

STATE OF LAKE WINNIPEG:

1999 to 2007



Canada

Manitoba

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

STATE OF LAKE WINNIPEG: 1999 TO 2007

Environment Canada
Manitoba Water Stewardship

June 2011

EXECUTIVE SUMMARY

The quality of Lake Winnipeg waters has deteriorated over time, with particular concern arising over the last few decades in response to the effects of accelerated nutrient enrichment. The frequency and intensity of algal blooms in the lake have increased in association with rising phosphorous and nitrogen loading from diffuse and point sources in the Lake Winnipeg watershed. The Province of Manitoba and the Government of Canada have recognized the need to address nutrient enrichment in Lake Winnipeg and progress has been made under the provincial Lake Winnipeg Action Plan and the federal Lake Winnipeg Basin Initiative to address these issues. While much information has been collected on Lake Winnipeg since 1999, before this report there has been no systematic attempt to summarize these data to form a greater understanding of changes seen more recently in the lake and to establish a baseline for the future.

The goal of this report is to present the recent state of knowledge on the Lake Winnipeg aquatic ecosystem as it pertains to eutrophication. This synthesis of research and monitoring on Lake Winnipeg in recent years is a collaborative effort amongst numerous federal and provincial agencies, universities, and independent researchers. The objectives of this report are to describe the temporal and spatial variation in Lake Winnipeg's physical, chemical, and biological characteristics, based on data collected primarily between 1999 to 2007, to explore current and emerging issues of concern to the health and integrity of Lake Winnipeg, and to highlight recent research on the lake. This recent baseline information is intended for use in the future development of performance indicators and nutrient objectives for Lake Winnipeg and its watershed.

Lake Winnipeg and its Watershed

Lake Winnipeg is a large, shallow lake covering four degrees of latitude and 23,750 square kilometres (km^2) in the province of Manitoba. The 10th largest freshwater lake in the world and sixth largest lake in Canada, Lake Winnipeg consists of a large, deeper north basin, and a smaller, relatively shallow south basin. The two basins are separated by the narrows through which waters from the south basin generally flow northward. The north and south basins of the lake differ not only in size and depth, but also in water quality and biological characteristics. The outflow of the lake is regulated, making Lake Winnipeg the third largest hydro-electric reservoir in the world. The lake is of substantial socio-economic and cultural importance, supporting numerous shoreline communities and sustaining a variety of uses including fisheries and recreation, important components of the provincial economy.

The Lake Winnipeg watershed extends across prairie and boreal ecoregions, four provinces and four states, covering about one million square kilometres. The watershed's large size presents unique challenges to water resource management given its geographical diversity, and the inter-jurisdictional and international nature of waters flowing to Lake Winnipeg. The watershed is dominated by agricultural land use and includes densely populated urban centres. The nature of land use in the watershed has implications for the quantity and quality of point and diffuse source discharges to Lake Winnipeg. The lake receives discharges from several large rivers and many smaller tributaries. The Saskatchewan, Red-Assiniboine, and Winnipeg river watersheds are the largest in area and contribute the greatest discharges to the lake. Lake Winnipeg and its main tributaries are regulated by numerous dams and diversions, having implications for nutrient sequestration both within the watershed and the lake.

Climate and Hydrology

The climate of Lake Winnipeg varies from north to south, with cooler, drier conditions to the north and warmer, wetter conditions to the south. Mean annual air temperature from 1999 to 2007 was 0.8^oC at The Pas in the north and 2.5^oC at Gimli in the south. Total annual precipitation ranged from 203 to 425 mm over the north basin, and from 72 to 151 mm over the south basin.

Precipitation over the lake and within the watershed is a key driver of water levels and water residence time in Lake Winnipeg. Precipitation in the immediate vicinity of Lake Winnipeg was below normal in 1999, 2003, 2006, and 2007, and above normal in 2004 and 2005. Flows in tributaries to Lake Winnipeg varied similarly to precipitation over the lake, with substantial reductions in flow in 2003 when there was drought throughout much of the watershed, and peak flows in 2005 associated with above normal precipitation.

Mean monthly discharge to Lake Winnipeg from 1999 to 2007 was 2,586 cubic metres per second (m^3/s). Mean monthly outflow from the Nelson River was 2,380 m^3/s. The Winnipeg River accounted for 49 % of discharges to the lake, the Saskatchewan 25 % and the Red 16 %; 10 % of discharges were from small rivers in the immediate vicinity of the lake and the Dauphin River. Mean monthly discharge for the Winnipeg River was 1,064 m^3/s, greater than that of the Red (346 m^3/s) and Saskatchewan (556 m^3/s) combined. However, flows on the Red River have increased since the early 1990s and the river now contributes proportionately more flow to Lake Winnipeg than historically. Mean monthly discharge to Lake Winnipeg was at a low for the period in 2003 at 537 m^3/s and at its highest in 2005 at 6,854 m^3/s. Water level in Lake Winnipeg's north basin was 216.98 metres above sea level (masl) in 2003 and 218.41 masl in 2005; while levels in the south basin in these years were 216.91 masl and 218.39 masl, respectively. Water residence time was also highest in 2003, at 7.0 years for the lake as a whole, twice the mean for 1999 to 2007.

Physical Characteristics

Water temperatures in Lake Winnipeg are typically cooler to the north and warmer to the south. Mean summer surface water temperatures from 1999 to 2007 were 19.7°C in the north basin and 21.5°C in the south basin and narrows. The shallow, wind-mixed waters of the lake are typically homogeneous in temperature with depth, although recent observations have recorded brief periods of thermal stratification in the north basin over the open water season (e.g., 2003) and under-ice (e.g., 2007).

Dissolved oxygen concentrations in Lake Winnipeg are typically uniform with depth, although recent observations have noted complex spatial and temporal variation in concentrations throughout the lake. Recent periods of under-saturation have occurred in association with thermal stratification events in the north basin, highlighting the potential for oxygen depletion in bottom waters of the lake. Mean open water concentrations of dissolved oxygen were greater than 8.8 miligrams per litre (mg/L) in the north basin from 1999 to 2007 and greater than 8.4 mg/L in the south basin; concentrations were for the most part above the water quality objective for the protection of aquatic life.

Total suspended solids concentrations in Lake Winnipeg are generally higher in the south basin than in the north basin, attributed in large part to discharges from the Red River. However, wind-driven resuspension and erosion in the lake also contribute to suspended solids. Mean annual concentrations over the open water season from 1999 to 2007 varied between 2.6 and 10.6 mg/L in the north basin and 7.7 to 17.4 mg/L in the south basin and narrows. Suspended solids in both basins were at their lowest concentrations in 2003 when tributary discharges to the lake were reduced in response to drought in the watershed.

Chemical Characteristics

Lake Winnipeg waters are fresh, alkaline, and well buffered with bicarbonate, sulphate, and calcium being the dominant ions. Calcium, magnesium, potassium, alkalinity, and pH are generally similar between lake basins. However, there are clear spatial differences in sulphate, sodium, and chloride in Lake Winnipeg. Sulphate is approximately 20 % higher in the south basin as compared to the north basin because of the influence of the Red River, which carries elevated concentrations of sulphate. Sodium and chloride concentrations are twice as great in the north basin which likely arises from the inflow of the Dauphin River, which is rich in sodium and chloride. In the south basin, the Winnipeg River effectively dilutes sodium and chloride concentrations in the lake given the large volume of dilute Winnipeg River water which flows into the lake. Comparison to data collected from 1999 to 2007 indicates that ion concentrations and measurements of conductivity were lower than those noted during a 1969 survey of Lake Winnipeg, and may be a result of higher water levels experienced during 1969.

Nutrients in Lake Winnipeg waters are strongly associated with inflowing waters from tributaries and total suspended solids concentrations. The highest concentrations typically occur in the south basin, near the inflow from the Red River, declining northward. Mean annual concentrations of total phosphorus over the open water season from 1999 to 2007 were 0.113 mg/L in the south basin and narrows, and 0.044 mg/L in the north basin. Concentrations were highest in 2005, corresponding with above normal precipitation and the greatest inflows from tributaries over that period. Total nitrogen concentrations from 1999 to 2007 varied spatially and temporally in a manner similar to those of total phosphorus. Mean concentration over the open water season from 1999 to 2007 was 0.869 mg/L in the south basin and narrows, and 0.654 mg/L in the north basin. The highest total nitrogen concentration, however, occurred in 2006 when algal blooms covered much of the north basin during the summer and fall. The dominance of elevated phosphorous concentrations in the south basin contributes to nitrogen-limitation (mean annual range 12:1 to 25:1), which transitions to phosphorous-limitation in the north basin (14:1 to 92:1). Despite low N:P ratios that favour the development of cyanobacterial blooms in the south basin, algal blooms are generally less extensive than in the north basin because higher suspended solids concentrations limit light penetration through the water column in the south basin.

Nutrient Loading to Lake Winnipeg

Nutrient loading to Lake Winnipeg increased between 1999 and 2007, attributed primarily to discharges from the Red River. The total phosphorous load to Lake Winnipeg was on average 7,655 tonnes per year and ranged from a high of 13,043 tonnes in 2005 to a low of 3,384 tonnes in 2003, in association with above and below normal precipitation and associated river flows. The Red River was the main source of phosphorus to Lake Winnipeg contributing on average 5,380 tonnes of phosphorus per year to the lake or 68 % of the annual total load. The total nitrogen load between 1994 and 2007 was on average 90,701 tonnes per year and ranged from a high of 122,491 tonnes in 2005 to a low of 51,737 tonnes in 2003. The Red River contributed 34 % of the annual total load over the study period and, on average, 31,476 tonnes of nitrogen per year. In addition to tributary discharges, nitrogen fixation and atmospheric deposition contributed to nitrogen loading (18 %). On average, about 60 % of the phosphorus entering Lake Winnipeg through the tributaries and atmospheric deposition was retained in the lake with about 40 % exiting through the Nelson River; approximately 54 % of the nitrogen entering Lake Winnipeg was retained with about 46 % exiting through the Nelson River.

Within the watershed, average annual phosphorous yield from 1999 to 2007 was 0.07 kilograms per hectare per year (kg/ha/y) while annual nitrogen yield was 0.89 kg/ha/y. Nutrient retention in numerous lakes, reservoirs, and wetlands, and the large proportion (31 %) of non-contributing areas throughout the watershed, contribute to relatively low yield as compared to watersheds of other large Canadian lakes. Annual average phosphorous and nitrogen export was highest in those river reaches characterized by urban and residential development, attributed to nutrient loading from wastewater treatment facilities serving larger populations and to higher rates of runoff from areas with low infiltration.

Biological Characteristics

Annual phytoplankton biomass (as chlorophyll a) varied between 5.6 micrograms per litre (μg/L) to 19 μg/L between 1999 and 2007, with species abundance and biomass dominated by cyanobacteria (between 13 and 94 % annually) and diatoms. The greatest occurrence of cyanobacteria (i.e., *Aphanizomenon*) was in 2006, when high chlorophyll a concentrations corresponded with satellite imagery showing large surface blooms (>5,000 km^2) in the north basin of the lake. This was the warmest summer recorded between 1999 and 2007. Diatoms, on the other hand, were prevalent in wet, cool years owing to their low light and lower optimal temperature requirements for growth.

Among the cyanobacteria, taxa capable of nitrogen fixation (e.g., *Aphanizomenon, Anabena*) were prevalent, with relative biomass between 11 and 97 % of total cyanobacteria biomass. *Aphanizomenon* dominance was coincident with the large peak in cyanobacteria in 2006. Blooms of non nitrogen-fixing cyanobacteria (e.g., *Microcystis*) also comprised a significant fraction of the cyanobacteria biomass in some years, with relative biomass varying from 4 to 89 % of the total cyanobacteria biomass. All large blooms of *Microcystis* (>10,000 mg/m^3) occurred in the south basin of Lake Winnipeg, favoured by turbid conditions. Over the nine-year period of record, *Microcystis* was not commonly found in the north basin.

The mean under-ice chlorophyll a concentration (3.2 μg/L) was 3.5 times lower than the open water concentration for the whole lake. Examination of species composition in 2001 revealed that *Aulacoseira* was among the dominant diatom taxa at all sites in the narrows and the south basin. The average under-ice phytoplankton biomass was approximately six times higher in the north basin and was mainly comprised of the dinoflagellate *Perdinium*.

Benthic organisms in the lake as a whole have undergone substantial increases in density in recent decades. Mean density in the lake more than tripled between the summers of 1969 and 2002. Midges and aquatic worms comprise a substantial component of the benthic macroinvertebrate community and increased markedly in density between 1969 and 2002 in the north basin of the lake. The benthic community may be responding both directly and indirectly to the increased availability of food resources as a consequence of nutrient enrichment in Lake Winnipeg.

Lake Winnipeg has long supported important subsistence, recreational, and commercial fisheries. Lake Whitefish harvest, upon which the commercial fishery was originally based, decreased in the 1930s and percids (Sauger and Walleye) began to dominate the fishery. Walleye harvest increased until the mid-1980s, declined until the mid-1990s, then increased to its highest level in 2007. Strong positive relationships between Walleye abundance and total phosphorous concentration, chlorophyll a concentration, and zooplankton density have been observed in large, shallow lakes in central Ontario. Small fishes that dominated the offshore waters of Lake Winnipeg included Emerald Shiner, Rainbow Smelt, and Cisco. Biomass of Emerald Shiner and Cisco was greater in the south basin and the narrows compared to the north basin. Non-native Rainbow Smelt are now an important part of the offshore prey fish community in the north basin. Biomass of Rainbow Smelt was greater in the north basin of the lake compared to the narrows and the south basin, increasing over time in the north basin.

Current and Emerging Issues

The health of the Lake Winnipeg ecosystem is not only affected by the loading of nitrogen and phosphorus to the lake, but also by other factors that may stress the system, causing changes in the water chemistry and biology of the lake, and potentially affecting biota and humans. These factors include the occurrence of algal blooms, the production of toxins by algae, the invasion of aquatic species into the lake, and climate change. The production and frequency of algal blooms have increased in Lake Winnipeg in recent years, with shifts towards a higher percentage of cyanobacteria and reduced diatom abundance since 1969, and the development of mid- to late summer surface algal biomass, concurrent with rising phosphorous concentrations in the lake and the occurrence of significant toxin levels. Toxin-producing species of cyanobacteria are common in Lake Winnipeg. Concentrations of microcystin in whole water and plankton net samples from 1999 to 2007 were generally low in pelagic areas of the lake, with elevated levels posing potential human risk occurring periodically offshore in association with intense offshore blooms and nearshore along beaches.

Escherichia coli (E. coli) is another potential concern in Lake Winnipeg. The vast majority of results over the last six years of monitoring bathing water, however, indicated that *E. coli* densities were within the recreational water quality objective, though higher in wet foreshore sand. Results from microbial source tracking techniques indicated that approximately 63 % of the total number of *E. coli* isolates originated from animals (primarily shorebirds and geese) and 5 % were from human sources. Concerns also surround aquatic invasive species. Manitoba has several aquatic invasive species, which include Common Carp, Rainbow Smelt, White Bass, *Eusbomina coregoni,* and Asian tapeworm. Of potential future concern are zebra mussels, which have recently appeared within the Lake Winnipeg watershed and could become costly to local economies, impacting water-based infrastructure, fish habitat and aquatic ecology, and recreation. In addition to the above issues, climate change also has the potential to affect the ecology and biology of Lake Winnipeg. Climate-forcing scenarios predict increases in mid-summer temperatures through the 21st century, with even greater increases in mid-winter temperatures, extending the open water season.

Knowledge Gaps

Gaps in knowledge regarding Lake Winnipeg limit progress toward a more comprehensive understanding of the lake, nature of stressors, and the response of the lake to these stresses. Information gaps still remain in regards to evaporation, groundwater, and rivers flowing from the eastern watershed of Lake Winnipeg. Spatial and temporal variability of the physical characteristics of Lake Winnipeg's waters require further investigation for a better understanding of the full range of conditions in the lake. Investigation of the nutrient balance of Lake Winnipeg has progressed substantially over the last decade, including modelling of nutrient-loading scenarios and preliminary examination of internal nutrient loading. Advancements have also been made in examining the watershed as a source of nutrients and the role of land use management practices in affecting nutrient loading.

A better understanding of nutrient dynamics in the lake would be gained from information on internal nutrient loading and resuspension, atmospheric nutrient deposition, variability in the rates of nitrogen fixation and denitrification, and nutrient-food web interactions. The biological component is perhaps the least understood in terms of trophic structure, food web linkages, and top-down bottom-up interactions. Knowledge of factors controlling bloom dynamics, the trophic position of Smelt, and the increased production of Walleye in the lake would be highly advantageous. In addition, further understanding of algal toxin dynamics and associated species and genotypes is needed, including the potential for transfer of microcystin to higher trophic levels.

TABLE OF CONTENTS

Continued...

Continued...

ACRONYMS

AIS - Aquatic Invasive Species
AVHRR - Advanced Very High Resolution Radiometer
CGCM3 - Version 3 of the Canadian Global Climate Model
Chla - Chlorophyll a
DFO - Department of Fisheries and Oceans (Fisheries and Oceans Canada)
DIN - Dissolved Inorganic Nitrogen
EC - Environment Canada
EGS - Ecological Goods and Services
GHG - Greenhouse Gas
IBA - Important Bird Area
LWBI - Lake Winnipeg Basin Initiative
LWIC - Lake Winnipeg Implementation Committee
LWRC - Lake Winnipeg Research Consortium
LWSAC - Lake Winnipeg Shoreline Advisory Committee
LWSB - Lake Winnipeg Stewardship Board
MAC - Maximum Acceptable Concentration
MERIS - Medium Resolution Imaging Spectrometer
MMREM - Manitoba Department of Mines Resources and Environmental Management
MSC - Meteorological Service of Canada
MWQSOG - Manitoba Water Quality Standards Objectives and Guidelines
MWS - Manitoba Water Stewardship
NH4 - Ammonium
NO3 - Nitrate
NTU - Nephelometric Turbidity Units
PAR - Photosynthetically Active Radiation
PO4 - Phosphate
PP - Particulate Phosphorus
SC - Statistics Canada
SE - Standard Error
SRP - Soluble Reactive Phosphorus
TDP - Total Dissolved Phosphorus
TN - Total Nitrogen
TP - Total Phosphorus
TSS - Total Suspended Solids
WASP - Water Analysis Simulation Program
WSC - Water Survey of Canada

ACKNOWLEDGEMENTS

This report is a first of its kind compilation of physical, chemical and biological information on Lake Winnipeg and would not have been possible without the hard work and coordination efforts of Lucie Lévesque (Environment Canada) and Elaine Page (Manitoba Water Stewardship).

Contributing Authors

Armstrong, Nicole; Manitoba Water Stewardship, Winnipeg, MB
Bunting, Lynda; University of Regina, Regina, SK
Cumming, Brian; Queens University, Kingston, ON
Donald, Dave: Environment Canada, Regina, SK
Engstrom, Daniel; Science Museum of Minnesota, St. Paul, MN
Franzin, William; Fisheries and Oceans Canada, Winnipeg, MB
Goldsborough, Gordon; University of Manitoba, Winnipeg, MB
Grosshans, Richard; International Institute for Sustainable Development, Winnipeg, MB
Guo, Jay; Environment Canada, Burlington, ON
Gurney, Sharon; Manitoba Water Stewardship, Winnipeg, MB
Hann, Brenda; University of Manitoba, Winnipeg, MB
Herbert, Claire; Parks Canada, Winnipeg, MB
Hiriart-Baer, Véronique; Environment Canada, Burlington, ON
Hobson, Keith; Environment Canada, Saskatoon, SK
Kling, Hedy; Algal Taxonomy and Ecology Inc., Winnipeg, MB
Kotak, Brian; AlgalTox International, Pine Falls, MB
Laird, Kathleen; Queens University, Kingston, ON
Leavitt, Peter; University of Regina, Regina, SK
Lévesque, Lucie; Environment Canada, Saskatoon, SK
Lumb, Chelsey; Manitoba Water Stewardship, Winnipeg, MB
McCullough, Greg; University of Manitoba, Winnipeg, MB
Ofukany, Amy; Environment Canada, Saskatoon, SK
Page, Elaine; Manitoba Water Stewardship, Winnipeg, MB
Parker, Brian; Environment Canada, Winnipeg, MB
Parks, Candace; Manitoba Water Stewardship, Winnipeg, MB
Ralley, Wendy; Manitoba Water Stewardship, Winnipeg, MB
Ramlal, Patricia; Fisheries and Oceans Canada, Winnipeg, MB
St. Amand, Ann; PhycoTech Inc., St. Joseph, MI
Stainton, Mike; Fisheries and Oceans Canada, Winnipeg, MB
Wassenaar, Len; Environment Canada, Saskatoon, SK
Watkinson, Doug; Fisheries and Oceans Canada, Winnipeg, MB
Watson, Sue; Environment Canada, Burlington, ON
Wrubleski, Dale; Ducks Unlimited Canada, Stonewall, MB
Yerubandi, Ram; Environment Canada, Burlington, ON
Zhang, Weitao; Environment Canada, Burlington, ON
Zhao, Jun; Environment Canada, Burlington, ON

Report Steering Committee

Lucie Lévesque; Environment Canada (Lead)
Elaine Page; Manitoba Water Stewardship (Lead)
Nicole Armstrong; Manitoba Water Stewardship
Malcolm Conly; Environment Canada
John Lawrence; Environment Canada
Dwight Williamson; Manitoba Water Stewardship

External Review

Les Swain; TriStar Environmental Consulting, Malahat, BC
Ellie Prepas; University of Alberta, Edmonton, AB

Editing, Graphic Design and Maps

Lloyd Crooks; Beaverbrook Communications Services, Winnipeg, MB
Emily Ritson-Bennett; Environment Canada, Saskatoon, SK

1.0 INTRODUCTION

Lake Winnipeg is the tenth largest freshwater body in the world and an important socio-economic and cultural resource for residents who live along its shores, and throughout the entire province of Manitoba. Nearly 23,000 permanent residents live along the lake in 30 communities. Eleven of these communities are First Nations, with a population of about 14,000 (Lake Winnipeg Stewardship Board 2006). The lake supports commercial and subsistence fisheries, shoreline communities, and recreational uses.

The first commercial fishery was established at Selkirk in 1880. The current commercial fishery on Lake Winnipeg is the largest in western Canada and the harvest of Walleye is the largest in North America at nearly three times that of Lake Erie (Ontario Ministry of Natural Resources 2005). From 2000 to 2004, the five-year average value of the commercial fishery on Lake Winnipeg was approximately $21 million (Lake Winnipeg Stewardship Board 2006), representing about 61 % of the value of all commercial fisheries in Manitoba. Lake Winnipeg is integral to the subsistence of fisheries-based communities and the traditional culture of First Nations. In addition to commercial and subsistence fishing, recreational fishing and bait fishing are also prominent on the lake. Recreational fishing on tributaries to Lake Winnipeg generates an estimated $17 million annually to the provincial economy.

The lake is also a source of drinking water. Several small communities along the shores of the lake, particularly those along the eastern side, draw water from the lake and treat it for community use. Domestic use of water also includes direct withdrawal of untreated water from the lake, as well as withdrawals from wells, tributaries, or local aquifers adjacent to the lake (Lake Winnipeg Stewardship Board 2006).

The lake is the third largest hydro-electric reservoir in the world, and the largest in North America.

Recreational and leisure uses of Lake Winnipeg include beaches, cottaging, all-season recreation, tourism, and eco-tourism, as well as world-class water sporting events, festivals, and all-season resorts. Grand Beach is a popular destination for tourists and residents. More than 10,000 cottages are located around the south basin (Lake Winnipeg Stewardship Board 2006), as well as nine provincial parks and several non-government camps. Manitoba Lowlands National Park has been proposed for the western and northwestern shores of the lake. Recreation and tourism along the Red River and Lake Winnipeg are estimated to contribute $110 million per year to the province of Manitoba.

Water levels on Lake Winnipeg have been regulated since 1976. The lake is the third largest hydro-electric reservoir in the world, and the largest in North America. In fiscal year 2005 to 2006, the sale of hydro-electricity generated nearly $1.9 billion for the Province of Manitoba (Manitoba Hydro-electric Board 2006). Approximately three-quarters of that revenue resulted from hydro-electricity exports to other Canadian provinces and the United States.

Threats to the health of the Lake Winnipeg ecosystem have implications for residents, lake users, and the entire province of Manitoba. Lake Winnipeg water quality has deteriorated over time, with particular concern arising over the past few decades related to the effects of accelerated nutrient enrichment. The frequency and intensity of algal blooms in the lake have increased in association with rising phosphorous and nitrogen loading to the lake from diffuse and point sources through the Lake Winnipeg watershed (McCullough 2007; Bourne *et al*. 2002; Jones and Armstrong 2001).

The lake's large inter-jurisdictional watershed is the second largest in Canada, spanning portions of the provinces of Alberta, Saskatchewan, Manitoba, and Ontario, as well as four U.S. states (Figure 1.1). The substantive size of this watershed relative to the area of the lake and the intensity of land uses contribute to the potential for excesses in anthropogenically-derived substances that may stress the lake's ecosystem beyond levels which the lake is able to assimilate and mediate (Lake Winnipeg Stewardship Board 2006). As well, the inter-jurisdictional and international extent of the watershed presents complex challenges for the management of diffuse and point source nutrient inputs to Lake Winnipeg.

Figure 1.1: *Lake Winnipeg watershed. Internal drainage consists of closed watersheds that do not drain to Lake Winnipeg via large rivers. Non-contributing drainage areas contribute flow only under the wettest of conditions.*

Municipal and industrial wastewater discharges from large cities in the basin contribute nutrients to Lake Winnipeg. In Manitoba alone, Lake Winnipeg receives effluent from nearly 200 wastewater treatment facilities, as well as effluent from 10 large facilities, including municipal and industrial plants (Bourne *et al.* 2002). In Manitoba, new or expanding large wastewater treatment facilities such as those serving the food processing sector or large communities are required to reduce phosphorus in discharges to 1.0 mg/L, and many are also required to reduce nitrogen to 15.0 mg/L. Extensive areas of cropland and intensive livestock operations can also affect the quality and quantity of waters draining to the lake. Agriculture contributes an estimated 5 % of the nitrogen load and 15 % of the phosphorous load to Lake Winnipeg (Bourne *et al.* 2002). Residential lakeshore developments are also sources of nutrients entering the lake.

The Province of Manitoba and the Government of Canada have recognized the need to address nutrient enrichment in Lake Winnipeg. In 2003, Manitoba Water Stewardship announced the Lake Winnipeg Action Plan which outlined a commitment to interim reductions in phosphorous and nitrogen loading to Lake Winnipeg. The Plan also included the establishment of the Lake Winnipeg Stewardship Board to advise the Province of Manitoba on nutrient reduction strategies. In addition, the Province's *Water Protection Act* addresses issues of nutrient enrichment through watershed planning and regulation. The Province of Manitoba is also working towards the development of long-term ecologically-relevant nutrient objectives for Lake Winnipeg. Environment Canada, Fisheries and Oceans Canada, the Lake Winnipeg Research Consortium, university scientists, independent researchers, and others are contributing to the science required to support the development of nutrient objectives. In 2007, as part of the Action Plan on Clean Water, Environment Canada established the Lake Winnipeg Basin Initiative. The Initiative prioritized science and monitoring on the lake and in the watershed, coupled with facilitation of governance and stewardship activities.

In 2008, Environment Canada and Manitoba Water Stewardship committed to producing a *State of the Lake* report for Lake Winnipeg. This report presents the recent state of knowledge on Lake Winnipeg waters and aquatic ecosystem principally as it pertains to eutrophication.

In 2008, Environment Canada and Manitoba Water Stewardship committed to producing a *State of the Lake* report for Lake Winnipeg.

This report describes the temporal and spatial variation in Lake Winnipeg's physical, chemical, and biological characteristics from 1999 to 2007, focussing on the role of nutrients in the aquatic ecosystem. This report also explores current and emerging issues of concern to the health and integrity of Lake Winnipeg, and highlights recent scientific research on the lake. Temporal and spatial variation in Lake Winnipeg conditions are described using data from Manitoba Water Stewardship's long-term water quality monitoring program, spatially-intensive surveys conducted by Fisheries and Oceans Canada and Environment Canada, and studies by universities and independent researchers (Section 2.0, Appendix 1). Metals, major ions, and pesticides information are also available for Lake Winnipeg, but the nature of these in the aquatic ecosystem are outside the scope of this report. This synthesis, the first of its kind for the lake, is intended to serve as a baseline for scientific information and investigations on the lake, and to support future development of performance indicators and nutrient objectives. It represents a collaborative effort among numerous federal and provincial agencies, universities, and independent researchers.

This report also highlights recent and on-going research projects on the lake, featured as text boxes at the end of each section of the report. These include investigation of nutrient sequestration in lakes and reservoirs in the watershed and the role of Netley-Libau Marsh in nutrient transfer to Lake Winnipeg, the physical limnology and hydrodynamics of Lake Winnipeg, nutrient source tracking, paleolimnological investigation of eutrophication in the lake, and preliminary examination of the potential for internal nutrient loading via sediments, as well as isotopic tracking of nutrient inputs to the food web of Lake Winnipeg.

2.0 HISTORY OF MONITORING AND RESEARCH

The first documented surveys on Lake Winnipeg took place in the mid-1920s. Early studies examined water chemistry and the biological characteristics of the lake (Neave 1932, 1933, 1934; Bajkov 1930, 1934; Lowe 1924). From the mid-1960s to the early 1970s, the Province of Manitoba conducted water quality and benthic macroinvertebrate monitoring programs in the north (e.g., Kristofferson *et al.* 1975; Pollard 1973; Rybicki 1966) and south (Crowe 1969, 1972a,b,c, 1973a,b) basins of the lake to characterize the status of the lake and to assess changes relative to earlier studies. Phytoplankton was also studied in the north basin of Lake Winnipeg (Derksen and Hangasjarvi 1979).

Intensive research and monitoring on Lake Winnipeg have been underway since 1999 in response to concerns about the 1997 Red River flood which discharged large volumes of sediment and nutrient-laden water into Lake Winnipeg.

One of the first temporally and spatially-intensive studies on Lake Winnipeg was conducted by Fisheries and Oceans Canada from 1969 to 1971. Samples were collected from 50 stations in the north and south basins over the open water season. Researchers reported on the morphological, hydrological, and watershed characteristics of the lake (Brunskill *et al.* 1980), light attenuation (Brunskill *et al.* 1979b), nutrient loading (Brunskill 1973), dissolved elements, conductance, metabolism (Brunskill *et al.* 1979a), and physical and chemical characteristics of off-shore sediments (Brunskill and Graham 1979). Reports also included results of biological investigations on crustaceous zooplankton (Patalas and Salki 1992; Patalas 1981) and zoobenthos (Chang *et al.* 1992, 1993, 1994; Flannagan and Cobb 1981, 1984, 1991, 1994).

Through the 1970s and 1980s opportunistic sampling was conducted on the lake, although data are limited for this period. The Province of Manitoba collected water samples in the south basin of Lake Winnipeg during the open water season in 1973 and under-ice in 1974 to monitor water quality, nutrients, major ions, and fecal and total coliforms. The Province of Manitoba also conducted an assessment of lake trophic status on the south basin of Lake Winnipeg in the early 1980s. The study included measurement of algal biomass and Secchi disk transparency in the open water seasons in 1980, 1981, and 1982 (Hughes 1983).

The Lake Winnipeg Research Consortium's research vessel MV Namao provides a science platform for scientific research on the lake.

Because of the limited historical monitoring on the lake, the Province began a program in 1992 to assess changes in water quality over time and space. Samples were collected during the open water and under-ice seasons at 38 stations from 1992 to 1997. Water quality parameters measured included macro- and micro-nutrients, ions, dissolved salts, metals, and algal biomass. Monitoring and research in the 1990s also included several whole-lake expeditions conducted by Fisheries and Oceans Canada, a geophysical, hydrographic, and paleolimnological survey of Lake Winnipeg in 1994 (Simpson *et al.* 2003; Kling 1998; Todd *et al*. 1996, 2000) and an evaluation of shoreline erosion and sedimentation processes in the south basin of Lake Winnipeg in 1998 (Baird and Stantec 2000).

Intensive research and monitoring on Lake Winnipeg have been underway since 1999 in response to concerns about the 1997 Red River flood which discharged large volumes of sediment and nutrient-laden water into Lake Winnipeg. The Lake Winnipeg Research Consortium (http://www.lakewinnipegresearch.org/) was established in 1998. The Consortium collaborates with provincial and federal agencies, universities, and independent researchers on issues pertaining to Lake Winnipeg. The Consortium coordinates scientific research on the lake and provides a science platform, the research vessel *MV Namao*. During the open water season, samples are collected in the pelagic zone during the spring, summer, and fall from a network of 65 stations (Figure 2.1).

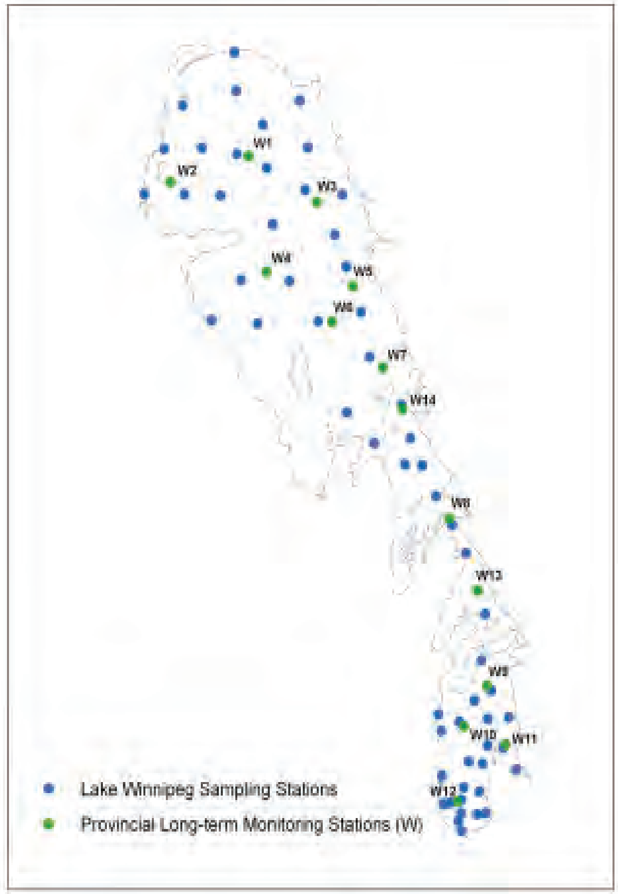


Figure 2.1: *Present-day monitoring stations on Lake Winnipeg. The original network of 65 monitoring stations was selected based on those sampled in 1969; additional stations were subsequently added to the network to maximize spatial representation of the lake.*

The Province of Manitoba manages a lake-wide, long-term monitoring program, established in 1999, to protect and maintain the health of Lake Winnipeg and to assess changes in water quality. Fourteen of the stations in the sampling network (Figure 2.1) are sampled regularly as part of Manitoba Water Stewardship's long-term water quality program. In addition to the open water season, these stations are also sampled once a year under-ice. Samples are analyzed for a variety of parameters including nutrients, metals, pesticides, dissolved oxygen, algal species composition and biomass, microcystin, and others. Manitoba Water Stewardship also collects water samples from the other 50 lake stations for a reduced suite of variables including pH, conductivity, turbidity, and nutrients. Zoobenthos are examined once a year at each location to provide an indicator of ecosystem health. Manitoba Water Stewardship also monitors fish communities in the lake and *Escherichia coli* at beaches in sand and water.

From 2001 to 2004, Fisheries and Oceans Canada conducted whole-lake expeditions each year in collaboration with the Lake Winnipeg Research Consortium to characterize the spatial variability in water chemistry (including investigation via satellite imagery) and to investigate aquatic ecosystem functioning of Lake Winnipeg. On-going research includes studies by Environment Canada (e.g., hydrodynamics and nutrient loading, nutrient source tracking, dissolved oxygen conditions, food web dynamics, phosphorus in sediments) and independent researchers (e.g., thermal modelling, suspended sediment dynamics, zoobenthos, zooplankton, phytoplankton, algal toxins, paleolimnology).

Monitoring, surveillance, and research on Lake Winnipeg faces a number of challenges. Given the size of the lake, collecting samples that are representative of the water quality and biology of the lake over the full range of seasons is difficult, and at times, dangerous, particularly under windy conditions. Samples are primarily collected using the *MV Namao,* and therefore represent pelagic rather than littoral and shoreline areas of the lake. As well, there are only three cruises of the *Namao* each year, limiting the temporal resolution of water quality and biological information on the lake. A study aimed at optimizing the sampling network on Lake Winnipeg is underway by Environment Canada. This study will help to identify the degree to which the current sampling network is representative of the lake as a whole.

3.0 LAKE WINNIPEG AND ITS WATERSHED

(Lucie Lévesque, Elaine Page)

3.1 Lake Winnipeg

Lake Winnipeg (Figure 3.1) is the sixth largest lake in Canada, extending 436 km from north to south and covering an area of 23,750 km^2 (Brunskill *et al.* 1980). The lake consists of two distinct basins. The north basin is approximately 111 km wide, extending from north of Berens Island to the northern shore of the lake. The north basin covers approximately 74 % of the total lake area and holds 81 % of the lake's 284 km^3 volume (Brunskill *et al.* 1980). The south basin is about one-third the width of the north basin, at 40 km, and extends from Black and Hecla islands south to the mouth of the Red River and the Netley-Libau Marsh. The south basin covers 11 % of the area and represents 10 % of the volume of Lake Winnipeg. The north and south basins are separated by a 2.6 km-wide channel known as the narrows. The narrows represents 15 % of the lake area and 9 % of the volume. Water typically flows from the south basin to the north basin through the narrows, although flow reversal occurs at times, largely generated by strong northerly winds, resulting in the flux of water from the north basin into the south basin.

The north and south basins of Lake Winnipeg are unique from one another, not only in terms of physical characteristics, but also in terms of water quality and biological characteristics.

The north and south basins of Lake Winnipeg are unique from one another, not only in terms of physical characteristics, but also in terms of water quality characteristics such as clarity and nutrient concentrations (Sections 5 and 6) and biological characteristics including abundance and composition of phytoplankton communities (Section 8). These variations are driven by differences in the sizes and depths of the basins, as well as the quantity and quality of waters flowing into Lake Winnipeg. Consequently, the basins may be considered as two relatively distinct water bodies. The narrows joining the north and south basins is most distinguished by the relatively rapid flow of water through this portion of the lake.

Lake Winnipeg is shallow compared to lakes in the Laurentian region, including the Great Lakes bordering Canada and the United States. The average depths of Lake Winnipeg's north and south basins are 13.3 m and 9 m, respectively (Brunskill *et al.* 1980). By comparison, the average depth of Lake Erie, the smallest of the Great Lakes by volume (484 km^3), is 19 m. Lake Superior, the largest of the Great Lakes by volume (12,100 km^3) and surface area (82,100 km^2), has an average depth of 147 m (http://www.great-lakes.net). The deepest water in Lake Winnipeg is in the narrows, reaching depths of nearly 60 m north of Black Island. The shallow waters and substantive tributary inflows to Lake Winnipeg (Section 3.2) contribute to water residence times ranging from three to five years, much shorter than those of other large lakes (e.g.191 years for Lake Superior).

The outflow of Lake Winnipeg is regulated, making the lake one of the world's largest hydro-electric reservoirs. Jenpeg Generating Station was constructed in 1976 downstream from the outlet of Lake Winnipeg on the Nelson River, upstream from Cross Lake. Regulation of the lake has increased its outflow capacity by 40 to 50 %, and has reduced the magnitude and frequency of flooding on Lake Winnipeg (Lake Winnipeg Stewardship Board 2006). Since the onset of regulation, water levels have remained relatively unchanged, that is, 217.44 metres above sea level (masl) (1914 to 1976) to 217.45 masl after regulation (1976 to 2007, Figure 3.2), a difference of one centimetre. Natural extremes in high and low water levels have been dampened by regulation, lessening the variability in lake levels.

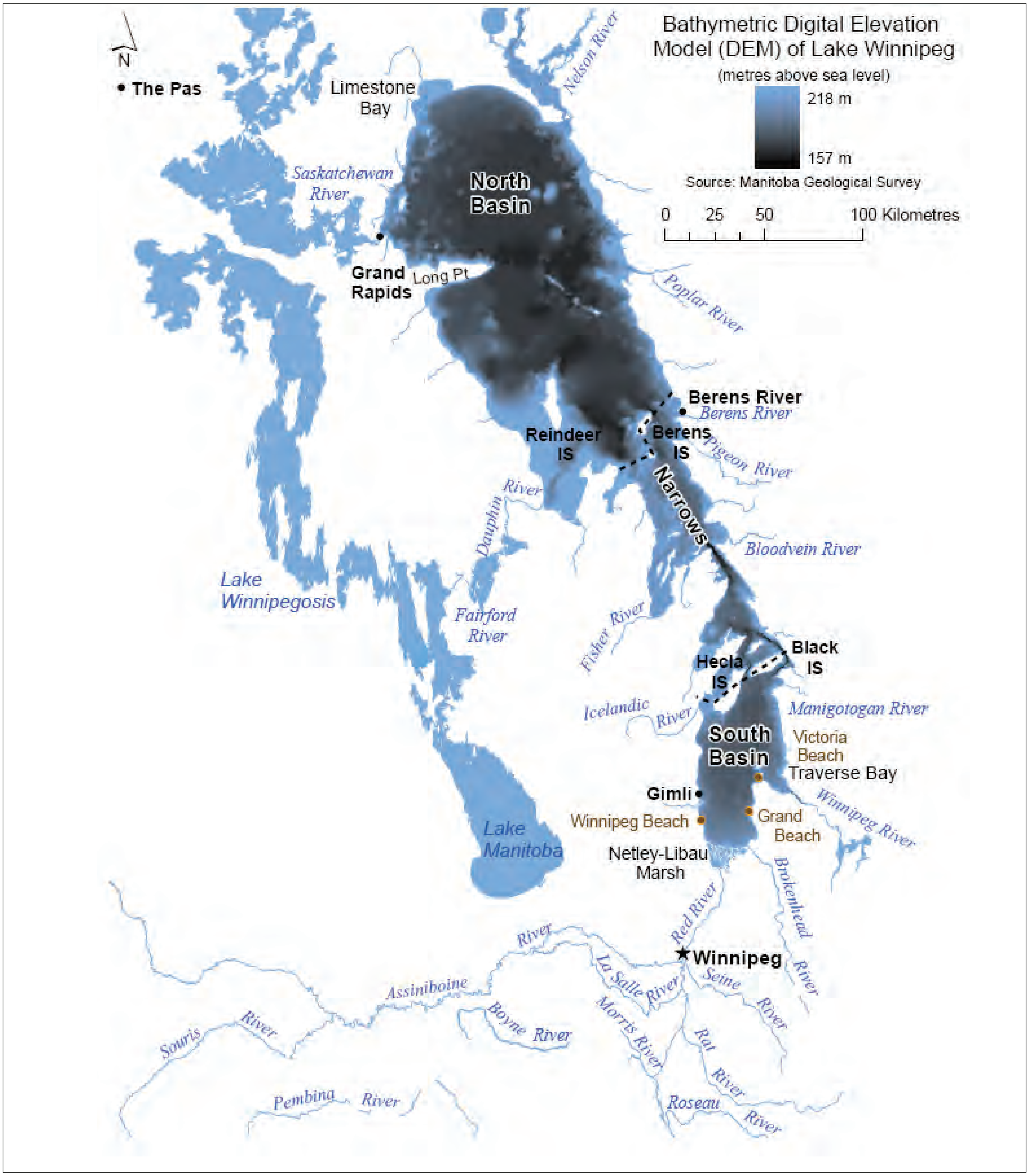


Figure 3.1: *Dominant features and bathymetry of Lake Winnipeg. Dark grey denotes deeper portions of the lake basin; blue, shallower.*

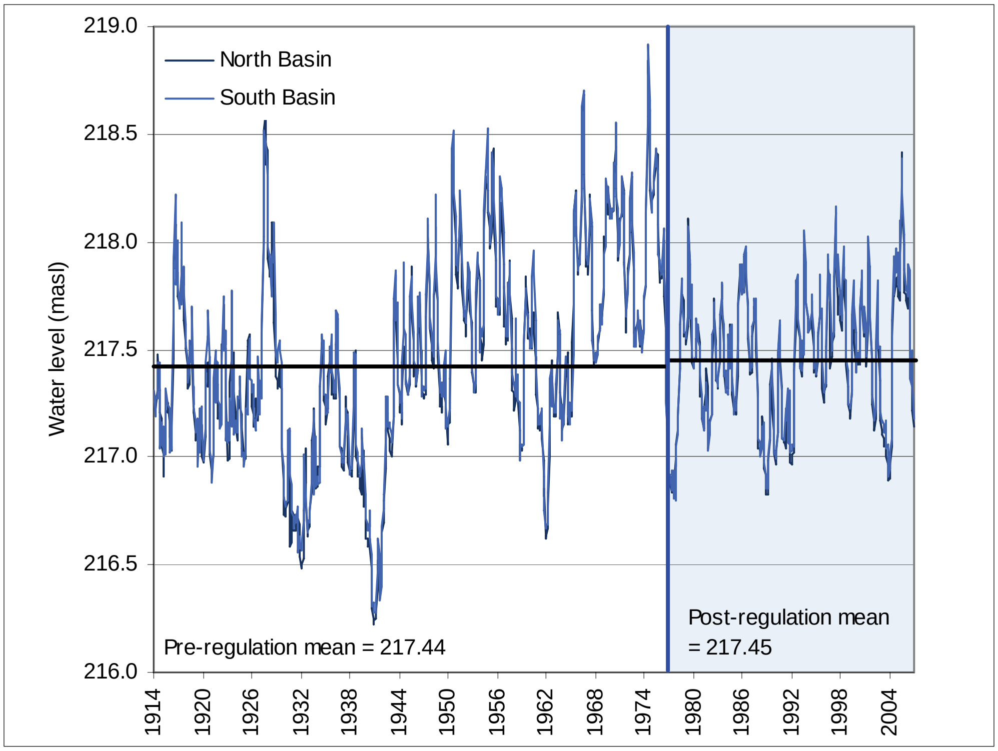


Figure 3.2: *Water levels in the north and south basins of Lake Winnipeg before and after regulation of outflow. (Data Source: Water Survey of Canada, modified by G. McCullough)*

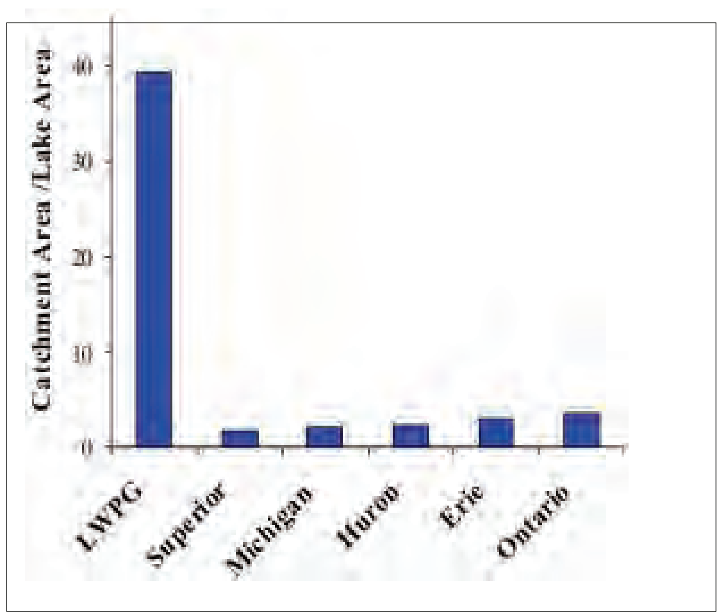


Figure 3.3: *Catchment area ratios for Lake Winnipeg and the five North American Laurentian Great Lakes. (Data Source: Environment Canada, S. Watson)*

3.2 The Watershed

Lake Winnipeg's watershed is the second largest in Canada at approximately one million square kilometres, second only in size to the Mackenzie River watershed. It includes portions of the provinces of Alberta, Saskatchewan, Manitoba, and north-western Ontario, as well as portions of the states of Montana, North Dakota, South Dakota, and Minnesota (Figure 1.1). The watershed-to-lake area ratio for Lake Winnipeg is approximately 40:1, which is much higher than other large lakes of the world (Lake Winnipeg Implementation Committee 2005). Area ratios for the Great Lakes are less than 5:1 (Figure 3.3).

Nearly 40 % of the Lake Winnipeg watershed lies within the Saskatchewan River sub-watershed, 31 % in the Red-Assiniboine sub-watershed and 17 % in the Winnipeg River sub-watershed (Figure 1.1). The substantive size of the watershed, its physiographic and climatic variability, and its inter-jursidictional and international nature make for unique challenges in identifying and managing sources of substances of concern to Lake Winnipeg, and in addressing issues such as the invasion of aquatic species.

Nearly 40 % of the Lake Winnipeg watershed lies within the Saskatchewan River sub-watershed, 31 % in the Red-Assiniboine sub-watershed and 17 % in the Winnipeg River sub-watershed (Figure 1.1).

Urban and agricultural land uses predominate in the watershed and represent sources of nutrients and other substances of concern that may contribute to deterioration of water quality and alteration of the aquatic ecosystem in Lake Winnipeg. Though point and diffuse sources of substances such as nutrients have more immediate local effects, transfers downstream through tributaries have implications for Lake Winnipeg. These may include nutrient enrichment of waters downstream of wastewater treatment plants, untreated discharges, and agricultural runoff.

The population of the Lake Winnipeg watershed is nearly 5.5 million in Canada and 1.1 million in the United States (Statistics Canada and U.S. Bureau of Census, as per Lake Winnipeg Stewardship Board 2006). Approximately 80 % of the human population is located in urban centres (Lake Winnipeg Stewardship Board 2006). Population densities are greatest in the Bow, central North Saskatchewan, and Red river sub-watersheds, associated with the presence of large cities such as Calgary and Edmonton, Alberta, and Winnipeg, Manitoba (Figure 3.4 on the following page). The lowest population densities are located in the sub-watersheds immediately bordering Lake Winnipeg.

Nearly 55 million hectares (ha) of farmland is located in the Canadian portion of the watershed, one-half of which is cropland. An additional 10 million ha of farmland are located in the United States. The areas of greatest agricultural land and fertilizer application are located to the west of Lake Winnipeg in the lower North and South Saskatchewan, Qu'Appelle, and Assiniboine sub-watersheds (Figure 3.5, page 11). In 2006, there were nearly 9.9 million head of cattle in the Lake Winnipeg watershed (www.statcan.gc.ca/start-debut-eng.html) with 800,000 in the United States portion of the watershed. Cattle populations were highest in the Red Deer and upper South Saskatchewan sub-watersheds and lowest in the sub-watersheds bordering Lake Winnipeg to the west and eastward into Ontario (Figure 3.5). There were 14.3 million pigs in the Lake Winnipeg watershed in 2006, with more than one-half of that population in Manitoba, and fewer than 500,000 in the United States portion of the watershed (Lake Winnipeg Stewardship Board 2006). Sheep population in the watershed was approximately 500,000 in 2006, with about 10 % in the United States. Ninety-seven million tonnes of manure were produced in the Canadian portion of the Lake Winnipeg watershed in 2006 (Figure 3.5).

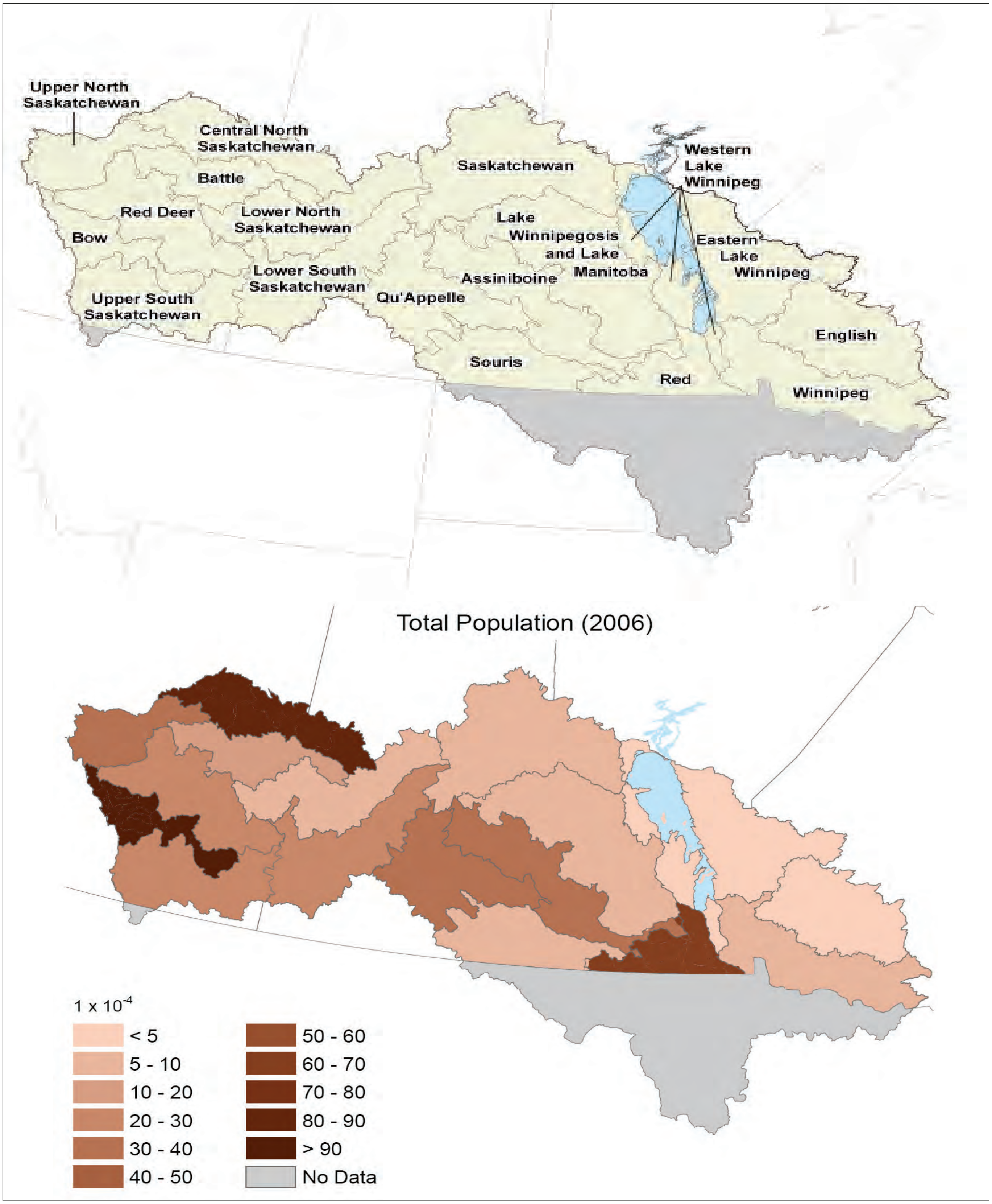


Figure 3.4: *Sub-drainages and total population in 2006. (Data Source: Statistics Canada, Canadian Environmental Sustainability Indicators land use analyses)*

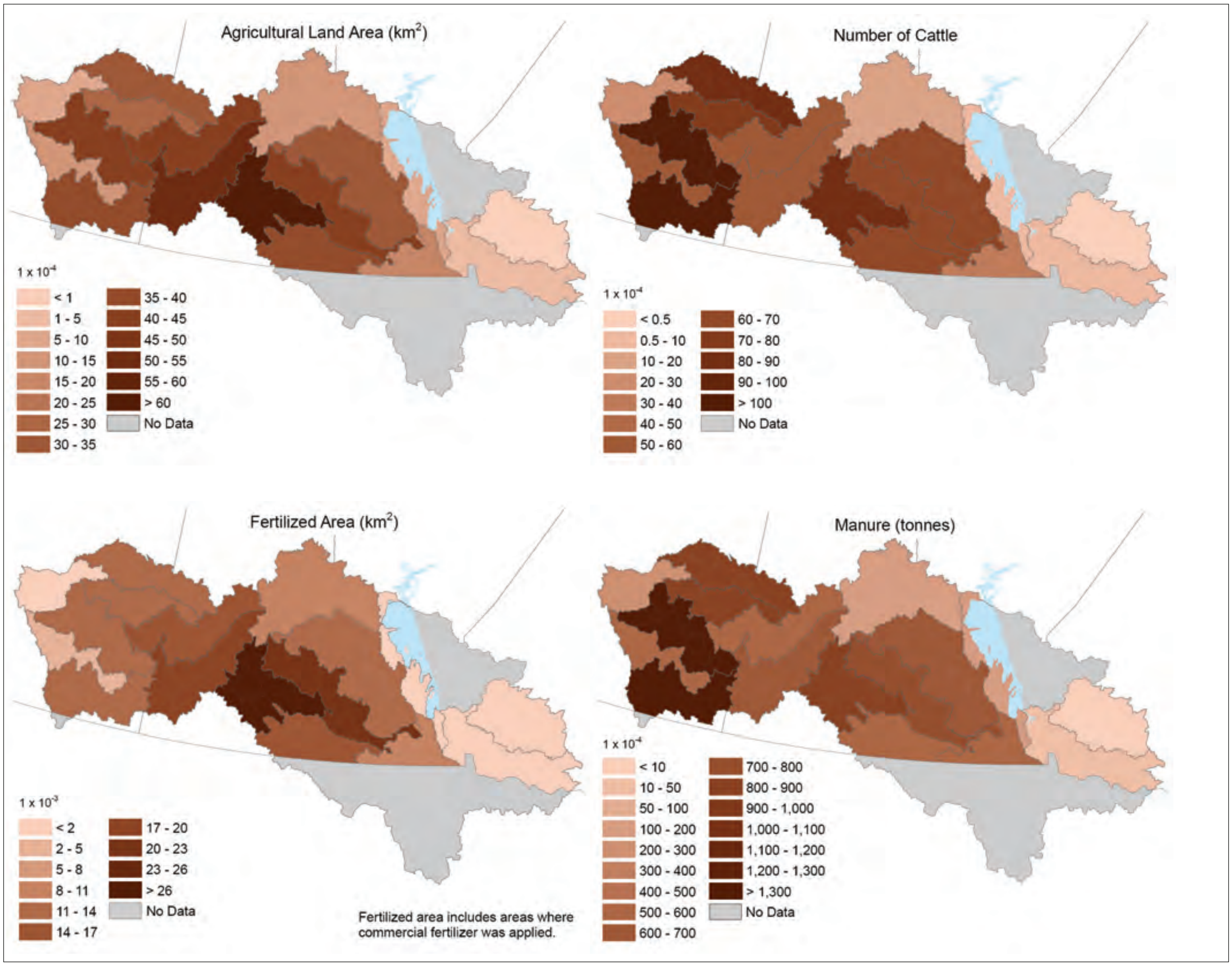


Figure 3.5: *Agricultural land area (<1000 to >60,000 km²), fertilized area (<2000 to >26,000 km²), number of cattle (<5000 to >100,000) and manure production (<100,000 to >14 million tonnes) in the Lake Winnipeg watershed in 2006. (Data Source: Statistics Canada)*

In addition to alteration of the landscape for urban and agricultural land use, the Lake Winnipeg watershed has also undergone intensive modification of water resources and hydrological characteristics. The first water control projects began in the 1890s (Baird and Stantec 2000), followed by the onset of hydro-electric development in the early 1900s. Rivers in the watershed now have numerous dams, diversions, and reservoirs with the potential to affect the quantity and quality of waters draining to Lake Winnipeg (Text Box 1). Drainage and alteration of wetlands and riparian areas in the watershed, and along the shoreline of the lake itself, have also taken place since settlement began, substantially reducing the distribution and density of these features (Lake Winnipeg Stewardship Board 2006). As a result, habitat for waterfowl, shorebirds, wildlife, and fish has been reduced, as has the efficacy of these altered landscapes in moderating erosive flows and in mediating nutrient and sediment loading to receiving waters (Text Box 2).

3.2.1 Climate and Physiography

Waters flowing to Lake Winnipeg are affected not only by land use, but also by the climate and physiography of the watershed. The climate of the Lake Winnipeg watershed is classified as cold continental, with arid conditions throughout, particularly over the warm season through the prairies and plains. Mean daily air temperatures throughout this large watershed vary substantially both seasonally and spatially (Table 3.1). The cold season is dominated by temperatures below 1.0°C and lasts for over one-half of the year; the warm season is dominated by temperatures greater than 10.0°C. Annual mean precipitation is for the most part between 400 and 600 mm, but as low as 200 mm in some areas. The numerous rivers, lakes, wetlands, and reservoirs in the Lake Winnipeg watershed are fed primarily by spring snowmelt runoff and rainfall during spring break-up and over the open water season, and groundwater during the winter.

Table 3.1: *Mean daily air temperature normals (°C; 1971 to 2000) in the Lake Winnipeg watershed. (Data source: http://atlas.nrcan.gc.ca)*

	North		South	
	Min	**Max**	**Min**	**Max**
Spring	-9	0	6	15
Summer	6	15	21	25
Fall	-4	0	6	15
Winter	-29	-15	-19	0

Sedimentary landscapes predominate, with semi-arid and temperate prairies throughout (Figure 3.6a). Boreal plain extends through the northern portion of the watershed and along the western shore of Lake Winnipeg. These sedimentary landscapes are dominated by cropland and grassland. Sedimentary soils are replaced by shallow, bedrock-underlain soils of the Precambrian Shield through the eastern portion of the watershed. Here, bogs and areas of aquatic rooted vegetation are numerous and cover an extensive portion of the landscape (Figure 3.6b). Rivers flowing through these resistant shield landscapes to the north and east of Lake Winnipeg are clearer-flowing than those in sedimentary prairie landscapes, where more easily erodible soils are plentiful.

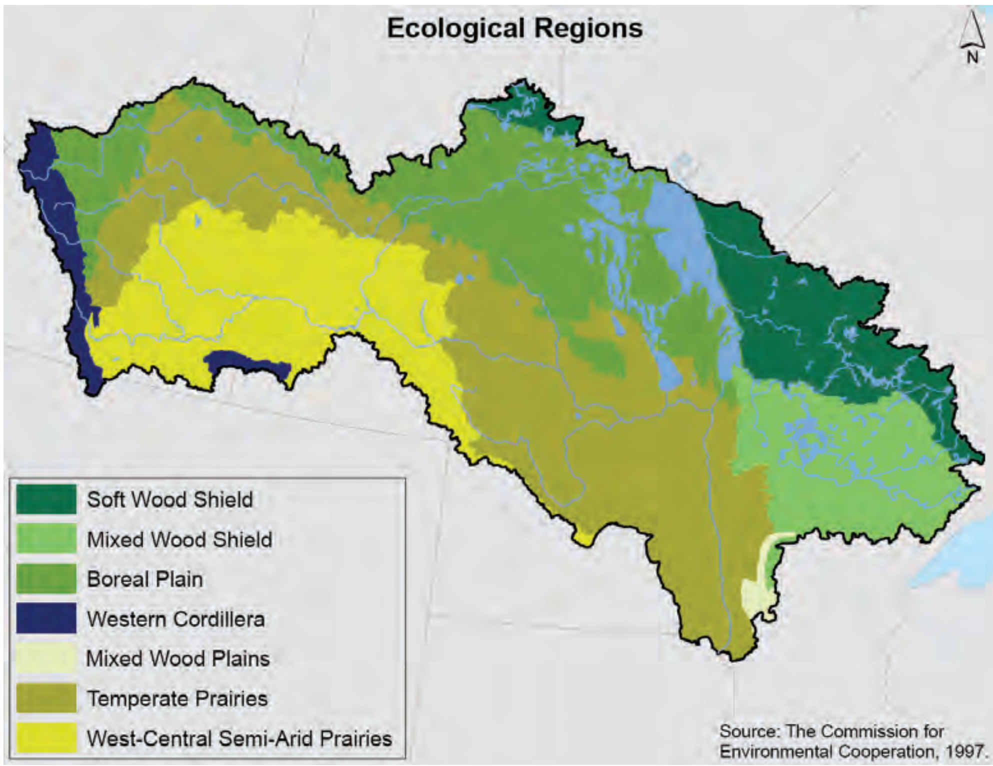


Figure 3.6a: *Ecological regions. (Data Source: Commission for Environmental Cooperation 1997)*

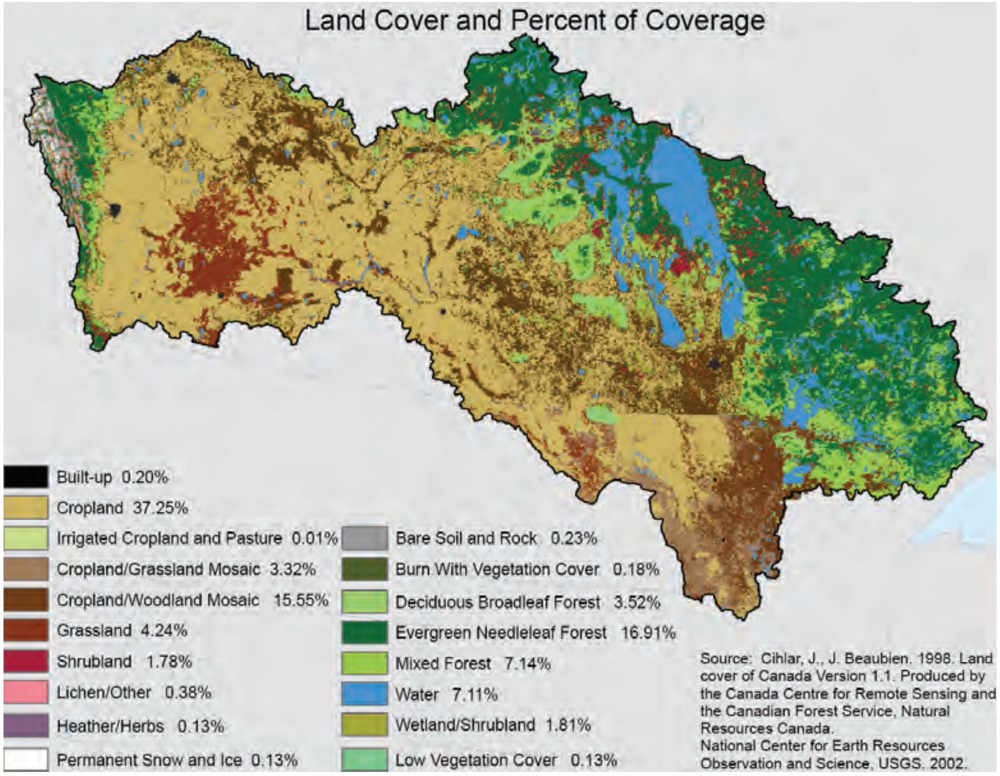


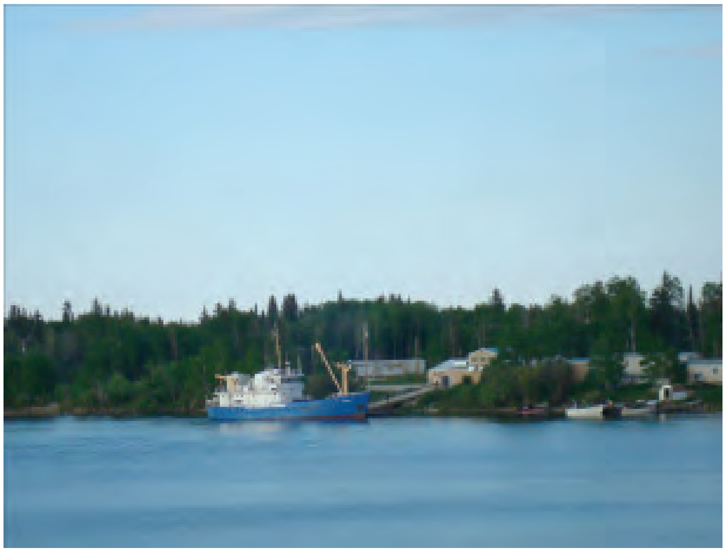
Figure 3.6b: *Land cover (light brown = cropland, dark brown = cropland/ grassland mosaic, light/dark green = forest. (Data Sources: NRC 1998 and USGS 2002) Land cover definitions vary slightly north and south of the Canada-United States border.*

Non-contributing areas in the sedimentary portion of the Lake Winnipeg watershed represent approximately one-third of the total watershed area (Figure 1.1). These areas may not contribute to annual runoff to the lake when precipitation is low. Water is retained as depression storage on the arid landscape (i.e., in prairie potholes or sloughs), or contributes to internal drainage systems (e.g., Quill Lakes, Saskatchewan). Under extreme wet and high antecedent moisture conditions these areas may spill over, increasing the effective drainage area of the watershed and further influencing nutrient yields to Lake Winnipeg. These areas are important contributors to groundwater recharge through the watershed. The role of groundwater discharge in the water budget of Lake Winnipeg relative to inflowing tributaries is not yet understood.

