Assessing nearshore water quality and biological condition in the Kawartha Lakes using a community science approach

by

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An oral defense of this thesis took place on August 18, 2022 in front of the following examining committee:

Examining Committee:

Chair of Examining Committee	Dr. Dario Bonetta
Research Supervisor	Dr. Andrea Kirkwood
Examining Committee Member	Dr. Denina Simmons
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The above committee determined that the thesis is acceptable in form and content and that a satisfactory knowledge of the field covered by the thesis was demonstrated by the candidate during an oral examination. A signed copy of the Certificate of Approval is available from the School of Graduate and Postdoctoral Studies.

ABSTRACT

The Kawartha Lakes, located in south-central Ontario, are a popular tourist destination, with a growing permanent resident population. Consequently, land development in its watersheds continues, and the specific land-use – water quality relationships in this region are unknown. The nearshore zone is where land use has its first impacts on the lake, and provides vital habitat for most lake inhabitants at some point in their life cycle. Despite its importance, the nearshore zone is rarely monitored and further investigation is required to understand human impacts on the nearshore zone. My thesis aimed to elucidate the relationships between land use and the abiotic and biotic condition of the nearshore zone in the Kawartha Lakes. To examine Lake Scugog's nearshore water quality patterns and relationship with land use, 12 volunteers collected water samples from spring to fall for three years. Land use had significant impacts on chloride at buffer scales and phosphorus at the sub-watershed scale. I also monitored the nearshore biotic community (phytoplankton, zooplankton, and macroinvertebrates) at eight sites in Lake Scugog. Two years of sampling found that macrophyte abundance significantly influenced the phytoplankton, zooplankton, and macroinvertebrate communities. During the pandemic restrictions in 2020, community scientists on 16 lakes collected monthly water samples from June-September. A subset of four lakes had nutrient data across 3-years (2019 - 2021), which allowed comparison of nutrient conditions before and during the pandemic. There were significant differences in water quality between watersheds and a notable impact of Lake Scugog on downstream lakes. There was not a significant impact of pandemic restrictions on nearshore water quality in these lakes. A focal study on Balsam, Cameron, Sturgeon, and Pigeon lakes involved nearshore water quality

monitoring (2019 and 2021) and biological sample collection (2021). There was a separation of distinct water quality profiles that grouped Balsam and Cameron, and Sturgeon and Pigeon. Exploring relationships between land use, water quality, and the biotic communities I found that phosphorus was important for driving phytoplankton, zooplankton, and macroinvertebrate community abundance. Overall, these findings provide important information for lake managers in understanding the role of land-use and nearshore ecological condition in lake health.

Keywords: nearshore zone; water quality; aquatic community; community science; land use

AUTHOR'S DECLARATION

I hereby declare that this thesis consists of original work of which I have authored. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I authorize the University of Ontario Institute of Technology (Ontario Tech University) to lend this thesis to other institutions or individuals for the purpose of scholarly research. I further authorize University of Ontario Institute of Technology (Ontario Tech University) to reproduce this thesis by photocopying or by other means, in total or in part, at the request of other institutions or individuals for the purpose of scholarly research. I understand that my thesis will be made electronically available to the public.

The research work in this thesis that was performed in compliance with the regulations of Research Ethics Board/Animal Care Committee under **REB Certificate #15910,** originally approved in May 2020.

Erin Smith

Erin Smith

STATEMENT OF CONTRIBUTIONS

Chapter 2 was published in the Lake and Reservoir Management as: Smith, E. D., Balika, D., & Kirkwood, A. E. (2021). Community science-based monitoring reveals the role of land use scale in driving nearshore water quality in a large, shallow, Canadian lake. *Lake and Reservoir Management*, *37*(4), 431-444. https://doi.org/10.1080/10402381.2021.1989525 Work performed in Chapter 2 was performed in partnership with community science volunteers recruited and trained by Kawartha Conservation who collected water samples. I conducted laboratory analysis, curated and analyzed the data, and wrote the manuscript. Content was altered to fit the thesis formatting.

The work described in Chapter 3 has been submitted to the journal Fundamental and Applied Limnology. I was responsible for collecting field samples, conducting laboratory analysis, curating and analyzing data, and writing the manuscript. Content was modified to fit thesis formatting.

The work described in Chapter 4 was conducted with the help of volunteers recruited through Kawartha Conservation and the Kawartha Lakes Conservation Authority (KLSA), volunteers were trained by myself and collected monthly water samples. I performed laboratory analysis, and curated and analyzed the data.

The work described in Chapter 5 was completed with the help of community science volunteers. Volunteers collected water samples in 2019 and 2020. I collected water samples and the biotic community samples in 2021. I was responsible for all laboratory analysis and curated and analysed the data.

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(Appendix B). TP = total phosphorus, TN = total nitrogen, TSS = total suspended solids. Phyto. = phytoplankton, Zoop. = zooplankton, Invert. = macroinvertebrate. 156

LIST OF ABBREVIATIONS AND SYMBOLS

%	Percent
º	Degrees
α	Alpha
µg	Micrograms
µm	Micrometers
µS	Micro Siemens
AIC	Akaike Information Criterion
ANOVA	Analysis of variance
Chla	Chlorophyll a
Chlor	Chloride
Cond	Conductivity
DCA	Detrended correspondence analysis
DO	Dissolved oxygen
EPT	Ephemeroptera + Plecoptera + Trichoptera
FBI	Hilsenhoff's Family biotic index
FD	Functional diversity
GIS	Geographic information systems
GLLVM	General linear latent variable model
IQR	Interquartile range
km	Kilometer
L	litres
m	Meter
MANOVA	Multivariate analysis of variance
mg	Milligrams
mL	Millilitres
mm	Millimeters

MLR	Multiple linear regression
MPN	Most probable number
NH3	Ammonia
NH4	Ammonium
nm	Nanometer
NMDS	Non-metric multidimensional scaling
NO2	Nitrite
NO3	Nitrate
PCA	Principal component analysis
PERMANOVA	Permutational analysis of variance
PWQO	Provincial water quality objective
R^2	Coefficient of determination
RDA	Redundancy analysis
SEM	Structural equation model
Spp.	Species
TDP	Total dissolved phosphorus
Temp	Temperature
TN	Total nitrogen
TON	Total organic nitrogen
TP	Total phosphorus
TSS	Total suspended solids
TSW	Trent Severn Waterway
Tur	Turbidity
VIF	Variance inflation factor
x	Times

Chapter 1. **Introduction**

1.1 The Kawartha Lakes

The Kawartha Lakes are a chain of 13 lakes located in southern Ontario, stretching across Victoria and Peterborough counties (Figure 1.1). These lakes were formed by the final retreat of the Laurentide Ice Sheet approximately 12,000 years ago (Todd & Lewis, 1993). The movements of the ice sheet as it retreated formed The Land Between, a unique geologic transition zone with limestone-dominated bedrock in the St Lawrence Lowlands to the south and granite-dominated Canadian Shield to the north (Marich, 2016). The Land Between represents not only a geologic transition but also a transition of physiography, climate, and elevation creating an ecotone, a unique habitat that can support high biodiversity (Goldblum & Rigg, 2010; Risser, 1990). Located in The Land Between, the Kawartha Lakes are not only an ecologically interesting study region, they are also import economically for tourism, which brings in over $109 million per year (City of Kawartha Lakes, 2021). Many people are drawn to the area every year for recreational activities on the water and to enjoy the rural and natural landscapes of the area.

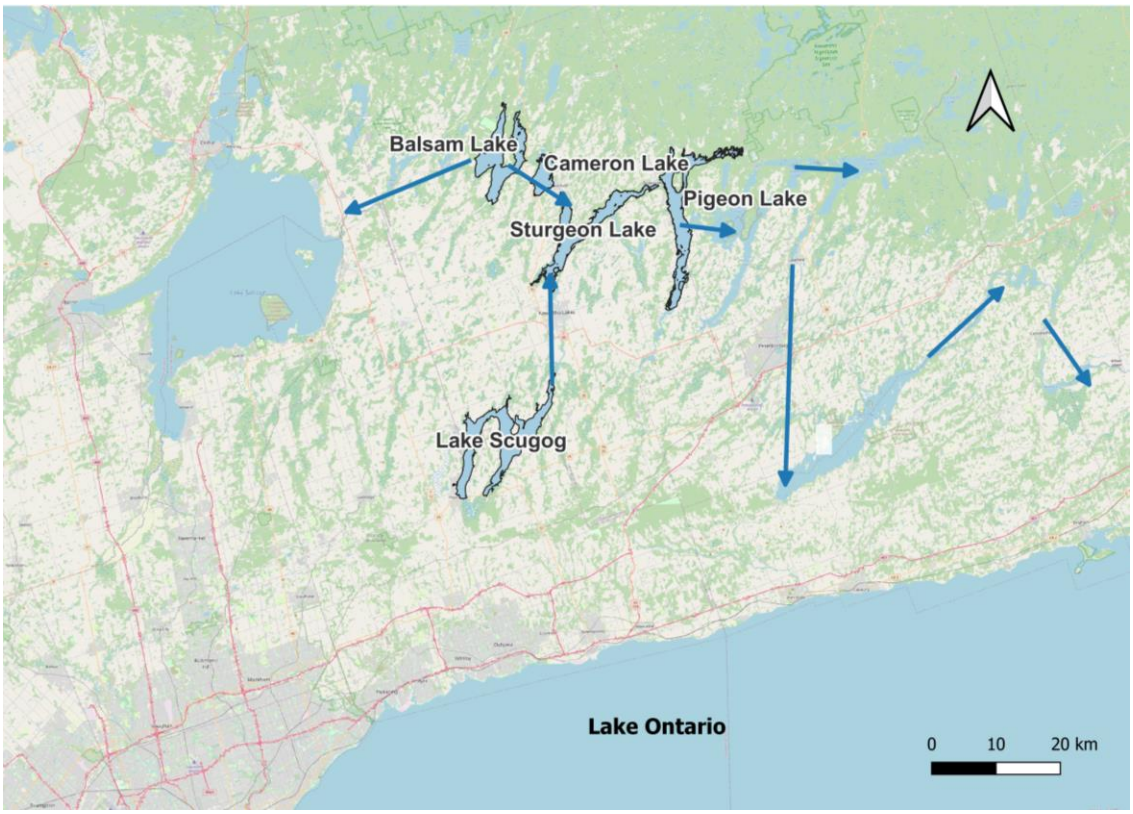

Figure 1.1. Regional map of primary study lakes: Balsam, Cameron, Scugog, Sturgeon, and Pigeon. Arrows indicate the direction of water flow through the Trent-Severn Waterway.

For thousands of years before European settlement, Indigenous peoples, including the Mississauga, Anishinabewaki, Wendake-Nionwentsïo, and Haudenosaunee peoples, sustainably hunted, fished and farmed in the Kawartha Lakes region (Native Land Digital, 2021). In the early 19th century, European settlement of the area began, and altered the landscape, setting it up for continuous development (Kawartha Conservation, 2016b). The forests were logged for timber and once cleared, used for agriculture, particularly in the southern portion of the catchment area. A series of dams and locks were constructed to connect the lakes and allow for the transportation of timber and other supplies. This system of connected lakes eventually formed part of the Trent-Severn waterway (TSW), which connects Georgian Bay to Lake Ontario at Trenton. By the time

the TSW was completed in 1920, the interconnected lakes were less important for transportation, but increasingly valued for the tourist economy (Foran, 2010). Interest in tourism for the region began before the TSW was completed, in the 1890s local tourism promoters wanted a new collective name for the area, and went to the Curve Lake First Nation peoples who suggested the name 'Kawatha'. Kawatha was said to mean 'shining waters and bright lands', and was slowly accepted by local town councils (Tatley, 1978). However, by 1900 the name had been changed to 'Kawartha', despite the R sound not existing in the Ojibwa language (McIntyre et al., 2006). The altered name stuck and the slogan attached to it 'shining waters and bright lands' was used to promote tourism in the area.

The natural landscape and activity in the Kawartha's following European settlement resulted in the current land-use patterns in the Kawartha Lakes watershed, with widespread agriculture interspersed with urban development (Figure 1.2). Cleared land used for farming resulted in the spread of small rural communities across the southern portion of the watershed. Advertisement of the region as a natural haven near Toronto resulted in many cottages built along shorelines, and small towns with tourism-based businesses thriving. Recently, more of these properties have been converted to permanent residences as owners retire and move out of large cities (Kawartha Conservation, 2015b). Seasonal tourism is important economically for the region, bringing in over 109$ million each year, however, it is also likely having negative impacts on the health of the lakes (City of Kawartha Lakes, 2021). Many tourists visit the area for sportfishing, however, in Lake Scugog a ban on fishing *Sander vitreus* (walleye) was implemented in 2016 due to declining stocks. The connectivity of these lakes

through the lock and dam system also influences the health of the lakes. With thousands of boats moving between the lakes each year there is an increased risk of introduction of non-native species. In particular, Lake Scugog has been plagued by invasive species, which will be discussed in Chapter 3. Ultimately, if nothing is changed, the appeal of the region may be lost due to the negative influence of human activity on water quality and the biotic communities in these lakes.

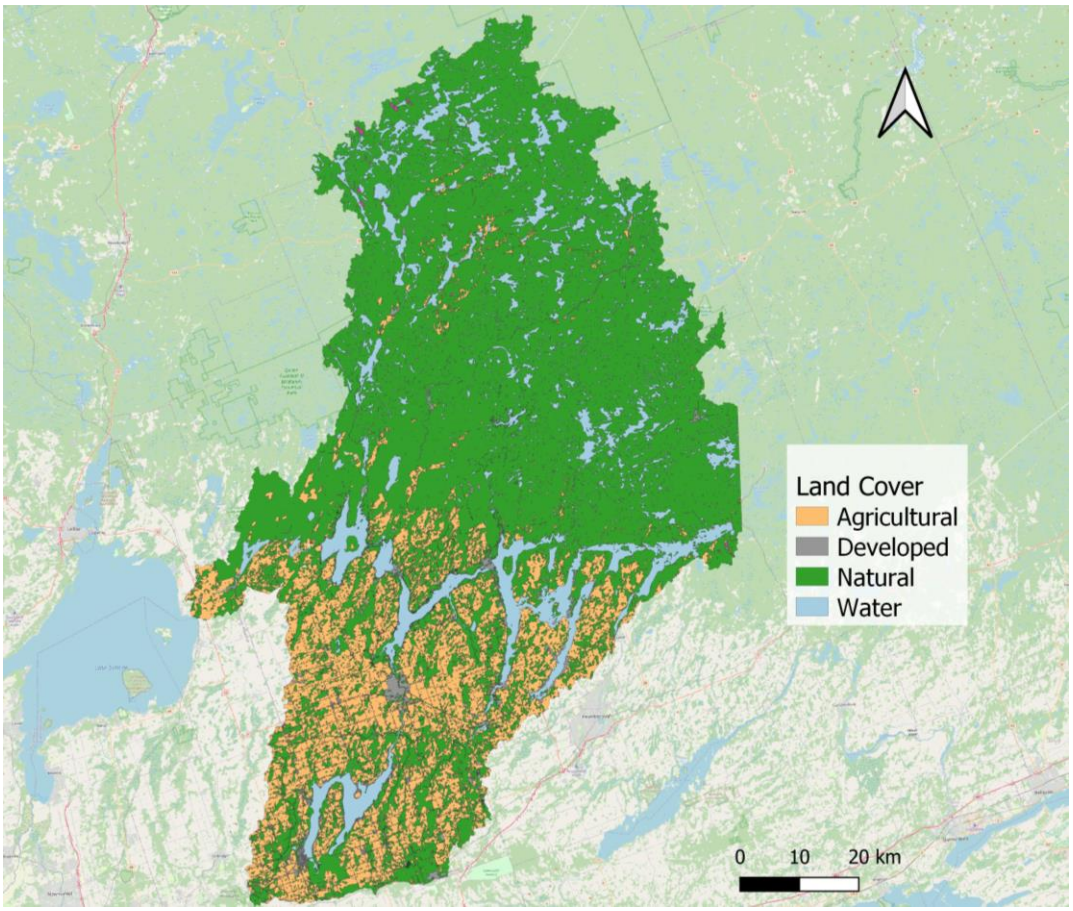

Figure 1.2 Regional map of the Kawartha Lakes watershed, with land classified as agricultural, developed or natural.

The climate in the Kawartha Lakes region is classified as a moist continental mid-latitude climate, in the Dfb climate category according to the Köppen Climate Classification System (Kawartha Conservation, 2014b). The Köppen Climate

Classification System is widely used and classifies regions by annual and monthly averages of temperature and precipitation (Peel et al., 2007). The most central climate station for the study area was the Lindsay Frost station, located in Lindsay ON, although it shut down in 2009. Data from the 36 years it was running indicates the area received an average of 890 mm of precipitation annually, spread fairly evenly across the months. January was the coldest month of the year with average temperatures of -8.4 ˚C, and July the hottest with an average temperature of 20.3 ˚C, and an average yearly temperature of 6.6 ˚C (Kawartha Conservation, 2015a). This data confirms the climate categorization of moist continental mid-latitude climate. In the Kawartha Lakes region it is predicted that climate change will result in warmer winters, more extreme precipitation events and more extreme weather events (Eimers et al., 2020; Kawartha Conservation, 2016a).

1.1.1 Lake Scugog

Lake Scugog is the southernmost Kawartha Lake and has a highly developed watershed compared to the other lakes included in this study. Lake Scugog's watershed is entirely within the limestone-dominated bedrock of the St. Lawrence Lowlands, making it very suitable for croplands. Scugog is fed by local tributaries and is a headwater for downstream Kawartha Lakes, including Sturgeon and Pigeon Lakes. Before European settlement, the Mississauga's of Scugog Island lived sustainably in the Lake Scugog watershed since the 1700s (Mississaugas of Scugog Island First Nation, 2021). In 1837, following European settlement of the area, a sawmill was constructed on the Scugog River at Lindsay (Kawartha Conservation, 2010). Following dam constructions, water levels in Lake Scugog rose by four feet and the wetland-riverine complex was flooded, resulting in the current morphometry of Lake Scugog (Figure 1.3). The raised water

levels allowed for the transport of materials between the lakes north of Scugog and the Port Perry railhead, connecting the Kawartha Lakes to the growing Greater Toronto Area (GTA). During European settlement much of Lake Scugog's watershed was cleared for timber and the land then used for agriculture. Since the 1930s Scugog has been advertised as an ideal vacation area and the tourism industry has been an important economic sector for the Township of Scugog, bringing in 1.2 - 1.5 million visitors each year (Township of Scugog, 2018).

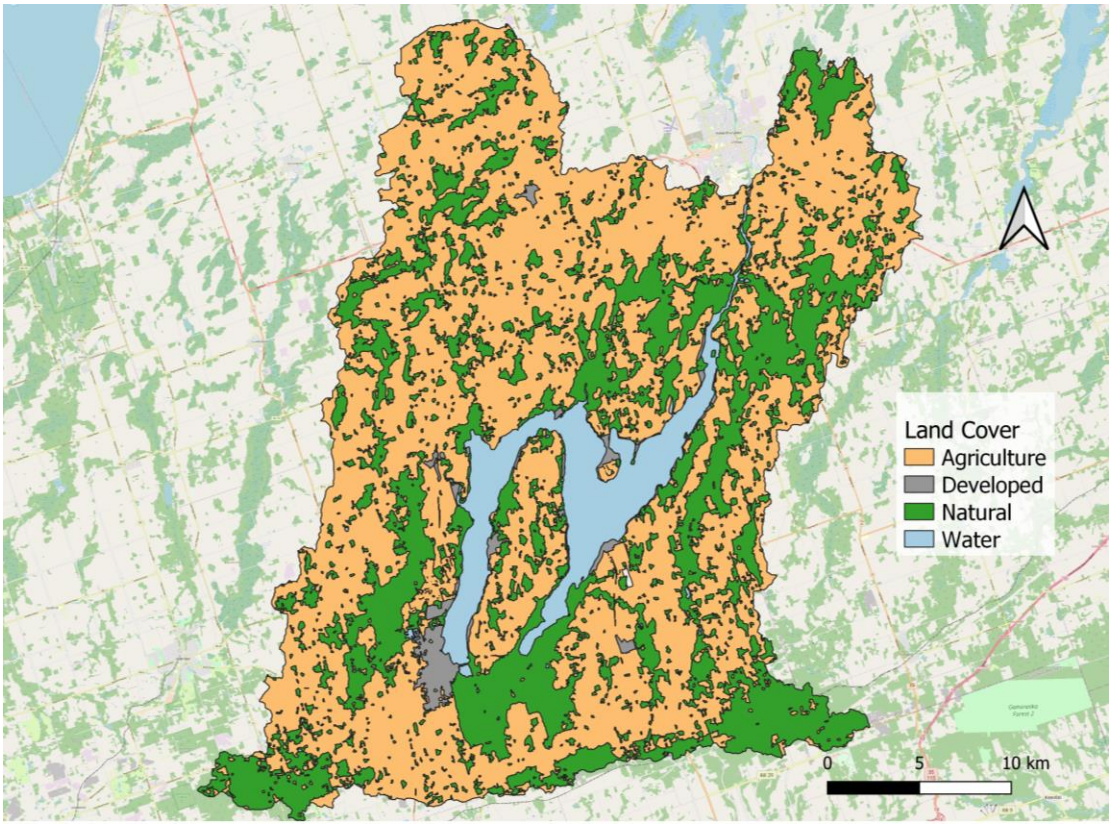

Figure 1.3 Map of Lake Scugog watershed with land cover classified as agricultural, natural or developed.

The land-use patterns in Scugog's 529.7 km^2 watershed reflect the changes made by European settlers in the 1800s. Agriculture dominates the landscape, making up 65% of the drainage basin, followed by natural land cover (33%) and development (2%). Due

to the fertile soils in the watershed, croplands are widespread and primarily composed of alfalfa, wheat, soybeans, and corn. Livestock is also common in the watershed, dominated by cattle and poultry (Kawartha Conservation, 2010). Although some soil conservation methods are used, agriculture at this scale is expected to have an impact on nutrients and potentially *Escherichia coli* levels in the lake. Natural land cover in Lake Scugog is primarily comprised of forests and wetlands which help with water filtration. In particular, there are two large wetlands, Osler Marsh and an unnamed wetland west of Lake Scugog which likely play an important role in filtering water from the surrounding agricultural lands. Finally, although developed land only makes up 2% of the Scugog watershed, the pattern of developed land use is important to consider for its impact on the lake's water quality and biology. Most development in the watershed is located in the Town of Port Perry on the southwest shore of the lake. This high concentration of developed land with hardened surfaces and lawns can result in increased levels of nutrients, chloride, suspended sediments, and *E. coli*. Most of the remaining developed land is located directly on the shoreline, in the form of cottages, trailer parks and permanent residences. Shoreline residences increases the area of hardened surfaces (roads, driveways, manicured lawns) near the water and the amount of altered shoreline.

When the lacustrine-wetland complex was flooded to make Lake Scugog, it resulted in a shallow lake with a thick layer of nutrient-rich sediment at the bottom. This combination, along with the abundant agriculture in the watershed resulted in a highly eutrophic lake by the mid-20th Century. Phosphorus levels were consistently above 20 µg/L, the current Provincial water quality objective (PWQO), reaching as high as 45 µg/L in 1976 (Ontario Ministry of the Environment and Ministry of Natural Resources,

1976). Since then, provincial regulations helped to reduce nutrient loading, resulting in declining phosphorus levels by the late 1990s, stabilizing below 20 µg/L (Robillard & Fox, 2006). More recently, monitoring results by Kawartha Conservation show that phosphorus levels have increased, this time in specific areas of the lake, especially the Port Perry area (Kawartha Conservation, 2010). This increase in phosphorus is mirrored by nitrogen, indicating that nutrient levels are increasing and shifting Scugog back towards a eutrophic state.

Lake Scugog is a highly productive lake with high macrophyte and algal growth, as well as productive fishery. Though Scugog is known for its excessive macrophyte growth that poses a nuisance for boaters and swimmers, macrophytes are important primary producers that can absorb contaminants, stabilize the lake bed, and reduce shoreline erosion (Beklioglu & Moss, 1996). Aquatic macrophytes also provide crucial habitat for the lake's inhabitants. Native macrophytes have provided these ecosystem services since the flooding of the lake, but the introduction of the invasive species *Myriophyllum spicatum* in the 1970s altered the abundance and diversity of macrophytes in Scugog (Borrowman et al., 2014). As a non-native species, *M. spicatum* easily dominated the aquatic macrophytes in Scugog, but it declined in the 1980s. It is believed that this decline was due to herbivory, specifically from the larva of the non-native moth *Acentria nivea* (Painter & McCabe, 1988). *Myriophyllum spicatum* beds were of concern again in the early 2000s, and herbivory was used to control its growth, but this time with the North American weevil *Euchrychiopsis leconteri* (Borrowman et al., 2014). Stocking specific areas with *E. leconteri* proved effective, and was used again in 2014 to reduce dense stands near Port Perry. However, at this point, a new invasive macrophyte,

Nitellopsis obtusa, was beginning to take over macrophyte beds in Scugog (Harrow-Lyle & Kirkwood, 2020). *Nitellopsis obtusa* not only results in high macrophyte density but also impacts the aquatic food web, increasing the bloom-forming cyanobacterial taxon *Microcystis* spp. and the invasive zebra mussel *Dreissena polymorpha* (Harrow-Lyle & Kirkwood, 2020). Due to its popularity as a boating and fishing destination, boat traffic has inadvertently introduced a variety of invasive species that continue to impact its water chemistry and biological community.

1.1.2 Balsam Lake

Balsam Lake is the western most lake in this study, and drains directly into Cameron Lake, indirectly feeding downstream Sturgeon and Pigeon Lakes. Balsam Lake is fed by Gull River and Corben Creek to the north and other small tributaries around the lake. Balsam Lake is the fourth largest Kawartha Lake (48km^2), its average depth is 4.8 m (Figure 1.4). Balsam's watershed is split between the limestone-dominated St. Lawrence Lowlands and granite-dominated Canadian Shield.

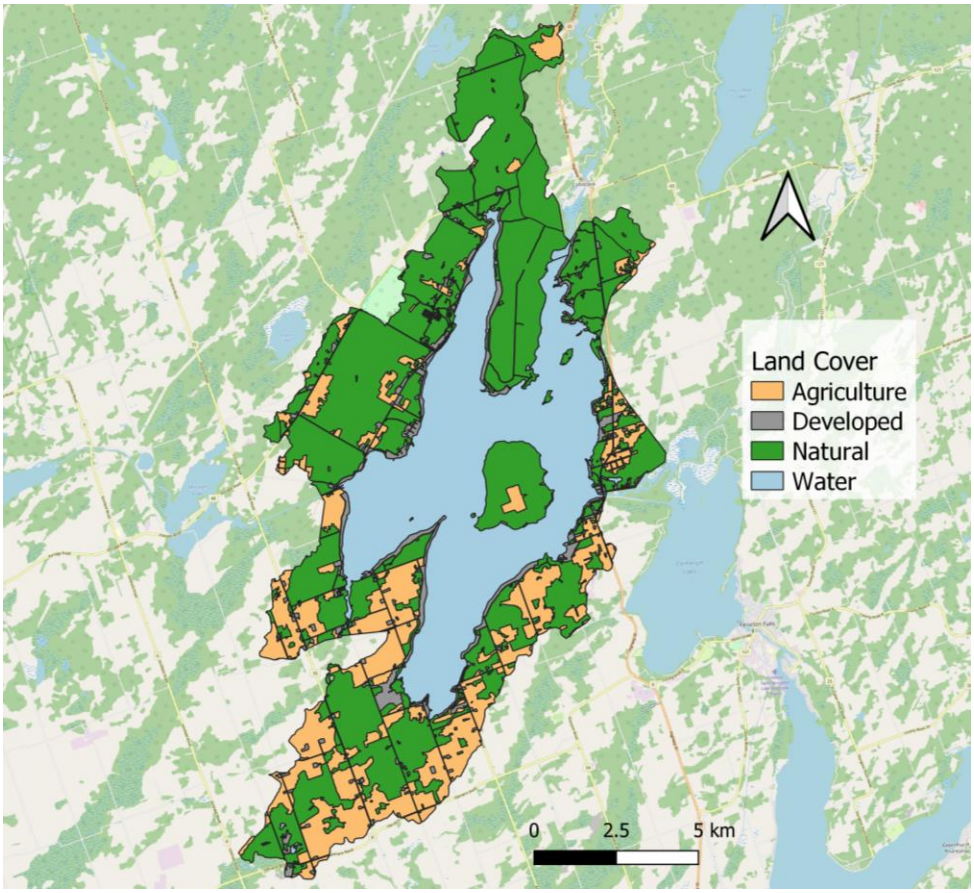

Figure 1.4 Map of Balsam Lake watershed with land cover classified as agricultural, natural or developed.

Land-use patterns in Balsam's watershed reflects the historical activity in the watershed and its underlying geology, with 55.5% natural land cover, 37.8% agricultural land use, and 6.7% developed land use. Natural land cover in the watershed is mainly composed of wetlands with some forests. The northern part of the watershed is dominated by natural land-cover due to the underlying geology (mosaic of limestone alvars and granite) being unsuitable for agriculture. Agricultural land use dominates the southern part of the watershed where the forests were logged and converted to farmland. Currently, hay and grain farming and beef cattle production dominate the agricultural landscape (Kawartha Conservation, 2015b). Developed land use is mostly spread along

the shoreline, as of 2012 1,380 properties covered 49.7% of the shoreline (Kawartha Conservation, 2015a). Similar to the other Kawartha Lakes there is a trend of an increasing number of permanent residents.

Balsam Lake is categorised as an oligotrophic lake, with average phosphorus levels in most areas of the lake below 10 µg/L (Kawartha Conservation, 2015a). Historically, phosphorus levels peaked in the late 1970s, around 17µg/L, and declined until the 1990s when they stabilised around 10 µg/L (Kawartha Conservation, 2015a; Robillard & Fox, 2006). More recently, monitoring of the lake's outflow at Rosedale from 2011-2013 indicates phosphorus levels declined further to an average of 8 µg/L (Kawartha Conservation, 2015a). Although nutrient levels are not currently of concern, they do require continued monitoring, especially since there is ongoing land development around the lake, and South Bay and East of Grand Island sampling locations are approaching 10 µg/L (Kawartha Conservation, 2015a).

Of the Kawartha Lakes included in this study, Balsam Lake is the least productive with low nutrient levels and a narrow nearshore zone. This combination of characteristics means that the available nearshore zone is vital to sustaining the aquatic food web by providing habitat and sustenance. Balsam Lake supports a range of cool/warm water fishes, including walleye and muskellunge, which rely on the nearshore zone for spawning and nursery grounds (Kawartha Conservation, 2015a). Although considered to be the most pristine of the five Kawartha Lakes in this study, Balsam Lake has not been immune to the invasion of non-native species, which inevitably impact the aquatic community. *Myriophyllum spicatum* is widespread in the lake, although it is likely not as problematic as in other lakes due to Balsam's lower nutrient concentrations, depth and

narrow nearshore zone. *Dreissena polymorpha* have impacted Balsam, resulting in increased water clarity following their arrival in the lake (Kawartha Conservation, 2015a). In 2013, the macroinvertebrate community in Balsam and Cameron Lake's tributaries were monitored. Sites were scored from Very Poor to Excellent on the Hilsenhoff Biotic Index. The Hilsenhoff Biotic Index is based on the pollution sensitivity of the identified taxa scores range between 0 to 10, with higher scores indicating a community more tolerant of pollution (Hilsenhoff, 1988). Out of the 16 sites in tributaries feeding Balsam and Cameron lakes 56% scored above Fair, while 44% scored worse than Fair on the Hilsenhoff Biotic Index (Kawartha Conservation, 2015a). The Balsam and Cameron tributary sites also had an average Biotic Index score of 5.17 which is lower than the tributaries of Sturgeon Lake, but slightly higher than Pigeons tributaries (Kawartha Conservation, 2016b).

1.1.3 Cameron Lake

Cameron Lake is second in the chain of lakes included in this project. It is fed directly by Balsam Lake and the Burnt River sub-watershed to the north. Cameron Lake has a surface area of 14.7 km^2, and an average depth of 6.9 m (Kawartha Conservation, 2015b). It is the smallest and deepest of the lakes included in this study. Similar to Balsam Lake, Cameron Lake's watershed is split between the limestone-dominated bedrock of the St. Lawrence Lowlands and granite bedrock of the Canadian Shield.

The Cameron Lake watershed land use is split between natural (51.1 %), and agricultural (43.2%) land cover, with interspersed development (5.7%) (Figure 1.5). The natural land in the watershed is primarily composed of treed wetlands, one in the Martins Creek South subwatershed and the other in the Pearns Creek subwatershed. Agricultural

land in the watershed is widespread but limited to the north due to the underlying geology. Agricultural land in the Cameron Lake watershed is mostly grain and hay croplands, followed by cattle pastures (Kawartha Conservation, 2015b). Although development only makes up a small proportion of the Cameron Lake watershed, it is concentrated along the shoreline, and continuing to increase with new permanent residents. Rosedale and Fenelon Falls are the main urban areas on the lake, located at the inlet and outlet, respectively. The rest of the development is rural, primarily cottages and permanent residences along the shoreline. As of 2012, 44.9% of the shoreline was developed with 588 residences, a 42% increase from 1970 (Kawartha Conservation, 2015a). This number has likely increased over the past 9 years and will continue to grow as this burgeoning bedroom community expands.

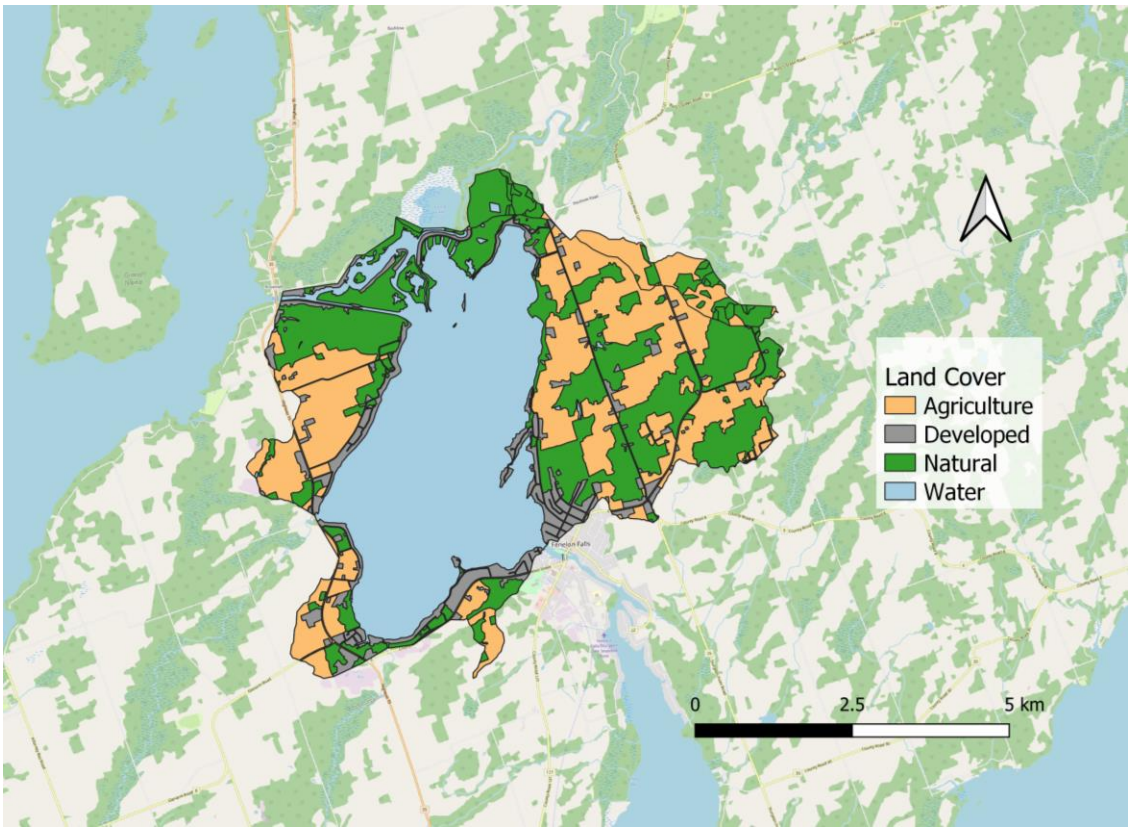

Figure 1.5 Map of Cameron Lake watershed with land cover classified as agricultural, natural or developed.

Local monitoring of Cameron Lake indicates it is oligotrophic, with an average phosphorus level of 10 µg/L (Dillon, 1975; Kawartha Conservation, 2015a). Historical trends in Cameron's phosphorus levels are similar to Balsam Lake, peaking in 1970 and declining until the 1990's when levels stabilised. Monitoring from 2011-2013 indicates average levels around 10 µg/L, with little variation across the lake. Nitrogen levels in Cameron were well below the levels of concern, averaging around 0.30 mg/L from 2011-2013. The only parameter of concern based on previous findings is *E. coli*, which had an annual geometric mean that exceeded the PWQO of 100 CFU/100 mL in 2010 and 2013. These elevated readings were taken at a popular beach in Fenelon Falls and were likely due to the abundant Canada Geese population, but examining *E. coli* levels around the lake will help determine if this is a local issue or lake-wide problem.

The biological community in Cameron Lake has not been monitored extensively, however, communities are expected to be similar to Balsam Lake due to similar water quality conditions between these connection ecosystems. Cameron Lake also has a comparatively small nearshore zone, limiting this important habitat for littoral organisms (Kawartha Conservation, 2015a). Although fish community data is not published, it is expected that Cameron supports a cool/warm water fishery, as the lake's tributaries provide habitat for both coolwater and warmwater fish. Additionally, there was no data available on the lake's macrophyte community. The macroinvertebrate community has been monitored in the Balsam and Cameron watershed tributaries, and Hilsenhoff Biotic Index ratings varied from Poor to Excellent, with 56% receiving a score of Fair or better (Kawartha Conservation, 2015a). Specific ratings within the Cameron Lake watershed

are not disclosed. It is expected that the invasive species *Myriophyllum spicatum* and *Dreissena polymorpha* are present in Cameron Lake, but this cannot be confirmed without recorded observations.

1.1.4 Sturgeon Lake

Sturgeon Lake lies in the middle of the study lakes, connecting Cameron, Scugog and Pigeon Lakes. Its central location makes it a popular lake for tourists and it experiences a high volume of recreational boat traffic. Sturgeon Lake is fed by Cameron Lake and Lake Scugog and a number of small and medium sized tributaries. Sturgeon Lake is the fifth-largest Kawartha Lake with a surface area of 45.6 km^2, but it is relatively shallow with an average depth of 3.5 m.

The Sturgeon Lake watershed is primarily composed of agricultural land use (49.5%), followed by natural land (40.9%), with scattered development (9.7%) (Figure 1.6). Agriculture is the main source of income for the area and it is present across the watershed. Similar to the rest of the region, grain and hay are the main crop and cattle farming is the second most common use of agricultural land (Kawartha Conservation, 2014a). Natural land in Sturgeon's watershed is primarily composed of wetlands surrounding tributaries and interspersed forests mostly north of the lake. Of the lakes included in this study, Sturgeon Lake has the most built-up areas, the town of Lindsay at the Scugog River inlet, Fenelon Falls at the Cameron Lake inlet, and Bobcaygeon at the Pigeon Lake outlet. There are also many shoreline communities spread around the lake, 1,774 waterfront residences cover 74% of the shoreline's length (Kawartha Conservation, 2014b).

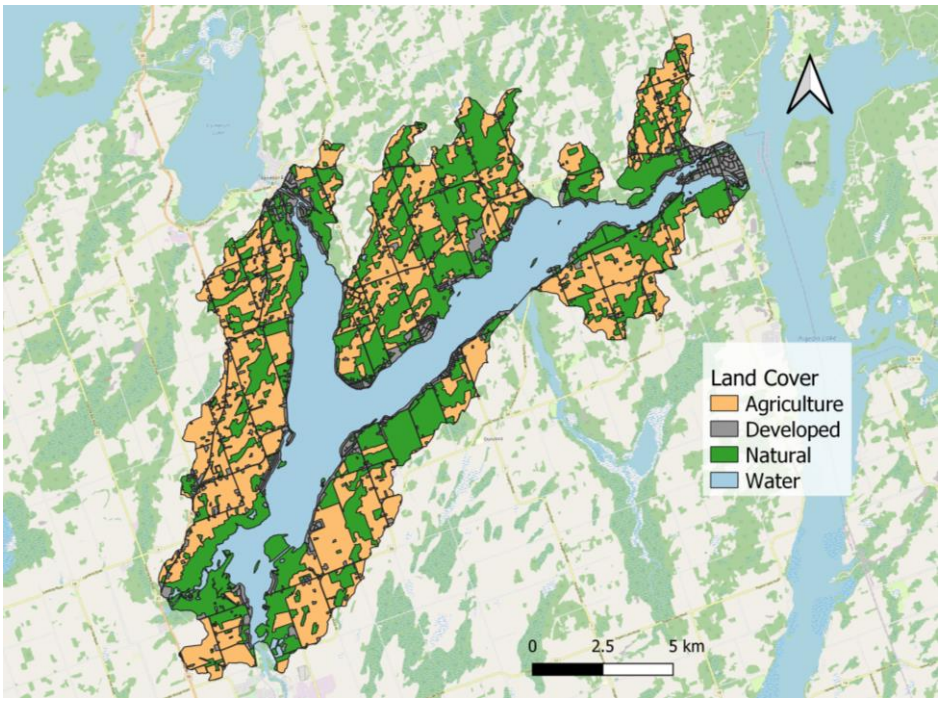

Figure 1.6 Map of Sturgeon Lake watershed with land cover classified as agricultural, natural and developed.

Sturgeon Lake has historically been one of the most productive Kawartha Lakes, following only Lake Scugog. Since the mid-1900s Sturgeon Lake was classified as eutrophic with phosphorus levels consistently exceeding 20 µg/L, in addition to nutrient pollution, toxic chemicals from industrial sources were dumped into the lake's tributaries eventually settling in the lakes sediments (Kawartha Conservation, 2014b). However, changes in policies, including reducing phosphates and improving wastewater treatment, have resulted in improved water quality since the 1970s, with decreasing nutrient levels and increased water clarity. Sturgeon Lake is now classified as a mesotrophic water body with phosphorus levels below 20 µg/L in most areas of the lake (Kawartha Conservation, 2014b). Overall Sturgeon is classified as mesotrophic; however, the southern arm of the lake remains eutrophic. The major phosphorus sources for this area of the lake are Lake Scugog and the Scugog River sub-watershed, which combined contribute 31% of the

lake's phosphorus inputs (Kawartha Conservation, 2014a). Nitrogen levels follow the same trend as phosphorus, with lower levels (< 0.50 mg/L) in all parts of the lake except the southern arm. From 2010-2012, *E. coli* was measured at public beaches on Sturgeon Lake and most locations measured well below the PWQO of 100 CFU/100 mL. Beach Park in Bobcaygeon exceeded the PWQO in over 30% of the samples taken in 2010, 2011, and 2012 (Kawartha Conservation, 2014b). Although waterfowl, specifically Canada Geese, are likely responsible for these high levels it is possible that there are other urban sources in Bobcaygeon contributing to *E. coli* levels.

