TROPHIC STATE AND METABOLISM IN A SOUTHEASTERN PIEDMONT RESERVOIR

by

Mary Callie Mayhew

(Under the direction of Todd C. Rasmussen)

Abstract

Lake Sidney Lanier is a valuable water resource in the rapidly developing region north of Atlanta, Georgia, USA. The reservoir has been managed by the U.S Army Corps of Engineers for multiple purposes since its completion in 1958. Since approximately 1990, Lake Lanier has been central to series of lawsuits in the "Eastern Water Wars" between Georgia, Alabama and Florida due to its importance as a water-storage facility within the Apalachicola-Chattahoochee-Flint River Basin. Of specific importance is the need to protect lake water quality to satisfy regional water supply demands, as well as for recreational and environmental purposes.

Recently, chlorophyll *a* levels have exceeded state water-quality standards. These exceedences have prompted the Georgia Environmental Protection Division to develop Total Maximum Daily Loads for phosphorus in Lake Lanier. While eutrophication in Southeastern Piedmont impoundments is a regional problem, nutrient cycling in these lakes does not appear to behave in a manner consistent with lakes in higher latitudes, and, hence, may not respond to nutrient-abatement strategies developed elsewhere.

Although phosphorus loading to Southeastern Piedmont waterbodies is high, soluble reactive phosphorus concentrations are generally low and phosphorus exports from the reservoir are only a small fraction of input loads. The prevailing hypothesis is that ferric oxides in the iron-rich, clay soils of the Southeastern Piedmont effectively sequester phosphorus, which then settle into the lake benthos. Yet, seasonal algal blooms suggest the presence of internal cycling driven by uncertain mechanisms.

This dissertation uses existing data sets and analyses of lake-eutrophication mechanisms to develop an improved understanding of nutrient cycling in Southeastern Piedmont impoundments. Water quality data for Lake Lanier between 1967 to 2001 are used to characterize the frequency, location, and magnitude of algal blooms, and to develop alternative conceptual models of nutrient cycling. The data suggest that phosphorus is not permanently sequestered in lake sediments, and is occasionally released, causing harmful algal blooms.

Three mechanisms were considered that could account for sediment-phosphorus release, including: i) Anoxic release (or reduced sorption) of phosphorous from oxyhydroxides in benthic (hypolimnetic) sediments; ii) Alkaline release (or reduced sorption) of phosphorus from oxyhydroxides from epilimnetic sediments as photosynthesis causes pH to exceed 8; and iii) Biologically mediated release into the photic zone by recycling of algal biomass in the metalimnion, as algal biomass is bound to clay and settles. The clay-algal complex includes sorbed phosphorus which may be available to algae and this complex explains the characteristic, heterograde, dissolved oxygen profile in summer. This setting also explains the relatively low algal biomass, which underestimates the rate of photosynthesis, and therefore of primary productivity.

These observations suggest that traditional measures of lake eutrophication (e.g., chlorophyll *a* and phosphorus concentrations) should be supplemented with metabolic parameters (e.g., oxygen and carbon dioxide production and consumption) to better understand, predict, and manage biologic productivity in Lake Lanier.

INDEX WORDS: Phosphorus, Lake Metabolism, Nutrient Cycling, Water Quality, Lake Lanier

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by

Mary Callie Mayhew

B.S., The University of Wisconsin-Milwaukee, 1969

M.S., The University of Wisconsin-Madison, 1974

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Mary Callie Mayhew

Approved:

Major Professor: Todd C. Rasmussen

Committee: William P. Miller
Amy D. Rosemond
David B. Wenner

Electronic Version Approved:

Maureen Grasso
Dean of the Graduate School
The University of Georgia
December 2009

Dedication

This dissertation is dedicated to the volunteers of LakeWatch, who gave their time and talent to monitor and protect a resource they value, to the memory of my husband, Ed Mayhew, who believed in the power of "instant scientists - just add water", and to my parents and children, who have supported and encouraged me throughout the years.

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Chapter 1

Introduction

Lake Lanier, a US Army Corps of Engineers (USACE) reservoir located on the upper Chattahoochee River 40 miles northeast of Atlanta, (Figure 1.1), is the primary water supply for the rapidly developing corridor north of Atlanta, as well as a significant recreational and wildlife resource. The quantity and quality of water in Lanier has become increasingly important as regional development, drought, and downstream demand for water resources have brought pressure to bear on the reservoir.

The purpose of this dissertation is to evaluate traditional measures of water quality, as well as measures of trophic state and metabolism, to determine what gives the most accurate description of productivity and water quality in Lake Lanier, to develop a conceptual model of the processes driving productivity and metabolism of the reservoir, and to form a hypothesize about future trends. Specifically, the focus of this dissertation is to investigate the role of phosphorus cycling in metabolism and productivity, and whether much of the considerable phosphorus load is sequestered in sediments, or whether anaerobic and/or alkaline release from sediments may be fueling an increasingly productive system.

1.1 Water Quality Standards and Total Daily Maximum Loads

The presence of the Atlanta metropolitan area, which has grown from a population of two million in the 1960s to almost five million in 2006, has put tremendous pressure on the water resources of the region. Eighty percent of the water supply for the Atlanta metropolitan region comes from Lake Lanier and other surface waters of the Upper Chattahoochee River watershed (Beck et al., 2002). The effects of rapid growth have been exacerbated by periodic drought, which resulted in the water level in Lanier reaching a record low of 320.4 m (1050.8 ft) msl in December 2007. The drought has intensified the decades long "eastern water wars" among Georgia, Florida and Alabama over how much water should flow down the Chattahoochee River, and how much should remain in reservoirs, especially Lanier.

There is also intense interest in water quality and the permitting of discharges into the lake. This public scrutiny, as well as the Consent Decree Order issued by the US federal District Court in the suit by the Sierra Club against the US Environmental Protection Agency (USEPA), culminated in the development of water quality standards by the Georgia Environmental Protection Division (GAEPD), which were put in place in 1999. As a result of mean annual chlorophyll *a* values above the standards, three areas of the lake were placed on the §303D list of impaired waters in 2006.

Monitoring data and algal growth potential tests conducted for the USACE in 1978-1979 (ESE, 1981) and in 1991 and 1998 by the USEPA for the Clean Lakes Study (Hatcher, 1998), indicate that phosphorus is the limiting nutrient in Lake Lanier. In response to these studies, GAEPD established Total Maximum Daily Loads (TMDLs) for phosphorus in 2007 for the major tributaries: 80,740 kg/yr (178,000 lb/yr) for the Chattahoochee River, 53,524 kg/yr (118,000 lb/yr) for the Chestatee River, and 6,532 kg/yr (14,400 lb/yr) for Flat Creek (GAEPD, 2007). Monitoring began in 2008 to evaluate lake water quality over time.

1.2 Trophic State

Research on cultural eutrophication during the last 50 years has established the importance of phosphorus as the limiting factor in many lakes (Vollenweider, 1968, Carlson, 1977, OECD, 1982: Schindler, 1980). This work also established the relationship between phosphorus, chlorophyll *a*, and secchi depth, which has become the basis for many approaches to the measurement and management of water quality, including many trophic state indices (Dillon and Rigler, 1974; Carlson, 1977, Henderson-Sellers, 1984).

Trophic status in lakes is classically determined by measuring autochthonous primary productivity, the rate of internal production by algae within the water body. Direct measurement of productivity requires radiotracer studies, measurement of ATP production, in-situ light-dark bottle incubations, or other rarely performed procedures. More commonly, trophic status is inferred using indirect measures, such as chlorophyll *a* concentration, phytoplankton biovolume, diatom index, secchi depth, or concentration or loading of the limiting nutrient, often phosphorus, but also nitrogen, and a variety of others (Henderson-Sellers, 1984; Lind et al., 1993).

However, these parameters are not always adequate predictors of primary productivity, and may not be consistent with each other or with other water quality parameters such as hypolimnetic anoxia. Natural lakes show some seasonal inconsistency between parameters (Carlson, 1977; Kufel, 2001), and trophic indices based on phosphorus limitation do not work for nitrogen-limited lakes (Carlson, 1977). Impoundments exhibit even more disparate values for standard trophic parameters. A study on Canyon Reservoir in Texas that used 22 different chemical and biological indices classified the reservoir as oligotrophic using 11 of the indices, mesotrophic by 4, and eutrophic by 7 (Henderson-Sellers, 1984).

Roosevelt Lake, another impoundment in Texas, is oligotrophic based on chlorophyll *a*, and eutrophic based on total phosphorus (Lind et al., 1993). Lake Blackshear, a reservoir on the Flint River in Georgia, is oligo-mesotrophic to mesoptrophic based on chlorophyll *a*, and eutrophic based on total phosphorus and secchi depth (Cofer et al., 2003). The Clean Lakes Study (Hatcher, 1998) classified Lake Lanier as mesotrophic based on total phosphorus and meso-oligotrophic based on chlorophyll *a*. A study done by the USEPA (2007) described Lake Lanier as oligotrophic based on chlorophyll *a*.

Lind (1993) attributes the variability in calculated trophic status when different parameters are used to fundamental differences between impoundments and natural lakes: (1) uncoupling of nutrient supply and primary production due to factors such as binding of nutrients by sediments, short water residence times, limited light availability due to turbidity, density currents from incoming rivers that plunge below the metalimnion and move through the impoundment without releasing nutrients to the euphotic zone; and (2) the high degree of of spatial and temporal heterogeneity in impoundments-a major source of which is the the up-lake to down-lake gradient in light, nutrients, etc, due to riverine inputs.

1.3 Metabolic Measures of Productivity

Measures of system metabolism, the production/consumption of O_2 and/or CO_2, have also been used to describe productivity. Ideally, trophic state is the result of system metabolism that is driven almost entirely by autochthonous production in the trophogenic zone. Limiting factors, often phosphorus in freshwater systems, determine production of algal biomass in the epilimnion. Low algal biomass results in little organic matter settling into the trophyolytic zone, resulting in low respiration in the hypolimnion, which remains oxic. Higher algal production in the epilimnion results in higher rates of respiration in the hypolimnion, which becomes anoxic.

Hypolimnetic anoxia, and the resultant clinograde oxygen profile, is classically definitive of eutrophy. This premise does not take into account other processes that may affect metabolism, such as alloochthonous production in the watershed that enters the lake as particulate and/or dissolved organic carbon. Del Giorgio et al. (1999) and den Heyer and Kalff (1998) found that one metabolic parameter, the rate of areal hypolimnetic oxygen deficit (AHOD) is not strongly coupled to algal biomass in the epilimnion. Del Giorgio et al.. (1999) attribute this to allochthonous organic material from the watershed, which increases respiration without increasing in-lake algal productivity. Wetzel (2001) notes that hypolimnetic CO_2 production, as well as oxygen deficit, may overestimate trophic state due to allocthonous inputs, or may underestimate it if an appreciable amount of organic production in the epilimnion is respired in the epilimnion and thus does not contribute to metabolism in the hypolimnion.

System metabolism, as measured by CO_2 production, may also be overestimated if groundwater contributes a significant amount of CO_2. CO_2 production has the important advantage of measuring anaerobic as well as aerobic respiration, which DO consumption cannot measure (Wetzel, 2001). . The disadvantage of using O_2 consumption, in addition to the ones discussed above, is that, since it measures only aerobic respiration, it is not useful if the hypolimnion becomes completely anoxic. Research on metabolism in lakes has traditionally focused on respiration, often as hypolimnetic oxygen deficit. Anoxia is extensive in many impoundments, including Lakes Blackshear and Lanier, indicating higher productivity and trophic state.

The other aspect of respiration, the production of CO_2 in the hypolimnion, can also be used as a measure of lake metabolism. However, it has not been used as extensively as hypolimnetic oxygen deficit. The interest in global climate change has focused attention on metabolic processes in aquatic systems, freshwater as well as marine, as sources and sinks of CO_2.

It is also possible to measure net ecosystem production (NEP) in the epilimnion by measuring the production and/or consumption of both metabolic parameters, O_2 and CO_2. Prairie et al. (2002) compared difference from saturation for O_2 and CO_2 in the epilimnion of several lakes to determine net hetero- or autotrophy. Their results, and those of other researchers (Cole et al., 1994; Cole et al., 2000; Del Giorgio, 1998) showed that most lakes, regardless of measured trophic status, were net heterotrophic. Carignan et al. (2000), however, found net epilimnetic autotrophy in the oligotrophic lakes they studied.

In spite of the uncertain relationship between metabolism and trophic state, metabolic parameters provide valuable information on the over-all production and respiration of a system, whether the origin of the organic material is the water body or the watershed. Metabolic measures, therefore, show disparity with other trophic indicators as extensively and as often as other parameters. However, they are integrators of ecosystem processes, in the sense that the total amount of organic matter, autochthonous or allocthonous, contributes to system metabolism.

It is clear that the relationship between trophic state and epilimnetic metabolic processes is no less ambiguous than the relationship between trophic state and hypolimnetic metabolic processes. A clearer picture of overall system productivity, if not of classic trophic state, may emerge if metabolic processes in the tropholytic, as well as the trophogenic, zone are examined. Hypolimnetic anoxia is common in reservoirs, but is often not considered in water quality standards.

An understanding of system metabolism, whether it is based on internal productivity or includes subsidized productivity from the watershed, may be more useful than traditional classifications of oligotrophy, eutrophy or mesotrophy for some purposes. Some knowledge of the rate and balance of metabolic processes is necessary to understand and manage the system intelligently.

1.4 Phosphorus Dynamics

The role of phosphorus in freshwater systems has been the subject of extensive research. Mortimer (1941) described the phosphorus cycle and the importance of iron in the binding of phosphorus under oxic, and its release under anoxic, conditions. The reduction and/or reversal of eutrophication due to a decrease in external phosphorus load in lakes with oxic sediments has been well documented (Edmondson and Lehman, 1981; Nurnberg, 1984), as well as the failure of phosphorus reduction programs due to continued internal loading from anoxic sediments (Gachter and Wehrli, 1998: Gachter and Muller, 2003).

Jensen et al. (1992) found that iron to phosphorus ratio (Fe:P) ratio in sediments is important in determining the rate of phosphorus release: an Fe: P ratio above 15, by weight, means that oxic conditions will likely control internal phosphorus loading. Other research has shown that anoxic release is not a universal phenomenon; some systems retain phosphorus under anoxic conditions, and others release it under oxic conditions (Caraco et al., 1993; Moosman et al., 2006). It has been proposed that sediments low in sulfate may not release phosphorus, and that sediments high is sulfate may do so when sulfate is reduced to sulfide, which permanently binds ferrous iron and prevents it from re-oxidizing and removing phosphorus from pore water (Caraco et al., 1993; Gachter and Muller, 2003).

Although external phosphorus load to Lake Lanier is high, phosphorus remains the limiting nutrient (ESEI, 1981; Hatcher, 1998), and anoxic release and internal loading of phosphorus has not been reported. Research (Reckhow, 1988; Mayhew et al., 2001; Parker, 2004) suggests that ferric oxy-hydroxides in clay soils characteristic of the Southeastern Piedmont bind phosphorus and sequester it in bottom sediments, even under anoxic conditions. Parker (2007) suggests that phosphorus may be released into pore waters under anoxic condition, but continued supply of ferric iron to the sediments, and mixing with ferric iron during mixis, re-binds any phosphorus released.

Release under oxic conditions occurs, but is not as common or extensive as anoxic release. One cause of oxic release is mineralization of organic matter, which also occurs, at lower rates, under anoxic conditions (Cole and Caraco, 1998). Another source of oxic phosphorus release is alkaline desorption (Lijklema, 1980).

Simulation of phosphorus sorption by ferric oxyhydoxides using MINTEQ (1991), a chemical equilibrium model for surface and ground water, shows that phosphorus is released from ferric iron as pH rises above 8. The effect of pH on phosphorus release from sediments has not received as much attention as anoxic release, perhaps because alkaline pH is not as common as circumneutral pH in many systems, especially over deeper sediments, where respiration often decreases pH.

However, elevated pH readings are not uncommon in soft water lakes, where photosynthesis drives pH to 9 and higher in the epilimnion (Wetzel, 2001). Many Southeastern Piedmont impoundments are such soft water systems; the surface pH in Lake Lanier may range from 7 to nearly 9.5 over a diurnal cycle in summer. During periods of high pH, the release of phosphorus bound to ferric iron suspended in the epilimnion, or in shallow sediments in contact with epilimnetic water, could fuel increasing rates of algal production.

1.5 Hypotheses

A conceptual model, shown in Figure 1.2, includes the relationships among metabolic processes, photosynthesis, respiration, in the water column and in the sediments, pH, phosphorus dynamics and oxic/anoxic conditions.

Specific hypotheses, and the chapters in which they are addressed, are that:

1. Parameters such as conductivity, secchi depth, chlorophyll *a* and epilimnetic phosphorus are consistent and indicate adequate water quality (Chapter 2);

2. Calculated trophic state varies with the parameter used to compute it because transparency, phosphorus and algal biomass (chlorophyll *a*) are not coupled in Lake Lanier (Chapter 3);

3. O_2 and CO_2 production/consumption, is the result of a higher, autochthonus metabolism than is reflected in chlorophyll *a* levels (Chapter 3);

4. Ninety percent of the phosphorus entering the lake is sequestered in the sediments (Chapter 4);

5. Phosphorus is released from, or sorbed more slowly by, ferric oxyhydroxides at elevated pH in the epilimnion and metalimnion during periods of intense photosynthesis (Chapter 4);

6. The low apparent algal biomass, and resultant low trophic status, is the result of binding of algae, bacteria, particulate and dissolved organic matter, and phosphorus by colloidal clay particles, which settle into the metalimnion, producing a positive heterograde oxygen profile in early summer, followed by negative heterograde profile as

respiration increases; causing accumulation and recycling at the metalimnion. (Chapter 4)

7. Organic matter accumulated in the metalimnion settles into the hypolimnion and sediments, producing hypolimnetic anoxic and sediment oxygen demand. (Chapter 4);

8. Current and historic information on O_2 production/consumption, CO_2 production, epilimnetic pH, redox conditions in hypolimnetic sediments and overlying waters, and sediment phosphorus concentration may provide a model for determining the temporal and spatial scale over which release of phosphorus may occur (Chapter 4).

1.6 Application

Lake Lanier is a USACE impoundment on the Chattahoochee River that is the primary water supply for the metropolitan Atlanta area and the developing corridor north of the city, as well as a recreational and wildlife resource. Water quality of this resource is of tantamount importance in the continuting economic well-being of the region. Lanier is a soft water system that supports an extensive fishery, but has relatively low chlorophyll *a* and phosphorus values, in spite of high phosphorus loading. As is often the case in reservoirs, trophic status varies spatically, temporally and with the parameter used, and hypolimnetic anoxia is well developed.

The explanation for low chlorophyll *a* values in reservoirs is usually light restriction due to suspended sediment by flow rates, for hypolimnetic anoxia, subsidies of organic material from the watershed; lack of internal loading of phosphorus, binding by interaction among ferric oxyhydroxides in Piedmont clay soil. pH, and algae. The purpose of this dissertation is to investigate these hypotheses, as well as the additional possibilities that phosphorus may be released from oxic, suspended or bentic sediments in the epilimnion, fueling algal productivity. Normally this should lead to higher chlorophyll *a* levels, however, the clay particles may settle phosphorus, algae and organic matter into the metalimnion, producing a heterograde oxygen profile into the metalimnion, and hypolimnetic anoxia as organic matter settles into the sediments.

Release or non-sorption of phosphorus at elevated pH, and possibly other, biotic mechanisms such mineralization by the microbial loop, may result in nutrient recycling in the metalimnion during the stratification period. The concentration of nutrients, algae and bacteria in the metalimnion, and subsequent settling into the hypolimnion may produce the considerable oxygen demand often seen in reservoirs. Such anoxia is often attributed to inputs of DOC and POC from the watershed, in systems with low apparent algal production. However, phosphorus release may be extensive enough to power a considerable autochthonous metabolism.

Management of the system has thus far focused on chlorophyll *a* and phosphorus levels, as measures of trophic state, and metabolic parameters such as hypolimnetic O_2 have not been considered. It is possible that the respiration of organic matter, and the production of it in the epilimnion, should be considered as indicators of ecosystem processes in the reservoir. Algal productivity is reflected more accurately by production/consumption of O_2 and production of CO_2 than by standing crop of chlorophyll *a*.

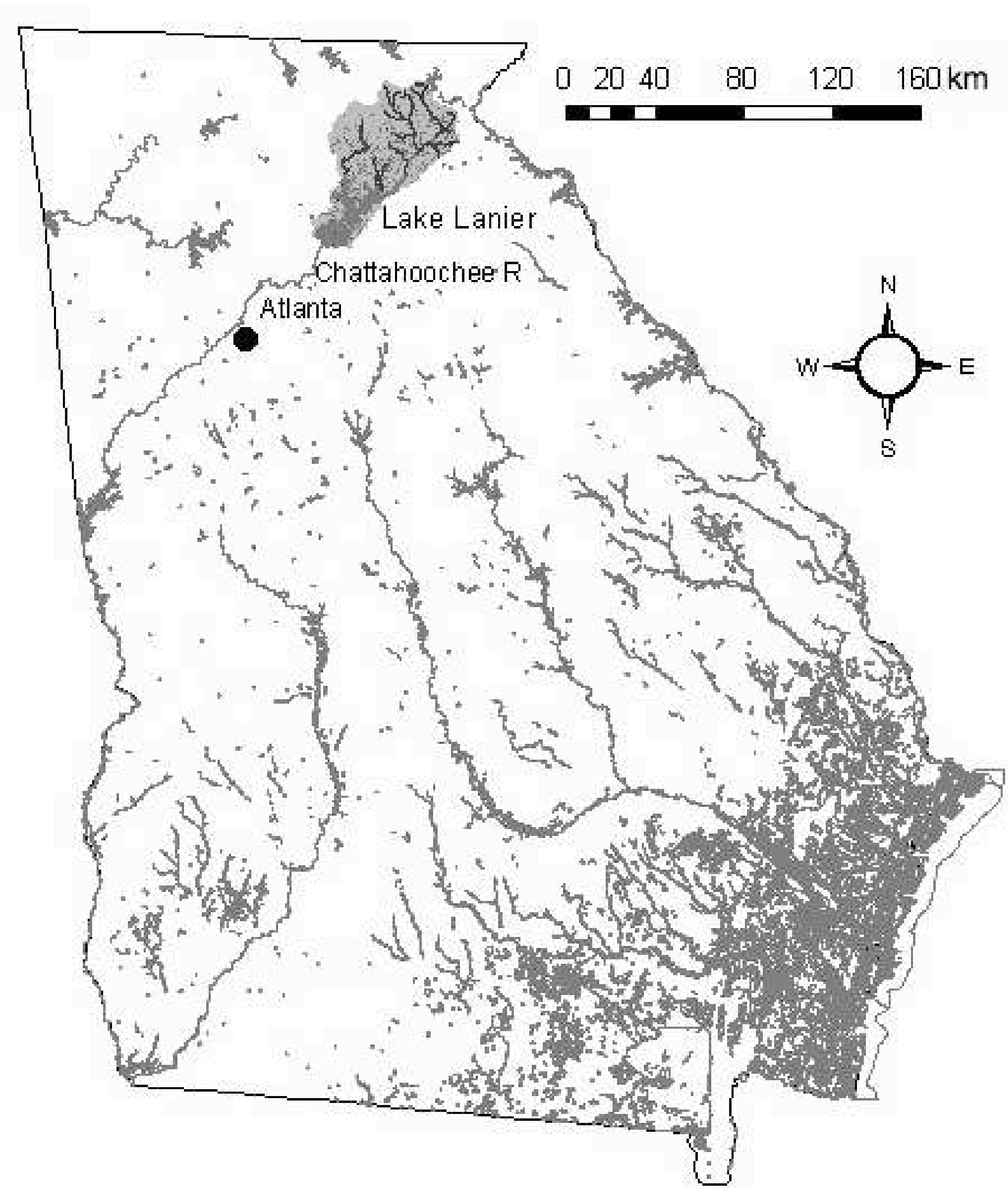


Figure 1.1: Location map for Lake Sidney Lanier and its watershed in Georgia.

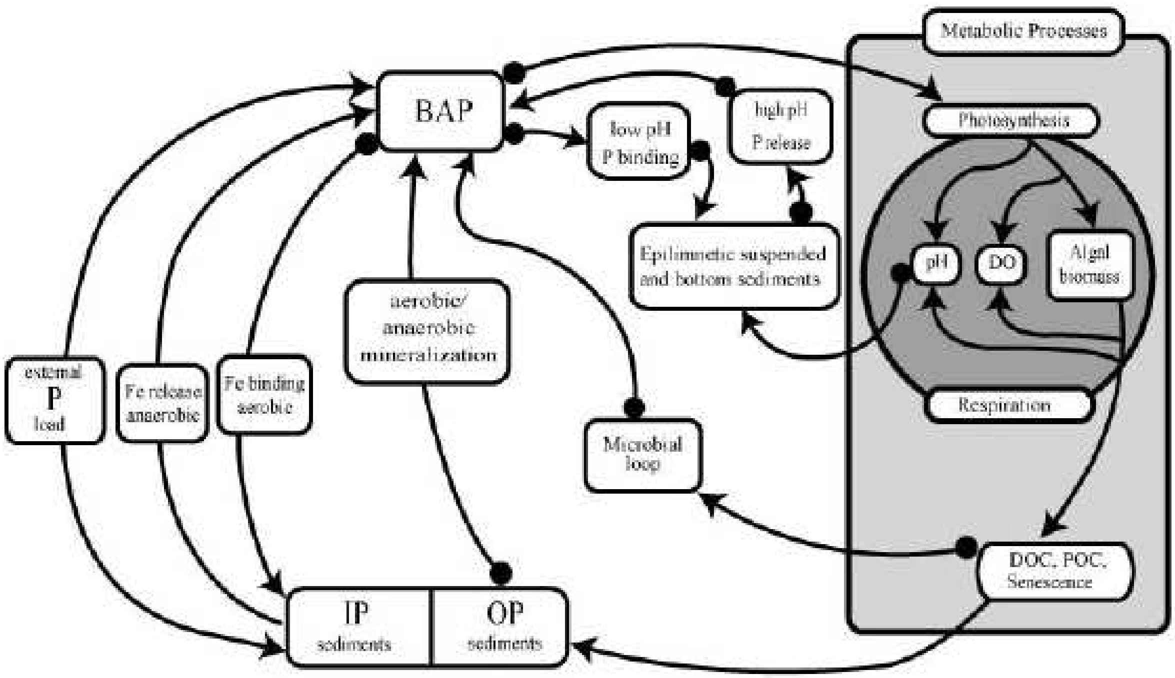


Figure 1.2: The hypothesized phosphorus cycle in a soft-water lake.

Chapter 2

Lake Lanier Water Quality, 1967-2002

Lake Lanier is a multi-purpose reservoir in the headwaters of the Chattahoochee River that is the primary water supply for the Atlanta metropolitan area, and is also used for disposal of treated sewage, flood control, recreation and wildlife habitat. The management of the reservoir has been the centerpiece of the *eastern water wars* among Georgia, Florida and Alabama over the allocation of water in the Appalachicola-Flint-Chattahoochee watershed among the thirsty, rapidly growing corridor north of Atlanta, and the downstream users in Georgia, as well as those in Alabama and Florida. The quantity of water stored in Lanier, or released for downstream users, is the main source of contention in the legal battle; however, regional development and the accompanying requirement for drinking water supply, has focused attention on water quality as well.

The purpose of this chapter is to examine historical data, as well as data collected for this study, for spatial and temporal patterns and/or trends in water quality in Lake Lanier. The correlation among parameters such as secchi depth, nutrient loading/ concentration, chlorophyll *a*, and dissolved oxygen level, which has been established to exist in many natural lakes, often do not exist in reservoirs (Henderson-Sellers, 1984; Lind, et al, 1993). The hypothesss addressed here is that traditional parameters such as conductivity, secchi depth, chlorophyll *a* and epilimnetic phosphorus are consistent and indicate adequate water quality.

2.1 Description of Study Area

Lake Lanier has a surface area of 153.8 km^2 (37,000 acres), a shoreline of 869 km (540 mi), and a watershed of 2,704 km^2 (1,044 mi^2). Maximum depth is 48.8 m (160 ft) near the dam at normal pool level of 326.4 m (1071 ft) msl. Lanier lies in the piedmont of the Appalachian mountains (Figure 1.1), an area of metamorphic, crystalline bedrock, overlain by weathered soils with a high clay content. The surface water in this region is low in mineral content and very soft, with a hardness of less than 10 ppm (Holder, 1967). The lake is monomictic, turning over in late November or early December and mixing until stratification begins in late March or early April. The hydraulic residence time is approximately 460 days.

