.	.	.	.	.
Seminole	Deep	75-21	4	10	.	.	.	.	.
St Lucie	David	75-10	4	10	27.0	8.6	0.5	0.1	0.0
Hillsborough	Place	75-22	4	10	.	.	.	.	.
Orange	Frederica	75-21	4	10	.	.	.	.	.
Union	Palestine	75-02	4	10	.	.	.	.	.
St Lucie	Phyllis	75-10	4	10	.	.	.	.	.
Polk	Martha	75-31	4	10	.	.	.	.	.
Leon	Carr	75-19	4	10	.	.	.	.	.
St Lucie	Margaret	75-10	4	10	20.0	7.7	0.7	0.0	0.9
Orange	Sunset	75-21	4	10	.	.	.	.	.
Nassau	East Nassau	75-10	4	10	.	.	.	.	.
Hernando	Tooke	75-17	4	10	.	.	.	.	.
Seminole	Mills	75-10	4	10	.	.	.	.	.
Putnam	Noname	75-10	4	10	.	.	.	.	.
Seminole	Florence	75-21	4	10	16.0	1.0	7.6	1.4	0.7
Seminole	Hodge	75-21	4	10	.	.	.	.	.
Lake	Idlewild	75-08	4	10	18.0	1.8	2.0	0.7	0.0
Highlands	Lelia	75-34	4	10	.	.	.	.	.
Orange	Gear	75-21	4	90	.	.	.	.	.
Leon	Harriman Pond	65-04	4	90	.	.	.	.	.
Putnam	Crescent	75-10	4	90	0.0	0.0	2.0	1.6	1.3
Alachua	Calf Pond	75-08	4	90	.	.	.	.	.
Leon	Killarney	65-04	4	90	.	.	.	.	.
Citrus	North	75-13	4	90	.	.	.	.	.
Highlands	Charlotte	75-34	4	90	26.0	3.8	0.9	0.7	0.2
Volusia	Ashby	75-10	4	90	34.0	4.5	3.2	3.4	1.2
Highlands	Glenada	75-34	4	90	6.0	0.4	2.9	1.0	0.0
Leon	McCord Pond	65-04	4	90	.	.	.	.	.

Orange	Apopka	75-08	4	90	2.0	1.1	4.1	2.7	0.6
Highlands	August	75-34	4	90	.	.	.	.	.
Orange	Lawsona	75-21	4	90	10.0	1.0	6.8	0.0	0.0
Orange	Como	75-21	4	90	.	.	.	.	.
Orange	Walker	75-21	4	90	.	.	.	.	.
Orange	Wade	75-21	4	90	.	.	.	.	.
Indian River	Blue Cypress	75-10	4	90	0.0	0.0	3.1	3.4	0.0
Polk	Smart	75-31	4	90	.	.	.	.	.
Gadsden	Tallavana	65-04	4	90	23.0	7.0	6.3	2.2	2.8
Leon	Susan	65-04	4	90	.	.	.	.	.
Putnam	Redwater	75-08	4	90	6.0	2.7	6.5	5.8	0.5
Putnam	Ross	75-10	4	90	2.0	0.1	5.2	0.0	0.0
Alachua	Wauberg	75-08	4	90	4.5	0.6	3.5	2.1	0.0
Lake	Beauclaire	75-08	4	90	17.0	5.7	4.1	2.3	2.6
Marion	Redwater Lake	75-08	4	90	.	.	.	.	.
Orange	Davis	75-21	4	90	4.0	0.2	2.8	0.0	1.3
Highlands	Huckleberry	75-34	4	90	.	.	.	.	.
Indian River	Stick Marsh	75-10	4	90	4.7	0.8	2.8	2.1	1.8
Leon	Goose Pond	65-04	4	90	.	.	.	.	.
Indian River	Farm 13	75-10	4	90	11.0	5.8	3.0	2.0	4.6
Seminole	Forest	75-10	4	90	.	.	.	.	.
Citrus	Park	75-13	4	90	.	.	.	.	.
Polk	Ring	75-31	4	90	.	.	.	.	.
Hernando	Harris Pond	75-13	4	90	.	.	.	.	.
Seminole	Jesup	75-10	4	90	0.0	0.0	4.8	3.2	0.1
Alachua	Newnan	75-08	4	90	12.7	3.0	6.1	3.2	3.3
Lake	Akron	75-10	4	90	.	.	.	.	.
Flagler	Dead	75-10	4	90	.	.	.	.	.
St Lucie	Mile	75-10	4	90	.	.	.	.	.
Taylor	Jaycee Pond	75-06	4	90	.	.	.	.	.
Alachua	Hidden	75-08	4	90	.	.	.	.	.

