ABSTRACT

Organic Matter Sources through Annual and Decadal Timescales
in a Polymictic Reservoir

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Phytoplankton productivity and allochthonous organic matter loading fluctuate through various timescales in response to changing environmental conditions. Organic matter compositional shifts can influence ecosystem structure and function by altering energy and nutrient flow pathways and turnover times. Organic matter carbon and nitrogen elemental and isotopic ratios were analyzed from sediment traps and a sediment core to determine organic matter provenance through annual and decadal timescales and to describe sediment transport mechanisms influencing organic matter delivery. Carbon to nitrogen ratios (C:N) indicated that allochthonous sources contributed much organic matter both annually and throughout the reservoir's life. Annually, C:N and carbon isotope ratios ($\delta^{13}C$) suggested a shift from primarily allochthonous organic matter in winter towards primarily autochthonous organic matter thereafter. Nitrogen isotope ratios ($\delta^{15}N$) indirectly recorded this seasonal organic matter source shift through nitrate utilization degree by phytoplankton. Decadally, C:N, $\delta^{13}C$, and $\delta^{15}N$ identified a shift from relatively large allochthonous organic matter contributions early towards increasing autochthonous organic matter contributions as the reservoir aged. Carbon, nitrogen, and phosphorus sediment concentrations recorded increasing phytoplankton productivity through time concurrent with phosphorus enrichment. $\delta^{13}C$ and $\delta^{15}N$ also suggested enhanced phytoplankton productivity with reservoir age. Urban growth and dairy operation intensification were possible sources of elevated external phosphorus loading leading to eutrophication. Phytoplankton productivity – $\delta^{13}C$ relationships opposed those advanced from stratified natural lakes possibly resulting from continuous water column mixing. However, variable isotopic compositions of dissolved inorganic nutrients may have influenced phytoplankton isotopic composition. Isotope ratio mixing models suggested that river plume sedimentation, sediment resuspension, and horizontal advection influenced organic matter delivery with individual mechanisms being more important seasonally. Linear regression models identified river discharge and wind-induced mixing as dominate factors influencing secondary sediment transport in the riverine and lacustrine regions respectively. Wind-induced mixing entrained deep-water advective river sediments into the photic zone rather than resuspending surface sediments. These findings suggest that allochthonous sources contribute much potential energy and nutrients annually and decadally in this reservoir and secondary sediment transport mechanisms influence organic matter delivery and potentially bioavailability.

Organic Matter Sources through Annual and Decadal Timescales
in a Polymictic Reservoir

by

Christopher Thomas Filstrup, B.S.

A Dissertation

Approved by the Department of Biology

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Submitted to the Graduate Faculty of
Baylor University in Partial Fulfillment of the
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of
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As an undergraduate at the University of Texas – Austin, I recall seeing a course entitled *Limnology and Oceanography* in the course catalogue. I thought *oceanography* sounded very interesting. However, what did a kid from the Midwest know about oceans? More importantly, what the heck is *limnology*? Naturally, I looked up the definition of *limnology* in my collegiate dictionary. *Limnology* is the study of biological, chemical, and physical features of lakes and other bodies of fresh water. Really! I can study this and potentially make a career of it? In my hometown of St. Joseph, Michigan, we did these things for fun. Thanks to this class and the teachings of Dr. Bassett Maguire Jr., I find myself here today.

I enrolled at Baylor University specifically to study with Dr. Owen T. Lind. I originally enrolled to earn a Master's degree with an expected duration of two years. Here I find myself nine years later. I don't think either of us anticipated that it would take this long. I thank you for devoting nearly a decade of your life to mentoring me. Through all the difficulties, you offered unwavering loyalty and encouragement. You also allowed me the freedom to find my own path. I believe I am a better scientist because of this.

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I have formed several personal relationships at Baylor University that made life tolerable for the past decade. Brad Christian reminded me that you can find a brother in the strangest place. Our friendship can best be described as that between either David Spade and Chris Farley or Forrest Gump and Bubba. I recall saying the Forrest Gump line "We are not of relations." at a Baylor Bears football game to which we got some laughs. Jeff Scales, Michael "Monkey" Mellon, Shannon Hill, and Sharon Conry also served as members of the original gang. Someday, we will all have doctorates. Holy cow! What were the Las Vegas odds on that? J. Thad Scott and Jeff Back were two colleagues that offered invaluable critiques of my research. It was a luxury to bounce ideas off of these two great scientific minds.

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DEDICATION

To Mom,

This document is a reflection of your dedication, patience, and love.

To this day, you remain *the wind beneath my wings.*

CHAPTER ONE

Introduction and Background

Organic Matter Composition

Organic matter primarily originates from two sources in aquatic ecosystems. Organic matter can be produced by primary producers, such as algae and macrophytes, within aquatic systems. This organic matter is referred to as autochthonous organic matter. In reservoirs, phytoplankton serves as the chief primary producer because abiogenic turbidity and large water-level fluctuations limit attached algae and macrophyte production (Kimmel and Groeger 1984; Kimmel et al. 1990). Organic matter can also be produced within the watershed and subsequently transported from the terrestrial landscape to aquatic systems. This external source of organic matter is termed allochthonous organic matter. Allochthonous organic matter loading from the watershed may be high in reservoirs compared to natural lakes because reservoirs are characterized by relatively larger watersheds and greater inflows (Thornton 1990b).

Traditionally, scientists have accepted that autochthonous organic matter production primarily supports aquatic food webs. Forbes (1887) postulated that lakes are ecosystems unto themselves and are completely isolated from surrounding terrestrial landscapes. However, limnologists have long recognized energy (organic matter) and nutrient transfers from watersheds to lakes thereby creating intimate linkages between lakes and surrounding terrestrial ecosystems (Wetzel 2001). Several studies (del Giorgio and Peters 1993; Cole et al. 1994; del Giorgio and Peters 1994; del Giorgio et al. 1997;

Cole et al. 2002) have demonstrated that community respiration commonly exceeds phytoplankton production in lakes. Allochthonous organic matter subsidizes autochthonous organic matter in supporting aquatic food webs in these systems.

It is requisite to characterize the organic matter base of lakes and reservoirs to adequately understand ecosystem structure and function. According to the "bottom – up" model of trophic structure regulation, the health of all trophic levels, and ultimately overall ecosystem health, depends on this autochthonous and allochthonous organic matter base. The relative proportions of organic matter originating from autochthonous and allochthonous sources may fluctuate through annual and decadal timescales. Shifts in organic matter composition influence energy (i.e. potential energy stored within the chemical bonds of reduced carbon compounds) flow through food webs, secondary producer trophic structure, and nutrient cycling including nutrient turnover times. For example, aquatic ecosystems with predominately autochthonous organic matter bases likely transfer much energy through a producer – primary consumer – secondary consumer food chain, such as a *Scenedesmus* sp. – *Daphnia* sp. – roach – heron food chain, whereas detrital pathways and the "microbial loop" transfer substantial energy in ecosystems receiving much allochthonous organic matter.

Trophic State Succession

Odum (1969) defined ecological succession as an orderly, directional process of community development resulting from environmental modification by the community that eventually culminates in a stabilized ecosystem. Lake succession is often confused with lake ontogeny, which is the gradual filling of lake basins with sediment that transforms the lake into swamp followed by a marsh and eventually a return to terrestrial conditions (Wetzel 2001). Lake succession involves the gradual enrichment of a lake from a less productive oligotrophic system to a more productive eutrophic system. Although this eutrophication process appears to contradict succession models from terrestrial ecosystems, Odum (1969) noted that external forcing factors, specifically nutrient loading from the watershed, disrupt ecological succession in lakes effectively reverting/maintaining the ecosystem in a younger "bloom" successional stage (i.e. *r*-selected species, low diversity, and low biomass to production ratios).

As reservoirs age, phytoplankton productivity fluctuates in response to changing environmental conditions. Two models, which differ based on factors limiting primary productivity, have been proposed concerning trophic state succession. The nutrient-limitation model predicts that phytoplankton productivity increases after reservoir impoundment fueled by a large terrestrial nutrient influx and decreases as these nutrients are exhausted (Kimmel and Groeger 1986; Holz et al. 1997). In contrast, the light limitation model predicts reduced phytoplankton productivity after reservoir impoundment due to increased inorganic turbidity and decreased light penetration (Hecky and Guildford 1984; Tadonléké et al. 2000). Following these initial trophic disequilibrium periods, several pathways exist depending on internal and external forcing factors.

Reservoirs are engineered systems constructed for anthropogenic purposes, such as potable water supplies, irrigation, commercial fish farming, and recreation. Improved understanding of trophic state succession improves management strategies designed for these purposes. Historically, trophic state succession received considerable interest because the initial highly productive period provided unrealistic expectations for sustained fisheries production (Kimmel and Groeger 1986). Identifying the current stage of trophic succession allows realistic targets to be set. Also, predicting possible trajectories enables mitigation efforts to be implemented before undesirable conditions manifest themselves.

Sediment Transport

Sediment transport mechanisms influence sediment distribution patterns and organic matter delivery in lakes and reservoirs. Seasonal differences in sediment transport mechanisms may influence both the quantity and quality of organic matter being deposited. Also, understanding of basin-wide deposition patterns aids interpretation of sediment cores to infer trophic state succession.

Numerous sediment distribution processes exist in lakes and reservoirs. Hilton et al. (1986) identified ten mechanisms of sediment distribution, including random redistribution of sediment and river plume sedimentation. Random redistribution of sediments involves continuous resuspension of bottom sediments by wave action with subsequent deposition being a random function of depth (Hilton et al. 1986). This mechanism is the dominant resuspension mechanism in several shallow lakes (Luettich et al. 1990; Hawley and Lesht 1992; Weyhenmeyer et al. 1997; Douglas and Rippey 2000). River plume sedimentation involves river water flowing as a discrete current through lakes at a depth dictated by water density with deposition of allochthonous materials occurring at distance from river inflow (Hilton et al. 1986). Density differences can be attributed to temperature, total dissolved solids (salinity), or suspended solids (Ford 1990). An additional mechanism, horizontal advection, involves the horizontal transport of autochthonous and allochthonous materials by a horizontal current system typically produced by river inflow, outflow, or wind shear in reservoirs (Ford 1990).

Sediment resuspension influences ecosystem function through increased turbidity and nutrient availability (Bengtsson et al. 1990). Sediment resuspension increases water column turbidity and therefore the potential for light limitation of phytoplankton productivity in lakes and reservoirs. Hellström (1991) demonstrated this relationship in a shallow natural lake. In turbid reservoirs, sediment resuspension deteriorates the already poor light environment. Also, sediment resuspension re-exposes previously unavailable organic matter to aquatic food webs (Meyers and Eadie 1993; Meyers and Ishiwatari 1993). This "second chance" energy and nutrient source can help support secondary production.

Statement of Problems

Improved knowledge of ecosystem structure and function requires better understanding of primary productivity and detrital organic matter provenance (Wetzel 2001). Autochthonous and allochthonous detritus may maintain ecosystem metabolic stability during short-term environmental and productivity fluctuations (Wetzel 2001). Therefore, further documentation of phytoplankton productivity and allochthonous organic matter loading fluctuations through annual and decadal timescales is required to understand lake ecosystems. This type of study would logically benefit from improved understanding of organic matter delivery (sediment transport) mechanisms. Also, stable isotopes have been an effective tool to elucidate food web dynamics and infer historical productivity in thermally-stratified natural lakes. However, these techniques have not been successfully applied to continuously mixing, shallow reservoirs.

Research Objectives

My research addressed the following objectives. 1) Examine annual and decadal allochthonous organic matter loading fluctuations in a eutrophic reservoir. 2) Explore decadal phytoplankton productivity trends as a reservoir matures. 3) Describe seasonal secondary sediment transport mechanisms influencing organic matter delivery in a polymictic reservoir. 4) Describe phytoplankton isotopic composition – productivity relationships in a shallow, rapidly-flushed reservoir.

Study Site

Lake Waco (31° 34' 28" N 97° 13' 13" W), which is located in McLennan County, Texas, USA, provides an ideal system in which to address these research objectives. Lake Waco is a relatively shallow ecosystem (mean depth = 6.4 m; maximum depth = 21.9 m) compared to other Texas reservoirs. The water column mixes completely from surface to bottom (holomictic) and continuously throughout the year with infrequent, brief thermal stratification (continuous polymictic). High sediment loading rates and sediment resuspension significantly contribute to high inorganic turbidity. The large watershed (120 watershed:lake area ratio) and predominately agricultural land uses create the potential for large allochthonous organic matter and nutrient loads from the watershed. Lake Waco is experiencing high phytoplankton production (eutrophication) resulting from these high nutrient loads.

Chapter Summaries

This thesis is sectioned into five chapters including the introduction (Chapter One) and conclusions (Chapter Five). The majority of content is contained within

Chapters Two, Three, and Four, which correspond to the three studies performed to address my research objectives. A brief summary of these chapters follows.

In Chapter Two, I describe spatiotemporal sediment resuspension dynamics. Total, inorganic, and volatile (organic) suspended solids deposition rates were determined by deploying sediment traps semi-monthly at four sampling stations during an annual cycle. Sediment resuspension was defined as gross sedimentation (deposition rates from a bottom sediment trap placed one meter above the sediment – water interface) minus net sedimentation (deposition rates from a photic sediment trap placed at four meters depth). Linear regression models were used to identify the primary factors (wind-induced mixing and river discharge) influencing sediment resuspension for the entire lake and at each sampling station.

In Chapter Three, I elucidate seasonal organic matter source shifts and sediment transport mechanisms. Carbon to nitrogen ratios (C:N) and carbon and nitrogen stable isotope ratios ($\delta^{13}C$ and $\delta^{15}N$) were measured on sedimenting particulate organic matter collected in photic and bottom sediment traps from a lacustrine station described in the previous study. Reservoir and natural lake characteristics that potentially contributed to our atypical seasonal patterns were acknowledged. Isotope ratio mixing models were used to identify secondary sediment transport mechanisms (sediment resuspension, horizontal advection, and river plume sedimentation) contributing excess particulate organic carbon and particulate organic nitrogen at our sampling location. Stable isotope mixing models allowed me to clarify conclusions from the previous study, which assumed that excess sediment deposition resulted entirely from sediment resuspension.

In Chapter Four, I chronicle decadal shifts in phytoplankton productivity and organic matter sources as the polymictic, sub-tropical reservoir aged. Organic matter elemental ratios (C:N and N:P) and isotopic ratios ($\delta^{13}C$ and $\delta^{15}N$) were analyzed from a dated sediment core that chronicled 35 years of reservoir and watershed history. Phytoplankton productivity proxies (primarily $\delta^{13}C$ and $\delta^{15}N$) were used to determine if paleolimnological techniques recorded a polymictic reservoir's eutrophication. Changing land use practices within the watershed that potentially influenced increasing phytoplankton productivity through time were identified.

CHAPTER TWO

Sediment Transport Mechanisms Influencing Spatiotemporal Resuspension Patterns in a Shallow, Polymictic Reservoir

Abstract

Although whole-lake sedimentation models have been developed for natural lakes, they may not apply to reservoirs due to differing physical and morphological characteristics along a reservoir's longitudinal axis. We measured sedimentation rates below the photic zone and near the sediment surface from a polymictic reservoir's riverine and lacustrine regions to identify transport mechanisms influencing sediment resuspension during an annual cycle. Lake-wide regression models revealed that wind-induced mixing depths explained <20% of sediment resuspension variability. However, site-specific mixing depth models explained 30% of sediment resuspension variability at a lacustrine station. Inverse relationships between mixing depth and sediment resuspension suggested that wind-induced mixing entrained deep-water advective river sediments into the photic zone rather than resuspending surface sediments. During our study, maximum mixing depths calculated from wind gusts never exceeded site depths which supported this hypothesis. Lake-wide and site-specific mixing depth models were not improved by considering short-duration, strong wind events suggesting that episodic winds did not generate enough momentum to effectively deepen mixing depths. Lake-wide regression models indicated that river discharge (r^2=0.19) was a better predictor of sediment resuspension than mixing depth. Site-specific discharge models explained 44 and 30% of sediment resuspension variability at riverine stations which emphasized the

influence of horizontal advection in riverine regions. River discharge and wind-induced mixing influenced sediment resuspension at one sampling station only indicating that the site may have been located in the transition region. Future reservoir sedimentation models should incorporate weighting factors to appropriately represent sediment transport mechanisms along a reservoir's longitudinal axis.

Introduction

Because sediment resuspension can strongly influence water column turbidity and nutrient concentrations in shallow lakes (Bengtsson et al. 1990), several whole-lake models have been developed to predict sedimentation processes in these systems. The dynamic ratio model estimates lake area subject to erosion/transportation or accumulation processes using only lake area and mean depth (Håkanson 1982). The exposure model uses either maximum fetch distances or lake exposure (i.e. the circular integral of fetch) as wave energy estimators to predict sediment distribution (Rowan et al. 1992). Carper and Bachmann (1984) calculated the wave base (i.e. mixing depth) from wind velocity and effective fetch using wave theory. The authors assumed the wave base equaled one-half of the wavelength.

However, whole-lake sedimentation models developed from natural lakes may not adequately characterize sedimentation processes in reservoirs which are more spatially dynamic than natural lakes. Reservoirs have been described as "river-lake hybrids" because they have intermediate morphological/hydrological characteristics between rivers and natural lakes (Kimmel et al. 1990). Hilton et al. (1986) stressed that different sediment transport mechanisms can dominate at different times and several may occur simultaneously in lakes. In reservoirs, different transport mechanisms can also drive sedimentation patterns among reservoir regions (Ford 1990) which complicates model application. For example, wind mixing and horizontal advection may strongly influence sedimentation patterns in the lacustrine and riverine regions respectively (Thornton 1990a).

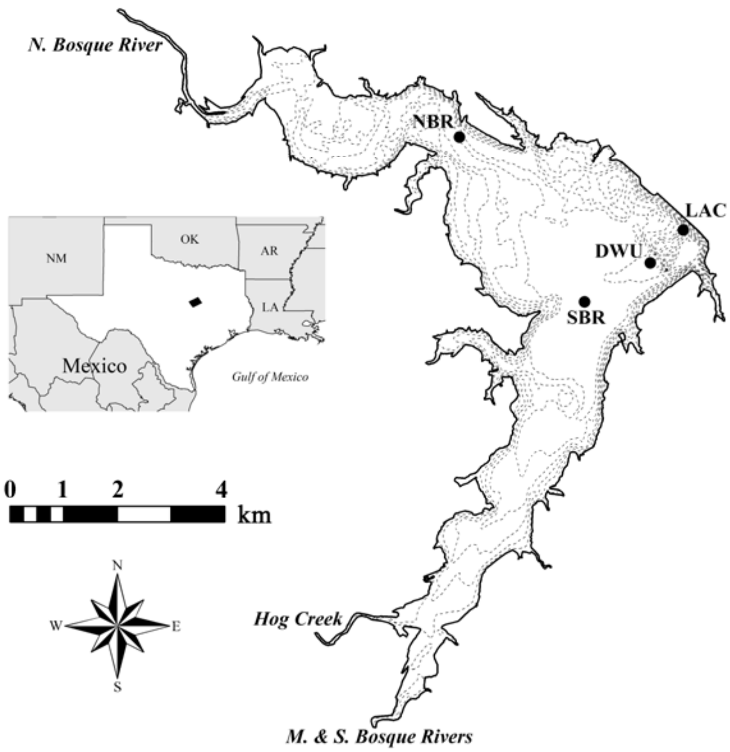

Figure 2.1: Lake Waco bathymetric map indicating North Bosque River (NBR), South Bosque River (SBR), deep-water upwelling (DWU), and lacustrine (LAC) sampling stations. Contour lines represent 5 ft (1.5 m) depth intervals and become deeper near dam. Bathymetry data were post-processed from original source (Texas Water Development Board 2003). Inset map indicated McLennan County, Texas.