Construction of the Lake Lanier was authorized by Congress in 1946 for the purposes of flood control, power production and navigation. The earthen dam and saddle dikes, completed in 1956, were constructed by the USACE, and the lake reached full pool in May 1959. Congress reauthorized Lake Lanier in the 1990s for the purposes of flood control, power production, navigation, water supply, recreation, and fish and wildlife management.

The Lake Lanier watershed encompasses Gwinnett, Dawson, Forsyth, Habersham, Hall, Lumpkin, and White counties. These counties have experienced rapid economic and population growth in recent years. Population in the study area increased 20.4 percent between 1980 and 1986 while the state's population increased only 11.7 percent during the same time period. Lake Lanier, as the most popular of the Corps of Engineer's projects nationwide, has shown an increasing trend in annual visits to the lake. Recreation is the biggest revenue producer on the lake, generating $2 billion per year. Lakeside property values have also increased in recent years, as much as 38 percent between 1992 and 1994 (Hatcher, 1998).

Agriculture is another important component of the Lake Lanier economy. The number of farms and the average value of farm lands and farm buildings per acre have increased in the Lake Lanier area, and the livestock and poultry contribution to the agricultural sector has increased dramatically from the late 1970s. The total amount of acreage in farms, however, has decreased. Agricultural land use is being replaced by residential development as the Atlanta metropolitan area continues to expand. Both the number of permits issued for new housing units in the basin and earnings from construction have increased. As residential development has increased, the number of requests for private dock permits on the lake has also increased. In the 2004 Shoreline Management Plan, the USACE established the maximum number of dock permits at 10,615 (USACE, 2007). In 2007, primarily due to safety concerns caused by low lake levels, the Corps placed a moratorium on the issuing of dock permits, of which approximately 1,000 remain.

As the amount of land used for agriculture has declined and residential land use has risen, industrial development has also increased since the impoundment of the reservoir. Prosperity in the Lake Lanier basin, and in the growing Atlanta metropolitan region to the south, is dependent upon the sustainability of the area's natural resources. Water quality and quantity in Lake Lanier and the Upper Chattahoochee River basin are foremost among the resources of the region.

2.2 History of Water Quality Monitoring

An early study of water quality (Holder, 1967) focused on the recreational value of the fishery in Lake Lanier. In the 1960s, there was a high quality, warm water, surface fishery consisting of large mouth bass (*Micropterus salmoidea*), black crappie (*Pomoxia nigromaculata*), white bass (*Roccus chrysopa*), blue gill (*Lapomia macrochirus*), catfish (*Italurus sp.*) and green sunfish (*Lepomis cyanellus*) (Holder, 1967). An experimental program in which hatchery-raised trout were stocked in the lake had just begun, and Holder was interested in establishing whether a density current flowed through the lake, even during stratification, without mixing with the waters above or below. Such a density current would supply cool, oxygen-rich water all year around, allowing the lake to support a deeper, cold water fishery as well as the surface, warm water one. Holder concluded that such a density current did not exist, which would preclude the establishment of a successful, long-term, trout fishery.

At the time of the study, the only significant point sources of effluent on the lake were in Gainesville, the Gainesville Sewage Treatment Plant (GSTP), located about 1.5 miles southeast of Thompson Bridge on the Chattahoochee arm of the reservoir, and Flat Creek Pollution Control Facility (FCPCF), located about three kilometers (two miles) from the lake on Flat Creek. Holder noted that, at that time, water quality in the area of discharge from GSTP did not appear to be affected.

However, water quality was adversely affected in Flat Creek and its embayment by nutrient enrichment, resulting in algal blooms and associated elevation in pH, decrease in secchi depth, supersaturation of epilimnetic oxygen, and depletion of hypolimnetic oxygen. Holder considered the poor water quality in Flat Creek to be the result of the lower dilution factor, compared to GSTP. The discharge from FCPCF was frequently as large, and periodically larger, than the flow in the receiving stream, and Flat Creek Bay is smaller and less well flushed compared to the lake channel at GSTP.

A water quality study was conducted by an engineering firm for the USACE in 1978-79 (ESEI, 1981), for the objectives of 1) establishing baseline data, 2) identification of water quality problems, 3) data collection for the development of reservoir control-discharge water quality relationships, 4) study special problems and collect data necessary for solutions for such problems, and 5) collection of data that will provide an adequate data base and understanding of project conditions to facilitate coordination with state agencies to implement watershed pollution control.

The impetus for this study included blooms of the blue-green alga *Anabaena* and accompanying *algophorous amoebae* in Flat Creek (Cook et al., 1974) and a fish kill at the hatchery below the dam in 1976 (Noell, 1977). The results of the study by ESEI were that Lake Lanier was oligo-mesotrophic, based on chlorophyll *a*, with the exception of Flat Creek, which was enriched by discharge from FCPCF and affected by toxicity in the sediments due to elevated levels of chlordane and lead. The study also noted that hypolimnetic anoxia produced elevated levels of manganese and iron late in stratification, and that such anoxia was more pronounced during high flow conditions.

As development, and the requirement for water resources, continued to increase in the Lake Lanier watershed in the 1980s and 1990s, public concern about the quality and quantity of water in the lake also increased. In 1986, in response to water quality issues in the Flat

Creek embayment, a private organization, Lake Watch, was launched for the purpose of identifying water quality problems and finding solutions for them. The group originally focused on Flat Creek and FCPCF, which discharged into it. In 1987, in partnership with Gainesville State College, the program was expanded to include a monitoring program on the whole reservoir. Much of the data collected by the Lake Watch program, which ended in 1997, is reported here.

In 1992 the GAEPD placed a moratorium on new discharge permits into the Chattahoochee River and Lake Lanier. In 1994 Gwinnett County requested a permit to discharge into Lake Lanier, and joined with several other counties in the watershed of Lake Lanier to form the Upper Chattahoochee Basin Group. The group commissioned a study that included a survey of current water quality and the development of a model to simulate the effects of the proposed discharge on the lake.

The report (Limno-Tech, Inc., 1998) indicated that non-point sources were a major contributor to water quality issues, and the model indicated that the discharge would have minimal effects on water quality. GAEPD issued a permit for the facility in 1998; however, a legal challenge by the Lake Lanier Association and Upper Chattahoochee Riverkeeper delayed the opening of the plant until modifications were made in the permit. The plant became operational in 2004. The UCBG continues a monitoring program at stations on the Chestatee arm, the Chattahoochee arm and on several, smaller tributaries of Lake Lanier.

The USEPA awarded a Clean Lakes grant to GAEPD in 1992 for a Diagnostic/ Feasibility Study of the reservoir. Phase I of the study was completed in 1994, and Phase II, in 1998 (Hatcher, 1998). The conclusions in the final report were that 1) productivity is limited by phosphorus, with a few areas, affected by high phosphorus loads, limited by nitrogen; 2) non-point source pollution is the greatest threat to water quality in the reservoir; and 3) the parameters of most concern are nutrients, phosphorus, followed by nitrogen; chlorophyll *a*; bacteria; and sediments. A range of options for addressing the concerns about water quality were proposed, including more interaction and cooperation among stakeholders and the numerous local, state and federal agencies regulating the reservoir; formation of a Lake Lanier Watershed Council; and implementation of more extensive best management practices (BMPs). The study also included three alternative water quality targets: 1) fewer inputs of pollutants, which should increase water quality; 2) current inputs at the time of the study, which should maintain current water quality, and 3) increased inputs, which should decrease water quality. GAEPD developed standards for Lake Lanier in 1999, and began compliance monitoring at 5 required and 5 supplementary stations in 2001.

In 2001-2002, a study of water quality was conducted for this dissertation at four stations on Lake Lanier. The four stations were Browns Bridge (BB) in the mixing zone; Flat Creek Bay (FC) , which empties into the mixing zone; lake off Flowery Branch Bay (FBL) in the lacustrine zone; and Flowery Branch Bay, which empties into the lacustrine zone (Table 2.1 and Figure 2.1, inset).

2.3 Methods

A. 1987-1997 data. Samples were collected at 100 stations (Table 2.2 and Figure 2.1) annually during August/September from 1987 to 1996, by a mix of Lake Watch volunteers and Gainesville State College (GSC) faculty. During the rest of the year, a subset of 34 stations was sampled, either bimonthly (every two months) or quarterly (every three months) by GSC faculty. From 1993-1996, only the August/ September survey was conducted. In 1997, the last year of the program, only the subset of 34 stations was monitored in August /September. Table 2.3 provides a summary of the stations, dates, parameters.

At each station a profile was taken of temperature and dissolved oxygen at 1 m increments using a YSI oxygen meter. Samples for water chemistry, chlorophyll *a*, and seston were collected with a Kemmerer or Van Dorn water bottle near the surface (1 m), at the thermocline during stratification (6 to 12 m) and at 10-11 m during mixis, and near the bottom (0.5 to 2 m above the bottom). Bacterial samples were collected just below the surface, stored on ice in the dark until transported back to the laboratory. Bacterial samples were analyzed within 24 hours for fecal coliforms by the membrane filtration technique (APHA, 1985). Water samples were brought to room temperature and analyzed for conductivity using a Fisher Scientific conductivity meter.

100 mL of water were filtered for chlorophyll *a* through glass fiber filters within 96 hours of collection and stored in 90% acetone for up to two months until processing. The samples were ground by hand in a tissue grinder, allowed to steep in the freezer for 2 to 24 hours and analyzed fluorometrically, with correction for phaeophytins (APHA, 1985). 100 mL of water were filtered through 0.45 μm cellulose filters within a week of collection for total filterable solids (seston) using a modification of APHA methods (Banse and Hobson, 1963). Inorganic (abioseston) and organic filterable (bioseston) solids were determined by ashing at 600^{o}C and subtraction of ash weight from seston weight.

The cadmium reduction method (APHA, 1985) was used to determine nitrite-nitrate (NO_2-NO_3) from December, 1988 to September, 1997. Samples were analyzed for soluble reactive phosphorus (SRP) from December,1988 to June, 1992 and for total phosphorus (TP) from August,1992 to September, 1997. Following persulfate digestion, samples were analyzed for TP by the stannous chloride method (APHA, 1985) using a 10-cm pathlength cell. Only the TP data is reported here.

B. 2001-2002 data. Samples were collected at four stations and a control station from July 2001 to March, 2002 (Table 2.1 and Figure 2.1). One station (BB) was located below Browns Bridge in the mixing zone; a second station (FC) was located in the embayment of Flat Creek, an area of historically poor water quality; a third station (FBL), was located in the lacustrine zone; and the fourth station (FB) was located in the embayment of Flowery Branch, which had no history of impairment. The control station (CON) was located in Flowery Branch Bay. Table 2.4 provides a summary of the stations, dates, parameters and location of data in this dissertation and in the electronic database.

DataSonde 4a Hydrolabs were deployed at three depths at each station for periods of two to four weeks from July 2001 to March 2002. The depths of deployment were: surface, 1.5-2 m; middle (metalimnion during stratification), 8-12 m; and bottom, 26-29 m. The hydrolabs were removed for servicing from 10/11/01 to 11/24/01, and were not deployed during this period. Secchi depth and profiles at one-meter intervals (two-meter intervals in the lower hypolimnion) were taken with a Hydrolab 4a DataSonde for temperature, dissolved oxygen, pH, conductivity, and sometimes ORP and turbidity, at each station when the deployed Hydrolabs were retrieved and re-deployed.

Samples for water chemistry were also collected at the four stations during April, August-September, and December 2001, and during February 2002. The samples were placed on ice until return to the laboratory, when they were frozen until transport to the Chemical Analysis Laboratory at the University of Georgia in Athens, Georgia. The samples were analyzed by inductively coupled plasma (ICP) for a suite of 20 elements, and by Bran-Luebbe Autoanalyzer for alkalinity, total inorganic carbon (TIC), total organic carbon (TOC), SRP, TP, ammonia (NH_4), NO_2, NO_3, total nitrogen (TN), and sulfate (SO4). TP and TN samples were digested with persulfate prior to analysis.

Relative areal hypolimnetic oxygen deficit (RAHOD) was calculated using the method of Wetzel and Likens (2000) from oxygen profile data collected in 1987-1992 by Lake Watch and Gainesville State College, and in 2001-2002 for this study. The volume of each one-meter increment of the hypolimnion was calculated using a GIS geodatabase of the bathymetry of Lake Lanier, developed by the Institute for Environmental and Spatial Analyis (IESA) at GSC

2.4 Results

Conductivity in this soft-water system is low. During the Lake Watch monitoring program in 1987 -1997, conductivity ranged from 22 to 51 μS/cm and averaged 34 μS/cm (Table 2.5). During most seasons, conductivity at the bottom was elevated above that at the surface by 2-24%. This elevation at depth was most pronounced in late summer, and increased over the ten year period, as shown by the increase in the ratio of hypolimnetic to epilimnetic conductivity (Table 2.5, Figure 2.2). May 1991 was distinctive in having 28% higher mid-depth average conductivity compared to the bottom The average and range of conductivity in the 2001-2002 sampling period were 33.7 and 21.3-111.4 μS/cm, respectively.

Annual and seasonal mean secchi depth for 34 or 100 stations from 1987 - 97 is shown in Table 2.5. Mean annual secchi depth varied less than 10% from year to year for the four years (1987-1988, 88-89,89-90, 91-92) for which seasonal data were collected. The lowest mean secchi depths, $\approx$20% below the mean, occurred in December in three of those years, and once in March. The highest mean secchi depths, $\approx$15 - 25% above the mean, occurred in summer in 1989, 1991and 1992. Mean annual Secchi depth at BB, FBL, FC and Buford Dam (DAM), for selected years over the period from 1966 - 2002, shows an increasing trend (Table 2.6).

A summary of the results of the survey of water chemistry for alkalinity, TIC-TOC and SO4 at BB, FC, FBL and FBB in 2001-2002 are shown in Table 2.7. The complete data for these parameters is available in the electronic data files. The range of TIC (2.5-6.5 mg/L) and TOC (1.4-3.3 mg/L) was low in the reservoir. Mean values for TIC were 20 - 30% higher at bottom depth at that at surface or middle depths, reflecting increasing concentrations of CO2 in the hypolimnion as summer stratification progressed. Mean values for TOC were higher at the surface and decreased with depth, most probably due to higher algal concentrations in the photic zone.

Alkalinity was at or below detection limits at all stations and depths throughout most of the sampling period. The exception occurred in April, when alkalinity at all stations and depths reached detectable levels ranging from 19.4-27.3 mg/L and in September at Flowery Branch Bay, when mid-depth alkalinity was 16.3 mg/L. These measureable levels of alkalinity were the result of elevated photosynthetic activity during a spring algal bloom in April, and at mid-depth in Flowery Branch Bay in September.

Sulfate concentrations were at or below the detection limit at most stations and depths throughout the sampling period. The highest levels of sulfate occurred in Flat Creek, 5.5 mg/L at the bottom in April and 8.25 mg/L at mid-depth September.

The results of the survey in 2001-2002 for TN, NO_2-NO_3, and NH_4 at BB, FC, FBL and FBB are shown in Table 2.8. The levels of all forms of nitrogen were low; TN never exceeded 1.5 mg/L. Overall, concentrations of N followed the spatial trend FC>BB >FBL >FBB. The highest values for TN were 1.39 mg/L, which occurred at FC at mid-depth in December. Values of TN close to 1.0 mg/L also occurred at bottom depths at BB in December and at FC in April and February. In April NO_2-NO_3 was higher at depth at all stations and NH_4 was undetectable at all depths and stations. By the end of stratification in December, as oxygen supplies were depleted and bacterial respiration moved on to other electron acceptors, NO_2-NO_3 was undetectable at the bottom of all stations, with the exception of FBB, where traces remained. NH_4 showed a concomitant increase with depth over the period of stratification. By December, NH_4, reached levels close to 1.0 mg/L at the bottom at BB and FC, comprising 95-100% of TN. The exception, as with NO_2-NO_3, was FFB, where NH_4 at the bottom depth was 60% of TN.

The results of ICP analysis for Fe and Mn in 2001-2002 is shown in Table 2.9. The data for the entire ICP metals analysis is available in the electronic data files. Both Fe and Mn were at or below detection limits at all stations and at all depths in february, during mixis, and in April, when stratification was just beginning. By September, after several months of stratification, manganese appeared at detectable concentrations (0.3 mg/L) in the mixing zone at the bottom of BB and of FC, but not in the lacustrine zone at FBL, or at FBB. By the end of stratification in December, manganese had reached detectable levels (0.3-0.4 mg/L) at the bottom of FBL and FBB, and had increased by 50-80% at BB and FC. Iron was undetectable at all stations and depths, with the exception of the bottom at FC in April, in which Fe was just above the detection limit, and the bottom of BB in December, which was at the detection limit.

The mean and standard deviation for annual and seasonal TP in 1990-92, and for summer TP in 1993-97, is shown in Table 2.10. These data show that epilimnetic TP is often below detection or quite low, and quite variable, and that hypolimnetic TP is higher and also quite variable. Figure 2.3 shows the variability between years in the mean annual summer phosphorus at surface, middle and bottom depths for 1990-1997. Epilimnetic and metalimnetic TP vary among years but show an increasing trend over time; hypolimnetic TP shows the greatest variability and the greatest increase over time.

Figure 2.4 shows the temporal and spatial variability in TP in Lake Lanier during 1991-1992. During stratification in August 1991, TP at the surface was, at most, 1.7 times higher than 50 μg/L at five surface stations; at depth, TP was up to three times higher than the baseline at ten stations. In December 1991, during mixis, the spatial pattern of TP was reversed. Concentrations of TP were up to 17 times higher than 50 μg/L at the surface of 15 stations, but only up to three times as high at nine stations at depth. In March 1992, at the beginning of stratification, TP was, as in August 1991, about 1.7 times higher than 50 μg/L at the surface of a few stations (two); and TP at the bottom was up to 14 times higher than the baseline at nine stations.

A comparison of surface and bottom TP during summer in 1990-1997 (Figures 2.5) shows that TP at the surface is slightly elevated, up to 1.8 times higher than 50 μg/L, at 4-5 stations in 1991 and 1993, and up to 23 times higher than the baseline at 37 stations in 1995. Twenty five of these stations, and the highest surface TP values occurred in the two riverine arms in 1995. TP at depth was >50 μg/L at more stations that the surface in all three years, 11 in 1991, 18 in 1993, and 51 in 1995, and the maximum levels of TP were also higher than maximum surface values in all three years, although 1991 bottom values were only about twice as high as surface values.

The results of the survey in 2001-2002 for TP and SRP are shown in Table 2.11. TP was below detection at all stations, all depths and all seasons, with the exception of the mid-depth sample at FC in September, which was at the detection limit of 0.04 mg/L. SRP was similarly below detection at almost all stations, depths and seasons. One sample, mid-depth in FBB in February, was above detection limits, and surface samples on that date at FBB and at FBL were at the detection limit. However, given the fact that TP was always at or below detection, and that normally SRP is expected to be the same or less than TP, it seems likely that P of either form was not present in significant amounts.

Table 2.12 shows a summary of TP data collected at ten stations on Lake Lanier for compliance monitoring by GAEPD (2002) for a time period, April-October 2001, similar to that for the data described above, collected for this study in 2001. This data was collected only in the epilimnion; however, TP shows a similar pattern to the data collected for this study (Table 2.11). Levels 1.5-2 times higher than the detection limit (0.02 mg/L) occurred in the spring at the stations in the Chattahoochee arm, the Chestatee arm, and in Balus Creek Bay, and in August in Little River Bay and Mud Creek Bay. The highest levels of TP occurred in August: 3.5 times the detection limit in Flat Creek Bay, and 22 times the detection limit, in Six Mile Creek. Bay. TP was not detectable on any sampling date in the main channel stations, FBL, BB and DAM.

The mean and standard deviation for seasonal and annual chlorophyll *a* for 1987-92, and mean summer chlorophyll *a* for 1992-95, are shown in Table 2.13. Mean annual epilimnetic chlorophyll *a* levels were moderate, ranging from $\approx$2.0 - 5.0 μg/L, and showed a similar seasonal pattern to Secchi depth. Winter (December/January) and early spring (March) chlorophyll *a* levels were 6 - 98% higher than the annual mean in all four years, with the exception of March 1989 and March 1992, which were 10% and 80% lower than the annual mean, respectively.

Table 2.14 shows seasonal and mean annual chlorophyll *a* in the Chattahoochee arm at Thompson Bridge, in the Chestatee arm at Bolling Bridge, SIX, FBL, BB, DAM and the mouth of FC in 1966-67 (Holder, 1967). Table 2.15 shows seasonal and mean annual chlorophyll *a* at required and supplementary GAEPD stations in 2001. The annual means at the required stations did not exceed the standards of 10.0 mg/L for the Chattahoochee and Chestatee arms or 5.0 mg/L for FBL, DAM or BB; however, the Chattahoochee arm, FBL and DAM were only a few percent below the standard. Mean annual chlorophyll *a* in 1966-67 at these stations was less than half that in 2001, except in FC, where it was about 15% lower in 2001 than in 1967-67.

However, in 2001 the station was in mid-bay, and in 1966-1967 it was at the mouth of Flat Creek, where nutrient levels were probably higher. Figure 2.6 shows seasonal chlorophyll *a* at Buford Dam and at Browns Bridge in 1966-67 and in 2001-2002. chlorophyll *a* was $<1.0\ \mu$g/L in both years during July and August at Browns Bridge and Buford Dam in 1966-1967, and at the dam in 2001; levels at Browns Bridge in the summer of 2001 were somewhat higher, about 2.0 μg/L.

The spring algal bloom occurred in May, and the fall bloom in October, at both stations in 2001, reaching 8.0 - 9.0 μg/L at Browns Bridge and 20-30% less than that at the dam. In 1966-1967chlorophyll *a* at Buford Dam did not exceed 2.0 μg/L. Levels at Browns Bridge also remained below 2.0 μg/L for most of the year, except for a winter bloom that exceeded 4.0 μg/L in January. Mean annual chlorophyll *a* in the Chestatee arm, at BB in the mixing zone, and at two stations in the lacustrine zone, FBL and DAM, has been quite variable, but increasing over the period from 1966 to 2001 (Figure 2.7).

Figure 2.8 shows seasonal profiles of dissolved oxygen (DO) at Browns Bridge in 1966-1967 and in 2001-2002. Hypolimnetic anoxia developed earlier in 2001, and at shallower depths, although by October the hypolimnion was anoxic below 20 m in both years. Positive, as well as negative, heterograde oxygen profiles occurred June through September of both years; positive heterograde decreased over that time frame, more than in 1967, and negative heterograde persisted longer in 1967. DO at mid-depth in the hypolimnion in September 1967 was still 2-3 mg/L; however, in September 2001 there was 1 mg/L or less. The hypolimnion was anoxic in both years in October and November. The reservoir remained stratified and anoxic in December 2001, but had begun to mix in December 1967.

Table 2.16 shows RAHOD for two main stem stations, BB in the mixing zone and FBL in the lacustrine zone, and two embayment stations, FC and FBB. RAHOD at the lacustrine station off Flowery Branch bay has increased steadily from 0.040 mg O2/cm2/day in 1988 to 0.060 in 2002. The oxygen deficit in Flowery Branch bay also increased from 0.031 to 0.56 mg O2/cm2/day over that time period, but was more variable. RAHOD at Browns Bridge in the mixing zone, and in Flat Creek bay, was higher than at the stations in the lower lake, and also more variable, being higher in 1989 and 2001 and lower in 1988, 1992 and 2002.

2.5 Summary and Conclusions

Lake Lanier is a soft water system, with low alkalinity and low nutrient levels, although chlorophyll *a* levels and hypolimnetic anoxia, have increased since 1967. Anoxia produced reducing conditions in December, 2001, which allowed the reduction of manganese, but only limited increases in ferrous iron; there was no apparent increase in hypolimnetic SRP. Anoxia in 1978-79 caused the release of higher levels of iron, however, SRP was only slightly elevated (ESEI, 1981). This was noteworthy, since it had been estimated that 90% of the P load entering the lake was bound and settled into sediments; however, internal loading was not observed (ESEI, 1981)

The phenomenon of reducing conditions without the accompanying release of phosphorus has been documented. Caraco, et al (1994) hypothesizes that phosphorus release does not occur, even under reducing conditions, unless sulfate levels are high, because sulfide produced by the reduction of sulfate binds and sequesters ferrous iron and prevents it from re-oxidizing and binding phosphorus. Sulfate levels are not high in Lake Lanier; levels in 2001 were generally at or below the detection limit of 3.0 mg/L (Table 2.7), which may explain the lack of internal loading. Another explanation is that reduction of ferric to ferrous iron is not extensive until other electron acceptors, nitrates and manganese, have been reduced. Reducing conditions in 2001 had reached the point where manganese was being reduced; however, only small amounts of ferrous iron had been released. The combination of high manganese concentrations, which must be reduced before iron is reduced, and the high concentrations of ferric iron, continuously settling from oxic epilimnetic water, in Piedmont impoundments such as Lanier, may be sufficient to bind P if it is released from sediments (Parker, 2004).

Chlorophyll *a* is low, although levels have increased since 1966. The prevailing hypothesis is that low algal biomass is caused by the lack of available P, most of which enters the lake bound to clay particles high in ferric oxyhydroxides, which settle and sequester P in bottom sediments. Internal loading of P from sediments, as a result of hypolimnetic anoxia, does not occur extensively in Lake Lanier, which may be explained by high iron content of sediments, or low levels of sulfate, or a combination of both. However, laboratory experiments show that P may be released from ferric iron at elevated pH (Lijklema, 1980), which may occur in soft water systems as a result of photosynthesis. Lake Lanier sediments have been shown to release some P at elevated pH in laboratory studies (Parker, 2004). There is also evidence that this may occur in Lake Allatoona, another Piedmont impoundment in Georgia (Ceballos and Rasmussen, 2007).

An alkaline release could result in higher algal biomass, which may have contributed to the increase in chlorophyll *a* since 1966. It is also possible that algal biomass has always been higher than indicated by epilimnetic chlorophyll *a*. The metalimnetic maxima and minima of oxygen may be the result of algal biomass that peaks in the metalimnion. This peak in biomass may be the result of algal cells settling with clay particles (Mayhew and Mayhew, 1992), or higher algal production because nutrients are more available as a result of settling or recycling in the metalimnion.

Positive heterograde oxygen curves that develop in summer, such as those in the metalimnion in Lanier in 1967, and, less extensively, in 2001, are usually attributed to a concentration of photosynthetic organisms (Horne and Goldman, 1994). Negative heterograde curves, such as those that developed at the bottom of the thermocline in Lanier during the summer in 1967 and 2001 (Figure 2.8), are generally due to increased respiration (Horne and

Goldman, 1994; Shapiro, 1960), although physical phenomena, such as density currents or lake morphology, may also create positivae or negative heterograde curves.

Lake Lanier does not have the distinctive morphology, steep sides with a distinct shelf, usually associated with a heterograde curve, and there is no evidence of a density current that could create one or both types of curves on an annual basis. Metalimnetic peaks in algal biomass could explain both positive and negative curves.

Table 2.1: Location of Lake Lanier stations sampled in 1999 and 2001-2002; Browns Bridge (BB), Flat Creek Bay (FC), lake off Flowery Branch Bay (FBL), and Flowery Branch Bay, and a control station located off Flowery Branch (CON).

	GPS Location	
Station	Easting	Northing
BB	228 225.17	3 795 062.20
FC	229 222.20	3 793 952.05
FBL	224 842.39	3 788 475.07
FB	225 635.35	3 787 726.50
CON	225 609.00	3 788 304.00

Table 2.2: Locations of Lake Lanier water quality sampling stations, 1987-1997.

