EFFECTS OF INTERNAL LOADING ON ALGAL BIOMASS IN LAKE ALLATOONA, A SOUTHEASTERN PIEDMONT IMPOUNDMENT

by

Elena Louise Ceballos

(Under the direction of Todd C. Rasmussen)

Abstract

Cultural eutrophication of lakes is the accelerated nutrient enrichment resulting in detrimental ecological effects such as algal blooms, lake anoxia and toxic metal release from sediments. Cultural eutrophication is a common occurrence in Piedmont impoundments in Georgia, as well as lakes and impoundments throughout the world. It often results in water unsafe for agricultural use, recreation and drinking.

To reduce the cultural eutrophication of local Piedmont impoundments, recent regulatory controls for nutrients were established as part of the Clean Lakes program and court-ordered total maximum daily loads (TMDLs). These regulatory efforts focus on the reduction and minimization of point-source watershed nutrient inputs, primarily phosphorus, into lake systems, as P is the limiting nutrient in Piedmont impoundments. Thus, reductions in phosphorus loading are expected to improve lake water quality.

However, in the Piedmont, as well as worldwide, many lakes continue to experience algal blooms and lake anoxia after sources of external loading are discontinued. The process of nutrient desorption from sediments, known as internal loading, has been identified to be a source of algal-available P, as well as other nutrients. The conditions under which internal loading takes place are region-specific as they vary based on local physical, chemical and biological conditions.

The purpose of our research was to quantify changes in algal biomass in response to internal loading in Southeast Piedmont impoundments. The results from a mesocosm experiment, physical and chemical sediment analysis, and algal assays were used to characterize algal-availabile phosphorus in Southeastern Piedmont impoundments.

INDEX WORDS: Algal Biomass, Lake Allatoona, Nutrients, Sediments

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B.A., The University of Georgia, 1994

A Thesis Submitted to the Graduate Faculty
of The University of Georgia in Partial Fulfillment
of the
Requirements for the Degree

Master of Science

Athens, Georgia

2007

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May 2007

Dedication

This project is dedicated to Ed Metz, with all due respect.

Acknowledgments

Several people generously contributed their time, ideas, laboratory space and equipment to this project. For this I am forever indebted to: Professor William Miller in the Department of Crop and Soil Sciences, for allowing me to take over his laboratory with physical and chemical analyses and algal assays, for help with physical and chemical assays, and for endless tolerance of discussions of methods and interpretations; Professor Marshall Darley in the Department of Plant Biology, for help with algal assay methods and loan of materials for algal assays; Professor Brian Binder and PhD candidate Christopher Burbage in the Department of Marine Sciences, for suggestions and instruction for algal assays; Professor Jeffrey Dean in the Warnell School of Forestry and Natural Resources, for challenging me to consider alternative theories to explain results and tirelessly editing my thesis; Researchers Paul Montello and Gisele Dean, and Professor Scott Merkle of the Warnell School of Forestry and Natural Resources for help with methods of algal assays; Dr. Gretchen Loeffler-Peltier for help in acquiring materials needed for experiments; Dr. Judy Meyer of the Institute of Ecology for suggestions for alternative methods to mesocosm experiments; PhD candidate Denise Carroll for warning me about the brutality involved in a thesis defense.

Special thanks to Dr. Todd Rasmussen for all of his support: for always making time to help with field work, laboratory work and writing, and especially for giving me the trust and financial support that allowed me to research and develop *my very own* methods.

This project was supported by a grant from the `Georgia Water Resources Institute`, 06HQGR0080, subagreement R7113-G2. Funding for water and sediment analyses and for laboratory experiments were also provided through a contract from `EnviroTrade`. Generous field logistical support was provided by `Blankenship Sand`.

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List of Abbreviations

BS	Base saturation, measure of total base cation (Ca, Mg, etc.), proportion relative to all cations
CE	Cultural cutrophication, accelerated eutrophication from anthropogenic sources
CEC	Cation exchange capacity, a measure of ability of soil to exchange cations, meq/100g
Chl *a*	Chlorophyll *a*, a measure of algal biomass, μg/L
Depth	Depth below water or sediment surface, m
DIP	Dissolved inorganic phosphorus, a measure of biologically available phosphorus, mg/L
DO	dissolved oxygen concentration, mg/L
DO%	Dissolved oxygen concentration, percent
DRP	Dissolved reactive phosphorus, a measure of biologically available phosphorus, mg/L
EC	Electrical conductivity, or Specific Conductance, a measure of total dissolved solids, μS/cm
LBC	Lime buffering capacity, a measure of amount of lime required to raise pH to neutral
NA	Not available
NTA	Nitrilotriacetic Acid
NTU	Nephelometric turbidity units, a water clarity scale
OM	Organic matter, mg/kg
PAAP	Potential algal-available phosphorus
pE	Electron activity, $pE = -\log_{10}[e^-]$
pH	Proton activity, $pH = -\log_{10}[H^+]$
SC	Specific conductance, a measure of the total dissolved solids concentrations, μS/cm
SRP	Soluble reactive phosphorus, a measure of biologically available phosphorus, mg/L
TDS	Total dissolved solids, mg/L

List of Abbreviations (continued)

TEA	Terminal electron acceptor
Temp	Temperature, °C
TKN	Total Kjeldahl nitrogen, a measure of organic nitrogen, mg/L
TON	Total oxidized nitrogen, $TON = NO_3 + NO_2$, mg/L
TRP	Total reactive phosphorus, mg/L
TSS	Total suspended solids, mg/L
TURB	Turbidity, a measure of water clarity, NTU
T-C	Total carbon, mg/kg
T-P	Total phosphorus, mg/kg
T-S	Total sulfur, mg/kg

Chapter 1

Introduction

1.1 Background

Lake eutrophication is a natural phenomenon of chemical, physical and biological change due to sedimentation and nutrient inputs over geologic time scales. Accelerated nutrient enrichment from anthropogenic sources is referred to as *cultural eutrophication*, and is a significant national water quality problem that can lead to adverse ecological effects, such as algal blooms that result in hypolimnetic anoxia (Koski-Vähälä and Hartikainen, 2001; Wetzel, 2001; López-Archilla et al., 2003), leading to fish kills (Kann and Smith, 1999) and toxic metals release from lake sediments (Fang et al., 2005).

The cause of these physical, chemical, and biological changes of lake water quality can be attributed to the introduction of excessive nutrients (primarily carbon, nitrogen and phosphorus) into aquatic systems, resulting in the stimulation of primary production. Lake eutrophication is often manifested by harmful algal blooms. A wide range of factors, including light, temperature, and nutrients, affect algal growth. The warm-temperate climate of the Southeastern Piedmont region, characterized by its extended growing season, provides excellent opportunities for algal growth. This growth is realized when sufficient nutrients are present.

Sources of nutrients include discharges from point sources, such as wastewater treatment facilities, as well as stormwater runoff from nonpoint sources, such as farms and communities. Lake Lanier tributaries dominated by point sources have elevated total dissolved solids and nitrate concentrations, and low sediment concentrations (Zeng and Rasmussen, 2005). Reductions in point-source phosphorus loads have been achieved by regulations aimed at

controlling the use of phosphate detergents along with advanced (e.g., tertiary) treatment of municipal wastewater.

Phosphorus from point sources is generally in the soluble *reactive* form (SRP) that is readily available for biological uptake. Stimulation of biological systems is readily observed when SRP is directly added to the photic zone of lake systems. Phosphorus from nonpoint sources can be in the SRP form, but can also be sorbed to soil minerals, as well as contained within organic matter. A dominant sediment sorption mechanism is reaction with iron hydroxides (FeOOH) which are ubiquitous within Southeastern Piedmont sediments. These iron minerals are commonly found coated on soil particles (sands, silts and clays). Sediment-bound phosphorus is either transported in suspension or as bedload in lake tributaries, and then subsequently deposited within lake embayments under quiescent conditions. In addition, nutrient spiraling within riverine systems may result in the partitioning of SRP into organic and sediment pools.

The primary sources of lake-nutrient enrichment can be from *allochthonous* (external) and *autochthonous* (internal) sources. Allochthonous inputs include point sources (wastewater treatment facilities) and nonpoint sources including agricultural, urban, and other land uses. Autochthonous inputs include release of phosphorus from sediment as well as from decaying organic matter. Internal loading results under chemical conditions where nutrients previously sorbed to lake sediments, desorb, and become available for algal growth. Of concern is the relative magnitude and importance of these two types of nutrient sources.

While current nutrient control strategies focus on limiting allochthonous inputs, the control of autochthonous inputs has not been addressed. Internal loading and its resulting effects are region-specific as they vary based on physical, chemical and biological conditions in lakes. Allochthonous nutrient loading is widely assumed to be the primary cause of cultural eutrophication. However, many lakes have continued to experience algal blooms and lake anoxia after sources of external loading were discontinued (Boström et al., 1988; Koussouris et al., 1992; Wetzel, 2001; Søndergaard et al., 2003). In lake systems with low concentrations

of orthophosphate in water and high concentrations in sediment, internal loading is the main mechanism providing P to algal biomass. For example, Santiago and Thomas (1992) found that in Lake Geneva internal P loading was inhibited when the orthophosphate concentration in the water was above approximately 14 μg/L.

The dominant limiting nutrient in impoundments is widely assumed to be phosphorus (Wetzel and Likens, 1991). Yet, nitrogen-fixing cyanobacteria are abundant during algal blooms in some lakes, which suggests that these lakes are nitrogen-limited rather than phosphorus-limited. One reason that phosphorus may be limiting is that an initial nitrogen-limitation allows nitrogen-fixing cyanobacteria to grow more rapidly than other phytoplankton. The subsequent abundance of these nitrogen-fixing cyanobacteria leads to increased nitrogen-cycling, and subsequent phosphorus limitation, as there is no equivalent biochemical mechanism to increase phosphorus (Schindler, 1977). Thus, while carbon and nitrogen are also of interest (as are sediments and toxic metals such as mercury, lead, and arsenic), the management of phosphorus loading is an integral part of lake management in these systems.

1.2 Problem Statement

Cultural eutrophication is found in lakes and reservoirs throughout the world, and is a common occurrence in southeastern Piedmont impoundments. Regional reservoirs are used within the Piedmont region of the southeastern U.S. to meet municipal and industrial water supply, wastewater dilution, recreation, and hydroelectric power generation needs of nearby communities. Poor water quality conditions caused by lake eutrophication adversely affects the use of reservoir waters to meet these needs. In addition, poor water quality also adversely affects native and introduced aquatic species by reducing their ability to forage, reproduce, and respire.

Lake Allatoona is a US Army Corps of Engineers reservoir located in northwest Georgia (Figure 1.1). Lake Allatoona was formed in January 1950 upon completion of Allatoona

dam. The surface area of the reservoir - when full - covers approximately 12,010 acres, the maximum depth at the dam is 145 feet, and the storage volume is approximately 367,500 acre-feet. The watershed area upstream of the dam is 1110 mi^2 and contains one large tributary (the Etowah River), and several smaller tributaries (the Little River, Noonday Creek, and Allatoona Creek). The designated uses for the reservoir include flood control, hydropower generation, water supply, recreation, fish and wildlife management, water quality, and navigation.

Suspended and bed sediments from influent tributaries accumulate in the upper reaches of the reservoir - primarily in the upper (Eastern) part of the lake near the Etowah and Little River inlets - forming depositional features including deltas and levees. These areas receive substantial inputs of sediments and nutrients from point and nonpoint sources. In addition, legacy sediments from historic sources have accumulated, providing long-term sources of sediment-related nutrients. Nutrients within these sediments may contribute to lake eutrophication via sediment release, thus contributing to internal loading.

Lake Allatoona, like many of Georgia's reservoirs, experiences seasonal algal blooms. These blooms are typically during the warm season (June through September) and are attributed to increased eutrophication from nutrient enrichment, primarily phosphorus. Phosphorus (P) is assumed to be the limiting nutrient in Southeastern Piedmont impoundments (Raschke, 1994). Causes of eutrophication for Lake Allatoona included phosphorus loads from watershed sources, including point and nonpoint sources. Point loads generally consist of soluble nutrients, while nonpoint loads consist of both soluble and particulate sources. Soluble loads are likely to become sorbed to suspended and bed sediments, as well as lost via biological uptake, while in transit to the lake.

Causes of eutrophication for Lake Allatoona included phosphorus loads from watershed sources, including point and nonpoint sources. Point loads generally consist of soluble nutrients, while nonpoint loads consist of both soluble and particulate sources. Soluble loads are

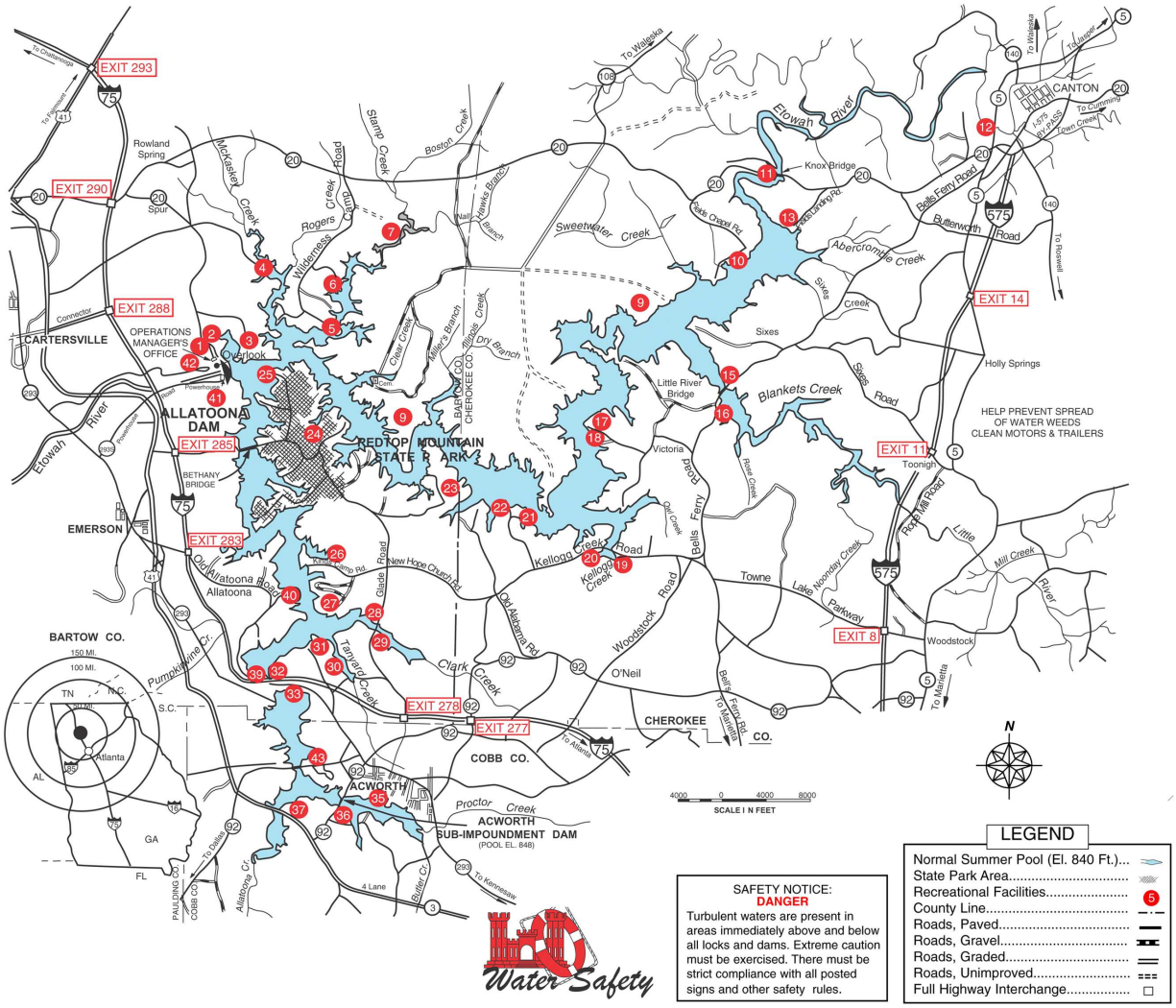

Figure 1.1: General location map of Lake Allatoona.

likely to become sorbed to suspended and bed sediments, as well as lost via biological uptake, while in transit to the lake.

Suspended and bed sediments from influent tributaries accumulate in the upper reaches of the reservoir - primarily in the upper (Eastern) part of the lake near the Etowah and Little River inlets - forming depositional features including deltas and levees. These areas receive substantial inputs of sediments and nutrients from point and nonpoint sources. In addition,

Table 1.1: Little River total phosphorus waste load allocation

NPDES Permit ID	Location	TP WLA (lb/yr)	TP WLA (kg/yr)
GA0024988	Cobb Noonday Creek WWTP	10,960	5,000
GA0001724	Seaboard Farms	5,000	2,300
GA0026263	Woodstock WPCP	760	350
Total		18,500	7,650

Source: `oaspub.epa.gov/pls/tmdl/waters_list.tmdl_report?p_tmdl_id=9825`

legacy sediments from historic sources have accumulated, providing long-term sources of sediment-related nutrients. Nutrients within these sediments may contribute to lake eutrophication via sediment release, thus contributing to internal loading.

In response to concerns about lake eutrophication, nutrient limits have been established for Lake Allatoona tributaries. Total maximum daily loads (TMDLs) are required by a federally mandated program to improve water quality. TMDLs are used to establish pollution limits for degraded waters. For Lake Allatoona, TMDLs have been established within the Little River Embayment (TMDL ID 9825) for Chlorophyll *a*. In response to this TMDL, waste load allocations (WLAs) have been established for total phosphorus for NPDES permits listed in Table 1.1, along with the total WLA for the embayment. Table 1.2 presents the total phosphorus loading limits for four tributaries. Of concern in this study are the first three sites, with a total annual load limit of 420,000 lb/yr, or approximately 191 metric tonnes/yr.

Dredging of nutrient-rich lake sediments has been proposed as a management tool for reducing within-lake nutrients. Small-scale dredging in Lake Allatoona has been conducted primarily for increasing navigability of the lake. Blankenship Sand operates dredges to remove sediment (primarily sand) from North Georgia lakes and rivers, including Lake Allatoona and its tributaries. Sediment dredging involves a mobile, floating, diesel-powered suction dredge

Table 1.2: Lake Allatoona total phosphorus load limits

Lake Tributary	TP Load	
	(lb/yr)	(kg/yr)
Etowah River at GA 140	340,000	155,000
Little River at GA 5	42,000	19,000
Noonday Creek at North Rope Mill Road	38,000	17,000
Shoal Creek at GA 108	9,200	4,2000

Source: `www.ganet.org/dnr/environ/rules_files/exist_files/391-3-6.pdf`

with a cutting head. Dredged materials are transported either to the shore or to a waiting clamshell barge that carries the sediments to an in-lake holding area that is re-dredged to the shoreline.

Dredged materials are separated into four parts;

1. A rubble fraction containing trash, large woody materials, and stones,

2. Sand that accumulates in piles, which are subsequently loaded onto road transport vehicles using front-end loaders,

3. Tailings composed of settleable solids (organic matter, fine sands, silts, and clays) that accumulate in holding (sedimentation) ponds, and

4. Tailwaters that are returned to the lake, containing the remaining unsettled clay fraction.

An important question is whether this dredging affects lake water quality. The dredging of sediments may reduce internal loading by removing internal sources of lake nutrients. While dredging may remove sand-sized particles from sediments, it may also result in the resuspension of clay and silt-sized particles during initial dredging, with subsequent transfer to the holding area and return of tailwaters to the lake. If desorption of P from the finer

materials occurs, then this action may degrade water quality. Yet, it is also possible that these finer materials may remove P from the water column by sorption and subsequent sedimentation. Removing within-lake sediments may benefit lake water quality by reducing the potential release of nutrients and toxic metals from these sediments.

Stakeholders would like to develop a program that would provide financial incentives for removal of lake phosphorus. Therefore, data documenting sediment phosphorus content is needed. Yet, not all phosphorus in sediments contributes to eutrophication and its resulting adverse effects. If a phosphorus trading policy is to be implemented to improve lake water quality, a method for determining algal-available benthic sediment phosphorus is needed.

1.3 Research Objectives

This project examines the possible benefits of removing benthic lake sediments on lake water quality by reducing the biologically available nutrients contained in these sediments. The key concerns are the total amount and release rates of phosphorus from lake sediments. There were several goals for this study. The first goal was to assess Lake Allatoona sediment composition and water quality. This was achieved by field data collection and sediment and water analyses. The second goal was to assess the amount of algal-available P in lake sediments. This was achieved by a laboratory algal assay in which algae were grown in flasks with sediment as the sole source of P. The final goal was to evaluate whether sediment removal by dredging would reduce algal biomass. This was achieved by a laboratory mesocosm study of sediment resuspension.

Field data collection and laboratory experiments were conducted to achieve the following specific objectives:

1. Gain a general understanding of Lake Allatoona sediment composition, especially with regards to differences based on sediment particle size

2. Assess changes in water quality and algal biomass that may be caused by current dredging methods, including sediment resuspension and sediment removal

3. Determine if there is a correlation between the results of chemical extraction of P in sediments and algal-available P as determined by algal assay

4. Make recommendations for future studies to assess potentially algal available phosphorus in sediments of Lake Allatoona

This study focused on quantifying the relative pools of nitrogen and phosphorus in lake nutrient cycling. Of specific concern was the biogeochemical cycling in Lake Allatoona, and how the cycling of nutrients was affected by lake dredging. Laboratory mesocosm and batch experiments were performed to quantify changes in water quality and algal biomass in response to current dredging methods.

1.4 Thesis Organization

The first chapter provides the general background along with a problem statement and research objectives. The second chapter provides a review of literature relevant to the studies performed. The third chapter summarizes tributary and within-lake sediment and nutrient conditions. The fourth chapter presents laboratory data to evaluate the bioavailability of lake sediment phosphorus. The final chapter summarizes research results and provides a number of recommendations for future research.

Chapter 2

Literature Review

2.1 Nutrient Limitations on Biological Productivity

Liebig's Law of the Minimum states that biological productivity is limited by a required nutrient or environmental parameter that is in the shortest supply. Phosphorus availability limits primary productivity in southeastern Piedmont impoundments (Raschke, 1994), as well as most lakes in the US (Vollenweider et al., 1974; Dillon and Rigler, 1974; Søballe and Kimmel, 1987), and the world (Schindler, 1978). In P-limited lakes, primary productivity and chlorophyll content are directly related to P concentration in the epilimnion (Schindler, 1978; Baines and Pace, 1994; Wetzel, 2001).

Primary productivity in this study refers to the algae that are oxygen-evolving photoautotrophs, including the photosynthetic protists (Domain Eukarya) and the cyanobacteria (Domain Bacteria, Kingdom Cyanophyta). Although the role of heterotrophic bacteria in lake P cycling will not be examined in this study, it is worth noting that these bacteria play a major role in mineralization of organic P compounds and thus algal-availability of P in lakes (Levine et al., 1986). They also compete directly with algae for phosphorus.

Studies of P competition between bacterioplankton and phytoplankton show a range of results depending on the parameter examined (uptake kinetics, P doubling rate, etc.), test organisms, and P concentrations used. However, empirical evidence has repeatedly supported the broad generalization that, when phytoplankton and bacterioplankton are both P-limited, bacterioplankton outcompete phytoplankton at low P concentrations, while phytoplankton tend to outcompete bacteria at higher P concentrations.

Heterotrophic microbial activity, as measured by respiration, may also be limited by other nutrients, availability of terminal electron acceptors and extracellular enzyme kinetics (used to break down larger organic compounds). Most heterotrophic bacteria and algae produce surface and extracellular phosphatases when the C:P ratio is high (Stewart and Wetzel, 1982). When excess orthophosphate is available, most algae store P in phosphate granules for use when P becomes limiting (Graham, 2000; Horne and Goldman, 1994).

2.2 Nutrient Limitation

One mechanism supporting widespread phosphorus limitation in lakes is that an initial nitrogen-limitation allows nitrogen-fixing cyanobacteria to grow more rapidly than other phytoplankton. The subsequent abundance of these nitrogen-fixing cyanobacteria leads to increased nitrogen-cycling, and subsequent phosphorus limitation, as there is no equivalent biochemical mechanism to increase algal-available phosphorus (Schindler, 1977). A second reason that P typically limits primary production is that P sorbs to Fe and Mn minerals in sediments of oxygenated waters (Mortimer, 1942).

Nitrogen (N) limitation is rare in the eastern U.S., but is more common in western lakes (Miller et al., 1978). Carbon (C) limitation is also rare (Schindler et al., 1972). In a study testing natural waters in the US, Miller et al. (1978) found that primary productivity was limited by trace elements in only 2% of 150 natural waters tested. N and P co-limitation is most common in eutrophic waters (Miller et al., 1978). Molar ratios of inorganic nitrogen (NO_3, NO_2 and NH_3) and soluble reactive P (PO_4^{3-}) in waters indicate macronutrient limitation. As a general rule, N:P ratios below approximately 10:1 indicate N limitation. N:P ratios between 10 and 12.1 generally indicate possible co-limitation. N:P ratios above 12 indicate P-limitation (Miller, 1978). Lake Allatoona samples from the Little River Marina, Noonday Creek Embayment and mid-lake at Marker 42E taken on July 15, 2005, were found to have N:P ratios $>$ 19-37, indicating P-limitation. Because all samples had SRP below detection limits, these ratios were calculated using the SRP detection limit of 50 μg/L.

Potentially algal-available phosphorus, or PAAP, is the total P available for algal growth in a given environment. Environmental factors affecting PAAP in lakes include redox potential, pH, temperature, sediment availability (suspended versus settled and thus available dependent on diffusion and bioturbation), mineralization, microbial processes, submerged macrophytes (Søndergaard et al., 2003) and algal phosphatases.

These factors are interconnected. For example, cool temperatures can affect the rate of nutrient remineralization by slowing production of exocellular enzymes made by algae and heterotrophic bacteria. Low redox potential and alkaline pH can work synergistically to favor orthophosphate release from sediments (Parker, 2004; Zeng et al., 2006). It is imperative that a method used to assess PAAP control for these environmental factors in order to be repeatable and for results to be comparable within and among lakes.

2.3 Sediment Phosphorus Release Mechanisms

Previous studies in the Chattahoochee River and tributaries within the Lake Lanier watershed have shown high correlation between stream discharge and the total suspended sediment concentration (Holmbeck-Pelham and Rasmussen, 1997) as well as between sediment and nutrient concentrations (Zeng and Rasmussen, 2005). These studies suggest that non-point sources of pollution in stormwater runoff cause episodic loading to the lake (Zeng and Rasmussen, 1999). Because phosphorus adsorbs strongly to particulate iron, and particulate iron is commonly found in sediments as a coating on soil particles, particulate phosphorus is a major component of non-point source inputs. These sediments tend to settle to the bottom of lentic waters, where they accumulate over time (Figure 2.1).