Do whole-lake models adequately describe sediment resuspension in shallow, polymictic reservoirs? To answer this question, we measured sedimentation rates at the photic zone base and near the sediment surface of riverine and lacustrine sampling locations for an annual cycle. We identified factors influencing sediment resuspension by relating percent resuspension to local wind patterns and river discharge. We hypothesized that sediment resuspension would be largely determined by wind/wave mixing in the lacustrine region. We also hypothesized that northwesterly winter winds and southerly-southeasterly winds during the rest of the year would enhance sediment resuspension at southeasterly and northwesterly sampling locations respectively. We hypothesized that short-duration, strong wind events (i.e. storms) would intensely affect sediment resuspension. Finally, we hypothesized that river discharge would strongly influence sediment resuspension in the riverine regions.

Table 2.1: Morphometric and watershed characteristics of Lake Waco. Water residence time calculated as 20 yr average (1980 – 2001 excluding 1990 and 1991 when river discharge was not recorded).

Characteristic	Value
Conservation pool elevation (m a.s.l.)	141
Surface area ($\times 10^7$ m^2)	3.53
Mean depth (m)	6.4
Maximum depth (m)	21.9
Volume ($\times 10^8$ m^3)	2.26
Water residence time (yr)†	0.37
Perimeter ($\times 10^5$ m)	1.12
Lake watershed area ($\times 10^5$ ha)	4.24
N. Bosque River watershed area ($\times 10^5$ ha)	3.19
Watershed / lake area	120

Methods

Study Site

Lake Waco (31° 34' 28" N 97° 13' 13" W) is a polymictic, sub-tropical reservoir located in McLennan County, Texas, USA (Fig. 2.1). The reservoir supplies potable water to Waco and surrounding communities. Lake Waco is a medium-sized, relatively shallow reservoir with a brief water residence time (Table 2.1). The North Bosque River provides ~75% of inflow (Lind and Barcena 2003) because of its proportionally large watershed (Table 2.1). Additional tributaries include the Middle and South Bosque Rivers and Hog Creek. Lake Waco's tributaries form dual reservoir axes that are oriented to prevailing regional wind patterns (Fig. 2.1). The northern arm (North Bosque

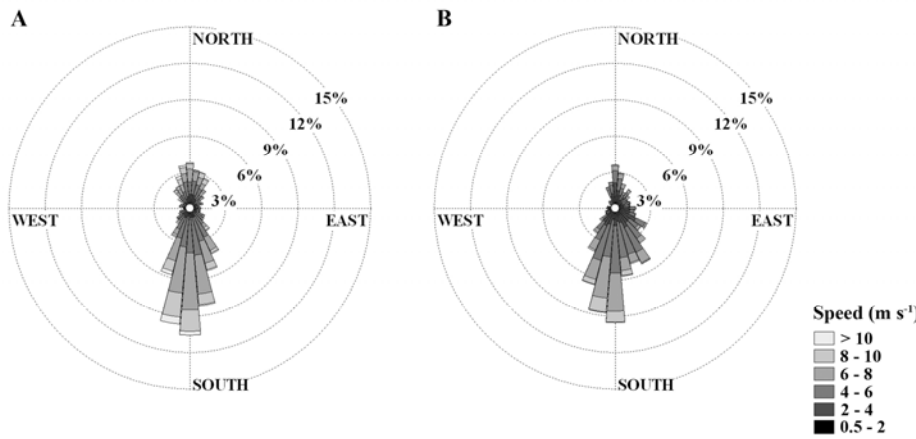

Figure 2.2: Local wind patterns during study (National Oceanic and Atmospheric Administration National Climatic Data Center 2009). Data were temporally grouped from A) November 2003 – May 2004 and B) June – October 2004 to show contrasting wind patterns.

River) aligns with northwesterly winds during winter, while the southern arm (Middle and South Bosque Rivers and Hog Creek) aligns with southwesterly winds during other seasons. Advective mixing from wind/wave action maintains persistent holomictic and polymictic circulation patterns although brief periods of thermal stratification have been reported (Lind 1971; Kimmel and Lind 1972). Lake Waco is prone to sediment resuspension due to its shallowness, orientation, and lack of wind-abating structures in the watershed.

Southerly winds predominated throughout our study (Fig. 2.2; WBAN 13959; National Oceanic and Atmospheric Administration National Climatic Data Center 2009). However, northerly and northeasterly winds were prevalent from November 2003 – May 2004 (Fig. 2.2A). Wind speeds >10 m s^{-1} were more common during this period than during the rest of our study. Wind speeds averaged 4.7 m s^{-1} with calm conditions (<0.5 m s^{-1}) occurring ~6% of the time from November 2003 – May 2004. Southeasterly winds occurred more frequently from June – October 2004 (Fig. 2.2B). Winds were gentler during this period (avg. = 3.7 m s^{-1}) with a greater occurrence of calm days (~8%).

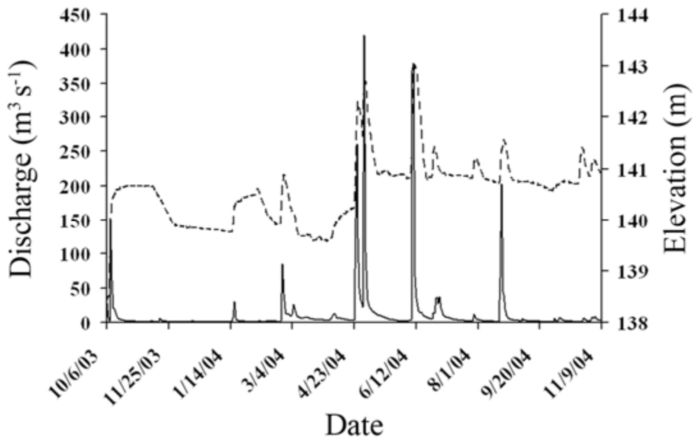

Figure 2.3: North Bosque River discharge (solid line; US Geological Survey Water Data for USA 2008) and Lake Waco water surface elevation (broken line; US Army Corps of Engineers-Ft. Worth District Hydrologic Data on Ft. Worth District Lakes 2008) during study.

North Bosque River discharge (Station No. 08095200; US Geological Survey Water Data for USA 2008) experienced several large peaks during our study (Fig. 2.3). North Bosque River discharge peaked (>100 $m^3 s^{-1}$) on April 25 (~260 $m^3 s^{-1}$), May 1 (~420 $m^3 s^{-1}$), June 10 (~380 $m^3 s^{-1}$), and August 20 (~200 $m^3 s^{-1}$). Lake Waco water surface elevation (US Army Corps of Engineers-Ft. Worth District Hydrologic Data on Ft. Worth District Lakes 2008) increased concurrent with peak discharge events on April 25 and May 1 (Fig. 2.3). Water surface elevation remained elevated thereafter.

Field and Laboratory Methods

We designated four sampling locations representing Lake Waco's different regions to adequately describe sediment resuspension in the reservoir. The North Bosque River (NBR) and South Bosque River (SBR) stations were located in the riverine/transition regions of the northern and southern arms respectively (Fig. 2.1). NBR was the shallowest location (z_{mean} = 8.7 ± 0.4 m) and was closest to North Bosque River inflow. SBR (z_{mean} = 10.5 ± 0.5 m) was closest to Middle and South Bosque River and Hog Creek inflow. The Lacustrine (LAC) and Deep Water Upwelling (DWU) stations were located in the lacustrine region (Fig. 2.1). LAC was the deepest location (z_{mean} = 15.0 ± 0.5 m) and farthest from tributary inflow. We believe DWU (z_{mean} = 11.6 ± 0.4 m) was a location of water upwelling resulting from its proximity to a submerged, former dam face. Lake Waco, which was filled in 1965, impounded a smaller reservoir that lost most of its storage capacity due to sedimentation.

We deployed buoy-carried, anchored sediment traps semi-monthly or monthly for approximately one week duration from November 2003 through October 2004. Cylindrical sediment traps were manufactured from 5.08-cm inner diameter polyvinyl chloride (PVC) pipes. Sediment traps had two vertical arms connected by a horizontal pipe to provide increased depositional surface area and balance in the water column. Sediment traps had 10:1 height:diameter ratios (H:D) similar to those used by Douglas and Rippey (2000). Sediment traps with H:D>5 have been recommended to avoid material loss (Bloesch and Burns 1980; Blomqvist and Håkanson 1981). We suspended sediment traps at two depths per station. Photic sediment traps were suspended at 4 m depth (average photic depth of Lake Waco's lacustrine region) to collect inorganic particles sedimenting from the upper mixed layer and organic matter produced within the reservoir. Bottom sediment traps were suspended one meter above the sediment-water interface to collect the aforementioned particles plus materials being secondarily-transported within the reservoir. Secondarily-transported materials include particles resuspended from the sediment surface, particles experiencing horizontal advection (i.e. sediment focusing), and particles sedimenting from river inflow (i.e. river plume sedimentation; Hilton et al. 1986; Ford 1990).

Sediment trap contents were emptied into polypropylene bottles and stored at 4°C. Sediment slurry aliquots were filtered onto pre-weighed glass fiber filters. Samples were dried at 104°C until reaching constant weights to calculate total suspended solids (TSS). Samples were combusted at 550°C to determine inorganic suspended solids (ISS). Volatile suspended solids (VSS) were calculated as the difference between TSS and ISS using the loss-on-ignition method (Dean 1974). Areal sedimentation rates (g m^{-2} d^{-1}) were calculated based on dry weights. Photic trap sedimentation rates measured net sedimentation, which consisted of allochthonous particles originating from the watershed and autochthonous in-lake production (James and Barko 1993). Bottom trap sedimentation rates measured gross sedimentation, which consisted of net sedimentation plus sediments that were secondarily-transported within the reservoir (James and Barko 1993). We defined resuspension and percent resuspension as follows:

$$R = S_{gross} - S_{net} \tag{1}$$

and

$$\%R = [(S_{gross} - S_{net}) / S_{gross}] \times 100\% \tag{2}$$

where R = resuspension (g m^{-2} d^{-1}), %R = percent resuspension (%), S_{gross} = gross sedimentation rates (g m^{-2} d^{-1}), and S_{net} = net sedimentation rates (g m^{-2} d^{-1}). Although we calculated site-specific resuspension variables, we acknowledge that resuspension rates and percentages included resuspended sediments horizontally transported from other locations.

Mixing Depth Calculations

We used ArcGIS 9.3 software (ESRI, Inc.) to measure fetch distances at 10° wind angle intervals for each sampling station. We calculated effective fetch distances following US Army Corps of Engineers (1962) to account for relatively narrow fetch widths common in canyon reservoirs. This method includes fetch distances ±45° at 6° intervals from the desired wind angle. Wavelength and wave period were calculated according to the following equations:

$$L = (g\,T^2) / 2\pi \tag{3}$$

and

$$(g\,T) / (2\pi\,U) = 1.2 \tanh [0.077\,((g\,F) / U^2)^{0.25}] \tag{4}$$

where L = wavelength (m), g = gravitational constant (9.8 m s^{-2}), T = wave period (s), U = wind velocity (m s^{-1}), and F = effective fetch (m; Bachmann et al. 2000). Wind speeds and directions were obtained for daily resultant winds and two-minute and five-second sustained wind maxima measured at Waco Regional Airport located adjacent to the reservoir (WBAN 13959; National Oceanic and Atmospheric Administration National Climatic Data Center 2009). We defined wind angles as direction from True North with North=0°, East=90°, South=180°, and West=270°. We calculated wavelengths using maximum sustained winds in addition to daily winds to account for storm events. Mixing depths were calculated as one-half the wavelength (Carper and Bachmann 1984).

Statistical Considerations

We used a one-way analysis of variance (one-way ANOVA) to identify significant differences in TSS percent resuspension among stations (time inclusive). We used linear regression to model TSS percent resuspension as functions of mixing depth, mixing depth-to-site depth ratios (z_{mix}:z_{site}), and North Bosque River discharge. Whole-lake and site-specific models were created for each independent variable. The aforementioned statistical analyses were evaluated at α=0.05. We used multiple linear regression (MLR) to model TSS percent resuspension as a function of z_{mix}:z_{site} and North Bosque River discharge at SBR. TSS percent resuspension was significantly influenced by both wind mixing and river discharge at SBR only. Stepwise selection was used with α=0.05 for variable entry and α=0.10 for variable removal. Prior to statistical analyses, data distribution normality were tested by the Shapiro-Wilk test statistic based on our low sample sizes (N<200). We used logarithmic and arcsine squareroot transformations to improve normality of positively skewed distributions and percentage data distributions respectively. Arcsine squareroot transformations typically improve proportional data normality by compressing middle values while spreading the ends (McCune and Grace 2002). Outliers, which were defined as values >1.5 interquartile ranges from Tukey's hinges, were identified and removed following data transformations. Statistical analyses were performed using SPSS 17.0 software (SPSS, Inc.).

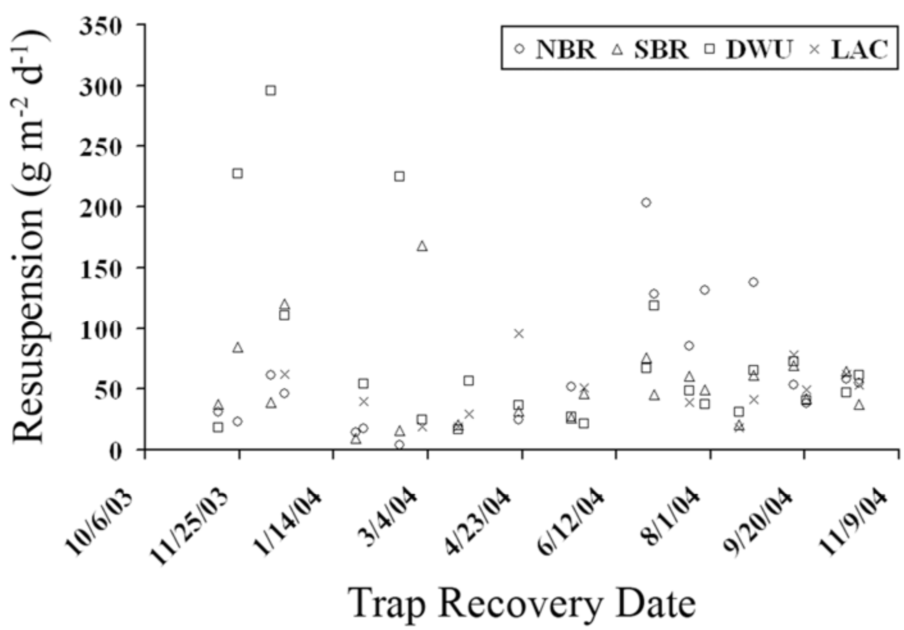

Figure 2.4: Total suspended solids resuspension rate (bottom trap sedimentation rate – photic trap sedimentation rate) through time at North Bosque River (NBR), South Bosque River (SBR), deep-water upwelling (DWU), and lacustrine (LAC) sampling stations.

Table 2.2: Annual averages (± standard deviations) for total suspended solids net and gross sedimentation rates (g m^{-2} d^{-1}), resuspension rates (gross – net sedimentation rates, g m^{-2} d^{-1}), and percent resuspension (resuspension rate / gross sedimentation rate, %) at North Bosque River (NBR), South Bosque River (SBR), deep-water upwelling (DWU), and lacustrine (LAC) sampling stations.

Station	Net sed rate	Gross sed rate	Resusp rate	% resusp
NBR	30.8 ± 19.4	91.9 ± 51.3	64.7 ± 53.0	60.4 ± 22.8
SBR	26.3 ± 17.4	80.6 ± 42.9	53.7 ± 36.9	64.6 ± 17.0
DWU	25.3 ± 19.8	102.9 ± 89.8	77.5 ± 75.7	71.2 ± 14.7
LAC	29.2 ± 19.7	71.6 ± 34.6	47.1 ± 22.7	66.4 ± 10.9

Table 2.3: Inorganic and organic percent composition of total suspended solids in photic and bottom sediment traps and resuspended materials at North Bosque River (NBR), South Bosque River (SBR), deep-water upwelling (DWU), and lacustrine (LAC) sampling stations. Values reported as annual averages ± standard deviations.

	Inorganic			Organic		
Station	Photic	Bottom	Resusp	Photic	Bottom	Resusp
NBR	81 ± 6	85 ± 4	88 ± 7	19 ± 6	15 ± 4	12 ± 7
SBR	79 ± 7	84 ± 4	88 ± 3	21 ± 7	16 ± 4	12 ± 3
DWU	80 ± 5	86 ± 3	89 ± 4	20 ± 5	14 ± 3	12 ± 4
LAC	81 ± 8	84 ± 5	85 ± 5	20 ± 8	16 ± 5	15 ± 5

Results

Sediment Resuspension

TSS resuspension rates varied greatly at each site during our study (Fig. 2.4 and Table 2.2). At NBR, TSS resuspension rates varied from 4 – 203 g m^{-2} d^{-1} with the highest rates occurring from June – August 2004 (Fig. 2.4). Contrastingly, TSS resuspension rates at SBR (9 – 168 g m^{-2} d^{-1}) and DWU (17 – 296 g m^{-2} d^{-1}) were typically greatest from November 2003 – February 2004. DWU had the three greatest resuspension rates lake-wide. TSS resuspension rates varied from 18 – 96 g m^{-2} d^{-1} at LAC. TSS resuspension rates were not significantly different among stations ($F_{3, 72}$=0.22, p=0.88) because of large within-site temporal variation. Resuspended sediments were composed of ~90% inorganic materials (Table 2.3).

Whole-lake and site-specific TSS resuspension percentages widely varied during our study (Fig. 2.5 and Table 2.2). From November 2003 – May 2004, resuspension patterns were spatially heterogeneous throughout Lake Waco. Lacustrine stations (DWU and LAC) typically had higher percentages of resuspended sediment than riverine stations (NBR and SBR; Fig. 2.5). TSS resuspension percentages were more similar spatially from June – October 2004. During this period, NBR commonly had the greatest or second greatest percentages of resuspended materials. NBR exhibited the greatest temporal variability (CV=38%) in resuspension percentages (Table 2.2). Lake-wide TSS resuspension percentages were highest during June 2004 (Fig. 2.5).

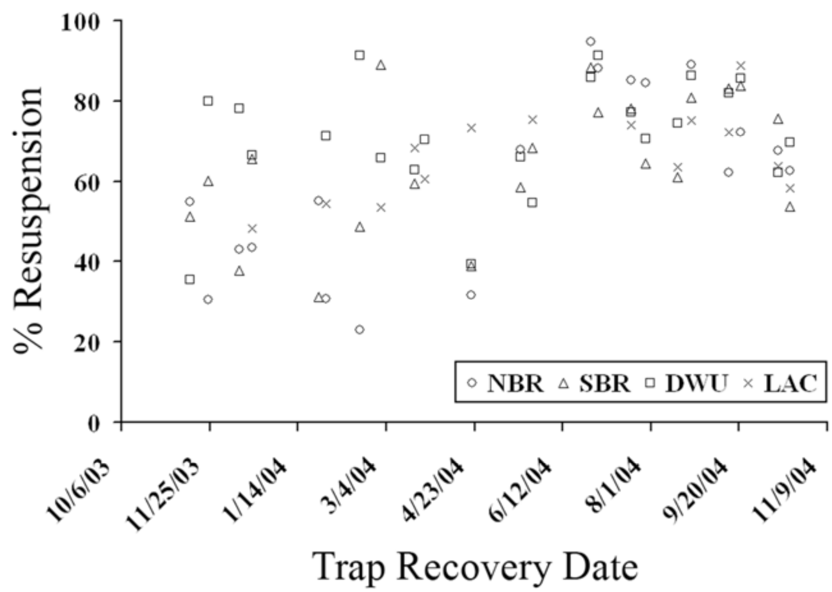

Figure 2.5: Total suspended solids percent resuspension (resuspension rate / bottom trap sedimentation rate) through time at North Bosque River (NBR), South Bosque River (SBR), deep-water upwelling (DWU), and lacustrine (LAC) sampling stations.