Station Number and Description		Location (UTM) Easting	Northing
1	Lula Br (Hwy 52)	251 031	3 811 373
2	Clark's Br (Hwy 284)	243 167	3 804 461
3	Limestone Ck	242 619	3 803 277
4	Laurel Park	242 169	3 804 148
5	Gainesville Water Intake	239 574	3 803 658
6	Upper Wahoo Cr	235 714	3 810 664
7	Middle Wahoo Ck	236 514	3 808 755
8	Lower Wahoo Ck	237 784	3 808 142
9	Little River	240 832	3 806 555
10	Thompson Br (Hwy 129)	238 575	3 804 498
11	Chatt. Country Club	237 315	3 803 449
12	STP Linwood	237 304	3 802 213
13	Upper Longwood Cove	237 602	3 800 403
14	Middle Longwood Cove	237 340	3 801 418
15	Mouth Longwood Cove	237 288	3 801 461
16	Ada Ck	236 122	3 803 794
17	Sardis Ck Access	234 803	3 802 930
18	Lanier Br (Hwy 53)	235 228	3 801 878
19	Gaineville Marina	235 203	3 800 906
20	Jefferson Park	235 282	3 799 620
21	Cove S of Marker 44	235 095	3 798 724
22	Channel Opp Jefferson Pk	234 498	3 799 251
23	Cove S of Marker 42	234 386	3 798 662
24	Cove E of River Forks Pk	233 203	3 797 929
25	N of River Forks Park	232 265	3 798 703
26	River Forks Swim Area	232 319	3 798 019
27	Wilkie Br (Hwy 139)	226 567	3 810 386
28	Cove Opp Toto Ck Access	227 058	3 809 651
29	Cool Springs	226 007	3 807 559
30	Thompson Ck Access	774 892	3 805 663
31	Cove Thompson Ck Mkr 6	775 628	3 806 786
32	Athens Boat Club	225 158	3 805 681
33	War Hill	227 822	3 803 100
34	Upper Julian Creek	225 149	3 803 871
35	Julian Creek Mouth	226 569	3 802 796

Table 2.2 (continued)

Station Number and Description		Location (UTM)	
		Easting	Northing
36	Upper Latham Ck	229 802	3 804 301
37	Middle Latham Ck	230 229	3 804 119
38	Upper Johnson Ck	231 936	3 803 959
39	Open Chan off Latham Ck	228 879	3 802 581
40	Little Hall Pk	229 057	3 800 251
41	Duckett Mill Pk S Cove	230 403	3 799 806
42	Chestatee Mouth W	229 384	3 798 463
43	Chestatee Mouth E	229 992	3 798 510
44	Confluence Chest & Chatt	230 617	3 797 783
45	Chestatee Bay Mouth	228 463	3 796 500
46	Brown's Br (Hwy 369)	228 182	3 795 626
47	Upper Chestatee Bay	226 705	3 796 473
48	Lan Mar Marina	228 029	3 794 048
49	Inlet Flat Ck Bay	232 470	3 795 314
50	Narrows Flat Ck Bay (FC9)	231 794	3 795 337
51	Mid Bay Flat Ck Bay	230 944	3 794 651
52	Inlet Balus Ck Bay	230 776	3 793 758
53	Balus Ck	231 751	3 793 507
54	Mouth Flat Ck Bay S	229 929	3 793 816
55	Mouth Flat Ck Bay N	229 658	3 793 972
56	Chan off Flat Ck	228 533	3 793 053
57	Sunrise Marina	228 923	3 792 427
58	Upper 2 Mile Ck	224 766	3 793 769
59	Vann's Tavern Access	225 912	3 791 694
60	Chan off Old Fed Pk	226 906	3 791 199
61	Old Fed Pk	227 173	3 790 624
62	Bethel Pk	224 591	3 790 912
63	Chan off Aqualand N	224 993	3 789 506
64	Chan off Aqualand S	225 100	3 789 350
65	Aqualand Marina	226 473	3 787 774
66	Starboard Marina	229 053	3 786 322
67	Upper Flowery Br Bay	227 835	3 787 082
68	Van Pugh Day Use	225 675	3 787 012
69	Mouth 2 Mile Ck Bay	775 545	3 789 348
70	6 Mile Ck above Br 369	772207	3 793 822

Table 2.2 (continued)

Station Number and Description		Location (UTM) Easting	Northing
71	Upper 6 Mile Ck	772 630	3 792 939
72	Lower 6 Mile Ck	773 795	3 792 589
73	Upper 4 Mile Ck	775 055	3 792 668
74	Conflu 4 Mile & 6 Mile Cks	774 383	3 790 648
75	University Yacht Club	224 844	3 784 718
76	Betwn Mrk 2&3 on 2 Mi Ck	222 301	3 789 098
77	Chan above Holiday Marina N	775 748	3 787 754
78	Chan above Holiday Marina S	775 893	3 787 363
79	Holiday Marina	776 142	3 785 829
80	Shady Grove Park	220 594	3 788 842
81	Lanier Islands	221 092	3 786 450
82	Mouth Shoal Ck	774 780	3 784 901
83	Upper Shoal Ck	775 810	3 784 181
84	Lanier Harbor Marina	774 455	3 783 387
85	Lanier Islands Beach	773 552	3 785 007
86	Upper Young Deer Ck	770 088	3 790 727
87	Middle Young Deer Ck	770 879	3 789 473
88	Young Deer Cove	771 250	3 789 769
89	Tidwell Access	770 998	3 787 796
90	Chan off Tidwell Access	771 466	3 787 391
91	Golf Course Lanier Islands	772 488	3 786 627
92	Mary Alice Pk	767 644	3 788 275
93	Habersham Marina	767 452	3 787 478
94	Middle Baldridge Bay	768 791	3 787 916
95	Swanee Campground	769 959	3 785 794
96	Chan off Baldridge Bay	770 417	3 784 943
97	Open Chan above Dam	771 043	3 785 034
98	Campground Lanier Islands	772 057	3 784 972
99	Above Buford Dam	770 139	3 784 115
100	Gwinnett Park	771 812	3 783 021

Table 2.3: Summary of monitoring data collected between August 1987 and September 1997, including number of stations and depths (S-Surface, M-Middle, B-Bottom), parameters measured, and location of data in chapters and appendices.

- Frequency
  - bimonthly (1987-88)
  - quarterly (1989-92)
  - annually (1992-97)
- Number of Stations
  - 100 in Aug/Sep 1987-1997
  - 34 in other seasons between 1987-1993 and in Aug 1997
- Parameters (depth)
  - Temperature and oxygen profiles (1-m intervals)
  - Secchi depth and fecal coliforms (surface)
  - Conductivity, TP, NO_2-NO_3, seston, abioseston, bioseston (SMB)
  - Chlorophyll *a* (SM)

Table 2.4: Summary of monitoring data collected in 1999 and 2001-2002, including number of stations, parameters and depths (S-Surface, M- Middle, B-Bottom).

- Dec 1999, Apr, Aug and Dec 2001
  - Four Stations
  - Parameters (depth):
    - CO_2 (3-5 m increments)
- Apr 2001, then every two to four weeks from Jul 2001- Mar 2002
  - Four Stations
  - Parameters (depth):
    - Secchi depth (S)
    - Temperature, oxygen, pH, conductivity profiles (S)
- Apr 2001 - Mar 2002
  - Four Stations plus one at a middle depth (M) as a control
  - Data loggers were deployed for two to four week rotations
  - Data was collected at 30 min intervals
  - Parameters (depth):
    - Temperature, oxygen, pH, conductivity (SM)
    - Temperature, oxygen, pH, conductivity, ORP (B)
- Apr, Aug, Dec 2001, Feb 2002
  - Four Stations
  - Parameters (depth):
    - Elemental Fe, Mn, P; alkalinity, TIC, TOC, NO_3, NH_4, TN, TP, SRP, SO_4 (SMB)
    - Chlorophyll a (three depths in photic zone)

Table 2.5: Mean seasonal and annual Secchi depth and specific conductance at three depths, and the ratio of hypolimnetic to epilimnetic conductivity (H/E) at 100 stations between 1987-1996, and 34 stations in 1997.

	Secchi	Specific Conductance (μS/cm)			
Date	Depth (m)	Surface	Middle	Bottom	H/E
Aug 1987	2.2	34	34	37	1.09
Oct 1987	2.4	28	30	30	1.07
Dec 1987	1.9	34	33	35	1.03
Mar 1988	2.2	40	37	38	0.95
Jun 1988	2.2	36	35	36	1.00
Annual Mean	2.2	34.4	33.8	35.2	1.02
Aug 1988	2.3	31	31	34	1.10
Dec 1988	2.3	31	28	34	1.10
Mar 1989	2.1	32	30	33	1.03
May 1989	2.1	36	34	35	0.97
Annual Mean	2.2	32.5	30.8	34.0	1.05
Aug 1989	2.7	24	25	26	1.08
Jan 1990	2.2	22	24	24	1.09
Mar 1990	1.4	22	22	24	1.09
Jun 1990	2.3	29	28	32	1.10
Annual Mean	2.2	24.3	24.8	26.5	1.09
Aug 1990	2.1	30	30	34	1.13
May 1991	2.1	37	51	40	1.08
Sep 1991	2.7	37	45	42	1.14
Dec 1991	1.8	40	44	37	0.93
Mar 1992	2.3	35	36	36	1.03
Jul 1992	3.0	34	37	40	1.18
Annual Mean	2.4	36.5	40.5	38.8	1.06
Sep 1992	2.9	32	32	37	1.16
Aug 1993	2.2	28	32	33	1.18
Aug 1994	2.6	35	35	37	1.06
Aug 1995	2.3	30	31	35	1.17
Aug 1996	2.5	35	35	38	1.09
Aug 1997	2.5	34	37	42	1.24

Table 2.6: Mean annual Secchi depth at Browns Bridge (BB), lake off Flowery Branch Bay (FBL), Buford Dam (DAM), and Flat Creek Bay (FC). 2001-2002 data for DAM (*) is from Upper Chattahoochee Riverkeeper. FC station in 1966 was the mouth of Flat Creek (**); in other years it was mid-bay.

	Secchi Depth (m)			
Period	BB	FBL	DAM	FC
1966-1967	1.8	2.4	2.7	**0.91
1988-1989	2.3	3.2	2.6	1.7
1991-1992	3.1	3.9	3.9	2.0
2001-2002	3.2	4.1	*3.3	3.3

Table 2.7: Elemental iron (detection limit = 50 μg/L) and manganese (detection limit = 100 μg/L) at Browns Bridge (BB), Flat Creek Bay (FC), lake off Flowery Branch Bay (FBL) and Flowery Branch Bay (FB) at three depths in 2001-2002. ND = no data. BDL = below detection limit.

		Iron (μg/L)				Manganese (μg/L)			
Station	Depth	4/12/01	9/7/01	12/2/01	2/15/02	4/12/01	9/7/01	12/2/01	2/15/02
BB	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Middle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Bottom	BDL	BDL	80	BDL	BDL	290	540	BDL
FC	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Middle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Bottom	120	BDL	BDL	BDL	BDL	340	510	BDL
FBL	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Middle	BDL	ND	BDL	BDL	BDL	ND	BDL	BDL
	Bottom	BDL	ND	BDL	BDL	BDL	ND	450	BDL
FB	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Middle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Bottom	BDL	BDL	BDL	BDL	BDL	BDL	330	BDL

Table 2.8: Ammonium and total nitrogen (detection limit = 0.02 mg/L) at Browns Bridge (BB), Flat Creek Bay (FC), lake off Flowery Branch Bay (FBL), and Flowery Branch Bay (FB) at three depths in 2001-2002. ND = no data. BDL = below detection limit.

		Ammonium (mg/L)				Total Nitrogen (mg/L)			
Station	Depth	4/12/01	9/7/01	12/2/01	2/15/02	4/12/01	9/7/01	12/2/01	2/15/02
BB	Surface	BDL	0.01	0.08	0.19	0.60	0.27	0.08	0.56
	Middle	BDL	0.10	0.42	0.20	0.50	0.60	0.42	0.53
	Bottom	BDL	0.22	0.93	0.20	0.65	0.50	0.93	0.56
FC	Surface	BDL	0.01	0.10	0.18	0.55	0.38	0.10	0.68
	Middle	BDL	0.07	0.37	0.19	0.53	0.56	0.37	0.77
	Bottom	BDL	0.24	0.72	0.19	0.90	0.63	0.72	0.86
FBL	Surface	BDL	0.01	0.05	0.14	0.44	0.27	0.05	0.37
	Middle	BDL	ND	0.18	0.16	0.40	ND	0.18	0.40
	Bottom	BDL	ND	0.56	0.18	0.52	ND	0.56	0.41
FB	Surface	BDL	0.01	0.07	0.15	0.41	0.28	0.07	0.43
	Middle	BDL	0.05	0.10	0.13	0.40	0.46	0.10	0.38
	Bottom	BDL	0.01	0.29	0.13	0.48	0.48	0.29	0.41

Table 2.9: Elemental phosphorus (detection limit = 0.04 mg/L) and nitrite-nitrate (detection limit is 0.02 mg/L) at Brown's Bridge (BB), Flat Creek Bay (FC), lake off Flowery Branch Bay (FBL) and Flowery Branch Bay (FB) at three depths in 2001-2002. ND = no data. BDL = below detection limit.

		Phosphorus (μg/L)				NO_2-NO_3 (mg/L)			
Station	Depth	4/12/01	9/7/01	12/2/01	2/15/02	4/12/01	9/7/01	12/2/01	2/15/02
BB	Surface	BDL	130	BDL	BDL	0.33	0.12	0.07	0.17
	Middle	BDL	80	BDL	BDL	0.27	0.40	BDL	0.10
	Bottom	BDL	50	BDL	BDL	0.45	0.26	BDL	0.16
FC	Surface	BDL	BDL	BDL	BDL	0.41	0.19	0.15	0.33
	Middle	BDL	40	BDL	10	0.27	0.40	0.82	0.36
	Bottom	BDL	150	BDL	BDL	0.78	0.29	BDL	0.40
FBL	Surface	BDL	250	BDL	BDL	0.26	0.11	0.13	0.09
	Middle	BDL	80	BDL	0.01	0.20	ND	0.11	0.10
	Bottom	BDL	ND	BDL	BDL	0.39	ND	BDL	0.14
FB	Surface	BDL	BDL	BDL	BDL	0.22	0.11	0.07	0.12
	Middle	BDL	120	BDL	BDL	0.19	0.30	0.15	0.08
	Bottom	BDL	70	BDL	BDL	0.37	0.35	0.06	0.12

Table 2.10: Total inorganic carbon and total organic carbon at Browns Bridge BB), Flat Creek (FC), lake off Flowery Branch Bay (FBL), and Flowery Branch Bay (FB) at three depths in 2001-2002. ND is no data.

		Total Inorganic Carbon (mg/L)				Total Organic Carbon (mg/L)			
Station	Depth	4/12/01	9/7/01	12/2/01	2/15/02	4/12/01	9/7/01	12/2/01	2/15/02
BB	Surface	2.88	2.80	3.18	ND	1.82	2.05	2.04	ND
	Middle	3.35	3.05	3.86	ND	1.62	1.36	1.55	ND
	Bottom	3.77	4.56	5.88	ND	1.48	1.43	1.67	ND
FC	Surface	2.93	2.98	2.56	3.27	1.79	2.30	2.11	3.34
	Middle	3.18	3.78	4.71	3.49	1.47	1.56	2.00	1.76
	Bottom	4.18	5.23	6.54	3.87	1.70	1.65	1.47	1.94
FBL	Surface	2.92	3.05	3.22	ND	1.70	2.25	2.19	ND
	Middle	3.46	ND	3.56	ND	1.56	ND	1.71	ND
	Bottom	3.79	ND	5.11	ND	1.51	ND	1.73	ND
FB	Surface	2.91	2.88	3.17	ND	1.70	2.14	2.15	ND
	Middle	3.47	3.48	3.68	ND	1.58	1.79	1.78	ND
	Bottom	3.84	3.76	4.76	ND	1.47	1.42	1.37	ND

Table 2.11: Alkalinity (detection limit = 10 mg/L) and sulfate (detection limit = 3 mg/L) at Browns Bridge (BB) Flat Creek (FC), lake off Flowery Branch Bay (FBL), and Flowery Branch Bay (FB) at three depths in 2001-2002. ND = no data. BDL = below detection limit.

		Alkalinity (mg/L)				Sulfate (mg/L)			
Station	Depth	4/12/01	9/7/01	12/2/01	2/15/02	4/12/01	9/7/01	12/2/01	2/15/02
BB	Surface	24.00	BDL	BDL	BDL	BDL	3.98	BDL	BDL
	Middle	22.78	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Bottom	21.87	BDL	BDL	BDL	BDL	BDL	BDL	BDL
FC	Surface	21.32	BDL	BDL	BDL	3.51	5.16	BDL	BDL
	Middle	27.27	BDL	BDL	BDL	BDL	BEL	8.25	BDL
	Bottom	25.03	BDL	BDL	BDL	5.48	3.65	BDL	BDL
FBL	Surface	22.35	BDL	BDL	BDL	BDL	3.35	3.15	BDL
	Middle	25.33	ND	BDL	BDL	BDL	ND	BDL	BDL
	Bottom	23.45	ND	BDL	BDL	BDL	ND	BDL	BDL
FB	Surface	24.30	BDL	BDL	BDL	BDL	3.87	BDL	BDL
	Middle	19.43	16.26	BDL	BDL	3.93	4.76	BDL	BDL
	Bottom	23.57	BDL	BDL	BDL	BDL	3.47	BDL	BDL

Table 2.12: Mean and standard deviation for total phosphorus (μg/L) for 100 stations sampled in 1990-1996 and 34 stations sampled in 1997 at three depths (S-surface, M-middle, B-bottom) on Lake Lanier.

	Total Phosphorus (μg/L)		
Date	Surface	Middle	Bottom
Aug 1990	11.3 ± 43.3	7.3 ± 17.8	7.3 ± 12.1
May 1991	5.9 ± 3.4	7.4 ± 9.4	7.0 ± 5.4
Sep 1991	59.6 ± 163.3	35.8 ± 35.2	53.3 ± 111.7
Dec 1991	25.8 ± 20.0	20.6 ± 14.5	140.9 ± 188.2
Mar 1992	80.9 ± 153.4	41.0 ± 23.7	38.6 ± 32.2
Sep 1992	41 ± 99.6	26.2 ± 26.8	46.7 ± 87.6
Aug 1993	12.2 ± 17.4	13.4 ± 25.0	65.7 ± 283.3
Aug 1994	18.1 ± 24.3	19.3 ± 26.7	46.1 ± 78.9
Aug 1995	61.0 ± 124.7	39.8 ± 51.3	114.5 ± 211.7
Aug 1996	18.6 ± 19.9	16.9 ± 9.7	35.3 ± 28.4
Aug 1997	17.1 ± 14.3	19.6 ± 9.6	72.4 ± 154.3

Table 2.13: Total phosphorus and soluble reactive phosphorus (detection limit = 40 μg/L) at Browns Bridge (BB), Flat Creek (FC), lake off Flowery Branch Bay (FBL), and Flowery Branch Bay (FB) at three depths in 2001-2002. ND = no data. BDL = below detection limit.

		Total Phosphorus (μg/L)				Soluble Reactive Phosphorus (μg/L)			
Station	Depth	4/12/01	9/7/01	12/2/01	2/15/02	4/12/01	9/7/01	12/2/01	2/15/02
BB	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Middle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Bottom	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
FC	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	Middle	BDL	40	BDL	BDL	BDL	BDL	BDL	BDL
	Bottom	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
FBL	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	40
	Middle	BDL	ND	BDL	BDL	BDL	ND	BDL	BDL
	Bottom	BDL	ND	BDL	BDL	BDL	ND	BDL	BDL
FB	Surface	BDL	BDL	BDL	BDL	BDL	BDL	BDL	40
	Middle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	70
	Bottom	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL

Table 2.14: Dissolved oxygen (DO, mg/L), pH, total nitrogen (TN, mg/L), and total phosphorus (TP, μg/L) for required GAEPD stations in 2001. Required stations are Chattahoochee arm at Lanier Bridge, Hwy 53 (CHA); Chestatee arm at Bolling Bridge, Hwy 53 (CHE); lake off Flowery Branch Bay (FBL); Browns Bridge (BB); and above Buford Dam (DAM)

	CHA				CHE				FBL			
DATE	DO	pH	TN	TP	DO	pH	TN	TP	DO	pH	TN	TP
Apr	10.39	7.78	0.72	30	10.62	7.83	0.34	30	10.53	7.27	<.28	BDL
May	9.80	9.42	0.49	20	8.44	8.28	0.42	BDL	9.25	7.99	0.36	BDL
Jun	7.72	7.96	0.42	BDL	7.85	7.53	0.22	BDL	7.99	7.41	0.62	BDL
Jul	7.28	7.98	0.19	BDL	7.58	7.52	0.27	BDL	7.28	7.21	<.31	BDL
Aug	7.42	8.05	0.32	BDL	7.75	7.88	0.19	BDL	7.40	7.26	0.30	BDL
Sep	7.53	7.94	0.22	20	7.40	7.20	0.20	BDL	7.62	7.48	0.28	BDL
Oct	8.00	7.13	0.22	BDL	7.49	720		BDL	7.73	7.15	<.21	BDL

	BB				DAM			
DATE	DO	pH	TN	TP	DO	pH	TN	TP
Apr	10.10	7.40	0.39	BDL	9.72	7.23	0.25	BDL
May	8.90	8.79	0.39	BDL	8.30	7.77	0.48	BDL
Jun	8.14	7.37	0.64	BDL	7.90	7.41	0.47	BDL
Jul	7.13	7.40	0.33	BDL	7.10	7.23	0.26	BDL
Aug	7.24	7.56	0.33	BDL	7.15	7.20	0.26	BDL
Sep	7.36	7.31	0.25	BDL	7.33	7.41	0.28	BDL
Oct	7.94	7.24	0.26	BDL	7.67	7.15	<0.20	BDL
Min	4.0	6.0						
Min avg	5.0							
Max		9.5	4.0					

Table 2.15: Dissolved oxygen (DO, mg/L), pH, total nitrogen (TN, mg/L), and total phosphorus (TP, μg/L) for supplementary GAEPD stations in 2001. Supplementary stations are Little River Bay (LTR), Flat Creek Bay (FC), Six Mile Creek Bay (SIX), Balus Creek Bay (BAL), and Mud Creek Bay (MUD). ND = no data. BDL = below detection limit of 20 μg/L.

	LTR				FC				SIX			
DATE	DO	pH	TN	TP	DO	pH	TN	TP	DO	pH	TN	TP
April	10.49	7.75	ND	ND	10.38	7.88	2.26	ND	7.43	0.58	9.08	BDL
May	8.26	9.14	0.44	BDL	10.89	9.39	0.80	BDL	7.88	0.34	8.50	BDL
Jun	7.76	7.71	0.47	BDL	7.97	7.67	0.64	BDL	7.50	0.47	7.59	BDL
Jul	7.59	7.67	<0.21	BDL	7.57	8.13	0.61	BDL	7.20	<0.22	7.10	BDL
Aug	7.40	8.27	0.29	40	7.80	8.07	0.86	70	7.53	0.32	6.87	440
Sep	7.09	7.11	<0.18	BDL	7.17	7.68	0.91	BDL	7.35	0.29	7.34	BDL
Oct	6.43	6.88	0.30	BDL	7.41	7.36	0.72	BDL	7.22	<0.26	7.37	BDL

	BAL				DUD			
DATE	DO	pH	TN	TP	DO	pH	TN	TP
Apr	10.19	7.54	0.85	ND	9.96	7.26	1.38	ND
May	10.53	9.21	0.58	40	9.37	7.69	0.53	BDL
Jun	7.98	7.46	0.66	BDL	7.67	7.47	0.46	BDL
Jul	7.31	7.50	0.42	BDL	7.39	7.33	<.33	BDL
Aug	7.38	7.97	0.48	BDL	7.60	7.78	0.33	30
Sep	7.10	7.41	0.36	BDL	7.06	7.41	0.29	BDL
Oct	7.35	7.35	0.20	BDL	7.45	7.33	0.27	30

Table 2.16: Mean and standard deviation for chlorophyll *a* for 100 stations sampled between 1987-1995 at three depths (S-Surface, M-Middle, B-Bottom) on Lake Lanier.

	chlorophyll *a* (μg/L)		
Date	Surface	Middle	Bottom
Aug 1987	6.87 ± 9.5	6.63 ± 5.4	7.2 ± 13.8
Oct 1987	3.53 ± 3.11	2.52 ± 2.10	2.91 ± 3.09
Dec 1987	5.48 ± 3.55	3.34 ± 1.60	3.99 ± 1.89
Mar 1988	6.39 ± 5.04	4.51 ± 2.16	4.79 ± 3.43
Annual Mean	5.13	4.26	4.25
Jun 1988	3.36 ± 2.31	4.29 ± 3.16	2.37 ± 2.35
Aug 1988	1.58 ± 1.72	2.45 ± 3.36	1.97 ± 2.23
Dec 1988	3.35 ± 4.12	2.67 ± 2.64	2.30 ± 1.76
Mar 1989	1.90 ± 1.36	0.94 ± 0.74	1.12 ± 1.15
May 1989	1.63 ± 1.10	1.03 ± 0.91	1.00 ± 1.10
Annual Mean	2.12	1.77	1.60
Aug 1989	2.24 ± 1.43	3.69 ± 2.89	2.05 ± 2.08
Jan 1990	6.73 ± 3.42	6.92 ± 3.21	3.94 ± 2.58
Mar 1990	5.15 ± 4.11	0.30 ± 0.17	1.83 ± 1.98
Jun 1990	2.55 ± 1.69	2.61 ± 1.89	2.51 ± 1.82
Annual Mean	3.84	3.38	2.58
Aug 1990	2.14 ± 1.36	2.81 ± 1.47	2.49 ± 2.44
May 1991	2.72 ± 1.45	1.47 ± 0.60	1.32 ± 0.79
Sep 1991	3.00 ± 4.22	3.21 ± 3.39	1.84 ± 3.41
Dec 1991	4.00 ± 1.89	4.10 ± 1.32	3.24 ± 1.40
Mar 1992	0.57 ± 0.32	0.56 ± 0.25	0.49 ± 0.08
Jul 1992	0.50 ± 0.00	0.61 ± 0.27	0.52 ± 0.12
Annual Mean	2.02	2.12	1.52
Sep 1992	0.58 ± 0.26	0.72 ± 0.54	0.61 ± 0.40
Aug 1993	0.96 ± 1.06	1.20 ± 1.59	1.01 ± 1.48
Aug 1994	1.41 ± 2.29	1.90 ± 2.50	1.44 ± 1.71
Aug 1995	2.92 ± 2.84	2.65 ± 2.71	2.50 ± 5.53

Table 2.17: Monthly and mean annual chlorophyll a (μg/L) at required and supplementary GAEPD stations in 2001. Required stations are Chattahoochee arm at Lanier Bridge, Hwy 60 (CHA); Chestatee arm at Bolling Bridge, Hwy 53 (CHE), lake off Flowery Branch Bay (FBL); Browns Bridge (BB; and above Buford Dam (DAM) Browns Bridge (BB; and above Buford Dam (DAM). BDL = below detection limit of 1 μg/L. Standard applies to the annual mean.

	Required stations					Supplementary stations				
DATE	CHA	CHE	FBL	DAM	BB	LTR	FC	SIX	BC	MUD
Apr	12.39	7.50	4.03	3.27	5.68	32.60	9.87	6.58	8.07	4.65
May	12.69	7.42	7.48	5.84	8.98	6.88	16.31	6.81	13.53	2.31
Jun	4.65	4.40	5.59	4.65	4.03	5.88	4.37	16.80	24.52	2.77
Jul	4.34	BDL	2.79	2.17	BDL	5.88	5.88	2.79	3.57	4.03
Aug	9.91	BDL	5.11	2.79	BDL	9.91	4.96	4.03	6.81	5.57
Sep	8.67	5.14	3.72	2.79	4.34	1.55	6.50	4.03	3.88	4.03
Oct	11.77	9.60	4.19	4.96	7.34	16.33	8.98	6.50	6.81	7.12
Annual Mean	9.20	5.20	4.70	3.80	4.60	11.35	8.12	6.79	9.60	4.35
Standard	10	10	5	5	5					

Table 2.18: Chlorophyll a (μg/L) in the Chattahoochee arm at Thompson Bridge, Hwy 129 (CHA); in the Chestatee arm at Bolling Bridge, Hwy 53 (CHE); Six Mile Creek Bay (SIX); the lake off Flowery Branch Bay (FBL); Browns Bridge (BB); Buford Dam (DAM); and the mouth of Flat Creek (FCM) in 1966-1967 (Holder (1967). ND = no data.

Date	CHA	CHE	SIX	FBL	BB	DAM	Date	FCM
May-Jun 1966	1.78	1.26	1.30	1.96	1.26	1.47	Jun 1966	19.05
Jun-Jul 1966	1.49	1.06	0.87	0.86	0.93	0.99	Jul 1966	4.52
							Aug 1966	4.90
Aug-Sep 1966	0.99	0.81	0.70	0.74	0.74	0.68	Sep 1966	3.88
							Oct-66	11.93
Oct-Dec 1966	3.75	1.34	1.59	1.28	2.11	1.18	Nov 1966	7.79
							Dec-66	15.38
Jan-Feb 1967	9.50	2.07	2.59	1.07	4.49	1.71	Jan 1967	ND
Mean	3.50	1.31	1.41	1.18	1.91	1.20	Mean	9.64

Table 2.19: Stations and sampling periods from 1987-1997 when fecal coliform bacteria exceeded 200 CFU/100 mL.