Non-contributing areas in the sedimentary portion of the Lake Winnipeg watershed represent approximately one-third of the total watershed area (Figure 1.1).

3.2.2 Saskatchewan River Sub-watershed

The Saskatchewan River has its headwaters in the Rocky Mountains at the western-most end of the Lake Winnipeg watershed (Figure 1.1). Here, small glacier-fed tributaries flow into the North Saskatchewan River, with the Bow, Oldman, and Red Deer rivers flowing to the South Saskatchewan River in Alberta. The North and South Saskatchewan rivers continue eastward through the arid and flat prairies and plains of Saskatchewan to the south and the boreal plain to the north, where they join to form the Saskatchewan River east of Prince Albert, Saskatchewan. Once in Manitoba, the river flows through the Saskatchewan River Delta, a wildlife area of nearly 10,000 km^2 (Partners for the Saskatchewan River Basin 2009). Downstream from the delta, the Saskatchewan River widens into the Cedar Lake hydro-electric reservoir before passing through the Grand Rapids Generating Station and flowing directly into the north basin of Lake Winnipeg. The Saskatchewan River has been



MV Namao in the Saskatchewan River at Grand Rapids.

regulated at Grand Rapids since 1965. The upper watershed is also regulated with numerous dams, diversions, and reservoirs that have altered the natural discharge of the river.

Runoff and tributary flow to the Saskatchewan River are highly variable and susceptible to periods of drought (Partners for the Saskatchewan River Basin 2009). The western prairies underwent an intensive drought in 2001. This deficit in moisture was attributable to below normal rain and snowfall through much of the basin in 2001 (Canadian Council of Ministers of the Environment 2003) and led to low flows on the Saskatchewan River. Flows from the Saskatchewan River to Lake Winnipeg have been in decline over past decades (Lake Winnipeg Stewardship Board 2006), as have flows in many rivers of the Canadian prairies (St. George 2006; Westmacott *et al.* 1997).

3.2.3 Red River Sub-watershed

The Red River originates in Minnesota near the borders of South Dakota, North Dakota and Minnesota (Figure 1.1). Approximately 80 % of the watershed is located within the United States and 20 % in Manitoba (Jones and Armstrong 2001). Tributaries of the Red River drain rich, productive soils in the predominantly agricultural landscapes of the Red River Valley, contributing to the Red River as it flows northward toward Lake Winnipeg. The river is joined from the west by the Assiniboine River at Winnipeg, then flows north to enter the south basin of Lake Winnipeg.

Discharges in the Red River have been rising since the 1990s and given the relatively large nutrient concentrations in the river, are of prime concern with respect to the health of Lake Winnipeg.

The Assiniboine River (its major tributaries are the Souris and Qu'Appelle rivers) originates in Saskatchewan and drains a flat, arid prairie landscape. Flood control structures on the Assiniboine River at the Manitoba-Saskatchewan border and at Portage La Prairie allow for control of flows in the Assiniboine, and re-direction of flows into Lake Manitoba and Delta Marsh, bypassing Winnipeg.

The general lack of topography in the Red River watershed contributes to slow movement of waters through the watershed (Burn and Goel 2001). The river has a long history of flooding. Flooding typically occurs in response to high fall moisture and large quantities of snowmelt runoff in combination with spring rainfall. More recent large floods occurred in 1997, 2005, and 2009, with the flood of 1997 being termed by some as the "Flood of the Century" when peak discharges in the river were at their highest in 145 years (Burn and Goel 2001). Discharges in the Red River have been rising since the 1990s (Lake Winnipeg Stewardship Board 2006) and given the relatively large nutrient concentrations in the river (see Section 7.0), are of prime concern with respect to the health of Lake Winnipeg.

3.2.4 Winnipeg River Sub-watershed

The Winnipeg River originates downstream from Lake of the Woods and flows northwest to Lake Winnipeg (Figure 1.1). The majority of the watershed is located in Ontario where tributaries flow through low, rolling Precambrian Shield terrain that is predominantly forested and has many lakes. The watershed also includes the Rainy River which flows from northern Minnesota into Lake of the Woods.

The flow of the Winnipeg River has been regulated since the 1880s (Lake of the Woods Control Board 2002). There are a number of hydro-electric generating stations controlling the flow including Pine Falls, McArthur Falls, Seven Sisters, Pointe du Bois, Slave Falls, and Great Falls. Flows are highly responsive to antecedent moisture conditions (St. George 2006) and water control within the watershed. Mean annual discharges have been increasing since 1924, attributed primarily to winter discharges, which have increased by between 61 and 110 % in response to rising precipitation in summer and fall (St. George 2006).

Winnipeg River

3.2.5 Lake Winnipeg Sub-watershed (Dauphin River, Interlake Region, Unmetered Rivers)

The Lake Winnipeg sub-watershed consists of the Dauphin River watershed and the Interlake Region to the west of Lake Winnipeg, as well as numerous rivers to the east of the lake (Figure 1.1). The Dauphin River watershed is also known as the Lake Manitoba watershed (The Lake Manitoba Regulation Review Advisory Committee 2003). This watershed includes part of east-central Saskatchewan and much of west-central Manitoba, with forested plains to the north and prairies dominated by agriculture to the south. The discharge of the Dauphin River is moderated by the presence of two large lakes within the basin - Lake Winnipegosis and Lake Manitoba. Lake Winnipegosis feeds into Lake Manitoba, which in turn drains though its only outlet, the Fairford River, into Lake St. Martin. From there flow continues eastward down the Dauphin River into the north basin of Lake Winnipeg. Water levels of Lake Manitoba and discharge in the Fairford River have been regulated since 1961.

Little is known of the quantity and quality of waters flowing from the numerous unmetered rivers immediately east of the lake.

The Interlake region immediately west of Lake Winnipeg is drained by small rivers and streams in forested and agricultural landscapes. East of Lake Winnipeg, numerous rivers, streams, and wetlands drain forested Precambrian Shield terrain. Water yields are high relative to rivers through the rest of the watershed (Section 4.3). Relative to other rivers in the Lake Winnipeg watershed, little is known of the quantity and quality of waters flowing from the numerous unmetered rivers immediately east of the lake.

Text Box 1. Nutrient Sequestration

(Brian Parker, David Donald)

Lake Winnipeg receives nutrient-rich surface water inputs from a large and complex watershed that includes small and large watercourses, thousands of natural lakes, and hundreds of man-made reservoirs. Lakes and reservoirs are thought to play an important role in limiting the eutrophication of Lake Winnipeg due to their hypothesized capacity to sequester or trap nutrients and thereby reduce the downstream transport of nutrients to Lake Winnipeg.

In 2008, Environment Canada began measuring inputs (from surface inflows and precipitation) and outputs (from lake/reservoir outflows) of nutrients from 25 larger (>3 km^2 in area) lakes and reservoirs in the Lake Winnipeg watershed with the goal of determining the amount of phosphorus and nitrogen sequestered in those systems. Study sites include lakes Dauphin, Manitoba, and Winnipegosis, the large mainstem and several mountain and foothill reservoirs of the Saskatchewan River system, the rapidly-flushing impoundments of the Winnipeg River, several smaller prairie lakes of the Pembina River chain and, with international collaboration, Lake Ashtabula in the Red River basin in North Dakota.

Based on preliminary data, it appears that the mainstem reservoirs of the Saskatchewan River basin and larger prairie lakes effectively sequester nutrients, particularly phosphorus. Sequestration rates for phosphorus varied from one percent in small rapidly flushing lakes to 90 % in large mainstem reservoirs with long water renewal times. In total, lentic waters in the western portion of the Lake Winnipeg basin sequestered more than 2,000 tonnes of phosphorus in 2009, significantly reducing downstream transport of this nutrient towards Lake Winnipeg. In contrast, phosphorus was released from run-of-river reservoirs of the Winnipeg River system to the east of Lake Winnipeg. Nutrient release was likely due to changes in surficial geology which varies from metamorphosed, nutrient-poor rock of the Precambrian Shield to comparatively nutrient-rich till and clay deposits along the lower portion of this river system.

Although the majority of the natural lakes and reservoirs studied appear to effectively sequester phosphorus, they do appear to vary in their function, as shown by high estimated retention rates for soluble phosphorus in natural lakes, but high retention of particulate phosphorus in reservoirs. Further, for water bodies with similar water renewal times, reservoirs sequester a higher proportion of incoming phosphorus than do lakes. The causes and persistence of these differences are under investigation.

In summary, natural lakes and man-made reservoirs in the Lake Winnipeg watershed show variable ability to sequester nutrients, but, in aggregate, sequester and prevent the downstream transport of large amounts of nutrients to Lake Winnipeg. Nutrient sequestration in upstream water bodies thus lowers the potential for even higher algal production in Lake Winnipeg. This research will continue through 2012, as part of the federal Lake Winnipeg Basin Initiative.

Text Box 2. Netley-Libau Marsh

(Gordon Goldsborough, Dale Wrubleski, Richard Grosshans)

At approximately 26,000 hectares (ha), Netley-Libau Marsh is one of the largest freshwater coastal wetlands in North America. Situated along the south shore of Lake Winnipeg, separated from it by a 25-km long series of barrier islands, the marsh is a complex of shallow lakes, lagoons, and channels ranging in depth up to three metres. The marsh is bisected by the Red River, which branches into three main channels before reaching Lake Winnipeg. The portion of the marsh lying west of the river is known locally as the Netley Marsh whereas the portion east of the river is the Libau Marsh. The term Netley-Libau is used to describe the entire complex. Over 90 % of the marsh is publicly owned including a 1,073 ha game bird refuge.

The marsh provided resources for early aboriginal people and European settlers. Recreational and agricultural uses through the late 19th and early 20th centuries were focussed on the abundant fish and wildlife, along with marsh vegetation used as a source of hay for livestock. Present uses include hunting, fishing, boating, bird-watching, and ecotourism. The marsh is recognized internationally as a habitat for nesting, staging, and moulting waterfowl, and has been designated as an Important Bird Area (IBA). The marsh also provides spawning, nursery, and feeding habitat for fish from Lake Winnipeg and the Red River. The marsh provides ecological goods and services such as filtering and sequestering nutrients from the Red River into Lake Winnipeg - a function that is increasingly understood to be a key component of an overall nutrient management strategy.

Waterbird populations have declined throughout the marsh over the past three decades. Between 1979 and 2001, loss of emergent plant species and the erosion of separating uplands between adjoining water bodies has been extensive, resulting in the amalgamation and expansion of many marsh bays and ponds. Half the entire complex (13,125 ha, 51 %) was open water in 2001, compared to 35 % (8,884 ha) in 1979. Cattail (*Typha spp.*) continues to be the dominant emergent species. However, hard- and soft-stem bulrush (*Schoenoplectus spp.*) declined ten-fold. Since droughts of the 1930s and 1940s, there have been few intervening dry periods that would allow new emergent plants to germinate. Without plant roots to anchor soft sediments, islands and upland habitats have eroded.

Netley Cut, a short channel dredged from the Red River into Netley Marsh in 1913, has been gradually eroded to a point where its channel is now wider than the river. Recent (2009) flow estimates reveal that at least one-third of the Red River volume is now passing into the marsh rather than through its historical channels into Lake Winnipeg. The consequences of this input on marsh hydrology and the associated loading of nutrients and other chemicals are unclear. This may explain the greater extent of vegetation change in Netley Marsh compared to Libau Marsh, which has only limited input of Red River water.

Continued...

Netley Cut, 1923.

Netley Cut, October 2003.

Text Box 2, Continued

Dredging the main channel at the Red River mouth ended in 1999. This has also likely contributed to the alteration of Red River flows through the marsh. High nutrient loads in the Red River, along with the arrival of common carp (*Cyprinus carpio*), may be contributing to enhanced algal growth. Nutrient-diffusing substrata have been deployed at various sites in the marsh, and the growth of periphyton on them has been measured over a period of three weeks, to assess the state of nutrient deficiency. Results have consistently demonstrated severe nitrogen limitation of algal growth, perhaps because loading from the Red River meets all requirements for phosphorus.

Since 2006, research has examined biomass and nutrient accumulation in marsh plants to understand nutrient cycling and storage within the marsh ecosystem. Cattails (*Typha spp.*) and reed grass (*Phragmites australis*) are particularly effective in absorbing nitrogen and phosphorus from sediment and water, assimilating them into biomass. Cattails absorb a significant amount of stored phosphorus from sediments during a single growing season. Plant tissue, water, and sediment from Netley Marsh have, on average, higher nitrogen and phosphorus than those from Libau Marsh.

Research has also examined the feasibility of harvesting plant biomass to prevent the return of nutrients into the wetland during natural decomposition. The removal of accumulated dead plant material removes significant stored phosphorus and stimulates plant growth in the spring. Harvested biomass provides feedstock for bioenergy production. Technologies for the conversion of biomass to energy are being examined, providing further insights on harvesting and utilization methods.

Netley-Libau Marsh lies at the interface of the Red River and Lake Winnipeg ecosystem. Its health is an integral part of reducing Lake Winnipeg's nutrient load. A healthy diverse marsh plant community can trap and store nitrogen and phosphorus that contribute to lake eutrophication.

Drought conditions in 2003 helped to re-establish some of the emergent plant communities of the marsh, but the recent return to wet conditions will make this reversal short-lived. It is believed that the restoration of vegetation in Netley-Libau Marsh would require a sustained period of low water, perhaps as much as two feet, over a period of two or more years. The current management of Lake Winnipeg for hydro-electric power production precludes low water levels. Consultations with stakeholders around Lake Winnipeg have shown widespread support for restoration of marsh vegetation but the implications for electricity generating potential need further consideration.

Netley-Libau Marsh resembles a shallow turbid lake more than a healthy coastal wetland. Many of the ecological goods and services benefits that the marsh could provide as wildlife and fisheries habitat, and in removing and storing nutrients that would otherwise enrich the lake, have been degraded or lost. Many of these benefits can be revitalized through marsh restoration and management. Several partners are working as part of a new Provincial Wetlands Working Group focussed on Netley-Libau and Delta Marsh, which clearly emphasizes the commitment and interest for the research, future management, and rehabilitation of these critical coastal wetlands.

4.0 CLIMATE AND HYDROLOGY: 1999 to 2007

(Lucie Lévesque)

Climate and hydrology vary widely throughout the Lake Winnipeg watershed. For the purposes of this report, air temperatures, precipitation, tributary discharges, and water levels from 1999 to 2007 are reported for the lake itself. Though climatic and hydrologic conditions throughout the Lake Winnipeg watershed affect the lake, the tributaries in the immediate vicinity provide a good indication of the potential drivers of aquatic chemistry and biology in the lake and the variability in these over recent years.

4.1 Air Temperature

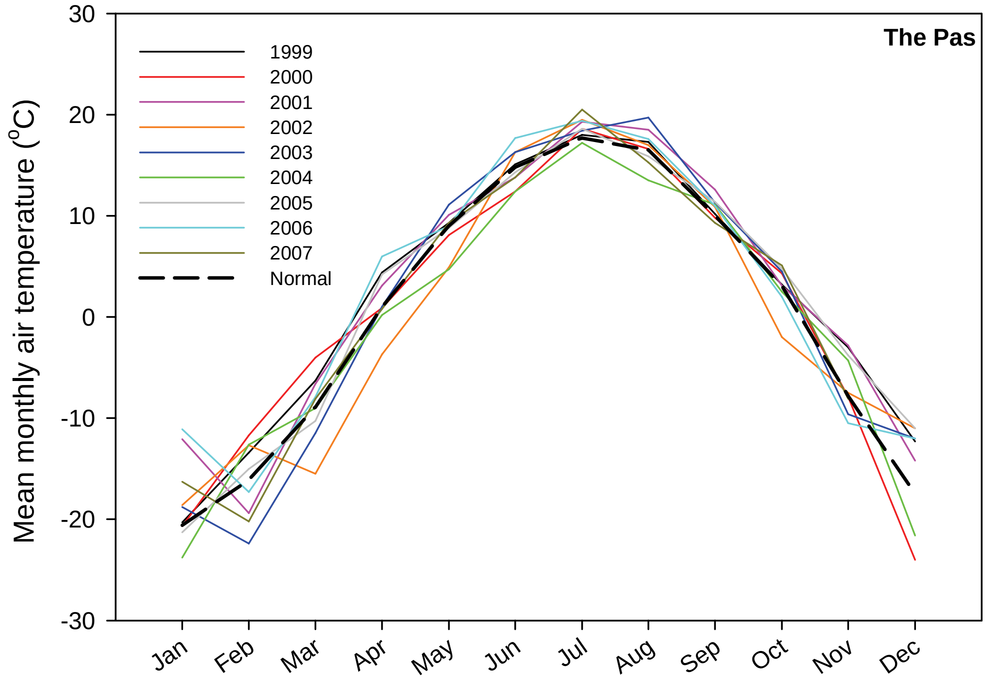
Long-term meteorological measurements are taken at two locations in the vicinity of Lake Winnipeg. These stations are located at The Pas, approximately 155 km upstream of the north basin of Lake Winnipeg, and at Gimli near the south end of the lake. Mean air temperature for the period of 1999 to 2007 was 0.8°C at The Pas, nearly two degrees lower than that at Gimli (2.5°C). At The Pas, temperatures ranged from a maximum of 37.6°C in 2002 to a minimum of -40.4°C in 2007. The maximum and minimum temperatures at the southern end of the lake were not substantially different (Gimli: 33.8°C to -40.6°C). Mean annual temperatures over the nine-year period were lowest in 2002 and 2004 (Table 4.1), close to a full degree below normal at both The Pas and Gimli (-0.2°C and -0.8°C, respectively at The Pas; 1.5°C and 0.9°C, respectively at Gimli). Mean annual air temperatures were highest at The Pas in 2001 and 2006 at 2.1°C and 2.0°C, respectively, close to two full degrees higher than normal. The highest mean annual temperatures at Gimli occurred in 1999 (3.6°C) and 2006 (3.7°C), nearly two degrees higher than normal.

Air temperatures near Lake Winnipeg vary seasonally, peaking typically in July or August and reaching their lowest in January or February (Figure 4.1). Mean monthly air temperatures at both The Pas and Gimli were, in most years from 1999 to 2007, higher than the 1971 to 2000 normal in early winter and below normal later in the winter, particularly in the month of February. Summer and fall air temperatures during that period were typically near normal. In 1999, 2001, and 2006, when mean annual air temperatures were highest, normals were exceeded most frequently and by the greatest magnitude over the winter and spring months (between 1.0°C and 9.5°C above normal at The Pas; between 4.1°C and 9.2°C above normal at Gimli). Projections of future air temperatures indicate that above normal temperatures in the spring will continue over the next century, lengthening the open water season and concomitantly resulting in rising water temperatures in Lake Winnipeg (Section 9.5).

Table 4.1: *Annual mean daily air temperature (°C) near Lake Winnipeg. (Data Source: Environment Canada)*

Year	Gimli	The Pas
1999	3.6	1.9
2000	1.8	0.2
2001	3.0	2.1
2002	1.5	-0.2
2003	2.1	0.7
2004	0.9	-0.8
2005	2.6	1.4
2006	3.7	2.0
2007	3.4	0.3
Normal 1971 to 2000	1.8	0.1

The effects of rising water temperatures on the Lake Winnipeg ecosystem (e.g., bioproductivity, biogeochemical processing, water balance) are not fully understood. In 2002 and 2004, when mean annual temperatures were below normal, mean monthly air temperatures from late winter through spring and over late fall and early winter were well below normal, by as much as 6.6°C at The Pas, and 5.9°C at Gimli. In contrast to the years with above normal temperatures, lower than normal air temperatures in the shoulder seasons may be



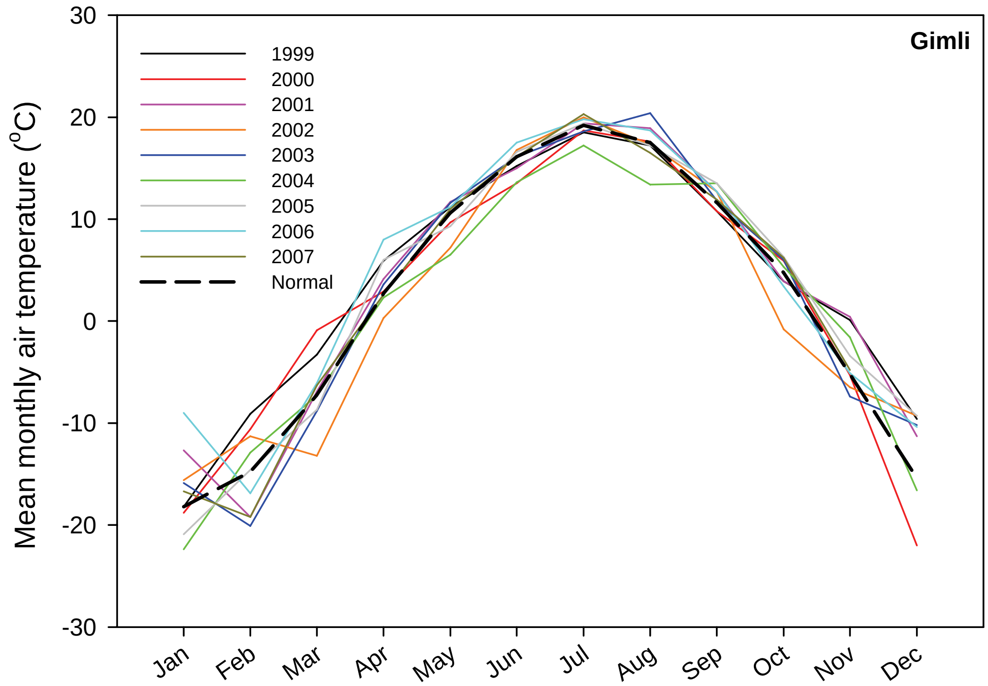


Figure 4.1: *Mean monthly air temperature at The Pas and Gimli, Manitoba, 1999 to 2007. Normal mean monthly temperature from 1971 to 2000. Note deviations from normal in the shoulder seasons. (Data source: Environment Canada Atmospheric Environment Service, http://www.climate.weatheroffice.ec.gc.ca, Gimli Station ID 5031038, lat. 50° 37.800' N long. 97° 1.200' W, The Pas Station ID 5052880, lat. 53° 58.200' N long. 101° 6.000' W)*

expected to abbreviate the open water season, promoting lower water temperatures and lesser rates of biological productivity and nutrient cycling in the lake. Below normal temperatures persisted over the warm season at both The Pas and Gimli in 2004; mean monthly air temperatures were as much as 4.0ºC below their respective normals (Figure 4.1).

4.2 Precipitation

Precipitation on Lake Winnipeg is measured at Environment Canada climate stations located on, and in the immediate vicinity of, the lake. Data from as many as 12 of these stations were used to calculate total annual and monthly precipitation from 1999 to 2007 (Appendix 2). Total annual precipitation on Lake Winnipeg from 1999 to 2007 varied between a low of 275 mm in 2006 and a high of 576 mm in 2004 (Table 4.2, Figure 4.2). Precipitation was at or below the 1971 to 2000 normal of 488 mm in most years. The 2001 drought that extended through much of the Lake Winnipeg watershed, and which lasted until as late as 2003 in the prairies, resulted in soil moisture deficits and reduced flows in tributaries to the lake (St. George 2006; Canadian Council of Ministers of the Environment 2003). Elevated precipitation in 2004 and 2005 promoted recovery from this drought.

Precipitation in the vicinity of Lake Winnipeg occurs primarily as rainfall, peaking in magnitude over the warm season. The cold season is characterized by lower monthly precipitation, which occurs as snowfall. Mean monthly precipitation for Lake Winnipeg from 1999 to 2007 was greatest May through October, peaking in June at 69.3 mm (±25.3 mm), and lowest January through April, dropping to 11.4 mm (±3.8 mm) in February (Figure 4.3). Data from meteorological stations at Gimli and The Pas indicate that snowfall represents

***Table 4.2** Total precipitation (mm) estimated for Lake Winnipeg (as per McCullough and Yerubandi, Appendix 2). Normal = 487.6 (sum of monthly normals for The Pas and Gimli, Environment Canada stations)*

Year	North Basin	South Basin	Lake Average
1999	295	105	400
2000	382	136	518
2001	355	126	481
2002	366	130	497
2003	286	102	387
2004	425	151	576
2005	400	142	543
2006	203	72	275
2007	274	97	371

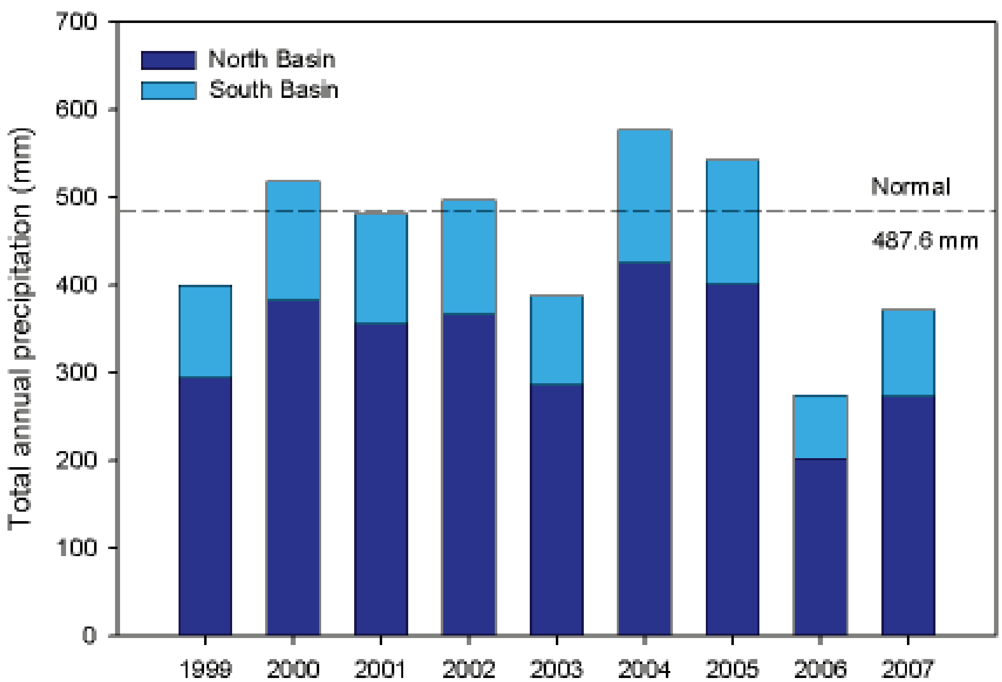


***Figure 4.2:** Total annual precipitation on Lake Winnipeg 1999 to 2007. Total annual precipitation normal calculated from the average monthly normals at Gimli and The Pas, 1971 to 2000. (Data source: Environment Canada http://www.climate.weatheroffice.ec.gc.ca)*

between approximately 15 and 20 % of total annual precipitation near southern Lake Winnipeg and up to twice that toward the north in some years. Similarly, precipitation through much of the Lake Winnipeg watershed is dominated by rainfall, contributing to high magnitudes and rates of runoff to water bodies in the summer in addition to that associated with spring snowmelt. Normal precipitation for The Pas and Gimli indicate that mean monthly precipitation is typically of greater magnitude and lower intra-annual variability toward the south than the north (Figure 4.3).

Currently, evaporation measurements are not taken on Lake Winnipeg. However, estimates for Lake Winnipeg have been made based on evaporation rates from nearby Dauphin Lake (as per G. McCullough, Appendix 2). Water balance estimates for Lake Winnipeg indicate that losses through evaporation are greatest from May to September (Figure 4.3). Wind-generated waves and high air temperatures over the ice-free period promote elevated rates of evaporation, estimated to reach greater than 130 mm per month in July and August. In mid-winter, gain to the water balance may occur when condensation under cold conditions could contribute to exchange of water between the lake and the atmosphere.

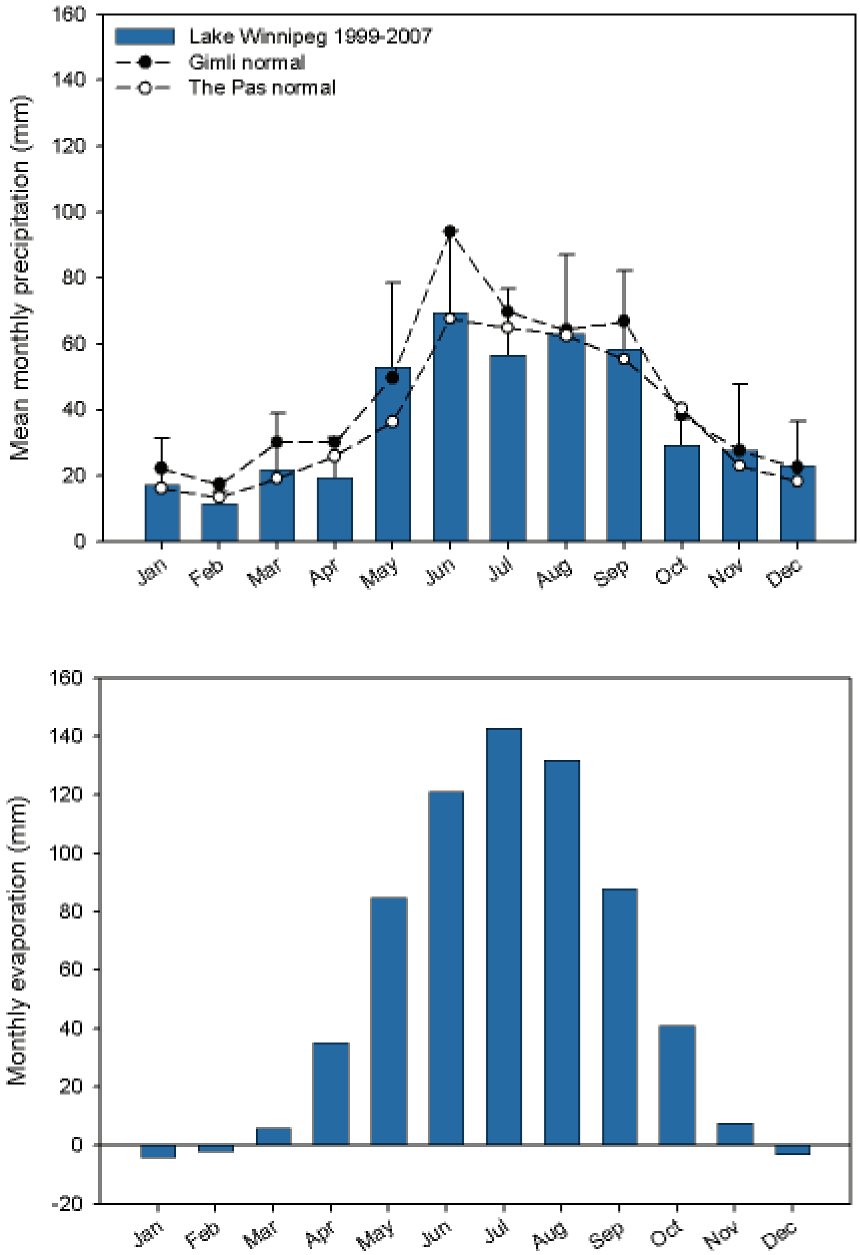


Figure 4.3: *Mean monthly precipitation on Lake Winnipeg (1999 to 2007) and estimated monthly evaporation from the lake. Mean monthly precipitation normals for The Pas and Gimli from 1971 to 2000. Inputs to Lake Winnipeg include not only precipitation but tributary flows as well; evaporation therefore may exceed precipitation. Negative values of evaporation indicate gains to the water balance via condensation. (Data source: Environment Canada http://www.climate.weatheroffice.ec.gc.ca, as per G. McCullough, Appendix 2)*

4.3 Hydrology

4.3.1 Tributary Discharges

Discharges through tributaries are the most significant sources of water, sediments (Section 5.3), and nutrients (Sections 6.0 and 7.0) to Lake Winnipeg. Originating from within the catchment, these discharges vary in magnitude inter-annually and seasonally, affecting the quantity and quality of waters in Lake Winnipeg, and ultimately the structure and function of the aquatic ecosystem. Mean monthly discharges to and from Lake Winnipeg were determined for the major inflows to, and the outflow from, Lake Winnipeg, as well as for unmetered streams draining to the lake (as per G. McCullough; Appendix 3). Over the period from 1999 to 2007, the Winnipeg River accounted for 49 % of inflow to Lake Winnipeg, the Saskatchewan River 25 %, the Red River 16 %, unmetered flows 6 %, and the Dauphin River 4 % (Figure 4.4). Relative to inflows from the 1964 to 2005 period of record, the Winnipeg and Red rivers contribute proportionately more to the total inflow to Lake Winnipeg from 1999 to 2007 than in earlier years (Winnipeg River 45 %, Red River 11 %; Lake Winnipeg Stewardship Board 2006).

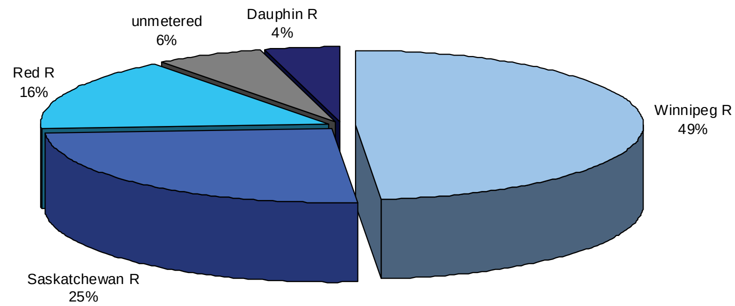


***Figure 4.4:** Mean monthly discharge into Lake Winnipeg as proportion of total tributary discharges from 1999 to 2007. (Data Source: Water Survey of Canada, G. McCullough)*

Rivers draining into the north basin of Lake Winnipeg represent nearly the same areal proportion of the Lake Winnipeg watershed as those draining to the south basin (Figure 1.1). Discharges from the former, however, represented only 29 to 36 % of total discharges to the lake from 1999 to 2007. Those draining into the south basin represented between 64 and 71 % of discharges to the lake. The lower overall discharge from large rivers to the north basin relative to the south basin may be attributed to the extensive distribution of non-contributing areas and internal drainage through the Saskatchewan River sub-watershed (Figure 1.1). Reduced contributions of discharges from this catchment to Lake Winnipeg's overall inflow were particularly noteworthy during the drought in the early 2000s.

The overall mean monthly inflow to Lake Winnipeg through tributaries from 1999 to 2007 was 2,586 m^3/s (Table 4.3). The lowest mean monthly discharge into the lake occurred in 2003 (537 m^3/s; Figure 4.5), following below normal precipitation (Figure 4.2) and drought conditions in the watershed. The greatest discharge into the lake occurred in 2005 (6,854 m^3/s), when precipitation was near its greatest and the Lake Winnipeg watershed had recovered from the preceding drought.

***Table 4.3:** Tributary discharge (m^3/s) to and from Lake Winnipeg, 1999 to 2007. (Data Sources: Water Survey of Canada; G. McCullough).*

River	Mean	Max	Min
All rivers	2586	6854	537
Winnipeg River	1064	2506	275
Red River	346	1985	22
Saskatchewan River	556	1512	42
Dauphin River	92	262	15
unmetered rivers (south basin)	127	318	32
unmetered rivers (north basin)	136	314	33
Nelson River (outflow)	2380	4665	765

In studying the water balance of Lake Winnipeg from 1996 to 2005, G. McCullough (Fisheries and Oceans Canada, unpublished data) determined that river discharge to Lake Winnipeg accounted for 88 % of inputs to the water balance of the lake, with precipitation accounting for 12 %. River discharge from the lake through the Nelson River accounted for 82 % of losses to the water balance, with the other 18 % attributable to evaporation. The contribution of groundwater to the water balance of the lake is not known.

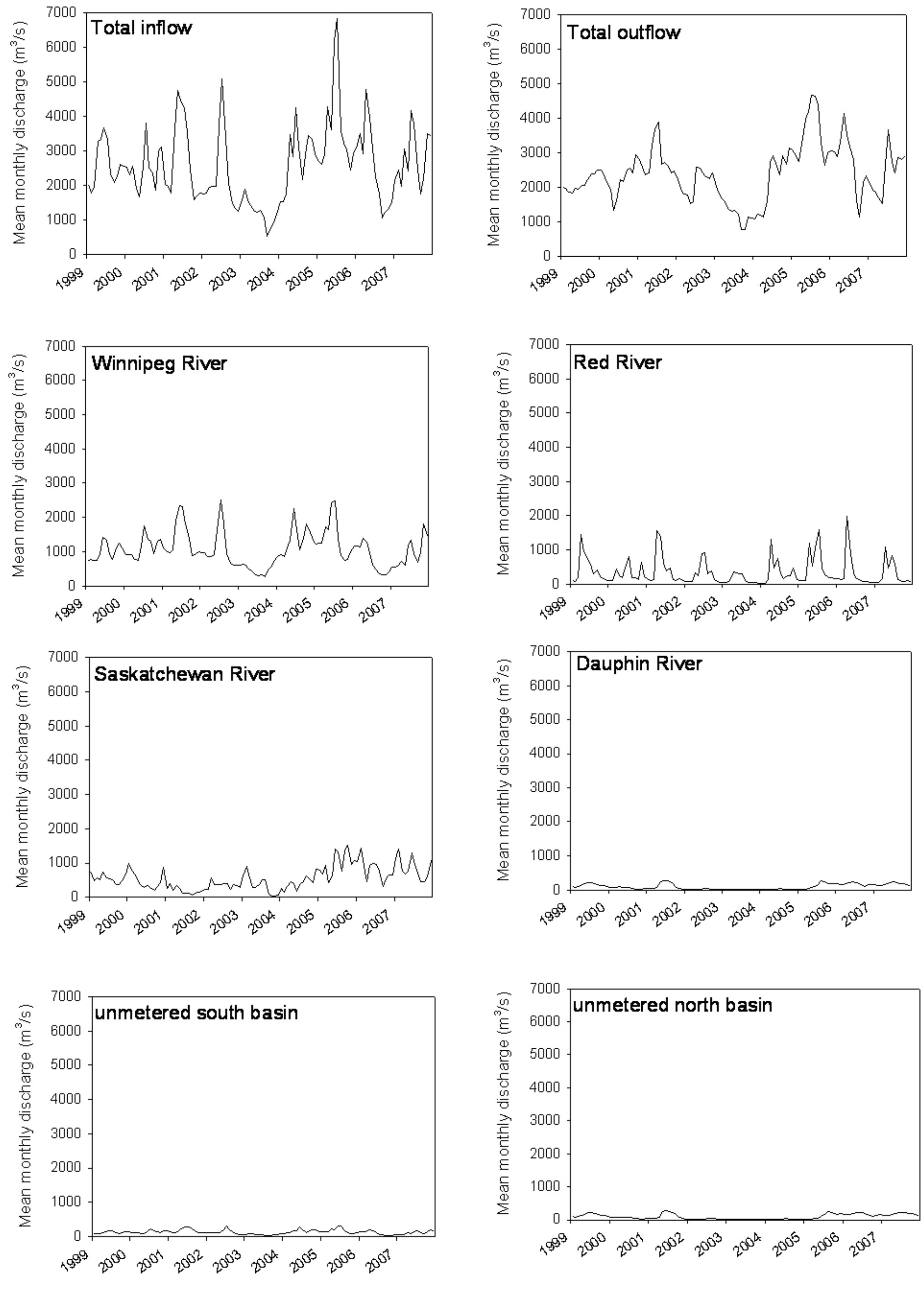


Figure 4.5: *Mean monthly discharges to and from Lake Winnipeg 1999 to 2007. (Data Source: Water Survey of Canada; G. McCullough)*

Winnipeg River

Mean monthly discharge from the Winnipeg River to Lake Winnipeg from 1999 to 2007 was 1,064 m^3/s. The lowest discharge occurred in 2003 (Table 4.3, Figure 4.5), associated with a period of drought during which above normal temperatures and increased spring and summer evaporation reduced runoff in the watershed (St. George 2006). Discharges from the Winnipeg River were high relative to other rivers flowing to Lake Winnipeg, with a maximum discharge from 1999 to 2007 of 2,506 m^3/s, nearly 1,000 m^3/s greater than that of the Saskatchewan River and more than 500 m^3/s greater than that of the Red River (Table 4.3). The dominance of Precambrian Shield in the Winnipeg River watershed contributes to lower soil moisture storage, hence greater runoff and tributary discharge than in sedimentary watersheds where runoff and discharge are highly dependent upon antecedent moisture conditions and depression storage (Section 3.2). In addition, the more resistant nature of the terrain in the Winnipeg River watershed accounts for typically low sediment and nutrient loads to Lake Winnipeg from this river relative to other rivers, such as the Red (Section 7.0).

Red River

Mean monthly discharge from the Red River to Lake Winnipeg from 1999 to 2007 was 346 m^3/s. With the exception of 2003, when mean monthly discharge was at its lowest for the period (144 m^3/s), discharges peaked dramatically in spring and summer nearly every year (Figure 4.5). Spring floods, and the high sediment and nutrient loads that they carry to Lake Winnipeg, are common occurrences on the Red River (Sections 3.2 and 7.0). The absence of high discharges in 2003 may be attributed to the preceding drought and low antecedent moisture conditions in the watershed, as well as below normal precipitation in 2003 (Figure 4.2). The substantive extent of non-contributing area in the Assiniboine River watershed (Figure 1.1) suggests that drought conditions would generate large deficits in moisture and reduce runoff to the Red River. This would also account for a mean discharge in the Red River from 1999 to 2007 that is approximately one-third that from the smaller Winnipeg River watershed (Table 4.3).

Spring floods, and the high sediment and nutrient loads that they carry to Lake Winnipeg, are common occurrences on the Red River.

Saskatchewan River

Mean monthly discharge from the Saskatchewan River to Lake Winnipeg from 1999 to 2007 was 556 m^3/s. Much like the Red River, the Saskatchewan River had its lowest discharges during the drought in 2003 and its highest discharges after the drought (Figure 4.5, Table 4.3). The wide extent of non-contributing areas in the Saskatchewan River watershed relative to watersheds of other large rivers draining to Lake Winnipeg (Figure 1.1) suggests that this river would be more susceptible to reduced flows under conditions of below normal precipitation and drought (e.g., 2001 through 2003 in Figure 4.5). Substantive deficits in soil moisture and reductions in contributing areas under drought conditions would delay recovery of high discharges, as observed in 2004. High discharges followed from 2005 to 2007, with which would be expected greater sediment and nutrient-loading to Lake Winnipeg (Sections 5.3 and 7.0).

Dauphin River and unmetered flows

The lowest discharges to Lake Winnipeg from 1999 to 2007 came from the Lake Winnipeg sub-watershed (Figure 1.1). Mean monthly discharge from the Dauphin River to Lake Winnipeg from 1999 to 2007 was the lowest of all the larger tributaries at 92 m^3/s (Table 4.3). Lower discharges in the Dauphin River are attributable to the smaller size of the watershed relative to the Winnipeg, Red, and Saskatchewan rivers, as well as the storage of runoff in Lake Winnipegosis and Lake Manitoba. As with the Saskatchewan River, discharges in the Dauphin River were greatest from

2005 to 2007, and lowest in 2003 and 2004 (Figure 4.5). Mean monthly discharge from unmetered rivers flowing into the south and north basins were estimated at 127 and 136 m³/s, respectively. Discharge estimates were based on rates of runoff in the metered portion of the catchment (Appendix 3). Therefore, inter-annual variation in discharges for the smaller, unmetered rivers reflect that of the larger rivers (e.g., reduced discharge from small rivers flowing to north basin from 2002 through 2004; Figure 4.5). More intensive monitoring of these smaller rivers is necessary to verify the magnitude and variability in discharges, as well as the quality of these waters (e.g., nutrient-loading, Section 7.0).

More intensive monitoring of the smaller, unmetered rivers is necessary to verify the magnitude and variability in discharges, as well as the quality of these waters.

Nelson River

Outflow from the lake through the Nelson River from 1999 to 2007 averaged 2,380 m³/s and, similar to inflows, outflows were lowest in 2003 and highest in 2005 (Table 4.3, Figure 4.5). The Nelson River is the only outlet from Lake Winnipeg and has been regulated for hydro-electric power generation since 1976. As a part of lake regulation, several channels and dikes were constructed at the north end of Lake Winnipeg to increase the outflow capacity of the lake and to divert a greater amount of water to the Jenpeg generating station along the west channel of the Nelson River. Channels constructed as a part of Lake Winnipeg regulation have increased the outflow capacity of the Nelson River by 40 to 50 % and have reduced the magnitude and frequency of flooding on Lake Winnipeg by reducing the natural extremes in water levels. Prior to lake regulation, outflow was typically greatest in the spring and early summer and lowest in the winter. However, since regulation, the seasonal outflow pattern can been reversed with greater outflow typically occurring during the fall and winter (Figure 4.5) during periods of peak hydro-electric power demand. However, from 1999 to 2007 there were many wet years with high inflows to Lake Winnipeg causing a more natural seasonal outflow pattern.

4.3.2 Lake Water Levels

Water levels in Lake Winnipeg are a function of tributary discharges to the lake as well as wind. Water levels in the north basin of the lake are slightly lower than those in the south basin, but vary in a similar manner seasonally and inter-annually. This variability is largely due to variation in tributary discharges. Water levels for Lake Winnipeg were calculated with data from Water Survey of Canada hydrometric stations (as per McCullough, Appendix 4). Water levels varied from a low of 216.98 masl in the north basin and 216.91 masl in the south basin in 2003, to a high of 218.41 masl in the north basin and 218.39 masl in the south basin in 2005 (Figure

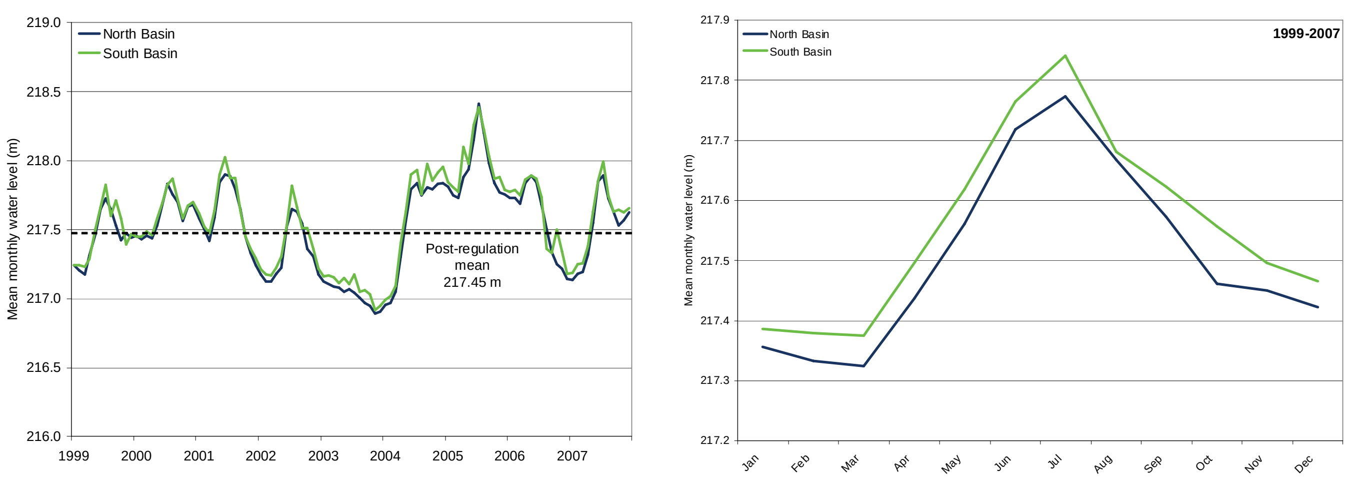


Figure 4.6: *Mean monthly and seasonal water levels from 1999 to 2007. (Data Source: Water Survey of Canada, G. McCullough)*

4.6). These years correspond with below and above normal precipitation on Lake Winnipeg (Figure 4.2). Water levels from 1999 to 2007 were higher and more variable in the south basin (range = 1.48 m) than the north basin (range = 1.43 m; Figure 4.6). Larger variability and height is likely related to the greater magnitude and variability, as well as extremes in peak flows in those rivers flowing to the south basin, versus those that flow to the north basin (Figure 4.5).

Water levels in both basins were below the 1976 to 2007 mean of 217.45 masl for much of 2003, when tributary discharges were low (Figure 4.5). Levels recovered through 2004 under cooler, wet summer conditions and rising inflows from most of the rivers draining to the lake. High water levels persisted through 2005 and much of 2006 (Figure 4.6). Seasonally, mean monthly water levels in Lake Winnipeg from 1999 to 2007 were highest in July (217.77 masl north basin, 217.84 masl south basin) after the height of the wet season, and lowest in March (217.32 masl north basin, 217.37 masl south basin) near the end of the ice-covered season. Regulation of flows into and out of Lake Winnipeg has reduced the natural extremes in highs and lows (Brunskill *et al.* 1980; Figure 4.6). In addition to the magnitude and frequency of inflows to and outflow from the lake, wind also affects water levels in Lake Winnipeg. "Wind-setup" or "storm surges" may raise the water level by 0.6 to 1.2 m, particularly in the fall and early winter (Baird and Stantec 2000).

The hydraulic retention times for Lake Winnipeg over the period of 1999 to 2007 averaged 3.5 years for the lake as a whole, 1.3 years for the south basin, and 4.3 years for the north basin.

4.3.3 Lake Water Residence Time

Water residence times in Lake Winnipeg have implications for the storage and release of nutrients and contaminants. Environment Canada examined the morphologic and hydrologic conditions of the lake from 1999 to 2007 (as per R. Yerubandi and W. Zhang, Appendix 5). Based on these data and the model, the hydraulic retention times for Lake Winnipeg over the period of 1999 to 2007 averaged 4.3 years for the lake as a whole, 1.3 years for the south basin, and 3.5 years for the north basin (Figure 4.7). Water residence time was longest in 2003 (whole lake: 7.0 years, south basin: 2.3 years, north basin: 5.8 years), when tributary discharges were at their lowest (1,109 m^3/s relative to an average of 2,586 m^3/s for the period of 1999 to 2007, Figure 4.5). The rate of flushing in Lake Winnipeg could be expected to have been lower in 2003, promoting nutrient retention. Shorter water residence times in the south basin than the north basin are likely attributable to the higher rates of tributary flows to the smaller south basin as compared to the larger north basin (i.e., Winnipeg and Red rivers, Table 4.3).

Water residence times as calculated by Yerubandi and Zhang for 1999 to 2007 are longer than those calculated by Brunskill *et al*. (1980) for the period of 1969 to 1974. From 1969 to 1974, water residence times for the south basin were between 0.43 and 0.83 years (whole lake range: 2.9 to 4.3 years); water residence times for 1999 to 2007 varied between 0.9 and 2.3 years for the south basin (whole lake range: 2.7 to 7.0 years), with residence times of one year or greater predominating in the 2000s.

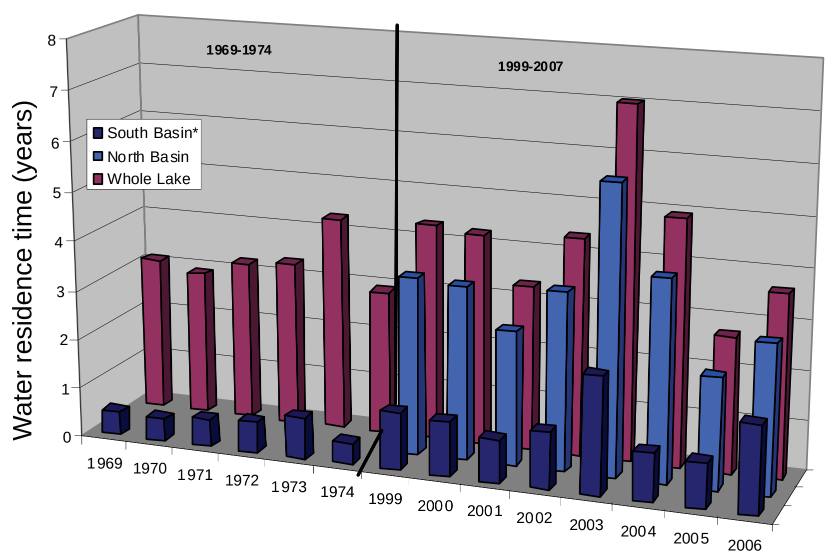


Figure 4.7: *Water residence time for Lake Winnipeg, 1969 to 1974 and 1999 to 2007. *defined as south basin and narrows. (Sources: Brunskill et al. 1980; Yerubandi and Zhang, unpublished; water level original data source: Water Survey of Canada, http://www.wsc.ec.gc.ca)*

5.0 PHYSICAL CHARACTERISTICS OF LAKE WINNIPEG WATERS

5.1 Thermal Regime

(Greg McCullough, Lucie Lévesque)

The thermal regime of a lake is controlled by the exchange of heat between the atmosphere and the water column (Wetzel 2001). There is also heat exchange between the lower water column and bottom sediments, but during the open water season at least, these transfers are small compared to the exchange at the water surface. In late winter and spring, solar radiation and convective atmospheric heating eventually melt the winter ice cover. In late summer and autumn, upwelling thermal radiation, and convective and evaporative cooling gradually lower the surface temperature until the lake again freezes over. Through the intervening open water period, the lake is warmed primarily by solar radiation being absorbed and converted into heat in the water column. Solar radiation heats clearer waters to a greater depth than murky waters, but without further disturbance, this process would in either case establish a smooth gradient from warmest at the surface to coolest below some maximum depth of light penetration. However, turbulent energy supplied by the wind does disturb this simple gradient, mixing warmer shallow water with cooler deep water to create an upper layer, or epilimnion, of nearly homogeneous temperature. Often the bottom of this layer is marked by a sharp decrease in temperature separating the epilimnion from cooler bottom water - the hypolimnion. Where the temperature gradient between the two layers is greater than 1.0°C/m, it is called a thermocline (as defined by Wetzel 2001).

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 biota in the lake.

The thermal regime of a lake has implications for the structure and function of its aquatic ecosystem. 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 biota in the lake (Wetzel 2001). Consequently, the rate of biological productivity is typically greater in warmer lakes, and will be responsive to climate-induced changes to the length of the open water season (Section 9.5). In lakes where thermal stratification develops, aquatic biota and internal nutrient loading may be affected by depression of oxygen levels at depth. This has recently become a concern with respect to Lake Winnipeg (Section 5.2).

Lake Winnipeg extends across four degrees of latitude from just north of 50° N to just south of 54° N. Air temperature near the north end of the lake is cooler than near the south end by about 4.0°C in May and November and about 2.0°C in mid-summer. This, along with the greater depth and thermal mass of the north basin, contributes to differences in water temperatures and ice cover between the north and south basins of the lake, and affects water circulation in the lake (Text Box 3). At mid-summer, the north basin waters tend to be 2.0 to 3.0°C cooler than the south basin. Cooler temperatures in the north basin contribute to a delay in ice-cover melt and break-up relative to the south basin by two weeks, on average. However, the greater thermal mass of the deeper north basin retards cooling so that the two basins often freeze over within days of each other.

Differences in geography and morphometry between the north and south basins of Lake Winnipeg affect the vertical distribution of heat through the water column and contribute to its classification as a cold polymictic lake (Lewis 1983). The south basin is continuously polymictic - that is, it is ice-covered in winter and ice-free and above 4.0°C in summer, but rarely, if ever, thermally stratified on more than a daily basis. In most of the few years for which there are multi-season records of thermal profiles, the north basin also appears to have remained well-mixed, with at most, only weak stratification and no distinct thermocline even in the warmest summer months (Brunskill *et al.* 1980; Kristofferson *et al.* 1975; Pollard 1973; Rybicki 1966; Bajkov 1930). However, in two years of recent record, thermoclines have been recorded in mid-summer at the 12 to 15 m depth. In summer 2003, the presence of a thermocline at stations in the north basin was associated with the depression of oxygen levels (Stainton 2005; Sections 5.2 and 8.2). This recent observation of thermoclines in the north basin indicates that this part of Lake Winnipeg should be considered discontinuously polymictic in at least some years.

Water temperature in Lake Winnipeg has been measured in recent years by Manitoba Water Stewardship, Fisheries and Oceans Canada, and Environment Canada. Seasonal variability in water temperatures and associated vertical thermal profiles, including those associated with the thaw and melt of ice as well as under ice, have been monitored and investigated extensively in recent years by Fisheries and Oceans Canada and Environment Canada in association with dissolved oxygen measurement and hydrodynamic modelling. Their data have been extended by modelling historic and future thermal conditions in Lake Winnipeg (McCullough 2005, Section 9.5).

5.1.1 Recent Conditions: 1999 to 2007

Water temperatures at long-term stations monitored by Manitoba Water Stewardship were recorded during the spring (May-June), summer (July-August), and fall (September-October) for the period from 1999 to 2007 (Appendix 1, Table 5.1). Survey dates varied between years, which may affect comparability in mean seasonal water temperatures. In particular, during spring and fall surveys, a week's difference in the mean sample date may account for a greater difference in temperatures between two years than would real inter-annual variability. Consequently, only mid-summer survey data are discussed below (Figure 5.1, following page).

Table 5.1: *Surface and bottom water temperatures (°C) in Lake Winnipeg from 1999 to 2007. Summer mean. (Data Source: Manitoba Water Stewardship)*

Year	Surface		Bottom	
	North Basin	South Basin and Narrows	North Basin	South Basin and Narrows
1999				
2000				
2001		20.6		19.4
2002	18.9	21.0	18.2	
2003	20.3	21.5	15.7	21.1
2004	17.1			
2005	19.4	20.2	19.3	19.8
2006	20.1	22.7	18.9	21.3
2007	22.2	22.9	17.2	22.1
Average 1999 to 2007	19.7	21.5	17.9	20.8

Average near-surface water temperatures measured at Manitoba Water Stewardship stations from 1999 to 2007 were cooler in the north basin than the south basin during mid-summer surveys (Figure 5.1). Mean temperatures in the deeper north basin ranged from 17.0 to 22.0°C, including maximums slightly above 20.0°C in 2003 and 2006, and with the highest being 22.2°C in 2007. Mean water temperatures in the south basin ranged from 20.0 to 23.0°C, with maximums typically 1.0 to 2.0°C higher than in the north basin, and the highest again being in 2007. The coolest mid-summer air temperatures in the period occurred in 2004 (Figure 4.1), and were associated with the lowest mid-summer water temperatures (north basin: 17.1°C; south basin: data not available). Mean mid-summer water temperatures near the lake bottom each year were similar to those at the surface, as might be expected, given Lake Winnipeg typically does not thermally stratify. Mean mid-summer bottom water temperatures in the south basin ranged from 0.5 to 5.4°C higher than those in the north basin (Figure 5.1, Table 5.1).

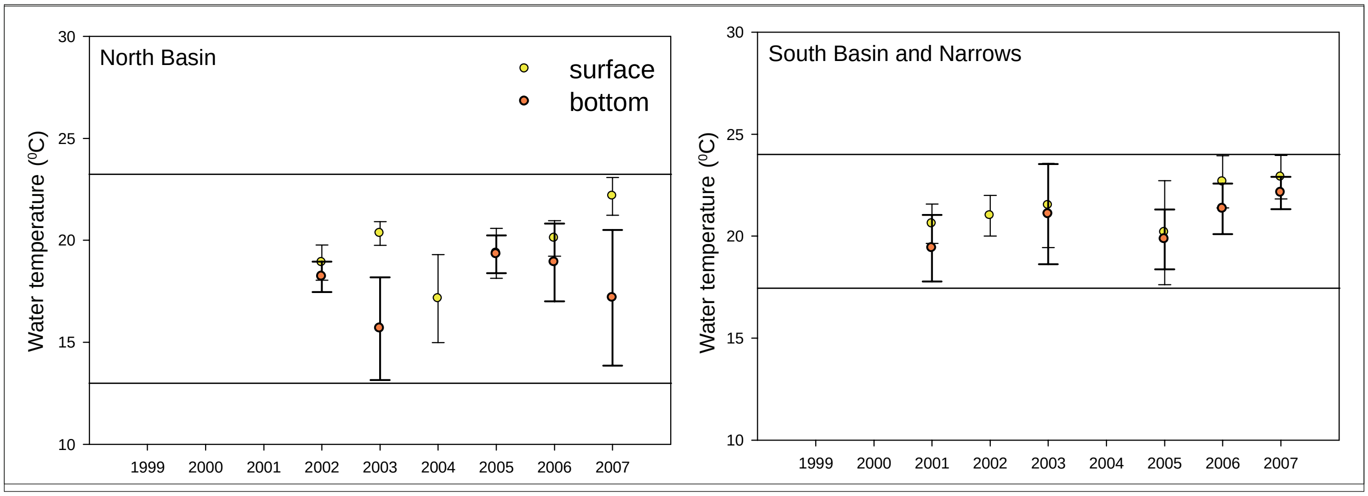


Figure 5.1: *Summer mean surface and bottom water temperature (± SE) in the north basin and south basin (and narrows) of Lake Winnipeg from 1999 to 2007.*

The difficulty with comparing seasonal water temperatures between years - due to variability in survey dates - can be remedied by modelling water temperatures from continuously recorded air temperature. Monthly surface water temperatures for Lake Winnipeg have been modelled from the monthly air temperature record at the Environment Canada weather station at Berens River (Figure 5.2; Appendix 6; McCullough 2005). The model predicts June to October monthly mean surface water temperatures as calculated from hourly records from 1999 to 2006 (Figure 5.3) with root mean square errors of 0.7°C (n = 8) and 0.9°C (n = 25) in the north and south basins, respectively. The model results are consistent with measurements from 1999 to 2007, which indicate that the south basin is warmer than the north basin in all seasons (though less so in autumn) and that 2004 was the coolest recent year in both basins (for all months from June through September, Figure 5.2).

The model also indicates that mean monthly water temperatures from 1999 to 2007 were not markedly cooler or warmer than any other decade since the mid-20th century.

The model also indicates that from 1999 to 2007, monthly mean water temperatures were not markedly cooler or warmer than any other decade since the mid-20th century. Indeed, the study period contains the second coolest August (2004) and the third warmest August (2003) mean surface water temperature in the modelled record. There is no significant long-term trend through the 60-year modelled record, and although the record indicates an inter-decadal oscillation roughly consistent with the well-known prairie drought cycle, this inter-decadal variability is small compared to inter-annual variability (Figure 5.2). Although gaps in local temperature records might preclude extending this model to the early 20th century, evidence from meteorological stations in the region indicates that summers were warmer in the late 1930s than in recent decades (Figure 5.4). On the other hand, summers in the earliest years of the 20th century tended to be cooler. Modelled air temperature data correlate well with mid-summer surface water temperature in Lake Winnipeg (Figure 5.5; r^2 = 0.80, 0.93, and 0.94, n = 4, 6, and 6 at the north, centre, and south basin Environment Canada moorings respectively, P <0.0001 in all cases). The relationship between long-term surface water and air temperatures (at least 1:1, Figure 5.5) is such that for each degree warmer the summer, the lake is likewise warmer by a degree or more. That is, Lake Winnipeg's mid-summer water temperatures would have been lower in the first decade of the 20th century (on average by more than 1.0°C), and higher in the 1930s (by at least 0.5°C) than through any extended period since.

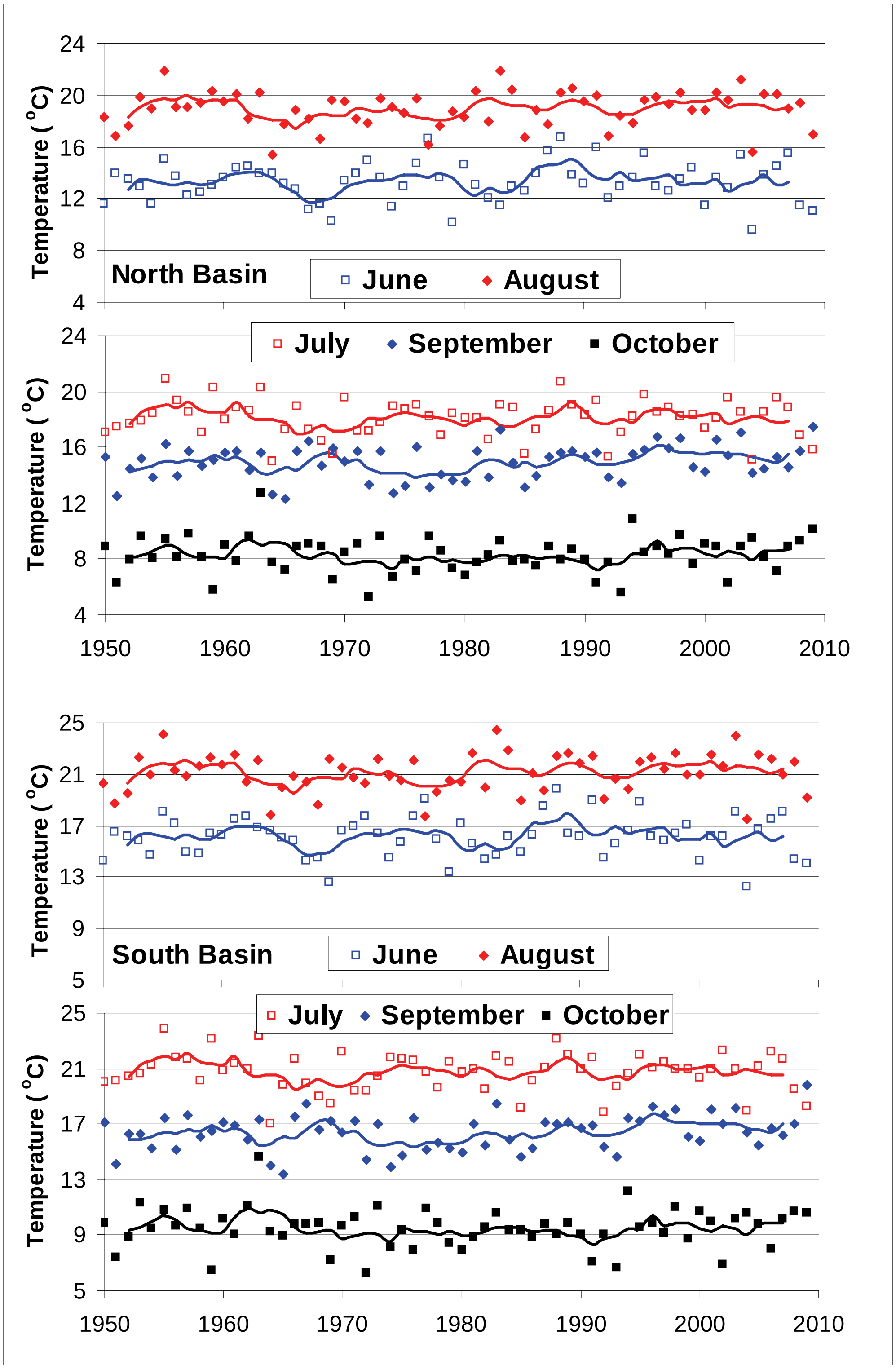


Figure 5.2: *Modelled monthly mean surface water temperature. Top panels = north basin, bottom panels = south basin. Lines are 5-year running means. (Data Source: see Appendix 6)*

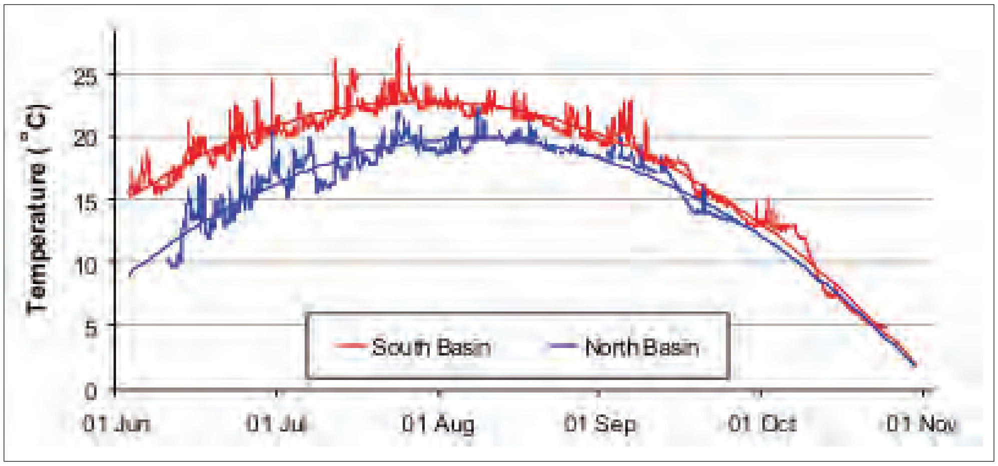


Figure 5.3: *Hourly air and surface water temperature records from Environment Canada moorings in the north and south basins of Lake Winnipeg, for the 2006 open water season. (Data Source: Environment Canada, B. Funk)*

McCormick and Fahnenstiel (1999) found 0.6 to 0.9°C temperature increases over the 20th century for surface waters in lakes Superior and Erie, respectively (using records at water intakes, 1906 to 1995 and 1918 to 1995 respectively; neither trend was statistically significant). If one considers the 1930s to be anomalous in the Lake Winnipeg record, then over the 20th century, the lake has probably warmed about as much as Lake Erie. It is instructive to note that McCormick and Fahnenstiel (1999) demonstrated that even this small temperature change has corresponded to a significant increase of 18 days in the period of summer stratification in Lake Erie since 1918.