While sediment burial removes phosphorus in the short term, the eventual release from benthic sediments may increase and eventually equal, or exceed, subsequent deposition. The role of sediment-bound phosphorus in lake eutrophication is controlled by the rate of release from benthic sediments to the overlying (pelagic) water column (Mayhew et al., 2001; Parker

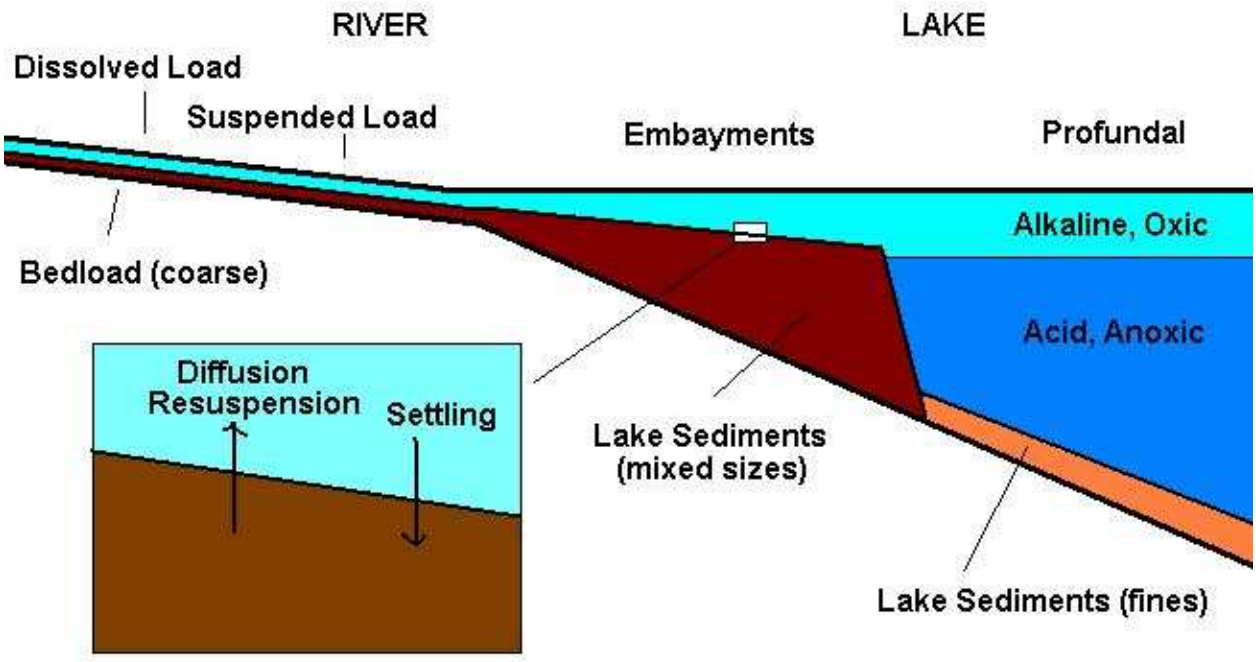

Figure 2.1: Typical sediment and chemical conditions in southeastern Piedmont reservoirs.

and Rasmussen, 2001; Zeng et al., 2006). The effects of internal phosphorus loading are often manifested as increases in lake primary productivity (Zeng et al., 2006).

Many laboratory and *in-situ* studies have evaluated internal phosphorus loading from lake sediments (Boström et al., 1988; Jin et al., 2005; Wang et al., 2005). Results vary widely due to variations in age and composition of sediments, organic matter content, particle size distribution, bioturbation, microbial community composition, history of lake phosphorus loading, and temperature (Boström et al., 1988; Lijklema, 1980; Froelich, 1988).

There are several mechanistic explanations for phosphorus release from sediments (Figure 2.2), including:

1. The dissolution of iron under anoxic conditions (Mortimer, 1941),
2. The exchange of hydroxy ion (OH^-) for adsorbed phosphate under alkaline conditions (Lijklema, 1980), and

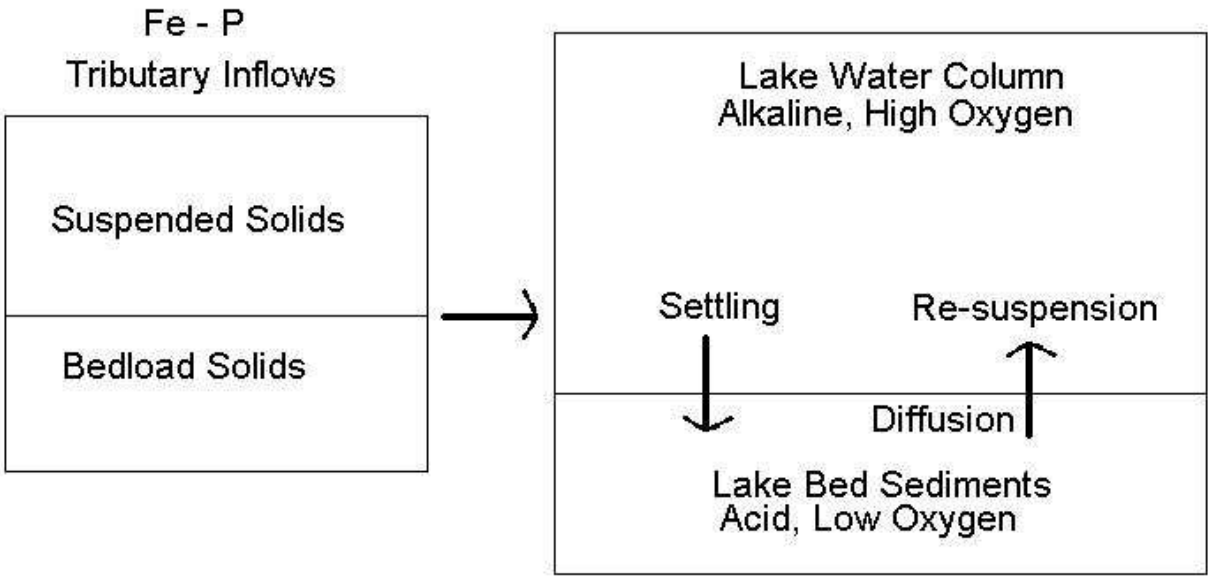

Figure 2.2: Phosphorus desorption mechanisms in Southeastern Piedmont impoundments.

3. The biologically mediated release of phosphate using alkaline phosphatases (Parker, 2004).

Understanding these release mechanisms provides us with the ability to estimate the conditions and controls of primary productivity with respect to phosphorus loading. Excessive primary productivity, manifested as algal blooms, can be quantified using chlorophyll *a* (Chl *a*) measurements because this is the primary pigment used by all photoautotrophs. Algal blooms are indicated by Chl *a* concentrations above $\approx$5 μg/L. Within the photic zone in the Southeastern Piedmont lakes, eutrophic conditions are also associated with elevated pH (>9) along with elevated dissolved oxygen (DO) concentrations (i.e., hyperoxic, DO>120% of saturation). Below the photic zone, respiration and decay of algal biomass causes hypoxia (DO<50%) or anoxia (DO<5%), along with lower pH (<7). Low DO and pH conditions are conducive to the reduction and subsequent dissolution of toxic metals, such as mercury.

Bio-accumulation of toxic metals is a risk for native aquatic species, as well as a risk to human health.

2.3.1 Low Oxidation-Reduction Potential

Sediment-phosphorus release is likely to occur in benthic sediments where organic loading is high. High organic loading causes a large biological oxygen demand that results in anoxic conditions. In Lake Allatoona, low oxygen concentrations are found in the late summer and fall when thermal stratification limits exchange between the hyper- and hypolimnion. These conditions foster slow iron reduction and subsequent P release. The rate of P release is limited by the relative concentrations of iron and phosphorus, as well as by the subsequent diffusion of phosphorus from the benthic zone into the pelagic zone.

Hypolimnetic DO declines as oxygen is used as the terminal electron acceptor (TEA) in aerobic respiration. At the same time, hypolimnetic oxygen production is limited as primary producers are light and/or nutrient limited at depth. Low redox potential refers to electron-rich conditions and a scarcity of oxygen and other electron acceptors. Under low redox potential Fe^{3+} is reduced to Fe^{2+} both biotically and abiotically. Chemical (abiotic) iron reduction occurs when redox potentials fall below approximately 200 millivolts (Boström et al., 1988). Yet, microbial (biotic) use of Fe^{3+} as a TEA may be responsible for more PO_4^{3-} release than abiotic reduction (Jones, 1983). Regardless of pathway, the chemical reduction of Fe^{3+} to Fe^{2+} releases Fe^{3+}-bound PO_4^{3-} from sediments (Mortimer, 1941).

2.3.2 Alkaline Conditions

Another sediment-phosphate release mechanism occurs during the summer in Southeastern Piedmont impoundments when warmer temperatures, abundant light, and available nutrients cause increased fixation of dissolved inorganic carbon (DIC) into biomass due to photosynthesis. Because DIC is the dominant buffer in Piedmont lakes, lowering DIC increases pH (López-Archilla et al., 2003).

The Redfield Ratio, when modified to include protons, (Equation 2.1) illustrates that for every 106 moles of carbon produced by photosynthesis, 124 moles of H^+ are consumed and 138 moles of O_2 are produced (Zeng et al. 2005).

$$106HCO_3^- + 16NO_3^- + HPO_4^{2-} + 124H^+ + 16H_2O \leftrightarrow (CH_2O)_{106}(NH_3)_{16}(H_3PO_4) + 138O_2 \quad (2.1)$$

In eutrophic lakes, such as Lake Allatoona, the change in pH due to carbon fixation can be dramatic, raising pH>9 (Kann and Smith, 1999; Parker, 2004; Ceballos, unpublished data). Shallow water depth, high water column stability and prolonged photoperiod also contribute to the alkaline release of PO_4^{3-} (Kann and Welch, 2005). Thermal stratification limits exchange between the O_2-rich epilimnion and hypolimnion where limited photosynthesis and detrital decay result in low O_2 and abundant CO_2 (Figure 2.3). This results in DO and pH profiles typical of hypereutrophic lakes (Wetzel, 2002).

Under alkaline conditions hydroxide anions (OH^-) compete with phosphate anions (PO_4^{3-}) to adsorb to sediment Fe- and Al-oxide solid phases (Macpherson et al., 1958; Lijklema, 1977; Lijklema, 1980). While under this mechanism one would expect phosphate release to increase with increasing pH, some evidence suggests that phosphate release is greatest in the pH range of 9 to 9.5 (MacPherson et al., 1958).

2.3.3 Phosphatases

The form of P that is immediately algal-available is orthophosphate. However, algae, as well as some zooplankters and heterotrophic bacteria, produce surface-bound and exuded enzymes that have the ability to cleave phosphorus from organic matter and iron minerals (Chrost, 1991). In general, low concentrations of P promote the production of phosphatases.

During the 1970s and 80s, alkaline phosphatase activity assays were used in efforts to measure algal primary production in P-limited environments (Stevens and Parr, 1977; Elser and Kimmel, 1986), and as an indicator of P limitation (Elser and Kimmel, 1986) with

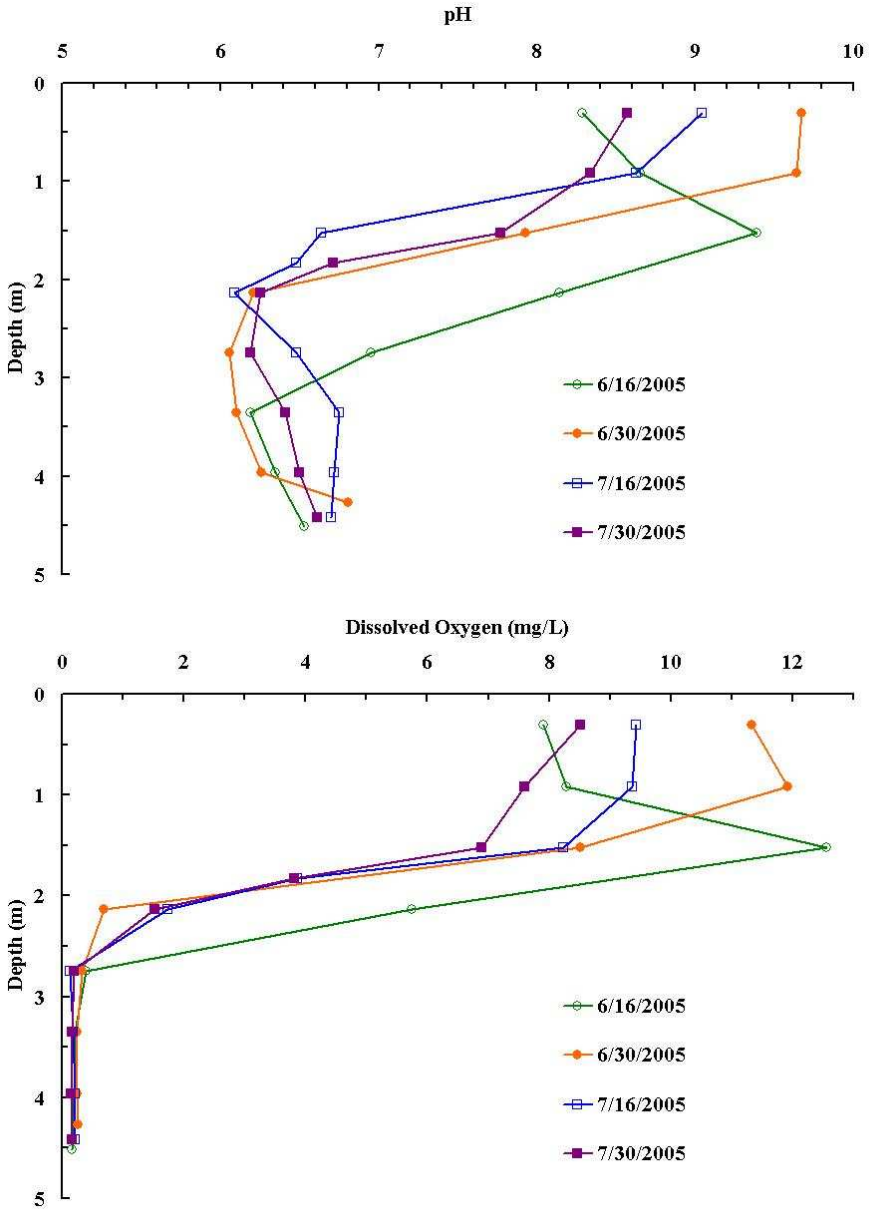

Figure 2.3: pH (top) and Dissolved Oxygen (bottom) profiles in Lake Herrick, a hypereutrophic lake in the Southeastern Georgia Piedmont located on the University of Georgia campus.

varying success (Currie and Kalff, 1984). Current use of the assay for these purposes is now scant in the literature.

2.3.4 Interrelationships Between Mechanisms

Sediments can be resuspended throughout the water column in shallow lakes and impoundments by wind, bioturbation and recreational boating (Kann and Welch. 2005; Ekholm and Kirkkala, 1997; Grobbelaar, 1983). Depending on the chemical equilibrium of P, this resuspension may or may not result in release of PO_4^{3-} from sediments into the water column (Fan et al., 2001; Koski-Vähälä and Hartikainen, 2001). As previously discussed, when algae are P-limited, their alkaline phosphatases may solubilize sorbed PO_4^{3-}.

Sediment resuspension may result in enhancing P release by reducing redox potential and increasing pH. If P released from resuspended sediments supports increased photosynthesis, the subsequent sinking and mineralization of algal biomass would further lower redox potential. The lower redox potential may then further increase P release from anoxic sediments. If photosynthesis in the epilimnion increases pH above 8.5, PO_4^{3-} release may also increase (Macpherson et al., 1952; Wang et al., 2005).

P released by sediment resuspension, and thus mixing with algae, likely includes the loosely sorbed-P bound to sediment surfaces, and a portion of the P released under low Eh and high pH. Currently there exists no chemical test to simulate this release.

2.4 Determination of Phosphorus Availability

Not all phosphorus in sediments is algal-available. The amount of labile phosphorus in lake sediments varies based on sediment and water physical, chemical and biological characteristics as discussed previously. To date there is no standard method or model that can be used to determine sediment P algal-availability under typical eutrophic conditions.

Many chemical and biological analyses have been developed in attempt to quantify the algal availability of sediment phosphorus. Chemical extractions have the advantage of being

rapid and inexpensive, yet in many circumstances the results do not accurately estimate algal-available phosphorus when compared to algal assays. Algal assays have the advantage of integrating algal physiological responses to environmental factors such as phosphorus concentration, light, temperature, redox potential and pH. However, the algal assays are very time and labor intensive and not typically performed in environmental analytical laboratories.

Limnocorrals are enclosures built around a portion of a lake for the purpose of manipulation of the enclosed environment. Limnocorrals have been used successfully for studies of food web dynamics (Burkholder and Cuker, 1991; Burkholder et al., 1998). However, to assess potentially algal-available phosphorus, limnocorrals are unable to maintain constant environmental factors, such as redox potential or pH. Traditionally, algal-available P is determined by algal assays in laboratories (Miller et al., 1978). Algal assays allow for control and manipulation of many environmental factors, as well as allow for replicates necessary for statistical analyses.

2.4.1 Chemical methods

Chemical extractions and biological assays are common methods used to estimate algal available sediment bound P. The chemical extractions typically employed remove P sequentially from sediments. The extraction sequence removes loosely sorbed P (using NH_4Cl) first, followed by P made labile by reducing conditions (Fe- and Al-bound P using NaOH) (Boström et al., 1988), P released at high pH and, lastly, calcium-bound P (apatite-P using HCl) (Boström et al., 1988).

Cowen and Lee (1976b) found that 6 to 25% of total P in suspended sediments from several New York lakes was available to *S. capricornutum*. Williams et al (1980) found that 8-50% of total P was algal available. Dorich et al (1984) found that 10 to 33% of total P in suspended sediments of agricultural origin was algal available. In this system, algal availability of P did not differ among particle sizes.

2.4.2 Algal assay methods

Algal assay methods vary widely, with the measured parameter being either growth rate, or primary production, (Santiago and Thomas, 1992), biomass (Ellis and Stanford, 1988; Engle and Sarnelle, 1990), or, occasionally, enzyme production (Elser and Kimmel, 1986).

The most common method for measuring the rate of primary production is by ^{14}C assimilation (Darley et al., 1981; Peterson et al., 1983; Sloterdijk et al., 1989). These experiments have the advantage of being rapid, typically lasting only a few hours. However, the method measures something between gross and net primary productivity (Miller, 2004).

There are three ways to measure algal biomass: pigment (typically Chl *a*), fluorescence (an indirect measurement of Chl *a*) and cell abundance (Miller, 2004). Each method has advantages and disadvantages. For example, a problem with measuring pigments is that a conversion factor is needed to obtain the C:chlorophyll ratio, which can range from 10 to 200. Counting cells, however, is time consuming. As well, *in vivo* fluorescence varies as a function of the physiological state of the cell.

The standard EPA algal assay method, which was developed in the 1970s, measures increases in algal biomass (dry weight) of a single green alga, *S. capricornutum*. This test, the Algal Assay Bottle Test (AABT) was developed to test water nutrient limitation and toxicity (Miller et al., 1978). The method has also been modified by researchers to evaluate algal availability of suspended sediment (Ellis and Stanford, 1988), agricultural runoff (Sharpley, 2003) and the effect of autoclaving sediments on algal-available P (Anderson and Magdoff, 2005). The green alga, *S. capricornutum*, was chosen for algal assays in this study because it is the standard alga used by the EPA and many other researchers for algal assays of nutrients and toxins. The non-motile and single-celled nature of *S. capricornutum* allows for relatively easy enumeration of cells using an improved Neubauer hemacytometer. It is also cosmopolitan and does not have a negative response to constant light or pH between 6-10 (Miller et al., 1978).

The variation in method that has received much attention from researchers is exposure of the test alga to sediments. Methods which separate algae from sediment tend to result in negligable growth (Boström et al., 1988). Algal assays that mix algae directly with sediment have the advantage of including the degradation of particulate P by algal phosphatases. However, the mixture of cells with sediment in an algal assay can make visual enumeration of cells difficult to impossible. If measuring Chl *a*, mixing of cells with sediments will result in a measurement of Chl *a* from experimental conditions as well as chlorophyll derivatives already in the sediment.

A variation of the AABT was chosen for this algal assay because the methods are explicitly stated. The method has also been repeatedly tested and revised as necessary by the EPA. The algal culture used varies, but typically a unicellular green alga, such as *Selenastrum capricornutum* or *Scenedesmus sp.* is chosen. While the use of the natural algal community for an algal assay may seem, intuitively, to be the best choice to assess response to environment, its use is not recommended (Miller et al., 1978). Rather, unialgal cultures from a reputable source are standard for algal assays. There are several reasons for this:

1. The assay must be repeatable. Because the dominant alga in a community will likely change throughout seasons, and algae respond differently to their environment, use of a natural community would compromise repeatability.

2. The success of an algal assay relies heavily on the accurate enumeration of organisms, pigments or carbon uptake. A natural algal community consists of multicellular and colonial genera, which are difficult to enumerate, contain different concentrations of pigments, and variable carbon:pigment ratios.

3. In the case of Lake Allatoona, comparison of potentially algal-available P from sediment is necessary for a nutrient trading system if the purpose of that system is to reduce the effects of eutrophication. Therefore, use of the same test organism, with the same environmental requirements, is necessary.

4. Natural algal communities typically deteriorate after enclosure (Boström et al., 1988).

2.4.3 Correlating chemical and algal assays

Shallow southeastern Piedmont impoundments typically produce algal blooms in the Summer or early Fall. If internal loading supports these blooms, then dredging of nutrient-rich sediments could be a valuable tool to decrease the effects of eutrophication. A nutrient trading system based on sediment P concentrations would need an appropriate method to quantify algal-available P in sediments. This could be achieved by correlating algal-available P as determined by bioassay with a chemical extraction of P.

As algal assays are more time consuming and expensive than chemical extractions, many studies have tried to correlate the results of chemical extractions of P with algal-availability of P. In some cases, the results of a particular extraction have been correlated to algal-available P (Sharpley, 1993; Anderson and Magdoff, 2005). While results vary from lake to lake, results from NaOH (Premazzi and Zanon, 1984), nitrilotriacetic acid (NTA) (Golterman, 1976) and iron-impregnated filter papers (Sharpley, 1993; Anderson and Magdoff, 2005) have been correlated to algal-available P as measured through algal assays.

However, to date no general agreement has been found between chemical extractions of P and assays for algal-available P (Søndergaard et al., 2003). These discrepancies are logical, as algal assay methods (sediment mixed or separated from algae), differences in sediment physico-chemical properties, algal culture used, previous algal nutrient limitation and parameter measured (growth or biomass) differ. A more thorough comparison of algal assay and chemical extraction results was reviewed by Boström et al. (1988).

Golterman's study (1976) was possibly the first published account of an effort to quantify algal uptake of P from sediment and to correlate results with a chemical extraction. In bioassays testing clay from the Kainju River in Uganda, Golterman estimated that 90 μg PO_4-P were available to the alga *Scenedesmus* per gram clay in contact with thin slices of

sediment filled agar. An extraction of sediment P with nitrilotriacetic acid (NTA) gave equal results (within 2-5%) of the amount uptaken by the test alga.

Ellis and Stanford (1988) performed sediment algal assays using *Selenastrum capricornutum* and suspended sediments $> 0.1\mu$m from tributaries to Flathead Lake in Montana. They found NTA-extracted P much higher than algal-available P measured by algal assay. The results of the extraction varied dramatically depending on the solution:solid ratio used. 1-36 μg-P/g suspended particulate matter (soils and sediments) was algal-available when tested by algal assay.

If chemical extractions are to be used to assess algal-availability of sediment P in Lake Allatoona, a correlation must be made between algal-availability and a particular extraction technique. To our knowledge there have been no studies comparing the results of chemical extractions with algal assays in Lake Allatoona.

Chapter 3

Field Measurements of Lake Allatoona Sediment and Nutrients

The objective of this chapter is to provide characterization data of Lake Allatoona water quality including sediment, nutrients, and metals. These data are then used to assess the magnitude of the water quality problems in Lake Allatoona, and to develop alternative management strategies for mitigating these problems.

3.1 Field and Laboratory Analyses

Both field and laboratory data were used for collecting data about lake water quality. Field measurements were collected using a Hydrolab Quanta (Austin, TX). The instrument provides field temperature, pH, specific conductance, turbidity, and dissolved oxygen concentrations. The instrument was calibrated prior to each field trip.

Field water and sediment samples that were subsequently analyzed in a laboratory were collected in acid-washed bottles, given a unique identifier label, and placed on ice during transport back to the University of Georgia. A minimum of two (duplicate) samples were generally collected from each location. Water samples from river sites were collected using a USGS DH-48 sampler lowered by a rope from a bridge. These samples were collected at approximately one- and two-thirds of the distance across the channel. Samples were collected in acid-washed bottles, stored on ice, and returned to the laboratory within ten hours.

Water and sediment samples were analyzed at three laboratories at the University of Georgia:

1. The UGA Soil, Plant, and Water Laboratory within the Agricultural and Environmental Services Laboratory, *aesl.ces.uga.edu*, was used for both sediment and water testing. Specific analytic methods are available at:

   - Measurement of Lime Buffer Capacity
     pubs.caes.uga.edu/caespubs/pubs/PDF/C874.pdf
   - Soil pH and Salt Concentration
     pubs.caes.uga.edu/caespubs/pubs/PDF/C875.pdf

   Organic Matter was determined by the *loss on ignition* method for three hours at 360°C. Sediment results are reported in units of mass in percent (%, g/100g) or in ppm (mg/kg), while water results are reported in units of mass per volume (mg/L). Other methods are available at: *aesl.ces.uga.edu/publications/soilcirc/index.html*.

2. The UGA Laboratory for Environmental Analysis, *www.uga.edu/lea*, within the College of Agricultural and Environmental Sciences and the Warnell School of Forestry and Natural Resources, was used for sediment testing. Metals were analyzed using a Elan 6000 inductively coupled (argon) plasma spectrometer (ICP) (Perkin Elmer, Waltham, MA) equipped with a mass spectrometer (MS) detector system. A Leco 2000 analyzer (Perkin Elmer, Waltham, MA) was used to determine C, N, and S in solid samples.

3. The UGA Stable Isotope & Soil Biology Laboratory, *www.uga.edu/sisbl*, within the Institute of Ecology was used to determine the soluble reactive phosphorus (SRP) concentration in water samples. This laboratory uses the molybdate color reaction chemistry (automated Murphy/Riley 1962) to analyze for SRP (orthophosphate) or SRP present as the product of a digest or extract procedure. A discussion of principles and practical considerations in continuous-flow colorimetry is available at *www.uga.edu/sisbl/contin.html*. The orthophosphate procedure used by this laboratory is specified in replicable detail at *www.uga.edu/sisbl/epa-po4.html*.

Data quality is emphasized at these laboratories, given the importance of accuracy in analysis for both research and regulatory arenas. A rigorous protocol of quality assurance/quality control (QA/QC) is followed for every sample batch run for metals analysis, including re-calibration and re-zeroing, replication, individual calibration of all elements in a sample run over the concentration range of interest, and use of check standards to ensure proper calibration. With larger sample batches, sample spikes and certified reference materials (NIST or equivalent) are run to confirm recovery of analyte elements. QA/QC information is provided with every sample set to evaluate detection limits, precision, and overall data quality.

These laboratories use multiple terms to describe phosphorus concentrations, including dissolved inorganic phosphorus (DIP), dissolved reactive phosphorus (DRP), soluble reactive phosphorus (SRP), as well as biologically available phosphorus (BAP). These terms correspond to specific laboratory methods used to estimate the phosphorus in the soil or water sample. While there are advantages and disadvantages for each method of analysis, there is not a unique analysis that is appropriate for all applications.

3.2 Site Visits

Sand, silt and water samples were collected on July 15, 2005, from Blankenship Sand's Little River and Knox Bridge locations (Figure 3.1). Measurements of in situ water quality were made using a Hydrolab Quanta (Hydrolab, Austin, Texas) from a pontoon boat. Water samples were collected near and down-lake from the dredging operation and the Little River and Noonday Creek embayments.

The weather was overcast with heavy rain during the afternoon. Lake water levels were approximately ten feet above normal pool due to a recent major flood. The water was muddy in both the Little River and Etowah River embayments. A clearly defined sediment wedge was present within the Knox Bridge embayment near the separations area, with elevated turbidity upstream and below the pre-storm event lake water.