Changes in Sturgeon Lake's water quality influenced its biotic community as it shifted from a murky, nutrient-enriched water body to a clear water mesotrophic system. The introduction of *Dreissena polymorpha* (zebra mussels) catalyzed this shift by increasing water clarity resulting in increased macrophyte growth (Kawartha Conservation, 2014b). Currently, macrophytes are abundant in the southern arm of the lake, specifically at Goose Bay, and dispersed in the northwest arm. Macrophyte growth in Sturgeon is limited in part by the bottom substrate which is too hard in the northeast arm of the lake and has variable regions of soft sediment in the northwest arm. The fish community in Sturgeon is similar to the other Kawartha Lakes, mainly supporting warmwater fish, with some non-permanent coolwater fish populations. In 2011 the macroinvertebrate community in the nearshore zone of Sturgeon was identified, and across 16 sites the average Simpsons Diversity Index value was 0.59. This relatively low score is likely due to the prevalence of scuds that made up over 50% of the total relative abundance. In addition, sensitive taxa, Ephemeroptera, Plecoptera, and Trichoptera comprised ~15% of the total relative abundance. These scores indicate a moderately

tolerant macroinvertebrate community, not as diverse as some healthy streams but better than the Lake Ontario nearshore community (Barrett et al., 2017; Johnson et al., 2004)

1.1.5 Pigeon Lake

Fed by Sturgeon Lake, Pigeon Lake is the easternmost lake included in this study. Pigeon Lake directly feeds Buckhorn Lake, and the rest of the downstream Kawartha Lakes indirectly. Pigeon Lake is elongated from north to south with Boyd Island at the north end. Pigeon has a total surface area of 55 km^2, and an average depth of 3.3 m (Kawartha Conservation, 2018). Bobcaygeon is the most developed area of Pigeon Lakes' watershed, located at the inlet from Sturgeon Lake. Pigeon Lake's watershed is the traditional land of the Curve Lake First Nation peoples, who hunted, fished, and lived on the land sustainably before European settlers arrived in the 1800s. The Curve Lake First Nations community has a deep connection with the lake, which includes the management of manoomin or wild rice (*Zizania* spp.).

Pigeon Lake's watershed is comprised of agricultural (49.6%) and natural (44.5%) land, with some development (5.9%) along the shoreline (Figure 1.7). Similar to the other Kawartha Lake watersheds, agriculture around Pigeon Lake consists of alfalfa, soybeans, corn, wheat and hay, and beef cattle is the most prominent livestock. Natural land cover is primarily to the north of the lake where the land is not suitable for farming. Wetlands are another important type of natural land cover in the watershed, surrounding some of Pigeon's tributaries and dispersed along the western edge of the lake. An estimated 29% of the shoreline is developed, although this is expected to increase greatly as the population of the City of Kawartha Lakes grows.

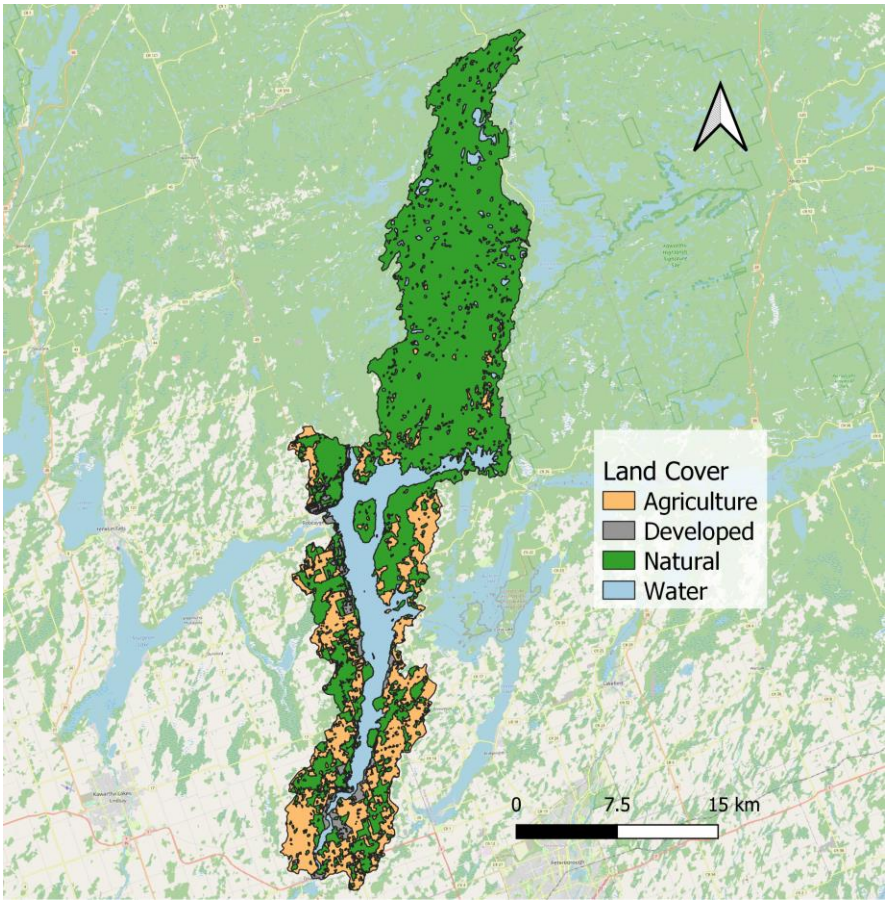

Figure 1.7 Map of Pigeon Lake watershed with land cover classified as agricultural, natural and developed.

Monitoring from 2011-2014 indicates that Pigeon Lake is mesotrophic, with phosphorus levels just below the 20 µg/L level (Kawartha Conservation, 2016b). Similar to the other Kawartha Lakes it experienced a decline in phosphorus levels following stricter nutrient pollution regulations in the 1970s. The lake has two distinct regions that have different water quality profiles. The southern part of the lake is much shallower, with higher phosphorus and nitrogen levels, approaching eutrophic conditions. The northern part of the lake is deeper and its main tributary, Nogies Creek has a mostly natural sub-watershed. *E. coli* levels in the lake measured consistently below the PWQO at the time, 100 CFU/100 mL, the only area of concern for *E. coli* was Omemee Beach on

the Pigeon River in the southern part of the watershed (Kawartha Conservation, 2016b). Omemee beach *E. coli* levels consistently exceeded the PWQO many times when sampled in 2011-2014.

The biotic community in Pigeon Lake reflects the contrasting water quality profiles between the northern and southern ends of the lake. The shallow, nutrient-rich southern end has high macrophyte growth, which was reported in a lake-wide vegetation survey in the 1970s and confirmed in 2015 where *Zizania* spp. was found to cover an estimated 10-15% of the lake's area (Kawartha Conservation, 2016b). The high plant abundance would be expected to provide habitat for many aquatic taxa, from macroinvertebrates to fish. Macroinvertebrates in the lake have not been examined, but in 2014, 16 tributary sites were measured. It was found that 82% of sites scored Fair or better based on the Hilsenhoff Biotic Index, which is comparable to Balsam and Cameron tributaries and better than Sturgeon's tributary HBI scores (Kawartha Conservation, 2016b).

1.2 The Nearshore Zone

The physical nearshore zone is defined as the area between the shoreline and the outer limit of surf cell circulation (Figure 1.8) (McLachlan, 1983). Although I refer to my study sites as being located in the nearshore zone to align with the Great Lakes Nearshore Monitoring Framework, they are also part of the littoral zone, which is defined as the area of the lake where sunlight reaches the sediment (Figure 1.9) (Zohary & Gasith, 2014). The littoral zone is typically where most primary production per unit area occurs in a lake because it supports the growth of both plants and algae. The littoral zone also provides habitat for almost all lake organisms at some point in their life cycle (Vadeboncoeur et

al., 2011). The nearshore zone, which usually encapsulates the littoral zone, is not only vital for aquatic organisms, but also for humans. It is where we go to the beach, swim, and fish. For the many roles it plays, we are just beginning to understand the nearshore zone and how its water quality and biotic communities are impacted by human activities.

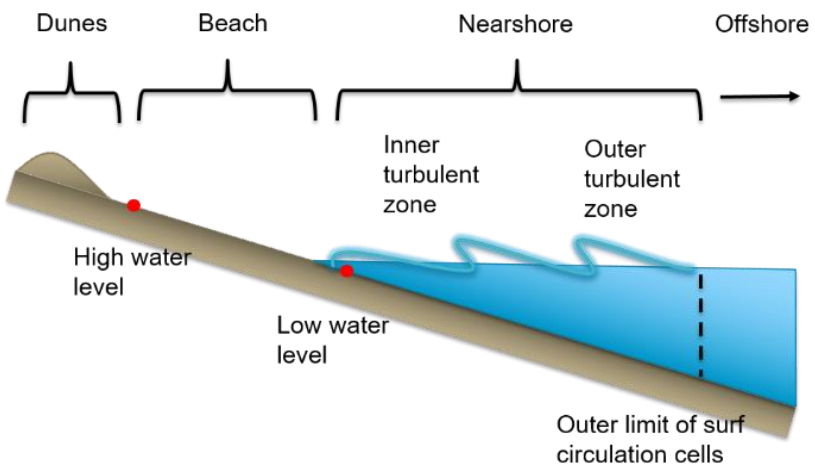

Figure 1.8 Physical nearshore zone of a lake.

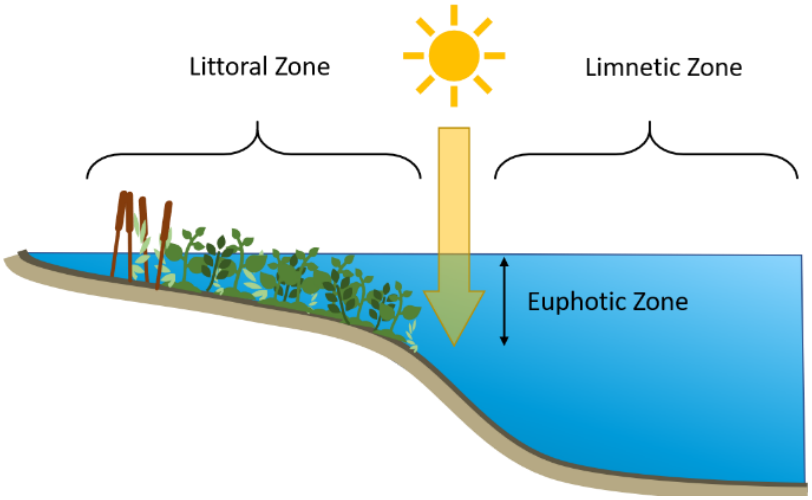

Figure 1.9 The littoral zone of a lake.

Traditional water quality monitoring typically occurs in the centre or deepest part of a lake's pelagic zone to get an overall measure of lake water quality. Yet, we know

that water quality in the nearshore zone are persistently distinct over a lake's ontogeny (Yurista et al., 2016). Only measuring water quality parameters in the offshore region of a lake reflects a major data gap in the habitat conditions of the nearshore zone. Nearshore water quality parameters such as conductivity and nutrients tend to be more variable, more correlated to landscape features, and of higher concentration than offshore measurements (Howell et al., 2012; Makarewicz et al., 2012; Yurista et al., 2015; Yurista et al., 2016). Much of the existing comparisons between nearshore and offshore water quality are on large lakes, such as the Laurentian Great Lakes, and further research is required to see if these trends persist in smaller inland lakes.

Land use is an important factor to consider for driving lake water quality, particularly in the nearshore zone. The nearshore zone is the first area of the lake that is directly impacted by watershed land use by receiving run-off and tributary discharge. Agricultural land in a watershed tends to contribute nutrients and *E. coli* and increases turbidity (Howell et al., 2014). Developed land does the same, but with higher inputs of chloride from road salts and occasionally trace metals (Howell et al., 2012; Read et al., 2015). Many of these impacts have been elucidated at the watershed scale, but there is growing evidence that the land use – water quality relationship varies regionally (Read et al., 2015; Soranno et al., 2015; Stachelek et al., 2020). Factors such as lake connectivity, morphometry, and configuration of watershed land cover impact the relationship between land use and water quality parameters (Gémesi et al., 2011; Read et al., 2015; Soranno et al., 2015). Additionally, different scales of land use are beginning to be examined to determine their relative importance within the watershed. Watershed-scale land use has the strongest relationship with water quality parameters in some large-scale studies

(Nielsen et al., 2012; Pratt & Chang, 2012), however, others have found that some water quality parameters have stronger relationships with fine-scale land use (Liu et al., 2017). Further investigation into the importance of each scale will provide vital information for future watershed management.

The biological community is also impacted by erosion and run-off caused by land use activities. At the base of the food web, primary producers feel the first impacts of shoreline development and the corresponding changes in water quality. Macrophytes in the nearshore zone tend to have reduced biomass and changes in structural plant type in response to shoreline development (Hicks & Frost, 2011). Even within a lake, macrophytes that are greater distance from docks had higher species richness, with more sensitive species present (Beck et al., 2013). Phytoplankton and periphyton in the nearshore zone are also impacted by land use. In freshwater systems, phytoplankton responds to increased phosphorus with increased biomass, and often a shift in the type of phytoplankton (Downing et al., 2001). An increase in phosphorus can lead to increased cyanobacteria algae blooms, and result in a higher proportion of inedible phytoplankton in the water column (Chang & Rossmann, 1988). Periphyton is also a useful indicator of shoreline development. Periphyton biomass increases in response to elevated nearshore nutrient levels, and can even respond to modest shoreline development (Lambert & Cattaneo, 2008; Rosenberger et al., 2008).

The impacts of human activity at the shoreline are also felt higher up in the food web, especially when macrophyte beds are removed and shorelines are hardened with armour stone, concrete or rip-rap. Macrophytes provide key refuge for many zooplankton species, but the removal of macrophytes does not impact zooplankton groups evenly, it

can reduce zooplankton biomass and shift a community to be dominated by cyclopoid copepods (Sagrario et al., 2009; Schriver et al., 1995). Zooplankton are also impacted by changes in the phytoplankton community in response to shoreline development. An increase in large colonial cyanobacteria can reduce the abundance of large daphniids due to their inedibility, shifting the community to calanoid copepods, rotifers and smaller Cladocera (Ger et al., 2014; Hansson et al., 2007). Macroinvertebrates are directly impacted by shoreline hardening, resulting in decreased richness and altered community composition (Porst et al., 2019). Although it is evident that shoreline hardening alters the macroinvertebrate community, there is no clear consensus on how the different groups are affected (Brauns et al., 2007; Chhor et al., 2020).

The nearshore zone of a lake is vital for the entire lake food web and our recreational enjoyment of the lake. The nearshore zone is the best location to examine the impacts of land use and determining the relationship between nearshore water quality and land use at multiple scales will provide more information about the relative impacts of development. Furthermore, shoreline modification has direct impacts on the biotic community that need to be better understood.

1.3 Community Science

Members of the public have long aided in scientific research, and since the 1990s we have referred to this as “citizen science” (Irwin, 2018). Recently, there has been an uptick in the number of citizen science research projects, however, there has also been debate about the language used to describe these projects (Dickinson et al., 2010). Citizen science projects are being relabelled as “community science” projects to make the language more inclusive to participants beyond national citizens. However, this

relabelling can be problematic because community science already exists with unique characteristics that are not present in many citizen science projects. Community science projects are based on the goals of the community, often linked to social action, and based on local priorities and perspectives, allowing the project to maintain power within the community (Cooper et al., 2021). Based on this set of specifications our project falls under the category of community science, as this project was developed based on the community's goals, I have shared information openly with the community, and it will inform strategies to benefit the communities around the lakes.

As community science grows, we are discovering the impacts of this research approach, and an increasing number of studies are examining the effectiveness of these projects. For the researcher there are some obvious benefits of the approach, such as increased spatial and temporal scale, access to private property, and the potential to answer multiple questions with a single methodology (Dickinson et al., 2010). As with any methodology, there are some limitations to community science. There are some questions about the accessibility and organization of community science data, but the biggest detractor is preconceptions about the credibility and validity of data collected by "amateur" scientists (Conrad & Hilchey, 2011).

To overcome the data accessibility issue, datasets are shared widely, especially as open data becomes more encouraged in academia. The credibility question is more often the barrier to community science, however, there are strategies to overcome this problem. Freitag et al. (2016), dissect the different strategies used by environmental monitoring community science projects to build credibility. There are three components of a study where strategies to increase credibility may be applied: (1) training and planning, (2) data

collection, and (3) data analysis and program evaluation (Freitag et al., 2016). Successful community science projects will have at least one strategy employed at each stage, and the most common strategies included: training, scientific advising, publication, and management use of project data (Freitag et al., 2016). In addition to these strategies, water quality monitoring projects can directly compare samples collected by volunteers to those collected by trained professionals. These comparisons support using community science for water monitoring as previous studies have not found a difference between volunteer and professional collected samples (Albus et al., 2020; Canfield Jr et al., 2002; Scott & Frost, 2017).

Involvement in a community science project is also beneficial for volunteers. As the number of community science projects has increased there has been more focus on identifying outcomes for the participants. Many projects found that participants gain scientific knowledge, particularly within the topic of the community science project (Bonney et al., 2016). Participants may also become more likely to engage in pro-environmental activities, and be involved with local decision-making (Bonney et al., 2016; Crall et al., 2013). Another potential outcome that requires more work to understand, is the impact on participants' attitudes towards science and the environment. Some studies have found an increase or perceived increase in scientific attitudes, while others did not detect any change between the start and end of a project (Brossard et al., 2005; Price & Lee, 2013; Toomey & Domroese, 2013). One finding that complicates this, is that many community science participants' are self-selected and already have positive attitudes towards science (Brossard et al., 2005). Other factors such as participants' level of involvement throughout the scientific process may impact outcomes, with more

involved projects resulting in increased understanding of science (Bonney et al., 2016). These conflicting findings indicate the need to understand how participants attitudes towards science and the environment change after involvement in a community science project.

Effective community science projects often require the cooperation between academic and non-academic actors, in a process called co-production. Academic actors being researchers usually from a university, while non-academic actors represent governmental or non-governmental environmental groups. The co-production model of research has also been growing, especially in fields where the results can benefit both groups engaged in the project (Norström et al., 2020). There are some basic components that are required to build a successful co-production project; the actors involved in the work must build trust, and be flexible with each other (Djenontin & Meadow, 2018). In environmental research, a successful co-production model provides information that can help to directly inform policy, while benefiting the wider academic community (Djenontin & Meadow, 2018). Co-production and community science often go hand-in-hand as they are both based on open science and working with a local community. Using a co-production method for a community science project can help ensure certain outcomes are achieved, such as increasing local decision-making and publication of the findings.

1.4 Bioindicators

One way to examine the impacts of human disturbance on the nearshore zone is through bioindicators, examining biological communities and their relationship to their environment. Biomonitoring can be used to detect changes to an environment over time

or space and has become a useful tool for water quality monitoring (Parmar et al., 2016). In particular, phytoplankton, zooplankton, and macroinvertebrates are a few aquatic organisms that have proven to accurately reflect the abiotic conditions (Katsiapi et al., 2016).

At the base of the aquatic food web, phytoplankton have a direct relationship with water quality, especially nutrients. As bioindicators phytoplankton react to changes in their environment rapidly, they have resistance ranges for different parameters and have sensitivities to a variety of different pollutants (Parmar et al., 2016). Phytoplankton will respond to changes with variation in biomass and community composition. For example, when there are excessive levels of phosphorus cyanobacteria will dominate the water column (Downing et al., 2001). Excessive cyanobacteria levels can become problematic as large colonies can be inedible for zooplankton, and some taxa release toxins that are harmful to fish and other organisms (Matsuzaki et al., 2018). Monitoring the phytoplankton community composition and biomass will provide information about its connection to water quality and provide information about its relationship with other organisms.

Next in the food chain, zooplankton feed on phytoplankton and are an important food source for fish. Zooplankton are also useful bioindicators for water quality and eutrophication as they are influenced by abiotic and biotic factors (Parmar et al., 2016). Measuring zooplankton seasonal variation and proportion of different functional groups can provide insight into environmental conditions. For example, large filter-feeding zooplankton like *Daphnia* are associated with high macrophyte, clear-water environments, while rotifers and cyclopoid copepods tend to dominate in a low

macrophyte, turbid-water state (Cottenie et al., 2001). Zooplankton also have sensitivities to salts, temperature, and dissolved oxygen (Swadling et al., 2000). Examining these relationships will provide more information about the impacts of human activity on the biological community.

Another useful aquatic bioindicator are macroinvertebrates. Macroinvertebrates have a variety of functional feeding groups from grazers and scrapers to predators and play important roles within the food web. Macroinvertebrates also have a range of sensitivities to water quality conditions which have allowed for the development of a variety of different metrics to measure the impact of a disturbance on an aquatic environment. These attributes have made macroinvertebrates the most commonly used aquatic bioindicator (Buss et al., 2014). When water quality is less degraded (low nutrient, high oxygen), it is expected that more sensitive taxa such as Ephemeroptera, Plecoptera, and Trichoptera will increase in abundance relative to more degraded environments (Lenat & Barbour, 1993). The macroinvertebrate community is also influenced by the physical structure of the nearshore zone, as the available habitats will dictate which taxa can survive (Brauns et al., 2007). Examining the macroinvertebrate community in the Kawartha Lakes will provide region-specific relationships with water quality and provide baseline metrics for future comparison.

1.5 Research Goals and Objectives

My thesis had two overarching goals: 1) to determine the effect of human activity in a watershed on nearshore water quality in an urbanizing watershed, and 2) assess the impact of shoreline alteration on the lower aquatic food web in the nearshore zone. My work aimed to achieve these goals by focusing on the Kawartha Lakes region, a popular

tourism destination in Southern Ontario with intensive agricultural land use. I first focused on Lake Scugog, a large headwater reservoir, where 12 local volunteers distributed around the lake collected samples from 2017 – 2019. Then expanded with community scientists on Balsam, Cameron, Sturgeon, and Pigeon lakes in 2019 – 2021. Sampling was adapted in 2020 to measure nutrients in the above lakes along with 11 additional lakes from Canal – Katchewanooka. By sampling nearshore sites on each lake, I was able to investigate land use – water quality relationships on multiple scales over a five-year period. I investigated the nearshore biotic communities in a subset of sites on Lake Scugog in 2018 and 2019, and in Balsam, Cameron, Sturgeon, and Pigeon Lakes in 2021. With multiple sampling dates at sites around each shoreline I was able to capture spatial and seasonal variation in the biotic communities and determine the important factors for driving diversity and abundance in each community.

To achieve my research goals, I developed four specific objectives:

1) Determine nearshore water quality spatial and temporal trends and relationship with land use at multiple spatial scales in Lake Scugog.
2) Assess nearshore phytoplankton, zooplankton, and macroinvertebrate communities and their relationship to nearshore habitat in Lake Scugog.
3) Examine spatial trends of nutrients across 16 Kawartha Lakes and determine if the change in human activity during the pandemic influenced nearshore nutrient levels in lakes with high shoreline development.
4) Determine the influence of land use and shoreline structure on nearshore water quality and biological communities in four central Kawartha Lakes: Balsam, Cameron, Sturgeon, and Pigeon.

By exploring nearshore water quality and biotic communities across the Kawartha Lakes the following chapters aim to achieve my research goals and address the identified research gaps. Chapter 2 investigates the spatial trends of water quality in Lake Scugog, demonstrating the influence of land use at multiple spatial scales on nearshore water quality. Chapter 3 reveals the importance of macrophytes in determining diversity and abundance in nearshore phytoplankton, zooplankton, and macroinvertebrates in Lake Scugog. Chapter 4 shows the nearshore nutrient trends across 16 Kawartha Lakes and compares nutrient levels before and during the COVID-19 pandemic in four of the lakes. Chapter 5 indicates the influence of land use on spatial water quality trends in the nearshore area of Balsam, Cameron, Sturgeon, and Pigeon Lakes and provides evidence of direct and indirect influence of human activity on phytoplankton, zooplankton, and macroinvertebrate community composition. My findings in Chapters 2, 4, and the "Water quality and land use" results section in Chapter 5 are based on data from volunteer collected water samples. The findings presented in Chapter 3 and Chapter 5 'Biomonitoring findings' are based on samples collected by myself and research assistants.

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Chapter 2. **Community science-based monitoring reveals the role of land-use scale in driving nearshore water quality in Lake Scugog**

2.1 Introduction

Monitoring of freshwater ecosystems is important for determining the current and changing condition of an ecosystem, as well as documenting the effects of human activities in watersheds. Land-use within a watershed is an important driver for lake water quality, with both urban and agricultural land use leading to increased nutrient levels (Howell et al., 2012; Nielsen et al., 2012; Read et al., 2015). Small-scale shoreline development can also result in increased nutrient, particulate matter and chloride levels (Howell et al., 2012; Rosenberger et al., 2008). Comparisons have been made between catchment and buffer-scale land use, but these studies are largely limited to riverine systems, and there is no consensus on the relative importance of catchment scales on water quality in lakes (Liu et al., 2017; Meador & Goldstein, 2003; Sliva & Williams, 2001). Additionally, few studies have compared the impact of shoreline and catchment-scale land use on nearshore water quality in lakes.

The best way to capture land use impacts on a lake is by monitoring at the location where the land-use effects are first experienced - the littoral or nearshore zone. As the first point of entry for stormwater run-off and subsequent erosion, the nearshore zone serves as a receiving and mixing zone for both point and non-point source contaminants. A nearshore water quality study conducted in Lake Ontario found that shoreline land use influenced highly variable nearshore water quality, but the effects diminished with distance from the shoreline into the lake (Howell et al., 2012). Other work has directly compared predictive models for nutrient concentration based on land

use and land cover at catchment and buffer scales. Nielsen et al. (2012) found the catchment scale was best at predicting lake nutrient level, however their maximum buffer size was 400 m. Soranno et al. (2015) found the 1000 m and 1500 m buffer scales were comparable to catchment scale models, and had moderately strong relationships with nutrients. However, they also found these relationships varied regionally and determined that further multi-scale studies are required to resolve the issue of land use scale. By targeting the nearshore zone, I expect to find a strong signal of watershed land use, however this signal will be dampened by mixing with the pelagic zone and sediment resuspension from wave action which can also act as an important source of nutrients in the nearshore zone (Nielsen et al., 2012; Read et al., 2015). The high nutrient and light availability in the nearshore zone contribute to a complex, but essential habitat for most aquatic organisms at some point in their life cycle (Vadeboncoeur et al., 2011). Despite the known importance of the nearshore zone, it is rarely monitored regularly for water quality (Vadeboncoeur et al., 2011).

Part of the deficiency in nearshore monitoring programs is lack of access to shallow sites adjacent to privately owned shorelines. Many lakes have extensive shoreline development for cottagers and permanent residents, which greatly reduces public access to the shoreline by land. In productive, shallow lakes, it can be difficult to access the nearshore zone by boat, especially in weedy lakes. One solution to this problem is community science, also known as “citizen science”. In recent years, there has been a shift to using the term “community science” to be more inclusive of volunteers with diverse backgrounds. In effect, any member of the local community who is willing and capable, can participate as a volunteer. By working with local community members, I can

gain access to otherwise inaccessible areas, such as privately-owned waterfronts. Increasingly, water quality monitoring programs have expanded to include community science initiatives, whereby local volunteers collect the water samples and record field observations (Jollymore et al., 2017; Poisson et al., 2020).

Community science monitoring programs vary by project, but typically involve volunteer recruitment, training, and sample collection, followed by processing and analysis by academic researchers or government agencies (Fernandez-Gimenez et al., 2008). One of the biggest barriers facing community science research is a misconception that data collected by lay persons is not as accurate or valid as data collected by skilled professionals, but community-science programs with effective training and data collection protocols generate the same quality of data as professional researchers (Canfield Jr et al., 2002; Freitag et al., 2016; Scott & Frost, 2017). By implementing a community science approach, research programs can capture more spatially explicit data with an increased number of sites and frequency of sample collection, without an increase in cost for hiring additional research personnel (Dickinson et al., 2010).

The present study reflects a co-production model whereby a lake stewardship group, a regional watershed authority, and a university laboratory worked together to coordinate, collect, and evaluate nearshore water quality in Lake Scugog, Ontario, Canada over 3 years (2017-2019). Notwithstanding the notable value of the lake to its residents and visitors alike, Lake Scugog has largely been understudied. Recent work has examined drivers of *Microcystis* blooms in the lake and found both significant biotic and abiotic drivers, including chloride, precipitation and total nitrogen (Tyler Harrow-Lyle &

Andrea E Kirkwood, 2020). In order to fill important data gaps surrounding ecosystem health in Lake Scugog, this nearshore monitoring project aimed to (1) establish current nearshore water quality conditions and trends in Lake Scugog, and (2) determine how nearshore water quality is influenced by land use at multiple scales.

2.2 Methods

2.2.1 Study Site

Lake Scugog is a 68 km^2 headwater reservoir in the Trent-Severn waterway with a mean depth of 1.4 m (Kawartha Conservation, 2010). Previously a riverine-wetland complex, Scugog was formed in the 1830s after the construction of a grinding mill on the Scugog River. Post dam construction, the lake became a flow-regulated reservoir with two basins extending north-south, split in the middle by Scugog Island. The Lake Scugog region serves as a bedroom community to the Greater Toronto Area (GTA), but also attracts tourists and fishers (Outdoor Canada, 2011). Since the 1950s, there has been increasing shoreline development and expansion of urban development in the watershed (Kawartha Conservation, 2010). Scugog has a 529.7 km^2, agriculturally dominated watershed, with 79% of the drainage basin within the Region of Durham and the remaining 21% within the City of Kawartha Lakes (Figure 2.1). The Township of Scugog is within the Region of Durham, and accounts for 70% of the watershed with a population of 21,439. Approximately 45% of Scugog residents live in the urban area of Port Perry, while the rest of the population is located in exurban communities, including 16 small hamlets located around the lake, many of which are shoreline communities (Kawartha Conservation, 2010).

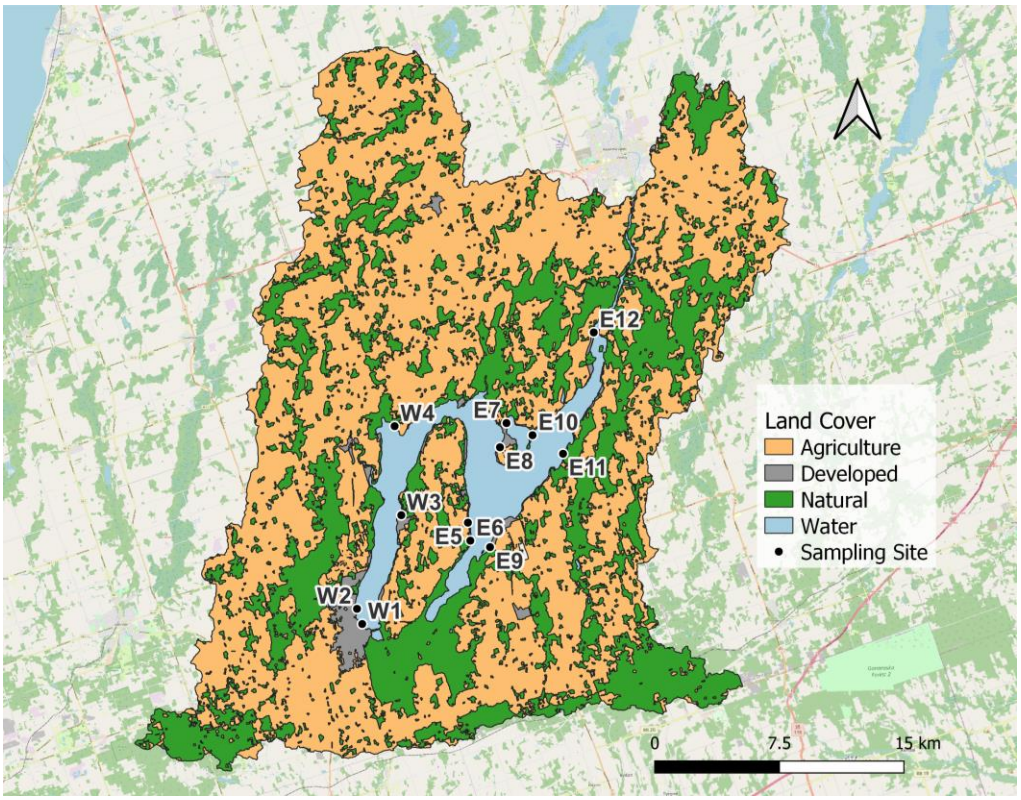

Figure 2.1 Map of Lake Scugog and surrounding watershed with delineated land use categories. The 12 study sites are labelled as black circles.

2.2.2 Volunteer Recruitment and Training

This project was born out of a pre-existing relationship between Ontario Tech researchers, Kawartha Conservation, and the Scugog Lake Stewards. As part of its mandate, Kawartha Conservation monitors and manages local lakes and tributaries, including Lake Scugog. The Scugog Lake Stewards are a volunteer stewardship group that promote research and conservation activities in Lake Scugog. All groups had a distinct, but inter-connected role in the co-production model developed for this project (Figure 2.2). In 2017, volunteer recruitment was targeted at residents in the Township of Scugog via Kawartha Conservation and Scugog Lake Stewards social media posts, as well as at municipal committees including the Scugog Environmental Advisory

Committee and the Healthy Lake Scugog Steering Committee. Upon completion of the recruitment period, 12 volunteers were confirmed, with four volunteers located on the west arm and eight volunteers on the east arm of the lake.

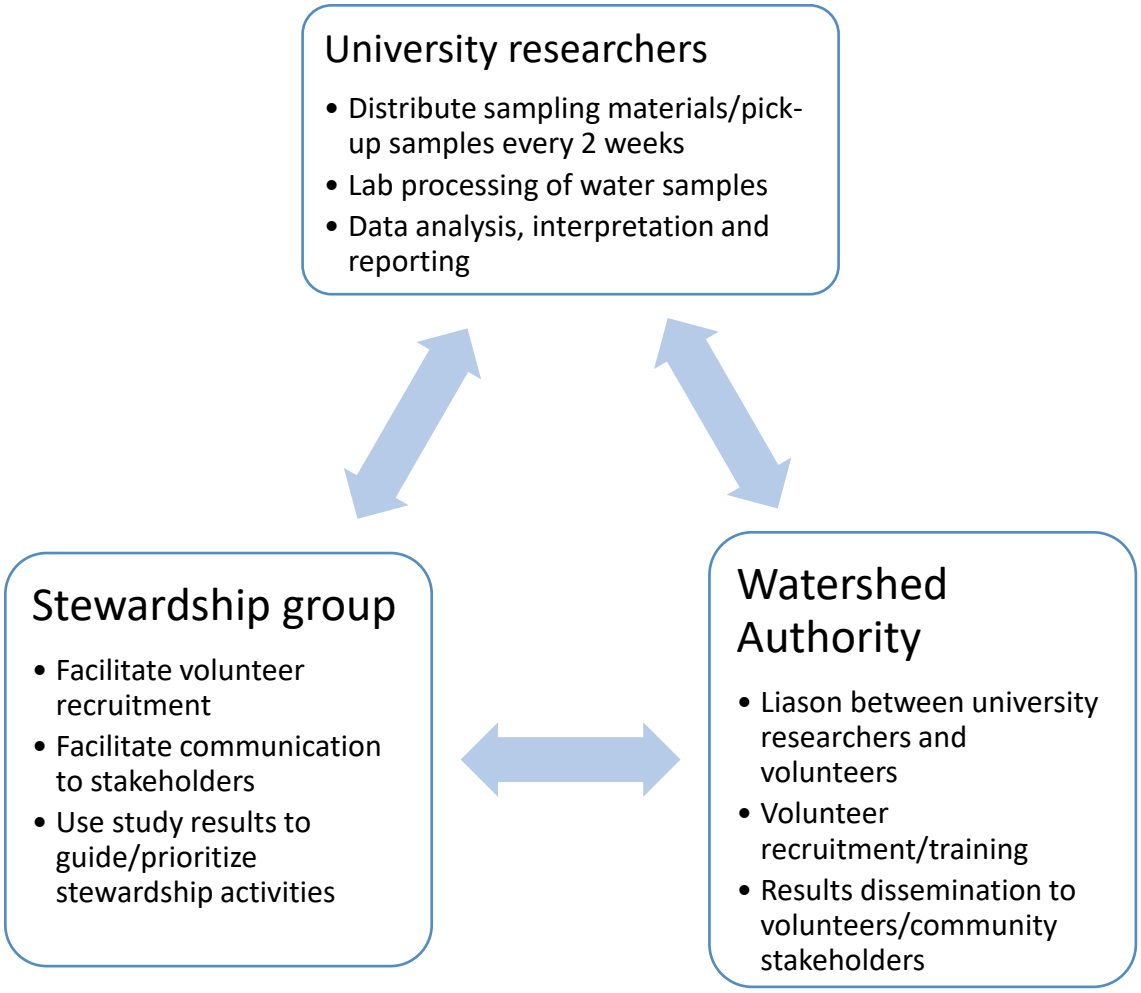

Figure 2.2 The community science co-production model involved a triad of research partners including a university (Ontario Tech), watershed authority (Kawartha Conservation), and a volunteer-run stewardship group (Scugog Lake Stewards). Each partner had a distinct, but important role in the execution and delivery of the water quality monitoring program from study conception to dissemination of results

Volunteer training is an essential step to for quality assurance in community science projects (Freitag et al., 2016). Volunteer training sessions occurred in spring 2017, and were conducted by a water quality specialist and volunteer liaison from Kawartha Conservation individually at the volunteer's dock. Training sessions included an explanation of the research project as well as a background briefing on Lake Scugog, water quality, and the methodology used to collect the samples. A sampling spot was selected based on a standard depth of 1 m. Volunteers were left with their first set of sampling equipment: two acid-washed 1 L Nalgene™ bottles, one sterile specimen cup, datasheets, gloves, a cooler bag, an ice pack, and a thermometer. The volunteer liaison at Kawartha Conservation reminded volunteers, by phone or email, 24 h in advance of each sampling event. By providing detailed background information about the project, demonstrating proper sample collection techniques, and selecting sampling sites our volunteer training methods provide some assurance of the quality of samples collected by the volunteers.

Water samples were collected between 08–09 h biweekly, from the end of May to early September. Sample collection was limited to between 08-09 h to avoid impacts of activities such as swimming and boating on lake water, and provided a time most volunteers were home and available to collect samples. Samples were stored in a cooler bag with an ice pack and placed in a shaded spot in front of the volunteer's home for university researchers to pick-up. During sample pick-up, two 1 L Nalgene™ bottles, and one sterile specimen cup were left for the next sampling event. Once retrieved, samples were kept on ice and transported to the laboratory for processing. All processing was done within 24 h of collection. At the end of each sampling season, volunteers received a

preliminary report of water quality results for their property, a thank-you card from the research team, and communications regarding next steps in the research project.

2.2.3 Water Sample Processing

Water samples from the Nalgene bottles were processed for chlorophyll a (Chla), total suspended solids (TSS), chloride, total phosphorus (TP), total dissolved phosphorus (TDP), and a nitrogen suite analysis to determine total nitrogen (TN). TSS was determined by filtering 250 ml of each water sample through a 47-mm GF/A glass microfiber filter, that had been pre-filtered with Milli-Q water. Chloride was measured directly with an Orion ion-selective electrode (ThermoFisher Scientific, Waltham, MA USA). A Genesys 20 Thermo Spectronic UV-vis spectrophotometer was used for the determination of Chla, which was extracted by filtering 300 mL of each lake water sample through a 47-mm GF/A glass microfiber filter. Chla was extracted from the filter using a modified 90% acetone extraction method (Kirkwood et al., 1999). TDP samples were filtered through 0.2-µm Whatman nylon membrane filters (GE Healthcare, Chicago, IL, USA). Total and dissolved phosphorus were determined using the modified ascorbic-acid method of Murphy and Riley (1962). Frozen nitrogen samples were shipped to an accredited lab (SGS Canada, Lakefield, Ontario, Canada) for nitrogen suite analysis to determine TN. Sterile specimen cup samples were used to determine total coliforms and *Escherichia coli* using the Colilert method (Edberg et al., 1990).

2.2.4 Land Use Classification

Land use data were determined using the open-source geographic information system software QGIS (QGIS Development Team, 2019). Watershed and land use files were sourced from Natural Resources Canada, land cover data is from 2001, however land cover patterns from the data file closely matched current land use patterns (Natural Resources Canada, 2015). Land cover was categorized into one of three categories: natural land cover, agricultural land use, or developed land use. Area and percentages for each category were determined at a 500 m and 1000 m radius around each site and for each sub-catchment zone (Figure 2.3). The sub-catchment area was determined using a digital elevation model (DEM) from Ontario Ministry of Natural Resources and Forestry (2019). Upslope area was calculated from an outlet adjacent to each sampling site, all outlets were located within 500 m of the sampling location.

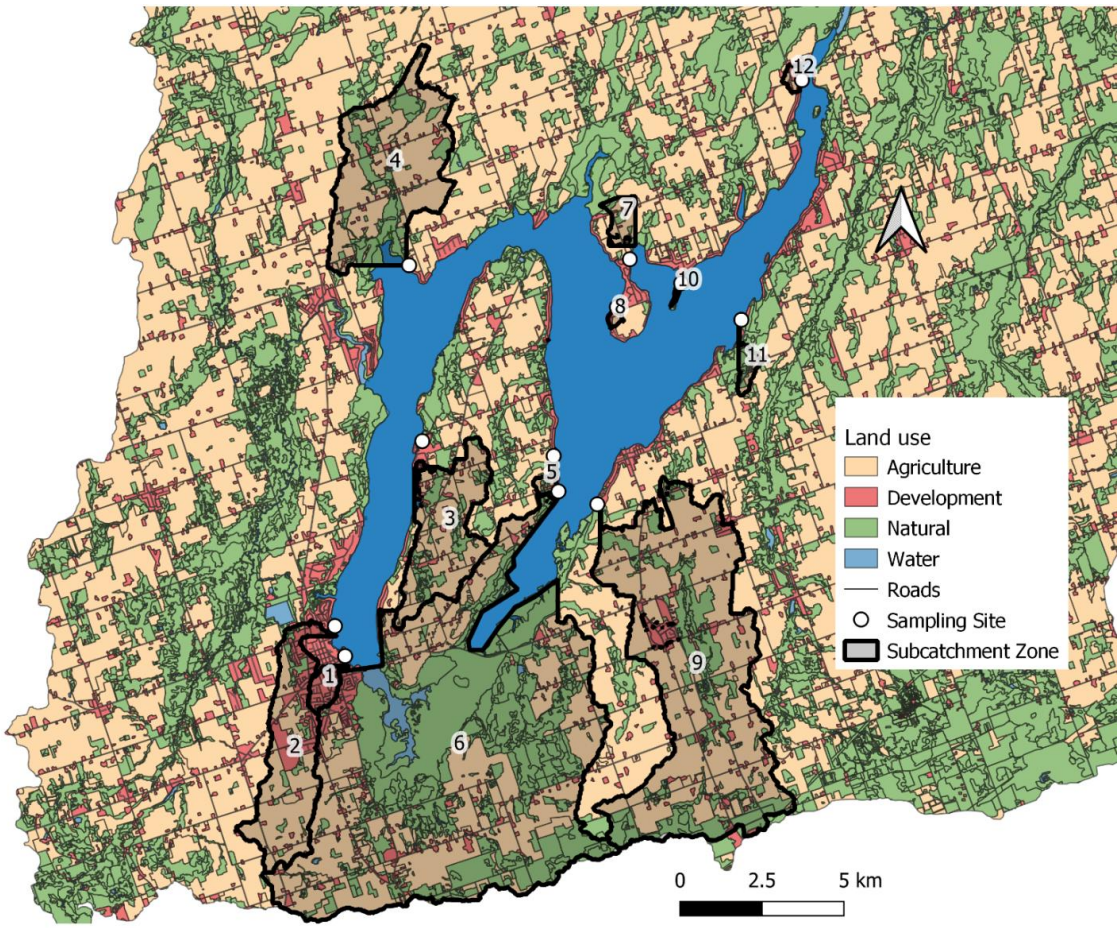

Figure 2.3 Map of Lake Scugog watershed with sub-catchment zones outlined for each sampling site.

