Spatial distribution of water quality and phytoplankton in the Upper Manitoba Great Lakes

By

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A Thesis submitted to the Faculty of Graduate and Postdoctoral Studies of

The University of Manitoba

In partial fulfillment of the requirements of the degree of

MASTER OF SCIENCE

Centre for Earth Observation Science

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ABSTRACT

Freshwater eutrophication in Canada poses significant threats to ecosystem health and community wellbeing, particularly in large lake systems like the upper Manitoba Great Lakes (uMBGL). Lakes Winnipegosis, Waterhen, and Manitoba form a critical buffer system within the Nelson River watershed, processing nutrients before they reach Lake Winnipeg and ultimately Hudson Bay and the North Atlantic Ocean. Despite their importance, these lakes remain severely understudied, with minimal spatial and temporal data available about nutrient dynamics and phytoplankton communities. This knowledge gap hinders evidence-based management decisions necessary to protect these valuable freshwater resources from eutrophication driven by modern challenges such as land use management and accelerated climate change.

This study provides the first spatially comprehensive, multi-year assessment of offshore water quality in the upperMBGL system. Over three open-water seasons (2016-2017), I collected and analyzed physical, chemical, and biological data across multiple basins to: (1) characterize in-situ offshore biogeochemical and physical conditions; (2) examine spatial and temporal variation through geostatistical analysis; and (3) document phytoplankton diversity and distribution patterns.

Results indicate that all three lakes are consistently mesotrophic to eutrophic. Nutrient concentrations generally increased from north to south in both Lakes Winnipegosis and Manitoba during the open water season. Filamentous cyanobacteria dominated summer phytoplankton biomass in both lakes, while cyanobacterial picoplankton dominated by abundance. Significant differences between years and basins suggest limited inter-basin mixing and differential impacts from local land use.

This research establishes a critical western science-based baseline for understanding water quality dynamics in the Upper Manitoba Great Lakes system, and highlights the urgent need for continued monitoring, community-engaged research, and the weaving of traditional ecological knowledge with western science approaches to ensure these lakes can continue functioning as effective nutrient buffers for Lake Winnipeg.

ACKNOWLEDGEMENTS

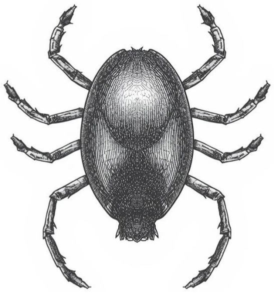
First, I would like to thank you, dear reader, for making it this far. If you’ve made it to this page then you’ve read the abstract, and likely care a bit about freshwater in the Prairies, so thanks!

Thank you to Dr. David Barber who was originally my advisor. Without him, this work would not have been possible. Thank you to Dr. Greg McCullough for his guidance, and to Dr. Jens Ehn who offered to step up as advisor when David died.

Thank you to Wayne Chan for listening to me ask multiple questions at random times of the day or night about math, and for spending innumerable hours recreating the algal counting program (phytobio) in python, as well as creating a digital counter for me so I don’t have to rely on reading my own handwriting to enter data!

Financial support was provided by Dr. David Barbers Canada Research Chair and the Lake Winnipeg Foundation.

I do NOT, however thank you, my wee friend. Except maybe thanks for the nightmares.



DEDICATION

I am so very grateful to all the people who have supported me over these long years. Thank you to Dr. David Barber for not only the funding he provided, but also the constant support and enthusiasm for my research and for the state of freshwater in Canada. His vision of ecosystem scale research highlights why all systems are important and interconnected, and that we cannot ignore one part if we wish to truly find solutions. Dr. Greg McCullough, my mentor and hero, who has had infinite patience with me, both in the field (try living with me for 10 days straight) and with my journey taking sooo long. There has never been a question about any topic area I've asked that Greg doesn't have an answer to, even if it's, "I'm not an expert, but…". I've never been bored in the field, whether it's learning about the landscape as we drive, being shown (by example), how to think through many different aspects of a question, or discussing any random topic we think up. Thank you Jens for being so encouraging and always asking interesting questions that make me think things through again from a different perspective. Dr. Tim Papakyriakou, thank you for being one of my freshwater peeps, and reading this! And for your constant support over the last decade in all things freshwater. To Hedy Kling, who has been a mentor to me for over two decades (OMG), has had infinite patience in trying to teach me the magical world of algae, not to mention fed me and made sure I have my vitamins. You are a treasure. Your farm is one of my favourite places. Thank you to George for putting up with me secluding myself for so long, and to my dearest possee for always supporting me, feeding me and making life livable. You don't have to read this thesis since you've lived every last minute with me. Thank you to my father, Brian Herbert, who made me the science nerd I am today, and to my mum for making me environmentally aware of the consequences of my actions long before it was cool. And thank you to Celia Mellinger!

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION

Freshwater is critical to human and animal health, yet only 3% of the earth's water is freshwater, and of that, only 0.5% is accessible (World Meteorological Organization 2021). Canada is home to 20% of the world's available freshwater resources, and just 7% of the world's renewable freshwater supply (i.e. that not retained in lakes, underground aquifers and glaciers) (Government of Canada, 2024). We must therefore take great care to ensure our valuable and limited freshwater resources are protected and maintained to the highest standard possible.

Globally, over the past two decades, freshwater lakes around the world have been subjected increasingly to multiple stressors, including land use (agricultural, industrial), resource extraction and climate pressures (McCullough et al., 2012; Michalak et al., 2013). Eutrophication, which is an increase in nutrient inputs into a waterbody, is slow naturally, but can be greatly accelerated due to human (anthropogenic) activity (Khan & Ansari, 2005). In large freshwater lakes such as Lakes Erie and Winnipeg in Canada, Lake Victoria in Africa and Lake Taihu in China, signs of lake stress are manifesting in symptoms like eutrophication, resulting in negative impacts such as an increasing algal presence, often in the form of surface blooms (Paerl et al., 2011). In aquatic ecosystems, human induced eutrophication has been shown to cause a multitude of problems, including a decrease in water clarity, oxygen depletion (from senescing blooms), fish kills and a loss of biodiversity (Carpenter et al., 1998; Environment Canada & Manitoba Water Stewardship, 2011; Kotak & Zurawell, 2007; Orihel et al., 2017; Paerl, 1988), leading to economic loss for communities (Environment Canada & Manitoba Water Stewardship, 2011; Kolker et al., 2013).

1.1 Eutrophication in the Canadian context

Eutrophication in Canada is leading to a demonstrated decrease in water quality in large and small water bodies that are critical to the health and welfare of humans and animals (Environment Canada & Manitoba Water Stewardship, 2011; Michalak et al., 2013). Rapid eutrophication of Canadian great lakes such as Winnipeg (Manitoba) and Erie (Ontario), are causing increased algal production and a decrease in algal species diversity, favouring potential toxin producing cyanobacterial species (Environment et al., 2020; Manitoba Clean Environment Commission, 2015; Schindler et al., 2012). In addition, the size and duration of algal blooms are not only aesthetically unappealing but can produce noxious taste and odours (Kotak & Zurawell, 2007), and compounds (cyanotoxins) toxic to humans, wildlife and aquatic biota (Kotak & Zurawell, 2007; Lewtas et al., 2015). Cyanotoxins are known to accumulate in aquatic species such as fish or bivalves, as well as in crops that are watered with contaminated water sources or via groundwater leaching. Consumption of either can lead to chronic issues such as permanent liver damage or a suppressed immune system in humans (Cheung et al., 2013; Lee et al., 2017). This is of particular concern in First Nation and other communities situated near the lakes, who may use them directly as their irrigation, recreation or drinking source. Human exposure to algal toxins at low levels via contaminated drinking water is an additional source of concern in First Nation or other communities whose water treatment plants may need repair, does not have the appropriate equipment to remove algal toxins (Chorus & Welker, 2021). Chronic or short-term exposure can result in a decrease in human and animal health and an increase in costs of water treatment and health care (Dodds et al., 2009; Kotak & Zurawell, 2007). A study conducted across Canada by Orihel (2012) on the concentrations of the algal toxin microcystin in 246 eutrophic lakes, ponds and reservoirs, found that over a 10-year period microcystin concentrations in eutrophic lakes was increasingly common, with the highest concentrations occurring when the nitrogen (N) to phosphorus (P) molar ratio was lower than 23.

In Lake Erie this eutrophication trend has resulted in a shift in cyanobacterial species dominance (from *Anabaena* sp. and *Aphanizomenon* sp. to *Microcystis* sp.), an increase in intensity and size of blooms, the introduction of aquatic invasive species such as benthic harmful algal blooms (e.g. *Cladophera*) and zebra mussels (*Dreissena polymorpha*), and a shift in trophic status within the littoral and pelagic regions (Binding et al., 2018; Hecky et al., 2004; Watson & Kling, 2017). In Lake Winnipeg there has been a rapid acceleration of eutrophication since the 1990s, resulting in total phosphorus concentrations that are double the current provincial objective of 0.05 mg L^{-1}, leading to an increase in algal bloom presence and size (Environment et al., 2020). The introduction of invasive species like zebra mussels has the potential to exacerbate eutrophication in Lake Winnipeg through effects like the phosphorus shunt (Hecky et al., 2004).

In Canada, the major drivers of eutrophication are excess nutrients, with excess phosphorus of primary consideration (Orihel et al., 2012; Schindler et al., 2008, 2012). Phosphorus induced cultural eutrophication is associated with four main causes: (a) increased erosion and runoff due to the conversion of forests and grasslands to cropland and for urban development; (b) recycling of organic wastes in agriculture; (c) urban sewage; and (d) use of inorganic fertilizers (Smil, 2000). In the Canadian Prairies, the main categories of high phosphorus loading into lakes can be divided into point source (discrete sources such as a wastewater treatment plant or combined sewer overflow systems); or non-point source (resulting from land use change from forest and grassland to agriculture or urban landscapes) (Schindler et al., 2012).

1.2 Agricultural Land Use Change

With the intensification of agricultural farming practices (McCullough et al., 2012), land use practices and animal husbandry (McCullough et al., 2012), a clearer understanding of the relative contribution of nitrogen and phosphorous and how their fractions interact in waterbodies is necessary to develop and implement effective land management practices for Canadian freshwater systems. Agricultural land use records show a distinct shift in many areas in Canada and the US in both the type and intensity of use (Dumanski et al., 2015; Rattan et al., 2017). From the 1920's onward, systematic drainage of wetlands (Dumanski et al., 2015; Khan & Ansari, 2005; Rattan et al., 2017) and construction of drainage channels was a common practice used to increase viable agricultural land. The draining of wetlands to make drainage channels establishes permanent surface water connections between isolated wetlands, ditches and streams, thereby increasing contributing areas and runoff volume (land no longer storing water long term) which, in turn, can increase flow and magnitude of flooding (Dumanski et al., 2015; Khan & Ansari, 2005). Runoff from flooded agricultural land also tends to be higher in soluble reactive phosphorous, (SRP) (the most biologically available portion of dissolved phosphorous (DP) (McCullough et al., 2012).

In addition, large-scale conversion of land cover type to agricultural use has caused a shift in nutrient flows to waterbodies by allowing increased nutrient input at vulnerable times of the year, such as during snowmelt or high precipitation events in the spring and fall (Rodgers, 2022). Both snowmelt and high precipitation events (which may also cause flooding for short periods of time), carry the capacity for increasing phosphorus input into a water body. Snowmelt runoff accounts for 80% of the annual surface runoff for Prairie streams (Khan & Ansari, 2005), resulting in a higher dissolved portion of phosphorus in spring, while runoff from rainfall travelling over non-frozen ground in summer and fall can pick up a higher portion of particulates (McCullough et al., 2012). The particulate form of phosphorous may be less bio-available but could impact other parameters such as water clarity which in turn impacts the rest of the food chain. These changes in land use exacerbate the effects of climate change on waterbodies, where increased precipitation events, particularly in the spring, coupled with increased surface water temperature and length of the open water season, can lead to higher total nutrient inputs into lakes, as well as causing temporary connectivity between non-contributing and contributing drainage basins (Ali & English, 2019; Donald et al., 2015).

Beusen et al. (2016) used a coupled nutrient-input-hydrology-in stream nutrient retention model to track the changes in the global freshwater nitrogen (N) and phosphorus (P) cycles over the 20th century and found that although in-stream retention and removal of N and P increased, nutrient delivery to rivers and the ocean was still greater than retention. They concluded that while the greatest retention of nutrients globally was due mainly to the creation of reservoirs, agriculture was still the dominant source of non-point nutrient inputs to surface waters, exceeding retention of nutrients globally.

1.2.1 Management Practices

While type of crop matters when identifying landscape impacts on phosphorus inputs, farming practices (type of tillage, planting methods and location), coupled with weather and land characteristics, have the most influence on reducing the impacts of soil erosion (Government of Canada). In Ontario for example, over the past 30 years, crop production has shifted from corn to soybean, and in 2025 soybeans were Manitoba's third largest crop, increasing 5-five-fold since 2007 (Ontario and Quebec producing the most soybean crops (Province of Manitoba, 2025).

With conventional tillage practices, soybean farming leaves behind less crop residue, which means that there is no soil cover in the winter and into spring (Lobb et al., 2024). This change in landscape use and the timing of fallow soil could shift the proportion of dissolved to particulate phosphorous running off the landscape (Michalak et al., 2013). Yet, since 1981, soil erosion risk overall in Canada has only increased slightly, with the most marked improvements effected by an increase in conservation tillage (most common tillage type in the Prairies), and adoption of no-till practices used on cereal crops, thereby reducing soil movement and preserving crop residue on the soil, preventing further erosion (Province of Manitoba, 2025).

Identifying agricultural nutrient management practices and how they relate to land management practices is important in understanding nutrient transport and mobilization through the watershed. On the prairies, dissolved N and P in snow melt may be higher than expected based on the type of fertilization regime, land type and management practices. For example, using conservation tillage practices as mentioned previously, leaving crop residue on top of the soil, or relying on a riparian zone with old vegetation coupled with freeze/thaw cycling occurring in spring can lyse vegetation on the soil and release higher than expected amounts of dissolved N and P into the system. This can cause a cascade effect which may carry nutrients further through the watershed than expected due to extended connectivity of water systems on the frozen landscape and lack of protection from the riparian zone buffer pre-spring vegetation growth (Michalak et al., 2013).

The type of fertilizer applied is also an important consideration. Since the 1930's, global consumption of inorganic phosphorus fertilizer has steadily increased. Inorganic fertilizer use has increased from 1 Mt P/year to 16.5 Mt P/year in 1988, with the 1988 peak expected to have been reached again by 2005 (Smil, 2000). The commonly used form of inorganic phosphorus, ordinary superphosphate (OSP) contained on average 8.7% available P, which is an order of magnitude more than the commonly recycled P-rich manures (Smil, 2000).

The Soil and Water Assessment Tool (SWAT) model was applied to factors that were observed leading up to the 2011 Lake Erie algal bloom (Michalak et al., 2013) to assess the impact of agricultural nutrient management practices on the lake. It showed that management practices such as fertilizer addition timing and type of tillage (e.g. no-till), combined with high rainfall events, would support the increase of dissolved phosphorus into the lake. The SWAT model showed that current management practices common within the Lake Erie watershed (autumn fertilizer application, fertilizer broadcast onto the surface as opposed to injected, and conservation tillage) should be re-evaluated in the light of the high dissolved portion of phosphorous being delivered to the lake (Michalak et al., 2013).

1.3 Climate Change Effects

Climate change is the second leading cause of biodiversity change globally, second only to land use (Lemmen et al., 2007). However, the effects of climate change in shallow, polymictic lakes (Bartosiewicz et al., 2019) like those that cover the Canadian western and central prairies, are not well studied, leading to a gap in our ability to make science-based management decisions. Both direct climate effects such as increased surface and air temperature, and indirect ones such as altered precipitation events or decreased wind days and speeds, can lead to a change in both lake biogeochemistry and phytoplankton community distribution, ultimately favouring cyanobacteria. Increased surface temperature due to climate change may also cause shifts, either short or long term, to the mixing status of lakes, particularly polymictic ones, resulting in increased stratification and microstratification and decreased CO_2 availability (Bartosiewicz et al., 2019; Carey et al., 2012; Hanson et al., 2008). Climate warming may not only affect lakes with earlier, stronger and longer stratification, but may even cause a fundamental shift in lake stratification processes, for example, causing polymictic lakes to become dimictic for long periods of time, dimictic lakes to become warm monomictic lakes and monomictic lakes to become oligomictic (Carey et al., 2012; Gerten & Adrian, 2000).

Eutrophication will also be exacerbated by a changing climate. Increasing air temperature has historically been shown to result in a decrease of in streamflow in response to increased evaporation coupled with decreased precipitation. This ultimately results in a decrease in the ability of streams to dilute nutrients (Lemmen et al., 2007), and due to decreasing lake levels and stream flow into and out of lakes, an increase in lake residence time. In other cases, increasing seasonal precipitation and/or increasing intensity and duration of precipitation may lead to an increase in flood frequency and magnitude, resulting in increased nutrient loading to lakes (McCullough et al., 2012; Schindler et al., 2012). Increased periods of droughts, due to higher and longer air temperatures, and flooding caused by an increase in high precipitation events, may increase erosion into streams, resulting in increased sediment loads and therefore nutrient inputs (Lemmen et al., 2007; Ontkean et al., 2005). Drought conditions also favour increased forest fire extents, leading to loss of vegetation and soil cover and therefore the ability of soils to store water locally. With no buffer for the runoff from soils, there will be an increase in the amount of nutrients released into streams and their associated lakes (Lemmen et al., 2007).

1.4 The Nelson River Watershed

The Nelson River watershed (NRW) (Figure 1.1), which includes the Lake Winnipeg basin, is over 1 million km^2, and is the second largest basin in Canada by population, with nearly

7 million people[1] (Delong et al., 2023). Zubrycki et.al. developed a framework dividing the benefits provided by the NRW and Churchill watershed into four categories; i) provisioning services such as food, freshwater, hydroelectricity, minerals and metals, ii) regulating services such as flood and drought management, water quality regulation, and erosion control, iii) cultural services which include recreational and aesthetic services and value to Indigenous communities and iv) supporting services such as habitat for fish, waterfowl and wildlife, and biodiversity. Their report highlights the value of freshwater in Canada, as a drinking and recreational water source, for hydroelectricity, as a commercial and sustenance fishery (valued at over 1.77 million 2011-2012), as a carbon sink, a spiritual source for Indigenous communities, for tourism and as important habitat supporting aquatic biodiversity and wildlife.

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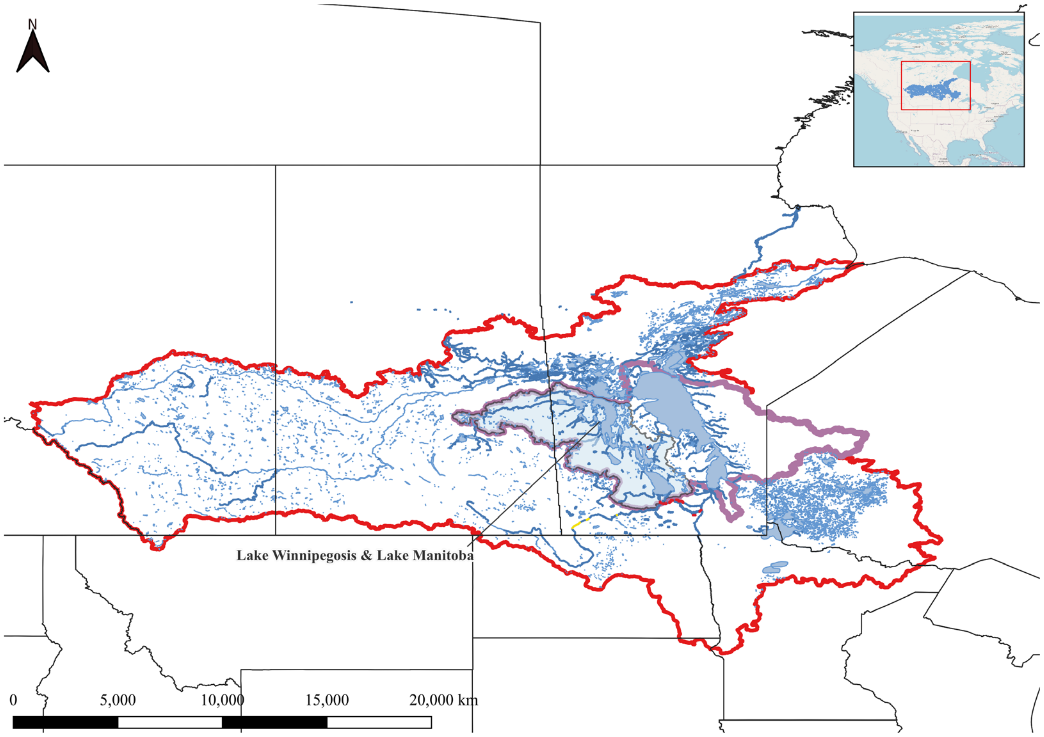
[1] https://www.gov.mb.ca/sd/water/lakes-beaches-rivers/lake-winnipeg.html#:~:text=Nearly%20seven%20million%20people%20live,First%20Nation%20and%20M%C3%A9tis%20communities.



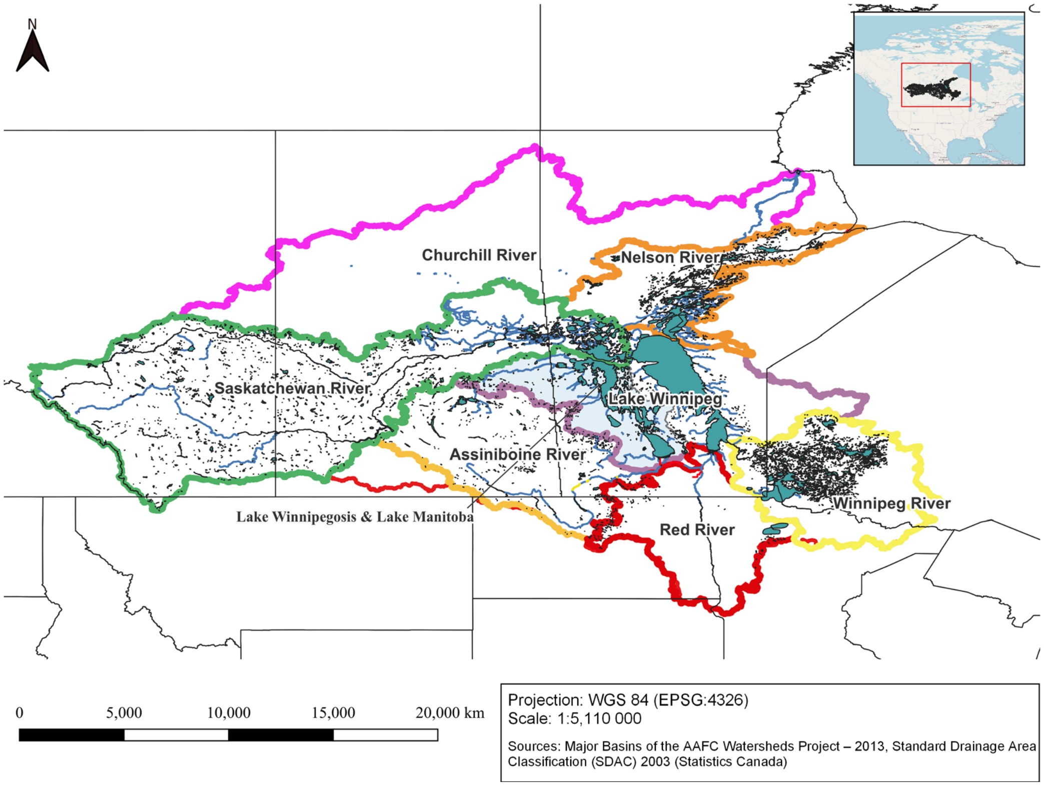
Projection: WGS 84 (EPSG:4326)
Scale: 1:5,110 000

Sources: Major Basins of the AAFC Watersheds Project – 2013, Standard Drainage Area Classification (SDAC) 2003 (Statistics Canada)

Figure 1.1. Nelson River watershed outlined in red, with the Lake Winnipegosis and Lake Manitoba watershed coloured blue and the Lake Winnipeg Major Basin outlined in purple

Within the NRW lies a series of interconnected lakes (Winnipegosis, Manitoba and Waterhen), which collectively form the upper Manitoba Great Lakes System (uMBGL). Lake Winnipegosis, the northernmost lake in the series, empties into Lake Waterhen, which in turn drains into Lake Manitoba. While multiple sub-basins exist within the NRW, all three upper MBGL lakes are situated within the Lake Winnipegosis and Lake Manitoba sub-basin (Agriculture and Agri-Food Canada, 2013), which extends from Manitoba into eastern Saskatchewan. These lakes ultimately feed, via the Fairford River, into Lake Winnipeg, forming another major basin (Lake Winnipeg), within the NRW (Agriculture and Agri-Food Canada, 2013). Lake Winnipeg is the only outlet in the NRW that flows into Hudson Bay, where freshwater and marine systems meet and are transformed before flowing into the arctic. The

Lake Winnipeg basin lakes are used extensively by the communities around them, both for sustenance and commercial fishing, as well as for recreation and drinking water (Figure 1.1, Figure 1.2) .



Projection: WGS 84 (EPSG:4326)
Scale: 1:5,110 000

Sources: Major Basins of the AAFC Watersheds Project – 2013, Standard Drainage Area Classification (SDAC) 2003 (Statistics Canada)

Figure 1.2. Nelson-Churchill River watershed highlighting the Lake Winnipegosis and Lake Manitoba sub-basin

Millions of dollars are spent each year protecting Canadian lakes, rivers and streams from urban and agricultural pollutants, excess nutrient input and in keeping fisheries healthy, yet we have large gaps in our understanding of the biogeochemical processes that occur within key drainage basins like the uMBGL. It has been well documented (Donald et al., 2015; Koiv et al., 2011) that large lakes can effectively sequester nitrogen and phosphorus, acting as nutrient filters for lakes downstream, but there are very few studies containing nutrient or phytoplankton data spatially and temporally on Lake Manitoba, one study on Lake Winnipegosis and none on Lake

Waterhen. Therefore, the effects on the aquatic food chain, including the relationship between phytoplankton and nutrients are not well understood. Current nutrient management strategies assume either that nitrogen (N) and phosphorus (P) co-limit primary productivity or that phosphorus is the main limiter to productivity in lakes, and that the relationship of chlorophyll to N and/or P is linear. It has, however, been demonstrated that particularly in lakes where the surrounding landscape is highly disturbed due to practices such as agriculture, the N and P biogeochemical processes may be disturbed and the relationship between nutrients, chlorophyll and primary productivity is also affected by the effects of light and algal composition (Dubourg et al., 2015; Filstrup & Downing, 2017).

1.5 Project Motivation

The majority of research conducted on the effects of a changing climate on nutrient processes and algal species diversity have occurred on lakes that are deeper than 9 metres and regularly stratify. Previous findings that nutrient concentrations are the key drivers for cyanobacterial dominance (Rigosi et al., 2014; Taranu et al., 2015) do not necessarily hold true for shallower polymictic lakes (Bartosiewicz et al., 2019). In shallow polymictic lakes, the effects of climate change-such as the impact of increasing water and air temperature-coupled with periods of intense heat (heat wave enhanced) that often coincide with calmer wind conditions, can lead to increased stratification in these lakes. Persistent stratification for longer periods of time can lead to epilimnetic carbon dioxide (CO_2) depletion, as well as altering the depth of light penetration, potentially favouring buoyant cyanobacteria which can utilize atmospheric N_2 and can move to access increased light and nutrients located deeper in the water column, or cyanobacteria which can use photoacclimation to maximize the amount of available light (Carey et al., 2012).

At present, management of the Lake Winnipeg watershed is heavily focused on decreasing phosphorus inputs into the lake but does not consider the large role lakes such as the uMBGL play in acting as nutrient sinks. We therefore have minimal science that can be used to make evidence-based decisions on whole lake ecosystem management, which is necessary to ensure the uMBGL and associated lakes can continue to act as buffers to prevent additional nutrients entering Lake Winnipeg. This thesis begins to address the lack of spatial and temporal data for two lakes on the uMBGL- Lakes Winnipegosis and Manitoba, and examines the relationships between in-situ nutrient concentrations and algal diversity for the open water season, to improve our understanding of how these lakes vary seasonally and annually, and how the effects of changing land use and climate may be impacting these lakes now and into the future.

1.6 Project Objectives

The goal of this thesis is to provide the first ever spatially comprehensive three-season (spring, summer, fall), multi-year report on offshore nutrient and biogeochemical data for Lakes Winnipegosis, Manitoba, and Waterhen. This will be accomplished by analyzing physical, chemical, and stoichiometric variables on the uMBGL and compare with algal taxa distribution to answer. Specifically, the sub-objectives of this project are to:

1. Provide an overview of selected in-situ off-shore biogeochemical and physical characteristics for the uMBGL- Lakes Winnipegosis, Waterhen and Manitoba.;
2. Conduct a geostatistical analysis to examine the spatial and temporal biogeochemical variation; and
3. Provide an overview of in-situ off-shore phytoplankton diversity for Lakes Winnipegosis and Manitoba

2. Review of the Upper Manitoba Great Lakes Study Area

2.1 Location

Lake Winnipegosis is located northwest of Lake Manitoba, and with an area of 5,403 km^2, is the 2nd largest lake in Manitoba, the 7th largest in Canada and the 12th largest lake in North America (Corkery, 1996) (Table 2.1). At its south end, Lake Winnipegosis drains into Lake Waterhen via the Little Waterhen and West Waterhen Rivers (Figure 2.1).

Waterhen Lake, with an area of 259 km^2 is the smallest of the uMBGL and is located north of Lake Manitoba and southeast of Lake Winnipegosis (Bajkov, 1930; Klein & Galbraith, 2017) (Figure 2.1). Waterhen Lake drains south, via the East Waterhen River, into the north basin of Lake Manitoba (Klein & Galbraith, 2017).

Lake Manitoba, with an area of 4,600 km^2 (McCullough, 2023), is the 3rd largest lake in Manitoba, 9th largest freshwater lake in Canada (Manitoba Fisheries, n.d.) and 14th largest lake in North America (Table 2.1). Lake Manitoba drains via the Fairford River through Pineimuta Lake into Lake St. Martin, before reaching its final destination in Anama Bay, Lake Winnipeg through the Dauphin River (Figure 2.1).

Projection WGS 84 (EPSG:4326)
Scale: 1:950,000

Sources: Watershed: Standard Drainage Area Classification (SDAC) 2003, Government of Canada Rivers: Lakes, Rivers and Glaciers in Canada - CanVec Series - Hydrographic Features

Figure 2.1. Map of the upper Manitoba Great Lakes

Table 2.1. Upper Manitoba Great Lakes (MBGL) morphometry and Canadian size rankings

Lake	Elevation (m)	Area (km²)	Length (km)	Width (km)	Maximum Depth (m)	Manitoba Ranking	Canadian Ranking
Winnipegosis	253.10	5,403.00	195.00	25.00	12.00	2.00	7.00
Manitoba	247.40	4,600.00	197.00	47.00		3.00	9.00
Waterhen	250.00	259.00	34.00	8.00	4.40		
Winnipeg	217.30	24,390.00	436.00	111.00	18.00	1.00	3.00

Last (1980) reported an elevation of 252.9 m for Lake Winnipegosis.

2.2 Geomorphology

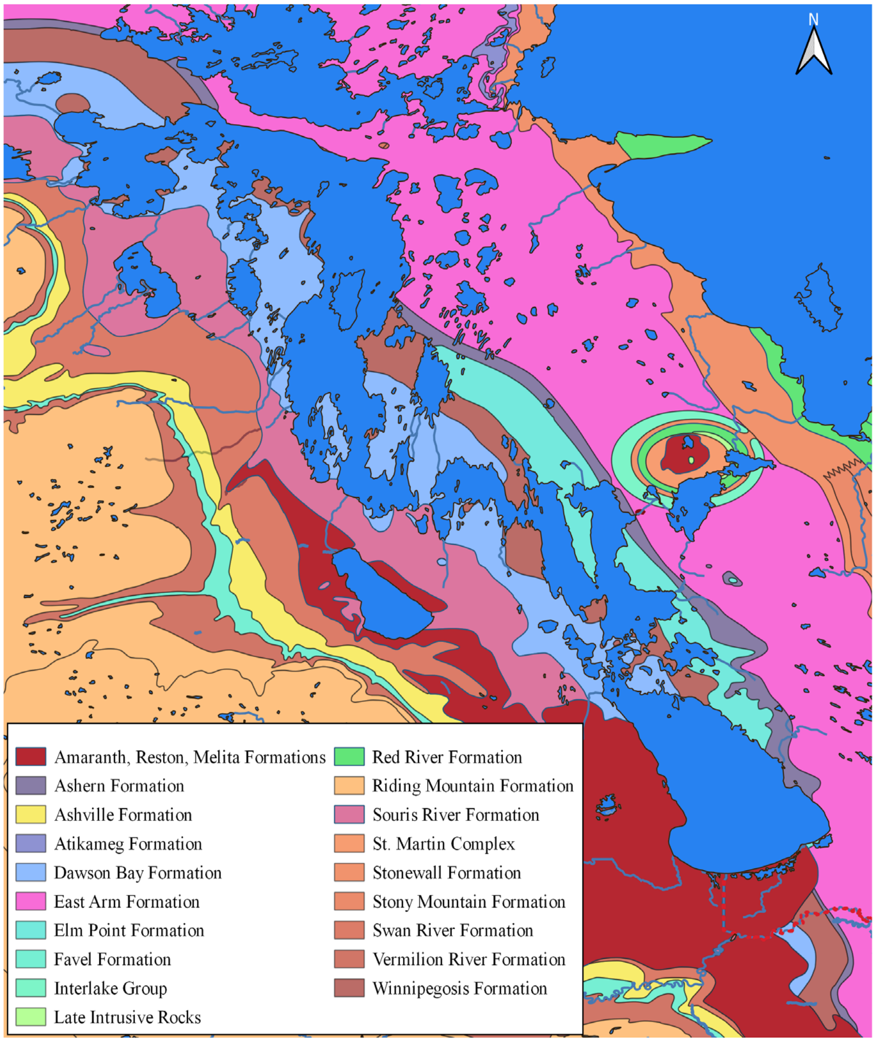
The Lake Winnipegosis and Lake Manitoba sub-basin is part of the Canadian Great Plains (CGP) geomorphic province of North America (F. M. Last & Last, 2012; W. M. Last & Ginn, 2005), the USA portion which was first described by Fenneman (1917). The CGP are part of the Great Plains, which are physiographic provinces based on landforms instead of climate or vegetation. The Great Plains encompasses ~3.6 million km^2, including 11 US states and 3 Canadian provinces. The CGP region of the Great Plains stretches ~ 450,000 km^2 from the Precambrian Shield east of Winnipeg, MB, west to the foothills of the Rocky Mountains. Commonly known as the prairie pothole region, the CGP is characterized by millions of mainly small (<100 km^2), shallow (z_{max} <10 m) and saline lakes, with some of the richest agricultural soils in the country and is characterized by flat to gently rolling topography (Last & Last, 2012). The actual number of lakes in this region varies depending on the minimum lake area included in the reporting and the hydroclimatic conditions at the time.

2.3 Geology

In the uMBGL, a thick sequence of glacial sediments deposited during the Pleistocene continental glaciation overlays the Paleozoic rock (Corkery, 1996), forming a thick layer of unconsolidated glacial, glaciofluvial and glaciolacustrine sediment (predominantly clay). This glacial deposition contributed to the formation of features such as The Pas end moraine, deposited at the edge of the glacial ice sheet, extending south from near The Pas, eastwards between Cedar Lake and Lake Winnipegosis, and forming Long Point in Lake Winnipeg (Figure 2.1) (Corkery, 1996).

Starting at the north end of Lake Winnipegosis and stretching south-east and along the eastern shores of the uMBGL to the south basin of Lake Manitoba, the upper layers of bedrock are comprised of shale and argillaceous dolomite (Ashern Formation) and fossiliferous and anhydritic dolomite of the Winnipegosis and East Arm (Interlake) formations (Figure 2.2). In the Lake Manitoba area, these formations were quarried as a source of limestone and aggregate (Last, 1980). Overlaying these sediments and descending from Overflowing Bay along the west shore of Lake Winnipegosis, terminating on the south-west side of Lake Manitoba, lies the Dawson Bay and Souris River formations, composed mainly of a shale carbonate sequence with minor evaporitic beds (Figure 2.2) (Last, 1980). The south-west basin of Lake Manitoba is underlain by the Jurassic Amaranth, Reston and Melita formations, which are not exposed to the surface (W. M. Last, 1980). These formations range from gypsum, anhydrite and shale (Amaranth) to sand, shale and minor carbonates, with minor amounts of carbonaceous shale and coal (Reston and Melita). The gypsum and anhydrite from the Amaranth formation have been mined for over 75 years at several locations around Lake Manitoba, including 20 km northeast of the Fairford River outlet (Figure 2.2) (Last, 1980). Overlying this is a thick sequence of siliceous, bentonitic and carbonaceous shales from the Cretaceous, with many of the formations exposed along the Manitoba Escarpment, a pre-glacial feature about 650 m thick, characterized by hard Odanah shale (Corkery, 1996), and lies along the eastern edge of Cretaceous rocks in Manitoba, creating a sharp break between the Manitoba Lowland Westlake/Interlake Plain and the Western Upland Plain (Last, 1980).

The Manitoba Lowlands (also called the Manitoba Lowland Saline Waterbelt) (Petch, 1990) are one of four physiographic regions in Manitoba. They extend from The Pas, along the western shore of Lake Winnipegosis and Lake Manitoba and include areas along the western bank of the Red River. They are bounded on the west by the Manitoba Escarpment (Corkery, 1996). The Manitoba Lowlands contain four sub-regions - the Interlake-Westlake Plain, the Red River Plain, the Lower Assiniboine Delta and the Southeast Section. To the south-east of Lake Manitoba, silty clay overlies bedrock, creating some of the most prosperous farming areas in Canada (Corkery, 1996). Pleistocene deposited silt and clay rich glacial tills as well as the glaciolacustrine sediment (mostly clay) form a low-permeability cap of varying thickness over the aquifer (Grasby & Betcher, 2002).



Upper Manitoba Great Lakes Lithology

Map Projection NAD83
EPSG: EPSG:4326
Scale: 820,000

Source: Manitoba Mineral Resources Division 1979,
Geological Map of Manitoba

Figure 2.2. Upper Manitoba Great Lakes Lithology

2.4 Hydrogeology

With the exception of the southwest portion of Lake Manitoba, the uMBGL is underlain by limestone and dolomite rocks deposited during several periods of the Paleozoic era (Corkery, 1996). As part of the Paleozoic outcrop belt in Manitoba, the uMBGL overlay a carbonate rock aquifer (Grasby, 2000), which is divided into fresh and saline portions along a north-south hydraulic divide (Grasby & Betcher, 2002). The aquifer is, in fact, a series of aquifers forming a continuous system, and in a series of west-dipping Middle Ordovician to Middle Devonian carbonates with minor shales and evaporites (Grasby & Betcher, 2002), which Grasby (2000) suggested occurred due to a hydrological head created by a significant influx of glacial meltwater during the Pleistocene. Follow up work by Grasby and Betcher (2002) found three dominant flow systems, caused by (i) a strong eastward gradient in the southwest related to the flow of saline waters from the Williston Basin; (*ii*) in the southeast, a northwest gradient related to the freshwater recharge from the Sandilands; and (*iii*) a westward and downdip flow of groundwater related to recharge in the Interlake. Salinity classified as per Hem (1985) along the aquifer, ranges from brine (>35,000 mg L^{-1}) to saline (10,000 – 35,000 mg L^{-1}) and brackish (2,000 – 10,000 mg L^{-1}) to freshwater (< 2000mg L^{-1}). Using the freshwater threshold of 2,000 mg L^{-1} as a cutoff, the carbonate aquifer demonstrates a clear freshwater to saline water boundary which closely follows the topographic lows defined by the Rat, Red, Saskatchewan and Assiniboine rivers, and lakes Manitoba and Winnipegosis (Grasby & Betcher, 2002), suggesting that glacial erosion of weaker geological beds created the current topography that defines the uMBGL morphology. The underlying carbonate geology supports the high carbonate values found in the uMBGL lakes compared to Lake Winnipeg, and the high saline groundwater input supports the saline characteristics and gradient that can be seen in the lakes from north to south (Bajkov, 1928; McKillop et al., 1992).

2.5 Lake Morphometry

The uMBGL were formed in a lowland basin consisting of limestone and shale bedrock which was scoured out by continental glaciers during the Pleistocene. 12,000 years ago, the basin was filled by glacial Lake Agassiz (Figure 2.3), which eventually drained, exposing the glacial lake bottom and leaving remnants, such as Lakes Winnipegosis and Manitoba, which appeared in their current form about 8,100 years ago (Teller et al., 1996). Additional large remnant lakes in the region include Cedar and Winnipeg (Welsted, Everitt, et al., 1996).

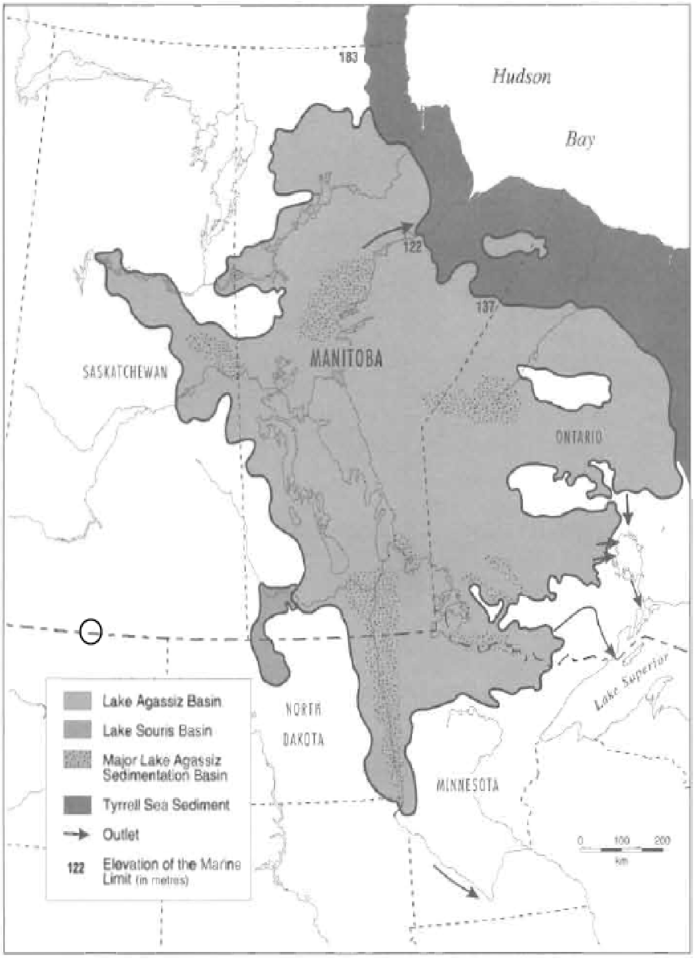


Figure 2.3. Map of Glacial Lake Agassiz
Note: Map from Corkery (1996).

Literature values for morphometric data on the uMBGL lakes can vary considerably with the source. Previous to the map created by McCullough (2023) for Lake Manitoba, Last (1980) provided one of the two documented methods for lake area calculations. Messager et.al. (2016) created the most comprehensive dataset for the uMBGL in the course of developing a database using a geo-statistical model to calculate morphological characteristics of freshwater lakes with a surface area of 10 ha or more.

Each lake in the uMBGL is here described using a series of common morphometric indicators to describe the physical properties of freshwater. Morphometric parameters allow comparison between lakes based on a set of standard metrics such as nutrient loading rates, thermal stability of the water column, and biological productivity. These parameters are vital to informing management techniques both at the lake and the watershed scale (Wetzel & Likens, 1991). General morphometric parameters described in this thesis can be divided into two categories - in terms of their physical and biological potential. Physical characteristics include lake length, width, depth, area and volume. Biological potential parameters discussed are shoreline development index (SDI), the watershed to lake area ratio (WA:LA), the dynamic sediment ratio (DR), and Carlson's trophic state index (TSI).

The uMBGL lakes are classified as continuous polymictic lakes as they may exhibit daily stratification but not stratification persisting through intervals of days to weeks. In comparison, Lake Winnipeg's south basin is generally considered continuous polymictic, but there is evidence the north basin may be discontinuous polymictic - stratifying at intervals of days (Environment and Climate Change Canada, 2020). Stratification patterns are important as they also inform the biology of a lake, for example, stratification persisting for days to weeks will affect the dispersion of biological organisms in the water column and their access to their energy sources differently than a lake which turns over completely daily (mixing surface to bottom waters) (Lewis Jr, 1983). Tables 2.3, 2.4 and 2.5 summarize the morphometric data for each lake.

The shoreline development index (SDI) is the ratio of shoreline length (S_L) relative to the length of the circumference of a circle whose area (A_0) is equal to that of the lake

$$SDI = \frac{\mathrm{S_L}}{2\sqrt{\pi A_0}} \qquad (2.1)$$

(Wetzel & Likens, 1991). The more complex the shoreline is, the more the SDI deviates from one, which is the value of a perfect circle. The SDI is used as an indicator of the extent of the littoral zone in a lake (the area of the lake presumed to be most productive); the higher the SDI value, the greater the potential for littoral zone development and the larger an area through which nutrient runoff from land may enter the lake (Page, 2011; Schwartz, 1982).

The watershed area to lake area ratio (WA:LA) is a morphological indicator used to provide an idea of the nutrient loading capability to a lake, as the larger the WA:LA ratio, the greater the amount of nutrients and sediments from river and stream inflows and runoff associated to the waterbody (Page, 2011).

The dynamic ratio (DR) model,

$$\mathrm{DR} = \frac{\sqrt{a}}{\bar{D}} \qquad (2.2)$$

as defined by Håkanson (1982), where a= lake area (km^2) and D = mean depth (m), is a measure used for lakes ranging from >1 km^2 to ~5000 km^2 to identify the percentage of a lake bottom area that is dominated by either fine particle (medium or finer silt) sediment erosion and transportation processes or sediment accumulation processes (Page, 2011). Håkanson (1982) divided the DR into 5 classes, with lakes with a DR of < 1.1 having less than 33% of areas of erosion and transportation, making the lake effectively a sediment trap, while lakes with DR > 3.8 are 100% dominated by resuspension activities. In shallow lakes, the resuspension of fine sediment particles can contribute to the water quality (e.g. water clarity, nutrient concentrations).

The Carlson TSI (CTSI) is a numerical trophic state index that uses multiple indices- Secchi disk depth (m), surface phosphorus (μg L^{-1}) and surface chlorophyll (μg L^{-1})

concentrations-as predictors of algal biomass, and ultimately as proxies to classify a lake's trophic state (Carlson, 1977), where:

$$\mathrm{TSI(SD)} = 10\left[6 - \frac{\ln \mathrm{SD}}{\ln 2}\right] \qquad (2.3)$$

$$\mathrm{TSI(Chl)} = 10\left[6 - \frac{2.04 - 0.68\ln \mathrm{Chl}}{\ln 2}\right] \qquad (2.4)$$

$$\mathrm{TSI(TP)} = 10\left(6 - \frac{\ln \frac{48}{\mathrm{TP}}}{\ln 2}\right) \qquad (2.5)$$

The TSI scale ranges from 0 to 100, with a mean TSI of <30 indicative of an oligotrophic lake, to a TSI of greater than 70 indicating a hypertrophic lake. Oligotrophic lakes typically have high water clarity, and primary and secondary productivity may be limited by a shortage of major nutrients, primarily nitrogen and phosphorus in temperate lakes (Environment et al., 2020; Khan & Ansari, 2005). As lakes become more productive, water clarity decreases and the amount of phosphorus and chlorophyll increases (Khan & Ansari, 2005; Schindler, 1974; Wetzel, 1975). Each indice can be plotted to compare seasonal or annual trends in lake trophic state between indices. The overall mean TSI for a lake is calculated by calculating the mean for all three indices. The eutrophication potential (EP) (Equation 2.6) of a lake is also calculated from the TSI. The EP is the potential of the lake, based on the current CTSI indice values, to stay eutrophic. The TSI and EP provides a simple empirical method to evaluate the potential for productivity of a lake and allows standardized comparisons across different waterbodies.

$$\text{Eutophication Potential (EP)} = \left(\frac{\text{Number of CTSI} > 50}{\text{Number of CTSI}}\right) \times 100\% \qquad (2.6)$$

2.6 Hydrology

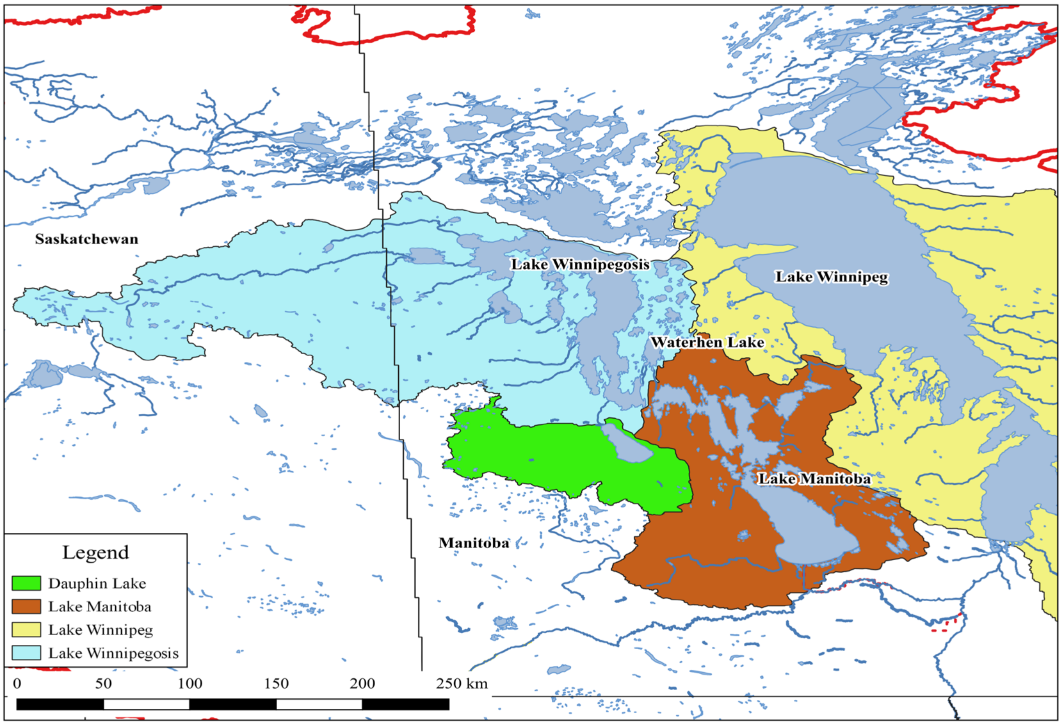
Lake morphological data is supplemented by and supports watershed modelling, which is critical in relating patterns in water quality to external and internal processes such as human activities and natural lake processes respectively (Benoy et al., 2016), thus enabling better support for science-based watershed management decisions (Wellen et al., 2015). Watersheds are natural hydrological units that capture all inputs and impacts (e.g. precipitation, runoff, groundwater, and stream flows), that affect the waterbodies contained within it. There are at least two major watershed classification systems in Canada that have been commonly used to delineate the hydrological networks and inform both physical and biological morphological data for freshwater systems. It is unclear from most works cited in this thesis which watershed classification system was used for any given dataset, therefore I summarize both below and in Table 2.2.

The National Scale Frameworks Hydrology-Drainage Areas classification is a compilation of the Water Survey of Canada and the National Atlas datasets designed to create a third dataset, the Fundamental Drainage Areas (FDA), which allow data to be aggregated to either of the two classification systems (Natural Resources Canada, 2009). This dataset allows analysis at the sub-sub-drainage basin level and higher (Statistics Canada, 2017). Under this classification, the uMBGL fall within the Lake Winnipegosis and Lake Manitoba drainage region, with an area of 82,719 km^2 (Figure 1.2, Table 2.2).

The Standard Drainage Area Classification (SDAC) and its variant - Drainage Area Classification (DAC) were created as a result of a partnership between Natural Resources Canada and Statistics Canada (Statistics Canada, 2018; Water Rangers, 2025) and were derived from the Water Survey of Canada's watershed boundary layer. The SDAC, created in 2003, provides the boundaries for the 11 major drainage, 164 sub-drainage and 974 sub-sub drainage areas in Canada, while the variant DAC can be built-up from the sub-sub-drainage areas of the SDAC. The SDAC and DAC were developed to provide more accurate boundaries and a refined watershed hierarchy which are used in reporting by Statistics Canada and allows a more standardized comparison with statistical data such as agricultural or population. Under this system, Lake Winnipegosis is contained within the Lake Winnipegosis watershed, and Lakes Manitoba and Waterhen are within the Lake Manitoba watershed (Table 2.2, Figure 2.4). To compare the SDAC watershed area to the FDA, the Lake Dauphin watershed should also be included, as it is in the Lake Winnipegosis-Lake Manitoba watershed (Table 2.2) FDA classification, creating a combined uMBGL watershed area of 82,860 km^2. Agriculture and Agri-Food Canada (AAFC) (Agriculture and Agri-Food Canada, 2016) used this SDAC classification for their AAFC Watershed Project, and the AAFC abbreviation is used to refer to the dataset in this thesis (Statistics Canada, 2018). When calculating watershed drainage areas, the AAFC effective drainage areas and non-contributing drainage areas should be considered (Figure 2.5). The effective drainage area is the area of the watershed that is expected to contribute to surface runoff under average runoff conditions, while the non-contributing drainage areas are areas that do not contribute to surface runoff during average runoff conditions and would only do so during years of high precipitation and discharge (Agriculture and Agri-Food Canada, 2013; Donald et al., 2015). Each of the three SDAC sub-watersheds contains regions of non-contribution drainage areas.

Table 2.2. Upper Manitoba Great Lakes watershed sizes according to the Agriculture and Agri-Food Canada (AAFC) SDAC and Water Survey of Canada (WSC) databases. The total area for the AAFC drainage basin is calculated by adding all three watershed areas together for a total drainage area of 82,860 km2.

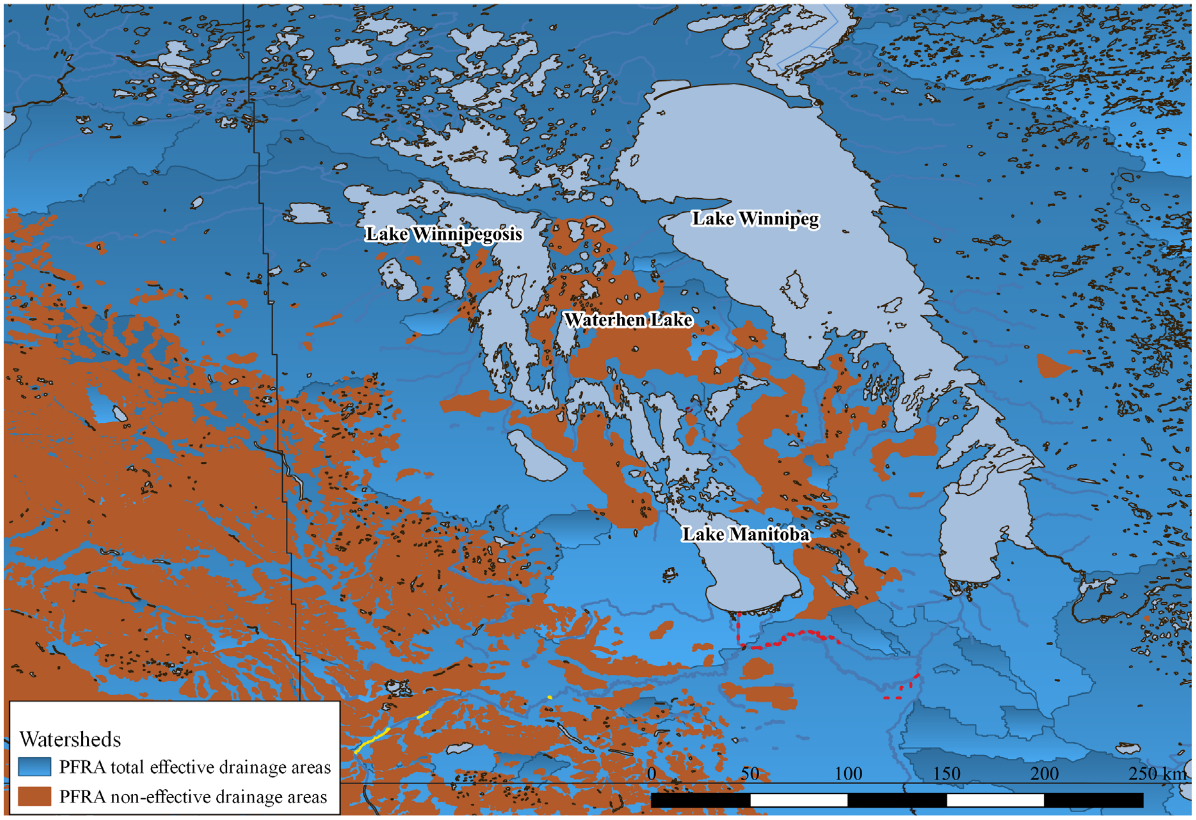
Lake	AAFC Watershed Area	WSC Watershed Layer	Lake Area	AAFC WA:LA	WSC WA:LA
Winnipegosis	46,355.1	82,746.6	5,403	9	15
Manitoba	27,795.2	82,746.6	4,600	6	18
Waterhen	Included in Winnipegosis watershed	82,746.6	2591	179	319
Dauphin	8,709.9	N/A	519.311	17	N/A



Scale 1:150,000

Source: Sub-basins of the AAFC watersheds project - 2013

Figure 2.4. Map of Agriculture and Agri-Food Canada (AAFC) sub-sub-districts



Projection WGS 84 (EPSG:4326)
Scale: 1:150,0000

Source: Areas of non-contributing drainage within total gross drainage areas of the AAFC watersheds project - 2013; Total effective drainage areas of the AAFC watersheds project - 2013; National hydrometric network basin polygons, Water Survey of Canada

Figure 2.5. Map of non-contributing drainage areas within total gross drainage areas of the AAFC watersheds project (2013)

2.6.1 Lake Winnipegosis

2.6.1.1 Physical.

Lake Winnipegosis, at an elevation of 253.1 metres above sea level (m.a.s.l), has an area of approximately 5,403 km^2 (Bajkov, 1930) (Table 2.1). The lake is elongated along a north-west to south-east axis, with a reported length of between 195-200 km long and 25-27 km wide (Herdendorf, 1984; Resources & Northern Development, 2022) and a shoreline length of 957 km (Herdendorf, 1984). Bajkov reported an average Secchi depth of 1.75 m. The Province of Manitoba recorded Secchi depths across the lake ranging from 0.78 - 4.1 m, similar to the

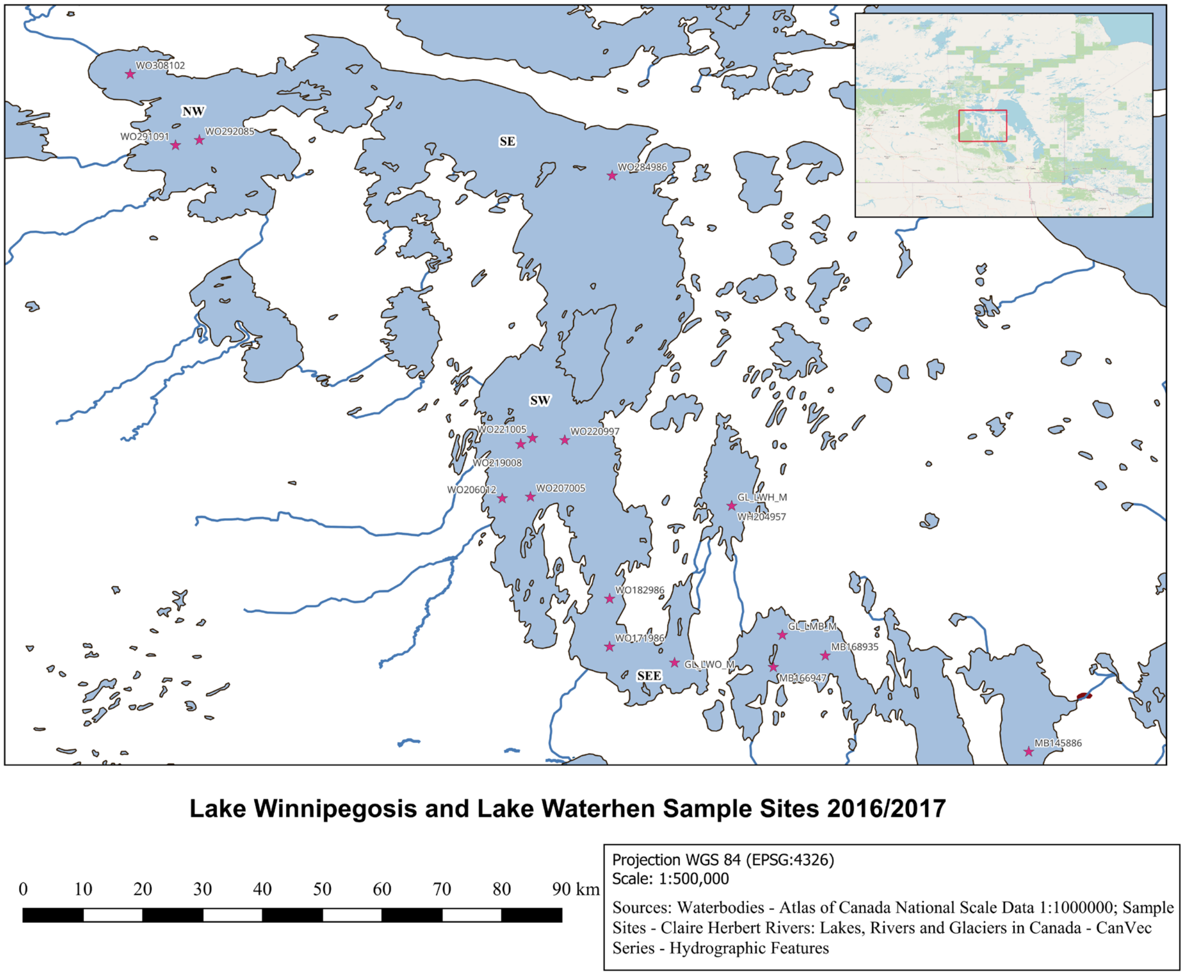
Coordinated Aquatic Monitoring Program (CAMP) values of 0.75-3.9 m (Coordinated Aquatic Monitoring Program, 2014) for the open-water season. For the purposes of this study, based on lake morphometry, I divided Lake Winnipegosis into four sub-basins: north-west (NW), south-east (SE), south-west (SW) and south-east east (SEE) (Figure 2.6).

