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Eutrophication Trend of Lakes in the Tampa Bay Watershed and the Role of Submerged Aquatic Vegetation in Buffering Lake Water Phosphorus Concentration

Max Jacobo Moreno Madriñan
University of South Florida

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by

Max Jacobo Moreno Madriñan

A dissertation submitted in partial fulfillment
of the requirements for the degree of
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Department of Environmental and Occupational Health
College of Public Health
University of South Florida

Co-Major Professor: Noreen D. Poor, Ph.D.
Co-Major Professor: Mark Ross, Ph.D.
Connie Mizak, Ph.D.
Amy Stuart, Ph.D.

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DEDICATION

This dissertation is dedicated to my son, Rafael Maximiliano Moreno-Schiaffino, an instrument of the LORD. Through his innocence, purity, and love I have been given the inspiration, strength and faith to complete my doctoral degree.

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cm	centimeter
EPA	Environmental Protection Agency
FDEP	Florida Department of Environmental Protection
ha	hectare
L	liter
m	meter
m^3	cubic meter
mg	milligram
µg	microgram
n	number of lakes
SAV	Submerged aquatic vegetation
PAC	Per cent area covered with vegetation
PVI	Per cent volume infested with vegetation
TBEP	Tampa Bay Estuary Program
TN	Total nitrogen
TN:TP	Ratio of total nitrogen to total phosphorus
TP	Total phosphorus

EUTROPHICATION TREND OF LAKES IN THE TAMPA BAY WATERSHED AND THE ROLE OF SUBMERGED AQUATIC VEGETATION IN BUFFERING PHOSPHORUS CONCENTRATION

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ABSTRACT

Twentieth century human settlement within the Tampa Bay watershed was linked to a dramatic mid-century decline in bay water quality and loss of seagrass acreage. Decades of direct and indirect nutrient discharges to the bay from phosphorus mining, fertilizer manufacturing, and wastewater treatment, as examples, impaired the estuary. In the past twenty years, regional stakeholders have worked to improve the bay water quality by reducing point and non-point source nutrient loading to the bay.

Lakes within the Tampa Bay watershed may play an important role in attenuating the flow of nutrients into the bay. This study hypothesized that between 1990 and 2007 lake water concentrations of total phosphorus (TP) and chlorophyll-α, as well as the ratio of total nitrogen to total phosphorus (TN:TP), have changed for selected lakes in the Tampa Bay watershed. During this period, the watershed underwent a rapid shift in land use as groves and farms became shopping malls and new homes. A two-way analysis of variance

(ANOVA) revealed that for 10 lakes clustered in the northern portion of the Tampa Bay watershed and classified as oligotrophic or mesotrophic, observed increases in water concentrations of TP and chlorophyll-α were statistically significant. For 6 lakes classified as hypereutrophic and scattered across the watershed, observed decreases in water TP concentrations were statistically significant, while chlorophyll-α concentrations did not change. For both groups of lakes, the TN:TP ratio declined significantly; however, oligotrophic and mesotrophic lakes were phosphorus-limited but hypereutrophic lakes were nitrogen-limited, based on this ratio.

A second hypothesis of this study was that lake water concentrations of TP, total nitrogen (TN) and chlorophyll-α were lower in lakes that had more coverage of submerged aquatic vegetation, as vegetation suppresses re-suspension of sediments and is a reservoir for nitrogen and phosphorus and a surface for biofilms. The results of a one-way ANOVA showed that for 34 lakes within the Tampa Bay watershed, lakes with a greater than 20 percent volume infested by macrophytes (PVI), water concentrations of TP and chlorophyll-α but not TN were statistically lower than for lakes with a less than 20 PVI.

CHAPTER 1.

INTRODUCTION

1.1 Purpose of the Study

Twentieth century human settlement within the Tampa Bay watershed was linked to a dramatic mid-century decline in bay water quality and loss of seagrass acreage. Decades of direct and indirect nutrient discharges to the bay from phosphorus mining, fertilizer manufacturing, and wastewater treatment, as examples, impaired the estuary. In the past twenty years, regional stakeholders have worked to improve the bay water quality by reducing point and non-point source nutrient loading to the bay (TBEP, 2006). Lakes within the Tampa Bay watershed may play an important role in attenuating the flow of nutrients into the bay, as these lakes ultimately discharge to the bay via canals, springs, creeks, streams, and rivers.

Analyses of temporal trends and a possible role for submerged aquatic vegetation in reducing eutrophication benefit those involved in lake water management within the Tampa Bay watershed (Figure 1.1).

Figure 1.1 Map of the Tampa Bay watershed and major bay segments (TBEP, 2006)

1.2 Problem Statement

Excessive nutrient loading to surface waters threatens environmental and human health (FDEP, 2004). Nutrient enrichment of surface wtaers can cause blooms of algae (Canfield, 1983) with the associated stench and stagnation (McCarthy, 2000). Sustained algal blooms can lower the water column dissolved oxygen concentration and light transparency, conditions which lead to a loss of biodiversity and an attending shift in dominant species (Paerl, 1988). Some species of blue-green algae (cyanobacteria) produce potent toxins. These toxins may enter the human body through ingestion, inhalation, or dermal contact with water or with fish from water polluted with cyanobacteria (Fleming et al., 2002; Karjalainen et al., 2007). Thus, eutrophication may impair a lake to the extent that swimming or fishing is not advised (Hansson et al., 1999).

The Florida Department of Environmental Protection (FDEP, 1996; 2000; 2006) reports an increasing percentage of Florida lakes with a degrading trend in eutrophication status. Urban land use has been suggested as the category with the higher contribution of phosphorus to receiving waters in Florida, followed by agricultural and forested land use (Reddy et al., 1999).

For a lake, water concentrations of nitrogen and phosphorus are determined by a combination of (net) external loading and internal cycling (James and Bierman Jr., 1995; Kittiwanich et al., 2006; Serpa et al., 2007). Phosphorus is most commonly the limiting nutrient in freshwater systems (ChunLei et al., 2004; Schauser et al., 2004; Schindler, 1977; Scinto and Reddy, 2003; Zhou et al., 2001). Factors that control the cycling of phosphorus within a lake are discussed in Chapter 2.

Land use (Figure 1.2) affects the external loading of nutrients to a lake both through runoff (Johnes et al., 1996; Reddy et al., 1999; Soranno et al., 1996) and atmospheric deposition (Poor et al., 2005). The eastern part of the Tampa Bay watershed is within a phosphorus mining region known as Bone Valley (Brown, 2005), and lakes in that area may be influenced by naturally-occurring phosphorus minerals or by phosphorus mining and fertilizer manufacturing activities. Much of the eastern portion of the watershed is agricultural, and cattle ranches, citrus groves, strawberry farms, and landscape plant nurseries are common sights. Applications of fertilizers or pesticides and seepage from piles or ponds of animal wastes are sources of nitrogen and phosphorus from agricultural land use (Arbukle and Downing, 2001; Bennett et al., 2001; Carpenter, 2005; Omernik, 1976). The northeastern sector of the watershed includes forested wetlands that drain into the Hillsborough River, which is the main source of drinking water for the City of Tampa (Schmidt and Luther, 2002; Xiana and Craneb, 2005). To a lesser extent, these forested wetlands contribute nutrients from natural flora and fauna to its surface waters (Reddy et al., 1999).

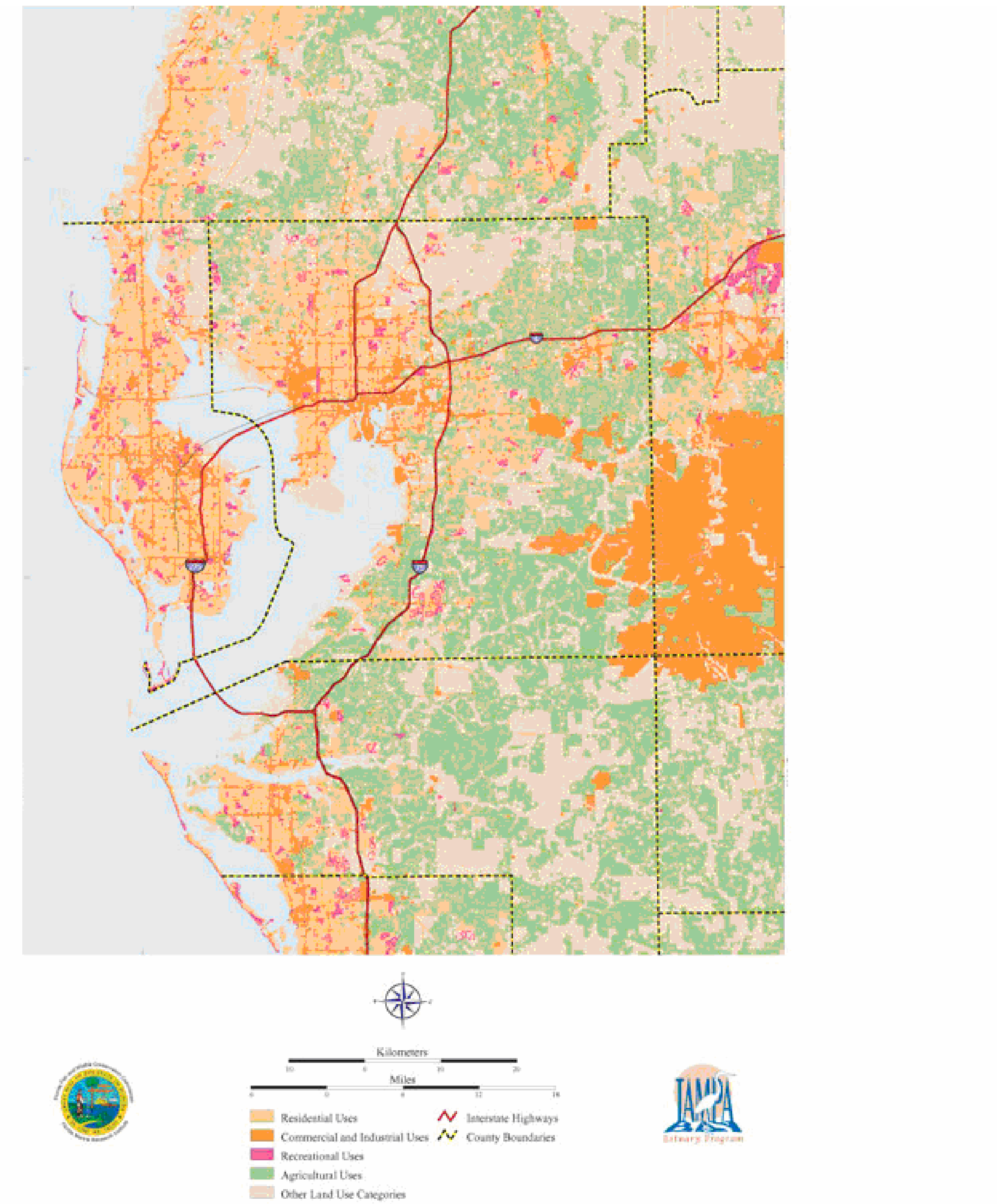
Closer to the bay the land use is predominantly urban or suburban, with heavy industry at the confluence of major rivers and the bay (Xiana and Craneb, 2005). The four-county Tampa Bay Metropolitan Area, which is host to the Cities of Clearwater, St. Petersburg, and Tampa, has a population of 2.7 million residents. Population growth was 30.45% between 1990 and 2006 (Hilssborough

County, 2007; US Bureau of the Census 2000, 2007), and was higher than the 20.34% of the total United States population for the same time period (Hilssborough County, 2007; US Bureau of the Census, 2006). The shift from rural to suburban or urban land use means more impervious cover and thus increased rainfall run-off (Xiana and Craneb, 2005). Run-off may contain nitrate or phosphate from inorganic fertilizers, or organic forms of nitrogen and phosphorus from leaves, insect debris, and animal excreta, as examples (Johnes et al., 1996; Reddy et al., 1999; Soranno et al., 1996). Atmospheric emissions of nitrogen oxide and ammonia from electrical utility, industrial, and transportation (including motor vehicle) sectors deposit to lake surfaces or adjacent drainage basins (Poor et al., 2005). Non-point source pollution from run-off or atmospheric deposition may increase with population density (Smith et al., 2003).

Septic tanks for household sewage treatment were typical in twentieth century development of Florida lakefront property, and many if not most of the homes bordering the lakes considered in this study are still on septic systems (Schmidt and Luther, 2002).

Much of the Tampa Bay watershed is underlain by karst geology (van Beynen et al., 2007), which in some places facilitate groundwater transport and for a few lakes provides a direct connection to the cleaner water of the Floridian Aquifer (Cheng and Kindinger, 2004).

Figure 1.2 Land Use in Tampa Bay watershed (Tampa Bay Estuary Program, 1999). Lakes analyzed in this study are located in areas with residential, commercial and industrial, and agricultural use mainly.



1.3 Research Hypotheses

The research hypotheses were:

- Average lake water concentrations of total phosphorus (TP), the ratio of total nitrogen to total phosphorus (TN:TP), and phytoplankton as measured by chlorophyll-α in selected lakes of the Tampa Bay watershed changed significantly between 1990 and 2007;
- Lakes with a greater abundance of submerged aquatic vegetation have significantly lower water concentrations of total phosphorus, total nitrogen (TN), and chlorophyll-α.

Eutrophication (primary productivity of natural waters) is traditionally measured based on the concentration of total phosphorus (TP) and total nitrogen (TN) as well as the concentration of phytoplankton (Canfield et al., 1985; Dillon and Rigler, 1974). For accuracy and functionality, phytoplankton is estimated by the measurement of chlorophyll-α in lake water (Canfield et al., 1985; Dillon and Rigler, 1974). Consequently, lake water concentration of TP, TN, and chlorophyll-α were water quality parameters used for this analysis. Special attention was given to lake water TP concentration and the metabolism of phosphorus in shallow lakes since this is the most common nutrient limiting phytoplanktonic productivity in freshwater ecosystems (ChunLei et al., 2004; Schauser et al., 2004; Schindler, 1977; Scinto and Reddy, 2003; Zhou et al., 2001).

This dissertation has been divided in complementary subtopics, each one covered in an independent chapter with specific objectives and results as follows:

- Chapter 2
  - Review of processes that determine the primary productivity of aquatic systems;
  - Definition of external factors that influence the processes involved in the phosphorus cycle in aquatic systems; and
  - Review of models of phosphorus fate and transport in aquatic systems.
- Chapter 3
  - Analysis of temporal trends of water concentration of total phosphorus (TP), ratio of water concentration of total nitrogen to water concentration of total phosphorus (TN:TP), and water concentration of chlorophyll-α in selected lakes of the Tampa Bay watershed;
  - Discussion of differences in trends of lake water concentration of TP, TN:TP ratio, and chlorophyll-α between lakes; and
  - Discussion of factors influencing the change in the lake water concentration of TP, TN:TP ratio, and chlorophyll-α.
- Chapter 4
  - Analysis of correlation between measures of submerged aquatic vegetation (SAV), lake water concentrations of TP, TN, and chlorophyll-α, and lake depth, volume, and area; and
  - Discussion of factors influencing lake water concentration of TP, TN, and chlorophyll-α ;

- Chapter 5
    - Analysis of association between submerged aquatic vegetation and lake water concentrations of TP, TN, and chlorophyll-α.
- Chapter 6
    - Implications of this research.

CHAPTER 2.

LITERATURE REVIEW

2.1 Introduction

Phosphorus is a fundamental component of life. Its presence in the adenosine di- and triphosphorus molecule make possible the conversion of energy from sunlight into chemical energy in plants, and fuels the indispensable reactions of photosynthesis and respiration (Chameides and Perdue, 1997). These processes do not just support the entire ecological web by providing energy to fuel all the metabolic transformations in living organisms but also determine the flow of phosphorus between biosphere and mineral reservoirs.

Essentially all the phosphorus significantly present in nature is in the form of orthophosphates (+5 oxidation state) since it is the only form stable in aqueous solution (Chameides and Perdue, 1997). Unlike other essential elements for life like nitrogen and carbon, phosphorus does not have a stable gaseous form of significance (Chameides and Perdue, 1997; Lahm, 2008; Schlesinger, 1991) and its occurrence in the atmosphere is generally limited to just minor amounts dissolved in moisture or contained in suspended dust particles (Graham and Duce, 1979). Its presence in nature is more prevalent in the form of insoluble compounds since its cycling depends mostly on slow geologic processes such as weathering of apatite and other calcium phosphorus minerals (Schlesinger, 1991). Although plant roots and mycorrhizae may accelerate this process, their effect is not enough to consider a significant role of microorganisms in increasing phosphorus bioavailability (Schlesinger, 1991). Therefore, it is frequently a limiting nutrient for photosynthetic productivity, especially in aquatic ecosystems (Chameides and Perdue, 1997) and even more so in freshwater ecosystems (ChunLei et al., 2004; Schauser et al., 2004; Schindler, 1977; Scinto and Reddy, 2003; Zhou et al., 2001).

Although this limitation slows down natural productivity, the increasing discharge of phosphorus into natural waterways as a result of human activities is accelerating the availability of this nutrient (Rast and Thorton, 1996). Hence, a basic, but clear, understanding of the phosphorus cycle in lakes is of vital importance for environmental management plans addressing the problem of eutrophication.

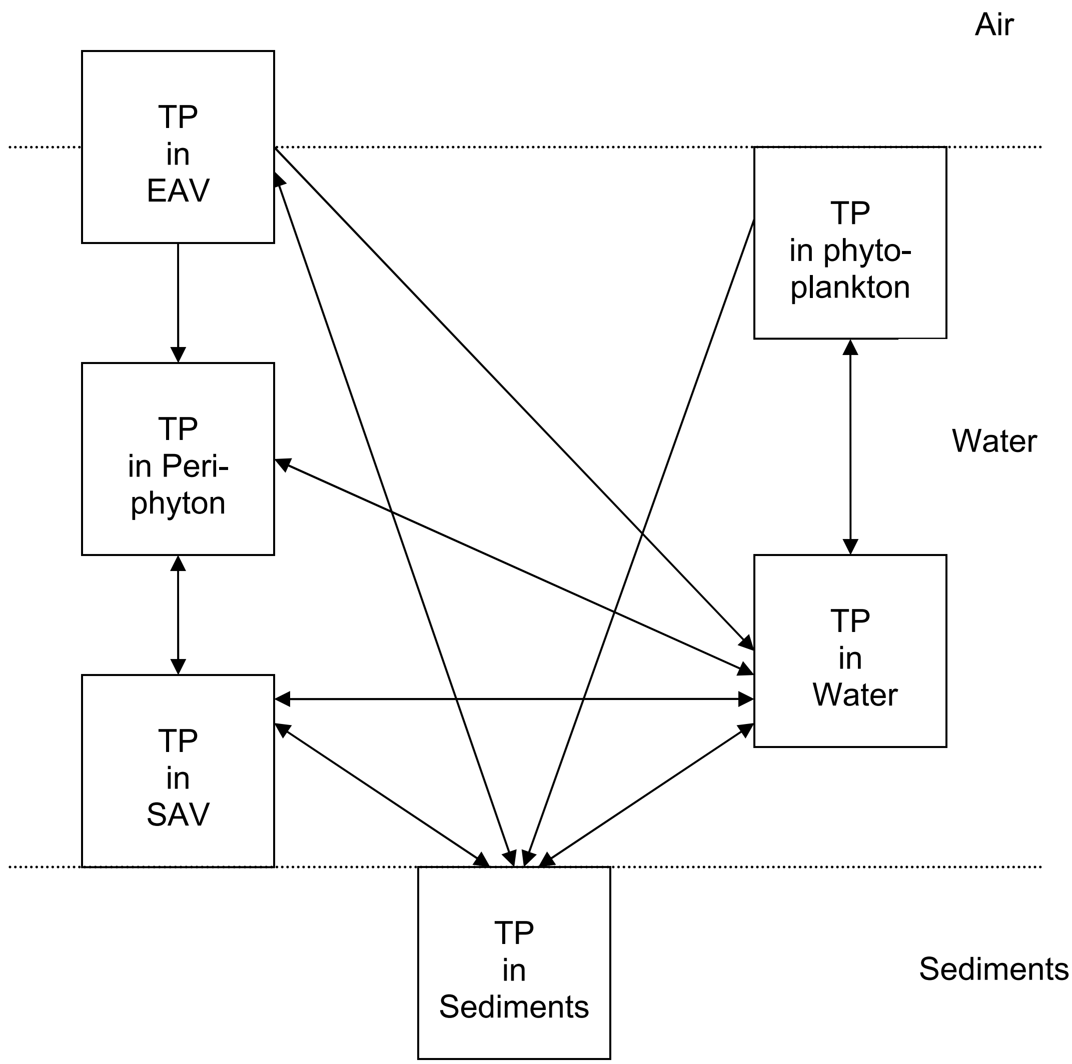
In aquatic ecosystems, phytoplankton growth removes soluble phosphorus from the water column while phytoplankton decomposition releases it back into the water and sediments (Fisher et al., 1982). This classical view of decomposition of organic matter and consequently regeneration of mineral nutrients to support primary production at the base of the food chain is correct but too simplistic (Sundby et al., 1992).

Phosphorus in aquatic ecosystems is present mostly as a particle but also in dissolved form (Schnoor, 1996; Wang and Mitsh, 2000), and each form behaves differently in the phosphorus cycle.   Particle phosphorus can settle by gravity while dissolved phosphorus can be assimilated by bacteria and plants.

Considering total phosphorus rather than differentiating phosphorus forms has been a more functional expression for modeling purposes because it simplifies the process (Wang and Mitsh, 2000). Since it is impossible to completely describe the phosphorus cycle, a basic description of the most important factors necessary for a reasonable understanding of the phosphorus cycle in shallow lakes follows. Figures 2.1 provide a simple visual description of some of the main reservoirs and flows of total phosphorus cycling in shallow freshwater systems.

Although not included in the figure, external inputs and outputs are determinant in the phosphorus budget of the system. The most important external contributions of phosphorus to the system are originated from non-point sources such as agriculture and urban activity (Carpenter et al., 1998; Howarth et al., 1996). Point sources are usually less important, although may be also significant (Carpenter, 2005; Cowen and Lee, 1976). In most cases, groundwater flow and atmospheric deposition are not important sources of phosphorus because it is normally not mobile in soils (Reddy et al., 1999) and its content in dust and suspended solids in the air is very low (Graham and Duce, 1979). This may not be the case for the lakes examined in this study for the following reasons: the central and northern part of Hillsborough County has a karst formation of sandy soil (van Beynen et al., 2007) that may facilitate the phosphorus leaching, and atmospheric deposition is actually a significant contributor of phosphorus to Florida lakes (Reddy et al., 1999).

Figure 2.1 Total phosphorus cycle in shallow aquatic systems. The connections in this figure represent the flows of phosphorus between the different components. The arrow heads point to the component receiving phosphorus. Those arrows with heads in both directions represent bi-flows of phosphorus. External inputs and outputs are not included in the diagram for visual clarity, but are important determinants in the phosphorus budget of the system.



TP: total phosphorus.
SAV: submerged aquatic vegetation.
EAV: emergent aquatic vegetation.

2.2 Inorganic Phosphorus

Organic and inorganic phosphorus compounds from the watershed are carried with runoff and discharged into the lake water. Eventually they settle into the lake sediments. Once in the sediments, persistent compounds of inorganic phosphorus stay buried in their original form while labile forms are dissolved into the sediment pore water. From the sediment pore water, phosphorus is released as phosphorus back into the water column, re-precipitated, or adsorbed by other compounds within the sediments (Sundby et al., 1992).

Under oxic conditions, binding of inorganic phosphorus to iron in the sediments has an important effect on the phosphorus mass balance in aquatic systems (Krom and Berner, 1980; 1981). The oxidized surface layer in sediments decreases the flux of dissolved inorganic phosphorus from the sediment to the water column by providing iron oxide that binds to the phosphorus, therefore trapping phosphorus (Mortimer, 1971). Hence, as the oxygen concentration decreases with depth in the sediments, the concentration of sequestered phosphorus decreases with depth as well (Sundby et al., 1992). This process of phosphorus sequestration has been defined in two steps: rapid adsorption on surfaces and then slow diffusion into the particles of iron oxide (Barrow, 1983). It has been suggested however, that particles at depths equal or greater than 10 cm may still maintain some capacity to retain phosphorus, despite the anoxic environment (Silverberg et al., 1987).

2.3 Organic Phosphorus

Usually most of the phosphorus present in the water column and sediment is organic (Rigler, 1956). Bacteria and fungi decompose plant and animal tissue into more simple organic matter (Fenchel, 1970). The resulting organic matter is further transformed by heterotrophic bacteria into bacterial protoplasm. Bacteria are in turn consumed by protozoan grazers, resulting in regeneration of inorganic phosphorus. According to Johannes (1965), bacteria alone release little phosphorus, instead they consume it, so bacterial grazers are needed for mineralization of this bacterial phosphorus. Barsdate et al. (1974), however, suggested based on laboratory experiments that little phosphorus from bacteria pass through grazers before is released into solution. This author explained that the reason for an increased water phosphorus concentration with a high presence of grazers is due to the fact that the physiology of bacterial population is changed by grazing pressure by selection of rapidly growing forms of bacteria. This change caused by grazing results in more rapid bacterial assimilation of organic phosphorus and subsequent faster regeneration of inorganic phosphorus, which is the form assimilated by phytoplankton (Barsdate et al., 1974).

Once organic phosphorus has been mineralized, it undergoes the same pathways as deposited inorganic phosphorus. A portion of the mineralized phosphorus is concentrated as phosphorus into the pore water where it is maintained in equilibrium with the portion adsorbed to the surface sites. Part of the phosphorus adsorbed to particles is then diffused to the interior of the iron oxides and stays sequestered. As dissolved phosphorus is released from the pore water back to the water column, it is also replaced by surface adsorbed phosphorus to maintain the equilibrium concentration (Sundby et al., 1992).