2.2.5 Statistical Analysis

Statistical analyses were conducted using R Studio version 3.6.1 (RStudio Team, 2016). Exploratory data analysis was conducted to identify outliers that may have been due to sample collection or processing errors. Values that fell out of the 1.5 times interquartile range (IQR) were examined to determine if there was an error that caused the value. No values that fell out of the IQR appeared to be due to sample collection or processing error and were thus kept for subsequent analysis. Parameters that did not have a normal distribution of residuals or exhibited heteroscedasticity were log-transformed

for analysis (Chla, fecal coliforms, and *E. coli*). A principal component analysis (PCA) was conducted to reduce dimensionality to better examine drivers of nearshore water quality. Prior to PCA, data were center standardized by first subtracting the mean value for the variable and then dividing by its standard deviation. The PCA was prepared using code written based on the FactoMineR package (Husson et al., 2013). Analysis of variance (ANOVA) was conducted to compare yearly trends in water quality variables, with significance at the $p < 0.05$ level. A Tukey's post-hoc test was conducted where significant models were found to determine specific relationships between years. Step-wise linear regressions were run on each water quality parameter (Nielsen et al., 2012). The final models presented are those for which the inclusion of land use parameters based on minimum Akaike Information Criterion (AIC) (Akaike et al., 1973).

2.3 Results

2.3.1 Water Quality

Nearshore water quality in Lake Scugog varied by location, with a few nutrient and contaminant hotspots (Table 2.1). In particular, sites W1, W2, W3, W4, E5, and E6 had elevated levels of TP and Chla compared to the rest of the sites. Average TP was above the provincial water quality objective (PWQO) of 20 µg/L for 75% of the sites sampled. TP and Chla were also significantly correlated ($r = 0.39$, $p < 0.05$). Chloride was also elevated at sites W1 and W2, both located in the Township of Port Perry. Fecal coliform and *E. coli* levels only exceeded the PWQO of 100 MPN on five occasions over the three-year sampling program, although average levels were slightly elevated at some sites, they were not at levels of concern.

Table 2.1. Seasonal means (May-September) reflecting three years of pooled data (2017-2019) for eight water quality parameters measured in Lake Scugog, Ontario. Standard deviations are reported in brackets.

Site	n	TP (µg/L)	TDP (µg/L)	Chla (mg/L)	TN (mg/L)	Coliform (MPN)	E. coli (MPN)	TSS (g/L)	Chloride (mg/L)
W1	23	33.6 (17.7)	4.9 (2.7)	16.8 (18.6)	0.60 (0.39)	82 (146)	31 (64)	0.014 (0.022)	49.9 (42.5)
W2	19	42.4 (21.2)	8.2 (7.8)	9.7 (6.6)	0.67 (0.34)	43 (56)	37 (106)	0.017 (0.47)	75.1 (60.7)
W3	15	32.3 (9.7)	4.6 (4.1)	10.5 (5.5)	0.58 (0.13)	55 (37)	33 (22)	0.006 (0.005)	15.3 (22.1)
W4	24	29.8 (11.6)	4.6 (3.2)	9.8 (5.2)	0.85 (0.43)	46 (43)	27 (42)	0.006 (0.006)	17.4 (55.5)
E5	23	30.4 (16.0)	4.6 (3.5)	12.6 (12.5)	0.81 (0.59)	46 (52)	13 (16)	0.006 (0.004)	11.4 (17.1)
E6	23	52.2 (19.0)	7.5 (8.3)	18.0 (33.6)	0.81 (0.29)	88 (130)	38 (85)	0.043 (0.145)	6.9 (12.4)
E7	22	21.8 (10.5)	3.4 (3.2)	6.3 (3.9)	0.68 (0.31)	52 (72)	14 (10)	0.006 (0.004)	7.1 (13.9)
E8	24	14.7 (8.0)	4.8 (6.6)	5.8 (3.8)	0.51 (0.33)	32 (28)	18 (21)	0.007 (0.006)	10.0 (16.1)
E9	20	14.6 (8.3)	5.0 (6.0)	6.2 (4.0)	0.45 (0.36)	40 (60)	15 (15)	0.010 (0.006)	9.8 (15.7)
E10	24	19.4 (10.4)	6.4 (10.3)	9.8 (10.8)	0.49 (0.34)	46 (69)	16 (20)	0.010 (0.007)	9.6 (16.3)
E11	16	18.9 (7.5)	11.3 (20.1)	6.9 (4.4)	0.46 (0.21)	59 (71)	20 (24)	0.026 (0.073)	4.7 (12.5)
E12	23	23.4 (10.7)	7.4 (10.7)	11.0 (13.0)	0.67 (0.35)	58 (69)	31 (59)	0.048 (0.168)	9.6 (15.1)

To further explore water quality patterns across the lake, a PCA was conducted and biplot produced (Figure 2.4). The first axis (PC1) explained 22.8 % of water quality variation across sites. PC1 was significantly driven by all water quality variables included in the analysis, with Chla, TP, and TN having the strongest influence on the axis (Table 2.2). There was some site separation on PC1, with lower site numbers (W1-W4, E5, E6) scoring slightly higher on the axis and high sites (E7-E12) scoring lower. This separation matches what was found from the site averages.

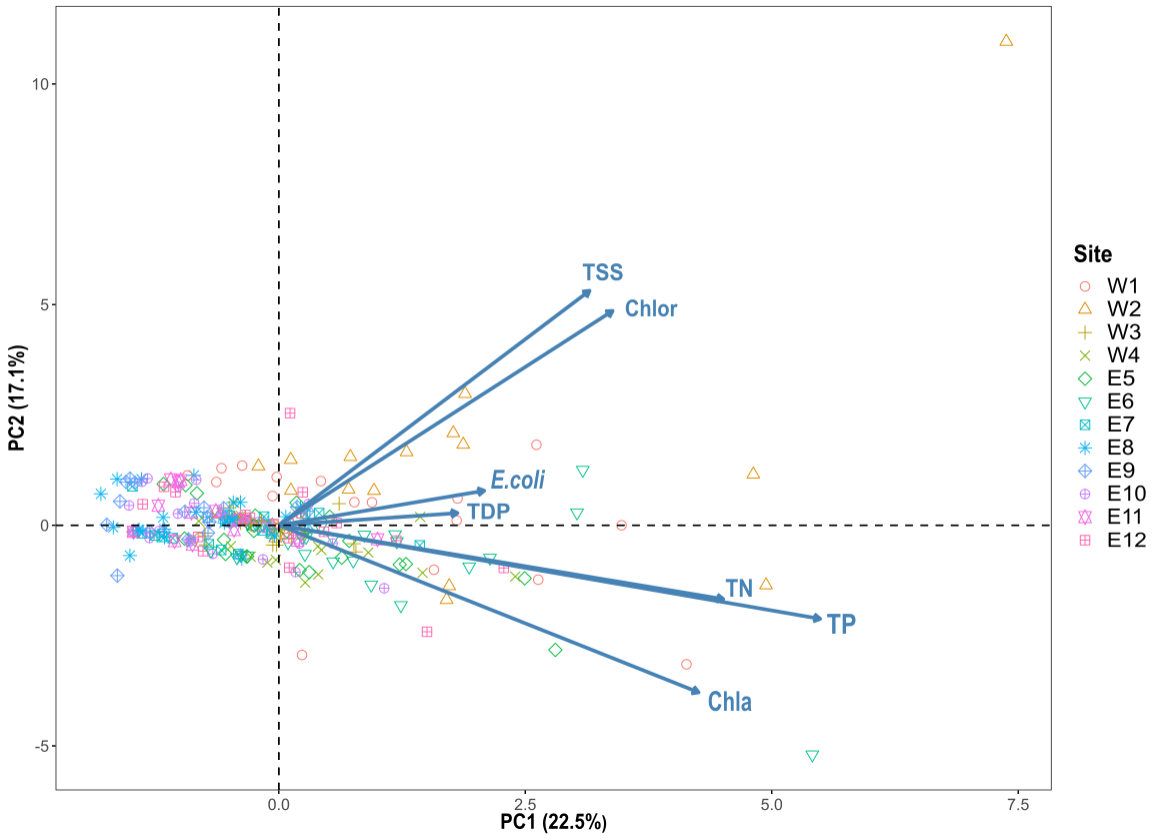

Figure 2.4 Principal component analysis biplot visualizing water quality profile across study years, data are labelled by site. Direction and length of arrow indicates the direction and correlation of each water quality parameter with the axes. TSS = total suspend solids, Chlor = chloride, TDP = total dissolved phosphorus, TN = total nitrogen, TP = total phosphorus, Chla = chlorophyll-a.

PC2 explained 17.1 % of variation in nearshore water quality, and was positively driven by chloride and TSS (Table 2.2). TP, Chla, and TN, were negative drivers of this axis and this separation may indicate different drivers from chloride and TSS. There is some site separation on PC2 with sites W1 and W2 scoring slightly higher than other sites, indicating there is likely one or more latent variables driving water quality at these sites.

Table 2.2. Parameter scores for the first two axes of the principal component analysis. Data was center-standardized and blank cells indicate water quality variables that were not statistically significant ($p < 0.05$) on the axis.

	Axis 1	**Axis 2**
TP	0.71	-0.25
Chla	0.56	-0.49
E. coli	0.30	
TN	0.54	-0.19
Chloride	0.44	0.60
TSS	0.37	0.70
TDP	0.24	
Eigenvalue	1.6	1.2
Variability explained (%)	22.8	17.1

An ANOVA was used to determine if water quality parameters varied by year. *Escherichia coli* and fecal coliforms were the only water quality variables with significant differences between years. A Tukey's post-hoc test found that 2017 and 2018 were both significantly higher than 2019 ($p < 0.05$). Climate was suspected to be driving the changes and water and air temperature were also compared across years, but there were no significant relationships. However, the occurrence of a storm event prior to sample collection was compared to *E. coli* and fecal coliform levels, and it was found that

fecal coliform levels were significantly higher following a storm event compared to no storm event, although there was no statistical significance found for *E. coli* levels ($p < 0.05$) (Figure 2.5).

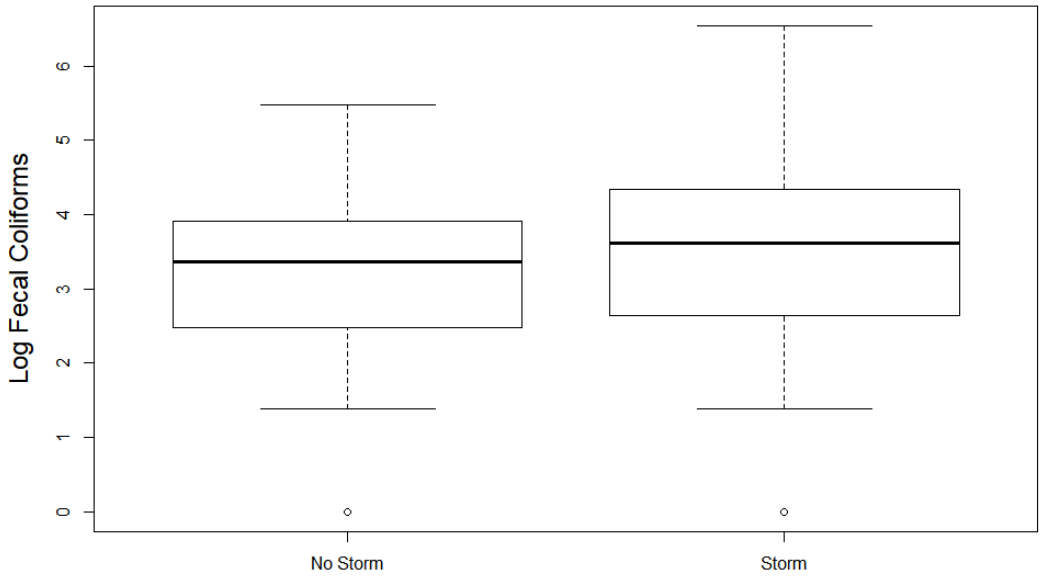

Figure 2.5 Boxplot of log transformed fecal coliforms with a storm event (>15 mm precipitation, n = 117) in the two weeks prior to sample collection or no storm event (n = 139) prior to sample collection.

2.3.2 Land Use

Land use type in the Lake Scugog watershed varied by scale and site location (Figure 2.6). Development in the Township of Port Perry is prevalent at sites W1 and W2. These sites had the highest levels of development at the 500 m and 1000 m scales. Agricultural and natural land cover dominated most sites, varying in proportion by scale. At the sub-catchment scale, agricultural land use was dominant at six sites, followed by natural at four sites, and development at two. At the 1000 m buffer scale there was a slight shift with six sites having primarily agricultural land use, and three of each with natural land cover and developed land use. Additionally, most sites had an increase in the

proportion of developed land use at the 1000 m buffer scale. This pattern continues for the 500 m buffer, where all sites had an increase in the proportion of developed land use. At the 500 m buffer scale, seven sites were dominated by developed land use, with three and two dominated by natural and agriculture land use, respectively. Another notable pattern at the buffer scales was the change in agricultural land use, where 11 out of 12 sites decreased in agricultural land use from the 1000 m buffer to the 500 m buffer. The increase in developed land at a smaller scale matches with the general pattern of land use on Lake Scugog, where most development is occurring adjacent to the shoreline, while the rest of the watershed remains dominated by agricultural land use.

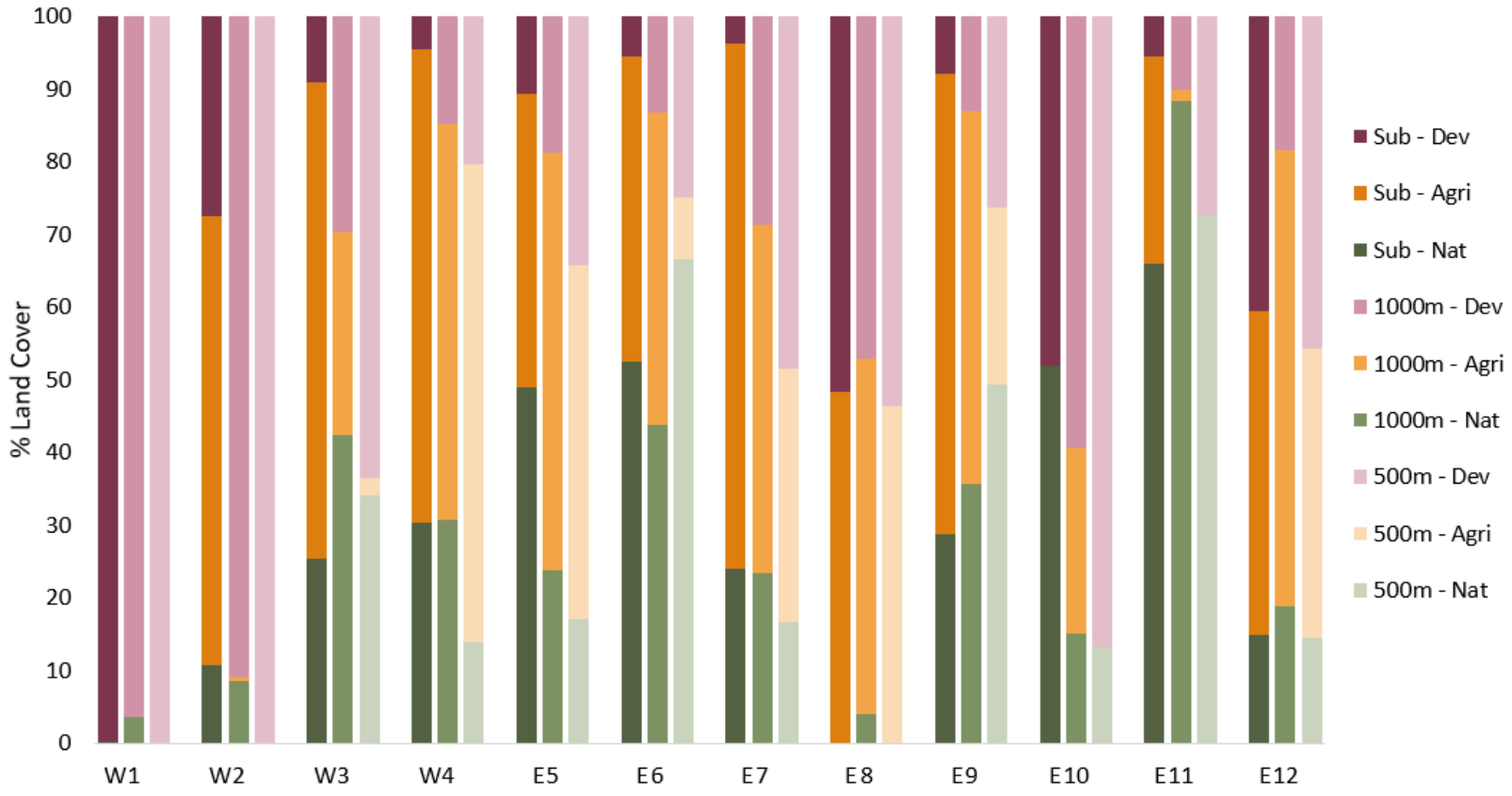

Figure 2.6 Bar plot of land use proportions for each sampling site at each scale (sub-catchment, 1000 m buffer, and 500 m buffer).

Land use proportions provide important information about the trends of land use in the watershed, but it is also important to consider the area of each land use type. The sub-catchment area for the sites varied substantially across the lake (0.07 – 74.02 km^2) (Figure 2.3). The range of sub-catchment areas is an important consideration because information can be lost when converted to relative proportions. One example of this is the sub-catchment for site W1. Proportionally, development is the highest, accounting for 100% of land use at the sub-catchment scale, but area-wise it is lower than three other sites (W2, E6, and E9). At the buffer scales, there is less variation due to buffer size standardization, however, the amount of land in the buffer did vary slightly by site. At the 1000 m scale, the average area was 1.60 ± 0.29 km^2, and at the 500 m scale, average area was 0.41 ± 0.08 km^2. These patterns indicate the importance of including area in predictive models.

2.3.3 Landscape – Water Quality Linkages

Step-wise linear regression models were used to examine the relationships between land use and water quality in the nearshore zone (Table 2.3). It is evident that land use scale has an impact on the relationship between water quality parameters and land use type. Chloride had the strongest and most consistent relationships with land use, where significant drivers were found at both buffer scales accounting for 75% and 47% of variation at the 1000 m buffer, and 500 m buffer scales, respectively. Although *E. coli* had marginally significant models at all three scales, there was no evident trend of increasing model strength by scale. Fecal coliforms were also marginally significant at both the sub-catchment and 500 m buffer scale. TDP was explained by each buffer scale, both explaining 27% of variance in the parameter. The rest of the parameters were only

statistically significant at a single scale. TP, suspended solids, and Chla, had significant land use predictors at the sub-catchment scale. Alternatively, TN only had significant land use predictors at the smallest, 500 m buffer scale.

Table 2.3. Results of forward stepwise linear regression to assess land-use type and scale as predictor variables for water quality parameters. All final models presented below are based on optimal AIC, final models that retained land use predictors are shown. Predictor variable statistical significance is denoted as follows: * $p \leq 0.05$, ** $p \leq 0.01$, *** $p \leq 0.001$.

Scale	Parameter	Development	Agriculture	Natural	Area	Roads	R^2	p
Sub-catchment	TP	+*					0.31	**0.034**
	Suspended Solids			+			0.16	0.105
	Chla		-	+		+	0.19	0.154
	Fecal coliforms	-	-	+		+	0.37	0.126
	E. coli	+					0.13	0.139
1000 m buffer	Chloride	+***					0.75	**<0.001**
	E. coli					+	0.10	0.172
	TDP		-	+			0.27	0.100
500 m buffer	Chloride	+**					0.47	**0.008**
	Fecal coliforms		-				0.10	0.156
	E. coli					+	0.23	0.065
	TDP	-	-*				0.27	0.101
	TN		+				0.16	0.11

Most notably, land area of sub-catchment and buffer scale was not an important predictor of water quality in the nearshore zone. In contrast, all land use variables were statistically significant in at least one of the predictive models, and development was included in the most models (6/13). Development was statistically significant in models for TP, TDP, and chloride. Road length, another indicator of development, was significant in models for Chla, and marginally significant for fecal coliforms and *E. coli*. Except for TDP, all of these relationships were positive, indicating the negative impact of development on water chemistry. Natural land cover was also an important predictor, having a positive relationship with suspended solids, fecal coliforms, *E. coli*, and chloride. Agricultural land use was included in models for Chla, fecal coliforms, TDP and TN.

2.4 Discussion

2.4.1 Water Quality

The water quality profiles revealed in Lake Scugog's nearshore zone show notable links with land use in the watershed at various catchment scales. They also serve as much needed baseline information for nearshore water quality condition. Previous work on Lake Scugog has focused on overall lake health, and indicates that Scugog has been eutrophic for a long time. For example, Agbeti (1992) measured TP at 42.5 µg/L, whereas Robillard and Fox (2006) reported an average TP of 27.0 µg/L based on data from 1980-2003. Interestingly, their yearly trends indicate a notable decline in lake TP concentration after 1990, mostly below 20 µg/L. Additionally, monitoring data from Kawartha Conservation confirms Lake Scugog shifted from an eutrophic lake to a mesotrophic lake in the early to mid-2000s, with average TP reported as below 20 µg/L

(Kawartha Conservation, 2010). In contrast, our findings from a more recent time period indicate that Scugog has shifted back to a eutrophic state, with TP levels well above 20 µg/L. This shift is supported by recent off shore monitoring by Harrow-Lyle and Kirkwood (2020), which also found elevated TP levels throughout the lake in the eutrophic range.

The positive relationship detected between TP and Chla indicate that phosphorus may be a limiting macronutrient for algae growth in Lake Scugog's nearshore. Both nutrients and Chla concentrations were highest in the western basin of the lake, at sites W1-W4. Sites W1-W3 are located on the southern portion of the west basin, which has the highest development associated with the Township of Port Perry. Sites E5 and E6 are on the west side of the east arm of the lake, and also had high levels of most water quality parameters, likely due to the high agricultural activity in their sub-catchments. Spatial patterns in water quality were reflected in the PCA biplot (Figure 2.4), where PC1 showed a clustering of sites with high nutrients and Chla. This result is unsurprising, given that previous monitoring of off-shore sites in Lake Scugog had elevated nutrients near these study sites (Kawartha Conservation, 2010). In addition, PC2 revealed the importance of urban development influencing TSS and chloride at sites W1 and W2 in the Town of Port Perry. Although parameters other than TP are not currently at levels of concern, they should continue to be monitored to ensure continued land development does not result in water quality impairment such as excessive nutrients and algal growth.

Temporal trends in nearshore water quality were only detected for fecal coliforms and *E. coli*, with 2019 being significantly lower than both 2017 and 2018. It has often

been found that *E. coli* levels are related to precipitation, and with major storm events in particular (Noble et al., 2003; Tornevi et al., 2014). Although I did not detect a statistically significant effect of the storm on *E. coli*, fecal coliforms were significantly elevated after a documented storm event (> 15 mm precipitation in 24 h). It is expected that as a result of climate change there will be an increase in frequency and intensity of rainfall in the region, which could have an impact on fecal coliform and *E. coli* levels in Lake Scugog (Eimers et al., 2020).

2.4.2 Land Use Patterns

Examining the effect of land use on water quality at buffer and sub-catchment scales provided key information about land use patterns around Lake Scugog. There was a clear shift in land use type when scaling up from the 500 m buffer zone to the sub-catchment scale. The shoreline areas contain the majority of development in the watershed, while the rest is composed of agricultural land use and natural land cover. Most of the changes in land use classification from 500 m to sub-catchment scale includes a decrease in developed land use and an increase in agricultural land use. This pattern of increasing development with proximity to the shoreline is a common land use pattern for many lakes, especially in areas that are popular for residents and cottagers (Howell et al., 2012; Schnaiberg et al., 2002).

The variation in land use also highlights the importance of studying land use from a multiscale perspective. Although popular for river and stream studies, this approach can help elucidate land use relationships when combined with nearshore water sampling in lakes (Liu et al., 2017; Pratt & Chang, 2012). Howell et al. (2012) found that shoreline

land use had impacts on shoreside water quality with the strength of this impact dropping off by distance from the shore. Other work has examined different scales, and found that larger buffer zones were better at predicting nutrient concentration than small buffers (Nielsen et al., 2012; Soranno et al., 2015).

2.4.3 Water Quality – Land Use Relationships

The regression models provide insight to the importance of each land use scale in influencing each water quality parameter. The sub-catchment scale was the only scale to produce a significant model for TP, indicating that the small-scale changes in land use near the shoreline were not as influential as the sub-catchment for this important nutrient. Since TP is already above the PWQO, these findings indicate that phosphorus mitigation should target watershed-scale development activities including erosion from construction activities, lawn and garden fertilizer application, and septic systems. These findings are not surprising as other studies have found the importance of watershed land use for TP, but the contribution of development and agricultural land use vary by study. Our findings are consistent with Howell et al. (2012), which found urban development was the most important watershed feature for nearshore TP. However, in many other studies agricultural land use has also been included in watershed level models of TP (Cross & Jacobson, 2013; Kim et al., 2016). It is noteworthy that given the large proportion of agricultural land use at the sub-catchment scale for most sites, agriculture was not an explanatory variable of water quality in this study.

At the buffer scales, TDP and chloride had statistically significant land use predictors, but only chloride had statistically significant models. The detection of

significant predictive relationships at the 1000 m and 500 m scales, but not the sub-catchment scale, provide evidence of the importance of small-scale land use for nearshore water quality. Although chloride levels in Lake Scugog are not currently at or near the PWQO, the relationship with shoreline development indicates the potential that these concentrations will increase as shoreline development and urbanization in Lake Scugog's watershed continues. The contribution of chloride from developed land is unsurprising since de-icing salt application on roads, driveways, and parking lots is common in urban areas (Hill & Sadowski, 2016; Scott et al., 2019). Moreover, it has been found that chloride retention is higher in urbanizing watersheds compared to heavily urban watersheds (Oswald et al., 2019). The direct connection of chloride pollution with shoreline development found in this study provides a rationale to target small-scale land use activities such as road salt application when developing and implementing mitigation initiatives.

The inclusion of the buffer scale also revealed the importance of shoreline-adjacent agriculture land as an explanatory variable for TN levels. Other work has found that near stream buffers were important for determining TN concentrations, especially for agriculturally dominated landscapes (Christensen et al., 2013; Stachelek et al., 2020). Although not statistically significant, the relationship detected in this study indicates the importance of the shoreline buffer at sites with a primarily agricultural sub-catchment.

Fecal coliform and *E. coli* also had marginally significant relationships with land use at multiple scales. In particular, development and roads were important at the sub-catchment and 500 m scales for *E. coli.* One potential explanation for this pattern is the

prevalence of both septic systems and manicured lawns at residential properties around the lake. Large manicured lawns tend to attract Canada Geese, which can be a major local source of coliform bacteria (Guerena et al., 2014). In addition, it is suspected that some of the septic systems may be old or not properly maintained which could result in contribution of *E. coli*, especially following storm events (Iverson et al., 2017; Sowah et al., 2014). The relationship with roads at the 500 m scale could be indicative of peri-urban lake-front communities impacting nearshore water quality.

Overall, agriculture and urban development in the Lake Scugog watershed are, in combination and at varying spatial scales, influencing nutrient and other contaminant concentrations in the nearshore zone. Considering that other popular lakes in the region have similar land use patterns in their watersheds, these findings may provide some insight into the possible drivers of nearshore water quality in those systems as well. Further confirmatory studies on those lakes will support a deeper understanding of the role of land use type and scale on the nearshore zone.

2.4.4 Management Implications

Our three-year study confirms that nearshore water quality monitoring can be effectively executed using a community science approach that incorporates a co-production model involving a local stewardship group, watershed authority, and scientific institution. Having a three-pronged research partnership created a sustainable monitoring program with multiple levels of built-in support. This co-production model, along with best practices that have been described in detail by others (Djenontin & Meadow, 2018;

Norström et al., 2020), could be adopted by any lake association with ties to lake managers and local colleges or universities.

Another key finding is the relative importance of buffer vs. catchment-scale land use influence on nearshore water quality in Lake Scugog. It is evident that local shoreline and urban development is an important driver of chloride and total phosphorus in the nearshore zone, which has important implications for aquatic organisms and lake trophic status. To maximize water chemistry mitigation on a limited budget, lake managers should target shoreline properties and developed areas in Lake Scugog as priority areas for stewardship activities, such as shoreline naturalization, septic system testing and repair, and alternative lawn and garden maintenance practices. Focus on tangible restoration activities at the local scale not only targets the source of water chemistry degradation, but provides an opportunity for shoreline residents to engage in meaningful stewardship of their beloved lake.

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Chapter 3. **Nearshore plankton and macroinvertebrate community structure is strongly associated with macrophyte abundance in a large lake with high shoreline development**

3.1 Introduction

An increase in human activity along the shorelines of lakes is altering shoreline structure and impacting the aquatic communities that rely on it for habitat (Brauns, Garcia, Walz, et al., 2007; Hicks & Frost, 2011; Porst et al., 2019). Therefore, it is imperative to focus on the spatially heterogenous nearshore area to measure the impact of human activities and provide insights for habitat conservation. There are several advantages of targeting the nearshore zone for monitoring because it can (1) offer insight on the effects of land-use and shoreline modification at the point of run-off entry into a lake, (2) fill in data gaps about a part of the lake where biological production tends to be highest, and (3) serve as an early-warning of change in ecological condition related to lake health (Chang & Rossmann, 1988; Vadeboncoeur et al., 2011). As the transition zone between the shoreline and offshore waters, changes in the nearshore zone like excessive nutrient loading and increased primary productivity can forecast eutrophication in the offshore waters (Chang & Rossmann, 1988; Lambert & Cattaneo, 2008).

Human activities in a lake's watershed can influence lake water quality, particularly as land-use area and intensity increases. Most types of land development lead to nutrient enrichment and, in some cases, contamination with metals or bacterial pathogens (Burant et al., 2018; Howell et al., 2012; Read et al., 2015). The nearshore area is also altered in lakes with high shoreline development, modifications including building docks, artificially reinforcing shoreline, removing vegetation, and constructing structures impact nearshore conditions (Christensen et al., 1996; Hicks & Frost, 2011; Johnson &

Jones, 2000). Seasonal and permanent residents may remove aquatic vegetation to make shorelines more aesthetically pleasing, but in the process, increase shoreline erosion and sediment suspension while removing important habitat for many organisms (Schriver et al., 1995; Walker et al., 2013). Alternatively, in an effort to reduce shoreline erosion, lakefront property owners may create hardened shorelines, using rip-rap, armour stone, or concrete. Although not intended, these shoreline alterations have been shown to reduce macroinvertebrate species richness and increase the abundance of pollution tolerant taxa, such as Chironomidae (Brauns, Garcia, Walz, et al., 2007).

The nearshore zone typically has the highest rates of primary productivity per unit area, because it encompasses the littoral area, where sunlight can penetrate down to the lake bed (Chang & Rossmann, 1988; Makarewicz et al., 2012). As such, photosynthetic organisms such as algae and plants can grow throughout the water column, from the sediments to the surface, if sunlight is not obstructed. Density of aquatic vegetation is typically highest in the nearshore as well, and it provides food and habitat for most lake organisms at some point in their life cycle, providing fish nursery grounds and habitat for macroinvertebrates and zooplankton (Schriver et al., 1995; Vadeboncoeur et al., 2011; Walker et al., 2013). The composition of the macrophyte community is also consequential, different structural plant types can alter water chemistry and impact the abundance of organisms such as macroinvertebrates (Walker et al., 2013; Wetzel, 2001). Since the nearshore zone plays an important role in the lake's food web, evaluating its composition and structure can provide important insights on the biological condition of a lake (Niemi et al., 2007). The proximity of the nearshore zone to the shoreline means it is the first place to experience the direct effects of human activities on the landscape.

Nutrients and contaminants may flow into the nearshore area from point sources, such as industrial effluent, or non-point sources, like agricultural runoff. Additionally, the magnitude of contaminant loadings and their impacts on the biological community depends on the extent of shoreline modification, Rosenberger et al. (2008) found shifts in the macroinvertebrate and periphyton communities at more developed sites.

Aquatic biological communities have long been used to serve as "bioindicators" of habitat or ecosystem condition. Evaluating lake condition based on the diversity, composition, and structure of aquatic communities can provide essential information for management of lake health. Phytoplankton, zooplankton, and macroinvertebrates have all been used extensively for biomonitoring to understand human impacts on aquatic systems (Buss et al., 2014; Parmar et al., 2016). In particular, lakes with high anthropogenic activity in the watershed require more intensive water quality and biomonitoring programs to track impacts of land-use.

Although the nearshore zone is recognized as containing vital habitats and an important contributor to lake ecosystem function, the monitoring bias focused on the pelagic zone (Fraterrigo & Downing, 2008; Gémesi et al., 2011) has rendered several knowledge gaps related to lake health. One knowledge gap relates to our limited understanding of the relationship between water quality, primary productivity, and the essential players in the lower aquatic food web community (i.e., phytoplankton, zooplankton, and macroinvertebrates) in the nearshore zone. I aimed to address this knowledge gap in the present study by 1) characterizing the nearshore phytoplankton, zooplankton and macroinvertebrate communities across a range of sites in a large, shallow lake (Lake Scugog), and 2) determining if there are distinct community profiles

associated with nearshore water quality and/or habitat (i.e. aquatic vegetation) conditions. I hypothesized that there would be an increase in taxa abundance at developed sites due to an increase in nutrients and corresponding increase in primary productivity (Rosenberger et al., 2008). However, diversity was expected to decline with development due to the elimination of sensitive taxa from pollution (Morse et al., 2003). Additionally, the reduction of habitat due to macrophyte removal is expected to have negative impacts on both macroinvertebrate and zooplankton abundance and diversity (Schriver et al., 1995; Walker et al., 2013).

Similar to other lakes situated in human-dominated watersheds, agriculture and urban development have had significant impacts on Lake Scugog's water quality (Smith et al., 2021). As a popular boating destination in the Trent-Severn waterway, Lake Scugog has been colonized over the decades by non-native invasive species, such as *Myriophyllum spicatum* (Eurasian watermilfoil) and dreissenid mussels. A recent study by Harrow-Lyle and Kirkwood (2020b), documented the increasing prevalence of *Dreissena polymorpha* and *Nitellopsis obtusa* in Scugog, and their role in driving periodic *Microcystis* spp. blooms in the lake. Lake Scugog is also a popular sportfishing destination, however a recent decline in *Sander vitreus* (walleye) indicates the changes in the lake are also impacting the upper trophic levels (Harrow-Lyle & Kirkwood, 2020a). The present study targeted Lake Scugog's nearshore zone to not only characterize the resident biological communities in the lake for the first time, but to disentangle the relative importance of water quality and aquatic vegetation in driving local community structure. These findings elucidate the importance of aquatic vegetation as habitat in the

nearshore zone, even in highly developed lakes such as Scugog, which can often be presumed to be too impaired for mitigation or restoration efforts.

3.2 Methods

3.2.1 Study Site

Lake Scugog is a large headwater reservoir located in Southern Ontario, Canada, formed in the 1830's after the damming of the Scugog River near the Town of Lindsay. Since then, Scugog has maintained the same size and morphometry, with an east and west basin separated by Scugog Island (Figure 3.1). Scugog was initially an important transportation route for loggers, but beginning in the 1950's, became an increasingly popular tourist destination due to its proximity to the Greater Toronto Area, and reputation as a sport-fishing destination. Tourism is one of two key industries for the Township of Scugog, the other being agriculture. These two industries define the large Lake Scugog watershed (529.7 km^2), agriculture consists of 53% of watershed land-use, and although developed land-use only comprises 4%, it is highly concentrated along the shoreline. The eight sites I selected for this study were primarily selected based on their (1) dispersed locations throughout the lake, and (2) land-owner approved access to the waterfront. Lake Scugog is bisected into two large basins by Scugog Island, thus I selected sites that covered both sides of the lake.

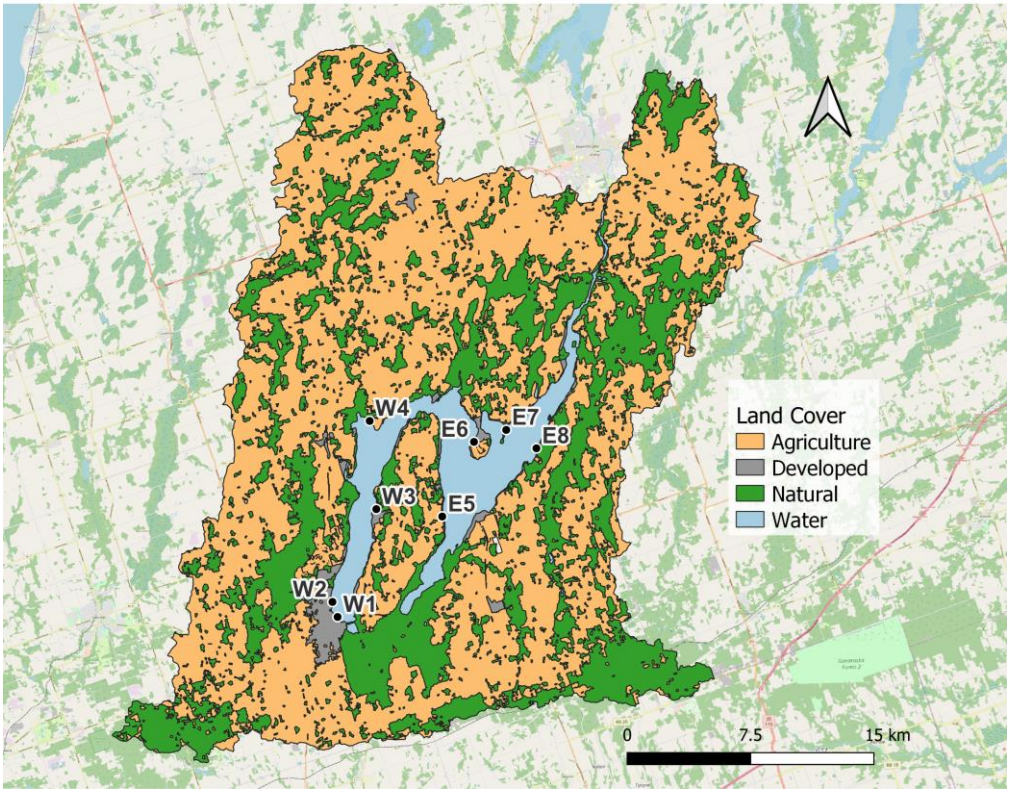

Figure 3.1 Map of Lake Scugog sampling sites and watershed land use, categorized in QGIS (version 3.12.0) as either agricultural, natural, or developed from provincial land cover files (Natural Resources Canada, 2015).

3.2.2 Sample Collection

Samples were collected every three weeks from June – September in 2018 and May-September in 2019. Scugog is a shallow lake with an average depth of 1.4 m, hence nearshore samples were taken at a location with a standard depth of approximately 0.5 m. Field measurements, including dissolved oxygen (DO) (mg L^{-1}), water temperature (ºC), pH, and conductivity (µS cm^{-1}), were recorded on each sampling date. Field measurements were collected using a 650 MDS multi parameter probe (YSI, Yellow Springs, Ohio, USA). Water samples were collected with a 4.2-L Van Dorn sampler submerged to a depth of 0.2 m, water from the Van Dorn was poured into three 1-L acid-

washed Nalgene™ bottles. Water samples were kept in iced coolers until they were returned to the lab on the same day as collection.

Phytoplankton samples were collected on every sampling trip. Sub-samples of lake water from the Van Dorn sampler were decanted into 120-mL borosilicate bottles and fixed with Lugol's solution. They were then stored at room temperature in the dark until identification. Zooplankton samples were collected by filtering 4.2 L of lake water from the Van Dorn sampler through a polyvinylchloride cylinder with a 63-µm filter. The concentrated zooplankton community was preserved with 70% ethanol and stored until identification. Macrophyte samples were collected once each year in August to ensure capture of peak growth. Macrophyte collection followed the method of Ginn (2011), at each site a lake rake was thrown once, to the full extension of the rope (5.0 m), allowed to sink and stabilize, and then it was pulled through towards the dock. All material trapped on the rake was placed into an extra-large plastic bag, sealed, and stored on ice until samples were returned to the lab and refrigerated. By standardizing the collection technique, macrophyte abundance at each site could be estimated in the lab (Carr et al., 2003).

The macroinvertebrate community was collected using Hester-Dendy artificial samplers. Samplers were attached to the dock at each site and submerged to a depth of 0.2 m. After three weeks, the samplers were removed and disassembled to extract all macroinvertebrates, which were then scraped into a glass jar and preserved with 70% ethanol. When the entire macroinvertebrate community had been removed, the sampler was cleaned of residue and placed at the same spot for the next deployment period. Periphyton was also scraped from a standard area of 21.2 cm^2 on top of the sampler, and

diluted with reverse osmosis water to 100 mL in a specimen cup. Periphyton samples were stored in the dark in an iced cooler until their same-day return to the lab.

3.2.3 Sample Processing

Upon return to the lab, water samples were stored in a fridge until processing was complete. All processing was completed within 24 hours of sample collection. Total suspended solids (TSS) was determined by filtering 250 mL of each water sample through a 47-mm GF/A glass microfiber filter that had been pre-filtered with Milli-Q water. Chloride was measured with a Cole-Parmer Replaceable Membrane ISE Probe (Cole-Parmer Canada Company, Montreal, QC, Canada). An aliquot of 250-300 mL of each lake water sample was filtered through a 47-mm GF/A glass microfiber filter for chlorophyll a (Chla) extraction. A modified 90% acetone extraction method was used to extract Chla from each sample. The Chla extract was measured with a Genesys 20 Thermo Spectronic UV-VIS spectrophotometer (ThermoFisher Scientific, Waltham, MA, USA) at 750 nm and 665 nm (Kirkwood et al., 1999). Total dissolved phosphorus (TDP) was extracted by filtering lake water samples through 0.2-µm Whatman nylon membrane filters (GE Healthcare, Chicago, IL, USA). TDP and total phosphorus (TP) were determined using the modified ascorbic-acid method of Murphy and Riley (1962) developed by the Ontario Ministry of the Environment (1983). Nitrogen suite analysis (NH3 + NH4, NO2, NO3, and total Kjeldahl nitrogen) was conducted by an accredited lab (SGS Canada, Lakefield, Ontario, Canada) for each lake sample.