Date	Station #	Fecal coliforms
Aug-87	3	240
	4	300
	21	2500
	45	1000
	50	1000
	75	367
	99	200
Dec-87	3	3500
	27	400
	66	495
Mar-89	66	1925
Jan-90	2	950
	3	820
	5	408
	27	1255
Mar-90	3	895
	5	1485
	12	1575
	15	376
	16	206
	18	264
	27	235
	34	228
	53	304
	66	340
	75	274
May-91	3	372
Sep-91	1	372
Mar-92	66	250
Jul-92	15	220
	79	1000
Sep-92	39	332
Sep-94	1	252
Aug-95	28	740
	43	484
Sep-96	1	360

Table 2.20: Relative areal hypolimnetic oxygen deficit (RAHOD) (mg-O_2/cm^2/day), number of days used to calculate RAHOD, and thickness of the hypolimnion (H_{THK}) at Browns Bridge (BB), Flat Creek Bay (FC), lake off Flowery Branch (FBL) and Flowery Branch Bay (FBB) between 1966-2001.

	BB		FBL		FC		FBB	
Year	Days (H_{THK})	RAHOD	Days (H_{THK})	RAHOD	Days (H_{THK})	RAHOD	Days (H_{THK})	RAHOD
1988	92 (26)	0.064	74 (26)	0.040	74 (11)	0.043	74 (12)	0.031
1989	61 (23)	0.083	61 (28)	0.043	61 (14)	0.098	61 (5)	0.019
1990	97 (26)	0.076	97 (35)	0.045	97 (10)	0.075	97 (7)	0.019
1992	106 (27)	0.062	109 (20)	0.048	130 (19)	0.055	109 (7)	0.017
2001	95 (18)	0.087	141 (17)	0.051	95 (19)	0.112	78 (12)	0.066
2002	100 (21)	0.058	94 (26)	0.060	100 (20)	0.055	100 (12)	0.056

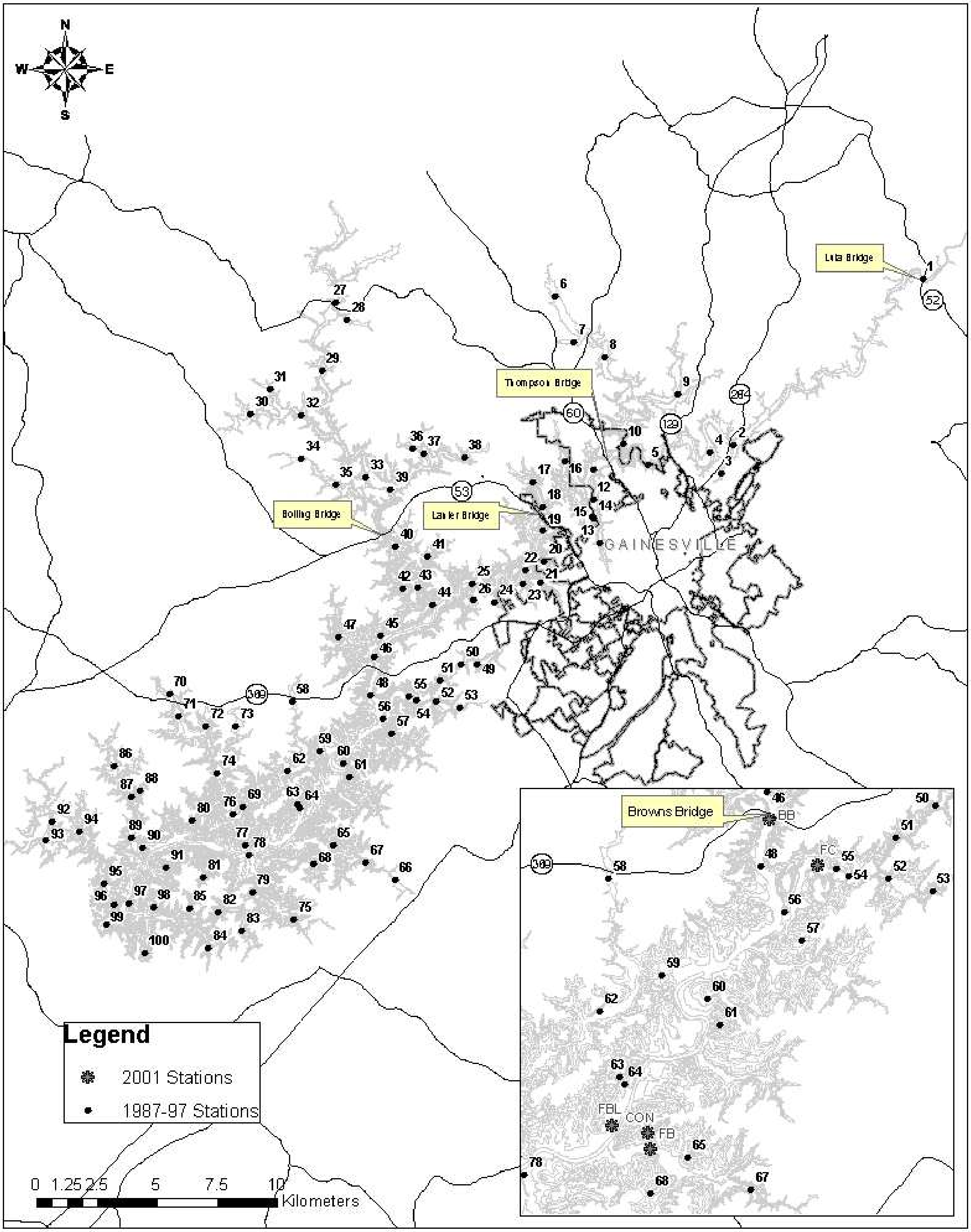


Figure 2.1: Lake Lanier sampling locations, 1987-1997 and 2001-2002, including Brown's Bridge (BB), Flat Creek Bay (FC), lake off Flowery Branch Bay (FBL), Flowery Branch Bay (FB) and control (CON) in Flowery Branch Bay.

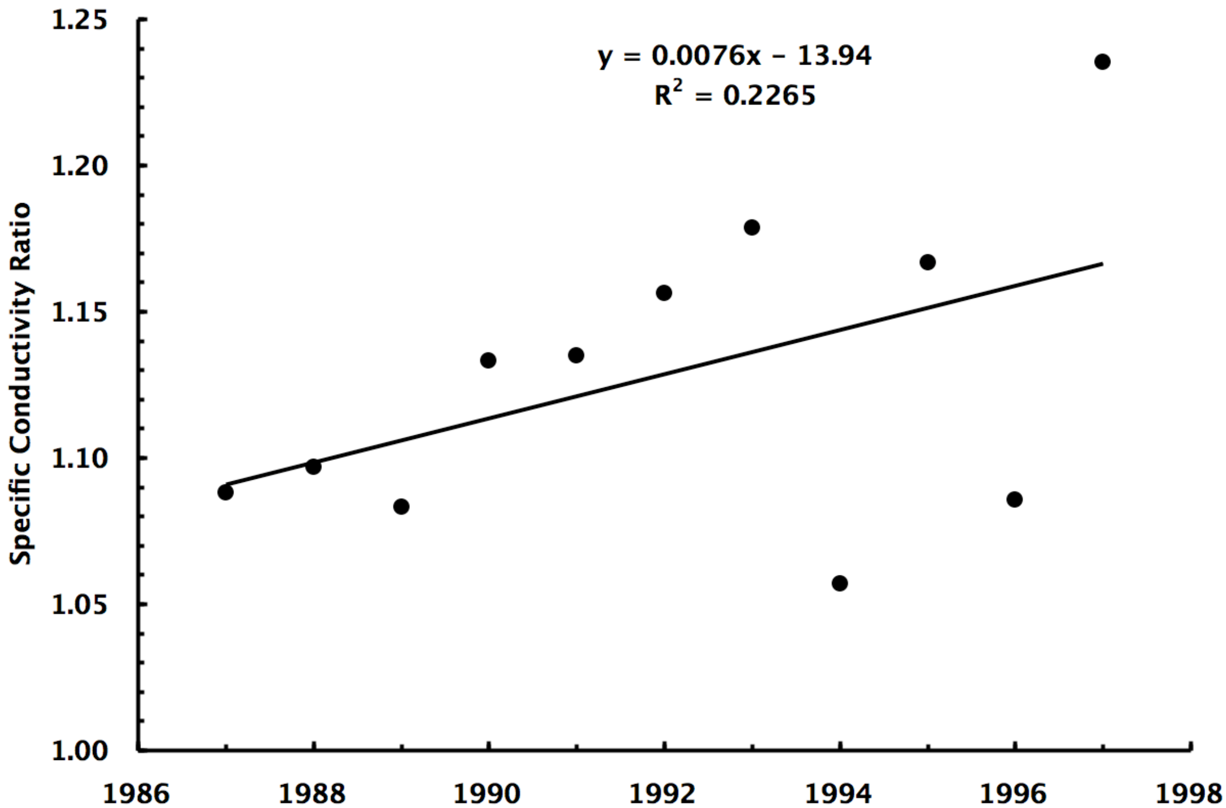


Figure 2.2: Ratio of mean specific conductivity (μS/cm) at the lake bottom (hyplimnetic) to surface (epilimnetic) observations during the late-summer (Aug-Sep) at 100 stations on Lake Lanier (34 in 1997).

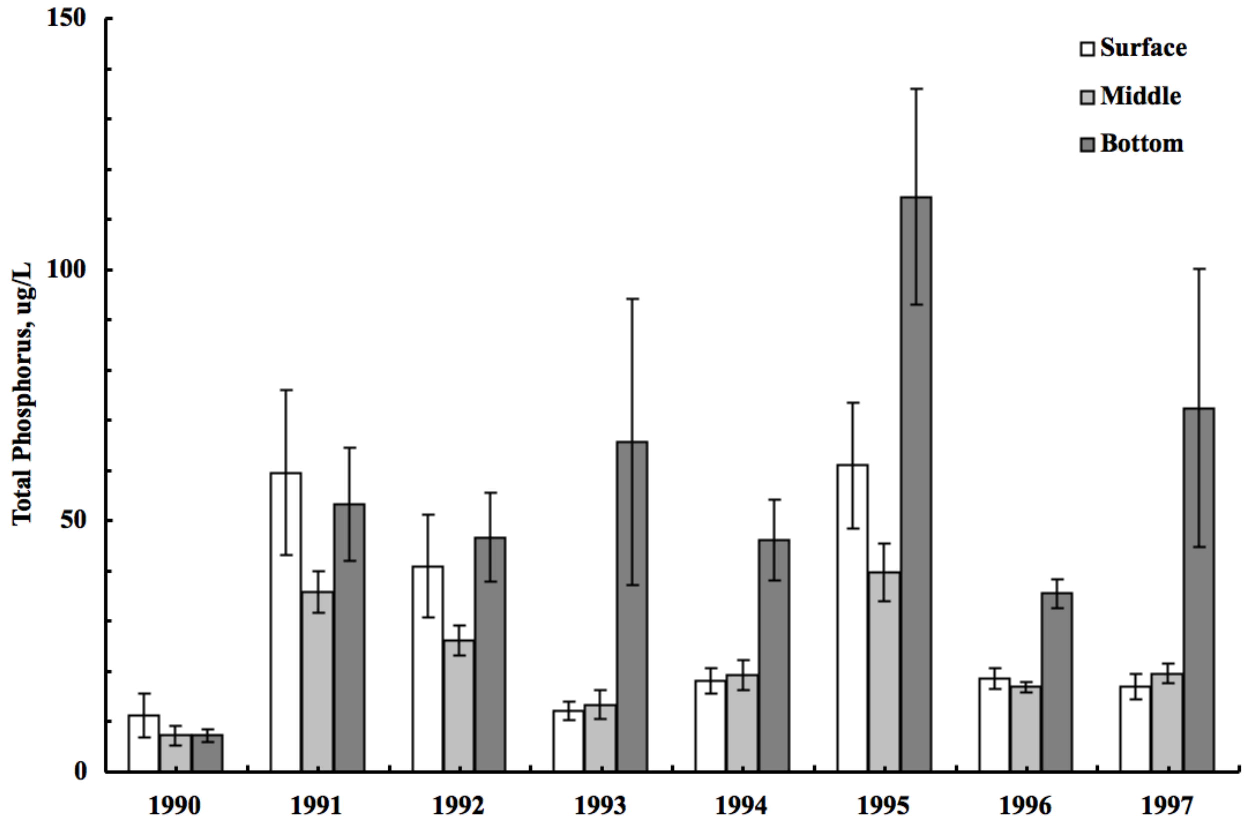


Figure 2.3: Mean total phosphorus (μg/L) in Lake Lanier at three depths (surface, middle, bottom) at 100 stations from 1990-1996 and 34 stations in 1997.

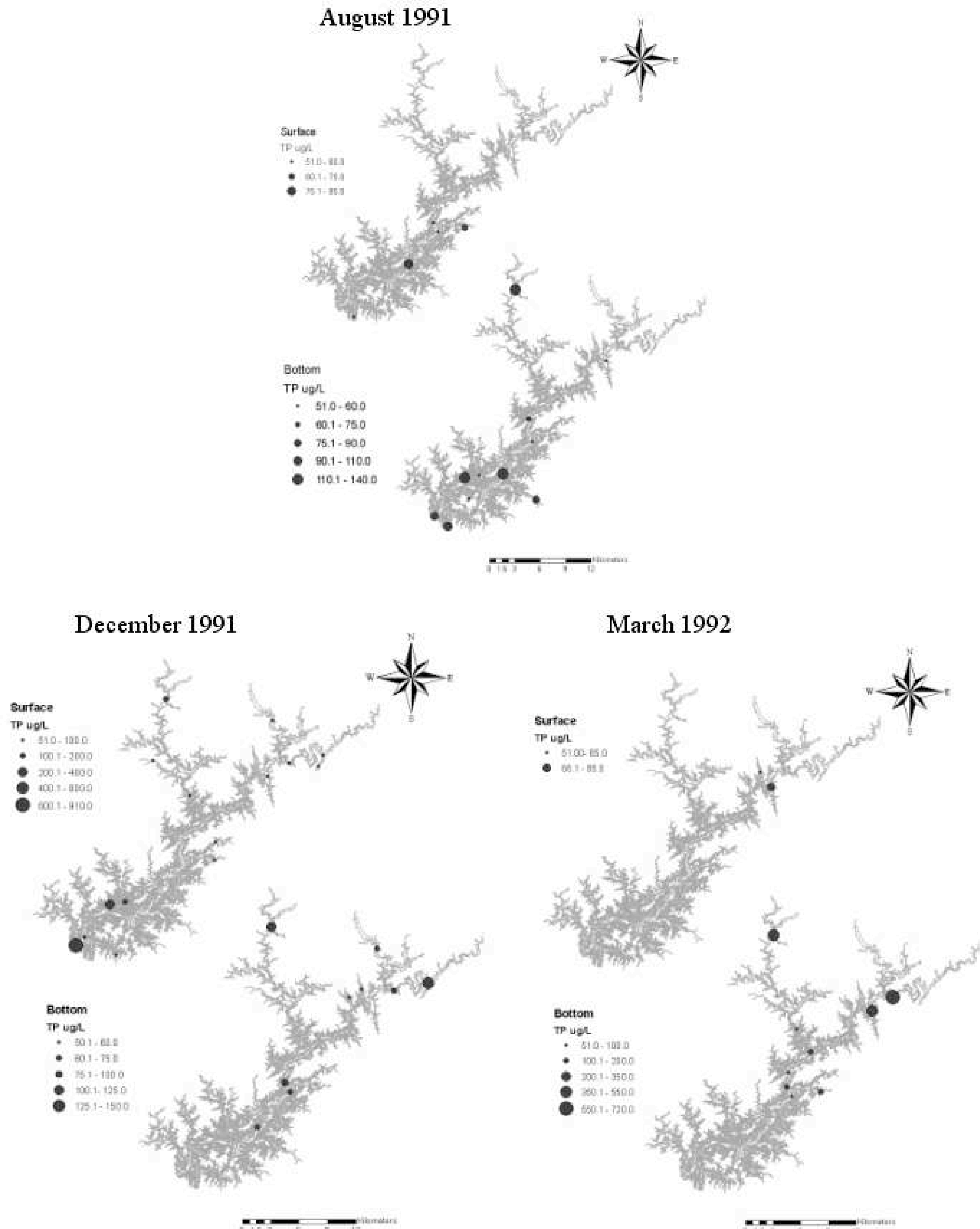
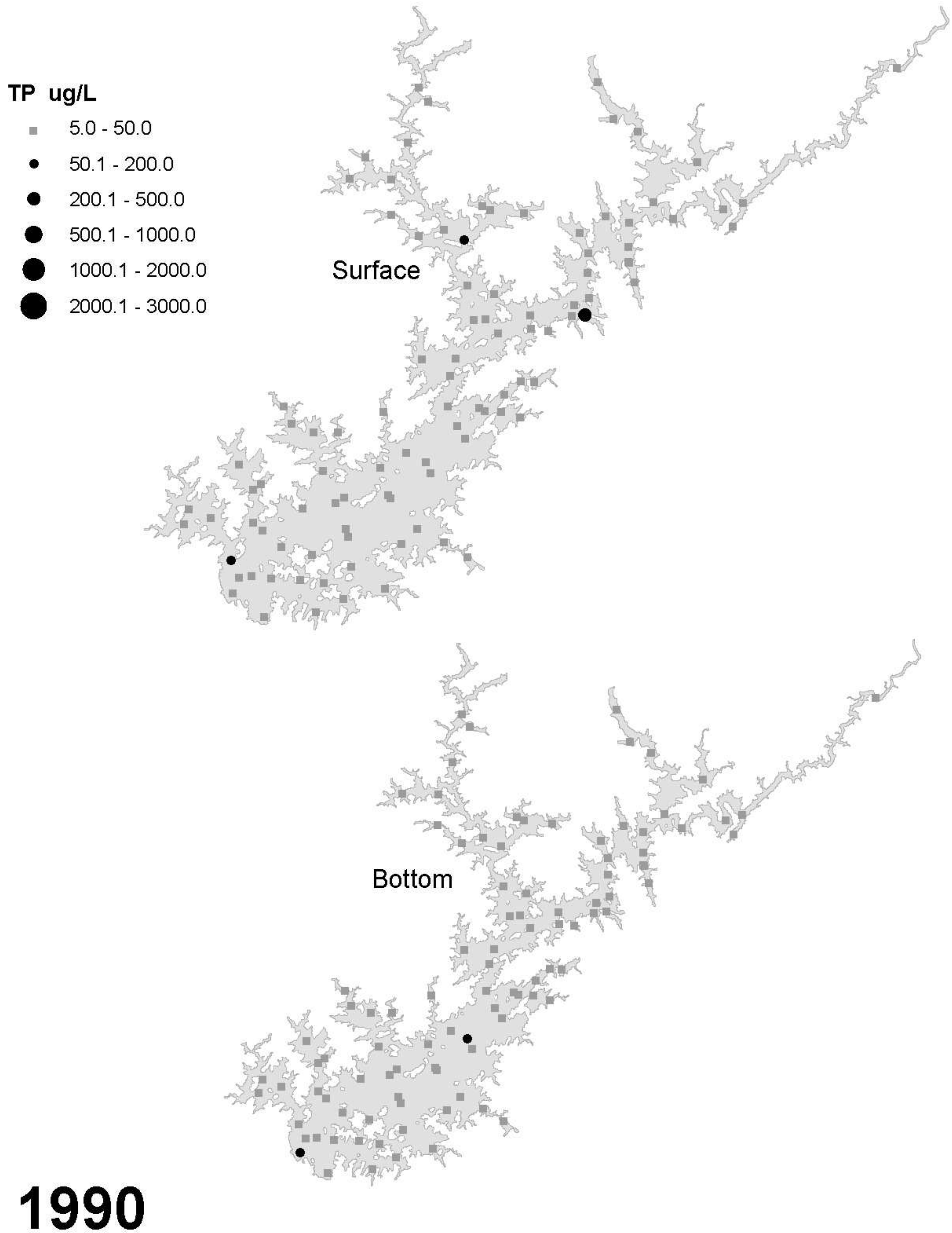
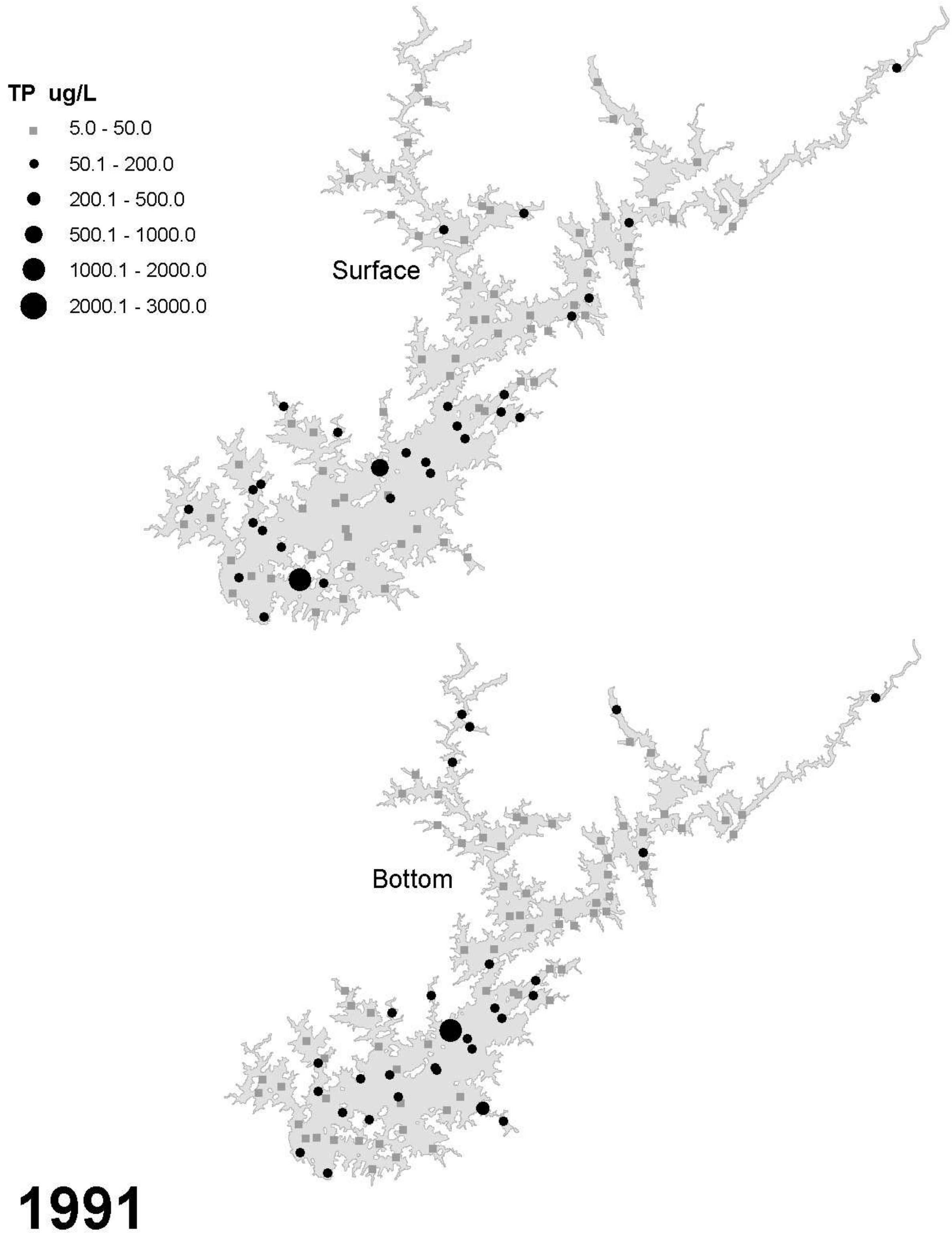
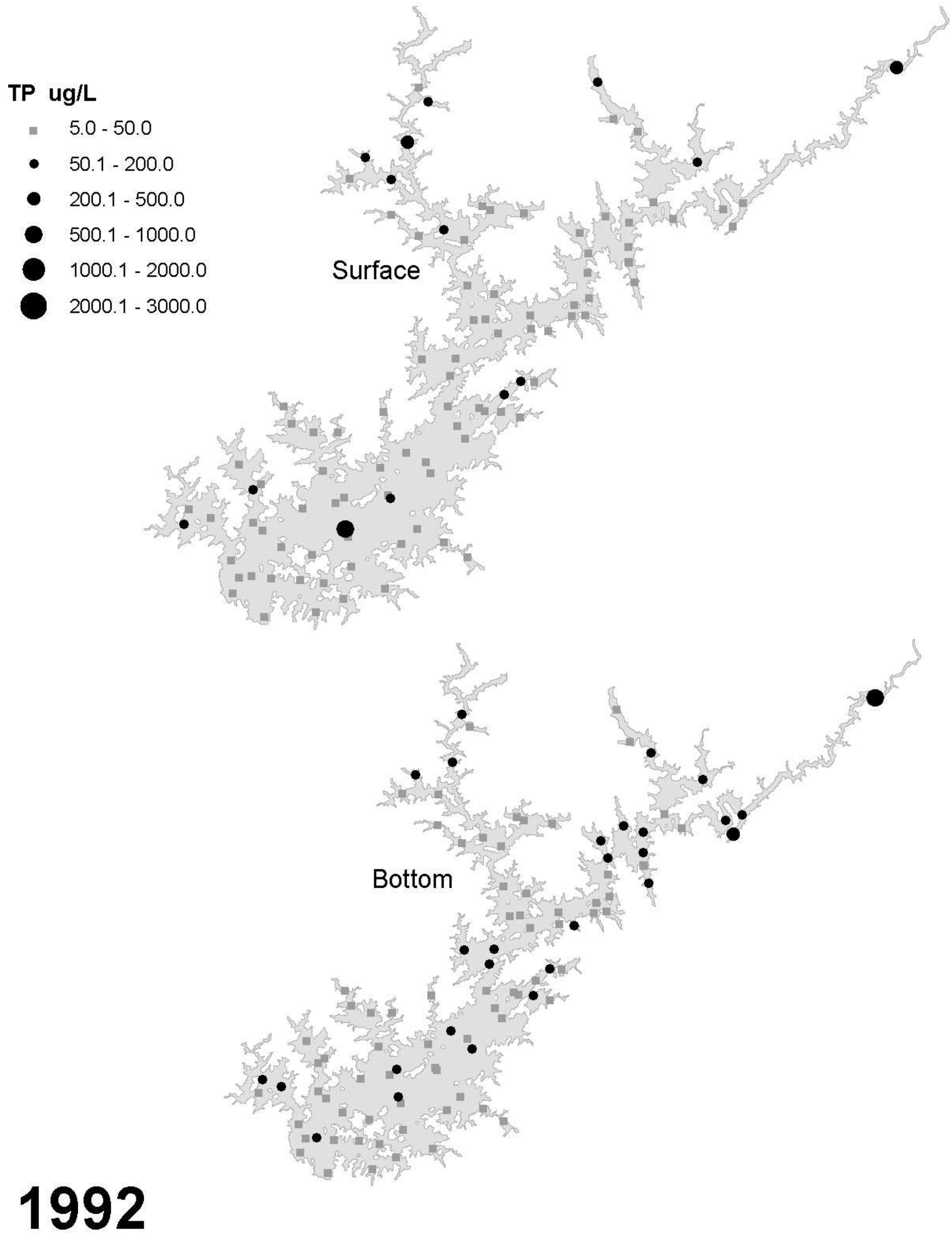
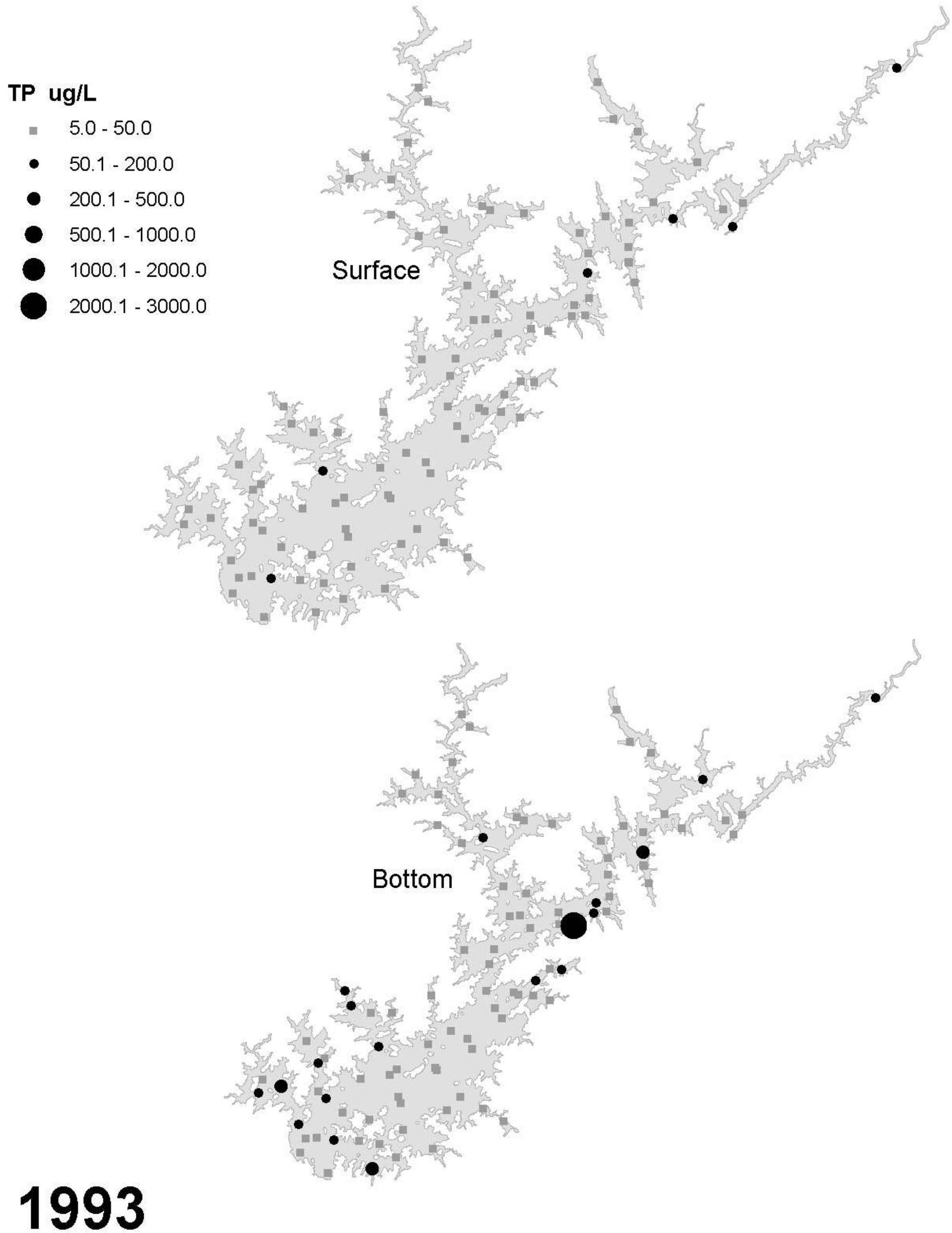


