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Impact of urban runoff on phosphorus, nitrogen, and dissolved oxygen in a shallow subtropical lake

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IMPACT OF URBAN RUNOFF ON PHOSPHORUS, NITROGEN, AND DISSOLVED OXYGEN IN A SHALLOW SUBTROPICAL LAKE

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ABSTRACT

This research assessed the current conditions of water quality in a shallow subtropical lake, influenced by a highly developed urban environment. Specifically, the research focused on the dynamics of phosphorus, nitrogen, and dissolved oxygen, as well as the effects of stormwater runoff on lake water quality. Furthermore, the research quantified gross primary production, net primary production, and respiration in order to discern seasonal variation in lake metabolism. A water quality monitoring platform with multi-parameter probes was deployed in the center of University Lake on the Louisiana State University campus. A series of lake water quality parameters including dissolved oxygen (DO) concentration, temperature, pH, specific conductivity, and cyanobacteria were recorded continuously at a 15-min interval from July 2008 to October 2009. In addition, water samples were collected monthly as well as after major rain events during the study period to determine changes in total phosphorus (TP), nitrate (NO_3^-), nitrite (NO_2^-), and total Kjeldahl nitrogen (TKN) concentrations. Results show a monthly average concentration of 0.286 mg/L TP (min – max: 0.167 - 0.621 mg/L), 0.053 mg/L NO_3-N (below detection – 0.24 mg/L), 0.045 mg/L NO_2-N (below detection – 0.012 mg/L), and 2.5 mg/L TKN (below detection – 5.12 mg/L). Mean storm event loading of 28.1 kg (7.5 – 47.8 kg) caused an immediate increase in total phosphorus within the water column by 14.1%. This resulted in a mean TP lake concentration of 0.383 mg/L. A similar trend was observed in a lesser degree in NO_3-N and TKN. Phosphorus loads in the lake were correlated with runoff volume ($r^2 = 0.71$), suggesting runoff volume is the most important factor effecting lake TP concentration after a storm event. Analysis of metabolism rates found a mean annual gross primary productivity value of 4.41 g O_2/m^2/day, a mean annual net primary production value of 2.13 g

O_2/m^2/day, and a mean annual respiration value of 5.90g O_2/m^2/day. Annually, 1610 g O_2/m^2 were produced while the annual sum of respiration was 2150 g O_2/m^2. Respiration rates were mostly equal to or greater than productivity rates throughout the year, indicating that this shallow subtropical urban lake was net heterotrophic throughout most of the year.

CHAPTER 1: INTRODUCTION

Today as a result of population increase and pollution on a global scale, water has become a more valuable resource. Although conditions appear to be improving, about 44% of all lakes, and about 59% of man-made lakes within the United States are still in fair or poor biological condition (USEPA, 2009). Lake and stream water quality is likely to become even more critical in the future as global climate change progresses. One of the first systems expected to be affected by climate change is river and lake systems (Whitehead *et al.*, 2009). Increases in air temperature will affect hydrologic aspects of lake ecosystems including flow patterns and evaporation. An increase in water temperature will affect chemical reaction kinetics as well as bacteriological processes within aquatic systems (Whitehead *et al.,* 2009). Because temperature and DO are highly correlated, an increase in temperature could have a drastic effect on freshwater ecosystems.

Increased consumption of fossil fuel and fertilizers and rapid change of land use have increased the active quantity of nutrients in global biogeochemical cycles. Lake eutrophication, the process of an ecosystem becoming more productive by nutrient enrichment, is a widely recognized problem for waterbodies throughout the world, which can cause algal blooms, fish kills and pungent smells. Nutrient enrichment in urban lakes and reservoirs, namely phosphorus and nitrogen, has increased on a large scale due to fertilizer use and municipal and industrial wastewater (Ryther and Dunstan, 1971; Tilman *et al*., 2001; Conley *et al*., 2009).

Rain events, in an urban environment, result in surface runoff that transports nutrients, sediment, and organic material that affect the biologic communities and the water quality of a waterbody (Wang *et al.,* 2003). Urban areas tend to have a large percentage of impervious

surface cover which alters the hydrology and geomorphology of drainage systems, including groundwater recharge and discharge. This results in decreased infiltration causing an increase in runoff and rate of travel. Urban runoff pollution tends to be difficult to control due to the large variety of pollutant sources and variable nature of source loadings due to the intermittent nature of precipitation and runoff (Li *et al*., 2007).

To better understand the effects of stormwater runoff on urban lake water quality, a comprehensive research project was conducted in 2008-2009. This thesis utilized data collected between July 2008 and December 2009 to achieve the following goals: (1) Quantify runoff effect on phosphorus and nitrogen loading, and its relationship to water quality parameters. (2) Determine runoff effect on changes of phosphorus, nitrogen, and dissolved oxygen concentrations. (3) Analyze seasonal variation of water quality parameters. (4) Assess lake metabolism processes in a shallow urban ecosystem using intensive dissolved oxygen measurements.

To achieve the objectives we deployed an Environmental Monitoring Buoy (EMB) in University Lake on the Louisiana State University Campus (Figure 1.1a), a 76-ha shallow lake that is surrounded by the highly developed residential areas. The monitoring program was part of an enhancement project to strengthen the capability of higher education and research in water quality through utilization of cutting-edge technology in the classroom (Figure 1.1b) and to increase awareness of urban environment problems and encourage community engagement in water quality protection. In addition to the EMB, lake volume was measured through the use of pressure transducers (Figure 1.1c) and, water samples were collected to analyze nutrient concentrations (Figure 1.1d).

Figure 1.1. (clockwise from top left): (a) Environmental monitoring buoy (EMB) with multi-probe sensors deployed in University Lake on the LSU campus; (b) Use of real-time data from the EMB system in the classroom; (c) Pressure transducer installed on University Lake to measure water depth and volume; (d) Water samples for nutrient analysis were collected using a kayak throughout the study period.

This thesis is divided into six chapters. Chapter 2 provides a literature review emphasizing the current status of knowledge on urban waterbodies, stormwater runoff, and nutrient availability. Chapter 3 presents the study on rainstorm effects on phosphorus transport to a subtropical urban lake. Chapter 4 presents the study of rainstorm effects on nitrogen transport to a subtropical urban lake. Chapter 5 discusses dissolved oxygen, temperature, and metabolic rates in a shallow subtropical urban lake. Chapters 3[*], 4, 5 are written as stand-alone journal publications. They each have their own introduction, methods, and results and discussion sections, and for that reason there will be some repetition among the chapters.

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CHAPTER 2: LITERATURE REVIEW

The scientific study of lake environments, also known as limnology, was initiated in the late 19th century through the work of Professor F. A. Forel. In its early development limnology dealt mainly with the geology, chemistry, and physics of lakes. Today this is known as forelian limnology. Through the work of others like G. E. Hutchinson limnology has truly become an interdisciplinary approach to understanding freshwater ecology and the interaction between chemical, physical, and biological processes (Dodds, 2002). The study of lake environments is continuously changing as our knowledge progresses. Today much of the research pertains to nutrient availability and the factors that affect nutrient cycles.

Freshwater pollution was not regulated in the United States until the introduction of the "Clean Water Act of 1972." The federal law focused on the control of wastewater and point source pollution management. Since then nonpoint source pollution has been recognized as a crucial contributor to water pollution (USEPA, 2009; Hoorman et al., 2008; Lathrop, 2007; Peters, 2009; Wang et. al., 2003). Nonpoint source pollution, which occurs from diffuse sources, includes runoff from agricultural or urban landscapes and can include nutrient, sediment, or pathogen pollutants. As a result of nonpoint source pollution not being limited to a single source, it is more difficult to control.

Waterbodies, especially lakes and ponds, are regarded as nutrient sinks because they are more vulnerable to the accumulation of pollutants and nutrients both in its water column and in its organisms than a stream or river. Phosphorus and nitrogen are important because they are fundamental drivers of lake primary production, and an excess of these nutrients can degrade

lakes through eutrophication. Both nutrients are found in fertilizers, organic matter such as detritus, and sewage.

Many variables are dependent upon lake depth, which is an important characteristic of a lake environment. These characteristics can substantially make a shallow lake distinct from deeper lakes (Petaloti *et al.*, 2004). The entire water column may be within the photic zone depending on the depth of the lake and the turbidity of the water. Shallow lakes usually support highly diverse biota due to more extensive littoral zone (Arora and Mehra, 2009). The lack of depth can also prevent the water column from becoming thermally stratified. Wind can also cause the resuspension of sediment and consequently provide a considerable portion of lake nutrients (Petaloti *et al.*, 2004). For these reasons, chemical and physical parameters of shallow lakes are highly variable.

2.1 Lake Eutrophication

Under natural conditions, the productivity of a lake increases as it ages. This process, also known as eutrophication, has been greatly accelerated by human influence over the past century. Increased consumption of fossil fuel and fertilizers and rapid change of land use have increased the active quantity of nutrients in global biogeochemical cycles. Today, lake eutrophication is a widely recognized problem for many waterbodies throughout the world, which can cause algal blooms, fish kills and pungent smells (Figure 2.1). Nutrient enrichment in urban lakes and reservoirs, namely phosphorus and nitrogen, has increased on a large scale due to fertilizer use and municipal and industrial wastewater (Ryther and Dunstan, 1971; Tilman *et al*., 2001; Conley *et al*., 2009). The reduction of nutrient input has been widely considered necessary to constrain lake eutrophication. However, the cost of reducing nutrient loads into waterways can be substantial, especially when it is still debated whether a reduction in phosphorus or a reduction in

both nitrogen and phosphorus is needed to produce results (Lathrop, 2007; Dodds *et al*., 2002; Conley *et al*., 2009; Schindler and Hecky, 2009).

Figure 2.1. Algal bloom in College Lake within the University Lake system on 5/28/2009.

Since eutrophication is increased nutrient input, any activities within the watershed of a lake that add nutrients causes eutrophication. This includes point and nonpoint source pollution resulting from both natural as well as anthropogenic events. Stormwater runoff from urban areas has been seen as a major source of nutrients, including nitrogen, phosphorus, and dissolved and suspended solids, for many urban lakes.

2.2 Urban Waterbodies

Rain events in an urban environment result in surface runoff (or urban runoff) that transports nutrients, sediment, and organic material that affect the biologic community and water quality of a waterbody (Wang *et al.,* 2003). Urban areas tend to have a large percentage of impervious surface cover which alters the hydrology and geomorphology of drainage systems, including groundwater recharge and discharge. This results in decreased infiltration causing an

increase in runoff and rate of travel. This is in fact the design of many stormwater management systems.

Urban runoff pollution tends to be difficult to control due to the large variety of pollutant sources and variable nature of source loadings due to the intermittent nature of precipitation and runoff (Li *et al*., 2007). For instance in an urban environment pollution loads do not tend to be proportionate to runoff volume. Urban environments tend to exhibit a first flush (Li *et al*., 2007; Rosenweig *et al*., 2008), which is when a disproportionate quantity of pollutant is transported in the early stages of a rain storm. Li et al. (2007) found that up to 63% of total phosphorus (TP) can be transported in the first 30% of runoff. Whether a first flush occurs and its characteristics depend on factors such as rainfall intensity, impervious area, watershed area, and antecedent conditions (Li *et al*., 2007). It is suggested that larger first flushes are associated with greater intensity rain events. The pollution load also tends to be larger the longer antecedent conditions are dry (Li *et al*., 2007). Not only do storm events effect pollution values within a lake environment, but they also effect temperature and dissolved oxygen concentrations by breaking down thermoclines (MacKinnon and Herbert, 1996).

Various common land uses in urban environments tend to increase nutrient loads. Municipal and industrial discharges, which can occur in urban environments, increase nutrient loads (Whitehead *et al.*, 2009). Construction sites, also common (Figure 2.1), can cause erosion rates to approach 50,000 metric tons/km^2/yr compared to 1,000 - 4,000 metric tons/km^2/yr in agricultural, and <100 metric tons/km^2/yr in undisturbed environments (Carpenter *et al.*, 1998). Fertilized turf, which is found in residential neighborhoods and on golf courses, has been shown to increase nitrogen and phosphorus in waterbodies (King *et al.*, 2007).

Figure 2.2. Construction site located adjacent to the shoreline of University Lake on 5/3/2009.

2.3 Phosphorus of Lake Waters

Phosphorus is a key nutrient in aquatic ecosystems and one of the most essential elements for all life forms on earth. The element is essential for primary production and is required for many metabolic processes in plants and animals. Organic phosphorus is present in living plants and animals, their by-products, and their residues. When in waterbodies the element can either be in the form of dissolved organic phosphorus (DOP) of particulate phosphorus (PP). However, in surface waters, phosphorus is often present as phosphate (PO_4) phosphorus. Typically, phosphorus is measured as total phosphorus (TP) which is the summation of DOP, PP, and PO_4. Generally, without phosphorus, as with nitrogen, aquatic plant growth will cease, regardless the availability of other nutrients.

Phosphorus is related to the effects of both urban and industrial land use (Huang *et al.*, 2007). Total phosphorus concentrations provide an indication of eutrophication by correlating phosphorus concentrations with chlorophyll *a*, a common measurement of algal biomass (Ruley and Rusch, 2004). Phosphorus concentrations have been correlated to chlorophyll in both

streams (Dodds *et al*., 2002) and lakes (Smith, 1982; Ruley and Rusch, 2004). Regressions tend to explain much more variability in lakes where r^2 values commonly exceed 90% (Dodds *et al*., 2002). However, many of these studies occurred in northern and northeastern United States lakes (Ruley and Rusch, 2002). Phosphorus and chlorophyll relationships tend to display a discernable breakpoint above which chlorophyll values are substantially greater. For mean total phosphorus Dodds *et al.* (2002) reported a break point occurred at about 30 μg/L, while greater chlorophyll concentrations are observed at lower latitudes and higher temperatures.

Internal phosphorus loading has shown to be an important factor for regulating eutrophication in many shallow lakes (Ramm and Scheps, 1997; Reddy *et al*., 2007). Internal phosphorus loading is the release of phosphorus from lake sediments into the water column. Internal loading is known to prevent the recovery of eutrophic lakes after external loadings have decreased (Reddy *et al*., 2007; Carpenter and Lathrop, 2008). In shallow eutrophic lakes internal loading can occur through several different methods. Loading has been correlated to sediment resuspension which is mainly the result of wave activity from wind (Chung *et al*., 2008; Holmroos *et al.*, 2009). In addition iron (Fe) largely controls internal phosphorus loading in lakes that exhibit hypoxia in the hypolimnion as iron sulfide (FeS) and iron disulfide (FeS_2) form under reduced conditions thus creating greater inputs due to internal loading during the summer months (Pensa and Chambers, 2004; Conley *et al*., 2009). As a result internal loading can exceed external loading during the summer months (Carpenter and Lathrop, 2008). Concentrations are then distributed through wind action and diffusion. Diffusion occurs if the concentration of phosphorus exceeds that of overlying water. If external loading rates are below a certain rate then sediments function as a net source of phosphorus, if rates are above the threshold sediments function as net phosphorus sinks (Reddy *et al*., 2007). Long term studies in Europe have shown

that internal loading decreases slowly after external sources are controlled. Short term studies do not take into account adaptive processes, and therefore there is a need for long term data (Schindler and Hecky, 2009).

2.4 Nitrogen of Lake Waters

Like phosphorus, nitrogen is a key nutrient in aquatic ecosystems and essential for primary production and for many metabolic processes in plants and animals. Although the most common form of nitrogen on earth, nitrogen gas (N_2), is abundant, it is difficult for most to use directly, the exception being nitrogen fixing bacteria such as cyanobacteria. Therefore, organic and inorganic nitrogen are common forms in aquatic ecosystems. Organic nitrogen includes amino acids, nucleic acids, proteins, and urea. Inorganic forms are more commonly measured in aquatic ecosystems and include nitrate (NO_3), nitrite (NO_2), and ammonium (NH_4). The summation of these three forms is commonly referred to as dissolved inorganic nitrogen (DIN). Like phosphorus, generally without the presence of nitrogen in aquatic ecosystems aquatic plant growth will cease, no matter the availability of other nutrients.

The export of nitrogen in stormwater runoff poses a threat to the health of the environment and humans, along with consuming a large share of public resources (NRC, 2001). It is important to analyze the interactions different parameters have on each other. For example, an increase in DO has been correlated to an increase of NO_3-N due to aerobic conditions being favored for rapid transformation of NH_3-N to NO_3-N (Dalkiran *et al.,* 2006). In urban areas like the proposed study site, nitrogen loading is more a function of processes that affect concentrations than of the catchment to convey precipitation as runoff (Lewis and Grimm, 2007). During rain events nitrogen is mainly transported in the form of nitrate. This is in part due to the fact that the more a nutrient is dependant on solid related particles, the more the concentration is

dependent on rainfall intensity (Budai and Clement, 2007). Therefore, streams in undistrubed forested environments show low nitrogen concentrations, while streams in agricultural and highly urbanized watersheds often display a high nitrogen concentration due to fertiler application and disturbance. For instance, a study in the Chesapeake Bay watershed has shown that more urbanized sites can export total nitrogen and nitrate at greater, less frequent flows (Shields *et al.*, 2008). That being said loads tend to be larger in areas with a greater percentage of impervious surface and commercial land use (Lewis and Grimm, 2007). Also, in urban areas it is important to note that there is seasonal variation in the concentrations of nitrogen (Li *et al.*, 2009; Kane *et al.*, 2008).

Both nitrogen and phosphorus are regarded as being the primary limiting factor for primary production. However, it is still being debated whether a reduction in both nitrogen and phosphorus, or just a reduction in phosphorus is needed to control eutrophication (Elser *et al.*, 1990; Lathrop, 2007; Dodds *et al*., 2002; Conley *et al*., 2009; Schindler and Hecky, 2009).

2.5 Dissolved Oxygen of Lake Waters

A crucial component of both organic and inorganic compounds, oxygen is representative of an ecosystems functionality and behavior, and as a result is regarded as an important indicator of the general health of waterbodies (Dodds, 2002). Dissolved oxygen (DO) is a crucial component in aquatic communities due to being involved in all metabolic processes. DO is also relatively easy to measure via classic chemical methods, or electrochemical techniques with sensors (D'Autilia *et al.*, 2004). Oxygen solubility in waters is negatively related to temperature but positively related to its partial pressure. This results in temperature and atmospheric pressure playing an important role controlling the saturation of oxygen in waterbodies. Temporal

concentrations of dissolved oxygen in water can be the result of several factors including metabolic activity rates, diffusion, temperature, and proximity to the atmosphere (Dodds, 2002). Pollution values play a critical factor in controlling the solubility of a waterbody because they tend to inversely affects solubility. In addition, wind can play an important role in atmosphere-water oxygen flux. Wind has shown to have a major impact on the magnitude and direction of oxygen flux at diel, day to day, and seasonal time scales in a hypereutrophic lake (Gelda and Effler, 2002).

Dynamic chemical, physical, and biological events can result in dissolved oxygen displaying an extremely irregular pattern over all time scales (D'Autilia *et al.*, 2004; Gelda and Effler, 2002). These factors can influence the magnitude and direction of oxygen flux in the water column (Gelda and Effler, 2002). On an annual time scale, respiration is an important factor throughout the year. During the winter DO has been shown to be mainly controlled by physical re-aeration due to lower temperatures, while during the summer months photosynthesis, driven by strong solar radiation, determines the release of considerable amounts of oxygen in the water (D'Autilia *et al.,* 2004). This results in the availability of DO during the winter months being distributed throughout the day, while during the summer oxygen is primarily available in greater concentrations during the sunlight hours. This can possibly lead to anoxic conditions, typically during the night, which can result in fish kills. The timing of metabolic processes from biotic communities can also impact the DO concentration at different time scales (D'Autilia *et al.,* 2004). Algal blooms cause dramatic fluctuations in DO which can be intensified by pollution and low-flow conditions (Whitehead *et al.,* 2009). Rain events resulting in storm runoff tend to cause hydrodynamic mixing which disrupts diurnal DO pattern of the water body and create oxygen profiles atypical of stratified lakes (Cornell and Klarer, 2008; McTammany *et al.,* 2003).

Therefore during the rainy season, rain events can further add to the variability of lake DO concentrations.

In many regions of the world over the course of a year, dissolved oxygen has shown peak values being observed during the coldest months of the year and the lowest values during the warmer seasons (Hull *et al.,* 2008). Supersaturation has been observed to occur early in spring after seasonal stratification occurs, followed by a peak in GPP during the spring. DO concentrations tend to drop during the summer due to increased temperatures and respiration. The drop in DO can continue throughout summer most likely resulting from a loss of nutrients and a reduction in solar radiation (MacKinnon and Herbert, 1996). On a diel time scale, high values have been observed during the most irradiated hours of the day or during the early evening hours, while low values are observed shortly before dawn (Hull *et al.,* 2008). This diel fluctuation in DO is well known to be the results of biologic activity and not temperature (Whitney, 1942). Sub-daily time periods continue to be an area that requires more research.

Many factors affect the intensity and direction of oxygen flux between air and water oxygen values. The difference between the DO concentration at saturation and the current DO concentration, the global transfer coefficient (KLA), which is specific for each body of water, wind speed, salinity or natural substances, and probably air moisture affect oxygen flux (Ginot and Herve, 1994). Models of photosynthesis and respiration tend to be more accurate when they include temperature.

Wind is one of the most important factors affecting dissolved oxygen concentrations within lakes (D'Autilia *et al.,* 2004; Hull *et al.,* 2008). Not only is it an important factor that controls reaeration from the atmosphere but it also physically mixes the waterbody, diffusing oxygen concentrations and temperature gradients (D'Autilia *et al.,* 2004; Nerini *et al.,* 2000).

High winds have been shown to have a greater impact on oxygen flux than departures from equilibrium conditions (Gelda and Effler, 2002) or salinity (Nerini *et al.*, 2000). Wind has also been shown to supersaturate water in aquatic systems, under laboratory conditions, when wind speeds average 2.5 m/s (Hull *et al.*, 2008).

The determination of dissolved oxygen is the basis of the biochemical oxygen demand test (BOD). BOD is commonly defined as the amount of oxygen utilized by bacteria while stabilizing decomposable organic matter under aerobic conditions. It is used to determine the pollutional strength of domestic and industrial waste in terms of oxygen depletion. BOD is performed by measuring the concentration of dissolved oxygen in water over time. BOD is positively correlated with temperature and has been shown to be greater during the spring and summer months (MacPherson *et al.*, 2007).

Dissolved oxygen is highly dependent on temperature. As temperature increases, DO saturation decreases leading to less oxygen availability. Algal growth rates will also increase due to the increase in temperature and therefore an increase in respiration (Whitehead et al., 2009). These reasons all seem to put shallow lakes at the greatest risk of being affected by rises in temperature, whether the result of thermal pollution or rises in temperature resulting from global warming.

2.6 Metabolic Processes of Aquatic Ecosystems

The two main metabolic processes in aquatic environments are photosynthesis and respiration. Respiration utilizes oxygen within the water, and can be a serious area of concern when a waterbody is polluted with excess nutrient such as nitrogen and phosphorus. These nutrients cause an increase in productivity and consequently an increase in respiration and photosynthesis. However, photosynthesis does not occur at greater depths or at night, causing

oxygen values to drop. Respiration dynamics have been shown to be variable and dependent upon the season (McTammany *et al.*, 2003). Therefore, it is currently hard to predict if and when a waterbody will become anoxic and can no longer support animal life.