Phytoplankton were counted and identified to group level at minimum (Wehr et al., 2015), and genus where possible, using an Evos xl core phase-contrast microscope (ThermoFisher Scientific, Waltham, MA USA) at 400X magnification. The zooplankton

community was counted and identified (Balcer et al., 1984) from each sample using the Evos xl core phase-contrast (40 – 200x) microscope. The zooplankton community was counted to reach 100 individuals or five 1-mL samples for each site, whichever came first. Copepod's were identified to order, Cladocerans to genus, and rotifers to species, where possible. Biomass estimates were calculated using previously established length-weight linear regressions (EPA, 2003). The macroinvertebrate community was counted and identified with a QZE stereomicroscope (Walter Products Inc., Windsor, ON Canada). Macroinvertebrates were identified to family level at minimum, and genus where possible using Peckarsky (1990). Family level was deemed acceptable for the multivariate analyses conducted based on the work by Bowman and Bailey (1997). Macrophyte samples were washed with reverse osmosis water, to remove sediment, and identified to species level. Once sorted macrophyte samples were weighed, dried in a convection oven at 80ºC, and reweighed to determine relative biomass (Carr et al., 2003). Periphyton chlorophyll *a* was determined by filtering 5 mL of the 100-mL periphyton sample through a 47-mm GF/A glass microfiber filter. Filters were folded, wrapped in aluminum foil, and frozen until extraction. Chla was extracted from the periphyton samples using the same modified 90% acetone extraction method as water column Chla samples (Kirkwood et al., 1999).

3.2.4 Statistical Analysis

Statistical analyses were performed using R Studio version 3.6.1 (RStudio, 2020). Variables were evaluated for normality with Q-Q plots and parameters that did not have a normal distribution or exhibited heteroscedasticity were log transformed for analysis (Chla). Bray-Curtis dissimilarity was calculated for each community using the *vegan*

package to determine beta diversity (Ferrier et al., 2007). Permutational analysis of variance (PERMANOVA) with a post hoc test using *pairwise.Adonis* (Martinez Arbizu, 2020) R package were calculated to determine differences between sampling sites. Redundancy analyses (RDA) were conducted for each community using the R package *vegan* (Oksanen et al., 2013). Final biplots were created using the *ggord* and *ggplot2* packages (Beck, 2017; Wickham, 2011), with only significant water quality vectors included.

A relatively new multivariate approach, generalised linear latent variable model (GLLVM), was used to examine species co-occurrence patterns while controlling for environmental variables, following the approach of Harrow-Lyle and Kirkwood (2022). The GLLVM extends the generalised linear model (GLM) to multivariate data following a factor analytic approach, which includes latent variables representing water quality and species specific factor loadings to determine correlations between species (Niku, Brooks, et al., 2019). Abundance data often has high variance which needs to be accounted for and, compared to other methods, GLLVMs are well equipped to handle data sets with a large number of species because the covariance model scales linearly (Warton et al., 2012). By using appropriate diagnostic tools, including Dunn-Smyth residual plots and Q-Q plots with 95% confidence intervals, the GLLVM can be used to select the best distribution to account for the mean-variance relationship (Niku, Hui, et al., 2019).

GLLVMs were generated for each community (phytoplankton, zooplankton, and macroinvertebrates) with the macrophyte community using the R packages *gllvm* (Niku, Hui, et al., 2019) and *mvabund* (Wang et al., 2022). The best distribution for each community was determined by examining residual plots for both negative binomial and

Poisson distributions, all models were best described by the negative binomial distribution. Two GLLVMs were generated for each level of the community; phytoplankton, zooplankton, and macroinvertebrates. One model included water quality parameters as latent variables, and the other removed them. The two models were compared to determine how much variation in species abundance was explained by the water quality parameters (Niku et al., 2017). Only the top 15 taxa for each community were included in the final correlation plot for a clear visualization of the relationships between the most common taxa (Dolédec & Statzner, 2010). Relationships between taxa were visualized with the *corrplot* (Wei et al., 2017) and *gclus* (Hurley & Hurley, 2019) packages.

Since macrophyte abundance was collected once each year during peak biomass it was turned into a categorical variable to compare with diversity and abundance of aquatic communities collected from May-September (Baattrup-Pedersen et al., 2006; Niemeier & Hubert, 1986). Macrophyte abundance was converted into a categorical variable with groups ‘high’ and ‘low’ based on a median split (Iacobucci et al., 2015). The division of sites based on the median split was verified by comparing categories with visual observations from the two years of data collection. Sites 3, 6, 7, and 8 were categorized as ‘low’ plant abundance sites and 1, 2, 4, and 5 were ‘high’ plant abundance sites. Macrophyte abundance categories were used in the RDA and for analysis of variance to determine phytoplankton, zooplankton, and macroinvertebrate abundance and diversity as a function of low and high macrophyte abundance.

3.3 Results

3.3.1 Water quality and submerged macrophytes

Water quality in the nearshore of Lake Scugog varied by site, with generally higher nutrients and primary productivity in the west basin (sites W1 – W4) of the lake (Table 3.1). TP and Chla were elevated at sites W1 – E5 compared to sites E6 – E8 on the northeastern shorelines. TDP and TN also followed this trend with the exception of site 3. Conductivity was also elevated at sites W1 and W2, both of which are located in the town of Port Perry.

Submerged macrophytes were measured once per year in August and also showed notable trends by location. The most abundant plant species were *Myriophyllum spicatum* (766.7 g), *Myriophyllum sibiricum*, (324.3 g) and *Elodea canadensis* (302.3 g). Sites W1, W2, and E5 had relatively consistent macrophyte taxa profiles across the years, while the rest had high variation (Figure 3.2). There was more variation in relative abundance at the low macrophyte sites compared to the high macrophyte sites. Although there were shifts taxa presence/absence the structural plant types stayed relatively consistent across both years of study for most sites.

Table 3.1. Seasonal means (May – September) from two years of pooled data (2018 – 2019) for each water quality parameter measured in Lake Scugog (n = 10). Standard deviation in brackets.

Site	Temperature (˚C)	pH	Conductivity ($\mu S \cdot cm^{-1}$)	DO ($mg \cdot L^{-1}$)	Chla ($mg \cdot L^{-1}$)	TP ($\mu g \cdot L^{-1}$)	TDP ($\mu g \cdot L^{-1}$)	TN ($mg \cdot L^{-1}$)	TSS ($g \cdot L^{-1}$)
W1	21.9 (2.7)	8.0 (0.7)	620.1 (116.6)	6.28 (2.45)	15.79 (11.83)	36.30 (12.79)	6.04 (5.57)	0.61 (0.11)	0.028 (0.073)
W2	22.7 (2.7)	8.3 (1.1)	668.0 (106.1)	6.37 (3.06)	18.84 (10.70)	55.86 (34.39)	5.44 (2.90)	0.70 (0.13)	0.003 (0.001)
W3	23.3 (3.3)	8.5 (0.7)	599.1 (138.0)	9.01 (2.24)	17.24 (22.10)	45.86 (30.68)	2.23 (1.17)	0.54 (0.12)	0.003 (0.002)
W4	22.8 (2.5)	8.2 (1.0)	505.4 (109.3)	5.45 (2.84)	25.30 (32.17)	45.86 (29.63)	6.47 (7.47)	0.71 (0.16)	0.003 (0.003)
E5	23.5 (3.6)	7.8 (0.8)	593.5 (158.7)	7.04 (3.57)	25.85 (19.52)	64.58 (42.63)	4.24 (2.20)	0.61 (0.17)	0.006 (0.003)
E6	23.4 (2.6)	8.5 (0.7)	544.7 (117.0)	8.26 (1.78)	6.48 (4.24)	18.15 (4.23)	2.36 (2.68)	0.47 (0.08)	0.004 (0.004)
E7	22.8 (3.5)	8.1 (0.7)	564.1 (128.0)	8.69 (2.61)	10.21 (6.50)	26.00 (8.41)	2.33 (2.11)	0.52 (0.10)	0.029 (0.082)
E8	23.3 (3.2)	8.2 (0.6)	583.0 (131.9)	8.48 (1.42)	8.52 (4.25)	20.37 (11.70)	3.75 (4.93)	0.55 (0.06)	0.003 (0.001)

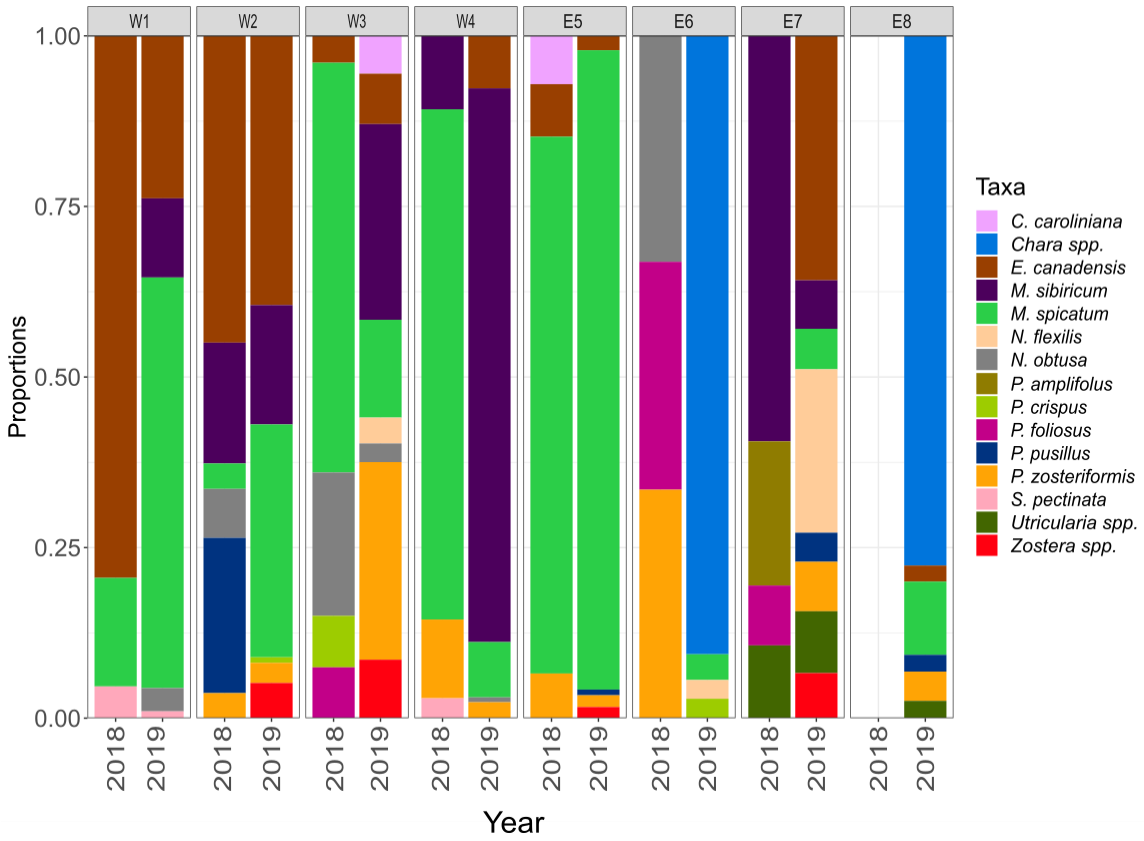

Figure 3.2 Taxa plot of macrophyte relative abundance across eight sites sampled in August of 2018 and 2019. The collection at site E8 in 2018 yielded no macrophyte biomass

3.3.2 Phytoplankton, zooplankton, and macroinvertebrate communities

The nearshore phytoplankton community composition varied seasonally and spatially over the two years of study. To investigate differences in the phytoplankton community across sampling sites, Bray-Curtis dissimilarity was calculated. It was found that sampling site significantly influenced phytoplankton community composition (PERMANOVA, $F_{7,90} = 2.07$, $p < 0.001$). Post hoc pairwise comparisons found significant differences between sites W2 and E8, and site W4 and sites E7 and E8 (pairwise adonis, $p_{adj} < 0.05$). A redundancy analysis (RDA) was conducted on the phytoplankton community with water quality and plant abundance as explanatory variables (Fig. 3.3a). Temperature, total phosphorus, dissolved oxygen, conductivity, turbidity, and plant abundance were the significant predictors included in the final model.

There was some grouping of sites along the first two axes, with sites E7 and E8 scoring lower on the second axis and sites W1 and W2 scoring higher on the second axis. To better understand the impacts of water quality on phytoplankton and macrophyte interactions, GLLVMs with and without water quality variables were determined. By comparing the two models I found that water quality variables explained approximately 92% of the variation in phytoplankton and macrophyte taxa abundance. With water quality accounted for, the correlation plot shows many significant relationships between phytoplankton and macrophyte taxa (Fig. 3.4). Most of the significant correlations with macrophytes were positive, with the exception of the invasive *Myriophyllum spicatum*, which had mostly negative relationships. The invasive *Nitellopsis obtusa* also had positive relationships with many phytoplankton taxa. Based on nested analysis of variance I found that macrophytes significantly influenced phytoplankton biomass, where high plant abundance resulted in increased phytoplankton biomass ($p < 0.05$, Table 3.2).

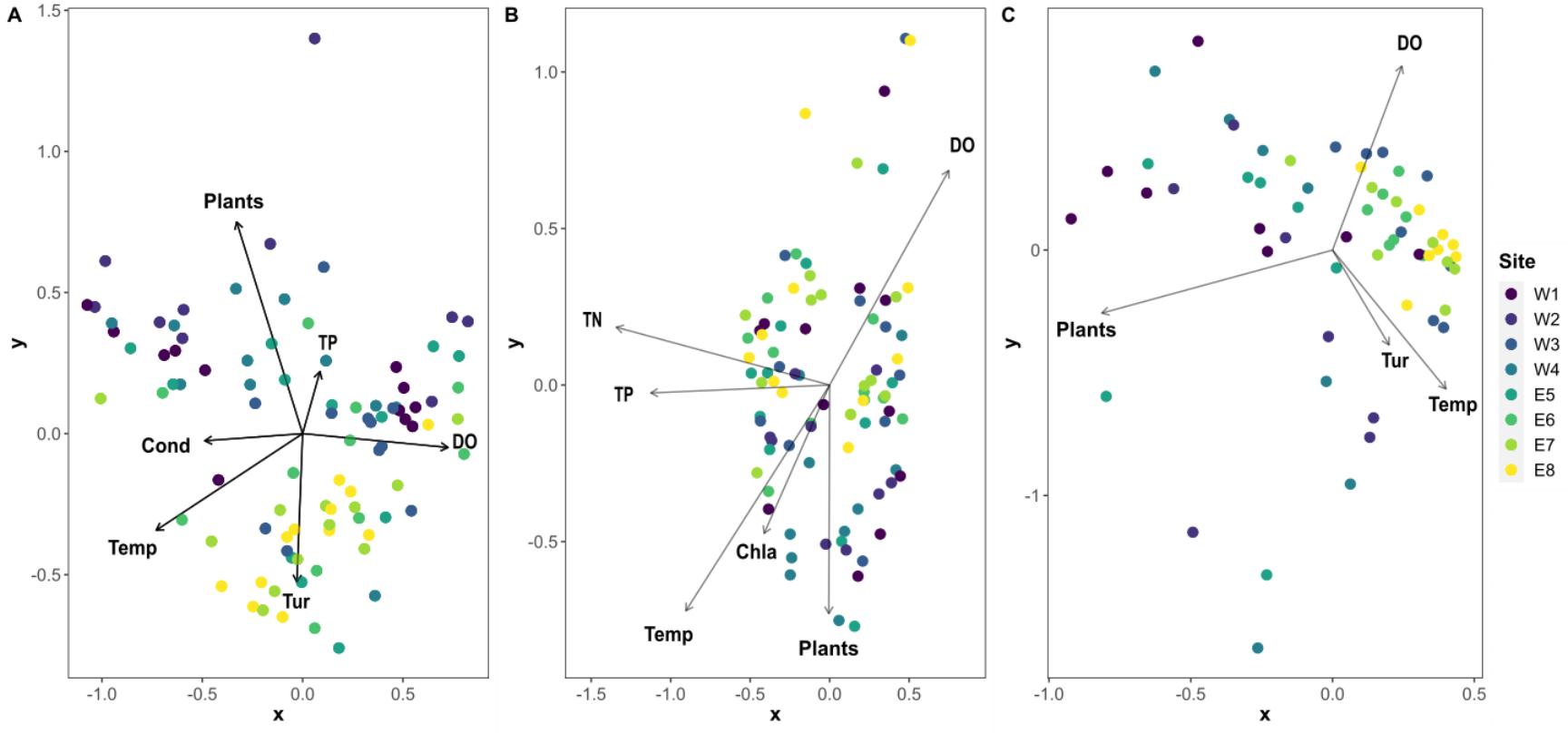

Figure 3.3 Redundancy analysis (RDA) biplots for (a) phytoplankton (n = 92), (b) zooplankton (n = 87), and (c) macroinvertebrate (n = 60) communities at eight sites across Lake Scugog. Environmental parameter arrows pointing in the same direction indicate positive correlations and arrows pointing in opposite directions indicate negative correlations. Arrow length is a direct representation of the variance explained by the environmental variable Cond = conductivity, Tur = turbidity, Temp = temperature, TP = total phosphorus, TN = total nitrogen, Chla = chlorophyll a, DO = dissolved oxygen

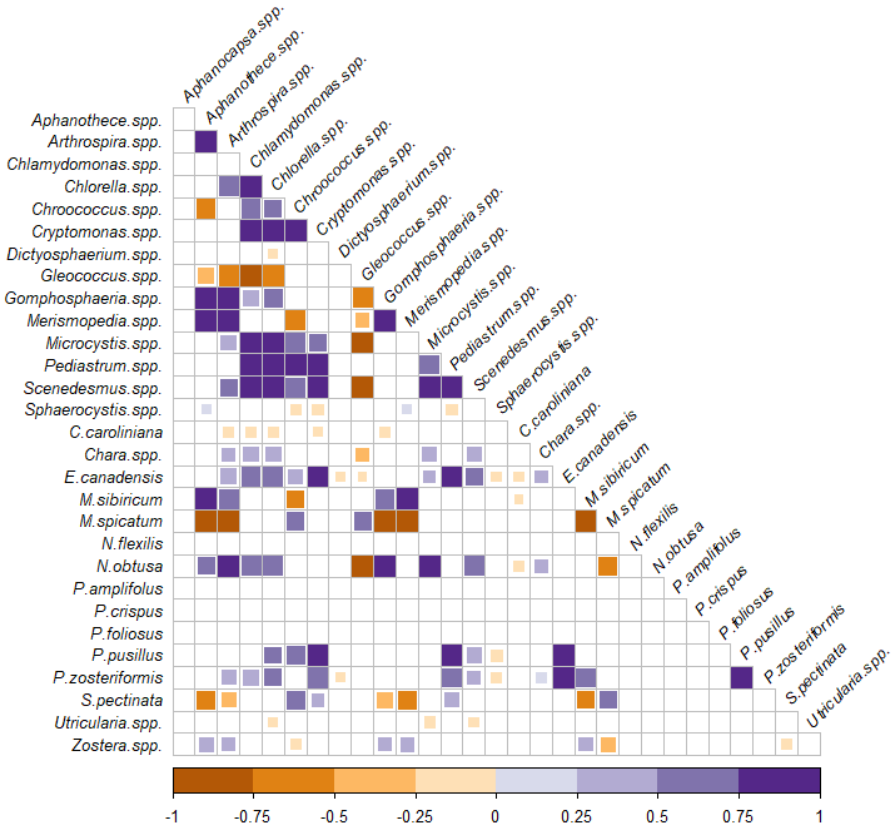

Figure 3.4 Generalized linear latent variable model (GLLVM) correlation matrix of species-specific interactions across the top 15 phytoplankton taxa and macrophyte taxa in Lake Scugog (n = 16) with water quality variables accounted for. The strength of the relationship between taxa is indicated by the colour and size of the box, blank boxes indicate the relationship between taxa was not significant.

The nearshore zooplankton community in Lake Scugog was primarily composed of nauplii copepods, *Bosmina*, and cyclopoids. A Bray-Curtis dissimilarity analysis did not find any differences in zooplankton diversity between sites. Examining the zooplankton community with water quality parameters provided some insight into what was driving nearshore zooplankton in Lake Scugog. Temperature, Chla, TP, TN and plants were significant drivers of the RDA component axes (Figure 3.3b). In addition, the results of the GLLVM's indicate that water quality had an influence on zooplankton and macrophyte community interactions. Comparing the models, I found that water quality

variables explained 58% of variation in zooplankton and macrophyte abundance. In the correlation plot most of the significant relationships included macrophytes, specifically *Potamogeton pusilus*, *N. obtusa*, *Najas flexilis*, and *Myriophyllum sibiricum* (Figure 3.5). There was significantly higher zooplankton biomass at high plant sites, but, there was no significant relationship between zooplankton diversity and plant abundance (ANOVA, $p < 0.05$, Table 3.2).

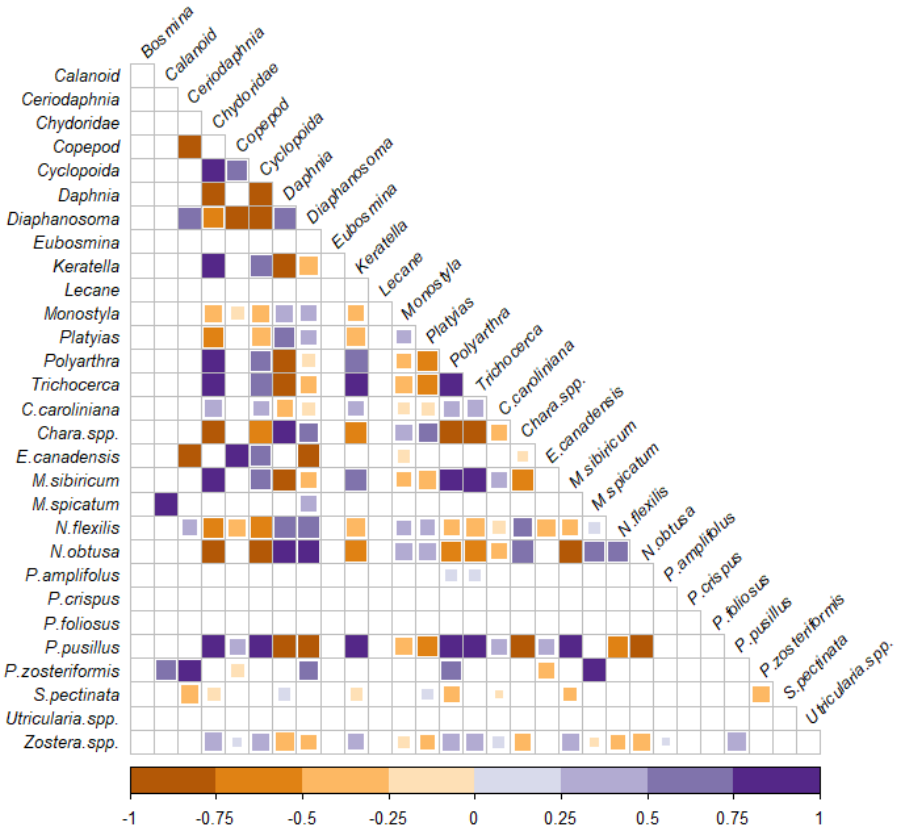

Figure 3.5 Generalized linear latent variable model (GLLVM) correlation matrix of species-specific interactions across the zooplankton and macrophyte taxa in Lake Scugog (n = 16) with water quality variables accounted for. The strength of the relationship between taxa is indicated by the colour and size of the box, blank boxes indicate the relationship between taxa was not significant.

Examining the macroinvertebrate community, there was significantly different community composition among sites based on Bray-Curtis dissimilarity (PERMANOVA,

$F_{7, 59} = 4.12$, $p < 0.001$). Pairwise tests showed significant differences between site W1 and sites W3, E6, E7, and E8, site W2 and sites W3 and E8, site W3 and sites W4, E5, and E6, site W4 and sites E7 and E8, and site E5 and site E8 (Table S1, $p_{adj} < 0.05$). The macroinvertebrate community had plants, temperature, turbidity, and DO driving the RDA component axes (Fig. 3.3c). Sampling sites appeared to score differently along the axes, with sites E6, E7 and E8 grouped together, scoring high on axis 1. To determine if macroinvertebrate and macrophyte community interactions were influenced by water quality parameters GLLVM's were created with and without water quality. Similar to the zooplankton community, most of the macroinvertebrates significant relationships were with macrophytes (Fig. 3.6). Specifically, Ephemerellidae, Gammaridae, Lymnaeidae, and Sericostomatidae had a mix of negative and positive relationships with macrophyte taxa. Overall, water quality variables explained 83% of the variation in taxa abundance.

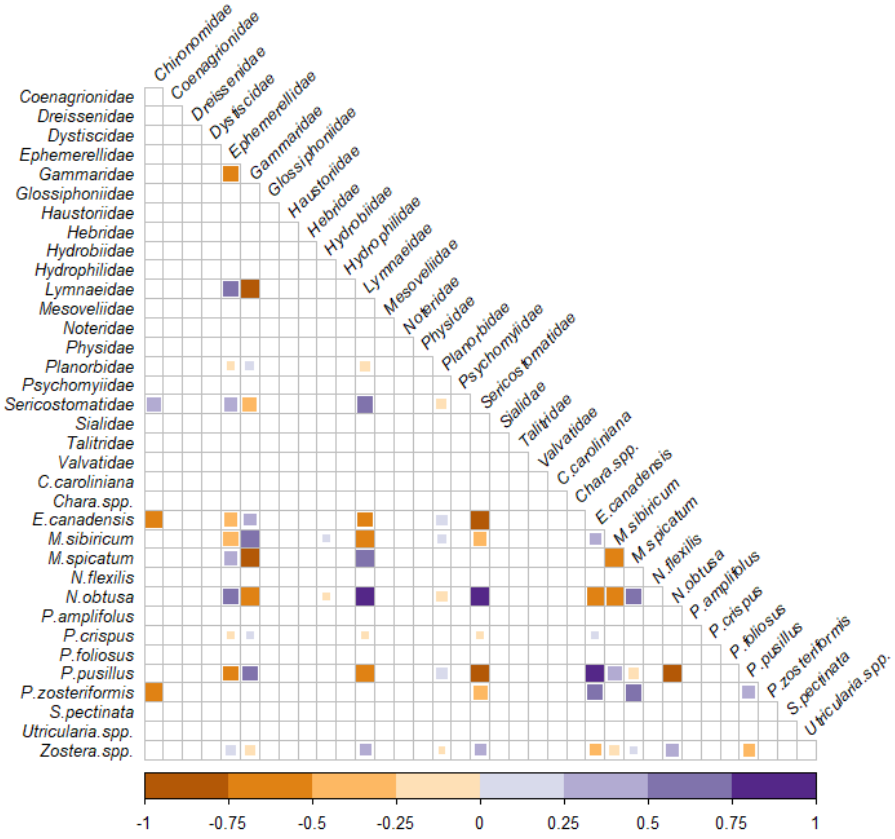

Figure 3.6 Generalized linear latent variable model (GLLVM) correlation matrix of species-specific interactions across the macroinvertebrate and macrophyte taxa in Lake Scugog (n = 16) with water quality variables accounted for. The strength of the relationship between taxa is indicated by the colour and size of the box, blank boxes indicate the relationship between taxa was not significant.

Table 3.2. Nested analysis of variance of plant biomass (high vs. low) with sampling site as a random effect on phytoplankton (n = 91), zooplankton (n = 87), and macroinvertebrate (n = 60) abundance and Simpson's diversity (D). Bold values indicate a statistically significant model ($p \leq 0.05$).

Group	Dependent variable	Estimate	Std. Error	p
Phytoplankton	Biomass(μg/mL)	7.69^{11}	3.85^{11}	**0.046**
	Diversity (D)	-0.109	0.047	0.059
Zooplankton	Biomass (μg/L)	2.34^{05}	9.47^{04}	**0.049**
	Diversity (D)	0.024	0.035	0.492
Macroinvertebrates	Density (indiv. /m^2)	-474.34	106.99	**0.005**
	Diversity (D)	0.186	0.061	**0.022**

The significance of plants and DO in the RDA indicate that primary productivity, specifically macrophyte abundance in the nearshore, may be influencing the macroinvertebrate community in Lake Scugog. In addition, the persistence of the significant correlations between macrophytes and macroinvertebrates in the GLLVM indicate their role in shaping the macroinvertebrate community. Macroinvertebrate diversity was significantly increased at high plant abundance sites, while abundance was decreased (ANOVA, $p < 0.05$, Table 3.2).

3.4 Discussion

In both nearshore water quality and biological communities, there are clear spatial patterns associated with local habitat and shoreline development. Water quality was the most degraded, with high nutrients and Chla, and low DO, at sites W1 – E5, which all

have high land-use impacts from development in their sub-watersheds (Smith et al., 2021). The west side of the watershed that drains into the west basin (sites W1- W4) has more urban and shoreline development compared to the east basin, including the growing town of Port Perry and several exurban communities. Additionally, Scugog Island has small-scale feedlots and pastureland that may be a source of high nutrients at site E5 in the eastern basin.

Water quality across sites clearly affected the nearshore biotic dynamics, starting with primary production. All of the high macrophyte sites had relatively high nutrient and Chla concentrations (Table 3.1). It was expected that low macrophyte sites would have increased nutrients due to lower nutrient uptake by aquatic plants (Levi et al., 2015). When considering the eutrophic status of Lake Scugog, nutrients are likely not at growth-limiting concentrations, and thus are in surplus supply. In addition, submerged macrophytes may be exhibiting the 'escape effect' by reaching the surface before algal shading can inhibit growth from the sediments (Scheffer et al., 1992). It is also quite likely that physical removal of macrophytes by people is occurring at low macrophyte sites, therefore confounding the effects of phytoplankton and nutrients on macrophyte abundance.

The nearshore phytoplankton community was driven by water temperature, conductivity, TP, turbidity, and DO and plant abundance ($p < 0.05$, Fig. 3.3a). To further explore why macrophytes would be important for phytoplankton community structure, the GLLVM correlation plot shows many statistically significant relationships between phytoplankton and macrophyte taxa. Specifically, the invasive charophyte *N. obtusa* had multiple significant, positive correlations with phytoplankton taxa, matching previous findings

that it plays an important role in driving the lower aquatic food web in its non-native range (Harrow-Lyle & Kirkwood, 2022). Although not significant, the nested analysis of variance indicates that phytoplankton diversity declined with macrophyte abundance. This finding was somewhat unexpected as it opposes previous findings on the impact of macrophytes on phytoplankton communities. Declerck et al. (2007) found that macrophytes had positive effects on phytoplankton richness, and Barrow et al. (2019), found that high macrophyte treatments had lower phytoplankton biomass, but higher taxonomic and functional diversity compared to low and no macrophyte treatments. Nonetheless, it is possible that phytoplankton diversity is increased at our low plant sites due to several factors including increased light availability and reduced grazing pressure from invertebrates in macrophyte stands (Søndergaard & Moss, 1998).

The nearshore zooplankton community was also influenced by water quality and macrophyte abundance (Fig. 3.3b). Temperature, TP, TN, Chl-a, DO, and plants were all statistically significant variables that contributed to the zooplankton RDA axes. However, water quality explained the least amount of variation in the zooplankton community compared to its influence on phytoplankton and macroinvertebrates in both the RDA and GLLVM. It was expected that macrophytes would play an important role in shaping the zooplankton community as other studies have found reduced zooplankton abundance and shifts in community composition in response to the removal of submerged macrophytes (Sagrario et al., 2009; Schriver et al., 1995). Although zooplankton diversity was not affected by plant abundance, biomass was significantly higher at high plant sites (Table 3.2). This is likely due to the increased amount of refuge habitat available for zooplankton in macrophyte beds (Paterson, 1993).

With respect to the nearshore macroinvertebrate community, it is clear that water quality is important for shaping co-occurrence interactions between macroinvertebrates and macrophytes. However, when accounting for the effect of water quality on macroinvertebrate community structure, there still remains a strong effect by macrophytes. I found that macrophytes had an overall positive influence on macroinvertebrate diversity, but a negative impact on abundance. This was unexpected as previously it has been found that macrophyte biomass increases both macroinvertebrate abundance and diversity (Warfe & Barmuta, 2004).

Looking closer at the macroinvertebrate community composition across sites provides some insight into why abundance and diversity responded differently. Sites with low plant abundance and high macroinvertebrate density were primarily Chironomidae. The Chironomidae are a largely pollution tolerant family of nematoceran flies, and in Lake Scugog, it appears they may be acting as an indicator of poor habitat conditions (Brauns, Garcia, Pusch, et al., 2007; Kaller & Kelso, 2007). Conversely, macroinvertebrate diversity was higher at high plant sites, and this is likely due to the increased habitat availability associated with submerged macrophyte beds. It is possible the increased macroinvertebrate diversity was also supported through increased food resources such as periphyton and zooplankton. Other studies that have focused on the role of habitat and macroinvertebrate communities have found that diversity and abundance increase with vegetation biomass (Taniguchi et al., 2003; Walker et al., 2013). Overall, the macroinvertebrate taxa present in Lake Scugog tend to be more tolerant of poor water quality, which aligns with the eutrophic, high development status of the lake. However, it is also clear that the presence and abundance of macrophytes, even in a degraded, weedy

lake such as Lake Scugog, can still increase the overall diversity of the nearshore macroinvertebrate community.

Although each biological community was influenced by unique water quality drivers, macrophyte abundance and species composition was shown to play an important role in the community structure of all biological communities studied. In particular, increased macrophyte abundance was a significant predictor of higher phytoplankton and macroinvertebrate diversity in the nearshore zone. Despite all sites reflecting some level of habitat degradation due to shoreline development (Smith et al., 2021), more species rich biological communities flourished where macrophytes were abundant. Alternatively, locations with comparatively better water quality (i.e., lower nutrients), but low macrophyte abundance, had less zooplankton and less diverse macroinvertebrate communities.

These findings have important implications for macrophyte management in productive lakes with high shoreline development and recreational use. The impulse for shoreline property owners to remove aquatic weed beds adjacent to their shorelines to improve aesthetics and boat navigation should be tempered with a desire to improve lake health. Our findings clearly show that sites with low macrophyte cover had lower abundance of zooplankton and lower diversity of phytoplankton and macroinvertebrates. Considering that aquatic insects and crustaceans are important prey for panfish and juvenile sportfish, as well as contributing to proper ecosystem function, there are direct and indirect negative consequences for overall lake health when macrophytes are intentionally removed.

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Chapter 4. **Community Science to the Rescue: Capturing Water Quality Data in the Kawartha Lakes During the COVID-19 Pandemic**

4.1 Introduction

The COVID-19 pandemic has impacted almost every aspect of human life, including some unexpected ecological changes. In the spring of 2020, the "anthropopause" had begun as a result of drastically constrained human activities such as industry and travel. Headlines were emerging that pandemic lockdowns were having a positive effect on the environment (Zerefos 2021). In Venice, Italy the canal bottom was seen for the first time in decades, and several large cities had improvements in air quality (Clifford 2020; He et al. 2020; Bherwani 2020). Although early indications are that the pandemic allowed some environmental renewal, it is still too soon to have a robust understanding of all the pandemic impacts on the environment (Berman and Ebisu 2020; Hallema et al. 2020).

One unfortunate effect of the COVID-19 pandemic was the pausing or cancelling of water quality monitoring programs. In the spring of 2020, much uncertainty surrounding the pandemic, and concerns about maintaining social distancing, lead to a reduction in the number of environmental monitoring programs. The U.S. National Park Service, for example, issued just 37% of its normal amount of research permits (Miller-Rushing et al. 2021), while the Canadian federal government cancelled all water quality monitoring programs (Zingel 2020). Within Ontario, the Lake Partner Program, a province-wide volunteer-based water quality monitoring program that includes over 550 inland lakes, paused for the summer of 2020 (Dorset Environmental Science Centre 2020). Along with federal and provincial governments, many local conservation authorities could not run their regular monitoring programs due to a lack of access to

laboratories and offices (Akinsorotan 2021). This multi-level monitoring shut-down reflected a massive data gap, where thousands of lakes and streams in Ontario were not monitored for water quality during the entire summer of 2020.

The pause in water quality monitoring left gaps in long-term monitoring programs and constrained the ability to measure the immediate impacts of the COVID-19 pandemic on Ontario's lakes. In Ontario, the pandemic lead to multiple lockdowns, restrictions on various activities, and changes in individuals' hygiene habits (Nielsen 2021). These changes improved air quality in many places, and it has recently been found that water quality may have been affected as well (He et al., 2020; Tobías et al., 2020). Although most field monitoring was cancelled in 2020, some researchers used remote sensing to examine turbidity and found it decreased in the Ganga River and Vembanad Lake in India during the lockdown period (Garg et al. 2020; Yunus et al. 2020). Alternatively in the Meriç-Ergen River Basin in Turkey, turbidity did not change but there were reductions in metal(loid) levels (Tokatlı & Varol, 2021). The reductions in water quality parameters were attributed to reduced effluent from industrial sources and reduced pollution from human activities, such as tourism, in the area. Another change in human activity that could impact water quality is the quantity of water consumed by households. With many people staying home and increasing hygienic behaviours such as hand washing, some studies found an increase in residential water consumption (Kalbusch et al. 2020; Abu-Bakar et al. 2021). In areas with household septic systems, there is the potential for these systems to become overloaded, resulting in poor treatment performance. Excessive septic seepage could lead to increased nutrient levels in nearby surface waters (Reay 2004; Oldfield et al. 2020).

This study aimed to fill the large monitoring gap created by pandemic restrictions in Ontario, Canada by implementing a community science approach that ensured physical distancing. The main objectives of this study were to: (1) Assess seasonal nutrient dynamics across 16 lakes in the Kawartha Lakes Region during the most restrictive pandemic year (2020), and (2) Compare pandemic nutrient conditions in four Kawartha Lakes during a pre-pandemic (2019) year and pandemic years (2020 and 2021), and (3) Evaluate lake-front residents' environmental attitudes and behaviours during the early months of strict pandemic measures in 2020. This study offered the unique opportunity to test if there was a detectable effect of the "anthropopause", with the hypothesis being that increased human activity in the Kawartha Lakes during the pandemic caused increased nutrient loadings to the study lakes.

4.2 Methods

4.2.1 Study Site

The Kawartha Lakes region, located in south-central Ontario, is within a 1-2 hour drive from the Greater Toronto Area in Ontario, Canada. The Kawartha Lakes are a popular tourist destination and an integral part of Ontario's cottage country (City of Kawartha Lakes 2020). They are also part of the nationally significant Trent-Severn Waterway (TSW), which connects Georgian Bay to Lake Ontario. Tourism is one of the biggest industries for the City of Kawartha Lakes and recreational activities on the lakes are a major draw for the area, helping to bring in over 1.6 million visitors annually (City of Kawartha Lakes 2020). The Kawartha Lakes watershed is part of "The Land Between" – a biodiverse ecotone that reflects a geological shift from limestone to granite (Alley,

2006). The watersheds of these lakes also have over 55 provincially significant watersheds that provide key ecosystem services (Kawartha Conservation 2022).

As part of the TSW, the Kawartha Lakes have controlled flow between the lakes through locks and dams. In our study lakes, the flow begins at Balsam Lake, feeds Canal and Mitchell to the west and Cameron to the east, and the flow continues east with Katchewanooka as the final downstream lake in our study area (Figure 4.1). Lake Scugog is a headwater lake that flows into Sturgeon, and Sandy lake is the only study lake not hydrologically connected to the TSW.

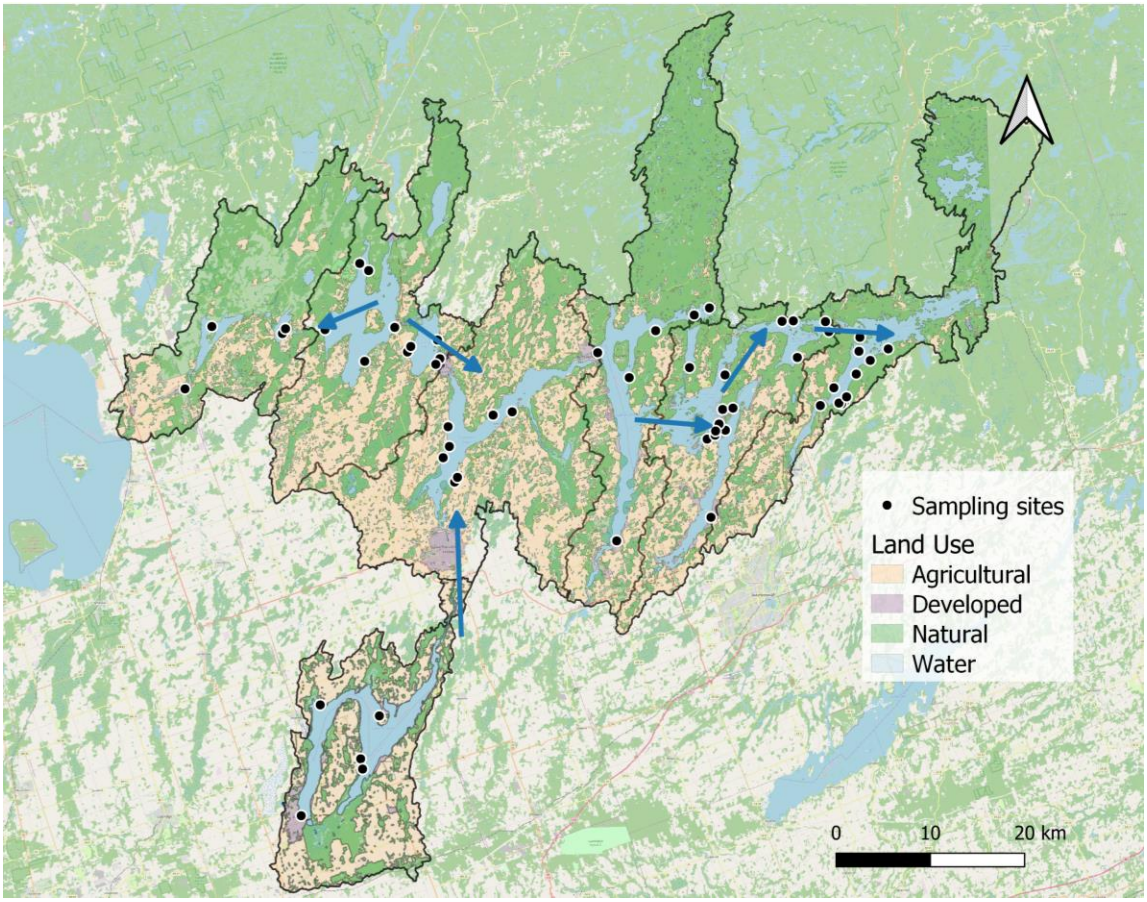

Figure 4.1 Map of 2020 water sampling sites in 16 Kawartha Lakes, arrows indicate the direction of water flow through the system and watersheds are outlined in black

4.2.2 Community Science Model

Our research group had been conducting research in the Kawartha Lakes using community science for several years before the pandemic, including four lakes in this study (Balsam, Cameron, Sturgeon and Pigeon). Not only had we established a network of lake associations and volunteers to tap into for this study, but community science was an ideal approach that could ensure physical distancing for all involved during the pandemic years. In the first year of the pandemic (2020), we recruited 58 community-science volunteers to sample from 60 sites across 16 lakes. Due to strict physical distancing requirements at this time, volunteers were asked to not only collect water samples with containers that we provided but to also store them in their freezer until the end of the study period in September 2020.