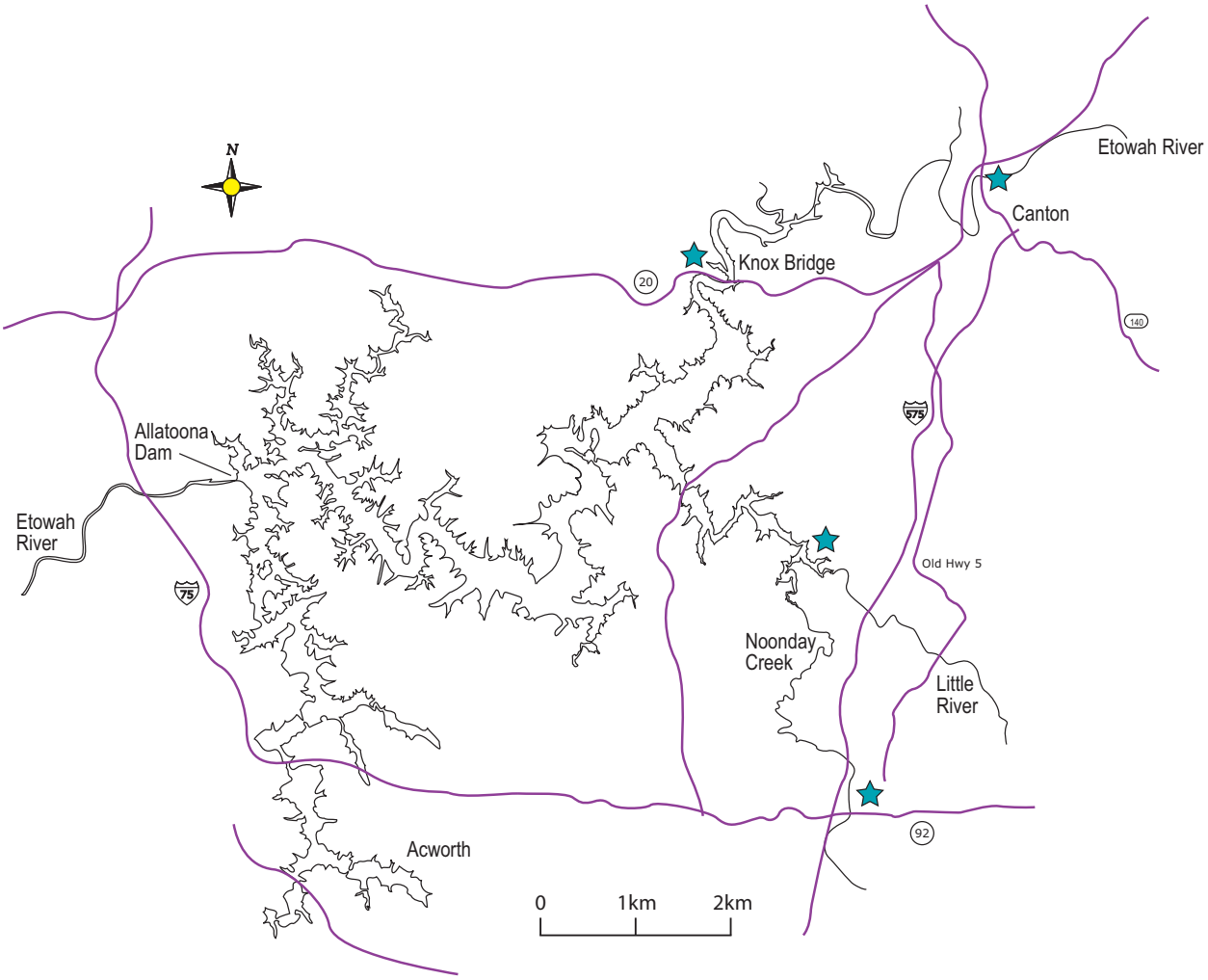

Figure 3.1: Map of Lake Allatoona, Georgia, showing lake tributaries and dredging sites (stars).

Lake water quality data collected during the July 15, 2005, visit can be used to provide a general overview of lake conditions. These data are provided in Table 3.1. The visit was conducted after a period of substantial rainfall, and the lake was approximately ten feet above normal pool. In general, lake turbidities were high, and shallow Secchi depths were observed, which can be attributed to the recent stormwater (allochthonous) inputs. The pH was well below 8.5, which is the minimum pH at which phosphorus should become bio-available due to alkaline desorption, so that algae did not appear to be abundant.

Table 3.1: Lake Allatoona water quality data collected on July 15, 2005.

Location	Knox Bridge		Marker	Little River	Noonday Creek
	Dredge	Separations	42E	Marina	Embayment
Sample	L1	L2	L3	L4	L5
Latitude, °N	34.22951	34.21551	34.18098	34.15961	34.13983
Longitude, °W	84.56548	84.57546	84.58952	84.57703	84.54491
Lake Depth, ft	21.4	>25	>50	>20	11.5
Secchi Depth, cm	NA	NA	62.5	32	32
Turbidity, NTU	507	NA	19.4	49.3	44.1
NH_4, mg/L	0.04	NA	0.12	0.17	0.06
NO_3, mg/L	0.09	NA	0.20	0.25	1.00
PO_4, mg/L	0.092	NA	BDL	BDL	BDL

Lake water quality data are illustrated in Figures 3.2-3.6. The deeper parts of the lake were generally of better quality (lower turbidities) than shallower parts, except the lowest dissolved oxygen reading (<1 mg/L) was found at the deepest part of the lake (50 m). Also, the highest pHs were also found at the surface in the deepest part. This indicates that the allochthonous (external) inputs are high in the shallow parts of the lake and the autochthonous (internal) inputs are highest in the deeper parts. This is consistent with data from Lake Lanier and many other Piedmont reservoirs.

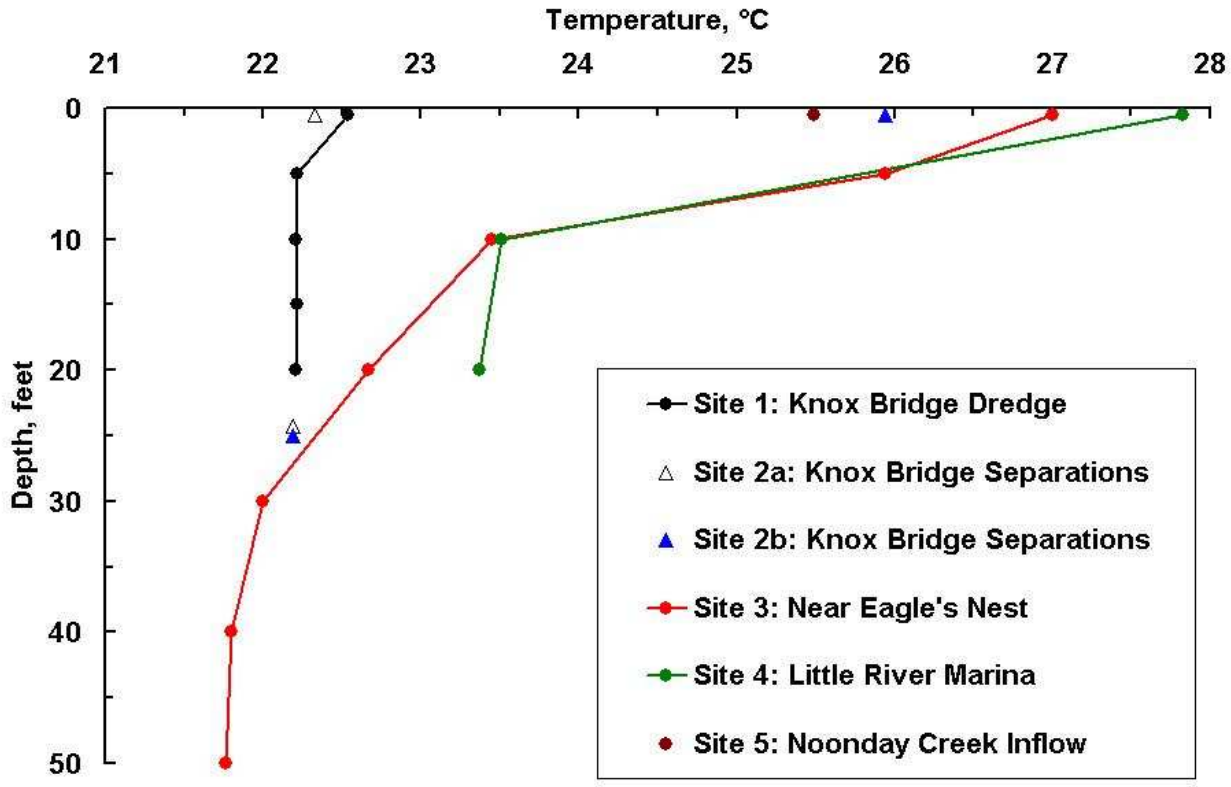

Figure 3.2: Lake Allatoona water temperature observations, July 15, 2005.

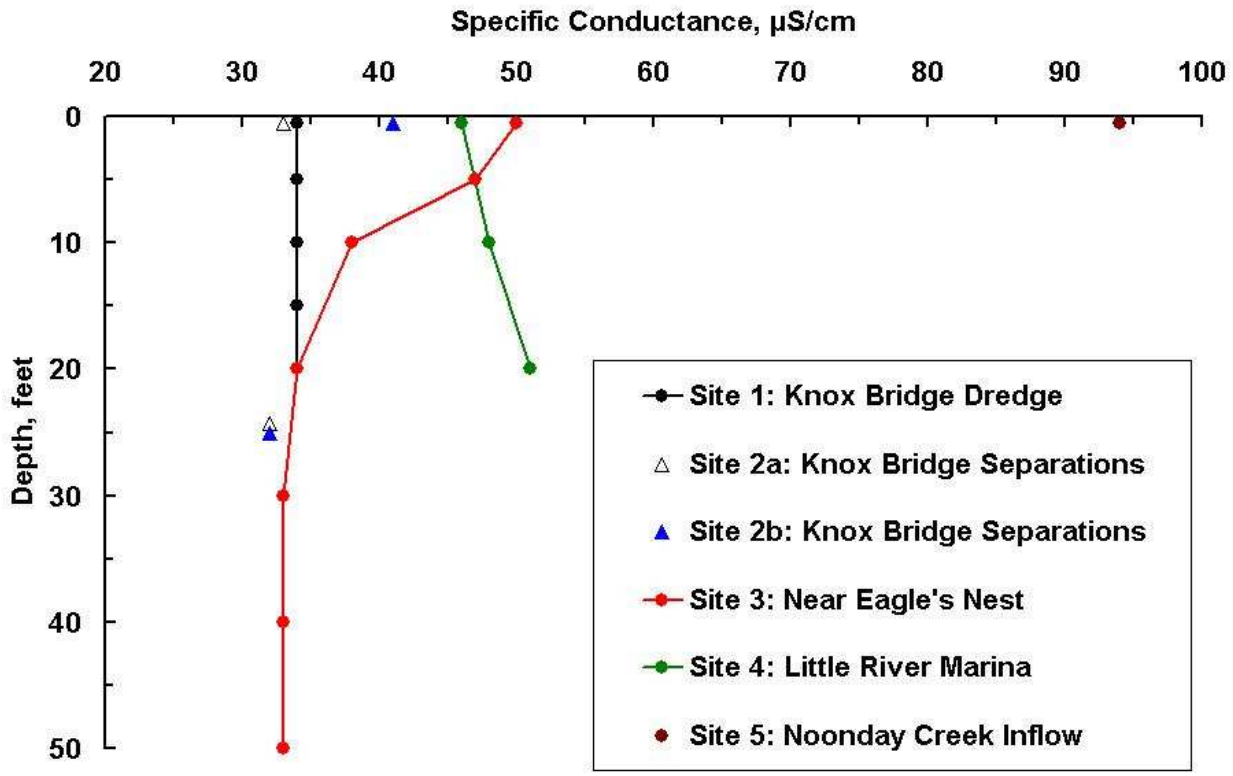

Figure 3.3: Lake Allatoona water specific conductance observations, July 15, 2005.

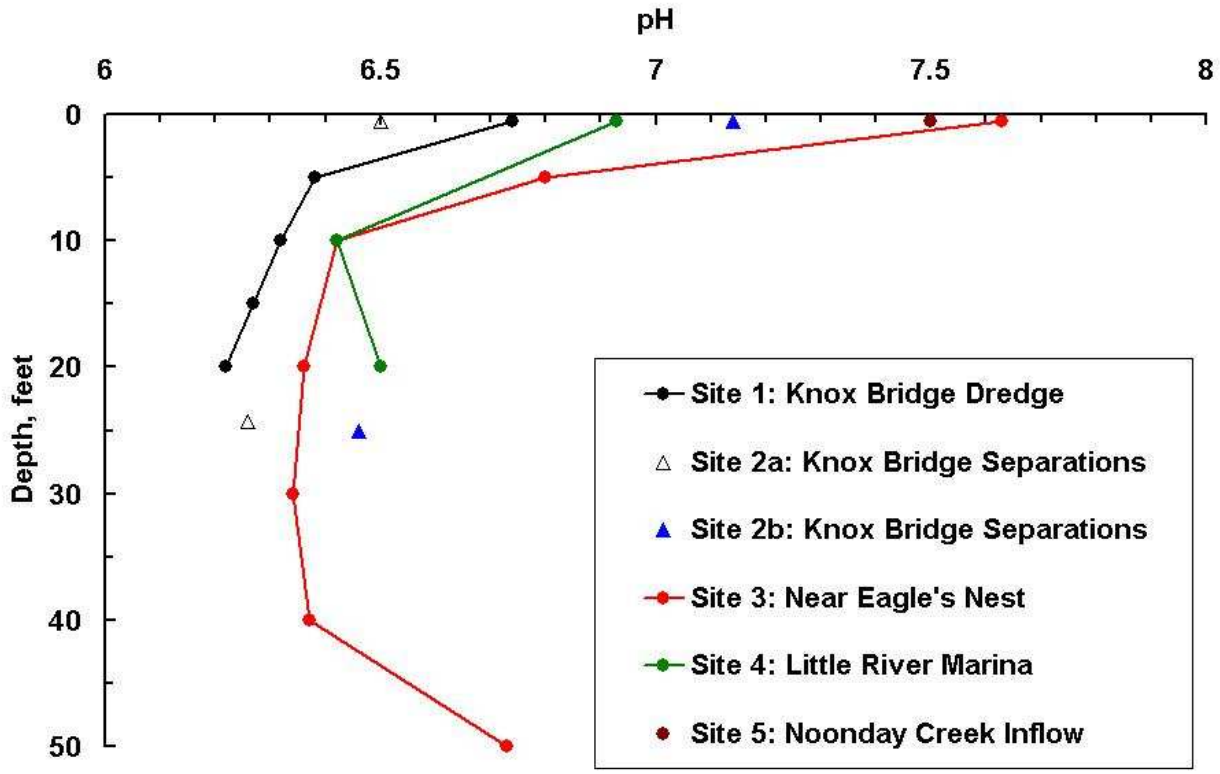

Figure 3.4: Lake Allatoona water pH observations, July 15, 2005.

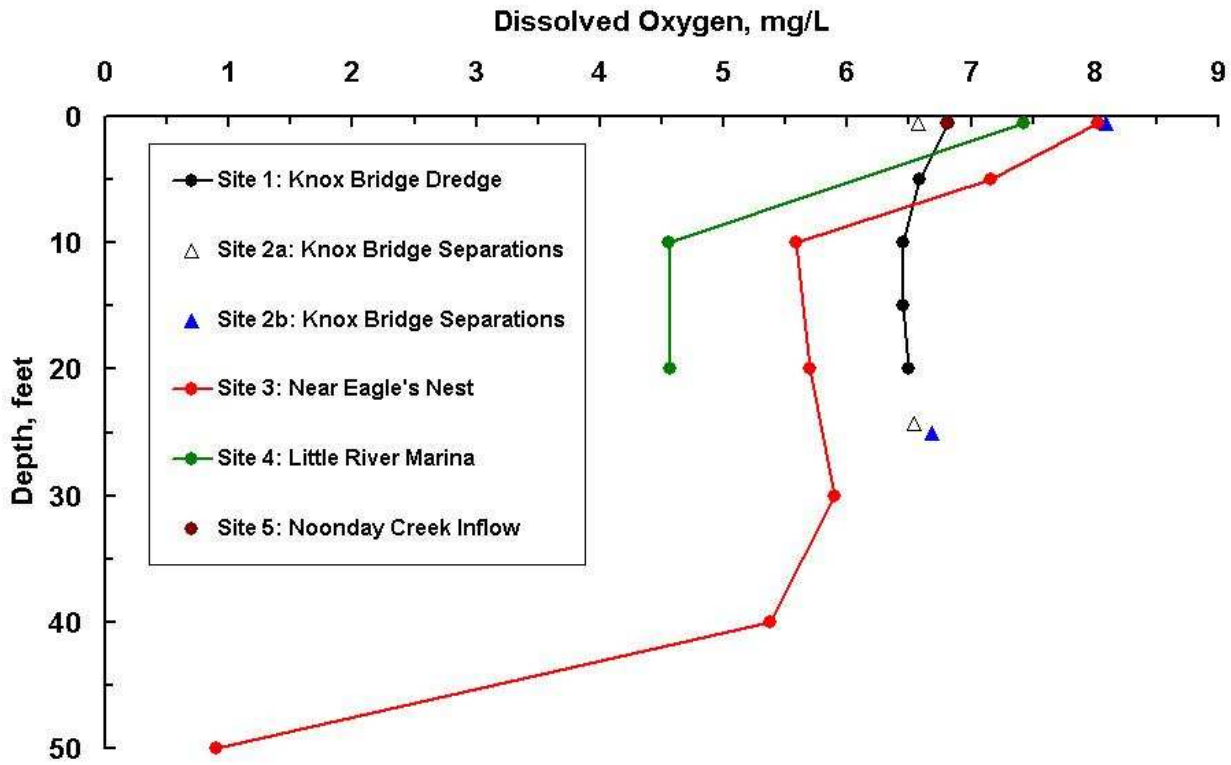

Figure 3.5: Lake Allatoona water dissolved oxygen observations, July 15, 2005.

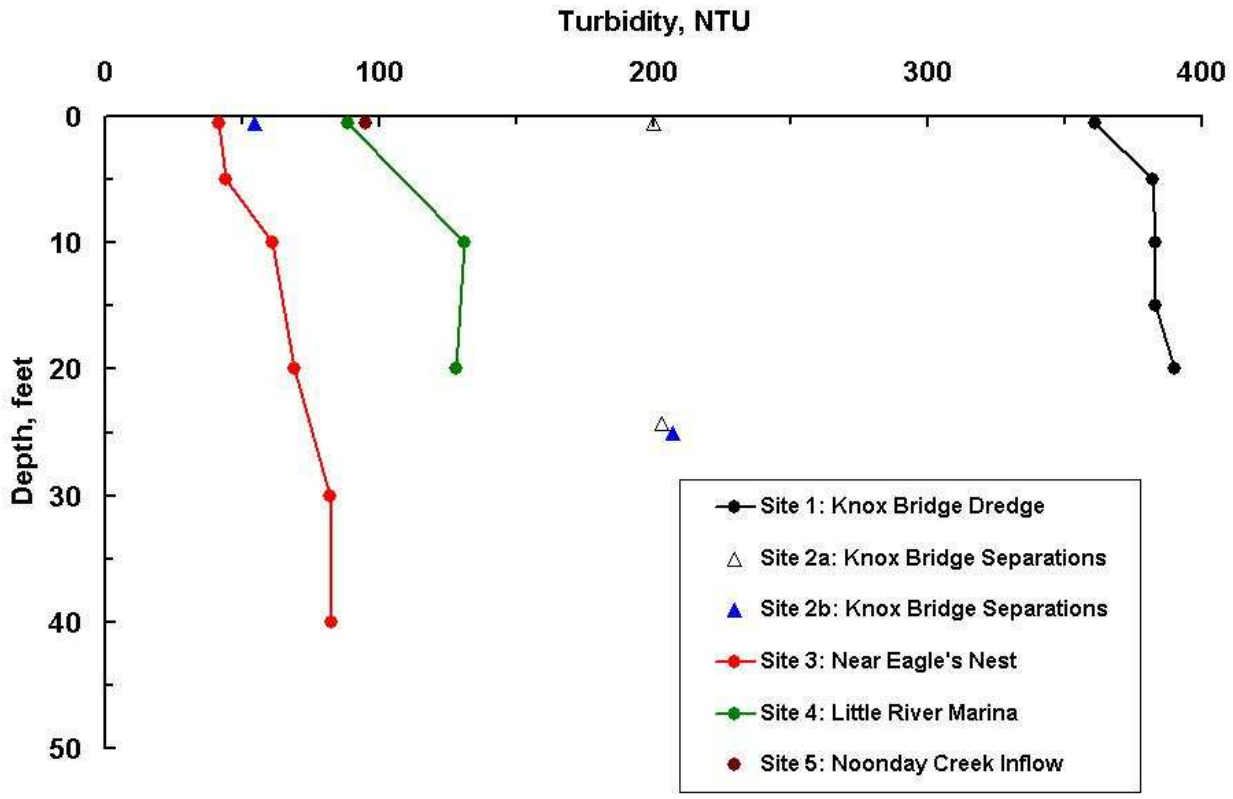

Figure 3.6: Lake Allatoona water turbidity observations, July 15, 2005.

While the lake generally had a green color, a distinct color line was observed in the lake where muddy water met the green water near the Knox Bridge Dredging Operations. This muddy water was clearly the result of tributary inflows from the Etowah River in response to heavy rainfall. This interface just happened to be at the Knox Bridge Separations Site at 11^{am} on July 15, 2005, (Site L2 in Table 3.1) but had moved up-lake by 5^{pm}. Based on the temperature and turbidity readings, it appears that the stormwater surge coming from upstream was cooler and more sediment-laden than prestorm lake water, and, hence, more dense. The turbid water appeared to be plunging under the pre-storm lake water. This hypothesis was supported by higher turbidities and lower temperatures (equal to the up-lake values) in deep-water observations.

The lake water quality did not vary to a substantial degree with depth in the muddy water zone, but was very stratified (vertically variable) downstream of the interface. The dredge operators noted that the muddy water was the result of the most recent storm and that earlier storms had already mixed into the lake.

On August 9, 2005, the Etowah River was sampled from the Highway 140 crossing at Canton. The Little River was sampled from the Old Highway 5 bridge near Woodstock. Noonday Creek was sampled from the Highway 92 bridge near Woodstock. Duplicate samples were collected from each location, at approximately one- and two-thirds of the distance across the channel. Water samples were collected using a USGS DH-48 sampler lowered by a rope from the bridge. Samples were collected in acid-washed bottles, stored on ice, and returned to the laboratory within ten hours.

A third set of samples were collected on October 7 and 8, 2005, following a storm event. On October 7th water samples in duplicate were collected from the Etowah River, Little River, and Noonday Creek. On October 8, 2005, sediment and water samples were collected from the Knox Bridge Dredging Operations, the Little River Dredging Operations, and from Noonday Creek Embayment. Access to the embayment was obtained using a Jon Boat.

Table 3.2: Lake Allatoona field water quality observations. DO is dissolved oxygen concentration

Location	Little River	Noonday Creek		
	tailwater	lake water	pore water	perched water
Date	8/9/5	10/8/5	10/8/5	10/8/5
Temperature, °C	25.38	22.03	21.35	20.94
Conductivity, μS/cm	99	168	417	297
Turbidity, NTU	1059	>5999	115	85
pH	7.08	7.48	6.08	7.58
DO, % saturation	101.1	87.2	3.5	69.5
DO, mg/L	8.17	7.48	0.33	69.5

Sediment samples were collected from an exposed bank of sediment next to the main channel in Noonday Creek embayment.

Field *in situ* Hydrolab water quality data for these visits are presented in Table 3.2. These data show the elevated conductivity of Noonday Creek, and the low pore water dissolved oxygen concentrations. Also note the high turbidity of these samples, undoubtedly due to stirring of sediments during measurement.

On November 5, 2005, approximately 40 L of sediment suspension was collected from the Lake Allatoona Noonday Creek embayment. This was the same location where the sediment samples on October 8, 2005, were collected. Access on this date was overland from a restricted access road within a subdivision. These sediments were returned to the laboratory for subsequent mesocosm investigations.

On March 3, 2006, a vertical sediment profile next to the Knox Bridge Boat Ramp was collected. A bucket auger with extension was used to collect mid-lake sediment samples from the surface to approximately 4 m at 30 cm intervals. A total of 12 samples were collected. Approximately 100 L of lake water was collected from the Little River Boat Ramp for laboratory mesocosm studies.

3.3 Lake Sediment and Nutrient Inputs

This section focuses on quantifying the sediment and nutrient inputs to Lake Allatoona from three tributaries (Etowah River, Little River, and Noonday Creek). Three sources of data are available for this assessment, including U.S. Geological Survey water quality database; data collected by the University of Georgia as part of this investigation, and data reported in a Clean Lakes study performed by Kennesaw State during the 1990s. These estimates are compared against each other and with the total maximum daily load (TMDL) estimates currently in place for these tributaries on Lake Allatoona.

3.3.1 U.S. Geological Survey Data

An important source for Lake Allatoona tributary water quality data is the U.S. Geological Survey national water quality database, available at `waterdata.usgs.gov/nwis/qw`. The database allows access to Lake Allatoona tributary inflow water quality data. Stations examined are provided in Table 3.3. An inspection of the available tributary water quality data is provided in Figures 3.7-3.11. The first two figures (Figure 3.7 and 3.8) show the time trend for turbidity and total phosphorus, respectively. Note that observations are generally low, but occasional large values are observed.

Table 3.3: US. Geological Survey river gaging stations used in this study.

- USGS 02392000 Etowah River at Canton GA
  Latitude 34°14'23.4", Longitude 84°29'41.08" NAD27
  Cherokee County GA, Hydrologic Unit 03150104
  Watershed area 613 mi^2, Annual mean discharge 1239 cfs (1897-1997)
- USGS 02392780 Little River at Georgia 5, near Woodstock GA
  Latitude 34°07'20", Longitude 84°30'16" NAD27
  Cherokee County GA, Hydrologic Unit 03150104
  Watershed area 139 mi^2
- USGS 02393000 Noonday Creek at AT GA 92, near Woodstock GA
  Latitude 34°05'09", Longitude 84°31'47" NAD27
  Cherokee County, Georgia , Hydrologic Unit 03150104
  Watershed Area 41.4 mi^2

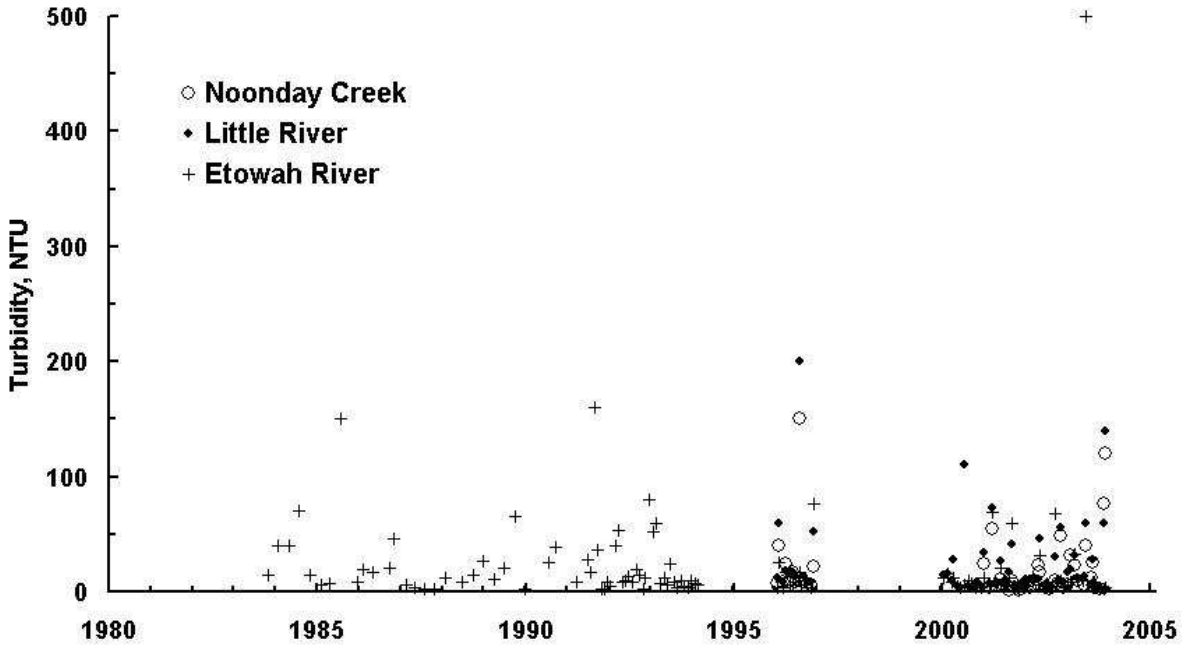

Figure 3.7: Plot of historical USGS turbidity time series for three Lake Allatoona tributaries.