The daily maximum dissolved oxygen deficit was highly correlated with daily rates of respiration (Mulholland *et al.,* 2005). Photosynthesis is the process responsible for the production of organic material. The rate at which biomass is produced by an ecosystem's producers (GPP) has been correlated to the amplitude of the diurnal DO deficit. However, disturbance, which is common in urban areas, causes a sharp decline of these indicators of stream metabolism (Mulholland *et al.*, 2005).

Wang et al. (2003) found urban streams to be heterotrophic throughout their entire study period. Many methods have been used to determine metabolism (Chapra and Di Toro, 1991, Cornell and Klarer, 2008, McBride and Chapra, 2005, McTammany *et al.,* 2003, Mulholland *et al*., 2005, Wang *et al*., 2003). Diurnal profiles have been found to be a fairly simple and valuable method used to calculate *in-situ* rates of metabolism (Mulholland *et al.,* 2005).

Previous short term measurements have depicted a double peak in primary production due to a slowdown in production (D'Autilia *et al.*, 2004). This has been attributed to a process known as photorespiration, a light-dependent process that develops carbon dioxide while consuming molecular oxygen (Hull *et al.*, 2008). Primary production slows down during irradiated summer days while water is highly oxygenated (Hull *et al.*, 2008). On a seasonal time scale a spring peak in primary production is typically observed in temperate lakes (MacKinnon and Herbert, 1996).

CHAPTER 3: RAINSTORM EFFECTS ON PHOSPHORUS TRANSPORT TO A SUBTROPICAL URBAN LAKE

3.1 Introduction

Lake eutrophication is a widely recognized problem for waterbodies throughout the world, which can cause algal blooms, fish kills and pungent smells. Nutrient enrichment in urban lakes and reservoirs, namely phosphorus (P) and nitrogen, has increased on a large scale due to fertilizer use and municipal and industrial wastewater (Ryther and Dunstan, 1971; Shindler, 1974; Tilman *et al*., 2001; Conley *et al*., 2009). The reduction of nutrient input has been widely considered necessary to constrain lake eutrophication. However, the cost of reducing nutrient loads into waterways can be substantial especially when it is still debated whether a reduction in phosphorus or a reduction in both nitrogen and phosphorus is needed to produce results (Dodds *et al*., 2002; Lathrop, 2007; Conley *et al*., 2009; Schindler and Hecky, 2009).

Nutrients are transported from a landscape into a waterbody through surface runoff. As a result, phosphorus has been shown to increase in a lake due to the effects of both urban and industrial land use in an urbanizing watershed (Huang *et al*., 2007). Urban runoff pollution is difficult to control because of the large variety of pollutant sources and intermittent nature of source loading due to the nature of precipitation and runoff (Li *et al*., 2007). For instance, in an urban environment, pollution loads are not always proportionate to runoff volume, due to what is known as “first flush” (Li *et al*., 2007; Rosenweig *et al*., 2008). First flush occurs when a disproportionate quantity of pollutant is transported in the early stages of a rain storm. Li *et al.*, (2007) found that up to 63% of total phosphorus (TP) can be transported in the first 30% of runoff. Whether a first flush occurs depends on factors such as rainfall intensity, impervious

area, watershed area, and antecedent conditions (Li *et al*., 2007). It is suggested that larger first flushes are associated with more intense rain events. Also, the pollution load tends to be larger the longer antecedent conditions are dry (Li *et al*., 2007).

Internal P loading has shown to be an important factor for regulating eutrophication in many shallow lakes (Ramm and Scheps, 1997; Reddy *et al*., 2007,). Internal P loading is the release of P from lake sediments into the water column. Internal loading is known to prevent the recovery of eutrophic lakes after external loadings have decreased (Reddy *et al*., 2007; Carpenter and Lathrop, 2008). Internal loading can occur both mechanically or chemically. Loading occurs mechanically when sediment is resuspended, which can occur through wave activity resulting from wind (Chung *et al*., 2008; Holmroos *et al.,* 2009). Chemically, iron largely controls internal phosphorus loading in lakes that exhibit hypoxia in the hypolimnion. Iron sulfate and iron sulfide form under reduced conditions thus creating greater phosphorus concentrations, due to internal loading, during the summer months (Pensa and Chambers, 2004; Conley *et al*., 2009). This can result in internal loading exceeding external loading during the summer months (Carpenter and Lathrop, 2008). Furthermore, concentrations are distributed through wind action and diffusion. Diffusion occurs if the concentration of P exceeds that of overlying water. If external loading rates are below a certain threshold then sediments function as a net source of P, if rates are above that limit, then sediments function as net P sinks (Reddy *et al*., 2007). Long-term studies have shown water quality can improve and cause internal loading to decreases slowly after external sources are controlled (Coveney et al., 2005; Schindler and Hecky, 2009). However, more long-term studies are necessary because short-term studies do not take into account adaptive processes (Schindler and Hecky, 2009).

TP concentrations provide an indication of eutrophication by correlating P concentrations with chlorophyll *a*, a common measurement of algal biomass (Ruley and Rusch, 2004). P concentrations have been correlated to chlorophyll in streams (Dodds *et al*., 2002) and lakes (Smith, 1982; Ruley and Rusch, 2004). Regression analyses used to predict P concentrations from chlorophyll *a* explain much more variability in lakes, where r^2 values commonly exceed 90% (Dodds *et al*., 2002). However, many of these studies occurred in lakes located in northern and northeastern United States (Ruley and Rusch, 2002). P and chlorophyll relationships tend to display a discernable breakpoint above which chlorophyll values are substantially greater. Dodds *et al*., (2002) reported a break point occurred at about 30 μg/L of TP. Also chlorophyll concentrations tended to be greater at lower latitudes and higher temperatures (Dodds *et al*., 2002).

This study aimed to assess stormwater effects on phosphorus transport. Specifically, the objectives of the study were to 1) quantify the runoff effect on phosphorus mass loading and its relationship to water quality parameters, 2) determine runoff effect on changes of phosphorus concentrations, and 3) analyze seasonal variation of water quality parameters.

3.2 Methods

3.2.1 Study Area

Located on the Louisiana State University campus in Baton Rouge, Louisiana, USA (Latitude 30° 24' North; Longitude 91°10' West), University Lake is a 74.6 ha shallow lake with a shoreline perimeter of about 6.7 km (Figure 3.1). Within the lake system are five small lakes which surround University Lake. The lake's watershed consists of a drainage area of about 187.4 ha (Reich Assoc., 1991). University Lake has one surface outflow and one surface inflow,

excluding periods of runoff, both of which are via overflow dams. The climate is considered humid-subtropical, with long hot summers and short mild winters. Data collected from Baton Rouge Ryan Airport by the National Weather Service (station ID# 160549) show a long-term (1931-2000) mean annual temperature of 19.9 °C, with the lowest monthly mean of 10.9 °C in January, and the highest monthly mean of 27.9 °C in July (Figure 3.2). The long-term data (1930-2000) shows an average annual rainfall of 147.7 cm, ranging from the highest monthly average of 15.9 cm in July, to the lowest monthly average of 8.05 cm in October (Figure 3.3). Overall, the summer season shows greater total rainfall, while the fall shows smaller total rainfall amounts. Weather information was gathered from a near-by weather station, LSU Ben Hur Agricultural Research Station, which is located approximately 5 km east of the study site. The annual average air temperature during the study period was about 19.4 °C, varying from 11 °C in January to 27.3 °C in July. The total precipitation during the 14 month study period (July 2008 to September 2009) was 153.0 cm ranging from 26.3 cm in September 2008, to 1.2 cm in October 2008.

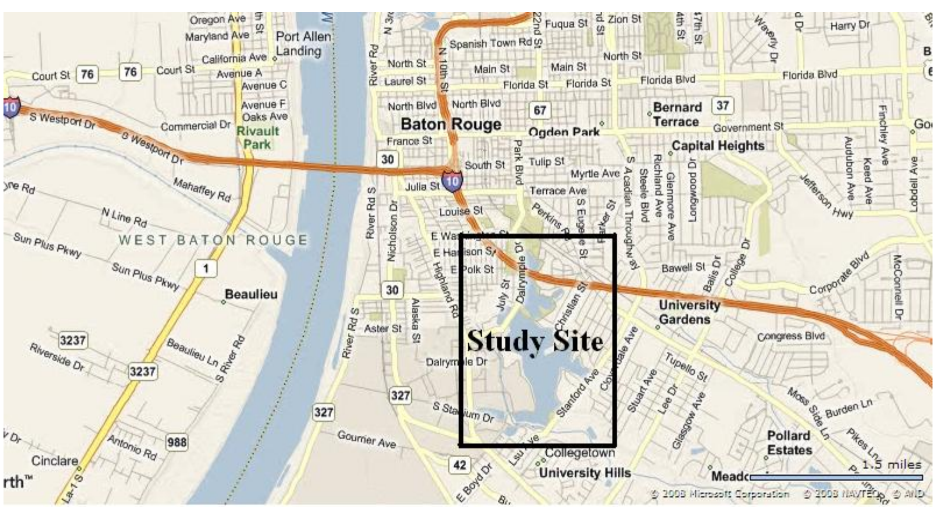


Figure 3.1. Location of University Lake located within Baton Rouge, Louisiana.

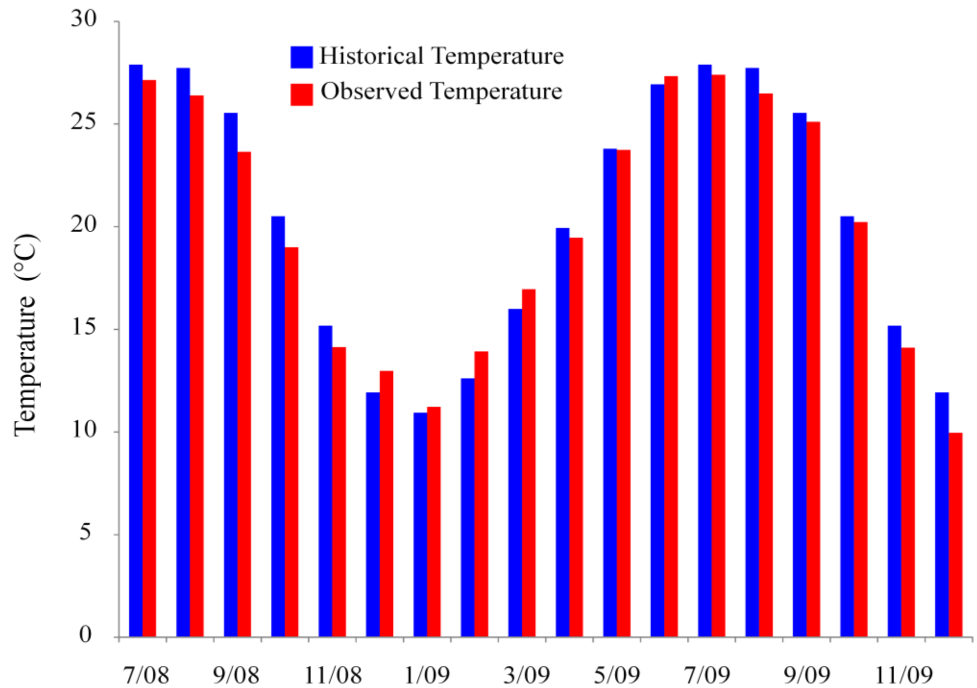


Figure 3.2. Temperature at Ben Hur Weather Station during study period compared against historical long-term data from Baton Rouge Ryan Airport (1930 – 2000).

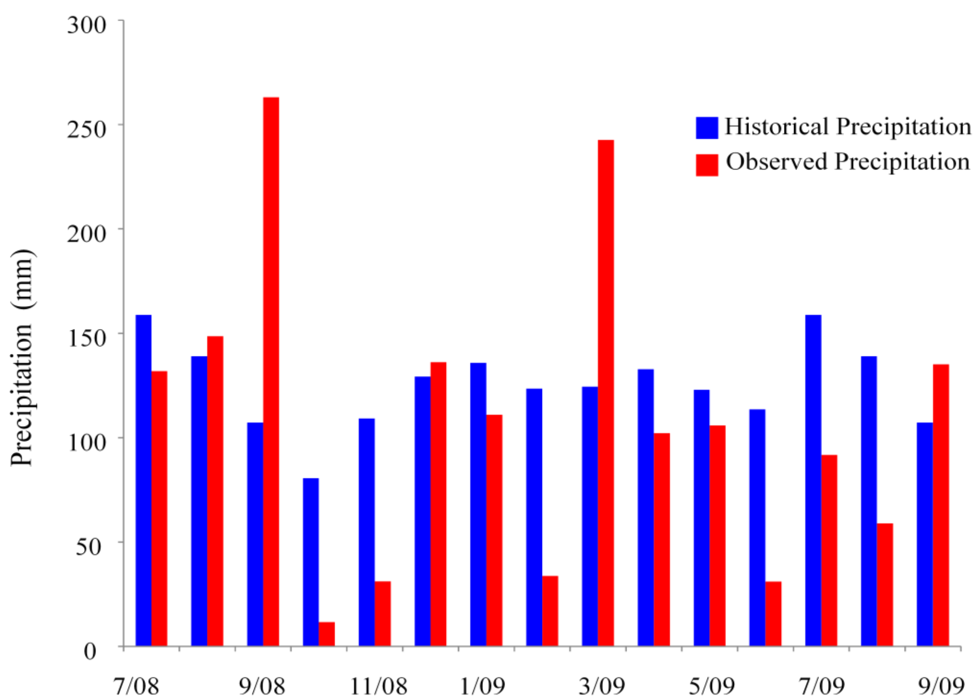


Figure 3.3. Monthly average precipitation at Ben Hur Weather Station during the study period compared against historical long-term data from Baton Rouge Ryan Airport (1930 – 2000).

The watershed is mostly urban, residential, recreational, and institutional. The lake was created when the area was initially dredged in the 1930's. This transformed a cypress swamp to an open water environment. The lake was most recently dredged in 1983 to remove bottom sediment and excess nutrients, resulting from surface runoff (Reich Assoc., 1991). City Park Lake, located immediately upstream of University Lake, had returned to pre-restoration phosphorus concentrations, while nitrogen concentrations were well below those reported after restoration by 2001 (Ruley and Rusch, 2002). This is also the case for University Lake. Chapter 4 indicates that nitrogen (nitrate, nitrite) concentrations are below analytical detection throughout most of the year.

3.2.2 Field Measurements

From July 2008 to October 2009, monthly *in-situ* measurements of water temperature, pH, conductivity, and dissolved oxygen were taken at seven locations using a YSI 556 multi-probe (YSI Inc., Yellow Springs, OH, USA). The probes were calibrated monthly before each monthly sampling event. Sites were located using a Trimble GeoXT GPS unit (Trimble Navigation Limited, Sunnyvale, CA, USA). At four sites designated along the shore (L1, L3, L5, and L7, Figure 3.4), measurements were taken 6 meters from shore on the bottom sediment. At the three sites designated in open water (L2, L4 and L6), measurements were taken at four depths – 30 cm and 60 cm below the surface, 30 cm above the bottom, and on the bottom sediment. In addition to the monthly measurements, measurements were conducted at the same locations within 24 hours following nine storm events. A storm event is defined as when greater than 1.0 cm of rain fell within one hour.

In addition, continuous measurements were collected using an Environment Monitoring Buoy (EMB), (YSI Inc., Yellow Springs, OH, USA) deployed in a central location of the lake.

The EMB system collected a series of water quality parameters at 15 minutes intervals, including; water temperature, pH, conductivity, chlorophyll *a*, turbidity, dissolved oxygen, and cyanobacteria concentration at about 1 m from the water surface. The probes were calibrated on a monthly basis.

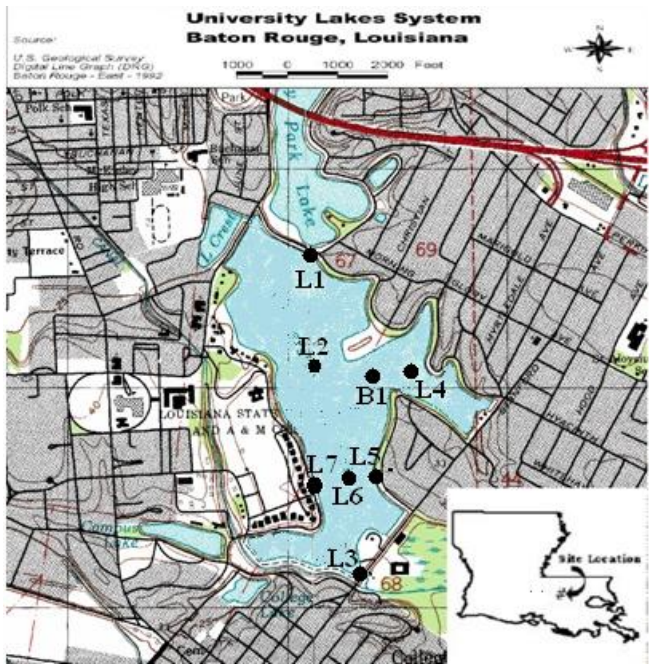


Figure 3.4. Seven sampling locations (L1 – L7) and the EMB location in University Lake on LSU campus.

Water depth measurements were taken using a wading rod in a grid pattern at 111 locations throughout the lake. Measuring locations were located using a Trimble GeoXT GPS unit (Trimble Navigation Limited, Sunnyvale, CA, USA). Utilizing the software package ArcMap 9.3 (ESRI, Redlands, CA), kriging analysis was performed to create depth contour lines (Figure 3.5). Contour lines were then digitized to calculate area of each contour which was in turn used to calculate average depth of University Lake. Using the surface area of 74.6 ha, calculated from spatial data, the volume is calculated at a given water level.

Figure 3.5. Bathymetric map of University Lake displaying contour lines.

Two HOBO pressure sensors (U20-001-01, Onset Computer Corp., Bourne, MA) were installed in the lake to record water level changes at 15 minute intervals. Depth was calculated using

$$Depth = (P_T - P_{atm})/(\rho * g) \tag{3.1}$$

where P_T is the pressure recorded by the transducer, P_{atm} is the atmospheric pressure, ρ is the density of water, and g is the gravitational constant. Measurements were correlated to the average measured depth and volume, to calculate lake depth and volume at 15 minute intervals. Stormwater runoff volume was determined to be the difference in lake volume extremes shortly before and after a rain storm.

Climatic data was obtained from the Louisiana Agriclimatic Information System (LAIS). Hourly precipitation, air temperature, maximum wind speed, average wind speed, relative humidity, solar radiation, soil temperature, and atmospheric pressure data were collected at LAIS' Ben Hur station, which was located about 5 km from University Lake.

3.2.3 Water Sampling

Water sampling was conducted at the same seven locations where *in-situ* measurements were taken on a monthly basis and within 24 hours following storm events (Figure 3.1). All of the samples were collected using the grab method. Samples collected from shore were taken from below the water surface, approximately 60 cm from shore. Depths were taken at open water sites using a measuring rod. Water samples during nine such storm events were collected. All samples were analyzed for TP by the Louisiana State University, Department of Agricultural Chemistry using EPA method 365.3 with a detection limit of 0.008 mg/L and reported as 0.004 mg/L.

3.2.4 Data Analysis

In order to determine storm runoff effects on TP within the lake, the change in TP mass (ΔTP) was quantified using a simple mass balance approach as follows:

$$\Delta TP = C_B V_B - C_A V_A \qquad (3.2)$$

where C_B is the TP concentration before the storm event, V_B is the lake volume before the storm event, C_A is the TP concentration after the storm event, and $V_A$is the lake volume after the storm event. The balance approach assumed that inflow and outflow were equal during the rain events; input as a result of groundwater was negligible; and internal phosphorus loading was negligible between monthly and storm event sampling. Event Mean Concentration (EMC) data is

commonly used to assess the pollution attributed to storm events. By using data concentration and volume data we can infer EMC using the formula:

$$EMC = \frac{C_A V_A - C_B V_B}{V_A - V_B} \quad (3.3)$$

Storm events were only included in the study if TP concentration measurements before a storm (C_B) were measured either from a monthly sampling or a previous storm event within 12 days of the storm event.

Regression analysis was performed on the relationships of ΔTP with all continuous, *in-situ*, and weather parameters. T-tests were used to determine if differences were recorded in *in-situ* measurements and concentrations before and after storm events. All statistical analysis were performed using the software package SAS 9.1.3 (SAS Institute Inc, 2003).

3.3 Results and Discussion

3.3.1 Base Conditions of University Lake

The 14-month study period received slightly less than average total precipitation (153 cm). Precipitation was not evenly distributed throughout the year (Figure 3.3). Greater than average rainfall was observed in September 2008 and March 2009 while the fall of 2008 experienced a drought with less than average rainfall. Rainfall during September 2008 was dominated by Hurricane Gustav and Hurricane Ike which totaled 228 mm of precipitation.

Over the 14-month study period, University Lake had an average depth of 0.86 m. Monthly average of depth varied from 0.63 m to 1.03 m. Average volume of University Lake was 643,000 m^3, ranging from 473,000 m^3 (August 2009) to 770,000 m^3 (September 2008). A seasonal variation in water depth was observed, with the highest water levels being recorded in

September 2008 and the lowest depths observed in August 2009. This variation was partially due to the seasonal variation in precipitation. During the wettest months of the study period, September 2008 and March 2009, the highest water levels and volumes were observed. Evapotranspiration also impacted the water levels and volumes of the lake, substantially during the summer months when temperatures and solar radiation are at their greatest (Peters, 2009, Xu and Wu, 2006).

Monthly TP concentrations during the 14-month study period averaged 0.286 mg/L. Values ranged from a high monthly concentration of 0.621 mg/L in August 2009 to a low monthly concentration of 0.167 mg/L in April 2009. Overall, smaller TP concentrations were observed during the spring. This is possibly due to TP being utilized for primary production resulting from an increase in solar radiation and nutrient availability. This time period has also been called the “clearwater phase” where a high zooplankton biomass grazes on phytoplankton (Søndergaard *et al.*, 2001). This may result in a reduction of organic sediment which in turn reduces the consumption of oxygen and enhances redox conditions. Due to this decrease in turbidity, solar radiation increase primary production. This consequently increases the uptake of P and oxidation of the benthic layer (Søndergaard *et al.*, 2001; Xie, 2006). Algal blooms are another possible cause for the decrease in TP. University Lake is most likely nitrogen limiting, which is typically the case in tropical and subtropical lakes (Phlips *et al*., 1997). Nitrogen fixation bacteria namely cyanobacteria, have been associated with blooms during the spring when temperatures and solar radiation increase and turbidity is still relatively low (Horne and Goldman, 1972). An increase in nitrogen availability may be causing the increase in primary productivity resulting in the consumption of P concentrations.

Similar to other studies of shallow eutrophic lakes (Petaloti *et al*., 2004; Ramm and Scheps, 1997), the concentrations and seasonal variability of TP in this study were high. Ramm and Scheps (1997) observed a mean TP concentration of 0.15 mg/L. Concentrations of TP were slightly less variable than lakes receiving agricultural and semi-treated domestic effluent which have ranged from 0.120 mg/L to 1.795 mg/L (Petaloti *et al*., 2004). The highest TP concentration was observed in August of 2009. Values were generally greater during the summer months of the study period. This is most likely due to two sources: internal loading from lake sediment and a reduction in water level. Researchers have found that several factors can affect phosphorus concentrations in shallow urban lake environments including total hardness, alkalinity, dissolved oxygen (DO), pH, thermal stratification, wind-induced mixing, and water level (Dalkiran *et al*., 2006). Alkalinity, pH, and DO tend to affect the internal phosphorus loading while temperature gradients and wind contribute to the distribution and concentration gradients throughout the lake. Water level has also been found to be directly correlated to TP, mainly during periods of more intense rainfall (Geraldes and Boavida, 2004). Higher temperatures contribute to greater TP concentrations by typically resulting in anoxic conditions which can lead to the reduction of ferric phosphates and the release of phosphorus into the water column (Ramm and Scheps, 1997; Reddy *et al*., 2007; Ruley and Rusch, 2004). Lower water levels result in greater lake concentrations of TP, because water with greater concentrations of TP (surface runoff, sediment pore water) is not diluted as much due to the smaller volume of lake water.

