A chironomid-inferred paleolimnological reconstruction of past hypolimnetic oxygen

concentrations in Loughborough Lake, Ontario: Implications for the local Lake Trout habitat

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ABSTRACT

Lake Trout (*Salvelinus namaycush*) are a rare and valuable species in Ontario with life history traits and habitat requirements that make them sensitive to environmental change. Nutrient loading and climate change threaten Lake Trout habitat by reducing hypolimnetic (deep water layer) dissolved oxygen (DO) concentrations to below conditions necessary for optimal Lake Trout metabolic function (6-7 mg/L). Loughborough Lake, Ontario, supports a naturally reproducing Lake Trout population. This lake is of significant management interest due to increasing shoreline development and agriculture in the watershed, as well as reported declines in end-of-summer hypolimnetic DO to below 5 mg/L in recent years. This study used sedimentary chironomid assemblages to create a paleolimnological reconstruction of past hypolimnetic DO in Loughborough Lake, and to assess the implications for the management of the Lake Trout population. The reconstruction revealed that baseline hypolimnetic oxygen conditions were maintained between the optimal range of 6-7 mg/L for Lake Trout for most of the inferred ~200 year history. Slight changes in inferred volume-weighted hypolimnetic oxygen (VWHO) suggest a decline in DO during periods of forestry in the late 1800s, followed by a recovery period in the 1900s. These observations were corroborated by reconstructed chlorophyll-*a* concentrations, an indicator of primary production. The increase in relative abundance of littoral chironomid taxa since the mid-1900s may indicate deteriorating hypolimnetic DO, which could be reflective of increased agricultural activity or the effects of climate warming. These results do not indicate an imminent need for changes to Lake Trout management; however, further investigation of multiple-proxies, including diatom analysis, would strengthen observations. This study provides historical context for making informed decisions regarding the future management of Lake Trout habitat in Loughborough Lake.

ACKNOWLEDGEMENTS

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LIST OF COMMON ABBREVIATIONS

Dissolved oxygen – DO

Volume-weighted hypolimnetic oxygen – VWHO

Ontario Ministry of Natural Resources and Forestry – OMNRF

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INTRODUCTION

Lake Trout habitat; the effects of nutrient loading and climate change

Lake Trout (*Salvelinus namaycush*) are one of Ontario's most valuable fisheries resources, drawing anglers from around the world. Only about one percent of Ontario's many lakes support Lake Trout populations; however, this represents ~20-25% of the world's distribution of Lake Trout lakes (OMNRF 2015). Of the 2432 of Ontario's inland lakes known to contain Lake Trout, 2261 lakes are managed for Lake Trout by the Ontario Ministry of Natural Resources and Forestry (OMNRF 2015). OMNRF policies and guidelines have been applied to these lakes to protect Lake Trout habitat and population viability (OMNRF 2015). It is therefore necessary to attain accurate information about present-day conditions and trajectories of environmental change occurring in these lakes to better manage this rare resource.

Lake Trout life histories and habitat requirements make them particularly vulnerable to environmental change and are therefore of specific concern for lake and fisheries managers. Lake Trout are slow to grow and mature, are large-bodied and long-living, and thus have low replacement rates (Shuter et al. 1998). They are well adapted to the cold, deep water of thermally stratified oligotrophic lakes, that have low levels of nutrients and high dissolved oxygen (DO) concentrations (Plumb and Blanchfield 2009). The primary factors that influence vertical movements of Lake Trout are water temperature and DO concentration, with ideal conditions of <12°C and >6-7 mg/L DO, respectively (Plumb and Blanchfield 2009). However, daily movement into conditions as high as 15°C and as low as 4 mg/L DO for short periods of foraging have been documented during summer months (Plumb and Blanchfield 2009). In thermally stratified lakes, the cold dense water ideal for Lake Trout is contained within the bottom-most layer called the hypolimnion. Adequate hypolimnetic DO concentrations are critical

for metabolic function in life support activities like feeding, swimming, growth, and reproduction. As summer progresses, natural processes like decomposition and respiration consume the finite supply of hypolimnetic DO. Therefore, end-of-summer DO concentrations can limit Lake Trout metabolic function, and should hypoxic concentrations of below 5 mg/L occur, maximum metabolic power capacity may decrease significantly in Lake Trout (Evans 2007).

Hypoxia is often caused by anthropogenic disturbances that increase loading of limiting nutrients. Phosphorus is the main limiting nutrient in Ontario Lake Trout lakes, and phosphorus loading often causes algal blooms, which in turn can deplete end-of-summer DO concentrations (Schindler et al. 1971, Dillon et al. 2003, Lienesch et al. 2005). Phosphorus loading and subsequent DO depletion can be caused by watershed disturbances like urban development, agricultural and sewage runoff, which cause increased erosion, and inflow of pesticides and fertilizer (Carpenter et al. 1998, Mallin et al. 2006). Anthropogenic eutrophication has been one of the most detrimental water pollution problems for aquatic life, especially for oxygen-sensitive fish (Pollock et al. 2007). With increased management of total phosphorus (TP) loading in lake catchments since the 1990s, there have been observed declines of TP in many Lake Trout lakes (Jeppesen et al. 2005, Winter et al. 2007). However, end-of-summer DO concentrations often remain below the habitable range of 6-7 mg/L for Lake Trout (Winter et al. 2007).

Alongside the negative impacts of nutrient loading on aquatic ecosystems, climate change is a potential threat multiplier (Smol, 2010). Climate warming may place additional stress on hypolimnetic DO concentrations due to changes in lake thermal regimes, including earlier onset of thermal stratification, longer ice-free periods, and later fall turnover (Stainsby et al. 2011). This amounts to an increase in the total length of the lake stratification period, now often

extending a month longer relative to historical averages, prolonging the period of DO depletion (Stainsby et al. 2011, Foley et al. 2012). Hypolimnetic hypoxia due to a longer stratification period may create an internal phosphorus load and continued algal blooms, which can often undermine management efforts for nutrient loading (Jankowski et al. 2006, Nürnburg et al. 2013). As many Ontario Lake Trout populations are located at or near the southern limit of their range in narrow hypolimnetic temperature and DO boundaries, they will be especially sensitive to climate warming.

Loughborough Lake, Ontario

Due to the potential impacts of climate change on temperate stratifying lakes, it has become important to understand to what degree the deep, low-nutrient Ontario Lake Trout lakes may have already been impacted, as well as possible future implications. Loughborough Lake (44° 26' 48" N, 76° 24' 26" W) is a Lake Trout lake of southeastern Ontario located within the South Frontenac Township of Frontenac County (Figure1). Its deep west basin, with a maximum depth of 38 meters, thermally stratifies providing the cold-water habitat necessary to support Lake Trout. The OMNRF designated Loughborough Lake for Lake Trout management in 2015, and lists it as a naturally reproducing population (defined as one that may have lost populations but has been identified for rehabilitation), rather than a put-grow-take lake (one that is stocked annually to draw anglers away from natural populations) (OMNRF 2015). However, the OMNRF has been stocking Loughborough Lake with juvenile Lake Trout since 2007. Loughborough Lake is part of Ontario Fisheries Management Zone 18, which was formed in 2008 and continues to this day to allow year-round recreational fishing of Lake Trout in specific lakes, including Loughborough Lake (OMNRF 2016).
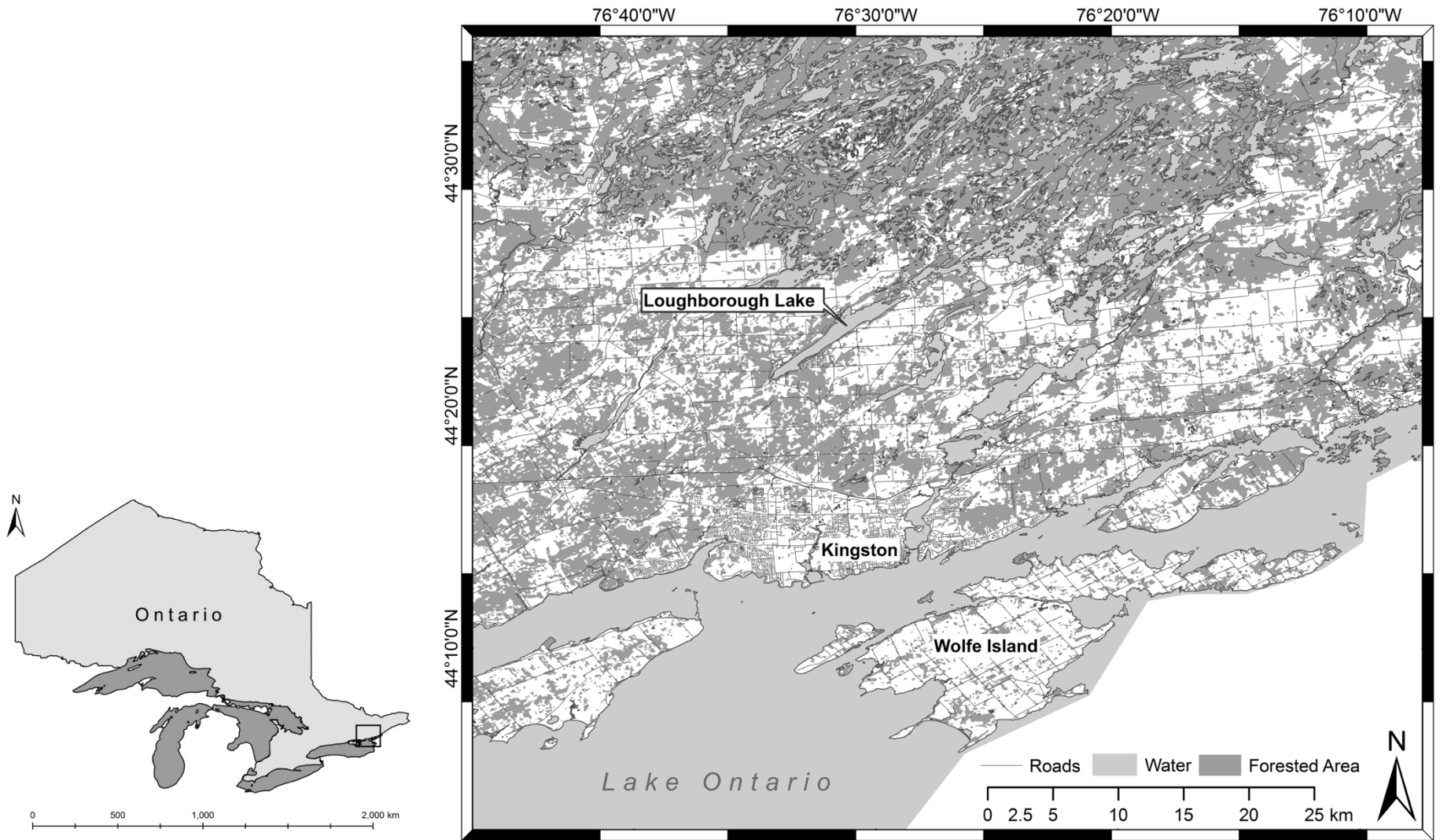
Figure 1: Map showing the location of Loughborough Lake in southeastern Ontario, Canada.