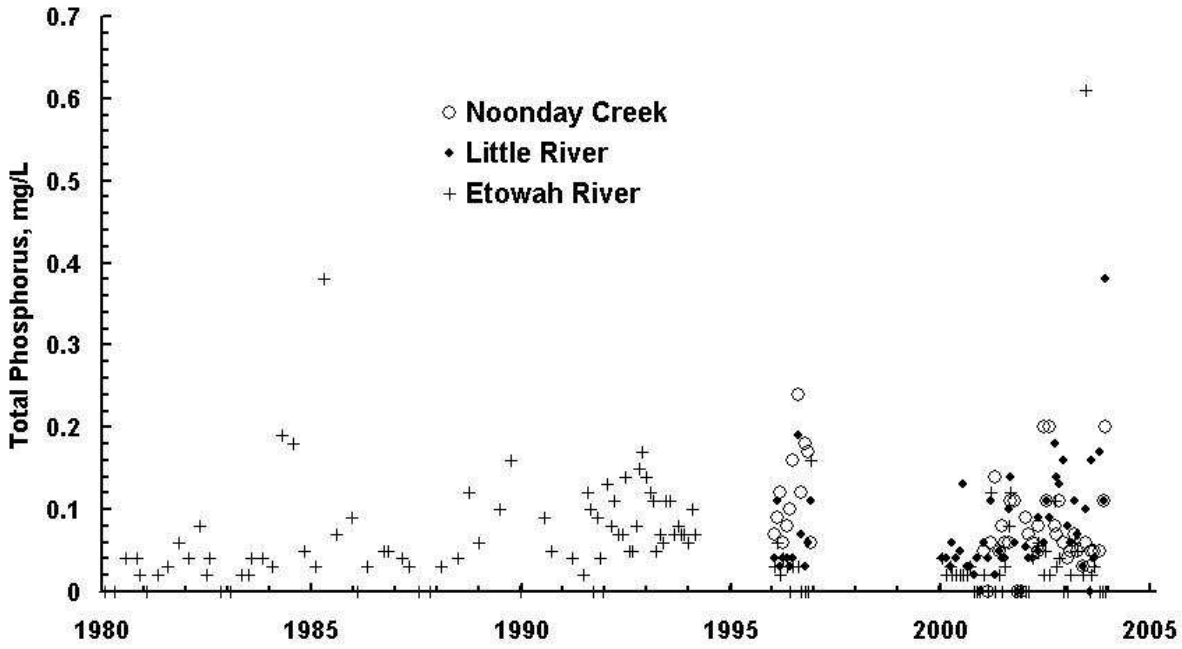

Figure 3.8: Plot of historical USGS total phosphorus time series for three Lake Allatoona tributaries.

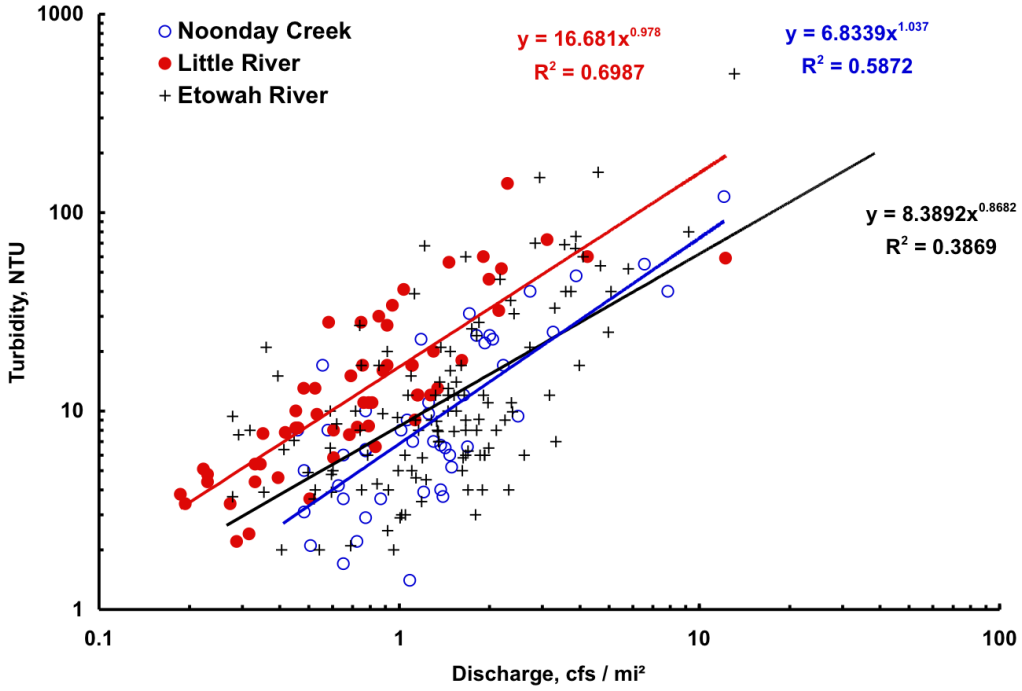

Figure 3.9: Plot of historical USGS discharge vs. turbidity for three Lake Allatoona tributaries. Note increasing trend in turbidity with runoff indicating effects of nonpoint source inputs during stormflow events.

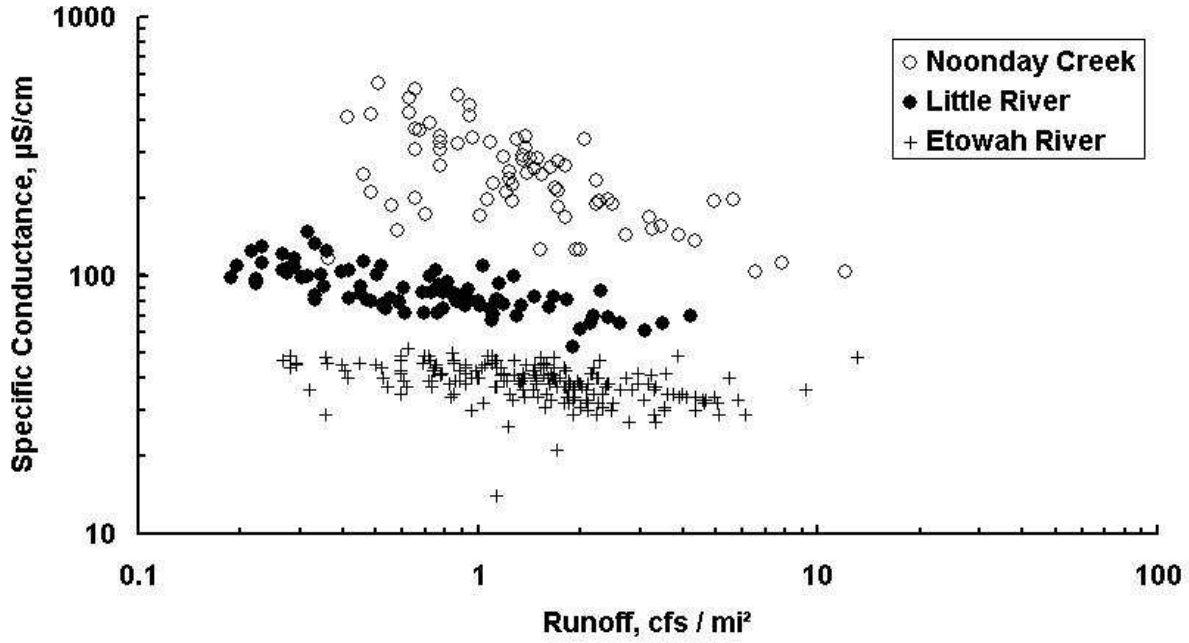

Figure 3.10: Plot of historical USGS discharge vs. specific conductance for three Lake Allatoona tributaries. Note decreasing trend in specific conductance with runoff indicating dilution of point source inputs during stormflow events.

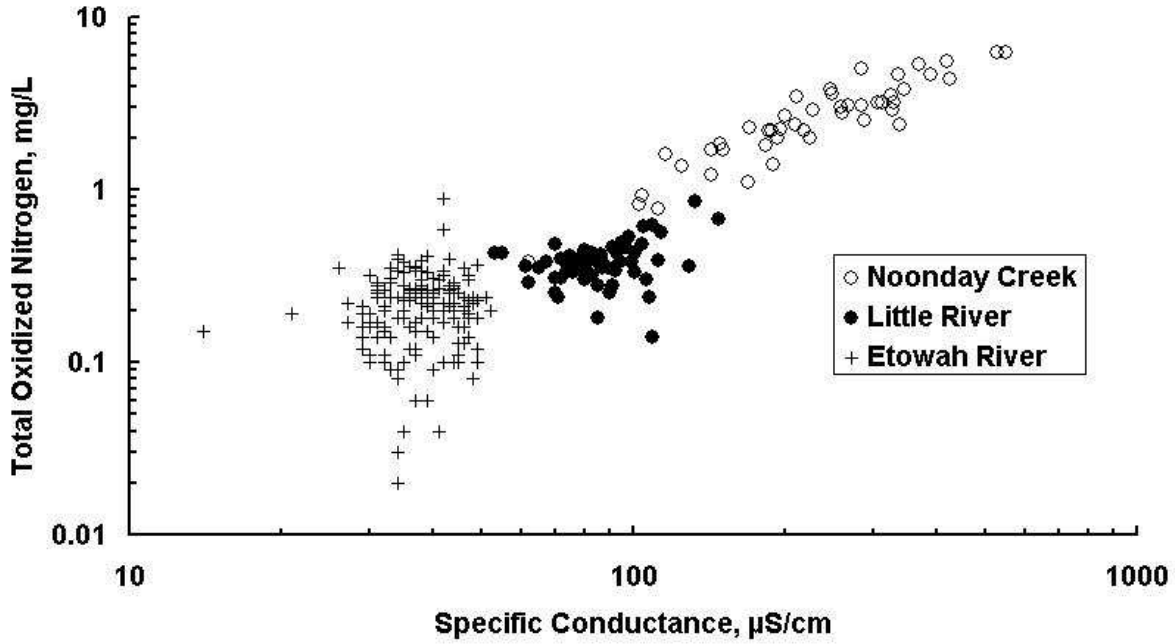

Figure 3.11: Plot of historical USGS specific conductance vs. nitrate concentration for three Lake Allatoona tributaries. Note increasing trend in nitrate concentration with specific conductance indicating point source origin.

Table 3.4: Upper Lake Allatoona tributary sediment and total phosphorus estimates using U.S. Geological Survey data. q is the water yield, cfs/mi^2. T is turbidity, NTU.

	Tributary		
	Etowah River (02392000)	Little River (02392780)	Noonday Creek (02393000)
Turbidity, NTU (r^2)	8.39 $q^{0.868}$ (0.387)	16.68 $q^{0.978}$ (0.699)	6.83 $q^{1.037}$ (0.587)
Mean Turbidity, NTU	21.3	35.6	14.1
Sediment Load, tonnes/day	64.6	24.5	2.86
Total Phosphorus, μg/L (r^2)	33.7 + 1.082 T (0.532)	43.8 + 1.207 T (0.454)	70.7 + 0.805 T (0.183)
Phosphorus Load, kg/day			
Soluble Fraction	102	30	14
Sediment Fraction	70	20	2
Total	172	60	15

These occasional higher values of turbidity and total phosphorus are likely due to storm-flows. Figure 3.9 show the positive correlations between stream discharge and turbidity. Figure 3.11 shows a strong positive relationship between specific conductance and nitrate, which is consistent with our supposition that nitrate loading is dominated by point sources.

Estimates of total annual loads of sediment and phosphorus can be obtained using the rating curve method (Holmbeck-Pelham and Rasmussen, 1997). The rating curve method uses the relationship between discharge and the desired parameters. The relationship between water yield, $q = Q/A$ (cfs/mi^2), and turbidity for the three sites is provided in Table 3.4. Note the similar relationships between Noonday Creek and Etowah River, and the larger value for Little River. The larger value for the Little River indicates that higher turbidities are observed at similar discharges.

The total sediment load can be calculated using the mean water yield, $\overline{q}$, from the Etowah River:

$$\overline{q} = \frac{\overline{Q}}{A} = \frac{1239 \text{ cfs}}{613 \text{ mi}^2} = 2.02 \text{ cfs/mi}^2$$

where $\overline{Q}$ is the mean discharge. Substituting this value into the sediment-discharge rating curves yields the flow-weighted average turbidity, $\overline{T}$. These relationships indicate the higher turbidity associated with Little River discharges.

The sediment load, T_{load} (tonnes/day), is then estimated using the product of the mean turbidity and the mean discharge, obtained using the average runoff rate for the ungaged streams:

$$T_{load} \text{ (tonnes/day)} = \overline{T} \text{ (NTU)} \times \overline{q} \ (cfs/mi^2) \times A \text{ (mi}^2\text{)}$$

where we must adjust for units $\times$ 28.3 (L/ft^3) $\times$ 10^{-9} (tonnes/mg) $\times$ 86,400 (s/day), and where we assume that 1 NTU $\approx$ 1 mg/L. This results in a load estimate shown in Table 3.4. This estimate is for the suspended (wash) load component of sediment transport, and neglects the bedload component of sediment transport. The actual load is also underestimated due to the inherent bias associated with the rating curve method. Thus, it is likely that greater sediment loads are actually present than indicated here. Daily (or more frequent) mean discharge data would be required to provide a less-biased estimate of the total sediment load.

The relationship between the total phosphorus, TP (mg/L), and sediment, T (NTU), concentrations can be established using a linear function between these two variables: The constant values (in units of mg/L) are independent of the sediment concentration, and can be used to estimate total soluble sources of phosphorus within the tributary. This assumes that the soluble form is associated with the sediment-free fraction. Note that the highest soluble phosphorus concentration is observed in Noonday Creek.

The turbidity-dependent values represents the phosphorus bound to the sediment. This assumes that the soluble fraction remains constant as the turbidity increases. The units for

this measure are in g/kg, or parts per thousand, which can be converted to units of pounds per ton by multiplying by two, yielding approximately 2.2, 2,4, and 1.6 pounds per ton for the three sites, respectively. Note that the highest sediment concentration is observed for the Little River.

An estimate of the total phosphorus load for both the soluble and sediment fractions can be obtained using the these observed relationships. For the soluble fraction, the average daily load is the product of the mean discharge and the turbidity-independent values. These estimates are provided in Table 3.4. For the sediment-bound fraction, the average daily load is the product of the sediment load and the turbidity-dependent values: Neglecting bedload transport is likely to further bias this estimate. The total load is estimated by summing the soluble and sediment fractions.

3.3.2 UGA Tributary Sampling Data

Water quality measurements were collected from Lake Allatoona tributaries on two sampling dates, August 9 and October 7, 2005 (Tables 3.5 - 3.7). Field water quality data are those collected using a Hydrolab Quanta (Table 3.5), along with laboratory data (Tables 3.6-3.7) for both dates from the UGA Soil, Plant, and Water laboratory within the Agricultural and Environmental Services Laboratory (AESL). Additional phosphorus data are provided for the second sampling date using the UGA Stable Isotope & Soil Biology Laboratory (SISBL) within the Institute of Ecology.

The total reactive phosphorus varied between sites; averaging 0.36 and 0.09 mg/L in the Etowah River and Little River, respectively. The amount of total phosphorus per unit sediment also varied slightly between sites. The smallest ratio (0.29 g/kg) was found in Etowah River, Little River having a higher ratio (0.92 g/kg), and Noonday Creek having the highest ratio (2.60 g/kg). Converting to English units of measure yields 0.58, 1.84, and 5.20 pounds phosphorus per ton of sediment for the three sites, respectively.

Table 3.5: Lake Allatoona tributary *in situ* water quality measurements.

Location	Etowah River				Little River			Noonday Creek	
	South	North	South	North	Center	South	North	West	East
Date	8/9/5	8/9/5	10/7/5	10/7/5	8/9/5	10/7/5	10/7/5	10/7/5	10/7/5
Temperature, °C	21.64	21.60	20.61	20.66	24.46	21.85	21.65	22.72	22.73
Conductivity, μS/cm	37	37	46	46	95	94	86	249	227
Turbidity, NTU	160	148	68.1	63.1	80	58.4	65.8	30.4	42.6
pH	6.40	6.56	6.92	6.94	6.80	7.17	7.22	7.43	7.35
DO, %	105.0	93.1	93.3	89.6	97.0	87.7	88.7	92.1	91.9
DO, mg/L	11.6	8.19	8.24	7.93	8.94	7.56	7.74	7.72	8.83

Nitrates were again highest in Noonday Creek and lowest in the Etowah River, with the Little River being intermediate. Specific conductance (SC), sodium, chloride, and hardness all follow these same trends, consistent with the magnitude of point source inputs on each tributary. Ammonium is low in all cases.

Table 3.6: Lake Allatoona tributary laboratory (AESL except for SISBL, †) water quality data collected on August 9, 2005.

	Etowah River		Little River	
	South	North	South	North
Sample	A3	A11	A8	A9
Hardness, mg/L	11.8	13.4	29.1	29.0
pH	7.10	6.50	7.06	7.11
TSS, mg/L	86	247	35	43
SO_4, mg/L	1.85	1.79	4.81	4.91
TKN, mg/L	1.86	1.02	0.74	0.88
NH_4, mg/L	0.03	0.04	0.05	0.07
NO_3, mg/L	0.24	0.24	0.55	0.56
NO_3(†), mg/L	0.22	0.21	0.52	0.51
TP, mg/L	<0.06	<0.06	0.066	<0.06
TP†, mg/L	0.08	0.10	0.08	0.43
PO_4, mg/L	<0.05	<0.05	<0.05	<0.05
PO_4†, mg/L	0.07	0.05	0.40	0.07
TRP, mg/L	0.24	0.48	0.08	0.10

Table 3.7: Lake Allatoona tributary laboratory (AESL except for SISBL, †) water quality data collected on October 7, 2005.

	Etowah River		Little River		Noonday Creek	
	South	North	South	North	West	East
Sample	A1	A2	A5	A10	A6	A7
Hardness, mg/L	14.7	14.7	25.8	27.1	70.7	65.93
pH	7.03	7.02	7.1	7.45	7.32	7.35
TDS, mg/L	50	68	90	66	170	140
TSS, mg/L	132	93	58	60	26	23
SO_4, mg/L	1.81	1.71	3.59	2.57	14.8	13.6
TKN, mg/L	0.91	0.88	<0.28	1.79	0.7	0.88
NH_4, mg/L	0.08	0.02	0.05	0.04	0.04	0.03
NO_3, mg/L	0.23	0.23	0.37	0.26	2.82	2.58
NO_3(†), mg/L	0.36	0.20	0.34	0.17	2.69	2.47
TP, mg/L	<0.06	<0.06	<0.06	<0.06	<0.06	<0.06
TP(†), mg/L	0.035	0.029	0.053	0.056	0.064	0.063
PO_4, mg/L	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
PO_4(†), mg/L	0.001	0.001	0.007	0.004	0.043	0.042

3.3.3 Comparison of Load Estimates

Table 3.8 summarizes the total phosphorus load (soluble plus sediment) for U.S. Geological Survey data, along with UGA data collected here. Also shown are estimates from the Lake Allatoona Phase I Diagnostic-Feasibility study report for 1992-1997 conducted by Kennesaw State University. Note that the Clean Lakes study estimates appear to be lower for all sites. Also shown for comparison are the regulatory limits established for these locations. Precise estimates are difficult to obtain, however, due to the paucity of stream discharge and contemporaneous nutrient data.

Table 3.8: Comparison of Lake Allatoona total phosphorus loads (soluble + sediment), kg/day.

Location	USGS Data	Clean Lakes Study	Limit
Etowah River	172	105	423
Little River	60	27.3	52.3
Noonday Creek	16	12.5	47.3

3.4 Lake Dredging

Dredging of lake sediments has been proposed as a mechanism for removing the source of internal loading. The dredging of sediments may reduce internal loading by removing the source of excess P. Small-scale dredging of Lake Allatoona has been conducted primarily for increasing navigability of the impoundment.

Blankenship Sand operates dredges to remove sediment (primarily sand) from North Georgia lakes and rivers, including Lake Allatoona and its tributaries (Figure 3.1). Dredging is conducted near the mouths of two tributaries, specifically near Know Bridge on the Etowah River and Rope Mill Park on the Little River. Table 3.9 summarizes recent sediment production from these locations. Sediment dredging involves a mobile, floating, diesel-powered suction dredge with a cutting head. Dredged materials are transported to the shore via a flexible pipe, or to a waiting clamshell barge that carries the sediments to an in-lake holding area that is re-dredged to the shoreline.

Dredged materials are separated into four parts:

1. A rubble fraction containing trash, large woody materials and stones;
2. A a sand fraction that is accumulated in sand piles, loaded using front-end loaders into road transport vehicles;
3. A tailings material that accumulates in holding (sedimentation) ponds; and
4. A tailwater that is returned to the lake.

The dredging currently used, while removing sand-sized particles from sediments, resuspends clay and silt-sized particles in the holding area and in tailwater return flows to the lake.

Samples of sediments dredged from Lake Allatoona were collected with the intention of characterizing their physical and chemical properties. The primary sediment physical properties are the particle size distribution, which is a measure of the relative proportion of sand, silt, and clay sized particles. The chemical properties include the nutrients (i.e., carbon,

Table 3.9: Sediment removal by Blankenship Sand, in tons.

Location	1997	1998	1999	2000	2001	2002	2003	2004 †	Average ‡
SANDS									
Knox Bridge	75,765	92,931	90,741	40,003	36,457	77,809	25,879	16,273	62,798
Little River	§	§	§	§	§	§	59,685	15,157	59,685
Blanton Bottoms	42,518	64,495	39,091	27,318	22,045	§	§	§	39,093
Gober	§	§	§	928	11,320	1,876	3,315	§	4,360
Goldkist	§	§	§	2,435	§	§	§	§	2,435
Keithburg	§	§	§	222	§	§	7,441	§	3,832
Total sand	118,283	157,426	129,832	70,906	69,822	79,685	96,320	31,430	103,182
TAILINGS									
Knox Bridge	13,370	16,400	16,013	7,059	6,434	13,731	4,567	2,872	11,082
Little River	§	§	§	§	§	§	10,533	2,675	10,533
Blanton Bottoms	7,503	11,381	6,898	4,821	3,890	§	§	§	6,899
Gober	§	§	§	164	1,998	331	585	§	770
Goldkist	§	§	§	430	§	§	§	§	430
Keithburg	§	§	§	39	§	§	1,313	§	676
Total Tailings	20,873	27,781	22,911	12,513	12,322	14,062	16,998	5,547	18,209
TOTAL									
Knox Bridge	89,135	109,331	106,754	47,062	42,891	91,540	30,446	19,145	73,880
Little River	§	§	§	§	§	§	70,218	17,832	70,218
Blanton Bottoms	50,021	75,876	45,989	32,139	25,935	§	§	§	45,992
Gober	§	§	§	1,092	13,318	2,207	3,900	§	5,129
Goldkist	§	§	§	2,865	§	§	§	§	2,865
Keithburg	§	§	§	261	§	§	8,754	§	4,508
Total sediment	139,156	185,207	152,743	83,419	82,144	93,747	113,318	36,977	121,391

† Partial year of operation.

‡ Average value is for full years of operation.

§ Dredge not operating at site.

nitrogen, and phosphorus), metals (i.e., iron, manganese, mercury, arsenic, and others), and other miscellaneous constituents (e.g., sulfur).

3.4.1 Sediment Physical Properties

The particle size distribution is used to provide a gross characterization of the sediment pool. Sediment particle sizes (sand, silt, and clay size fractions) are characterized, along with the

nutrient and mercury content of each fraction. The working hypothesis is that nutrients and toxic metals are sorbed to the clay fraction, which may be present as a surface coating on the larger fractions. Also, some nutrients (primarily P) sorb to iron oxide coatings.

The particle-size distributions for sediment samples collected from the Knox Bridge and Little River dredging operations are presented in Table 3.10. Sediments were sampled from both the sand pile as well as the tailings piles. Note that all samples from the sand piles were dominated by sand-size particles. Also note that the tailings sediment samples were more variable, and are dominated by loams (loams, sandy loams, silt loams, or loamy sands) with one finer sample (clay).

3.4.2 Sediment Nutrient Properties

Sediment nutrient properties for Knox Bridge sand and tailings piles are presented in Table 3.11. Nutrient properties for Little River sand and tailings piles are presented in Table 3.12. Tailwater nutrient properties from both locations are presented in Table 3.13.

Table 3.10: Lake Allatoona dredged sediment physical characteristics (AESL).

Knox Bridge sand pile

Date	7/15/05	8/9/05	10/8/05	10/8/05	10/8/05	10/8/05	10/8/05
Sample	KB-A	A21	A35	A37	A50	A52	B84
Sand, %	97	96	98	98	98	98	98
Silt, %	1	2	0	0	0	0	0
Clay, %	2	2	2	2	2	2	2
Soil Type	Sand	Sand	Sand	Sand	Sand	Sand	Sand

Knox Bridge tailings pile

Date	7/15/05	8/9/05	8/9/05	10/8/05	10/8/05	10/8/05	10/8/05	10/8/05
Sample	KB-B	A24	A25	A61	B95	A54	A55	A59
Sand, %	20	22	44	42	40	12	40	44
Silt, %	13	68	50	48	50	70	54	50
Clay, %	67	10	6	10	10	18	6	6
Soil Type	Clay	Silt Loam	Sandy Loam	Loam	Loam	Silt Loam	Silt Loam	Sandy Loam

Little River sand pile

Date	8/9/05	10/8/05	10/8/05	10/8/05
Sample	A36	A89	A92	A94
Sand, %	96	94	94	92
Silt, %	2	4	4	6
Clay, %	2	2	2	2
Soil Type	Sand	Sand	Sand	Sand

Little River tailings pile

Date	8/9/05	10/8/05	10/8/05	10/8/05	10/8/05	10/8/05
Sample	A31	A84	B96	A86	A82	A91
Sand, %	36	74	80	20	10.8	76
Silt, %	54	20	18	60	77.2	20
Clay, %	10	6	2	20	12	4
Soil Type	Silt Loam	Sandy Loam	Loamy Sand	Silt Loam	Silt Loam	Loamy Sand

Table 3.11: Lake Allatoona dredged sediment nutrient characteristics, Knox Bridge dredging operations (AESL).

