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Documenting the History of Oxygen Depletion in Lake St. Croix, Minnesota, Using Chironomidae Remains in the Sedimentary Record

Caitlin E. Stewart
University of Massachusetts Amherst

DOCUMENTING THE HISTORY OF OXYGEN DEPLETION IN LAKE ST. CROIX, MINNESOTA, USING CHIRONOMIDAE REMAINS IN THE SEDIMENTARY RECORD

A Thesis Presented

by

CAITLIN EYRE STEWART

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Approved as to style and content by:

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Donna Francis, Chair

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William McCoy, Member

_____
Michael L. Williams,
Department Head, Geosciences

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ABSTRACT

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SEPTEMBER 2009

CAITLIN EYRE STEWART, B.S., SUNY COLLEGE AT ONEONTA

M.S., UNIVERSITY OF MASSACHUSETTS AMHERST

Directed by: Professor Donna Francis

Lake St. Croix is a natural impoundment located at the southern end of the St. Croix River. Land use changes since European settlement (c. 1850) have resulted in nutrient runoff, eutrophication, and periodic oxygen depletion in the hypolimnion of Lake St. Croix. Establishing sound lake management practices requires knowledge of historical conditions obtained through paleoecological studies. Remains of non-biting midges (Insecta: Diptera Chironomidae) in lake sediments have been shown to be reliable indicators of past hypolimnetic oxygen conditions. Cores from two sub-basins in the lake were collected in 2006. Midge analysis indicated that shifts in species assemblages

correspond to the times of land use change. *Chironomus* and *Procladius*, which are tolerant of low oxygen levels, increased in relative abundance as land use changes adversely impacted the St. Croix River's watershed. Volume-weighted hypolimnetic oxygen concentrations were estimated using a transfer function developed for southern Ontario. Mean post-settlement chironomid reconstructed average volume-weighted hypolimnetic oxygen values were 0.73 mg/L lower than mean pre-settlement values for sub-basin 1, near Prescott, WI and 0.45 mg/L lower for sub-basin 3, near Lakeland, MN. These results indicate that oxygen depletion has occurred in the lake since the time of European settlement, and are supported by increases in the relative abundance of eutrophic midge bioindicators and the decrease in relative abundance of bioindicators of less productive conditions since the 1850s. This study, in conjunction with other historical and paleoecological studies of Lake St. Croix, provides historical data for setting management goals and strategies for Lake St. Croix.

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LIST OF SYMBOLS AND ABBREVIATIONS

CCA	Canonical Correspondence Analysis
cm	Centimeter
cm^3	Cubic centimeter
DCA	Detrended correspondence analysis
DCCA	Detrended canonical correspondence analysis
DNR	Department of Natural Resources
g	Grams
H’	Shannon-Wiener diversity index
km	Kilometer
LSC	Lake St. Croix
m	Meter
MCES	Metropolitan Council Environmental Services
mg/L	Milligram per liter (parts per million)
ml	Milliliter
MPCA	Minnesota Pollution Control Agency
MSCL	Multisensor Core Logger
P	Phosphorus
P	P-value, or probability value
PCA	Principal component analysis
PLS	Partial least squares
ppb	Parts per billion

r^2	Coefficient of determination
r^2_{jack}	Squared correlation between jackknife predicted and observed values
RDA	Redundancy Analysis
RMSEP	Root mean square error of prediction, predictive error
s	Second
SCR	St. Croix River
SI	International System of Units
T	Tons
µm	Micrometer
VWHO	Volume-weighted hypolimnetic oxygen
WA	Weighted averaging
WAPLS	Weighted averaging partial least squares
Years BP	years before present

CHAPTER 1

INTRODUCTION

1.1 Purpose and Approach

Humans have the ability to adversely affect the integrity of the environment, and these actions often harm river and lake ecosystems (Meybeck and Helmer, 1989; Smith, 2003). Land use change in the St. Croix River's (hereafter, SCR) watershed beginning at the time of European settlement has resulted in nutrient runoff, eutrophication, and periodic oxygen depletion in the hypolimnion of Lake St. Croix (hereafter, LSC), Minnesota (hereafter MN)/Wisconsin (hereafter WI) (Troelstrup et al., 1993; Triplett et al., 2009; Edlund et al., 2009; Lafrancois et al., 2009). LSC is a riverine lake at the lower end of the SCR whose water quality is impacted by the inlet of the river and tributaries that flow into the lake. Total phosphorus and other pollutants enter the lake through these inlets. In addition, deforestation, agricultural practices, urbanization, and recreation have resulted in changes to the SCR watershed that impact LSC.

The hypolimnion is the bottom, most dense layer in a thermally stratified lake with inadequate light penetration for photosynthesis to occur (Brönmark & Hansson, 1998). In addition, oxygen concentration is naturally low in this layer compared to the epilimnion (upper, wind mixed layer of a thermally stratified lake) and metalimnion (transition zone where the thermocline, or the depth where temperature gradient is greatest during the summer, occurs) due to the absence of wind mixing that contributes oxygen from the atmosphere to the lake. Hypolimnetic oxygen depletion occurs when organic matter in the epilimnion sediments down to the benthos, and bacterial decomposition of organic matter depletes the hypolimnion of oxygen (Little and Smol,

2001). In addition, aerobic respiration consumes oxygen in the hypolimnion (Cornet, 1989).

Indicators of eutrophication, including altered diatom communities and increased phosphorus loading, occurred in the late 1800s and in the 1950s, and have raised concern in both the scientific and public communities for the health of the LSC ecosystem (Triplett et al., 2009). In the 1990s, sediment and nutrient loading were designated as the focus of riverway management strategies (Davis, 2004). In 2008, the states of MN and WI placed LSC on their list of impaired waters due to excess nutrients and eutrophication (Magdalene, 2009). In order to effectively implement plans that will improve the water quality of LSC, it is important to understand what conditions were like before European settlement and how those conditions changed over time as a result of human disturbance.

The purpose of this project was to reconstruct historical oxygen conditions in Lake St. Croix and test previous historical and paleoecological eutrophication studies that conclude LSC has been adversely impacted by land use change since the time of European settlement. Water quality of LSC has been monitored by the Metropolitan Council Environmental Services (hereafter MCES) since 1976, providing recent data on variables including total nitrogen, total chlorophyll *a*, and total phosphorus (Lafrancois et al., 2009). In order to determine if modern day aquatic conditions in LSC are the result of anthropogenic eutrophication or are natural, it is essential to understand what lake conditions were like before European settlement. Paleoecology uses information in the sediment record to reconstruct past environmental conditions. In this study, another indicator of eutrophication, deep-water oxygen loss, was examined using subfossil midge (Chironomidae) remains from two sediment cores. Results of this study will provide

historical data for the interagency St. Croix Basin Planning Team who determines goals and management strategies for the SCR, as well as other concerned organizations, residents, and visitors. Objectives of this study were to:

1) Reconstruct historical oxygen conditions in LSC using subfossil Chironomidae remains from lake sediment cores.
2) Identify changes in Chironomidae communities in LSC resulting from eutrophication and low oxygen conditions.
3) Correlate Chironomidae community shifts and oxygen depletion with other signals of eutrophication that have been studied in LSC, including diatom species shifts and increased phosphorus concentrations.

1.2 Site Description

LSC is a 37 km-long natural impoundment located at the southern end of the 266-km SCR, stretching from Stillwater, MN to Prescott, WI, where the SCR flows into the Mississippi River (Troelstrup et al., 1993) (Figure 1). LSC was created c. 9500 years BP by two events (Eyster-Smyth et al., 1991). First, at the confluence of the Mississippi and Chippewa Rivers, an alluvial fan formed that impounded the Mississippi River, forming Lake Pepin (Troelstrup et al, 1993). The alluvial fan impounded water that was forced upstream. Second, outflow from glacial Lake Agassiz decreased that resulted in the formation of an alluvial deposit at Point Douglas, impounding LSC (Troelstrup et al, 1993). Four sub-basins were created in LSC as a result of secondary deposition from side valley tributaries of the Kinnickinnic River, Valley Creek, and Willow River (Figure 2). Table 1 describes the four sub-basins.

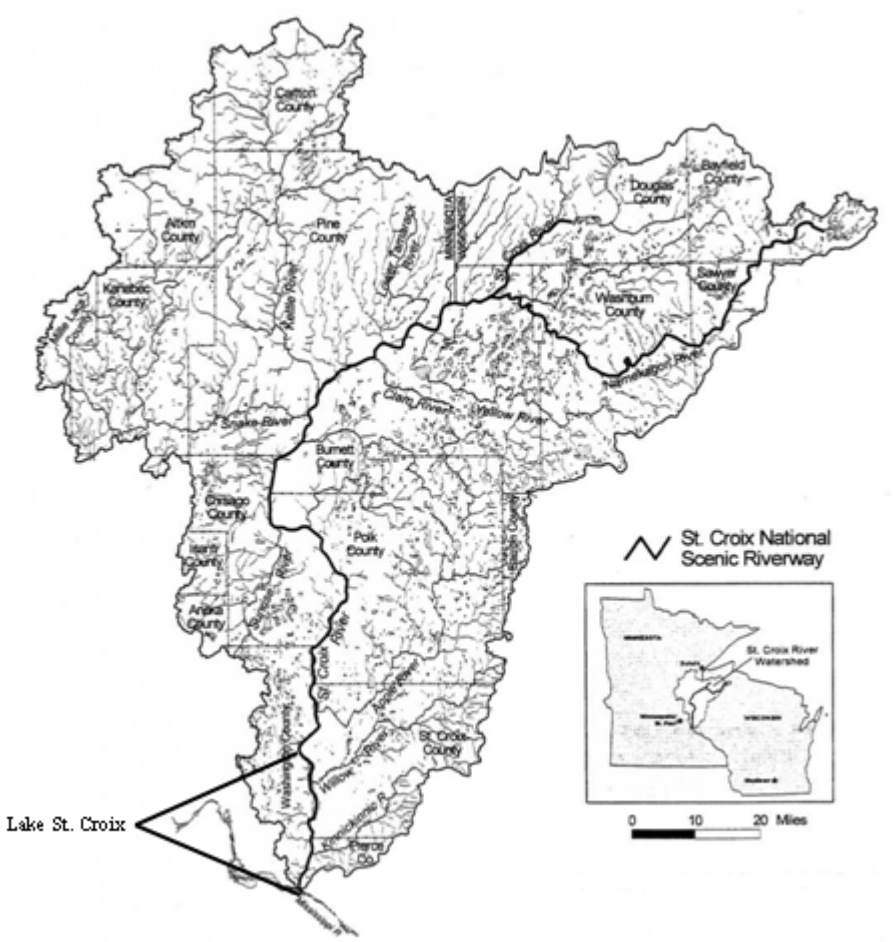

Figure 1. Map of the St. Croix River watershed, the St. Croix National Scenic Riverway, and Lake St. Croix (modified from the National Park Service National Scenic Riverway).

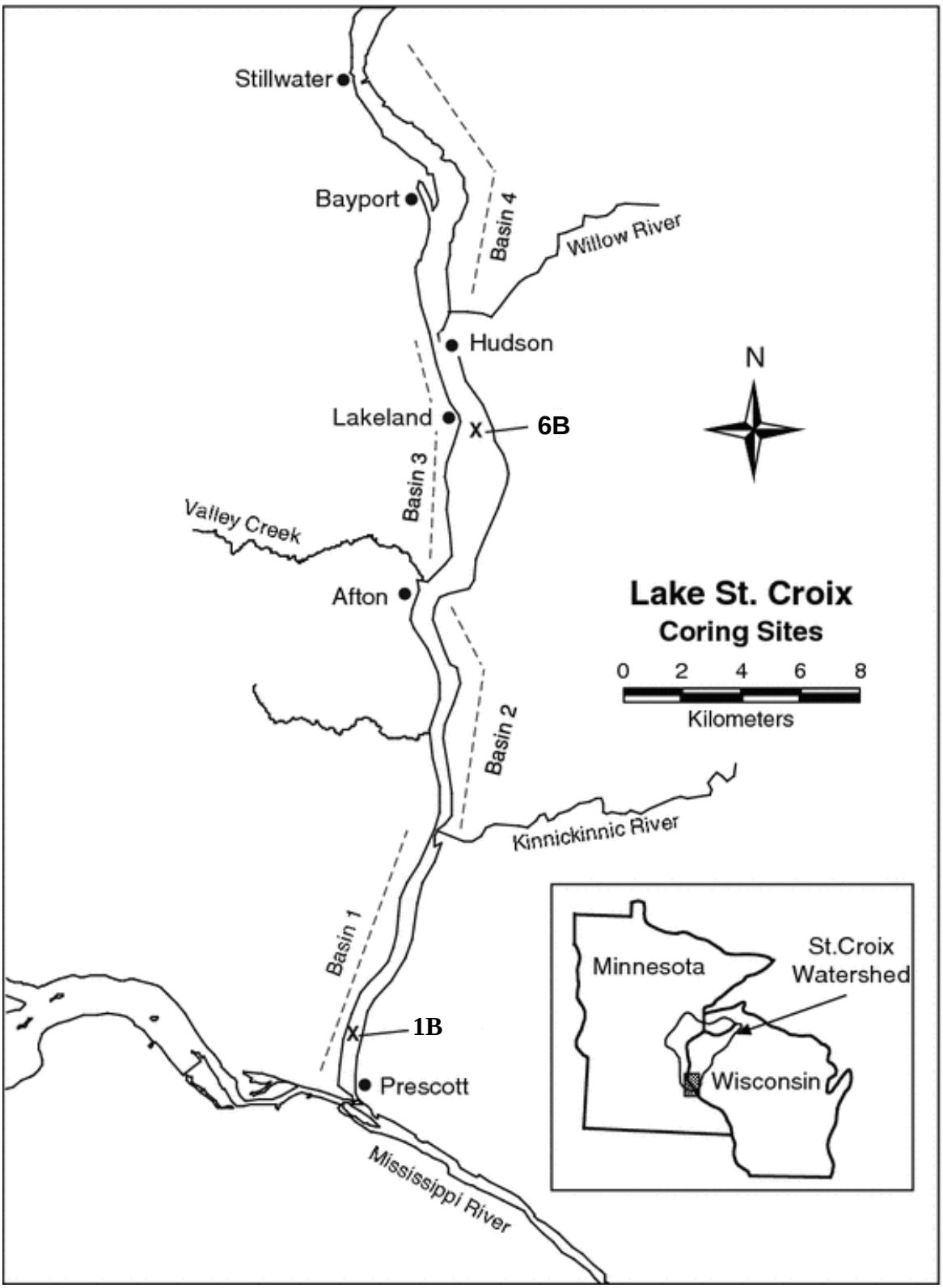

Figure 2. Locations of 1B and 6B coring sites and corresponding sub-basins in Lake St. Croix (modified from Edlund et al., 2009).

Table 1. Description of the 4 sub-basins in Lake St. Croix.

Basin	Location	Maximum depth
Four	Stillwater to Willow River Bar, Hudson, WI	c. 10 m
Three	Willow River Bar to Catfish Bar (Valley Creek mouth), Afton, MN	c. 20 m
Two	Afton, MN to the Kinnickinnic Bar	>22 m
One	Kinnickinnic Bar to mouth of St. Croix River, Prescott, WI	>17 m

LSC is classified as riverine, meaning that the transportation of water, sediment, and nutrients occurs along one linear flow axis (Triplett et al., 2008). The total drainage area of the SCR that empties into LSC is 22,196 km^2 (Graczyk, 1986). Lake surface area is 35 km^2, and water residence time is 20 to 50 days (Triplett et al., 2008). Fine grained sediments have been deposited on the bottom of LSC beginning at the time of formation, and the substrate is composed of organics and sand (Anderson and Varro, 2002).

In 1968, 406 km of the SCR and Namekagon River, WI, the SCR's largest tributary, were designated the St. Croix National Scenic Riverway under the National Wild and Scenic Rivers Act because of its scenic, recreational, and environmental qualities (Anderson and Varro, 2002). The lower portion of the SCR below St. Croix and Taylors Falls was incorporated in 1972, and the riverway presently stretches from northern WI to the Mississippi/SCR confluence. The National Park Service administers the Riverway from the St. Croix and Namekagon River headwaters to Stillwater, MN, and the Minnesota Department of Natural Resources (hereafter DNR) and the Wisconsin DNR administer the section below Stillwater. Under the designation of National Scenic

Riverway, scenic easements and private lands were acquired, and land and water use is regulated. Despite these protection measurements, the NPS does not regulate tributaries emptying water contaminated with nutrients and sediment into the SCR. Federally listed endangered species that reside in this area include the Higgins' eye and winged mapleleaf mussels, cougars, wolves, peregrine falcons, bald eagles, and the Karner blue butterfly (Jennings and McGuiness, 2009). The river provides the necessary habitat to support an ecosystem rich in biodiversity, and also provides fertile soils for agriculture, and a landscape that is sought after by recreationists and developers alike.

1.3 Land Use Change

The SCR watershed is 19,900 km^2 and has undergone a sequence of land use changes that have impacted LSC. Before 1850, Native Americans impacted the land by intentional and unintentional burning of vegetation, but these land use changes did not significantly alter water quality (Curtis, 1959 as cited in Triplett et al., 2009). By 1840, European milling and logging operations were impacting the landscape. Throughout the 1900s, agriculture was a major land use in the region, and after 1950, urbanization and increasing population impacted the watershed (Triplett et al., 2009). In the early 1990s, the northern portion of the SCR watershed is dominated by forested areas, while the southern portion is dominated by agriculture and developed areas (Figure 3).

The St. Croix Valley was first inhabited by the Dakotas, hunters and gatherers who established villages on large lakes in present day Minnesota and Wisconsin (McMahon and Karamanski, 2002). The Ojibwa Indians were also early peoples of the SCR. At the start of the 18^{th} century, French fur traders navigated up the Mississippi

River and in 1679, Daniel Greysolon, Sieur du Luth declared Dakota territory on the SCR for the French (McMahon and Karamanski, 2002). British fur traders arrived in the region in the 1780s followed by the Americans. European colonists arrived in the region in the 1830s, and as trappers exhausted animal populations in the region, milling and logging operations began in the 1840s.

Lumbering was Minnesota's major industry during the mid 1800s, and the SCR was used for log driving (Bachmann, 1945). In 1838, the Dakota and Ojibwa Indians signed treaties that allowed logging practices in the SCR Valley. White pines were heavily harvested from the tributaries to the SCR and areas north of St. Croix Falls and Taylors Falls. During the winter months, logs were cut and stacked near the tributaries. Snowmelt during the spring raised water levels and logs were driven from the tributaries to the SCR. In the late 1800s, the SCR's logging era peaked and gave way to land clearance and agricultural practices.

By 1880, the St. Croix valley prairies were completely settled and spanning the early 1900s, agricultural operations cycled from wheat to dairy to corn (Triplett et al., 2009). In 1920, population in the St. Croix watershed reached 250,000, and by 1992, had increased to 400,000. St. Paul and Minneapolis saw a population explosion in the 1950s and 1960s. These increases in population resulted in urban and recreation stress to the SCR (Davis, 2004).

Today, the northern portion of the St. Croix Basin there is increasing development around small lakes and riparian areas as large lakes have reached full development (Anderson and Varro, 2002). The northern portion is not suitable for farming, and is forested. The southern portion is dominated by agriculture, including dairy operations,

and grain and vegetable production (Figure 3). From 1973 to 1993, recreational uses in the southern portion of the Saint Croix Basin doubled. Despite the classification of the lower SCR as wild and scenic, zoning decisions have allowed large structures to be erected as close as 4 m from the riverbank, threatening the integrity of the river (Jennings and McGuiness, 2009).