Loughborough Lake is of significant interest due to management concerns associated with shoreline development, agriculture in the watershed, and long-term changes in hypolimnetic DO concentration. While most of Ontario's Lake Trout lakes occur on the Precambrian Shield, Loughborough Lake is situated on the boundary between shield and limestone plains geology (Currie and Ermanovics 1971). The west basin is mostly surrounded by limestone bedrock whereas the east basin transitions into Precambrian shield, and has much less development in its surrounding watershed relative to the west basin. According to the Ontario Flow Assessment Tool (OFAT) provided by the OMNRF, the surrounding watershed of the west basin has been mostly cleared, with agriculture and other rural land use representing 54.3% of the watershed, and community/infrastructure representing 4.7% of the watershed (excluding open water) (Figure 2). Historically, the township of South Frontenac was settled at the beginning of the 1800s by United Empire Loyalists, with increasing development of farms, factories, mills and mines through the mid-1800s (Rollason 1982). Most of the original timber was cleared for lumber, mining, or farmland by 1880 (Rollason 1982). The small town of Battersea is located on Milburn Creek, the outflow from the Loughborough watershed. In the latter half of the 19th century, a dam was built to run a sawmill, that by 1870 was producing $4400 worth of lumber annually (Rollason 1982). Lumber and mining activities came to an end around the 1890s, and the forest became re-established, however agriculture on the arable land of the limestone plains continued (Rollason 1982). Today, Loughborough Lake is a popular destination for vacation homes and recreational fishing.
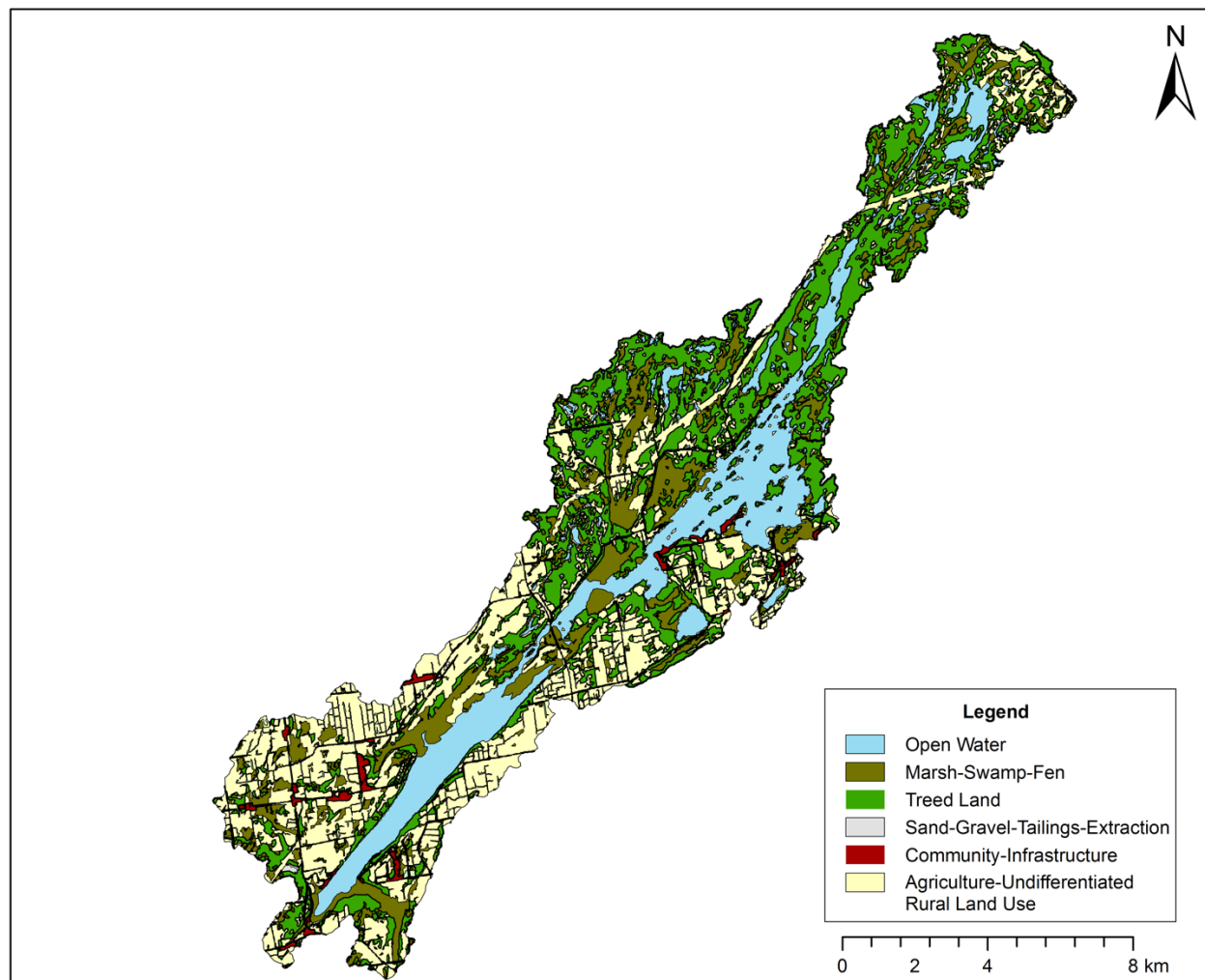
Figure 2: Present-day land-use in the Loughborough Lake watershed. Data was taken from the Ontario Flow Assessment Tool, made available by the OMNRF.