Magnuson *et al.* (2000) found that on average, in the northern hemisphere (Europe, Asia, North America), spring break-up has advanced seven days per century and that fall freeze-up has shifted back six days per century over as much as 150 years of record.

Results of other studies suggest that the cool period recorded at the turn of the 20th century followed an even cooler 19th century. Wahl (1970) showed that summers in North Dakota were roughly 2.0°C cooler from 1850 to 1870 than from 1930 to 1960. Rannie (1983) showed that the median date of break-up on the Red River was twelve days earlier, and freeze-up ten days later in the 20th compared to the 19th century, and calculated that median spring and fall temperatures in southern Manitoba were 2.5°C lower in the 19th century.

In a global study, Magnuson *et al.* (2000) found that on average, in the northern hemisphere (Europe, Asia, North America), spring break-up has advanced seven days per century and that fall freeze-up has shifted back six days per century over as much as 150 years of record. They estimated that this corresponded to air temperature increases of 1.2°C per century. Wahl's (1970; summer) and Rannie's (1983; spring and fall) results indicate that climate in southern Manitoba and North Dakota has warmed more than this global average since the 19th century. Given the coherence of long-term temperature records among meteorological records at widely distributed stations in the Manitoba portion of the Lake Winnipeg watershed, it is likely that this higher than average rate of historic warming holds for the climate over Lake Winnipeg, and hence, for Lake Winnipeg itself. Warmer summers and longer open water seasons in the 20th century would have contributed more nutrient processing and higher bioproductivity than in the cooler 19th century, even in the absence of increased nutrient-loading. The same may be said for open water season warming, earlier ice break-up and later freeze-up projected for the 21st century (Section 9.5).

Long-term changes in temperatures have the potential to affect the length of the open water season and, therefore, water temperatures and levels of biological productivity and nutrient processing in lakes. However, historical records and modelling results for Lake Winnipeg indicate that the historic changes of 1.0 to 2.0°C through the 19th and 20th centuries are exceeded by inter-annual variability in the mid-summer temperature of the lake (Figure 5.4). Although changes in air and water temperatures over the long-term have the potential to alter the structure and function of Lake Winnipeg (Section 9.5), the frequency and magnitude of extremes (e.g., elevated water temperatures in the 1930s and 2004) may be more significant.

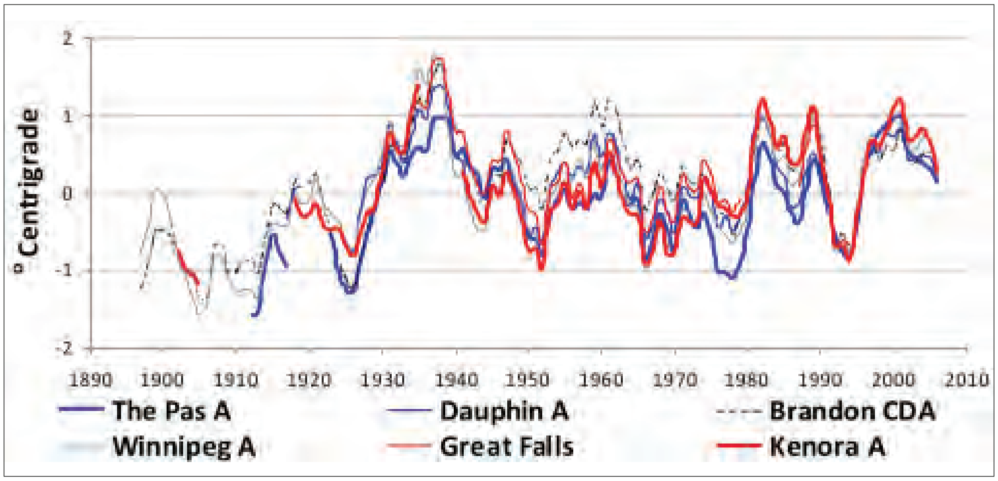


Figure 5.4: *Mean July-August air temperature at stations west, south, and southeast of Lake Winnipeg. Data are adjusted for historic changes in instruments, site locations, and operating practices (AHCCD 2010), normalized (expressed as the difference from the mean temperature from 1990 to 1999) and smoothed to five-year running means.*

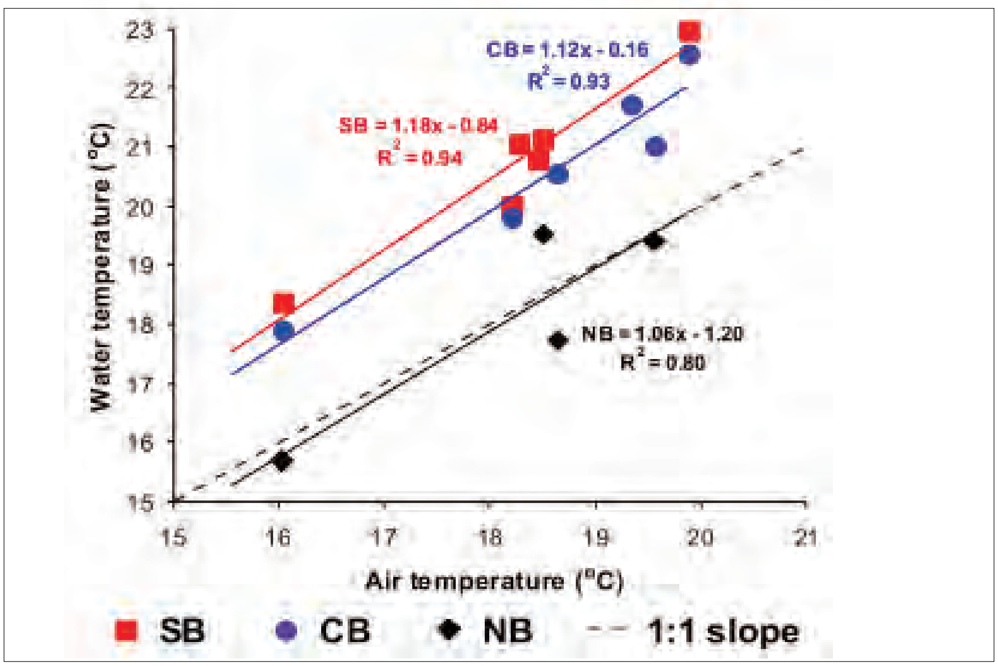


Figure 5.5: *Mean August surface water temperature recorded at Environment Canada moorings in the south basin (SB), the narrows (CB), and the north basin (NB) of Lake Winnipeg, shown as functions of the mean July-August air temperature recorded at the Environment Canada weather stations at The Pas and Kenora. The dashed line indicates a 1:1 relationship.*

5.1.2 Seasonal and Spatial Variability

Figure 5.6 is a series of satellite images showing seasonal variations in surface water temperatures in Lake Winnipeg. In late May, with remnants of ice in the north basin still remaining, average surface water temperature was 3.0 to 5.0°C, with slightly higher temperatures in the shallow shoreline waters along the northern and western portion of the basin and towards the narrows. Water temperatures in the ice-free south basin had reached more than 15.0°C by the same date. By mid-summer, both basins had warmed to over 20.0°C, but the north basin remained 2.0 to 3.0°C cooler than the south basin in late August. The differences in surface water temperatures between the north and south were no longer evident by the end of September, when temperatures of the two basins had for the most part drawn to within 1.0°C of each other (17.0 to 18.0°C, likely due to the faster rate of cooling of the shallow south basin). Shallow water in the north basin along the eastern shore and near the outlet of Dauphin River had cooled a few degrees further than the rest of the north basin. These patterns of cooling and convergence of north and south basin water temperatures by the end of the open water season (November) are similar to those recorded at Environment Canada moorings in 2006 (Figure 5.3). Although there are few temperature records available for later than the end of October in the north basin, this extrapolation to November shown in Figure 5.3 supports the observation that in spring, the north basin typically loses ice cover and warms later than the south basin, in the fall, the two basins typically freeze within days of each other.

Weather observer records for the south basin at Gimli indicate that the open water season from 1946 to 1991 has been, on average, 220 days from first melt in spring to complete ice cover in early winter.

The length of the open water season on Lake Winnipeg varies with the timing and duration of break-up and freeze-up. Weather observer records for the south basin at Gimli indicate that the open water season from 1946 to 1991 has been, on average, 220 days from first melt in spring to complete ice cover in early winter (on average April 21st to November 27th). However, due to the length of both break-up and freeze-up processes, the south basin of the lake was on average completely ice-free for only 180 days. No such record exists of break-up and freeze-up dates on the north basin. However, given that spring break-up typically occurs two weeks later in the north and that the ice cover re-forms at the same time or very soon after freeze-up in the south basin, the average open water season in the north basin may be

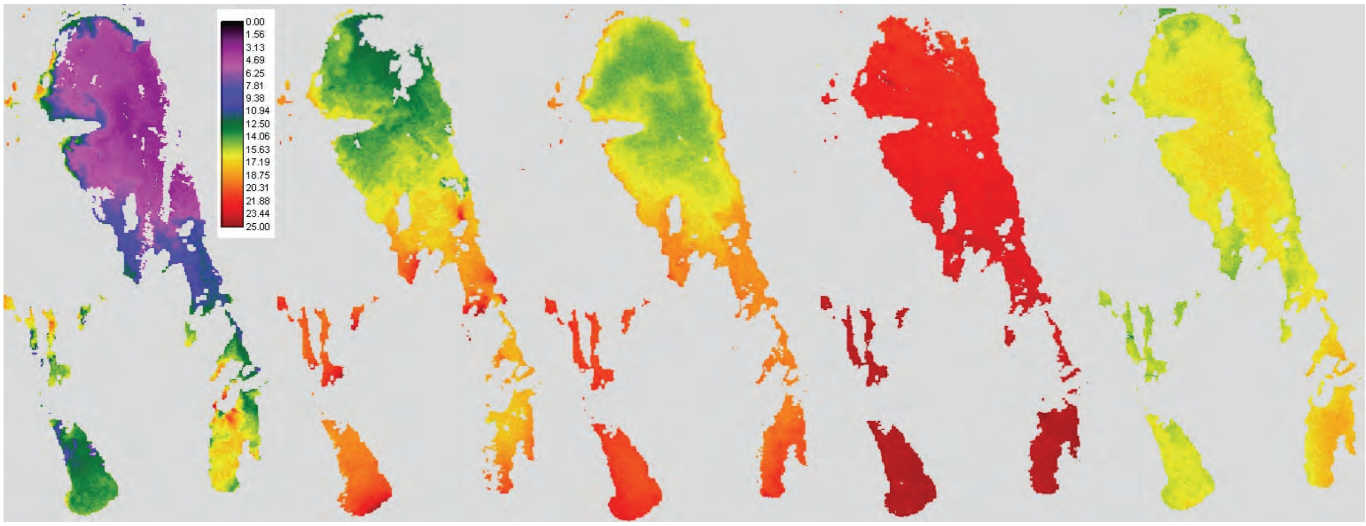


Figure 5.6: *Surface water temperature (0 = black, 25°C = red/brown) of Lakes Winnipeg and Manitoba on 26 May, 2 June, 6 July, 27 August and 30 September, 2001. Grey areas indicate land, cloud, or - in May - ice. (Data Source: AVHRR (U.S. National Oceanic and Atmospheric Administration's Advanced Very High Resolution Radiometer satellite data using general equations and have not been validated by comparison with local field surface temperature measurements)*

estimated to be roughly two weeks shorter than that in the south basin of the lake. Satellite images recorded from 1983 to 2009 indicate that break-up typically occurs more slowly in the north basin, spanning two to three weeks compared to one to two weeks in the south basin (Figure 5.7). Similar to the historic observations at Gimli, the satellite-derived record indicates the first melt in the south basin occurred as early as the second week in April with the last remnant ice occurring the first week in June. The earliest break-up in the satellite record was in the south basin in 2005 when the basin was completely ice-free by April 22nd. From 1983 to 2008, the north basin was on average about half clear of ice by May 19th (that is, two weeks later than the south basin, although the difference was as little as five days (1999) and as great as 22 days (2004)). Satellite images also confirm that freeze-up occurs in each basin within a few days of the other, sometime between mid-November and mid-December (Figure 5.7). Although frequent cloud and fog cover during the freeze-up period prevented as thorough an analysis with satellite data as for break-up, images from 2001 to 2008 show that mid-freeze-up (the date when the developing ice cover first extended over at least half of the basin) was never more than five days later in the north than in the south basin and the mean difference was less than two days.

Satellite images from 2001 to 2008 show that mid-freeze-up was never more than five days later in the north than in the south basin and the mean difference was less than two days.

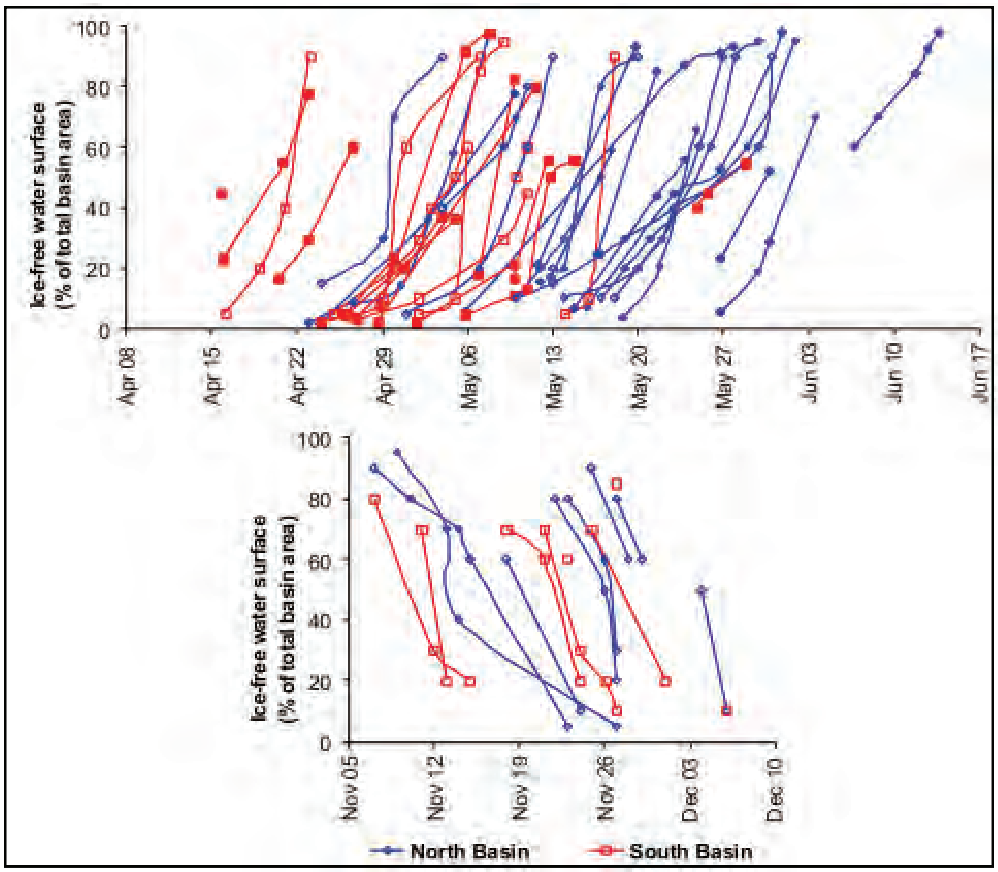


Figure 5.7: *Progression of melt and break-up (upper panel) and freeze-up (lower panel) of the ice cover on the north and south basins of Lake Winnipeg derived from satellite observations. The process is indicated as series of observations (% open water determined from individual images). Lines join same-year points, so that each curve indicates the progressive loss (gain) of ice cover through a single season. Open symbols indicate observations using MODIS (U.S. National Aeronautics and Space Administration's Moderate Resolution Imaging Spectroradiometer) data from 2001 to 2009; solid symbols (top panel only) indicate observations using AVHRR (U.S. National Oceanic and Atmospheric Administration's Advanced Very High Resolution Radiometer) data from 1987 to 2000.*

5.1.3 Water Temperature Stratification

Surface and bottom water temperatures in Lake Winnipeg are typically similar, reflecting the shallow, well-mixed nature of the lake, particularly during the open water season. Vertical temperature profiles for surface and bottom waters measured by Fisheries and Oceans Canada through the north and south basins in mid-summer from 1999 to 2008 indicate that at most stations and on most surveys, the whole water column had been mixed to within one or two degrees of the temperature at the surface (Figure 5.8 and, 2006 in Figure 5.9).

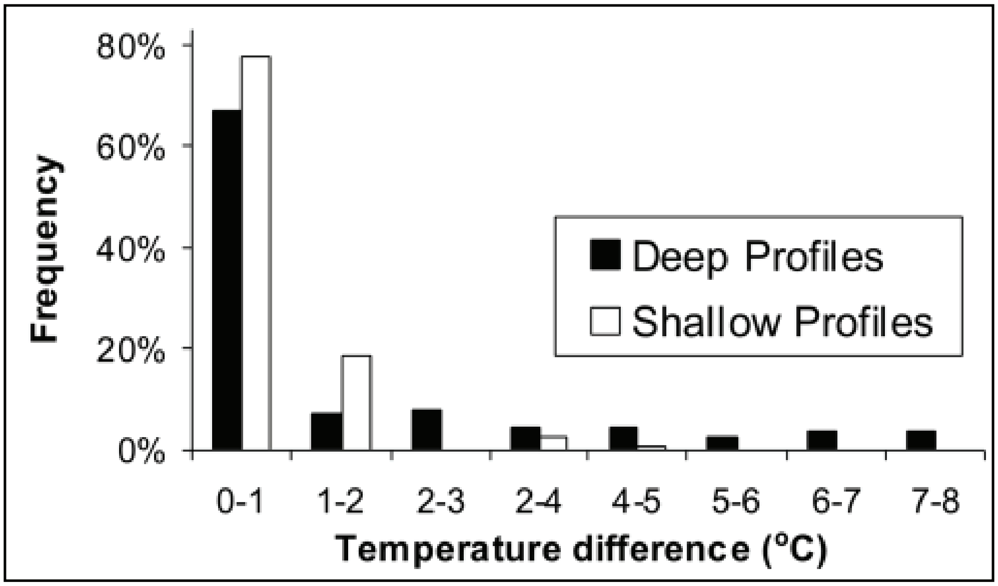


***Figure 5.8:** Difference between surface and bottom temperature in 170 deep profiles (depth >10 m) and 117 shallow profiles (depth <10 m) recorded on mid-summer surveys at stations throughout the north and south basins of Lake Winnipeg from 1999 to 2008. (Data Sources: Fisheries and Ocean Canada, M. Stainton; Environment Canada, L. Wassenaar)*

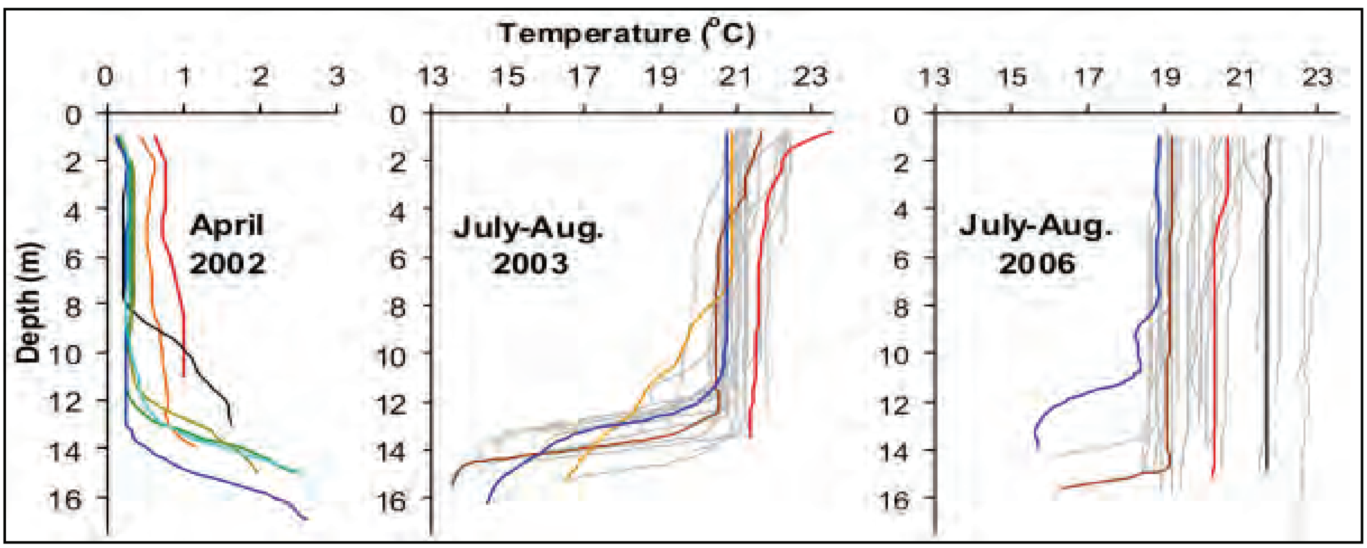


***Figure 5.9:** Selected thermal profiles in the north basin of Lake Winnipeg under ice (left) and during the open water season (centre and right). For mid-summer 2003 and 2006, every profile recorded on each July - August survey of the north basin is shown in light grey. Representative profiles are coloured to distinguish the range of vertical distributions. Note the expanded temperature scale for late winter compared to mid-summer profiles. (Data Source: Fisheries and Oceans Canada, M. Stainton)*

However, thermal stratification has been observed in Lake Winnipeg in some years, particularly in the deep north basin both in late winter and at mid-summer. By late winter, most of the water column has cooled to less than 1.0ºC (2002 in Figure 5.9). However, the deepest water remains slightly warmer, and in some years a thin layer near the bottom may be isolated long enough that late winter hypoxia develops. Environment Canada findings indicate that oxygen-depleted waters may develop in Lake Winnipeg over the winter (Section 5.2.3).

In the summer of 2003, a thermocline was observed at the 12 to 15 m depth in the north basin (Figure 5.9) over a hypolimnetic layer that had been isolated from the surface since early July. The result was that bottom waters became depleted of oxygen and enriched in carbon dioxide (O^2 as low as 2.5 mg/L, CO^2 as high as 2,000 mg/L in samples within one metre from the bottom). This particular thermocline was eroded by turbulence due to strong winds in late August. Other instances of depressed oxygen levels (<4 mg/L) were reported (Section 5.2) for the winter of 2007, and the summers of 2006 and 2007. If these occasional observations represent a condition that is becoming more frequent than in the past, it is likely that the population structure of Lake Winnipeg biota will change. For example, the benthic community may shift toward more hypoxia-tolerant species and fewer pelagic species that feed at the sediment-water interface (Barton and Taylor 1996; Chapman 1986).

5.2 Dissolved Oxygen

(Lucie Lévesque, Len Wassenaar)

Dissolved oxygen in lakes originates from atmospheric re-aeration and phytoplankton productivity in the photic zone (Falkowski and Raven 1997) and is consumed by ecosystem respiration and biological decay (Wetzel 2001). In shallow water bodies in particular, dissolved oxygen concentrations depend not only on biological processes, but on aeration and water temperature as well, the latter of which affects the solubility of oxygen (Wetzel 2001). Studies in Lake Erie, where eutrophication has been of concern since the mid-1900s, have shown that the distribution and concentration of dissolved oxygen in the water column are functions of vertical mixing, exchanges between the water and the atmosphere, biological productivity (photosynthesis, community respiration, respiration at the sediment-water interface), and horizontal transport (Edwards *et al.* 2005; Charlton 1987; Patterson *et al.* 1985). Patterson *et al.* (1985), in modelling the biological and physical processes that dominate oxygen dynamics and budget in the central basin of Lake Erie, found that production and respiration dominated oxygen levels in the epilimnion, whereas surface fluxes predominated under conditions of elevated wind speed. In Lake Winnipeg, where eutrophication is also of concern but where depth and stratification are less pronounced, biological productivity (e.g., phytoplankton productivity associated with blooms, Sections 8.1 and 9.1) and wind-induced mixing and exchanges at the air-water and sediment-water interfaces (e.g., Section 5.3.3) may be expected to be prominent drivers of dissolved oxygen levels and dynamics in the lake.

Reduction in concentrations of dissolved oxygen may have adverse effects on aquatic organisms including invertebrate and fish kills in instances of severe oxygen depletion.

Excessive phytoplankton growth in response to eutrophication and associated demands for oxygen consumption that occur during subsequent decomposition have the potential to depress oxygen levels in lakes. Elevated rates of respiration and decay near the sediment-water interface may result in the depletion of dissolved oxygen in bottom waters. Reduction in concentrations of dissolved oxygen may have adverse effects on aquatic organisms including invertebrate and fish kills in instances of severe oxygen depletion. Oxygen depletion may also affect internal nutrient loading in lakes. Decomposition of algal biomass and other organic materials consumes oxygen and reduces the depth to which lake bottom sediments are oxygenated. Anoxic or hypoxic conditions at the sediment-water interface promote nutrient release (e.g., phosphorus) from sediments through diffusion, particularly in shallow waters (Anderson and Ring 1999). Therefore, oxygen depletion may positively feedback to eutrophication through the redistribution of nutrients into more bio-available forms. The role of respiration at the sediment-water interface in oxygen depletion and internal nutrient loading in Lake Winnipeg is not well understood. A preliminary study has shown that sediment-associated phosphorus increased in the latter half of the 20th century (Text Box 6), consistent with increased external nutrient loading to the lake (Section 7.1). Oxygen depletion would promote the release of phosphorus into the water column, amplifying the potential for eutrophication in Lake Winnipeg.

Over the past decade, Lake Winnipeg has developed signs of cultural eutrophication (i.e., driven by anthropogenic activities) as revealed by the increasing size and frequency of algal blooms (Section 9.1) and by record-breaking fisheries productivity (Manitoba Water Stewardship 2008). The dependence of dissolved oxygen on phytoplankton productivity, aquatic community respiration, and biological decay means that the health of the lake is susceptible to modification in response to eutrophication. One of the most commonly observed negative impacts of cultural eutrophication is bottom water oxygen depletion in thermally stratified lakes leading to so-called "dead zones", as seen in the Gulf of Mexico and in Lake Erie (Diaz and Rosenberg 2008; Yerubandi *et al.* 2008; Rabalais *et al.* 2002). Historically, and although spatial and temporal data are few, there were no reports of such depletion in Lake Winnipeg from the original surveys of the lake in 1964 to 2002 (Brunskill *et al.* 1979a; Kristofferson *et al.* 1975; Rybicki 1966). However, in the summer of 2003, oxygen-depleted waters (<2.0 mg/L) below a thermocline were detected at several stations in the deepest portions of the north basin of Lake Winnipeg (Stainton 2005). Dissolved oxygen concentrations in the summer of 2003 were below the Manitoba Water Quality Objective for the protection of aquatic life (instantaneous minimum of 5 mg/L). This oxygen depletion event has led to the suggestion that Lake Winnipeg's eutrophication trajectory currently poses a threat to dissolved oxygen (Basin 2009), one of the most fundamental requirements for the survival of aquatic life.

5.2.1 Recent Conditions: 1999 to 2007

Dissolved oxygen concentrations in Lake Winnipeg were measured by Manitoba Water Stewardship from 1999 to 2007 at long-term water quality monitoring stations on the lake. Measurements were made in the spring, summer, and fall in surface and bottom waters. Inter-annual comparisons of mean dissolved oxygen concentrations in Lake Winnipeg over the long-term are limited by the absence, or low number, of measurements in some seasons (e.g., north basin, summer 2001; Figure 5.10; Appendix 1). Dissolved oxygen concentrations were measured less frequently in winter than over the open water season. More rigorous examinations of under-ice dissolved oxygen have recently been undertaken by Environment Canada (Section 5.2.3).

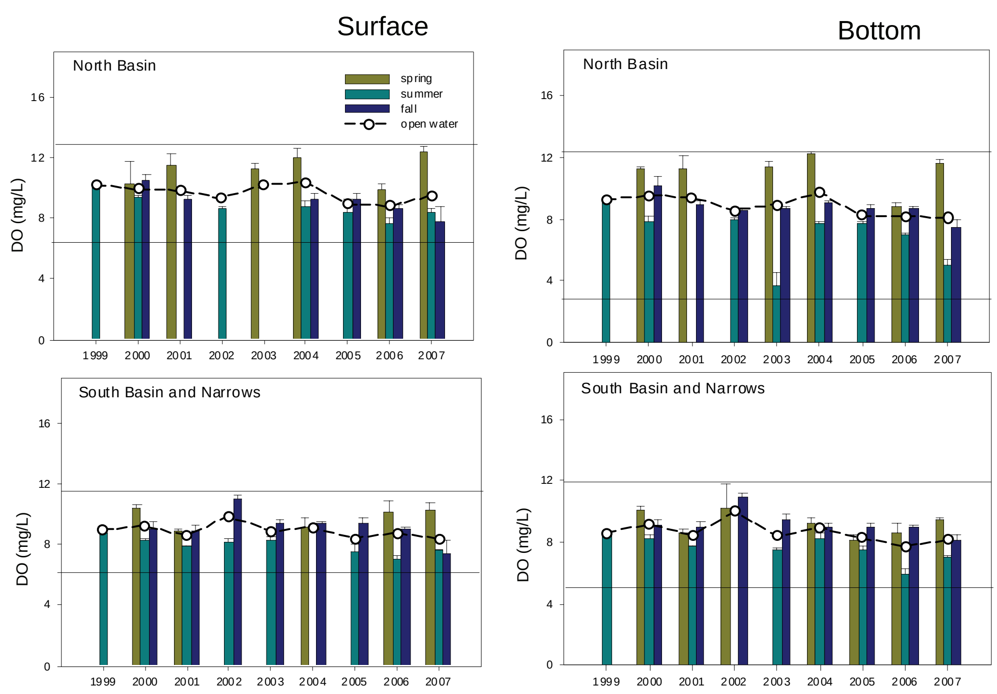


Figure 5.10: *Annual and seasonal mean dissolved oxygen concentrations (± SE) in the north basin and south basin (and narrows) of Lake Winnipeg from 1999 to 2007. Line graph indicates season-weighted mean for the open water season (n range for surface: north basin = 11 to 26, south basin = 8 to 30. n range for bottom: north basin = 12 to 78, south basin = 8 to 56).*

Dissolved oxygen concentrations in surface waters of Lake Winnipeg from 1999 to 2007 were slightly lower in the south basin than in the north basin (Table 5.2, Figure 5.10). Mean annual dissolved oxygen levels varied between 8.8 and 10.4 mg/L in the north basin. Concentrations in the south basin were approximately 0.4 mg/L lower than those of the north basin (mean annual range: 8.4 to 9.9 mg/L). Approximately 2 % of measurements in both basins were below the Manitoba Water Quality Objective for the protection of aquatic life (5.0 mg/L, Williamson 2002), indicating the potential for detrimental effects on aquatic biota.

Table 5.2: *Dissolved oxygen concentrations (mg/L) in Lake Winnipeg from 1999 to 2007. Season-weighted means. (Data Source: Manitoba Water Stewardship)*

SURFACE	North Basin				South Basin and Narrows			
Year	Mean	Min	Max	N	Mean	Min	Max	N
1999	10.2	8.3	12.6	21	9.0	8.2	11.9	30
2000	9.9	4.7	12.6	19	9.2	7.7	11.1	14
2001	9.8	8.2	12.9	16	8.6	7.8	9.7	10
2002	9.3	8.5	12.2	11	9.9	7.8	12.3	8
2003	10.2	6.5	12.4	11	8.8	7.8	10.1	14
2004	10.4	6.1	13.8	20	9.2	7.5	10.9	12
2005	8.9	6.0	10.9	13	8.4	6.3	10.8	10
2006	8.8	5.9	11.0	26	8.8	6.1	13.3	20
2007	9.5	3.9	13.6	21	8.4	3.6	12.2	20
Average 1999 to 2007	9.7				8.9			

BOTTOM	North Basin				South Basin and Narrows			
Year	Mean	Min	Max	N	Mean	Min	Max	N
1999	9.3	8.1	10.1	21	8.6	7.9	9.9	26
2000	9.6	5.5	11.7	18	9.1	7.8	10.8	15
2001	9.4	7.7	12.9	15	8.4	7.6	9.8	11
2002	8.6	7.4	11.3	12	10.0	7.6	13.1	8
2003	9.0	2.4	12.7	14	8.4	7.0	10.3	14
2004	9.7	6.3	13.3	58	8.9	7.3	11.2	35
2005	8.2	6.7	10.2	39	8.3	4.7	10.9	33
2006	8.2	2.9	10.5	78	7.7	2.1	13.2	57
2007	8.1	1.7	12.7	59	8.2	5.0	10.1	56
Average 1999 to 2007	8.9				8.6			

Concentrations of dissolved oxygen in bottom waters were lower than those in surface waters, as would be expected in response to differences in temperatures and aeration with depth. However, the discrepancy between bottom and surface waters in Lake Winnipeg is not substantial compared to lakes with more pronounced stratification and less extensive vertical mixing by winds (e.g., Lake Erie, Lake Simcoe - Ontario Ministry of Environment 2010; Edwards *et al.* 2005). Dissolved oxygen concentrations in the bottom waters of Lake Winnipeg from 1999 to 2007 were slightly higher in the north basin of the lake (Table 5.2, Figure 5.10), ranging between 1.7 and 13.3 mg/L. Those in the south basin ranged between 2.1 and 13.2 mg/L. Dissolved oxygen concentrations in bottom waters of the lake more frequently exceeded guidelines than those in surface waters, with approximately 6.0 % of measurements in the north basin and 2.4 % in the south basin falling below the Manitoba water quality objective for the protection of aquatic life.

Mean seasonal dissolved oxygen concentrations on both surface and bottom waters were typically lower in the summer relative to the spring and fall. This is consistent with seasonal variations in water temperatures and rates of biological productivity, which would typically be at their greatest in the summer. Lower dissolved oxygen concentrations are associated with higher water temperatures in the lake (Figure 5.11). This is attributed mainly to the lower solubility of dissolved oxygen in warmer waters as well as increased consumption of dissolved oxygen with rising rates of respiration and decay at the sediment-water interface (Wetzel 2001). In addition, differences in water temperatures between the north and south basins, as well as the greater depth and lesser degree of mixing and aeration in the north basin, account for the slightly higher concentrations of dissolved oxygen in the north basin relative to those further south.

The reduction in dissolved oxygen concentrations in bottom waters over the summer was particularly pronounced in 2003 in the north basin, when oxygen-depleted waters were first detected in the lake. Concentrations as low as 2.4 mg/L and up to 5.2 mg/L were measured (Stainton 2005). Similarly, concentrations of dissolved oxygen in the north basin were as low as 2.9 and 1.7 mg/L in the summers of 2006 and 2007, respectively. In all years, dissolved oxygen concentrations had recovered by the fall.

The reduction in dissolved oxygen concentrations in bottom waters over the summer was particularly pronounced in 2003 in the north basin, when oxygen-depleted waters were first detected in the lake.

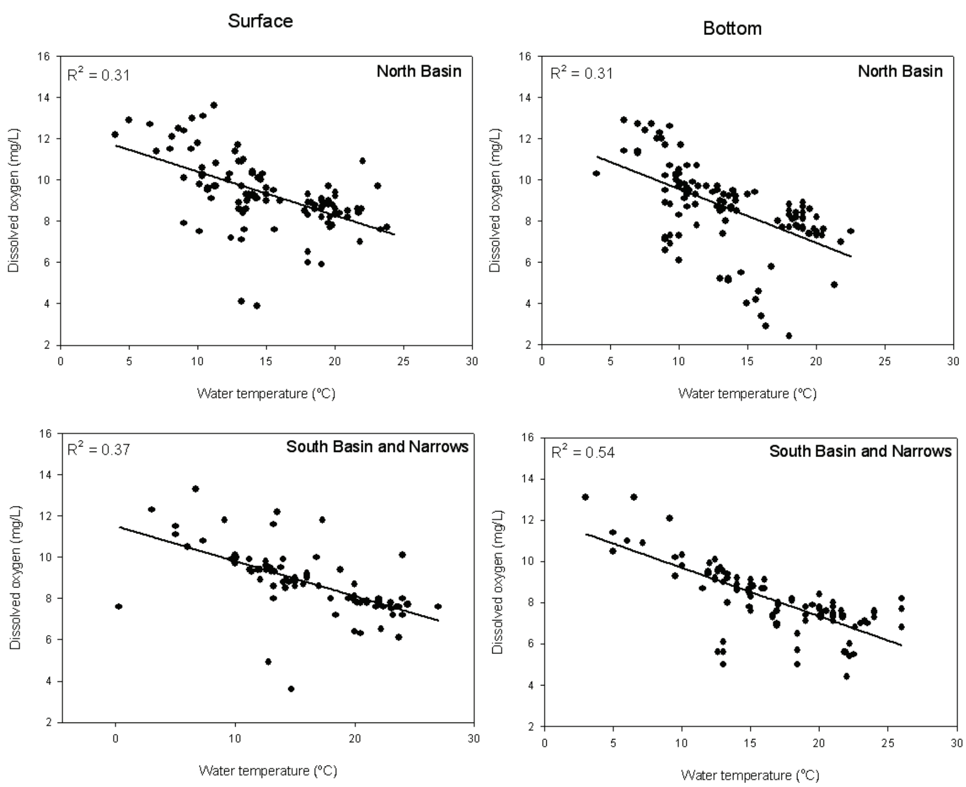


***Figure 5.11:** Dissolved oxygen concentration versus water temperature in surface (<1 m depth) and bottom (north basin typically >13 m, south basin >7 m, and narrows >20 m) waters of the north basin and south basin (and narrows).*

5.2.2 Seasonal and Spatial Variation

In 2006, Environment Canada began an intensive multi-year investigation of the status and dynamics of dissolved oxygen in Lake Winnipeg (Appendix 1). The first objective of the study was to quantify the seasonal spatial-temporal dissolved oxygen patterns in Lake Winnipeg from 2006 to 2011 by means of extensive and detailed vertical water quality depth profiling at up to 65 stations in the north and south basins. Particular attention was to be paid to zones of high primary productivity and to the detection of bottom water hypoxia. The second objective was to attempt to quantify current spatial-temporal productivity patterns in the lake through the application of stable oxygen isotope analyses.

Surveys from the spring, summer, and fall of 2007 provided an indication of the nature of spatial variability in dissolved oxygen concentrations in Lake Winnipeg and the processes governing this variability (Figure 5.12). Findings were consistent with those for mean concentrations between 1999 and 2007 (Figure 5.10, Table 5.2). Dissolved oxygen concentrations in surface waters were typically higher in the north basin than the south basin. In the spring and summer, dissolved oxygen concentrations in the lake increased from north to south, reaching concentrations above 10.0 mg/L north of Reindeer Island (Figure 5.12). Dissolved oxygen levels dropped from spring to summer in both basins, falling by approximately 2.0 to 3.0 mg/L from the spring to summer. In the fall, the northward gradient in dissolved oxygen concentrations was less pronounced, and concentrations were typically greater than those in the summer.

Surveys from the spring, summer, and fall of 2007 provided an indication of the nature of spatial variability in dissolved oxygen concentrations in Lake Winnipeg and the processes governing this variability.

Dissolved oxygen concentrations in bottom waters over the open water season in 2007 typically varied spatially and seasonally in a manner similar to those near the surface (Figure 5.12). The similarity of dissolved oxygen concentrations in surface and bottom waters over the open water season, consistent with the shallow, well-mixed nature of Lake Winnipeg's waters and the general lack of thermal stratification (Section 5.1), was most notable in the south basin. The south basin was isothermal over all seasons sampled and is unlikely to undergo thermal stratification due to its shallow depth and high rate of re-aeration.

The difference in dissolved oxygen concentrations between surface and bottom waters in the summer of 2007 was most notable in the north basin. Vertical depth profiles from stations in the north and south basins in 2007 indicated that dissolved oxygen concentrations and water temperatures in the south basin did not vary substantially between the surface and the 10 m depth in the summer and fall, and those in the north basin did not vary substantially between the surface and the 16 m depth in the spring and fall (Figure 5.13). However, in the summer, dissolved oxygen concentrations dropped throughout the lake, particularly in bottom waters of the central north basin, indicating the potential for stratification. The lowest concentrations occurred near Long Point in the north basin, with concentrations increasing southward into the narrows and decreasing slightly into the south basin. Dissolved oxygen concentrations at a station in the north basin declined from near 10.0 mg/L in the upper 10 m of the water column where water temperatures were near 20.0^{o}C, to near 6.0 mg/L at depths between 12 and 16 m where water temperatures declined to near 12.0^{o}C (Figure 5.13). The deepest parts of the north basin are most susceptible to depressed oxygen concentrations, particularly in the summer when a seasonal thermocline may form in this region of the lake and rates of biological respiration and decay would be at their greatest. Dissolved oxygen

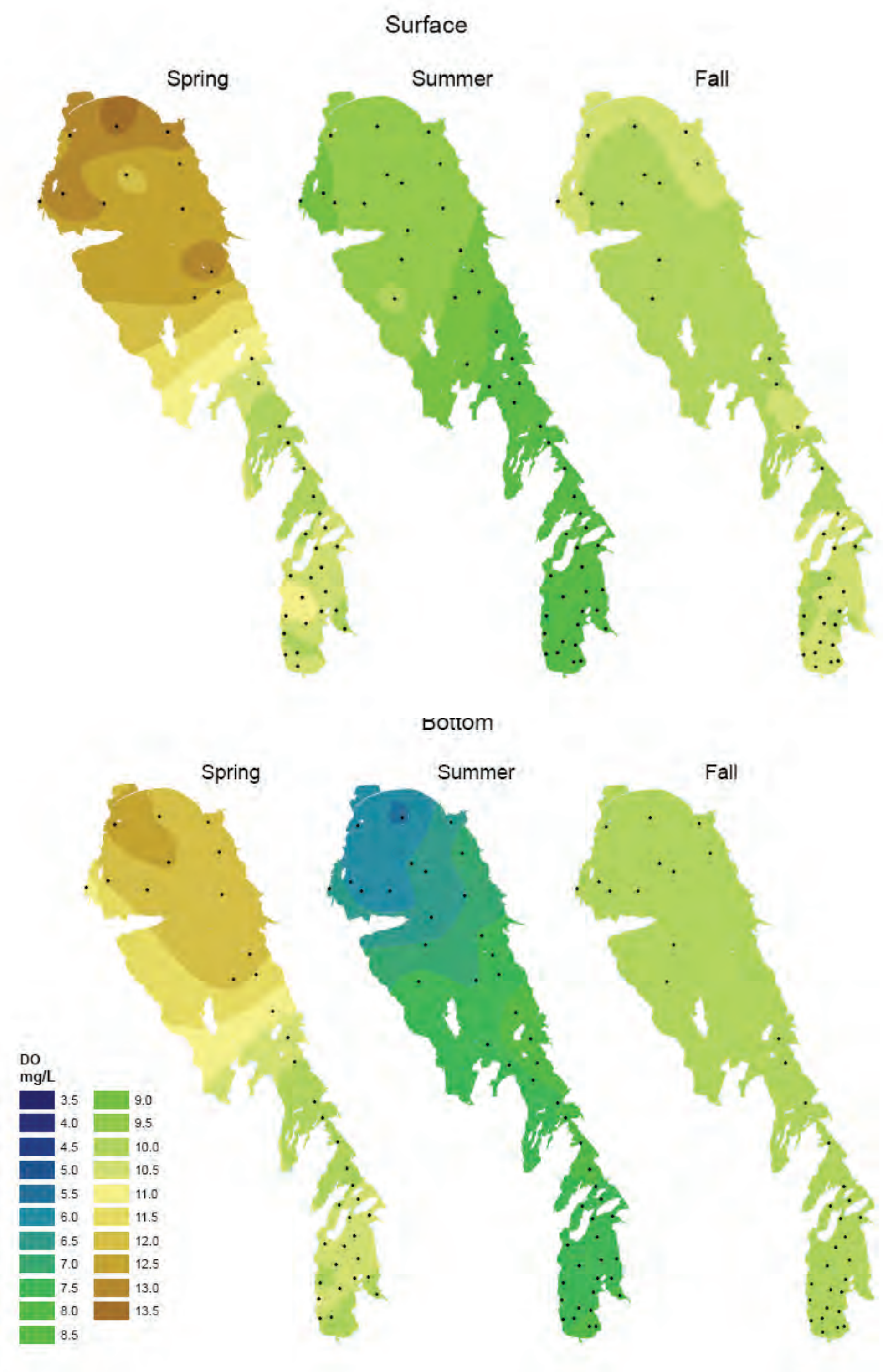


Figure 5.12: *Dissolved oxygen concentrations in Lake Winnipeg surface and bottom waters over the spring (May-June), summer (July-August), and fall (September-October) of 2007.*

concentrations in surface and bottom waters of the lake recovered to near 10.0 mg/L in the fall and thermal stratification in the north basin was no longer evident (Figures 5.12 and 5.13).

Although dissolved oxygen concentrations in Lake Winnipeg are typically lower in the south basin than the north basin, reduced in the summer relative to the spring and fall, and at times stratified in the north basin, investigation of dissolved oxygen in the lake has revealed that concentrations are highly variable inter- and intra-annually, as well as between basins. Dissolved oxygen concentrations in Lake Erie are less variable over the long-term than inter-anually (Charlton 1987). Continued seasonal, inter-annual, and spatial measurement of dissolved oxygen is necessary to develop a more thorough understanding of the range and variability in concentrations in Lake Winnipeg, and the linkages of these to climatic conditions, nutrient-loading, and biological productivity.

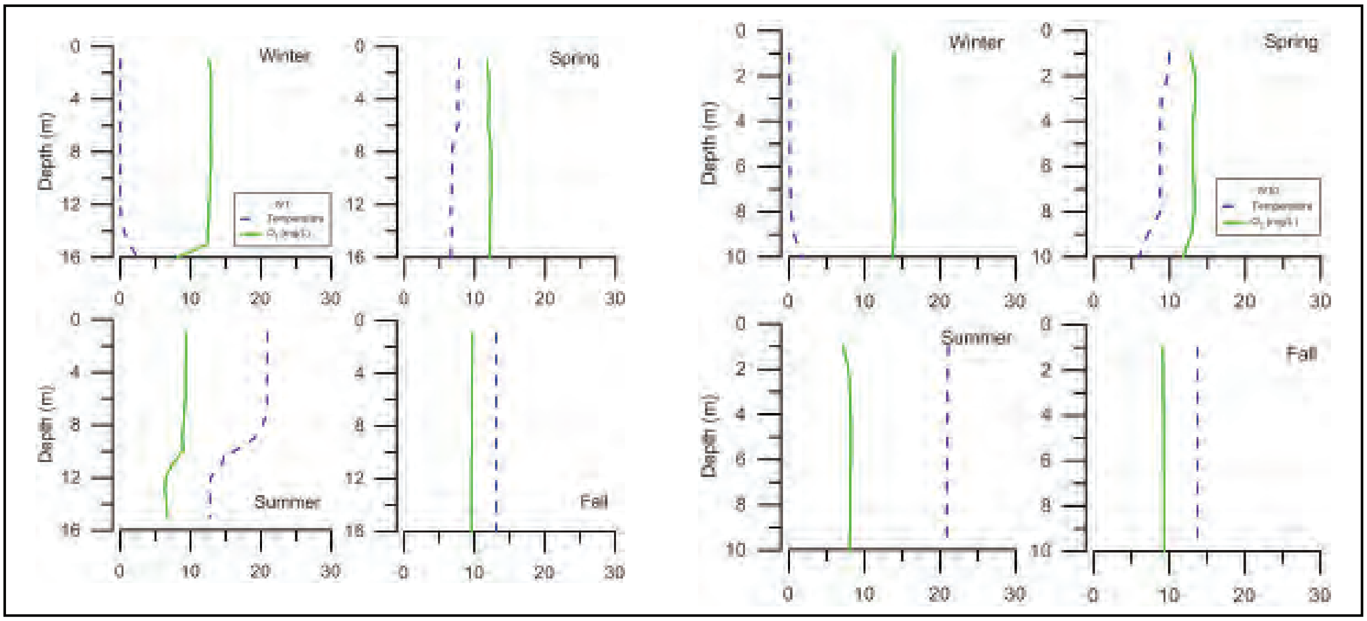


Figure 5.13: *Dissolved oxygen concentrations and water temperature over vertical profiles in the north basin (left) and south basin (right) of Lake Winnipeg in 2007. (Data Source: Environment Canada)*

5.2.3 Dissolved Oxygen Depletion

Oxygen depletion in Lake Winnipeg was first noted with the observation of thermal stratification and depressed oxygen concentrations in the north basin in 2003 (Stainton 2005). Since 2003, more thorough study of dissolved oxygen has taken place. In addition to dissolved oxygen concentration, the percent dissolved oxygen saturation (normalized for water temperature) in Lake Winnipeg was measured by Environment Canada during the open water season and under ice from 2006 to 2009. Dissolved oxygen concentrations seek temperature-dependent equilibrium (100 % saturation) with the atmosphere by in- and out-gassing. However oxygen super- or under-saturation is common in lakes, governed by the balance between photosynthesis and respiration relative to re-aeration or gas exchange (Dubois *et al.* 2009; Wetzel 2001; Carignan *et al.* 2000; del Giorgio and Peters 1994; Odum 1956). The percent dissolved oxygen saturation in waters, therefore, provides an indication of potential for depletion (e.g., <100 % saturation).

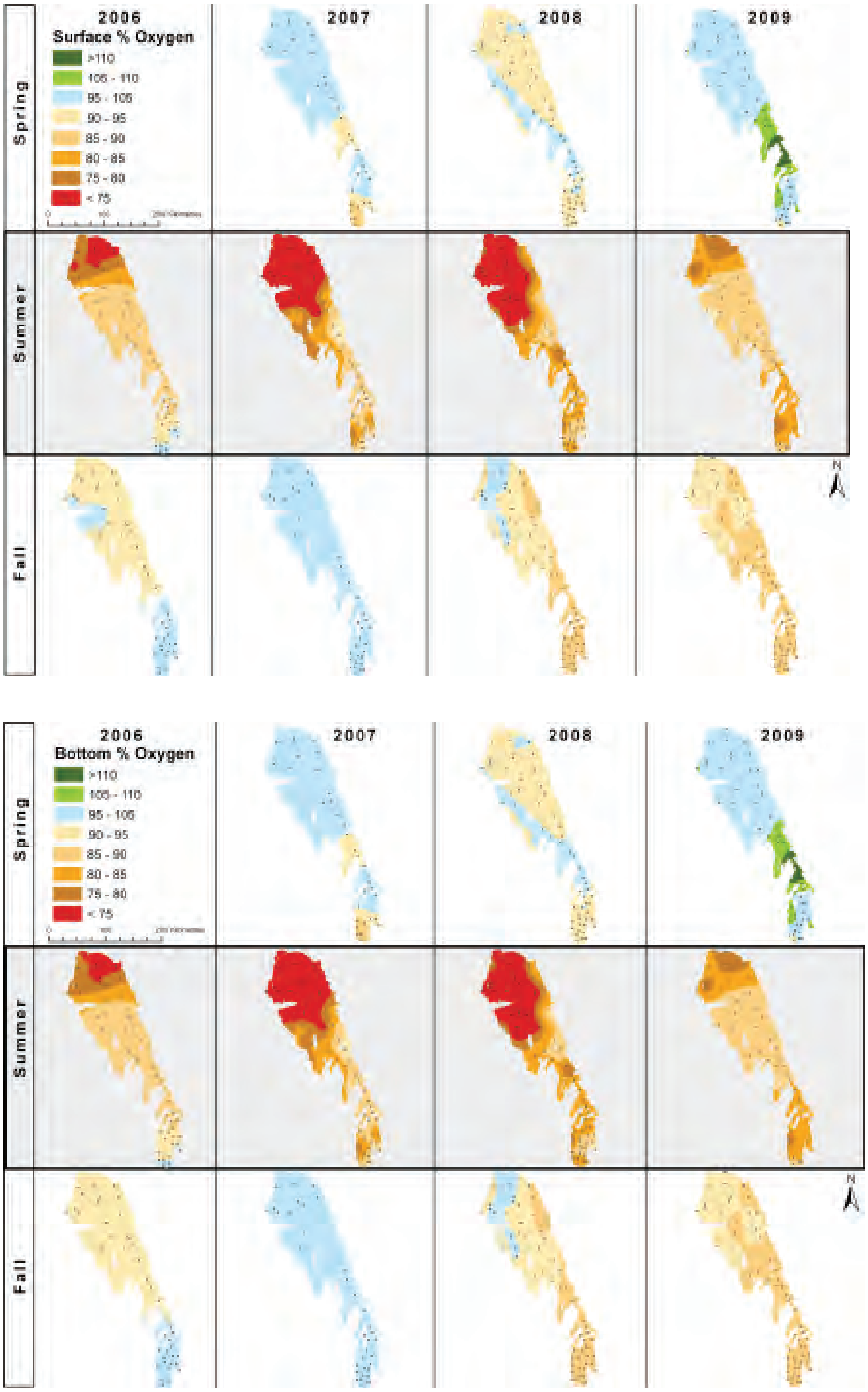


Figure 5.14: *Percent dissolved oxygen in Lake Winnipeg surface and bottom waters over the spring, summer, and fall of 2006 to 2009. Dark green = >110%, Red = <75%. (Data Source: Environment Canada, L. Wassenaar)*

The percent dissolved oxygen saturation in surface and bottom waters of Lake Winnipeg from 2006 to 2009 varied considerably from season to season and year to year (Figure 5.14), reflecting the complexity of dissolved oxygen dynamics in Lake Winnipeg. Reductions in dissolved oxygen saturation were more evident in bottom waters than surface waters, particularly in the summer. Percent saturation in the summers was for the most part between 75 and 105 % near the surface, and saturation in bottom waters was typically between <75 % to 85 %. Saturation levels of <75 % were common in surface and bottom waters of the north basin every summer except in 2009 and, on three occasions concentrations fell below critical thresholds (5 mg/L instantaneous minimum, Manitoba Water Quality Objective for the protection of aquatic life). Levels recovered by fall each year to near or above 100 % saturation.

Relatively low dissolved oxygen saturation was also recorded at depth in the winter. Surveys from 2006 to 2008 indicated that bottom waters may be under-saturated to levels lower than 75 % (Figure 5.15). Vertical profiles of dissolved oxygen concentrations at stations in the north and south basins illustrate the potential for reduction in dissolved oxygen concentrations in association with under-ice thermal stratification and restricted re-aeration. Dissolved oxygen concentrations at a station in the north basin and one in the south basin decreased by 5.0 to 10.0 mg/L at depth in association with an inverse thermocline (Figure 5.13). Concentrations reached as low was 2.7 mg/L in the north basin in 2007.

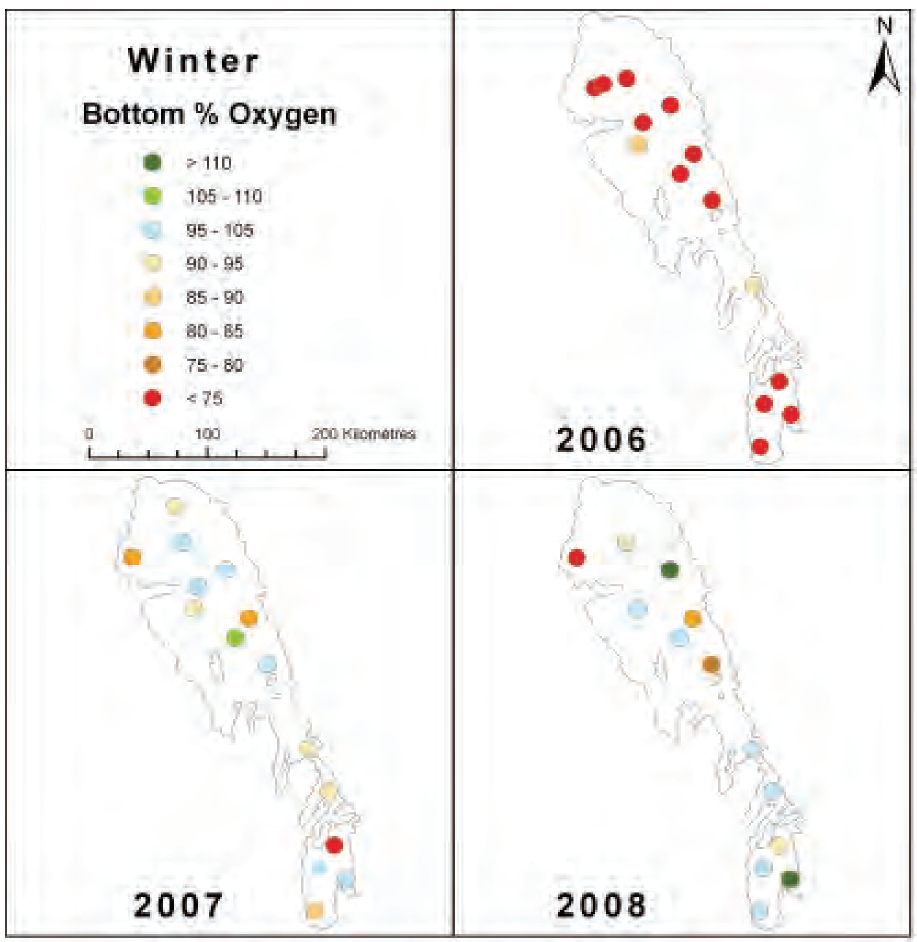


Figure 5.15: *Percent dissolved oxygen in Lake Winnipeg bottom waters in winter 2006 to 2008. Dark green = >110%, Red = <75%. (Data Source: Environment Canada, L. Wassenaar)*

While occurrences of dissolved oxygen concentrations below critical thresholds in the north basin were few, the spatial extent of low oxygen zones in Lake Winnipeg are not known. Given oxygen depletion generally occurred in bottom waters of the north basin and only below a thermocline, the controls on bottom water oxygen depletion in Lake Winnipeg will likely be re-aeration constraints and temperature stratification. This suggests that weeks of prolonged hot weather with little wind are the most undesirable conditions, enabling the formation of summertime thermocline and oxygen depletion in the north basin. Patterson *et al.* (1985) showed that mildly eutrophic lakes, such as Lake Erie, do not regain saturation of dissolved oxygen as quickly as oligotrophic lakes (e.g., <1 day) except over long periods of low irradiance and elevated wind speed. In addition to thermocline development, respiration following large algal blooms may promote reduction of oxygen levels in bottom waters, particularly in winter when re-aeration is constrained by ice. The duration and spatial extent of thermocline development in the north basin during the summer and the spatial and temporal extent of bottom water dissolved oxygen concentrations over winter are key information gaps.

The history of oxygen levels in Lake Erie illustrates the complexities of oxygen depletion and its effects in lakes. Oxygen depletion was first noted in the hypolimnion of the central basin of Lake Erie in the 1950s (Charlton 1987). Though oxygen depletion is natural in lakes, the degree and extent of depletion in Lake Erie was attributed to elevated total phosphorous loadings and concentrations in the lake, which peaked as late as the early 1970s (Bertram 1993). Examinations of historical dissolved oxygen in the lake indicated that depletion rates and the extent of the anoxic area increased from 1929 to 1980, attributed to increasing phosphorous concentrations and associated increases in productivity and decomposition (Rosa and Burns 1987). Reduction in phosphorous loadings and concentrations were observed from the 1970s through the 1980s, however, oxygen depletion in the lake (and primary production) did not fall correspondingly. Given oxygen depletion seemed to represent integration of effects over many years, it was expected that oxygen levels would respond to reduced loadings after a 10 to 15-year lag (Charlton 1987). Indeed, hypolimnetic oxygen depletion began to fall in the early 1990s, promoting recovery of meostrophic species and benthic communities (Carrik 2004; Kane *et al.* 2004), and re-establishment of the fish community, including species unable to tolerate anoxia (Ludsin *et al.* 2001). However, in recent years, expansion of oxygen depletion has been once again observed in Lake Erie (Edwards *et al.* 2005), as have increased phosphorous concentrations (Rockwell *et al.* 2005) and shifts toward eutrophic phytoplankton and increased biomass (Conroy *et al.* 2005). Although external nutrient loading has been a driver of eutrophication and oxygen depletion in the lake, the renewed depletion in Lake Erie has been attributed to sediment-oxygen demand and hypolimnion respiration. Theory suggests that the introduction of invasive mussels has played a significant role in this recent change by consuming oxygen and excreting nutrients in the hypolimnion, as well as potentially promoting remineralisation of nutrients (Conroy *et al.* 2005; Arnott and Vanni 1996). This in effect, increases algal productivity and the formation of blooms (Edwards *et al.* 2005). The history of eutrophication and dissolved oxygen depletion in Lake Erie suggests that reductions in dissolved oxygen depletion and associated effects on aquatic biota and trophic status (Section 6.0) in Lake Winnipeg may be expected to begin nearly a decade or more after the implementation of reduction measures for external nutrient loading. The response of the lake to reduction in external nutrient loadings may be expected to be confounded by internal nutrient loading (e.g., release of sediment-associated phosphorus, Text Box 6) decomposition of algal biomass, the potential establishment of new invasive species (e.g., zebra mussel, Section 9.4) and changes in climate (Section 9.5).

The duration and spatial extent of thermocline development in the north basin during the summer and the spatial and temporal extent of bottom water dissolved oxygen concentrations over winter are key information gaps.

5.3 Total Suspended Solids

(Greg McCullough, Lucie Lévesque)

Suspended solids in lakes are particles comprised of materials derived externally from watershed erosion and biomass production in tributary rivers and lakes, and to a lesser degree from atmospheric deposition. Suspended solids also originate within lakes from littoral erosion, resuspension of materials formerly deposited as bottom sediments, and biomass production within the lake itself. The concentration of these materials in the water column, operationally defined here as the weight of solids captured on a 1.2 μm filter, divided by the volume of water passed through the filter, is referred to as total suspended solids (TSS). It is important to understand that TSS as determined by filtration includes both mineral and biological particles. Samples collected from Lake Winnipeg between 2002 and 2004 indicated that TSS in the south basin was on average almost two-thirds mineral (predominantly clay and silt) by weight and in the north basin, almost three-quarters organic material (i.e. phytoplankton, zooplankton and organic detritus; McCullough, unpublished data). Within lakes, TSS varies spatially and temporally as a function of external loading and internal transport (e.g. internal circulation of water masses), internal biological productivity, and disturbance of the water column (most frequently wind-induced turbulence causing littoral erosion or resuspension of bottom sediments, although thermal circulation and gravity currents play a part as well). TSS also varies with depth when wind-generated turbulence is insufficient to overcome settling of particles in the water column.

In Lake Winnipeg waters are relatively shallow. This, along with long fetches, contributes to the frequent occurrence of wind-developed turbulence, which disturbs bottom sediments, resuspending them in the water column.

Suspended solids are integral to the physical, chemical, and biological nature of freshwater lake ecosystems. Because the mineral component of TSS may carry absorbed nutrients, and the biological component is itself composed largely of nutrients, deposition and resuspension of particles affects nutrient transport through, and storage in, the lake and availability in the water column. Suspended solids which have settled onto the bottom may represent a considerable reserve of nutrients (Text Box 6). In Lake Winnipeg, however, waters are relatively shallow. This, along with long fetches, contributes to the frequent occurrence of wind-developed turbulence sufficient to disturb bottom sediments and resuspend them into the water column. In addition, some sediments deposited through the winter may be returned to the water column during spring overturn of the water column and dissolution of nutrients at the sediment-water interface (most likely to occur whenever bottom water or sediments become hypoxic or anoxic) may return nutrients from the bottom sediments to the water column. However, the extent of internal sediment and nutrient loading through either resuspension or dissolution from bottom sediments has not been quantified for Lake Winnipeg.

More directly, suspended solids are a major determinant of light distribution in lakes. By absorbing and scattering light, they limit its transmission into the water column. In Lake Winnipeg, transparency and the depth of penetration of light are highly and inversely correlated with TSS (Brunskill *et al.* 1980; and results in the section following). Elsewhere, low light has been demonstrated to limit primary productivity (e.g. Wang *et al.* 2005) and to affect community composition and trophic structure of lake ecosystems (e.g., phytoplankton - Scheffer *et al.* 1997).

The effect of sedimentary processes on spatial and temporal distributions of light in Lake Winnipeg is better documented than is the effect of these processes on internal storage and loading. Both early and current studies demonstrate that turbidity tends to be highest (and water column transparency lowest) in the south basin near the mouth of the most sediment-rich tributary, the Red River, and lowest (transparency highest) in the north, where tributaries are relatively poor in suspended solids, and the deeper bottom is less subject to disturbance by wind-induced turbulence (Brunskill *et al.* 1979b; Bajkov 1930; results in the section following). Phytoplankton, zoobenthos, and fish have all been recorded in lower abundance in the more turbid waters of the south basin compared to the relatively transparent waters of the north basin (Bajkov 1930). Historical satellite data have been analyzed to demonstrate that widespread, intense surface blooms of phytoplankton were common in the relatively clear waters of the north basin of the lake almost a decade before they began to occur frequently in the more turbid south basin (Section 9.1). However, widespread surface blooms are now almost annual occurrences in both basins. Cyanobacteria (which readily form surface blooms) thrive in low-light conditions and have been shown to achieve greater dominance in turbid than in clear waters (e.g., Scheffer *et al.* 1997; Section 8.1). This characteristic has implications for the distribution of algal blooms and associated toxins within the lake (Section 9).

5.3.1 Recent Conditions: 1999 to 2007

Turbidity (a measure of light scattering by suspended particles) and Secchi depth (the depth to which a submerged 25 cm disc is visible to the eye, and therefore an approximate measure of water transparency) are related to TSS in lakes. All three parameters were measured by Manitoba Water Stewardship at long-term water quality monitoring stations in Lake Winnipeg from 1999 to 2007 (Appendix 1). Samples were collected and measurements were made at the surface and through the euphotic zone in the spring, summer, and fall. However, the ability to make inter-annual comparisons of mean TSS in the lake is limited because of the absence or low number of measurements in some seasons.

Total suspended solids concentrations in the north basin were typically lower than those in the south basin (Table 5.3; Figure 5.16). In the north basin, mean annual TSS varied between 2.6 and 10.6 mg/L, in the south basin the annual means varied between 7.7 and 14.4 mg/L. Outliers were excluded from the annual open water and seasonal means to eliminate the influence of extreme values (Appendix 1). These outliers reached as high as 118 mg/L in the north basin (fall 2004) and 108 mg/L in the south basin (fall 2007). Higher TSS in the south basin as compared to the north basin over the period of 1999 to 2007 is consistent with previous studies that have indicated clearer water in the north basin than the south basin (McCullough *et al.* 2001; Brunskill *et al.* 1979b). Higher sediment concentrations in the south basin can be attributed to the high sediment loads delivered by the Red River (Brunskill *et al.* 1980) and to the greater susceptibility of the shallow south bottom sediments to wind-driven resuspension (McCullough *et al.* 2001).