2.4 Oxygen

Changes in the oxygen concentration in the overlying water can cause drastic changes in the phosphorus flux from sediments, with anaerobic conditions increasing phosphorus flux to the overlying water (Moore et al., 1998; Moore et al., 1991), thereby making it available to phytoplankton and eventually promoting algal blooms (ChunLei et al., 2004). As phytoplankton from a bloom sink to the sediments, decomposer bacteria consume the decaying material (Fenchel, 1970) and the remaining dissolved oxygen, further exacerbating the oxygen depletion. In opposite circumstances, high photosynthetic activity of benthic microalgae decrease phosphorus release by increasing dissolved oxygen levels (Spears et al., 2008). Additionally photosynthesis can lead to elevation in pH, and high pH under calcareous conditions favors co-precipitation of phosphorus with calcium carbonate (Dierberg et al., 2002; Spears et al., 2008). It has also been suggested that an elevated pH can affect ion exchange processes that decrease the capacity of iron and aluminum compounds to bind phosphorus, resulting in release of phosphorus from sediments (Bostrom et al., 1988; Zhou et al., 2001).

The flux of phosphorus from sediments to water column is regulated primarily by redox reactions involving iron and aluminum in sediments, and then by the gradients in the concentration of phosphorus between pore water and overlying water (Moore et al., 1998). Levels of nitrate and sulfate have been inversely correlated to the flux of phosphorus from sediments to overlying water (Fisher and Reddy, 2001). This might be due to the oxygen content in these compounds that can raise the overall oxygen level in the media hence influencing redox reaction in phosphorus chemistry. By contrast Caraco et al. (1989) suggest greater release of phosphorus from sediments at higher sulfate concentrations.

In lakes with a thermocline, the release of phosphorus from sediments is typically controlled by the oxygen concentration in the hypolimnion, and decomposition of phytoplankton sinking into this layer is a significant cause of oxygen depletion (Genkai-Kato and Carpenter, 2005). Deeper in the sediment column, even below aerobic overlying water, conditions are increasingly anoxic, facilitating the release of phosphorus from iron and aluminum bound phosphorus undergoing reduction (Moore et al., 1991; Sundby et al., 1992). This newly regenerated phosphorus can migrate upward to participate in exchange reactions between adsorption sites and pore water, or can be released into the overlying water (Sundby et al., 1992).

Additionally to the reduction of iron and aluminum, part of the increased phosphorus release from sediments under anoxic conditions may be also explained by metabolic changes in microbial population that cause release of phosphorus from the cells (Bostrom et al., 1988; Gatcher et al., 1988). Complexed phosphorus and iron (II) is released into water solution when the cells, under anoxic conditions and in the absence of nitrates, use iron (III) as an alternative electron acceptor (Jones et al., 1983). Moreover, microbial growth yield is typically low under anoxic conditions, hence the release of phosphorus previously bound to carbon in bacteria and cyanobacteria is higher (Bostrom et al., 1988).

2.5 Submerged Aquatic Vegetation

Aquatic macrophytes in general (both emergent and submerged) can absorb nutrients both from the water and from the sediments depending upon the species and relative nutrient concentrations in water and sediments (Denny, 1972; Graneli and Solander, 1988; James et al., 2006; Rattray et al., 1991). Emergent macrophytes take phosphorus from the sediments under all conditions while submerged aquatic vegetation takes phosphorus from both water and sediments although the latter pathway is more prevalent under normal conditions as well as under conditions of high water phosphorus concentration (Graneli and Solander, 1988).

Barko and Smart (1980), Barsdate et al. (1974) and Mcroy (1972) indicated that aquatic macrophytes can pump nutrients from sediments to the water column making it available to phytoplankton. Wang and Mitsh (2000) included this effect in a phosphorus cycling model. Graneli and Solander (1988) suggested that both types of aquatic vegetation, submerged and emergent, can release minimal or important quantities of phosphorus to the water column depending on the vegetative stage. Growing vegetation would release minimal quantities of phosphorus via plant material, and decaying vegetation would release considerable amounts. According to Gumbricht (1993), the main phosphorus removal mechanism in designed wetlands would be nutrient uptake by submerged aquatic vegetation followed by harvesting. Other theories suggest that aquatic macrophytes should not be removed because they decrease recycling of phosphorus from sediments back into the overlying water by suppressing resuspension of sediments. This principle was included in prediction models developed by Genkai-Kato and Carpenter (2005).

Inhibition of sediment resuspension is a simple mechanism that explains at least part of the effect of submerged aquatic vegetation in the suppression of phosphorus recycling (Hamilton and Mitchell, 1996; Scheffer, 2004). Emergent and submerged aquatic vegetation neutralize the effect exerted by waves and wind in causing vertical mixing of the overlying water in shallow lakes (Bachmann et al., 2004; Hamilton and Mitchell, 1996). Additionally a possible intense photosynthesis caused by aquatic macrophytes and associated periphyton can increase the pH and lead to coprecipitation of phosphorus with calcium carbonate under alkaline conditions (Dierberg et al., 2002). In contrast, elevated pH in the interface water sediments resulting from photosynthesis may cause release of phosphorus from iron and aluminum compounds mostly because orthophosphate is replaced by hydroxide ions in ligand exchange reactions (Bostrom et al., 1988). It has been explained that aquatic vegetation in general affects the water chemistry by regulating oxygen and pH therefore influencing the phosphorus cycle in lakes (Graneli and Solander, 1988).

Emergent and submerged aquatic vegetation provide substrate surfaces for epiphytes and periphyton to grow (Cattaneo and Kalff, 1980; Dierberg et al., 2002), which can offer another important regulatory impact on the phosphorus flux. Periphytic algae and other epiphytes growing on the surface of submerged aquatic vegetation remove phosphorus directly from the water column (Dierberg et al., 2002; Scinto and Reddy, 2003). Submerged aquatic vegetation can also directly uptake part of its required nutrients from the water column in addition to sediments (Graneli and Solander, 1988). This absorption and translocation of phosphorus through out the entire plant material is another important part of the lake water phosphorus cycle (Barko and Smart, 1980).

The above-mentioned mechanisms refer to the availability of phosphorus in the water column. Yet there are also other ways by which submerged aquatic vegetation influence the growth of phytoplankton. Macrophytes can indirectly suppress phytoplankton growth by providing shelter from fish to the zooplankton that graze on phytoplankton (Scheffer, 2004; Scheffer et al., 2001). Zooplankton that graze on bacteria, however, may also accelerate the mineralization process that makes phosphorus available in the water column (Barsdate et al., 1974). Aquatic vegetation, in general, provides a refuge against cladocera (microscopic order of crustacean and part of zooplankton population), which are the most efficient grazers of bacteria and a favorite prey for fish (Moss, 1990) . In a laboratory experiment, Rigler (1956) found that some species of bacteria that utilize inorganic phosphorus grow well in suspension, but others grow well on the walls of storage vessels. Hence, in a natural environment, macrophytes might provide surfaces for phosphorus-consuming bacteria to grow.

Independent of the mechanisms utilized by submerged aquatic vegetation for regulation of phosphorus levels, there is extensive literature demonstrating the phosphorus removal capacity of submerged aquatic vegetation in constructed wetlands and lakes (Dierberg et al., 2002; Gu et al., 2001; Gumbricht, 1993; Knight et al., 2003). Gumbricht (1993) stated that the phosphorus absorption rate of submerged aquatic vegetation is proportional to the phosphorus concentration within the surrounding water. These results are consistent with experiments done by James et al. (2006) showing a direct relationship between nutrient concentration in submerged vegetation tissue and water column nutrient concentration. This is of special interest regarding the potential use of submerged aquatic vegetation as a treatment method to remove nutrients in nutrient-rich water.

Although the nutrient removal capacity of submerged aquatic vegetation and mechanisms through which submerged aquatic vegetation may regulate the phosphorus cycle in lakes has been documented, the association between nutrient concentration and submerged aquatic vegetation biomass in Florida lakes has not been well established. It has been theorized based on correlations found in some studies in Florida that submerged aquatic vegetation would cause lower water nutrient concentrations (Bachmann et al., 2002; Bachmann et al., 2004), except at high nutrient levels, when the influence would work in the opposite way. Other authors indicate that elevated nutrient concentrations in lake water would cause the absence of submerged aquatic vegetation because of light attenuation (Duarte, 1995; Graneli and Solander, 1988). A study conducted in Québec, Canada, reported that slope of the littoral zone is a more important determinant of the variability in submerged aquatic vegetation biomass (Duarte and Kalff, 1986).

In summary, extensive submerged aquatic vegetation may lead to low nitrogen and phosphorus concentrations in the water column, which in turn leads to low phytoplankton growth. These two associations may explain why submerged aquatic vegetation has been reported as inversely influencing phytoplankton (as estimated by chlorophyll-α) under Florida conditions (Canfield and Hoyer, 1992). This would be an indirect connection between submerged aquatic vegetation and phytoplankton by means of the concentration of total phosphorus in the water column (see Figure 2.1). The authors indicated that a percentage area cover (PAC) of submerged aquatic vegetation greater or equal to approximately 30% of lake area is required for noticeable reductions in phytoplankton biomass. They found that a small amount of aquatic vegetation does not play an important role in phytoplankton biomass reduction. A study conducted in New Zealand lakes, however, provided evidence that submerged aquatic vegetation dominates phytoplankton in shallow eutrophic lakes (Hamilton and Mitchell, 1996), and the suggested mechanism of control was the stabilization of lake sediments and the inhibition of sediment resuspension.

In literature reported on Florida lakes, the association between submerged aquatic vegetation and nutrients is still unclear, however, a strong direct association has been reported between water nutrient concentration and phytoplankton concentration, as measured by chlorophyll-α (Bachmann et al., 2002; Brown et al., 2000; Canfield et al., 1984).

Drastic increases in lake water concentration of nutrients and turbidity have been reported when submerged aquatic vegetation was removed by herbicide treatment (O'Dell et al., 1995), or by hurricanes (Bachmann et al., 1999). Similarly, lakes have been reported to switch from turbid to clear water state when planktivorous fish were removed and submerged aquatic vegetation increased (Ozimek et al., 1990).

2.6 Light

Light can affect the phosphorus cycle indirectly through the presence of submerged aquatic vegetation, phytoplankton, turbidity, and benthic microalgae. Light can influence water phosphorus concentration through phytoplankton (Philips et al., 1997), by favoring growth of this microscopic algae and cyanobacteria, consequently removing phosphorus from solution and incorporating it into easily sedimented biomass (Krivtsov et al., 2000).

Likewise the intensity of light reaching the sediments determines the distribution and abundance of submerged aquatic vegetation (Hoyer et al., 2004). There is some discrepancy in the literature regarding the feedback that nutrient loading may have on submerged aquatic vegetation through shading by phytoplankton. Spence (1982) found light availability as an important consequence of excess nutrient loading and one of the major factors determining the submerged aquatic vegetation distribution. The increase in phytoplankton as a result of larger phosphorus inputs would block light from reaching the submerged aquatic vegetation, reducing their distribution to just the very shallow zone. Subsequently, their capacity to restrain phosphorus recycling back into the water column would also be reduced (Genkai-Kato and Carpenter, 2005).

Light penetration to the sediments also allows photosynthetic activity of benthic microalgae. This regulates the flux of phosphorus from sediments to water by affecting the oxygen concentration and pH level and thus phosphorus sequestration in sediments (Spears et al., 2008).

It has been speculated that periphytic algae attached to the surface of submerged aquatic vegetation might be the cause of the shading effect responsible for loss of submerged macrophytes under conditions of high nutrients loads (Philips et al., 1978). Results of Bachmann et al. (2002) suggested that this is unlikely under conditions of Florida lakes, because they did not find an association between increasing nutrient loading and periphyton abundance in their study. They explain this by suggesting that the shading effect caused by phytoplankton as a result of increasing nutrient concentration keeps periphytic algae from receiving enough light for photosynthesis. Periphytic algae are attached to macrophyte surface and therefore unable to move toward the source of light, while phytoplanktonic algae can move in the water column toward the source of light, blocking it from reaching the periphyton (Bachmann et al., 2002). In this case, if periphyton competes with submerged aquatic vegetation for light and periphyton is reduced with increased nutrient loading because is shaded by phytoplankton, then a high nutrient concentration in the water column would add shading from the phytoplankton but reduce shading from the periphyton to submerged aquatic vegetation.

Light penetration is dependent in great measure on turbidity. Scheffer's basic model for alternative stable states in shallow lakes (Scheffer, 2004) assumes that an increase in nutrient levels causes an increase in turbidity, turbidity is reduced by submerged vegetation, but submerged vegetation disappears under conditions of extreme turbidity. Likewise Bachmann et al. (2002) indicate that submerged aquatic vegetation tolerate increasing water nutrient concentrations (which is one of the factors leading to turbidity) below excessive levels. Thus, as long as the nutrient concentration is not extreme, submerged aquatic vegetation amount determines nutrient concentrations in lake water (Bachmann et al., 2002; Bachmann et al., 2004).

Silica can obstruct the effect of light by triggering the growth of diatoms (Krivtsov et al., 2000). Diatoms (phytoplanktonic algae) grow in abundance under high concentrations of silica, thereby shading submerged aquatic vegetation and periphyton, and taking more phosphorus from the water column, making it less available for periphyton.

2.7 Mean Depth and Temperature

Temperature affects lake water phosphorus cycling primarily through its effect on biological activity (Bostrom et al., 1988). Increased temperature favors microbial activity, which in turn decreases oxygen concentrations and pH via microbial consumption of oxygen. As previously described, low oxygen levels in sediments cause the release of phosphorus bound to iron and aluminum complexes from sediments into solution (Moore et al., 1991; Richardson, 1985; Sundby et al., 1992). High temperature coupled with high pH can also have the opposite effect by removing phosphorus from solution via co-precipitation with calcium carbonate (Bostrom et al., 1988; Dierberg et al., 2002). In a broad sense, temperature influences the phosphorus cycling in aquatic systems by affecting the rate at which chemical processes take place (Simpson and Eaton, 1986).

Empirically-derived models developed by Genkai-Kato and Carpenter (2005) indicated that the reversibility of conditions of high concentration of phytoplankton back to a clear water state was affected considerably by mean depth, temperature, and presence of submerged aquatic vegetation. In shallow lakes (<1.9 m) with submerged aquatic vegetation, phosphorus recycling from sediments decreased as depth decreased because of more vegetation at shallow depths, increasing the likelihood of restoration back to the clear water state for shallow lakes. In the absence of submerged aquatic vegetation, however, shallow lakes were more vulnerable to eutrophication and less likely to return to clear water state because phosphorus recycling increased with temperature and with decreased depth. Submerged aquatic vegetation was not found to have an effect at depths greater than 10 m. Dilution of phosphorus in the hypolimnion and reduction of temperature are the possible factors suppressing recycling of phosphorus at depths greater than 28 m. An intermediate mean depth (1.9-28 m) is most resistant to reversing from eutrophication to the clear water state because these lakes were too deep to be affected by submerged aquatic vegetation and too shallow for phosphorus dilution in the hypolimnion.

2.8 Bacteria and Aquatic Animals

An increase in microbial activity leads to increase in oxygen and nitrate consumption. As oxygen concentration decreases, the capacity of complexes of iron, aluminum, and manganese to retain phosphorus decreases as well (Bostrom et al., 1988). Anaerobic conditions cause changes in the metabolism of bacteria and cyanobacteria that increases phosphorus release from the cells (Gatcher et al., 1988). In addition, the biomass of microbial population is reduced under anaerobic conditions hence less retention of phosphorus in microbial mass means more phosphorus is available in solution (Bostrom et al., 1988).

It is not clear the role of bacteria in the phosphorus cycling but most literature seems to agree that in general they do not actively excrete phosphorus, and demand more phosphorus from the water column than they release (Andersson et al., 1988; Johannes, 1968). The release of phosphorus caused by bacteria (Azam et al., 1999) is actually presumed to be an indirect effect through zooplankton excretion after bacteria is consumed by zooplankton (Ammerman and Azam, 1985) or when bacteria die or are infected by viruses. Other authors suggested that although bacterial grazers in fact stimulate the release of phosphorus from bacterial biomass into the water column, this happen because the grazing pressure makes faster and more efficient the phosphorus release process in bacteria but not because phosphorus actually pass through the grazers in significant amounts (Barsdate et al., 1974).

In the absence of a large population of bacterial grazers, heterotrophic bacteria utilize a greater amount of inorganic phosphorus in the process of organic matter decomposition (Johannes, 1965). Since most organic matter comes from carbohydrates, which are poor in essential nutrients, bacteria use inorganic phosphorus from free water as an extra source of phosphorus for growth (Fenchel, 1970). This would reduce the amount of available phosphorus in the water column under conditions not limited by carbon, since bacteria can successfully compete with phytoplankton for uptake of phosphorus due to a faster growth rate (Rhee, 1972).

Macroinvertebrates affect phosphorus cycling in many ways. They promote the rate of phosphorus mineralization (Hansen et al., 1998) and increase phosphorus release through their digestive and excretory processes (Gardner et al., 1981). Bioturbation generated by macroinvertebrates in the sediments cause water movement, which favors the diffusion of phosphorus into the overlying water. Burrows made by some animals further increase the surface of contact between interstitial and overlying water (Bostrom et al., 1988), which enhances the exposure of sediments to oxygen, thereby augmenting retention of phosphorus by iron oxide, but also facilitates the transport of phosphorus contained in the pore water out to the overlying water (Hansen et al., 1998). Burrowing animals also transport oxidized and reduced compounds back and forward between the reduced and oxidized zone, promoting redox reactions (Canfield et al., 1993).

Mussels and other macroinvertebrates also have a strong regulatory impact on phosphorus cycling by regulating the population of phytoplankton, zooplankton, and bacteria (Barsdate et al., 1974; Berman and Richman, 1974; Johannes, 1968; Scheffer, 2004). Fish are especially important since they impose a direct and indirect effect on the water column phosphorus in aquatic systems. They exert a predatory pressure on bacteria, zooplankton, phytoplankton and benthic invertebrates, which indirectly affect the flux of phosphorus (Andersson et al., 1988; Kitchell et al., 1979; Scheffer, 2004). Fish also directly affect phosphorus cycling by increasing resuspension of sediments in search for benthic food (Scheffer et al., 2001). The processes by which fish and invertebrates influence the phosphorus cycle are strongly related to vegetative cycles and are more intense during high temperature periods (Andersson et al., 1988).

2.9 Land Use / Land Cover

Conditions in the watershed influences lake water quality through the inputs of materials and nutrients via runoff (Reddy et al., 1999; Soranno et al., 1996). A study conducted by Smith et al. (2003) reported that runoff, area of the watershed, and population, were the three variables more strongly associated to loading of dissolved nitrogen and phosphorus into receiving waters. Trends in land use, agriculture, human population (Johnes et al., 1996; Smith et al., 2003) as well as mining and forestry practices in watersheds have increased the transport and discharge of phosphorus to water bodies (U. S. Environmental Protection Agency 1990 as quoted by Soranno et al., 1996). Contributions of phosphorus to lakes can come from point sources such as waste water treatment plants, as well as from non-point sources such as runoff from agricultural and urban lands (Carpenter, 2005; Cowen and Lee, 1976).

Non-point sources resulting mainly from agricultural and urban activity are the most important sources of phosphorus to natural bodies of water in the United States and other developed countries (Carpenter et al., 1998; Howarth et al., 1996). According to Reddy et al. (1999), urban land use is the category with the higher contribution of phosphorus to receiving waters, followed by agricultural and forested land use. This is explained by the greater fraction of impervious layer in urban areas, which reduces infiltration and increases flow of runoff. This subsequently results in greater phosphorus export. Groundwater flow is commonly not an important passage of phosphorus from the watershed to lakes because of the generally low mobility of phosphorus in the soil. The high porosity of the karst formation in northeastern and central Hillsborough County (van Beynen et al., 2007) might prove the exception.

In agricultural land use, phosphorus surplus from excessive fertilization and manure production is accumulated in the soil further finding its way with the runoff into natural water bodies (Bennett et al., 2001; Carpenter, 2005). A study conducted in agricultural areas of the eastern United States reported that the magnitude of runoff nutrient concentration is usually relative to the percentage of the watershed covered with cultivated land (Omernik, 1976). Other study conducted in the highly agricultural region of Iowa found that watersheds dominated by animal agriculture constitute the main source of phosphorus into natural waters, while intensive row crop agriculture-dominated watersheds are larger sources of nitrogen (Arbukle and Downing, 2001). Reddy et al. (1999) says that atmospheric phosphorus deposition (especially dry deposition) is a significant and local contributor of phosphorus to Florida lakes and that the loading is higher for agricultural lands as compared with forested watersheds.

In Florida, phosphorus mining is an important economical activity but has heavily impacted waters that reach Tampa Bay (Baskaran and Swarzenski, 2007). The area in the eastern part of the Tampa Bay watershed extends into Bone Valley, which has been mined for phosphorus since the late 1800s. That area is characterized by an abundance of wetlands and its natural ecology that has been disrupted by the phosphorus mining. After 20 years of active restoration conducted in the region, however, has water quality improvements have been seen (Brown, 2005). Phosphorus removal by wetland restoration and treatment of point sources can significantly reduce the frequency of algal blooms (Billen and Garnier, 1997). Land use and its corresponding contributions of phosphorus to lakes must be considered in phosphorus budget calculations for planning lake management options (Cowen and Lee, 1976).

2.10 Public Health Implications

Beside the widely known adverse effect in the ecosystem: light attenuation, odors, lowering of dissolved oxygen, and fish kills (Paerl, 1988); eutrophication represents also a serious threat to aesthetics and most importantly to public health (FDEP, 2004). A study at the University of Colorado concluded that nutrient enrichment in lakes and ponds, cause conditions favourable to abundance of trematode parasites; and also suggested that favour conditions for mosquito vectors of malaria, cholera causing bacteria, and swimmers itch (Johnson et al., 1999).

There are species of phytoplanktonic algae, that can be toxic and that are triggered by surface water enrichment. Abundant productivities of any toxic algae are better known as harmful algal blooms (HABs) and are a growing concern for public health in Florida because their potential effect in surface drinking water resources and recreational sites (FDEP, 2004). Cyanobacterial species are a phylum of phytoplankton with a nitrogen-fixing capability. They would be favored over other species of phytoplankton by increasing conditions of relative limited nitrogen and abundant phosphorus. As a consequence, low ratios of total nitrogen to total phosphorus (TN:TP) could lead to increase of cyanobacterial water concentration (Hecky and Kilham, 1988; Levich, 1996;

Levich and Bulgakov, 1992). This can be a concern to public health because certain species of cyanobacteria produce cyanotoxins that cause oxidative stress in affected cells and may promote tumors in the nervous, hepatic, and dermatologic systems (Fleming et al., 2002; Karjalainen et al., 2007). These toxins may enter the human body through consumption of drinking water, inhalation of aerosolized toxins, and by contact with water or fish from water polluted with cyanobacteria (Fleming et al., 2002; Karjalainen et al., 2007).

The relative abundance of cyanobacteria over other phytoplanktonic species causes interferences in the food-web (Levich, 1996). They are not consumed by many species, and this lack of predatory pressure further helps them to dominate over competitor species that are producers in the food chains (Levich, 1996). The cyanobacteria produced toxin can be transferred through the food chain through consumption of some grazers, affecting the upper levels of the trophic web and even humans (USEPA, 1997). In small amounts cyanobacteria may complement the diet of tolerant grazers, but when ingested in excess the cyanotoxins may decrease the eggs production of some species of zooplankton and reduce feeding and growth rates of fish larvae (Karjalainen et al., 2007). The process of detoxication for the possible consumers of cyanobacteria imply a metabolic cost, resulting in a decreased growth and condition and subsequently in their susceptibility to be predated for other higher consumers (Karjalainen et al., 2007).

Some literature suggest that harvesting of submerged aquatic vegetation may increase the risk of cyanobacterial blooms (Scheffer, 2004) but other found that, at least under conditions of low nutrient levels, cyanobacteria did not increase as a result of harvesting submerged aquatic vegetation (Morris et al., 2006).

2.11 Summary of Phosphorus Cycle Review

Among the most important processes regulating phosphorus cycling in shallow freshwater lakes are sedimentation, plant nutrient uptake, and regulation of ion exchange processes via dissolved oxygen and pH. Emergent and submerged aquatic vegetation enhance sedimentation mainly by suppressing water turbulence and thus favouring conditions for particle settling, and by facilitating phosphorus co-precipitation with calcium complexes by raising pH or binding with iron and aluminium by altering water column oxygen due to photosynthesis. Both types of aquatic vegetation provide surface of substrate for periphytic algae, an algae that relies mostly on phosphorus dissolved in the water column.

Submerged aquatic vegetation assimilates nutrients from either or both media, water and/or sediments, depending upon the relative availability of nutrients in each media. Emergent aquatic vegetation relies on sediments for a supply of nutrients. Under conditions of high lake water phosphorus concentration, submerged aquatic vegetation would play an important role accumulating in its biomass phosphorus that has been directly removed from the water column. Under conditions of high concentration of phosphorus in the sediments, like those in lakes with a long history of phosphorus loading, emergent aquatic vegetation would play a major role in removing phosphorus from the sediments. This phosphorus would eventually return to the water column as the vegetative decay. Nutrient uptake from the water column or from the sediments would be a key difference between submerged and emergent aquatic vegetation that may determine the importance of these two types of aquatic vegetation in regard to lake management plans directed toward improvement of water quality.

Additionally, external inputs of phosphorus are determinant regulators of the phosphorus status in lakes. These depend on land use conditions on the surrounding watershed. Urban and agricultural land uses are usually the most important contributors of phosphorus to lakes, mainly through runoff, although highly porous soil profile might facilitate underground transport of phosphorus. Atmospheric deposition of phosphorus-laden dust may be an important input of phosphorus to Florida lakes.

