STATE OF LAKE WINNIPEG

2nd EDITION

Manitoba

Canada

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Photos throughout this report have been provided courtesy of many individuals who have worked on Lake Winnipeg and throughout its watershed over the past several years.

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Environment and Climate Change Canada

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EXECUTIVE SUMMARY

Lake Winnipeg is a valuable freshwater resource known for its fisheries, its plentiful beaches, and its importance to the traditional livelihood of many First Nations and Métis communities. The lake supports both commercial and angling fisheries that add significantly to Manitoba's economy through recreational spending and through commercial sales to domestic and international markets. The beaches are a strong economic driver for local communities attracting visitors, cottagers and permanent residents alike. Beaches in the south basin of Lake Winnipeg are easily accessible, only a short drive from Winnipeg and visitors to the beaches can exceed 30,000 per day, especially during the busy summer months when people head to the shore to enjoy the lake. The lake is also an important part of Manitoba's hydroelectric system. Its outflow is regulated allowing the production of electricity at northern generating stations.

Growing concerns over the increased presence of large and sustained algal blooms in the lake in the 1990s, especially following the flood of 1997, led to the development of a multilateral partnership with a focus on monitoring and understanding the state of the lake's aquatic ecosystem. In addition to commitments provided by the Government of Canada and the Manitoba government, academic institutions and non-government organizations have been pivotal in supporting the research and monitoring activities that have significantly improved our knowledge of the lake over the past twenty years. The inaugural State of Lake Winnipeg report, published in 2011, documented the physical, chemical and biological characteristics of Lake Winnipeg for the 1999–2007 period and represented the first comprehensive multi-agency assessment and reporting on the status of Lake Winnipeg. The 2011 report became the foundation for future lake assessments and presented key information that provided important context to support the development of ecologically relevant nutrient objectives and loading targets for Lake Winnipeg. With increased monitoring and improved knowledge, comes greater understanding of the threats to Lake Winnipeg. Climate change, invasive species, changes in land use, land and water management practices, and fishing pressures present challenges in managing the lake to achieve a healthy aquatic ecosystem.

Lake Winnipeg is a very large body of water extending 436 km from north to south and covering an area of 23,750 km^2. Air temperatures can vary considerably between its north and south basins and its east and west shores. Within the immediate region around the lake, mean annual air temperatures range from -0.7°C (north) to 1.6°C (south). Due in part to latitude and continentality, the climate of the Prairie Provinces along with the Lake Winnipeg region is warming much faster than the global average. Since 1970, the region surrounding Lake Winnipeg has been warming at twice the rate per decade for the globe. Although there is an expectation that these increasing air temperatures would affect lake temperatures, this has not yet been observed, likely because of high annual variability. However, over the last ten years, there have been more examples of vertical thermal stratification in the north basin of the lake. Although the cause of the increased frequency of observed thermoclines has not been explored, the effects of higher temperatures and other variables that influence the development of thermoclines must be better understood to determine whether we are witnessing inter-annual or decadal variability, or some long-term trend ultimately related to climate change.

Lake Winnipeg lies within a watershed of nearly 1,000,000 km^2. Precipitation varies across the basin and strongly influences the hydrology of the lake. An average of 498 mm of precipitation is recorded across the Lake Winnipeg region annually. Rainfall accounts for 76% of the average annual precipitation. Between the months of November and February, most of the observed precipitation falls as snow, with an average snowfall for the region of 132 cm. There is significant inter-annual variability in precipitation totals across the Lake Winnipeg basin, resulting in periods of drought and flooding. Over the last several decades, the hydrology of the lake and tributary rivers appears to be changing. The water residence time of the lake has varied over the last 50 years from 2.5 to almost eight years in length. From 1967 to 1998 the average residence time was 4.5 years, which decreased to 4.1 years during the 1999–2007 period, and further decreased to 3.1 years during the 2008–2016 period. The average streamflow of gauged streams flowing into Lake Winnipeg was almost 20% higher in the 2008–2016 period compared to the 1999–2007 period, and 44% higher in the 2008-2016 period compared to the historical average from 1977 to 1998. The greatest increases in flow between the 2008–2016 and 1999–2007 periods were in the Dauphin (125%) and Assiniboine (85%) rivers. The Winnipeg River, which accounted for the majority of flow into the lake (43% from 2008 to 2016), was the only source where no change was noted between the 2008–2016 and 1999–2007 periods. Similar to the inflows to Lake Winnipeg, the average outflow consistently increased over time and was 24% higher in the 2008–2016 period compared to the 1999–2007 period, and 53% higher in the 2008–2016 period than the 1977–1998 historical average. Climate models suggest that by the 2051–2080 period, under both low and high-carbon scenarios, winter, spring and fall seasons will all become wetter.

In general, waters of Lake Winnipeg are alkaline and well buffered. The north and south basins are unique from each other in physical characteristics, which influences the water chemistry of each basin. The south basin is shallower, smaller and generally warmer than the north basin. Although the south basin warms sooner, and remains warmer through to midsummer, it also cools faster than the north basin and the two basins often freeze within a few days of each other. Despite this, ice melt and breakup occur, on average, about two weeks earlier in the south basin, where surface water temperatures can reach 5^{o}C or higher before ice has cleared off the north basin. Differences in basin size and depth also influence the dissolved oxygen concentrations in each basin. In the 2008–2016 period, dissolved oxygen concentrations were generally higher in the north basin than in the south in spring and summer. In autumn, they were about the same in each basin, but in winter tended to be higher in the south. In both basins, average dissolved oxygen concentrations of both the euphotic and bottom waters were greater than 9 mg/L in spring, fall and winter and greater than 7 mg/L in summer. Similar to the 1999–2007 period, only 2.4% of over 900 observations from 2008–2016 were below 5.0 mg/L, which is Manitoba's water quality objective for the protection of aquatic life for dissolved oxygen at temperatures above 5^{o}C. At 5^{o}C or below, the objective falls to 3 mg/L. Because temperature was not considered in the above statistic, it is likely that the number of samples that did not meet the objective is even lower than 2.4%.

Differences in inflows to each basin also influence the water quality of each basin. Water chemistry in the south basin and narrows is controlled in part by the influence of the Red River and Winnipeg River chemistries while the Saskatchewan and Dauphin rivers influence water chemistry in the north basin. Median concentrations of potassium, total suspended solids, turbidity, total organic carbon, and the majority of trace elements were significantly higher in the south basin and narrows compared to the north basin. In contrast, conductivity, alkalinity, sodium, chloride, total dissolved solids, and total inorganic carbon were higher in the north basin compared to the south basin and narrows. Median lake pH and concentrations of calcium, magnesium, and hardness were similar between basins.

The total load of nutrients entering the lake varies annually, generally in response to changes in streamflow. From 1994 to 2016 the phosphorus loads varied between just over 3,000 tonnes per year to just under 11,000 tonnes per year, and in the last five years have been near the average (7,368 tonnes/year) or below. The Red River is the single largest source of phosphorus, accounting for 69% of the average total load. Since the 1990s, flows have increased in the Red River resulting in greater nutrient loads being transported to the lake compared to the historic record. Phosphorus concentrations in the south basin, which averaged 0.104 mg/L from 1999 to 2016, about double the phosphorus objective (0.05 mg/L), continue to be high in comparison to estimated historical concentrations. In the north basin, phosphorus concentrations averaged 0.039 mg/L from 1999 to 2016 and were below the long-term average in the last five years of this period of record. In both basins, from 1999 to 2016, phosphorus concentrations tended to be higher in the bottom waters than near the surface of the lake, although the difference between the surface and the bottom was greatest in the north basin. The internal phosphorus loading to the lake from the combined effect of diffusion of dissolved phosphorus from sediments and wind-induced resuspension of particulate phosphorus in lake sediments is estimated to be equal to or higher than phosphorus entering the lake from streams. Further knowledge on the influence of this internal phosphorus loading is needed to improve the understanding of the phosphorus cycle and phosphorus balance of the lake.

From 1994 to 2016, nitrogen loads to the lake varied between 52,470 tonnes/year to 136,676 tonnes/year. The loading to the lake has been below the average (91,263 tonnes/year) in three of the last five years, mainly attributable to lower loads than average from the Red and Winnipeg rivers. The Red and Winnipeg rivers are the largest sources of nitrogen to the lake, contributing on average 34% and 22% of the load, respectively. Nitrogen concentrations are typically highest at the very south end of the lake near the inflow of the Red River. Concentrations in the south basin tend to peak in the summer and decline in the fall, whereas in the north basin, nitrogen concentrations remain similar in summer and fall. Nitrogen concentrations from 2012 to 2016 appear to be lower than earlier in the period of record. The average total nitrogen concentration of the south basin (0.85 mg/L) from 1999 to 2016 is slightly above the nitrogen objective (0.75 mg/L), while the average total nitrogen concentration of the north basin (0.63 mg/L) is below the nitrogen objective. Nitrogen fixation and denitrification rates are not well studied and more research in this area would enable an improved understanding of the nitrogen cycle and nitrogen balance of the lake.

Although the primary focus of the water chemistry programs has been on tracking nutrient concentrations and general lake chemistry, the levels of other contaminants and trace elements in the lake have also been explored. In-use contaminants (e.g. pesticides, pharmaceuticals, brominated flame retardants) and legacy organochlorine persistent organic pollutants do not seem to pose an acute or chronic risk to the Lake Winnipeg ecosystem. Pesticide detections are low in Lake Winnipeg and no samples exceeded Manitoba's guidelines for protection of aquatic life. However, for some of the commonly used pesticides that are prevalent in the basin (e.g. atrazine, 2,4-D, dicamba, glyphosate, and MCPA), research on long-term exposure scenarios would fill an information gap. Concentrations of microplastics appear elevated in Lake Winnipeg relative to Lake Huron and Superior, but are similar to concentrations found in Lake Erie. Unlike the Laurentian Great Lakes, where fragments and pellets were the dominant type of microplastics, the majority of particles identified in Lake Winnipeg are fibers. The impacts of microplastics to the aquatic ecosystem of Lake Winnipeg are currently unknown and require more research. Concentrations of trace elements in Lake Winnipeg are low and many are often below detection. Exceedances of guidelines for the protection of aquatic life occur rarely for most metals other than aluminum and iron, which have naturally high concentrations in most Manitoba freshwaters. Between 1999 and 2016, the concentration of most trace elements in Lake Winnipeg were similar to typical concentrations measured for other Manitoba freshwaters.

Lake Winnipeg supports a diverse biological community. While phytoplankton are part of the lake's natural freshwater ecosystem, excess amounts, particularly large amounts of cyanobacteria, are a concern for their potential impacts to recreation, fisheries and drinking water. Over the last two decades, phytoplankton species composition and biomass varied considerably, likely responding to major flooding events and inter-annual changes in wind, nutrients, light and temperature. In the last five years, however, the total phytoplankton biomass has been stable and similar to the 1999–2016 average. Although there is significant year-to-year variation in the relative proportions of the different phytoplankton classes, on average, cyanobacteria has accounted for about half of the total phytoplankton biomass in Lake Winnipeg during the open-water period. Cyanobacteria biomass was greatest in 2011 and comprised almost 83% of the total phytoplankton biomass. The spatial extent, intensity and duration of algal blooms also show significant year-to-year variation. Higher phytoplankton biomass is typically observed in the north basin, where light and nutrients are optimal for growth. From 2003-2018, blooms lasted on average from about 40 days to 80 days over a season although each year, some locations had blooms that lasted up to 150 days. The cyanobacterial toxin microcystin is not frequently detected in the offshore areas of Lake Winnipeg but is found more frequently in the nearshore areas of the south basin. However, concentrations remain quite low and are below the recreational objective in the majority of samples.

Other components of the aquatic food web are under pressure or have undergone changes that are more pronounced over the recent years. Six aquatic species in Lake Winnipeg (three fish, one mollusc, one reptile and one amphibian) are either considered at risk or are of special concern, including: Lake Sturgeon, Chestnut Lamprey, Bigmouth Buffalo, Mapleleaf Mussel, Snapping Turtle and Northern Leopard Frog. Within the last decade, two new invasive species have established a presence in the lake that will likely impact invertebrate and fish populations. Spiny Water Flea, native to large areas of Eurasia, were first found in Lake Winnipeg in 2011 in the stomachs of fish captured in the south basin of the lake. By the fall of 2012, the species had colonized the north basin. In the fall of 2013 Zebra Mussels, which are native to Europe, were detected in several harbours in the south basin of the lake. The mussels are now well established in the south basin of the lake and have spread to the north basin of the lake. The changes that may occur to the lake's ecosystem due to the presence of both these invaders is unclear because of the complexity of various physical and biological interactions.

The Lake Winnipeg fishery has also undergone recent changes. The status of the historic Walleye fishery is in decline as it has been threatened by the collapse of the non-native Rainbow Smelt population in the north basin of the lake, and from several years of high mortality rates attributed to intensive fishing pressure. The status of the Sauger population is also of concern. The population is in a multi-decadal decline with fishery yields decreasing over 90% in the past several decades.

Analysis of fish tissue indicates that the mercury content of Lake Winnipeg fish is among the lowest for water bodies monitored in Manitoba and is below the Health Canada guideline for retail fish. Water quality monitoring of the indicator bacteria *E. coli* at Lake Winnipeg beaches from 2004 to 2018 showed that in the majority of cases most beaches were within recreational water quality objectives and showed no major changes over that time. Studies to determine the sources of *E. coli* at Lake Winnipeg beaches indicated that much of the indicator *E. coli* bacteria came from animal sources, mostly shorebirds and geese.

Lake Winnipeg is a complex system. Over the past two decades our knowledge of its function and structure has improved, but many knowledge gaps still exist. Some of the gaps identified in the first State of Lake Winnipeg Report have been addressed in this update (*e.g.* fish populations, temperature and oxygen profiles, spatial variability in chemical constituents), while others are currently under study (*e.g.* littoral zones, internal nutrient loading/sediment resuspension). A significant gap remains in efforts to include indigenous knowledge along with western science to inform better our understanding of the state of the lake. Furthermore, new questions have emerged regarding the impacts of invasive species, climate change and uncertainty around the effects of microplastics on different biota.

The management of the lake remains a challenging task. Improvement will take time given the large watershed, the many small and diffuse sources of nutrients, and internal loading from legacy nutrients in the lake sediments. In the meantime, information from this report will be used to develop indicators of lake health that can be updated and reported on regularly. Efforts to improve the health of Lake Winnipeg will continue throughout the Lake Winnipeg basin including: wastewater treatment facility upgrades, implementing beneficial management practices to reduce nutrient loading, and actions to limit the spread of aquatic invasive species. With multilateral partnerships established and a keen stakeholder interest, a key next step in achieving the goal of a healthy aquatic ecosystem in Lake Winnipeg might include consideration of an adaptive management approach. Adaptive management uses the best available data and information for lake management and allows adjustments to be made as additional knowledge in areas such as climate change, invasive species and in-lake processes become known.

SOMMAIRE

Le lac Winnipeg est une précieuse source d'eau douce; il est reconnu pour ses pêcheries, ses nombreuses plages et le rôle important qu'il joue dans le mode de subsistance traditionnel de nombreuses communautés des Premières Nations et métisses. Le lac soutient les secteurs de la pêche commerciale et sportive et enrichit considérablement l'économie du Manitoba en stimulant les dépenses de loisirs et les ventes commerciales aux marchés national et international. Les plages constituent un important moteur économique pour les collectivités locales et attirent tant les visiteurs que les villégiateurs et les résidents permanents. À courte distance en voiture de Winnipeg, les plages du bassin sud du lac Winnipeg sont faciles d'accès et peuvent accueillir plus de 30 000 visiteurs par jour, en particulier durant la haute saison estivale, alors que les gens se rendent sur les rives pour profiter du lac. Le lac est aussi un élément important du réseau hydroélectrique du Manitoba. Son débit sortant est régulé afin de permettre la production d'électricité dans les centrales du nord.

Les préoccupations croissantes face à la présence accrue d'efflorescences algales étendues et de longue durée au cours des années 1990, en particulier après la crue de 1997, ont donné naissance à un partenariat multilatéral ayant pour but de surveiller et de comprendre l'état de l'écosystème aquatique du lac. Outre les engagements du gouvernement du Canada et du gouvernement du Manitoba, des établissements universitaires et des organisations non gouvernementales ont joué un rôle crucial en soutenant des activités de recherche et de surveillance qui nous ont permis d'améliorer considérablement notre connaissance du lac au cours des vingt dernières années. Le premier rapport sur l'état du lac, publié en 2011, décrivait les caractéristiques physiques, chimiques et biologiques du lac entre 1999 et 2007 et constituait le premier rapport d'évaluation multi-organismes complet de l'état du lac Winnipeg. Le rapport de 2011 a jeté les bases des évaluations ultérieures et contenait des renseignements clés qui ont servi de fondement à l'établissement d'objectifs écologiquement pertinents en matière d'éléments nutritifs et de charges cibles pour le lac Winnipeg. Une meilleure surveillance et des connaissances plus approfondies ont permis de mieux comprendre les menaces qui pèsent sur le lac Winnipeg. Les changements climatiques, les espèces envahissantes, la réaffectation des sols, les changements dans la gestion des terres et des eaux et les pressions exercées par la pêche présentent des défis dont il faut tenir compte dans la gestion du lac pour assurer la santé de l'écosystème aquatique.

Le lac Winnipeg est un très grand plan d'eau qui s'étend sur 436 km du nord au sud et occupe une superficie de 23 750 km^2. Les températures de l'air peuvent varier considérablement entre les bassins nord et sud et les rives est et ouest. Aux environs immédiats du lac, les températures annuelles moyennes de l'air varient entre -0,7 °C (au nord) et 1,6 °C (au sud). En raison notamment de sa latitude et de sa continentalité, le climat des provinces des Prairies et de la région du lac Winnipeg se réchauffe beaucoup plus rapidement que le taux moyen mondial. Depuis 1970, la région qui entoure le lac Winnipeg s'est réchauffée deux fois plus rapidement que le taux décennal planétaire. Bien que cette hausse des températures de l'air soit censée influer sur les températures du lac, ce phénomène n'a pas encore été observé, probablement en raison de la forte variabilité annuelle des températures. Cependant, au cours des dix dernières années, un plus grand nombre de cas de stratification thermique verticale ont été recensés dans le bassin nord du lac. Bien que la cause de ces thermoclines plus fréquentes n'ait pas été explorée, il nous faudrait mieux comprendre les effets des températures plus élevées et d'autres variables qui influent sur le développement de thermoclines afin de déterminer si sous assistons à une variabilité interannuelle ou décennale ou à une tendance à long terme qui, ultimement, serait liée aux changements climatiques.

Le lac Winnipeg est situé dans un bassin hydrologique de près de 1 000 000 km^2. Les précipitations varient dans l'ensemble du bassin et exercent une forte influence sur l'hydrologie du lac. En moyenne, 498 mm de précipitations sont enregistrés annuellement dans l'ensemble de la région du lac Winnipeg. La pluie représente 76 % des précipitations annuelles moyennes. Entre novembre et février, la plus grande partie des précipitations tombent sous forme de neige, l'enneigement moyen pour la région étant de 132 cm. Les précipitations totales sont marquées par une importante variabilité interannuelle dans l'ensemble du bassin versant du lac Winnipeg, ce qui donne lieu à des périodes de sécheresse et d'inondation. Au cours des dernières décennies, l'hydrologie du lac et de ses affluents semble avoir évolué. Le temps de séjour de l'eau du lac est passé, au cours des 50 dernières années, de 2,5 à près de 8 ans. Entre 1967 et 1998, le temps de séjour moyen était de 4,5 ans; il est tombé à 4,1 ans entre 1999 et 2007, puis à 3,1 ans pendant la période d'observation la plus récente, soit de 2008 à 2016. Le débit moyen des cours d'eau jaugés qui se jettent dans le lac Winnipeg était presque 20 % plus élevé pendant la période de 2008 à 2016 qu'entre 1999 et 2007, et 44 % plus élevé durant la période de 2008 à 2016 que la moyenne historique de 1977 à 1998. Les plus fortes hausses de débit entre les périodes 2008-2016 et 1999-2007 ont été mesurées dans les rivières Dauphin (125 %) et Assiniboine (85 %). La rivière Winnipeg, qui représente le principal apport d'eau au lac (43 % de 2008 à 2016), était la seule source pour laquelle aucun changement n'avait été observé entre les périodes 2008-2016 et 1999-2007. Comme dans le cas des débits entrants, le débit sortant moyen du lac Winnipeg a constamment augmenté avec le temps et était 24 % plus élevé pendant la période 2008-2016 que durant la période 1999-2007, et 53 % plus élevé pendant la période 2008-2016 que la moyenne historique de 1977-1998. Les modèles climatiques indiquent qu'au cours de la période 2051-2080, dans un scénario de faibles émissions ou d'émissions élevées de carbone, les hivers, les printemps et les automnes deviendront plus humides.

Globalement, les eaux du lac Winnipeg sont alcalines et tamponnées. Les bassins nord et sud présentent des caractéristiques physiques distinctes qui influent sur la chimie de l'eau de chaque bassin. Le bassin sud est moins profond, plus petit et généralement plus chaud que le bassin nord. Bien que le bassin sud se réchauffe plus tôt et demeure plus chaud jusqu'au milieu de l'été, il se refroidit aussi plus rapidement que le bassin nord, et les deux bassins gèlent souvent à quelques jours d'intervalle. Cependant, la fonte de la glace et la débâcle surviennent en moyenne environ deux semaines plus tôt dans le bassin sud, où la température des eaux de surface peut atteindre 5 °C ou plus avant que la glace ait disparu du bassin nord. Les différences de taille et de profondeur entre les deux bassins ont aussi un effet sur les concentrations d'oxygène dissous dans chaque bassin. Pendant la période 2008-2016, les concentrations d'oxygène dissous étaient généralement plus élevées dans le bassin nord que dans le bassin sud au printemps et en été. À l'automne, elles étaient à peu près les mêmes dans les deux bassins, mais pendant l'hiver, elles étaient généralement plus élevées dans le bassin sud. Dans les deux bassins, les concentrations moyennes d'oxygène dissous dans les eaux de la zone euphotique et les eaux de fond étaient supérieures à 9 mg/L au printemps, en automne et en hiver et dépassaient les 7 mg/L en été. Comme pour la période de 1999 à 2007, à peine 2,4 % des quelque 900 observations réalisées de 2008 à 2016 étaient inférieures à 5,0 mg/L, soit l'objectif de qualité de l'eau du Manitoba pour la protection de la vie aquatique établi pour l'oxygène dissous à des températures supérieures à 5 °C. À des températures de 5 °C ou moins, l'objectif est de 3 mg/L; la température n'étant pas prise en compte dans cette statistique, le nombre d'échantillons n'atteignant pas l'objectif pourrait être inférieur à 2,4 %.

Les différences dans les débits entrants influent aussi sur la qualité de l'eau de chaque bassin. La chimie de l'eau du bassin sud et du passage est régie en partie par la rivière Rouge et de la rivière Winnipeg tandis que les rivières Saskatchewan et Dauphin influent sur la chimie de l'eau du bassin nord. Les mesures médianes de potassium, de matières en suspension totales, de turbidité, du carbone organique total et de la plupart des oligoéléments étaient sensiblement plus élevées dans le bassin sud et le passage que dans le bassin nord. En revanche, la conductivité, l'alcalinité, le sodium, le chlorure, la quantité totale de matières dissoutes et le carbone inorganique total étaient plus élevés dans le bassin nord que dans le bassin sud et le passage. Le pH médian de l'eau, les concentrations de calcium et de magnésium et la dureté étaient comparables dans les deux bassins.

La charge totale en éléments nutritifs pénétrant dans le lac varie d'année en année, généralement sous l'effet des variations de débit. De 1994 à 2016, les charges de phosphore ont varié entre un peu plus de 3 000 tonnes par an et un peu moins de 11 000 tonnes par an, et au cours des cinq dernières années, elles se situaient autour ou en deçà de la moyenne (7 368 tonnes/an). La rivière Rouge est la plus importante source de phosphore, représentant 69 % de la charge totale moyenne. Depuis les années 1990, les débits ayant augmenté dans la rivière Rouge, les charges en éléments nutritifs transportées dans le lac dépassaient les niveaux historiques. Les concentrations de phosphore dans le bassin sud, qui s'établissaient en moyenne à 0,104 mg/L de 1999 à 2016, soit environ le double de l'objectif fixé pour le phosphore (0,05 mg/L), demeurent élevées par rapport aux concentrations historiques estimées. Dans le bassin nord, les concentrations de phosphore s'établissaient à 0,039 mg/L en moyenne de 1999 à 2016 et se situaient sous la moyenne à long terme au cours des cinq dernières années de cette période d'observation. Dans les deux bassins, de 1999 à 2016, les concentrations de phosphore étaient généralement plus élevées dans les eaux de fond que près de la surface du lac, cet écart étant toutefois plus marqué dans le bassin nord. La charge interne de phosphore du lac, qui résulte à la fois de la diffusion du phosphore dissous des sédiments et de la remise en suspension du phosphore particulaire des sédiments lacustres sous l'effet du vent, devrait être égale ou supérieure à la quantité de phosphore acheminée dans le lac par les cours d'eau. Une meilleure connaissance de l'influence de cette charge interne de phosphore est nécessaire pour comprendre le cycle du phosphore et le bilan de phosphore du lac.

De 1994 à 2016, les charges d'azote du lac variaient entre 52 470 tonnes/an et 136 676 tonnes/an. Les charges se situaient sous la moyenne (91 263 tonnes/an) pendant trois des cinq dernières années, surtout en raison de charges plus faibles que la moyenne en provenance des rivières Rouge et Winnipeg. Les rivières Rouge et Winnipeg sont les principales sources d'azote du lac, représentant en moyenne 34 % et 22 % de la charge, respectivement. Les concentrations d'azote sont généralement les plus élevées à l'extrémité sud du lac, près de l'embouchure de la rivière Rouge. Les concentrations dans le bassin sud tendent à culminer pendant l'été et à diminuer à l'automne, tandis que dans le bassin nord, elles demeurent constantes pendant l'été et l'hiver. De 2012 à 2016, les concentrations d'azote semblaient plus faibles qu'au début de la période d'observation. La concentration moyenne d'azote total dans le bassin sud (0,85 mg/L) entre 1999 et 2016 est légèrement supérieure à l'objectif (0,75 mg/L), tandis que dans le bassin nord (0,63 mg/L), elle est inférieure à l'objectif. Les taux de fixation de l'azote et de dénitrification ont été peu étudiés, et d'autres travaux de recherche permettraient de mieux comprendre le cycle de l'azote et le bilan d'azote du lac.

Bien que les études de la chimie de l'eau visent essentiellement à faire le suivi des concentrations d'éléments nutritifs et à surveiller la chimie générale du lac, les niveaux de certains autres contaminants et oligoéléments présents dans le lac ont aussi été examinés. Les contaminants en usage (p. ex., pesticides, produits pharmaceutiques, ignifugeants bromés) et les polluants organiques organochlorés hérités ne semblent pas présenter de risque aigu ou chronique pour l'écosystème du lac Winnipeg. Le niveau de détection de pesticides est faible dans le lac Winnipeg, et aucun échantillon ne dépassait les lignes directrices du Manitoba pour la protection de la vie aquatique. Cependant, d'autres travaux de recherche sur des scénarios d'exposition de long terme permettraient de combler une lacune pour certains pesticides couramment utilisés dans le bassin versant (p. ex., atrazine, 2,4-D, dicamba, glyphosate et MCPA). Les concentrations de microplastiques semblent plus élevées dans le lac Winnipeg que dans les lacs Huron et Supérieur, mais elles sont comparables aux concentrations mesurées dans le lac Érié. À la différence des Grands Lacs laurentiens, où les microplastiques sont surtout présents sous forme de fragments et de granules, la majeure partie des particules recensées dans le lac Winnipeg sont des fibres. Les incidences des microplastiques sur l'écosystème aquatique du lac Winnipeg sont actuellement inconnues et nécessitent des recherches plus poussées. Les concentrations d'oligoéléments sont faibles dans le lac Winnipeg et souvent sous le seuil de détection. La plupart des métaux autres que l'aluminium et le fer, qui sont naturellement présents en fortes concentrations dans la plus grande partie des eaux douces du Manitoba, dépassaient rarement les lignes directrices pour la protection de la vie aquatique. Entre 1999 et 2016, les concentrations de la plupart des oligoéléments dans le lac Winnipeg étaient comparables aux concentrations types mesurées dans les autres eaux douces du Manitoba.

Le lac Winnipeg abrite une communauté biologique diversifiée. Bien que le phytoplancton fasse partie des écosystèmes d'eau douce naturels, la présence de quantités excessives d'algues, en particulier de cyanobactéries, est préoccupante en raison de leurs effets potentiels sur les activités récréatives, la pêche et l'eau potable. Au cours des deux dernières décennies, la composition des espèces et la biomasse du phytoplancton ont beaucoup varié, probablement sous l'effet des importantes inondations et des variations interannuelles des vents, des éléments nutritifs, de la lumière et de la température. Au cours des cinq dernières années toutefois, la biomasse totale du phytoplancton est demeurée stable et comparable à la moyenne de la période 1999-2016. En dépit de variations annuelles importantes dans la proportion relative des différentes classes de phytoplancton, les cyanobactéries représentaient, en moyenne, environ la moitié de la biomasse totale du phytoplancton dans le lac Winnipeg pendant la période d'eau libre. La biomasse des cyanobactéries était plus importante en 2011 et constituait près de 83 % de la biomasse totale du phytoplancton. L'étendue, l'intensité et la durée des efflorescences algales présentaient également des variations annuelles notables. La biomasse du phytoplancton est généralement plus élevée dans le bassin nord, où les conditions lumineuses et les éléments nutritifs présents favorisent la croissance. De 2003 à 2018, les efflorescences duraient en moyenne 40 à 80 jours au cours d'une même saison bien que chaque année, des efflorescences durant jusqu'à 150 jours aient été enregistrées à certains endroits. La microcystine, une toxine cyanobactérienne, n'est pas souvent décelée au large du lac Winnipeg, mais elle est plus souvent présente près des rives du bassin sud. Cependant, les concentrations demeurent assez faibles et sont inférieures à l'objectif fixé pour les eaux utilisées à des fins récréatives dans la plupart des échantillons.

D'autres composants du réseau trophique aquatique sont soumis à des pressions ou ont subi des changements qui étaient plus prononcés au cours des dernières années. Six espèces aquatiques du lac Winnipeg (trois poissons, un mollusque, un reptile et un amphibien) sont considérées en péril ou préoccupantes : l'esturgeon jaune, la lamproie brune, le buffalo à grande bouche, la mulette feuille d'érable, la tortue serpentine et la grenouille léopard. Au cours de la dernière décennie, deux nouvelles espèces envahissantes ont commencé à s'implanter dans le lac et auront vraisemblablement une incidence sur les populations d'invertébrés et de poissons. Le cladocère épineux, une espèce très commune en Eurasie, a été découvert dans le lac Winnipeg en 2011 dans l'estomac de poissons capturés dans le bassin sud. À l'automne 2012, cette espèce avait colonisé le bassin nord. À l'automne 2013, des moules zébrées, originaires d'Europe, ont été découvertes dans plusieurs ports du bassin sud. Les moules sont maintenant bien établies dans le bassin sud du lac et se sont propagées au bassin nord. Les changements qui peuvent survenir dans l'écosystème du lac en présence de ces espèces envahissantes sont incertains en raison de la complexité des diverses interactions physiques et biologiques.

La pêche dans le lac Winnipeg a aussi subi des changements récents. Le secteur historique de la pêche au doré jaune est en déclin, menacé par l'effondrement de la population d'éperlans arc-en-ciel non indigènes dans le bassin nord du lac et plusieurs années marquées par une forte mortalité, attribuable à la pression exercée par la pêche intensive. L'état de la population de dorés noirs est aussi préoccupant. La population connaît un déclin multidécennal, les rendements de la pêche ayant diminué de plus de 90 % au cours des dernières décennies. Une analyse des tissus des poissons indique que les teneurs en mercure des poissons du lac Winnipeg de 2010 à 2016 sont parmi les plus faibles enregistrées pour les plans d'eau surveillés au Manitoba et sont inférieures à la recommandation de Santé Canada pour le poisson vendu au détail.

La surveillance de la qualité de l'eau à l'aide de la bactérie indicatrice *E. coli* sur les plages du lac Winnipeg de 2004 à 2018 a montré que dans la majorité des cas, les plages respectaient généralement l'objectif de qualité de l'eau pour les eaux utilisées à des fins récréatives et qu'aucun changement significatif n'était survenu au cours de cette période. Les études menées sur les sources d'*E. coli* sur les plages du lac Winnipeg ont indiqué qu'une grande partie des bactéries *E. coli* provenaient de sources animales, principalement des oiseaux de rivage et des oies.

Le lac Winnipeg est un système complexe. Au cours des dernières décennies, notre connaissance de sa fonction et de sa structure s'est améliorée, mais il subsiste de nombreuses lacunes. Certaines de ces lacunes ont été relevées dans le premier rapport sur l'état du lac Winnipeg et sont abordées dans la présente mise à jour (p. ex., populations de poisson et profils de température et d'oxygène, variabilité spatiale des composants chimiques), tandis que d'autres sont à l'étude (p. ex., zones littorales, charge interne en éléments nutritifs et remise en suspension des sédiments). Il faudrait notamment redoubler d'efforts pour intégrer le savoir autochtone et la science occidentale afin de mieux comprendre l'état du lac. Les incidences des espèces envahissantes, des changements climatiques et de l'incertitude qui entoure les effets des microplastiques sur différents biotes suscitent par ailleurs de nouvelles questions.

La gestion du lac demeure une tâche ardue. Il faudra du temps pour améliorer la situation, compte tenu de l'étendue du bassin versant, des nombreuses petites sources diffuses d'éléments nutritifs et de la charge interne en éléments nutritifs hérités qui sont présents dans les sédiments lacustres. Dans l'intervalle, les renseignements contenus dans ce rapport seront utilisés pour mettre au point des indicateurs de la santé du lac qui pourront être mis à jour et faire l'objet de rapports périodiques. Des efforts pour améliorer la santé du lac Winnipeg se poursuivront dans l'ensemble du bassin hydrographique du lac Winnipeg et comprendront une modernisation des usines de traitement des eaux usées, la mise en œuvre de pratiques de gestion bénéfiques afin de réduire la charge en éléments nutritifs et des mesures visant à limiter la propagation des espèces aquatiques envahissantes. Dans le cadre des partenariats multilatéraux établis et compte tenu du vif intérêt manifesté par les intervenants, la prochaine étape clé pour assurer la santé de l'écosystème aquatique du lac Winnipeg consisterait notamment à envisager l'adoption d'une démarche de gestion adaptative. Une gestion adaptative utilise les meilleurs renseignements et les meilleures données disponibles pour la gestion du lac et permet d'apporter des ajustements pour tenir compte des nouvelles connaissances dans des domaines tels que les changements climatiques, les espèces envahissantes et les processus intralacustres.

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ACRONYMS

A

AAFC	Agriculture and Agri-Food Canada
Ag	silver
Al	aluminum
As	arsenic
AIS	aquatic invasive species
ATZ	atrazine

B

Ba	barium
Be	beryllium
BDE	brominated diphenyl ether
Bi	bismuth
BMF	biomagnification factors
Bo	boron
BTBPE	bis(2,4,6-tribromophenoxy)ethane

C

Ca	calcium
CAMP	coordinated aquatic monitoring program
CB	total chlorobenzenes
CBWM	community based water monitoring
CBZ	carbamazepine
CCME	Canadian Council of Ministers of the Environment
Cd	cadmium
CHL	Total chlordane
Cl	chloride
CLO	clothianidin
Co	cobalt
COSEWIC	Committee of the Status of Endangered Wildlife in Canada
Cr	chromium
Cr(VI)	hexavalent chromium
Cs	cesium
Cu	copper

D

DBDPE	decabromodiphenylethane
DCP	Dechlorine Plus
DDT	dichlorodiphenyltrichloroethane
DIN	dissolved inorganic nitrogen
DOC	dissolved organic carbon
DP	dissolved phosphorus

E

ECCC	Environment and Climate Change Canada
E. Coli	*Escherichia coli*
EGS	ecological goods and services
ESA	European Space Agency

F

Fe	iron
FWMC	flow weighted mean concentration

G

GCM	Global Climate Models

H

HBCD	hexabromocyclododecane
HC	Health Canada
HCH	Hexachlorocyclohexane
HCO_3^-	bicarbonate

I

IMI	imidacloprid

K

K	potassium

L

Li	lithium
LWBI	Lake Winnipeg Basin Initiative
LWCBMN	Lake Winnipeg community based monitoring network
LWF	Lake Winnipeg Foundation
LWRC	Lake Winnipeg Research Consortium

M

MARD	Manitoba Agriculture and Resource Development
MeHg	methylmercury
MERIS	medium-spectral resolution imaging spectrometer
MFMD	Manitoba fish mercury database
Mg	magnesium
Mn	manganese
Mo	molybdenum
MODIS	moderate resolution imaging spectroradiometer
MOU	Memorandum of Understanding
MP	microplastics
MSY	maximum sustainable yield

N

NIS	non-indigenous species
N	nitrogen
Na	sodium
Ni	nickel
NTU	nephelometric turbidity units

O

OC	organochlorine
OLCI	Ocean and Land Colour Instrument

P

P	phosphorus
Pb	lead
PCA	pentachloroanisole
PCB	polychlorinated biphenyl
PCIC	Pacific Climate Impacts Consortium
POP	persistent organic pollutants
PP	particulate phosphorus

R

Rb	rubidium

S

SARA	Species at Risk Act
Se	selenium
Sn	tin
SO_4^{2-}	sulphate
Sr	strontium

T

TDS	total dissolved solids
Te	tellurium
THM	thiamethoxam
Ti	titanium
TIC	total inorganic carbon
Tl	thallium
TN	total nitrogen
TOC	total organic carbon
TP	total phosphorus
TSS	total suspended solids

U

U	uranium

V

V	vanadium

	W		Z
W	tungsten	Zn	zinc
WQG	water quality guidelines	Zr	zirconium
WSC	Water Survey of Canada		

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1.0 INTRODUCTION

By: Elise Watchorn (Environment and Climate Change Canada)

Lake Winnipeg and its Watershed

Lake Winnipeg, a remnant of post-glacial Lake Agassiz, consists of a large north basin connected by a narrow channel to a smaller south basin (Figure 1-1). With an area of 23,750 km^2 and extending 436 km from south to north, it is Canada's sixth largest lake (Brunskill et al. 1980) and, excluding the Caspian Sea, the eleventh largest freshwater lake by surface area in the world (Oxford World Atlas, 2015). Geologically, sedimentary bedrock (limestone, dolomite and sandstone) underlies the western half of the lake and dominates its western shores, whereas Precambrian shield underlies the eastern half (Todd et al. 1998). Muskeg wetlands and forested areas are typical of the northern shores (Watchorn et al. 2012, AAFC 2010); agricultural land predominates along the southwest. The outflow of Lake Winnipeg has been regulated for hydroelectric generation since 1976, making the lake one of the world's largest reservoirs.

Figure 1-1: Lake Winnipeg, illustrating the north and south basins and the narrows. Source: Environment and Climate Change Canada.

Compared to the Laurentian Great Lakes, Lake Winnipeg is relatively shallow. Though the deepest point, northeast of Black Island, reaches some 60 m, mean depths in Lake Winnipeg's north and south basins are 13.3 m and 9 m, respectively (Brunskill et al. 1980). By contrast, Lake Erie, the shallowest of the Great Lakes bordering Ontario and the United States, has an average depth of 19 m; that of Lake Superior is 147 m (USEPA 2018). Lake Winnipeg's shallow depth, comparatively small volume (284 km^3) and considerable tributary inflows make for water residence times that have varied between 2.5 and almost 8 years (*see Section 3.0*), much shorter than residence times of the Great Lakes (such as 191 years for Lake Superior).

The shallow nature of Lake Winnipeg, combined with an unbroken fetch of over 300 km along the direction of prevailing northwest winds, enables the formation of waves over 6 m in height (EC 2016a). The lake's morphometry also contributes to the development of large seiches in which water levels at the northern or southern extremes of the lake can rise or fall as much as 1.2 m in a matter of hours (Einarsson and Lowe 1968).

Lake Winnipeg's north and south basins are unique from each other in physical characteristics. Differences in basin size and depth, as well as the quantity and quality of water flowing into each basin, also cause the north and south basins to differ from each other in terms of water quality characteristics, including clarity and nutrient concentrations (EC and MWS 2011). This dissimilarity in water quality drives biological differences between basins, such as the composition and abundance of phytoplankton and zooplankton communities, and the structure of the fish community (EC and MWS 2011).

Figure 1-2: The Lake Winnipeg drainage basin. Source: Environment and Climate Change Canada.

Lake Winnipeg lies within a watershed of nearly 1,000,000 km^2 (Figure 1-2), encompassing parts of Alberta, Saskatchewan, Manitoba and Ontario, as well as portions of Minnesota, North and South Dakota, and Montana.

The lake is strongly influenced by a large watershed: the ratio of watershed to lake surface area is approximately 40:1, many times higher than that of any Great Lake (LWIC 2005).

Approximately 6.7 to 6.9 million people reside in the Lake Winnipeg basin (SC 2016a, USCB 2010), the majority in large cities such as Calgary, Edmonton and Winnipeg. The climate and landscape of the Lake Winnipeg basin is defined, from west to east, by the Montane Cordillera, Prairie, Boreal Plains, and Boreal Shield ecozones (ESWG 1995); easily erodible sedimentary soils underlie the bulk of the watershed, with the less erodible Precambrian Shield forming the landscape in the east. Land use in the watershed is dominated by agriculture, covering over 500,000 km^2; 65% of which is cropland (SC 2016b; USDA 2012). These urban and agricultural landscapes represent sources of nutrients and other substances of concern that are carried downstream via tributaries. Both point and non-point sources of nutrients and other contaminants, including wastewater discharge and agricultural runoff, affect the aquatic ecosystem of, and contribute to deteriorating water quality, in Lake Winnipeg. The watershed is thus large, interjurisdictional and international, and physiographically and climatically diverse, compounding the challenges of managing aquatic inputs to Lake Winnipeg.

The largest of Lake Winnipeg's subwatersheds, that of the Saskatchewan River, drains eastward from the Rocky Mountains through Prairie and Boreal Plains. Multiple dams and reservoirs on headwater streams, the North and South Saskatchewan Rivers, and the mainstem river absorb the highly variable flows of this watershed (EC and MWS 2011), and allow for the sequestration of some nutrients acquired along the courses of its waterways (Donald et al. 2015).

The Red River subwatershed includes the eastward flowing Assiniboine River and the northward flowing Red River, which drain through flat, nutrient rich prairie soils. The low relief landscape is prone to recurring floods, which have increased in frequency and severity through recent climatic change, with a corresponding increase in nutrient loading to Lake Winnipeg (McCullough et al. 2012). Major flood control works to manage this flooding include a channel for the interwatershed diversion of an increasing portion of Assiniboine River flows (and associated nutrient loads) into adjacent Lake Manitoba.

The Winnipeg River subwatershed drains northwestward from Ontario and southeast Manitoba. Boreal Shield comprises a majority of this subwatershed, with forested terrain and shallow soils interspersed by many lakes including the large Lake of the Woods.

The Dauphin River subwatershed drains eastward, through Boreal Plains terrain, via the other great lakes of this region: lakes Winnipegosis and Manitoba, each of which sequesters a major fraction of the nutrients it receives (Donald et al. 2015). Major recent flooding on Lake Manitoba and downstream of Lake St. Martin, due to increased local runoff and the diversion of water from the Assiniboine subwatershed, has led to the construction of channels providing a new outlet directly into Lake Winnipeg.

The remainder of the Lake Winnipeg basin is made up of smaller rivers draining directly to Lake Winnipeg, with forested and agricultural Interlake streams immediately west of the Lake, and myriad rivers and wetlands draining Precambrian Shield to the east. Noncontributing areas, primarily in the prairies, may represent approximately a third of the total Lake Winnipeg watershed in low precipitation years, but increase the effective drainage area to the lake when extreme wet conditions facilitate their hydrologic connection to tributaries.

There are nearly six million discrete wetlands within the Lake Winnipeg watershed, comprising over 150,000 km^2 (P. Badiou, unpublished). These range from the small palustrine potholes of the Prairie ecozone, large riverine wetlands such as the Saskatchewan River Delta, vast peatlands of the Boreal Shield, and coastal wetlands directly abutting Lake Winnipeg such as Netley-Libau Marsh. These wetlands provide ecosystem services including the attenuation of eutrophication through nutrient sequestration, the provision of spawning and feeding habitat for sport and commercial fisheries, migration and breeding habitat for waterfowl and wildlife (Goldsborough 2015) and the mitigation of flood damage through water storage (Simonovic and Juliano 2001). However, palustrine wetlands have been extensively drained or degraded by agricultural and other activities, throughout the Lake Winnipeg basin but particularly in the Red River subwatershed (Hanuta 2001). Coastal wetlands face impairment from hydrological regime change (Watchorn 2015), lake level regulation, shoreline development, and destructive invasive species (Wrubleski et al. 2018). These restrictions in wetland extent and function limit their ability to modulate water quality, which can adversely impact the downstream environment, including Lake Winnipeg.

Wetlands play an important role in managing water quality and quantity in the landscape.

TEXT BOX 1: Traditional Knowledge Perspectives

By: Lake Winnipeg Indigenous Collective, Wendy Ross and Thomas Beaudry

"Traditional knowledge" is a term largely used by organizations and academics to describe a body of knowledge which is collectively generated, shared, preserved and passed down in a community through oral tradition, practical experience, or first-hand observation (WIPO, 2012). In the current literature, this body of knowledge describes deep interconnected relationships of all living beings with each other where no living being or relationship has primacy and all things on the land are considered alive (Little Bear 2009, McGregor 2013). It can take many forms including artistic or cultural expressions, scientific works, inventions, discoveries, names, ceremonies, practices, songs and skills. Traditional Knowledge cannot be universally defined as each community has a unique knowledge system co-developed within the context of their own history, language and culture over many generations and in response to a continuously changing environment (WIPO 2012, McGregor 2006).

> *Traditional knowledge is not a product or object that can be defined and studied in isolation. It is participatory and experiential (Leroy Little Bear 2012)*

Traditional knowledge can include ways of understanding the natural environment, and is referred to as the Indigenous science (Snively and Williams 2016). It is often cultivated through extensive observation, spiritual connections, teachings and lived experiences on the land and waters (Berkes et al. 2000, FSCC 2004). For Lake Winnipeg, this science has been held in the Indigenous communities around the lake since time immemorial. It is a form of intellectual property, generated in the communities, sustained in oral histories and through on-the-land teachings, and passed down through generations and between resource users.

Indigenous nations globally have been living in harmony with natural systems and using their local Indigenous science to sustainably manage natural resources for thousands of years (Mazzocchi 2006). It is recognized internationally that Indigenous science plays a crucial role in achieving sustainable resource management and biological diversity (UNCBD 1992, Gadgil et al. 1993). It has been incorporated into ecological restoration and management, and conservation biology, and is recognized as equivalent to western science (UNEP 1998, Berkes et al. 2000). Several similarities exist between Indigenous and western science knowledge systems. Both systems reflect a way of knowing based on observation and creating order out of disorder (Berkes 2005). Indigenous science is evidence-based and can be applied to modern challenges and understanding of environmental changes. Like western science, it depends on protocols and processes that reference sources, invite peer review and criticism, and identify biases. Although often complementary, Indigenous science is openly embedded within a system of values, such as the principles of respect, reciprocity, spiritual connection, and responsibility to the environment (Kimmerer 2000).

Continued next page

Text Box 1 continued

We contend that Indigenous and western sciences together have a great deal to offer for the holistic understanding of Lake Winnipeg and how it can be managed sustainably for future generations. Buck (2009) describes the importance of acknowledging a variety of other knowledge systems:

> *We as individuals tend to view our civilization as "the best" and when our teachings, knowledge, and belief systems are ridiculed, marginalized and then utterly dismissed as "quaint", we begin to question our worldview. This has happened and is still happening to First Nations people as well as all colonized peoples. Until other worldviews are proposed and considered, there will be a distinct "difference" and "quaintness" about all that is not mainstream.*

In *Ininew* (Cree) and *Anishinabe* (Ojibway) philosophies there is a term *mino-pimatisiwin* and *mino-bimaadiziwin,* respectively, which translates into 'living the good life'. This philosophy is described as an active process of achieving balance in relationship with all thing. The word for water in Ininew is *nipi,* which directly refers to *pimatisiwin,* or my way of living, giving rise to the 'water is life'. Within this philosophy water has a spirit, and all things are living and related. *Makeso Sakahikan* (Fox Lake) elders describe this concept:

> *Mino-pimatisiwin relates to the overall health of our people…Specifically, our lands and waters should be whole and healthy, both of which are prerequisites of a peaceful existence (FLCN, 2011).*

Many Indigenous people around Lake Winnipeg have a fundamental relationship with the natural environment, and particularly, the water. Water is given by the Creator, and is viewed as a life-giving spiritual entity and the blood of mother earth (Cave and McKay, 2016; Craft, 2014). The Indigenous women, in particular, share a sacred connection to the water through their role as child-bearers and water carriers (Cave and McKay, 2016; Craft, 2014). The women have the ultimate role and responsibility in protecting, nurturing, and speaking for the health of the water (Craft, 2014), while expressing respect, honor, and gratitude for its spirit through ceremony and song (Cave and McKay, 2016). For the Indigenous nations residing on the shores of Lake Winnipeg, the water provides for many spiritual, cultural, recreational and economic needs. It supports the commercial and subsistence fishery, as well as medicines, plants, and berries. It provides for the animals needed for hunting and trapping, while also providing recreational and cultural opportunities such as angling, boating, paddling, and swimming. Water is also an integral part of the spiritual identity of many of these nations.

Indigenous peoples who have resided near the shores of Lake Winnipeg since time immemorial have passed along detailed and comprehensive information about the health of the lake, and how it has changed over time, that have not been documented or observed by western science. To date, the State of Lake Winnipeg reporting has not included Indigenous science. Since western science does not and cannot have a complete picture of the lake, this presents a significant knowledge gap in all of the data reported.

State of the Lake Reporting

The growing concerns over the presence of large and sustained algal blooms in Lake Winnipeg in the 1990s prompted an increased effort to understand and monitor the state of the lake's aquatic ecosystem. The Lake Winnipeg Research Consortium (LWRC) was established in 1998 with a mandate to coordinate federal and provincial agencies, academic researchers, and other lake stakeholders to facilitate multidisciplinary scientific research on the condition of the lake and its basin. Since 1999, the Consortium has facilitated and conducted seasonal lake-wide scientific surveys of a network 65 monitoring stations each spring, summer and fall (Figure 1-3). Additionally, the Province of Manitoba maintains a long-term, lake-wide monitoring program to assess ecosystem health and water quality in Lake Winnipeg. Established in 1999, this program centers around 14 key stations in the sampling network (Figure 1-3), monitored once per year under-ice as well as three times in the open water season. Variables analyzed include nutrient and metals concentrations, dissolved oxygen, measures of water clarity, pesticides, algal biomass and species composition, algal toxins such as microcystin, and zoobenthos. During the spring, summer and fall surveys, water samples are also collected from the Consortium's larger network but are analyzed for a reduced suite of parameters including nutrients, conductivity, turbidity and pH.

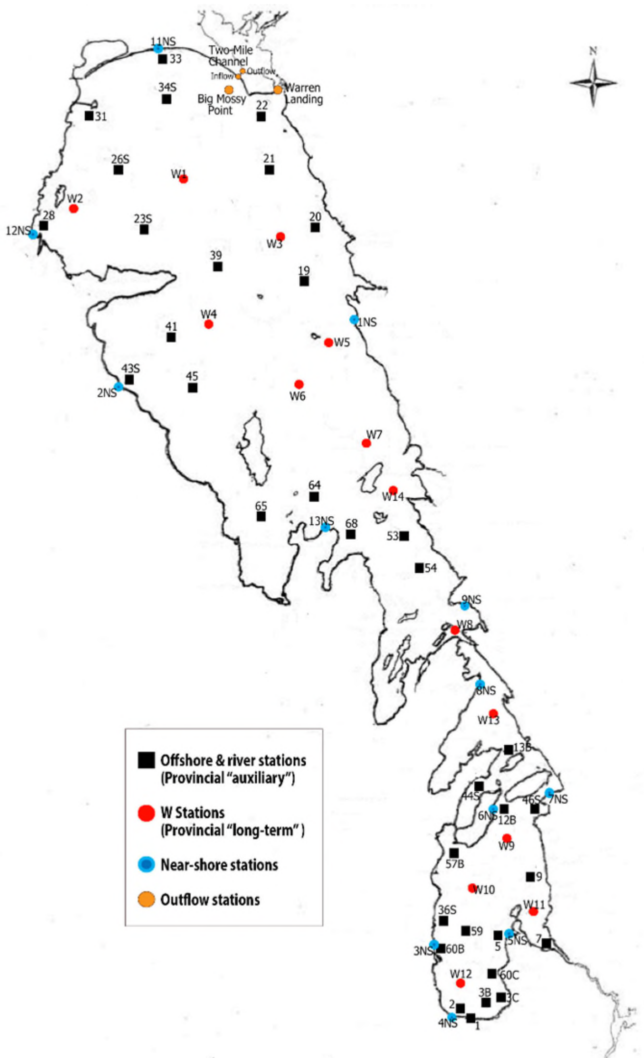


Figure 1-3: The current network of routinely monitored pelagic, littoral and outflow stations in Lake Winnipeg. An original network of 65 stations was selected based on those sampled in 1969; additional stations were subsequently added to maximize spatial representation of the lake, while others were retired to allow for a new focus on littoral monitoring. Source: LWRC.

In 2011, a report on the state of Lake Winnipeg was published (EC and MWS 2011), a first of its kind compilation of physical, chemical and biological characteristics of Lake Winnipeg and its watershed for the 1999 to 2007 period. This comprehensive document included contributions from three dozen authors from federal and provincial government agencies, from academia, and from other public and private science organizations. Its objectives were to describe the temporal and spatial variation in Lake Winnipeg's physical, chemical and biological characteristics, to explore existing and emerging issues of concern to the health and integrity of the lake, and to highlight recent research.

Since 2007, further research and ongoing monitoring programs have advanced our understanding of Lake Winnipeg and its watershed. Lake Winnipeg, too, has changed since 2007. The effects of climate change are becoming more palpable and new invasive species have colonized the lake—the Spiny Water Flea (*Bythotrephes longimanus*) in 2011, and the Zebra Mussel (*Dreissena polymorpha*) in 2013—whose impacts on the lake's biology and chemistry are beginning to be marked, and merit investigation. Forage fish community structures have seen dramatic shifts in recent years, with the north basin crash in Rainbow Smelt (*Osmerus mordax*) populations (Lumb et al. 2012). Contaminants of emerging concern, such as pharmaceuticals and microplastics, are being observed in Lake Winnipeg biota, sediments, and water columns (Challis et al. 2018, Warrack 2017).

The Lake Winnipeg Research Consortium facilitates and conducts seasonal lake-wide scientific surveys aboard their research vessel the MV Namao.

In the decade since the last report, numerous scientific studies have increased our knowledge about the lake and the processes that influence the state of the lake. Advancements have been made on topics such as: improving the ability to track and predict algal blooms with remote sensing (*e.g.* Binding et al. 2018); nutrient loading, sequestration, and source fingerprinting (*e.g.* Donald et al. 2015, Hobson et al. 2012, Rattan et al. 2017); coastal wetlands (Watchorn et al. 2012), beneficial management practices for nutrient retention (*e.g.* Wong et al. 2014, Liu et al. 2014), watershed and in-lake eutrophication models (*e.g.* Booty et al. 2011, Zhang and Yerubandi 2012); microplastics and other contaminants (*e.g.* Challis et al. 2018, Warrack 2017); and the aquatic biology of the lake (*e.g.* Hann et al. 2017, Hann and Salki 2017, Lumb et al. 2012). Research funded under the Lake Winnipeg Basin Initiative (LWBI) of Environment and Climate Change Canada (ECCC) is summarized in the final reports for Phases I (EC 2013) and II (ECCC 2018a) of the LWBI.

Under the Canada-Manitoba Memorandum of Understanding (MOU) Respecting Lake Winnipeg and the Lake Winnipeg Basin (EC and MWS 2010), Canada and Manitoba developed a Science Subsidiary Arrangement that includes a commitment to improve reporting on the status of and trends in water quality and aquatic ecosystem health in the Lake Winnipeg Basin. To address this commitment, ECCC and Manitoba Agriculture and Resource Development (MARD) collaborated on the production of an updated State of Lake Winnipeg report. This updated report serves to expand the state of collective knowledge of Lake Winnipeg and its watershed to cover information and knowledge gained since publication of the first report.

2.0 CLIMATE

By: Ryan Smith (Prairie Climate Centre, University of Winnipeg)

Lake Winnipeg's historical, cultural and ecological significance is intimately tied to climate and climate change. The lake owes its very existence to the last major climate event the region experienced approximately 11,000 years ago when the large glaciers that covered much of North America melted to form Lake Agassiz. Following this event, and continuing until very recently, the climate system entered a long period of relative stability. This stability supported a vibrant lake ecosystem, which in turn, supports Manitoba's modern fishing, tourism and hydroelectric economies and Indigenous communities.

The lake is also no stranger to periodic, short-term climatic extremes. Since detailed and reliable meteorological record keeping began at the end of the 19th century, the region has had to deal with severe drought, severe seasonal and flash flood events, and severe thunderstorms capable of tornadoes. As a result, Lake Winnipeg's flora, fauna and human inhabitants are exceptionally adept at climate resilience, as there are few places on Earth where a system must cope with nearly -50°C in the winter and +40°C in the summer.

Despite this ability to deal with climatic extremes, Lake Winnipeg remains highly vulnerable to present-day, human-caused climate change. Seemingly small projected changes in annual or seasonal mean temperatures across the lake's vast drainage basin disguise the more insidious side of climate change, which is that extreme heat events are expected to occur much more frequently in the coming decades. Heat extremes have enormous impacts on water availability for agriculture, hydroelectric production, and municipal uses and they are directly related to the frequency and intensity of forest fires, the intensity of severe weather, and human health. More importantly, from a lake point-of-view, water levels, temperature, nutrient load, dissolved oxygen concentration, ice thickness and extent, and aquatic ecosystem integrity are impacted or threatened by changes in climate and extreme weather. Understanding the historical climate of Lake Winnipeg, as well as how its climate is projected to change in the future, is therefore of critical importance.

Methods

Baseline (historical) climate normals for the lake were constructed from data for four communities adjacent to the lake: Grand Rapids, Norway House, Berens River and Arborg. Combined, these data provide an adequate description of the spatial variations in climate across the north basin, narrows, and south basin of Lake Winnipeg, as well as the east and west shores. As is standard practice in climatology, the 30-year period of 1981 to 2010 was used to construct these climatic normals.

Importantly, these land-based weather stations describe atmospheric characteristics at points adjacent to the lake, which are closely related to, but ultimately different from, climatic characteristics *over* the lake. Furthermore, Lake Winnipeg is sufficiently large enough to greatly impact its surrounding climate, modulating nearby temperatures, raising atmospheric humidity levels and creating local-scale winds and gust-fronts that have an impact on the weather locally and downwind of the lake.

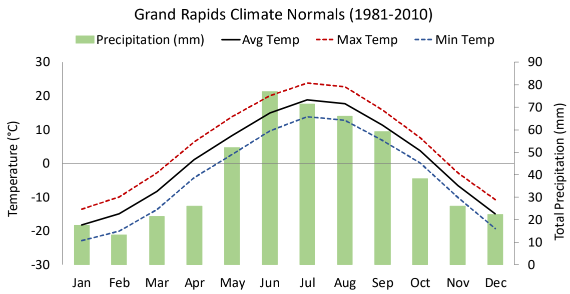
An ensemble of 12 statistically downscaled Global Climate Models (GCMs) made available by the Pacific Climate Impacts Consortium (PCIC) were used to project future climate changes over the lake under two scenarios, RCP8.5 (‘High Carbon’) and RCP4.5 (‘Low Carbon’), and over two 30-year periods, 2021 to 2050 and 2051 to 2080. For comparative purposes, modelled baseline statistics for the period 1976 to 2005 are also shown. The data sources and methods used to construct the baseline and projected future climate statistics for the Lake Winnipeg region are described in the Appendix.

Lake Winnipeg’s Baseline (Historic) Climate Context

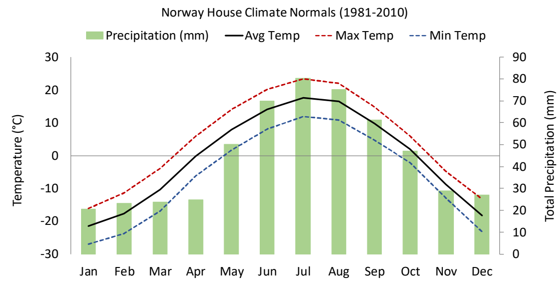
Temperature

Lake Winnipeg is a very large body of water and, as such, average temperatures vary between its north and south basins and its east and west shores (Figure 2-1). Within the immediate region around the lake, mean annual temperatures range from 1.6°C in Arborg (south) to -0.7°C in Norway House (north) and from 0.6°C in Berens River (east) to 1.1°C in Grand Rapids (west).

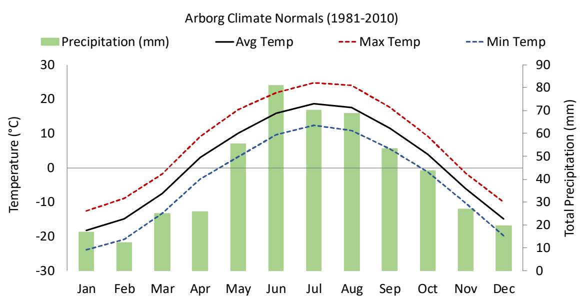
On average, the Lake Winnipeg region experiences a 116-day frost-free season, due in large part to the long days and abundant solar radiation the region experiences during the summer. July is the warmest month of the year, when average mean temperatures range from 17.6°C in Norway House to 18.8°C in Arborg. The average maximum July temperature across the region is 23.8°C, while the record high for the region within the baseline period was 37.5°C, set in Grand Rapids.