Community-science volunteers were recruited from a pre-existing network established by the local watershed authority Kawartha Conservation. Additional volunteers were recruited through the Kawartha Lake Stewards Association (KLSA), which spread the word via email and virtual meetings. We also partnered with Curve Lake First Nation, whose traditional lands and waters encompass the Kawartha Lakes Region. Five volunteers from Curve Lake First Nation selected lake sites to monitor across their reserve territory, located between Buckhorn and Chemong Lake.

Volunteer training was conducted virtually to ensure the safety of all study participants. A training video was created and shared on YouTube with a direct link sent to all volunteers. Two live video follow-up sessions were booked about a week after the release of the recorded video to answer questions and share additional information.

Volunteers were also sent a document with visual and written instructions for collecting and storing their water samples, as well as recording field observations.

4.2.3 Water Sample Collection

Volunteer sample kits were distributed across four pick-up locations that included local marinas and volunteers' homes, one week before the first sample collection date. Sample kits included 8 – 200 mL specimen cups, gloves, collection instructions and a field datasheet. Water samples were collected monthly from June – September 2020. Volunteers collected samples to fill two specimen cups on the last Tuesday of the month between 8 – 9 am, for consistency across sites. If volunteers were unable to take the sample at the requested time, they recorded the date and time of their sample collection. Once samples were collected, the labelled specimen cups were placed in the volunteers' freezer until sample pick-up in late September.

4.2.4 Volunteer Survey Deployment

With various pandemic-related lockdown measures in place in 2020 and 2021, I wanted to examine if there was an impact of these measures on homeowners' habits. Of particular interest were, changes to the number of people and time spent at the waterfront property, changes in habits that impact septic tank load, and changes to waterfront property maintenance. To investigate the changes in these habits, an anonymous online survey was sent to all community science volunteers (Appendix A, REB #:15910). Survey questions asked participants to compare their activities at the waterfront property to the previous year (2019) with follow-up questions for respondents to specify their activities such as gardening habits and some questions to gather demographic information. These comparisons were used to analyze trends in waterfront property

owner habits before and during the COVID-19 pandemic. Volunteers were sent a link to the anonymous online survey during the summer of 2020. When the survey was closed in the fall of 2020, there were 45 responses.

4.2.5 Water Sample Processing

Water samples from 2019 and 2021 were received from community scientists on the day of sample collection and kept on ice until returned to the laboratory. Aliquots were poured for phosphorus and nitrogen analysis and frozen within 24 hours of sample collection. Frozen samples were sent to an accredited lab (SGS Canada, Lakefield, ON) for nitrogen suite analysis (NH2, NH3, NH3+NH4, and total Kjeldahl nitrogen). Total phosphorus was determined using the method of Murphy and Riley (1962). Water samples collected in 2020 were frozen immediately by the community scientist and frozen samples were returned to the laboratory in September. Samples were then thawed for total phosphorus and chlorophyll-a (Chla) analysis in the laboratory and aliquots were sent for nitrogen suite analysis (SGS Canada, Lakefield, ON). Samples collected in 2020 were measured for Chla fluorescence with an Aquafluor handheld fluorometer (Turner Designs, Sunnyvale, CA). 2020 Chla could not be compared to 2019 or 2021 samples because a different method was used for measuring Chla in those years (spectrophotometric method).

4.2.6 Statistical Analysis

Statistical analyses were completed using R Studio (RStudio Team 2016). Exploratory data analysis was conducted to identify outliers. Points that fell out of the 1.5 times IQR were examined to determine if there was a sampling error. One site on Lake Scugog that has been monitored previously had extremely elevated values that appeared

to be due to sampling error and were thus removed from future analysis. TP and Chla were log-transformed due to the non-normal distribution of residuals. Spatial variation was analyzed with analysis of variance (ANOVA) to determine if there were differences between lakes and watersheds, with Tukey's post hoc test to determine specific differences between groups. Non-metric multidimensional scaling (NMDS) was performed to examine water quality profiles across the watersheds. Permutational analysis of variance (PERMANOVA) was conducted with *vegan* (Oksanen et al., 2013) based on Bray-Curtis dissimilarity to determine if differences between watersheds were significant. A pairwise post hoc test was performed using the *pairwise.adonis* (Martinez Arbizu, 2020) R package. Seasonal variation was examined with ANOVA between months and years for the four lakes with 2019, 2020 and 2021 data. Welch's t-test was conducted on survey data to determine if there were significant differences between property owners' habits in 2019 and 2020. Principal component analysis (PCA) was conducted on data from the four lakes studied from 2019-2021. The longest gradient from the detrended correspondence analysis (DCA) was less than three, so PCA was conducted with the R packages *vegan* (Oksanen et al., 2013), *factoextra* (Kassambara & Mundt, 2017), and *ggplot2* (Wickham, 2011).

4.3 Results

4.3.1 Community Science Survey

The survey of waterfront property owner's habits was sent to all 58 community scientists in June 2020, and 78% responded by the deadline in October 2020. 57% of the respondents were in the 65+ age category, and the average length of residency at the

waterfront property was 25 years. 57% of respondents identified as male, and 78% completed a degree or diploma beyond high school.

Only two respondents indicated that they did not have a septic system, most septic systems were older, and 63% of respondents with septic systems indicated their system was installed over 15 years ago. Most respondents reported regular septic tank maintenance, 60% of tanks were pumped out within the last two years, and only one respondent had not pumped their tank in the previous five years (Figure 4.2).

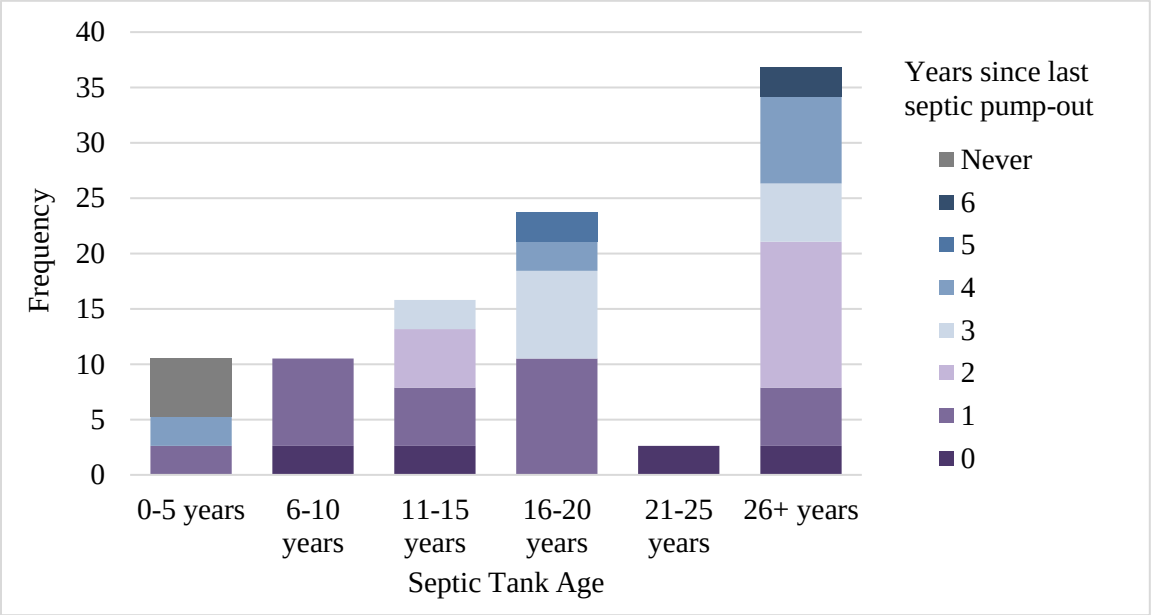

Figure 4.2 Stacked bar plot showing participant responses when asked about the age of their septic tank and when it was last pumped out, expressed as per cent of total response.

When comparing volunteers' habits in the spring of 2020 (pandemic) to 2019 (pre-pandemic) there was no significant difference in the number of days spent at their waterfront residence however, there was a slight but not significant decrease in the number of people visiting their property. Habits relating to septic system use did appear to change between 2019 and 2020, with many respondents indicating they increased their

use of detergents, water, and handwashing (Figure 4.3). Soap use had the biggest increase from 2019, with 60% of respondents indicating they increased or greatly increased their soap use in 2020. Overall, water use was reported to have increased by 33% of respondents.

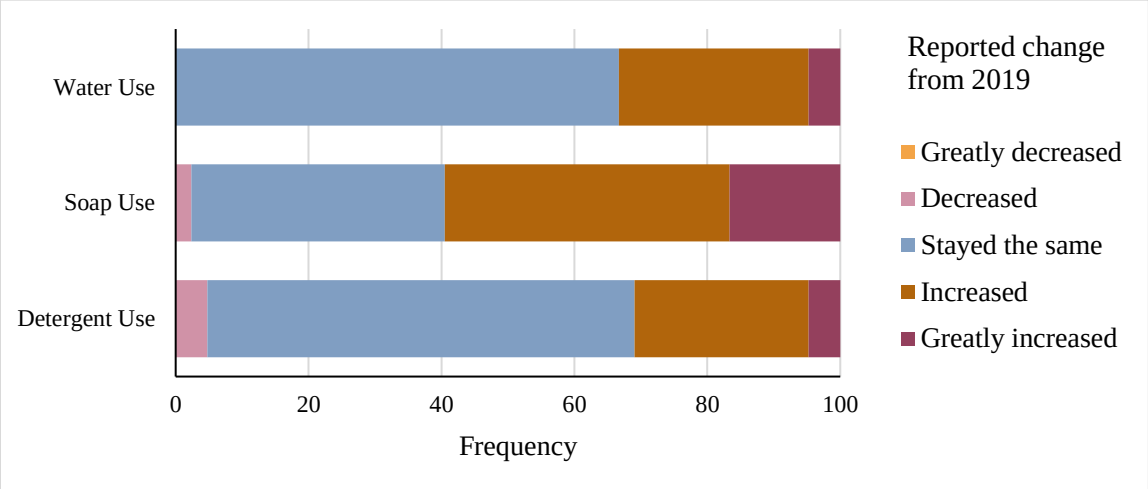

Figure 4.3 Stacked bar plot showing participant responses when asked about changes in frequency of their detergent, soap, and water in 2020 compared to 2019, expressed as per cent of total response.

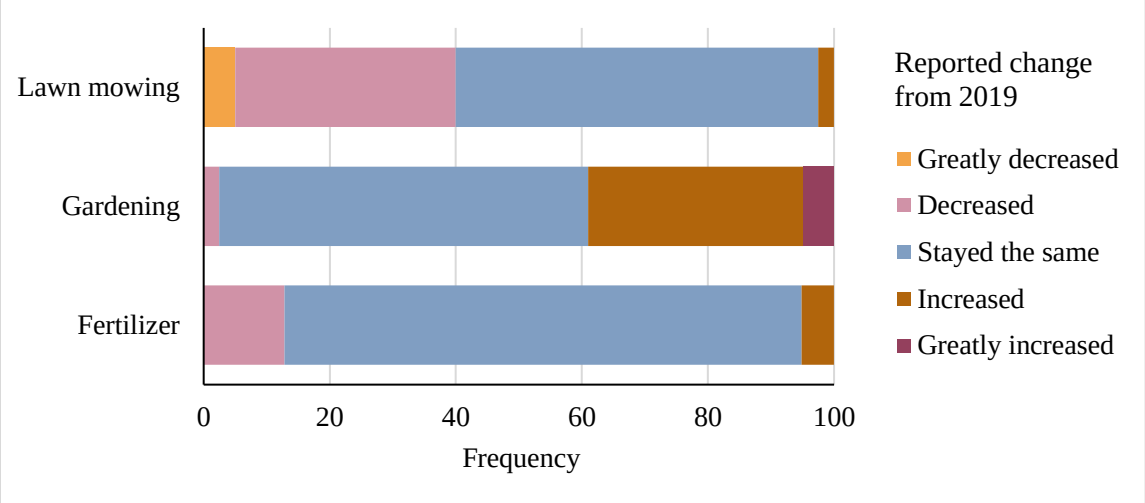

Figure 4.4 Stacked bar plot showing participant responses when asked about any changes in frequency of their lawn mowing, gardening and fertilizer use in 2020 compared to 2019, expressed as per cent of total response.

Property maintenance habits of waterfront property owners also changed slightly from 2019 to 2020. 40% of respondents indicated that they had decreased the amount of time mowing their lawn compared to 2019. Alternatively, time spent gardening increased for 39% of respondents. The amount of fertilizer used on lawns and gardens did not change much from 2019 to 2020 (Figure 4.4).

4.3.2 Kawartha Lakes Water Quality

Nutrient concentrations across the 16 Kawartha Lakes studied in 2020 reveal distinct spatial patterns, with higher nutrient levels in Lake Scugog and downstream of Lake Scugog (Figure 4.5). Lake Scugog had significantly higher total phosphorus compared to the rest of the lakes (ANOVA, $p < 0.05$). Patterns in water quality across sites sampled in 2020 were explored with non-metric multidimensional scaling (NMDS) (Figure 4.6). Two NMDS axes sufficiently comprised variation (stress = 0.041). PERMANOVA revealed significant differences in water quality profiles between watersheds ($F_{6,220}=11.21$, $p < 0.01$). The Cameron Lake Dam – Fenelon River watershed was significantly different from all watersheds except the Talbot River – Trent-Severn Waterway watershed ($p < 0.05$). The Lake Scugog – Scugog River watershed was significantly different from all other watersheds ($p < 0.05$).

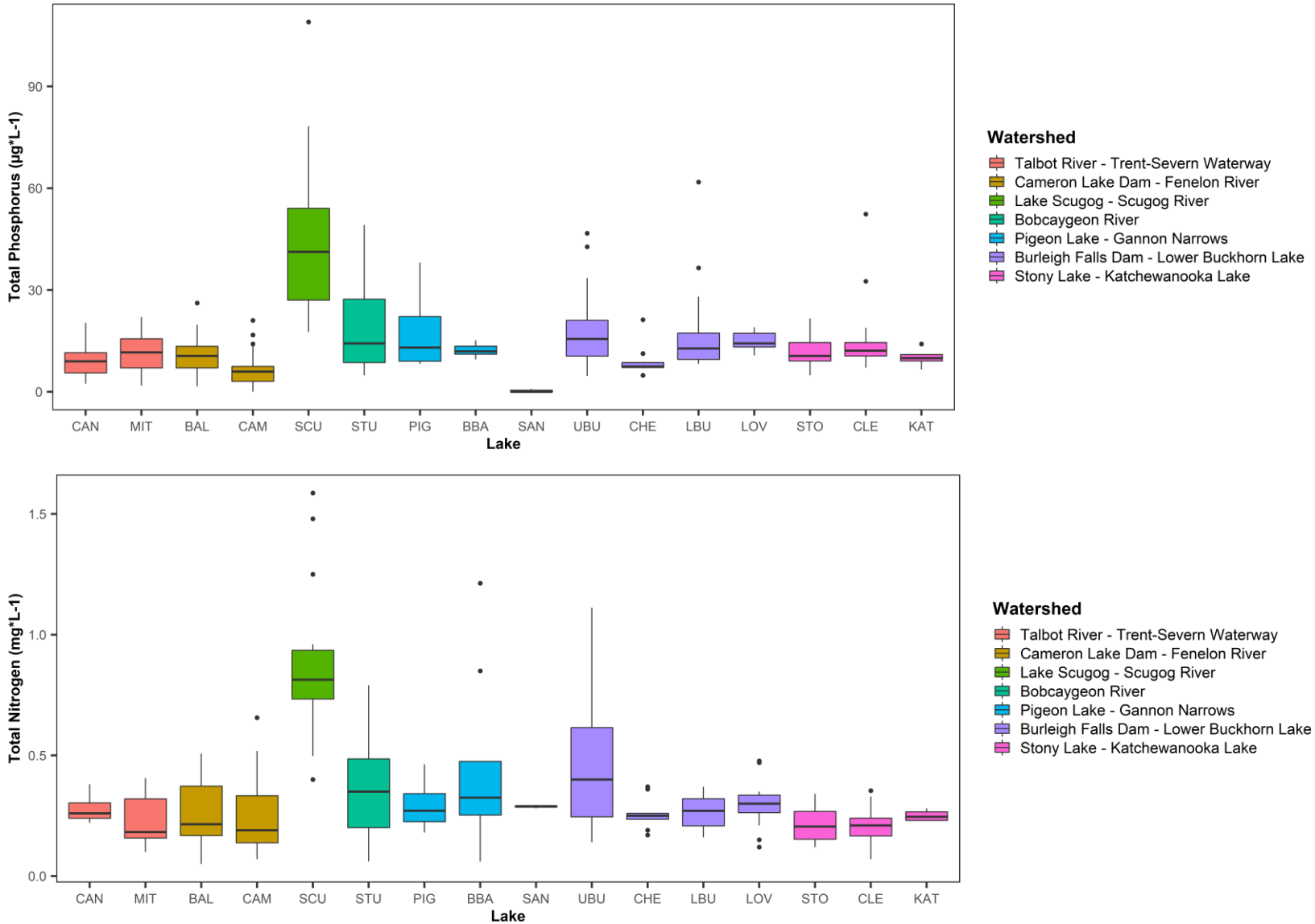

Figure 4.5 Boxplots of A) total phosphorus (µg*L-1) and B) total nitrogen (mg*L-1) in each lake, with watershed indicated by box colour.

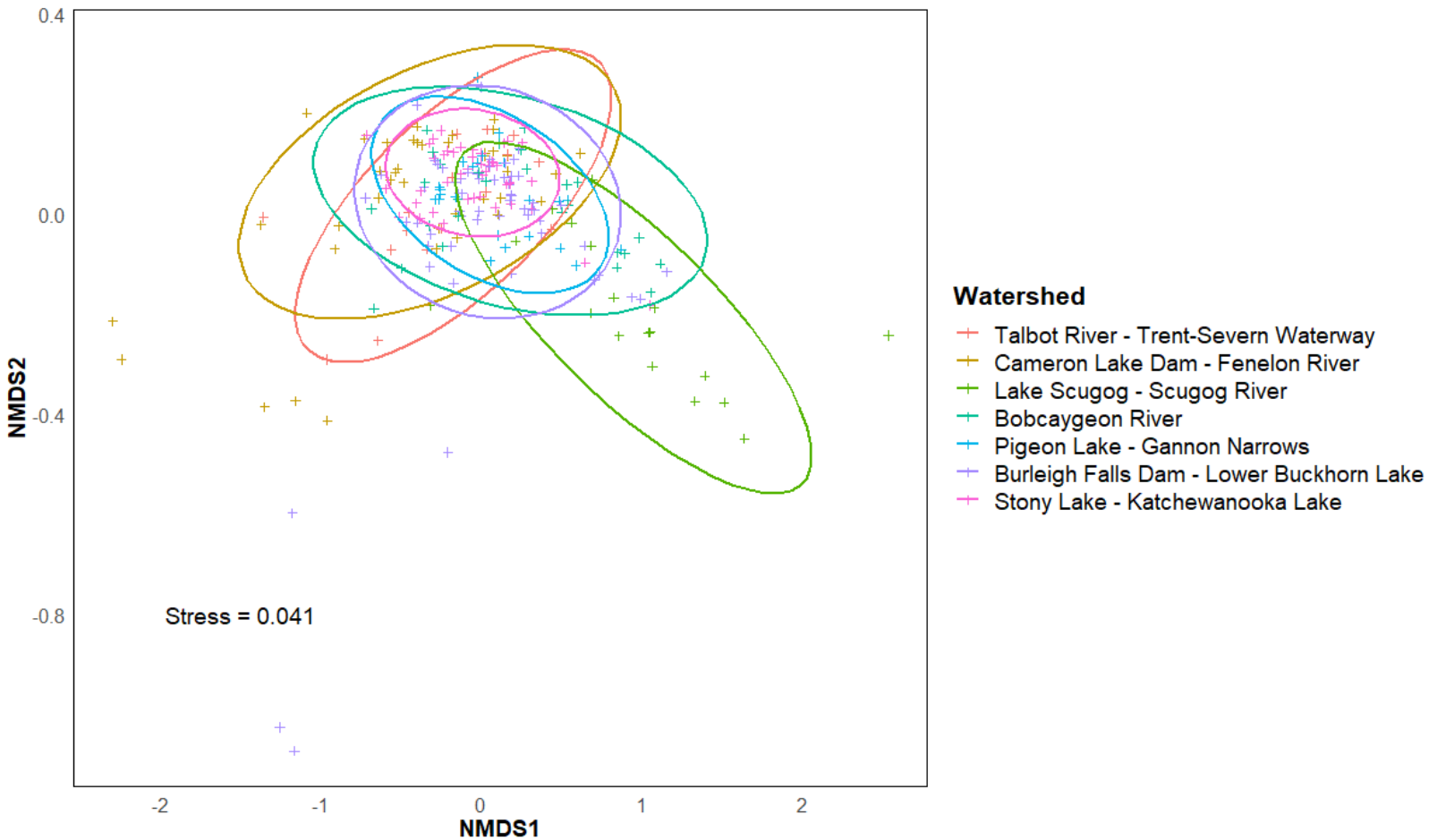

Figure 4.6 Non-metric multidimensional scaling (NMDS) biplot based on Bray-Curtis dissimilarities for water quality at sampling sites studied in 2020 (n = 228), points and ellipses indicated by watershed color.

Proportions of natural, agricultural and developed land were calculated for the quaternary watersheds and for the sub-watershed draining to each site. Natural and agricultural were the most abundant land use/cover types in all the watersheds (Figure 4.7). All watersheds except Lake Scugog – Scugog River and Bobcaygeon River had natural land cover as the dominant land cover type. Stepwise regressions were run for each water quality parameter with land use. Sub-watershed development was the only predictor selected and was significant for predicting total nitrogen, chlorophyll-a, and total phosphorus ($p < 0.05$). The strongest relationship was between total nitrogen and development ($R^2 = 0.18$), followed by chlorophyll a and development ($R^2 = 0.14$), and total phosphorus and development ($R^2 = 0.08$).

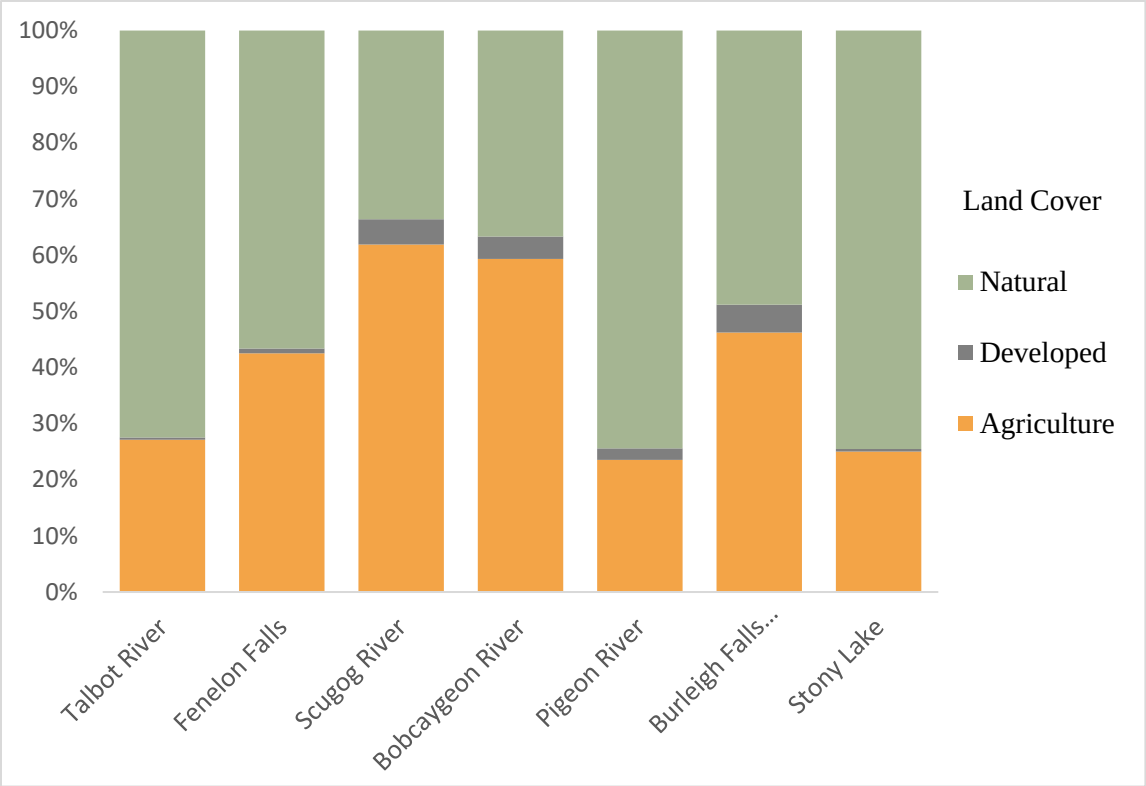

Figure 4.7 Bar plot of land cover proportions in the seven watersheds studied.

Seasonal water quality trends were also investigated for all lakes studied in 2020. Total phosphorus has some monthly variation, with trends varying by watershed (Figure 4.8). Cameron Lake Dam – Fenelon River, Pigeon Lake – Gannon Narrows, Burleigh Falls Dam – Lower Buckhorn Lake, and Stony Lake – Katchewanooka Lake watersheds had similar trends across their lakes. The biggest difference in trends within a watershed was for Canal and Mitchell Lakes in the Talbot River – Trent-Severn Waterway watershed. Monthly precipitation peaked in August at Ken Reid Conservation Area, Lindsay (112.5 mm).

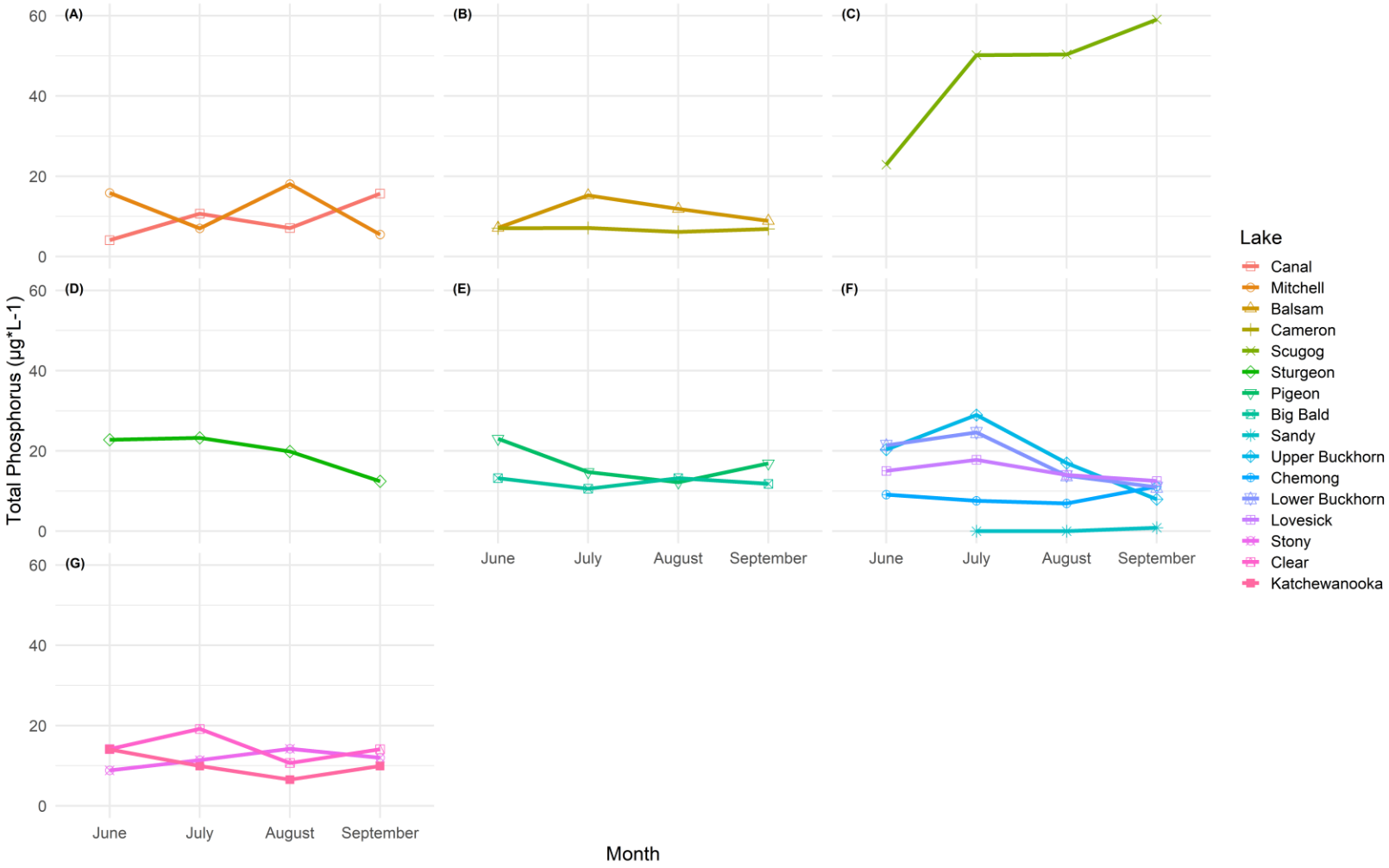

Figure 4.8 Seasonal total phosphorus (µg*L-1) levels for lakes in each watershed A) Talbot River – Trent-Severn Waterway, B) Cameron Lake Dam – Fenelon River, C) Lake Scugog – Scugog River, D) Bobcaygeon River, E) Pigeon Lake – Gannon Narrows, F) Burleigh Falls Dam – Lower Buckhorn Lake, and G) Stony Lake -Katchewanooka Lake.

4.3.3 Annual Trends

Phosphorus and nitrogen samples were collected in 2019, 2020, and 2021 in four study lakes (Balsam, Cameron, Sturgeon and Pigeon). A principal component analysis (PCA) was conducted to examine nutrient patterns across the three study years (Figure 4.9). The first axis explained 34% of the variation in the data and was driven by total organic nitrogen and total phosphorus (Table 4.1). The second axis explained 26.7% of the variation in the data and was driven by nitrates and nitrites. Ellipses were drawn by year, with all three years overlapped and spread along the first axis.

Table 4.1. Parameter scores for water quality variables on axes 1 and 2 of the principal component analysis.

	Dim.1	**Dim.2**
TP	0.792	-0.391
NO2	0.376	0.667
NO3	0.134	0.698
NH	0.544	0.392
TON	0.785	-0.315
Eigenvalue	1.70	1.34
Variance Explained (%)	34.0	26.7

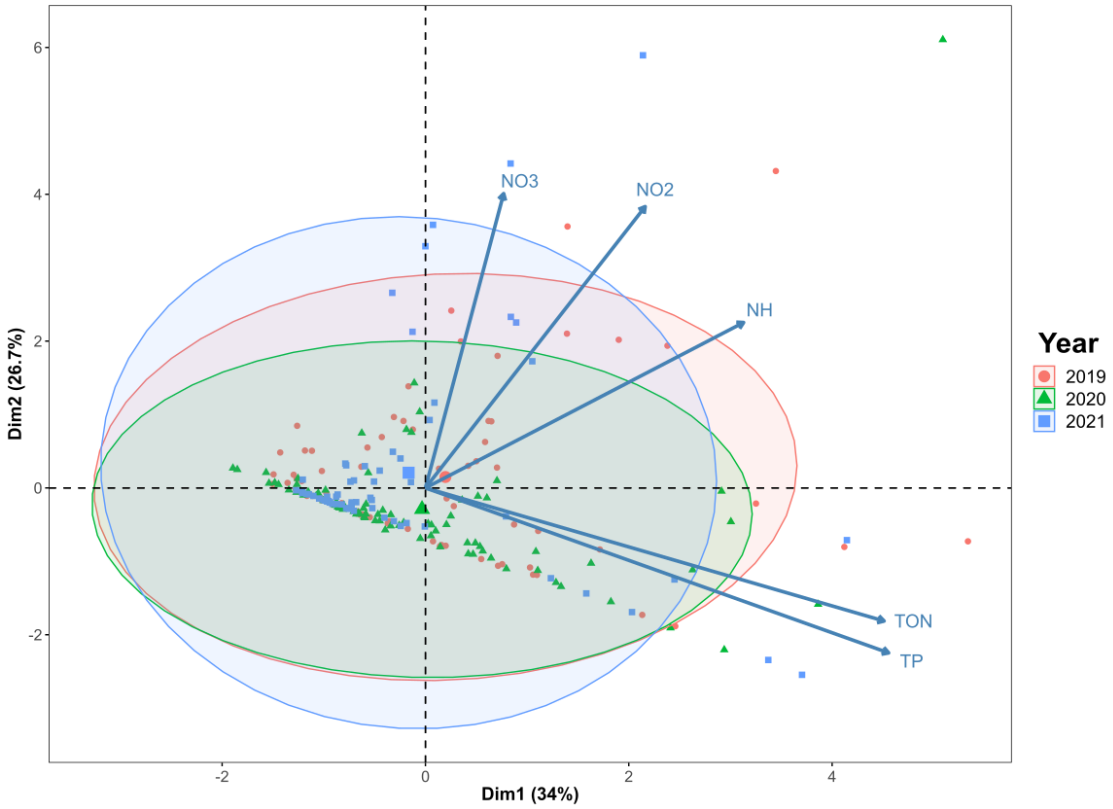

Figure 4.9 Biplot of principle component analysis axes 1 and 2 with observations from 2019 (n = 70), 2020 (n = 83), and 2021 (n = 66), with ellipses drawn by year representing multivariate normality. Direction and length of arrow indicate the association with the axes and strength of driver for each water quality variable. TP = total phosphorus, TON = total organic nitrogen, NH = ammonia/ammonium, NO2 = nitrates, NO3 = nitrates.

Nutrient levels before and during the COVID-19 pandemic were compared with ANOVA. Total phosphorus was significantly lower in 2021 than in 2019 and 2020, ammonia/ammonium levels were significantly higher in 2019 than in 2020 and 2021, and nitrates were significantly higher in 2021 than in 2020 ($p < 0.05$, Figure 4.10). Precipitation and water temperature data were compared across years to examine the potential climate impacts on water quality. Accumulated precipitation in the four days before sample collection was significantly higher in 2021 than in 2019 and 2020 ($p < 0.05$). There were no significant differences in water temperature across the years. I also

considered the impact of storm events (> 15 mm precipitation) in the week before sample collection and found no significant relationship to any water quality variable.

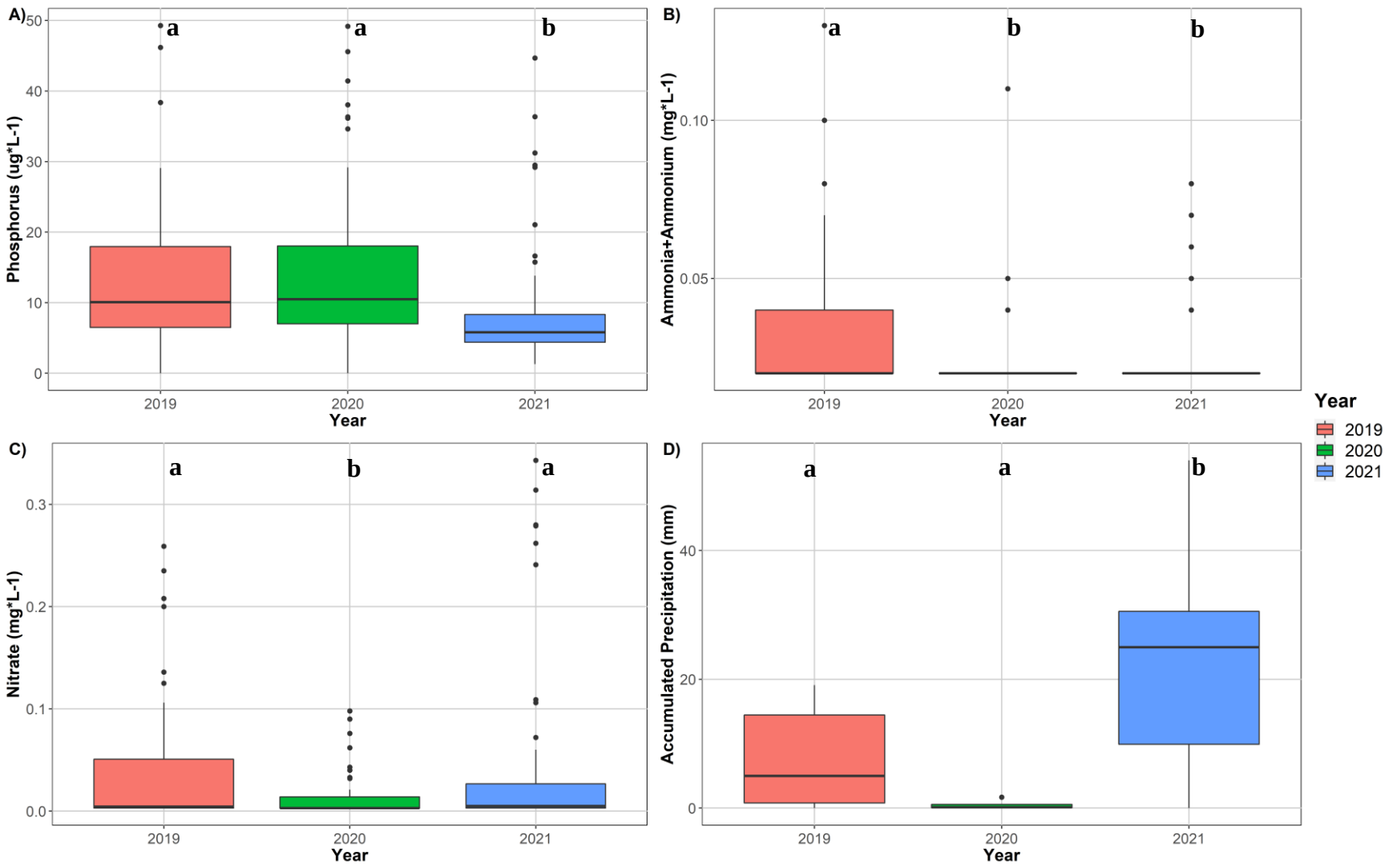

Figure 4.10 Boxplots of total phosphorus, ammonia/ammonium, nitrates, and accumulated precipitation in the four days before sampling for each sampling year from 2019-2021 (n = 24).

4.4 Discussion

The community survey results indicate that although the number of people and the amount of time spent at lake-front residences did not appreciably change from the pre-pandemic year (2019), there was an increase in detergent use, water use, and handwashing. Initially, I thought that the use of waterfront properties would have been lower in 2020 compared to the pre-pandemic year due to strict pandemic travel restrictions (Tam 2021). However, given that many of the respondents are permanent residents and other respondents retreated to their cottages to minimize COVID-19 exposure in cities, it is perhaps not surprising that the use of lake-front properties did not decrease during the first year of the pandemic. The increase in water/hygienic habits related to the COVID-19 pandemic was not surprising, as other studies have found similar changes during health crises (Głąbska et al., 2020; Park et al., 2010). The increase in these habits can incur strain on septic systems (Gray, 1995), combined with the older age of most systems (i.e., most were >25 years old), could result in septic system failure. However, I also found that most septic tanks were regularly maintained, which would reduce the likelihood of contamination of nearby waters (Macintosh et al., 2011). However, due to the anonymity of the survey, I could not directly compare septic system use to water quality parameters.

By examining the spatial patterns across the Kawartha Lakes in 2020, I found an impact of watershed land use on lake water quality (PERMANOVA, $p < 0.01$). Lake Scugog was significantly different from all other watersheds. As a headwater lake with a highly agricultural watershed and high shoreline development, it is unique in this connected system. The Cameron Lake Dam – Fenelon River watershed is one of two

watersheds not downstream of Lake Scugog, and the only watershed that it is not significantly different from is Talbot River – Trent-Severn Waterway, the other watershed not downstream of Lake Scugog. Although the Cameron Lake Dam – Fenelon River watershed also feeds the rest of the downstream lakes, this difference in water quality profiles may indicate the influence of Lake Scugog on downstream lakes. These findings indicate the importance of a lake's position in a hydrologically connected system for determining nutrient levels, matching what previous work has found (Soranno et al., 2015).

Lake Scugog had the highest percentage of agriculture in its watershed and significantly higher phosphorus levels compared to the other lakes. The impact of the high level of agricultural land use is not surprising as many others have found it to be important in driving lake nutrient levels (Arbuckle & Downing, 2001; Zampella et al., 2007). Developed land use in the sub-watershed had a positive, significant relationship with total nitrogen, total phosphorus, and chlorophyll a. The increased nutrient and chlorophyll a levels with sub-watershed development matched previous findings (Fraterrigo & Downing, 2008; Howell et al., 2012). Despite the relatively low level of land development in most of the watersheds, there was still a considerable effect on nearshore water quality.

There was no clear relationship between precipitation and total phosphorus levels in the lakes studied in 2020 (Figure 4.8). Lakes within the same watershed mostly had similar seasonal total phosphorus trends. The main deviation from this was in the Talbot-River – Trent Severn Waterway watershed, where Mitchell and Canal Lakes had opposite trends in phosphorus levels. This finding is surprising because they are connected, with

Mitchell Lake feeding Canal Lake. It is not clear if Mitchell is acting as a source of phosphorus for Canal Lake as they share a watershed. Water flow is highly regulated along the Trent-Severn Waterway, and the Kirkfield Lift Lock, located between Mitchell and Canal Lakes, is the second-largest of its type in the world (Lake Simcoe Region Conservation Authority & Kawartha Conservation, 2016). Further investigation into water flow within the watershed may reveal the cause for the difference in nutrient trends.

Nutrient levels also varied annually in the four lakes studied from 2019-2021. The PCA explained 60.7% of the variation in the data with the first two axes however, there was a high amount of overlap in the ellipses for each year, indicating there was not much difference in overall water quality between years. When annual relationships were explored for total phosphorus, ammonia/ammonium, and nitrates some patterns began to emerge. It was expected that there would be a difference in nutrient levels during the pandemic, when lockdown measures were in place, compared to before the pandemic. Only ammonia/ammonium complied with this expectation, where levels were significantly higher in 2019 than in 2020 and 2021. In the agriculture-dominated watersheds, fertilizer and manure from farming operations are common sources of ammonia/ammonium. However, farming activities did not decline, but surged during the pandemic as an essential activity (Statistics Canada, 2022). Since I only had data from one pre-pandemic year in this study, I may lack the statistical power to confirm if the lower ammonia/ammonium measured in 2020 and 2021 was an effect of the pandemic.