Lake water increase in phosphorus concentration can lead to nitrogen limiting conditions that favour the increase in abundance of toxic cyanobacteria with the consequent adverse effects in public health.

2.12 Modeling Review

Information on the dynamics and quantity of phosphorus is critical in the assessment of water quality (Komatsu et al., 2006) since availability of this element is a limiting factor in the production of phytoplankton in most freshwater systems (Schauser et al., 2004; Schindler, 1977; Scinto and Reddy, 2003; Zhou et al., 2001).

Some components within the aquatic ecosystem are naturally placed as key factors to buffer drastic fluctuations in phosphorus levels and consequently in algal productivity. Submerged aquatic vegetation and associated periphyton have been identified as potential controllers of lake water phosphorus concentration and water quality (Bachmann et al., 2002; Bachmann et al., 2004; Dierberg et al., 2002; Scheffer, 2004). Results presented in chapters 4 and 5 show inverse correlations between submerged aquatic vegetation and water phosphorus concentrations even higher than those reported on the literature, and support the theory that submerged aquatic vegetation plays an important role as a regulator of water phosphorus levels and water quality in general and as such, needs to be included in a lake water quality model.

Since early in the 1970's, environmental managers started using models as a tool for analysis and formulation of plans (Komatsu et al., 2006). Today, models are obligatory tools to solve environmental problems. In eutrophication, simple empirical/regression models have been designed to estimate chlorophyll-α based on a known total phosphorus (TP) concentration (Bachmann et al., 2002; Canfield and Hoyer, 1992; Canfield et al., 1984; Canfield, 1983). Regression models, however, do not accurately account for the non-linearity of the flows between the components of ecological systems. In contrast, dynamical eutrophication models better predict the reservoir's reaction to nutrient inputs

(Komatsu et al., 2006), since they incorporate a mechanistic approach that includes time-dependent non-linear closed-loop interactions between determinant components of the system (Schnoor, 1996).

Many dynamical models that simulate phosphorus cycling in aquatic systems have been published. These models represent all scales from lakes (Everett et al., 2007; James and Bierman Jr., 1995; Komatsu et al., 2006; Schauser et al., 2004; Schauser et al., 2006; Spears et al., 2008; Zhou et al., 2001), estuaries (Doering et al., 1995; Kittiwanich et al., 2006; Serpa et al., 2007), wetlands (Lantzke et al., 1999; Richardson et al., 2005; Wang and Mitsh, 2000) to global cycles (Chameides and Perdue, 1997); and different levels of complexity , from three compartments or reservoirs (Harte, 1988; Lahm, 2008) to more than ten (Jorgensen, 2003; Kittiwanich et al., 2006; Tett and Wilson, 1999).

Complicated models with a large number of reservoirs and factors considered might be too specific to be applied to more general circumstances. As it is the case with any of the components of aquatic ecosystems, given the complexity of interactions between factors affecting phosphorus cycling in a shallow lake, it is impossible to know and realistically consider all of them (Carr et al., 1997). It is therefore important to define what the most important factors, parameters, speciation, and mechanisms involved in water phosphorus dynamics are, in order to limit the complexity of the model. Some important parameters of phosphorus dynamics of shallow aquatic lakes assuming conditions of a closed system are given in Table 2.1.

Table 2.1 Imortant parameters measured and considered in phosphorus (P) cycling models of shallow aquatic systems.

Description	Type of aquatic system	Value in terms of phosphorus (P)	Source
P content in peryphiton	Wetlands	0.10 – 0.29 mg g^{-1} dry weight	Scinto and reddy, 2003
P content in submerged aquatic vegetation	Lakes	1.41 mg g^{-1} dry weight	Bachmann et al., 2002
P content in lake pore water	Lakes	≤6 mg L-1	Moore et al., 1991
P content in lake pore water	Lakes	0.1 - > 1 mg L^{-1}	Moore et al., 1998
P content in lake sediment	Lakes	0.54-3.84 mg g^{-1} dry weight	Perkins and Underwood, 2000
P content in wetland sediment	Wetlands	0.28 – 0.37 mg g^{-1}	Wang et al., 2006

For instance, phosphorus modeling would ideally consider separate forms of phosphorus, particulate and dissolved, as only dissolved forms can be assimilated by primary productivity. Kinetic rates for both forms of phosphorus might not be available (Wang and Mitsh, 2000). Additionally, particulate phosphorus represents the most of the phosphorus present in aquatic systems (Table 2.2), therefore it is practical for modeling efforts to use total phosphorus (TP) as an alternative for differentiating the two forms of phosphorus (Wang and Mitsh, 2000). For costly decisions related to water quality management, however, a more detailed approach to the problem may be required (Schnoor, 1996).

Table 2.2 Percent of particulate and dissolved phosphorus content in water column in shallow aquatic systems.

Source	Percentage Particulate	Percentage Dissolved
Meybeck (1982) in Wang and Mitsch (2000)	95	5
Wang and Mitsch (2000)	>75	<25
Schnoor (1996)	70	30
Perkins and Underwood (2000)	84-85	16-15

Whenever a required level of detail cannot be achieved, assumptions and inputs from other similar conditions might be needed and expected to offer helpful hints into phosphorus dynamics of aquatic systems. Efforts then should attempt to synthesize scientific information from literature into the missing elements in the process of model construction. Sections from 2.1 to 2.10 of this chapter describe important factors influencing the metabolism of phosphorus in shallow lakes. In the present section of this literature review, basic processes of phosphorus retention and release in shallow aquatic systems are examined with a goal to gain insight regarding phosphorus cycling and the role of submerged aquatic vegetation in lakes. Such processes will be discussed based on their role in potential basic components or reservoirs of a model that has not been proposed but is theorized in this document.

2.12.1 Interaction between Sediments and Water Column

It is well known that the concentration of phosphorus in aquatic systems with an active inflow and outflow of water depends directly on the phosphorus concentration in the inflow (Richardson, 1996; Schnoor, 1996), and inversely on the hydraulic detention time and the sedimentation rate (Schnoor, 1996). In addition to better conditions for sedimentation, which is the main factor responsible for retention of phosphorus (Schnoor, 1996; Wang and Mitsh, 2000), a large hydraulic detention time, also increases the opportunity of primary productivity for a higher nutrient uptake (Wang and Mitsh, 2000).

The overall cycling of phosphorus in aquatic systems is influenced in great measure by sediments, acting either as a sink or source of phosphorus (Bostrom et al., 1988; Fisher and Reddy, 2001). Some of the phosphorus that settles down to the sediments is recycled back into the lake water column, especially at higher temperature (Genkai-Kato and Carpenter, 2005) and anoxic conditions (Schnoor, 1996). Some recycled phosphorus from sediments to water has been mentioned in modeling literature not specifying the pathway (Schnoor, 1996) or has being modeled through resuspension caused by benthic organisms and storms; or through emergent macrophytes that pump phosphorus from sediments up to water through litter pathways (Wang and Mitsh, 2000). The latter process happens because emergent macrophytes rely exclusively on sediments for nutrients up-take (Graneli and Solander, 1988).

In general, sediments play a key role as potential source of phosphorus into the overlying water (Zhou et al., 2001) and consequently in the recovery of deteriorated aquatic systems (Komatsu et al., 2006). That explains why sediments have been included in many published models of phosphorus cycling in aquatic systems (Carpenter, 2005; Kittiwanich et al., 2006; Komatsu et al., 2006; Schauser et al., 2004; Schauser et al., 2006; Serpa et al., 2007; Spears et al., 2008; Wang and Mitsh, 2000; Zhou et al., 2001).

Concomitant to the importance of sediments in the lake water phosphorus cycle are the concentration of oxygen, iron, and aluminum (Moore et al., 1991; Richardson, 1985; Sundby et al., 1992) in the sediments, as these metals determine the capability of sediments to sequester phosphorus. Therefore, phosphorus release rates from sediments into the water clearly need to be considered in model building.

2.12.2 Interaction between Sediments and Submerged Aquatic Vegetation

Recycling of phosphorus from sediments to the water column represents a critical factor for phosphorus budget calculations (Reddy et al., 1999). Wang and Mitsh (2000) included emergent macrophytes as a pathway for returning phosphorus from sediments back into the water column. The authors calculated that harvesting of macrophytes could remove phosphorus from the system at a rate of 0.32 – 1.6 g m^{-2} $year^{-1}$ depending on the biomass of the macrophytes. Submerged macrophytes have also been modeled as suppressing phosphorus release from sediments to the water column (Genkai-Kato and Carpenter, 2005; Hamilton and Mitchell, 1996; Scheffer, 2004).

Literature indicates that for emergent vegetation, absorption and translocation of sediment phosphorus to plant material has important effect on the phosphorus cycle of lacustrine systems (Barko and Smart, 1980; Barsdate et al., 1974, as quoted in Barsdate et al., 1974; Mcroy et al., 1972). While submerged aquatic vegetation influences phosphorus concentration in lake water through its effect in decreasing sediments resuspension (Bachmann et al., 2002; Bachmann et al., 2004; Scheffer, 2004). Models that intended to determine the concentration of suspended sediments in the lake water column based on the stress induced by waves have resulted in error because submerged aquatic vegetation was not considered (Hamilton and Mitchell, 1996).

Both types of aquatic vegetation have different effects in the phosphorus cycle based, among other reasons, on their source of nutrients. These sources of nutrients are sediments for emergent macrophytes but both, sediments and water, for submerged macrophytes (Graneli and Solander, 1988). In the model formulated by Genkai-Kato and Carpenter (2005), shallow depths favor the capacity of submerged macrophytes to suppress phosphorus recycling in lakes. A shallow depth was assumed in the model by Wang and Mitsh (2000) since it was applied to a wetland.

2.12.3 Interaction between Submerged Aquatic Vegetation and Water Column

Other pathway for submerged aquatic vegetation effect in water phosphorus cycling is by increasing nutrient-up take directly (Graneli and Solander, 1988) and through periphyton and other epiphytes (Dierberg et al., 2002; Scinto and Reddy, 2003). Except for few dynamical models (Everett et al., 2007; Genkai-Kato and Carpenter, 2005) and a regression model (Canfield et al., 1984) submerged aquatic vegetation has not often been taken into account in modeling for water quality. Based on the results seen in chapters 4 and 5, including submerged aquatic vegetation may improve prediction of behaviors in shallow lakes. Models that do not include the submerged aquatic vegetation component but other types of biomass have calculated the rate of phosphorus up-take in proportion to the net growth of biomass (Chameides and Perdue, 1997; Wang and Mitsh, 2000), and this same principle may be applied to potential models that include submerged aquatic vegetation. For simplicity, submerged aquatic vegetation might be considered as a single factor, or alternately, as affected by light, temperature, nutrients, carbon, water velocity and so on (Carr et al., 1997).

2.12.4 Historical Land Use and Trend of Population Growth

So far the components considered in this chapter have been about the minimum required for estimating water phosphorus concentration under conditions of a closed system or assuming known inputs and outputs of phosphorus in an open system. In a more realistic approach, however, inputs of phosphorus to the system are not known and also need to be estimated. These inputs come with the inflow loads mostly from streams and runoff, are dependent on land use (Reddy et al., 1999), and can be predicted with the use of models (Omernik, 1976). The inclusion of external components in phosphorus cycling can be complex and require a large amount of data (Soranno et al., 1996), increasing uncertainties and the possibility of error for the overall model. Research and careful modeling, however, can account for this downside and improve the accuracy of the overall estimations.

Models for use in conservation and water quality management have predicted increase in the inputs of phosphorus based on historical trends of land use and human population (Johnes et al., 1996). Predictions made based upon current increase of urban land cover on Lake Mendota watershed estimated slight increases in annual phosphorus loading but still enough for significant effects on eutrophication. If the entire watershed were urbanized (Soranno et al., 1996), the phosphorus loading would double and the effects in water quality would be severe.

Accurate consideration of the trends of phosphorus input into the system will determine the likelihood of the trends of eutrophication or its reversibility. Lakes with long history of heavy inputs of phosphorus would be more difficult to recover or may not be able to recover (Carpenter et al., 1999). This is because the phosphorus that enters into the system is mostly accumulated in the sediments and the biomass from where it can be recycled to the water column at a faster rate than what can be lost from the system (Carpenter, 2005). This recycling of phosphorus can continue for a long time after external inputs have been decreased (Carpenter, 2005). Failure in reaching a goal level for lake water phosphorus concentration projected with a residence time model has been attributed to a recycling of phosphorus from the sediments (Larsen et al., 1979).

For this reason internal loadings are more important for predictions than external loadings unless sediments are removed from the system (Reddy et al., 1999).

Yet, probably even more important than recycling from the sediments is a slow and constant flux of phosphorus from the watershed soil (Carpenter, 2005). This happens as a consequence of accumulation of phosphorus in the soil and is more common in agricultural areas as a consequence of over-fertilization (Bennett et al., 2001). According to model estimates, when phosphorus inputs are decreased, recovery of eutrophic lakes can be fast if phosphorus recycling from sediments and flux from soils are slow, but this recovery can take hundred of years if these two processes are fast, (Carpenter, 2005).

Florida lakes present especial difficulties for prediction models (Reddy et al., 1999). The flat and low landscape of Florida make it difficult for watershed boundaries to be delineated, hence calculations of non-point inputs are difficult. The sandy limestone foundation of the peninsula cause seepage lakes where no surface inflow or outflow can be easily identified, complicating the calculations of inputs and outputs of phosphorus. Additionally, atmospheric inputs of dry phosphorus need to be included in the calculations since this represents a significant external source according to what was reported for Reddy et al. (1999).

2.12.5 Some Parameters Found in Literature

Hence, submerged aquatic vegetation should be another important reservoir to be considered in modeling of phosphorus cycling in shallow aquatic systems. Table 2.3 shows some rates for important fluxes between water and sediments, periphyton, and phytoplankton compiled from literature. These standard rates can be useful for modeling efforts addressing the metabolism of phosphorus in shallow lakes. Notice that no rates are included for submerged aquatic vegetation, which reflect the insufficient inclusion of this component in modeling studies.

Table 2.3 Standard rates in phosphorus cycling of shallow aquatic systems.

Description	Type of aquatic system	Value in g P m^{-2} $year^{-1}$	M:measured C:calculated	Source
Phytoplankton and periphyton up-take	Wetlands	0.12 – 0.22	C	Wang and Mitsch, 2000
Sedimentation rate	Wetlands	0.62 - 1.08	C	Wang and Mitsch, 2000
Sedimentation rate	Lakes	3.17	M	Schauser et al., 2004
Flux by leaching and decomposition of bottom	Wetlands	0.20 – 0.66	C	Wang and Mitsch, 2000
Flux from sediments by resuspension	Wetlands	0.27 – 2.47	C	Wang and Mitsch, 2000
Flux from sediments by resuspension	Lakes	1.46 – 25.55	M	Sondergaard et al., 2004
Flux from sediments	Lakes	2.11	C	Schauser et al., 2004
Flux from sediments	Wetlands	2.37	M	Fisher and Reddy, 2001
Flux from sediments	Lakes	0.99	M	Moore et al., 1991
Flux from sediments	Lakes	0.36	M	Moore et al., 1998
Flux from sediments	Wetlands	3.41	M	Lai and Lam, 2008
Flux from sediments	Lakes	0.38	M	Ogburn (1984) in Reddy et al. (1999)
Atmospheric deposition	lakes	0.044 - 0.058	M	Ogburn (1984) in Reddy et al. (1999)
Sediments accumulation rates	Wetlands	6 to 29 mm $year^{-1}$	C	Wang and Mitsch, 2000

2.13 Discussion

Considering the inverse relationship observed in Chapters 4 and 5 between the prevalence of submerged aquatic vegetation and the concentration of TP in lake water, it seems more likely that the process modelled by Genkai-Kato and Carpenter (2005) in which submerged aquatic vegetation function as a suppressor for recycling of phosphorus from sediment back to overlying water is more prevalent than the possible pumping effect from sediments to water modelled by Wang and Mitsh (2000). This also agrees with the water quality model of James and Bierman (1995), which calculated that the amount of phosphorus removed by sedimentation from water solution in Lake Okeechobee exceeded the net flux of phosphorus from sediments into the overlying water. Table 2.3, however, shows similar rates reported for phosphorus sedimentation and phosphorus flux from sediments to water.

The inverse association between lake water TP concentration and depth also is consistent with the effect of temperature modeled by Genkai-Kato and Carpenter (2005). According to this the higher temperature associated to shallow depth favour the chemical reactions leading to release of phosphorus from the sediments into the overlying water. This would, subsequently increase the concentration of phosphorus in solution. Unfortunately limited information was found about models including the processes of phosphorus uptake from sediments and water to submerged aquatic vegetation that could be used to interpret the associations found. It is, therefore, difficult to test for a relationship between submerged aquatic vegetation and lake water phosphorus concentration due to the lack of observational or calculated data for up-take rates from sediments and water to submerged aquatic vegetation as well as release rates from submerged aquatic vegetation to water column. In addition, another difficulty is the lack of literary sources that include all the rates for the three proposed reservoirs in the same system.

A simplistic model suggested here to estimate the status of TP concentration in lake water of urban lakes based on the status of submerged aquatic vegetation should be composed by three reservoirs: TP contained in sediments, TP contained in water column, and TP contained in submerged aquatic vegetation. This model would test assumptions made in literature and results obtained in Chapter 4 and 5 about association and possible causation between submerged aquatic vegetation and concentration of phosphorus in lake water. Since phosphorus does not have a stable gaseous form of significance (Chameides and Perdue, 1997; Lahm, 2008; Schlesinger, 1991), this model would not include any reservoir or input for phosphorus in gaseous form nor consider any output from the system into the atmosphere. It is important to make the observation, however, that in lakes were other inputs of phosohorus are not large enough, then phosphorus atmospheric deposition may be considered significant.

Since lakes are open systems with a strong dependence from external inputs and outputs, these need to be included as flows in and out of the system. Among the possible external sources of phosphorus in the lakes with lower levels of total water phosphorus concentration may be runoff from urban areas.

Although phosphorus is known for not been soluble and readily transported with water, the fact that the underground soil profile surrounding these lakes is sandy karst (van Beynen et al., 2007) suggests the possibility that leaching of some forms of soluble phosphorus through the underground profile may be another external source of phosphorus to the lakes. Note that some of the lakes with higher concentrations of total phosphorus in the water column are located toward the eastern part of the Tampa Bay watershed, which also is the part of the watershed that is contained within the area of the Southern Bone Valley (Brown, 2005). Landscape alterations, runoff, groundwater flow of soluble phosphorus, or discharges from mining activities may explain their high phosphorus levels. Naturally occurring phosphorus, suburban and agricultural land runoff may be the main sources of phosphorus for Lake Thonotosassa, which is the lake with the highest level in water column phosphorus concentration among those examined here.

The slightly decreasing trend in total phosphorus concentration in the eutrophic and hypereutrophic lakes may be indicative that loading of phosphorus to these lakes was not excessive to the point that eutrophication could not be reversed. Otherwise due to excessive phosphorus loading, the labels would have produced such accumulation of phosphorus in the system that recycling from sediments and flux from soils were faster than phosphorus loss from the system (Carpenter, 2005; Larsen et al., 1979). Special consideration must be given to the amount of phosphorus loading over time since it would determine in great measure the possibility of recovering (Bennett et al., 2001; Carpenter,

2005) and also to the land use in the watershed since it would determine the loading (Johnes et al., 1996; Soranno et al., 1996).

A good alternative tool to construct this model is Stella ® software, an iconographic computational platform helpful to visualise and analyze equations and processes (Costanza and Voinov, 2001), that has been commonly applied for biogeochemical modeling in aquatic systems (Carpenter, 2005; Jorgensen, 2003; Jorgensen et al., 2002; Krivtsov et al., 2000; Tett and Wilson, 1999).

CHAPTER 3.

TEMPORAL TRENDS IN LAKE WATER CONCENTRATION OF TOTAL PHOSPHORUS, RATIO OF TOTAL NITROGEN TO TOTAL PHOSPHORUS, AND CHLOROPHYLL-α FOR LAKES OF DIFFERENT EUTROPHICATION STATUS IN TAMPA BAY WATERSHED: 1990-2007

3.1 Introduction

This chapter provides an analysis of the nutrient and chlorophyll-α concentration trend behavior in a group of lakes of the Tampa Bay watershed, which may prove to be useful indicators of overall watershed trends. These findings will present evidence to further corroborate or contradict the theory of cultural eutrophication associated with watershed development (Smith et al., 2003). On a broader scale, the knowledge of a trend, if one exists, would assist surface water managers and community stakeholders in their efforts to create sustainable development in the Tampa Bay watershed.

The 5,7000 ha Tampa Bay, watershed lies within the Counties of Hillsborough, Pinellas, and Manatee and extends to parts of Sarasota, Pasco, and Polk Counties. Between 2001 and 2030, the population within the Tampa Bay watershed is expected to increase by an estimated two million people and 940,000 jobs are projected to be created during the same time period (Tampa

Bay Regional Planning Council, 2007). Concern over the impact of this region's fast growth on the cultural eutrophication of natural lakes located in the Tampa Bay watershed relates to not only the bodies of water within the watershed, but more broadly, to the possible effects of ecosystem flux on the receiving bay.

3.2 Objectives and Hypotheses

Population growth and watershed development have often been associated with increased nutrient concentration in naturally occurring bodies of water (Rast and Thorton, 1996; Smith et al., 2003). Terrell et al. (2000) however, did not find a trend of increasing water concentration of total phosphorus (TP), total nitrogen (TN), and chlorophyll-α across 127 Florida lakes during a time period of growing population between 1967 and 1997. Reports 305 (b) from the Florida Department of Environmental Protection (FDEP), however, have reported over time a decreasing percentage of lakes with a stable water quality. The percentage of lakes with stable levels of eutrophication estate parameters (TP, TN, and chlorophyll-α) were 71, 58, and 41% for the years 1996, 2000, and 2006, respectively. The percentage of lakes reporting a degrading trend in eutrophication parameters (increases in TP, TN, and chlorophyll-α) has increased: 9, 22, and 33% for the years 1996, 2000, and 2006, respectively. The percentage of lakes showing an improving trend in eutrophication parameters (decrease in TP, TN, and chlorophyll-α) has also increased, but to a lesser extent: 20, 20, and 26% for the years 1996, 2000, and 2006, respectively. The number of lakes assessed by the FDEP for the above mentioned reports was 627, 541, and 358, respectively (FDEP, 1996; 2000; 2006).

With those different findings as a background, the following objective and hypothesis are addressed in this chapter for lakes in the Tampa Bay watershed:

- Objective: To determine if there is a change in lake water concentration of eutrophication-related parameters netween 1990 and 2007.
    - *Null Hypothesis (H_o)*: Lake water concentration of total phosphorus (TP), ratio of total nitrogen to total phosphorus (TN:TP), and chlorophyll-α did not change between 1990 to 2007.
    - *Alternate Hypothesis (H_a)*: Lake water concentration of total phosphorus (TP), ratio of total nitrogen to total phosphorus (TN:TP), and chlorophyll-α did change between 1990 to 2007.

3.3 Methods

3.3.1 Data Gathering and Sampling Methods

To examine the temporal variability of water chemistry, existing record data for lake water concentrations of TN, TP, and chlorophyll-α on 16 lakes located in the Tampa Bay watershed were compiled. Out of 649 lakes located in Hillsborough, Pinellas, Manatee, and Polk Counties, for which information is provided by the Water Atlas (2008), web site of the Florida Center for Community Design and Research at the University of South Florida, only 16 lakes met both inclusion criteria: containment within the Tampa Bay watershed and data availability for at least 75% of the 18 year study period (1990 to 2007). The locations of the lakes chosen for this analysis are represented graphically in Figure 3.1. For more information about the lakes, refer to Appendix A.

All of the data analyzed in this study were obtained from the Water Atlas (2008). Most of the water samples were collected by citizen volunteers sponsored by the water quality monitoring program, LAKEWATCH, and these samples were analyzed in the laboratory of the Department of Fisheries and Aquatic Sciences at the University of Florida. The methods used to collect the data are described in Brown et al. (1998). TP was determined by oxygenating phosphorus with potassium persulfate (Menzel and Corwin, 1965) and measuring the liberated phosphorus with the colorimetric technique of Murphy and Riley (1962) as cited in Menzel and Corwin (1965). TN was determined by a persulfate oxidation technique (D'Elia et al., 1977) followed with nitrate-nitrogen determination by ultraviolet derivative spectroscopy of second order (Bachmann and Canfield, 1996; Simal et al., 1985; Wollin, 1987). Determination of chlorophyll-α was done by extracting the pigment with ethanol (Sartory and Grobbelar, 1984) and then measuring it with spectrophotometry following the Standard Method (SM) 10200 H method (APHA, 1989; 1998).

Water samples from Lake Thonotosassa were collected and analyzed by the Environmental Protection Commission of Hillsborough County using a combination of EPA and APHA Standard Methods. TP was determined by EPA 365.4; and TN was the sum of Total Kjeldahl Nitrogen (TKN) and nitrate/nitrite nitrogen, where Total Kjeldahl Nitrogen (TKN) was determined by EPA 351.2 while SM 4500 NO3 F (APHA, 1989; 1998) was used for nitrate/nitrite nitrogen. Chlorophyll-α was determined by SM 10200 H (APHA, 1989; 1998).

Samples from Behula, Bonnet, and Hunter Lakes were collected by the City of Lakeland Division of Lakes and Stormwater and analyzed by the City of Lakeland Wastewater Laboratory. The method used for TP analysis was EPA 365.4; and for TN analysis the methods were EPA 353.2 for nitrate and PAI DK03 (method approved by the Environmental Protection Agency) for Total Kjeldahl Nitrogen (TKN). For chlorophyll- α, a modification of Standard Methods 10200ha (APHA, 1998) was used. Data for Ward Lake came from STORET, a computerized environmental database of United States Geological Survey, which was also was made available by Water Atlas (2008).