Published values for Lake Winnipegosis range from a mean depth of 3 m in the south basin to a maximum depth of 18 m (Bajkov, 1930; Herdendorf, 1990; Resources & Northern Development, n.d.) in the north basin. In the south basin, Bajkov (1930) reported a maximum depth of 4 - 5 m (Table 2.3).

2.6.1.2 Biological.

There is no reported SDI value or WA:LA ratio for Lake Winnipegosis, therefore I calculated the WA:LA ratio from both the AAFC watershed sub-sub-basin and the WSC layers. The WA:LA for Lake Winnipegosis ranges from 9 to 15 depending on whether the ratio is calculated using the AAFC watershed layer (WA:LA = 9) or the WSC watershed layer (WA:LA = 15) (Table 2.2).

Herdendorf (1984) reported a DR value for Lake Winnipegosis of 3.7, indicating the lake is almost 100% dominated by resuspension. As the lake can be divided into basins, however, each basin should have the DR calculated separately to identify differences based on basin morphology.



Lake Winnipegosis and Lake Waterhen Sample Sites 2016/2017

Projection WGS 84 (EPSG:4326)
Scale: 1:500,000

Sources: Waterbodies - Atlas of Canada National Scale Data 1:1000000; Sample Sites - Claire Herbert Rivers: Lakes, Rivers and Glaciers in Canada - CanVec Series - Hydrographic Features

Figure 2.6. Lake Winnipegosis, showing sample sites and basin divisions (NW, SE, SW, SEE)

Table 2.3. Lake Winnipegosis morphometric data

Units	Whole Lake	Herbert
Watershed Area (Km^2)	54,000[1]	46,355[2]
Lake Area (Km^2)	5403[3]	5401[4]
Maximum Length (Km^2)	200[5]	
Maximum Width (Km)	25[5]	
Shoreline Length (Km)	957[6]	
Wetlands/km shoreline (Km^2)	0.800	
Mean Depth (m)		7.0
Max Depth (m)	12[5]	9.7
Depth Ratio (z:zm)	-	0.73
DR	-	10
Shoreline Development Index		3.7[6]
Watershed:Lake Area Ratio (WA:LA)	-	9
Estimated Volume (Km^3)	16[6]	

Wetlands/km shoreline from Watchorn et.al. (2012). Values from Herbert table were used for calculations in this thesis and unless otherwise cited were generated from this research. Lake statistics are for the entire lake as most statistics per basin do not exist, with the exception of Bajkov (1930b) who reported a south basin mean depth of 4-5 m. [1]Last(1980), [2]AAFC(2013),[3]Bajkov(1930),[4]Atlas of Canada(2013),[5]Klein&Galbraith(2017), [6]Herdendorf(1984)

2.6.2 Lake Waterhen

2.6.2.1 Physical.

Waterhen Lake is located east and downstream of Lake Winnipegosis and northwest and upstream of Lake Manitoba (Figure 2.1). At 34 km long and 8 km wide (Galbraith et al., 2017), it is short and narrow compared to the other two lakes in this study. It is also very shallow, with a maximum depth of 4.4. m (Galbraith et al., 2017) (Table 2.1). Messager et al. (2016) calculated the area of Waterhen Lake to be 160 km^2, which is half the area reported by Bajkov (1930) of 259 km^2.

2.6.2.2 Biological.

Messager (2016) reported an SDI of 5.8, indicating the lake has the potential to be highly productive (Table 2.3).

There is no reported DR or WA:LA for Lake Waterhen so I calculated them based on data from (Table 2.3). The DR for Lake Waterhen was 4.2, indicating that Lake Waterhen has the potential for 100% resuspension of sediment. Using both watershed layer data from (Table 2.2), I calculated the WA:LA to vary from 179 (Agriculture and Agri-Food Canada, 2013) to 319.

Table 2.4. Lake Waterhen morphometric data

Units	Lake
Watershed Area (Km²)	46,355[1]
Lake Area (Km²)	160[2]
Maximum Length (Km²)	34[3]
Maximum Width (Km)	8[3]
Shoreline Length (Km)	262[2]
Wetlands/km shoreline5 (Km2)	n.d.
Mean Depth (Km)	3.8
Max Depth (m)	4.4[3]
Depth Ratio (z:zm)	
DR	4.2
Shoreline Development Index	5.8
Watershed:Lake Area Ratio	289
Estimated Volume (Km³)	0.43[2]

[1] Statistics Canada (2018)

[2] Messager et.al. (2016)

[3] Klein&Galbraith(2017)

2.6.3 Lake Manitoba

Lake Manitoba is located between Dauphin Lake and Lake Winnipeg, to the southeast of Lakes Waterhen and Winnipegosis (Figure 2.1). Though the most commonly reported surface area of Lake Manitoba is 4,700 km^2, McCullough (2023), using a digitized bathymetric map and satellite data (McCullough, 2023), calculated the area of the lake to be 4,596 km^2, not including Ebb and Flow Lake, similar to that reported by the International Joint Commission (1977), Bajkov (1930) and Donald et al. (2015) (Table 2.5). Depending on wind direction, Ebb and Flow Lake exchanges water with the south basin of Lake Manitoba at the west side of the Narrows but may not be included in all area calculations for Lake Manitoba, accounting for some of the differences in reported lake area. For example, Messager et al. (2016) calculated the area of Lake Manitoba to be 4,751 km^2, including Ebb and Flow Lake. In this thesis, a rounded area of 4,600 km^2 will be used. Reports on the lake's length range from 197 km (Page, 2011) to 225 km (Lake Manitoba Regulation Review Advisory Committee, 2003). At its widest Lake Manitoba is between 40 (Lake Manitoba Regulation Review Advisory Committee, 2003) and 47 km (Last, 1980) making it the 3rd largest lake in Manitoba, 9th in Canada and 14th in North America (Table 2.1). For the purposes of this study, I will use a length of 197 km and maximum width of 47 km, as reported by Page (2011) (Table 2.5).

The north basin of Lake Manitoba has an area of 1,808 km^2 (McCullough, 2023), accounts for 39% (McCullough, 2023) of the total lake surface area, and is made up of a very irregular shoreline dominated by numerous, bedrock-controlled features such as islands, peninsulas and straits (Last, 1980). The north basin is 92 km long and 22 km wide, with a total shoreline length of 523 km (Lake Manitoba Regulation Review Advisory Committee, 2003; Page, 2011) and runs along a NW-SE axis (Last & Teller, 2002). Page (2011) and McCullough

(2023) divided the north basin into 3 smaller elongate sub-sub-basins, connected to the south basin by a narrow strait southwest of Dolly Bay, referred to as “the Narrows” (Figure 2.1), which is 4 km long and 0.8 - 1 km wide (Last & Teller, 2002; Page, 2011). The three northern sub-basins are divided from each other by narrow straits created by land features, notably Peonan Point between the south-east portion of the north-west basin (SE) and the west-north-west basin (NW), collectively called the northwest basin with the third basin occurring to the northeast (NE) (Table 2.5, Figure 2.1) (McCullough, 2023). For this study, I used these same divisions for the north basin and considered the south basin as a single basin (Figure 2.7).

In contrast, the south basin of Lake Manitoba is slightly trough-shaped, with steeply sloping sides and a flat, featureless bottom, characterized by a relatively smooth, regular, 392 km shoreline with fewer islands than the northern basins (Last, 1980), that gradually converges north towards the Narrows (Page, 2011). The south basin is 105 km long, 47 km wide and has a shoreline length of 392 km (Last, 1980). The surface area of the south basin is reported by Page and Last as 3,107 km^2 (70% of the total lake surface area), however, McCullough (2023) reported the south basin surface area as 2,792 km^2 (61% of the total lake surface area) which is the area that will be used in this study (Table 2.5).

Page reported an SDI of 1.98 for the south basin and 3.70 for the north basin. As indicated by the higher SDI, the north basin of Lake Manitoba has a larger littoral zone, and therefore a greater potential for biological productivity than many other Great Lakes in Canada. Downing & Duarte (2009) and Herdendorf (1984), however, used a different shoreline length for Lake Manitoba (810 km) to calculate SDI, and reported values for Lake Winnipeg of 2.5 and for Lake Manitoba of 3.4 which, while lower than Page’s reported values still indicates that Lake Manitoba has a greater potential for biological productivity than many large lakes in Canada.

Page (2011) reported a WA:LA ratio of 40 for the north basin of Lake Manitoba, and a WA:LA ratio of 6 for the south basin. When partial flow of the Assiniboine River is diverted into the south basin of Lake Manitoba during floods (via the Assiniboine River Diversion (ARD)), the WA:LA increases to 19.

Table 2.5. Lake Manitoba morphometric data

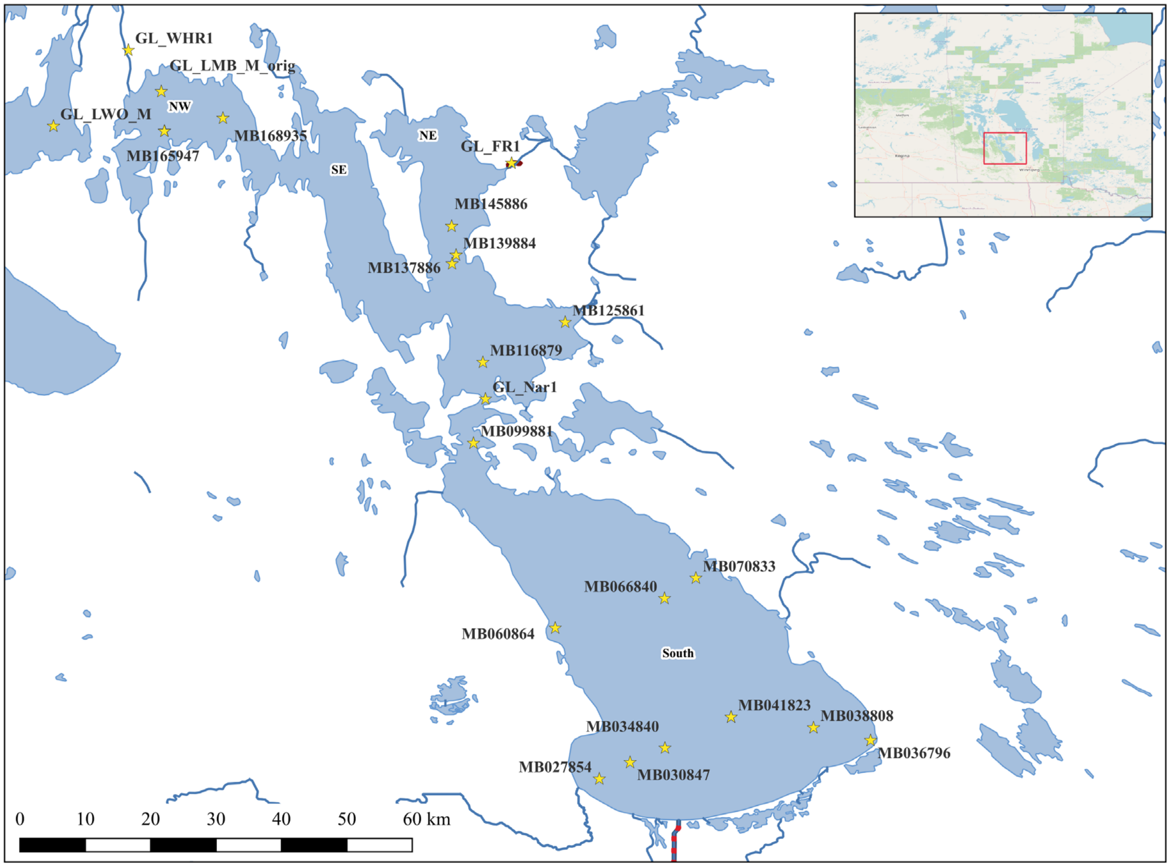
Units	South Basin	North Basin	South basin[2]	NE basin[2]	NW-basin[2]	Whole Lake
Watershed Area (Km^2)	17,343	62,956				80,299
Watershed Area (with ARD) (Km^2)	58,843	62,956				121,799
Lake Area (Km^2)	3,107[1]	1,593[1]	2,792	812	992	4,596[2]
Maximum Length (Km)	105[1]	92[1]				197
Maximum Width (Km)	47[1]	22[1]				47
Shoreline Length (Km)	392[1]	523[1]				915
Wetlands/km shoreline5 (Km^2)						0.616
Mean Depth (m)	5[1]	4.5	4.1	3.1	2.3	3.5[2]
Max Depth (m)	7.0	7.0	6.5	6.3	6.9	6.9[2]
Depth Ratio (z:zm)		0.71	0.64			
Dynamic Ratio (DR)		11.1	8.9			
Shoreline Development Index (SDI)		1.98	3.70			
Watershed:Lake Area Ratio		6[1]	40			17[1]
Watershed:Lake Area Ratio (with ARD)		19				
Estimated Volume (Km^3)	15.5	7.2	11.5	2.5	2.3	16.3

The NW-basin is a combination of the NW-basin (NW & SE sub-basins) from McCullough (2023).

Except where noted, values are from Page (2011).

[1]Data are from Last (1980).

[2]Data are from McCullough (2023)



Lake Manitoba sample sites 2016/2017

Projection WGS 84 (EPSG:4326) Scale: 1:500,000	Sources: Waterbodies - Atlas of Canada National Scale Data 1:1000000; Sample sites - Claire Herbert Rivers: Lakes, Rivers and Glaciers in Canada - CanVec Series - Hydrographic Features

Figure 2.7. Lake Manitoba sample sites 2016-2017 and basin divisions (NW, SE, NE, South)

2.6.4 Wetland Classification

Watchorn et al. (2012) created a comprehensive database of coastal wetlands for Lakes Winnipegosis, Manitoba and Winnipeg using the Manitoba Forest Resource Inventory (FRI) cover types for wetlands. Wetlands consisted of three main wetland classes - barrier-protected (BAR), lacustrine (LAC) or riverine (RIV) which were further divided into subclasses. Broadly speaking, a barrier-protected wetland is a wetland where a barrier bar separates it from the adjacent lake, a lacustrine wetland is a wetland confined to the high and low shoreline and littoral zones, and a riverine wetland is a wetland located at the mouth of a river.

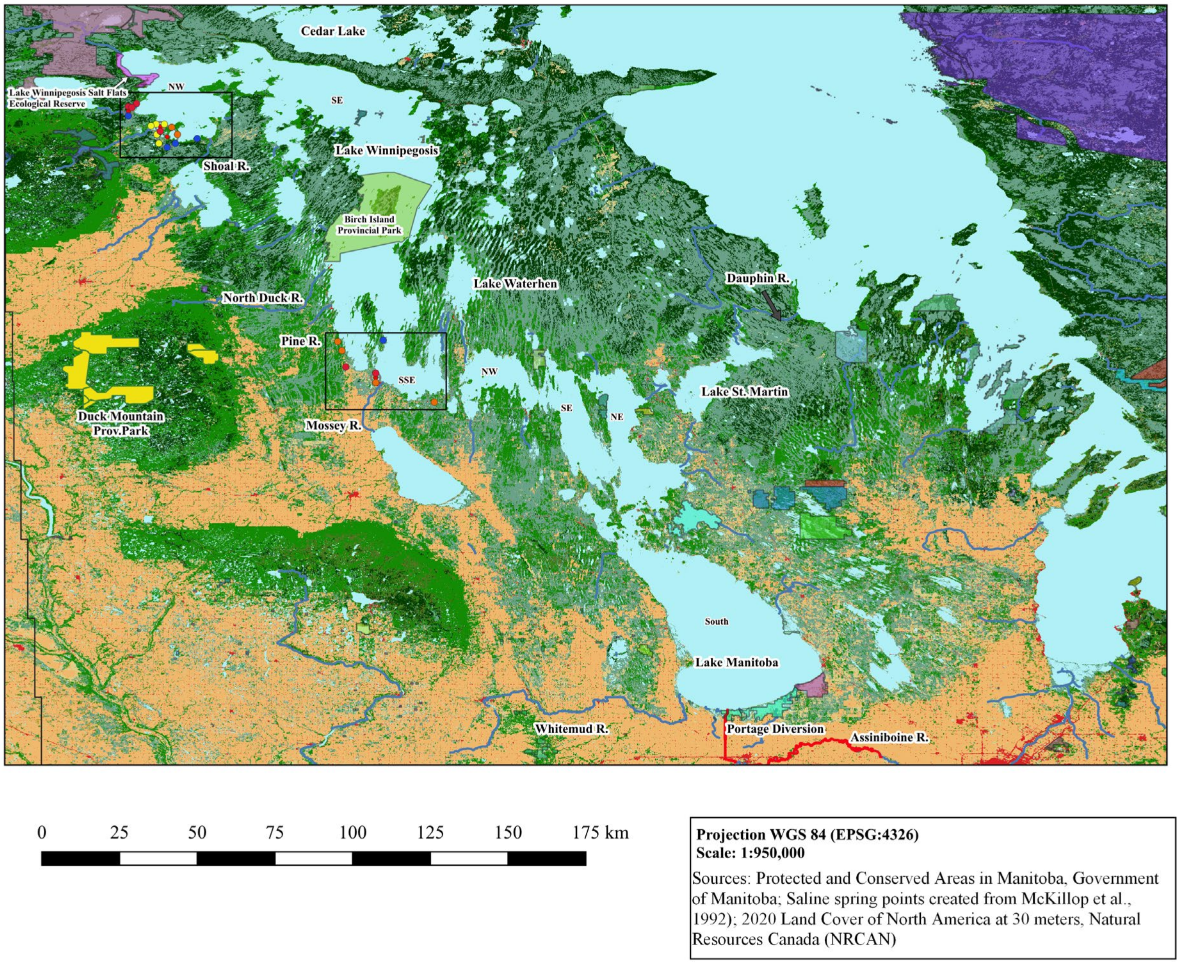
2.6.4.1 Lake Winnipegosis

The uMBGL watershed drains an area of 82,746 km^2 (Table 2.2) in western Manitoba and eastern Saskatchewan, with its headwaters originating from the western part of the Manitoba Escarpment of Riding Mountain, Duck Mountain and Porcupine Mountain and the Pasquia hills in eastern Saskatchewan (Corkery, 1996; Intermountain Conservation District & Province of Manitoba, n.d.; Page, 2011), with drainage generally occurring from west to east. Nested within the uMBGL watershed is the Lake Winnipegosis watershed, which contains Lakes Winnipegosis and Waterhen and covers an area of 46,355 km^2. The headwaters of the uMBGL flow into the north-west basin of Lake Winnipegosis via the Overflowing, Red Deer, Steeprock and Shoal Rivers, into the southwest basin from the North Duck and Pine Rivers, and into the southeast-east (SEE) basin via the Mossy River, before ultimately exiting from the SEE basin into Lake Waterhen through the Waterhen River. Additional smaller tributaries feeding into Lake Winnipegosis include Pelican, Sclater and Point Rivers (Figure 2.1) (Resources & Northern Development, 2022). The SE and SW basins are connected via two narrow sections of the lake, which are divided in half by Birch Island Provincial Park (Figure 2.8). The SW and SEE basins are connected via a narrower portion of the lake on the east side, between Red Deer point and Salt Point (Figure 2.1). Located within Duck Mountain Provincial Forest and Park are several small dams (Shanty, Pine River and Beaver Lake), which were constructed by the Prairie Farm

Rehabilitation Administration (PFRA) between 1956 and 1962 to help manage downstream flooding and drought conditions (Intermountain Conservation District & Province of Manitoba, n.d.). Lake Winnipegosis' SW basin is dotted on the west and east sides by areas of non-contributing drainage, which may contribute to the hydrologic budget during high water events or seasons (Figure 2.5).

Surrounding Lake Winnipegosis are 72, 242 ha of coastal wetland, equal to 27% of the total coastal wetland area of 270,994 ha (Watchorn et al., 2012). Lake Winnipegosis wetlands are dominated by riverine type (26%), with the Open Drowned River-Mouth dominated by Wet Meadow (81%). LAC (Mudflats/Saltflats, Wet Meadow and Marsh) and BAR (Mudflats/Saltflats and Willow) making up equally (20% and 22%) the remainder of the coastal wetland areas.

The carbonate rock aquifer is part of the Manitoba Lowland Saline Water belt (Teller, 1984), dividing the saline and brackish waters to the west of Lakes Winnipegosis and Manitoba from the freshwater regions. Dotted along the west side of Lake Winnipegosis are a series of freshwater, brackish, hypo- and hypersaline seeps, springs and marshes which periodically feed into the west side of Lake Winnipegosis (McKillop et al., 1992). First described by La Vérendrye around 1741 (McKillop et al., 1992; Petch, 1990), these springs have been utilized since at least AD 1100 (Figure 2.8) for salt making.



Projection WGS 84 (EPSG:4326)
Scale: 1:950,000
Sources: Protected and Conserved Areas in Manitoba, Government of Manitoba; Saline spring points created from McKillop et al., 1992); 2020 Land Cover of North America at 30 meters, Natural Resources Canada (NRCAN)

Figure 2.8. Upper Manitoba Great Lakes, showing McKillop historical saline sample sites and provincial parks

2.6.4.2 Lake Waterhen

Lake Winnipegosis drains into Waterhen Lake through the Little Waterhen and West Waterhen rivers, flowing into Lake Manitoba via the East Waterhen River (Galbraith et al., 2017) (Figure 2.1). Lake Waterhen is surrounded on the west, north and east sides by areas of non-contributing drainage, and therefore may not receive much runoff from the land surrounding it, except in high water years or during high precipitation events (Figure 2.5).

Lehner et al. (2025) classified wetlands around Lake Waterhen as a mixture of palustrine and seasonally saturated to regularly flooded. Lake Waterhen wetlands were not classified specifically by Watchorn (2012).

Lake Waterhen lies along the east side of the carbonate aquifer, along the 2000 mg L^{-1} isocon, directly along the saline-freshwater divide (Grasby & Betcher, 2002).

2.6.4.3 Lake Manitoba

Watchorn et al. (2012) identified 56, 365 ha of coastal wetlands surrounding Lake Manitoba, equating to 21% of total coastal wetlands for the Manitoba Great Lakes. BAR comprises 41% (mainly Barrier-Protected Beach Lagoons) and includes Delta Marsh, located along the south shore of the south basin. LAC, dominated by the subclass Open and Protected embayments makes up 31%, and RIV, dominated by Open Drowned River-Mouths made up the remaining 29%.

Due to the differences in lake morphology and major drainage processes, the north and south basins of Lake Manitoba are considered two distinct hydrological systems (Page, 2011). Water is transported from Lake Winnipegosis into Lake Waterhen via the ungauged West and Little Waterhen Rivers. The outflow of Lake Waterhen into Lake Manitoba is the East Waterhen River, providing about 42% of the total inflow into Lake Manitoba, with precipitation accounting for an additional 40% of input to the lake surface (Last, 1983). The small, irregularly shaped north basin flows through to the larger and less articulated south basin, with an average residence time of two years (Last, 1983; Page, 2011). In the north basin, two coastal wetlands of significance are in the Proulx Lake and Basket Creek areas (Page, 2011) (Figure 2.1). In the south basin, Ebb and Flow Lake, west of the narrows, exchanges water with Lake Manitoba during larger setup and seiche events.

Inflow from the Waterhen River largely bypasses the south basin of Lake Manitoba, leaving the latter's budget dominated mainly by evaporation (up to 60%). The Whitemud River at the southwest end of the south basin contributes approximately 6% of the flow into the south basin (Page, 2011). In years with large inflows from the ARD, however, the ARD may dominate the inflow into Lake Manitoba's south basin (Last, 1983; Page, 2011).

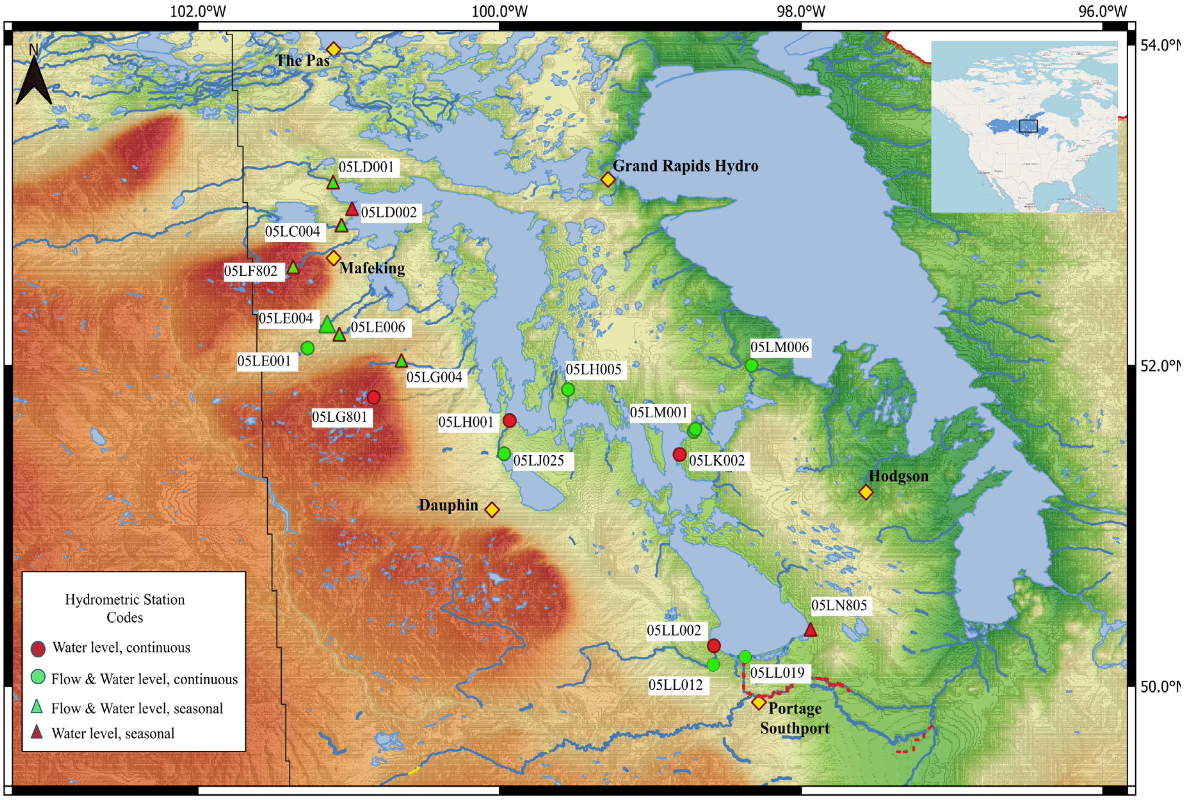
2.6.5 Water Flows

2.6.5.1 Lakes Winnipegosis and Waterhen

The Water Survey of Canada currently records seasonal lake flow and level at the inlets for Lake Winnipegosis at the Overflowing, Red Deer, and North Duck rivers, and continuous flow and water level at the Mossy River (Figure 2.9). During the study years 2016-2017, the highest mean annual discharge (MAD) is from the Red Deer River (RDR), with a maximum daily discharge of 174 m s^{-1} in October 2016, six times higher than the MAD of 28.0 m^3 s^{-1} recorded for the 68 year period from 1956-2024 (Figure 2.10). Both study years also exceeded the 90th percentile for flow.

Donald et al. (2015) calculated a mean water residence time of 3.29 years and a mean annual outflow of 154 m^3 s^{-1} for Lake Winnipegosis from 2008-2011, while Lehner et al. (2025), using the HydroLAKES database model data (Messager et al., 2016) reported a mean residence time of 6.6 years. The HydroLAKES dataset calculated discharge for the period 1971-2000, therefore better representing a long-term average.

Welsted et al. (1996) reported a discharge of 77.1 m^3 s^{-1} at the Waterhen River, while between 1950-2024, the Water Survey of Canada data showed a MAD of 91.7 m^3 s^{-1} (Environment & Climate Change Canada, n.d.). Both study years contained the second and ninth highest flows for the entire record (Figure 2.10).



Environment and Climate Change Canada (ECCC) weather stations with climate normal data 1991-2020. Water Survey of Canada (WSC) hydrometric station locations.

Map Projection WGS 84
EPSG:4326

ECCC Weather Stations and WSC hydrometric Stns

Scale: 1:1,300,000

Layers: Digital Elevation Model of Canada - Canada3D, 2001 Government of Canada; Hydrometric stations derived from Water Survey of Canada hydrometric data; weather station data from ECCC historical weather station data

Figure 2.9. Upper MBGL, showing Water Survey of Canada hydrological stations and Environment and Climate Change Canada weather stations

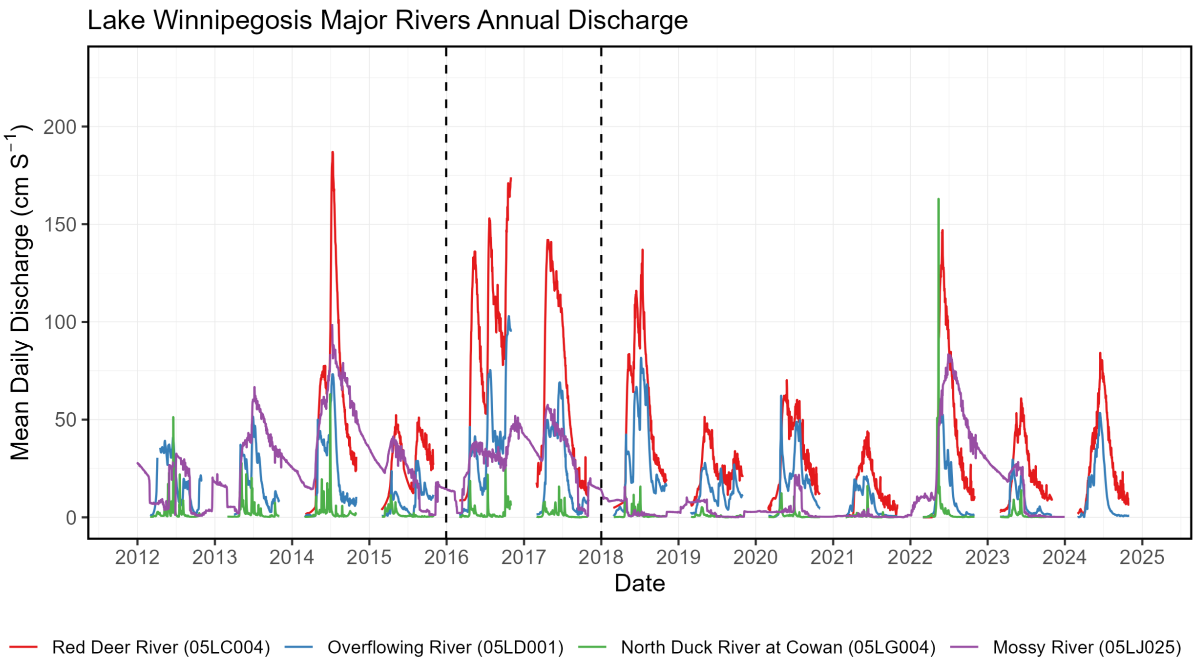


Figure 2.10. Lake Winnipegosis Major Rivers annual discharge, 2012-2024

2.6.5.2 Lake Manitoba

The major inflow into the north basin of Lake Manitoba is from Lake Winnipegosis via the Waterhen River, providing 79% of the average annual river inflow (Page 2011), with mean daily average flows ranging between 69.7 and 489.0 m^3 s^{-1} from 1959 - 2024, and a mean annual discharge of 91.7 m^3 s^{-1} for the same 65-year period (Environment & Climate Change Canada, n.d.). The annual median discharge (MAD) from the Waterhen River in the study years was 161 and 229 m^3 s^{-1} respectively, 2.5 times higher than the 65-year MAD (Environment & Climate Change Canada, n.d.). The remaining inflows enter the north basin of Lake Manitoba from various ungauged streams, drains, groundwater inputs (Page, 2011), and, when the wind is blowing from the north or west, Ebb and Flow Lake, located to the west of Lake Manitoba (Figure 2.1). In years the Assiniboine River diversion is not operating, flows from the Waterhen River can account for up to 90% of the inflow into the north basin (Page, 2011).

In the south basin, the main inflow is the Whitemud River (6% of annual flow) (Page, 2011), located on the west side, near Lynch's Point (Figure 2.1), which had average flows ranging between 0 and 314.0 $m^3 s^{-1}$ for the period 1973-2023. The MAD for the 50-year period was 7.2 $m^3 s^{-1}$, while flows for 2016 and 2017 were up to six times higher at 46 and 29 $m^3 s^{-1}$. East of the Whitemud is the Assiniboine River (Portage) Diversion (ARD), a man-made 29 km long channel designed to divert water from the Assiniboine River upstream of Portage la Prairie, into Lake Manitoba during high water events (International Joint Commission, 1977). When the ARD is operating, it provides an additional 11 percent of the average annual inflow into the south basin (Page, 2011). Since its operation in 1975, flows on the ARD have ranged from 0 to 1,000 $m^3 s^{-1}$. The ARD operated during the 2017 study year, with a maximum flow of 702 $m^3 s^{-1}$ on April 12, almost 100 times higher than the average annual discharge from the Whitemud River (Figure 2.11) (Environment & Climate Change Canada, n.d.).

The Fairford River is the only outlet from Lake Manitoba to Lake Winnipeg, via Lake St. Martin (Figure 2.1). Since 1933, the Fairford River has had its water levels controlled, and, since the current structure and channel changes in 1961, a minimum flow of 23 $m^3 s^{-1}$ is required. Flows were recorded intermittently from 1912, with continuous monitoring beginning in 1955 (Environment & Climate Change Canada, n.d.). Between 1955-2024, mean flows ranged from 1.7 to 415 $m^3 s^{-1}$, with flows of 170 and 269 $m^3 s^{-1}$ in 2016 and 2017, and the peak flow in 2016 occurring in November. The 2017 mean flow was twice the MAD of 99 $m^3 s^{-1}$ (Figure 2.12) (Environment & Climate Change Canada, n.d.).

The Dauphin River carries the uMBGL water from Lake St. Martin to Lake Winnipeg and follows a flow pattern similar to the Fairford River, with mean average flows ranging from 16 to 391 $m^3 s^{-1}$. The mean average flows in 2016-2017 were 174 and 253 m3 s^{-1} respectively, twice the MAD of 105 $m^3 s^{-1}$ for the 46-year period from 1977-2023 (Figure 2.12), but similar to the outflow from the Fairford River.

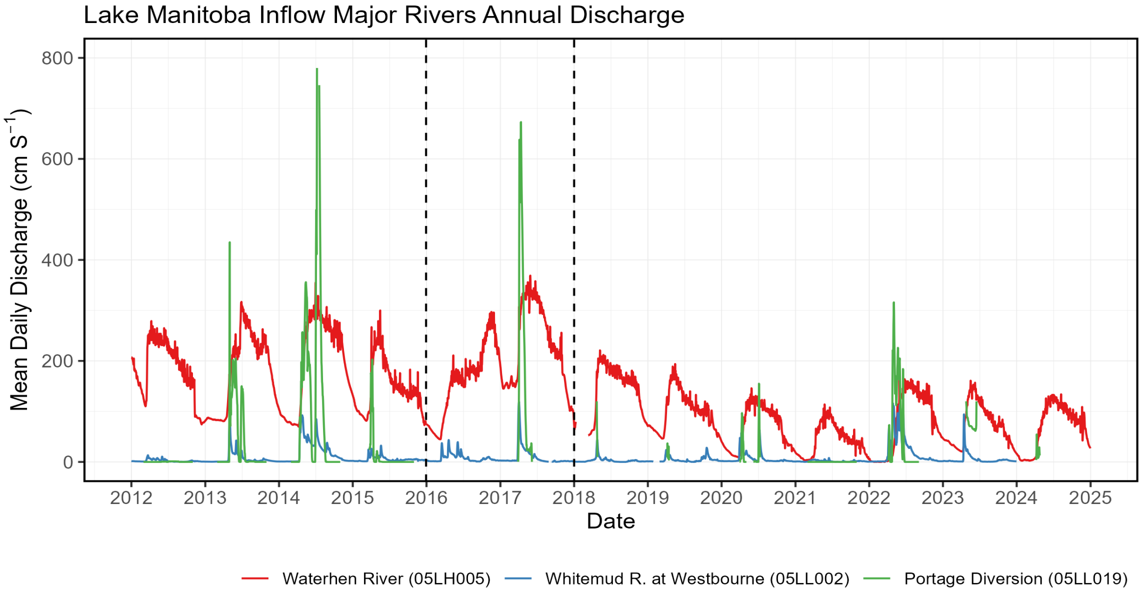


Figure 2.11. Lake Manitoba Major River outflows

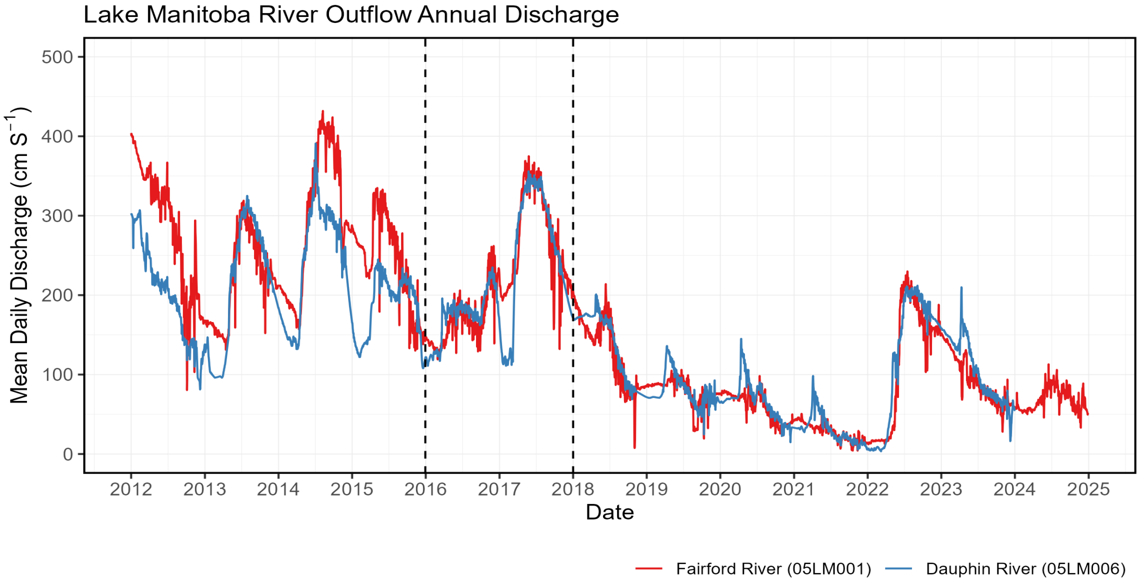


Figure 2.12. Fairford and Dauphin Rivers Annual Discharge 2012-2024

2.7 Climate

Manitoba's climate is controlled mainly by its location as the geographic centre of North America, or its "continentality". The continentality of Manitoba enhances the already strong latitudinally controlled temperature fluctuations within the province, characterized by very low temperatures in the winter and very high temperatures in the summer (Welsted, Stadel, et al., 1996). Manitoba is a dry province, with the south-eastern edge receiving an annual amount of about 600 mm precipitation and most areas receiving less than 525 mm. Two-thirds of the precipitation falls as rain from May-September.

The uMBGL are divided into two ecozones (Boreal and Plains) and three ecoregions (Mid-Boreal Lowland, Interlake Plain and Lake Manitoba Plain) (Figure 2.13). The north and central basins of Lake Winnipegosis to the western shore of Lake Winnipeg are part of the Mid-Boreal Lowland ecoregion (Boreal Plains ecozone) (Smith, 1998). The climate in the Boreal Plains is strongly influenced by continental climatic conditions, is sub-humid and has typically cold winters and moderately warm summers (Smith, 1998). Of the Environment and Climate Change Canada (ECCC) weather stations that provide climate normal data, the closest in the Lake Winnipegosis northern region is The Pas. The maximum daily average temperature for 1991-2020 was 18.4 °C and ranged between -23.4 °C to 23.9 °C. Average maximum precipitation was 80.1 mm (Meteorological Service of Canada, n.d.) (Table 2.6).

The central and southern basins of Lake Winnipegosis, Lake Waterhen and the northern and upper eastern side of the southern basin of Lake Manitoba are part of the Interlake Plain Ecoregion (Boreal Plains Ecozone). This ecoregion stretches in a broad arc from the USA-Canada border at the southeastern edge of the Manitoba Plain, northwest across the southern Interlake-Westlake region to the Saskatchewan border at Red Deer Lake (Smith, 1998). The region is characterized by short, moderately warm summers and long, cold winters. The closest Environment and Climate Change Canada climate normal weather station is at Mafeking, to the west of Swan Lake (Meteorological Service of Canada, n.d.). Climate normals for 1991-2020 reached a maximum daily average temperature of 18.7 °C and ranged between -21.2 °C and 24.8 °C. The average maximum monthly precipitation was 105.8 mm (Meteorological Service of Canada, n.d.) (Table 2.6).

The west and southeast portion of Lake Manitoba is part of the Lake Manitoba Plain ecoregion (Prairies Ecozone). This ecoregion stretches northwest from the International Boundary to Lake Dauphin, with the Manitoba Escarpment marking its western boundary (Figures 2.13, 2.14). This ecoregion is part of the Grassland transition ecoclimatic region and is characterized by short, warm summers and long, cold winters (Canada et al., 2011). The closest Environment and Climate Change Canada climate normal weather station is located at Dauphin (Meteorological Service of Canada, n.d.). Climate normals for 1991-2020 reached an average maximum daily temperature of 18.8 °C and ranged from -20.9 °C to 25.1 °C. The average maximum monthly precipitation was 86.6 mm (Table 2.6) (Meteorological Service of Canada, n.d.). Along the east side of Lake Manitoba, closer to Lake Winnipeg in the Interlake ecoregion is Hodgson station. Climate normals reached a maximum daily temperature of 18.3 °C and ranged from -24.3 °C to 25.2 °C. The maximum monthly precipitation was 85.2 mm (Meteorological Service of Canada, n.d.) (Table 2.6). At the south end of Lake Manitoba, the Portage Southport climate station recorded an average daily temperature of 19.8 °C, with a range between -20.9 °C and 25.4 °C. The average maximum monthly precipitation in this region was 80.4 mm (Meteorological Service of Canada, n.d.) (Table 2.6).

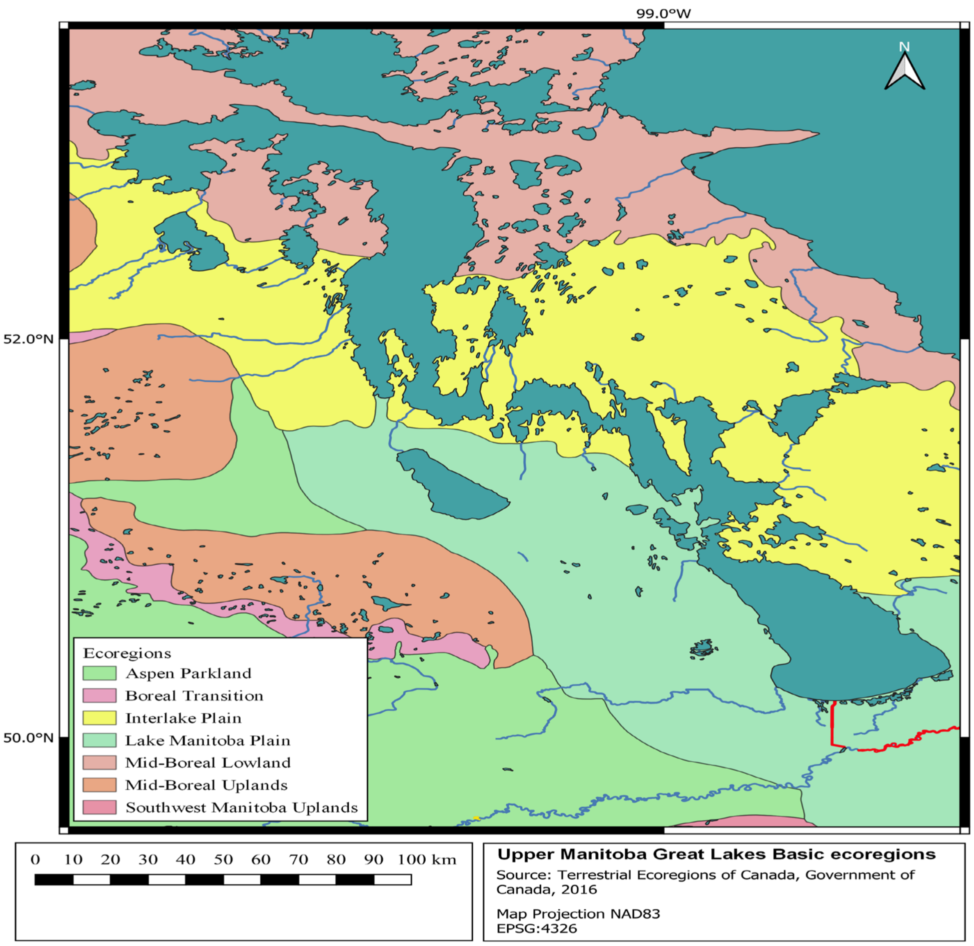
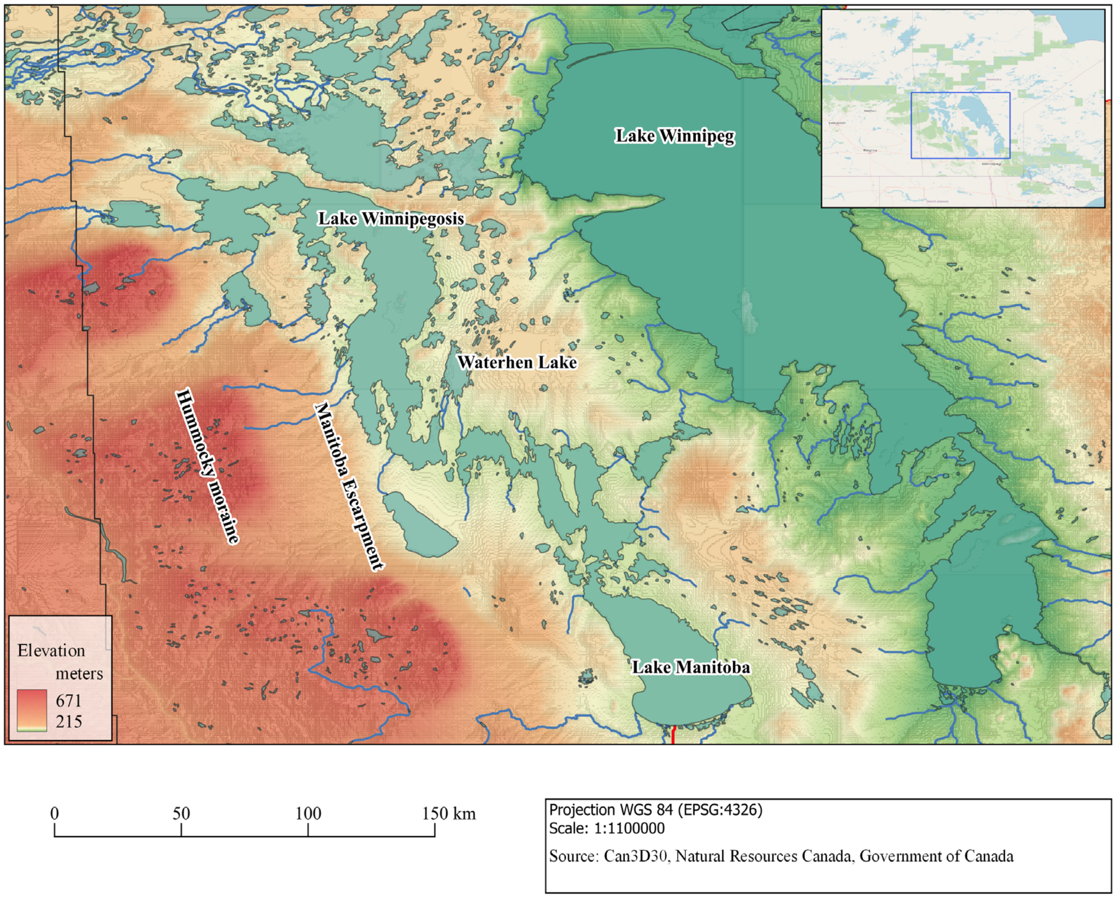


Figure 2.13. Upper Manitoba Great Lakes (uMBGL) Ecoregions

Table 2.6. Temperature and Precipitation climate normals and averages for 1991-2020

	Temperature (°C)			Precipitation (mm)
Location	**Max.daily.average**	**Min.daily**	**Max.daily**	**Avg.max..monthly**
The Pas	18.4	-23.4	23.9	80.1
Grand Rapids Hydro	17.5	-24.0	24.1	79.3
Mafeking	18.7	-21.2	24.8	105.8
Portage Southport	19.8	-19.1	25.4	80.4
Dauphin	18.8	-20.9	25.1	86.6
Hodgson	18.3	-24.3	25.2	85.2



Projection WGS 84 (EPSG:4326)
Scale: 1:1100000

Source: Can3D30, Natural Resources Canada, Government of Canada

Figure 2.14. Upper Manitoba Great Lakes (uMBGL) Digital Elevation Map, 1:1,100,000 scale

2.8 Vegetation

Following the northwest-to-southeast climate trends, vegetation and soils in Manitoba strongly correlate with increasing moisture availability (increased precipitation from west to east) (Scott, 1996). Wetland locations are, however, an exception to this trend, whose formation patterns can be attributed to topographical depressions created by glacial activity (Last & Ginn, 2005).

2.8.1 Mid-Boreal Lowland Ecoregion

The Mid-Boreal Lowland region (Boreal Plains ecozone) which houses the north end of Lake Winnipegosis, is co-dominated by Eutric Brunisols, which are associated with the sandy beaches of Lake Agassiz, and shallow to deep Organic Mesisols and Fibrisols. Limestone bedrock outcroppings are common (Smith, 1998). The clays, silts and sand of the remnants of Lake Agassiz are covered over by organic deposits in the form of poorly drained flat bogs and horizontal fens. This ecoregion is characterized by mixed boreal forest composed of black spruce, trembling aspen, balsam polar and jack pine. In the marshy areas, black spruce and ericaceous shrubs and mosses dominate, while in fens swamp birch and tamarack, sedges and brown mosses are the majority (Smith, 1998).

2.8.2 Interlake Plain Ecoregion

The Interlake Plain ecoregion is dominated by imperfectly drained Chernozemic Dark Gray soils in the southern and northwestern parts of the ecoregion. Inclusions include well to imperfectly drained Chernozemic Black soils. Eutric Brunisols, shallow Gray Luvisols, Organic Mesisols and peaty phase Himic Gleysols dominate in the central and northern section of this region. Dominant vegetation is a closed cover of trembling aspen, balsam poplar and understory of mixed herbs and tall shrubs. White spruce and balsam fir exhibit moderate to good growth but are not widely represented due to past fires. In sandy areas jack pine occurs, while in poorly drained sites sedges, willow, black spruce and tamarack are present (Smith, 1998).

2.8.3 Lake Manitoba Ecoregion

The Lake Manitoba Plain, where the central and south portions of Lake Manitoba are situated, contains some of the most productive soils in Manitoba (Black, Dark Gray and Dark Brown Chernozems), with the most productive soils located at the south end of the basin (Figure 2.16). The native tall-grass prairie in this ecozone has been replaced by agricultural crops, or rangeland in the driest parts of the plain.

2.9 Land use

2.9.1 Interlake Plain Ecoregion

Arable soil production is still moderately to severely limited in this region (Figure 2.16). Around the central basin of Lake Winnipegosis, land use is mainly pasture by large farms >540 hectares, with some smaller farms and pasture with high crop input (>$65/ha).

2.9.2 Mid-Boreal Lowland Ecoregion

Salt has been utilized in the Manitoba Lowlands Saline Waterbelt since A.D. 1100 (Petch, 1990). European settler journals indicate that the local Indigenous population used the salts and brines for medicinal purposes, tanning and food preservation. In the Lake Winnipegosis area, the saline springs provide minerals such as halite, silicate, clay, gypsum and aggregate resources (Last & Ginn, 2005). Along the shorelines and in deltas of these lakes, coarse clastics such as sand and gravels are used by communities. On the western shore of Lake Winnipegosis, adjacent to the Sapotaweyak Cree Nation Lands, 90 kilometres south of The Pas, there is a 560-

hectare Salt Flat Ecological Reserve. This salt flat complex is composed of poorly drained lake flats of saline till (Figure 2.8) (Manitoba's Protected Areas Initiative, 2015). The vegetation found in the salt flats is typical of salt tolerant plants commonly found in northern oceanic coastlines. This area is an important hunting, trapping, fishing and traditional activity area for generations by Indigenous peoples (Manitoba's Protected Areas Initiative, 2015). The portion of the salt flats extending into Overflowing Bay are important habitat for the endangered piping plover and yellow rail which is a Species at Risk (Manitoba's Protected Areas Initiative, 2015).

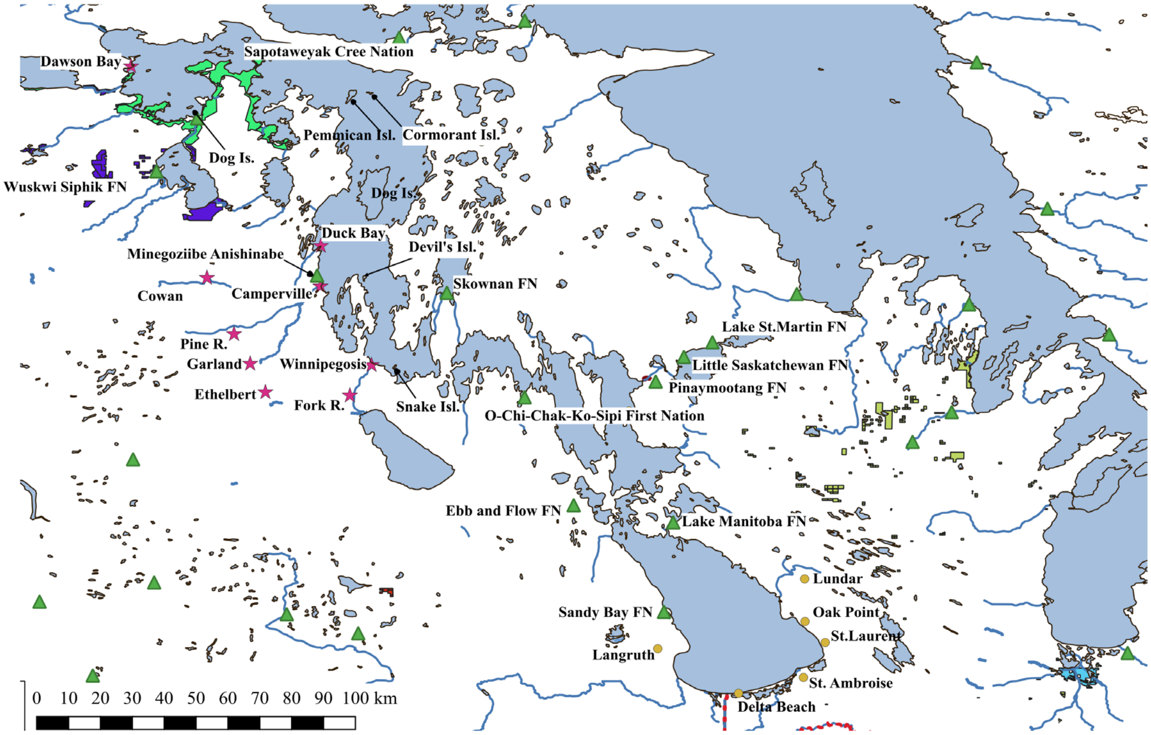
Eight communities and one First Nation are located around Lake Winnipegosis. Along the west side, lies the village of Winnipegosis, Camperville, Cowan, Duck Bay, Ethelbert, Fork River, Garland, Pine River and Pine Creek First Nation lie along the western shore, and between these two communities lies Pine Creek First Nation. Dawson Bay is located along the northwestern shore (Intermountain Conservation District & Province of Manitoba, n.d.) (Figure 2.15).

Agriculture Canada (Ag-Canada) crop inventory and land use for 2016 in this region shows a mix of wetlands, broadleaf, shrubland, mixed wood, and coniferous vegetation. According to the Ag-Canada Land Inventory (ACLI), which describes the ability of the land to support certain types of crops and agriculture management practices, the majority of soils surrounding the NW and SE basins of Lake Winnipegosis, in the Mid-Boreal Lowlands, are organic and not capable of supporting arable culture or permanent pasture (Figure 2.16) (Intermountain Conservation District & Province of Manitoba, n.d.). Class 2 and 3 soils, however, do support crop production, cattle ranching, and hay production (Intermountain Conservation District & Province of Manitoba, n.d.).

2.9.3 Lake Manitoba Ecoregion

In the south basin of Lake Manitoba, approximately 65 percent of the total area is forest or grassland (Page, 2011). The major land use in the Lake Manitoba ecoregion is either cropland or pasture. Pasture crops include high inputs (> $65/ha), small to large farm pastures (< 540 ha), and areas of higher crop diversity (pasture, summer fallow, flax) (Smith, 1998). About eight percent of the south basin produces cereal grain crops, while oilseed crops are more prevalent in the northern part of the south basin, where the soil composition is less desirable (thin, stony, nutrient deficient) (Page, 2011). In addition to crops, livestock production is the predominant agricultural activity (Page, 2011).

Major towns and communities around Lake Manitoba include St. Laurent, Oak Point, Lundar, Lake Manitoba First Nation along the east shore, and Langruth, Steep Rock, St. Ambroise, Sandy Bay, Delta Beach and Ebb and Flow First Nation along the western and southern shores (Figure 2.15).



Projection WGS 84 (EPSG:4326)
Scale: 1:600,000

Sources: Map Labels - CanVec Series - Toponymic Features; Waterbodies - Atlas of Canada National Scale Data 1:1000000,Government of Canada;Treaty Land Entitlement Sites in Manitoba, Government of Manitoba.

Figure 2.15. Upper Manitoba Great Lakes (uMBGL) major towns, communities, and First Nations

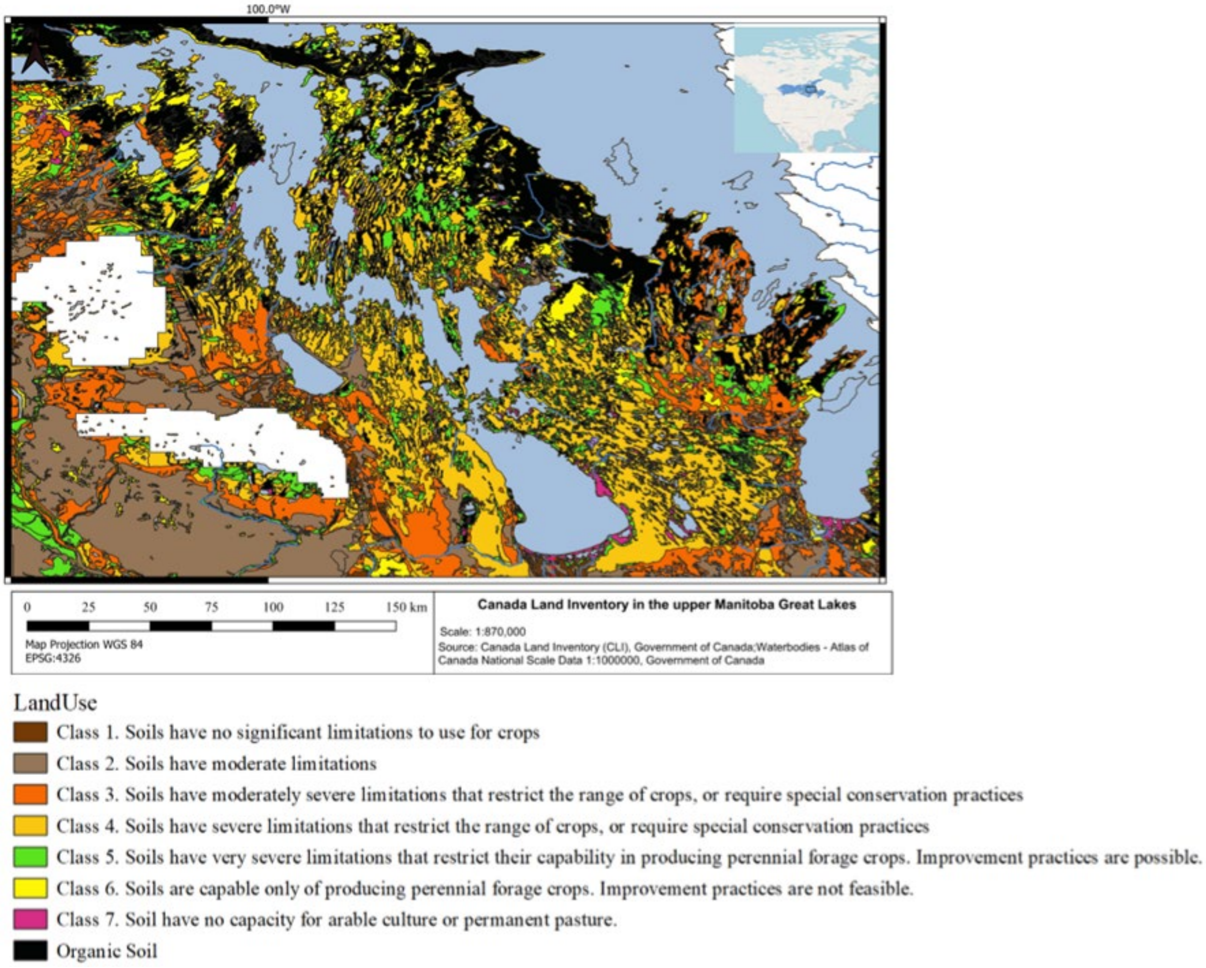


Figure 2.16. Upper Manitoba Great Lakes (uMBGL) land use as described in the Canada Land Use Inventory

2.10 Summary of previous water quality surveys

2.10.1 Lake Winnipegosis

Other than fish population surveys, there have been few recorded water quality studies on Lake Winnipegosis. The first documented survey of the uMBGL was conducted by Bajkov (1930), who measured total solids, turbidity, ions (Cl) and alkalinity. Bajkov (1930) recorded an average Secchi disk depth of 1.75 m, showing that the water was clearer than Lake Winnipeg at the time of his survey (Table 2.7).