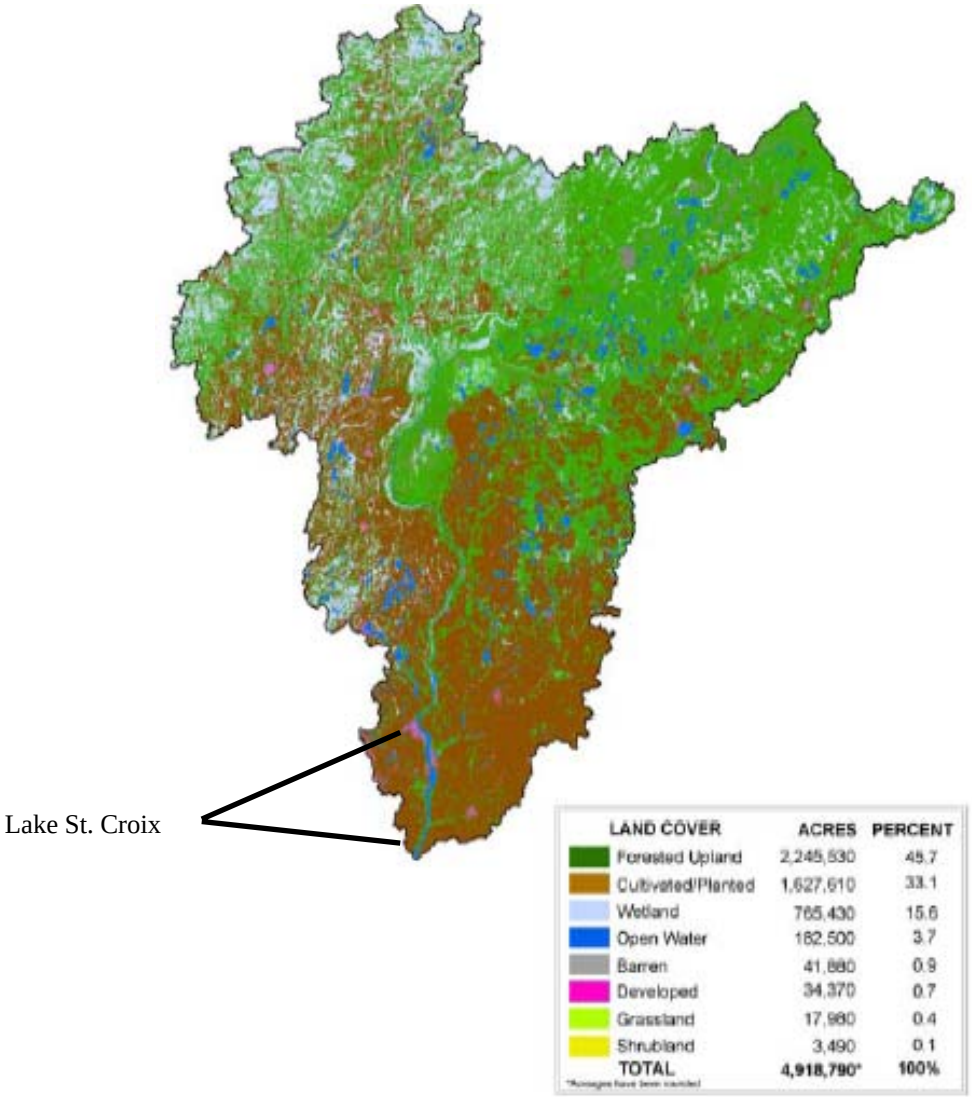

Figure 3. St. Croix River watershed 1992 land cover (from Davis, 2004, map produced by the National Park Service).

1.4 Eutrophication of Lake St. Croix

Since the time of European settlement, the water quality of LSC has been altered as a result of land use change (Troelstrup et al., 1993, Triplett et al., 2009, Edlund et al., 2009, Lafrancois et al., 2009). Recently, LSC has been impacted by pressures from an increasing population in Washington County, MN and St. Croix County, WI, urban development in the Minneapolis/St. Paul metropolitan area, and the annual influx of over 1 million recreationists. Tributaries and the main stem of the SCR have contributed elevated nutrient loads to LSC, and eutrophication has resulted (Troelstrup et al., 1993). In the 1990s, environmental organizations and resource managers raised concerns about the intense recreation and land development that had been impacting LSC (Davis, 2004). In 2008, the Minnesota Pollution Control Agency (hereafter MPCA) classified LSC as impaired due to high phosphorus levels (MPCA, 2008). In addition, the MPCA classifies LSC as eutrophic based on Secchi depth, total phosphorus, and chlorophyll *a* (a green photosynthetic pigment found in plants and algae) measurements (MPCA, 2001).

The productivity of a lake, or its trophic status, is measured in terms of total phosphorus levels that promote algal growth and decrease water clarity (Carlson, 1977). Robert Carlson developed a trophic state index that measures lake productivity using Secchi depth readings, chlorophyll *a* and total phosphorus data (Carlson, 1977). Phosphorus is a limiting factor for algae growth; that is, the concentration of this nutrient will inhibit or promote algae and plant productivity, abundance, and growth. Water transparency is determined by the amount of algae in the water column as well as suspended sediments (Simpson, 1991). Under this index, lakes may be classified into 4 categories: oligotrophic (clear water, oxygen present year round in the hypolimnion),

mesotrophic (moderately clear water, hypolimnetic anoxia may occur during the summer), eutrophic (anoxia occurs in the hypolimnion during the summer, plants are abundant, and transparency is low), and hypereutrophic (heavy algal blooms persist during the summer, dense macrophytes grow but light penetration limits abundance) (Carlson, 1977).

From the time of its formation, a lake naturally ages as it fills in with organic matter, sediments, and silt (Simpson, 1991). This aging process may be accelerated by anthropogenic activities, such as agricultural runoff, waste water treatment plant overflows, or urban runoff, that increase the amount of nutrients in a body of water. High levels of nutrients promote the abundance of algae and plants and decrease water transparency. As dead plants and algae sink to the bottom of the lake, decomposition takes place, depleting the water of oxygen. Hypoxia (low amounts of oxygen) or anoxia (no oxygen) may occur in the hypolimnion that could result in fish kills and dead zones (Brönmark and Hansson, 2005).

Phosphorus sources to LSC include point sources discharged from a visible pipe, factory, or outlet, and non-point sources, where rain or snowmelt picks up pollutants and carries them into a water body. Natural background non-point sources to LSC, such as surface runoff of nutrients, groundwater discharge of nutrients, and windblown sediments, have remained constant at 166 T/yr since 1880 (Triplett et al., 2009). Anthropogenic nonpoint source pollution to LSC includes human-induced stream bank erosion, surface runoff from concentrated animal feeding operations, urban runoff, inorganic fertilizers, livestock feed supplements, and individual sewage treatment systems. Anthropogenic point source pollution to LSC includes wastewater treatment

facilities, industrial discharges, separated or combined sewer outfalls, construction sites, and municipal separate storm water sewer systems. Anthropogenic nonpoint source pollution accounted for 60 % of total nonpoint source pollution to LSC in the 1990s, while anthropogenic point source pollution, calculated from wastewater treatment discharge data, accounted for 11 % of total anthropogenic point source loads in the 1990s (Magdalene, 2009).

1.5 Previous Studies of Eutrophication

A number of paleolimnological studies have shown that a plethora of eutrophic signals have occurred in LSC since European settlement. Troelstrup et al. (1993) extracted sediment cores from LSC near Bayport, MN (LSC1), Lakeland, MN (LSC2), and Afton, MN (LSC3) in order to analyze trophic changes since the time of European settlement. Times of deforestation and agricultural practices correlated to times of increased sediment organic matter and carbonates. European settlement and white pine harvests in the late 1840s and early 1850s correspond to peaks in organic matter. Beginning in the 1950s, primary production has shown a large increase, and since the 1960s, cynaobacterial blooms have been documented in LSC (Brook, 1966 as cited in Troelstrup et al., 1993). Chlorophyll, carbonate, and organic levels increased since the 1950s, and these increases resulted from anthropogenic eutrophication of LSC (Troelstrup et al., 1993).

Troelstrup et al. (1993) also analyzed 18 to 20 core sections of all 3 cores for subfossil chironomids based on the stratigraphic patterns in organic matter, carbonate, and chlorophyll content. Chironomids were often identifiable to genus, but graphs were

constructed using subfamily data at a low resolution. No quantitative reconstructions were performed. Midge density was higher in the deepest core sections and displayed an up core decrease. Density was low, ranging from 1 to c. 15 per cm^3. For normal temperate lakes, midge densities should be greater than 100 per cm^3 (Walker, 1993). Troelstrup et al. (1993) attributed low midge densities to anoxic conditions in LSC. Taxa ranged from 4 to 16 genera per section. Troelstrup et al. (2003) discovered that the communities of all 3 cores were dominated by *Chironomus* and *Procladius*. Taxa richness in LSC2 extracted in sub-basin 3 was highest in the upper core sections. The majority of taxa found in high abundance were classified as littoral or profundal. Dissolved oxygen levels were at or near 0 in the LSC sub-basins, resulting in the high relative abundance of *Chironomus*.

Triplett et al. (2009) extracted 24 cores from 8 transects in the 4 sub-basins of LSC in 1999. The rate of sediment accumulation and phosphorus loading were analyzed. Sediment accumulation showed an increase in 1850, and a peak occurred from 1950-1960. These accumulation levels were 8 times higher than pre-European settlement levels. Total phosphorus (TP) load to LSC showed a significant increase after 1940, with post-settlement values 4 times higher than pre-settlement values. Logging and agricultural practices from 1850 to 1890 resulted in minor impacts to LSC. The sediment and phosphorus load peaks that occurred in the 1950s caused by urbanization lend to the conclusion that during this time, land use change significantly contributed to the eutrophication of LSC more than land use change in the mid 1800s.

Cores from the Triplett et al. (2009) study from sub-basins 1 and 3 were used for diatom, biogenic silica, and fossil pigment analysis in a study conducted by Edlund et al.

(2009) in order to reconstruct historical water column total phosphorus. It was concluded that a 3-fold increase in inorganic sediment accumulation took place from the 1850s to the present. After the mid 1950s, a 6-fold increase was seen in biogenic silica accumulation, a 20 to 50 fold increase was seen in diatom accumulation and a shift from benthic to planktonic diatom taxa occurred. Schelske (1999) reported that in Lake Apopka, Florida, increased phosphorus levels drove the shift from a benthic dominated diatom community to one dominated by planktonic taxa. Diatom bioindicators of eutrophy, including *Fragilaria crotonensis*, *Cyclostephanos invisitatus*, and *C. tholiformis*, increased in abundance after 1950 (Edlund et al 2009). Fossil pigment concentrations showed an increase in the 1960s, and diatom-inferred total phosphorus doubled from 1910 to 1990. As was concluded in the Triplett et al. (2009) study, land use changes in the late 1800s and early 1900s had little impact on nutrient mass transport and water quality; however, the mid 1900s showed many eutrophication signals that indicated water quality was degraded.

In addition to these paleolimnological studies, recent water quality measurements indicate LSC is eutrophic. Based on the Carlson Trophic State Index, LSC is classified as eutrophic (MPCA, 2001). Transparency was measured with a Secchi disk, and the mean depth in LSC from 1997 to 2006 was 1.2 m; mean total phosphorus (1997 to 2006) was 45 ppb; and mean chlorophyll *a* (1997 to 2006) was 16.7 ppb (MPCA, 2001). Troelstrup et al. (1993) reported that LSC experiences hypolimnetic oxygen depletion via decomposition, and anoxia occurs at the sediment-water interface.

In 2007, the Great Lakes Inventory and Monitoring Network initiated a water quality monitoring program in the St. Croix National Scenic Riverway, with 3 stations on

LSC (VanderMeulen and Elias, 2008). High nitrate+nitrite-nitrogen levels were reported in the Willow and Kinnickinnic Rivers resulting from fertilizer application for agricultural development and increasing urbanization. The ratio of nitrogen to phosphorus helps to determine which of these 2 nutrients is impacting primary productivity the most. Phosphorus is limited when TN:TP values are high, and nitrogen is limited when TN:TP values are low. High TN:TP values were reported in LSC, with most greater than 15. Even though these results indicate phosphorus is the limiting nutrient in LSC, algal blooms of nitrogen fixers have been reported in the lake since the 1960s (Brook, 1966 as cited in Troelstrup 1993), and nitrogen-fixing cyanobacteria flourish in low, not high, TN:TP conditions. During the summer, LSC became stratified in relation to dissolved oxygen and temperature. Stratification prevents oxygen from mixing throughout the lake, and near-bottom dissolved oxygen levels were 1.18 and 0.07 mg/L at 2 LSC stations. VanderMeulen and Elias (2008) concluded that water quality was degraded in downstream locations in LSC.

Lafrancois et al. (2009) analyzed long-term water monitoring data from 1976 to 2004 at the inlet and outlet of LSC and compared these values to diatom inferred total phosphorus that was reconstructed in Edlund et al. (2009) study. At the inlet of LSC, a decrease in ammonium and total phosphorus concentrations was seen from 1976 to 2004, probably due to strict regulations of point source pollution sources. Nitrate concentrations increased at the inlet during this time period due to point source pollution changes and nonpoint source pollution increases. Modern day total phosphorus concentrations are significantly higher than the levels seen before 1950. Today, agriculture and urbanization contribute to point and nonpoint source pollution. Trends in

water quality variables were similar at the inlet and outlet of LSC. Reconstructed phosphorus levels were similar to measured phosphorus in the 1980s and 1990s.

In a study conducted by Lafrancois et al. (2006), dissolved oxygen and temperatures were taken at 7 to 9 sample sites in August 2005 and August-September 2006 in LSC. Dissolved oxygen levels in the deepest portions of the lake often were below 4 mg/L. Lafrancois et al. (2006) concluded that oxygen loss does occur in LSC, resulting in hypoxic conditions, and these conditions are widespread throughout the lake.

1.6 Chironomid Ecology

Non-biting midges (Insecta: Diptera: Chironomidae), or chironomids, are a family of macroinvertebrates that are diverse and abundant in freshwater ecosystems, often greatly outnumbering all other invertebrate families (Ferrington et al., 2008). Ten subfamilies comprise the Chironomidae, with 8000 to 20,000 species existing from the Himalayas to equatorial east Africa to Antarctica (Porinchu and MacDonald, 2003), but only 5,000 species have been described by entomologists (Brooks et al., 2007). Larvae occupy habitats ranging from lentic (standing water) to lotic (moving water) environments, from hot springs and glacial melt trickles to brackish water and marine environments, and from phytotelmata (minute aquatic environments held by plants) to hydrated soil (Pinder, 1995b). They form an essential link in freshwater ecosystems; their niche is both prey and consumer (Ferrington et al., 2008).

Chironomids are holometabolous, undergoing complete metamorphosis from the aquatic egg, larvae, and pupae stages to two-winged adults emerging from the water (Brooks et al., 2007) (Figure 4). The chironomid life cycle is complete in one year in

temperate regions or up to three years in higher latitudes. Mating occurs in aerial swarms, and adult females oviposit eggs on the water surface along with a gelatinous matrix for protection. Often, the egg matrix will become attached to leaf litter or macrophytes (Pornichu and MacDonald, 2003). Water temperature is the most influential environmental variable that determines the rate of egg development (Pinder, 1995a), with interspecific and intraspecific competition, pH, DO, salinity, and photoperiod also being important variables (Pornichu and MacDonald, 2003). The number of eggs in a mass varies from 20 to 2,000 depending on the size of the species (Pinder, 1995a).

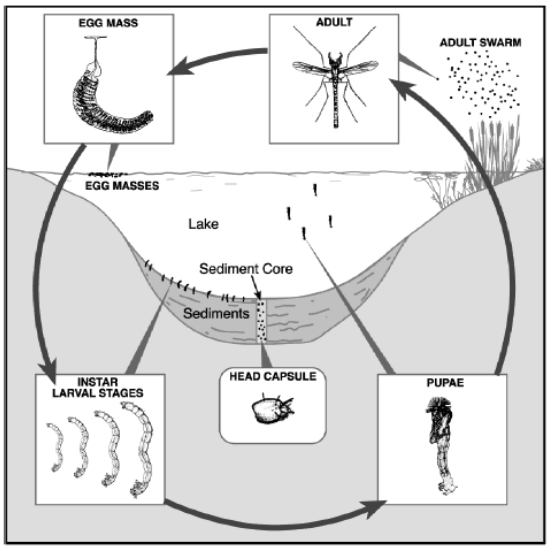

Figure 4. Complete metamorphosis of the Chironomidae (from Porinchu and MacDonald, 2003).

Larvae emerge from eggs within a few days to a month (Pornichu and MacDonald, 2003) and consume coarse detrital matter (shredders), medium detrital particles from sediments (gatherers and scrapers), fine detrital particles in suspension (filter-feeders), transport (gatherers), or deposited (scrapers), algae (scrapers, gatherers,

filter-feeders), vascular plants (miners), fungal spores and hyphen (gatherers), animals (predators or parasites) (Ferrington et al., 2008). Four larval stages, or instars, are present in all subfamilies, and ecdysis, or molting of the exoskeleton, occurs between each instar. As larvae, some taxa construct a case and filter feed while others are free-living (Ferrington et al., 2008). Most larvae range in length from 2 to 30 mm, and may be red, green, or pale yellow in color (Brooks et al., 2007). The thorax is composed of 3 segments, and the abdomen, 9 segments. One pair of ventral prolegs, or false legs, protrudes from the prothorax as well as the terminal end (Bouchard, 2004). Hooks are present at the terminus of the prolegs. The anal segment has paired procerci with setae (Cranston, 1995b). The sclerotized (hardened integument or skin) head capsule is non-retracting with mandibles moving on a horizontal plane.

Head capsule morphology is diverse among Chironomidae subfamilies. Chironominae (Figure 5) and Orthocladiinae (Figure 6) head capsules possess ventromental plates and a mentum. The mentum is a double-walled plate with teeth, and the ventromental plates extend posterolaterally or laterally from the mentum and are well developed in the Chironominae and poorly developed in the Orthocladiinae. The number of mandibular teeth is also diagnostic. Antennae of the Chironominae have 5 to 8 segments, the mentum is well developed, and ventromental plates of almost all taxa are striated. The Orthocladiinae are a morphologically diverse subfamily and span a wide variety of environments (Epler, 2001). Antennae are well developed with 3 to 7 segments, and the mentum and premandibles are also well developed.

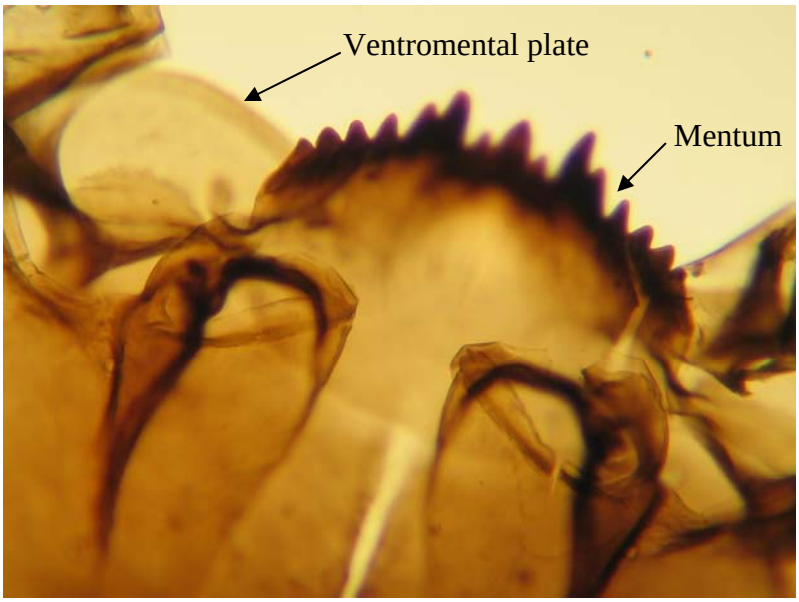

Figure 5. Subfossil Chironominae (*Chironomus*) head capsule showing mentum and ventromental plate.

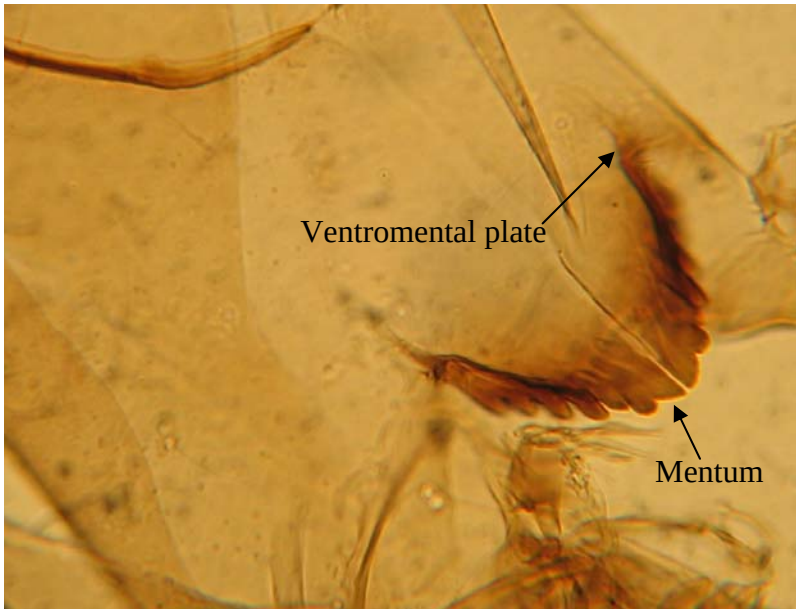

Figure 6. Subfossil Orthocladiinae head capsule (*Cricotopus/Orthocladius*) showing mentum and ventromental plates.