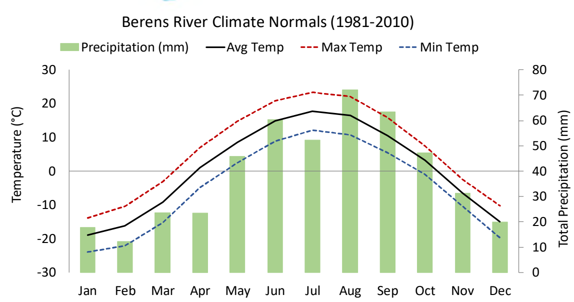
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Extreme Maximum (°C)	7.5	9.0	17.0	28.0	32.5	36.5	37.5	35.0	31.7	25.0	17.5	7.8
Extreme Minimum (°C)	-43.0	-41.1	-38.5	-26.1	-13.3	-3.5	2.0	-1.0	-9.0	-16.0	-32.5	-39.4
Avg. Wind Speed (km/h)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Most Frequent Wind Direction	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Snowfall (cm)	17.6	13	17.6	12	4.8	0	0	0	0.3	8.3	20.4	22.4
Extreme Daily Snowfall (cm)	11.6	12.5	41.2	22.5	30	11.4	0	0	7.4	21.2	18.8	18.8



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Extreme Maximum (°C)	7.7	9.0	14.0	25.9	33.0	33.8	34.7	32.2	31.1	22.9	12.8	5.4
Extreme Minimum (°C)	-49.4	-45.6	-41.5	-32.7	-12.0	-2.0	1.8	-1.4	-7.0	-22.8	-35.9	-45.0
Avg. Wind Speed (km/h)	10.0	10.6	11.6	13.0	13.1	12.5	11.4	11.0	12.1	12.2	10.9	9.9
Most Frequent Wind Direction	NW	NW	S	S	NE	SW	SW	SW	NW	NW	NW	NW
Snowfall (cm)	26.5	28.7	26.5	17.8	5.9	0.1	0	0	1.5	21.4	33.1	33.6
Extreme Daily Snowfall (cm)	25.4	21.4	23.2	18.8	19	0.8	0	0	14.4	22.6	24.6	20



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Extreme Maximum (°C)	7.5	7.8	16.7	30.0	37.0	37.0	36.5	36.1	36.5	28.5	20.6	9.5
Extreme Minimum (°C)	-45.6	-48.3	-42.2	-29.5	-14.0	-5.0	1.1	-2.0	-7.8	-21.0	-40.0	-41.1
Avg. Wind Speed (km/h)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Most Frequent Wind Direction	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Snowfall (cm)	16.9	10.1	15.3	7.8	1.2	0	0	0	0.4	6.9	18.3	18.7
Extreme Daily Snowfall (cm)	30.5	26.9	26	22.4	11	0	0	0	6	41.9	30.5	22



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Extreme Maximum (°C)	7.1	9.4	14.4	24.7	31.7	37.3	35.2	35.6	29.8	22.4	11.9	6.0
Extreme Minimum (°C)	-42.7	-42.8	-40.1	-30.0	-11.0	-2.7	0.5	0.1	-6.9	-18.1	-36.8	-40.5
Avg. Wind Speed (km/h)	11.6	11.5	12.2	13.0	13.0	12.3	12.2	12.0	12.6	13.1	12.4	11.8
Most Frequent Wind Direction	NW	NW	NW	NW	NW	NW	W	NW	N	N	S	NW
Snowfall (cm)	20.7	13.3	17.6	6.7	2.2	0	0	0	0.3	13	26.7	22.7
Extreme Daily Snowfall (cm)	14	11.4	28.6	13.2	7.7	0	0.2	0	3	18.2	14.6	13.8

Figure 2-1: Climate normals (temperature, precipitation and wind) and noteworthy extremes (1981 to 2010) for Norway House, Berens River, Arborg and Grand Rapids, Manitoba. Note: Updated normals data for Gimli, Manitoba were not available at the time of writing. Arborg, located 34 km northwest of Gimli, was used instead since mean annual temperature and precipitation values are nearly identical for the two stations despite Arborg being slightly more inland. Wind data was only available for Norway House and Berens River. (Data Source: ECCC 2017a).

Winter temperatures are correspondingly low and typically sub-zero, due to very short days, long nights and the frequent advection of continental polar air masses into the region, supporting the development of a thick layer of ice and snow over the lake. January is the coldest month of the year, with average January mean temperatures ranging from -21.5°C in Norway House to -18.2°C in Arborg. The average minimum January temperature for the region is -24.4°C and the record lowest temperature for the region within the baseline period, was -49.4°C, recorded in Norway House.

Precipitation

An average of 498 mm of precipitation is recorded across the Lake Winnipeg region annually. Between the months of March and October, the majority of this precipitation falls as rain. In Arborg and Grand Rapids, the wettest month of the year is typically June, with 80.9 mm and 76.9 mm of precipitation, respectively, observed on average. In contrast, the wettest month in Norway House is July, with on average 80.2 mm of precipitation, and the wettest month in Berens River is August, with an average of 72.2 mm of precipitation (Figure 2-1).

Between the months of November and February, most of the observed precipitation falls as snow, with an average snowfall for the region of 132 cm. Norway House receives the largest amount of snowfall on average (195 cm) whereas Arborg receives the least amount of snowfall on average (96 cm).

Importantly, very large annual and inter-annual variability in precipitation totals is common across the Lake Winnipeg region. Extreme one-day rainfall and snowfall events can greatly impact the amount of precipitation a station reports in a given year. The extreme daily rainfall record for the region is held by Arborg, which received 107 mm of rainfall on August 29, 1992. Arborg also holds the record for largest one-day snowfall with 41.9 cm observed on October 8, 2007.

Wind

As with temperature and precipitation, wind follows a distinct seasonal pattern across Lake Winnipeg. During the winter months, when the polar jet stream is most active and an upper-level trough typically forms over the region, surface winds most frequently blow out of the northwest or north. During summer, as the polar jet stream moves further north and becomes weaker, upper-level ridges become more common, which more frequently facilitate surface winds that blow from the west, southwest and south. Throughout the year, average monthly wind speeds range between 10 and 13 km/h. Across daily and sub-daily time periods, wind speed and direction are enormously variable. Storms track into the region during all times of the year, generating large wind gusts. Norway House and Berens River have both experienced wind gusts just shy of 100 km/h.

The Projected Future Climate of Lake Winnipeg

Due in part to latitude and continentality, the climate of the Prairie Provinces, including the Lake Winnipeg region, is warming much faster than the global average. Since 1970, a statistically significant linear trend in the average annual temperature of 0.3°C per decade is detectable in the region surrounding Lake Winnipeg, compared to 0.15°C per decade for the globe (GISS 2017) over this same time period (Figure 2-2).

Temperature

Lake Winnipeg's climate will continue to warm, even under the 'Low Carbon' scenario (Figure 2-3) where the mean annual temperature is projected to rise by 3°C by 2051 to 2080. Under the 'High Carbon' warming scenario, models project much more warming with an approximate 4 to 5°C increase in mean annual temperature by 2051 to 2080.

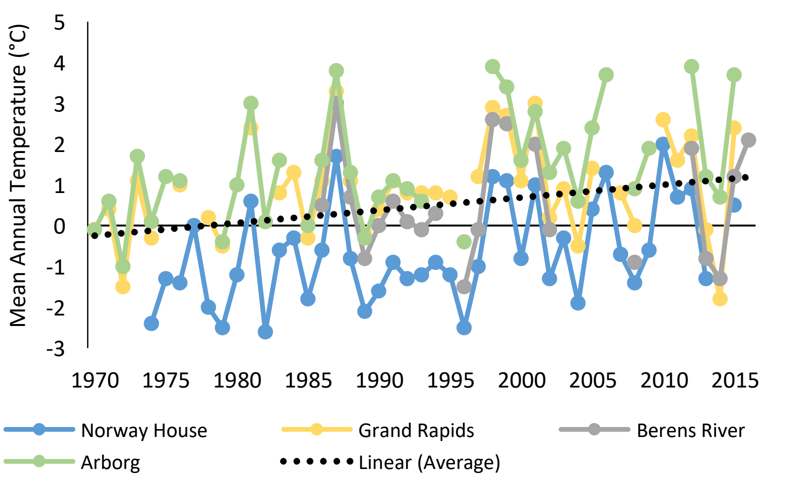


Figure 2-2: Mean annual temperature data for Norway House, Grand Rapids, Berens River and Arborg, Manitoba. The linear trend line was computed from the average of all four stations. Gaps in the data are unfortunately common and due mainly to weather station mechanical issues and observer errors. (Data source: ECCC 2017b).

Changes in temperature extremes best illustrate the profound climatic shifts projected for the region (Table 2-1). For the 2051 to 2080 period, the number of hot days (days when the maximum temperature reaches or exceeds 30°C) is projected to increase dramatically, from an average of zero to three hot days over the baseline period to an average of 11 to 26 hot days under the 'High Carbon' scenario. Conversely, the number of cold days (days when the minimum temperature reaches -30°C or colder) is projected to drop from an average of 14 to 23 days over the baseline period to an average of just one to six cold days under the 'High Carbon' scenario. The warming is projected to result in a large increase in the length of the frost-free season—a mean increase of one month—from 114 to 141 days in the baseline period to 143 to 175 days under the 'High Carbon' scenario.

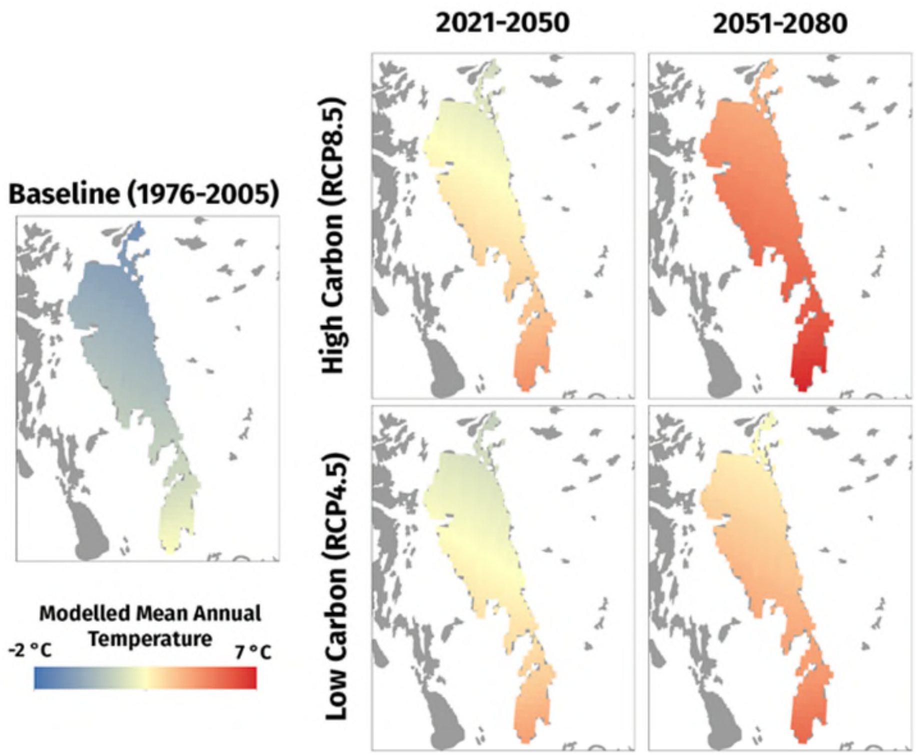


Figure 2-3: Projected changes in mean annual temperature over Lake Winnipeg under the RCP8.5 (High Carbon) and RCP4.5 (Low Carbon) warming scenarios for two future time periods (2021 to 2050 and 2051 to 2080), as well as the modelled baseline mean annual temperature for the period 1976 to 2005 (Data Source: PCIC 2017).

Precipitation

Climate change is expected to impact seasonal precipitation totals across the Lake Winnipeg region. Importantly, precipitation is far more difficult to model than temperature, and there is generally lower confidence in precipitation projections. The winter, spring and fall seasons are all projected to become wetter in the future, while summers are projected to become slightly drier (Table 2-1). Annually, the region is expected to see an increase in precipitation; however, this may not necessarily lead to an increase in runoff and lake water levels. This is because much higher temperatures will likely increase evapotranspiration rates, increasing the risk of a summer moisture deficit and climatological drought.

Table 2-1: Projected climate statistics for Lake Winnipeg. For each time period, the 10th and 90th percentile projections within the 12-model ensemble are given. This was done in part because the lake is very large and a singular statistic does not necessarily capture the range in values often observed between the north and south basins.

	Modelled Baseline	Low Carbon Scenario (RCP4.5)		High Carbon Scenario (RCP8.5)	
	1976-2005	2021-2050	2051-2080	2021-2050	2051-2080
Annual Temp. (°C)	-0.3 to 0.8	1.8 to 3.0	3.0 to 3.9	2.0 to 3.3	3.8 to 6.2
Winter Temp. (°C)	-19.1 to -16.9	-16.7 to -14.2	-15.0 to -12.7	-16.3 to -13.7	-13.7 to -10.4
Spring Temp. (°C)	-0.9 to 0.7	0.7 to 2.4	1.7 to 3.1	0.7 to 2.3	2.1 to 4.5
Summer Temp. (°C)	16.0 to 17.3	17.8 to 18.8	18.8 to 19.7	18.0 to 19.3	19.7 to 21.7
Fall Temp. (°C)	2.1 to 3.4	4.0 to 5.2	5.0 to 6.2	4.0 to 5.9	6.2 to 8.3
Annual Precip. (mm)	425 to 480	447 to 498	460 to 502	446 to 499	464 to 518
Winter Precip. (mm)	58 to 71	66 to 82	68 to 85	66 to 81	72 to 94
Spring Precip. (mm)	71 to 92	76 to 102	83 to 104	81 to 101	89 to 114
Summer Precip. (mm)	167 to 200	167 to 200	164 to 191	166 to 196	160 to 192
Fall Precip. (mm)	108 to 131	119 to 140	120 to 140	117 to 143	115 to 141
Days ≥ 30 °C	0 to 3	1 to 7	5 to 14	3 to 9	11 to 26
Days ≤ -30 °C	14 to 23	4 to 14	2 to 10	3 to 15	1 to 6
Frost-free Season (days)	114 to 141	124 to 159	134 to 161	128 to 162	143 to 175
First Frost	17-Sep to 07-Oct	23-Sep to 16-Oct	29-Sep to 16-Oct	24-Sep to 18-Oct	01-Oct to 25-Oct
Last Frost	16-May to 31-May	07-May to 28-May	02-May to 20-May	04-May to 25-May	28-Apr to 16-May
10 °C Degree Days	656 to 785	848 to 974	997 to 1126	899 to 1055	1129 to 1434

Note: Climate model data were available for the period 1950 to 2095 (although some individual models had data to the year 2100). The High and Low Carbon scenarios begin in the year 2006. As such, the period 1981 to 2010 (the period for which climate normals are computed by ECCC) was not used as a baseline period; instead, a baseline period of 1976 to 2005 was used.

Summary

This section presented some baseline climate normals for four weather stations adjacent to Lake Winnipeg and summarized some future climate projections across the Lake. The modelled projections make it very clear that historic climate normals will very soon be out of date, even under a so-called ‘Low Carbon’ future emissions scenario. Indeed, weather observations over the last 50 years or so indicate that change has already occurred. This underscores the urgent need for climate change adaptation planning in the region. If the models’ high carbon pathway becomes a reality, it is clear that dramatic and challenging climate changes will likely have an impact on the Lake Winnipeg ecosystem.

3.0 HYDROLOGY

By: Carly Delavau and Mark Lee (Manitoba Agriculture and Resource Development) and Diana Fred (Environment and Climate Change Canada)

The Lake Winnipeg watershed covers approximately 1,000,000 km^2, extending from the Canadian Rockies in western Canada to within kilometres of Lake Superior in eastern Canada, and covers portions of four U.S. states to the south (*see Figure 1-2*). Although the climate and geology vary greatly across the basin, the basin's hydrology is generally typical of prairie hydrology and is characterized by variability and extremes.

Inflow to the lake varies greatly throughout the year and from year to year. The majority of inflow typically occurs in spring and early summer due to snowmelt; however, summer rainfall can generate significant inflows under certain conditions. The inter-annual variability in lake inflow can differ by orders of magnitude as conditions range from severe drought to severe flooding. Streamflow into Lake Winnipeg from each subwatershed also varies widely due to differences in a variety of factors, including contributing drainage area, land surface characteristics, climate, and regulated infrastructure, among others.

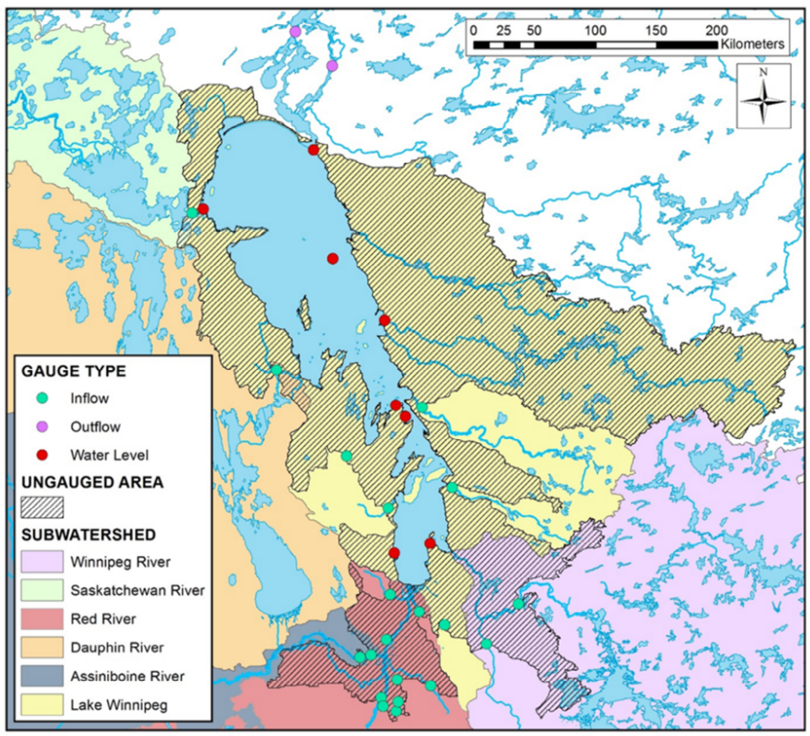


Figure 3-1: Lake Winnipeg subwatersheds and the streamflow and water level gauges included in the analysis. Black hatching represents the ungauged drainage area to Lake Winnipeg.

Water levels and mean discharges into and out of Lake Winnipeg have been determined for the subwatersheds of Lake Winnipeg (Figure 3-1) for three different timeframes to compare recent conditions during the current reporting period (2008–2016) to the previous reporting period (1999–2007) and to historical observations (1977–1998). All streamflow values are reported as a monthly mean discharge in cubic meters per second (m^3/s) unless stated otherwise. A summary of methodologies and gauge station information is included in the Appendix.

Gauged Inflows

Saskatchewan River Subwatershed

Average streamflow of the Saskatchewan River from 2008 to 2016 was 731 m^3/s, which is 31 to 41% higher than it was during 1999 to 2007 and 1977 to 1998, respectively (Table 3-1), thus indicating a wetter period in the Saskatchewan River Basin than in previous years. The highest monthly streamflows on record (post-1977) occurred in 2011 (2,098 m^3/s), 2013 (1,807 m^3/s) and 2014 (1,786 m^3/s). The maximum discharge in 2011 measured 39% above the highest streamflow of the previous reporting period (1,512 m^3/s in 2005) and 64% above the historical period's maximum streamflow (1,283 m^3/s in 1993). The wet conditions observed in 2011 not only occurred in the Saskatchewan River subwatershed, but also were widespread across the Assiniboine, Red and Dauphin River subwatersheds, resulting in major flooding throughout these regions.

The North Saskatchewan River headwaters begin at the Saskatchewan Glacier in Banff National Park.

The driest year between 2008 and 2016 was 2009, particularly throughout the spring and summer months. The minimum discharge of the current reporting period (128 m^3/s) occurred during April of 2009. Streamflows in 2010 were also much lower than average, particularly during the spring. However, the low flows that occurred in 2009 and 2010 were less severe and shorter in duration compared to the drought conditions observed in 2001 to 2003 and 1992 to 1993.

Red and Assiniboine River Subwatershed

Over the current reporting period, the Red River's average discharge was 292 m^3/s (Table 3-1). This value is slightly higher (9%) than the average over the 1999 to 2007 reporting period (269 m^3/s), and significantly higher (73%) than the historical average (169 m^3/s). Between 2008 and

2016, the Red River experienced several major events; the largest in 2009 resulted in the maximum discharge of the reporting period (2,257 m^3/s in April 2009), followed by another major event two years later in April 2011 (1,736 m^3/s). The 2009 event exceeded the maximum streamflow during the previous reporting period (1,812 m^3/s in April 2006); however, it was not as significant as the May 1997 spring melt (2,485 m^3/s). Following the flooding observed in 2011 was a year of drought conditions from spring 2012 to 2013, which speaks to the significant variability in streamflows typical of prairie watersheds. The minimum discharge during the study period occurred in February 2013 (18 m^3/s). This drought event was comparable to the conditions in 2003 to 2004, however was not as significant as the 1988 to 1991, 1980 to 1981 or 1977 drought events in the historical period of record.

Table 3-1: Average, minimum and maximum monthly mean discharges in cubic meters per second (m^3/s) to Lake Winnipeg.

Subwatershed	1977-1998			1999-2007			2008-2016		
	Average	Min	Max	Average	Min	Max	Average	Min	Max
Winnipeg River	883	129	2,519	1,058	7	2,538	1,084	278	2,531
Red River	169	3	2,485	269	14	1,812	292	18	2,257
Saskatchewan River	520	90	1,283	556	42	1,512	731	128	2,098
Dauphin River	65	1	313	94	1	238	211	70	579
Assiniboine River	40	5	254	60	5	290	111	14	519
Lake Winnipeg (Local Gauged Inflows)	59	4	298	71	3	342	94	7	397
Total Gauged Inflow	1,749	531	6,775	2,108	653	6,253	2,515	1,404	6,211
Total Combined Outflow	1,920	763	4,272	2,372	765	4,673	2,931	1,556	5,095

The Assiniboine River basin has unequivocally experienced above average runoff throughout most of the 2008 to 2016 reporting period. During this time, the average streamflow was 111 m^3/s, 85% higher than the past reporting period (60 m^3/s), and 178% higher than the historical period (40 m^3/s) (Table 3-1). There were two major events in the Assiniboine River basin during the current reporting period: 2011 and 2014. The 2011 event was a spring freshet and rainfall-driven flood that generated the maximum streamflow of the current reporting period, while 2014 was a result of intense summer rainfall. The Assiniboine River's 2011 daily peak streamflow at Brandon was the highest on record at 1,280 m^3/s, a return period of 250 years (Province of Manitoba 2013). To reduce flooding on the downstream Assiniboine River, approximately 5.83 million cubic decametres of floodwaters were diverted into Lake Manitoba through the Portage Diversion, which accounted for 50% of the total inflow to Lake Manitoba that year (Province of Manitoba 2013). During both the previous reporting period and historical record (post-1977), there have not been any high streamflow events comparable to the 2011 and 2014 floods. As the Assiniboine River Basin was wetter than average during the current reporting period, the minimum discharge (13.9 m^3/s in January 2009) was higher than it was during 1999 to 2007 and 1977 to 1998 (4.7 m^3/s).

Winnipeg River Subwatershed

Over the current reporting period, the average discharge of the Winnipeg River was 1,084 m^3/s (Table 3-1). This discharge is comparable to the average over the 1999 to 2007 period (1,058 m^3/s); however, it is 23% higher than the historical average of 883 m^3/s. The Winnipeg River observed record flooding throughout several months in 2014. Streamflows exceeded the 90th percentile beginning in May and ending in September. Within this period, flows were record high for approximately two straight months during July and August. The maximum flow during the current reporting period was 2,531 m^3/s in July 2014, which is comparable to the maximum from the previous and historical reporting periods (2,538 m^3/s in 2002 and 2,519 m^3/s in 1992, respectively). Although record high streamflows occurred in 2014, overall, the previous reporting period observed more high flow months (> 1,576 m^3/s, the 90th percentile) than the current period.

In the Winnipeg River subwatershed, 2011 was the driest year within the current reporting period, although record (or near-record) flooding was taking place in the Assiniboine River, Red River, and Dauphin River subwatersheds at the same time. The minimum streamflow between 2008 and 2016 was 278 m^3/s in September of 2011. Overall, 2008 to 2016 had fewer low flow months (< 433 m^3/s, the 10th percentile) than 1999 to 2007. Variability in streamflow between 2008 and 2016 is similar to that of the historical period, while the 1999 to 2007 period showed increased variability.

Water Survey of Canada water level and flow monitoring station.

Dauphin River Subwatershed

Similar to the Assiniboine River Basin, the Dauphin River subwatershed experienced above normal streamflow and lake levels for the duration of the current reporting period. The average streamflow during 2008 to 2016 was 211 m^3/s (Table 3-1), which is 125% greater than the past reporting period and 225% greater than the historical period. During the current reporting period, the average streamflow remained above the 90th percentile (214 m^3/s) one third of the time, and dropped below the 50th percentile (74 m^3/s) only once, during February of 2010, further demonstrating these above-average conditions. Comparing the minimum discharge of the current reporting period (70 m^3/s) to the previous and historical reporting period values of 1.1 m^3/s and 0.7 m^3/s, respectively, demonstrates the unique high flow regime throughout 2008 to 2016.

Due to record flooding within the Assiniboine River Basin during the spring of 2011, unprecedented flows were sent along the Portage Diversion into Lake Manitoba, contributing to the highest lake levels on record for both Lake Manitoba and Lake St. Martin (both had 400-yr return periods; Province of Manitoba 2013). During this time, Lake Winnipegosis was also at a record high level (125-yr return period), as was the Waterhen River (conveys water from Lake Winnipegosis to Lake Manitoba), which also contributed to the high levels on Lake Manitoba (Province of Manitoba 2013). These conditions resulted in a maximum streamflow at the Dauphin River gauge of 579 m^3/s in July of 2011. To lower the levels on Lake Manitoba and Lake St. Martin, an emergency channel was constructed from Lake St. Martin running east to Big Buffalo Lake, eventually entering the Dauphin River just upstream of its confluence with Lake Winnipeg. The emergency channel was operated from November 1st, 2011 to November 21st, 2012, drawing down the water levels in Lake St. Martin much faster than under unregulated conditions (Province of Manitoba 2013).

Lake Winnipeg Subwatershed

Approximately 80% of the Lake Winnipeg subwatershed is ungauged (Figure 3-1), including the Pigeon, Berens and Poplar rivers, which likely provide significant local inflows to Lake Winnipeg. For this report, the tributaries used to determine the gauged local inflow to Lake Winnipeg include the Bloodvein, Manigotagan, Icelandic, Fisher and Brokenhead rivers. Of these tributaries, the Bloodvein River is the largest contributor, accounting for over 80% of the gauged monthly mean inflows, on average. It should be noted that there are missing data between 1997 and 2010 at the Manigotagan gauge, and only spring data are available at the Icelandic and Fisher River gauges between 2000 and 2010.

Based on the available data, the average gauged local inflow to Lake Winnipeg was 94 m^3/s during the current reporting period, which is 32% higher than the previous reporting period and 59% higher than historical observations (Table 3-1), similarly indicating a wetter period in the Lake Winnipeg subwatershed. Four of the top five highest monthly streamflows since 1977 occurred during the current reporting period in 2009 and 2010, with the maximum value in July of 2010 (397 m^3/s). The period between fall 2011 and spring 2012 was the driest during the current reporting period, with a minimum discharge of 7.2 m^3/s in February of 2012. However, this period of lower flows was not as severe as the streamflow conditions experienced in 1999 during the previous reporting period, and in 1977 and 1988 within the historical period.

Inflow Summary

Between 2008 and 2016, the Winnipeg River accounted for 43% of the inflow to Lake Winnipeg on average, followed by the Saskatchewan River (29%), Red River (12%), Dauphin River (8%), Assiniboine River (4%), and local tributaries which contributed approximately 4% of the total inflow (Figure 3-2). Interestingly, during the 2008 to 2016 period, average contributions from the Winnipeg River subwatershed decreased by 7%, from 50 to 43%, since the previous reporting period (8% from the historical average). Dauphin River contributions doubled from 4% to 8% and Saskatchewan River contributions increased 3% from 26 to 29%, while all other subwatershed contributions were within 1– 2% of the previous reporting period and historical observations. The decrease in contributions from the Winnipeg River is not due to decreased streamflow during the current reporting period (the average discharge during 2008 to 2016 is almost identical to that during 1999 to 2007), but rather is attributed to increased streamflow observed in the Saskatchewan, Red, Assiniboine, Dauphin and Lake Winnipeg subwatersheds.

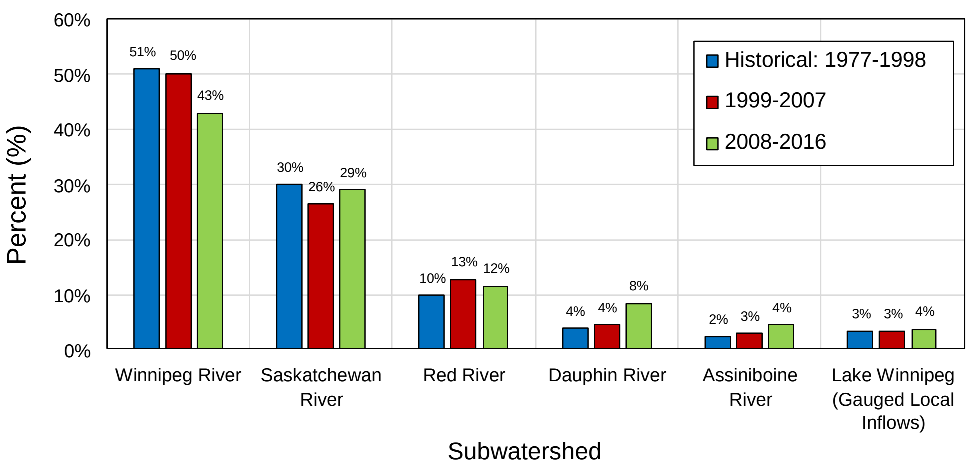


Figure 3-2: Average gauged inflow contributions (%) to Lake Winnipeg for the historical period of 1977 to 1998 (blue), the previous reporting period of 1999 to 2007 (red) and the current reporting period of 2008 to 2016 (green).

The current reporting period has an average streamflow (2,515 m^3/s) that is considerably higher than in past time periods (Figure 3-3). This flow is almost 20% higher than the average of the pervious reporting period (2,108 m^3/s), and 44% greater than the historical average (1,749 m^3/s). Two of the five highest monthly discharges since 1977 occurred within the 2008 to 2016 period, including the 2011 (ranked 3rd) and 2014 (ranked 4th) flood events. However, when considering the average annual inflow, 2011 is the highest on record, followed by 2014, 2005, 2009 and then 1997.

Due to the wet cycle observed during the current reporting period, there were very few low-flow years as far as Lake Winnipeg inflows are concerned. Both 2015 and 2016 had monthly peak streamflows that were relatively low (less than 3,000 m^3/s), although the minimum streamflow of the current reporting period occurred in the fall of 2012 (1,404 m^3/s). However, due to elevated baseflow conditions, this minimum streamflow is much higher than other low flow years such as 2003 (653 m^3/s) in the previous reporting period or 1987 to 1988 (531 m^3/s) in the historical period of record.

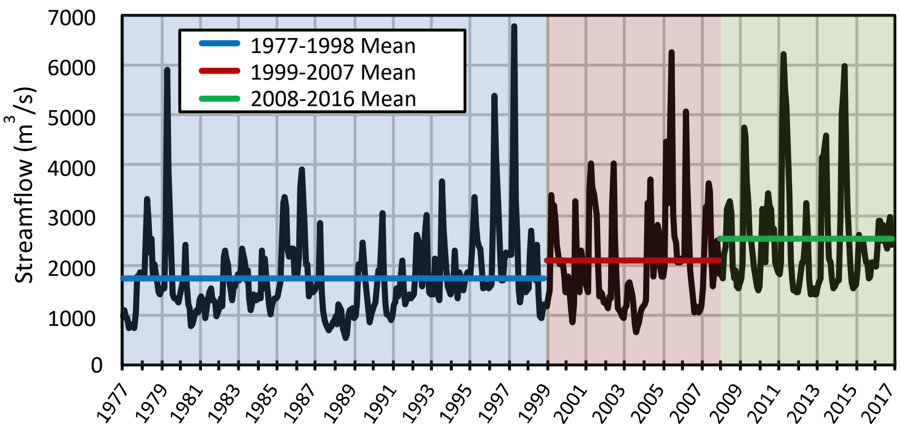


Figure 3-3: Monthly mean gauged inflow to Lake Winnipeg in cubic meters per second (m3/s). Average streamflow over each of the pre-defined time periods is indicated by the blue (1977–1998), red (1999–2007) and green (2008–2016) lines.

Outflow Discharges

Similar to the inflows to Lake Winnipeg, the average outflow has consistently increased over time. During the historical period, the average outflow was 1,920 m^3/s, then increased to 2,372m^3/s during the previous reporting period, and reached 2,931 m^3/s for the current reporting period (Table 3-1).

While the 2011 flood had a marked impact on the increased outflows during the current reporting period, it is not the only event contributing to this increase. In 2009, 2010, 2013 and 2014, the monthly mean outflows also exceeded 4,000 m^3/s. The outflow has been above the 4,000 m^3/s mark in previous periods, but much less frequently, and for shorter durations (e.g. 2005, 1997 and 1979). Similar to Lake Winnipeg inflows, the low outflow years in the current reporting period are much higher than in the previous or historical periods. The minimum outflow between 2008 and 2016 was 1,556 m^3/s, which is just over twice as large as the minimum outflows during 1999 to 2007 (765 m^3/s) and 1977 to 1998 (763 m^3/s).

Manitoba Hydro began regulating Lake Winnipeg outflows in 1976 under an Interim Licence issued under the Manitoba Water Power Act and Regulations. This regulation has had a significant influence on outflow volumes and patterns from the lake. Manitoba Hydro has the authority to adjust outflows when the mean, wind eliminated water levels on the Lake fall within the Operating Range of their licence. The Operating Range is defined in the licence as falling within the range of 711 to 715 feet above sea level. Lake Winnipeg operating decisions are typically made on a seasonal basis and consider current water supply conditions in addition to other factors such as inflow forecasts and Manitoba's electrical demand (Manitoba Hydro, 2014). From time to time, lake levels rise above 715 feet and when this occurs Manitoba Hydro is obligated by their licence to increase to maximum discharge until the lake levels return to 715 feet or less. The purpose of the project is two fold: 1) to provide Manitoba Hydro with a secure supply of water for their generating stations located on the Lower Nelson River, and, 2) to provide a degree of flood protection to the communities located around the shores of the Lake.

Lake Water Levels

Water levels on Lake Winnipeg can vary significantly at various locations across the lake due to weather conditions such as wind and barometric pressure. In order to determine the average lake level (termed the smoothed *wind-eliminated level)*, which is used for regulating the lake, Manitoba Hydro uses daily average readings from eight Water Survey of Canada water level gauges (*see Appendix*). The current reporting period had an average lake level of 217.77 m, which is 0.25 m higher than the average water level of the previous reporting period, and 0.32 m higher than the post-1977 historical average. Similar to Lake Winnipeg outflows, the wind-eliminated water level of Lake Winnipeg illustrates a strong response to regulation. The wind-eliminated water levels and lake outflows demonstrate similar seasonal fluctuations, particularly during high water level years such as 2005, 2009 to 2011 and 2014, when Manitoba Hydro maximizes Lake Winnipeg discharge. During the current reporting period, the water level peaked in July 2011 at 218.51 m (Figure 3-4).

The minimum lake level during the current reporting period occurred in early 2013 (217.34 m). In the previous reporting

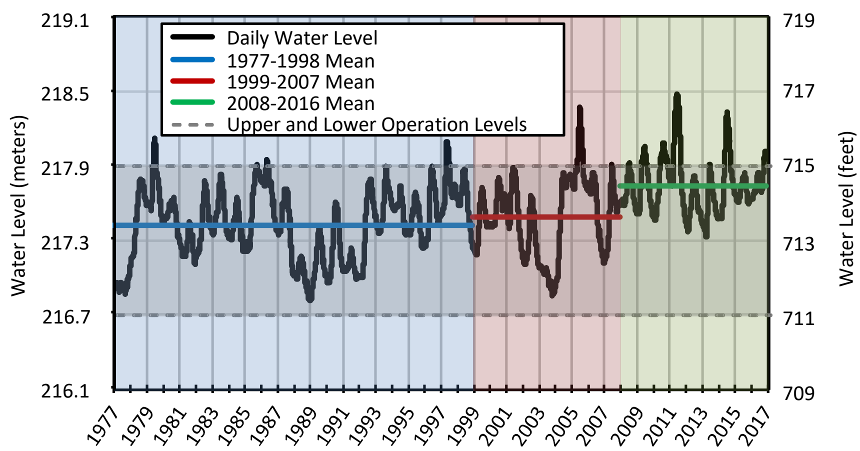


Figure 3-4: Daily average wind-eliminated water level of Lake Winnipeg shown in both meters (left) and feet (right) above sea level. Average water levels over each of the pre-defined time periods are indicated by the blue (1977–1998), red (1999–2007), and green (2008–2016) lines. The upper and lower operation levels are denoted by grey lines and shading.

period, lake levels dropped below this value on numerous occasions, most notably during 2002 to 2003 (216.87 m) and 2006 to 2007 (217.14 m). Historically, 1976 to 1977 (216.88 m) and 1988 to 1992 were also periods of low water levels, whereby the lowest level since 1977 occurred at the beginning of 1989 measuring 216.83 m. This value is over 0.5 m less than the lowest level from the current reporting period.

Lake Water Residence Time

Residence time is a measurement of how long water resides within a lake. This influences the storage and release of nutrients and contaminants. Residence time for Lake Winnipeg was determined using the outflow from Jenpeg Generating Station and the combined morphology of the lake (collected by Brunskill et al. 1980) using:

Residence Time = Lake Volume / Outflow Discharge (Brunskill et al. 1980)

Lake Winnipeg's residence time has varied over the last 50 years from 2.5 to almost eight years in length. The years with the longest residence time, 1977 (7.2 years), 1988 (7.9 years) and 2003 (6.8 years) correspond to periods of drought. The shortest residence times (< 3 years) in 1969, 2005, 2009, 2011, and 2014, correspond to high water flood years.

The overall average residence time from 1967 to 2016 is 4.2 years. The period of 1967 to 1998 had an average residence time of 4.5 years, while the residence time from 1999 to 2007 was 4.1 years, which further decreased to 3.1 years during the current reporting period of 2008 to 2016. The decrease in the residence time of the lake is due to the observed increase in streamflow.

4.0 PHYSICAL CHARACTERISTICS

By: Gregory K. McCullough (University of Manitoba)

The thermal regime and spatio-temporal distributions of suspended sediments and oxygen were described for Lake Winnipeg through the period 1999–2007 in the first State of Lake Winnipeg report (EC and MWS 2011). The following sections provide an update on the 2008–2016 period and describe patterns in temperature, dissolved oxygen, total suspended solids, turbidity and transparency (Secchi depth) using data collected by Environment and Climate Change Canada (ECCC) at surface buoys in the north and south basins and the narrows, and by Manitoba Agriculture and Resource Development (MARD) from whole-lake surveys. More information on sampling locations and frequency, as well as instrumentation is provided in the Appendix.

Wave action on the shores of Lake Winnipeg.

Thermal Regime

The reader may refer to the first State of Lake Winnipeg report (EC and MWS 2011) for a brief, general discussion of processes that determine the distribution of heat in lakes. The thermal regime of a lake has implications for the structure and function of its aquatic ecosystem. Many species have limited tolerance for either high or low temperatures. The lengths of the ice-covered and open water seasons, as well as the degree to which waters are warmed, affect the energy available for the metabolism of all biota in the lake (Wetzel 2001). The thermal regime of Lake Winnipeg develops differently in its two main basins in response to differences in bathymetry, morphometry and climate (see Section 5.0 in EC and MWS 2011). The south basin is shallower than the north basin. On average, compared to the north end of the lake, the air temperature near the south end is about 4.0°C warmer in May and November and about 2.0°C warmer in mid-summer. This, and the smaller depth and thermal mass of the south basin, contributes to differences in ice cover period, water temperature, and vertical thermal stratification between the two basins. Ice melts and clears off the south basin a week or two earlier than off the north. The south basin warms quickly and is shallow enough that winds normally keep the water column well mixed from early spring through fall. Any thermal gradients that develop in hot, calm periods tend to be short-lived. The much larger and deeper north basin warms more slowly, and heat is mixed down through the water column only gradually. Although the north basin is isothermal for at least brief intervals early in the open water season, and more persistently from late summer through fall, in some years it develops a deep thermocline which persists for weeks through the mid- to late summer.

Profundal surface water temperature record

Water temperature in Lake Winnipeg has been monitored at long-term profundal stations by MARD (*see Appendix*) in spring (May–June), summer (July–August), and fall (September–October) since 1999. Survey dates varied between years, which reduces the value of this dataset for inter-annual comparability or long-term trends in mean seasonal water temperatures. In spring or fall, a week's difference in the survey dates may account for a greater difference in temperatures recorded than would real inter-annual variability. Consequently, only mid-summer survey data are discussed below. Except where otherwise noted, the temperature data discussed below is from observations made in the upper metre of the water column.

Surface buoy with monitoring equipment.

Thermal data from whole lake surveys is supplemented by records at surface buoys in the north and south basin and the narrows, where air and surface water temperature (and wind and waves) have been recorded since the early 1990s. Early weather buoy records have proven difficult to decipher, and all suffer from discontinuities (some minor, some affecting almost entire seasons) due to mechanical failures. However, for most years since 1999 there remain many months of continuous data. Here, the weather buoy data is used to describe seasonal and long-term trends in Lake Winnipeg surface water temperature. Records from the MARD surveys are used primarily with reference to the thermal structure of the water column.

From 1999 to 2016, daily mean surface water temperatures in the north and south basins fell within a range of roughly 7°C (Figure 4-1). The south basin warmed earlier than the north; by mid-June, it was 12–20°C while surface water at the north buoy was roughly 6°C cooler, from 6°C to 14°C. Summer temperatures peaked at 18–24°C in the south compared to 15–20°C in the north. The between-basin difference generally narrowed in late summer (dropping to 16–21°C in the south compared to 14–17°C in the north by 1st September). In 2013, for example, even though there was an unusual warming in the south basin late in the summer, the peak was short-lived (and may only have occurred in the near-surface) so that by mid-September, it was only 1°C warmer than in the north. In fact, though the south basin warms sooner, and remains warmer through to midsummer, it also cools faster than the north basin so that the two basins often freeze within a few days of each other (see Figures 5.6 and 5.7 in EC and MWS 2011). On the other hand, ice melt and breakup occur, on average, about two weeks earlier in the south basin, where surface water temperatures can reach 5°C or higher before ice has cleared off the north basin.

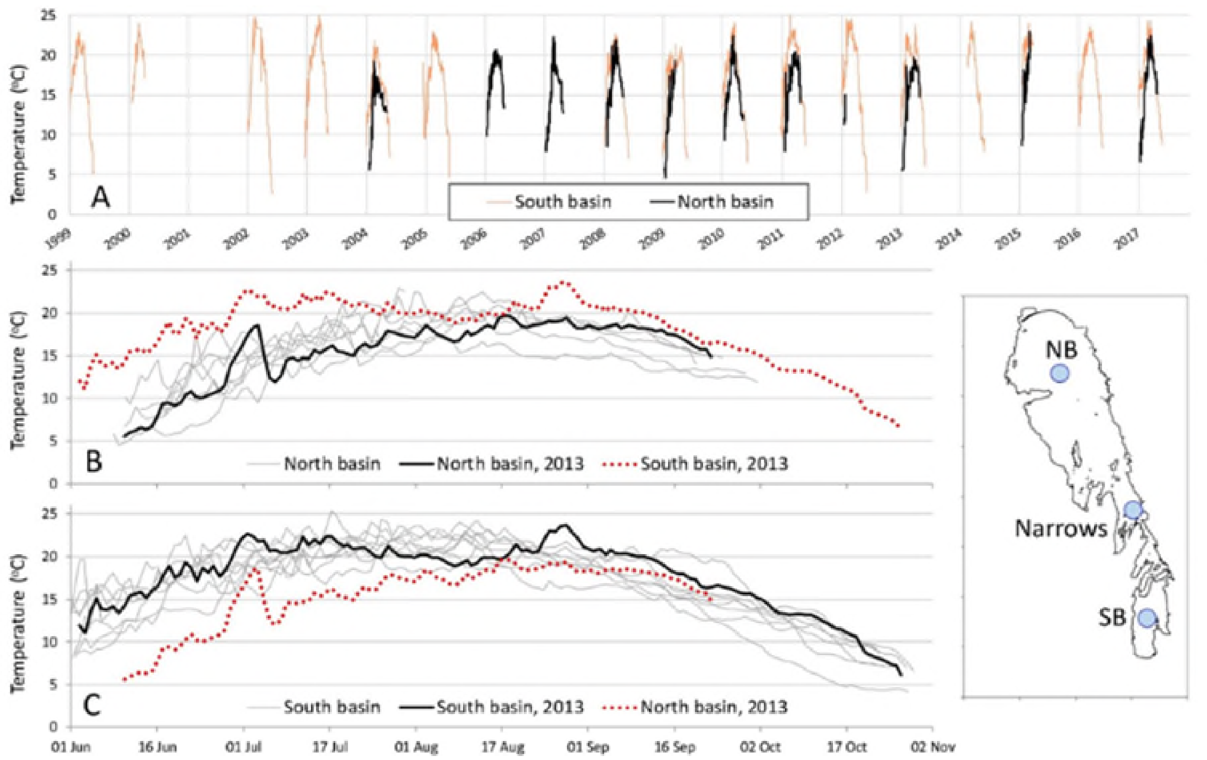


Figure 4-1: A) Daily mean surface water temperature recorded at buoys in the north and south basins, from 1999 to 2017 (excluding some records with brief time series, and some anomalously high values in early spring). Note that the vertical grid identifies the 1st of June of each year. B, C) Seasonal time series (including only years when fairly continuous were recorded in both basins; i.e. 2004, 2008–2011, 2013, 2015 and 2017). The 2013 records are highlighted in both panels to illustrate typical between-basin differences in the seasonal pattern. Location of buoys shown at right. Data supplied by Environment and Climate Change Canada.

In the years for which there were no significant data gaps, July–August monthly mean surface water temperatures ranged from 14.9 to 19.5°C in the north basin, and from 18.4 to 23.2°C in the south basin (Table 4-1, Figure 4-2). When the differences in individual years are averaged, the north basin was 4.5°C cooler in July, but only 2.2°C cooler in August. In the south basin, the September mean temperature ranged from 14.9 to 19.5°. There are insufficient complete September records for the north basin to generate meaningful statistics; however, by inference from Figure 4-1 the mean temperature is similar in a typical September. Monthly mean temperatures in the narrows were on average 0.5°C lower than at the south basin buoy, but the difference was not significant (s.d. = 0.7°C, n = 30). In the south basin, through the period 2008–2016, the August and September mean surface water temperature was on average 0.5°C and 0.6°C higher (respectively) than through the 1997–2007 period reported earlier (EC and MWS 2011). It was 0.1°C lower in July. None of these differences is significant. For the north basin, there are too few complete monthly mean records in the 1999–2000 period to make a meaningful comparison. On the other hand, inter-annual variability in the record was considerable; standard deviations of July and August monthly means from 1999 to 2016 range from 1.1 to 1.5°C (Table 4-1). Even between successive years, there were some large differences; for instance, at the south basin and narrows buoys, the mean August temperature was 4.7°C lower in 2004 than in 2003.

Table 4-1: July, August and September surface water temperatures recorded at Environment and Climate Change Canada buoys in the north and south basins and narrows of Lake Winnipeg, for months with less than three days of missing data, 2008–2016. n.d. indicates insufficient continuous data to provide meaningful comparable monthly statistics. Units are °C.

	mean	s.d.	min	max	n
July					
North basin	16.4	1.2	14.9	17.9	6
Narrows	20.9	1.6	17.9	23	7
South basin	20.9	1.4	18.4	23.2	9
August					
North basin	18.9	0.5	18.3	19.5	4
Narrows	20.6	1	18.7	21.3	7
South basin	21.1	1	18.9	22	8
September					
North basin	n.d.	n.d.	n.d.	n.d.	2
Narrows	16.6	1.8	14.5	19.3	6
South basin	17.1	1.5	14.9	19.5	8

Such large inter-annual variability would mask any likely long-term trend in such a short record, but in fact, no significant trends have been shown even in longer datasets (see Section 5.0 in EC and MWS 2011). Nonetheless, given that regional air temperature records are 1–2°C higher than in the late 19th–early 20th century, and that there is a strong linear relationship between air and surface water temperature in Lake Winnipeg (Figures 5.4 and 5.5 in EC and MWS 2011), it is expected that Lake Winnipeg waters have warmed and will be warmer yet in response to general regional warming predicted through the 21st century (*see Section 2.0)*.

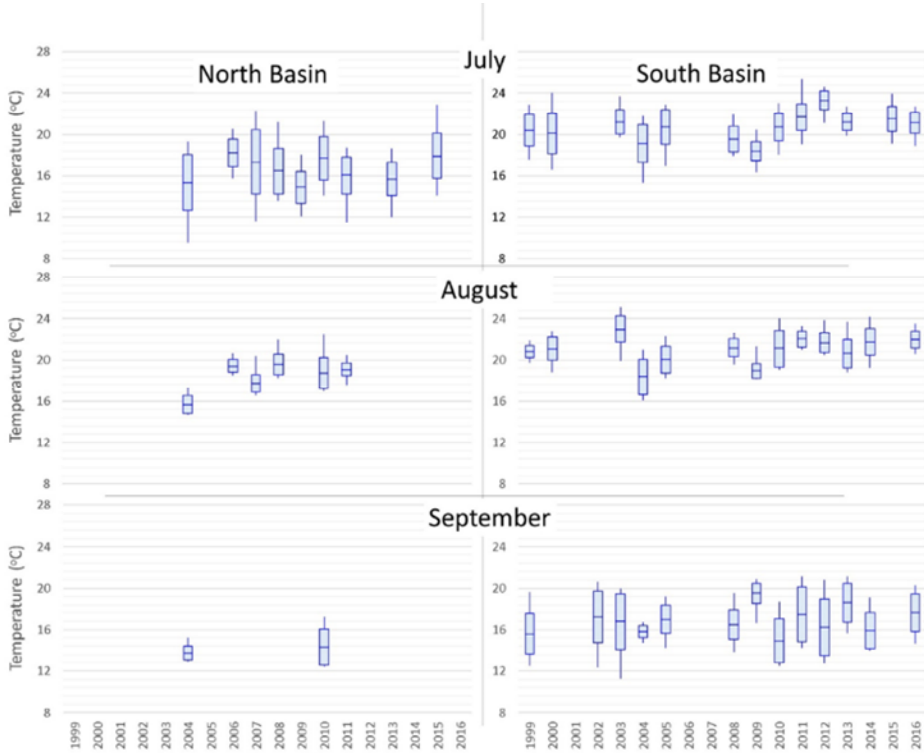


Figure 4-2: July–September surface water temperatures recorded at buoys in Lake Winnipeg for all months with less than five days missing data. Boxes show mean +/- 1 standard deviation (calculated from daily means) for each group; whiskers indicate the total range. Note lower temperature range for September data. Buoy locations are shown on the map in Figure 4-1. Data supplied by Environment and Climate Change Canada.

Nearshore stations

Littoral zone monitoring at five stations in the north basin and seven in the south basin and narrows was initiated in 2013. Each station is sampled along a transect perpendicular to shore, at stations with bottom depths of 1, 2 and 3 m. Table 4-2 reports selected data from mid-summer surveys at these nearshore stations.

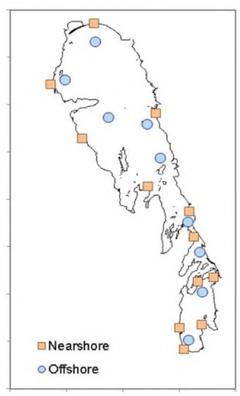
Lake Winnipeg shoreline, Gimli, Manitoba.

In the south basin and narrows, the surface temperature was essentially the same in the nearshore zone as at nearby profundal long term monitoring stations. This was true for turbidity as well. For the southernmost stations, this may be because the nearest profundal station lies in the plume of the Red

River. Indeed, the specific conductivity was lower at the profundal station (though the difference is not statistically significant), which may be explained by the higher charged ion content of Red River water; specific conductivity is consistently higher near the mouth of the Red River than anywhere else in Lake Winnipeg (Brunskill et al. 1979a) In the north basin, surface water temperature tended to be higher near shore (by 1.7^{o}C, s.d. = 1.1) as might be expected given the great difference between profundal and littoral depths. Even here, though, turbidity was not consistently higher nearshore, in spite of the shallow sediments available for resuspension in the frequently energetic environment. However, the nearshore environment is difficult to sample in rough weather, so that the observations should not be considered random, at least with respect to wave energetics.

*Table 4 2: Means (and standard deviations) of surface observations at 12 nearshore stations observed in summer surveys in 2015 and 2016, compared with mean conditions at paired offshore stations.*All values are means of instrument records from 0.5 to 1.5 m depth at three substations (1, 2 and 3 m bottom depth) per nearshore station. Difference = value at nearshore station minus value at the nearest offshore station, if the two were sampled on the same or consecutive days. Locations are shown in the map at the right. Data supplied by Environment and Climate Change Canada and Manitoba Agriculture and Resource Development.*

	Mean (s.d.) at nearshore stations	Difference (s.d.) from offshore stations
North basin		
Temp. (^{o}C)	20.6 (1.2)	1.7 (1.1)
DO (mg/l)	8.6 (0.6)	-0.63 (0.93)
DO (%)	95 (8)	-3.6 (10.8)
Cond. (µS/cm at 25^{o}C)	390 (59)	-9.2 (28.1)
Turb. (NTU)	25 (29)	16 (21)
South basin and narrows		
Temp. (^{o}C)	21.3 (1.1)	0 (0.5)
DO (mg/l)	8.5 (0.7)	0.73 (0.63)
DO (%)	96 (8)	8.2 (6.7)
Cond. (µS/cm at 25^{o}C)	378 (159)	-38.1 (144.3)
Turb. (NTU)	21 (7)	-4 (14)



** For the nearshore stations, n = 160 (144 for turbidity)–i.e. about 7 observations along each transect in each year. For the nearshore–offshore pairs, n = 7 and 8 in the north and south basin-narrows respectively over the two years. About 11% of nearshore turbidity values ranging from 407 to 409 NTU were treated as outliers and excluded from statistical calculations. The highest non-excluded turbidity was 241 NTU.*

Thermal stratification

Figure 4-3 shows spring and summer warming, and early fall cooling of the water columns in the north and south basin, recorded at ECCC moorings through the summer of 2008. The moorings consisted of chains of thermistors at 1 to 2 m intervals, from 2 to 16.5 m depth in the north basin (nine thermistors) and from 3 to 11 m depth in the south basin (seven thermistors). The water column in the south basin was nearly isothermal at 12°C by mid-June but developed a shallow thermocline a few days later when the surface warmed rapidly to about 22°C, while below 5 m the temperature remained unchanged until 28 June. On that day, the surface quickly cooled several degrees as the full water column became mixed by turbulence generated by strong winds (as recorded at the ECCC south basin weather buoy). The water column remained nearly isothermal through most of July. A weak gradient developed late in July but did not persist; the basin was isothermal for the rest of the summer, through to the end of September.

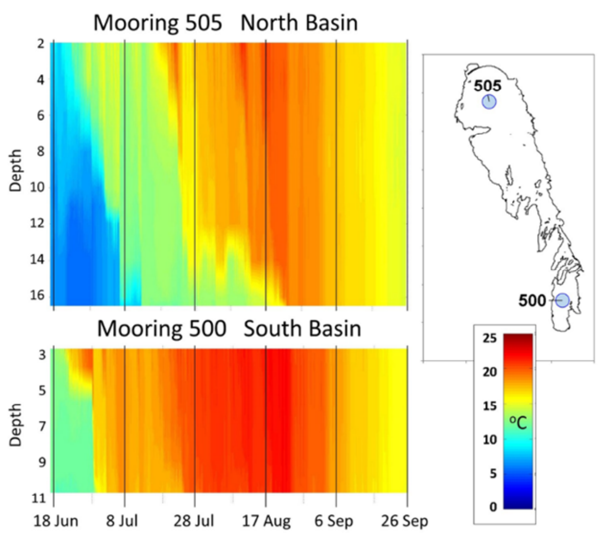


Figure 4-3: Temperature records from thermistor logger chains installed in 2008 in the north and south basins of Lake Winnipeg. Colour temperature scale (°C) at lower right. Charts produced by R. Yerubandi, Environment and Climate Change Canada.

In the north basin, the surface warmed only gradually until the third week of July, when the surface, down to at least 4 m, warmed briefly to about the same 22°C as the south basin at that time. Warming near the bottom lagged the surface by at least three weeks, with < 10°C water persisting until the second week of July, and < 15°C water persisting until mid-August. A strong thermocline developed at about 14 m depth, and persisted until about 23 August, when the entire water column was cooled, again by strong winds (as recorded at the ECCC north basin weather buoy). In this case, the thermocline isolated a < 3 m thick layer of deep water from more oxygenated near-surface waters for at least three weeks. The basin remained continuously isothermal throughout the rest of the mooring record, from late August until near the end of September.

Environment and Climate Change Canada maintained similar moorings in both basins through several open water seasons. The 2008 record is fairly typical of these moorings records. The south basin tends to warm to over 20°C and becomes isothermal between late June and mid-July; rarely does it develop more than a weak, ephemeral thermal gradient after that, at least within the few years of continuous records. Warming in the north basin lags the south, especially below the surface. In every year of moorings records in the north basin, a thermocline was identifiable at about the 14–15 m depth, even if sometimes weak, for as little as a week to more than a month, usually in early summer.

Figure 4-4 shows thermal profiles from summer surveys from 2013–2016. No well-developed thermoclines were recorded in the south basin (profiles to less than 12 m depth) in any year, although there is one record of a small, but sharp thermal gradient at about 7 m depth (B in Figure 4-4) and this is associated with a small, but distinct drop in dissolved oxygen, suggesting that it had persisted for at least some days. There was substantial thermal stratification in deep water in each of these years. It varied from particularly sharp—with a strong thermocline at 13–16 m depth in 2013, in particular—to weak, as in 2014 when overall stratification was split between gradients in the upper water column and again at the 14–15 m depth. The hypolimnetic water under some of these thermoclines was as much as 6 to 8°C lower than the overlying, often nearly isothermal epilimnion, from which we can infer that the water column had not mixed completely since much earlier in the open water season, when the surface had not yet warmed above 10 to 15°C.

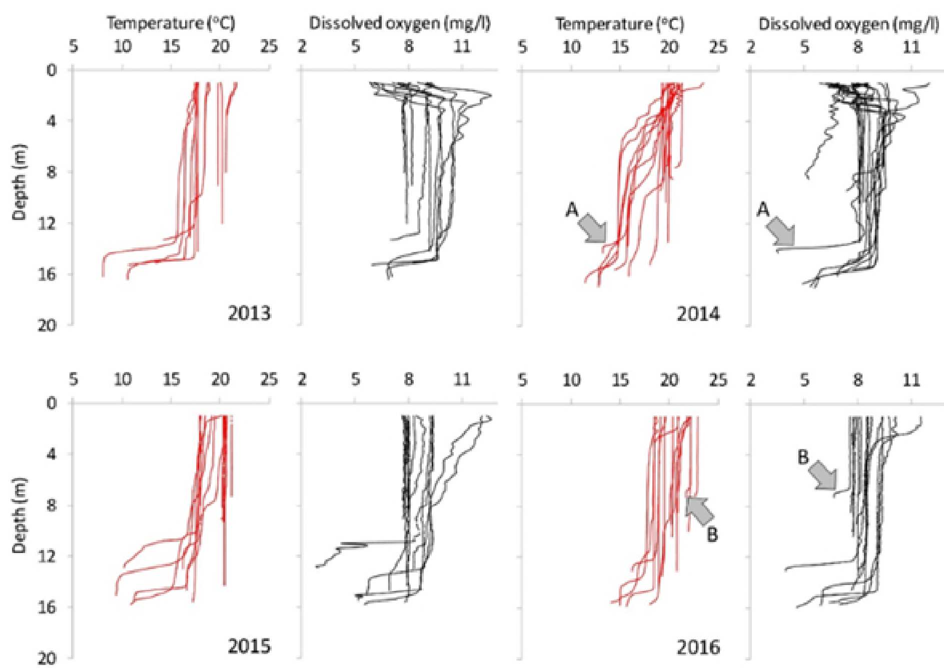


Figure 4-4: Temperature and dissolved oxygen profiles at long-term monitoring stations. Arrows indicate instances of oxygen depletion in very thin hypolimnia that had formed at (A) Station W2 in the north basin on 3 August 2014 and at (B) W12 on 30 July 2016. Data supplied by Environment and Climate Change Canada.

There are increasingly frequent examples of vertical thermal stratification in water column profiles recorded in mid-summer surveys of the lake, mostly in deep water in the north basin. In 80% of the shallower profiles recorded on mid-summer surveys over the last two decades, there was less than 1^{o}C difference between surface and bottom temperature, and this did not change between the periods 1999–2007 and 2008–2016 (Figure 4-5). These shallower profiles were mostly in the south basin, and this statistic reflects the generally well-mixed nature of that basin. However, it bears noting that even here, the temperature near the bottom was sometimes (specifically, in 5% of recorded summer profiles) as much as 2 to 4^{o}C lower than near the surface. In deep water (> 10 m, mostly in the north basin) from 1999–2007, the bottom was < 1^{o}C lower than the surface in 64% of recorded profiles. However, from 2008–2016, this reasonably well-mixed character was observed in only 54% of summer profiles in deep water, while the frequency with bottom water 2 to 4^{o}C lower than surface water, doubled. Development of thermal gradients through the summer period became a more common occurrence.

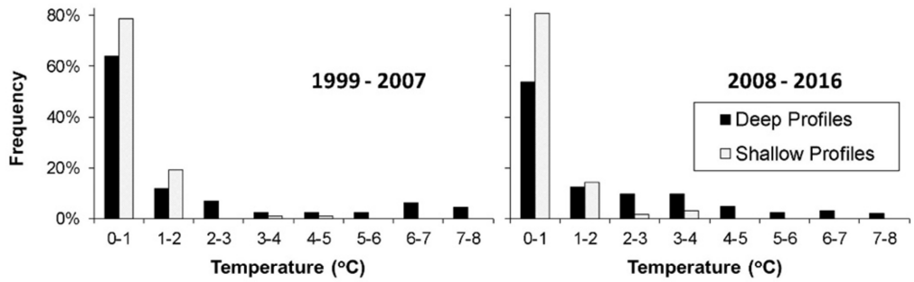


Figure 4-5: Differences between surface (0.5–1.5 m) and bottom (about 1–2 m above the bottom) temperature in deep (bottom depth >10 m) and shallow (5–10 m) water column profiles recorded during mid-summer whole-lake surveys of Lake Winnipeg. Deep profiles, n = 177 and 268; shallow profiles, n = 94 and 162, for 1999–2007 and 2008–2016, respectively. 1999–2006 data supplied by Fisheries and Oceans Canada; 2007–2016 data supplied by Environment and Climate Change Canada.