3.3.2 Statistical Analysis

The total data set of values for TP, TN:TP, and chlorophyll-α from each lake were used over the 18-year time period.

Values for TN were analyzed relative to TP. The importance of lake water TN concentration as a limiting factor depends on its abundance relative to that of lake water TP. Therefore, a TN:TP ratio is more meaningful in regard to the potential for eutrophication and likelihood of algal abundance. According to the nutrient limitation criteria based on Brezonick (1984), ratios greater than 30 correspond to phosphorus-limited lakes while ratios less than 10 correspond to nitrogen-limited lakes. Nutrient limitation in lakes is balanced (both nutrients are limiting) if TN:TP ratio is between 10 and 30.

The 16 lakes studied to assess the overall trend of TP, TN:TP, and chlorophyll-α were sorted in two groups depending on the overall mean of lake water TP concentration for each lake during the study period. The grouping criteria followed the Trophic State Classification System of Forsberg and Ryding (1980) (Fig 3.1). This system suggests uses for surface waters based on the water nutrient concentration and not necessarily implying an adverse effect; however, as discussed in Chapter 1, elevated nutrient levels have been associated with adverse human and environmental health consequences.

The Florida Department of Environmental Protection Agency (FDEP) among other parameters considers surface waters as good when TP concentrations are between 0 to 64 mg L^{-1}, fair if TP concentrations range from 65 to 112 mg L^{-1}, and poor for TP concentrations between 112 and 567 mg L^{-1}.

The first group of lakes was comprised of 10 lakes exhibiting both oligotrophic and mesotrophic conditions (less than 25 µg TP L^{-1}), while the second group comprised 6 hypereutrophic lakes (more than 100 µg TP L^{-}1). The first group were located in the northwestern corner of Hillsborough County, an area that drains into Old Tampa Bay according to Lewis and Estevez (1988) (Figure 1.1). Lakes in the second group were found in more distant parts of the Tampa Bay watershed, for example, in Hillsborough, Pinellas, Polk, and Manatee Counties. Figure 3.2 shows all the lakes included in this analysis.

Since the objective of this research was to characterize patterns of response and change in a variable (TP, TN:TP, or chlorophyll-α) over time, this analysis falls under the definition of longitudinal research (Ware, 1985). As it is usual in most longitudinal studies, this one violates some assumptions required for standard regression analysis. Measurements taken over time were not independent, residuals were not normally distributed, and data were not collected at a constant set of time points and many values are missing: characteristics that make standard regression analysis inapplicable (Lin and Ying, 2003; Ware, 1985; Zeguer et al., 1988). For each parameter in each trophic group of lakes, however, least squares linear regressions were plotted as a preliminary visual assessment of the overall 18-year trends.

Points that were 5 times greater or smaller than the mean and that individually influenced the mean value of the eutrophication parameter were removed as high influence points. The total numbers of points removed by this method were 7 out of 2101 for TP and 2 out of 1707 for chlorophyll-α

A randomized complete block design (Ott, 1993) was used for this analysis. Period was the treatment. To account for lake effect, the data was blocked by lake. The response was lake water concentration of TP, TN:TP ratio, and chlorophyll-α, and the experimental unit was lake water in Tampa Bay watershed. It was an assumption that samples were collected at random (randomly distributed in time). The null hypotheses for each of the sets are given below:

1. The means of the periods are equal.

2. The means of the lakes are equal.
3. There is no interaction between periods and lakes.

Lakes were first separated into low and high eutrophication groups based on TP concentration, as previously discussed.

A two-way analysis of variance (ANOVA) was conducted with SYSTAT® to test the research hypotheses. The first factor was period. The data set for each trophic group was divided in two periods: Period 1 extended from 1990 until 1998 and Period 2 from 1999 until 2007. The second factor was lake (lake drainage basin).

ANOVA is a tool typically applied to controlled experiments rather than to "natural" experiments; however, ANOVA has been used in natural studies to test for changes in TP (Smith and McCormick, 2001) and mercury (Babiarz et al., 1998) concentration over time or space.

T-test considering unequal variances was applied to detect differences between periods within each lake. A significant level of $\alpha = 0.05$ was used to perform all the statistical tests.

Annual averages of TP, TN:TP ratio, and chlorophyll-α concentrations across all lakes were plotted against the corresponding year to easily visualize cycling in the data. Regression coefficients that represent the population of data rather than the original specific data are called “population-averaged” as described by Zeguer et al. (1988). A

Annual estimates of population data for the Tampa Bay Metropolitan Area and Hillsborough County were tabulated by year to appreciate the annual growth in population.

Figure 3.1 Trophic State Classification System (Forsberg and Ryding, 1980). Concentrations of constituents for each trophic state, and typical uses of waterbodies. http://www.hillsborough.wateratlas.usf.edu

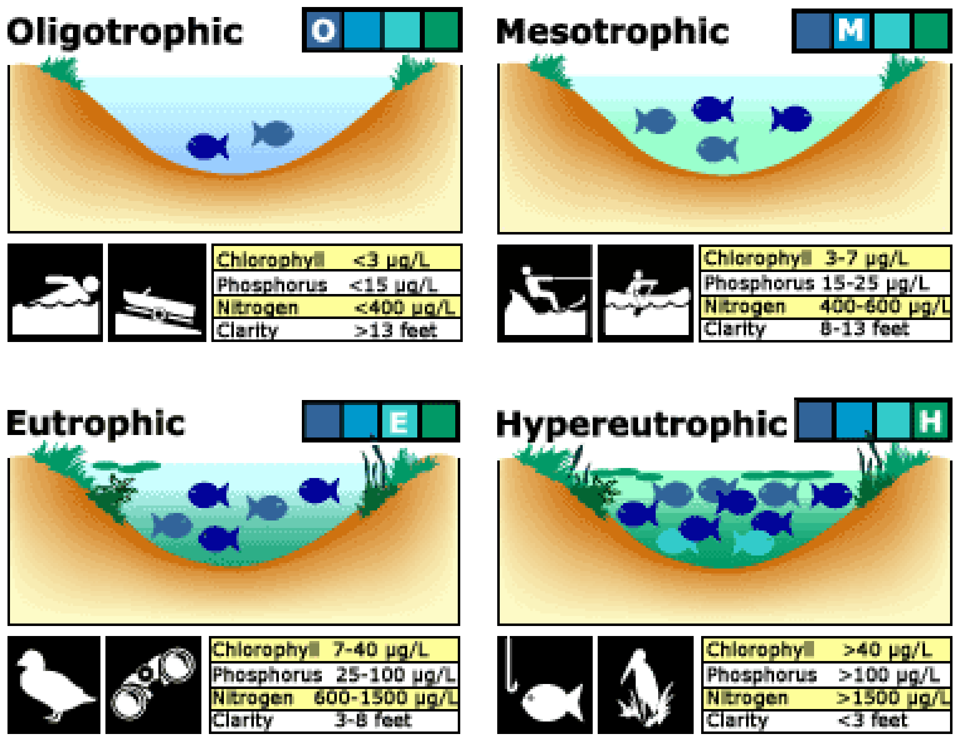
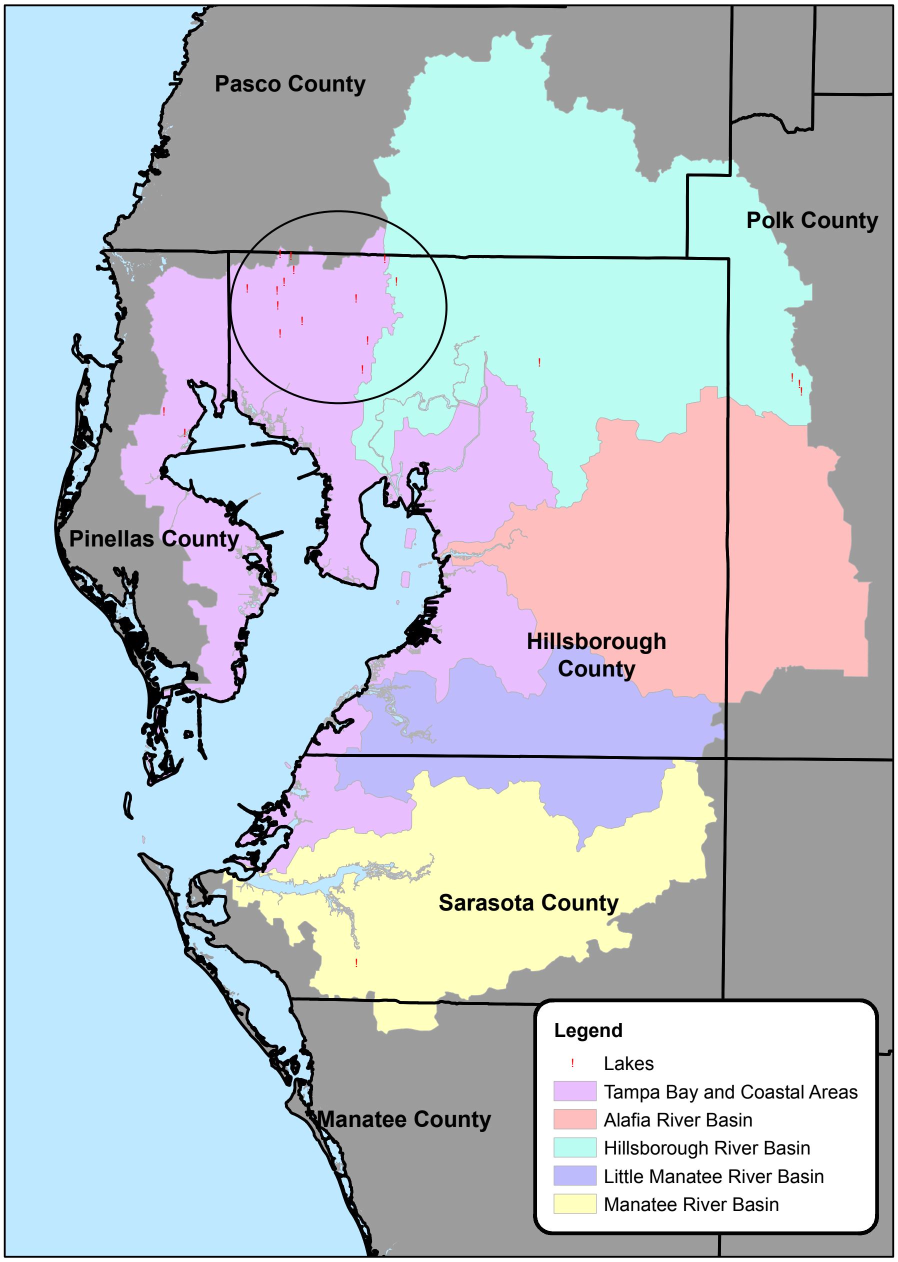


Figure 3.2 Lakes analyzed for trends in trophic state variables during the time period 1990 to 2007. Lakes are shown in red. Lakes within the circle are oligotrophic and mesotrophic.



Map Created by Pete Reehling – University of South Florida

3.4 Results and Discussion

Results from the two-way ANOVA showed a significant difference between means of lakes in all the groups for all the trophic state variables considered ($p<0.0001$). Interactions were found between periods and lakes in all the groups for all the trophic state variables considered ($p<0.003$). Over the 18-year study period, overall average annual TP concentrations for the ten oligotrophic and mesotrophic lakes ranged from 10.47 to 19.09 µg L^{-1}. Individual lake values ranged from 2.00 to 59.00 µg L^{-1} (Table 3.1). Many values in the upper annual range exceeded the limit of 25 µg L^{-1} of TP used as trophic class separation. However, the means for each one of these lakes remained within the 25 µg L^{-1} limit according to the criteria. The overall annual average of TP concentration increased from 11.71 µg L^{-1} in 1990 to 15.00 µg L^{-1} in 2007 (Table 3.1). The best-fit line for a plot of original data suggested a weak correlation (r = 0.3) and a positive slope (Figure 3.3). A significant difference was found with two-way ANOVA between the means of the first and second periods ($p< 0.0001$). The t-tests performed on TP concentration for each lake between Periods 1 and 2 showed a statistically significant difference ($p \leq 0.05$) in all the lakes of this group except for two. The fact that each lake did not increase at the same rate between period 1 and 2 was evidence of significant interaction (Figure 3.4).

The average annual TP concentration for the group of hypereutrophic lakes ranged from 456.47 µg L^{-1} in 1997 to 158.29 µg L^{-1} in 2007 (Table 3.2). The individual values ranged from 10 to 2300 µg L^{-1}. The best-fit line for a plot of original data showed a low coefficient of determination and a slightly decreasing slope (Figure 3.5). Two-way ANOVA showed significant differences between the two periods (p = 0.03). Such results were the opposite of the increasing trend suggested by the group of oligotrophic and mesotrophic lakes for the same parameter and time period. The t-tests performed for each lake of this group for TP concentration found no statistical significant difference between the first and second period in two lakes ($p > 0.05$), and a significant increase in one lake ($p \leq 0.05$). Since the effects of period on lake water TP concentration did not remain the same for different lakes then there was interaction (Figure 3.6).

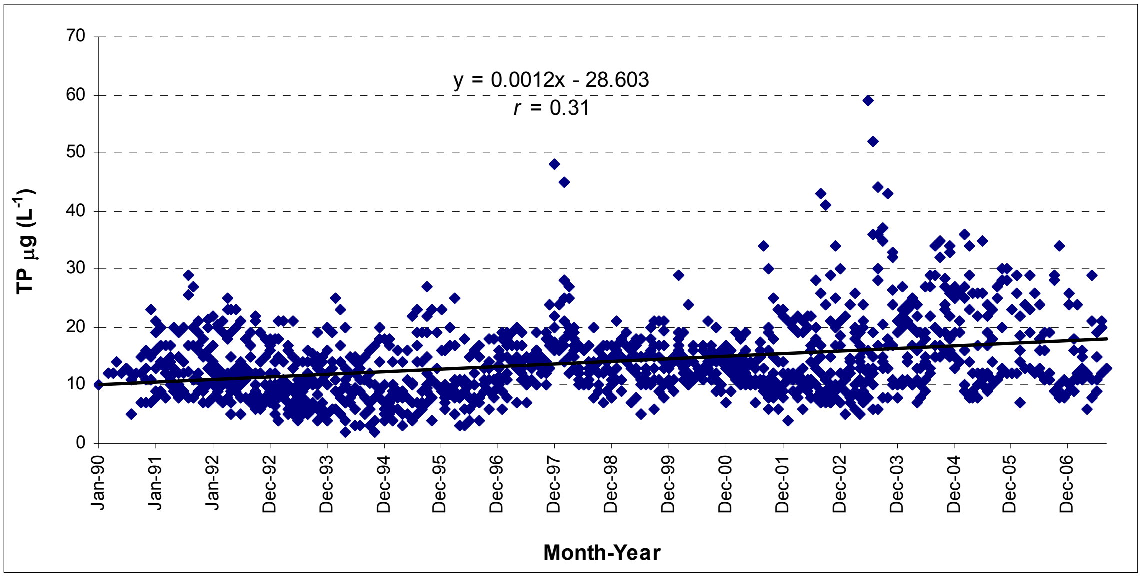
For the group of hypereutrophic lakes, reasons for changes in lake water TP concentration may be related to implementation of lake management plans, lake restoration, and storm water treatment projects (City of Lakeland, 2001; Southwest Florida Water Management District, 2003). The plan of water improvement for Lake Thonotosassa, for instance, had a reduction in phosphorus loadings from point sources since the Sno-Man seafood processing plant closed in 1992 and the Plant City Wastewater Tretament Plant ceased discharges in 1997. In regards to non-point sources, a 21-ha marsh was constructed to intercept water from Baker Creek before entering the lake. A strategy to control exotic submerged and floating plants, such as water hyacinth (*Eichhornia crassipes*) and water lettuce *(Pistia stratiotes,* has been implemented to allow penetration of light for submerged aquatic vegetation species. Such programs for lake improvement may have given priority to those lakes with the worse water quality in order to maximize the efficiency of limited funding. The reduction in lake water TP concentration observed in these highly eutrophied lakes demonstrate the efficacy of lake management plans.

Differences in the amplitude of TP concentrations between oligotrophic and mesotrophic lakes and hypereutrophic lakes were seen. Standard deviation (SD) in TP concentrations for oligotrophic and mesotrophic lakes ranged from 3.77 to 10.09 µg L^{-1} (Table 3.1) and coefficient of variation (CV) from 27% to 60%, while for hypereutrophic lakes SD went from 72.35 to 482.01 (Table 3.2) and CV from 45% to 105%.

Table 3.1 Summary statistics for annual TP concentration in µg L^{-1} collected by LAKEWATCH from 10 oligotrophic and mesotrophic lakes in two counties from 1990 through 2007

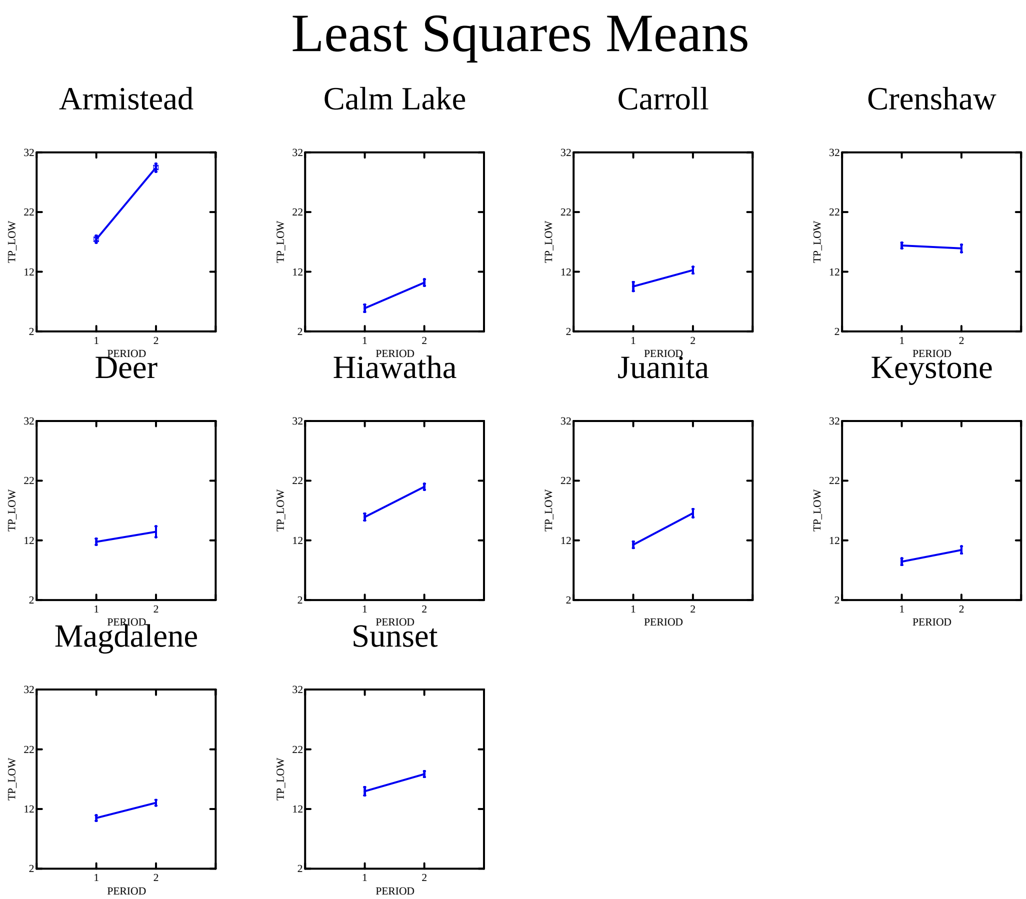
Year	Annual Average (µg L^{-1})	SD (µg L^{-1})	Minimum (µg L^{-1})	Maximum (µg L^{-1})	n (Lakes)	n (Samples)
1990	11.71	3.71	5.00	23.00	7	24
1991	14.14	4.91	7.00	29.00	8	68
1992	12.66	5.02	5.00	25.00	10	88
1993	10.51	4.24	4.00	21.00	9	92
1994	10.47	5.05	2.00	25.00	8	73
1995	10.64	5.55	3.00	27.00	9	72
1996	10.63	4.48	3.00	25.00	9	69
1997	14.16	3.39	6.00	24.00	10	74
1998	16.17	7.02	7.00	48.00	9	68
1999	13.96	3.77	5.00	21.00	8	70
2000	13.97	3.43	9.00	29.00	9	73
2001	13.52	4.82	6.00	34.00	9	78
2002	14.12	7.28	4.00	43.00	10	91
2003	17.40	10.59	5.00	59.00	10	86
2004	19.09	6.90	7.00	35.00	10	74
2005	18.93	7.83	8.00	36.00	9	59
2006	17.30	7.29	7.00	34.00	8	41
2007	15.00	5.90	6.00	29.00	6	32

Figure 3.3   Plot of TP concentration for 10 oligotrophic and mesotrophic lakes.



As mentioned earlier, analysis of TN is expressed here as a ratio to TP since phosphorus is the nutrient limiting natural productivity in most of these lakes and because cyanobacteria abundance is a concern under conditions of high lake water TP concentration relative to lake water TN concentration. The overall group of 10 lakes, oligotrophic and mesotrophic, presented annual mean TN:TP ratio slightly decreasing from 55.35 in 1990 to 44.14 in 2007 (Table 3.3) and ranged from 35.82 to 55.35 (Table 3.3). Individual values ranged from 12.41 to 132.00, the annual SD ranged from 10.04 to 18.14 and CV from 26% to 33%. A plot of TN:TP ratios for individual values shows a decreasing line with a weak correlation (r = 0.26, Figure 3.7) for this group of lakes.

Figure 3.4 Change in TP concentration in 10 oligotrophic and mesotrophic lakes between Periods 1 and 2.



Two-way ANOVA confirmed this trend by finding a significant difference between the first and second periods, being lower in the second ($p < 0.001$). Even though all the 10 lakes showed a decrease in TN:TP ratio from the first to the second period, when tested individually by t-test, this change was not statistically significant ($p > 0.05$) for four lakes. The effect of period on TN:TP ratio was not consistent between lakes in this group (Figure 3.8), indicating interaction between period and lake. The overall decreasing trend in TN:TP ratio might indicate that TP concentration has increased at a faster pace than TN concentration. As the plot of ratios shows, all lakes in this group are included within the phosphorus limitation criteria of Brezonick (1984), that is, all ratios ≥ 30 (Figure 3.7). However, if the downward trend continues, it would result in a gradual tendency from being phosphorus-limited toward being phosphorus- and nitrogen-limited.

Table 3.2 Summary statistics for annual TP concentration in μg L^{-1} collected by LAKEWATCH from 6 hypereutrophic lakes in three counties from 1990 through 2007

Year	Annual Average (μg L^{-1})	SD (μg L^{-1})	Minimum (μg L^{-1})	Maximum (μg L^{-1})	n (Lakes)	n (Samples)
1990	456.47	482.01	10.00	2300.00	6	53
1991	311.98	251.78	57.00	890.00	6	53
1992	463.32	449.27	67.00	2290.00	5	28
1993	247.03	155.21	51.00	680.00	6	37
1994	254.53	157.27	64.00	1010.00	6	40
1995	287.53	121.68	50.00	608.00	6	60
1996	263.10	121.50	50.00	504.00	6	61
1997	298.72	251.06	20.00	1800.00	6	58
1998	375.88	328.34	60.00	2000.00	6	57
1999	306.90	277.60	30.00	1940.00	6	58
2000	284.15	199.55	90.00	1050.00	6	55
2001	260.39	109.76	80.00	508.00	6	57
2002	228.64	90.34	68.00	483.00	6	42
2003	300.49	227.86	39.00	1620.00	6	53
2004	304.04	201.79	80.00	1030.00	6	52
2005	250.98	144.73	54.00	727.00	6	49
2006	191.78	120.64	47.00	638.00	6	36
2007	158.29	72.35	51.00	296.00	5	17

Figure 3.5 Plot of TP concentration for 6 hypereutrophic lakes.