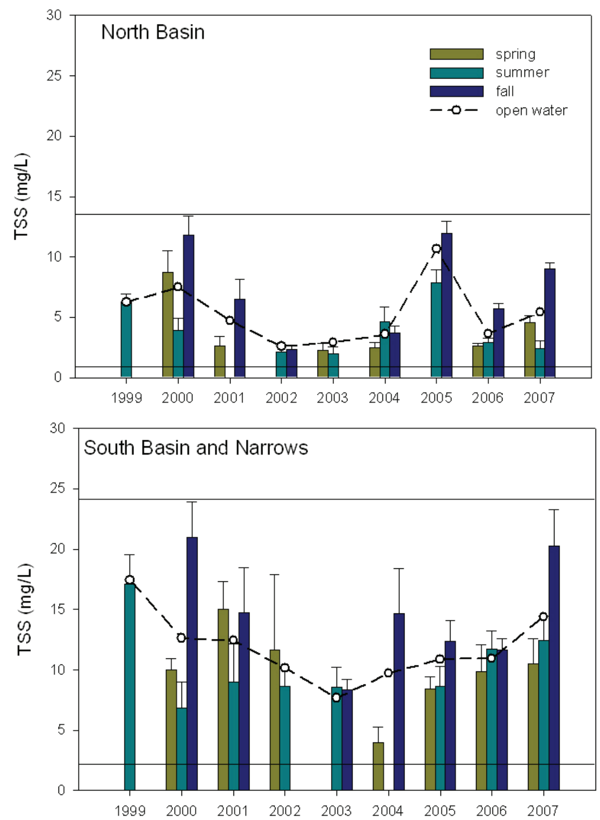


Figure 5.16 *Annual and seasonal mean total suspended solids concentrations (± SE) in the north basin and south basin (and narrows) of Lake Winnipeg from 1999 to 2007. Concentrations represent water samples collected from the surface and euphotic zone. Line graph indicates season-weighted mean for the open water season (n range for north basin = 12 to 59, south basin = 8 to 96).*

Mean seasonal TSS was typically highest in the fall in both basins (Figure 5.16). The Canadian Council of Ministers of the Environment guideline for the protection of aquatic life is defined as 25 mg/L above background (Canadian Council of Ministers of the Environment 2005). Background concentrations of water quality parameters for Lake Winnipeg have yet to be

Table 5.3. *Total suspended solids concentrations (mg/L), turbidity (NTU) and secchi disc depth (m) in Lake Winnipeg from 1999 to 2007. Season-weighted means. (Data Source: Manitoba Water Stewardship)*

TSS	North Basin				South Basin and Narrows			
Year	Mean	Min	Max	N	Mean	Min	Max	N
1999	6.2	2.5	13	21	17.4	2.5	58	30
2000	7.5	2.5	15	20	12.6	2.5	60	15
2001	4.7	0.5	35	20	12.4	4.0	24	12
2002	2.6	1.0	6	12	10.1	4.0	40	9
2003	2.9	0.5	8	13	7.7	2.0	91	14
2004	3.6	0.5	14	28	9.7	2.0	41	14
2005	10.6	4	21	35	10.9	2.0	49	51
2006	3.6	0.5	10	59	10.9	2.0	66	82
2007	5.4	0.5	14	59	14.4	2.0	108	98
Average 1999 to 2007	5.2				11.8			

Turbidity	North Basin				South Basin and Narrows			
Year	Mean	Min	Max	N	Mean	Min	Max	N
1999	9.0	2.2	35	21	24.4	10.0	57	30
2000	9.2	1.9	25	20	29.6	6.7	91	15
2001	5.5	1.4	17	19	16.4	5.7	28	12
2002	2.9	1.0	7.4	13	21.9	3.5	50	9
2003	3.3	0.9	13	12	20.6	1.6	80	14
2004	5.3	1.0	18	20	13.9	2.2	38	12
2005	11.6	5.8	27	12	13.9	5.9	22	11
2006	6.1	1.4	17	26	13.1	3.2	94	21
2007	7.7	2.3	26	21	21.5	3.2	94	21
Average 1999 to 2007	6.7				19.5			

Secchi Disk Depth	North Basin				South Basin and Narrows			
Year	Mean	Min	Max	N	Mean	Min	Max	N
1999								
2000	1.55	0.50	3.12	15	0.53	0.25	1.00	13
2001	1.27	1.10	3.12	21	0.43	0.25	0.75	15
2002	1.77	0.85	4.00	35	0.70	0.20	1.40	29
2003	2.13	1.50	2.75	11	0.65	0.13	2.48	14
2004	1.18	0.50	2.50	10	0.30	0.20	0.40	4
2005	0.66	0.12	1.50	33	0.57	0.18	1.50	45
2006	1.33	0.25	2.50	57	0.76	0.15	1.90	82
2007	1.33	0.50	2.75	58	0.69	0.25	1.50	93
Average 1999 to 2007	1.40				0.60			

established. Although TSS from 1999 to 2007 in Lake Winnipeg was for the most part below 25 mg/L, concentrations in excess of this occurred commonly in the fall, particularly in the south basin (48 % of observations were greater than 25 mg/L). In the south basin, higher TSS late in the open water season is explained largely by more frequent or more persistent strong winds in autumn compared to spring or summer (McCullough *et al.* 2001; Section 5.3.2). On the other hand, in the north basin, higher concentrations in late summer and autumn are generally the cumulative consequence of phytoplankton growth and reproduction over the summer. However, even in the north basin, relatively high TSS in shallow (<12 m deep, e.g. south of Berens Island) and/or nearshore waters (e.g., south of Berens Island, and along the east and north shores) is more likely due to littoral erosion and bottom resuspension associated with high winds.

Turbidity was generally greater and Secchi depth shallower in the south basin relative to the north basin in autumn compared to spring and summer (Figure 5.17). Mean annual turbidity in the north basin from 1999 to 2007 varied between 2.9 and 11.6 NTU (minimum: 0.9, maximum: 35.0 NTU), and mean annual Secchi depth varied between 0.66 and 2.13 m (minimum: 0.10, maximum: 4.00 m). In the south basin, mean annual turbidity ranged from 13.1 to 24.4 NTU (minimum: 1.6, maximum: 61.6 NTU). More turbid water in the south basin restricted the depth to which light penetrated the water column, so that Secchi depths tended to be less than one-half those in the north basin (mean annual: 0.30 to 0.76 m, minimum: 0.13

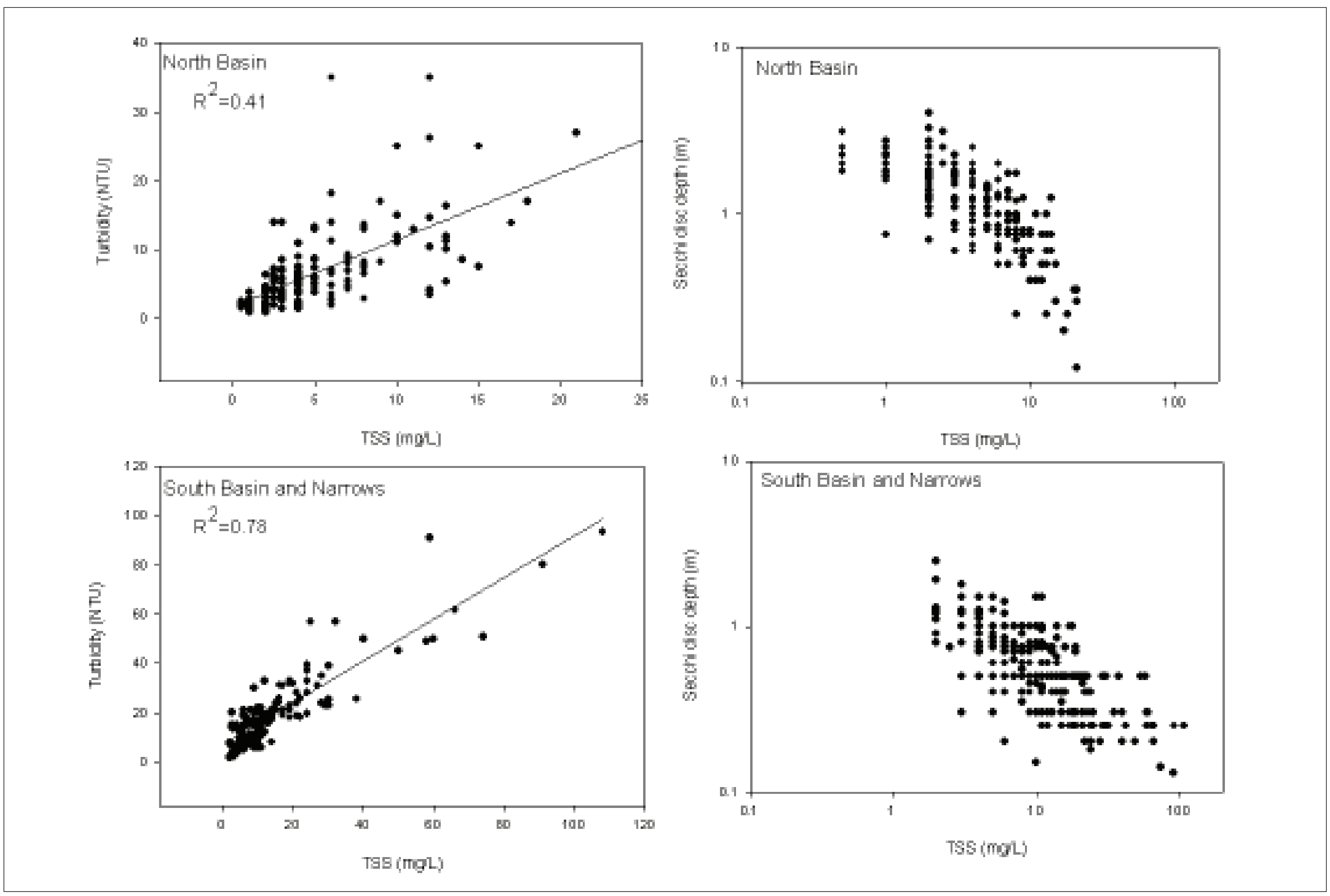


Figure 5.17: *Total suspended solids concentrations versus turbidity and Secchi disc depths in surface waters (0-1m depth-integrated) of the north basin and south basin (and narrows) of Lake Winnipeg from 1999 to 2007.*

m, maximum: 1.90 m). Seventy-seven percent of Secchi depth measurements in the south basin were less than one metre, and 80 % of those (shallower Secchi depths) were recorded in the summer and fall. Conversely, 69 % of the measurements from the north basin were greater than one metre, and again most (70 %) of the shallower ones were recorded in the fall. Consequently, primary productivity, particularly that of species that may be light-limited, would be expected to be lower in the south basin relative to the north basin through much of the open water season. Indeed, different light regimes between the two basins do support differences in the composition of the two phytoplankton communities (Sections 8.1, 9.1), and may in turn help explain differences in the overall trophic structures of these two regions of the lake.

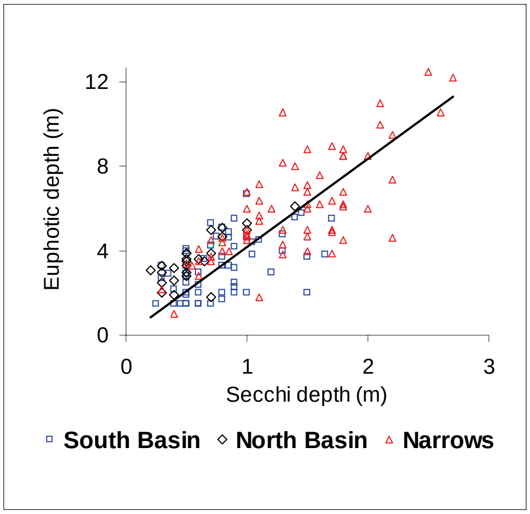


Figure 5.18: *Euphotic depth as a function of Secchi depth. Diagonal line shows the least squares fit of pooled data.*

Profiles of photosynthetically active radiation (PAR) in the water column recorded in 2007 and 2008 by Environment Canada quantify the effect of suspended solids in limiting the depth to which light penetrates the water column in the lake. The study included vertical profiles of both PAR and turbidity at about 60 standard stations (Appendix 1). Euphotic zone depth, that is, the depth down to which PAR in the water column is at least 1 % of down-welling PAR just above the lake surface, was determined from these profiles. Over all stations and all seasons sampled in 2008, the euphotic depth was generally three times to five times the Secchi depth (Table 5.4, Figure 5.18), although there was wide variability in this relationship. Euphotic depth was on average, about twice as deep in the north basin as in the south and tended to be shallower in autumn than in spring (Table 5.4).

The north and south basins of Lake Winnipeg are frequently considered as being similar to the central and western basins of Lake Erie in water transparency. In fact, these data show that Lake Winnipeg now is generally less clear than Lake Erie was at its most turbid in the 1970s before action was taken to reverse the severe eutrophication by reducing phosphorous loading to the lake. From

Table 5.4: *Mean and standard deviation (in parentheses) of turbidity (NTU), Secchi depths (m) and euphotic depths (m) recorded by Environment Canada during three whole lake surveys of Lake Winnipeg in 2008. Turbidity data have been averaged over the euphotic depth in each profile. Euphotic depth is here calculated as the depth through which measured PAR is >1% of PAR measured above the water surface.*

Basin	Turbidity	Secchi Depth	Euphotic Depth	Euphotic Depth/Secchi Depth	N
Spring					
South	17.8 (8.6)	1.0 (0.4)	4.6 (1.0)	4.4	19
Narrows	21.2 (7.9)	0.8 (0.3)	4.5 (0.9)	5.9	10
North	12.6 (7.1)	1.6 (0.5)	7.1 (2.5)	4.5	22
Summer					
South	32.7 (11.5)	0.6 (0.2)	3.2 (0.9)	5.2	20
Narrows	51.1 (5.1)	0.4 (0.1)	2.8 (0.5)	8.0	7
North	15.1 (9.1)	1.4 (0.6)	6.7 (2.9)	4.9	19
Autumn					
South	30.4 (10.5)	0.7 (0.3)	1.9 (0.7)	2.6	19
Narrows	46.2 (17.7)	0.6 (0.2)	3.1 (1.0)	5.6	8
North	19.4 (11.1)	1.2 (0.5)	4.7 (1.3)	3.8	24

1971 to 1973, June to August mean Secchi depths in the western basin of Lake Erie ranged from 1.2 to 2.2 m (Rathke and Edwards 1985), more than two times the 0.5 to 0.6 m in the south basin of Lake Winnipeg from 1999 to 2007. Mean Secchi depths in the central basin of Lake Erie ranged from 4.2 to 6.1 m, three to almost five times deeper than the 0.9 to 2.1 m in the north basin of Lake Winnipeg. Data describing euphotic depth also indicate that present-day Lake Winnipeg is more turbid than Lake Erie in the 1970s, although the differences are less than those for Sechhi depths. Through May to October, 1975, the mean euphotic depth in the western basin of Lake Erie was 4.0 to 6.0 m (Bukata *et al.* 1995) compared to the 3.2 m averaged over the three surveys of the south basin of Lake Winnipeg in 2008. The mean euphotic depth in the central basin of Lake Erie in 1975 was 10 to 15 m, about twice the mean of 6.2 m in the north basin of Lake Winnipeg in 2008. In the north basin of Lake Winnipeg, where the suspended solids load is predominantly biological, the lower transparency compared to central Lake Erie can be taken as evidence that Lake Winnipeg is currently more eutrophic than Lake Erie at its peak. On the other hand, in the south basin (and in the turbid north narrows region south of Berens Island), Secchi depth is predominantly determined by the concentration of suspended mineral, not organic, particles, and is not an indicator of phytoplankton biomass or productivity.

5.3.2 Seasonal and Spatial Variation

In 2002, total suspended solids concentrations were measured in Lake Winnipeg by Fisheries and Oceans Canada as one of several whole-lake expeditions aimed at gathering information on the spatial and seasonal variability of water quality throughout the lake (Appendix 1). The information provided by this expedition revealed the same general patterns of temporal and spatial variability in TSS in 2002 as from 1999 to 2007. There was a pronounced south-north (decreasing northward) gradient in all seasons and higher concentrations through the whole lake in fall compared to spring and summer (Figure 5.19).

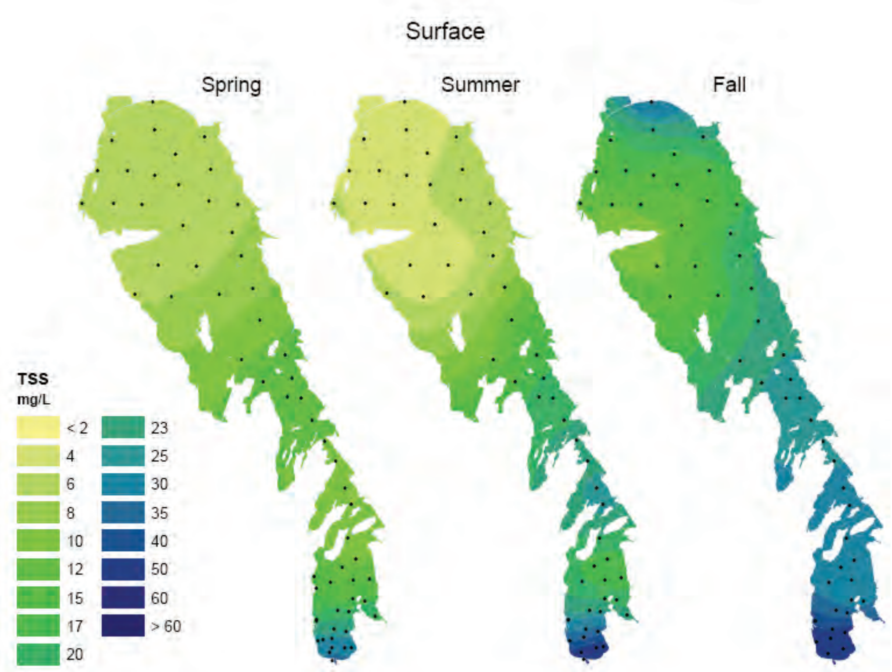


Figure 5.19: *TSS concentrations in Lake Winnipeg surface waters over the spring (May-June), summer (July-August), and fall (September-October) of 2002.*

In the south basin, TSS was highest near the mouth of the Red River. This is consistent with previous studies that have shown higher suspended sediment loading to the south basin by sediment-laden discharges from the Red River watershed (McCullough *et al.* 2001; Brunskill *et al.* 1980) which is underlain by deep glacial sediments. Concentrations were lower near the mouth of the Winnipeg River. It is a larger river, but it flows through a series of large lakes on the Precambrian Shield and develops an order of magnitude lower TSS than the Red. Discharges from the Winnipeg River dilute suspended solids in Traverse Bay at its mouth. In the narrows, in spring and fall, TSS was more uniformly distributed than in either the north or south basin. In the north basin, TSS tended to decrease both from south to north, and from east to west. Overall, concentrations were lowest north and northeast of Long Point. In part, low TSS here can be attributed to Saskatchewan River water which is like Winnipeg River water, low in suspended solids (Jones and Armstrong 2001). TSS was highest in the shallower waters south of Berens Island and in Dauphin Bay, and near the highly erodible clayey till shores at the north end of the lake.

Wind-induced resuspension also plays a significant role in this seasonal pattern (of TSS), contributing to elevated concentrations along the shallow eastern shore of the north basin and the shallow south basin as a whole.

In the south basin, TSS tends to be highest in autumn, although in years of flooding along the Red River, it peaks near the river mouth in April and May. The autumn peak is more readily explained by more frequent and stronger wind events towards the end of the open water season. As it is in the south basin, TSS tends also to peak in autumn in the north basin. Reasons include the progressive transfer of sediments from the south to the north through the open water season, and the cumulative production of biological matter over the course of the open water season. However, as in the south basin, wind-induced resuspension also plays a significant role in this seasonal pattern, contributing to elevated concentrations throughout the relatively shallow region south of Berens Island, and in shallow water along the eastern shore. The east side of the north basin is also the predominant path of flow carrying relatively turbid water from the south basin and narrows north towards the outflow of the lake (Text Box 3).

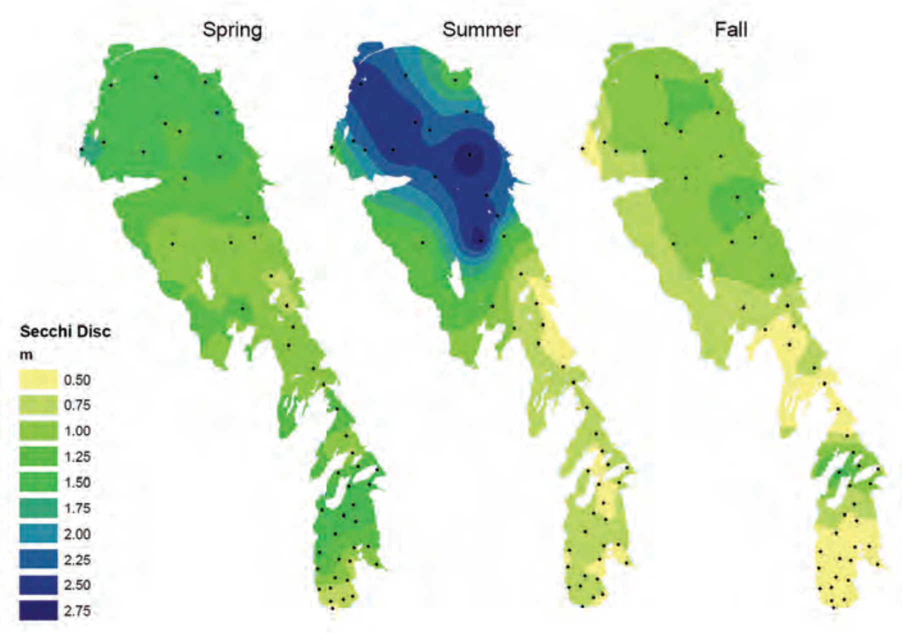


Figure 5.20: *Secchi disc depths in Lake Winnipeg surface waters over the spring (May-June), summer (July-August), and fall (September-October) of 2002.*

Similar spatial distributions and seasonal variations in TSS and associated turbidity in surface waters were found in a study conducted by McCullough *et al.* (2001) for Fisheries and Oceans Canada. Satellite remote sensing was used to document suspended sediment and productivity in Lake Winnipeg, providing more detail regarding spatial distributions than that provided by data acquired from sampling at long-term stations. Moreover, the latter are confined to the pelagic zone (Figures 5.16, 5.19) whereas satellite-derived maps show distributions throughout shallow regions not represented by the long-term stations. McCullough *et al.* (2001) examined satellite data from the U.S. National Oceanic and Aeronautical Administration's orbiting Advanced Very High Resolution Radiometer (AVHRR) recorded over the period of 1985 to 2000. TSS determined from these data were validated using *in situ* TSS determined on samples collected by Fisheries and Oceans Canada. There were no significant trends in TSS anywhere in the lake over the 15-year period.

McCullough *et al.* (2001) also tested their data for relationships between observed TSS and various climatic and hydrologic variables, using June to September data, for the period 1985 to 2000. Although, as expected, discharge from rivers into the lake was demonstrated to affect the spatial and seasonal distribution of suspended solids in the lake. In fact, antecedent winds were the most significant contributor to suspended sediment dynamics (Figure 5.21). Correlations between either six-hour mean or two-day mean antecedent wind and TSS were strongest along the eastern shore of the north basin, indicating a strong link between wind-generated littoral erosion and nearshore bottom resuspension of sediments there (Figure 5.19). While TSS and antecedent wind conditions were correlated in the south basin, the relationship was not as strong as in the north basin. Tributary inputs of suspended sediments (particularly from the Red River) likely play a stronger role in determining overall sediment concentrations in the south basin.

Although discharge from rivers into the lake also affects the spatial and seasonal distribution of suspended solids in Lake Winnipeg, antecedent winds were found to be the most significant contributor to suspended sediment dynamics

5.3.3 Vertical Variability

The preceding discussion has depended almost entirely on samples collected either in near-surface waters or integrated through the euphotic zone. In 2004, Fisheries and Oceans Canada collected paired near-surface and near-bottom samples, enabling an examination of the representativeness of surface samples in Lake Winnipeg. Excluding all pairs for which total suspended solids concentrations were <1 mg/L (near the analytical precision), near-bottom concentrations ranged from 9 mg/L lower to 127 mg/L higher than those near the surface, or in relative terms, near-bottom TSS ranged from 0.3 to 12 times near-surface TSS. However, in most cases, there were large relative differences between small absolute values of TSS; differences larger than 10 mg/L accounted for only 18 % of these paired observations. In relative terms, near-bottom concentrations of total suspended solids that were more than two times as large as those near-surface accounted for only 20 % of observations, indicating that although higher suspended solids concentrations occurred in bottom waters, waters were at least moderately well-mixed most of the time. This is consistent with the shallow depth and large fetches that characterizes Lake Winnipeg, such that wind-driven mixing can reach the bottom, even of the north basin, at least occasionally during most open water seasons. But interestingly, there is no significant relationship between either the absolute or the relative difference between near-bottom and near-surface concentrations and the total water column depth ($P = 0.761$ and $P = 0.659$ respectively, $n = 94$), indicating that some degree of suspended solids stratification may at times occur in either the relatively shallow south basin or the deeper north basin.

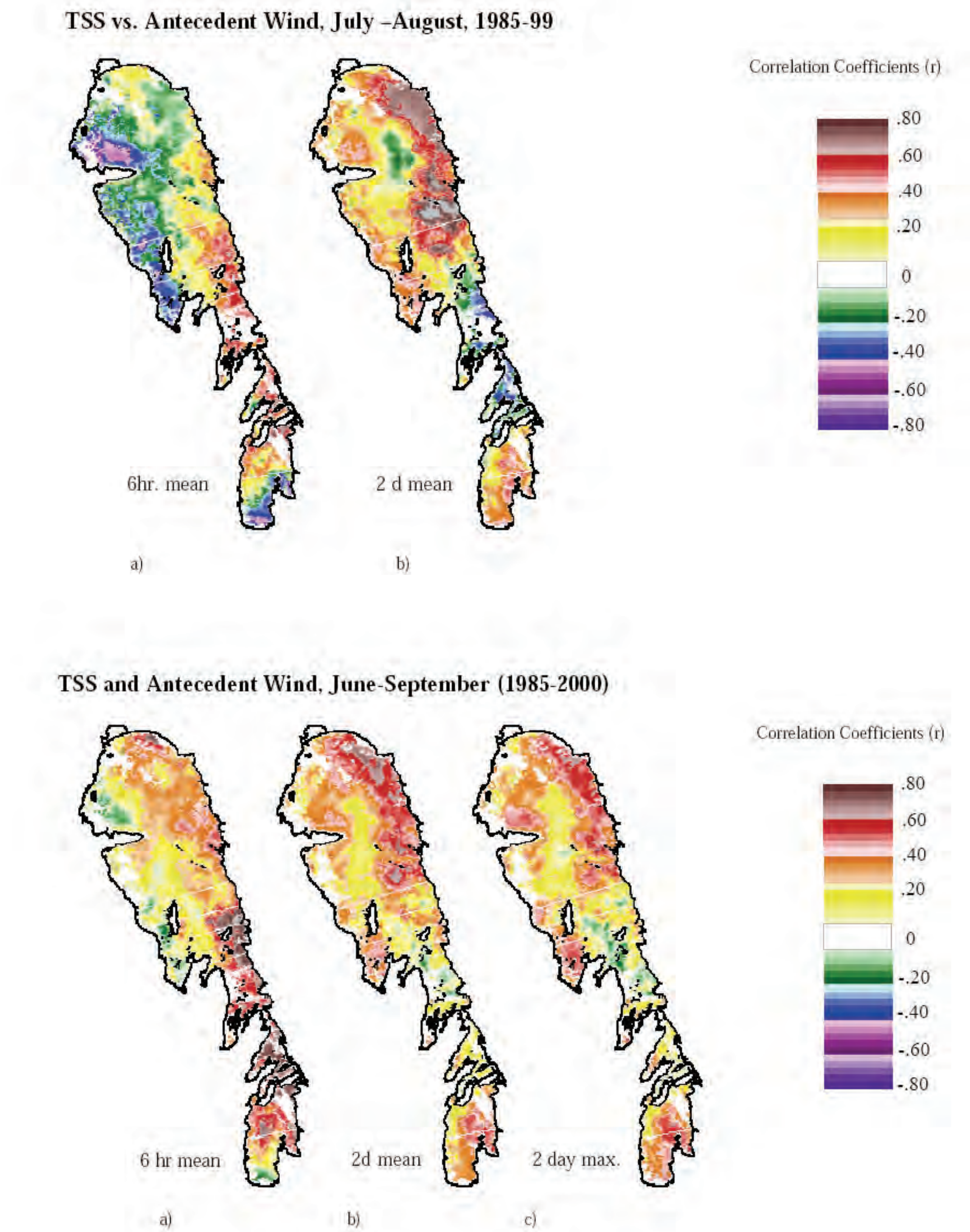


Figure 5.21 *Correlations (r) between TSS concentrations and antecedent wind for 1985 to 2000 late July-August (6-hour mean and 2-day mean wind; n = 12, P <0.05 for r >0.58, P <0 for r >0.71) and 1985 to 1999 for June-September (6-hour, 2-day mean, and 2-day maximum wind; n = 20, P <0.05 for r >0.45, P <0.01 for r >0.57). (Data Source: NOAA AVHRR)*

Turbidity profiles recorded by Environment Canada during a mid-summer cruise in 2008 (Figure 5.22) support these findings. Turbidity typically did not vary much with depth, except occasionally within 1 to 4 m of the bottom, where a more turbid layer was frequently observed. In one exceptional case, the upper 5 m of the water column were substantially more turbid than the lower 12 m. This same profile was marked by a chlorophyll a fluorescence maximum in the upper 5 m (Figure 5.22), so the turbid layer in the upper water column may have been due to phytoplankton rather than mineral suspended sediments. Turbidity was higher in the south basin, contributing to shallower PAR penetration into the water column as compared to the north basin (Figure 5.22). Large outliers, where they occurred within a few decimetres from the bottom, were removed prior to averaging. Consequently, excursions near the bottoms of these profiles represent substantial thicknesses of relatively turbid water lying near the bottom of the water column. In the north basin in particular, several of these near-bottom turbid layers are marked by high fluorescence, indicating the presence of organic matter, possibly healthy phytoplankton but more likely dying cells settling out of the population above. In the south basin, near-bottom turbid layers never exhibit elevated fluorescence, and the elevated turbidity near the bottom is more likely due to mineral particles (silt and clay) and organic detritus resuspended from the bottom sediment.

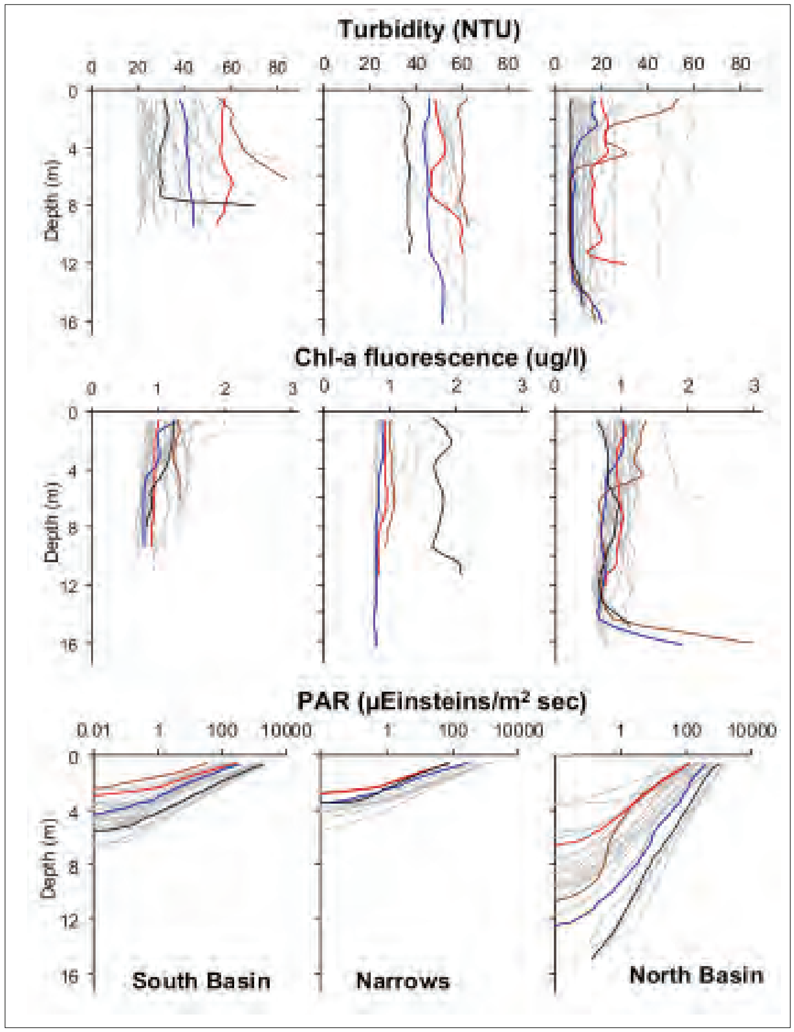


Figure 5.22: *Vertical profiles of turbidity (top panel: NTU), chlorophyll-a fluorescence (middle panel) and PAR (bottom panel) for stations in the south and north basins, and the narrows region. Colour coding is used to identify turbidity and PAR in identical CTD casts in the two panels. Data shown have been reduced to averages at 1 m intervals. Outliers in the raw backscatter data were removed prior to averaging, where they occurred within a few decimetres of the bottom. All profiles were recorded from 22 July to 15 August, 2008. (Data Source: Environment Canada, L. Wassenaar)*

Text Box 3. Physical Limnology and Hydrodynamic Modelling - May to October 2007

(Ram Yerubandi, Jun Zhao)

Lake Winnipeg is one of the largest freshwater bodies in the world, characterized by shallow waters that are highly susceptible to isothermal and turbid conditions due to wind action. Inputs from tributaries flowing to the south basin and resuspension of bottom sediments affect Lake Winnipeg, particularly with respect to nutrients. The movement of water masses and associated particulates (including nutrients) from the south basin to the north basin is important to the overall nutrient balance of the lake. Movement of water masses within basins is critical to this understanding as well. To increase the understanding of the circulation, thermal structure, and mixing of this shallow lake, and to advance development of hydrodynamic and water quality models, currents, water temperature, and meteorological parameters, light transmission and bottom layer oxygen were measured in the lake from May to October 2007.

Water temperature vertical profiles indicated that the shallow south basin (mooring 500, Plate 1) warmed more quickly than the deep north basin (mooring 504). The south basin and narrows remained isothermal, whereas the north basin was stratified from late June through much of July. Water quality measurements at two stations two metres above bottom indicated that dissolved oxygen decreased from 12.1 to 4.2 mg/L between June 11 and July 11, 2007, after which concentrations increased as stratification deteriorated. Turbidity peaked in association with episodes of high wind activity. Observations of currents in Lake Winnipeg indicated that the current moved predominantly northward in the south basin and narrows, and northeastward in the north basin toward the outflow from the lake.

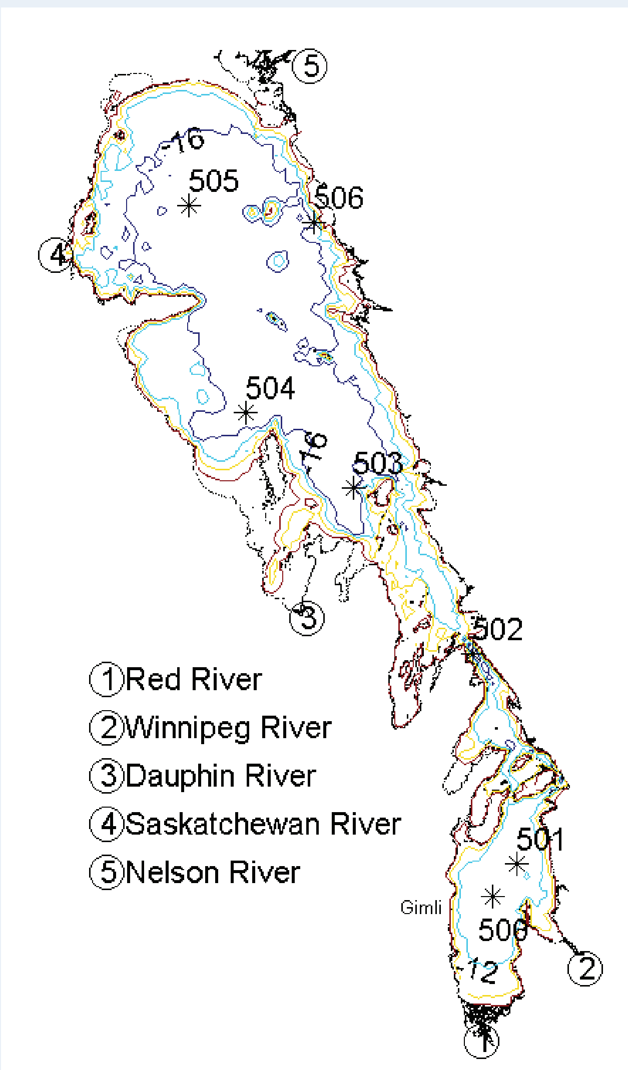


Plate 1. *Moorings were deployed and located within the north and south basins and narrows, as well as on shore.*

Circulation and the thermal structure of the lake were investigated with a three-dimensional hydrodynamic modelling system (Estuary Lake Coastal Ocean Model - ELCOM). The model successfully simulated the thermal structure, surface currents, and water levels of Lake Winnipeg. The large scale circulation pattern of Lake Winnipeg surface waters as simulated by the model was similar to the general pattern of wind over the area. Current in the north basin varied spatially, though generally flowed northeastward at the surface and north and northeastward at depth, with two cyclonic gyres near Long Point (Plate 2). Current in the narrows was primarily northward, while that in the south basin was characterized by a weak cyclonic gyre with a southward current along Winnipeg Beach on the west side and northward along Victoria Beach on the east side. These gyres may be significant to the transport, mixing, and fate of water quality constituents in Lake Winnipeg.

Continued...

Text Box 3 continued...

Circulation in Lake Winnipeg is highly variable, attributed to wind action on the surface of the lake. The exchange of water between the north and south basins through the narrows is influenced by seiches. Inter-basin and intra-basin circulation, and thermal and mixing dynamics may be of substantive importance to the movement and residence time of nutrients and biota between and within basins. The Estuary Lake Coastal Ocean Model simulations indicated that the model will be a useful tool for the advancement of water quality modelling and water resources management on Lake Winnipeg.

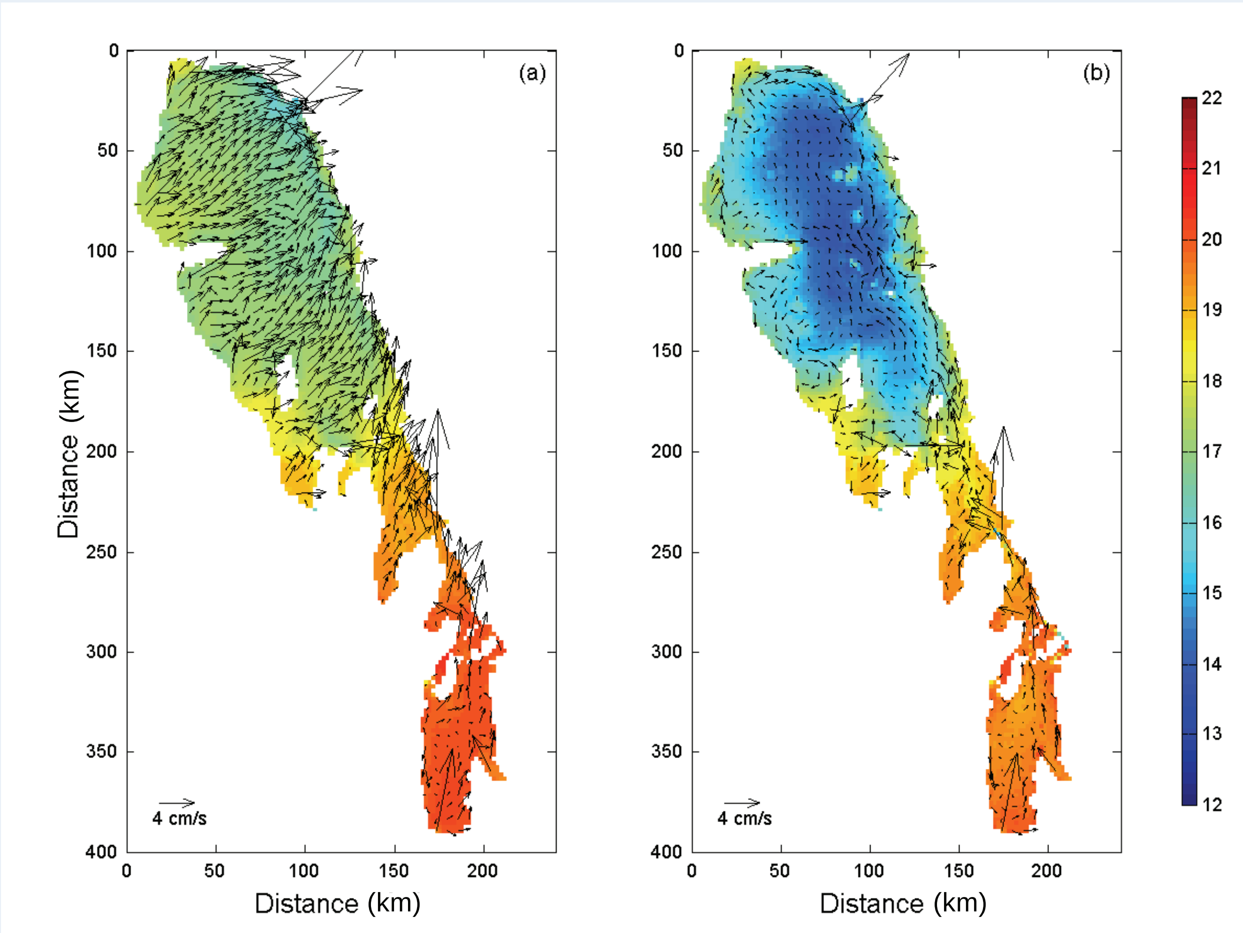


Plate 2. *Simulated mean summer circulation and temperature at the surface (left) and averaged over depth (right).*

6.0 CHEMICAL CHARACTERISTICS OF LAKE WINNIPEG

(Elaine Page)

Nitrogen and phosphorus are the major macronutrients that affect plant growth, biomass, and species composition in an aquatic environment. Nitrogen and phosphorus may also indirectly affect the growth, biomass, and composition of higher trophic levels including benthic invertebrates, zooplankton, and planktivorous and piscivorous fish. Nutrients occur in particulate and dissolved forms, the latter being more bio-available. Nutrients originate in soils, rocks, vegetation, and organic debris and are essential to a healthy functioning aquatic ecosystem. However, accelerated nitrogen and phosphorous loading from anthropogenic activities may negatively affect aquatic life, and is one of the largest problems facing many countries worldwide (Carpenter *et al*. 1998). Anthropogenic sources of nutrients include municipal and industrial wastewaters, fertilizers, livestock manure, and runoff from urban areas. However, because of the tremendous number of potential non-point and point sources of nutrients, tracking these sources and reducing nutrient levels in water bodies remains a challenge (Text Box 4).

Visible effects of accelerated nutrient enrichment in water bodies include an increase in phytoplankton biomass (e.g., McCauley *et al.* 1989; Dillon and Rigler 1974; Sakomoto 1966) which may be manifested as algal blooms, and a shift in community composition towards the dominance of cyanobacteria (Pick and Lean 1987; Text Box 5). While algal blooms are a natural phenomenon, particularly in naturally eutrophic prairie lakes where soils are rich in nutrients, blooms become more frequent and widespread with cultural eutrophication. Large blooms of cyanobacteria have been increasing in size and frequency in the north basin of Lake Winnipeg (Section 9.1). Nuisance blooms of cyanobacteria are typically favoured under high nutrient concentrations, low N:P ratios, and warmer water temperatures, all of which are characteristic of summer and fall conditions in Lake Winnipeg. Other common impacts of eutrophication in lakes include oxygen depletion (Section 5.2), release of algal toxins (Section 9.2), loss of species diversity, degradation of aquatic habitat, and reduction in water transparency.

Recent nutrient budget work suggested that Lake Winnipeg is retaining phosphorus and nitrogen.

Reservoirs and lakes with longer water residence times are particularly sensitive to accelerated nutrient loading given their efficiency in retaining phosphorus (Text Box 1). Although Lake Winnipeg has a relatively short water residence time, trend analysis for the Red, Winnipeg, Saskatchewan, and Nelson rivers, and recent nutrient budget work suggested that Lake Winnipeg is retaining phosphorus and nitrogen (Jones and Armstrong 2001; Section 7.0). Once nutrients are transported into a lake, they may be recycled through a number of processes. In the shallower south basin of Lake Winnipeg, nutrients may be entrained into the water column through wind-induced resuspension (Text Box 6). During low oxygen events, phosphorus and ammonia may be remobilized, making lake restoration efforts more challenging.

6.1 General Water Chemistry and Major Ions

Measurements of pH from 1999 to 2007 indicated that Lake Winnipeg was alkaline. These measurements were similar to values previously reported for Lake Winnipeg (Brunskill *et al.* 1979; Table 6.1). Although measurements of pH did not differ considerably between the north basin (8.00) and the south basin and narrows (7.93), lower individual measurements of pH typically occurred along the eastern shoreline of the lake near the inflows of the Winnipeg, Manigotogan, and Berens rivers.

The alkaline nature of the lake is related to the sedimentary geology that is characteristic of the southern and western parts of the drainage area (Brunskill *et al.* 1980). The eastern watershed drains soils, peatlands, and water bodies of the Precambrian Shield, where water tends to be more acidic. All individual measurements of pH were compared to the Manitoba Water Quality Standards, Objectives, and Guidelines (Williamson 2002) and were considered to be protective of aquatic life. Total and bicarbonate alkalinity were generally similar between basins, although concentrations were slightly lower in the south basin (Table 6.1). Mean alkalinity in 1969 was at the lower end of the range in mean concentrations for the 1999 to 2007 period. According to the B.C. Ministry of Environment, Lands and Parks classification for alkalinity (B.C. Ministry of Environment, Lands and Parks 1998), Lake Winnipeg has a low sensitivity to acidification (alkalinity >20 mg/L) and is considered to be well buffered, likely because of the underlying sedimentary geology.

Table 6.1: *Summary of water quality variables and major ions for the north basin and south basin and narrows of Lake Winnipeg. The 1999 to 2007 data were summarized as the mean of surface and euphotic samples during the open water season (May to October). The range represents the range in annual average values from 1999 to 2007. Mean values from 1969 represent samples collected at depths ranging from 0 to 4 m during the open water season. (Data sources: 1999 to 2007 from Manitoba Water Stewardship; 1969 from Department of Fisheries and Oceans)*

		North Basin			South Basin and Narrows		
			1999 to 2007			1999 to 2007	
Variable	Units	1969 Mean	Mean	Range	1969 Mean	Mean	Range
pH	pH Units	8.22	8.00	7.60 - 8.46	8.13	7.93	7.58 - 8.21
Total Alkalinity (as $CaCO_3$)	mg/L	97	97	86 - 104	85	93	84 - 115
Bicarbonate Alkalinity	mg/L	118	116	103 - 124	104	113	102 - 135
Specific Conductivity	µS/cm	282	313	262 - 355	250	287	232 - 360
Sulphate	mg/L	27	28	23 - 32	31	36	24 - 55
Calcium	mg/L	28	27	22 - 33	25	26	23 - 34
Sodium	mg/L	14	17	12 - 23	8.9	11	9.7 - 17
Chloride	mg/L	18	19	11 - 26	7.0	8.9	6.6 - 12
Magnesium	mg/L	12	11	9.1 - 13	11	11	10 - 16
Potassium	mg/L	2.4	2.7	2.5 - 3.1	2.7	3.4	2.9 - 4.3

Measurements of specific conductance from 1999 to 2007 indicated that Lake Winnipeg waters were fresh (<1500 us/cm; Table 6.1). Specific conductance was comparatively higher in the north basin (313 µS/cm) relative to measurements in the south basin and narrows (287 µS/cm) during that period. Spatial differences in specific conductance are likely associated with the influence of the Dauphin River, which transported ion-rich waters (1200 to 1600 µS/cm) from Lake Manitoba into the north basin of Lake Winnipeg. Waters in the south basin and narrows were also diluted by the inflow of the Winnipeg River which had a comparatively lower specific conductance (100 µS/cm). Specific conductance varies annually with changes in evaporation, flow, and precipitation, and is typically higher in years with elevated rates of evaporation, lower precipitation, and reduced river inflow.

Runoff waters from urban and agricultural areas are typically high in dissolved salts and may also contribute to the variability in specific conductance. Measurements of specific conductance by Fisheries and Oceans Canada in 1969 (Brunskill *et al.* 1979) were low in comparison to the 1999 to 2007 average (Table 6.1). However, the 1969 basin averages were within the range of specific conductance measurements recorded from 1999 to 2007. Lower specific conductance in 1969 may be at least partially explained by higher annual water levels (218.12 masl) relative to the 1999 to 2007 period (range 217.09 masl to 217.84 masl) with exception of 2005, where lake levels were almost as equally high, reaching 218.03 masl.

Bicarbonate, sulphate, and calcium were the dominant ions in the lake from 1999 to 2007.

Bicarbonate, sulphate, and calcium were the dominant ions in the lake from 1999 to 2007 (Table 6.1). Dominant ion chemistry was generally similar between the two lake basins with the exception of sulphate. Sulphate was approximately 20 % higher in the south basin (36 mg/L) in comparison to the north basin (28 mg/L). Basin differences in sulphate were associated with the influence of the Red River, which carried higher concentrations of sulphate (134 mg/L) into the south basin of Lake Winnipeg. In comparison to the 1999 to 2007 average sulphate concentration in the south basin and narrows, sulphate appeared lower in 1969, but was within the range of the most recent (1999 to 2007) concentrations (Table 6.1). Although sodium (Na) and chloride (Cl) were not dominant ions in Lake Winnipeg, concentrations were approximately two times greater in the north basin (Na = 17 mg/L and Cl = 19 mg/L) relative to the south basin and narrows (Na = 11 mg/L and Cl = 9 mg/L). Basin differences were associated with the inflow of the Dauphin River, which transported higher average concentrations of chloride (305 mg/L) and sodium (184 mg/L) to the north basin of the lake.

Studies on Lake Winnipeg chemistry in the late 1960s attributed the higher chloride concentrations in the north basin to the inflow of the Dauphin River (Brunskill *et al.* 1979). Lower average concentrations of sodium and chloride in the south basin and narrows were also influenced by the inflow of the Winnipeg River (Na = 2 mg/L and Cl = 1 mg/L) which effectively dilutes water in the south basin. Comparisons to 1969 data indicated that sodium was almost 20 % lower as compared to the 1999 to 2007 period, and the average 1969 concentration for the south basin and narrows (9 mg/L) was just outside of the 1999 to 2007 range for sodium (10 to 17 mg/L). As with specific conductance, it is possible that sodium was diluted because of higher than normal lake water levels in 1969.

6.2 Phosphorus

6.2.1 Recent Conditions: 1999 to 2007

From 1999 to 2007, Manitoba Water Stewardship measured phosphorous concentrations in Lake Winnipeg during the open water season at long-term water quality monitoring stations through the north and south basins and narrows (Appendix 1a). The average total phosphorous concentration was almost three times higher in the south basin (0.113 mg/L) compared to the north basin (0.044 mg/L; Table 6.2). Considering all individual surface and euphotic zone samples collected during that period, the range in total phosphorous concentrations was approximately four times greater in the south basin and narrows (from 0.016 to 0.508 mg/L) as compared to the north basin (ranged from 0.005 to 0.13 mg/L). Basin differences were associated with the nutrient-rich inflow of the Red River, which transported higher average concentrations of phosphorus (0.354 mg/L) to the south basin of the lake. The Winnipeg River, which is comparatively low in phosphorus (0.027 mg/L), dilutes water in the south basin, particularly along the length of the eastern side of the south basin (see Section 6.2.2). Eighty-nine percent of individual samples (n = 773) exceeded the narrative Manitoba Water Quality Guideline for total phosphorus (0.025 mg/L in lakes; Williamson 2002). Of the samples that exceeded the narrative guideline, 60 % occurred in the south basin and narrows region of the lake.

Total phosphorous concentrations were almost three times higher in the south basin as compared to the north basin.

Table 6.2: *Total phosphorous concentrations (mg/L) in Lake Winnipeg from 1999 to 2007. Data represent the season-weighted means of all surface and eutrophic zone samples from May to October. (Data source: Manitoba Water Stewardship)*

	North Basin				South Basin and Narrows			
Year	**Mean**	**Min**	**Max**	**n**	**Mean**	**Min**	**Max**	**n**
1999	0.045	0.024	0.130	21	0.100	0.030	0.232	32
2000	0.036	0.013	0,091	29	0.078	0.040	0.166	24
2001	0.054	0.008	0.231	23	0.118	0.033	0.215	15
2002	0.045	0.014	0.116	85	0.109	0.023	0.296	87
2003	0.031	0.017	0.069	13	0.117	0.029	0.252	14
2004	0.044	0.014	0.204	30	0.099	0.028	0.200	14
2005	0.061	0.028	0.103	35	0.138	0.026	0.508	51
2006	0.038	0.009	0.080	60	0.124	0.030	0.414	82
2007	0.046	0.005	0.095	59	0.131	0.028	0.390	98
Average 1999 to 2007	0.044				0.113			

Annual mean total phosphorous concentrations were highest in 2005 in both the north basin (0.061 mg/L) and the south basin and narrows region (0.138 mg/L) of the lake (Figure 6.1). These high phosphorous concentrations corresponded to high flows in the major tributaries that year because of heavy spring and summer precipitation events. This represents the greatest total phosphorous load delivered to the lake of any year from 1994 to 2007 (13,043 tonnes/year (t/y); Section 7.1). Mean annual total phosphorous concentrations were lowest in the north basin in 2003 (0.031 mg/L) and in the south basin and narrows in 2000 (0.078 mg/L) when inflow and phosphorous loads to the lake were relatively low. The average annual total phosphorous concentrations for the south basin and narrows of Lake Winnipeg exceeded 0.1 mg/L in most years. Average total phosphorous concentrations in the south basin and narrows appeared higher from 2005 to 2007 relative to concentrations from 1999 to 2004. However, this pattern was not apparent in the north basin (Figure 6.1).

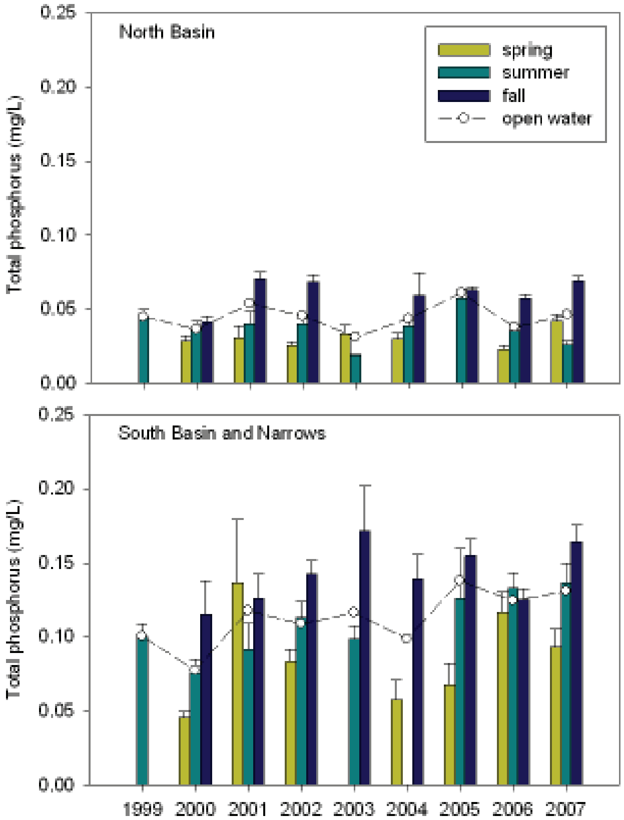


***Figure 6.1:** Annual and seasonal mean total phosphorous concentrations (± SE) in the north basin and south basin (and narrows) of Lake Winnipeg from 1999 to 2007. Concentrations represent water samples collected from the surface and euphotic zone (n range for north basin = 13 to 85; narrows and south basin = 14 to 98) from May to October.*

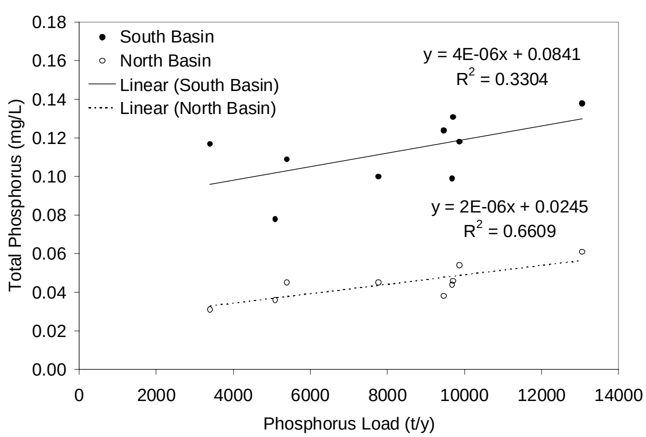


***Figure 6.2:** Relationship between total phosphorous loading (Section 7.1) and mean annual total phosphorous concentrations (surface and euphotic zone samples; May to October) in the north basin and south basin and narrows of Lake Winnipeg.*

Except for 2001, concentrations of total phosphorus in the south basin and narrows were lowest in spring and generally highest in fall (Figure 6.1). Seasonal patterns in total phosphorus are similar to those for total suspended solids (Section 5.3). However, higher concentrations of total phosphorus also occur subsequent to peak summer temperatures (Section 5.1) that drive biological productivity in the lake and the production of particulate organic matter and associated phosphorus.

Mean open water phosphorous concentrations in Lake Winnipeg were positively correlated with total phosphorous loading to the lake (Figure 6.2). However, the relationship between load and in-lake phosphorous concentrations was much weaker for the south basin which highlights the complexity of other processes affecting nutrient cycling such as sedimentation and wind induced resuspension.

In comparison to the 1969 average phosphorous concentration in the south basin and narrows (0.096 mg/L) and the north basin (0.035 mg/L), phosphorus was within the range of the most recent (1999 to 2007) concentrations. The 1969 mean phosphorous concentration for the north basin was at the lower end of the 1999 to 2007 range (0.031 to 0.061 mg/L), whereas the 1969 phosphorous concentration was within the middle of the 1999 to 2007 range for the south basin and narrows (0.078 to 0.138 mg/L). A study to assess the impact of the 1997 Red River flooding on the south basin (Stewart *et al.* 2000) compared data collected by Fisheries and Oceans Canada in 1969 and data collected by Manitoba Environment in the 1990s (1992, 1994, 1996, and 1998) and found no apparent increase in total phosphorous concentrations in the south basin between these two time periods. A subsequent review of Lake Winnipeg nutrient data (North South Consultants 2006) found that the 1969 nutrient concentrations were at the lower end of the range of nutrient concentrations for the 1992 to 2005 period. However, similar to Stewart *et al.* (2000), the review also indicated that the 1969 data in the south basin were similar to mean values at the higher end of the range (perhaps indicating no substantial change in the south basin).

Other lakes in the world experiencing nutrient enrichment include the large (3,555 km^2), shallow (mean depth = 7.1 m) Lake Peipsi in Estonia, which has a comparatively lower phosphorous concentration (0.042 mg/L; Nõges *et al.* 2010) than Lake Winnipeg. In western Lake Erie, total phosphorous concentrations were comparatively low in more recent years, with average spring concentrations of approximately 0.020 mg/L (Millard and Howell 2009). In comparison to Lake Erie, spring phosphorous concentrations in Lake Winnipeg were approximately two to four times higher in the north basin (range 0.034 to 0.081 mg/L) and two to six times higher in the south basin and narrows (range 0.040 to 0.116 mg/L). Recent spring total phosphorous concentrations for Lake Simcoe (0.013 mg/L) were similar to those reported for Lake Erie (Young *et al.* 2010).

Annual mean total phosphorus data from 1999 to 2007 for each of Lake Winnipeg's basins were compared to the Organisation for Economic Cooperation and Development trophic classification (Organisation for Economic Cooperation and Development 1982). Trophic classification of lakes has been used to assess the relative degree of nutrient enrichment and biological production in lakes. Lakes are classified as oligotrophic, mesotrophic, eutrophic, and hypereutrophic on the basis of annual mean total phosphorous concentrations, algal biomass (as chlorophyll a), and water clarity. An oligotrophic lake has low concentrations of phosphorus and chlorophyll a and high water clarity, whereas a hypereutrophic lake has high concentrations of phosphorus and chlorophyll a and low water clarity. According to the Organisation for Economic Cooperation and Development trophic classification for total phosphorus, the south basin and narrows region of Lake Winnipeg was considered eutrophic in 2000 (0.078 mg/L) and 2004 (0.099 mg/L) and hypereutrophic (>0.1 mg/L) in all other years. The north basin was generally considered eutrophic from 1999 to 2007 with exception of 2003, when the mean annual total phosphorous concentration was lowest. That year the north basin was considered mesoeutrophic.

From 1999 to 2007, particulate phosphorus generally comprised a greater fraction of total phosphorus in the north basin of the lake (Figure 6.3). The average annual particulate phosphorous fraction ranged from 41 % in 1999 to as high as 60 % in 2004 and 2006. The relative proportions of dissolved phosphorus in the north basin of the lake are similar to those in the Saskatchewan River where the average (1994 to 2007) dissolved phosphorous fraction varied from approximately 25 to 45 % from May to September (Section 7.1). In the south basin, the percent contribution of dissolved and particulate phosphorus was much more variable from year to year compared to the north basin. Particulate phosphorus accounted for 36 % of the total phosphorus in 1999 and as much as 64 % in 2005 when tributary inflows and total phosphorous loads to the lake were highest.

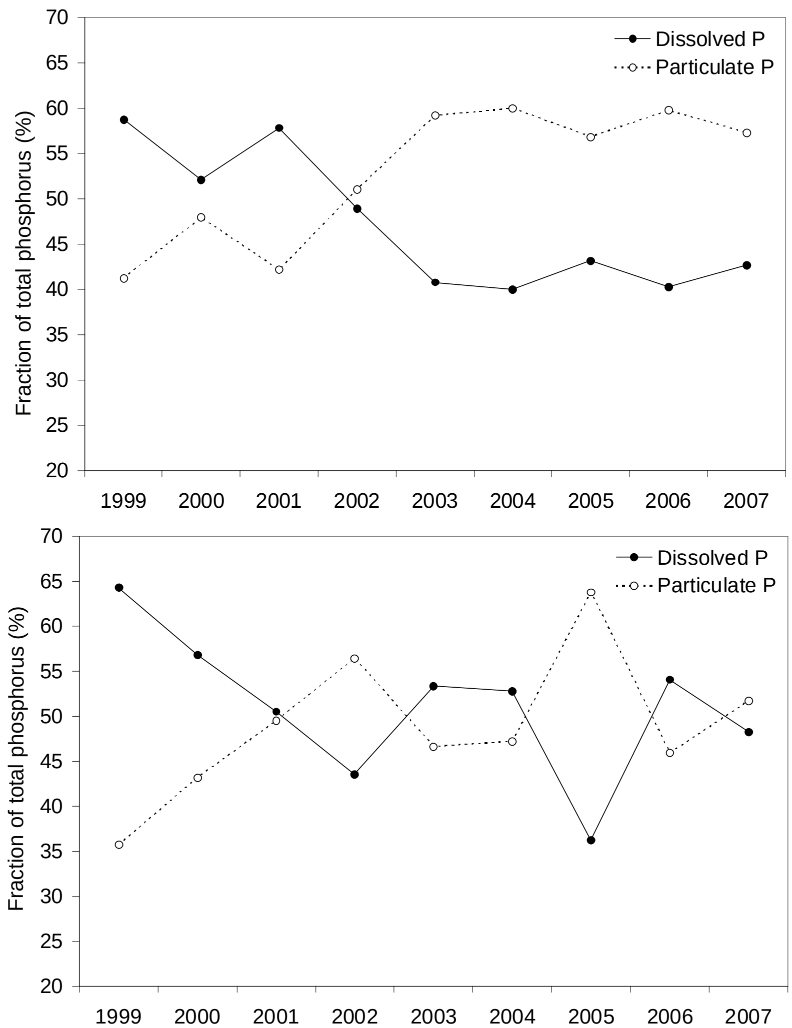


Figure 6.3: *Average annual open water (May to October) percent dissolved and particulate phosphorus in the north basin (top panel) and the narrows and south basin (bottom panel) of Lake Winnipeg from 1999 to 2007.*

6.2.2 Seasonal and Spatial Variation

In 2002, Fisheries and Oceans Canada measured phosphorous concentrations in Lake Winnipeg as part of a whole-lake expedition (Appendix 1). Total, particulate, and dissolved phosphorus were shown to vary seasonally and spatially in Lake Winnipeg. In 2002, north-south gradients were generally evident with higher phosphorous concentrations in the south basin near the inflow of the Red River and lower concentrations in the north basin, particularly north of Long Point near the inflow of the Saskatchewan River (Figure 6.4, on facing page). The north-south gradient is not surprising given the higher phosphorous loads entering Lake Winnipeg through the Red River as compared to the Saskatchewan River (Section 7.1). Total phosphorous concentrations remained fairly constant in the north basin during the spring and summer and increased to approximately 0.06 mg/L in the fall. In the south basin, there was a progressive increase in total phosphorus from approximately 0.05 to 0.06 mg/L in the spring to about 0.1 mg/L in the fall.

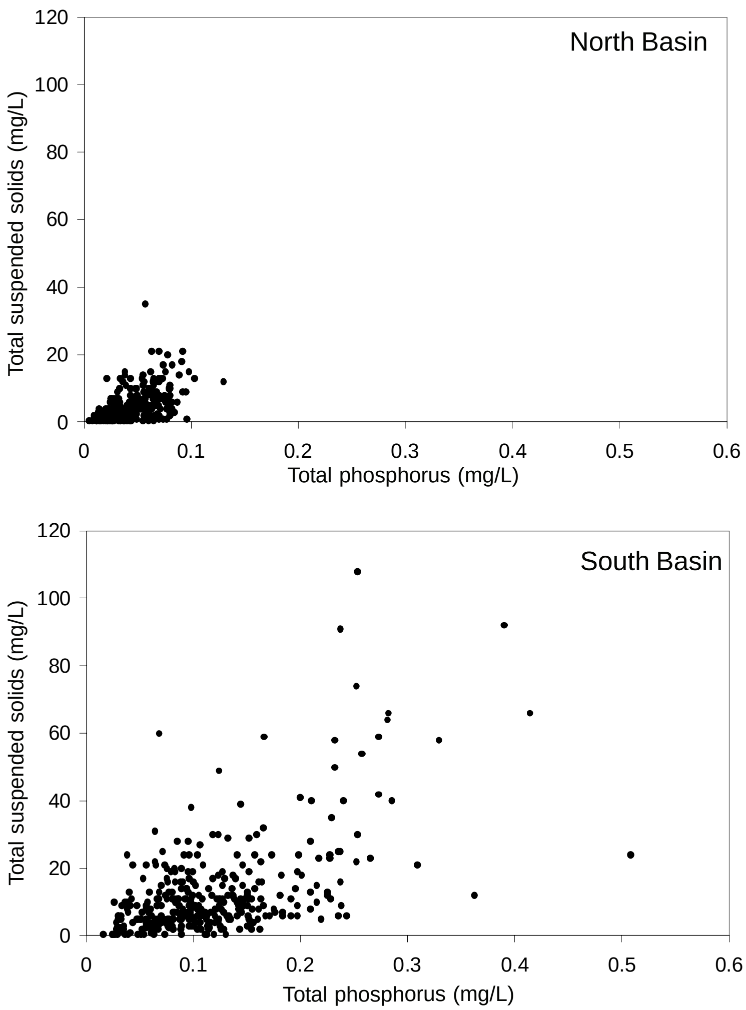


Figure 6.5: *Relationship between total phosphorus and total suspended solids concentrations in the north basin and the south basin and narrows region of Lake Winnipeg (1999 to 2007). Values represent both surface/euphotic zone samples and samples collected 0.5 m above the lake bottom.*

Particulate and dissolved phosphorus were also highest in the fall, subsequent to peak river discharges and peak biological productivity in the spring and summer. However, seasonal patterns in the dissolved and particulate fractions differed from seasonal patterns in total phosphorus (Figure 6.4). In the north basin, particulate phosphorus peaked in the spring, declined in the summer, and increased to approximately 0.05 mg/L in the fall. The most evident decline in particulate phosphorus in the north basin during the summer of 2002 occurred north of Long Point where water transparency is typically highest in the lake (Figure 5.20). Particulate phosphorous concentrations were generally similar over all seasons in the south basin. Dissolved phosphorus was lowest in both basins during the spring and increased to a maximum concentration of about 0.05 mg/L in the north basin and between 0.08 and 0.12 mg/L in the south basin. Dissolved phosphorus was highest at the very south end of the lake near the inflow of the Red River and was lower along the eastern edge of the south basin in Traverse Bay near the inflow of the comparatively nutrient-poor Winnipeg River.