A similar increase is evident in the frequency of thermoclines identifiable in water column profiles. From 1999 to 2007, thermoclines were identified in less than a third (31%) of all mid-summer profiles recorded in deep water [> 10 m deep, with a thermocline defined as a thermal gradient of at least 1^{o}C per meter depth (per Wetzel 2001)]. In the 2008–2016 period, the frequency of such thermoclines was almost double that (58%). Previously Lake Winnipeg has been described as a cold polymictic lake (i.e. well-mixed, referring to both basins and the narrows) with at most only brief periods of stratification (EC and MWS 2011, Wassenaar 2012). If indeed the north basin was continuously polymictic in the past, it may be in the process of becoming discontinuously so, with a deep thermocline preventing full vertical circulation for several weeks in the early summer of most years. The cause of these increases in large temperature ranges in mid-summer profiles, and the related increase in frequency of observed thermoclines has not been explored. Development and persistence of deep thermoclines must be related to interaction of several factors, including the rate of surface warming in spring, the mid-summer peak temperature, and water column transparency and frequency of strong winds in both periods. The relative effect of these and possibly other forcing variables must be better understood if we are to judge whether we are witnessing interannual or decadal variability, or some long-term trend ultimately related to climate change.

Dissolved oxygen

Dissolved oxygen enters and leaves the lake at the water surface according to temperature-driven solubility. Concentrations are increased when photosynthesis occurs in the euphotic zone and are reduced when it is converted to carbon dioxide during plant respiration and organic decay. In a eutrophic lake, the latter is largely respiration and decomposition of phytoplankton, which can occur at any depth. In the upper, euphotic zone, or anywhere in a vertically well-mixed lake, respiration and decomposition rarely reduces oxygen to levels dangerous to aquatic life before it is replenished. However, since much organic detritus settles to the bottom and decomposes, there is a potential for oxygen depletion whenever vertical mixing is prevented for any length of time, as by a persistent thermocline.

In Lake Winnipeg, dissolved oxygen concentrations were generally higher in the north basin than in the south in spring and summer. In autumn, they were about the same in each basin, but in winter tended to be higher in the south (Figure 4-6). They were highest in winter (in near surface water but not at the bottom) and lowest in summer. On average, the water column in both the north and south basins was well-mixed with respect to oxygen in both spring and fall. In summer it was lower in bottom compared to surface water. The average difference was nearly 20% in the south basin, and 6–12% in the north. It was generally lower in bottom water in winter as well, except in the south basin in the earlier period (1999–2007) where it was on average about the same in surface and bottom samples.

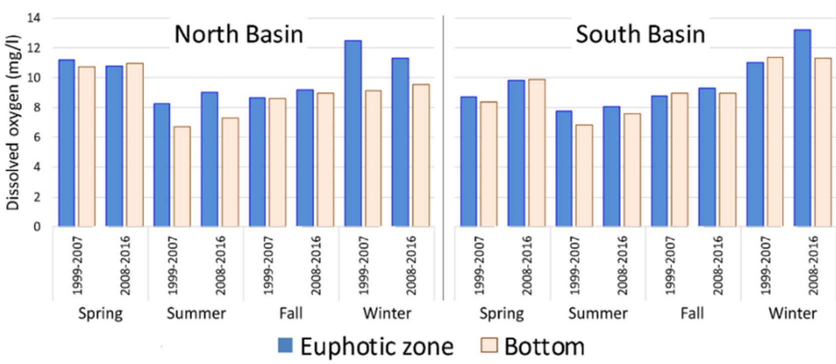


Figure 4-6: Euphotic zone and bottom seasonal mean dissolved oxygen concentration in Lake Winnipeg for the periods 1999–2007 and 2008–2016. Chart created from data in Table 4-3.

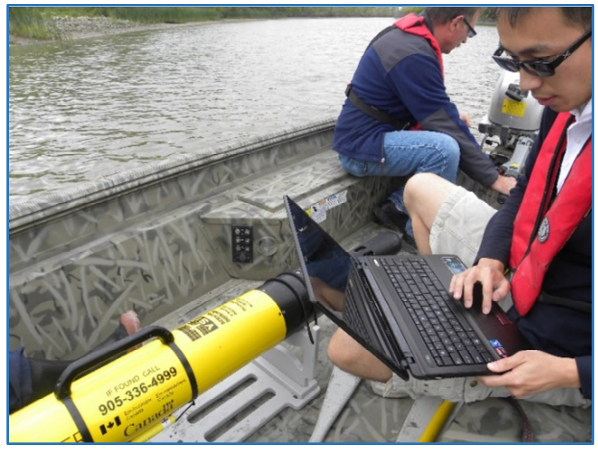
Overall, from 1999 to 2016, observed concentrations of dissolved oxygen ranged from 1.7 to 15.5 mg/L in the north basin of the lake, and from 0.1 to 18.7 mg/L in the south basin (Table 4-3). However, concentrations that would be considered hypoxic remain uncommon. Manitoba's water quality objective for protection of aquatic life for dissolved oxygen is temperature dependent — at temperatures above 5°C, the lower limit for dissolved oxygen is 5 mg/L, whereas at temperatures of 5°C or below, the limit is 3 mg/L (MWS 2011). Similar to the 1999–2007 period, only 2.4% of over 900 observations from 2008–2016 were below 5.0 mg/L; because temperature is not considered in this statistic, it is likely that the number of samples that did

Table 4-3: Euphotic zone and bottom water temperature and dissolved oxygen concentration in Lake Winnipeg for the periods 1999–2007 and 2008–2016. The dataset has been filtered to include only stations for which both temperature and oxygen were reported for both the euphotic zone and bottom water. Data provided by Manitoba Agriculture and Resource Development.

			Temperature (°C)					Dissolved oxygen (mg/l)				
			mean	s.d.	min.	max	n	mean	s.d.	min.	max	n
North Basin												
spring	1999-2007	euph.	10.5	2.8	4	15	28	11.2	1.4	7.1	13.6	28
		bot.	9.1	2	4	12.8	28	10.7	1.5	7.2	12.9	28
	2008-2016	euph.	9.6	3.1	3.2	16.1	55	10.8	1.9	5.4	13.5	56
		bot.	7.7	2.6	3.1	14.1	56	11	1.5	6.2	13.8	56
summer	1999-2007	euph.	19.8	1.9	15.6	23.8	32	8.3	0.8	5.9	9.7	32
		bot.	17.9	2.1	13	22.5	34	6.7	1.7	2.4	8.7	34
	2008-2016	euph.	19.4	1.9	12.6	22.4	63	9	1.1	6.4	14.4	63
		bot.	15.7	3	9.1	21.5	63	7.3	1.5	3.2	10	63
fall	1999-2007	euph.	15.2	2.4	11	19.5	34	8.7	1.4	3.9	10.3	34
		bot.	15.1	2.6	11.3	20	33	8.6	0.8	5.2	9.6	33
	2008-2016	euph.	15.7	1.9	11.7	20.4	61	9.2	0.9	3.2	10.5	61
		bot.	15.3	1.7	11.4	19	62	9	0.6	6.9	10.1	62
winter	1999-2007	euph.	0	0.1	0	0.2	21	12.5	2	5.6	14.3	21
		bot.	1	1.1	0	2.8	21	9.1	3	1.7	12.6	21
	2008-2016	euph.	0.1	0.4	0	3	46	11.3	3.5	4.2	15.5	46
		bot.	1.5	0.7	0.1	3	44	9.5	3.3	3.1	15.2	44
South Basin												
spring	1999-2007	euph.	15.8	2.4	12.6	20.2	13	8.7	2.9	2.2	12.2	13
		bot.	14.9	2.5	11.5	20.1	14	8.4	1.2	5.7	10	14
	2008-2016	euph.	13.8	2.3	8.9	19.8	36	9.8	1.8	2.2	13.2	36
		bot.	11.5	2.9	4.8	16.3	35	9.9	1.4	7	12.8	35
summer	1999-2007	euph.	22.1	1.9	18	24	17	7.7	0.8	6.1	10.1	17
		bot.	21.6	1.7	18	24	17	6.8	1.4	3.4	8.2	17
	2008-2016	euph.	21.4	1.6	17.9	25.3	36	8.1	0.5	7.1	9	36
		bot.	20.6	1.5	17.1	22.9	36	7.6	0.8	5.2	8.8	36
fall	1999-2007	euph.	14	1.6	10	16	16	8.8	1.1	4.9	9.9	16
		bot.	14	1.6	10	16	16	8.9	0.5	8.2	10.3	16
	2008-2016	euph.	15.4	1.8	13.3	20.3	29	9.3	0.4	8.5	10.1	29
		bot.	14.8	1.7	12.1	19.5	30	9	0.5	7	9.8	30
winter	1999-2007	euph.	0	0	0	0.1	6	11	5.6	0.1	14.9	6
		bot.	1.6	1.3	0	3.2	11	11.4	2.4	6.5	16.7	11
	2008-2016	euph.	0.1	0.1	0	0.3	28	13.2	3.5	5.2	18.7	28
		bot.	1.5	0.9	0.1	3.2	28	11.3	2.9	4.7	14.7	28

not meet the objective is even lower. Twenty occurrences of dissolved oxygen at less than 5 mg/L were in the north basin and only three in the south. Just under two-thirds were recorded in bottom samples, in either summer or winter. Of the remaining third near the surface, most occurred under ice in winter.

Wassenaar (2012) reported that Lake Winnipeg is on average under-saturated with respect to oxygen despite its high productivity. This was the case through the period 2008–2016, in all seasons (Figure 4-7). Wassenaar postulated that super-saturation is uncommon because the lake is generally very turbid (so that most photosynthesis occurs in a shallow euphotic zone near the surface, where excess oxygen can readily equilibrate with the atmosphere) and well-mixed (so that oxygen in deeper water is regularly circulated back to the surface). At least in the north basin, which is often neither turbid nor well-mixed, it may be that the dominance of *Aphanizomenon* spp. and sometimes *Microcystis* spp. explains the lack of super-saturation in spite of high productivity. Both taxa regulate their buoyancy, and typically float to the surface when weather conditions allow. That is, both tend to photosynthesize and release oxygen near the surface where it will quickly equilibrate with the atmosphere. Conversely, many of the observed instances of super-saturation, especially in the north basin, occurred in spring (Figure 4-7) when other algal taxa dominate the phytoplankton community. These spring algae tend to be neutrally buoyant, are carried with currents circulating throughout the mixed layer, and generate oxygen throughout the euphotic zone—some of it at depths where oxygen may be generated faster than it can dissipate.



Downloading water quality data from a data sonde.

Many of the cases of severe oxygen depletion (Figure 4-7) that were recorded during summer in the north basin developed in hypolimnetic water under thermoclines that persisted for weeks, at least. However, given the frequency of thermocline formation, i.e. every year from 2013 to 2016, it is perhaps surprising that hypoxia was not observed more often. It is instructive to consider how hypoxia and even anoxia develop in the central basin of Lake Erie where the phenomenon has been studied since at least the 1970s. There, Charlton (1980) demonstrated that the rate of oxygen depletion varied inversely with the hypolimnion thickness, simply because thickness determined whether oxygen demand from decaying, settled organic matter acted on a larger or smaller volume of confined water. In Lake Erie, the hypolimnion has varied historically from about 2 to 7 m thick; estimated oxygen depletion rates have ranged from 2 to 3 mg/L per month (at 4–7 m thickness) up to 4 mg/L per month (in a 2-m thick hypolimnion). Severe hypoxia can develop at any of these thicknesses since in Lake Erie, the thermocline can persist throughout the summer. In several case studies reported by Charlton, it required from mid-June to late August to reduce oxygen from over 10 mg/l to < 2 mg/L.If a thermocline develops in the north basin of Lake Winnipeg, it typically confines a hypolimnion only 2 to 3 m thick (Figure 4-5, showing profiles to about 1 m above the bottom). Although there is no evidence in several years' records of ECCC moorings of a thermocline persisting more than about 30 days in the north basin, the deep thermocline under which hypoxia was recorded in 2003 may have persisted longer. It likely formed in early July and there is ancillary evidence that it was eroded by strong winds in late August (EC and MWS 2011). Dissolved oxygen levels were as low as 2.5 mg/L in the hypolimnion when they were measured in late July. At depletion rates calculated for Lake Erie, conditions in the hypolimnion in the north basin of Lake Winnipeg in the summer of 2003 would have approached anoxia before the water column re-mixed. One of the thinnest hypolimnia recorded in 2014 is associated with the lowest dissolved oxygen concentration measured that year (A in Figure 4-4). Interestingly, in 2016 a small decrease in temperature just above the bottom in the south basin was associated with a significant drop in dissolved oxygen concentration (B in Figure 4-5).

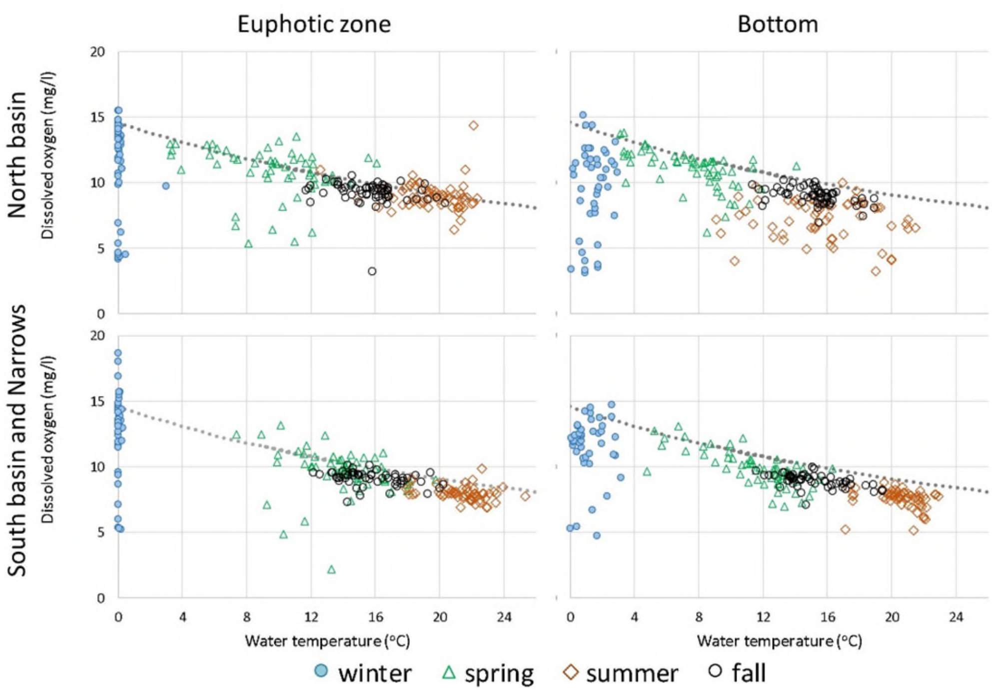


Figure 4-7: Dissolved oxygen concentration as a function of water temperature in the euphotic zone (near surface in winter) and bottom waters from 2008 to 2016 in the north basin, south basin and narrows of Lake Winnipeg. Euphotic zone samples were typically collected from surface to 2–6 m depth in the north basin, and surface to 1–4 m in the south basin. In winter, upper water column samples were collected immediately under the ice. The dotted line indicates oxygen saturation (in lake water at one atmosphere pressure, with no primary production; APHA 2001). Data supplied by Manitoba Agriculture and Resource Development.

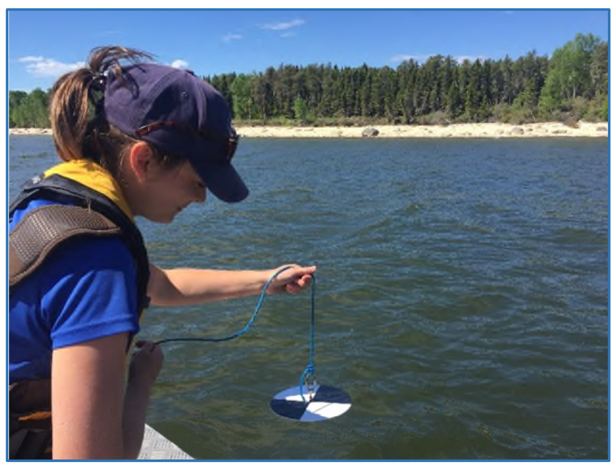
However, such a weak gradient in shallow water would be unlikely to persist for more than a few, very calm days. Hypolimnia of < 2 m thickness, all exhibiting some degree of oxygen depletion, have been recorded in several mid-summer surveys in the north basin (Figure 4-4). If these thermoclines persisted for weeks or more after the summer survey, the hypolimnetic waters may well have become hypoxic or even anoxic. However, it would require additional surveys from mid-summer to autumn, or continuous moorings records between summer and fall surveys (at least in years when strong thermoclines are observed at midsummer) to indicate whether this inference is ever borne out.

Although a few samples under ice were super-saturated with respect to oxygen, many more were under-saturated, indicating that respiration dominated production in winter. Although, as noted above, oxygen super-saturation was more common in spring than later in the open water season, the water column was very often deeply undersaturated in the same period, both in the euphotic zone and at the bottom.

Suspended sediment and light

Suspended solids in lakes comprise plankton generated by biological production within the lake, and mineral particles delivered by tributary rivers or derived from bank erosion or resuspension of littoral and bottom sediments. Their interaction with light by scattering and absorption is the cause of turbidity and is typically the main limitation on transparency in lakes.

From 1999 to 2016, total suspended solids (TSS) concentrations tended to be very low in winter, when the water column was undisturbed by winds and external sources were low (river discharge) or nil (bank erosion). Only two of 182 observations exceeded 10 mg/l (both in the north basin, in 2009 and 2015; Figure 4-8). Under ice, median TSS ranged only from 1.0 to 2.5 mg/l among the two basins and the two averaging periods (n = 23 to 95; Figure 4-8). (Due to the high skewness, samples are illustrated in Figure 4-8 and described here in terms of medians and quartiles rather than means and standard deviations.)



Using a Secchi disk to measure light penetration into the water column.

Through the open water season, TSS ranged from < 1 to 320 mg/L. However, the overall sample is highly skewed, so that 90% of observations were < 20 mg/l and only 2% were > 40 mg/L (n = 1165 from 1999 to 2016). In the north basin, seasonal medians ranged from 3 to 7 mg/L (n = 75–126); in the south basin and narrows, they ranged from 8 to 16 mg/L (n = 50–152). TSS were 5–10X higher during the open water season than under ice.

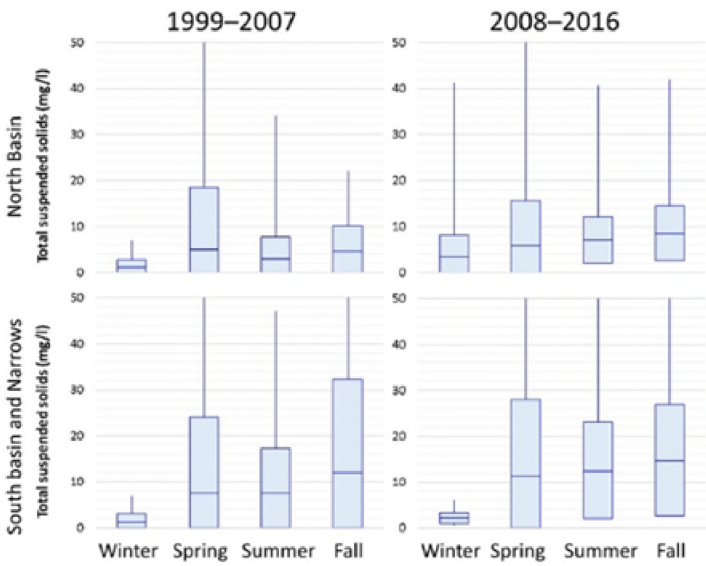


Figure 4-8: Total suspended solids (mg/L) in the north basin (upper panels) and the south basin and narrows (lower panels) showing seasonal statistics and comparing the periods 1999–2007 and 2008–2016. Boxes show medians and quartiles for each group; whiskers indicate the total range. Eighteen values exceed the chart range (50–152 mg/L); all are included in the calculation of the medians and quartiles. Data supplied by Manitoba Agriculture and Resource Development.

As with TSS, turbidity was lowest in winter. The winter medians ranged only from 5.2 to 6.1 NTU among the two basins and the two averaging periods (n = 56–115; Fig. 4-9) (NTU = nephelometric turbidity units). Through the open water period, in the north basin, seasonal medians ranged from 5 to 6 NTU (n = 119–212); in the south basin and narrows, they ranged from 9 to 20 NTU (n = 118–183). In the north basin, there was no distinct seasonal pattern, and indeed, very little range in turbidity through the open water season. However, in the south basin and narrows, turbidity increased through the calendar year, from 4 to 10 NTU in winter and 5 to 15 NTU in spring, to a broader range of 12 to 30 NTU in summer and autumn (referring to quartiles in the bottom panels in Figure 4-9).

Except in winter, transparency was greater in the north basin than the south. In the north basin, from 1999–2016, seasonal median Secchi depths ranged from 0.8 to 1.4 m; from 2008 to 2016, they ranged from 0.15 m deeper (in spring) to 0.25 m deeper (in autumn) compared to the earlier period, 1999 to 2007. In the south basin and narrows, mean Secchi depths ranged from 0.4 to 0.9 m, but there was < 0.1 m difference in median values in any season between the two periods. Seasonal variation and patterns of Secchi depth were generally greater and more revealing than the patterns in either TSS or turbidity. In the north basin, transparency as indicated by the Secchi depth increased from winter to a peak in spring or mid-summer (with generally little difference between the two seasons) and then decreased into autumn, while in the south basin, it decreased continuously from maximum transparency under ice through to a minimum in the fall (Figure 4-10).

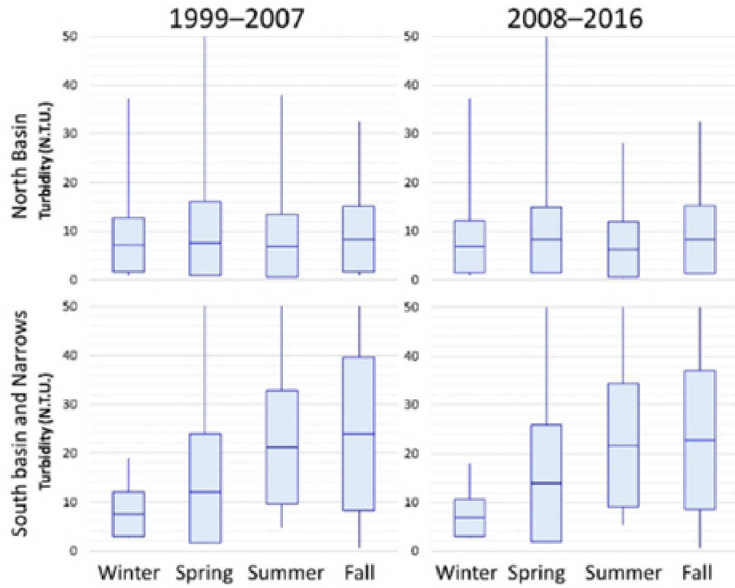


Figure 4-9: Turbidity (NTU) in the north basin (upper panels) and the south basin and narrows (lower panels) showing seasonal statistics and comparing the periods 1999–2007 and 2008–2016. Boxes show medians and quartiles for each group; whiskers indicate the total range. 26 values that exceed the chart range (of a total of 1607 observations) are included in the calculation of medians and quartiles. Data supplied by Manitoba Agriculture and Resource Development.

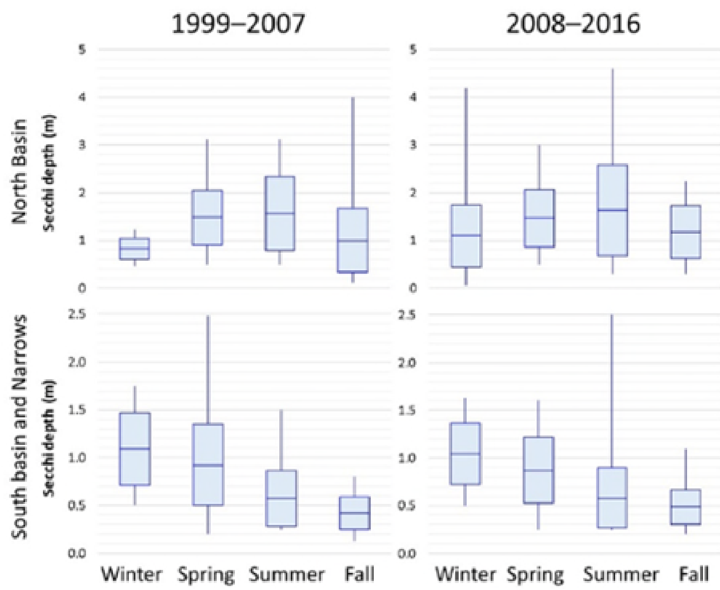


Figure 4-10: Secchi disc depths (m) in the north basin (upper panels) and the south basin and narrows (lower panels) showing seasonal statistics and comparing the periods 1999–2007 and 2008–2016. Boxes show medians and quartiles for each group; whiskers indicate the total range. Note the different scales between regions. One south basin, summer value (Secchi = 5 m) that exceeded the chart range is included in the calculation of median and quartiles. Data supplied by Manitoba Agriculture and Resource Development.

Suspended sediment dynamics in the lake are strongly correlated with antecedent wind energy (Figure 5.2 in EC and MWS 2011) and wind energy over the lake typically increases from spring through to fall (McCullough 2015). Given the extent to which transparency and turbidity are dependent on suspended solids concentration, the patterns observed in both the south basin and narrows are consistent with patterns in the seasonal wind climate over Lake Winnipeg. The relationship is most obvious in the south basin, where turbidity peaks, and transparency is lowest in the windiest season, as would be expected if both are determined primarily by suspended solids mixed into the water column by resuspension of bottom and littoral sediments. One might expect the same pattern in the north basin, and indeed, the transparency is lower in autumn than in spring. That the north basin is as clear in summer as in spring may be due to the seasonal progression of stratification. The water column is typically nearly isothermal in spring and autumn, so that wind-generated turbulence can potentially disturb bottom sediments and mix them through the water column up to the surface. Because the lake tends to stratify in late spring or early summer, wind-generated turbulence cannot propagate down to the sediments and mixing upwards through the water column cannot happen. Hence, in summer, the north basin remains relatively clear of bottom sediments that are mixed upwards during storm events in autumn. This is so even though algal biomass generally peaks in late summer; its effect on transparency in summer, on average, must be less than the combined effect of biomass and resuspended bottom sediments in the fall.

Lake Winnipeg Shoreline, Hecla, Manitoba.

5.0 GENERAL CHEMISTRY AND TRACE METALS

By: Andrew Burton (Manitoba Agriculture and Resource Development)

Water chemistry in Lake Winnipeg has been shown to vary annually, seasonally and spatially with clear geospatial differences between and within basins (EC and MWS 2011). Weather and climate play a critical role influencing lake water chemistry through runoff processes and the characteristics of the surrounding terrestrial watershed (e.g. surface runoff, erosion and weathering of catchment geologies, and hydrological flow pathways). In addition, water chemistry is influenced by anthropogenic activities (e.g. urban stormwater runoff, agriculture, wastewater discharge) which also contribute to changes in the chemical properties of the lake.

Water sampling equipment used onboard the research vessel MV Namao.

Since 1999, Manitoba Agriculture and Resource Development (MARD), in partnership with the Lake Winnipeg Research Consortium Inc., has operated an extensive water quality monitoring program on Lake Winnipeg during the spring, summer, fall and winter periods. Between 1999 and 2016, water quality information has been collected for general chemistry including pH, alkalinity, conductivity, hardness, dissolved oxygen (DO), total suspended solids (TSS), turbidity, nutrients and carbon constituents at approximately 65 stations across the lake (*see Figure 1-3*). An expanded suite of chemistry is measured at 14 long-term stations including ionic constituents and trace elements (e.g. metals).

While much work is underway to characterize and better understand the nutrient dynamics and algal blooms in the lake, few, if any comprehensive assessments of trace elements in the lake have been published to date. For the purpose of this report, the term 'trace' refers to small concentrations of elements usually below 1 mg/L. Many trace elements exist naturally in the environment and some are required micronutrients for plant and animal growth (Drever 1997); however, above certain thresholds trace elements have been shown to negatively impact water uses (e.g. drinking water supply, agriculture) and water resources (e.g. aquatic life, wildlife). This section provides a summary of general chemistry and trace element concentrations associated with the provincial long-term Lake Winnipeg Water Quality Monitoring Program for the 1999 to 2016 period.

Whole Lake Overview

Summary statistics (i.e. 5th, 50th, and 95th percentiles, mean and standard deviation) for general chemistry and trace element concentrations in Lake Winnipeg surface, euphotic and bottom waters measured between 1999 and 2016 are provided in Table 5-1. To calculate summary statistics, two different approaches were used to estimate values below the detection limit depending on the degree of censoring (*see Appendix*). Trace elements beryllium [Be], bismuth [Bi], cesium [Cs], hexavalent chromium [Cr(VI)], selenium [Se], silver [Ag], tellurium [Te], thallium [Tl], and tungsten [W]) were excluded from statistical analyses because concentrations were highly censored (i.e., greater than 80% of measurements were below detection limit). The general chemistry and trace element dataset was primarily derived from 14 long-term monitoring stations (i.e. seven stations in the north basin and seven stations in the south basin and narrows); however, a reduced suite of chemical variables have also been measured at approximately 50 auxiliary stations, with occasional samples collected elsewhere in the lake.

In general, waters of Lake Winnipeg are alkaline (median pH of 8.2) and well buffered (median total alkalinity of 99 mg/L) by the presence of bicarbonates, carbonates, and phosphates primarily owing to the underlying sedimentary geology and catchment soil chemistry. Between 1999 and 2016, lake chemistry was dominated by bicarbonate (HCO_3^-), sulphate (SO_4^{2-}), and calcium (Ca^{2+}) ions resulting in moderate to high hardness and conductivity. Total organic carbon (TOC) concentration (median 8.8 mg/L) was within the range of other natural surface waters in general (Thurman 1985).

Between 1999 and 2016, surface, euphotic and bottom water samples collected from Lake Winnipeg were analyzed for total concentrations of trace elements (n = 33) and dissolved concentrations of aluminum (Al). In general, concentrations of trace elements in Lake Winnipeg are low and many are often below detection. Total Al, iron (Fe) strontium (Sr), barium (Ba), and boron (B) represent the elements with the highest median concentrations and were among the most commonly detected trace elements in Lake Winnipeg waters (Table 5-1). Between 1999 and 2016, the concentration of most trace elements in Lake Winnipeg were similar to typical concentrations measured for Manitoba freshwaters. For example, the concentration of total Fe in Lake Winnipeg ranged from 50 µg/L to 1,425 µg/L (median of 300 µg/L) which is below the median concentration of 390 µg/L in Manitoba. The concentration of total arsenic (As) in Lake Winnipeg ranged from 0.9 µg/L to 3.3 µg/L (median of 1.6 µg/L) which is similar to the median concentration of 1.8 µg/L in Manitoba. Nevertheless, some trace elements (e.g. Al, copper [Cu], and tin [Sn]) were measured towards the higher range of concentrations and were detected more frequently than in other Manitoba freshwaters. For example, the concentration of total Al in Lake Winnipeg ranged from 63 to 1,600 µg/L (median of 310 µg/L), which is higher than the median concentration of 283 µg/L in Manitoba.

Annual Variability

Box and whisker plots for each water quality variable and trace element were used to investigate the inter-annual variability in range and median concentrations in the north basin, south basin, and narrows from 1999 to 2016. Box and whisker plots for total Al (Figure 5-1) and total Fe (Figure 5-2) are provided to illustrate the significant variability from year to year. Median pH and concentrations of alkalinity, conductivity, Ca^{2+}, sodium [Na^{+}], potassium [K^{+}], SO_4^{2-}, total carbon, total dissolved solids (TDS), hardness and many trace elements (i.e. As, Ba, lithium [Li], molybdenum [Mo], Sr, and uranium [U]) were consistently highest in 2012 (data not shown) when lake water levels and river flows were below average (see Section 3.0). However, this general pattern was not visually apparent in other low water years (i.e. 2003) and there was no clear dilution effect for most variables during high water periods (i.e. 2005, 2009, 2011, 2014) with high river inflows. The lack of a pattern between concentrations for most chemical variables and trace elements during the driest and wettest years may be a result of the residence time in the lake (estimated to range from less than three to eight years [see Section 3.0]) and lake regulation. In addition, environmental factors other than precipitation (e.g. temperature, wind), anthropogenic activity (e.g. land

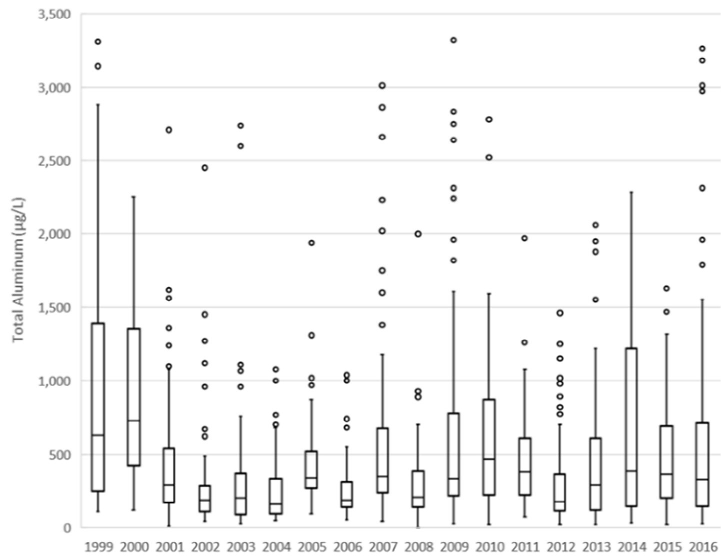


Figure 5-1: Box and whisker plot showing total aluminum concentration (µg/L) measured in Lake Winnipeg surface, euphotic and bottom waters from 1999 to 2016.

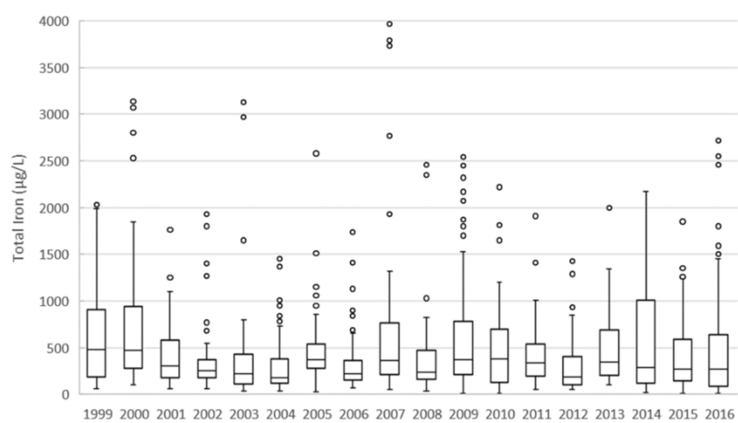


Figure 5-2: Box and whisker plot of total iron concentration (µg/L) measured in Lake Winnipeg surface, euphotic and bottom samples from 1999 to 2016.

Table 5-1: Summary statistics (5th, 50th, 95th percentiles, mean and standard deviation [SD]) of general chemistry and trace elements in all samples (surface, euphotic and bottom) collected from Lake Winnipeg for the period 1999 to 2016.

Variable	Units	5th	50th	95th	Mean	SD	Count Total
pH	pH units	8.5	8.2	7.5	8.0	7.9	3,659
Conductivity	µS/cm at 25° C	123.1	322.0	554.0	335.0	127.9	3,483
Alkalinity Total ($CaCO_3$)	mg/L	48.0	99.0	142.0	99.5	30.5	1,729
Alkalinity Bicarbonate (HCO_3)	mg/L	58.6	120.0	168.0	119.6	35.9	1,729
Ca^{2+} Total	mg/L	13.9	28.4	43.9	28.8	8.8	1,743
Mg^{2+} Total	mg/L	4.3	12.5	21.2	12.9	5.1	1,743
Na^{+} Total	mg/L	2.8	16.3	29.7	16.2	8.5	1,743
K^{+} Total	mg/L	1.1	3.0	5.4	3.2	1.3	1,634
Cl^{-} Dissolved	mg/L	2.2	13.5	28.2	14.9	12.4	1,095
Cl^{-} Total	mg/L	1.7	15.2	35.3	16.7	10.1	638
SO_4^{2-} Total	mg/L	3.9	45.9	108.0	49.5	28.9	638
SO_4^{2-} Dissolved	mg/L	5.0	30.5	64.0	32.2	17.0	1,095
Hardness Total ($CaCO_3$)	mg/L	52.9	122.0	196.0	124.2	41.9	1,716
Total Dissolved Solids (TDS)	mg/L @180° C	85.0	196.0	297.8	197.3	65.1	1,585
Total Suspended Solids (TSS)	mg/L	1.0	6.2	37.1	11.2	0.3	3,463
Turbidity	NTU	1.9	8.2	33.4	12.2	13.6	1,733
Total Inorganic Carbon (TIC)	mg/L	12.0	24.0	39.6	24.6	8.3	2,570
Total Organic Carbon (TOC)	mg/L	5.2	8.8	12.0	8.9	2.7	1,689
Total Carbon (TC)	mg/L	23.0	32.4	50.2	33.8	8.6	2,570
Al Dissolved	µg/L	4.4	30.0	140.0	47.1	59.7	1,742
Al Total	µg/L	63	310	1,600	503	582	1,742
As Total	µg/L	0.9	1.6	3.3	2.3	11.6	1,742
Ba Total	µg/L	14.0	32.8	55.8	33.7	12.3	1,742
B Total	µg/L	5.0	28.0	52.9	29.4	26.4	1,742
Cd Total*	µg/L	0.002	0.01	0.06	0.02	0.001	1,742
Cr Total*	µg/L	0.1	0.5	2.5	0.8	0.025	1,742
Co Total*	µg/L	0.04	0.2	0.7	0.2	0.006	1,742
Cu Total	µg/L	1.3	2.1	4.3	2.6	2.4	1,742
Fe Total	µg/L	50.0	300.0	1,424.5	465.1	517.8	1,712
Pb Total*	µg/L	0.04	0.24	1.30	0.41	0.01	1,742
Li Total	µg/L	2.6	11.0	23.0	12.0	7.7	1,742
Mn Total	µg/L	3.8	18.0	88.5	30.3	55.7	1,743
Mo Total	µg/L	0.2	0.8	1.5	0.8	0.4	1,742
Ni Total*	µg/L	0.7	1.7	3.8	1.9	0.02	1,742
Rb Total	µg/L	1.3	2.0	4.7	2.4	1.2	1,742
Sb Total*	µg/L	0.1	0.2	0.9	0.3	0.01	1,742
Sn Total*	µg/L	0.002	0.1	2.3	0.7	0.2	1,742
Sr Total	µg/L	28.1	100.0	183.1	104.5	46.2	1,720
Th Total*	µg/L	0.01	0.07	0.43	0.13	0.01	1,537
Ti Total	µg/L	1.7	10.0	55.6	17.3	21.1	1,742
U Total	µg/L	0.2	0.8	2.3	1.0	0.7	1,742
V Total	µg/L	0.6	1.7	6.0	2.3	1.9	1,742
Zn Total*	µg/L	0.7	2.8	11.7	4.1	0.1	1,742
Zr Total*	µg/L	0.1	0.4	2.2	0.7	0.02	1,742

development, air pollution, and discharge of wastewater effluent), within lake conditions (e.g. pH, organic matter, hardness, redox reactions) and biogeochemical processes (e.g. biological, sedimentation) each play a critical role in influencing general chemistry and trace elements in the lake. Further work is required to better understand the relationship between inflows, lakes levels, and other drivers of inter-annual variability (e.g. wind, resuspension, redox) of dissolved constituents and trace elements in Lake Winnipeg.

Spatial and Seasonal Variability

Table 5-2 compares the summary statistics (5th, 50th, and 95th percentile, mean and standard deviation) of general chemistry and trace elements measured in surface, euphotic and bottom samples collected from the north basin versus the south basin and narrows from 1999 to 2016. In general, median concentrations of K^+, TSS, turbidity, TOC, and the majority of trace elements (i.e. Al, As, cadmium [Cd], chromium [Cr], cobalt [Co], Cu, Fe, lead [Pb], lithium [Li], manganese [Mn], nickel [Ni], rubidium [Rb], thallium [Th], titanium [Ti], U, vanadium [V], zinc [Zn], and zirconium [Zr]) were significantly higher in the south basin and narrows compared to the north basin. In addition, fewer non-detects occurred in the south basin and narrows than in the north basin for most trace elements. In contrast, conductivity, alkalinity, Na^+, chloride (Cl^-), TDS, total inorganic carbon (TIC) and trace elements Ba, Sn, and Sr were significantly higher in the north basin compared to the south basin and narrows. Median lake pH and concentrations of Ca^{2+}, magnesium (Mg^{2+}), hardness and trace elements B, Mo and Sb were similar (i.e. less than 10% difference) between basins.

Many factors cause seasonal changes in chemical constituents and trace element concentrations in the north basin, south basin, and narrows. For example, Lake Winnipeg completely freezes over during the winter but in the spring, a large flow of water associated with the snowmelt enters the lake within a short time period and effectively dilutes concentrations of specific variables (e.g. conductivity, cations and anions) and may cause increases in other variables (e.g. suspended solids, trace elements). Throughout the open water season, Lake Winnipeg is considered to be a polymictic lake (i.e., well mixed); however, there are locations (specifically in the north basin) that are deep enough to promote weak stratification. Nevertheless, an increase in strong winds during the spring, summer and fall can cause increased turbulent mixing of the lake waters resulting in resuspended sediments and elevated trace element concentrations. Furthermore, throughout the summer and fall, increased water temperatures promote evaporation from the lake, which may lead to increased ionic concentrations (e.g. Ca^{2+}, Mg^{2+}, Na^+, K^+, SO_4^{2-}, and Cl^-). In general, water chemistry within the south basin and narrows is characterized by large fluctuations in the concentrations of chemical constituents and trace elements through the seasons; whereas, water chemistry in the north basin exhibits less variability and more consistent concentrations of chemical constituents and trace elements.

Table 5-2: Summary statistics (5th, 50th, 95th percentiles, mean and standard deviation [SD]) of general chemistry and trace elements in samples from the north basin versus the south basin and narrows of Lake Winnipeg from 1999 and 2016.

		South Basin and Narrows						North Basin					
Variable	Units	5th	50th	95th	Mean	SD	Count	5th	50th	95th	Mean	SD	Count
pH	-	7.5	8.2	8.5	8.0	0.3	2,015	7.5	8.2	8.5	8.0	0.3	1,644
Conductivity	µS/cm at 25° C	107.0	300.0	665.0	329.1	161.4	1,925	239.0	339.0	437.0	342.4	65.5	1,558
Alkalinity Total ($CaCO_3$)	mg/L	46.0	92.0	156.0	94.3	38.4	841	75.4	102.5	133.7	104.3	19.1	888
Alkalinity Bicarbonate (HCO_3)	mg/L	56.1	111.0	181.0	113.6	44.9	841	90.0	124.0	160.0	125.2	23.2	888
Ca^{2+} Total	mg/L	12.9	27.3	48.1	28.1	11.0	852	20.4	28.8	39.2	29.3	6.0	891
Mg^{2+} Total	mg/L	4.0	12.0	24.8	12.7	6.6	852	8.7	12.7	17.8	13.0	2.9	891
Na^{+} Total	mg/L	2.5	11.7	26.3	12.6	7.8	852	10.0	19.1	30.7	19.7	7.6	891
K^{+} Total	mg/L	1.0	3.3	6.3	3.3	1.7	799	2.3	2.9	3.8	3.0	0.5	835
Cl^{-} Dissolved	mg/L	1.7	8.2	16.5	8.6	4.6	511	7.5	19.7	31.9	20.5	14.2	584
Cl^{-} Total	mg/L	1.6	9.5	21.2	9.9	6.2	331	11.8	23.1	38.1	24.1	8.2	307
SO_4^{2-} Total	mg/L	3.5	47.7	128.0	54.9	38.7	331	30.1	44.5	58.9	43.8	8.0	307
SO_4^{2-} Dissolved	mg/L	4.2	32.0	81.3	34.0	21.9	511	20.1	30.2	41.0	30.6	11.0	584
Hardness Total ($CaCO_3$)	mg/L	48.8	116.0	216.0	121.7	53.7	831	87.0	124.0	171.0	126.6	26.0	885
Total Dissolved Solids (TDS)	mg/L @180 °C	76.0	181.0	347.7	189.4	84.5	767	142.9	204.0	263.0	204.7	37.5	818
Total Suspended Solids (TSS)*	mg/L	1.7	9.2	48.4	15.3	0.5	1,911	0.8	3.9	19.2	6.2	0.2	1,552
Turbidity	NTU	3.4	14.2	40.0	17.4	16.7	842	1.5	5.4	19.2	7.3	6.9	891
Total Inorganic Carbon (TIC)	mg/L	10.7	22.3	43.9	24.4	10.3	1,471	18.0	25.0	33.0	24.9	4.5	1,099
Total Organic Carbon (TOC)	mg/L	6.9	9.8	12.9	9.8	2.7	953	4.5	7.8	11.0	7.7	2.3	736
Total Carbon (TC)	mg/L	21.7	32.0	56.7	34.5	10.7	1,471	27.0	32.7	39.7	32.9	4.4	1,099
Al Dissolved	µg/L	6.0	31.7	170.0	54.7	0.1	853	3.8	27.0	120.0	39.8	50.1	889
Al Total	µg/L	110.0	450.0	2,125.0	701.5	0.7	851	51.2	219.0	910.0	314.3	365.5	891
As Total	µg/L	0.8	1.9	3.8	3.1	0.02	851	1.0	1.4	2.2	1.5	0.7	891
Ba Total	µg/L	12.0	29.3	50.9	30.0	0.01	851	22.0	35.0	57.6	37.2	11.6	891
B Total	µg/L	5.0	29.0	60.5	29.7	0.03	851	19.5	28.0	44.5	29.1	27.3	891
Cd Total*	µg/L	0.004	0.014	0.051	0.019	0.001	851	0.0003	0.004	0.073	0.019	0.003	891
Cr Total*	µg/L	0.2	0.8	3.2	1.1	0.04	851	0.1	0.4	1.6	0.5	0.02	891
Co Total*	µg/L	0.1	0.3	0.9	0.3	0.01	851	0.02	0.1	0.4	0.1	0.01	891
Cu Total	µg/L	1.5	2.5	5.0	2.9	0.002	851	1.2	1.9	3.7	2.3	2.7	891
Fe Total	µg/L	140.0	470.0	1,760.0	651.0	0.6	852	49.0	190.0	764.3	280.9	361.0	860
Pb Total*	µg/L	0.1	0.4	1.3	0.5	0.01	851	0.02	0.2	1.1	0.3	0.02	891
Li Total	µg/L	2.0	12.0	28.9	13.0	0.01	851	7.0	10.0	16.1	11.0	4.9	891
Mn Total	µg/L	5.8	26.3	95.7	37.1	0.05	852	3.0	12.8	63.5	23.9	62.3	891
Mo Total	µg/L	0.1	0.8	1.7	0.8	0.0005	851	0.5	0.8	1.0	0.8	0.2	891
Ni Total*	µg/L	1.0	2.2	4.8	2.4	0.04	851	0.7	1.3	2.3	1.4	0.02	891
Rb Total	µg/L	1.5	2.5	5.7	2.9	0.001	851	1.2	1.7	3.3	1.9	0.8	891
Sb Total*	µg/L	0.1	0.2	0.9	0.3	0.01	851	0.1	0.2	0.9	0.3	0.01	891
Sn Total*	µg/L	0.001	0.05	2.08	0.68	0.34	851	0.002	0.07	2.53	0.75	0.25	891
Sr Total	µg/L	25.7	86.7	170.2	89.0	0.05	838	64.0	113.0	187.0	119.2	40.4	882
Th Total*	µg/L	0.02	0.10	0.57	0.18	0.01	741	0.01	0.05	0.27	0.08	0.004	796
Ti Total	µg/L	4.0	15.3	74.2	24.3	0.02	851	1.3	6.5	34.3	10.6	14.7	891
U Total	µg/L	0.1	1.0	2.8	1.2	0.001	851	0.5	0.7	1.1	0.8	0.2	891
V Total	µg/L	0.8	2.8	7.5	3.2	0.002	851	0.5	1.2	2.9	1.4	0.9	891
Zn Total*	µg/L	1.0	3.6	12.6	4.8	0.15	851	0.5	2.2	9.9	3.3	0.1	891
Zr Total*	µg/L	0.2	0.7	2.8	1.0	0.03	851	0.1	0.3	1.2	0.4	0.02	891

Note: Orange shading highlights the variables with concentrations greater than 10% higher in one basin over the other. For variables with less than 15% of observations measured BDL, this value was replaced by ½ the detection limit. The symbol '' denotes the variables whereby MLE was employed to calculate summary statistics owing to the high number of values BDL (i.e. between 15% and 80%).*

Water chemistry in the south basin and narrows is controlled in part by the influence of the Red River and Winnipeg River. The Winnipeg River, which drains the Precambrian Shield and contains relatively dilute water (i.e. acidic to neutral pH with low ionic concentrations), accounts for nearly half of all water transported to Lake Winnipeg. The Winnipeg River effectively dilutes elevated concentrations associated with the alkaline and ion rich waters of the Red River, which drains the prairie landscape. In terms of seasonal patterns in the south basin, average lake pH, conductivity, hardness, total dissolved solids, and concentrations of alkalinity, Ca^{2+}, Mg^{2+}, total carbon (both inorganic and organic fractions), B, and Sr did not vary greatly during the spring, summer, and fall between 1999 and 2016 (Table 5-3). For most chemical constituents and trace elements, concentrations were lower during the winter compared to the spring, summer and fall in the south basin and narrows. During the summer and fall, elevated SO_4^{2-}, K^+, and Cl^- typically occurred, likely because of increased evaporative concentration of ions during the warmer months with this pattern being more pronounced during low water level years. In addition, turbidity and trace elements (i.e. Al, As, Ba, Fe, Li, Mn, Mo, Rb, Ti, and V) were higher during the summer and fall than the winter and spring, likely due to strong winds and turbulent mixing and the shallow nature of the basin.

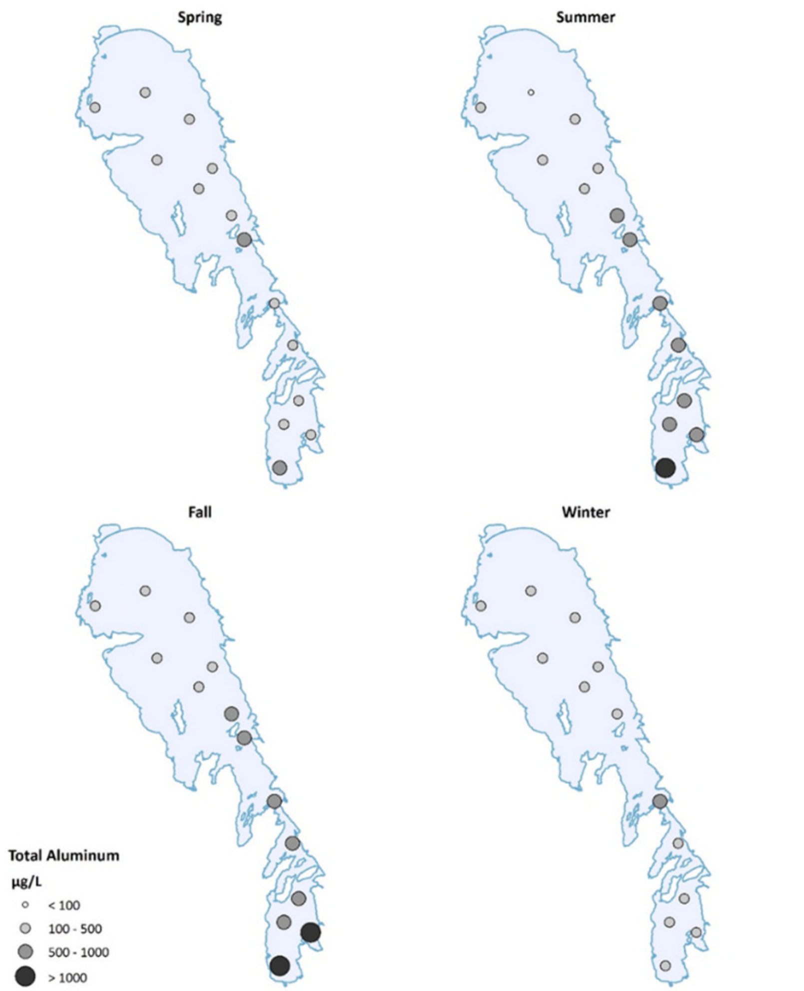


Figure 5-3: Average total aluminum concentrations (µg/L) measured in surface and euphotic samples at the 14 W stations in Lake Winnipeg during the spring, summer, fall and winter from 1999 to 2016.

Water chemistry in the north basin is influenced by the Saskatchewan and Dauphin rivers, which drain the alpine, prairie and boreal plain landscape; characterized by neutral to basic pH with high concentrations of alkalinity and ionic constituents. Elevated ionic concentrations in the north basin are contributed in part by the ionic rich waters draining Lake Manitoba and the Fairford and Dauphin river systems. In the north basin, average pH and concentration of alkalinity, conductivity, total dissolved solids, Ca^{2+}, K^+, Na^+, Cl^-, TIC, total carbon, and trace elements Ba, Cu, Li, and Rb did not vary greatly between the spring, summer, fall and winter between 1999 and 2016 (Table 5-3). However, some constituents did exhibit seasonal variability. For example, the average concentration of TOC, Mg^{2+}, hardness, and some trace elements (i.e. B, Mo, Sr, and U) were higher during the winter compared to spring, summer and fall. Concentrations of turbidity, and trace elements Al, As, Fe, Ti, and V were higher in the fall compared to the spring, summer and winter. Nevertheless, further work is required to determine why the concentration of many trace elements were lower during the summer and fall when elevated concentrations would have been expected.

Table 5-3: Average concentration of chemical constituents and trace elements in surface and euphotic samples at fourteen long-term water quality monitoring stations by season in the south basin and narrows of Lake Winnipeg from 1999 to 2016.

Variable	Units	North Basin				South Basin			
		Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter
pH	pH units	7.8	8.1	8.0	7.9	8.0	7.9	7.9	7.6
Conductivity	µS/cm at 25° C	324.9	323.8	318.4	351.9	282.6	289.0	290.5	217.8
Alkalinity Total ($CaCO_3$)	mg/L	99.7	100.5	101.8	111.4	93.5	93.0	94.8	77.5
Alkalinity Bicarbonate (HCO_3)	mg/L	119.2	120.1	122.1	135.4	112.3	112.0	114.1	93.8
Ca^{2+} Total	mg/L	28.3	28.4	28.6	31.9	27.4	27.8	28.3	22.6
Mg^{2+} Total	mg/L	12.7	12.3	12.6	14.2	12.2	12.8	12.9	9.3
Na^{+} Total	mg/L	19.5	18.7	18.7	20.3	12.1	12.7	12.9	8.3
K^{+} Total	mg/L	2.9	2.9	2.9	3.2	3.1	3.4	3.7	2.2
Cl^{-} Total	mg/L	25.4	23.6	22.8	23.2	9.2	10.5	10.3	6.2
SO_4^{2-} Total	mg/L	43.3	42.3	43.7	45.9	51.1	56.4	55.9	36.3
Hardness Total ($CaCO_3$)	mg/L	122.8	121.1	123.0	138.2	118.3	121.9	122.3	94.8
Total Dissolved Solids (TDS)	mg/L @180° C	202.7	198.2	194.7	221.7	185.2	193.3	195.1	149.0
Total Suspended Solids (TSS)	mg/L	5.1	5.9	8.0	4.6	11.3	11.9	16.7	4.6
Turbidity	NTU	5.6	6.1	8.1	6.2	11.3	19.3	23.8	6.6
Total Inorganic Carbon (TIC)	mg/L	22.9	22.6	23.8	25.1	21.0	21.9	21.7	17.6
Total Organic Carbon (TOC)	mg/L	7.7	8.1	8.4	9.2	9.8	9.9	10.0	11.0
Total Carbon (TC)	mg/L	30.9	30.8	32.6	34.0	31.0	31.9	32.0	28.8
Al Total	µg/L	241.9	217.2	315.3	286.9	382.8	773.0	895.6	331.0
As Total	µg/L	1.3	1.4	1.7	1.6	1.7	2.0	7.3	1.8
Ba Total	µg/L	33.9	34.6	36.4	38.4	25.7	30.6	33.2	20.9
B Total	µg/L	29.3	25.9	27.3	31.4	26.2	27.2	28.3	30.7
Cu Total	µg/L	2.1	2.4	2.0	2.2	2.4	2.6	2.9	3.1
Fe Total	µg/L	207.9	186.3	307.4	227.0	376.9	737.7	867.5	300.3
Li Total	µg/L	10.3	10.6	11.5	11.8	11.8	13.5	14.2	7.6
Mn Total	µg/L	20.3	10.8	14.8	6.4	25.4	41.1	48.8	9.6
Mo Total	µg/L	0.7	0.7	0.8	0.9	0.8	0.9	0.9	0.5
Rb Total	µg/L	1.6	1.7	2.0	1.9	2.1	3.1	3.5	2.0
Sr Total	µg/L	110.1	113.0	117.0	129.6	86.7	90.4	90.7	61.9
Ti Total	µg/L	7.6	7.0	10.9	8.9	13.9	27.5	31.5	11.2
U Total	µg/L	0.8	0.7	0.8	0.9	1.2	1.2	1.2	0.7
V Total	µg/L	1.1	1.1	1.5	1.4	2.3	3.5	4.0	1.8

Note: For variables with less than 15% of observations measured below detection limit (BDL), this value was replaced by ½ the detection limit. Average concentrations for variables with greater than 15% of observations measured below detection limit were excluded from the seasonal comparison.

Not only are there distinct differences in the concentration of chemical constituents and trace elements between basins by season, but also within each basin. For example, spatial and temporal variability within Lake Winnipeg for total Al for the 1999 to 2016 period is illustrated in Figure 5-3. In general, the average total Al concentration was higher in the south basin and narrows (612 µg/L) compared to the north basin (263 µg/L). In the south basin, average total aluminum was observed to be higher during the summer (773 µg/L) and fall (896 µg/L) compared to the winter (331 µg/L) and spring (383 µg/L) (Table 5-3). Figure 5-4 shows the seasonal weighted means of total Al in the south basin and narrows compared to the north basin from 1999 to 2016. Elevated concentrations during the summer and fall (particularly in the south basin) are likely associated with re-suspended Al from the lake sediment induced by strong winds and lake mixing. In the north basin, the average total Al concentration was highest during the fall (315 µg/L), but remains close to the average during the spring (242 µg/L), summer (217 µg/L) and winter (287µg/L).

Modeled spatial and temporal variability within Lake Winnipeg for conductivity from 1999 to 2016 is illustrated in Figure 5-5. Clear spatial and seasonal differences in conductivity were apparent in the north basin and south basin and narrows. High river flows associated with the spring snowmelt dilute lake water, keeping conductivity low throughout the majority of the lake. A strong east to west pattern in conductivity is evident during the summer months owing to lower conductivity water draining into the lake from the Winnipeg River and other rivers on the east side of Lake Winnipeg. Rivers in the south and west (e.g. Red River, Dauphin River) contain ionic-rich waters. During the fall, lake mixing promotes more uniform conductivity levels in the lake.

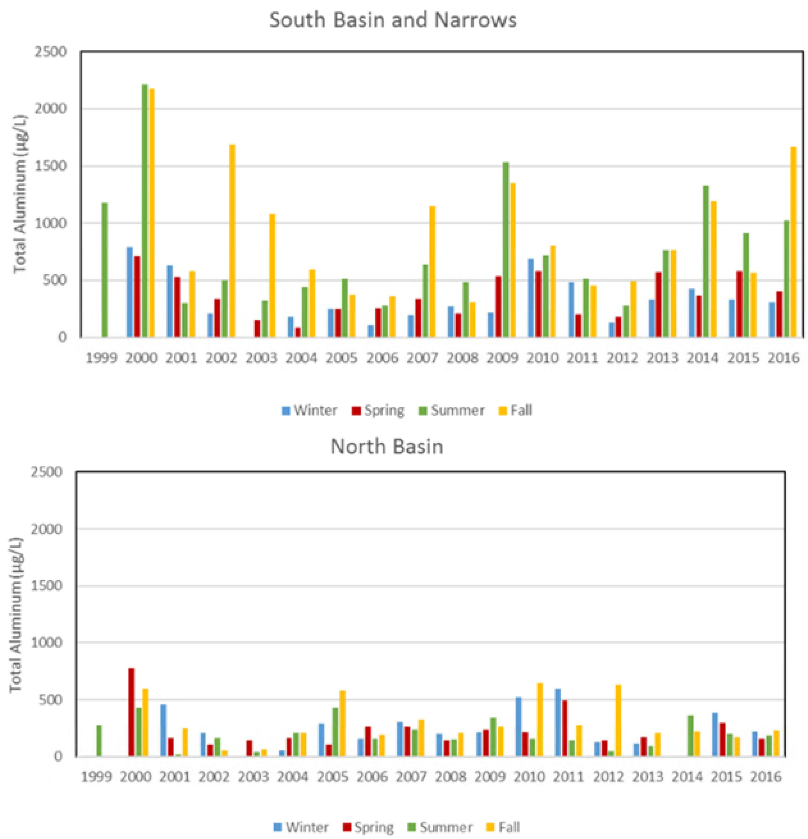


Figure 5-4: Average total aluminum concentration measured in surface and euphotic samples at 14 long-term water quality monitoring stations by season in the south basin and narrow (top panel) and north basin (bottom panel) of Lake Winnipeg form 1999 to 2016.

Despite the shallow nature of the lake and strong wind-induced mixing, vertical gradients in trace elements are apparent in Lake

Winnipeg. For example, there is evidence of higher concentrations of trace elements (i.e. Al, Co, Cu, Fe, Mn, Th, and Ti) at the lake bottom compared to the surface and euphotic zone; particularly at sites exhibiting weak stratification during calm conditions (data not shown). However, further investigation is required to understand the mechanisms (e.g. physical, chemical and biological processes, re-release of stored trace elements into the water column from bottom sediments) promoting these changes.