Alachua	Bivans Arm	75-08	4	90	.	.	.	.	.
Highlands	Wolf	75-34	4	90	0.0	0.0	4.7	0.9	0.1
Alachua	Johnson Pond	75-08	4	90	.	.	.	.	.
Lake	Trout	75-08	4	90	0.0	0.0	5.7	0.0	0.0
Polk	Conine	75-31	4	90	.	.	.	.	.
Leon	Spring Pond	65-04	4	90	.	.	2.8	0.9	0.0
Hernando	Bystre	75-13	4	90	.	.	.	.	.
Lee	Donut	75-36	5	10	.	.	.	.	.
Sarasota	Loon	75-36	5	10	.	.	.	.	.
Osceola	Laurel	75-24	5	10	.	.	.	.	.
Hillsborough	St. Clair	75-36	5	10	.	.	.	.	.
Lee	Little Murex	75-36	5	10	.	.	.	.	.
Polk	Lowrey	75-36	5	10	.	.	.	.	.
Polk	Weohyakapka	75-35	5	10	42.7	26.0	3.4	0.5	1.9
Lee	Murex	75-36	5	90	.	.	.	.	.
Hillsborough	Grady	75-36	5	90	.	.	.	.	.
Polk	Livingston	75-36	5	90	.	.	.	.	.
Hillsborough	Thonotosassa	75-25	5	90	.	.	.	.	.
Hillsborough	Fantasia	75-36	5	90	.	.	.	.	.
Lee	Dunes	75-36	5	90	.	.	.	.	.
Sarasota	Upper Myakka	75-36	5	90	.	.	.	.	.
Pinellas	Skipper	75-28	6	10	.	.	.	.	.
Columbia	Jeffery	65-06	6	10	.	.	.	.	.
Pinellas	Hewitt	75-28	6	10	.	.	.	.	.
Pinellas	Harbor	75-28	6	10	.	.	.	.	.
Pinellas	Fusion	75-28	6	10	.	.	.	.	.
Pinellas	Spring	75-28	6	10	.	.	.	.	.
Alachua	Meta	65-06	6	10	.	.	.	.	.
Polk	Little Bass	75-30	6	90	43.4	4.9	3.0	0.5	0.0
Pinellas	Flamingo	75-28	6	90	.	.	.	.	.
Alachua	Alice	65-06	6	90	.	.	.	.	.

Pinellas	Egret	75-28	6	90	.	.	.	.	.
Pinellas	Heron	75-28	6	90	.	.	.	.	.
Pinellas	Placido	75-28	6	90	.	.	.	.	.
Alachua	Gwynn Oaks	65-06	6	90	.	.	.	.	.

LIST OF REFERENCES

[APHA] American Public Health Association. 2005. Standard methods for the examination of water and waste water, 21st ed. Washington (DC).

Bachmann RW, Hoyer MV, Canfield DE Jr. 2000. The potential for wave disturbance in shallow Florida lakes. Lake Reserv Manage. 16:281-291.

Bachmann RW, Horsburgh CA, Hoyer MV, Mataraza, LK, Canfield DE Jr. 2002. Relations between trophic state indicators and plant biomass in Florida lakes. Hydrobiologia. 470: 219-234.

Bachmann, RW, Bigham DL, Hoyer MV, Canfield DE Jr. 2012a. A strategy for establishing numeric nutrient criteria for Florida lakes. Lake Reserv Manage. 21:1-8.

Bachmann, RW, Bigham DL, Hoyer MV, Canfield DE Jr. 2012b. Factors determining the distributions of total phosphorus, total nitrogen, and chlorophyll a in Florida lakes. Lake Reserv Manage. 28 (1):10-26.

Bigham DL. 2012. Analyses of temporal changes in trophic state variables in Florida lakes. [dissertation]. [Gainesville (FL)]: University of Florida.