The next known consistent sampling on Lake Winnipegosis occurred many decades later when, beginning in 2008, Manitoba Hydro's Coordinated Aquatic Monitoring Program (CAMP), began monitoring five sites periodically on Lake Winnipegosis. One site has been monitored annually on the lake for the entire 2008-2025 period (Table 2.7) (Coordinated Aquatic Monitoring Program, 2019, 2025).

Since 1973, the Water Science and Watershed Management, Water Stewardship section of the Province of Manitoba has sporadically monitored various streams that flow into Lake Winnipegosis and have sampled on-lake sites in the short term (Table 2.7) (Bourne et al., 2002). Water parameters analyzed include various contaminants, metals, ions and nutrients. Between 2008 and 2011, Donald et al. (2015) sampled river inflows and outflows and calculated nutrient sequestration for Lake Winnipegosis (Table 2.7). From 2012-2022, the UM Manitoba Great Lakes Program placed one mooring in the south basin of Lake Winnipegosis, and collected data for conductivity, temperature, salinity (2012-2021) and dissolved oxygen (2019-21) (Table 2.7) (Barber et al., 2021).

2.10.2 Lake Waterhen

Bajkov (1930) noted that the hydrological and hydrobiological conditions on Lake Waterhen were very similar to Lake Winnipegosis.

The Province of Manitoba monitors one station on the Waterhen River for water quality related to the lake (Table 2.7) (Bourne et al., 2002). The same variables as described for Lake Winnipegosis were analyzed). From 2012-2020, the UM Manitoba Great Lakes Program placed one mooring in south-central Lake Waterhen, and recorded data for conductivity, salinity, temperature (2012-2020) and dissolved oxygen (2019-2021) (Table 2.7) (Barber et al., 2021).

2.10.3 Lake Manitoba

Pre-1990

Bajkov (1930) also documented the first water quality results on Lake Manitoba. Since then, there have been sporadic studies over the decades that included water quality parameter reporting. Many of these have been limited to the south basin and include studies by Thomas (1953-54), Cober (1966-1969), Crowe (1972), Tudorancea and Green (1975), Gilliland (1965), van Everdingen (1971), Last (1980) and the Province of Manitoba (Table 2.7), (Herbert, 2025b; Page 2011).

Since 1970, the effects of the Assiniboine River Diversion have also been monitored for water quality parameters and studied for impacts on suspended sediment loading to Lake Manitoba (load, particle size distribution) (Table 2.7) (Crowe, 1974; Fred, 2013; Page, 2011).

Post-1990

Since August 1991, the Water Science and Watershed Management, Water Stewardship section of the Province of Manitoba has monitored inputs and outputs at Lake Manitoba quarterly (four sites) and monthly (two sites). Donald et al. (2015) also measured nutrient sequestration for Lake Manitoba based on tributary loading and export in the Fairford River (Table 2.7). From 2012-2021, the UM Manitoba Great Lakes Program collected data for conductivity, salinity, temperature, and dissolved oxygen from one mooring in the center of the north-west basin of Lake Manitoba (Table 2.7), (Barber et al., 2021).

2.11 Chemical Limnology

Nutrient enrichment in many Canadian lakes has led to larger and longer lasting toxic algal blooms, decreased water clarity and the potential for economic loss (Environment Canada & Manitoba Water Stewardship, 2011; Michalak et al., 2013). Aquatic systems are not only affected by eutrophication (increased nutrient loading), but also by changes in the algal composition and stoichiometry of the waterbody. The composition and ratio of the elements carbon (C), nitrogen (N) and phosphorus (P) (C:N:P) are the basis of stoichiometric analysis, and when coupled with seston (suspended particulate matter, particularly suspended mineral matter) provide fundamental insights into understanding primary productivity and food web interactions in lake systems, helping unravel the complex web of ecological interactions including nutrient cycling and plankton community dynamics (Ren et al., 2024). Elevated N:P ratios, for example, are associated with phytoplankton P limitation, poor food quality for zooplankton, and potential production of methane by P-limited microbes (Elser et al., 2022).

2.11.1 Lake Winnipegosis

After Bajkov's (1930) survey in the 1920's, almost 50 years passed before the next water quality samples were collected by the Province of Manitoba, beginning in the 1970's (Water Quality Management Section, n.d.). In his survey, Bajkov reported one low oxygen value (3.0 ppm) at 4.5 m at a site 2 miles west of Snake Island (Figure 2.6). All other oxygen values were above 10.2 ppm. Bajkov also reported a distinct north - south dissolved salt gradient, ranging from 1,200 ppm to 600 ppm. He attributed the decrease in dissolved solids in the south basin to the input of the Mossy River. From 2008-2011, Donald et al. (2015) measured total, dissolved organic and inorganic nitrogen, soluble phosphorus and nitrate-nitrite nitrogen. They calculated that Lake Winnipegosis sequestered around 73 percent of phosphorus and up to 90 percent of the nitrogen entering the watershed.

Data collected between 2008-2014 from the Coordinated Aquatic Monitoring Program-CAMP (Coordinated Aquatic Monitoring Program, 2017), found concentrations of chloride above the provincial guideline. The high values were attributed to the discharge of saline waters from the carbonate aquifer system (Coordinated Aquatic Monitoring Program, 2017). Water quality varied across the lake, with total phosphorus (TP) and chlorophyll a (Chla) resembling the north basin of Lake Winnipeg (Coordinated Aquatic Monitoring Program, 2017). Based on CAMP nutrient data, Lake Winnipegosis is classified as meso-eutrophic to eutrophic and is P-limited. CAMP also recorded one instance of a site (5) exceeding the Manitoba Water Quality Standards, Objectives and Guidelines (MWQSOG) for mercury, and exceedances for Selenium, Silver and Chloride (Coordinated Aquatic Monitoring Program, 2017).

2.11.2 Lake Waterhen

Bajkov (1930) measured total solids, minerals and ions for Lake Waterhen, and measured a Secchi depth of 1.25 m, CO_3 value of 63.3 ppm and a pH between 8.4-8.5. In addition to open water mooring data collection, the Manitoba Great Lakes Project (Barber et al., 2021) deployed under ice moorings in 2020 and 2021 and found under ice anoxia occurred from March to April 2020.

2.11.3 Lake Manitoba

Lake Manitoba is currently classified as slightly brackish, mesotrophic to eutrophic, with high specific conductivity, ranging from 624 $\mu S\ cm^{-1}$ in the spring to 2010 $\mu S\ cm^{-1}$ in late summer/early fall (Page, 2011). Lake Manitoba is considered turbid, with the south basin having higher turbidity than the north (south basin total suspended solid (TSS) average = 30 mg L^{-1}) (Page, 2011). The lake is also considered alkaline, with an average pH of 8.55, and is very hard, with values ranging from 265 mg L^{-1} to 382 mg L^{-1}. The alkalinity in Lake Manitoba is atypical of most alkaline prairie lakes as it is associated with the saline springs that occur along the west side. The lake has high total alkalinity ($CaCO_3$) in the south basin, with an average value of 218 mg L^{-1}, indicating that it has a high acid buffering capacity (Page, 2011).

Page (2011) reported average nutrient values of 1.25 mg L^{-1} for total nitrogen and 0.07 mg L^{-1} total phosphorus for the south basin, and an N:P molar ratio in the south basin of 49, ranging throughout the season from 3 to 147.

Table 2.7. List of historical water quality samples taken on the upper Manitoba Great Lakes

Lake	Sample Years	Variables	Reference	Source
Manitoba	1926-28	WQ	Bajkov (1930a)	
Manitoba	1953-54	WQ,T	Thomas (1959)	as cited by Last, 1980
Manitoba	1963	WQ	Gilliland (1965)	as cited by Last, 1980
Manitoba	1966-69	WQ	Cober (1968), Crowe (1972)	
Manitoba	1968-69	WQ	van Everdingen (1971)	as cited by Last, 1980
Manitoba	1971	WQ	Janusz (1972)	as cited by Last, 1980
Manitoba	1973-2018	WQ, MWQS-data	Manitoba Water Quality Survey (MWQS) (2018)	
Manitoba	1973-74	WQ	Tudorancea and Green (1975)	as cited by Last, 1980
Manitoba	1978	WQ	Last (1980)	
Manitoba	1980-82	WQ	Hughes (1983)	as cited by Last, 1980
Manitoba	1997-2000	WQ	in Hughes (2002)	as cited by Last, 1980
Manitoba	2005-2007	WQ	Page (2011)	
Manitoba	2008-2011	WQD	Donald et.al. (2015)	
Manitoba	2012-2022	mooring	MBGL (Barber et al., 2021)	
Waterhen	1926-28	WQ	Bajkov (1930a)	
Waterhen	1966-1978	WQ	Last (1980)	
Waterhen	2012-2022	mooring	MBGL (Barber et al., 2021)	
Winnipegosis	1926-28	WQ	Bajkov (1930a)	
Winnipegosis	1973-2018	WQ,MWQS	Manitoba Water Quality Survey (MWQS) (2018)	
Winnipegosis	2008-2011	WQD	Donald et.al. (2015)	
Winnipegosis	2008-2024	CAMP	CAMP (2025)	
Winnipegosis	2012-2022	mooring	MBGL (Barber et al., 2021)	

Notes:

Base water quality analysis (WQ) data includes pH, Total Hardness, Total Alkalinity, Specific conductivity, total dissolved solids, Ca, Mg, Na, P, Fe, Cl, So4, Carbonate, Bicarbonate, Phosphate, Nitrate, Fluoride, Boron, Secchi, DO, temperature

Manitoba Water Quality (2018) (MWQS-data) data also includes temperature, total residue , OH alkalinity , As , Cd, Cu, Pb , Ni, Zn , Mn, O2 , BOD . total organic and inorganic carbon , turbidity, total organic nitrogen, ammonia , orthophosphate , and O2 saturation

Thomas (1959)(T) also reports water temperature , O2, colour, turbidity, suspended matter, silica, non-carbonate hardness, % sodium, saturation and stability indices

Donald et al (2015) reported the variables (WQD) Nitrate/Nitrite,total phosphorus, total dissolved phosphorus, soluble reactive phosphorus, total nitrogen, total dissolved nitrogen, dissolved inorganic nitrogen

Barber et al., 2021 mooring data measured turbidity, chlorophyll, fluorescence, phycocyanin, temperature and conductivity. From 2019-2021 oxygen was also measured.

2.12 Physical Limnology

2.12.1 Thermal Mixing properties

2.12.1.1 Lake Winnipegosis

In the northern basin of Lake Winnipegosis, Bajkov (1930) identified deep spots ranging from 35 ft. (10 m) to 60 ft. (18 m). The south basin is much shallower, with an average depth of 15 ft. (4 m). Bajkov (1930) recorded a temperature difference of up to 7°C from the bottom of the lake, with a low dissolved oxygen count recorded during warm weather. The Coordinated Aquatic Monitoring Program (Coordinated Aquatic Monitoring Program, 2017) recorded thermal stratification at 4 of the 5 sample sites between 2008-2013. Profile data collected as part of the CAMP study showed a maximum temperature difference between the surface and bottom of the lake of 4°C, with an average temperature difference of 1°C.

2.12.1.2 Lake Waterhen

Historically, Lake Waterhen has been observed to be clear and deep in the middle. No recorded profiles exist. As mentioned earlier, the only under ice work recorded on the lake was conducted under the CEOS Manitoba Great Lakes Program in 2021. Anoxia was recorded from March to April 2021, with a fish kill noted in the spring (Skownan First Nation community member, personal communication, July, 2021). More observational research needs to be conducted to understand the stratification pattern of Lake Waterhen.

2.12.1.3 Lake Manitoba

Lake Manitoba exhibits weak thermal stratification that persists for a few hours, characterizing it as a polymictic lake (Bajkov, 1930; Kenney, 1979; Last, 1980). Temporary micro thermoclines from 1 to 2 meters in the water column were noted during successively calm and hot days (Tudorancea et al., 1979 as cited in Page, 2011) but differ from a true thermocline in their lack of long-term stability. Bajkov (1930), at a location near Fairford, was the first to conduct thermal and oxygen profiles, and noted that the difference between the depths never varied by more than three degrees. Kenny (1979, as cited in Page, 2011) also measured water temperature at 5 depths in the lake and noted a temperature difference over 4.2 m of no greater than 0.4°C, even during high winds (22 km/hr).

Last (1980) recorded inverse thermal stratification at 16 sites in the south basin under ice, with a surface temperature of 1°C and a bottom temperature of 4°C. Tudorancea et al. (1979, as cited by Page, 2011) also noted bottom temperatures ranging from 0.5°C near the Whitemud River, to 2.5°C under ice in the south basin (0.5 to 7 km offshore).

2.12.2 Sediment Resuspension

2.12.2.1 Lake Winnipegosis and Lake Waterhen

Neither Lake Winnipegosis nor Lake Waterhen have any recorded information on sediment resuspension, however CAMP analysis of sediment samples from Lake Winnipegosis found the bottom was mainly sand (93%) (Coordinated Aquatic Monitoring Program, 2017).

2.12.2.2 Lake Manitoba

Sediment resuspension in Lake Manitoba is quite variable due to wind and wave action on the littoral zone (Thomas, 1959 as cited by Page, 2011). Wind generated waves are estimated to resuspend about 66% of the lake bottom (Last, 1980).

Kenney (Kenney, 1979) found sand contributed the greatest mass to suspended sediment samples in the upper 2.7 m of the water column, while below that depth, silt comprised the largest fraction. Kenney attributed this shift in sediment size to the occurrence of two distinct water currents moving at different speeds and directions. The along shore currents move easterly or south-easterly and were found at water depth greater than 2.7 m, while the faster onshore currents occurred in the top half of the water column and moved in the same direction as the prevailing wind.

2.13 Biological Characteristics

2.13.1 Plankton

Information on the phytoplankton and zooplankton on all three upper MBGL lakes is sparse, with Bajkov (1930) recording the first observations, followed by sporadic identification by Kling (2025) for phytoplankton. Beginning in 2008, the CAMP program began recording counts for Lake Winnipegosis (Coordinated Aquatic Monitoring Program, 2017).

There are no records of zooplankton counts at any of the uMBGL. As part of my data collection, zooplankton samples were collected in both study years for all three lakes but have not yet been enumerated.

2.13.1.1 Lake Winnipegosis

There is only one previously published plankton analysis directly on Lake Winnipegosis, with data collected from 2008-2024 by the CAMP program (Coordinated Aquatic Monitoring Program, 2017). Five sites on Lake Winnipegosis were sampled by CAMP in 2008 and 2009 (Coordinated Aquatic Monitoring Program, 2017). Except for the most southerly sample station (5) on Lake Winnipegosis, phytoplankton biomass was highest in spring and fall, ranging from 11,235 to 24,952 mg m^{-3}. Phytoplankton communities were dominated by either cyanobacteria or bacillariophyta (diatoms) (Coordinated Aquatic Monitoring Program, 2014).

As part of the CAMP monitoring project, where chlorophyll concentrations exceed 10 ug L^{-1}, samples are also analyzed for Microcystin-LR (MC-LR). In the 2008/2009 sampling season, three stations (1, 2 and 5), were tested for Microcystin-LR. All stations tested negative (detection limit <0.2 µg L^{-1}) (Report prepared for the Manitoba/Manitoba Hydro MOU Working Group by North/Sourth Consultants Inc., 2014).

Kling (2025) previously identified multiple algal taxa on Lake Winnipegosis from one location near Swan River in the spring of 2005.

2.13.1.2 Lake Waterhen

Previous to this survey, there is no known algal identification data for Lake Waterhen.

2.13.1.3 Lake Manitoba

Bajkov (1930) generated a phytoplankton species list for Lake Manitoba and found that generally green and blue-green algae dominated the phytoplankton species composition with some diatoms being 'very common'.

Kling (2025) identified multiple algal taxa on Lake Manitoba from 2003 and 2009 samples.

2.13.2 Benthic Invertebrates

Benthic data for the upper MBGL is also very sparse, with Bajkov (1930) providing the first observations on all three lakes and Crowe (1974) following almost 50 years later with observations on Lake Manitoba. On Lake Winnipegosis, the next benthic survey after Bajkov occurred over 80 years later by the CAMP (Coordinated Aquatic Monitoring Program, 2017).

2.13.2.1 Lake Winnipegosis

Bajkov (1930) found the bottom of Lake Winnipegosis to be somewhat hard, with some places of very soft mud, and covered by dead molluscan shells and Chara.

The CAMP program has sampled annually for benthic macroinvertebrates since 2010 and assessed benthic macroinvertebrates (BMI) using 4 indicators and separating nearshore from offshore. Mean abundance for BMI is a commonly used indicator in lakes and is what will be reported for comparison in this report. Overall, BMI abundance on Lake Winnipegosis was more similar to abundance in Cedar or South Moose Lakes rather than Lake Winnipeg. In the sample years 2010-2013, more than 70% of the BMI were non-insects, with Amphipoda dominating at > 50%, followed by Oligochaeta at 20%. Ephemeroptera comprised the majority of the Insecta species. In the offshore habitat, Insecta (Chironomidae) dominated in 2010 and 2013 (70-80%), while in 2011 and 2012 non-insects (Oligochaeta) dominated, comprising 60-80% of the samples. Simpson's Diversity Index, which is a measure of species diversity, indicates there is a slightly higher species diversity in the nearshore habitat vs the offshore habitat (Coordinated Aquatic Monitoring Program, 2017).

2.13.2.2 Lake Waterhen

Bajkov (1930) reported that Lake Waterhen was rich in Amphipoda and insect larvae, with Amphipoda being so common they clogged gill nets. There are no other records for benthic invertebrates on Lake Waterhen.

2.13.2.3 Lake Manitoba

Bajkov (1930) found mollusk and Chironomidae larvae to be the main food items of fish in the lake, with Chironomidae larvae being found in whitefish stomachs. The next reported benthic survey was by Crowe (1974), almost 50 years later. Crowe used benthos to assess the productivity of lakes and found Lake Manitoba to be highly productive (Page, 2011). Midge (Chironomidae) larvae made up almost 60 percent of the benthos, with snails comprising 24% and fingernail clams 14% (Crowe, 1974; Page, 2011).

2.13.3 Fishery

All three upper MBGL lakes are multi-use fisheries that have been used for over a century by both the Indigenous rights holders for subsistence harvesting, commercial gill netting and recreational angling (Agriculture & Resource Development, 2020; Galbraith et al., 2017; Manitoba Fisheries, n.d.), as well as by the Hudson Bay company men and Icelandic fishers from the settlement of Gimli, MB. The use of the fisheries was so extensive, that somewhere between 1881 and 1882, the Indigenous income from fishing had so declined that, although they requested exclusive fishing rights on some lakes such as Lake Winnipeg, the Indigenous rights holders were denied by the federal government, and were instead required like every other individual to purchase a permit to fish, even for subsistence (Nicholson, 2007). Manitoba fisheries, including Lakes Manitoba and Winnipegosis, also experienced a crisis of over-harvesting caused by multiple factors such as lack of proper oversight by the federal government which allowed the development of large American owned consortiums (Nicholson, 2007). This led to the closing of both summer fisheries in 1905 (Manitoba) and 1906 (Winnipegosis), though Lake Manitoba fishery did reopen in the summer for a period during and after World War 1 (Nicholson, 2007). The winter fishery was allowed to stay open but allowed people to be employed only part-time as fishers anywhere but on Lake Winnipeg (Nicholson, 2007). By allowing only part-time fishers, the government hoped to discourage farmers who fished in the winter as a supplement to their income to turn to fishing full time.

2.13.3.1 Lake Winnipegosis

Lake Winnipegosis, which has been commercially fished since the late 1890's, is Manitoba's third largest commercial fishery, generating between $1-$2 million dollars per year. T Nicholson (2007) noted records of fishermen from the Icelandic settlement of Gimli, MB reaching Lake Winnipegosis in 1897, responding to reports of a rich fishery. The summer whitefish fishery re-opened on the lake in 1926, this time with a harvest limit. A fall pickerel season was also included, but the main fishery was still in winter (Nicholson, 2007). Bajkov (1930) reported important whitefish grounds in the northern part of the lake, extending from Cormorant Island to Dawson Bay (Figure 2.15). Oxygen concentration in the south end of the lake was thought to be too low to support whitefish in the summer. Between the 1960's and 1980's, a drastic decline in pickerel was noted, decreasing from one million to 400,000 pounds of landed catch (2007).

There are currently 26 species present in the lake, with the winter commercial fishery based mainly on white sucker and northern pike. In the summer, walleye are fished under a limited harvest quota licence (Nicholson, 2007; Resources & Northern Development, n.d.). Both fisheries generate between \$1 - \$2 million dollars in landed catch each year. Lake Winnipegosis is also stocked with between 1 to 5 million walleye fry each year (Resources & Northern Development, n.d.).

The CAMP program samples fish annually in the north and south basins of Lake Winnipegosis. For the sampling period 2008-2013, though the population size caught was small, CAMP found species similar to Lake Winnipeg, including walleye, yellow perch and white sucker (Coordinated Aquatic Monitoring Program, 2017).

2.13.3.2 Lake Waterhen

The communities of Skownan First Nation, Waterhen and Mallard are located along the shores of Lake Waterhen, which has been commercially fished since 1931, and was the first freshwater eco-certified fishery for northern pike (*Esox Lucius*) and lake walleye (*Sander vitreus*) in North America. Together with Cedar Lake Fishery, Lake Waterhen is the only source of Marine Stewardship Council (MSC) certified sustainable northern pike in the world (International Institute for Sustainable Development, 2022). The landed value for all fish species on the lake has ranged between $9,069 in 2003/2004 down from $347,111 in 1991/1992. In year one of this study (2016), the total landed value of all fish was just over $60,000. In 1993, 2003 and 2011 Lake Waterhen was also stocked with either walleye or lake whitefish (Klein & Galbraith, 2017).

In 2020, the CEOS MBGL sampling program observed under-ice anoxia for a month in February (Barber et al., 2021). Subsequently, in spring 2020 a large number of dead fish were reported along the lake shoreline, creating concern that anoxia may be an issue on the lake. As of 2025, no other under ice monitoring has occurred.

2.13.3.3 Lake Manitoba

The first fishery documented on Lake Manitoba was an Indigenous fishery, found in records from 1804 (Manitoba Fisheries, n.d.). Icelanders, moving from the parent colony in Gimli, MB, started fishing on the shores of Lake Manitoba in 1885, and at the Narrows in 1890 (Nicholson, 2007). Regulation of the commercial fishery on Lake Manitoba began in 1895 (Manitoba Fisheries, n.d.), There is also a coarse fish (lake carp and suckers) fishery allowed year-round and a limited Delta Marsh fishery (Manitoba Fisheries, n.d.).

Bajkov (1930), recorded the first complete fish survey, and identified the pickerel, tullibee and whitefish fisheries as the top three fisheries on Lake Manitoba. Annual fish production declined drastically since 1950, with this change mainly attributed to the regulation of water levels in 1961 at the Fairford dam, causing the loss of spawning habitat and drainage of wetlands, although it is argued that the drought of the 1960's, an increase in fishing licenses and a decrease in fishing net mesh size were in fact the major contributors (Nicholson, 2007; Page, 2011). Nicholson (2007) also noted that the drought in the 1930's and soil runoff from farmers' fields may have also contributed to the change in type of dominant species caught over the decades from 1902-1941.

As of 2021, the Lake Manitoba fishery (commercial and recreational), is valued at about $9 million per year, with walleye and sauger the main commercially fished species (Manitoba Fisheries, n.d.). The walleye population have increased significantly since the voluntary increase of the minimum mesh size in the 2016-2017 fishing season, however sauger while increasing, are still in a collapsed condition (Manitoba Fisheries, n.d.). The walleye fishery is also supplemented by larval stocking supplied by local hatcheries (Manitoba Fisheries, n.d.). If current catch rates continue on the lake, an eventual decline in both species is predicted due to non-sustainable harvesting (Manitoba Fisheries, n.d.). Catches of cisco, yellow perch and northern pike, as well as carp are also part of the commercial fishery, showing a decline in the last five years from the 2001/2002-2005/2005 fishing seasons and the 2006-2007 to 2010/2011 seasons (Manitoba Fisheries, n.d.).

Oxygen depletion is an ongoing concern in Lake Manitoba as well as Lake Waterhen. 'Death waves' of dead fish, likely from severe oxygen depletion under ice, have been reported below the ice surface and in the spring (Page, 2011). The decrease in oxygen in such a shallow lake, was attributed, in this instance, to red algae blooming below the ice (Page, 2011).

3. Methods

3.1 Open water sample collection

Water samples were collected from 16 and 18 locations in 2016 and 2017, respectively, from Lake Manitoba (Figure 2.7) and 10 locations in 2016 and 2017 from Lake Winnipegosis (Figure 2.6). Two samples were collected from Lake Waterhen (Figure 2.6), both in the same area as the Waterhen mooring, so while the methodology is the same, Lake Waterhen was not used for algal genus comparison (Table 3.1). Table 3.2 defines all variables reported in this study.

In 2016, only surface water samples (10 cm below surface of the water) were collected, while in 2017, surface and bottom (0.5 - 1 m from the bottom of the lake) samples were collected in addition to the surface water samples. Samples were collected one to two times per season (May-June, July-August, September-October) to capture seasonal changes representing spring, summer and fall. During each field survey (spring, summer, fall), samples were also collected on a regular basis from the Fairford, Waterhen, Overflowing and Red Deer Rivers, to characterize nutrient and particulate loading to the upper MBGL, but the data will not be discussed as part of this thesis. Sampling locations were selected based on accessibility via the sample boat from the nearest boat launch, and spatial distribution within the lake (with more central locations and capturing the most spatially uniform distribution prioritized). In order to facilitate comparison with longer term datasets, where possible sample locations close to current provincial water quality sites were selected.

Table 3.1. Sample summary for Manitoba Great Lakes, 2016-2017

Lake	Year	Stations (n)	Surface (n)	Bottom (n)
Lake Manitoba	2016	16	30	0
	2017	18	46	12
Lake Waterhen	2016	2	5	0
	2017	1	6	1
Lake Winnipegosis	2016	10	17	0
	2017	10	22	14

Note: Surface depth = 0 m; Bottom depth > 0 m, n=number of samples or stations

Table 3.2. Definitions, abbreviations, and units describing light, nutrient, chemical and phytoplankton parameters

Category	Parameter	Abbreviation	Units
Physical			
	Sample depth	S_depth	Meter
	Air temperature	A_{temp}	°C
	Surface water temperature	S_{temp}	°C
	Bottom water temperature	B_{temp}	°C
Light			
	Maximum depth	Z_{max}	Meter
	Secchi disk depth	Secchi	Meter
	Total suspended solids	TSS	mg L^{-1}
	Volatile suspended solids	VSS	mg L^{-1}
	Fixed suspended solids	FSS	mg L^{-1}
	Particulate organic matter	POM	mg L^{-1}
Nutrient			
	Total phosphorus	TP	µg L^{-1}
	Total dissolved phosphorus	TDP	µg L^{-1}
	Dissolved reactive phosphorus	DRP	µg L^{-1}
	Particulate phosphorus	PP	µg L^{-1}
	Total nitrogen	TN	µg L^{-1}
	Total dissolved nitrogen	TDN	µg L^{-1}
	Dissolved organic carbon	DOC	µmol L^{-1}
	Particulate nitrogen	PN	µg L^{-1}

	Particulate organic carbon to particulate phosphorus ratio	OC:P	molar ratio
	Particulate nitrogen to particulate phosphorus ratio	N:P	molar ratio
	Particulate carbon to particulate nitrogen ratio	OC:N	molar ratio
Chemical			
	Conductivity	Cond	μS cm^{-1} @ 25 °C
	pH	pH	unitless
Phytoplankton			
	Total chlorophyll	TChl	μg L^{-1}
	Chlorophyll-a	Chl a	μg L^{-1}
	Particulate organic carbon	POC	μg L^{-1}
	Particulate organic carbon to total chlorophyll ratio	POC:TotC	molar ratio
	Total phytoplankton biomass	Phyto	mg m^{-3}
	Cyanobacteria biomass	Cyano	mg m^{-3}
	Chlorophyta biomass	Chloro	mg m^{-3}
	Euglenophyta biomass	Euglo	mg m^{-3}
	Chrysophyceae biomass	Chryso	mg m^{-3}
	Haptophyta biomass	Hapto	mg m^{-3}
	Bacillariophyta Biomass	Bacill	mg m^{-3}
	Cryptophyta biomass	Crypto	mg m^{-3}
	Dinoflagellata biomass	Dino	mg m^{-3}

Water samples were collected from an anchored 16-foot Lund boat with a 40 hp Mercury outboard motor, which was pulled using a Ford F-150 truck. Samples were collected according to the protocols described by (Herbert & Kamula, 2019) and are described here for collection from a boat or bridge. After anchoring, sampling was not conducted for a minimum of 5 minutes to let any disturbed sediment settle, and samples were taken off the opposite side of the boat from the anchor. Care was taken to not sample if an observable plume of sediment was noted. Field observations and metadata were collected in a Rite-in-the-Rain field logbook and digitized as pdf files which are available upon request. Field log notes were also uploaded as a csv file to the Canadian Watershed Information Network (CanWIN) project page (Herbert, 2018). Initial sample location coordinates were recorded using a Garmin GPS in datum WGS84, the model as noted in the field logbook (Herbert, 2018). Each time we returned to the sample location; the exact coordinates were recorded again. In some instances, for example in bad weather, if we could not return to within 500 m of the original sample location, a new location was recorded. Therefore, some stations may have been sampled only once or twice over the two-year period, but they are still within 2.5 km of the original station. At each station air and water (surface and bottom) temperature using a digital thermometer were recorded, with the air temperature measured first, over the water and in water on the shady side of the boat. Surface water temperature as provided by the Lowe boat sonar was also recorded beginning in 2017 when the sonar was installed.

Additional field observations recorded included weather conditions, wave height (m), cloud cover according to Environment and Climate Change Canada cloud cover scale, wind speed (Beaufort Wind Scale[2]) and wind direction. Additional samples collected at each station were also recorded as yes/no tick boxes in the field log (phytoplankton, zooplankton, algal toxins and nitrogen fixation). At certain locations, additional samples were taken and processed as per the Lake Winnipeg Foundation protocols for community-based monitoring (CBM) to compare phosphorus results from different analytical methods.

3.1.1 Water quality collection

Whole water samples were taken using a 500 ml acid washed high density polyethylene (HDPE) Nalgene brand screw-cap bottle, which was triple rinsed downstream with lake water prior to collecting the sample. Samples were collected from the opposite side of the boat from which the anchor was deployed. Surface water samples were collected by placing the sample bottle 10 cm below the surface of the water and letting it fill to rim of the bottle neck, in order to leave a headspace. When removed from the water, the sample was capped, shaken vigorously and a sub-sample was decanted into a 20 ml glass scintillation vial pre-filled with acid Lugol's for phytoplankton taxonomy. The remaining sample was recapped and stored in a cooler with ice for further processing on shore.

In 2017, bottom samples (0.5 m above the lake bottom) were taken from sites where the lake depth was greater than 4 m. Lake depth was measured using the depth sounder in the Lowe boat as well as via a weighted tape measure. Depth using the tape measure was recorded as the depth from the surface of the water to the bottom of the lake. Bottom samples were collected using a Kemmerer water sampler with a calibrated multi braided polypropylene line marked at 1 m intervals. When the desired depth was achieved, a metal messenger was sent down the line to

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[2] https://www.canada.ca/en/environment-climate-change/services/general-marine-weather-information/understanding-forecasts/beaufort-wind-scale-table.html

trigger the open ends of the sampler to close. The sampler was then brought to the surface, where a 500 ml high density polyethylene (HDPE) Nalgene brand screw-cap bottle that had been acid washed and cleaned according to (Herbert & Kamula, 2019) was rinsed three times with the lake water and then filled using the same method as for the surface water sample, including collecting a sub-sample for phytoplankton taxonomy. When collecting a bottle sample at a station, the bottom sample was collected before any other sample was taken to ensure minimal disruption of the lake bottom.

River samples were collected either from the Lowe boat using the same sampling method described previously, or if sampling from a bridge, by inserting the 500 ml Nalgene bottle into a stainless-steel sample holder which was lowered from the center of the bridge channel using a multi braided polypropylene line which was tied to the bridge railing. The sampler was lowered into the water to just below the surface and submersed until the bottle was full. The bottle was also rinsed three times and rinse water was dumped downstream of sample collection. A phytoplankton sample was sub-sampled as above.

In 2017, water clarity was measured *in situ* using a 0.22 m diameter black and white Secchi disk attached to a multi braided polypropylene line marked at 0.1 m intervals to a depth of 1 m, then subsequently marked at 0.5 m intervals. The Secchi disk was held over the shady side of the boat and slowly lowered into the water until the white sections were no longer visible, then it was lowered an additional 0.2 m and slowly raised until it was visible again. The mark where the line met the surface of the water was recorded as Secchi disk depth (Secchi).

3.1.2 Probe collection

Vertical lake profiles for conductivity, temperature, pressure and turbidity (CTD) were measured *in situ* at each site using an Idronaut Model 304 during every sample survey. The Idronaut was attached to a multi braided polypropylene line which was marked at 1 m intervals. The Idronaut was held in the air for thirty seconds, submerged just under the surface of the water for one minute, then hand lowered slowly at ~1 m per second intervals until it touched the bottom of the lake. The Idronaut was then raised back to the surface, and the bottom rinsed with lake water to remove any sediment. If sediment was stirred up from the bottom or attached to the instrument when it was raised, it was noted in the logbook. The Idronaut records data at a frequency of 6-7 times per second for each profile, with data averaged to approximately sixteen second intervals. Conductivity at 25°C and salinity are derived values calculated from the raw conductivity values using the UNESCO formulae (IOC et al., 2010).

In 2017, a Li-Cor Model Li-192 Underwater Quantum Sensor with an LI-1000 DataLogger was used to collect vertical light profiles at 0.5 m intervals. Surface incident readings were taken at the start and end of every profile. Profiles were conducted twice at each station and the readings were averaged. In August 2017 vertical profile data was also collected using a Seabird SBE19plus for temperature, salinity, turbidity, fluorescence, oxygen and photosynthetic irradiance (PAR). In July 2019, a bbe Fluoroprobe was used to record profiles of fluorescence emission and excitation wavelengths for algal taxa identification. The Idronaut, Li-Cor, Seabird and Fluoroprobe data are not reported in this thesis but will be available in CanWIN[3] and will be used in planned future publications once this thesis is published.

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[3] https://canwin-datahub.ad.umanitoba.ca/data/project/mbgl

3.2 Plankton Collection

3.2.1 Phytoplankton

Samples for phytoplankton analysis were collected using two methods. In 2016 and 2017 whole water samples were sub-sampled from the 500 ml HDPE sample bottle collected for chemistry analysis, into 20 ml glass scintillation vials with inverted cone lids. The scintillation vials were pre-filled with 0.5 ml of acid Lugol's Iodine for preservation. In 2016, additional samples were taken using a 63-micron phytoplankton net, which was lowered from the surface of the lake to 1 m above the bottom. The sample was then poured into a 20 ml glass scintillation vial filled with acid Lugol's Iodine. Samples are stored at room temperature in cardboard boxes in Lab 566 in the Wallace building at CEOS.

3.2.2 Algal Toxin Analysis

In both years, 1000 - 2000 ml of lake surface water, collected as above, was passed through a 47 mm glass fiber (GF/F) filter, with a nominal particle retention size of 0.7 microns. Filter papers and filtrate were saved, kept cold using freezer packs until returning to the UM, then frozen at -20ºC. In addition, a second sample collection method for toxins was utilized, by slowly lowering a zooplankton net with a 63-micron mesh size from the surface of the lake to 1 m above the bottom. The net was raised to the surface again, and, using lake water from the sample site, was rinsed into a Whirlpak bag via the plastic collection tube at the bottom of the net. The Whirlpak was labelled with the date, station id and sample depth. The top of the Whirlpak bag was folded over 2 - 3 times, then, holding both bag end twist ties, the bag was whirled around to seal it with air. Samples were stored in coolers with ice until return to the UM, where the bag was frozen at -20ºC. Whole water and zooplankton haul samples were freeze dried using a heat evaporator unit located at the Department of Fisheries and Oceans, Freshwater

Institute, Winnipeg, MB. Samples were sent to the Zastepa lab at Environment and Canada Change, Burlington, ON for toxin analysis. Results were not available at the time of publication.

3.2.3 Zooplankton

In both study years, zooplankton samples were collected using a Wisconsin net of 25 cm mouth diameter and 73-micron mesh size. The net was hauled vertically from just above the lake bottom to the surface at a rate of 0.5 m s^{-1}. Keeping the mouth of the net above the water surface, the sides of the net were rinsed down twice, draining the sample into a 250 ml glass sample container between rinses. Zooplankton samples were preserved in 10% formalin, stored in a cardboard box with dividers, kept cool and in the dark and sent to Salki Consultants for enumeration. Data was not available at the time of this thesis publication.

3.3 Sample Processing

After collection, samples were immediately placed in a cooler with frozen ice packs until they could be processed. Samples were generally processed within 24 h of collection, with a maximum time between sampling and processing of 48 h. Whole water samples were divided into particulate and dissolved fractions in the field using an OEM hand vacuum pump with a pressure gauge and a magnetic filter unit (Herbert & Kamula, 2019) attached to a glass Erlenmeyer flask, or at CEOS using a vacuum line and the same magnetic filter unit and flask. Between surveys all filtration equipment and sample bottles were cleaned at CEOS. Equipment was rinsed three times with hot water, scrubbed with a bottle brush and rinsed with distilled water three times. Equipment was then soaked in a 10% hydrochloric acid (HCl) solution for 24 h, filled with distilled water and left for 24 h. Equipment was then rinsed and placed on a drying rack until dry.

Dissolved and particulate phases are defined as what either passes through or is retained on either a pre-ignited (16 hours at 500 °C) Whatman 42.5 mm glass fibre (GF/C) filter paper (nominal particle retention size 1-2 μm) or 42.5 mm GF/F filter paper (nominal particle retention size 0.7 μm). In this study, different retention sizes were used for different parameters. For the parameters phosphorus, chlorophyll, total suspended solids (TSS), carbon and nitrogen, a measured amount of sample water from 100-200 ml was passed through a GF/C filter paper (one for phosphorus, one for chlorophyll, one for carbon and nitrogen combined) into the Erlenmeyer flask. Between each sample pour (for each parameter), the sample bottle was capped and shaken vigorously to ensure homogeneity. Each filter paper was placed into its own petri dish, labelled with the corresponding particulate analysis (phosphorus, nitrogen, carbon or chlorophyll) and placed in a desiccator for 24 h. Filter papers were then removed, each analysis type was combined into one plastic bag, and all filter papers were frozen at -20 °C (Herbert & Kamula, 2019) until analysis. The first filtrate from each sample site was discarded, and 125 ml was subset into a high-density polyethylene (HDPE) Nalgene brand screw-cap bottle for total dissolved phosphorus (TDP), dissolved reactive phosphorus (DRP) and total dissolved nitrogen (TDN) analysis. The sample bottle was labelled with sample site, sample date and analysis (TDN/TDP) and stored in the fridge at 4 °C until analysis. The filtration funnel and flask were rinsed with distilled water before a new site was processed. For quality control, at the beginning of each filtration day in the field, a blank filter paper and a filter paper with 100 ml of distilled water passed through it were also desiccated and sent to the lab for phosphorus analysis.

Samples for dissolved organic carbon were sub-sampled from the whole water sample and transferred into either 4 ml or 20 ml pre-combusted (16 hours at 500 °C) glass vials. Vials were previously cleaned using the same 10% hydrochloric acid bath process described above.

Vial lids were dried at 65 ºC in a drying oven at the Center for Earth Observation Science (CEOS) (Binder APT. line). While wearing polypropylene gloves, filtrate from the Whatman GF/C filter papers was sub-sampled into the glass vials, and samples were preserved with 50 µL 2M HCl. Vials were recapped, lids were wrapped using Parafilm and samples were stored at 4ºC in the dark until analysis. Due to capacity, most DOC samples were not analyzed at CEOS, but were sent to the FWI laboratory and analyzed according to (Stainton et al., 1977) via the gas chromatography method.

Filters for TSS determination were pre-weighed and analyzed at the Centre for Earth Observation Science using the method described by Fischer (2024) and (Herbert & Kamula, 2019). In brief, frozen filter paper samples were thawed in a desiccator until dry, then were placed on pre-combusted (16 hours at 500 °C) aluminum foil squares that sat in foil crucibles. The crucibles were placed in a drying oven for 16 h to overnight at 104 °C. Crucibles were then removed and placed in a covered desiccator to cool down. When cool, all filter papers were weighed on a precision A&D weighing GR-200 scale with an error of ±1 mg, a scale interval of 0.1 mg, and minimum and maximum weights of 10 and 210 mg. A check weight was used at the start of every sample weighing data and recorded. An empty crucible and foil paper were then placed on the scale, tared, and the filter paper sample was transferred into the crucible using flat tip forceps. After the reading stabilized (5-10 s), the sample weight was recorded, and the sample was placed back in the desiccator. The increase in the weight of the filter, divided by the filtered volume, is assumed to be the TSS (Stainton et al., 1977). Every 10-15 samples a sample was weighed twice for quality control. When finished, the filter papers were placed back in the oven and left at 104 °C for another 16 h, after which the process above was repeated. If the weight difference between day 1 and day 2 for a sample was > 0.04 mg the sample was placed back in the oven for a third day. Results were averaged to obtain the final TSS value in mg L^{-1}.

Volatile suspended solids (VSS) act as a proxy for the amount of organic matter lost on ignition of the TSS filter paper at 500 °C (Fischer, 2024; Herbert & Kamula, 2019; Stainton et al., 1977). To calculate VSS, once all samples were within the acceptable error margin, they were transferred to a Barnstead/Thermolyne Type 30400 with Eurotherm 2416 Temperature/Process Controller oven, where the temperature was ramped up to 500 °C, held there for 4 h and ramped back down to 0 °C. The same process was conducted until the samples were within the 0.04 margin of error. Results were averaged and divided by the filtered volume to obtain the final VSS value in mg L^{-1} (Fischer, 2024; Herbert & Kamula, 2019; Stainton et al., 1977).

Fixed suspended solids (FSS) act as a proxy for inorganic suspended solids (also called tripton) and were calculated as the difference between VSS and TSS (Fischer, 2024; Herbert & Kamula, 2019; Stainton et al., 1977).

Using the same sample water as above, samples for algal particulates (αP) were filtered onto a 42.5 mm Whatman GF/F filter paper using the OEM hand vacuum pump was above. The filtrate from the αP was collected for coloured dissolved organic matter (CDOM) analysis, transferred to a brown glass bottle pre-combusted at 500 ºC for 16h, then wrapped in foil which had also been combusted at 500 ºC for 16 h. Filter papers were stored at -20ºC and filtrate at 4ºC in the Barber lab at CEOS. Samples were analyzed at CEOS in the Ehn/Mundy lab using a Lambda 650S UV/Vis spectrophotometer following the method described by (Fischer, 2024) and (Herbert & Kamula, 2019). CDOM and αP data were not reported on for this study.

3.4 Water Quality Analysis

With the exception of TSS, VSS and FSS, all water samples for nutrient and chemical analysis reported in this study were analyzed at the Freshwater Institute (FWI), Department of Fisheries and Oceans, Winnipeg, MB according to the methods described by (Stainton et al., 1977). Samples were hand-delivered in coolers with ice packs after each survey. Filtrate and filter paper sample storage time was usually less than 48 hours but could range up to four days based on date transferred to FWI (e.g. samples sent on the weekend would not be analyzed until the following week). Previous sample storage time analysis conducted at FWI did not indicate a significant loss of dissolved nutrient concentration with a longer storage time (Stainton, personal communication, 2018).

Chlorophyll was analyzed using a Shimadzu RF-551 PC Spectrofluorometric detector with a High-Pressure Liquid Chromatography column (HPLC), resulting in a value for total chlorophyll (Tchl) for analysis without the HPLC column, and phaeophytin corrected chlorophyll concentration (Chl a) using the HPLC column. The Shimadzu provides excitation around 440 nm and detects emissions around 660 nm (Stainton et al., 1977), with arbitrary units calibrated to a known chlorophyll a standard. The method can overestimate TChl in waters with higher sediment or that are aphotic, however it is the measure commonly reported for Lake Winnipeg and other prairie lakes where chlorophyll was analyzed at FWI, therefore, of the two, TChl is the parameter that will mainly be reported on in this study.

Specific conductance and pH were analyzed using portable lab meters. Analysis on 2016 samples was conducted on the filtrate instead of the whole water samples as whole water was not provided to the lab in large enough quantities to conduct pH and conductivity measurements.

This issue was corrected for the 2017 field season. Samples for specific conductance were put into a water bath at 25 °C, therefore results were reported as specific conductance at 25 °C.

DOC was analyzed using perchloric acidification and gas chromatography. Particulate organic carbon (POC) was analyzed on the same filter paper as particulate nitrogen (PN). The filter paper was thawed, then cut in half using sterile scissors. Samples were analyzed using a Perkin Elmer Model 240 stop-flow type CHN analyzer. TDN and TDP were analyzed using photocombustion and zinc reduction with a Technicon Autoanalyzer II system. Dissolved reactive phosphorus (DRP), often referred to as soluble reactive phosphorus (SRP) was analyzed using a Col-Parmer 2100 UV spectrometer and particulate phosphorus (PP) was analyzed using the ammonium molybdate method and the Technicon system.

3.4.1 Phytoplankton analysis

Lakes Winnipegosis and Manitoba were divided into basins representing various regions based on geographic and hydrologic features of each lake, to allow for an evaluation of spatial variability of effects of nutrient chemistry and suspended sediments on plankton distribution. In some cases, individual phytoplankton samples for each site were combined into one sample per region by subsampling 2-4 ml from each station into a new 20 ml glass vial for each region and season (spring, summer, fall) for each year to facilitate counting. Counts were conducted by either counting 10 fields, counting along a single transect at 63x magnification, or by counting half the sample chamber at 40x magnification using Integrated Modulation Contrast (IMC). Counting is not complete, so a subset of the data is used in this thesis.

Algal samples were scanned for major taxa to identify dominant genera. The 2016 net samples were used to identify genera but were not used for counting as they were too concentrated. A limited subset of samples (n=6) representing the major basins in Lake

Winnipegosis and Lake Manitoba were sent to Hedy Kling, Algal Taxonomy and Ecology Inc. for species enumeration. In addition, I identified major taxa using a Leica DMIL LED Inverted Routine Fluorescence Microscope and eyepieces with 12.5x magnification, with identification verified by Hedy Kling. Phytoplankon were identified to species where possible and visually counted using a Leica diavert microscope following (Findlay & Kling, 2003) and (Lund et al., 1958), and final biomass and biovolume were calculated using the software program Phytobio (Fee, 1980; Fee et al., 2022).

Measurements of phytoplankton nutrient status consisted of four particulate composition ratios: organic carbon to phosphorus (OC:P), nitrogen to phosphorus (N:P), organic carbon to nitrogen (OC:N) and carbon to chlorophyll (OC:Tchl), calculated on an atom:atom basis (molar ratio). For a subset of samples in 2016 (n=6 on Lake Winnipegosis and n= 8 on Lake Manitoba), one metabolic indicator (nitrogen fixation) was calculated. Samples were analyzed using the acetylene reduction assay (ARA) method as per Findlay (1994) and Flett (1976) and modified for in-field use as per Higgins et al. (2017). Due to the low number of analyses, the data was not used in this thesis.

3.5 Statistics

All analysis was conducted using R Statistical Software v 4.5.1 (2025-06-13 ucrt). For the parameters VSS and TSS, negative values were adjusted to 0 as they were below detection. No other adjustments were made to the data. Summary statistics per lake were generated (Appendix A) for each water quality parameter, consisting of the mean, standard deviation (SD), median, median absolute deviation (MAD), minimum and maximum values. The median and MAD were used to determine the variability in the dataset as MAD is considered more robust against outliers than the SD (Helsel et al., 2020; Shao & Wang, 2009). The data were assessed for normality by applying the Shapiro-Wilk (S-W) test, and non-normality visually verified by generating Quantile-Quantile (QQ) plots with the S-W *p*-value overlaid. In the S-W test, the *W* statistic indicates the goodness of fit of the data to a normal distribution, and the *p* value is the probability of observing the data if the null hypothesis is true (the data is normally distributed) (Helsel et al., 2020). The *p* value was considered statistically significant at $p <= 0.05$ (Mukherjee & Bhonge, 2025) (Table B.1). The QQ plots compare the empirical (theoretical) quantiles against the observed data (Dytham, 2011; Meals & Dressing, 2005). The red line represents the normal distribution with the same mean and SD as the dataset. The S-W test is more robust with small samples sizes so was used instead of some other common normality tests such as the Kolmogorov-Smirnov test (Papadaki et al., 2023; Shapiro & Wilk, 1965; Shapriro et al., 1968). According to the S-W test, most of the water quality parameters were not normally distributed, therefore non-parametric analysis was applied for further analysis (Table B.1). Using non-parametric analysis has several advantages over transforming non-normal data such as for skewed distributions like water-quality data, greater power is achieved, and censored data (above or below detection limits) can be used with imputing values (Helsel, 1987).

3.5.1 Carlson's trophic state index (TSI)

Carlson's trophic state index (TSI) is a multiparameter index that uses algal biomass as the basis for its classification system by applying three water quality parameters, SD, TChl and TP as independent indicators of algal biomass (Carlson, 1977). The TSI differs from other multiparameter indices in that it reflects a continuum of trophic states rather than a discrete single classification, allowing for the changing status of a lake and acknowledging that although nutrients cause eutrophication, the effects of nutrient additions can be modified by other factors, for example, climate related (weather), and biological (plankton composition) (Carlson, 1977; Carlson & Simpson, 1996).

3.5.2 Non-parametric statistics

The non-parametric two-sided Wilcoxon Rank Sum Test (Mann-Whitney U) with adjusted p-values was applied to test for median differences between years for each variable (Table C.2) with each lake treated as one basin, and median differences for each variable between basins over both years (Table C.1,).

Because the datasets consist of a short time frame (two years), a Mann-Kendall (τ) pairwise correlation analysis was applied to each lake as one basin, to investigate whether any correlations between the variables showed a strong positive or negative association (Appendix C). The analysis compared each variable between years to detect change over time (Table C.3), as well as compare all variables per year within each lake to assess correlation between variables. Each lake was treated as one basin (Figures C.1, C.2, C.3). The Mann Kendall test is commonly used for water quality data because the data does not need to be normally distributed or linear, the test is robust for outliers and can be used for small temporal datasets (Dixon & Chiswell, 1996; Helsel et al., 2020). A Theil-Sen slope was calculated to show rate of change of any significant trends detected. The Theil-Sen slope is closely related to Kendall's τ and *S* statistic and is useful to detect trends in non-normal data as it is resistant to multiple outliers and has better efficiency for small datasets than other regression methods (Helsel et al., 2020). The Theil-Sen slope calculates the median of all pairwise slopes and provides a value for the magnitude of the trend (Sen slope and *p* value). The Sen slope indicates the amount the variable increases per unit per year, and the *p* value is the significance of the change. Differences were considered significant at an alpha level of 0.05. Because the Mann-Kendall and Theil-Sen slope analysis are pairwise comparisons, Lake Waterhen was included in the analysis.

A Kendall correlation matrix was created to illustrate the data, showing the *tau* (whether the relationship between variables is positive or negative and the strength of the relationship, from -1 to -1) as a colour gradient, and the p-value as a significance star with * $p <= 0.05$, ** $p <= 0.01$ and *** $p <= 0.001$ in the box for each variable (Appendix C). Supplemental table 1 (Herbert, 2025b) provides the Theil-Sen values, *p* value which is closely related to *tau* and indicates the significance of the slope for the correlation matrix data. The slope shows the amount the variable (y) increases per unit per year (Helsel et al., 2020).

3.6 Principal Component Analysis

A principal component analysis (PCA) was conducted on each lake to identify the major biogeochemical components per lake and sublocation. Ellipses at the 95th percentile illustrated the spread of samples (coloured dots) for each basin. Arrows are the loadings calculated for each variable and indicate how strongly the water quality variable influences the distribution of the points (length and orientation). In order to conduct the PCA, missing data for each variable was calculated by replacing any missing values with the median of the available values (simple median imputation). Simple median imputation was used to preserve the sample size of the datasets. Data for the PCA was normalized before analysis by scaling variables to have a mean of zero and standard deviation of 1 (unit variance). Data was normalized in order to ensure each variable was weighted equally and prevent dominance by data with larger variances. Clustering was applied to examine variables for similarity between basins. After using PCA to reduce dimensionality of the data, K-means clustering was used to confirm PCA results and simplify visualization. The optimal number of clusters to use was determined using the sum of squared distances based on the PCA loading scores.

4. Results

4.1 Geospatial analysis - Objectives 1 and 2

The goal of objective 1 was to provide an overview of in-situ offshore biogeochemical and physical characteristics for the uMBGL by examining the spatial and temporal patterns of key biogeochemical variables (e.g., nitrogen, phosphorus, carbon) during the open-water season.

The aim of Objective 2 was to conduct a geostatistical analysis of spatial and temporal biogeochemical variation across the uMBGL system. In this analysis, I examined how physical and chemical characteristics differ among basins within each lake and between lakes, while also exploring correlations between hydrological and climatic factors (e.g., inflows, precipitation, temperature) and the observed spatial and seasonal variability in water quality.

To address these two objectives, descriptive statistics were calculated to establish seasonal conditions, after which non-parametric statistics were applied (Chapter 5). Data is presented throughout the entire open water season, from May to October for 2016 and 2017. Occasionally a site was sampled twice in a season (spring, summer, fall).

4.1.1 Descriptive Statistics

The mean, standard deviation, median, median absolute deviation (MAD), minimum and maximum values were calculated for each lake per year. The median and MAD were used to determine the variability in the dataset as MAD is considered more robust against outliers than the standard deviation (SD) (Appendix A) (Meals & Dressing, 2005). As the Shapiro-Wilk test indicated that the majority of water quality parameters were not normally distributed (Table B.1, Figures B1-B15), further analysis was conducted using non-parametric methods.

4.1.2 Nitrogen

Across the uMBGL total nitrogen (TN) increased from north to south, with the highest concentrations occurring in the south basin of Lake Manitoba (Figures 4.1, D.1). Total nitrogen concentration in each study year was similar, although seasonally, spring concentrations were slightly lower in 2017 (Herbert, 2025a).

4.1.2.1 Lake Winnipegosis

Over the 2016 and 2017 open-water seasons, the median TN concentration for the entire lake was 701 and 744 μg L^{-1} (± 62.0) respectively, with individual values ranging from 596.0 - 893.0 μg L^{-1}. Medians for total dissolved nitrogen (TDN), including both inorganic and organic forms, and suspended nitrogen (SuspN) were 553.0 μg L^{-1} (± 34.0) and 175.0 μg L^{-1} (± 34.0) in 2016 and 524.0 μg L^{-1} (±24.0) and 184.5 μg L^{-1} (±50.5) in 2017 (Tables A.1, A.2.). Of the constituents measured, in both years, the dissolved portion made up over 70% of the total nitrogen pool, while the particulate portion constituted 25%.

4.1.2.2 Lake Waterhen

Over the open-water season for both years, total dissolved nitrogen accounted for 80% (655 μg L^{-1} in 2016 and 611 μg L^{-1} in 2017) of the total nitrogen (825 μg L^{-1} (±60.0) in 2016 and 755 μg L^{-1} (±17.0) in 2017). Suspended nitrogen accounted for about 20% (166 μg L^{-1} in 2016 and 142 μg L^{-1} in 2017) of the total (Tables A.3, A.4).

4.1.2.3 Lake Manitoba

On Lake Manitoba, the median concentrations of total nitrogen in 2016 and 2017 were 985.0 and 904 μg L^{-1} (± 93.5 & 76.0) respectively, with individual values ranging from 490.0 - 1,843.0 μg L^{-1} (Tables A.5, A.6). Over the two years, total dissolved nitrogen comprised on average 69% of the total, with suspended nitrogen comprising the rest (334.0 μg L^{-1} in 2016 and 242.0 μg L^{-1} in 2017). Total nitrogen concentrations in each study year was similar, and the seasonal gradient was the same, with concentrations generally moving from lower to higher from north to south (Figures 4.1, D.1). In 2016, TN at the Narrows site was 60% higher in the fall and 38% higher in the summer, than the median at a station in the south basin in each respective season.

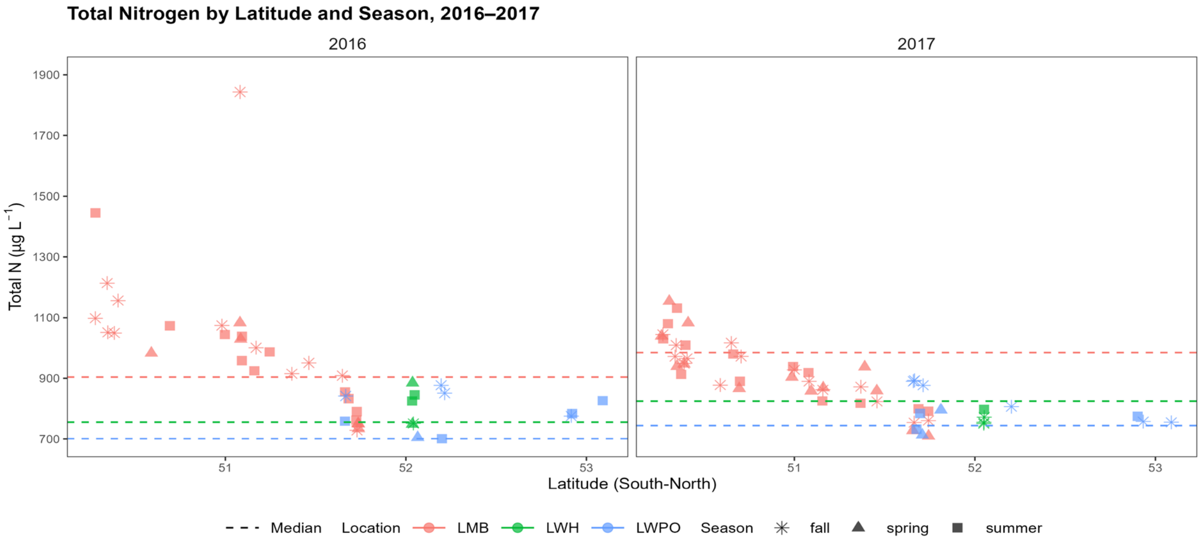

Figure 4.1. Total Nitrogen (μg L^{-1}) for upper Manitoba Great Lakes, by latitude and season. The dashed coloured lines represent the median for each lake.

4.1.3 Phosphorus

Overall, total phosphorus (TP) was highly variable in 2017 across the uMBGL, with concentrations 4 times higher than 2016 (Figures 4.2, D.2), but no discernible gradient. In 2016, TP increased slightly from north to south, with the highest concentrations occurring in the fall and summer in the south basin of Lake Manitoba (Herbert, 2025a).

4.1.3.1 Lake Winnipegosis

From 2016 to 2017, the median TP concentration increased from 20 μg L^{-1} (± 2.0) to 81 μg L^{-1} (± 59.0), with individual values ranging from 14.0 - 370.0 μg L^{-1} (Tables A.1, A.2). In 2016, particulate phosphorus (PP) made up 70% of the total phosphorus, while in 2017, PP comprised 22%.

Seasonally, in 2016 TP decreased from north to south, peaking in the fall. In 2017, phosphorus concentrations were variable throughout the season, with the highest concentrations occurring in summer and fall (Figure 4.2), generally following the same decrease in concentration from north to south, but a clear gradient is masked by the variability.

4.1.3.2 Lake Waterhen

Median TP concentrations were 21 μg L^{-1} (± 2.0) and 35 μg L^{-1} (± 20.0) in 2016 and 2017 respectively. In 2016, PP accounted for 62% of the total, while in 2017 PP accounted for 37% of TP (Tables A.3, A.4).

Seasonally, TP concentrations in both study years followed the same gradient, with concentrations being highest in the fall and lowest in the spring (Figures 4.2, D.2).