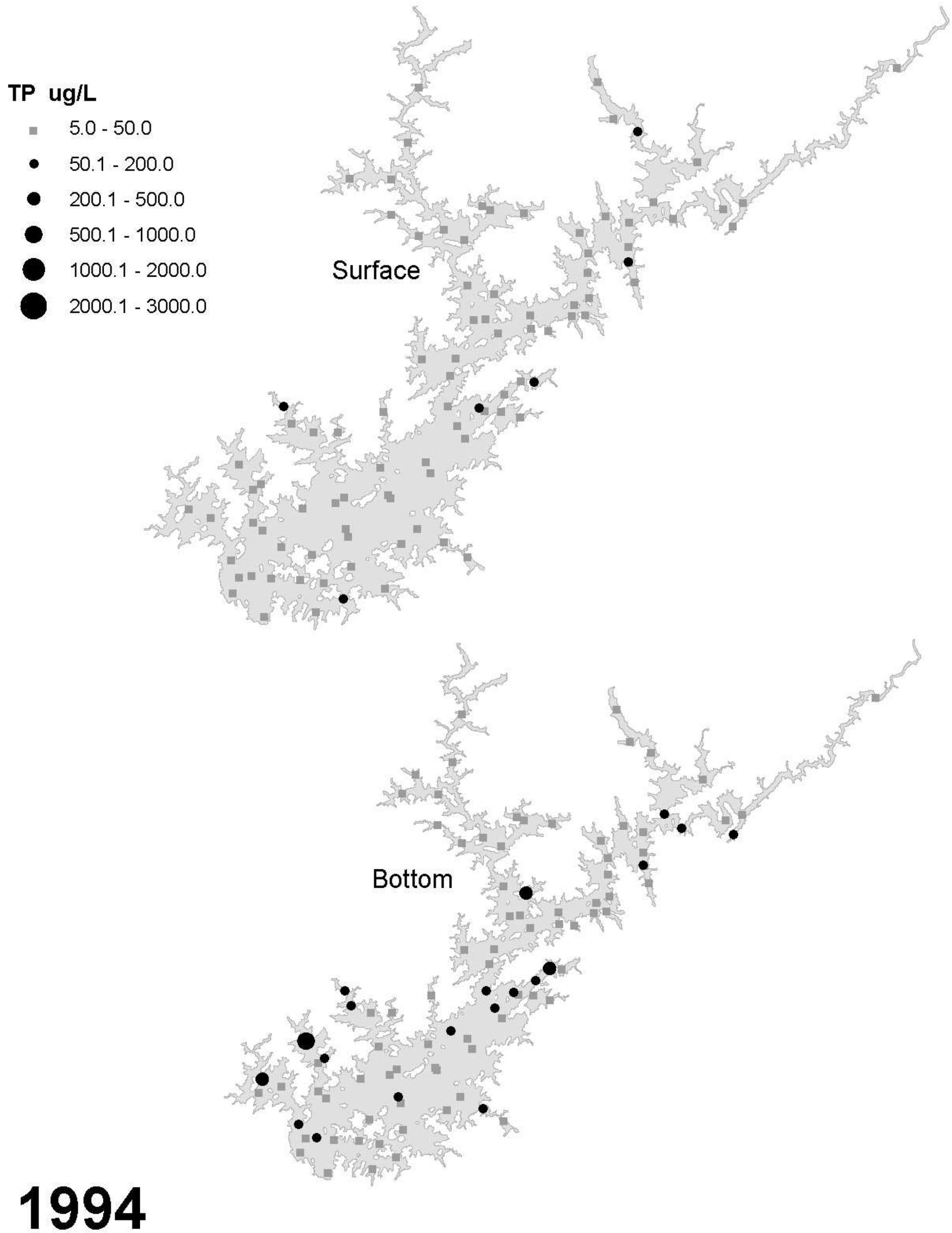
Figure 2.4: Seasonal distribution of total phosphorus in Lake Lanier during August 1991 (summer), December 1991 (winter), and March 1992 (spring).

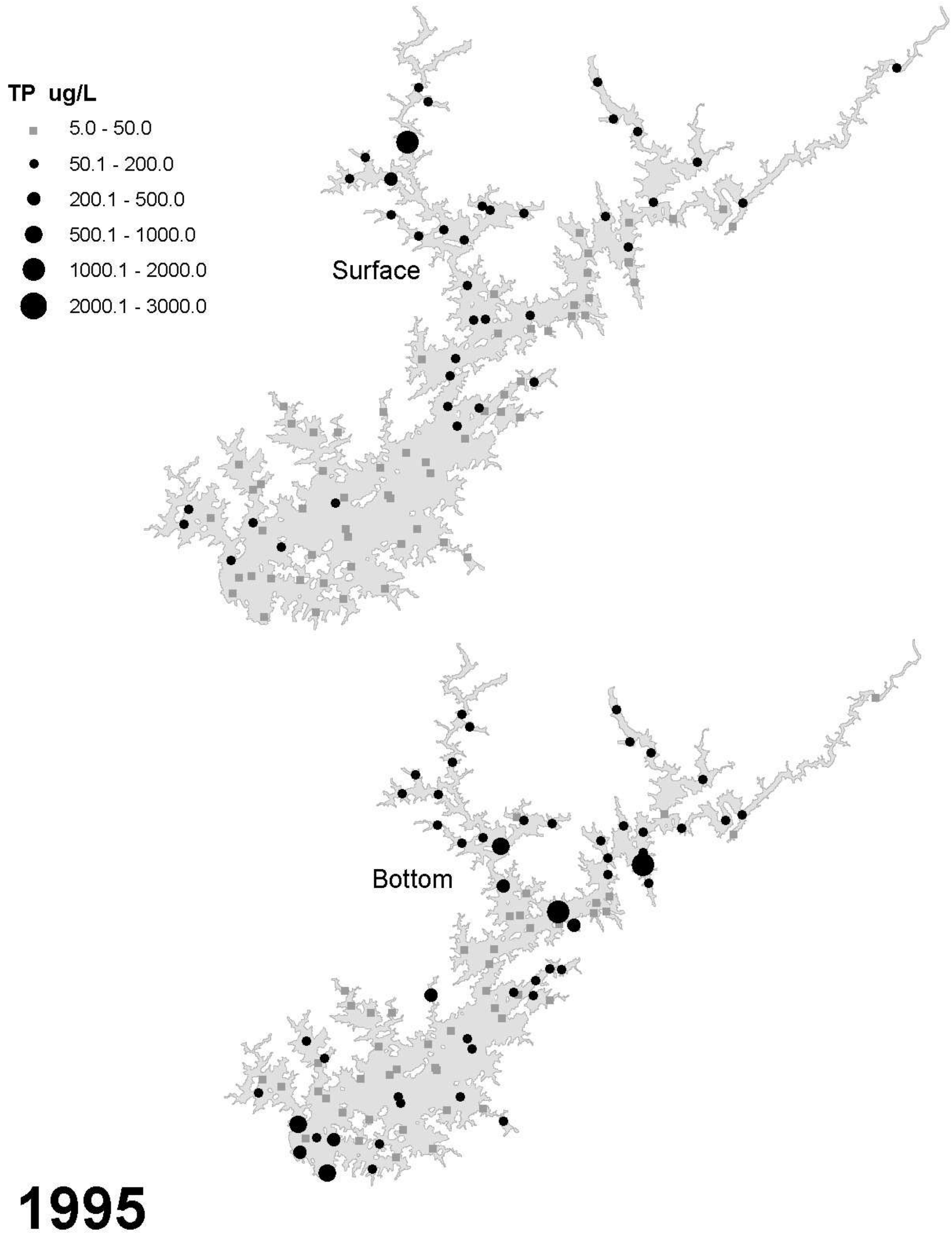

TP ug/L
5.0 - 50.0
50.1 - 200.0
200.1 - 500.0
500.1 - 1000.0
1000.1 - 2000.0
2000.1 - 3000.0
Surface
Bottom
1990


TP ug/L
5.0 - 50.0
50.1 - 200.0
200.1 - 500.0
500.1 - 1000.0
1000.1 - 2000.0
2000.1 - 3000.0
Surface
Bottom
1991

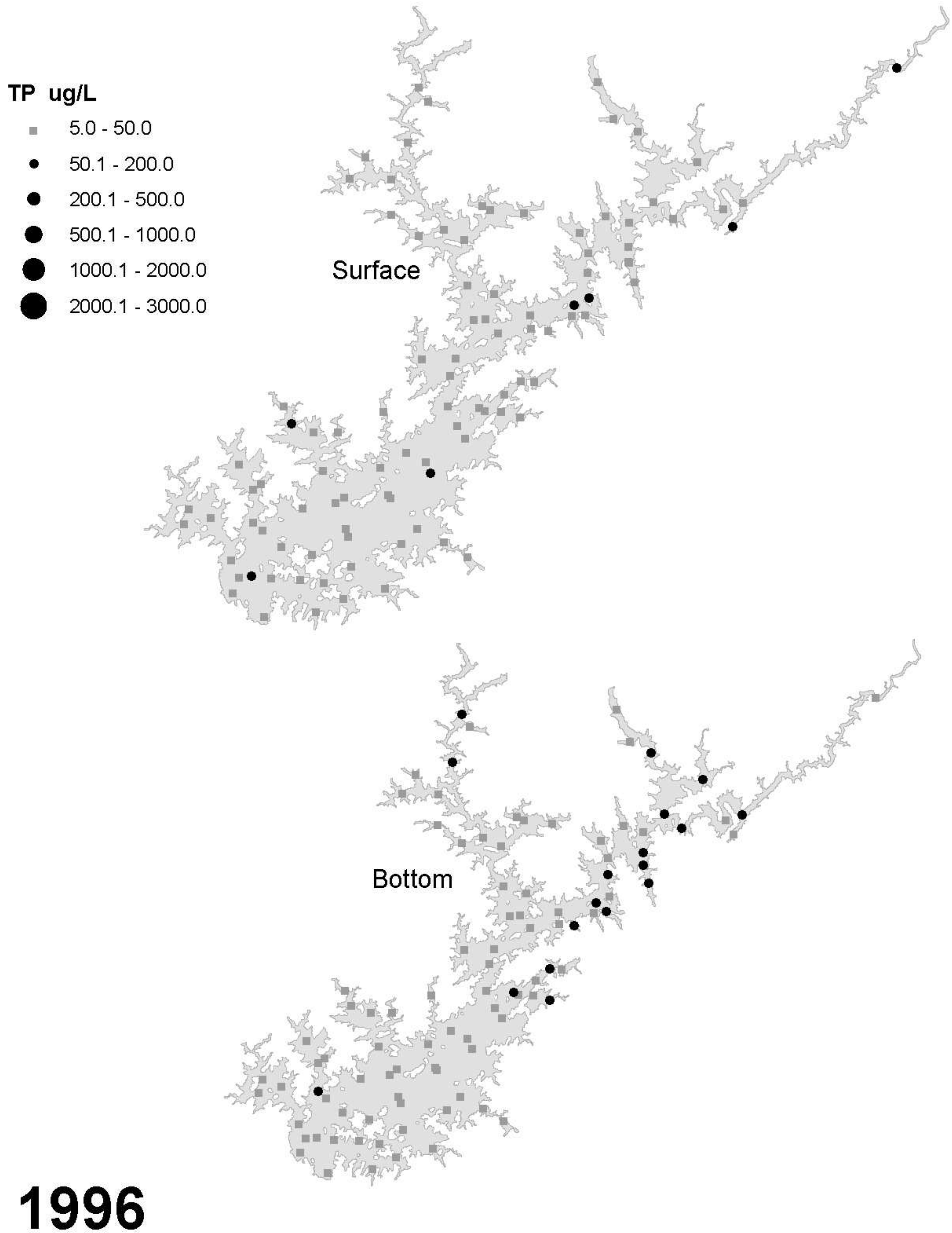

TP ug/L
5.0 - 50.0
50.1 - 200.0
200.1 - 500.0
500.1 - 1000.0
1000.1 - 2000.0
2000.1 - 3000.0
Surface
Bottom
1992


TP ug/L
5.0 - 50.0
50.1 - 200.0
200.1 - 500.0
500.1 - 1000.0
1000.1 - 2000.0
2000.1 - 3000.0
Surface
Bottom
1993


TP ug/L
5.0 - 50.0
50.1 - 200.0
200.1 - 500.0
500.1 - 1000.0
1000.1 - 2000.0
2000.1 - 3000.0
Surface
Bottom
1994



TP ug/L
5.0 - 50.0
50.1 - 200.0
200.1 - 500.0
500.1 - 1000.0
1000.1 - 2000.0
2000.1 - 3000.0
Surface
Bottom
1995


TP ug/L
5.0 - 50.0
50.1 - 200.0
200.1 - 500.0
500.1 - 1000.0
1000.1 - 2000.0
2000.1 - 3000.0
Surface
Bottom
1996

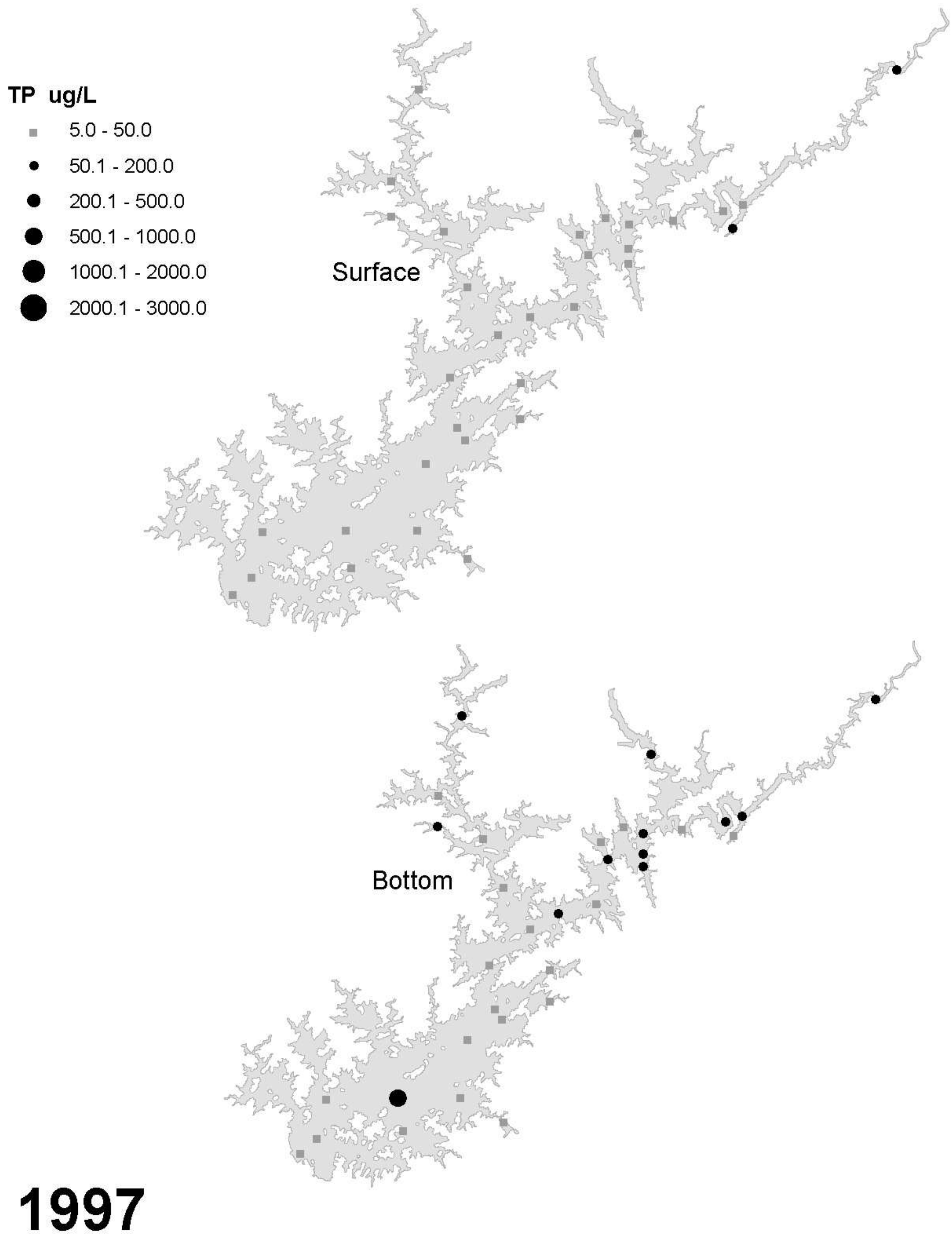


Figure 2.5: Distribution of Lake Lanier epilimnetic (surface) and hypolimnetic (bottom) concentrations of total phosphorus in August, 1990-1997.

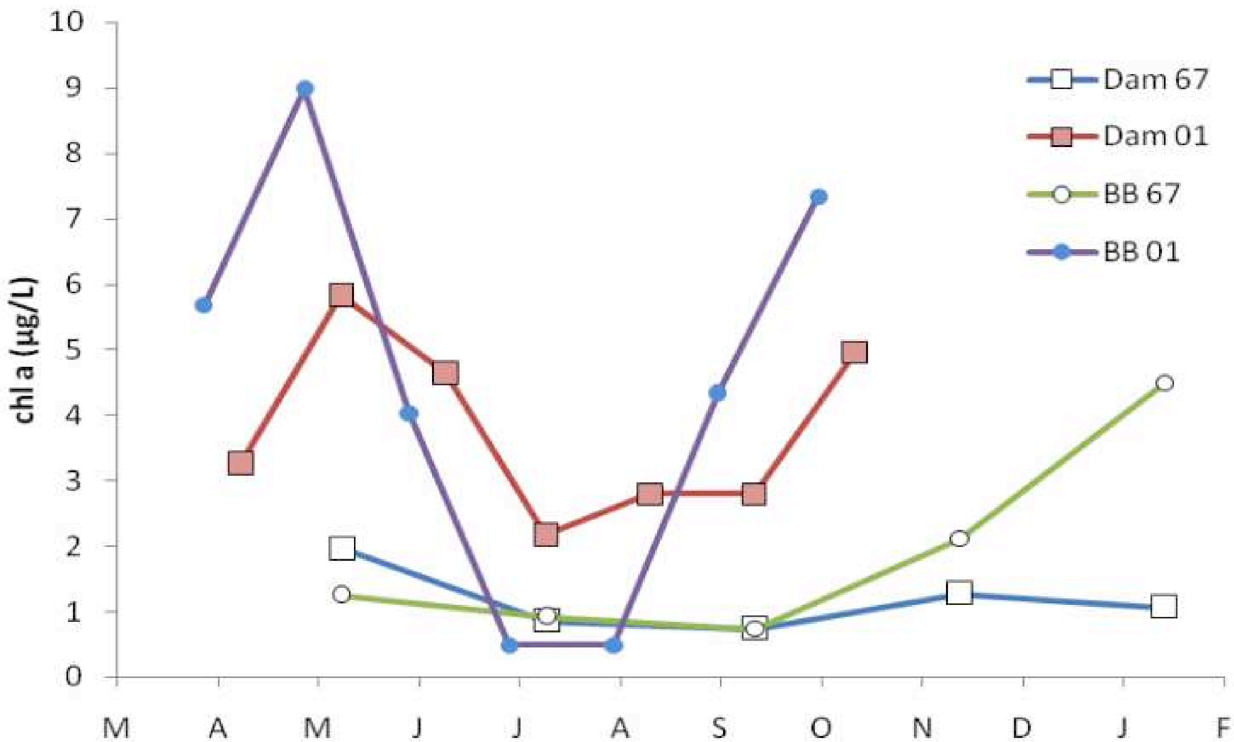


Figure 2.6: Seasonal variation chlorophyll *a* concentration at Browns Bridge (BB) and above Buford Dam (Dam) in 1967 and 2001.

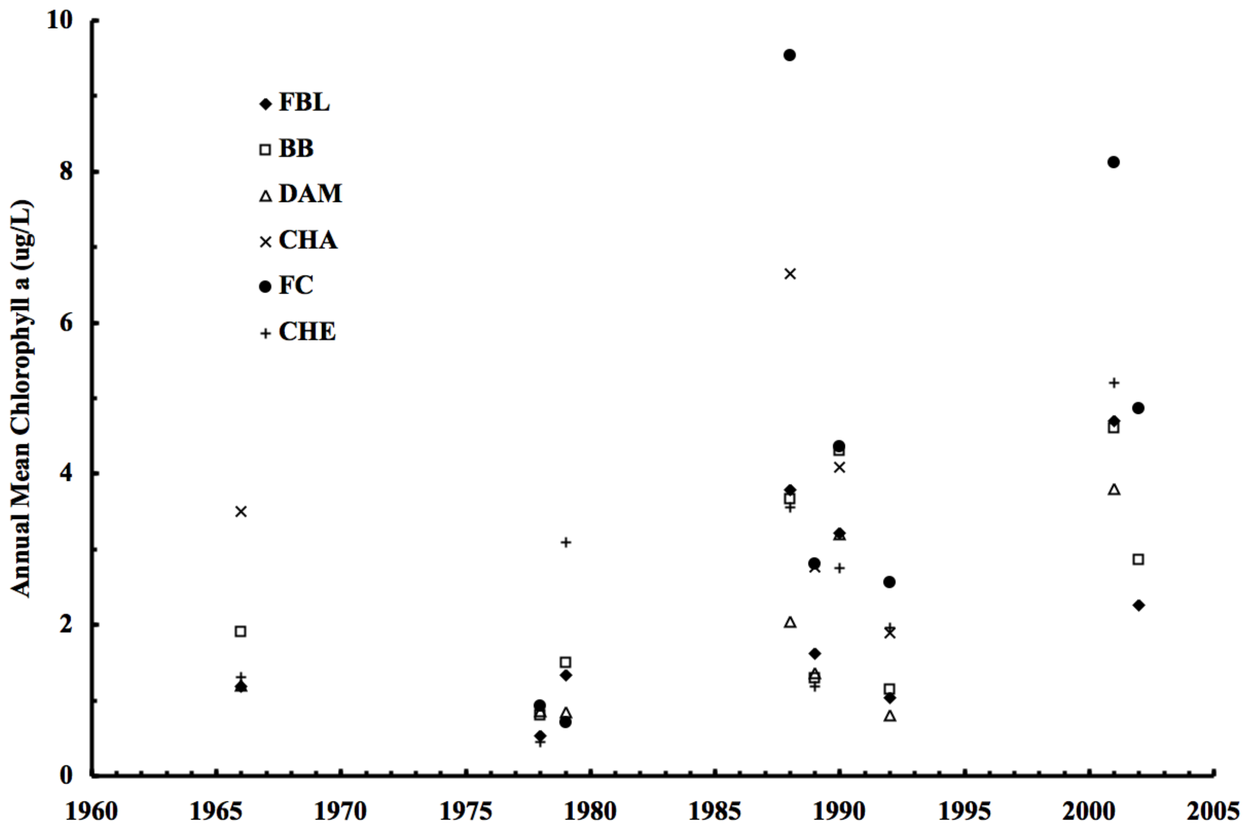


Figure 2.7: Mean annual chlorophyll *a* between 1966-2001 in the Lake Lanier off Flowery Branch Bay (FBL), at Browns Bridge (BB), above Buford Dam (DAM), and within the Chattahoochee (CHA), Flat Creek (FC), and Chestatee (CHE) arms.

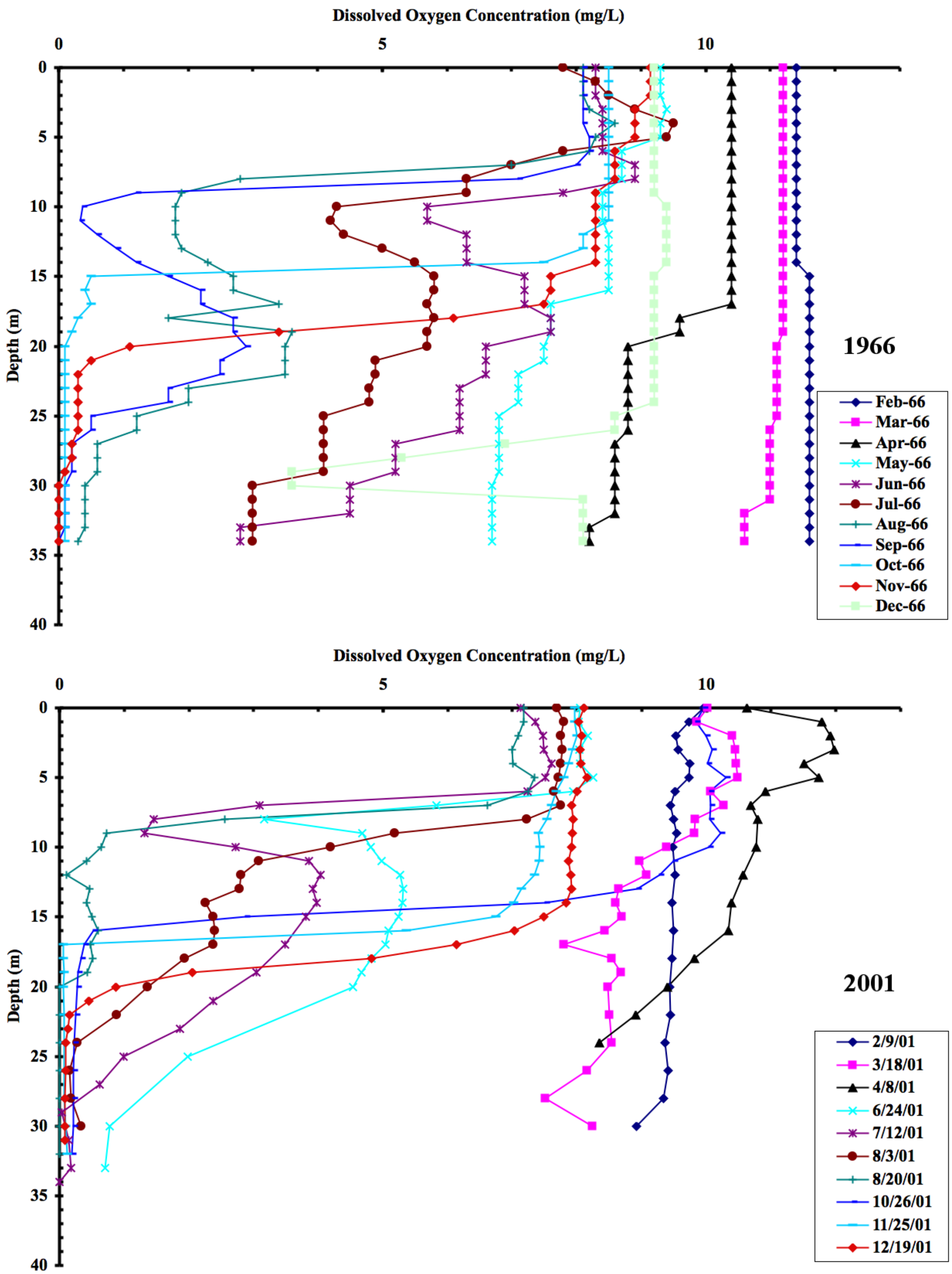


Figure 2.8: Monthly dissolved oxygen profiles at Browns Bridge in 1966 and 2001.