Subfossil specimens of the Tanypodinae are sometimes difficult to identify because diagnostic features, such as antennae, claws, and maxillary palpi (sensory structures located on the maxilla, or mouthpart) are often missing (Rieradevall and Brooks, 2001). Taxa of this subfamily always possess a ligula, or a toothed plate that is a component of the feeding apparatus (Figure 7). Antennae are retractile, and premandibles are not present. Cephalic setation patterns on the head capsule are diagnostic, and ventral (lower or bottom side) and dorsal (upper or top side) sockets and pores are almost always visible on head capsules (Rieradevall and Brooks, 2001). Cephalic setae are tactile mechano-receptors and pores are sensory pits that respond to mechanical forces. Larvae are predacious with an apical tooth on the mandible that aids in capturing prey (Epler, 2001).

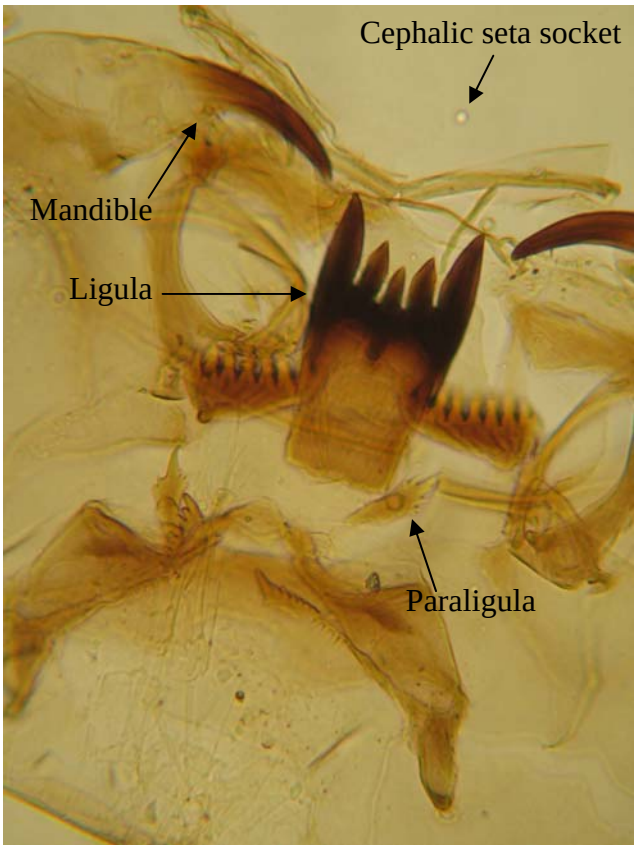

Figure 7. Subfossil Tanypodinae (*Procladius*) head capsule showing ligula, mandible, cephalic seta socket, and paraligula.

During the fourth instar larval stage, the thorax that houses pupal and adult features becomes swollen and the exuvia (larval skin) is shed. Pupae, as with larvae, may be sedentary and housed in a case or free living (Oliver, 1971). This life stage lasts a few hours to a few days. Eclosion occurs when the pupae swim to the surface of the water and the adult emerges.

Adult life stages are short; they mate in swarms and die within 2 weeks after emergence. If they feed at all, it is not on blood, but on nectar, pollen, or honeydew (Brooks et al, 2007). Adults resemble mosquitoes, but wings do not have scales, and the proboscis is short (Pornichu and MacDonald, 2003).

1.7 Chironomids as Bioindicators of Water Quality

A biological indicator organism is one that, through its absence or presence in a community, indicates specific environmental conditions such as pH, oxygen, or temperature. Chironomid larvae are useful in aquatic biological monitoring because they are ubiquitous, abundant, and distributed world-wide in freshwater ecosystems (Porinchu and MacDonald 2003). Unlike chemical or physical water quality parameters that show short term changes in streams or lakes, these organisms are indicative of changes that occur over time scales as great as centennial to millennial. Because chironomids are sensitive to changes in their aquatic environment, species compositions indicate specific gradients of water quality, such as oxygen levels, over time (Pornichu and MacDonald, 2003). Short life cycles allow chironomids to respond quickly to perturbations, and species distributions are in near equilibrium with their aquatic environment (Pornichu et al., 2003).

Larvae are impacted by aquatic conditions, and respond to fluctuating variables such as temperature and oxygen in different ways (Brodersen and Quinlan, 2006). Larval growth and survival is most dependent on temperature, food availability, and oxygen concentration (Brodersen and Quinlan, 2006) (Figure 8). Habitat and food web links affect chironomids on smaller temporal and spatial scales.

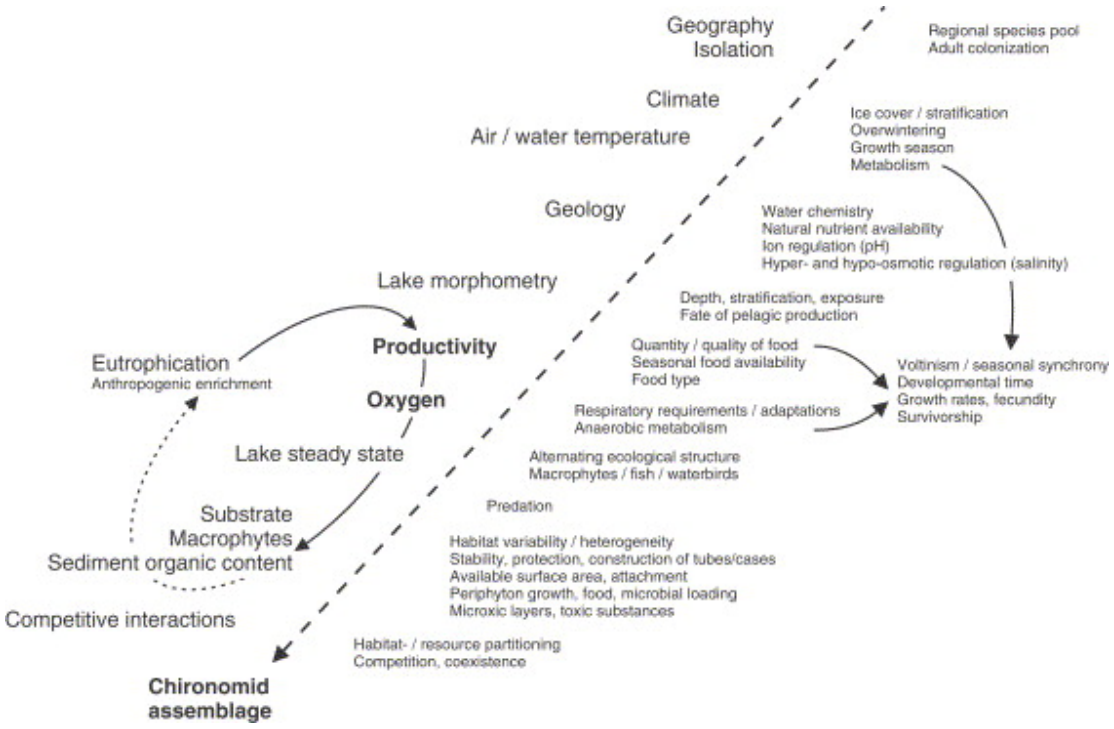

Figure 8. Interacting environmental factors that effect living chironomid communities and subfossil assemblages (from Brodersen & Quinlan, 2006).

Chironomids are often used as biological indicators of lake trophic status because chironomid abundance is often regulated by concentrations of dissolved oxygen (Kajak, 1997). Brodersen et al. (2004) discovered that oxy-regulatory capacity adaptations allow some chironomid taxa, such as *Chironomus*, *Dicrotendipes,* and *Procladius* to thrive in low oxygen conditions, while other taxa, such as *Micropsectra* and *Heterotrissocladius*, cannot tolerate low oxygen conditions. Concentrations of hemoglobin allow some chironomid taxa, such as *Chironomus,* to survive periods of hypoxia and anoxia. In addition, some taxa such as *Chironomus* have the ability to slow their metabolic rates, or

transition to anaerobic metabolism (Hamburger et al., 1994). Size is another factor that allows chironomids to tolerate low amounts of dissolved oxygen. For taxa such as *Chironomus*, *Stictochironomus*, and *Procladius*, large body size allows for more adequate ventilation of larval tubes (Int Panis et al, 1996).

Chironomid community distributions are affected not only by exposure to low amounts of oxygen, but by duration of exposure (Brodersen and Quinlan, 2006). For example, hypoxia-intolerant (oligotrophic) taxa are impacted by decreasing amounts of oxygen, while mesotrophic and eutrophic taxa may be able to withstand months of hypoxia (Hamburger et al, 1995).

In the past, a simplistic relationship was thought to exist between lake trophic status, hypolimnetic oxygen, and midge ecology, with high oxygen and the bioindicator *Micropsectra* equating to oligotrophic lakes, while low oxygen and the bioindicator *Chironomus* equated to eutrophic lakes (Brodersen and Quinlan, 2006). These over-simplistic relationships are not fully accurate in the real world because the rate of oxygen depletion and the concentration of oxygen in the hypolimnion are caused by interacting biological, physical, and chemical conditions, such as lake volume, depth, fetch (the available distance for air to flow over a lake), the annual retention of phosphorus, mean summer water temperature, ice cover duration and thermal stratification strength and timing (Papst et al., 1980, Cornett, 1989, Ohlendorf et al., 2000).

In 1921, German freshwater biologist Dr. August Thienemann was the first scientist to use chironomids in limnology (Smol, 2002). He developed a system to classify lake trophic status based on the dominant chironomid species found in the lake (Smol, 2002). He discovered that oligotrophic lakes were dominated by *Tanytarsus*, a

species that thrives in clear, cold, nutrient poor water, while eutrophic lakes were dominated by *Chironomus*, a species that thrives in nutrient rich water. In addition, he also determined that summer hypolimnetic oxygen concentration affected deep-water profundal species composition (Crisman, 1988).

Thienemann's classification was modified by Lars Brundin in 1949, who believed that trophic state was associated with specific profundal chironomid species and that profundal taxa were greatly influenced by the amount of oxygen present in the hypolimnion (Brodersen and Quinlan, 2006). Brundin's classification system associated ultra-oligotrophic lakes (very low nutrients) with *Heterotrissocladius subpilosus*, oligotrophic lakes with *Tanytarsus lugens*, mesotrophic lakes with *Stictochironomus rosenschöldi* and *Sergentia coracina*, eutrophic lakes with *Chironomus anthracinus* and *C. plumosus*, and dystrophic lakes (variable nutrients, humic compounds result in brown water color) with *C. tenuistylus*.

Data sets spanning centennial time scales are not in existence for lakes, making it difficult to determine if low oxygen conditions are caused by natural variables or are the result of anthropogenic impacts to the landscape (Quinlan and Smol, 2001). Proxy data retrieved from lake sediment cores, such as chironomid head capsules, are useful in estimating paleoenvironmental conditions (Quinlan and Smol, 2001). Chironomids are used as paleoenvironmental indicators of oxygen conditions and lake trophic state for a number of reasons. During the larval stages, or instars, chironomids undergo ecdysis and molt their sclerotized chitinous head capsules, which then sink to the lake bottoms where they are preserved in the sediment (Smol, 2002). Head capsules retain diagnostic features that are essential for taxonomic identification. Secondly, some chironomid taxa

are stenotopic, that is, they are capable of tolerating only a narrow range of aquatic environmental conditions (Brooks et al., 2007). Third, short life cycles allow chironomids to respond quickly to aquatic perturbations, and distributions are in near equilibrium with the aquatic environment (Pornichu and MacDonald, 2003). Because species vary in their tolerance to environmental factors, changes in species assemblages over time can be used to infer historical environmental change. Chironomidae larvae are relatively stationary in their environment, and this *in situ* metamorphic stage ensures assemblages respond to local environmental changes such as point source pollution.

Meriläinen et al. (2000) conducted a paleolimnological study on the progression of trophic status in Lake Lappajärvi, Finland and how profundal chironomid taxa altered as a result of eutrophication and decreased hypolimnetic oxygen levels. *Heterotrissocladius subpilosus*, *Paracladopelma nigritula*, and *Micropsectra* spp., all bioindicators of oligotrophic conditions, typified the pre-industrial era before 1935. *Heterotrissocladius subpilosus* became extinct with increased nutrient loading from 1935 – 1960, and *Micropsectra*, *Paracladopelma*, and *Sergentia* decreased in numbers. With a period of increased erosion and heightened nutrient loading from 1960 to 1970 came significant increases in *Chironomus* type *anthracinus*, a bioindicator of moderate eutrophy. The succession to meso-eutrophic conditions saw an increase in *Chironomus* type *plumosus*, another bioindicator of eutrophication.

Warwick (1980) used chironomids as biological indicators of anthropogenic eutrophication of the Bay of Quinte, Ontario, Canada, and expected to find chironomid assemblages characteristic of eutrophic lakes. Instead, the data showed that littoral (shallow water) chironomids declined while cold-stenothermous, profundal (deep water)

oligotrophic-tolerant chironomids dominated the sediment samples. Deforestation eroded nutrient poor soils into the Bay of Quinte, resulting in low primary productivity that explained the oligotrophic-tolerant chironomids found in the sediments in place of the expected eutrophic-tolerant chironomids. As eroded soil entered the bay, turbidity increased, displacing the thermocline which resulted in the decline of littoral chironomids. Phytophilous, or plant feeding chironomid populations decreased as turbidity increased. Water clarity decreased due to increased turbidity, and deep water macrophyte populations declined due to a lack of sunlight entering the water. In the late 1900s, eutrophic-tolerant chironomids replaced oligotrophic chironomids, as Warwick expected (1980).

1.8 Quantitative Reconstructions

Lake monitoring data, such as measured dissolved oxygen, covers short time periods on the scale of decades, while long term data sets cover longer time periods on the scale of centuries. In order to understand the changes that occur in the present, past conditions must be known for comparison, and long term data sets are useful in showing what conditions were like before the impact of European settlement. Quantitative reconstructions use paleo biological data, such as chironomid head capsules recovered from sediment cores, to reconstruct an environmental variable, such as oxygen conditions, over long periods of time (Quinlan and Smol, 2001). Inferences of fluctuating levels of hypolimnetic oxygen over centennial time scales are useful components of eutrophication studies in freshwater lakes (Quinlan et al., 1998). Fossil

Chironomidae proxy data sets in lake sediments are useful in reconstructing the hypolimnetic oxygen conditions of the past (Quinlan et al., 1998).

A transfer function is a mathematical model that relates modern day species distributions to biological, physical, and chemical environmental gradients. That model is then used to interpret fossil assemblages. Multivariate statistical techniques are used to obtain transfer function equations, and these techniques express how valuable an environmental variable is in terms of the faunal composition data, quantifying the relationship between species distribution and environmental variables. Several steps are involved in developing a transfer function. Environmental data and surficial sediment samples are collected from a suite of lakes along an environmental gradient. Taxa are sorted and identified from each sample site. Ordination methods (the arrangement of species along environmental gradients) are used to determine which environmental variables are most influential on species distributions. The transfer model can then be developed using statistical techniques such as weighted-averaging regression. Error estimation techniques, such as jackknifing, bootstrapping, and cross validation, are methods used to evaluate the robustness of the transfer function. Finally, the transfer function is run using down-core chirionomid assemblage data for paleo-reconstructions (Pornichu and MacDonald 2003).

A number of assumptions are made when running a transfer function (Birks, 1998). First, taxa identified in the training set are related to the habitat they were sampled from. Second, the reconstructed environmental variable is either significant in the ecological system or linearly linked to an environmental variable that is significant in the ecological system. Third, the response of the training set taxa and the fossil taxa to

aquatic perturbations is the same. Fourth, regression and calibration techniques sufficiently model taxa responses to the environmental variable. Fifth, all other environmental variables are insignificant to the organisms of interest.

CHAPTER 2

METHODS

2.1 Coring

On 7 June 2006, a Livingston corer outfitted with a 2.5 m polycarbonate tube (6.5 cm inner diameter) and operated with rigid drive rods from an anchored boat (Wright 1991), was used to extract four sediment cores from LSC at two different sample sites (Figure 2). Two cores were taken at each sample site to ensure enough sediment would be available for sediment processing and for chironomid head capsules. Core 1B was extracted from sub-basin 1 upstream of Prescott, WI at 44°45'27.6"N and 92 °48'25.6"E from a water depth of 12.11 m, and was 1.57 m long. Core 6B was extracted from sub-basin 3 near Lakeland, MN at 44°56'50.7"N and 92 °45'20.6"E from a water depth of 14.98 m and was 1.96 m long. Sub-basins 1 and 3 were chosen as coring sites due to spatial variability that occur in upstream (core 6B) and downstream (core 1B) locations. In addition, these coring sites were chosen to make comparisons to other studies that were previously conducted in the same sub-basins (Troelstrup et al., 1993; Edlund et al., 2009; Lafrancois et al., 2009; Triplett et al., 2009) Table 1 details the sub-basins in LSC. Cores were kept upright in the boat and taken to shore.

Cores were secured upright to posts in the boat, and once the boat reached shore, cores were carried upright off the boat, taking care not to disturb the uppermost sections of the cores that had high water content. Disturbance leads to unreliable chironomid subfossil data, decreasing the accuracy of historical reconstructions (Tomkins et al., 2007). The uppermost sections of each core were sectioned immediately in the field, and the remainders of the cores were sectioned after magnetic susceptibility analysis. Down-

core smearing was removed from core exteriors by using a spatula to remove sediment. Before magnetic susceptibility was analyzed, the upper 22 sections of core 1B-1 and 6B-2 were sectioned on shore at a high resolution of 16 cm for the uppermost 16 cm and 2 cm for 18 – 22 cm.

After magnetic susceptibility analysis was complete, cores were sectioned at the St. Croix Watershed Research Station at 2 cm intervals for 16 to 120 cm, and at 5 cm intervals for the bottommost 120 to 150 cm. Before magnetic susceptibility was analyzed, the upper 21 sections of core 6B-1 and 6B-2 were sectioned on shore at a high resolution of 1 cm intervals. After magnetic susceptibility analysis was complete, the cores were sectioned at the St. Croix Watershed Research Station at 2 cm intervals for 22 to 140 cm and 5 cm intervals for 140 to 150 cm. Samples were placed in screw-top polypropylene jars and stored at 4^{o}C. The upper sections of the cores were sectioned at a high resolution in order to show changes taking place at the time of European settlement through the present.

2.2 Dating Sediment Cores

Dating young sediment cores (100-150 years) with lead-210 (^{210}Pb) is a popular high-resolution method that involves uranium decay (Cohen, 2003). Radioactive decay is the process by which an unstable atom, or radioactive nuclide, emits radiation and changes into a different species as the ratio of protons to neutrons alters, until no radioactivity remains and the atom becomes stable. During decay, alpha or beta particles, or gamma rays are emitted from the nucleus of an atom. The radioactive element uranium (U) decays to a sequence of nuclides having known half-lives (Rapp and Hill,

1998). Uranium-series dating utilizes this uranium decay sequence terminating in stable lead to estimate the age of a sediment core (Bradley, 1999).

Two intermediate isotopes (an atom of an element that is composed of the same number of protons, but a different number of neutrons) in the ^{238}U decay series, Radium-226 (^{226}Ra) and Radon-222 (^{222}Rn), naturally occur in rock, soil, and water and are released from the earth's crust into the atmosphere (Bradley, 1999). Radium-226 decays at a half life of 1622 years to the gas ^{222}Rn that is found in the atmosphere (Cohen 2003). Radon-222 rapidly decays to ^{210}Pb, and is re-deposited as unsupported ^{210}Pb that either precipitates from the atmosphere or returns to the earth as dry fallout and collects in lacustrine sediments (Preiss et al., 1996). Manganese and iron oxides, as well as organic matter, carry re-deposited ^{210}Pb to lakes where sedimentation and decay to bismuth-214 (^{214}Bi) take place (Cohen, 2003). Once the sedimented ^{210}Pb is buried, decay results in stable ^{206}Pb (Cohen, 2003). Not only is unsupported ^{210}Pb atmospherically deposited, but supported ^{210}Pb exists in all samples due to the decay of its parent isotopes *in situ* (Cohen, 2003). If it is assumed that ^{226}Ra is in equilibrium with supported ^{210}Pb, total supported ^{210}Pb can be inferred by measuring mean ^{226}Ra (Cohen, 2003). Unsupported ^{210}Pb is calculated by subtracting total supported ^{210}Pb from total ^{210}Pb (Cohen, 2003).