Knox Bridge sand pile

Date	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A21	A35	A37	A50	A52	B84
LBC	51	46	44	46	40	53
pH, $CaCl_2$	5.15	5.2	5.4	5.6	5.3	5.4
pH	5.75	5.8	6.0	6.2	5.9	6.0
BS, %	76.31	NA	NA	NA	NA	NA
CEC, meq/100g	0.6673	NA	NA	NA	NA	NA
OM, %	0.38	0.64	0.76	0.4	0.5	0.7
C, ppm	1,162	1,247	720	680	1,752	1,775
S, ppm	21	<5	6.2	<5	<5	<5
N, ppm	87	61	151	102	179	88
TKN, ppm	NA	316	173	175	158	143
NH_4, ppm	1.5	2.23	2.77	2.98	2.88	2.62
NO_3, ppm	2	4.06	3.074	2.77	2.87	3.18
TP, ppm	NA	77.78	105.6	116.0	114.6	224.8
PO_4, ppm	2.74	3.86	3.31	4.97	1.93	4.14

Knox Bridge tailings pile

Date	8/9/2005	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A24	A25	A61	B95	A54	A55	A59
LBC	733	375	501	526	672	579	325
pH, $CaCl_2$	5.45	5.65	5.6	5.7	5.4	5.7	5.5
pH	6.05	6.25	6.2	6.3	6.0	6.3	6.1
BS, %	59.86	86.32	NA	NA	NA	NA	NA
CEC	4.565	5.756	NA	NA	NA	NA	NA
OM, %	4.96	6.96	6.44	6.8	5.7	7.84	3.02
C, ppm	18,430	23,890	25,000	26,210	19,370	29,540	10,980
S, ppm	386	464	5,459	4,991	5,185	1,139	2,171
N, ppm	1,047	1,195	1,107	1,224	1,025	1,249	454
TKN, ppm	NA	NA	1,434	1,460	1,240	1,489	605
NH_4, ppm	14	1	21.49	16.71	24.73	29.27	9.85
NO_3, ppm	1.5	1.5	4.27	3.82	3.26	3.18	3.27
TP, ppm	NA	NA	301.2	322.4	429.8	282.4	154.64
PO_4, ppm	7.18	6.79	12.41	11.45	11.72	13.38	11.72

Table 3.12: Lake Allatoona dredged sediment nutrient characteristics, Little River dredging operations (AESL).

Little River sand pile

Date	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A36	A89	A90	A92	A94
LBC	63	95	87	65	69
pH, $CaCl_2$	5.61	6.1	6.1	5.6	5.6
pH	6.21	6.7	6.7	6.2	6.2
BS, %	81.94	NA	NA	NA	NA
CEC	0.7606	NA	NA	NA	NA
OM, %	0.30	0.76	0.44	0.68	0.46
C, ppm	907	3,581	2,100	25,940	17,120
S, ppm	38	<5	767	5.4	<5
N, ppm	86	238	0216	1,145	134
TKN, ppm	NA	352	217	239	186
NH_4, ppm	0.5	10.12	9.46	8.54	2.52
NO_3, ppm	2	5.1	3.3	4.9	3.6
TP, ppm	NA	81.8	52.6	46.8	126.8
PO_4, ppm	1.6	12.6	11.5	6.4	6.5

Little River tailings pile

Date	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A31	A84	B96	A86	A82	A91
LBC	565	256	263	759	419	244
pH, $CaCl_2$	6.07	6.0	6.1	6.1	6.1	6.0
pH	6.67	6.6	6.7	6.7	6.7	6.6
BS, %	86.55	NA	NA	NA	NA	NA
CEC	5.294	NA	NA	NA	NA	NA
OM, %	5.04	4.02	2.54	7.56	6.66	5.84
C, ppm	21,380	11,160	7,272	30,310	18,020	12,050
S, ppm	489	737	<5	1,337	1,085	1,475
N, ppm	1,049	527	540	1,551	813	518
TKN, ppm	NA	730	522	1,774	1,226	505
NH_4, ppm	18.0	27.7	28.5	26.6	19.6	22.0
NO_3, ppm	1.0	3.1	2.6	4.2	2.6	3.7
TP, ppm	NA	NA	103	392	235	111
PO_4, ppm	11.3	17.1	18.2	16.3	13.4	12.1

Table 3.13: Lake Allatoona dredged sediment nutrient characteristics, tailwater from Knox Bridge and Little River dredging operations (AESL except for SISBL, †).

Dredging Tailwaters

Location	Little River		Knox Bridge	
Date	8/9/2005	8/9/2005	10/8/2005	10/8/2005
Sample	A4	A12	A40	A41
Hardness, mg/L	31.32	30.73	26.28	26.47
pH	6.90	7.03	7.33	7.22
TDS, mg/L	NA	NA	126	100
TSS, mg/L	1,178	1,116	50	49
SO_4, mg/L	4.17	4.11	7.39	7.41
TKN, mg/L	2.42	2.70	0.67	1.86
NH_4, mg/L	0.14	0.14	0.06	0.07
NO_3, mg/L	0.49	0.49	0.06	0.05
NO_3(†), mg/L	0.52	0.52	0.04	0.05
TP, mg/L	<0.06	<0.06	<0.06	<0.06
TP(†), mg/L	0.120	0.100	0.055	0.059
PO_4, mg/L	<0.05	<0.05	<0.05	<0.05
PO_4(†), mg/L	0.080	0.080	0.001	0.002
TRP, mg/L	0.46	0.18	NA	NA

Note that the pH of the sediment samples is slightly acidic (5.7-6.7) while the tailwaters are more neutral (6.9-7.3). Also note that the Base Saturation (BS) of the sediments ranges from approximately 60-87%, which indicates that the base cations (Ca^{2+}, Mg^{2+}, etc.) dominated over the acid cations (H^+, Na^+, etc.). Soils with higher Base Saturations are more productive in an agronomic setting. As expected, the Cation Exchange Capacity (CEC) was higher for the tailings (4.6-5.8) than for the sands (0.67-0.76), which was consistent with the finer texture of the tailings. Soils with high CEC are also more productive.

The organic matter (OM) composition is lower in the sands (0.3-0.8%) than the tailings (3.5-7.8%). Carbon (C) composition is also lower in the sands (0.07-0.36%) than the tailings

(1.1-7.6%). The (N) nitrogen composition of the sands was low (0.006-0.11%), while the tailings were generally higher (0.05-0.12%).

The dredged sand materials had measurable quantities of nutrients (e.g., NH_4, NO_3, and PO_4). The ammonium (NH_4) concentration in the sands ranged from 0.5 to 10.1 ppm, with higher concentrations in samples with more organic matter. The total phosphorus (TP) concentration in the sands ranged from 46.76 to 224.8 ppm, but was less well-correlated with the organic matter content.

Like the BS, CEC, OM, C, and N concentrations, nutrients were higher in the tailings than in the sands, which was readily explained by the more abundant organic matter in these materials. NH_4 concentrations ranged from 1 to 29.3 ppm and TP ranges from 103 to 430 ppm. Again, while increased ammonium concentrations tracked organic matter concentrations, this trend was not observed with total phosphorus.

Tailwater nutrient concentrations were generally low; $NH_4 \leq 0.14$ mg/L, $NO_3 \leq 0.48$ mg/L, $PO_4 \leq 0.08$ mg/L, and TP ≤ 0.12 mg/L. This was in spite of the fact that of high sediment content within the tailwaters, with total suspended sediment concentrations exceeding 1000 mg/L on occasion (August 9, 2005 at the Little River dredging operation).

3.4.3 Lake Sediment Properties

Sediment data collected from Noonday Creek Embayment are shown in Table 3.14. In addition to these data, a vertical sediment profile of lake sediments was collected from the Etowah Embayment at the Knox Bridge Boat Ramp on Highway 20 (Table 3.15). Of all the water quality parameters measured, only one variable (NH_4) showed any variation with depth (Figure 3.12). The increased ammonium concentration with depth ($R = 0.83$) wss likely due to changes in oxidation. Total phosphorus concentrations were highly correlated with iron ($R = 0.98$) (Figure 3.13), while total nitrogen was correlated with organic matter ($R = 0.97$) (Figure 3.14). Nitrate did not correlate with other variables ($R < 0.30$), the best correlation being with the percent clay fraction ($R = 0.29$).

Table 3.14: Lake Allatoona sediment characteristics, benthic sediments from Noonday Creek embayment collected on 10/8/2005 (AESL).

Sample	A80	B98	A81	B99	A87	B97	A85	A88
Sand, %	50.0	42.0	NA	NA	38.0	40.0	56.0	94.0
Silt, %	38.0	50.0	NA	NA	46.0	44.0	34.0	4.0
Clay, %	12.0	8.0	NA	NA	16.0	16.0	10.0	2.0
Soil Type	Loam	Loam	NA	NA	Loam	Loam	Sandy Loam	Sand
LBC	534.0	494.0	418.0	429.0	517.0	591.0	662.0	503.0
pH, $CaCl_2$	5.8	5.9	5.3	5.4	6.7	6.6	6.9	6.7
pH	6.4	6.5	5.9	6.0	7.3	7.2	7.5	7.3
C, %	3.321	2.834	2.298	2.579	2.424	2.631	2.983	1.904
OM, %	10.2	8.62	9.78	9.34	6.04	7.98	7.04	6.28
S, ppm	1,844	174	803	1,134	787	1,339	254	697
N, ppm	1,986	1,605	1,257	1,630	1,438	1,484	1,578	950
TKN, ppm	2,158	2,358	2,300	2,372	1,496	1,807	1,716	1,156
NH_4, ppm	134	152	30	58	33	46	73	28
NO_3, ppm	3.10	3.06	3.39	3.88	3.46	3.25	4.28	2.78
P, ppm	14.34	16.55	17.1	15.17	20.41	20.13	21.1	20.41
TP, ppm	340.4	360	220.2	212.6	404.6	480.2	464.8	311.6
Ca, ppm	1,760	1,611	1,288	1,599	2,106	2,360	2,449	1,904
K, ppm	181.3	166.4	105.5	122.8	159.6	178.3	184.4	157.6
Mg, ppm	168.9	160.8	105.1	129.6	168	189.7	206.5	151.5
Mn, ppm	502.6	525.7	229.1	297.9	1085	1001	1994	989.3
Zn, ppm	39.53	38.68	24.11	29.25	44.98	47.32	49.58	33.77

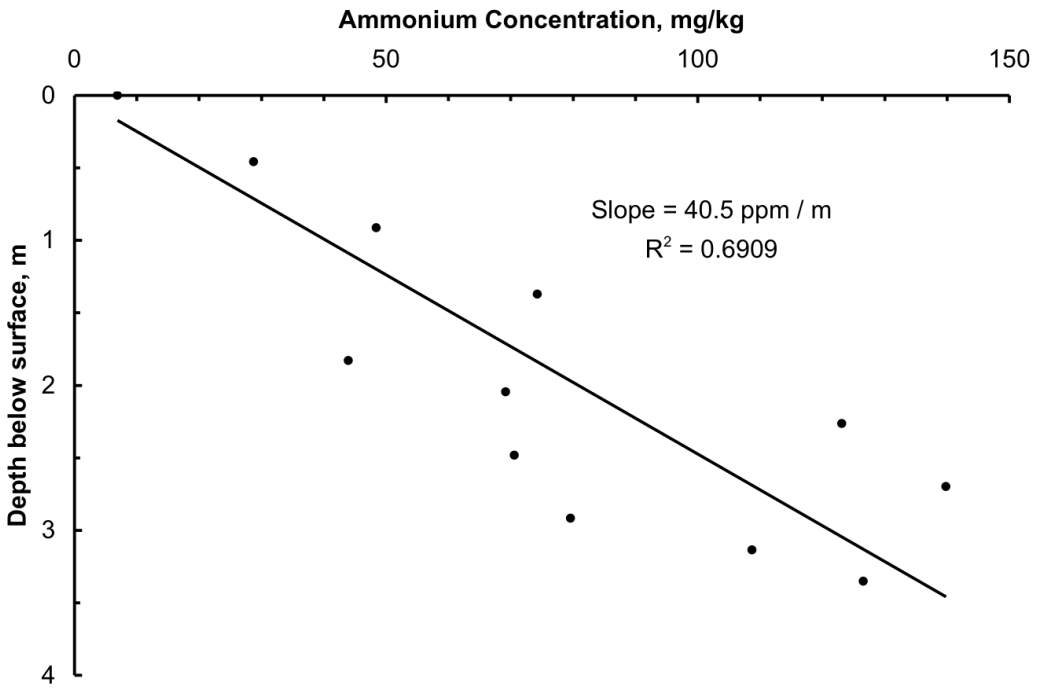

Figure 3.12: Etowah Embayment sediment samples; depth vs. ammonium content

Table 3.15: Sediment Profile (Knox Bridge Boat Ramp): Physical properties and nutrients. Results for total sediment concentration indicated with the prefix *T*-.

Depth	Sand	Silt	Clay	Soil Type	CEC	LBC	pH	pH	BS
m	%	%	%		meq/100g	ppm/pH	$CaCl_2$	equiv	%
0.00	36	46	18	Loam	5.79	648	4.98	5.58	61.5
0.46	38	46	16	Loam	4.93	704	4.69	5.29	42.6
0.91	36	44	20	Loam	5.59	770	4.99	5.59	52.9
1.37	24	58	18	Silt Loam	5.72	725	5.02	5.62	57.4
1.83	64	28	8	Sandy Loam	3.90	513	4.90	5.50	52.7
2.05	54	34	12	Sandy Loam	4.21	548	5.20	5.80	60.9
2.26	44	40	16	Loam	4.79	616	5.45	6.05	67.9
2.48	74	18	8	Sandy Loam	2.74	347	5.36	5.96	66.1
2.70	50	36	14	Loam	5.71	666	5.34	5.94	68.3
2.92	64	26	10	Sandy Loam	3.56	425	5.43	6.03	69.7
3.14	54	34	12	Sandy Loam	5.22	587	5.46	6.06	72.1
3.35	46	42	12	Loam	6.76	648	6.68	7.28	99.6

Depth	T-C	OM	T-S	S	T-P	P	T-N	NH_4-N	NO_3-N
m	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm
0.00	23,130	61,400	248.6	429.8	328.2	8.30	1518	6.9	4.0
0.46	23,650	58,800	393.6	530.3	354.4	9.48	1634	28.8	1.0
0.91	25,560	65,000	265.8	439.6	374.8	11.26	1847	48.5	2.0
1.37	33,760	77,600	366.6	665.3	409.4	10.18	2424	74.3	1.0
1.83	13,810	30,800	158.0	381.1	208.0	9.44	706	43.9	2.0
2.05	21,180	50,800	214.2	307.2	262.2	10.02	1201	69.2	1.0
2.26	20,810	63,200	217.6	410.8	299.0	10.66	1532	123.1	1.5
2.48	10,920	27,200	114.0	213.7	170.5	10.15	612	70.6	1.5
2.70	42,720	89,400	346.8	519.3	317.6	10.70	2594	139.8	1.5
2.92	17,480	42,800	168.2	42.3	215.6	10.18	1006	79.6	1.0
3.14	31,940	68,600	275.4	347.0	241.8	11.77	1762	108.7	1.5
3.35	27,430	58,400	277.6	427.7	344.8	11.04	1863	126.6	1.5

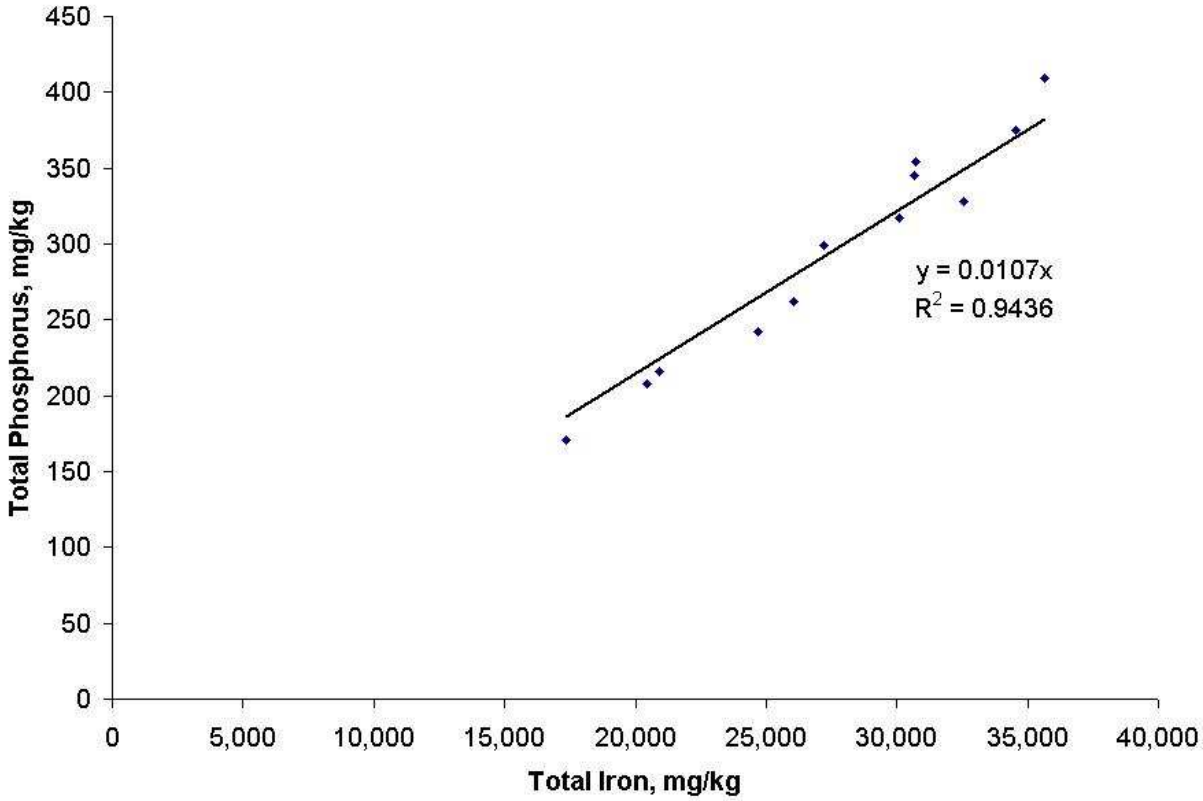

Figure 3.13: Etowah Embayment sediment samples; total iron vs. total phosphorus

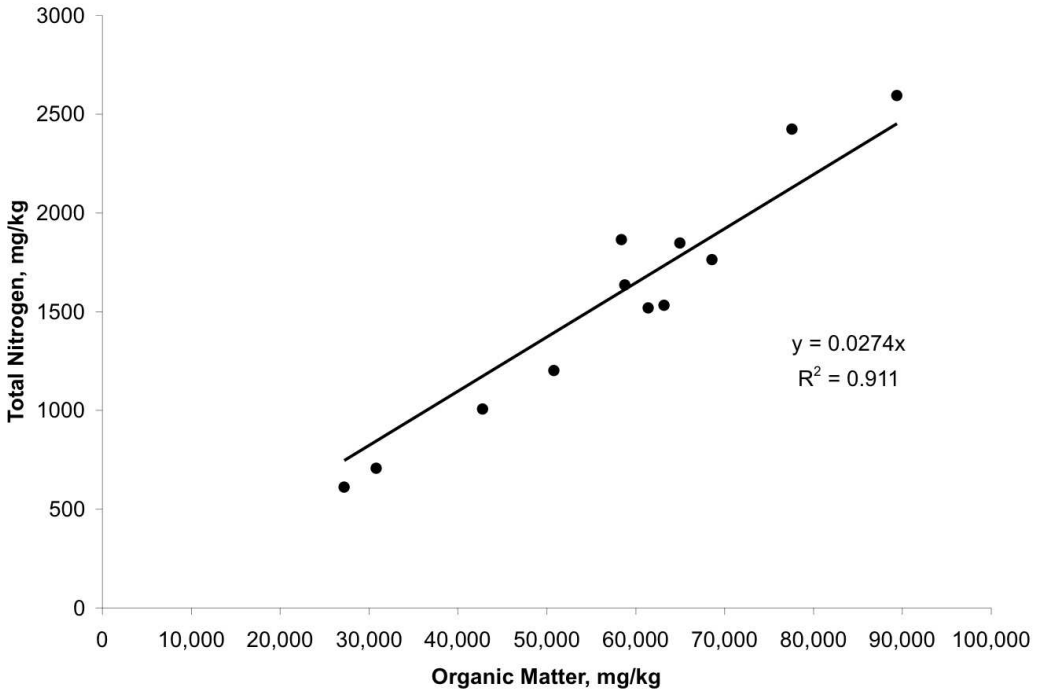

Figure 3.14: Etowah Embayment sediment samples; organic matter vs. total nitrogen

3.4.4 Sediment Metals Properties

Metals concentrations are of interest for two reasons; i) certain metals - primarily iron - have been shown to control phosphorus availability, and ii) certain metals - such as lead and mercury - are known to be toxic at elevated levels. Data from sediment and water samples were analyzed for metals so that these two factors could be examined.

Tributary metals concentrations for two dates are shown in Tables 3.16 and 3.17. Sediment metals concentrations from samples collected in Noonday Creek embayment are provided in Table 3.18. Etowah River embayment sediment data are provided in Tables 3.19 and 3.20 for major and minor metals, respectively. Tables 3.21 and 3.22 present the sediment metal properties for the Knox Bridge and Little River dredged sediments, respectively.

Toxic metal concentrations in Lake Allatoona sediments were low (Table 3.23). All reported samples for toxic metals lie well below regulatory limits.

The total phosphorus to total iron ratio is slightly less variable, with a ratio of 75 g/kg in the Etowah River, 108 g/kg in the Little River, and 266 g/kg in Noonday Creek. The total iron per unit sediment varies even less, with 4.0 g/kg in the Etowah River, 8.6 g/kg in the Little River, and 9.8 g/kg for Noonday Creek. This indicates that Noonday Creek is enriched in iron (per unit sediment) as well as total phosphorus (both per unit iron and per unit sediment).

The form of iron present in sediments also varied. The acid oxalate extraction dissolves X-ray amorphous iron, while the dithionite extraction dissolves crystalline iron (McKeague and Day, 1966). In general, the likely form that would most readily sorb phosphorus is the amorphous (oxalate) form. Note that the tailings generally had a greater quantity of the amorphous form of iron, while the sand product was dominated by the crystalline form (Table 3.24).

Table 3.16: Lake Allatoona tributary metals concentrations collected on August 9, 2005 (AESL).

	Etowah River		Little River	
	South	North	South	North
Sample	A3	A11	A8	A9
Al, mg/L	1.902	2.700	0.653	0.637
B, mg/L	0.011	0.011	0.017	0.015
Ca, mg/L	3.140	3.594	7.225	7.197
Cd, mg/L	<0.005	<0.005	<0.005	<0.005
Cl, mg/L	3.090	6.880	6.800	7.090
Cr, mg/L	<0.005	<0.005	<0.005	<0.005
Cu, mg/L	<0.005	0.005	0.007	0.015
F, mg/L	0.030	0.030	0.070	0.070
Fe, mg/L	1.300	2.043	1.000	0.980
K, mg/L	1.587	1.67	2.505	2.443
Mg, mg/L	0.976	1.077	2.692	2.693
Mn, mg/L	0.0226	0.186	0.071	0.072
Mo, mg/L	<0.005	<0.005	<0.005	<0.005
Na, mg/L	1.670	1.796	5.004	4.927
Ni, mg/L	<0.01	<0.01	<0.01	<0.01
Si, mg/L	5.722	6.679	7.342	7.339
Zn, mg/L	0.026	0.084	0.028	0.029

Table 3.17: Lake Allatoona tributary metals concentrations collected on October 7, 2005 (AESL).

	Etowah River		Little River		Noonday Creek	
	South	North	South	North	West	East
Sample	A1	A2	A5	A10	A6	A7
Al, mg/L	0.219	0.322	0.246	0.239	<0.025	<0.025
B, mg/L	<0.01	<0.01	0.016	<0.01	0.055	0.049
Ca, mg/L	3.978	3.983	6.351	6.953	23.01	21.37
Cd, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Cl, mg/L	1.85	1.79	5.62	3.79	15.24	14.06
Cr, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Cu, mg/L	<0.005	<0.005	<0.005	<0.005	0.0073	<0.005
F, mg/L	0.05	0.05	0.10	0.09	0.23	0.20
Fe, mg/L	0.391	0.469	0.533	0.481	0.239	0.239
K, mg/L	1.508	1.574	2.868	2.672	4.581	4.339
Mg, mg/L	1.174	1.165	2.420	2.376	3.245	3.091
Mn, mg/L	<0.005	<0.005	0.010	0.010	0.006	<0.005
Mo, mg/L	0.006	<0.005	<0.005	<0.005	<0.005	<0.005
Na, mg/L	2.115	2.127	5.179	4.299	16.71	15.52
Ni, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Si, mg/L	4.820	4.931	6.243	6.598	5.890	5.244
Zn, mg/L	<0.005	0.007	0.014	0.008	0.012	0.018

Table 3.18: Lake Allatoona sediment characteristics, benthic sediments from Noonday Creek embayment collected on 10/8/2005 (AESL). Analysis using double acid (Mehlich) extraction.

Al, ppm	11.73	11.43	3.65	3.77
B, ppm	0.015	0.015	<0.01	<0.01
Ca, ppm	7.94	7.77	7.31	7.40
Cd, ppm	<0.005	<0.005	<0.005	<0.005
Cl, ppm	9.05	6.59	2.85	3.08
Cr, ppm	0.009	0.009	<0.005	<0.005
Cu, ppm	<0.005	<0.005	<0.005	<0.005
F, ppm	0.08	0.08	0.07	0.08
Fe, ppm	13.45	13.22	2.64	2.74
K, ppm	2.944	2.985	1.657	1.657
Mg, ppm	2.804	2.764	1.964	1.956
Mn, ppm	3.204	2.968	0.012	0.012
Mo, ppm	<0.005	<0.005	<0.005	<0.005
Na, ppm	4.808	4.812	1.517	1.500
Ni, ppm	<0.01	<0.01	<0.01	<0.01
Si, ppm	19.41	19.30	5.601	5.531
Zn, ppm	0.041	0.050	0.019	0.016

Table 3.19: Sediment profile for Etowah River Embayment (Knox Bridge Boat Ramp), major metals (AESL). Analysis for total sediment concentration (indicated with the prefix *T*-) performed using total acid digestion, others using double acid (Mehlich) extraction.