3.3.2 Storm Runoff

During the 14 months of this study, nine storms events were observed. Precipitation for the storm events averaged 57 mm, ranging from 23.6 mm to 95.5 mm. The storms generated an average runoff volume of 87,500 m^3, ranging from 19,400 m^3 to 169,000 m^3. Continuous water

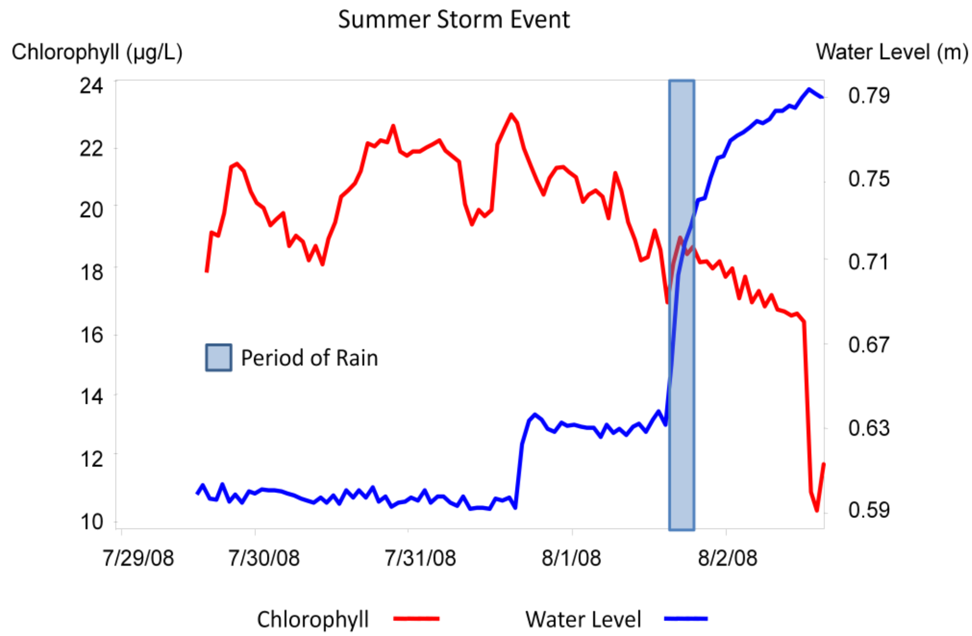
level measurements showed an instantaneous effect from precipitation during all nine storm events. Figure 3.6 depicts continuous monitoring during a summer sampling event and a winter sampling event. In an urban environment, the infiltration rate is much slower than a natural environment and therefore urban areas generate more surface runoff. This is due to the high percentage of impervious surfaces associated with urban environments. Urban and suburban watersheds typically have a runoff coefficient (RC; runoff as a percentage of precipitation) of 30 – 40% (Peters, 2009). Although spikes in chlorophyll *a* (Figure 3.3b) were commonly observed after storm events, storm events resulted in an immediate decrease in chlorophyll *a* concentrations.

Mean TP concentrations for storm event sampling during the 14-month study period was 0.383 mg/L. Values ranged from a high storm concentration of 0.636 mg/L to a low monthly concentration of 0.173 mg/L. Event mean concentrations (EMC) of runoff water was calculated to have a mean TP concentration of 0.636 mg/L ranging from 0.260 mg/L to 1.703 mg/L. Variation in EMC are dependent on rainfall, antecedent conditions, and rainfall intensity (Luo *et al.*, 2009). Brezonik and Stadelmann (2002) observed a mean TP EMC of 0.58 mg/L consisting of 561 monitored events in urban environments. The variation of TP EMC was much less than that observed by Luo *et al.* (2009) in a coastal area of southern China which calculated values up to 12.91 mg/L.

Average climatic data for the day of each rain event was recorded (Table 3.1) was typical for a subtropical climate on the day of a storm event. Solar radiation was low due to cloud cover throughout the day. Recorded air temperatures and soil temperatures were similar and fluctuated throughout the year, with the highest temperatures being observed in July and August with the

lowest temperatures in January. Wind speeds and precipitation totals varied depending upon individual storm event.

a)



b)

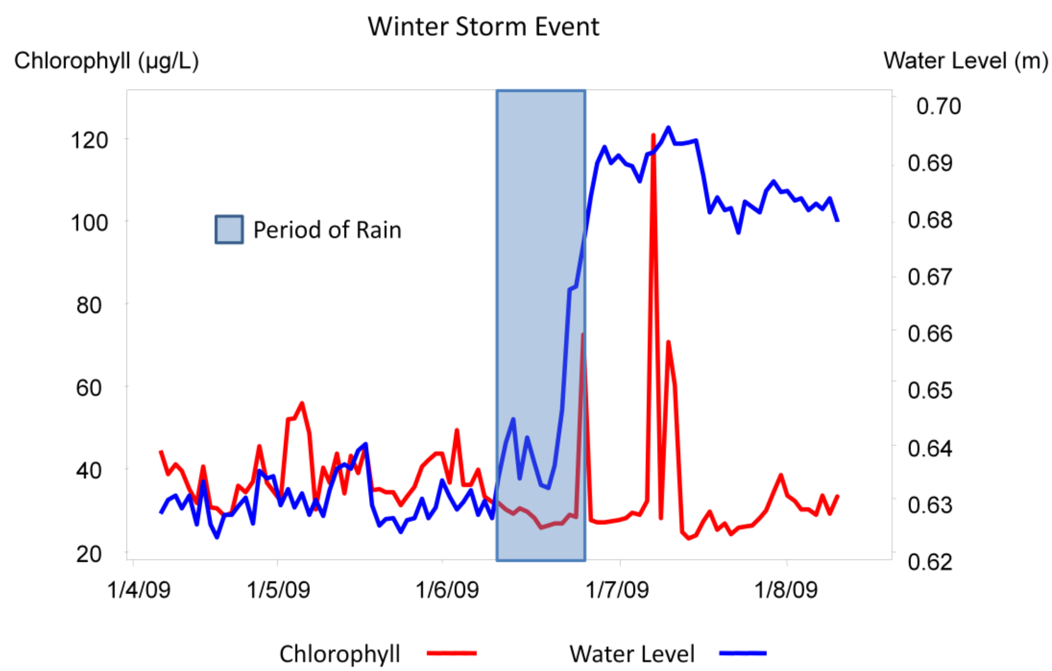


Figure 3.6. Continuous measurement of depth and chlorophyll *a* during a) summer and b) winter storm events at University Lake.

Table 3.1. Mean daily weather parameters measured for nine storm events at LSU Ben Hur weather station.

Storm Date	Avg. Temp. (°C)	Avg. Max Wind Speed (kph)	Avg. Wind Speed (kph)	Relative Humidity (%)	Solar Radiation (kw/m^2)	Soil Temp. (°C)	Total Precipitation (mm)	Antecedent Dry Time (hours)
7/31/2008	26.2	20.3	9.7	82.9	0.227	30.6	43.9	170
8/24/2008	22.8	32.3	17.6	90.7	0.024	26.4	52.8	112
1/6/2009	16.4	26.3	14.3	91.1	0.026	18.6	46.5	36
3/26/2009	19.1	26.8	10.8	93.0	0.053	20.6	95.5	234
5/4/2009	22.8	14.4	7.2	89.2	0.117	25.5	76.7	350
8/8/2009	25.9	14.8	6.1	83.4	0.128	30.1	23.6	262
9/5/2009	25.5	13.5	6.1	81.4	0.153	28.5	31.2	167
9/9/2009	24.7	11.7	5.6	88.0	0.135	28.0	23.9	68
9/23/2009	24.7	11.7	5.8	90.6	0.092	27.8	19.3	223

Differences in lake physicochemical parameters were observed between monthly and storm event sampling (Table 3.2). There was a significant difference between the temperatures of monthly and storm event sampling when separated by season. Both summer and spring storm events significantly decreased the temperature of the lake ($p < 0.006$). Winter storm events significantly increased the temperature of the lake ($p = 0.0006$). Temperature change resulting from storm events was highly dependent on the season. Summer months caused a decrease of lake temperature due to cooler precipitation temperatures than lake temperatures. The short time period surface runoff was in contact with land surfaces was not sufficient to raise runoff temperatures to the temperature of the lake. During the winter, land surface temperatures were sufficient to raise the temperature of runoff above the ambient lake temperature.

Table 3.2. Mean *in-situ* parameters recorded during sampling events at University Lake.

Sampling Date	Event	Temperature (°C)	Sp. Cond. (µS)	DO (% Sat.)	DO (mg/L)	pH
7/23/2008	Monthly	30.8 *(30.0-32.0)*	0.169 *(0.165-0.175)*	64.5 *(17.3-130.8)*	4.76 *(1.32-9.67)*	8.36 *(7.49-9.11)*
8/1/2008	Storm	30.2 *(29.3-31.2)*	0.173 *(0.159-0.184)*	46.9 *(4.7-101.6)*	3.52 *(0.35-7.51)*	8.00 *(7.41-8.80)*
8/23/2008	Monthly	30.8 *(29.6-32.0)*	0.190 *(0.148-0.266)*	67.7 *(1.4-124.7)*	5.02 *(0.11-9.21)*	7.95 *(7.15-9.03)*
8/25/2008	Storm	26.8 *(26.4-27.3)*	0.183 *(0.141-0.265)*	77.7 *(0.9-113.8)*	6.20 *(0.07-9.08)*	7.87 *(7.01-8.55)*
9/24/2008	Monthly	27.7 *(26.6-29.3)*	0.185 *(0.135-0.352)*	76.1 *(0.8-141.0)*	5.93 *(0.07-10.94)*	8.30 *(7.00-9.30)*
10/26/2008	Monthly	20.6 *(19.5-22.1)*	0.210 *(0.160-0.350)*	88.1 *(1.1-140.3)*	7.89 *(0.1-12.64)*	7.91 *(7.21-8.93)*
11/21/2008	Monthly	14.0 *(12.7-14.7)*	0.203 *(0.174-0.353)*	85.8 *(1.0-114.2)*	8.85 *(0.1-11.66)*	7.88 *(7.48-8.48)*
12/31/2008	Monthly	15.7 *(15.1-16.0)*	0.175 *(0.144-0.293)*	85.7 *(2.1-113.3)*	8.52 *(0.21-11.17)*	7.79 *(7.37-8.20)*
1/6/2009	Storm	19.2 *(18.7-19.7)*	0.180 *(0.143-0.299)*	75.1 *(2.7-99.1)*	6.92 *(0.25-9.13)*	7.50 *(7.20-7.90)*
1/29/2009	Monthly	14.2 *(13.5-14.9)*	0.176 *(0.158-0.268)*	92.3 *(2.9-125.2)*	9.47 *(0.3-12.68)*	7.81 *(6.92-8.45)*
2/25/2009	Monthly	17.1 *(16.0-18.7)*	0.191 *(0.170-0.286)*	93.4 *(1.7-129.2)*	8.97 *(0.17-12.27)*	7.92 *(6.82-8.64)*
3/26/2009	Storm	21.7 *(21.5-21.8)*	0.194 *(0.175-0.247)*	75.0 *(6.6-100.5)*	6.60 *(0.58-8.81)*	7.58 *(7.06-8.02)*
3/29/2009	Monthly	19.0 *(17.9-19.8)*	0.191 *(0.151-0.268)*	90.4 *(4.8-121.6)*	8.38 *(0.45-11.26)*	7.77 *(7.07-8.93)*
4/28/2009	Monthly	25.7 *(24.6-27.5)*	0.210 *(0.187-0.297)*	82.2 *(4.4-130.2)*	6.66 *(0.36-10.42)*	7.97 *(6.99-8.99)*
5/4/2009	Storm	25.6 *(24.9-26.3)*	0.201 *(0.184-0.246)*	67.7 *(6.1-106.1)*	5.52 *(0.50-8.58)*	7.42 *(7.01-7.85)*
5/28/2009	Monthly	29.1 *(27.2-31.0)*	0.195 *(0.175-0.268)*	95.1 *(3.7-178.4)*	7.24 *(0.29-13.24)*	8.23 *(7.04-9.09)*
6/16/2009	Monthly	31.3 *(30.3-32.3)*	0.229 *(0.205-0.298)*	75.7 *(5.7-155.7)*	5.56 *(0.42-11.32)*	8.11 *(6.96-9.08)*
7/26/2009	Monthly	31.0 *(29.0-31.9)*	0.262 *(0.235-0.341)*	96.5 *(3.0-191.9)*	7.14 *(0.22-14.03)*	8.30 *(7.02-9.11)*
8/9/2009	Storm	30.8 *(29.6-33.0)*	0.264 *(0.249-0.312)*	63.3 *(3.2-147.2)*	4.68 *(0.24-10.67)*	8.05 *(7.29-9.07)*
8/29/2009	Monthly	30.5 *(29.0-31.5)*	0.275 *(0.255-0.355)*	89.5 *(1.6-187.3)*	6.66 *(0.12-13.79)*	8.66 *(7.28-9.73)*
9/6/2009	Storm	29.5 *(28.0-33.0)*	0.275 *(0.246-0.351)*	58.9 *(1.9-160.0)*	4.42 *(0.15-12.09)*	8.03 *(7.32-9.14)*
9/10/2009	Storm	28.1 *(27.0-29.5)*	0.282 *(0.239-0.366)*	47.2 *(2.2-102.7)*	3.68 *(0.17-7.99)*	7.72 *(7.14-8.48)*
9/24/2009	Storm	27.3 *(25.9-30.6)*	0.260 *(0.233-0.340)*	96.9 *(4.1-166.9)*	7.62 *(0.33-12.80)*	8.07 *(7.00-8.99)*
9/27/2009	Monthly	28.4 *(27.3-30.6)*	0.250 *(0.226-0.324)*	60.9 *(2.1-138.6)*	4.69 *(0.16-10.42)*	7.91 *(6.98-8.90)*

** Numbers in italics represent minimum and maximum values.*

There was a significant reduction in pH observed during storm event sampling overall ($p < 0.0001$) (Table 3.2). However, when separated by season, only the spring exhibited a significant change ($p < 0.0001$). DO was reduced after a storm event during both the spring and winter months ($p = 0.0149$, $p < 0.0001$ respectively), and did not change during the summer ($p = 0.9717$). Specific conductivity only decreased significantly during the summer months ($p = 0.0006$).

3.3.3 Total Phosphorus

From, July 2008 through August 2009, a significantly greater concentration of TP after storm events ($p = 0.004$) was observed (Figure 3.7). However, during the month of October 2009, several heavy storm events within a relatively short period of time resulted in a dilution of lake TP concentrations. Therefore, overall TP concentrations were not significantly greater after a storm event ($p = 0.4571$). The mean ΔTP was 28.1 kg, ranging from 7.5 kg to 47.8 kg (Table 3.3). The highest observed ΔTP values were observed on 7/31/2008. The TP concentration for this storm event was measured nine days before the storm event. Some of the increase in concentration was due to internal loading because of the high temperatures during this time of the year; however it is unclear how much of an increase.

Mean TP concentrations for each sampling location from both storm and monthly sampling events are presented in Table 3.4. Concentrations at the inflow (L1) were much greater than concentrations throughout the lake and continued to decrease toward the outflow of University Lake (L3). This most likely means that most of the TP sources are located above University Lake. The decrease in concentration as water moves down flow is consistent with what would be expected as TP settles out and used for algal growth. Mean TP concentration at

L4 had the second greatest concentration within University Lake (0.331 mg/L). This is most likely due to large runoff volumes resulting from a longer shoreline.

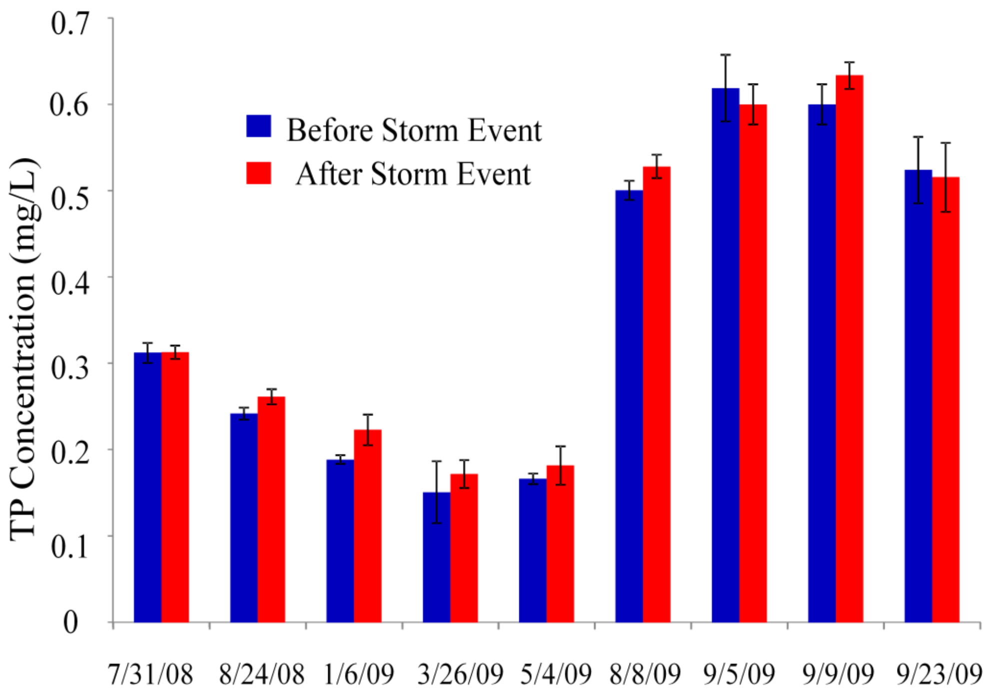


Figure 3.7. Total phosphorus concentrations before and after storm events at University Lake.

Table 3.3. Mean total phosphorus concentrations and lake water volumes before and after storm events at University Lake.

Storm Date	TP Before Storm (mg/L)	TP After Storm (mg/L)	Volume Before Storm (m^3)	Volume After Storm (m^3)	Precipitation (mm)	Storm Runoff (m^3)	Δ TP (kg)
7/31/2008	0.313	0.314	666814	817843	43.9	151029	47.80
8/24/2008	0.243	0.262	715235	806307	52.8	91074	37.55
1/6/2009	0.189	0.224	693929	747601	46.5	53672	35.91
3/26/2009	0.152	0.173	693437	862754	95.5	169317	43.78
5/4/2009	0.167	0.183	669411	709941	76.7	40530	17.73
8/8/2009	0.502	0.530	470889	490284	23.6	19395	23.33
9/5/2009	0.621	0.602	417171	442618	31.2	25446	7.45
9/9/2009	0.602	0.636	425845	439418	23.9	13573	22.92
9/23/2009	0.526	0.517	471946	511548	19.3	39602	16.44

Table 3.4. Mean TP by site from University Lake with standard deviation.

Site	Mean TP (mg/L)	SD
L1	0.345 *(0.164 - 0.806)*	0.172
L2	0.316 *(0.150 - 0.687)*	0.168
L3	0.308 *(0.148 - 0.721)*	0.176
L4	0.331 *(0.159 - 0.666)*	0.176
L5	0.309 *(0.124 - 0.625)*	0.164
L6	0.318 *(0.109 - 0.639)*	0.177
L7	0.326 *(0.142 - 0.735)*	0.180

**Numbers in italics represent minimum and maximum values*

3.3.4 Correlations

Regression analyses performed on *ΔTP* and climatic (Table 3.1), *in-situ* (Table 3.2), and continuous measurements (Table 3.5) revealed that stormwater runoff volume explained 71% of variability (r^2=0.71) (Figure 3.8). This result may mean that a first flushes were not observed or were not very dramatic during the sampled storm events which has been observed in other studies in urban environments (Flint and Davis, 2007; Huang *et al.*, 2009). Specific conductivity explained 41% of variance but was found to not be significant. The possible correlation is most likely due to dissolved phosphorus being a substantial component of the total phosphorus concentrations. Surprisingly, no correlation was found between *ΔTP* and the antecedent dry period before a storm event, which has been observed in other studies (Li *et al.*, 2007). Chlorophyll *a* concentrations were inversely related to *ΔTP* possibly due to a dilution of chlorophyll *a* by stormwater (r^2= 0.86).

Table 3.5. Mean continuous monitored parameters the day of storm event at University Lake.

Storm Date	Avg. Temp. (°C)	Avg. Sp. Conductivity (µS)	Avg. pH	Avg. Chlorophyll (µg/L)	Avg. Turbidity (NTU)	Avg. DO (% sat.)	Avg. DO (mg/L)	Cyanobacteria (cells/mL)	Lake Level Change (m)
7/31/2008	30.77	0.178	8.88	21.40	20.2	97.91	7.29	61995	0.202
8/24/2008	27.56	0.172	7.77	20.56	149.3	79.79	6.29	55817	0.122
1/6/2009	19.14	0.159	7.26	31.31	642.1	52.00	4.80	35229	0.072
3/26/2009	21.90	0.179	7.19	17.45	117.2	56.67	4.96	14140	0.227
5/4/2009	25.89	0.203	9.01	55.42	67.4	67.78	5.48	29824	0.054
8/8/2009	31.60	0.283	10.17	59.86	1399.4	10.99	0.81	155770	0.026

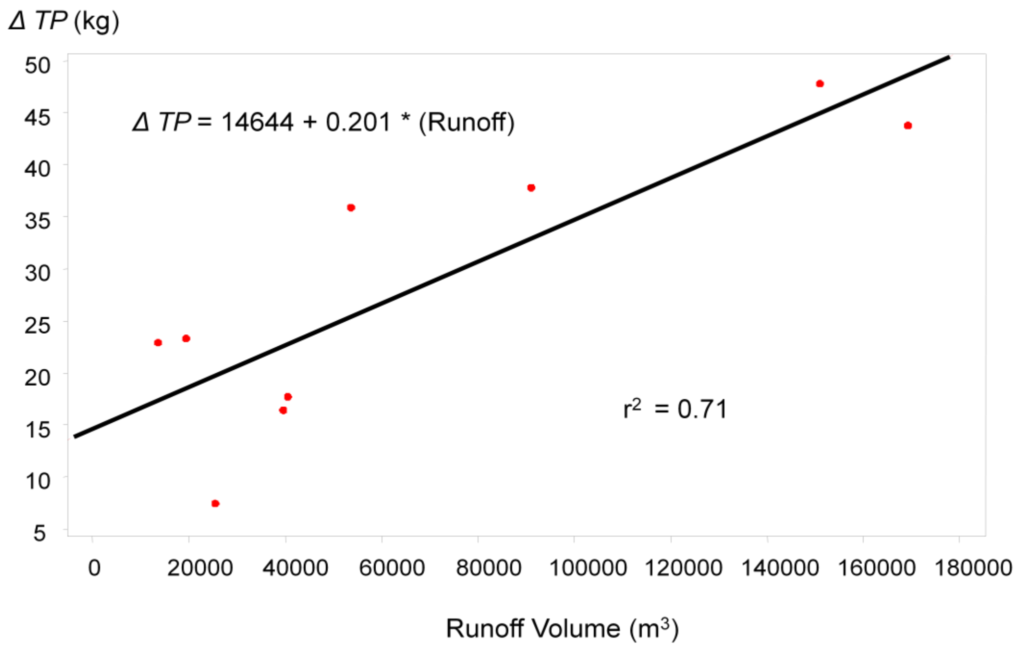


Figure 3.8. Relationship of the net change in TP mass load (*Δ TP*) with runoff volume from University Lake.