Blindlow I, Andersson G, Hargeby A, Johansson S. 1993. Long-term patterns of alternate stable states in two shallow eutrophic lakes. Freshw Biol. 30:159-167.

Blindlow I, Hargeby A, Andersson G. 1997. Alternate stable states in shallow lakes: what causes a shift. In: Jeppesen E, Søndergaard Ma., Søndergaard Mo., Christoffersen K, editors. The role of macrophytes in structuring the biological community and biogeochemical dynamics in lakes Springer-Verlag, New York. p. 353–360.

Brenner M, Binford MW. 1988. Relationships between concentrations of sedimentary variables and trophic state in Florida lakes. Can J Fish Aquat Sci. 45(2): 294-300.

Brenner M, Keenan LW, Miller SJ, Schelske CL. 1999. Spatial and temporal patterns of sediment and nutrient accumulation in shallow lakes of the Upper St. Johns River Basin, Florida. Wetl Ecol Manag. 6:221-240.

Brown CD, Hoyer MV, Bachmann RW, Canfield DE Jr. 2000. Nutrient-chlorophyll relationships: an evaluation of empirical nutrient-chlorophyll models using Florida and northern-temperate lake data. Can J Fish Aquat Sci. 57:1574-1583.

Brown MT, Vivas MB. 2005. Landscape development intensity index. Environ Monit Assess. 101: 289-309.

Burkholder JM, Wetzel RG, Klomparens KL. 1990. Direct comparison of phosphate uptake by the adnate and loosely attached microalgae within an intact biofilm matrix. Appl Environ Microbiol. 56:2882-2890.

Canfield DE Jr. 1983. Prediction of chlorophyll a concentrations in Florida lakes: the importance of phosphorus and nitrogen. J Am Wat Res Assoc. 19(2): 255-262.

Canfield DE Jr, Langeland KA, Maceina MJ, Haller WT, Shireman JV, Jones JR. 1983. Trophic state classification of lakes with aquatic macrophytes. Can J Fish Aquat Sci. 40:1713-1718.

Canfield DE Jr, Linda SB, Hodgson LM. 1984. Relations between color and some limnological characteristics of Florida lakes. Wat Res Bull. 20(3):323-329.

Canfield DE Jr, Hoyer MV. 1988. Regional geology and the chemical and trophic state characteristics of Florida lakes. Lake Reserv Manage. 4(1): 21-31.

Canfield DE Jr, Hoyer MV, Duarte CM. 1990. An empirical method for characterizing standing crops of aquatic vegetation. J Aquat Plant Manage. 23:25-28.

Canfield DE Jr, Brown CD, Bachmann RW, Hoyer MV. 2002. Volunteer lake monitoring: testing the reliability of data collected by the Florida LAKEWATCH program. Lake Reserv Manage. 18(1): 1-9.

Carignan R. 1982. An empirical model to estimate the relative importance of roots in phosphorus uptake by aquatic macrophytes. Can J Fish Aquat. Sci. 39:243-247.

Carignan R, Kalff J. 1980. Phosphorus sources for aquatic needs: water or sediments. Science 202: 987-988.

Carpenter, S. R., 1980. Enrichment of Lake Wingra, Wisconsin, by submersed macrophyte decay. Ecology (USA) 61:1145-l 155.

Carrick HJ, Aldridge FJ, Schelske CL. 1993. Wind influences phytoplankton biomass and composition in a shallow, productive lake. Limnol Oceanogr. 38: 1179-1192.

Christophoridis C, Fytianos K. 2006. Conditions affecting the release of phosphorus from lakes sediments. J Enviro Qual. 35:1181-1192.

Clark MA, Bortles J, Darr J, Denaham L, Johnson C, Sheridan R. 2008. Water quality summary report. Orange County Environmental Protection Division, Orlando, Florida.

Curtis PJ, Adams HE. 1995. Dissolved organic matter quantity and quality from freshwater and saltwater lakes in east-central Alberta. Biogeochemistry 30:59-76.

Davis CB, van der Valk AG. 1978. Litter decomposition in prairie glacial marshes. In: Good RE, Whigham DF, Simpson RL, editors. Freshwater wetlands: ecological processes and management potential. Academic Press, NY. p. 99-113.