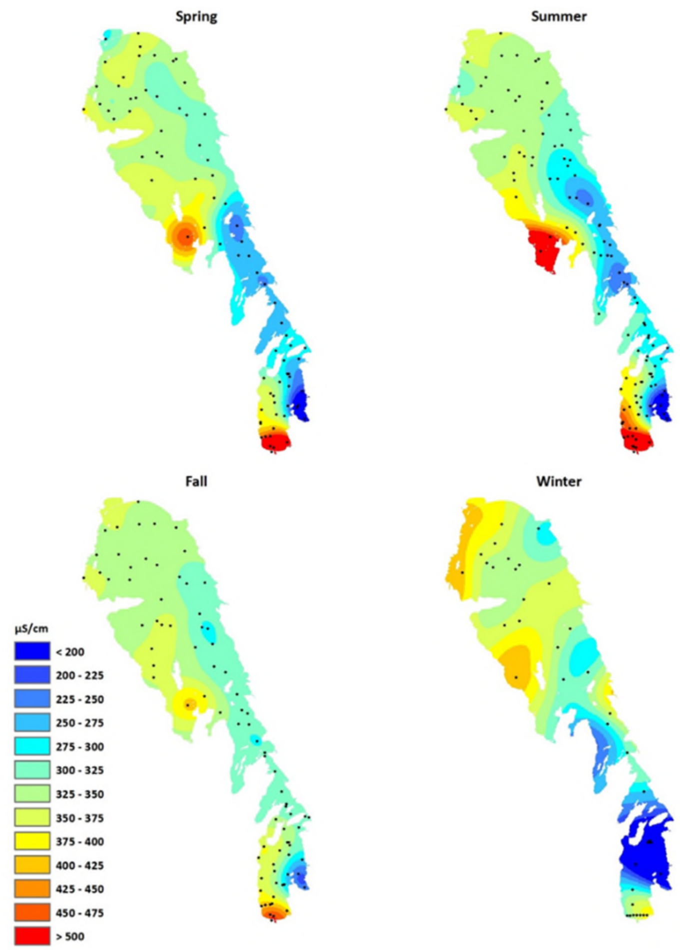


Figure 5-5: Average conductivity (µS/cm) measured in surface and euphotic samples in Lake Winnipeg during spring, summer, fall and winter from 1999 to 2016.

Exceedance of Water Quality Objectives and Guidelines

Chemical constituents were compared to water quality objectives and guidelines for the protection of aquatic life, where possible (Table 5-4). Exceedances of provincial water quality objectives and federal water quality guidelines occurred frequently for total trace elements Al (87%) and Fe (49%); less frequently (i.e. between 1% and 5%) for total Ag, Co, Cu, and Se; and rarely (< 1%) for total As, Cd, Cu, Pb, Se, Zn and dissolved Cr(VI). Lake pH was within the acceptable range for the protection of aquatic life at all times.

Exceedances for total Al and total Fe occurred in all eighteen years monitored and there was an apparent geospatial pattern in distribution whereby a higher percentage of exceedances occurred in the south basin and narrows compared to the north basin. For example, 95% of the samples in the south basin and narrows exceeded the water quality guideline for total Al compared to 80% in the north basin. In total, 69% of samples in the south basin and narrows exceeded the water quality guideline for total Fe compared to 29% in the north basin.

Exceedances for total Al comprised a nearly equal mixture of surface, euphotic and bottom depth samples whereas exceedances for total Fe occurred primarily in the bottom depth samples. On occasion (4.6% of all samples measured), the federal water quality guideline for the protection of aquatic life for total Ag was exceeded; however, no exceedances have occurred since 2010. Similar to Fe, Ag exceeded the water quality guideline more frequently in the south basin (6%) compared to the north basin (4%), and bottom depth samples exceeded the guidelines more frequently than surface or euphotic samples. Nevertheless, the majority of samples measured for total Ag were below the detection limit (91%).

Table 5-4: Trace elements measured in Lake Winnipeg waters from 1999 to 2016 compared to provincial water quality objectives and federal water quality guidelines for the protection of aquatic life.

Variable	Reference	Water Quality Guideline for Protection of Aquatic Life		% Exceedance	
		Short-Term	Long-Term	Short-Term	Long-Term
Aluminum Total	CCREM 1987	-	5 µg/L if pH < 6.5 or 100 µg/L if pH ≥ 6.5	-	87.2
Arsenic Total	USEPA 1995	340 µg/L	150 µg/L	0.1	0.2
Boron Total	CCME 2009	29,000 µg/L	1,500 µg/L	0.0	0.0
Cadmium Total	US EPA 2001	e^(1.0166*ln[Hardness]-3.924)	e^(0.7409*ln[Hardness]-4.719)	0.0	0.9
Chloride Total	CCME 2011a	640,000 µg/L	120,000 µg/L	0.0	0.0
Chromium Hexavalent Total	US EPA 1995	16 µg/L	11 µg/L	0.1	0.1
Cobalt Total	EC 2017	-	e^(0.414*ln[Hardness]-1.887)	-	1.4
Copper Total	US EPA 1999	e^(0.9422*ln[Hardness]-1.700)	e^(0.8545*ln[Hardness]-1.702)	0.8	1.5
Iron Total	CCREM 1987	-	300 µg/L	-	49
Lead Total	US EPA 1980	e^(1.273*ln[hardness]-1.460)	e^(1.273*ln[hardness]-4.705)	0.0	0.9
Molybdenum Total	CCME 1999a	-	73 µg/L	-	0.0
Nickel Total	US EPA 1995	e^(0.8460*ln[Hardness]+2.255)	e^(0.8460*ln[Hardness]+0.0584)	0.0	0.0
pH	CCREM 1987	-	6.5 - 9.0	-	0.0
Selenium Total	CCREM 1987	-	1 µg/L	-	1.6
Silver Total	CCME 2015	-	0.25 µg/L	-	4.6
Thallium Total	CCME 1999b		0.8 µg/L	-	0.0
Uranium Total	CCME 2011b	33 µg/L	15 µg/L	0.0	0.0
Vanadium Total	EC 2016b		120 µg/L	-	0.0
Zinc Total	USEPA 1995	e^(0.8473*ln[Hardness]-0.884)	e^(0.8473*ln[Hardness]-0.884)	0.1	0.1

Note: Provincial water quality objectives and guidelines for the protection of aquatic life are based on MWS (2011). The Government of Manitoba adopts U.S. EPA acute and chronic water quality criterion for protection of aquatic life for some trace elements (i.e. As, Cd, Cr[VI], Cu, Pb, Ni, Zn), and federal water quality guidelines for protection of aquatic life based on the Canadian Council of Ministers of the Environment (CCME 2018) for Ag, Al, B, Cl, Fe, Mo, pH, Se, Tl, and U. Environment and Climate Change Canada developed Federal Environment Quality Guidelines for protection of aquatic life (i.e. Co, V) (ECCC 2018b). The dash (-) symbol represents not applicable (i.e. guideline does not exist).

Exceedances of water quality objectives and water quality guidelines for the protection of aquatic life for a variety of trace elements (e.g. Al, Fe and Ag) have also been reported in other Manitoba surface waters (unpublished data, MARD 2018). For example, between 2005 and 2016, 78% of all surface water samples analyzed for total Al across the province exceeded the provincial water quality guideline. However, note that an exceedance of a water quality guideline does not necessarily mean that it is causing an adverse impact to aquatic life. For example, water quality guidelines for the protection of aquatic life are derived using a variety of laboratory toxicological studies (i.e., subjecting various species to different exposure conditions) and often follow a statistical approach or incorporate a safety protection factor. Therefore, it is possible for a water quality guideline for the protection of aquatic life to exceed the natural background concentration (or natural condition) at a given site. In Manitoba, the concentration of many trace elements (including Al, Fe, and Ag) in surface waters are naturally high, primarily due to the naturally occurring geology in the province. Therefore, it may be more appropriate to use the natural background concentration as the water quality guideline for the protection of aquatic life based on the assumption that the biological community has adapted to the local conditions. However, additional studies would be required if anthropogenic activity was observed to be increasing the concentration of a trace element and/or causing impacts to aquatic life above the natural background concentration.

Chemical constituents were compared to provincial water quality objectives and guidelines for the protection of agricultural uses (i.e., irrigation, livestock). Exceedances of water quality objectives and guidelines for livestock rarely occurred between 1999 and 2016 with the exception of As (0.3%). Exceedances of irrigation water quality guidelines occasionally occurred for B (5%) and rarely (<1%) for TDS, As, hexavalent Cr, Fe, and Mn.

Potential Sources and Factors Affecting Trace Elements in Lake Winnipeg

The most abundant trace elements in the earth's crust (i.e. Al, Ba, Fe, Mn, and Sr) were also the most common and highest detected elements in Lake Winnipeg waters and it is likely that natural sources (e.g. soil erosion and geochemical weathering of catchment geology and lithology) partly influence the observed concentrations in Lake Winnipeg.

Although the Winnipeg River typically contributes half of the flow to Lake Winnipeg and effectively dilutes the water chemistry in the south basin, the majority of trace elements in the lake water appears to be associated with discharge from the Red River. Loading calculations near the discharge point to Lake Winnipeg along the major rivers (Winnipeg, Red, Dauphin) were computed and the Red River was determined to be the major contributor for total Al, As, Ba, B, Fe, Li, Mn, Mo, Pb, Rb, Sr, Ti, U, and V. Anthropogenic activities such as agricultural practices (e.g. application of pesticides, manure and biosolids; animal husbandry), logging, mining, land development, stormwater runoff from urban areas, discharge from industrial and municipal wastewater treatment facilities, and leachate from landfill sites could all be sources of trace elements to Lake Winnipeg (COSEWIC 2002). Although most constituents remain low and are typically below environmental quality guidelines, further investigation into impacts of trace elements in Lake Winnipeg is warranted.

Conclusion

This summary includes a review of general chemistry (i.e., water quality variables), and for the first time, a comprehensive assessment of trace elements in Lake Winnipeg waters from 1999 to 2016. Significant annual, spatial, and temporal variability in chemical composition and trace element concentrations highlights the complexity of understanding water chemistry in Lake Winnipeg. Changes in water quality conditions in Lake Winnipeg are highly influenced by watershed characteristics (e.g. soils and geology), environmental factors (e.g. weather and climate), internal physical and biogeochemical lake processes, and the chemistry of the major rivers discharging into the lake. Further work is required to better understand the potential sources of trace elements in Lake Winnipeg and the biogeochemical factors influencing their concentrations

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6.0 LAKE NUTRIENT CONCENTRATIONS

By: Elaine Page (Manitoba Agriculture and Resource Development)

Excessive concentrations of nitrogen and phosphorus are contributing to an increase in the frequency and severity of algal blooms in Lake Winnipeg. Scientific studies in Manitoba demonstrate that phosphorus and nitrogen concentrations in some streams have increased from natural conditions by nearly 200% due to human activities. Algal blooms spoil drinking water, ruin beaches, reduce property values, and can have adverse effects on fish and other aquatic life. There are many types of algae that can form blooms, including cyanobacteria, commonly known as blue green algae. Some blue green algae produce toxins that can be dangerous to people, livestock and pets. Algal blooms also have a negative effect on our economy by decreasing tourism and over time, may reduce the productivity of commercial and recreational fisheries.

Algal bloom, Lake Winnipeg.

There have been a number of limnological surveys of Lake Winnipeg through time beginning in the mid-1920, 1960s, 1970s and early 1990s. Further details of the history of these surveys is summarized in the previous State of Lake Winnipeg report (EC and MWS 2011). Much of the intensive research and annual monitoring of Lake Winnipeg has occurred since the late 1990s following concerns around the increasing frequency and severity of algal blooms and the impact of the flood of 1997 on Lake Winnipeg. The results presented in the following section will summarize the nutrient conditions in Lake Winnipeg since 1999 when annual lake monitoring began.

Since 1999, Manitoba Agriculture and Resource Development has maintained a long-term water quality program to track changes in lake water quality over a network of 65 stations on Lake Winnipeg (*see Figure 1-3*). Chemical and biological samples are generally collected during the spring, summer and fall lake cruises from the Motor Vessel *Namao* in partnership with the Lake Winnipeg Research Consortium Inc. In 2014, 12 nearshore stations were added to the monitoring network to better characterize the nearshore chemistry of Lake Winnipeg. Water samples collected at nearshore and offshore stations are analyzed for nutrients, chlorophyll-a, and other chemical parameters at all stations during each cruise. More intensive monitoring occurs at 14 long-term monitoring sites for additional analyses including metals, major ions, sediment chemistry and particle size analysis, phytoplankton biomass and species composition, microcystin-LR and benthic invertebrate densities (*see Figure 1-3*). Water samples are collected at various depths depending on the station (e.g., surface, euphotic and 0.5 meters above lake sediments). Sites are accessed by a Bombardier track machine and helicopter during the ice cover season to collect samples from a reduced suite of stations in collaboration with Environment and Climate Change Canada. The objective of the provincial long-term water quality monitoring program on Lake Winnipeg is to determine the impact of anthropogenic activities on water quality and aquatic life and to provide information to support and protect the health of the lake.

Phosphorus

The south basin of Lake Winnipeg is rich in phosphorus and is considered hyper-eutrophic. For the 1999 to 2016 period, total phosphorus (TP) concentrations were almost three times higher in the south basin and narrows (0.104 mg/L) than the north basin (0.039 mg/L) of the lake (Figure 6-1). Compared to Lake Erie, TP concentrations in Lake Winnipeg are approximately two times higher in the north basin and four times higher in the south basin in the spring (mean spring TP of 0.020 mg/L in Lake Erie; Millard and Howell 2009).

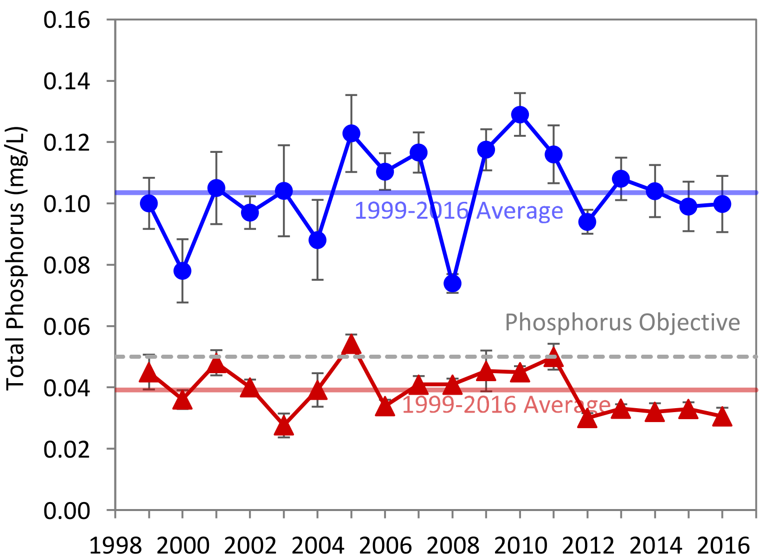


Figure 6-1: Annual open water mean TP concentrations (± SE) in the north basin and the south basin and narrows of Lake Winnipeg from 1999 to 2016.

The TP objective for Lake Winnipeg is 0.05 mg/L and was derived using paleolimnological inference models that reconstructed historical changes in nutrient influx, algal production and phytoplankton composition (Bunting et al. 2016). That work indicated that Lake Winnipeg was mesotrophic (0.02 mg/L) until the 1900s (Bunting et al. 2016) and there was an acceleration of eutrophication through the 1900s with nitrogen, phosphorus and algal pigments increasing through time. By the 1990s, the inferred phosphorus concentrations were approximately 0.05 mg/L and since that time there has been a rapid acceleration of eutrophication with phosphorus concentrations closer to 0.1 mg/L in the south basin of the lake. These recent TP concentrations in the south basin are two times the objective and significant nutrient reductions will be required across all sectors to achieve the phosphorus objective in Lake Winnipeg.

In the south basin, there is no apparent increasing or decreasing trend in TP, and recent concentrations are within the range of variability observed for the 1999 to 2016 period (Figure 6-1). Phosphorus has been quite stable in the north basin over the same period, although concentrations appear to be slightly lower in recent years. It is unknown why this is the case and further analyses of trends will be required to determine whether this is in fact a significant trend. Generally, high TP concentrations in the south basin corresponded with high river flows and TP loading to the lake (i.e. in 2005, 2009 and 2011). However, this does not appear to be the case for the north basin, likely because some of the loads from the Red River are attenuated in the south basin of the lake before reaching the north basin. Annual average TP concentrations are positively correlated with TP loading to the lake (EC and MWS 2011). However, the relationship was much weaker in the south basin, which highlights the complexity of other processes affecting nutrient dynamics in the south basin (e.g. sedimentation and wind-induced resuspension).

Major geospatial gradients of phosphorus concentrations indicated that phosphorus concentrations are typically highest at the very south end of the lake near the inflow of the Red River and decline moving northwards for the 2008 to 2016 period (Figure 6-2). Spatial patterns in phosphorus concentrations found in Lake Winnipeg appear to be associated with the phosphorus-rich inflow of the Red River, which transports approximately 69% of the TP load to Lake Winnipeg, on average (*see Section 7*). The Winnipeg River is the second largest source of phosphorus to Lake Winnipeg and contributes approximately 14% of the TP loading to Lake Winnipeg. The remaining contributions are relatively low in comparison and their combined contribution is similar to that of the Winnipeg River. A strong south to north gradient was apparent through spring, summer and fall periods and can be partly explained by high concentrations of dissolved phosphorus (DP > 0.1 mg/L) in the south basin, which comprises nearly 60% of the TP in this basin (Figure 6-2).

For the 2008 to 2016 period, seasonal patterns in TP concentrations were lowest in the spring and showed increasing trends through the summer and fall months in the south basin (Figure 6-2). Total phosphorus showed a different pattern in the north basin with the lowest concentrations in summer (0.02 to 0.03 mg/L) and highest concentrations in fall (0.04 to 0.07 mg/L). The pattern of increasing TP from spring to fall in the south basin is typical of shallow eutrophic lakes and is driven by internal re-mobilization and resuspension processes (Søndergaard et al. 2003).

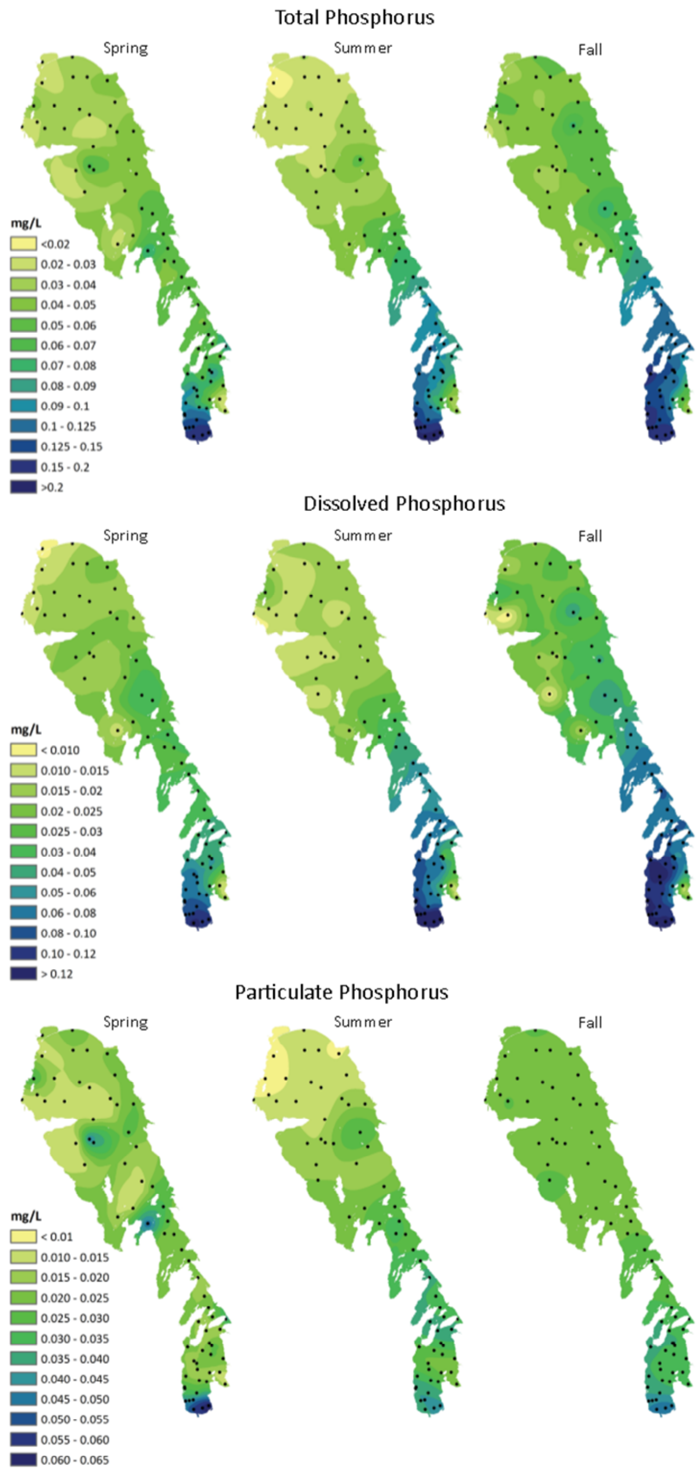


Figure 6-2: Total, particulate and total dissolved phosphorus in Lake Winnipeg during the spring, summer and fall, average of the 2008–2016 period. Note the change in scale.

Phosphorus may also be re-mobilized from the lake sediments during events when oxygen is depleted. Low oxygen events have been observed on occasion in the north basin of Lake Winnipeg; however, the south basin is comparatively shallow and typically well oxygenated at depth in most cases. Matisoff et al. (2017) found that the majority of sediment in Lake Winnipeg was resuspended from the lake bottom (~95%) and that sediment-associated internal TP loads estimated for the south basin were double the rate in the north basin. In contrast, Nürnberg and LaZerte (2016) found that much of the total internal phosphorus loading was contributed by the north basin (57–79%) with smaller loads estimated for the narrows (16–20%) and the south basin (18–23%). Internal phosphorus loads were found to be comparable to external loads (Matisoff et al. 2017) or 1.3 times greater than external loads to the lake (Nürnberg and LaZerte 2016). Given the high internal loads, Lake Winnipeg, like many nutrient-rich shallow lakes, is expected to take much longer to observe improvements in water quality because of legacy phosphorus in the lake sediments. Furthermore, Lake Winnipeg retains approximately 47% of external phosphorus loads on average making remediation efforts all the more challenging.

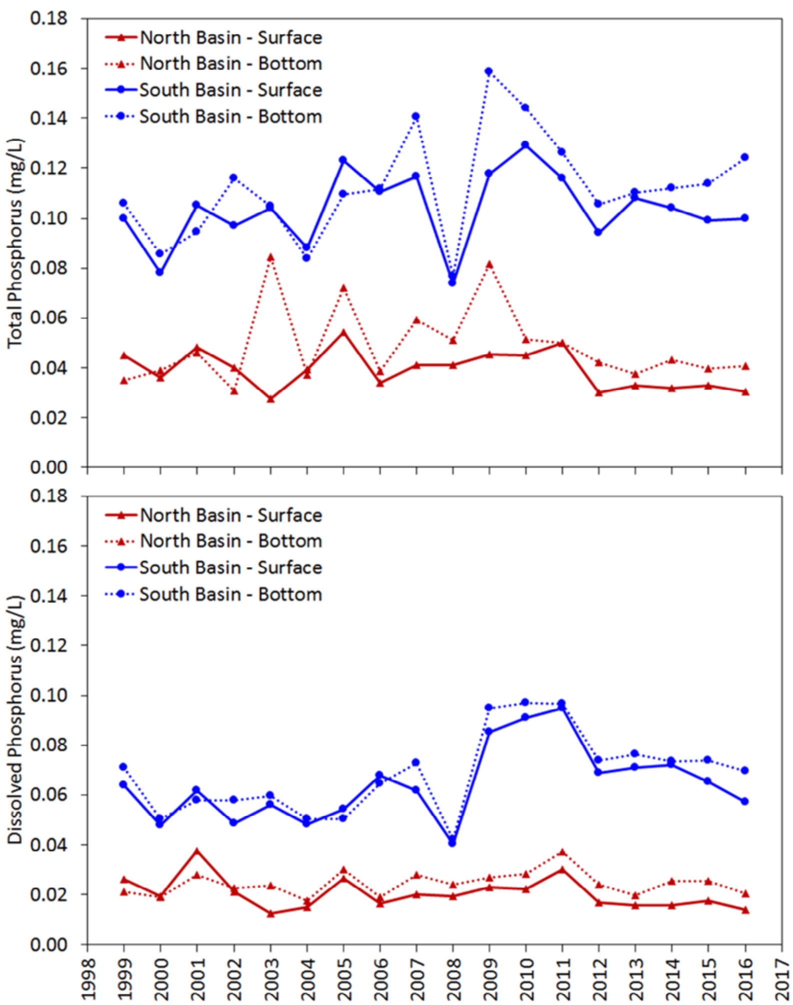


Figure 6-3: Annual open water mean total and dissolved phosphorus concentrations in the north basin and the south basin and narrows of Lake Winnipeg collected from the lake surface and euphotic zone (surface) and 0.5 m above the lake sediments (bottom) from 1999 to 2016.

A comparison of annual average TP and DP concentrations at the lake surface (including samples collected from the surface and euphotic zone) and 0.5 m above the lake bottom indicate that phosphorus is released from the sediments of Lake Winnipeg in most years in both basins of the lake (Figure 6-3). In the north basin, average phosphorus concentrations at depth for the 1999 to 2016 period (0.049 mg/L) are approximately 25% higher than surface phosphorus concentrations (0.039 mg/L). However, in some years in the north basin, TP concentrations just above the lake bottom were more than double the concentrations measured at the lake surface. In the south basin, annual average TP concentrations at depth ranged from 11% below to 24% above surface TP concentrations for the 1999 to 2016 period. In the north basin, peak concentrations of TP were apparent at depth in 2003, 2005 and 2009, and appear to be related to increases in particulate phosphorus, as the dissolved fraction did not show similar increases during these years. Average DP:TP ratios in the north basin (not shown) indicate that DP was slightly higher at depth (0.58) as compared to surface (0.52) on average for the 1999 to 2016 period.

Vertical differences in phosphorus were also observed in the south basin and narrows with the average TP concentration about 9% higher (0.112 mg/L) at depth as compared to the average concentration at the surface (0.104 mg/L) for the 1999 to 2016 period. Annual average TP concentrations at depth ranged from 11% below to 24% above surface TP concentrations in the south basin of Lake Winnipeg for the 1999 to 2016 period. Similar to the north basin, the south basin and narrows also had peaks in TP concentrations at depth in certain years such as 2007 and 2009; however, the TP concentrations at depth occurred in different years in some cases. Average DP:TP ratios in the south basin and narrows (not shown) indicate that DP was similar at depth as compared to surface (0.60) on average for the 1999 to 2016 period.

Wind speed, water column stability, dissolved oxygen, temperature, iron and redox processes are important mechanisms that would affect the internal phosphorus resuspension and remobilization in Lake Winnipeg. However, further analysis is required to explain the differences and mechanisms controlling the internal phosphorus load from the north and south basins of Lake Winnipeg, given their distinct characteristics.

Nitrogen

Similar to phosphorus, the south basin of Lake Winnipeg had higher total nitrogen (TN) concentrations (0.85 mg/L) as compared to the north basin (0.63 mg/L) of the lake for the 1999 to 2016 period (Figure 6-4). Inter-annual variability in nitrogen may be driven by a number of factors, including nitrogen fixation and denitrification processes, nitrogen loading from tributary rivers, internal loading and wind-induced resuspension. From 1999 to 2016, there are no clear trends in TN in the south basin of

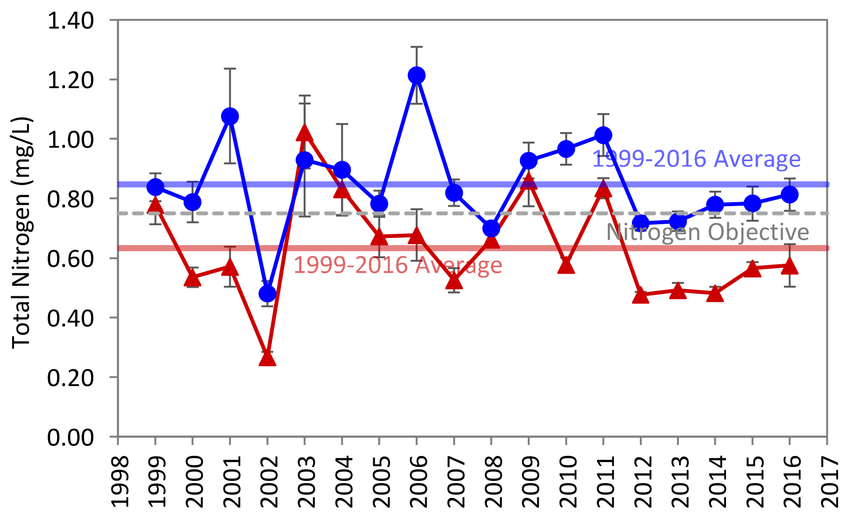


Figure 6-4: Annual open-water mean TN concentrations (± SE) in the north basin and the south basin and narrows of Lake Winnipeg from 1999 to 2016.

Lake Winnipeg although nitrogen concentrations since 2012 appear slightly lower as compared to earlier years in the record with the exception of 2002 (Figure 6-4). TN in the north basin generally followed a similar pattern over the same period with concentrations appearing slightly lower in recent years.

Similar to patterns in phosphorus, major geospatial gradients in TN and dissolved inorganic nitrogen (DIN) are partly associated with the nitrogen-rich inflow of the Red River which transports approximately 34% of the TN load to Lake Winnipeg, on average (Figure 6-5). Other major sources of nitrogen to Lake Winnipeg include the Winnipeg River, atmospheric deposition and nitrogen fixation. In Lake Winnipeg, DIN may comprise up to 31% of TN in the north basin and 45% in the south basin and narrows.

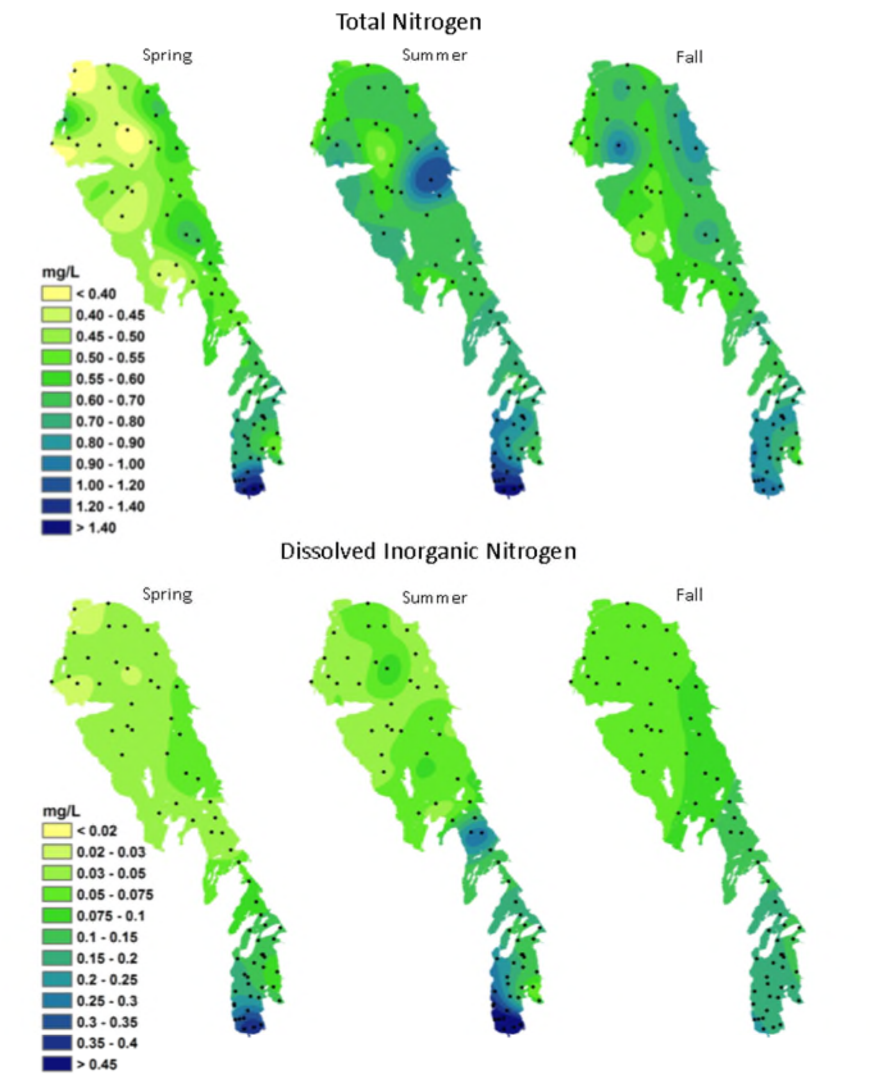


Figure 6-5: Total nitrogen and dissolved inorganic nitrogen in Lake Winnipeg during the spring, summer and fall, average of the 2008–2016 period. Note the change in scale.

For the 2008 to 2016 period, nitrogen concentrations were typically highest at the very south end of the lake near the inflow of the Red River and declined moving northwards. Spatial gradients were pronounced during the spring and summer period with highest nitrogen concentrations at the south end of the lake. TN and DIN concentrations peaked during the summer period in the south basin (> 1.4 mg/L TN, > 0.45 mg/L DIN) and may be attributed to the high concentrations of nitrogen being transported to the south basin by the Red River. Following peak summer concentrations, TN and DIN concentrations declined in the fall in the south basin. However, in the north basin, nitrogen concentrations were generally lowest in the spring, increased in the summer and remained similar across the summer and fall periods, likely owing to the extensive blooms of cyanobacteria in the north basin, which fix nitrogen from the atmosphere.

Nitrogen fixation studies were carried out in 2003 to measure nitrogen fixation rates by nitrogen fixing cyanobacteria (L. Hendzel, unpublished). These data were used to derive a nitrogen load estimate from nitrogen fixation for the lake (9,300 tonnes/year). However, it is unknown how fixation would vary from year to year given differences in cyanobacterial bloom composition, frequency, magnitude and duration each year. Moreover, no measurements of denitrification are available for Lake Winnipeg even though this can represent an important nitrogen loss from the lake. Findings from other eutrophic lakes indicate that losses through denitrification processes may be considerable (e.g. Nõges et al. 1998) and this warrants further study in Lake Winnipeg. Other nitrogen losses from Lake Winnipeg include the Nelson River, which exports approximately 51% of the TN loading to the lake, while 49% of nitrogen is retained within the lake.

Nutrients in the Nearshore and Offshore

Nearshore water quality monitoring began in 2014 at 12 stations in the nearshore areas of the lake. These stations were established to better characterize the general water quality conditions in the nearshore areas and to assess potential water quality changes that may relate to Zebra Mussels (*Dreissena polymorpha*), which were first detected in Lake Winnipeg in 2013. At each site, water samples are collected at the lake surface in a transect from shore at 1, 2 and 3 m depths along the transect. Data are briefly summarized for the 2014 to 2016 period to provide an initial characterization of the major differences in nutrients in nearshore and offshore areas of the lake. Nutrient data for the same period (2014 to 2016) were summarized for surface water samples collected from offshore stations.

Sampling water at a nearshore station.

Overall, TP and particulate phosphorus (PP) concentrations appear to be higher in the nearshore areas of Lake Winnipeg as compared to offshore regions for the 2014–2016 period (Figure 6-6). The average TP concentration was 30% and 11% higher in the nearshore areas of the north basin and south basin and narrows (respectively), as compared to the offshore areas for the same period. Similarly, average PP concentrations were also elevated in the nearshore areas of the north basin (nearshore mean=0.030 mg/L; offshore mean=0.016 mg/L) and south basin and narrows (nearshore mean=0.046 mg/L; offshore mean=0.036 mg/L). This is likely attributed to the resuspended sediment and sediment associated nutrients from the shallow nearshore areas that are subject to intense wave action during sustained wind events on the lake. Elevated TP concentrations are apparent at nearshore stations at the very southern end of the lake. The Red River is the single largest source of phosphorus to the lake and its contribution would also help to explain the elevated phosphorus concentrations in the nearshore areas at the very southern end of the lake. Unlike total and particulate fractions of phosphorus, average dissolved phosphorus concentrations in the north basin (0.015 mg/L) and the south basin and narrows (~0.65 mg/L) did not differ between nearshore and offshore areas of Lake Winnipeg (Figure 6-6).

Similar to total and particulate phosphorus, TN was also higher in the nearshore areas as compared to the offshore in both basins of Lake Winnipeg (Figure 6-7). Specifically, TN was 13% higher in the nearshore areas of the north basin and 16% higher in the nearshore areas of the south basin and narrows as compared to the offshore regions of the lake within each respective basin. As with phosphorus, some of the elevated nearshore TN concentrations may be associated with the resuspended sediments and nutrients in the dynamic nearshore area that is subject to strong wind and wave action. Elevated nitrogen concentrations in the nearshore zones of the lake may also be partly related to nitrogen-fixing cyanobacteria, which are predominant in the lake during the summer and fall periods. However, it is unknown how nitrogen fixation rates would differ between nearshore and offshore areas, as nitrogen fixation has only been measured in the offshore areas of Lake Winnipeg (L. Hendzel, unpublished). Similar to TN, average DIN concentrations were higher in the nearshore as compared to the offshore in the north basin. This was not the case in the south basin, where average DIN was higher at offshore sites, especially those located at the very southern end of the lake nearest the inflow of the Red River.

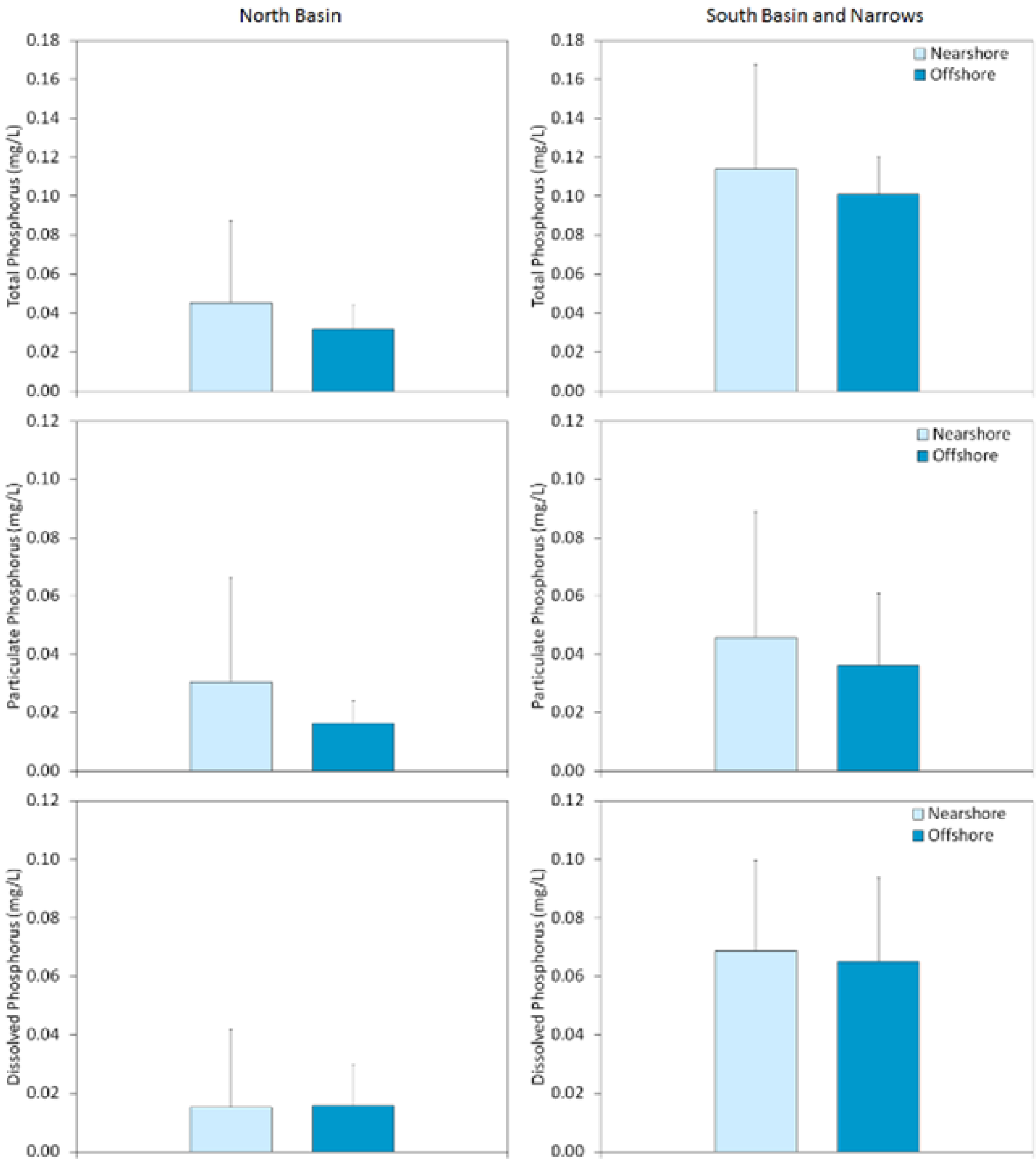


Figure 6-6: Average total, particulate and dissolved phosphorus concentrations (mg/L, ±SE) in the nearshore and offshore of the north basin and south basin and narrows regions of Lake Winnipeg, 2014 to 2016. Note scale change for dissolved and particulate phosphorus.

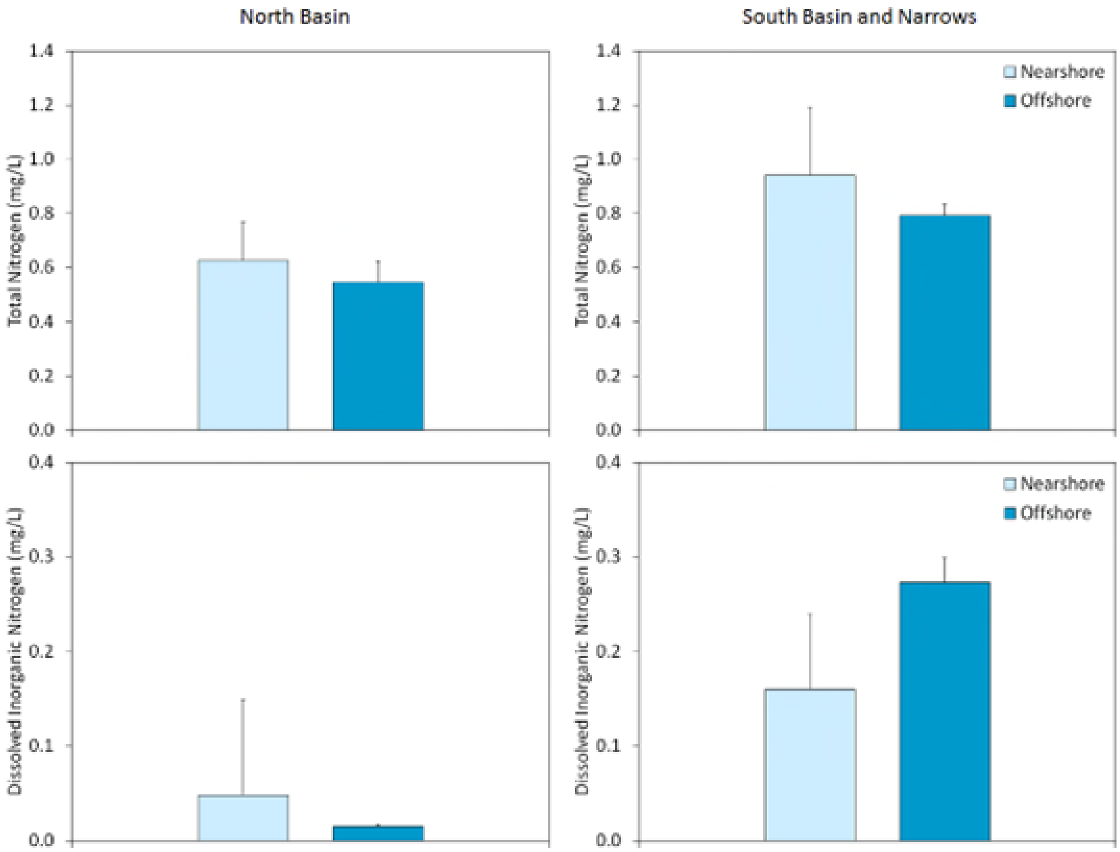


Figure 6-7: Average TN and DIN concentrations (mg/L, ±SE) in the nearshore and offshore of the north basin and south basin and narrows regions of Lake Winnipeg, 2014 to 2016. Note scale change for DIN.

Nutrient Limitation (N:P)

Elemental nutrient ratios of nitrogen and phosphorus are widely used to provide information on the nutrient status of the phytoplankton in marine and fresh waters. Nutrients are typically considered sufficient for growth at a molar nitrogen to phosphorus ratio (N:P ratio) of 16:1 (Redfield et al. 1963), and a departure from the optimal Redfield ratio is generally indicative of nutrient deficiency. When ratios are below 10:1, phytoplankton are generally considered nitrogen deficient, and phosphorus deficiency generally occurs at ratios above 20:1. Phytoplankton community structure is affected by N:P ratios (e.g., Smith 1983), as some species of phytoplankton are either better nitrogen or phosphorus competitors. Cyanobacteria are generally considered better nitrogen competitors, and as such, are usually dominant at low N:P ratios. From 1999 to 2016, the south basin and narrows region of Lake Winnipeg was generally considered either co-limited by both nitrogen and phosphorus or phosphorus limited with average TN:TP ratios ranging from 15:1 to 29:1 (Figure 6-8). Average annual open water N:P ratios were lowest in 2002, coinciding with low TN concentrations (0.48 mg/L). The average N:P molar ratio for the south basin and narrows is 22:1 for the 1999 to 2016 periods. Although N:P ratios indicate that the south basin is generally co-limited by nitrogen and phosphorus or at times nitrogen-limited, phytoplankton in the south basin are constrained by the turbidity and high sediment load transported to the lake from the Red River, which limits the amount of light available for phytoplankton growth. The north basin of Lake Winnipeg is phosphorus limited with an average ratio of 42:1 over the 1999 to 2016 period. However, ratios of N:P were much more variable among years, with ratios ranging from 21:1 to 103:1. TN concentrations are much more variable in the north basin of Lake Winnipeg and phosphorus limitation was most pronounced in 2003 when TN was the highest in the north basin of Lake Winnipeg.

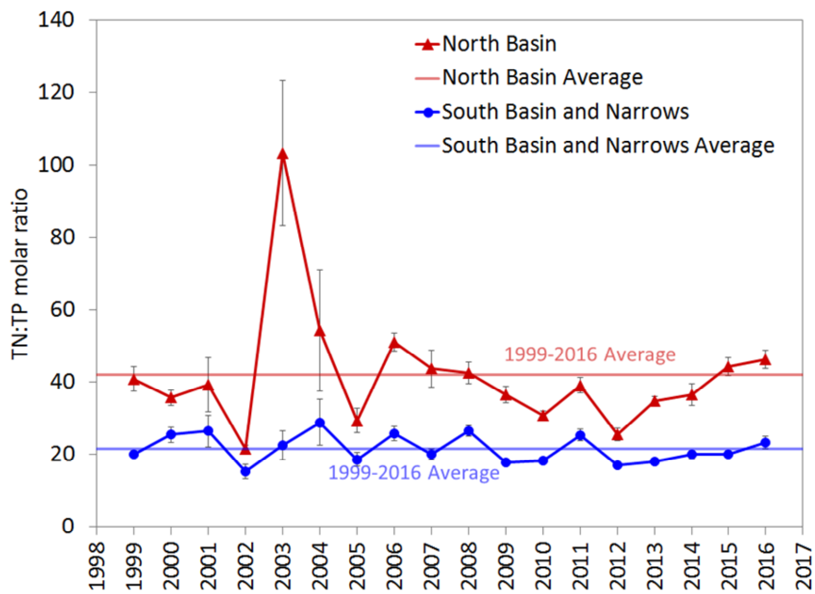


Figure 6-8: Annual open water mean TN:TP molar ratio (± SE) in the north basin and the south basin and narrows of Lake Winnipeg, from 1999 to 2016.

In terms of seasonal changes in the TN:TP ratio, on average, the north basin was phosphorus limited through the spring, summer and fall (Figure 6-9). However, the north basin was highly phosphorus limited (50:1) during the summer period as compared to the spring and fall. Not surprisingly, TP concentrations were relatively low (~0.010–0.015 mg/L) during the summer in the north basin for the 2008–2016 period. The south basin of Lake Winnipeg was phosphorus limited or co-limited by both nitrogen and phosphorus in the spring. Traverse Bay was the most phosphorous-limited in the south basin, with higher N:P ratios (~40:1–50:1) as compared to the rest of the south basin where ratios ranged between 25:1 and 35:1. During fall, the south basin became more nitrogen-limited which was coincident with increasing total phosphorous concentrations later in the open water season (Figure 6-9). Despite low N:P ratios favouring bloom formation in the south basin of Lake Winnipeg, cyanobacteria blooms are less extensive as compared to the north basin of the lake because phytoplankton are light-limited due to the higher turbidity in the south basin.

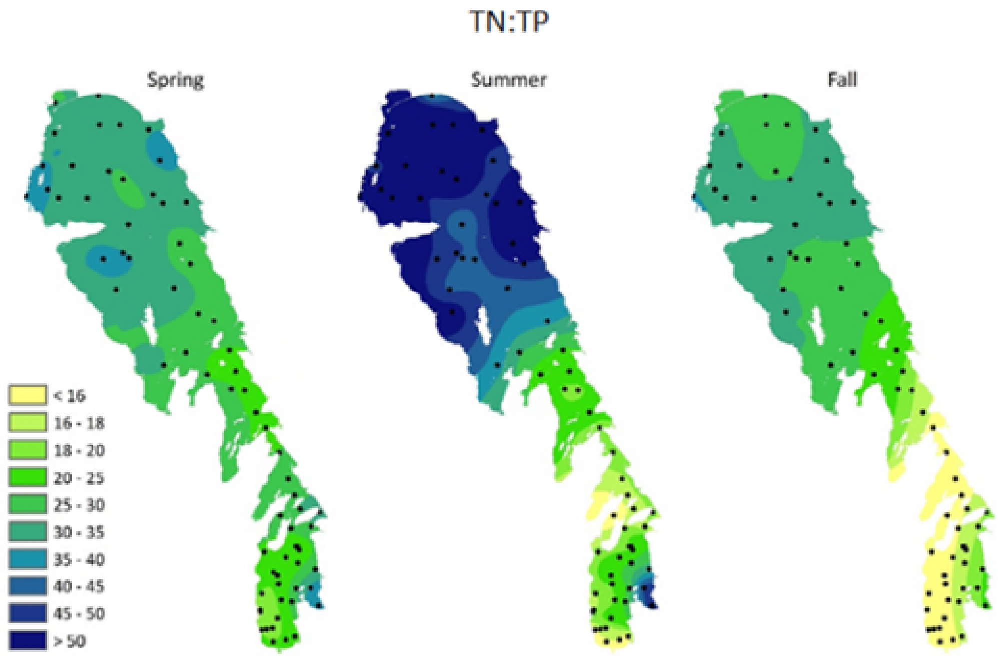


Figure 6-9: TN:TP molar ratio in Lake Winnipeg during the spring, summer and fall, average of the 2008–2016 period.

7.0 NUTRIENT LOADING

By: Brian Wiebe (Manitoba Agriculture and Resource Development)

Nutrient loading is arguably the dominant factor impacting nutrient concentrations and associated eutrophication in Lake Winnipeg. Actual concentrations of nutrients within Lake Winnipeg will vary both spatially and temporally as a result of complex interactions between nutrient additions to the lake, internal lake processes, nutrient losses from the lake and variability in flows both into and out of the lake. Three main tributaries contribute the majority of nutrient loads and flows entering Lake Winnipeg (Figure 7-1). The Red and Winnipeg rivers flow into the south basin and the Saskatchewan River flows into the north basin, and combined, they contribute on average 85% of the flow, 68% of the nitrogen, and 88% of the phosphorus entering the lake. However, several smaller tributaries also contribute significant flows and nutrient loads. These include the Dauphin, Fisher and Icelandic rivers on the west side of the lake, the Black, Manigotagan, Bloodvein, Pigeon, Berens and Poplar rivers on the east side, and the Brokenhead River in the south between the Red and Winnipeg rivers. Both point and diffuse sources contribute nutrients via these tributaries, but because of the large surface area of the lake, atmospheric deposition (nitrogen and phosphorus) as well as nitrogen fixation are also significant

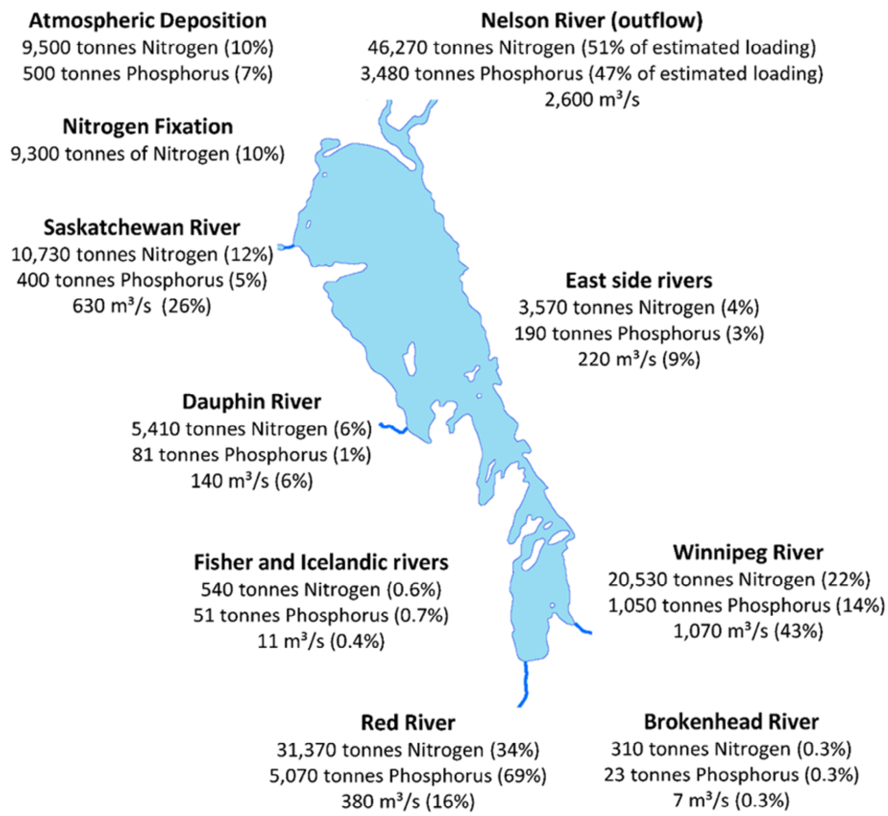


Figure 7-1: Estimated loading of total nitrogen and total phosphorus to Lake Winnipeg (annual averages, 1994–2016).

contributors (Tables 7-1 and 7-2). On average, the Winnipeg River provides over 40% of the flow and about a fifth of the nitrogen entering Lake Winnipeg. In contrast, the Red River contributes almost 70% of the phosphorus and a third of the nitrogen while contributing only about 16% of the flow (Figure 7-1). The Saskatchewan River supplies 26% of the flow while contributing only 5% of the phosphorus and 12% of the nitrogen on average. The Dauphin, Icelandic, Fisher, Brokenhead and east side rivers contribute a further 16% of the flow and provide 11% of the nitrogen and 5% of the phosphorus. The only outlet is the Nelson River, which flows north to Hudson Bay. It is estimated that over the 1994–2016 time period, about half of the nitrogen and phosphorus entering Lake Winnipeg left the lake via outflow to the Nelson River.

Water quality in tributaries to Lake Winnipeg is monitored by Manitoba Agriculture and Resource Development (MARD), Environment and Climate Change Canada (ECCC) and their partners; water quantity is monitored through the Canada-Manitoba Hydrometric Agreement. Water quality and quantity data are used to describe the recent loadings of TP and TN to Lake Winnipeg over the period 1994 to 2016 using methods described in the previous report (EC and MWS 2011).

Phosphorus

Phosphorus loading to Lake Winnipeg shows large annual fluctuations and, as expected, the greatest annual loads are associated with the highest flows entering the lake. On average, the estimated total annual phosphorus contribution to Lake Winnipeg was 7,368 tonnes. The lowest annual load of 3,093 tonnes occurred in 2003 (average flow of 1140 m³/s) while the highest annual loads occurred in the flood years of 1997 (10,581 tonnes), 2005 (10,849 tonnes) and 2011 (10,932 tonnes) with average annual flow rates of 2,780 m³/s, 3,300 m³/s, and 3,340 m³/s, respectively. Phosphorus loading has been near or below the average (1994-2016) for four of the last five years (Figure 7-2) despite flows being above average in 2013, 2014 and 2015 largely due to increased flows in the Saskatchewan River. Flows in the Red River were mostly below average over the same time

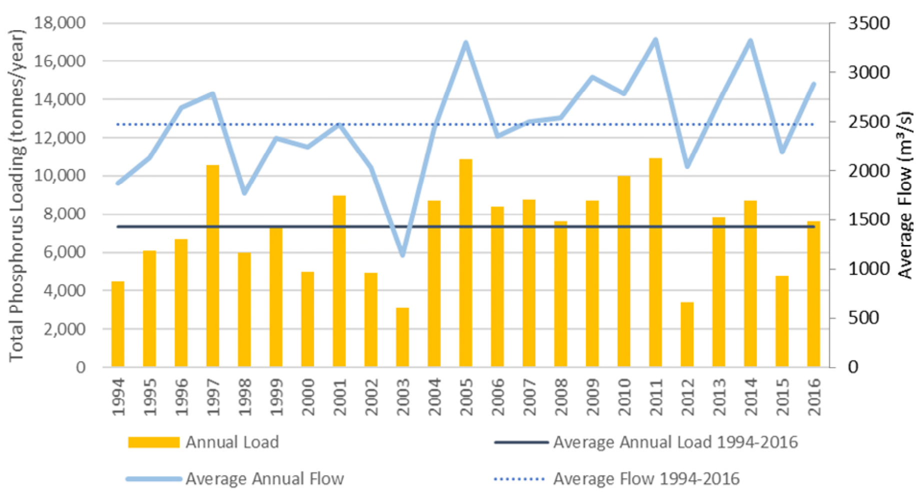


Figure 7-2: Annual estimates of TP loading to Lake Winnipeg.

period. It is clear that flow is a critical variable (flow-load relationship is highly significant in all tributaries with adjusted R^2 values ranging from 0.59 to 0.91) but is not the only factor that impacts loading. The Lake Winnipeg watershed is large and highly variable and it would be expected that different factors would be important in different parts of the watershed.

The Winnipeg River flows out of a watershed dominated by forest and granitic bedrock and is also impacted by several man-made reservoirs (for hydro-electric power generation) and lakes along its tributaries. Although it contributes about half of the flow into Lake Winnipeg, it accounts for only 14% of the annual phosphorus load. The Flow Weighted Mean Concentration (FWMC) was 0.031 mg/L, an order of magnitude less than the Red River (Table 7-1). The combination of lower development, low phosphorus applications to the watershed (majority of watershed is natural forest), and several reservoirs, lakes and wetlands, which allow for settling of suspended particles (and associated phosphorus) all contribute to the much lower FWMC in the Winnipeg River. The range between maximum and minimum annual load is about 1,000 tonnes, with the maximum in 2004 (1,433 tonnes) the year after the minimum (448 tonnes). The four-year moving average of loads shows some fluctuation but is consistently near the mean, which suggests that little has changed in the watershed between 1994 and 2016. Except for 2014 and 2016 (the two years with the highest flows), the Winnipeg River has been below its average load since 2011 (Figure 7-3).

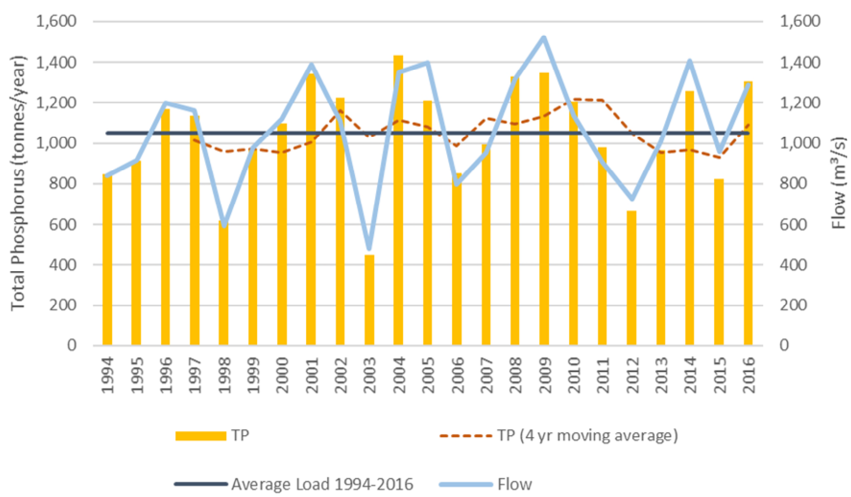


Figure 7-3: Annual estimates of TP loading from the Winnipeg River.

The Red River is the largest single source of phosphorus into Lake Winnipeg accounting for an average of 69% of phosphorus loading to the lake despite contributing on average only 16% of the flow. FWMC of phosphorus (0.418 mg P/L) is an order of magnitude greater than most other tributaries (1994–2016) and both flows and loads are much more variable in the Red River. The difference between minimum and maximum annual loads from the Red River is almost 7,000 tonnes (Table 7-1). The minimum load of 1,337 tonnes occurred in 2012, while the maximums occurred in the flood years of 1997 (8,176 tonnes), 2005 (8,024 tonnes) and 2011 (8,109 tonnes). The relationship between flow and load is also much less consistent than for the Winnipeg River (Figure 7-4). This is not surprising considering the greater variability within the watershed, which

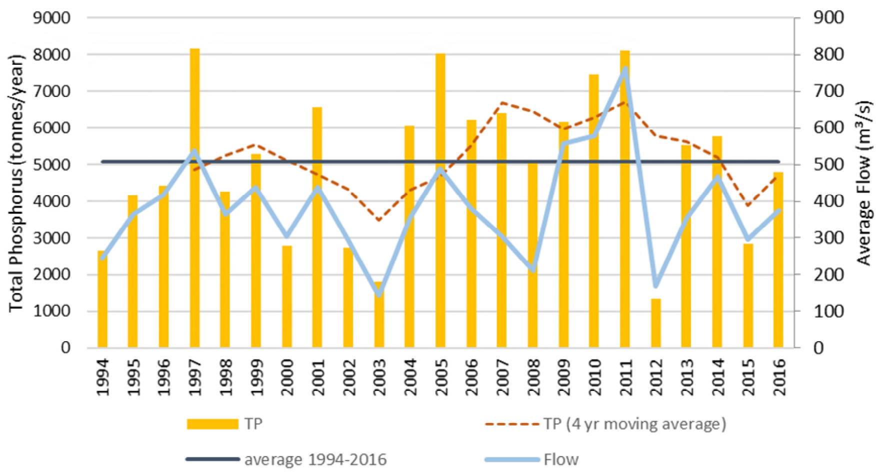


Figure 7-4: Annual estimates of TP loading from the Red River.

includes both the Red and Assiniboine rivers. The watershed includes large regions of intensive agriculture as well as other areas dominated by perennial pasture, hay-land and natural areas.