3.4 Conclusions

This study quantified the change in TP mass resulting from storm events in a shallow eutrophic urban lake. Mean monthly lake TP concentration was 0.286 mg/L. Mean storm event loading of 28.1 kg TP, with an estimated average concentration of 0.636 mg/L caused an immediate increase in TP within the water column by 14.1%. This resulted in a mean lake concentration of 0.383 mg/L TP after storm events. Much of the variation in TP concentrations was due to internal P loading which led to increased concentrations primarily during the summer months. Phosphorus loads in the lake were correlated with runoff volume ($r^2 = 0.71$), suggesting runoff volume is the most important factor effecting lake TP concentration after a storm event. Storm events had a seasonal effect on the temperature of the waterbody; increasing the temperature during the winter months and decreasing the temperature during the summer months. Storm events also tended to lower the pH of this lake system, while DO was more seasonal and decreased during the winter months, possible due to the increase in water temperature.

CHAPTER 4: RAINSTORM EFFECTS ON NITROGEN TRANSPORT TO A SUBTROPICAL URBAN LAKE

4.1 Introduction

Human export of nitrogen (N) to streams, rivers and reservoirs is of great concern due to its impact on the health of the environment and on humans. Like phosphorus, N can be a limiting nutrient in aquatic ecosystems and is required for plant growth. Many times excess N loading is a nonpoint source pollutant associated with activities in agricultural and urban environments. In urban and suburban environments N yield has been observed to be as much as 10 times greater than forested watersheds (Groffman *et al.,* 2004). Nonpoint source pollutants are more irregular than point source pollutants because they are impacted by seasonal changes, such as precipitation and flow rates, as well as anthropogenic effects, like construction and other soil disturbances. In urban environments N load transported by a storm event has been shown to both, be a function of characteristics affecting concentration (Lewis and Grimm, 2007) and runoff volume (Rosenzweig *et al.,* 2008). For these reasons mitigation to reduce nutrient loads consumes a large share of public resources (NRC, 2001).

In urban waterbodies, inorganic N is typically dominant. This includes N in the form of nitrate (NO_3), which is typically the most abundant, nitrite, and ammonia. During rain events nitrogen is typically transported from surface and subsurface soil to near waterbodies, often in the form of nitrate. This is in part because the more a nutrient is dependent on solid related particles, the more the concentration are dependant on rainfall intensity (Budai and Clement, 2007). Therefore in forested environments, low-density suburban, and agricultural catchments nitrogen concentrations are found to be transported at relatively low flows. However, results from the Chesapeake Bay watershed have shown that more urbanized sites export total nitrogen

and nitrate at greater less frequent flows as well (Shields *et al.*, 2008). That being said, N loads tend to be larger in urban areas due to a greater percentage of impervious surface and commercial land use. These conditions can lead to seasonal variation occurring in concentrations of nitrogen (Li *et al.*, 2009; Kane *et al.*, 2008).

Nitrogen fixation typically contributes when there is an extremely low supply of nitrogen in the waterbody. Typically in shallow lakes throughout the summer phosphorus concentrations increase as nitrogen declines. This can result in planktonic nitrogen fixing cyanobacteria blooms in freshwater when phosphorus is abundant and nitrogen availability is low. Even in these cases nitrogen can still remain the limiting nutrient in an ecosystem dominated by nitrogen fixing bacteria. Cyanobacteria are undesirable because they can be toxic, cause hypoxia, and disrupt food webs (Groffman *et al.*, 2004).

Typically, phosphorus is regarded as being the primary limiting factor of primary production; however it is important to acknowledge nitrogen as a possible limiting factor in lakes (Elser *et al.*, 1990). It is still not clear whether a reduction in both nitrogen and phosphorus, or just a reduction in phosphorus is needed to control eutrophication (Lathrop, 2007; Dodds *et al*., 2002; Conley *et al*., 2009; Schindler and Hecky, 2009). It is therefore important to assess the impact of an urban environment and rain events on N concentrations. This study aimed to assess stormwater effects on nitrogen transport. Specifically, the objectives of the study were to 1) evaluate the effect stormwater has on concentrations of species of nitrogen resulting from rainstorm events and 2) use lake parameters, namely cyanobacteria concentrations, to explain variation in nitrogen loadings.

4.2 Methods

4.2.1 Study Area

Located on the Louisiana State University campus in Baton Rouge, USA (Latitude 30° 24’ North; Longitude 91°10’ West), University Lake is a 74.6 ha shallow lake with a perimeter of about 6.7 km (Figure 4.1). Within the lake system are five small lakes which surround University Lake. The lake’s watershed consists of a drainage area of about 187.4 ha (Reich Assoc., 1991).

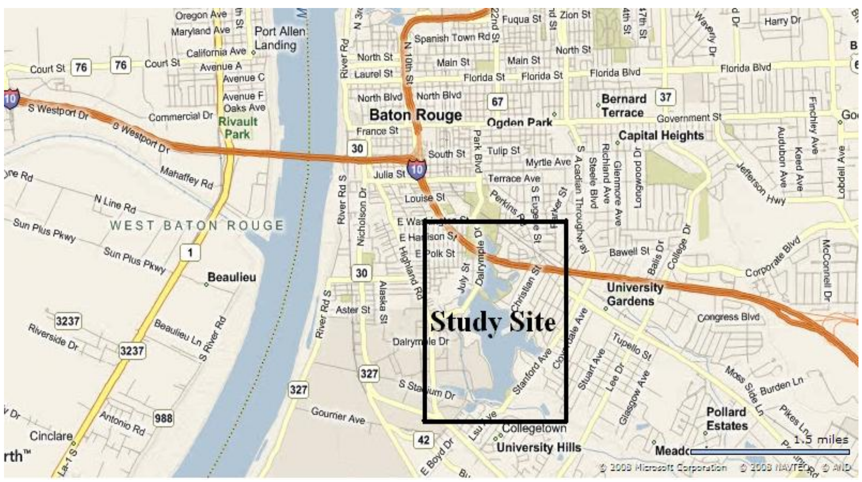


Figure 4.1. Map of University Lake located within Baton Rouge, Louisiana.

The watershed is almost entirely consisting of some form of urban use, mostly residential, recreational, and institutional. The lake was created when the area was initially dredged in the 1930’s. This transformed a cypress swamp to an open water environment. The lake was most recently dredged in 1983 to remove bottom sediment and excess nutrients, resulting from surface runoff (Reich Assoc., 1991). City Park Lake, located immediately upstream of University Lake, had returned to prerestoration phosphorus values, while nitrogen values were well below restoration values by 2001 (Ruley and Rusch, 2002).

The climate is considered humid-subtropical, with long hot summers and short mild winters. Data collected from Baton Rouge Ryan Airport by the National Weather Service (station ID# 160549) show a long-term (1931-2000) mean annual temperature of 19.9 °C, with monthly mean temperatures ranging from 10.9 °C in January to 27.9 °C in July (Figure 4.2). The long-term data (1930-2000) shows an average annual rainfall of 147.7 cm, ranging from a high monthly average of 15.9 cm in July to a low of 8.05 cm in October (Figure 4.3). The summer displays the highest rainfall while the fall has the lowest. Weather information during the study period was gathered from a near-by weather station, LSU Ben Hur Agricultural Research Station, which is located approximately 5 km east of the study site. The annual average air temperature during the study period (July 2008 to December 2009) was about 19.4 °C, varying from 9.9 °C in December 2009 to 27.3 °C in July 2009. The total precipitation during the 18 month study period (July 2008 to December 2009) was 213.9 cm ranging from a low monthly total of 1.2 cm in October 2008 to a high of 39.7 cm in December 2009.

4.2.2 Data Collection

From July 2008 to December 2009 water sampling was conducted at seven locations throughout University Lake, both on a monthly basis and with 24 hours following storm events (Figure 4.4). A storm event was defined as when greater than 1.0 cm of rain fell within one hour. Monthly and storm event samplings are presented as the mean of all seven sampling sites. All of the samples were collected using the grab method. Samples collected from shore were taken from below the water surface, approximately 60 cm from shore. Water samples collected in open water were collected 0.3 m below the water surface. All samples were analyzed for nitrate, nitrite, and total Kjeldahl nitrogen (TKN). Nitrate and nitrite were filtered through a 47μm glass fiber filter (GF/F Whatman International Ltd, Maidstone, England) and analyzed using EPA

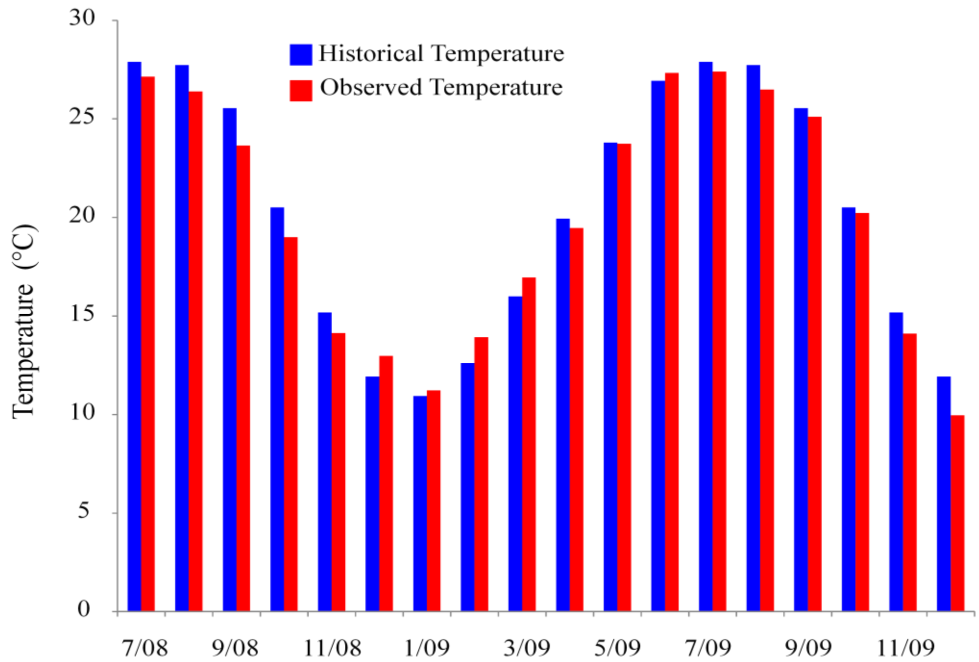


Figure 4.2. Temperature at Ben Hur Weather Station during study period compared against historical long-term data from Baton Rouge Ryan Airport (1930 – 2000).

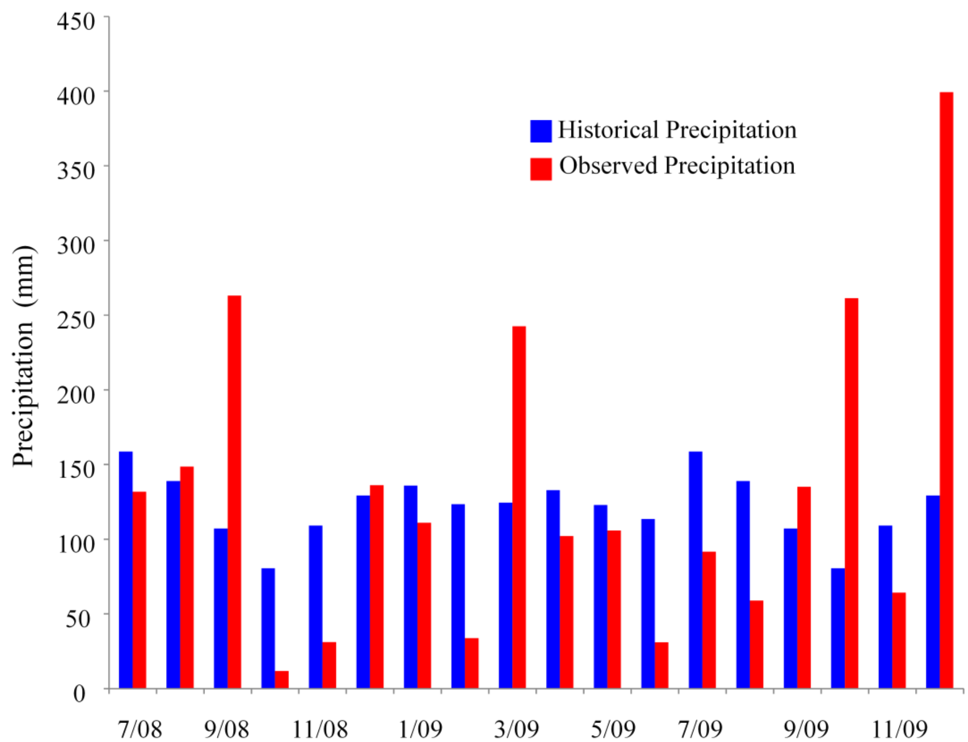


Figure 4.3. Monthly average precipitation at Ben Hur Weather Station during the study period compared against historical long-term data from Baton Rouge Ryan Airport (1930 – 2000).

method 353.3 with a detection limit of 0.05 and 0.003 mg/L respectively and reported as 0.025 and 0.0015 mg/L respectively. TKN was analyzed using method EPA 351.2 with a detection limit of 3.0 mg/L and reported as 1.5 mg/L. Samples collected from July 2009 on were filtered as above and analyzed for ammonia using EPA method 351.2 with a detection limit of 0.25 mg/L and reported as 0.175 mg/L. Chemical analysis was performed at the Louisiana State University, Department of Agricultural Chemistry.

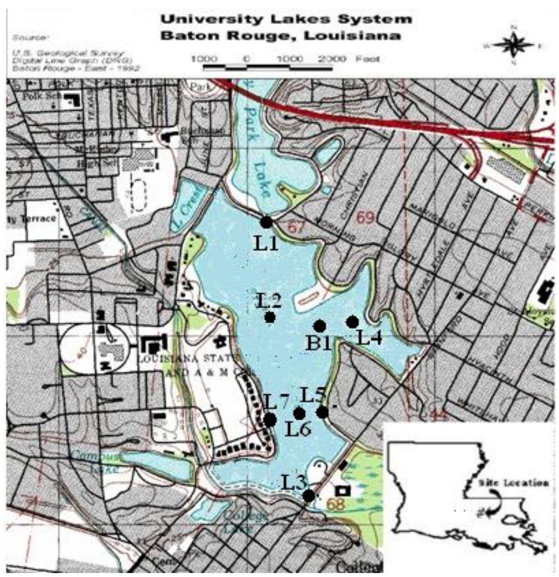


Figure 4.4. Seven sampling locations (L1 – L7) and the monitoring buoy location in University Lake on LSU campus.

Continuous measurements were collected using an Environment Monitoring Buoy (EMB), (YSI Inc., Yellow Springs, OH, USA) deployed in a central location of the lake. The EMB system automatically collected a series of water quality parameters, at 15 minutes intervals, including water temperature, pH, conductivity, chlorophyll *a*, turbidity, dissolved oxygen, and cyanobacteria concentration at about 3 ft from the water surface. The probes were calibrated on a monthly basis.

4.3 Results and Discussion

Monthly concentration data for the study period consisted of 16 sampling events (October 2009 and November 2009 were not sampled). TKN concentration averaged 2.51 mg/L ranging from below detection limit (3.00 mg/L) to 5.12 mg/L in December 2009. Mean storm event concentration was 3.45 mg/L ranging from below detection limit (BDL) to 6.78 mg/L in September 2009. TKN storm event concentrations were significantly greater than monthly concentrations (t-test, df=183, p=0.0386) (Figure 4.5). Monthly mean was BDL because observations BDL were included in the mean as a value of 1.5 mg/L. Of the TKN samples collected 72.4% of samples were BDL.

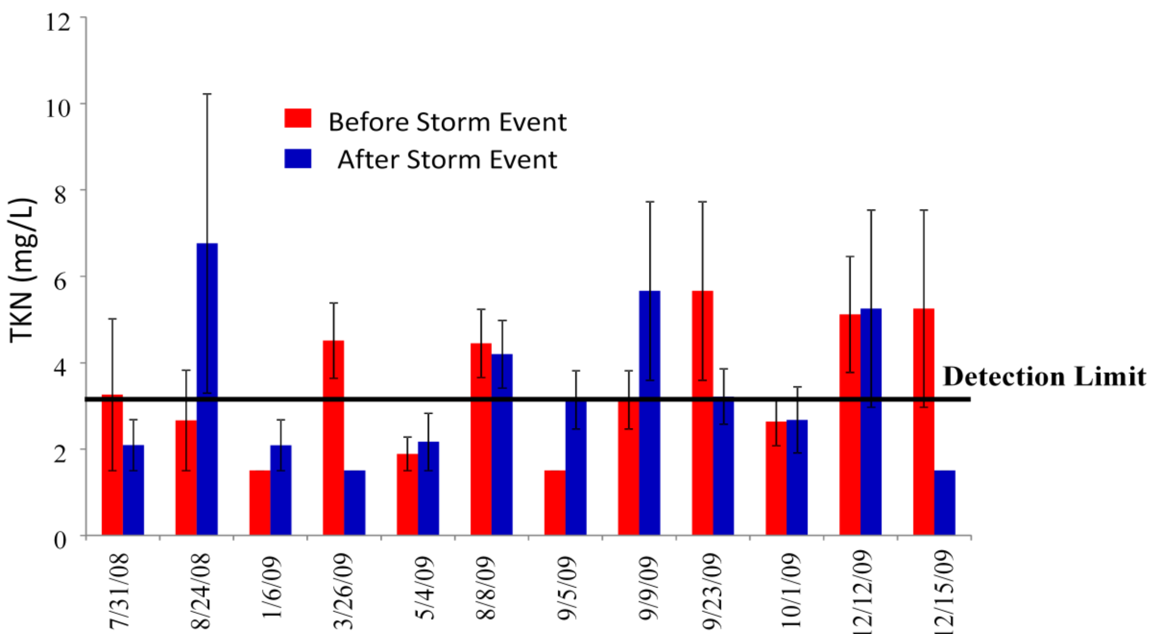


Figure 4.5. TKN concentrations in University Lake before and after storm events. Concentrations below detection limit were recorded as 1.5 mg/L.

Mean monthly nitrate-N was 0.054 mg/L, ranging from BDL (0.05 mg/L) to 0.24 mg/L in December 2009. Mean storm event concentration was 0.075 mg/L ranging from below detection limit to 0.211 mg/L in December 2009. Nitrate storm event concentrations were significantly

greater than monthly concentrations (t-test, df=223, p=0.0348) (Figure 4.6). Observations BDL were included in the mean as a value of 0.025 mg/L. Of the nitrate samples collected 67.6% of samples were BDL.

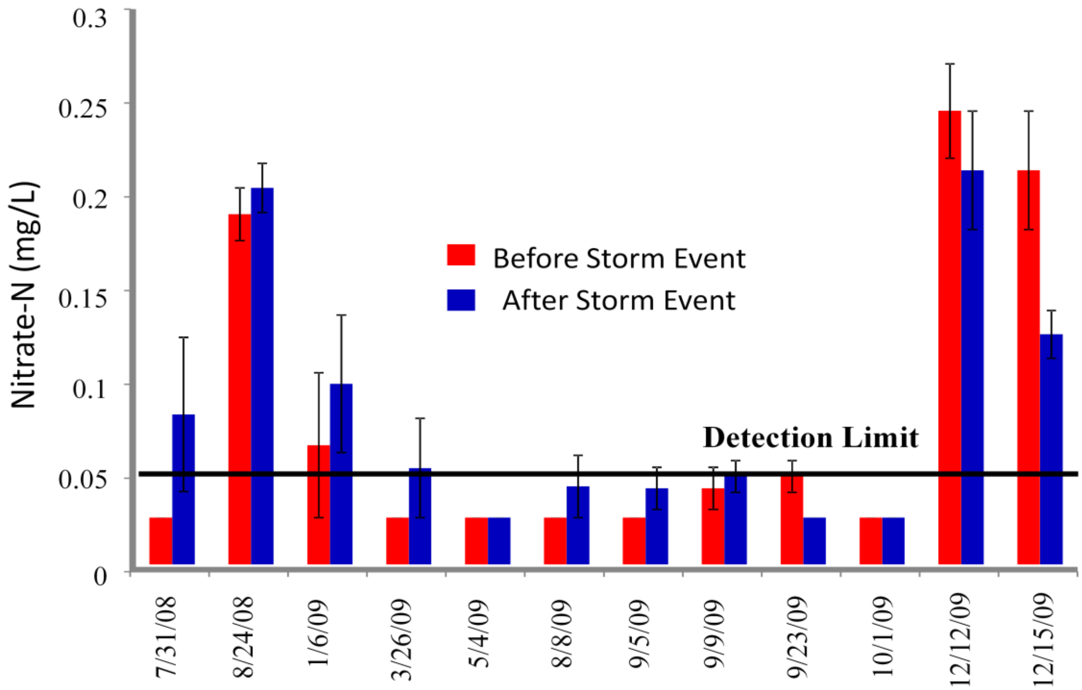


Figure 4.6. Nitrate-N concentrations in University Lake before and after storm events. Concentrations below detection limit were recorded as 0.025 mg/L.

Mean monthly nitrite-N was 0.004 mg/L, ranging from BDL (0.003 mg/L) to 0.012 mg/L in November 2008. Mean storm event concentration was 0.004 mg/L ranging from BDL to 0.012 mg/L in August 2008. Nitrite monthly concentrations were not significantly greater than storm event concentrations (t-test, df=223, p=0.1410) (Figure 4.7). Observations BDL were included in the mean as a value of 0.0015 mg/L. Of the nitrate samples collected 47.6% of samples were BDL.

Mean monthly ammonia was 0.23 mg/L, ranging from BDL (0.25 mg/L) to 0.36 mg/L in December 2009. Mean storm event concentration was 0.20 mg/L ranging from BDL to 0.27 mg/L in September 2009. Ammonia monthly concentrations were not significantly greater than storm event concentrations (t-test, df=82, p=0.1554) (Figure 4.8). Observations BDL were included in the mean as a value of 0.18 mg/L. Of the nitrate samples collected 82.1% of samples were BDL.

These results suggest that nitrogen is a possible limiting nutrient for University Lake. This is common in lakes with a large supply of phosphorus. Greater concentrations of TKN and nitrate observed after storm events indicated that stormwater runoff is a source of nitrogen to the system.

Table 4.1 presents the mean nitrogen concentrations by site. Greater concentrations of TKN, nitrate, and ammonia are observed at the inflow (L1) indicating that a substantial source of nitrogen is located upstream of University Lake. However, the rest of the lake has a fairly homogeneous supply of nitrogen indicating a rapid utilization of nitrogen.

Nitrogen fixing bacteria, namely blue-green algae (cyanobacteria) were not found to be correlated to the forms of nitrogen analyzed (Figure 4.9). All species of inorganic nitrogen were not observed to display a strong seasonal variation. Greater concentrations of nitrite were not observed during the winter months; however due to a lack of samples being above detection limit further research is need to support this trend.