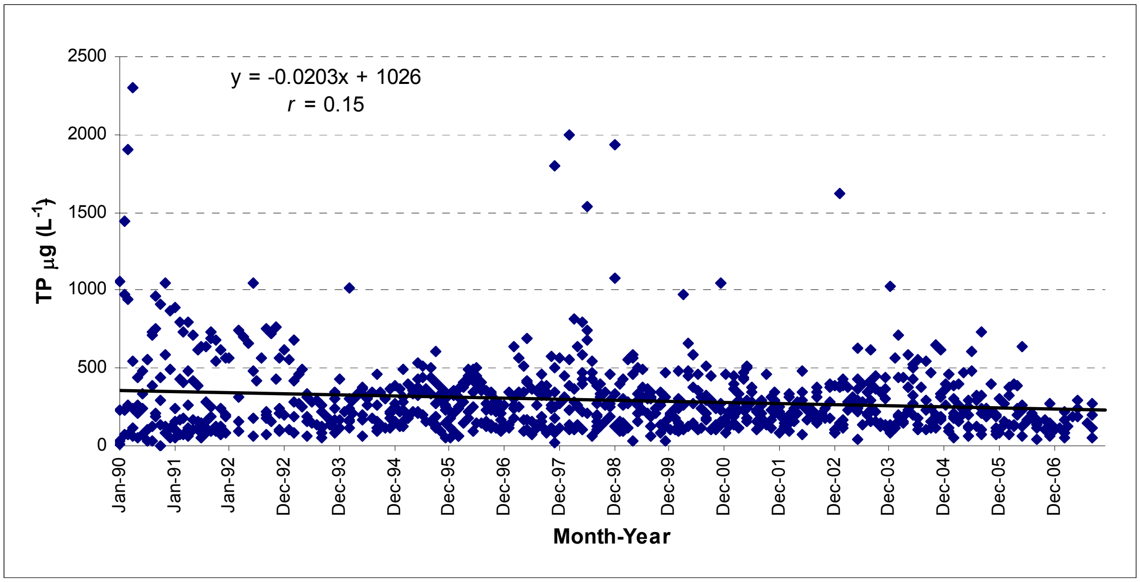
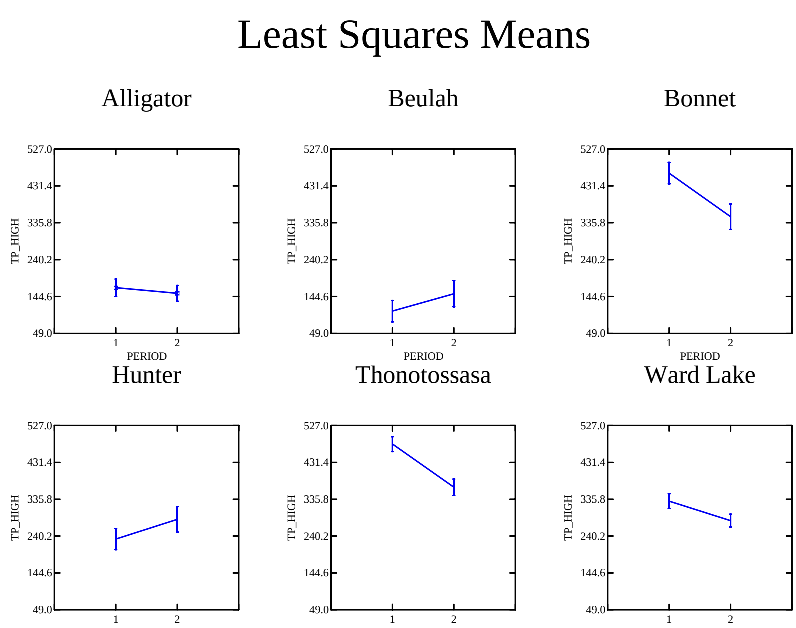


Figure 3.6 Change in TP concentration in 6 hypereutrophic lakes between Periods 1 and 2.



Low TN:TP ratios could lead to increase of cyanobacterial water concentration (Hecky and Kilham, 1988; Levich, 1996; Levich and Bulgakov, 1992). This group of organisms produces potent toxins that have been associated to harmful algal blooms (USEPA, 1997) and possibly an increased risk of primary hepatocellular carcinoma (Fleming et al., 2002). As a consequence, control of nitrogen would be increasingly more important in the formulation of environmental management plans for these lakes.

Only 5 out of the initial 6 hypereutrophic lakes were examined for TN:TP ratio because no TN data were available for one of them. Data for this ratio plotted for this group of lakes ($r = 0.15$, Figure 3.9), show a slightly decreasing slope within the nitrogen limitation zone. This regression line suggests a significant ($p < 0.01$) weak negative tendency over time which was confirmed by two-way ANOVA ($p < 0.001$). Individual t-tests, however, detected a statistically significant change of increase ($p \leq 0.05$) in only two of the lakes. There was interaction for this group too because each lake did not increase in a consistent way between Period 1 and Period 2 (Figure 3.10).

For hypereutrophic lakes, TN:TP ratios ranged from 0.41 to 53.92 (Table 3.4) and overall annual average ratios ranged from 4.66 to 15.89 (Table 3.4). These ratios are at the interface between balanced (both nitrogen- and phosphorus -imited) and nitrogen-limited (TN:TP = 10) primary productivity. The lower TN:TP ratio found in hypereutrophic lakes as compared to oligotrophic and mesotrophic lakes suggested different criteria for lake management options between these two groups of lakes. The SD of TN:TP ratios ranged from 2.7 to 10.74 and CV from 47% to 101%.

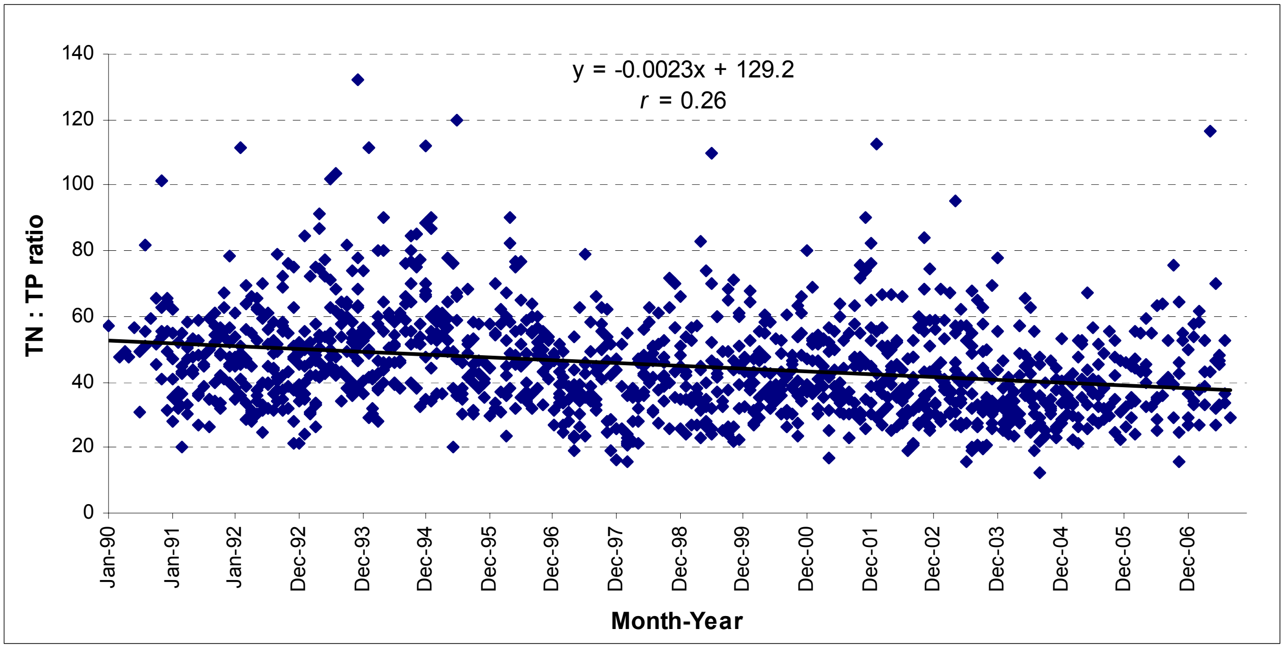
Table 3.3 Summary statistics for the annual ratio of TN:TP ratio collected by LAKEWATCH from 10 oligotrophic and mesotrophic lakes in two counties from 1990 through 2007

Year	Annual Average	SD	Minimum	Maximum	n (Lakes)	n (Samples)
1990	55.35	14.69	30.77	101.43	6	24
1991	45.82	10.90	20.00	78.57	8	68
1992	47.22	14.87	21.43	111.25	10	88
1993	53.53	18.14	21.43	132.00	9	92
1994	55.13	16.36	28.24	111.67	8	73
1995	52.90	17.92	20.00	120.00	9	72
1996	49.60	13.49	23.60	90.00	9	69
1997	40.43	11.69	19.00	79.00	10	74
1998	40.10	12.84	15.56	71.43	9	68
1999	43.13	16.40	21.90	110.00	8	70
2000	42.78	10.70	23.50	67.78	9	73
2001	44.52	13.67	16.84	90.00	9	78
2002	43.18	15.73	19.23	112.50	10	91
2003	40.29	14.36	15.93	95.00	10	86
2004	35.82	11.47	12.41	78.00	10	75
2005	37.96	10.04	21.18	67.00	9	59
2006	41.99	12.89	15.42	75.56	8	42
2007	44.14	17.57	26.67	116.67	6	32

The temporal distribution of data points plotted for chlorophyll-α concentrations from oligotrophic and mesotrophic lakes indicated an increasing trend (Figure 3.11, $r = 0.23$). These results were consistent with the two-way ANOVA ($p < 0.001$). The t-tests of chlorophyll-α concentrations for each lake revealed that concentrations in 3 of the 8 lakes showed no significant change between Period 1 and Period 2 ($p > 0.05$) and confirmed an interaction between periods and lakes (Figure 3.12).

The study period in this group of lakes started with an overall annual average of 6.52 μg L^{-1} in 1990 and ended with an average of 5.27 μg L^{-1} in 2007 (Table 3.5). Averages values ranged from 4.38 to 13.80 μg L^{-1} and lake values ranged from 1.00 to 48.00 μg L^{-1} (Table 3.5) with SD ranging from 3.17 in 2007 to 11.05 in 2003 (Table 3.5) and CV from 60% to 101% for the same years.

Figure 3.7 Plot of TN:TP ratio for 10 oligotrophic and mesotrophic lakes.



For hypereutrophic lakes, no overall trend for lake water chlorophyll-α concentrations was apparent from the visual analysis (r = 0.04, Figure 3.13). Preliminary results from regression analysis were confirmed by two-way ANOVA, which found no significant difference between the first and second periods in the concentration of chlorophyll-α in the lake water of these hypereutrophic lakes (p = 0.717). The t-tests conducted on individual lakes showed that 3 lakes had a statistically significant increase ($p \leq 0,5$) in chlorophyll-α concentrations one lake showed a significant decrease, and one more showed no significant change. The fact that each lake did not behave in the same way between periods 1 and 2 was evidence of interaction (Figure 3.14). Lake chlorophyll-α concentrations had a wide range from 1.00 to 643.50 μg L^{-1}, and a much higher overall SD (Table 3.6) from 44.05 to 150.53 and CV from 69 to 128% as compared with the less eutrophied lakes. The overall annual averages ranged from 47.59 in 1990 to 116.70 μg L^{-1} in 2007 (Table 3.6).

Figure 3.8 Change in the TN:TP ratio in 10 oligotrophic and mesotrophic lakes between Periods 1 and 2.

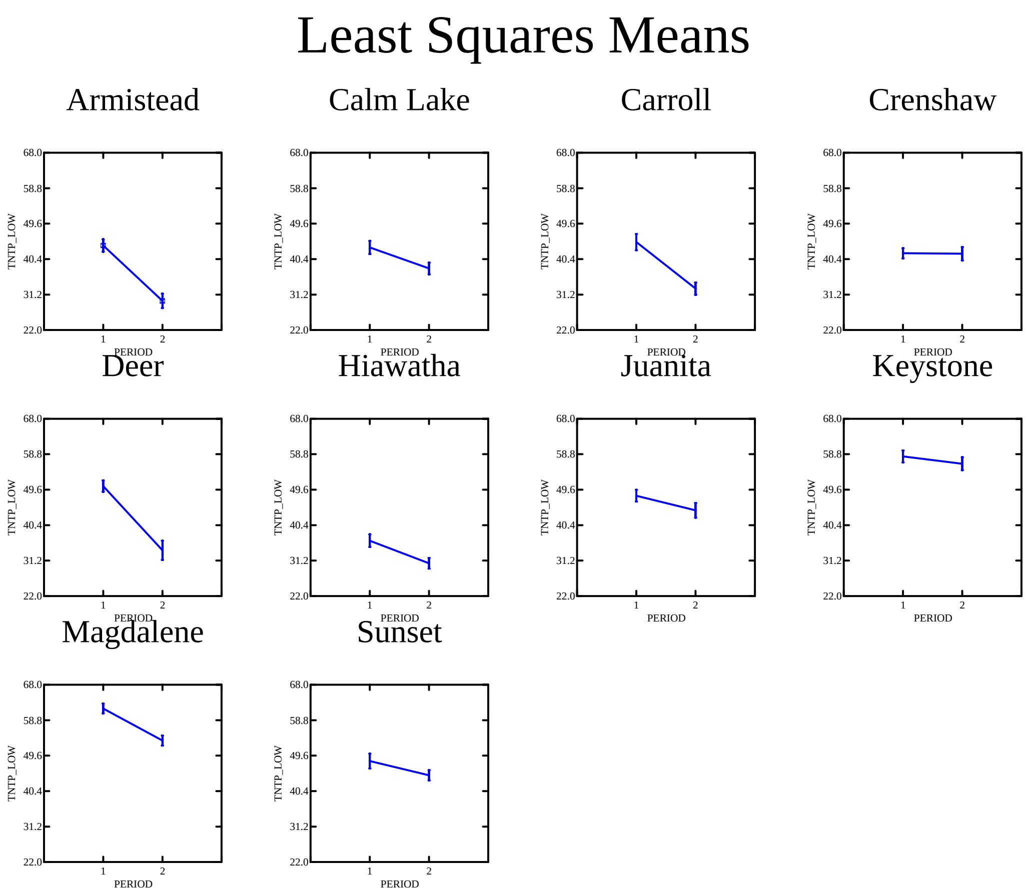


Table 3.4 Summary statistics for annual TN:TP ratio collected by LAKEWATCH from 5 hypereutrophic lakes in three counties from 1990 through 2007

Year	Annual Average	SD	Minimum	Maximum	n (Lakes)	n (Samples)
1990	13.15	9.20	0.84	37.14	5	48
1991	12.83	9.33	1.96	38.43	5	53
1992	10.54	10.74	0.42	51.64	4	24
1993	13.19	9.02	2.18	35.29	5	25
1994	9.07	5.17	3.09	24.06	5	27
1995	5.67	4.24	0.41	18.47	5	48
1996	6.66	5.54	1.88	25.96	5	48
1997	7.02	5.56	1.06	23.53	5	46
1998	4.66	2.69	0.35	12.10	5	46
1999	5.89	2.95	0.38	13.15	5	41
2000	9.17	8.86	1.13	48.95	5	40
2001	8.24	5.73	0.72	34.58	5	40
2002	9.21	4.52	2.08	20.39	5	30
2003	7.01	8.05	0.55	53.92	5	40
2004	5.70	2.70	1.66	13.28	5	29
2005	8.90	4.48	2.95	19.87	5	27
2006	11.33	4.81	4.29	21.00	5	22
2007	15.89	7.95	10.48	33.04	4	8

Figure 3.9 Plot of TN:TP ratio for 5 hypereutrophic lakes.

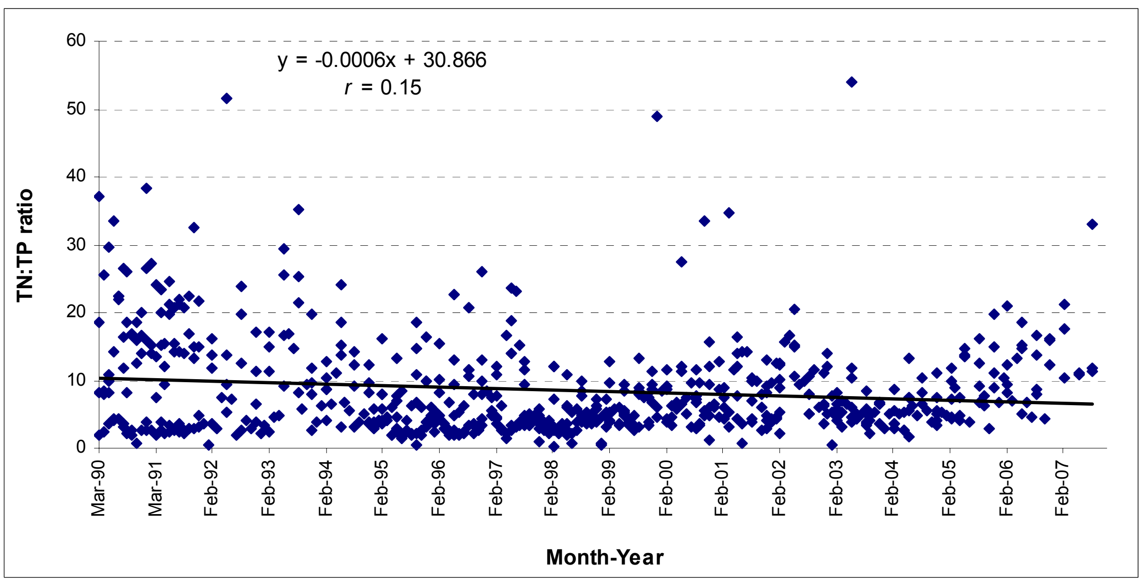


Figure 3.10 Change in the TN:TP ratio in 5 hypereutrophic lakes between Periods 1 and 2.

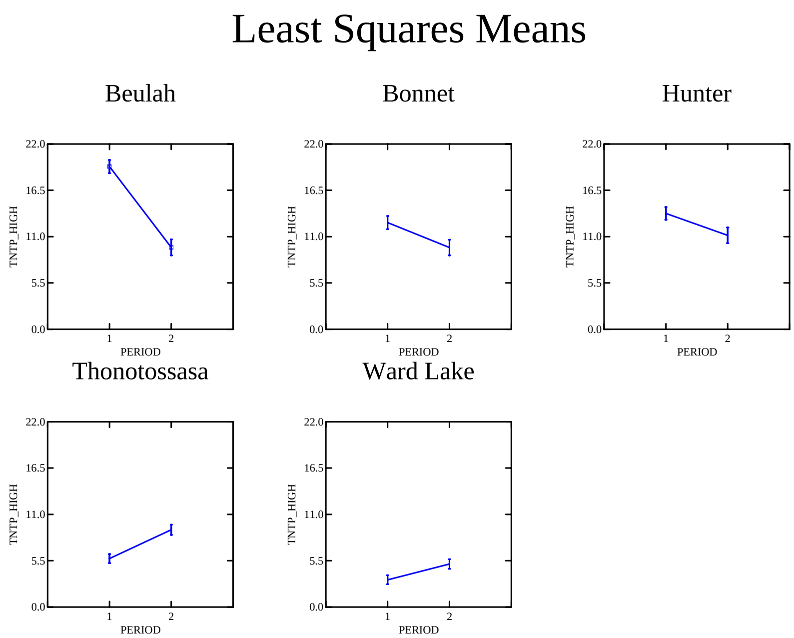


Table 3.5 Summary statistics for chlorophyll-α in μg L^{-1} collected by LAKEWATCH from 8 oligotrophic and mesotrophic lakes in two counties from 1990 through 2007

Year	Annual Average (μg L^{-1})	SD (μg L^{-1})	Minimum (μg L^{-1})	Maximum (μg L^{-1})	n (Lakes)	n (Samples)
1990	6.52	5.38	2.00	22.00	6	25
1991	8.79	7.36	1.00	34.00	7	71
1992	5.18	3.93	1.00	17.00	8	77
1993	3.83	2.67	1.00	11.00	7	71
1994	5.32	4.50	1.00	20.00	7	59
1995	5.32	5.51	1.00	23.00	7	57
1996	4.38	3.60	1.00	18.00	7	55
1997	6.69	3.84	1.00	18.00	7	64
1998	8.00	4.83	2.00	24.00	7	55
1999	6.76	3.96	1.00	20.00	6	62
2000	6.31	4.54	2.00	25.00	8	64
2001	5.73	4.63	1.00	24.00	8	73
2002	6.84	6.70	1.00	30.00	8	81
2003	10.85	11.05	1.00	48.00	8	73
2004	9.87	7.03	2.00	36.00	8	69
2005	13.80	10.98	2.00	41.00	7	55
2006	11.42	8.78	2.00	39.00	6	38
2007	5.27	3.17	2.00	11.00	5	26

Figure 3.11 Plot of chlorophyll-α concentration for 8 oligotrophic and mesotrophic lakes.

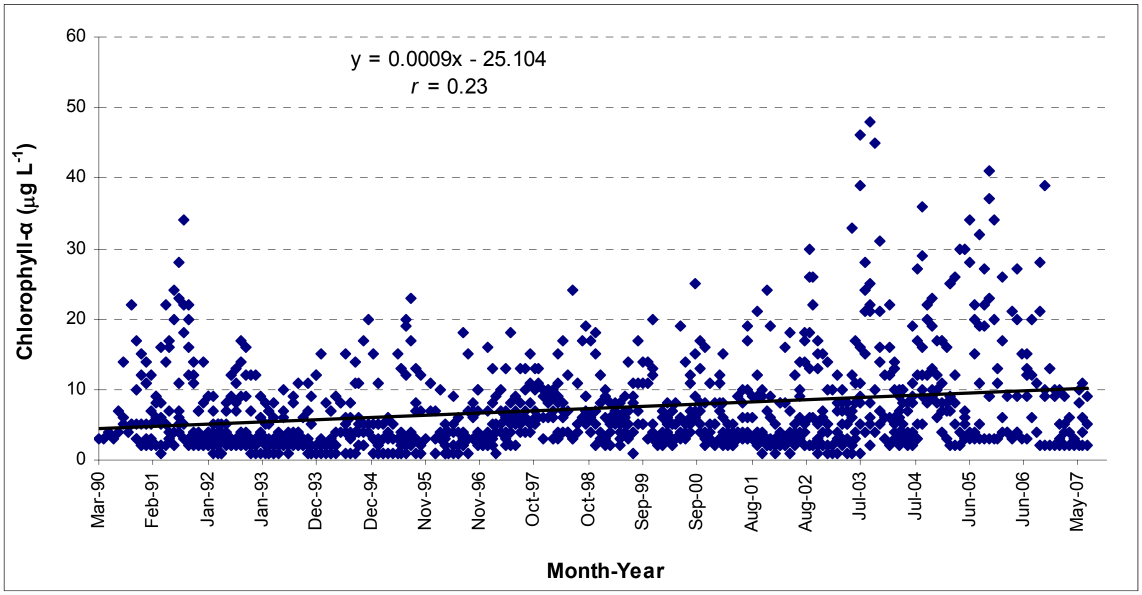
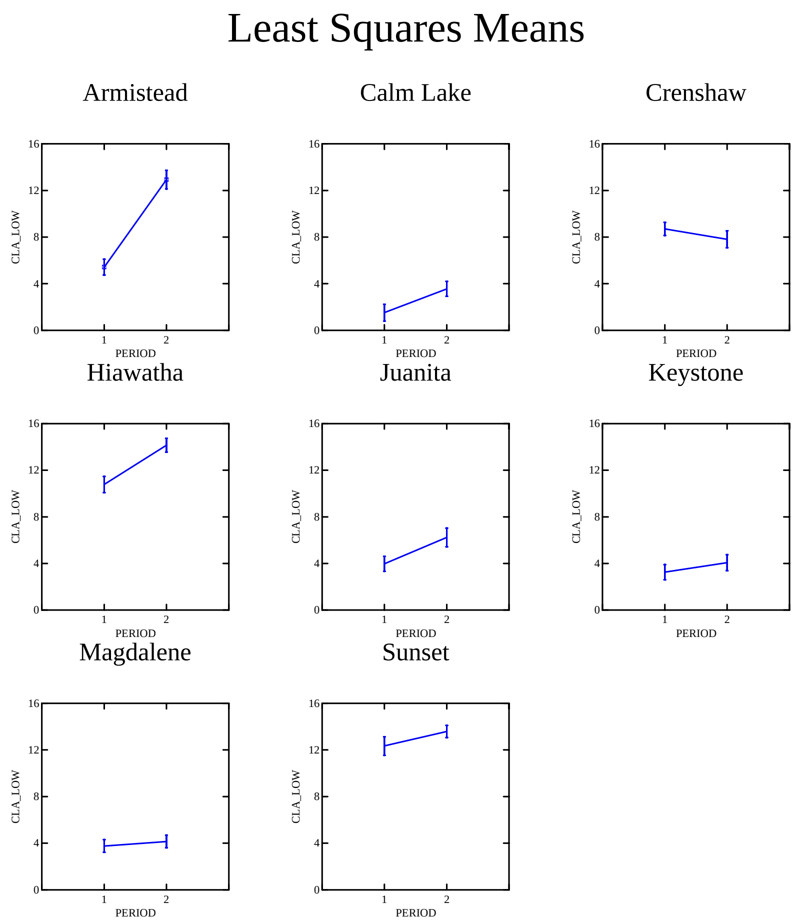


Table 3.6 Summary statistics for annual Chlorophyll-α concentration in µg L^{-1} collected by LAKEWATCH from 5 hypereutrophic lakes in three counties from 1990 through 2007

Year	Annual Average (µg L^{-1})	SD (µg L^{-1})	Minimum (µg L^{-1})	Maximum (µg L^{-1})	n (Lakes)	n (Samples)
1990	116.70	150.53	1.00	643.50	4	46
1991	80.59	73.33	20.30	447.40	5	48
1992	90.52	76.80	5.13	338.16	5	28
1993	52.42	48.26	3.02	185.76	5	36
1994	54.61	59.57	3.23	236.20	5	37
1995	63.72	44.05	5.10	151.41	5	36
1996	79.27	76.23	1.70	279.50	5	36
1997	60.85	62.55	0.50	241.00	5	35
1998	58.88	66.84	1.00	325.00	5	35
1999	70.78	55.79	1.70	273.79	5	37
2000	92.78	78.08	10.40	263.68	5	35
2001	73.92	60.60	3.30	213.33	5	36
2002	72.41	56.16	7.80	182.50	5	36
2003	84.00	54.55	12.00	194.10	5	32
2004	83.04	71.82	8.40	231.80	5	33
2005	88.56	73.22	2.40	366.80	5	41
2006	68.01	85.51	4.40	383.00	5	26
2007	47.59	47.58	6.70	139.60	4	15

Figure 3.12 Change in chlorophyll-α concentration in 8 oligotrophic and mesotrophic lakes between Periods 1 and 2.



Lake water concentrations of TP, TN:TP ratios, and chlorophyll-α were averaged and plotted by year for the interval 1990-2006. These results were consistent with those from analysis done upon the entire cloud of single data points in terms of the direction and significance of the association. An additional benefit of using the averages instead of the original data is a more clear visualization of a possible cycling effect for all the variables in both groups of lakes.