The Assiniboine River is also influenced by several large reservoirs as well as many smaller lakes and wetlands, which can remove phosphorus from surface water. This regional difference within the watershed can be demonstrated by comparing the three flood years. Figure 7-4 shows very similar flow-load relationships for 1997 and 2005. Both floods were predominantly in the Red River Valley and adjacent to the Red River channel. The 1997 flooding occurred during the spring snowmelt/freshet period and the 2005 event occurred in June–July due to prolonged heavy rainfall after many crops had been planted, yet the relationship is similar. The 2011 flooding began with the spring freshet, but heavy rains in the Assiniboine River watershed in both Manitoba and Saskatchewan resulted in high flows and flooding continued through June, July and into August. Despite flows being about 50% higher in 2011, phosphorus loading was very similar to 1997 and 2005. The lower phosphorus concentration in 2011 was likely the result of differences on the landscape (e.g. soils, topography, agricultural practices) along with the ameliorating effect of reservoirs, lakes and wetlands. The load was below average in 2016 and has been below average for three of the last five years (Figure 7-4).

The Saskatchewan River contributed on average 26% of the flow but only 5% of the phosphorus load to Lake Winnipeg with an average FWMC of 0.020 mg P/L (Figure 7-1 and Table 7-1). Loads ranged from 90 tonnes in 2001 to a high of 822 tonnes in 2011 (flows 200 m^3/s and 1050 m^3/s, respectively). Increased precipitation in the watershed has resulted in both flows and loads increasing dramatically in 2005 and remaining higher than average with the exception of 2008, 2009 and 2015 (Figure 7-5). The four-year moving average of loading mirrors the changes in flow. In 2016, the Saskatchewan River contributed about 9% of the TP loading to Lake Winnipeg. The Saskatchewan River watershed contains large areas of intensive agriculture, not unlike the Red River watershed, but the presence of several reservoirs and lakes along the Saskatchewan River as well as flow regulation for hydroelectric generation, are likely the main reasons for the much lower FWMC and hence lower phosphorus loading.

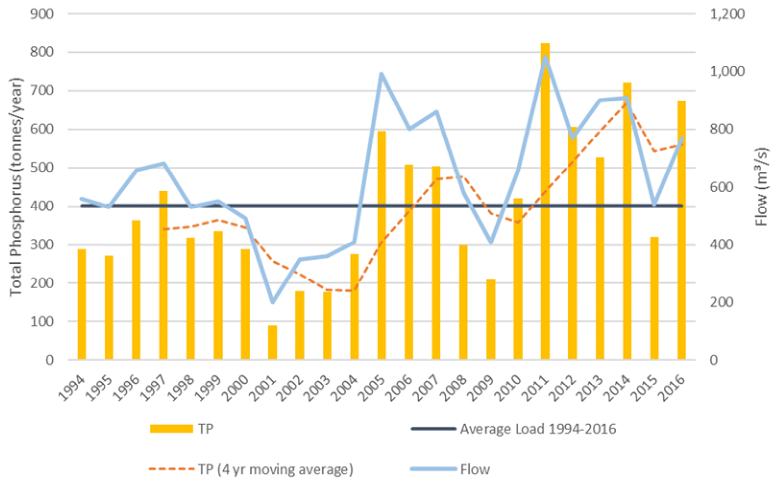


Figure 7-5: Annual estimates of TP loading from the Saskatchewan River

Table 7-1: Estimated annual phosphorus loading (tonnes/year) to Lake Winnipeg from major tributaries and atmospheric deposition.

	Red River	Winnipeg River	Sask. River	Dauphin River	East Side Rivers	Broken-head River	Fisher River	Icelandic River	Atmos. Dep.	Total load	Outflow Nelson River
1994	2,661	846	289	18	135	15	2	2	500	**4,467**	1,536
1995	4,165	912	271	55	152	13	5	7	500	**6,079**	1,754
1996	4,420	1,171	362	79	154	10	8	17	500	**6,723**	2,055
1997	8,176	1,135	440	89	188	14	18	22	500	**10,581**	1,987
1998	4,266	619	318	65	116	13	43	38	500	**5,977**	2,249
1999	5,278	968	336	80	184	5	11	8	500	**7,371**	1,813
2000	2,783	1,098	288	41	182	16	34	37	500	**4,980**	1,989
2001	6,559	1,343	90	74	233	18	76	51	500	**8,944**	4,535
2002	2,743	1,228	179	18	248	12	6	7	500	**4,942**	3,358
2003	1,826	448	177	15	113	6	4	5	500	**3,093**	1,849
2004	6,052	1,433	275	15	235	43	113	37	500	**8,703**	2,313
2005	8,024	1,211	594	99	256	67	38	58	500	**10,849**	6,343
2006	6,211	851	507	101	160	21	13	29	500	**8,393**	4,211
2007	6,402	996	503	53	219	54	20	19	500	**8,767**	3,620
2008	5,116	1,331	300	91	219	16	28	30	500	**7,631**	3,946
2009	6,168	1,348	210	91	220	40	84	64	500	**8,727**	5,298
2010	7,455	1,206	421	103	186	66	38	15	500	**9,990**	5,082
2011	8,109	980	822	247	176	36	32	30	500	**10,932**	7,206
2012	1,337	668	606	114	170	3	2	1	500	**3,400**	4,137
2013	5,529	966	527	89	172	6	12	10	500	**7,812**	3,273
2014	5,779	1,261	721	126	219	35	27	27	500	**8,693**	3,712
2015	2,833	825	319	100	171	10	7	11	500	**4,776**	2,531
2016	4,780	1,304	673	105	225	16	11	17	500	**7,632**	5,273
average	5,073	1,050	401	81	188	23	27	24	500	**7,368**	3,481
FWMC (mg/L)	0.418	0.031	0.02	0.018	0.027	0.101	0.123	0.174			0.042

The Dauphin River has also experienced increased flows since 2005 (with the exception of 2007) contributing annual phosphorus loads above its long-term average (Figure 7-6). The majority of the flow in the Dauphin River originates from Lake Manitoba and its flow is regulated by the Fairford dam. The watershed includes both agricultural and natural (mostly forested) areas but drains into either Lake Winnipegosis and then into Lake Manitoba or directly into Lake Manitoba. The FWMC of the Dauphin River (0.018 mg P/L) is very similar to the Saskatchewan River, which is consistent with the fact that most of the water has flowed through substantial lake systems prior to entering the Dauphin River. As a result, even though the watershed is large and contributes 6% of the flow, it only contributes 1% of the annual phosphorus loads. Annual loading ranged from 15 to 247 tonnes of phosphorus with the higher values mostly seen since 2005 as high inflows into Lake Winnipegosis and Lake Manitoba resulted in higher flows in the Dauphin River.

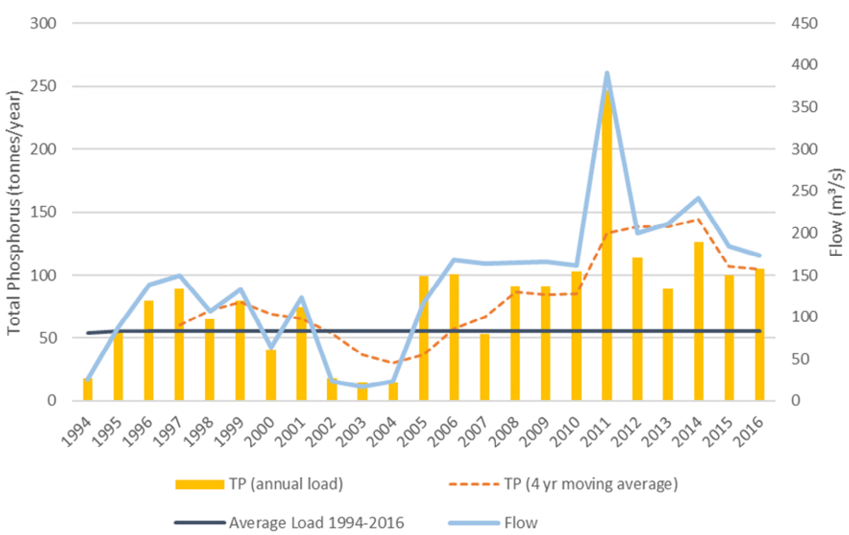


Figure 7-6: Annual estimates of TP loading form the Dauphin River.

The Fisher, Icelandic and Brokenhead rivers contribute about 1% of the annual phosphorus load to Lake Winnipeg while contributing a little less than 1% of the flow. These smaller tributaries have FWMC values intermediate between the Red River and the other three larger rivers. FWMC were 0.123 mg P/L for the Fisher River, 0.174 mg P/L for the Icelandic River, and 0.101 mg P/L for the Brokenhead River (Table 7-1). Their watersheds are largely agricultural, but the agriculture is less intensive than that of the Red River watershed, and although they also do not have lakes or reservoirs on their channels, their watersheds contain many wetlands, which help reduce phosphorus entering these tributaries.

The other east-side rivers (the Black, Manigotagan, Bloodvein, Pigeon, Berens, and Poplar rivers) are no longer monitored for chemistry and their contributions have been estimated based on flow and concentration relationships with the Winnipeg River. Their estimated contribution is 188 tonnes of phosphorus (3% of average annual loading) while contributing 9% of annual flow. Because they are based on the Winnipeg River values, they like the Winnipeg River, show annual variation with changes in flows but a consistent flow-load relationship.

Direct atmospheric deposition has been estimated (see ECCC and MWS 2011) to contribute 500 tonnes of phosphorus annually to Lake Winnipeg. This is about 7% of the annual load on average and is the third largest contributor after the Red and Winnipeg Rivers.

The relationship between phosphorus loads and flows is apparent, but as the variability between the various tributaries demonstrates, conditions on the landscape and within the watershed as well as the timing of runoff events, can also impact the flow-load relationship. To capture the variability in concentration during runoff events, MARD increased sampling frequency during the spring freshet as well as other high flow periods beginning in 2009. Prior to 2009, only major flood events were subject to increased sampling frequency. Figure 7-7 shows sampling dates, flows and phosphorus concentrations in the Red River at Selkirk for 1996 and 1997. The 1997 higher frequency sampling captures much more of the variability in phosphorus concentrations during the spring runoff period compared to the monthly sampling used in 1996. For consistency and to be better able to compare annual loading estimates pre- and post-2009, monthly average concentrations and flows were used to calculate loads for the entire 1994–2016 period where monthly (or more frequent) concentration data was available.

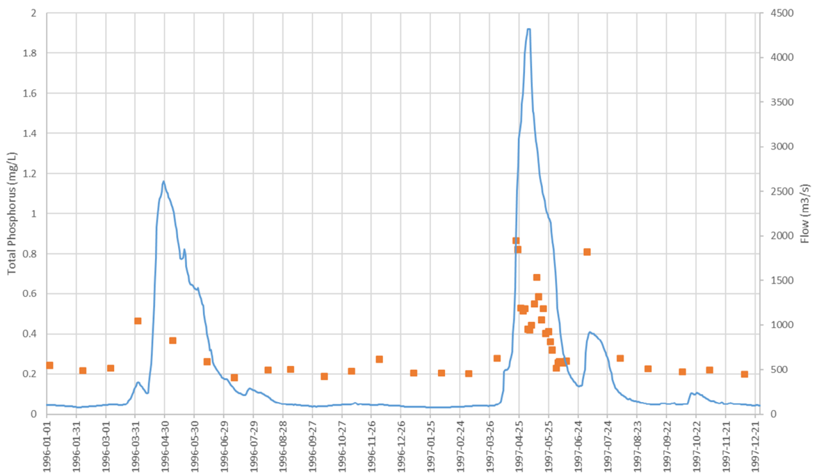


Figure 7-7: TP concentration and flows, Red River at Selkirk, 1996–1997.

The Nelson River is the only outlet to Lake Winnipeg and the average FWMC of the outflow was 0.042 mg P/L (1994–2016, Table 7-1), an order of magnitude less than that of the Red River but about 1.4 times greater than the Winnipeg River and double that of the Saskatchewan River. It is estimated that about 47% of the phosphorus entering Lake Winnipeg exits via the Nelson River. The remainder of the phosphorus is retained within the lake due to a variety of in-lake processes.

Nitrogen

As with phosphorus, nitrogen loading to Lake Winnipeg shows large annual fluctuations in response to changes in flows (flow-load relationship is highly significant in all tributaries with adjusted R^2 values ranging from 0.67 to 0.95). Estimated average annual loading of nitrogen to Lake Winnipeg was 91,263 tonnes; 72,463 tonnes was contributed by the various tributaries and the remaining 18,800 tonnes from atmospheric deposition and nitrogen fixation. The lowest loading occurred in 2003 (31,280 tonnes N) and the highest annual load of 136,676 tonnes occurred in 2011 (average annual flows of 1,140 m³/s and 3,340 m³/s, respectively). Loads have been below average (1994–2016) for three of the last five years (Figure 7-8).

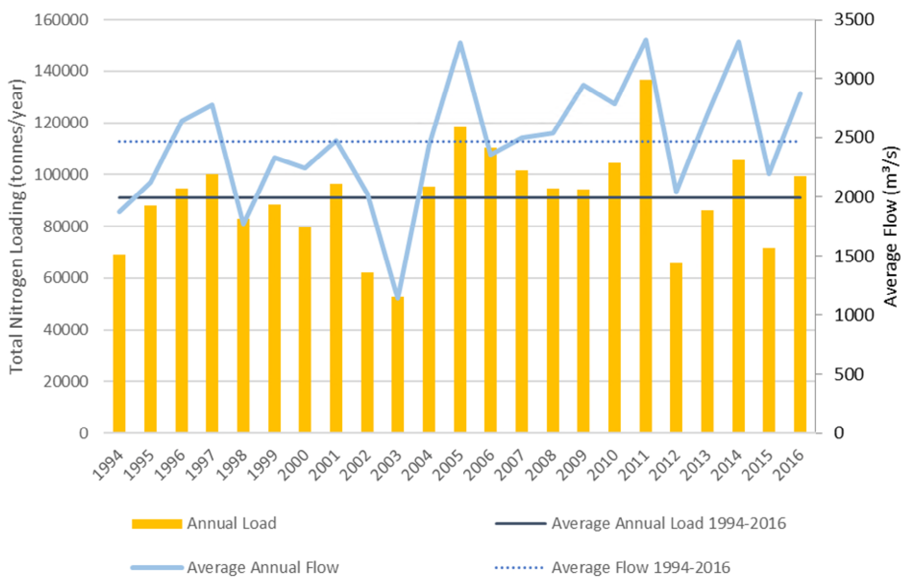


Figure 7-8: Annual estimates of TN loading to Lake Winnipeg.

The nitrogen load in the Red River was near average in 2016, and except for 2014, has been at or below average since 2012 (Figure 7-9). The Saskatchewan River has experienced higher than average flows in recent years and except for 2015, has contributed above average nitrogen loads (Figure 7-10). The Dauphin River has also experienced increased flows since 2006, and except for 2010, has contributed above average annual nitrogen loads (Figure 7-11). The impact of increased runoff entering lakes Manitoba and Winnipegosis are clearly observed in higher flows and loads entering Lake Winnipeg via the Dauphin River since 2005.

The nitrogen loads from the Winnipeg River have been at or below average since 2010, 2016 being the sole exception (Figure 7-9). Although it contributes about 14% of the phosphorus load to Lake Winnipeg, it accounts for 22% of the nitrogen annual load. This higher contribution can be seen in the average FWMC, which is about one-quarter that of the Red River (0.061 mg/L – Table 7-2). The average annual contribution was 20,530 tonnes of nitrogen, which ranged from a minimum annual load of 12,711 tonnes in 1998 to a maximum in 2008 of 33,422 tonnes. The nitrogen load-flow relationship is less consistent than for phosphorus. The four-year moving average of loads shows above average values from 2004 through 2011 with below average values from 2012 through 2016 despite flows maintaining a fairly consistent fluctuation about the mean (Figure 7-9). The flow-load relationship appears to have changed beginning in 2009.

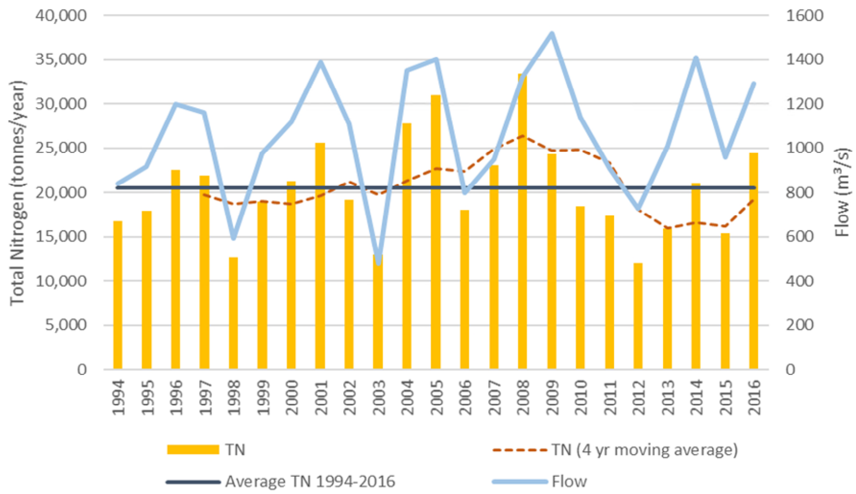


Figure 7-9: Annual estimates of TN loading from the Winnipeg River.

Table 7- 2: Estimated annual nitrogen loading (tonnes/year) to Lake Winnipeg from major tributaries, atmospheric deposition, and nitrogen fixation.

	Red River	Winnipeg River	Sask. River	Dauphin River	East Side Rivers	Broken-head River	Fisher River	Icelandic River	Atmos. Dep.	Nitrogen Fixation	Total load	Outflow
1994	22,121	16,754	6,938	1,221	3,079	251	21	19	9,500	9,300	**69,204**	26,077
1995	36,370	17,932	7,016	4,029	3,488	159	51	69	9,500	9,300	**87,913**	32,793
1996	34,577	22,536	9,467	5,288	3,545	173	93	181	9,500	9,300	**94,658**	45,385
1997	37,871	21,884	11,654	5,465	3,794	197	201	228	9,500	9,300	**100,094**	54,494
1998	35,303	12,711	8,627	4,004	2,306	192	480	388	9,500	9,300	**82,809**	36,059
1999	32,836	18,942	9,074	4,908	3,520	82	123	85	9,500	9,300	**88,371**	30,320
2000	24,479	21,258	7,889	2,515	3,701	346	382	381	9,500	9,300	**79,752**	41,886
2001	37,755	25,573	3,037	4,560	4,864	507	843	522	9,500	9,300	**96,460**	55,737
2002	13,172	19,159	5,167	1,117	4,292	390	67	73	9,500	9,300	**62,237**	24,379
2003	11,930	13,028	5,440	880	2,188	117	40	47	9,500	9,300	**52,470**	28,704
2004	34,163	27,782	7,026	1,406	4,104	442	1263	380	9,500	9,300	**95,366**	39,103
2005	39,205	30,929	16,211	6,844	4,686	828	425	600	9,500	9,300	**118,529**	59,099
2006	42,403	18,048	16,030	10,773	3,115	369	310	338	9,500	9,300	**110,186**	55,170
2007	34,579	23,061	13,535	7,334	3,427	529	235	147	9,500	9,300	**101,647**	38,774
2008	18,294	33,422	11,938	7,496	3,996	175	303	295	9,500	9,300	**94,719**	71,190
2009	34,444	24,297	5,329	5,977	4,243	354	313	495	9,500	9,300	**94,252**	51,842
2010	45,973	18,444	11,180	5,078	3,425	766	604	281	9,500	9,300	**104,550**	57,669
2011	63,781	17,410	19,177	13,255	3,245	359	373	277	9,500	9,300	**136,676**	75,951
2012	11,187	12,084	13,948	6,867	2,820	43	49	14	9,500	9,300	**65,812**	40,904
2013	26,695	15,942	14,502	6,862	3,281	89	145	103	9,500	9,300	**86,419**	40,874
2014	33,573	21,100	20,336	6,862	4,145	258	353	237	9,500	9,300	**105,663**	54,026
2015	19,395	15,430	9,167	5,717	2,899	146	75	123	9,500	9,300	**71,753**	44,456
2016	31,503	24,461	14,101	6,028	3,918	291	196	200	9,500	9,300	**99,497**	59,340
average	31,374	20,530	10,730	5,412	3,569	307	302	238	9,500	9,300	**91,263**	46,271
FWMC (mg/L)	2.59	0.61	0.54	1.22	0.51	1.34	1.35	1.77			**1.17**	0.56

TEXT BOX 2: The Lake Winnipeg Community-Based Monitoring Network

By: Alexis Kanu and Chelsea Lobson (Lake Winnipeg Foundation)

Community-based water monitoring (CBWM) programs are becoming increasingly prevalent across Canada in response to a growing need for data to understand a changing environment (WWF 2017). A range of CBWM initiatives exists, designed in some cases to create education and engagement opportunities, and in others to generate robust data to support local and regional decision-making and water management.

The use of citizen-generated data to complement government and industry water monitoring programs and inform policy decisions is an increasingly common strategy. Internationally, the United States Environmental Protection Agency's (US EPA) external advisory group, the National Advisory Council for Environmental Policy and Technology, recommended that the agency integrate CBWM into their data collection activities and subsequently use these data in cross-sectoral planning and decision-making (USEPA 2006). In Canada, the Northwest Territories' Water Strategy successfully integrates CBWM data with agency datasets to support decision-making at multiple levels (GNWT and INAC 2010). In Manitoba, CBWM groups have been active for years, although they have generally operated independently of each other. In 2015, the Lake Winnipeg Foundation (LWF) facilitated the building of the Lake Winnipeg Community-Based Monitoring Network (LWCBMN), with the primary goal of generating credible phosphorus concentration data to inform water quality management decisions.

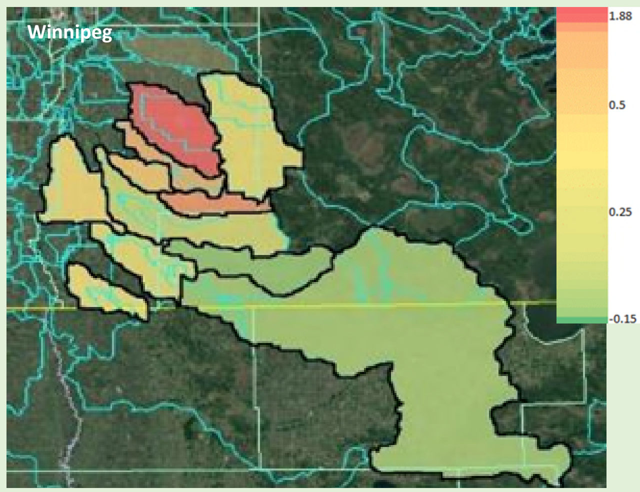


Figure A: TP export coefficients (kg/ha/year) for sub-watersheds in the Seine-Rat River Conservation District in 2017.

Network sampling and analysis protocols, developed by LWF's science advisors, test for total and dissolved phosphorus using collection methods and lab analyses that are compatible with those of provincial and federal agencies. These existing agency datasets provide a valuable overview of TP loading to Lake Winnipeg from major tributaries. The LWCBMN data complements agency data at the sub-watershed level by increasing sampling frequency during peak-flow events to identify "phosphorus hotspots" – localized areas contributing a larger proportional phosphorus load to Lake Winnipeg. For example, in 2017 in the Seine-Rat River Conservation District, the LWCBMN found TP export coefficients ranging from 0.05 to 1.64 kg/ha/year (Figure A). High spatial variation in TP exports within this district highlights the value and significance of sampling frequently at multiple sites with the support of a spatially dispersed network of partners and volunteers.

Continued next page

Text Box 2 continued

The LWCBMN was piloted in 2016 through a partnership between LWF and the Seine-Rat River and La Salle-Redboine conservation districts, with funding from the Manitoba government. Twelve sites across the two conservation districts were sampled from April to October, with 200 samples collected. Since the pilot project in 2016, the LWCBMN has grown rapidly. In 2018, the network had expanded to include nine conservation district partners and 41 citizen volunteers, collecting samples at over 100 sites. Government partners, including Manitoba Agriculture and Resource Development and Agriculture and Agri-Food Canada, provide advice to the network through the LWCBMN steering committee, while AAFC also provides in-kind support for lab analyses.

To support overall network co-ordination, LWF provides each partner with shared protocols, equipment and training; notifies partners of optimal sampling times based on spring melt, extreme weather events and a regular spring/summer schedule; analyzes all water samples for total and dissolved phosphorus concentrations; and provides raw data and contextualized information back to each partner.

Conservation district partners and volunteers participate in training and commit to using approved network protocols and equipment; collect samples regularly and in response to high-water events; and submit samples and corresponding metadata to LWF for analysis. Participation in the LWCBMN provides conservation districts and community volunteers with access to credible lab analyses and an aggregated dataset shared online annually.

LWCBMN sampling sites are located at Water Survey of Canada (WSC) metered flow stations. By linking phosphorus concentration with WSC hydrometric data, the CBWM program can determine phosphorus load and export coefficients for each sampling site. Furthermore, WSC hydrometric data is updated in real-time on the WSC website so volunteers and partners can be notified when a peak-flow event is occurring at their site, enabling a flow-based sampling schedule (Figure B).

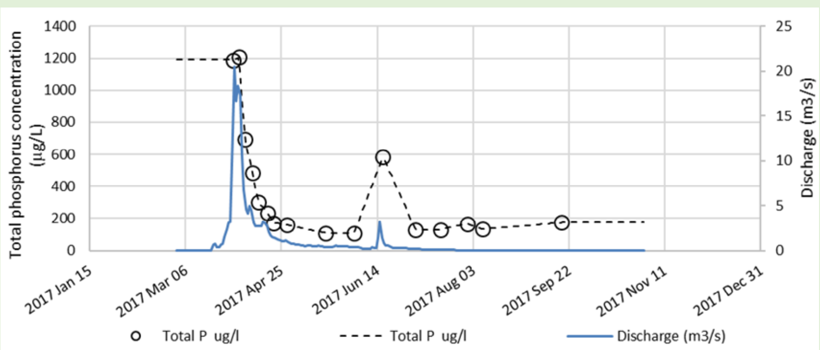


Figure B: Flow-based sampling schedule at the Roseisle Creek in 2017.

The LWCBMN is committed to ensuring that all data collected is shared openly in a timely manner. All network partners and volunteers receive their sampling results, as well as the broader network results, in a format that enables them to use what they have learned in local and regional decision-making. For example, conservation district partners are using LWCBMN data to prioritize projects in watersheds where high phosphorus exports are being reported. LWCBMN data can also be used to identify and quantify the phosphorus reductions achieved through remedial action, and evaluate return on investment.

As with phosphorus, the Red River is also the largest single source of nitrogen into Lake Winnipeg, but its role is much less dominant accounting for on average 34% of nitrogen loading to the lake while contributing on average 16% of the flow. The FWMC (2.59 mg N/L), though still the highest among tributaries is only about double that of the Dauphin, Brokenhead, Icelandic and Fisher rivers and four to five times greater than the Saskatchewan, the Winnipeg and other eastern rivers. The load was very close to the average in 2016, and has been at or below average for the last five years (Figure 7-10). The loading in 2011 was by far the highest (63,781 tonnes) since 1994, almost 20,000 tonnes higher than the next highest year (2010) and almost 25,000 tonnes greater than the 2005 flood year. The minimum annual values, 11,930 tonnes (2003) and 11,187 tonnes (2012), are more than 60,000 tonnes less than the maximum in 2011.

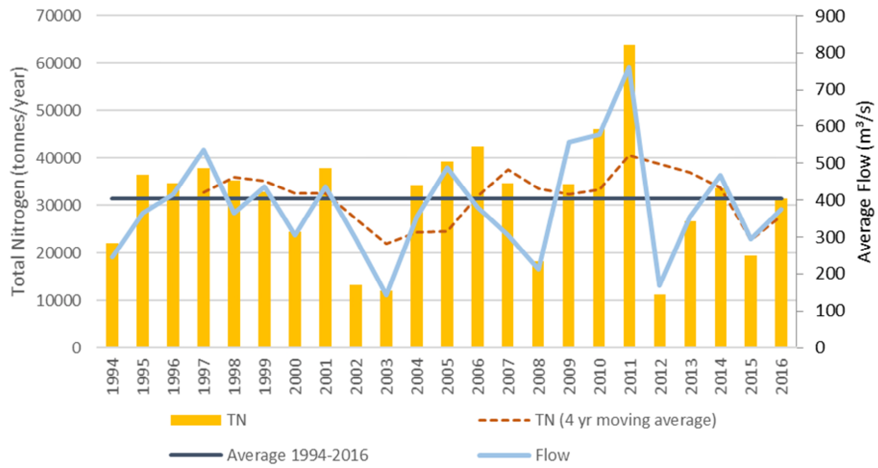


Figure 7-10: Annual estimates of TN loading from the Red River.

The relationship between flow and load is variable (Figure 7-10) with several years (1995, 2002, 2006, 2007, 2009, 2011) standing out with loads either higher or lower than one would expect from the annual flow alone. The great variability within the watershed, which includes both the Red and Assiniboine rivers, and timing of runoff events may be responsible in part for this variability. The watershed includes large regions of intensive agriculture as well as other areas dominated by perennial pasture, hay-land and natural areas. The impact of reservoirs, lakes and wetlands would also be expected to affect nitrogen differently than phosphorus. Comparing nitrogen and phosphorus loading during the three flood years of 1997, 2005 and 2011, demonstrates this difference. Despite much higher flows in 2011, phosphorus loading was very similar in all three years (Figure 7-4). The higher flows from the Assiniboine did not result in a comparable increase in phosphorus loading. However, in regard to nitrogen, the three years respond very similarly to flow with the high flows from the Assiniboine River basin resulting in a corresponding high nitrogen load (Figure 7-10). The differences in watersheds and timing appear to have had a much smaller impact on the flow-load relationship for nitrogen than phosphorus loading.

The Saskatchewan River contributed on average 12% of the nitrogen load and 26% of the flow to Lake Winnipeg (Figure 7-11). Loads ranged from 3,037 tonnes in 2001 to 20,336 tonnes in 2014 (flows 200 m³/s and 910 m³/s, respectively) (Table 7-2 and Figure 7-11). Increased precipitation in the watershed has resulted in both flows and loads increasing dramatically in 2005 and remaining higher than average with the exception of 2009 and 2015 (Figure 7-11). The four-year moving average of loading mirrors the changes in flow. In 2016, the Saskatchewan River contributed about 14% of the TN loading to Lake Winnipeg. The Saskatchewan River watershed contains large areas of intensive agriculture, not unlike the Red River watershed, but the presence of several reservoirs and lakes along the Saskatchewan River, as well as flow regulation for hydroelectric generation, are likely the main reasons for the lower FWMC (0.54 mg N/L), hence lower nitrogen loading in comparison to the Red River.

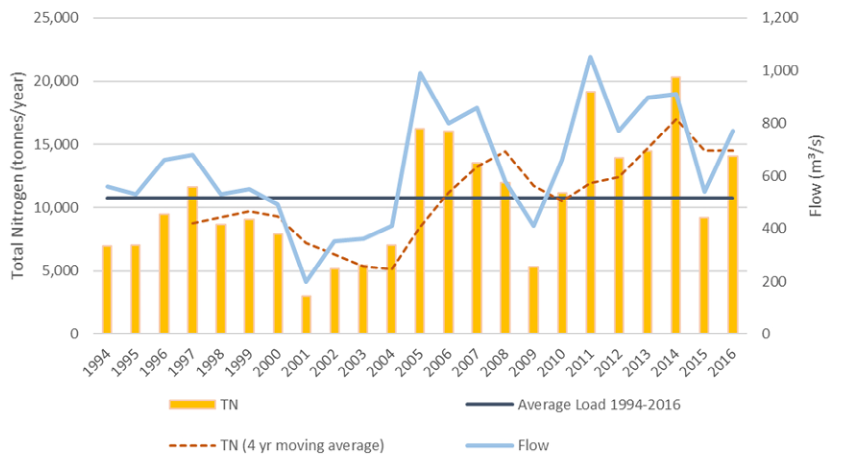


Figure 7-11: Annual estimates of TN loading from the Saskatchewan River.

The Dauphin River has also experienced increased flows since 2005 as high inflows into Lake Winnipegosis and Lake Manitoba resulted in higher flows in the Dauphin River. With the exception of 2010, the Dauphin River has contributed annual nitrogen loads above its long-term average beginning in 2005 (Figure 7-12). Annual loading ranged from 880 to 13,255 tonnes of nitrogen with the higher values mostly seen since 2005 (Table 7-2). Although the flow-load relationship is fairly consistent, the years 2005 through 2008 are notable in

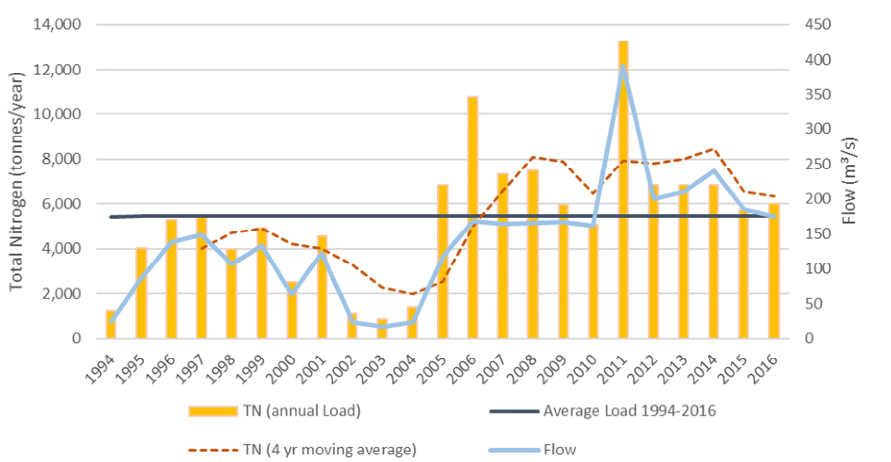


Figure 7-12: Annual estimates of total nitrogen loading from the Dauphin River.

that the nitrogen loads are higher than other years with similar or even somewhat higher flows (2012 through 2016). The watershed includes both agricultural and natural (mostly forested) areas but drains into either Lake Winnipegosis and then into Lake Manitoba or directly into Lake Manitoba before flowing into the Dauphin River. Its flow is regulated by the Fairford dam. Unlike phosphorus, the FWMC of nitrogen in the Dauphin River (1.22 mg/L) is about double that of the Saskatchewan River, despite also travelling through considerable lake systems prior to entering Lake Winnipeg. However, its FWMC is comparable to the Fisher and Brokenhead rivers, which have similar land use. The watershed is large and contributes 6% of the flow and 6% of the annual nitrogen loads.

The Fisher, Icelandic and Brokenhead rivers contribute about 1% of the annual nitrogen load to Lake Winnipeg while contributing a little less than 1% of the flow. These smaller tributaries have FWMC values intermediate between the Red River and the Saskatchewan River and very similar to the Dauphin River. FWMC were 1.35 mg N/L for the Fisher River, 1.77 mg N/L for the Icelandic River, and 1.34 mg N/L for the Brokenhead River (Table 7-2). Their watersheds are largely agricultural, but agricultural production is less intensive than that of the Red River watershed and, although they also do not have lakes or reservoirs on their channels, their watersheds contain many wetlands, which help reduce nitrogen entering these tributaries.

The other east-side rivers (Black, Manigotagan, Bloodvein, Pigeon, Berens and Poplar) are no longer monitored for chemistry and their contributions have been estimated based on flow and concentration relationships with the Winnipeg River. Their estimated contribution (Table 7-2) is 3,569 tonnes of nitrogen (4% of average annual loading) while contributing 9% of annual flow (Figure 7-1). Because they are estimated from Winnipeg River values, they show similar annual variation and flow-load relationships.

Collecting river water samples from a bridge.

Due to the large surface area of Lake Winnipeg, nitrogen fixation and direct atmospheric deposition are also substantial contributors to nitrogen loading. Direct atmospheric deposition has been estimated (EC and MWS 2011) to contribute 9,500 tonnes of nitrogen annually to Lake Winnipeg. Nitrogen fixation is estimated to contribute on average 9,300 tonnes of nitrogen annually. Each of these is about 10% of the annual load on average and are the fourth and fifth largest contributors after the Red, Winnipeg and Saskatchewan rivers.

The average FWMC of the lake's outflow (0.056 mg N/L) to the Nelson River was very similar to the Saskatchewan, Winnipeg and other east side rivers (1994–2016, Table 7-2). On average the Nelson River carried 46,271 tonnes of nitrogen out of Lake Winnipeg each year (about 51% of the load entering the lake). The remainder of the nitrogen is either lost to the atmosphere or retained within the lake due to a variety of in-lake processes.

8.0 PHYTOPLANKTON

By: Elaine Page (Manitoba Agriculture and Resource Development)

Phytoplankton are useful indicators of nutrient enrichment in aquatic ecosystems because of their relatively short life cycles, which are integrative of water chemistry conditions (Makarewicz 1993, Munawar and Munawar 1982). In Lake Winnipeg, one of the most visible symptoms of nutrient enrichment is the development of large blooms of cyanobacteria in the north basin of the lake. Dense blooms also wash up along beaches in the south basin and are evident in the offshore areas of the south basin. In addition to bloom formation during the summer months, noticeable changes have occurred during the ice-cover season with algae coating fishers' nets and making nets more visible to fish. It is well recognized that algal blooms have occurred historically in Lake Winnipeg. However, there is evidence to indicate that bloom frequency and intensity has increased over time. Many of the bloom-forming cyanobacteria in Lake Winnipeg (for example *Aphanizomenon*, *Anabaena* and *Microcystis*) have the potential for toxin production. Some of these toxins have been detected at elevated concentrations in cyanobacterial blooms in Lake Winnipeg.

Collecting phytoplankton samples.

From 1999 to 2016, euphotic zone water samples were collected from 14 water quality monitoring stations (*see Figure 1-3*) as a part of Manitoba Agriculture and Resource Development's long-term water quality monitoring program on Lake Winnipeg. Samples were analyzed for phytoplankton species composition, biomass and chlorophyll-a and are summarized below. In addition to phytoplankton monitoring, Manitoba also routinely monitors for the algal toxin microcystin in the nearshore and offshore areas of Lake Winnipeg to provide information on recreational water quality at beaches in the south basin and to assess the toxicity of blooms of cyanobacteria in the lake. Microcystin data are summarized and presented for the 1999–2018 period.

Phytoplankton Biomass and Species Composition

From 1999 to 2016, the average lake-wide biomass was dominated by cyanobacteria (Cyanophyceae), which comprised 51% of the mean total biomass during the open water period in the euphotic zone of the lake (Table 8-1). Similar to the 1999–2007 reporting period,

Aphanizomenon, *Anabaena* and *Microcystis* accounted for the majority of the total cyanobacteria biomass in Lake Winnipeg. Diatoms (Bacilliariophyceae) were the next largest group comprising 34% of the mean total biomass during the open water period from 1999 to 2016. Large centric diatoms including *Aulacoseira* and *Stephanodiscus* comprised a predominant fraction of the total diatom biomass in Lake Winnipeg. The relative contribution of other phytoplankton in the euphotic zone of Lake Winnipeg was low in comparison to cyanobacteria and diatoms, with the mean relative contribution of the Cryptophyceae, Chlorophyceae, Chrysophceae, Dinophceae and Euglenophyceae ranging between about 0.2 to 8% of the total mean biomass in Lake Winnipeg during the open water period (Table 8-1).

Table 8-1: Summary of the mean phytoplankton biomass (mg/m^3) and relative composition (%) from the euphotic zone of Lake Winnipeg (May to October), 1999–2016.

Class	Mean Phytoplankton Biomass (mg/m^3)	Relative Composition (%)
Bacillariophyceae	1799	34%
Chlorophyceae	149	3%
Chrysophyceae	105	2%
Cryptophyceae	432	8%
Cyanophyceae	2658	51%
Dinophyceae	74	1%
Euglenophyceae	13	0.20%

Phytoplankton species composition and biomass varied considerably over the 1999–2016 period (Figure 8-1) with more than a 60-fold difference between years with low phytoplankton biomass (319 mg/m^3 in 2002) and high biomass years (21,098 mg/m^3 in 2011). The average open water phytoplankton biomass for Lake Winnipeg during the 1999 to 2016 period was 5,229 mg/m^3 with peak biomass occurring during years with major flood events (2011) and warm years like 2006, which followed a major flood event in 2005. Phytoplankton biomass has been relatively stable in the most recent five years (2012–2016) and similar to the long-term average (5,229 mg/m^3) with biomass ranging from approximately 5,000 to 5,600 mg/m^3 in the open water season.

From 1999 to 2016, average annual cyanobacteria biomass was lowest in 2000 (29 mg/m^3) and comprised only 9% of the total phytoplankton biomass. Cyanobacteria biomass was greatest in 2011 (17,589 mg/m^3), comprised almost 83% of the total phytoplankton biomass and was predominated by *Aphanizomenon*. Large increases in cyanobacteria biomass were accompanied by decreases in diatom biomass. For instance, diatoms comprised only 7% of the mean total algal biomass in 2011 (Figure 8-1) when cyanobacteria biomass was at its greatest.

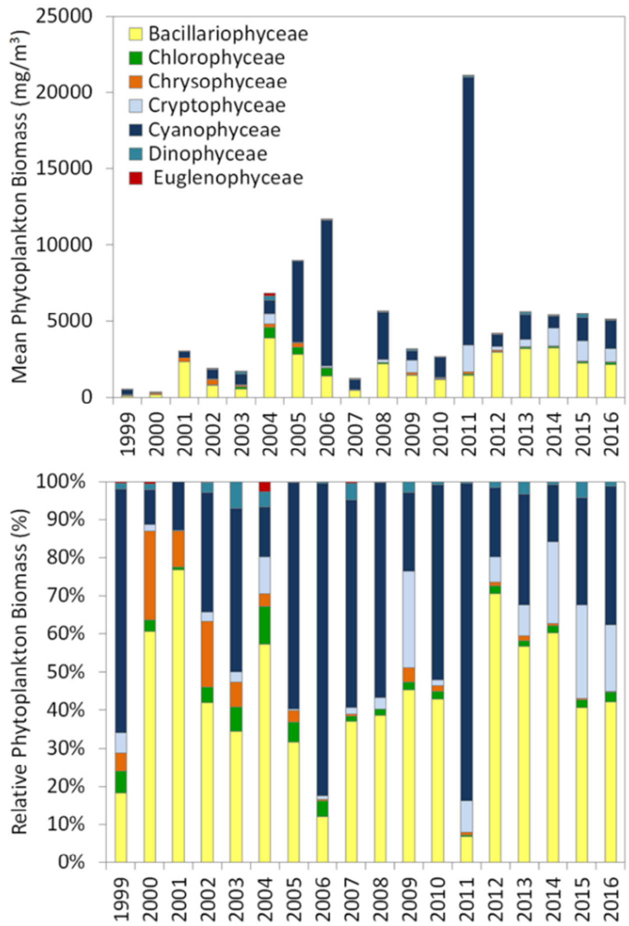


Figure 8-1: Mean annual phytoplankton biomass and composition by class (as mg/m^3 and %) in Lake Winnipeg, 1999–2016. Annual means represent the mean of euphotic zone samples collected between May and October in each year.

From 1999 to 2016, the open water average diatom biomass ranged from 95 mg/m^3 (in 1999) to 3,898 mg/m^3 (in 2004) (Figure 8-1). Diatoms comprised 77% of the total phytoplankton biomass in 2001 when the proportion of cyanobacteria was relatively low (13%). The high contribution of diatoms in 2001 was likely related to two relatively high biomass samples collected from the north basin in October (22,280 mg/m^3 and 9,037 mg/m^3) that were dominated by *Aulacoseira* and *Stephanodiscus*. From 1999 to 2016, the relative contribution of chlorophytes and cryptophytes ranged from < 1 to 10% and < 1 to 25%, respectively. From 1999 to 2007, cryptophytes comprised between 0 and 10% of the mean annual biomass for the lake, but higher cryptophyte biomass was observed in the more recent period of record in Lake Winnipeg from 2008 to 2016. For example, the 1999–2016 mean cryptophyte biomass was 432 mg/m^3. However, cryptophyte biomass in more recent years (i.e. 2014–2016) ranged from 891 to 1,333 mg/m^3. The relative contribution of chrysophytes was highly variable and comprised as much as 23% of the total biomass in 2000 and was less than 1% of the phytoplankton biomass in a number of years. Mean annual relative dinophyte contribution was also variable (< 1 to 7%) and euglenophytes did not contribute significantly to the total biomass, ranging from < 1 to 3%.

Chlorophyll-a

From 1999 to 2016, annual average chlorophyll-a concentrations in the euphotic zone of the north basin of Lake Winnipeg ranged from 4.1 µg/L (in 2000) to 35 µg/L (in 2006) with a mean open water chlorophyll-a concentration of 12.2 µg/L (Figure 8-2). Mean annual chlorophyll-a concentrations in the euphotic zone of the south basin and narrows were about half the chlorophyll-a concentrations observed in the north basin of Lake Winnipeg in most years (mean = 7.3 µg/L, range = 3.6 to 21.1 µg/L). From 2012 to 2016, chlorophyll-a concentrations have remained below the 1999–2016 long-term average in the north basin (12.2 µg/L) and south basin and narrows (7.3 µg/L) with the exception of the north basin in 2013.

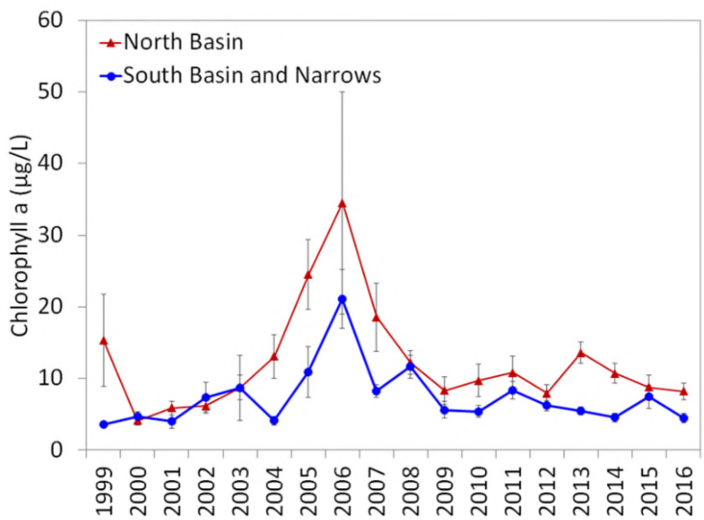


Figure 8-2: Mean annual chlorophyll a concentration for the north basin and south basin and narrows region (µg/L; ± SE) of Lake Winnipeg, 1999 to 2016. Annual means represent average of euphotic zone samples collected from May to October in each year.

Mean annual chlorophyll-a concentrations were highest in both basins of the lake in 2006 and corresponded to satellite imagery showing large surface blooms (> 5,000 km^2) in Lake Winnipeg during the late summer months. Elevated nutrient loads and precipitation in 2005, and above average air temperatures in 2006, may partly explain elevated chlorophyll-a concentrations in 2006. Phytoplankton biomass and chlorophyll-a were reasonably well related and followed the same overall temporal pattern in most years. However, the chlorophyll-a record did not show the same peak that was captured in the phytoplankton biomass in 2011.

Chlorophyll-a concentrations were mapped geospatially to describe the seasonal and spatial variation in concentrations in Lake Winnipeg for the 2008–2016 period (Figure 8-3). Overall, chlorophyll-a concentrations were generally lowest in spring and increased progressively through summer and fall in the north basin. Concentrations increased from less than 10 µg/L in the spring to approximately 20 to 25 µg/L in certain areas of the north basin in the summer period.

Maximum chlorophyll-a concentrations were observed in the fall period with concentrations between 30 and 35 µg/L on average for the 2008 to 2016 period. In contrast, the narrows region of the lake showed lower average chlorophyll-a concentrations in the summer and fall period (< 5 µg/L) relative to the spring (10 µg/L) and could be partly related to the strong water currents carrying turbid water through the narrows, which precludes the development of intense blooms of cyanobacteria in this area of the lake. In the south basin, chlorophyll-a concentrations showed little change from spring to fall with concentrations generally less than 10 µg/L. This is not surprising given the highly turbid waters of the south basin, which limit the growth of phytoplankton in Lake Winnipeg. It is clear that the pattern in phytoplankton bloom development and progression varies annually, as chlorophyll-a has been lowest in the summer in some years (e.g. 2007) with higher spring and fall biomass and a progressive increase from spring to fall in other years (e.g. 2006).

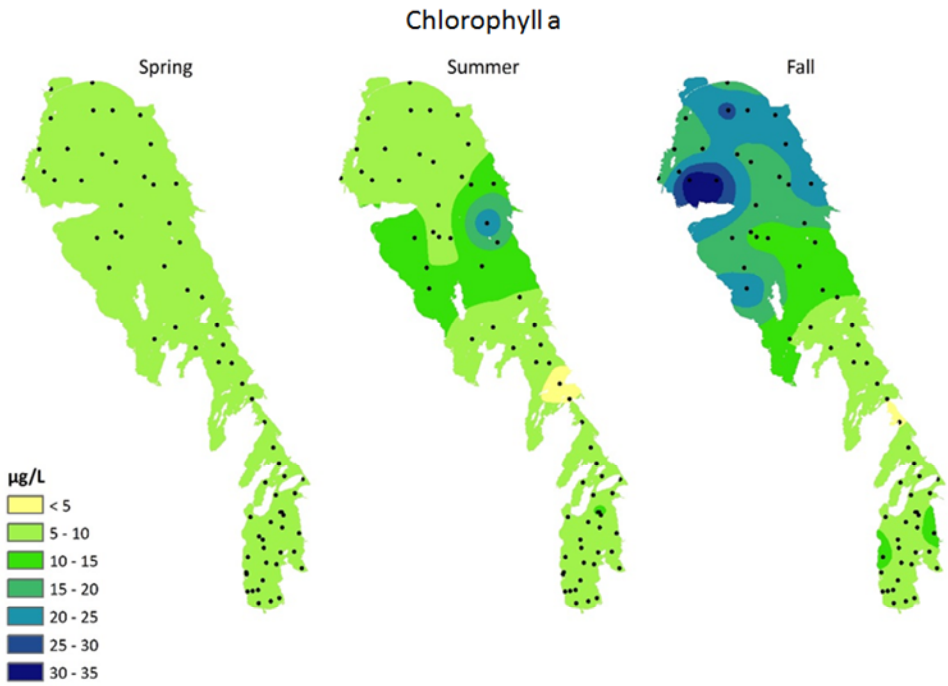


Figure 8-3: Seasonal and spatial variation in chlorophyll a (µg/L) in Lake Winnipeg during the spring, summer and fall, average of the 2008-2016 period.

Analysis of the nearshore and offshore chlorophyll-a data for the 2014–2016 period indicates that the nearshore areas of Lake Winnipeg are more productive (Figure 8-4). On average, chlorophyll-a concentrations in the nearshore areas of the north basin and south basin and narrows regions of Lake Winnipeg were 31 and 37% higher, respectively as compared to concentrations in the offshore regions. Both phosphorus and nitrogen are higher in nearshore areas of Lake Winnipeg and could partly explain the higher chlorophyll-a observed. It is also likely that blooms which form further offshore are blown into the nearshore areas of the lake causing increases in chlorophyll-a. Chlorophyll-a to TP ratios were calculated over the same period in the nearshore and offshore but no difference was observed within each basin (Figure 8-4). However, average chlorophyll-a:TP ratios were higher in the north basin of Lake Winnipeg

(0.27) compared the south basin (0.08). The south basin is turbid and phytoplankton growth is light-limited; therefore, less chlorophyll is produced per unit of phosphorus in the south basin.

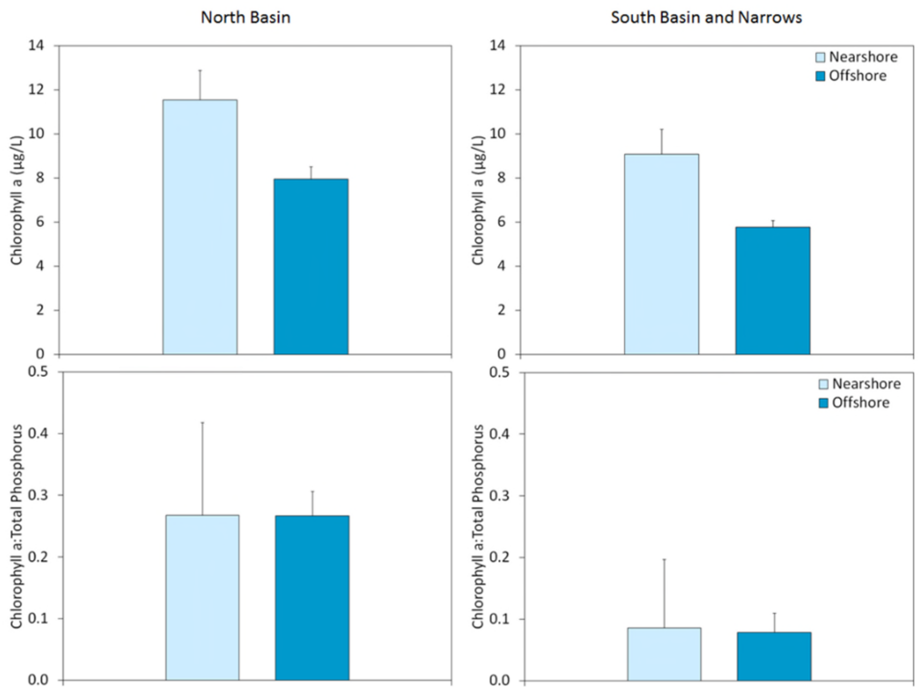


Figure 8-4: Average chlorophyll a concentrations (mg/L) and average total phosphorus to chlorophyll a ratios (±SE) in the nearshore and offshore of the north basin and south basin and narrows regions of Lake Winnipeg for the May to October period, 2014 to 2016.

Microcystin

A total of 596 surface and euphotic zone water samples were collected at offshore sites for the 1999–2018 period by Manitoba Agriculture and Resource Development. Samples were collected from both long-term water quality monitoring stations, and at various offshore locations where evidence of blooms was present, and were analyzed for total microcystin (as microcystin-LR). For the 1999–2018 period, Manitoba collected 426 water samples from beaches in the south basin of Lake Winnipeg. The nearshore beach samples were collected as a part of Manitoba's Clean Beaches Program when significant blooms along the beaches were evident.

Overall, microcystin was less frequently detected in the offshore areas of Lake Winnipeg compared to the nearshore sites (Table 8-2). Rates of detection for microcystin in the offshore of the north basin and south basin and narrows were 21 and 15%, respectively whereas microcystin detection in the nearshore was 54%. Average microcystin concentrations in the nearshore areas (1.27 µg/L) were higher in comparison to the offshore areas of the north basin (0.73 µg/L) and south basin and narrows region (0.13 µg/L) of the lake. Maximum microcystin concentrations detected in Lake Winnipeg for the 1999–2018 period were 331, 85 and 3.16 µg/L in the nearshore of the south basin, the offshore of the north basin and the offshore of the south basin and narrows, respectively. However, data indicate that microcystin concentrations are relatively low or below detection limits in most cases in Lake Winnipeg. For instance, only about 5% of samples exceeded 1.5 µg/L (which is the guideline for Canadian drinking water quality) in the offshore areas of the north basin and the nearshore area of the south basin and narrows, respectively for the 1999–2018 period. High concentrations were not frequently detected in the offshore of the south basin, with only 1% of samples exceeding 1.5 µg/L. Of the 303 water samples collected in the north basin between 1999 and 2018, only one sample exceeded the recreational water quality objective of 20 µg/L at a site located along the eastern side. Microcystin did not exceed 20 µg/L in the 293 samples collected from the offshore areas of the south basin and narrows over the same period.

Table 8-2: Summary statistics for microcystin-LR collected from nearshore and offshore areas in the north basin and south basin and narrows regions of Lake Winnipeg, 1999 to 2018.

	North Basin - Offshore	South Basin and Narrows - Offshore	South Basin and Narrows - Nearshore
Average (µg/L)	0.73	0.13	1.27
Minimum (µg/L)	0.05	0.05	0.05
Maximum (µg/L)	85	3.16	331
Standard Deviation (µg/L)	5.6	0.2	16.1
Standard Error (µg/L)	0.32	0.01	0.78
Total Number of Samples	303	293	426
Number of Samples with MC-LR Detections	64	45	229
Number of Samples > 1.5 µg/L	14	3	22
Number of Samples > 20 µg/L	1	0	3
% Detection	21	15	54
% of Samples > 1.5 µg/L	4.6	1	5.2
% of Samples > 20 µg/L	0.33	0	0.7

9.0 ALGAL BLOOMS

By: Caren Binding (Environment and Climate Change Canada)

Optical sensors on board Earth Observation (EO) satellites enable quantitative assessments of algal blooms via estimates of chlorophyll-a concentrations, offering a low cost solution for frequent, lake-wide observations of the extent and severity of blooms. This section presents a summary of satellite-derived indices for algal bloom intensity, spatial extent, duration and severity as previously reported by Binding et al. (2018). These bloom indices are now derived operationally by Environment and Climate Change Canada (ECCC) using data from the OLCI sensor (Ocean and Land Colour Instrument) on the European Space Agency's (ESA) Sentinel-3A satellite, launched in February 2016. Historical results for the 2003-2011 time period are also presented, derived from OLCI's predecessor, ESA's MERIS sensor.

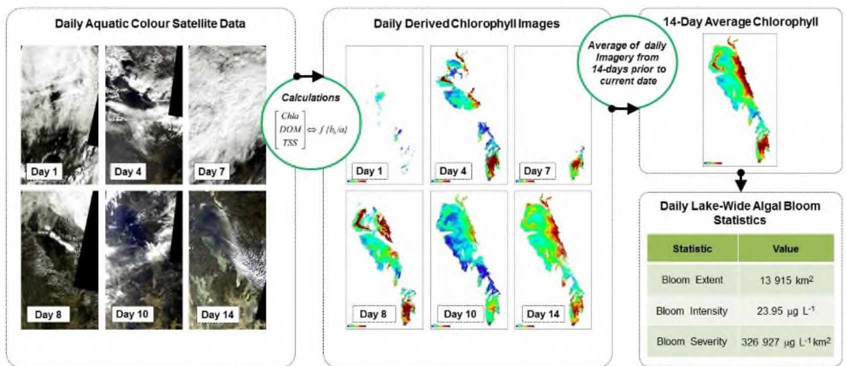


Figure 9-1: Schematic describing image processing work-flow for satellite-derived bloom indices.

The spatial and temporal coverage of satellite data is extensive, with Lake Winnipeg containing more than a quarter of a million lake observations per image (for MERIS/OLCI at full resolution) and approximately 50 valid observations per pixel over the June to October time period. The greatest challenge in accurately delineating and quantifying the bloom extent and intensity is still, however, the partial loss of data due to cloud cover over individual scenes. Methods have been further refined since Binding et al. (2018) to mitigate data anomalies brought about by cloud cover. Here we calculate a 14-day rolling average

Table 9-1: Definitions of remotely sensed algal bloom indices.

Algal Bloom Index	Definition
Algal Bloom Flag	= Chlorophyll > 10 $\mu g\ l^{-1}$
Bloom Intensity ($\mu g\ l^{-1}$)	= Mean Chlorophyll concentration within area flagged as bloom
Bloom Duration (Days)	= Number of days pixel flagged as bloom
Bloom Extent (km^2)	= Total area of pixels flagged as bloom
Bloom Severity ($\mu g\ l^{-1}\ km^2$)	= Bloom Intensity x Bloom Extent

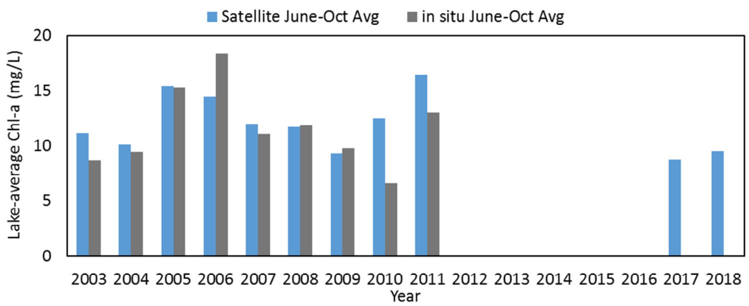


Figure 9-2: Annual lake-wide average satellite-derived and in situ chlorophyll, 2003– 2018.

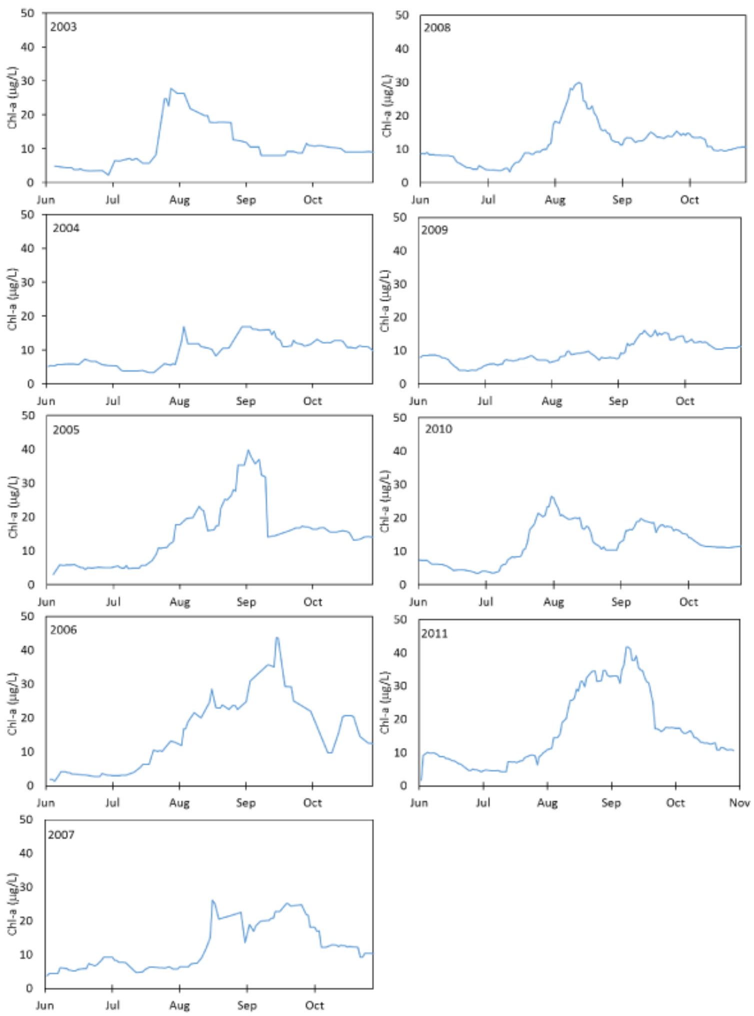


Figure 9-3: Seasonal variability in lake-average chlorophyll, 2003–2011, documenting year to year differences in the timing of peak algal biomass.

data product to provide near daily full-lake coverage for more comprehensive bloom assessments (Figure 9-1). From each daily image, chlorophyll concentrations and a suite of algal bloom indices are retrieved as defined in Table 9-1.