Deevey ES Jr. 1940. Limnological studies in Connecticut. V.A. contribution to regional geology. Am J Sci. 238:717-741.

Dillon PJ, Rigler RH. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol Oceanogr. 20:28-39.

Ecological Society of America. 2012. Frontiers in Ecology and the Environment, Volume 10 (13 articles on volunteer monitoring in ecology).

Enfield DB, Mestas-Nuñez AM, Trimble PJ. 2001. The Atlantic multidecadal oscillation and its relation to rainfall and river flows in the continental US. Geophys Res Lett. 28(10): 2077-2080.

Fore LS, Frydenborg R, Wellendorf N, Espy J, Frick T, Whiting D, Jackson J, Patronis J. 2007. Assessing the biological condition of Florida lakes: development of the Lake Vegetation Index (LVI). Tallahassee (FL): Florida Department of Environmental Protection; Final Report; [cited 18 June 2013]. Accessed from www.seminole.wateratlas.usf.edu/upload/documents/lvi_final 07 .pdf

Florida LAKEWATCH. 2003. Florida LAKEWATCH annual data summaries 2002. Department of Fisheries and Aquatic Sciences, University of Florida/Institute of Food and Agricultural Sciences, Gainesville, Florida.

Gergel SE, Turner MG, Kratz TK. 1999. Dissolved organic carbon as an indicator of the scale of watershed influence on lakes and rivers. Ecol App. 9:1377-1390.

Graneli, W., 1984. Reed Phragmites australis (Cav.) Trin. Ex Steudel as an energy source in Sweden. Biomass 4:183-208.

Granéli W, Solander D. 1988. Influence of aquatic macrophytes on phosphorus cycling in lakes. Hydrobiologia 170(1): 245-266.

Griffith GE, Canfield DE Jr, Horsburgh CA, Omernik JM. 1997. Lake Regions of Florida. Corvallis (OR): US Environmental Protection Agency; National Health and Environmental Effects Research Laboratory. EPA/R-97/217; [cited 25 May 2013]. Available from http://www.epa.gov/wed/pages/ecoregions/fl.eco.htm

Forsberg C, Ryding SO. 1980. Eutrophication parameters and trophic state indices in 30 Swedish waste-receiving lakes Arch Hydrobiol. 89:189-207.

Hakanson, L. 1982. Lake bottom dynamics and morphometry: the dynamic ratio. Water Resources Research. 18:1444-1450.

Hamilton D, Mitchell S. 1997. Wave-induced shear stresses, plant nutrients and chlorophyll in seven shallow lakes. Freshw Biol. 38(1): 159-168.

Hanrahan JL, Kravtsov SV, Roebber, PJ. 2010. Connecting past and present climate variability to the water levels of Lakes Michigan and Huron. Geophys Res Lett. 37(1).

Hansson LA. 1990. Quantifying the impact of periphytic algae on nutrient availability for phytoplankton. Freshw Biol. 24:265-273.

Havens KE, Sharfstein B, Brady MA, East TL, Harwell MC, Maki RP, Rodusky AJ. 2004. Recovery of submerged plants from high water stress in a large subtropical lake in Florida, USA. Aquat Bot. 78: 67-82.

Hesslein RH, Broeker WS, Quay PD, Schindler DW. 1980. Whole-lake radiocarbon experiment in an oligotrophic lake at the Experimental Lakes Area, northwestern Ontario. Can J Fish Aquat Sci. 37:454-463.

Hoyer MV, Horsburgh CA, Canfield DE Jr, Bachmann RW. 2005. Lake level and trophic state variables among a population of shallow Florida lakes and within individual lakes. Can J Fish Aquat Sci. 62: 2760-2769.

Hutchinson GE. 1975. A treatise on limnology. Volume III. Limnological Botany. J. Wiley & Sons, NY, 660 pp.

Jackson HO, Starrett WC. 1959. Turbidity and sedimentation at Lake Chautauqua, Illinois. J Wildl Manage. 23(2): 157-168.

Jaynes ML, Carpenter SR. 1986. Effects of vascular and nonvascular macrophytes on sediment redox and solute dynamics. Ecology (USA) 67: 875-882.