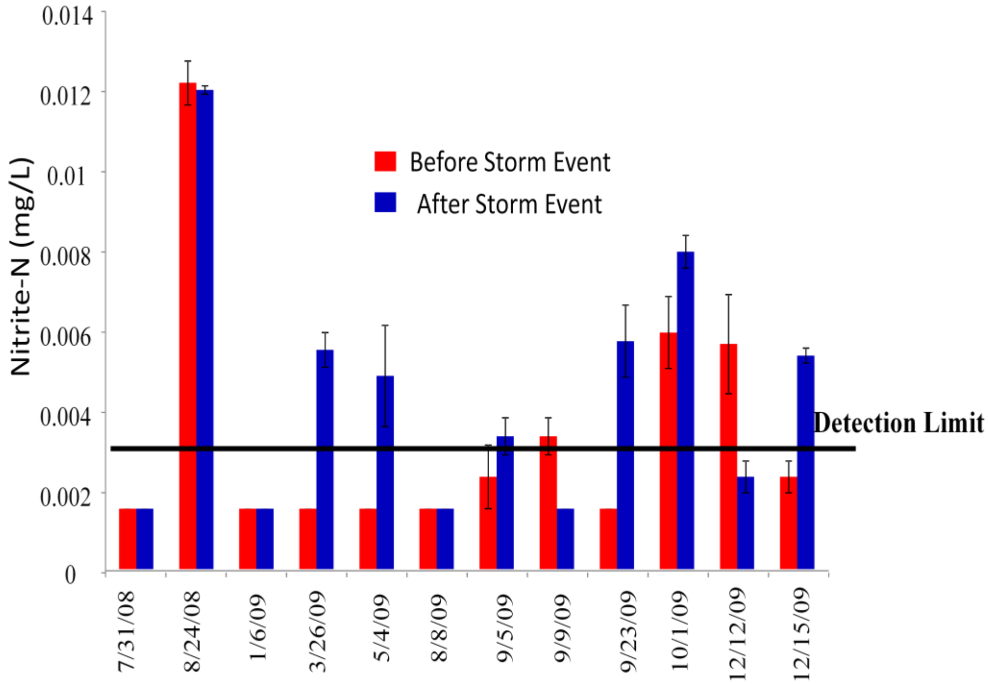


Figure 4.7. Nitrite-N concentrations in University Lake before and after storm events. Concentrations below detection limit were recorded as 0.0015 mg/L.

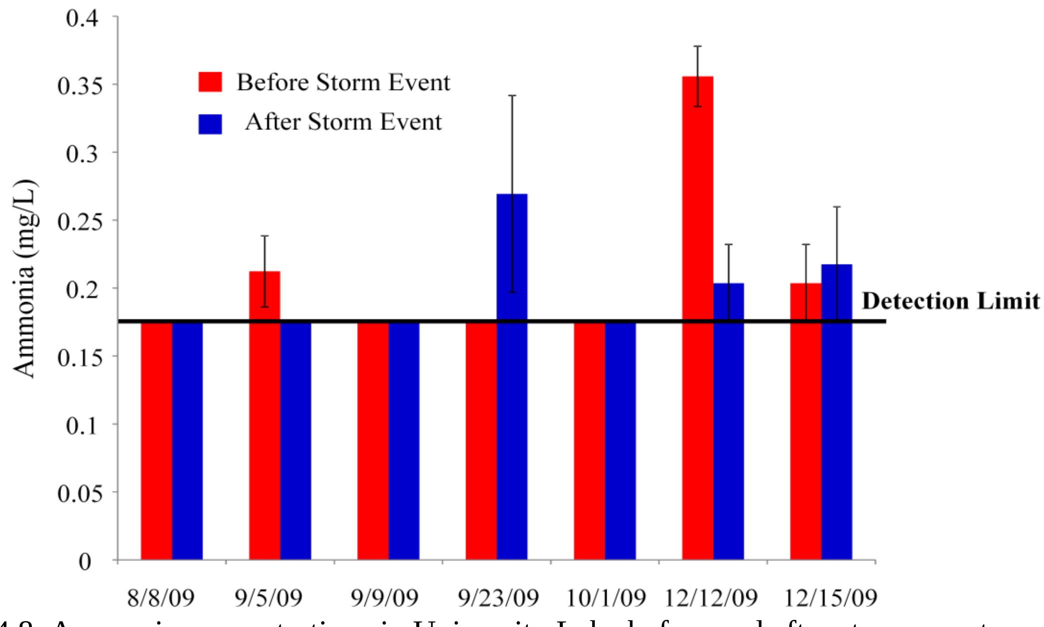
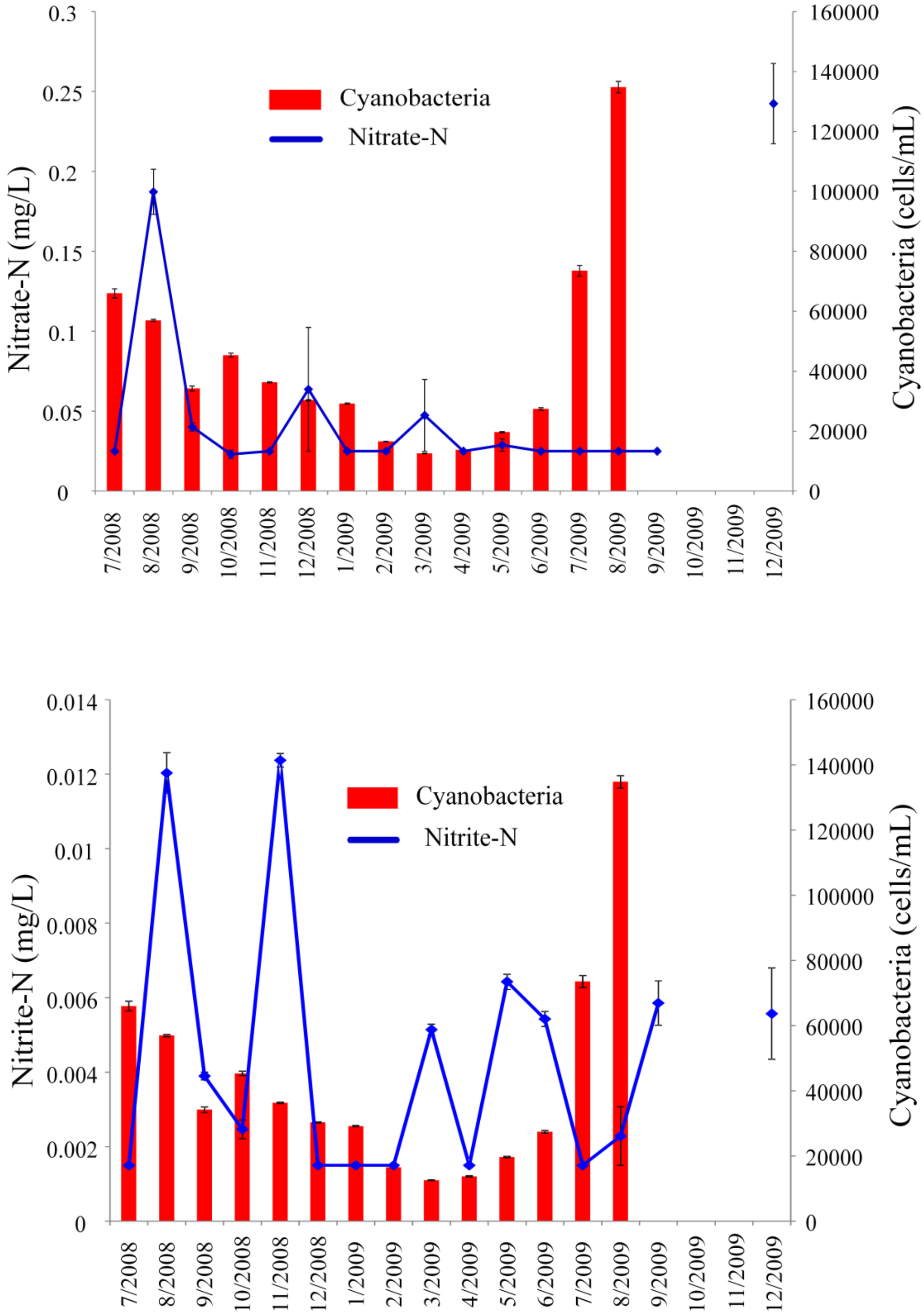


Figure 4.8. Ammonia concentrations in University Lake before and after storm events. Concentrations below detection limit were recorded as 0.175 mg/L.

Table 4.1. Average concentrations of nitrogen observed at seven locations in University Lake during July 2008 through December 2009.

Site	TKN(mg/L)		Nitrite-N (mg/L)		Nitrate-N (mg/L)		Ammonia (mg/L)	
	Mean (min-max)	SD	Mean (min-max)	SD	Mean (min-max)	SD	Mean (min-max)	SD
L1	3.43 (BDL-23.10)	4.93	0.005 (BDL-0.015)	0.0038	0.111 (BDL-0.347)	0.109	0.31 (BDL-0.68)	0.179
L2	2.90 (BDL-16.70)	3.29	0.005 (BDL-0.013)	0.0037	0.064 (BDL-0.313)	0.079	0.19 (BDL-0.32)	0.045
L3	2.66 (BDL-11.70)	2.82	0.004 (BDL-0.012)	0.0034	0.054 (BDL-0.229)	0.061	0.20 (BDL-0.38)	0.063
L4	2.88 (BDL-13.80)	2.66	0.004 (BDL-0.012)	0.0032	0.050 (BDL-0.228)	0.055	0.19 (BDL-0.29)	0.037
L5	3.12 (BDL-16.50)	3.26	0.004 (BDL-0.012	0.0034	0.062 (BDL-0.260)	0.071	0.21 (BDL-0.35)	0.068
L6	2.92 (BDL-18.00)	3.41	0.004 (BDL-0.013)	0.0035	0.048 (BDL-0.188)	0.050	0.19 (BDL-0.37)	0.061
L7	2.20 (BDL-7.75)	1.65	0.005 (BDL-0.013)	0.0039	0.076 (BDL-0.330)	0.094	0.21 (BDL-0.35)	0.068
Mean	**2.87**		**0.004**		**0.066**		**0.21**	

a)



b)

Figure 4.9. Relationship of mean monthly nitrate-N (a), nitrite-N (b), TKN (c), and cyanobacteria concentrations at University Lake.

c)

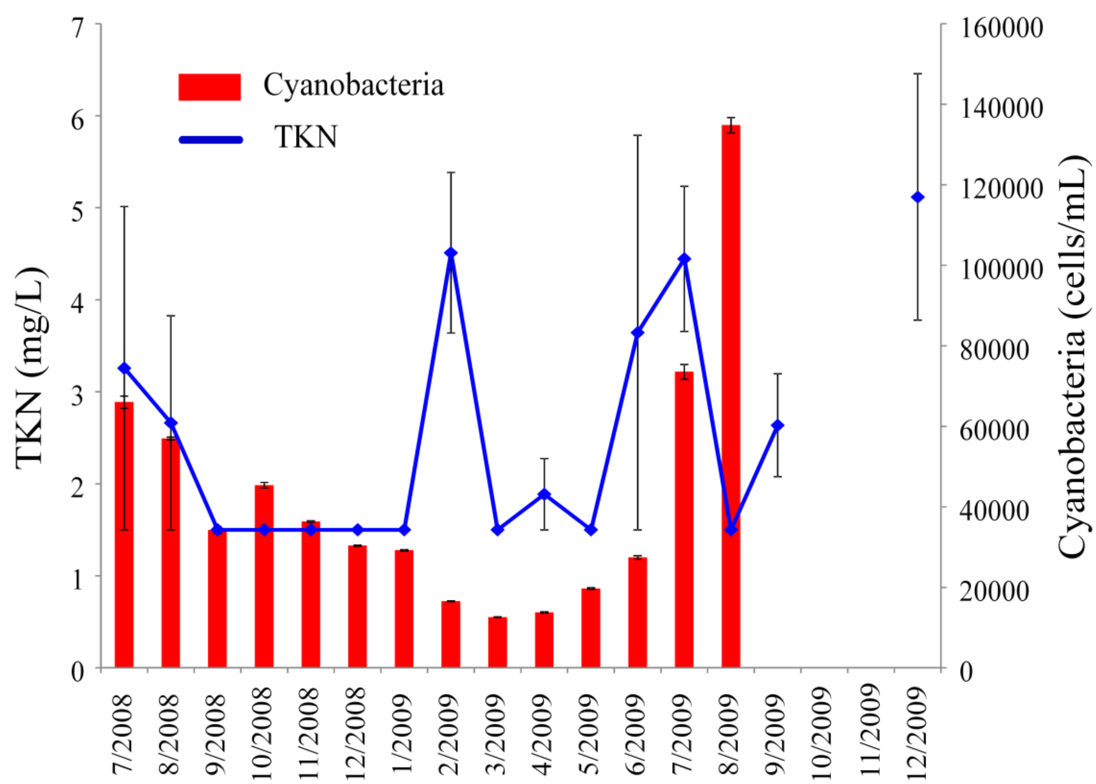


Figure 4.9, (continued).

4.4 Conclusions

This study has assessed changes in nitrogen concentration after rainstorm events and during several seasons. Over the entire study period, University Lake showed low concentrations of all species of nitrogen, although there was a significant increase of nitrate nitrogen and total Kjeldahl nitrogen following rainstorm events. Due to the high mean monthly concentration of phosphorus reported in Chapter 3 (0.286 mg/L), it is probable that this shallow lake system is nitrogen limited. Furthermore, the study found that species of nitrogen were not correlated to concentrations of nitrogen fixing bacteria, namely cyanobacteria, which may be the result of extremely low concentrations of nitrogen or that the growth of cyanobacteria is controlled by another variable.

CHAPTER 5: DISSOLVED OXYGEN, TEMPERATURE, AND METABOLIC RATES IN A SHALLOW SUBTROPIC URBAN LAKE

5.1 Introduction

Population increase and intensified human activities have altered physical, chemical and biological processes of many natural waterbodies. Today, surface water pollution has become a serious environmental issue for many urban areas. In the United States, about 44% of all lakes, and about 59% of man-made lakes are in fair or poor biologic condition (USEPA, 2009). Many urban lakes are impaired due to excess nutrients and organic enrichment, which can cause algal blooms, fish kills, and pungent smells. It is becoming increasingly obvious in urban areas, where a heavier strain is put on aquatic resources.

One main reason for the impairment of urban lakes is the impact of surface runoff. Surface runoff in urban areas introduces nutrients, sediment, and organic matter into a waterbody, which can affect the aquatic environment and biologic community (Wang *et al.*, 2003). The large percentage of impervious surface in urban environments alters regional hydrologic processes, such as infiltration, surface and subsurface flow, and groundwater recharge, while municipal and industrial discharges increase nutrient loads (Whitehead *et al.*, 2009). This leads to urban runoff pollution being difficult to control due to the large variety of pollutant sources and variable nature of source loadings due to the intermittent nature of precipitation and runoff (Li *et al.*, 2007). Therefore, having continuous real-time monitoring becomes a critical tool in assessing the health of an urban waterbody.

Rain events resulting in storm runoff can have dramatic effects on a lake body. They can cause hydrodynamic mixing which disrupt the diurnal dissolved oxygen (DO) pattern of the water body (Cornell and Klarer, 2008; McTammany *et al.*, 2003). DO, a crucial component of both organic and inorganic compounds, is representative of an ecosystems functionality and behavior. The concentration of dissolved oxygen (DO) in a waterbody is often regarded as an important indicator of the health of the biologic community because of the importance of DO in biological and chemical processes (Dodds, 2002). The concentration of DO in a lake system is affected by chemical, physical, and biological interactions including metabolic activity rates, atmospheric diffusion, and temperature resulting in irregular patterns over time (Dodds, 2002; D'Autilia *et al.*, 2004; Gelda and Effler, 2002; Ginot and Herve, 1994; Lopez-Archilla *et al.*, 2004; Odum, 1956). These factors can influence the magnitude and direction of oxygen flux in the water column (Gelda and Effler, 2002). During the winter, DO has been shown to be mainly controlled by physical re-aeration due to lower temperatures, while during the summer months photosynthesis, driven by high solar radiation, determines the release of considerable amounts of oxygen in the water (D'Autilia *et al.*, 2004). The decline in DO is increased throughout the summer due to warmer temperatures, high DO consumption due to respiration, and possibly due to a consumption of nutrients (MacKinnon and Herbert, 1996).

On a diel time scale, high values have been observed during the most irradiated hours of the day or during the early evening hours, while low values have been observed shortly before dawn. This diel fluctuation in DO is well known to be the results of biologic activity and not temperature (Whitney, 1942). Sub-daily time periods continue to be an area that requires more research. This can possibly lead to anoxic conditions, typically during the night, which can result in fish kills.

The two main metabolic processes in aquatic environments are photosynthesis and respiration. Photosynthesis is the process responsible for the synthesis of organic material. Respiration utilizes oxygen within the water, and can be a serious area of concern when a waterbody is nutrient enriched. Nutrients such as nitrogen and phosphorus stimulate productivity, consequently causing an increase photosynthesis and eventually respiration. However, photosynthesis does not occur below the photic zone or at night, causing dissolved oxygen concentrations to decline. Respiration dynamics have been shown to be variable and dependent upon the season (McTammany *et al.,* 2003).

Many methods have been used to estimate metabolism including the delta method, the diel oxygen change method, and the extreme value method (Chapra and Di Toro, 1991; Cornell and Klarer, 2008; McBride and Chapra, 2005; McTammany *et al*., 2003; Mulholland *et al*., 2005; Wang *et al*., 2003). Most methods deal with changes in dissolved oxygen concentrations, however some methods are more complex than others. Many of these methods are designed to calculate metabolism in streams, and incorporate variables unnecessary in calculating lake metabolism. In an effort to reduce error due to measurement error, the use of simple diurnal profiles are used to calculate *in-situ* rates of metabolism (Mulholland *et al.*, 2005).

Previous studies have found that metabolic rate varies at different time scales. On an annual time scale, respiration is an important factor throughout the year. Primary production and GPP and tends to increase in early spring following seasonal stratification in temperate and deeper lakes resulting in DO supersaturation (MacKinnon and Herbert, 1996). On a diel time scale previous short term measurements have depicted a double peak in primary production due to a slowdown in production (D'Autilia *et al.,* 2004). This has been attributed to a process known as photorespiration, a light-dependent process that develops carbon dioxide while consuming

molecular oxygen (Hull *et al.*, 2008). Primary production slows down during irradiated summer days while water is highly oxygenated (Hull *et al.*, 2008). Respiration is typically assumed to remain constant throughout the day.

Metabolic rates are affected by many variables including solar radiation, temperature, and biochemical oxygen demand (Cornell and Klarer, 2008; McBride and Chapra, 2005; Wang *et al*., 2003). The timing of metabolic processes from biotic communities can also impact DO concentration (D'Autilia *et al.*, 2004). Algal blooms cause dramatic fluctuations in DO which can be intensified by pollution and low-flow conditions (Whitehead *et al.*, 2009). These variables have considerable variation at multiple time scales and can have a dramatic effect on lake metabolism.

Many lake characteristics such as water depth can also affect DO concentrations and metabolic rate. Shallow lakes display unique characteristics distinct from commonly studied deeper lakes (Petaloti *et al.*, 2004). The entire water column may be within the photic zone depending on the depth of the lake and the turbidity of the water. Shallow lakes usually support highly diverse biota due to more extensive littoral zones (Arora and Mehra, 2009). Wind can also cause resuspension of sediment and consequently a considerable portion of nutrients (Petaloti *et al.*, 2004). Lake sediment can also affect metabolic rates within a lake system resulting in it becoming heterotrophic (Urabe, 2005). For these reasons, biologic, chemical, and physical parameters of shallow lakes are highly variable and prediction of DO in shallow waters is complex (Hull *et al*., 2008).

Although some studies have been performed at short time intervals (D’Autilia *et al.*, 2004), information is still lacking in regard to DO. Specifically, the objectives of the study were to; 1) assess trends in dissolved oxygen and temperature in a shallow urban lake environment, 2)

quantify rates of metabolism including respiration production and gross primary production, 3) utilize weather and lake parameters to explain trends in metabolism, 4) analyses calculated metabolic rates to determine the autotrophic/heterotrophic nature of the lake.

5.2 Methods

5.2.1 Site Description

Located in Baton Rouge Louisiana on the Louisiana State University campus (Latitude 30° 24’ North; Longitude 91°10’ West), University Lake is a 74.6 ha shallow lake with a perimeter of about 6.7 km (Figure 5.1). There are five small lakes surrounding University Lake and the entire lake watershed has a drainage area of about 187.4 ha (Reich Assoc. 1991). The climate is considered humid-subtropical, with long hot summers and short mild winters. Long-term data was collected from Baton Rouge Ryan Airport by the National Weather Service (station ID# 160549). Long-term (1931-2009) annual temperature of was 19.9 °C, ranging from 10.9 °C in January to 27.9 °C in July (Figure 5.2). The annual average air temperature at Ben Hur weather station during the 18 month study period (July 2008 to December 2009) was 19.9 °C. Mean monthly temperature ranged from a low of 9.9 °C in December 2009 to 27.4 °C in July 2009. During most of the 18 month study period, monthly averages were colder than average long-term temperatures. Exceptions were during the winter of 2008-2009, where above average temperatures were observed. The long-term data (1930-2000) shows an average annual rainfall of 147.7 cm, ranging from a high monthly average of 15.9 cm in July to 8.05 cm in October (Figure 5.3). The highest precipitation totals occurred during the summer, while the fall typically received the least. Total precipitation during the study period was 235.9 cm ranging from a low monthly total of 1.2 cm in October 2008 to a high of 39.9 cm in December 2009.

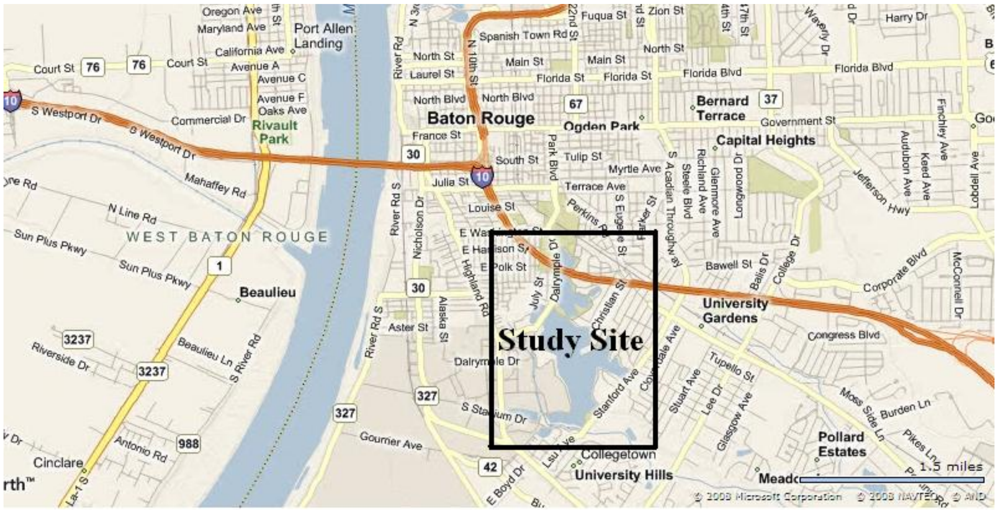


Figure 5.1. Location of University Lake in Baton Rouge, Louisiana.