Figure 3.13 Plot of chlorophyll-α concentration for 5 hypereutrophic lakes.

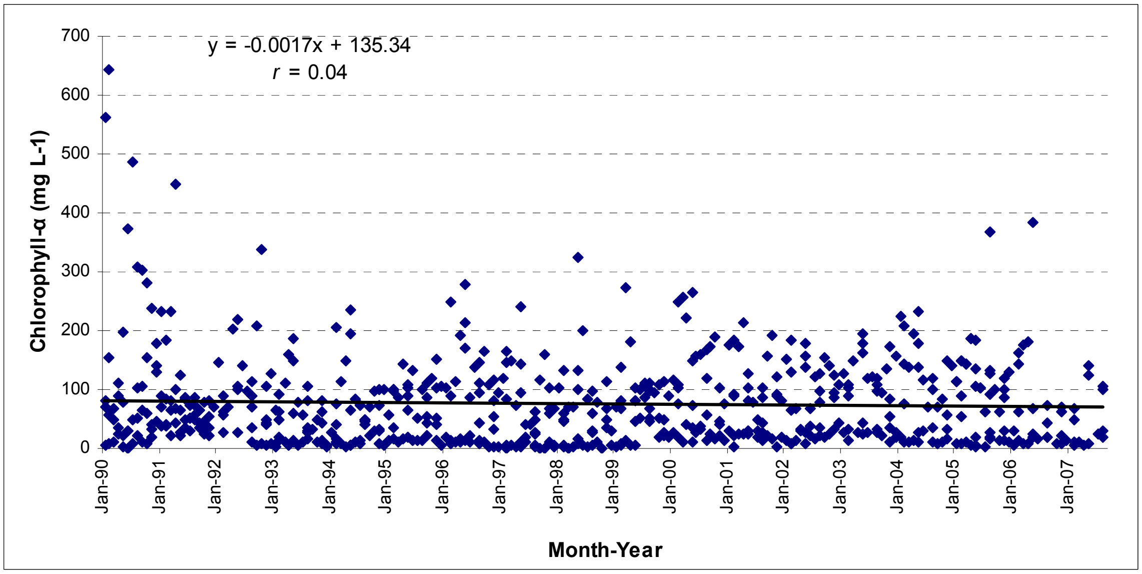
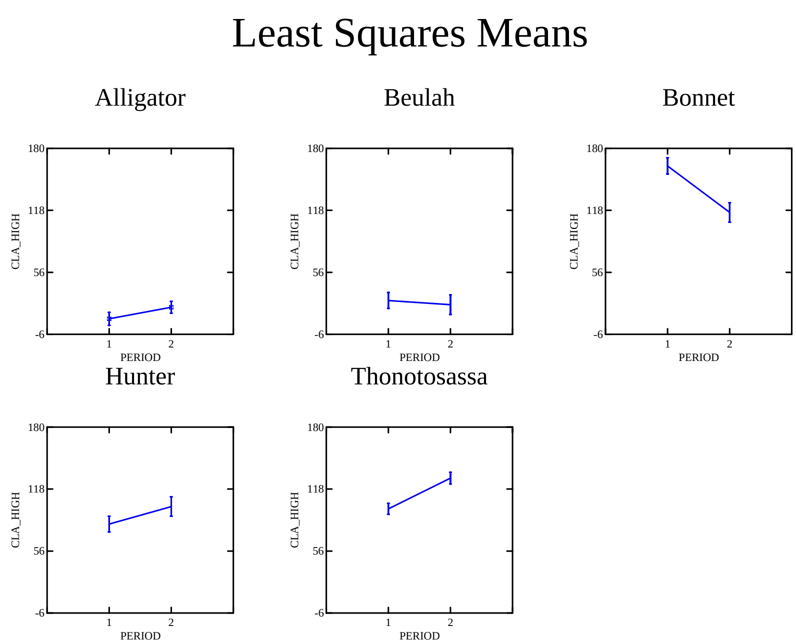


Figure 3.14 Change in chlorophyll-α concentration in 5 hypereutrophic lakes between periods 1 and 2.



Plots of annually averaged values against time showed a strong direct increase of TP and chlorophyll-α (r = 0.78 and 0.68 respectively, p <0.01 and p = 0.02, respectively, Figures 3.17 and 3.19).

Figure 3.15 Population growth estimates in the Tampa Bay Metropolitan Area from 1990 until 2006 (Hilssborough County, 2007; US Bureau of the Census 2000, 2007).

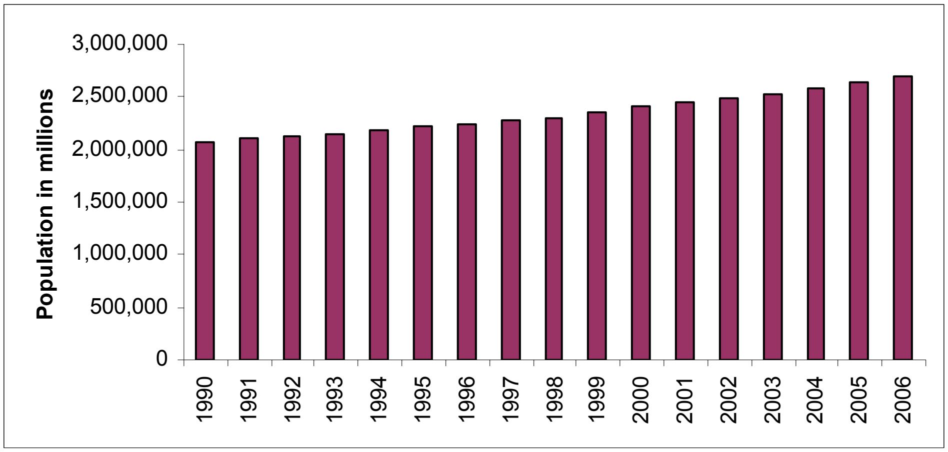
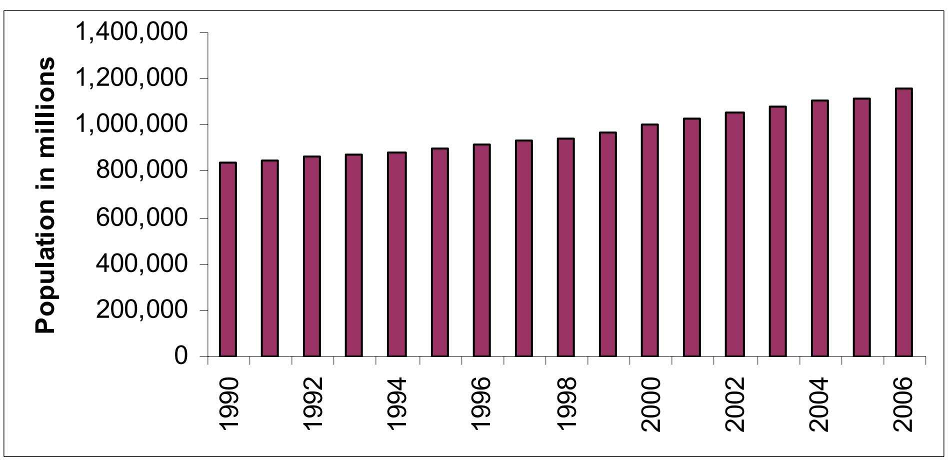


Figure 3.16 Population growth estimates in Hillsborough County from 1990 until 2006 (Hilssborough County, 2007; US Bureau of the Census 2000, 2007).



Rising lake water TP and chlorophyll-α concentrations in northeastern Hillsborough County appear to trend along with County population growth and may be evidence of an urban signature in lake water quality. Figures 3.15 and 3.16 show a continuous increase in population for the Tampa Bay Metropolitan Area (TBMA) and Hillsborough County: a 30.5% and 38.8% increase in the estimated populations of Tampa Bay Metropolitan Area and Hillsborough County, respectively, for the time period between 1990 and 2006 (Hilssborough County, 2007; US Bureau of the Census 2000, 2007). Figure 1.2 shows that residential and commercial uses are prevalent in the area. This might also have some association with the results.

The increase in lake water TP and chlorophyll-α concentrations in northeastern Hillsborough County may be related the characteristic karst formation of the area (van Beynen et al., 2007), which may help the transport of nutrients along with the groundwater flow from urban sources to the lakes or for some lakes as a consequence of the drought and over-pumping of the Floridian Aquifer, and thus loss of a relatively clean water supply. This study, however, cannot conclude the increase in TP concentrations in oligotrophic and mesotrophic lakes is due to new phosphorus been discharged into the lakes or to old phosphorus recycled from the sediments.

The historically-elevated lake water TP concentrations in lakes located in the eastern part of the Tampa Bay watershed may be explained in part by the mining activity in the Bone Valley. This economical activity has been operating in the area since the late 1800s (Brown, 2005) (Figure 1.2)

Figure 3.17 Change in annual averages of lake water TP concentration over time in years in oligotrophic and mesotrophic lakes of the Tampa Bay watershed.

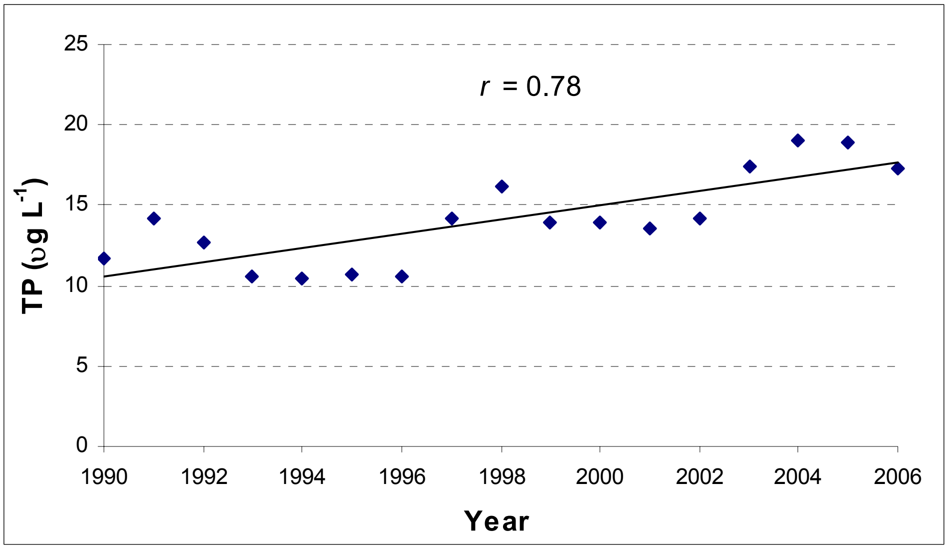


Figure 3.18 Change in annual averages of the TN:TP over time in years in oligotrophic and mesotrophic lakes of the Tampa Bay watershed.

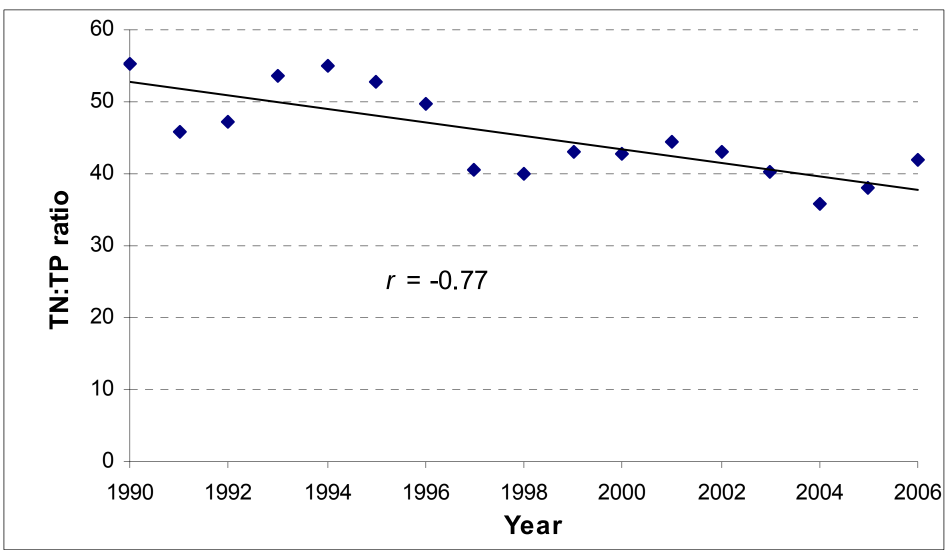
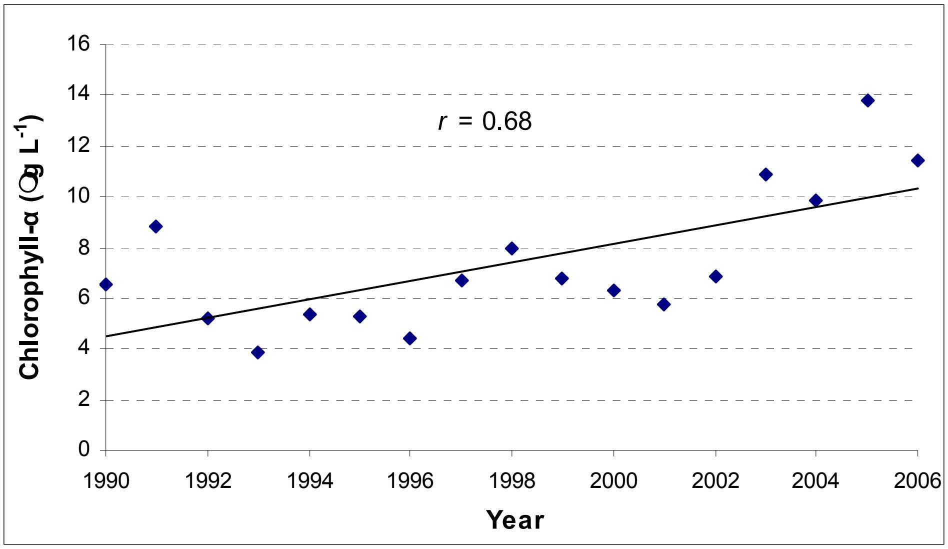


Figure 3.19 Change in annual averages of lake water chlorophyll- α concentration over time in years in oligotrophic and mesotrophic lakes of the Tampa Bay watershed.



The plot of the TN:TP for oligotrophic and mesotrophic lakes suggests a strong decline with time (r = -0.77, $p < 0.01$, Figure 3.18). This is again consistent with the result from previous analyses, which was explained by a greater trend of increasing lake water TP concentration as compared with TN concentration along the time period studied (Figure 3.7).

Figure 3.20 Change in annual averages of lake waterTP concentration over time in years in hypereutrophic lakes of the Tampa Bay watershed.

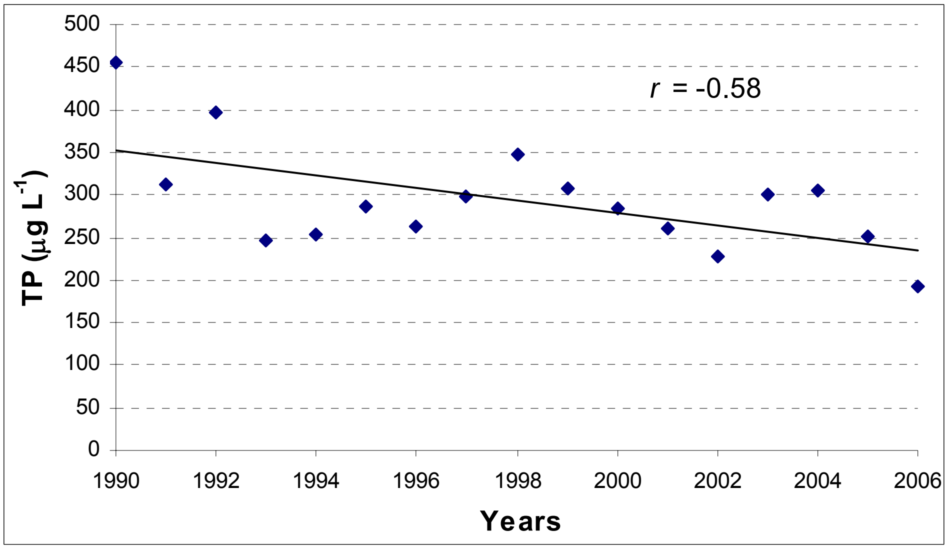


Figure 3.21 Change in annual averages of the TN:TP over time in years in hypereutrophic lakes of the Tampa Bay watershed.

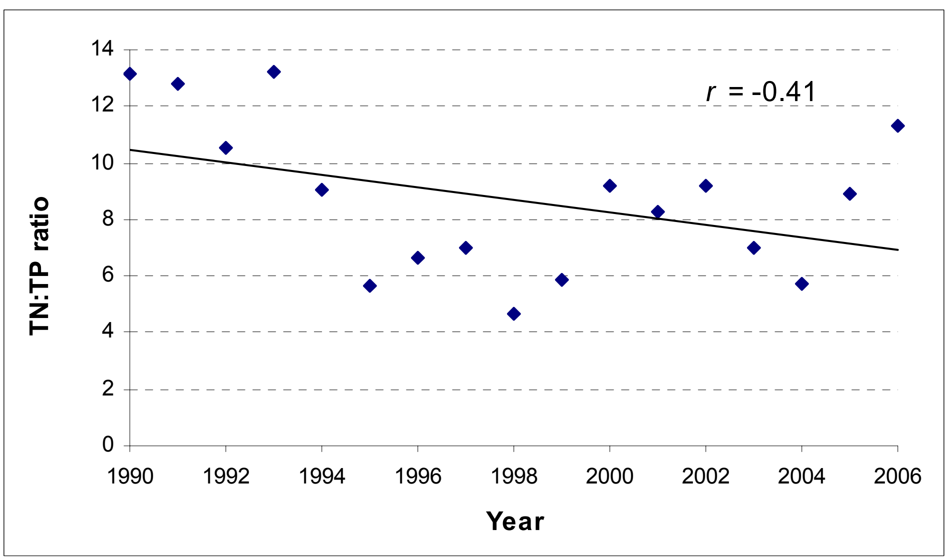
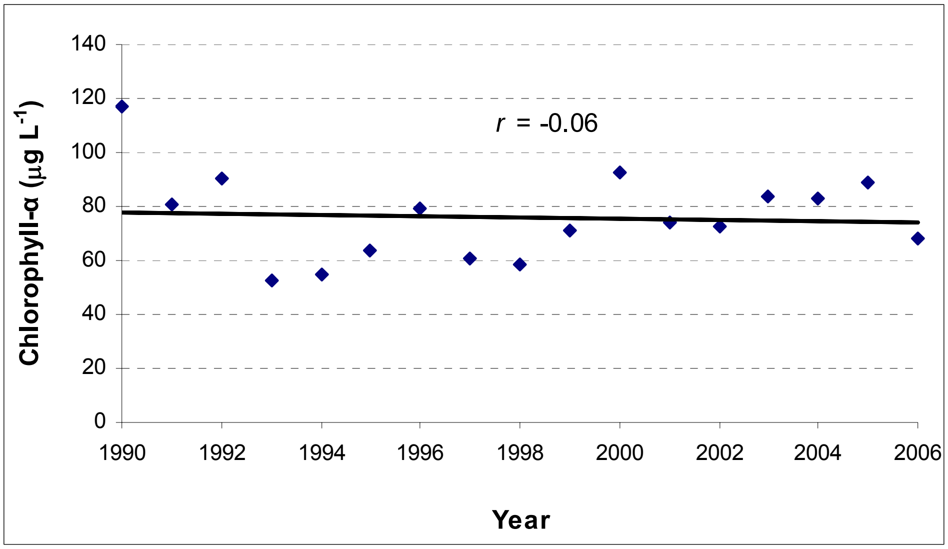


Figure 3.22 Change in annual averages of lake water chlorophyll-α concentration over time in years in hypereutrophic lakes of the Tampa Bay watershed.



In the group of hypereutrophic lakes, annually averaged lake water concentrations of TP showed similar tendencies as with linear regressions with the individual data points and ANOVA.. This curve suggested a significant medium inverse associations (r = -0.58, p < 0.01 for total phosphorus, Figure 3.20). Unlike results from the analysis on the individual data points, the analysis of annual average TN:TP ratios indicated a medium association (r=-0.41) but it was not significant (p > 0.05, Figure 3.21). Results for chlorophyll-α oncentrations against years were consistent with previous analyses and showed no significant correlation (r = -0.06, Figure 3.22).

3.5 Summary

The results from this chapter indicate that 16 lakes in the Tampa Bay watershed had a small but significant change in the concentration of trophic state variables between 1990 and 2007. There was a different behavior in water quality trends depending upon the degree of lake water eutrophicatuion.

Concentrations of TP and chlorophyll-α increased during the 18-years study period for oligotrophic and mesotrophic lakes. The TN:TP ratio showed that the study group of oligotrophic and mesotrophic lakes is phosphorus-limited. This ratio showed a significant decline and may reflect that lake water TP concentrations are increasing at a faster pace than lake water TN concentrations.

As expected from the increase in TP and the phosphorus limitation of these lakes, an analysis of the chlorophyll-α concentration in lake water also suggested a slight increase over time. One likely explanation for the increase in TP and chlorophyll-α (and TN, results not included) is that population growth in Hillsborough County has increased the phosphorus and nitrogen loading to these lakes. For example, increased fertilizer use in lawns and/or higher run-off rates would likely result I elevated chlorophyll-α concentration. An important factor that may be facilitating this process is the sandy karst characteristic of the soil profile, which facilitates the leaching of nutrients into the lakes. These possible explanations for phosphorus increase over time, however, are just suggestions since there is not conclusive evidence showing if the extra phosphorus responsible for the TP increase in the lake water column is recycled from the sediments or if it is a new external input to the system. Whatever is the case, the fact is that these results obtained from this group of 10 oligotrophic and mesotrophic lakes meet the alternative hypotheses forTP, TN:TP, and chlorophyll-α concentrations.

Hypereutrophic lakes presented a different behavior as compared to oligotrophic or mesotrophic lakes. This second group of lakes had a decreasing trend in water concentration of TP during the 18-year study period. The ratio of TN:TP showed that these highly eutrophied lakes were nitrogen-limited. Chlorophyll-α did not show indications of any trend over time at all in these hypereutrophic lakes, and are more in accordance with what was indicated by Terrell et al., (2000), in a bigger sample of lakes.

Decrease in lake water TP concentration in this second group of lakes may be explained by the implementation of lake management plans (City of Lakeland, 2001; Southwest Florida Water Management District, 2003) that may have prioritized highly eutrophied lakes over less eutrophied ones. The fact that both groups of lakes resulted having a decreasing trend in the TN:TP ratio raise concerns for a shift in phytoplankton composition to more noxious species.

CHAPTER 4.

IDENTIFICATION OF IMPORTANT VARIABLES AFFECTING WATER QUALITY IN LAKES OF THE TAMPA BAY WATERSHED

4.1 Introduction

Aquatic vegetation can contain important proportion of the total nutrient content of the lake, it is therefore a key element to consider when assessing the potential concentration of nutrients in the lake water column (Canfield et al., 1983). As it is known based on the literature reviewed in Chapter 2 (Bachmann et al., 2002; Dierberg et al., 2002; Hamilton and Mitchell, 1996), submerged rather than emergent aquatic vegetation has a greater potential to be associated to low levels of TP, TN, and chlorophyll-α in lake water, hence it is one of the variables examined in this chapter for possible association with eutrophication status, along with lake water total phosphorus (TP), total nitrogen (TN) concentration, and lake area, depth, and volume.

It has been very well documented the strong and clear direct association between phytoplankton as measured by chlorophyll-α and nutrients dissolved in lake in Florida (Bachmann et al., 2002; Brown et al., 2000; Canfield et al., 1984). Although less conclusive, there has been also documentation reporting that large amounts of submerged aquatic macrophytes have some association with reduced productivity of lake phytoplankton (Bachmann et al., 2002; Canfield and Hoyer, 1992; Canfield et al., 1984; Landers, 1982).

In this chapter a 34-lake database that includes information on the variables submerged aquatic vegetation (two forms); lake water concentration of TP, TN, and chlorophyll-α; and area, depth and volume is introduced and described. Correlation between these lake variables are examined and compared with results from previous studies. Some of the species of submerged aquatic vegetation more abundant in the lakes studied were: *Vallisneria Americana*, *Algal ssp*., *Hydrilla verticilata*, *Egeria densa,* and *Potamogeton spp.,* among others.

4.2 Methods

4.2.1 Data Sampling Program

The 34 urban and suburban lakes examined in this chapter are distributed over an area with mixed land use: residential, recreational, and agricultural, in the northern and eastern portions of Hillsborough County (Figure 4.1). The lakes are distributed over four subbasins within the Tampa Bay watershed; Sweetwater Creek, Rocky / Brushy Creek, Brooker Creek, and Curiosity Creek. The analysis was carried out by using existing data from two different sources. Data on lake water TP, TN, and chlorophyll-α concentrations, as well as data on submerged aquatic vegetation, was collected by the Florida Center for Community Design and Research at the University of South (Koenig and Eilers, 2006-2007). Data availability was the determining factor for inclusion of the 34 lakes in this analysis. Values on general lake variables: lake surface area, mean depth, and lake volume, were collected and reported by volunteer citizens from the Florida LAKEWATCH water quality monitoring program and were obtained as part of the Hillsborough County sampling program (2008).