The Ontario Ministry of Environment (OMOE) has conducted sporadic water monitoring on Loughborough Lake since 1987, including measurement of end-of-summer DO concentrations. Recorded temperature profiles show that the thermocline consistently occurred between ~10-13 m depth in all years monitored. In 1998, 2002, and 2005, the hypolimnion was hypoxic for most of its depth, at DO concentrations of below 5 mg/L (Figure 3). In 2009, DO did not drop to consistently below 5 mg/L until after a depth of 25 m (Figure 3). Prior to September 2012, the hypolimnetic DO concentrations at the end-of-summer were consistently below the

optima for Lake Trout of 6-7 mg/L, and dropped below the minimal survival threshold of 4 mg/L in the bottom 10 m between 1998 and 2009 (Figure 3). The average lake TP in 2012 was 7.5 µg/L, which would classify it as an oligotrophic lake (<10 µg/L), however there have been incidences in the past where the TP has increased to the range of a mesotrophic lake (10-20 µg/L). It appears Lake Trout habitat has fallen below optimal in Loughborough Lake in recent history, however without long-term data it remains unknown whether this has been a persistent feature of the lake or a recent phenomenon.
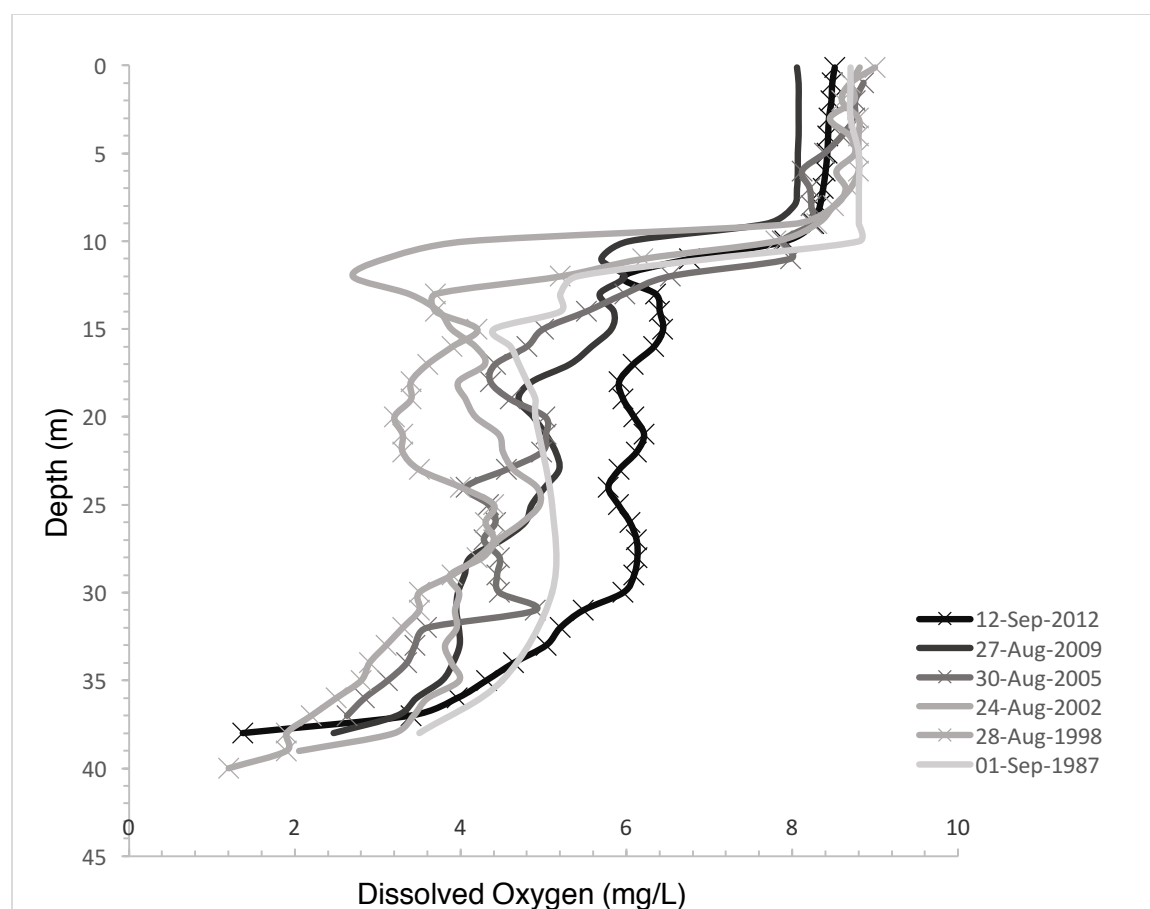
Figure 3: End-of-summer dissolved oxygen concentrations (mg/L) at increasing depth (m) in Loughborough Lake, Ontario, available measurements from 1987 to 2012 from the OMOE and the Loughborough Lake Association.

Paleolimnology – Chironomids as indicators of DO

The direct monitoring data available for Loughborough Lake provides important information on recent limnological conditions; however, the conditions prior industrialization remain unknown. Without historic context for the ecology of Loughborough Lake, it is not possible to ascertain whether recent evidence of sub-optimal Lake Trout habitat is due to anthropogenic eutrophication, climate change, or indicative of 'normal' conditions. To determine pre-disturbance hypolimnetic oxygen conditions in Loughborough Lake, I will apply paleolimnological techniques that use the biological, chemical and physical indicators preserved in dated lake sediments to infer baseline conditions (Smol 2009). Paleolimnological techniques have been used extensively to determine the impact of a wide variety of anthropogenic stresses on aquatic ecosystems, including long-term changes related to climate (Smol 2009).

Chironomids (Family: Chironomidae) are non-biting midges that can be used to reconstruct historical deep-water dissolved oxygen concentrations. Chironomids are one of the most abundant insects found preserved in lake sediments globally (Walker 2001), as their larvae develop in freshwater (four larval instars before emerging as adult flies) with sclerotized chitinous head capsules that preserve well in lake sediments (Walker 2001). The distribution and abundance of different chironomid taxa is dependent on limnological conditions, including oxygen concentration, so the composition of chironomid assemblages preserved in lake sediments reflects the environment present at the time of deposition (Brodersen and Quinlan 2006). Chironomids have physiological, behavioural, or morphological features that make them tolerant to varying levels of dissolved oxygen concentrations. For example, some chironomid taxa have larger body size and high hemoglobin content, and are thus more tolerant to (and competitive at) low levels of DO (Int Panis et al. 1996a). Other adaptations like tube ventilation

and the ability to migrate through the water column can also provide some chironomids an advantage in hypoxic conditions (Int Panis et al. 1996b, Bazzanti et al. 1998).

Initially, interpretation of chironomid assemblages provided qualitative assessments of DO concentrations, but quantitative chironomid-based inference models have been developed to reconstruct background DO conditions, among other environmental variables (Brodersen and Quinlan 2006). The ecological optima and tolerance of taxa can be determined from surface-sediment calibration sets through the association of chironomid assemblages with varying lake environments, both chemical and morphometric (Smol 2009). Transfer functions can then be used to infer past lake conditions from relative chironomid taxa abundances, like DO concentration (Smol 2009). It is important to use large calibration sets because DO concentration can be variable both within and among lakes (Smol 2009). Some of the first transfer functions were developed for the duration of anoxia (Anoxic Factor, AF; Quinlan et al. 1998), and late-summer hypolimnetic oxygen levels (Little and Smol 2001, Quinlan and Smol 2001a). Later, a volume-weighted hypolimnetic oxygen (VWHO) model was developed by Quinlan and Smol (2001a) using chironomid assemblages from the surface sediments of 59 south-central Ontario lakes, along with the late summer DO concentrations of the total hypolimnetic volume of the lakes. This VWHO transfer function can then be applied to the chironomid assemblages from full sediment cores to reconstruct VWHO trends through time, this approach was then validated in an analysis of Peninsula Lake, Ontario (Quinlan and Smol 2001a). More recently, it has been demonstrated that the inclusion of the ratio between *Chaoborus* (Diptera: Chaoboridae) mandibles and chironomid head capsules (chaob:chir ratio) into subfossil chironomid-based VWHO models improves the quality of the inferences in anoxic lakes (Quinlan and Smol 2010). *Chaoborus,* known as the phantom midge, has an aquatic larval form with chitinous mandibles

that remain preserved in lake sediments. *Chaoborus* can oxy-regulate in low DO conditions, allowing them to take refuge in an expanded predator-free zone, resulting in their increased abundance in hypoxic and anoxic hypolimnia (Quinlan and Smol 2010). A chaob:chir calibration set developed by Quinlan and Smol (2010) from the surface sediments of 80 thermally-stratifying shield lakes in south-central Canada has now been added to the Quinlan and Smol (2001a) chironomid training set.

In this study, I will use the chironomid/chaoborid sedimentary assemblages of Loughborough Lake and the VWHO model developed by Quinlan and Smol (2001a, 2010) to assess long-term trends in DO conditions and address the following questions: (1) How have the chironomid sedimentary assemblages and VWHO concentrations in Loughborough Lake changed over the past ~150 years? and (2) Do trends in VWHO concentrations present a concern for management of the current Lake Trout population? Based on the current monitoring data, the history of agricultural development in the watershed, and the potential recent effects of climate change, I expect to find long-term changes in the chironomid assemblages of Loughborough Lake associated with changes in hypolimnetic DO concentrations.