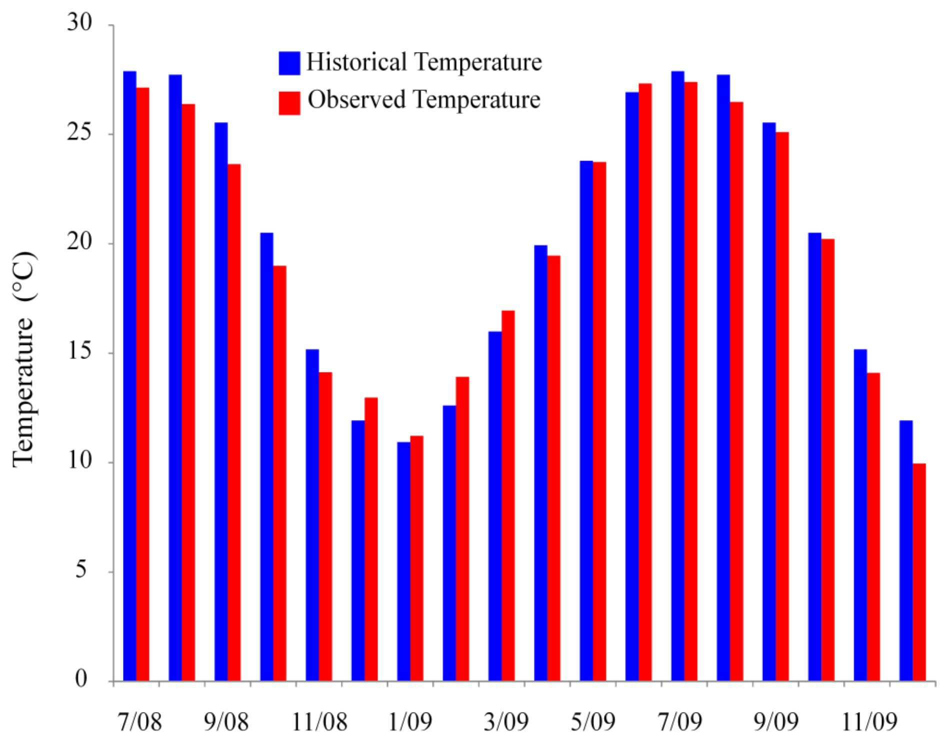


Figure 5.2. Temperature at Ben Hur Weather Station during study period compared against historical long-term data from Baton Rouge Ryan Airport (1930 – 2000).

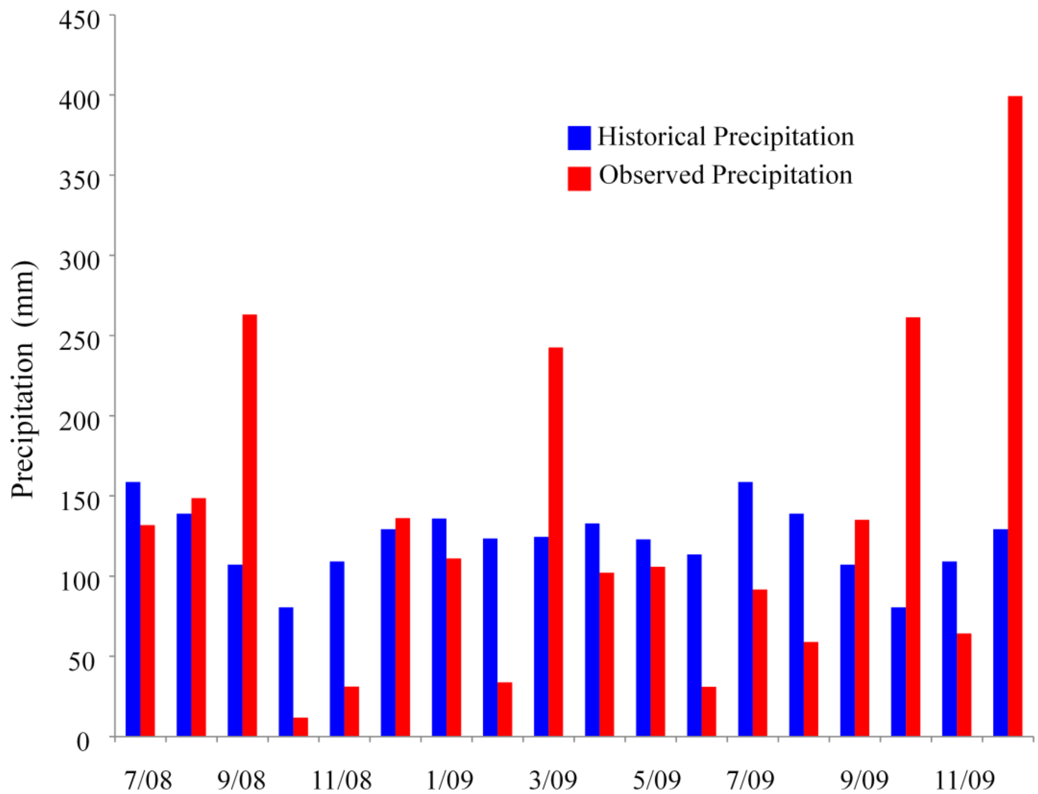


Figure 5.3. Monthly average precipitation at Ben Hur Weather Station during the study period compared against historical long-term data from Baton Rouge Ryan Airport (1930 – 2000).

The watershed is almost entirely consisting of some form of urban use, mostly residential, recreational, and institutional. The lake was created when the area was initially dredged in the 1930's. This transformed a cypress swamp to an open water environment. The lake was most recently dredged in 1983 to remove sediment and excess nutrients, resulting from surface runoff (Reich Assoc., 1991).

5.2.2 Field Measurements

An Environment Monitoring Buoy (EMB) (YSI Inc., Yellow Springs, OH, USA) was deployed in an approximately central location of the lake. The EMB system consisted of a heavy-duty floating platform with a YSI 6600 multi-probe, solar panel, and data transmission antenna. The system automatically collected a series of water quality parameters at 15 minute

intervals, including water temperature, pH, conductivity, chlorophyll *a*, turbidity, dissolved oxygen, and cyanobacteria concentration data at about 3 ft from the water surface. The probes were calibrated on a monthly basis.

Water samples were collected at seven sites located throughout the lake; four sites located along the shoreline and three sites were located in open water. Samples were collected 0.3 m below the water surface. Samples were analyzed for five day biological oxygen demand (BOD_5) using a modified version of the 'Standard Methods for the Examination of Water and Wastewater', Section 5210B (APHA, 2005). The samples were collected and immediately returned to the laboratory for analysis. Modification included; reaeration being performed daily if dissolved oxygen concentration was below 3.00 mg/L due to no dilution being performed; samples were stored at room temperature and corrected for temperature. The reported BOD_5 values are the mean values of the seven sampling sites. Figure 5.4 depicts the seven sampling sites and the location of the EMB.

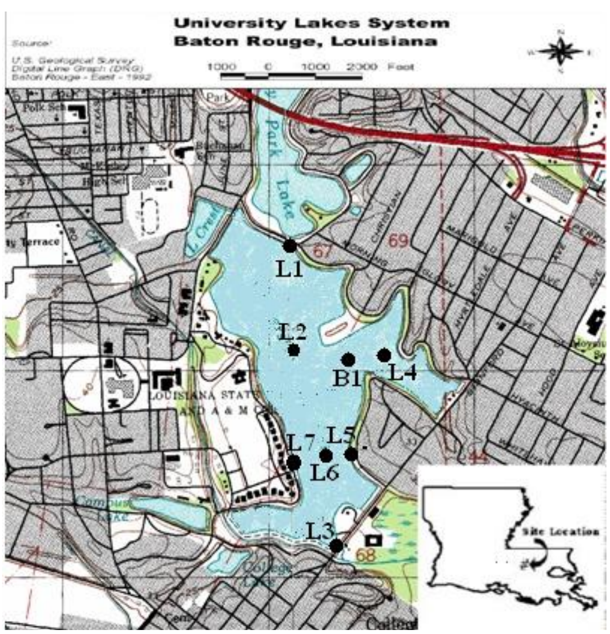


Figure 5.4. Seven sampling locations (L1 – L7) and monitoring buoy (B1) location within University Lake.

From July 2008 to December 2009, monthly *in-situ* measurements of dissolved oxygen were taken at seven locations (Figure 5.4) using a YSI 556 multi-probe (YSI Inc., Yellow Springs, OH, USA). The probes were calibrated monthly before each monthly sampling event. Sites were located using a Trimble GeoXT GPS unit (Trimble Navigation Limited, Sunnyvale, CA, USA). At the four sites that were located along the shore (L1, L3, L5, and L7, Figure 3.4) measurements were taken about 6 meters from shore on the bottom sediment. For the three sites that were located in open water (L2, L4 and L6) measurements were taken at four depths – 30 cm and 60 cm below the surface, 30 cm above the bottom, and on the bottom sediment. In addition to the monthly measurements, measurements were conducted at the same locations within 24 hours following twelve storm events. In this study, a storm event was defined as when greater than 1.0 cm of rain fell within one hour.

Two pressure sensors were installed in the lake to record data at 15-minute intervals. Atmospheric and hydrostatic pressure data was used to calculate average lake depth of the University Lake at 15-minute intervals.

Climatic data was obtained from Louisiana Agriclimatic Information System (LAIS). The data included hourly precipitation, air temperature, maximum wind speed, average wind speed, relative humidity, solar radiation, soil temperature, and atmospheric pressure measured at Ben Hur, which is located about 5 km from University Lake.

5.2.3 Estimation of Lake Productivity

We used a single station diel oxygen change method in an open system to measure the individual components of metabolism including community respiration (R) and net primary

productivity (NPP) (Odum, 1956). From these measurements we are able to calculate gross primary production (GPP).

Reaeration resulting from oxygen diffusion with the atmosphere was not calculated in this study. Several studies have determined that diffusion rates were negligible when calculating metabolic rates (Cornell and Klarer, 2008). Three assumptions are involved in this study. We assume that the water column is evenly mixed throughout the lake environment. This assumption is well known to be false in most lakes (Ginot and Herve, 1994). Secondly, we assume that biomass and nutrients are evenly distributed throughout the study period. We also assumed meteorological variables, in particular wind speed, remain constant.

Net primary productivity (*NPP* in g O_2/m^2/day) is calculated by summing the flux in oxygen during daylight hours (sunrise to sunset) using

$$NPP = \sum_{sunrise}^{sunset} (\Delta[O_2]) \times Z_i \tag{5.1}$$

where ΔO_2 is the hourly change in DO concentration in mg/L, and Z is the average hourly water depth in meter.

Respiration was assumed to be consistent throughout both during the day light hours and during the dark night hours. Average hourly respiration (R_{hr} in g O_2/m^2/hour) was determined between one hour after sunset and one hour before sunrise to ensure that no photosynthesis was occurring using

$$R_{hr} = \text{-}(\sum_{i=(sunset+1)}^{sunrise-1} (\Delta[O_2]/\Delta t) \times Z/(24-(sunset-sunrise))) \tag{5.2}$$

where O_2 is the dissolved oxygen concentration in mg/L, Z is the average depth of the lake, *sunrise* is the time the sunrise occurred, and *sunset* is the time the sunset occurred. (5). Gross primary productivity (*GPP* in g O_2/m^2/day) was determined using

$$GPP = NPP + (R_{hr} \times (24 - (sunset - sunrise))) \tag{5.3}$$

where *GPP* is gross primary productivity, *NPP* is net primary productivity, R_{hr} is hourly respiration, *sunrise* is the time sunrise occurred, and *sunset* is the time sunset occurred. Respiration (*R* in g O_2/m^2/day) is calculated using

$$R = R_{hr} \times 24 \tag{5.4}$$

where R_{hr} is hourly respiration. Calculated mean monthly *GPP* and *R* values were used to determine the productivity respiration ratio (*P/R*). The ratio provides insight about the heterotrophic and autotrophic processes within the system. *P/R* was calculated using

$$P / R = GPP / R \tag{5. 5}$$

where *GPP* is gross primary productivity and *R* is respiration.

Estimation error can occur in respiration and productivity rates due mainly to two sources: mechanical and/or physical processes controlling DO concentrations (Cornell and Klarer, 2008). In a natural system, if DO concentration is solely the result of metabolism, the respiration rate and the *GPP* should be greater than zero and the net primary productivity (*NPP*) should be less than *GPP*. In this study, hence, we removed data points from the calculations when respiration estimates were calculated negative or *NPP* estimates were greater than *GPP*.

5.3 Results and Discussion

Table 5.1 depicts the mean monthly average, standard deviations, minimum, and maximum of weather and lake parameters. Table 5.1 also shows the results of the mean of monthly averages of the calculated metabolism rates.

Table 5.1. Mean *(min-max)* monthly weather and lake parameters of University Lake.

Parameters	Mean *(min-max)*	SD
Air Temp. (°C)	19.7 *(9.9-27.1)*	5.9
Total Precipitation (mm)	105.1 *(11.2-239.4)*	72.2
Avg. Wind Speed (kph)	10.1 *(6.2-13.5)*	1.5
Relative Humidity (%)	73.0 *(67.1-81.9)*	4.0
Solar Radiation (kw/m^2)	0.186 *(0.092-0.291)*	0.065
Soil Temp. (°C)	22.6 *(12.9-32.2)*	12.6
Water Temp. (°C)	22.6 *(14.4-31.0)*	6.3
Sp. Conductivity (µS)	0.180 *(0.160-0.204)*	0.014
PH	7.86 *(7.48-8.06)*	0.18
Chlorophyll *a* (µg/L)	26.51 *(20.31-33.47)*	4.30
Turbidity (ntu)	106.4 *(32.9-303.1)*	87.3
DO Saturation (% sat)	81.8 *(50.2-111.7)*	19.2
DO (mg/L)	7.30 *(4.25-10.2)*	2.22
Cyanobacteria (cells/mL)	32300 *(12600-65400)*	17800
Depth (cm)	90.0 *(73.9-101.6)*	8.7
BOD_5 (mg/L)	7.2 *(5.4-9.5)*	1.15
BOD_5 (N Inhibited) (mg/L)	3.9 *(2.6-5.4)*	0.98
NPP (g O_2 / m^2 /day)	2.13 *(0.83-3.79)*	0.97
Respiration (g O_2 / m^2 /day)	5.90 *(2.38-11.77)*	2.87
GPP (g O_2 / m^2 /day)	4.41 *(1.96-7.68)*	1.71

5.3.1 Dissolved Oxygen

EMB dissolved oxygen concentrations had a mean monthly concentration of 7.26 mg/L ranging from 4.34 to 10.1 mg/L in October 2008 and February 2009 respectively (Figure 5.5).

Overall, winter mean daily DO concentrations were significantly greater than summer mean daily DO concentrations (t-test, df = 244, t = -10.0, p = <.0001). Daily variance in dissolved oxygen was significantly greater (t-test, df=244, t=7.09, p= <.0001) during the summer season (April – October) than the winter season (November – March). Summer DO concentrations also appeared to have greater variation at finer time scales as well.

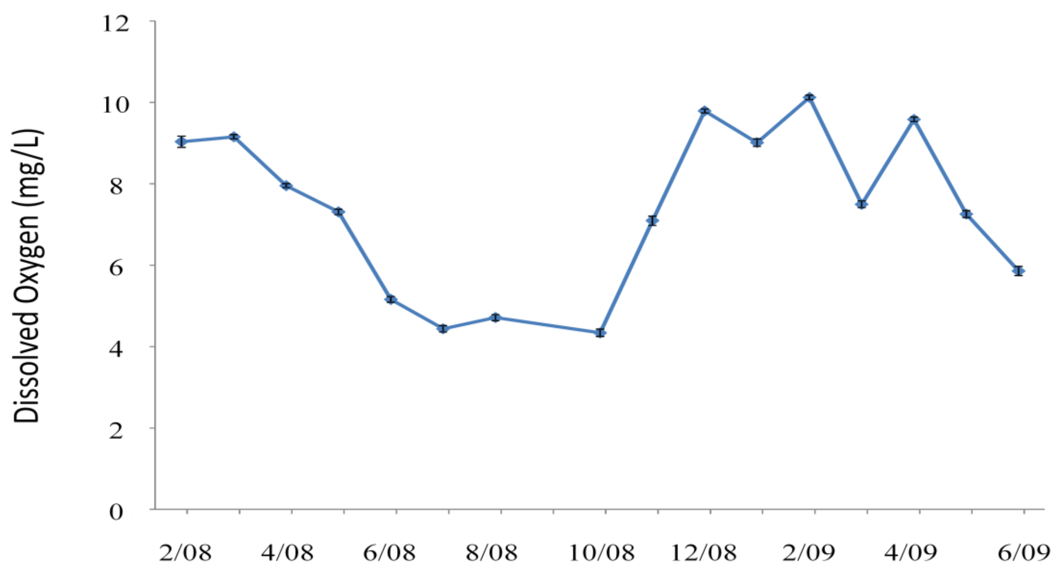


Figure 5.5. Mean monthly dissolved oxygen concentrations at University Lake.

Our results confirmed results in other regions that greater DO concentrations are observed during the winter (Hull *et al.*, 2008). This is mainly the result of lower temperatures during the winter. The variation of lake parameters can be considerably different when viewed at different time scales or during different time periods. This is especially true for dissolved oxygen concentration variation. It was observed that summer daily total variation, or the wave height, was greater than daily winter variation (Figure 5.6). Similar results were observed by other researchers (Xu *et al.*, 2010) in an intensive study on diurnal dissolved oxygen dynamics conducted in a forested stream in central Louisiana. This is most likely due to an increase in both primary production and respiration.

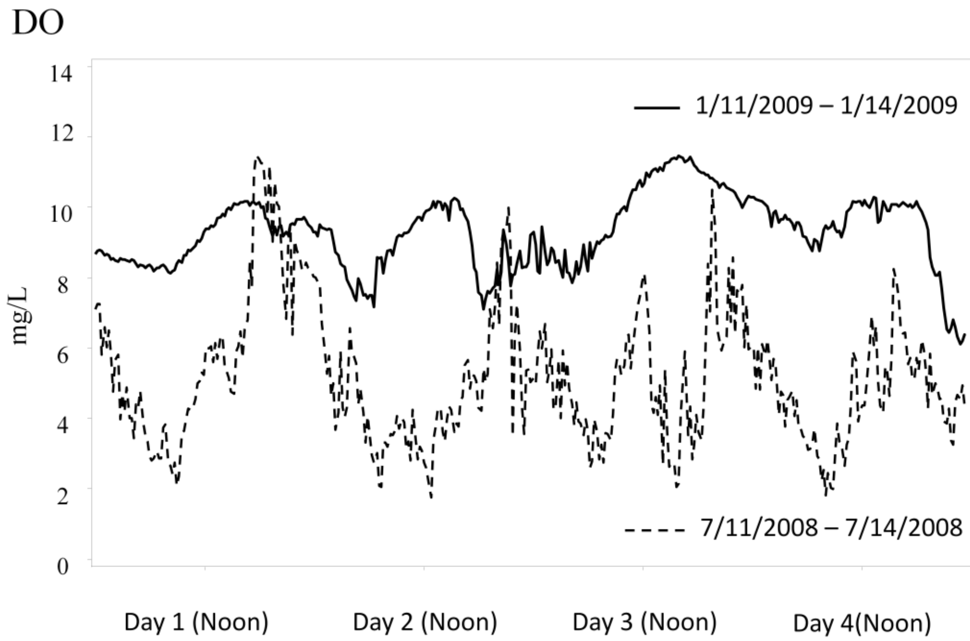


Figure 5.6. Difference in diurnal fluctuation of DO during winter and a summer season at University Lake.

D'Autilia *et al.* (2004) found that the rate of oxygen accumulation is faster than its utilization, particularly in the early morning. This was observed in our study at the daily time frame, particularly during the winter months. This is the result of smaller respiration rates during the winter.

Sampled dissolved oxygen concentrations had a mean monthly event concentration of 7.31 mg/L ranging from 4.75 to 10.78 mg/L in July 2008 and December 2009 respectively. Sampled dissolved oxygen concentrations had a mean storm event concentration of 6.95 mg/L ranging from 4.68 to 15.18 mg/L in August 2009 and December 2009 respectively. Overall, there was no significant difference between DO concentration before and after sampling (t-test, $n = 12$, $p = 0.36$) (Figure 5.7).

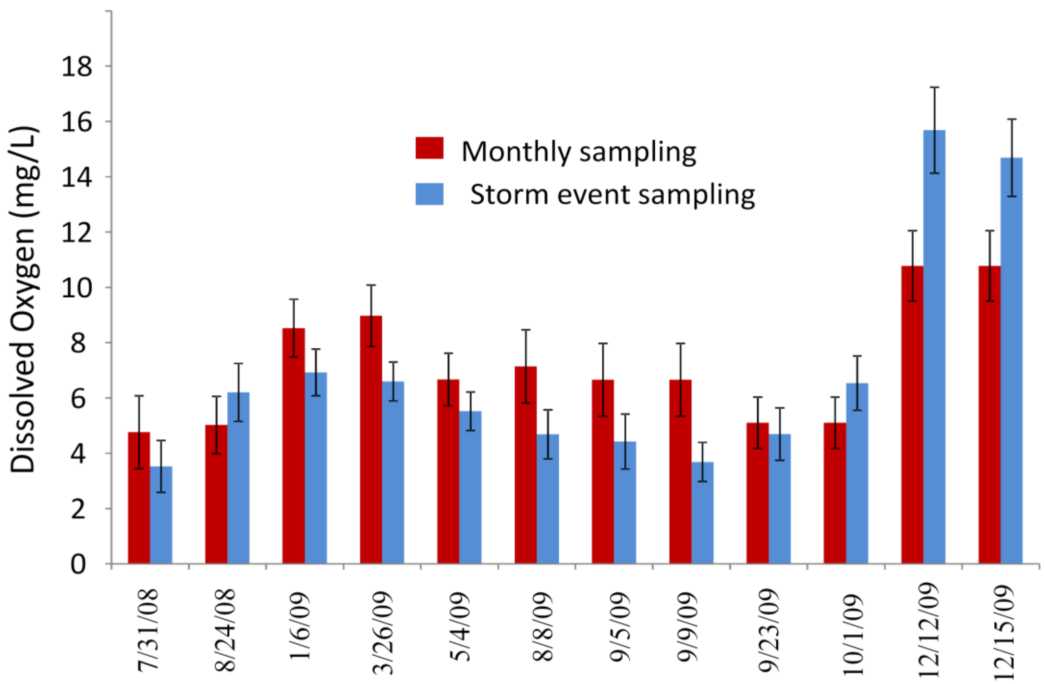


Figure 5.7. Comparison of measured dissolved oxygen concentrations before and after storm event at University Lake.

Throughout most of the year storm events decreased lake DO concentrations. This effect is most likely the result of the effect of storm events on lake temperature. Well below average temperatures were observed throughout most of December 2009, which could have resulted in increases in DO concentrations within University Lake. If the month of December 2009 is omitted from analysis, storm events would have resulted in a significant decrease in dissolved oxygen concentrations (t-test, n = 10, p = 0.03).

5.3.2 Long-Term Metabolism

During the study period, 299 days were recorded where continuous sampling occurred throughout the day in order to calculate respiration and productivity rates. During the month of September 2008, technical problems prevented accurate measurement of dissolved oxygen and consequently, data was omitted from analysis. After screening (R>0, GPP>NPP) 246

observations remained for analysis, therefore 82.3% of the data passed screening. Of the data points removed only 40.4% received rainfall (>0.25 cm) or above average wind speeds (10.14 kph). Data that did not pass screening was probably the result meteorological factors or the result of mechanical malfunction.