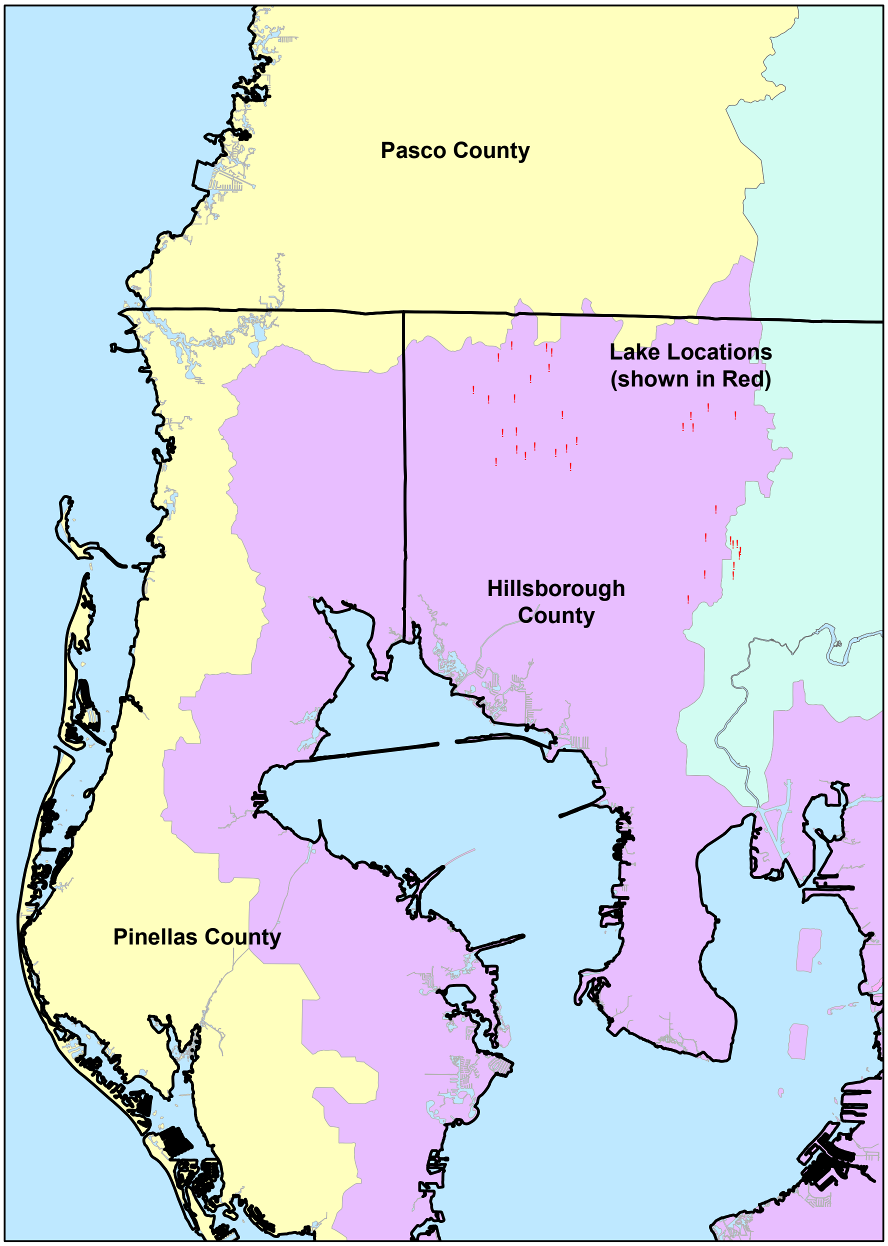
Samples taken to determine TP, TN, and chlorophyll-α were analyzed by the Hillsborough County Environmental Protection Commission laboratory (Chapter 3). Submerged aquatic vegetation is expressed as percentage of area covered with vegetation (PAC) and percentage of volume of the lake infested with vegetation (PVI). Variables for each lake correspond to only one measurement done in 2006 or 2007 (depending on the lake). These data proceed from measurements carried out at one point in time. They are, however, useful to examine potential long term associations between the mentioned variables because each particular lake water variable depends on a historical trend, and no variable changed suddenly and independently from their past conditions.

According to Griffin (2008) data on PAC were determined by first selecting 100 randomly ordered bathymetric points, then reaching the points by boat to determine the presence of submerged aquatic vegetation by the soft return data obtained with a fathometer and expressing the results in terms of percentage. PVI was calculated by measuring the depth of the soft returns (top of vegetation) and the depth of the hard returns (lake bottom) using the bathymetric trace for each point as shown in Equation (1):

Equation (1): $$\frac{\sum_{0}^{100}\left(\frac{(Lake\ Depth - Depth\ of\ Vegetation)}{Lake\ Depth}\right)}{100} = \text{PVI}$$

For points where no vegetation exists, the numerator is zero and that point is counted as zero.

Figure 4.1 Location of lakes with recent observations on submerged aquatic vegetation (Koenig and Eilers, 2006-2007).



Map Created by Pete Reehling – University of South Florida

4.2.2 Statistical Analysis

Analyses of correlation were used to determine significant associations between the eutrophic state variables lake water TP, TN, and chlorophyll-α concentration and variables of submerged aquatic vegetation, lake area, depth, and volume. This analysis was conducted using the formula for the Pearson correlation in Excel shown in Equation (2), where *X* and *Y* are the lake variables, *X* bar and *Y* bar are the respective means, and SD_x and SD_y are the respective standard deviations.

Equation (2): $$r = \frac{\sum_{i-1}^{n}\left(X_i - \overline{X}\right)\left(Y_i - \overline{Y}\right)}{(n-1)SD_x SD_y}$$

The variables TP and TN were taken as indicators of eutrophication (dependent variables) but also as factors associated to eutrophication (independent variables) when eutrophication was indicated by chlorophyll-α. A correlation > 0.30 was statistically significant at a 95% of confidence level.

4.3 Results and Discussion

Values in Table 4.1 present summary statistics for physical and chemical variables analyzed in this study. Variables for each lake correspond to one measurement done in 2006 or 2007. The lakes had an average mean depth of

2.51 m, ranging from 0.91 to 3.96 m therefore approaching the concept of shallow lakes used by Scheffer (2004) according to which lakes depth less than 3 m were considered shallow. Lake surface area ranged from 1.21 to 174.43 ha with a mean of 25.42 ha. The average volume was 713,300 m^3 with a minimum of 2,087 and maximum of 5, 714,000 m^3.

According to the Trophic State Classification System of System Forsberg and Ryding (1980, Figure 3.1) the lakes ranged from mesotrophic to eutrophic with average values of 25.18 μg L^{-1}, 0.80 mg L^{-1}, and 7.90 μg L^{-1} for TP, TN, and chlorophyll-α respectively. Ranges for these variables were: 3.00 to 50.00 μg L^{-1} for TP, 0.37 to 1.27 mg L^{-1} for TN, and 1.20 to 44.61 μg L^{-1} for chlorophyll-α. The average percent of study lake area covered by submerged aquatic vegetation (PAC) was 38.78% with a minimum and maximum of 2.00 and 85.00% respectively. On average the volume of the study lakes occupied by submerged aquatic vegetation (PVI) was 16.17% and ranged between 0.52% and 47.00% (Table 4.1).

The analysis of correlation between all the variables examined in this study (Table 4.2) presented significant inverse correlation between chlorophyll-α and both measures of submerged aquatic vegetation: PAC and PVI, with r = -0.72 and -0.60, respectively. TP, r = 0.46; and TN, r = 0.41. TP was also inversely correlated with both forms of submerged aquatic vegetation, r = -0.52 and -0.41 for PAC and PVI, respectively, and additionally with mean depth, r = -0.36. TN was not significantly correlated ($p > 0.05$) with any variable other than chlorophyll-α. The fact that chlorophyll-α presented a higher correlation with TP as compared to TN may reflect the phosphorus limitation of most of the lakes studied.

The greater negative correlation found between chlorophyll-α and both measures of submerged aquatic vegetation as compared with the positive correlation between chlorophyll-α and TP and TN may suggest that submerged aquatic vegetation rather than TP and TN might be the factor most associated to phytoplankton. If this is the case, this result suggests that submerged aquatic vegetation may be associated with chlorophyll-α also through some other additional way not involving TP and TN. Batchman et al. (2002), however, reported a smaller negative correlation between chlorophyll-α and submerged aquatic vegetation (r = -0.29) as compared to those between chlorophyll-α and TP (r = 0.82), and chlorophyll-α and TN (r = 0.70). Yet the authors consider that under conditions where water nutrient concentration is not excessively elevated, submerged aquatic vegetation reduce nutrients and consequently phytoplankton concentration as measured by chlorophyll-α, rather than dissolved nutrients reducing submerged aquatic vegetation growth. Other authors also suggest association between chlorophyll-α with TP and TN (Brown et al., 2000; Canfield et al., 1984).

A suggested inverse relationship between submerged aquatic vegetation and chlorophyll-α in the water column might be explained by the effect possibly caused by submerged aquatic vegetation in reducing concentration of limiting nutrients in the water column, and the subsequent reduced availability of phytoplankton (Bachmann et al., 2004; Scheffer, 2004); and by provision of shelters from predatory fish for zooplankton that directly prey on phytoplankton (Scheffer, 2004).

In general, the mechanism by which submerged aquatic vegetation reduce nutrient concentration has been described in part by the attenuation of water turbulence, which results in less resuspension and recycling of nutrients back into the water column (Bachmann et al., 2004; Hamilton and Mitchell, 1996; Scheffer, 2004). Other contributing mechanism may be the provision of substrate surface for periphyton that up-take nutrients from the water column (Bachmann et al., 2004; Cattaneo and Kalff, 1980); Up-take of nutrients from the water column by submerged aquatic vegetation directly (Denny, 1972; Graneli and Solander, 1988); and by influencing ion exchange reactions via regulation of dissolved oxygen and pH (Graneli and Solander, 1988). All these reasons may help explain the inverse relationship found between submerged aquatic vegetation and TP, and consequently also with chlorophyll-α.

The present study found mean depth to be directly correlated with both forms of submerged aquatic vegetation, r = 0.34 and 0.31 for PAC and PVI respectively. This may seems to be unexpected since the required light penetration for photosynthesis is reduced with depth, however, analysis of regression between submerged aquatic vegetation abundance and depths (results not presented) showed an increase in submerged aquatic vegetation up to a depth of approximately 2.5 m and then a decrease. The fact that most of the lakes studied were shallower than that depth, explains why the overall results showed a direct correlation between both variables. Other additional possible explanation is that shallow depths might favor dominance of emergent aquatic vegetation over submerged aquatic vegetation. As also expected, both mean depth and area were correlated with volume r = 0.38 and 0.98, respectively. Since PAC and PVI are both expressions of the same variable submerged aquatic vegetation, they were consequently highly correlated, r = 0.91. Association of these two related sub variables has been reported in the literature (Canfield and Hoyer, 1992; Canfield et al., 1984).

The inverse characteristics of the relationship between TP and mean depth may be due to the greater distance between the source of resuspended phosphorus in the bottom sediments and the superior layers of the water column. Stronger turbulence would be required to resuspend phosphorus through the entire water column. Another possible reason is the obvious direct correlation between depth and volume. An increasing depth would be associated with a greater volume of water and consequently greater dilution of phosphorus.

An analysis of correlation of the eight variables of shallow lakes grouped by groups would result as follow: (1) submerged aquatic vegetation variables, PAC and PVI; (2) eutrophication variables, TP, TN, and chlorophyll-α; and (3) lake size variables, area, depth, and volume. Lakes with more submerged aquatic vegetation have less eutrophication especially chlorophyll-α concentration followed by TP concentration. Bigger lakes have less eutrophication and depth is the most important. Bigger lakes tend to have more submerged aquatic vegetation (for shallow lakes).

Table 4.1 Summary statistics of Hillsborough lakes examined for association between lake variables.

	n	Median	Mean	SD	Minimum	Maximum
PAC	34	38.00	38.78	26.75	2.00	85.00
PVI	34	12.46	16.17	12.58	0.52	47.00
Volume (m^3)	34	381700	713300	1050000	2087	5714000
Area (ha)	34	15.99	25.42	32.85	1.21	174.43
Mean Depth (m)	34	2.59	2.51	0.77	0.91	3.96
TN (µg L^{-1})	34	0.86	0.80	0.24	0.37	1.27
TP (µg L^{-1})	34	24.50	25.18	11.39	3.00	50.00
Chlorophyll-α (µg L^{-1})	33	6.60	7.90	5.19	1.20	21.70

Table 4.2 Matrix table showing the analysis of correlation between lake variables. Values in dark correspond to resulting significant associations between trophic state parameters (TN, TP, chlorophyll-α) and theirs factors controlling for water quality at 95% confidence.

	PAC	PVI	Volume (m^3)	Area (ha)	Depth (m)	TN (mg/L)	TP (µg L^{-2})	Chla (µg L^{-2})
PAC	1.00							
PVI	0.91	1.00						
Volume (m^3)	0.24	0.24	1.00					
Area (ha)	0.22	0.23	0.98	1.00				
Depth (m)	0.34	0.31	0.38	0.28	1.00			
TN (mg L^{-2})	-0.27	-0.21	0.11	0.10	-0.27	1.00		
TP (µg L^{-2})	**-0.52**	**-0.41**	-0.13	-0.14	**-0.36**	0.26	1.00	
Chla (µg L^{-2})	**-0.72**	**-0.60**	-0.29	-0.27	-0.27	**0.41**	**0.46**	1.00

4.4 Summary

In summary, among the eight lake indicators (TP, TN, chlorophyll-α, PAC, PVI, mean depth, lake surface area, and lake volume) examined in this chapter, the strongest statistically significant association was found to be the inverse relationship between chlorophyll-α and submerged aquatic vegetation as represented by PAC and PVI. The second strongest statistically significant association was the relationship between chlorophyll-α and lake water TP concentration. Submerged aquatic vegetation was not significantly associated to lake water TN concentration. In general, more submerged aquatic vegetation is associated with less eutrophication.

The higher correlation of chlorophyll-α with abundance of submerged aquatic vegetation as compared to water total phosphorus was a finding not expected and may indicate that submerged aquatic vegetation is associated to chlorophyll-α also through some other way that does not involve nutrients. This is consistent with the idea that phytoplankton is affected by submerged aquatic vegetation through both effects on TP concentration and through other effect s, for example, as a shelter for phytoplankton-grazing zooplankton. Batchman et al. (2002) and Batchman et al. (2004) although suggested a weak inverse relationship between both parameters, indicated that aquatic macrophytes did not significantly affect the phosphorus versus chlorophyll relationships in their studies.

In general, for shallow lakes, bigger lakes had more submerged aquatic vegetation, Bigger lakes were less eutrophied and the largest correlation was between depth and lake water TP concentration.

CHAPTER 5.

EFFECT OF SUBMERGED AQUATIC VEGETATION ON WATER TOTAL PHOSPHORUS CONCENTRATION IN LAKES OF THE TAMPA BAY WATERSHED

5.1 Introduction

Among the different types of aquatic vegetation, submerged aquatic vegetation has been found to play an important role in regulation of nutrient concentrations and subsequently lake phytoplankton (Bachmann et al., 2004; Brenner et al., 1999; Jeppesen et al., 1997; Knight et al., 2003), and also in a more direct way, probably shelter for grazers (Scheffer, 2004). The nature and extent of these relationships, however, still remain vague (Bachmann et al., 2002). It has been speculated that at high levels of nutrient concentrations, nutrients may control submerged aquatic vegetation while in waters with more moderated and lower nutrient concentrations, nutrients may be controlled and further reduced by aquatic macrophytes especially submerged aquatic vegetation (Bachmann et al., 2002; Bachmann et al., 2004).

Studies conducted in other geographic areas seem to be even less unifying in terms of clarifying the nature of this relationship. Nutrient levels in water have been found to trigger growth of submerged aquatic vegetation

(Ozimek, 1978, as quoted by Duarte and Kalff, 1986), not to cause a clear effect (Carpenter and Adams, 1979), and to decrease prevalence of submerged aquatic vegetation (Duarte, 1995) especially at a large increase in phosphorus level (Graneli and Solander, 1988). It has been reported in literature that lakes have changed from being in a clear water state to turbid water state (with a higher concentration of nutrients and suspended solids), when submerged aquatic vegetation was removed by herbicide treatment (O'Dell et al., 1995), or by hurricanes (Bachmann et al., 1999). Likewise, lakes have been reported to switch from a turbid to clear water state when planktivorous fish were removed and submerged aquatic vegetation increased (Ozimek et al., 1990).

It was shown in Chapter 4 that submerged aquatic vegetation, and total phosphorus (TP) to a lesser extent, were the most significant variables associated with phytoplankton productivity. Consequently it is important to further examine the relationship between these two variables. In this chapter the same data analyzed in Chapter 4 from a group of 34 lakes in Hillsborough County was examined for a possible association between the percentage of volume and area of the lake that is occupied by submerged aquatic vegetation (PVI and PAC) versus the concentration of total phosphorus (TP), total nitrogen (TN), and chlorophyll-α in lake water. The difference is that this analysis only covers submerged aquatic vegetation variables and eutrophication variables (not size variables) because these were the more strongly correlated in Chapter 4. Also examined was the association between submerged aquatic vegetation density and inter-annual variability of lake water TP concentration in the mentioned parameters in a subsample of 24 lakes.

A strong link between water nutrients concentration and phytoplankton biomass has been reported in studies conducted in Florida lakes (Bachmann et al., 2002; Canfield, 1983). The relationship between nutrients and chlorophyll-α with submerged aquatic vegetation in lake water has shown to be more complex and difficult to clarify. Some authors, however, have already approached some measure of relationship between nutrients and submerged aquatic vegetation in Florida lakes (Bachmann et al., 2002) and other geographical areas (Cattaneo and Kalff, 1980; Duarte, 1995; Graneli and Solander, 1988; Scheffer, 2004), and the speculated mechanisms (sediments stabilization, plant up-take, precipitation, redox reactions, and shelters) by which they may be related (Bachmann et al., 2004; Cattaneo and Kalff, 1980; Duarte, 1995; Graneli and Solander, 1988; Scheffer, 2004). Other authors have reported information about the relationship between phytoplankton and submerged aquatic vegetation (Canfield and Hoyer, 1992; Canfield et al., 1984). Still in general, much needs to be done to confirm the relationship, if one exists, between eutrophication variables and submerged aquatic vegetation.

5.2 Objectives and Hypothesis

The analysis conducted in this chapter was intended to provide evidence regarding the relationship between submerged aquatic vegetation and TP concentrations. As a complementary analysis, submerged aquatic vegetation was examined in relation to chlorophyll-α and TN concentrations, as was any influence of submerged aquatic vegetation in the inter-annual variability of water TP concentration. The results could be applied to develop best management practices to control cultural eutrophication associated with watershed development. The following objectives and hypothesis are addressed in this chapter:

- Objective: To provide evidence that the presence of submerged aquatic vegetation plays a significant role in lowering total phosphorus (TP) concentration in lake water in lakes of Hillsborough County.
  - *Hypothesis 1:* Lakes with little vegetation have higher TP, TN, and chlorophyll-α concentrations.

- Objective: To determine if the presence of submerged aquatic vegetation in urban and suburban lakes of the Tampa Bay watershed is strongly and inversely associated to the inter-annual fluctuation in lake water TP concentration.

5.3 Methods

5.3.1 Data Description

Methods are described in Chapter 4.

5.3.2 Statistical Analysis

In order to detect any possible effect on lake water TP, TN, and chlorophyll concentration presumably due to the presence of submerged aquatic vegetation, the lakes were separated in two groups, those with a high presence of submerged aquatic vegetation versus those with a low presence. The separation criteria used to delineate a difference between the two groups was defined by Bachmann et al. (2002). According to this, high presence of submerged aquatic vegetation is considered as macrophyte-dominated and included those lakes with PVI>80 while low presence of submerged aquatic vegetation corresponded to phytoplankton-dominated, which are those lakes with PVI<20. Since none of the 34 lakes in this study had a PVI greater than 80, a PVI of 20 was considered as the cut-point for separation of the two groups. Hence for the purpose of this study, lakes with a PVI greater than 20 were considered macrophyte-dominated while lakes with a PVI less than 20 were considered phytoplankton-dominated. One-way ANOVA was used to determine if the means of both groups of lakes were significantly different for each one of the three variables considered (TP, TN, and chlorophyll-α). Bachmann et al.

(2002) conducted similar analysis using one-way ANOVA to determine differences between the means of two lake groups for eutrophication variables. A t-test assuming unequal variances was additionally used to confirm the results. Both tests were performed at the 95% confidence level ($\alpha = 0.05$). Graphics with frequency distributions were made for visual comparison between the two groups of lakes for each one of the three trophic state variables examined.

A linear regression was used to represent a possible relationship between variability of lake water TP concentration and submerged aquatic vegetation dominance in 24 lakes. The variability of TP concentration was indicated by expressing the coefficient of variation (CV) of this eutrophication variable for the available values from 2005 until present in terms of the mean of this variable for the same time period. The year 2005 as a cut-off point for inclusion of values of eutrophication was chosen arbitrarily intending to have a short period but long enough to show variability. A long period of time for observations in eutrophication variables would have increased the uncertainty for the corresponding values in submerged aquatic vegetation, which are unknown. Submerged aquatic vegetation dominance was indicated by one-time value of both PAC and PVI since this was the only available value for this parameter.

5.4 Results and Discussion

Out of the overall group of 34 lakes, 9 were classified as macrophyte-dominated. The average percent volume infested (PVI) in this group was 34.15

with a minimum and maximum of 23.00 and 47.00 respectively; 9.69 was the average PVI for the 25 lakes composing the phytoplankton-dominated group, with a minimum and maximum of 0.52 and 19.00 respectively (Table 5.1). The average percent of area covered with submerged aquatic vegetation (PAC) in the macrophyte-dominated group was 74.44 with a minimum and maximum of 63.00 and 85.00 respectively. The average for the same variable in the phytoplankton-dominated group was 25.93 and ranged between 2.00 and 56.00 (Table 5.2).

Table 5.1 Summary table for PVI values of lakes with macrophyte-dominance and phytoplankton-dominance.

Lake dominance	n	Median	Mean	SD	Minimum	Maximum
Macrophyte-dominated	9	33	34.1	8.4	23	47
Phytoplankton-dominated	25	10	9.6	5.3	0.52	19

Table 5.2 Summary table for PAC values of lakes macrophyte-dominance and phytoplankton-dominance.

Lake dominance	n	Median	Mean	SD	Minimum	Maximum
Macrophyte-dominated	9	76	74.4	7.6	63	85
Phytoplankton-dominated	25	30	25.9	17.7	2	56

The TP concentration in the group of lakes with macrophyte-dominance ranged from 3 to 31 μg L^{-1} with an average of 18.44 μg L^{-1}, while the group with phytoplankton-dominance showed a range from 10 to 50 μg L^{-1} with an average of 27.6 μg L^{-1} (Table 5.3). According to the Trophic State Classification System of Forsberg and Ryding (1980, Figure 3.1) some lakes from both groups overlap under the classification of oligotrophic ([TP] <15 μg L^{-1}), mesotrophic (15< [TP] <25 μg L^{-1}), and eutrophic ([TP] >25 μg L^{-1}). No lake in either group fell under the classification of hypereutrophic ([TP] >100 μg L^{-1}). The range in macrophyte-dominated lakes for chlorophyll-α went from 1.2 to 6.9 μg L^{-1} with an average of

3.60 μg L^{-1}, while 3.6 to 44.61 μg L^{-1} with average of 10.91 μg L^{-1} was the range for the same variable in phytoplankton-dominated lakes (Table 5.4). These ranges show substantial overlap for both variables in both lake groups, which do not guarantee accurate prediction of water quality based only on submerged aquatic vegetation prevalence. Despite the overlap, however, the results of a t-test indicated a significant difference between both groups of lakes for TP and chlorophyll-α (p = 0.0002 and 6.4 x 10^{-9} respectively). This difference was confirmed in one-way ANOVA for the same parameters (p = 0.036 and 0.017 respectively).

TP and chlorophyll-α values were significantly higher for lakes with a PVI lower than 20 (phytoplankton-dominated) as compared to those with a PVI higher than 20 (macrophyte-dominated). This not just support results of significant association of submerged aquatic vegetation with TP and chlorophyll-α described in Chapter 4 but also indicates a strong relationship between submerged aquatic vegetation and TP and chlorophyll-α. Results are additionally supported by literature that indicate the nutrient removal capacity of submerged aquatic vegetation from water column (Dierberg et al., 2002; Gu et al., 2001; Knight et al., 2003). Some possible mechanisms by which submerged aquatic vegetation reduces nutrient concentrations in the water column are described in Chapter 2 and more briefly in Chapter 4.

Table 5.3 Summary table for TP values (µg p L^{-1}) for lakes with macrophyte-dominance and phytoplankton-dominance.

Lake dominance	n	Median	Mean	SD	Minimum	Maximum
Macrophyte-dominated	9	19	18.44	8.99	3	31
Phytoplankton-dominated	25	26	27.60	11.35	10	50

Table 5.4 Summary table for chlorophyll-α values (µg L^{-1}) for lakes with macrophyte-dominance and phytoplankton-dominance.

Lake dominance	n	Median	Mean	SD	Minimum	Maximum
Macrophyte-dominated	9	3.80	3.60	1.82	1.20	6.90
Phytoplankton-dominated	25	8.80	10.91	8.64	3.60	44.61

Analysis of frequencies in both group of lakes for TP (Figures 5.2 and 5.3) and chlorophyll-α concentrations (Figures 5.4 and 5.5) showed that more lakes in the group with macrophyte-dominance were toward the upper limit of values in both parameters (skewed to the left). And more lakes in the phytoplankton-dominated group were toward the lower limit of the range for this group (skewed to the right).