The sediment cores for this study were recovered from the same locations as cores recovered in a 1999 study by Triplett et al. (2009) for comparative purposes. Triplett et al. (2009) determined chronology of the cores by analyzing the concentration of ^{210}Pb in 18-25 samples from each core through its grand-daughter product, polonium-210 (^{210}Po). The short half life of ^{210}Po (138 days) allows for the assumption that it is in equilibrium with ^{210}Pb (Cohen, 2003). Polonium-209 was added as an internal yield tracer to freeze-

dried sediment, carbonate was removed from samples by a treatment of concentrated HCl, and finally, isotopes were distilled at 550^{o}C. Polonium isotopes were plated on silver planchettes from a solution of 0.5 M hydrochloric (HCL). An Ortec alpha spectrometry system measured activity for 0.8-3 x 10^5 s. Unsupported ^{210}Pb was quantified by subtracting lower core mean activity levels (supported ^{210}Pb) from activity in the upper samples. Asymptotic activity was used to determine supported ^{210}Pb in the lower core samples, and supported ^{210}Pb was subtracted from measured total activity in the upper core samples to determine unsupported ^{210}Pb activity.

The constant rate of supply (CSR) model was used to calculate sedimentation rates and core dates (Appleby and Oldfield, 1978). Cores 1B and 6B displayed monotonic downcore declines in ^{210}Pb activity and surface activities were low. Change in slope in ^{210}PB activity profiles resulted from fluctuations in sediment flux, justifying the use of the CSR model that allows for such sediment accumulation fluctuations while presuming a constant ^{210}PB flux.

The same core increments that were ^{210}Pb-dated were also analyzed for Cesium-137 in order to check the accuracy of the chronology (Edlund et al., 2009). From 1963-1964, maximum atmospheric deposition of ^{137}Cs occurred during the period of nuclear bomb testing. Extremely well defined peaks of ^{137}CS occurred in the sediment chronology, allowing the dates of 1963-1964 to be precisely placed at the depths where peaks occurred.

In addition, terrestrial organic matter samples from the lower sections of cores 8C, 5B, 3A and 1B were radiocarbon dated. Depths were chosen based on fluctuations in magnetic susceptibility in order to correlate ^{14}C dates to other cores (Edlund et al., 2009).

Lead-210 results were cross-checked with ^{137}Cs peaks, and results indicated that ^{210}PB dates displayed a reasonable correlation to ^{137}Cs date, supporting the reliability of ^{210}Pb dates from the mid 1900s to the present (Edlund et al., 2009). CALIB returned 1 possible calendar date for 1B samples of 1413 AD. CALIB returned 3 possible calendar dates for 5B samples of 1330, 1350, and 1390 AD, but no date overlapped with ^{210}Pb dates and the median (1350) was used. A single pre-European settlement sediment accumulation rate was calculated using calibrated ^{14}C dates for cores 8C, 5B, 3A and 1B that represented the block of time from ^{14}C sample depth to European settlement around 1850. The calibrated 14C date for 5B was applied to transect 6, where core 6B was extracted. In core 1B, the ratio of post- to pre-settlement sediment accumulation was determined. Post-settlement was designated as 1838 (marked by a rise in magnetic susceptibility) to modern times. Pre-settlement was designated as 1838 to the date 1413 that was ^{14}C calibrated.

Figures 9a and 9b display lead-210 chronologies with error bars for both cores using the Pb-210 dates from Edlund et al. (2009) study. Oldest dates showed the greatest amount of error, while the youngest dates displayed little error. Estimated uncertainty for the age models was relatively low, verifying the use of the Pb-210 dates from Edlund et al. (2009) in this study.

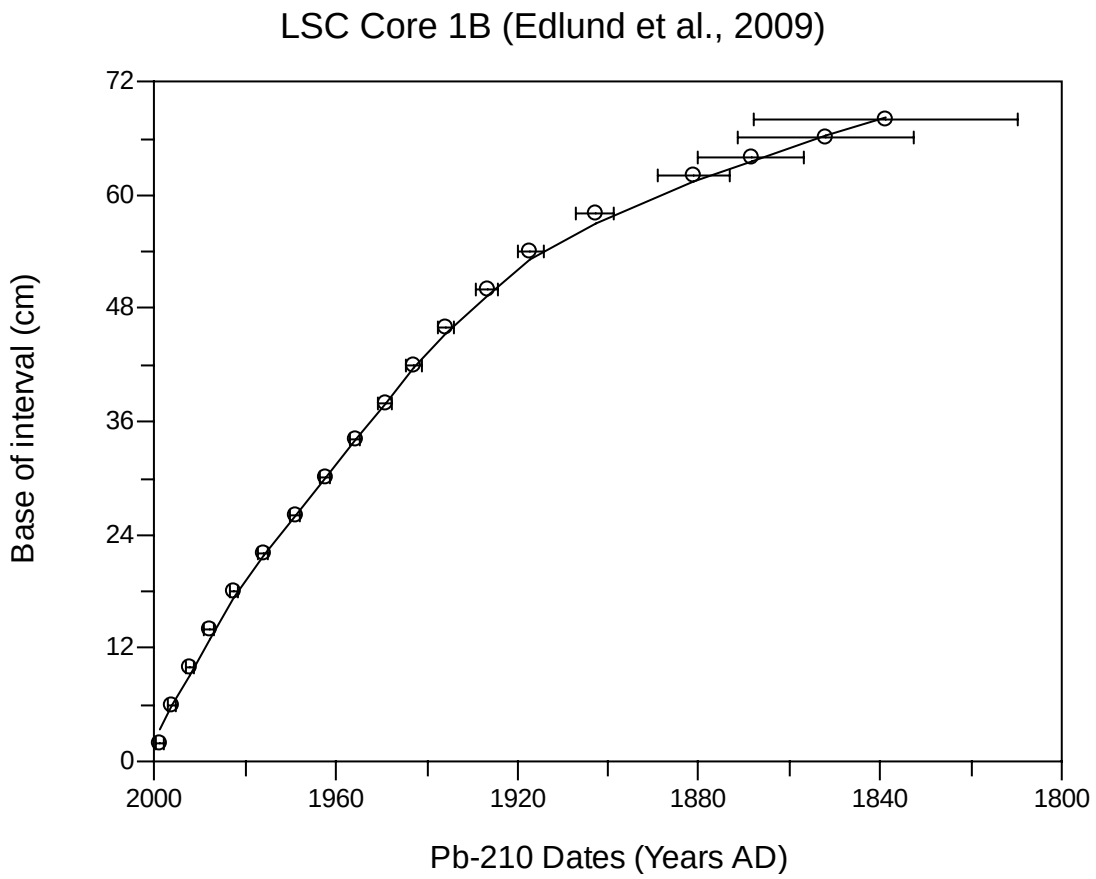

Figure 9a. Core 1B calculated age (Years AD) versus base of core interval (cm) based on the constant rate of supply model. (Data are from Edlund et al., 2009).

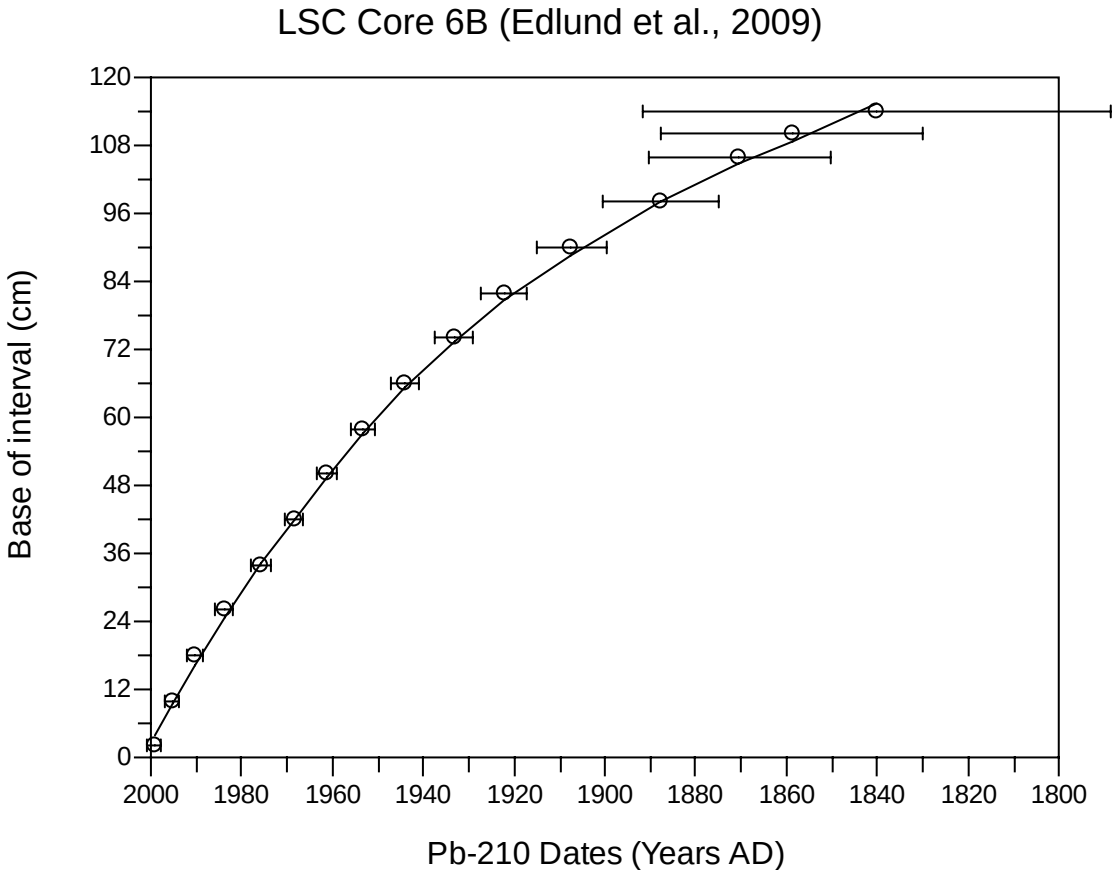

Figure 9b. Core 6B calculated age (Years AD) versus base of core interval (cm) based on the constant rate of supply model. (Data are from Edlund et al., 2009).

2.3 Magnetic Susceptibility

Measuring the magnetic susceptibility of a sediment core is a non-destructive method that applies a magnetic field that magnetizes the iron-bearing minerals in the sample, and the ease of magnetization is recorded (Thompson et al., 1975). A lake's sediment record may be impacted by variables such as climate change, soil erosion, and anthropogenic land use changes (Guerrero et al., 2000), which can all lead to fluctuations

in the magnetic properties of a sediment core. These fluctuations can be linked to times of human induced environmental change as indicated by paleo-proxy data (Guerrero et al., 2000).

Magnetic susceptibility analysis for cores 1B and 6B took place at the Limnological Research Center Core Facility at the University of Minnesota using a Geotek Multisensor Core Logger (MSCL) with an automated track and a Bartington MS2 core logging sensor. Cores were brought to room temperature and sectioned at 1.6 m in the polycarbonate tubes to fit the automated track. Measurements were logged every centimeter.

The major and minor tie points in the magnetic susceptibility profiles of cores 1B and 6B from Edlund et al. (2009) and 1B-1 and 6B-1 from this study were used to correlate cores for dating (Figures 10a and 10b). It is important to note that the sensitivity of the MSCL has changed from when the Edlund et al (2009) cores were logged. In addition, the length of the cores was not the same. Core 1B (Edlund et al., 2009) was longer than core 1B-1, and core 6B (Edlund et al., 2009) was shorter than core 6B-1. Magnetic susceptibility of the undated cores of this study was plotted against depth and age (Years AD) of the dated cores from Edlund et al., 2000 (Figures 10a and 10b). Patterns in the magnetic susceptibility profiles were used to determine major and minor tie points of the dated and undated cores. Ages from the dated Pb^{210} cores were assigned to the tie points of the undated cores, and linear interpolation was assumed between tie points. Even though core lengths were different, figures 9a and 9b were produced using the same number of core sections for each core.

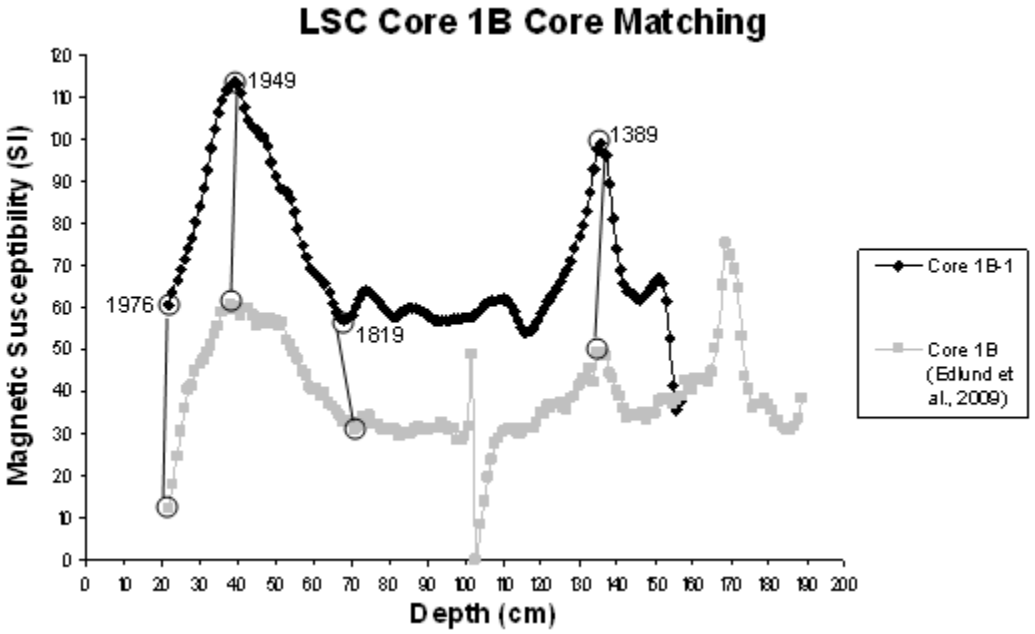

Figure 10a. Core 1B core matching. Major and minor tie points are circled. The Pb-210 dates of core 1B in years AD (data are from Edlund et al., 2009) were applied to the tie points in core 1B-1.

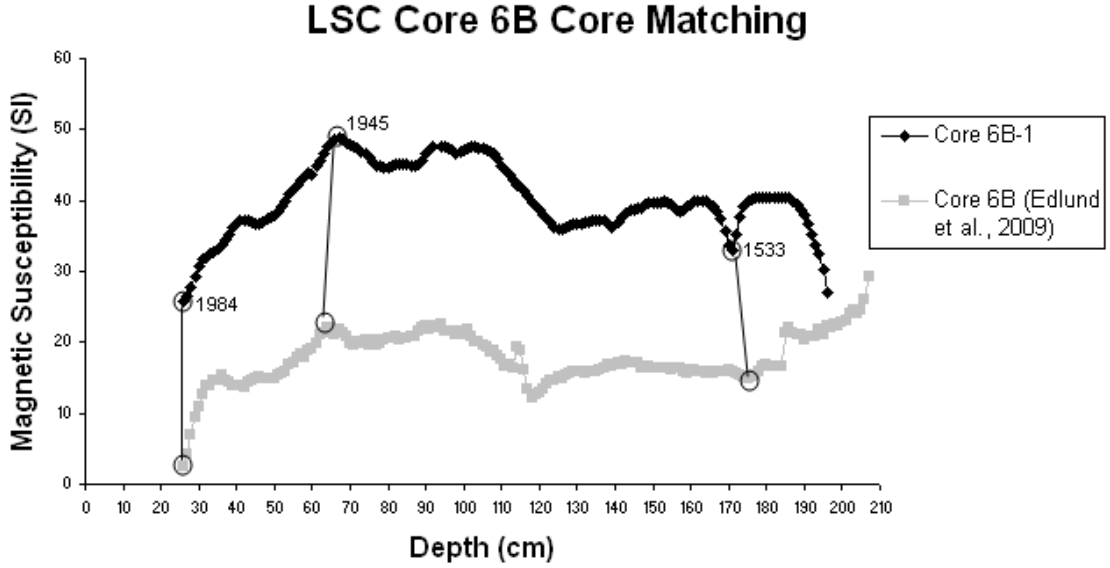

Figure 10b. Core 6B core matching. Major and minor tie points are circled. The Pb-210 dates of core 6B in years AD (data are from Edlund et al., 2009) were applied to the tie points in core 6B-1.

2.4 Sediment Analysis for Chironomid Head Capsules

Aliquots of 6.99 – 35.08 g of wet sediment were processed with the addition of approximately 100 ml of 10 % hydrochloric acid (HCl) and allowed to digest overnight to remove carbonates. The amount of sediment that was processed was determined by the total number of head capsules extracted from each core section, as a minimum of 40 head capsules per section was necessary for statistical accuracy (Quinlan et al., 1998). Sediment was rinsed with deionized water through a 100 µm sieve, backwashed into a glass beaker, digested in approximately 100 ml of 5 % potassium hydroxide (KOH) for 20 minutes at 80°C in order to remove organic matter. Once cool, sediment was rinsed a final time through the 100 µm sieve with deionized water and backwashed into the glass beaker. The surfactant Brij® 35 was added to each sample in order for the head capsules to be picked more easily from the sample, and ethyl alcohol (EtOH) was added as a preservative to prevent fungal growth. Chironomid head capsules were picked from the sample using a dissecting microscope at 50x, a Bogorov counting tray (Gannon, 1971), and a wire loop. Head capsules were placed in a drop of deionized water on glass cover slips. Deionized water was allowed to evaporate, leaving head capsules on cover slips. Glass microscope slides were prepared by mounting cover slips to slides using Euparal®. Slides have been archived at the University of Massachusetts Geoscience Department Quaternary Laboratory.

Head capsules were identified to the lowest taxonomic level possible using a compound microscope at 400x and taxonomic keys of Ferrington et al. (2008), Brooks et al. (2007), Epler (2001), Oliver and Roussel (1983), Rieradevall and Brooks (2001), and

Walker (2007). If differentiation between 2 genera was not possible, they were documented together (*Corynoneura/Thienemanniella*). Chironominae early instar, tribe Tanytarsini, tribe Macropelopiini, tribe Pentaneurini, and Tanypodinae early instar groups were used to categorize head capsules that could not be identified to lower taxonomic levels.

2.5 Quantitative Reconstruction of average Volume-Weighted Hypolimnetic Oxygen

This study used a chironomid-based inference model for end-of-summer hypolimnetic oxygen developed by Dr. Roberto Quinlan, York University, Toronto, Ontario and Dr. John P. Smol, Queen's University, Ontario (Quinlan and Smol, 2001). Developing a transfer function specific to LSC was beyond the scope and budget of this study. The use of the transfer function developed by Quinlan and Smol for this study is justified by the fact that the geographic area in which data was collected showed similar land use changes to LSC. In the mid 1800s, the Muskoka-Haliburton area was completely deforested for the logging industry and land clearance for agriculture (Quinlan and Smol, 2001). After the decline of the logging industry in the early 1900s, the region was re-forested, and today is dominated by mixed deciduous-coniferous successional forest (Quinlan and Smol, 2001). The region is almost void of agriculture due to poor soils, however, recreational activity adds a great amount of stress to the watershed. Resorts, residential areas, and golf courses, as well as undeveloped areas, are prevalent in the region.

Surficial sediments of 86 lakes in the District Municipality of Muskoka and the County of Haliburton, south-central Ontario, Canada were analyzed for subfossil

chironomid head capsules. The lake sites spanned a variety of land uses such as undeveloped, residential, and resort and a range of environmental conditions from shallow to deep, ultra-oligotrophic to eutrophic, and anoxic to oxic. Sediment cores were extracted from the deepest locations of each lake and ^{210}Pb dated. A YSI dissolved oxygen meter profiled temperature and oxygen measurements in each lake. End-of-summer, spring turnover, or year-round measurements were taken.

Hypolimnetic volume was calculated using the Lind method, which defines the top of the hypolimnion as the point of intersection of 1 tangential line drawn through the thermocline and a second tangential line drawn through the bottom portion of the temperature profile. End-of-summer volume-weighted hypolimnetic oxygen (VWHO) was determined using bathymetric maps and measured oxygen-temperature profiles (Quinlan and Smol, 2001). Measurements in the hypolimnion were taken at 1, 2, 5, or 10 m intervals. Temperature and oxygen values were linearly interpolated at 1m intervals between the hypolimnetic oxygen measurements. VWHO values were determined from the interpolated profiles. The percentage of hypolimnetic area and lake surface area that was underlain by anoxic ([DO] < 1 mg L^{-1}) and hypoxic ([DO] < 4 mg L^{-1}) water was determined from oxygen profiles. Bottom oxygen ([botO_2]) was defined as the DO concentration 1m above sediments at the deepest point.