4.1.3.3 Lake Manitoba

Phosphorus concentrations were three to four times as high in 2017 versus 2016 (Figure 4.2). The median TP concentration for 2016 was 28.5 μg L^{-1} (± 11.0), with individual values ranging from 16.0 - 88.0 μg L^{-1}, while for 2017 median TP was 90.0 μg L^{-1} (± 38.0), with individual values ranging from 17.0 - 302.0 μg L^{-1} (Tables A.5, A.6). In 2016, PP comprised 67% of the TP, while in 2017, PP made up 20% of the total.

In 2017, phosphorus concentrations were variable throughout the season, masking any clear gradients, with the highest concentrations occurring in summer (Figures 4.2, D.2).

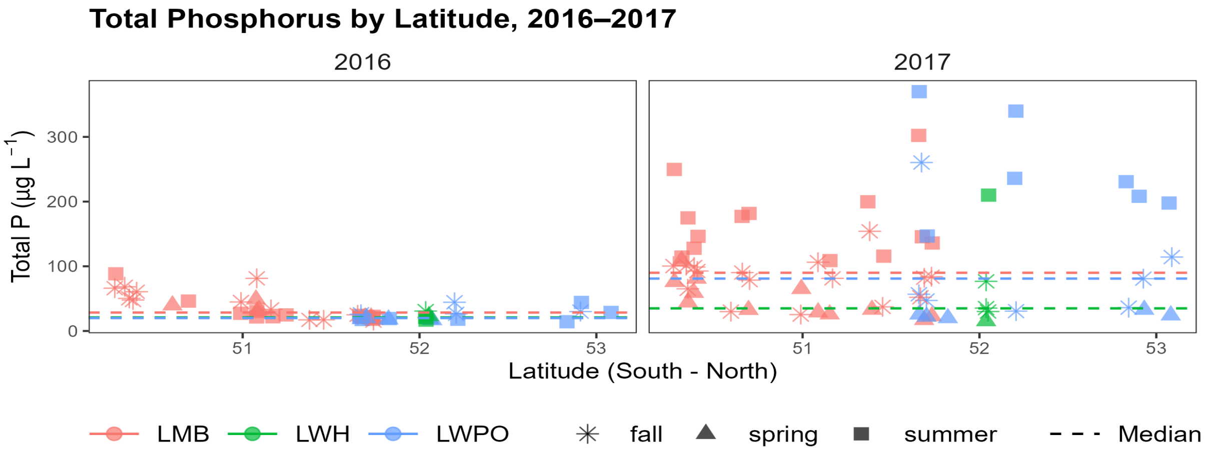

Figure 4.2. Total Phosphorus (μg L^{1}) for upper Manitoba Great Lakes by season, 2016 & 2017.The dashed coloured lines represent the median for each lake

4.1.4 Carbon

Particulate organic (POC) and dissolved organic carbon (DOC) were the forms of carbon measured for this study. Overall, POC showed a slight gradient increase from north to south, with the highest values occurring in the fall. POC follows the same gradient, decreasing from north to south as TP for 2016, but follows the opposite trend in 2017, with TP concentrations being higher in the south end of Lake Winnipegosis during the summer. DOC was more variable across the uMBGL, and decreased slightly from north to south, with the highest peaks occurring in the spring across all lakes (Figures 4.3, D.3, D.4) (Herbert, 2025a).

4.1.4.1 Lake Winnipegosis

On Lake Winnipegosis median POC was 1,610.0 μg L^{-1} (± 270.0) in 2016 and 1,470 (±325.0) μg L^{-1} in 2017. Median DOC in 2016 was 924.5 μmol L^{-1} (± 42.0) and was 1,185.0 μmol L^{-1} (± 55.0) in 2017 (Figure 4.3 (panel a), Tables A.1, A.2).

POC values were 20% higher in fall 2017 versus 2016. DOC exhibited higher variability spatially, with the highest value appearing in the south basin in summer 2016 and in the north basin in fall 2017 (Figure 4.3, panel a). PC values followed a gradient from low to high moving from the north to south end of the lake for both years. DOC had higher variability in 2016 and 2017, with a gradient similar to PC (Figures D.3, D.4).

4.1.4.2 Lake Waterhen

On Lake Waterhen, the median for POC was 1,690 μg L^{-1} (± 40.0) in 2016 and 1,430 μg L^{-1} (± 180.0) in 2017. From 2016-2017, individual POC values ranged from 1,010 μg L^{-1} to 1,810 μg L^{-1}. The DOC median concentration in 2016 was 937.0 μmol L^{-1} (± 99.0) and 1,249 μmol L^{-1} (± 34.0) in 2017, with individual values ranging from 838-2,915 μmol L^{-1} over both study years (Figure 4.3) (Tables A.3, A.4).

4.1.4.3 Lake Manitoba

POC showed a slight gradient in both years from low to high, decreasing from north to south (Figure 4.3). In 2016, median PC was 3,720 μg L^{-1} (± 900.0), and in 2017 the PC median was slightly lower at 2,755 μg L^{-1} (± 810.0) (Tables A.5, A.6). DOC was 967.50 μmol L^{-1} (± 83.5) in 2016 and 1,085.5 μmol L^{-1} (± 121.5) in 2017.

There was one very high POC value in fall 2016 (15,930 μg L^{-1}) at the Narrows (Figure 4.3 (panel a)). Any spatial gradient in DOC is masked by high variability, with DOC peaking in summer 2016 in the NE and south basins, while in 2017 DOC values were highest in the spring along the north to south gradient (Figures 4.3 (panel b), D.3, D.4).

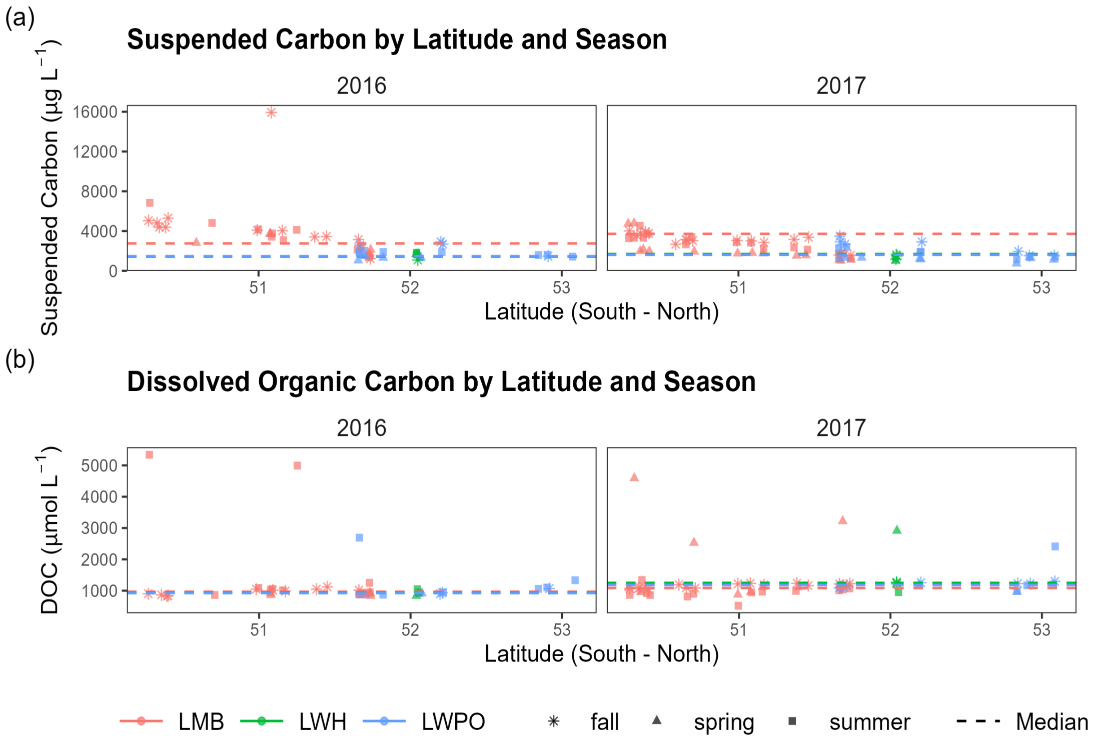

Figure 4.3. Particulate Organic Carbon (Suspended Carbon) (µg L^{1}) and Dissolved Organic Carbon µmol L^{-1} @ 25°C for upper Manitoba Great Lakes by Latitude and Season. The dashed coloured lines represent the median for each lake

4.1.5 Water Clarity

4.1.6 Secchi

Secchi depth was only measured in 2017. Overall, Secchi depth decreased across the uMBGL, with the highest values on Lake Waterhen and the lowest in the south basin of Lake Manitoba (Figures 4.4, D.5) (Herbert, 2025a).

4.1.6.1 Lake Winnipegosis

Median Secchi depth for Lake Winnipegosis was 1.51 m (± 0.48), with values ranging from 0.92 - 3.4 m. Secchi depth was highly variable over the season, though typically higher in spring, and varied highly from north to south, with no clear spatial gradient (Table A.1, A.2).

4.1.6.2 Lake Waterhen

Lake Waterhen has only two measured values for Secchi depth. The highest value occurred during the summer (3.80 m), with a spring value of 2.91 m (Figure 4.4, Table A.3, A.4).

4.1.6.3 Lake Manitoba

Lake Manitoba median Secchi depth was 0.90 m (± 0.26), with values ranging from 0.44 m to 2.48 m (Table A.5, A.6). Secchi depth decreased in a north-south gradient, with the highest values occurring in the spring (Figure 4.4).

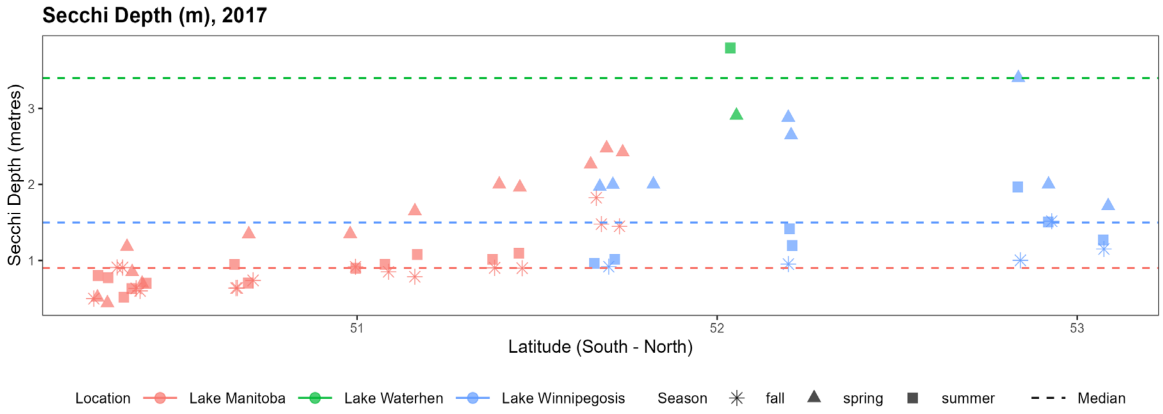

Figure 4.4. Secchi Depth (m) in upper Manitoba Great Lakes by Latitude and Season, 2017. The dashed coloured lines represent the median for each lake

4.1.7 Suspended Solids

Over the uMBGL, TSS (Figure D.6) and FSS (Figure D.7) exhibited a spatial gradient from low to high, moving north to south (Figure 4.5). The gradient was less clear with VSS (Figure D.8) as there is more variability in the data. In 2017 the VSS showed a slight gradient, moving from higher to lower from north to south, the opposite of TSS and FSS gradients (Herbert, 2025a).

4.1.7.1 Lake Winnipegosis

For 2016 there is no TSS (a), FSS (b) or VSS (c) data for the north basin of Lake Winnipegosis, therefore spatial gradient patterns were based on 2017 data (Figure 4.5). In 2016, medians for TSS, VSS and FSS in the south basin were 3.2 mg L^1 (± 1.2), 0.25 mg L^1 (± 0.10) and 2.5 mg L^1 (± 0.5) respectively, and in 2017 values were 4.25 mg L^1 (± 4.0), 0.87 mg L^1 (± 0.21) and 0.67 mg L^1(± 0.67) (Tables A.1, A.2).

4.1.7.2 Lake Waterhen

TSS, VSS and FSS medians were two to three times higher in 2016 versus 2017; TSS = 4.1 mg L^1 (± 1.17), VSS = 0.45 mg L^1 (± 0.25) and FSS = 2.27 mg L^1 (± 1.35) in 2016, decreasing to TSS = 1.17 mg L^{-1} (±1.17), VSS = 0.83 mg L^{-1} (±0.02) and FSS = 0.5 mg L^{-1} (±0.5) (Tables A.3, A.4). Overall, TSS, FSS and VSS were highest in the spring, with the exception of VSS in 2016 where summer had the highest concentration.

4.1.7.3 Lake Manitoba

TSS and FSS followed a similar gradient, increasing from north to south, while the VSS gradient moved generally from higher to lower in 2017, with no clear gradient in 2016, possibly due to the smaller dataset size. The median TSS in 2016 was 9.0 mg L^{-1} (±3.20), FSS was 0.44 mg L^{-1} (±0.36) and VSS was 5.10 mg L^{-1} (±3.77). In 2017, median TSS, FSS and VSS values were 7.17 (±3.5) , 0.69 (±0.16) and 2.83 mg L^{-1} (±1.33) (Tables A.5, A.6).

In 2017 TSS in the spring at two stations in the south basin were 120 - 143% higher than the median. In the NW basin at the mooring site, TSS in the fall of 2017 was 143 % higher than the median, while values were lowest in the narrows and in the south basin (Figure 4.5).

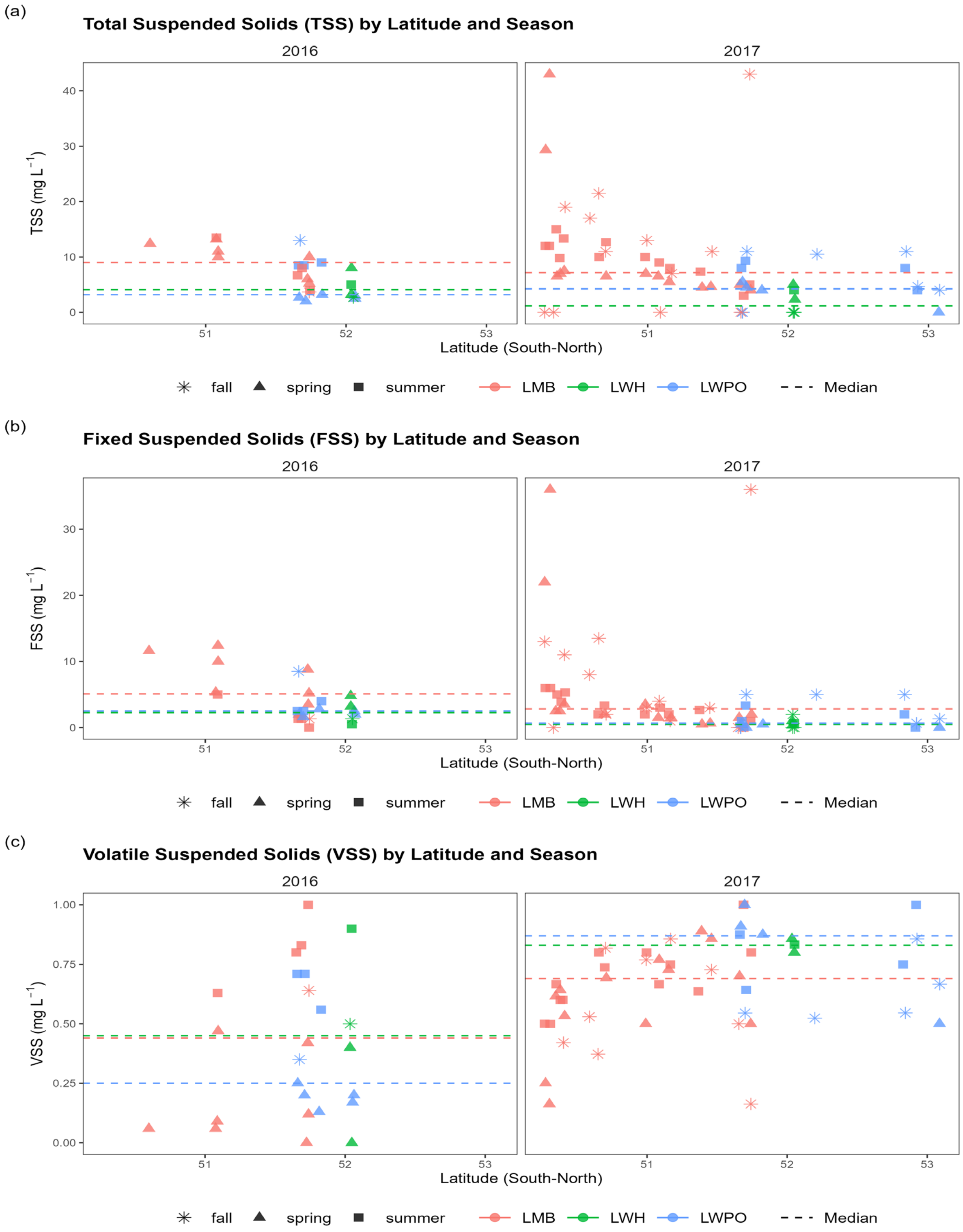

Figure 4.5. Suspended solids (TSS,FSS,VSS) (mg L^{1}) in the upper Manitoba Great Lakes by latitude and season, 2016-2017. The dashed coloured lines represent the median for each lake

4.1.8 Conductivity

Specific Conductivity (conductivity at 25ºC) is used in this report as a proxy for salinity (McCleskey et al., 2025; Talling, 2009). Overall, the specific conductivity (COND) gradient across the uMBGL was more pronounced in 2017 vs 2016, with concentrations lowest at the north end of Lake Winnipegosis (Dawson and Overflowing Bay) and increasing from north to south (Figures 4.6, D.9) (Herbert, 2025a).

4.1.8.1 Lake Winnipegosis

Specific conductivity was highly variable throughout the seasons (Figures 4.6, D.9). The north end of Lake Winnipegosis contained the lowest conductivity values, with a median value in 2016 of 999.0 µS cm^{-1} (± 34.0) and 931.50 µS cm^{-1} (± 49.0) in 2017. Individual conductivity values ranged from 731.0 to 1,044 µS cm^{-1} in 2016 and 559.0 to 1052.0 µS cm^{-1} in 2017 (Tables A.1, A.2).

4.1.8.2 Lake Waterhen

Specific conductivity followed a seasonal gradient, with concentrations lowest in spring and highest in the summer in 2016 and fall in 2017 (Figure 4.6). Conductivity median in 2016 was 905 µS cm^{-1} (± 30.0) and 942.5 µS cm^{-1} (± 17.5) in 2017 (Tables A.3, A.4).

4.1.8.3 Lake Manitoba

In 2017 Lake Manitoba conductivity concentrations were lowest in the spring at the north basin and highest in the south basin (Figure 4.6). Specific conductivity median was similar between years, with a value of 951.0 µS cm^{-1} (± 25.5) in 2016 and 959.5 µS cm^{-1} (± 32.5) in 2017 (Tables A.5, A.6).

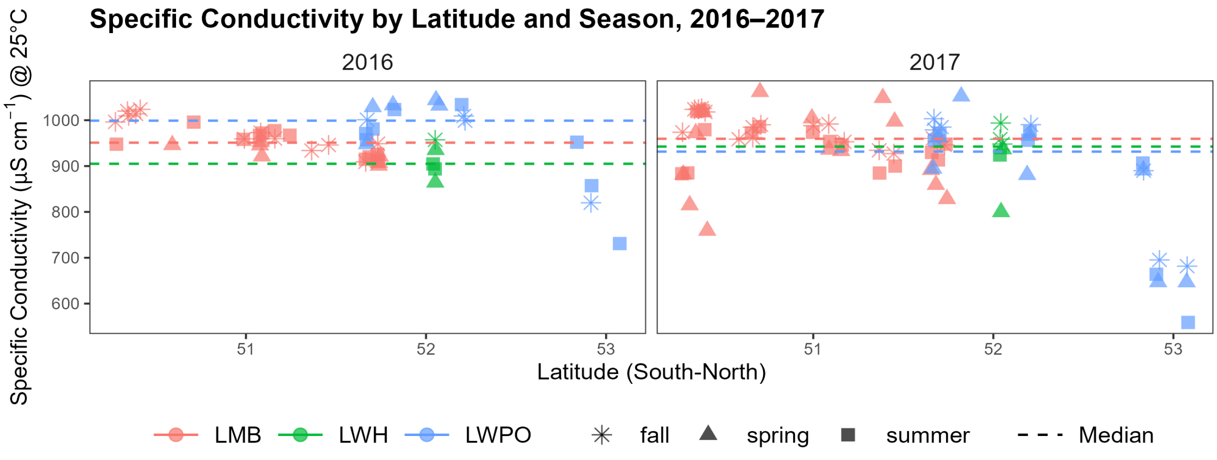

Figure 4.6. Specific Conductivity µS cm^{-1} @ 25°C for upper Manitoba Great Lakes by latitude and season. The dashed coloured lines represent the median for each lake

4.1.9 Chlorophyll

Overall, chlorophyll values in the uMBGL exhibited a north-south gradient from low to high, with the exception that in the fall of both years Lake Winnipegosis chlorophyll concentrations were higher in the north than the south (Figure 4.7).

4.1.9.1 Lake Winnipegosis

Chlorophyll data were obtained using both the GF method (Total Chlorophyll (TChl)), and the HPLC method (Chlorophyll-a (Chl a)). Descriptive statistics for both are reported in Appendix A. Chl a values were half the concentration of TChl, however both datasets follow the same spatial and temporal gradient, so only TChl will be discussed in further detail. Median TChl concentrations were 4.86 μg L^{-1} (± 1.97) in 2016 and 6.38 (± 2.59) μg L^{-1} in 2017. Overall, individual TChl values ranged from 1.71 - 15.67 μg L^{-1} over both study years. Although TChl in Lake Winnipegosis was variably distributed, the highest concentrations generally occurred in the fall in both years. Concentrations trended from higher to lower from north to south (Tables A.1, A.2, Figure 4.7).

4.1.9.2 Lake Waterhen

Of the three uMBGL, Lake Waterhen contained the lowest chlorophyll concentrations, with a median in 2016 of 3.21 (± 0.69) μg L^{-1} and 2.76 (± 0.35) μg L^{-1} in 2017. Overall, in both years, chlorophyll did not differ significantly, with individual values ranging from 1.52-4.16 μg L^{-1} (Tables A.3, A.4). Seasonally values showed no discernible pattern, however the sample size was small (6) and therefore it is difficult to detect changes (Figure 4.7).

4.1.9.3 Lake Manitoba

Lake Manitoba chlorophyll values had the highest spatial and temporal variation of the three lakes. Median concentrations for both years was around 7.8 μg L^{-1} (± 3.83-4.30) (Tables A.1, A.2), however individual values ranged from 1.59 - 38.74 μg L^{-1} in 2016 and from 2.16 - 29.84 μg L^{-1} in 2017. Seasonally, chlorophyll concentrations in 2016 were higher in the fall, while in 2017 the temporal gradient is masked by higher variability in the data. 2016 also showed a peak chlorophyll value 400% higher than the median. Figure 4.7 shows TChl values for 2016 and 2017 with the dashed line showing the median for each year. Chl a values exhibit a similar pattern.

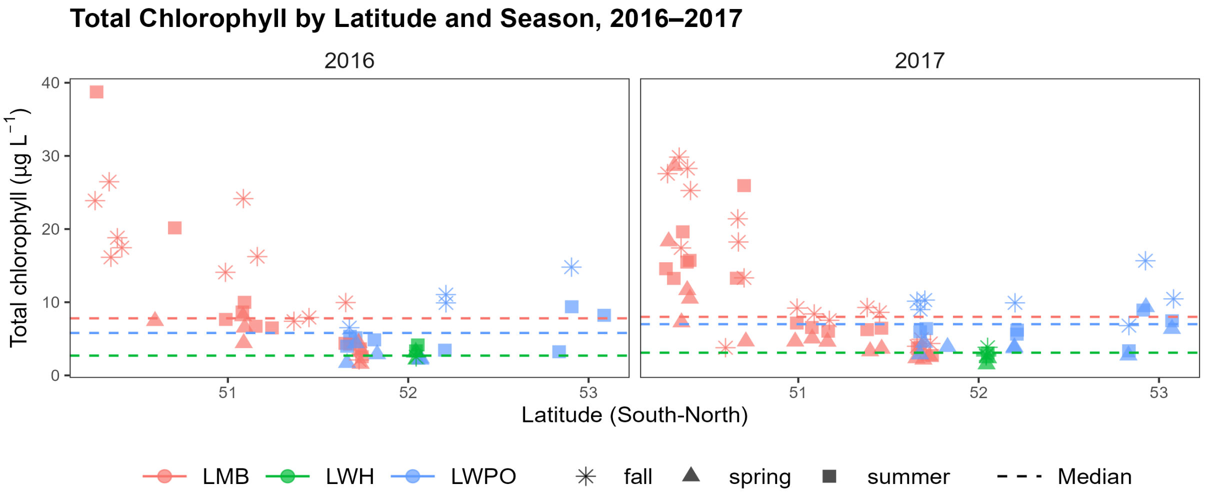

Figure 4.7. Total Chlorophyll (μg L^1) for upper MBGL lakes by season, 2016 & 2017. The dashed coloured lines represent the median for each lake

4.2 Phytoplankton diversity - Objective 3

The goal of Objective 3 was to provide an overview of in-situ offshore phytoplankton diversity for Lakes Winnipegosis and Manitoba. This was accomplished by examining the relationship between nutrient concentrations and phytoplankton diversity and biomass, assessing whether patterns of nutrient stoichiometry (OC:N, N:P ratios) indicated nutrient limitation and how the results influenced phytoplankton community composition.

Two sites from Lake Winnipegosis (2016 & 2018) and six sites from Lake Manitoba (2016-2017) were counted and identified to species (Figure 4.8). Lake Manitoba samples in the north basin (NB) were pooled to represent the entire basin, while along the west side of the south basin, a transect of individual samples were counted (S3, S4, S6, S7). The combined samples for the north basin of Lake Manitoba (LMB NB 2016 and 2017) were composed of stations GL_LMB_M (mooring), MB168935 and MB165947 (NW basin). Samples from one inflow (Swan River) in the north and one outflow (Fairford River) in the south were also counted. For this study, microbluegreens, also commonly called picocyanobacteria, are algae with cells between 0.2 to 3.0 µm (Jakubowska & Szeląg-Wasielewska, 2015). On Lake Winnipegosis, 65 unique taxa were counted, while on Lake Manitoba there were 95 unique taxa.

4.2.0.1 Lake Winnipegosis

Both basins of Lake Winnipegosis were dominated by cyanobacteria. By biomass, in August 2016, Cyanobacteria comprised 51% of the total, with *Microcystis* sp. dominating, followed by a dinoflagellate (*Ceratium* sp.) in the north basin site. The filamentous genera *Planktothrix* was the seventh highest biomass, and the second dominant cyanobacteria. Plastic green fiber was also found in the north basin sample (Dawson Bay), with a biomass of 125.7 mg m^{-3} (17% of total algal biomass) but is not included in the phytoplankton biomass graph (Figure 4.8). By abundance, microbluegreens dominated.

In the south basin site in 2018, filamentous cyanobacteria dominated (*Planktolyngbya* spp.), followed by micro and small bluegreens (*Cyanodictyon* spp.) (Figure 4.8). In both basins, by abundance, microbluegreens and smaller blue green taxa (< 3.5 microns) dominated, comprising almost 97% of the population. In addition, *Aphanizomenon* heterocysts were present in the south basin. Biomass to abundance ratios were similar in 2016 at the south basin station (57% biomass to 92% abundance).

The Swan River site in May 2017 was dominated by plastics (785.4 mg m^{-3}), which is not included on the plots. The distribution of algal biomass at the site was dominated by diatoms (61%) (mainly *Tryblionella* sp. *and Nitzschia* sp.), with chrysophytes (*Ochromonads* spp.) making up 28% and chlorophytes (*Koliella* sp., *Monoraphidium minutum*) 11%.

4.2.0.2 Lake Manitoba

In the summer of 2016, by biomass, the Lake Manitoba north basin was dominated by chrysophytes (*Chrysamoeba* sp.), followed by filamentous cyanobacteria (*Pseudanabaena* sp.*,* *Planktolyngbya* spp.*)* but by abundance was dominated by microgreens and microbluegreens. In summer 2017 in the north basin, microbluegreens dominated by biomass and abundance, with filamentous cyanobacteria (*Planktolyngbya* spp.) occurring as the second dominant genus by biomass. *Dolichospermum* spp. and *Aphanizomenon* spp. heterocysts were detected in the summer 2016 combined sample but not in 2017. In the south basin a transect on the west side of the lake in August 2017 was counted. Sites S6 and S7 are near the west side of the lake, near MB027854, while Sites S3 and S4 are near MB030847 (Figure 2.7). By biomass, all sites were dominated by filamentous cyanobacteria (e.g. *Planktolyngbya* spp., *Cuspidothrix issatchenkoi*, *Aphanizomenon* sp.). By abundance, microbluegreens dominated, except at site S7 where *Coelospaerium sp., Planktolyngbya* spp*,* and *Woronochinia* sp. co-dominated. *Dolichospermum* spp. and *Aphanizomenon* spp. heterocysts were also detected in all the south basin samples.

The Fairford river biomass composition was similar to the northeast basin of Lake Manitoba, being comprised mainly of filamentous cyanobacteria (*Planktolyngbya* spp.), followed by microbluegreens and other picoplankton (e.g. *Cyanodictyon reticulatum*). By abundance microbluegreens dominated this site also. *Dolichospermum* spp. heterocysts and akinetes were also present.

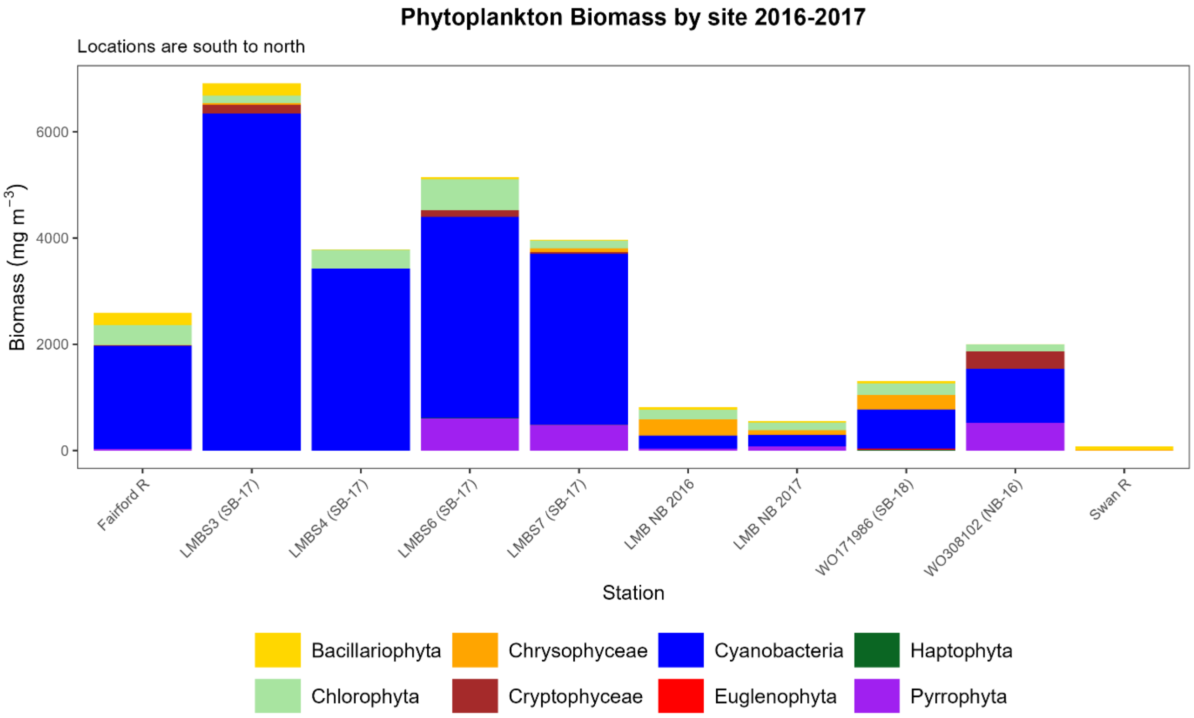

Figure 4.8. Phytoplankton biomass distribution from north to south along the upperMBGL. Except for the Fairford River (upperMBGL outflow) station, stations are sorted from left-right and south-to-north

5. A comparison of biogeochemical and physical characteristics of the upperMBGL - Objective 1 Discussion

5.1 Climatological conditions and flows

Hydrologic and meteorological analysis was conducted using the r packages *fasstr* (Goetz & Schwarz, 2023) and *tidyhydat* (Albers, 2017) for hydrologic data and *weathercan* (LaZerte & Albers, 2018) for weather. Temperature and total precipitation between 2016 and 2017 exhibited similar patterns at the six weather stations (Grand Rapids, Swan River, Mafeking, Dauphin, McCreary, and Oak Point) closest to the upper MBGL. Total precipitation (mm) from ECCC weather stations is reported here as the sum of total rainfall and the water equivalent of total snowfall as it is the parameter most frequently reported in the ECCC dataset. Climate normals for all weather stations within the upper MBGL reported similar minimum and maximum daily temperature and precipitation, so The Pas was used as a representative station for graphing (Table 2.6). Temperature and precipitation data for the study years were compared to the climate normals for 1991-2020 The Pas weather station (Figure 2.9). Precipitation at the Oak Point weather station was consistently above the climate normal throughout the year, with both winter and fall in 2016 and winter and spring in 2017 recording peaks that were double the monthly average precipitation climate normal (80 mm) (Figure 5.1). In fall 2016, the Dauphin weather station also recorded a precipitation peak almost twice the climate normal. The McCreary station reported precipitation events in spring and fall 2016 and fall 2017 above the climate norm. Both winters were dry for Dauphin and McCreary weather stations and wet for Oak Point. Weather stations at Grand Rapids, Mafeking and Swan River in spring and summer 2016 and summer and fall 2017 reported peaks double the climate normal (Figure 5.2). Both winters were dry, with total precipitation of less than 50 mm each month. Monthly mean temperatures (°C) at all stations were within daily maximum and minimum climate normals (Figure 5.3).

Of the Water Survey of Canada hydrometric stations on tributary rivers, only the Mossy, Waterhen, Fairford and Dauphin Rivers (Figure 2.9) provide continuous flow and level records. On Lake Winnipegosis, in both study years, three of the four inflows (Overflowing (Figure 5.5), Red Deer (Figure 5.6) and Mossy (Figure 5.7)), were at or above the 90th percentile flow (P90), indicating high flow conditions relative to the long term mean annual discharge (MAD). In 2016, the Overflowing and Red Deer Rivers experienced three seasonal peaks (spring, mid-summer and fall), while in 2017 the Red Deer River peaked in the spring, with a gradual decrease into the winter. The Overflowing River peaked mid-July. The Mossy River's peak flow occurred in fall 2016, with 2017 high flows occurring in spring 2017. The 2017 peak flows followed the high precipitation events recorded at the Grand Rapids, Mafeking and Swan River weather stations (Figure 5.11). Flows at the North Duck River (Figure 5.9) were above the MAD in 2016 and below in 2017, with spring and summer peaks occurring later in 2016 than 2017, and no fall peak in 2017.

In the south end of Lake Manitoba at the Whitemud River (Figure 5.10), flows were above the MAD, but well below the P90. There was one high peak in the flow in 2017 (spring), which coincided with the Assiniboine River Diversion (ARD) peak flow (Figure 5.12). In 2016, the Waterhen River exhibited higher flow in the fall, and during 2017 the flow was above P90 and increased at the same time as the spring peak at the Whitemud and ARD, but due to storage in Lake Winnipegosis remained higher until decreasing in late summer.

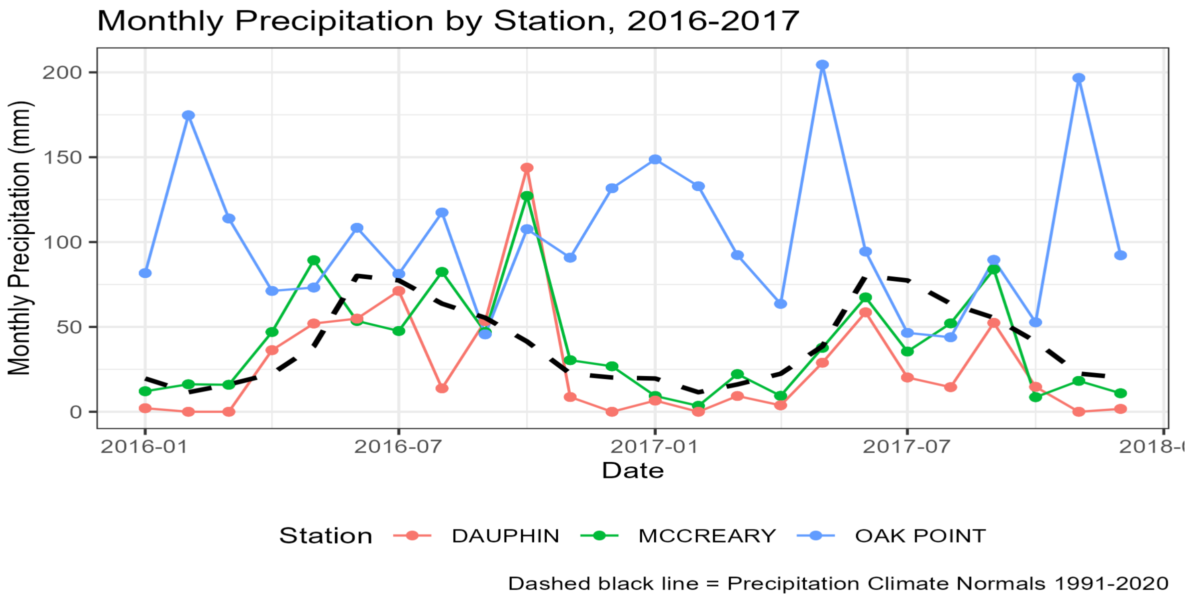

Figure 5.1. Monthly Precipitation by Station (2016-2017) with The Pas Climate Normals 1991-2020

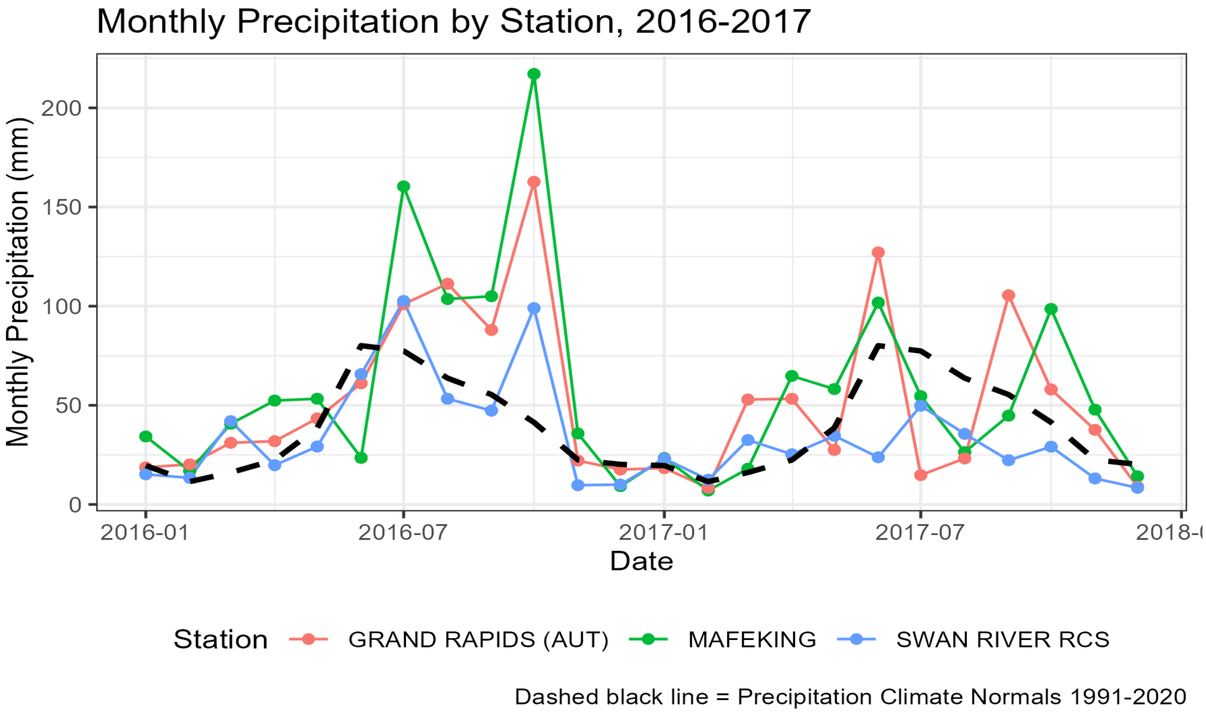

Figure 5.2. Monthly Precipitation by Station (2016-2017) with The Pas Climate Normals 1991-2020

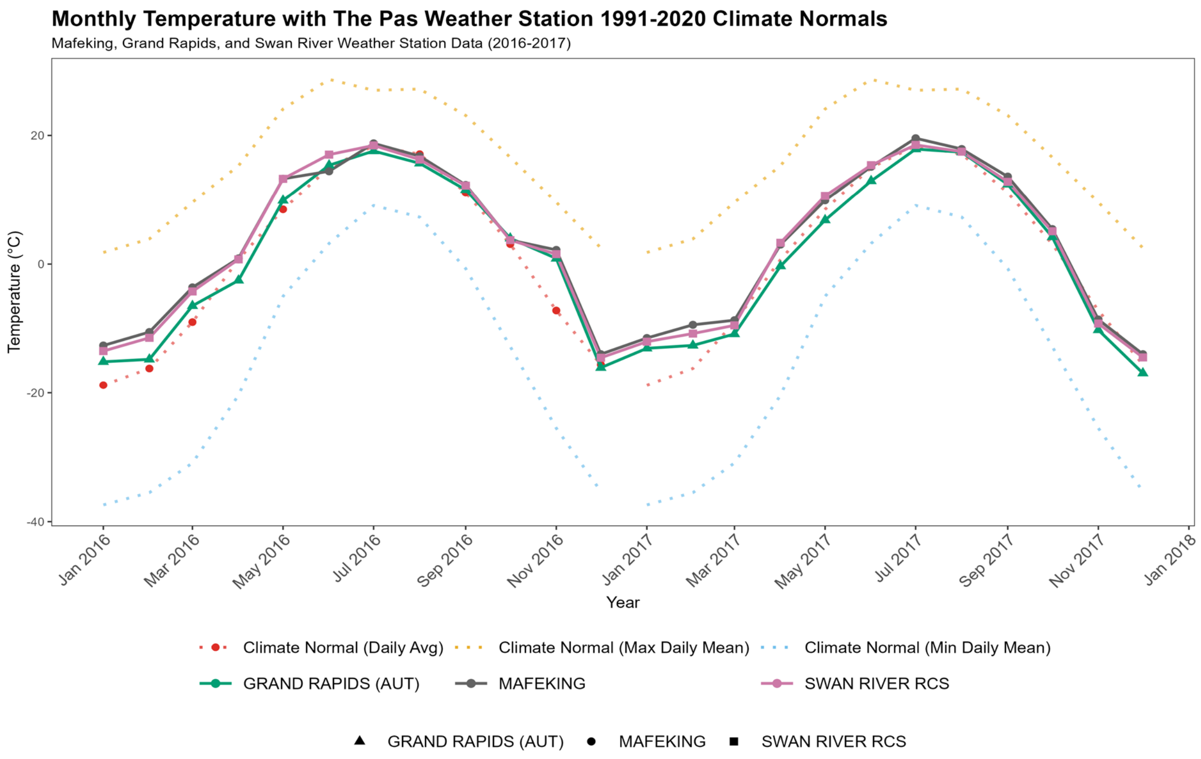

Figure 5.3. Monthly Mean Temperatures (°C), north section of upperMBGL, with average, minimum and maximum daily climate normal as dashed lines (1991-2020)

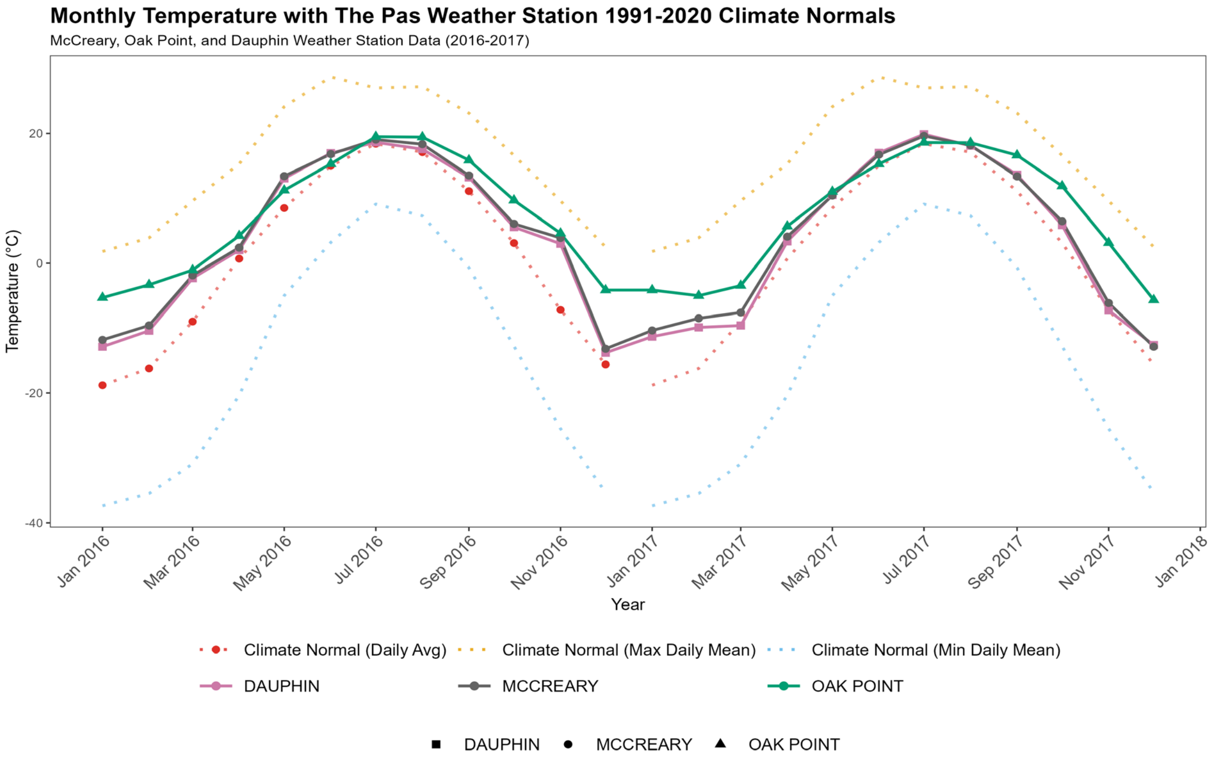

Figure 5.4. Monthly Mean Temperatures (°C), south section of upperMBGL, with average, minimum and maximum daily climate normals as dashed lines (1991-2020)

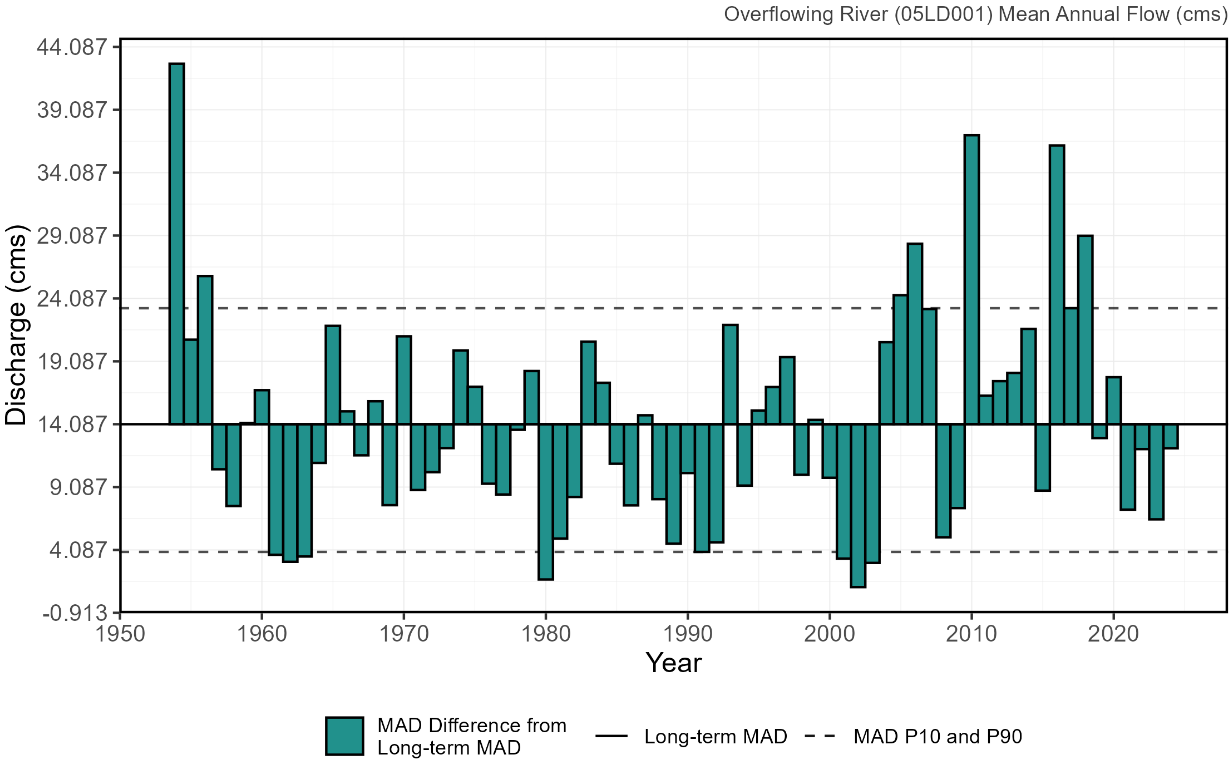

Figure 5.5. Overflowing River Annual Mean discharge plotted against the long-term annual mean

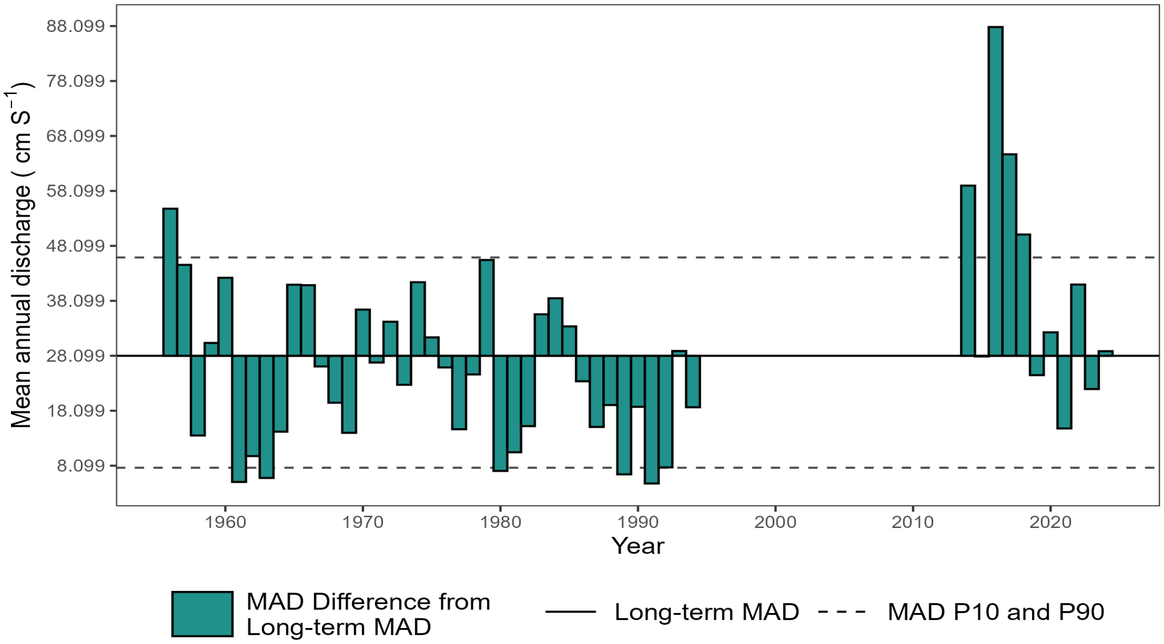

Figure 5.6. Red Deer River Annual Mean discharge plotted against the long-term annual mean

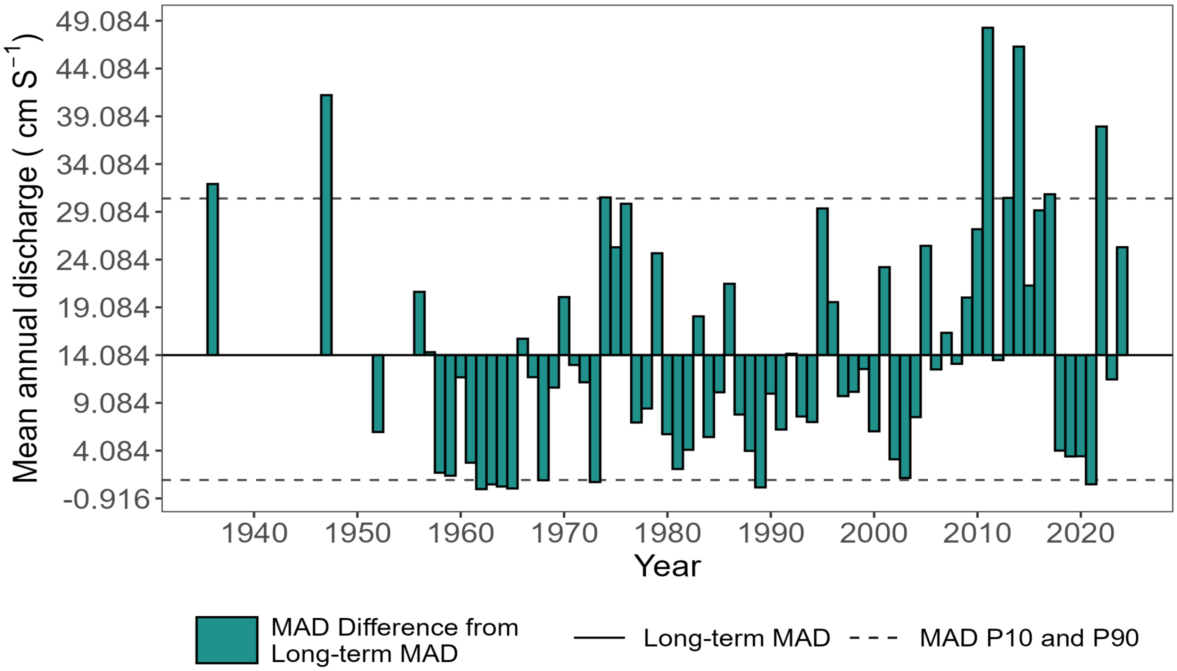

Figure 5.7. Mossy River Annual Mean discharge plotted against the long-term annual mean

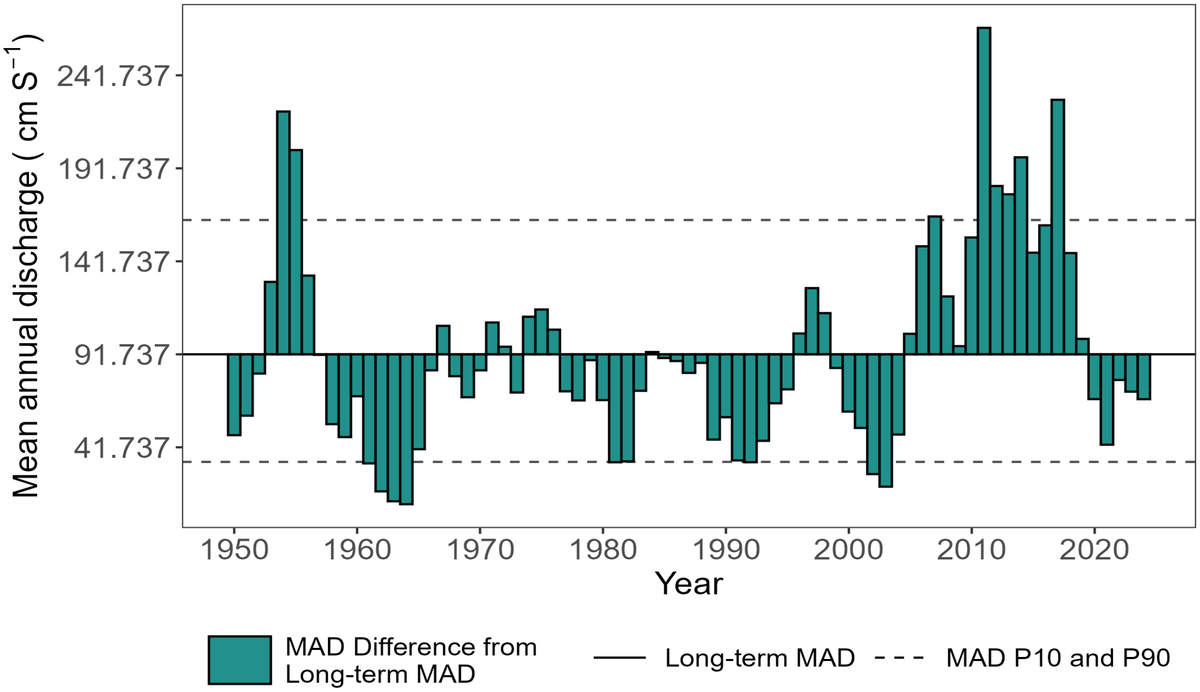

Figure 5.8. Waterhen River Annual Mean discharge plotted against the long-term annual mean

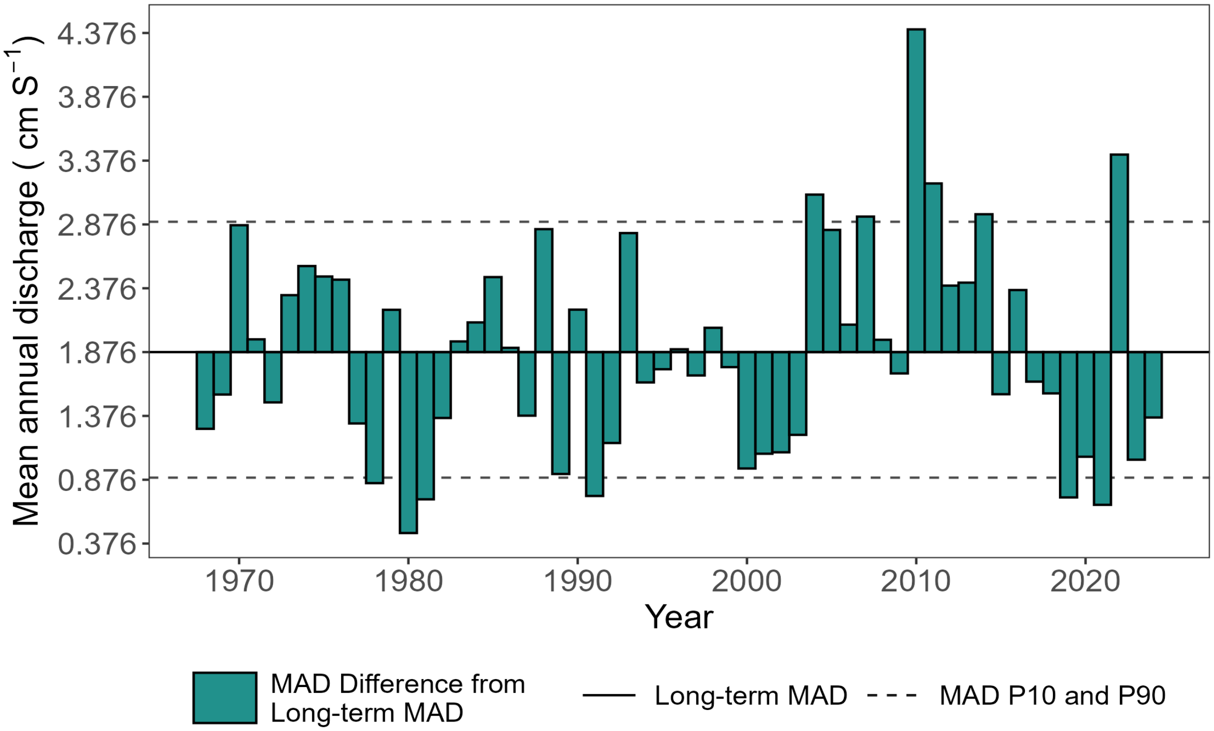

Figure 5.9. North Duck River Annual Mean discharge plotted against the long-term annual mean

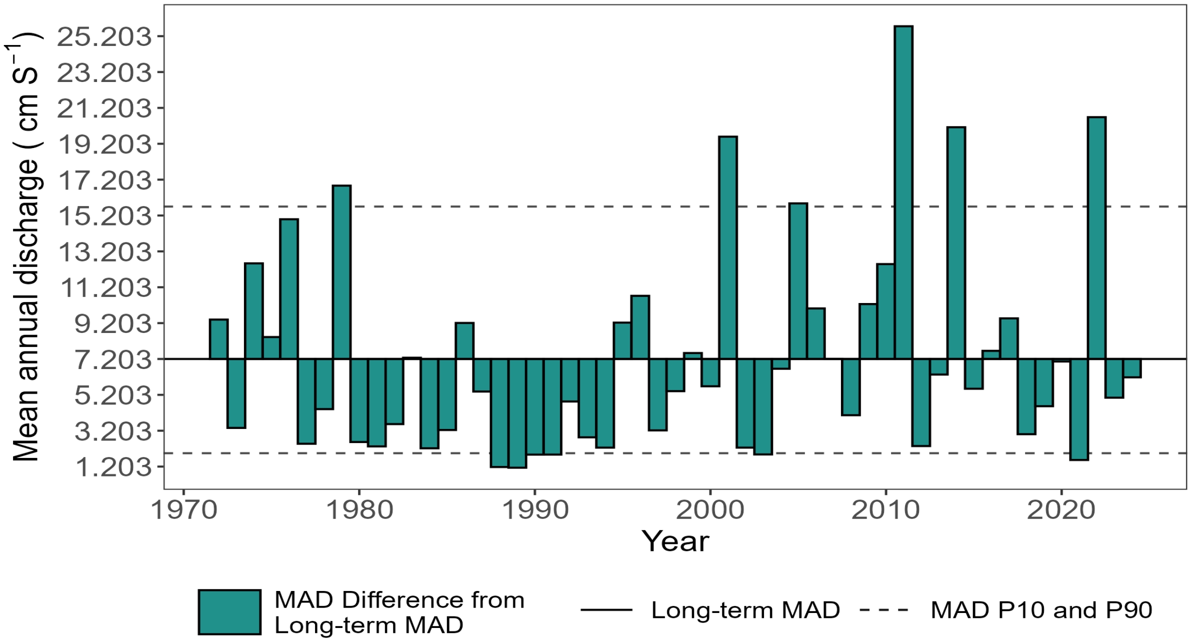

Figure 5.10. Whitemud River Annual Mean discharge plotted against the long-term annual mean

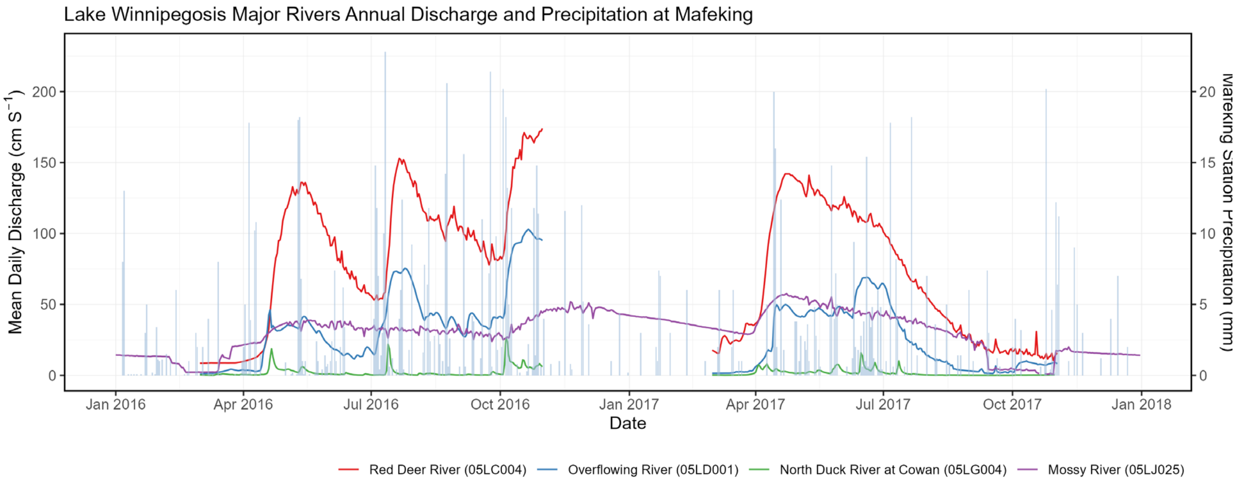

Figure 5.11. Daily flow and Precipitation - Lake Winnipegosis Major Rivers and Mafeking Weather Station, 2016 and 2017

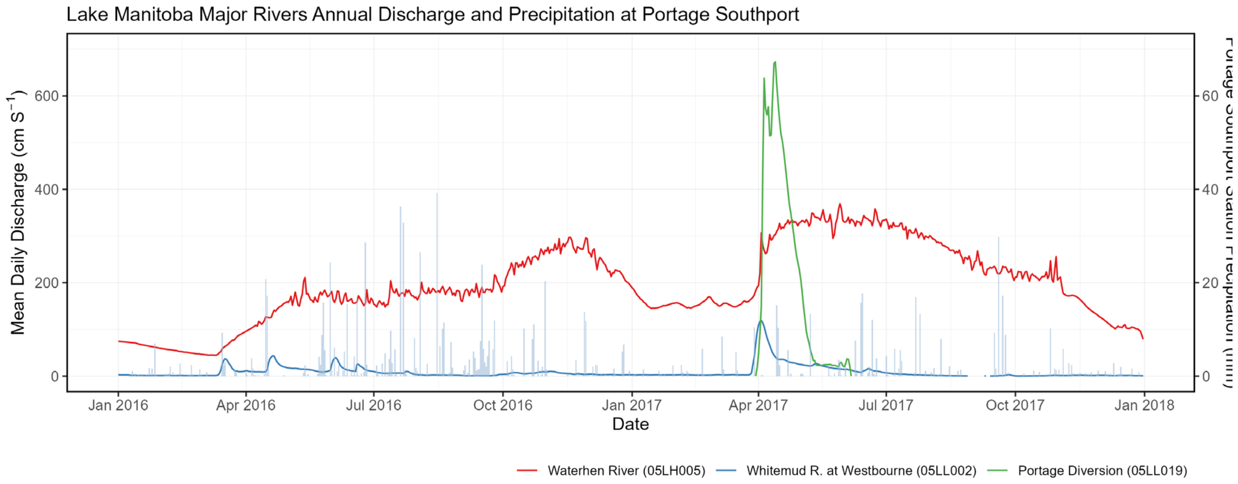

Figure 5.12. Daily flows, Whitemud and Waterhen Rivers and Assiniboine River Diversion (ARD)

5.2 *Non-parametric statistics*

Lake Waterhen was not used in the Wilcoxon Rank Sum Test as there was only one sample location. Differences were considered significant at an alpha level of ≤ 0.05. Significant differences were detected in Lakes Winnipegosis and Lake Manitoba both between years (Figure 5.13) and between basins (Figure 5.14).