Sediment Resuspension Models

Mixing depth models did not adequately predict TSS resuspension percentages for the entire lake (Table 2.4 and Table 2.5). Average mixing depth and average z_{mix}:z_{site} based on daily resultant winds explained only 13 and 17% of TSS percent resuspension variability. However, mixing depth models were better at predicting TSS percent resuspension at DWU and SBR. Average mixing depth explained 30% of TSS percent resuspension variation at DWU (Table 2.4). Average z_{mix}:z_{site} explained 20% of TSS percent resuspension variability at SBR (Table 2.5). Unexpectedly, TSS resuspension percentages were inversely related to mixing depth predictors for whole-lake and site-specific models. Mixing depth models were not significant at NBR and LAC. Models were not improved by considering mixing depths based on two-minute or five-second sustained winds.

Table 2.4: Linear regression models of total suspended solids percent resuspension (arcsine squareroot transformed) as a function of average mixing depth (log_{10} transformed) based on daily resultant wind speeds and directions for the entire lake and at deep-water upwelling (DWU) sampling station. Models are presented for significant regressions (α=0.05) only. SE = Standard Error. SEE = Standard error of the estimate.

	Slope				
Station	Value	SE†	r^2	*p-value*	SEE††
Whole-lake	-0.21	0.06	0.13	0.002	0.11
DWU	-0.24	0.08	0.30	0.010	0.08

Table 2.5: Linear regression models of total suspended solids percent resuspension (arcsine squareroot transformed) as a function of mixing depth-to-station depth ratio (z_{mix}:z_{site}; log_{10} transformed) based on daily resultant wind speeds and directions for the entire lake and at South Bosque River (SBR) sampling station. Models are presented for significant regressions (α=0.05) only. SE = Standard Error. SEE = Standard error of the estimate.

	Slope				
Station	Value	SE†	r^2	*p-value*	SEE††
Whole-lake	-0.24	0.06	0.17	<0.001	0.11
SBR	-0.27	0.12	0.20	0.040	0.11

North Bosque River discharge models (r^2=0.19) explained slightly more TSS percent resuspension variability than mixing depth models (r^2=0.17) for the entire lake (Table 2.6 and Table 2.5). North Bosque River discharge also explained the most variability of TSS percent resuspension at the riverine stations (Table 2.6). North Bosque River discharge explained 44 and 30% of sediment resuspension at NBR and SBR respectively. SBR was the only sampling location where sediment resuspension was significantly influenced by mixing depth and river discharge. Multiple linear regression explained 47% of TSS percent resuspension as a function of North Bosque River discharge and z_{mix}:z_{site} at SBR (Table 2.7). River discharge models were not significant at DWU and LAC. Models were not improved by considering North Bosque River discharge prior to the deployment period (+3, 5, 7, and 10 days).

Table 2.6: Linear regression models of total suspended solids percent resuspension (arcsine squareroot transformed) as a function of average North Bosque River discharge ($log_{10} - log_{10}$ transformed) for the entire lake and at North Bosque River (NBR) and South Bosque River (SBR) sampling stations. Models are presented for significant regressions (α=0.05) only. Variable was not normally distributed in the whole-lake model. SE = Standard Error. SEE = Standard error of the estimate.

	Slope				
Station	Value	SE†	r^2	*p-value*	SEE††
Whole-lake‡	0.16	0.04	0.19	<0.001	0.11
NBR	0.29	0.08	0.44	0.003	0.12
SBR	0.18	0.07	0.30	0.011	0.10

Table 2.7: Multiple linear regression model of total suspended solids percent resuspension (arcsine squareroot transformed) as a function of average North Bosque River discharge ($log_{10} - log_{10}$ transformed) and z_{mix}:z_{site} (log_{10} transformed) based on daily resultant wind speeds and directions for the South Bosque River (SBR) sampling station. SE = Standard Error. SEE = Standard error of the estimate.

	Slope		
Variable	Value	SE†	*p-value*
MLR model (r^2=0.47, *p-value*=0.003, SEE††=0.09)			
NB discharge	0.17	0.06	0.008
z_{mix}:z_{site}	-0.25	0.10	0.026

Discussion

The dynamic ratio model (DR = √lake area / mean depth) identifies the importance of sediment erosion and transportation processes versus sediment accumulation processes in lakes (Håkanson 1982). Based on Florida lakes, lakes with DR>0.8 had sediment surfaces that were vulnerable to wave disturbance across the entire lake bottom (Bachmann et al. 2000). Central Texas reservoirs typically have DR≈0.8 and smaller surface areas than large natural lakes (Fig. 2.6). Lake Waco (DR=0.93) sediments should theoretically be prone to erosion and transportation processes across

<33% of the lake bottom (Håkanson 1982). However, dynamic ratio classifications may be biased in reservoirs due to morphological differences with natural lakes which tend to be more circular. The dynamic ratio does not consider either wind speeds or effective fetch distances. Reservoirs may behave similarly to large, shallow natural lakes concerning sedimentation processes if their longitudinal axes are oriented to prevailing wind patterns such as Lake Waco.

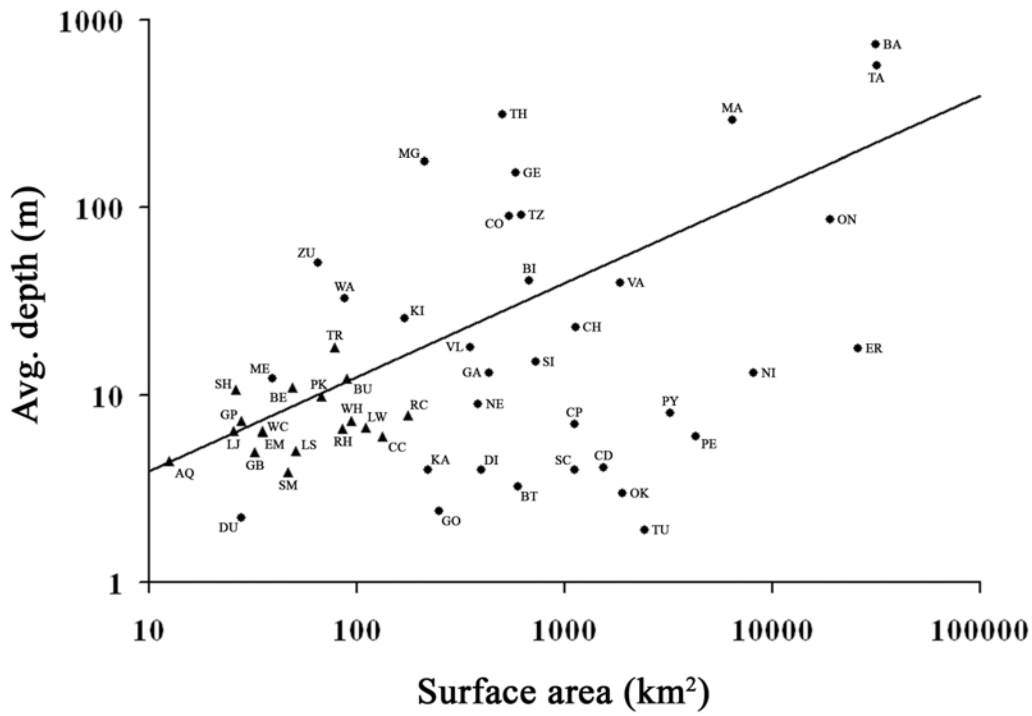

Figure 2.6: Central Texas reservoirs (triangles; Texas Water Development Board Reservoir Summary Report 2009) and large world natural lakes (circles; International Lake Environmental Committee World Lakes Database 2009) characterized by mean depth and surface area (modified from Havens et al. 2007). Trendline represents dynamic ratio (DR = √surface area (in km^2) / avg. depth (in m); Håkanson 1982) of 0.8 with lakes below this trendline (DR>0.8) being increasingly affected by wind/wave mixing. Central Texas reservoirs include Aquilla (AQ), Belton (BE), Buchanan (BU), Cedar Creek (CC), Eagle Mountain (EM), Granbury (GB), Grapevine (GP), Lewisville (LW), Limestone (LS), Lyndon B. Johnson (LJ), Possum Kingdom (PK), Ray Hubbard (RH), Richland-Chambers (RC), Somerville (SM), Stillhouse Hollow (SH), Travis (TR), Waco (WC), and Whitney (WH). World natural lakes include Baikal (BA), Biwa (BI), Balaton (BT), Chad (CD), Champlain (CH), Constance (CO), Chapala (CP), Dianchi (DI), Dong-hu (DU), Erie (ER), Gatiun (GA), Geneva (GE), George (GO), Kasumiguara (KA), Kinneret (KI), Malawi (MA), Mendota (ME), Maggiore (MG), Neagh (NE), Nicaragua (NI), Okeechobee (OK), Ontario (ON), Peipsi (PE), Poyang (PY), St. Clair (SC), Simcoe (SI), Tanganyika (TA), Tahoe (TH), Tai-hu (TU), Taupo (TZ), Vattern (VA), Valencia (VL), Washington (WA), and Zurich (ZU).

We hypothesized that wind/wave mixing primarily determined sediment resuspension in Lake Waco particularly in the lacustrine region. Mixing depth models did not support our hypothesis for the entire lake. Average mixing depth explained <20% of TSS percent resuspension variability (Table 2.4). We increased model explanatory power by considering depth differences between sites. However, z_{mix}:z_{site} still explained <20% of TSS percent resuspension variability (Table 2.5). The model's poor explanatory power suggested that other sediment transport mechanisms, such as horizontal advection, induced sediment resuspension. Bailey and Hamilton (1997) emphasized the importance of including horizontal circulation in resuspension models. Alternatively, poor model explanatory power may have indicated that simple linear regression could not describe sedimentation processes in this spatially dynamic system.

Site-specific models revealed that wind/wave mixing significantly influenced sediment resuspension in the lacustrine region. Average mixing depth explained 30% of TSS percent resuspension variability at DWU (Table 2.4). This location coincided with both the northern and southern reservoir axes (Fig. 2.1). Prevailing wind patterns oriented along these relatively long effective fetches may have induced sediment resuspension throughout the year. This hypothesis was supported by the significant regression between TSS percent resuspension and z_{mix}:z_{site} at SBR (r^2=0.20, p<0.05; Table 2.5) which had similar fetch orientation. Filstrup et al. (2009) identified sediment resuspension as a primary mechanism influencing gross sedimentation rates at DWU using organic matter isotopic composition mixing models. We believe that the nearby submerged dam face created localized upwelling areas that enhanced wind-induced sediment resuspension. Low sediment trap recovery rates (~60%) likely influenced sediment resuspension – mixing depth relationships at LAC.

Unexpectedly, sediment resuspension was inversely related to mixing depth in Lake Waco (Table 2.4 and Table 2.5). This result modified our conceptual model of sediment transport in Lake Waco. During horizontal transport, sediments likely settled and became concentrated with depth as distance from North Bosque River inflow increased. Wind-induced mixing likely entrained deep-water suspended sediments thereby creating a more homogeneous water column (i.e. smaller sedimentation rate difference between photic and bottom sediment traps). During our study, mixing depths never reached the sediment surface (z_{mix}:z_{site}<1) even when considering five-second sustained wind gusts. Therefore, wind-induced mixing likely did not resuspend surface sediments but rather mixed suspended sediments into the photic zone. Filstrup et al. (2009) documented the importance of horizontal advection on organic matter sedimentation at DWU which supports this proposed mechanism.

We hypothesized that northwesterly winter winds and prevailing southerly-southeasterly winds promoted sediment resuspension at southeasterly and northwesterly sampling locations respectively. TSS resuspension rates supported this hypothesis at stations where sediment resuspension was significantly influenced by wind-induced mixing. Resuspension rates were greatest at DWU and SBR from November 2003 – February 2004 (Fig. 2.4) when strong northerly winds were common (Fig. 2.2). Although NBR exhibited greater resuspension rates from June – October 2004 (Fig. 2.4) when winds were primarily from the south – southeast (Fig. 2.2), sediment resuspension and wind-induced mixing were not related at this site. TSS percent resuspension did not support this hypothesis. Greater TSS resuspension percentages at NBR during summer (Fig. 2.5) were largely influenced by North Bosque River discharge (See commentary below). TSS resuspension percentages at DWU were similar year-round despite abnormally low values in November and May (Fig. 2.5). At SBR, percent resuspension values were typically lower from November 2003 – May 2004 which opposed our hypothesis. We believe this discrepancy originated from the unexpected reciprocal relationship between mixing depth and sediment resuspension in Lake Waco.

We hypothesized that storms strongly influenced sediment resuspension in Lake Waco. The accentuated influence of strong wind events on sediment resuspension has been demonstrated in several natural lakes (Bengtsson et al. 1990; Luettich et al. 1990; Kristensen et al. 1992; Evans 1994; James et al. 2008). Linear regression models were not substantially improved by including mixing depth variables based on two-minute or five-second sustained winds. We selected these two variables to characterize storm events because they were simple and readily-available. We acknowledged that these theoretical maximum mixing depths were not likely to be realized in Lake Waco due to their short duration. However, mixing depths never exceeded site depths suggesting that our sampling sites were sediment transport regions not sediment erosion regions. According to the dynamic ratio, erosion occurred over less than a third of Lake Waco's sediment surface (Håkanson 1982). Sediment erosion regions may have occurred at shallower locations in the riverine regions with sediments being subsequently horizontally-advected throughout the reservoir.

Finally, we hypothesized that river discharge primarily determined sediment resuspension in the riverine regions. Large river discharge events along with wind speed explained suspended inorganic particle concentrations in a Swedish lake (Markensten and Pierson 2003). Evans (1994) suggested that river currents be added to mechanisms directly influencing resuspension. North Bosque River discharge was the greatest single predictor of site-specific sediment resuspension at NBR (r^2=0.44, p=0.003; Table 2.6) emphasizing the importance of river inflow on sediment transport in riverine regions. Both resuspension rates and percentages increased in June and remained elevated thereafter at NBR (Fig. 2.4 and Fig. 2.5). This shift was potentially a delayed response to the large discharge events in late-April through early-June and the resulting lake elevation rise (Fig. 2.3). Additionally, North Bosque River discharge explained more sediment resuspension variability than wind-induced mixing at SBR (r^2=0.30, p=0.011; Table 2.6). North Bosque River discharge may have served as an indicator of ungauged South and Middle Bosque River and Hog Creek discharge at SBR because their discharges are typically correlated. SBR may have been located in the transition region because it was the only station influenced by both river discharge and mixing depth (Table 2.7). Interestingly, lake-wide regression models revealed that river discharge (r^2=0.19, p<0.001; Table 2.6) predicted TSS resuspension percent better than wind-induced mixing (r^2=0.17, p<0.001; Table 2.5). These relationships suggested that our resuspension metrics may have been influenced by horizontal advection in addition to sediment resuspension especially in riverine regions.

Conclusions

Reservoir managers require models to adequately predict sedimentation processes and their influences on water quality. In Florida natural lakes, Bachmann et al. (2000) demonstrated that greater dynamic ratios indicated poor water quality using total phosphorus, total nitrogen, chlorophyll *a*, and Secchi depth as water quality indicators. Similar studies relating widely-applicable sedimentation models to water quality should be performed in reservoirs. However, sedimentation models require careful scrutiny and validation before reservoir application. Our findings suggested that models developed from natural lakes may not apply to reservoirs due to differing morphological and hydrological characteristics along reservoirs' longitudinal axes. We propose that future reservoir models incorporate weighting factors to account for these differing sediment transport mechanisms among reservoir regions.

Acknowledgments

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CHAPTER THREE

Allochthonous Organic Matter Supplements and Sediment Transport in a Polymictic Reservoir Determined Using Elemental and Isotopic Ratios

Abstract

Because allochthonous organic matter (OM) loading supplements autochthonous OM in supporting lake and reservoir food webs, C and N elemental and isotopic ratios of sedimenting particulate OM were measured during an annual cycle in a polymictic, eutrophic reservoir. Particulate organic C and N deposition rates were greatest during winter and lowest during spring. C:N ratios decreased through our study indicating that OM largely originated from allochthonous sources in winter and autochthonous sources thereafter. $\delta^{13}C$ were influenced by C_4 plant signatures and became increasingly light from winter through autumn. $\delta^{15}N$ indirectly recorded the OM source shift through nitrate utilization degree with maximum values occurring in May as nitrate concentrations decreased. Unlike relationships from stratified systems, $\delta^{13}C$ decreased with increasing algal biomass. This relationship suggests that minimal inorganic C fixation relative to supplies maintained photosynthetic isotopic discrimination during productive periods. Water column mixing likely maintained adequate inorganic C concentrations in the photic zone. Alternatively, OM isotopic composition may have been influenced by changing dissolved inorganic nutrient pools in this rapidly flushed system. $\delta^{15}N$ also recorded increased N_2 fixation as nitrate concentrations declined through autumn. Secondary sediment transport mechanisms strongly influenced OM delivery. Particulate organic C and N deposition rates were 3× greater near the sediment-

water interface. Isotopic ratio mixing models suggested that river plume sedimentation, sediment resuspension, and horizontal advection influenced excess sediment deposition with individual mechanisms being more important seasonally. Our findings suggest that allochthonous OM loading and secondarily-transported OM seasonally supplement phytoplankton production in productive reservoirs.

Introduction

Organic matter (OM) produced within watersheds (allochthonous) can be transported to lakes and reservoirs where it supplements OM produced within lakes (autochthonous). The traditional paradigm of autochthonous OM supporting aquatic food webs has recently been scrutinized. Traditional aquatic food web studies ignore large allochthonous OM loading rates of some systems or assume allochthonous OM is recalcitrant (Cole et al. 2002). Community respiration exceeds C fixation by algae in lakes worldwide (Cole et al. 1994). In these net heterotrophic systems, community respiration must be subsidized by allochthonous OM (Cole et al. 2002). Allochthonous OM supplements are thought less important in more productive systems. Algal production typically approaches or exceeds community respiration in eutrophic lakes worldwide (del Giorgio and Peters 1993; del Giorgio and Peters 1994). However, allochthonous OM contributions may be seasonally important in eutrophic reservoirs. Minor allochthonous contributions can maintain consumers during periods of low autochthonous production (Vander Zanden and Sanzone 2004). Temperate lake models suggest baseline respiration rates of allochthonous OM exist independently of algal production (del Giorgio et al. 1999), but bacteria rapidly respond to short periods of algal-derived DOC even in oligotrophic systems (McCallister and del Giorgio 2008).

Reservoirs may receive greater allochthonous loads than lakes because of relatively large watersheds and greater inflows (Thornton 1990b).

Understanding both OM transport and processing in lakes and reservoirs is critical to accurately determining global C budgets (Lehmann et al. 2002; Downing et al. 2006; Downing et al. 2008). Reservoir sediments accumulate 160 Tg organic C yr^{-1}, roughly 4× that of natural lakes (Dean and Gorham 1998). Downing et al. (2008) estimated that medium-sized reservoirs may bury 2× more organic C than previously thought. However, little of the C fixed in lakes and reservoirs may be preserved in sediments (Meyers et al. 1984; Meyers 1994). Secondary sediment resuspension and transport mechanisms re-expose previously sedimented OM to bacterial and benthic utilization (Meyers and Eadie 1993; Meyers and Ishiwatari 1993). Therefore, C burial efficiencies of lakes and reservoirs are potentially reduced. Secondary transport of previously deposited particles complicates sediment deposition rate measurements (Bernasconi et al. 1997) and C burial calculations.