Another important factor to consider when interpreting inter-annual variation in nutrient levels is precipitation, especially storm events which can drive nutrients from the

landscape into lakes. There was an increase in precipitation in the days before sample collection in 2021, but only nitrates had a corresponding increase in 2021 (Figure 4.10). Fertilizers are a common source of nitrates, and the high levels of agricultural land use in the watersheds may play a role in determining nitrate levels. Alternatively, small-scale land use has also been shown to be important for determining nearshore water quality (Smith et al., 2021), shoreline septic systems and fertilizers used for gardening, combined with more hard surfaces may be contributing to increased surface runoff of nitrates. Finally, total phosphorus had significantly lower levels in 2021 than in 2019 and 2020, which was the opposite of precipitation trends. This is an interesting finding since other studies have shown precipitation to be an important driver of phosphorus in surface waters (Fraser et al. 1999; Hart et al. 2004). Overall, the similar phosphorus concentrations between 2019 and 2020 are in line with the community survey results, which indicated no change in the number of people or duration of stay at lake-front properties between 2019 and 2020.

In summary, this study did not detect a significant signal in lake nutrient patterns associated with the pandemic. In effect, the anthropopause did not appear to cause a change in baseline nutrient conditions across the Kawartha Lakes. The lack of signal from the pandemic indicates that the positive environmental impacts of lockdown measures seen in other studies may be restricted to more densely populated areas where changes in industrial activity, tourism, and transportation have consequential impacts on the environment. In rural areas like the Kawartha Lakes, agricultural practices continued, and the reduced tourism did not have a detectable impact on the nearshore water quality of lake-front seasonal cottages and residences. The Kawartha Lakes also did not have as

much of a reduction in tourism as other areas in the summer of 2020 due to fairly consistent domestic travel (Giunta 2020). Even though I did not detect an impact of the pandemic on nutrient levels, this study is a critical reminder that environmental monitoring can and needs to continue throughout disturbances such as lockdown measures. I recommend that lake managers consider developing robust community science programs in their jurisdictions as at least a fall-back measure to minimize disruption to annual lake monitoring programs.

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Chapter 5. **Evaluating the role of land-use type and scale in nearshore water quality and biological community structure in the Kawartha Lakes (Balsam, Cameron, Sturgeon, and Pigeon)**

5.1 Introduction

Humans are drawn to live near water, but as we alter the physical structure and composition of lake shorelines, this often degrades the aquatic features that originally drew us to the water in the first place. Initial impacts of human activity come from physical alteration of the shoreline: building docks and other structures, artificially reinforcing shorelines, and removing vegetation. These changes often lead to a loss of aquatic plants, coarse woody structure, and canopy shade. As a result, all of these changes can have negative impacts on aquatic communities (Christensen et al., 1996; Hicks & Frost, 2011; Johnson & Jones, 2000). Additionally, there are direct chemical alterations from increased nutrient and contaminant sources, such as fertilizers, road salts, and septic systems (Reay, 2004; Rosenberger et al., 2008). These impacts are further enhanced by the physical alterations that increase hard surfaces resulting in more surface runoff (Arnold Jr & Gibbons, 1996).

Shoreline modifications have direct impacts on nearshore nutrients (Dillon et al., 1994), sediment loads (Meadows et al., 2005), and bacterial contamination (Reay, 2004). Of particular interest are nutrient inputs, which drive primary production in the lake. It has been established that watershed development leads to increased phosphorus and nitrogen levels (Soranno et al., 2015; Zampella et al., 2007). Additionally, recent studies have found the importance of considering multiple spatial scales to determine land use – water quality relationships in rivers and lakes (Lei et al., 2021; Pratt & Chang, 2012;

Vera Mercado & Engel, 2021). As the intersection between the terrestrial and aquatic environments, the nearshore zone is the ideal location to examine land use impacts on water quality. The spatial variation in the nearshore zone will likely provide an accurate reflection of land use at both large and small scales. This information will be particularly useful because nearshore water quality can help forecast overall lake health (Chang & Rossmann, 1988; Lambert & Cattaneo, 2008).

The nearshore zone is where humans enjoy the lake, but it is also vital for aquatic organisms, with almost all lake inhabitants spending some of their life cycles in the nearshore zone (Vadeboncoeur et al., 2011). Consequently, shoreline alterations impact the entire food web, from primary producers to top predators (Chang & Rossmann, 1988; Dustin & Vondracek, 2017; Goforth & Carman, 2009; Hall et al., 2003; Rosenberger et al., 2008). Physical alterations reduce the complexity of the nearshore zone and can decrease habitat availability. Nutrient additions alter primary productivity and can have direct and indirect effects on aquatic organisms. Although we know that shoreline alterations impact all levels of the food web in the nearshore, it is not clear how interactions between these organisms are affected (Meadows et al., 2005). Previous work has examined the impact of human activity on multitrophic interactions, but these relationships have not been examined for the nearshore zone of lakes (Stamenković et al., 2021; Taranu et al., 2021; Usio et al., 2017).

The current study examines the nearshore zone of four contiguous Kawartha Lakes: Balsam, Cameron, Sturgeon, and Pigeon. Located just outside the Greater Toronto Area (GTA), these lakes have been a popular tourist destination since the early 1900s, and recently the number of permanent residents has also been growing (Kawartha

Conservation, 2015b). The watersheds have a mix of natural land cover and agricultural land use. Development is concentrated along the shoreline of these lakes, the degree of shoreline alteration ranges from 29% on Pigeon to 74% on Sturgeon Lake (Kawartha Conservation, 2014b, 2016). Nearshore water quality was monitored in these lakes by working with a local watershed authority (Kawartha Conservation) and local community volunteers who collected samples from their property shorelines. Water quality parameters from these samples were compared with land use at multiple spatial scales, from 100 m buffers to sub-watersheds. Additionally, the biological communities were monitored at a subset of sampling sites to examine the relationships between nearshore phytoplankton, zooplankton, and macroinvertebrate communities, water quality, and human activity at the shoreline. It was expected that land use at large and small scales would be associated with nearshore water quality. It was also predicted that associated effects of land-use on water quality would have direct and/or indirect effects on nearshore biological communities.

5.2 Methods

5.2.1 Sample collection and processing

The four lakes included in this study, Balsam, Cameron, Sturgeon, and Pigeon, are located centrally in the chain of Kawartha Lakes that are part of the Trent Severn Waterway. Water flows through these lakes in the following order: Balsam, Cameron, Sturgeon, Pigeon (Figure 5.1). Background on these lakes and their land-use patterns is provided in Chapter 1, and physical characteristics of each lake are provided in Table 5.1.

Water quality samples were collected by community scientists recruited through local organizations. Our research partner, Kawartha Conservation, had existing

relationships with the Balsam Lake Association, who recruited volunteers on their lake. Sturgeon Lake had an existing community scientist volunteer group previously coordinated by Kawartha Conservation. Social media and public meetings were also used by Kawartha Conservation to recruit additional volunteers. Additionally, some volunteers recruited through the Kawartha Lake Stewards Association (KLSA) for the 2020 pandemic monitoring study (Chapter 3), collected samples in 2021 as well. Sampling sites were limited to where community scientists had waterfront access and extra effort was placed on recruiting volunteers in areas without nearby sampling sites. Although we had a large group of community scientists (n = 47), there were some spatial gaps (i.e. west shore of Pigeon Lake) that could not be filled by community scientist volunteers.

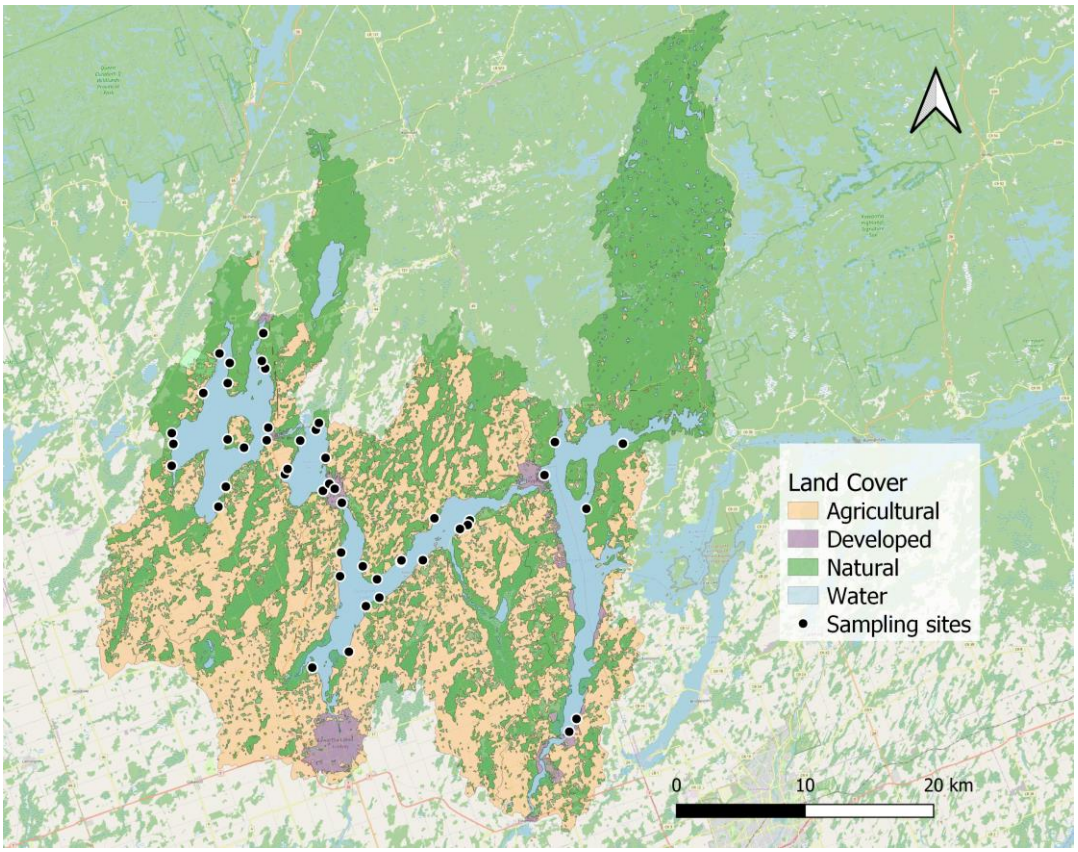

Figure 5.1 Water quality sampling sites (n = 47) and watershed land use for Balsam, Cameron, Sturgeon and Pigeon lakes.

Community science volunteers were trained following the methods described in Chapter 2 (if recruited in 2019), or virtually (if recruited in 2021), as described in Chapter 4. All samples were processed following the methods described in Chapter 1, except for chloride, which was not measured due to the method detection limit of the ion-selective probe (~ 60 mg/L) not being low enough for most samples in this set of Kawartha Lakes. Findings from volunteer collected samples are presented in the results section "Water quality and land use".

Table 5.1. Physical characteristics of each study lake, from Kawartha Conservation (2014b, 2015b, 2016)

	Average Depth (m)	**Shoreline Length (km)**	**Volume (m^3)**	**Flushing Rate**	**Shoreline land development (%)**
Balsam	4.8	98	2.37[8]	3.5	45
Cameron	9.3	43	1.00[8]	12.5	50
Sturgeon	3.5	100	1.63[8]	20	74
Pigeon	3.3	145.2	1.89[8]	12	29

Biomonitoring was conducted entirely by myself and research assistants at a sub-set of the community science water monitoring sites (eight sites on Balsam and Pigeon, seven sites on Cameron and Sturgeon). Biotic communities and water samples were collected monthly from June – September in 2021. All biological communities were collected following the methodologies described in Chapter 3. The only change was that Hester-Dendy artificial substrates were deployed for four weeks instead of three weeks. Additionally, shorelines at each biological monitoring site were classified as concrete, armour stone, rip-rap, or natural (Table 5.2).

Table 5.2. Count (n) of shoreline type on each lake.

	Natural	Rip-rap	Concrete	Armour stone
Balsam	4	3	1	0
Cameron	2	0	3	1
Sturgeon	4	1	2	0
Pigeon	4	0	1	3

5.2.2 Statistical Analysis

All statistical analyses were completed using RStudio (RStudio Team, 2016). Exploratory data analysis found an outlier at a site on Pigeon Lake. A note was made during sample processing that a large clump of filamentous algae was present in the sample indicating improper sample collection, thus the sample was removed from future analysis. Any other outliers from community scientist collected samples were compared to previous values at the site, including those collected by myself during biomonitoring to determine if the outlier was caused by sampling error. No other sampling points were removed following data inspection. Environmental and biological parameters were evaluated for normality with the Shapiro-Wilk test, variables that had a non-normal distribution of residuals and were log-transformed for all analyses.

Differences in water quality parameters between years and shoreline types were determined using analysis of variance (ANOVA) with significance set at $\alpha = 0.05$. A principal component analysis (PCA) was used to examine the role of sub-watershed land use in influencing water quality parameters and lake groupings. The PCA was conducted using the *factoextra* (Kassambara & Mundt, 2017) and *ggplot2* (Wickham, 2011) R packages, all of the environmental variables were standardized for the analysis. Permutational analysis of variance (PERMANOVA) was conducted to examine differences in water quality between the lakes using the *vegan* package (Oksanen et al.,

2013). Correlations between environmental parameters were assessed with Spearman correlation analysis and visualized with *corrplot* (Wei et al., 2017).

Land use analysis was conducted with land cover data from Kawartha Conservation, data files were last updated in 2017. Land cover was categorised as either natural, agricultural, developed or water. Log-transformed water quality parameters were regressed with sub-watershed land use (agriculture and development) to examine the impact of watershed land use on nearshore water quality. To determine the importance of land use at various scales, stepwise multiple linear regressions (MLR) were completed for each water quality parameter at each land use scale (100 m, 500 m, 1000 m, sub-watershed) following the method of Vera Mercado and Engel (2021). Only significant models ($p < 0.05$) determined by stepwise selection are presented.

Taxonomic diversity was calculated using Simpson's Index of Diversity using the *vegan* R package (Oksanen et al., 2013). Functional diversity (FD) was determined using taxa traits and community composition matrices to build dendrograms for each community. FD was calculated from the sum of branch lengths for the taxa in each community, based on the code from Petchey and Gaston (2002). Traits were selected based on ecological relevance and data availability. Traits for the phytoplankton community included: cell/colony shape, cell size, cell motility, tendency to form colonies, and heterotroph/mixotroph/autotroph status (Longhi & Beisner, 2010; Rimet & Druart, 2018). Zooplankton traits included: body length, feeding type, and food source (Barnett et al., 2007; Obertegger & Flaim, 2015). Macroinvertebrate traits included: size, habitat, trophic status, thermal tolerance, and respiration type (Vieira et al., 2006). Two additional metrics for assessing macroinvertebrate communities were calculated:

Hilsenhoff's Family Biotic Index (FBI) and EPT index (% Ephemeroptera, Plecoptera, and Trichoptera taxa). Higher FBI scores result from more tolerant taxa present in a given community, potentially indicating poor water quality conditions. Higher EPT scores result from a higher abundance of the sensitive families listed above, potentially indicating better water quality conditions.

Multiple linear regressions were conducted with abundance and richness for each community, calculated with the *vegan* (Oksanen et al., 2013) package, and environmental variables. Detrended correspondence analysis (DCA) was conducted for each community to determine if taxa had a linear or unimodal response to environmental variables. The longest DCA gradient was < 3 for the phytoplankton and macroinvertebrate communities and < 4 for the zooplankton community. Although the longest DCA gradient for zooplankton was above three, the data were Hellinger transformed to allow for linear ordination methods (Legendre & Gallagher, 2001). The phytoplankton and macroinvertebrate communities were also Hellinger transformed to give low weight to rare species. Based on the result of the DCAs, redundancy analysis (RDA) was conducted for each Hellinger transformed community to examine the relationship between taxa and water quality parameters. Environmental drivers for each community matrix were determined with multivariate analysis of variance (MANOVA), and only significant ($p < 0.05$) environmental parameters were included in the final RDA model. The RDA matrix was calculated based on Bray-Curtis dissimilarity calculated with *vegan* (Oksanen et al., 2013), *mice* (Van Buuren & Groothuis-Oudshoorn, 2011), and *dplyr* (Wickham et al., 2015) and plotted with *ggplot2* (Wickham, 2011).

To determine the underlying factors driving phytoplankton, zooplankton, and macroinvertebrate abundance and diversity a piecewise structural equation model (SEM) was constructed using the *PiecewiseSEM* package (Lefcheck, 2016). Piecewise SEM was used because it considers the hierarchal structure of the data (sampling sites within lakes) by fitting each mixed effect model separately. The piecewise SEM is preferable to the traditional SEM because it allows for complex causal models with relatively low sample sizes (Shipley, 2000). This approach allowed us to test the relative role of human impacts on density and diversity of each community (phytoplankton, zooplankton, and macroinvertebrates) through direct and indirect relationships.

Each linear mixed effect model was constructed with the *nlme* package (Pinheiro et al., 2017), with lake as a random effect and environmental parameters and human impacts as fixed effects. A hypothesized basis model was constructed based on the literature, and a directional separation (d-sep) test for independence claims, built into the *Piecewise SEM* package, was used to check for any paths missing from the model (Lefcheck, 2016). Variance inflation factors (VIFs) were checked for each model using the *DAAG* package (Maindonald et al., 2015), variables with VIF > 3 were removed to avoid multicollinearity. To find the best model the method of Stamenković et al. (2021) was used, subsequently removing explanatory variables to reduce Akaike's Information Criterion (AIC). The resultant graph was visualized with NodeXL (Smith et al., 2010).

5.3 Results

5.3.1 Water quality and land use

Water quality results from across the study region were all below levels of concern according to the provincial water quality objectives (PWQO) (Appendix B,

Table B1.2). Phosphorus levels indicate oligotrophic status for Balsam and Cameron lakes and mesotrophic status for Sturgeon and Pigeon lakes (Figure 5.2). Phosphorus levels were significantly higher in Sturgeon and Pigeon lakes than in Balsam and Cameron lakes (PERMANOVA, $p < 0.05$). Nitrogen levels were also relatively low but followed the same pattern as phosphorus, with higher levels in Sturgeon and Pigeon (Figure 5.3, $p < 0.05$). Chlorophyll a levels also follow this pattern, where Pigeon in particular had very high chlorophyll a levels in 2019, when it also had much higher phosphorus levels. Only Pigeon had a significant positive relationship between log chlorophyll a and log phosphorus (Figure 5.4, $p < 0.05$).

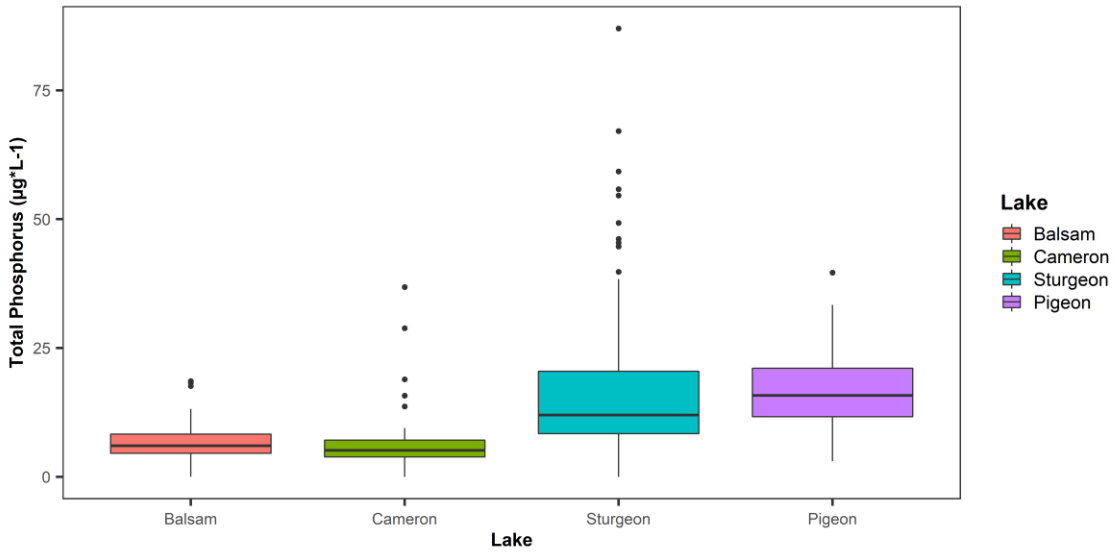

Figure 5.2 Boxplot comparing total phosphorus (µg*L-1) across the four study lakes. ANOVA was used to compare sample means ($\alpha = 0.05$).

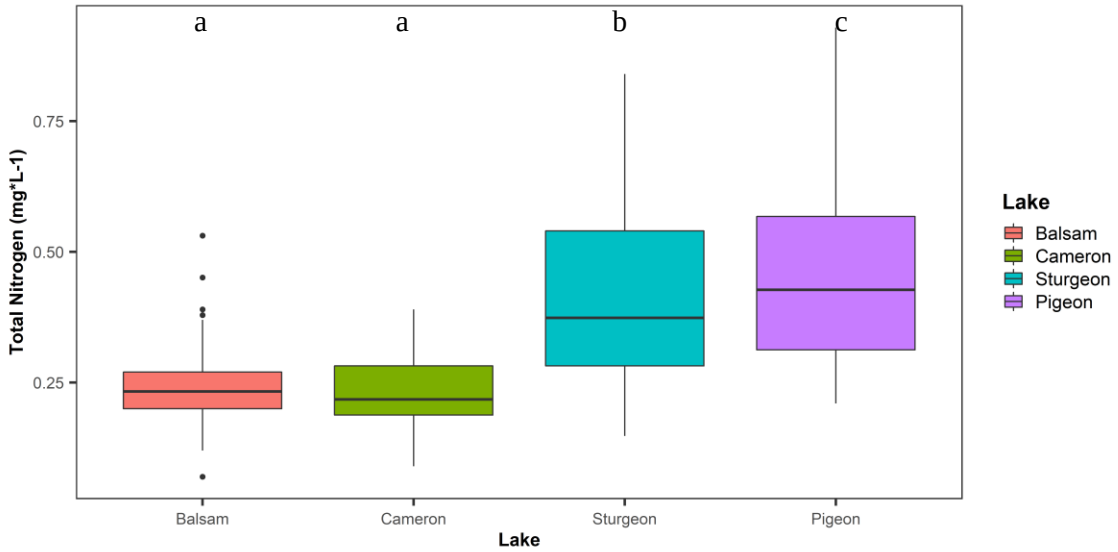

Figure 5.3 Boxplot comparing total nitrogen (mg*L-1) across the four study lakes. ANOVA was used to compare sample means (α = 0.05). One outlier removed for image clarity, full figure in Appendix B.

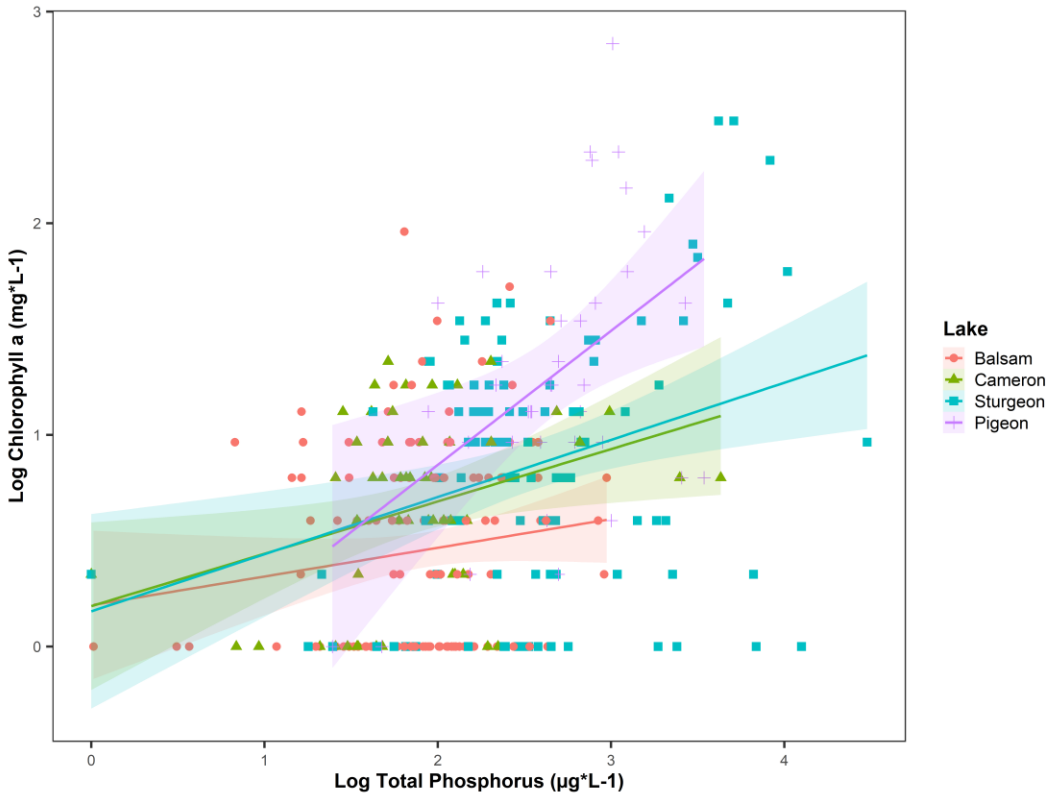

Figure 5.4 Scatterplot showing the significant linear regression (R^2adj=0.25, F(7,317) = 16.74, p < 0.001) between log (x + 1) transformed total phosphorus (µg*L-1) and log (x + 1) transformed chlorophyll a (mg*L-1) for each lake.

A principal component analysis (PCA) was conducted with community science data and sub-catchment land use for each site, the first two axes of the PCA explain 43.2% of variation in the data (Figure 5.5). The lakes show clear groupings with Balsam and Cameron and Sturgeon and Pigeon grouped, and a PERMANOVA statistically confirmed the groupings of lakes with Balsam and Cameron significantly different from Sturgeon and Pigeon ($p < 0.05$). Natural land cover appears to be important for this separation of the sites, driving the positive scores for Balsam and Cameron on axis 1 (Table 5.3). Agricultural and developed land use were associated with many water quality parameters, driving negative scores on axis 1. Road density was associated with nitrates, nitrites, and ammonia + ammonium. Alternatively, *E. coli* and fecal coliforms do not appear to have a strong association with land use at the sub-watershed scale. Correlation analysis was used to examine relationships between environmental variables in the PCA (Figure 5.6). Chla had significant relationships with TP, TON, turbidity, and nitrates (Pearson correlation, $p < 0.05$). TP and TON had a strong, positive relationship, and both also correlated with turbidity, *E. coli*, and fecal coliforms ($p < 0.05$). The impact of a storm event (> 15 mm precipitation in 24 hours) in the week before sample collection was investigated with repeated measures ANOVA, and only fecal coliforms and TSS were significantly higher when a storm event occurred ($p < 0.05$).

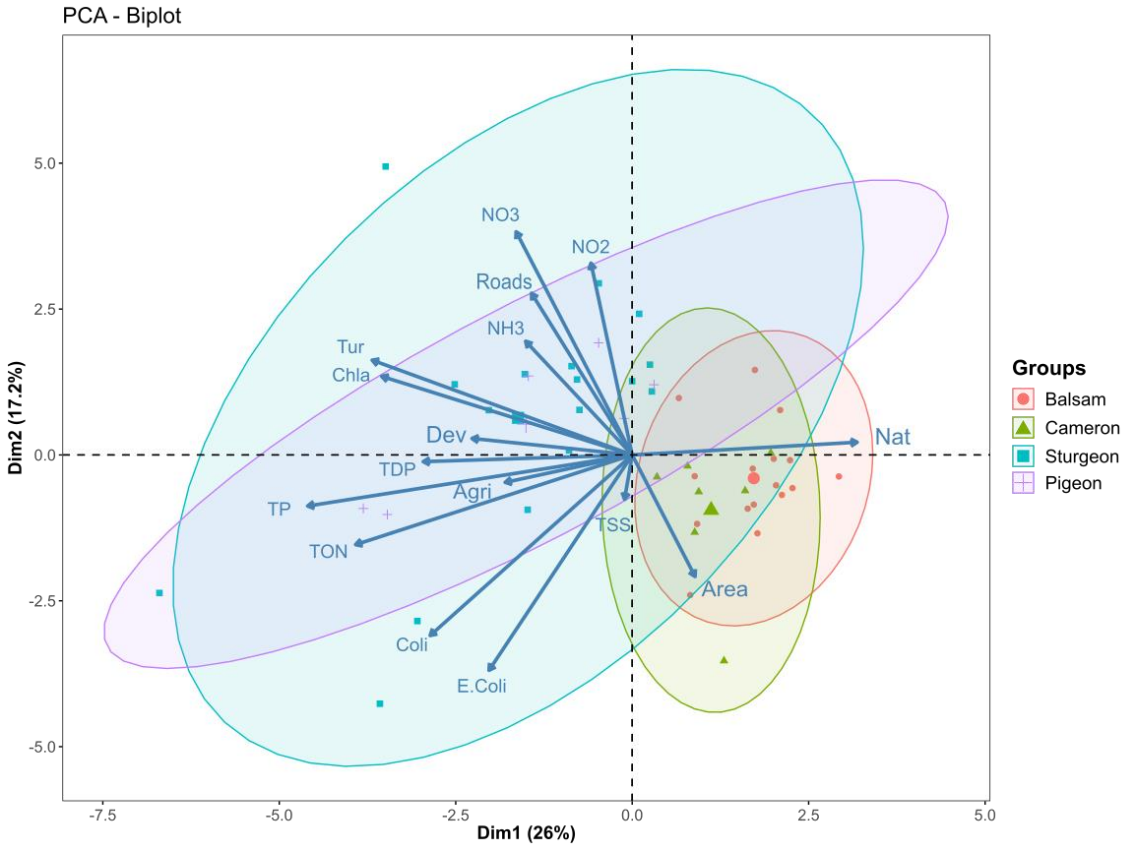

Figure 5.5 . Principal component analysis biplot of water quality observations from Balsam (n = 131), Cameron (n = 48), Sturgeon (n = 102), and Pigeon (n = 38) lakes. Points are grouped by lake, direction and length of arrow indicate strength of correlation with the axes. TP = total phosphorus, TDP = total dissolved phosphorus, TON = total organic nitrogen, NH = ammonia/ammonium, NO2 = nitrates, NO3 = nitrates, Tur = turbidity, Chla = chlorophyll-a, TSS = total suspended solids, Coli = fecal coliforms, Dev = sub-watershed developed land use, Agri = sub-watershed agricultural land use, Nat = sub-watershed natural land cover, Area = sub-watershed area.

Table 5.3. Water quality and land use scores on PCA axes 1 and 2.

	Axis 1	Axis 2
Dev	-0.440	0.055
Agri	-0.349	-0.091
Nat	0.618	0.042
Area	0.173	-0.404
Roads	-0.277	0.538
TP	-0.892	-0.170
TDP	-0.574	-0.022
Tur	-0.715	0.315
TSS	-0.021	-0.147
Chla	-0.689	0.261
Coli	-0.555	-0.600
E. coli	-0.393	-0.716
NO2	-0.113	0.638
NO3	-0.319	0.742
NH3	-0.293	0.378
TON	-0.760	-0.298

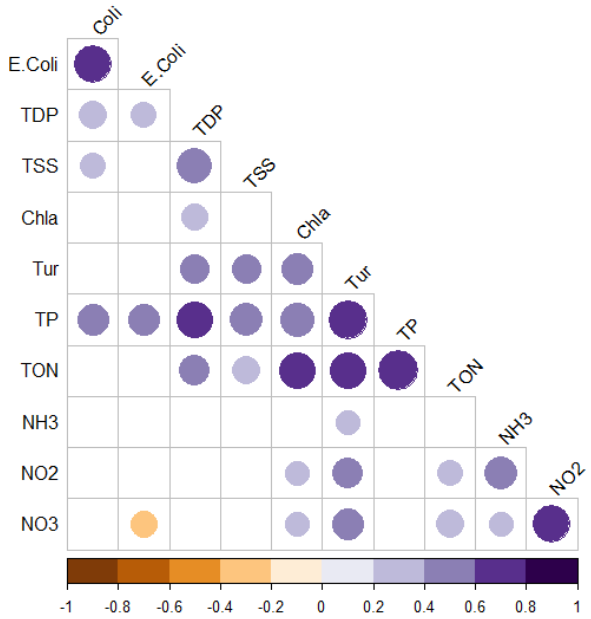

Figure 5.6 Correlation plot for environmental variables with only significant relationships displayed, size and colour of the circle indicates strength of the relationship.

Sub-watershed land use (agricultural and developed) was compared to water quality variables with an interaction of lake, the water quality variables tested were selected based on associations in the PCA. Only Sturgeon and Pigeon lakes had a positive, significant relationship with watershed land use and total phosphorus (Figure 5.7, $p < 0.05$). Nitrogen and turbidity were significantly higher in Sturgeon and Pigeon, but there were no significant interactions with land use. Pigeon had significantly higher chlorophyll a levels, but there was no significant relationship between chlorophyll a and land use. Relationships between land use and water quality parameters were evaluated with stepwise regression models at buffer and sub-catchment land use scales (Table 5.4). Most of the significant relationships between land use and water quality were at the sub-watershed scale, and the 1000 m and 500 m buffer scales each had one significant model. At the sub-watershed scale road density was included in all the models, indicating its importance in influencing water quality at this scale. Developed land use was also included in three models, one at each scale and agricultural land use was only selected as a significant predictor in the 500 m buffer regression model. Model R^2 values were relatively low for all the models indicating that there are likely other important factors influencing water quality in these lakes.

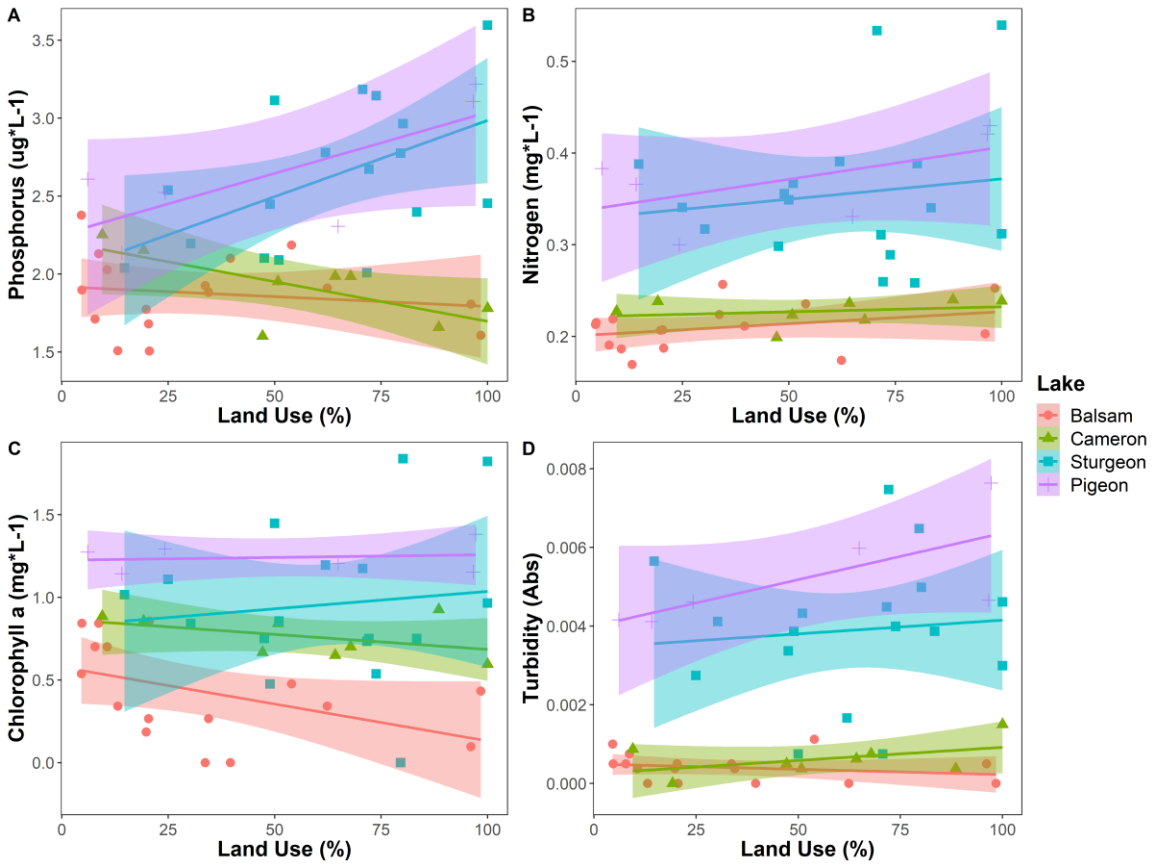

Figure 5.7 Scatterplots with regressions lines visualizing the relationship between sub-watershed land use (agricultural plus developed) and log (x+1) transformed water quality variables with 95% confidence intervals for each lake. Linear regressions with an interaction between land use and lake found significant relationships between land use and total phosphorus for Sturgeon and Pigeon lakes ($R^2 = 0.60$, $p < 0.001$).

Table 5.4. Results of forward stepwise linear regression to assess land-use type and scale as predictor variables for water quality parameters. Final models are presented below based on optimal AIC, and statistically significant models ($p < 0.05$). Values for each land-use type are regression model estimates, blank cells indicate the land-use type was not selected from the forward selection.

	Agricultural	**Developed**	**Natural**	**Road density**	**Model R^2**
Sub-watershed					
E. coli		0.203		-0.740	0.13
Chla				0.528	0.13
NO2				0.0015	0.09
NO3				0.077	0.22
1000 m buffer					
E. coli		0.730			0.08
500 m buffer					
Turbidity	0.005	0.009			0.21

Although land-use models provided some insight into drivers of nearshore water quality there remained a lot of unexplained variance for each water quality parameter. As an interconnected system, water flow needs to be considered as well, since upstream lakes are also contributing to downstream lake water quality. Although all lakes receive water from northern tributaries that flow through forested watersheds, Sturgeon and Pigeon lakes have agriculture-dominated southern tributaries that appear to be contributing to increased phosphorus levels in these lakes (Figure 5.8). Due to their different water quality profiles Sturgeon and Pigeon were analyzed separately for within

lake trends, and it was found that latitude had a significant, negative relationship with phosphorus ($p < 0.05$).

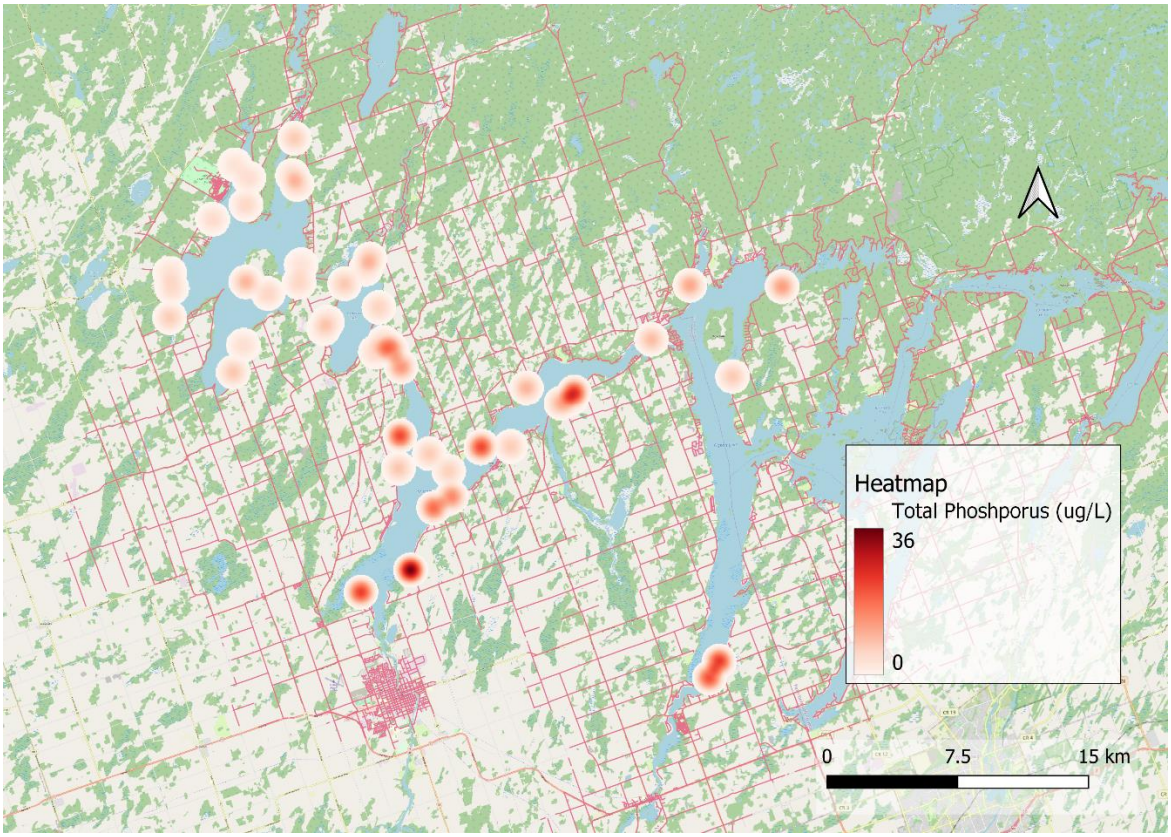

Figure 5.8 Heatmap of average total phosphorus (µg*L-1) from community science samples.

5.3.2 Biomonitoring findings

Water quality samples collected during biological sample collection showed similar spatial patterns to water samples collected by community scientists (Table 5.5). Balsam and Cameron lakes had phosphorus levels below 10 $\mu g*L^{-1}$ and chlorophyll a levels were relatively low as well (< 1.0 mg/L). Sturgeon and Pigeon phosphorus levels were above 10 µg/L and they had higher chlorophyll a levels. The increased primary productivity in Sturgeon and Pigeon lakes extended to submerged macrophytes that were sampled in August. Though not statistically significant, macrophyte biomass was slightly

higher at Sturgeon and Pigeon Lake sites (Figure 5.9). Two-way ANOVA tests determined that shoreline type interacted with lake and affected total phosphorus concentration ($F(5,132) = 3.83$, $p < 0.05$). Rip-rap shorelines had significantly lower total phosphorus compared to armour stone and natural shorelines (Figure 5.10, $p < 0.05$). The same relationship with shoreline type existed for Chla, without a significant interaction of lake ($p < 0.05$). Total phosphorus in Balsam and Cameron Lakes was significantly lower than in Sturgeon and Pigeon Lakes ($p < 0.05$). Two-way ANOVAs were also run to test for relationships between shoreline type and each biotic community's abundance and diversity and no significant relationships were detected.

Table 5.5. Means and standard deviations (in brackets) for environmental variables collected concurrently with biological samples in 2021 for each lake (n = 144).