Significance

It is important to understand the background conditions and natural variability of a system to properly assess any observed changes. The paleolimnological reconstruction of hypolimnetic DO concentrations from Loughborough Lake will provide information that will allow lake managers to put modern environmental stressors like climate change, land-use change, and other anthropogenic activities into historical context, and determine their effect on aquatic ecosystem services. If climate change is increasing the likelihood of hypoxic and anoxic

hypolimnetic conditions in Lake Trout lakes, then the protocols and models currently used by managers need to be modified to account for this additional stressor. The proper stewardship of rare Ontario Lake Trout lakes is dependent on the ability to create accurate water quality goals that reflect the effects of DO trends on available Lake Trout habitat. The survival of Lake Trout, a valuable Canadian natural resource, will impact the multi-million-dollar angling and tourism industries, and therefore local and regional economy. The baseline data determined from this study can be used to create future evidence-based policies, and for further research projects that model the present and forecast the future of Lake Trout lakes.

MATERIALS AND METHODS

Sample collection

On October 15, 2014, a sediment core was collected from the deepest point of the west basin (38 meters) of Loughborough Lake using a gravity corer (Glew 1989). The core measured 20.5 cm and was sectioned at 0.5 intervals on-site using a vertical extruder (Glew 1988). Sediment samples were kept in Whirlpak® bags, and then stored in a cold room at Queen's University Paleoecological Environmental Assessment and Research Laboratory (PEARL) prior to analysis.

An EG&G Ortec High Purity Germanium Gamma Spectrometer was used to determine the core chronology by measuring activities of radioisotopes ^{210}Pb and ^{137}Cs. The constant rate of supply (CRS) model was used to calculate ^{210}Pb activities (Appleby 2001). ^{137}Cs was used as an independent marker for the 1963 peak in radioactive fallout following the test-ban treaty of thermonuclear weapons (Appleby 2001). The methods described by Binford (1990) were used to calculate uncertainty, and all procedures and equipment followed methods outlined in Schelske et al. (1994).

Chironomid processing and identification

Processing and identification of chironomid assemblages followed standard methods (Walker 2001). Approximately 2 g of wet sediment were taken from the same intervals that were used to establish the core chronology (every other 0.5 cm for the top 10 cm, followed by 2 cm intervals between 10-20 cm), then deflocculated in 80 mL beakers with a treatment of 5% potassium hydroxide (KOH) and deionized water solution, heated on a hotplate at 80°C for 20 minutes with occasional stirring. Following deflocculation, the samples were sieved through a

100-µm mesh to remove smaller sediment particles. The sample was rinsed thoroughly with

deionized water to remove KOH and debris, and transferred back into the 80 mL beaker to be

stored with some drops of 95% ethanol. Chironomid head capsules and *Chaoborus* mandibles

were picked out of the sample solution using a Bogorov counting tray and fine forceps, making

four passes under a dissecting microscope at ~20x magnification. Isolated midge remains were

mounted on a slide using one drop of Entellan[®] (Electron Microscopy Sciences) per cover slip.

Finally, chironomid and *Choaborus* remains were identified to the lowest possible taxonomic

level, either family, genus, or species level using Brooks et al. (2007) and Anderson et al. (2013)

identification guides. Remains were identified using bright-field optics at 200-400x

magnification on a Leica DM 2500 LED microscope. A minimum count of 40-50 full

chironomid head capsules per interval was the target to properly quantify the sedimentary

assemblages (Quinlan and Smol 2001b).

Downcore data analysis

The relative abundances of taxa within the sedimentary assemblages were applied to a

VWHO model comprising of 44 midge taxa from 59 stratified south-central Ontario lakes

(Quinlan and Smol 2001a, 2010) to infer changes in the VWHO (mg/L) and chaob:chir of

Loughborough Lake through time. The concentration of head capsules within the dry sediment

picked for each interval sample was also determined. The VWHO model was applied using the

"rioja" package (Juggins, 2015) for the R software environment (R Core Team 2017).

Biostratigraphic zones were identified by applying a constrained incremental sum of squares

(CONISS) cluster analysis (Grimm 1987), and the broken stick model (Bennett 1996) to the

assemblage data using the "vegan" package (Oksanen et al. 2016) for the R software environment (R Core Team 2017).

Chlorophyll-a

      To examine trends in past levels of primary production in Loughborough Lake, chlorophyll-*a* concentrations in the lake sediments were spectrally-inferred. Sediment samples from each interval were freeze-dried and sieved through a 125 μm mesh to remove potential sources of interference with the spectral signal. The sediment reflectance spectra obtained using a Visible Near-Infrared Spectroscometer (NIRS) were applied to a linear regression model with a reflectance metric to infer chlorophyll-*a* concentrations (Michelutti *et al.,* 2010).

RESULTS

Chironomid assemblages

Of the 100 different chironomid taxa found in the Loughborough Lake sedimentary assemblages, 17 taxa exceeded a minimum of 3% relative abundance in at least two intervals; only these were included in the stratigraphy for the sake of clarity (Figure 4). Of these 17, there were 5 profundal taxa and 12 littoral identified. *Micropsectra,* with a DO optima of 6.1 mg/L (Quinlan and Smol 2001a), was the most dominant profundal taxon with relative abundances consistently between 20% and 30% (Figure 4). Of the other profundal taxa, *Chironomus* (3.3 mg/L) and *Sergentia coracina* (5.3 mg/L) (Quinlan and Smol 2001a), maintained abundances between ~5-10% throughout most of the core (Figure 4). Of the 12 littoral taxa represented in the reconstruction, *Dicrotendipes nervosus* (4.1 mg/L) and *Tanytarsus lugens* (4.1 mg/L) (Quinlan and Smol 2001a) were the most consistently dominant throughout the core, at relative abundances between ~5-12% (Figure 4). However, *Tanytarsus lugens* dropped to lower relative abundances of <5% between ~1916-1954 and post~1997 (Figure 4).

The CONISS cluster analysis did not indicate any stratigraphic zonation between taxa assemblages throughout the core. However, subtle changes in chironomid taxa relative abundances can be observed between the late 19th century, early 20th century, and late 20th century. In ~1854, *Dicrotendipes nervosus* (4.1 mg/L) experienced an increase in abundance from ~10% to ~16%, while *Micropsectra* (6.1 mg/L) decreased from ~24% to ~13% (Figure 4) (Quinlan and Smol 2001a). *Paratanytarsus,* a littoral taxon, was mostly not present in the core, except between ~1872-1916 where it increased in abundance up to ~12% (Figure 4). *Parakiefferiella nigra*, with a higher DO optima of 8.0 (Quinlan and Smol 2001a), increased from below ~3% relative abundance to ~5-9% between ~1916-1967, before dropping again

thereafter (Figure 4). *Sergentia coracina,* (5.3 mg/L) (Quinlan and Smol 2001a), dropped below

~5% between ~1942-1988. The littoral taxa *Cricotopus/Orthocladius* (3.9 mg/L), *Corynoneura*

edwardsi (4.2 mg/L), *Psectrocladius sordidellus* (4.3 mg/L), *Polypedilum* (4.3 mg/L), and

Tanytarsus latctescens (4.7 mg/L) (Quinlan and Smol 2001a) were all rarely present in the core

until they became more abundant in the late 20th century (post ~1942) (Figure 4). This is

especially notable for *Corynoneura edwardsi,* which went from being very rare in the early