Depth	T-Al	T-Fe	T-Mg	T-K	T-Ca	T-Mn	T-Na	T-Zn	T-Cr
m	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm
0.00	33,620	32,540	3578	3346	1124.8	361.0	98.24	78.14	30.24
0.46	38,020	30,700	3960	3652	1014.4	372.4	105.08	81.50	33.36
0.91	34,100	34,540	3866	3322	1273.4	485.2	106.00	83.36	33.36
1.37	36,760	35,660	4038	3536	1431.0	561.8	111.00	94.36	33.04
1.83	21,320	20,420	2348	2060	698.8	266.2	61.52	51.10	21.20
2.05	27,100	26,080	2762	2494	1060.4	389.8	65.06	65.46	32.42
2.26	25,580	27,220	2766	2372	1097.0	436.8	72.44	76.08	26.18
2.48	19,260	17,328	1984	1808	717.4	235.0	60.24	47.36	23.38
2.70	30,460	30,100	3392	2966	1672.4	490.0	62.68	133.52	30.86
2.92	22,720	20,900	2400	2208	938.0	291.0	104.42	64.14	31.48
3.14	27,120	24,700	3132	2768	1628.0	313.0	59.92	109.84	25.88
3.35	38,320	30,680	3206	3154	3026.0	445.6	99.34	192.82	31.48

Depth	Al	Fe	Mg	K	Ca	Mn	Na	Zn	Cr
m	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm
0.00	485.2	362.8	59.70	58.70	548.5	42.70	39.68	5.58	0.105
0.46	489.8	674.0	24.36	49.78	327.6	75.80	30.63	6.47	0.172
0.91	490.6	758.0	30.51	57.90	469.2	52.35	48.66	6.45	0.220
1.37	455.3	776.5	48.66	50.05	519.5	55.90	35.60	10.97	0.163
1.83	359.5	591.5	27.69	39.18	317.0	37.69	32.35	5.85	0.163
2.05	444.4	666.5	46.73	43.63	388.5	38.70	27.02	8.30	0.191
2.26	529.3	665.5	65.95	62.40	479.0	49.81	33.48	11.86	0.220
2.48	312.1	482.9	35.19	33.40	263.6	27.24	26.76	7.47	0.144
2.70	530.1	638.5	68.75	53.95	603.0	38.82	39.35	30.55	0.172
2.92	402.0	558.5	49.38	44.12	365.4	37.63	29.82	10.37	0.172
3.14	457.0	580.0	50.90	45.19	619.5	21.31	29.77	22.37	0.163
3.35	435.1	653.0	59.10	46.97	1197.0	72.45	30.65	45.93	0.220

Table 3.20: Sediment Profile for Etowah River Embayment (Knox Bridge Boat Ramp): Minor metals. Analysis for total sediment concentration (indicated with the prefix *T*-) performed using total acid digestion, others using double acid (Mehlich) extraction.

Depth	T-Ni	T-Cd	T-Pb	T-B	T-Cu	T-Mo
m	ppm	ppm	ppm	ppm	ppm	ppm
0.00	10.70	5.80	<5	8.36	<1	10.76
0.46	11.02	6.38	<5	4.34	<1	<1
0.91	13.62	6.96	<5	3.22	<1	<1
1.37	15.90	6.08	<5	3.38	<1	<1
1.83	5.52	3.18	<5	<2	<1	<1
2.05	12.32	5.22	<5	<2	<1	<1
2.26	7.14	5.80	<5	<2	<1	<1
2.48	5.84	1.74	<5	<2	<1	<1
2.70	10.38	4.34	<5	<2	<1	<1
2.92	10.70	4.06	<5	<2	<1	<1
3.14	7.78	3.76	<5	<2	<1	<1
3.35	10.70	6.08	<5	2.08	<1	<1

Depth	Ni	Cd	Pb	B	Cu	Mo
m	ppm	ppm	ppm	ppm	ppm	ppm
0.00	0.294	0.092	1.024	0.069	3.36	< 0.02
0.46	0.441	0.165	1.185	< 0.024	2.93	< 0.02
0.91	0.294	0.174	1.616	0.069	3.34	< 0.02
1.37	0.608	0.220	1.401	< 0.024	3.56	< 0.02
1.83	0.284	0.119	1.293	< 0.024	2.50	< 0.02
2.05	0.314	0.165	1.482	< 0.024	3.12	< 0.02
2.26	0.343	0.183	1.563	< 0.024	3.63	< 0.02
2.48	0.127	0.137	1.078	< 0.024	2.22	< 0.02
2.70	0.431	0.183	1.482	0.074	2.84	< 0.02
2.92	0.284	0.165	1.482	0.082	2.70	< 0.02
3.14	0.284	0.174	1.913	0.057	2.54	< 0.02
3.35	0.196	0.183	1.482	< 0.024	2.56	< 0.02

Table 3.21: Lake Allatoona dredged sediment metals characteristics, Knox Bridge dredging operations (AESL).

Knox Bridge sand pile

Date	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A21	A35	A37	A50	A52	B84
Al, ppm	38.06	NA	NA	NA	NA	NA
B, ppm	0.261	NA	NA	NA	NA	NA
Ca, ppm	67	131.2	99.44	123.7	70.47	122.8
Cd, ppm	0.050	NA	NA	NA	NA	NA
Cr, ppm	0.103	NA	NA	NA	NA	NA
Cu, ppm	0.401	NA	NA	NA	NA	NA
K, ppm	7.395	18.73	16.47	17.88	15.15	22.49
Fe, ppm	64.3	NA	NA	NA	NA	NA
Mg, ppm	6.09	19.17	16.46	18.88	14.38	18.81
Mn, ppm	8.22	21.26	21.47	29.40	15.79	24.63
Mo, ppm	0.027	NA	NA	NA	NA	NA
Na, ppm	24.0	NA	NA	NA	NA	NA
Ni, ppm	0.105	NA	NA	NA	NA	NA
Pb, ppm	0.474	NA	NA	NA	NA	NA
Zn, ppm	4.98	6.12	5.42	6.35	5.65	5.88

Knox Bridge tailings pile

Date	8/9/2005	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A24	A25	A61	B95	A54	A55	A59
Al, ppm	196.4	182.25	NA	NA	NA	NA	NA
B, ppm	0.161	0.184	NA	NA	NA	NA	NA
Ca, ppm	401	807	2,119	2,267	1,475	2,058	908
Cd, ppm	0.126	0.157	NA	NA	NA	NA	NA
Cr, ppm	0.200	0.172	NA	NA	NA	NA	NA
Cu, ppm	1.760	2.706	NA	NA	NA	NA	NA
K, ppm	31.5	40.6	116.6	116.8	161.4	132.4	94.2
Fe, ppm	351.9	408.5	NA	NA	NA	NA	NA
Mg, ppm	61.0	83.8	185.7	200.2	206.1	205.3	125.5
Mn, ppm	108.5	77.7	211.5	215.8	274.7	264.6	167.3
Mo, ppm	<.02	<.02	NA	NA	NA	NA	NA
Na, ppm	31.71	30.87	NA	NA	NA	NA	NA
Ni, ppm	0.335	0.414	NA	NA	NA	NA	NA
Pb, ppm	0.931	0.861	NA	NA	NA	NA	NA
Zn, ppm	6.7	15.3	32.6	34.7	17.4	25.4	11.5

Table 3.22: Lake Allatoona dredged sediment metals characteristics, Little River dredging operations (AESL).

Little River sand pile

Date	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A36	A89	A90	A92	A94
Al, ppm	27.245	NA	NA	NA	NA
B, ppm	0.1807	NA	NA	NA	NA
Ca, ppm	80.3	382.4	272.3	247.8	307.0
Cd, ppm	0.0503	NA	NA	NA	NA
Cr, ppm	0.0483	NA	NA	NA	NA
Cu, ppm	0.231	NA	NA	NA	NA
K, ppm	10.02	46.78	38.49	36.71	36.24
Fe, ppm	67.55	NA	NA	NA	NA
Mg, ppm	10.7	48.0	38.0	36.7	41.9
Mn, ppm	43.6	308.7	250.1	174.1	148.1
Mo, ppm	<0.02	NA	NA	NA	NA
Na, ppm	24.6	NA	NA	NA	NA
Ni, ppm	0.079	NA	NA	NA	NA
Pb, ppm	0.123	NA	NA	NA	NA
Zn, ppm	1.99	6.59	5.42	3.47	3.70

Little River tailings pile

Date	8/9/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005	10/8/2005
Sample	A31	A84	B96	A86	A82	A91
Al, ppm	187.7	NA	NA	NA	NA	NA
B, ppm	0.057	NA	NA	NA	NA	NA
Ca, ppm	716	827	737	1,721	1,433	1,071
Cd, ppm	0.258	NA	NA	NA	NA	NA
Cr, ppm	0.221	NA	NA	NA	NA	NA
Cu, ppm	2.131	NA	NA	NA	NA	NA
K, ppm	53.3	91.8	90.9	130.9	119.8	82.9
Fe, ppm	750	NA	NA	NA	NA	NA
Mg, ppm	84.1	115.8	101.4	227.7	199.6	112.4
Mn, ppm	1,153	910	845	1,635	1,492	828
Mo, ppm	<.02	NA	NA	NA	NA	NA
Na, ppm	38.01	NA	NA	NA	NA	NA
Ni, ppm	0.631	NA	NA	NA	NA	NA
Pb, ppm	1.265	NA	NA	NA	NA	NA
Zn, ppm	9.67	10.32	9.16	17.88	13.75	6.90

Table 3.23: Lake Allatoona sediment metals content. Samples analyzed by University of Georgia Laboratory for Environmental Analysis. All units are ppm.

Location	KB sand	KB silt	LR sand	LR silt	GAEPD	Lake Lanier	
						Avg	Max
Cr	28.34	32.83	10.67	41.01	100	30	73
As	1.07	1.87	0.45	1.82	20	<4	16
Se	0.17	0.32	0.06	0.63	2	<0.2	1
Ag	0.02	0.08	0.01	0.07	2	<0.2	0.3
Cd	0.02	0.12	0.01	0.09	2	<0.5	0.5
Ba	15	129	35	308	1,000	125	330
Hg	0.070	0.180	0.000	0.060	0.500	0.032	0.300
Pb	2.85	13.75	2.45	15.06	75	15	48

Table 3.24: Iron extracts for Lake Allatoona samples, i) Acid Oxalate, Fe_o, and ii) Citrate Dithionite, Fe_d. Analysis performed in the UGA Crop and Soil Science Soil Characterization Laboratory by Vicki Hufstetler, (706)542-0910, under the direction of Professor Larry West).

Location	ID	Extractable Iron, %		Ratio
		Fe_o	Fe_d	(Fe_o/Fe_d)
KB tailings	A22	1.3	1.6	0.83
KB tailings	A23	0.9	1.2	0.75
KB Sand	A26	0.2	0.5	0.31
LR Sand	A30	0.1	0.4	0.30
LR tailings	A32	1.2	2.0	0.60

3.5 Summary

The sediment load to Lake Allatoona is substantial. Sediment inputs from three tributaries studied here were estimated using U.S. Geological Survey suspended sediment concentration and discharge data to vary from approximately 1,000 tonnes per year from the Noonday Creek watershed, to 9,000 tonnes from the Little River watershed, to over 23,000 tonnes from the Etowah River basin. These enormous sediment loads settle to the bottom of the reservoir, primarily near the discharge points of these tributaries into the reservoir. These

are conservative estimates of the sediment load because they do not account for bedload transport. Bedload transport of sand-sized and larger sediments is expected to be a significant - yet unmeasured - input to the reservoir.

A crude estimate of the time required to fill the reservoir can be calculated by assuming:

- The total sediment loading to the reservoir is limited to the inputs from the Etowah River, Little River, and Noonday Creek, totaling 33,000 tonnes per year;
- The sediment density in the lake is 1.2 tonnes per cubic meter, yielding a sediment input of approximately 27,500 m^3/yr; and
- The total storage volume is 367,500 acre feet, equivalent to approximately 454 million cubic meters.

The resulting time to fill the lake with sediment is approximately 16,500 years. Yet this estimate grossly underestimates the total sediment inputs by neglecting bedload transport as well as other lake tributaries. In addition, the sediments will accumulate in the shallower zones nearer to tributary inputs, thus adversely affecting shallower zones than near the dam.

Large quantities of nutrients are contained within tributary sediments. Using U.S. Geological Survey water quality data, the total phosphorus concentration within tributary suspended sediments was estimated to be 1.1, 1.2, and 0.8 g/kg for the Etowah River, Little River and Noonday Creek. These values are equivalent to 2.2, 2.4, and 1.6 pounds per ton, respectively. Corresponding estimates for stormwater samples collected for this project on the Etowah River (2.6 g/kg) and the Little River (1.8 g/kg) were slightly higher.

The total phosphorus load to Lake Allatoona was estimated using U.S. Geologic Survey water quality and discharge data to be approximately 5.84 tonnes per year from Noonday Creek, 21.9 tonnes per year from the Little River, and 62.8 tonnes per year from the Etowah River basin. These estimates are larger than - but generally consistent with - estimates obtained independently by the Lake Allatoona Phase I Diagnostic Feasibility Study conducted by Kennesaw State University between 1992 and 1997 as part of an EPA Clean Lakes

Study. The total phosphorus load estimated in the Clean Lakes Study yielded an estimate of 4.6 tonnes per year from Noonday Creek, 10 tonnes per year from the Little River, and 38.4 tonnes per year from the Etowah River Basin. Discrepancies between these estimates can be attributed to differences in the data and methods of analysis used. Additional studies using in-lake sediment accumulation surveys are needed to independently corroborate these estimates.

Lake Allatoona sediments are removed by commercial dredging operations by Blankenship Sand. These dredging operations focus on the separation and removal of the sand-size fraction. Dredging operations at the Knox Bridge facility yields a product with approximately 97.7% sand, 0.3% silt, and 2% clay. Dredging operations at the Little River facility yields a product with approximately 94% sand, 4% silt, and 2% clay. There is a small amount of residual organic matter within this material, ranging from 0.4 to 0.7% at Knox Bridge, and 0.3 to 0.8% at Little River.

The texture of undisturbed benthic sediments samples obtained from the Noonday Creek embayment was highly variable, but was generally dominated by the sand- and silt-sized fraction. The average texture was a sandy loam, with approximately 53% sand, 36% silt, and 11% clay. In addition to the mineral fraction, the samples contained approximately 8.2% organic matter.

Tailings are also produced as a waste byproduct at these facilities, and are currently handled separately than the sand product. The composition of these tailings was highly variable, ranging between 12 and 44% sand at Knox Bridge, and between 11 and 80% sand at Little River. Silt composition varied between 48 and 70% at Knox Bridge, and between 18 and 77% at Little River. The clay composition varied from 6 to 18% at Knox Bridge, and 2 to 20% at Little River. Organic matter varied from 3 to 7% at Knox Bridge, and 2.5 to 7.5% at Little River. The tailings ranged in soil texture from a silt loam to a loamy sand.

Another waste byproduct is dredging tailwater effluent. Samples collected from tailwater at the Knox Bridge dredging operation showed low levels of suspended solids (approximately

50 mg/L), while samples collected at the Little River dredging operation showed higher levels (approximately 1100 mg/L). The predominant texture of dredging tailwaters is likely to be clay due to extended holding times provided by the tailings collection basins. Estimates of total return loads from tailwater effluent are unavailable, as estimates of discharge volumes would be required for this estimate. Due to the nature of the operations, however, routine measurements of discharge and concentration would be required for adequate precision to be achieved. Yet the total magnitude of return loads appears to be small in comparison to the sand- and silt-size fractions removed.

Dredging operations by Blankenship Sand yielded a sand product at the Knox Bridge dredging operations that contained approximately 0.13 g/kg total phosphorus (0.26 pounds per ton) and a tailings byproduct that contained approximately 0.60 g/kg total phosphorus (1.2 pounds per ton). The sand product at the Little River dredging operations contained approximately 0.077 g/kg total phosphorus (0.15 pounds per ton) and a tailings byproduct that contained approximately 0.21 g/kg total phosphorus (0.42 pounds per ton). Lake benthic sediment collected from the Noonday Creek embayment contains approximately 0.35 g/kg total phosphorus (0.7 pounds per ton).

The phosphorus contents of the dredged and benthic sediment materials were slightly less that those estimated using U.S. Geological Survey tributary water quality data. One explanation that may reconcile these conflicting estimates is the possibility that the phosphorus was more likely associated with the clay-size fraction. Because dredging operations focus on the removal of the larger sediments with correspondingly lower clay content, these materials are less likely to contain phosphorus.

Dividing the sediment phosphorus concentration by the percent clay content yielded substantially higher sediment phosphorus concentrations; 6.4 g/kg for Knox Bridge sands, 3.2 g/kg for Knox Bridge tailings, 3.8 g/kg for Little River sands, 2.3 g/kg for Little River tailings, and 3.3 for Noonday Creek sediments. These values are substantially larger than those

found for the bulk sediments, and are even larger than tributary observations, indicating that this is a possible reason for the lower bulk total phosphorus concentrations.

Biologically available phosphorus (i.e., the form of phosphorus that phytoplankton can use for growth) concentrations are usually low in lake water, as P tends to sorb to soil minerals (e.g., iron oxyhydroxides) and calcium cations. Because clay particles are smaller, they have much greater surface area per unit volume than do the larger sand particles, and so the majority of phosphates bond to the finer materials.

While results varied depending on the method of analysis, the smaller clay fraction of the sediment consistently contained considerably more phosphorus than the larger sand fraction. Three fractions of sediment from both the Little River and the Knox Bridge sites were tested, including the dredged product and waste tailings from near or within the settling pond. Using the alkaline desorption (NaOH) method, clay contained 75.7 ppm phosphorus, while sand contained 20.7 ppm. The water in the settling ponds, and from two tributaries of the lake, was also tested. The amount of phosphorus in the water samples was small relative to that in the sediment. All values of total phosphorus in water samples were less than 0.5 mg/L, and many were below detection limits.

Analysis of the iron composition in the sediments indicated that the amorphous form dominated the tailings while the crystalline form dominated the sand product at both dredging locations. The sorption of phosphorus to iron is favored on the amorphous form, suggesting that the sorption of phosphorus to iron within the tailings material is likely.

Soil organic matter is the fraction of the soil composed of anything that once lived. As with phosphorus, a much higher percent organic matter was found within the tailings (5.7%) than within the sand samples (0.034%). In general, soils in Georgia contain only small amounts of organic matter (1 to 2%) compared to the very fertile soils found in the Midwest (4 to 5%). The decay of organic materials provides another source of nutrients (including phosphorus, nitrogen, and carbon) for biological systems.

The removal of benthic sediments from Lake Allatoona by dredging or other methods will remove nutrients (especially phosphorus) that are bound to these sediments. The amount of nutrients removed by dredging depends on the size of the sediment particles removed. The presence of these nutrients mean that the dredged tailings are an excellent medium for plant growth, having ideal soil textures, including loams, silt loams, sandy loams, and loamy sands. Coupled with plentiful organic matter, these materials could be marketed for landscaping purposes. While health risks associated with toxic metals contamination are a concern, test results failed to show metals contamination, and all results were well below state regulatory limits. A current limitation for the use of these materials for landscaping purposes is the poor soil structure. Improvements in soil structure by composting or other means would allow more widespread use of these materials as soil amendments.

Reductions of autochthonous nutrient loadings within Lake Allatoona requires both reductions in lake loading, as well as reductions in the internal sediment pool. Removal of benthic sediments containing nutrients is a key component of improving lake water quality. While sand removal is currently the only economically viable option for sediment removal, incentives to remove what are currently dredged byproducts should be considered.

Chapter 4

Laboratory Investigations of Lake Allatoona Sediments

This chapter focuses on laboratory physical, chemical and biological assays of sediments collected from the Little River Embayment. Of particular interest is the effect of sediment removal and sediment resuspension on water quality parameters including algal biomass. A series of laboratory experiments was performed to evaluate sediment physical and chemical properties, some of which had previously been determined by other methods in professional laboratories on the University of Georgia campus. Based on a thorough literature review and previous laboratory and field results, it was hypothesized that the overall effect of sediment removal would be a decrease in algal biomass. It was also hypothesized that sediment resuspension would result in an increase in algal biomass.

4.1 Materials and Methods

4.1.1 Mesocosm Experiment

Six clear acrylic columns (Spartech Townsend, Des Moines, Iowa) were used to investigate the interaction between sediment and lake productivity. The columns were 122 cm tall by 26.7 cm in diameter. The bottom ends were sealed using a PVC cap with an embedded o-ring (UGA Instrument Shop, Athens, GA) to prevent leakage. Five of the six columns are shown in Figure 4.1 - the sixth column seal failed, resulting in the loss of column contents.

Sediments collected from Noonday Creek embayment on November 5, 2005, were used for these column experiments. On November 11, 2005, a slurry of approximately 20 L of sediment and 40 L of water was created using a blunger (mixing tool) in a 75 L container. Equal amounts of the slurry were transferred to four of the six columns. All columns were

Figure 4.1: Laboratory mesocosm experimental columns (122 cm in height, 26.7 cm in diameter). A control column (left) filled with lake water, while the remaining columns filled with lake water plus approximately 21 cm of lake sediment. Resuspension of sediment in two columns (second and fourth from left) led to increased algal growth.

then filled to a total height of 111 cm with water collected from Lake Herrick, a local impoundment on the University of Georgia campus. A single control column was filled with lake water to a total height of 111 cm. The columns were allowed to reach equilibrium over a period of eight weeks. After settling, the sediment depth was approximately 21 cm in the experimental columns.

The columns were maintained in a 12:12 light:dark cycle illuminated by GE wide-spectrum plant and aquarium fluorescent grow lamps. After four weeks, the fluorescent lamps were replaced with three GE R400 Multi-vapor lamps such that two columns shared a single light source. A small water pump (FountainPro, JeanTech Inc., www.fountainpro.com) was placed approximately 25 cm below the surface in each column to circulate water at a rate of approximately 1.5 L per minute. Lake water was added as needed to compensate for evaporative loss.

On December 26, 2005, three columns were mixed with a blunger: two containing sediment and two controls without sediment. The two experimental columns containing sediment were mixed until they held 20-22 g/L suspended solids as read by an ASTM soil hygrometer (HB Instrument Co., Collegeville, PA). Two columns containing sediments were left undisturbed. Temperature, specific conductance (SC), pH, and dissolved oxygen (DO) were measured using a Hydrolab Quanta (Austin, TX). Soluble reactive phosphorus (SRP) in 0.22 μm filtered water was measured using a LaMotte Smart 2 colorimeter (LaMotte, Chestertown, Maryland), turbidity by a Hach 2100P turbidimeter (Hach, Loveland, CO) (Hach, 1991-2004), and chlorophyll *a* (Chl *a*) using a Turner 10-AU fluorometer (Turner, Sunnyvale, CA) (Turner, 1998). These parameters were measured daily between 5 and 8^{pm} for 14 days.

Chl *a* is the primary photosynthetic pigment in algae and cyanobacteria (Wetzel, 2001). The fluorometer measures Chl *a* indirectly as photoemission after exposure of the organisms to 460 nm (blue) exciting wavelength. The conjugated double bonds in the Chl *a* molecule are excited by this wavelength to emit a 685 nm (red) light, which is measured by the fluorometer (Turner, 1998).

4.1.2 Physical and Chemical Tests

A sediment algal assay and a P-extraction technique previously found to correlate with algal assay results were performed using sediment from the Little River Embayment. The purpose of these experiments was to quantify potentially algal-available phosphorus (PAAP) in Lake

Allatoona sediment, and to compare these results with the results of chemical extractions in effort to find a correlation between a chemical extraction analyses and algal productivity assays.

Collection of Sediments, Sediment Drying and Sediment Sterilization

Sediments used for analyses of particle-size distribution, sorption isotherms, desorption experiments and algal assays were collected from approximately 0.5-m below the sediment surface in the Noonday Creek Embayment (Figure 3.1). Sediments were air-dried at room temperature and sieved through a 2 mm (#10) mesh. Particle size distribution was determined by the micropippette method (Miller and Miller, 1987).

Sediments used for physical and chemical tests, as well as algal assays, were air-dried at room temperature (22-23°C) after collection and stored in plastic bags until use (approximately six months). While this assay could be performed with moist sediments, air-dried sediments were used in order to minimize the interference of other algae with the test alga.

While all methods of drying soil affect the extractability of nutrients, air-drying typically causes less drastic changes than drying by oven or microwave (Payne and Rechcigl, 1989). The extent to which a soil property is changed depends on soil type (composition and structure) and on the severity of the method used (Payne and Rechcigl, 1989).

The effect of air-drying on soil pH is variable, and depends on soil type. However, the effect is small, typically raising or lowering pH less than 0.3 units (Bartlett and James, 1980; Payne and Rechcigl, 1989). Drying typically increases dispersion and solubility of organic matter (Bartlett and James, 1980). Air-drying can increase extractable P as measured by the molybdovanadophosphoric acid and Mehlich methods (Payne and Rechcigl, 1989; Rechcigl et al., 1992), but the effect is partly reversed by rewetting (Bartlett and James, 1980).

The sediment-algal assay may also be performed with sterilized sediments. Sterilization of sediment used in this assay was considered for the purpose of eliminating algal spores and other microorganisms in sediment from competing with the test alga *S. capricornutum*.

Because the assay is an attempt to quantify PAAP, the uptake of P by bacteria would likely result in a lower quantity of P available for algal uptake, reducing PAAP.

The standard method of sterilizing sediments for algal assays is by autoclaving (Anderson and Magdoff, 2005). Other methods of sterilizing sediments include addition of sterilants such as of chloroform ($CHCl_3$), ethylene and propylene oxides, methyl bromide, mercuric chloride ($HgCl_2$) and sodium azide (NaN_3), as well as irradiation using ^{6}0Co and microwaves. All methods have advantages and drawbacks with most drawbacks concerning the alteration some sediment physical and/or chemical properties.

Autoclaving soils and sediments increases algal-available P. This is due, in part, to the release of cell content from lysing of organisms. Anderson and Magdoff (2005) found that autoclaving sediments resulted in an increase in sediment-algal assay (using *S. capricornutum*) cell density of 58% over non-autoclaved sediments. Autoclaving also resulted in similar increases in chemically extracted P.

PO_4^{3-} Sorption Isotherm

To quantify the maximum sorption capacity of sediments used in algal assays, a sorption isotherm was performed by adding 50mL 0, 2.5, 5, 10, 15, 20, 30, 40 50 and 60 mg/L PO_4^{3-} solution to 0.5 g sediment. Samples were incubated at 21°C and 40 rpm in an end-over-end shaker for 12 hours followed by centrifugation at 3700 rpm for 10 minutes. Supernatant samples (20 mL) were filtered through 0.22 μm filters, and the samples diluted to approximately 0.01-0.5 mg/L final concentration. SRP concentrations in the diluted samples were analyzed using the ascorbic acid method (Eaton et al., 2005) with measurements read at 880 nm wavelength using a Shimadzu UVmini 1240 spectrophotometer (Shimadzu Corp., Tokyo, Japan).

Sediment P Desorption with Iron-Impregnated Filter Papers

Phosphorus was extracted using iron-impregnated filter papers as described in Sparks (1996), with the exception of the quantity of filter paper used. The published method suggests using five 2×10-cm strips to sorb P from a gram of sediment suspended in 40 mL of deionized water. However, preliminary tests found this quantity of filter paper excessive and unnecessary. Consequently, for the assays reported here, one 47-mm diameter filter circle cut into three strips was used for each replicate. Two standard concentrations, 50 and 100 μg/L, were processed as samples. Three standard concentrations, 20, 40 and 100 μg/L, were not processed as samples, but analyzed by ascorbic acid method and quantified by spectrophotometer. Phosphorus concentration was determined using the ascorbic acid method (Eaton et al., 2005) with 25-mL sample and 4-mL mixed reagent. Two standard curves were made, one based on the results of standards processed as samples, and a second based on laboratory standards.