Jeppesen E, Søndergaard M, Søndergaard M, Christoffersen K, editors. 1998. The structuring role of submerged macrophytes in lakes. Springer-Verlag. New York.

Jones JR, Bachmann RW. 1976. Prediction of phosphorus and chlorophyll levels in lakes. J Wat Pol Con Fed. 48:2176-2182.

Jones JR, Bachmann RW. 1978. A survey of water transparency in Iowa lakes. Proc Iowa Acad Sci. 85:6-9.

Knowlton MF, Jones JR. 1997. Trophic status of Missouri River floodplain lakes in relation to basin type and connectivity. Wetlands. 17(4):468-475.

Knowlton MF, Jones JR. 2003. Connectivity influences temporal variations of limnological conditions in Missouri River scour lakes. Lake Reserv Manage. 19(2):160-170.

Knowlton MF, Jones JR. 2006. Temporal variation and assessment of trophic state indicators in Missouri reservoirs: implications for lake monitoring and management. Lake Reserv Manage. 22(3):261-271.

Kufel L, Kufel I. 2002. Chara beds acting as nutrient sinks in shallow lakes—a review. Aquat Bot. 72:249-260.

Landers DH. 1982. Effects of naturally senescing aquatic macrophytes on nutrient chemistry and chlorophyll a of surrounding waters. Limnol Oceanogr. 27:428-439.

Maceina MJ, Shireman JV. 1980. The use of a recording fathometer for the determination of the distribution and biomass of hydrilla. J Aquat Plant Manage. 18:34-39.

McDowell WH, Likens GE. 1988. Origin, composition, and flux of dissolved organic carbon in the Hubbard Brook Valley. Ecol Monogr. 58:177-195.

Menzel DW, Corwin N. 1965. The measurement of total phosphorus in seawater based on the liberation of organically bound fractions by persulfate oxidation. Limnol Oceanogr. 10:280-282.

Morris JT, Lajtha K. 1986. Decomposition and nutrient dynamics of litter from four species of freshwater emergent macrophytes. Hydrobiologia. 13(1):215-223.

Murphy J, Riley JP. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal Chim Acta. 27:31-36.

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

Moyle JB. 1949. Some indices of lake productivity. Trans Am Fish Soc. 76:322-334.

Moyle JB. 1956. Relationship between chemistry of Minnesota surface water and wildlife management. J Wildl Manage. 20:303-320.

Nagid EJ, Canfield DE Jr., Hoyer MV. 2001. Wind-induced increases in trophic state characteristics of a large (27 km2), shallow (1.5 m mean depth) Florida lake. Hydrobiologia. 455:97–110.

Noges P, Jarvet A, Tuvikene L, Noges T. 1998. The budgets of nitrogen and phosphorus in shallow eutrophic Lake Vortsjarv (Estonia). Hydrobiologia 363:219–227.

Nürnberg GK. 1996. Trophic state of clear and colored, soft-and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. Lake Reserv Manage. 12: 432-447.

Otsuki A, Wetzel RG. 1972. Co-precipitation of phosphate with carbonates in a marl lake. Limnol Oceanogr. 17(5):763-767.

Pace ML, Cole JJ. 2002. Synchronous variation of dissolved organic carbon and color in lakes. Limnol Oceanogr. 47:333-342.

Reckhow KH, Arhonditsis GB, Kenney MA, Hauser L, Tribo J, Wu C, Elcock KJ, Steinberg LJ, Stow CA, McBride SJ. 2005. A predictive approach to nutrient criteria. Environ Sci Technol. 39(9): 2913-2919.

Rørslett B. 1985. Death of submerged macrophytes—actual field observations and some implications. Aquat Bot. 22(1):7-19.

Saunders PA, Shaw WH, Bukaveckas PA. 2000. Differences in nutrient limitation and grazer suppression of phytoplankton in seepage and drainage lakes of the Adirondack region, NY, U.S.A. Freshw Biol. 43:391-407.

SAS Institute Inc. 2000. JMP Statistics and Graphics Guide [computer program]. SAS Institute Inc., Cary, N.C.

Schafer MD, Dickinson RE, Heaney JP, Huber WC. 1986. Gazetteer of Florida lakes, Pub. 96. Gainesville (FL): Florida Water Resource Research Center.