Of the 86 lakes, 59 were used for numerical analysis with assemblages of 44 chironomid taxa. Taxa with less than 2 occurrences with a relative abundance less than 2% were eliminated. Principal components analysis (PCA) is an unconstrained ordination technique that assumes unimodal distributions (Holland, 2008), and in this study, was used to identify the sample sites to be excluded from ordination analysis. If,

on the first 2 axes of PCA of environmental data, sample sites fell outside the 95 % confidence limits of sample score means, then they were excluded (Quinlan and Smol, 2001). Detrended correspondence analysis (DCA) is an ordination technique that corrects for two problems that arise in PCA (Holland, 2008). DCA corrects for the arch effect that occurs on the second axis by detrending. DCA also corrects for the uneven spacing of samples along the first axis by rescaling. If, on the first 2 axes of DCA of screened species data, sample sites fell outside the 95 % confidence limits of sample score means, then they were excluded (Quinlan and Smol, 2001).

In order to determine if unimodal-or linear-based ordination techniques would be the most useful in conducting numerical analyses, DCA with detrending by segments, non-linear rescaling, and downweighting of rare taxa was used to determine the gradient length of chironomid composition in relation to environmental variables for the first 2 DCA axes. Direct gradient analysis such as redundancy analysis (RDA) and canonical correspondence analysis (CCA) determine which environmental variables best explain species abundance. The gradient length of chironomid composition in relation to environmental variables was determined using detrended canonical correspondence analysis (DCCA) with detrending by segments, non-linear rescaling, and downweighting of rare taxa determined. The environmental variables that best explained chironomid assemblages, as determined by forward selection in RDA (linear model) and CCA (unimodal model) were VWHO and [botO_2], with $P > 0.05$.

Oxygen inference models were developed from linear- and unimodal-based modeling because the first and second DCA axes both showed intermediate gradient lengths. Linear regression models used partial least squares (PLS), and unimodal

regression models used weighted averaging (WA) with inverse or classical deshrinking with or without tolerance downweighting. In addition, weighted averaging partial least squares regression (WA-PLS) was also utilized. WA is a reciprocal averaging technique that utilizes the modern training set to determine the optimum environmental variable of interest for all species. PLS is a linear technique that relates response variables (Y) to explanatory variables (X) in order to determine the linear combination of X variables that best model Y dependent variables. WA-PLS is a unimodal based technique that is best used for gradients of intermediate length. The predictive error (root mean square error of prediction) was used to assess the models. Jackknifed RMSEP values were compared among models to assess error. The inference models that were developed were statistically robust with moderate predictive power for measured VWHO and [botO_2].

Oxygen inference models were utilized in conjunction with the Clerk et al (2000) fossil chironomid data in order to reconstruct VWHO and [botO_2]. The Clerk et al (2000) fossil chironomid dataset from Peninsula Lake, Ontario Canada displayed chironomid fluctuations in response to European land use change including logging and deforestation. Quinlan and Smol (2001) concluded that oxygen inference models are accurate, based on the results seen in the reconstruction of VWHO and [botO_2] using the Clerk et al. (2000) chironomid dataset.

The transfer function developed by Quinlan and Smol (2001) was applied to the LSC fossil chironomid dataset from cores 1B and 6B in order to reconstruct average VWHO in LSC. Hypolimnetic oxygen is a crucial environmental variable that greatly impacts the survival and distribution of aquatic life (Little and Smol, 2001). In addition, hypolimnetic oxygen is an important indicator of eutrophication because as nutrients

promote the production of plant growth in the epilimnion, organic matter dies and sinks to the benthos of the lake, and decomposition decreases the amount of oxygen in the hypolimnion. Average VWHO was used in this study as opposed to other variables that impact chironomids because Quinlan and Smol (2001) concluded that the strongest explanatory variables for sub-fossil chironomid assemblage variation were average end-of-summer VWHO and bottom oxygen concentration.

The program C2 (Juggins, 2003) was used to run the transfer function. Oxygen inference models were developed for LSC using WA, WAPLS, and PLS. Leave-one-out cross validation method and square root species transformation were selected for all regressions. Performance statistics showed that estimated avgVWHO by weighted averaging (inverse deshrinking) was a robust inference model based on the moderate coefficient of determination ($r^2_{jack} = 0.49$) and a low predictive error (RMSEP = 2.32) (Table 2).

Table 2. Performance statistics for avgVWHO inference models. RMSE, RMSEP, Max. Bias units are mg O_2 L^{-1}.

Model	Component/ Deshrinking	r^2	r^2_{jack}	RMSE	RMSEP	Max. Bias
WA	Inverse	0.615	0.489	2.009	2.316	2.148
WA	Classical	0.615	0.506	2.562	2.710	1.873
WATOL	Inverse	0.645	0.475	1.930	2.351	2.049
WATOL	Classical	0.645	0.487	2.404	2.710	1.447
PLS	1	0.545	0.456	2.184	2.389	2.709
PLS	2	0.681	0.483	1.831	2.340	2.050
PLS	3	0.761	0.423	1.584	2.539	1.585
PLS	4	0.823	0.345	1.364	2.832	1.020
PLS	5	0.846	0.313	1.269	3.001	0.851
WAPLS	1	0.615	0.489	2.012	2.317	2.056
WAPLS	2	0.761	0.467	1.585	2.405	1.316
WAPLS	3	0.831	0.373	1.333	2.807	1.144
WAPLS	4	0.863	0.323	1.199	3.014	0.907
WAPLS	5	0.879	0.321	1.126	3.105	1.147

2.6 Chironomid Analysis

A number of equations and mathematical operations were used to analyze chironomid taxa data. Taxa richness is the number of different taxa that are present in each core section. This measure of biodiversity is widely used due to its ability to make comparisons among different taxonomic groups (Prendergast et al., 1993). Relative

abundance measures the amount of each taxon in a core section by dividing the abundance of each taxon by the total abundance of all taxa, and multiplying that result by 100. Head capsule concentration was determined by dividing the total number of head capsules in each core section by the grams of wet sediment that were used in processing the head capsules.

Diversity indices are used to quantify taxa rarity and commonness in a community. The Shannon-Wiener diversity index (H') measures the degree of uncertainty of predicting a taxon in a random sample. For example, a community that is dominated by a single taxon has a lower uncertainty of prediction than a community that is biologically diverse. As diversity increases, uncertainty increases. Figure 10 shows the equation where S is the total number of taxa in the community (richness) and p_i is the proportion of S made up of the *i*th taxa (Whittaker, 1975). If H' is 0, then there is only 1 taxon in the sample. If H' is the maximum calculated value, then all taxa are equally abundant. The meaning of median values is obscure, an obvious draw back in this index. A large H' value indicates a great amount of taxa diversity, while a small H' value indicates low biodiversity.

$$H' = -\sum_{i=1}^{S} pi(\ln(pi))$$

Figure 11. Shannon-Wiener diversity index equation.

The programs Tilia and TGView 2.0.2 (E. Grimm, Illinois State Museum, Research and Collections Center, Springfield, Illinois, USA) were used to generate chironomid profiles. Taxa with less than 3% abundance were eliminated. A cluster analysis was generated with CONISS (Grimm, 1987) using chironomid count data with a dendrogram scale of total sum of squares. Zones were determined based on major species shifts seen in the dendrogram clusters.

CHAPTER 3

RESULTS

3.1 Magnetic Susceptibility

Cores 1B and 6B displayed the same general trends of low, constant values in the deepest core sections, followed by an increase to peak susceptibility values, and a decrease in the upper core sections. Figure 12a shows magnetic susceptibility profiles for cores 1B-1 and 1B-2. Core 1B-1 values ranged from 35.5 SI – 113.6 SI. A decrease in values occurred from c. 1389 – 1515 AD, followed by fairly constant values from c. 1515 1819 AD with local minima of 56.7 SI (c. 1661 AD), 57.6 SI (c. 1737 AD), and 56.9 SI (c. 1819 AD). Susceptibility values increased from c. 1819 – 1949 AD, then decreased up core to modern times. In core 1B-2, values ranged from 29.8 SI (114 cm) – 116.3 SI (36 cm), and showed a similar profile to core 1B-1. An increase in susceptibility values occurred from 114 cm (29.8 SI) – 108 cm (60 SI), followed by fairly constant values from 107 - 36 cm. The same up core decrease in susceptibility values to modern times occurred in core 1B-2.

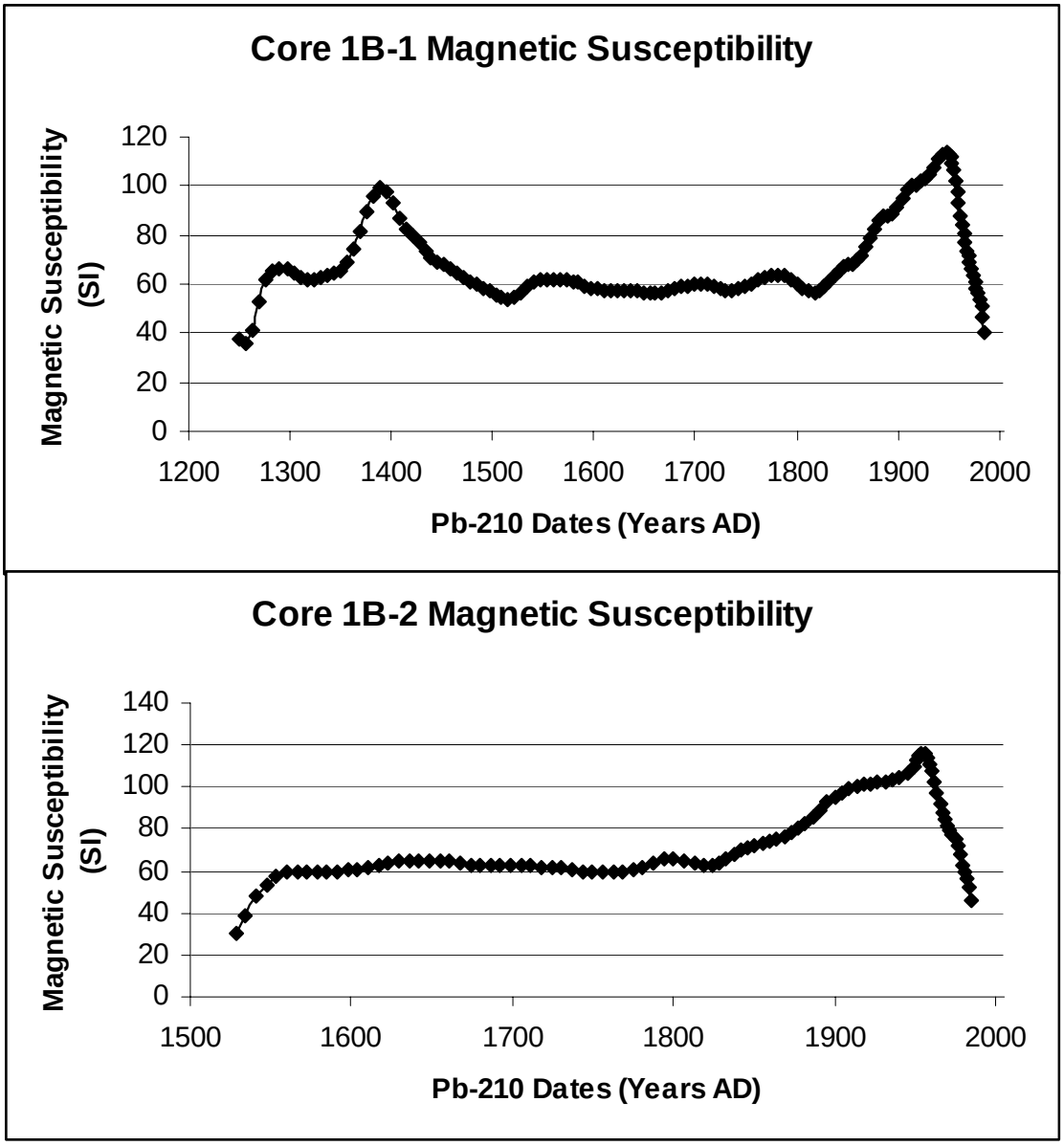

Figure 12a. Magnetic susceptibility profiles for cores 1B-1 and 1B-2.

Figure 12b shows magnetic susceptibility profiles for cores 6B-1 and 6B-2. In core 6B-1, values ranged from 25.9 SI to 48.8 SI. A significant decrease in values occurred between c. 1549 – 1561 AD, with a local minimum value of 32.9 SI, followed by an increase between c. 1561 – 1591 AD to 40.0 SI. Values were fairly uniform until an increase occurred between c. 1730 – 1813 AD to a peak of 47.6 SI, after which

susceptibility was fairly uniform with a local maximum of 47.7 SI at c. 1851 AD. An increase to the highest peak at c. 1945 AD began at 1904 AD, and susceptibility decreased to modern times, with a small increase occurring between c. 1967 – 1969 AD. Relatively uniform susceptibility values occurred up core to modern times. In core 6B-2, values ranged from 14.1 SI to 930.2 SI. Values increased from c. 1495 – 1523 AD, then decreased to 30.9 SI at c. 1553 AD. Values remained relatively uniform, until an increase to 52.7 SI occurred at c. 1745 AD. Very large spikes in magnetic susceptibility were seen from c. 1757 – 1775 AD. Values fell sharply and quickly to 46.3 SI at c. 1798 AD, and then increased rapidly to 921.9 SI at c. 1813 AD. An up core decrease in values began in c. 1945 AD. Magnetic susceptibility profiles of cores 1B-1 and 6B-1 showed excellent correlation to core 1B and 6B from the Edlund et al. (2009) study (Figures 9a and 9b).

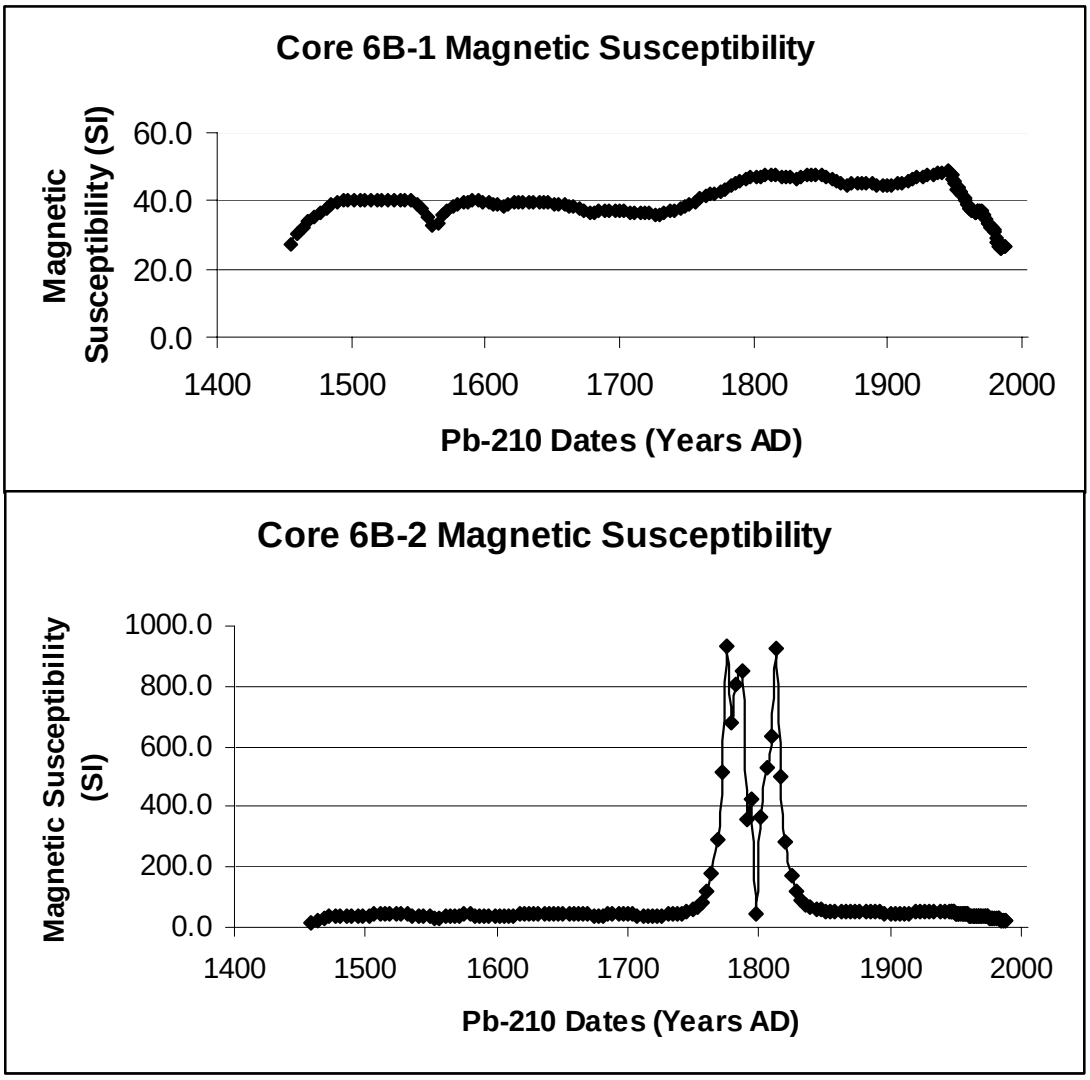

Figure 12b. Magnetic susceptibility profiles for cores 6B-1, and 6B-2.

3.2 Chironomids

A total of 49 chironomid taxa were identified in LSC core 1B and 44 taxa in core 6B (Table 3) excluding Chironominae early instar, Tribe Tanytarsini, Tribe Macropelopiini, Tribe Pentaneurini, and Tanypodinae Early Instar. In core 1B, the Tanypodinae were the most abundant, followed by the Chironomini, the Orthocladiinae, the Tanytarsini, and Pseudochironomini (Figure 13a). In core 6B, the Chironomini were

the most abundant, followed by the Tanytarsini, the Tanypodinae, the Orthocladiinae, and the Pseudochironomini (Figure 13b). In both cores, the Chironominae were the most abundant subfamily.

The majority of chironomid taxa identified in core 1B and 6B were littoral. However, profundal taxa, including *Chironomus*, *Cryptochironomus*, *Endochironomus*, *Glyptotendipes*, *Lauterborniella*, *Pagastiella*, *Orthocladius*, and *Procladius***,** were present in both cores (Ferrington et al., 2008). Total head capsules of all taxa were summed for all core sections, and in core 1B, *Procladius* showed the most occurrences (1397 head capsules), followed by *Chironomus* (604 head capsules), *Cricotopus/Orthocladius* (336 head capsules), and *Polypedilum* (183.5 head capsules). In core 6B, *Chironomus* showed the most occurrences (392 head capsules), followed by *Procladius* (356 head capsules), *Cladotanytarsus mancus* type (294.5 head capsules), *Polypedilu*m (227 head capsules), and *Cricotopus/Orthocladius* (176 head capsules). LSC cores showed a good representation of both littoral and profundal taxa, and even though more taxa were classified as littoral, profundal taxa showed the most occurrences in both cores.

Table 3. Chironomid taxa found in Lake St. Croix sediment cores with associated subfamily and tribe information.