	TP (µg/L)	TDP (µg/L)	Chla (mg/L)	TSS (mg/L)	TSOS (mg/L)	Tur (Abs @750nm)	TN (mg/L)	TON (mg/L)	pH	Cond (µS/cm)	DO (mg/L)
Balsam	4.49 (2.53)	1.35 (3.07)	0.42 (0.63)	2.8914 (6.7319)	0.0607 (2.426)	0.0010 (0.0415)	0.210 (0.033)	0.201 (0.033)	8.1 (0.5)	222.90 (46.3&)	7.62 (3.45)
Cameron	5.46 (3.87)	0.66 (1.36)	0.88 (0.77)	0.3111 (1.1647)	0.0071 (0.212)	0.0015 (0.046)	0.241 (0.043)	0.218 (0.028)	8.0 (0.3)	225.10 (40.16)	7.47 (4.09)
Sturgeon	11.70 (9.56)	1.29 (2.22)	1.76 (1.84)	4.6122 (13.1268)	0.0096 (0.353)	0.0017 (0.0635)	0.445 (0.202)	0.339 (0.171)	8.0 (0.3)	365.91 (168.89)	6.95 (3.12)
Pigeon	11.97 (6.59)	1.97 (3.74)	2.66 (2.38)	0.7264 (3.8916)	0.0015 (0.057)	0.0007 (0.0261)	0.358 (0.161)	0.315 (0.163)	7.9 (0.5)	266.58 (159.06)	6.57 (4.08)

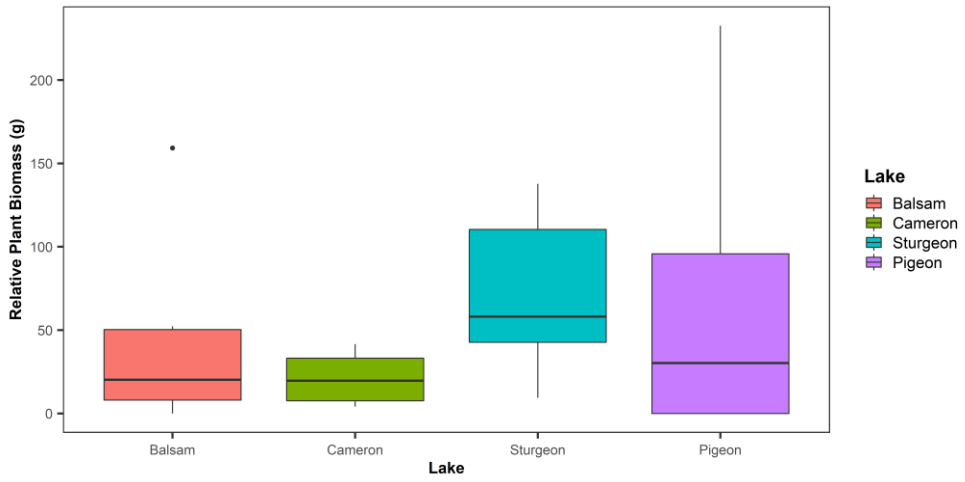

Figure 5.9 Boxplot of plant biomass by lake. ANOVA found no significant differences between lakes (n = 29).

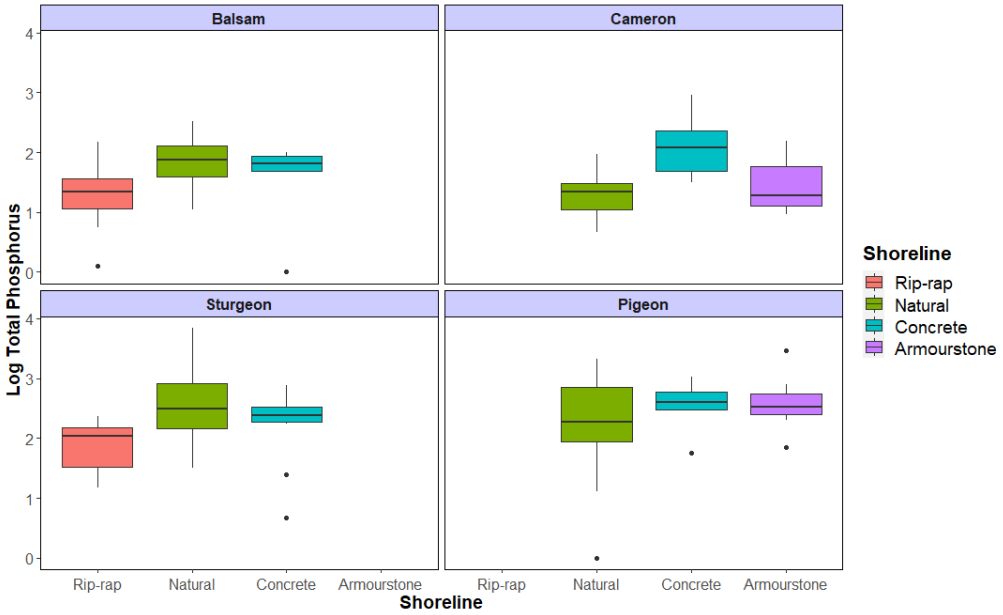

Figure 5.10 Boxplots of log-transformed total phosphorus by shoreline type for each lake (Balsam, Cameron, Sturgeon, and Pigeon). Two-way ANOVA found a significant effect of shoreline type on total phosphorus ($F(5,132) = 3.83$, $p < 0.05$). Total phosphorus was significantly lower at rip-rap shorelines compared to armour stone and natural shorelines ($p < 0.05$).

Table 5.6 Mean of community metrics for nearshore phytoplankton (n = 144), zooplankton (n = 144), and macroinvertebrates (n = 115) across the study lakes.

	Phytoplankton			**Zooplankton**			**Macroinvertebrates**		
	Biovolume (µg/L)	Simpsons Diversity	Functional Diversity	Abundance (indiv./L)	Simpsons Diversity	Functional Diversity	Abundance (indiv./m^2)	Simpsons Diversity	Functional Diversity
Balsam	644.5	0.47	0.057	175	0.61	0.25	122	0.48	0.30
Cameron	560.91	0.58	0.051	323	0.49	0.20	108	0.50	0.28
Sturgeon	1608.3	0.55	0.101	822	0.58	0.29	231	0.57	0.32
Pigeon	1810.4	0.49	0.075	732	0.50	0.35	215	0.57	0.34

The phytoplankton communities had similar abundances and diversity across the study lakes. Although there were no statistically significant differences, Pigeon Lake had higher phytoplankton abundance than the other lakes (Table 5.6). Phytoplankton abundance and richness were regressed with environmental variables to determine potential drivers of phytoplankton richness in these lakes (Table 5.7). Only total dissolved phosphorus had a significant, positive relationship with phytoplankton abundance ($p < 0.05$). To better examine the relationship between community composition and water quality parameters a redundancy analysis (RDA) was conducted with only significant environmental drivers included in the final biplot (Figure 5.11). According to multivariate analysis of variance (MANOVA) dissolved oxygen, turbidity, and depth of the sampling site had significant relationships with the phytoplankton community ($p < 0.05$). Examining species scores in the RDA, Cyanophyta scored low on axis 1, indicating higher abundance associated with increased dissolved oxygen, and Cryptophyta scored high on axis 1, indicating association with increased site depth. Other groups had low scores and little correlation with water quality parameters.

Table 5.7. Multiple linear regression (MLR) models of phytoplankton abundance and richness, bold p-values indicate statistically significant predictors. All water quality variables were log-transformed prior to analysis (n = 143).

	Abundance (R^2 = 0.15)		**Richness (R^2 = 0.04)**	
	Estimate	p-value	Estimate	p-value
Total phosphorus	93.46	0.072	-0.07	0.363
Total dissolved phosphorus	132.63	**0.0018**	-0.11	0.077
Total nitrogen	245.39	0.400	-0.173	0.694
Turbidity	-24621.21	0.188	38.49	0.174
Temperature	243.84	0.396	0.82	0.062
Conductivity	74.90	0.229	0.11	0.255
pH	-652.37	0.323	-0.30	0.760

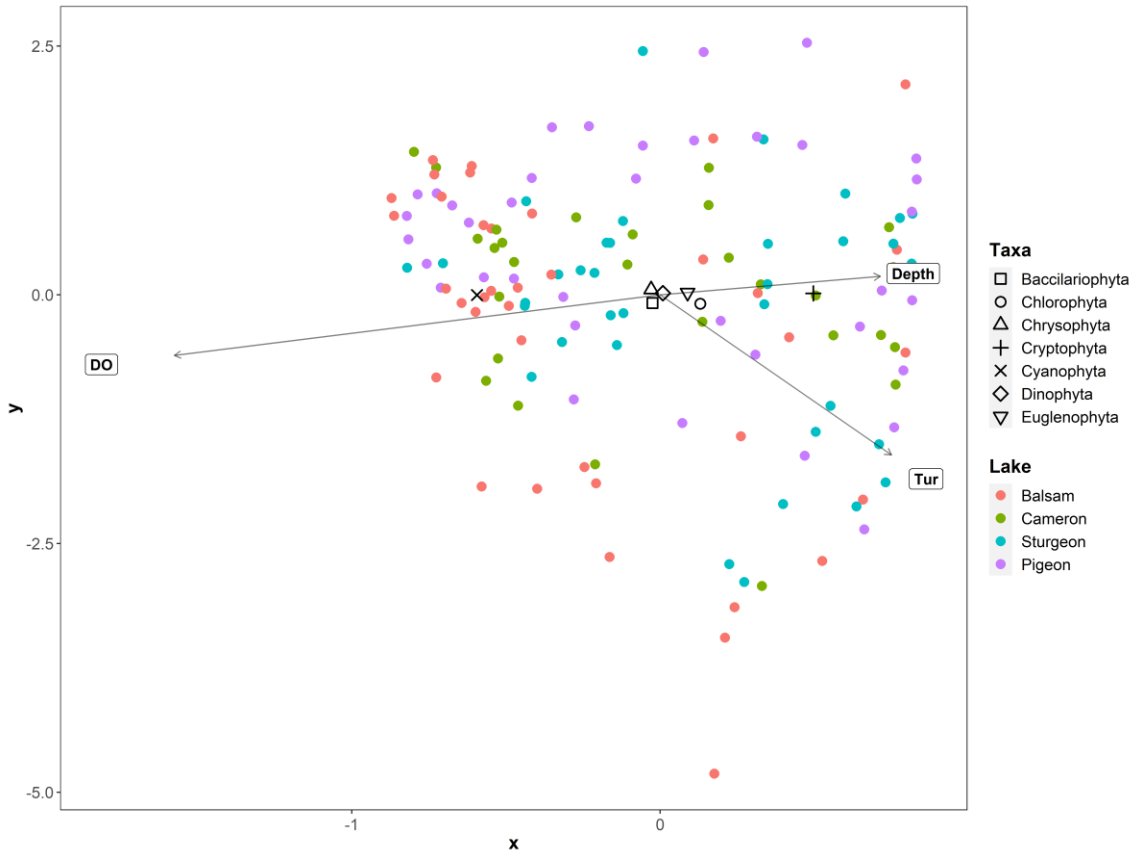

Figure 5.11 Biplot from redundancy analysis of the phytoplankton communities in Balsam, Cameron, Sturgeon, and Pigeon Lakes. Only significant environmental drivers are included: dissolved oxygen (DO), depth, and turbidity (Tur), arrow length corresponds with the strength of influence on the axes. The first constrained axis explains 9.0% of variance and the second constrained axis explains 0.3% of variance.

There were no significant differences in zooplankton abundance or taxonomic diversity across the study lakes (Table 5.6). However, Pigeon Lake had significantly higher functional diversity than Cameron Lake ($p < 0.05$). Zooplankton abundance had a significant, positive relationship with total phosphorus, total nitrogen, and Chla (Table 5.8). The MANOVA indicated that total phosphorus, chlorophyll a, and sample site depth were the significant water quality parameters for zooplankton community composition, and were included in the RDA biplot (Figure 5.12). Examining the RDA species scores, *Bosmina* scored the highest on axis 1, indicating some association with TP, while

Keratella scored low on the same axis, indicating some association with increased site depth.

Table 5.8. Multiple linear regression (MLR) models of zooplankton abundance and richness, bold p-values indicate statistically significant predictors. All water quality variables were log-transformed prior to analysis (n = 131).

	Abundance (R^2 = 0.45)		**Richness (R^2 = -0.02)**	
	Estimate	p-value	Estimate	p-value
Total Phosphorus	18.40	**<0.001**	-0.02	0.702
Total dissolved phosphorus	1.91	0.585	-0.01	0.782
Total nitrogen	93.87	**< 0.001**	-0.25	0.265
Chla	16.87	**0.002**	-0.02	0.646
Turbidity	-1368.16	0.379	7.10	0.625
Temperature	-50.13	0.055	-0.10	0.687
Conductivity	5.36	0.306	-0.01	0.784
pH	83.56	0.157	-0.19	0.735

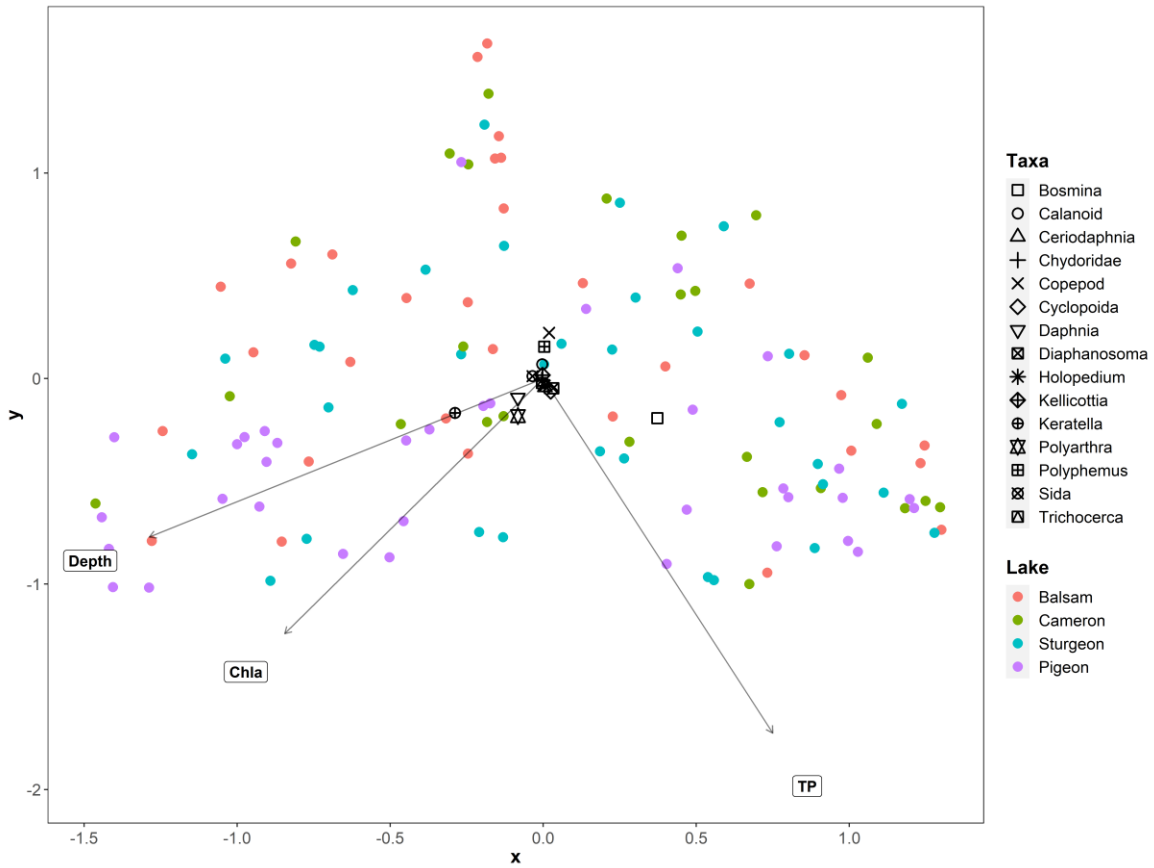

Figure 5.12 Biplot from redundancy analysis of the zooplankton communities in Balsam, Cameron, Sturgeon, and Pigeon Lakes. Only significant environmental drivers are included: total phosphorus (TP), depth, and chlorophyll-a (Chla), arrow length corresponds with the strength of influence on the axes. The first constrained axis explains 27.5% of variance and the second constrained axis explains 2.3% of variance.

Macroinvertebrate samples had significantly higher abundance in Sturgeon and Pigeon compared to Balsam and Cameron, and functional diversity was significantly higher in Pigeon compared to Cameron ($p < 0.05$, Table 5.5). Hilsenhoff's Family Biotic Index (FBI) was calculated for each community to examine the tolerance of each community. Pigeon had significantly higher FBI than Balsam, indicating a more tolerant macroinvertebrate community ($p < 0.05$). EPT index was also calculated for each community and Pigeon had a significantly higher score than both Balsam and Sturgeon ($p < 0.05$). Macroinvertebrate abundance had positive, significant relationships with total

phosphorus and Chla and a negative, significant relationship with conductivity ($p < 0.05$, Table 5.9). The macroinvertebrate MANOVA indicated the significant water quality drivers to be included in the RDA biplot: pH, conductivity, water temperature, chlorophyll a, and total phosphorus (Figure 5.13). Mollusca, which was primarily composed of *Dreissena polymorpha* (zebra mussels), scored high on RDA axis 1, while Amphipoda had the lowest score on RDA axis 1.

Table 5.9. Multiple linear regression (MLR) models of macroinvertebrate abundance and richness, bold p-values indicate statistically significant predictors. All water quality variables were log-transformed prior to analysis (n = 115).

	Abundance (R^2 = 0.20)		**Richness (R^2 = 0.001)**	
	Estimate	p-value	Estimate	p-value
Total Phosphorus	52.37	**0.023**	0.05	0.458
Total dissolved phosphorus	13.70	0.401	-0.003	0.945
Total nitrogen	-114.09	0.343	0.47	0.183
Chla	65.68	**0.002**	-0.05	0.424
Turbidity	-10541.65	0.113	13.80	0.476
Temperature	-35.30	0.752	0.29	0.369
Conductivity	-60.79	**0.003**	0.03	0.633
pH	152.74	0.509	-0.73	0.278

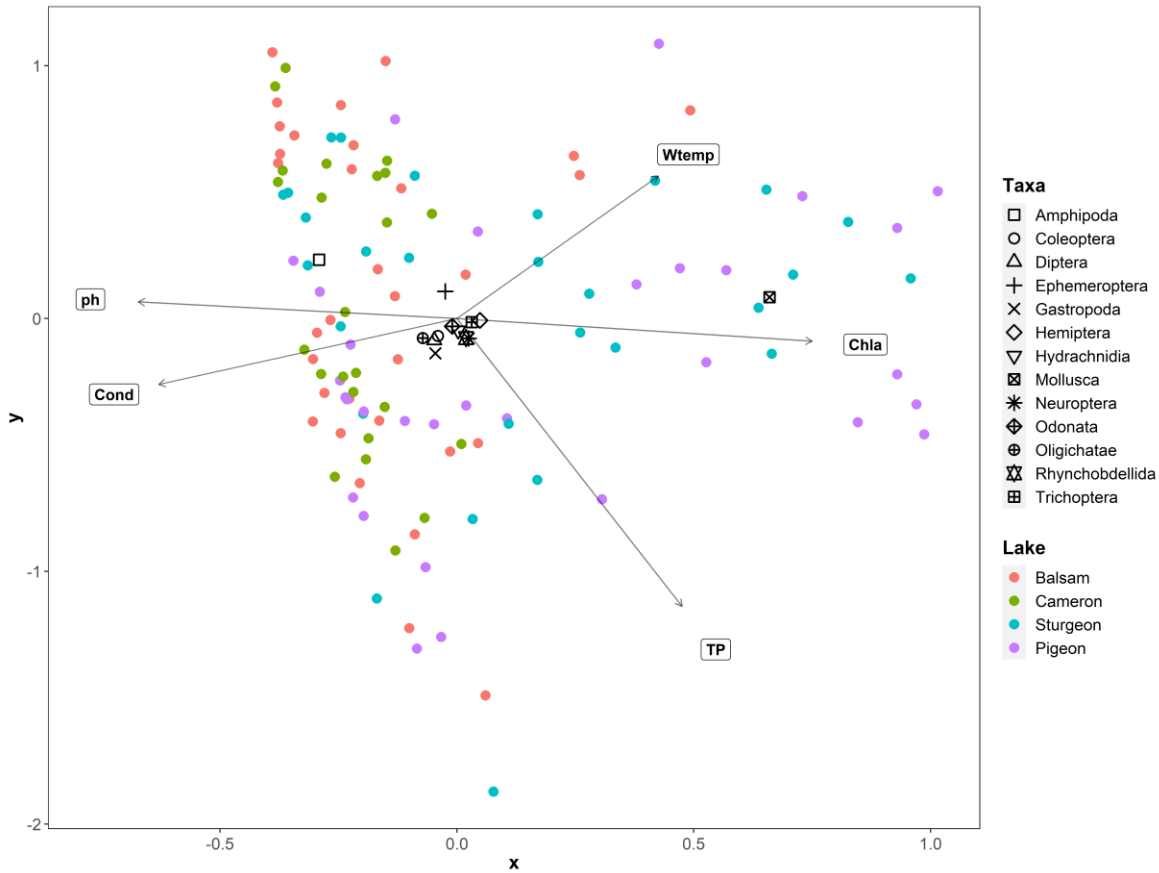

Figure 5.13 Biplot from redundancy analysis of the macroinvertebrate communities in Balsam, Cameron, Sturgeon, and Pigeon Lakes. Only significant environmental drivers are included: total phosphorus (TP), chlorophyll-a (Chla), water temperature (Wtemp), pH, and conductivity (Cond) arrow length corresponds with the strength of influence on the axes. The first constrained axis explains 9.9% of variance and the second constrained axis explains 2.3% of variance.

Path analysis was conducted using *Piecewise SEM* in R and visualized as a concept map (Figure 5.14). Statistically significant relationships ($p < 0.05$) are shown with green and red arrows (positive and negative relationships), and the width of the arrow is scaled to the strength of the relationship. Land use (agricultural and developed) had a positive, significant impact on total nitrogen, while shoreline alteration had no direct, significant impacts on water quality or the biological communities. Total suspended solids (TSS) had a positive significant relationship with periphyton. Total

phosphorus had positive, significant relationships with phytoplankton density and diversity, zooplankton density, and macroinvertebrate density and diversity.

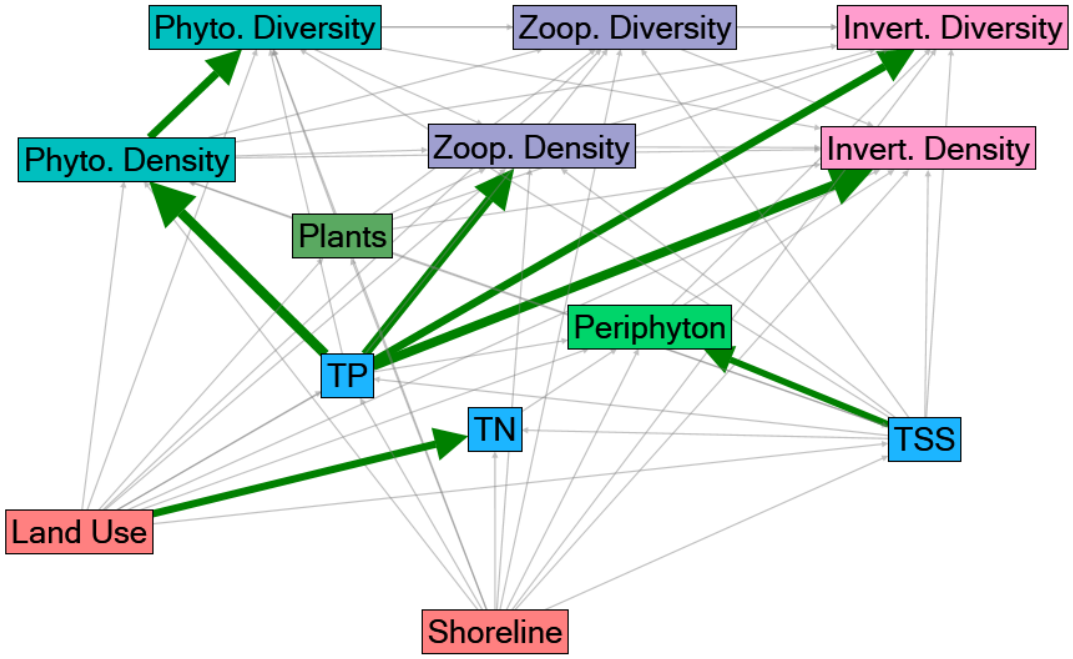

Figure 5.14 . Final concept map of piecewise SEM (Fisher's C = 44.12, p = 0.302, df = 40), for phytoplankton, zooplankton, and macroinvertebrate density and diversity. Green arrows show significant positive effects, gray arrows show nonsignificant effects ($p < 0.05$; Appendix B). Arrow width was scaled by the value of the path coefficient (Appendix B). TP = total phosphorus, TN = total nitrogen, TSS = total suspended solids. Phyto. = phytoplankton, Zoop. = zooplankton, Invert. = macroinvertebrate.

5.4 Discussion

5.4.1 Water quality and land use patterns

One clear pattern across the study lakes was the apparent bifurcation of lakes according to water quality profile, with Balsam and Cameron lakes grouping together and Sturgeon and Pigeon lakes grouping together (PERMANOVA, $p < 0.05$). Results from the principal component analysis indicate that more sites in Balsam and Cameron Lakes were associated with natural land cover, while Sturgeon and Pigeon had more sites associated with development and agriculture. This finding indicates the importance of

land use for driving this binary distinction of water quality profiles. The confirmed oligotrophic conditions in Balsam and Cameron lakes' was expected based on water quality data reported from 2011-2013 monitoring (Kawartha Conservation, 2015a). Higher nutrient levels and greater variability within Sturgeon and Pigeon lakes' was also expected based on the most recent monitoring data (Kawartha Conservation, 2014b, 2016). As pointed out in previous reports, the southern portions of Sturgeon and Pigeon have tributaries contributing high levels of nutrients. Scugog River flows from eutrophic Lake Scugog, through the town of Lindsay, and into the shallow southern arm of Sturgeon. All these factors are likely contributing to increased nutrients and primary productivity in this portion of the lake. Previous monitoring indicates phosphorus levels in the Scugog River are eutrophic upstream of the developed area of Lindsay and increased downstream of the town (Kawartha Conservation, 2014b). Similarly, Pigeon River, Reforestation Creek, and Potash Creek feed the shallow, southern portion of Pigeon Lake, with Pigeon River flowing through the town of Omemee. Although combined these tributaries only contribute about 5% of annual inputs to the lake, they have a large impact on water quality in the southern portion of the lake. Notably, Reforestation Creek has a drainage area of just 15.3 km^2 however, 52% of it is agricultural land use resulting in average phosphorus levels of 116 µg/L (Kawartha Conservation, 2016). Evidently, the southern tributaries in Sturgeon and Pigeon have a substantial influence on water quality in the southern region of the lakes, differentiating them from Balsam and Cameron.

There was a significant relationship between storm events and total suspended solids and fecal coliforms in the nearshore sampling sites. Although in this study *E. coli*

was significantly correlated with fecal coliforms and in previous studies a correlation with turbidity has been found (Davies-Colley et al., 2008; Till et al., 2008), *E. coli* did not have a significant relationship with storm events. I had expected to detect this relationship since *E. coli* has been shown to be influenced by weather patterns, especially storm events (McKergow & Davies-Colley, 2010). The lack of relationship between storm events and *E. coli* levels may be because of the consistently low levels of *E. coli* detected throughout the study, and it is possible that with more frequent monitoring a relationship between land use, storm events, and *E. coli* would have been detected.

By comparing watershed land use (agriculture and development) with water quality variables I found that only phosphorus was significantly correlated with land use, and only for Sturgeon and Pigeon lakes. The lack of relationship between phosphorus and land use in Cameron and Balsam lakes may be due to the sporadic patterns of land use in the watershed, as well as their relatively low total coverage. In Sturgeon and Pigeon, urban and agricultural land-use is more concentrated in the southern portion of their watersheds which may strengthen the land use – nutrient relationship (Uuemaa et al., 2005). The relationship between latitude and phosphorus in Sturgeon and Pigeon also supports the conclusion that phosphorus loadings are higher in the southern portions of the watersheds for these lakes.

The impact of land use on nearshore water quality was investigated at multiple scales from a 100 m buffer to a sub-watershed scale. Most of the significant relationships were detected at the sub-watershed scale, confirming previous findings that the strongest land use – water quality relationships exist at large spatial scales (Lei et al., 2021; Pratt & Chang, 2012). Road density was a significant predictor for nitrates and nitrites at the sub-

watershed scale, which might indicate an impact of the rural communities around the lakes. As has previously been found, hard surfaces, in this case, roads, significantly increase nitrate and nitrite levels (Zampella et al., 2007). Although *E. coli* was not associated with land use in the principal component analysis, it was predicted by development at both the sub-watershed and 1000 m scales. Although I cannot confirm the exact source of *E. coli*, there are two likely main sources in the watersheds of the study lakes: manicured lawns in developed areas are prime *Branta canadensis* (Canadian Goose) habitat and septic systems, which are the primary sewage treatment method around these lakes (Guerena et al., 2014; Iverson et al., 2017). At the 500 m scale, the only significant relationship detected was between turbidity and agricultural and developed land use. Although we know that land use increases turbidity levels, previous work has found the strongest relationship occurs at a large scale (Vera Mercado & Engel, 2021). There were no significant relationships detected between water quality variables and land use at the 100 m buffer scale. Other studies have found significant relationships between water quality and small buffer scale land use (Fraterrigo & Downing, 2008; Soranno et al., 2015), however, Fraterrigo and Downing (2008) found that a watershed's ability to transfer materials, transport capacity, played an important role in determining the most effective land use scale for predicting nutrient levels. Factors such as hydrology, topography, soils and geology can all affect a watersheds transport capacity (D'Arcy & Carignan, 1997; Kirchner & Dillon, 1975). These findings highlight the importance of studying multiple spatial scales as land use relationships vary regionally (Read et al., 2015; Soranno et al., 2015).

5.4.2 Biotic communities

Water quality trends from the biological monitoring component of the study closely match those from samples collected by community scientists: oligotrophic conditions in Balsam and Cameron lakes and mesotrophic conditions in Sturgeon and Pigeon lakes. The nearshore biomonitoring reflects these patterns, starting with primary productivity. The higher plant biomass at Sturgeon and Pigeon sites is likely due to the increased nutrient availability promoting primary productivity. However, this pattern was only seen in the phytoplankton community in Pigeon, which had the highest phytoplankton abundance. It is possible that other factors, like lake morphometry and predation, are dictating phytoplankton levels (Carpenter et al., 2001; Matsuzaki et al., 2018).

The role of shoreline modifications was also investigated to examine its influence on nearshore water quality and biotic communities. It was expected that natural shorelines would play a role in filtering nutrients from surface runoff (Dosskey et al., 2010), however, rip-rap shorelines had significantly lower phosphorus levels than natural shorelines. Previous studies have found that rip-rap shorelines were generally less degraded than other altered shoreline types (Jennings et al., 1999; Patrick et al., 2014), and our results compliment these findings. It is possible that with more natural shorelines, particularly with higher densities of shoreline plants, there would be higher rates of decomposition, which can serve to release nutrients into the nearshore zone. Surprisingly, there were no significant relationships between shoreline type and any of the biotic communities' abundance or diversity. Other work has found that altered shorelines negatively impacted plant abundance (Patrick et al., 2014) and macroinvertebrate

diversity (Brauns, Garcia, Walz, et al., 2007). An important consideration for the shoreline analysis is the heterogeneity of the shorelines around these lakes. Although the sampling site that was sampled had a specific shoreline type, in most cases there were neighbours with different shoreline types that would influence the overall habitat quality and function of the nearshore zone across sites. Additionally, much of the development along these shorelines has existed for > 20 years and the biotic communities have likely already adapted to current shoreline structure (Kawartha Conservation, 2014a, 2015b).

Phytoplankton had no significant trends in diversity or abundance across the study lakes, despite the distinct nutrient trends that usually impact phytoplankton (Dubey & Dutta, 2020; Leibold, 1999). However, multiple linear regression indicated that total dissolved phosphorus was the only significant predictor for phytoplankton abundance ($p < 0.05$). In addition, there was a positive, significant relationship between log phosphorus and log chlorophyll a in Pigeon Lake. Clearly, phosphorus is an important nutrient for phytoplankton in these lakes, particularly in the nutrient-rich Pigeon Lake. There are many factors that can affect the TP – Chla relationship, which may have caused the lack of significant relationship in Balsam, Cameron, and Sturgeon. Chlorophyll content of algal cells can vary widely, and lake morphometry, and watershed characteristics can impact the TP – Chla relationship (Quinlan et al., 2021; Yuan & Jones, 2020). Additionally, the invasive *Dreissena polymorpha* has been shown to alter the TP – Chla relationship (Higgins et al., 2011) however, I identified *D. polymorpha* communities in all of the study lakes. Although there was not a significant TP – Chla relationship for Balsam, Cameron, and Sturgeon, they still had positive slopes (Figure 5.4).

The phytoplankton multivariate analysis of variance indicated dissolved oxygen, depth, and turbidity were significantly driving the phytoplankton community. The significance of dissolved oxygen should not be interpreted as driving community structure, as phytoplankton are photosynthetic organisms, and the apparent relationship may be due to other factors influencing both oxygen levels and the phytoplankton community. One important consideration is the diel oxygen cycle. Since our sites were sampled in the same order each sampling day, it is possible that the timing of our site visits is confounding the relationship between oxygen and phytoplankton community composition. Turbidity and site depth infer the importance of light availability for phytoplankton. The negative relationship between cyanobacteria and depth was unexpected, as they can outcompete other taxa in deeper waters due to their light-harvesting accessory pigments and their ability to regulate buoyancy in the water column (Walsby et al., 1995).

The zooplankton community in Pigeon Lake had the highest abundance and functional diversity, matching the higher phytoplankton levels in Pigeon. Multiple linear regression showed that total phosphorus, total nitrogen, and chlorophyll a were positive, significant drivers of zooplankton abundance. The RDA indicated that *Bosmina* were positively associated with total phosphorus. It appears that the high nutrient sites are dominated by tolerant taxa, such as *Bosmina*, which were highly abundant in this system. Specifically, in Pigeon Lake, *Bosmina* and *Polyphemus* were abundant, and they are known to be tolerant of a large range of environmental conditions (Swadling et al., 2000). Despite differences in water quality, the lack of clear groupings of zooplankton communities between lakes is not surprising given the interconnectedness of these lakes

and the continuous dispersal of zooplankton (Dodson et al., 2005). In addition, environmental gradients (e.g., temperature, pH, etc.) are not broad enough to shape distinct zooplankton communities across these lakes.

The nearshore macroinvertebrate communities had higher abundance and diversity in the higher nutrient lakes (Pigeon and Sturgeon). The positive relationships between total phosphorus and chlorophyll a and macroinvertebrate abundance matches previous findings that macroinvertebrate abundance increases with nutrient enrichment (Blumenshine et al., 1997). Some previous studies have found nutrient enrichment to decrease macroinvertebrate diversity by decreasing habitat heterogeneity (Donohue et al., 2009) however, I did not find any significant relationships between nutrients and macroinvertebrate richness. Although nutrient enrichment resulted in more abundant communities, the taxa present tended to be more tolerant of the poor conditions, indicated by the higher FBI scores in Sturgeon and Pigeon. Alternatively, other studies found a weak impact of trophic status and habitat structure was more important in determining macroinvertebrate communities (Brauns, Garcia, Pusch, et al., 2007). I did not find any significant relationships between macroinvertebrate community metrics and macrophyte biomass or shoreline hardening. This was unexpected as I found an effect of macrophyte biomass on the macroinvertebrate community in Chapter 3, and previous work has demonstrated the importance of shoreline structure in providing food resources and habitat (Brauns, Garcia, Walz, et al., 2007; Walker et al., 2013)

Overall, the structural equation model (SEM) did a good job of showing significant connections between human activity, water quality, and phytoplankton, zooplankton and macroinvertebrate communities. Interestingly, there was only one direct,

but significant impact of watershed land use, which was a positive relationship with total nitrogen. This relationship is not surprising considering that the dominant land use in these watersheds is agricultural. Previous work has found agricultural land use, especially croplands to be a significant source of nitrogen (Soranno et al., 2015; Zampella et al., 2007). However, I expected that land use would significantly drive total phosphorus, as was found in previous studies examining land use – nutrient relationships (Soranno et al., 2015; Tong & Chen, 2002).

Even though the major source(s) of phosphorus was not revealed in this study, total phosphorus was an explanatory variable of phytoplankton, zooplankton, and macroinvertebrate abundance. A positive relationship with phytoplankton density was expected, as phosphorus is a limiting resource for phytoplankton in freshwater environments (Dubey & Dutta, 2020; Leibold, 1999). The relationships between phosphorus and zooplankton and macroinvertebrate density is not as clear. It was expected phosphorus might have indirect impacts on these communities by increasing the primary productivity of phytoplankton and periphyton. An increase in phytoplankton density was detected, however, there was no significant relationship between phytoplankton and zooplankton. This was unexpected as they are directly connected in the aquatic food web, and previous studies have found relationships between phytoplankton and zooplankton abundance (Leibold, 1999; Stamenković et al., 2021).

There was no significant relationship between periphyton and phosphorus or macroinvertebrates. Previous studies have found positive effects of nutrients on macroinvertebrates mediated through increased periphyton growth (Miltner et al., 1998; Rader & Richardson, 1992). Although periphyton was not significantly influenced by

nutrient levels, total suspended solids had a positive, significant effect on periphyton. It has been found that TSS stimulated periphyton growth at intermediate levels (100 mg/L) (Birkett et al., 2007), and TSS levels were much below this threshold in the lakes that I studied.

Also, most interestingly, macrophytes did not have any significant, direct effects on any of the biological communities studied. Previous work, including within this thesis (Chapter 3) has shown the important role of macrophytes in the littoral zone, especially for zooplankton and macroinvertebrate communities (Porst et al., 2019; Sagrario et al., 2009; Schriver et al., 1995). In contrast to environmental variables, there were few significant relationships revealed between the biotic communities in the SEM. Only phytoplankton density had a positive relationship with phytoplankton diversity.

The Kawartha Lakes population is projected to continue growing and this study provides current nearshore water quality data for comparison, and insights on how to mitigate impacts of development (City of Kawartha Lakes, 2019). By examining the nearshore water quality in these lakes, I identified clear water quality profiles that grouped Balsam and Cameron lakes and Sturgeon and Pigeon lakes. It appears that this division is driven at least partially by land use, with a strong influence from the southern tributaries in Sturgeon and Pigeon. Land-use impacts were detected for different water quality parameters from buffer scales (500 m and 1000 m) to the sub-watershed scale, providing valuable information on sources of contaminants at each scale and demonstrating the importance of studying land use at multiple scales. Nearshore nutrient levels in these lakes were significant drivers of abundance for all three aquatic communities studied. Although I did not detect an impact of shoreline macrophytes or

physical structure on the biotic communities in these lakes, future work should further investigate impacts of shoreline alteration on the biotic community. A balanced study design and more physical factors measured (shade, substrate etc.) could provide insight on the role of shoreline structure in these lakes with high shoreline development.

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Chapter 6. **General Discussion and Conclusions**

My thesis aimed to determine the effect of different human land-uses on nearshore water quality, and to assess the impact of shoreline alteration on the lower aquatic food web in the nearshore zone in the Kawartha Lakes. To achieve these goals I set-out to address four specific objectives: 1) Determine nearshore water quality trends and their relationship to land use at multiple spatial scales in Lake Scugog using a community science approach (Chapter 2); 2) Asses nearshore phytoplankton, zooplankton, and macroinvertebrate communities and their relationship to nearshore habitat (water quality and macrophytes) in Lake Scugog (Chapter 3); 3) Examine the spatial patterns of nutrients across 16 Kawartha Lakes and determine if expected change in human activity during the COVID-19 pandemic influenced nearshore nutrient levels in four lakes with high shoreline development (Chapter 4); and 4) Determine the influence of land use and shoreline structure on nearshore water quality and biological communities in four central Kawartha Lakes (Chapter 5). Addressing these objectives in Chapters 2 to 5 has enabled me to show that at scales as small as 500 m land use, particularly developed land use, negatively impacted nearshore water quality in the Kawartha Lakes. In Lake Scugog, macrophytes had a strong effect on all three biotic communities studied (phytoplankton, zooplankton, and macroinvertebrates). The COVID-19 pandemic did not have a significant effect on nearshore water quality in the Kawartha Lakes. Finally, nearshore nutrient levels played a significant role in determining phytoplankton, zooplankton, and macroinvertebrate abundance and community composition for zooplankton and macroinvertebrates in four central Kawartha Lakes.

6.1 Building a community science water quality monitoring program

Lakes face multiple stressors from climate change to land development. Consequently, effective water monitoring programs require adequate spatial and seasonal coverage to capture environmental variation over time. Community science is a method that has increased in popularity for environmental monitoring because it provides useful information for management programs and academic researchers (Latimore & Steen, 2014). Previous work has shown that when community science is properly implemented, it can produce data comparable to professionally collected samples (Canfield Jr et al., 2002; Scott & Frost, 2017). Additionally, it is becoming clear that community scientist collected samples are a major contribution to broad-scale ecological databases and are important for examining long-term water quality trends (Lottig et al., 2014; Poisson et al., 2020).

My thesis research implemented a co-production community science research model to monitor water quality in the Kawartha Lakes for five years, with a total of 16 lakes sampled over this period. I found spatial patterns of within and between lake variation, and showed some clear patterns in nutrient levels in this system. In Lake Scugog, I found spatial patterns of elevated nutrients, chloride, and *E. coli* levels in the western basin of the lake compared to the eastern basin (Table 2.1). In the four central Kawartha Lakes (Balsam, Cameron, Sturgeon, and Pigeon Lakes), I found the expected pattern of lower nutrient status (oligotrophic) for the upper lakes (Balsam and Cameron) and comparatively higher nutrient status (mesotrophic) in the lower, down-gradient lakes (Sturgeon and Pigeon) (Figure 5.2). I also found some within-lake trends for Sturgeon and Pigeon Lakes, where the southern sampling sites in both lakes had higher nutrient

concentrations compared to sites in the northern portion of the lakes (Figure 5.7). Since these patterns reflect notable shifts in land-cover and land-use in the entire catchment, it indicates that local tributaries are an important driver of nearshore water quality, including the Scugog River, which drains Lake Scugog into Sturgeon Lake. This impact of Lake Scugog outflow was further substantiated in the 2020 water monitoring findings, with an increase in nutrient levels in lakes downstream of the Scugog River (Figure 4.3). By implementing a large-scale community science monitoring program in 2019, spatial patterns in water quality across the Kawartha Lakes have been documented for the first time, and offer important information to local lake managers in their development of lake management plans. The spatial patterns of water quality that I identified can help lake managers pinpoint areas that require efforts to improve water quality conditions. By finding relationships between water quality and land use and shoreline structure I provided information that can be used to mitigate impacts of human activity on the lake. Finally, by sharing what factors impact the biotic communities' abundance and diversity in these lakes' lake managers can make plans to protect these communities.