assemblages to being consistently present in relative abundances between ~2-13% post ~1942
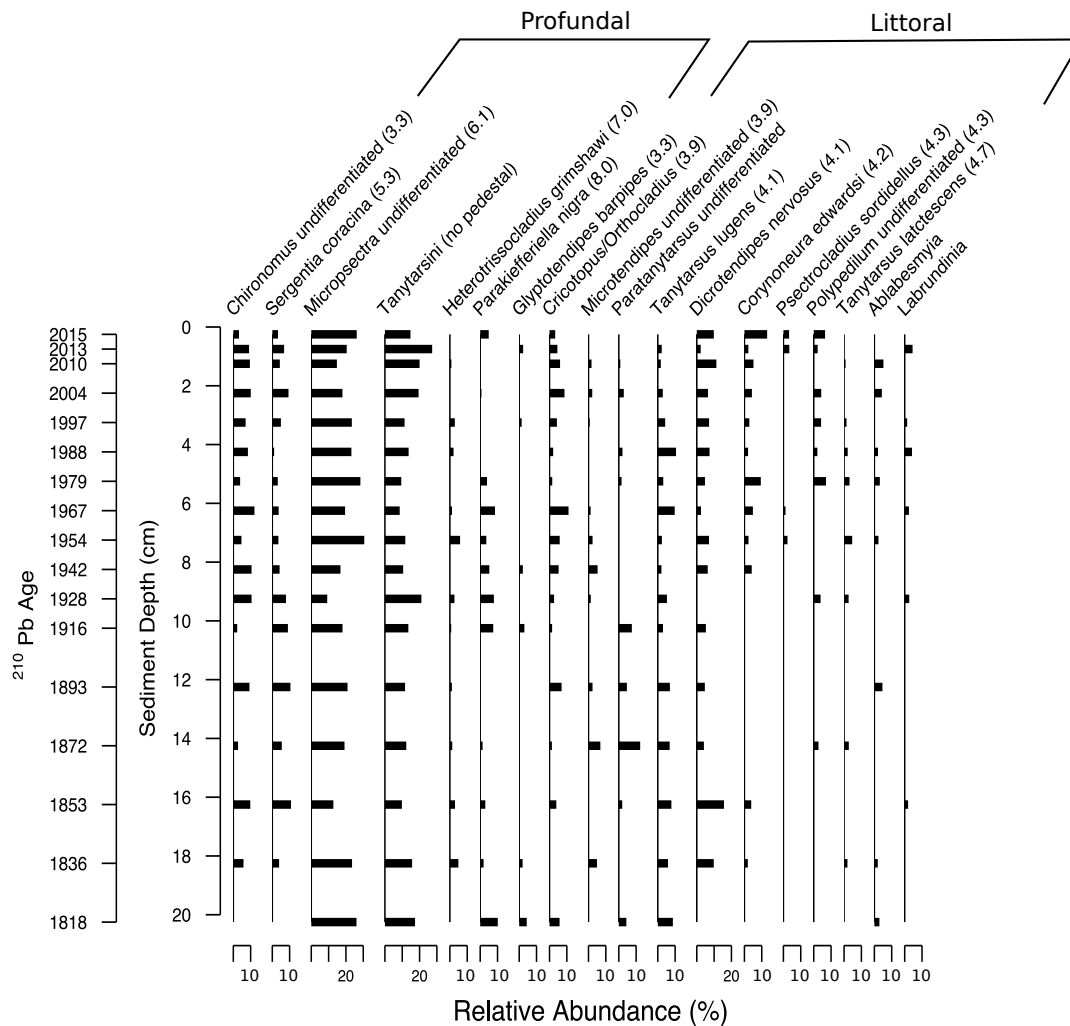
(Figure 4).

Figure 4: Relative abundances of chironomid taxa in the sediment core collected from the west basin of Loughborough Lake. Individual taxa were classified as either profundal or littoral according to Brooks et al. (2007), and ordered within each group according to DO optima indicated in parentheses following the taxon name (Quinlan and Smol 2001a). Some chironomid taxa were grouped to better visualize trends in figures, if they showed similar or no trends and had similar life history strategies.

Chironomid-inferred VWHO

      Tanytarsini (no pedestal) represented a large proportion of the total count for several intervals, and was included in the VWHO inference model as *Micropsectra*, which was the most consistently abundant *Tanytarsini* taxon encountered (~3 times more abundant than all other *Tanytarsini* taxa combined). The ratio of chironomid:chaoborid remains in the sediments was uniform throughout the core, as only one chaoborid mandible was found. The chironomid head capsule count did not meet the target count of 40 in four of the 17 analyzed sediment intervals (0.0-0.5, 9.0-9.5, 14-14.5, and 20.0-20.5), while a count between 40 and 70.5 was achieved in the others (Figure 5). The chironomid-inferred VWHO is relatively 6-7 mg/L throughout the reconstruction (Figure 5). However, there are some slight changes: in ~1818 VWHO increased to 8.6 mg/L, and in ~1853 it decreased to just below 6.0 mg/L (Figure 5). Between ~1915-1954 VWHO increased to 7.3-7.8 mg/L, while in ~1941 it decreased again just below 6.0 mg/L (Figure 5).
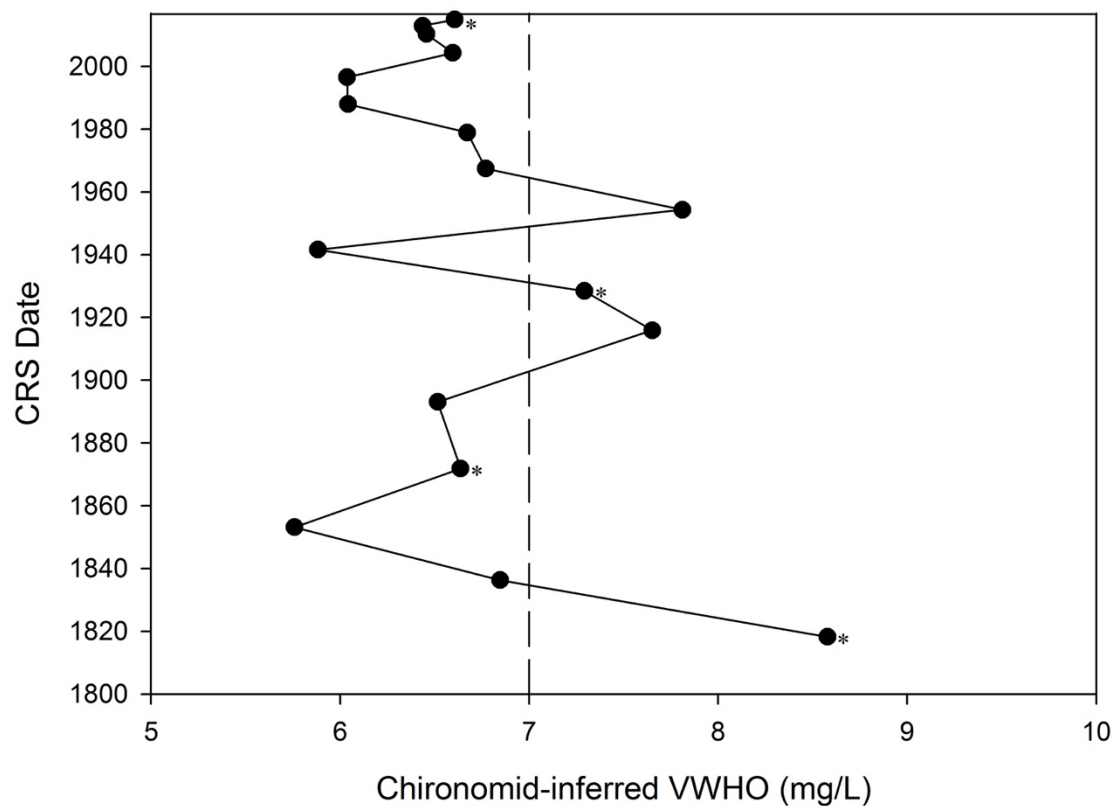
Figure 5: Chironomid-inferred VWHO for Loughborough Lake using the Quinlan and Smol (2001a, 2010) VWHO inference model. Constant Rate of Supply (CRS) dates indicated. Points highlighted in white come from total chironomid counts of fewer than 40 whole head capsules (30.5, 38, 36.5, and 34.5 from top to bottom). The dashed line indicates the Lake Trout DO optima of 7 mg/L.

Chlorophyll-a

The VRS-inferred chlorophyll-*a* profile has three major periods of change throughout the reconstruction. In the mid to late 19th century, there was a distinct peak in chlorophyll-*a* dry mass (Figure 6). During this same period of ~1845 to ~1890, chironomid-inferred VWHO experienced a decline (Figure 6). This was followed by a period of chlorophyll-*a* decline between ~1905 to

~1980, during which chironomid-inferred VWHO peaked (Figure 6). Lastly, chlorophyll-*a* once

again experienced an increase in dry mass at ~1980 to ~2000, at which time chironomid-inferred