C and N elemental and isotopic ratios are routinely used to determine OM sources and biogeochemical processing in natural lakes. C:N ratio source distinctions are well-documented. Vascular land plants are largely composed of carbohydrates, such as cellulose and lignin, for structural support though phytoplankton is not (Meyers and Ishiwatari 1993). Vascular land plants have C:N ratios ≥20, whereas phytoplankton has C:N ratios between 4 and 10 (Meyers and Teranes 2001). The application of isotopic ratios to understand biogeochemical C and N cycling is complex. OM $\delta^{13}C$ and $\delta^{15}N$ have been shown to record phytoplankton productivity (Cifuentes et al. 1988; Schelske and Hodell 1995; Hodell and Schelske 1998) and nitrate utilization (Altabet and Francois

1994; Teranes and Bernasconi 2000) respectively in marine and stratified natural lake systems. Sedimenting phytoplankton preferentially removes ^{12}C and ^{14}N from the epilimnion (Hodell and Schelske 1998). Phytoplankton becomes isotopically heavy as epilimnetic CO_2 and NO_3 concentrations are depleted because isotopic discrimination is reduced (Hodell and Schelske 1998). However, these relationships may not apply to polymictic systems due to persistent water column mixing. More studies documenting seasonal variations of OM isotopic compositions are required to understand C and N cycling (Bernasconi et al. 1997; Gu et al. 2006) particularly in polymictic systems.

Additionally, system differences between reservoirs and natural lakes may complicate application of stable isotope techniques, which were largely advanced from natural lakes, to reservoirs. Reservoirs have been called "river-lake hybrids" because they possess intermediate characteristics between rivers and natural lakes concerning morphology/hydrology, primary nutrient sources (external loading versus internal recycling), and primary OM sources (allochthonous versus autochthonous; Kimmel et al. 1990). Reservoirs tend to be more spatiotemporally dynamic than natural lakes of similar size. Reservoirs exhibit longitudinal gradients in environmental conditions resulting from morphology/hydrology with many reservoirs also displaying vertical gradients attributed to thermal stratification which predominate in natural lakes (Kennedy and Walker 1990). Often, reservoirs receive large, pulsed inflows that result in extreme, irregular water level fluctuations and rapid, variable water retention times (WRT; Wetzel 1990). Thus, models developed from natural lakes should be applied cautiously to reservoirs (Wetzel 1990). Lind et al. (1993) demonstrated difficulties associated with applying natural lake trophic-state classification methods to reservoirs. They suggested that short WRT was likely the most important factor explaining these difficulties. Similar difficulties may complicate the use of stable isotopes to determine phytoplankton productivity. Also, stable isotope models assume that OM is primarily comprised of phytoplankton (Schelske and Hodell 1995). OM source distinctions may mask productivity relationships in reservoirs due to enhanced sediment loading from watersheds.

In this study, we investigated OM sources and secondary sediment transport mechanisms during an annual cycle in a shallow (Z_{mean} = 6.4 m), polymictic, eutrophic reservoir. We hypothesized that sedimenting particulate organic matter (SPOM) primarily originated from allochthonous sources in winter/spring and autochthonous sources in summer/autumn. We also hypothesized that enhanced SPOM deposition near the sediment-water interface (SWI) was more strongly influenced by surface sediment resuspension in winter/spring and horizontal advection of autochthonous OM in summer/autumn. Particulate organic C (POC) and N (PON) deposition rates, C:N ratios, and C and N stable isotopic composition of SPOM were measured to test these hypotheses. We identify and comment on complications associated with applying stable isotope techniques to reservoirs and propose sampling strategies to account for these difficulties.

Methods

Study Site

Lake Waco (31° 34' 28" N 97° 13' 13" W) is a eutrophic, polymictic reservoir located near the western city-limits of Waco, McLennan County, Texas, USA (Fig. 3.1).

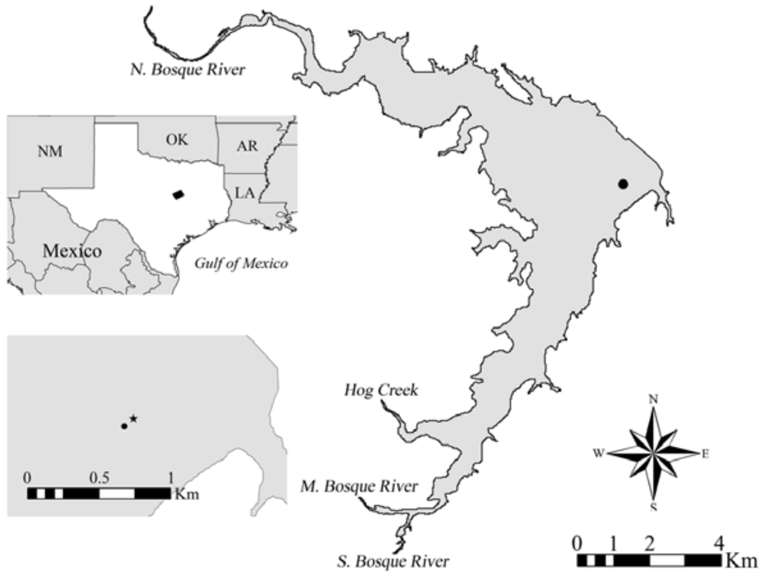

Figure 3.1: Lake Waco map indicating sampling stations. The upper inset map indicates McLennan County, Texas. The lower inset map with associated scale bar indicates sediment trap (circle) and sediment core (star) locations.

Morphometric and watershed characteristics are summarized in Table 2.1. The calculated water residence time during our study (0.47 yr) was longer than the 20 year average water residence time. Chemical characteristics are summarized in Table 3.1. The North Bosque River watershed provides ~75% of reservoir inflow (Lind and Barcena 2003). North Bosque River discharge (USGS Gauge Station No. 08095200; US Geological Survey Water Data for USA 2008) and Lake Waco water surface elevation (US Army Corps of Engineers-Ft. Worth District Hydrologic Data on Ft. Worth District Lakes 2008) are displayed in Figure 2.3. Additional small tributaries include the Middle and South Bosque Rivers and Hog Creek. Predominate landuses include rangeland in the North Bosque River watershed and row-crop agriculture and rangeland in the remaining watersheds (Dworkin 2003). Numerous dairy operations occur near the North Bosque River headwaters. Six municipalities discharge treated wastewater into the North Bosque River above the reservoir. Water column mixing is holomictic (Lind 1971) although brief periods of thermal stratification have been reported (Kimmel and Lind 1972). Based on phytoplankton productivity, Lake Waco was classified soon after impoundment as eutrophic (Kimmel and Lind 1972) and later as mesotrophic (Lind 1986). Phosphorus has been identified as the primary limiting nutrient (McFarland et al. 2001) with short periods of N limitation occurring during late summer (Scott et al. 2008). Light limitation of phytoplankton productivity resulting from high inorganic turbidity (i.e. clays) has been reported (Lind 1986; Rendón-López 1997).

Table 3.1: Chemical characteristics of Lake Waco between October 2003 and September 2004. Values reported as averages and ranges (in parentheses) of monthly samples collected at sediment trap location (0.3 m depth; Scott et al. 2008). pH reported as median. Total alkalinity reported as 1996 value (Rendón-López 1997). Total alkalinity range based on 1976 – 1979 values (Lind unpublished data).

Characteristic	Value
pH	8.2 (7.7 – 8.4)
Total alkalinity (mg $CaCO_3$ L^{-1})	161 (119 – 167)
Total N (mg L^{-1})	1.10 (0.52 – 1.92)
NO_3^- (mg L^{-1})	0.39 (0.02 – 1.04)
Total P (mg L^{-1})	0.10 (0.02 – 0.62)
Chlorophyll *a* (μg L^{-1})	10.8 (5.5 – 16.6)
Turbidity (NTU)	7.5 (3.1 – 11.5)

Organic Matter Sources

Sedimenting particulate matter was collected by buoy-carried, anchored duplicate sediment traps. Sediment traps were deployed semi-monthly or monthly for approximately one-week duration from November 2003 to October 2004 (Fig. 3.1). Cylindrical sediment traps, which had a 10:1 height-to-diameter ratio, were not treated with anti-microbial agents or preservatives. Little OM (~10%) is lost through mineralization by bacteria with exposure times of one week (Bloesch and Burns 1980). Sediment traps were suspended at four meters, which represents the average photic depth of Lake Waco's lacustrine region, and at one meter above the SWI. Photic sediment traps were intended to collect primary sedimentation of autochthonous and allochthonous particles. Bottom sediment traps were intended to collect the aforementioned particles and material experiencing secondary transport within the reservoir. Secondary transport mechanisms include surface sediment resuspension, horizontal advection (i.e. sediment focusing), and river plume sedimentation (Hilton et al. 1986; Ford 1990).

Sediment slurry samples were dried at 104°C, powdered, and treated to remove inorganic carbon (i.e. calcite) prior to analyses. HCl (~1.2N) was slowly added to dry sediments until complete removal of inorganic carbon as indicated by effervescence termination (Vreca and Muri 2006). After adequate rinsing with de-ionized water, samples were dried at 104°C until reaching constant weight. POC and PON concentrations (weight percent) and $\delta^{13}C$ and $\delta^{15}N$ were analyzed by an Elemental Analyzer coupled to an Isotope Ratio Mass Spectrometer (EA-IRMS). POC and PON deposition rates were calculated as the product of elemental concentrations and bulk sediment deposition rates (Teranes and Bernasconi 2000; Lehmann et al. 2004b). C:N values were reported as atomic ratios. $\delta^{13}C$ and $\delta^{15}N$ were reported versus the Pee Dee Belemnite carbonate standard (V-PDB) and atmospheric N_2 (AIR) respectively.

Sediment Transport Mechanisms

Bernasconi et al. (1997) created two-variable mixing models to determine which secondary sediment transport mechanism(s) (sediment resuspension, sediment lateral transport, and allochthonous organic matter loading) contributed to excess POC and PON deposition. Although several mechanisms may concurrently contribute to sediment deposition (Hilton et al. 1986), we were able to begin distinguishing among primary processes influencing deposition seasonally in this polymictic reservoir. We defined excess sediment deposition as bottom minus photic sediment trap deposition rates. To determine relative contributions of resuspended OM, we used the following equation:

$$\delta_{Bottom} = \delta_{Photic} \times Prct_{Photic} + \delta_{Sediment} \times Prct_{Sediment} \quad (5)$$

where:

$$Prct_{Photic} + Prct_{Sediment} = 1 \quad (6)$$

and δ_{Bottom} is the predicted isotopic composition in bottom sediment traps, and δ_{Photic} and $\delta_{Sediment}$ are observed isotopic compositions in photic sediment traps and surface sediments respectively. $\delta_{Sediment}$ was determined from the upper 1.5 cm of a sediment core extracted in July 2004 (Fig. 3.1). $\delta^{13}C$ and $\delta^{15}N$ were analyzed as previously described for sediment trap samples. $Prct_{Photic}$ is the percent contribution of photic to bottom sediment trap deposition (i.e. photic divided by bottom sediment trap deposition rates), and $Prct_{Sediment}$ is the percent contribution of excess sediment to bottom sediment trap deposition (i.e. [bottom minus photic sediment trap deposition rates] divided by bottom sediment trap deposition rates). Sediment resuspension influences excess material deposition when predicted and observed SPOM isotopic compositions in bottom sediment traps are correlated. To determine relative contributions of horizontally-transported autochthonous material, we re-arranged the aforementioned equation:

$$\delta_{Sediment} = [\delta_{Bottom} - (\delta_{Photic} \times Prct_{Photic})] / Prct_{Sediment} \quad (7)$$

where variables are defined as previously stated except $\delta_{Sediment}$ is the predicted SPOM isotopic composition of surface sediments, and δ_{Bottom} is the observed SPOM isotopic composition in bottom sediment traps. In the latter equation, $\delta_{Sediment}$ and $Prct_{Sediment}$ refer to isotopic compositions and percent contributions of excess material regardless of source. Specifically, these variables refer to excess material primarily originating from autochthonous sources because we compared predicted values of excess material to observed compositions in photic sediment traps. Horizontal advection in the water column influences excess material deposition when predicted excess material isotopic compositions and observed isotopic compositions in photic sediment traps are correlated.

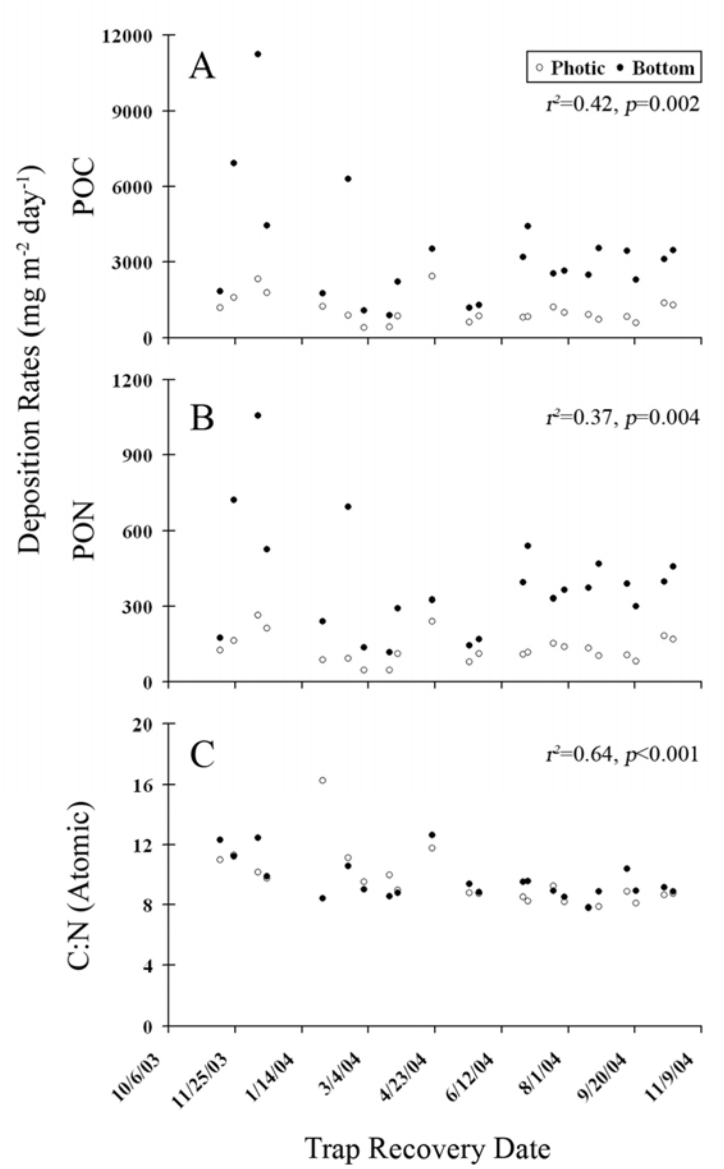

Figure 3.2: Seasonal variation of (A) particulate organic carbon and (B) particulate organic nitrogen deposition rates and resulting (C) C:N ratios of sedimenting particulate organic matter in photic (open circles) and bottom (closed circles) sediment traps. Coefficients of determination and probability values for significant correlations (p<0.05) between photic and bottom sediment trap data have been included. January samples were excluded from analyses because the photic sediment trap C:N ratio was identified as an outlier.

We used the term horizontal advection to replace lateral transport as originally used by Bernasconi et al. (1997) to remove ambiguity. Lateral transport may be interpreted as material movement from the littoral to profundal zones. We defined horizontal advection as sediment movement along the reservoir's longitudinal flow axis. These mixing models assume that carbon and nitrogen isotopic compositions of SPOM are not altered (Cifuentes et al. 1988; Bernasconi et al. 1997).

Statistical Considerations

We performed statistical analyses using SPSS 16.0 software (SPSS Inc.). Outliers (values outside of three standard deviations from the mean) were excluded from statistical analyses. We tested for differences between photic and bottom sediment traps for all variables using paired samples t-tests ($\alpha=0.05$). Normality of data distributions were determined by the Shapiro-Wilk test statistic based on our low sample size (i.e. $N<200$). Paired variables were transformed as required to meet normality assumptions. Mean differences and standard deviations were expressed as untransformed variable units, while significance testing was performed using normalized variables.

Results

Organic Matter Sources

POC and PON deposition rates and SPOM C:N ratios are presented in Figure 3.2. We identified the January sample in photic sediment traps as an outlier based on C:N ratios (16.26; Fig. 3.2C). Although reasons for this value are unknown, it may have been influenced by a small North Bosque River discharge peak and resulting 12 cm lake-level rise (Fig. 2.3). We excluded January samples from statistical analyses for POC and PON

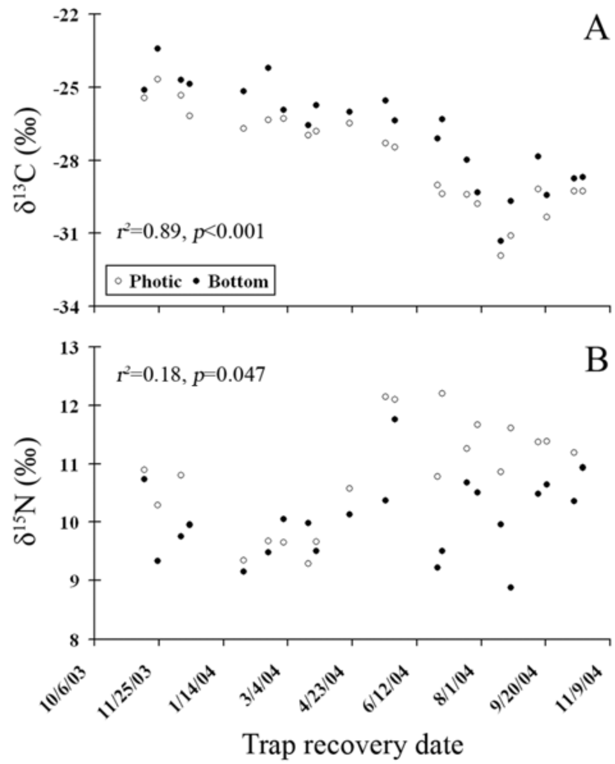

Figure 3.3: Seasonal variation of (A) carbon and (B) nitrogen isotopic compositions of sedimenting particulate organic matter in photic (open circles) and bottom (closed circles) sediment traps. Coefficients of determination and probability values for significant correlations (p<0.05) between photic and bottom sediment trap data have been included.

deposition rates and C:N ratios. POC deposition rates varied temporally from 388 – 2433 and 868 – 11231 mg C m^{-2} d^{-1} in photic and bottom sediment traps respectively (Fig. 3.2A). PON deposition rates varied from 47 – 266 and 118 – 1055 mg N m^{-2} d^{-1} in photic and bottom sediment traps respectively (Fig. 3.2B). During winter (November – February), POC and PON deposition rates were typically greater and more variable than other seasons (Fig. 3.2A and Fig. 3.2B). Bottom sediment traps collected more POC and PON relative to photic sediment traps in winter. POC and PON deposition rates were typically lowest during spring (March – May) and were most similar between photic and bottom sediment traps. However, photic and bottom sediment traps collected more POC and PON in April compared to other spring samples. During summer/autumn (June –

October), POC and PON deposition rates were slightly greater in photic sediment traps and 2× greater in bottom sediment traps relative to spring. SPOM C:N ratios varied temporally from 7.82 – 11.76 and 7.81 – 12.61 in photic and bottom sediment traps respectively (Fig. 3.2C). C:N ratios generally decreased throughout the study with exponential (r^2=0.59, p<0.001) and linear (r^2=0.37, p=0.003) functions best describing trends in photic and bottom sediment traps respectively. April samples, which had maximum C:N ratios for both photic and bottom sediment traps, deviated from these general trends.