The average of mean monthly net primary productivity (NPP) was 2.13 g O_2/m^2/day. Values ranged from 0.83 in November 2008 to 3.79 g O_2/m^2/day in June 2009 (Figure 5.8a). Estimated annual sum of NPP was 780 g O_2/m^2/year. Although NPP did appear to exhibit a seasonal trend, summer (April – October) and winter (November – March) NPP was not significantly different (t-test, df=236, t=1.79, p= .075). Mean monthly R was 5.90 g O_2/m^2/day ranging from 2.38 to 11.77 g O_2/m^2/day in November 2008 and June 2009 respectively (Figure 5.8b). Estimated annual sum of R was 2150 g O_2/m^2/year. Respiration was significantly greater during the summer months (t-test, df=234, t=6.87, p= <.0001). Mean GPP value was 4.41 g O_2/m^2/day ranging from 1.96 to 7.68 g O_2/m^2/day in November 2008 and June 2009 respectively (Figure 5.8c). Estimated annual sum of GPP was 1610 g O_2/m^2/year. GPP did exhibit a seasonal trend. Summer (April – October) rates were significantly greater (t-test, df=238, t=3.57, p= .0004) than rates during the winter (November – March). Respiration was observed to have the most variation throughout the year followed by GPP, then productivity.

Calculated metabolic rates appear to fluctuate substantially depending on time period, location, and calculating method. Wang (2003) observed photosynthetic rates ranging from 9.6 to 74.8 g /m^2/day in an agricultural environment and 4.4 to 12.6 g O_2/m^2/day in a metropolitan environment using the “extreme value method.” Respiration rates ranged from 17.24 to 169.88 g O_2/m^2/day in an agricultural environment and 23.01 to 57.74 g O_2/m^2/day in a metropolitan environment. Lopez-Archilla *et al.* (2004) list the GPP and respiration values from many studies

from various shallow lake studies within the United States and throughout the world. Our results tended to agree with multiple previous studies that found that GPP and respiration rates drop from July until November (Cornell and Klarer, 2008). However, this is in part due to our short study period. Given a longer study period we would have a more accurate seasonal trend in metabolic rates.

a)

b)

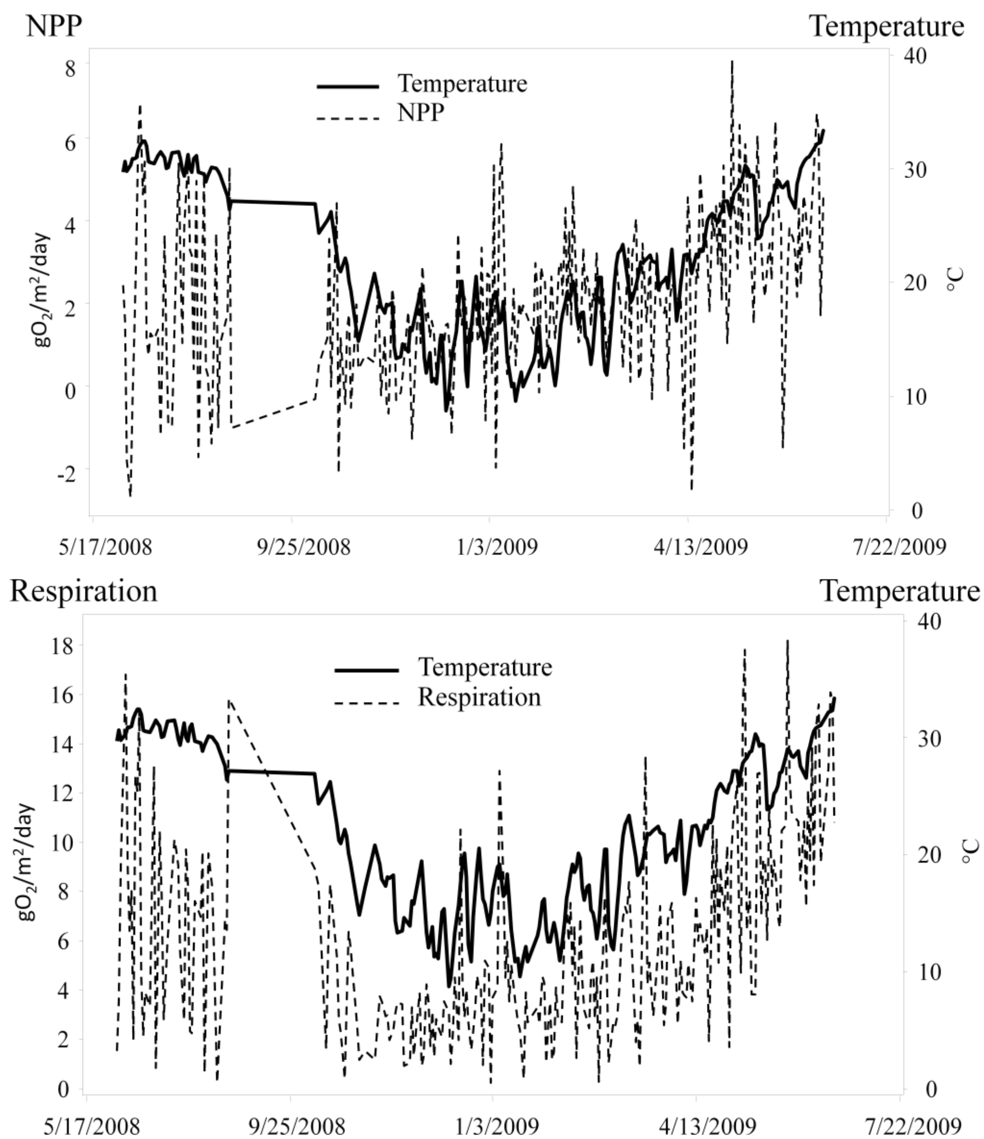


Figure 5.8. Relationship of net primary production (a), respiration (b), and gross primary production (c) with water temperature at University Lake.

c)

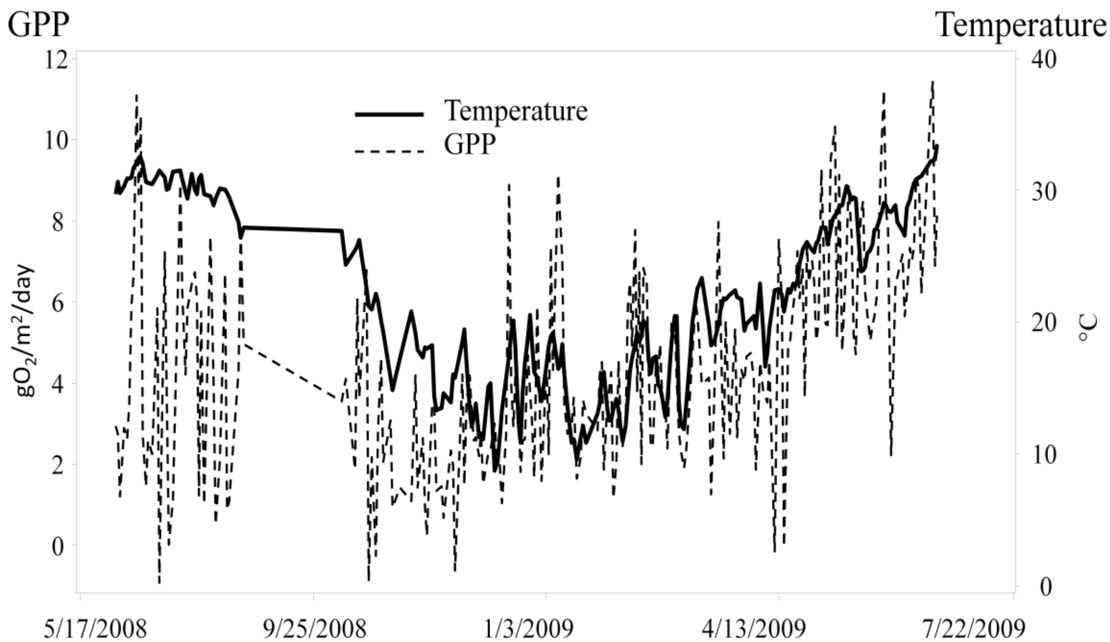


Figure 5.8, (continued).

5.3.3 Relationships of Productivity and Respiration with Environmental Conditions

Water temperature had a monthly mean of 22.6 °C, ranging from 14.4 to 31 °C in December 2008 and July 2008 respectively. Trends in GPP and water temperature were closely related when analyzed at an annual time frame (Figure 5.8c). Both variables had a distinct seasonal variation with greater values during the summer months and smaller values during the winter months. Mean daily water temperature explained 20% (regression, n=246, r^2=0.20) of the variation in calculated GPP (Figure 5.9a). Respiration also exhibited seasonal variation and was better correlated with water temperature (Figure 5.9b). Mean daily water temperature explained about 37% (regression, n=246, r^2=0.37) of the variation in the calculated respiration (Figure 5.9c), while only explaining about 7% (regression, n=246, r^2=0.07) of productivity.

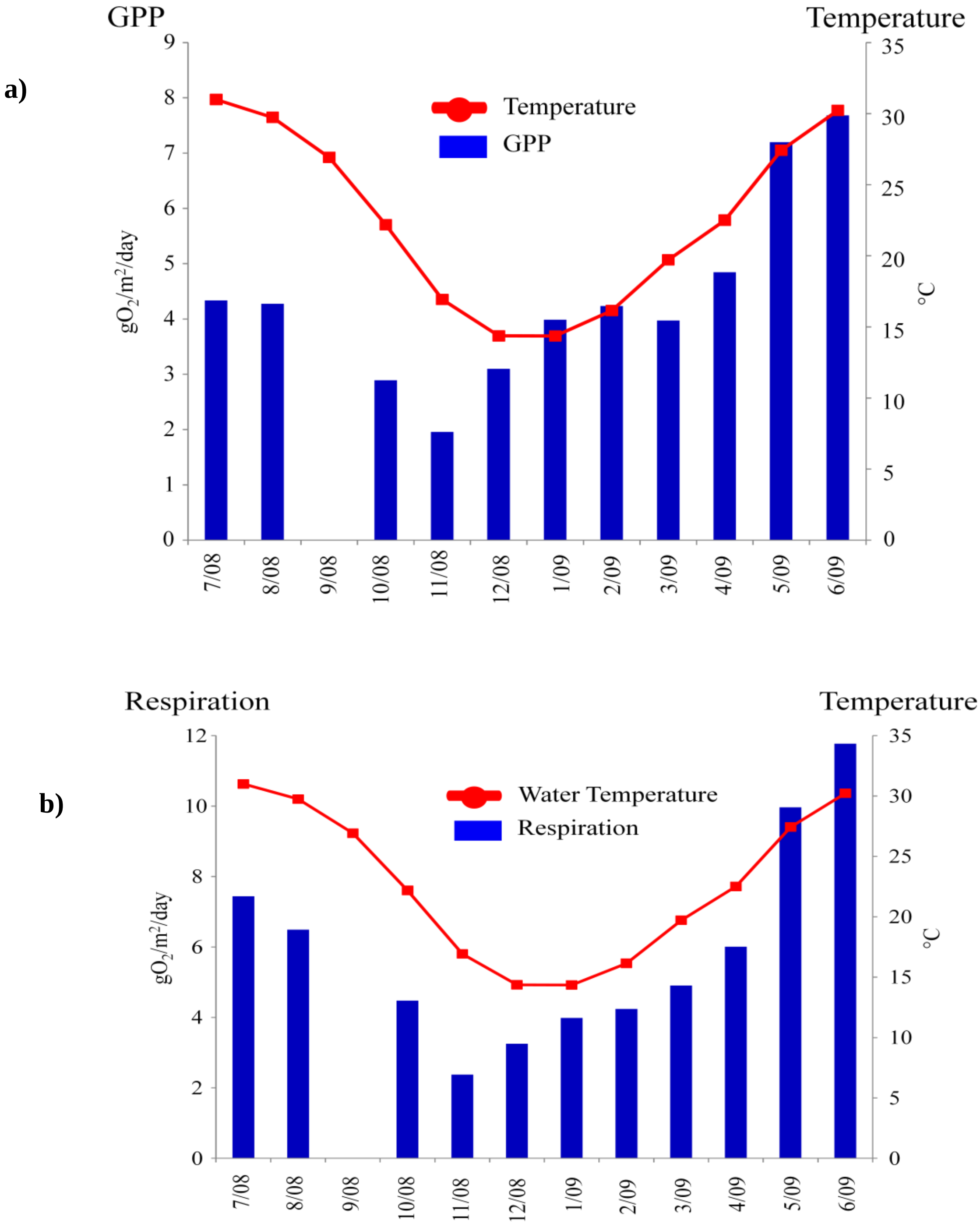


Figure 5.9. Relationship of average gross primary production (a) and respiration (b) with water temperature at University Lake.

Our results display a correlation between water temperature and metabolic processes. This supports previous research, because it is understood that biologic processes are accelerated with increasing temperature. Being such a shallow lake environment, our results also support Hull *et al.* (2008) who stated that there is a strong correlation between oxygen consumption at the sediment layer and overhanging water temperature.

Solar radiation also depicted a highly seasonal variation. Solar radiation explained only 15% (regression, n=246, r^2=0.15) of GPP (Figure 5.10a), 19% of respiration (regression, n=246, r^2=0.19) (Figure 5.10b), and only 8% of net productivity (regression, n=246, r^2=0.08) (Figure 5.10c).

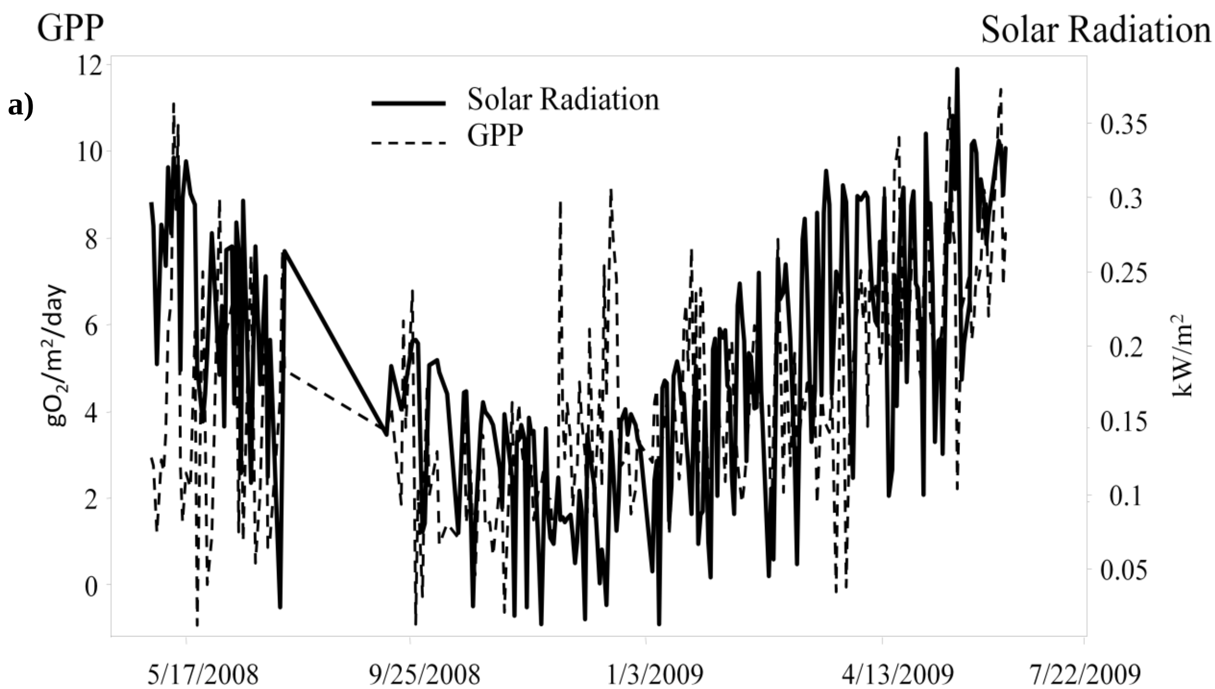
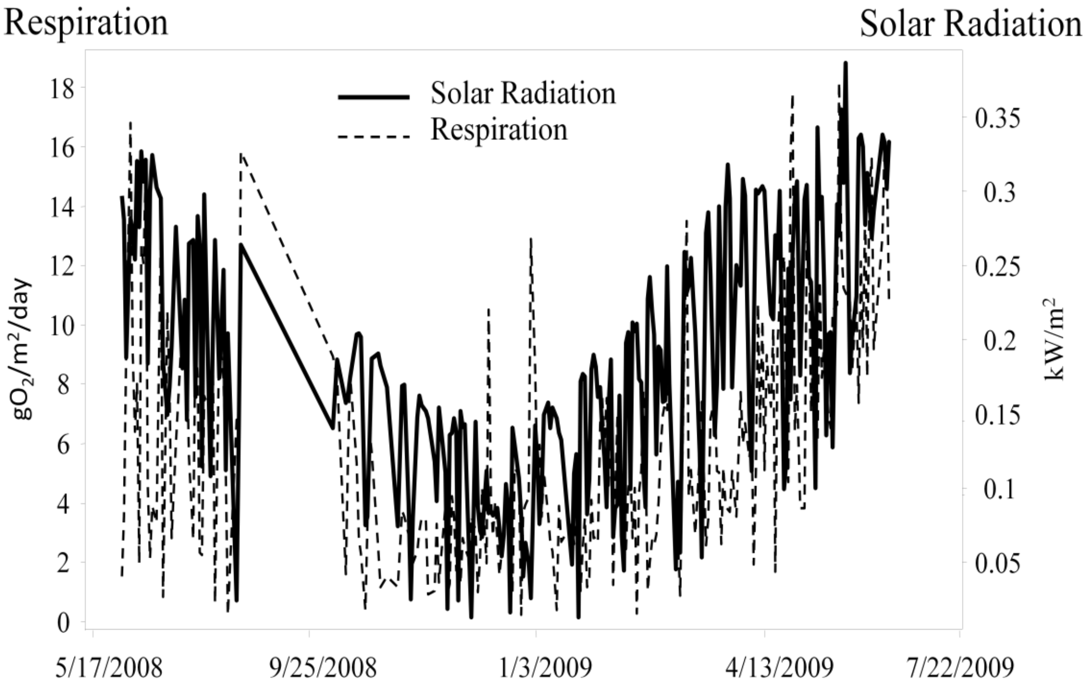


Figure 5.10. Relationship of gross primary production (a), respiration (b), and net primary production (c) with solar radiation at University Lake.

b)



c)

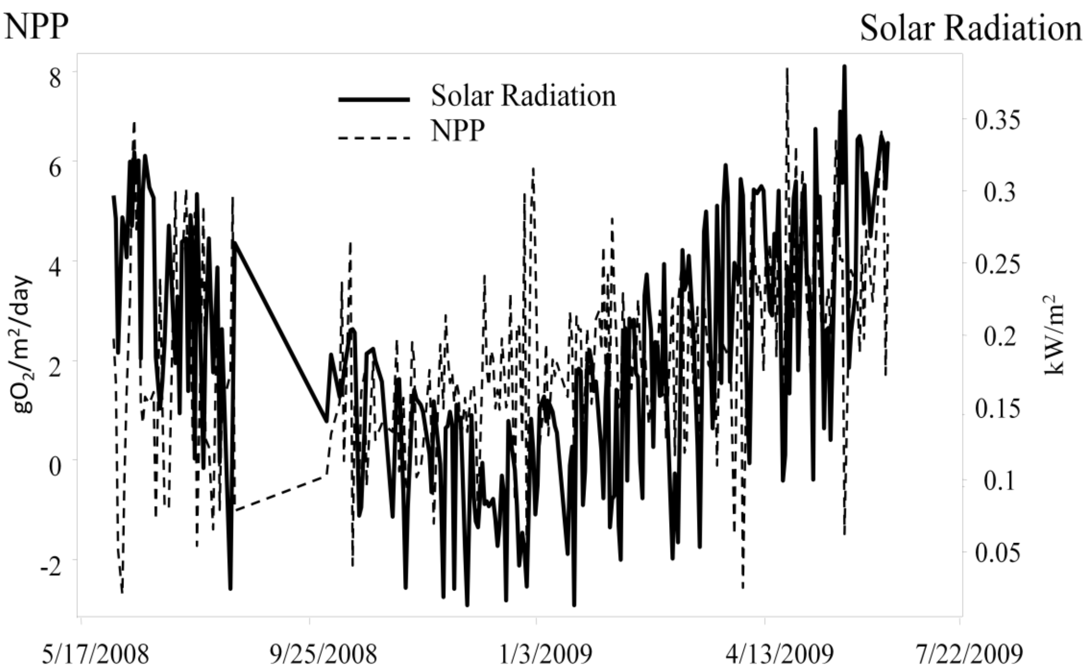


Figure 5.10, (continued).

When analyzing the seasonal relationship between metabolic processes and environmental conditions it may be easier to use monthly means. When comparing GPP and solar radiation (Figure 5.11a) we are able to observe the lowest monthly average of GPP (1.96 g O_2 / m^2 /day) occurred in November 2008 and rates increased during the winter, when solar radiation and water temperature are at their minimums. A similar seasonal pattern was observed for rates of respiration (Figure 5.11b). This same pattern was observed for the relationship between water temperature and GPP (Figure 5.9a) and water temperature and respiration (Figure 5.9b).

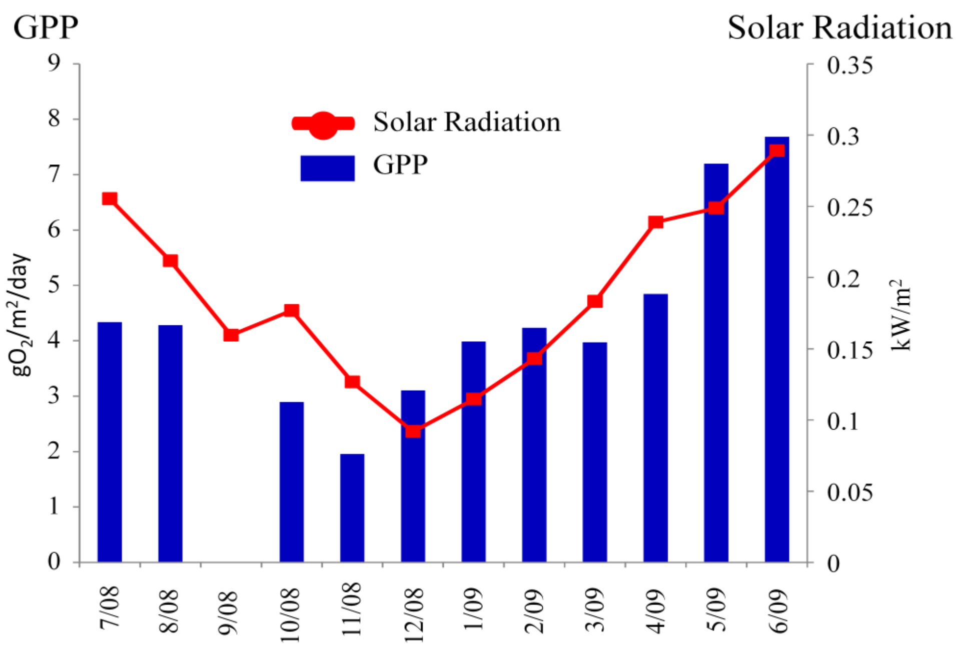
Mean BOD_5 during the study period was observed to be 7.2 mg/L ranging from 5.4 to 9.5 mg/L in March 2009 and June 2009 respectively (Figure 5.12). The nitrogen inhibited BOD_5 had a mean value of 3.9 mg/L ranging from 2.6 to 5.4 mg/L in March 2009 and August 2008 respectively. BOD_5 results explained 18% of the variability in GPP (regression, n=11, r^2=0.18) (Figure 5.13a), and 39% of respiration (regression, n=11, r^2=0.39) (Figure 5.13b).