Chapter 3

Lake Lanier Trophic State and Metabolism

Lakes, ponds, and other impoundments are commonly characterized according to their trophic state. Trophic states range from pristine, low-nutrient, oligotrophic conditions, through moderately enriched, mesotrophic conditions, to enriched, eutrophic conditions. Eutrophic systems can be further extended to include meso-eutrophic and hyper-eutrophic conditions when extremely enriched. Streams may also be categorized by trophic state, although the importance of allocthonous material in stream ecosystems, and it s perceived lesser importance in lakes and reservoirs, has meant that trophic state in streams is seen as a somewhat different phenomenon than in lakes.

Yet many lakes are net heterotrophic (Cole et al., 1994; Cole et al., 2000; Del Giorgio, 1998), which emphasizes the need to consider metabolic parameters as well as nutrients and algal biomass in aquatic systems of all types. Dodds (2000, 2007) has propsed that trophic state for streams should be divided into net autotrophic and net heterotrophic, in order to allow consideration of external carbon sources as well as internal ones. This may be a useful division for other aquatic systems as well.

While classifying lentic systems according to their trophic state provides a useful management tool, there is no universally applicable system that satisfactorily classifies every lake and impoundment (Horne and Goldman, 1994). Large impoundments are especially difficult to categorize, and the trophic status often varies with the classification method and/or the variable selected as the basis of classification (Lind, et al., 1993).

This chapter examines how metabolic measures can be used to evaluate lake trophic status. Specific hypotheses to be tested include:

1. Lake transparency (Secchi depth), phosphorus concentration, and algal biomass (chlorophyll *a*) are not adequate for establishing the trophic state of Lake Lanier because they are inconsistent with respect to each other; and

2. Alternative metabolic parameters (e.g., O_2 and CO_2 production/consumption) are more consistent and accurate measures of productivity, and indicate higher productivity than is reflected in chlorophyll *a* levels.

These hypotheses will be evaluated using data from the following sources: Master's thesis (Holder, 1967); water quality management study done for the USACE (ESIE, 1981) in 1978-1979; monitoring program by Lake Watch and Gainesville State College (GSC) from 1987-1997; this study,conducted in 2001-2002; compliance monitoring by GAEPD (2002) in 2001; and a study by the USEPA (2008) on production/respiration and sediment oxygen demand/nutrient exchange (SOD/NUTX).

Data from these sources were used to calculate Carlson's (1977) TSI, relative areal hypolimnetic oxygen deficit (R_{AHD}), and hypolimnetic percent difference from saturation, for selected years. TSI, hypolimnetic percentdifference from saturation and RAHD are compared to each other and to consumption of CO_2 in the epilimnion and production of CO_2 in the hypolimnion. These results are also compared to the direct measurement of production/respiration in the water column and SOD/NUTX (USEPA, 2008).

3.1 Background

Trophic status in lakes, ideally, should be a measure of autochthonous primary productivity, the rate of internal production by algae within the water body. However, direct measurement of productivity is difficult. Therefore, trophic status is ofteb inferred using indirect measures, such as chlorophyll *a* concentration, phytoplankton biovolume, diatom index, secchi depth, or concentration or loading of the limiting nutrient, and a variety of other parameters (Henderson-Sellers, 1984; Lind, et al, 1993).

Metabolic measures, most frequently hypolimnetic oxygen deficit, may also be used to estimate productivity (Hutchinson, 1957; Lind and Davalos-Lind, 1993), although most currently used indices do not include metabolic parameters. The association between oxygen production/consumption and trophic state has been well established (Strom, 1931; Hutchinson, 1938, 1957) and is the basis for the description of different oxygen profiles as characteristic of oligo- and eutrophy.

Hypolimnetic oxygen deficit, calculated as volumetric oxygen deficit (VHOD) or areal oxygen deficit (AHOD), may be used as a measure of trophic state (Wetzel and Lind, 2000; Matthews and Effler, 2006). AHOD is more useful for comparisons between systems, and is therefore more widely used than VHOD; however, neither method may be used once anoxia is reached, since it is based on the rate at which oxygen is consumed.

A more easily calculated parameter is the anoxic factor (AF), which can be used in anoxic waters (Nurnberg, 1995). AF is the number of days that a sediment area, equal to the whole-lake surface area, is overlain by anoxic water. AF does not correlate well with AHOD, perhaps due to differences in geochemistry and lake trophy (Nurnberg, 1995). Yet, AF is positively correlated to TP, according to Nurnberg (1995) and to Reckhow's (1977) probability of anoxia. AF correlates with lake morphometry, as does AHOD. AF also has a positive correlation with the morphometric ratio (i.e., $z/\sqrt{A}$, where z and A are the lake depth and area, respectively) while AHOD has a negative correlation (i.e., lake shape has an inverse effect on AF and AHOD) (Nurnberg, 1995). The primary factors affecting AF are nutrient concentrations and lake shape; DOC plays only a minor role.

Nurnberg's results support the generally accepted hypothesis that oxygen regime is controlled by productivity and morphometry (Hutchinson, 1957; Reckhow and Chapra, 1983). In spite of difficulties with AHOD (Hutchinson, 1957; Matthews and Effler, 2006), AHOD shows significant correlation to measures of primary production: TP (Rast and Lee, 1978; Chapra and Canale, 1991, chlorophyll a (Vollenweider and Janus, 1981) and Secchi disc transparency (Lasenby, 1975). Oxygen is also of interest because of its effects on geo- and aqueous chemistry via oxidation-reduction processes, as well as its powerful influence on biological processes, diversity and abundance.

Increasing interest in the effects of climate change has lead to increasing interest in metabolic parameters, CO_2 as well as O_2. Some research (Cole, et al, 2000, Streigel and Michmerhuizen, 1998) shows that many lakes are net heterotrophic, and contribute a small but globally significant amount of carbon to the atmosphere, about 0.14×10^{15} g/yr (Anderson, et al, 1999). Some lakes, however, are balanced on an annual cycle, being net autotrophic in summer and net heterotrophic in winter (Staehr and Sand-Jensen, 2007). Epilimnetic CO_2 may be used to determine net auto- or heterotrophy. Epilimnetic CO_2 levels below saturation are considered indicative of net autotrophy, which is generally associated with eutrophy-values above saturation are considered indicative of net heterotrophy, which is generally associated with oligotrophy. However, lakes of a range of trophic states measured by other parameters have been found to be supersaturated in CO_2 (Cole et al., 2000).

Hypolimnetic accumulation of CO_2 may also be used, as oxygen deficit is used, to estimate productivity. Hypolimnetic CO_2 has an advantage over O_2 as a measure of metabolism, because CO_2 may be measured after anoxia occurs. However, CO_2 levels may be affected by groundwater inputs or biological processes such as methanogenesis (Wetzel and Likens, 2000).

3.2 Methods

The methods for collection of data for secchi depth, dissolved oxygen, chlorophyll *a* and total phosphorus during the Lake Watch/GSC program in 1987-1997, and for secchi depth, dissolved oxygen and chlorophyll *a* in 2001-2002, are described in Chapter 2. The frequency, depth and number of stations for these parameters are shown in Table 2.3 for 1987-1997, and in Table 2.4 for 2001-2002.

The frequency, depth and number of stations for CO_2 in 2001-2002 are shown in Table 2.4; the methods for CO_2 data collection are described here. Samples were collected at multiple depths throughout the water column using a Van Dorn water bottle. The sample jars were filled to minimize bubbles and air space by placing the end of the tube from the water bottle in the bottom of the jar and allowing the jar to overflow. Samples were stored on ice and transported to the Marine Sciences Laboratory at the University of Georgia, where they were analyzed for CO_2 by equilibration with water samples of known pCO_2 concentration. Headspace gas was analyzed for CO_2 by nondispersive infrared gas analyzer (NDIR).

The Carlson (1977) Trophic State Index (TSI) was used to calculate the trophic status of Lake Lanier for all three parameters used by Carlson, mean annual secchi depth, chlorophyll *a*, and epilimnetic TP. The formulas for each of these indices are:

$$TSI_{SD} = 60 - 14.41 \ln SD \tag{3.1}$$

$$TSI_{Chl} = 30.6 + 9.81 \ln Cha \tag{3.2}$$

$$TSI_{TP} = 4.15 + 14.42 \ln TP \tag{3.3}$$

where SD is the mean annual Secchi depth (m), Cha is the mean annual photic zone chlorophyll *a* concentration (μg/L), and TP is the mean annual epilimnetic total phosphorus concentration (μg/L). The scale for Carlson's TSI and corresponding trophic state are shown in Table 3.1.

RAHD was calculated according to the method of Wetzel and Likens (2000). Values of RAHD that correspond to the upper limits of oligotrophy, 0.025 mg-O_2/cm^2/d, and the lower limits of oligotrophy, 0.055 mg-O_2/cm^2/d, were suggested by Mortimer (Hutchinson, 1957). These values were used to construct a scale of RAHD values that correspond to the TSI scale used by Carlson (1977). This constructed RAHD scale is shown in Table 3.1, with the corresponding Carlson TSI and trophic state.

3.3 Results

Table 3.2 shows the TSI for chlorophyll *a*, Secchi depth and total phosphorus at Thompson Bridge in the Chattahoochee arm, Bolling Bridge in the Chestatee arm, Flat Creek bay, Browns Bridge, lake off Flowery Branch bay and Buford Dam from 1966 to 2001. TSI$_{SD}$ was the most consistent of the three parameters, indicating upper mesotrophic conditions in the Chestatee and Chattahochee arms throughout the 45 year period; conditions in Flat Creek bay were similarly mesotrophic, except in 1989, when Flat Creek reached the lower end of eutrophy. TSI$_{SD}$ in the mixing zone at Browns Bridge, and in the lacustrine zone at Flowery Branch Lake and Buford Dam was more mesotrophic. TSI$_{SD}$ shows a decreasing trend, more pronounced in the mixing zone (BB) and the lacustrine zone(DAM), than in the tributary arms, over time.

TSI$_{Chl}$ was lower and more variable than TSI$_{SD}$. The disparity in TSI$_{SD}$ and TSI$_{Chl}$ was most pronounced in 1966, when TSI$_{SD}$ was 30 percent higher than TSI$_{Chl}$, and in 1991, when it was 20-30 percent higher. Trophic state based on chlorophylla ranged from mid-mesotrophic to oligo-mesotrophic. TSI$_{Chl}$, unlike TSI$_{SD}$, shows an increasing trend over time. However, TSI$_{Chl}$ decreases, as does TSI$_{SD}$ with distance downstream from the riverine zone.

Phosphorus has not been as routinely measured as secchi depth and chlorophyll *a*, but available TSI$_{TP}$ shows much more variability than TSI$_{SD}$ and TSI$_{Chl}$, ranging from lower mesotrophic to eutrophic. The greatest disparity occurred between TSI$_{TP}$ and TSI$_{Chl}$ in 1978, when TSI$_{TP}$ exceeded TSI$_{Chl}$ by 50 -150 percent.

Epilimnetic pCO_2 (Table 3.3 and Figure 3.1) at Browns Bridge, Flat Creek Bay, and lake off Flowery Branch Bay during 1999 and 2000 is below the atmospheric saturation level of 380 μatm for much of the year, indicating net autotrophic conditions. Hypolimnetic pCO_2 (Table 3.3 and Figure 3.2) is very high throughout the stratified period. Hypolimnetic pCO_2 doubled from 3200 and 3700 μatm in April at Browns Bridge and Flat Creek Bay, respectively, to 7900 μatm in December.

Hypolimnetic DO deficits are concomitantly high. Figure 3.3 shows the difference from saturation in hypolimnetic oxygen at Browns Bridge and Flowery Branch lake over the period from 1987 to 1997. The DO deficit at Browns Bridge was more extensive than at

Flowery Branch throughout that period, increasing approximately 15 percent, from 85 to 100 percent; the deficit at Flowery Branch lake was less severe, but showed a greater increase of approximately 20 percent, from 70 to 90 percent.

3.4 Discussion

Lake Lanier shows the variability in trophic state, depending on the parameter used, that has often been reported for reservoirs. TSI_{SD} was 30 percent higher than TSI_{Chl} in 1966, and 20 -30 percent higher in 1991. Holder (1967) attributed the disparity in secchi depth and chlorophyll *a* levels in 1966 to heavy rains in April and May, which caused very turbid conditions and may have contributed to light limitation of algal production. TSI_{Chl} was lower, and more variable, than TSI_{SD}. TSI_{TP} was the most variable and least consistent of the three trophic state indices.

TSI_{SD} was more spatially and temporally consistent than TSI_{TP} and TSI_{Chl}. Spatially, TSI_{SD} shows the pattern seen in many reservoirs: more turbid, and therefore more eutrophic, conditions in the riverine arms, and less turbid, and therefore more oligotrophic conditions, with distance downstream into the mixing zone and lacustrine zone. Temporally, TSI_{SD} shows a decrease, which may be the result of lower sediment loads over time. If secchi depth is affected by non-algal turbidity, as appears to be the case, then TSI_{SD} is not a useful measure of productivity.

TSI_{Chl} is more variable than TSI_{SD} , but does show a similar spatial pattern, decreasing from more eutrophic conditions in the riverine arms to more mesotrophic conditions in the mixing and lacustrine zones. Temporally, there is an increase to more eutrophic conditions, which could be explained by the increasing clarity of the water, and increased light availability for photosynthesis. However, the fact that the spatial distribution of TSI_{Chl} shows a pattern of decreasing trophic level with distance downstream cannot be explained by increasing clarity of water-that should result in increased algal growth, and increased TSI, with distance downstream. Increasing clarity of water accompanied by increasing nutrients over time could explain the pattern. However, TSI_{TP} is so variable that no clear pattern of phosphorus availability can be discerned. If anything,the pattern looks more like decreasing availability of phosphorus over time. Part of the explanation for the variability in phosphorus levels, and hence in TSI_{TP}, may be that limits of detection for phosphorus are often 10 or 20 μg/L, while the limits for oligotrophy are around 5 μg/L. Another explanation may be that phosphorus bound to sediments may not be as readily measured as dissolved and organic phosphorus, although the acidification used in most chemical measures of phosphorus should release most of it.

The data for metabolic parameters indicate more eutrophic conditions than the TSI data. Epilimnetic CO_2 levels below saturation, and large hypolimnetic accumulations of CO_2, during stratification, indicate eutrophy. Hypolimnetic oxygen deficits are extensive, and increasing over time, which is also characteristic of eutrophy.

Most of the results of a study done by the USEPA (2008) for the GAEPD in the summer of 2007 indicate that Lake Lanier is low in productivity, and oligo-mesotrophic. The exception are SOD values, which were in the eutrophic range. Chlorophyll *a* was low, ranging from 1.8-5.5 μg/L. However, chlorophyll *a* was only sampled during July, when chlorophyll levels are often quite low. Light and dark bottle measurements also showed that photsynthesis and respiration (P&R) were negligible at all stations sampled. Algal Growth Potential Tests (AGPT) indicated that P was limiting in the Chattahoochee arm and in Flat Creek Bay, and N was limiting in the West Fork of Little River.

Maximum standing crop of algae in AGPT were low, ranging from 0.44-4.2 μg/L, which also indicate lower trophic index. NUTX rates for sediments at all stations were negligible for all forms of N and P, which support the low productivity indicated by chlorophyll sampling, AGPT, and light/dark bottle experiments. All of these results support the general conclusion of the study that productivity is low and the reservoir is oligo-mesotrophic. These results are in agreement with the low TSI_{Chl}, that occurred periodically in Lake Lanier, as shown in Table 3.2. The results of the SOD experiments, however, indicate more eutrophic conditions.

SOD ranged from -1.8561 in Flat Creek to -1.1000 g-O_2/m^2/day in Flowery Branch, values that are more in agreement with the RAHD (Table 3.3), percent difference from saturation (Figure 3.2) and consumption/production of CO_2 (Table 3.4) reported here.

The de-coupling of Secchi depth, chlorophyll *a* and phosphorus in reservoirs is usually explained by non-algal turbidity, which decreases secchi depth, but also decreases light availability for algae. The disparity between chlorophyll *a* and phosphorus is more difficult to explain, although it is probably due in part to the difficulty in measuring low levels of phosphorus. That explains mesotrophic levels of P, but not eutrophic levels. Sporadically, P reaches very high levels in Lanier, resulting in elevated annual average P values. This could be due to release from anoxic sediments or to alkaline desorption, or to large, intermittent, external loading events.

It is of interest that metabolic rate in the water column and in the sediments is also not coupled. Organic matter that is metabolized in the sediments must have been present in the water column at some time, but there is little evidence of autochthonous production, or of allocthonous transport, in the overlying water.

3.5 Conclusions

Calculated trophic state in Lake Lanier, as is often the case in reservoirs, varies with the parameter used to compute it. TSI based on transparency indicates upper mesotrophy, and a decreasing trend in trophic state over time,and distance downstream, which may be due to reduced turbidity. TSI based on chlorophyll *a* is lower than that based on transparency, but shows a concomitant increasing downstream trend. This trend in TSI_{Chl} may be the result of increasing transparency, but may also be due to increasing phosphorus. TSI_{TP} data presented here is not extensive enough to determine if trophic state based on phosphorus is increasing over time, although it appears that phosphorus is quite variable, and sporadically quite high. Phosphorus data described in Chapter 2 is evidence that epilimnetic and hypolimnetic levels of phosphorus are increasing in Lanier.

Metabolic parameters do indicate higher productivity, although there is variability and some inconsistency. The most notable inconsistency is the very low productivity and respiration in the light/dark bottle experiments conducted by the USEPA (2008), compared with the much higher productivity indicated by the other metabolic parameters,RAHD, percent difference from saturation, CO_2 production/consumption, and SOD, that are reported here. It is apparent that organic material is metabolized in the sediment, resulting in hypolimnetic oxygen deficit and CO_2 accumulation. The low P&R in the water column, as shown in the light/dark bottle experiment, does not explain the CO_2 deficit, indicative of net autotrophy during the growing season. It is proposed that epilimnetic production is higher that indicated by measures of algal biomass or light/dark experiments because algae are bound to clay particles and settled out of the photic zone into the sediments, as well as into the metalimnetic plate during stratified periods.

Table 3.1: Trophic status for the Carlson's trophic state index (TSI) and corresponding ranges for parameters used to calculate the index (secchi depth, SD; total phosphorus, TP; chlorophyll *a*, Cha), and the scale for a trophic state index based on areal hypolimnetic oxygen deficit (AHOD).

Trophic Status	TSI	SD (m)	TP (μg/L)	Cha (μg/L)	AHOD (mg/cm^2/d)
oligotrophic	20	16	3.0	0.34	0.0125
mesotrophic	40	4	12	2.6	0.0255
eutrophic	60	2.7	48	20	0.0800
meso-eutrophic	80	0.25	192	154	0.1750
hyper-eutrophic	100	0.06	768	1183	0.4500

Table 3.2: Trophic state indices for chorophyll *a* (TSI_{Chl}), Secchi depth (TSI_{SD}) and total phosphorus (TSI_{TP}) at Thompson Bridge in the Chattahoochee arm (CHA), Bolling Bridge in the Chestatee arm (CHE), Flat Creek Bay (FC), Browns Bridge (BB), lake off Flowery Bay (FBL) and Buford Dam (DAM) from 1966-2001. (nd = no data)

Date	CHA			CHE			FC		
	TSI_{Chl}	TSI_{SD}	TSI_{TP}	TSI_{Chl}	TSI_{SD}	TSI_{TP}	TSI_{Chl}	TSI_{SD}	TSI_{TP}
1966	37.4	53.92	nd	31.7	52.03	nd	nd	nd	nd
1978	nd	nd	nd	22.7	nd	nd	29.8	nd	nd
1979	nd	nd	nd	41.6	nd	nd	27.1	nd	nd
1987	49.2	51.4	nd	43.1	48.5	nd	52.7	53.9	nd
1988	40.7	53.7	nd	32.4	51.4	nd	40.7	52.4	nd
1989	44.4	53.6	nd	40.7	51.8	nd	45.1	62.9	nd
1991	36.9	53.1	56.1	37.4	47.7	51.0	39.9	50.3	50.8
2001	53.2	50.0	43.9	46.7	49.5	41.0	51.1	49.8	46.3

Date	BB			FBL			DAM		
	TSI_{Chl}	TSI_{SD}	TSI_{TP}	TSI_{Chl}	TSI_{SD}	TSI_{TP}	TSI_{Chl}	TSI_{SD}	TSI_{TP}
1966	32.8	51.3	nd	31.3	47.32	nd	32.4	45.8	nd
1978	28.5	nd	73.2	24.3	nd	nd	29.1	nd	68.6
1979	34.5	nd	53.9	33.4	nd	nd	29.0	nd	53.2
1987	43.4	45.9	nd	43.7	42.4	nd	37.4	45.8	nd
1988	33.1	48.4	nd	35.2	43.3	nd	33.9	46.0	nd
1989	44.9	45.2	nd	42.0	43.2	nd	42.0	44.3	nd
1991	32.4	43.7	51.0	30.6	40.5	53.7	28.4	40.5	58.3
2001	45.5	46.0	37.4	46.4	40.5	37.4	45.8	43.6	37.4

Table 3.3: Relative areal hypolimnetic oxygen deficit (R_{AHD}) (mg-O_2/cm^2/d) and trophic state index (TSI) based on R_{AHD} at Browns Bridge (BB), Flat Creek Bay (FC), lake off Flowery Branch (FBL) and Flowery Branch Bay (FBB) from 1966-2001; annual inflow (km^3) to Lake Lanier also shown. (nd = no data)

Year	Inflow	BB		FBL		FC		FBB	
	(km^3)	R_{AHD}	TSI_{AHD}	R_{AHD}	TSI_{AHD}	R_{AHD}	TSI_{AHD}	R_{AHD}	TSI_{AHD}
1966	1.75	0.038	46.0	0.042	48.2	nd	nd	nd	nd
1988	0.72	0.057	54.7	0.034	43.6	0.056	54.5	0.028	39.3
1989	1,51	0.048	51.2	0.033	43.1	0.066	57.9	nd	nd
1990	2.47	0.058	55.2	0.045	49.7	0.054	53.5	0.017	29.0
1992	1.76	0.057	54.8	0.044	49.1	nd	nd	0.016	27.2
2001	0.88	0.057	55.0	0.036	44.8	0.073	60.1	0.035	44.6
2002	0.74	0.043	48.8	0.042	48.3	0.040	47.3	0.023	35.4

Table 3.4: pCO_2 (μatm) at Flat Creek Bay (FC), Browns Bridge (BB), channel off Flowery Branch Bay (FBL) and Flowery Branch Bay in 1999-2001.

Date	Epilimnetic pCO_2				Hypolimnetic pCO_2			
	FC	BB	FBL	FB	FC	BB	FBL	FB
Dec-1999	1423	nd	nd	nd	6348	nd	nd	nd
Apr-2001	284	335	nd	nd	3797	3210	nd	nd
Aug-2001	237	237	250	nd	7913	7913	nd	nd
Dec-2001	435	250	630	587	4281	4425	5217	4589

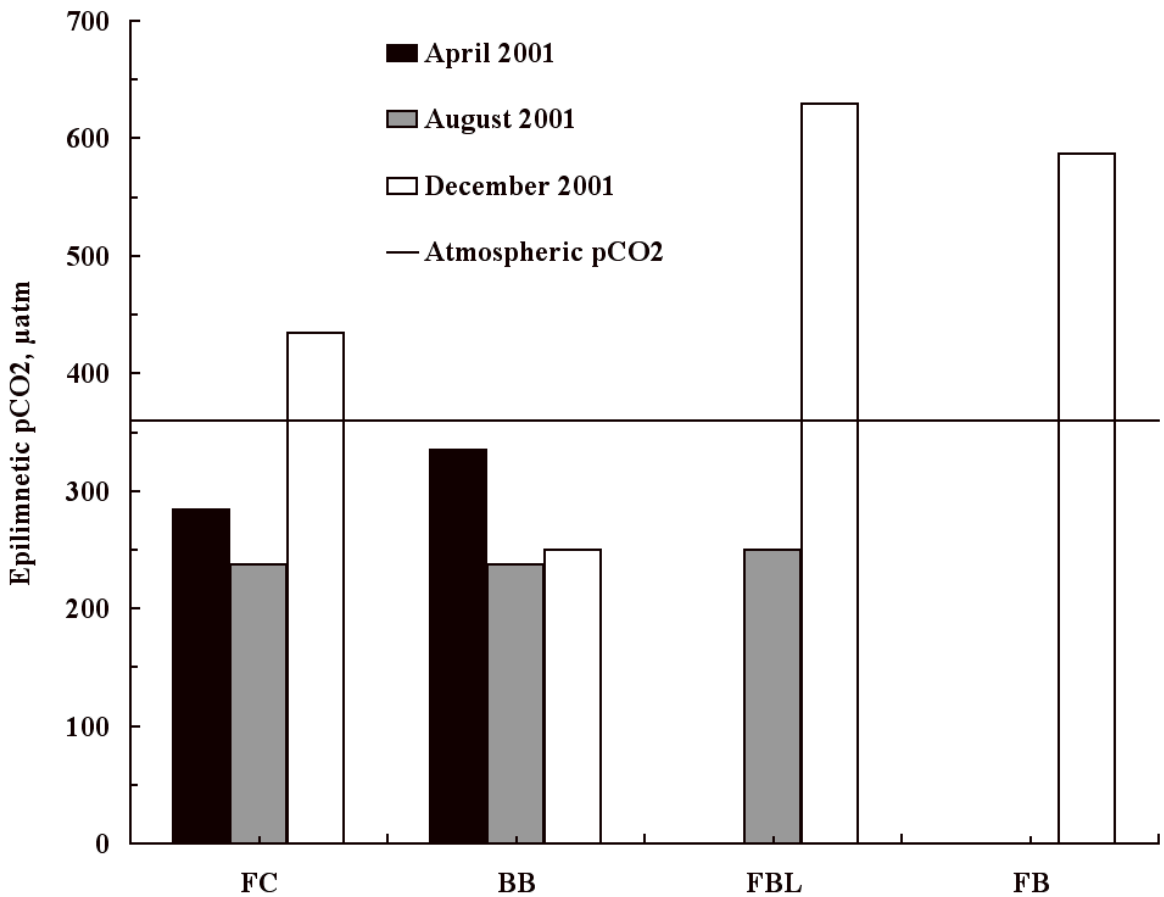


Figure 3.1: Epilimnetic pCO_2 at Flat Creek Bay (FC), Browns Bridge (BB), Flowery Branch Lake (FBL) and Flowery Branch Bay (FB) in 2001-2002.