DO declined to moderate values (Figure 6).
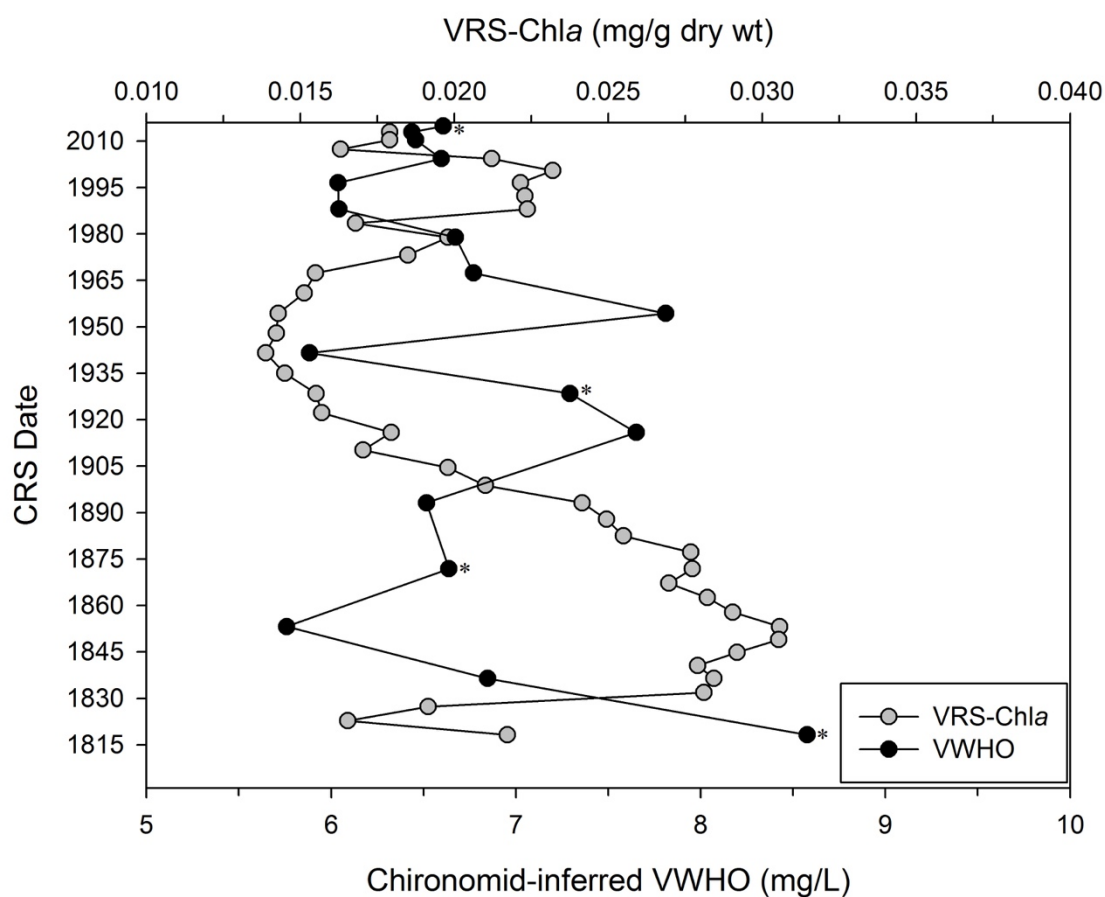
Figure 6: VRS-inferred chlorophyll-*a* plotted against chironomid-inferred VWHO for

Loughborough Lake. Constant Rate of Supply (CRS) dates indicated. Points highlighted with an

asterisk come from total chironomid counts of below 40 whole head capsules.

DISCUSSION

The chironomid-inferred VWHO reconstruction of Loughborough Lake revealed that over the past ~200 years DO concentrations have relatively been stable at ~6-7 mg/L, optimal conditions for Lake Trout habitat. Stable conditions have persisted through pre-settlement and modern history, with indications of a slight decline in VWHO during the 1800s, followed by a recovery period in the 1900s. Changes in human activity in the watershed throughout history may account for variations in chironomid assemblages, including periods of increased forestry, agriculture, and settlement. Modern climate warming may influence future lake conditions; however, these results do not suggest an imminent need for increased management of DO within Loughborough Lake.

Chironomid assemblages

Chironomids of the genus *Micropsectra* dominated the profundal taxa assemblages, with relative abundances between ~20-30% throughout the core (Figure 4). This genus is indicative of high oxygen levels with an optimum of ~6.06 mg/L (Quinlan and Smol 2001a). *Micropsectra* are typical of mesotrophic-oligotrophic lakes, and many are cold stenotherms, meaning that they function within a narrow cold temperature range (Brooks et al. 2007). Their dominance throughout the core indicates that hypolimnetic DO has likely not entered periods of prolonged hypoxia during the past ~200 years, but has rather stayed between moderate values.

During the mid to latter half of the 19th century, there were slight changes in chironomid assemblages that may indicate a decrease in the quality of hypolimnetic DO. In ~1853, *Micropsectra* declined from ~24% to ~13% relative abundance, while *Dicrotendipes nervosus,* which is indicative of low oxygen levels, increased from ~10% to ~16% relative abundance

(Figure 4). *Dicrotendipes nervosus* is often associated with macrophytes in littoral zones, and is typical of mesotrophic to eutrophic water bodies (Brooks et al. 2007). Another littoral taxon that is often associated with macrophytes is *Paratanytarsus*, which increased in relative abundance between ~1872 to ~1916 (Figure 4).

Parakiefferiella nigra is an indicator of high oxygen, typical of oligotrophic lakes, with an optimal VWHO of 7.95 mg/L (Quinlan and Smol 2001a). Its relative abundance was very low, between 0-2% early and late in the core; however, it had a distinct period of increased abundance up to 8.5% between ~1916-1967 (Figure 4). This might be a recovery trend; at the bottom-most interval, its relative abundance was ~10% (Figure 4), but then dropped off during a potential period of decreasing hypolimnetic DO, and recovered back to higher relative abundances during a potential period of increasing hypolimnetic DO. Its subsequent disappearance from the assemblages post ~1988 may be indicative of decreasing hypolimnetic DO concentrations in recent history.

Littoral taxa are important indicators of lake processes despite not being direct indicators of hypolimnetic conditions. During seasonal changes and the natural movement of water throughout a lake, there is a re-deposition of sediment from littoral to profundal zones. Taking a core from the deepest point in the lake not only captures profundal species abundances but also captures littoral and pelagic communities. An increase in the relative abundance of littoral taxa in the sediment can be indicative of diminishing profundal communities in response to deteriorating DO concentrations, especially periods of anoxia (Clerk et al. 2000).

Loughborough Lake does not appear to have experienced periods of prolonged anoxia; however, chironomid assemblages reveal that several littoral taxa experienced increases in relative abundance post approximately 1942 (Figure 4). *Corynoneura edwardsi* went from being

very rare in the early assemblages to being consistently present in relative abundances at between ~2-13% post ~1942 (Figure 4). This species is commonly associated with temperate lakes with abundant macrophytes in the littoral zone (Brooks et al. 2007). The collective group *Cricotopus/Orthocladius* experienced a similar change in relative abundance to *Corynoneura edwardsi* (Figure 4). This group of chironomids are eurytopic (able to withstand diverse habitat conditions), but tend to be associated with eutrophic conditions and found amongst macrophytes (Brooks et al. 2007). Similarly, the group *Polypedilum,* which is found in eutrophic lakes of regions with a temperate climate, started to increase in abundance post ~1979 (Figure 4) (Brooks et al. 2007). This observed increase in littoral relative abundance during the second half of the 20th century may be indicative of increasing nutrients in the lake that is more favourable for littoral species than profundal.

Chironomid-inferred VWHO and chlorophyll-a

The Loughborough Lake VWHO reconstruction revealed that hypolimnetic oxygen conditions have persisted at ~6-7 mg/L over the past 200 years (Figure 5). However, there were subtle changes that show a potential recovery trend, most likely from human disturbance in the watershed. The reconstruction of VWHO and chlorophyll-*a* exhibit concurrent changes, with the late-1800s experiencing a period of increased production and slight decrease in VWHO, immediately followed by the reverse trends in the mid 1900s (Figure 6). The decrease in chironomid-inferred VWHO to below 6 mg/L during the 1850s (Figure 5) coincides with the period of peak forestry in the watershed between 1850-1880, when early settlers were taking advantage of the virgin forests for logging (Rollason 1982). Logging would have elevated erosion and runoff into the lake, in turn increasing nutrient levels, corroborated by the

chlorophyll-*a* reconstruction indicating an increase in production during roughly the same period of ~1820-1890 (Figure 6). Thus, the intense logging may be have resulted in a period of degraded hypolimnetic water quality.

By the early 1900s, most of South Frontenac was deforested, and the demand for logging activity from northern mining industries was ending (Rollason 1982). This might have led to a recovery period in the watershed as forests became re-established, and hypolimnetic DO concentrations increased in quality. The chironomid-inferred VWHO increased to a peak of ~7.3-7.8 between ~1920-1960 (Figure 5). Of note is a drop in VWHO down to 5.9 in ~1941 (Figure 5), that may be associated with temporary decreases in the abundances of taxa indicative of high VHWO conditions such as *Micropsectra* and *Tanytarsini* (no pedestal), being replaced by taxa better suited to low VWHO conditions such as *Chironomus* and *Dicrotendipes nervosus* (Figure 5). It is unclear, whether this single sample is indicative of a regression in the recovery or an outlier, as the inferred chlorophyll-*a* values indicate a continuous decrease between ~1920-1980 (Figure 6).

Agricultural intensity would have increased throughout the 20th century, especially with advances in technology available, including chemical fertilizers and improved machinery. Agriculture and other rural land use makes up over 50% of the current land-use distribution of the Loughborough Lake watershed (Figure 2). The increase in chlorophyll-*a* from ~1980-2000 might be associated with increasing nutrient load from agricultural activity in the watershed, which may also be why the VWHO has since decreased from its 1960s peak (Figure 5).