Higher total phosphorous concentrations were generally associated with high suspended solids concentrations in the lake (Figure 6.5). This may be related to wind-induced resuspension of lake sediments, particularly in the south basin where relatively shallow lake depths (<14 m; average depth of 9.7 m) allow for considerable sediment-water interaction. High phosphorous concentrations are also associated with sediment-laden inflows to the south basin of the lake, where suspended solids are generally >30 mg/L near the mouth of the Red River (Figure 5.19).

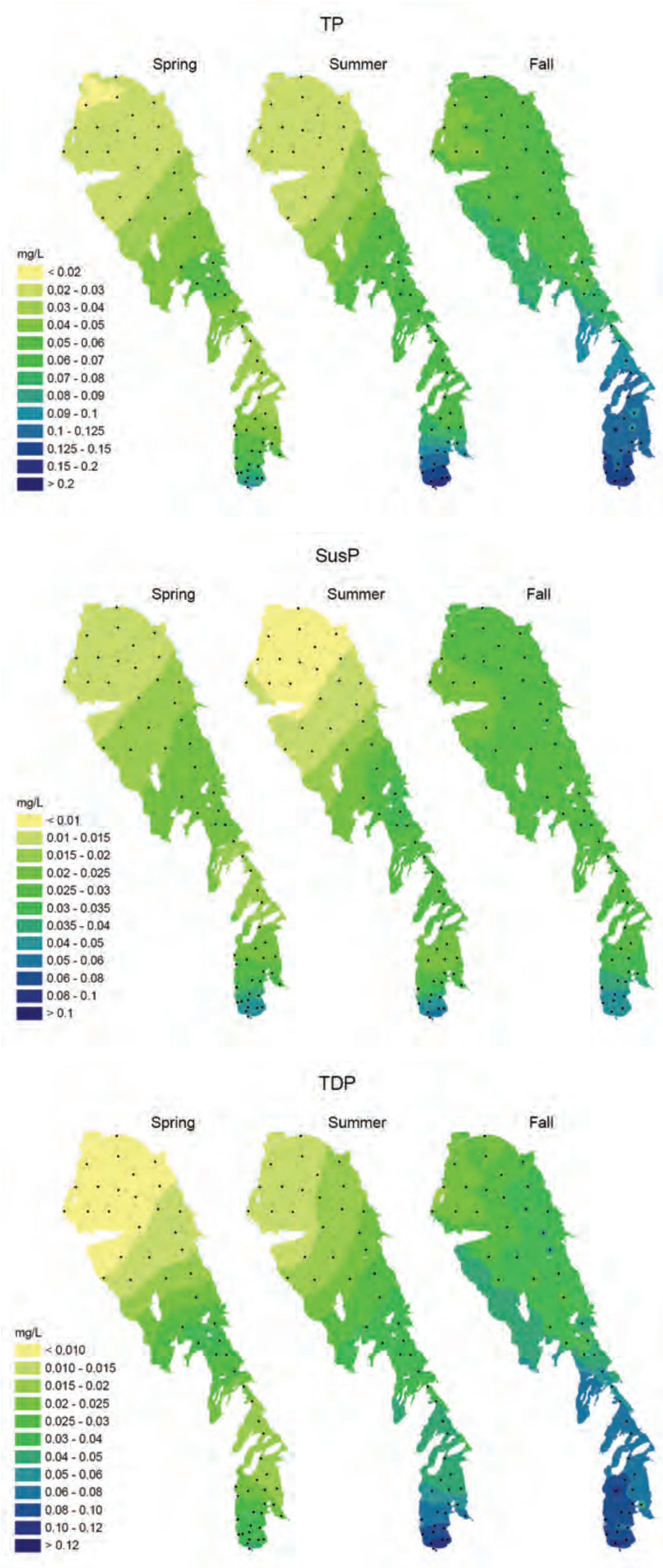


Figure 6.4: *Total phosphorus, particulate phosphorus, and dissolved phosphorus in Lake Winnipeg during the spring, summer, and fall of 2002. Note the change in scale.*

6.3 Nitrogen

6.3.1 Recent Conditions: 1999 to 2007

From 1999 to 2007, Manitoba Water Stewardship measured nitrogen concentrations in Lake Winnipeg over the open water season at long-term water quality monitoring stations throughout the north and south basins and narrows (Appendix 1a). Considering all individual samples collected during the open water season, total nitrogen ranged from 0.105 to 2.75 mg/L in the north basin and from 0.105 to 3.45 mg/L in the south basin and narrows. Currently, there is no total nitrogen guideline for the protection of aquatic life in Manitoba. However, other jurisdictions have adopted a chronic total nitrogen guideline of 1.0 mg/L (Alberta Environment 1999). Twenty percent of all total nitrogen samples taken on Lake Winnipeg (n = 722) exceeded the Alberta water quality guideline. Of the total nitrogen values that exceeded the 1.0 mg/L guideline, 79 % were from the south basin.

Total nitrogen concentrations were quite variable from year to year and there appeared to be no clear change from 1999 to 2007.

The average open water total nitrogen concentration (1999 to 2007) was 0.653 mg/L for the north basin and 0.869 mg/L for the south basin and narrows (Table 6.3). The range in total nitrogen concentrations in the north basin and south basin and narrows of Lake Winnipeg were appreciably greater in comparison to recent total nitrogen averages for Lake Simcoe (range 0.44 to 0.56 mg/L; Young *et al.* 2010). From 1999 to 2007, annual average open water nitrogen concentrations ranged from a minimum of 0.27 mg/L in 2002 to a maximum 1.02 mg/L in 2003 in the north basin of Lake Winnipeg (Figure 6.6). In the south basin, the lowest annual total nitrogen concentration also occurred in 2002 (0.48 mg/L) and was highest in 2006 (1.21 mg/L) when algal blooms covered much of the north basin during the summer and fall (Section 8.1). Total nitrogen concentrations were quite variable from year to year and there appeared to be no clear change from 1999 to 2007. In comparison, the 1969 mean total nitrogen concentrations for the north basin (0.40 mg/L) and the south basin and narrows (0.64 mg/L) were within the ranges for 1999 to 2007, but were generally at the mid- to lower end of the ranges. In most years, total nitrogen concentrations in the north basin increased over the open water season and were generally highest in the fall, most notably in 2004, 2006, and 2007 (Figure 6.6).

Table 6.3: *Total nitrogen concentrations (mg/L) in Lake Winnipeg from 1999 to 2007. Data represent the season-weighted means of all surface and euphotic zone samples from May to October.*

	North Basin				South Basin and Narrows			
Year	**Mean**	**Min**	**Max**	**n**	**Mean**	**Min**	**Max**	**n**
1999	0.778	0.505	1.61	21	0.838	0.405	1.60	32
2000	0.535	0.305	0.92	29	0.788	0.430	1.56	24
2001	0.571	0.120	1.13	22	1.077	0.120	2.62	15
2002	0.267	0.105	0.81	85	0.480	0.105	1.91	87
2003	1.023	0.405	1.81	13	0.929	0.105	2.90	14
2004	0.830	0.105	2.03	30	0.896	0.105	2.25	14
2005	0.673	0.110	2.44	35	0.782	0.105	1.97	51
2006	0.677	0.305	1.55	59	1.214	0.305	2.93	82
2007	0.525	0.105	1.71	59	0.819	0.105	2.42	98
Average 1999 to 2007	0.653				0.869			

Elevated fall nitrogen concentrations in the north basin may be attributable to peak biological nitrogen fixation in the summer. In contrast, total nitrogen concentrations in the south basin and narrows were generally highest during the spring and lower in the summer and fall. Lower total nitrogen concentrations in the south basin during the summer and fall may be associated with lower nutrient loads from the Red and Winnipeg rivers and peak biological uptake during this period.

Dissolved inorganic nitrogen concentrations in the south basin were approximately twice those in the north basin of Lake Winnipeg (Figure 6.7). High nitrogen loading from the Red and Winnipeg rivers as compared to the Saskatchewan River (Section 7.2) likely influences this north-south gradient in concentrations. Percent dissolved inorganic nitrogen in the north basin did not vary in a manner similar to that in the south basin from 1999 to 2007. Dissolved inorganic nitrogen in the north basin ranged from 4 % of total nitrogen in 1999 to 31 % in 2001. Dissolved inorganic nitrogen in the south basin comprised between 12 % of total nitrogen in 2000 and 45 % in 2003.

Ammonia concentrations were compared against the Manitoba Water Quality Objectives for the protection of aquatic life (chronic objectives, 30-day averaging duration for cool water species; Williamson 2002) that are associated with high pH and water temperatures typical of summer months. Chronic objectives for ammonia ranged from 0.51 to 5.93 mg/L in the summer (at water temperatures >5°C). All ammonia concentrations measured during the summer in Lake Winnipeg from 1999 to 2007 met chronic objectives for ammonia, as ammonia concentrations never exceeded 0.480 mg/L in the lake. Similarly, all ammonia concentrations measured during the winter met chronic objectives for ammonia. Winter ammonia objectives (at water temperatures <5°C) were much higher than those calculated for the summer (>1.3 mg/L at all sites) whereas actual ammonia concentrations ranged from <0.01 to 0.820 mg/L.

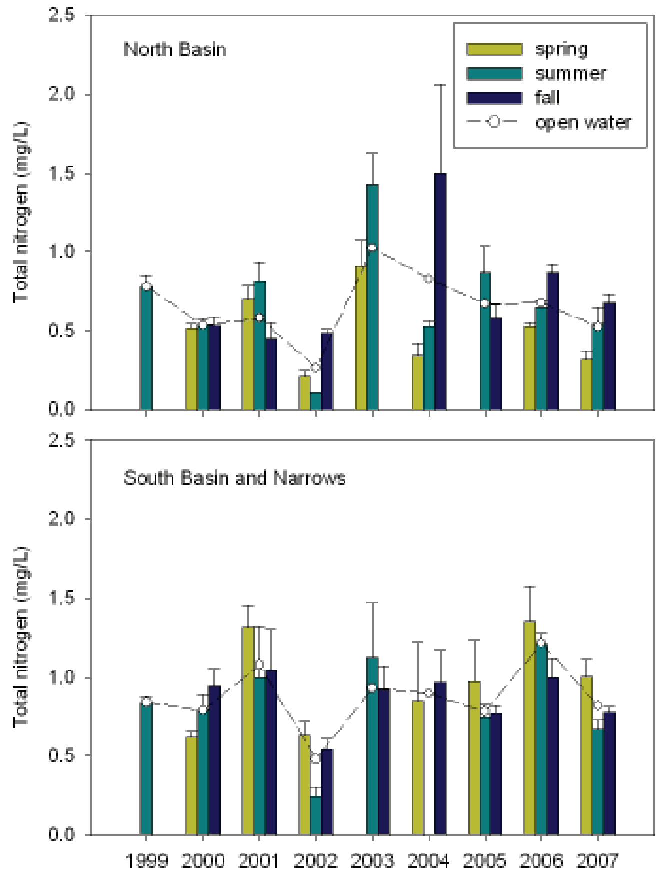


Figure 6.6: *Annual mean open water (May to October) total nitrogen (± SE) in the north basin and south basin and narrows of Lake Winnipeg from 1999 to 2007. Concentrations represent water samples collected from the surface and euphotic zone of Lake Winnipeg (n range for north basin = 13 to 85; narrows and south basin = 14 to 98).*

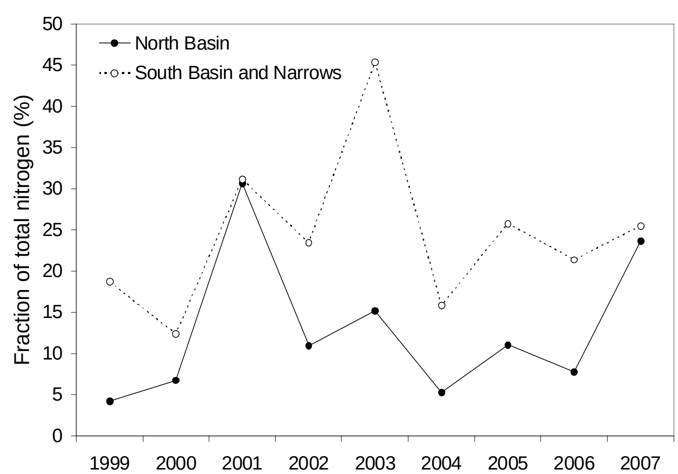


Figure 6.7: *Average annual open-water (May to October) percent dissolved inorganic nitrogen in the north basin and south basin and narrows of Lake Winnipeg from 1999 to 2007.*

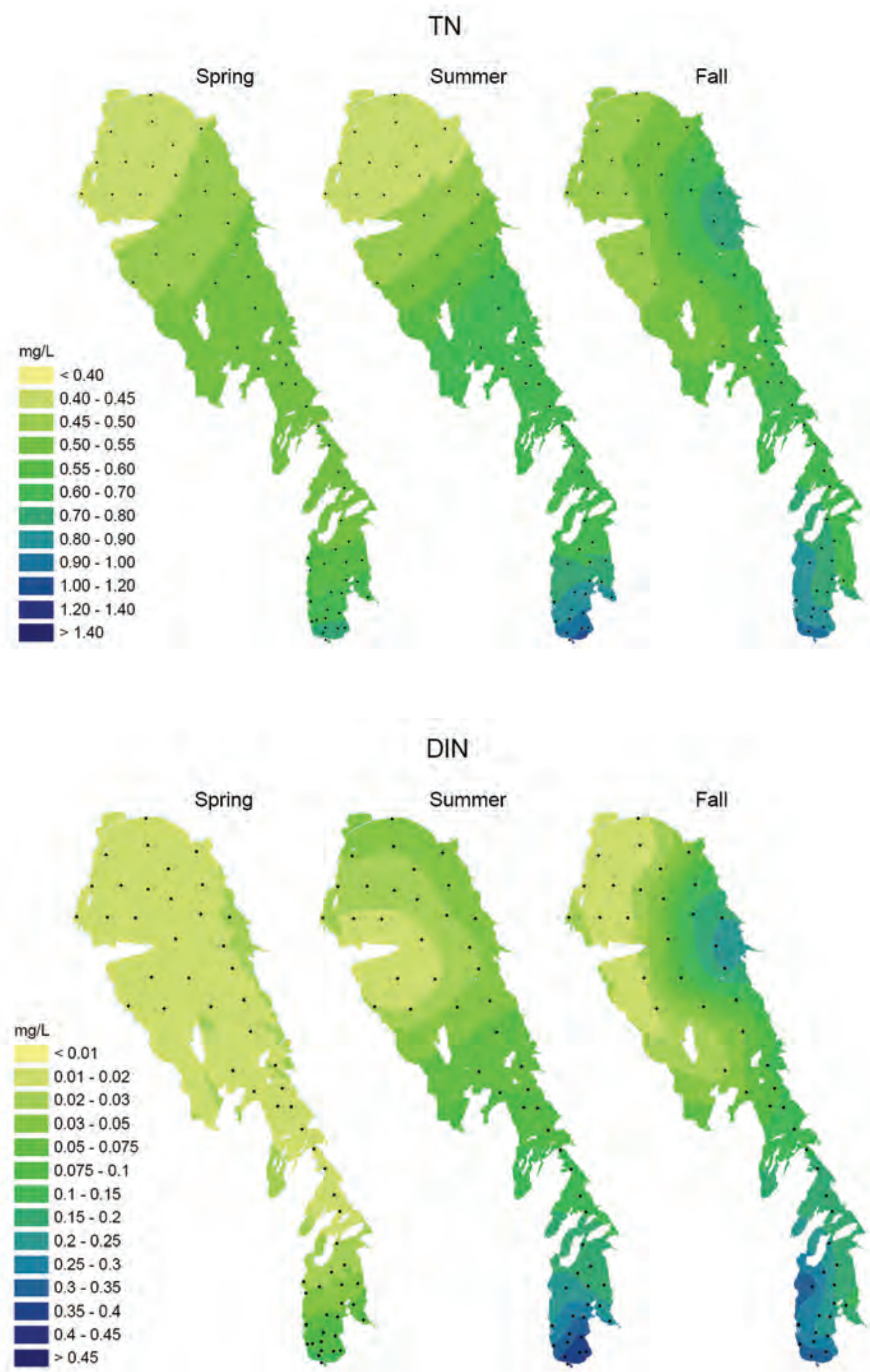


Figure 6.8: *Total nitrogen and dissolved inorganic nitrogen in Lake Winnipeg during the spring, summer, and fall of 2002.*

6.3.2 Seasonal and Spatial Variation

In 2002, Fisheries and Oceans Canada measured nitrogen concentrations in Lake Winnipeg as part of a whole-lake expedition (Appendix 1). Spring total nitrogen concentrations ranged from 0.45 mg/L in the north basin to 0.80 mg/L at the very south end of the lake near the inflow of the Red River (Figure 6.8). During the summer, total nitrogen concentrations increased to approximately 1.0 mg/L in the south basin particularly near the mouth of the Red River which carries high nutrient loads (Section 7.2). Total nitrogen concentrations remained very similar between spring and summer in the north basin of the lake with concentrations ranging from 0.45 mg/L to approximately 0.6 mg/L. In the fall, total nitrogen concentrations were greater than 0.5 mg/L in the north basin and were slightly higher along the east side (approximately 0.7 mg/L), where total suspended solids concentrations were elevated, and where wind-driven resuspension is common (Figure 5.18 and Section 5.3.2) and the northeastward circulation pattern within the lake (Text Box 3, Section 5.1). Total nitrogen concentrations were similar in the summer and fall in the south basin. However, during the fall, the highest total nitrogen concentrations (approximately 1.0 mg/L) were prevalent along the west side of the south basin (Figure 6.8).

Dissolved inorganic nitrogen concentrations were relatively low during the spring of 2002 and ranged from 0.02 mg/L in the north basin to 0.05 mg/L in the south basin. In the summer, dissolved inorganic nitrogen concentrations were approximately two times higher than in spring, with concentrations ranging from 0.02 mg/L to as high as 0.45 mg/L in the south basin. Dissolved inorganic nitrogen concentrations did not increase in the summer in the vicinity of Long Point (approximately 0.02 mg/L), reflecting low dissolved inorganic nitrogen concentrations arising from the Saskatchewan River in the summer months (Section 7.2). The range of dissolved inorganic nitrogen concentrations in the fall was similar to that in the summer. However, the spatial pattern of inorganic nitrogen concentrations in the fall differed between the north and south basins of the lake. In the north basin, elevated dissolved inorganic nitrogen concentrations (approximately 0.3 mg/L in the fall) occurred along the east side of the basin while concentrations were approximately ten times lower (0.02 mg/L in the fall) along the west side. In the south basin, relatively high dissolved inorganic nitrogen concentrations extended along the west side of the basin (0.35 mg/L in the fall), while low dissolved inorganic nitrogen concentrations were prevalent in southeastern portion of the basin (approximately 0.2 mg/L in the fall).

6.4 Internal Nutrient Loading Estimate

An initial estimate of internal phosphorous load was calculated for the north basin of Lake Winnipeg to gain some preliminary insight into the potential for internal nutrient release during low oxygen events. The internal phosphorous load estimate was calculated using a method that was applied to estimate internal nutrient loading for Lake Simcoe (Eimers and Winter 2005). Considering all low oxygen events from 1999 to 2007 during the summer in the north basin of Lake Winnipeg (<5 mg/L), net total phosphorous increase measured at 0.5 m above the lake bottom (calculated as the summer concentration minus the spring concentration) ranged from between 0.017 to 0.058 mg/L with an average net summer increase of 0.038 mg/L. Using an average net increase of 0.038 mg/L of total phosphorus over the summer in the 14 to 18 m zone of the north basin (volume of 245 $m^3 \times 10^8$ from Brunskill *et al.* 1980), approximately 930 tonnes of phosphorus may be remobilized into the north basin water column during low oxygen events. In comparison, this internal loading in the north basin has the potential to contribute more phosphorus than the average load from the Saskatchewan River and more than three-quarters of the average phosphorous load from the Winnipeg River. Internal nutrient load will most certainly vary from year to year given the nature of oxygen concentrations at the bottom of the lake, considering that in some years, water at the bottom of the lake has been well oxygenated. This initial loading estimate is considered conservative because it does not consider the phosphorus that may be reintroduced into the water column by wind-induced resuspension of bottom sediments and does not account for what may be remobilized during the ice cover period. Future nutrient budget work is expected to provide a more comprehensive assessment of the internal phosphorous load using mass balance modelling approaches.

From 1999 to 2007, the south basin of Lake Winnipeg was generally nitrogen limited (with ratios around 10:1) or co-limited by both nitrogen and phosphorus (with ratios ranging between 10:1 and 20:1).

6.5 Phytoplankton and Nutrient Limitation (N:P)

6.5.1 Recent Conditions: 1999 to 2007

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 phosphorous 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 2007, the south basin of Lake Winnipeg was generally nitrogen-limited (ratios around 10:1) or co-limited by both nitrogen and phosphorus (ratios ranging between 10:1 and 20:1) (Figure 6.9). In the south basin of Lake Winnipeg, N:P ratios varied inter-annually during this period. Average annual open water N:P ratios were lowest in 2002 (12:1), coinciding with low total nitrogen concentrations (0.48 mg/L) and highest in 2000 and 2004 (25:1) with an average of 20:1 for the south basin (1999 to 2007). N:P ratios were generally highest in the spring (range 20:1 to 30:1) and declined over the open water season to the lowest ratios in the fall, between <10:1 to 20:1, as phosphorous concentrations increased. Nitrogen limitation was also apparent in past studies of the south basin. Using data collected in 1969 and the mid-1990s, Stewart *et al.* (2000) identified a strong relationship between nitrogen and chlorophyll a ($R^2 = 0.990$) in the south basin of Lake Winnipeg, although no relationship was found between total phosphorus and chlorophyll a.

Although N:P ratios indicate that the south basin is generally nitrogen-limited, large blooms of cyanobacteria are less frequent and less extensive in comparison to the north basin. This is likely because phytoplankton in the south basin are at times light-limited from the high sediment loads arising from the Red River. In contrast, the north basin of the lake was mostly phosphorous-limited. The overall ratio in the north basin was 41:1, with average annual ratios ranging from 14:1 (2002) to 92:1 (2003). The lowest N:P ratios from 1999 to 2007 occurred in 2002, in association with the lowest total nitrogen concentrations for the period (Figure 6.9). Phosphorous limitation was most pronounced in 2003, when average annual total nitrogen concentration was at its greatest. There was no clear seasonal pattern in the N:P ratios in the north basin of the lake. The inconsistent seasonal pattern may be related to nitrogen-fixing cyanobacteria that dominate the phytoplankton in the north basin, forming blooms in some years (Sections 8.1, 9.1). These algal blooms may have the ability to shift the N:P ratio through fixation of atmospheric nitrogen. Internal nutrient loading under hypoxic conditions (e.g., 2003) may also affect N:P ratios.

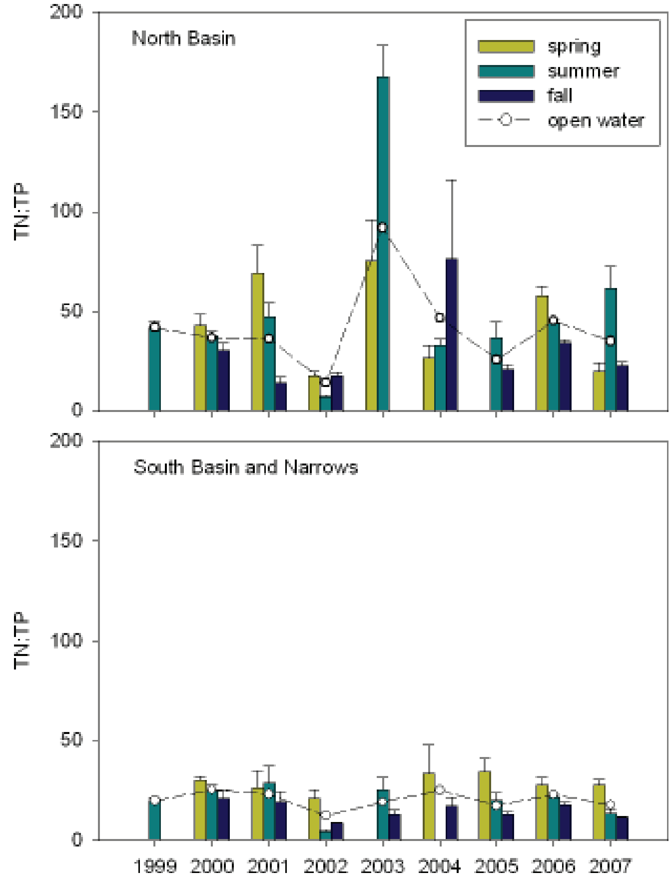


Figure 6.9: *Annual mean open water (May to October) total nitrogen to total phosphorus molar ratio (± SE) in the north basin and south basin and the narrows of Lake Winnipeg from 1999 to 2007. Ratios are calculated for water samples collected from the surface and euphotic zone of Lake Winnipeg.*

6.5.2 Seasonal and Spatial Variation

During the spring of 2002, Lake Winnipeg was primarily phosphorous-limited with N:P ratios greater than 18:1 at all lake stations (Figure 6.10, next page). The north basin was more strongly phosphorous-limited relative to the south basin in the spring, particularly north of Reindeer Island, where ratios ranged from 40:1 to 50:1. The Saskatchewan River is relatively low in phosphorus in comparison to the Red River (Section 7.1), driving the north-south gradient in nutrient limitation. In the south basin, Traverse Bay and the eastern shoreline were the most phosphorous-limited, with higher N:P ratios (approximately 40:1 to 50:1) as compared to the rest of the south basin where ratios ranged between 25:1 and 35:1. Nitrogen to phosphorus ratios were near the optimal Redfield ratio, indicating that primary production near the inflow of the Red River (20:1) was not nutrient-limited during the spring freshet. During the summer, the north basin remained more strongly phosphorous-limited than the south basin of the lake. However, from spring to summer, N:P ratios appeared to decline in the north basin, and ratios also declined in Traverse Bay and along the east shore of the south basin. During the fall, much of the south basin was nitrogen-limited in association with rising phosphorous concentrations later in the open water season (Figure 6.5). Ratios were below 16:1, with exception of Traverse Bay and the eastern shoreline of the south basin, which remained slightly phosphorous-limited (ratios of 18:1 to 25:1). Despite low N:P ratios in the south basin that favour the development of cyanobacteria, blooms in the south basin are less extensive than in the north basin of the lake because of light limitation by suspended solids (Section 5.3, Figure 5.20).

Phosphorous limitation in the north basin declined over the open water season, with ratios of approximately 40:1 in the summer and 25:1 in the fall. Contributions of phosphorus from rivers and increases in biological productivity over the open water season likely account for this seasonal pattern in the north basin.

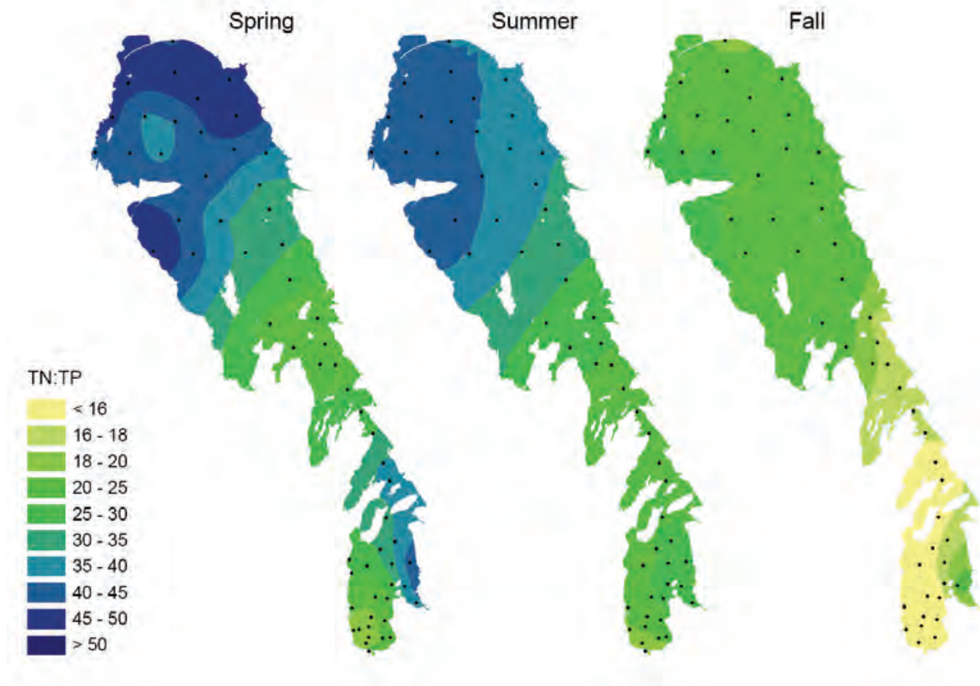


Figure 6.10: *Phytoplankton nutrient limitation (N:P molar ratio) in Lake Winnipeg during the spring, summer, and fall of 2002.*

Text Box 4. Phosphorus Isotope Source Tracking in Lake Winnipeg

(Véronique Hiriart-Baer, Len Wassenaar)

Phosphorus is a naturally-occurring, essential nutrient for terrestrial and aquatic plants in freshwater ecosystems, and often limits the growth of algae because concentrations are typically low relative to the biological demand. However, human activities on the landscape can lead to the eutrophication of lakes and streams. In Lake Winnipeg, one of the primary factors driving the ongoing eutrophication process is excess phosphorous loading from both point (e.g sewage treatment plants), and non-point (e.g. manure and fertilizers applied to agricultural and urban landscapes) sources.

To understand the dynamics of phosphorus in lakes and rivers, the concentration (or loading) of the different forms of phosphorus in the different biogeochemical compartments of a lake (e.g. water, sediment, biota) are typically measured and monitored over time and space. While this approach results in an understanding of the flow and exchange of phosphorus between the different compartments, information on the relative contribution of the different input sources (e.g. inorganic fertilizer versus treated sewage or animal waste) of phosphorus to a particular ecosystem remains unknown. An understanding of both the sources and in-lake processing of phosphorus is required to better manage and adapt remediation strategies.

The need to identify specific sources of nutrients to aquatic ecosystems is not new. For decades, researchers have been using naturally-occurring stable isotopes (not radioactive) of hydrogen, carbon, nitrogen, oxygen, and sulphur as forensic 'fingerprints' to determine the origin and follow the fate of these elements in the environment. Although phosphorus has only one stable isotope, most of the phosphorus in the environment is tightly bound to oxygen (as phosphate PO_4), which has three stable isotopes, so the relative abundance of the oxygen isotopes can be measured. In phosphates, the bond between phosphorus and oxygen is stable at most environmental temperatures and does not readily swap oxygen molecules with water without biological mediation. This means that different sources of phosphorus with distinct phosphate 'fingerprints' can potentially be identified, particularly in nutrient-rich lakes, such as Lake Winnipeg where biological overprinting of the signature will likely be at a minimum.

Given the large watershed of Lake Winnipeg and the numerous potential sources of phosphorus to the lake, this study began by focussing on measuring the oxygen stable isotope ratios of dissolved inorganic phosphates ($\delta^{18}O_{DIP}$) spatially and temporally in the lake. Spatial variability in $\delta^{18}O_{DIP}$ was observed across the entire lake suggesting that the original signature of the phosphorus was retained and source tracking efforts were possible (Plate 1). By comparing the $\delta^{18}O_{DIP}$ to the theoretical equilibrium, there are at least two dominant sources of phosphorus with distinct signatures, and the relative contribution of these sources appears to differ between the north and south basins and the time of year (Plate 2 on the following page).

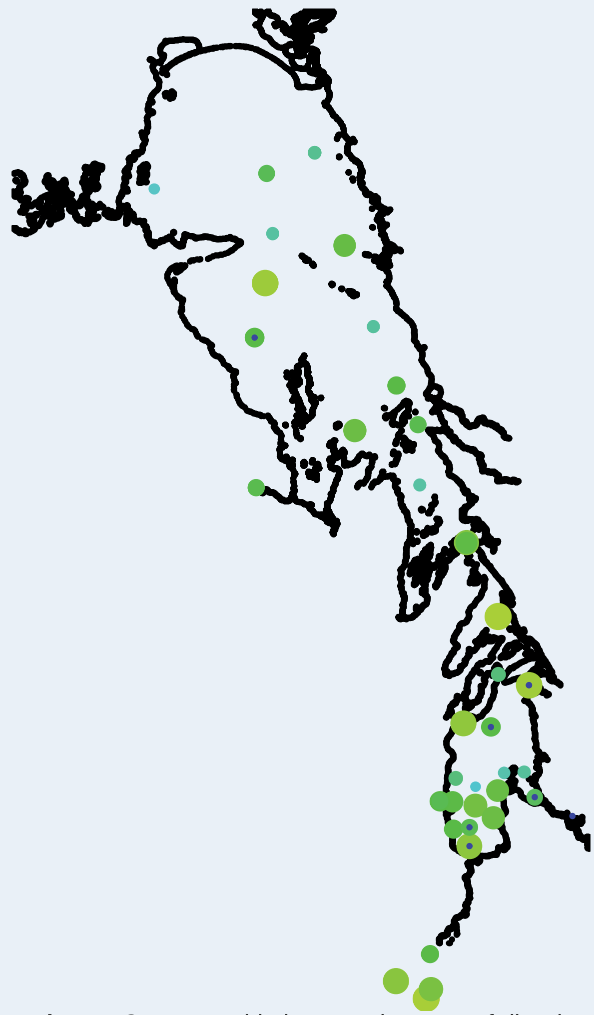


Plate 1. *Oxygen stable isotope signature of dissolved inorganic phosphates ($\delta^{18}O_{DIP}$) across Lake Winnipeg for all stations sampled in 2007 and 2008. The magnitude of the $\delta^{18}O_{DIP}$ values, represented by both color and circle size, ranged from 2.6‰ to 24.6‰ with an average of 13.7±4.4‰.*

Continued...

Text Box 4 continued...

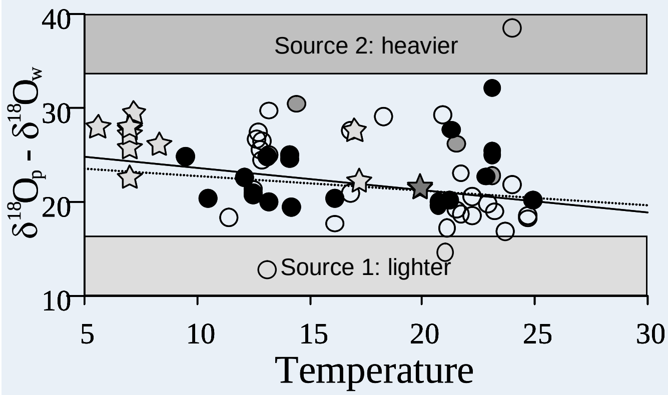


Plate 2. *The $\delta^{18}O_{DIP}$ - $\delta^{18}O_w$ as a function of temperature for all months. The y axis is written as $\delta^{18}O_{DIP}$ - $\delta^{18}O_w$ so that the equilibrium values can be represented in a linear relationship with temperature. Legend: filled circles = north basin; open circles = south basin; shaded circles = narrows; dark shaded stars = Rivers; light shaded stars = STP.*

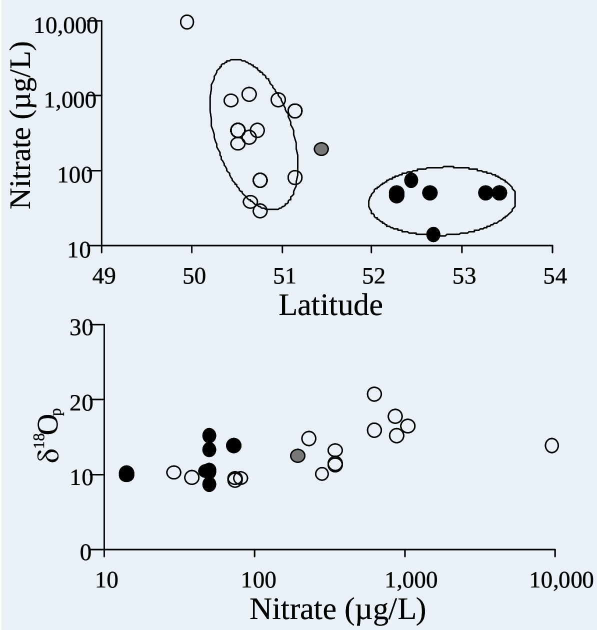


Plate 3. *Spatial variability in lake water nitrate concentrations as a function of latitude and variability in oxygen stable isotope signature of dissolved inorganic phosphates ($\delta^{18}O_{DIP}$) as a function of lake water nitrate concentrations. Legend: filled circles = north basin; open circles = south basin; shaded circles = narrows.*

One likely source of phosphorus is agricultural inputs. Nitrate concentrations are significantly ($p<0.001$) lower in the north compared to the south basin and as nitrate concentrations increase, the isotopic signature of $\delta^{18}O_{DIP}$ becomes more enriched (Plate 3).

Temporal variations in the $\delta^{18}O_{DIP}$ were also observed in addition to the spatial variability. In the north basin there was no notable change in the mean and range of $\delta^{18}O_{DIP}$ over the spring, summer, and fall seasons, while in the south basin there was enrichment in the $\delta^{18}O_{DIP}$ in the spring and fall (data not shown). This pattern was also reflected in the nitrate concentrations, with higher concentrations in the fall. This suggests that delivery of a phosphorous source with an enriched $\delta^{18}O_{DIP}$ is increased in the south basin during times of high flow and that this source is also delivering higher levels of nitrate.

Research efforts are on-going to better characterize the $\delta^{18}O_{DIP}$ signature of the possible sources (for example, sewage treatment plants, rivers and sediment) which will allow phosphorous source apportionment through mixing models.

Text Box 5. Historical Changes in Water Quality of Lake Winnipeg's South Basin

(Lynda Bunting, Peter Leavitt, Kathleen Laird, Brian Cumming, Ann St. Amand, Daniel Engstrom)

Researchers from the University of Regina, Queen's University, the Science Museum of Minnesota, and PhycoTech Inc. are partnering with Manitoba Water Stewardship to quantify the timing, magnitude, and potential causes of eutrophication in the south basin of Lake Winnipeg.

Fossils preserved in lake sediments were used to measure historical changes in the abundance and species composition of algae and invertebrates, and concentrations of major nutrients. This information will be used to reconstruct historical changes in water quality.

Three sediment cores were collected from the south basin of Lake Winnipeg during 2006 to 2008 to allow researchers to better determine the overall patterns of historical change in communities of algae within the water column.

Preliminary analysis from a centrally-located core reveals that total algal abundance has increased about 300 to 500 % during the 20th century (Plate 1a). In addition, all algal groups have increased during the past 70 years, including diatoms, cryptophytes, green algae, and colonial cyanobacteria. Furthermore, sediment analyses revealed that blooms of cyanobacteria capable of nitrogen fixation expanded nearly ten-fold after 1990, even though they have been present in the lake for at least 150 years (Plate 1b). This shift from well-mixed algae prior to 1990 to floating cyanobacteria during the past 20 years is consistent with documented changes in algal community composition known from direct measurements of species in the lake's water column.

Further work is being conducted on two additional cores from the south basin, as well as five sediment sites from the north basin to determine how the timing and extent of eutrophication may have varied within the lake. In addition, historical records of climate change and land use are being collected for comparison with the fossil data to determine whether water quality loss is affected more by climate, crop production, livestock, or urbanization. Along with other information, these data will be used to establish baseline conditions and ecologically-relevant goals for remediation of Lake Winnipeg.

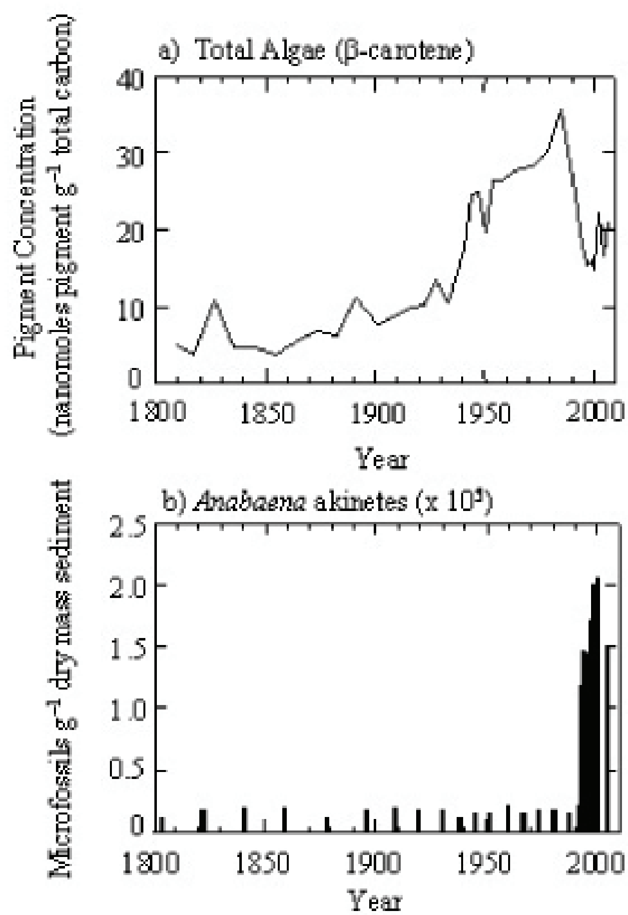


Plate 1. *Historical changes in total algal abundance (Plate 1a) and cyanobacteria capable of nitrogen fixation (as Anabaena akinetes; Plate 1b) in a sediment core collected from the south basin of Lake Winnipeg.*

Text Box 6. Phosphorus in Sediments of Lake Winnipeg

(Lucie Lévesque, Sue Watson, Jay Guo, Mike Stainton, Patricia Ramlal)

Eutrophication is often tied to excess loading of nutrients such as phosphorus. Phosphorous loading occurs from external and internal sources. External loading is derived from precipitation and watershed runoff delivered to the lake via tributaries and groundwater. Internal phosphorous loading occurs through the exchange between sediments and the water column, and may constitute a significant proportion of total loading and an important part of phosphorous cycling in lakes (Wetzel 2001). This is particularly true for shallow lakes where wind-driven resuspension and diffusion to the water column under anoxia result in the redistribution of sediment-associated phosphorus into the water column (Kalff 2002; Sondergaard *et al.* 2001). Internal loading of phosphorus may result in substantive delays in the effects of remedial reduction in external inputs (e.g., Sondergaard *et al.* 2003; Lau and Lane 2002).

Sediment cores collected by Fisheries and Oceans Canada in 1994 and 1999 were analyzed for organic and inorganic fractions of phosphorus to assess the historic rate of phosphorous deposition in Lake Winnipeg. Total phosphorous concentrations in cores collected from the south basin increased from the 1930s through to the 1960s. More pronounced rates of phosphorous deposition in the second half of the 20th century were attributed to anthropogenic inputs (Mayer *et al.* 2006). Higher levels of organic phosphorus near the sediment-water interface were attributed to elevated rates of primary productivity and associated contributions of algal biomass. A sediment core from the north basin also indicated an increased deposition rate since the 1960s, and further acceleration since the 1990s.

Surface sediments were collected in the north and south basins of Lake Winnipeg in 1969, 1994, 1999, and 2002 (Brunskill *et al.* 1980; Fisheries and Oceans Canada). Elevated total phosphorous concentrations in the late 1990s and into the 2000s (Plate 1) are consistent with increases in sediment and nutrient loading to the lake from the watershed (see Section 7.0). Surficial concentrations of sediment-associated phosphorus in mid-summers of 1994 and 1999 were greatest in the south basin near the inflow of the Red River, reaching levels as high as 1.86 mg/g dry weight total phosphorus in 1994 and 1.26 mg/g dry weight in 1999. Concentrations in the north basin typically did not reach much higher than 1.0 mg/g dry weight and were greatest in the northwestern portion of the basin near the inflow of the Saskatchewan River. The 1997 Red River flood resulted in record-high flow and sediment loading to Lake Winnipeg from the Red River. Sediment and associated internal nutrient loads from this flood were distributed to the north basin, contributing to the formation of a substantive cyanobacterial bloom in 1999. Elevated concentrations of sediment-associated total phosphorus in the north basin in the spring of 2002 (as high as nearly 2.0 mg/g dry weight) may be attributed to the contributions of detrital algal material from antecedent blooms. Slight reductions in total phosphorous concentrations in the summer through the fall of 2002 are consistent with increased rates of biological productivity and decay in response to the seasonal rise in water temperatures.

Phosphorus in sediments is present in a number of particulate (and dissolved) fractions, some of which are more bio-available and therefore may have the most immediate effect on algal growth and ability to develop excessive biomass. The estimation of bio-available phosphorus with laboratory bioassays is expensive and time consuming, and alternative chemical analyses have been developed to estimate bio-available phosphorus which show reasonably good agreement with the bioassays (Young *et al.* 1985; DePinto *et al.* 1981; Dorich *et al.* 1980; Williams *et al.* 1980; Sagher 1976; Sagher *et al.* 1975; Golterman *et al.* 1969). Furthermore, where benthic invertebrate activity and/or resuspension and within-lake sediment transport is significant, as in relatively shallow systems like Lake Winnipeg, the surficial layers are often highly disturbed and transient, and the top ~5 cm provides a better indication of the more stable surficial layer. With the exception of a few cores taken in the early 1990s (Mayer 1991), there has been little work done to date to evaluate the distribution and levels of bio-available phosphorus in this layer. Therefore, during the summer *MV Namao* cruises in 2008 and 2009, basin-wide samples of the top 0 to 5 cm sediment layer were collected by Environment Canada at scheduled sampling sites in the north and south basins with a Ponar. Samples were analysed for total and bio-available phosphorus with an improved chemical extraction method modified from Mayer (1991). The preliminary results of these surveys suggest that the percent bio-available portion of total sediment phosphorus is on average higher in the south basin than the north basin (Plate 2 on page 80).

Continued...

Text Box 6 continued...

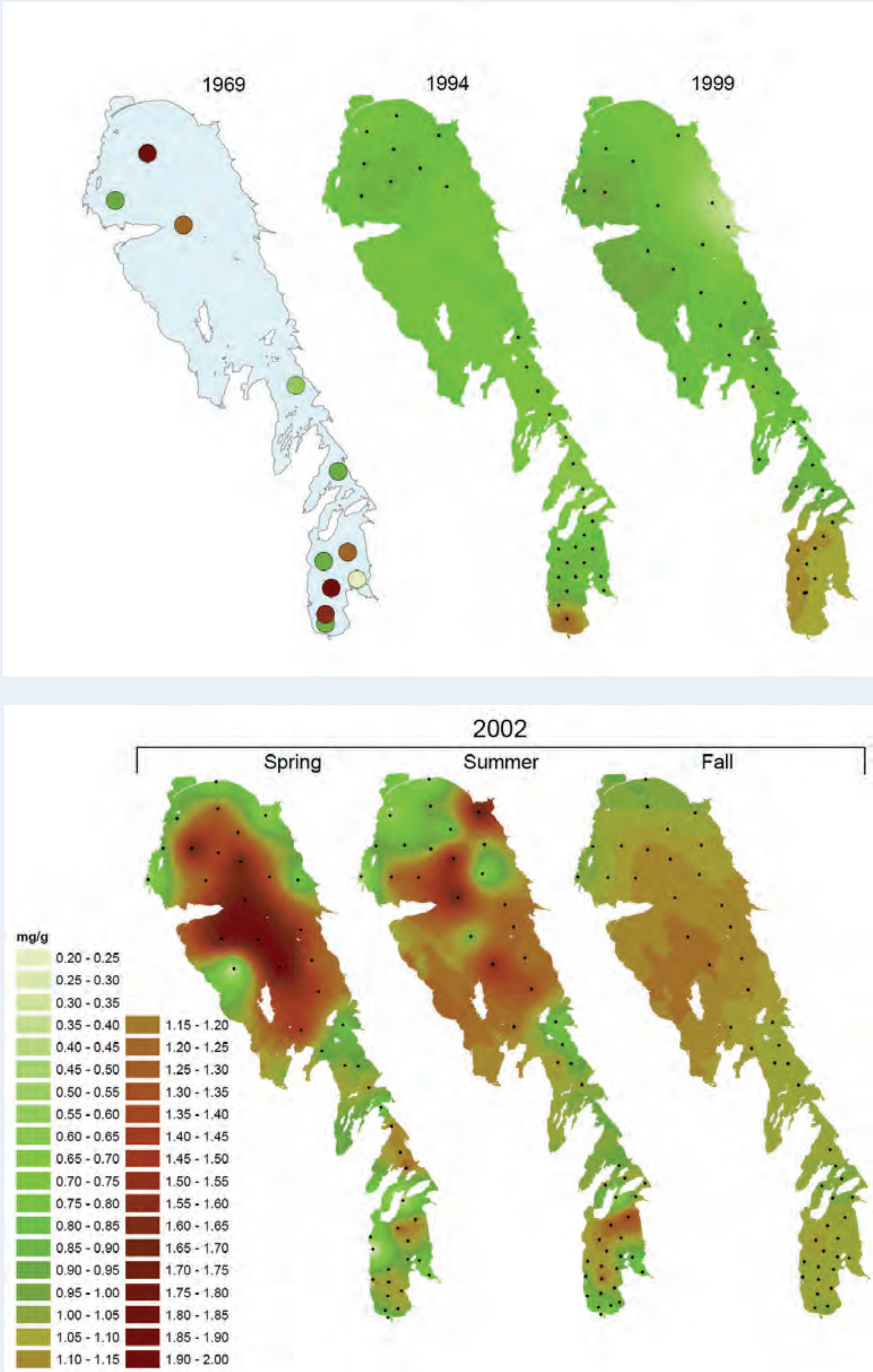


***Plate 1.** Total phosphorous concentrations in surface sediment mid-summer 1969, 1994, and 1999, and spring, summer, and fall 2002. Scale ranges from 0.2 (light green) to 2.0 mg/g dwt at intervals of 0.05. (1969 = Brunskill et al. 0 to 5 cm, Eckman/Ponar; 1994/1999 = Fisheries and Oceans Canada (Stainton), 0 to 1 cm, Eckman grab; 2002 = Fisheries and Oceans Canada (Ramlal), 0 to 1 cm, Ponar)*

Continued...

Text Box 6 continued...

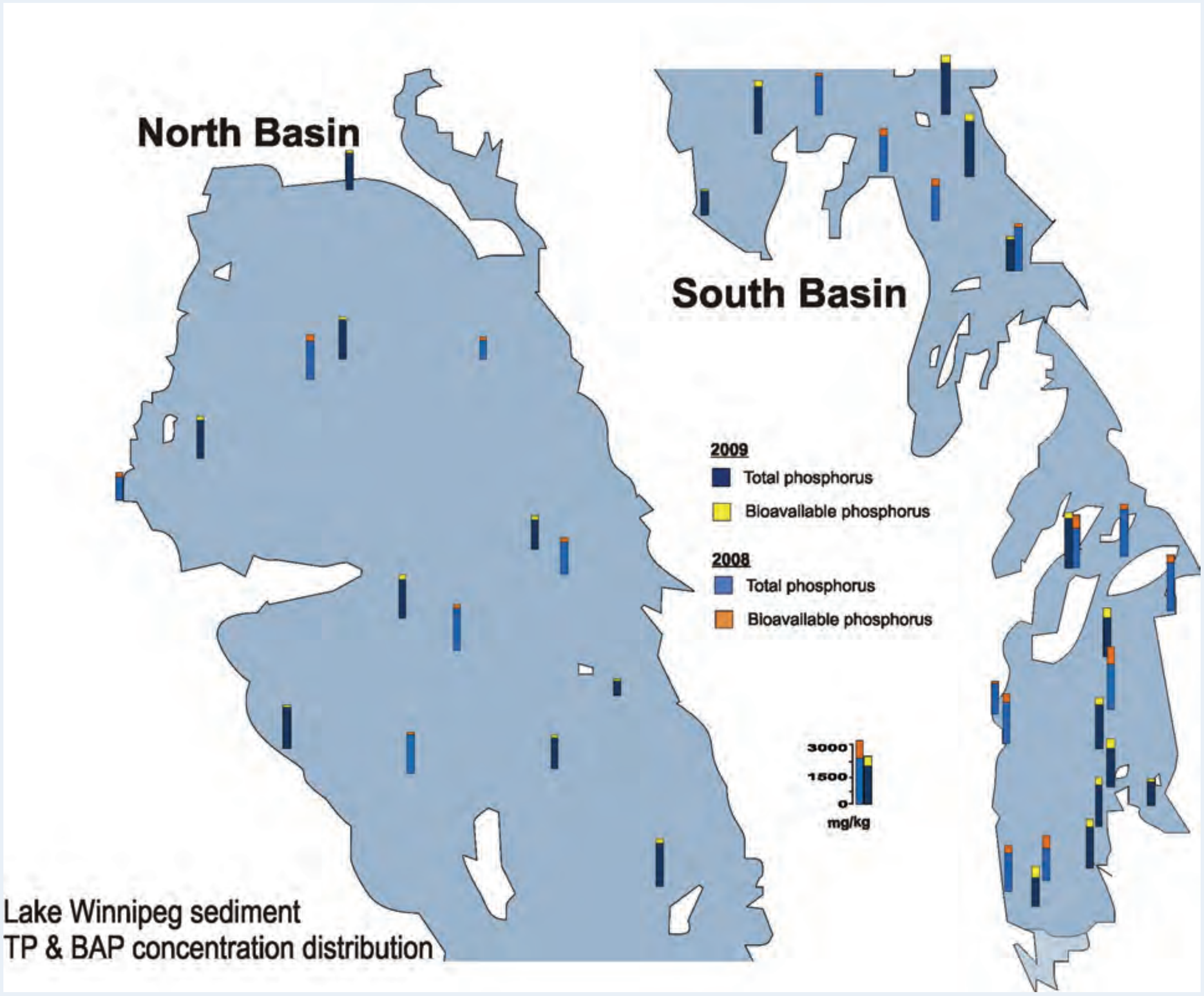


Plate 2. *Total and bio-available phosphorus in surficial (0 to 5 cm) sediment collected from focal study sites in Lake Winnipeg summer surveys, 2008 and 2009.*

7.0 NUTRIENT LOADING TO LAKE WINNIPEG

(Nicole Armstrong, Greg McCullough)

Nutrient levels in Lake Winnipeg and associated eutrophication dynamics are affected by nutrient loads from tributaries to the lake (Text Box 7). Nutrient loads include those from large transboundary rivers (Red, Saskatchewan, Winnipeg) as well as numerous smaller rivers that discharge directly into Lake Winnipeg, transporting nutrients originating from point and diffuse sources within their catchments. Given the large surface area of the lake, atmospheric deposition is also an important source of nutrients to Lake Winnipeg. Water quality in tributaries to Lake Winnipeg is monitored by Manitoba Water Stewardship, Environment Canada 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 total phosphorus and total nitrogen to Lake Winnipeg over the period 1994 to 2007 (Appendix 7).

7.1 Phosphorus

Average total phosphorous concentrations for the years 1994 through 2007 in the tributaries to Lake Winnipeg ranged from 0.019 mg/L in the Saskatchewan River to 0.354 mg/L in the Red River (Table 7.1). While a formal statistical trend analysis was not conducted on these data, Figure 7.1 (next page) suggests that total phosphorous concentrations increased between 1994 and 2007 for the Winnipeg, Red, and Brokenhead rivers (the only three tributaries where data were available for every year during that period).

On average, the total phosphorous load to Lake Winnipeg was 7,655 tonnes per year (t/y) (Table 7.2) or 27 tonnes per cubic kilometre per year (t/km^3/y). In comparison, the total phosphorous load to Lake Simcoe in Ontario ranged from 53 to 157 t/y between 1990 and 2007 (Lake Simcoe Region Conservation Authority 2009; Winter *et al.* 2007) and was considerably less per unit volume at 4.6 to 13.5 t/km^3/y. Substantial federal and provincial efforts have been focussed on reducing phosphorous loading to Lake Simcoe from its relatively small watershed (2,899 km^2). Phosphorous loads to the Great Lakes are no longer routinely monitored, but historical loads ranged from a high of almost 20,000 t/y to Lake Erie in the late 1970s to under 3,000 t/y to Lake Superior in the early 1990s (Neilson *et al.* 1995).

On average, the total phosphorous load to Lake Winnipeg was 7,655 t/y or 27 t/km^3/y.

Table 7.1: *Annual average total phosphorous concentrations (mg/L) in tributaries to Lake Winnipeg.*

	Annual Average Total Phosphorus				
Year	Red River at Selkirk	Winnipeg River at Point du Bois	Saskatchewan River at Grand Rapids	Dauphin River	Brokenhead River
1994	0.310	0.022	0.020	0.023	0.088
1995	0.297	0.022	0.016	0.020	0.047
1996	0.257	0.023	0.018	0.018	0.054
1997	0.324	0.022			0.050
1998	0.300	0.024			0.063
1999	0.285	0.026			0.066
2000	0.276	0.024			0.060
2001	0.362	0.026	0.018		0.056
2002	0.299	0.033	0.018		0.081
2003	0.419	0.027	0.019		0.083
2004	0.503	0.030	0.022	0.023	0.115
2005	0.444	0.030	0.019	0.030	0.157
2006	0.360	0.027	0.023	0.021	0.177
2007	0.517	0.035	0.022	0.014	0.220
Average	**0.354**	**0.027**	**0.019**	**0.021**	**0.094**

Figure 7.1: *Change in annual average total phosphorous concentrations in the Red River at Selkirk, the Brokenhead River, and the Winnipeg River at Point du Bois (secondary axis) between 1994 and 2007.*

Table 7.2: *Annual average total phosphorous loads to Lake Winnipeg from major tributaries and atmospheric deposition and outflow from the Nelson River.*

	Phosphorus (tonnes per year)													
	1994	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**
Red River	2,661	4,165	4,418	8,176	4,266	5,425	2,782	7,344	3,082	2,050	6,800	9,917	7,044	7,188
Winnipeg River	779	855	1,051	1,089	553	1,047	1,093	1,380	1.313	503	1,607	1,410	968	1.032
Saskatchewan River	289	271	362	578	397	424	353	98	202	199	309	668	569	566
Dauphin River	19	53	78	188	123	157	73	154	12	2	20	111	113	72
East Side Rivers	135	152	154	188	116	184	182	233	248	113	235	256	160	219
Brokenhead River	15	6	10	14	13	5	16	20	14	7	43	72	23	62
Fisher River	2	4	8	17	41	10	33	72	6	3	108	36	33	25
Icelandic River	2	8	22	27	47	10	46	63	9	6	46	72	40	26
Atmospheric Deposition	500	500	500	500	500	500	500	500	500	500	500	500	500	500
Total	**4,401**	**6,013**	**6,604**	**10,777**	**6,055**	**7,762**	**5,077**	**9,864**	**5,384**	**3,384**	**9,667**	**13,043**	**9,451**	**9,691**
Outflow from Nelson River	1,536	1,754	2,055	1,987	2,249	1,813	1,989	2,840	3,864	2,195	2,574	8,119	4,795	4,634

Except for Lake Erie, phosphorous loads to the Great Lakes were also lower per unit volume than those to Lake Winnipeg, ranging from 0.3 t/km^3/y to Lake Superior to 7.3 t/km^3/y to Lake Ontario. In Lake Erie, phosphorous loads ranged from 15.5 to 39.3 t/km^3/y. The comparatively high phosphorous load per unit volume to Lake Winnipeg as compared to the Great Lakes and Lake Simcoe supports the need for action to reduce nutrient loading to Lake Winnipeg.

Total annual phosphorous load to Lake Winnipeg ranged from a high of 13,043 tonnes in 2005 to a low of 3,384 tonnes in 2003. Similar to rivers in the Upper Mississippi, Ohio, and Great Lakes river basins (Lorenz *et al.* 2009; David and Gentry 2000), inter-annual variability in phosphorous loads to Lake Winnipeg is mostly determined by variability in annual stream flow, as demonstrated by the relationship between annual stream flow and annual phosphorous load. However, the strength of the relationship varies across rivers and is weakened by long-term changes in concentration in some of the tributaries to Lake Winnipeg such as the Red River (Jones and Armstrong 2001). For example, in the Winnipeg River, stream flow accounts for 79 % of the variability in phosphorous load. In contrast, only about 50 % of the variability in phosphorous load in the Red River can be attributed to stream flow. Nevertheless, when extreme years are considered, the correspondence between annual flow and annual phosphorous loading is high even for the Red River. The three years with highest annual flow in the 1994 to 2007 record (1997, 2001, 2005) were the years of highest annual phosphorous loading by the Red; conversely, the three lowest flow years (1994, 2002, 2003) were the three lowest phosphorous loading years. McCullough (2001) also noted that annual phosphorous yields tended to be well correlated with annual stream flow in Lake Winnipeg tributary rivers such as the Saskatchewan, Winnipeg, Dauphin, and Red rivers (81 to 94 % of the variability explained by stream flow, using annual data for the shorter period 1994 to 2001).

In particular, the low phosphorous load to Lake Winnipeg in 2003 can be attributed to low inflows (Figure 4.5, Section 4.3) from all major tributaries. The annual average inflow from the Red and Winnipeg rivers in 2003 was the lowest recorded from 1994 to 2007, less than one-half the average for the period, and the annual stream flow in the Saskatchewan River was third lowest in the period of study. Conversely, in 2005, flows were relatively high in all three main tributaries to Lake Winnipeg (Figure 4.5, Section 4.3). On the Saskatchewan River that year, flows were the highest observed during the study period and were almost double the average observed through the period from 1994 to 2007. Flows on the Winnipeg and Red rivers were the second highest in 2005. Peak flow years on the Winnipeg and Red rivers occurred in 2001 and 1997, respectively. In 2005, concentrations of total phosphorus were at or above average for the study period in the Red, Winnipeg, and Saskatchewan rivers, likely contributing further to the record large total phosphorous load to Lake Winnipeg in 2005. However, at least at the extremes, the Red River clearly dominates this relationship between tributary flow and total phosphorous loading to Lake Winnipeg. The three years with the highest stream flow in the Red River and the three lowest flow years, were not only the years of highest and lowest annual phosphorous load respectively, in the Red River (as mentioned above), they were also the three highest (and lowest) years of phosphorous loading to Lake Winnipeg.

Over the period 1994 to 2007, the Red River was the main source of phosphorus to Lake Winnipeg contributing on average 5,380 tonnes of phosphorus per year to the lake or 68 % of the annual total load (Table 7.3, Figure 7.2). The Red River basin itself, exclusive of its large western tributary, the Assiniboine River, supplied on average 86 % of the total Red River load. The Red-Assiniboine basin contributed between 55 and 76 % of the annual phosphorous load to the lake. Those years when the Red River contributed a relatively small portion of the total phosphorous load (2000 and 2002) were characterized by below average flows and total phosphorous concentrations in the Red River but above average flows in the Winnipeg River.

***Table 7.3:** Average annual % contribution of rivers, atmospheric deposition, and nitrogen fixation to the total nutrient load in Lake Winnipeg from 1994 to 2007.*

Source	% Total Phosphorous Load	% Total Nitrogen Load
Red River (at Selkirk)	68	34
Winnipeg River (at Pine Falls)	15	25
Saskatchewan River	5	10
Dauphin River	1	4
East Side Rivers	3	4
Brokenhead, Fisher and Icelandic Rivers	1	1
Atmospheric Deposition	7	11
Nitrogen Fixation	-	11

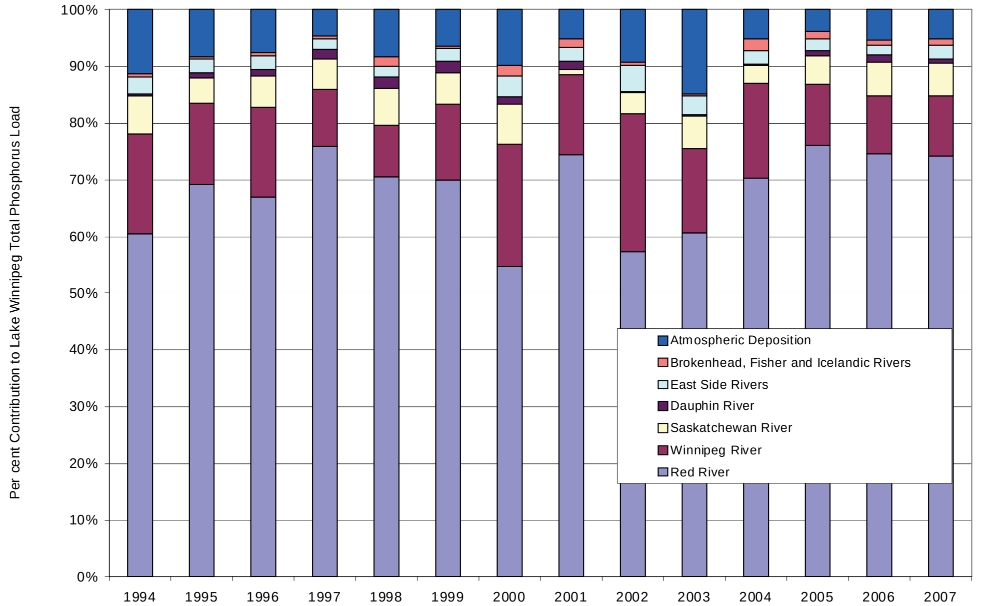


Figure 7.2: *Contribution of rivers and atmospheric deposition (%) to the total phosphorous load in Lake Winnipeg (1994 through 2007).*

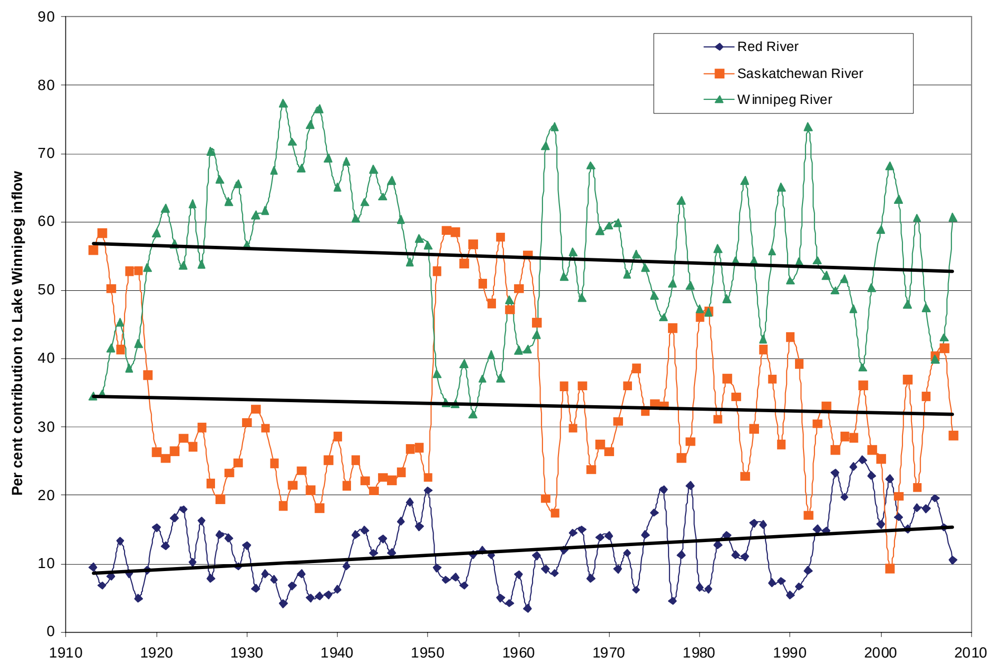


Figure 7.3: *Contribution of the Red, Winnipeg, and Saskatchewan rivers (%) to hydrologic inputs to Lake Winnipeg between 1913 and 2008.*

However, the entire study period from 1994 through 2007 was characterized by high flows in the Red River compared to the previous 80 years of record and by a correspondingly high relative contribution of the Red River to inflow to Lake Winnipeg (Figure 7.3). Moreover, Jones and Armstrong (2001) reported that stream flow independent phosphorous concentrations in the Red River at Selkirk increased 29 % between 1978 and 1999. Therefore, compared to the earlier record, it is likely that the relative contribution of the Red River to the total phosphorous load in Lake Winnipeg was high from 1994 through 2007.