Scheffer M, Jeppesen E. 1997. Alternate stable states. In: Jeppesen E, Søndergaard Ma., Søndergaard Mo., Christoffersen K editors. The role of macrophytes in structuring the biological community and biogeochemical dynamics in lakes. Springer-Verlag, New York. p. 397–406.

Scheffer, M. 1998. Ecology of shallow lakes. Chapman and Hall, London.

Schindler DW. 1988. Confusion over the origin of alkalinity in lakes. Limnol Oceanogr. 33(6):1637-1640.

Schindler DW, Bayley SE, Curtis PJ, Parker BR, Stainton MP, Kelly CC. 1992. Natural and man-made factors affecting the abundance and cycling of dissolved organic substances in precambrian shield lakes. Hydrobiologia 229:1-21.

Shannon EE, Brezonik PL. 1972. Limnological characteristics of north and central Florida lakes. Limnol Oceanogr. 17:97-110.

Syers JK, Harris RF, Armstrong DE. 1973. Phosphate chemistry in lake sediments. J Environ Qual. 2(1):1-11.

Twilley RR, Ejdung G, Romare P, Kemp WM. 1986. A comparative study of decomposition, oxygen consumption and nutrient release for selected aquatic plants occurring in an estuarine environment. Oikos 47: 190-198.

[USEPA] United States Environmental Protection Agency. 2000. Ambient water quality criteria recommendations information supporting the development of state and tribal nutrient criteria: Lakes and reservoirs in nutrient ecoregion VI. Office of Water. Washington, D.C.

[USEPA] United States Environmental Protection Agency. 2010. Water quality standards for the state of Florida's lakes and flowing waters. Federal Register 75:131 (26 January 2010) 4173–4226; [cited 25 May 2013]. Available from http://www.gpo.gov/fdsys/pkg/FR-2010-12-06/pdf/2010-29943.pdf

van der Linden, M. J. H. A., 1986. Phosphorus economy of reed vegetation in the Zuidelijk Flevoland polder (The Netherlands): seasonal distribution of phosphorus among shoots and rhizomes and availability of soil phosphorus. Acta Oecologie Oecol. Plant. 7:397-405.

Van Donk E, Gulati RD, Grimm MP. 1989.Food web manipulation in Lake Zwemlust: positive and negative effects during the first two years. Hydrobiol Bull. 23:19-34.

Vermaat JE, Santamaria L, Roos PJ. 2000. Water flow across and sediment trapping in submerged macrophyte beds of contrasting growth forms. Arch Hydrobiol. 148:549-562.

Vollenweider RA. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters with particular reference to nitrogen and phosphorus as factors in eutrophication. Organization for Economic Cooperation and Development, Technical Report DAS/CSI/68:27.

Wetzel RG. 1983. Limnology. 2nd ed. Saunders College Publishing, Philadelphia (PA)

Wetzel RG. 1990. Reservoir ecosystems: Conclusions and speculations. In: Thornton KW, Kimmel BL, Payne FE, editors. Reservoir Limnology: Ecological Perspectives. New York (NY): John Wiley and Sons. p. 227–238.

Wetzel RG. 1992. Gradient-dominated ecosystems: sources and regulatory functions of dissolved organic matter in freshwater ecosystems. Hydrobiologia 229(1):181-198.

BIOGRAPHICAL SKETCH

Chris Anderson was born and raised in Florida. He completed his undergraduate studies at the University of Florida where he majored in wildlife ecology and minored in fisheries and aquatic sciences. After graduating *cum laude* in 2010, he began working for the University of Florida's LAKEWATCH program as a fisheries field biologist. He worked on a project centered on the long-term monitoring of water quality, fish communities and aquatic plant communities in lakes throughout the State of Florida. It was during this time that his fascination with limnology and fisheries management really grew and became his primary research interests. After working at LAKEWATCH for over a year, he was offered a graduate assistantship by Dr. Daniel Canfield Jr., a professor in the Fisheries and Aquatic Sciences Program at UF. During Chris' time as a graduate student, he continued to work on LAKEWATCH's long-term monitoring projects as well as becoming a teaching assistant for an introductory fisheries science class. In his free time, he enjoys playing and watching sports, as well as fishing, cooking, and spending time with his family and friends.