Previous work has identified the role of community science for creating spatially and temporally explicit data sets that can be used to answer a range of research questions, and these have been especially valuable in the United States, where over half of the observations for common water quality parameters came from community scientists (Poisson et al., 2020). Alternatively, projects that use the co-production model, can provide benefits to regional organizations, local volunteers and academic researchers (Latimore & Steen, 2014). My work provided another example of the role community science can play in producing a large, high quality data set using a unique co-production

model as described in Ch. 2. It also demonstrated the adaptability and rapid-response capability of community science during the pandemic, which allowed me to collect data from a broad range of lakes in a year (2020) when most environmental monitoring was paused. At a time of pronounced environmental change (e.g., climate heating, biodiversity loss, etc.) it is exceedingly important to have consistent and reliable environmental data, and community science can help achieve this goal. In lakes with increasing land development along private shorelines, community science is the most viable and economical way for lake researchers and managers to access the nearshore zone for long-term monitoring.

6.2 Determining the influence of land use on nearshore water quality

Lakes are influenced by different kinds of watershed land use, such as agriculture, mining, forestry, and urbanization. However, it is the spatial extent and relative amounts of land uses across watersheds that determine the magnitude of water quality degradation (Kim et al., 2016; Pratt & Chang, 2012). It is also apparent that the impacts of land use vary by spatial scale. Many studies have examined the effect of land use at multiple spatial scales in rivers and streams and found impacts on water quality at both large and small scales (Lei et al., 2021; Pratt & Chang, 2012; Vera Mercado & Engel, 2021). My research focused on the nearshore area of lakes which are ecologically important, valued for human enjoyment, and the location where the first receives run-off and tributary discharge sourced from the landscape. I found that land use had significant impacts on nearshore water quality at multiple spatial scales, from 500 m buffers up to the sub-watershed scale. I also found that the relationship between water quality and land use varied not only by the spatial scale, but the lake as well.

In the headwater Lake Scugog, where agriculture is the dominant land-use in the watershed, total phosphorus was significantly influenced by urban development at the watershed scale, while chloride was strongly affected by urban development at the 1000 m buffer scale (Table 2.3). Conversely, in the chain of Kawartha Lakes (Balsam, Cameron, Sturgeon, and Pigeon Lakes), development was a statistically significant explanatory variable for *E. coli* at the sub-watershed and 1000 m buffer scales, whereas agriculture and development were significant explanatory variables of turbidity at the 500 m buffer scale (Table 5.3). Additionally, the combination of agricultural and developed land use significantly affected phosphorus levels in Sturgeon and Pigeon Lakes, but not Balsam and Cameron Lakes (Figure 5.6). Evidently, the different land use patterns at each scale impact nearshore water quality, at smaller scales an increase in road density from shoreline neighbourhoods combined with hard surfaces such as driveways and lawns result in direct addition of chloride and suspended solids that contribute to turbidity. Alternatively, the impacts of development at the sub-watershed scale on phosphorus and *E. coli* may be due to an overall increase in hard surfaces leading to increased run-off from multiple different sources, including the extensive agriculture land use.

These findings suggest that nearshore water quality is not just influenced by local activities and shoreline conditions, but rather different water quality parameters are driven by different scales of land-use activity. Previous work has shown that land use – water quality relationships are region specific (Read et al., 2015), but I found that even within the same region, the nearshore zone of lakes had different relationships with different scales of catchment land use. Other work has highlighted factors such as soil

transport capacity and hydrologic connectivity that have important implications for land use – water quality relationships (Fraterrigo & Downing, 2008; Soranno et al., 2015). Although we could not examine soil transport capacity with this study, we did find patterns that indicated the role of hydrologic connectivity in determining water quality in these lakes.

My findings, along with previous studies, make it clear that improved best practices for reducing runoff from agricultural and urban areas need to be implemented to protect our lakes. More importantly, my results also showed that urban development had a disproportionate affect on water quality compared to agriculture, where only a small percentage of urban development resulted in significant increases in phosphorus and chloride, for example. Actions such as naturalizing shorelines, septic system maintenance, and reducing road salt application can positively impact nearshore water quality. The lakes examined in this project sit on the outskirts of the highly developed Greater Toronto Area, and as urban sprawl continues, it is increasingly important to understand how increasing human presence and activity in the Kawartha Lakes is affecting lake health.

6.3 Water quality trends during the COVID-19 pandemic

The COVID-19 global pandemic provided the setting for a global experiment that has never happened before. In environmental research, there were some positive findings of improved air quality and water clarity in highly developed regions (Bherwani et al., 2020; Yunus et al., 2020). At the same time, there was reduced environmental monitoring due to lockdown orders closing laboratories, and the suspension of most monitoring programs. I adapted an existing community science project to expand to 16 lakes in 2020,

four of which (Balsam, Cameron, Sturgeon, and Pigeon) I had pre-pandemic (2019) and pandemic (2020 and 2021) nutrient data to compare. Although this provided a serendipitous opportunity to compare pandemic data with pre-pandemic data, there was no statistically significant difference between years with respect to nutrients, despite the notable change in human presence and activity. Although I did detect some seasonal trends, there was no clear effect of the pandemic on nearshore nutrient levels (Figure 4.10). It is difficult to ascertain why there was no strong pandemic signal or effect, but it is likely that there were several confounding factors that influenced nutrient sources to the nearshore zone. It is also possible that my study lacked enough statistical power to detect an effect.

Since other studies have mainly found impacts of the pandemic on industrial activity and transportation (Garg et al., 2020; Yunus et al., 2020), it should not be too surprising that the largely rural Kawartha Lakes, with no major urban areas or industry, did not respond strongly to the pandemic. Farming and tourism are the main economic activity in the Kawartha's (City of Kawartha Lakes, 2021). Even with international tourism stopping in the first year of the pandemic, there was persistent domestic travel (Giunta, 2020). Additionally, farming was considered essential throughout the pandemic and continued throughout. The net impact of these consistent pre-pandemic and pandemic activities likely resulted in minimal change to human activity across the study period.

Although I did not detect an effect of the COVID-19 pandemic on nearshore water quality, there is still much we can learn from the 'anthropopause'. Changing activity in highly industrialized areas had an immediate and strong impact on environmental conditions (Bherwani et al., 2020; Yunus et al., 2020), in contrast to the

lack of change in rural settings such as the Kawartha's. Although we can still detect land use impacts in this region, they are subdued compared to urban areas. Finally, by adapting my community science sampling approach I found a way to continue collecting water quality data even as most other environmental monitoring was paused.

6.4 Land use and shoreline modifications impact nearshore biotic communities

Human activity in a lake's watershed has a direct impact on water quality and shoreline structure and consequently influences the aquatic food web. It has been shown that water quality influences the nearshore biotic communities, especially primary producers (Lambert & Cattaneo, 2008; Vadeboncoeur et al., 2001). Additionally, macrophyte abundance and shoreline alteration affect habitat availability, and ultimately influence the abundance and diversity of aquatic organisms such as macroinvertebrates (Brauns et al., 2007; Porst et al., 2019). In my thesis research, I demonstrated the role of nearshore water quality and macrophytes in determining the abundance and diversity of phytoplankton, zooplankton, and macroinvertebrates in Lake Scugog. I also found an indirect effect of land use on nearshore community abundance and diversity in the Kawartha Lakes.

I found that macrophytes significantly associated with increased zooplankton abundance and macroinvertebrate diversity in Lake Scugog (Table 3.3). This finding supports the hypothesis that macrophytes have a positive effect on plankton and macroinvertebrate communities by providing habitat and a food source. It also highlights the downside of macrophyte removal by homeowners, which negatively affects the aquatic food web. In Balsam, Cameron, Sturgeon, and Pigeon lakes, I found that water quality was important for driving abundance and diversity in the lower aquatic food web

(Figure 5.14). Land use indirectly impacted the biotic community by increasing nutrient levels in the nearshore zone (Figure 5.3). Alternatively, there was no direct impact of shoreline structure or macrophyte abundance on the biotic communities studied. Overall, these findings show the important role humans play in dictating nearshore chemical and physical conditions that influence the abundance and diversity of organisms in the nearshore aquatic food web.

The finding that both nearshore structure and water chemistry in the Kawartha lakes play important roles in determining nearshore community composition is meaningful for understanding overall lake health. They also point to key features to include in future analyses of these and other lakes with high shoreline development. My findings have particularly relevant implications in this region because of the importance of the lower aquatic food web in supporting an important sport fishery. It is crucial to understand the drivers of the biotic community in the nearshore area that provides essential nursery grounds for fish. The Kawartha Lakes are a hotspot for sportfishing in Ontario, and since numbers of some taxa, such as *Sander vitreus* (walleye), are in decline (Harrow-Lyle & Kirkwood, 2020), it is imperative to improve our understanding of the entire aquatic food web as it relates to sustaining a viable fish community and recreational fishery.

6.5 Study limitations and future directions

By adopting a community science approach for this project there were some expected limitations related to sample collection. As was expected, there were some missed sample collections each year. Although this loss of data required extra work to perform data analysis, it was much lower than other community science projects have

reported (Bos et al., 2019; Scott & Frost, 2017), and overall did not impede my ability to perform statistical analyses. Implementing a community science approach also raises concern about the accuracy of sample collection and field observations. To overcome this potential problem, I examined the shape of my data to find any outliers that may have been due to collection errors. Although the community science method resulted in some limitations, overall this method resulted in a much larger, spatially explicit data set than what could have been collected by myself. Additionally, it was a study approach that had "physical distancing" already built in to it, which allowed me to collect and analyze samples in 2020 while other organizations and laboratories were closed.

Though the community science approach granted exclusive access to private shorelines, which enabled Hester-Dendy deployments and other biological samples to be collected, it also constrained the spatial coverage of my sampling locations. Site selection was dependent on there being a willing community scientist to volunteer their lake-front property for months-long field studies. Overall, I accrued enough sites in each lake, I had unequal sample sizes for many of my comparison groups (lake, plant abundance, shoreline type). Fortunately, I was able to apply statistical methods designed to overcome this limitation.

Future work could benefit by focussing on the role of water flow and discharge in the Kawartha Lakes system. Although it was not within the scope of my thesis research, I think that some of the unexplained variation in my statistical results is due to the influence of up-gradient lake water quality on down-gradient lakes. These lakes have regulated flow and flushing rates are variable between lakes, therefore water discharge should be considered as indicated in previous work (Dillon, 1975; Soranno et al., 2015).

Additionally, I found an impact of shoreline type on phosphorus and chlorophyll-a (Chapter 5) and a strong impact of macrophyte abundance on biotic communities in Lake Scugog (Chapter 3). Future work can expand on this finding by considering additional nearshore habitat features, such as shade, coarse woody debris, and bottom substrate. These variables are all altered by shoreline development and affect nearshore habitat. Coarse woody debris and bottom substrate determine the physical structure of the nearshore zone (Christensen et al., 1996; Creque et al., 2010), and shade alters water temperature (Johnson & Jones, 2000), indirectly affecting abiotic and biotic processes.

6.6 Final thoughts

My research has shown the impact of human activity on the abiotic and biotic components of the nearshore area. These findings provide evidence of the negative effect of land development and shoreline alteration on water quality, but also potential ways to reverse or minimize the effects of human activities on these lakes. I found that even in rural watersheds there was a strong effect of development on nearshore water quality, and indirectly the biotic communities. This finding demonstrates the powerful effect of development, even in small amounts, and highlights the importance of continuous monitoring, to capture incremental changes. By monitoring these changes lake managers and policy-makers can find ways to mitigate the environmental impacts of continued population growth.

Finally, I consider my use of a community science approach to garner a large, spatially explicit data set further validates its utility as a sampling approach for scientific research. By accepting and expanding community science research, we can increase data collection while connecting with local communities that need environmental monitoring

data. In particular, co-production can connect western science with Indigenous knowledge to explore important research questions and increase our shared knowledge. Community science and co-production can play a role in decolonization, especially when it comes to environmental research, and it is a vital next step to improving our understanding of aquatic systems.

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Appendix A.

Research Ethics Board Approval Notice

Approval Notice - REB File #15910
1 message

researchethics@uoit.ca <researchethics@uoit.ca>
To: "Kirkwood Andrea(Primary Investigator)" <andrea.kirkwood@uoit.ca>
Cc: "Smith Erin(Student Lead/Post-Doctoral Lead)" <erin.smith20@uoit.net>, researchethics@uoit.ca

Date:	*May 29, 2020*
To:	*Andrea Kirkwood*
From:	*Paul Yielder, REB Vice-Chair*
File # & Title:	*15910 - Survey of Waterfront Resident Habits and Activities du*
Status:	***APPROVED***
REB Expiry Date:	***May 01, 2021***

Documents Approved:

Consent Form - received May 19, 2020
PI TCPS2 Certificate - received May 19, 2020
Student Lead TCPS2 Certificate - received May 19, 2020
Online Survey Questions - received May 19, 2020
Reminder Email - received May 19, 2020
Email Invitation - received May 19, 2020

Notwithstanding this approval, you are required to obtain/submit, to Ontario Tech Research Ethics Board, any relevant approvals/permissions required, prior to commencement of this

The Ontario Tech Research Ethics Board (REB) has reviewed and approved the research study named above to ensure compliance with the Tri-Council Policy Statement: Ethical Conduct f Tech Research Ethics Policy and Procedures and associated regulations. As the Principal Investigator (PI), you are required to adhere to the research protocol described in the REB applica you are responsible for obtaining any further approvals that might be required to complete your project.

Please note a collegial suggestion to amend data storage procedures to delete all data after 5-7 years, as this timeline is in accordance with standard practice.

Under the TCPS2 2018, the PI is responsible for complying with the continuing research ethics reviews requirements listed below:

Renewal Request Form: All approved projects are subject to an annual renewal process. Projects must be renewed or closed by the expiry date indicated above ("Current Expiry"). Proje automatically suspended by the REB; projects not renewed 60 days post expiry date will be automatically closed by the REB. Once your file has been formally closed, a new submission w

Change Request Form: If the research plan, methods, and/or recruitment methods should change, please submit a change request application to the REB for review and approval prior t

Adverse or Unexpected Events Form: Events must be reported to the REB within 72 hours after the event occurred with an indication of how these events affect (in the view of the Princ continuation of the protocol (i.e. un-anticipated or un-mitigated physical, social or psychological harm to a participant).

Research Project Completion Form: This form must be completed when the research study is concluded.

Always quote your REB file number (**15910**) on future correspondence. We wish you success with your study.

Sincerely,

Dr. Paul Yielder
REB Vice-Chair
paul.yielder@uoit.ca

Emma Markoff
Research Ethics Assistant
researchethics@uoit.ca

Survey of Waterfront Resident Habits and Activities during the COVID-19 Pandemic

Consent Form

Title of Research Study: Survey of Waterfront Resident Habits and Activities during the COVID-19 Pandemic

Name of Principal Investigator (PI): Dr. Andrea Kirkwood

PI's contact number(s)/email(s): Dr. Andrea Kirkwood: andrea.kirkwood@uoit.ca, (905)721-8668 ext.3622

Names(s) of Co-Investigator(s), Faculty Supervisor, Student Lead(s), etc., and contact number(s)/email(s): Erin Smith (Student Lead): erin.smith@uoit.ca, (905)721-8668 ext. 3050

Departmental and institutional affiliation(s): Ontario Tech University, Faculty of Science

You are invited to participate in a research study entitled Survey of Waterfront Resident Habits and Activities during the COVID-19 Pandemic. You are being asked to take part in a research study. Please read the information about the study presented in this form. The form includes details on the study's procedures, risks, and benefits that you should know before you decide if you would like to take part. You should take as much time as you need to make your decision. You should ask the Principal Investigator (PI) or study team to explain anything that you do not understand and make sure that all of your questions have been answered before signing this consent form. Before you make your decision, feel free to talk about this study with anyone you wish including your friends and family. Participation in this study is voluntary.

This study has been reviewed by the University of Ontario Institute of Technology (Ontario Tech University) Research Ethics Board #15910 on May 29, 2020.

Purpose:

The COVID-19 pandemic is changing the way people operate around the world, and is already having impacts on air quality and other environmental conditions

The purpose of this study is to determine if the habits of waterfront residents and cottagers in the Kawartha and Durham regions are having an impact on the health of their lakes.

You have been invited to participate in this survey because you have a property on the waterfront of a study lake and have previously participated in research studies conducted by researchers at Ontario Tech University.

Procedures:

This study is a virtual online survey which takes a single session to complete, at the time and place of your choosing. The survey is expected to take between 10-15 minutes to complete.

Approximately 50 people will be taking part in this study.

The participant is responsible for completing the survey as accurately as possible and submitting the survey upon completion.

The data collected from the survey responses will be analysed for trends and presented in aggregate form for any presentations or manuscripts. The results of this study will benefit research in this field by providing information on how water quality may be impacted in rural areas. Water quality monitoring has been paused at all levels of government during this time and the results of this survey will aid in determining potential impacts of the COVID-19 pandemic on water quality.

Potential Benefits:

You will not directly benefit from participating in this study

Potential Risk or Discomforts:

There are no known or anticipated risks to you by participating in this study.

The researchers will advise you if there is any new information that could alter your decision to participate in the study.

Use and Storage of Data:

Data will be collected through the online survey tool, SurveyMonkey, and stored temporarily through the Canadian Data Centre. Then, data will be stored on an USB key which will be kept in a locked cabinet. Only the researchers listed in this letter will have access to the USB.

Data will be stored indefinitely following the termination of the study. Some demographic information will be collected as part of this study (age range, gender, level of schooling)

All information collected during this study, including your demographic information, will be kept confidential and will not be shared with anyone outside the study unless required by law. You will not be named in any reports, publications, or presentations that may come from this study.

Confidentiality:

This research study includes the collection of demographic data which will be aggregated (not individually presented) in an effort to protect your anonymity. Despite best efforts it is possible that your identity can be determined even when data is aggregated, however,

due to the anonymity of the data the risk of this is very low.

Please note that communication via e-mail is not absolutely secure. Thus, please do not communicate personal sensitive information via e-mail.

Voluntary Participation:

Your participation in this study is voluntary and you may partake in only those aspects of the study in which you feel comfortable. You may also decide not to be in this study, or to be in the study now, and then change your mind later. You may leave the study at any time without affecting your relationship with the researchers. You will be given information that is relevant to your decision to continue or withdraw from participation. Such information will need to be subsequently provided.

You may refuse to answer any question you do not want to answer by simply choosing the 'No answer' option

Right to Withdraw:

If you withdraw from the research project at any time, any data that you have contributed will be removed from the study and you do not need to offer any reason for making this request.

In some research projects, the withdrawal of data may not be feasible. This is an anonymous survey and thus data cannot be withdrawn after you have completed the study.

If you stop answering questions or close your online browser early your data will not be collected, you will be considered withdrawn from the survey.

Conflict of Interest:

As previous volunteers who have worked with the research team on previous projects you may feel obligated to complete the survey. The survey is designed to be completely anonymous so that researchers will not be able to determine if you completed the survey, your decision about completing the survey will not affect your relationship with the researchers.

Researchers have an interest in completing this study. Their interests should not influence your decision to participate in this study.

Compensation, Reimbursement, Incentives:

Participants will not incur any expenses as a result of their participation in the study.

No incentives, compensation, or reimbursement action will take place for the successful completion of this study.

Debriefing and Dissemination of Results:

If you are interested in the results of this study you may contact Erin Smith at erin.smith@uoit.ca to find out how the results will be published.

Participant Rights and Concerns:

Please read this consent form carefully and feel free to ask the researcher any questions that you might have about the study. If you have any questions about your rights as a participant in this study, complaints, or adverse events, please contact the Research Ethics Office at (905) 721-8668 ext. 3693 or at researchethics@uoit.ca.

If you have any questions concerning the research study or experience any discomfort related to the study, please contact the researcher Erin Smith at (905)721-8668 ext. 3050 or erin.smith@uoit.ca

By signing this form, you do not give up any of your legal rights against the investigators, sponsor or involved institutions for compensation, nor does this form relieve the investigators, sponsor or involved institutions of their legal and professional responsibilities.

Consent to Participate:

Online Consent

1. I have read the consent form and understand the study being described.

2. I have had an opportunity to ask questions and my questions have been answered. I am free to ask questions about the study in the future.

3. I freely consent to participate in the research study, understanding that I may discontinue participation at any time without penalty. A copy of this Consent Form has been made available to me.

I agree

I do not agree

Property Usage During the Pandemic

The following are some questions about where your waterfront property is, and how much time you are spending at it during the COVID-19 pandemic.

1. Do you own or lease a waterfront property in the Kawartha Lakes?

Yes

No

No answer

2. Which of the following lakes is your property on? Please choose one of the following:

Balsam

Cameron

Sturgeon

Pigeon

Scugog

Other (Please specify) _____

No answer

3. Have you been residing at your waterfront property for anytime during the COVID-19 pandemic?

Yes

No

No answer

4. During the following time periods how many days per week did you spend at your waterfront property?

	March 15 – April 15	April 16 – May 15	May 16 – June 15
2019	0, 1, 2, 3, 4, 5, 6, 7, No answer	0, 1, 2, 3, 4, 5, 6, 7, No answer	0, 1, 2, 3, 4, 5, 6, 7, No answer
2020	0, 1, 2, 3, 4, 5, 6, 7, No answer	0, 1, 2, 3, 4, 5, 6, 7, No answer	0, 1, 2, 3, 4, 5, 6, 7, No answer

5. During the period of March 15 to June 15, what is the total number of people (including yourself) that have visited or stayed at your waterfront property?

1

2

3

4

5

6+

No answer

Questions about Your Septic System

This section asks you specific questions about your current septic system

6. Does your waterfront property use a septic system?

Yes

No

I don’t know

No answer

[If no skip to next page]

7. How long have you had your current septic system?

0-5 years

6-10 years

11-15 years

16-20 years

21-25 years

26+ years

No answer

8. When was the last time you had your septic system pumped-out?

Last year

Two years ago

Three years ago

Four years ago

Five years ago

Six or more years ago

I have never had my septic system pumped out

I don’t know

No answer

Sanitation Habits during COVID-19

This section has specific questions about your sanitary habits during the COVID-19 pandemic.

While staying at your waterfront property, how have the following household habits changed during the COVID-19 pandemic (March – June, 2020) compared to the same time period last year (March – June, 2019)?

9. Using detergents and/or cleaning solutions to clean different areas of your household (e.g., kitchen or bathroom):

Greatly increased

Increased

Stayed the same

Decreased

Greatly decreased

No answer

10. Handwashing with soap (liquid soap or bar soap):

Greatly increased

Increased

Stayed the same

Decreased

Greatly decreased

No answer

11. Water usage related to cleaning surfaces, washing clothes, hand-washing, and bathing:

Greatly increased

Increased

Stayed the same

Decreased

Greatly decreased

No answer

12. Do you use low phosphate or phosphate free soaps and cleaners?

Yes

No

Sometimes

No answer

Property Maintenance Habits

These questions are specific to your property maintenance during the COVID-19 pandemic.

13. Does your waterfront property have a lawn and/or garden (e.g., flower bed, vegetable garden)?

Yes

No

No answer

[If no skip directly to Q#23]

14. Did you apply fertilizer to your lawn and/or garden at your waterfront property last year (2019)?

Yes

No

No answer

15. Have you applied fertilizer to your lawn and/or garden this year (2020) at your waterfront property?

Yes

No

No answer

16. How does your fertilizer use at your waterfront property this year compare to last year?

Greatly increased

Increased

Stayed the same

Decreased

Greatly decreased

No answer

17. What type of fertilizer does your household typically use if and when you fertilize your lawn or garden? Please check all that apply.

Liquid fertilizer

Granular fertilizer

Organic fertilizer

Phosphorus-free fertilizer

Don't know

Not applicable: my lawn and/or garden is not fertilized

Other

No answer

18. If you have a garden at your waterfront property, please check the following garden types that best describes the type of garden(s) you have. Please check all that apply:

Flowerbeds

Vegetable garden

Ornamental shrubs and trees

Native wildflower/Pollinator garden

Shoreline plants for erosion control/shoreline naturalization

Not applicable: there are no maintained gardens at my waterfront property

No answer

19. Compared to last year (2019) how has the amount of time spent gardening changed this year at your waterfront property?

Greatly increased

Increased

Stayed the same

Decreased

Greatly decreased

No answer

20. If your gardening activities have changed this year compared to last year at your waterfront property, what specific changes have occurred? Check all that apply.

Added garden plants

Removed garden plants

Naturalized the shoreline with native plants

Removed plants from the shoreline

Enhanced or created a rain garden/BlueScaping (i.e., gardens that trap stormwater run-off)

Removed a garden to add new landscape structures (e.g., stone patio)

Other

Not applicable: No changes in gardening activities occurred this year

No answer

21. Have you added (or plan to add) any new landscaping features to your waterfront property this year (2020)?

Add a hardened shoreline (armour stone, retaining wall, riprap etc.)

Maintain/fix a hardened shoreline

Remove a hardened shoreline

Remove lawn area to replace with a hard surface (interlocking, gravel, patio stones, concrete etc.)

Replace hard surfaces with vegetation (e.g., lawn or garden)

Not applicable: no new landscaping features were added or planned to be added to my waterfront property this year

No answer

22. Compared to last year (2019) how has your lawn mowing frequency changed this year (2020)?

Greatly increased

Increased

Stayed the same

Decreased

Greatly decreased

No answer

Tell us about yourself

We are asking these questions so we can develop a broad understanding of the respondents in this survey.

23. How long have you lived in or seasonally resided in your current waterfront property?

Only years in numbers may be entered in this field

24. What is your gender? Please choose only one (1) of the following:

Female

Male

Transgender

Non-binary

Prefer not to say

No Answer

25. In which age range do you fall? Please choose only one (1) of the following:

0 to 17 years of age

18 to 24 years of age

25 to 34 years of age

35 to 44 years of age

45 to 55 years of age

56 to 64 years of age

65 years of age or older

No answer

26. What is the highest level of education you have successfully completed to date? Please choose only one of the following:

No schooling

Elementary School

High School Diploma or equivalent

Trade/Vocational Certificate (including apprenticeships)

College diploma or certificate

Bachelor's degree

Post-graduate studies (e.g., Masters or PhD)

Other

No answer

Thank you for completing the survey!

Dear Participant,

During this time it is important to stay indoors and practice social distancing. The following link gives you ideas about how to stay occupied during the pandemic. https://www.vicnews.com/trending-now/40-things-to-at-home-during-the-coronavirus-pandemic/

If you would like to know more about COVID-19 please visit this link:

https://www.publichealthontario.ca/en/diseases-and-conditions/infectious-diseases/respiratory-diseases/novel-coronavirus

If you have any questions about our research, including learning about the aggregated results please contact Erin Smith at erin.smith@uoit.ca

Thank you again for your participation.

Sincerely,

Erin Smith

Ontario Tech University

Faculty of Science

Appendix B.

Chapter 4

Figure B1.1. Mean (standard deviation) nutrient and fluorometer chlorophyll-a values for samples collected from each lake in 2020.

Watershed	Lake	n	TP (µg/L)	TN (mg/L)	NH (mg/L)	NO2 (mg/L)	NO3 (mg/L)	TKN (mg/L)	Chla (mg/L)
Talbot River – Trent Severn Waterway	Canal	8	9.38 (5.21)	0.28 (0.06)	0.040 (0.017)	0.002 (0.000)	0.004 (0.003)	0.28 (0.06)	1.81 (0.43)
	Mitchell	8	11.60 (6.07)	0.23 (0.11)	0.033 (0.24)	0.002 (0.000)	0.006 (0.004)	0.23 (0.10)	0.96 (0.25)
Cameron Lake Dam – Fenelon River	Balsam	20	10.79 (5.71)	0.25 (0.13)	0.021 (0.004)	0.002 (0.001)	0.008 (0.009)	0.24 (0.13)	1.20 (0.44)
	Cameron	20	6.81 (5.45)	0.24 (0.15)	0.026 (0.020)	0.003 (0.005)	0.010 (0.008)	0.23 (0.14)	0.95 (0.32)
Lake Scugog – Scugog River	Scugog	15	45.90 (24.19)	0.87 (0.30)	0.073 (0.090)	0.002 (0.001)	0.004 (0.002)	0.86 (0.30)	5.27 (5.28)
Bobcaygeon River	Sturgeon	27	19.45 (13.12)	0.34 (0.18)	0.024 (0.009)	0.002 (0.002)	0.016 (0.026)	0.33 (0.18)	1.30 (0.45)
Pigeon Lake – Gannon Narrows	Pigeon	16	16.72 (8.44)	0.29 (0.09)	0.020 (0.000)	0.002 (0.001)	0.014 (0.017)	0.28 (0.08)	1.63 (0.58)
	Big Bald	8	12.21 (1.78)	0.44 (0.37)	0.031 (0.017)	0.002 (0.000)	0.005 (0.004)	0.44 (0.36)	1.31 (0.32)
Burleigh Falls Dam – Lower Buckhorn Lake	Sandy	3	0.29 (0.41)	0.29 (0.00)	0.020 (0.000)	0.002 (0.000)	0.003 (0.000)	0.29 (0.00)	1.79 (0.29)
	Upper Buckhorn	18	18.73 (11.49)	0.45 (0.25)	0.052 (0.030)	0.002 (0.000)	0.005 (0.005)	0.45 (0.24)	1.31 (0.40)
	Chemong	10	9.04 (4.35)	0.26 (0.06)	0.023 (0.009)	0.002 (0.000)	0.005 (0.004)	0.26 (0.06)	1.84 (0.52)
	Lower Buckhorn	15	17.95 (13.98)	0.27 (0.06)	0.023 (0.010)	0.002 (0.000)	0.003 (0.001)	0.27 (0.06)	1.54 (0.75)

	Lovesick	12	14.83 (2.64)	0.30 (0.10)	0.022 (0.006)	0.002 (0.00)	0.0003 (0.001)	0.30 (0.10)	1.66 (0.46)
Stony Lake – Katchewanooka Lake	Stony	16	11.61 (4.13)	0.21 (0.08)	0.020 (0.000)	0.002 (0.000)	0.003 (0.000)	0.21 (0.08)	1.16 (0.28)
	Clear	24	14.52 (9.34)	0.20 (0.07)	0.022 (0.007)	0.002 (0.000)	0.006 (0.006)	0.20 (0.07)	1.32 (0.43)
	Katchewanooka	4	10.11 (2.67)	0.25 (0.02)	0.020 (0.000)	0.002 (0.002)	0.013 (0.008)	0.24 (0.01)	1.37 (0.19)

Chapter 5

Table B1.2 Seasonal average of water quality parameters pooled from both years (2019 and 2021) of the community science water collection program. Standard deviation in brackets.

Lake	Year	TP (µg/L)	TDP (µg/L)	Turbidity Abs (750nm)	TSS (g/L)	Chla (mg/L)	Coliforms (MPN 100mL)	*E. coli* (MPN 100mL)	TN (mg/L)	Nitrite (mg/L)	Nitrate mg/L	NH4 mg/L	TON mg/L
Balsam	2019 (n=76)	7.16 (3.18)	1.40 (1.93)	0.0004 (0.008)	4.10 (2.90)	0.89 (1.18)	6 (10)	2(5)	0.239 (0.059)	0.0000 (0.003)	0.0154 (0.0220)	0.012 (0.028)	0.2111 (0.0551)
	2021 (n=59)	5.80 (3.22)	0.42 (0.80)	0.0004 (0.008)	0.10 (0.40)	0.61 (0.82)	14 (24)	3 (5)	0.235 (0.064)	0.0007 (0.0040)	0.0050 (0.0089)	0.008 (0.028)	0.2205 (0.0516)
Cameron	2019 (n=21)	8.09 (8.91)	2.28 (6.21)	0.0003 (0.004)	10.69 (35.11)	1.01 (0.86)	4 (5)	1 (2)	0.185 (0.060)	0.0002 (0.009)	0.0210 (0.0230)	0.009 (0.018)	0.1548 (0.0561)
	2021 (n=30)	6.00 (2.97)	0.72 (0.91)	0.0007 (0.0011)	11.05 (58.33)	1.17 (0.82)	16 (22)	3 (6)	0.256 (0.050)	0.0001 (0.007)	0.0146 (0.0184)	0.001 (0.007)	0.2403 (0.0374)

Sturgeon	2019 (n=61)	19.54 (16.23)	4.92 (4.26)	0.0005 (0.0014)	5.39 (18.99)	1.87 (2.18)	13 (23)	5 (11)	0.403 (0.164)	0.0016 (0.0028)	0.0532 (0.0747)	0.023 (0.032)	0.3251 (0.1487)
	2021 (n=58)	14.17 (11.05)	2.66 (5.44)	0.0039 (0.0055)	3.03 (7.11)	1.78 (1.95)	20 (37)	4 (7)	0.424 (0.160)	0.0014 (0.0020)	0.0913 (0.1168)	0.014 (0.024)	0.3172 (0.1410)
Pigeon	2019 (n=28)	23.79 (22.59)	6.84 (4.47)	0.0048 (0.0193)	2.86 (2.43)	4.41 (4.00)	12 (19)	5 (9)	0.591 (0.733)	0.0016 (0.0023)	0.0362 (0.0641)	0.077 (0.301)	0.4756 (0.4435)
	2021 (n=21)	13.61 (7.61)	1.70 (2.92)	0.0052 (0.0046)	0.22 (0.51)	2.46 (1.77)	13 (12)	3 (5)	0.443 (0.149)	0.0011 (0.0018)	0.0651 (0.0948)	0.014 (0.027)	0.3645 (0.1418)

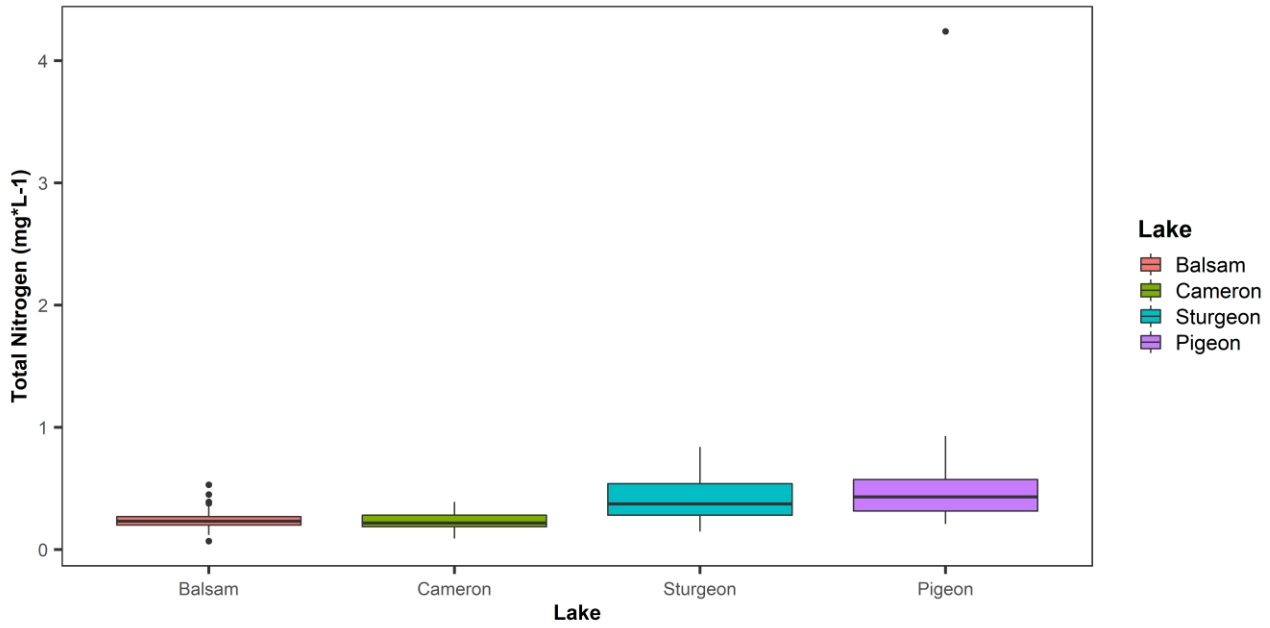

Figure B1.1. Boxplot comparing total nitrogen (mg*L^{-1}) across the four study lakes with outlier included.

Table B1.3 Results of piecewise structural equation model (SEM) for the final selected model (Fisher's C = 44.12, p = 0.302, df = 40), showing unstandardized path coefficients (Estimate), standard error of regression weight (Std. Error), degrees of freedom (DF), critical value for regression weight (Critical Value), level of significance for regression weight (p), and standardized path coefficients (Std. Estimate). p-values < 0.05 are bolded.

Response variable	**Predictor variable**	**Estimate**	**Std. Error**	**DF**	**Critical Value**	**p**	**Std. Estimate**
Total suspended solids	Shoreline	-0.0871	0.1062	22	-0.8195	0.4213	-0.0996
Total suspended solids	Land Use	0.0007	0.004	22	0.1658	0.8698	0.0207
Total suspended solids	Macrophytes	-0.0015	0.0024	22	-0.5997	0.5548	-0.0751
Total Phosphorus	Shoreline	0.0212	0.0721	23	0.2941	0.7713	0.041
Total Phosphorus	Land Use	0.0029	0.0027	23	1.0489	0.3051	0.1504
Total Phosphorus	Total suspended solids	-0.0174	0.0471	114	-0.3689	0.7129	-0.0293
Total Nitrogen	Shoreline	-0.0079	0.0096	23	-0.8213	0.4199	-0.0875
Total Nitrogen	Land Use	0.0011	0.0004	23	3.1252	**0.0048**	0.3429
Total Nitrogen	Total suspended solids	0.0116	0.0074	114	1.5701	0.1192	0.1127
Macrophytes	Shoreline	-11.3006	8.5881	23	-1.3158	0.2012	-0.2499
Macrophytes	Land Use	-0.198	0.3192	23	-0.6203	0.5412	-0.1184
Periphyton	Shoreline	-0.1477	0.1467	23	-1.0065	0.3246	-0.1383
Periphyton	Land Use	-0.0042	0.0059	23	-0.7083	0.4859	-0.1055
Periphyton	Total Phosphorus	0.4734	0.2764	83	1.7125	0.0905	0.2296
Periphyton	Total suspended solids	-0.5054	0.1312	83	-3.8533	**0.0002**	-0.4136
Periphyton	Total Nitrogen	1.9357	1.7659	83	1.0962	0.2762	0.1631
Phytoplankton density	Total Phosphorus	1.1617	0.3553	20	3.2691	**0.0038**	0.629
Phytoplankton density	Shoreline	-0.1137	0.1556	20	-0.7309	0.4733	-0.1189

Phytoplankton density	Macrophytes	-0.0029	0.0035	20	-0.8155	0.4244	-0.1349
Phytoplankton density	Land Use	-0.0039	0.0059	20	-0.6729	0.5087	-0.1116
Phytoplankton density	Total suspended solids	-0.0011	0.2251	20	-0.0048	0.9962	-0.001
Phytoplankton diversity	Total Phosphorus	-0.0898	0.053	19	-1.6936	0.1067	-0.4428
Phytoplankton diversity	Shoreline	-0.0006	0.02	19	-0.0305	0.976	-0.0058
Phytoplankton diversity	Macrophytes	0.0007	0.0005	19	1.6122	0.1234	0.3152
Phytoplankton diversity	Land Use	0.0013	0.0008	19	1.6724	0.1108	0.325
Phytoplankton diversity	Total suspended solids	-0.0093	0.0256	19	-0.3632	0.7204	-0.0775
Phytoplankton diversity	Phytoplankton density	0.0582	0.0261	19	2.2299	**0.038**	0.5296
Zooplankton diversity	Total Phosphorus	-0.1189	0.0966	19	-1.2314	0.2332	-0.3297
Zooplankton diversity	Shoreline	-0.0207	0.0371	19	-0.5567	0.5842	-0.1106
Zooplankton diversity	Macrophytes	-0.0013	0.0008	19	-1.5756	0.1316	-0.323
Zooplankton diversity	Phytoplankton density	0.0649	0.0478	19	1.36	0.1898	0.3325
Zooplankton diversity	Land Use	0.0008	0.0014	19	0.5581	0.5833	0.1133
Zooplankton diversity	Total suspended solids	0.0382	0.0446	19	0.8572	0.402	0.1786
Zooplankton density	Total Phosphorus	0.4039	0.1642	19	2.4598	**0.0237**	0.5422
Zooplankton density	Shoreline	-0.0329	0.0631	19	-0.5214	0.6081	-0.0852
Zooplankton density	Macrophytes	0.0007	0.0014	19	0.5207	0.6086	0.0879
Zooplankton density	Phytoplankton density	-0.0017	0.0812	19	-0.0213	0.9832	-0.0043
Zooplankton density	Land Use	0.0038	0.0024	19	1.5955	0.1271	0.2665
Zooplankton density	Total suspended solids	0.0213	0.0757	19	0.281	0.7817	0.0482
Macroinvertebrate diversity	Phytoplankton density	-0.0184	0.0185	16	-0.9997	0.3323	-0.2369

Macroinvertebrate diversity	Zooplankton density	-0.0146	0.0457	16	-0.3186	0.7542	-0.0755
Macroinvertebrate diversity	Zooplankton diversity	-0.0473	0.0797	16	-0.5928	0.5616	-0.1186
Macroinvertebrate diversity	Macrophytes	-0.0005	0.0003	16	-1.4654	0.1622	-0.3057
Macroinvertebrate diversity	Shoreline	0.0093	0.0139	16	0.6684	0.5134	0.1247
Macroinvertebrate diversity	Total Phosphorus	0.1041	0.0403	16	2.5858	**0.0199**	0.7241
Macroinvertebrate diversity	Land Use	-0.0002	0.0005	16	-0.3054	0.764	-0.0606
Macroinvertebrate diversity	Periphyton	-0.0071	0.0136	16	-0.5206	0.6098	-0.1013
Macroinvertebrate diversity	Total suspended solids	0.0212	0.0166	16	1.2811	0.2184	0.2489
Macroinvertebrate density	Phytoplankton density	0.0869	0.1733	16	0.5017	0.6227	0.1234
Macroinvertebrate density	Zooplankton diversity	-0.1144	0.7487	16	-0.1528	0.8805	-0.0317
Macroinvertebrate density	Zooplankton density	-0.616	0.4294	16	-1.4346	0.1707	-0.3528
Macroinvertebrate density	Macrophytes	-0.0012	0.0032	16	-0.385	0.7053	-0.0834
Macroinvertebrate density	Shoreline	-0.0478	0.1305	16	-0.3663	0.7189	-0.0709
Macroinvertebrate density	Total Phosphorus	0.8112	0.3781	16	2.1457	**0.0476**	0.6236
Macroinvertebrate density	Land Use	-0.0041	0.0051	16	-0.8073	0.4313	-0.1661
Macroinvertebrate density	Periphyton	-0.213	0.1274	16	-1.6717	0.114	-0.3375
Macroinvertebrate density	Total suspended solids	0.1249	0.1554	16	0.8034	0.4335	0.162

Table B1.2. Summary of variance explained by each model, marginal R^2 values indicate the total proportion of variance explained by predictor variables.

Response Variable	Marginal R^2	Conditional R^2
Total suspended solids	0.01	0.95
Total phosphorus	0.02	0.59
Total nitrogen	0.13	0.57
Macrophytes	0.08	0.92
Periphyton	0.17	0.39
Phytoplankton Density	0.33	0.93
Phytoplankton Diversity	0.23	0.92
Zooplankton Density	0.2	1
Zooplankton Diversity	0.42	1
Macroinvertebrate Density	0.36	1
Macroinvertebrate Diversity	0.32	1