These relatively high r^2 values are somewhat discredited by the small sample size. Figure 8 shows the monthly average of respiration and BOD_5. This graph does show a seasonal response to BOD_5, however the trend does not follow respiration closely. The BOD_5 values were on average comparable but less variable than results observed by Arora and Mehra (2009) found in a shallow urban lake located in Delhi, India.

Average depth during the study period was 90.0 cm ranging from 73.9 to 98.9 cm observed in November 2008 and September 2008 respectively. Figure 5.14 shows the monthly average GPP and monthly average water depth. Overall depth only explained 3% of the

variability in GPP (regression, n=246, r^2=0.03). However, when the water level was below average (90.0 cm) it explained 12% of variability (regression, n=104, r^2=0.12).

a)



b)

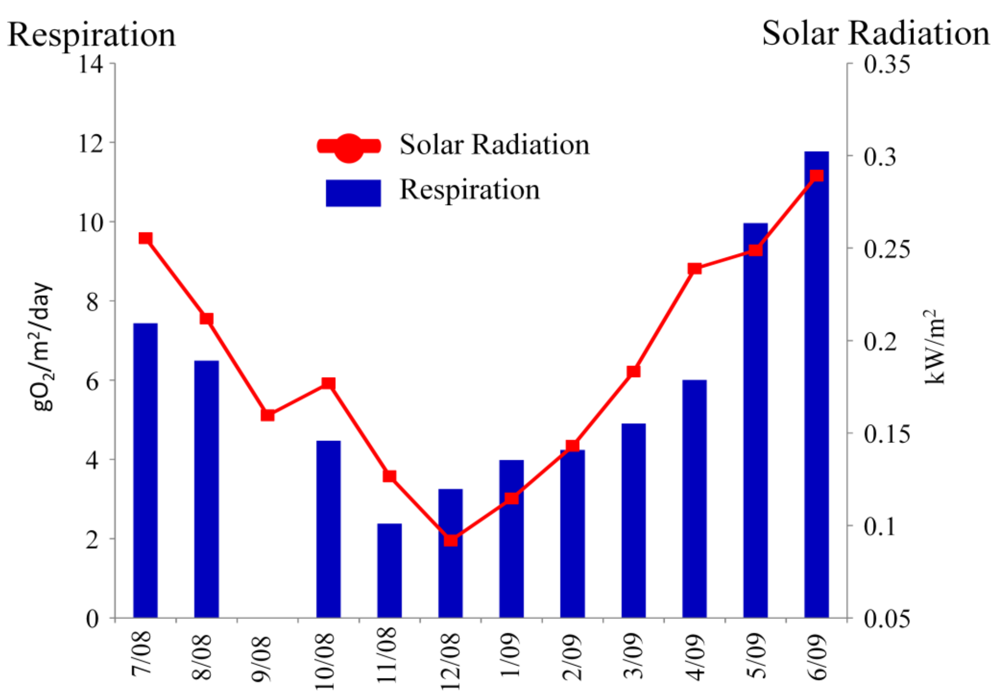


Figure 5.11. Relationship of mean monthly gross primary production (a) and respiration (b) with solar radiation at University Lake.

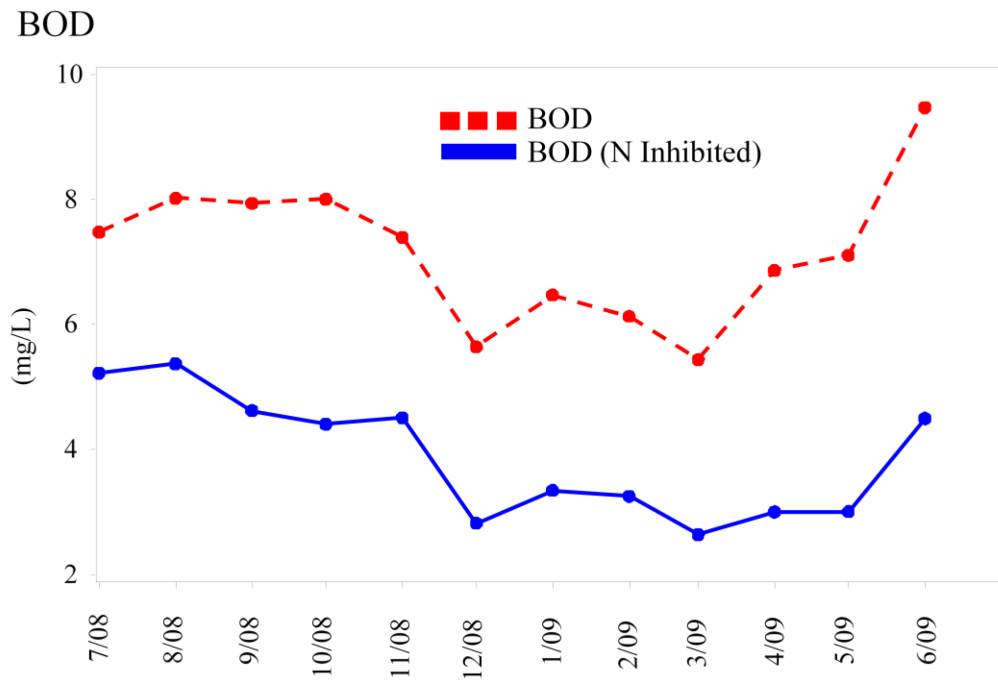


Figure 5.12. Monthly BOD_5 and nitrogen inhibited BOD_5 at University Lake.

a)

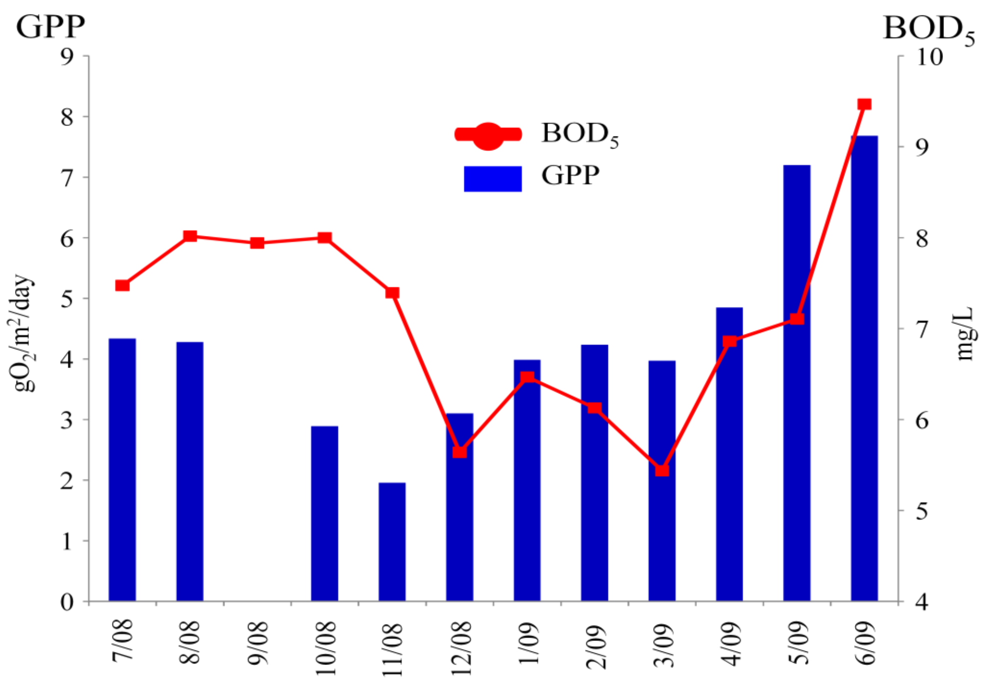


Figure 5.13. Relationship between gross primary production (a) and respiration (b) with BOD_5 at University Lake.

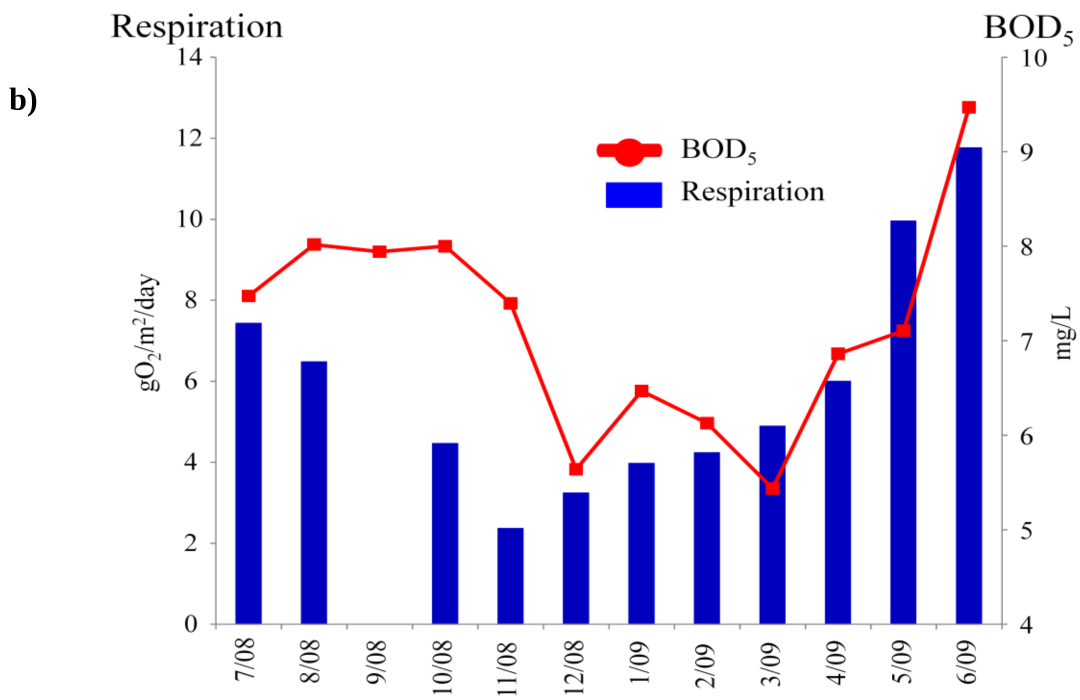


Figure 5.13, (continued).

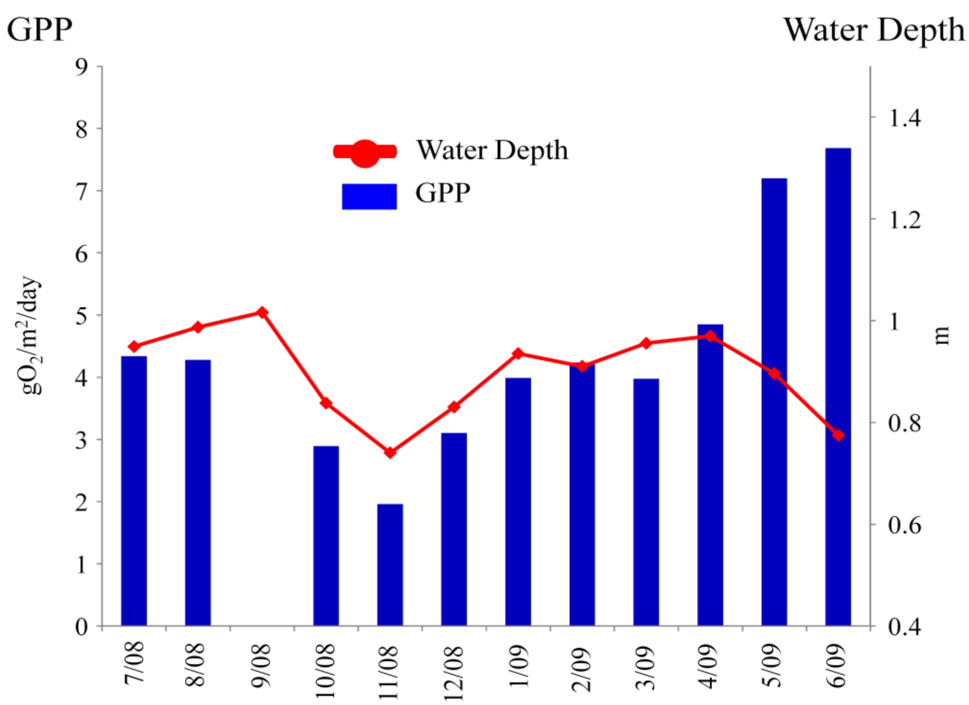


Figure 5.14. Relationship between mean monthly water depth and gross primary production (GPP) at University Lake.

The discrepancies between the correlations of metabolism and water temperature and solar radiation during the winter may in part be due to low water levels resulting from below average precipitation. Low water levels have shown to affect the alkalinity and nutrient concentrations of shallow eutrophic lakes which can consequently effect the phytoplankton composition of a lake (Nõges and Nõges, 1999). Robinson *et al.* (1997) found that with increasing water depth phytoplankton productivity increased in a prairie wetland. This means that water depth becomes an important variable controlling metabolic rates at low water levels, due to the water column's proximity to the sediment layer. It must also be considered that another variable that has not been measured is affecting metabolic rates during the fall and winter season.

Mean monthly *GPP* was plotted against respiration in Figure 5.15a portraying insight to the heterotrophic and autotrophic nature of the study site. Figure 5.15b depicts the ratio of mean monthly *P*/*R* over time. A ratio of < 1 implies that the system is net hetrotrophic, meaning that organic carbon from other ecosystems is supporting the lake ecosystem. When the *P/R* ratio is > 1, the system is net autotrophic, and is producing enough organic carbon to support the lake ecosystem.

Results indicate that the *P*/*R* ratio was less than one throughout most of the year, indicating that respiration was greater than *GPP* throughout most of the year. However, during the winter months of January and February 2009 *GPP* was nearly equal to respiration. These results indicate that University Lake, a shallow urban lake, is net heterotrophic throughout most of the year. Winter months were observed in the lower left side of Figure 5.15a, indicating less respiration and productivity occurring during that time period, while the summer months, in the upper right of the Figure 5.15a, indicate greater values of respiration and productivity. In

addition, Figure 5.15b shows a greater P/R ratio during the winter months while during the spring and summer months. This indicates a greater net production of carbon during the winter.

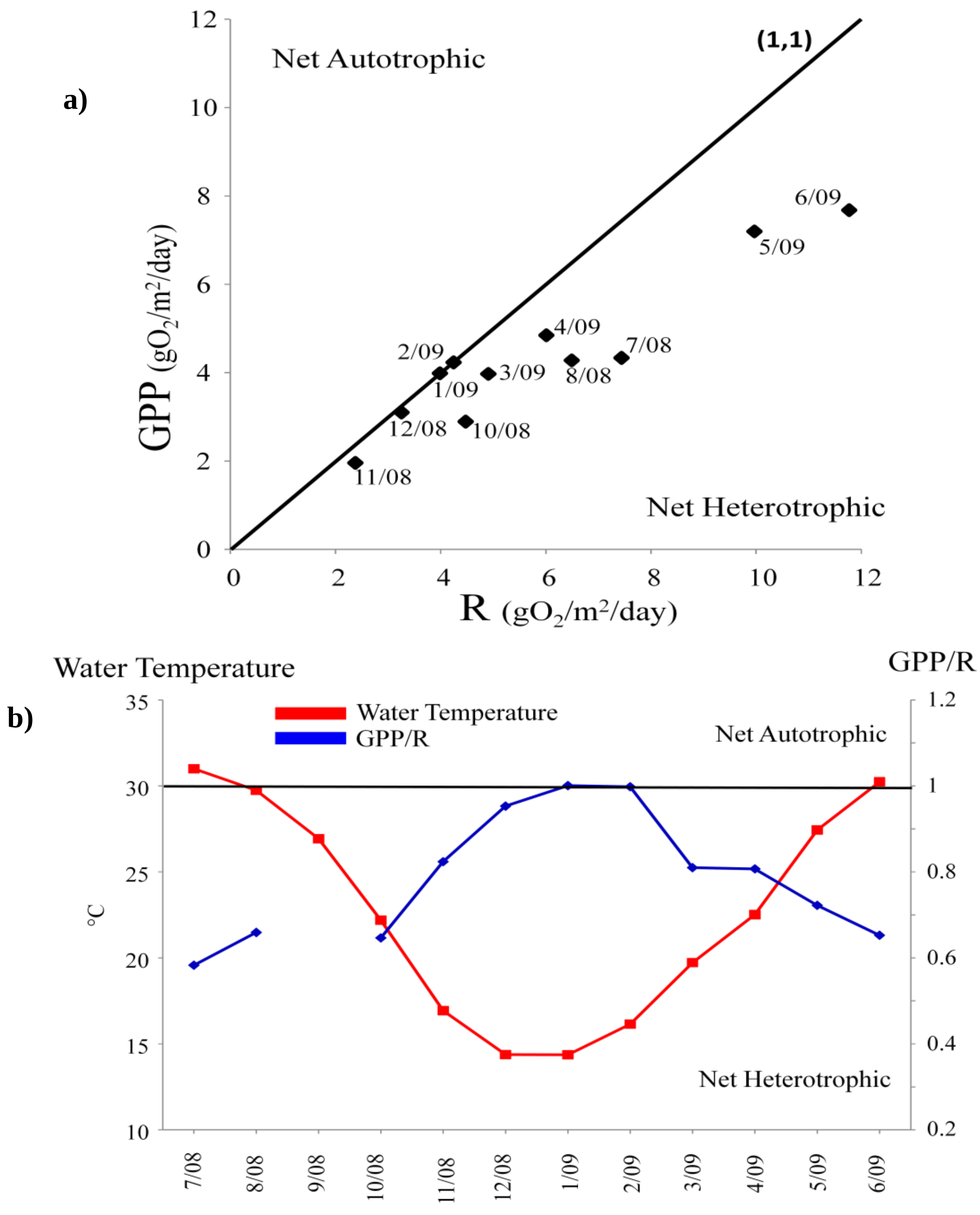


Figure 5.15. Ratio of mean monthly gross primary productivity (*GPP*) and respiration (*R*) (a), and *P/R* ratio through time (b) at University Lake.

P/R ratios < 1 can be common in unproductive oligotrophic lakes (Urabe et al., 2005). However, *P/R* ratio may be dependent on the geographic region and method of calculating metabolic rates. An urban stream has been observed by Wang *et al.* (2003) to remain heterotrophic throughout their study period (July-September) in Indiana, USA, while a non-urban site became autotrophic. In urban lake environments surface runoff makes up a large percentage of the water budget. Chapter 3 presents volume measurements of stormwater runoff from storm events. Stormwater runoff transports many nutrients including nitrogen and phosphorus, and organic matter from the surrounding landscape which can have an impact on respiration and production (Brezonik and Stadelmann, 2002, Mctammany *et al.,* 2003). This influx of nutrients and organic material has most likely lead to respiration being greater than productivity in this shallow subtropical urban lake throughout most of the year. The tendency of smaller metabolic rates to occur during the winter while greater rates occurring during the summer is most likely primarily determined by temperature, due to metabolic rates running faster at higher temperatures (Whitehead *et al.,* 2009).

5.4 Conclusion

Our results have shown that urban, shallow, subtropical lakes may display seasonal variation similar to that of many deeper stratified lakes in regard to oxygen dynamics. Dissolved oxygen concentrations were significantly greater and less variable during the winter months compared to smaller more variable concentration during the summer months. Mean monthly net primary production (*NPP*) was 2.13 g O_2/m^2/day ranging from 0.83 to 3.79 g O_2/m^2/day in November 2008 and June 2009 respectively. Mean monthly respiration (*R*) was 5.90 g O_2 / m^2 /day ranging from 2.38 to 11.77 g O_2/m^2/day in November 2008 and June 2009 respectively. Our mean gross primary production (*GPP*) value was 4.41 g O_2/m^2/day ranging from 1.96 to 7.68 g

O_2/m^2/day in November 2008 and June 2009 respectively. Metabolic rates displayed seasonal variation mainly due to changes in water temperature which explained 20 and 37% of the variability in *GPP* and respiration respectively. Biochemical oxygen demand (BOD), solar radiation, and water depth also appeared to influence lake metabolic rates in a shallow, urban, subtropical lake. Annually 1610 g O_2/m^2 were produced while the annual sum of *R* was 2150 g O_2/m^2. This urban study site was shown to be net heterotrophic throughout most of the year. During the winter months gross primary productivity was equal to respiration rates.

CHAPTER 6: SUMMARY

This thesis research assessed the current conditions of water quality in a subtropical shallow lake that is influenced by the highly developed urban environment. The research utilized data collected between July 2008 and December 2009 to achieve the following goals: (1) to quantify runoff effect on phosphorus and nitrogen loading, and its relationship to water quality parameters; (2) to determine runoff effect on changes of phosphorus, nitrogen, and dissolved oxygen concentrations; (3) to analyze seasonal variation of water quality parameters; and (4) to assess lake metabolism processes in a shallow urban ecosystem using intensive dissolved oxygen measurements.

The study on phosphorus dynamics in the shallow urban lake showed that storm events immediately caused an increase in TP mass within the system. Mean monthly lake TP concentration was 0.286 mg/L. Mean storm event loading of 28.1 kg of TP caused a mean increase in TP concentration by 14.1%. This resulted in a mean lake concentration of 0.383 mg/L of TP after storm events. Much of the variation in TP concentrations was due to internal phosphorus loading which led to increased concentrations primarily during the summer months. Phosphorus loads in the lake were correlated with runoff volume (r^2 = 0.71), suggesting runoff volume is the most important factor effecting lake TP concentration after a storm event. Storm events had a seasonal effect on the temperature of the waterbody; raising the temperature during the winter months and decreasing the temperature during the summer months.

The study on nitrogen in the lake assessed changes in nitrogen concentration after rainstorm events and during several seasons. Over the entire study period, University Lake showed low concentrations of all species of nitrogen, although there was a significant increase of

nitrate nitrogen and total Kjeldahl nitrogen following rainstorm events. Due to the high mean monthly concentration of phosphorus reported (0.286 mg/L), it is probable that this shallow lake system is nitrogen limited.

The results of the study on DO show that urban, shallow, subtropical lakes may display seasonal variation similar to that of many deeper stratified lakes in regard to oxygen dynamics. This includes greater and less variable DO concentrations during the winter months compared to smaller more variable concentration during the summer months. Mean monthly metabolic rates were 2.13, 5.90, and 4.41 g O_2/m^2/day for net primary production, respiration, and gross primary production respectively. The lowest values of all three metabolic rates occurred in November of 2008, while the greatest rates of metabolic activity occurred in June 2009. Annually 1610 g O_2/m^2 were produced while the annual sum of respiration was 2150 g O_2/m^2. This is consistent with studies in temperate lakes which have found a spring peak in primary production. Metabolic rates displayed seasonal variation mainly due to changes in water temperature which explained 20 and 37% of the variability in gross primary productivity and respiration respectively. The study site was shown to be net heterotrophic throughout most of the year. During the winter months gross primary productivity was equal to respiration rates.

This thesis research is only the beginning of an intensive study to determine the effect of an urban environment on water quality. Long term data is continuing to be collected on University Lake, which will allow further analysis of the adaptive processes of shallow eutrophic waterbodies.

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