Annual Average Chlorophyll

Annual lake-wide average satellite-derived chlorophyll shows significant inter-annual variability (Figure 9-2), with highest concentrations in 2005 and 2011 and lowest concentrations in 2009 and 2017. In situ monitoring data (*see Figure 8-2*) suggested chlorophyll concentrations for 2012 to 2016 remained below the long-term average, and for 2017 and 2018, suggest a continuation of that trend. Work is underway to validate algorithms and image processing routines for the MODIS sensor in order to fill the 2012–2016 data gap.

Seasonal Chlorophyll

Seasonal variability in chlorophyll was found to be broadly consistent with that suggested by in situ monitoring data (*see Section 8.0*), with spring concentrations less than 10 µg/L increasing to lake-average peak summer/fall concentrations ranging from 15 to 40 µg/L (Figure 9-3). As noted in Section 8.0, there are clear inter-annual differences in the seasonal progression of blooms, and the increased spatial and temporal resolution afforded by satellite imagery documents that seasonal variability in more detail (Figures 9-3 and 9-4). Figure 9-3 captures differences in the timing of seasonal chlorophyll maxima year to year, ranging from August 1st in 2003 and 2010, and considerably later (September 16th and 19th respectively) in 2006 and 2009.

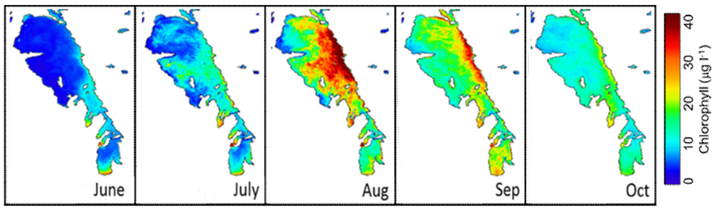


Figure 9-4: Monthly variability in chlorophyll (averaged over 2003–2011), documenting spatial variability in seasonal chlorophyll.

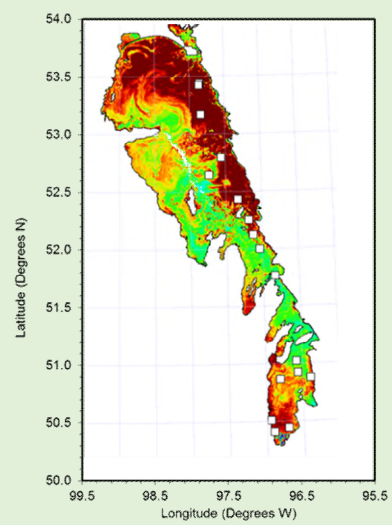
Average monthly chlorophyll maps document the spatial progression of blooms on the lake (Figure 9-4). Imagery captures spring bloom conditions in the south basin and narrows, typically dominated by turbid adapted diatoms such as *Aulacoseira.* Blooms occur in the narrows and north basin where nutrient-rich waters of the turbid south basin first reach more favourable light conditions in the clearer north basin. Widespread cyanobacteria blooms occur in the summer and fall in both basins, with nitrogen-fixing taxa (*Dolichospermum, Aphanizomenon)* typically dominating in the north and non nitrogen-fixing cyanobacteria (*Microcystis, Planktothrix)* dominating in the south basin. Blooms are frequently observed to concentrate along the northeast shore of the north basin consistent with flow and prevailing winds.

Annual Algal Bloom Indices

All bloom indices showed significant inter-annual variability (Figure 9-5). Average spatial extent of the bloom over the 2003–2018 period ranged from a minimum of 6,852 km^2 in 2017 to 13,149 km^2 in 2011,

Validation of Satellite Chlorophyll Retrievals

Pixel-wise validation of satellite-derived chlorophyll against in situ measures show reasonable agreement (Binding et al. 2018) but there are inherent challenges in comparing a single discrete surface water sample with a satellite measure, which represents average conditions over a 300 m x 300 m area. Annual (June–October) lake-wide average satellite-derived chlorophyll concentrations over the 2003–2011 period were, therefore, compared with in situ chlorophyll averaged over the same period (at the 14 long term monitoring stations plus additional ancillary stations). The two measures agree well in both magnitude and temporal variability, with the exception of 2006, 2010 and 2011 where there are significant differences (Figure 9-2). Image analysis confirms those differences can be attributed to temporal or spatial variability in the bloom relative to field sampling stations, producing either positive or negative bias in sampling the bloom those years. For example, in 2006, in situ samples showed a positive in situ sampling bias (Mean/Median Chlorophyll; in situ=3.8, satellite = 1.3).



OLCI-derived 14-day average chlorophyll, September 23rd 2006, with the location of sampling stations during that same 14-day period.

while peak bloom extent reached as much as 23,648 km^2 in 2006, covering 96% of the lake surface. Average bloom intensity peaked at 25.9 µg/L of chlorophyll in 2006, in agreement with observations from chlorophyll monitoring data (*see Figure 8-2*). The most intense peak bloom conditions were observed in 2005, 2006 and 2011. Blooms lasted on average from ~40 days in 2017 and 2018 to nearly 80 days in 2005, although most years had locations where prolonged blooms lasted up to 150 days. Bloom severity incorporates both spatial extent and intensity; while 2006 may have been the most intense bloom in the record, 2005 and 2011 were the most extensive, so all three years ranked the most severe years. Consistent with the phytoplankton monitoring data, these observations coincide with years of major flood events (2011) and warm summers, such as 2006. Binding et al. (2018) found that annual average bloom severity over the MERIS mission could be reasonably predicted by TP loads and summer temperatures.

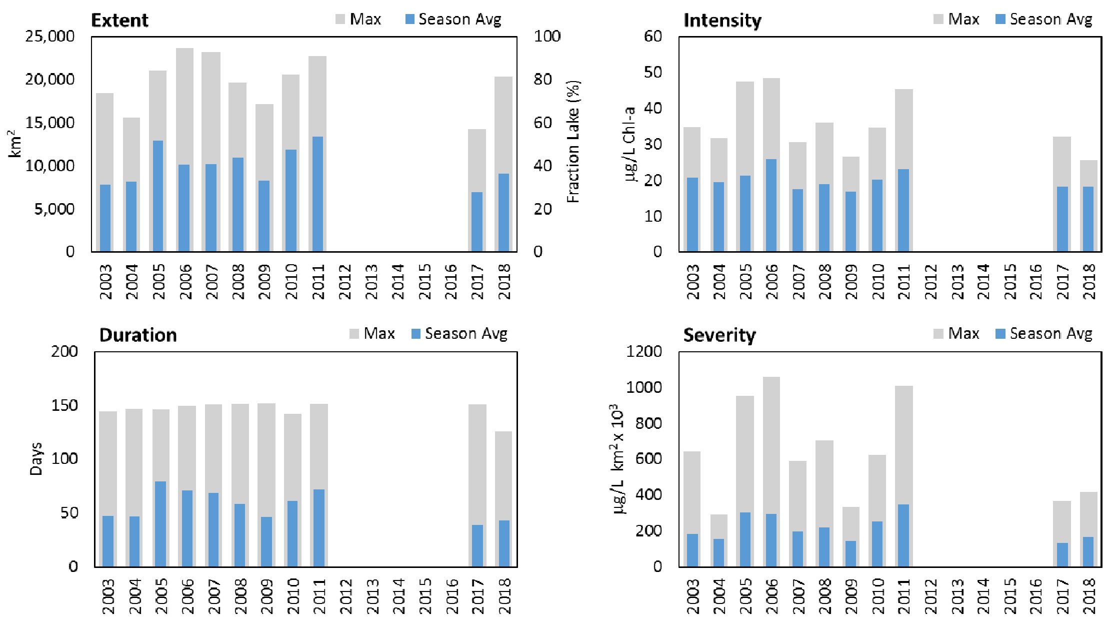


Figure 9-5: Annual lake-average (June 1st – Oct. 31st) bloom indices — spatial extent, intensity, duration and overall severity.

10.0 ZOOPLANKTON AND ZOOBENTHOS

By: Brenda Hann (University of Manitoba) and Alex Salki (Salki Consultants Inc.)

Crustacean Zooplankton Community

Zooplankton are essential components of the pelagic food web of aquatic ecosystems, comprising a vital intermediate trophic link, subject to both top-down predation as well as bottom-up dietary factors driven by nutrient availability. Changes in predators and food quality or quantity combine to influence the abundance and composition of the zooplankton community that reflect changes occurring in the lake ecosystem. Recently, Hann and Salki (2017) reviewed the long-term patterns in crustacean zooplankton community abundance in Lake Winnipeg for the period 1969–2006. Some of the key findings are reported here. Although zooplankton samples are being collected as part of the ongoing monitoring program on Lake Winnipeg, there are no readily available data for assessment post-2006 for inclusion in this report.

Zooplankton sampling in the nearshore.

What species of zooplankton are in Lake Winnipeg?

The species of crustacean zooplankton reported in Lake Winnipeg (see Hann and Salki 2017) showed little change over the last 80 years. The dominant taxa in the lake in summer include: *Leptodiaptomus ashlandi, Acanthocyclops vernalis, Diacyclops thomasi, Daphnia retrocurva, Daphnia mendotae, Diaphanosoma birgei, Eubosmina coregon,* and *Bosmina longirostris.* All of these were included in the suite of "core" species designated by Patalas and Salki (1992) with lake-wide distributions based on the 1969 surveys. The other core species, though not dominants, but also with relatively unchanged lake-wide distribution and densities were: *Limnocalanus macrurus, Epischura lacustris, E. nevadensis* (see Figure 8a in Patalas and Salki, 1992, mistakenly presented as *E. lacustris*), *E. oregonensis*, and *Leptodora kindti.* However, there were taxa whose distribution over the years consistently showed overwhelming prevalence in particular basins: north basin—*Eubosmina coregoni, Daphnia longiremis, Chydorus* cf. *sphaericus, Leptodiaptomus sicilis, L. minutus;* south basin—*L. siciloides* and *Tropocyclops* *prasinus mexicanus* (Figures 10-1 to 10-3). *E. coregoni*, a cladoceran invader, expanded its range in the north basin. It is not yet successfully established in the south basin despite its likely original point of entry being via the Winnipeg River.

The increasing dominance of the herbivore *Leptodiaptomus ashlandi* throughout the lake, and the declining proportions of *Leptodiaptomus minutus* and *Limnocalanus macrurus* in the north basin over the last decade (Figure 10-1), parallel the growing predominance of cyanobacteria among the primary producers (Kling et al. 2011). Large calanoids, such as *Heterocope, Epischura* and *Limnocalanus* were never abundant in Lake Winnipeg. *Limnocalanus macrurus*, a large calanoid (female body size 2.2-3.2 mm; Balcer et al. 1984), declined in parallel with cultural eutrophication and increased abundance of Rainbow Smelt (*Osmerus mordax*), similar to the pattern observed in Lake Erie (Kane et al. 2004). Among Cyclopoida, *Acanthocyclops vernalis* and *Diacyclops thomasi* comprised the dominant taxa in both basins of Lake Winnipeg (Figure 10-2). Cladocera are consistently more abundant in the north basin with its higher transparencies compared to the south basin (Figure 10-3). *Chydorus* cf. *sphaericus* and *Eubosmina coregoni,* found exclusively in the north basin, were more abundant in years with extensive cyanobacterial blooms.

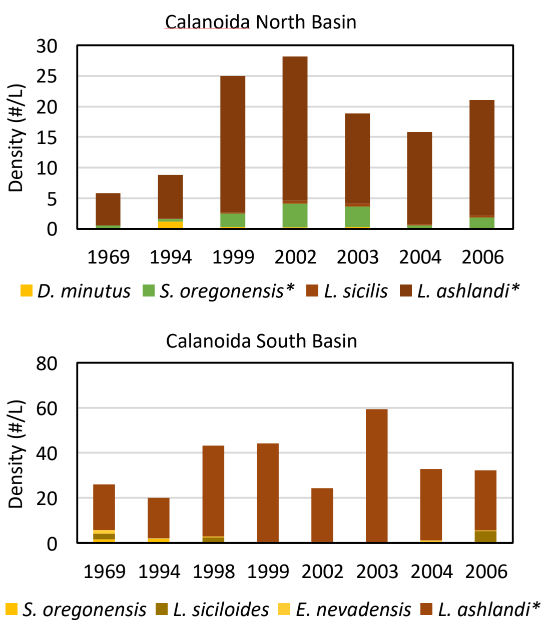


Figure 10-1: Densities of dominant taxa in Lake Winnipeg in the north basin (top panel) and south basin (bottom panel). Calanoida: Skistodiaptomus oregonensis, Leptodiaptomus siciloides, Epischura nevadensis, Leptodiaptomus ashlandi, Diaptomus minutus, Leptodiaptomus sicilis.

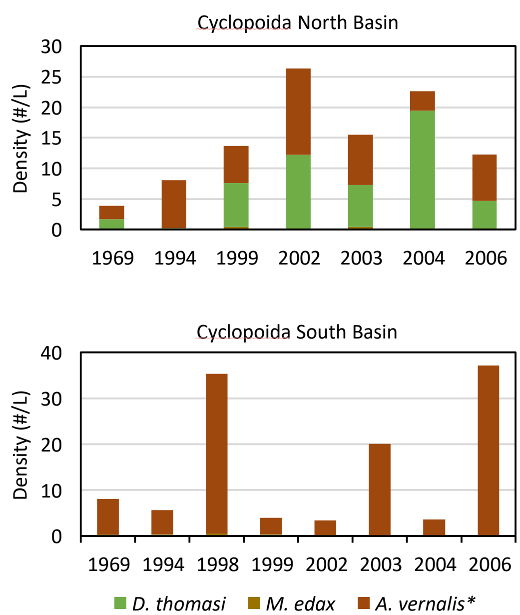


Figure 10-2: Densities of dominant taxa in Lake Winnipeg in the north basin (top panel) and south basin (bottom panel). Cyclopoida: Mesocyclops edax, Diacyclops thomasi, Acanthocyclops vernalis.

The four common species of cyclopoid copepods in Lake Winnipeg represent a considerable range of adult body size: *Acanthocyclops vernalis* (0.8–1.8 mm) and *Mesocyclops edax* (1.3–1.7 mm) are large cyclopoids, *Diacyclops thomasi* (1.0–1.4 mm) is of medium size, and *Tropocyclops prasinus mexicanus* (0.5–0.9 mm) is of small body size (Balcer et al. 1984).

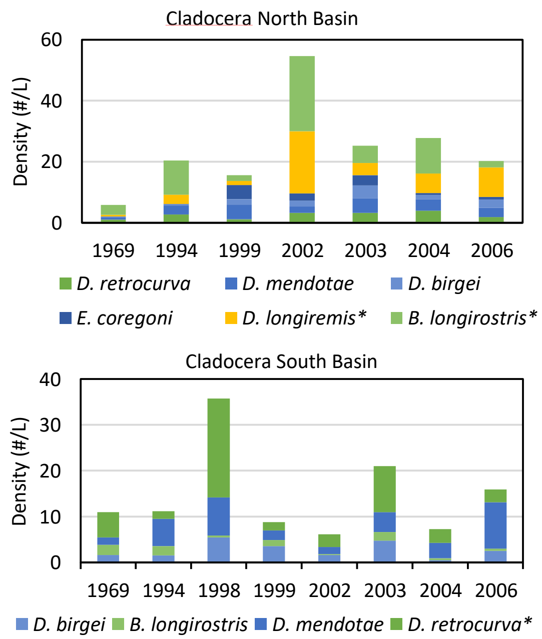


Figure 10-3: Densities of dominant taxa in Lake Winnipeg in the north basin (top panel) and south basin (bottom panel). Cladocera: Diaphanosoma birgei, Bosmina longirostris, Daphnia mendotae, Daphnia retrocurva, Eubosmina coregoni, Daphnia longiremis.

In Lake Winnipeg, Cladocera are consistently more abundant in the north basin despite the higher transparency compared with the south basin. Fish predation might be a strong factor to explain the pattern of the Cladocera populations in light of differing turbidity levels in each respective basin. Schulze (2011) showed that even in turbid environments, fish predation can be a main driver in shaping the zooplankton community in a top-down manner. Total biomass of planktivores in Lake Winnipeg was higher in the south basin than in the north basin and Emerald Shiner (*Notropis atherinoides*) biomass was greater than any other species in the lake (Lumb et al. 2012). The most abundant planktivores in the south basin are Emerald Shiners, found in significantly higher abundance in surface trawls (Lumb et al. 2012) where they undoubtedly feed heavily on cladocerans. Cladocera are predominantly herbivores, grazing on algae, protists, and bacteria and, in turn, are prey for planktivores, both fish and predatory invertebrates, including *Chaoborus* and cladocerans, *Leptodora* and *Bythotrephes longimanus* (a recent invader into Lake Winnipeg). Thus, cladocerans play a pivotal role in food web dynamics, energy transfer, and nutrient cycling in freshwaters (Carpenter et al. 1998, Urabe et al. 2002).

Crustacean zooplankton community abundance in Lake Winnipeg in summer (1969–2006)

Changes in the overall density of the zooplankton community from 1969 to 2006 were moderate despite changes in nutrient levels in the lake between 1969 and 2002–2006, particularly in TP (McCullough et al. 2012), and especially in years with elevated flows in the Red River over the last two decades, leading to cultural eutrophication of the entire lake. However, other reviews of nutrient data in Lake Winnipeg (Stewart et al. 2003, North/South Consultants 2006) found no substantial change in the nutrient status of the South basin. Total zooplankton densities in the entire lake showed a gradual increase in densities after 1969, peaking in 2003; however, this peak may be a consequence of high surface water temperatures and long residence time, in addition to nutrient loading. In 2003, TP concentration in the South basin was elevated (0.117 mg/L) despite very low flows, hence low TP load from the Red River into the South basin; the potential sources of this additional TP are unidentified. As long as there is a high probability of recurring flooding in the Red River basin (McCullough et al. 2012), productivity of Lake Winnipeg will not likely return to 1969–1990 levels. Climate-induced flooding and warming, coupled with other impacts including from agricultural intensification (Schindler et al. 2012), have raised lake productivity to a higher baseline level as substantiated by zooplankton densities.

Variation in zooplankton community density between north basin and south basin of Lake Winnipeg in summer (1969–2006)

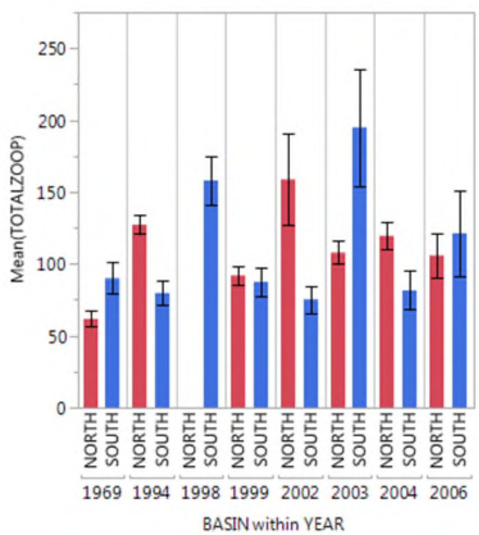


Figure 10-4: Densities of total zooplankton in Lake Winnipeg.

Crustacean zooplankton have increased more consistently in the north basin, possibly as a consequence of lower densities of pelagic planktivorous fish and higher primary production compared with the more turbid south basin (Brunskill et al. 1979b, 1980). Calanoid copepods play a larger role in the south basin food web in contrast to cyclopoid copepods and Cladocera in the north basin.

There were significant differences among years within each basin for all taxonomic groups. In the north basin, for total zooplankton, densities were significantly lower in 1969 than in all subsequent years, with no differences among those years (Figure 10-4). In the south basin, for total zooplankton, there were again significant differences among years, but 1998 and 2003 were significantly higher than all other years (Figure 10-4). Patterns for total Cladocera, total Cyclopoida, and total Calanoida in each basin followed the total zooplankton patterns in each basin (data not shown). Thus, it appears that the zooplankton community in the north basin is now fluctuating around a somewhat higher "set point", whereas the south basin has not responded in this manner to environmental changes.

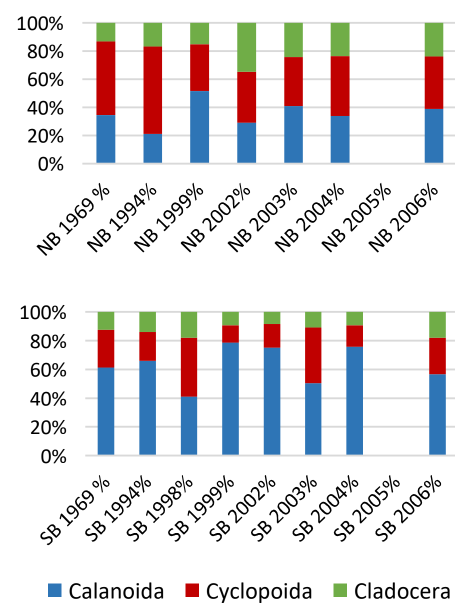


Figure 10-5: Proportions of major zooplankton groups in summer in north (top panel) and south (bottom panel) basins of Lake Winnipeg. NB = north basin. SB = south basin.

Proportions of Calanoida, Cyclopoida and Cladocera were relatively stable over time but characteristic of each basin (Figure 10-5). Cladocera and Cyclopoida together comprised a larger percentage of the zooplankton community than Calanoida in the north basin over time; by contrast, Calanoida were dominant in the south basin throughout the period of study except in 1998 and 2003. In 1998, cyclopoids increased relative to calanoids and cladocerans in the south basin, possibly as a result of inputs of detritus and sediment during the 1997 flood (Stewart et al. 2003). To some extent, a similar response in cyclopoid proportions occurred in 2006 after the very high flushing in summer and fall in 2005 on the Red River. Calanoida (essentially one species, *Leptodiaptomus ashlandi)* were more prevalent in the more eutrophic, turbid south basin, whereas Cyclopoida and Cladocera prevailed in the less eutrophic, more transparent north basin.

Zooplankton densities were routinely higher in the north basin than in the south basin. This reflects differences in morphometry (e.g. greater water depth) and higher, non light-limited primary productivity, as well as notable differences in primary producers (phytoplankton, cyanobacteria) and reduced predation pressure from north basin planktivorous fish communities (Lumb et al. 2012). Climate played a role. In warm, dry years (i.e. 1969, 1994, 1999, 2006), phytoplankton biomass was primarily comprised of cyanobacteria (Kling 1996, EC and MWS 2011), with three genera (*Microcystis, Anabaena, Aphanizomenon)* comprising up to 95% of the biomass (EC and MWS 2011), whereas in cool, wet years, diatoms and cryptophytes constituted a larger proportion of the phytoplankton biomass (EC and MWS 2011). A complicating factor for zooplankton feeding is basin-specific differences in predominant cyanobacterial bloom-forming taxa. For example, different zooplankton species have differing abilities to consume the non-N-fixing *Microcystis* blooms in the nutrient-rich, turbid conditions of the south basin versus the nitrogen fixing-*Anabaena* and *Aphanizomenon* blooms in the north basin.

Processing nearshore zoobenthos samples.

Zoobenthos in Lake Winnipeg (2000-2016)

Zoobenthos in Lake Winnipeg (1969–2013), including seasonal and spatial patterns and trends, was examined in detail by Hann et al. (2017), based in part on historical data summarized in various Department of Fisheries and Ocean (DFO) reports and in part on samples collected throughout the lake (2002–2013) by the Lake Winnipeg Research Consortium (LWRC). These samples were collected during spring, summer and fall cruises from 57 of 65 stations (*see Figure 1-3*) using a standard Ekman dredge (0.0225 m^2). They were rinsed with lake water through a 200 µm mesh net, preserved with 10% buffered formalin, returned to the laboratory for identification and enumeration, and preserved in 70% ethanol. All data were summarized into groups: Oligochaeta (Tubificinae, Lumbriculidae), Mollusca (Sphaeriidae), Amphipoda (Pontoporeiidae), Ephemeroptera (Ephemeridae), Trichoptera (Leptoceridae, Molannidae), and Diptera (Chironomidae) and Nematoda. This work illustrated two prevailing patterns in terms of zoobenthos abundance and community composition. Firstly, lake-wide zoobenthos density in the 2002–2013 period was higher than in 1969, and the increases in density were more pronounced in the north basin than either the south basin or narrows. Secondly, the proportion of oligochaetes and chironomids increased in the north basin, while amphipods decreased in both basins and were no longer the dominant taxon in the north basin (Hann et al. 2017).

Concurrent to the LWRC zoobenthos sampling, Manitoba Agriculture and Resource Development (MARD) established a set of long-term water quality monitoring stations (W1–W12) distributed throughout the lake basins in 2000, with 2 additional stations (W13, W14) added in 2007 (*see Figure 1-3*). Benthos samples were collected during spring only (except 2005 when samples were collected in the fall) using a Ponar dredge (0.0523 m^2), with three replicates taken and sediments pooled for analyses. Sediments were washed through a 400 µm mesh net, preserved in 70% ethanol and analyzed by ALS at its Environmental Laboratory in Winnipeg. Thus, zoobenthos in Lake Winnipeg was sampled using two standard protocols at an overlapping set of stations, W1–W14. This provided a unique opportunity to evaluate the efficacy of the two sampling methods, specifically for this set of stations, i.e. how do densities of zoobenthic communities differ when the sampling protocols differ with respect to:

1. Mesh size: 200 µm vs 400 µm for stations W1–W14
2. Number of sampling stations: W1–W14 stations sampled in spring (MARD) vs the larger array of stations sampled in spring (LWRC)
3. Season of sampling: spring, summer and fall (LWRC) vs spring only (LWRC).

(1) Effect of Mesh Size

Total whole-lake zoobenthos densities at stations W1–W14 for the LWRC samples collected with a 200 µm mesh (mean = ~8000/m^2) were 2.5–3X higher than for samples collected by MARD with a 400 µm mesh (mean = ~2500/m^2) (Figure 10-6). More organisms were retained by a smaller mesh size. Total zoobenthic densities included taxonomic groups of organisms of substantially differing body size and biomass (not evaluated), sampled differentially by the two distinct mesh sizes employed in the surveys. For example, *Diporeia,* a relatively large amphipod, was a dominant component of the benthos in samples sieved with the 400 µm mesh in comparison with the much smaller Chironomidae, Oligochaeta and Nematoda, comprising a larger

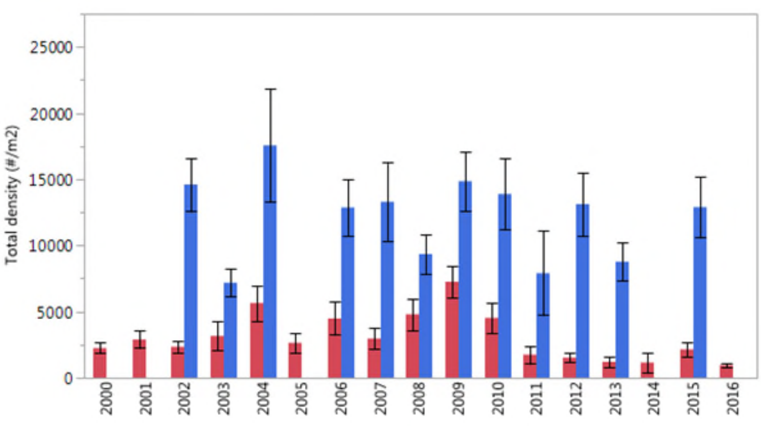


Figure 10-6: Total whole-lake zoobenthos density (mean ± S.E. including Nematoda, #/m^2) in spring samples collected at stations W1–W14 using a 200 µm mesh (red bars) and a 400 µm mesh (blue bars).

numerical component in samples sieved with the 200 µm mesh. Individuals of Chironomidae, Oligochaeta and especially Nematoda were consistently present in much lower densities in samples sieved through the larger mesh size as they were not retained as effectively in the net (Figure 10-7).

When analyzed by basin, the MARD data showed the narrows with the highest density in many years, whereas the LWRC data showed the north basin having the highest density in most years (Figure 10-8). The MARD data showed a substantial decline during 2011–2016, which was not evident in the LWRC data.

Sediments at the stations in the narrows comprised highly variable particle sizes, ranging from gravel to fine organic sediments. Sediments in the south basin and north basin were more consistently clay and silt (unpublished data). MARD sampling protocol prescribed sampling using a Ponar grab in addition to sieving of the samples through 400-micron mesh netting in contrast with LWRC sampling protocol using an Ekman grab and sieving through 200-micron mesh. The design of the Ponar grab (heavier apparatus, stronger jaws, larger surface area sampled) may enable sampling the diversity of sediment particle sizes more efficiently and at more stations in the narrows than is possible using an Ekman grab. However, type of gear as well as mesh size are confounded in these two sampling protocols, making interpretation of observed patterns challenging.

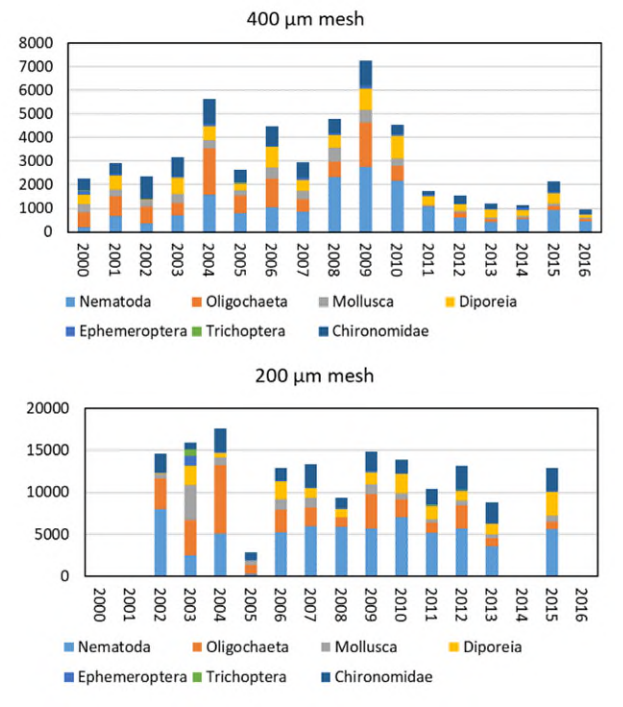


Figure 10-7: Spring densities (mean ± S.E. including Nematoda, #/m^2) of major zoobenthic taxa at stations W1–W14 in samples sieved through a 400 µm mesh (top) or a 200 µm mesh (bottom).

Diporeia (Pontoporeidae), a relatively large amphipod, routinely comprised a dominant component of the zoobenthic community in the narrows, especially in the spring (Hann et al. 2017). Using the MARD sampling protocol, *Diporeia* individuals would be retained with the 400 micron mesh whereas many of the smaller taxa would pass through the mesh. Hence, large-bodied organisms constitute a larger proportion of the community in the narrows than in either north or south basin. In the north basin, *Diporeia* represented a less substantial constituent of the zoobenthic community, and was virtually absent from the South basin. Thus, total density of zoobenthic organisms in the narrows, sampled using the MARD protocol, would be expected to be correlated with the abundance of the larger taxa (Figure 10-8).

In contrast, smaller zoobenthic organisms predominated in the north basin, followed by the south basin, and then the narrows. This was the pattern for total density observed among basins according to the LWRC sampling protocol (Figure 10-8).

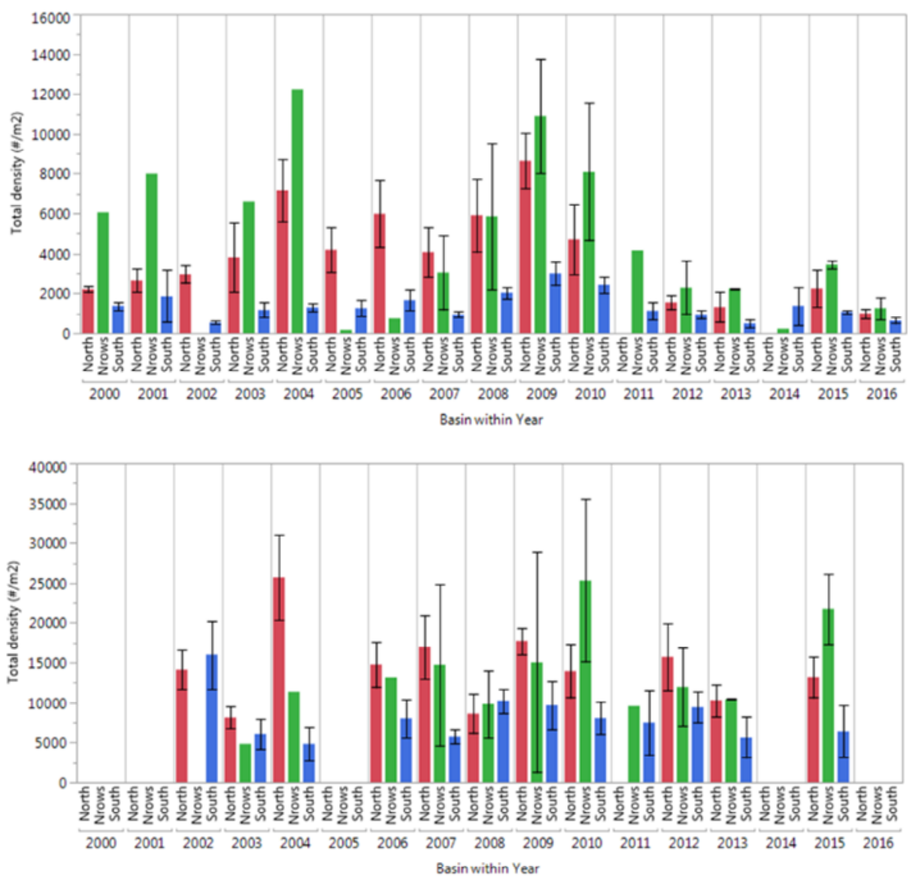


Figure 10-8: Total zoobenthos density (mean ± S.E. including Nematoda, #/m²) in the north basin (red bars), narrows (green bars), and south basin (blue bars) of Lake Winnipeg at stations W1–W14 from spring samples using a 400 µm mesh by Manitoba Agriculture and Resource Development (top), and a 200 µm mesh by Lake Winnipeg Research Consortium (bottom).

(2) Effect of Number of Sampling Stations (spring only)

Zoobenthos densities in spring (LWRC) averaged over only 14 stations (W1-W14) (Figure 10-8 bottom) were slightly higher than when ~50 stations throughout the lake (Figure 10-9) were included. The W1–W14 stations are located in the more homogeneous offshore region of Lake Winnipeg, whereas the larger complex of LWRC sampling stations include a wider spectrum of habitats, many nearer to shore and comprising a more diverse array of sediment types and habitats.

(3) *Effect of Season: Spring only vs spring, summer and fall*

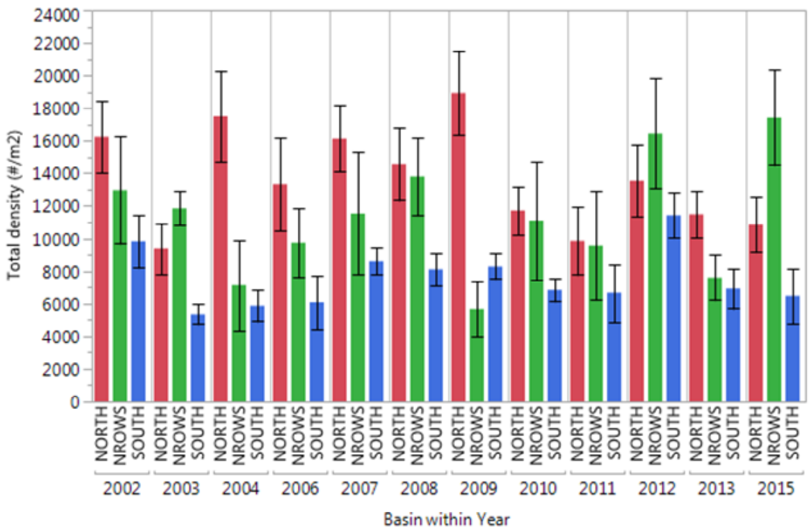


Figure 10-9: LWRC zoobenthos densities (mean ± S.E.) averaged over all ~50 stations (total zoobenthos density including Nematoda, #/m²) by year and basin (spring only).

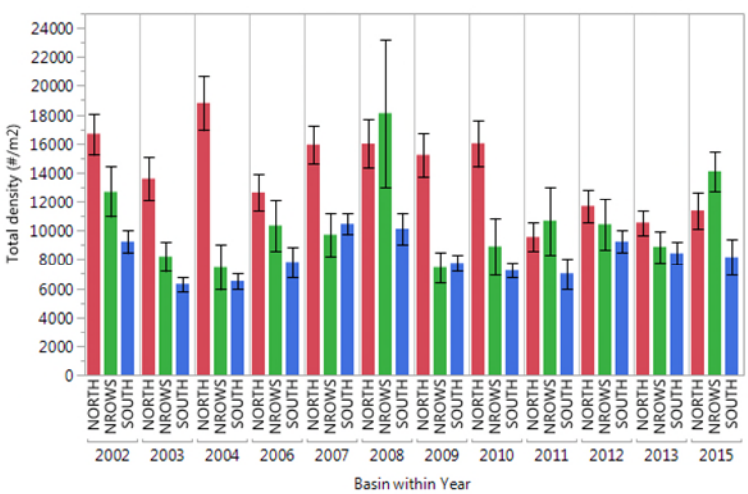


Figure 10-10: LWRC benthos densities (mean ± S.E.) averaged over all ~50 stations (total zoobenthos density including Nematoda, #/m²) by year and basin (spring, summer, and fall).

Zoobenthic invertebrate taxa display a wide variety of life cycles, including those with one-year and multi-year life cycles. Hence, representatives of the diverse array of organisms found in and on the benthic sediments may be of different body sizes in each season. Sampling with larger mesh size will selectively sample larger-bodied organisms (Figure 10-7). For example, mainly adult *Diporeia* organisms will be sampled in spring, whereas juveniles will comprise the majority of organisms sampled in summer and fall. In contrast, many immature insects develop from eggs in spring and grow larger through the open water season.

Comparison of zoobenthic densities at the larger array of stations (LWRC data) in spring only (Figure 10-9) or in all three seasons (Figure 10-10) showed that the densities were similar with higher densities in the north basin. However, it is apparent that densities in the narrows were higher when sampling in spring only in comparison with sampling in all three seasons, again largely attributable to the preponderance of large-bodied *Diporeia* in the narrows in spring. Densities in the south basin were lowest when sampled in spring only or in all three seasons.

11.0 FISH

By: Chelsey Lumb, Doug Watkinson (Fisheries and Oceans Canada), and Geoff Klein (Manitoba Agriculture and Resource Development)

In Lake Winnipeg, fishing provides an important source of protein and food security for many communities around the lake. In addition, fish populations have long supported valuable angling and commercial fisheries. In fact, the commercial fishery on Lake Winnipeg is the second largest commercial inland fishery in North America, second only to Lake Erie (LWQRTF 2011). Annual in-lake monitoring of the fish community, recreational catches and commercial catch records are important in providing information about status and trends of fish populations that support the subsistence, angling and commercial freshwater fisheries in Lake Winnipeg. The following sections summarize the monitoring information and status of prey fish, Walleye (*Sander vitreus*) and Sauger (*Sander canadensis*) populations, and information about the angling and commercial fisheries, to provide an assessment of the health of the fish community.

Prey Fish Survey

Prey fish community monitoring provides information about dynamics of native and non-native fish populations over time and space, is used to infer fish community resilience and key trophic linkages, and provides an early indication of year class strength of Walleye, that recruit to the commercial fishery some four years later. Since 2002, midwater trawl samples have been collected near offshore monitoring stations lake wide during spring, summer and fall, in all years except 2005, when samples were only collected during fall, and in 2013 and 2014, when no trawl samples were collected during spring. In all years of the survey, Emerald Shiner (*Notropis atherinoides*), Rainbow Smelt (*Osmerus mordax*), and Cisco (*Coregonus artedi*) were the dominant species by weight (Figure 11-1). Also commonly caught in trawl tows were Walleye, White Bass (*Morone chrysops*), and Yellow Perch (*Perca flavescens*). Other species made up a relatively small proportion of catches. In total, 29 species were caught in trawl tows since 2002.

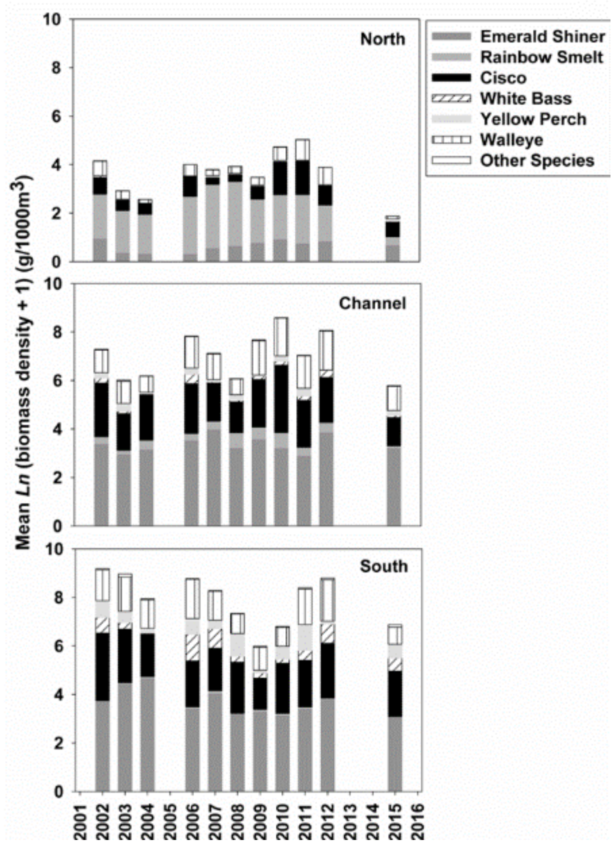


Figure 11-1: Mean biomass density (g/1000 m^3) of natural log (x + 1) transformed offshore pelagic species biomass in trawl samples collected from three geographic regions of Lake Winnipeg, 2002–2004, 2006–2012 and 2015. The trawl survey was not conducted in one or more seasons in 2005, 2013 and 2014. Other species equals the combined catch of all other species.

Rainbow Smelt biomass in the north basin was high during fall 2006 (88.9 g/1000 m^3), spring 2007 (75.5 g/1000 m^3), and summer 2008 (79.1 g/1000 m^3) (Figure 11-2); however by 2015 Rainbow Smelt biomass of about 1 g/1000 m^3 was roughly 5% of the long-term mean of summer (19.1 g/1000 m^3) and fall (35.3 g/1000 m^3). In 2017, no Rainbow Smelt were collected in the trawl survey. North basin Emerald Shiner summer biomass also decreased post-2011; however, fall biomass was higher for some years from 2009 to 2015 compared to 2002–2008 (Figure 11-2). In the narrows, Emerald Shiner summer biomass hovered around the long-term mean (123.6 g/1000 m^3) from 2006 to 2015, with the exception of 2011 (Figure 11-3). Fall biomass of Emerald Shiner in the narrows was more variable than during spring or summer, although the long-term mean fall biomass (119.8 g/1000 m^3) was comparable to the long-term mean summer biomass (123.6 g/1000 m^3). South basin Emerald Shiner long-term mean summer biomass (221.4 g/1000 m^3) was nearly double that in the narrows (123.6 g/1000 m^3) (Figure 11-4). Annually, Cisco biomass was generally less than Emerald Shiner biomass in the north, narrows and south basin. Cisco biomass was generally greater in the south basin compared to the narrows, and was greater in the south basin during summer (long-term mean biomass 27.4 g/1000 m^3) and fall (28.0 g/1000 m^3) compared to spring (10.1 g/1000 m^3). Mainly juvenile Cisco were caught in trawls, unlike small-bodied species, such as Rainbow Smelt and Emerald Shiner, which are vulnerable to being caught in the trawl during all life stages.

White Bass and Yellow Perch biomass in the south basin were generally an order of magnitude smaller than Emerald Shiner biomass (Figure 11-5). Like Cisco, mostly age-0 White Bass and Yellow Perch were caught during the summer trawl survey, providing some insight into recruitment variability. Peak mean summer biomass of White Bass was observed in 2007 (21.9 g/1000m^3). Yellow Perch summer biomass peaked in 2008 (47.3 g/1000 m^3). Interestingly, mean summer biomass of Walleye peaked in 2011 (39.5 g/1000 m^3) following the 2011 Manitoba flood. Although Sauger was not abundant in trawls, there was evidence of some age-1 Sauger sampled during spring 2012 (3.8 g/1000 m^3). As percids are spring spawners, favourable conditions likely affected Yellow Perch, Walleye and Sauger in 2011, leading to good recruitment in that year.

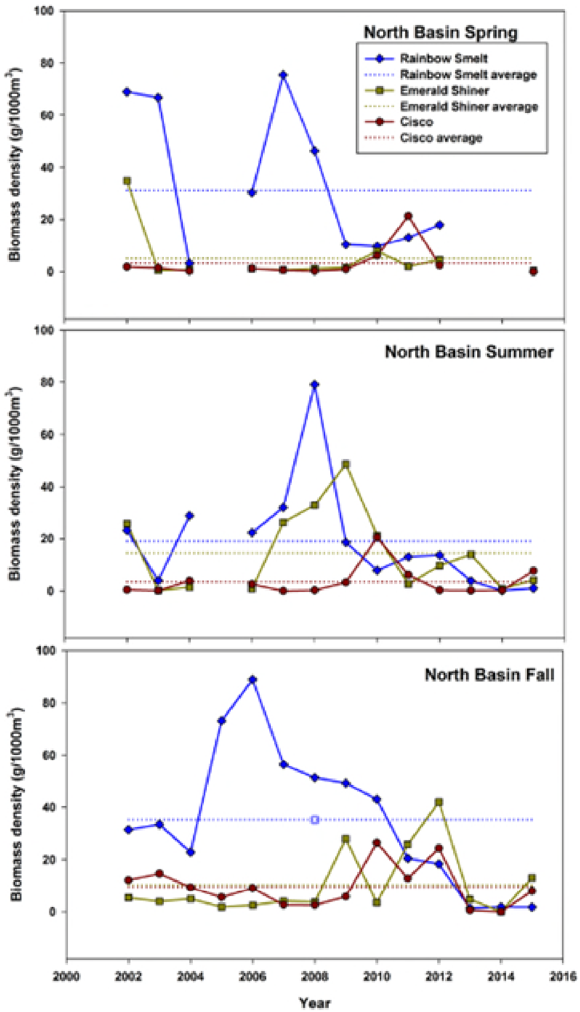


Figure 11-2: Mean biomass density (g/1000 m^3) of the most commonly captured species in offshore pelagic trawl samples collected from the north basin, during spring, summer and fall, 2002–2015. Dashed lines represent mean catch rate for each species for the entire time period.

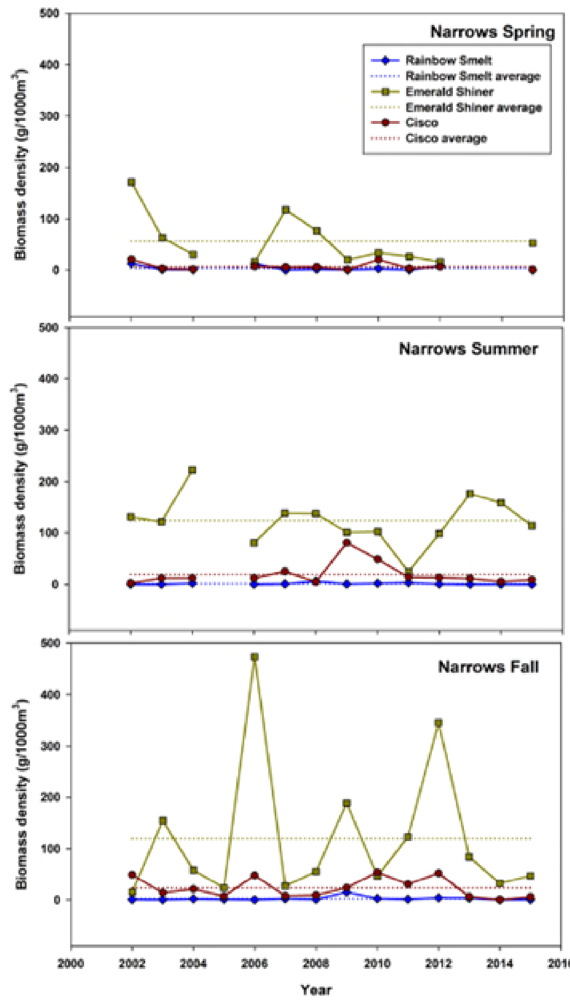


Figure 11-3: Mean biomass density (g/1000 m^3) of Rainbow Smelt, Emerald Shiner, and Cisco captured in offshore pelagic trawl samples collected from the narrows, during spring, summer and fall, 2002–2015. Dashed lines represent mean catch rate for each species for the entire time period.

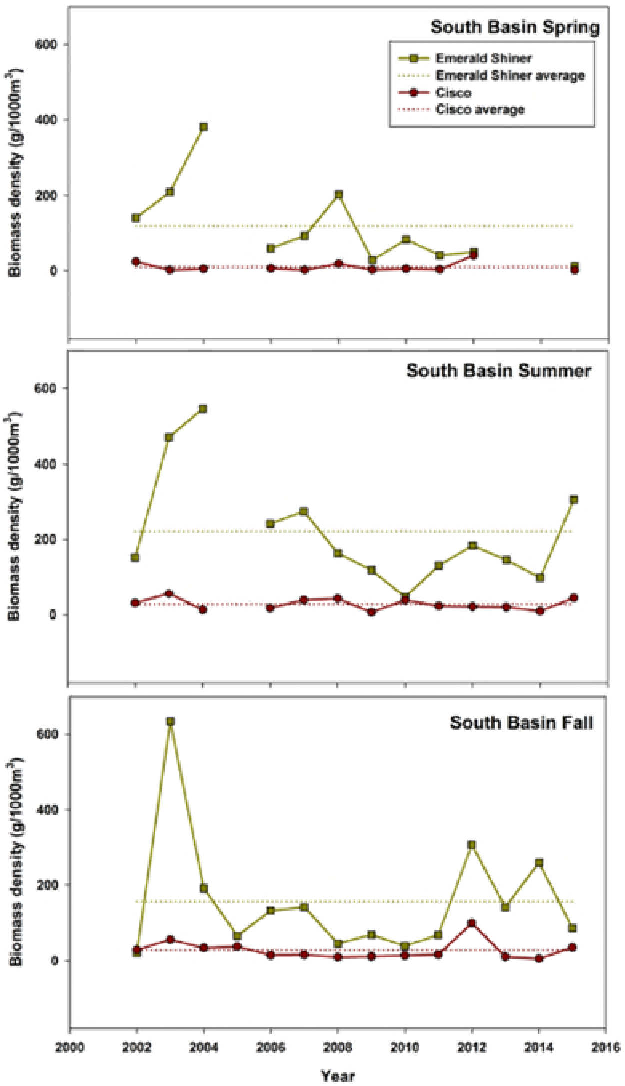


Figure 11-4: Mean biomass density (g/1000 m3) of Emerald Shiner, and Cisco captured in offshore pelagic fish trawl samples collected from the south basin during spring, summer and fall, 2002-2015. Dashed lines represent mean catch rate for each species for the entire time period.

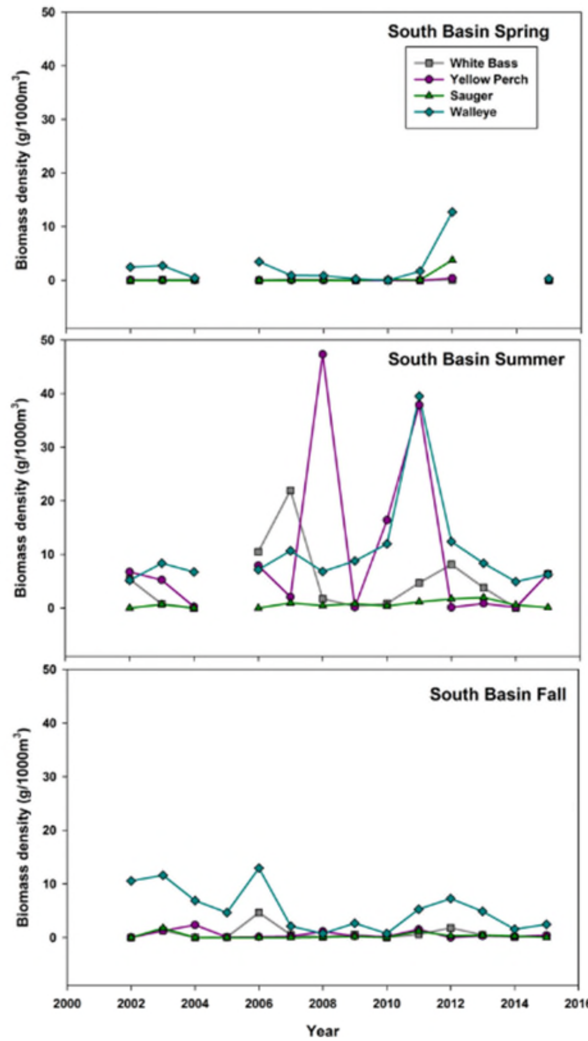


Figure 11-5: Mean biomass (g/1000 m3) of White Bass, Yellow Perch, Sauger and Walleye in offshore pelagic fish trawl samples collected from the south basin during spring, summer and fall, 2002-2015.

Walleye mean biomass was greater in the south basin and narrows than the north basin, and greatest in the summer, when juvenile Walleye were caught in trawls (Figure 11-6). Mean summer biomass of juvenile Walleye was generally higher in the south basin than the narrows, except in 2010 (24.1 g/1000 m^3) and 2013 (27.6 g/1000 m^3), when biomass was higher in the narrows than the south basin. Spatial differences in Walleye biomass across the lake could suggest there is more spawning in the south basin and narrows than the north basin, where mean biomass of Walleye was lower.

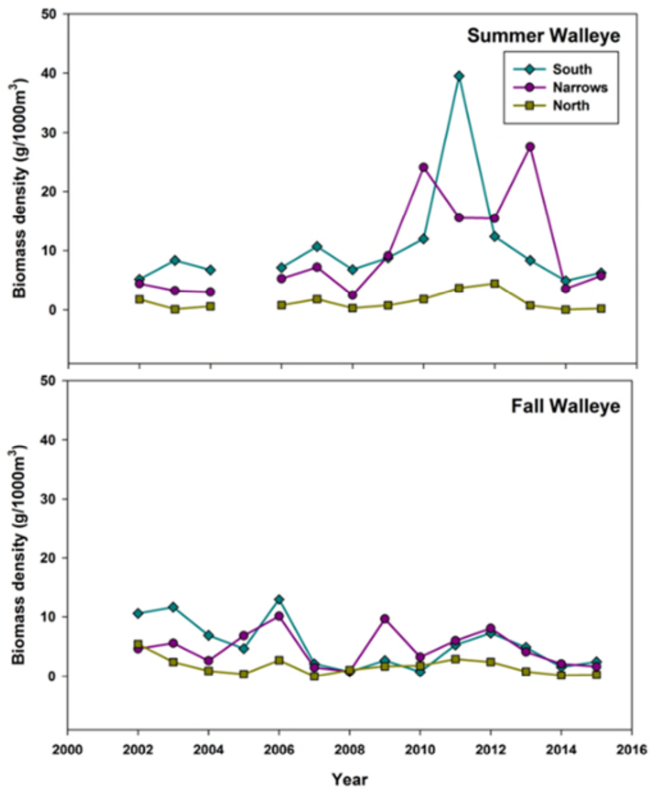


Figure 11-6: Mean biomass density (g/1000 m^3) of Walleye in offshore pelagic fish trawl samples collected from three geographic regions of Lake Winnipeg, during summer and fall, 2002–2015. In 2005, only a fall cruise was conducted.

Reasons for the decline of Rainbow Smelt in the north basin are not clear. The timing of the decline from 2009 to 2011 overlapped with part of the period of high predator (Walleye) abundance (see sections below). Similarly in the south basin, predation pressure likely increased on Emerald Shiner from roughly the mid-2000s until about 2014, with increased relative abundance of Walleye and Sauger. Emerald Shiner maintained a relatively stable biomass, even with increased predatory pressure, while Rainbow Smelt in the north basin virtually collapsed. The reason(s) for this difference remain unclear. Perhaps because the south basin was more productive than the north basin, prey fish production was able to increase so that it matched increasing Walleye predatory demand over time with the net result being no large temporal change in biomass of Emerald Shiner (Lumb et al. 2012). Also, the more diverse prey fish community in the south basin compared to the north basin likely reduced the predatory pressure on a single prey fish species, as evidenced by Walleye and Sauger diets (Sheppard et al. 2015).

Rainbow Smelt in the north basin were likely exposed to other factors, in addition to predation, that may have contributed to declines in biomass. Bottom-up limitation in the form of low, and then reduced productivity due to decreased TP, or higher densities of Spiny Water Flea (*Bythotrephes longimanus*) than in the south basin, could have reduced growth and survival of Rainbow Smelt (Parker Stetter et al. 2005). At large stock sizes (> 10 kg/ha), competition or cannibalism by older Rainbow Smelt on age-0 recruits could have been an important factor related to Rainbow Smelt recruitment, as it was in Lake Huron (O'Brien et al. 2014). Recruitment of age-0 Rainbow Smelt was also regulated by spring precipitation influencing stream spawning habitats in Lake Huron (O'Brien et al. 2014). Non-native Rainbow Smelt appear to be poorly suited to maintain relatively stable biomass compared to native prey fish species. Lake Winnipeg is isothermal and the lake's water temperatures are likely not ideal for Rainbow Smelt feeding, growth and survival (Brandt et al. 1980, Wismer and Christie 1987, Lantry and Stewart 1993). Rainbow Smelt are nearly absent from the south basin where water temperatures are higher.

The status of the north basin prey fish community is poor and the trend is decreasing abundance. During most of the years of the survey, the prey fish community was dominated by non-native Rainbow Smelt until it underwent a strong decline. The species composition and density of the prey fish community in the north basin before the arrival of Rainbow Smelt is uncertain because the arrival of Rainbow Smelt pre-dates the pelagic trawl survey.

The status of the south basin prey fish community is good and the trend is stable. The prey fish community in the south basin is more diverse and composed of more native fish species, lending resilience and stability in the event of increased stressors. Biomass trends of dominant species, Emerald Shiner and Cisco, were stable over the years of the survey. Recruitment of White Bass, Yellow Perch, Sauger and Walleye was variable, with some years of strong recruitment of percid and other species.

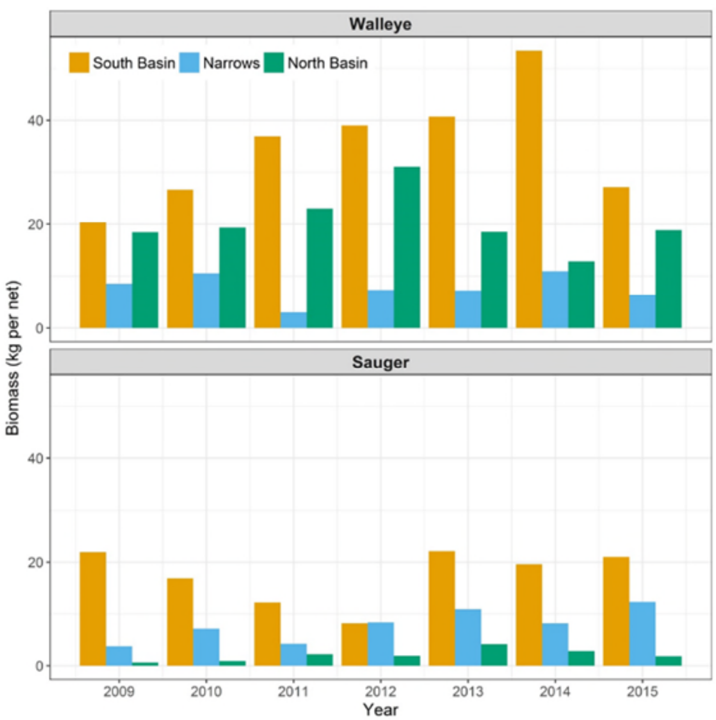


Figure 11-7: Catch per unit effort (kg per net) of Walleye and Sauger from 2009 to 2015, by basin, from the annual index gill net survey of Lake Winnipeg. Figure courtesy of C. Charles, Fisheries and Oceans Canada.

Walleye and Sauger Survey

Relative abundance, condition and mortality of Walleye and Sauger are determined annually through the provincial index net program including index gill nets set at two locations in the south basin, the narrows, and the north basin during June and July; methods have been standardized since 2009 (MSD 2019). From 2009 to 2012, Walleye biomass in both the south and north basins increased, but from 2013 to 2014, while biomass increased in the south basin, it decreased in the north basin (Figure 11-7). This represents a large decrease in biomass for the lake as the surface area of the north basin is more than six times the size of the south basin, with ample suitable Walleye habitat (Lester et al. 2004). The timing of decreased Walleye biomass in the north basin and the apparent increase in biomass in the south basin corresponds with declining biomass of Rainbow Smelt, the main diet item of Walleye in the north basin. By 2015, Walleye biomass decreased in both the north and south basins.

From 2009 to 2015 Sauger biomass was higher in the south basin than the narrows or north basin (Figure 11-7). Biomass increased in the south basin from 2013 to 2015. Sauger were less abundant in the north basin (Johnston et al. 2012) and less reliant on Rainbow Smelt than Walleye (Sheppard et al. 2015). In the south basin where Sauger biomass was the highest in the lake, the prey fish community was made up of more species, and biomass of the prey fish community was relatively stable over the time series.