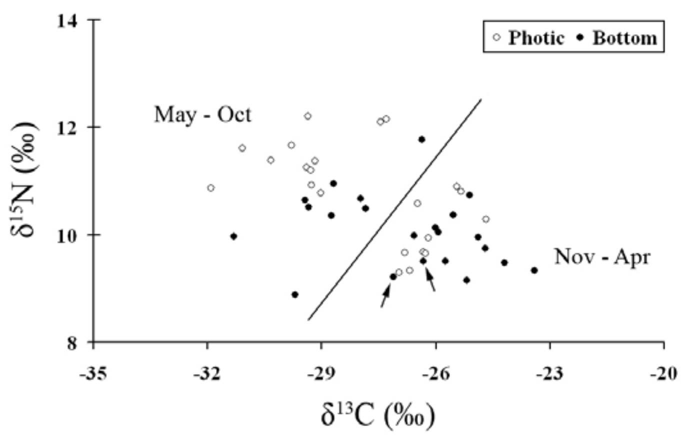

Figure 3.4: Scatterplot of nitrogen versus carbon isotopic compositions of sedimenting particulate organic matter in photic (open circles) and bottom (closed circles) sediment traps. Winter/spring samples occurred to the right of the line, while summer/fall samples occurred to the left. The two arrowed samples from 6/28 and 7/2 were exceptions.

SPOM C and N isotopic compositions are displayed in Figures 3.3 and 3.4. $\delta^{13}C$ varied temporally from -31.92 to -24.68‰ and -31.31 to -23.41‰ in photic and bottom sediment traps respectively (Fig. 3.3A). $\delta^{15}N$ varied from 9.29 to 12.20 and 8.88 to 11.76‰ in photic and bottom sediment traps respectively (Fig. 3.3B). $\delta^{13}C$ decreased throughout the study with linear functions best describing trends in photic (r^2=0.81, p<0.001) and bottom (r^2=0.75, p<0.001) sediment traps (Fig. 3.3A). Beginning in June,

$\delta^{13}C$ decreased more rapidly and reached minima in August in both photic and bottom sediment traps. $\delta^{15}N$ decreased from November to January in both photic and bottom sediment traps (Fig. 3.3B). $\delta^{15}N$ increased from January to April before becoming dramatically heavier in May. Photic sediment trap $\delta^{15}N$ decreased from May to October. During this period, bottom sediment trap $\delta^{15}N$ varied widely and reached a minimum in August. Samples separated temporally based on seasonal isotopic composition differences (Fig. 3.4). Winter/spring samples were isotopically heavy and light for C and N respectively, whereas summer/autumn samples were isotopically light and heavy for C and N respectively. Bottom sediment trap samples from June were exceptions and clustered with winter/spring samples.

Table 3.2: Particulate organic carbon (POC) and particulate organic nitrogen (PON) deposition rates, C:N ratios, and C and N isotopic compositions of sedimenting particulate organic matter in photic (4 m depth) and bottom (1 m above sediment-water interface) sediment traps. POC and PON deposition rates and resulting C:N ratios were calculated as annual averages ± standard deviations. January samples were excluded from POC, PON, and C:N calculations (see text for explanation). $\delta^{13}C$ and $\delta^{15}N$ values were calculated as annual weighted averages relative to POC and PON deposition rates respectively. Sediment $\delta^{13}C$ and $\delta^{15}N$ values were obtained from the homogenized upper 1.5 cm of a sediment core. $\delta^{13}C$ and $\delta^{15}N$ are reported in conventional δ-notation versus the Pee Dee Belemnite carbonate standard (V-PDB) and atmospheric N_2 (AIR) respectively.

	Deposition rates				
	POC (mg C m^{-2} d^{-1})	PON (mg N m^{-2} d^{-1})	C:N (atomic)	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Photic trap	1080 ± 558	133 ± 58	9.3 ± 1.2	-27.56	10.84
Bottom trap	3429 ± 2377	398 ± 225	9.7 ± 1.4	-26.34	9.92
Sediment				-25.99	9.83

Sediment Transport Mechanisms

POC and PON deposition rates were approximately 3× higher in bottom sediment traps than in photic sediment traps (Table 3.2). Annual mean differences between bottom and photic sediment traps were statistically significant (p<0.001 for both POC and PON;

Table 3.3: Paired samples t-test analyses of particulate organic carbon and particulate organic nitrogen deposition rates, C:N ratios, and $\delta^{13}C$ and $\delta^{15}N$ values for photic versus bottom sediment traps. Mean differences were calculated with respect to bottom sediment traps with positive values indicating greater/heavier values and negative values indicating lesser/lighter values. January samples were excluded from POC, PON, and C:N calculations (see text for explanation). POC, PON, and C:N ratios were transformed prior to significance testing to meet normality assumptions.

	Deposition rates				
	POC (mg C m^{-2} d^{-1})	PON (mg N m^{-2} d^{-1})	C:N (atomic)	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Mean diff.	2349	265	0.42	1.11	-0.74
SD	2061	196	0.83	0.69	0.87
p-value	<0.001	<0.001	0.030	<0.001	0.001

Table 3.3). Deposition rates in photic and bottom sediment traps were significantly correlated for both POC (r^2=0.42, p=0.002) and PON (r^2=0.37, p=0.004; Fig. 3.2A and Fig. 3.2B). C:N ratios were greater in bottom sediment traps than in photic sediment traps (Table 3.2). C:N ratios were significantly different between paired sediment traps (p=0.030; Table 3.3). However, annual mean differences varied greatly with respect to the mean (CV=198%). C:N ratios were significantly correlated (r^2=0.64, p<0.001) in photic and bottom sediment traps (Fig. 3.2C).

Predicted C and N isotopic compositions in bottom sediment traps are presented in Figure 3.5. Predicted $\delta^{13}C$ were significantly correlated (r^2=0.80, p<0.001) to observed values in bottom sediment traps (Fig. 3.5A). Predicted and observed $\delta^{15}N$ in bottom sediment traps were significantly correlated (r^2=0.46, p<0.001; Fig. 3.5B). Paired values were not significantly different between sediment traps for $\delta^{13}C$ (t=-0.511, DF=21) and $\delta^{15}N$ (t=-1.530, DF=21). Predicted $\delta^{13}C$ underestimated observed values during winter (November – February) but overestimated observed values during summer/autumn (July – October; Fig. 3.5A). There was generally close agreement between values during spring through early summer (March – June). Predicted $\delta^{15}N$ tended to overestimate observed values during winter through late summer (November –

August) and underestimated observed values thereafter (Fig. 3.5B). The four largest differences in $\delta^{15}N$ between photic and bottom sediment traps occurred from May through August.

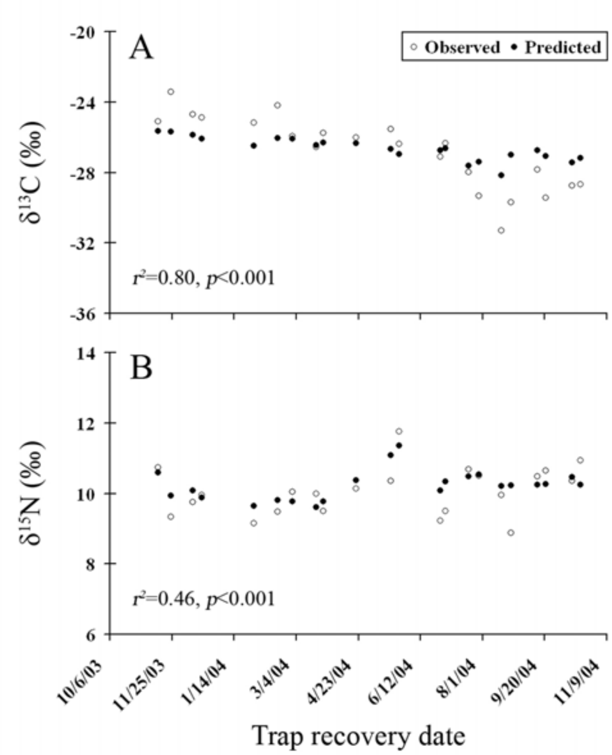

Figure 3.5: Time-series of (A) carbon and (B) nitrogen isotopic compositions of sedimenting particulate organic matter observed in bottom sediment traps (open circles) and predicted isotopic compositions based on organic matter isotopic compositions in photic sediment traps and surface sediments (closed circles). Coefficients of determination and probability values for significant correlations (p<0.05) between observed and predicted values have been included.

Predicted C and N isotopic compositions of excess SPOM are displayed in Figure 3.6. Excess material was defined as material collected in bottom sediment traps that was additional to material collected in photic sediment traps (i.e. bottom minus photic sediment trap deposition rates). Predicted excess material $\delta^{13}C$ and observed values in photic sediment traps were significantly correlated (r^2=0.75, p<0.001; Fig. 3.6A). Predicted excess material was always enriched in ^{13}C compared to photic sediment trap

SPOM. Predicted excess material $\delta^{15}N$ and observed values in photic sediment traps were not correlated (α=0.05; Fig. 3.6B). Predicted excess material was typically depleted in ^{15}N compared to photic sediment trap SPOM with the largest differences occurring from May through August.

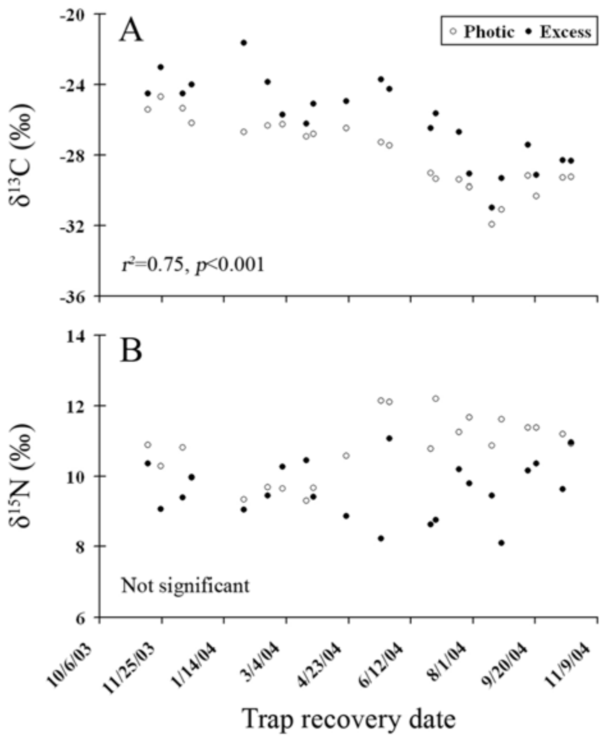

Figure 3.6: Time-series of (A) carbon and (B) nitrogen isotopic compositions of sedimenting particulate organic matter observed in photic sediment traps (open circles) and predicted isotopic compositions of excess organic matter required to obtain observed values in bottom sediment traps (closed circles). Excess organic matter was defined as the fraction of organic matter remaining after organic matter in photic sediment traps had been subtracted. Coefficients of determination and probability values for significant correlations (p<0.05) between observed and predicted values have been included.

Discussion

Organic Matter Sources

C:N ratios indicated that allochthonous sources contributed substantially to SPOM in Lake Waco during our study. Annual average C:N ratios and maxima in photic

(9.3 ± 1.2 and 11.8 respectively) and bottom (9.7 ± 1.4 and 12.6 respectively; Table 3.2 and Fig. 3.2C) sediment traps were near upper limits of typical phytoplankton C:N ratios (4 to 10; Meyers and Teranes 2001). For comparison, Pyramid Lake (Nevada, USA) surface sediments had C:N ratios of 6.7 to 11.5 in areas where algae contributed 90% of bulk organic C to the water column (Tenzer et al. 1997). We likely underestimated basin-wide allochthonous OM contributions because our sampling station was located at distance from river input (Fig. 3.1). OM is primarily supplied by allochthonous sources in reservoir riverine zones and by autochthonous sources in reservoir lacustrine zones (Kimmel et al. 1990). Also, surface sediments had larger C:N ratios closer to allochthonous OM input points (i.e. near shore) in Lake Victoria (East Africa; Talbot and Lærdal 2000). A sampling strategy with stations located within riverine, transition, and lacustrine zones would be required to adequately characterize allochthonous OM delivery to Lake Waco. Although large allochthonous OM loading indicates the potential for net heterotrophy in Lake Waco, we did not investigate incorporation of allochthonous organic matter into food webs.

C and N elemental and isotopic ratios supported our hypothesis that SPOM primarily originated from allochthonous sources in winter/spring and autochthonous sources in summer/autumn. Large SPOM compositional changes were implied by widely varying C:N ratios (7.8 to 12.6; Fig. 3.2C). Decreasing C:N ratios during our study identified a seasonal SPOM compositional shift with allochthonous OM composing more of SPOM in winter relative to autumn (Fig. 3.2C). C and N isotopes also suggested seasonal SPOM compositional shifts with SPOM being ^{13}C-enriched and ^{15}N-depleted in winter/spring relative to summer/autumn samples (Fig. 3.4). δ^{13}C generally decreased from winter through autumn with accelerated decreases beginning in June and an annual minimum in August (Fig. 3.3A). This trend likely tracks declining allochthonous OM contributions and associated C_4 plant signatures (isotopic shifts of -8 to -12‰) relative to the algae (isotopic shifts of approximately -20‰; Meyers and Ishiwatari 1993). During our study, lake-wide chlorophyll a concentrations increased beginning in late-March and remained elevated from April through September (average ± sd = 16.0 ± 1.8 μg L^{-1}) with the exception of August (8.0 μg L^{-1}; Scott et al. 2008) therefore supporting this claim. $\delta^{15}N$ indirectly recorded SPOM source shifts through water column nitrate utilization. Near-minimum $\delta^{15}N$ in January increased gradually through April before dramatically increasing in May (Fig. 3.3B), which coincided with a period of nitrate depletion reported by Scott et al. (2008). Algae are enriched in ^{15}N as nitrate pools are depleted and increasingly ^{15}N-enriched (Hodell and Schelske 1998). Our results counter the suggestion that $\delta^{15}N$ may not record nitrate utilization in eutrophic lakes (Lehmann et al. 2004a).

Interestingly, relationships between algal productivity and C isotopic compositions in Lake Waco appeared to contrast those advanced from stratified natural lakes (Bernasconi et al. 1997; Hodell and Schelske 1998; Hollander and Smith 2001; Lehmann et al. 2004a) and estuaries (Cifuentes et al. 1988). Although we lack seasonal productivity data, we know that algal biomass was elevated from April through September (Scott et al. 2008). We assumed that algal photosynthetic rates increased with increasing algal biomass, higher summer/autumn temperatures, and improved summer photic conditions. If this assumption held true, SPOM $\delta^{13}C$ became lighter with increased productivity. In stratified systems, epilimnetic DIC pools are enriched in ^{13}C as settling algae preferentially utilize and export ^{12}C to the hypolimnion (Hodell and Schelske 1998; Lehmann et al. 2004a). Algae become isotopically heavy because photosynthetic isotope fractionation decreases as CO_2 becomes limiting (Hodell and Schelske 1998; Lehmann et al. 2004a). These Rayleigh fractionation models require reactant quantities to be finite (Teranes and Bernasconi 2000) and are not valid when reactants are continuously replenished (Teranes and Bernasconi 2000). In polymictic systems, algae may continue to preferentially assimilate ^{12}C during highly productive periods because strong holomictic circulation patterns likely prevent ^{12}C draw-down. In Lake Waco, total alkalinity concentrations and pH indicate that DIC pools are abundant with bicarbonate as the primary DIC species (Table 3.1). Teranes and Bernasconi (2000) presented a similar argument in suggesting that $\delta^{15}N$ may not identify nitrate utilization degree when only small portions of DIN pools are consumed.

Alternatively, SPOM $\delta^{13}C$ may have been influenced by changing DIC isotopic compositions in Lake Waco. SPOM $\delta^{13}C$ decreased throughout our study (Fig. 3.3A) unlike the annual cycles typical of natural lakes. Because of Lake Waco's short water residence time (0.47 yr during our study), DIC isotopic compositions may have drastically changed during our study. We cannot assume consistent DIC $\delta^{13}C$, which weakens our proposed relationships between algal productivity and OM isotopic compositions in polymictic reservoirs. In contrast, studies in Lake Ontario benefited from the assumption of conservative ion chemistry between years based on Lake Ontario's long water residence time (7 yrs; Schelske and Hodell 1991; Hodell and Schelske 1998). Variable DIC isotopic compositions likely complicate phytoplankton productivity relationships in most reservoirs, which are characterized by relatively short water residence times compared to natural lakes (Kennedy and Walker 1990). Future reservoir studies involving stable isotopes should measure DIC $\delta^{13}C$ concurrently with algae $\delta^{13}C$ to accurately determine photosynthetic isotopic fractionation.

Also, widely variable inflow rates and dissolved inorganic nutrient isotopic compositions may complicate stable isotope studies in reservoirs. Relatively large, irregular water level fluctuations are characteristic of reservoirs compared to natural lakes (Wetzel 1990). Lake Waco experienced two large North Bosque River discharge peaks (261 and 419 m^3 sec^{-1}) and a subsequent lake-level rise following April's sampling (Fig. 2.3). SPOM compositions differed seasonally along this timeframe with ^{13}C-enriched, ^{15}N-depleted SPOM during low water elevation (November – April) and ^{13}C-depleted, ^{15}N-enriched SPOM during high water elevation (May – October; Fig. 3.4). The only exceptions to this seasonal clustering were bottom sediment trap samples from June which clustered with winter samples (See arrowed samples, Fig. 3.4). June sampling was preceded by two discharge peaks (380 and 38 m^3 sec^{-1}; Fig. 2.3) which likely delivered large allochthonous OM loads to Lake Waco. $\delta^{15}N$ values of algae (+8‰) and land plants (+1‰) differ because they retain nitrogen source distinctions (Meyers and Ishiwatari 1993). While we hypothesize that seasonal differences resulted from differing allochthonous OM contributions, we cannot discredit the claim that differences may have been strongly influenced by differing isotopic compositions of North Bosque River inflow. Future studies should include $\delta^{13}C$ and $\delta^{15}N$ analyses of dissolved inorganic nutrient pools in major tributaries as well as those in the lake proper to remove this bias.

SPOM N isotopic composition was likely influenced by cyanobacterial N_2 fixation during our study. Lake Waco experiences large seasonal cyanobacterial blooms of *Anabaena* sp. Photic sediment trap $\delta^{15}N$ decreased from May through October (Fig. 3.3B) suggesting increased utilization of atmospheric N_2 by algae during summer. Our observations agree with documented increased N_2 fixation rates during July through September (Scott et al. 2008). Atmospheric N_2 and DIN have $\delta^{15}N$ values of 0 and +7 to 10‰ respectively (Peters et al. 1978; Meyers and Ishiwatari 1993). Low algal $\delta^{15}N$ values have been attributed to N_2 fixation by cyanobacterial blooms (Gu et al. 1996; Gu et al. 2006; Scott et al. 2007). As summer nitrate concentrations decreased, cyanobacteria likely began fixing atmospheric N_2 to alleviate N limitation in Lake Waco.