There was no significant difference found between the group of lakes with macrophyte-dominance and those with phytoplankton-dominance regarding values in the concentration of TN (t-test, $p<0.071$; ANOVA, $p<0.326$; Figures 5.6 and 5.7). This result matches the lack of association detected between TN and submerged aquatic vegetation in the analysis of regression and correlation conducted in the previous chapter, but differs with results of Batchman et al. (2002) and Batchman et al. (2004) that suggested such an association. Average lake water TN concentration in macrophye-dominated lakes was 0.73 mg L^{-1} with a minimum and maximum of 0.37 and 1.27 mg L^{-1} respectively. The range for the same parameter in phytoplankton-dominated lakes went from 0.41 to 1.13 mg L^{-1} with an average value of 0.82 mg L^{-1}. As expected from the t-test and ANOVA analysis, the overlap for both groups of lakes regarding this parameter was much greater; in fact the range of the phytoplankton-dominated lakes was totally included within the range of macrophye-dominated lakes.

Table 5.5 Summary table for TN (mg L^{-1}) values of lakes with macrophyte-dominance and phytoplankton-dominance.

Lake dominance	n	Median	Mean	SD	Minimum	Maximum
Macrophyte-dominated	9	0.60	0.73	0.34	0.37	1.26
Phytoplankton-dominated	25	0.87	0.82	0.19	0.41	1.13

As it can be seen from the analysis of frequencies (Figures 5.6 and 5.7), the distribution of frequencies of lake water TN concentration were opposite of those showed by TP and chlorophyll-α. Most of the macrophyte-dominated lakes were in the lower values of nitrogen concentration of the scale (skewed to the right). Most of the lakes in the phytoplankton-dominated group were toward the higher concentration of the range (skewed to the left). These results seems to support the hypothesis that submerged aquatic vegetation is associated with lower concentrations of nitrogen in the water column, however, the t-test and ANOVA analysis proved different.

Figure 5.1 TP in macrophyte-dominated lakes.

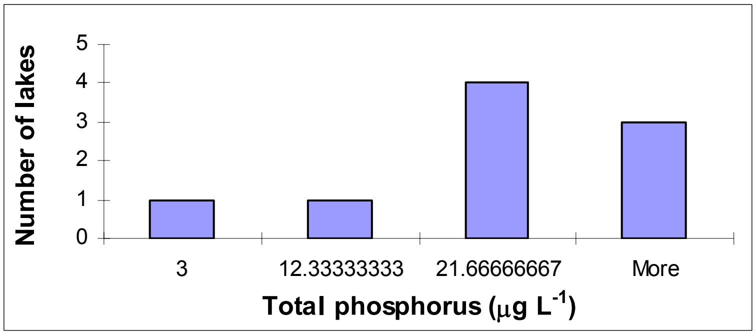


Figure 5.2 TP in phytoplankton-dominated lakes.

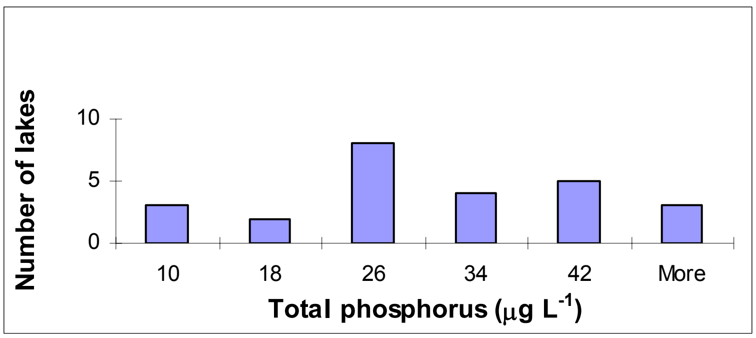


Figure 5.3 Chlorophyll-*a* in macrophyte-dominated lakes.

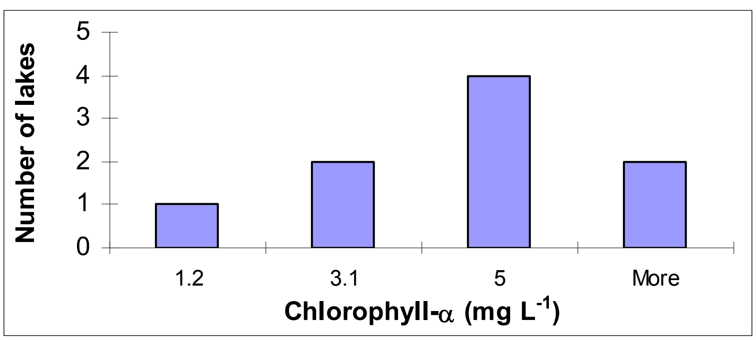


Figure 5.4 Chlorophyll-*a* in Phytoplankton-dominated lakes.

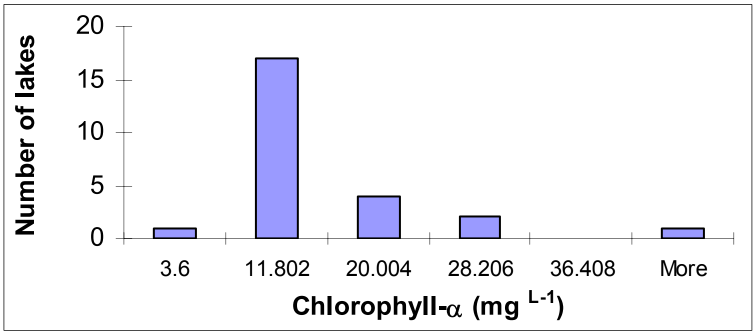


Figure 5.5 TN in macrophyte-dominated lakes.

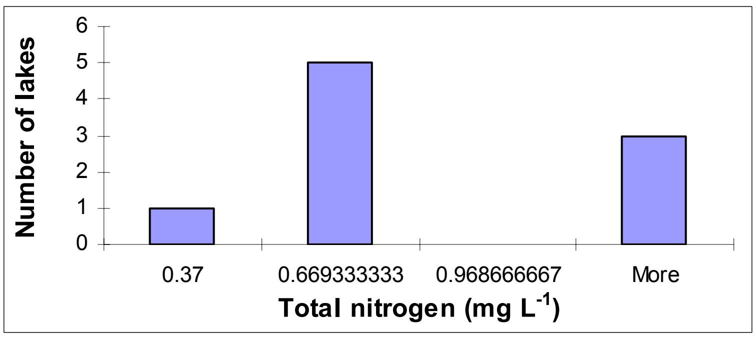
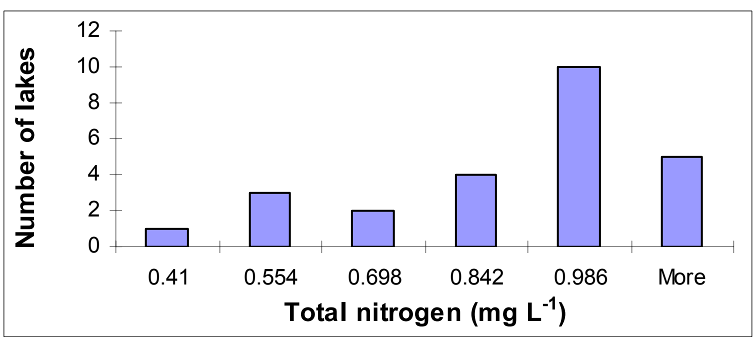


Figure 5.6 TN in Phytoplankton-dominated lakes.



Contrary to expected, the linear regression analysis for a possible relationship between the variability of lake water TP concentration (as expressed by CV) and submerged aquatic vegetation did not show any association for any of the two measures of submerged aquatic vegetation: PAC and PVI (Figures 5.8

and 5.9). This result, however, is not conclusive because of limitations caused by unavailability of repeated measures of submerged aquatic vegetation dominance for the lakes studied.

Figure 5.7 Variability of lake water total phosphorus with increment in area covered by vegetation.

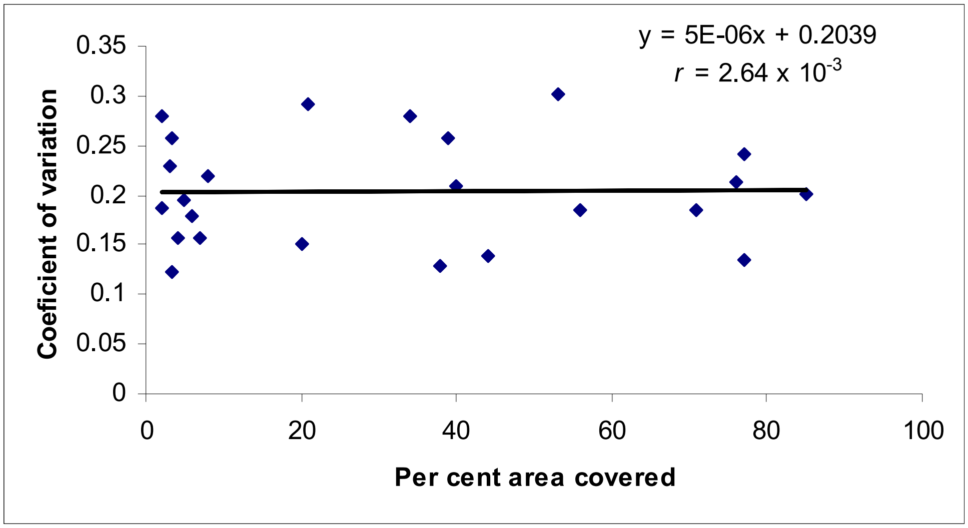
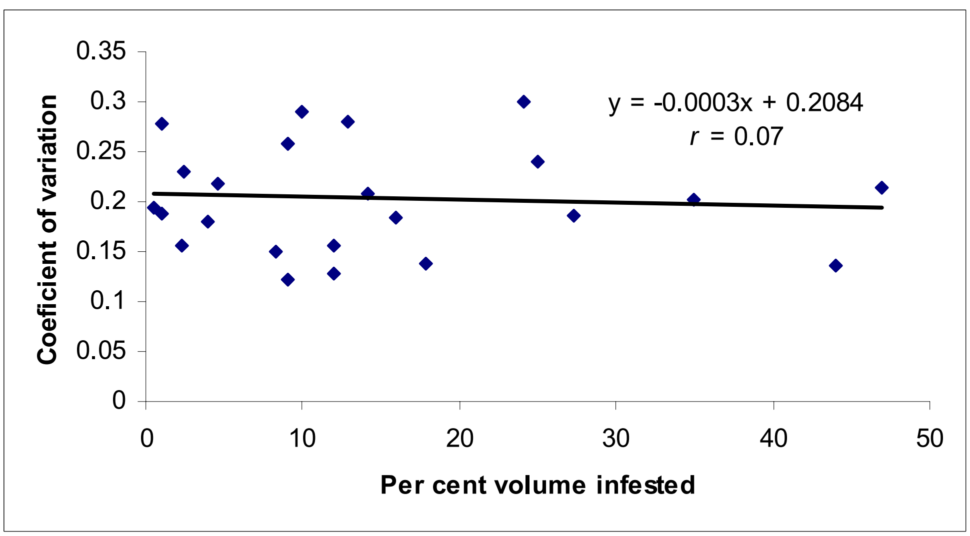


Figure 5.8 Variability of lake water total phosphorus with increment in volume of the lake infested by vegetation.



5.5 Summary

The analysis of t-test and one way ANOVA conducted in this chapter indicated that lakes dominated by macrophytes show a significantly lower lake water concentration of TP and chlorophyll-α as compared to those lakes dominated by phytoplankton. The same type of analysis did not show a significant difference between both groups of lakes in regard to TN concentration. This result confirms the strong correlation between submerged aquatic vegetation and lake water chlorophyll-α and lake water TP concentration discussed in Chapter 4. Furthermore, these results strongly suggest that at least under TP levels <50 μg L^{-1}, the association of submerged aquatic vegetation with concentrations of TP and chlorophyll-α did not extend to lake water TN concentrations.

This analysis did not show a buffering effect of submerged aquatic vegetation in the inter-annual variability of lake water TP concentration in urban lakes.

CHAPTER 6.

IMPLICATIONS OF RESEARCH

Eutrophication of lakes located in the Tampa Bay watershed show significant trends between 1990 and 2007. There are three main findings regarding these trends. First, the concentration of phosphorus (as TP) and phytoplankton (as measured by chlorophyll-α) increased over time for lakes classified as oligotrophic or mesotrophic. Second, in hypereutropic lakes phosphorous concentrations decreased with time and no significant trend was seen for chlorophyll-α. Third, the ratio of nitrogen to phosphorus (TN:TP) declined for lakes with both low and high levels of eutrophication. Historical and recent human settlement patterns and population growth in Hillsborough County, coupled with a karst geology, may have contributed to the observed increase in lake water phosphorus concentrations for oligotrophic and mesotrophic lakes. Lake management plans that have included reducing point and non-point source nutrient flows may be responsible for the declining water phosphorus concentrations in hypereutrophic lakes.

For many of the oligotrophic and mesotrophic lakes of the region a trend line suggests that in ~20 years primary productivity may be nitrogen-limited, but for hypereutrophic lakes, primary productivity is already nitrogen-limited, as suggested by TN: TP ratios $\leq$ 10. Nitrogen-limited primary productivity may have undesirable conseuences related to the increase in cyanobacterial populations (Hecky and Kilham, 1988; Levich, 1996; Levich and Bulgakov, 1992), and thus a threat thread to public health due to effects caused by toxins produced by this blue-green algae (Fleming et al., 2002; Karjalainen et al., 2007).

As population growth and development seem inevitable and the underlined karst formation is a permanent condition (in the case those factors play in fact a role in the eutrophication of these lakes), this study explored submerged aquatic vegetation as a possible factor (based on the literature read) in controlling eutrophication for lakes in the Tampa Bay watershed. Among a group of lakes composed mostly of those in the low eutrophication subgroup, submerged aquatic vegetation was found to be the most significant factor associated to eutrophication. The strongest variables associated to eutrophication as estimated by water concentration of chlorophyll-α, were (in order of significance), percentage of lake area covered with submerged aquatic vegetation (PAC), percentage of lake volume occupied by submerged aquatic vegetation (PVI), and concentration of phosphorus and nitrogen in lake water. Submerged aquatic vegetation (both expressions) also had the strongest association with lake water phosphorus concentration, followed by mean depth. When water nitrogen concentration was examined as dependent variable, there were no other variables significantly associated with it.

Hypothesis-testing revealed that phosphorus and chlorophyll-α concentrations were significantly higher for lakes with low coverage of submerged aquatic vegetation than for lakes with high coverage. These results support the theory that submerged aquatic vegetation is a strong candidate to be considered a controlling factor or at least a strong indicator of water quality.

Measuring the relationship between submerged aquatic vegetation and phosphorus levels, may in fact, be indirectly measuring the relationship between submerged aquatic vegetation and phytoplankton productivity. This is because phytoplankton productivity is mostly limited by water phosphorus concentration in this group of lakes. Hence the mechanisms discussed here as means by which submerged aquatic vegetation may influence water phosphorus concentration are indirectly those by which submerged aquatic vegetation may influence phytoplankton productivity as well. These mechanisms are: sedimentation of suspended total phosphorus, direct phosphorus uptake from the water column, provision of surfaces for periphyton and bacteria, and influences on ion exchange reactions via regulation of dissolved oxygen concentration and pH levels.

Phytoplankton productivity (as measured by chlorophyll-α), however, was found to be more strongly associated with submerged aquatic vegetation than with total phosphorus concentration. This may indicate that in addition to the indirect mechanisms mentioned above, submerged aquatic vegetation may exert also a more direct effect in phytoplankton productivity, for example, by sheltering zooplankton from fish, which increases zooplankton predation of phytoplankton.

Eutrophication and change of nutrient ratios in freshwater urban and suburban lakes may represent a threat to public health by promoting productivity of toxic algae. Oral, respiratory, and cutaneous exposure to toxins released by toxic algae can result in diseases of the nervous, gastrointestinal, and hepatic systems. Furthermore, alterations of environmental aesthetics and ecosystem balance are additional ways through which eutrophication can impact the human wellbeing, from a more holistic view of public health.

A growing body of research is still needed in order to determine key variables or factors that might be manipulated to control lake water eutrophication. Results of such research and monitoring programs will contribute to the formulation of effective management plans for sustainable conditions of human wellbeing.

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APPENDICES

Appendix A: General Information about Lakes Examined for Eutrophication Trends

Hillsborough County						
Lakes	**Area (ha)**	**Mean Depth (m)**	**Volume (m^3)**	**Latitude**	**Longitude**	**Watershed**
Thonotosassa	344	2.4	N/A	28^{O} 03' 39"	-82^{O} 16' 39"	Pemberton Creek
Keystone	175	3.3	5714175	28^{O} 07' 59"	-82^{O} 35' 24"	Brooker Creek
Magdalene	83	2.4	2385712	28^{O} 04' 55"	-82^{O} 28' 55"	Sweet Water Creek
Carroll	82	2.4	2044183	28^{O} 03' 04"	-82^{O} 29' 15"	Sweet Water Creek
Hiawatha	55	3.3	1873595	28^{O} 10' 10"	-82^{O} 34' 54"	Anclote river
Calm	47	2.7	1477849	28^{O} 08' 32"	-82^{O} 34' 54"	Brooker Creek
Armistead	14	2.7	347739	28^{O} 06' 04"	-80^{O} 33' 35"	Rocky/Brushy Creek
Deer	14	3.6	501064	28^{O} 10' 04"	-82^{O} 27' 45"	Rocky/Brushy Creek
Sunset	13	2.4	352140	28^{O} 08' 06"	-82^{O} 37' 32"	Brooker Creek
Keene	13	2.7	317223	28^{O} 08' 40"	-82^{O} 26' 53"	Cypres Creek
Crenshaw	12	1.5	42304	28^{O} 07' 33"	-82^{O} 29' 45"	rocky/Brushy Creek
Juanita	10	2.7	246887	28^{O} 07' 03"	-82^{O} 35' 20"	Brooker Creek
Dead Lady	1.2	0.9	2087	28^{O} 09' 18"	-82^{O} 34' 14"	Brooker Creek

Appendix A: (Continued)

Pinellas County						
Lakes	**Area (ha)**	**Mean Depth (m)**	**Volume (m^3)**	**Latitude**	**Longitude**	**Watershed**
Alligator	32	N/A	N/A	27^{O} 58’ 55”	-82^{O} 41’ 50”	Old Tampa Bay
Chautauqua	22	N/A	N/A	28^{O} 00’ 15”	-82^{O} 43’ 21”	Old Tampa Bay

Manatee County						
Lakes	**Area (ha)**	**Mean Depth (m)**	**Volume (m^3)**	**Latitude**	**Longitude**	**Watershed**
Ward Lake	103	N/A	N/A	27^{O} 25’ 40”	-82^{O} 29’ 09”	Manatee River

Polk County						
Lakes	**Area (ha)**	**Mean Depth (m)**	**Volume (m^3)**	**Latitude**	**Longitude**	**Watershed**
haunter	87	N/A	N/A	28^{O} 01’ 58”	-81^{O} 57’ 57”	Hillsborough River
Bonnet	32	N/A	N/A	28^{O} 02’ 51”	-81^{O} 58’ 36”	Hillsborough River
Beulah	7	N/A	N/A	28^{O} 02’ 26”	-81^{O} 58’ 06”	Hillsborough River

Appendix B: One Time Values of Variables of Lakes Examined for Submerged Vegetation

Lake	PAC %	PIV %	TP (µg L^{-1})	TN (mg L^{-1}	Chlorophyll-α (µg L^{-1})	Area (ha)	Mean Depth (m)	Volume (M^3)
Alice	85	41	19	0.37	1.20	37.23	2.74	941847
Carroll	85	35	23	0.45	1.40	81.75	2.44	2044183
White Trout	77	44	14	0.60	3.20	30.35	3.35	1011391
Reinheimer	77	25	12	1.20	3.80	8.09	1.83	236600
Magdalene	76	47	14	1.07	3.80	83.37	2.44	2385713
Eckles	71	27.3	30	1.27	5.40	11.33	2.13	256854
Mound	69	32	20	0.52	2.50	30.35	3.96	1280673
Raleigh	67	23	3	0.60	6.90	9.71	2.74	254667
George	63	33	31	0.50	4.20	10.93	3.66	378654
Cypress	56	16	10	0.54	4.40	6.48	3.66	225021
Round	56	17.1	21	0.45	3.60	4.05	2.74	99847
Horse	46	19	21	0.89	3.80	10.93	2.13	146282
Rogers	44	13	17	0.95	14.40	38.04	2.44	746805
Pine	44	17.9	40	0.99	8.60	3.24	2.44	555292
Noreast	40	14.1	27	0.72	9.70	3.24	1.52	87071
Calm	39	9	22	0.41	4.00	46.54	3.35	1477849
Island Ford	38	12	25	0.87	8.80	36.02	3.05	1131957
Keystone	38	12	25	1.13	3.70	174.43	3.35	5714176
Crescent	35	10	35	0.94		18.21	2.74	553353
Dead Lady	34	12.9	50	0.94	6.60	1.21	0.91	2087
Elizabeth	30	10	24	0.86	6.30	7.69	3.66	272512
Taylor 2	30	13	12	0.64	6.00	19.02	2.74	543649
Rainbow	26	9	10	0.77	8.40	19.02	2.74	544936
Juanita	21	10	10	0.99	11.40	9.71	2.74	246887
Crenshaw	20	8.2	22	0.83	11.40	12.14	1.52	42304
Church	15	4.6	26	0.52	5.90	25.09	1.22	138039
Cedar East	8	4.6	33	0.61	6.60	1.21	1.52	33639
Armistead	7	12	45	1.09	21.30	13.76	2.74	347739
Rock	6	4	35	0.91	21.70	21.45	2.13	431011
Brant	5	0.5	35	0.93	9.00	22.26	1.83	384649
Saddleback	3.5	9	27	1.08	11.40	12.55	1.52	226065
Cedar West	3	2.4	41	0.78	17.30	2.02	1.83	18916
Pretty	2	1	33	0.95	11.90	32.78	3.35	1068395
Josephine	2	1	44	0.85	12.00	20.24	2.13	422013

Appendix B: (Continued)

Lake	Latitude	Longitude	Watershed
Alice	28°07'56"	-82° 36 ' 14 "	Brooker Creek
Carroll	28° 03 ' 04 "	-82° 29 ' 15 "	Sweetwater Creek
White Trout	28° 02 ' 21 "	-82° 29 ' 46 "	Sweetwater Creek
Reinheimer	28° 07 ' 48 "	-82° 29 ' 12 "	Rocky/Brushy Creek
Magdalene	28° 04 ' 55 "	-82° 28 ' 55 "	Sweetwater Creek
Eckles	28° 03 ' 19 "	-82° 28 ' 19 "	City of Tampa
Mound	28° 08 ' 51 "	-82° 34 ' 19 "	Brooker Creek
Raleigh	28° 06 ' 21 "	-82° 35 ' 02 "	Brooker Creek
George	28° 04 ' 07 "	-82° 29 ' 14 "	Sweetwater Creek
Cypress	28° 07 ' 32 "	-82° 33 ' 52 "	Rocky/Brushy Creek
Round	28° 07 ' 14 "	-82° 30 ' 00 "	Rocky/Brushy Creek
Horse	28° 06 ' 38 "	-82° 34 ' 44 "	Brooker Creek
Rogers	28° 06 ' 32 "	-82° 35 ' 19 "	Brooker Creek
Pine	28° 03 ' 38 "	-82° 28 ' 20 "	Curiosity Creek
Noreast	28° 03 ' 45 "	-82° 28 ' 07 "	Curiosity Creek
Calm	28° 08 ' 32 "	-82° 34 ' 54 "	Brooker Creek
Island Ford	28° 09 ' 08 "	-82° 35 ' 56 "	Brooker Creek
Keystone	28° 07 ' 59 "	-82° 35 ' 24 "	Brooker Creek
Crescent	28° 09 ' 29 "	-82° 35 ' 31 "	Brooker Creek
Dead Lady	28° 09 ' 18 "	-82° 34 ' 14 "	Brooker Creek

Appendix B: (Continued)

Lake	Latitude	Longitude	Watershed
Elizabeth	28° 09 ' 26 "	-82° 34 ' 24 "	Brooker Creek
Taylor 2	28° 08 ' 12 "	-82° 36 ' 43 "	Brooker Creek
Rainbow	28° 07 ' 00 "	-82° 35 ' 46 "	Brooker Creek
Juanita	28° 07 ' 03 "	-82° 35 ' 20 "	Brooker Creek
Crenshaw	28° 07 ' 33 "	-82° 29 ' 45 "	Rocky/Brushy Creek
Church	28° 06 ' 11 "	-82° 35 ' 58 "	Brooker Creek
Cedar East	28° 03 ' 56 "	-82° 28 ' 13 "	Curiosity Creek
Armistead	28° 06 ' 04 "	-82° 33 ' 35 "	Rocky/Brushy Creek
Rock	28° 06 ' 48 "	-82° 33 ' 24 "	Rocky/Brushy Creek
Brant	28° 07 ' 35 "	-82° 28 ' 20 "	Rocky/Brushy Creek
Saddleback	28° 07 ' 13 "	-82° 29 ' 41 "	Rocky/Brushy Creek
Cedar West	28° 03 ' 55 "	-82° 28 ' 21 "	Curiosity Creek
Pretty	28° 06 ' 27 "	-82° 34 ' 04 "	Rocky/Brushy Creek
Josephine	28° 06 ' 35 "	-82° 33 ' 43 "	Rocky/Brushy Creek

ABOUT THE AUTHOR

Max Jacobo Moreno Madriñan received his Bachelor's degree in Agriculture Science in 1993 from the Universidad Nacional de Colombia, in Colombia. He received his Master's degree in Environmental Management for Sustainable Development with Emphasis on Coastal Zones from the Pontificia Universidad Javeriana, in Colombia.

During his doctoral studies at University of South Florida, he has presented the results of his research at three regional meetings of the Air and Waste Management Association and at one meeting of the Implementation and Integration Workshop of the Gulf of Mexico Alliance. He received two awards for his research studies.