Chironomid Taxa	Core
Subfamily Chironominae	
Tribe Chironomini	
Chironomus	1B, 6B
Cladopelma	6B
Cryptochironomus	1B, 6B
Cryptotendipes	1B, 6B
Dicrotendipes	1B, 6B
Endochironomus	1B, 6B
Endochironomus sp. A	1B
Glyptotendipes	1B, 6B
Harnischia	1B, 6B
Lauterborniella/Zavreliella	1B, 6B
Microchironomus	1B, 6B
Microtendipes	1B, 6B
Pagastiella	1B, 6B
Parachironomus	1B, 6B
Paracladopelma	6B
Paralauterborniella	1B, 6B
Paratendipes	1B, 6B
Phaenopsectra	1B
Polypedilum	1B, 6B
Subfamily Chironominae	
Tribe Chironomini	
Robackia	1B, 6B
Saetheria	1B
Sergentia	1B, 6B
Stenochironomus	6B
Stictochironomus	1B, 6B
Tribelos	1B, 6B
Xenochironomus	1B
Tribe Tanytarsini	
Cladotanytarsus group A	1B, 6B
Cladotanytarsus mancus type	1B, 6B
Micropsectra type	1B, 6B

Continued on page 54

Tribe Tanytarsini (continued)	
Paratanytarsus	1B, 6B
Stempellina	1B, 6B
Stempellinella/Zavrelia	1B, 6B
Tanytarsus lugens/Corynocera oliveri type	1B, 6B
Tribe Pseudochironomini	
Pseudochironomus	1B, 6B
Subfamily Orthocladiinae	
Tribe Orthocladiini	
Corynoneura/Thienemanniella	1B, 6B
Cricotopus/Orthocladius	1B, 6B
Eukiefferiella/Tvetenia	1B, 6B
Epoicocladius	1B
Euryhapsis	1B
Limnophyes	1B
Metriocnemus	6B
Nanocladius	1B, 6B
Parakiefferiella	1B, 6B
Parakiefferiella sp. B	1B, 6B
Psectrocladius	1B, 6B
Rheocricotopus	1B
Synorthocladius	1B
Zalutschia	1B, 6B
Subfamily Tanypodinae	
Tribe Coelotanypodini	
Coelotanypus	1B, 6B
Subfamily Tanypodinae (continued)	
Tribe Macropelopiini	
Djalmabatista	1B, 6B
Procladius	1B, 6B
Tribe Pentaneurini	
Ablabesmyia	1B, 6B
Labrundinia	1B, 6B

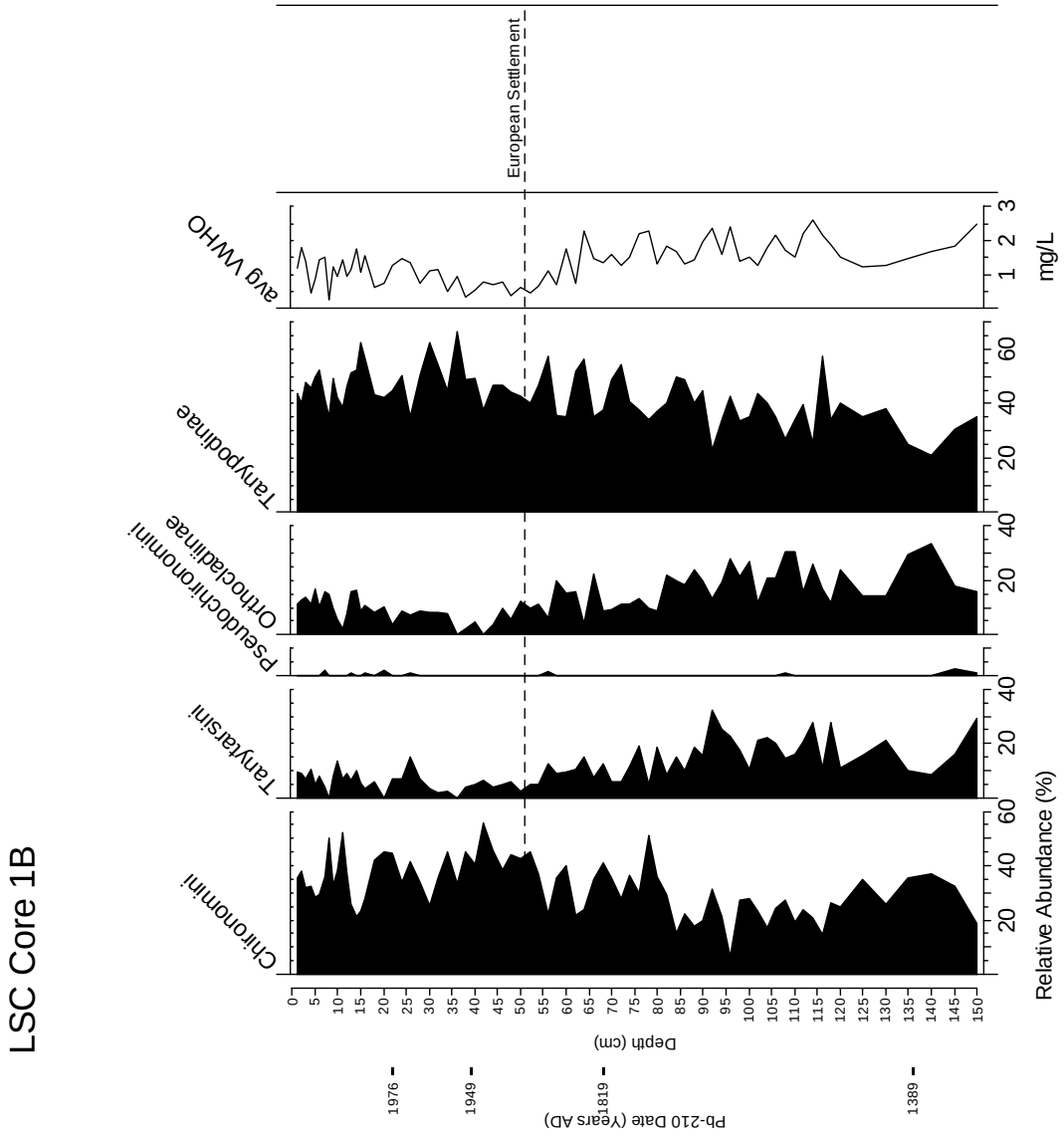

Figure 13a. Core 1B relative abundance profiles for Chironomidae groups. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

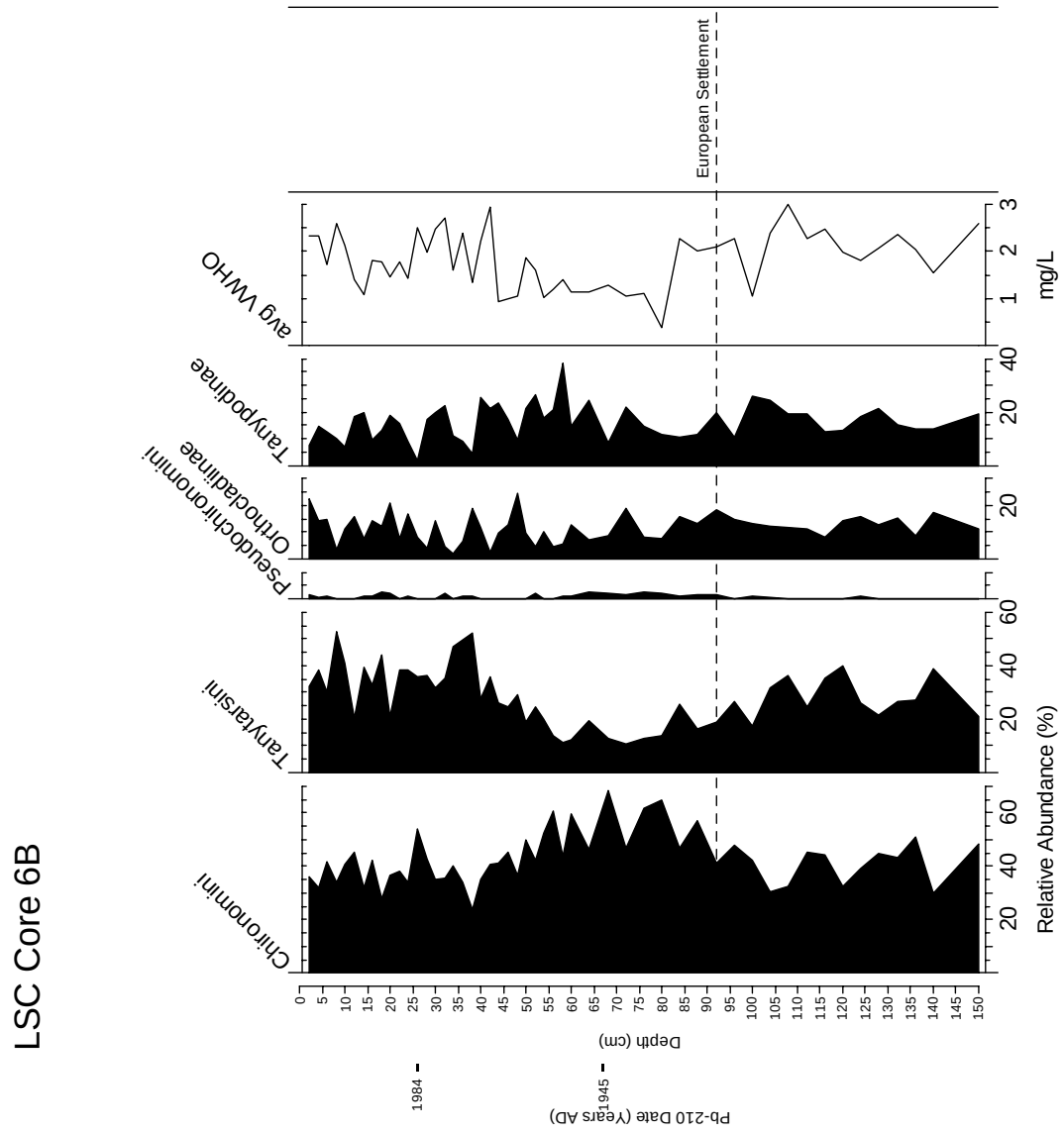

Figure 13b. Core 6B relative abundance profiles for Chironomidae groups. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

Number of head capsules per grams wet sediment (HC / g wet sediment) was low in both cores, and many samples did not result in a head capsule count of 40 after the first processing and picking. More sediment was processed to reach the appropriate total. HC / g wet sediment for core 1B ranged from 1.02 - 7.71 (Figure 14a). The average of HC/g wet sediment for core sections before European settlement (2.17) was slightly lower than the average for core sections following European settlement (2.26). Low values were seen in the lowest sections of the core, with increasing values at c. 1541 AD, followed by lower values at the time of European settlement and increasing values in the upper section of the core to modern times. HC / g wet sediment values were higher for core 6B than 1B, ranging from 2.06 - 5.44 (Figure 14b). The average of HC/g wet sediment for core sections before European settlement (4.23) was higher than the average for core sections following European settlement (3.40). HC / g wet sediment values were high at c. 1809 AD, with lower values seen after settlement, followed by higher values after c. 1965 AD.

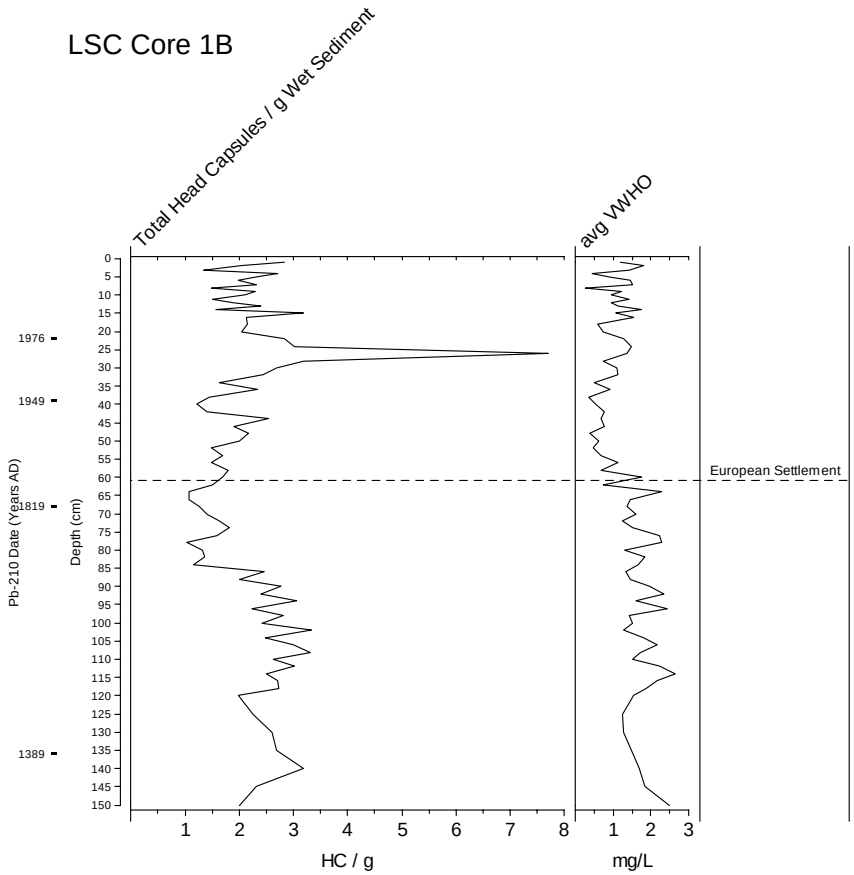

Figure 14a. Core 1B total head capsules per grams wet sediment. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

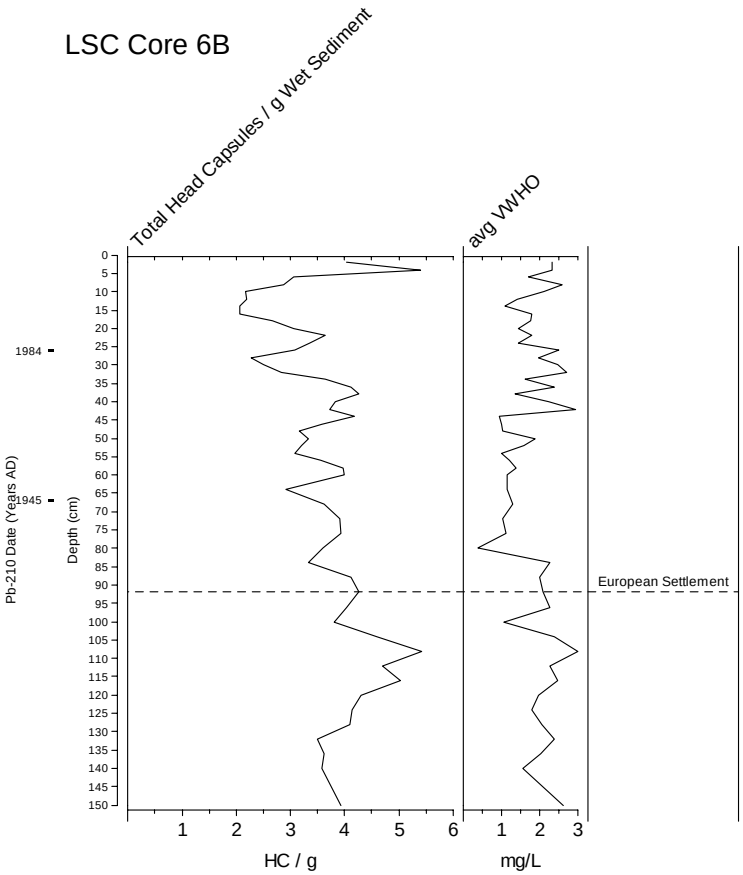

Figure 14b. Core 6B total head capsules per grams wet sediment. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

Core 1B taxa richness ranged from 6 – 23 taxa (Figure 15a). The average of taxa richness values for core sections before European settlement (16.26) was higher than the average for core sections following European settlement (12.28). Taxa richness values showed a significant decrease c. 1363 AD, with higher values seen in c. 1566 – 1900 AD. Lower values were seen up core to modern times with higher variation. Core 6B taxa richness ranged from 10 – 24 taxa (Fig. 15b). Values increased to a peak c.1809 AD. Another peak occurred c. 1941 AD, followed by significantly lower values from c. 1948 – 1969 AD. Higher values occurred up core to modern times. Taxa richness values were higher in core 6B than in 1B.

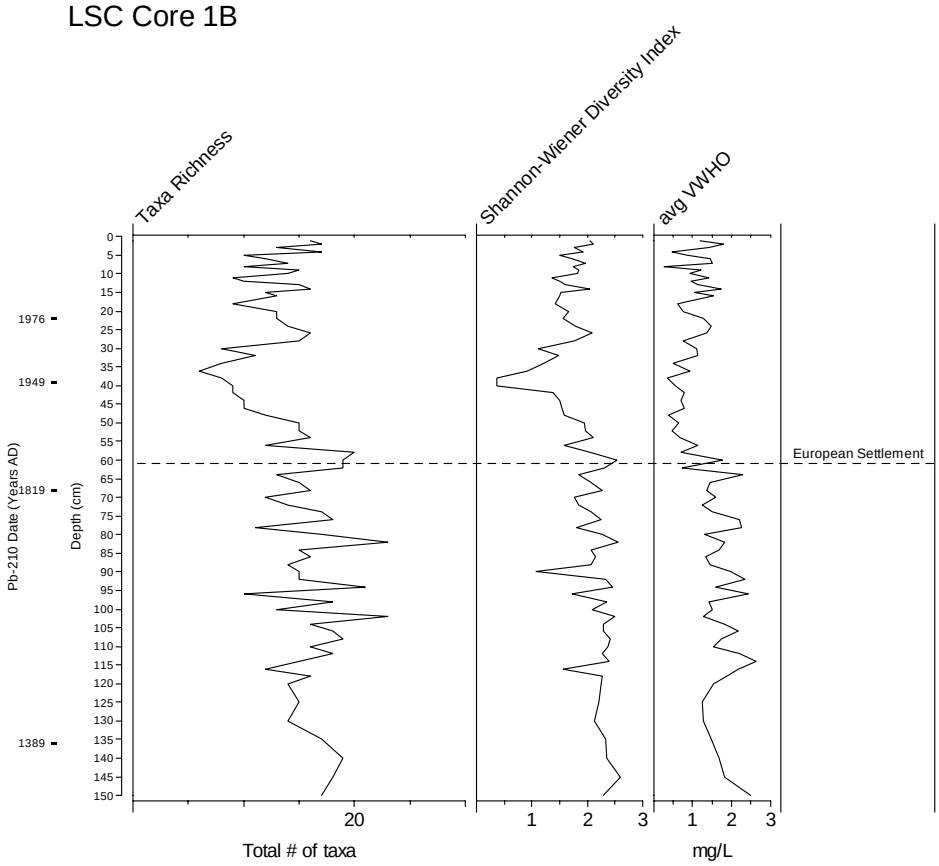

Figure 15a. Core 1B Chironomid taxa richness (total number of taxa) and Shannon Wiener diversity index. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

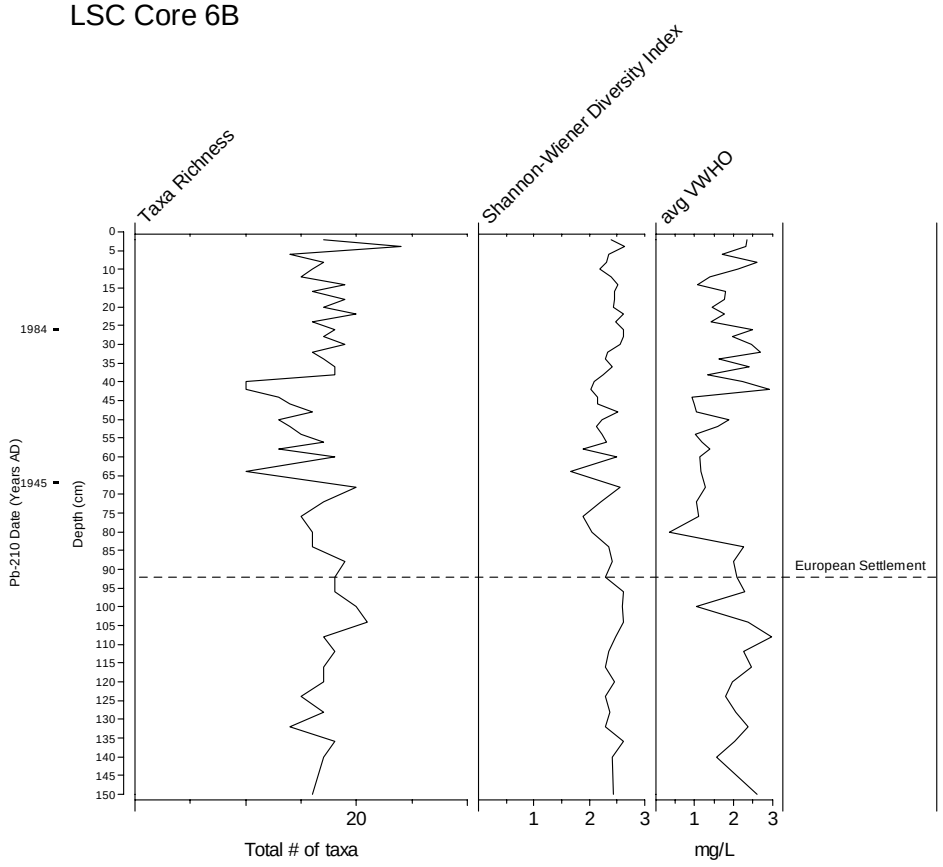

Figure 15b. Core 6B Chironomid taxa richness (total number of taxa) and Shannon-Wiener diversity index. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

The Shannon-Wiener diversity index (H') for core 1B ranged from 0.36 – 2.60 (Figure 15a). Values closest to 0 occurred in c. 1945 and 1951 AD, while largest values occurred in c. 1330, 1730, 1855, and 1864. Pre-settlement values ranged from 1.56 - 2.60. Post-settlement values ranged from 0.36 – 2.11. The H' for core 6B ranged from 1.65 – 2.64 (Figure 15b). Values closest to 0 occurred in c. 1911, 1948, and 1952 AD, while largest values occurred in c. 1689, 1809, 1839, 1982, 1984, and 1987. Pre-settlement values ranged from 2.28 – 2.62. Post-settlement values ranged from 1.65 – 2.64.