Bottom sediment trap deposition rates were relatively high and variable during our study. POC and PON deposition rates in bottom sediment traps were an order of magnitude greater in Lake Waco (3429 mg C m^{-2} d^{-1} and 398 mg N m^{-2} d^{-1}; Table 3.2) compared to Lake Lugano (Switzerland), a monomictic sub-alpine lake (299 mg C m^{-2} d^{-1} and 30 mg N m^{-2} d^{-1}; Bernasconi et al. 1997). Deposition rates varied seasonally with winter POC and PON deliveries disproportionately comprising annual POC and PON deposits. From November through February, bottom sediment traps collected 45 and 41% of annual POC and PON deposits respectively. Elevated bottom sediment trap deposition rates and greater C:N ratios during winter (Fig. 3.3) suggested that allochthonous sources greatly contributed to annual POC and PON budgets. Lind (1971) demonstrated that allochthonous OM composed 22% of annual OM budgets in Lake Waco, which is plausible based on our findings. However, we showed that allochthonous sources supplied more OM seasonally. We anticipated our sediment trap estimates would be greater than inflow estimates of Lind (1971) because secondarily transported OM would be biased towards more refractory allochthonous OM during sediment focusing.

Bottom sediment trap deposition rates likely poorly estimate POC and PON burial rates in Lake Waco. Our deposition rates likely overestimate burial rates because this location experiences strong sediment focusing. Filstrup and Lind (unpublished data) showed that 71% of total suspended solids (60% of volatile suspended solids) in bottom sediment traps were attributed to secondary sediment transport. However, deposition rates from our sampling location may underestimate basin-wide deposition rates because deposition rates typically decrease exponentially along a reservoir's flow axis (Thornton 1990b). Even natural lakes can display a high degree of spatial variability concerning deposition rates. Schelske (2006) found that 70% each of total dry mass and OM was deposited on only 39% of Lake Apopka (Florida, USA) bottom area. Additional sampling stations located throughout the various reservoir regions are required to adequately estimate basin-wide burial rates. Additionally, bottom sediment trap deposition rates would need to be corrected for OM degradation at the SWI which can be significant. Hedges et al. (1988) discovered that 60 and 70% of POC and PON respectively were degraded at the SWI in Dabob Bay, Washington (USA).

Sediment Transport Mechanisms

Secondary sediment transport mechanisms potentially exposed OM to prolonged microbial mineralization during our study. Bottom sediment traps collected significantly more POC and PON than photic sediment traps (Table 3.2 and Table 3.3). Higher particulate organic nutrient deposition rates near the SWI have been reported for numerous natural lakes (Meyers and Eadie 1993; Bernasconi et al. 1997; Hodell and Schelske 1998). POC and PON deposition rates in bottom relative to photic sediment traps were an order of magnitude greater in Lake Waco (~200% for both POC and PON;

Table 3.2) compared to Lake Lugano (~10 and ~21% for POC and PON respectively; Bernasconi et al. 1997). Strong, frequent water column mixing, as well as relatively shallow depth, likely explained enhanced excess material deposition attributed to secondary sediment transport mechanisms in Lake Waco. Filstrup and Lind (unpublished data) documented that excess total suspended solids deposition was related to maximum wind speed direction at this location. Holomictic circulation patterns were suggested by significant correlations between photic and bottom sediment traps for deposition rates and C:N ratios (Fig. 3.2). However, OM elemental and isotopic differences between photic and bottom sediment traps suggested that allochthonous OM composed greater proportions of SPOM in bottom sediment traps (Table 3.2 and Table 3.3).

Although several mechanisms likely contributed to secondary sediment transport, we hypothesized that excess POC and PON deposition was strongly influenced by surface sediment resuspension in winter/spring and horizontal advection of autochthonous OM in summer/autumn. Significant correlations between predicted and observed isotopic compositions for $\delta^{13}C$ (r^2=0.80, p<0.001) and $\delta^{15}N$ (r^2=0.46, p<0.001; Fig. 3.5) identified sediment resuspension as a primary mechanism influencing excess sediment deposition in Lake Waco. Although predicted and observed $\delta^{13}C$ agreed well during spring/early-summer, they did not agree as closely as we anticipated during winter (Fig. 3.5A). Relatively light observed values suggest that other mechanisms, including density plume sedimentation of allochthonous OM and associated C_4 plant signatures, contributed excess material during winter. Predicted and observed $\delta^{13}C$ disagreed after June with relatively heavy observed values (Fig. 3.5A). Predicted excess material $\delta^{13}C$ and observed values in photic sediment traps were significantly correlated (r^2=0.75,

$p<0.001$; Fig. 3.6A) indicating that horizontal advection of autochthonous OM influenced excess material deposition. Strong correlations between predicted and observed $\delta^{13}C$ ($r^2=0.92$, $p<0.001$) during May through October and lack of correlation during November through April suggest that horizontal advection was a primary transport mechanism during summer/autumn but not during winter/spring. Predicted excess material $\delta^{15}N$ were not correlated to observed photic sediment trap values (Fig. 3.6B) similar to findings from Lake Lugano (Bernasconi et al. 1997). Other factors, including additional transport mechanisms and biogeochemical N cycling, likely influenced SPOM N isotopic compositions during our study.

The aforementioned models rely on stable isotopes serving as conservative tracers of autochthonous and allochthonous OM sources (Cifuentes et al. 1988; Bernasconi et al. 1997). The ability of microbial degradation to mask OM isotopic source distinctions is debatable. Laboratory studies have demonstrated that microbial degradation can significantly alter OM isotopic composition in cultures (Macko and Estep 1984; Lehmann et al. 2002). However, several field studies have shown that C and N isotopic compositions were not largely changed by diagenesis and that sediment OM reflects original algal signals (Meyers and Eadie 1993; Meyers 1994; Schelske and Hodell 1995; Hodell and Schelske 1998). In Lake Waco, $\delta^{13}C$ became heavier from photic to bottom sediment traps to surface sediments whereas $\delta^{15}N$ became lighter (Table 3.2). The ~1‰ $\delta^{13}C$ increase with depth in Lake Waco (Table 3.3) was similar to observations from Lake Michigan (Meyers and Eadie 1993) which the authors attributed to lateral transport of autochthonous OM from highly productive coastal regions. This explanation does not seem plausible for our observations. $\delta^{15}N$ decreases with depth were highly variable

(Table 3.3) and likely resulted from several nitrogen cycling processes. This decrease could potentially be explained by intermediate stages of OM processing which was hypothesized to explain a 1 to 2‰ $\delta^{15}N$ decrease with depth in Lake Michigan (Meyers and Eadie 1993). Based on greater POC and PON deposition rates in bottom sediment traps (Table 3.3), it seems more likely that isotopic composition changes resulted from greater allochthonous OM contributions to bottom sediment traps. Therefore, isotopic compositions served as conservative tracers.

Conclusions

Our findings suggest that allochthonous OM loads and secondarily-transported OM significantly contribute to SPOM in reservoirs. These supplements to autochthonous OM production may be important in maintaining food webs particularly during periods of low phytoplankton productivity. C and N elemental and isotopic ratios appeared to nicely record OM source differences in this reservoir. However, either mixing regime or reservoir characteristics obscured phytoplankton productivity inferences as indicated by atypical C isotope patterns. Future studies concerning stable isotope fractionation – phytoplankton productivity relationships in reservoirs are warranted and should include isotopic measurements of DIC and DIN pools both within reservoir and in major tributaries. A stable isotope study performed in a monomictic reservoir would imply whether our observations differed from previous studies because of mixing regimes or system differences.

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CHAPTER FOUR

Sediment Elemental and Isotopic Compositions Record a Polymictic Reservoir's Eutrophication

Abstract

Paleolimnological studies are rarely performed on reservoirs over concern that the sediments may not accurately chronicle reservoir history. Eutrophication indicators may behave differently in polymictic reservoirs and stratified natural lakes due to system and/or mixing regime differences. We measured particulate organic carbon (POC), particulate organic nitrogen (PON), and total phosphorus (TP) concentrations, carbon:nitrogen (C:N) and nitrogen:phosphorus (N:P) ratios, and carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) stable isotopes from a sediment core to demonstrate that sufficient information can be derived from sediments to permit a historical reconstruction. Widely variable POC were likely biased by seasonal/annual variability in allochthonous organic matter (OM) loading. Upwardly increasing PON supported historic primary productivity (PP) data suggesting that PON may be better PP indicators than POC. Upwardly increasing TP documented historic P enrichment. C:N declined with reservoir age and identified an OM source shift from largely allochthonous to increasingly autochthonous sources. Unexpectedly, N:P increased through time implying that N-fixation rates have increased to compensate for increasing N limitation as P loading increased. $\delta^{13}C$ decreased with increasing PP producing an atypical relationship compared to stratified natural lakes. OM source shifts likely biased the $\delta^{13}C$ – PP relationship and may weaken $\delta^{13}C$-inferred PP reconstructions in similar reservoirs. We attributed progressively

heavier $\delta^{15}N$ to dissolved inorganic N (DIN) source changes rather than nitrate utilization. Watershed urban growth and dairy operation intensification potentially contributed greater loads of isotopically heavy DIN. These paleolimnological indicators have great potential to assist eutrophication assessment and management efforts in reservoirs.

Introduction

Paleolimnology provides tools enabling lake and reservoir managers to retrospectively model long-term primary productivity (PP) and nutrient loading fluctuations. Commonly, paleolimnological techniques are the only methods to indirectly obtain historical water quality data when direct historical monitoring data are lacking (Smol 1992; Battarbee 1999). When available, historical monitoring data are biased by short-term variations that mask long-term trends (Battarbee 1999) and analytical techniques which may not be comparable to modern techniques (Smol 1992). Reconstructing water quality history from several independent studies is additionally complicated by differing sampling schemes and methodologies. While paleolimnological studies of natural lakes are common, they are rarely performed on reservoirs because of concern over sediment disturbance (Shotbolt et al. 2005; Shotbolt et al. 2006). Additional studies demonstrating successful paleolimnological reconstructions and identifying confounding factors in reservoirs are requisite before paleolimnological techniques can be robustly applied to reservoirs.

Sediment organic carbon and nitrogen isotopic compositions are often used to infer historical phytoplankton production of natural lakes. Sediment carbon isotopes were successfully employed as paleoproductivity proxies in the Great Lakes (Schelske and Hodell 1991; Schelske and Hodell 1995; Hodell and Schelske 1998). The relationship's mechanism relies on preferential assimilation and hypolimnetic transport of the lighter isotope (^{12}C) by phytoplankton which subsequently creates an isotopically heavy epilimnetic dissolved inorganic carbon (DIC) pool (Hodell and Schelske 1998). It is unclear if this relationship holds in well-mixed reservoirs. Filstrup et al. (2009) documented an atypical sedimenting organic matter (OM) carbon stable isotope trend during an annual cycle in a polymictic reservoir suggesting that system characteristics and/or mixing regimes obscure phytoplankton productivity relationships. Sediment nitrogen isotopes chronicled epilimnetic nitrate utilization in stratified natural lakes based on a reservoir effect mechanism similar to that of carbon stable isotopes (Hodell and Schelske 1998; Teranes and Bernasconi 2000). However, dissolved inorganic nitrogen (DIN) sources strongly influence phytoplankton nitrogen isotopic composition and can obscure relationships associated with nitrate utilization, especially over decadal and centurial timescales (Teranes and Bernasconi 2000; Lehmann et al. 2004b). The isotopic composition of DIN will change as watershed land use practices change during a reservoir's life.

Lake Waco presents a model system in which to test the application of paleolimnological techniques as eutrophication indicators in polymictic reservoirs. This sub-tropical reservoir is relatively shallow (Z_{mean} = 6.4 m) with holomictic and polymictic circulation patterns although ephemeral thermal stratification exists (Lind 1971). Based on storage capacity, the reservoir is medium-sized (upper 40th percentile) relative to Texas reservoirs (Texas Water Development Board Reservoir Summary Report 2009). Lake Waco has experienced accelerated eutrophication due to P enrichment which has been sporadically documented in the literature (Kimmel 1969; Kimmel and Lind 1972; Lind 1979; Lind 1986; Rendón-López 1997; Adams and McFarland 2001; Adams et al. 2005). As a result, drinking water treatment costs increased 2.5× from 1996 to 2004 with >50% of current costs related to removal of taste and odor compounds (Conry 2008).

Can sediment elemental concentrations and ratios and OM isotopic compositions record eutrophication in such a polymictic reservoir? To answer this question, we measured particulate organic carbon (POC), particulate organic nitrogen (PON), and total phosphorus (TP) concentrations and ratios as well as carbon and nitrogen stable isotopes ($\delta^{13}C$ and $\delta^{15}N$) from a dated sediment core. We hypothesized that upwardly increasing POC, PON, and TP concentrations would chronicle eutrophication through increased algal biomass delivery to sediments and reservoir TP enrichment. We also hypothesized that decreasing carbon:nitrogen (C:N) ratios identified an OM source shift from allochthonous to autochthonous sources and that decreasing nitrogen:phosphorus (N:P) ratios recorded increasing nitrogen limitation concurrent with TP enrichment. Finally, we hypothesized that OM $\delta^{13}C$ and $\delta^{15}N$ suggested increased PP but were strongly biased by allochthonous OM loading and dissolved inorganic nutrient source shifts.

Methods

Study Site

Lake Waco (31° 34' 28" N 97° 13' 13" W) is an impoundment of the North Bosque River located in McLennan County, Texas, USA (Fig. 3.1). The Middle and South Bosque Rivers and Hog Creek provide minor inflows. The reservoir is relatively shallow with a brief water residence time and large watershed (Table 2.1). Lake Waco

Table 4.1: Historic productivity and chemical characteristics of Lake Waco. Values reported as averages and ranges (in parentheses) for net primary productivity (NPP), chlorophyll *a* (Chl *a*), phosphate-P (PO_4-P), total phosphorus (TP), nitrate-N (NO_3-N), total Kjeldahl nitrogen (TKN), turbidity (TURB), and light extinction coefficient (LEC). Values represent lacustrine stations only except 1970 nutrient and turbidity data and 1980 data which include riverine and transition stations. Phosphate values from 1970s reported as soluble reactive phosphorus. Turbidity values from the 1960s reported as Jackson Turbidity Units. Data obtained from the following sources: 1960s (Kimmel 1969) except NPP (Kimmel and Lind 1972), 1970s (Lind 1979), 1980s (Lind 1986), 1990s (Adams and McFarland 2001) except NPP and LEC (Rendón-López 1997), and 2000s (Adams et al. 2005) except NPP and LEC (Gifford and Lind unpublished data).

			Decade		
Variable	1960s†	1970s‡	1980s§	1990s¶	2000s††
NPP (mg C m^{-2} day^{-1})	890 (220-1449)	474 (56-2550)		512 (132-1534)	3160
Chl *a* (µg L^{-1})		10 (3-54)	7.1	17 (4-35)	19 (3-41)
PO4-P (µg L^{-1})	2000 (0-7200)	5 (0-16) ‡‡		10 (0-70)	4 (0-31)
TP (µg L^{-1})		53 (26-79)	53	100 (20-470)	75 (9-210)
NO_3-N (mg L^{-1})	<0.02	0.14 (0-0.85)		0.41 (0.01-1.90)	0.48 (0.01-2.44)
TKN (mg L^{-1})				0.61 (0.15-1.54)	0.58 (0.06-1.54)
TURB (NTU)	44§§ (26-80)	15 (6-24)	26.6		
LEC (m^{-1})	1.4 (0.9-2.5)			1.3 (0.8-1.7)	1.0

was constructed in 1965 for water supply and flood control purposes. The reservoir flooded a previous impoundment constructed in 1929 that became a marsh due to high sedimentation rates. Lake Waco has experienced accelerated eutrophication due to nutrient enrichment. Net PP and chlorophyll *a* concentrations have increased from the 1970s to 2000s (Table 4.1). Dissolved and total P concentrations increased between the 1970s and 1990s before declining in the 2000s (Table 4.1). Nitrate-N concentrations increased from the 1960s to 2000s whereas total Kjeldahl N concentrations were similar during the 1990s and 2000s (Table 4.1). P is the primary limiting nutrient (McFarland et al. 2001) with brief N limitation occurring in late summer (Scott et al. 2008). Inorganic light limitation (clays) also influences phytoplankton productivity (Lind 1986; Rendón-López 1997).

Lake Waco is located in a historically agricultural watershed dominated by row-crop agriculture (cotton) and rangeland (Lind 1971). Little bluestem (*Schizachyrium scoparium* (Michx.) Nash) is the predominant prairie species. Ashe juniper (*Juniperus ashei* Buchh.) and plateau live oak (*Quercus fusiformis* Small) primarily compose upland woodlands, while pecan (*Carya illinoinensis* (Wangenh.) K. Koch) primarily covers riparian zones. The North Bosque River receives outfall from six municipal wastewater treatment plants. Many dairy operations occur in Erath County which contains the North Bosque River headwaters. In Erath County, dairy cow numbers roughly doubled between 1987 and 1993 (35400 to 70000 head respectively) and peaked in 1997 (93300 head; US Department of Agriculture National Agriculture Statistics Service 2008). Dairy cows decreased by 16300 head from 2002 – 2003 and currently number 58000 head (US Department of Agriculture National Agriculture Statistics Service 2008). The human population inhabiting the watershed was 93315 in 2000 (US Census Bureau 2002). For the six watershed counties, the population increased by 37% between 1970 and 2000 (US Census Bureau 1973; US Census Bureau 2002). We assumed a similar trend within the watershed.

Site Selection and Sediment Coring

We used bathymetric maps and multifrequency acoustic profiles to determine a suitable coring location. We restricted site selection to deepwater locations because they are typically sediment accumulation zones that contain continuously-deposited undisturbed sediments (Shotbolt et al. 2005). Previous multifrequency acoustic profiles (Dunbar et al. 1999) revealed a deepwater zone with thick sediment deposits. Additional multifrequency acoustic surveying revealed a deepwater depression that appeared to contain relatively undisturbed sediments. Although nearby locations experience sediment focusing (Filstrup and Lind unpublished data), we selected this depression because sediments likely remained relatively undisturbed after deposition and integrated signals over the lake basin.

A ~2.4 m sediment core was extracted from the selected location in July 2004 using a vibracorer head and aluminum core barrel (Fig. 3.1). The sediment core was sectioned at 1.5-cm intervals during vertical extrusion from the core barrel. After homogenization, samples were dried at 104°C until reaching constant weight. Water content (percent weight) was calculated based on sediment wet and dry weights. Samples were manually powdered by mortar and pestle. Samples were stored at room temperature awaiting further analyses.

Sediment Core Dating

A sediment core dating model was established using stratigraphic markers for peak radiocesium (^{137}Cs) activity, reservoir re-impoundment date, lead concentration peak, and core collection date. ^{137}Cs activity was measured by high-resolution germanium diode gamma detectors and multi-channel analyzer according to Schottler and Engstrom (2006). ^{137}Cs atmospheric deposition peaked in 1963 – 1964 resulting from nuclear weapons testing (Albrecht et al. 1998). Maximum ^{137}Cs activity, which represents 1963 in minimally disturbed sediments (Schottler and Engstrom 2006), would have occurred in the original reservoir's sediments. Reservoir re-impoundment (1965) likely deposited a nearby visual stratigraphic marker (terrestrial OM, soils, etc.) that could confirm the ^{137}Cs peak. Lead (Pb), aluminum (Al), and calcium (Ca) concentrations were measured by an inductively coupled plasma mass spectrometer (ICP-MS) and an inductively coupled plasma atomic emission spectrometer (ICP-AES) following concentrated acid digestion (nitric, hydrochloric, hydrofluoric, and perchloric). Sediment Pb concentrations decreased in dated sediment cores from four reservoirs throughout the United States and were attributed to government restrictions on leaded gasoline and its subsequent replacement by unleaded gasoline (Callender and Van Metre 1997). In White Rock Lake (140 km NNE of Lake Waco), Al-normalized Pb concentrations peaked during 1978 in a ^{137}Cs-dated sediment core (Van Metre and Callender 1997). Al-normalized Pb concentrations accounted for varying clay (i.e. aluminosilicate) sedimentation rates (Van Metre and Callender 1997). We calculated (Al + Ca)-normalized Pb concentrations to remove biases from varying clay and calcite sedimentation rates. The core surface represented recent deposition (2004).