Between 1994 and 2007, the Winnipeg River (at Pine Falls) was the second largest contributor of total phosphorus to Lake Winnipeg. On average, it contributed 15 % of the annual total load to the lake (average 1,049 t/y). The relative percent contribution of total phosphorus from the Winnipeg River to Lake Winnipeg ranged from 9 to 24 %. The Winnipeg River contributed least to Lake Winnipeg nutrient loads in years when flows in the Winnipeg River were relatively low, and both flow and phosphorous concentration of the Red River were relatively high (2006, for example).

Other individual sources contributed less than 10 % each to the total phosphorous load to Lake Winnipeg. Estimated atmospheric deposition was the largest of these small sources. Deposition provided on average 7 % of the total phosphorous load, and possibly as much as 15 % in years of low flow such as 2003. Given that a single estimate of atmospheric deposition is used for the entire study period, these percentage contributions should be interpreted with caution.

Similarly, where rivers were sampled at less than monthly intervals (for example, Dauphin, Brokenhead, Fisher, and Icelandic rivers), annual phosphorous loads calculated for these rivers should also be interpreted with caution. McCullough (2001) calculated annual nutrient loads for tributaries to Lake Winnipeg and concluded that there was good confidence in annual loads where concentrations were measured monthly. In contrast, McCullough (2001) concluded that there was high uncertainty in annual loads for those rivers sampled only four times per year and for which hydrology was flashy (short but intense spring freshet and large peaks during summer rains). David and Gentry (2000) also expressed concern regarding nutrient loads calculated with concentration data collected every six weeks in Illinois. They noted that particulate nutrient loads may be underestimated, especially in smaller watersheds, when water quality sampling is not targeted to include storm events. Additional analysis of the impact of sampling frequency on annual nutrient load for the La Salle River in 2007 is provided in Appendix 7.2. Even with monthly water quality monitoring, calculation of annual nutrient loads during years with large flood events (such as 1997 on the Red River) would benefit from increased water quality monitoring during the flood event (McCullough 2001). Manitoba Water Stewardship's current water quality monitoring program has been adjusted to better characterize nutrient loads during floods and spring freshets.

Manitoba Water Stewardship's current water quality monitoring program has been adjusted to better characterize nutrient loads during floods and spring freshets.

Average annual phosphorous export from Lake Winnipeg through the two channels of the Nelson River was 3,029 t/y (1994 through 2007). On average, about 60 % of the phosphorus entering Lake Winnipeg through the tributaries and atmospheric deposition was retained in the lake with about 40 % exiting through the Nelson River.

Seasonal total phosphorous loads to Lake Winnipeg from the three main tributaries followed a predictable pattern with the largest loads delivered during spring freshet (Figure 7.4). The pattern observed was driven by seasonal variation in the Red River which, as discussed, was the largest source of total phosphorus to Lake Winnipeg. In contrast, seasonal trends in total phosphorous loading to Lake Winnipeg from the Saskatchewan and Winnipeg rivers were not consistent with a typical, spring freshet-dominated load. Both rivers are heavily regulated by hydro-electric dams. In addition, the Winnipeg River flows through the Precambrian Shield which generates low concentrations of suspended solids. During the period 1994 through 2007, peak flows on the Saskatchewan River occurred in the winter, consistent with maximizing

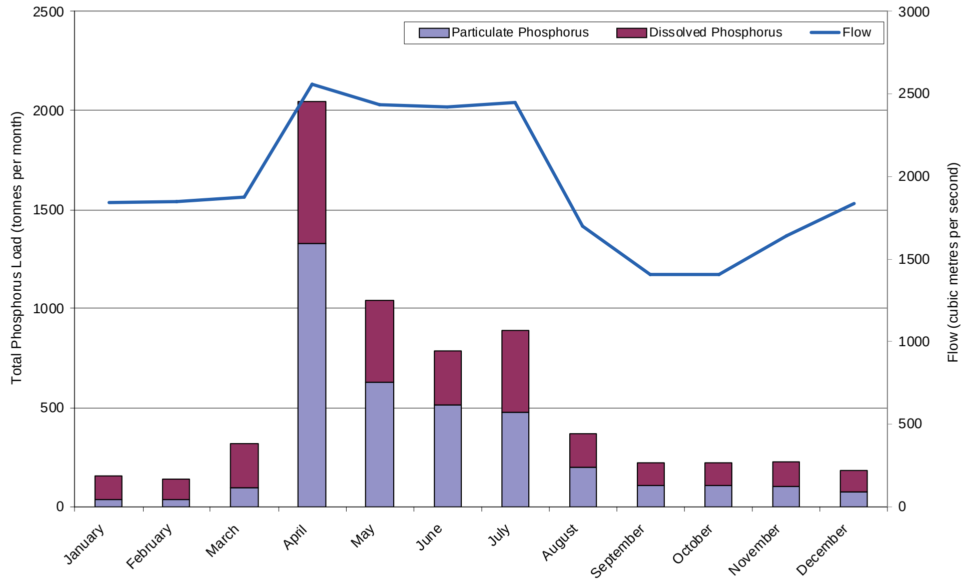


Figure 7.4: *Seasonal variation in total phosphorous load to Lake Winnipeg from the Red, Saskatchewan, and Winnipeg rivers (monthly average from 1994 to 2007 where data were available). Winnipeg River data are from Point du Bois due to a lack of seasonal data for 1994 to 2001 at Pine Falls.*

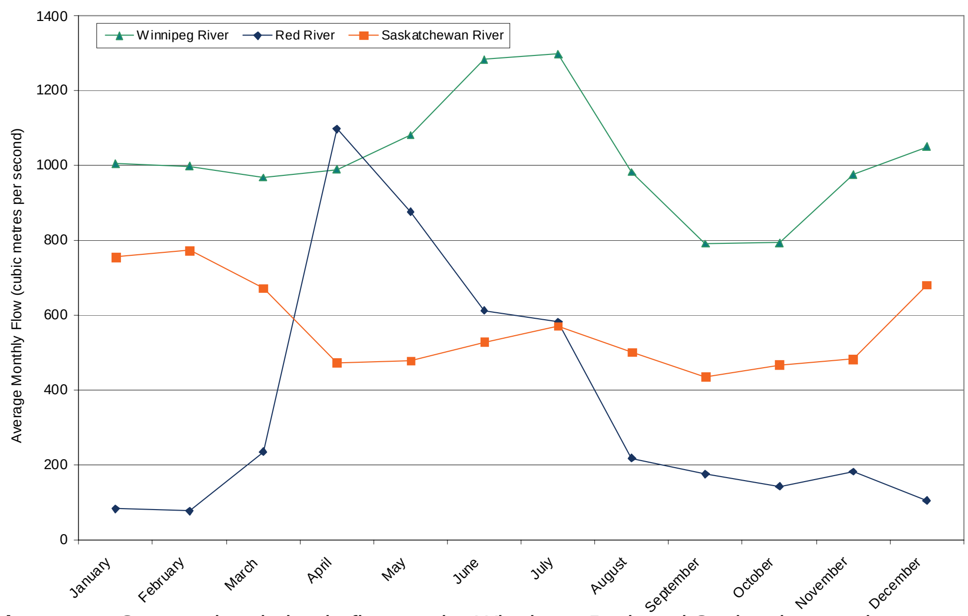


Figure 7.5: *Seasonal variation in flow on the Winnipeg, Red, and Saskatchewan rivers (monthly average from 1994 to 2007).*

power generation during the coldest time of the year (Figure 7.5). However, peak phosphorous loading from the Saskatchewan River occurred in the summer when phosphorous concentrations were almost three times higher than those in the winter (Figure 7.6). The results were not surprising given that the water samples were collected from the forebay of the Grand Rapids Generating Station in Cedar Lake and therefore, nutrient dynamics are consistent with a lentic rather than a lotic environment. In contrast, a typical lotic pattern is observed further upstream on the Saskatchewan River near the Manitoba-Saskatchewan border (Figure 7.7). Total phosphorous loads in the Winnipeg River (Point du Bois) demonstrate relatively little seasonal variation as compared to those in the Saskatchewan and Red rivers (Figure 7.6). Higher loads were observed in mid-summer and early winter during times of peak flow and peak phosphorous concentrations, respectively.

Overall, about 44 % of the phosphorous load to Lake Winnipeg from its watershed from 1994 to 2007 was carried in dissolved form (Red River 46 %, Winnipeg River at Pine Falls 39 %, Saskatchewan River at Grand Rapids 40 %). The proportions varied seasonally (Figure 7.8). In the major tributaries, the dissolved fraction was highest during the ice-covered season (especially January through March) and lowest during spring freshet (April through June). These differences were less in the Saskatchewan River at Grand Rapids where the influence of Cedar Lake (a 2,779 km^2 reservoir created by the Manitoba Hydro dam at Grand Rapids) may have dampened natural variation.

Figure 7.6: *Seasonal variation in total phosphorous load from the Saskatchewan River (top figure) (monthly average for non-missing years from 1994 to 2007), and the Winnipeg River at Point du Bois (bottom figure) (monthly average from 1994 to 2007).*

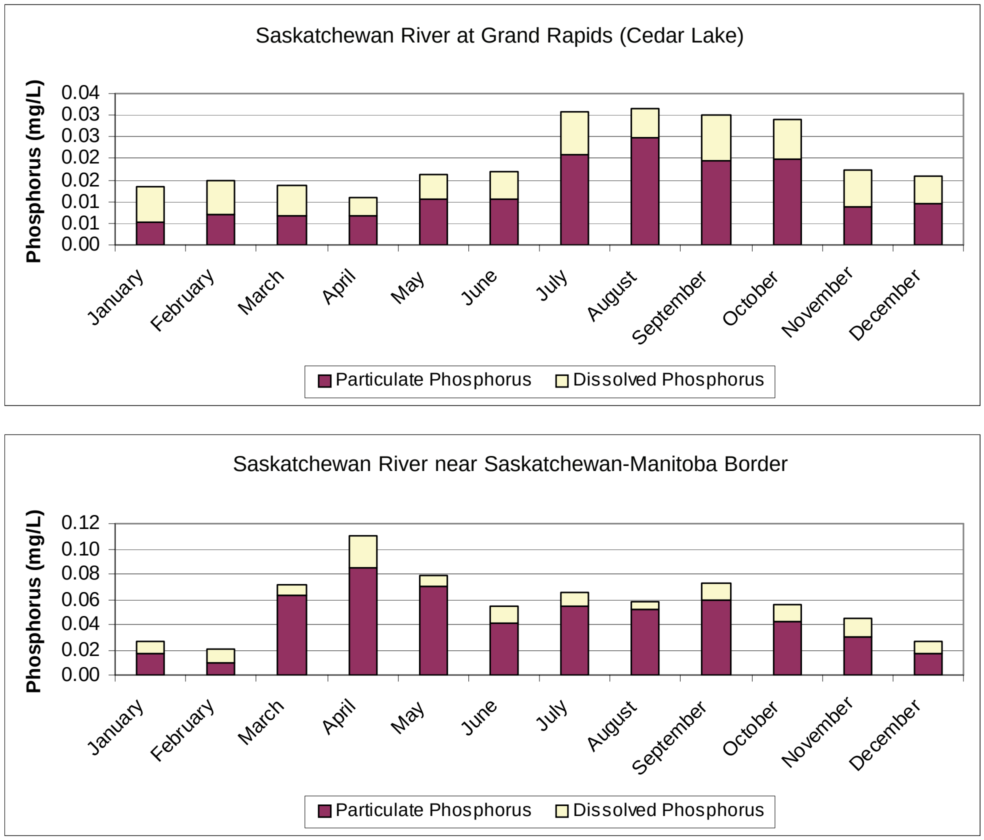


Figure 7.7: *Seasonal variation in total phosphorous concentrations in the Saskatchewan River downstream at Grand Rapids (Cedar Lake) and upstream near the Manitoba-Saskatchewan Border (monthly average from 1994 to 2007 where data were available).*

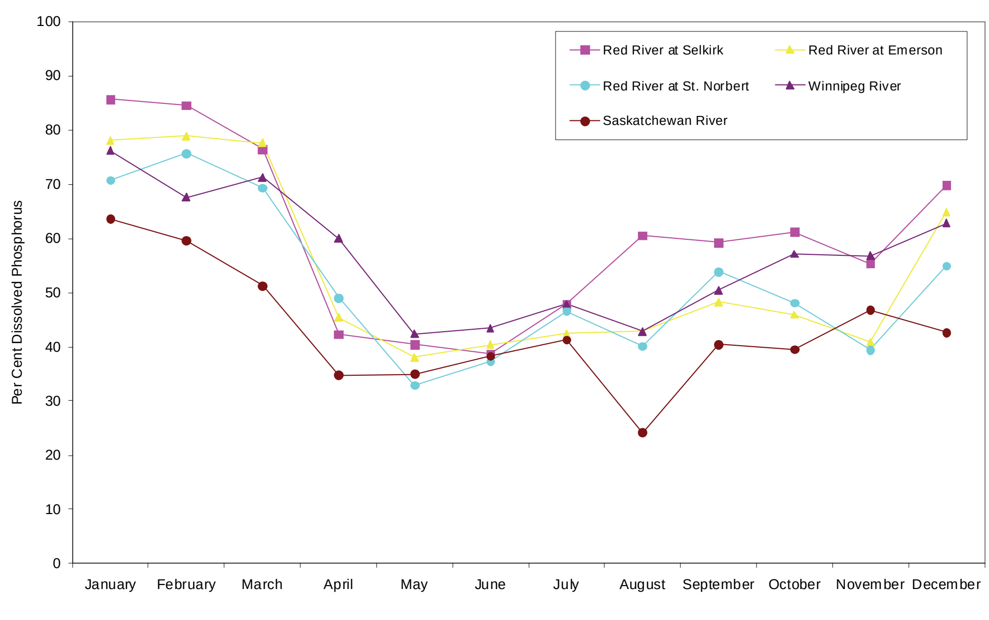


Figure 7.8: *Dissolved phosphorus as a fraction of total phosphorus in the Red, Winnipeg, and Saskatchewan rivers (averages for the period 1994 through 2007).*

7.2 Nitrogen

Average total nitrogen concentrations in the tributaries to Lake Winnipeg ranged from 2.52 mg/L in the Red River to 0.49 mg/L in the Saskatchewan River (Table 7.4). Unlike phosphorus, changes in total nitrogen concentrations between 1994 and 2007 were not readily apparent.

Table 7.4: *Annual average total nitrogen concentrations (mg/L) in tributaries to Lake Winnipeg.*

	Annual Average Total Nitrogen				
Year	Red River	Winnipeg River	Saskatchewan River	Dauphin River	Brokenhead River
1994	2.4	0.50	0.40	1.5	1.5
1995	2.4	0.49	0.42	1.5	0.94
1996	2.2	0.52	0.45	1.2	0.89
1997	2.1	0.45			0.70
1998	2.6	0.48			0.94
1999	2.3	0.50			1.1
2000	2.7	0.49			1.3
2001	2.3	0.55	0.50		1.4
2002	1.7	0.58	0.46		2.3
2003	3.4	0.52	0.48		1.4
2004	3.2	0.53	0.51	1.9	1.1
2005	2.5	0.55	0.51	1.9	1.7
2006	2.6	0.53	0.66	2.0	2.8
2007	3.1	0.54	0.53	1.4	1.9
Average	**2.5**	**0.52**	**0.49**	**1.6**	**1.4**

The total nitrogen load to Lake Winnipeg between 1994 and 2007 was on average 90,701 t/y (Table 7.5, next page). The nitrogen load ranged from a high of 122,491 t/y in 2005 to a low of 51,737 t/y in 2003. As with phosphorous loads, inter-annual variability in nitrogen loading can primarily be attributed to variability in stream flow rather than in concentration (Figure 4.5, Section 4.3), as demonstrated by the relationship between annual stream flow and annual nitrogen load. However, the relationship with stream flow varies among rivers and moreover, is weakened by long-term trends in concentration in some of the tributaries to Lake Winnipeg including the Red River (Jones and Armstrong 2001). For example, in the Winnipeg River, stream flow accounts for 97 % of the variability in nitrogen load. In contrast, only 63 % of the variability in nitrogen loads in the Red River can be attributed to stream flow. However, as for phosphorous loading, the correspondence between annual flow and phosphorous fluxes in Red River is high, particularly in years of high or low flow. McCullough (2001) also noted that annual nitrogen yields tended to be well-correlated with annual stream flow in the Saskatchewan, Winnipeg, Dauphin and Red rivers (73 to 96 % of the variability explained by stream flow). Low nitrogen loading in 2003 was associated with the lowest annual inflows from the Red and Winnipeg rivers for the period 1994 through 2007, while high nitrogen loading occurred in 2005, the year with greatest flow in the Saskatchewan River and the second greatest in the Red and Winnipeg rivers (Figure 4.5, Section 4.3).

As with phosphorus, the Red River was the main source of nitrogen to Lake Winnipeg.

As with phosphorus, the Red River was the main source of nitrogen to Lake Winnipeg. The Red River contributed 34 % of the annual total load over the study period, or on average, 31,476 t/y of nitrogen to the lake (Figure 7.9, Table 7.3). As with phosphorus, most of the nitrogen load in the Red River (88 % on average) is

Table 7.5: *Annual average total nitrogen loads (tonnes per year) to Lake Winnipeg from major tributaries and atmospheric deposition.*

	Nitrogen (tonnes per year)													
	1994	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**
Red River	22,121	36,370	34,558	37,871	35,303	33,681	24,459	37,755	12,849	11,157	34,155	42,840	42,996	34,554
Winnipeg River	19,291	21,119	25,770	24,700	12,939	21,861	23,446	30,949	26,647	13,345	27,735	31,303	17,512	18,996
Saskatchewan River	6,938	7,016	9,467	13,760	8,870	9,592	7,678	3,037	5,167	5,440	7,026	16,211	16,012	13,841
Dauphin River	1,144	3,637	5,102	5,102	3,539	4,349	2,308	4,287	835	602	1,652	6,825	10,804	7,296
East Side Rivers	3,079	3,488	3,545	3,794	2,306	3,520	3,701	4,864	4,292	2,188	4,104	4,686	3,115	3,427
Brokenhead River	251	111	173	197	192	81	346	507	390	117	398	793	369	548
Fisher River	21	50	91	198	471	121	375	828	66	39	1,242	418	385	286
Icelandic River	19	70	185	234	397	87	391	535	75	49	389	615	344	221
Atmospheric Deposition	9,500	9,500	9,500	9,500	9,500	9,500	9,500	9,500	9,500	9,500	9,500	9,500	9,500	9,500
Nitrogen Fixation	9,300	9,300	9,300	9,300	9,300	9,300	9,300	9,300	9,300	9,300	9,300	9,300	9,300	9,300
Total	**71,664**	**90,662**	**97,691**	**104,656**	**82,818**	**92,093**	**81,505**	**101,562**	**69,121**	**51,737**	**95,501**	**122,481**	**110,336**	**97,969**
Outflow from Nelson River	26,077	32,793	45,385	54,494	36,059	30,320	41,886	50,403	23,588	32,024	39,655	65,336	53,165	50,822

contributed by the Red River basin, exclusive of the Assiniboine River. The annual contribution from the Red-Assiniboine rivers basin ranged from 19 to 43 %. Years when the Red River contributed a relatively small portion of the total nitrogen load were characterized by below average flows (2003) or below average total nitrogen concentrations (2002) in the Red River.

The Winnipeg River was the second largest contributor of total nitrogen to Lake Winnipeg, contributing on average 25 % of the annual total load to the lake (average 22,544 t/y). The contribution of total nitrogen from the Winnipeg River to Lake Winnipeg ranged from 16 to 39 % between 1994 and 2007. The Winnipeg River provided a small proportion of the total nitrogen load in those years when flows on

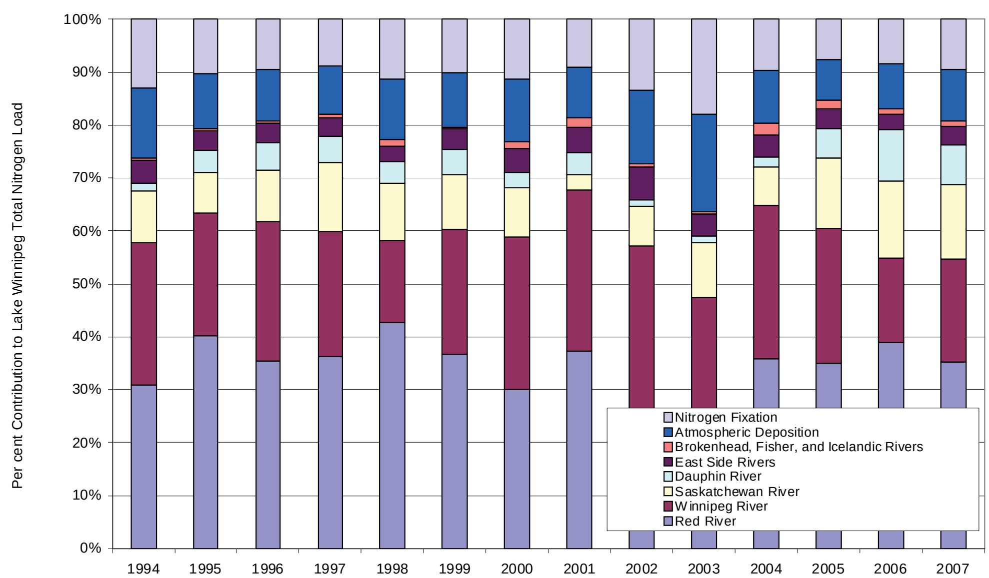


Figure 7.9: *Contribution of rivers, atmospheric deposition, and nitrogen fixation (%) to the total nitrogen load in Lake Winnipeg (1994 through 2007).*

the river were low (1998, 2006, and 2007). In contrast to phosphorus, for which the annual loading from the Red River was two to four times as large as from the Winnipeg River, nitrogen loading from the Red River was on average only 1.5 times larger than those from the Winnipeg River. In fact, in two years (2002 and 2003), nitrogen loading from the Winnipeg River was the larger of the two.

Other sources of nitrogen to Lake Winnipeg were relatively small compared to those from the Red and Winnipeg rivers. The Saskatchewan River, atmospheric deposition, and nitrogen fixation provided on average about 10 % each of the nitrogen load, although in the very low flow year (2003) nitrogen fixation and atmospheric deposition contributed 18 % to the lake. While a single estimate of atmospheric deposition is used for the entire study period, studies in the United States (including the U.S. portion of the Red River watershed) indicate that nitrogen deposition has a narrow range of annual fluctuation (Ruddy *et al.* 2006). In contrast, rates of nitrogen fixation can be expected to vary on a diurnal and annual scale (Ferber *et al.* 2004; Levine and Lewis 1984). As described previously, nutrient loads for the smaller contributing watersheds (for example the Dauphin, Brokenhead, Icelandic, and Fisher rivers) calculated from quarterly samples should also be interpreted with caution.

Average annual nitrogen export from Lake Winnipeg through the two channels of the Nelson River was 41,572 t/y (1994 through 2007). On average, about 54 % of the nitrogen entering Lake Winnipeg through the tributaries and atmospheric deposition was retained in the lake, with about 46 % exiting through the Nelson River.

On average (1994 through 2007), about 54 % of the nitrogen entering Lake Winnipeg through the tributaries and atmospheric deposition was retained in the lake, with about 46 % exiting through the Nelson River.

As with phosphorus, seasonal patterns of nitrogen loading to Lake Winnipeg were dominated by the seasonal flow regime of the Red River where inflow of nitrogen during April was more than double that observed in any other month (Figure 7.10, on the following pages). In contrast, there was only weak seasonal variability in nitrogen loading from the Saskatchewan and Winnipeg rivers (Figure 7.10), likely due to the fact that these flows are regulated. In the Saskatchewan River, small peaks in nitrogen loading corresponded with peak flows in the winter (February and March) and peak nitrogen concentrations in September. In the Winnipeg River, seasonal variation in nitrogen loading was strongly tied to variation in flow, with peak loading during peak flow in June and July (Figure 7.10 on the following pages).

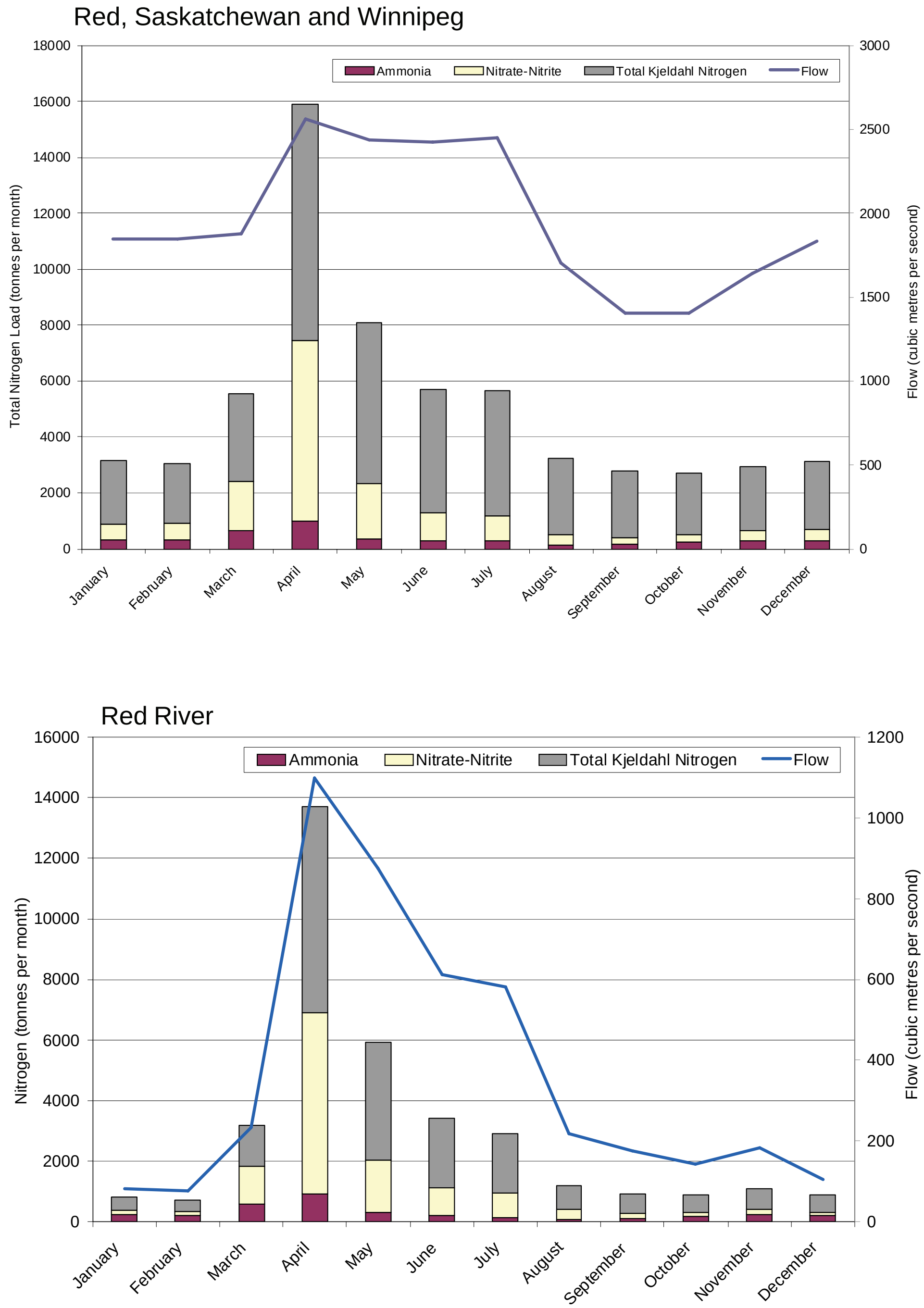
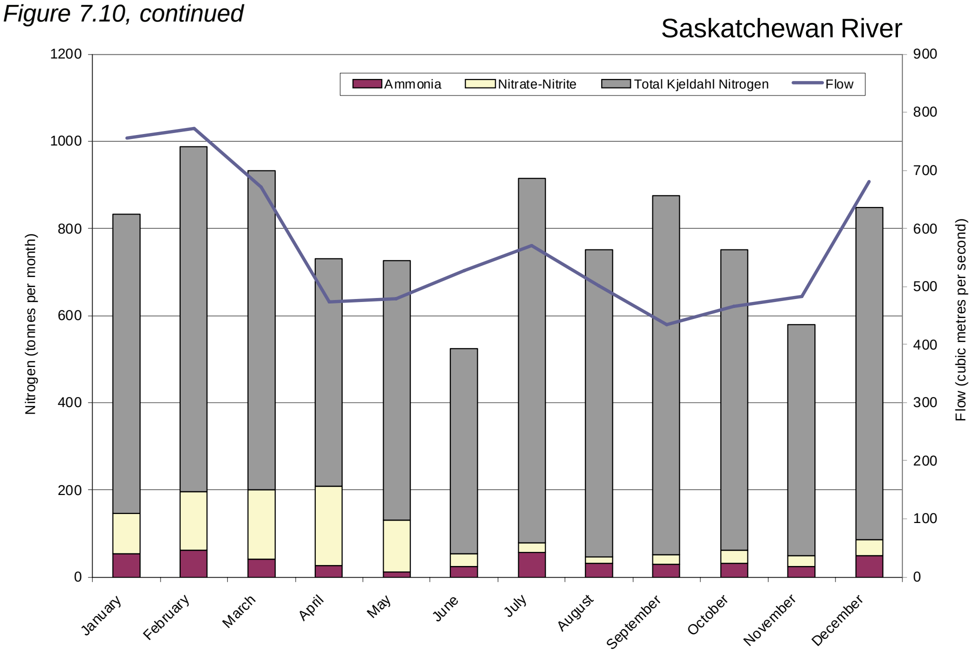
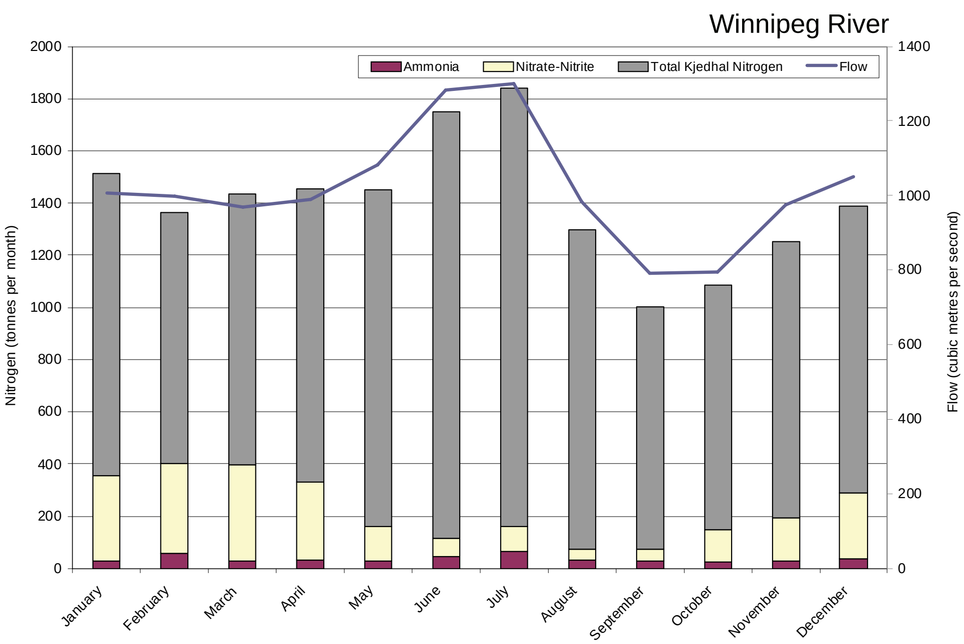


Figure 7.10: *Seasonal variation in total nitrogen loads to Lake Winnipeg from the Red, Saskatchewan, and Winnipeg rivers (monthly average from 1994 to 2007 where data were available).*

Continued ...

Figure 7.10, continued





7.3 Nutrient Yield from the Lake Winnipeg Watershed

Export of nutrients from land to surface water depends on the hydrological and precipitation regime, soil characteristics, and vegetation cover. Land use practices and management activities also greatly influence the transport of nutrients from land to surface waters. For example, in some instances, rates of nutrient export can be reduced by the presence of riparian vegetation along stream channels and lake shores, and by retention of nutrients in reservoirs, lakes, and wetlands. In contrast, the development of drainage channels and wetland drainage can have the opposite effect, resulting in increased nutrient export to surface waters. Finally, nutrients may also be taken up and transformed within streams themselves such that nutrient loads delivered downstream are significantly modified (Bernhardt *et al.* 2003).

A relatively large proportion of the Lake Winnipeg watershed is non-contributing and therefore does not contribute to nutrient export in most years.

Average annual phosphorous export from the watersheds tributary to Lake Winnipeg was 0.07 kilograms per hectare per year (kg/ha/y) (range 0.03 to 0.13 kg/ha/y between 1994 and 2007). This is low compared to, for instance, the Lake Simcoe watershed, where export ranged between 0.18 and 0.54 kg/ha/y from 1990 to 2007 (calculated from Lake Simcoe Region Conservation Authority 2009 and Winter *et al.* 2007). It was also considerably less than the approximate phosphorous export from each of the Laurentian Great Lake watersheds as calculated from published watershed areas (Lake Superior Centre 1998) and phosphorous loads during the 1970s, 1980s, and early 1990s (Neilson *et al.* 1995). Phosphorous export ranged from a low of 0.16 kg/ha/y from the Lake Superior watershed to 2.4 kg/ha/y from the Lake Erie watershed (excluding the upper Great Lakes above the St. Clair River) in the 1970s. Nutrient retention in the many lakes, reservoirs, and wetlands across the vast Lake Winnipeg watershed likely contribute to the relatively low rate of nutrient export. In addition, a relatively large proportion of the Lake Winnipeg watershed is non-contributing (31 %) and therefore does not contribute to nutrient export in most years (Figure 1.1). Non-contributing areas of the watershed are those marshes, wetlands, and other natural storage areas that prevent runoff from reaching the main stream in a year of average runoff (Prairie Farm Rehabilitation Administration 2008). However, even if the contributing area of the watershed is used to calculate export (704,086 km^2), rates of phosphorous export remain relatively low at 0.11 kg/ha/y (range 0.05 to 0.19 kg/ha/y between 1994 and 2007).

Figure 7.12 shows the close relationship between total inflow to Lake Winnipeg and the maximum rate of watershed phosphorous export observed during each year. Phosphorous export from the watershed was highest in 2005 when inflows to Lake Winnipeg were highest during the study period (Figure 4.5, Section 4.3). In contrast, phosphorous export from the watershed was at a minimum in 2003 when inflow to Lake Winnipeg was lowest.

The median phosphorous and nitrogen concentrations at the mouths of the Saskatchewan (0.016 and 0.473 mg/L, respectively), Dauphin (0.021 and 1.51 mg/L, respectively), and Winnipeg rivers (0.035 and 0.633 mg/L, respectively) are as much as an order of magnitude less than the 0.292 and 2.28 mg/L, respectively, at the Red River. Moreover, average export from each of these three major tributaries (0.01 to 0.08 kg/ha/y for phosphorus and 0.23 to 1.26 kg/ha/y nitrogen), is generally considerably less than from the Red River basin (0.19 kg/ha/y phosphorus and 1.10 kg/ha/y nitrogen). There are several reasons for these differences. The Winnipeg River basin is less populated and the land less intensively developed than the basins of the other major tributaries (Figures 3.4 and 3.5), so it develops lower nutrient concentrations, particularly for phosphorus, than the Red River basin. In addition, the Winnipeg River drains a Precambrian Shield watershed underlain by igneous

Figure 7.12: *Changes in total inflow to Lake Winnipeg and maximum observed phosphorous export in the watershed between 1994 and 2007.*

bedrock while the others drain plains watersheds characterized by thick sediments overlying sedimentary rocks (Figure 3.6). Ahl (1988), in a review of data collected in undeveloped forested watersheds, showed that basins on igneous rocks naturally export only one-half as much phosphorus as basins on sedimentary rocks. Furthermore, the Red River not only drains a sedimentary watershed, it and its tributaries frequently flood the deep lacustrine plain that forms its central valley. Floods are associated with marked increases in phosphorous concentrations as measured at the river mouth.

Another, perhaps more significant feature distinguishing the Red River from the Saskatchewan and Winnipeg rivers, is the absence of large lakes and reservoirs (Section 3.2). Lakes and reservoirs are efficient traps for both sediments and nutrients carried in rivers (Text Box 1). Nutrients contained in, or adsorbed onto, mineral sediments such as the silts and clays carried in plains rivers, tend to be trapped in lakes because they settle to the bottom and become incorporated into permanent lake sediments (Ahl 1988). Lakes can also function as traps for dissolved nutrients if they are consumed by algae, some of which also settles to the bottom. Although nutrients may also be recycled out of lake bottom sediments back into the water column (Text Box 6), the net flow of these nutrients is usually into the permanent sediments that accumulate at the bottoms of lakes. This tendency was reported as early as the 1970s by Larsen and Mercier (1976) who reported phosphorous retention in lakes ranging from 0 to 90 % with an average of 57 % retention. Lake Winnipeg itself clearly functions as a nutrient trap, retaining roughly the same proportion of phosphorus (60 %) as Larsen and Mercier found in their investigations. Lake Winnipeg retains a slightly lower proportion (46 %) of the nitrogen that is carried into it.

The Red River and its tributaries frequently flood the deep lacustrine plain that forms its central valley. Floods are associated with marked increases in phosphorous concentrations as measured at the river mouth.

The Saskatchewan River drains a landscape in Alberta that is as intensively farmed as the Red River basin (Figure 3.5). Nonetheless, it delivers much less phosphorus to Lake Winnipeg than the Red River. Median total phosphorous concentrations in spring runoff in streams draining agricultural watersheds in southern Alberta ranged up to 1.1 mg/L in one study (Ontkean *et al.* 2005) - higher than those monitored in southern Manitoba and more than an order of magnitude higher than the median concentration at the mouth of the Saskatchewan River. Some of the loss between headwaters and mouth of the Saskatchewan River can likely be attributed to in-channel storage of sediments in slow-flowing reaches or man-made reservoirs, but since dissolved phosphorus alone ranged up to 1.0 mg/L in the headwaters (in the same study of agricultural watersheds), some retention is likely due to sedimentation of plankton produced in the many reservoirs along the river's course.

Total phosphorous export from the land drained by tributaries and direct runoff into the Red River between Emerson and Selkirk was 0.66 kg/ha/y - the highest export coefficient calculated for a predominantly non-urban region in Manitoba.

Even in the southern Manitoba portion of the Lake Winnipeg watershed, there is sufficient variety of geology, soils, vegetation, land use, and hydrology that export of phosphorus varied considerably, from 0.01 to 0.33 kg/ha/y, across monitored sub-watersheds there (Table 7.6). Where water quality was monitored at multiple stations along major rivers, rates of nutrient export could also be determined for successive reaches (Table 7.7). Total phosphorous export from the land drained by tributaries and direct runoff into the Red River between Emerson and Selkirk (predominantly agricultural land on lacustrine clay deposited by glacial Lake Agassiz) was 0.66 kg/ha/y - the highest export coefficient calculated for a predominantly non-urban region in Manitoba. The total export from the next downstream reach (that is, from Headingley on the Assiniboine River and St. Norbert on the Red River, above the city of Winnipeg, to Selkirk, below the city) was 1.58 kg/ha/y, but this value includes the output of the City of Winnipeg's three wastewater treatment plants. To

Table 7.6: *Annual average nutrient loads and export to Lake Winnipeg, 1994 to 2007.*

Location	Total Nitrogen (tonnes/year)	Total Phosphorus (tonnes/year)	Gross Watershed Area (hectares)	TN kg/ha/y	TP kg/ha/y
Assiniboine River and Tributaries					
Assiniboine at Kamsack	502	52	1,300,000	0.39	0.04
Assiniboine at Brandon	2,440	410	9,370,000	0.26	0.04
Assiniboine at Treesbank	2,512	479	9,570,000	0.26	0.05
Assiniboine at Headingley	3,793	750	16,200,000	0.23	0.05
Qu'Appelle River	525	77	5,090,000	0.10	0.02
Souris River, Westhope	858	135	4,370,000	0.20	0.03
Souris River, Treesbank	1,222	239	6,110,000	0.20	0.04
Red River and Tributaries					
Red River at Emerson	17,282	2,554	10,200,000	1.69	0.25
Red River at St. Norbert	22,016	3,683	11,900,000	1.85	0.31
Red River at Selkirk	31,476	5,380	28,700,000	1.10	0.19
LaSalle River	265	59	180,000	1.47	0.33
Roseau River	722	67	502,000	1.44	0.13
Rat River	232	34	142,000	1.63	0.24
Pembina River	1,066	199	750,000	1.42	0.27
Seine River	69	10	30,200	2.29	0.33
Brokenhead River	319	23	158,000	2.02	0.14
Dauphin River	4,106	84	8,230,000	0.50	0.01
Fisher River	328	28	171,000	1.92	0.17
Icelandic River	258	30	124,000	2.08	0.24
Saskatchewan River, above Carrot River	11,721	1,094	37,640,000	0.31	0.03
Saskatchewan River, at Grand Rapids	9,290	377	40,600,000	0.23	0.01
Winnipeg River at Point du Bois	16,854	845	12,600,000	1.34	0.07
Winnipeg River at Pine Falls	22,544	1,049	13,600,000	1.66	0.08

Table 7.7: *Annual average nutrient loads and export to Lake Winnipeg from river reaches of the Assiniboine, Winnipeg, and Red rivers from 1994 to 2007.*

Location	Total Nitrogen (tonnes/year)	Total Phosphorus (tonnes/year)	Gross Watershed Area (hectares)	TN kg/ha/y	TP kg/ha/y
Assiniboine River					
Export between Kamsack and Brandon	1,938	358	8,070,000	0.24	0.04
Export between Brandon and Treesbank	72	70	200,000	0.36	0.35
Export between Treesbank and Headingley with Souris removed	59	33	520,000	0.11	0.06
Red River					
Export between Emerson and Selkirk (includes Assiniboine River watershed)	14,195	2,826	18,500,000	0.77	0.15
Export between Emerson and Selkirk (excludes Assiniboine River watershed)	10,402	2,076	2,300,000	4.52	0.90
Export between Emerson and St. Norbert	4,735	1,130	1,700,000	2.79	0.66
Export between St. Norbert and Selkirk with Assiniboine River, La Salle, Seine removed	5,004	615	389,800	12.8	1.58
Export between St. Norbert and Selkirk with Assiniboine River, La Salle, Seine and City of Winnipeg wastewater removed	5,004	207	389,800	3.67	0.53
Winnipeg River					
Nutrient Export between Point du Bois and Pine Falls	5,690	204	1,000,000	5.69	0.20

place these values in a global context, export rates are compared below with published data for other North American and European watersheds (Lorenz *et al.* 2009; Scott *et al.* 2006; House *et al.* 1997; Jarvie *et al.* 1997; Bayley *et al.* 1992). North American and European watersheds were chosen for comparison based on data availability and to provide a wide range of population density, land use, and geology.

The range of phosphorous export rates from sub-watersheds that make up the Lake Winnipeg watershed were similar to those modelled in the Chesapeake Bay watershed. Rates of phosphorous export across the Chesapeake Bay watershed ranged from less than 0.05 kg/ha/y to greater than 0.40 kg/ha/y (Chesapeake Bay Program 2009). Rates of phosphorous export were also similar in some Ontario watersheds such as the Barrie Creeks sub-watershed near Lake Simcoe (1.2 kg/ha/y) (Lake Simcoe Region Conservation Authority 2009). Similarly, rates of phosphorous export in watersheds across the Red, Upper Mississippi, Ohio, and Great Lakes River basins in the United States (Lorenz *et al.* 2009) ranged from a low of 0.03 kg/ha/y in the Souris River watershed in the Assiniboine/Red River basin - characterized by low precipitation and runoff, large headwater reservoirs, and a high proportion of non-contributing drainage area - to a high of 6.49 kg/ha/y in the primarily urban Salt Creek watershed in the upper Mississippi River basin.

Figure 7.13 shows the relationship between phosphorous load and catchment area for tributaries to Lake Winnipeg and selected North American and northern European rivers (Lorenz *et al.* 2009; Scott *et al.* 2006; House *et al.* 1997; Jarvie *et al.* 1997; Bayley *et al.* 1992). Phosphorous load increased with increasing catchment size from small forested headwater streams at the Experimental Lakes Area in northwestern Ontario to large tributaries to the North Sea in Europe. To facilitate

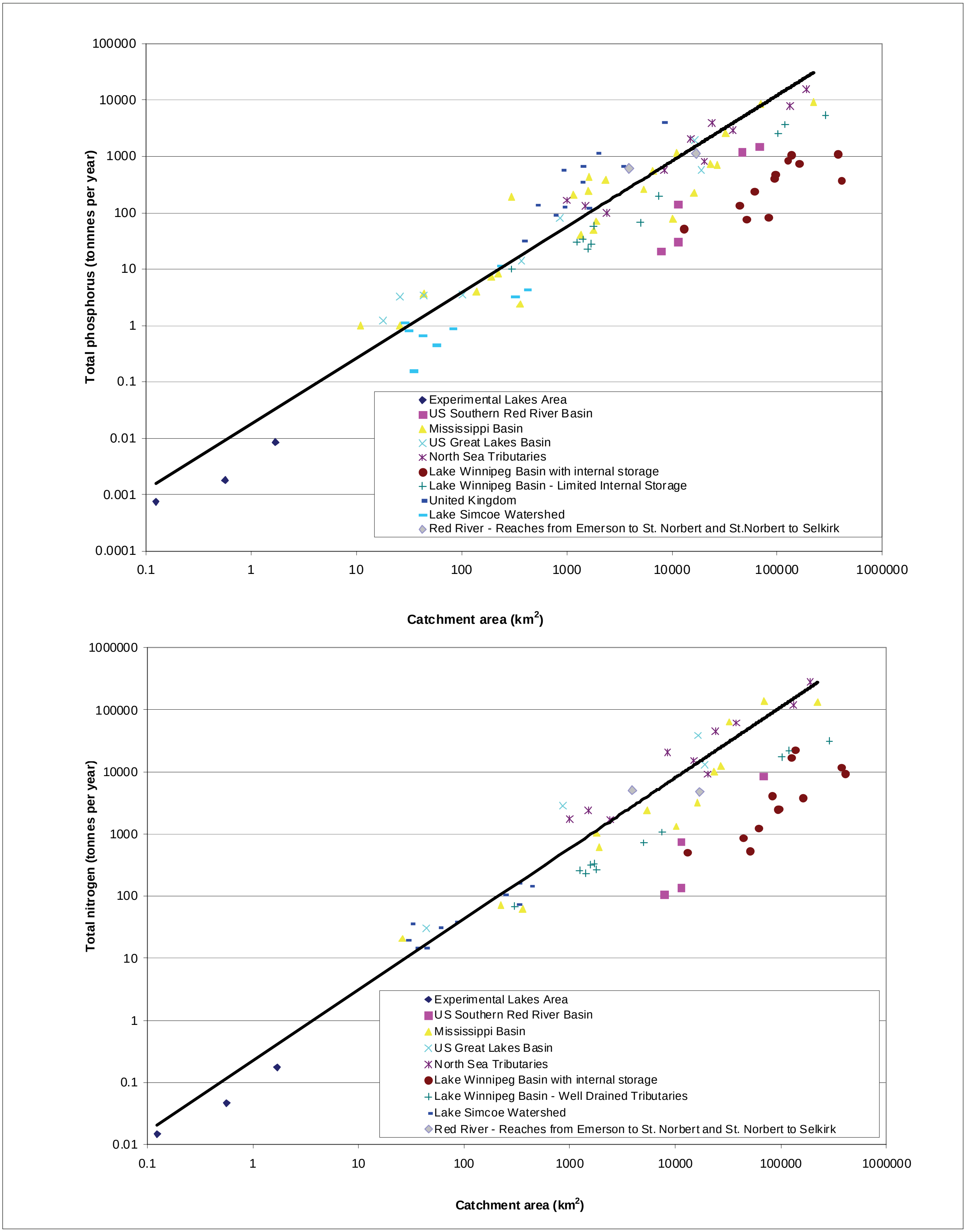


Figure 7.13: *Relationship between phosphorous and nitrogen loads and catchment area for rivers across North America, northern Europe, and in the Lake Winnipeg watershed.*

comparison, a trend line is drawn on Figure 7.13, calculated on data from the upper Mississippi River, Experimental Lakes Area, Lake Simcoe, and the northern European data sets. Among rivers in the Lake Winnipeg basin, those with large potential for internal storage in lakes and reservoirs and/or with large non-contributing areas (e.g. the Winnipeg, Dauphin, Saskatchewan, Qu'Appelle, Assiniboine, and Souris rivers; Figure 1.1) fall well below the trend line, indicating a relatively small phosphorous load per unit catchment area compared to the global rivers. In contrast, phosphorous load per unit catchment area is relatively high for the Red River reach from Emerson, at the international boundary, to St. Norbert just upstream of the city of Winnipeg and even higher from just upstream of the city of Winnipeg at St. Norbert to downstream of the city at Selkirk. Along these reaches of the Red River, phosphorous load per unit catchment area is similar to those measured for the North Sea and in the Mississippi River watershed.

Scatter in the relationship between phosphorous load carried and catchment area can be attributed in part to differences in land use and intensity of use. For example, those rivers draining to the North Sea tend to fall above the trend line. They drain densely populated, highly industrialized, and intensively farmed catchments (Jarvie *et al.* 1997). The North Sea catchment has an average population density of 193 people per square kilometre (Jarvie *et al.* 1997), so it is to be expected that phosphorous export from those rivers would be larger per unit area than across much of the Lake Winnipeg watershed. Similarly, some areas of the Mississippi, Ohio, and Great Lakes basins in the United States are also densely populated and intensively farmed, and have as many as 2,170 people/km^2 (Lorenz *et al.* 2009). The Manitoba portion of the Red River watershed (including the city of Winnipeg but excluding the less densely populated Assiniboine and Souris river watersheds) has a population density of just 26 people/km^2 (Statistics Canada 2007). However, the population density around the city of Winnipeg is relatively high compared to across the entire Nelson River drainage basin which has only 5.1 people/km^2 (Statistics Canada 2007). The comparatively high population density in the city of Winnipeg, the lack of tertiary wastewater treatment, a lack of potential for internal storage along these reaches, and the numerous point and non-point sources in this area likely contribute to the relatively high rate of phosphorous export along the main stem of the Red River (see below). In addition, others have noted (e.g. Jarvie *et al.* 1997), that differences in sampling methods, sample collection frequency, and load estimation methods can greatly impact estimation of phosphorous loads. Some smaller Lake Winnipeg basin sub-watersheds were monitored only four times per year, so that for these catchments, nutrient loads and export rates may have been underestimated.

In the Lake Winnipeg basin, between 1994 and 2007, annual average phosphorous export was highest in those river reaches characterized by urban and residential development.

In the Lake Winnipeg basin, between 1994 and 2007, annual average phosphorous export was highest in those river reaches characterized by urban and residential development such as along the Assiniboine River reach including Brandon (0.35 kg/ha/y), along the Winnipeg River through cottage country (0.20 kg/ha/y), and particularly on the reach of the Red River that includes Winnipeg (1.58 kg/ha/y). Relatively high phosphorous export from urban areas can be attributed to nutrient loading from wastewater treatment facilities serving larger populations and to higher rates of runoff from areas with low infiltration.

For the Red River within Winnipeg, the three wastewater treatment facilities provide more than 50 % of the phosphorous load exported from this river reach. These wastewater treatment facilities discharged about 375 tonnes of phosphorus per year, contributing between 4 and 18 % of the total phosphorous load in the Red River at Selkirk, with a higher proportion contributed in low flow years such as 2003. Much of

These results support the need for a multi-faceted approach to nutrient reduction that targets point and non-point sources within the city of Winnipeg.

the phosphorous load from the Winnipeg's wastewater treatment plants (>80 %) is readily available for algae growth (orthophosphorus). During the time of this study (1994 to 2007), the City of Winnipeg provided only secondary wastewater treatment with no additional tertiary treatment to remove nitrogen and phosphorus. However, even if the phosphorous load provided by the wastewater discharges is removed, phosphorous export in this reach would remain relatively high at 0.55 kg/ha/y. This can be attributed to point sources such as onsite wastewater management systems and dispersed sources such as lawn fertilizers and pet feces. Also, runoff from impermeable surfaces such as sidewalks, driveways, streets, and roofs is higher than from more permeable agricultural or naturally vegetated areas where infiltration reduces runoff volumes. McLeod (2007) studied urban runoff quality in Saskatoon and showed that total Kjeldahl nitrogen loads from urban runoff were estimated to exceed those contributed by the local wastewater treatment facility. Estimated phosphorous loads in runoff were smaller than those contributed by the wastewater treatment facility but were still a significant source of phosphorus delivered to the South Saskatchewan River. Overall, these results support the need for a multi-faceted approach to nutrient reduction that targets point and non-point sources within the city of Winnipeg.

Phosphorous export from the United States portion of the Red River watershed (as measured at Emerson, 1994 to 2007) was about 0.25 kg/ha/y (Table 7.6). A similar rate was reported by Lorenz *et al.* (2009) for the Red River at Grand Forks (0.22 kg/ha/y). Including the very low phosphorous export from the Assiniboine River (only 0.05 kg/ha/y), the export from the whole Red River watershed (as measured at Selkirk) was lower at 0.15 kg/ha/y (Table 7.7). Several lakes and reservoirs along the Assiniboine and Souris rivers (including lakes on the Qu'Appelle River, Lake of the Prairies on the Assiniboine River, the Rafferty and Alameda dams, and the Boundary Dam on the Souris River) likely contribute to a relatively low rate of nutrient export to the Assiniboine River as compared to the Red River. Even with the Assiniboine River excluded from the Red River watershed, rates of phosphorous export were considerably higher in the Canadian portion of the Red River watershed (0.90 kg/ha/y) than from the American portion. The higher rate of phosphorous export in the Canadian portion can only partly be attributed to nutrient loading from the City of Winnipeg's three wastewater treatment plants. Even upstream from Winnipeg, export to the reach from Emerson to St. Norbert is 0.66 kg/ha/y, three times the rate upstream from Emerson. Nearly one-half of this remaining difference can be attributed to differences in contributing drainage areas. Approximately 35 % of the United States portion of the watershed is non-contributing as compared to only 8 % in the Canadian portion (Prairie Farm Rehabilitation Administration 2008).

Phosphorous export was relatively high from the Seine River and La Salle River sub-watersheds (0.33 kg/ha/y). Both watersheds are part of the larger Red River watershed and are characterized by intensive agricultural activities (cultivated land and livestock operations). Flaten *et al.* (2003) noted that in 2000 and 2001, census regions in the Seine and La Salle sub-watersheds (and others in the larger Red River watershed) received phosphorus (from manure and inorganic fertilizer) in excess of crop requirements. In addition, both the Seine and La Salle rivers receive municipal wastewater discharged from wastewater treatment lagoons providing secondary treatment. Secondary treatment typically removes about 60 % of the phosphorus and 10 % of the nitrogen in wastewater (Chambers *et al.* 2001).

Phosphorous export from tributaries of the Red River in Manitoba such as the Roseau, Rat, Seine, and La Salle rivers (0.13 to 0.33 kg/ha/y) is at most one-half of that calculated for the main stem of the Red River between Emerson and St. Norbert

(0.66 kg/ha/y). A similar discrepancy exists with regard to nitrogen export. Nitrogen export calculated for the land draining into the Red River reach from Emerson to St. Norbert (2.79 kg/ha/y) is higher than the export calculated for any smaller sub-watershed in the region (1.44 to 2.29 kg/ha/y). One possible explanation for this apparent discrepancy is that nutrient loads, and therefore exports, calculated from quarterly water quality samples on the tributaries are underestimated compared to loads and export calculated from the monthly record for the main stem. However, nutrient loads from those tributaries not included in the study (e.g. Morris, Boyne, Little Morris rivers) may have contributed a relatively large proportion of the nutrient load to the Red River, or perhaps erosion within the main stem of the Red River significantly contributed to phosphorous export. Efforts to enhance water quality sampling in tributaries to the Red River should provide additional clarity in the future.

Between 1994 and 2007, annual average phosphorous export was lowest in the Dauphin and Saskatchewan river watersheds (0.01 kg/ha/y). In view of the discussion above, it is significant that both rivers drain through either reservoirs (the Saskatchewan) or lakes (the Dauphin) and more critically, that the largest lakes in these watersheds are situated only a short distance upstream from their final outlets into Lake Winnipeg (Section 3.2). The Saskatchewan River watershed is also marked by a significant proportion of non-contributing drainage area. Consequently, although both rivers drain considerable agricultural, industrial, and urban lands, in both cases, most of the nutrient load developed in upper reaches of their watersheds is trapped and stored as sediments before it can reach Lake Winnipeg (Text Box 1, Section 3.2). Relatively low rates of nutrient export calculated for other watersheds - including the Assiniboine, Souris, and Qu'Appelle rivers - can be attributed to either or both high nutrient retention in lakes and reservoirs or high proportions of non-contributing area.

Average total nitrogen export from the Lake Winnipeg watershed was 0.89 kg/ha/y (range 0.51 to 1.2 kg/ha/y between 1994 and 2007). As with phosphorus, nitrogen export was highly variable across the Lake Winnipeg watershed ranging from 0.1 kg/ha/y from the Qu'Appelle River sub-watershed (Assiniboine River) to 2.3 kg/ha/y from the Seine River sub-watershed (Red River) (Table 7.6). As for phosphorus, nitrogen export calculated for some main stem reaches was higher than for any individual tributary watersheds. In particular, nitrogen export in the Red River within Winnipeg was 12.8 kg/ha/y (Table 7.7). Nitrogen export in the United States portion of the Red River (as measured at Emerson) was about 1.69 kg/ha/y (Table 7.6) and 1.24 kg/ha/y at Grand Forks. With the Assiniboine River included, nitrogen export in the Canadian portion of the Red River watershed was somewhat lower (0.77 kg/ha/y) than from the United States portion of the Red River watershed (Table 7.7). However, when the Assiniboine River was excluded, nitrogen export was higher (4.52 kg/ha/y). As with phosphorus, higher nitrogen export in Canada is partly attributable to relatively high nutrient loading from the Winnipeg region and partly to the larger proportion of non-contributing areas in the United States portion of the Red River watershed. The City of Winnipeg's three wastewater treatment facilities discharge about 3,300 tonnes of nitrogen per year, contributing between 8 and 30 % of the total nitrogen load in the Red River at Selkirk, with a higher proportion contributed in low flow years such as 2003.

Average total nitrogen export from the Lake Winnipeg watershed was 0.89 kg/ha/y (range 0.51 to 1.2 kg/ha/y between 1994 and 2007).

Rates of nitrogen export across the Lake Winnipeg watershed were similar to those in the Chesapeake Bay watershed where nitrogen export ranged from 1.5 to 12 kg/ha/y (Chesapeake Bay Program 2009). Upper rates of nitrogen export were somewhat higher in several watersheds across the Red, Upper Mississippi, Ohio, and Great Lakes basins (Lorenz *et al.* 2009) where export ranged from 0.12 kg/ha/y in the Souris River to 33.1 kg/ha/y in the primarily agricultural Auglaize River in Ohio.

As with phosphorus, there is a strong relationship between nitrogen load and catchment area in selected rivers across North America and northern Europe (Figure 7.13; Lorenz *et al.* 2009; Scott *et al.* 2006; Jarvie *et al*. 1997; Bayley *et al.* 1992). Nitrogen load increased with increasing catchment size from the small headwater streams at the Experimental Lakes Area to the larger tributaries to the North Sea in Europe. Nitrogen loads per unit catchment area were also relatively large for the reaches of the Red River from Emerson at the international boundary to upstream of the city of Winnipeg at St. Norbert and then from St. Norbert downstream to Selkirk. As with phosphorus, contributions from wastewater receiving only secondary treatment, a lack of potential for internal storage, and numerous point and non-point sources in urban areas likely contributed to the relatively high nitrogen loads per unit catchment area (see below). Rivers in the Lake Winnipeg basin with large potential for internal storage in reservoirs and large non-contributing areas (Saskatchewan River, Qu'Appelle River, Assiniboine River, Souris River, etc.) fall well below the general trend line, indicating that they transport relatively little nitrogen per unit catchment area compared to the global rivers. As described above for phosphorus, scatter in the relationship between nitrogen load carried and catchment area can be attributed in part to differences in land use and intensity of use. For nitrogen, variation in atmospheric deposition may also contribute to differences in the mass of nitrogen carried per unit of catchment area. Lorenz *et al.* (2009) noted that nitrogen deposition varied from 74 to 921 kg/km^2 across the Red, Mississippi, Ohio, and Great Lakes basins. Rivers from the United States portion of the Red River watershed, along with rivers throughout the Canadian portion of the Lake Winnipeg watershed, may export less nitrogen per unit of catchment area due to reduced atmospheric deposition of nitrogen as compared to watersheds in Europe, central Ontario and the Mississippi, Ohio, and Great Lakes basins in the United States. Finally, as noted for phosphorus, differences in sampling methods, sample collection frequency, and load estimation methods can greatly impact estimation of nitrogen loads. Therefore, the relatively infrequent sampling of some Lake Winnipeg basin tributaries may have underestimated the nitrogen loads per unit catchment area (see Appendix 7.2 for a discussion on the La Salle River).

Nitrogen export was also highest in those river reaches characterized by urban and residential development, particularly on the Winnipeg River through cottage country and on the Red River within Winnipeg and between Emerson and St. Norbert.

Nitrogen export was also highest in those river reaches characterized by urban and residential development, particularly on the Winnipeg River through cottage country (5.69 kg/ha/y) and on the Red River within Winnipeg (12.8 kg/ha/y) and between Emerson and St. Norbert (2.79 kg/ha/y). For the Red River within Winnipeg, the three wastewater treatment facilities provide more than 70 % of the nitrogen that is exported from this river reach. During the time period of this study (1994 to 2007), the City of Winnipeg provided only secondary wastewater treatment with no additional tertiary treatment to remove nitrogen and phosphorus. However, even if the nitrogen load provided by the wastewater discharges is removed, nitrogen export in this reach is relatively high at 3.67 kg/ha/y. Nitrogen export was also high on the Seine River at 2.3 kg/ha/y, the Brokenhead River at 2.0 kg/ha/y, and the Icelandic River at 2.1 kg/ha/y. About 23 % of the Brokenhead River sub-watershed in its western, downstream portion devoted to cereal and forage crop production while the eastern portion of the watershed is primarily forested uplands and wetlands. The Icelandic River watershed also contains agricultural land, primarily in the south-central portion of the watershed with relatively undeveloped forests and wetlands in the north and western portions of the watershed (East Interlake Conservation District 2007).

Text Box 7. Eutrophication Model for Lake Winnipeg

(Weitao Zhang, Ram Yerubandi)

The health of Lake Winnipeg has declined over the past three decades in response to increased loadings of nitrogen and phosphorus to the lake. Increased nutrient loading and associated eutrophication have contributed to the increasing frequency and magnitude of algal blooms (McCullough 2009), as well as shifts in phytoplankton composition (Kling 1998). A four box spatially segmented eutrophication model was developed using the Water Analysis Simulation Program (WASP) to simulate the major nutrient and algal dynamics in Lake Winnipeg (Plate 1).

The eutrophication model for Lake Winnipeg includes two major nutrient cycles (nitrogen and phosphorus) and three functional phytoplankton groups (non-cyanobacteria, nitrogen-fixing cyanobacteria, non-nitrogen-fixing cyanobacteria) for the north and south basins of the lake (Plate 2). The distinct morphological and hydrological features and climate conditions of the two lake basins were also used in the eutrophication model. Calibration and validation of the model has recently been completed for the 2002 to 2007 period using physical, chemical, and biological data collected from the lake, and further model refinements are currently underway. The model will be used to support nutrient management strategies and to assist in establishing ecologically-relevant nutrient objectives for Lake Winnipeg.

Future water quality modelling of Lake Winnipeg will consider incorporation of silica kinetics and zooplankton grazing dynamics into the model structure, integration of a hydrodynamic model, a long-term simulation period to study feedbacks associated with scenarios of nutrient reduction or climate change, and uncertainty analysis. Data partners: M. Stainton, S. Page (Fisheries and Oceans Canada), E. Page (Manitoba Water Stewardship).

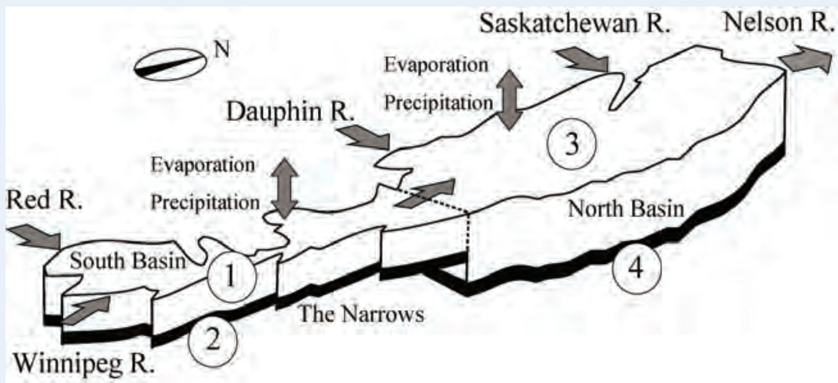


Plate 1. *The schematic of segmentation and water balance of Lake Winnipeg WASP model. Segment 1: South Basin; Segment 2: South Basin Sediment; Segment 3: North Basin; Segment 4: North Basin Sediment.*

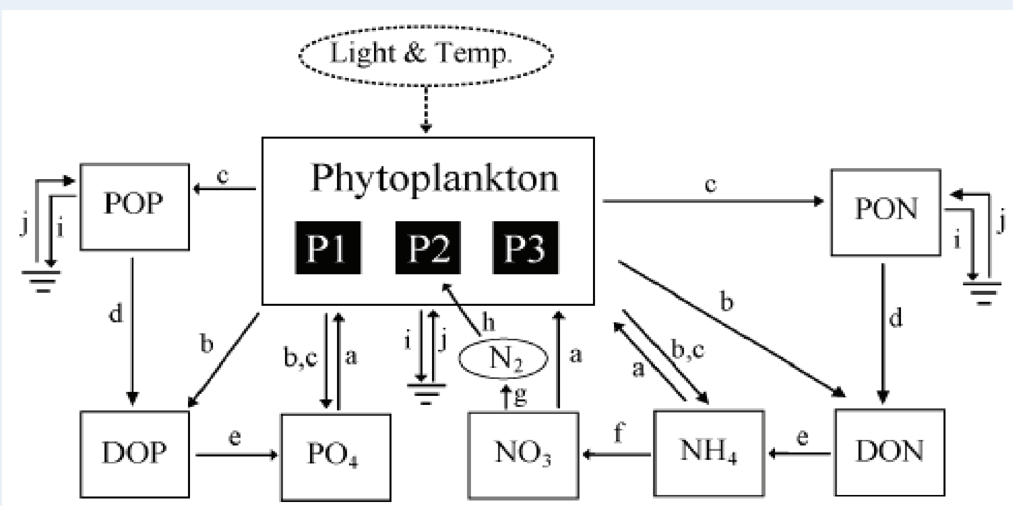


Plate 2. *The phosphorous and nitrogen cycles of the Lake Winnipeg WASP model. The solid line arrows indicate flows of matter through the system. P1: Phytoplankton group 1(non-cyanobacteria); P2: Phytoplankton group 2 (nitrogen-fixing cyanobacteria); P3: Phytoplankton group 3 (non-nitrogen-fixing cyanobacteria). (a) phytoplankton uptake; (b) phytoplankton respiration; (c) phytoplankton death; (d) detritus dissolution; (e) mineralization; (f) nitrification; (g) denitrification; (h) cyanobacteria N-fixation; (i) settling; (j) resuspension.*

8.0 BIOLOGICAL CHARACTERISTICS OF LAKE WINNIPEG

8.1 Phytoplankton

(Elaine Page)

Phytoplankton biomass and species composition vary spatially and temporally in aquatic ecosystems and are regulated by a number of factors including light, temperature, water column stability, nutrients, grazing, pathogens, and toxins. 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 has been the development of large blooms of cyanobacteria in the north basin of the lake. Although more limited in areal extent, dense blooms have also been apparent along beach shorelines and in the offshore areas of the south basin. In addition to bloom formation during the summer months, noticeable changes have also 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 (Section 9.1, Text Box 5). 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 bloom material sampled from offshore and nearshore areas of the lake (Section 9.2).