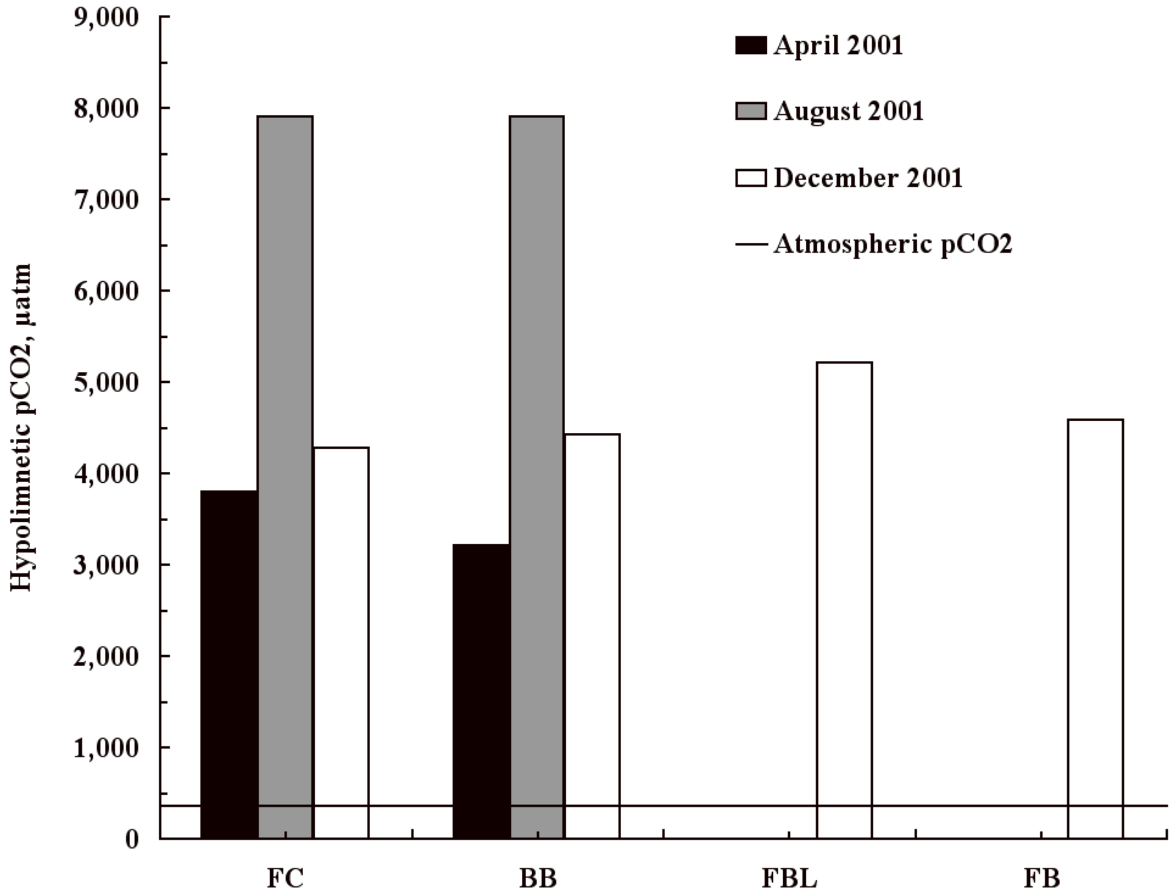


Figure 3.2: Hypolimnetic pCO_2 at Flat Creek Bay (FC), Browns Bridge (BB), Flowery Branch Lake (FBL) and Flowery Branch Bay (FB) in 2001-2002.

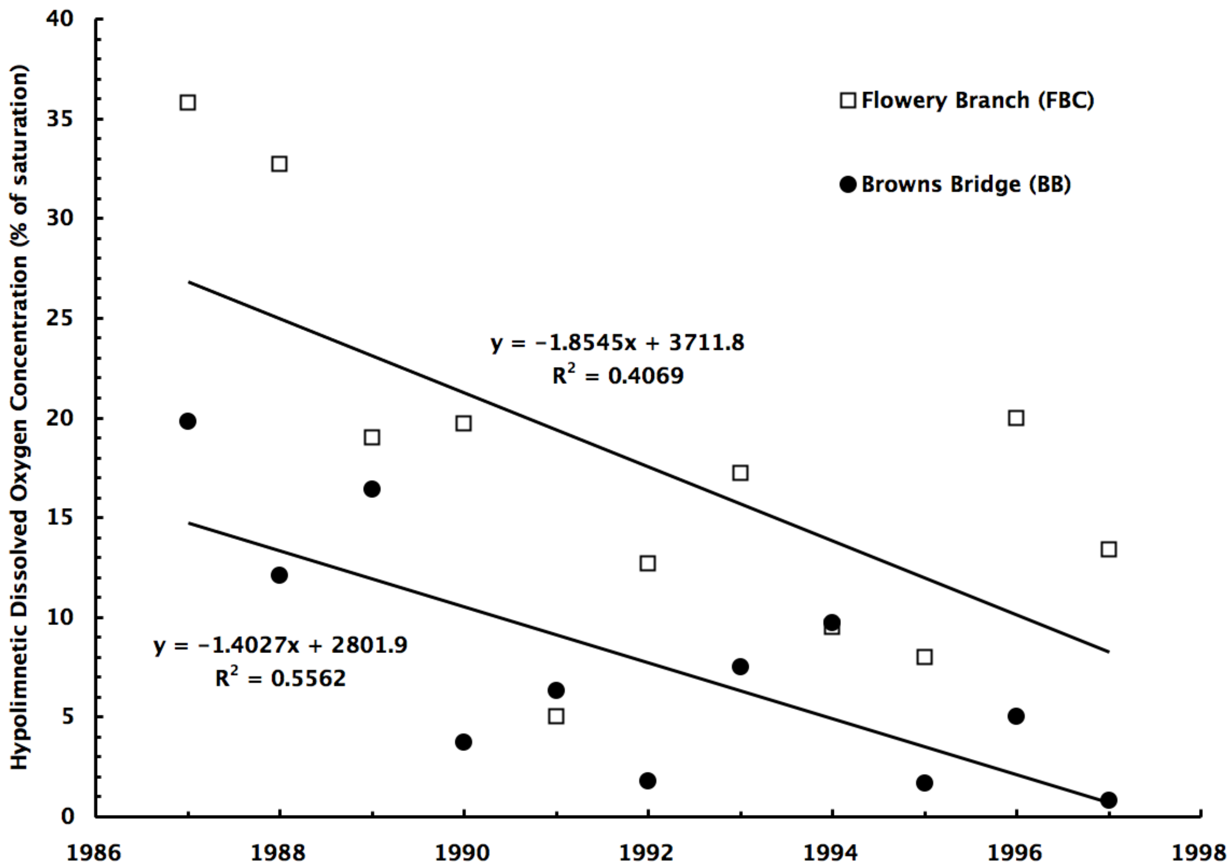


Figure 3.3: Hypolimnetic dissolved oxgyen at Browns Bridge (BB) and Flowery Branch Lake (FBL) Stations, 1987-1997.

Chapter 4

The Role of Phosphorus in Lake Lanier Productivity and Metabolism

Phosphorus is often the limiting factor in freshwater systems, and its role in eutrophication has been the subject of extensive study. Control of external phosphorus loading has, in many cases, produced dramatic improvement in trophic state and water quality, with little or no residual internal phosphorus loading (L.Washington, Jensen, et al, 1992, 2006; Nurnberg, 1984, 1986). Such improvements are presumably due to the reduction in algal growth and concomitantly higher hypolimnetic oxygen levels, which increase the binding capacity of benthic sediments for phosphorus, due to the increase in ferric iron species. However, there have been notable exceptions to the rule that oxic sediments bind, and anoxic sediments release, phosphorus (Caraco, 1993; Gachter and Muller, 2003).

Although ferric iron undoubtedly is the key player in the oxic binding and anoxic release of phosphorus, there are many variables that affect this process (Mortimer, 1941, 1971). These variables include the ratio of iron to phosphorus in sediments (Bortleson, 1974; Jensen, 1992, 1996); continuous supply of ferric iron to anoxic hypolimnia, which bind phosphorus released from anoxic sediments (Parker, 2004);. high levels of calcium, which compete with phosphorus for sorption sites on ferric hydroxides, (Stauffer, 1985); binding of ferrous iron by sulfides, which prevents re-oxidation to ferric iron (Caraco, et al, 1993; Gachter and Muller, 2003); nitrogen, rather than phosphorus, limitation (Cullen and Forsberg, 1988), oxic mineralization of organic matter to release phosphorus (Caraco, et al, 1993), oxic uptake/anoxic release of phosphorus by bacteria (Gachter and Meyer, 1993; Hupfer, et al, 1995), and desorption of phosphorus from ferric iron at pH above 8.5 (Lijklema, 1980; Parker, 2004).

Lake Lanier is one of those notable exceptions. The phosphorus load to Lake Lanier was estimated to be 95,000 kg/yr in 1973, 55,677 kg/yr in 1991, and 31,259 kg/yr in 1996, a loading of about 0.04 g/m^3/yr in 1973 and about 0.01 g/m^3/yr (0.4 lb/acre-ft/yr) in 1996, of which only 10 percent is estimated to leave the reservoir through the dam (Hatcher, 1998). Algal growth potential tests, which indicate that phosphorus is the limiting factor (ESEI, 1981, Hatcher, 1998; USEPA, 2007), and low chlorophyll *a* and epilimnetic SRP, support the hypothesis that much of the phosphorus in the remaining 90 percent, about 85,000 kg/yr based on 1973 loading and about 28,000 kg/yr based on 1996 loading, is bound and settled in sediments each year. Should conditions develop in the reservoir that favor the release of benthic phosphorus, internal loading could be quite high.

Lanier is a soft-water system low in calcium and sulfates and high in iron, which favors binding and sequestration of phosphorus in sediments. In southeastern Piedmont impoundments such as Lanier this lack of phosphorus release, even though phosphorus loading is high, has been attributed to the high iron levels, which may be present in oxidized form above the sediments, so that, even if phosphorus is released, it is quickly resorbed and precipitated (Reckhow, 1988; Lind, 1993; Hatcher, 1994; Parker, 2004).

This would explain the moderate phosphorus and chlorophyll levels in Lanier, even though phosphorus loading is high. The result should be an oligotrophic or meso-oligotrophic system, with low chlorophyll *a* and SRP levels, as is generally the case in the reservoir. However, while Laniers trophic state is variable, often oligotrophic-mesotrophic based on chlorophyll levels, and often mesotrophic based on phosphorus levels and secchi depth, hyplimnetic oxygen depletion ranks it as eutrophic (chapter 2, Mayhew and Mayhew, 1991).

Such oxygen depletion in reservoirs is frequently attributed to subsidies of organic material from the watershed, and not to autochthonous production. Enough organic material, allocthonous and/or autochthonous, is available in Lanier to power an extensive community, including a very productive fishery Some component of the organic material in the reservoir must be supplied by the watershed. Although DOC/DOM and POC/POM are low in tributaries (Zeng and Rasmussen, 2005), bacterial loading is periodically quite high, and organic matter may enter in increments during high flow or in subsurface density currents, that are not measured. It is also quite likely that significant amounts of organic material are produced autochthonously, using the phosphorus that arrives, primarily bound to sediments, from the watershed.

The phosphate buffer (Froelich, 1988) is a mechanism that causes release of phosphorus from sediments when tributaries enter receiving bodies, making it sporadically available to algae. This may be related to alkaline release at pH above 8.5. Some algae may utilize phosphorus that is attached to suspended sediments (Froelich, 1988; Burkholder, 1992), and there is evidence that re-suspended sediments in another Piedmont impoundment, Lake Allatoona, increase algal biomass (Ceballos and Rasmussen, 2007). The increase in availability of phosphorus from re-suspended sediments in Lake Allatoona may be linked to another mechanism of phosphorus release, desorption from oxic, epilimnetic bottom or suspended sediments at elevated pH (> 8.0). Photosynthesis in soft-water systems raises pH. During daylight hours in spring and summer, pH often rises above 8.0 in Lake Allatoona and in Lake Lanier.

The possibility of alkaline release has been demonstrated by chemical models (MINTEQ, 1991) and laboratory experiments (Likjelma, 1980, Parker, 2004), but verification of this mechanism in the field is difficult. Chemical models indicate that alkaline release occurs, but not the rate at which it occurs. Adsorption-desorption experiments are generally done over periods of 24 hours or more, while the increases in pH due to photosynthesis may occur over minutes to a few hours.

Another possible explanation for low internal loading of phosphorus in Lanier is that the amount of phosphorus in the sediments is not as high as has been assumed. Ninety percent of the phosphorus load to Lanier remains in the reservoir, and it is reasonable to assume that much of that phosphorus reaches bottom sediments. However, if a significant portion of the phosphorus in benthic sediments is organic, release of that component by mineralization will be more moderate, and there will be less inorganic phosphorus bound to iron.

Additionally, if inorganic phosphorus levels are high in sediments, it is possible that conditions may develop in the future that could result in extensive internal loading. However, low inorganic phosphorus levels in benthic sediments could explain the lack of internal loading. The question then arises: If the sediment phosphorus levels are low, what is happening to the 90 percent of the load that remains in the reservoir?

It is possible that more of the phosphorus is incorporated into the food web by autochthonous production than is apparent from the low algal biomass, that algae may bind to sediments and utilize attached phosphorus, and that algal biomass is underestimated due to more rapid settling of algal-clay complexes. These flocs could collect at the metalimnion, contributing to the upper metalimnetic oxygen maximum, and the lower metalimnetic minimum, as they continue to settle and receive less light. Some algae are mixotrophs, and may concentrate at the metalimnetic plate, where bacterial food is available to supplement photosynthesis (Bird and Kalff, 1989). The microbial loop may also recycle some nutrients from the clay-algae complexes and make them available in the mixing zone.

The hypotheses that phosphorus levels in the sediments are low, data on an analysis of sediments for organic and inorganic P are presented. The hypothesis that phosphorus may be available due to alkaline desorption at high pH is tested by examining the results of field experiments on the effects of pH on phosphorus sorption by clay sediments, and by examining data on seasonal pH, dissolved oxygen, and redox. These seasonal data are also used to examine the hypothesis that phosphorus is released by reduction of iron under anoxic conditions.

4.1 Methods

4.1.1 Sediment Phosphorus

Sediment samples were collected in the summer of 2007 and analyzed for inorganic, organic and total phosphorus. Samples were collected at the four stations studied in 2001-2002, Flat Creek Bay (FC), Browns Bridge (BB), Flowery Branch Bay (BB), and the main lake off

Flowery Branch (FBL). Littoral samples at 1.5 and 2.0 m, and pelagic samples at 25 and 19 m, were collected at FC and FB; pelagic samples only, at 34 m, were collected at BB and FBL. Sediment samples were collected with a ponar grab sampler and stored on ice until return to the laboratory, where they were refrigerated.

The procedures for analysis are those described by Murdoch, et al (1996). Three replicates of each sample were dried at 60^{o}C for 24 hours, ground in a mortar and pestle, and stored in a dessicator. 0.5 g of sediment and 50 mL of 1.0 N HCl were placed in 125 plastic jars and shaken for 24 hours. Approximately 15 mL of sample were centrifuged for 5 minutes at high power (2000 rpm). The supernatant was diuted to 1:10 and analyzed for SRP with a Hach DRS 890 colorimeter and PhosVer method 79 (ascorbic acid-molybdate blue). The MDL is 0.01 mg/L.

4.1.2 Pond Enclosure Experiments

The effect of pH on phosphorus sorption to sediments was tested by adding soil-phosphorus solutions to enclosures of epilimnetic water and sediments in a Piedmont pond in June 2006. The pond is a small, man-made impoundment on the campus of Gainesville State College, located in Hall County, Georgia (Figure 4.1). Each enclosure was constructed by removing the bottom of a plastic bucket, ranging in diameter from 20 to 22 cm. The plastic cylinders were pushed into shallow sediments, so that water depth ranged from 8 to 15 cm inside the mesocosms, and volume ranged from 2.8 to 4.0 L. pH was measured by suspending a YSI 660 sonde inside a cylinder from a ring stand, and moving the ring stand and sonde among the mesocosms.

Two experiments, one of 135 min and the other of 240 min duration, were conducted. Soil-P slurries were made 18 - 24 hours before an experiment by mixing 20 mL of standard $K_2H_2PO_4$ solution (1 mL = 0.1 mg P/L) with 5.0 g of Bt horizon soil. The soil, which had been collected in Habersham County, GA, was air dried and sieved through a #20 (840 μm) mesh sieve. Control soil-H_2O slurries were made by adding 20 mL of distilled H_2O to 5.0 g of sieved soil. The pH of the soil-H_2O slurries varied fro 4.40 - 4.48; that of the soil-P slurries ranged from 4.05 - 4.42. 0.1 N HCl was added to reach a pH of 4.3 - 4.7 in acid enclosures and 0.1 N $NaOH$ was added to reach a pH of 9.3 - 10.0 in alkaline enclosures.

Controls were left at ambient pH. Soluble reactive phosphorus (SRP) was measured using a Hach DRS 890 colorimeter and PhosVer method 79 (ascorbic acid-molybdate blue). SRP was measured 5 min after addition of soil-P or soil-H_2O slurry, and at 30 min or 60-75 min increments thereafter. Results were corrected for the volume of water in the enclosures.

4.1.3 Seasonal data on pH, Dissolved Oxygen and Oxidation-Reduction Potential

Data from hydrolabs deployed during 2001-2002, as described in Chapter 2, are examined. Data for two of the stations located in the main channel, Browns Bridge and Flowery Branch lake, are presented here.

4.2 Results

4.2.1 Sediment Phosphorus

Phosphorus in the terrestrial soil samples from the watershed of Lake Lanier contained negligible amounts of inorganic phosphorus, less than a 0.5 mg/kg of soil, and ssmall amounts of organic phosphorus, about 5 and 17 mg/kg of soil (Table 4.1). The shallow sediments collected in Flat Creek and Flowery Branch Bays contained negligible amounts of both forms of phosphorus. Pelagic sediments in Flat Creek Bay and in the mixing zone at Browns Bridge had about 30 mg/kg of organic phosphorus and about 50 - 60 mg/kg of inorganic phosphorus.

Inorganic phosphorus was about 70 percent lower, and organic phosphorus about 45 percent lower, in the channel at Flowery Branch Lake than at Browns Bridge. Sediments from Flowery Branch Bay itself were lower in inorganic phosphorus, 88 percent lower than those at Browns Bridge, than the channel, but somewhat higher in organic phosphorus than the channel, about 25 percent lower than at Browns Bridge. The ponds had higher total phosphorus, especially organic phosphorus, with about 54 percent higher in Duck Pond and 150 percent higher in Pine Pond. Shallow sediments in both ponds had less phosphorus of both types that the deeper pond sediments, but more than the shallow lake sediments.

4.2.2 Pond Enclosure Experiments

Results are shown in Tables 4.2 and 4.2 and Figures 4.2 and 4.3. SRP was below detection in the control soil - DH_2O enclosures throughout both trials. pH in these control enclosures varied only slightly, decreasing about 1-3 percent from the initial value of 6.72 (135 min trial), or increasing about 2 percent from the initial value of 6.96 (240-min trial). In the control soil - P enclosure (240 min trial), some phosphorus, 17 percent of the amount added to the soil slurry, was still in solution five minutes after addition, however, SRP was below detection for the rest of the trial. The change in pH in the soil - P control enclosure was larger than in the soil - DH_2O enclosures, increasing 14 percent from an initial pH of 7.19 to 8.22.

Initial pH in the experimental acid soil-P enclosure (135-min trial) was 7.10 and 4.28 5 min after the addition of the soil-P slurry; by the end of the trial, pH had increased 30 percent, to 6.42. The amount of phosphorus remaining in solution 5 min after addition of the soil slurry, 16.5 percent, was similar to that in the control soil - P enclosure. Liskewise, SRP was below detection for the remainder of the trial. Initial pH in the alkaline soil - P enclosure (135 min trial) was 6.85 and 9.30 5 min after addition of the soil slurry. At 5 min, 31 percent of the phosphorus added in the soil remained in solution, and about 10 percent was still in solution by the end of the trial.

Initial pH in the acid - P enclosure (240 min trial) was 7.66; at the 5 min mark pH was 4.10, increasing about 10 percent to 5.35 at the end of the trial. Initial pH in the alkaline enclosure was 7.02, and 9.29 at the 5 min mark, and 8.82, about 5 percent lower. The acid experimental enclosure showed a pattern more similar to the alkaline enclosure in the 240 min trial. At 5 min, 28 percent of the added phosphorus still remained in solution in the alkaline enclosure and 24 percent remained in the acid enclosure, and the amount remaining declined at a similar rate in both enclosures until 240 min, by which time SRP was below detection in the acid enclosure and 9 percent still remained in the alkaline enclosure.

4.2.3 Seasonal Data on pH, Dissolved Oxygen, and Oxidation-Reduction Potential

Figures 4.4, shows summer conditions at Browns Bridge, in the mixing zone, and in the lake off Flowery Branch Bay. Surface pH shows a diel increase due to photosynthesis, often above 7.5 and reaching 8.0 or above during daylight. Mid-depth pH is much more variable, ranging from 7.5 down to 6.5, reflecting periodic high rates of photosynthesis over a period of several days, interspersed with periods of high respiration; after August 15, respiration predominates. Bottom pH is steady at 6.3 or so until August 6, after which it declines to about 5.6, with some periods of much lower pH, down to 5.1 or 2, due to increasing respiration.

Surface DO during this period is steady and shows a similar diel pattern to pH, as does DO at mid-depth; DO in August is very low at mid-depths, often less than 4.o and sometimes less than 2.0 mg/L. DO at the bottom is somewhere between 0.0 and 1.0 mg.L at the end of July, and declines to 0.0 and remains there by the 6 or 7th of August. ORP at mid-depth remains hight, aroud 600 mv; at mid-depth, ORP is only slightly lower than the surface during the last week of July; however, it is variable during the firest two weeks of august, decreasing below 400 and sometimes below 300 m, almost to 200 mv, which is the ORP at which reduction of ferric iron may occur.

Conditions in the lake off Flowery Branch Bay, later in summer (Sep-Oct), are more moderate; surface pH is often above 7.5, but rarely above 8.0; mid-depth pH is only slightly lower than the surface and mid-depth oxygen levels are often slightly higher. Photosynthesis is occurring at almost as high a rate at mid-depth as at the surface; however, respiration rates are not increasing there, as they did at Browns Bridge. . pH at the bottom is steady at about 6.4, reflecting moderate respiration rates; DO is also higher and more steady than at Browns Bridge, although there is a decline from nearly 3.0 to 2.0 mg/L. ORP at mid-depth is similar to that at Browns Bridge, but much higher at the bottom, showing a decline over time from levels as high as the mid-depth, around 550, to about 525, with some periodic decreseds to 500 or less. However, ORP has not approached the 200 mv mark, when internal loading could occur.

Condition at Browns Bridge and the lake off flowery Branch Bay in late fall/winter are shown in Figure 4.5. pH, oxygen and bottom ORP show that mixis is occurring around the middle of December, a big spike in ORP, and a smaller one in DO occurring around the 10^th. Before that, ORP was well below 200,mv, 100 mv or less, and returned to that level between mixing periods.

Conditions off Flowery branch are more uniform later in December and the beginning of January. Mixis is complete, pH is similar at all depths, as is oxygen, although declining at mid-depth and the bottom by a mg/L. pH and oxygen levels increase during January, perhaps as a result of mixing of nutrients and increased photosynthesis, or due to the recovery from mixis, which mixed the oxygen demanding materials in the hypolimnion into the epilimnion. ORP at mid-depth is still slightly depressed, less than 500 mv, at the end of December, for similar reasons, but is increased to 550 or so during early January. ORP at the bottom is lower than in the summer,and fairly steady, at 300 or a little higher.

Figure 4.6 shows the conditions at Browns Bridge in winter, February. Data were not collected at lake off Flowery Branch during February or March. pH is above 7.0 at all depths during mixis, stratification has not yet isolated the hypolimnion from the epilimnion. pH is higher at the surface and mid-depth, due to photosynthesis. It is above 7.5 , but does not reach 8.0, and is slightly lower at mid-depth. DO is also fairly high at all depths, although DO at middepth matches or surpasses that at the surface for much of the period from feb 10 to 18.DP at the bottom is in the range of the mid-depth and surface for the fest few days, then begiss to decrease somewhat, although still above 8.0 mg/L. ORP is high at the surface, 500 or higher, however, it is less than 400 at the bottom, and remains at that level. It may be that sediment oxygen demand is depleting oxygen at the bottom even during mixis.

4.3 Discussion and Conclusions

The range of concentration of phosphorus in natural lake sediments is large, and the relationship between sediment concentration and release into the water column is not direct. A variety of factors may affect the degree to which phosphorus is released from sediments that are high in phosphorus; however, if phosphorus levels are relatively low, then release of phosphorus, even under anoxic conditions, should be low, since phosphorus levels in pore waters of sediment may be quite high before diffusion into the overlying water occurs.

Levels of phosphorus in sediments include 600 - 2800 mg/kg for shallow, eutrophic lakes in China (Wang, et al, 2008; Li, et al, 2006), 1,200 - 3,000 mg/kg for Lake Memphremagog in Vermont (Carignan and Flett, 1981), 1,240 -10,100 mg/kg for six lakes in Wisconsin ranging from oligotrophic to mesotrophic (Bortleson, 1974), and 700 - 1,000 mg/kg for Lake Winnipeg in Canada (Mayer, et al., 2006).

Benthic sediment from another Piedmont reservoir in Georgia, Lake Allatoona, contained about 350 mg/kg of phosphorus (Rasmussen and Cebellos, 2009). Total phosphorus load to Allatoona has been estimated to be about 53,000 kg/yr in the Clean Lakes Study and to be about 90, 540 kg/yr using USGS protocol (Rasmussen and Cebellos, 2009), which gives a loading of about 0.1 g/m^3/yr for the lower estimate and 0.2 g/m^3/yr for the higher estimate.

Allatoona has a higher phosphorus load than Lanier, and has a higher phosphorus concentration in sediments. The re-suspension of sediments in Allatoona has contributed to internal loading in that reservoir. Lanier is a larger, deeper reservoir, however, resuspension of sediments may also contribute to internal loading in Lanier, not by the traditional method of anoxic release, but by aerobic release at elevated pH.

The results of the pond enclosure experiments indicate that elevated pH may increase desorption, and/or slow sorption, of phosphorus by sediments, which may be a source of internal loading in soft water systems where pH frequently rises above 8. The sorption/desorption of phosphorus is complicated by algal uptake, but the initial pH when sediment carrying P reaches a water body appears to be significant. Even though the pH in the control mesocosm reached 8.22, there was no measurable desorption over the time interval of the experiment.

It is also possible that mineralization of organic material in sediments may produce some phosphorus loading, whether the sediments are oxic or anoxic (Caraco, et al., 1993). Parker (2004) found that adding organic substrates to lake Lanier sediments decreased ORP and resulted in increased release of phosphorus from sediments.

If most of the phosphorus load is sequestered in sediments, the lake should be oligotrophic to meso-oligotrophic, as indeed it has been described, based on chlorophyll *a* levels. A report by the USEPA (2007) concludes that the lake is oligotrophic to oligo-mesotrophic based on AGPT, estimates of productivity based on light-dark bottle experiments, and sediment nutrient exchange rates. Table 4.4 shows a summary of the chlorophyll *a*, nutrient exchange, and sediment oxygen demand from the USEPA (2007) study.

There was negligible exchange of nitrogen and phosphorus from Lanier sediments at the five locations in July 2007. Total and dissolved phosphorus were generally below detection. Traces of ammonia, average of 0.04 $g/m^2/day$, and of total kjeldahl nitrogen (TKN), average of 0.10 $g/m^2/day$, and a trace of total phosphorus, were released from sediments in Little River. AGPT showed that phosphorus was limiting at Hwy 53 in the Chestatee arm and in Flat Creek bay, but nitrogen was limiting in the west fork of the Little River. Standing crop of chlorophyll *a* in the AGPT indicate that the reservoir is oligotrophic to meso-oligotrophic (EPA, 2007). Gross primary production was low at all three stations sampled.

Respiration exceeded photosynthesis in Little River, indicating net heterotrophy; photosynthesis exceeded respiration at Hwy 53 in the Chestatee, indicating net autotrophy. The results in Flat Creek Bay were anomalous: there was some DO increase in one of the dark bottles, which gave a negative respiration, and a P/R ration of - 0.91. However, SOD was measureable and in the range of eutrophic lake sediments.

Table 4.1: Phosphorus concentrations in Lake Lanier sediments, Duck and Pine Pond sediments, and terrestrial soil.

		Depth	Phosphorus (mg/kg)		
Location		(m)	Inorganic	Organic	Total
Lake	Browns Bridge	34.0	58.3	29.7	88.0
Embayment	Flat Creek	2.0	1.0	0.0	1.0
Embayment	Flat Creek	25.0	52.3	32.3	84.7
Embayment	Flowery Branch	1.5	0.0	0.0	0.0
Embayment	Flowery Branch	19.0	6.3	23.7	30.0
Channel	Flowery Branch	34.0	17.0	16.7	33.7
Pond	Duck Pond	0.2	11.7	12.0	23.7
Pond	Duck Pond	2.0	56.3	45.7	102.0
Pond	Pine Pond	0.2	21.0	49.3	70.3
Pond	Pine Pond	1.5	17.0	82.3	99.3
Soil	Hall County	-	0.0	16.7	16.7
Soil	Habersham County	-	0.3	4.7	5.0

Table 4.2: Phosphorus (P) remaining in solution (mg/kg and percent), after addition of acidic and alkaline soil-P slurry to pond enclosures: 135 min trial.

Time	Control Soil-DH_2O		Acid Soil		Alkaline Soil	
(min)	pH	P	pH	P	pH	P
0	6.72	bd	7.10	bd	6.85	bd
5	6.64	bd	4.28	66.0 (16.5%)	9.30	123.2 (31.0%)
60	6.46	bd	5.48	bd	9.00	50.4 (12.5%)
135	6.66	bd	6.42	bd	8.75	39.2 (9.8%)

Table 4.3: Phosphorus, as μg/g and percent remaining in solution, after addition of acidic and alkaline soil-P slurry to pond mesocosms: 240 min trial.