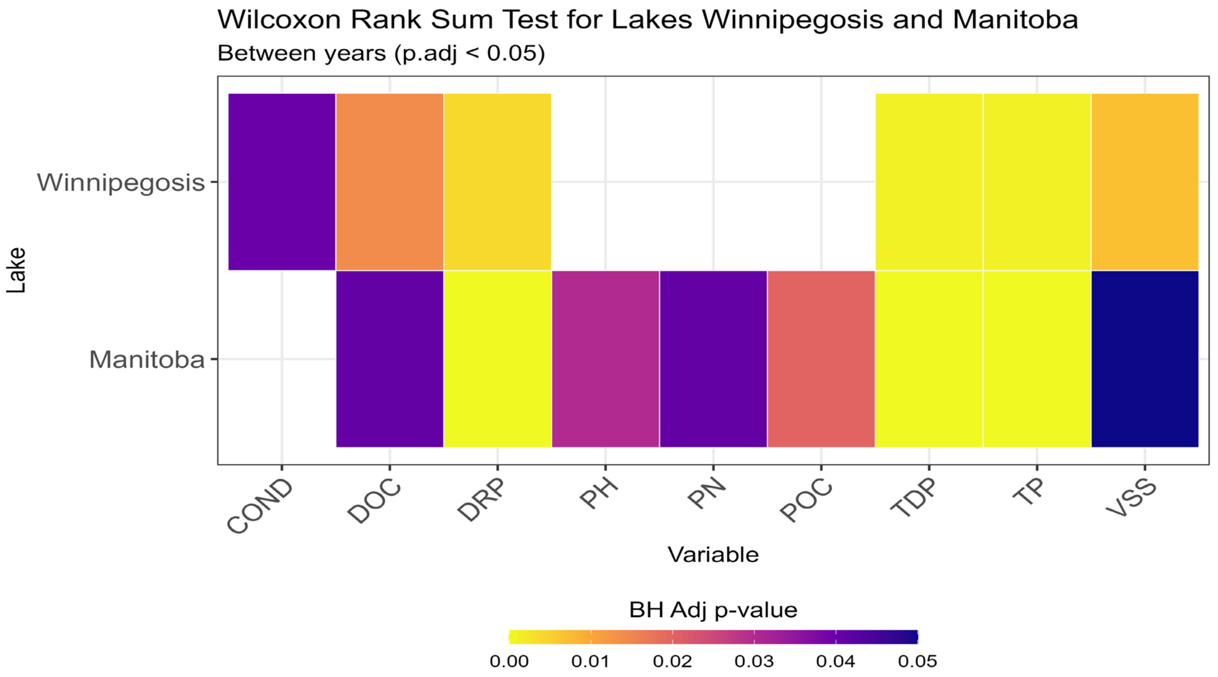

Figure 5.13. Wilcoxon Rank Sum Test for Lakes Winnipegosis and Manitoba between years. White spaces indicate a $p > 0.05$. Variables that had a $p > 0.05$ in both years are not shown

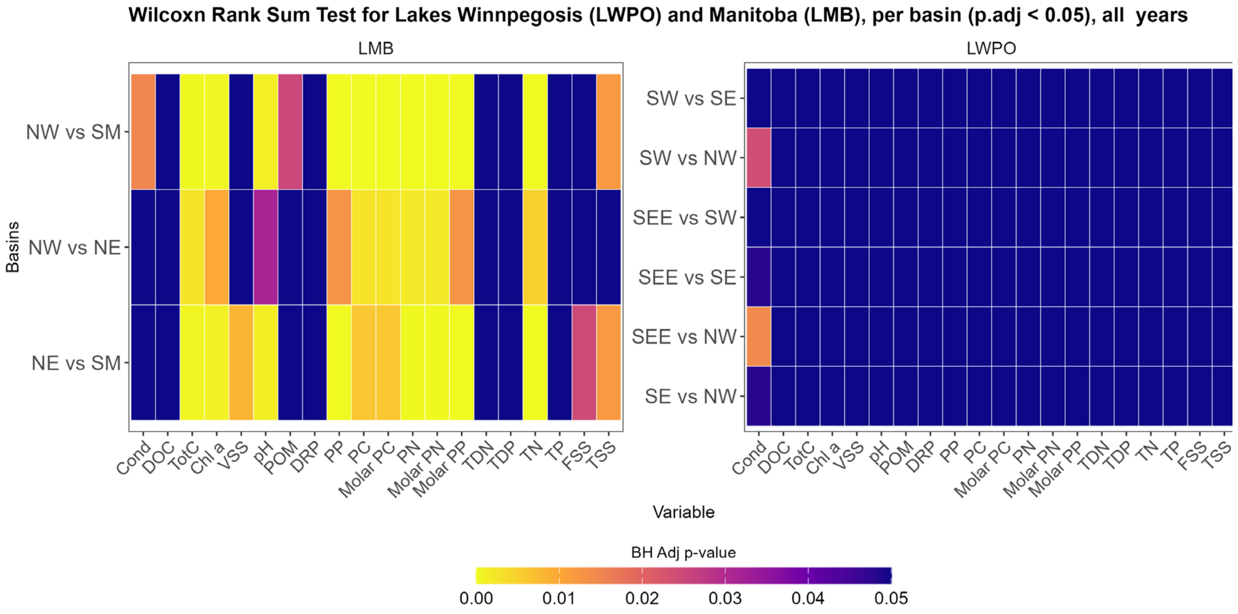

Figure 5.14. Wilcoxon Rank Sum Test for differences between basins in Lakes Winnipegosis and Manitoba, using data for all years. Only variables with p values > 0.05 are shown

In the Mann-Kendall correlation matrix, previous abbreviations for variables were used and are described in Appendix C. Matrixes for each lake are shown in Figures C.1, C.2 and C.3. The Mann-Kendall (τ) and p values and Theil-Sen slope were calculated for each variables to assess correlations and magnitude of change between variable pairs for each year and lake, and for each variable between years for each lake (Table C.3) (Herbert, 2025b). There were only 4 variables between the three lakes that showed significant differences in trends between years. On Lakes Manitoba and Winnipegosis, only DOC showed a significant trend, increasing between years ($p \leq 0.05$) for Lake Winnipeg and $p \leq 0.001$ for Lake Winnipegosis. On Lake Waterhen, TSS and FSS ($p \leq 0.05$) showed a significant decreasing trend between years and DRP showed a significant increasing trend between years ($p \leq 0.05$) (Table C.3).

5.2.1 Nitrogen

The wilcoxon rank sum test showed no difference in any nitrogen form between years for Lake Winnipegosis, but there was a significant correlation ($p \leq 0.05$) between basins for all forms (total, dissolved, suspended). On Lake Manitoba, suspended nitrogen (PN) differed significantly between years ($p \leq 0.04$) (Figure 5.13) and differed significantly ($p \leq 0.05$) to very significantly between basins ($p \leq 0.001$), while TN ($p \leq 0.001$) and PN ($p \leq 0.005$) differed highly between basins in both years. TDN differed highly between the north basins and the NW-south basin in 2016 ($p \leq 0.001$) but not 2017 (Tables C.1, C.2).

Mann-Kendall test results showed that on Lake Winnipegosis, TN (total_n) was positively correlated to particulate nutrients (suspended nitrogen, phosphorus and carbon, total dissolved nitrogen and algal biomass (TChl, Chl a) with high significance ($p \leq 0.001$) (Figure C.1). TDN was significantly positively correlated ($p \leq 0.05$) to PN. On Lake Waterhen, TN was highly significantly correlated with TDN ($p \leq 0.01$), and significantly correlated with algal biomass ($p \leq 0.05$). Particulate nitrogen (PN) and suspended carbon were very significantly correlated ($p \leq 0.001$). All correlations were positive (Figure C.2). Total nitrogen on Lake Manitoba was very highly positively correlated with suspended nutrients (N, P, C), algal biomass, TSS, conductivity and pH ($p \leq 0.001$) and negatively correlated with VSS ($p \leq 0.05$) (Figure C.2). TDN and PN were significantly positively correlated ($p \leq 0.05$) (Figure C.3). Between years the Mann-Kendall test did not detect any significant trend in any nitrogen form for any lake (Table C.3).

5.2.2 Phosphorus

The wilcoxon rank sum test showed that Lakes Winnipegosis and Manitoba followed the same pattern between years for total and dissolved forms of phosphorus. Both were highly significantly different from each other between years ($p \leq 0.01$). On Lake Winnipegosis, all forms of phosphorus were significantly different ($p \leq 0.05$) between basins, while on Lake Manitoba TP, TDP and SRP were significantly different between basins ($p \leq 0.05$) and PP was highly significantly different ($p \leq 0.01$) (Tables C.1, C.2).

Mann-Kendall test results on Lake Winnipegosis showed TP was very highly significantly correlated to PP, TDP, DRP, PN and algal biomass ($p \leq 0.001$), highly significantly correlated to PN, VSS and pH ($p \leq 0.01$), and negatively significantly correlated to FSS and conductivity ($p \leq 0.05$). PP was also very highly correlated to PN ($p \leq 0.001$) and significantly correlated to the dissolved portions of DRP and TDN ($p \leq 0.05$) (Figure C.1). TP on Lake Waterhen was highly significantly correlated to the dissolved components of phosphorus (DRP, TDP $p \leq 0.01$) (Figure C.2). There was a significant increasing trend ($p \leq 0.05$) for DRP between years on Lake Waterhen (Figure C.2). TP on Lake Manitoba was very highly significantly correlated to dissolved and particulate forms of phosphorus (DRP, TDP, PP), algal biomass and pH ($p \leq 0.001$). PN was significantly correlated to TDN and TP ($p \leq 0.05$) (Figure C.3).

5.2.3 Carbon

The wilcoxon rank sum test showed that particulate organic carbon (POC) did not differ between years on Lake Winnipegosis but did differ significantly between basins ($p \leq 0.05$) (Table C.2). DOC was highly significantly different between years ($p \leq 0.01$), and significantly different between basins ($p \leq 0.05$) (Table C.1). On Lake Manitoba, POC differed very significantly ($p \leq 0.01$), and DOC differed significantly between years ($p \leq 0.05$). POC and DOC differed very significantly between all basins on Lake Manitoba ($p \leq 0.01$) (Tables C.1, C.2).

Mann-Kendall test results on Lake Winnipegosis showed POC was very significantly highly correlated ($p \leq 0.001$) to other particulate nutrients (nitrogen and carbon) as well as algal biomass and TN. TSS and conductivity were highly significantly correlated ($p \leq 0.01$) with POC, while the suspended solids FSS and POM as well as pH were significantly correlated ($p \leq 0.05$) (Figure C.1). DOC was very significantly negatively correlated ($p \leq 0.01$) with conductivity and significantly correlated with DRP ($p \leq 0.05$) (Figure C.1). Over the two-year sampling period, DOC also increased significantly ($p \leq 0.03$) (Table C.3). POC was very highly positively correlated to PN ($p \leq 0.01$) while DOC was not correlated to any other variable on Lake Waterhen (Figure C.2). POC on Lake Manitoba followed a pattern similar to Lake Winnipegosis, in that it was very significantly highly positively correlated ($p \leq 0.001$) to particulate nutrients (nitrogen and phosphorus) and algal biomass. POC was also very significantly positively correlated to suspended solids (TSS, FSS, POM), as well as conductivity, pH and TN ($p \leq 0.001$) and significantly negatively correlated to VSS ($p \leq 0.05$). DOC was not correlated with any other variable (Figure C.3), however DOC did increase highly significantly ($p \leq 0.007$) between years (Table C.3).

5.2.4 Conductivity

Results from the wilcoxon rank sum test on Lake Winnipegosis indicated conductivity differed significantly between years ($p \leq 0.05$) (Table C.2), but there was no difference on Lake Manitoba (Table C.2). Conductivity differed between all basins on Lake Winnipegosis, ranging from highly significantly ($p \leq 0.001$) to significant ($p \leq 0.05$) (Table C.1). On Lake Manitoba, in 2016, conductivity differed highly significantly between both north basins and the south basin ($p \leq 0.01$) and differed significantly between the NE and south basin ($p \leq 0.05$). In 2017, only the NW and south basins differed significantly ($p \leq 0.05$) (Table C.1).

Mann-Kendall test results on Lake Winnipegosis also indicated conductivity was very significantly positively correlated with POC ($p \leq 0.01$) and positively significantly correlated with FSS ($p \leq 0.05$). Conductivity was very significantly negatively correlated with TDP, TP and DOC ($p \leq 0.01$) and negatively significantly correlated with VSS ($p \leq 0.05$) (Figure C.1). On Lake Waterhen conductivity did not correlate with any other variables (Figure C.2). Between variables, on Lake Manitoba, conductivity was very positively highly significantly correlated with particulate nutrients (PN, PP, POC), algal biomass, pH and TN ($p \leq 0.001$) and positively significantly correlated with TDN ($\mathrm{p} \leq 0.05$) (Figure C.3). Conductivity did not differ significantly between years on any of the lakes (Herbert, 2025b).

5.2.5 Chlorophyll

Neither measure of chlorophyll was significantly different between years on Lake Winnipegosis or Manitoba. TChl (TotC) and Chl a were significantly ($p \leq 0.05$) to highly significantly ($p \leq 0.001$) different between the north basins and both north basins and the south basin in both years (Tables C.1, C.2).

Mann-Kendall results on Lake Winnipegosis indicate both forms of chlorophyll were very highly positively significantly ($p \leq 0.001$) correlated with particulate nutrients (PN, PP,POC), with each other, and with TN and TP. Chlorophyll was also highly significantly positively correlated with TDP ($p \leq 0.01$) (Figure C.1). On Lake Waterhen, both measures of chlorophyll were positively significantly ($p \leq 0.05$) correlated to TN, while TChl was positively significantly correlated and Chl a was highly positively significantly correlated ($p \leq 0.01$) to pH (Figure C.2). Similar to Lake Winnipegosis, chlorophylls on Lake Manitoba were highly positively significantly correlated to particulate nutrients ($p \leq 0.001$), each other, and TSS, conductivity, pH, TN and TP. TChl was highly significantly correlated ($p \leq 0.01$) to POM while Chl a was highly significantly correlated to FSS. Chl a was also significantly correlated to TDP ($p \leq 0.05$) and DRP (Figure C.3). Between years the Mann-Kendall test did not detect any significant trend in chlorophyll between years on any of the lakes (Herbert, 2025b).

5.2.6 Transparency

Transparency in the uMBGL increases generally from north to south from Lake Winnipegosis to Manitoba, with the lowest transparency at the south end of Lake Manitoba. Lake Waterhen, in between these lakes, exhibited the highest water clarity of the three lakes, but since the lake was sampled at one location, spatial variability in transparency on the lake is not well represented. Transparency was higher in the spring, decreasing throughout summer into the fall (Figures 4.4, D.5). Secchi depth was only measured in 2017 so there is no between year comparison.

Secchi depth and turbidity are inversely related indicators of transparency, as the higher the turbidity values, the lower the transparency. In addition to Secchi depth, measures of transparency related to turbidity measured for this project were TSS, FSS and VSS. The gradient of TSS is the inverse of the Secchi gradient, with TSS values being higher at the south end of Lake Manitoba and lower at the north end of Lake Winnipegosis and on Lake Waterhen (Figure 4.5). VSS, the organic component of TSS, while generally following the gradient decreasing from north to south, is highest in both study years in the NW basin of Lake Manitoba. In 2016, the inorganic component of TSS, FSS, was highest at the Narrows, and comprised 70-80% of the total (Figure D.6).

Between study years, the wilcoxon rank sum test showed that both VSS and FSS values showed a significant difference ($p \leq 0.05$), with no significant difference between basins in either year (Table C.2, Figure 5.14), which may be due to the lack of data in 2016. There was a significant to very significant differences between years for VSS (LOI) ($p \leq 0.01$) on both Lakes Winnipegosis and Manitoba (Table C.2). Both lakes also had very significant ($p \leq 0.01$) to significant ($p \leq 0.05$) differences in TSS, VSS (LOI) and FSS (Tripton) between all basins (Table C.1).

The Mann-Kendall test on Lake Winnipegosis showed that TSS was very significantly positively correlated POC and pH ($p \leq 0.01$), and significantly positively correlated with PC and pH ($p \leq 0.05$) (Figure C.1). VSS was significantly positively correlated with TDP ($p \leq 0.05$), and very significantly positively correlated with pH and TP ($p \leq 0.01$) (Figure C.1). VSS was very significantly negatively correlated with FSS ($p \leq 0.01$), and significantly correlated with conductivity ($p \leq 0.05$) (Figure C.1). FSS was significantly positively correlated with PC and conductivity ($p \leq 0.05$), very highly positively significantly correlated with TSS ($p \leq 0.001$), very significantly negatively correlated with VSS ($p \leq 0.01$) and significantly correlated with TP ($p \leq 0.05$). On Lake Waterhen, the only significant correlation between variables was a negative correlation between FSS and VSS ($p \leq 0.05$) (Figure C.2). TSS and FSS exhibited a significant ($p \leq 0.01$ and $p \leq 0.02$) decreasing trend over the two years (Table C.3). Particulate nutrients and algal biomass were very highly positively correlated ($p \leq 0.001$) with TSS and FSS on Lake Manitoba. TSS and FSS were also very highly positively correlated with each other. FSS was very highly positively correlated with TN ($p \leq 0.001$). TSS was highly to very highly positively correlated with pH, TN and POM ($p \leq 0.001$) and very highly negatively correlated with VSS ($p \leq 0.01$) (Figure C.3). VSS was also very highly to significantly negatively correlated the particulate nutrients and TN ($p \leq 0.001$), and significantly positively correlated to SRP ($p \leq 0.05$) (Figure C.3).

On Lakes Waterhen and Manitoba, FSS in both years comprised a similar proportion to TSS (40-50%). In 2017 the highest TSS and FSS values occurred on the west side of Lake Manitoba at a station closest to shore and near the input of the Whitemud River (Figure D.6). The 2017 high values were collected in spring and summer, both during months (May, August) which recorded precipitation peaks at the McCreary weather station above the climate normal (Figure 5.1). In 2017, the ARD was also operating, recording the fourth highest flow (April) since it came into operation, and potentially bringing in higher turbidity inorganic waters.

5.3 Summary

Where possible, provincial sample stations near study locations and at major river inflows and outflows were used to determine potential effects from river flows. The Assiniboine River Diversion (ARD) operated for over two months (66 days) in 2017, with flows beginning on March 31st, and peak flow on April 14th (674 $m^3 s^{-1}$) (Figure 5.12). Since it began operation, there are only three other instances where flow has been higher in the ARD: in 1976 (736 $m^3 s^{-1}$), 2011 (983 $m^3 s^{-1}$) and 2014 (929 $m^3 s^{-1}$), making 2017 the fourth highest flow since the operation of the ARD began in 1970 (Province of Manitoba, n.d.). Water quality data from provincial station MB05MJS045 (Assiniboine River at reservoir of Portage La Prairie W.T.P.) was used to ascertain potential ARD inputs (Table E.1, Figure E.1).

5.3.1 Conductivity

Overall, the highest conductivity occurred either at the south end of Lake Manitoba or in the SW and SEE basins of Lake Winnipegosis. The source of the higher conductivity waters at the sample stations (higher salinity) is likely due to groundwater recharge from ancient basin brine (Pleistocene melt water which occurred during the formation of Lake Agassiz), mixed with modern water (rain and snow) that dissolved evaporite beds (Grasby & Betcher, 2002), whereas the freshwater east of the carbonate rock aquifer are related to modern precipitation (meteoric) recharge Grasby & Betcher (2002). In the south basin of Lake Manitoba, the high conductivity values may also be due to the high evaporative characteristics of the south basin (Page, 2011).

5.3.2 Nutrients

TN and TP followed the same patterns in both study years in the uMBGL, increasing from north to south, with Lake Manitoba's south basin consistently containing the highest concentrations.

The highest concentration of TP on Lake Winnipegosis occurred in summer 2017 at the south end (370 $\mu g L^{-1}$), decreasing by over half in the fall (158.5 $\mu g L^{-1}$) (Figure D.2). Lake Waterhen and Lake Manitoba's north basin followed a similar gradient, also containing the highest TP values of the two years in summer 2017. Patterns in 2016 may be skewed as data was collected on Lake Manitoba at one site, and on Lake Winnipegosis only in the SW and SEE basins (Herbert, 2025a).

On Lake Manitoba, in 2016, the highest TP and TN concentrations occurred in the narrows (Figures D.1, D.2), (Herbert, 2025a). The reason for this is unclear as values to the north and south sampled the day after are close to half the narrows values, and the provincial sample at the narrows from Oct.18 recorded a TP of 30 µg L^{-1}. Study TN and provincial station TN values recorded similar values of 1,843 and 1,140 µg L^{-1} respectively. One reason for the higher TP values may be that the narrows site is typically sampled from a bridge, near a high traffic area housing a waterfront restaurant, and therefore it may be more influenced by anthropogenic sources. The sampling location to the south of the narrows is close to where Ebb and Flow Lake may enter Lake Manitoba, which may also influence the TP concentrations, but TP has not been measured in that area. The northern station may be affected by water flowing from the NE basin water, which was lower in TP, particularly if the wind was blowing from the north (Figure 2.1), (Herbert, 2025a). Peak flows in the northern Lake Winnipegosis rivers (Overflowing, Red Deer and Mossy), corresponded with high precipitation events during the open water season. While there was no clear correspondence to higher nutrient levels, this may be reflective of sample timing and not potential nutrient influx into the system, particularly since nutrients did increase over the course of the season, indicating nutrient input and/or internal nutrient recycling. As all lakes were also phosphorus limited during the majority of both years, I hypothesized that the runoff associated with precipitation events were a potential source of nutrients into the lake, causing pulses of nutrients which were then uptaken by algae or macrophytes in the lakes. In addition to algal nutrient uptake, Lake Winnipegosis in particular, in the north basin contains macrophytes that both grow from the bottom to the surface, and occasionally occur as free-floating “islands” of cattails and rushes that appear to have broken off from wetlands bordering the lake. We observed these islands floating during the course of both sampling years.

In the south basin of Lake Manitoba in 2016, even with low flow and water levels (Figure 5.12), the Whitemud river appeared to influence TP concentrations more than Assiniboine river inflows. In October (18th and 27th) 2016, TP values measured by the province increased at stations along the Assiniboine river, ranging from 79 to 84 µg L^{-1} up to 90 µg L^{-1} at Lake Manitoba near Delta Beach (MB05LLS013), while study sites in the same week (Oct. 24th) recorded TP values ranging from 66 µg L^{-1} near the Whitemud river on the west side and 50 µg L^{-1} on the east side of the basin (Figure D.2), (Herbert, 2025a). The same pattern occurs in spring 2017 samples, with TP generally decreasing from west to east in the south basin. During the operation of the ARD, TP values decreased with declining flows, with individual values ranging from 515 – 1,810 µg L^{-1} at its peak (Figure E.1). In the south basin, however TP values remained at least twice as high as 2016, likely due to the initial influx of nutrients from the ARD. Unlike TP, TN did not differ significantly between years even when the ARD was operational, which may indicate that when the ARD is operational, phosphorus is a more significant nutrient input into the lake than nitrogen.

5.4 *Carlson’s trophic state index (TSI)*

The TSI describes the existing trophic condition of the waterbody using three separate indices as proxies for algal biomass (SD, TP, TChl). Each indice can be used to individually assess trophic state, however TChl is considered the primary indicator (Figure 5.15). The overall TSI is calculated by taking the mean of all three indices. The mean eutrophication potential (EP) of the water body was also calculated and ranked as a value between 0 and 100 (Equation 2.6) (SD, TP, TChl). The EP describes the potential of the system to continue to eutrophy based on current nutrient conditions. A TSI of < 40 indicates oligotrophic conditions, 40-50 = mesotrophy, 50-70 = eutrophy and above 70 indicates hypereutrophic conditions (Carlson, 1977; Carlson & Simpson, 1996). Results for this study were based on 2017 data as SD was not collected in 2016, but a similar result would be expected based on TP and TChl values for 2016. The TSI was calculated for each basin on Lakes Winnipegosis and Manitoba. Overall, every basin for Lakes Winnipegosis and Manitoba had a eutrophication potential of 60 or higher. The TSI for all basins on Lakes Winnipegosis and Manitoba had indices of 50 or above, indicating eutrophic conditions at the time of sampling. Lake Waterhen had a very low eutrophic potential (16%) but based on current conditions was classified as mesotrophic (Figure 5.16).

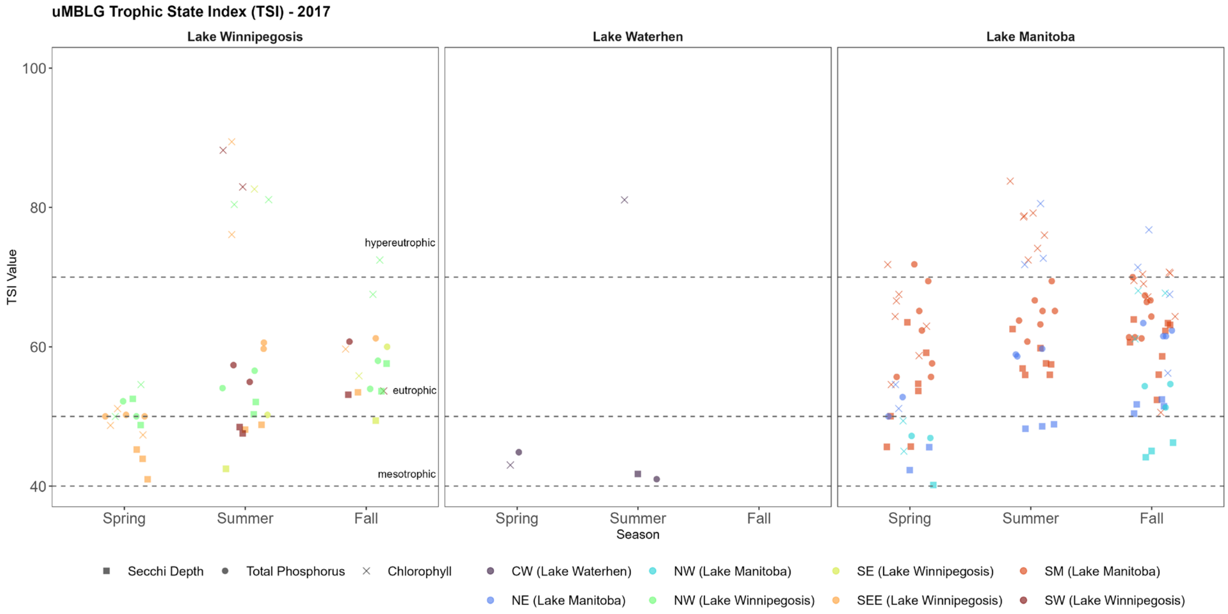

Figure 5.15. Carlson Trophic State Index 2017 showing individual sample points

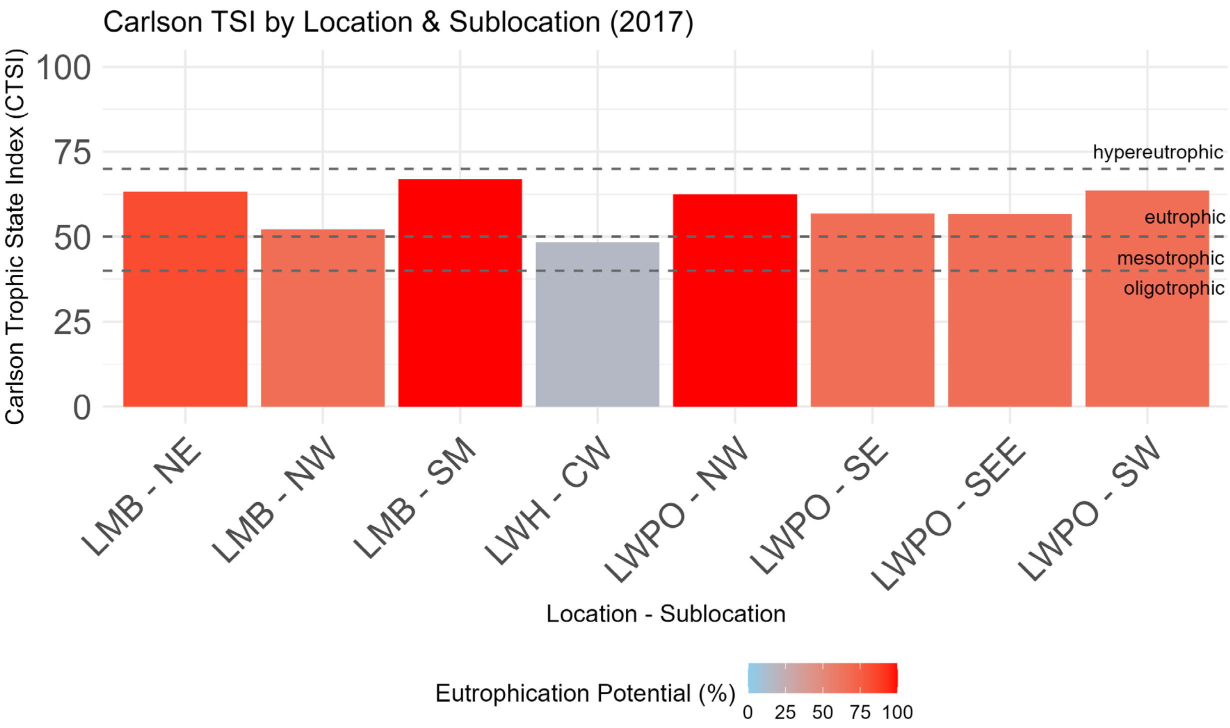

Figure 5.16. Carlson Trophic Status Index 2017. Bar colour shows eutrophication potential

6. Geostatistical analysis and distribution - Objectives 2 and 3 Discussion

6.1 Geostatistical Analysis between Lakes Winnipegosis, Waterhen and Manitoba - Objective 2

Analysis was conducted on minimally cleaned data, therefore variable names are abbreviated differently. Original names that are used in this results sections are followed in brackets by the current names used in the rest of this thesis: suspc (POC), doc (DOC), hplc (chl a), gf_chla (TChl), suspn (PN), total_n (TN), tdn (TDN), susp (PP), srp (DRP), tdp (TDP), total_p (TP), tripton (FSS), loi (VSS) and cond (COND), POM (POM).

6.1.1 Lake Winnipegosis

On Lake Winnipegosis for the combined 2016-2017 sample period, the first 4 PCA components explain 78.9% of the variability with components 1-4 accounting for 31.4%, 21.5%, 17,2% and 8.8% successively (Figure 6.1). 31.4% of the variation in the lake is influenced by particulate nutrients (PN, PP), and algal biomass (chlorophylls), followed by POC (Figure 6.2). The second strongest variances in the lake are due to phosphorus (dissolved and total), while water clarity indicators (TSS, FSS) are the third strongest influence. PCA supports the Mann-Kendall results of negative correlations between conductivity and TSS versus DOC, TP, DRP and TDP (Figure 6.3).

Using K-means clustering, 4 groups were identified. Group 1 locations were most influenced by particulate nutrients (PP, POC, PN), TN and algal biomass, representing areas of high productivity, higher nitrogen and high turbidity. Group 2 locations had the broadest distribution and were most influenced by phosphorus (dissolved and total), particulate organic matter and pH, representing phosphorus rich areas, potentially dominated by organic matter.

Group 3 was most influenced by total and inorganic solids (FSS) and conductivity, representing areas with more mineralized, turbid water. The last group only occurred in the NW basin so did not form a 95th percentile (Figure 6.3), indicating no clear dominant variable.

The results indicate that the lake basins are less diverse biogeochemically than Lake Manitoba, but still exhibit different dominant characteristics. The SEE basin represents a diverse, nutrient rich, productive area with high conductivity and lower water clarity. The SW basin is also very diverse, representing a slightly less turbid, but highly productive, nutrient rich area dominated by phosphorus. The SE basin also represents a more nutrient rich area than the north basin, influenced by phosphorus and higher conductivity. The NW basin is an area of lower conductivity, and is dominated by particulate nutrients (N,P) and algal biomass (Figure 6.4), representing a region driven where productivity may be limited by nutrient availability and light.

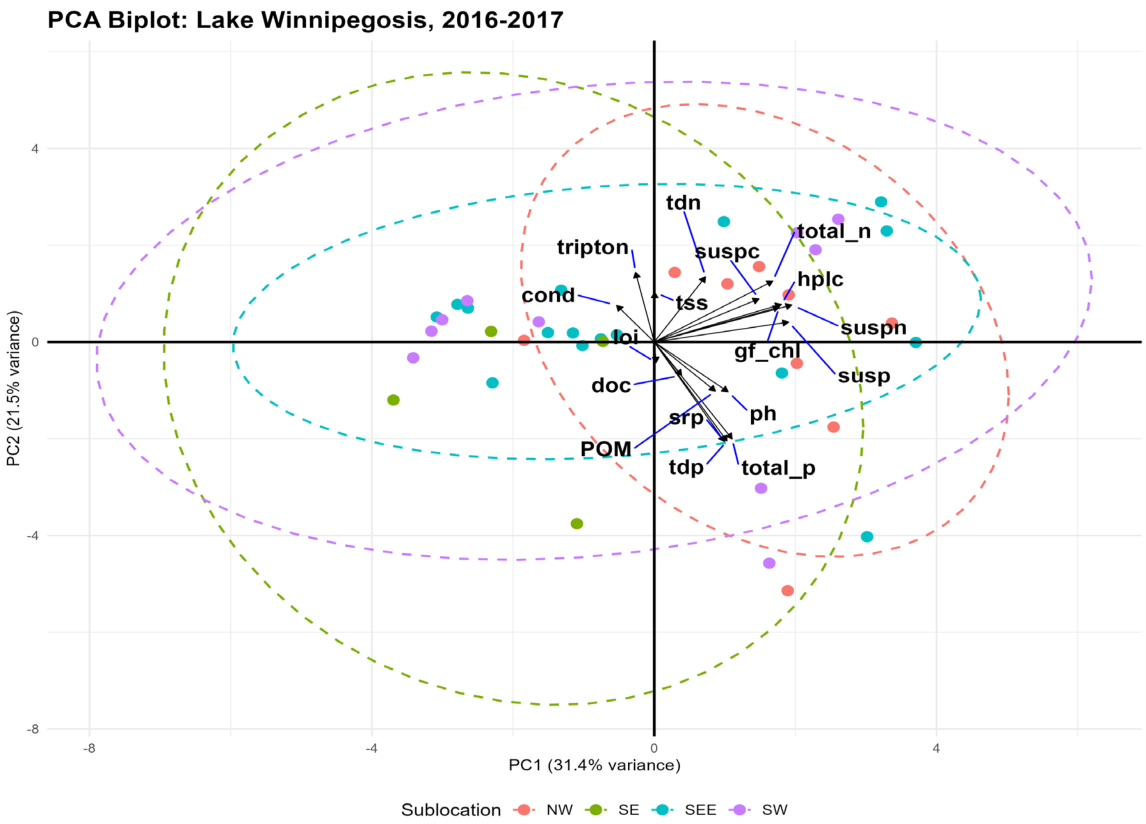

Figure 6.1. Lake Winnipegosis PCA showing basins, 2016-2017

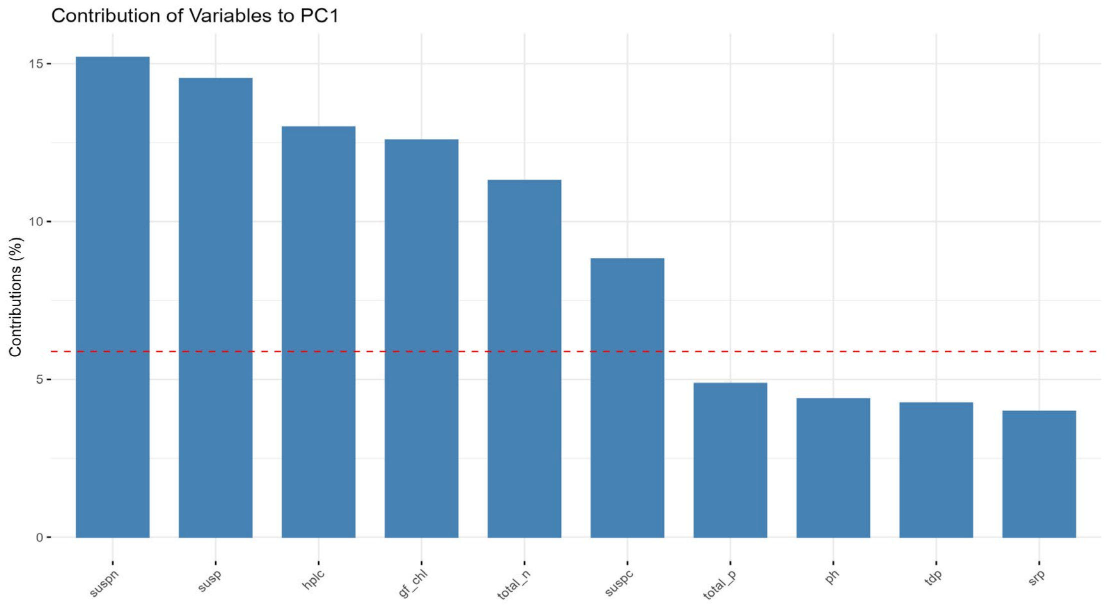

Figure 6.2. Lake Winnipegosis Principal Component Analysis Scree Plot showing axis 1 major contributors.

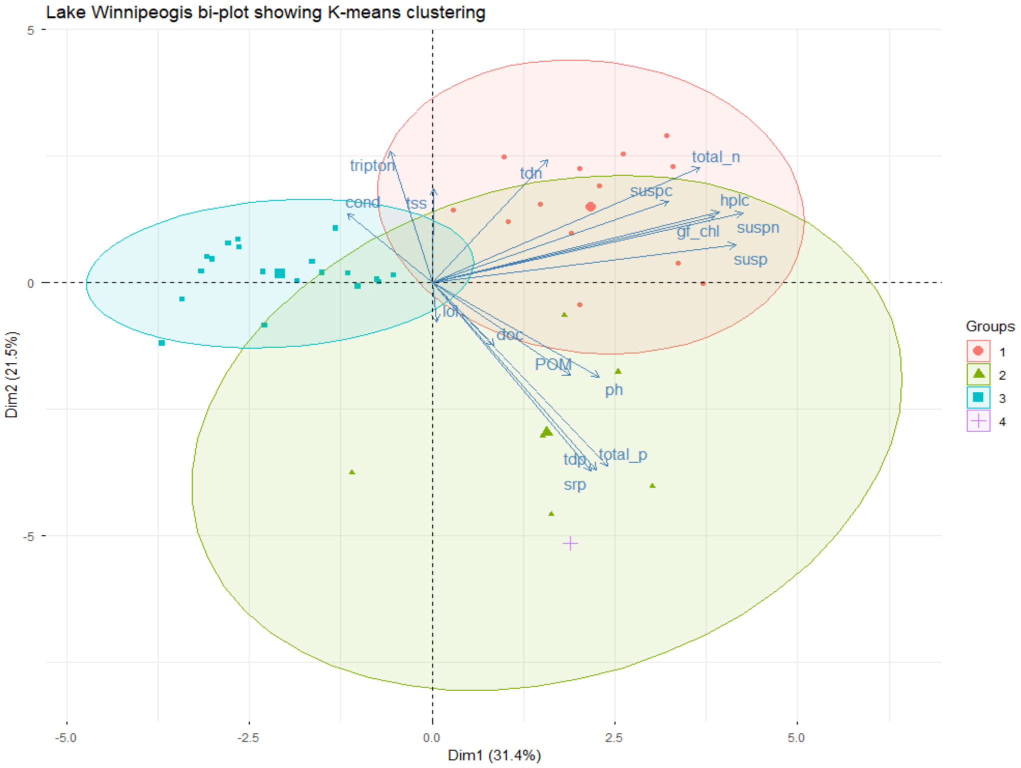

Figure 6.3. Lake Winnipegosis biplot showing variable loadings and K-means clusters

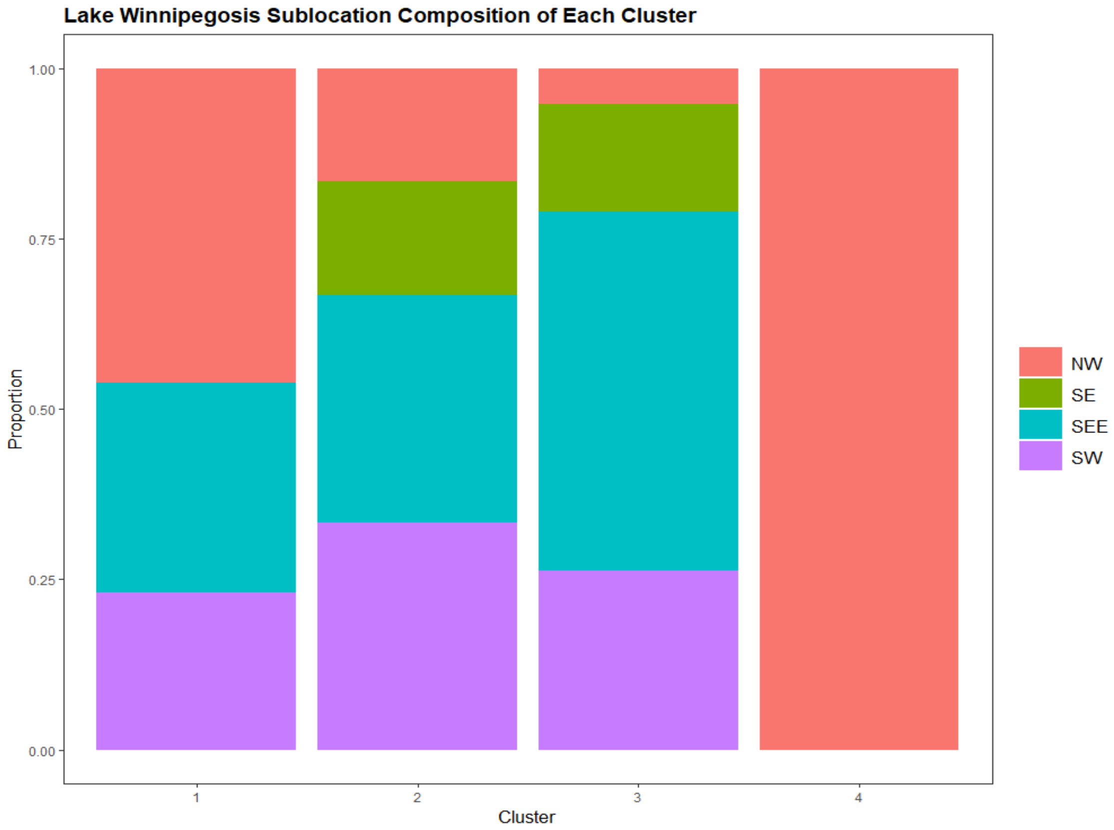

Figure 6.4. Lake Winnipegosis K-means clusters related to sublocations

6.1.2 Lake Waterhen

On Lake Waterhen K-means clustering was not applied as the lake is treated as one basin. PCA analysis showed that 51.8% of the variability in the data can be explained by the first two PCA Axis (Figure 6.5). Particulates (POC,PN,PP) contributed to the highest positive variance in the data (Figure 6.6), while DOC is negatively correlated. On PC2, conductivity. FSS (tripton) and pH are strongly negatively correlated.

The distribution of the data (Figure 6.7) indicates that particulate nutrients (POC, PN and PP) are highly correlated with each other, as are inorganic particulates (TSS and VSS). Phosphorus variables are highly correlated with algal biomass, suggesting algal productivity is tied more closely to phosphorus than nitrogen, and somewhat correlated with pH. DOC is strongly negatively correlated with TN.

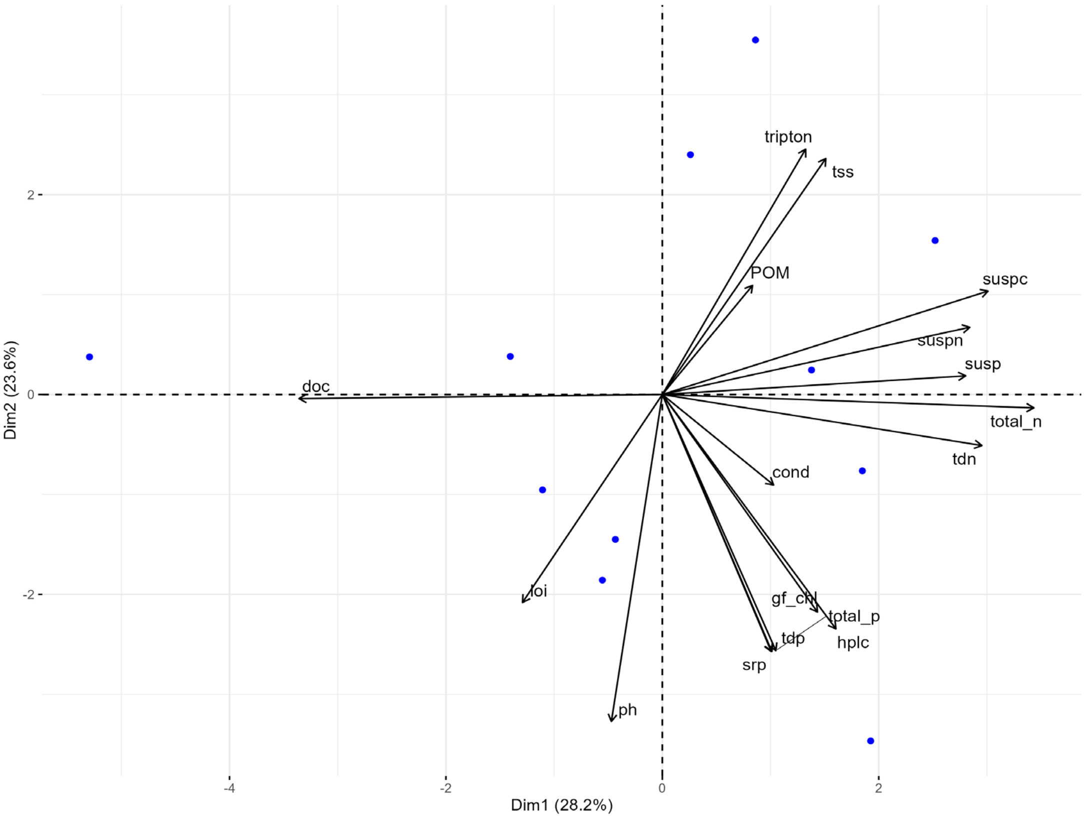

Figure 6.5. Lake Waterhen PCA showing loadings and sample data, 2016-2017

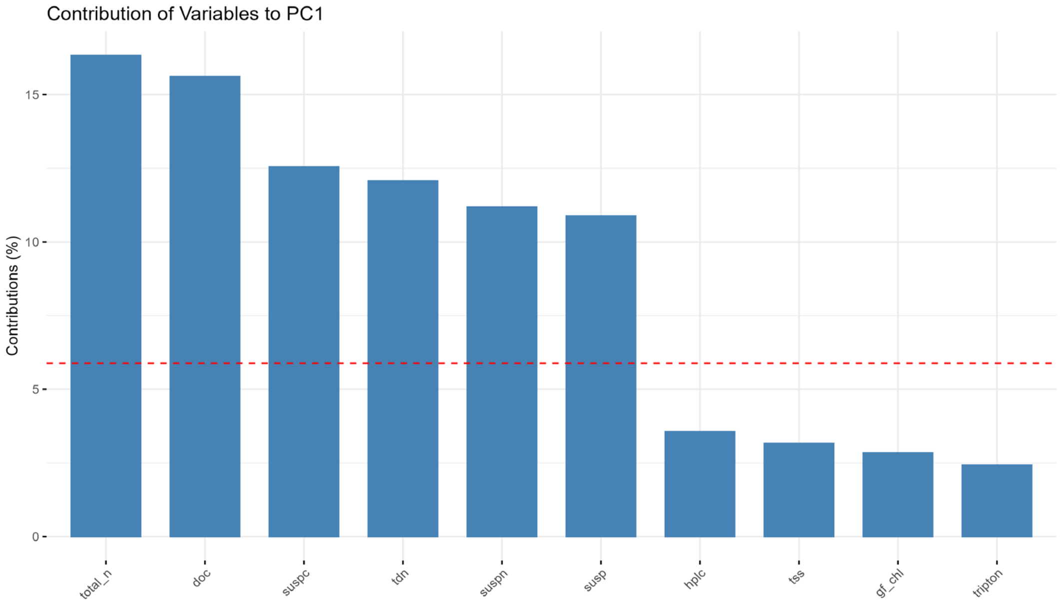

Figure 6.6. Lake Waterhen Principal Component Analysis Scree Plot showing axis 1 major contributors

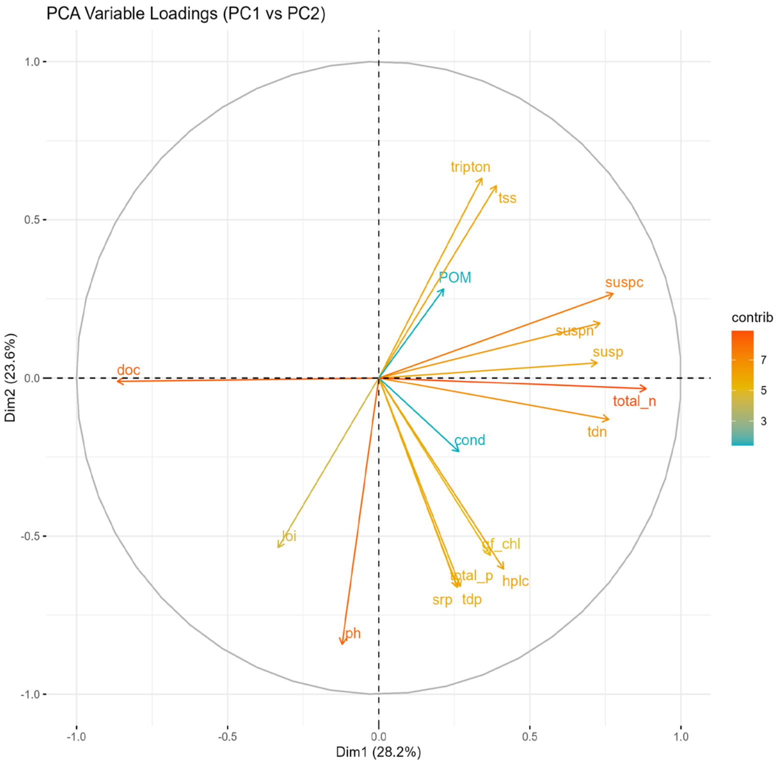

Figure 6.7. Lake Waterhen PCA showing loading contribution, 2016-2017

6.1.3 Lake Manitoba

On Lake Manitoba (Figure 6.8), the first two axis explain 51.8% of the variability in the data. Particulate nutrients (POC, PP, PN), TN and TChl contributed the highest variance to Axis 1, explaining 33.2% of the variability in the data. DRP, TDP and TP were the highest contributors to axis 2, explaining 18.6% of the variability (Figure 6.9). The south and NE basins appear to have similar biogeochemical parameters relative to the NW basin.

The K-means for the data was plotted using 5 clusters. Variables in Group 1 exhibited similar characteristics to Lake Winnipegosis, being most heavily influenced by high particulate nutrients, TN (POC, PN, PP), inorganic particulates (FSS (tripton) and TSS), and algal biomass (chlorophylls), representing areas of high productivity and high turbidity. The Group 2 cluster was most heavily influenced by organic particulates (VSS (loi)), representing an area characterized by organics. Group 3 cluster overlapped with groups one, two and four, and consisted of areas most influenced by dissolved organic carbon, conductivity and total dissolved nitrogen, representing an area with high conductivity and low nutrients. Group 4 represented low nutrient, low pH influenced areas, representing low productivity areas. Group 5 cluster was most diverse, and strongly influenced by phosphorus (dissolved and total) and pH (Figure 6.10).

Bi-plot analysis indicates that in Lake Manitoba, the south basin is highly turbid, nutrient rich and productive, dominated by phosphorus and has the highest conductivity. The NE basin has a distinct signature and appears to be driven by organic particulates. The NW basin represents the area with the lowest concentration of nutrients (Figure 6.11).