Analysis of species shifts in the dendrograms resulted in the establishment of 5 chironomid zones in core 1B (Figure 16a), and 3 zones in core 6B (Figure 15b). Core 1B Zone I spanned from c. 1297 – 1541 AD. *Chironomus* relative abundance peaked in c. 1397 AD, with decreasing values towards the top of the zone. *Dicrotendipes* first appeared in c. 1330 AD, then was absent from c. 1395 – 1490 AD. *Glyptotendipes* first appeared in c. 1363 AD with few occurrences throughout the zone. *Polypedilum*, *Cricotopus/Orthocladius*, and *Procladius* were ubiquitous throughout this zone, with high abundance. *Stempellina* first appeared in c. 1503 AD. Of all 5 zones, the highest relative abundance value for *Nanocladius*, *Cricotopus/Orthocladius* and *Polypedilum* were seen in Zone I. *Harnischia*, *Lauterborniella/Zavreliella*, *Parachironomus*, and *Paratanytarsus* relative abundance values were absent from this zone.

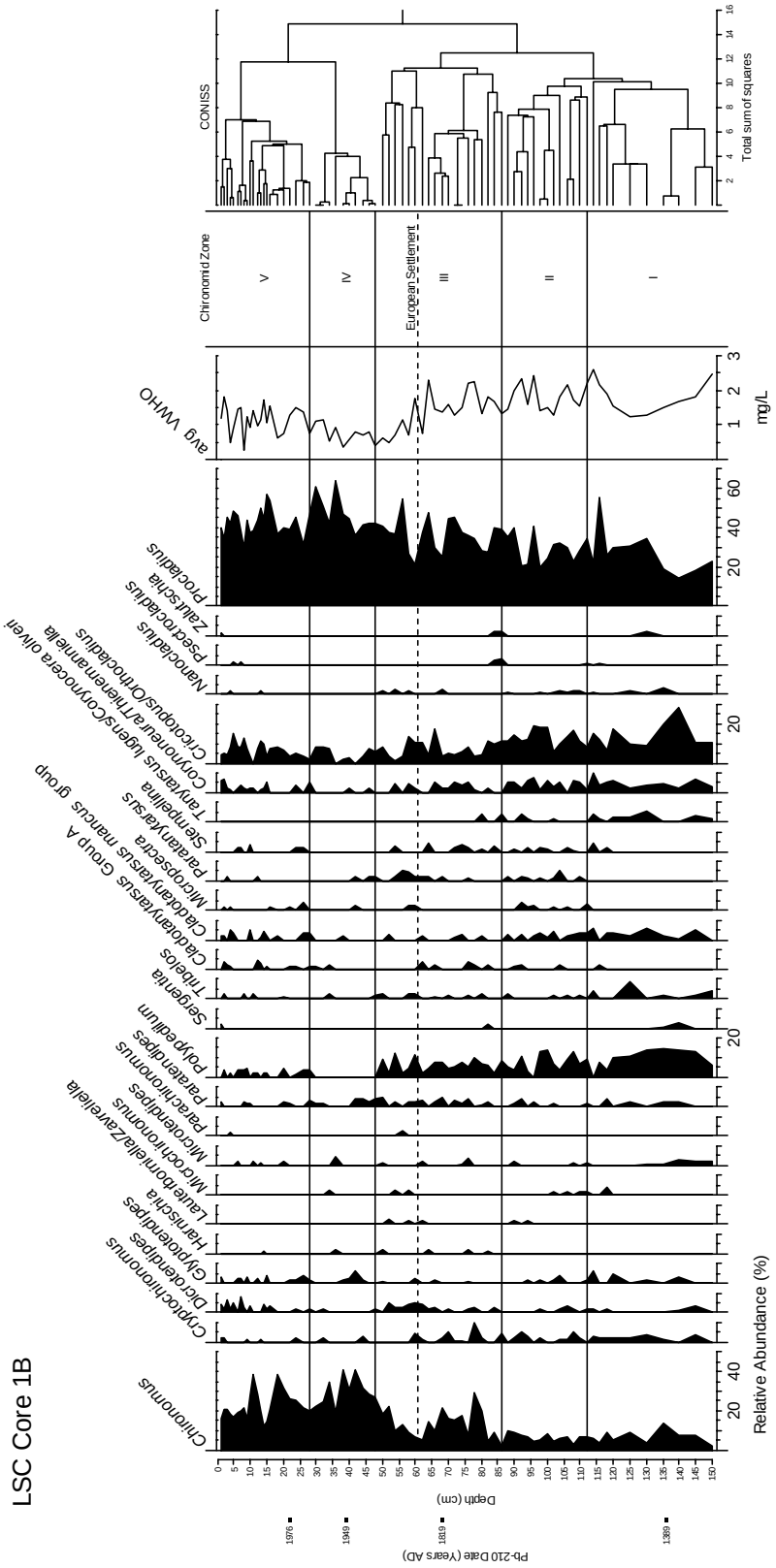

Figure 16a. Summary diagram for core 1B of chironomid zones established from species shifts. Chironomid taxa used to reconstruct oxygen are arranged according to subfamily from right to left: Chironominae, Orthocladiinae, and Tanypodinae. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

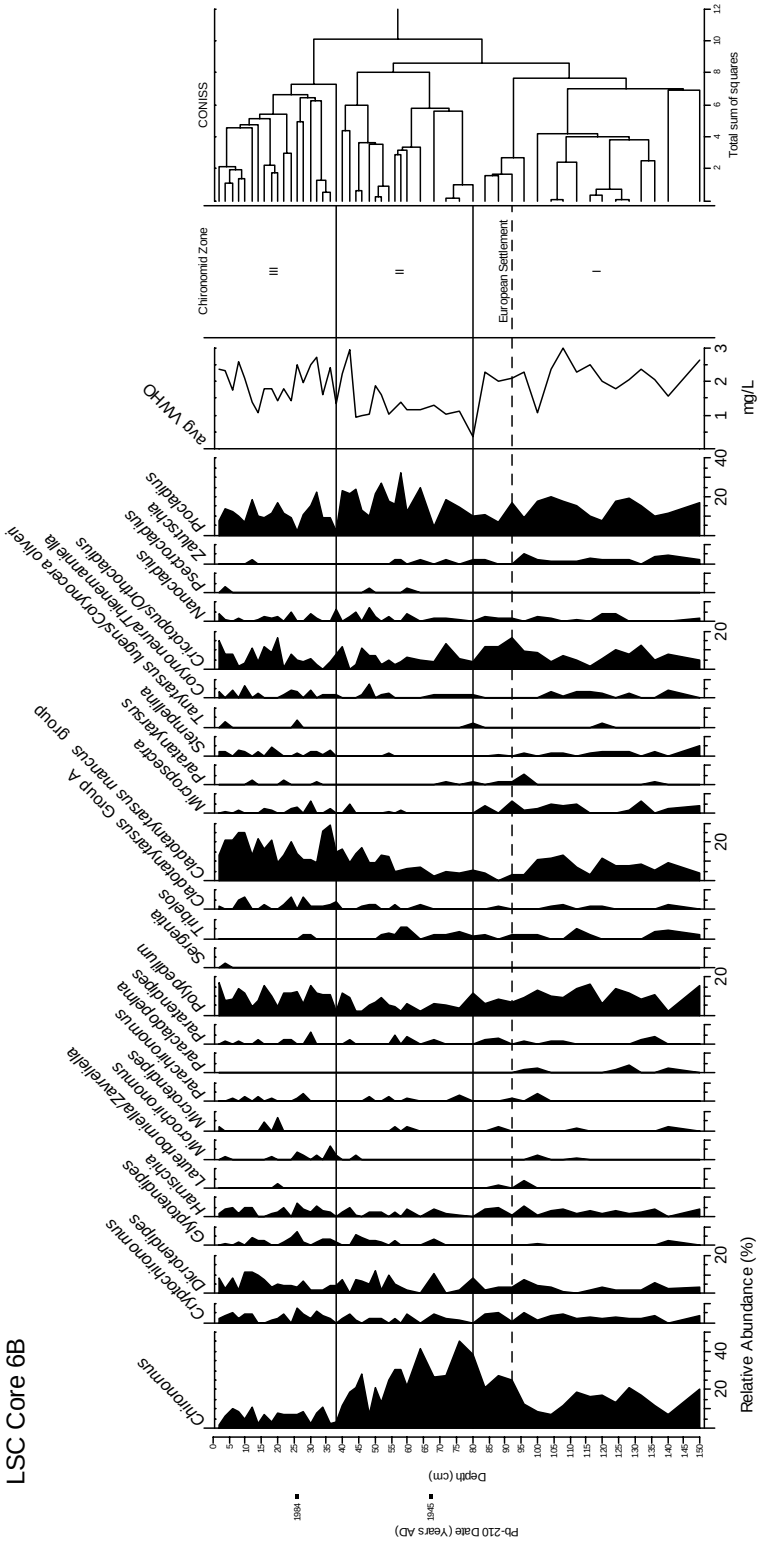

Figure 16b. Summary diagram for core 6B of chironomid zones established from species shifts. Chironomid taxa used to reconstruct oxygen are arranged according to subfamily from right to left: Chironominae, Orthocladiinae, and Tanypodinae. Chironomid reconstructed average volume-weighted hypolimnetic oxygen in mg/L.

Core 1B Zone II spanned from c. 1541 – 1705 AD. *Chironomus* relative abundance began to significantly increase in c. 1667 AD to a peak in c. 1693 AD. *Polypedilum* peaked in c. 1629 AD. *Cricotopus/Orthocladius* displayed high values from c. 1617 – 1642 AD. *Harnischia*, *Parachironomus*, and *Sergentia* were absent from this zone, with *Lauterborniella/Zavreliella* first appearing in c. 1655 and *Paratanytarsus* first appearing in c. 1553.

Core 1B Zone III spanned from c. 1718 – 1909 AD. *Chironomus* relative abundance significantly increased in c. 1718 AD, peaking in c. 1756. Values were low at the time of European settlement, but increased towards the top of the zone. *Cricotopus/Orthocladius* relative abundance was high at the time of European settlement, and *Procladius* displayed a peak in c. 1873 AD. *Lauterborniella/Zavreliella*, *Microchironomus*, *Micropsectra, Nanocladius*, *Corynoneura/Thienemanniella, and Cricotopus/Orthocladius* showed peaks at the time of European settlement. *Harnischia* first appeared in c. 1730 AD with low values throughout this zone, and *Parachironomus* first appeared after European settlement in c. 1873 AD then decreased to 0%. Of all 5 zones, the relative abundance value was highest for *Stempellina* in Zone III.

Core 1B Zone IV spanned from c. 1909 – 1973 AD. *Chironomus* relative abundance values were high throughout this zone. *Glyptotendipes* displayed a peak in 1934. *Procladius* displayed 2 peaks in 1951 and 1957 AD. *Polypedilum* was absent from 1912 – 1960 AD, and reappeared in c. 1936 AD. *Sergentia*, *Stempellina*, *Lauterborniella/Zavreliella*, *Parachironomus*, *Tanytarsus Lugens/Corynocera*, *Nanocladius, Psectrocladius*, and *Zalutschia* were absent from this zone. Of all 5 zones, the relative abundance value for C*hironomus* and *Procladius* was highest in Zone IV.

Core 1B Zone V spanned from c. 1973 – 1999 AD. *Chironomus* showed 2 peaks in c. 1982 and 1990 AD. *Polypedilum* relative abundance values were low throughout this zone. *Procladius* values remained high. *Stempellina* was rare. *Harnischia, Lauterborniella/Zavreliella, Microchironomus, Tanytarsus Lugens/Corynocera, and Zalutschia* were absent from this zone.

Core 6B Zone I spanned from c. 1640 – 1896 AD. *Chironomus* relative abundance displayed peaks in c. 1640, 1719, and 1839 AD, followed by low values at the time of European settlement, followed by a decrease beginning in c. 1794 AD. An increase in values began in c. 1839 AD, with large values occurring towards the top of the zone. *Procladius* relative abundance values displayed a significant increase beginning in c. 1749 AD to a peak in c. 1809 AD, and values decreased at the time of European settlement followed by a peak in c. 1851 AD. *Cricotopus/Orthocladius* relative abundance values displayed a peak in c. 1704 AD, and an increase in values began in c. 1764 AD to a peak in c. 1851 AD, with values remaining high. *Dicrotendipes* displayed a peak in c. 1779 AD, while *Polypedilum* displayed a peak in c. 1824 AD. *Stempellina* relative abundance values were high before settlement, with low values occurring after settlement, and dropping to 0% following settlement. Of all 3 zones, the highest relative abundance value for *Stempellina* and *Cricotopus/Orthocladius* was highest in Zone I. *Psectrocladius* was absent from this zone.

Core 6B Zone II spanned from c. 1896 – 1973 AD. *Chironomus* relative abundance displayed peaks in c. 1911, 1948, 1952, and 1965 AD. *Procladius* relative abundance values were high, and displayed peaks in c. 1955, 1959, and 1967 AD. *Cricotopus/Orthocladius* relative abundance peaked in c. 1926 AD, and with lower

values occurring throughout the rest of the zone with 2 peaks in c. 1965 and 1969 AD. *Polypedilum* relative abundance values were low, displaying peaks in c. 1959 and 1969 AD. *Glyptotendipes, Microchironomus, Psectrocladius, and Ablabesmyia* were absent from this zone. Of all 3 zones, the highest relative abundance value for *Nanocladius, Procladius, and Chironomus* was highest in Zone II. *Lauterborniella/Zavreliella* and *Paracladopelma* were absent from this zone.

Core 6B Zone III spanned from c. 1978 – 1999 AD. *Chironomus* relative abundance values were consistently low in this zone. *Procladius* relative abundance values increased to a peak in c. 1976 AD. *Cladotanytarsus mancus* group, *Cricotopus/Orthocladius* and *Polypedilum* both displayed consistently high relative abundance values throughout this zone. Of all 3 zones, the highest relative abundance value for *Polypedilum* was seen in Zone III. *Psectrocladius* did not occur in this zone until c. 1998 AD. The only occurrence of Sergentia in all zones was in c. 1998 AD. *Stempellina, Lauterborniella/Zavreliella, Microchironomus,* and *Paracladopelma* were absent from this zone.

3.3 Average Volume-Weighted Hypolimnetic Oxygen

For core 1B and 6B, average VWHO values were used to compare the pre-settlement mean to the post-settlement mean and percent decrease in oxygen values. In core 1B, post-European settlement depths ranged from 1 – 60 cm, totaling 37 samples. Pre-settlement depths ranged from 62 - 150 cm, totaling 37 samples. The mean of these values in post-settlement sediment samples was 0.98 mg/L, and the mean of these values in pre-settlement sediment samples was 1.71 mg/L (Table 4). Therefore, mean pre-

settlement values were 0.73 mg/L higher than mean post-settlement values. By comparing an equal number of 33 core samples in core 1B for each of pre and post settlement average VWHO values, results indicate that an 82 % decrease was seen in post-settlement levels compared to pre-settlement values. Average VWHO values began to increase in c. 1427 AD, peaking at 2.17 mg/L in c. 1503 AD (Figure 15a). Values remained high, and a significant decrease to 0.85 mg/L occurred in c. 1837. At the time of European settlement, values were low at 0.68 mg/L (c. 1855) and 1.14 mg/L (c. 1864). Values remained low until c. 1945 when an increase began that led to a peak of 1.51 mg/L in c. 1970 AD. Values then decreased to 0.60 mg/L in c. 1979, followed by higher values up core.

Table 4. Mean pre-European settlement versus mean post-European settlement average VWHO values for cores 1B and 6B.

	Core 1B	Core 6B
Mean of Pre-settlement average VWHO value	1.71 mg/L	2.14 mg/L
Mean of Post-settlement average VWHO value	0.98 mg/L	1.70 mg/L

In core 6B, post-settlement depth ranged from 2 – 92 cm, totaling 38 samples. Pre-settlement depths ranged from 96 – 150 cm totaling 13 samples. The mean of these values in post-settlement sediment samples was 1.70 mg/L, and the mean of these values in pre-settlement sediment samples was 2.14 mg/L (Table 4). Therefore, mean pre-settlement values were 0.45 mg/L higher than mean post-settlement values. By comparing an equal number of 13 core samples in core 6B for each of pre and post

settlement average VWHO values, results indicate that a 52 % decrease was seen in post-settlement levels compared to pre-settlement values. Average VWHO values increased in the deepest sections of the core, reaching a peak in 1838, followed by a significant drop to 1.05 mg/L in 1846 AD (Figure 15b). In 1883 AD, values increased to 2.28 mg/L. Values increased and remained high until 1928 AD, when a decrease to 0.36 mg/L occurred. Values increased to a peak of 2.94 mg/L in 1974 AD, followed by lower values up core, with a low value of 1.08 mg/L in 1995 AD.

Sample-specific errors for the average VWHO reconstructions of both cores were calculated using bootstrapping as the cross-validation technique (Figures 17a and 17b). Errors for bootstrapped estimated average VWHO by weighted averaging with inverse deshrinking were large for both cores. The magnitude of changes seen in reconstructed VWHO in both cores is less than the error terms. Despite the high errors, the reconstructions are valid because the timing of chironomid species changes is in agreement with other indicators of eutrophication. The increase in relative abundance of the eutrophic indicators *Chironomus* and *Procladius* correlates well to the timing of up core increases of phosphorus levels, biogenic silica and diatom accumulation, a shift from benthic to planktonic diatom production, the appearance of eutrophic diatom bioindicators, pigment concentrations and the appearance of diatom eutrophic indicators beginning after European settlement and significantly increasing in the mid 20th century.

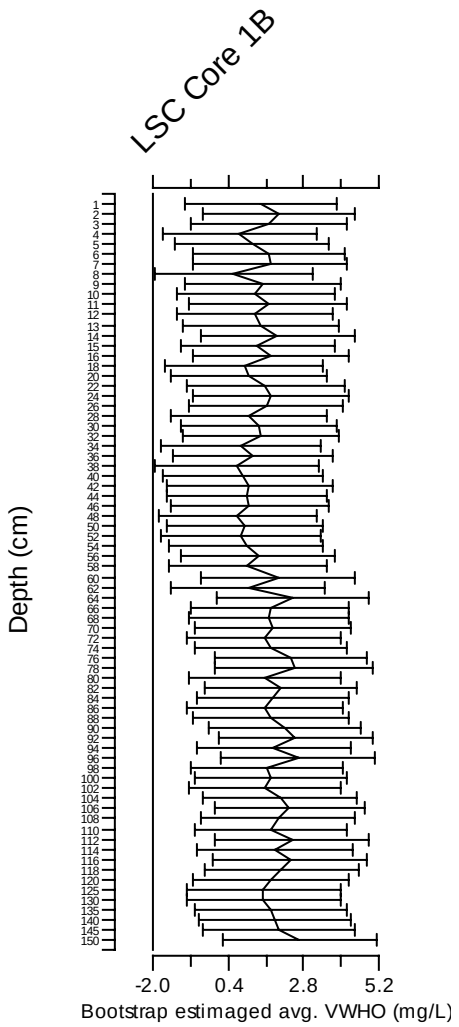

Figure 17a. Core 1B bootstrap estimated average VWHO (mg/L) with error bars vs. core depth (cm).

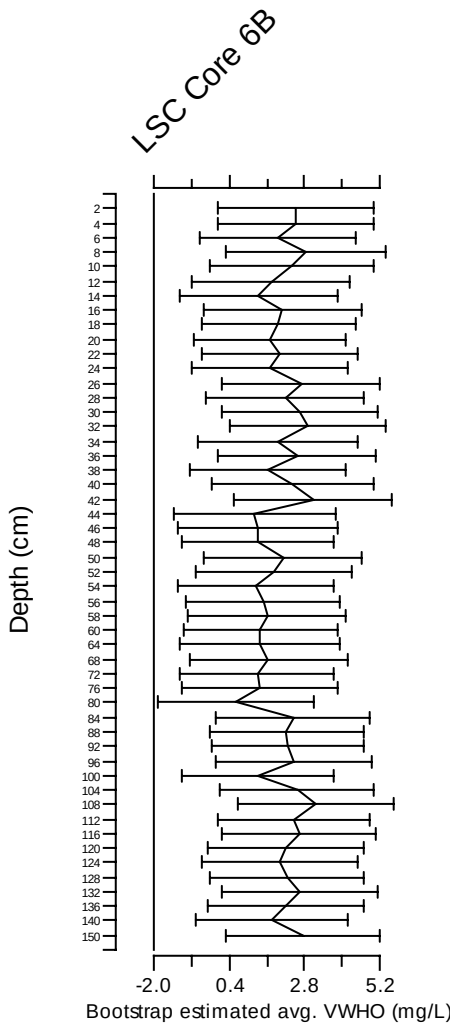

Figure 17b. Core 6B bootstrap estimated average VWHO (mg/L) with error bars vs. core depth (cm).