Assessment of condition is one tool used to evaluate the status of a population. Relative weight is an index of plumpness commonly used to measure the overall health of a fish by calculating the ratio of the weight of a fish to what a growing healthy fish of the same length should weigh (Brouder et al. 2009). Relative weight can also be used to compare condition of a population to populations from similar kinds of lakes in North America. Relative weights between 80% and 100% are considered within the range found in healthy populations. Relative weights less than 80% are considered severely thin, indicating a lack of food for the fish. In Lake Winnipeg, Walleye between 400 and 600 mm in length in the north basin were above average plumpness in 2009 to 2012, with relative weights in the range of 98% to 103%. When compared to 83 populations of Walleye, Lake Winnipeg north basin Walleye were above the 75th percentile (Figure 11-8; Brouder et al. 2009). In the south basin, during the same time period, Walleye had mean relative weights of 87%, in the 50th percentile. In 2013, when Rainbow Smelt biomass in the north basin during the summer had decreased to 20% of the long-term mean, relative weight of Walleye in the north basin decreased to 93% and in the south basin to 85%, both still within the healthy range. By 2015, however, relative weight of Walleye in both the North and the south basins were at and below levels which indicate a lack of food (80% in the north basin, 78% in the south basin), both within the 5th percentile

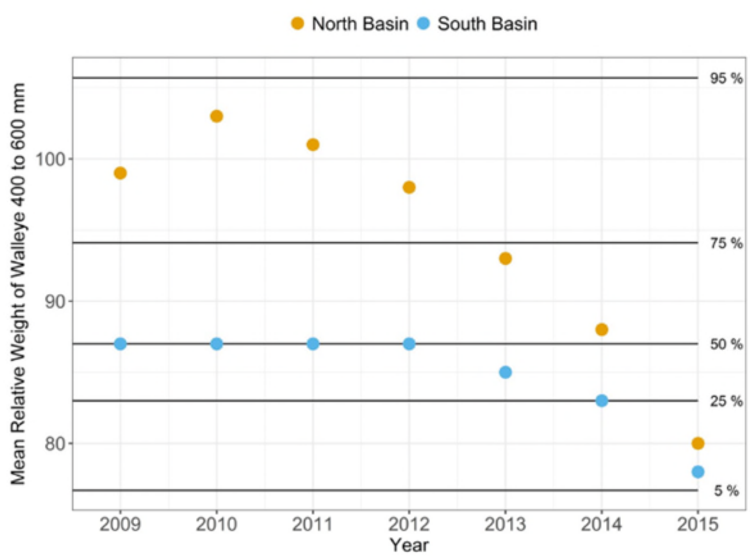


Figure 11-8: Relative weight of Walleye between 400 and 600 mm in length from the north and south basins (fish weight compared to a length-specific standard weight, which is an index of plumpness). Percentiles on the right vertical axis show how the condition of Lake Winnipeg fish compared to 83 populations of Walleye from similar kinds of lakes. Figure courtesy of C. Charles, Fisheries and Oceans Canada.

compared to other populations in North America. The decline in condition is concerning as it occurred while relative abundance of Walleye decreased (from 2013 to 2015), resulting in less density dependent effects on growth.

Lake Winnipeg – Red River Angling Fishery

The Lower Red River flows into Lake Winnipeg at the southern end of the lake, supporting an intensive angling fishery for many species of fish, notably Channel Catfish (*Ictalurus punctatus*) and Walleye. Based on results of the most recent five-year national recreational fishing survey available, ending in 2010, over 50,000 anglers fished more than 389,000 days on the Red River and Lake Winnipeg combined (FOC 2012). The most harvested/kept species were Channel Catfish, Walleye, and Yellow Perch. The Red River and Lake Winnipeg angling fisheries contribute an estimated $20,000,000 to $32,000,000 annually and represent 19.3% of total expenditures directly attributable to recreational fishing in Manitoba.

The Red River from the St. Andrews Lock and Dam, to Lake Winnipeg sustains the most intensive winter angling fishery in Manitoba. Annually, the number of ice fishing shelters on the Red River increased to a maximum of approximately 520 shelters in 2009/2010. In 2014/15, there were approximately 350 to 375 shelters on the Red River and 150 shelters on Lake Winnipeg, and many more if portable shelters were included (Manitoba Agriculture and Resource Development Conservation Officer, personal communication). From January to April 2017, there were a total of 239 permanent shelters, 1087 portable shelters and 579 groups fishing with no shelters on the south basin of Lake Winnipeg (Kroeger et al. 2017). A total of 12,660 people were surveyed using several methods (interviews, cards, and enforcement contacts), as a part of a creel survey of the south basin of Lake Winnipeg during the winter of 2017 (Kroeger et al. 2017). A total number of 14,792 fish were harvested by 7577 anglers, 96% of harvested fish were Walleye (14,170 fish). Other species harvested included Sauger (246), Yellow Perch (155), Burbot (*Lota lota*) (107), Northern Pike (*Esox lucius*) (47) and Lake Whitefish (*Coregonus clupeaformis*) (10). Roughly 54% of Walleye caught were harvested and 46% (11 990 Walleye), were released (Kroeger et al. 2017).

Lake Winnipeg Commercial Fishery

Lake Winnipeg contributes 57% of the province's total production and 75% of the landed value of the harvest in Manitoba. Average annual landings from Lake Winnipeg from 2009 to 2015 were 6,539,739 kg, with an average value of $15,357,975 (Manitoba Agriculture and Resource Development, Wildlife and Fisheries Branch, personal communication). From 2009 to 2015, 57% of all licensed fishers in Manitoba were employed on Lake Winnipeg, totaling 814 fishers and helpers per year with an average income per licensee of $18,889 (Manitoba Agriculture and Resource Development, Wildlife and Fisheries Branch, personal communication).

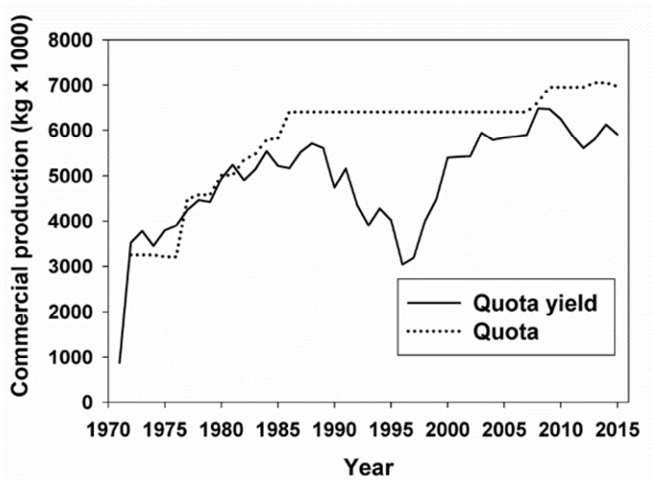


Figure 11-9: Lake Winnipeg commercial production for all quota species combined (Walleye, Sauger and Lake Whitefish) from 1971 to 2015, and total quota from 1972 to 2015. The fishery was closed in 1970 for 18 months due to mercury contamination in Northern Pike, Yellow Perch, Sauger, Walleye, and Freshwater Drum (Keleher 1970).

The Lake Winnipeg commercial fishery is managed using a Quota Entitlement system. The system was introduced in 1972 and modified in 1986 to allow transfer of individual quotas. This system comprises a total lake quota for Sauger, Walleye, and Lake Whitefish combined (LWQRTF 2011), based on individual fishers' yields during 1976 to 1985. Quota has increased since 2008 (Figure 11-9).

Using annual catches of all species (quota and non-quota species) and fishing effort (estimated by the number of deliveries reported by the Freshwater Fish Marketing Corporation) from 1973 to 1982, maximum sustainable yield (MSY) for the Lake Winnipeg fishery was estimated to be 5.2 million kg or 2.2 kg/ha/y (Lysack 1986, Leach et al. 1987). Until the 2000s, whenever landings of just quota species exceeded 5.2 million kg, declines in landings followed.

The total lake quota of 6.4 million kg for Walleye, Sauger, and Lake Whitefish from 1986 to 2007 exceeded the MSY estimate for all quota and non-quota species combined by 1.2 million kg. From 2008 to 2014 the quota increased on average by 0.66 million kg in the absence of current information about Lake Whitefish stocks status. Quota on Lake Winnipeg in 2014 was 6.7 million kg plus an additional 0.64 million kg through the whitefish optimization program for a total of 7.3 million kg. Due to a quota buyback program, the 2015 quota decreased to 6.97 million kg (W. Galbraith, Manitoba Agriculture and Resource Development, personal communication).

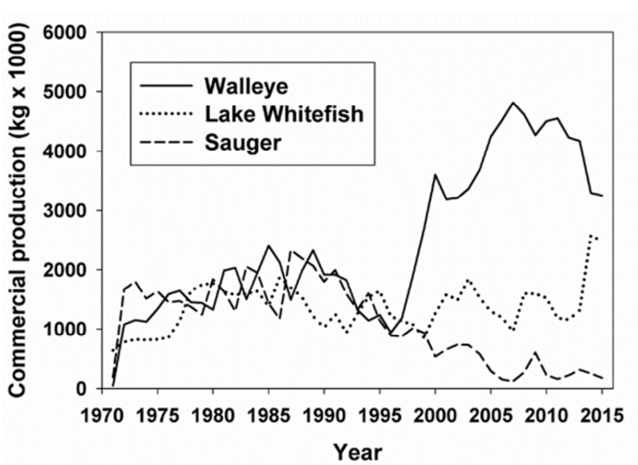


Figure 11-10: Lake Winnipeg commercial production for the three quota species separately, 1971-2015.

Walleye have been the most valuable species since at least the 1980s (Lysack 1986, MSD 2017) and therefore, targeted by the commercial fishery. Increased quota in the mid-1980s caused a large contraction in the fishery until the mid-1990s. Following the arrival of Rainbow Smelt (Campbell et al. 1991) and increased phosphorus loading to the lake (McCullough et al. 2012), commercial landings of Walleye increased from the mid-1990s, reaching its highest level in 2007, when landings were 4.8 million kg (2.02 kg/ha), nearly double any previous peak (Figure 11-10).

TEXT BOX 3: Mercury in Fish Tissue

By: Wolfgang Jansen (North/South Consultants Inc.)

Mercury is naturally found in small amounts in rocks, soils, water and organisms. It enters temperate zone aquatic systems directly through atmospheric deposition of mercury released mainly from smelters, thermal power generating plants, and volcanic eruptions, via run-off from the watershed, municipal effluents, or the flooding of land. Most of this mercury is inorganic (Hg^{2+}), but some enters waterbodies as the more toxic organic form methylmercury (MeHg). MeHg can also be formed by primarily bacterial methylation of inorganic mercury (Jensen and Jernelöv 1967) in the sediments, the water column, or fringing wetlands of lakes and rivers. MeHg is readily incorporated into biota and bioaccumulates and biomagnifies up the food chain (Watras et al. 1998). Thus, the primary pathway fish take up MeHg is through their diet (Hall et al. 1997, Hrenchuck et al. 2012), and concentrations typically increase at successively higher trophic levels such that piscivorous species have the highest MeHg levels. In most large-bodied freshwater fish, total mercury in the axial musculature primarily (> 90%) is in the form of MeHg (Lockhart et al.1972, Van Walleghem et al. 2007, Lescord et al. 2018). Henceforth, mercury is used to denote MeHg.

While there is a mounting body of evidence from mainly laboratory experiments that mercury in environmentally relevant concentrations has the potential to affect fish physiology, reproduction and behavior (Sandheinrich and Wiener 2011), detrimental effects on wild fish populations have not been conclusively documented (Bilodeau 2016). The main concern of elevated fish mercury concentrations is the transfer to and further bioaccumulation of mercury in humans, because it represents a potent neurotoxin that is particularly damaging to the developing brain (Clarkson 2002). For most people, fish is the primary source of dietary mercury intake (Mergler et al. 2007).

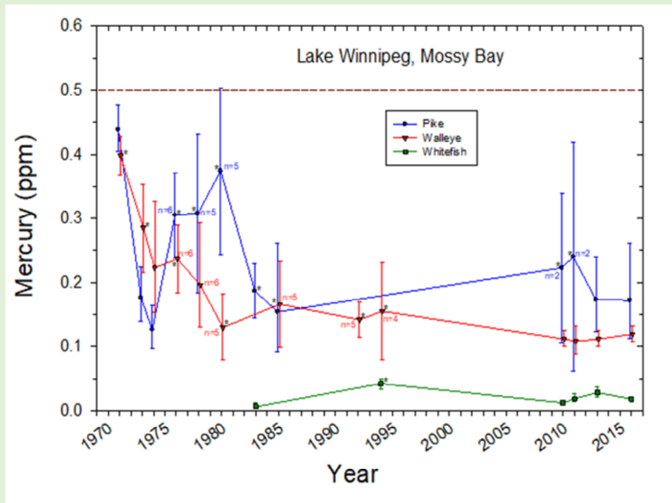


Figure A: Mean (95% confidence limits, CL) length standardized muscle mercury concentrations of Northern Pike, Walleye, and Lake Whitefish from Lake Winnipeg, Mossy Bay for years 1971–2016; an asterisk indicates that the relationship between fish length and mercury concentration was not significant and the arithmetic mean was used; n represents sample size (provided if less than 10); the 0.5 ppm Health Canada standard for retail fish is indicated by the stippled line.

Mercury concentrations in fish from Lake Winnipeg measured for the past 45 years have guided fisheries management decisions, including the closure of the fishery from April 1970 to the end of 1972, and marketing restrictions on predatory species for some years thereafter (Bligh 1970, Derksen 1978a, Derksen 1978b, Derksen 1979, Moshenko and McGregor 1978). Despite the potential impacts to the Lake Winnipeg fishery, relatively little published information exists on fish mercury concentrations from Lake Winnipeg collected after the early 1970s.

In this summary, data for Lake Whitefish (*Coregonus clupeaformis*), Northern Pike (*Esox Lucius*), and Walleye (*Sander vitreus*) from Mossy Bay at the northern tip of Lake Winnipeg collected under the Coordinated Aquatic Monitoring Program (CAMP) between 2010 and 2016 were combined with earlier (1971–1994) results for species captured from

Continued next page

Text Box 3 continued

or close to Mossy Bay and available in the Manitoba Fish Mercury Database (MFMD). A full discussion on methods used in this analysis is contained in the Appendix. Although most of the historic yearly data were subject to the limitations outlined in the Appendix, the entire dataset allowed some cautious interpretation of time trends in mercury concentrations for three important commercial fish species.

A Comparison of Mercury Concentrations in Lake Winnipeg Fish over Time

Mean mercury concentrations of Northern Pike and Walleye from Mossy Bay were highest when analyzed from large numbers of individual fish from commercial and survey samples in 1971 (Figure A). The standard mean for Northern Pike in 1971 was 0.44 ppm, and 52% of the 82 individuals exceeded the current Health Canada (HC) standard of 0.5 ppm for retail fish. In partial contrast, only 18% of the 62 Walleye and 11% of the 19 Sauger analyzed in 1971 exceeded the HC standard, resulting in arithmetic means of 0.40 ppm and 0.36 ppm, respectively.

Concentrations in Northern Pike and Walleye declined significantly in 1973 and 1974 (Figure A), the last year fish from Mossy Bay were analyzed with larger samples prior to the re-institution of fish mercury monitoring under CAMP in 2010. Concentrations in both these piscivores fluctuated substantially, particularly in pike, from 1976 to 1985 when sample sizes mainly ranged between five and six fish (Figure A). By 1985, mean concentrations in Northern Pike and Walleye were 0.15 ppm and 0.17 ppm, respectively, not significantly different from means measured in subsequent years until 2016 (Figure A). Current (2010-2016) standard means are consistently at 0.17 ppm for Northern Pike and 0.11–0.12 ppm for Walleye. With the exception of one very large pike captured in 2016, none of the 294 fish (including 123 whitefish) analyzed over this time period have exceeded the HC standard (see Figure B for data from 2010-2013).

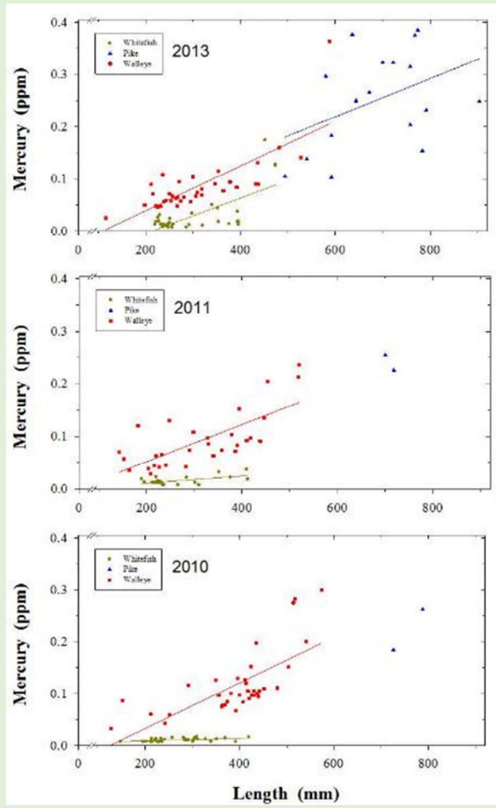


Figure B: Relationship between mercury concentration and fork length for Lake Whitefish, Northern Pike, and Walleye from Mossy Bay of Lake Winnipeg from 2010–2013. Significant linear regression lines are shown.

Standard means of Lake Whitefish, which mainly feed on invertebrates, were first measured in 1985 (0.01 ppm) and have ranged between 0.01–0.03 ppm since (Figure A). The one higher concentration of 0.04 ppm is for an arithmetic mean of 13 fish collected in 1994, which were on average substantially larger than the standard length for the species (350 mm). Apart from the above species, Yellow Perch (*Perca flavescens*) is the only other species for which individual mercury concentration are available for Mossy Bay. Ten perch captured in 1985 measuring between 230 and 310 mm had an arithmetic mean of 0.20 ppm, and two perch of 70 and 82 mm lengths had concentrations of 0.008 ppm and 0.011 ppm, respectively. As the timeline of mercury concentrations in Northern Pike and Walleye presented here indicates, current (2013–2016) concentrations have declined compared the maxima measured in 1971 and have remained stable since the mid-1980s (Figure A). Concentrations measured in Northern Pike, Walleye, and Lake Whitefish in 2013 and 2016 are among the lowest for the 25 waterbodies regularly monitored in Manitoba (CAMP 2017).

Until the 2000s, when commercial landings of Walleye reached roughly 2.3 million kg (0.97 kg/ha/y) in a given year, they decreased in subsequent years (Figure 11-11). After 2000, the Walleye stock may have been able to support higher yields given higher total phosphorous and establishment of Rainbow Smelt, though an updated target for lake-wide sustainable yield has not been identified.

In the early to mid-1990s, Walleye landings decreased, following Walleye landings that exceeded 2.3 million kg (0.97 kg/ha), when total annual mortality was 70% (W. Lysack, Manitoba Conservation and Water Stewardship). Based on guidance from Lester et al. (2000, 2014) the total annual mortality for a safe fishing level would be 44% for Walleye in Lake Winnipeg. Population assessments using index gill net survey data found total annual mortality of Walleye (ages 5 to 9) increased from 48% in 2013, to 50% in 2014, and 54% in 2015, based on five year averages of cohort-specific mortality rates. Maximum sustainable yield (MSY) of Lake Winnipeg Walleye would be achieved at 39% (G. Klein, Manitoba Agriculture and Resource Development, unpublished data). Walleye mortality rate has been higher than MSY since 2013.

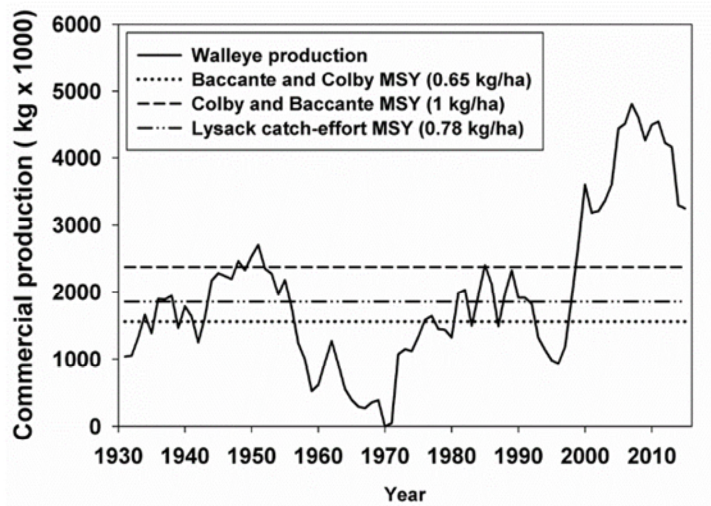


Figure 11-11: Annual yield of Walleye (kg x 1000) from the Lake Winnipeg fishery from 1931 to 2015 (MCWS 2016, MSD 2017). Three estimates of Walleye maximum sustainable yield (MSY) are shown: 2.37 million kg (1 kg/ha·y from Colby and Baccante 1996), 1.56 million kg (0.65 kg/ha·y from Baccante and Colby 1996), and 1.85 million estimated from catch and effort data from the Lake Winnipeg fishery from 1971 to 2001 (0.78 kg/ha·y from W. Lysack, Manitoba Conservation). The fishery was closed in 1970 for 18 months (Keleher 1970).

Population assessments from the index gill net survey provided estimates of Sauger total annual mortality (all age classes), which increased from 46% in 2013 and 48% in 2014, to 51% in 2015, based on five year averages of cohort-specific mortality rates. By 2017, the total annual mortality rate for Sauger had reached 56%, higher than would support maximum fishery yield of this species, which would occur at a total annual mortality rate of 55% (G. M. Klein, Manitoba Agriculture and Resource Development, unpublished data). Deliveries of Sauger are much lower than their historic average (Figure 11-10). The decreasing trend in deliveries began in the early 1990s. Under a technical definition that fisheries are considered collapsed when deliveries decrease by 90% or more over time, the Sauger fishery on Lake Winnipeg has collapsed. The commercial fishery failed to deliver 100,000 kg in both the 2017 or 2018 fishing years. There is evidence that interspecific competition does exist between Sauger and Walleye (Bellgraph et al. 2008, Butt et al. 2017) and direct predation of Sauger by Walleye likely occurs at some level (Treml et al. 2014). The same process that may have resulted in higher Walleye yields since the 2000s, the arrival of Rainbow Smelt (Campbell et al. 1991) and increased phosphorus loading to the lake (McCullough et al. 2012), may have potentially been detrimental to Sauger populations but that mechanism is uncertain. Thus, it is possible that low Sauger abundance is in part related to increased Walleye in Lake Winnipeg, but overfishing is occurring relative to harvest levels that would provide for maximum sustained yield.

Stock Status Summary

The overall lake-wide status of Walleye is fair and the trend is deteriorating as relative abundance and landings decreased following at least several years of high mortality rates. Recent total annual mortality rate estimates for Walleye from 2013 to 2015 (54% in 2015) were higher than a target reference point for maximum yield that could be sustained (39%). Not only has the population contracted, Walleye condition has decreased below that of a healthy population, indicating a lack of food, despite lower Walleye densities. Prey fish biomass has undergone a strong decline in the north basin, where Rainbow Smelt biomass is less than 5% of long-term mean. With effective fisheries management, the Walleye stock may have been able to support higher yields given higher total phosphorous and Rainbow Smelt establishment after the year 2000, although an updated target for a lake-wide sustainable yield was not identified. The current ability of the food web to support Walleye landings observed in the 2000s is diminished with the virtual disappearance of Rainbow Smelt and uncertainty following the establishment of Zebra Mussels (*Dreissena polymorpha*) and lower nutrient loadings.

Historic fishery independent relative abundance and commercial fishery landings suggest Sauger have declined since at least the early 1990s. The stock size is collapsed compared to historic levels, the current status of Sauger is poor and the overall lake trend is deteriorating. Total annual mortality estimates from 2013 to 2015 (51% in 2015) were below a MSY target (55%), but climbed to 56% in 2017. Biomass of Sauger increased from 2013 to 2015 when annual mortality rates were lower than MSY. Sauger were less dependent on Rainbow Smelt for prey, and thus less affected by the strong decline of Rainbow Smelt biomass in the north basin. In the south basin, where Sauger are most abundant in the lake, the prey fish community is diverse, made up of mainly native fish species, and prey fish biomass is relatively stable.

There are no recent fishery independent survey data available to assess the status of Lake Whitefish in Lake Winnipeg and current information is, therefore, not available.

Before the quota was increased in 2008, the maximum harvest allowed by the multi-species quota of 6.4 million kg was never landed. The increases in quota since 2008 pose additional risk to a sustainable fishery, in addition to potential changes in lake productivity following Zebra Mussel detection in 2013, Rainbow Smelt biomass less than 5% of long term means, and efforts to reduce nutrient loading to address eutrophication.

Long-term monitoring can be used to provide insights into effects of changes in the food web, lake trophic state, establishment of non-native species, and climate change, on dynamics of fish populations and fisheries (Casselman et al. 1999). Lake-wide monitoring of prey fish populations has provided considerable insight into predator-prey dynamics within the Lake Winnipeg ecosystem, particularly the interaction between Walleye and Rainbow Smelt. Rainbow Smelt populations have now collapsed and uncertainty remains about the response of Walleye populations in Lake Winnipeg, specifically the north basin. Long-term fish community and fisheries monitoring programs accompanied by surveillance of other ecosystem components is important given the uncertainty of the north basin prey fish community, Walleye and Sauger stock status and trend, and consumer sustainability concerns (e.g. MBASW 2015). If decreased pelagic productivity results from establishment of zebra mussels and reduced nutrient loading, juvenile and prey fish growth and biomass may be impacted due to bottom-up limitations. These bottom-up limitations would directly affect biomass of top predators and fisheries yields.

12.0 AQUATIC INVASIVE SPECIES

By: Scott Higgins (IISD Experimental Lakes Area Inc.) and Wolfgang Jansen (North/South Consultants Inc.)

Aquatic invasive species (AIS) are non-native plants, animals, viruses or parasites that occur outside their native ranges and threaten native aquatic species, aquatic ecosystem stability, and/or any ecological goods and services (e.g. commercial, agricultural, recreational activities) provided by the aquatic ecosystems in which they establish. Not all non-indigenous (NIS) or non-native species are considered invasive; the definition of AIS implies measurable or serious 'harm' or the threat of serious harm to native aquatic communities and/or ecological goods and services (EGS). In fact, many NIS pose minimal risk to invaded ecosystems and, in some cases, these species may confer benefits (e.g. economic, recreational, biodiversity). Further, species defined as AIS may vary considerably in terms of their risk and impacts to native aquatic communities and EGS. The recognition that not all NIS or AIS are equivalent in terms of their risk is important from a science and management perspective, allowing researchers and management agencies to focus their efforts on a smaller subset of species that pose the greatest risk.

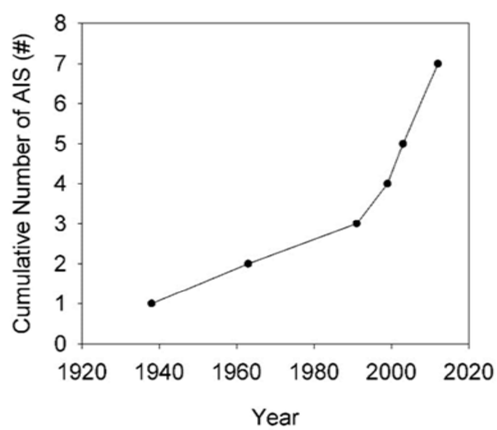


Figure 12-1: Cumulative number of known aquatic invasive species in Lake Winnipeg.

While Lake Winnipeg has considerably fewer NIS than the Laurentian Great Lakes, the number has risen in recent years (Figure 12-1, Table 12-1) and has included some AIS that pose a threat to native biota and the EGS (e.g. water quality, fishery, industry) that the lake ecosystem provides. Additionally, there are species present in Manitoba but not yet present in Lake Winnipeg that could represent a future threat (Table 12-2). Since the previous State of Lake Winnipeg report (EC and MWS 2011), both Spiny Water Flea (*Bythotrephes longimanus*) and Zebra Mussels (*Dreissena polymorpha*) have established a presence in the lake. This section provides an update on their status in Lake Winnipeg.

Table 12-1: Aquatic invasive species detected in Lake Winnipeg.

Common name	Taxonomic name	First observance
Common carp	*Cyprinus carpio*	1938
White Bass	*Morone chyrsops*	1963
Rainbow Smelt	*Osmerus mordax*	1991
Eubosmina	*Eubosmina coregoni*	1999
Asian tapeworm	*Bothriocephaus acheilognathi*	2003
Spiny waterflea	*Bythotrephes cederstroemi*	2012
Zebra Mussel	*Dreissena polymorpha*	2012

Table 12-2: Aquatic invasive species detected in Manitoba but absent from Lake Winnipeg.

Common name	Taxonomic name	Location	First observance
Feral Goldfish	*Carassius auratus*	Retention ponds in Winnipeg area	
Freshwater jellies	*Craspedacusta sowerbyi*	Star Lake, Cady Lake	1960's
Black algae	*Lyngbya wollei*	Lakes in Whiteshell region (White Lake, Betula Lake, Jessica Lake)	
Purple loosestrife	*Lythrum salicaria*		
Flowering rush	*Butomus umbellatus*	Winnipeg River, King's Park (Winnipeg)	
Invasive giant reed	*Phragmites australis australis*	Winnipeg, south eastern Manitoba, Whiteshell provincial park	2011
Koi herpes virus (KHV)		Lake Manitoba, Lake St. Martin	2008
Rusty Crayfish	*Oroconectes rusticus*	Whiteshell region, Winnipeg River	2005

Zebra Mussels (*Dreissena polymorpha*)

Zebra Mussels are native to the Ponto Caspian region of Eastern Europe. The species was first detected in North America in 1986, within a harbour of western Lake Erie, and was presumed to have been translocated from Europe in ballast waters of a Trans-Atlantic shipping vessel (Carlton 2008). By 1993, the species had established populations outside of the Great Lakes basin into the Mississippi River basin and expanded its range > 3500 km southward to the Gulf of Mexico (Cope et al. 1997).

Adult zebra mussels colonizing a settling plate deployed in Gimli Harbour.

The first reports of Zebra Mussels in Lake Winnipeg occurred in the autumn of 2013, and subsequent inspections of harbours and infrastructure revealed numerous adult Zebra Mussels in four harbours on both east and west sides of the southern basin, and one private dock (near Winnipeg Beach). However, subsequent to this finding, a specimen (confirmed to be *D. polymorpha*) was provided to the Province of Manitoba that had been discovered on a piece of floating debris during the summer of 2012 on the east side of the southern basin, suggesting the first arrival of Zebra Mussels may have occurred in 2012 or earlier.

The historically dominant vectors (pathways of introduction) of AIS to Lake Winnipeg are the Red River and Winnipeg River (Table 12-1). Lake Winnipeg has an enormous watershed (23,750 km^2) and these results demonstrate the vulnerability of Lake Winnipeg to AIS invasions from 'upstream' jurisdictions (e.g. Ontario, North Dakota, Saskatchewan, Alberta). While the vector of Zebra Mussel introduction to Lake Winnipeg is not known with certainty, subsequent reports of Zebra Mussels in upstream portions of the U.S. and Canadian portions of the Red River suggest downstream transport of either veligers (i.e. mussel larvae that mainly reside in the water column) or adults on floating debris was the most probable.

As with other systems, Zebra Mussels can spread quickly within large lakes either moving with currents or aided by human activities (e.g. attachment to the hulls of vessels). While adult mussels were detected in several harbours in the fall of 2013, veligers were not detected until the following year. During 2014, veligers were detected in numerous locations in the southern basin. By 2014, veligers were detected in the northern basin and the connecting channel. While detections were more limited in 2016 (Figure 12-2), this result is most probably related to a mismatch in timing between reproductive and sampling events.

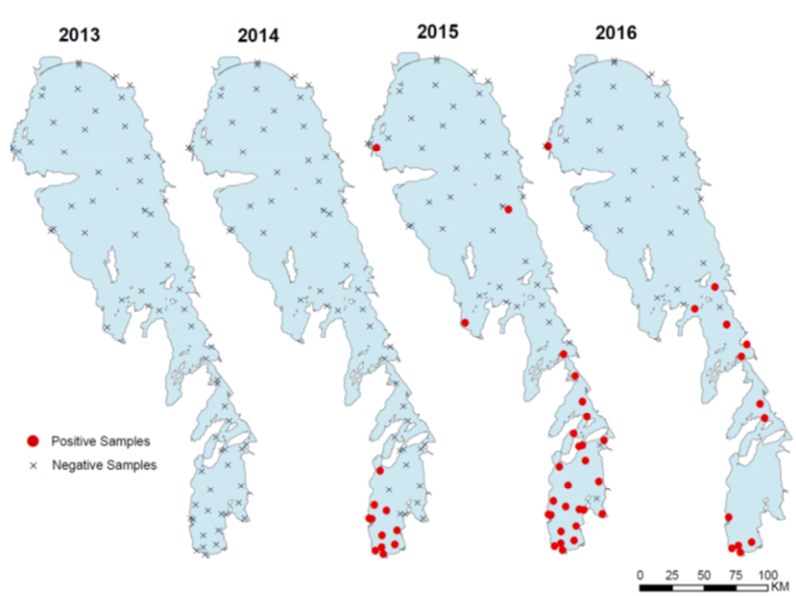


Figure 12-2: Distribution of Distribution of Zebra Mussel veligers in Lake Winnipeg from 2013–2016. Data provided courtesy of the Province of Manitoba, Fisheries Branch.

By 2016 adult mussels on settling plates (Figure 12-3) and other hard surfaces (e.g. piers, docks, etc.) were reported in only the southern portions of Lake Winnipeg's northern basin. However, given detections of veligers in various portions of the northern basin, it is likely that localized areas in the northern basin may already have been colonized, but have yet remained undetected. Such localized populations were common in the Laurentian Great Lakes during early colonization, and represent potential seed populations for colonization of remaining habitats. At present, there is reasonable evidence to suggest that the distribution of Zebra Mussels in the northern basin will increase dramatically within the next five years.

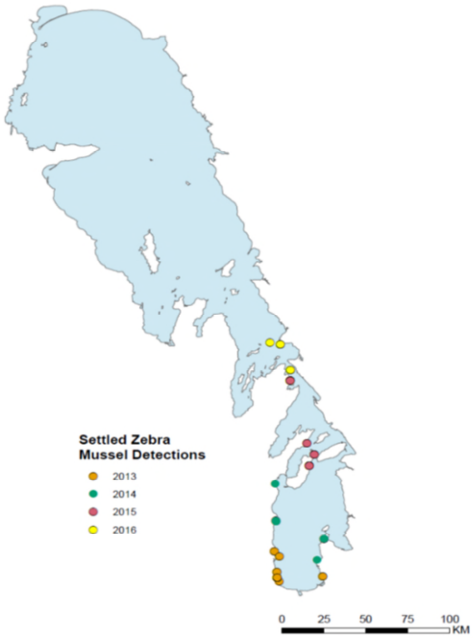


Figure 12-3: Distribution of adult Zebra Mussels detected on settling plates or other hard surfaces from 2013–2016, courtesy of the Province of Manitoba, Fisheries Branch.

Spiny Water Flea (*Bythotrephes longimanus*)

The history and status of the Spiny Water Flea (*Bythotrephes*) in Lake Winnipeg has recently been described by Jansen et al. (2017) and the following is a summary of that work. The species is native to large areas of Eurasia (Grigorovich et al. 1998, Strecker 2011) and was first detected in North America in Lake Ontario in 1982 (Johannsson et al. 1991). Similar to Zebra Mussels, the Spiny Water Flea likely arrived in North America via ballast water from a shipping vessel (Sprules et al. 1990). The species spread throughout the rest of the Laurentian Great Lakes and by 1989-1991 was present in inland lakes in southern Ontario (Yan et al. 1992) and in northern Minnesota, USA (Schuldt and Merrick, pers. comm in Yan et al. 1992).

The Spiny Water Flea was first documented in the Lake Winnipeg watershed at Saganaga Lake, Ontario in 2003 (USGS 2016a, in Jansen et al. 2017). By 2007, *Bythotrephes* was found in Lake of the Woods, the source of the Winnipeg River, which flows into the south basin of Lake Winnipeg. In 2011, several specimens were found in the stomachs of fish that were captured in Lake Winnipeg near the mouth of the Winnipeg River. Additionally, individual specimens of *Bythotrephes* were caught with plankton nets in the lake, near the location where the fish were caught (Jansen et al. 2017). Targeted sampling of *Bythotrephes* by ship-based plankton net tows in Lake Winnipeg started in 2012.

In the summer of 2012, *Bythotrephes* were present throughout much of the south basin of the lake but were most abundant near the bay at the mouth of the Winnipeg River. The species was captured at one site at the south end of the narrows but there was no evidence of the presence of the species in the north basin. By fall 2012, the species had dispersed throughout the narrows and had colonized the north basin of the lake (Figure 12-4).

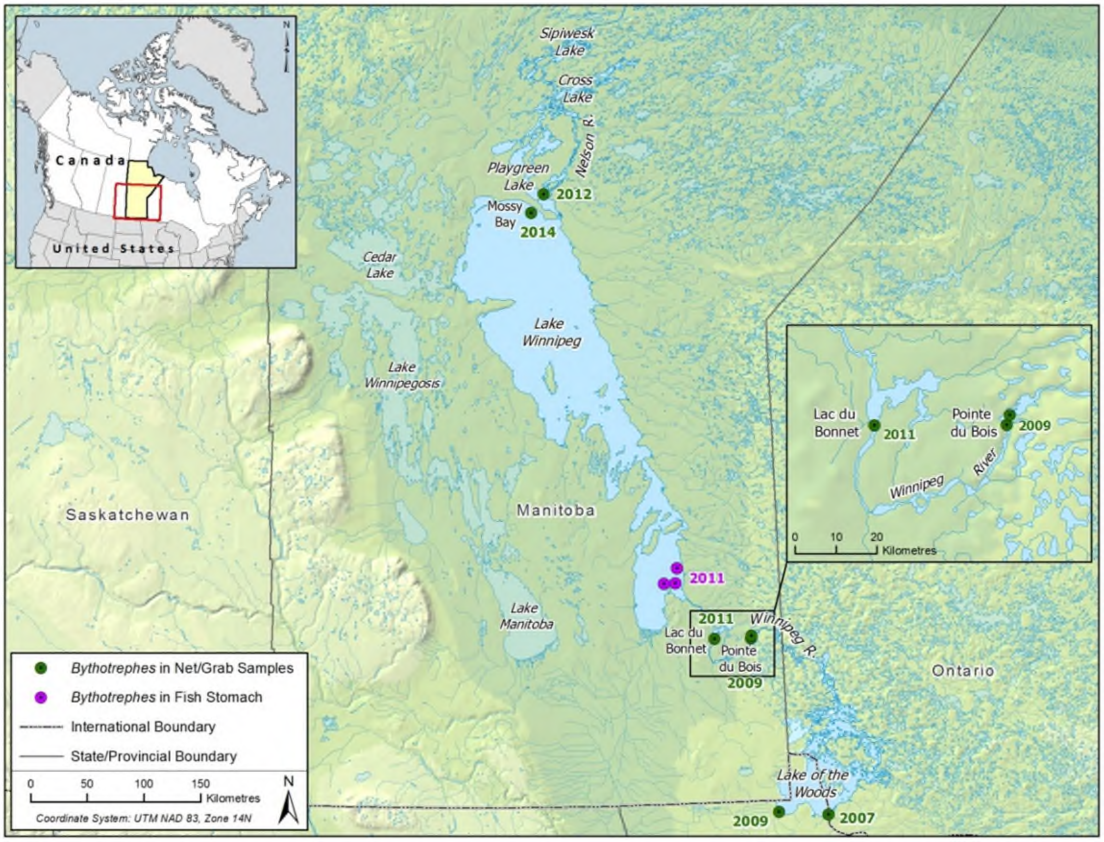


Figure 12-4: Invasion history of the Spiny Water Flea (Bythotrephes longimanus) in Manitoba waterbodies from non-targeted sampling. Years indicate the first record at a location. Note that targeted sampling in 2012 documented B. longimanus in Lake Winnipeg just south of Mossy Bay (see Figure 11-6). The first record within the Lake Winnipeg watershed was in 2003 in Saganaga Lake, Ontario (350 km east of Lake of the Woods). Reproduced from Jansen et al. (2017) with permission.

Mean densities of Spiny Water Flea in the South basin, the North basin, and the narrows of Lake Winnipeg in 2012 ranged from 2.2–9.2 individuals/m^3 (for densities at each station see Figure 12-5) and are at the lower end of ranges of densities reported elsewhere in North America (e.g. Barbiero and Tuchman 2004, Walsh et al. 2016). However, inconsistent sampling methodologies, particularly net diameter and mesh size, among the different studies make comparisons difficult. The relatively low densities in Lake Winnipeg may be a result of the recent introduction of the species and it may be expected that densities will increase over time.

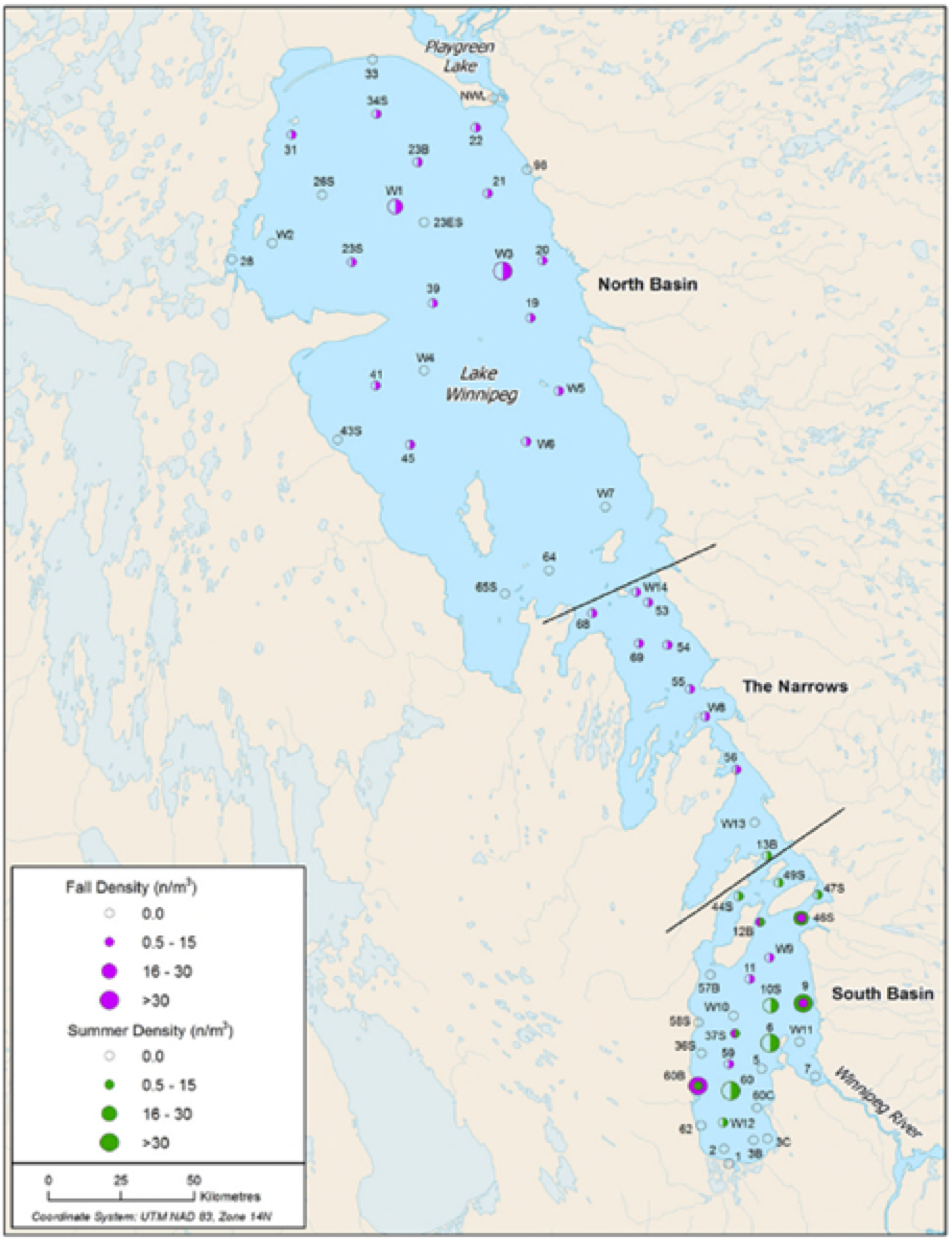


Figure 12-5: Distribution and density of Bythotrephes longimanus in Lake Winnipeg based on ship-based pelagic plankton tows at 65 sites in the summer (13 July–9 August) and fall (13 September–10 October) of 2012; three lake areas are separated by lines; n= number of individuals. Reproduced from Jansen et al. (2017) with permission.

Aquatic Invasive Species and Lake Management

The most significant challenge with AIS management, particularly in large systems like Lake Winnipeg and the Red River, is the lack of feasible and cost-effective control measures. Thus, there is a strong incentive for policy and management of AIS to focus on prevention, including: legislative tools (e.g. regulations, prohibitions), coordination and enforcement (e.g. check stops for trailered boats, communication and coordination between jurisdictions), education (e.g. information campaigns and materials), and pro-active measures such as the installation of decontamination stations at high use boat launches. In 2015, the Government of Canada introduced new legislation to target the possession and movement of AIS within Canada, providing both federal (i.e. Canadian Border Services) and provincial agencies with additional regulatory and enforcement tools (e.g. prohibition of importation, transportation and release; authority for inspections; requirements of decontamination). At the same time provincial governments also drafted provincial regulations providing guidance and tools for managing the threats of AIS. These legislative and policy tools provided necessary powers for regulation and enforcement of potential AIS vectors. However, the success of prevention programs are also heavily dependent on public education and awareness of AIS, voluntary public action, and access to information and infrastructure associated with decontamination.

In the context of Lake Winnipeg, an evaluation of the long-term effects of Zebra Mussel and Spiny Water Flea on the food web will require careful monitoring, research and ecosystem modelling. The introduction of Spiny Water Flea has been associated with lower diversity, abundance and biomass of zooplankton (e.g. Yan et al. 2002, Boudreau and Yan 2003, Barbiero and Tuchman 2004, Weisz and Yan 2011), reduced fish growth (Feiner et al. 2015), and lower epilimnetic zooplankton productivity (Strecker and Arnott 2008) resulting in less food for planktivorous fish. Following Zebra Mussel invasion to Lake Erie, nearshore areas experienced a resurgence of coastal algal blooms, increases in pathogenic bacteria, increased fish and avian deaths, and large shoreline wash-ups of dead dreissenid shells (Higgins et al. 2008). Further, while improvements in offshore water quality appeared to be occurring following decades of point source phosphorus control, model outputs used to set phosphorus loading targets began to diverge from monitoring data (Hecky et al. 2004). The 'nearshore shunt hypothesis' proposed by Hecky et al. (2004), was an attempt to reconcile the apparently conflicting changes in the nearshore and offshore water quality and poor model performance. Their hypothesis suggested that dreissenid mussel filtration rates in nearshore waters created a long-term process where nutrients were stripped out of coastal waters, making them more bioavailable in the nearshore and also leading to higher burial rates. The potential impact of this process in the Lake Winnipeg context should be carefully evaluated.

Because of the complexity of potential physical and biological interactions and potential synergistic negative effects of both species on zooplankton abundance, the directions and magnitude of the changes in the Lake Winnipeg food web are difficult to predict. For example, Zebra Mussel invasion may increase water clarity, which would create habitat conditions that would favour higher Spiny Water Flea abundance. Evidence from other invaded systems suggests that for effective lake management, monitoring programs should include: 1) estimates of invasive species across different habitat types; 2) monitoring of trophic levels at the base of the food web (i.e. phytoplankton and benthic algae); 3) monitoring of key diet items for fish and clams; 4) monitoring of water quality indicators such as light attenuation (e.g. Secchi depth, extinction coefficients), and soluble and particulate nutrient concentrations; and, 5) monitoring of fish populations, including those of interest to indigenous, commercial and recreational stakeholders. An improved understanding of the potential interactions between the Spiny Water Flea, Zebra Mussels and current ecological issues in Lake Winnipeg (e.g. phosphorus, algal blooms and toxicity, species at risk, climate change), and improvements (e.g. inclusion of Zebra Mussel filtration) of ecological modeling capacity would be useful in terms of lake management.

13.0 AQUATIC SPECIES AT RISK

By: Eva Enders and Doug Watkinson (Fisheries and Oceans Canada)

Wildlife species that are Canadian indigenous species, subspecies or distinct populations and at risk of becoming extirpated or extinct are protected by the Canadian *Species at Risk Act* (SARA). The Committee on the Status of Endangered Wildlife in Canada (COSEWIC), an independent advisory panel to the Minister of Environment and Climate Change Canada, assesses the conservation status of wildlife species based on a status report. Members of COSEWIC are wildlife biology experts from academia, government, non-governmental organizations, the private sector, and Aboriginal knowledge-holders. Once COSEWIC has determined the designation for a given species, the Minister of Environment and Climate Change Canada considers the species for listing under SARA. Under the *Act*, species or designatable units (DUs) thereof and their critical habitats receive protection. Depending on their status, species at risk are listed in one of five categories:

- *extinct* – a wildlife species that no longer exists;
- *extirpated* – a wildlife species that no longer exist in the wild in Canada, but exists elsewhere;
- *endangered* – a wildlife species that is facing imminent extirpation or extinction;
- *threatened* – a wildlife species likely to become an endangered species if nothing is done to reverse the factors leading to its extirpation or extinction; and
- *special concern* – a wildlife species that may become a threatened or an endangered species because of a combination of biological characteristics and identified threats.

The goals of the *Species at Risk Act* are to avoid wildlife species from becoming extinct or extirpated, help with the recovery of extirpated, endangered or threatened species, and ensure that species of special concern to not become endangered or threatened.

There are many anthropogenic threats that may endanger wildlife species (Salafsky et al. 2008; Faber-Langendoen et al. 2012: Master et al. 2012), the ones that can affect freshwater aquatic resources include:

- *Residential and commercial development* – Housing, urban, commercial, industrial, tourism, and recreation areas can all alter or destroy habitat such that it is a detriment to a species.
- *Agriculture and aquaculture* – Aquaculture can lead to pollution, negative species interactions resulting from escapees, and disease and pathogen transmission.

- *Energy production and mining* – Mining and quarrying can result in habitat loss or degradation if it occurs below the high water mark or in the flood plain.
- *Transportation and service corridors* – Roads, railways, utility and service lines can modify or reduce habitat and may result in barriers to movement.
- *Biological resource use* – Fishing and harvesting aquatic resources, either by targeted fishing for a particular species or as incidental harvesting such as fisheries bycatch can affect a the fish community.
- *Human intrusions and disturbance* – Recreational activities, works and activities can modify habitat, reduce fitness or kill individuals.
- *Natural system modifications* – Fire and/or fire suppression, dams, water management, and other ecosystem modification can alter habitat.
- *Invasive and other problematic species and genes* – Invasive non-native species, problematic native species, and introduced genetic material can affect the resident species, e.g., new species can parasitize or predate upon residents, hybridize, compete for food, bring unfamiliar diseases, modify habitats and/or disrupt important interactions.
- *Pollution* – Discharge of household/urban waste, industrial, military, agricultural and forestry effluents, garbage and solid waste, and air-borne pollutants into the environment may affect species abundance.
- *Climate change* – Climate change can result in habitat shifting and alteration, droughts, temperature extremes, storms, and flooding making it difficult or even impossible for many species to survive in their current distribution range.

In most instances, more than one of these anthropogenic factors is having an impact on biodiversity. The most current List of Wildlife Species at Risk in Canada is available at the Species at Risk Registry website.

Profiles for Listed Aquatic Species in the Lake Winnipeg Basin

Mapleleaf *(Quadrula quadrula)*

COSEWIC Designation: Endangered

SARA Status: Endangered

Special significance: Other members of the genus *Quadrula* have suffered population declines throughout their range; *Quadrula quadrula* may represent the best opportunity to maintain representation of the genus for the long term in North America. Canadian populations occur in disjunct watersheds that are separate from the main distribution of this species in the US. They have unique haplotypes and ecological information important in conserving the diversity of the species.

Distribution: In Manitoba, Mapleleaf populations have been documented in the Red River, the lower reaches of the Assiniboine and Roseau rivers, and recently in the several tributaries on the east side of Lake Winnipeg (Bloodvein, Brokenhead, Wanipigow, Bradbury, Maskwa, and Winnipeg rivers).

Habitat needs and life history: Mapleleaf is usually found in medium to large rivers with slow to moderate currents and firmly packed substrate of sand, coarse gravel or clay/mud. During spawning, males release sperm into the water and females siphon it out of the water and filter it with their gills. In the marsupium (special area of the gill) of the female, the eggs are fertilized and develop into glochidia (larvae). This brooding period is generally short, lasting from late spring to early summer. Once released, glochidia require a fish host, Channel Catfish *(Ictalurus punctatus),* for 50 to 60 days to develop into juveniles. They then drop off their host and exist free-living or in the substrate.

*Threats**:*** In Manitoba, Mapleleaf is threatened by habitat loss and degradation due to aquatic invasive species, specifically the recently introduced zebra mussel, habitat modifications such as riprap and diking, and water quality deterioration due to nutrient enrichment from point and non-point sources associated with municipal and urban effluent and extensive agriculture, respectively.

Chestnut Lamprey *(Ichthyomyzon castaneus)* - Saskatchewan-Nelson River Drainage Unit

COSEWIC Designation: Non-active

SARA Status: Special Concern

Special significance: The parasitic Chestnut Lamprey is endemic to Canada. The Chestnut Lamprey has a closely related, nonparasitic species, the Southern Brook lamprey *(Ichthyomyzon gagei)* that does not occur in Canada. Another parasitic-nonparasitic species pair, Silver Lamprey *(Ichthyomyzon unicuspis)* and Northern Brook Lamprey *(Ichthyomyzon fossor)* does occur together in Canada. Why one species pair occurs in Canada and the other does not is being studied to determine if these are pairs or ecomorphotypes of a single species. The species is generally not viewed favourably by the public, which puts the species inherently at risk.

Distribution: Chestnut Lamprey occur in streams, rivers, and lakes of the Lake Winnipeg drainage as far north as Dog Head Point in Lake Winnipeg.

Habitat: Chestnut Lampreys inhabit large creeks, moderate-sized rivers, and lakes. Spawning occurs over coarse gravel areas in streams or rivers. Juveniles need softer sediments to burrow into and adults prefer substrate and cover to hide during day.

Biology: The life cycle consists of three stages, a larval stage that lasts up to seven years, a metamorphosis stage than begins in summer and lasts until winter, and finally an adult stage that lasts 18 months. Chestnut Lamprey mate once between 7 and 9 years of age; they mate, spawn, and then die shortly thereafter. The juveniles burrow into softer sediment and feed on protozoans, phyto- and zooplankton. The adults attack a wide variety of fishes by attaching themselves to the body of their hosts consuming body fluids and muscles. Chestnut Lamprey are nocturnal.

Threats: Suitable spawning areas of Chestnut Lamprey are disappearing due to siltation and pollution. Chemical pollution can cause mortality at all ages; eutrophication can cause mortality in juveniles. The deterioration of river environments also threatens their food supply.

Lake Sturgeon (*Acipenser fulvescens*)

COSEWIC Designation: Endangered

SARA Status: Under Consideration

Special significance: Sturgeons are considered living fossils, having changed little from their ancestors of the Devonian Period. Lake Sturgeon is the only strictly freshwater species of sturgeon in Canadian waters.

Distribution: Lake Sturgeon occur in Lake Winnipeg and its tributaries, including the Bloodvein, Pigeon, Poplar, Winnipeg, Red, Saskatchewan, and Berens rivers.

Habitat: Lake Sturgeon are generally bottom-dwelling fish found in large rivers and lakes, typically at depths between 5 to 10 m, sometimes greater. Spawning occurs in the spring in fast-flowing water at depths between 0.3 to 23 m over cobble, boulder, gravel, and bedrock.

Threats: Human activities represent the most important threat to Lake Sturgeon. Historically, commercial fishing led to population declines. The after effects of these declines still dominate the demographics and recovery of many populations, including the Lake Winnipeg population. Contemporary harvest still threatens some populations, but to what extent is unclear. More recently, the direct and indirect effects of dams pose important threats. Dams result in habitat loss and fragmentation, altered flow regimes, and may increase mortality by entrainment in turbines. Habitat degradation resulting from invasive species, sedimentation from erosion, and pollution from industrial, agricultural and urban sources are also threats to the Lake Sturgeon population in Lake Winnipeg.

Bigmouth Buffalo *(Ictiobus cyprinellus)*

COSEWIC Designation: Special Concern

SARA Status: Special Concern

Special significance: Bigmouth Buffalo is considered a delicacy by some in the United States and a commercial fishery existed in Saskatchewan, dating from the 1940s ending in 1983. There is currently limited demand for Bigmouth Buffalo as a food source in Canada, but they may be found in the live food fish market in Ontario. There is some scientific interest in the species related to its taxonomy.

Distribution: Bigmouth Buffalo is found in the Red and Assiniboine rivers as well as the south basin of Lake Winnipeg. Although there has been an increase in the extent of occurrence and area of occupancy in Manitoba (Saskatchewan-Nelson River Biogeographic Zone), the species is not believed to be abundant.

Habitat: Bigmouth Buffalo are found in lakes and in slower water zones (i.e., deeper pools) of medium- to large-sized rivers. Bigmouth Buffalo are often observed in schools at midwater or near the bottom. They prefer warm, highly eutrophic waters and are less impacted by turbidity or high water temperatures than other freshwater fishes. Spawning is apparently dependent on spring flooding to provide access to flooded vegetation in shallow bays, small tributary streams, ditches, marshy areas, and backwaters.

Threats: As successful reproduction appears to be associated with flooding of shoreline vegetation, the loss of spawning habitat associated with regulated water levels is a threat to Bigmouth Buffalo. In addition, the river connectivity in the Assiniboine and Red Rivers are impeded by several barriers that limit the free movement of Bigmouth Buffalo that are shown to undertake large transboundary migration in unimpeded river sections.

Northern Leopard Frog *(Lithobates pipiens) - Western Boreal/Prairie populations*

COSEWIC Designation: Special Concern

SARA Status: Special Concern

Special significance: The Northern Leopard Frog can serve as an indicator species of ecosystem health as it plays an important ecological role as both predator and prey species. It remains one of the most widespread amphibians in Canada and has been extensively studied and used for educational purposes.

Distribution: Northern Leopard Frog occurs in North America, from southeastern British Columbia to Labrador, and from the southcentral Northwest Territories down through the central and southwestern United States, near Mexico. In the mid 1970s the species recovered from sharp declines in Manitoba, where it is now believed to be common throughout the southern regions of the province. Populations in Manitoba are not currently monitored.

Habitat: Northern Leopard Frog uses several distinct habitat types to meet its needs throughout the year. Separate sites are used for overwintering, breeding and foraging, and contiguity between these habitats is necessary for the species' survival. Overwintering sites are well-oxygenated bodies of water that do not freeze to the bottom. Streams, creeks, rivers, spillways below dams, and deep lakes and ponds may all provide appropriate overwintering conditions. Breeding occurs in pools, ponds, marshes and lakes, and may occasionally occur in slow-moving streams and creeks. The tadpoles also use these types of streams. A typical breeding pond is 30 to 60 m in diameter and 1.5 to 2.0 m deep; it is located in an open area with abundant vegetation and no fish. In the summer, the frogs are found in a wide variety of habitats, particularly moist upland meadows and native prairie; riparian areas and ponds facilitate dispersal and provide additional corridors for movement between habitats.

Biology: The Northern Leopard Frog emerges from overwintering ponds in early spring shortly after the ice has melted, when the water temperature rises to 7 to 10 °C. The eggs hatch in 9 to 14 days, depending on water temperature. Tadpoles initially remain close to the egg mass after hatching, but disperse after a few days. Free-swimming tadpoles feed on floating vegetation and dead and decaying organic matter. Tadpoles take 60 to 90 days to complete their metamorphosis and transform into frogs. Adult frogs feed on insects, spiders, worms, snails, slugs, small birds, fish, snakes and other frogs. The Northern Leopard Frog typically lives for a maximum of four to five years.

Threats: Observed declines in Northern Leopard Frog are often associated with habitat loss, degradation and fragmentation. The species' diversified habitat requirements make it particularly vulnerable to such changes. Removal or modification of even one of the three habitat types used by Northern Leopard Frogs may render the landscape unsupportive of the species' requirements. Recreational and commercial harvest is still permitted in Manitoba. Collection of individuals may contribute to the decline of the population. Embryo mortality may be attributed to ultraviolet radiation.

Snapping Turtle *(Chelydra serpentine)*

COSEWIC Designation: Special Concern

SARA Status: Special Concern

Special significance: Healthy populations of Snapping Turtles play significant ecological roles in wetland ecosystems, consuming dead fish and other vertebrates, reducing plant biomass, and creating channels in wetlands that fish, amphibians and other reptiles use. Their eggs provide a significant source of nutrients to mammalian mesocarnivores at a critical time when these mammals are raising their young, and at other times of year juveniles and hatchlings are consumed by a wide range of vertebrate predators. Snapping Turtle have great significance for First Nations' people.

Distribution and Population: The Snapping Turtle occurs from southern Manitoba to Texas. Although the Snapping Turtle is one of Canada's more widespread turtle species, long-term studies of two populations in Ontario have demonstrated that large and apparently secure populations are vulnerable to even slight increases in adult mortality and do not recover quickly from declines. The Snapping Turtle remains relatively abundant in eastern Canada but is less often encountered in Saskatchewan and Manitoba.

Habitat: Although Snapping Turtles have been observed in shallow water in almost every kind of freshwater habitat, the preferred habitat of the species is characterized by slow-moving water with a soft mud bottom and dense aquatic vegetation. Established populations are most often located in ponds, sloughs, shallow bays or river edges, and slow streams, or areas combining several of these wetland habitats. Snapping Turtles commonly bask on offshore logs and protruding rocks, depending on air and water temperature. Females generally nest on sand or gravel banks along waterways. Upon emergence from the nest in early fall, hatchling Snapping

Turtles usually move to water, after which they bury themselves under leaf litter or debris. Snapping Turtles overwinter underwater, buried beneath logs, sticks or overhanging banks in small streams that flow continuously throughout the winter.

Biology: In Canada, females reach sexual maturity at the late age of 15 to 20 years. Mating takes place in early spring, and nesting takes place from late May to late June. Females show strong nest site fidelity. The eggs hatch from mid- to late September. Snapping Turtles are primarily diurnal, although activity is occasionally observed at night. Snapping Turtles have been known to bask to raise their body temperature. As their name implies, they have a reputation for ornery behaviour. When confronted on land, adults strike quickly with their long neck extended. Many Snapping Turtles return annually to their hibernation site.

Threats: Most of the serious threats to Snapping Turtles are events that increase adult mortality. Legal and illegal harvesting of adults, persecution and road mortality are the most prominent causes of premature death in adult Snapping Turtles. A new, rapidly increasing and far more serious threat is the illegal wildlife trade. There is a highly organized trade in turtles for food, medicine, pets and trinkets. Snapping Turtles are most vulnerable to mortality from vehicular collisions during the reproductive season because females cross roads frequently in search of suitable nesting sites, but also because soft gravel road shoulders make attractive nesting sites. Another threat is unnaturally high rates of nest predation by large populations of raccoons and other mammals. Other factors threatening the survival of the species include boat propeller strikes, bycatch from recreational and commercial fishing, dredging, road grading, and other practices.

Summary

All Canadians have a role to play in the conservation of wildlife species and their habitats. With increasing human activities and climate change, the List of Wildlife Species at Risk is likely to grow. Education and awareness of conservation needs and actions to protect habitats are essential for limiting the growth of the list. The ability for citizens to contribute directly to species at risk protection and to species conservation is a high priority under SARA. To assist with that, the Government of Canada provides resources through the Habitat Stewardship Program for projects that protect species at risk and their habitat.