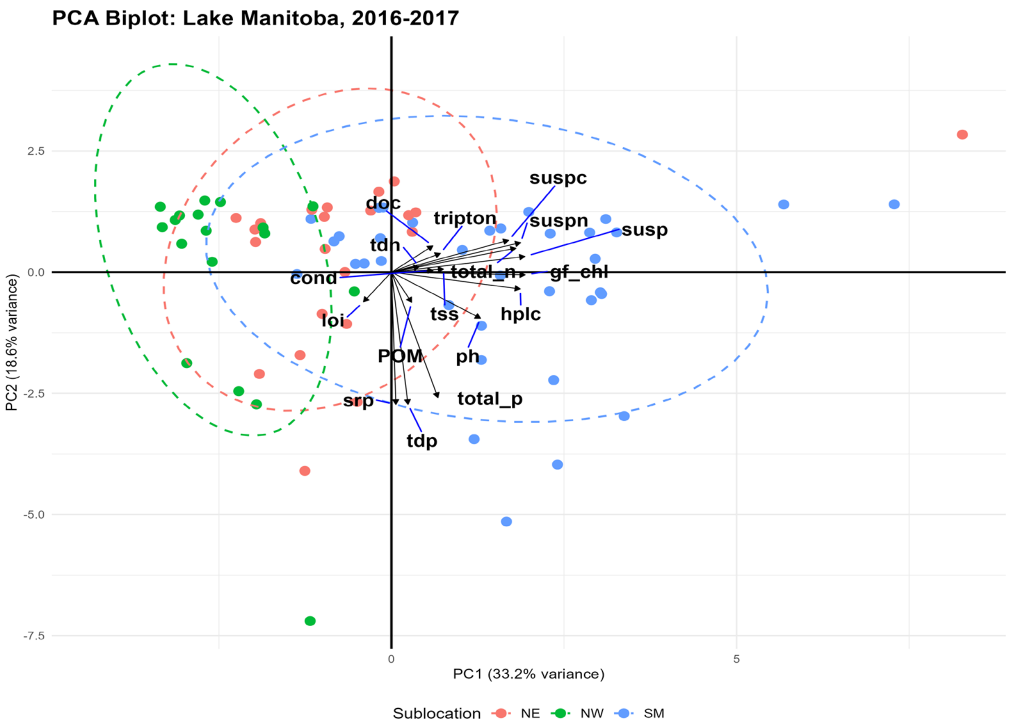

Figure 6.8. Lake Manitoba PCA showing basins, 2016-2017

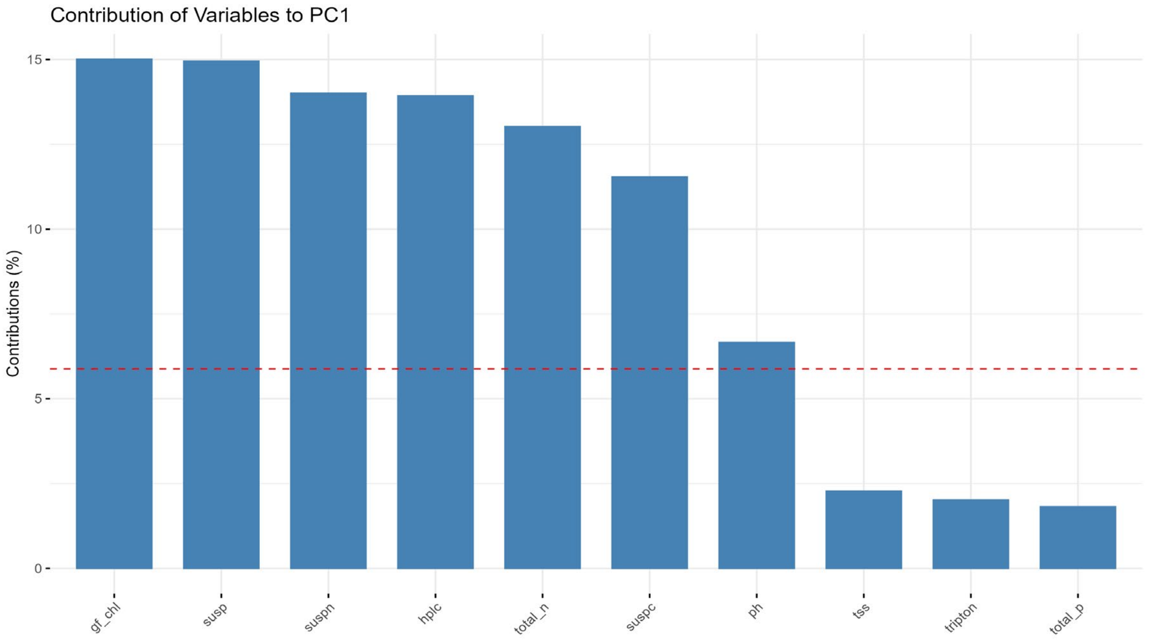

Figure 6.9. Lake Manitoba Principal Component Analysis Scree Plot showing axis 1 major contributors

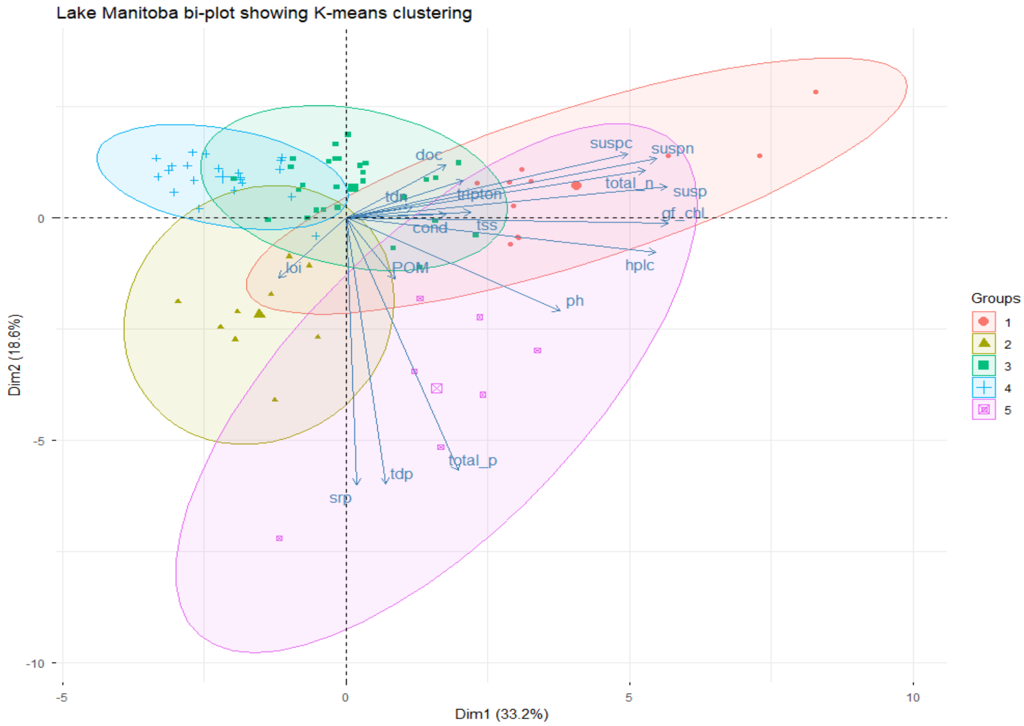

Figure 6.10. Lake Manitoba biplot showing variable loadings and K-means clusters

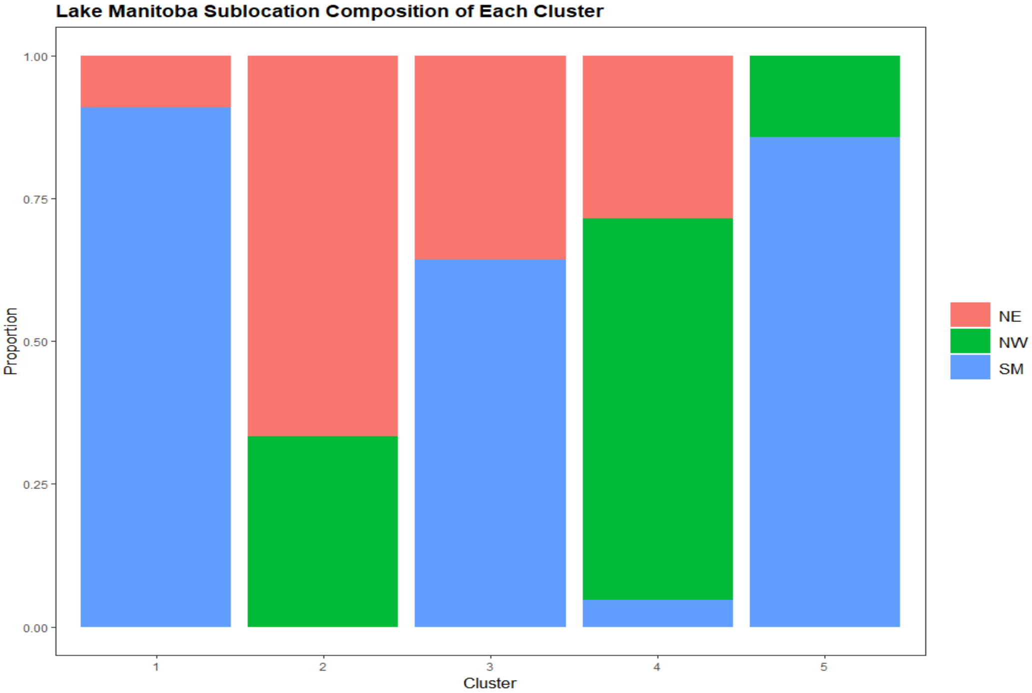

Figure 6.11. Lake Manitoba K-means clusters related to sublocations

6.2 Discussion

PCA with K-means clustering supports my division of the lakes into multiple basins, with overlapping characteristics between adjacent basins. On Lake Winnipegosis, the NW basin characterizes the least productive basin. Moving south, basins become more productive and turbid. The SEE basin on Lake Winnipegosis is also distinct due to its high conductivity and more mineralized water.

Lake Waterhen was treated as one basin and exhibits the same strong correlation between algal biomass and phosphorus. It is also unique in the strong inverse correlation between DOC and TN, with both parameters influencing variability on the lake.

Lake Manitoba basins were the most distinct, with the NW basin being the least productive, and the south basin being the most productive. The NE basin had a distinct organic particulate signal, and the SB was distinguished also by its high conductivity. The overlap between basins as shown on the PCA biplot (Figure 6.1) and scores plots (Herbert, 2025b) indicate that the biogeochemistry of the NE basin of Lake Manitoba is influenced by mixing of water from the NW and south basins, which both ultimately flow out of the lake through the Fairford River.

6.3 Phytoplankton diversity - Objective 3 Discussion

Total chlorophyll (TChl) is commonly used as a proxy for algal biomass (Axler & Owen, 1994) and was also compared to algal biomass data to see if a similar pattern was observed across Lakes Winnipegosis and Manitoba. TChl increased from north to south in the lakes, with the highest concentrations occurring at the south end of Lake Manitoba (Figures 6.12, D.10). The same overall pattern can be observed in the algal biomass analysis, with phytoplankton biomass two to three times higher in the south basin of Lake Manitoba (Figure 6.12), with both lakes dominated by cyanobacteria.

To compare different lakes to each other, biomass which is a standardized value is always used, however it is worth noting the difference in dominant algae by biomass versus abundance (Appendix F). In examining phytoplankton distribution and composition for Lakes Winnipegosis and Manitoba, I found the cyanobacterial community to be highly diverse, with filamentous cyanobacteria both fixing and non-fixing to be dominant by biomass, but cyanobacterial picoplankton (<2.0 µm) to mainly dominate by abundance. The abundance of picoplankton may explain the lack of surface algal blooms in the open water areas of the lakes, as they have been shown to have a competitive advantage in the water column due to their small cell size (slower sedimentation rate and higher ratio of cell biomass to volume), increased ability over other species to absorb and process a wider spectrum of light, and ability to use phycocyanin to supplement nitrogen deficiency (Jakubowska & Szeląg-Wasielewska, 2015). Picocyanobacteria are able to survive and regrow after periods of total darkness and can grow in light ranging from 45 to 2000 µmol/photon m^2 s^{-1} (Jakubowska & Szeląg-Wasielewska, 2015). Li et.al. (2019) showed that picocyanobacteria are also likely major contributors to primary production due to their ability to preferentially accumulate, store and metabolize polyphosphate (polyP), a polymer which can contain hundreds of orthophosphate units, the most biologically available form of phosphorus for algae to use.

Even though they do contain many nitrogen-fixing filamentous algae, the taxa commonly associated with surface algal blooms, unlike Lake Winnipeg, the uMBGL did not exhibit evidence of surface water blooms during either sampling season. Although the lakes are shallow, the north basin of Lake Winnipegosis does reach depths similar to Lake Winnipeg (6 - 9 m), and all lakes were often calm on the days we sampled. Therefore, the theory that the lakes are always well mixed and therefore cannot form surface algal blooms even for short periods of time is not entirely supported. However, the abundance of picoplanktic algae may also explain part of the difference between these lakes and Lake Winnipeg in terms of bloom formation. In addition, communities around the lakes have observed blooms in protected bays, common in areas where people recreate.

On both lakes, the dominance of cyanobacteria (filamentous, microbluegreen and coccoid), support current literature and the results in this study (trophic status, nutrient ratios), that phosphorus limitation influences phytoplankton composition (Schindler, 1974). The presence of *Dolichospermum* (*Anabaena*) and *Aphanizomenon* heterocysts (Tables F2-F6, F8-F9), also support study findings of nitrogen limitation during both summers (Appendix G).

The dominance by biomass of *Microcystis* sp. in 2016 in the north basin of Lake Winnipegosis supports literature (Chang et al., 2020) and this study's results that in regions where light may be limited and/or nitrogen is not limited, bluegreen algal taxa that can control their buoyancy but do not fix nitrogen (e.g. *Microcystis*) may have a competitive advantage over those that control their buoyancy but also can fix nitrogen (*Dolichospermum, Aphanizomenon*), as the former use nitrogen more efficiently and grow better under lower light conditions (Chang et al., 2020).

Of the cyanobacterial taxa detected, 14 are potential toxin producers of one or more of seven varieties of toxins (microcystin, lipopolisacharid, cylindrospermopsin, BMAA, anatoxin-a, saxitoxin, paralytic shellfish poison or nodularin and further analysis should be conducted to understand the presence and distribution of these toxins in the lakes (Jakubowska & Szeląg-Wasielewska, 2015; Li et al., 2019).

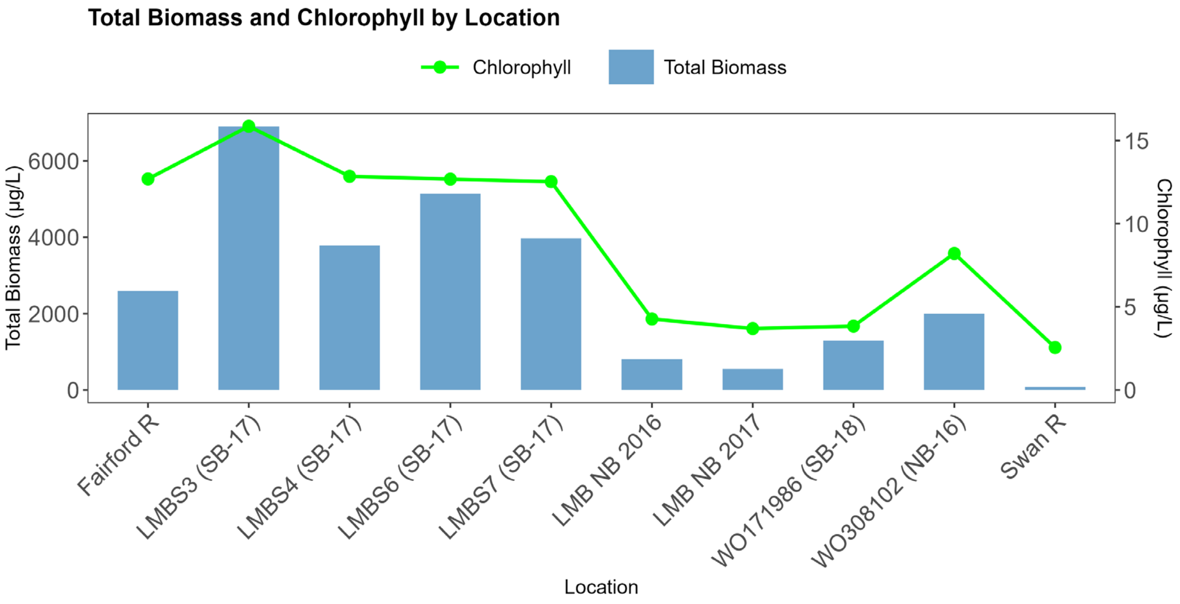

Figure 6.12. Algal biomass (mg m-3) and Chlorophyll (µg/L). Chlorophyll values for the combine NB stations are median values from all stations. Except for the Fairford River (upperMBGL outflow) station, stations are sorted from left-right and south-to-north

6.3.1 Stoichiometry

Algae respond quickly to changes in their environment and are often used as indicators of water quality. It is important to understand the factors driving algal community composition and dynamics, particularly in our rapidly changing climate (Ayub et al., 2024; Litchman & Klausmeier, 2008). Water column nutrient concentrations are not related directly to nutrient bioavailability for phytoplankton, therefore a preferred method to detect in situ nutrient deficiency is by using a variety of nutrient status indicators (Dubourg et al., 2015; Hecky & Kilham, 1988). Except for total chlorophyll, all measurements use the particulate form of the variable, are calculated on an atom to atom ratio, and indicate a specific type of limitation, OC:P (P limitation), N:P (P limitation), OC:N (N limitation) and OC:Chl (N and P limitation). Thresholds for nutrient limitation are based on values established in previous literature

(Guildford & Hecky, 2000; Healey, F. P., 1975; Healey & Hendzel, 1980) and are used to indicate levels of nutrient deficiency ranging from none to moderate to extreme (Table 6.1).

Overall, all measures indicated that phosphorus and nitrogen were limiting in all lakes, ranging from moderate to severe based on season and location. In 2016 the P status indicators (OC:P, N:P, OC:Chl) (Figures G.2 - G.4) were in agreement for all lakes, and indicated moderate to severe P limitation throughout the year. On Lake Manitoba in both years, P limitation was highest in the NE and most northern southern basin and lowest in the southern basin. On Lake Winnipegosis, in both years, P limitation was severe in the SEE and SW basins, and moderate in the NW basin.

In 2016 and 2017, N status indicators (OC:N, OC:Chl), agreed that on Lake Winnipegosis, the NW basin was the only basin that exhibited no or borderline nitrogen limitation during all seasons. Throughout both years and all seasons, the rest of the lake was moderately nitrogen limited (Figures G.1, G3). Lake Waterhen was moderately nitrogen limited during both years also. Lake Manitoba was moderately nitrogen limited over both years and all seasons, with the exception of station MB145886 in fall 2017 which was extremely N limited (15.73 µmol) (Herbert, 2025a).

Table 6.1. Nutrient status indicator thresholds applied to phytoplankton communities. Values are from Dubourg et al. (2015).

Indicator	Nutrient	No deficiency	Moderate	Extreme deficiency	Deficient
OC:N	N	< 8.3	8.3-14.6	>14.6	
OC:P	P	< 129	129-258	> 258	
N:P	P	< 22			> 22
OC:Chl	N/P	< 4.2	> 4.2	> 8.3	

7. Conclusion

7.1 Summary

The upper Manitoba Great Lakes are part of a diverse watershed that provides significant economic, recreational and societal benefits to communities, both that live around its shores, and that travel to utilize its waters. The region contains two of the largest lakes in the Prairies, spans two ecozones and three ecoregions. The north (Mid-Boreal Lowlands) and central (Interlake Plain) basins of Lake Winnipegosis are characterized by lower population density and contain a higher proportion of organic soils in the form of flat bogs, horizontal fens and marshy areas, mixed with areas of richer soils that support cattle and bison ranching and hay production. The south end of Lake Manitoba (Lake Manitoba ecoregion) lies in the most productive region in the Canadian prairies, as well as being the most highly populated region of the three uMBGL. This region contains one of the only two eco-certified fisheries in North America for northern pike and lake walleye yet is one of the most understudied in Canada. Personal conversations with communities, both Indigenous and settler, along both sides of these lakes have revealed a deep concern for the health of the lake, and the effects of the changing water quality on both the aquatic biota and human health.

These lakes act as key buffers to nutrient inputs into Lake Winnipeg; therefore it is vital to gain an understanding of how climate change and land use management and practices may affect their ability to process nutrient inputs. This study was undertaken with three objectives - to (1) provide the first ever spatially comprehensive three-season multi-year report on offshore nutrient and biogeochemical data, (2) provide a western science baseline for comparing and understanding of the differences in water quality variables between the lakes, and (3) examine the spatial and temporal distribution of phytoplankton in the basin.

My research concluded that for objective 1, all three lakes can be classified as mesotrophic to eutrophic in all seasons. Paleolimnological evidence suggests that Lake Manitoba has been slowly eutrophying over the last few decades (Gushulak et al., 2024) in the south basin. My research supports the evidence that Lake Manitoba is eutrophic not only in the south, but also in the north basin, and that the dominant algal taxa are a mix of fixing and non-fixing filamentous cyanobacteria during the summer. The same general gradient of increasing nutrient concentrations moving from north to south during the open water season was observed on both Lake Winnipegosis and Manitoba but could not be shown on Lake Waterhen due to lack of sample locations. Lake Winnipegosis and Lake Waterhen were also dominated by a mix of fixing and non-fixing filamentous cyanobacteria during the summer and had a higher diversity and different major taxa than Lake Winnipeg (Kling, 1998; Kling et al., 2011; McCullough et al., 2012). The Fairford River, the outlet to Lake Winnipeg, presents a nutrient and algal composition more similar to the lakes than other rivers (e.g. Swan River), indicting that there is high potential for transfer of excess nutrients and algal taxa into Lake St. Martin and Lake Winnipeg.

High water flowing into Lake Winnipegosis did correspond with high precipitation events over the two years, but this pattern was not as clear on Lake Manitoba except when the ARD was operational in 2017, which may indicate that Lake Winnipegosis water quality may be more impacted by nutrient pulses caused by precipitation events versus runoff from land.

For objective 2, I compared multiple basins within each lake, and did find significant differences between years and basins, indicating that mixing between basins may be limited, and basins may be impacted differently by local land use. Basins that receive water from larger river systems, such as the Overflowing or Red Deer Rivers, may be impacted more in high or low water years than a more central basin that is further away from river influences. Both 2016 and 2017 were high water years, with 2017 also having the ARD in operation. In particular, the ARD has been shown to significantly affect water quality in the south basin of Lake Manitoba but may be mitigated by inflows from the Whitemud River. It is therefore important when trying to understand water quality in these systems, to consider both at what scale the question is being asked (local, regional) and what stake or rights holder is asking the question. This was the first comprehensive water quality report for Lake Waterhen, which indicated that it is a mesotrophic lake where particulate nutrients and algal biomass explain just over 50% of the variability in the data.

In examining phytoplankton distribution and composition (Objective 3), I found the cyanobacterial communities on Lakes Winnipegosis and Manitoba were more diverse than Lake Winnipeg and dominated by filamentous cyanobacteria both fixing and non-fixing. I found filamentous cyanobacteria to be dominant by biomass, but cyanobacterial picoplankton to mainly dominate by abundance. The abundance of picoplankton may explain the lack of surface algal blooms in the open water areas of the lakes, as they may have a competitive advantage in the water column due to their small cell size and increased ability over other larger species to absorb and process a wider spectrum of light (Jakubowska and Szeląg-Wasielewska 2015). Phytoplankton biomass and chlorophyll concentrations followed the same pattern, and similar cyanobacterial species are known to be dominant in Lake Winnipeg (Kling et al. 2011).

There was a general agreement between total chlorophyll and algal biomass, with lower chlorophyll occurring at lower biomass locations (Figure 6.12). There are also many potential toxin producing algae which occur consistently in the uMBGL lakes, meriting a much more thorough investigation into the potential distribution and impacts for this region.

Although Lake Winnipeg does contain some similar species to the uMBGL, the biomass and general species composition is very different; for example, there is virtually no *Aphanizomenon flos-aquae* in the uMBGL, while it dominates in Lake Winnipeg, and the smaller coccoid species which dominate in the uMBGL lakes never reach similar biomass levels in Lake Winnipeg (Kling, 1998; Kling et al., 2011). The dominant taxa in the uMBGL, when present in Lake Winnipeg, enter via the Dauphin River (Figure 2.1) but do not survive into the center of the basin or become a major contributor to the algal community on Lake Winnipeg (H.Kling, personal communication, September 20, 2025).

7.2 Recommendations and Future Research

Spatial variability analysis for Lake Waterhen was not possible as only 1 site was continuously monitored. In spring 2016, the north end of Lake Winnipegosis and the south end of Lake Manitoba did not have good spatial coverage either. Travelling and sampling these lakes required time and money, due to the lack of access to launch points for boats, the limitations of boat size (many launch areas have < 1.0 m of water to launch from) and the lack of reliable weather data from stations close to the lakes. This makes sampling unpredictable and can cause days of delay if the weather station data is inaccurate. Other water quality parameters such as silica, nitrate and nitrite and ions such as sodium and calcium were not measured but would be useful in identifying sources of conductivity/salinity to relate to algal productivity. This thesis reports data from a western science perspective and ignores the valuable information traditional knowledge would provide to the history of lake water quality. We did make an effort to connect with every community around the lake, inform them of our research and listen to community specific concerns, but community concerns could not be addressed specifically within the scope of this work.

Additional data from methods such as lake core analysis for algal species and nutrient concentrations should continue to be collected from Lake Manitoba's north basin, Ebb and Flow Lake and Lake Winnipegosis, to provide a longer-term record for historical comparison. Other historical and current datasets (e.g. nutrient, contaminants), related to these lakes were also not openly accessible, and when accessed were not standardized to any national or international vocabulary, and metadata surrounding the data was either unclear, spread out over multiple documents, or both. Analytical methods from private labs are not transparent in changes from reference EPA or other common methods that were used, making it very hard to compare the same variable across different agencies. Ebb and Flow Lake, which is connected to Lake Manitoba, has been ignored in the literature, so there is no data on how inflows from the lake may be affecting Lake Manitoba, and vice-versa. Ebb and Flow Lake is a lake of great cultural significant to Ebb and Flow community (Ebb and Flow community members, personal communication, June, 2025) and an understanding of its impacts on Lake Manitoba and from Lake Manitoba to Ebb and Flow is of particularly important to that community, and so should be a priority. Ebb and Flow also has a large zebra mussel infestation, which makes it a good comparison lake. The impact of Lake Manitoba outflow on water levels and water quality on Lake St. Martin is also not well studied in western science and should also be considered in any monitoring program. This study also supports previous published results that show an increase in nutrient and sediment (Page, 2011) delivery occurs due to the ARD, but the impact of the Assiniboine River water through Lake Manitoba and into Lake Winnipeg has not been well studied either. To summarize key future actions that could be taken:

1. Curation of key provincially and federally funded datasets to international standardized vocabularies. Data should be made open by default with appropriate metadata, including methods and relationships between common variables mapped.
2. Curation of other datasets that have been collected (e.g. Mooring data from all three lakes, profile data for conductivity, temperature and turbidity, light profiles) should be undertaken and also made open and accessible.
3. Working with First Nation and other Indigenous communities around the lakes to identify key issues relevant to their communities. Research should be undertaken with these themes in mind, ideally with a community led component to the work.
4. Detailed counts for current sample locations for algal composition should be undertaken to gain a more thorough understanding of spatial and temporal changes, particularly as algae act as key indicators of nutrient issues.
5. Analysis for a variety of toxins should be conducted at sample sites and in collaboration with communities around the lakes to assess presence.
6. A comparison of algal biomass, in-situ chlorophyll data and satellite data should be conducted to assess the feasibility of using remote sensing to monitor these lakes, particularly in light of the different algal composition as compared to Lake Winnipeg.
7. Sampling on Ebb and Flow Lake and Lake St. Martin should be conducted in order to assess the effects of inflows and outflows into/from Lake Manitoba and the potential for zebra mussel incursion. Sampling should include zebra mussel veligers and could include eDNA.
8. When possible, sampling regimes should include sampling during high precipitation events to better understand correlations between flow, lake levels and events. Engaging

community-based monitors in this type of work is critical to the long term success of freshwater research in Manitoba.

In conclusion, I have identified over a short time span key biogeochemical drivers of water quality on three upper Manitoba Great Lakes, but there is much more work that should be done to understand and mitigate the impacts of land use, and climate change on these key buffer systems into Lake Winnipeg.

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APPENDICES

Appendix A — Upper MBGL descriptive statistics

Table A.1. Descriptive statistics for Lake Winnipegosis, 2016 water quality variables

Variable	Count	Mean	SD	MAD	Min	Median	Max
Secchi Depth (m)							
Total N (µg/L)	17	736.88	75.14	50.00	648.00	701.00	877.00
PN (µg/L)	17	181.24	51.01	34.00	91.00	175.00	263.00
TDN (µg/L)	17	555.65	44.54	34.00	462.00	553.00	623.00
TP (µg/L)	17	23.71	8.84	2.00	14.00	20.00	44.00
PP (µg/L)	17	15.94	6.16	3.00	10.00	14.00	34.00
DRP (µg/L)	13	1.92	1.89	0.00	1.00	1.00	8.00
TDP (µg/L)	17	6.53	2.94	1.00	4.00	5.00	16.00
POC (µg/L)	17	1,780.00	523.61	270.00	1,050.00	1,610.00	2,910.00
DOC (µmol/L)	14	1,097.86	476.44	42.00	851.00	924.50	2,692.00
TChl (µg/L)	17	5.84	3.66	1.97	1.71	4.86	14.80
Chl a (µg/L)	17	3.65	2.44	1.67	0.48	3.22	8.51
TSS (mg/L)	9	5.82	3.97	1.20	2.00	3.20	13.00
VSS (mg/L)	9	0.36	0.23	0.10	0.13	0.25	0.71
FSS (mg/L)	9	3.16	2.11	0.50	1.60	2.50	8.50
POM (%)	9	2.66	2.61	0.27	0.40	0.67	6.00
COND (µS/cm)	17	965.94	86.68	34.00	731.00	999.00	1,044.00
pH	17	8.59	0.02	0.01	8.56	8.58	8.63
POC (µmol/L)	17	148.32	43.63	22.50	87.49	134.16	242.48
PN (µmol/L)	17	12.94	3.64	2.42	6.50	12.49	18.78
PP (µmol/L)	17	0.51	0.20	0.10	0.32	0.45	1.10

Table A.2. Descriptive statistics for Lake Winnipegosis, 2017 water quality variables

Variable	Count	Mean	SD	MAD	Min	Median	Max
Secchi Depth (m)	20	1.68	0.69	0.48	0.92	1.52	3.40
TN (µg/L)	22	732.45	91.39	63.00	596.00	743.50	893.00
PN (µg/L)	22	189.36	57.99	50.50	91.00	184.50	288.00
TDN (µg/L)	22	543.09	49.60	24.00	471.00	524.00	632.00
TP (µg/L)	19	130.58	115.41	59.00	20.00	81.00	370.00
PP (µg/L)	22	18.50	6.97	5.00	7.00	18.00	32.00
DRP (µg/L)	19	52.11	57.95	21.00	1.00	22.00	173.00
TDP (µg/L)	22	51.41	56.00	9.50	4.00	14.50	175.00
POC (µg/L)	22	1,765.00	723.08	325.00	740.00	1,470.00	3,490.00
DOC (µmol/L)	13	1,252.62	361.52	55.00	965.00	1,185.00	2,413.00
TChl (µg/L)	22	6.98	3.23	2.59	2.73	6.38	15.67
Chl a (µg/L)	22	4.59	2.50	2.11	1.50	4.38	11.04
TSS (mg/L)	20	4.50	4.03	4.00	0.00	4.25	11.00
VSS (mg/L)	16	1.03	0.84	0.21	0.50	0.87	4.00
FSS (mg/L)	15	1.62	1.97	0.67	0.00	0.67	5.00
POM (%)	22	3.66	2.61	1.75	0.00	4.25	7.00
COND (µS/cm)	22	870.59	146.63	49.00	559.00	931.50	1,052.00
pH	22	8.60	0.03	0.02	8.53	8.60	8.64
POC (µmol/L)	22	147.07	60.25	27.08	61.66	122.49	290.81
PN (µmol/L)	22	13.52	4.14	3.61	6.50	13.17	20.56
PP (µmol/L)	22	0.60	0.22	0.16	0.23	0.58	1.03

Table A.3. Descriptive statistics for Lake Waterhen, 2016 water quality variables

Variable	Count	Mean	SD	MAD	Min	Median	Max
Secchi Depth (m)							
TN (µg/L)	5	811.00	59.73	60.00	749.00	825.00	885.00
PN (µg/L)	5	162.60	41.57	22.00	96.00	166.00	205.00
TDN (µg/L)	5	648.40	38.35	15.00	591.00	655.00	697.00
TP (µg/L)	5	22.40	5.27	2.00	17.00	21.00	31.00
PP (µg/L)	5	12.80	1.30	1.00	11.00	13.00	14.00
DRP (µg/L)	4	2.25	2.50	0.00	1.00	1.00	6.00
TDP (µg/L)	5	8.00	3.67	1.00	5.00	6.00	14.00
POC (µg/L)	5	1,588.00	284.64	40.00	1,090.00	1,690.00	1,810.00
DOC (µmol/L)	3	941.00	105.06	99.00	838.00	937.00	1,048.00
TChl (µg/L)	5	3.08	0.77	0.69	2.17	3.21	4.16
Chl a (µg/L)	5	1.98	0.64	0.50	1.34	2.01	2.98
TSS (mg/L)	4	4.72	2.40	1.17	2.67	4.10	8.00
VSS (mg/L)	4	0.45	0.37	0.25	0.00	0.45	0.90
FSS (mg/L)	4	2.46	1.93	1.35	0.50	2.27	4.80
POM (%)	4	2.26	1.99	1.58	0.00	2.27	4.50
COND (µS/cm)	5	911.00	36.15	30.00	864.00	905.00	957.00
pH	5	8.55	0.03	0.02	8.50	8.54	8.59
POC (µmol/L)	5	132.32	23.72	3.33	90.83	140.82	150.82
PN (µmol/L)	5	11.61	2.97	1.57	6.85	11.85	14.64
PP (µmol/L)	5	0.41	0.04	0.03	0.35	0.42	0.45

Table A.4. Descriptive statistics for Lake Waterhen, 2017 water quality variables

Variable	Count	Mean	SD	MAD	Min	Median	Max
Secchi Depth (m)	2	3.36	0.63	0.44	2.91	3.36	3.80
TN (µg/L)	5	748.20	49.39	17.00	666.00	755.00	797.00
PN (µg/L)	5	135.20	15.22	4.00	109.00	142.00	146.00
TDN (µg/L)	5	613.00	37.23	26.00	557.00	611.00	655.00
TP (µg/L)	5	73.40	79.75	20.00	15.00	35.00	210.00
PP (µg/L)	6	13.00	2.53	1.50	9.00	13.00	16.00
DRP (µg/L)	5	27.60	38.08	8.00	1.00	9.00	93.00
TDP (µg/L)	5	33.40	39.83	9.00	5.00	14.00	102.00
POC (µg/L)	5	1,344.00	243.89	180.00	1,010.00	1,430.00	1,610.00
DOC (µmol/L)	5	1,520.00	790.17	34.00	953.00	1,249.00	2,915.00
TChl (µg/L)	6	2.71	0.77	0.35	1.52	2.76	3.84
Chl a (µg/L)	6	1.82	0.60	0.41	1.06	1.76	2.77
TSS (mg/L)	6	1.89	2.24	1.17	0.00	1.17	5.00
VSS (mg/L)	3	0.83	0.03	0.02	0.80	0.83	0.86
FSS (mg/L)	6	0.67	0.76	0.50	0.00	0.50	2.00
POM (%)	6	1.22	2.29	1.67	-2.00	1.00	4.00
COND (µS/cm)	6	927.00	66.62	17.50	800.00	942.50	994.00
pH	6	8.56	0.05	0.00	8.47	8.58	8.59
POC (µmol/L)	5	111.99	20.32	15.00	84.16	119.16	134.16
PN (µmol/L)	5	9.65	1.09	0.29	7.78	10.14	10.42
PP (µmol/L)	6	0.42	0.08	0.05	0.29	0.42	0.52

Table A.5. Descriptive statistics for Lake Manitoba, 2016 water quality variables

Variable	Count	Mean	SD	MAD	Min	Median	Max
Secchi Depth (m)							
TN (µg/L)	30	984.07	242.95	93.50	490.00	985.00	1,843.00
PN (µg/L)	29	353.48	203.24	71.00	130.00	334.00	1,170.00
TDN (µg/L)	30	649.03	50.76	39.00	529.00	655.00	741.00
TP (µg/L)	30	36.07	20.05	11.00	16.00	28.50	88.00
PP (µg/L)	30	25.90	16.33	5.50	11.00	19.00	70.00
DRP (µg/L)	25	2.24	1.76	1.00	1.00	2.00	8.00
TDP (µg/L)	30	8.47	4.26	2.50	3.00	7.00	18.00
POC (µg/L)	29	3,875.86	2,676.65	900.00	1,230.00	3,720.00	15,930.00
DOC (µmol/L)	26	1,289.12	1,147.15	83.50	827.00	967.50	5,340.00
TChl (µg/L)	30	11.12	8.80	3.83	1.59	7.78	38.74
Chl a (µg/L)	30	7.06	5.85	2.67	0.56	4.94	27.21
TSS (mg/L)	12	8.64	3.53	3.20	3.67	9.00	13.50
VSS (mg/L)	12	0.43	0.35	0.36	0.00	0.44	1.00
FSS (mg/L)	12	5.48	4.29	3.77	0.00	5.10	12.40
POM (%)	12	3.15	2.69	1.62	0.00	2.42	8.50
COND (µS/cm)	30	954.80	35.34	25.50	901.00	951.00	1,024.00
pH	30	8.61	0.05	0.04	8.50	8.61	8.71
POC (µmol/L)	29	322.96	223.04	74.99	102.49	309.97	1,327.39
PN (µmol/L)	29	25.24	14.51	5.07	9.28	23.85	83.53
PP (µmol/L)	30	0.84	0.53	0.18	0.35	0.61	2.26

Table A.6. Descriptive statistics for Lake Manitoba, 2017 water quality variables

Variable	Count	Mean	SD	MAD	Min	Median	Max
Secchi Depth (m)	41	1.07	0.54	0.26	0.44	0.90	2.48
TN (µg/L)	45	907.38	110.25	76.00	710.00	904.00	1,154.00
PN (µg/L)	46	259.89	97.11	66.00	117.00	242.00	505.00
TDN (µg/L)	45	648.78	60.98	37.00	551.00	641.00	840.00
TP (µg/L)	41	98.88	63.30	38.00	17.00	90.00	302.00
PP (µg/L)	46	25.54	15.62	6.00	10.00	18.00	79.00
DRP (µg/L)	41	31.51	31.61	17.00	1.00	19.00	138.00
TDP (µg/L)	45	38.89	31.62	19.00	5.00	31.00	150.00
POC (µg/L)	46	2,648.26	1,002.10	810.00	1,020.00	2,755.00	4,780.00
DOC (µmol/L)	38	1,239.87	715.20	121.50	520.00	1,085.50	4,592.00
TChl (µg/L)	46	11.12	8.25	4.30	2.16	7.96	29.84
Chl a (µg/L)	46	8.33	6.58	3.27	1.21	5.90	23.39
TSS (mg/L)	42	10.05	9.56	3.50	0.00	7.17	43.00
VSS (mg/L)	39	0.83	1.04	0.16	0.16	0.69	7.00
FSS (mg/L)	38	5.70	8.47	1.33	0.00	2.83	36.00
POM (%)	42	5.03	4.03	2.06	-13.00	5.94	10.00
COND (µS/cm)	46	952.57	62.89	32.50	759.00	959.50	1,062.00
pH	46	8.64	0.05	0.04	8.55	8.64	8.73
POC (µmol/L)	46	220.67	83.50	67.49	84.99	229.56	398.30
PN (µmol/L)	46	18.55	6.93	4.71	8.35	17.28	36.05
PP (µmol/L)	46	0.82	0.50	0.19	0.32	0.58	2.55

Appendix B — Normality Test Results

Table B.1. Shapiro-Wilk's test for normality - Lakes Winnipegosis (LWPO), Manitoba (LMB) and Waterhen (LWH), 2016-2017

Location	n	W	p	Variable
LMB	75	0.771	< .001	suspn
LWH	10	0.973	0.915	suspn
LWPO	39	0.964	0.233	suspn
LMB	75	0.972	0.097	tdn
LWH	10	0.971	0.904	tdn
LWPO	39	0.955	0.122	tdn
LMB	66	0.720	< .001	srp
LWH	9	0.588	< .001	srp
LWPO	32	0.664	< .001	srp
LMB	76	0.822	< .001	susp
LWH	11	0.964	0.820	susp
LWPO	39	0.930	0.018	susp
LMB	75	0.761	< .001	tdp
LWH	10	0.577	< .001	tdp

LWPO	39	0.629	< .001	tdp
LMB	75	0.669	< .001	suspc
LWH	10	0.902	0.231	suspc
LWPO	39	0.910	0.004	suspc
LMB	76	0.872	< .001	gf_chl
LWH	11	0.992	0.999	gf_chl
LWPO	39	0.929	0.017	gf_chl
LMB	76	0.866	< .001	hplc
LWH	11	0.952	0.670	hplc
LWPO	39	0.937	0.030	hplc
LMB	54	0.758	< .001	TSS
LWH	10	0.921	0.367	TSS
LWPO	31	0.633	< .001	TSS
LMB	54	0.636	< .001	FSS
LWH	10	0.843	0.048	FSS
LWPO	31	0.807	< .001	FSS
LMB	51	0.377	< .001	VSS
LWH	7	0.845	0.110	VSS

LWPO	27	0.664	< .001	VSS
LMB	76	0.960	0.017	cond
LWH	11	0.927	0.382	cond
LWPO	39	0.819	< .001	cond
LMB	76	0.967	0.044	ph
LWH	11	0.839	0.031	ph
LWPO	39	0.941	0.042	ph
LMB	64	0.439	< .001	doc
LWH	8	0.643	< .001	doc
LWPO	27	0.617	< .001	doc
LMB	75	0.864	< .001	total_n
LWH	10	0.950	0.667	total_n
LWPO	39	0.959	0.167	total_n
LMB	71	0.836	< .001	total_p
LWH	10	0.580	< .001	total_p
LWPO	36	0.678	< .001	total_p

n = number of samples. The W statistic indicates how close to a normal distribution the data is, and the p-value indicates how significantly different the data is from normal. P < 0.05 is considered statistically significant and indicates data is not normally distributed.

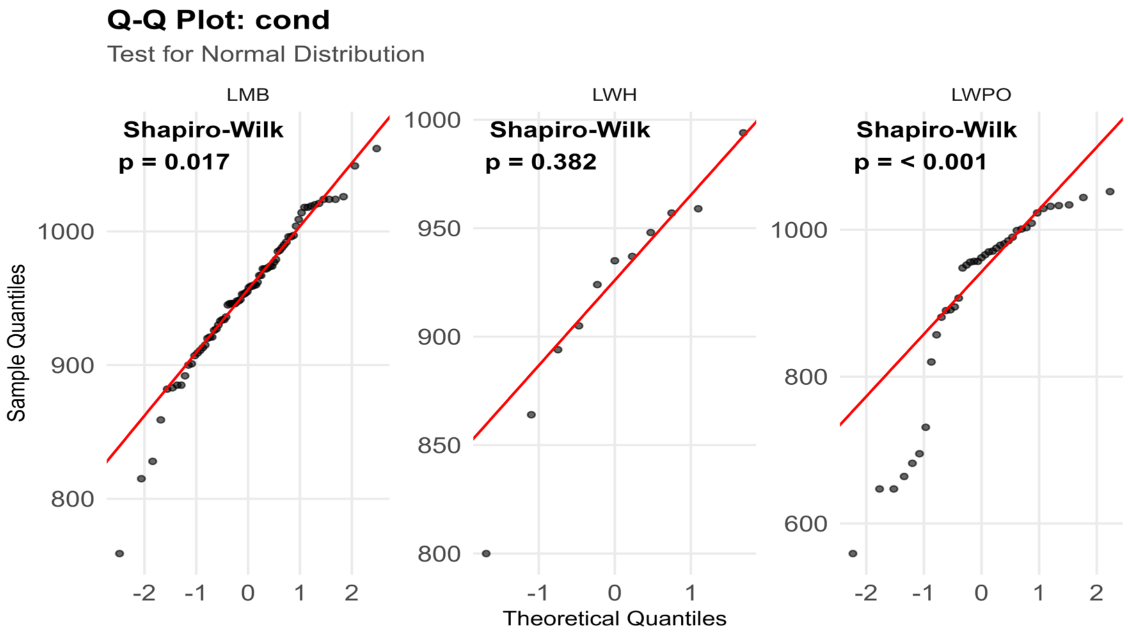

Figure B.1. QQ plot for Specific Conductivity with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

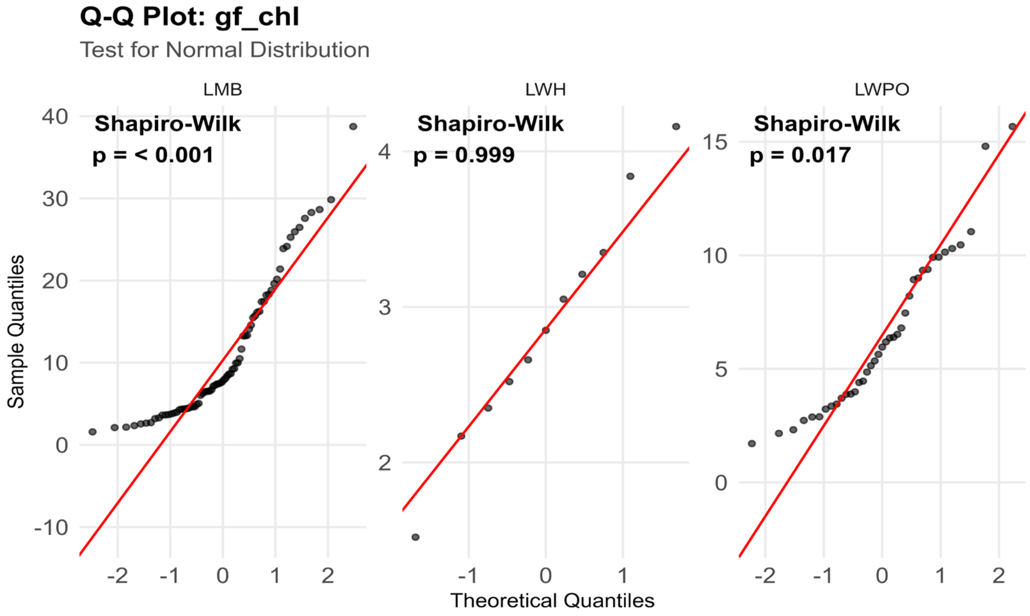

Figure B.2. QQ plot for total chlorophyll (TotC) with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

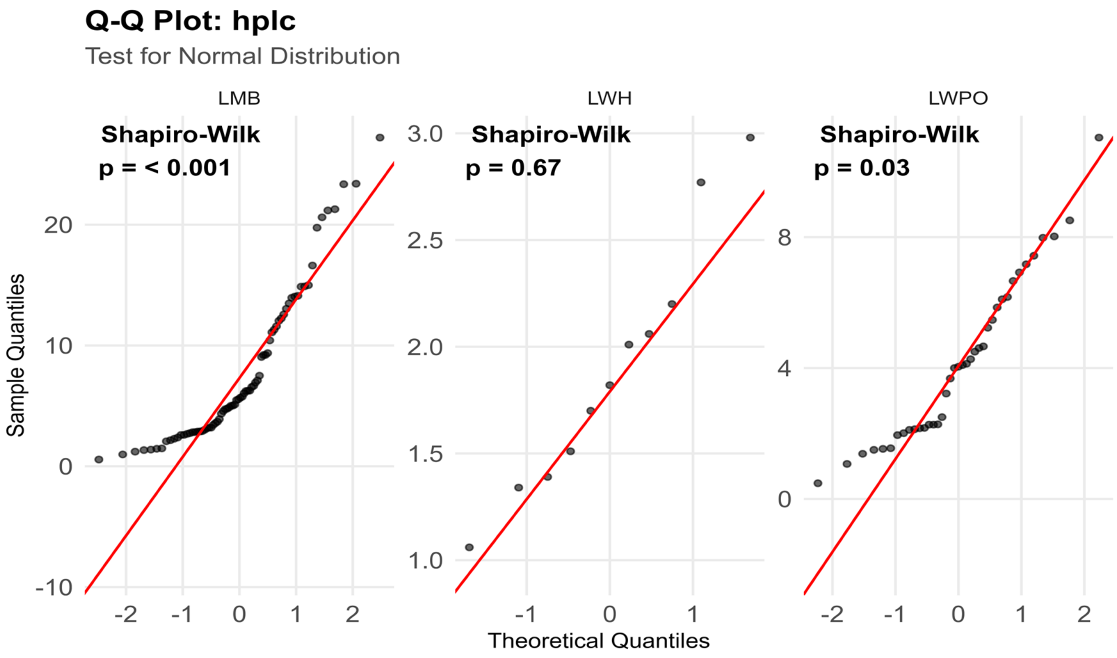

Figure B.3. QQ plot for chlorophyll with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

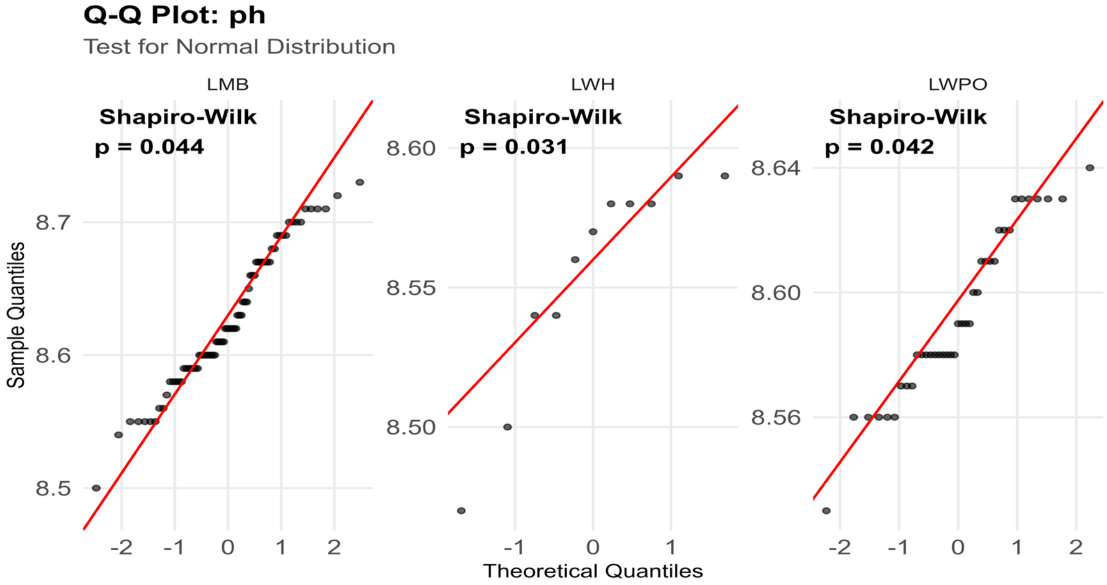

Figure B.4. QQ plot for pH with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

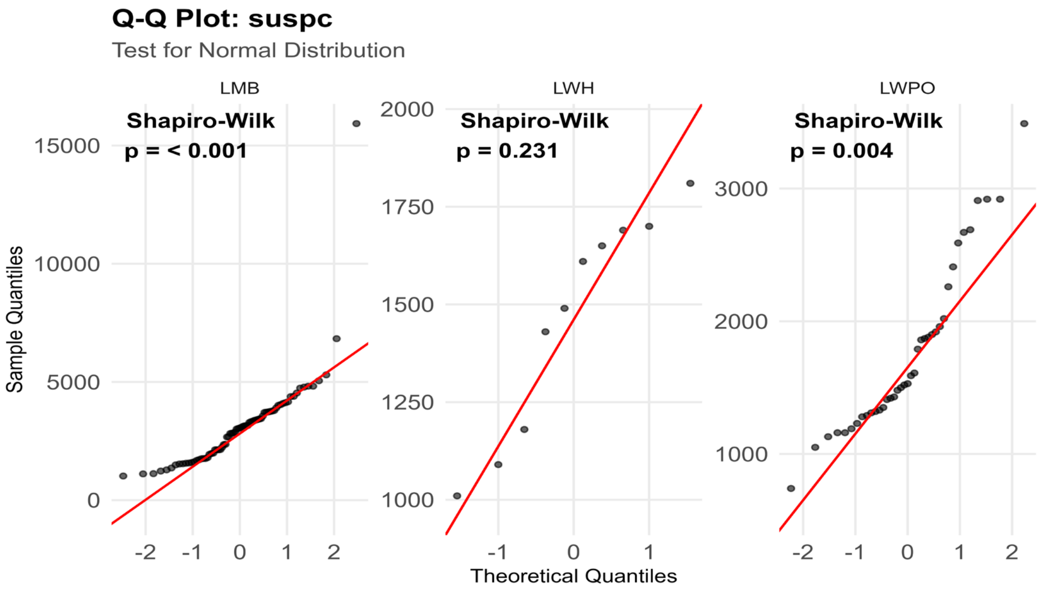

Figure B.5. QQ plot for suspended carbon with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

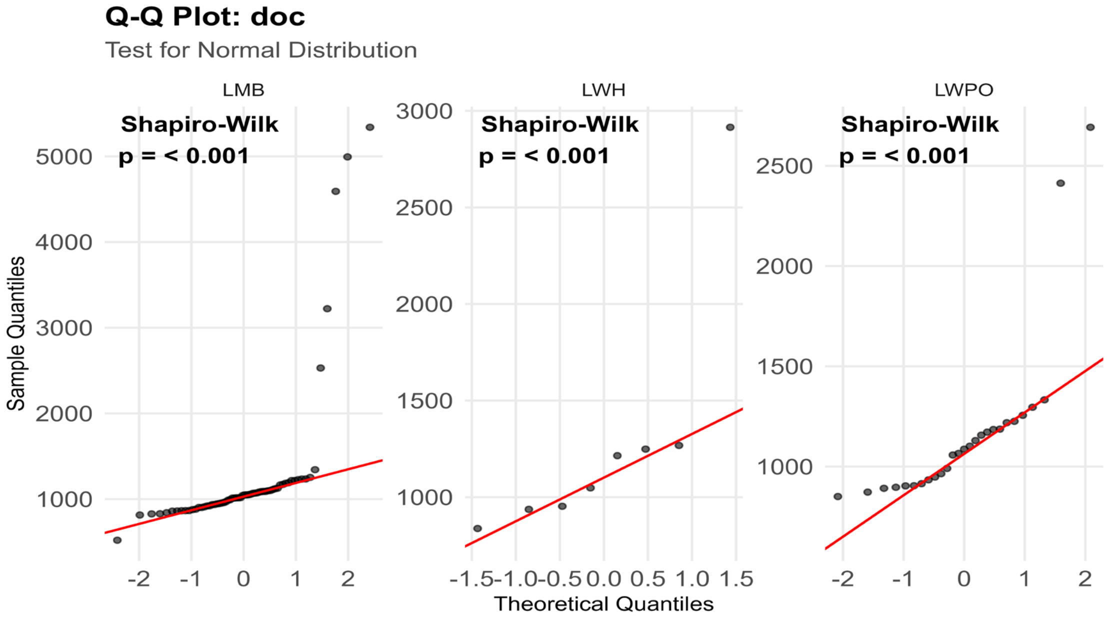

Figure B.6. QQ plot for dissolved organic carbon with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

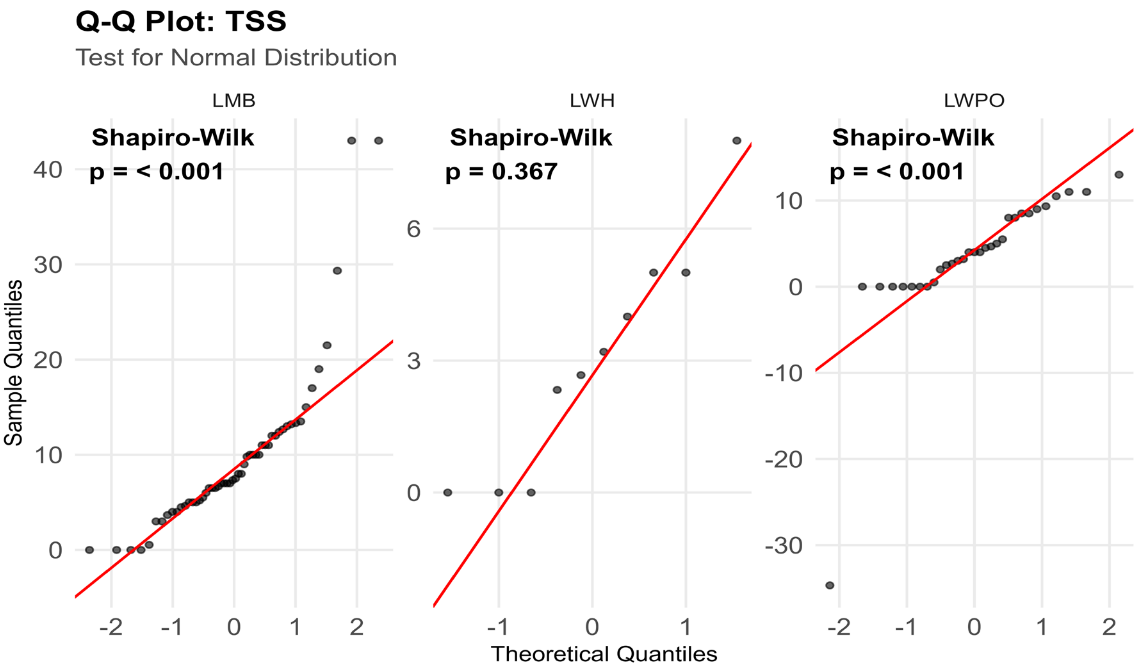

Figure B.7. QQ plot for total suspended solids with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

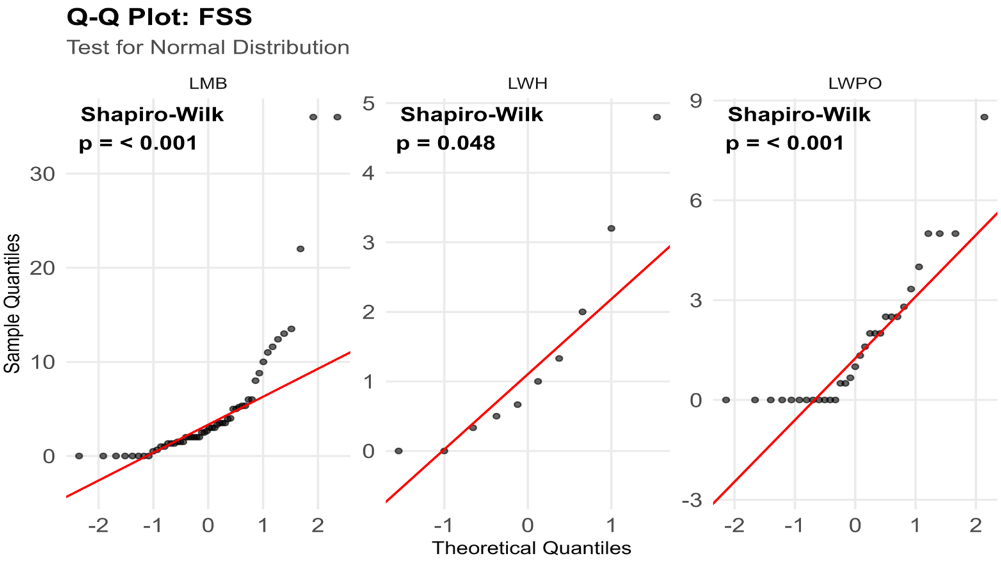

Figure B.8. QQ plot for fixed suspended solids with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

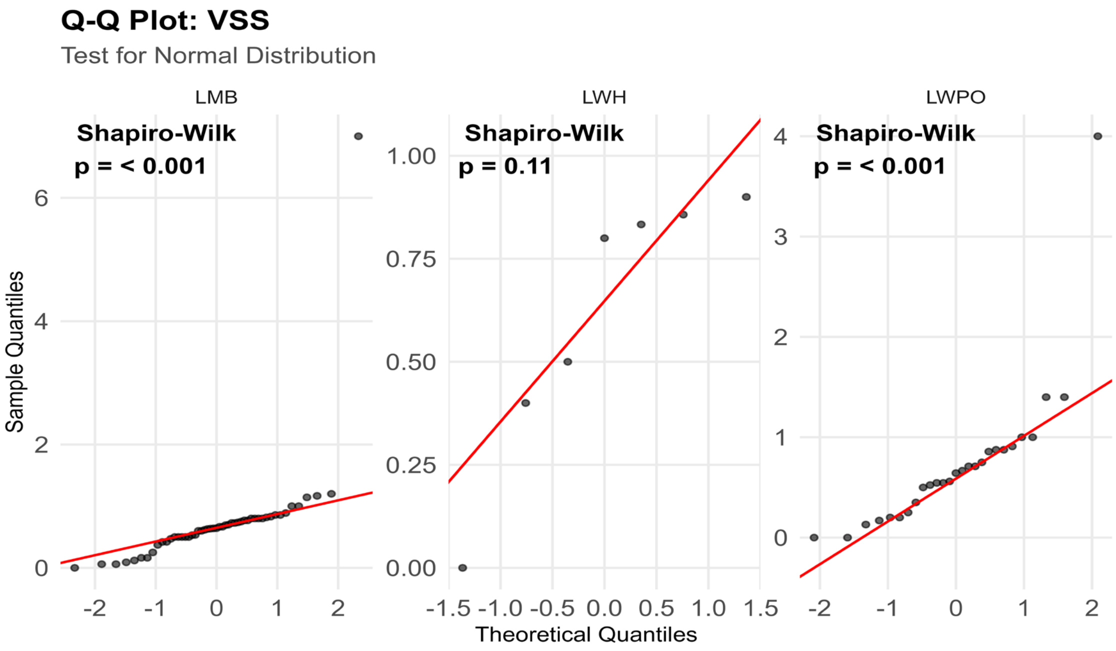

Figure B.9. QQ plot for volatile suspended solids with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

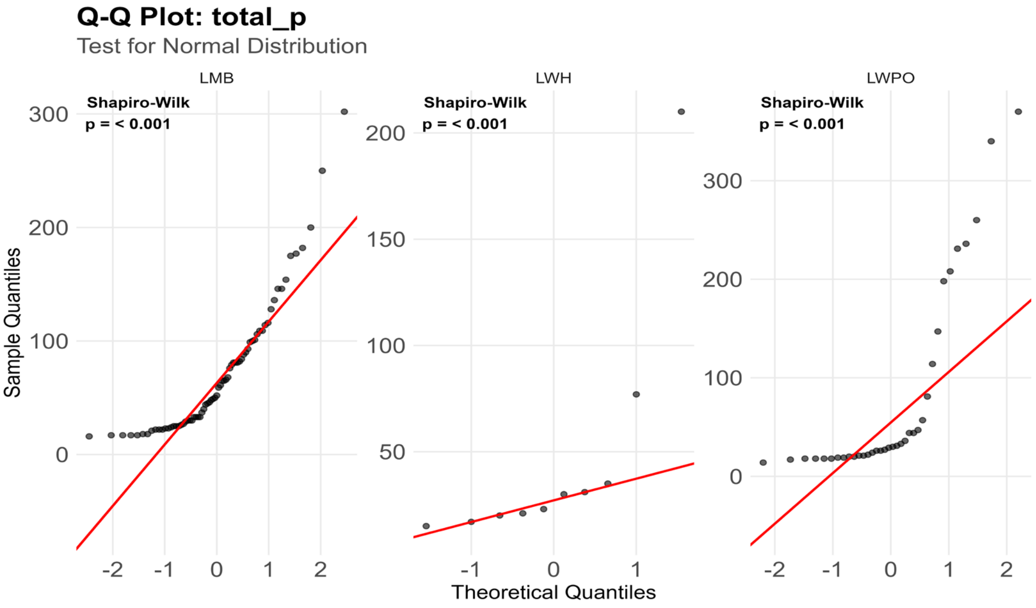

Figure B.10. QQ plot for total phosphorus with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