Sediments were dated by linearly interpolating between the core surface (2004) and the (Al + Ca)-normalized Pb concentration peak (1978). We corrected for sediment compaction by interpolating with respect to linear dry weight sedimentation rates (g yr^{-1}) rather than linear sedimentation rates (cm yr^{-1}). Dry weight density (g cm^{-3}) was calculated from water content (percent weight) using the following equation:

$$\rho_{DW} = \rho_W \times \rho_G \times (1 - WC) / [(\rho_G \times WC) + (\rho_W \times (1 - WC)] \qquad (8)$$

where ρ_{DW} is dry weight density (g cm^{-3}), ρ_W is water density (1 g cm^{-3}), ρ_G is assumed grain density (2.6 g cm^{-3}), and WC is water content factor (percent weight; Dunbar personal communication). Core section dry weight (g sediment) was calculated as the product of dry weight density and core section volume (68.4 cm^3). We used the average annual dry weight sedimentation rate to extrapolate the core bottom date from the normalized Pb concentration peak.

Elemental Concentrations, Ratios, and Stable Isotopes

POC and PON concentrations (weight percent) and C and N stable isotopic compositions were analyzed by an elemental analyzer coupled to an isotope ratio mass spectrometer (EA-IRMS). Dried, powdered samples were treated with HCl (~1.2N) to remove inorganic C (i.e. calcite) prior to analyses (Vreca and Muri 2006). Samples were rinsed with de-ionized water, centrifuged, and dried at 104°C until reaching constant weights. TP concentrations were determined by flow injection analysis (FIA) using the molybdenum blue method following Kjeldahl digestion. Dried, powdered, untreated samples were digested with sulfuric acid, potassium sulfate, and copper sulfate in a block digestor. Samples were digested at 350°C for 120 minutes after a linear ramp to 250°C and 30 minute hold. Phosphate-phosphorus was measured on a Lachat 8500 based on the molybdenum blue method (Murphy and Riley 1962). POC, PON, and TP concentrations were reported as sediment concentrations (mg g sediment^{-1}). C:N and N:P ratios were reported as atomic ratios. $\delta^{13}C$ and $\delta^{15}N$ were reported versus the Pee Dee Belemnite carbonate standard (V-PDB) and atmospheric N_2 (AIR) respectively.

Land Use/Land Cover Changes

We investigated watershed land use/land cover (LULC) changes to determine potential nutrient source and loading rate shifts as Lake Waco aged. Watershed LULC were determined using satellite data obtained from the EROS Data Center Distributed Active Archive Center (EDC DAAC) from the North American Landscape Characterization (NALC) program. Data were acquired for October 1973, 1986, and 1991 from the multispectral scanner sensor (MSS) and July 2001 from the Enhanced Thematic Mapper (ETM+). Data were classified into five LULC categories: deciduous hardwood forest, juniper forest, grass/agriculture, urban/bare ground, and water (Anderson et al. 1990). We subsequently combined deciduous hardwood and juniper forests into a single LULC (forests). We used the unsupervised classification algorithm Iterative Self-Organizing Data Analysis Algorithm (ISODATA) provided with ER

Mapper image processing software (Earth Resource Mapping Pty Ltd., West Perth, W.A., Australia) to classify LULC.

Statistical Considerations

We performed statistical analyses using SPSS 16.0 software (SPSS Inc.). Outliers were identified and removed prior to regression and correlation analyses. Outliers were defined as values >1.5 interquartile ranges from Tukey's hinges. Decadal data trends were revealed by simple linear regression. Elemental concentrations and ratios influencing C and N stable isotopes were identified by bivariate correlation. Correlations were analyzed using Pearson product-moment correlation coefficients and two-tailed significance tests.

Results

Sediment Core Dating

Radiocesium activity and lead concentration profiles are displayed in Figure 4.1. ^{137}Cs activity increased from a near-surface minimum (0.07 pCi g^{-1}) to a core bottom maximum (0.36 pCi g^{-1}; Fig. 4.1A). The sediment core contained neither the ^{137}Cs atmospheric deposition peak (1963) nor non-detectable activities which pre-dated atmospheric nuclear weapons testing (pre-1952). The ^{137}Cs profile recorded dilution events at 30, 92, and 188 cm. Unexpectedly, near-surface ^{137}Cs activity was elevated (0.27 pCi g^{-1}; Fig. 4.1A). Lead concentrations increased with sediment core depth but by variable rates (Fig. 4.1B). Pb concentrations increased slowly until 90 cm (0.008 ppm cm^{-1}), increased rapidly from 90 to 174 cm (0.066 ppm cm^{-1}), and increased more slowly thereafter (0.007 ppm cm^{-1}). The core section at 174 cm depth represented a breakpoint

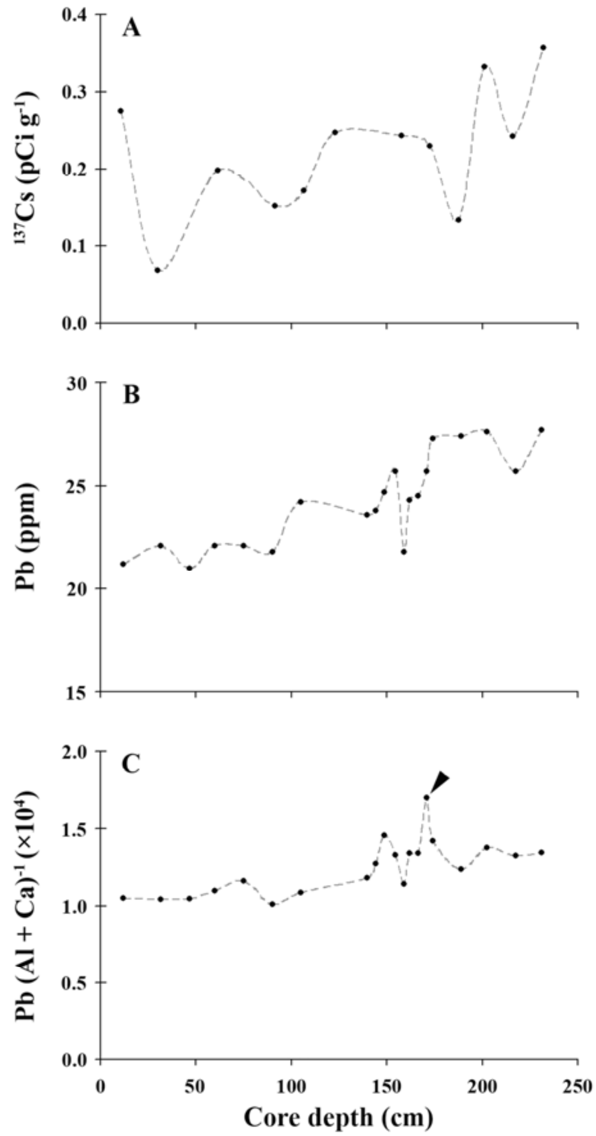

Figure 4.1: Stratigraphic markers used for sediment core dating. (A) ^{137}Cs activity, which obtained peak atmospheric deposition rates in 1963 – 1964 after atmospheric nuclear weapons testing. (B) Lead concentrations and (C) aluminum- and calcium-normalized lead values (×10^{4}), which obtained peak values in 1978 (arrowed) from a White Rock Lake (Texas) sediment core attributed to bans on leaded gasoline sales. Al- and Ca-normalized Pb values accounted for varying deposition rates of clays and calcite respectively. Connecting lines were added as visual aids.

between elevated Pb concentrations (below) and rapidly decreasing Pb concentrations (above). The nearby apparent dilution event (159 cm) emphasized the need to account for varying sedimentation rates (Fig. 4.1B). Aluminum- and calcium-normalized Pb values clearly peaked at 171 cm (arrowed; Fig. 4.1C). We assigned the date 1978 to this

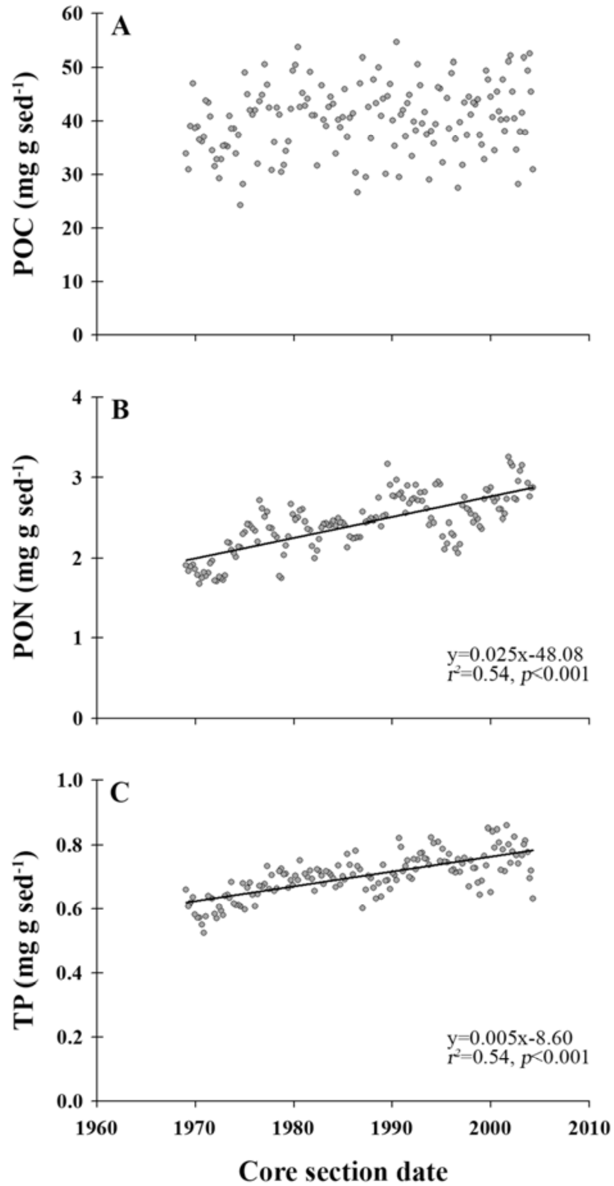

Figure 4.2: (A) Particulate organic C, (B) particulate organic N, and (C) total P concentrations from sediment core. Linear trend lines were fit to PON and TP data.

depth. The resulting linear sedimentation rate was 6.6 cm yr^{-1} calculated by linear interpolation between core surface (2004) and the normalized Pb concentration peak. The corresponding linear dry weight sedimentation rate was 138.7 g yr^{-1} for the sediment core surface area. The sediment core bottom represented 1969 based on linear extrapolation of this sedimentation rate from the normalized Pb concentration peak.

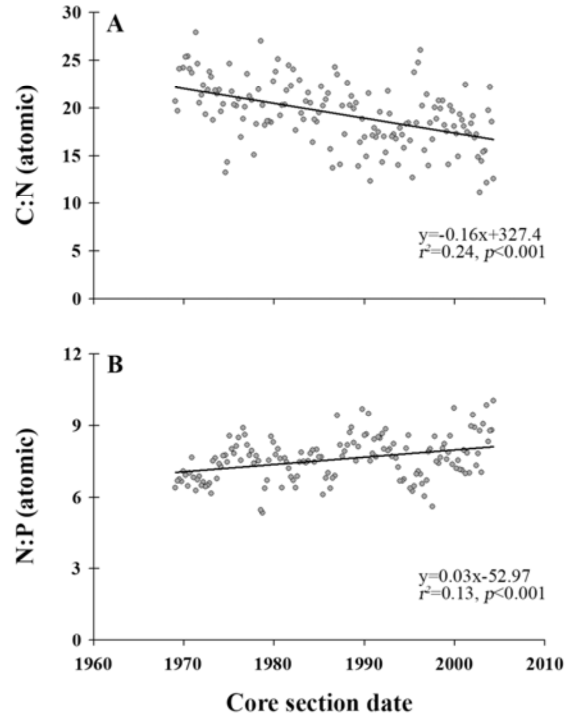

Figure 4.3: (A) C:N and (B) N:P atomic ratios from sediment core. Linear trend lines were fit to data.

Elemental Concentrations and Ratios

POC, PON, and TP concentrations are presented in Figure 4.2. POC concentrations did not display a conspicuous temporal trend as section concentrations were widely scattered (Fig. 4.2A). However, POC concentrations significantly linearly increased (r^2=0.04, p<0.05, N=154) from 1969 to 2004 although with little explanatory power. POC concentrations varied temporally from 24.2 – 54.6 mg g sediment^{-1}. Contrastingly, PON and TP concentrations had conspicuous temporal trends. PON concentrations linearly increased (r^2=0.54, p<0.001, N=154) as the reservoir aged (Fig. 4.2B). PON concentrations varied from 1.7 – 3.3 mg g sediment^{-1}. TP concentrations also linearly increased (r^2=0.54, p<0.001, N=152; Fig. 4.2C) from 1969 to 2004 but 5× more slowly than PON concentrations. TP concentrations ranged from 0.5 – 0.9 mg g sediment^{-1}. PON and TP concentrations were significantly correlated (r=0.61, p<0.001, N=150), whereas POC and TP concentrations were uncorrelated (α=0.05) in this P-limited system.

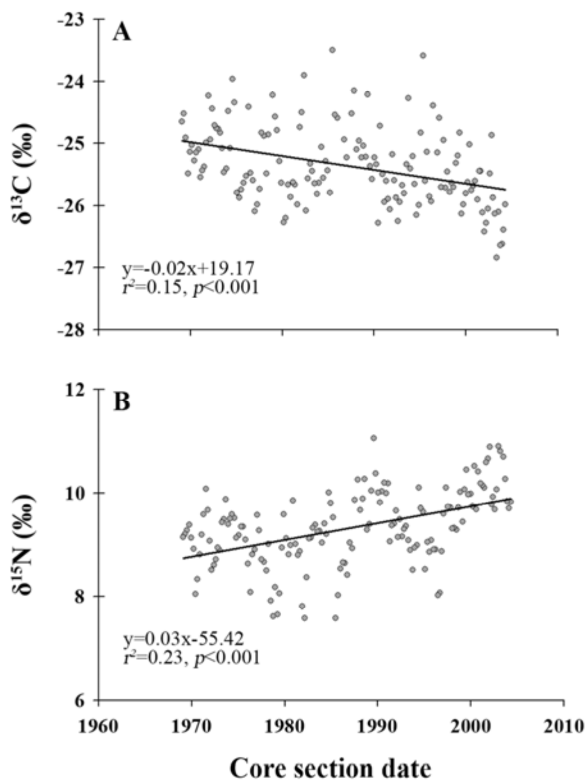

Figure 4.4: (A) C and (B) N stable isotopic compositions of particulate organic matter from sediment core. Linear trend lines were fit to data.

C:N and N:P ratios are displayed in Figure 4.3. C:N ratios linearly decreased (r^2=0.24, p<0.001, N=153; Fig. 4.3A) throughout the sediment core resulting from relatively constant POC concentrations and increasing PON concentrations (Fig. 4.2A and Fig. 4.2B). C:N ratios varied from 11.1 – 27.9. N:P ratios linearly increased (r^2=0.13, p<0.001, N=154; Fig. 4.3B) through time based on differential rates of change between PON and TP concentrations (Fig. 4.2A and Fig. 4.2B). However, N:P ratios increased relatively slowly and the regression explained little N:P ratio variation. N:P ratios varied temporally from 5.3 – 10.1.

Table 4.2: Correlation matrix for C and N stable isotopic compositions versus particulate organic C (POC), particulate organic N (PON), and total P (TP) concentrations and C:N and N:P ratios. Values reported as Pearson correlation coefficients (*r*). Relationships are significant at **p*<0.05, ***p*<0.01, and ****p*<0.001. NS = Not significant.

	Concentrations						
	POC	PON	TP	C:N	N:P	$\delta^{13}C$	$\delta^{15}N$
$\delta^{13}C$	-0.74***	-0.56***	-0.30***	NS	-0.49***	1	-0.44***
$\delta^{15}N$	0.27**	0.65***	0.25**	-0.32***	0.54***	-0.44***	1

Table 4.3: Decadal land use/land cover (LULC) change for Lake Waco watershed. Agricultural lands and rangeland (Ag/Range) have been combined although agricultural lands compose ~90% of the LULC.

	Decade			
LULC	1970s	1980s	1990s	2000s
Forest	32	26	25	44
Ag/Range	59	58	65	43
Urban	7	15	10	12
Water	1	1	1	1

Stable Isotopes

C and N stable isotopic compositions are presented in Figure 4.4. $\delta^{13}C$ linearly decreased (r^2=0.15, p<0.001, N=151) from 1969 to 2004 (Fig. 4.4A). However, $\delta^{13}C$ varied widely around this long-term trend as indicated by the low coefficient of determination. $\delta^{13}C$ became increasingly negative near the sediment core surface. $\delta^{13}C$ varied temporally from -26.8 to -23.5‰. $\delta^{13}C$ and $\delta^{15}N$ were negatively correlated (r=-0.44, p<0.001, N=150; Table 4.2) and therefore displayed a reciprocal relationship through time (Fig. 4.4). $\delta^{15}N$ linearly increased (r^2=0.23, p<0.001, N=155) from the sediment core bottom to surface (Fig. 4.4B). $\delta^{15}N$ had extreme values of +7.6 and +11.1‰ in the sediment core.

Pearson correlation coefficients (r) between C and N stable isotopes and elemental concentrations and ratios are provided in Table 4.2. In order of decreasing strength, $\delta^{13}C$ were significantly correlated to POC concentrations (r=-0.74, p<0.001), PON concentrations (r=-0.56, p<0.001), N:P ratios (r=-0.49, p<0.001), $\delta^{15}N$, and TP concentrations (r=-0.30, p<0.001). All correlations were negative. $\delta^{13}C$ were not significantly correlated to C:N ratios. In order of decreasing strength, $\delta^{15}N$ were significantly correlated to PON concentrations (r=0.65, p<0.001), N:P ratios (r=0.54, p<0.001), $\delta^{13}C$, C:N ratios (r=-0.32, p<0.001), POC concentrations (r=0.27, p<0.01), and TP concentrations (r=0.25, p<0.01). $\delta^{15}N$ were positively correlated to all elemental concentrations and ratios except $\delta^{13}C$ and C:N ratios.

Land Use/Land Cover Changes

Decadal LULC changes are provided in Table 4.3. In the 1970s, agricultural lands and rangeland composed ~60% watershed area with forests (32%) composing the second largest LULC. These LULC categories covered >90% watershed area. Urban lands increased ~2× in the 1980s at the expense of forests as agricultural lands and rangelands remained unchanged. Agricultural lands, rangelands, and forests covered >80% watershed area. In the 1990s, agricultural land and rangeland coverage increased slightly (65%) while urban lands decreased (10%). Forested area remained unchanged. Watershed LULC shifted largely from the 1990s to 2000s. Forests increased by ~20% concurrent with reciprocal decreases in agricultural lands and rangelands. Water consistently composed 1% watershed area.