Sediment P Desorption by $CaCl_2$

Three replicates each of 2, 4, 8 and 16 g air-dried sediment in 45 mL of 1 mM $CaCl_2$ solution and three replicates of 24-g air-dried sediment in 40 mL of 1-mM $CaCl_2$ solution in 50-mL centrifuge tubes were mixed for 24 hours using an end-over-end shaker. Samples were centrifuged for 15 minutes at 1500 rpm and filtered through 0.45 μm syringe filters. Orthophosphate concentrations in the filtered supernatant were measured using the ascorbic acid method (Eaton et al., 2005)

4.1.3 Algal assays

The purpose of the algal assays was to estimate the amount of P in Little River sediment that was potentially available to *S. capricornutum* under neutral pH at 22°C. This is accomplished by determining the P:biomass curve, which is the cell density of *S. capricornutum* when grown for a range of P concentrations

Media and Glassware

Two media differing mainly in their concentrations of macronutrients were evaluated for use in P:biomass curves and sediment algal assays. Bold's Basal Medium (Carolina Biological Supply, year) is a broad-range growth medium for algae with high concentrations of macronutrients. Note that Modified BBM* corresponds to the standard BBM (Anderson, 2005) with the exception that 1.72 mmol K_2PO_4 (no KH_2PO_4) and 0.18 mmol $NaHCO_3^-$ were added to supplement the buffering system. A second medium, Synthetic Algal Nutrient Medium (SANM aka NAAM) (Miller et al., 1978) was tested for an initial P:biomass curve. Because SANM severely limited biomass of *S. capricornutum*, modified BBM* was chosen for the subsequent algal assays.

All glassware was washed using Micro-90 (International Products Corporation, Burlington, New Jersey), rinsed thoroughly, and soaked for at least 12 hours in a 10% solution (v/v) bath of HCl. Glassware was rinsed at least three times with deionized water, dried, mouth down, at room temperature, and covered with aluminum foil until use.

After media was placed in flasks, flasks were stoppered with a foam (#6) plug and covered to the neck with aluminum foil. Flasks were sterilized by autoclaving (30 minutes at 121°C and 100 Pa), and then left 48 hours at room temperature (22°C) before algal inoculation to allow for equilibration with atmosphere.

P:Biomass Curve

A P:biomass curve was generated by assessing the resultant cell densities when *S. capricornutum* was grown under varying P concentrations. Modified BBM (100 mL) with varying levels of P (four replicates each of 0, 10, 25, 50 and 100 μmol PO_4-P) was placed in 250 mL Erlenmeyer flasks. Two days after autoclaving, an aliquot (77 μL) of P-starved *S. capricornutum* was used to inoculate each flask. (Standard inoculation culture density was 3.872×10^6 $\pm$ 5% cells/mL as measured using an improved Neubauer hemacytometer, and 11.0 raw units fluorescence represented approximately 3000 cells/mL in the diluted inoculum.)

Samples were incubated on an orbital shaker (Lab-line, Melrose Park, IL) at 100 rpm under continuous illumination from six Philips F40T12/DX Alto 40 W fluorescent lamps (PAR = 17 μEi/cm^2) until cells reached a stationary growth phase (<10% increase in density over 24 hours). A standard curve of cell densities to fluorescence was made by serial dilution and enumeration of samples by improved Neubauer hemacytometer. Resulting cell densities were measured by fluorometry.

Note that the Modified BBM* corresponds to the standard BBM (Anderson, 2005) with the exception of 1.72 mmol K_2PO_4 (no KH_2PO_4) and 0.18 mmol $NaHCO^{3-}$ added.

Sediment Algal Assay

The purpose of the sediment algal assay was to test the change in *Selenastrum capricornutum* (UTEX1645) (UTEX, Austin, Texas) cell biomass when grown in varying clay turbidities. The algal assay methods were based on the Algal Assay Bottle Test (AABT), which was developed by the U.S. Environmental Protection Agency (Miller et al., 1978). Our method deviated from the AABT as follows:

1. The AABT suggests an initial inoculum of 1000 cells/mL. Our method used an initial inoculum of 3000 cells/mL, as recommended by Schultz et al. (1994). The higher initial inoculum resulted in more rapid increases in biomass, hence more rapid results.

2. The standard AABT method uses an electronic particle counter to count cells, and converts these data to maximum standing crop as dry weight. Another acceptable measurement technique includes cell counting using a hemacytometer which was used for this study.

Sand, silt and floating organic matter were removed from sediment primarily to facilitate enumeration of algae. However, removal of sand was also consistent with the theory that, in a lake, sand would likely contribute little phosphorus to algal biomass as it would settle out of the photic zone within a few minutes.

Clay-sized sediment particles (<2 μm) were extracted by mixing sediment with deionized water for 24 hours in an end-over-end shaker, followed by removal of supernatant after 24 hours settling time. A ratio between NTU and mass solids (in mg/L) was estimated by drying 512 Nephelometric Turbidity Units (NTU) samples in aluminum weighing boats in an 88°C oven for six hours.

Turbidities of 70, 146 and 298 NTU were created by serial dilution of the turbid suspension with deionized water. Concentrated P-free BBM was added after target turbidities were reached. Initial pHs ranged from 4.9 in the control flask (containing no sediment) to 6.6 in the 298 NTU flasks.

Aliquots (100 mL) of each turbid suspension, and a control with no sediment, were placed in 250-mL Erlenmeyer flasks. Two flasks at each turbidity level were autoclaved for 30 minutes, and three flasks at each turbidity level were not autoclaved. Two days after autoclaving, four flasks at each turbidity level (two autoclaved and two not autoclaved) were inoculated with *S. capricornutum* to create an initial cell concentration of 3000 cells/ mL. One control at each concentration was not inoculated with the test alga or autoclaved in order to enumerate *S. capricornutum* cells from sediment. Samples were incubated on an orbital shaker at 100 rpm under constant illumination by Philips F40T12/DX Alto 40-W fluorescent lamps.

Cell concentrations were determined using an improved Neubauer hemacytometer. Cell concentrations were determined daily beginning on day 10, and continued until cells reached a stationary growth phase (¡10% increase in density over 24 hours). The highest cell concentration in each sample reached within two weeks was used for statistical analysis. Counting of autoclaved sampled was discontinued due to evidence that the autoclaving procedure used was not sufficient kill sediment biota.

4.2 Results

4.2.1 Mesocosm Experiment

Disturbed and undisturbed conditions were compared using microcosms maintained under ambient conditions that simulated the natural setting. Changes in environmental conditions, including dissolved oxygen (DO), pH, and Chl *a* were used to evaluate the effects of benthic sediment removal on the overlying (pelagic) water quality. Photographs of the sediment columns are provided in Figure 4.2. Water quality data in both tabular and graphical formats are presented in Tables 4.1 and Figure 4.3, respectively.

Note that, after mixing, pH and DO concentrations within the sediment-mixed experimental columns initially decreased to pH between 6.1 and 6.6 and DO between 1 and 3% saturation. These subsequently began to increase with pH increasing to between 8.7 and 10.3, and DO increasing to between 120% and 170% of equilibrium with the atmosphere from 5 to 10 days after mixing. Surface Chl *a* increased to peak concentration 7-9 days after mixing, while periphyton biomass increased throughout the experiment. Control columns showed no similar rise or fall in Chl *a*, pH or DO. Note how the clarity and algal abundance vary with treatment. Also note from the figures the resulting reduction in biological activity when water was added on January 5, 2006. Immediate decreases in pH, DO, and Chl *a* were all observed.

The filamentous green alga *Oedogonium* (Chlorophyta) dominated the algal community in all columns (Figure 4.4. Long filaments of this alga attached to the walls of all columns. By the end of the experiment (Day 13) the filaments were shortest, approximately 2 cm, in the column without sediment. The filaments in the unmixed columns containing sediment were, on average, approximately 6 cm. The filaments in the mixed column were long enough to reach from the walls to the center of the columns where they entangled with filaments attached to the opposite wall (>13 cm).

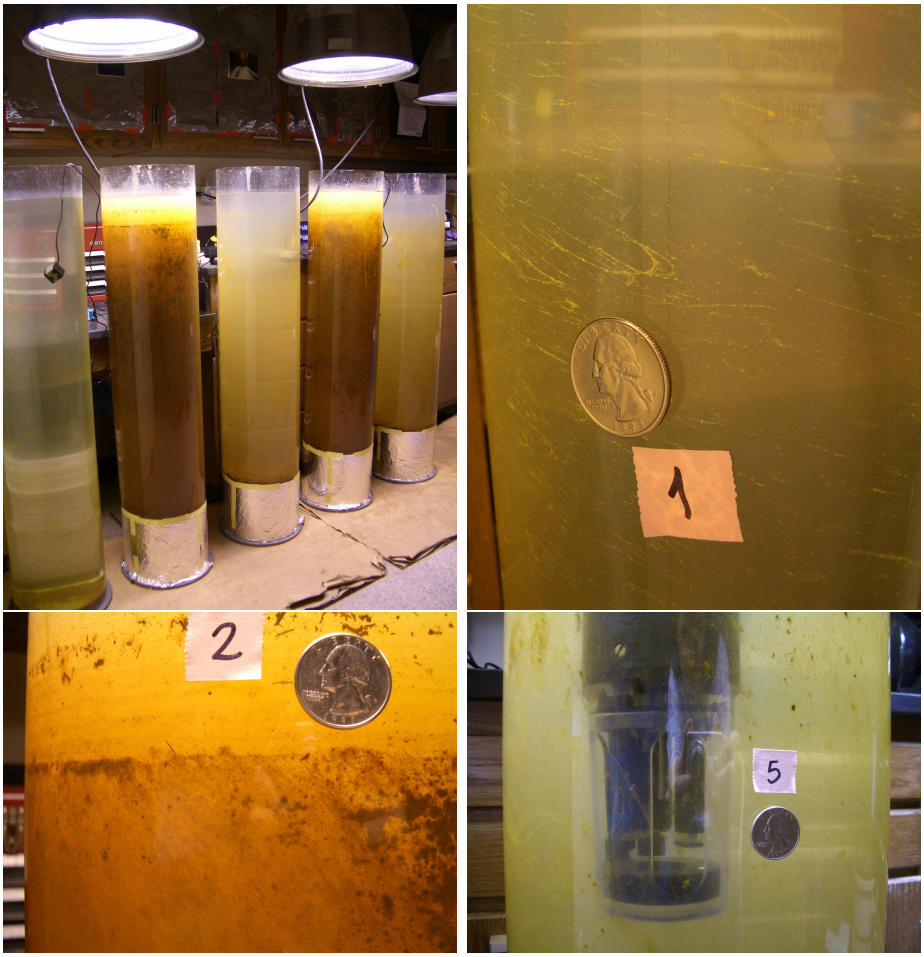

Figure 4.2: Photographs of sediment column studies. Column 1 (on left) is water with no sediment. Columns 2 and 4 are columns with disturbed (mixed) sediments. Columns 3 and 5 are columns with undisturbed sediments.

Table 4.1: Column water quality observations.

Date	Column				
	1 control	2 mixed	3 unmixed	4 mixed	5 unmixed
12/26/2006 (before mixing)					
Temperature, °C	24.44	25.09	25.13	24.96	24.62
Conductivity, μS/cm	89	95	95	97	95
Dissolved O_2, mg/L	7.14	6.03	7.02	7.04	7.59
pH	7.01	6.98	7.04	7.02	7.11
Dissolved O_2, %	85.7	70	85.1	85.3	91.7
Turbidity (Quanta), NTU	4	6.5	8.2	8.6	3.3
SRP (0.22 μm), mg/L	0	0	NA	NA	NA
12/26/2005 (after mixing)					
Temperature, °C	NA	26.3	NA	25.66	NA
Conductivity, μS/cm	NA	152	NA	124	NA
Dissolved O_2, mg/L	NA	0.16	NA	0.27	NA
pH	NA	6.15	NA	6.57	NA
Dissolved O_2, %	NA	1.4	NA	3.1	NA
SRP (0.22 μm), mg/L	NA	0.06	NA	0.17	NA
12/27/2005					
TSS, mg/L	NA	2	NA	1	NA
SRP (0.22 μm), mg/L	NA	0.03	NA	0.03	NA
Chl *a*, raw	0.157	NA	0.34	NA	0.095
Turbidity (Hach), NTU	NA	497	NA	293	NA
12/28/2005					
Temperature, °C	24.11	29.21	25.01	28.13	24.53
Conductivity, μS/cm	89	136	96	121	94
Dissolved O_2, mg/L	7.33	2.84	8.13	3.21	7.85
pH	7.23	8.05	7.55	7.44	7.44
Dissolved O_2, %	88.2	34.1	98.2	41.6	94.3
Turbidity (Hach), NTU	NA	344	NA	272	NA
SRP (0.22 μm), mg/L	NA	0.02	NA	0.02	NA
TSS, mg/L	NA	0	NA	0	NA

Table 4.1. Column water quality observations (cont.).

Date	Column				
	1 control	2 mixed	3 unmixed	4 mixed	5 unmixed
12/29/2005					
Temperature, °C	23.8	26.71	24.44	26.55	23.73
Conductivity, μS/cm	90	135	96	117	95
Dissolved O_2, mg/L	7.13	3.29	7.93	5.26	7.81
pH	7.69	7.52	8.01	7.9	7.74
Dissolved O_2, %	85.7	41.7	95.5	66.1	92.7
Turbidity (Hach), NTU	3.69	350	6.98	244	5.66
SRP (0.22 μm), mg/L	0	0	0	0	0
12/30/2005					
Temperature, °C	21.9	26.1	22.28	25.38	21.18
Conductivity, μS/cm	91	131	98	117	96
Dissolved O_2, mg/L	7.74	5.67	8.27	5.82	8.21
pH	7.78	8.18	8.04	7.82	7.45
Dissolved O_2, %	89	70	95.5	71	91.1
Turbidity (Hach), NTU	3.82	229	767	197	6.36
SRP (0.22 μm), mg/L	0	0.01	0	0.02	0
12/31/2005					
Temperature, °C	24.09	28.19	24.91	27.89	24.39
Conductivity, μS/cm	92	122	99	108	98
Dissolved O_2, mg/L	7.62	7.65	9.03	7.36	8.13
pH	6.91	8.95	8.04	8.68	7.77
Dissolved O_2, %	91.1	98.5	109.6	95.4	98.8
Turbidity (Hach), NTU	3.15	194	8.59	174	9.56
SRP (0.22 μm), mg/L	0	0.02	0	0	0

Table 4.1. Column water quality observations (cont.).

Date	Column				
	1 control	2 mixed	3 unmixed	4 mixed	5 unmixed
1/1/2006					
Temperature, °C	24.23	28.47	25.08	27.86	24.54
Conductivity, μS/cm	92	134	99	109	98
Dissolved O_2, mg/L	8.31	9.74	9.33	9.54	7.88
pH	7.93	9.65	8.26	8.98	7.96
Dissolved O_2, %	100.3	126	113.4	118.7	92.8
Turbidity (Hach), NTU	3.06	203	8.94	150	9.45
SRP (0.22 μm), mg/L	0	0	0	0	0
1/2/2006					
Temperature, °C	25.93	30.28	27.21	29.53	26.42
Conductivity, μS/cm	92	153	100	113	99
Dissolved O_2, mg/L	7.7	11.45	8.02	8.97	7.83
pH	7.9	9.9	8.27	9.49	7.75
Dissolved O_2, %	98.4	152	102.5	118.1	99
Turbidity (Hach), NTU	2.83	208	8.49	134	10.2
SRP (0.22 μm), mg/L	0.01	0.02	0.01	0.02	0
1/3/2006					
Temperature, °C	25.11	28.32	25.98	27.98	25.24
Conductivity, μS/cm	92	153	101	125	99
Dissolved O_2, mg/L	8.22	12.35	7.8	10.59	8.22
pH	8.41	10.31	8.15	10.16	7.77
Dissolved O_2, %	99.4	156.4	95.9	136.2	99.3
Turbidity (Hach), NTU	3.33	152	10.9	115	10
SRP (0.22 μm), mg/L	0.01	0.02	0	0.01	0

Table 4.1. Column water quality observations (cont.).

Date	Column				
	1 control	2 mixed	3 unmixed	4 mixed	5 unmixed
1/4/2006					
Temperature, °C	23.63	27.29	24.54	26.66	23.72
Conductivity, μS/cm	90	154	99	133	97
Dissolved O_2, mg/L	8.4	12.94	7.63	13.76	8.44
pH	8.27	10.01	7.85	9.88	7.72
Dissolved O_2, %	99.7	164.4	92.4	171.4	100.2
Turbidity (Hach), NTU	3.03	112	11.1	106	10.8
SRP (0.22 μm), mg/L	0.01	0.02	0	0.02	0
1/5/2006					
Water added, L	4	5	3.5	5	4
Temperature, °C	24.18	25.32	25.14	24.89	24.87
Conductivity, μS/cm	87	127	96	117	92
Dissolved O_2, mg/L	8.72	6.16	7.96	8.57	8.66
pH	8.39	7.17	8.06	7.51	8.49
Dissolved O_2, %	105	75	96.9	103.9	105.8
Turbidity (Hach), NTU	3.51	331	10.6	215	9.59
SRP (0.22 μm), mg/L	0	0	0	0	0
1/6/2006					
Temperature, °C	20.82	21.02	20.91	20.68	20.29
Conductivity, μS/cm	86	127	96	116	92
Dissolved O_2, mg/L	7.74	4.65	6.9	5.91	7.09
pH	7.96	7.1	7.57	7.27	7.45
Dissolved O_2, %	87.1	52.5	78	66.2	77.5
Turbidity (Quanta), NTU	6.6	319	12.1	205	6.7

Table 4.1. Column water quality observations (cont.).

Date	Column				
	1 control	2 mixed	3 unmixed	4 mixed	5 unmixed
1/6/2006					
Temperature, °C	23.51	24.54	24.35	24.05	23.58
Conductivity, μS/cm	86	125	96	114	93
Dissolved O_2, mg/L	8.37	6.74	8.27	8.18	8.36
pH	8.28	7.38	7.89	7.57	8.17
Dissolved O_2, %	98.9	79.8	99.5	97.3	97.9
Turbidity (Hach), NTU	3.54	231	11.8	149	9.59
SRP (0.22 μm), mg/L	0	0	0	0	0.01
1/7/2006					
Temperature, °C	21.64	22.77	22.3	22.01	21.48
Conductivity, μS/cm	87	125	97	115	94
Dissolved O_2, mg/L	9.75	7.69	8.31	7.98	8.57
pH	8.54	7.74	7.9	7.94	8.02
Dissolved O_2, %	111.3	89.7	95.9	92.3	98.7
Turbidity (Hach), NTU	3.61	204	11.2	127	9.13
SRP (0.22 μm), mg/L	0.01	0.03	0.01	0.04	0
1/8/2006					
Temperature, °C	25.35	31	26.48	30.8	27.6
Conductivity, μS/cm	87	120	97	110	93
Dissolved O_2, mg/L	9.7	6.56	8.54	7.53	7.95
pH	8.61	8.31	7.92	8.52	8.17
Dissolved O_2, %	118.4	88.8	106	99.4	102.1
Turbidity (Hach), NTU	3.22	195	11.4	118	8.81
SRP (0.22 μm), mg/L	0.01	0.03	0.01	0.03	0.01

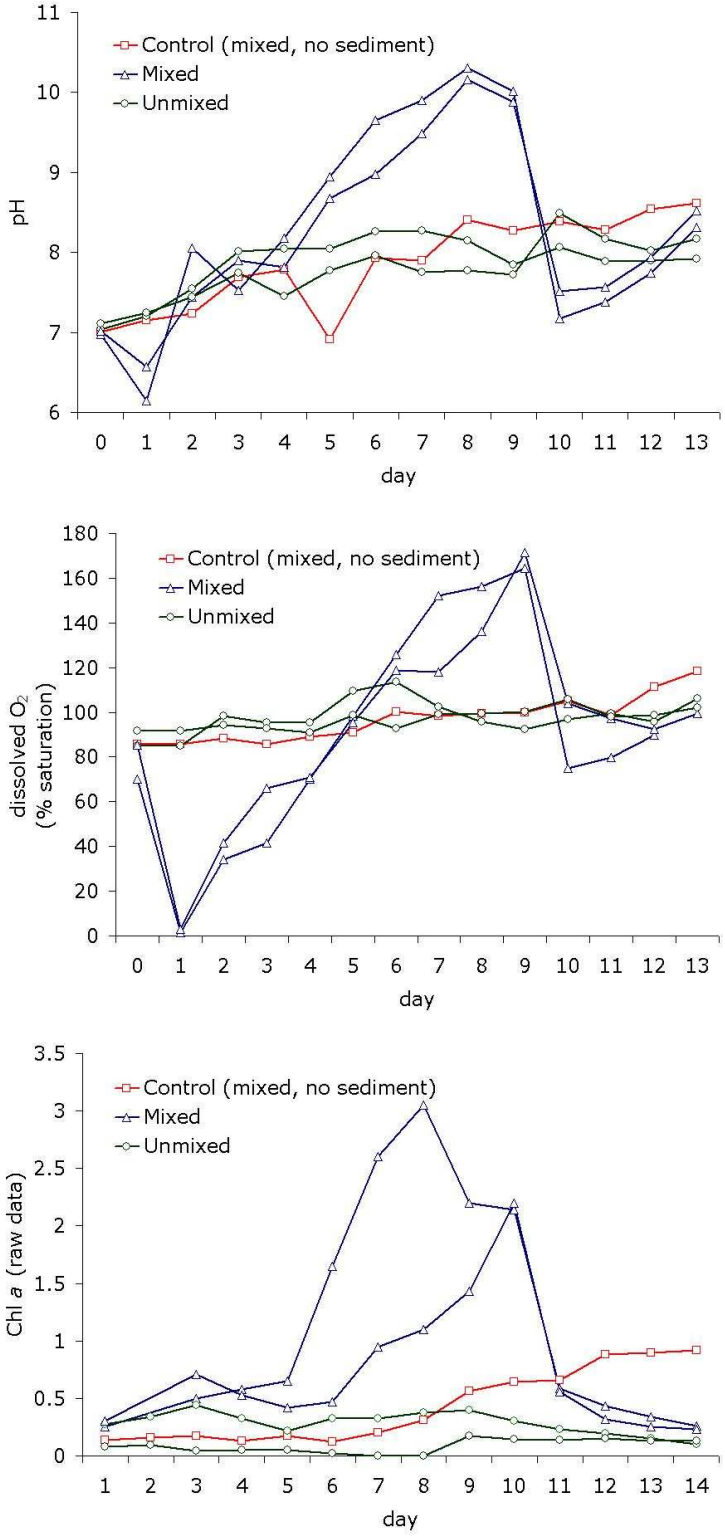

Figure 4.3: Effects of mixing on water quality

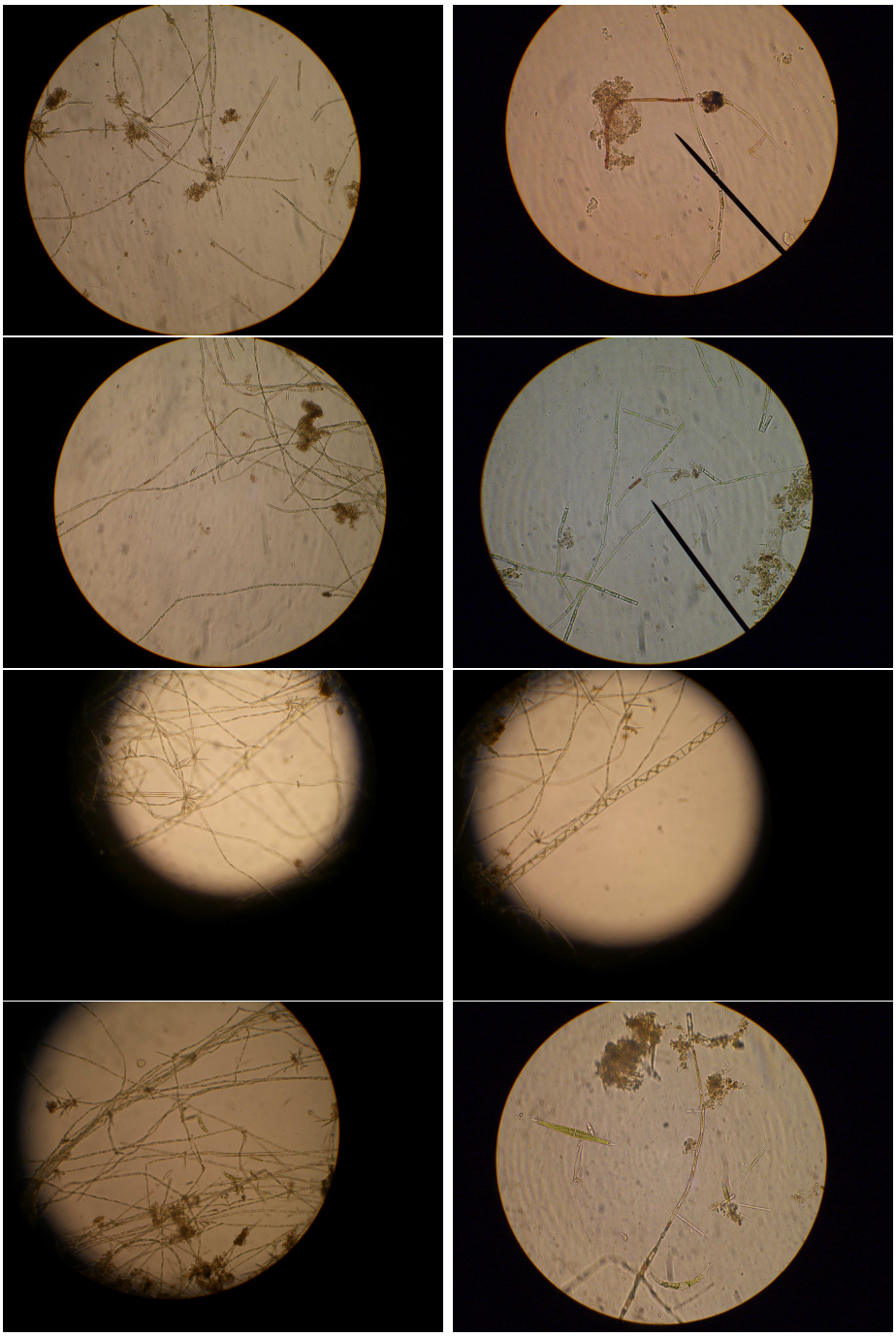

Figure 4.4: Microphotographs of sediment column studies. Top two rows are disturbed sediment columns. Bottom two rows are undisturbed sediment columns

4.2.2 Physical and Chemical Tests

Particle size distribution

The particle size distribution of Little River Embayment sediment used for all batch experiments was 38.5% sand, 59% silt and 2.5% clay. Clay-sized particles in reservoirs tend to be mostly inorganic silicates as well as Fe and Al oxides (Kirk, 1994).

PO_4^{3-} Sorption Isotherm

Maximum orthophosphate sorption capacity was approximately 1200 mg PO_4^{3-}-P/kg sediment (Figure 4.5). A logarithmic curve was added to the figure to show a best-fit for the data.

Sediment P Desorption with Iron-Impregnated Filter Papers

The mean absorbance of duplicate samples was 0.04. Based on the standard curve of known concentrations, 6.3 μg P were extracted from 1 gram of sediment. Based on the curve of standards processed as samples, 3.6 μg P were extracted from 1 gram of sediment (Figure 4.6).

Sediment P Desorption by $CaCl_2$

No detectable P was desorbed from sediment by incubation in 1 mM $CaCl_2$.

4.2.3 Algal Assays

PO_4^{3-} Algal Biomass Curve

Cell density reached a maximum of 7-8 million cells at approximately 25-50 μM PO_4^{3-}-P. concentrations above approximately 50 μM did not lead to increased algal biomass (Figure 4.7.