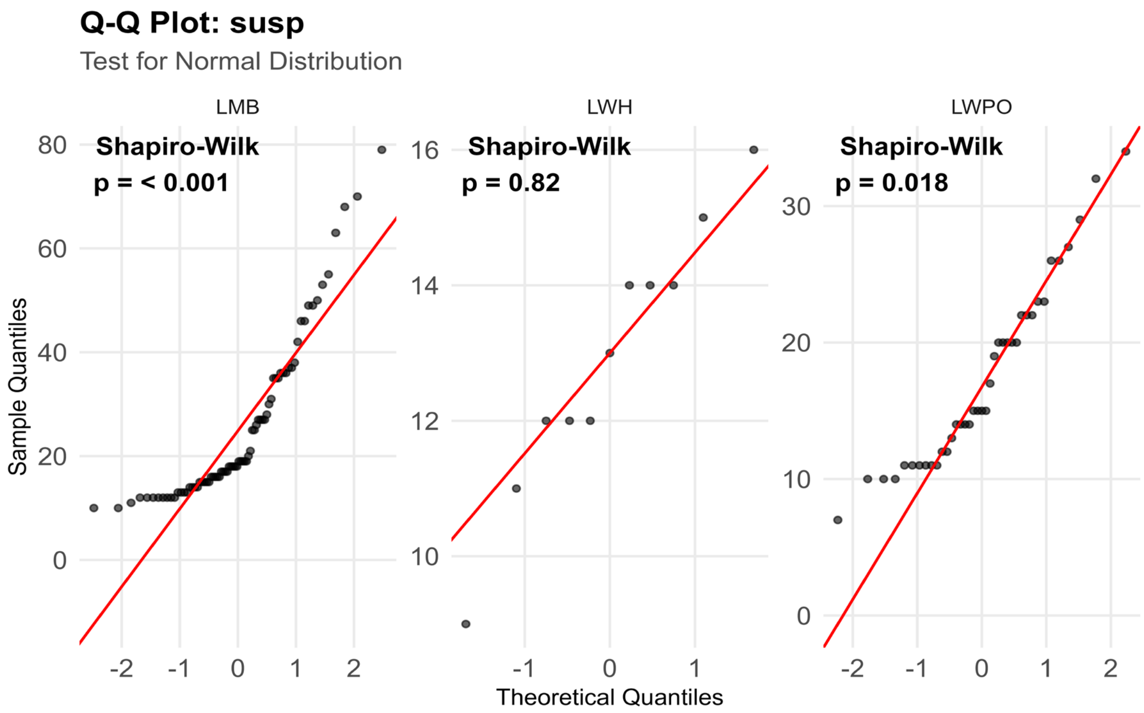

Figure B.11. QQ plot for suspended phosphorus with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

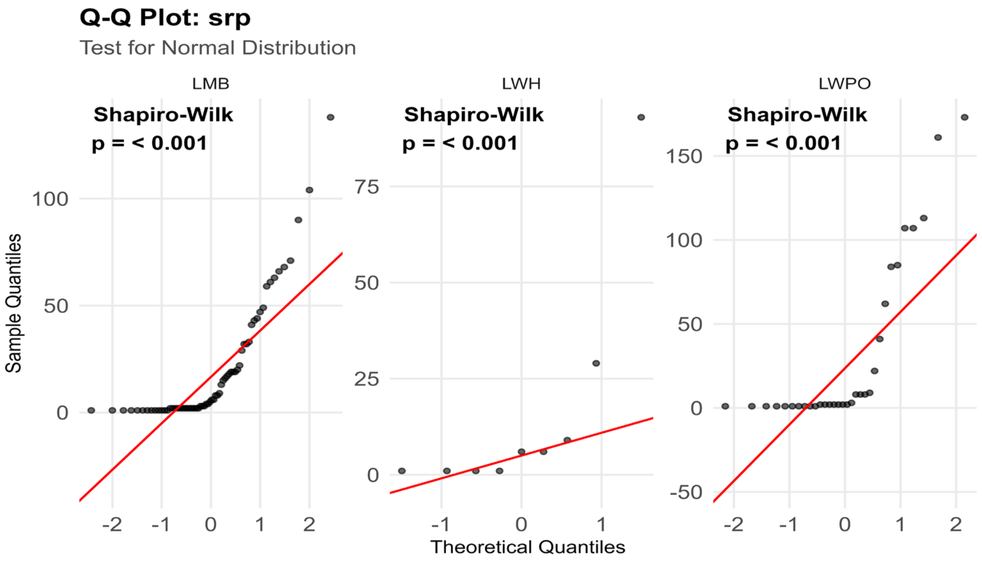

Figure B.12. QQ plot for soluble reactive phosphorus with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant.The red line represents a normal distribution with the same mean and standard deviation as the data

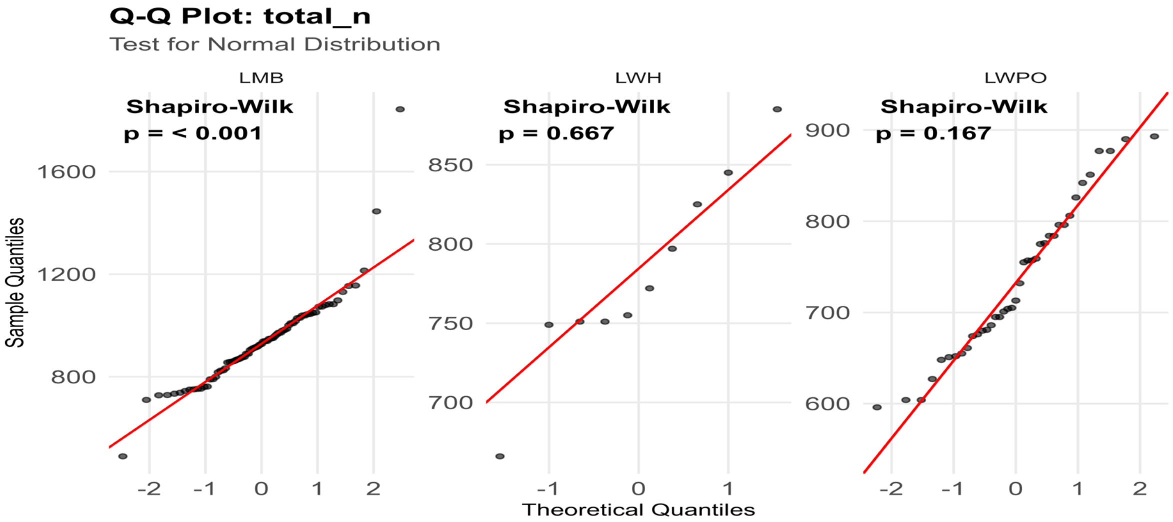

Figure B.13. QQ plot for total nitrogen with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

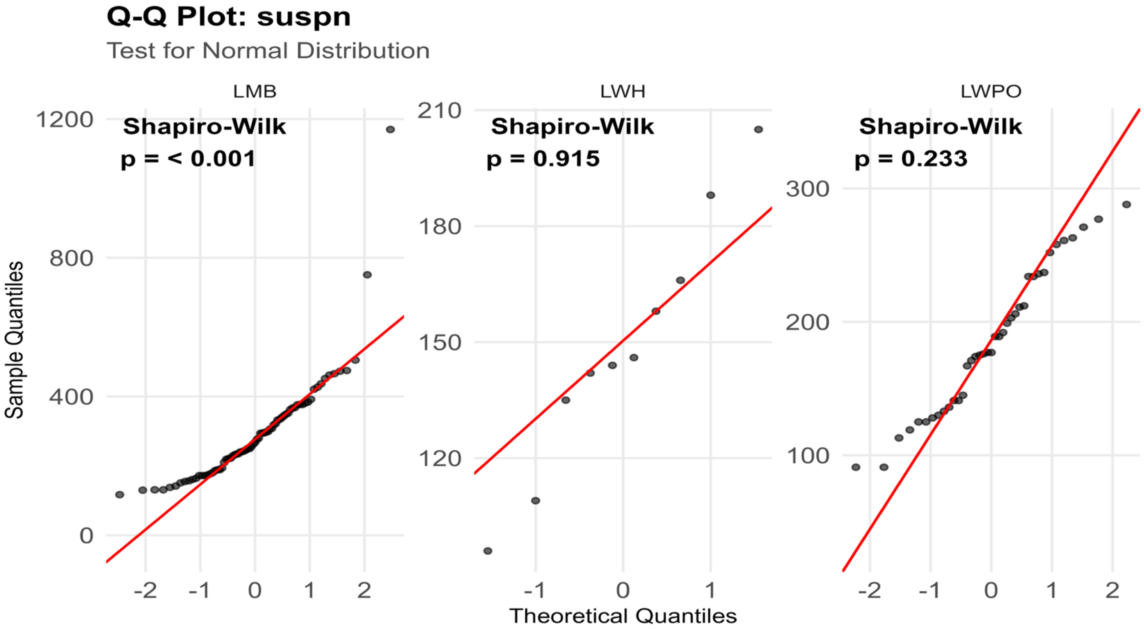

Figure B.14. QQ plot for suspended nitrogen with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

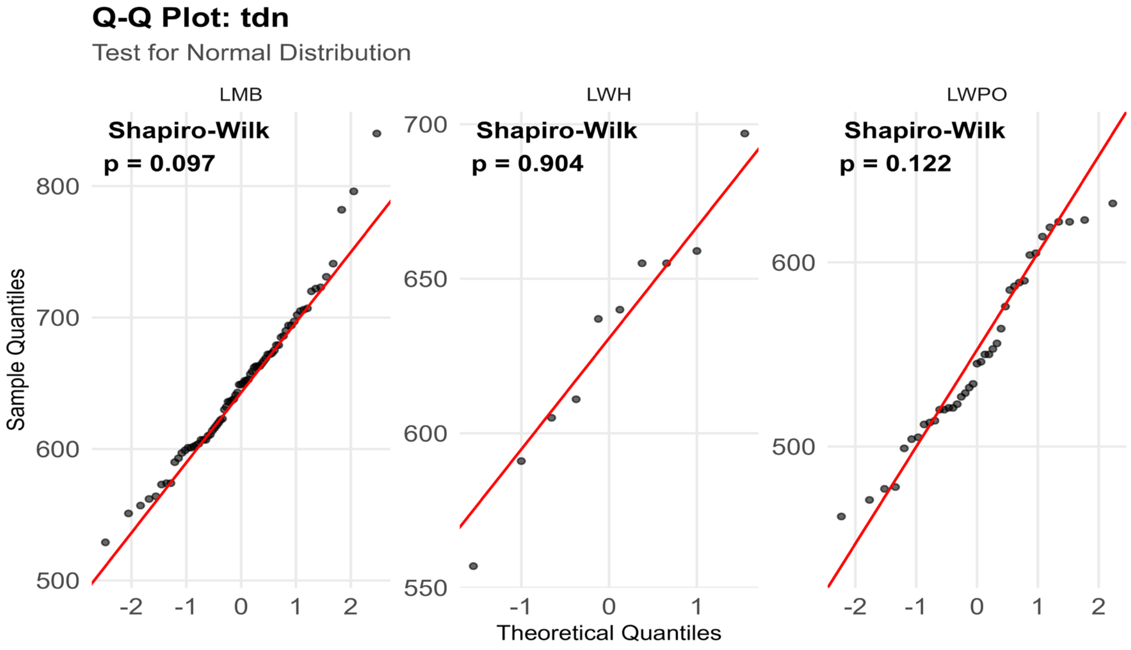

Figure B.15. QQ plot for total dissolved nitrogen with Shapiro-Wilk's p-value. A p-value of < 0.05 is considered statistically significant. The red line represents a normal distribution with the same mean and standard deviation as the data

Appendix C — Non-parametric test results

Table C.1. Wilcoxon Rank Sum Test (Mann Whitney U) for Lakes Winnipegosis and Manitoba, per basin, $p <= 0.05$

Lake	Year	Basin Pair	n1	n2	p	Variable	p.adj
Manitoba	2016	NW vs NE	9	10	0.0007	suspn	0.0011
Manitoba	2016	NW vs SM	9	10	0.0004	suspn	0.0011
Manitoba	2016	NE vs SM	10	10	0.0211	suspn	0.0211
Manitoba	2016	NW vs NE	9	11	0.0007	tdn	0.0019
Manitoba	2016	NW vs SM	9	10	0.0012	tdn	0.0019
Manitoba	2016	NW vs NE	9	11	0.0064	susp	0.0078
Manitoba	2016	NW vs SM	9	10	0.0003	susp	0.0008
Manitoba	2016	NE vs SM	11	10	0.0078	susp	0.0078
Manitoba	2016	NW vs SM	9	10	0.0007	tdp	0.0021
Manitoba	2016	NE vs SM	11	10	0.0033	tdp	0.0049
Manitoba	2016	NW vs NE	9	10	0.0004	suspc	0.0006
Manitoba	2016	NW vs SM	9	10	0.0004	suspc	0.0006
Manitoba	2016	NE vs SM	10	10	0.0257	suspc	0.0257
Manitoba	2016	NW vs NE	9	11	0.0024	gf_chl	0.0036
Manitoba	2016	NW vs SM	9	10	0.0005	gf_chl	0.0016
Manitoba	2016	NE vs SM	11	10	0.0151	gf_chl	0.0151
Manitoba	2016	NW vs NE	9	11	0.0226	hplc	0.0226
Manitoba	2016	NW vs SM	9	10	0.0005	hplc	0.0016
Manitoba	2016	NE vs SM	11	10	0.0183	hplc	0.0226
Manitoba	2016	NW vs NE	7	4	0.0138	tss	0.0414
Manitoba	2016	NW vs NE	9	11	0.0009	cond	0.0014
Manitoba	2016	NW vs SM	9	10	0.0005	cond	0.0014
Manitoba	2016	NE vs SM	11	10	0.0484	cond	0.0484
Manitoba	2016	NW vs NE	9	11	0.0191	ph	0.0287

Manitoba	2016	NW vs SM	9	10	0.0032	ph	0.0095
Manitoba	2016	NE vs SM	11	10	0.0355	ph	0.0355
Manitoba	2016	NW vs NE	9	11	0.0024	total_n	0.0036
Manitoba	2016	NW vs SM	9	10	0.0003	total_n	0.0008
Manitoba	2016	NE vs SM	11	10	0.0151	total_n	0.0151
Manitoba	2016	NW vs NE	9	11	0.0431	total_p	0.0431
Manitoba	2016	NW vs SM	9	10	0.0003	total_p	0.0008
Manitoba	2016	NE vs SM	11	10	0.0101	total_p	0.0151
Manitoba	2016	NW vs NE	9	10	0.0004	suspC_uM_L	0.0006
Manitoba	2016	NW vs SM	9	10	0.0004	suspC_uM_L	0.0006
Manitoba	2016	NE vs SM	10	10	0.0257	suspC_uM_L	0.0257
Manitoba	2016	NW vs NE	9	10	0.0007	suspN_uM_L	0.0011
Manitoba	2016	NW vs SM	9	10	0.0004	suspN_uM_L	0.0011
Manitoba	2016	NE vs SM	10	10	0.0211	suspN_uM_L	0.0211
Manitoba	2016	NW vs NE	9	11	0.0064	suspP_uM_L	0.0078
Manitoba	2016	NW vs SM	9	10	0.0003	suspP_uM_L	0.0008
Manitoba	2016	NE vs SM	11	10	0.0078	suspP_uM_L	0.0078
Manitoba	2017	NW vs NE	9	12	0.0020	suspn	0.0020
Manitoba	2017	NW vs SM	9	25	0.0000	suspn	0.0000
Manitoba	2017	NE vs SM	12	25	0.0001	suspn	0.0001
Manitoba	2017	NW vs NE	9	12	0.0129	susp	0.0129
Manitoba	2017	NW vs SM	9	25	0.0000	susp	0.0000
Manitoba	2017	NE vs SM	12	25	0.0000	susp	0.0000
Manitoba	2017	NW vs NE	9	12	0.0018	suspc	0.0026
Manitoba	2017	NW vs SM	9	25	0.0000	suspc	0.0000
Manitoba	2017	NE vs SM	12	25	0.0061	suspc	0.0061
Manitoba	2017	NW vs NE	9	12	0.0025	gf_chl	0.0025
Manitoba	2017	NW vs SM	9	25	0.0000	gf_chl	0.0001

Manitoba	2017	NE vs SM	12	25	0.0003	gf_chl	0.0004
Manitoba	2017	NW vs NE	9	12	0.0095	hplc	0.0095
Manitoba	2017	NW vs SM	9	25	0.0001	hplc	0.0003
Manitoba	2017	NE vs SM	12	25	0.0007	hplc	0.0010
Manitoba	2017	NW vs SM	9	22	0.0079	tss	0.0118
Manitoba	2017	NE vs SM	11	22	0.0071	tss	0.0118
Manitoba	2017	NE vs SM	10	22	0.0082	tripton	0.0245
Manitoba	2017	NE vs SM	10	20	0.0028	loi	0.0083
Manitoba	2017	NW vs SM	9	25	0.0049	cond	0.0148
Manitoba	2017	NW vs NE	9	12	0.0315	ph	0.0315
Manitoba	2017	NW vs SM	9	25	0.0004	ph	0.0012
Manitoba	2017	NE vs SM	12	25	0.0010	ph	0.0015
Manitoba	2017	NW vs NE	9	12	0.0050	total_n	0.0050
Manitoba	2017	NW vs SM	9	24	0.0000	total_n	0.0000
Manitoba	2017	NE vs SM	12	24	0.0000	total_n	0.0000
Manitoba	2017	NW vs SM	9	22	0.0084	POM	0.0251
Manitoba	2017	NW vs NE	9	12	0.0018	suspC_uM_L	0.0026
Manitoba	2017	NW vs SM	9	25	0.0000	suspC_uM_L	0.0000
Manitoba	2017	NE vs SM	12	25	0.0061	suspC_uM_L	0.0061
Manitoba	2017	NW vs NE	9	12	0.0020	suspN_uM_L	0.0020
Manitoba	2017	NW vs SM	9	25	0.0000	suspN_uM_L	0.0000
Manitoba	2017	NE vs SM	12	25	0.0001	suspN_uM_L	0.0001
Manitoba	2017	NW vs NE	9	12	0.0129	suspP_uM_L	0.0129
Manitoba	2017	NW vs SM	9	25	0.0000	suspP_uM_L	0.0000
Manitoba	2017	NE vs SM	12	25	0.0000	suspP_uM_L	0.0000
Winnipegosis	2017	SEE vs SE	8	3	0.0321	cond	0.0481
Winnipegosis	2017	SEE vs NW	8	6	0.0024	cond	0.0143
Winnipegosis	2017	SW vs NW	5	6	0.0080	cond	0.0239

Winnipegosis	2017	SE vs NW	3	6	0.0275	cond	0.0481

Table C.2. Wilcoxon Rank Sum Test (Mann Whitney U) between years (2016-2017) for Lakes Winnipegosis and Manitoba, $p <= 0.05$

Lake	Year 1	Year 2	n1	n2	p	Variable	p.adj
Manitoba	2016	2017	29	46	0.0180	suspn	0.0399
Manitoba	2016	2017	25	41	0.0000	srp	0.0000
Manitoba	2016	2017	30	45	0.0000	tdp	0.0000
Manitoba	2016	2017	29	46	0.0049	suspc	0.0197
Manitoba	2016	2017	12	39	0.0263	loi	0.0497
Manitoba	2016	2017	30	46	0.0093	ph	0.0311
Manitoba	2016	2017	26	38	0.0171	doc	0.0399
Manitoba	2016	2017	30	41	0.0000	total_p	0.0000
Manitoba	2016	2017	12	42	0.0273	POM	0.0497
Winnipegosis	2016	2017	13	19	0.0008	srp	0.0054
Winnipegosis	2016	2017	17	22	0.0001	tdp	0.0008
Winnipegosis	2016	2017	9	16	0.0017	loi	0.0083
Winnipegosis	2016	2017	17	22	0.0149	cond	0.0495
Winnipegosis	2016	2017	14	13	0.0045	doc	0.0181
Winnipegosis	2016	2017	17	19	0.0000	total_p	0.0008

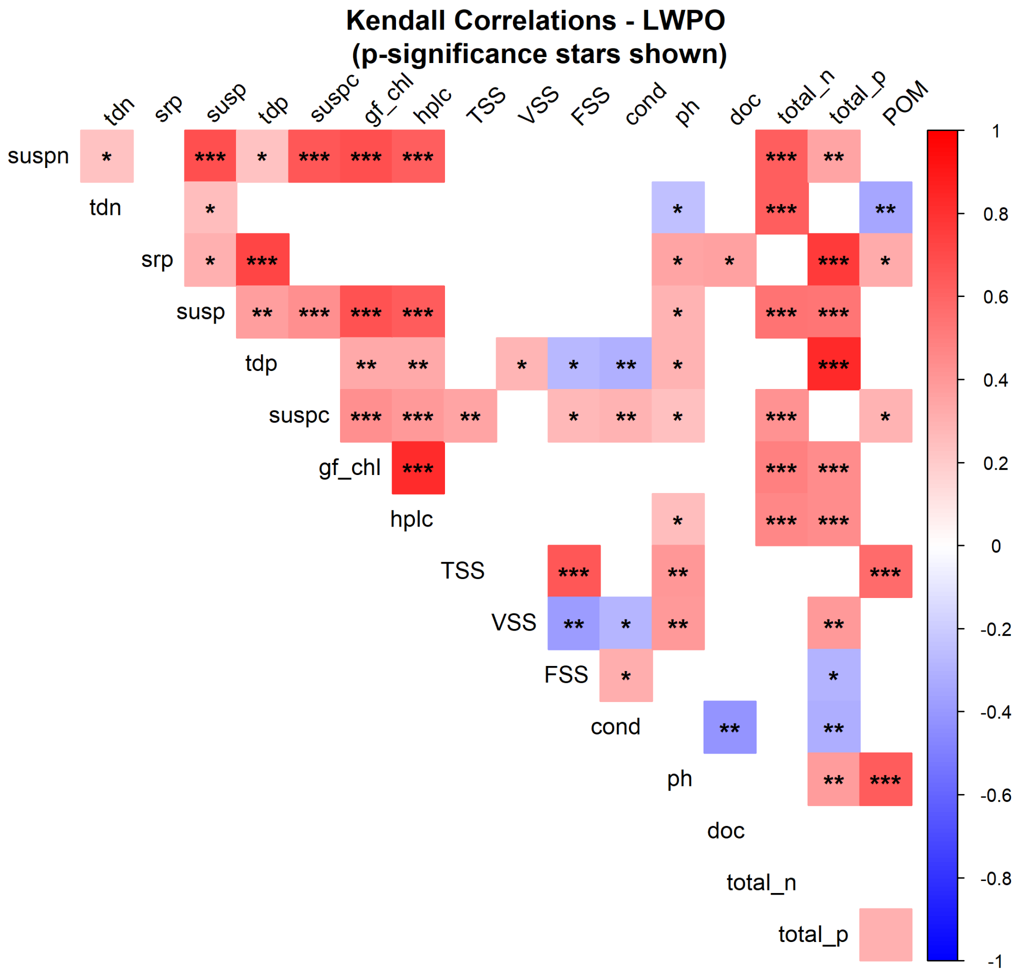

Figure C.1. Lake Winnipegosis Kendall Correlation Matrix. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Red = positive correlation, blue = negative correlation.

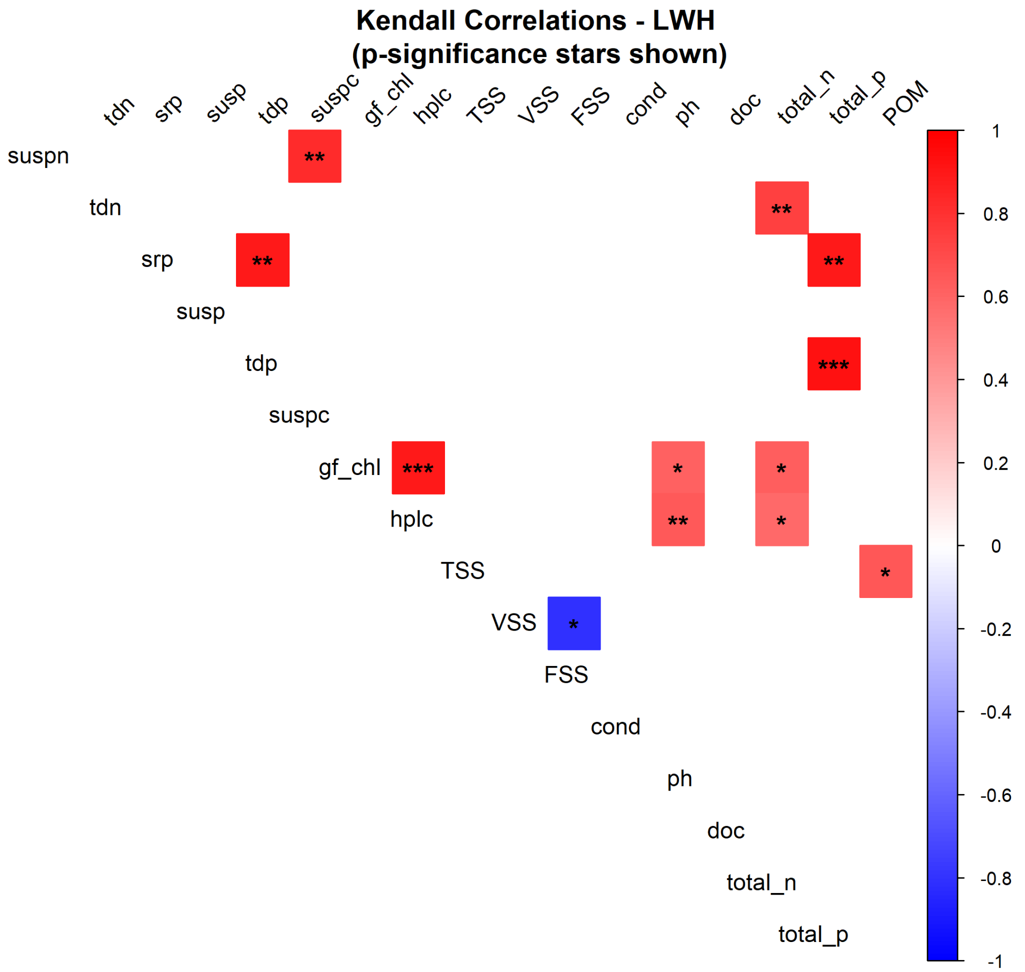

Figure C.2. Lake Waterhen Kendall Correlation Matrix. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Red = positive correlation, blue = negative correlation.

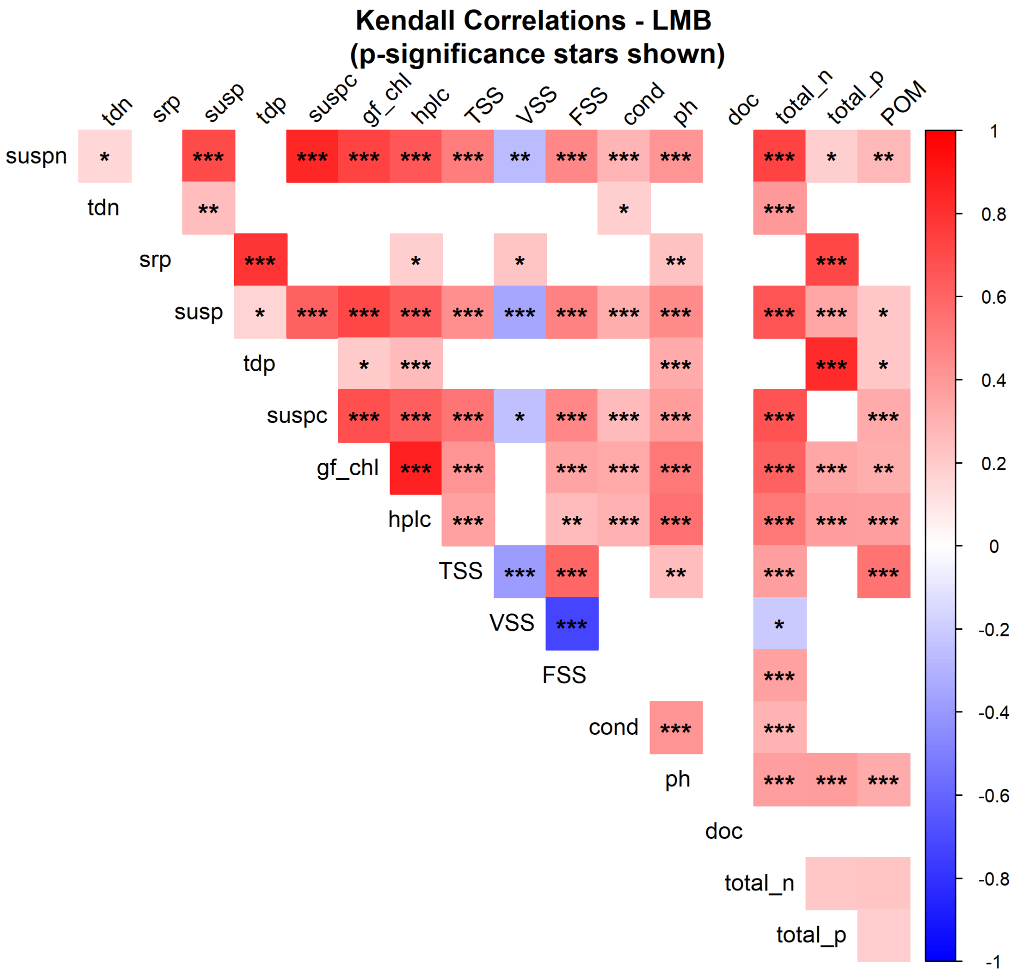

Figure C.3. Lake Manitoba Kendall Correlation Matrix. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Red = positive correlation, blue = negative correlation.

Table C.3. Mann-Kendall trend table for Lakes Winnipegosis and Manitoba, for variables between years, p < = 0.05

Lake	Variable	n	Tau	p value	S statistic	Theil Sen Slope per day	Theil Sen Slope per year	Trend	Significance
Manitoba	DOC	76	0.1825	0.0335	368	0.3630	132.5838	Increasing	*
Waterhen	TSS	11	-0.6222	0.0142	-28	-0.0090	-3.3054	Decreasing	*
Waterhen	FSS	11	-0.5778	0.0248	-26	-0.0053	-1.9428	Decreasing	*
Waterhen	DRP	11	0.5278	0.0473	19	0.0170	6.2273	Increasing	*
Winnipegosis	DOC	39	0.3732	0.0067	131	0.5667	206.9750	Increasing	**

Appendix D — Chemistry Maps of the upper MBGL for 2016-2017

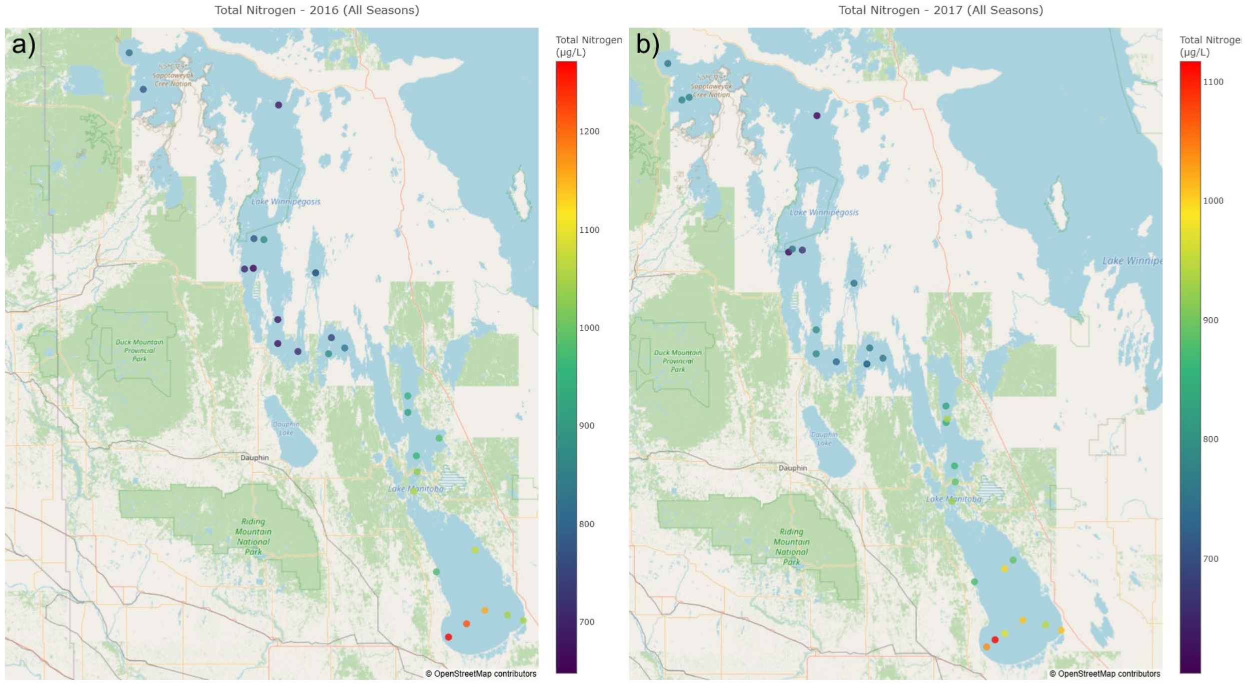

Figure D.1. Map of upperMBGL Total Nitrogen, 2016 (a) and 2017 (b)

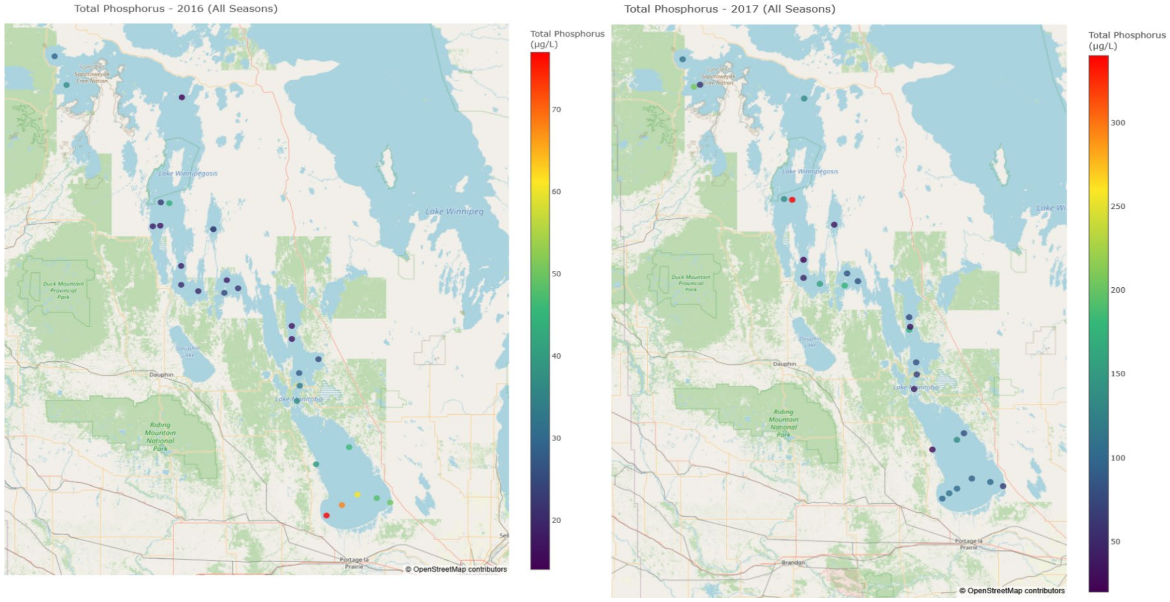

Figure D.2. Map of upperMBGL Total Phosphorus, 2016-2017

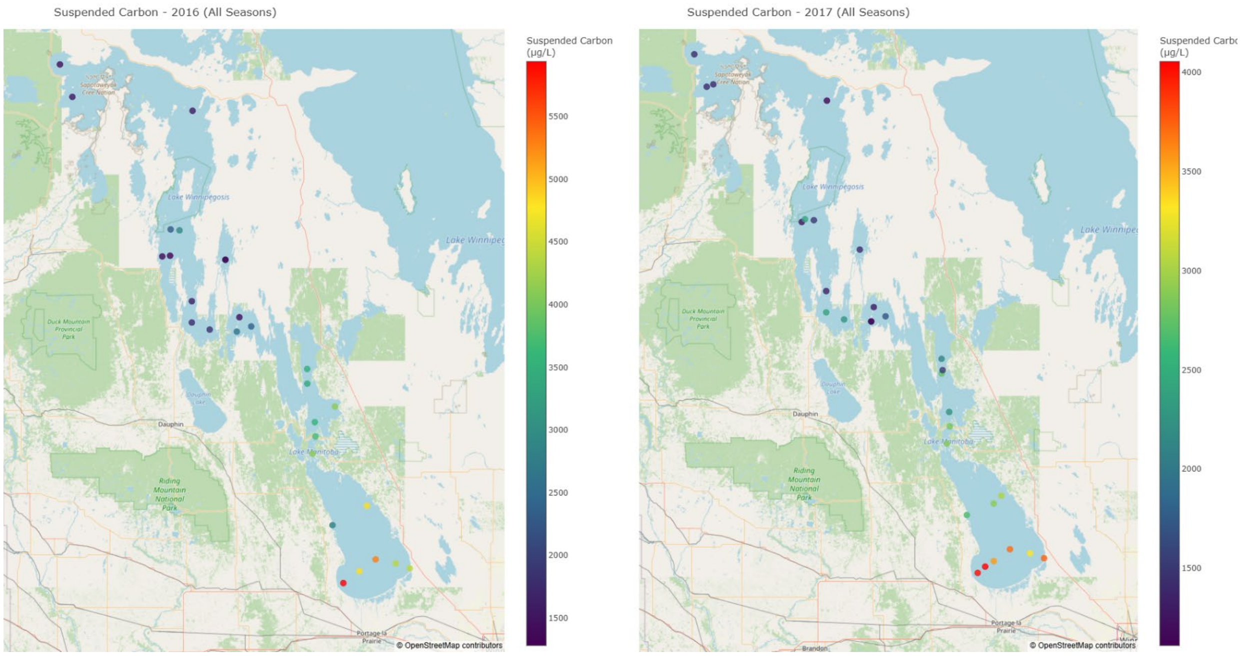

Figure D.3. Map of upperMBGL Suspended Carbon, 2016-2017

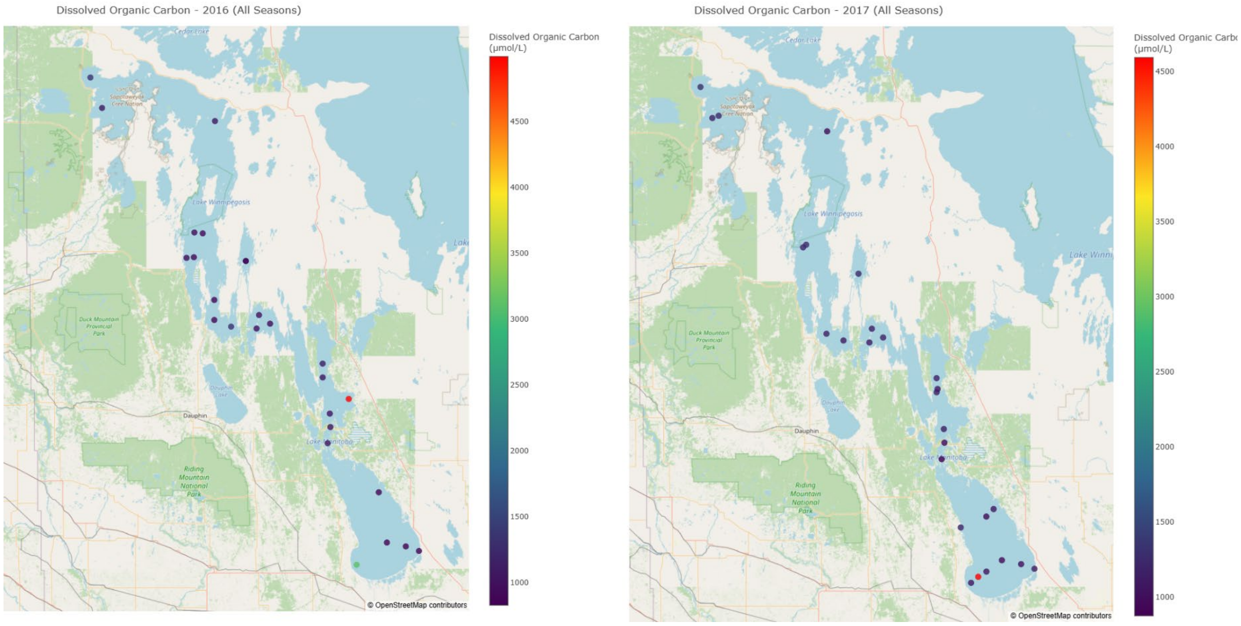

Figure D.4. Map of upperMBGL Dissolved Organic Carbon, 2016-2017

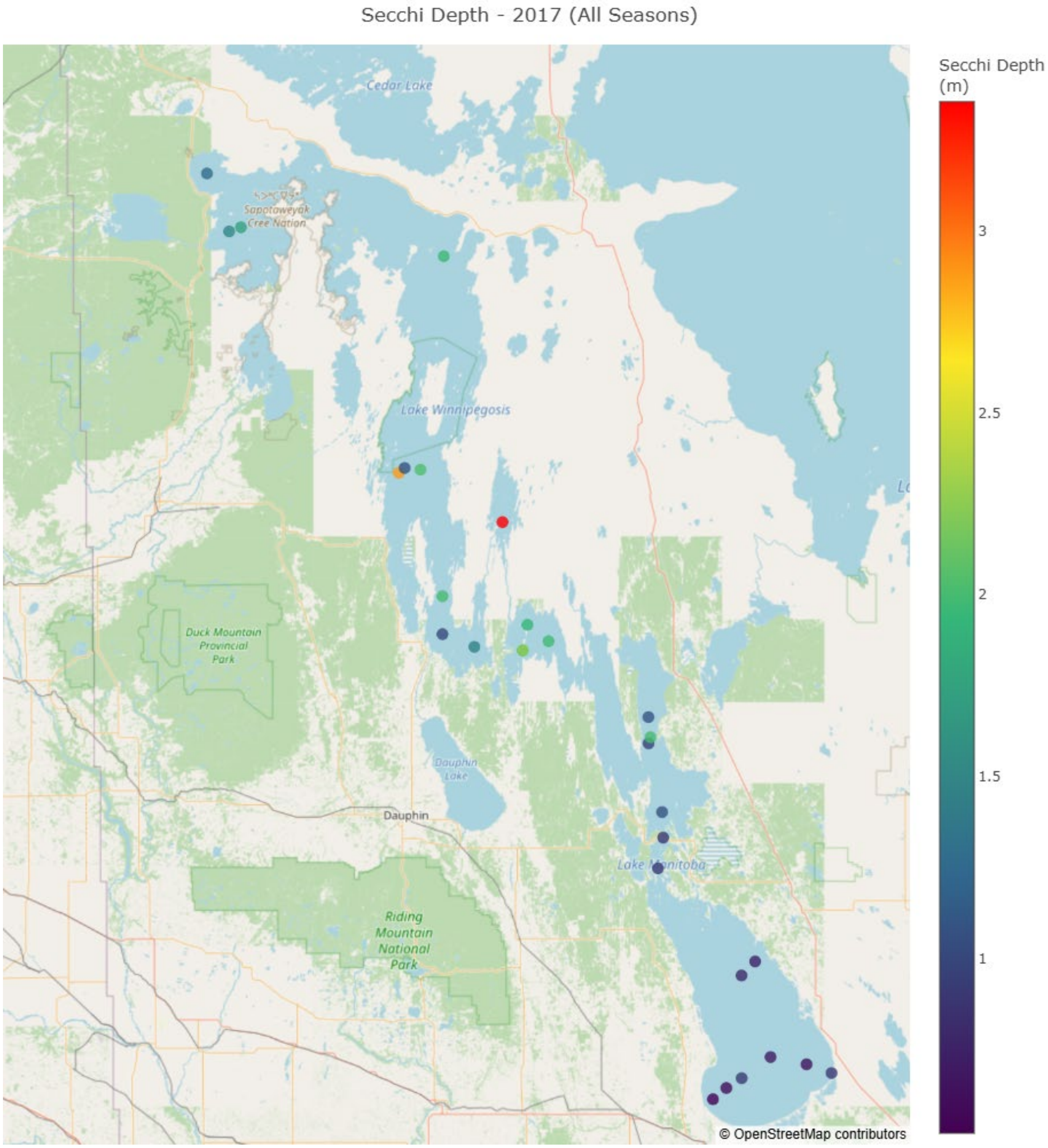

Figure D.5. Map of upperMBGL Secchi Depth (m), 2017

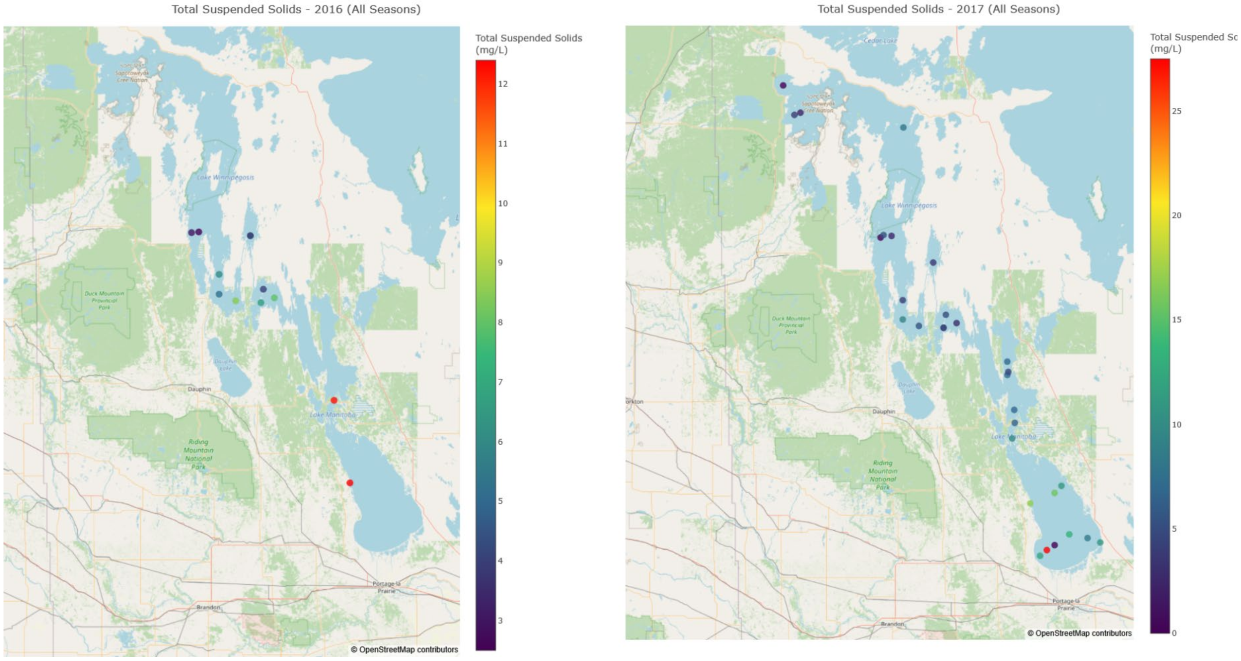

Figure D.6. Map of TSS in the upperMBGL, 2016-2017

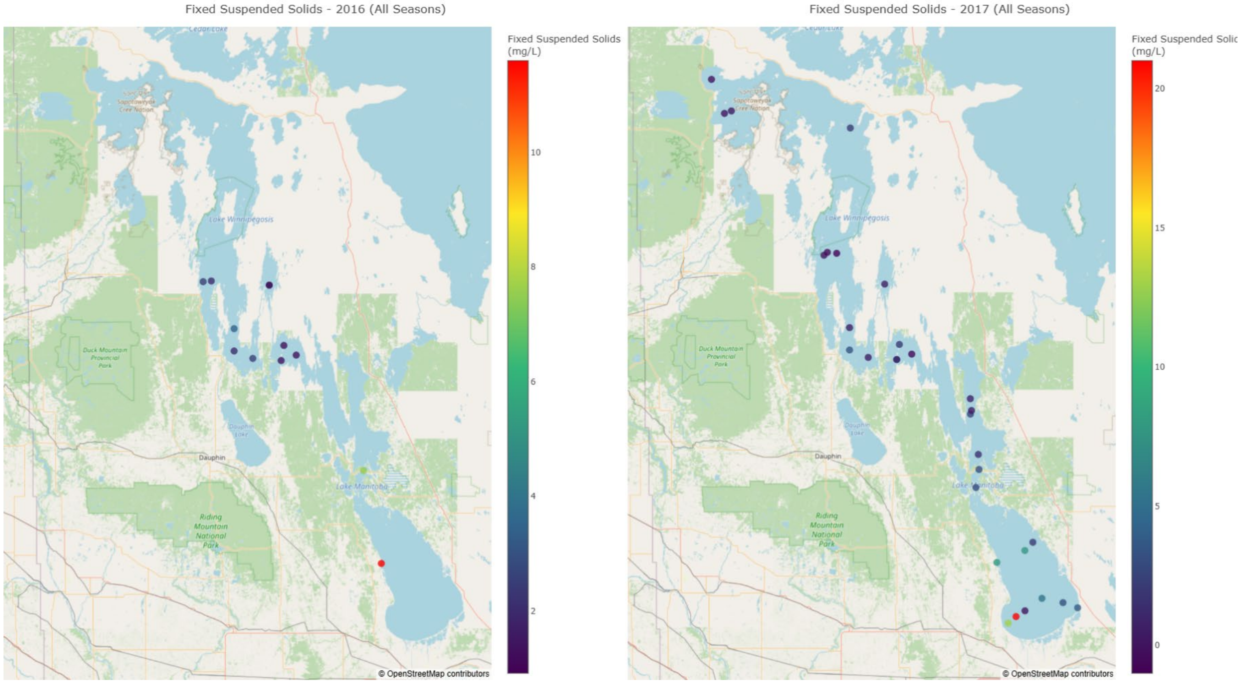

Figure D.7. Map of FSS in the upperMBGL, 2016-2017

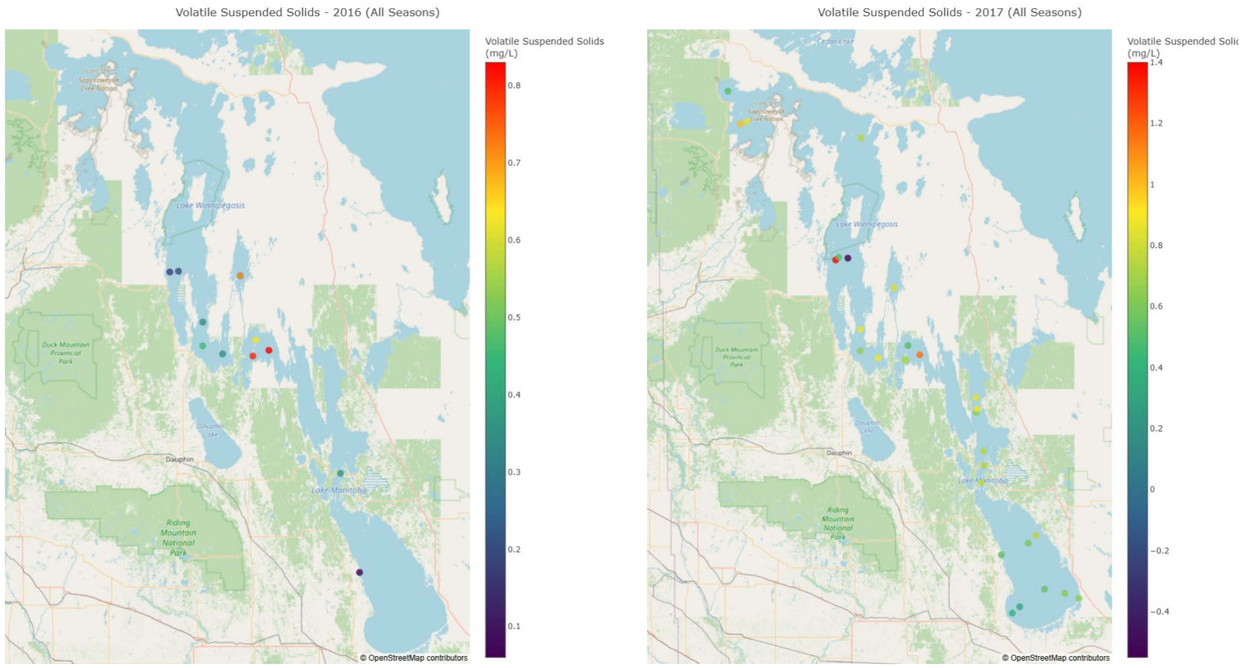

Figure D.8. Map of VSS in the upperMBGL, 2016-2017

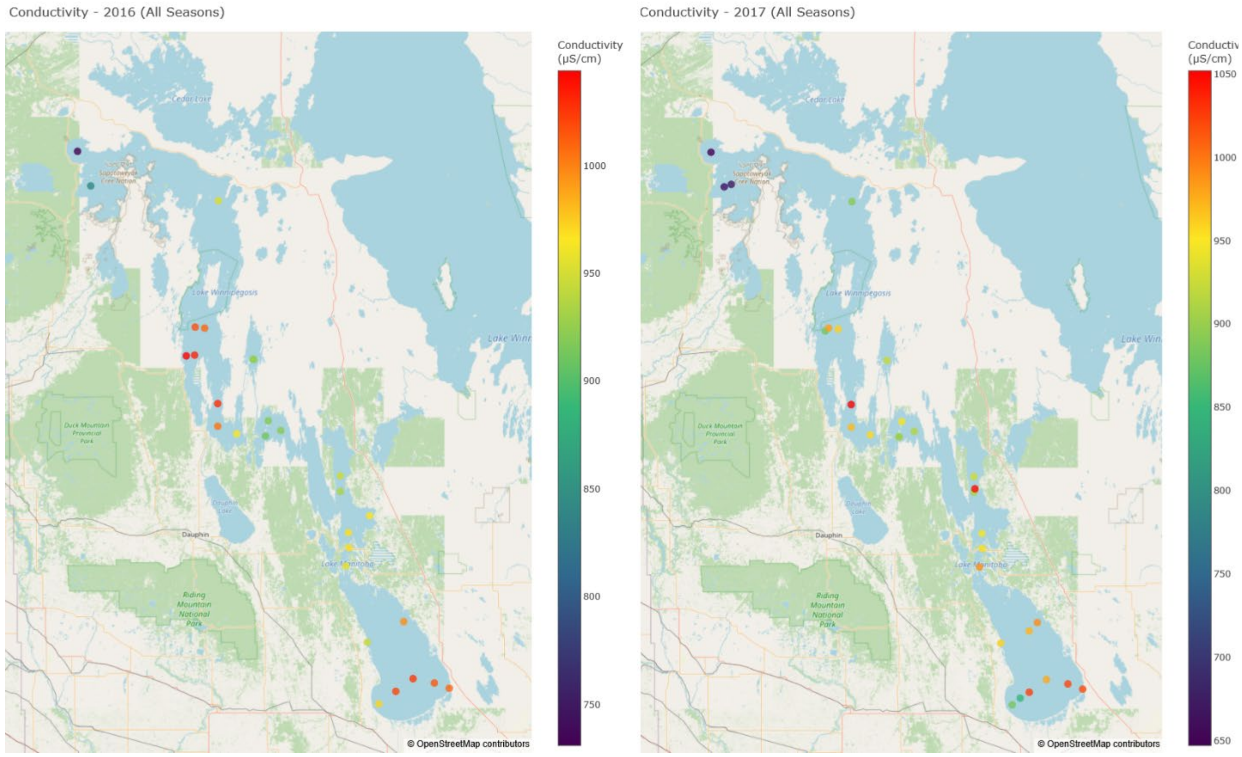

Figure D.9. Map of upperMBGL Conductivity, 2016-2017

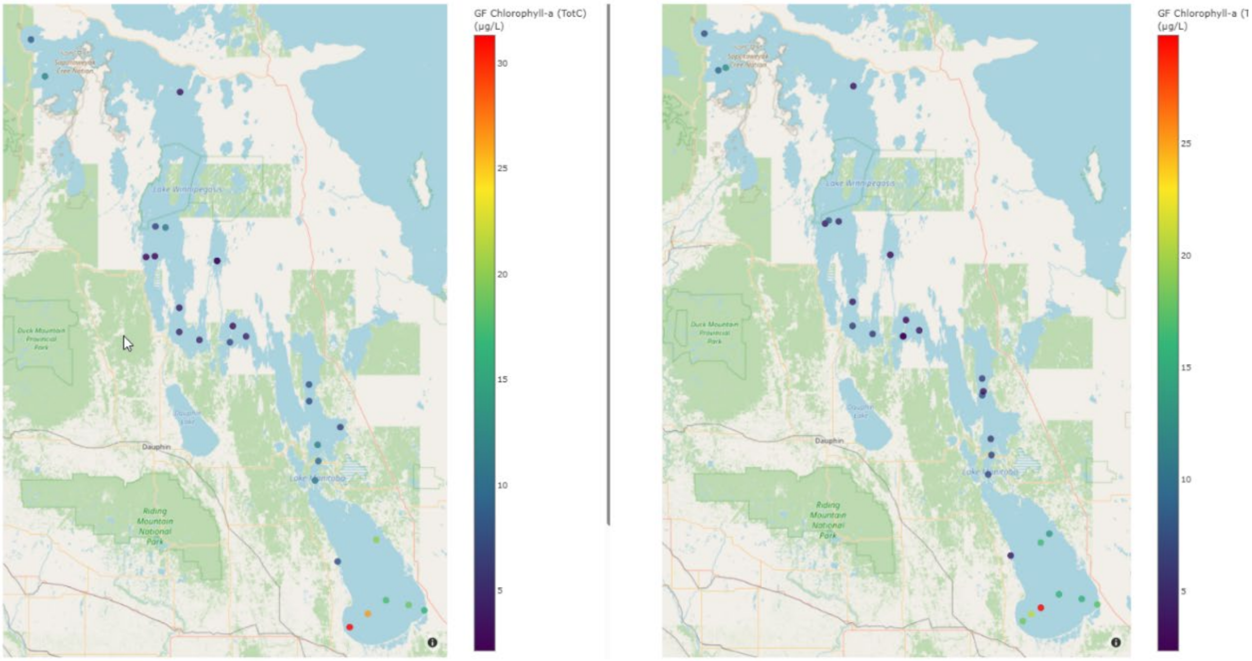

Figure D.10. Map of upperMBGL Total Chlorophyyll, 2016-2017

Appendix E — Provincial Chemistry Data

Table E.1. Provincial Chemistry stations and select median chemistry values, 2016 and 2017

Year	Station No.	Station Description	TN (µg/L) median	Total Organic Carbon (µg/L) median	Temperature (°C) median	Total Dissolved Solids (µg/L) median	pH median	COND (µS/cm @25°C) median	TP (µg/L) median	Total Carbon (µg/L) median
2016										
	MB05LJS006	MOSSY RIVER AT PR #364 NEAR WINNIPEGOSIS	955		6.70	343.0	8.270	520.0		54,850
	MB05LKS009	LAKE MANITOBA NARROWS AT PTH 68	1,005		4.20	613.0	8.340	999.5	16.8	56,850
	MB05LLS001	WHITEMUD RIVER AT PTH 16 AT WESTBOURNE	1,750	14,100	5.20	435.0	8.200	658.0		72,050
	MB05LLS013	NEAR DELTA FIELD STATION	1,260	12,100	6.50	561.0	8.460	918.0		53,700
	MB05LLS046	DELTA BEACH, LAKE MANITOBA			20.50					

Year	Station No.	Station Description	TN (µg/L) median	Total Organic Carbon (µg/L) median	Temperature (°C) median	Total Dissolved Solids (µg/L) median	pH median	COND (µS/cm @25°C) median	TP (µg/L) median	Total Carbon (µg/L) median
2017	MB05LMS003	DAUPHIN RIVER MIDWAY BETWEEN ANAMA BAY AND GYPSUMVILLE	1,110	13,200	6.10	552.0	8.150	937.0	12.7	55,400
	MB05MJS045	ASSINIBOINE RIVER AT RESERVOIR OF PORTAGE LA PRAIRIE W.T.P.	1,460	10,850	8.10	668.5	8.320	953.0		69,700
	MB05MJS047	ASSINIBOINE RIVER AT TCH, EAST OF PORTAGE LA PRAIRIE	1,630	10,850	9.90	665.0	8.290	945.0		71,000
	MB05MJS053	ASSINIBOINE RIVER AT PR #334, SOUTH OF HEADINGLEY	1,515	10,850	7.55	682.5	8.345	975.5		70,700
	MB05LJS006	MOSSY RIVER AT PR #364 NEAR WINNIPEGOSIS	890		12.10	324.0	8.260	474.0		51,100

Year	Station No.	Station Description	TN (µg/L) median	Total Organic Carbon (µg/L) median	Temperature (°C) median	Total Dissolved Solids (µg/L) median	pH median	COND (µS/cm @25°C) median	TP (µg/L) median	Total Carbon (µg/L) median
	MB05LKS009	LAKE MANITOBA NARROWS AT PTH 68	1,010		12.90	592.0	8.620	915.0		54,500
	MB05LLS001	WHITEMUD RIVER AT PTH 16 AT WESTBOURNE	1,345		9.80	497.0	8.285	676.0		62,700
	MB05LLS013	NEAR DELTA FIELD STATION	970		13.60	592.0	8.600	898.0	18.1	52,900
	MB05LMS003	DAUPHIN RIVER MIDWAY BETWEEN ANAMA BAY AND GYPSUMVILLE	1,040		8.75	551.5	8.320	905.0	14.7	52,200
	MB05MJS045	ASSINIBOINE RIVER AT RESERVOIR OF PORTAGE LA PRAIRIE W.T.P.	1,390		7.20	689.0	8.170	863.0		62,800
	MB05MJS047	ASSINIBOINE RIVER AT TCH, EAST OF PORTAGE LA PRAIRIE	1,420		15.20	716.0	8.510	919.0		67,200

Year	Station No.	Station Description	TN (µg/L) median	Total Organic Carbon (µg/L) median	Temperature (°C) median	Total Dissolved Solids (µg/L) median	pH median	COND (µS/cm @25°C) median	TP (µg/L) median	Total Carbon (µg/L) median
	MB05MJS053	ASSINIBOINE RIVER AT PR #334, SOUTH OF HEADINGLEY	1,585		8.05	710.0	8.150	830.5		61,950