CHAPTER 4

DISCUSSION

4.1 Magnetic Susceptibility

Clinkers (Figure 18) were found in Core 6B-2, 1 large mass at 88-86 cm, and a second at 110-104 cm, accounting for the large spikes in the magnetic susceptibility profile (Figure 12b). Ships that navigated the St. Croix River utilized coal-burning boilers to generate steam to run the vessels (Braunschweiger, 2005). When coal is burned, non-combustible metals, such as pyrites, coagulate to form lumps known as clinkers (Allen, 2003). In order to maintain the health of the fire and keep it well oxygenated, firemen removed clinkers from the boilers (Braunschweiger, 2005). When clinkers were seen, the fireman sliced the fire, a process that included raking a slice bar across the firebars where clinkers were located, holding them above the flames to be broken with a slice bar, or removed with a devil's claw (Braunschweiger, 2005). A pricker bar was used to remove clinkers from the underside of the grating. Clinkers removed from the fire were placed in deck level hoppers, pumped through an inclined pipe via pressurized water and ejected away from the ship into the water.

Figure 18. Clinker extracted from core 6B-2.

During the period of subsistence and pioneer agriculture in early 1800s, food and supplies were shipped up the St. Croix River on steamboats, but these shipments were infrequent during favorable weather and non-existent during the colder months (Larson, 1972). Market agriculture blossomed in the lower St. Croix River valley during the 1850s and 1860s (Anderson et al., 1996), and oats, potatoes, and wheat were exported downriver on ships (Robinson, 1915). 1838 marked the year of the first steamboat voyage on the St. Croix River, docking at the Dalles (Anderson et al., 1996). Tourists enjoyed taking in the sights on a steamboat (Anderson et al., 1996).

The magnetic susceptibility profiles of cores 1B-1 and 6B-1 suggest long-term land use changes in the LSC region since the time of European settlement as peaks in magnetic susceptibility are the result of eroded sediment entering the lake (Triplett et al., 2009). The timing of initial susceptibility increases in the mid to late 1800s corresponds well to the onset of logging operations in the 1840s, to the complete settlement in the prairies by 1880, and to wheat, dairy, and corn agricultural operations in the early 1900s (Anderson et al., 1996). In core 1B-1, a decrease in susceptibility levels occurred at 1800 AD, with increasing levels beginning in 1813 AD, followed by a continual increase until 1948 AD. Core 6B-1 also showed an increase in susceptibility levels, with slight decreases occurring from 1841 – 1848 AD and 1902 – 1928 AD. Despite these slight decreases, the times of susceptibility increases in core 6B-1 correlate to times of land use change. The decrease in susceptibility levels to modern times in the second half of the 20^{th} century in both cores corresponds well to times of decreasing agricultural practices, soil conservation, and reforestation beginning in the 1960s (Anderson et al., 1996).

Magnetic susceptibility peak values were higher in downstream core 1B than upstream core 6B. These results were also seen in the Edlund et al. (2009) magnetic susceptibility profiles. Varying sediment mineralogy, grain size, and concentration of magnetic grains result in differences in magnetic susceptibility (Triplett et al., 2009). The origin of sediment may be different for the upstream half and the downstream half of LSC. It appears that sediment has been deposited by side-valley tributaries in the downstream portion of LSC, leading to higher magnetic susceptibility peaks in core 1B. Triplett et al. (2009) reported that dredging of the Kinnickinnic River delta occurred in the 1900s in order to allow for ship passage. Dredging disturbed sediment that initially lay undisturbed in the delta, resulting in higher peaks in core 1B than in 6B.

4.2 Chironomids

The Chironominae are often the most common subfamily in temperate, mesotrophic and eutrophic lakes (Sæther, 1979). In both core 1B and 6B, the Chironominae were more abundant than the Orthocladiinae and the Tanypodinae. Chironomid communities of both cores were composed mostly of littoral taxa (Figures 13a and 13b). Littoral chironomid head capsules are carried from shallow habitats and deposited in the benthos of the lake by water movement, resulting in a full representation of chironomids associated with different lake habitats in sediment cores (Francis, 2001).

Troelstrup et al. (1993) found that in the midge analysis of LSC 2 (extracted near Lakeland in the same sub-basin of core 6B of this study), the percentage of Tanytarsini and Chironomini showed an up core increase, and Orthocladiinae did not show

significant trends. These results show a good correlation to the results of this study. Percentage of *Procladius* was high in lowest core sections. In core 6B, the percentage of Chironomini and Tanytarsini, like LSC2, showed an up core increase, with Chironomini showing a slight decrease in the upper-most core sections. In core 6B, the Orthocladiinae were much more abundant in the lower core sections than in LSC2. *Procladius* was abundant in the lowest core sections (Figure 14b). It would be expected that tribe profiles for LSC 2 and core 6B would be similar as they both were extracted from sub-basin 3.

Head capsules / g wet sediment were low in cores 1B and 6B, and the profiles appear to track chironomid reconstructed average VWHO values, indicating that oxygen levels may influence head capsule density. Low head capsules / g wet sediment values somewhat correlated to low average VWHO values. Low head capsule counts were consistent with low midge densities seen in 3 LSC cores analyzed in the Troelstrup et al. (1993) study. Troelstrup et al. (1993) attributed low midge densities in LSC to anoxia, correlating well to the results of this study with low head capsules / g wet sediment occurring at times of low average VWHO.

Taxa diversity is considered to consist of several components, including taxa richness, or number of taxa. Healthy ecosystems are often associated with high taxa richness. However, as aquatic conditions degrade, the diversity of pollution tolerant taxa may increase. Troelstrup et al. (1993) discovered that chironomid taxa richness in LSC ranged from 4 – 16 taxa; lower values than were seen in this study. The significant increase in total phosphorus load to LSC after 1940 (Triplett et al., 2009), and the increase in fossil pigments after 1950 (Edlund et al., 2009) indicate an increase in productivity in LSC that could lead to decreased oxygen levels and degraded water

quality. The timing of the two eutrophic signals corresponds well to lower taxa richness values seen in core 1B from 1865 – 2006 AD, and significantly low values from 1934 – 1960 AD, and in core 6B from 1883 – 1974 AD, with significantly low values from 1949 – 1974 AD. In core 1B, the post-European average of taxa richness was 12.28, while the pre-European average was 16.26, and in core 6B, the post-European average was 16.28 while the pre-European average was 17.25, indicating richness was greater before European settlement. These results correlate well with the lower average VWHO seen after European settlement, and may indicate that land use changes resulting in eutrophication and decreased oxygen levels may have resulted in decreased taxa richness. The high richness values seen in core 1B beginning in 2002, and in 1986 in core 6B could be explained by the decrease in total phosphorus concentrations to LSC beginning in 1976 due to enforced regulations of point-sources of pollution (Kloiber, 2004). Enforced regulations of pollution in the 1970s may have led to less productive conditions in modern times, which may explain why higher average VWHO values which were seen up core beginning in 1972 AD in core 1B and 1974 AD in core 6B.

Shannon-Wiener diversity index values (H') for core 1B were closest to 0 in the late 1940s and early 1950s, indicating a low amount of taxa diversity during these times, and high values occurred in the 1840s, indicating a large amount of diversity and a greater parting among taxa. H' of core 6B was closest to 0 in 1949, and highest in 2004. Core 6B displayed higher H' values than 1B, indicating a greater amount of biodiversity in the upstream sub-basin. Chironomid reconstructed average VWHO values were low during the early to mid 1900s, correlating well to low Shannon-Wiener diversity index values and implying low biological diversity resulting from degraded water quality.

Taxa richness appeared to track the Shannon-Wiener diversity index in both cores. When taxa richness was high, H' was high, and when taxa richness was low, H' was low. If it is assumed that low taxa richness and H' values are the result of degraded water quality, than the low values of both indices seen after European settlement, especially in the 1960s and 1970s correlate well to the appearance of diatom eutrophic indicators after 1950 (Edlund et al., 2009), the significant total phosphorus increase seen after 1940 (Triplett et al., 2009), and the significant increase in primary production seen after 1950 (Troelstrup et al., 2003).

The MPCA (2001) reports that LSC is eutrophic based on the Carlson Trophic State Index. Eutrophication is often caused by increased nutrient levels, changes in lake morphometry, or alterations in the interactions of intra-lake biota (Whiteside, 1983). During the mid and late 20th century, lake eutrophication was often caused by increased inorganic nutrients, such as nitrogen and phosphorous (Wetzel, 2001). When a lake is impacted by nitrogen and phosphorus runoff, primary producer productivity accelerates, changing the chemical and physical characteristics of the lake. Physical and chemical alterations are seen in the fluctuating species composition of the lake's fauna and flora (Whiteside, 1983).

Bioindicators of eutrophy such as *Chironomus*, *Procladius*, *Dicrotendipes*, *Glyptotendipes*, and *Polypedilum*, all displayed an increase in abundance around the time of European settlement in core 1B. This correlates well with the findings of Edlund et al. (2009), who discovered that after the 1950s, significant increases in eutrophication signals, including biogenic silica, diatom accumulation, and a shift from benthic to planktonic diatom community occurred.

Core 6B displayed an up core decrease in the relative abundance of *Glyptotendipes, Chironomus* and *Procladius* beginning around the 1970s. During the 1970s and 1980s, reconstructed oxygen levels were higher than the early and mid 20th century. Edlund et al. (2009) reported an increase in siliceous microfossil accumulation in core 6B from 1950 to 1960s when reconstructed total phosphorus levels were high, followed by decrease in accumulation during the 1970s and 1980s that was not seen in core 1B. In addition, in core 6B, Edlund et al. (2009) reported an increase in biogenic silica accumulation from 1940 to the mid 1960s, followed by a decrease until around 1980. Nutrient loads have been shown to increase diatom productivity (Conley et al., 1993). The decreases in siliceous microfossil accumulation and biogenic silica seen in the 1970s and 1980s may indicate less productive conditions that would lead to an increase in oxygen levels during this time. This assumption is further strengthened by the appearance and persistence of the oligotrophic indicator *Stempellina* c. 1974 AD, perhaps indicating less productive conditions. The improved aquatic conditions during this time may explain the decrease in relative abundance of chironomid bioindicators of eutrophic conditions, particularly *Chironomus* and *Procladius*. In 1992 in the northern portion of LSC was dominated by development and cultivated/planted land use that may contribute nutrients such as fertilizers to the lake that decrease hypolimnetic oxygen. Beginning c. 1994, higher relative abundance values of *Dicrotendipes*, *Polypedilum*, *Chironomus* and *Procladius* were seen, perhaps responding to these land uses.

Some chironomid taxa are bioindicators of less productive aquatic conditions. *Stempellina* is associated with oligotrophic lakes (Epler, 2001), *Sergentia* is associated with mesotrophic lakes with moderate oxygen depletion (Quinlan et al. 1998), and

Paratanytarsus is associated with mesotrophic lakes (Sæther, 1979). Francis (2001) discovered that *Sergentia* was present in Grapevine Point (Douglas Lake, Michigan) before European settlement, but relative abundance decreased to 0% and remained there following settlement. In addition, Stahl (1959) discovered that as Myers Lake, Indiana, filled in with sediments, hypolimnetic volume decreased, resulting in a decrease of oxygen, as well as a decrease in *Sergentia* abundance. In core 1B, *Stempellina*, *Sergentia*, and *Paratanytarsus* all decreased in relative abundance after European settlement, perhaps indicating deteriorating water quality resulting from more productive conditions. In core 6B, *Stempellina* occurrences were rare, and abundance reached 0 % after European settlement with abundance increasing after 1977 AD. Perhaps this increase was the result of enforced regulations of point-sources of pollution beginning in 1976 (Kloiber, 2004) *Sergentia* appeared only once in 2004, while *Paratanytarsus* values decreased and were rare after settlement. These findings correlate well with the more productive conditions in LSC as indicated by the increase in relative abundance of diatom bioindicators of eutrophy after 1950 (Edlund et al., 2009).

Meriläinen et al. (2000) studied how profundal chironomid taxa in Lake Lappajärvi, Finland were impacted by eutrophication and decreased hypolimnetic oxygen levels. *Micropsectra* spp., a bioindicator of oligotrophic conditions, typified the pre-industrial era before 1935. *Micropsectra*, *Paracladopelma*, and *Sergentia* decreased in numbers. With a period of increased erosion and heightened nutrient loading from 1960 to 1970 came significant increases in *Chironomus* type *anthracinus*, a bioindicator of moderate eutrophy. The succession to meso-eutrophic conditions saw an increase in *Chironomus* type *plumosus*, another bioindicator of eutrophication. Chironomid results

in the Meriläinen et al. (2000) study correlate well to this study of LSC. In core 1B, relative abundance of *Micropsectra* was greater before European settlement, and showed decreasing abundance following settlement. In core 6B, *Micropsectra* displayed a significant decrease in relative abundance after settlement. An increase was seen c. 1984, but relative abundance decreased up core. Nutrient inputs from agriculture and industry in the mid 1900s corresponded to an increase in *Chironomus* in core 1B and 6B.

4.3 Average Volume-Weighted Hypolimnetic Oxygen

Reconstructed oxygen levels decreased after European settlement, and average VWHO values were low in the 1940s, correlating well with increases in total phosphorus load to LSC (Triplett et al., 2009), increases in biogenic silica concentration (Edlund et al., 2009), and the appearance of diatom bioindicators of eutrophic conditions (Edlund et al., 2009), all occurring in the mid 1900s. Eutrophication results from increases in phosphorus to a lake that promotes primary production, and ultimately decreases oxygen levels. The lower post-European settlement average VWHO values correlate well with the previously mentioned eutrophication signals that indicate the water quality of LSC has changed since the 1850s. Oxygen depletion after 1950 may have resulted from an increase in primary production after 1950 (Troelstrup et al, 1993), and an increase in total phosphorus after 1940 (Triplett et al., 2009).

Land use change impacts average VWHO. In core 1B, a decrease in average VWHO occurred in c. 1855 AD, with low values persisting until c. 1954 AD. In core 6B, decreases began c. 1851 AD, with low values persisting until c. 1967 AD. These low

values correspond well to agricultural operations that persisted in the region during the late 1800s and early 1900s (Anderson et al., 1996). Agriculture contributed nutrients to the lake, resulting in oxygen depletion. This conclusion is augmented by the increase seen in the diatom reconstructed total phosphorus levels reported by Edlund et al. (2009).

Maximum total phosphorous levels occurred in the 1990s, as reported by Edlund et al. (2009). Total phosphorus load to LSC reached a maximum from 1980 to 2000 AD (Triplett et al., 2009). It would be expected that reconstructed oxygen would be lower during the 1990s in response to high phosphorus levels, however, oxygen levels were higher than levels seen during the mid 1900s in core 1B and 6B. In both cores, values were low in the early 1990s and displayed an up core increase. However, values during the 1990s were still lower than pre-European settlement values.

Reconstructed average VWHO values were higher in core 6B than in core 1B. The core 1B post-European settlement mean for all core sections of average VWHO was 0.98 mg/L, while the core 6B mean was 1.70 mg/L. It would be expected that because core 6B displayed a significantly higher historical accumulation of biogenic silica and higher siliceous microfossil accumulation than core 1B, reconstructed average VWHO values would be lower in core 6B due to the indication of more productive conditions by these diatom findings of Edlund et al. (2009). This introduces some uncertainty in the strength of chironomid reconstructed average VWHO.

Despite the increase in average VWHO seen in the uppermost sections of both cores that do not correlate to increased phosphorus levels and diatom accumulation reported by Edlund et al. (2009) and Triplett et al. (2009), the reconstructions of this study are reliable. Even though chironomids respond to a variety of environmental

variables, such as water temperature maximum lake depth, and major ion chemistry, Quinlan and Smol (2001), concluded that the strongest explanatory variable to chironomid data was end-of-summer average VWHO and bottom oxygen concentration. Performance statistics of the WA with inverse deshrinking inference model of the Quinlan and Smol (2001) study resulted in an r^2_{jack} of 0.544 and a RMSEP of 2.147 (Quinlan and Smol, 2001), slightly higher values than the performance statistics of this LSC study (r^2_{jack} = 0.489, RMSEP = 2.316).

Oxygen values from the chironomid reconstructions correlate well to measured modern oxygen data, further strengthening the reliability of LSC average VWHO reconstructions. Lafrancois et al. (2009) reported that in August 2005 AD, dissolved oxygen at 6-14 m was 4 mg/L, and in August 2006, dissolved oxygen at 6-9 m was 4 mg/L. In 2005, 6 out of 7 sample sites showed dissolved oxygen levels close to 0 mg/L at approximately below 13 m (Lafrancois et al. 2009). In 2006, all 9 sample sites showed dissolved oxygen levels close to 0 mg/L at approximately below 10 m (Lafrancois et al. 2009). It would be expected that average VWHO values would be low in the 1990s due to increased phosphorus loads; however, values were higher than the mid 1950s in both cores. Despite this limitation, high values seen in the 1990s were still lower than pre-European settlement values, helping to supporting the modern measured data. VanderMeulen and Elias (2008) collected water quality data that profiled dissolved oxygen at 4 sample sites on LSC once a month during the open water season for 2007. Mean dissolved oxygen for 4 monitoring sites in LSC were 8.8, 10.97, 9.1, and 8.5 mg/L. These results indicate relatively high oxygen concentrations, and may help to explain the

higher reconstructed average VWHO values seen in the uppermost sections of core 1B and 6B.

Using a transfer function developed for Ontario, Canada for a riverine lake in Minnesota introduces some uncertainties. The lakes in Quinlan and Smol's training set were not riverine in nature. Bedrock geology was different between these two areas, resulting in differences to water chemistry. Despite these limitations, the use of a transfer function developed for Canadian lakes is justifiable for use in this study due to similar climate. The climate of the Muskoka-Haliburton region is moderately moist and cool, annual precipitation ranges from 900 – 1200 mm, mean January temperatures are -11.2^{o}C and mean July temperatures are 19.1^{o}C (Hutchinson et al., 1994). The climate of southwestern Ontario is humid continental, while the climate of central Ontario is humid continental. The climate of Minnesota is continental, with cold winters and warm summers. Mean January and July temperatures for Minneapolis-St. Paul 23^{o}C and -11^{o}C respectively.

CHAPTER 5

CONCLUSION

Because of its scenic, recreational, and environmental qualities, a portion of the SCR, including LSC, was classified as the St. Croix National Scenic Riverway under the National Wild and Scenic Rivers Act (Anderson and Varro, 2002). Due to this classification, it may be incorrectly assumed that water quality is excellent in LSC. Conditions have become degraded as a result of land use change since the time of European settlement. Historical oxygen conditions in LSC were reconstructed using subfossil Chironomidae remains from lake sediment cores. Mean post-settlement chironomid reconstructed average volume-weighted hypolimnetic oxygen values were 0.73 mg/L lower than mean pre-settlement values in core 1B and 0.45 mg/L lower in core 6B. Both cores appeared to show a significant decrease in post-European settlement average VWHO values compared to pre settlement values, with core 1B values decreasing by 82 % and core 6B decreasing by 52 %.

Changes were seen in the Chironomidae communities in LSC that resulted from eutrophication and low oxygen conditions. In the post-settlement era of LSC, eutrophic chironomid bioindicators dominated the communities in core 1B and 6B. High relative abundance values occurred after 1950 for most eutrophic indicators, including *Chironomus* and *Procladius*. Even though some chironomid eutrophic indicators decreased in relative abundance in core 6B in the later part of the 20th century, values were still higher than pre-European settlement. Indicators of less productive conditions often showed up core decreases in relative abundance after European settlement,

including *Paratanytarsus* and *Stempellina*. In general, reconstructed average VWHO values were low when the relative abundance of *Chironomus* and *Procladius* were high.

The timing of Chironomidae community shifts and oxygen depletion correlated well to other signals of eutrophication that have been studied in LSC. Up core increases in nutrients, biogenic silica and diatom accumulation, a shift from benthic to planktonic production, the appearance of eutrophic diatom bioindicators, increases in sedimentation rates, pigment concentrations, and the accumulation of organic matter and carbonates are all signals of eutrophication. The majority of these signals did not occur immediately after European settlement, but in the early to mid 1900s, when *Chironomus* and *Procladius* relative abundance values were significantly high and average VWHO values were low. Chironomid community shifts and reconstructed average VWHO values provide another eutrophication signal in the LSC record. This study provides historical data for the interagency St. Croix Basin Planning Team which determines goals and management strategies for the SCR, as well as concerned environmental planners, policy makers and citizens.

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