Discussion

Sediment Core Dating

Reservoir sediment cores are difficult to date because their young age precludes dating techniques commonly employed on their natural lake counterparts. Sediment cores from pre-1950s reservoirs typically contain four stratigraphic markers: reservoir impoundment date, introductory detectable ^{137}Cs activity (1952), peak ^{137}Cs activity (1963), and core collection date (Callender and Van Metre 1997). For reservoirs impounded in the mid-1960s, sediment core stratigraphic markers may be restricted to only reservoir impoundment and core collection date. Additional post-1963 dating markers, such as the Pb concentration peak (1978) and local flood events, may be vital to establishing reliable sediment core dating models in these reservoirs.

Our sediment core documented a truncated history of Lake Waco. Absence of the ^{137}Cs activity peak (Fig. 4.1A) indicated that the sediment core post-dated 1963. This finding agreed with the absence of visual stratigraphic markers likely deposited during reservoir re-impoundment (1965). Both the actual and normalized Pb concentration profiles indicated reduced Pb accumulation above ~170 cm (Fig. 4.1B and Fig. 4.1C). Similar trends would be anticipated following federal government restrictions on leaded gasoline sales (Callender and Van Metre 1997; Van Metre and Callender 1997). The low Pb concentrations and broadly shaped peak (Fig. 4.1B) agreed with sediment Pb concentration profiles from reservoirs with agricultural and rural watersheds in Georgia and Iowa (Callender and Van Metre 1997). According to our dating model, the sediment core did not record the initial four years of lake history.

To independently assess dating model accuracy, we identified historic flood markers using sediment grain size and temporally correlated peaks with North Bosque River discharge and lake surface elevation peaks. We assumed that floods would result in increased delivery of larger sediment grains at this deepwater location. Grain size diameter at the 50^{th} (median) and 95^{th} (largest) percentiles recorded peaks at 39, 77 – 107, and 191 cm (Fig. 4.5). ^{137}Cs activity dilution events (Fig. 4.1A) supported designation of these core sections as flood markers. Our dating model assigned the dates 1999, 1989 – 1993, and 1975 respectively to these peaks. Assigned dates closely agreed (±1 to 2 years) with historic North Bosque River discharge and lake elevation (Fig. 4.6). The large flood of 1991 – 1992 (peak B; Fig. 4.6) likely deposited much sediment resulting in broad grain size peaks (peak B; Fig. 4.5). Because we were interested in decadal trends, our dating model proved adequate.

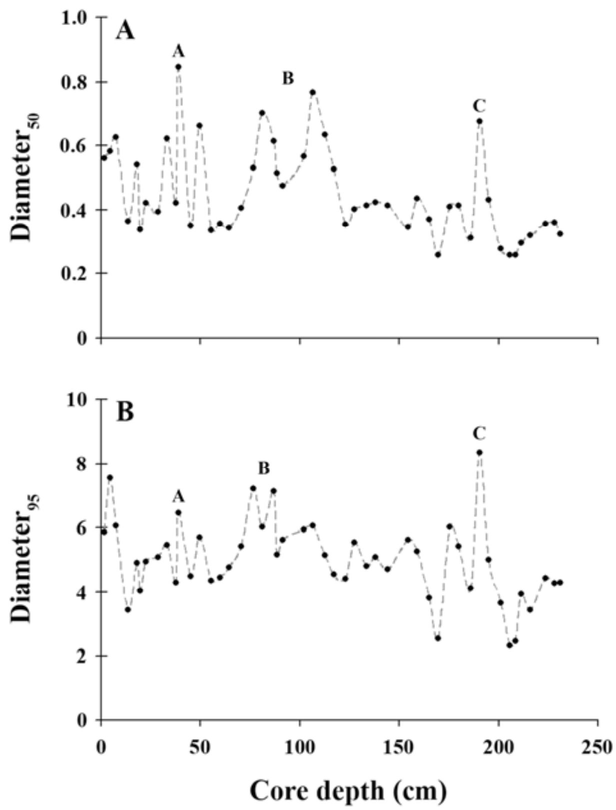

Figure 4.5: Sediment grain size profiles for (A) 50th percentile (median) and (B) 95th percentile (largest) diameters. Grain sizes were determined by X-ray attenuation using a Sedigraph. Peaks were assigned the dates 1999 (A), 1989 – 1993 (B), and 1975 (C) based on our dating model.

The high local sedimentation rate (6.6 cm yr^{-1}) indicated that the coring location experienced sediment focusing. This sedimentation rate was ~9× greater than the surface area-averaged sedimentation rate (0.74 cm yr^{-1}) that we calculated from volumetric surveys conducted in 1970 and 1995 (Texas Water Development Board 2003). These findings agreed with previous studies that identified a nearby location as receiving much secondarily-transported material (Filstrup and Lind unpublished data; Filstrup et al. 2009). Despite temporal resolution loss, our sediment core likely contained continuously-deposited sediments that integrated signals across the watershed which are ideal sediment core characteristics (Shotbolt et al. 2005; Shotbolt et al. 2006). We anticipated minor temporal resolution loss because sediment focusing is an annual process due to Lake Waco's shallowness and mixing potential.

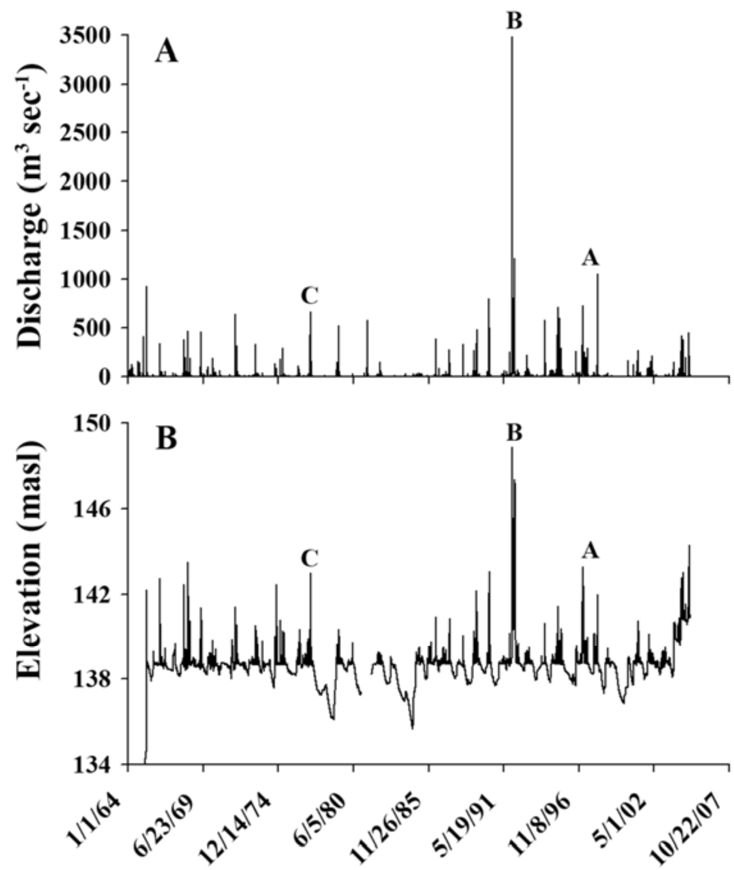

Figure 4.6: (A) North Bosque River discharge (USGS Gauge Station No. 08095200; US Geological Survey Water Data for USA 2009) and (B) Lake Waco water surface elevation (US Army Corps of Engineers-Ft. Worth District Hydrologic Data on Ft. Worth District Lakes 2009). Denoted flood events occurred during 1997 – 1998 (A), 1991 – 1992 (B), and 1977 (C).

Paleolimnological Eutrophication Indicators

Reservoir sediments are important environmental archives that are currently underutilized (Shotbolt et al. 2005; Shotbolt et al. 2006). We attempted to determine if sediment elemental concentrations and ratios and OM isotopic compositions chronicled a polymictic reservoir's eutrophication history. We hypothesized that POC and PON concentrations indicated increasing algal biomass and TP concentrations indicated reservoir nutrient enrichment. Data supported this hypothesis but also indicated that PON concentrations better approximate historic algal biomass than POC concentrations. Both POC (r^2=0.04, p<0.05) and PON (r^2=0.54, p<0.001) concentrations significantly increased through time (Fig. 4.2A and Fig. 4.2B). However, scattered POC concentrations suggested that seasonal allochthonous OM loading differences created short-term "noise" that masked decadal trends. POC concentrations are strongly biased by variable allochthonous OM loading based on vascular land plant elemental compositions (C-rich, N-poor; Meyers and Teranes 2001). Contrastingly, PON concentrations adequately represented decadal net PP trends (Table 4.1). The significant correlation between PON and TP concentrations (r^2=0.37, p<0.001) and lack of correlation between POC and TP concentrations further supported the use of PON concentrations as algal biomass indicators in this P-limited system. We agree with Lehmann et al. (2004b) that PON concentrations may be better PP proxies than POC concentrations in systems receiving substantial allochthonous OM loads.

We hypothesized that decreasing C:N ratios recorded an OM source shift towards increasingly algal-derived OM and decreasing N:P ratios recorded increasing N limitation. C:N ratios significantly decreased (r^2=0.24, p<0.001) throughout the sediment core therefore supporting our hypothesis (Fig. 4.3A). Based on different structural requirements, phytoplankton has C:N ratios between 4 and 10 whereas vascular land plants have C:N ratios ≥20 (Meyers and Teranes 2001). C:N ratios (11.1 – 27.9) suggested that sediment OM composition fluctuated between primarily algal to largely terrestrial provenance. Short-term variation was likely caused by seasonal sediment OM source shifts (Filstrup et al. 2009). Decadal trends indicated that algal OM composed increasingly greater sediment OM fractions as the reservoir became P-enriched which agreed with historic net PP and P data (Table 4.1).

Unexpectedly, N:P ratios increased (r^2=0.13, p<0.001) through time therefore countering our hypothesis (Fig. 4.3B). We anticipated that N:P ratios would decrease as Lake Waco experienced P enrichment and PP became increasingly N limited. However, N limitation may not have been realized in this reservoir because N-fixing cyanobacteria potentially maintained P limitation. As P loading increased with reservoir age (Fig. 4.2C), primary productivity may not have become N limited because increasing N-fixation rates compensated for lower relative nitrate concentrations. Scott et al. (2008) documented a similar seasonal mechanism in which N-fixation rates were highest during July through September concurrent with low nitrate concentrations and warm temperatures. Phytoplankton likely fixed the minimum amount of atmospheric N_2 required to remove N deficiency because the process is energetically expensive. Therefore algal elemental composition likely remained relatively consistent which potentially explained the relatively flat decadal N:P ratio trend (Fig. 4.3B). Additionally, a phytoplankton compositional shift towards cyanobacteria, which have a relatively high

N content (Rodriguez et al. 1989), could have explained relatively consistent N:P ratios with P enrichment.

We hypothesized that OM $\delta^{13}C$ indicated increased PP as Lake Waco aged. OM $\delta^{13}C$ did not directly record decadal PP fluctuations as has been documented from stratified natural lakes (Schelske and Hodell 1991; Schelske and Hodell 1995; Hodell and Schelske 1998). Decreasing $\delta^{13}C$ (r^2=0.15, p<0.001; Fig. 5A) suggested that $\delta^{13}C$ and PP (as documented by historic data; Table 4.1) were inversely related. In stratified natural lakes, phytoplankton becomes increasingly ^{13}C-enriched during highly productive periods because epilimnetic DIC pools become progressively heavier as sedimenting phytoplankton preferentially removes ^{12}C from the epilimnion (Hodell and Schelske 1998). However, this reservoir effect may not occur in well-mixed reservoirs (Filstrup et al. 2009). The weak correlation between $\delta^{13}C$ and TP (r=-0.30, p<0.001) indicated that decadal $\delta^{13}C$ trends were largely influenced by factors other than PP (Table 4.2).

OM source distinctions likely biased $\delta^{13}C$ – PP relationships in this reservoir. Large and varying allochthonous OM loads can obscure relationships between $\delta^{13}C$ and PP (Schelske and Hodell 1991; Schelske and Hodell 1995; Lehmann et al. 2004b) complicating $\delta^{13}C$ profile interpretation in reservoirs. We were unfortunately not able to adequately characterize terrestrial $\delta^{13}C$ end-members for this large, diverse watershed. Typically, $\delta^{13}C$ cannot discriminate algae from C_3 land plants but can distinguish algae and C_3 land plants from C_4 land plants (Meyers and Teranes 2001). C_3 pathways (approximately -20‰) produce greater isotopic shifts than C_4 pathways (-8 to -12‰; Meyers and Ishiwatari 1993). Declining $\delta^{13}C$ likely recorded decreasing relative contributions of terrestrial OM, especially C_4 prairie grasses, as PP increased. This decadal trend supported long-term OM source shifts identified from C:N ratios. We did not expect $\delta^{13}C$ and C:N ratios to be correlated (Table 4.2) because of source distinction differences (C_3 versus C_4 plants and aquatic versus land plants respectively). Short-term $\delta^{13}C$ variability likely resulted from seasonal/annual allochthonous OM loading fluctuations. $\delta^{13}C$ were most strongly correlated to POC concentrations (r=-0.74, p<0.001) which suffered from similar biases (Table 4.2).

We hypothesized that OM $\delta^{15}N$ indirectly estimated PP through water column nitrate utilization degree. Decadal $\delta^{15}N$ trends appeared to support our hypothesis. Numerous studies (Altabet and Francois 1994; Hodell and Schelske 1998; Teranes and Bernasconi 2000) have documented the successful application of sediment OM $\delta^{15}N$ in recording epilimnetic nitrate utilization. Phytoplankton decreasingly discriminates against ^{15}N as epilimnetic DIN pools become progressively heavier resulting from preferential assimilation and hypolimnetic transport of ^{14}N (Hodell and Schelske 1998). Increasing OM $\delta^{15}N$ (r^2=0.23, p<0.001) suggested that phytoplankton progressively assimilated more nitrate relative to water column concentrations (Fig. 4.4B). Decadal trends indirectly recorded increased PP through time because phytoplankton assimilated progressively greater fractions of increasing DIN supplies (see nitrate concentrations; Table 4.1). $\delta^{15}N$ were most strongly correlated to PON concentrations (r=0.65, p<0.001) which were reliable PP indicators. Annual nitrate concentration drawdown and seasonal N-fixation (Scott et al. 2008; Scott et al. 2009) may have influenced short-term $\delta^{15}N$ fluctuations.

Alternatively, external N source changes associated with watershed land use change may have artificially produced $\delta^{15}N$ – PP relationships in Lake Waco. DIN $\delta^{15}N$

shifts can more strongly control OM $\delta^{15}N$ than nitrate assimilation fractionation especially through decadal and centurial timescales (Teranes and Bernasconi 2000; Lehmann et al. 2004a; Lehmann et al. 2004b). Agricultural runoff and wastewater discharge likely influenced DIN isotopic composition. Animal waste and sewage are isotopically heavy ($\delta^{15}N$=10 to 25‰; Kendall 1998; Fry 2006). Human population growth and dairy operation intensification potentially shifted DIN pools to progressively heavier isotopic compositions. Also, watershed urbanization in the 1970s and increased agricultural land and rangeland activities (i.e. dairy manure application) in the 1990s probably induced sustained shifts in DIN $\delta^{15}N$ (Table 4.3). Watershed afforestation in the 2000s did not lead to isotopically lighter DIN pools (Table 4.3 and Fig. 4.4B). The weak correlation between $\delta^{15}N$ and TP (r=0.25, p<0.01) indicated that factors other than PP, such as DIN source shifts, strongly affected OM isotopic compositions (Table 4.3).

Management Implications

Reservoir managers are failing to take advantage of powerful management techniques used by natural lake managers because of concerns associated with recent sediment dating models and possible sediment disturbance. Our study documented difficulties encountered that may be experienced by others. Paleolimnological studies should preferentially be performed on sediment cores containing pre-impoundment soils to establish an inclusive dating model. Despite missing this stratigraphic marker, we developed an adequate dating model (±1 to 2 years) to investigate decadal PP trends. Dating models for recent reservoir sediments (post-1960s) will likely benefit from stratigraphic markers associated with local flood events and peak atmospheric Pb deposition. Seasonal fluctuations in elemental and isotopic signals, particularly POC concentration and $\delta^{13}C$, confounded interpretation of decadal trends. This "noise" can be removed by sectioning at coarser intervals (at least annual), combining finer sections pre-analyses, or averaging finer sections post-analyses if investigators are primarily interested in long-term trends. The convention of sectioning at 1-cm intervals may not apply to reservoir sediments. Current data concerning algal and land plant stable isotope end-members and dissolved inorganic nutrient stable isotopic compositions will strengthen stable isotope interpretations. Additional paleolimnological studies performed on reservoirs will help refine methodologies and data interpretations so reservoir managers can extract the maximum information content stored in these archives. In particular, studies performed on small, well-mixed reservoirs will help identify system and/or mixing pattern characteristics that create indicator – PP relationships atypical of stratified natural lakes.

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CHAPTER FIVE

Conclusions

My research investigated organic matter provenance through annual and decadal timescales and sediment transport mechanisms influencing organic matter delivery to reservoir sediments. I examined annual and decadal allochthonous organic matter loading fluctuations, explored decadal phytoplankton productivity trends, determined seasonal sediment transport mechanisms, and assessed phytoplankton isotopic composition – productivity relationships in this eutrophic, shallow, rapidly-flushed reservoir.

Allochthonous sources contributed much organic matter through both annual and decadal timescales. Annually, organic matter originated from primarily allochthonous sources during winter and autochthonous sources thereafter. Decadally, organic matter provenance shifted from largely allochthonous sources after impoundment towards autochthonous sources with time.

Phytoplankton productivity increased through time concurrent with phosphorus enrichment. Paleolimnological indicators adequately recorded documented reservoir eutrophication. However, variable allochthonous organic matter loading and changing dissolved inorganic nutrient sources associated with changing land use complicated paleolimnological indicator interpretation.

Sediment transport mechanisms including river plume sedimentation, sediment resuspension, and horizontal advection, influenced organic matter deposition with individual mechanisms being more important seasonally. Sediment resuspension was largely influenced by river discharge and wind-induced mixing in the riverine and lacustrine regions respectively. Wind-induced mixing entrained deep-water advective river sediments into the photic zone rather than resuspending surface sediments.

Phytoplankton carbon isotopic composition – productivity relationships during this study opposed those advanced from stratified natural lakes. Phytoplankton maintained photosynthetic isotopic discrimination during productive periods because water column mixing likely maintained adequate inorganic carbon concentrations in the photic zone. However, phytoplankton isotopic composition may have also been influenced by changing dissolved inorganic nutrient pools in this reservoir.

Although reservoirs have been described as river-lake hybrids, these ecosystems may metabolically function as either rivers or natural lakes based on organic matter composition. Ecosystem structure and function are dynamic as organic matter composition varies through time. Frequent disturbances, including nutrient loading from the watershed, maintain these ecosystems in primitive successional states. Management efforts should address these disturbances therefore allowing autogenic succession to proceed.

APPENDICES

APPENDIX A

Publications Related to This Research

Chapter Two

Filstrup CT, Lind OT. In Revision. Sediment transport mechanisms influencing spatiotemporal resuspension patterns in a shallow, polymictic reservoir. Lake and Reservoir Management.

Chapter Three

Filstrup CT, Scott JT, Lind OT. 2009. Allochthonous organic matter supplements and sediment transport in a polymictic reservoir determined using elemental and isotopic ratios. Biogeochemistry 96:87-100. DOI 10.1007/s10533-009-9346-4.

Chapter Four

Filstrup CT, Scott JT, White JD, Lind OT. In Revision. Sediment elemental and isotopic compositions record a polymictic reservoir's eutrophication. Lakes and Reservoirs: Research and Management.

APPENDIX B

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