The Government develops recovery strategies and action plans for endangered and threatened species, and management plans for species of special concern. Action plans identify the conservation measures required to meet the population and distribution objectives outlined in the recovery strategy. An action plan must also to the extent possible identify critical habitat or complete the identification of critical habitat if it is not fully identified in the recovery strategy. The plan also includes information on measures proposed to protect that critical habitat, methods proposed to monitor the recovery of the species, and an evaluation of the socio-economic costs of the action plan and benefits to be derived from its implementation. A management plan differs from a recovery strategy and an action plan in that it identifies conservation measures needed to prevent a species of special concern from becoming threatened or endangered but does not identify critical habitat.

Numerous recovery activities are conducted to protect and conserve species at risk including research, education and awareness, habitat restoration and enhancement initiatives, monitoring, and assessment. To guide the effective use of limited resources, national and regional planning partners establish the overall priorities annually and then specific projects are developed.

14.0 RECREATIONAL WATER QUALITY

By: Cassie McLean (Manitoba Agriculture and Resource Development)

Lake Winnipeg is known for its plentiful beaches that offer a variety of opportunities for recreational activities during the open water season including swimming, boating, kite-boarding and windsurfing. The beaches are a strong economic driver for local communities attracting visitors, cottagers and permanent residents alike. Beaches in the south basin of Lake Winnipeg are easily accessible, only a short drive from Winnipeg, and visitors to the beaches can exceed 30,000 per day, especially during the busy summer months when people head to the shore to enjoy the lake.

Manitoba Agriculture and Resource Development, in conjunction with Manitoba Health, has developed the Manitoba Clean Beaches Program to provide valuable information to Manitobans on how to protect our beautiful beaches and reduce public health risks. As part of this program, Manitoba routinely monitors densities of the fecal indicator bacteria, *Escherichia coli* (*E. coli*) at Lake Winnipeg beaches. *Escherichia coli* is a bacterium found in large numbers in all warm-blooded animals including humans, livestock, wildlife, and birds. *Escherichia coli* is widely used as an indicator of fecal contamination of freshwater beaches and studies elsewhere have linked densities of *E. coli* to illness rates of bathers in recreational waters. While *E. coli* itself does not generally cause illness, when it is present in large numbers, the risk of becoming ill from other organisms is elevated.

Figure 14-1: Beach monitoring locations on Lake Winnipeg (Clean Beaches Program, Manitoba Agriculture and Resource Development). Map source: Manitoba Agriculture and Resource Development.

Many jurisdictions, including Canada and Manitoba, have recreational water quality objectives in place to protect public health and recreational water quality at beaches. Manitoba's recreational water quality objective is 200 *E. coli* per 100 mL of water. Currently, the Province of Manitoba routinely monitors densities of *E. coli* at 19 beaches in the south basin of Lake Winnipeg as part of the Clean Beaches Program (Figure 14-1). Approximately ten Lake Winnipeg beaches were monitored historically in the 1980s and 1990s but only the most extensive dataset for the 2004 to 2018 period is presented and summarized in this section.

Generally, beaches on the east side of Lake Winnipeg are characterized by long, soft sandy beaches with a gently sloping bathing area, while smaller rocky beaches with steeper bathing areas are characteristic of the west side (Figure 14-2). Water quality at beaches can be affected by a number of factors including wind, waves, water temperature, beach characteristics, runoff and natural and anthropogenic sources of pollutants. Some beaches around Lake Winnipeg are located near cottage developments, towns and small communities; storm drains and licensed wastewater discharges (that are required to meet a 200 *E. coli* per 100 mL standard); or are more isolated and are not directly affected by discharges from urban development.

Figure 14-2: West Grand Beach (left) has characteristically smooth soft sand with gentle sloping shoreline and shallow bathing area. Winnipeg Beach (right) has rocky sand with sharper shorelines and a deeper bathing area.

Lake Winnipeg beaches are typically monitored for *E. coli* on a weekly or bi-weekly basis and are compared to the recreational water quality objective of 200 *E. coli* per 100 mL to determine the recreational risk to human health. When *E.coli* is below the objective, water quality is considered acceptable for swimming and recreation. However, when *E. coli* exceeds the recreational water quality objective, advisory signs are posted to warn swimmers of elevated bacterial counts, as a precaution to reduce the risk of illness.

Escherichia coli at Lake Winnipeg Beaches: 2004 to 2018

Recreational water quality at Lake Winnipeg beaches was typically 'good', fairly stable and showed no major changes for the 2004 to 2018 period (Figure 14-3). Recreational water quality is considered 'good' when more than 80% of water samples were below the recreational water quality objective meaning that water quality is generally protected, conditions rarely depart from desirable densities, and usually return within acceptable limits within 24 hours. The current reporting period is consistent with historical data collected from Lake Winnipeg beaches in which the vast majority of

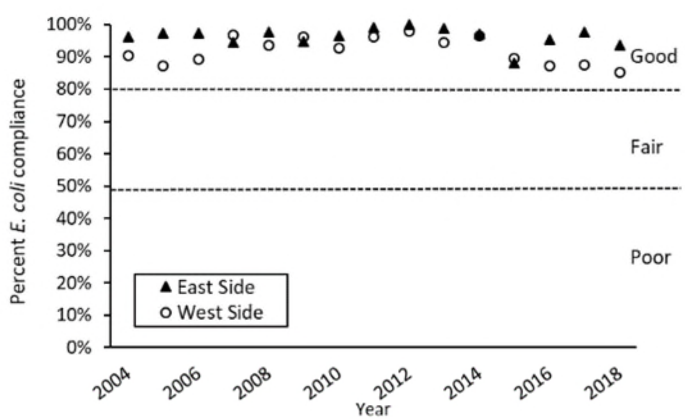


Figure 14-3: Percent of total annual E. coli geometric means from east and west side Lake Winnipeg beaches that meet the acceptable recreational objective of 200 E. coli per 100 mL.

samples were within acceptable limits (EC and MWS 2011, Williamson 1985, 1988, MWS unpublished data).

Samples collected from the west side beaches tended to exceed the recreational objective more frequently than east side beaches in most years and resulted in more frequent beach advisories and slightly lower percent compliance rates (Figure 14-4, Table 14-1). While it is uncertain why west side beaches tend to have more frequent exceedances as compared to the east side, a number of factors including wind, lake currents, beach morphology and external factors from the 'beachshed' (runoff, storm drains, discharges) may partly explain these differences. Regardless, beach water quality is generally considered good at both the east and west side Lake Winnipeg beaches.

When the average percent compliance for *E. coli* in the last five years (2014–2018) is compared to the average rate of compliance for the historic record (2004–2018), compliance at most beaches is very similar suggesting little change in beach conditions between the two periods. The lowest annual rate of compliance at each beach was generally above 70% at all beaches for the 2004–2018 period. However, the majority of beaches for the 2004–018 period showed more than an average 80% compliance, meaning recreational water quality was generally protected and within acceptable limits (Figure 14-4, Table 14-1).

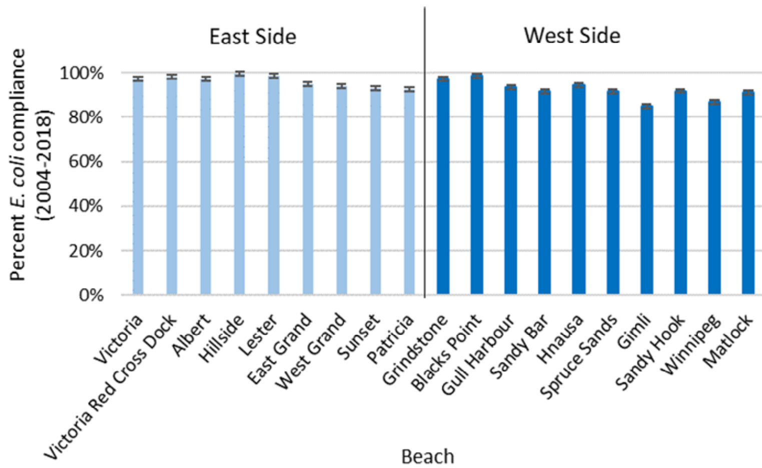


Figure 14-4: Average E. coli geometric means (2004-2018) by beach from both east and west side Lake Winnipeg beaches. Note: Location of Matlock Beach changed in 2017 to Milne Beach approx. 1km north of Matlock.

Historical and current data for fecal indicator bacteria gathered from Lake Winnipeg beaches provide strong evidence that microbiological conditions can change dramatically within a 24-hour period. Whitman et al. (2004) was also able to demonstrate that the time of sampling was a significant factor responsible for diurnal changes in *E. coli.* While other factors such as cloud cover, turbidity, etc. also influenced daily temporal changes in density, significant decreases in *E. coli* were observed from early morning to afternoon. Presumably, the decrease of *E. coli* densities in bathing water was due to environmental conditions, the most lethal of which is ultraviolet radiation (McCambridge and McMeekin 1981).

On Lake Winnipeg, intensive studies have shown that large numbers of *E. coli* are present in the wet sand of beaches and densities are significantly higher than the adjacent lake water (EC and MWS 2011, Williamson et al. 2004). This is consistent with other studies, which have found that

Table 14-1: Rate of compliance with the recreational water quality objective for E. coli (200 E.coli per 100 mL) at east and west side Lake Winnipeg beaches for the historic period of record (2004–2018) and the last five years (2014–2018).

Beach		Percent Compliance (%)				Number of Samples	
		Summary 2004-2018	Min 2004-2018	Max 2004-2018	5 Year Summary	Summary 2004-2018	5 Year Summary
East Side	Victoria	97%	88%	100%	97%	258	92
	Victoria Red Cross Dock	98%	90%	100%	99%	156	63
	Albert	97%	83%	100%	96%	266	94
	Hillside	100%	95%	100%	100%	257	89
	Lester	99%	82%	100%	97%	260	91
	East Grand	95%	88%	100%	93%	271	95
	West Grand	94%	68%	100%	89%	853	171
	Sunset	93%	71%	100%	93%	203	98
	Patricia	93%	80%	100%	92%	280	97
West Side	Grindstone	97%	85%	100%	95%	244	96
	Blacks Point	99%	94%	100%	98%	216	93
	Gull Harbour	93%	83%	100%	95%	277	97
	Sandy Bar	92%	70%	100%	91%	281	98
	Hnausa	95%	86%	100%	95%	262	93
	Spruce Sands	92%	82%	100%	90%	280	100
	Gimli	85%	67%	97%	78%	893	179
	Sandy Hook	92%	78%	100%	89%	281	100
	Winnipeg	87%	67%	100%	82%	305	109
	Matlock	91%	70%	100%	87%	277	104

wet beach sand can also support significantly higher densities of *E. coli* as compared to lake water (Edge et al. 2010, Ishii et al. 2007, Kon et al. 2007, Whitman and Nevers 2003). This is because of the optimal conditions that support bacterial growth in wet beach sand (e.g. light, moisture, organic content). Long-term studies on Lake Winnipeg indicate elevated densities of bacteria occur in lake water when strong north winds cause lake levels to rise and wave action washes *E. coli* out of the foreshore sand and into the bathing area (Williamson et al. 2004).

Intensive genetic ribotyping studies to determine the sources of *E. coli* at Lake Winnipeg beaches indicate that less than 10% of the *E. coli* is from human sources with 63% being contributed by animal sources (Williamson et al. 2004). Of the animal component, the largest identifiable source of *E. coli* was from shorebirds and geese. The importance of shorebirds as a source of contamination to bathing water has also been documented in other studies (Edge et al. 2010, Edge and Hill 2007, Fogarty et al. 2003, Whitman and Nevers 2004, Feare et al. 1999).

Overall, recreational water quality at Lake Winnipeg beaches is typically good, though it may be occasionally impaired due to elevated *E. coli* densities exceeding acceptable recreational objectives. Further research is required to develop predictive models that correlate changes in lake levels, wind direction and wind speed on observed *E. coli* densities in bathing water. Additionally, advancements in technology that allow shorter lag times between sample collection and receipt of sample results, or the development of real-time *E. coli* measurements could improve turn-around times associated with posting beach advisories.

15.0 CONTAMINANTS IN LAKE WINNIPEG

By: Jonathan Challis, Mark Hanson, Dana Moore, Sarah Warrack (University of Manitoba), Michael Rennie (Lakehead University) and Charles Wong (University of Winnipeg)

This section summarizes a select number of publications reporting the occurrence and dynamics of contaminants, including pesticides, pharmaceuticals, microplastics, brominated flame retardants and legacy organic contaminants in water, sediments, and biota of Lake Winnipeg. While other publications exist regarding emerging contaminants within this watershed (e.g. Carlson et al. 2013a, Anderson et al. 2013, Anderson et al. 2015) for sake of brevity and relevance to the State of Lake Winnipeg, this discussion focuses primarily on data relevant to Lake Winnipeg proper.

Contaminants to Lake Winnipeg come from a number of anthropogenic sources.

Pesticides

The largest collection of pesticide occurrence data comes from riverine systems within the Lake Winnipeg watershed as a result of monitoring efforts by the provincial and federal governments. However, there is a paucity of data and publications reporting the occurrence and fate of pesticides in Lake Winnipeg itself. Between 1995 and 2014, 26 of 39 pesticides detected in surface waters in the province of Manitoba had their greatest concentrations measured in three drainage regions (Lake Winnipeg West, the Red River, and the Assiniboine River), including one (Fenoxaprop – 1.61 µg/L) taken from Lake Winnipeg proper, three in streams (Brokenhead River and Icelandic River North) draining into the western side of Lake Winnipeg, seventeen in the Red River basin draining into Lake Winnipeg, and five in the Assiniboine River basin draining into the Red River. Of the pesticides detected in these basins, a total of eleven exceeded Provincial Water Quality Guidelines (WQG) for Protection of Aquatic Life (2,4-D, chlorothalonil, chlorpyrifos ethyl, deltamethrin, lindane, MCPA, pentachlorophenol), Irrigation Waters (bromacil, bromoxynil, dicamba, MCPA, simazine) and Livestock (MCPA). Of these eleven pesticides, six were observed to have relatively elevated detection rates (> 9.5%) (2,4-D, clothianidin, dicamba, glyphosate, imazamethabenz methyl, MCPA). No sites in Lake Winnipeg proper exceeded the WQG's for the Protection of Aquatic Life, but one site exceeded the guidelines for protection of irrigation water once each for bromoxynil and MCPA, and six times for dicamba. Overall, rates of pesticide detection are low in Lake Winnipeg proper (Table 15-1).

Table 15-1: Summary monitoring totals and rates of pesticide detection in the drainage basins monitored for pesticide residues in surface waters in Manitoba, 1995–2014.

Drainage Basin	# Sub-watersheds	# Sites	# Samples	# Analyses	# Detects	% Detect
Saskatchewan River	2	14	18	693	1	0.14
Lake Manitoba	8	30	860	37,689	300	0.8
Assiniboine River	6	41	755	34,436	759	2.2
Souris River	2	17	233	9,947	206	2.07
Red River	9	39	952	40,171	1,335	3.32
Winnipeg River	3	9	98	4,921	29	0.59
Lake Winnipeg East	5	12	40	1,679	0	0
Lake Winnipeg West	7	28	303	13,976	177	1.27
Grass River	5	21	37	1,395	3	0.22
Nelson River	1	3	6	222	0	0

Recently, Challis et al. (2018) used passive sampling to measure pesticide levels and sources in the Red River and Lake Winnipeg in 2014 and 2015. Concentrations of atrazine (ATZ) and the neonicotinoids thiamethoxam (THM), clothianidin (CLO) and imidacloprid (IMI) were measured in the south basin, narrows, and north basin of Lake Winnipeg, and at several locations in the Red River. A brief summary of the methodology is described in the Appendix. Results specific to Lake Winnipeg are reported here. Lake-wide concentrations of neonicotinoids and atrazine ranged from 0.05–6 ng/L and 13–96 ng/L, respectively (Figure 15-1). While concentrations appeared to be elevated in 2015 compared to 2014, obvious spatial and temporal trends were not observed. Based on a lake volume of 284 km^3 (Brunskill et al. 1980) mass loadings can be estimated from the observed pesticide concentrations. Average estimated lake-wide masses of THM, CLO, and ATZ were 116, 346, and 4600 kg respectively. Compared to 2014 and 2015, average mass loadings in the Red River at Selkirk of 72, 80, and 615 kg (Challis et al. 2018), burdens of these pesticides in the lake are approximately 2–6 fold greater. Greater mass burdens in the lake are expected due to other major inputs (e.g. inflow from Lake Manitoba, Saskatchewan River) and that riverine inputs are occurring on a much shorter timescale compared to turnover in Lake Winnipeg (average residence time of 4.3 years) (Brunskill et al. 1980). Regardless, the concentrations of these three pesticides observed in Lake Winnipeg pose little risk to aquatic life. Even for ATZ, where dilution does

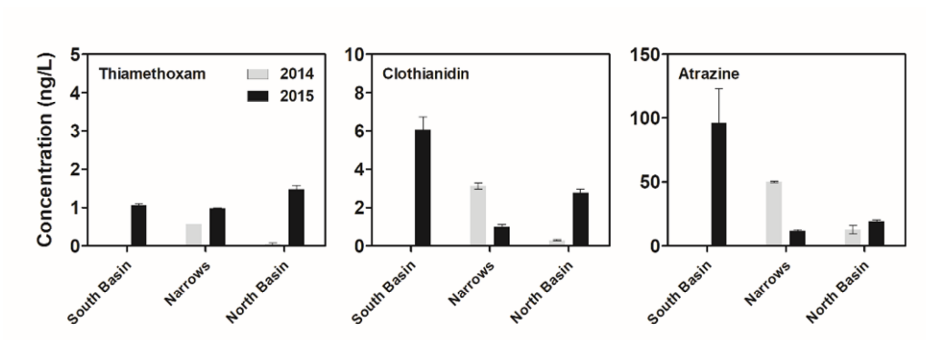


Figure 15-1: Concentrations of thiamethoxam, clothianidin and atrazine in Lake Winnipeg measured during the 2014 and 2015 sampling seasons. Error bars represent the standard deviation of the mean for triplicate measurements.

not appear to be significant when comparing average levels in the Red River at Selkirk (61 ng/L) and Lake Winnipeg (56 ng/L), concentrations are still ≈30-fold less than the CCME guideline for protection of aquatic life (1800 ng/L), ≈160-fold less than the CCME guidelines for the protection of irrigation water (10000 ng/L) and ≈80-fold less than the CCME guideline for the protection of livestock water (5000 ng/L).

For comparison purposes, average ATZ concentrations from 1991 to 1994 in the Laurentian Great Lakes were ≈4 ng/L in Superior, ≈36 ng/L in Michigan, ≈22 ng/L in Huron, 30–109 ng/L in Erie (measured in east, central and west of Lake), and ≈84 ng/L in Ontario (Schottler et al. 1997). More recently, Kurt-Karakus et al. (2010) reported atrazine concentrations in the five Great Lakes ranging from 5.5 to 61 ng/L, very similar to the levels observed in Lake Winnipeg. While neonicotinoid levels in the Great Lakes have yet to be characterized, a recent publication (Hladik et al. 2018) reports concentrations in a number of tributaries to the Great Lakes. Across ten major tributaries, the three most frequently detected neonicotinoids were imidacloprid, clothianidin and thiamethoxam with medians ranging from 7.0 to 39 ng/L and a maximum observed concentration for an individual neonicotinoid of 203 ng/L (Hladik et al. 2018). These are difficult to compare directly to Lake Winnipeg levels; however, these concentrations are similar to observations in the Red River.

Pharmaceuticals

Challis et al. (2018) also measured pharmaceutical concentrations in Lake Winnipeg and the Red River as an indicator of systems influenced by municipal inputs. Carbamazepine (CBZ) was the only pharmaceutical (out of 17 that were analyzed) that was detected consistently in Lake Winnipeg. CBZ is an anti-epileptic drug, commonly prescribed for treatment of anxiety and depression, and is ubiquitous in impacted waters given its use and relative persistence. Concentrations of CBZ in Lake Winnipeg did not differ significantly through time or space. The average lake-wide concentration was 1.3 ng/L, ranging from 0.44–2.4 ng/L. For comparison, at Lake Ontario shoreline sites (Toronto and Hamilton Harbour), which are impacted much more so by urban and industrial inputs compared to Lake Winnipeg, Li et al. (2010) detected 17 pharmaceuticals ranging from 0.13–35 ng/L. At an open water site in the western basin of Lake Ontario, a site more comparable to the Lake Winnipeg south basin data, fewer pharmaceuticals were detected and concentrations ranged from 0.04–8 ng/L. CBZ had a concentration of 1.4 ng/L, very comparable to Lake Winnipeg (Li et al. 2010).

There were six other pharmaceuticals of note (antibiotics clarithromycin, sulfapyridine, sulfamethoxazole, trimethoprim and β-blockers metoprolol, propranolol) detected in the study but they were only detected in the Red River, downstream of Winnipeg's North End Water Pollution Control Centre (Challis et al. 2018). The specific pharmaceuticals detected there were typical of effluent impacted surface waters. Concentrations ranged from 0.2–35 ng/L with notable one-time spikes in the levels of sulfapyridine (250 ng/L) and clarithromycin (170 ng/L) (Figure 15-2). Concentrations of CBZ in Lake Winnipeg and the Red River are well below the CCME long term water quality guideline of 10,000 ng/L. While toxicity thresholds for the other pharmaceuticals are not well developed, like CBZ, the concentrations observed in Lake Winnipeg and the Red River are unlikely to pose any significant risk to non-target organisms.

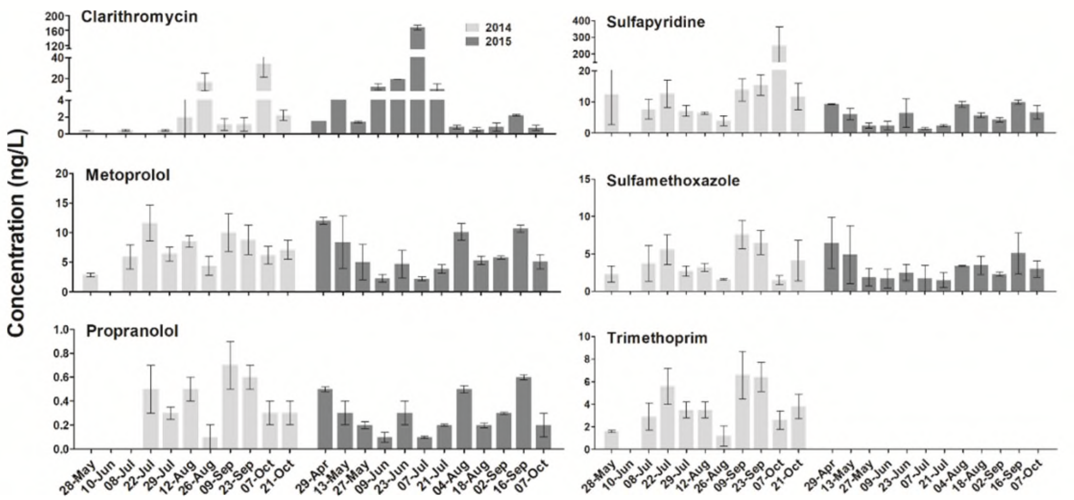


Figure 15-2: Concentrations of six pharmaceuticals at the North End Red River site. Left to right on each plot is spring to fall samples in 2014 (light gray bars) and 2015 (dark gray bars). Each bar represents the mean and standard deviation (error bars) of triplicate samples. Trimethoprim was not detected in any samples in 2015. Figure taken from Challis et al. (2018).

Microplastics

Microplastics (MPs, broadly defined as plastic particles < 5mm in size) are of growing concern given their ubiquitous nature and potential impact on aquatic organisms including zooplankton, zoobenthos and fish (Anderson et al. 2016). The only published study that reports densities of MPs in Lake Winnipeg (Anderson et al. 2017) illustrates that MPs are elevated relative to the Great Lakes (Figure 15-3). Compared to a survey of Great Lakes (Erikson et al. 2013), the average density of MPs in Lake Winnipeg was not significantly different from Lake Erie, but was significantly elevated relative to Lake Huron and eastern Lake Superior (Anderson et al. 2017). Lake Erie has a watershed roughly 1/10th of that of Lake Winnipeg, but supports a much more

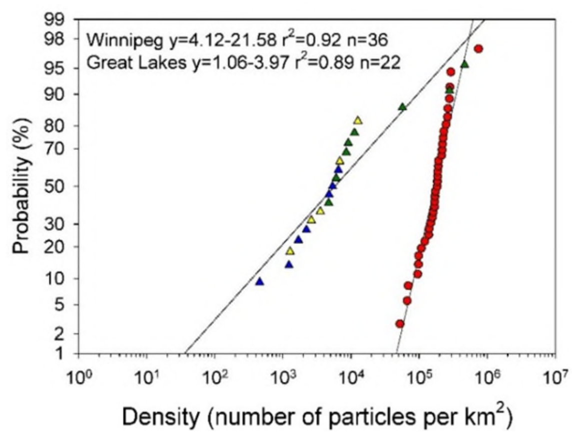


Figure 15-3: Exposure distributions for microplastics in Lake Winnipeg (red circles) and the Laurentian Great Lakes (triangles; blue = Lake Huron, green = Lake Erie, yellow = Lake Superior). Great Lakes data are from Eriksen et al. (2013), Lake Winnipeg data from Anderson et al. (2017).

concentrated population within that watershed (12 million around Lake Erie vs. nearly 7 million people in the Lake Winnipeg watershed), with a much greater concentration of industrial activity (Anderson et al. 2017).

Unlike the Great Lakes, where samples were dominated by fragments (42%) and pellets (48%) (Eriksen et al. 2013), the majority of particles identified in Lake Winnipeg were fibers (86% on average; Figure 15-4). Fibers may be generated from the breakdown of larger plastic particles (Eriksen et al. 2014). However, a major source of fibers has also been identified to come from synthetic fabrics used in athletic and outdoor clothing (Hartline et al. 2016, Mason et al. 2016).

The annual loading of MPs to Lake Winnipeg is estimated to be approximately 1.2 billion particles, 33% of which come from the Red River (Warrack 2017). A number of assumptions were made to generate this estimate. Given that no inter-annual differences in MP densities in the lake was found (Anderson et al. 2017), a lake-wide average density of 200,000 MPs per km^2 was used to reflect a “steady-state” estimate of MPs in Lake Winnipeg. Scaling to lake area, this represents a total particle count in Lake Winnipeg of 4.8 billion. Assuming a turnover rate of 4 years (EC and MWS 2011), one quarter (or 1.2 billion) of the total lake-wide quantity of MP is lost annually. Therefore, to maintain a stable density of 4.8 billion, the annual load is estimated to be equivalent to annual losses (1.2 billion).

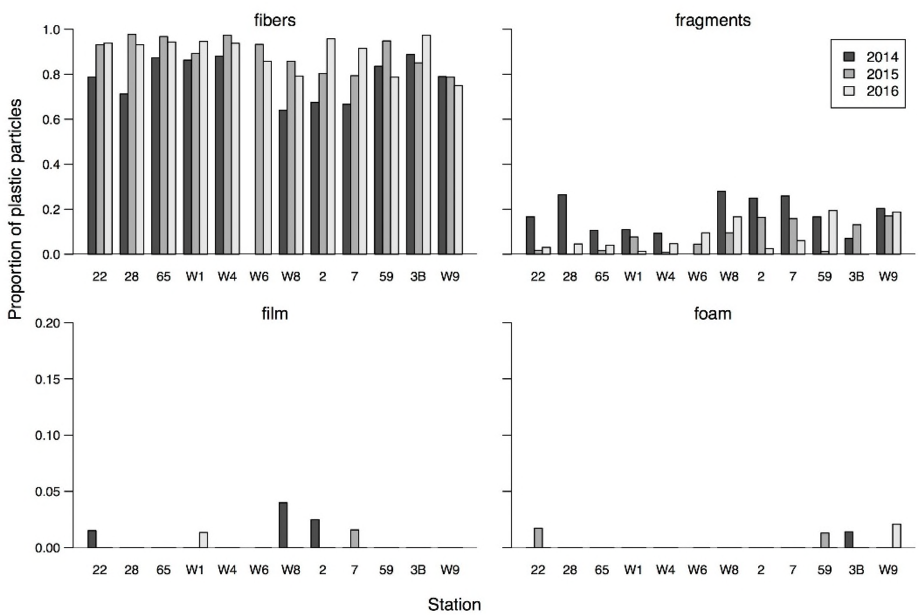


Figure 15-4: Proportion of the four types of microplastic particles (fibres, fragments, film and foam) found at the 12 stations across Lake Winnipeg , 2014–2016. Beads are absent from the figure as none were detected. Note difference in y-axis scaling for lower panels. North basin stations are 22, 28, 65, W1, W4, W6, W8; south basin stations are 2, 7, 59, 3B and W9. Reproduced from Anderson et al. (2017) with permission.

Brominated Flame Retardants

Brominated flame retardants are common industrial chemicals. Many are produced in large volumes and often found in quantifiable amounts in wildlife (Darnerud 2003). Tomy et al. (2007) measured isomers of the chlorinated flame retardant DP in archived Lake Winnipeg food web samples. The syn-DP isomer was detected in most samples, while the anti-isomer was only detected in 45% of samples. Concentrations of DP were generally greatest in Burbot (*Lota lota*), Walleye (*Sander vitreus*), Goldeye (*Hiodon alosoides*), mussels and zooplankton; however, levels of syn- and anti-isomers differed between biota. Concentrations of syn- and anti-DP isomers in Lake Winnipeg sediments were 11.7 and 18.3 pg/g (dry wt), respectively. In biota, concentrations ranged from 24 pg/g (anti-DP in Lake Whitefish (*Coregonus clupeaformis*)) to 760 pg/g (anti-DP in Goldeye). The anti-isomer was dominant in higher trophic level organisms like Walleye (730 pg/g lipid wt.) and Goldeye (760 pg/g), while the syn-isomer was greatest in lower trophic organisms like zooplankton (550 pg/g) and mussels (430 pg/g). Differences between the measured bio-magnification potentials for the two isomers suggest that interspecies differences can drive the distribution of these isomers in the environment. Overall, DP levels were significantly lower than other, more common flame retardant compounds (PBDE and HBCD) (Law et al. 2006).

The bioaccumulation and trophic transfer of brominated flame retardants was assessed in a Lake Winnipeg food web (Law et al. 2006). Brominated diphenyl ether (BDE) congeners, hexabromocyclododecane (HBCD) diastereoisomers (α, β, γ), decabromodiphenylethane (DBDPE), and bis(2,4,6-tribromophenoxy)ethane (BTBPE) were measured in south basin fish, mussels, zooplankton, sediment and water. Fifteen individual BDE congeners were detected consistently in the dissolved phase water samples, with concentrations ranging from 0.60 pg/L (BDE 85) to 17.0 pg/L (BDE 47). ΣPBDE concentrations (49 pg/L) represented the single most abundant class of flame retardants found in water samples, followed by α-HBCD (11 pg/L) and BTBPE (1.9 pg/L). In sediments, again ΣPBDE represented the largest concentration of flame retardant, ranging from 1160-1610 ng/g. Levels of BDE 209 accounted for approximately 50% of this ΣPBDE concentration. The isomer γ-HBCD was found at 50 pg/g in the south basin sediments. Ten BDE congeners, α, β, γ-HBCD and BTBPE were consistently detected in all species sampled. Mean ΣPBDE concentrations ranged from 240 ng/g (lipid wt) in Burbot to 11 ng/g in Lake Whitefish. The lipid-correlated biomagnification factors (BMFs) of individual predator/prey feeding relationships suggested that certain BDEs, HBCD and BTBPE biomagnify in Lake Winnipeg food webs. The greatest BMFs were observed within the walleye-to-whitefish relationship for these three flame retardants. In general, the levels of these flame retardants observed in water, sediment and biota were deemed comparable to other freshwater systems.

Legacy Organochlorine Persistent Organic Pollutants (POPs)

Persistent organic pollutants are chemicals of concern because they tend to bio-magnify and bio-accumulate in ecosystems, and have negative effects on human health and the environment. Rawn et al. (2000) used dated sediment cores from the north and south basin to assess historical deposition of organochlorine (OC) chemicals in Lake Winnipeg. Relatively elevated concentrations of PCBs were observed in sediments dated 1960–1970. PCB flux to sediment was estimated at 5–11 µg/m^2/y and burdens in the north and south basin were similar (388 and 337 µg/m^2); however, sources differed. Loadings to the north basin indicated mostly atmospheric sources of PCBs while south basin sediment cores were indicative of agricultural, industrial and urban inputs. Inventories of DDT were 37 and 127 µg/m^2 in the north and south basin cores, respectively. Hexachlorocyclohexane (HCH) was observed in the more recent (post-1980) core slices with fluxes ranging from 177 to 914 ng/m^2/y. Total chlordane (ΣCHL), pentachloroanisole (PCA), and total chlorobenzenes (ΣCBz) were present in the surface sediment core slices in both basins. The legacy OC insecticides in the Lake Winnipeg sediment cores revealed a steep increase beginning in 1940s (corresponding to use in North America) reaching a maximum level in the late 1960s. While deposition into the sediments represents a major sink for these POPs in Lake Winnipeg, periods of flooding, high flow and erosion could impact the flux measurements reported here.

Stewart et al. (2003) investigated the short-term effects of the 1997 Red River flood on OC levels in water, sediments and biota from Lake Winnipeg. OC concentrations in sediments of the south basin did not change markedly following the 1997 Red River flood; however, concentrations in pre-flood sediment cores (Rawn et al. 2000) appear to fall at the low range of those observed post-flood (Table 15-2). Water concentrations of ΣPCB post-flood ranged from 0.05 to 0.83 ng/L, which were significantly elevated compared to concentrations observed in samples taken years later in 2002 (3.4–24 pg/L) (Gewurtz et al. 2006). Significant increases in ΣPCB and ΣDDT levels in years post-flood (1999) were observed in the top predators (Walleye and Burbot). Even over a two-month period in the summer following the flood, significant increases in OC concentrations were measured in zooplankton and Yellow Perch (*Perca flavescens*) (> 2-fold in ΣPCB, ΣDDT, ΣCHL, ΣCBz) and in Walleye (1.4 fold ΣPCB). ΣDDT levels in Walleye (14 ng/g) and Burbot liver (average 412 ng/g) represented the single Lake Winnipeg exceedance of a CCME guideline (14 ng/g wet wt., CCME 1999c). Changes in specific congener patterns over time suggested that changes in OC levels in fish pre- and post-flood were likely due to shifts at the base of the food web in plankton communities, though this also reflects a period where Rainbow Smelt (*Osmerus mordax*) were becoming established in the lake as well.

Table 15-2: Sediment core concentrations of organochlorine compounds pre- (Rawn et al. 2000) and post-1997 flood (Stewart et al. 2003) in the south basin of Lake Winnipeg.

South basin sediment samples (ng/g dry wt)	ΣPCB	ΣDDT	ΣHCH	ΣCHL	ΣCBz
Pre-1997* (Rawn et al. 2000)	≈0.5–20	≈0.5–4.5	≈0.05–0.35	≈0.05– 0.45	≈0.01– 0.70
Post-1997** (Stewart et al. 2003)	5.5–38	1.3–6.6	<0.22–0.85	<0.01–1.19	<0.15–1.39

**Cores taken in 1995, dated 1932-1991. Approximate values for this report obtained by reading off of graph. **Cores taken in 1998.*

Gewurtz et al. (2006) examined bioaccumulation dynamics of 103 PCB congeners from 2000 to 2002 in the Lake Winnipeg food web. Concentrations of PCBs were measured in the water ([ΣPCB] in south basin = 24 pg/L; north basin = 3.4 pg/L), sediments ([ΣPCB] in south basin = 1.8 ng/g; north basin = 1.0 ng/g), and biota (adult Walleye, Sauger (*Sander canadensis*), Lake Whitefish, Cisco, emergent mayflies and plankton). Water and sediment concentrations were lower than observed by Rawn et al. (2000) and Stewart et al. (2003) (Table 15-2).The extent of biomagnification of PCBs per unit trophic level did not differ significantly between the north and the south basins; however, ΣPCB concentrations were significantly greater in south compared to north basin biota. Biomagnification was observed in the south basin from the invertebrates and herbivorous whitefish to the top predators, but was much less evident in the north basin, a result attributed to the higher proportion of lower chlorinated congeners and the presence of Rainbow Smelt in the north basin. These findings suggest that differences in PCB concentrations and congener patterns between basins are largely due to differences in sources of loadings (riverine inputs to the south basin and atmospheric deposition to the north basin) rather than food web processes. These results are similar to the observations of Rawn et al. (2000). Finally, this study concluded that the high nutrient-associated DOC in Lake Winnipeg water likely decreases PCB bioavailability to lower trophic level organisms, and thus the entire food web.

It is possible to provide some context around the levels of legacy contaminants in Lake Winnipeg through selected comparisons with other large lakes that are geographically close and subject to both primarily atmospheric background deposition (e.g. Lake Superior) and significant anthropogenic inputs (e.g. Lake Ontario). Concentrations of ΣPCBs in sediments from Lake Ontario and Erie ranged from 2.6 to 255 ng/g (mean = 100 ng/g) and from 1.9 to 245 ng/g (mean = 98 ng/g), respectively (Marvin et al. 2004). These concentrations are elevated compared to Lake Winnipeg sediments, which generally had single digit ng/g ΣPCB concentrations. Canadian Sediment Quality Guidelines for total PCBs is 277 ng/g (CCME 1999c). Total PCB water concentrations ranged from 100 pg/L in Lake Superior to 1.6 ng/L in the western basin of Lake Erie (Anderson et al. 1999). In Lake Winnipeg, water concentrations appear to fall in between Lake Superior and Erie levels, ranging from 3.4 (Gewurtz et al. 2006) to 830 pg/L (Stewart et al. 2003).

McGoldrick et al. (2006) measured many of these same contaminants in fish from the Great Lakes. The greatest average ΣDDT fish concentration (wet wt.) was observed in Lake Ontario (280 ng/g) followed by Michigan (220 ng/g), Huron (130 ng/g), Superior (110 ng/g) and Erie (68 ng/g), all exceeding the tissue residue guideline for the protection of wildlife consumers of aquatic biota of 14 ng/g (CCME 1999c). Similarly, high levels of ΣPCBs are observed in fish from the Great Lakes, with highest concentrations observed in Lake Michigan (935 ng/g) followed by Ontario (692 ng/g), Huron (653 ng/g), Erie (625 ng/g) and Superior (372 ng/g). Levels of these same contaminants are comparatively low in Lake Winnipeg. ΣDDT fish concentrations (wet wt.) between 1995 and 2000 ranged from 5.5–14 ng/g in Walleye to 2.8–5.1 ng/g in Sauger to 2.5–6 ng/g in Yellow Perch (Stewart et al. 2003). ΣPCB fish concentrations (wet wt.) from the same study ranged from 8.6–26 ng/g in Walleye to 4.9–10 ng/g in Sauger to 6–13 ng/g in Yellow Perch (Stewart et al. 2003). The average concentration of the sum of tetra-, penta-, and hexa-BDEs were highest in Lake Ontario (85 ng/g) followed by Superior (63 ng/g), Michigan (55 ng/g), Huron

(43 ng/g) and Lake Erie (18 ng/g). Levels compared to Lake Winnipeg were elevated in Burbot (ΣPBDE = 240 ng/g lipid wt), but similar in other fish, from 11 ng/g in Lake Whitefish to 54 ng/g in Walleye (Law et al. 2006). Taken together, levels of legacy POPs in Lake Winnipeg water, sediment and biota appear to be similar or lower than those observed in the Great Lakes.

Summary

In general, concentrations of present day and legacy contaminants do not appear to pose an acute or chronic risk to the Lake Winnipeg aquatic ecosystem; however, significant gaps in our understanding of some of the current impacts and future risks still exist. For example, while detection frequencies of pesticides in the lake proper are generally very low, there are a number of pesticides that warrant continued monitoring (based on prevalence in the watershed), in order to better understand long-term exposure scenarios in the lake (e.g. atrazine, 2,4-D, dicamba, glyphosate, and MCPA). Furthermore, uncertainty remains regarding the potential sources and transport mechanisms of microplastics to the lake, the effects of environmentally-relevant concentrations of microplastics on different biota, and the extent of macroplastic contamination in the lake. Finally, with respect to legacy POPs and other organics (e.g. PCBs), although the concentrations of these contaminants in the various compartments are below thresholds of concern (where standards have been developed), and while these contaminants do not represent a pressing concern to the Lake Winnipeg ecosystem, it is approaching 20 years since many of these legacy POP measurements were conducted. It would be interesting to conduct a current survey of water, sediment and biota in Lake Winnipeg to understand long-term temporal trends for these contaminants.

16.0 CONCLUDING REMARKS

The growing concerns over the presence of large and sustained algal blooms in Lake Winnipeg in the 1990s prompted an increased effort on understanding and monitoring the state of the lake's aquatic ecosystem. Both the Government of Canada and the Manitoba government, as parties to the Memorandum of Understanding Respecting Lake Winnipeg and the Lake Winnipeg Basin (MOU), committed to a collaborative approach to understand and protect the water quality and ecological health of Lake Winnipeg and its basin (EC and MWS 2010). In 2011, the first State of Lake Winnipeg report was published (EC and MWS 2011), which documented the physical, chemical and biological characteristics of Lake Winnipeg for the 1999–2007 period. Continued and ongoing support for monitoring and research expanded the amount of information available for the lake, allowing the production of this updated report.

Although efforts over the last 10-year period have increased the amount of information and knowledge regarding the aquatic ecosystem of Lake Winnipeg, many significant gaps in knowledge still exist. The first State of Lake Winnipeg report provided an overview of knowledge gaps specific to the information in that report (EC and MWS 2011):

Hydrology – the lake water balance remained incomplete; specifically, estimates of evaporation, groundwater inflow and contributions from remote ungauged rivers flowing into the east side of the lake.

Physical and chemical characteristics of Water – information on the length of the open-water season, under-ice conditions, littoral areas and the degree of spatial (horizontally and with depth) and temporal variability of temperature and chemistry.

Nutrients – atmospheric loading of nutrients (N and P), rates of nitrogen fixation by cyanobacteria, losses of nitrogen through denitrification, and internal loading of nutrients (through diffusion and resuspension) remained unquantified.

Biology – trophic structure, food web linkages, top-down and bottom-up interactions were not well understood. Examples include lack of knowledge around factors controlling phytoplankton biomass and species composition, the drivers for increased Walleye production and the impacts of non-native rainbow smelt on the food web.

Algal toxins – the factors driving toxicity of blooms, knowledge on which phytoplankton species were producing microcystin, and an evaluation of the potential for transfer of microcystin to higher trophic levels were unknown.

With respect to the gaps identified in the first State of Lake Winnipeg report, some have been reported on in this update (*e.g.* fish populations, temperature and oxygen profiles, and spatial variability in chemical constituents), others are currently being studied but have not yet been reported on (*e.g.* littoral zones, internal nutrient loading/sediment resuspension), but most others remain. In fact, new questions regarding the impacts of Spiny Water Flea and Zebra Mussels on the food web have arisen with the introduction of these invasive species. Although eutrophication remains the prime concern regarding the health of the lake, recent work has also identified other areas where either information or knowledge is lacking, specifically:

Microplastics - uncertainty remains regarding the potential sources and transport mechanisms of microplastics to the lake, the effects of environmentally relevant concentrations of microplastics on different biota, and the extent of macroplastic contamination in the lake.

Pesticides - while detection frequencies of pesticides in the lake proper are generally very low, there are a number of pesticides that warrant ongoing reporting to better understand long-term exposure scenarios in the lake (e.g. atrazine, 2,4-D, dicamba, glyphosate and MCPA).

Furthermore, both the first report, and this updated report, have not included traditional knowledge/indigenous science into the assessment of the state of the lake. Indigenous peoples living around Lake Winnipeg and throughout the basin have been gathering, sharing, and passing down traditional ecological knowledge for generations. These residents have used their expertise derived from experience to monitor the lake and sustainably manage its subsistence, cultural and spiritual resources. This knowledge represents important information and understanding of the state of the lake and the lack of inclusion represents a gap in our comprehensive knowledge of the lake.

Lake Winnipeg is a complex system and although knowledge of the aquatic system has improved over the past two decades, management of the lake remains a challenging task. An adaptive management approach lends itself well to Lake Winnipeg where the effects of climate change and invasive species are unknown and uncertainty remains about the role of some processes in influencing nutrient concentrations and subsequent impacts on the food web. With multijurisdictional partnerships and stakeholder interest already well established, a key next step for achieving the goal of a healthy aquatic ecosystem in Lake Winnipeg might include consideration of an adaptive management approach.

The MOU has provided the framework for, and helped guide the coordinated science, programs and activities on Lake Winnipeg. As part of the MOU, Canada and Manitoba agreed to adopt appropriate indicators for assessing the status and trends in ecosystem health of Lake Winnipeg. The first in the series focused on fish populations and was published in 2019. Additional indicators, based on some of the information included in this report, are currently under development. The current MOU expires in 2020, efforts are underway to explore options to help ensure that research, and the implementation of solutions for achieving a healthy Lake Winnipeg continues collaboratively with all groups.

APPENDIX: METHODS AND DATA SOURCES

Section 2.0: Climate

Baseline Climate Normals (1981–2010)

Climate normals for Arborg, Grand Rapids, Norway House and Berens River were obtained from Environment and Climate Change Canada's climate normals website:

http://climate.weather.gc.ca/climate_normals/index_e.html

These data summarize climatological statistics over the period 1981–2010. Importantly, climate normals are only produced for weather stations that have at least 15 years of data within this 30-year time period. At the turn of each decade, Environment Canada updates these climate normals.

Annual Temperature Trend (1960-2016)

Long-term annual temperature trends were computed using Environment and Climate Change Canada's Adjusted Historical Canadian Climate Data (AHCCD), available online at:

https://www.canada.ca/en/environment-climate-change/services/climate-change/modelling-projections- analysis/adjusted-homogenized-canadian-data.html

These data pass through a rigorous checking process and are considered the best data available for analysis of long-term trends. Missing data is unfortunately quite common, especially for the stations adjacent to Lake Winnipeg.

Projected Climate Changes (2021–2080)

Future projections of temperature and precipitation under the RCP4.5 (low carbon) and RCP8.5 (high carbon) forcing scenarios were made using downscaled climate model data supplied by the Pacific Climate Impacts Consortium available at:

http://pacificclimate.org

This extraordinary dataset includes bias corrected, spatially downscaled daily maximum and minimum temperatures and daily total precipitation values, at 10 km by 10 km resolution across all of Canada, from 12 latest-generation CMIP5 GCMs (selected to be representative of the whole CMIP5 ensemble), for the period 1950–2095 (although some models had data available until the year 2100).

A computer program was written using IDL to extract data points completely contained by a Lake Winnipeg shapefile. These extracted values were then used to compute various temperature and precipitation-based climate change statistics that were subsequently summarized over two scenarios and three time periods: 1976–2005 (referred to in this report as the modelled baseline period), 2021–2050 and 2051–2080. In this report, we present the 10th and 90th percentile projections from the 12-model ensemble.

It is important to note the inherent limitations of these projected datasets. Visit PCIC's website to learn more about how these data were constructed and how these data should best be used. We acknowledge that these data are not ideally suited for computing frost-free season lengths.

Section 3.0: Hydrology

Only those hydrometric gauges with a sufficient period of record (from 1977 to present; 40 years of data) were used to calculate the tributary discharges into Lake Winnipeg. All streamflow and lake water level data were obtained from the Water Survey of Canada and Manitoba Hydro:

https://www.canada.ca/en/environment-climate-change/services/water-overview/quantity/monitoring/survey.html

Inflows

The Saskatchewan River streamflow was calculated using the Grand Rapids gauge (05KJ001). The Red River and Assiniboine River contributions are reported separately. For data availability reasons, the Red River at Ste. Agathe streamflow gauge (05OC012) was the primary gauge used for the analysis. Additional local inflows to the Red River downstream of Ste. Agathe included: Netley Creek, Devils Creek, Grassmere Creek, Marsh River, Seine River, Rat River, and Tourond Creek. The Assiniboine River at Headingley gauge (05MJ001) was the main gauge used for calculating the Assiniboine streamflow; additional local contributions from Sturgeon Creek (05MJ004) were also included. The Dauphin River at Dauphin River gauge (05LM006) was used to analyze Dauphin River contributions to Lake Winnipeg. A portion of Assiniboine River streamflow is diverted northward into Lake Manitoba through the Portage Diversion, located three kilometres upstream of Portage la Prairie, Manitoba, and is captured in the Dauphin River contributions to Lake Winnipeg. The Slave Falls Generating Station gauge on the Winnipeg River was the main gauge used to calculate the contribution from the Winnipeg River (05PF063); the Whitemouth River contributions were also included (05PH003).

Lake Winnipeg Inflow Gauges:

Gauge ID	Name	Drainage Area (km^2)	Latitude	Longitude
Saskatchewan River Subwatershed				
05KL001	Saskatchewan River at Grand Rapids	406,000	53.16389	-99.3489
Red River Subwatershed				
05OC012	Red River at Ste. Agathe	115,000	49.56667	-97.17667
05OJ008	Netley Creek near Petersfield	641	50.32722	-97.04417
05OJ016	Devils Creek Near Libau	240	50.19505	-96.71383
05OH007	Seine River near Ste. Anne	580	49.64372	-96.609
05OE011	Seine River Diversion Near Ile Des Chenes	1,420	49.69494	-96.98764
05OJ017	Grassmere Creek Drain near Middlechurch	462	49.99439	-97.10136
05OE010	Marsh River near Otterburne	403	49.50239	-97.15594
05OE001	Rat River near Otterburne	1,420	49.46198	-97.009
05OE009	Tourond Creek near Tourond	210	49.53203	-96.98478
Assiniboine River Subwatershed				
05MJ001	Assiniboine River near Headingley	162,000	49.86828	-97.40528
05MJ004	Sturgeon Creek at St. James	556	49.88175	-97.27983
Dauphin River Subwatershed				
05LM006	Dauphin River near Dauphin River	82,300	52.00195	-98.32978
Winnipeg River Subwatershed				
05PF063	Winnipeg River at Slave Falls	126,000	50.225	-95.57083
05PH003	Whitemouth River near Whitemouth	3,750	49.93878	-96.95683

Tributaries used to determine the gauged local inflow to Lake Winnipeg include the Bloodvein River, Manigotagan River, Icelandic River, Fisher River, and Brokenhead River. It should be noted that there are missing data between 1997 and 2010 at the Manigotagan gauge, and only spring data are available at the Icelandic and Fisher River gauges between 2000 and 2010.

Lake Winnipeg Local Inflows

Gauge ID	Name	Drainage Area (km^2)	Latitude	Longitude
05SD003	Fisher River Near Dallas	1,710	51.35617	-97.51222
05RB003	Bloodvein River above Bloodvein Bay	9,090	51.70342	-96.60453
05RA001	Manigotagan River Near Manigotagan	1,830	51.10128	-96.28303
05SC002	Icelandic River Near Riverton	1,240	50.96472	-97.03747
05SA002	Brokenhead River at Beausejour	1,580	50.09014	-96.42833

Outlfow

The Lake Winnipeg outflow is the summation of streamflow measured at the Nelson River West Channel at Jenpeg Generating Station (05UB009) and the Nelson River East Channel below Sea River Falls (05UB008).

Lake Winnipeg Outflow Gauges

Gauge ID	Name	Drainage Area (km^2)	Latitude	Longitude
05UB009	Nelson River West Channel at Jenpeg	N/A	54.49806	-98.04806
05UB008	Nelson River East Channel at Jenpeg	N/A	54.24417	-97.59083

Water Levels

Mean water level was determined from the average of readings at these eight gauges:

Lake Winnipeg Water Level Gauges

Gauge ID	Name	Drainage Area	Latitude	Longitude
05RF001	Lake Winnipeg at Montreal Point	1,020,000	53.625	-97.84444
05SG001	Lake Winnipeg at Mission Point	1,020,000	53.19112	-99.21198
05RE003	Lake Winnipeg at George Island	1,020,000	52.81842	-97.62956
05RD005	Lake Winnipeg at Berens River	1,020,000	52.35331	-97.02217
05SD002	Lake Winnipeg at Matheson Island Landing	1,020,000	51.72394	-96.91544
05SD001	Lake Winnipeg at Pine Dock	1,020,000	51.63969	-96.80331
5SB006	Lake Winnipeg at Gimli	1,020,000	50.63058	-96.98202
05SA003	Lake Winnipeg at Victoria Beach	1,020,000	50.69511	-96.56217

Section 4.0: Physical Characteristics

Since about 1999, Manitoba Agriculture and Resource Development (MARD, formerly Manitoba Water Stewardship, and Manitoba Sustainable Development) has reported water temperature at the surface or in the euphotic zone, and near the bottom in spring, mid-summer, fall at roughly 60 stations each year, and in late winter at about 14 stations. Prior to 2007, water column temperature profiles were recorded (also conductivity and turbidity) at the same stations by Fisheries and Oceans Canada, using an RBR water column profiling instrument (RBR TD-410). Since then, Environment and Climate Change Canada (ECCC) has used a Seabird instrument (SBE 19+) to record vertical thermal profiles at every station (also conductivity, light, dissolved oxygen, turbidity, light and chlorophyll fluorescence). Throughout both periods (i.e. 1999 to the present) MARD has measured temperature, dissolved oxygen, total suspended solids and a suite of other water quality parameters in discrete water samples collected at the same stations. To ensure that statistics for temperature and oxygen were calculated from identical sets, data were filtered to include only those station-dates where both parameters were reported for both the euphotic zone and bottom. In 2013, the Lake Winnipeg Research Consortium, in partnership with ECCC and MARD began a nearshore monitoring program that has supplemented these records with similar measurements along transects at in the littoral zone (currently five transects in the north basin, and seven in the south basin and narrows).

Section 5.0: General Chemistry and Trace Elements

General chemistry and trace metals were collected and analyzed for surface (represented as the top 20 cm) and/or euphotic (represented as the depth to which 1% of surface irradiance penetrates into the water column), and bottom samples (represented as the lower layer near the sediment water interface and/or below which stratification occurs).

One of the challenges with analyzing such large water quality datasets is understanding how to deal with censored data (i.e. observations below detection limit [BDL] and multiple detection limits). The presence of censored data can lead to unwanted biases during statistical analyses (e.g. calculation of mean and standard deviation). Therefore, the following approaches were employed to address the issues of observations BDL: Trace elements with less than 15% of observations BDL (i.e. dissolved and total aluminum [Al], arsenic [As], barium [Ba], boron, (B], copper [Cu], iron [Fe], lithium [Li], manganese [Mn], molybdenum [Mo], rubidium [Rb], strontium [Sr], titanium [Ti], uranium [U] and vanadium [V]) were replaced by a value of one-half the detection limit. This is a simple substitution method that has been widely accepted and used for calculating summary statistics whereby a high percentage of the observations were detects. For trace elements with observations between 15% and 80% BDL (i.e. cadmium [Cd], chromium [Cr], cobalt [Co], lead [Pb], nickel [Ni], antimony [Sb], tin [Sn], thallium [Tl], zinc [Zn], and zirconium [Zr], maximum likelihood estimation (MLE) was used to compute summary statistics following Hensel (1990) and Zhang (2007). Trace elements with observations greater than 80% BDL (i.e. beryllium [Be], bismuth [Bi], cesium [Cs], dissolved Cr[VI], selenium [Se], silver [Ag], tellurium [Te], thallium [Tl], and tungsten [W]) were excluded from statistical analyses and are not discussed further.

Trace element data from the fourteen long-term stations in Lake Winnipeg were used for the spatial and temporal comparison between the north basin and south basin and narrows. Trace elements with observations greater than 15% BDL were excluded from this analysis. Trace elements with less than 15% of observations below detection limit were replaced by a value of one-half the detection limit. General chemistry data from all stations in Lake Winnipeg were used for the spatial and temporal comparison between the north basin, south basin and narrows. A significant difference was defined as ± 10% from the yearly average.

Sections 6.0 Lake Nutrient Concentrations and 7.0 Nutrient Loading

All total phosphorus concentrations measured by Manitoba Agriculture and Resource Development from April 2001 through March 2009 in Lake Winnipeg and its tributaries have been adjusted to account for a change in laboratory analysis technique. Low range total phosphorus concentrations (<0.2 mg/L) measured during this period were approximately 12 % higher than those observed with the laboratory technique used in 2000 and in 2009 through the present (McCullough 2015b). A high range correction factor could not be derived. The 12 % correction factor was also applied across the total phosphorus data over the same period (April 2001 to March 2009) for data >0.2 mg/L because comparative data were insufficient to derive a correction factor. Particulate phosphorus was recalculated using the adjusted total phosphorus concentrations for the April 2001 through March 2009 period.

All nutrient loads calculated per the 2011 State of Lake Winnipeg report (EC and MWS 2011) with water quality data from Manitoba Agriculture and Resource Development and flow data from Water Survey of Canada.

Section 10.0 Zooplankton/Zoobenthos

Zooplankton

Field collections were done at predetermined stations in Lake Winnipeg located in all regions of the lake; all stations were not sampled in every survey. In 1969, six field surveys were conducted aboard M.V. *Bradbury* at approximately biweekly intervals throughout the open water season. In all other years, field collections were done three times annually during the open water season, representing spring, summer, and fall periods using the M.V. *Namao*, a scientific vessel owned and operated by the Lake Winnipeg Research Consortium. Collection of water samples during the 1969 cruises was described in Patalas and Salki (1992) with detailed station data provided in Salki and Patalas (1992). Some post-1969 Lake Winnipeg zooplankton samples were collected opportunistically through efforts organized by other agencies: the Geological Survey of Canada in 1994, the International Joint Commission in 1998 (south basin only) and the Lake Winnipeg Research Consortium (1999–2006) (see Hann and Salki 2017). Zooplankton samples in all years of the study were collected using consistent methods, i.e. an integrated zooplankton haul, from 1 m above the bottom sediments to the surface with the use of a single net, 1 m long, with 73 µm mesh and net opening of 0.049 m^2. Samples were made up to a standard volume of 125 mL and preserved with 10% formalin.

Zoobenthos

The zoobenthic samples collected from L. Winnipeg for Dr. B.J. Hann, Department of Biological Sciences, University of Manitoba, Winnipeg, Manitoba by the Lake Winnipeg Research Consortium (LWRC) and were hand-picked from the rinsed sediment using a dissecting microscope at low magnification (10X). Specimens were identified primarily to the family level using higher magnifications (25X, 40X) and published reference keys (Brinkhurst and Jamieson 1971; Merritt and Cummins 1996; Pennak 1953, 1989; Thorp and Covich 2010), enumerated, and preserved in 70% ethanol. Picked invertebrates are archived in Department of Biological Sciences, University of Manitoba. All data were summarized into groups: Oligochaeta (Tubificinae, Lumbriculidae), Mollusca (Sphaeriidae), Amphipoda (Pontoporeiidae), Ephemeroptera (Ephemeridae), Trichoptera (Leptoceridae, Molannidae), and Diptera (Chironomidae), and Nematoda.

Section 11.0: Fish

Collection and processing of trawl samples were funded through grants from the Fisheries and Oceans Canada Species at Risk Program and other Canadian Government funds, the Fisheries Enhancement Fund, and the Fish and Wildlife Enhancement Fund, in partnership with the Lake Winnipeg Research Consortium.

Mercury in Fish Tissue

Historic data on fish mercury concentrations were obtained from the Manitoba Fish Mercury Database (MFMD), which was originally assembled in 2005 by combining multiple subsections of the "Canadian Food Inspections Agency Survey Database" and the "Freshwater Institute National Contaminants Information System" that were stored at the Fisheries and Oceans Canada Freshwater Institute. The MFMD only contains data for individual fish. In the 1970s, these were obtained from either survey sampling or commercial individual sampling (*see* MH and the Province of Manitoba 2015). Data for 1971–1992 are assigned to "Area 1" of Lake Winnipeg in the database. The associated geographic coordinates place this location approximately 9 km south and offshore from the closest location included in the cluster of sampling sites representing Mossy Bay for CAMP fish collections since 2010 (see below). Two other small sets of data have been attributed to Mossy Bay but lack geographic coordinates. The older set is for three Northern Pike, 18 Walleye, and 20 Sauger from commercial individual samples (Derksen 1978a) and are not included in the MFMD. These data are not listed in either Derksen (1978b) or Derksen (1979), and thus were likely collected sometime between September 1970 and December 1972. The second set consists of survey samples of 13 Lake Whitefish and four Walleye for 1994. Recent (since 2010) data were obtained under CAMP from near shore areas (< 3.5 km offshore) within Mossy Bay. All of these data are referred to as from Mossy Bay in this analysis.

Much of the historic data included in the MFMD come from relative small size spectra of relatively large fish because muscle samples were mainly taken from individuals selected from commercial catches (i.e. commercial individual samples). In contrast, starting with the survey samples in the early 1970s and continuing until the present, fish collections have aimed to capture a broad spectrum of size classes around an average fork length of 350 mm for Lake Whitefish, 400 mm for Walleye, and 550 mm for Northern Pike. These lengths are referred to as the species' standard length and recognize methods applied by fish mercury monitoring programs for Northern Manitoba from the 1970s to the 1990s (Jansen and Strange 2007).

Comparisons of mean mercury concentrations must account for differences in fish sizes between years and lakes because fish accumulate mercury over their life time such that older, larger individuals have higher concentrations than younger, smaller fish (Green 1986; Evans et al. 2005). To reduce the effect of fish size on mean mercury concentrations, concentrations were standardized for fish length, i.e. adjusted to the standard length of a species, using unique regression equations generated from the relationship between logarithmic transformations of the muscle mercury concentrations (µg/g or parts per million [ppm]) and fork lengths (mm) of each individual. These standardized mean mercury concentrations are referred to as standard means. To present data in more familiar units, all standardized means and their measures of variance have been retransformed to arithmetic values. If the relationship between fish length and mercury concentration was not significant (usually associated with small sample size) and length standardization was not meaningful, arithmetic means were used for statistical comparisons.

Section 15.0: Contaminants in Lake Winnipeg

Pesticides and Pharmaceuticals in Lake Winnipeg

Polar organic chemical integrative samplers (POCIS) were deployed from three weather buoys located in the south basin, narrows, and north basin, as part of the lake-wide surveys conducted on the M.V. *Namao*. Samplers were deployed in 2014 and 2015 during the spring or summer cruises for 30-60 days. In 2014, only data from the narrows and north basin is presented as the POCIS were lost from the south basin buoy. Pharmaceuticals and pesticides were extracted from the samplers according to Carlson et al. (2013b). A total of 23 contaminants (6 pesticides and 17 pharmaceuticals) were analyzed by liquid chromatography tandem mass spectrometry (LC-MS/MS). A full description of the analytical methodology is described in Challis et al. (2018).

Microplastics

Microplastics were collected in Lake Winnipeg using a 3 m, 333 µm mesh manta trawl with a 61 cm wide by 18 cm high opening, as described in Anderson et al. (2017). Lake-based samples were taken from 12 stations in Lake Winnipeg once a year from 2014 to 2016. All samples were processed using wet-peroxide oxidation and particles identified visually.

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