Algal blooms have occurred historically in Lake Winnipeg. However, there is evidence to indicate that bloom frequency and intensity has increased over time.

The phytoplankton community composition of Lake Winnipeg was initially monitored in the early 1920s (Lowe 1924), with subsequent surveys in 1928 and 1929 (Bajkov 1930, 1934), in the mid-1960s (Rybicki 1966), the late 1960s (Brunskill, unpublished), and periodically through the 1990s (Kling 1996, 1998; Manitoba Water Stewardship, unpublished). From 1999 to 2007, phytoplankton biomass and community composition were determined at a select number of sites from a network of 14 stations that are part of the Manitoba Water Stewardship long-term monitoring program on Lake Winnipeg (Figure 2.1; Appendix 1). However, sampling effort was not consistent each year and varied depending on station accessibility during strong wind events. Smaller research vessels were often used to collect samples, particularly between 2000 and 2002 and in 2005 when the *MV Namao* was not accessible, which prevented safe access to certain stations. Phytoplankton samples were collected consistently from all 14 long-term stations in 2006 and 2007, and were used to summarize the spatial and seasonal variation in phytoplankton biomass and species composition in Lake Winnipeg (Section 8.1.2).

Phytoplankton species composition and biomass has also been intensively sampled and investigated by other agencies and individuals working on Lake Winnipeg (see Section 9.1). Data in this section and Section 9.1 have limited comparability given differences in sampling effort, sample collection methods, taxonomic identification, and biomass estimates. For instance, phytoplankton samples collected by Manitoba Water Stewardship were collected as depth integrated euphotic zone samples and analyzed by ALS Laboratories, whereas samples in Section 9.1 were collected as discrete surface samples and analyzed by Algal Taxonomy and Ecology Inc. Further details as to the differences in methods applied in the phytoplankton investigations may be found in Appendix 1.

8.1.1 Chlorophyll a

Temporal Variation

From 1999 to 2007, annual average chlorophyll a concentrations for the north basin of Lake Winnipeg ranged from 4.1 µg/L to 34.5 µg/L with a mean open water chlorophyll a concentration of 14.4 µg/L (Figure 8.1). Mean annual chlorophyll a concentrations in the south basin and narrows were similar or low in comparison to the north basin of Lake Winnipeg (mean = 8 µg/L, range = 3.6 to 21.1 µg/L). 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 (Figures 8.1 below, and 8.2 on the following page). Similar to chlorophyll a concentrations, there was large inter-annual variation in mean annual phytoplankton biomass with the greatest mean biomass occurring in 2006 (Figure 8.2).

Chlorophyll a concentrations were greatest in 2006 and corresponded to satellite imagery showing showing large surface bloom development during the late summer months.

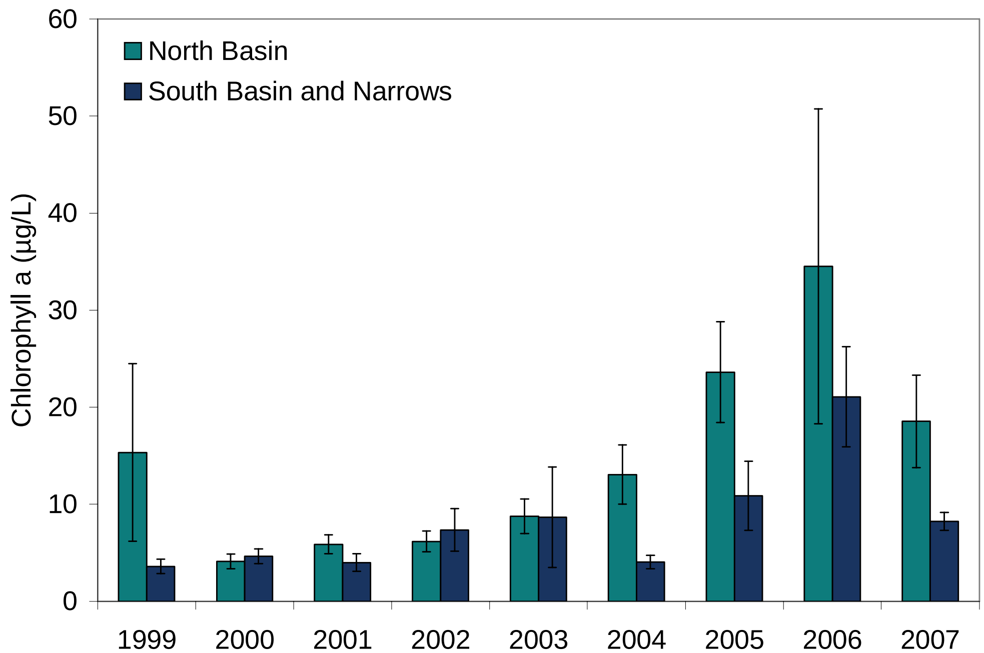


Figure 8.1: *Mean annual chlorophyll a concentration for the north basin and south basin and narrows region (µg/L; ± SE) of Lake Winnipeg, 1999 to 2007. Annual means represent average of euphotic zone samples collected from May to October in a given year.*

Unpublished data from the 1969 survey of Lake Winnipeg were summarized to characterize historical algal biomass in the lake. Direct comparison of the 1969 data to the most recent data presents some challenges because of the difference in the type of samples, the depth of sample collection, and the inter-annual variability in the sampling network. Discrete water samples were collected from a range of depths in 1969, whereas samples summarized in the current section represent depth integrated euphotic zone samples. As such, the 1969 chlorophyll a data were summarized using discrete samples collected between 0 to 4 m given that a majority (95 %) of euphotic zone samples collected from 1999 to 2007 were collected within

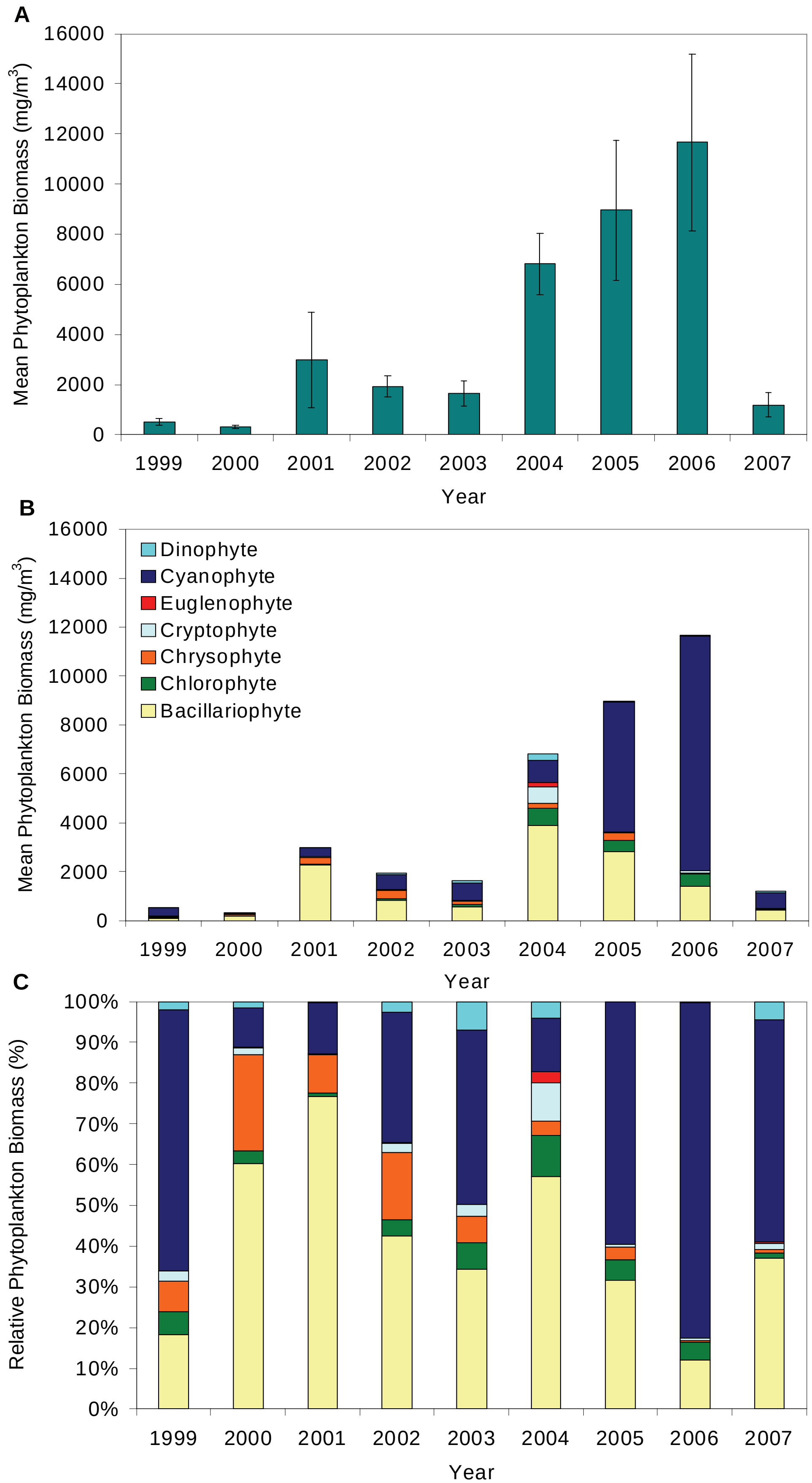


Figure 8.2: *Mean annual total phytoplankton biomass (mg/m^3; ± SE; A) and absolute (mg/m^3; B) and relative phytoplankton composition by class (%; C) in Lake Winnipeg, 1999 to 2007 (n = 10 to 40). Annual means represent the mean of euphotic zone samples collected between July and October in a given year.*

this same depth range. The 1969 mean open water chlorophyll a concentration for the south basin and narrows (3.7 µg/L) was at the lower end of the 1999 to 2007 range in mean annual chlorophyll a concentrations (range = 3.6 to 21.1 µg/L). In the north basin, the mean open water chlorophyll a concentration in 1969 (2.3 µg/L) was below the mean annual range of concentrations from 1999 to 2007 (range = 4.1 to 34.5 µg/L). Differences between 1969 and the most recent data may be partly related to differences in the sampling methodologies as noted earlier. However, other work on Lake Winnipeg has also found increases in phytoplankton biomass between the 1969 survey and data collected in the 1990s and 2000s (Section 9.1, Text Box 5).

According to the trophic classification for lake waters (Environment Canada 2004; Organisation for Economic Cooperation and Development 1982), mean annual chlorophyll a concentrations for the two lake basins suggest that the lake ranged from mesotrophic (2.5 to 8 µg/L) to eutrophic (8 to 25 µg/L) conditions between 1999 and 2007. However, on the basis of total phosphorous trophic classification, both basins of the lake were generally considered eutrophic in most years (Section 6.2.1).

Seasonal and Spatial Variation

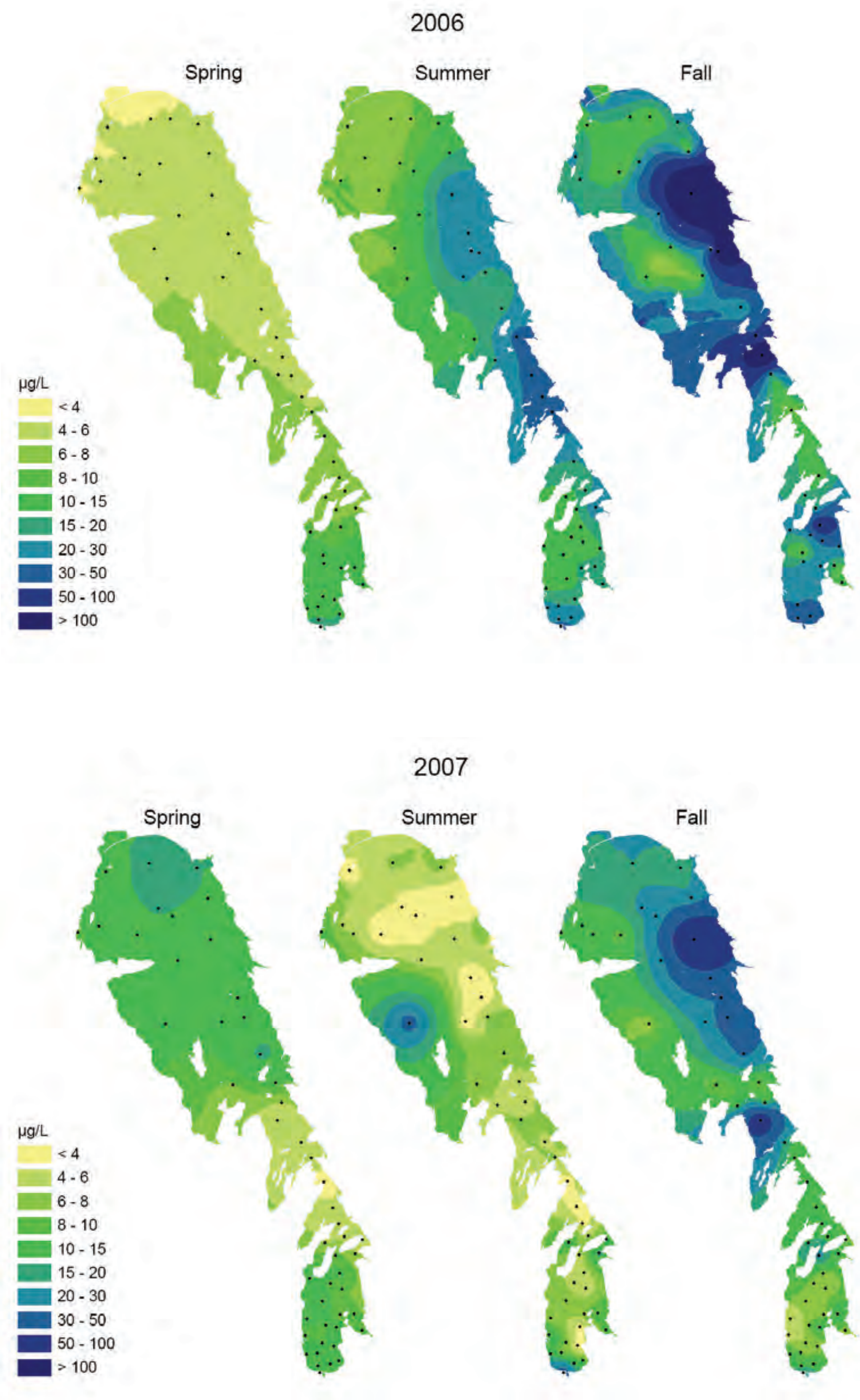
Chlorophyll a samples were collected most intensively in 2006 and 2007 and were mapped geospatially to describe the seasonal and spatial variation in chlorophyll a concentrations in Lake Winnipeg. Seasonal patterns in chlorophyll a differed between the two years in both basins of Lake Winnipeg (Figure 8.3 on the following page). In 2006, chlorophyll a concentrations were generally lowest in spring and increased progressively through summer and fall. In 2007, chlorophyll a in the north basin was generally lower in the summer (in most areas) with greater spring and fall biomass, while there were little seasonal differences in chlorophyll a in the south basin. In eutrophic north-temperate lakes, chlorophyll a concentrations are typically highest in the spring and fall with biomass minima in the summer (Marshall and Peters 1989). It is possible that lower chlorophyll a concentrations in the north basin during the mid-summer period of 2007 may have been partly related to nutrient deficiency in Lake Winnipeg.

In 2006, the lowest chlorophyll a biomass occurred in the spring at the very north end of the lake and near the mouth of the Saskatchewan River where concentrations were less than 4 µg/L. In 2007, chlorophyll a concentrations less than 4 µg/L generally occurred during the summer in areas of the north basin and narrows, and at a few isolated stations along the east side of the south basin near Grand Beach (Figure 8.3, next page). In spring 2006, there was a clear south-north gradient in chlorophyll a ranging from 67.8 µg/L near the inflow of the Red River to 1.8 µg/L at the north end of the lake near Limestone Bay. However, the spring south-north gradient in chlorophyll a was not apparent in 2007, with lower concentrations in the narrows region (4 to 6 µg/L) and similar chlorophyll a concentrations in the south and north basins (ca. 8 to 10 µg/L; Figure 8.3).

In some years, prolific blooms are observed along the east side of the north basin where light and nutrients may be optimal for phytoplankton growth.

Particularly evident in 2006 was the progression of bloom development along the east side of the north basin and the narrows through the summer and fall periods. In some years, higher chlorophyll a and prolific blooms are observed along the east side of the north basin where light and nutrient concentrations are likely optimal for phytoplankton growth. Large algal blooms in this particular area of the north basin may also be partly associated with the prevailing winds from the northwest that effectively concentrate large surface scums along the eastern shoreline.

__Figure 8.3:__ Seasonal and spatial variation in chlorophyll a in Lake Winnipeg in 2006 and 2007.



8.1.2 Phytoplankton Biomass and Composition

A total of 146 genera were identified in the 280 samples collected from Lake Winnipeg from 1999 to 2007. Overall, a large proportion (>80 %) of the total phytoplankton biomass in Lake Winnipeg was represented by as few as five genera belonging to cyanobacteria (*Aphanizomenon, Anabaena, Microcystis*) and diatoms (*Aulacoseira, Stephanodiscus*) while most other genera contributed to <5 % of the total biomass (Table 8.1). The five dominant genera are generally characteristic of eutrophic systems (Wetzel 2001; Reynolds 1998) and most are known to dominate the phytoplankton assemblages of other large shallow eutrophic lakes (e.g., Laugaste 1996). Considering the relative occurrence of phytoplankton genera in all Lake Winnipeg samples, only 11 genera were in more than 50 % of all samples collected from 1999 to 2007. *Cryptomonas, Aulacoseira*, and *Aphanizomenon* were among the most ubiquitous genera occurring in >80 % of all samples collected over the nine-year period in Lake Winnipeg (Table 8.1).

Temporal Variation

The average (1999 to 2007) lake-wide biomass was dominated by the cyanobacteria (cyanophytes), which comprised 51 % of the mean total biomass (Table 8.2, next page). Twenty-six cyanobacteria were identified to genus, with three major genera *Aphanizomenon, Anabaena*, and *Microcystis* together accounting for 95 % of the total cyanobacteria biomass. Of these three genera, *Aphanizomenon* was by far the most dominant, comprising over 60 % of the mean biomass (Table 8.1). Phytoplankton species composition varied over the nine-year period coincident with large variation in phytoplankton biomass and chlorophyll a (Figure 8.2B; Figure 8.3). From 1999 to 2007, average annual cyanobacteria biomass was lowest in 2000 and comprised only 10 % of the total phytoplankton biomass. Cyanobacteria biomass was greatest in 2006 and comprised almost 82 % of the total phytoplankton biomass and was largely related to elevated densities of *Aphanizomenon*. It is of note that the mean monthly air temperature for April 2006 was the warmest April recorded between 1999 and 2007 (Section 4.1) and may have been one of several factors influencing both the timing and development of cyanobacteria.

Table 8.1: *Relative biomass (%) of common phytoplankton taxa and relative occurrence (%) of taxa in samples collected from May to October in Lake Winnipeg, 1999 to 2007. Common taxa were arbitrarily defined as taxa with >0.1% of the total biomass.*

Group		Total Biomass (%)	Relative Occurence in samples (%)
Bacillariophyte			
	Aulacoseira (Melosira)	6.8	83.2
	Stephanodiscus	5.2	70.7
	Fragilaria	1.2	34.6
	Amphiprora	1.1	9.3
	Asterionella	0.9	42.1
Chlorophyte			
	Coelastrum	1.3	46.4
	Pediastrum	1.1	65.4
	Eudorina	0.4	11.4
	Oocystis	0.3	58.9
	Dictyosphaerium	0.2	24.6
	Closterium	0.1	46.4
	Gloeococcus	0.1	6.1
Chrysophyte			
	Unidentified Chrysophytes	1.2	55.0
	Uroglenopsis	0.2	0.4
	Mallomonas	0.1	42.5
Cryptophyte			
	Cryptomonas	0.9	86.4
	Rhodomonas	0.6	78.2
Euglenophyte			
	Phacus	0.2	4.6
Cyanophyte			
	Aphanizomenon	61.2	82.1
	Anabaena	5.3	54.3
	Microcystis	5.2	17.1
	Coelosphaerium	1.6	7.1
	Aphanocapsa	1.2	14.6
	Oscillatoria	0.1	14.3
	Gomphosphaeria	0.1	28.6
Dinophyte			
	Peridinium	0.5	19.6
	Ceratium	0.4	24.3

Table 8.2: *Mean phytoplankton biomass (mg/m³), relative composition (%), the number of taxa summarized by class, and a list of the dominant taxa identified in euphotic zone samples collected from Lake Winnipeg from 1999 to 2007.*

Group	Mean Phytoplankton Biomass (mg/m3)	Relative Composition (%)	Total Number of Taxa	Dominant Taxa
Bacillariophyte	1,393	35	25	*Melosira, Stephanodiscus, Fragilaria*
Chlorophyte	211	5	60	*Coelastrum, Pediastrum*
Chrysophyte	155	4	18	Unidentified Chrysophytes, *Uroglenopsis*
Cryptophyte	103	3	10	*Cryptomonas, Rhodomonas*
Euglenophyte	23	1	4	*Phacus*
Cyanophyte	2,058	51	26	*Aphanizomenon, Anabaena, Microcystis*
Dinophyte	62	2	3	*Peridinium, Ceratium*

Diatoms (bacilliariophytes) were the next largest group comprising 35 % of the mean (1999 to 2007) total biomass in Lake Winnipeg. Although a total of 25 genera were identified, nearly 80 % of the diatom biomass in the lake was comprised of the centric diatoms *Aulacoseira (Melosira)* and *Stephanodiscus*, and the pennate diatom *Fragilaria*. Large increases in cyanobacteria biomass were accompanied by decreases in diatom biomass. For instance, diatoms comprised only 12 % of the mean total algal biomass in 2006 (Figure 8.2B, C) when cyanobacteria biomass was at its greatest. Diatoms comprised up to 77 % of the total algal biomass in 2001 (3,844 mg/m³; 78 % total biomass), when the proportion of cyanobacteria was relatively low. 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/m³ and 9,037 mg/m³) that were dominated by *Aulacoseira* and *Stephanodiscus*. Because a limited number of samples were collected in 2001 (n = 7) as compared to other years, the composition and biomass in 2001 may not be representative of the phytoplankton assemblage in that particular year.

Cyanobacteria and diatoms comprised more than 80 % of the total phytoplankton biomass in Lake Winnipeg from 1999 to 2007.

Earlier studies of Lake Winnipeg phytoplankton have also found large inter-annual differences in phytoplankton composition and suggested that temporal shifts in composition may have been partly explained by climate (Kling 1996). For instance, in wet, cool years, such as 1992, diatoms and cryptophytes comprised a large proportion of the biomass. In contrast, phytoplankton composition was dominated by cyanobacteria in dry, warm years such as 1969 and 1994 (Kling 1996). Overall, the relative diatom contribution in the Manitoba Water Stewardship data set was lowest in the two warmest years (1999 and 2006) from 1999 to 2007. Consequently, these were also the same years when the relative cyanobacteria biomass was the highest in Lake Winnipeg (Figure 8.2C).

The relative contribution of chlorophytes (green algae) to the mean (1999 to 2007) phytoplankton biomass of Lake Winnipeg was small in comparison (5 %) to diatoms and cyanobacteria, but this group was considerably more diverse than the other major phytoplankton groups (60 genera). The mean (1999 to 2007) relative contribution of the chrysophytes, cryptophytes, dinophytes, and euglenophytes was small, comprising between about 1 to 4 % of the total mean biomass in Lake Winnipeg from 1999 to 2007. From 1999 to 2007, the relative contribution of chlorophytes and cryptophytes ranged from <1 (in several years) to 10 % in 2004. The relative contribution of chrysophytes was highly variable and comprised as much as 25 % of the total biomass in 2000 and was less than 1 % of the phytoplankton biomass in 2006 and 2007 (Figure 8.2C). Mean annual relative dinophyte contribution was also variable (<1 to 7%) and euglenophytes did not contribute significantly to the total biomass from 1999 to 2007.

Seasonal and Spatial Variation

Samples for analyses of phytoplankton biomass and community composition were collected most intensively by Manitoba Water Stewardship in 2006 and 2007. These data were summarized to characterize the seasonal and spatial dynamics of phytoplankton in Lake Winnipeg during the open water period of these two years. In the spring of 2006 and 2007, phytoplankton communities were generally dominated by diatoms in Lake Winnipeg (Figures 8.5 and 8.6, on the following pages). In both years, *Aulacoseira* sp. and *Stephanodiscus* sp. were among the dominant genera. However, *Cyclotella* sp. and *Asterionella formosa* were also dominant in 2006 and 2007, respectively. Other algal groups contributing to the spring phytoplankton assemblage in Lake Winnipeg included the chlorophytes (in 2006) and dinophytes (in 2008; mainly *Perdinium* sp.). Diatoms are typically dominant in the spring because of their lower optimal temperature and low light requirements for growth as compared to other phytoplankton groups. In the summer, there was a subsequent shift to cyanobacteria dominance at most sites in 2006, while in summer of 2007, the phytoplankton assemblage was much more variable with a mixed assemblage of cyanobacteria, diatoms, and chrysophytes (Figures 8.5 and 8.6). Possible nutrient deficiency and cooler conditions in 2007 may partly explain the relatively low biomass and the mixed phytoplankton assemblage in the summer of 2007. *Aphanizomenon* sp., *Anabaena* sp., and *Microcystis* sp. were among the dominant cyanobacteria during the summer in both years. In the fall, cyanobacteria generally comprised a considerable amount of the phytoplankton biomass in Lake Winnipeg in both years. However, diatoms were an important component of the fall phytoplankton assemblage in the north basin in 2007.

Diatoms are typically dominant in the spring because of their lower optimal temperature and low light requirements for growth as compared to other phytoplankton groups.

Similar to seasonal patterns in chlorophyll a, total phytoplankton biomass in 2006 was generally lowest in the spring and highest in the fall (Figures 8.5 and 8.6). In 2006, much of the seasonal increase in phytoplankton biomass was attributed to the increase in cyanobacteria biomass in Lake Winnipeg. Wet and cool conditions prevailed during 2007 and phytoplankton biomass was generally an order of magnitude lower as compared to 2006 (note ten-fold scale differences between Figure 8.5 and Figure 8.6). In 2007, seasonal patterns in phytoplankton biomass were also similar to seasonal patterns in chlorophyll a. In the north basin, biomass was generally higher in the spring and fall with biomass minima during the summer. Seasonal fluctuations in the south basin were generally not considerable in 2007, with exception of a single site located centre of the south basin (W10) during the fall which was characterized by a large *Microcystis* sp. bloom.

Spatial variation in phytoplankton biomass was generally similar to spatial patterns in chlorophyll a. In 2006, with exception of a few sites, the south basin biomass was generally low in comparison to the north basin. Phytoplankton biomass was typically higher along the east side of the north basin (at W3, W5, and W7) in the spring, summer, and fall, while the north and northeastern areas of the north basin (W1, W2, and W4) had notably lower phytoplankton biomass. These spatial patterns in phytoplankton distribution through the north basin were consistent with the west-to-east gradient in nutrients (Figure 6.4 and 6.8) and total suspended solids (Figure 5.19). In the open water season of 2007, phytoplankton biomass was generally lowest in the narrows and highest in the north basin (Figure 8.6). Higher phytoplankton biomass occurred along the east side of the north basin in the spring of 2007, although did not appear to progress through the summer and fall as observed in 2006.

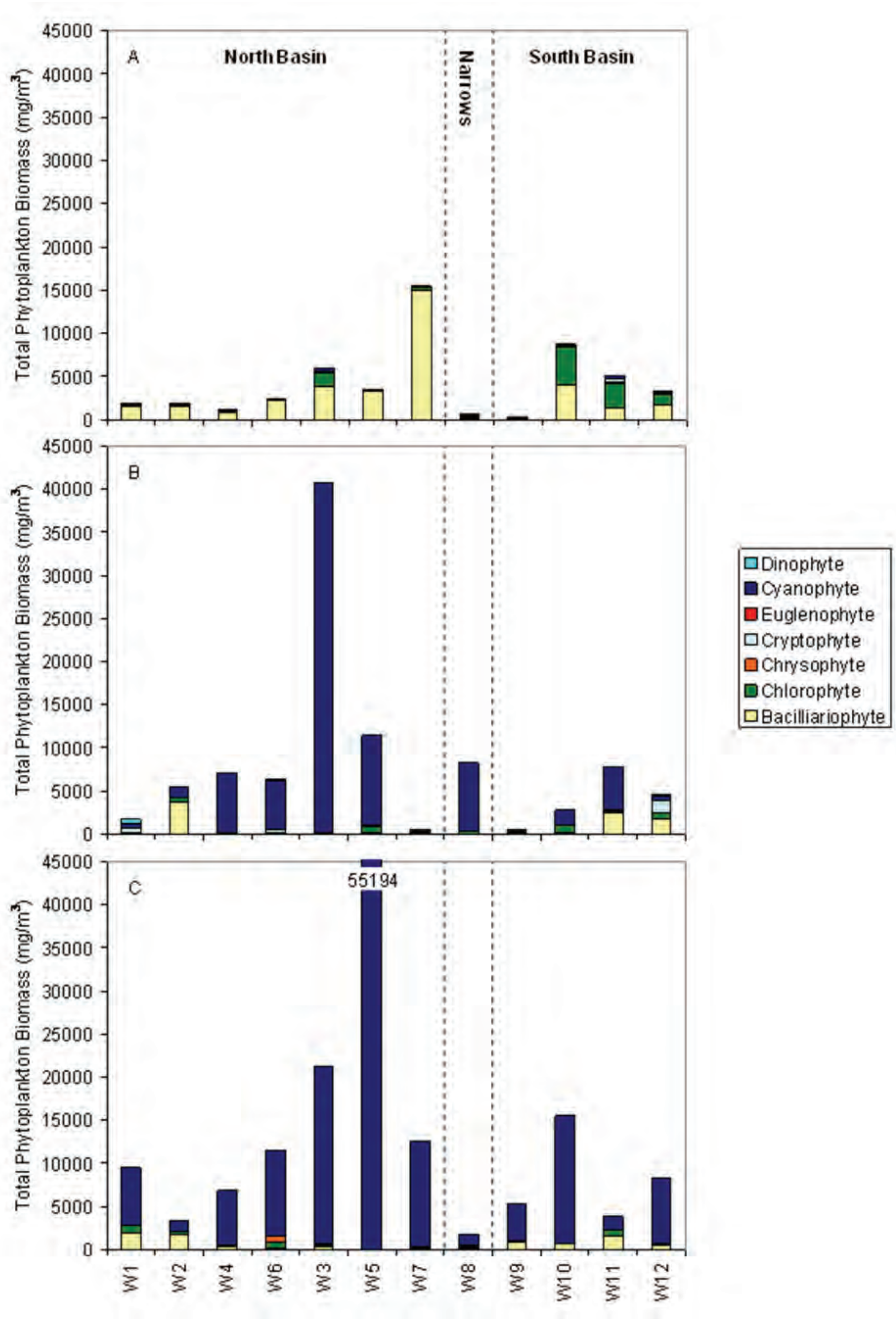


Figure 8.5: *Spatial variation in total phytoplankton biomass (mg/m³) and community composition at the 12 long-term stations located in the north basin, narrows, and south basin of Lake Winnipeg during the spring (A), summer (B), and fall (C) monitoring periods of 2006.*

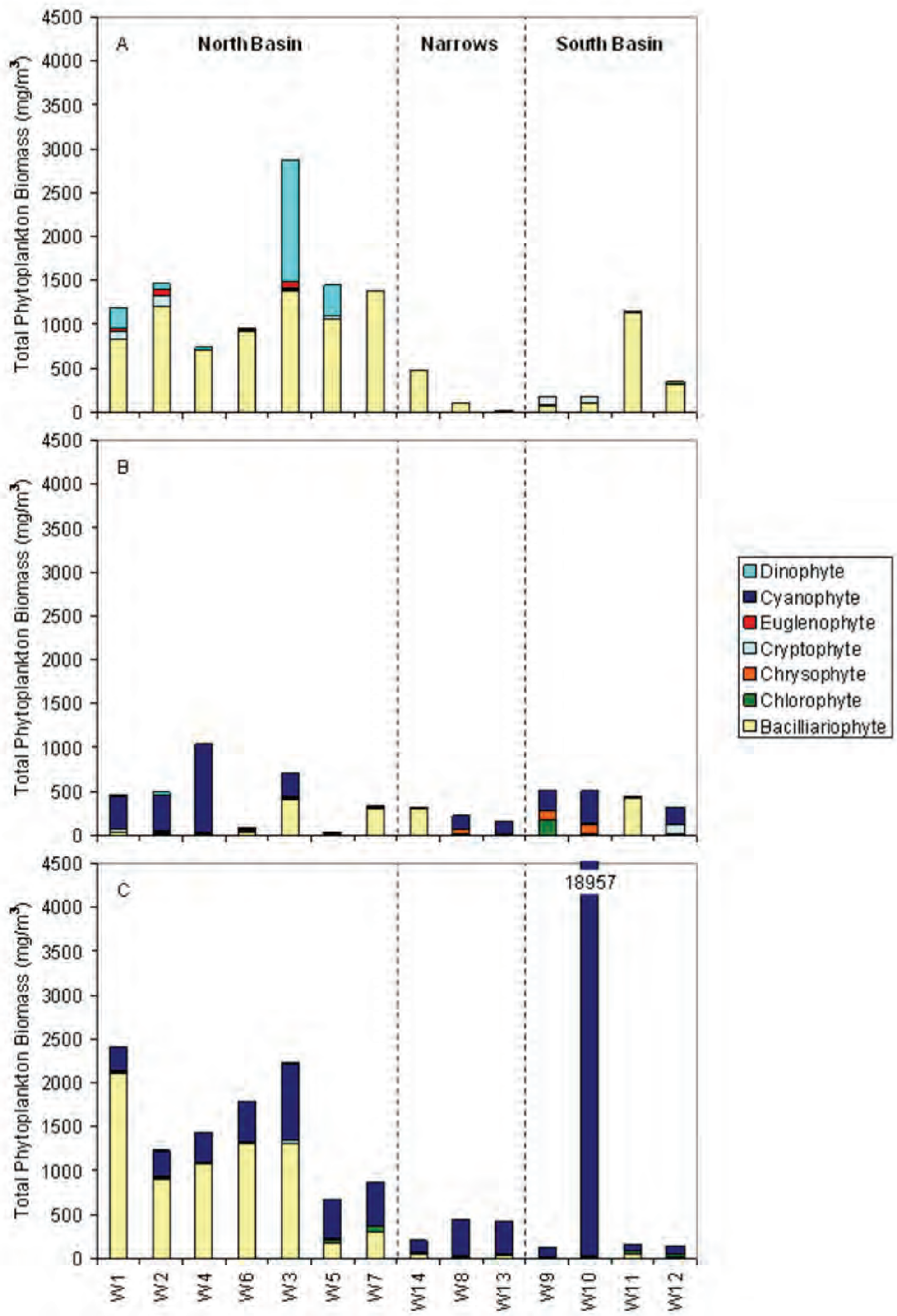


Figure 8.6: *Spatial variation in total phytoplankton biomass (mg/m^3) and community composition at the 14 long-term stations located in the north basin, narrows, and south basin of Lake Winnipeg during the spring (A), summer (B), and fall (C) monitoring periods of 2007.*

8.1.3 Under-Ice Conditions

Under-ice phytoplankton biomass may be limited by a number of factors including light and temperature. Chlorophyll a and phytoplankton biomass and community composition have been measured under-ice (January to April) by Manitoba Water Stewardship. Chlorophyll a concentrations were measured at a limited number of stations from 2000 to 2007. Under-ice phytoplankton species composition data were collected in several years over the study period with the most extensive record collected in 2001.

Phytoplankton biomass is typically lowest under ice cover, and may be constrained by light transmission through the ice, water temperature, and dissolved oxygen concentrations in the lake sediments.

The mean (2000 to 2007) under-ice chlorophyll a concentration (3.2 µg/L) was 3.5 times lower than the mean (2000 to 2007) open water concentration for the whole lake (11.2 µg/L). Phytoplankton biomass is typically lowest under ice cover, and may be constrained by light transmission through the ice, water temperature, and dissolved oxygen concentrations in the lake sediments. Under-ice biomass is also dependant on the viability of the resting stages, the size of the seed bank, and endogenous factors (e.g., maturation period) (Rengefors and Anderson 1998). Mean under-ice concentrations of chlorophyll a in Lake Winnipeg were highly variable from year to year with the lowest concentration in 2005 (1.08 µg/L) and the highest in 2007 (8.8 µg/L). Considering all individual samples collected from 2000 to 2007, under-ice biomass ranged from <0.5 µg/L to 43 µg/L in the north basin and from <0.5 µg/L to 13.3 µg/L in the south basin and narrows. The range in chlorophyll a concentrations was small in certain years where under-ice biomass remained consistently low across the lake (e.g., 2005, range = <0.5 to 2 µg/L). Under-ice chlorophyll a data collected in 2007 indicated that biomass was highest near Long Point (43 µg/L) and in the centre of the north basin (30 µg/L) (Figure 8.7). Higher chlorophyll a concentrations were also found north of the narrows (12 µg/L) and northeast of Reindeer Island in the centre of the north basin (12 µg/L). Chlorophyll a concentrations in the south basin were comparatively low and did not exceed 2 µg/L.

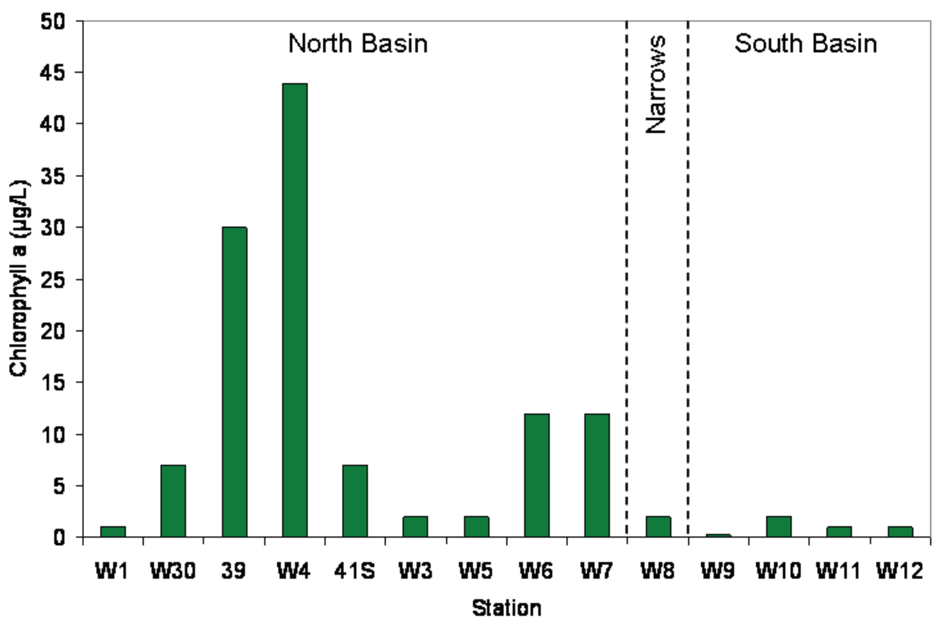


Figure 8.7: *Chlorophyll a concentrations (µg/L) measured at 14 stations during the under-ice period (January-April), 2007.*

Under-ice phytoplankton community composition and biomass were best characterized in 2001 when a greater number of samples were collected. In 2001, phytoplankton biomass remained low (ca. 100 mg/m^3) in the narrows and south basin of the lake and was mainly comprised of diatoms (Figure 8.8). *Aulacoseira* was among the dominant diatom at all sites in the narrows and the south basin in 2001. Bajkov (1930, 1934), found that the large centric diatom *Melosira* (now *Aulacoseira*) as well as other diatoms was characteristic of winter and early spring conditions in Lake Winnipeg. Kling *et al.* (2006) reported that biomass maxima under-ice ranged from 2,000 to 4,000 mg/m^3 and was dominated by *Aulacoseira islandica*. The average phytoplankton biomass during the period in this report was approximately six times higher in the north basin than in the south and was mainly comprised of the dinoflagellate *Perdinium* (Figure 8.8). Dinophytes and cryptophytes have been found under-ice because of their ability to reposition themselves in the water column to harvest light (Phillips and Fawley 1998). *Aulacoseira* is typically dominant in years with little snow cover while dinophytes (e.g., *Perdinium*) are dominant during high snow cover conditions (H. Kling, pers. comm.). Cryptophytes, chrysophytes, and chlorophytes were found in the north basin in 2001. However, the relative composition of the three aforementioned groups was less than 7 % of the total under-ice phytoplankton community in the north basin. Chlorophytes and chrysophytes were sub-dominant groups under-ice in the south basin and comprised approximately 15 % of the total phytoplankton biomass (Figure 8.8). Cyanobacteria were not abundant under-ice in 2001 and have not been found to contribute significantly to the under-ice community of Lake Winnipeg.

Aulacoseira is typically dominant (under-ice) in years with little snow cover while dinophytes (e.g., *Perdinium*) are dominant during high snow cover conditions.

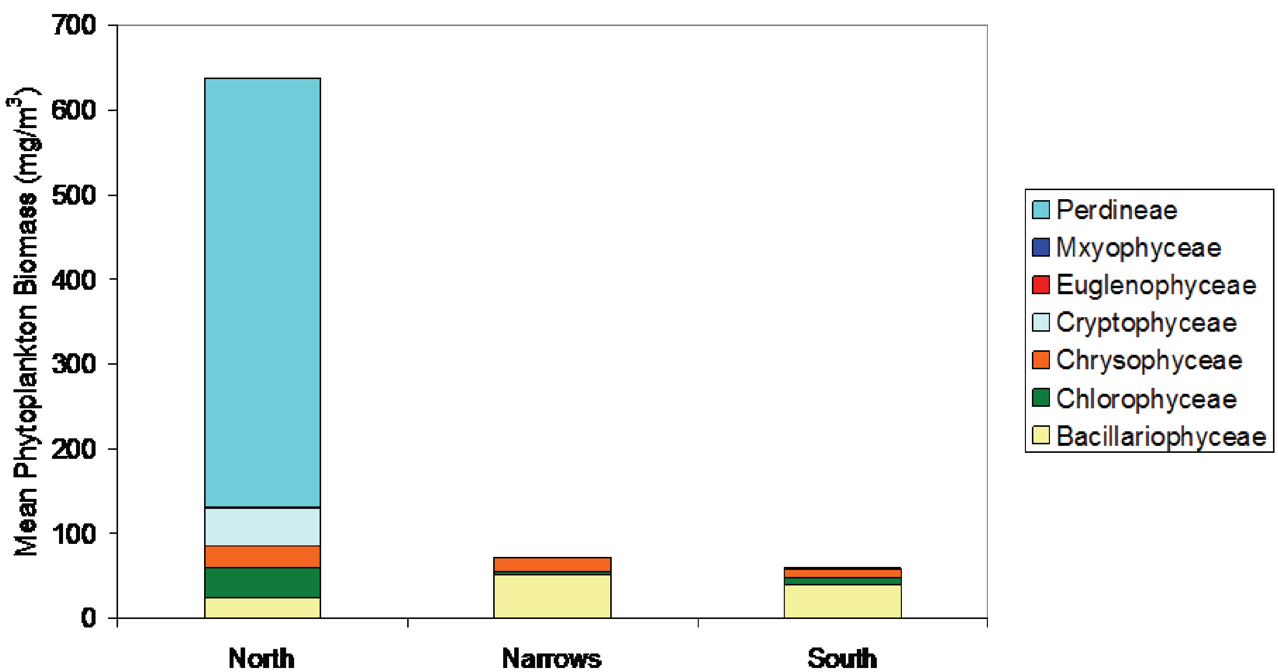


Figure 8.8: *The under-ice phytoplankton community composition and biomass (January to April; mg/m^3) in the north basin (n = 12), narrows (n = 2), and south basin (n = 6) of Lake Winnipeg in 2001.*

8.1.4 Cyanobacteria Dynamics

Together *Aphanizomenon*, *Anabaena*, and *Microcystis* comprised 95 % of the cyanobacteria biomass in Lake Winnipeg from 1999 to 2007. Among the cyanobacteria, genera capable of nitrogen fixation (N-fixing), *Aphanizomenon* and *Anabaena*, for example, were prevalent during the summer and fall from 1999 to 2007 in Lake Winnipeg. The relative biomass of N-fixing genera comprised between 11 % (in 2007) and 97 % (in 2003) of the total cyanobacteria biomass (Figure 8.9). Considering the relative biomass across all years, the N-fixing cyanobacteria comprised almost 70 % of the cyanobacteria in Lake Winnipeg. The dominant N-fixing genera varied depending on year. *Anabaena* was dominant in 2003 and 2004, while *Aphanizomenon* dominance was coincident with the large peak in cyanobacteria in 2006 (Figure 8.9).

Microcystis was largely responsible for increases in non-N-fixing cyanobacteria biomass in the south basin of Lake Winnipeg.

In certain years (e.g., 1999, 2005, and 2007), blooms of non-N-fixing cyanobacteria (for example *Microcystis*) have also comprised a significant fraction of the cyanobacteria biomass (Figure 8.9). Relative biomass of non-N-fixing genera has comprised from 4 % (in 2000) to 89 % (in 2007) of the total cyanobacteria biomass. In 1999, 2005, and 2007, non-N-fixing cyanobacteria comprised 74, 78, and 89 % of the summer and fall cyanobacteria biomass, respectively. *Microcystis* was largely responsible for increases in non-N-fixing biomass. All large blooms of *Microcystis* (>10,000 mg/m^3) occurred in the south basin of Lake Winnipeg at stations located in the northern part of the south basin in the offshore regions of the lake. Of all the samples containing *Microcystis* (n = 47), almost 90 % of observations occurred in the narrows and the south basin. Over the nine-year period of record, *Microcystis* was not commonly found in the north basin and biomass remained relatively low, never exceeding 750 mg/m^3. *Microcystis* blooms typically occur in lakes with high summer chlorophyll a concentrations and low water transparency (Chorus and Bartram 1999) during calm conditions of the late summer months (Reynolds 1984). The south basin and narrows region of Lake Winnipeg are turbid and nutrient-rich (Sections 5.3 and 6) and may favour the development of *Microcystis* blooms during calm conditions.

Algae on Lake Winnipeg off Hecla Island.

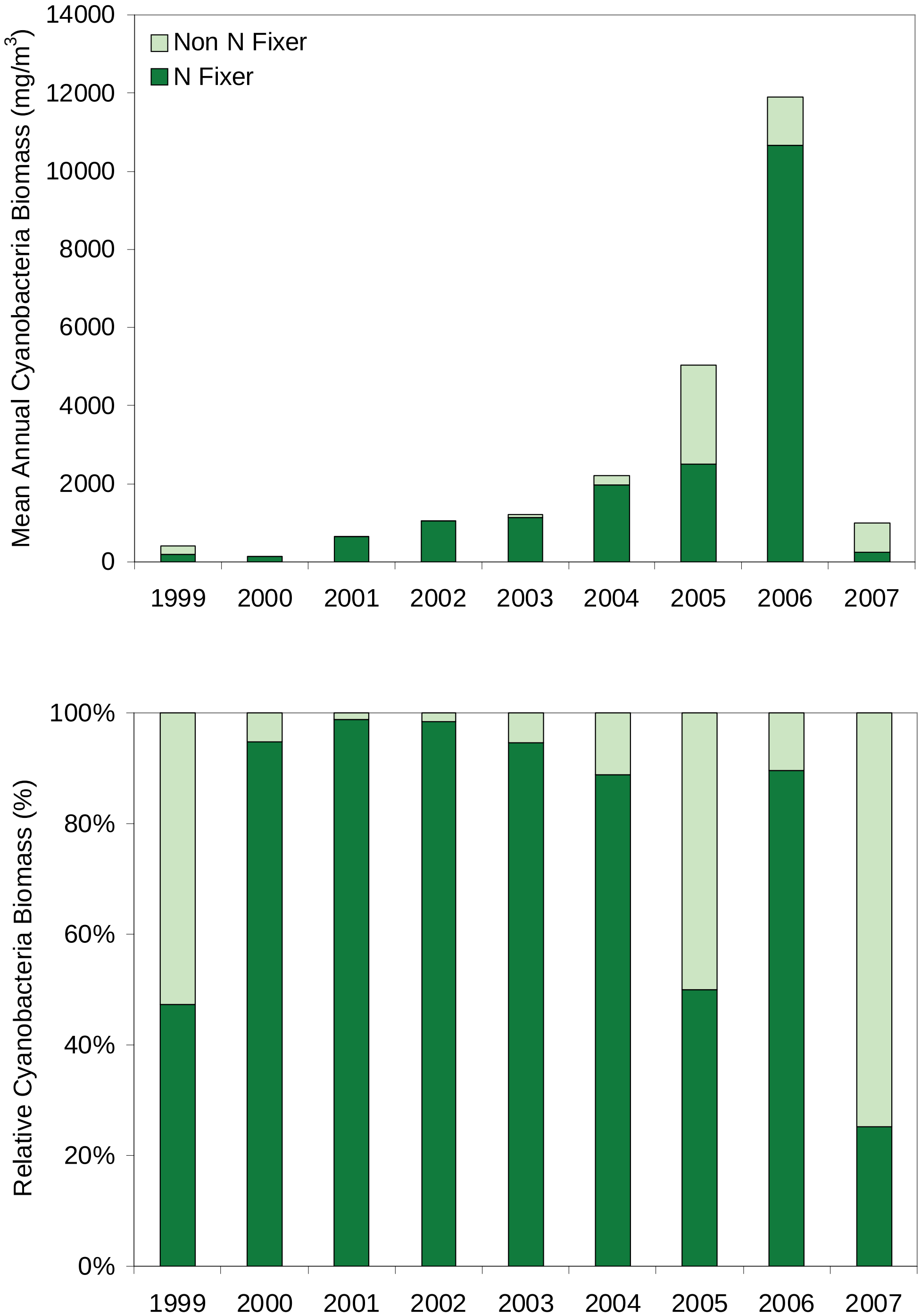


Figure 8.9: *Absolute phytoplankton biomass (mg/m³) and relative biomass of N-fixing and non-N-fixing cyanobacteria in Lake Winnipeg, 1999 to 2007. N-fixers include Aphanizomenon and Anabena and non-N-fixers include Microcystis.*

8.2 Zoobenthos

(Brenda Hann)

Benthic macroinvertebrates inhabit the bottom sediments of lakes for at least part of their life cycle (Resh and Rosenberg 2010). Benthic macroinvertebrates have been recommended for use in assessing water quality and ecosystem health because they are ubiquitous, long-lived, and the group may comprise several hundred species. The species-rich community includes those with broad tolerance to environmental conditions (known as "generalists") as well as others with narrow tolerance to specific environmental stressors (known as "specialists"). These latter species may be particularly useful as indicators of environmental stressors in the habitat.

Benthic macroinvertebrates play a critical role in the aquatic food web in lakes, serving as a major component of the diet of many fish. For example, populations of large crustaceans (e.g., *Diporeia,* a freshwater shrimp, and insect larvae and *Hexagenia,* a mayfly) are important in fish diets in the Laurentian Great Lakes (Krieger *et al.* 2007; Nalepa *et al.* 2007). Fluctuations in densities of crustaceans in these lakes have been associated with large changes in other components of the food web, including commercially important fish species.

Benthic macroinvertebrates play a critical role in the aquatic food web in lakes, serving as a major component of the diet of many fish.

In Lake Winnipeg, benthos was sampled qualitatively in 1929 (Bajkov 1930), quantitatively during open water periods in 1969 (Flannagan *et al.* 1994), and sporadically during the 1970s, typically using an Ekman dredge, but occasionally a Ponar dredge, as required by coarse bottom sediments. Quantitative sampling began again in 2002, and included up to 60 stations in the north basin, the narrows, and the south basin of Lake Winnipeg during spring (May to June), summer (August), and fall (September to October) with a standard Ekman dredge and 200 micron mesh sieve.

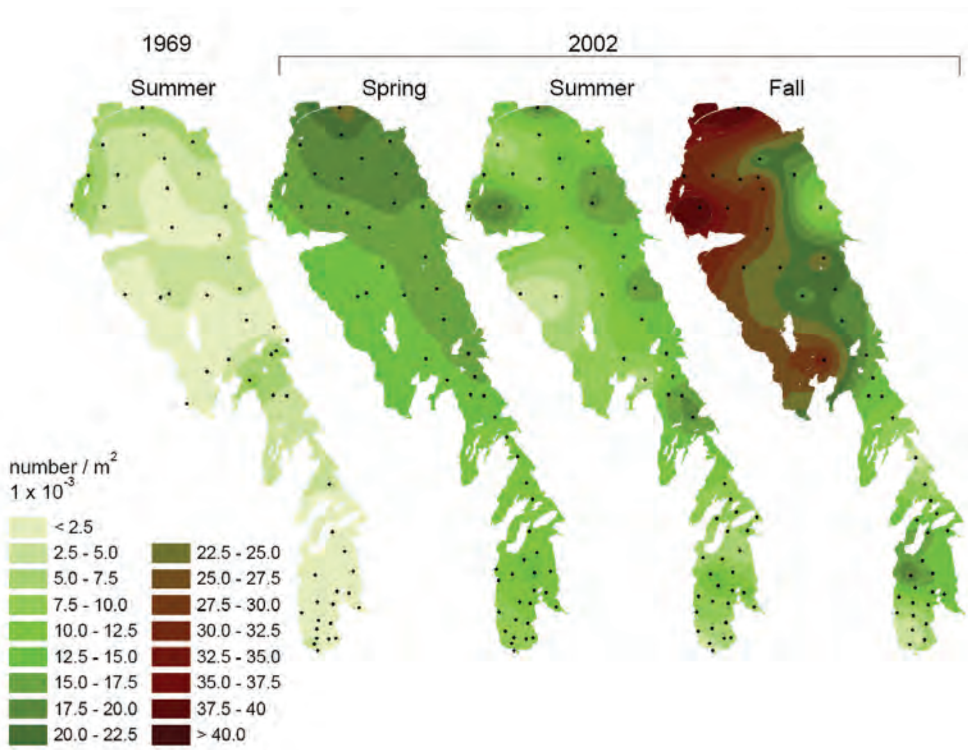


Figure 8.9: *Total density (numbers per m^2) of benthic macroinvertebrates in Lake Winnipeg. (Data Source: University of Manitoba, B. Hann)*

8.2.1 Seasonal and Spatial Dynamics

Benthic organisms in Lake Winnipeg as a whole have undergone substantial increases in density in recent decades, especially when comparing 1969 and 2002 (representative of the most recent decade) (Figure 8.9), where mean density in the lake more than tripled. Increased density in the north basin was particularly noticeable and varied seasonally and spatially within the basin (Figure 8.9). Highest densities of benthic macroinvertebrates occurred in spring and fall, in part a function of the life cycles of major groups of organisms. Many insect species overwinter as late stage larvae and emerge as flying adults during spring. These adults mate and lay eggs. The larvae develop over the summer months, by fall reaching sizes large enough to be retained on the sieves used to separate the organisms from the sediments. The highest densities of benthic macroinvertebrates were found in fall 2002 in the western portions of the north basin, especially near the mouth of the Saskatchewan River as it enters from Cedar Lake.

Midges (Family Chironomidae) comprise a substantial component of the benthic macroinvertebrate community. Their overall density increased markedly in the north basin between 1969 and 2002 (Figure 8.10). Seasonal variation in 2002 was present but patchy with highest densities in regions of the north basin, generally near river mouths.

The highest densities of benthic macroinvertebrates were found in fall 2002 in the western portions of the north basin, especially near the mouth of the Saskatchewan River.

Aquatic worms (Class Oligochaeta) have shown a most remarkable increase in density between 1969 and 2002, again primarily in the north basin (Figure 8.11). The seasonal pattern reflects that seen for the midges, with higher densities in spring and fall. The pattern, however, is not attributable to life cycle features but rather may be more indicative of real changes in population sizes.

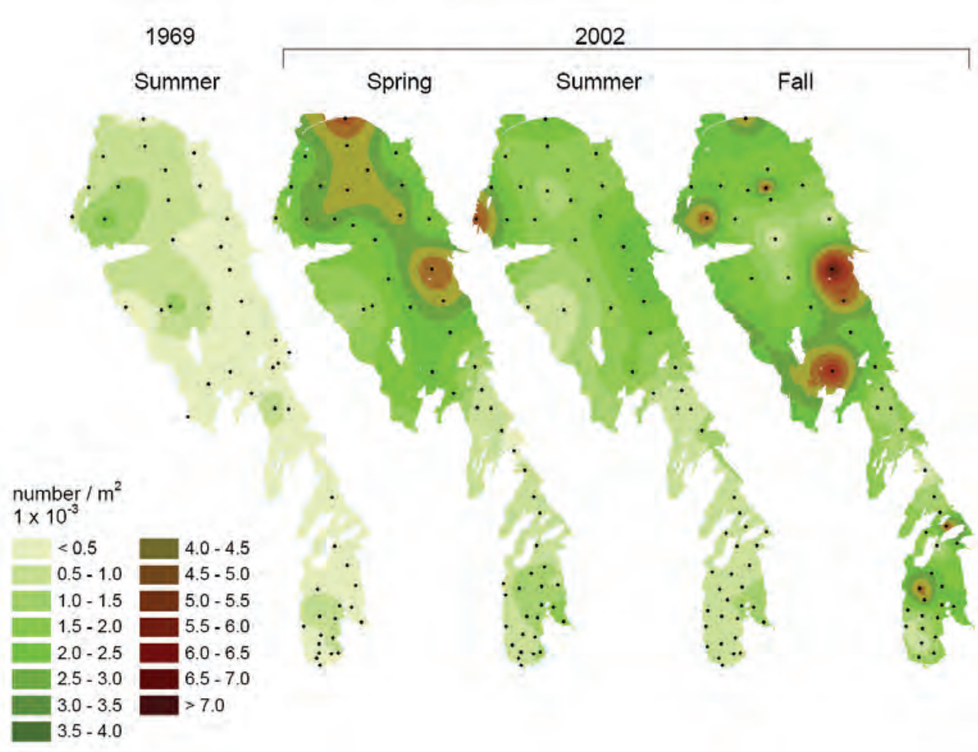


Figure 8.10: *Density (numbers per m^2) of Chironomidae in Lake Winnipeg. (Data Source: University of Manitoba, B. Hann)*

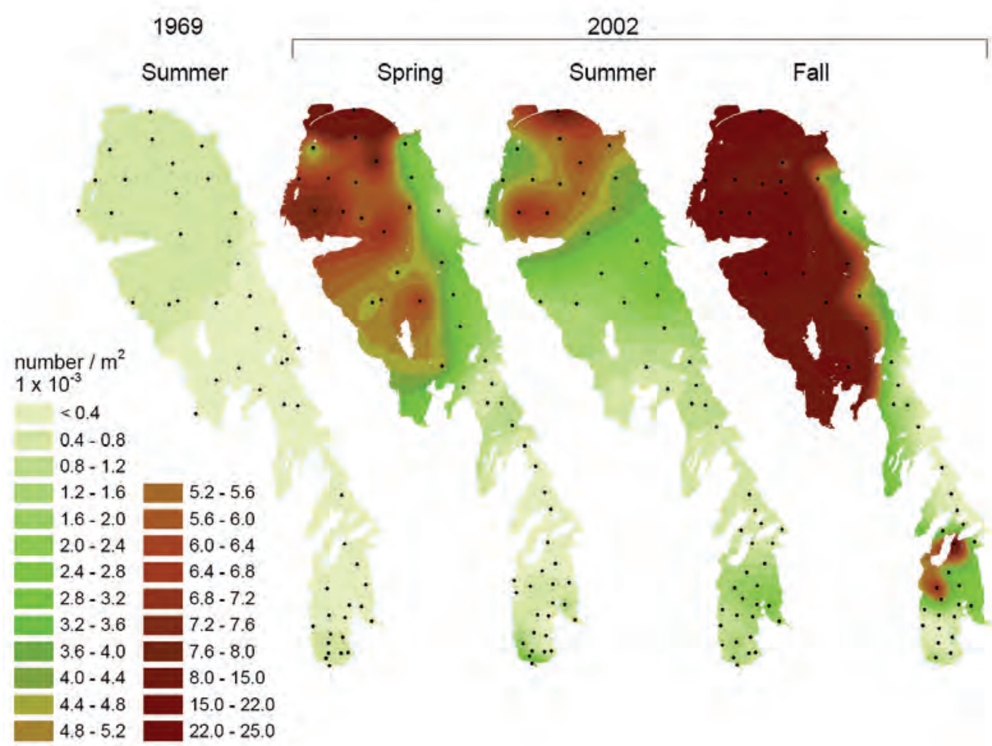


Figure 8.11: *Density (numbers per m^2) of Oligochaeta in Lake Winnipeg. (Data Source: University of Manitoba, B. Hann)*

8.2.2 Zoobenthos and Nutrient Loading

Nutrient loading into Lake Winnipeg has increased substantially over recent years (Section 7.0), primarily from the three major in-flowing rivers (Red, Winnipeg, Saskatchewan). The resulting eutrophication of the lake is demonstrated in the dramatic changes in algal community composition (Section 9.1). The benthic community may respond both directly and indirectly to the increased availability of food resources as a consequence of this enrichment. In contrast with more typical phytoplankton, most of which is consumed by zooplankton while in the water column, dense cyanobacterial blooms, stimulated by the availability of nutrients and largely inedible by zooplankton, settle to the bottom sediments and may be consumed immediately or after decomposition by deposit-feeding benthic organisms. Populations of bacteria and protists may also be stimulated to increased growth with more available nutrients and serve as supplemental food resources for benthos.

8.2.3 Zoobenthos: 2000 to 2007

(Elaine Page)

From 2000 to 2007, benthos samples were also collected over the open water season by Manitoba Water Stewardship at each of the long-term water quality monitoring stations through the north and south basins and narrows (Appendix 1a). Initially, sampling consisted of 12 stations and was expanded to include two additional stations (W13, W14) in 2007. Benthic invertebrate samples have generally been collected during the *MV Namao*'s spring cruise. However, in some years, samples were collected in the summer and fall (e.g., 2005). The following section only summarizes data that were collected during the spring. Further details on sample collection and analysis are provided in Appendix 1c.

Aquatic worms (Oligochaeta) and round worms (Nematoda) made up more than one-half of the total mean zoobenthic density (2000 to 2007) in Lake Winnipeg with mean densities of 1,008 individuals/m^2 and 821 individuals/m^2, respectively (Table 8.3). Other co-dominant taxa from 2000 to 2007 included midges (Chironomidae), and scuds (Amphipoda: mainly *Diporeia hoyi*). Fingernail clams (Sphaeriidae) and mayflies (Ephemeroptera: mainly *Hexagenia sp.*) accounted for a smaller fraction of the 2000 to 2007 mean benthic density in Lake Winnipeg. Many of the sites sampled were located offshore where soft clay sediments predominate and are more conducive to colonization by aquatic worms, round worms, and midges.

Aquatic worms (Oligochaeta) and round worms (Nematoda) made up more than one-half of the total mean zoobenthic density (2000 to 2007) in Lake Winnipeg.

Annual spring mean benthic density for the whole lake ranged from a low of 2,096 individuals/m^2 in 2000 to a high of 6,503 individuals/m^2 in 2004 (Table 8.3). The highest annual density in 2004 was attributed to increases in the aquatic worms, nematodes, and mayflies. Together these three groups comprised more than 80 % of the benthic density in Lake Winnipeg in 2004. Although sampling methods and number of stations differed between the 1969 Fisheries and Oceans Canada benthic survey (Flannagan *et al*. 1994; refer to Appendix 1c) and the most recent survey by Manitoba Water Stewardship (2000 to 2007), benthic densities in 1969 fell within the range of densities recorded between 2000 and 2007.

Table 8.3: *Summary of spring (May and June) mean whole lake benthic invertebrate densities (number/m^2) collected from Lake Winnipeg, 2000 to 2007. (Data Source: Manitoba Water Stewardship)*

Taxanomic Group		2000	2001	2002	2003	2004	2005*	2006	2007	2000-2007 Mean
Annelida										
	Oligochaeta	538	1132	977	659	2,540	-	609	603	1,008
Nematoda		201	667	321	875	1,746	-	1,081	854	821
Diptera										
	Chironomidae	547	474	529	933	1,084	-	891	659	731
Ephemeroptera		197	59	23	48	94	-	26	95	77
Amphipoda										
	Haustoriidae	351	512	0	929	747	-	1,021	466	575
Mollusca										
	Sphaeriidae	262	189	265	390	293	-	389	313	300
Mean Total Density		**2,096**	**3.033**	**2,116**	**3,834**	**6,503**	**-**	**4,017**	**2,990**	

** Benthos samples were collected in the fall of 2005*

Diporeia hoyi is a burrowing amphipod which is an important energy-rich food source for Lake Whitefish, and is an important component of aquatic food webs in lakes. *Diporeia* is typically found in cool, deep waters (Nalepa *et al.* 2003) and is particularly sensitive to dissolved oxygen depletion (Hellawell 1986). Densities of *Diporeia* have declined during low oxygen events in the deepest areas of the north basin of Lake Winnipeg (Hann and Kowalchuck 2004). A decline in *Diporeia* may have implications for Lake Whitefish and the lake food web given their importance as a food source for Lake Whitefish.

Densities of *Diporeia* are generally higher in the narrows and the north basin where cool, deep waters support *Diporeia* growth.

Based on the provincial surveys from 2000 to 2007, the highest *Diporeia* densities were generally found in the narrows region of the lake with an average of 3,089 individuals/m^2 at depths ranging from 9 to 20 m. The 2000 to 2007 average density for the narrows was similar to the average density reported in 1969 (2,457 individuals/m^2; Flannagan and Cobb 1994). Densities were also relatively high in the north basin (2000 to 2007 mean = 1,420 individuals/m^2). However, the relatively high north basin mean density was largely attributed to a single station along the east side of the basin (near Georges Island) where annual densities ranged from 108 individuals/m^2 in 2000 to as high as 11,698 individuals/m^2 in 2006. Although sampling methods differed between Manitoba Water Stewardship and Fisheries and Oceans Canada (Appendix 1a), densities of *Diporeia* during the 1969 survey of the lake were also highest in the vicinity of Georges Island (Flannagan and Cobb 1994). *Diporeia* densities remained low at all other stations sampled in the north basin between 2000 and 2007, never exceeding 60 individuals/m^2. In the south basin, *Diporeia* was only found at a single station in Traverse Bay in 2000, 2003, and 2006. Densities at this station ranged from 6/m^2 to 536 individuals/m^2 with an average of 228 individuals/m^2 for the south basin. Densities of *Diporeia* are generally higher in the narrows and the north basin where cool, deep waters support *Diporeia* growth.