Climate Change

While agricultural activity in the watershed may be associated with increasing nutrient input to the catchment, the effects of climate change might also be affecting recent changes in chironomid assemblages. An increase in littoral species abundance might be driven by warmer autumnal air temperature, and the associated warmer autumnal water temperature, and a later ice over date (Palmer et al. 2014). Temperature is closely linked with lake productivity (trophic status), so chironomids would be responding to climate-induced changes in productivity not in temperature itself (Brodersen and Quinlan 2006). As the epilimnion and metalimnion become warmer, and nutrient production subsequently increases, favourable conditions for littoral species may occur, like increased macrophyte growth and lower DO concentrations. With warmer surface temperatures comes increasingly stabilized lake stratification, further isolating the hypolimnion, prolonging the period for DO consumption before lake turnover (Palmer et al. 2014). This might put stress on profundal communities, including profundal chironomid assemblages with high DO optima. However, direct effects of temperature on chironomids may be hard to distinguish from other changes in lake processes (Brodersen and Quinlan 2006).

Limitations of the study

The inability to distinguish taxa of *Tanytarsini* due to missing pedestals and other head capsule components creates a degree of uncertainty in interpreting chironomid assemblages. Their large proportion represented in the down-core assemblages of Loughborough Lake may be affecting inferred VWHO. Furthermore, several of the intervals throughout the core were below the target count of 50 head capsules. Caution in the interpretation of these values should be exhibited, especially for those below 40. As total counts decrease, each additional chironomid

has a greater effect on the inferred-VWHO, meaning rare outlier taxa would have a greater weight. Continued data collection to top up these intervals could be completed to improve the quality of the reconstruction. Lastly, a multi-proxy approach would strengthen any interpretations of chironomid inferences by providing additional information on historic lake conditions, creating a more complete understanding of environmental change. The investigation of diatom assemblages is currently underway, which are used as indicators of total phosphorus, the limiting nutrient in most Ontario Lake Trout lakes (Smol 2009).

Conclusion: Management significance

The paleolimnological reconstruction of past oxygen conditions in Loughborough Lake indicate that good Lake Trout habitat was a persistent feature of the last ~200 years, despite a period of slight deterioration and recovery coincident with initial land clearance of the watershed in the 19th century. Current monitoring data indicates that end-of-summer hypolimnetic DO concentrations have been either hypoxic or near hypoxic between 1987 and 2009 (Figure 3). Increased relative abundances of littoral taxa since the mid-1900s may be reflective of a deteriorating hypolimnion as indicated by the monitoring data; however, profundal taxa indicative of high DO remain prevalent throughout this period. These results do not present an immediate concern for the quality of Lake Trout habitat, as a considerable downward trend in hypolimnetic DO concentrations is not apparent. Loughborough is a large, deep lake of 38 m, with a large hypolimnetic volume, so Lake Trout have a large habitat to support them. Because this is a stocked population, it is important to assess whether the lake has the capacity to maintain abundant recruits between years so that the government is not investing money into a futile effort. The nuanced indications from the chironomid assemblages of possible stress on the

hypolimnetic DO may present the need for future management efforts on maintenance of water

quality, including the control of agricultural runoff. These results highlight the value of

paleolimnological indicators by providing some long-term context to evaluate recent conditions.

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APPENDIX 1.

Table A.1. Loughborough Lake chironomid-inferred VWHO model output.

Interval (cm)	Midpt (cm)	Age	VWHO (mg/L)	Proportion inference.taxa	Lake chaob.chir	chir.total
X0.0.0.5	0.25	2014.86	6.61	100	0	30.5
X0.5.1.0	0.75	2012.92	6.44	95.45	0	44
X1.0.1.5	1.25	2010.38	6.46	98.26	0	57.5
X2.0.2.5	2.25	2004.31	6.60	100	0.007	69.5
X3.0.3.5	3.25	1996.51	6.04	92.91	0	70.5
X4.0.4.5	4.25	1988	6.04	95.79	0	47.5
X5.0.5.5	5.25	1978.96	6.67	100	0	63.5
X6.0.6.5	6.25	1967.39	6.77	97.56	0	41
X7.0.7.5	7.25	1954.34	7.81	100	0	42.5
X8.0.8.5	8.25	1941.58	5.88	93.68	0	47.5
X9.0.9.5	9.25	1928.46	7.29	89.47	0	38
X10.0.10.5	10.25	1915.92	7.65	93.23	0	66.5
X12.0.12.5	12.25	1893.12	6.52	97.67	0	43
X14.0.14.5	14.25	1871.84	6.64	97.26	0	36.5
X16.0.16.5	16.25	1853.14	5.76	96.078	0	51
X18.0.18.5	18.25	1836.38	6.85	98.04	0	51
X20.0.20.5	20.25	1818.22	8.58	91.30	0	34.5

APPENDIX 2.

Table A.2. The total chironomid count data from Loughborough Lake for each interval down-core. Taxa and counts shaded in grey indicate sections of taxa that were grouped together for the stratigraphy.

	Loughborough Lake	Interval	0.0-0.5	0.5-1.0	1.0-1.5	2.0-2.5	3.0-3.5	4.0-4.5	5.0-5.5	6.0-6.5	7.0-7.5	8.0-8.5	9.0-9.5	10.0-10.5	12.0-12.5	14.0-14.5	16.0-16.5	18.0-18.5	20.0-20.5
y		Midpt (cm)	0.25	0.75	1.25	2.25	3.25	4.25	5.25	6.25	7.25	8.25	9.25	10.25	12.25	14.25	16.25	18.25	20.25
		Age	2014.86	2012.92	2010.38	2004.31	1996.51	1988	1978.96	1967.39	1954.34	1941.58	1928.46	1915.92	1893.12	1871.84	1853.14	1836.38	1818.22
C	Chironomus anthracinus-type	CHIRON		6	2	8	6	2	3	2	4	4	4	3	2		2		
C	Chironomus plumosus-type	CHIRON		2	9	6	4	2	2	8		6	4		6		2	6	6
C	Chironomus spp.	CHIRON	2				4										2		
C	Cladopelma lateralis-type	CLADOP	2	2	2			2	3					2	2		2	2	2
C	Cryptochironomus	CRYPCH								1		2			3				
C	Demicryptochironomis													1					
C	Dicrotendipes nervosus-type	DICROT	6	2	13	9	10	7	6	2	6	6		7	4	3	16	10	
C	Dicrotendipes notatus-type	DICROT															2	4	
C	Endochironomus albipennis-type	ENDOCH													4			2	
C	Endochironomus tendens-type	ENDOCH				2													
C	Einfeldia dissidens-type	GLYPTO						2											
C	Glyptotendipes barpipes-type	GLYPTO		2		2								2	4			2	3
C	Glyptotendipes severini-type	GLYPTO				2	2												
C	Glyptotendipes pallens-type	GLYPTO		2	1									4		2	2		
C	Lauterborniella						2							2					
C	Microchironomus	MICROC					2											2	
C	Microtendipes pedellus-type	MICROT				3						2						2	
C	Microtendipes spp.	MICROT			2		1			1		5	1			2	5	3	
C	Pagastiella	PGASTL								2				2					
C	Parachironomus varus-type	PARACH				2													
C	Paratendipes albimanus-type	PARATE	4					2				2							2
C	Paratendipes nudisquama-type										1								
C	Phaenopsectra type A	SERGEP					2												
C	Polypedilum nubeculosum-type	POLYPE	2			6	2		2										
C	Polypedilum nubifer-type	POLYPE	2	2			2	2	5				3				2		
C	Polypedilum sordens-type	POLYPE					2		2										
C	Sergentia coracina-type	SERGEN	2	6	5	13	7	1	4	3	3	4		12	9	4	11	4	
C	Sergentia longiventris-type	SERGEN															2	2	
C	Stictochironomus rosenschoeldi-type	STICTO							2			2							
C	Stictochironomus type B	STICTO			1	1								1			2	2	
C	Tribelos	TRIBEL													1				
C	Xenochironomous												2						
O	Chaetocladius dentiforceps-type	CHAETO						4	4				1						
O	Chaetocladius piger-type	CHAETO			2														1
O	Corynoneura carriana-type	CORYTH			2			4									2		
O	Corynoneura edwardsi-type	CORYTH	8	2	6	6	4	2	12	4	2	4					4	2	
O	Corynoneura Type A								2			2							
O	Corynoneura spp.	CORYTH			4							2							
O	Cricotopus spp.	CRICOR				2	2										1	2	
O	Cricotopus bicinctus-type			1	2	2				2		1	2						
O	Cricotopus (Isocladius) sylvestris-type	CRICOR												2					
O	Cricotopus (Isocladius) type C	CRICOR										2							1
O	Cricotopus cylindraceus-type	CRICOR			2							2			2	4			2
O	Cricotopus intersectus-type	CRICOR			4	4	2			7		3						2	
O	Cricotopus trifascia-type	CRICOR								2									
O	Cricotopus type B		1																
O	Eukiefferiella claripennis-type	EUKIEF					2												
O	Eukiefferiella fittkaui-type								2										
O	Heterotrissocladius grimshawi-type	HTRTRS			1		4	2		1	5		2	1	1	1	3	5	