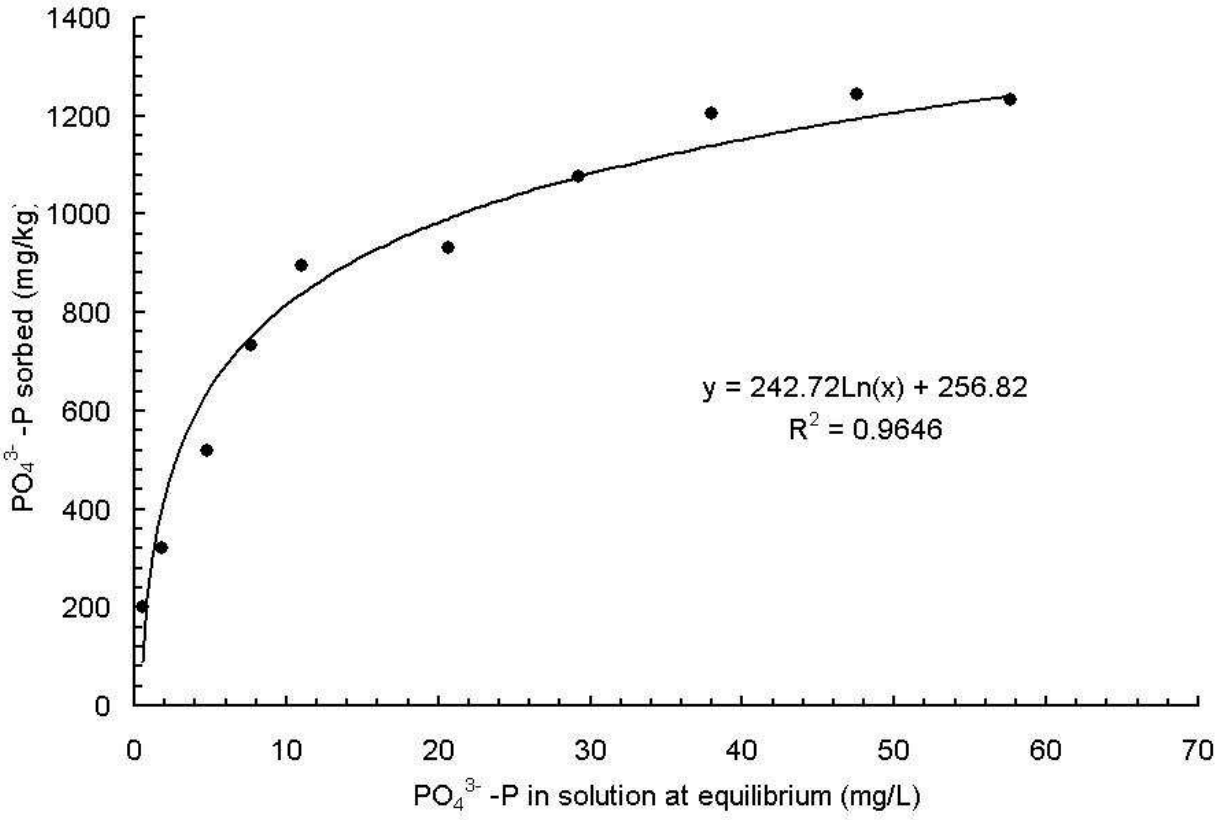

Figure 4.5: Equilibrium Lake Allatoona sediment-water phosphate-P partitioning at 22°C.

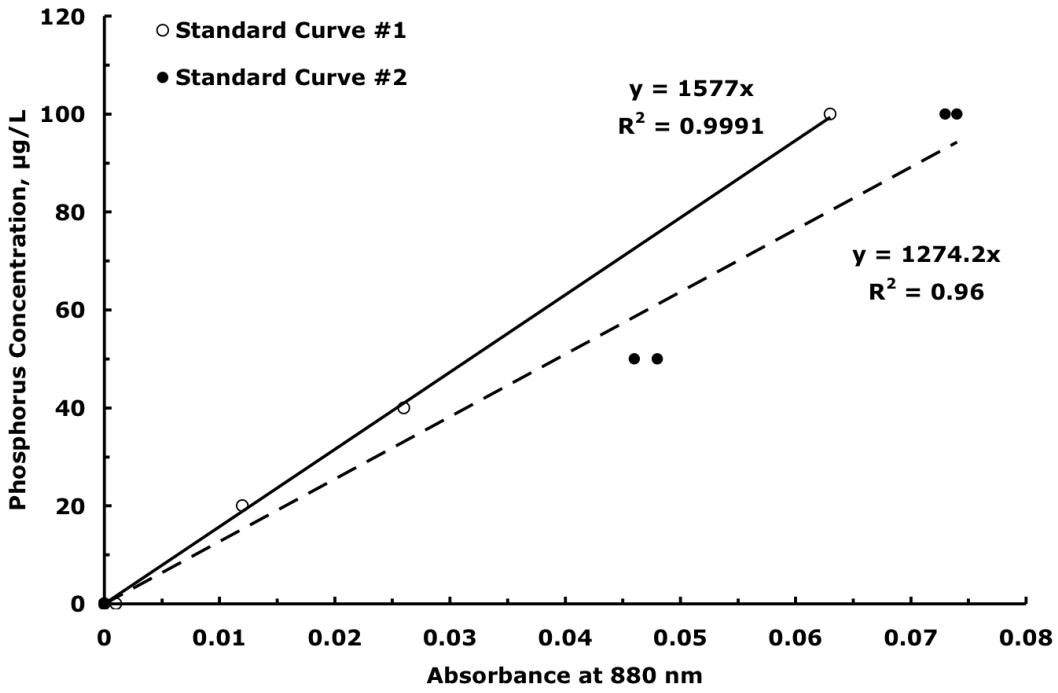

Figure 4.6: Standard curves for relationship between Absorbance and P concentrations. Standard curve #1 obtained using known P concentrations allowed to react with the filter strips, extracted, and then analyzed. Standard curve #2 obtained using known P concentrations and then analyzed without reaction with filter strips and subsequent extraction.

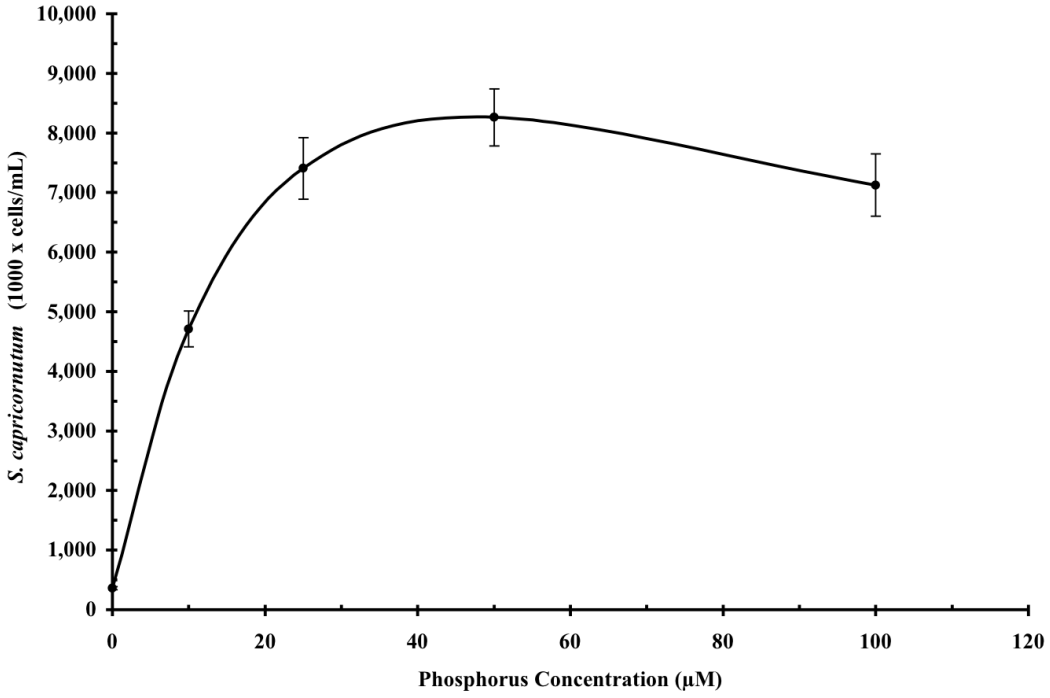

Figure 4.7: Algal P assay curve; algal biomass versus P concentration (at zero turbidity).

The controls, containing standard N (2.94 mM), zero P, zero N, or zero P SANM, showed no significant differences in biomass. This suggests that N was not limiting in the BBM medium.

Sediment algal assay

Figure 4.8 shows that concentrations of *S. capricornutum* increased linearly with increasing sediment turbidity in non-autoclaved samples ($y = 3{,}368\ x + 286{,}656$, $R^2 = 0.8253$). The results supported the hypothesis that higher turbidity due to clay-sized particles would support higher density of the test alga. Most of the variation in cell density is explained by the linear relationship with turbidity. An increase in 1 NTU was associated with an average increase of 3,400 *S. capricornutum* cells.

The conversion to sediment mass required a relationship be found between turbidity and the total solids concentration. A 512-NTU sample was determined to contain 413-mg/L solids. This relationship, in conjunction with the previously determined relationship between phosphorus concentration and biomass, yields the relationship between P concentration and total solids, shown in Figure 4.9. Note that the resulted estimated sediment P concentration is approximately 269 ppm.

4.3 Discussion

The purpose of this research was to assess whether removal of sediments from Lake Allatoona would decrease algal-available phosphorus. Because the lake is shallow, recreational boating, current dredging methods, and strong winds and storms can resuspend sediments. Algal growth in response to sediment resuspension was evaluated in two experiments. The purpose of the experiments was to determine whether mixing results in increased algal biomass.

The mesocosm experiment evaluated the effect of SR on several water quality parameters. The sediment-algal assay evaluated the effects of prolonged SR on algal biomass. Both experiments were preliminary, and more research is needed in order to state with confidence

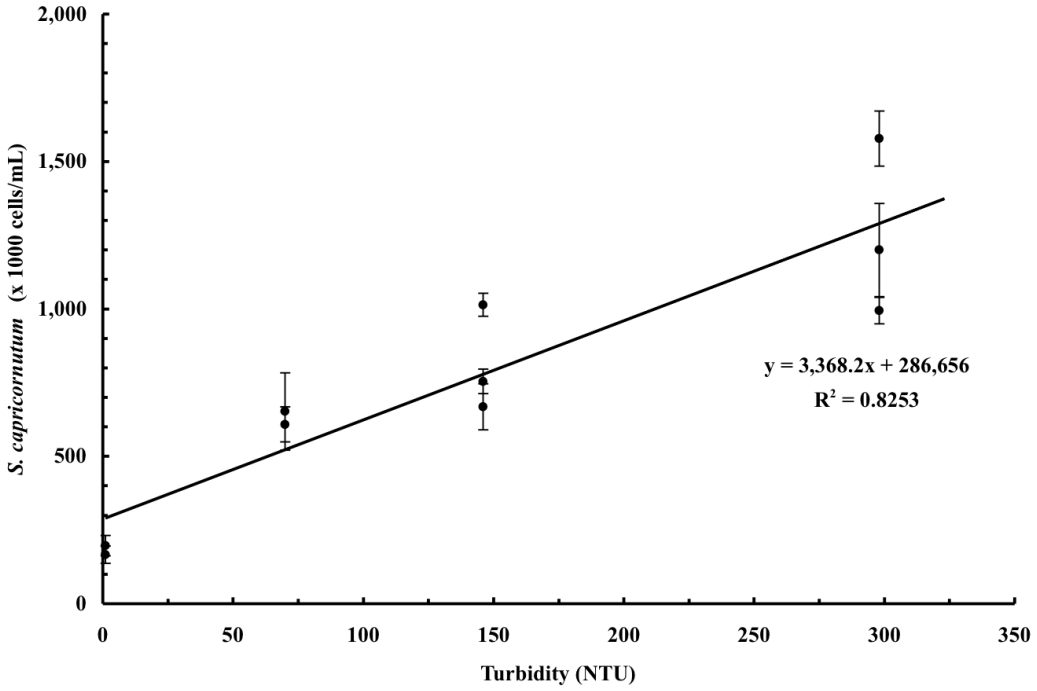

Figure 4.8: Algal P biomass, determined using cell counts, versus initial turbidity.

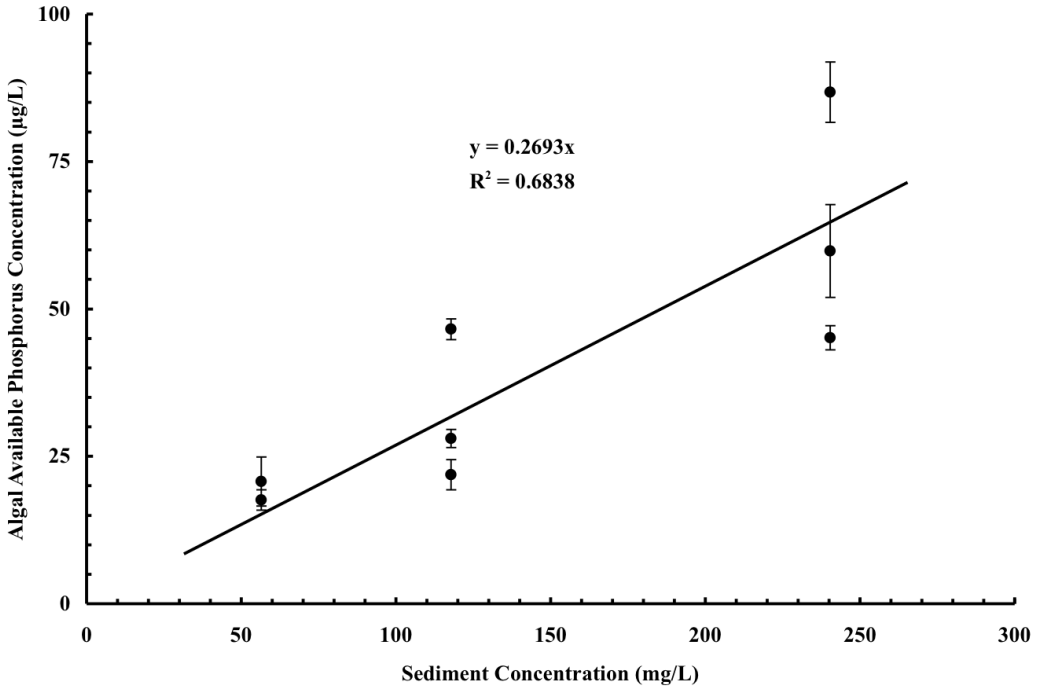

Figure 4.9: Estimated algal-available P concentration as a function of sediment concentration.

that removal of sediments from Lake Allatoona would decrease the frequency and intensity of algal blooms.

4.3.1 Mesocosm Experiment

The changes in Chl *a*, pH and DO seen in this experiment are consistent with the hypothesis that the SR resulted in release of P from suspended particles, promoting algal growth. The immediate fall in pH and dissolved oxygen (DO) of the experimental mesocosms after mixing was likely due to the integration of reduced anoxic, acidic sediments with the overlying water. As discussed in the literature review, low redox potential increases P desorption from sediments when Mn and Fe minerals are reduced (Mortimer, 1941). Therefore, mixing likely provided greater algal-available P to the water column than was available in the unmixed mesocosms.

The subsequent rise of pH and DO observed was likely due, initially, to gradual resettling of suspended particles and associated microbial biomass. The subsequent rise in Chl *a*, pH and DO in the mixed mesocosms all suggest increased rates of photosynthesis over unmixed and sediment-free mesocosms. If this is correct, the rise inChl *a* was due to increased rates of photosynthesis, the rise in DO a by-product of photosynthesis, and the rise in pH was due to fixation of carbonates in photosynthesis.

A second hypothesis that could account for the increase in Chl *a*, DO and pH in the mixed columns is the reseeding of the euphotic zone with algal spores released from sediments. In this scenario any P release from resuspended sediments may or may not contribute to the increase in photosynthetic biomass.

Another hypothesis that could account for the increase in Chl *a* is that light-limited algae produce more Chl *a* per cell than non-light limited algae (Mashall Darley, personal communication). While this effect may account for increases in Chl *a*, it would not contribute to the rise in pH or DO. Again, the rise in Chl *a*, pH and DO were most likely due to increases in primary productivity. pH has been used to estimate increases in algal biomass (Miller et

al., 1978; Lopez-Archilla et al., 2003) based on the premise that it is due to the removal of carbonates during photosynthesis.

SR as a source of algal-available P is well documented in the primary literature. Many researchers have evaluated P flux from sediment cores to overlying water over a range of pH and redox potentials (Haggard et al., 2005; Moore and Reddy, 1994; Anderson, 1975).

While it has already been established that P limits algal biomass in Lake Allatoona, the fact that a non-diazotrophic (not able to fix N) alga dominated all mesocosms supports the theory that P was also the limiting nutrient in the mesocosms. Theoretically, if N were the limiting nutrient in this system, one would expect N-fixing algae to dominate the algal community. The fact that a non-diazotrophic alga was dominant suggests that N was not limiting. The diatom *Synedra* was abundant as well, which suggests that silica was also not limiting. The predominance of *Oedogonium* was likely due to the high surface area:volume ratio in the columns, which gave the alga much more area for attachment than would exist in a lake.

This mesocosm experiment has the advantage of simulating a water column in the reservoir. As Lake Allatoona is very shallow, diffusion of nutrients from the mesocosm sediment to the water's surface is likely similar to the diffusion seen in shallow areas of the lake. Because the sediment used in the experiment was given ample time to settle (8 weeks), it is also likely that the microbial community of the lake sediments was able to reestablish itself in the mesocosms.

The results of this experiment would be strengthened by a more accurate measurement of algal biomass. The fluorometric measurement of Chl *a* in the mixed mesocosms was affected by high concentrations of suspended sediment blocking both excitation and emission wavelengths. Therefore, Chl *a* concentration from the mixed columns were likely higher than recorded. More accurate measurements could be achieved by a Chl a extraction using ethanol, such as the Welschmeyer technique (Welschmeyer, 1994). An accurate method to quantify Chl a of periphyton on column walls would also have been helpful.

Sediment was collected using a shovel, and collected sediments were mixed with lake water before being placed in mesocosms. This method could be improved significantly. Anoxic sediments were mixed with oxic water, thus altering the established redox potential and microbial community. A more efficient method of collection would be to use a sediment corer. Sediments could then be transferred directly to mesocosms. This would (1) reduce the amount of time necessary for sediment settling and microbial recovery, and (2) result in a microbial community more similar to natural conditions.

4.3.2 Sediment Physical and Chemical Tests

PO_4^{3-} Sorption Isotherm

The maximum sorption capacity of PO_4^{3-}-P for Noonday Creek sediment was approximately 1,200 ppm. This value is considerably higher than values reported by Wang et al. (2005) for sediments from Lake Taihu, China. Maximum sorption capacities of PO_4^{3-}-P ranged from 300 ppm for sediments from two areas of the lake considered slightly contaminated (Gonghu and East Taihu), to 500 ppm in the area considered highly eutrophied. Lake Taihu receives industrial and domestic wastewater, as well as runoff from agriculture and land development projects.

While the maximum sorption capacity of Noonday Creek sediments may be useful for modeling purposes, caution should be taken in applying these laboratory results to field conditions. The maximum sorption capacity of these sediments under field conditions will be considerably less than the maximum sorption capacity of these sediments as measured in the laboratory. As previously discussed, a rise or drop in pH reduces sorption capacity of sediments. Also, thorough saturation of sediment particles using an end-over-end shaker likely does not simulate field conditions. Whether sediments in Lake Allatoona function as a source or sink for P depends on numerous field conditions as well as the form of P in overlying waters. These conditions are discussed in the second chapter.

Sediment P Desorption with Iron-Impregnated Filter Papers

To correlate the results of this chemical assay with results of algal assays would require that both tests be performed on several different sediments. An interesting question is whether sediments further away from the Etowah River, the main tributary to Lake Allatoona, contain less algal-available P. Sediments collected from the eastern riverine portion of the lake, mid-lake and near the dam could be assayed to address this question. However, as the orthophosphate desorption with iron-impregnated filter paper is a very time consuming test, other tests should also be performed in effort to correlate results between chemical and biological tests.

Sediment P Desorption by CaCl2

The fact that no P was found in this desorption experiment could be due to a number of factors. P desorbed by mixing may have been immediately re-sorbed to the sediment. Alternatively, desorbed P may have been taken up by microorganisms, such as algae and bacteria with the microorganisms subsequently removed by filtration.

4.3.3 Algal Assays

P:Biomass Curve

The fact that the *S. capricornutum* populations did not increase significantly with added P above 25 μM suggests that, when incubated in modified BBM, with P above 25 μM, algal growth is limited by some other nutrient or environmental parameter.

Sediment Algal Assay

The purpose of the algal assay was to measure potentially algal-available phosphorus (PAAP) adhering to clay-sized particles. Under calm conditions, sand and larger silt particles resettle quickly after resuspension. For this reason, as well as to facilitate enumeration of the test alga, only clay was used in the assay. However, in a shallow reservoir, such as the eastern

potion of Lake Allatoona, the entire depth of the lake may be euphotic, with enough light reaching benthic sediments that it does not limit algal biomass. Under such circumstances, silt and sand would lie in close proximity to algae, and nutrients adsorbed to these particles would be algal-availble.

The mean of the highest algal biomass of samples incubated without a source of P was approximately 260,000 cells/mL. The increase from a starting concentration of 3,000 cells/mL to 260,000 cells/mL must have been supported entirely by P stored as phosphate granules in the inoculum cells or P released and remineralized after cell death.

Samples grown with clay turbidity contained algae from dried sediment. The quantity of these algae, while not enumerated, was small compared to the quantity of test algae. Because the sediment-algal assay did not include enumeration of other algae growing in samples, the estimation of sediment-sorbed P based on comparison with the P biomass curve was likely underestimated. This problem could be eliminated by sediment sterilization prior to performing the assay. As described in more detail in the Materials and Methods section, none of the sediment sterilization techniques tested to date appear to alter sediment physical and/or chemical properties, suggesting this could be a useful approach for future studies.

A concentration of approximately 1.5 million *S. capricornutum* cells/mL resulted from incubation in 298 NTU. Based on the final concentrations of cells incubated with 0 and 25 μM K_2HPO-P in the P biomass curve, a population of *S. capricornutum* cells/mL would be expected from incubation in modified BBM with a corresponding concentration of soluble P (Figure 4.7). When grown under these laboratory conditions, P-containing sediments are associated with *S. capricornutum* abundance, as shown in Figure 4.8. Based on the proportion of the mass of solids per unit turbidity, then the resulting plot of algal-available P per unit solids (shown in Figure 4.9) yields a slope of 269 ppm. This is much higher than the Fe-filter strip estimates, but is more consistent with the total phosphorus estimate provided in the third chapter for the tailings fraction.

The interpretation of results using this algal assay is limited to neutral pH and aerobic conditions. The incubation conditions - a large water surface area in contact with gas inside the flask, flasks stoppered using gas-permeable stoppers and active water movement from placement an orbital shaker - allowed for rapid exchange (especially carbonates and O_2) between flask contents and the outside environment. This rapid exchange likely served as a buffer from the high pH typically associated with rapid photosynthesis in algal cultures.

Future experiments should examine modifications to pH and redox potentials to more closely simulate conditions in the lake. While a natural algal community is not recommended for this assay, the assay could include other algal species found in the lake and obtained from a reputable supplier. Use of several algae in a this assay may be necessary if variations in growth responses to pH and/or redox potential are to be examined, as not all algae can tolerate these changes.

The sediment-algal assay has the advantage of being a variation on a well-established and relatively easily reproduced protocol. The assay could also be easily modified to quantify algal biomass in response to mixing regime. For example, one treatment could be sediment and algae left undisturbed, a second treatment would be lightly shaken, and a third treatment total sediment resuspension.

Removal of Lake Allatoona Sediments

The results of the mesocosm and sediment-algal assay experiments suggest that, in Lake Allatoona, sediment resuspension results in increases in algal biomass. Therefore, removal of these sediments may benefit water quality by reducing the frequency of algal blooms that result from activities that resuspend sediments.

Chapter 5

Conclusions and Recommendations

Accumulation of sediments in lakes adversely affect lake function by reducing water storage, interfering with recreation, degrading sport fishing, and inducing eutrophication. The effects of sediment in Lake Allatoona (a U.S. Army Corps of Engineers reservoir located in northwest Atlanta, Georgia) are a source of ongoing concern, especially due to the rapid development in the upstream watershed. During the winter months, lowering of lake water levels reveal substantial accumulations of lake sediments, which are generally flooded during the summer months once water levels are raised. Growing season algal blooms in embayments have led to the development on phosphorus loading limits, which restrict the discharges of nutrients to the lake.

While ongoing lake dredging activities are removing some of the sediment entering the lake, significant quantities of sediments from previous and ongoing sedimentation persist. These lake dredging operations are limited by the ability to market dredged materials, and a lack financial incentives for upgrading dredging systems.

This research was partially supported by stakeholders interested in establishing a nutrient trading program. Parties dredging sediments from Lake Allatoona would earn a marketable credit based on the quantity of nutrients within the sediments that promote algal growth during the growing season. An important question is how much credit should be given for the sediment removed.

5.1 Sediment Characterization

One objective of this research was to gain a general understanding of Lake Allatoona sediment composition, especially with regards to differences based on sediment particle size. Results of field samples collected from Little River and Knox Bridge dredging sites indicate that sand dredged from the lake has much less P per kilogram (85 mg/kg TP, 5.3 mg/kg PO_4-P) than the smaller particles, or fines, of silt and clay (295 mg/kg TP, 12.7 mg/kg PO_4-P). Due to the surface area: volume ratio, and the fact that most sediment P is sorbed on the particle's surface, very small particles contain much more P per mass than do larger particles. Or, in other words, total P in sediment particles is inversely proportional to size.

If sediment is to be removed for the purpose of improving water quality, then the removal of finer particles would have greater impact on improving water quality than the removal of larger particles. For this reason we suggest that if a P trading program is to be implemented, credit for P removal should be partially based on the particle size of the sediments removed, in that sediments containing smaller particles will contain higher nutrient concentrations.

5.2 Determination of Algal-Available P

There are many forms of P, and several methods to quantify each form. If a P credit or trading system is to be implemented, the question of which form of P to quantify and which method to use is paramount to determine the effects of such a program on the ecological health of the lake. Ideally, the method used would measure potentially algal-available P of sediment. It will be important to determine the correlation between the results of chemical extraction of P in sediments and algal-available P as determined by algal assay.

While the results of a sediment P algal assay can give an estimate of algal-available P, the assay is time-consuming and relatively expensive when compared to chemical methods of P determination. Because the ratio of sediment PAAP to total P is highly variable, we

suggest that chemical determination of total P of sediments not be the only basis for a P trading program.

In order to properly estimate the quantity of algal-available P removed by dredging of sediments, a model is needed that estimates algal-available P under the various conditions found in Lake Allatoona. Such a model should incorporate ranges of pH, redox potential, DO, temperature, light, and other factors affecting P algal-availability. To create such a model, further sediment P assays incorporating varying environmental conditions are needed, as well seasonal observational data from Lake Allatoona.

5.3 Evaluation of Dredging Effects

Another objective of this research was to assess changes in water quality and algal biomass that may be caused by current dredging methods, including sediment resuspension and sediment removal. The results of the mesocosm study suggest that the removal of sediments could be an effective method of reducing internal loading and algal blooms.

The results also suggest that dredging, which induces sediment resuspension within the photic zone during the growing season, has the potential to contribute to the development of algal blooms. For this reason we suggest that future dredging be performed in a way that does not increase lake turbidity within the photic zone when conditions are favorable to algal growth (i.e., during the summer growing season).

Current dredging practice allows clay to return to the lake after traversing the holding ponds. The influence on sediment discharge may also increase phytoplankton activity if there is phosphorus release within the photic zone during the growing season.

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