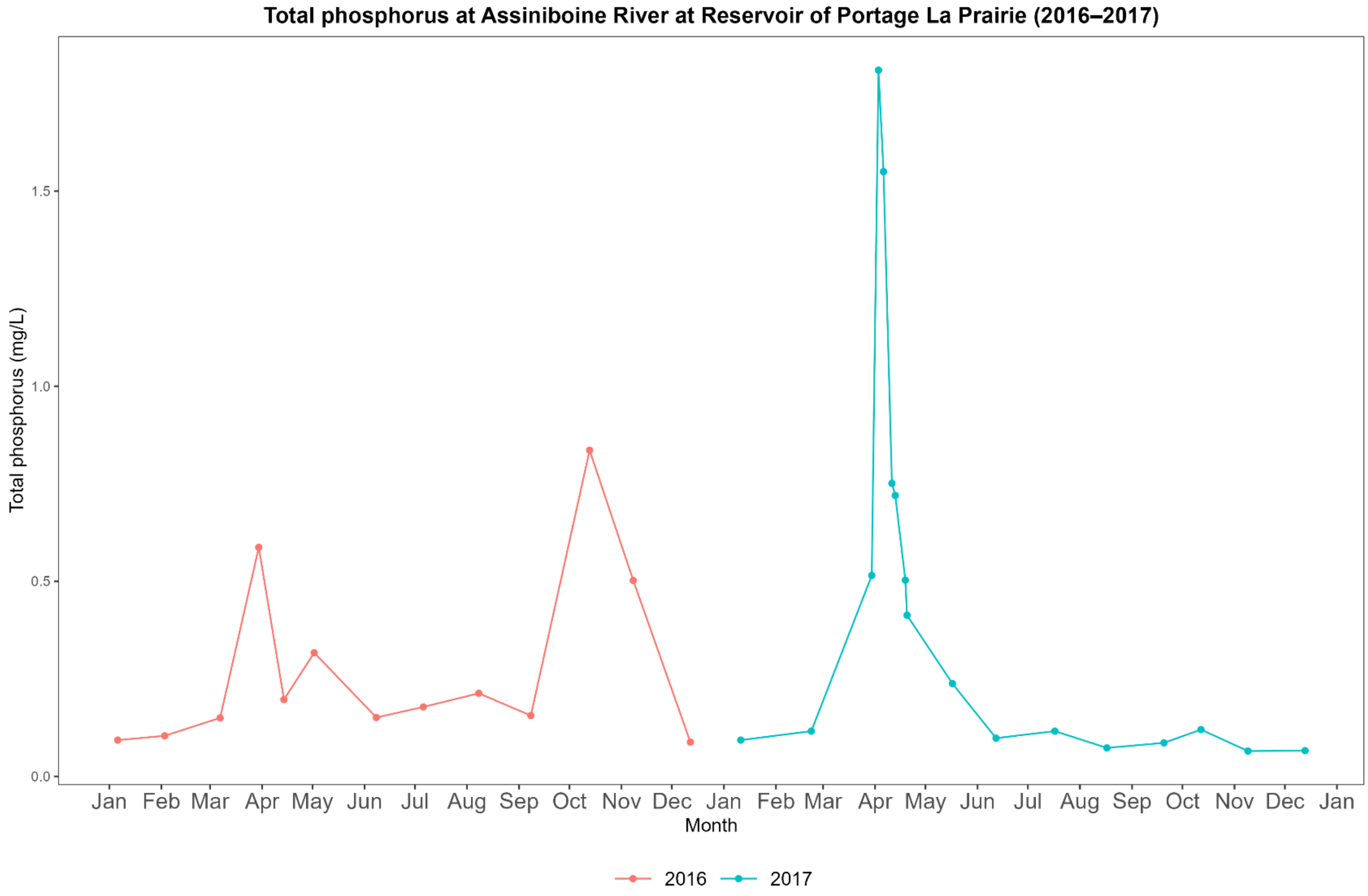

Figure E.1. Total phosphorus (mg/L) for provincial station MB05MJS045, Assiniboine River at Portage La Prairie reservoir for 2016 and 2017

Appendix F — Phytoplankton Distribution by Abundance and Biomass

Table F.1. Fairford River Sept. 2016, Algal Abundance and Biomass.

taxa_name	Group	Cells/L	mg/m³	Station	Date
Planktolyngbya limnetica, subtillus	Cyanobacteria	4,535,376.0	641.2	FR1	27-Sep-17
micro bluegreens	Cyanobacteria	381,926,400.0	345.6	FR1	27-Sep-17
Cyanodictyon reticulatum	Cyanobacteria	250,639,200.0	226.8	FR1	27-Sep-17
Planktolyngbya circumcreta	Cyanobacteria	2,387,040.0	176.4	FR1	27-Sep-17
Oocystis lacustris	Chlorophyta	1,491,900.0	156.8	FR1	27-Sep-17
Synedra cf acus v angustissima	Bacillariophyta	208,866.0	140.0	FR1	27-Sep-17
Aphanothece cf bachmanii(Anathece)	Cyanobacteria	76,385,280.0	120.9	FR1	27-Sep-17
Aphanocapsa incerta	Cyanobacteria	122,932,560.0	111.2	FR1	27-Sep-17
Pseudanabaena limnetica	Cyanobacteria	11,935,200.0	102.9	FR1	27-Sep-17
Botryococcus sp	Chlorophyta	29,838.0	42.7	FR1	27-Sep-17
bluegreen cells(free)	Cyanobacteria	5,370,840.0	31.5	FR1	27-Sep-17
Planktolyngbya capillaris, microspira	Cyanobacteria	865,302.0	31.3	FR1	27-Sep-17
Merismopedia tenuissima	Cyanobacteria	21,483,360.0	30.9	FR1	27-Sep-17
micro greens	Chlorophyta	28,644,480.0	30.2	FR1	27-Sep-17
Chroococcus sp	Cyanobacteria	7,161,120.0	30.0	FR1	27-Sep-17
Oocystis sp	Chlorophyta	596,760.0	26.9	FR1	27-Sep-17
Denticula sp	Bacillariophyta	89,514.0	26.5	FR1	27-Sep-17
Chroococcus minimus	Cyanobacteria	28,644,480.0	25.9	FR1	27-Sep-17
Fragilaria crotonensis	Bacillariophyta	28,000.0	25.3	FR1	27-Sep-17

Gymnodinium sp4	Pyrrophyta	59,676.0	24.2	FR1	27-Sep-17
Planktolyngbya contorta	Cyanobacteria	238,704.0	17.6	FR1	27-Sep-17
cyanodictyon filiformis	Cyanobacteria	14,322,240.0	15.1	FR1	27-Sep-17
Planktolyngbya bipunctata	Cyanobacteria	29,838.0	15.0	FR1	27-Sep-17
Didymocystis sp	Chlorophyta	1,193,520.0	14.0	FR1	27-Sep-17
Oocystis nephrocytium	Chlorophyta	89,514.0	13.7	FR1	27-Sep-17
Lemmermania tetrapedia	Chlorophyta	1,193,520.0	13.0	FR1	27-Sep-17
Small centrics (C. cf meneghiniana	Bacillariophyta	328,218.0	12.9	FR1	27-Sep-17
Unid green cells	Chlorophyta	656,436.0	11.3	FR1	27-Sep-17
Mougeotia spp	Chlorophyta	22,000.0	10.6	FR1	27-Sep-17
Botryococcus braunii	Chlorophyta	1,000.0	9.4	FR1	27-Sep-17
Navicula radiosa	Bacillariophyta	4,000.0	8.4	FR1	27-Sep-17
Dinobryon bavaricum	Chrysophyceae	89,514.0	6.9	FR1	27-Sep-17
Monoraphidium griffithii	Chlorophyta	119,352.0	6.0	FR1	27-Sep-17
Nitzschia acicularis	Bacillariophyta	89,514.0	6.0	FR1	27-Sep-17
Quadrigula sp	Chlorophyta	119,352.0	5.9	FR1	27-Sep-17
Staurodesmus sp	Chlorophyta	1,000.0	5.6	FR1	27-Sep-17
Tetraedron caudataum	Chlorophyta	59,676.0	5.2	FR1	27-Sep-17
Tetraedron minimum	Chlorophyta	89,514.0	5.2	FR1	27-Sep-17
Radiocystis geminata	Cyanobacteria	1,216,000.0	5.1	FR1	27-Sep-17
Gomphosphaeria aponina	Cyanobacteria	48,000.0	4.9	FR1	27-Sep-17
Scenedesmus cf ecornis	Chlorophyta	596,760.0	4.7	FR1	27-Sep-17
Urosolenia eriense	Bacillariophyta	119,352.0	4.6	FR1	27-Sep-17

Plagioselmis nanoplanktica/R.minuta	Cryptophyceae	89,514.0	4.2	FR1	27-Sep-17
Planktolyngbya undulata	Cyanobacteria	29,838.0	4.2	FR1	27-Sep-17
Cryptomonas sp	Cryptophyceae	29,838.0	4.1	FR1	27-Sep-17
Scenedesmus sp	Chlorophyta	179,028.0	4.1	FR1	27-Sep-17
Unid green cells	Chlorophyta	29,838.0	4.1	FR1	27-Sep-17
Chroococcus limneticus (Limnococcus)	Cyanobacteria	32,000.0	2.9	FR1	27-Sep-17
Eucapsa cf alpina	Cyanobacteria	136,000.0	2.9	FR1	27-Sep-17
Planktonema lauterborni	Chlorophyta	59,676.0	2.1	FR1	27-Sep-17
Katablepharis ovalis	Mixotrophs	89,514.0	2.0	FR1	27-Sep-17
Oocystis solitaria	Chlorophyta	2,000.0	1.8	FR1	27-Sep-17
Nephrochlamy	Chlorophyta	119,352.0	1.6	FR1	27-Sep-17
green algal cysts	Chlorophyta	2,000.0	1.5	FR1	27-Sep-17
Crucigenia quadrata	Chlorophyta	119,352.0	1.3	FR1	27-Sep-17
Cyclotella meninghiniana	Bacillariophyta	2,000.0	1.1	FR1	27-Sep-17
Cryptomonas erosa	Cryptophyceae	1,000.0	0.9	FR1	27-Sep-17
Aphanocapsa elachista	Cyanobacteria	128,000.0	0.8	FR1	27-Sep-17
Gymnodinium sp3	Pyrrophyta	3,000.0	0.8	FR1	27-Sep-17
Pseudopediastrum boryanum	Chlorophyta	32,000.0	0.8	FR1	27-Sep-17
Asterionella formosa	Bacillariophyta	6,000.0	0.7	FR1	27-Sep-17
Woronochinia cf compacta	Cyanobacteria	152,000.0	0.7	FR1	27-Sep-17
Coelospaerium kuetzingianum	Cyanobacteria	256,000.0	0.5	FR1	27-Sep-17
Staurastrum chaetoceros	Chlorophyta	2,000.0	0.5	FR1	27-Sep-17
Synedra nanana	Bacillariophyta	12,000.0	0.5	FR1	27-Sep-17

Chroococcus cf dispersus	Cyanobacteria	24,000.0	0.3	FR1	27-Sep-17
Planktosphaeria gelatinosa	Chlorophyta	16,000.0	0.3	FR1	27-Sep-17
Anabaena akinete	Cyanobacteria	2,000.0	0.2	FR1	27-Sep-17
bluegreen colony benthic	Cyanobacteria	3,000.0	0.2	FR1	27-Sep-17
Dolichospermum lemmermannii v minor	Cyanobacteria	8,000.0	0.2	FR1	27-Sep-17
Fragilaria sp	Bacillariophyta	2,000.0	0.2	FR1	27-Sep-17
Elakatothrix spirochroma	Chlorophyta	1,000.0	0.1	FR1	27-Sep-17
Staurastrum tetracerum	Chlorophyta	1,000.0	0.1	FR1	27-Sep-17
Anabaena heterocysts	Heterocyst	1,000.0	0.0	FR1	27-Sep-17

Table F.2. Lake Manitoba Station S3 August 2016, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m³	Station	Date
Planktolyngbya limnetica and undulata	Cyanobacteria	7,638,528.0	2,351.7	S3-5	12-Aug-17
Cuspidothris issatchenkoi	Cyanobacteria	1,670,928.0	1,543.3	S3-5	12-Aug-17
Aphanizomenon gracile	Cyanobacteria	1,432,224.0	1,358.1	S3-5	12-Aug-17
Aphanizomenon cf skujae	Cyanobacteria	477,408.0	225.0	S3-5	12-Aug-17
Chroococcus limneticus	Cyanobacteria	477,408.0	221.2	S3-5	12-Aug-17
Stephanodiscus agassiziensis	Bacillariophyta	59,676.0	205.9	S3-5	12-Aug-17
Planktolyngbya contorta	Cyanobacteria	1,909,632.0	164.6	S3-5	12-Aug-17
Planktothrix agardhii	Cyanobacteria	119,352.0	147.0	S3-5	12-Aug-17
Plagioselmis nanoplanktica/R.minuta	Cryptophyceae	4,177,320.0	125.4	S3-5	12-Aug-17
Coelospaerium kuetzingianum	Cyanobacteria	23,870,400.0	100.0	S3-5	12-Aug-17

Woronichinia klingae	Cyanobacteria	5,728,896.0	98.8	S3-5	12-Aug-17
Mallomonas pseudocoronata	Chrysophyceae	29,838.0	43.1	S3-5	12-Aug-17
Dictyosphaerium ehrenbergiana	Chlorophyta	477,408.0	32.3	S3-5	12-Aug-17
Crucigenia fenestrata	Chlorophyta	2,566,068.0	32.2	S3-5	12-Aug-17
Cryptomonas reflexa	Cryptophyceae	29,838.0	31.4	S3-5	12-Aug-17
Closterium kuetzingii	Chlorophyta	29,838.0	28.1	S3-5	12-Aug-17
Woronichinia elorantae	Cyanobacteria	3,819,264.0	25.6	S3-5	12-Aug-17
Dolichospermu flos aquae	Cyanobacteria	358,056.0	20.7	S3-5	12-Aug-17
Pediastrum tetras	Chlorophyta	477,408.0	18.4	S3-5	12-Aug-17
Anabaenopsis cf circularis	Cyanobacteria	716,112.0	16.5	S3-5	12-Aug-17
Aphanocapsa sp	Cyanobacteria	7,638,528.0	16.4	S3-5	12-Aug-17
Dolichospermum mendotae	Cyanobacteria	417,732.0	14.3	S3-5	12-Aug-17
Dolichospermu spiroideum	Cyanobacteria	119,352.0	12.5	S3-5	12-Aug-17
Microcystis sp	Cyanobacteria	656,436.0	11.3	S3-5	12-Aug-17
Dolichospermu circinalis	Cyanobacteria	835,464.0	11.0	S3-5	12-Aug-17
Nitzschia palaceae, palea	Bacillariophyta	119,352.0	10.2	S3-5	12-Aug-17
Synedra acus v. radians	Bacillariophyta	29,838.0	8.6	S3-5	12-Aug-17
Raphidocelis sp	Chlorophyta	1,909,632.0	8.2	S3-5	12-Aug-17
Lemmermaniella palida	Cyanobacteria	7,638,528.0	8.1	S3-5	12-Aug-17
Closterium acutum	Chlorophyta	29,838.0	7.2	S3-5	12-Aug-17
Planktonema lauterborni	Chlorophyta	59,676.0	7.2	S3-5	12-Aug-17
Elakatothrix spirochroma	Chlorophyta	59,676.0	1.7	S3-5	12-Aug-17
Elakatothrix genevensis	Chlorophyta	59,676.0	0.7	S3-5	12-Aug-17

Monoraphidium contortum	Chlorophyta	29,838.0	0.6	S3-5	12-Aug-17
Monoraphidium irregulare	Chlorophyta	29,838.0	0.2	S3-5	12-Aug-17
Anabaena heterocysts1	Heterocyst	149,190.0	0.0	S3-5	12-Aug-17
Anabaena heterocysts2	Heterocyst	29,838.0	0.0	S3-5	12-Aug-17
Anabaena heterocysts3	Heterocyst	29,838.0	0.0	S3-5	12-Aug-17
Aphanizomenon heterocysts1	Heterocyst	2,148,336.0	0.0	S3-5	12-Aug-17
Aphanizomenon heterocysts2	Heterocyst	1,670,928.0	0.0	S3-5	12-Aug-17
Aphanizomenon heterocysts3	Heterocyst	954,816.0	0.0	S3-5	12-Aug-17

Table F.3. Lake Manitoba Station S4 August 2016, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m^3	Station	Date
Planktolyngbya sp	Cyanobacteria	7,399,824.0	1,138.2	S4	12-Aug-17
Cuspidothris issatchenkoi	Cyanobacteria	1,909,632.0	707.9	S4	12-Aug-17
Aphanizomenon gracile	Cyanobacteria	716,112.0	330.7	S4	12-Aug-17
Microcystis novacekii	Cyanobacteria	15,277,056.0	262.1	S4	12-Aug-17
Planktothrix agardhii	Cyanobacteria	119,352.0	216.0	S4	12-Aug-17
micro bluegreens	Cyanobacteria	403,409,760.0	211.2	S4	12-Aug-17
Woronichinia klingae	Cyanobacteria	19,096,320.0	168.0	S4	12-Aug-17
Dolichospermum crassum	Cyanobacteria	537,084.0	166.7	S4	12-Aug-17
Oocystis solitaria	Chlorophyta	358,056.0	161.3	S4	12-Aug-17
Small greens	Chlorophyta	57,288,960.0	94.1	S4	12-Aug-17
Dolichospermu flos aquae	Cyanobacteria	954,816.0	55.3	S4	12-Aug-17

Aphanizomenon skujae	Cyanobacteria	238,704.0	48.0	S4	12-Aug-17
Dictyosphaerium pulchellum	Chlorophyta	477,408.0	43.9	S4	12-Aug-17
Collodictyon triciliatum	Mixotrophs	29,838.0	39.2	S4	12-Aug-17
Closterium aciculare	Chlorophyta	29,838.0	36.0	S4	12-Aug-17
Planktolyngbya contorta	Cyanobacteria	179,028.0	24.3	S4	12-Aug-17
Cyanodictyon tubaeforme	Cyanobacteria	10,502,976.0	22.5	S4	12-Aug-17
Radiocystis geminata	Cyanobacteria	3,819,264.0	22.4	S4	12-Aug-17
Dolichospermum cfcompactum	Cyanobacteria	895,140.0	15.4	S4	12-Aug-17
Aphanizomenon sp	Cyanobacteria	537,084.0	10.6	S4	12-Aug-17
Microcystis sp	Cyanobacteria	208,866.0	7.0	S4	12-Aug-17
Planktonema lauterborni	Chlorophyta	149,190.0	5.9	S4	12-Aug-17
Aphanocapsa elachista	Cyanobacteria	1,909,632.0	5.8	S4	12-Aug-17
Coenococcus planctonica(Eutetramorus planktonica)	Chlorophyta	477,408.0	5.5	S4	12-Aug-17
Coelastrum pseudmicroporum/asteroideum	Chlorophyta	238,704.0	4.1	S4	12-Aug-17
Plagioselmis nanoplanktica	Cryptophyceae	119,352.0	2.7	S4	12-Aug-17
Coelomoron pusillum	Cyanobacteria	954,816.0	2.0	S4	12-Aug-17
Snowella lacustris	Cyanobacteria	477,408.0	2.0	S4	12-Aug-17
Nitzschia acicularis	Bacillariophyta	29,838.0	1.9	S4	12-Aug-17
Pseudanabaena sp	Cyanobacteria	29,838.0	1.9	S4	12-Aug-17
Urosolenia eriense	Bacillariophyta	59,676.0	1.8	S4	12-Aug-17
Merismopedia warmingiana	Cyanobacteria	1,909,632.0	1.7	S4	12-Aug-17
Spirulina sp	Cyanobacteria	119,352.0	1.5	S4	12-Aug-17
Chrysochromulina parva	Haptophyta	149,190.0	1.3	S4	12-Aug-17

Katablepharis ovalis	Mixotrophs	59,676.0	1.2	S4	12-Aug-17
Monoraphidium contortum	Chlorophyta	59,676.0	1.1	S4	12-Aug-17
Oocystis sp	Chlorophyta	29,838.0	1.0	S4	12-Aug-17
Tetrastrum staurogeniaforme	Chlorophyta	119,352.0	0.9	S4	12-Aug-17
Monoraphidium minutum	Chlorophyta	89,514.0	0.8	S4	12-Aug-17
Dolichospermum heterocyst	Cyanobacteria	29,838.0	0.3	S4	12-Aug-17
Anabaena heterocysts	Heterocyst	59,676.0	0.0	S4	12-Aug-17
Anabaena heterocysts	Heterocyst	29,838.0	0.0	S4	12-Aug-17
Anabaena heterocysts	Heterocyst	29,838.0	0.0	S4	12-Aug-17
Aphanizomenon heterocysts	Heterocyst	2,148,336.0	0.0	S4	12-Aug-17
Aphanizomenon heterocysts	Heterocyst	716,112.0	0.0	S4	12-Aug-17
Aphanizomenon heterocysts	Heterocyst	238,704.0	0.0	S4	12-Aug-17

Table F.4. Lake Manitoba Station S6 August 2016, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m³	Station	Date
Aphanizomenon gracile	Cyanobacteria	1,193,520.0	926.0	S6	12-Aug-17
Cuspidothrix issatchenkoi	Cyanobacteria	775,788.0	836.0	S6	12-Aug-17
Planktolyngbya limnetica and some undulata	Cyanobacteria	2,983,800.0	711.9	S6	12-Aug-17
Kolkwitziella acuta (diplopsalis acuta marine type LWPG)	Pyrrophyta	29,838.0	601.6	S6	12-Aug-17
Aphanizomenon skuja	Cyanobacteria	775,788.0	214.5	S6	12-Aug-17
Planktothrix agardhii	Cyanobacteria	208,866.0	184.8	S6	12-Aug-17
Anabaenopsis cf circinalis	Cyanobacteria	5,072,460.0	156.2	S6	12-Aug-17

Small greens	Chlorophyta	48,218,208.0	147.2	S6	12-Aug-17
Woronichinia klingae	Cyanobacteria	15,277,056.0	134.4	S6	12-Aug-17
micro bluegreens	Cyanobacteria	455,924,640.0	122.2	S6	12-Aug-17
Crucigenia quadrata/ fenestrata	Chlorophyta	10,741,680.0	117.3	S6	12-Aug-17
Dolichospermum cf compacta	Cyanobacteria	6,355,494.0	109.0	S6	12-Aug-17
Oocystis sp	Chlorophyta	3,282,180.0	98.5	S6	12-Aug-17
Aphanocapsa holsatic	Cyanobacteria	91,662,336.0	82.9	S6	12-Aug-17
Microcystis sp	Cyanobacteria	4,774,080.0	81.9	S6	12-Aug-17
Anabaena / Komvophoron benthic	Cyanobacteria	59,676.0	64.0	S6	12-Aug-17
Rhodomonas minuta (Plagioselmis nanoplanktica)	Cryptophyceae	2,088,660.0	62.7	S6	12-Aug-17
Chlorella sp	Chlorophyta	26,257,440.0	56.3	S6	12-Aug-17
Franceia sp	Chlorophyta	29,838.0	55.1	S6	12-Aug-17
Cryptomonas reflexa	Cryptophyceae	59,676.0	52.3	S6	12-Aug-17
Chroococcus minutus	Cyanobacteria	537,084.0	39.4	S6	12-Aug-17
Anabaena sp (benthic species)	Cyanobacteria	1,790,280.0	38.4	S6	12-Aug-17
Dictyosphaerium tetrachototum	Chlorophyta	1,432,224.0	36.9	S6	12-Aug-17
Planktolyngbya contorta	Cyanobacteria	238,704.0	36.7	S6	12-Aug-17
Planktonema lauterborni	Chlorophyta	596,760.0	30.9	S6	12-Aug-17
Microcystis sp	Cyanobacteria	596,760.0	26.5	S6	12-Aug-17
Synedra sp.	Bacillariophyta	59,676.0	25.6	S6	12-Aug-17
Coenococcus planctonica (Eutetramorus planktonica)	Chlorophyta	1,909,632.0	21.9	S6	12-Aug-17
Katablepharis ovalis	Mixotrophs	596,760.0	13.7	S6	12-Aug-17
Merismopedia tenuissima	Cyanobacteria	14,322,240.0	13.0	S6	12-Aug-17

Staurastrum pingue	Chlorophyta	29,838.0	11.2	S6	12-Aug-17
Urosolenia eriense	Bacillariophyta	298,380.0	9.0	S6	12-Aug-17
Planktolyngbya microspira	Cyanobacteria	477,408.0	8.6	S6	12-Aug-17
Monoraphidium contortum	Chlorophyta	596,760.0	8.1	S6	12-Aug-17
Cryptomonas marssonii	Cryptophyceae	29,838.0	7.1	S6	12-Aug-17
Chrysochromulina parva	Haptophyta	596,760.0	6.9	S6	12-Aug-17
Lagerheimia citriforme	Chlorophyta	29,838.0	3.1	S6	12-Aug-17
Aphanocapsa sp	Cyanobacteria	2,864,448.0	2.6	S6	12-Aug-17
Aphanotheca sp	Cyanobacteria	954,816.0	1.5	S6	12-Aug-17
Anabaena heterocysts	Heterocyst	1,491,900.0	0.0	S6	12-Aug-17
Anabaena heterocysts	Heterocyst	179,028.0	0.0	S6	12-Aug-17
Anabaena heterocysts	Heterocyst	59,676.0	0.0	S6	12-Aug-17
Aphanizomenon heterocysts	Heterocyst	984,654.0	0.0	S6	12-Aug-17
Aphanizomenon heterocysts	Heterocyst	596,760.0	0.0	S6	12-Aug-17
Aphanizomenon heterocysts	Heterocyst	596,760.0	0.0	S6	12-Aug-17

Table F.5. Lake Manitoba Station S7 August 2016, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m³	Station	Date
Planktolyngbya limnetica	Cyanobacteria	9,249,780.0	1,423.9	S7	12-Aug-17
Aphanizomenon gracile	Cyanobacteria	775,788.0	597.1	S7	12-Aug-17
Peridinium polonicum	Pyrrophyta	29,838.0	426.6	S7	12-Aug-17
Planktothrix agardhii	Cyanobacteria	179,028.0	405.0	S7	12-Aug-17

Cuspidothris issatchenkoi	Cyanobacteria	1,193,520.0	240.0	S7	12-Aug-17
Woronichinia klingae	Cyanobacteria	7,638,528.0	172.8	S7	12-Aug-17
Coelospaerium kuetzingianum	Cyanobacteria	19,096,320.0	80.0	S7	12-Aug-17
Woronochinia compacta	Cyanobacteria	11,457,792.0	77.8	S7	12-Aug-17
Chroococcus turgidus	Cyanobacteria	59,676.0	49.4	S7	12-Aug-17
Glenodinium sp3	Pyrrophyta	29,838.0	49.0	S7	12-Aug-17
Aphanizomenon skujae	Cyanobacteria	59,676.0	38.0	S7	12-Aug-17
Chroococcus limneticus	Cyanobacteria	119,352.0	37.0	S7	12-Aug-17
Mallomonas sp	Chrysophyceae	59,676.0	35.3	S7	12-Aug-17
Dictyosphaerium pulchellum	Chlorophyta	358,056.0	32.9	S7	12-Aug-17
Planktonema lauterborni	Chlorophyta	477,408.0	32.3	S7	12-Aug-17
Plagioselmis nanoplanktica/R.minuta	Cryptophyceae	566,922.0	29.3	S7	12-Aug-17
Planktolyngbya contorta	Cyanobacteria	298,380.0	28.5	S7	12-Aug-17
Dolichospermu lemmermannii	Cyanobacteria	298,380.0	23.0	S7	12-Aug-17
Urosolenia eriense	Bacillariophyta	328,218.0	22.2	S7	12-Aug-17
Ochromonas sp	Chrysophyceae	149,190.0	20.5	S7	12-Aug-17
Oocystis sp	Chlorophyta	566,922.0	14.9	S7	12-Aug-17
Oocystis solitaria	Chlorophyta	29,838.0	13.4	S7	12-Aug-17
Chroococcus dispersus	Cyanobacteria	716,112.0	12.3	S7	12-Aug-17
Dinobryon stipitatum	Chrysophyceae	89,514.0	12.3	S7	12-Aug-17
Crucigenia fenestrata	Chlorophyta	716,112.0	12.0	S7	12-Aug-17
Crucigenia quadrata	Chlorophyta	1,402,386.0	11.7	S7	12-Aug-17
Coelomoron sp	Cyanobacteria	954,816.0	11.0	S7	12-Aug-17

Planktolyngbya sp (P. microspira)	Cyanobacteria	745,950.0	10.6	S7	12-Aug-17
Pediastrum kawraiskyi	Chlorophyta	208,866.0	8.4	S7	12-Aug-17
Gymnodinium sp4	Pyrrophyta	29,838.0	7.1	S7	12-Aug-17
Staurastrum tetracerum	Chlorophyta	29,838.0	6.9	S7	12-Aug-17
Planktolyngbya contorta	Cyanobacteria	29,838.0	5.5	S7	12-Aug-17
Monoraphidium contortum	Chlorophyta	208,866.0	5.1	S7	12-Aug-17
Dolichospermu circinalis	Cyanobacteria	358,056.0	4.7	S7	12-Aug-17
Chrysosporum minderi	Cyanobacteria	238,704.0	4.2	S7	12-Aug-17
Katablepharis ovalis	Mixotrophs	179,028.0	4.1	S7	12-Aug-17
Chrysochromulina parva	Haptophyta	358,056.0	3.1	S7	12-Aug-17
Elakatothrix genevensis	Chlorophyta	119,352.0	1.8	S7	12-Aug-17
Lagerheimia citriforme	Chlorophyta	29,838.0	1.5	S7	12-Aug-17
Koliella tatrae	Chlorophyta	179,028.0	1.1	S7	12-Aug-17
Romeria sp	Cyanobacteria	149,190.0	1.0	S7	12-Aug-17
Monoraphidium minutum	Chlorophyta	119,352.0	0.8	S7	12-Aug-17
Elakatothrix spirochroma	Chlorophyta	29,838.0	0.7	S7	12-Aug-17
Bicosoeca lacustris	Chrysophyceae	29,838.0	0.4	S7	12-Aug-17
Anabaena heterocysts1	Heterocyst	59,676.0	0.0	S7	12-Aug-17
Anabaena heterocysts2	Heterocyst	29,838.0	0.0	S7	12-Aug-17
Aphanizomenon heterocysts1	Heterocyst	984,654.0	0.0	S7	12-Aug-17
Aphanizomenon heterocysts2	Heterocyst	208,866.0	0.0	S7	12-Aug-17
Aphanizomenon heterocysts3	Heterocyst	119,352.0	0.0	S7	12-Aug-17

Table F.7. Lake Manitoba Pooled North Basin Stations Summer 2017, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m³	Station	Date
Micro bluegreens	Cyanobacteria	257,800,320.0	69.1	North Basin Pooled	15-Jul-17
Planktolyngbya limnetica	Cyanobacteria	238,704.0	50.4	North Basin Pooled	15-Jul-17
Gymnodinium sp2	Pyrrophyta	4,000.0	29.0	North Basin Pooled	15-Jul-17
Large chrysophytes (Ochromonads)	Chrysophyceae	89,514.0	27.8	North Basin Pooled	15-Jul-17
Stauridinium tetras	Chlorophyta	596,760.0	27.3	North Basin Pooled	15-Jul-17
Chrysamoeba sp	Chrysophyceae	29,838.0	21.9	North Basin Pooled	15-Jul-17
Small greens	Chlorophyta	1,253,196.0	21.5	North Basin Pooled	15-Jul-17
Glenodiniums sp 4	Pyrrophyta	29,838.0	16.1	North Basin Pooled	15-Jul-17
Parvodinium inconspicuum	Pyrrophyta	5,000.0	15.4	North Basin Pooled	15-Jul-17
Medium chrysophytes (Ochromonads)	Chrysophyceae	387,894.0	15.0	North Basin Pooled	15-Jul-17
Aphanocapsa holsatica	Cyanobacteria	15,277,056.0	13.8	North Basin Pooled	15-Jul-17
Anathece bachmanii	Cyanobacteria	8,593,344.0	13.6	North Basin Pooled	15-Jul-17
Unid green cells	Chlorophyta	89,514.0	12.3	North Basin Pooled	15-Jul-17
Oocystis sp.	Chlorophyta	1,044,330.0	12.2	North Basin Pooled	15-Jul-17
Tetraedron minimum	Chlorophyta	328,218.0	12.1	North Basin Pooled	15-Jul-17
Radiocystis geminata	Cyanobacteria	96,000.0	11.0	North Basin Pooled	15-Jul-17
Tiny chrysophytes	Chrysophyceae	835,464.0	9.6	North Basin Pooled	15-Jul-17
Micro-greens	Chlorophyta	8,056,260.0	8.5	North Basin Pooled	15-Jul-17
Gymnodinium sp4	Pyrrophyta	29,838.0	7.1	North Basin Pooled	15-Jul-17
Parvodinium pusillum	Pyrrophyta	8,000.0	7.1	North Basin Pooled	15-Jul-17

Aphanocapsa delicatissima	Cyanobacteria	7,638,528.0	6.9	North Basin Pooled	15-Jul-17
Tetraedron caudataum	Chlorophyta	29,838.0	5.9	North Basin Pooled	15-Jul-17
Botryococcus canadensis	Chlorophyta	4,000.0	5.7	North Basin Pooled	15-Jul-17
Planktolyngbya sp	Cyanobacteria	29,838.0	5.7	North Basin Pooled	15-Jul-17
Cyanodictyon reticulatum	Cyanobacteria	3,819,264.0	5.5	North Basin Pooled	15-Jul-17
Mougeotia spp	Chlorophyta	15,000.0	5.5	North Basin Pooled	15-Jul-17
Aphanocapsa elachista	Cyanobacteria	1,280,000.0	5.4	North Basin Pooled	15-Jul-17
Plagioselmis lacustris	Cryptophyceae	29,838.0	5.4	North Basin Pooled	15-Jul-17
Fragilaria sp	Bacillariophyta	29,838.0	5.0	North Basin Pooled	15-Jul-17
Pseudopediastrum boryanum	Chlorophyta	119,352.0	4.8	North Basin Pooled	15-Jul-17
Navicula radiosa	Bacillariophyta	2,000.0	4.6	North Basin Pooled	15-Jul-17
Oocystis nephrocytium	Chlorophyta	59,676.0	4.6	North Basin Pooled	15-Jul-17
Planktothrix aghardhii	Cyanobacteria	2,000.0	4.2	North Basin Pooled	15-Jul-17
Cryptomonas reflexa	Cryptophyceae	5,000.0	3.9	North Basin Pooled	15-Jul-17
Anathece sp	Cyanobacteria	2,387,040.0	3.8	North Basin Pooled	15-Jul-17
Pediastrum duplex	Chlorophyta	16,000.0	3.4	North Basin Pooled	15-Jul-17
Woronichinia sp	Cyanobacteria	256,000.0	3.4	North Basin Pooled	15-Jul-17
Crucigenia quadrata	Chlorophyta	298,380.0	3.3	North Basin Pooled	15-Jul-17
Plagioselmis nanoplanktica	Cryptophyceae	89,514.0	3.2	North Basin Pooled	15-Jul-17
Small blue-greens	Cyanobacteria	268,542.0	3.1	North Basin Pooled	15-Jul-17
Chroococcus minimus	Cyanobacteria	3,341,856.0	3.0	North Basin Pooled	15-Jul-17
Nitzschia sp	Bacillariophyta	89,514.0	3.0	North Basin Pooled	15-Jul-17
Cyclotella meninghiniana	Bacillariophyta	1,000.0	2.9	North Basin Pooled	15-Jul-17

Nephrochlamy	Chlorophyta	149,190.0	2.9	North Basin Pooled	15-Jul-17
Dolichospermum flos aquae	Cyanobacteria	42,000.0	2.8	North Basin Pooled	15-Jul-17
Scenedesmus cf spinosa	Chlorophyta	59,676.0	2.8	North Basin Pooled	15-Jul-17
Dinobryon bavaricum	Chrysophyceae	29,838.0	2.7	North Basin Pooled	15-Jul-17
Dinobryon stipitatum	Chrysophyceae	29,838.0	2.7	North Basin Pooled	15-Jul-17
cf Raphidiopsis	Cyanobacteria	29,838.0	2.6	North Basin Pooled	15-Jul-17
Small centrics	Bacillariophyta	59,676.0	2.4	North Basin Pooled	15-Jul-17
Oedogonium sp	Chlorophyta	7,000.0	2.3	North Basin Pooled	15-Jul-17
Achnanthidium sp	Bacillariophyta	29,838.0	2.0	North Basin Pooled	15-Jul-17
Bluegreen cells(free)	Cyanobacteria	29,838.0	1.7	North Basin Pooled	15-Jul-17
Staurastrum chaetoceros	Chlorophyta	2,000.0	1.6	North Basin Pooled	15-Jul-17
Didymocystis sp	Chlorophyta	59,676.0	1.4	North Basin Pooled	15-Jul-17
Nitzschia sp	Bacillariophyta	3,000.0	1.4	North Basin Pooled	15-Jul-17
Lemmermania tetrapedia	Chlorophyta	119,352.0	1.3	North Basin Pooled	15-Jul-17
Monoraphidium komarkova	Chlorophyta	29,838.0	1.3	North Basin Pooled	15-Jul-17
Scenedesmus cf ecornis	Chlorophyta	298,380.0	1.3	North Basin Pooled	15-Jul-17
Stichococcus sp	Chlorophyta	89,514.0	1.2	North Basin Pooled	15-Jul-17
Synedra sp	Bacillariophyta	4,000.0	1.2	North Basin Pooled	15-Jul-17
Cyanodictyon filiformis	Cyanobacteria	954,816.0	1.0	North Basin Pooled	15-Jul-17
Kirchneriella contortum	Chlorophyta	29,838.0	1.0	North Basin Pooled	15-Jul-17
Monoraphidium arcuatum	Chlorophyta	29,838.0	1.0	North Basin Pooled	15-Jul-17
Unid green cells	Chlorophyta	1,000.0	1.0	North Basin Pooled	15-Jul-17
Monoraphidium minutum	Chlorophyta	119,352.0	0.9	North Basin Pooled	15-Jul-17

Snowella cf atomus	Mixotrophs	954,816.0	0.9	North Basin Pooled	15-Jul-17
Chrysochromulina parva	Haptophyta	59,676.0	0.8	North Basin Pooled	15-Jul-17
Aphanocapse nubilum	Cyanobacteria	320,000.0	0.7	North Basin Pooled	15-Jul-17
Katablepharis ovalis	Mixotrophs	29,838.0	0.7	North Basin Pooled	15-Jul-17
Aphanizomenon cf aphanizomenoides	Cyanobacteria	1,000.0	0.5	North Basin Pooled	15-Jul-17
Chroococcus cf hyalina	Cyanobacteria	8,000.0	0.5	North Basin Pooled	15-Jul-17
Coelospaerium kuetzingianum	Cyanobacteria	240,000.0	0.5	North Basin Pooled	15-Jul-17
Cryptomonas marssonii	Cryptophyceae	2,000.0	0.5	North Basin Pooled	15-Jul-17
Synedra nanana	Bacillariophyta	10,000.0	0.5	North Basin Pooled	15-Jul-17
Monoraphidium contortum	Chlorophyta	29,838.0	0.4	North Basin Pooled	15-Jul-17
Coelastrum cf sphaericum	Chlorophyta	8,000.0	0.3	North Basin Pooled	15-Jul-17
Scenedesmus dimorphus	Chlorophyta	4,000.0	0.3	North Basin Pooled	15-Jul-17
Snowella littoralis	Cyanobacteria	64,000.0	0.3	North Basin Pooled	15-Jul-17
Cyanodictyon tubaeforme	Cyanobacteria	128,000.0	0.2	North Basin Pooled	15-Jul-17
Lemmermaniella palida	Cyanobacteria	179,028.0	0.2	North Basin Pooled	15-Jul-17
Pediastrum kawraiskyi	Chlorophyta	4,000.0	0.2	North Basin Pooled	15-Jul-17
Planktolyngbya contorta	Cyanobacteria	2,000.0	0.2	North Basin Pooled	15-Jul-17
Planktonema lauterborni	Chlorophyta	6,000.0	0.2	North Basin Pooled	15-Jul-17
Chroococcus sp	Cyanobacteria	12,000.0	0.1	North Basin Pooled	15-Jul-17
Coelosphaerium subarcticum	Cyanobacteria	64,000.0	0.1	North Basin Pooled	15-Jul-17

Table F.8. Lake Winnipegosis Station WO171986, August 2018, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m³	Station	Date
Planktolyngbya sp	Cyanobacteria	1,193,520.0	158.0	986	10-Aug-18
Planktolyngbya contorta	Cyanobacteria	1,193,520.0	147.0	986	10-Aug-18
Pseudanabaena sp	Cyanobacteria	238,704.0	86.4	986	10-Aug-18
Ochromonas sp	Chrysophyceae	149,190.0	80.6	986	10-Aug-18
Cyanodictyon planktonica	Cyanobacteria	75,430,464.0	79.6	986	10-Aug-18
Dictyosphaerium pulchellum	Chlorophyta	1,909,632.0	74.1	986	10-Aug-18
Large chrysophytes (Ochromonads)	Chrysophyceae	238,704.0	74.1	986	10-Aug-18
micro bluegreens	Cyanobacteria	210,059,520.0	56.3	986	10-Aug-18
Medium chrysophytes (Ochromonads)	Chrysophyceae	954,816.0	55.3	986	10-Aug-18
Oocystis lacustris	Chlorophyta	387,894.0	40.8	986	10-Aug-18
Tiny chrysophytes	Chrysophyceae	3,103,152.0	35.7	986	10-Aug-18
Cyanodictyon reticulatum	Cyanobacteria	35,328,192.0	32.0	986	10-Aug-18
Radiocystis geminata	Cyanobacteria	7,638,528.0	32.0	986	10-Aug-18
Aphanizomenon skujae	Cyanobacteria	89,514.0	31.9	986	10-Aug-18
Synechococcus sp	Cyanobacteria	12,412,608.0	25.3	986	10-Aug-18
Cyclotella bodanica	Bacillariophyta	29,838.0	25.2	986	10-Aug-18
Stichogloea doederlenii	Chrysophyceae	238,704.0	25.1	986	10-Aug-18
Koliella longiseta	Chlorophyta	716,112.0	24.3	986	10-Aug-18
micro greens	Chlorophyta	21,483,360.0	22.7	986	10-Aug-18
Chroococcus dispersus	Cyanobacteria	1,909,632.0	21.9	986	10-Aug-18
Limnothrix	Cyanobacteria	119,352.0	21.6	986	10-Aug-18

Euglena sp long pointed	Euglenophyta	29,838.0	19.8	986	10-Aug-18
Nephrochlamy	Chlorophyta	716,112.0	18.4	986	10-Aug-18
Chrysochromulina parva	Haptophyta	1,670,928.0	14.6	986	10-Aug-18
Cyanodictyon tubaeforme	Cyanobacteria	5,370,840.0	14.4	986	10-Aug-18
Merismopedia warmingiana	Cyanobacteria	15,277,056.0	13.8	986	10-Aug-18
Anathece sp	Cyanobacteria	3,819,264.0	8.6	986	10-Aug-18
Lagerheimia genevensis	Chlorophyta	238,704.0	8.5	986	10-Aug-18
Aphanocapsa sp	Cyanobacteria	8,354,640.0	7.6	986	10-Aug-18
Oocystis sp	Chlorophyta	238,704.0	7.2	986	10-Aug-18
Urosolenia eriense	Bacillariophyta	89,514.0	6.0	986	10-Aug-18
Tetraedron minimum	Chlorophyta	238,704.0	5.9	986	10-Aug-18
Stichococcus sp	Chlorophyta	238,704.0	5.5	986	10-Aug-18
Planktonema lauterborni	Chlorophyta	89,514.0	4.6	986	10-Aug-18
Colorless flagellate	Mixotrophs	238,704.0	4.1	986	10-Aug-18
Tetraedron minimum	Chlorophyta	89,514.0	3.3	986	10-Aug-18
Dinobryon elegantissimum	Chrysophyceae	29,838.0	3.0	986	10-Aug-18
cyanodictyon filiformis	Cyanobacteria	1,521,738.0	2.5	986	10-Aug-18
Scenedesmus cf ecornis	Chlorophyta	358,056.0	2.4	986	10-Aug-18
Bitrichia chodatii	Chrysophyceae	29,838.0	1.8	986	10-Aug-18
Katablepharis ovalis	Mixotrophs	59,676.0	1.4	986	10-Aug-18
Crucigenia tetrapedia	Chlorophyta	119,352.0	0.9	986	10-Aug-18
Scenedesmus sp	Chlorophyta	59,676.0	0.8	986	10-Aug-18
Romeria sp	Cyanobacteria	119,352.0	0.2	986	10-Aug-18

Spermatozopsis exaltans/similis	Chlorophyta	29,838.0	0.1	986	10-Aug-18
Aphanizomenon heterocysts	Heterocyst	89,514.0	0.0	986	10-Aug-18

Table F.9. Lake Winnipegosis Station WO308102, August 2016, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m³	Station	Date
Microcystis cf viridis	Cyanobacteria	4,833,756.0	214.3	WO308102	14-Aug-16
Ceratium furcoides	Pyrrophyta	5,000.0	209.4	WO308102	14-Aug-16
Cryptomonas reflexa	Cryptophyceae	179,028.0	172.5	WO308102	14-Aug-16
Ceratium hirundenella	Pyrrophyta	2,000.0	157.1	WO308102	14-Aug-16
Plagioselmis nannoplanctica	Cryptophyceae	1,193,520.0	125.4	WO308102	14-Aug-16
Planktothrix agardhii	Cyanobacteria	119,352.0	120.0	WO308102	14-Aug-16
Kolkwitziella acuta	Pyrrophyta	4,000.0	98.3	WO308102	14-Aug-16
Woronochinia cf compacta	Cyanobacteria	14,322,240.0	96.0	WO308102	14-Aug-16
Aphanotheca minutissiuma	Cyanobacteria	59,198,592.0	93.7	WO308102	14-Aug-16
Dolichospermum skujae-laxum	Cyanobacteria	149,190.0	82.7	WO308102	14-Aug-16
Lyngbya birgei	Cyanobacteria	1,000.0	70.4	WO308102	14-Aug-16
Planktolyngbya undulata	Cyanobacteria	11,248,926.0	58.9	WO308102	14-Aug-16
Limnothrix redekei	Cyanobacteria	59,676.0	58.8	WO308102	14-Aug-16
Peridinium polonicum	Pyrrophyta	2,000.0	55.9	WO308102	14-Aug-16
Dolichospermum spiroides/crassum	Cyanobacteria	130,000.0	46.1	WO308102	14-Aug-16
Mougeotia spp	Chlorophyta	89,514.0	36.0	WO308102	14-Aug-16
Planktolyngbya brevicellularis	Cyanobacteria	119,352.0	33.7	WO308102	14-Aug-16

Aphanocapsa sp.	Cyanobacteria	20,051,136.0	28.8	WO308102	14-Aug-16
Microcystis wesenbergi	Cyanobacteria	416,000.0	28.1	WO308102	14-Aug-16
Planktolyngbya limnetica	Cyanobacteria	59,676.0	23.0	WO308102	14-Aug-16
Unid green cells	Chlorophyta	298,380.0	17.3	WO308102	14-Aug-16
Cryptomonas marssonii	Cryptophyceae	59,676.0	14.3	WO308102	14-Aug-16
Woronochinia naegeliana	Cyanobacteria	576,000.0	13.2	WO308102	14-Aug-16
Ankyra juday	Chlorophyta	59,676.0	11.9	WO308102	14-Aug-16
Cyanodictyon reticulatum	Cyanobacteria	11,935,200.0	10.8	WO308102	14-Aug-16
Limnothrix obliqueacuminata	Cyanobacteria	59,676.0	10.8	WO308102	14-Aug-16
Planktonema lauterborni	Chlorophyta	179,028.0	10.6	WO308102	14-Aug-16
Aphanocapsa incerta	Cyanobacteria	20,000,000.0	10.5	WO308102	14-Aug-16
Coelastrum cambricum	Chlorophyta	32,000.0	9.9	WO308102	14-Aug-16
Plagioselmis lacustris	Cryptophyceae	59,676.0	9.6	WO308102	14-Aug-16
Crucigeniella tetrapedia	Chlorophyta	358,056.0	8.8	WO308102	14-Aug-16
Chroococcus sp	Cyanobacteria	149,190.0	8.6	WO308102	14-Aug-16
Crucigeniella quadrata	Chlorophyta	238,704.0	5.9	WO308102	14-Aug-16
Stephanodiscus agassiziensis	Bacillariophyta	2,000.0	5.4	WO308102	14-Aug-16
Oocystis lacustris	Chlorophyta	29,838.0	4.1	WO308102	14-Aug-16
Merismopedia cf warmingiana	Cyanobacteria	7,459,500.0	3.9	WO308102	14-Aug-16
Oocystis sp.	Chlorophyta	328,218.0	3.8	WO308102	14-Aug-16
Pediastrum duplex	Chlorophyta	16,000.0	3.4	WO308102	14-Aug-16
Scenedesmus cf spinosa	Chlorophyta	358,056.0	3.1	WO308102	14-Aug-16
Scenedesmus sp	Chlorophyta	119,352.0	2.7	WO308102	14-Aug-16

Ochromonas sp	Chrysophyceae	29,838.0	2.7	WO308102	14-Aug-16
Tiny chrysophyte flagellates	Chrysophyceae	387,894.0	2.3	WO308102	14-Aug-16
Closterium limneticum	Chlorophyta	1,000.0	2.1	WO308102	14-Aug-16
Chroococcus turgidus	Cyanobacteria	2,000.0	2.1	WO308102	14-Aug-16
Chroococcus dispersus/distens	Cyanobacteria	32,000.0	1.9	WO308102	14-Aug-16
Pediastrum boryanum	Chlorophyta	12,000.0	1.6	WO308102	14-Aug-16
Scenedesmus cf opoliensis	Chlorophyta	119,352.0	1.6	WO308102	14-Aug-16
Bluegreen cells	Cyanobacteria	358,056.0	1.5	WO308102	14-Aug-16
Dictyosphaerium primarium/simplex	Chlorophyta	954,816.0	1.4	WO308102	14-Aug-16
Monoraphidum cf irregulare	Chlorophyta	179,028.0	1.3	WO308102	14-Aug-16
Cyanocatena	Cyanobacteria	1,432,224.0	1.3	WO308102	14-Aug-16
Fragilaria sp.	Bacillariophyta	10,000.0	1.3	WO308102	14-Aug-16
Dolichospermum flos-aquae	Cyanobacteria	14,000.0	1.0	WO308102	14-Aug-16
Achnanthidium sp	Bacillariophyta	29,838.0	0.8	WO308102	14-Aug-16
Closterium acutum	Chlorophyta	2,000.0	0.6	WO308102	14-Aug-16
Radiocystis geminata	Cyanobacteria	64,000.0	0.3	WO308102	14-Aug-16
Anabaena heterocysts	Heterocyst	1,000.0	0.0	WO308102	14-Aug-16
Aphanizomenon heterocysts	Heterocyst	59,676.0	0.0	WO308102	14-Aug-16

Table F.10. Swan River May 2016, Algal Abundance and Biomass.

Taxon	Group	Cells/L	mg/m³	Station	Date
Tryboniella sp.	Bacillariophyta	29,838.0	39.2	GLSR1	19-May-16
Nitzschia sp.	Bacillariophyta	208,866.0	11.1	GLSR1	19-May-16
Large chrysophytes (Ochromonads)	Chrysophyceae	29,838.0	10.6	GLSR1	19-May-16
Asterionella formosa	Bacillariophyta	40,000.0	7.5	GLSR1	19-May-16
Synedra ulna v.danica	Bacillariophyta	6,000.0	4.4	GLSR1	19-May-16
Navicula sp.	Bacillariophyta	2,000.0	2.1	GLSR1	19-May-16
Dinobryon sp.	Chrysophyceae	119,352.0	1.6	GLSR1	19-May-16
Koliella sp	Chlorophyta	29,838.0	0.6	GLSR1	19-May-16
Achnanthidium sp.	Bacillariophyta	29,838.0	0.6	GLSR1	19-May-16
Monoraphidium minutum	Chlorophyta	29,838.0	0.3	GLSR1	19-May-16
Nitzschia sp.	Bacillariophyta	2,000.0	0.2	GLSR1	19-May-16
Nitzschia acicularis	Bacillariophyta	1,000.0	0.2	GLSR1	19-May-16
Nitzschia sp.	Bacillariophyta	1,000.0	0.2	GLSR1	19-May-16

Appendix G — Nutrient Deficiency Indicators

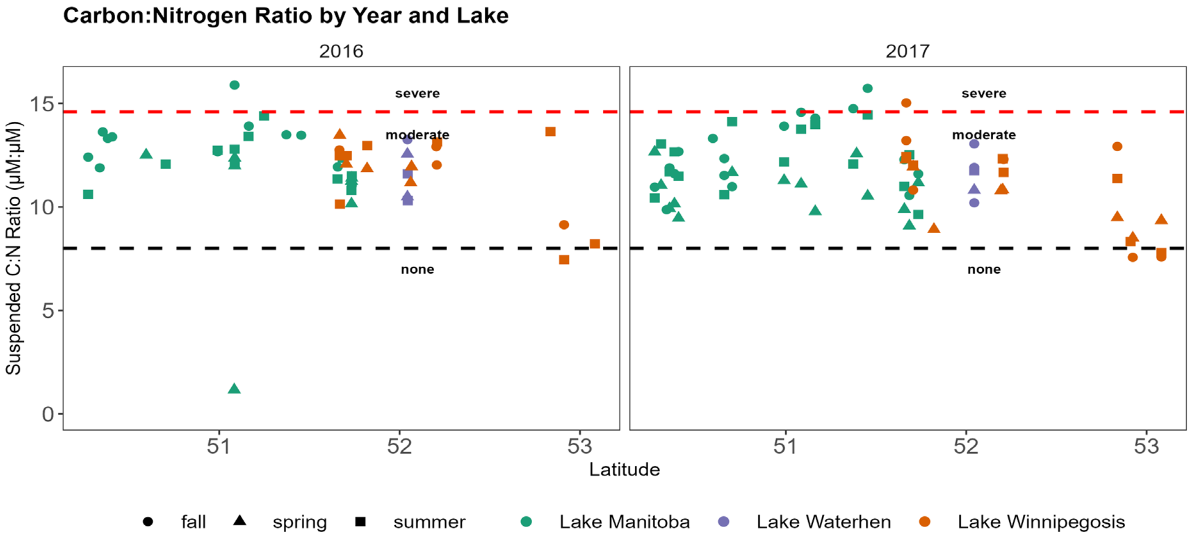

Figure G.1. Carbon to Nitrogen molar ratio for the uMBGL by year and season. Latitude is from south to north

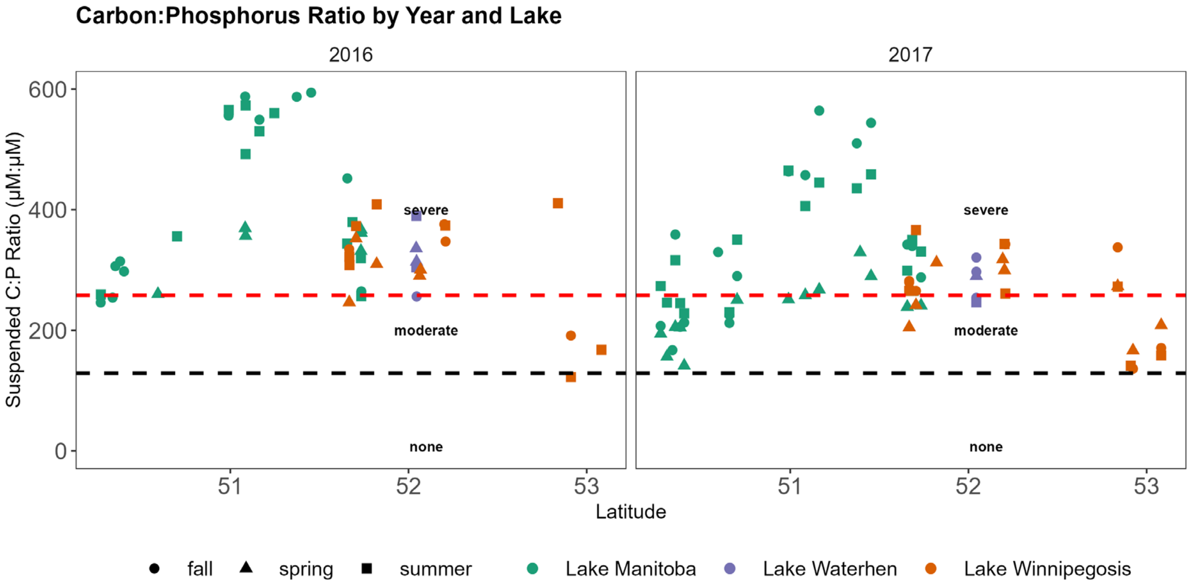

Figure G.2. Carbon to Phosphorus molar ratio for the uMBGL by year and season. Latitude is from south to north

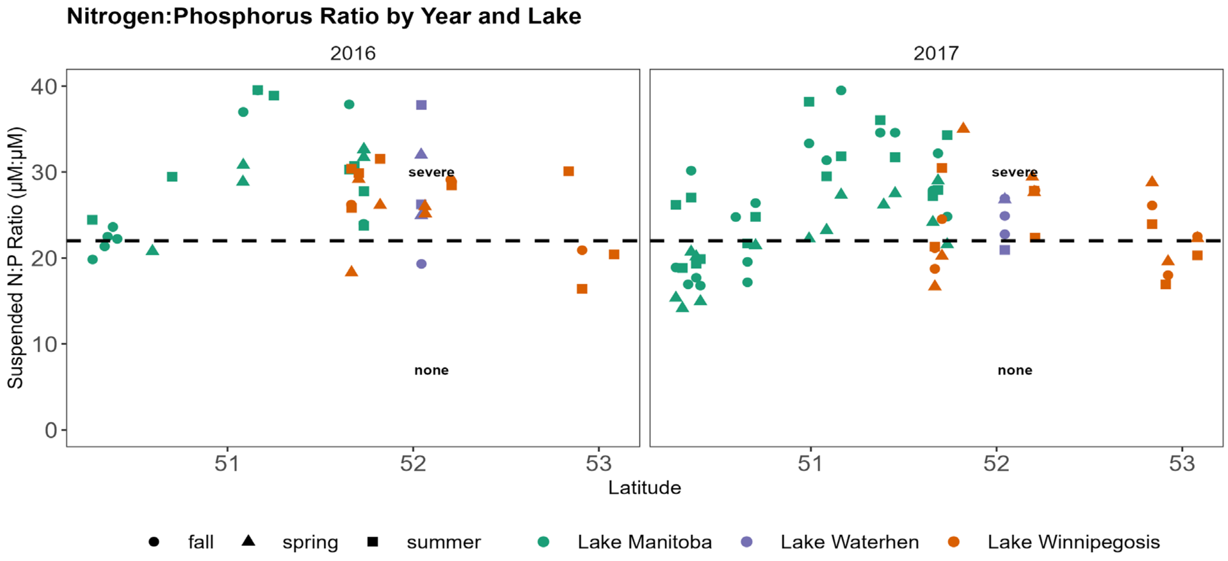

Figure G.3. Nitrogen to Phosphorus molar ratio for the uMBGL by year and season. Latitude is from south to north

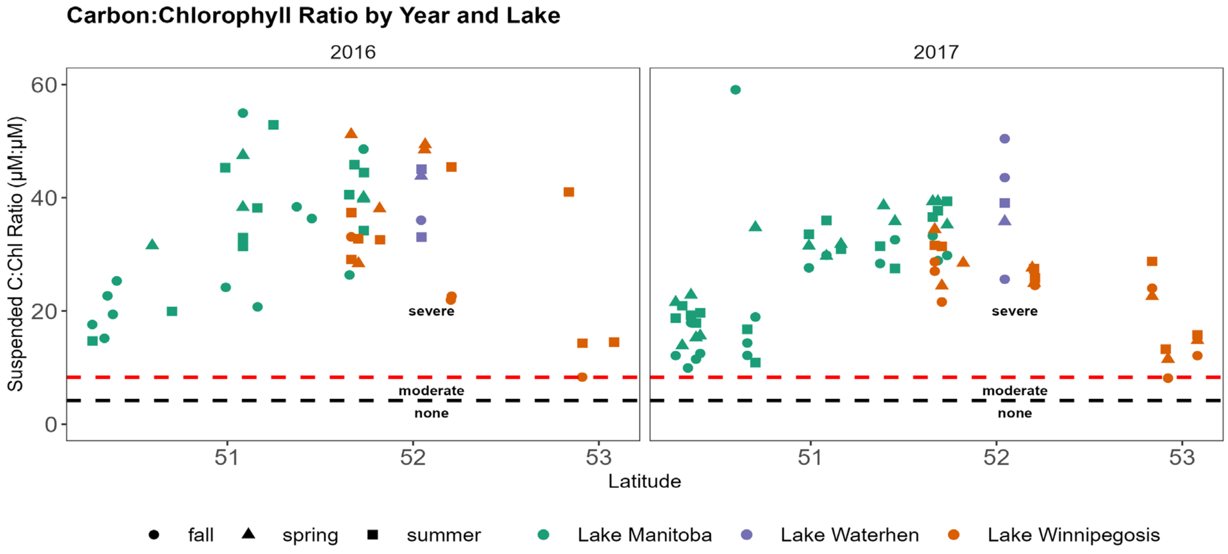

Figure G.4. Carbon to Chlorophyll molar ratio for the uMBGL by year and season. Latitude is from south to north