Group	Taxon	Code	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
O	Heterotrissocladius maeaeri-type	HTRTRS																	
O	Heterotrissocladius marcidus-type	HTRTRS		1	2	2			1				1	2				2	
O	Limnophyes-Paralimnophyes	LIMNOP										1		4	2				1
O	Metriocnemus terrester										2					2			2
O	Nanocladius branchicolus-type	NANOCL		2			2		2			2	2				2		
O	Nanocladius rectinervis			2															
O	Orthoclad (indistinguishable)	ORTHSP			1		2	2								2			1
O	Orthocladius oliveri-type	CRICOR				2													
O	Orthocladius (Pognocladius) consabrinus-type	CRICOR				2													
O	Orthocladius Type S					2													
O	Paracricotopus	PSDSMA					2						2						
O	Parakiefferiella bathophilia-type	PARAKA			1														
O	Parakiefferiella nigra-type	PARAKA	3		3	1			5	7	3	5	6	10		1	3	2	7
O	Parakiefferiella spp.					2													
O	Psectrocladius (Mesopsectrocladiu) barbatipes-type	PSECTA																2	
O	Psectrocladius (Psectrocladius) sordidellus-type	PSECTP	2	3						1	2								
O	Pseudorthocladius	PSEDOR					2					2							2
O	Pseudosmittia							1											
O	Smittia-Parasmittia	PSDSMA		2			2												
O	Thienemanniella type E							2											
O	Thienemanniella clavicornis-type	CORYTH										2							
P	Pseudochironomus	PSEDCH						3							1				1
Tanyp	Conchapelopia														2				
Tanyp	Ablabesmyia				2	6		2	4		2				4			2	2
Tanyp	Derotanypus	DEROTA																	
Tanyp	Djalmabatista												2						
Tanyp	Guttipelapia	GUTTIP					2												
Tanyp	Labrundina	LABRUN		4			2	4					2					2	
Tanyp	Procladius	PROCLD									2				2				
Tanyt	Cladotanytarsus mancus-type 1	CLDTYM							1							2		2	
Tanyt	Cladotanytarsus mancus-type unknown	CLDTYM															2		
Tanyt	Corynocera oliveri-type	CORCER					2			3	2	2	1	2	2				
Tanyt	Micropsectra spp.		2									2							
Tanyt	Micropsectra contracta-type	MICROP	3	4	3	11	4	6	2	2	2	4		1	4			2	
Tanyt	Micropsectra insignilobus-type	MICROP	7	6	7	6	8	10	6	6	13	6	5	15	7	5	8	4	6
Tanyt	Micropsectra junci-type	MICROP																	
Tanyt	Micropsectra pallidula-type	MICROP						2											
Tanyt	Micropsectra radialis-type	MICROP	4	8	7	8	21	4	28	8	9	6	2	8	7	9	5	18	12
Tanyt	Neozavrelia	NEOZAV									2	2	4	2					
Tanyt	Paratanytarsus spp.	MICROP			1	4		2	2					10	4	9	2		3
Tanyt	Paratanytarsus austriacus-type	MICROP										4							
Tanyt	Paratanytarsus pencillatus-type	MICROP											2						
Tanyt	Paratanytarsus type A																	2	
Tanyt	Rheotanytarsus	RHEOTA																	2
Tanyt	Stempellina-Zavrelia	STMPLL													1				
Tanyt	Tanytarsini (no pedestal)	TANYSL	9	24	23	27	16	13	12	7	10	10	16	18	10	9	10	16	12
Tanyt	Tanytarsus glabrescens-type	TANYSL			2								2	4					
Tanyt	Tanytarsus latctescens-type	TANYSL			1		2	2	4		4		2			2		2	
Tanyt	Tanytarsus lugens-type	TANYLU		2	2	4	6	10	4	8	2	2	4	4	6	5	8	6	6
Tanyt	Tanytarsus mendax-type	TANYSL						2		2									
Tanyt	Tanytarsus pallidicornus-type 1	TANYSL								2			2						2
Tanyt	Tanytarsus spp. (no mandible)	TANYSL	1		2	2	3			2		2		2			2		1
	Caoborus mandible	C-FLAV				1													

APPENDIX 3.

Table A.3. Loughborough Lake VRS-inferred chlorophyll-*a* in mg/g dry mass.

Core Depth(cm)	VRS-chla (mg/g dry wt)
0.75	0.018
1.25	0.018
1.75	0.016
2.25	0.021
2.75	0.023
3.25	0.022
3.75	0.022
4.25	0.022
4.75	0.017
5.25	0.020
5.75	0.018
6.25	0.016
6.75	0.015
7.25	0.014
7.75	0.014
8.25	0.014
8.75	0.015
9.25	0.016
9.75	0.016
10.25	0.018
10.75	0.017
11.25	0.020
11.75	0.021
12.25	0.024
12.75	0.025
13.25	0.026
13.75	0.028
14.25	0.028
14.75	0.027
15.25	0.028
15.75	0.029
16.25	0.031
16.75	0.031
17.25	0.029
17.75	0.028
18.25	0.028
18.75	0.028
19.25	0.019
19.75	0.017
20.25	0.022

APPENDIX 4.

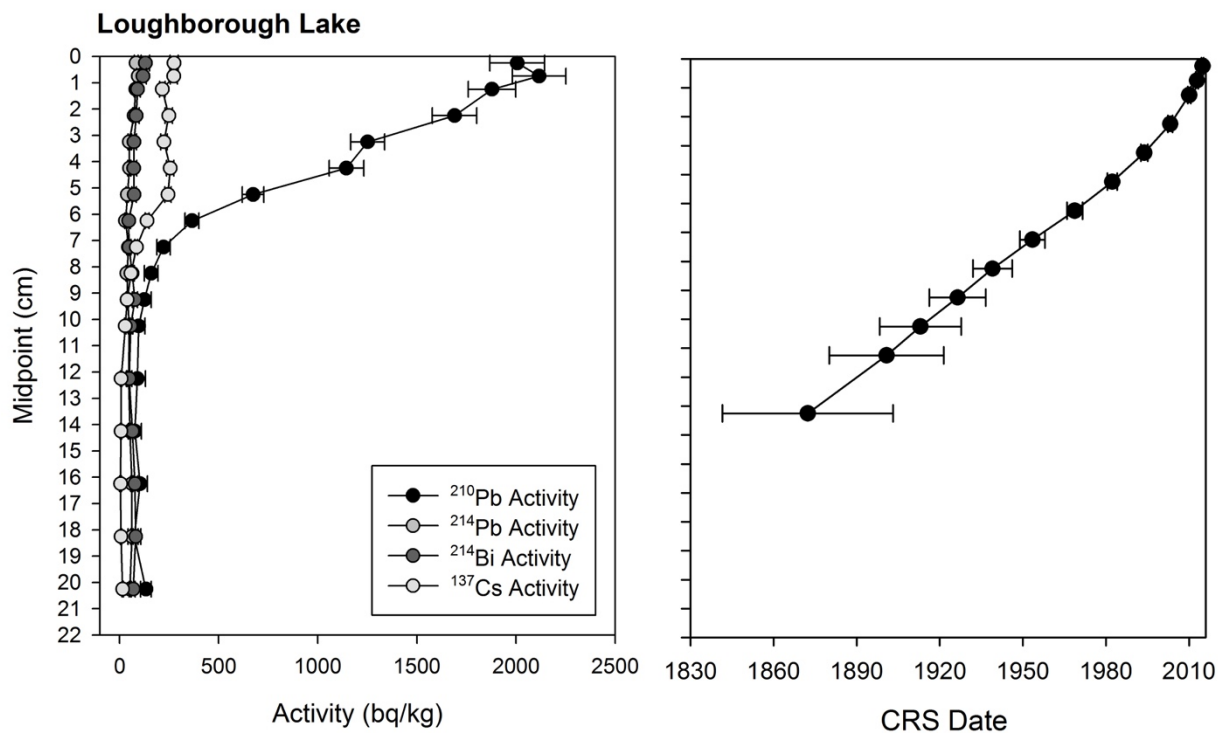Figure A.1. The radioactivity of unsupported ^{210}Pb, ^{214}Pb and ^{214}Bi (for inferring supported

^{210}Pb), and ^{137}Cs, as well as associated error of Loughborough Lake, Ontario, Canada on the left.

The Constant Rate of Supply (CRS) derived dates and associated error on the right.