Time	Control Soil-Water		Control Soil-P			Acid Soil-P			Alkaline Soil-P		
(min)	pH	P	pH	P		pH	P		pH	P	
0	6.96	bd	7.19	bd		7.66	bd		7.02	bd	
5	6.92	bd	7.16	68.0	(17.0%)	4.10	95.6	(24%)	9.29	110.0	(28%)
30			8.35	bd							
60	6.82	bd	8.22	bd		4.90	60.0	(14.5%)	8.91	69.6	(17.5%)
120	7.10	bd				5.04	48.0	(12%)	8.85	63.8	(16%)
180						5.13	42.0	(10.5%)	8.80	46.4	(11.5%)
240						5.35	bd		8.82	34.8	(9%)

Table 4.4: Summary of sediment oxygen demand (SOD) (O_2 g/m^2/day), sediment nutrient exchange rates (g/m^2/day) for total phosphorus (TP), dissolved phosphorus DP), ammonium (NH_4), nitrite-nitrate (NO_2-NO_3) and total kjeldahl nitrogen TKN) at the West Fork of the Little River (LTR), Chattahoochee arm at Hwy 53 CHATT), Flat Creek Bay (FC), Chestatee arm at Hwy 53 (CHES), Browns Bridge (BB), Mud Creek Bay (MUD) and Flowery Branch Bay (FB) (USEPD, 2008).

		Nutrient Exchange Rates				
Location	SOD	TP	DP	NH_4	NO_2-NO_3	TKN
LTR	nd	0.01	0.00	0.04	0.00	0.10
CHA	-1.36	0.00	0.00	0.00	0.00	0.02
FC	-1.86	0.00	0.00	0.00	-0.03	-0.01
CHE	-1.36	0.00	0.00	0.00	0.00	-0.03
BB	-1.32	0.00	0.00	0.00	0.00	-0.01
MUD	-1.57	0.00	0.00	0.00	-0.01	0.12
FB	-1.10	0.00	0.00	0.00	-0.01	0.03

Table 4.5: Summary of algal growth potential tests AGPT) as chlorophyll *a* (Chl a) and limiting nutrient (Limit. Nut.), of gross primaryproductivity (GPP) and respiration (R) (O2 g/m^2/day), and the ratio of GPP to R (P/R) at the West Fork of the Little River (LTR), Chattahoochee arm at Hwy 53 (CHA), and Flat Creek Bay (FC) (USEPA, 2008).

	AGPT		Productivity		
Location	Chl *a*	Limiting Nutrient	GPP	R	P/R
LTR	0.44	nitrogen	2.16	11.12	0.19
CHAT	0.50	phosphorus	1.22	0.48	2.53
FC	4.20	phosphorus	1.64	-1.81	-0.91

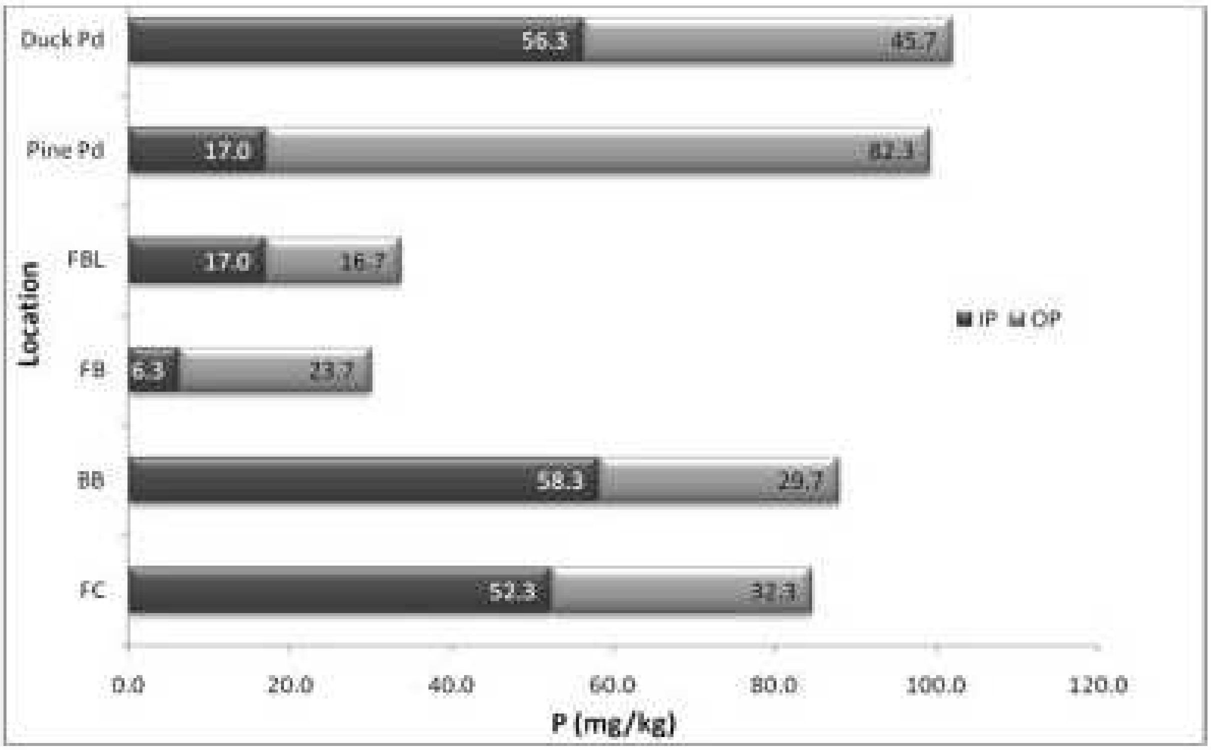


Figure 4.1: Inorganic (IP) and organic phosphorus (OP) concentrations (mg/kg) in pond and Lake Lanier sediments.

FPL = Lake Lanier off Flowery Branch, FB = Flowery Branch embayment,
BB = Browns Bridge, FC = Flat Creek embayment.

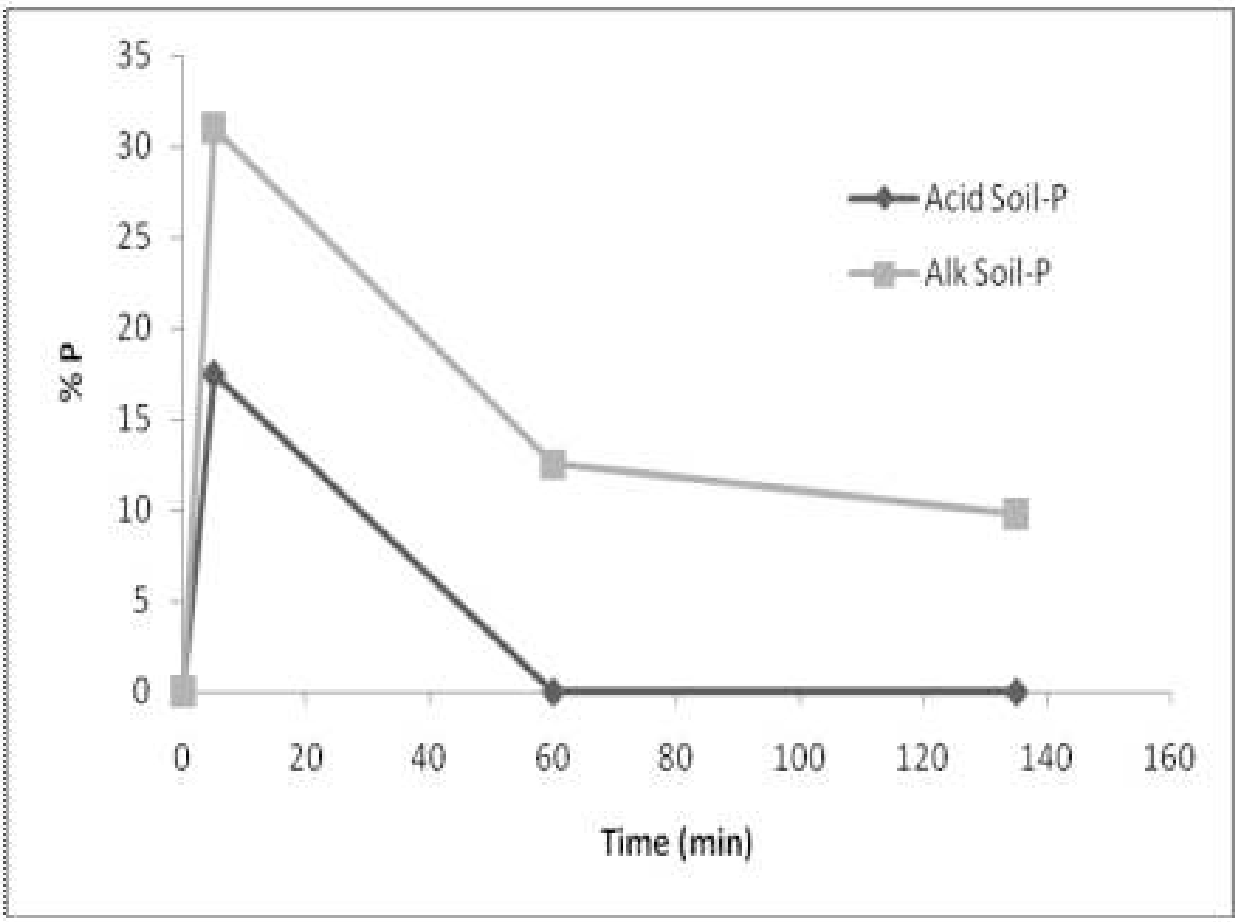


Figure 4.2: Phosphorus (P, mg/kg) remaining in solution after addition of acidic and alkaline soil-P slurry to pond enclosures: 135 min trial.

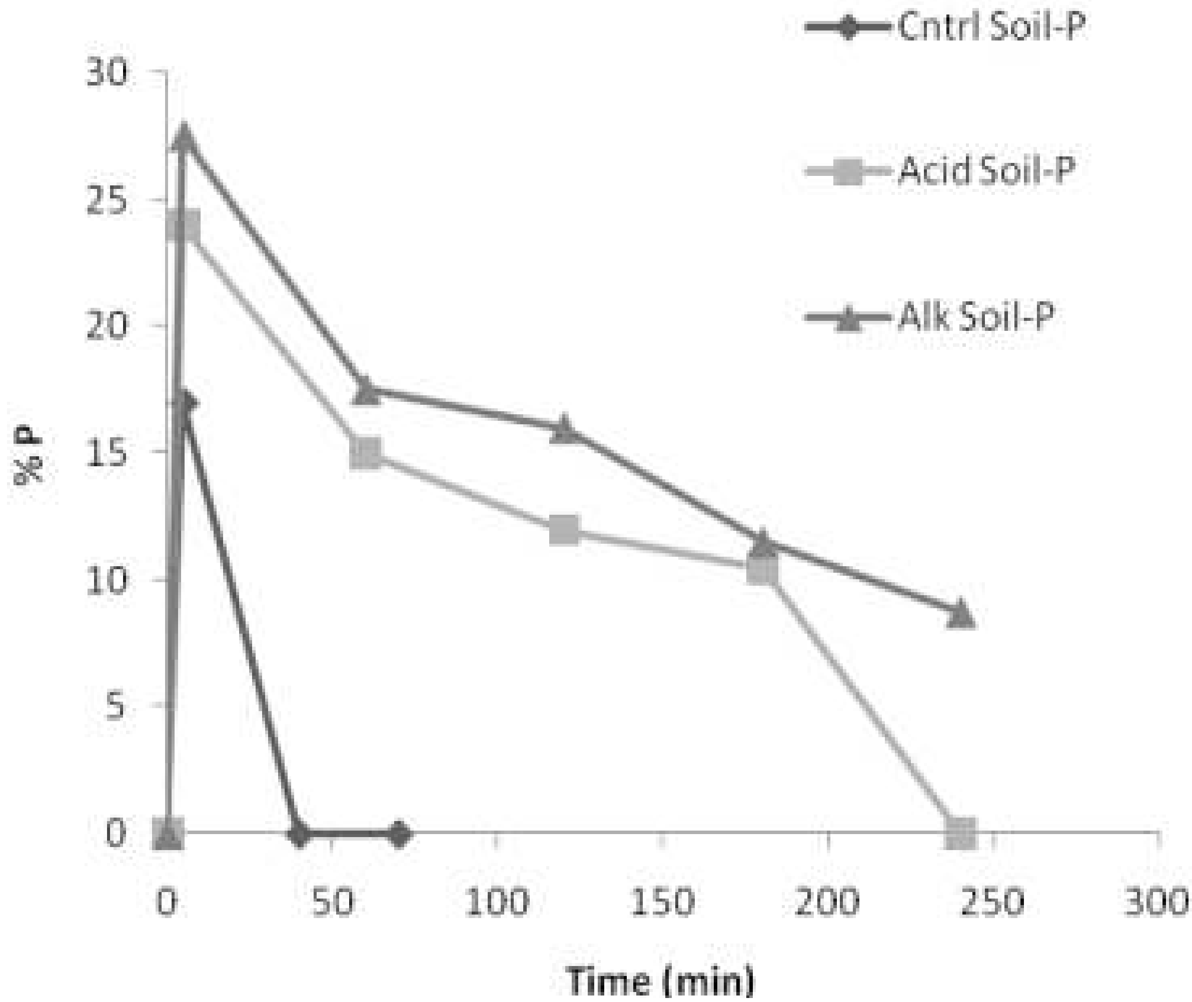


Figure 4.3: Percent phosphorus (P) (μg/g) remaining in solution after addition of acidic and alkaline soil-P slurry to pond enclosures: 240 min trial.

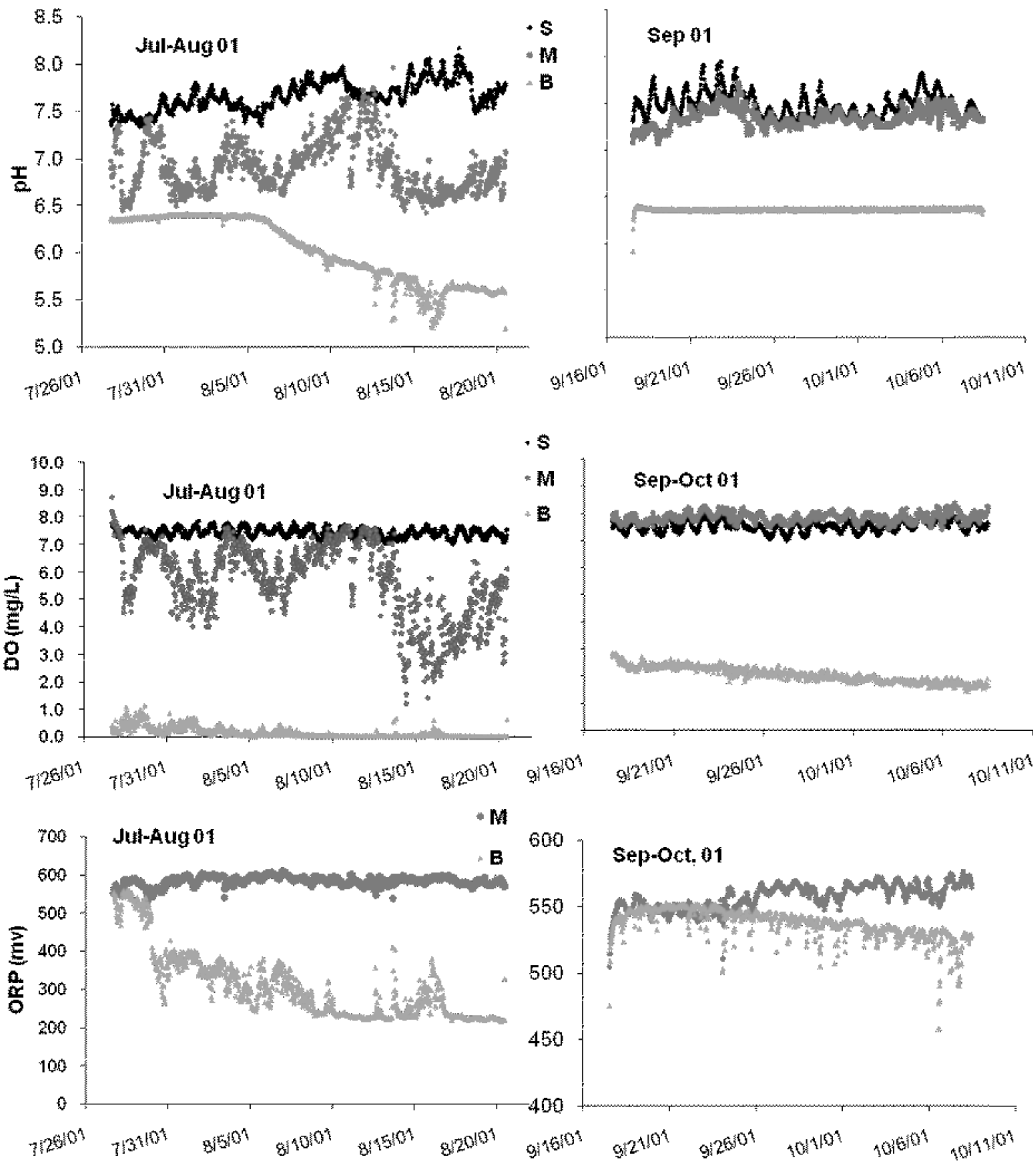


Figure 4.4: pH and dissolved oxygen (DO) at surface (S), middle (M) and bottom (B) depths and oxidation-reduction potential (ORP) at middle (M) and bottom (B) depths at Browns Bridge in July (left) and at Flowery Branch Lake in September 2001 (right).

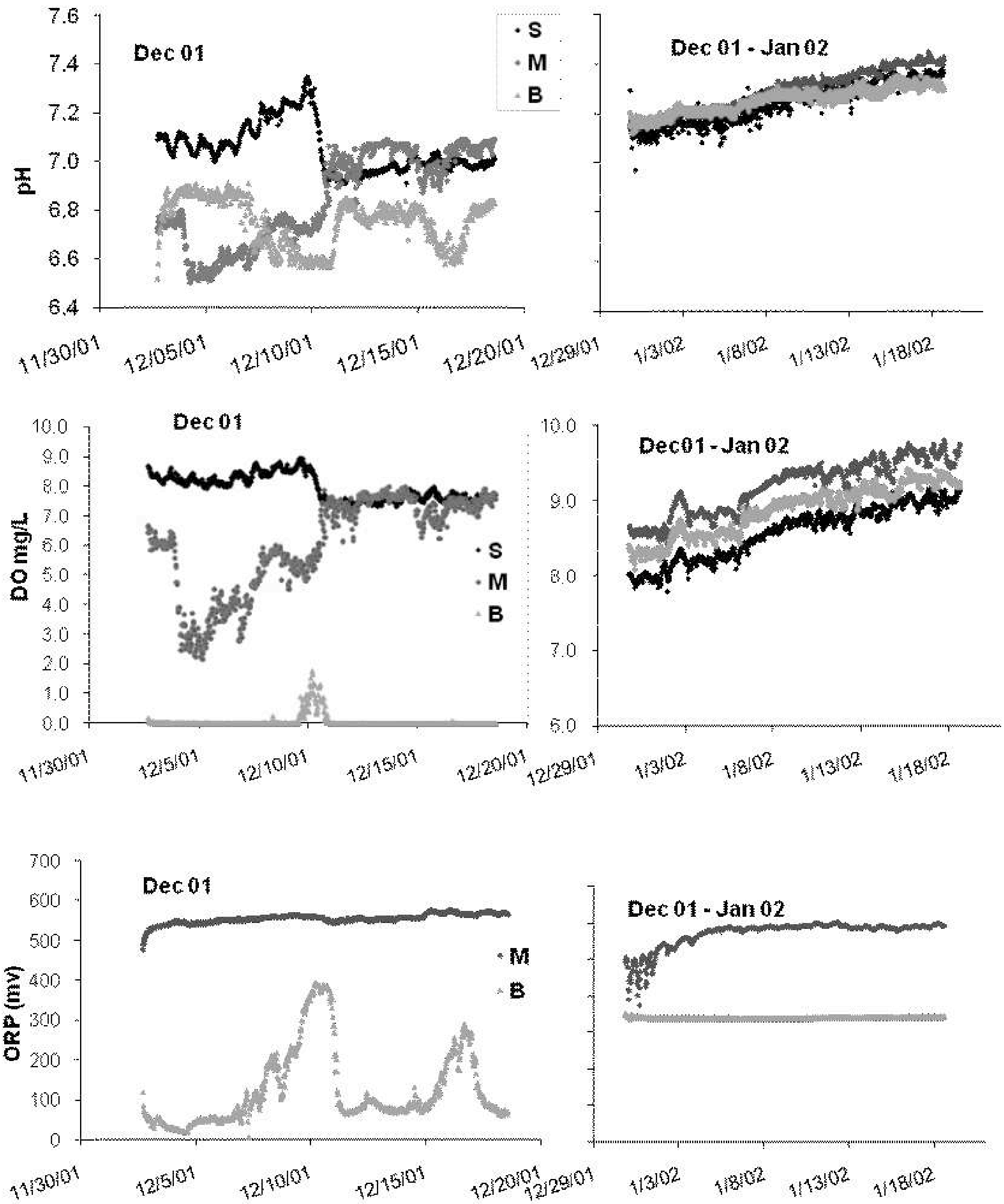


Figure 4.5: pH and dissolved oxygen (DO) at surface (S), middle (M) and bottom (B) depths and oxidation-reduction potential (ORP) at middle (M) and bottom (B) depths at Browns Bridge in December 2001(left) and at Flowery Branch Lake in January 2002 (right).

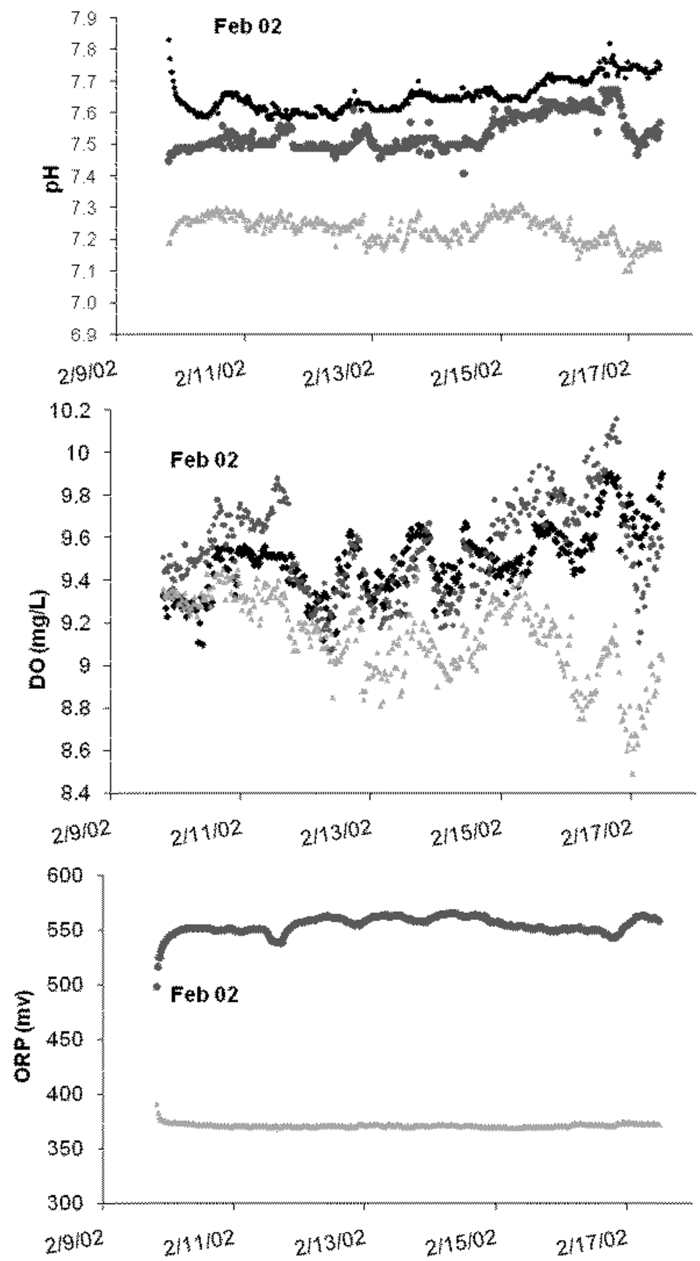


Figure 4.6: pH and dissolved oxygen (DO) at surface (S), middle (M) and bottom depths andoxidation-reduction potential (ORP) at middle (M) and bottom (B) depths at Browns Bridge in February 2001.

Chapter 5

Summary and Conclusions

Anoxic conditions in impoundments are often attributed to allocthonous material brought in by riverine interflows, and not to authochthonous production. However, in some soft-water systems such as Lanier, with high levels of fine, particulate clays with high iron content, internal production may be driven by alkaline release in shallow, euphotic zones and sediments, and by algal release from sediments as they are flocculated and settled (Cuker, 1990; Burkholder, 1992). Clay binds algal cells and settles them by flocculation, especially during periods of high rainfall and runoff .

This data could be interpreted to mean that epilimnetic productivity is low, due to turbidity attenuating light, and that hypolimnetic anoxia is caused by subsidies of organic material from the watershed. However, both total inorganic and organic carbon are low in all regions of the lake, including the tributary arms of the Chattahoochee and Chestatee. While the community in lake Lanier could be based on such a heterotrophic food web, it is very possible that much of the production is autochthonous, based on P released by oxic, alkaline release as tributary water reaches the lake, in a phosphate buffer process (Froelich, 1988), and possibly by biotic processes (Burkholder, 1992), as well.

The accelerated sedimentation of organic matter and mineral nutrients such as P is well documented in aquatic systems. The phenomenon of marine snow has fresh water parallels, and has been used for the purposes of clarifying treated water and reducing algal concentrations and eutrophication in water bodies. Dissolved organics, algae and bacteria have been sedimented from the water column in the laboratory studies (Cuker, et al, 1990; Mayhew and Mayhew, 1992), and field studies have shown that algae bound to the sedimenting clay can release and use P also bound to the clay (Engle and Sarnelle, 1990; Burkholder, 1992).

Zooplankton also feed on aggregates of clay and sorbed organics (Arruda, et al, 1983). Metabolic processes may not clearly distinguish autochthonous from allocthonous sources of productivity; however, it may be possible to determine if material produced in the epilimnion is respired in the epilimnion or metalimnion. On the other hand, it may also be difficult to determine if material being respired in the metalimnion is due to riverine density currents, instead. However, the pattern of metabolism of both dissolved oxygen and carbon dioxide throughout the water column should give a better picture of system metabolism.

In summary, the calculated trophic index of Lanier, as with many reservoirs, varies with the parameter used to compute it. This inconsistency is due to the fact that P loads, primary productivity, and algal biomass (chla) in the epilimnion are not coupled in the reservoir. Phosphorus entering the lake is bound to ferric oxyhydroxides in the clay soil, however, the phosphate buffer (Froelich, 1988); desorption, or delay in sorption, of phosphorus at elevated pH; the ability of algae bound to clay to release phosphorus (Burkholder, 1992); and recycling of phosphorus by the microbial loop in the metalimnion, may release enough nutrients to exert a *middle out* trophic effect. Grazing of settled algae and clay flocculates, plus microbial recycling in the metalimnion, plus alkaline release from sediments, may transfer nutrients and energy rapidly into herbivorous zooplankton, and from there to the planktivorous fish, primarily shad in Lanier, so that an extensive community is maintained.

The evidence for this is in the extensive fishery supported in the lake, the apparently low levels of chlorophyll a in the epilimnion, the evidence of increased metalimnetic metabolic activity, and, especially, the lack of phosphorus in the sediments. The assumption that 90 percent of the phosphorus entering the lake is sequestered in the sediments is not accurate; much of that phosphorus is recycled within the community, specifically by algae and components of the microbial loop in the metalimnion during the stratified period. Phosphorus is not released from the sediments because much of the P that appears to be sequestered and settled into the sediments is not. Instead, it is recycled back into the community by a variety of mechanisms.

As a result, metabolic parameters, dissolved oxygen consumption and carbon dioxide production, provide a more meaningful measure of productivity than classic trophic status, and also provide a more useful tool for mapping historic trends in system productivity than traditional measures of trophic status. The measurement of metabolic parameters may be important in the management of soft water systems such as Lanier, in spite of the apparent low algal biomass and phosphorus levels in the epilimnion. Management by the GAEPD has focused on phosphorus. This research supports the importance of phosphorus, but the lake is not oligotrophic, as indicated by the USEPAs sediment oxygen demand study (2008).

Chapter 6

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