8.3 Fish

(Chelsey Lumb, William Franzin, Doug Watkinson)

Fish communities are influenced by energy inputs, nutrients, habitat variables, and interactions among species (Evans *et al.* 1987). Long-term fish community monitoring can provide insights into effects of changes in lake trophic state, establishment of non-native species, and climate change on dynamics of fish populations and fisheries (Casselman *et al.* 1999; Text Box 8). Limnological variables that describe climate, nutrients, lower trophic levels, and lake morphometry have been used to predict potential fish production (reviewed by Leach *et al.* 1987). For example, total phosphorous concentrations and the ratio of benthic invertebrate biomass to mean lake depth were demonstrated to have significant positive associations with fish yield and fish biomass (Hanson and Leggett 1982). While increased system productivity has been linked with increased total fish biomass, increased productivity has also been linked with changes in fish community composition. Fish species dominance was suggested to shift along a gradient of eutrophication from salmonids (for example, Lake Trout) to coregonids (Cisco and Lake Whitefish), from coregonids to percids (Yellow Perch, Sauger, and Walleye), and eventually to osmerids (Rainbow Smelt), centrarchids (Black Crappie), and cyprinids (minnows such as different species of Shiner) (Colby *et al.* 1972). This type of shift was documented in European lakes, where proportions of coregonids in commercial catches decreased with increasing lake trophic status, and proportions of cyprinids increased (Jeppesen *et al.* 2000; Eckmann and Rösch 1998).

Changes in fish community composition are related to harvest and to changes in environmental conditions that favour certain species and act unfavourably on others, depending on species tolerance to eutrophy (hypoxia, turbidity) (Ludsin *et al*. 2001; Colby *et al.* 1972). Hypoxic conditions can negatively affect abundance of some types of benthic invertebrates (Hellawell 1986), which in turn can have negative impacts on benthic-feeding fish species, such as Lake Whitefish (Ludsin *et al.* 2001). Initial favourable response of percids to nutrient enrichment has been linked to increased phytoplankton and zooplankton production in nearshore areas, important nursery habitat for juvenile percids (Leach *et al.* 1977). Growth rate of young and adult percids increased in a number of nutrient-enriched lakes to some threshold level, and then decreased (Leach *et al.* 1977). Loss of spawning habitat associated with deposition of organic matter has been implicated in percid declines in eutrophic lakes (Leach *et al.* 1977). Changes in light and temperature, important abiotic environmental variables that influence feeding and reproduction in Walleye (Ryder 1977), have been shown to affect the amount of habitat area available (Lester *et al.* 2004). Habitat area was used to predict Walleye yields from inland lake fisheries in Ontario (Lester *et al.* 2004). Highly turbid habitats appear to favour Sauger over Walleye (Stewart and Watkinson 2004; Scott and Crossman 1973). Relative abundance of Sauger in Lake Winnipeg has declined in areas where there were suspected declines in turbidity (Johnston *et al.* 2010). While nutrient enrichment may initially lead to a favourable response in percids, somewhere along the continuum of trophic conditions the responses become unfavourable (Leach *et al.* 1977).

Highly turbid habitats appear to favour Sauger over Walleye, which may partly explain declined relative abundance of Sauger in areas of Lake Winnipeg where there were suspected declines in turbidity.

Over the last 30 years, Lake Winnipeg has been subject to increased nutrient concentrations in rivers (Jones and Armstrong 2001; Section 7.0). Total phosphorous and total nitrogen concentrations have increased in the lake in the last century (Hesslein *et al.* 2007). Indications of nutrient enrichment include larger and more frequent algal blooms during the last ten years (McCullough *et al.* 2004; Section 9.1), changes in phytoplankton species composition consistent with anthropogenic eutrophication (Kling 1998; Section 9.1), and increased abundance of zooplankton (Patalas and Salki 1992). Increased nutrient loading leads to increased production of phytoplankton and zooplankton, and increases oxygen demand of lake bottom waters during the decomposition of organic matter. Reduction in dissolved oxygen concentrations in bottom waters was observed in areas of the north basin of Lake Winnipeg during the summers of 2003, 2006, and 2007 (Stainton and McCullough 2004; Section 5.2). Following excessive phosphorous loading in Lake Erie, anoxic conditions in bottom waters eliminated important benthic macroinvertebrate prey species and reduced thermal habitat for cold water fishes in some areas of the lake (Ludsin *et al.* 2001). Similar changes could be observed in Lake Winnipeg, if conditions of depressed oxygen levels become more frequent.

Lake Winnipeg has long supported important domestic, recreational, and commercial fisheries.

Lake Winnipeg has long supported important domestic, recreational, and commercial fisheries. The Lake Winnipeg commercial fishery began primarily as a gill net fishery for Lake Whitefish in the late 1800s, although other species were harvested, including Lake Sturgeon and Northern Pike (*Esox lucius;* Heuring 1993). By the end of the 1920s, Lake Sturgeon harvest declined to minimal levels due to overfishing (Franzin *et al.* 2003). Lake Whitefish harvest decreased in the 1930s and percids (Sauger and Walleye) began to dominate the commercial fishery (Figure 8.12). Harvest of Lake Whitefish, Sauger, and Walleye declined during the 1960s and increased after 1970, when the fishery was closed for about 18 months due to mercury contamination in Northern Pike, Yellow Perch, Sauger, Walleye, and Freshwater Drum (*Aplodinotus grunniens;* Keleher 1970). The Sauger harvest has declined from the mid-1980s to the present. The Walleye harvest has fluctuated

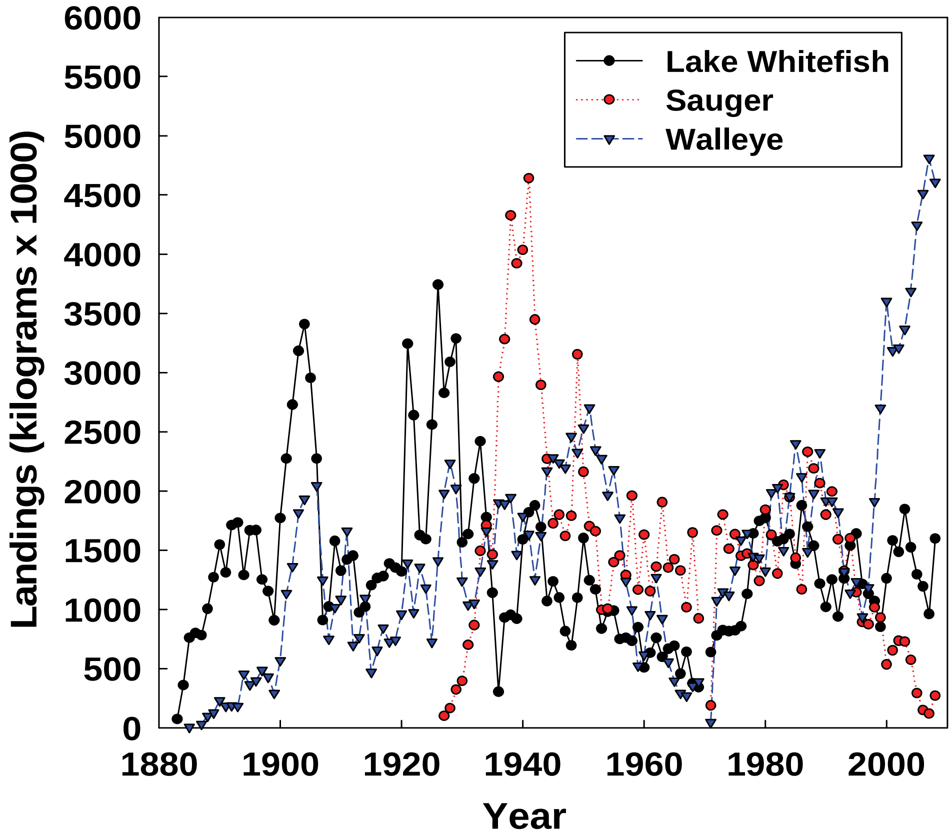


Figure 8.12: *Annual landings of Lake Winnipeg commercial fishery by species (kg x 1000). Sauger (Sander canadensis) was not recognized as a species separate from Walleye (Sander vitreus) before 1928. The fishery was closed in 1970 due to high concentrations of mercury in Sauger and Walleye.*

over the decades, increasing until the mid-1980s, then declining until the mid-1990s. Since then it has increased, reaching its highest level in 2007, when the harvest was nearly double any previous peak (Figure 8.12). Strong positive relationships between Walleye abundance and total phosphorous concentration, chlorophyll a concentration, and zooplankton density have been observed in large, shallow lakes in central Ontario (Robillard and Fox 2006) and may help to explain increased Walleye harvest from Lake Winnipeg.

In 2004, a workshop was held to identify science and research needs for Lake Winnipeg. Top priorities identified for the support of managing Lake Winnipeg's aquatic resources included the development of fish community sampling programs (comprised of multi-mesh index gill net surveys, offshore trawl surveys), a small inshore program (for example, electro-fishing), and spawning stock surveys (Ayles and Rosenberg 2005). Multi-mesh index gill net surveys of the lake are currently conducted annually by Manitoba Water Stewardship, Fisheries Branch. In some jurisdictions, such as the Laurentian Great Lakes, data collected from fish community sampling programs are used to describe the status of a suite of indicators that have been chosen to objectively represent the condition of components of the ecosystem (State of the Lakes Ecosystem Conference 1999). Within the category of indicators called "Biotic Communities," one of the fish indicators is prey fish populations. Prey fish populations are assessed to monitor abundance and diversity of fishes, and to infer the stability of predator species necessary to maintain the biological integrity of each lake (Environment Canada and the U.S. Environmental Protection Agency 2009).

Strong positive relationships between Walleye abundance and total phosphorous concentration, chlorophyll a concentration, and, zooplankton density have been observed in large, shallow lakes and may help to explain increased Walleye harvest from Lake Winnipeg.

To describe seasonal distribution and abundance of small fishes in the offshore waters of Lake Winnipeg, including prey fish species and juvenile life stages of large-bodied fish species, mid-water trawl tows were conducted from 2002 to 2008 near 65 long-term monitoring stations around the lake. Trawl samples were collected during spring, summer, and fall cruises of the Lake Winnipeg Research Consortium's *MV Namao* in all years except 2005, when samples were only collected during the fall (Appendix 1c). Data from these surveys were used to describe inter-annual, seasonal, and spatial variability of biomass estimates of small fishes within the lake.

Since 2002, 25 species were caught in trawl tows. Emerald Shiner, Rainbow Smelt, and Cisco were the dominant species by mass in all years of the survey (Figure 8.13, on the page following). Biomass of Emerald Shiner was greater in the south basin and the narrows compared to the north basin (Figures 8.13a-c). Biomass of Rainbow Smelt was greater in the north basin of the lake compared to the south basin and the narrows in all seasons (Figures 8.13d-f). Cisco biomass was generally greater in the south basin and the narrows compared to the north basin (Figures 8.13g-i). Biomass of Cisco and Walleye (Figures 8.13j-l) were greater in the summer and the fall when young-of-the-year fish were caught in trawls, than in the spring catches when young-of-the-year fish were absent. Biomass of Yellow Perch (Figures 8.13m-o) and White Bass (Figures 8.13p-r) were greatest during the summer in the south basin. In the summer and the fall, Walleye biomass was greater in the south basin and the narrows, compared to the north basin.

Non-native Rainbow Smelt, first observed in Lake Winnipeg in the early 1990s (Campbell *et al.* 1991), are now an important part of the offshore prey fish community in the north basin of the lake. Thermal tolerance of adult Rainbow Smelt may be exceeded during the summer in the south basin, which may partly explain the species distribution mainly in the cooler north basin of the lake (Franzin *et al.*

2005). Potential impacts of climate change, including increased growing season length (water temperatures above 10.0ºC; Sections 5.1.2, 9.5), and warmer open water temperatures, may be detrimental to some coldwater species, but may benefit other species, possibly by improving over-winter survival of young fish (Franzin *et al*. 2005).

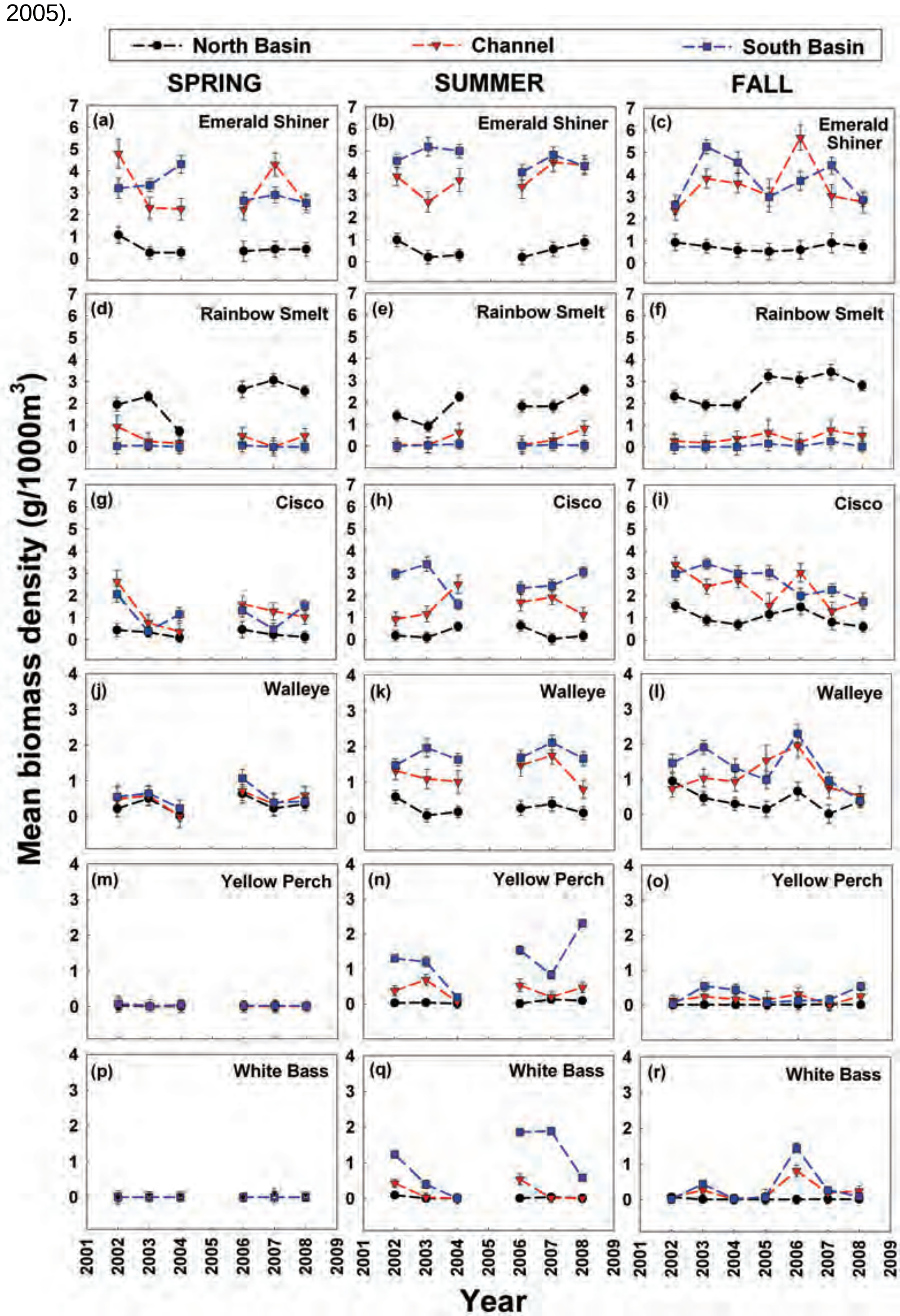


Figure 8.13: *Mean biomass density (g/1000m³) time series of log (x + 1) transformed trawl data of the most commonly captured species. Note differences in y-axis among rows of panels. In 2005, only a fall cruise was conducted. Collection and processing of trawl samples in 2008 were funded through grants from the Fisheries Enhancement Fund.*

Text Box 8. Food Web Isotope Model for Lake Winnipeg

(Keith Hobson, Len Wassenaar, Amy Ofukany)

The measurement of naturally-occurring stable isotopes of carbon (C), nitrogen (N), oxygen (O), sulphur (S), and hydrogen (H) in biota can provide a means of tracing the origins and fate of nutrients and contaminants to higher trophic levels. For several groups of consumers, this analytical approach can also provide an estimate of trophic position and the rate of bioaccumulation or depuration of contaminants. Baseline isotopic studies on Lake Winnipeg by Wassenaar and colleagues have provided inorganic and primary productivity isotopic basemaps or isoscapes for the lake. Isoscapes provide baseline isotopic structure that varies spatially in the lake and which can be used to readily define sources of water and attendant nutrient inputs to the Lake Winnipeg food web. This project included a series of isotopic measurements on fish (2007 to 2009) and more recently (2009) on piscivorous Double-crested Cormorants (*Phalacrocorax auritus*).

Fish isotope measurements have shown considerable isotopic structure that can be used to evaluate the relative importance of nutrient inputs beyond eutrophication. Lake Winnipeg shows more enriched $\delta^{15}N$ levels in the south basin as a result primarily of agricultural nitrogen inputs (Plate 1 top). The narrows region of the lake shows intermediate isotope values reflecting a mixture of the two basins (Plate 1 bottom). Taking the north basin as an example, fish differed in their isotopic compositions and generally occupied two distinct trophic levels with Walleye the top piscivorous predator as indicated by highest muscle $\delta^{15}N$ values. Extensive mixing of the water column results in less $\delta^{13}C$ discrimination between benthic and pelagic food webs but Yellow Perch and Cisco are the most pelagic compared with more benthic Trout, Perch and Stickleback. The introduced Rainbow Smelt occupied the lower forage fish trophic position that is used by Walleye. Work now continues on applying isotopic measurements of additional elements such as O, S, and H, and on the use of GIS and spatial statistical tools to link the inorganic isoscapes with the biotic food web.

Secondary applications will be to determine movement patterns and stock isolation of fish within the lake by comparing baseline isoscape patterns with fish values once effects of isotopic discrimination have been incorporated. Also, trace metal concentrations in fish, with an emphasis on lead, will be examined to see how food web isotope patterns can inform the transport of these contaminants from fish to piscivorous birds. This ecotoxicological component of the food web isotope work will be conducted as an M.Sc. thesis (A. Ofukany) in the Department of Toxicology, University of Saskatchewan, Canada.

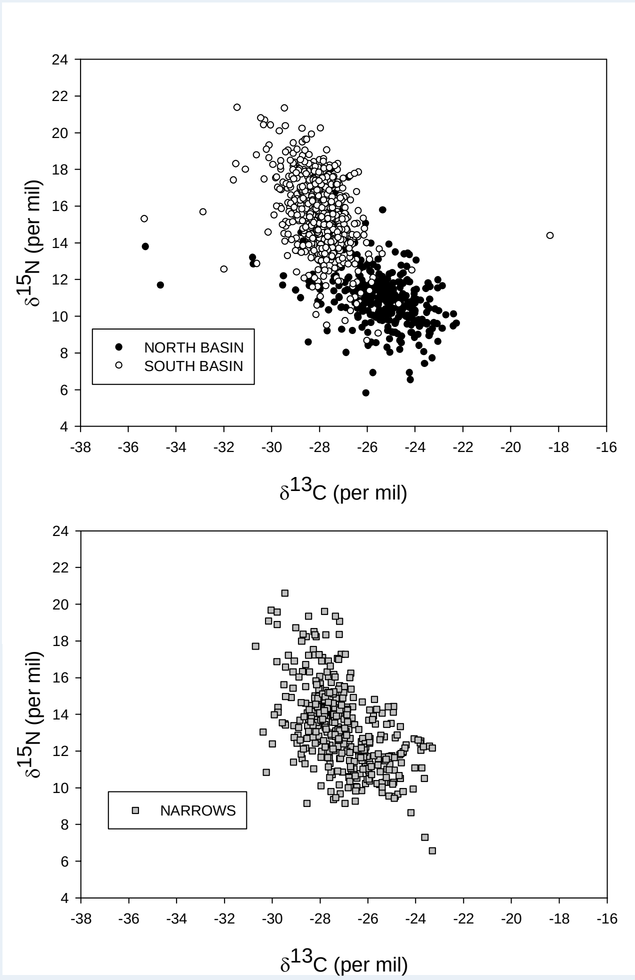


Plate 1. *Stable-carbon and nitrogen isotope results for muscle tissue of all fish sampled in (top) north and south basins and (bottom) narrows of Lake Winnipeg (2002-2008).*

9.0 CURRENT AND EMERGING ISSUES

The health of the Lake Winnipeg ecosystem is not only affected by the loading of nitrogen and phosphorus to the lake, but also by other factors that may stress the system. These stressors, which include algal blooms, the production of toxins by algae, the invasion of aquatic species, and changes in the length of the open water season have the potential for causing changes in the water chemistry and biology of the lake. These in turn may have deleterious effects on biota and humans.

9.1 Bloom Development and Phytoplankton Succession in Lake Winnipeg: A Comparison of Historical Records with Recent (2003 to 2007) Data

(Sue Watson, Hedy Kling, Greg McCullough, Mike Stainton)

Lake Winnipeg has become progressively more eutrophic in recent years, resulting in an increase of algal blooms, which may have detrimental effects on the aquatic ecosystem as well as human use of Lake Winnipeg. Huge surface blooms of cyanobacteria have increased both in frequency and severity in the north and south basins since the mid-1990s (Figure 9.1). Thick wind-blown mats and scum of algae bloom material have also fouled beaches and recreational areas in the south basin of the lake (Section 9.3). In addition, commercial fishers have reported increased densities of diatom algae attached to their fishing nets in the winter and spring fishery. These forms of algae can either clog nets or, because of the increased visibility of the nets to the fish, cause reduced catches. Algae in Lake Winnipeg also have the potential to produce toxins (Kling *et al*. 2007; Section 9.2).

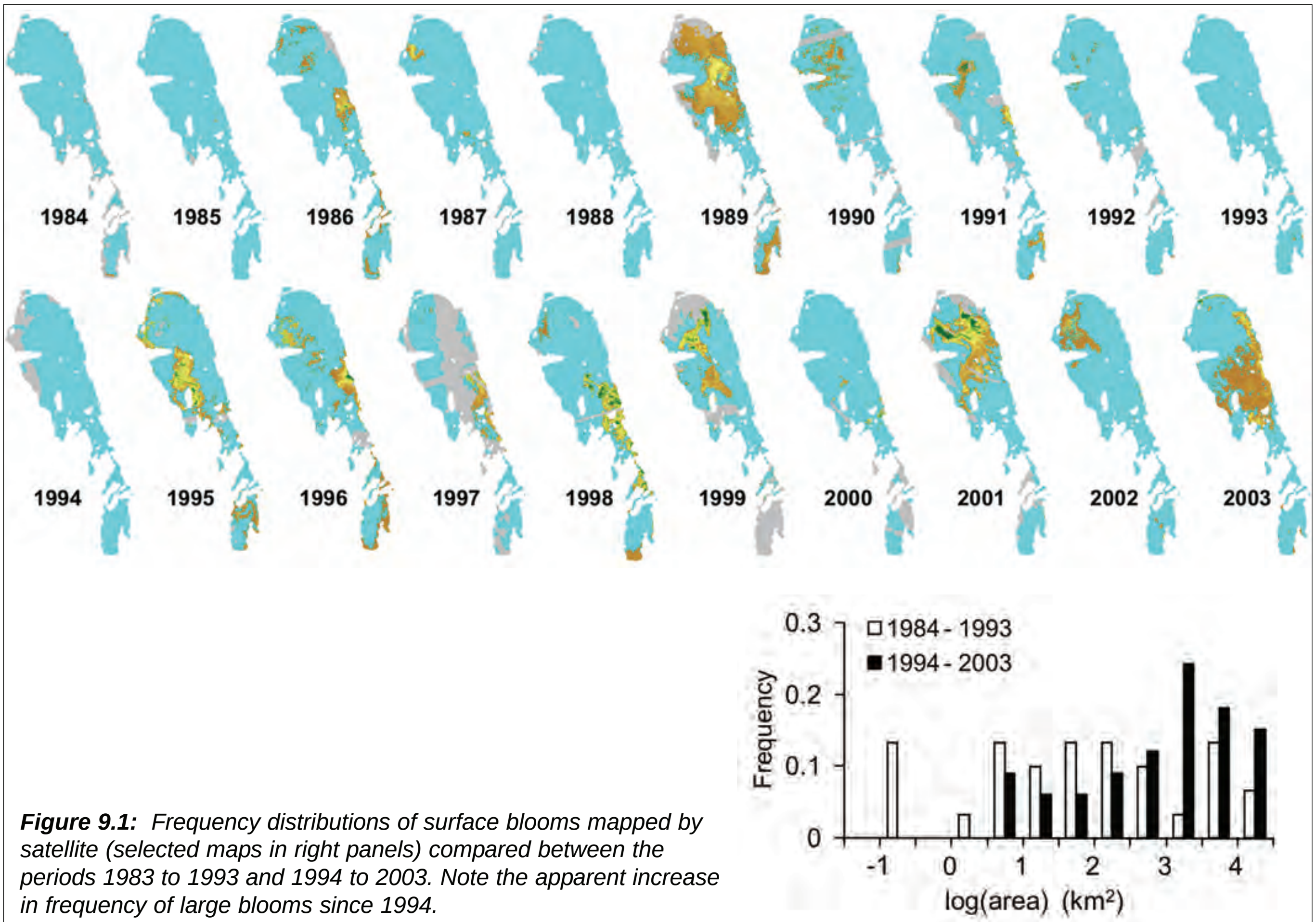


Figure 9.1: *Frequency distributions of surface blooms mapped by satellite (selected maps in right panels) compared between the periods 1983 to 1993 and 1994 to 2003. Note the apparent increase in frequency of large blooms since 1994.*

Few quantitative early data exist with respect to algal blooms in Lake Winnipeg, with the exception of an early assessment in 1969, which recorded a diverse assemblage of over 400 species of algae (Flanagan *et al.* 1994). Nonetheless, anecdotal and sediment records (Kling 1998) suggest that blooms are not limited to the late 20th century, although earlier bloom-forming events were less frequent and often dominated by a different plankton assemblage. Incidental reports from the late 20th century also indicate a potential for periodic bloom development, indicative of a mesotrophic state and moderate levels of productivity and nutrients. A photomicrograph of material from a summer bloom in 1924 (Figure 9.2) shows a predominance of centric diatoms (*Stephanodiscus niagarae*, a few *Aulacoseira sp.*) and some large dinoflagellates (*Ceratium furcoides*), but few cyanobacteria (*Anabaena*). In contrast, in the 1920s and early 1930s, Lowe (1924) recorded a thick scum of *Anabaena flos-aquae*, while Bajkov (1930, 1934) noted cyanobacteria (mainly *Aphanizomenon flos aquae* with some *Anabaena*) blooms 'as thick as carpets' in some areas of the lake.

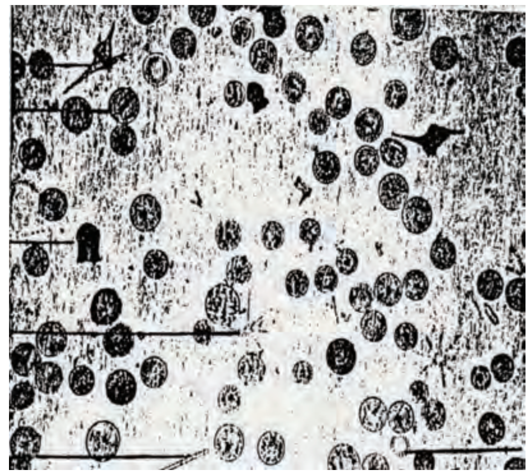


Figure 9.2: *Light microscope image of a bloom sample from Lake Winnipeg taken during summer. (Lowe 1924)*

To address this data gap, phytoplankton were sampled during spring, summer, and fall lake-wide surveys starting in 2002. Surface (one metre) samples were collected at up to 60 stations in the north and south basins of Lake Winnipeg and analyzed for species composition and abundance (see Appendix 1c for further details on sampling and analysis). Dominant taxa were reported, where possible to the species or species complex level (Findlay and Kling 1998). Samples were related to reconstructed Lake Winnipeg paleolimnology from sediment cores. A representative sediment core was analyzed using methods described by Kling (1998). Data were collected at discreet dates and sites with satellite imaging to obtain a better understanding of spatial and temporal bloom dynamics using large-scale remote surveillance mapping. Satellite imaging maps were derived from MERIS (the European Space Agency's Medium Resolution Imaging Spectrometer); chlorophyll biomass was derived from these data as described by McCullough (2007). A synopsis of major changes in the phytoplankton interpreted from these data was developed and early assemblages (prior to 2000) were compared to two recent surveys from 2003 and 2007.

The dated sediment core retrieved in 1994 (Figures 9.3 and 9.4, on the page following) showed a distinct increase in phosphorus, carbon, nitrogen, and chlorophyll a since the late 1950s. This increase corresponds to increased human population and agricultural development in the watershed, hydro-electric development on the Saskatchewan River, and lake level regulation. The sediment core nutrient data were consistent with a Lake Winnipeg nutrient model developed by Fisheries and Oceans Canada (Stainton *et al.* 2008). This model showed increased average phosphorous concentration in the lake, attributed largely to increased discharge from the Red River watershed, with a small peak associated with Red River floods in the 1950s. Phosphorous concentration declined briefly during the drought in the 1960s, and subsequently increased beginning in the early 1970s, with a dramatic rise in the 1990s (Figure 9.5; Section 7.0). During this period there was an increase in water transparency in the north basin (Salki *et al.* 2006), likely due to the construction of reservoirs on the Saskatchewan River which trapped and reduced suspended levels of clay and silt before they reached the lake. The resultant increase in light penetration would have increased phytoplankton use of the phosphorous inputs to this basin, accounting for the increased abundance of non-siliceous and siliceous algal microfossils found in the cores. This increase in

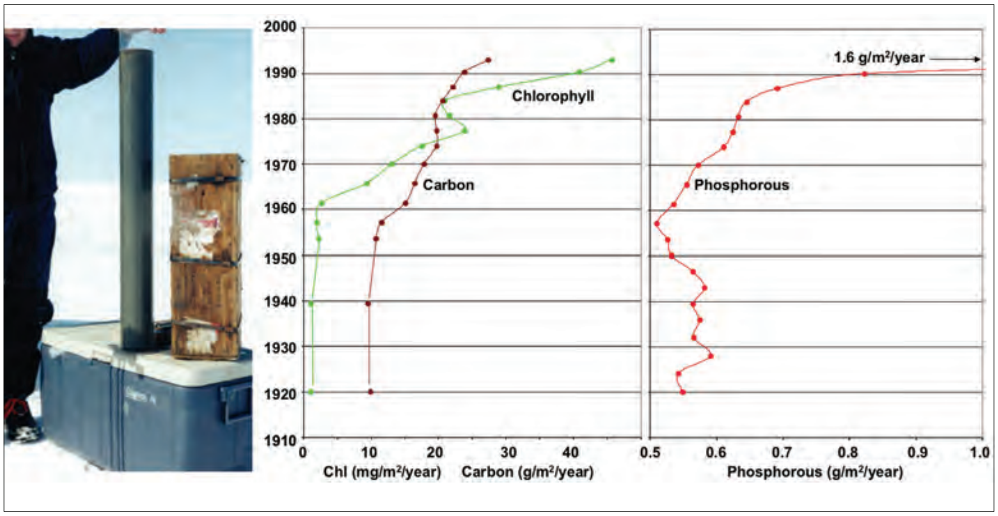


Figure 9.3: *Phosphorus, (P), carbon (C) and chlorophyll measured in 1994 core from the central north basin of Lake Winnipeg. Note increased deposition since the late 1950s, coincident with increased agricultural practices in the watershed, hydro-electric development, and impoundment of the Saskatchewan River since early 1960s and lake level regulation since 1976.*

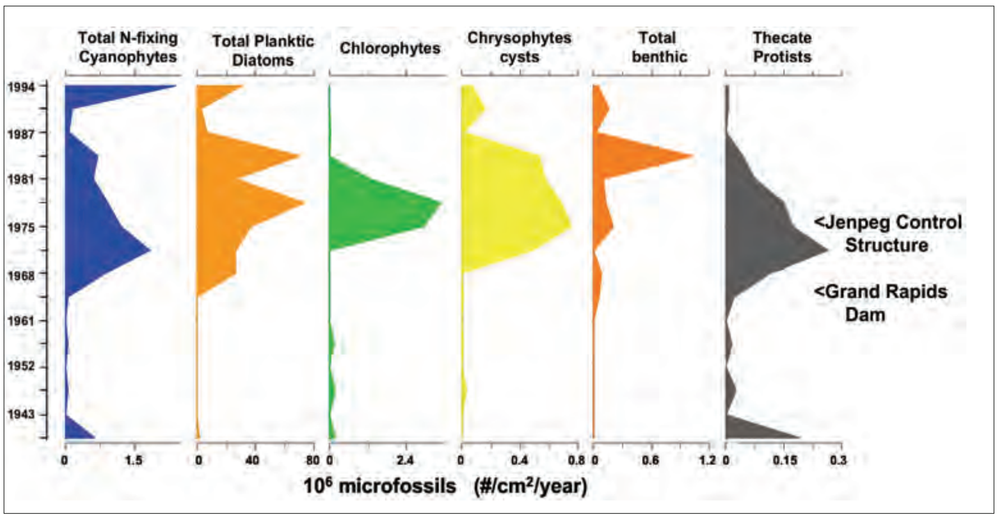


Figure 9.4: *Summary of non-siliceous and siliceous algal microfossils in Nam 2 core deposition rates fluctuate correlating with nutrient increases and hydro-electric construction on Saskatchewan River and control of Lake Winnipeg outlet (Jenpeg). Major taxonomic and functional biotic groups as indicated above.*

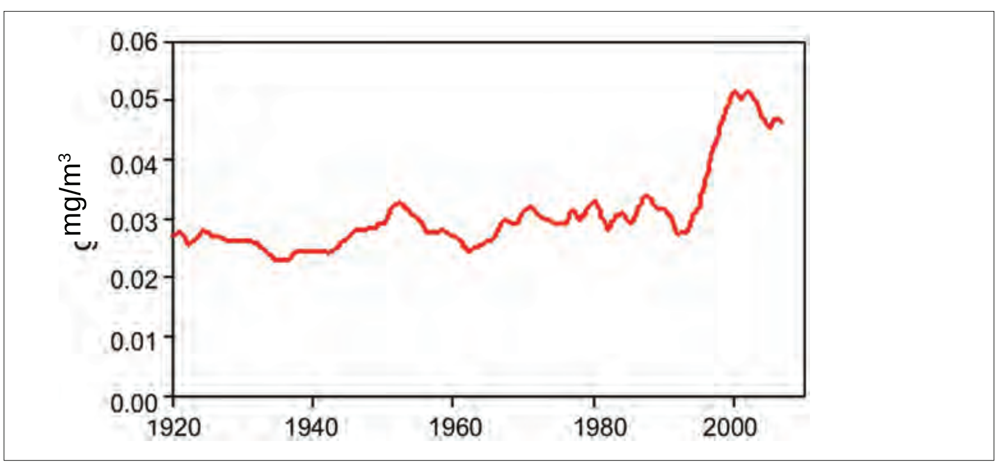


Figure 9.5: *Mean phosphorous concentration in major Lake Winnipeg tributaries. The sharp increase in the mid-1990s is due to precipitation changes, doubling the Red River flows. Phosphorous concentration in the Red River (200 to 300 mg /m³) is nearly an order of magnitude higher than that of other major tributaries (20 to 40 mg /m³). Data are flow-weighted mean concentrations estimated from concentration-discharge-time regressions, smoothed by averaging antecedent 5-yr intervals. (Stainton et al. 2008)*

abundance was correlated strongly with the increasing nutrient concentrations in the core (Figure 9.3) and corresponded with hydro-electric construction on the Saskatchewan River and Lake Winnipeg outlet (Figure 9.4).

Phytoplankton biomass increased over the latter half of the 20th century in association with rising phosphorous concentrations. Seasonal and spatial phytoplankton surveys in 1969 (Figure 9.6) showed a maximum recorded biomass, across all stations and dates, of 10,000 mg/m^3. The basin-wide summer (July to August) average biomass in 1969 was less than 2,000 mg/m^3 and comprised a diversity of taxonomic groups. In marked contrast, since 1999, total biomass has exceeded 40,000 mg/m^3 in several individual samples, and there has been a progressive increase in mean biomass to over 4,000 mg/m^3 by 2007 accompanied by a shift to a predominance of cyanobacteria during the summer blooms (Section 8.1).

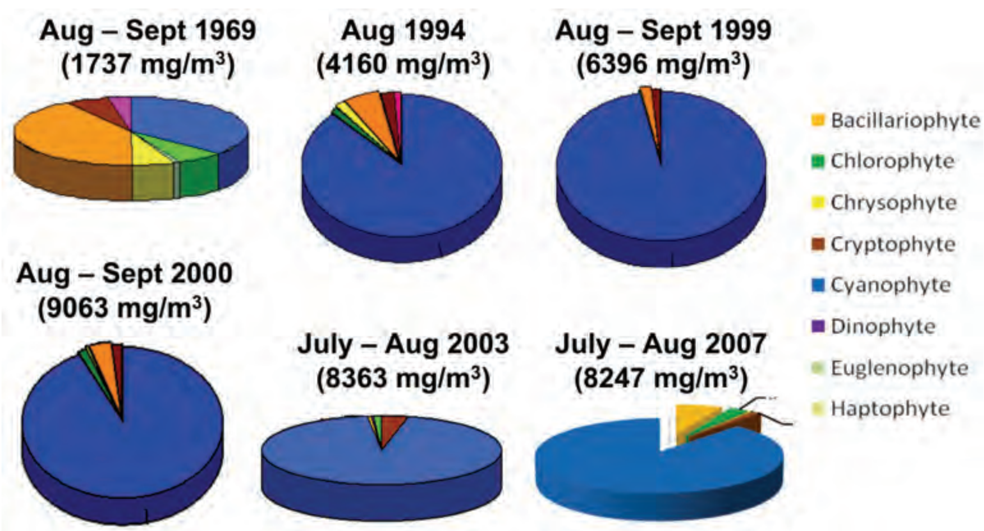


Figure 9.6: *Summary of whole-lake mean phytoplankton composition for sampled years from 1969 to 2007. Numbers in parentheses are mean total biomass for the survey. 2007 data are from a preliminary analysis. (Data Source: Watson et al.)*

Phytoplankton assemblages associated with blooms have been shown to vary between the north and south basins of Lake Winnipeg, as well as seasonally. Data from the 2003 spring survey showed a plankton assemblage that was dominated by chain-forming diatoms (Figure 9.6; *Aulacoseira islandica, A. granulate, A. ambigua, A. subarctica, and S. binderanus*). High summer biomass of cyanobacteria appeared in both basins of the lake, dominated by different taxa. The south basin showed high biomass of non nitrogen-fixing cyanobacteria (*Microcystis; Oscillatoria (syn. Planktothrix suspensa)*) in early July, while a surface bloom of nitrogen-fixing taxa (*Aphanizomenon flos-aquae* complex) was observed extending from the Saskatchewan River to the Berens River (Section 8.1). Depending on the year and climatic conditions, some years had a mixed bloom including *Aphanizomenon flos aquae* complex but dominated by *Anabaena flos aquae* and *A. lemmermannii* in areas of the north and/or south basin in mid- to late July (data not shown). However, usually by fall, the predominant cyanobacteria in the bloom are species and/or morphotypes belonging to the *Aphanizomenon flos aquae* complex. Mixed cyanobacterial assemblages remained through September into November with centric diatoms (chain-forming *Aulacoseira ambigua, A. granulata, A. subarctica* and other single-celled *Stephanodiscus* and *Cyclostephanos* centric species) again dominating in the fall.

Satellite imaging has provided an effective new method of monitoring the development of algal and cyanobacterial blooms across the entire lake, and has demonstrated that vast surface blooms of cyanobacteria (10,000 to 15,000 km^2 in extent) can develop in the north basin. An example is provided in Figures 9.7A and 9.7B, which show satellite reflectance and phytoplankton patterns in Lake Winnipeg on August 15, 2003. Upwelling reflectance is directly correlated with total suspended solids in the upper water column, which, in this image, ranged from <2 mg/L in the darkest areas of the north basin to >30 mg/L in the brightest areas in bays along the southwest side of the narrows. High levels of cholorophyll along the east shore of the north basin corresponded to a surface bloom of cyanobacteria; concentrations were in excess of 50 mg/m^3 (the upper limit of meaningful resolution via this technique). Chlorophyll tended to be highest in regions of relatively low total suspended solids where light penetrates relatively deeply into the water column. This is most apparent from the south basin through to Berens Island near the south end of the north basin. Here, where most of the matter scattering light upwards and out of the water column was silt or clay, chlorophyll tended to be inversely correlated with reflectance (and therefore, with total suspended solids). However, throughout most of the north basin, reflectance tended to be positively correlated with chlorophyll, because most of the reflecting matter was phytoplankton. The satellite images correspond very well with actual biomass estimates from surface samples taken during the lake-wide summer survey in August 2003 (Figure 9.7C).

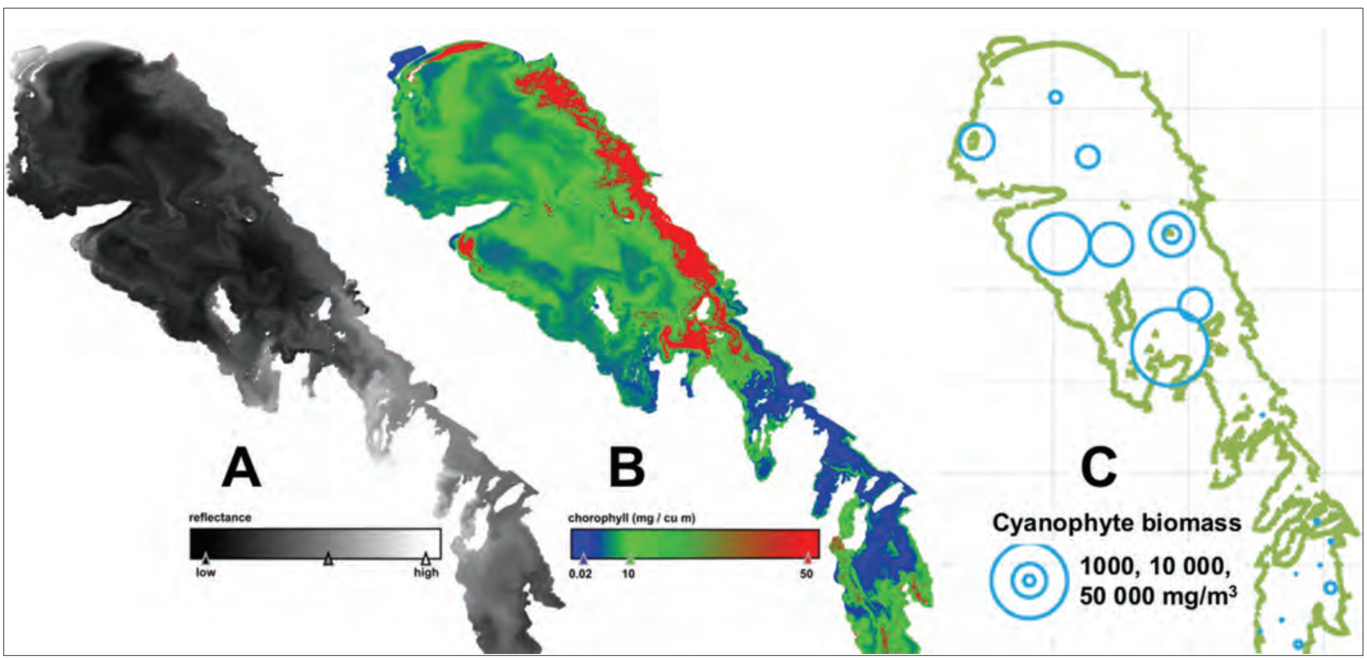


Figure 9.7A, B: *At-satellite reflectance and phytoplankton patterns in Lake Winnipeg, August 15, 2003. Left: satellite image showing upwelling reflectance at 560 nm (MERIS Band 5; MERIS = European Space Agency's Medium Resolution Imaging Spectrometer).* ***C:*** *Estimated contours of total cyanobacterial biomass from microscope counts of 1 m samples collected from summer survey across Lake Winnipeg August 2003.*

More recent in-lake surveys have shown an increasing dominance of potentially toxic cyanobacteria taxa (Figure 9.6; Section 9.2). Toxins (microcystins, and others) were detected in several samples taken from surface blooms between 1999 and 2003, particularly in samples collected near the shore. Microphotographs of plankton in which toxins were detected are shown in Figures 9.8a-c (this page and next). These were taken from bloom material sampled from 1999 to 2003 where the highest biomass recorded was approximately 40,000 mg/m^3 with >200 µg/L of the toxin microcystin in a sample taken off Elk Island in the south basin on September 26, 2001.

Records of bloom development and phytoplankton succession in Lake Winnipeg provide clear evidence of recent increases in the production and frequency of severe algal blooms with shifts towards higher percentage cyanobacteria in last 10 to 15 years (see also Text Box 5). Diatoms, in particular, have been greatly reduced in abundance during the open water season since 1969. Over the last decade, several large surface algal blooms have developed in mid- to late summer, often comprised of in excess of 90 % cyanobacteria. They are responsible for almost all the annual, widespread surface blooms which have formed on Lake Winnipeg in most years since the mid-1990s, concurrent with the increases in phosphorous concentrations in the lake, mostly derived from the Red River basin. Of particular concern, some samples taken in surface blooms contained significant toxin levels with important implications for food webs and human risk (Section 9.2). The combined effects of nutrient increases, algal species shifts, and toxin production represent a significant threat to the sustainability of ecosystem function and productivity.

(Cyanobacteria) are responsible for almost all the annual, widespread surface blooms which have formed on Lake Winnipeg in most years since the mid-1990s, concurrent with the increases in phosphorous concentrations in the lake.

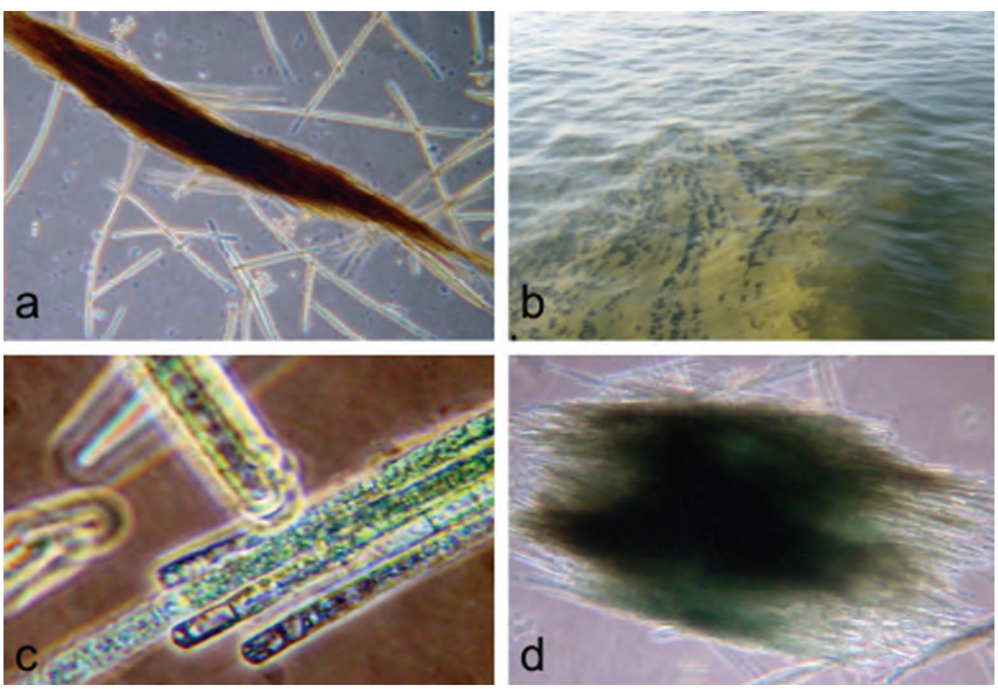


Figure 9.8a: *Morphotypes of Aphanizomenon flos aquae complex: a) A. cf flos aquae, b) surface scum, c) A. cf klebahni, and d) colony morphology (ranges between morphologies described for 4 to 5 morpho-species).*

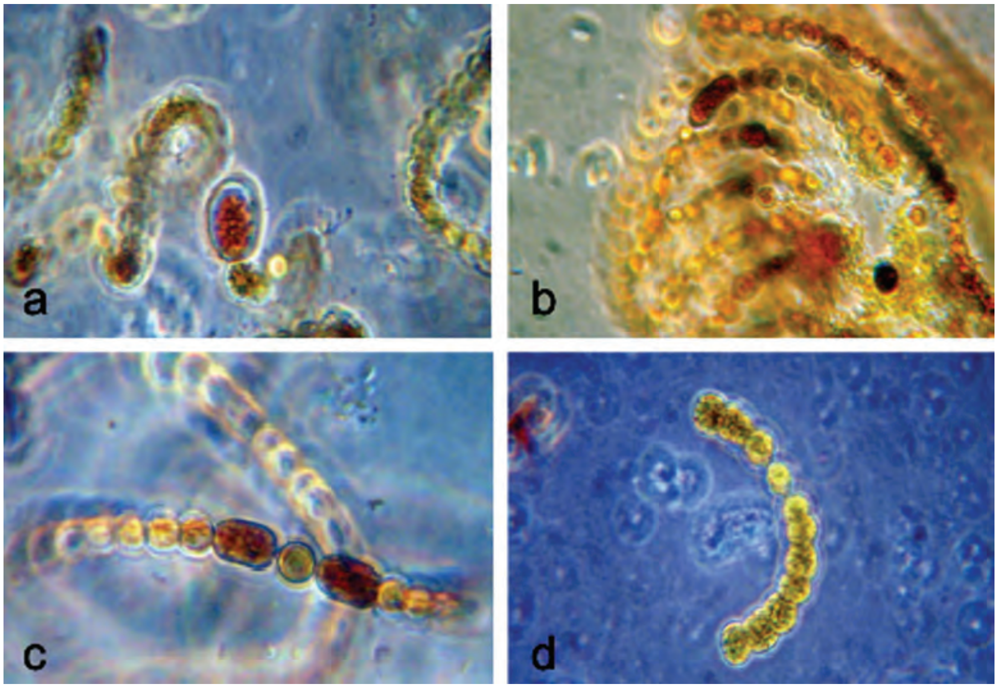


Figure 9.8b: *Morphotypes of Anabaena: a) A.flos aquae /A. spiroides complex, b) A. cf fusca (similar to A. flos aquae but with very tight coiling colony morphology), and c) A. lemmermannii, d) A. crassa /A. circinalis complex.*

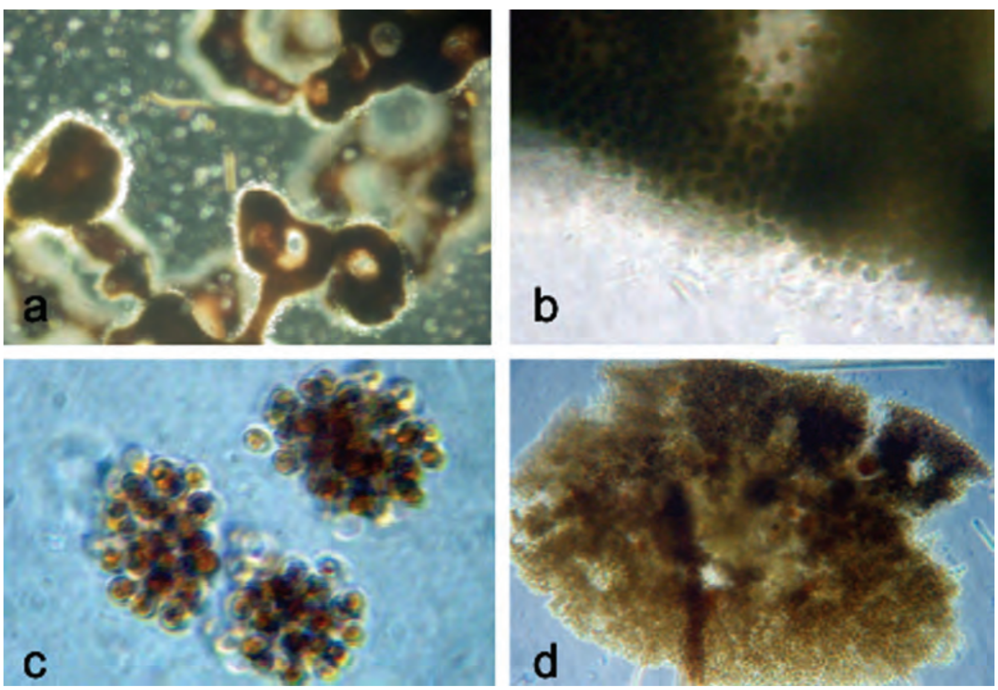


Figure 9.8c: *Top: Microcystis botrys; Bottom left: M. novacekii; Right; M. flos aquae (flattened colony). Note: other species present were M. ichthyoblabe, M. wesenbergii , M. viridis and M. aeruginosa.*

9.2 Cyanobacterial (Blue-green Algal) Toxins in Lake Winnipeg

(Brian Kotak, Sharon Gurney, Claire Herbert, Hedy Kling)

Cyanobacteria, also known as blue-green algae due to sharing some pigment and other characteristics with other groups of algae, can greatly impair water quality in lakes and rivers worldwide. Large accumulations (blooms - Section 9.1) that resemble paint-like slicks, particularly along shorelines, make water unappealing for recreation, can lead to fish kills when the cyanobacteria bloom dies, decomposes, and uses up dissolved oxygen in the water, and can greatly increase the cost of drinking water treatment. In addition, many species of cyanobacteria are also known to produce toxins that are harmful to wildlife and humans (Carmichael 1992).

In freshwater environments, cyanobacteria are known to produce three classes of toxins: neurotoxins (nerve toxins), hepatotoxins (liver toxins), and dermatotoxins (contact irritants). Neurotoxins such as anatoxin-a, anatoxin-a(s), homoanatoxin, and paralytic shellfish poisons affect the nervous system and can result in paralysis and respiratory failure (Kotak and Zurawell 2007). Documented cases of blooms that produce cyanobacterial neurotoxins in Canada appear to be rare. However, this may be due to the lack of analysis of samples for such neurotoxins in freshwater environments as well as the general labile (unstable) nature of the toxins in water. The cyanobacterial hepatotoxin, microcystin, has been much better documented within the Canadian and United States portions of the Lake Winnipeg watershed (Chen *et al.* 2009; Lindon and Heiskary 2009; Kotak and Zurawell 2007; Jones 1998, 1999, 2002) and elsewhere in Canada (for example Giani *et al.* 2005), and worldwide. There are more than 70 closely related structural variations of microcystin, with all of them inducing liver damage and dysfunction. Microcystin-LR is a common type of the microcystin reported worldwide. However, many microcystins can also be produced by cyanobacteria simultaneously, and other microcystin variants can be dominant in phytoplankton communities. This has been observed in Lake Winnipeg (Herbert and Kling, unpublished data). Dermatotoxins and irritant toxins cause irritations of exposed tissues and are the class of toxins that result in skin rashes.

In freshwater environments, cyanobacteria are known to produce three classes of toxins: neurotoxins (nerve toxins), hepatotoxins (liver toxins), and dermatotoxins (contact irritants).

Both freshwater and marine cyanobacteria produce toxins. The primary toxin-producing cyanobacteria genera include *Anabaena, Aphanizomenon* (including *Cuspidothrix*), *Cylindrospermopsis, Microcystis,* and *Planktothrix* (*Oscillatoria*) (reviewed by Zurawell *et al.* 2005). With the exception of *Cylindrospermopsis*, all other genera have been identified in Lake Winnipeg between 1969 and 2007. More than one species within these genera can be toxic and many of the toxic species can form blooms (Backer 2002). The toxicity of a bloom can vary spatially within a lake on a particular day, and temporally (hourly, daily, monthly, and between years) in the same lake (Kotak *et al.* 1995). Although the factors leading to the development of blooms are reasonably well known, the factors leading to the dominance of toxic strains or those that influence toxin production at the cellular level are less clear. The factors that are optimal for growth, such as nutrients and light, may not be the same as those required for optimal toxicity (Kotak *et al.* 2000).

Recently, Guildford *et al.* (2008) observed that nutrient status (in particular, phosphorus) at the cellular level had an influence on microcystin production in Lake Ontario, Lake Erie, and several small reservoirs in southern Ontario. Other aquatic organisms can also have an effect on the phytoplankton community, and by extension, the presence of toxic cyanobacterial species. There is some evidence to suggest that aquatic invasive species such as zebra mussels can alter phytoplankton community composition through selective filter feeding on toxin-producing species of *Microcystis* (Vanderploeg *et al.* 2001; Brittain *et al.* 2000). Zebra mussels consume the toxic cells and selectively isolate and eject them in their pseudofeces, giving the *Microcystis* cells a competitive advantage as they also pick up nutrients from the pseudofeces. Zebra mussels have now been reported in the Red River basin in Minnesota, the first reported occurrence in the Lake Winnipeg watershed (Section 9.4).

The potential for accumulation of certain cyanobacterial toxins such as microcystin in the aquatic food chain has also received some attention. Research in Alberta has shown that filter-feeding aquatic invertebrates, especially clams and snails, can accumulate the cyanobacterial liver toxin, microcystin (Zurawell *et al.* 1999, 2006; Prepas *et al.* 1997; Kotak *et al.* 1996a). Accumulation of microcystins in fish has also been reported in tilapia, catfish, carp, and yellow perch (Wilson *et al.* 2008; Mohamed *et al.* 2003; Zimba *et al.* 2001) among other species. Direct toxicity of microcystins to fish has also been demonstrated in the laboratory (Kotak *et al.* 1996b) and has been linked to mortality of fish in natural environments (Zimba *et al.* 2001).

9.2.1 Exposure

Recreational Contact

There is a potential health risk for children or adults who accidentally ingest or inhale toxic cyanobacteria during recreational activities.

Adverse health effects from exposure to cyanobacteria and their toxins in recreational water is a growing concern. The three routes of exposure include direct contact, accidental swallowing, and inhalation. Direct contact can result in skin irritations, dermatitis, rashes, and eye and ear irritations, while accidental swallowing and inhalation can lead to asthma, hay fever symptoms, diarrhea, vomiting, abdominal pain and, in extreme cases, liver damage. Even at very low concentrations, people with sensitive skin may show allergic reactions. Higher densities of cyanobacteria and a greater length of exposure will often result in an increase in the severity of symptoms. Toxicity related to direct contact during recreation is primarily a concern during the development of a bloom, and immediately after a bloom has developed.

There have been few reported cases of cyanobacterial toxin-related illness due to recreational water activity in Canadian waters (Health Canada 2009). However, there is a potential health risk for children or adults who accidentally ingest or inhale toxic cyanobacteria during recreational activities (Health Canada 2009). High levels of microcystin-LR have been detected along Lake Winnipeg shorelines since the mid-1990s. For example, on September 9, 1996, microcystin-LR concentrations of 300 μg/L were found along Victoria Beach (Jones 1998).

Evidence suggests that potential adverse effects from ingestion are more severe than dermal contact. Inhalation may also be a significant route of exposure in areas where blooms are dense. This uptake route may be relevant for individuals participating in waterskiing, jet-skiing, wake boarding, and other water sports where an aerosol of the bloom is created.

The Federal-Provincial-Territorial Working Group on Recreational Water Quality of the Federal-Provincial-Territorial Committee on Health and the Environment has developed draft guidelines for recreational contact with cyanobacteria and certain toxins (Health Canada 2009).

The recommended guideline values for cyanobacteria and their toxins in recreational waters are total cyanobacteria (100,000 cells/mL) or total microcystins (20 µg/L-expressed as microcystin-LR). Above these guidelines, the water is considered unsuitable for primary-contact recreation, and a swimming advisory should be issued. No recreational guideline levels have been developed for other cyanobacterial toxins. The Province of Manitoba has developed a beach monitoring protocol for Lake Winnipeg and other beaches in the watershed for cyanobacterial blooms and microcystin (Section 9.3). Details of this program can be found on the Manitoba Water Stewardship website (www.gov.mb.ca/waterstewardship/quality/beaches.html).

Exposure through Drinking Water

Exposure to cyanobacterial toxins in drinking water can occur if the water is not adequately treated. Human deaths attributed to cyanobacterial toxins in freshwaters are rare, although there is a growing body of evidence to link long-term exposure of cyanobacterial liver toxins (by consuming inadequately-treated drinking water) to increased incidence of colorectal and liver cancers (Zhou *et al.* 2002; Ueno *et al.* 1996). Drinking raw, untreated water is not recommended, even in remote backcountry areas. In addition, boiling water is not adequate to destroy toxins such as microcystin. Water treatment processes that include activated carbon filtration, ultraviolet, and ozonation are preferred for the removal/destruction of cyanobacterial toxins, provided that the water treatment systems are adequately maintained.

There is a growing body of evidence to link long-term exposure of cyanobacterial liver toxins (by consuming inadequately-treated drinking water) to increased incidence of colorectal and liver cancers.

Several communities on Lake Winnipeg, primarily on the east shore, draw water from the lake and provide treatment for community use. These include Victoria Beach, Seymourville, and several First Nations communities. Pine Dock has a community system supplied with well water which serves most of the community, but a few individual homes may be drawing water directly from the lake. Most homes on Matheson Island have wells, but some use lake water. In some communities such as Loon Straits and Princess Harbour, a community system is not available and individual households use water directly from the lake. It is likely that other individuals living along the lakeshore may use lake water for drinking water, irrigation, or for livestock use (Lake Winnipeg Stewardship Board 2006).

Health Canada works with the provinces and territories to establish drinking water quality guidelines. The maximum acceptable concentration for the cyanobacterial toxin microcystin-LR in drinking water is 0.0015 mg/L (1.5 µg/L). This guideline is designed to protect human health against acute exposure to other microcystins (total microcystins) that may also be present (Federal-Provincial-Territorial Committee on Drinking Water 2002). However, this guideline level does not account for the tumour-promoting activity of microcystins, or synergistic effects between toxins.

Within the Manitoba portion of the Lake Winnipeg watershed, low levels of microcystin-LR have been found in both raw and treated water from municipal water supplies (Jones 1998, 1999, 2002).

9.2.2 Recent Conditions

While many water samples have been collected from 1999 to 2007 on Lake Winnipeg, only a subset of these has been analyzed for microcystins. In addition, very few samples have been analyzed for other cyanobacterial toxins. Three distinct data sets were included in the analyses for this section. The collection and analytical methods varied by member organizations of the Lake Winnipeg Research Consortium (http://www.lakewinnipegresearch.org/) and are detailed in Appendix 8.

Microcystin in Whole Water Samples from Pelagic Sites

Whole water euphotic zone samples at pelagic sites were collected by Manitoba Water Stewardship between 1999 and 2007, and by AlgalTox International in 2006 and 2007. Few differences are evident in the data collected between these two agencies, and therefore, these data have been combined for presentation purposes. In addition, as there was little difference between years, all data have been combined and presented by lake region.

Table 9.1 demonstrates that there is almost no difference in the mean and median concentrations of microcystin among the three regions of Lake Winnipeg. In addition, concentrations are consistently low at pelagic sites. Most samples analyzed did not contain detectable concentrations of microcystin (detection limit of 0.10 µg/L for most analyses). Microcystin concentrations observed are substantially lower than those expected to pose a human health threat from recreational contact, and are also substantially lower than the drinking water guidance level of 1.5 µg/L (Federal-Provincial-Territorial Committee on Drinking Water 2002) for finished drinking water. The highest concentration observed in these two data sets was 2.31 µg/L (July 2007) from a site located in the north basin. At the time, a significant algal bloom was also observed at this site, with a chlorophyll a concentration of 41.9 µg/L. However, this elevated microcystin concentration is still well below that which would be considered unsafe for recreational contact (and the sampling site was located far from any shoreline where recreational contact or drinking water withdrawals would take place). Data from whole water samples therefore indicate that microcystin concentrations are consistently low or below detection limits at pelagic sites in Lake Winnipeg.

Table 9.1: *Summary statistics for microcystin in pelagic (offshore) water samples collected in Lake Winnipeg from 1999 to 2007.*

	Total Microcystin (µg/L MC-LR equivalents)		
	South	**Narrows**	**North**
Minimum	0.05	0.05	0.05
Maximum	0.79	0.22	2.31
Mean	0.08	0.09	0.09
Median	0.05	0.05	0.05
Number of samples	107	60	131

Relatively low microcystin concentrations observed at pelagic sites on Lake Winnipeg may be a result of several factors. First, the whole water samples collected represent integrated water samples, collected from the surface of the lake to the bottom of the euphotic zone, sometimes to a depth of as much as five metres. Second, sampling over such a large depth range may cause substantial dilution of algae within the sample. When surface blooms are present, a euphotic zone sample will cause an underestimation of the phytoplankton community at the surface. Third, if the phytoplankton biomass is also low, this will make detection of the toxin more difficult. Fourth, production of microcystin by the phytoplankton community may be low because those species capable of producing the toxin are producing toxin at a low rate or because the species may be producing other toxins not measured. Given that some samples have been collected during intensive pelagic blooms, where the phytoplankton biomass was high, it is plausible that the phytoplankton community in Lake Winnipeg at pelagic sites simply does not produce large concentrations of the toxin. This hypothesis is supported by additional data discussed on the following pages.

Microcystin in Samples Collected With a Plankton Net from Pelagic Sites

To better understand microcystin production at pelagic sites in Lake Winnipeg, phytoplankton samples have also been collected with a plankton net, and the biomass material dried and analyzed for microcystin. Samples were collected from pelagic sites in this manner in 2003 and 2004 by Algal Taxonomy and Ecology Inc. Microcystin concentrations were then expressed as µg/L based on the results of the protein phosphatase assays (Appendix 8). However, these concentrations do not represent the concentration of microcystin in lake water and therefore, cannot be directly compared to the results described above. An estimate of the concentration of microcystin (µg/L) in the lake water was not possible due to sampling methods (towing the plankton net behind the *MV Namao* resulted in an underestimation of the phytoplankton biomass due to clogging of the net).

Figure 9.9(A-D), presented on the following page, depicts microcystin concentration in Lake Winnipeg over four time periods: July to August 2003, September to October 2003, July to August 2004, and September to October 2004. The figure demonstrates that variations in microcystin concentrations do occur in Lake Winnipeg. Areas of higher concentration tend to be patchy, and these areas can vary seasonally and between years at sites. Note that the concentrations shown in Figure 9.9 represent the concentrations from the assays, and do not represent concentrations in the lake.

An algal bloom viewed from the bow of the MV Namao.

To explore the relationship between microcystin detected in whole water and net samples, both sample types were collected simultaneously from pelagic sampling sites in 2007 by AlgalTox International. Consistent with data collected from other years in Lake Winnipeg, microcystin concentrations in both sample types were low. Only 27 % of the more than 80 whole water samples contained detectable (i.e., >0.10 µg/L) concentrations of microcystin but 81 % of the net samples contained detectable concentrations of microcystin (Table 9.2; Kotak 2009). Upon further analysis of data where both whole water and net samples were collected at the same time (sometimes this was not always possible due to logistical problems), more than half of the net samples contained microcystin while the corresponding whole water sample did not. This likely indicates that the phytoplankton are producing microcystin at such a low level that concentration of the sample with a plankton net is necessary to actually detect the toxin at pelagic sites. This is consistent with the hypothesis that Lake Winnipeg phytoplankton in pelagic areas produce low concentrations of microcystin.

Table 9.2: *Summary statistics for whole water and net samples containing and not containing detectable microcystin (MC) in pelagic (off-shore) samples from Lake Winnipeg in 2007. (from Kotak 2009)*

Category	Number of Samples	%
Whole water samples-detectable MC	23	26.7
Whole water samples - no detectable MC	63	73.2
Net samples - detectable MC	64	81.0
Net samples - no detectable MC	15	19.0

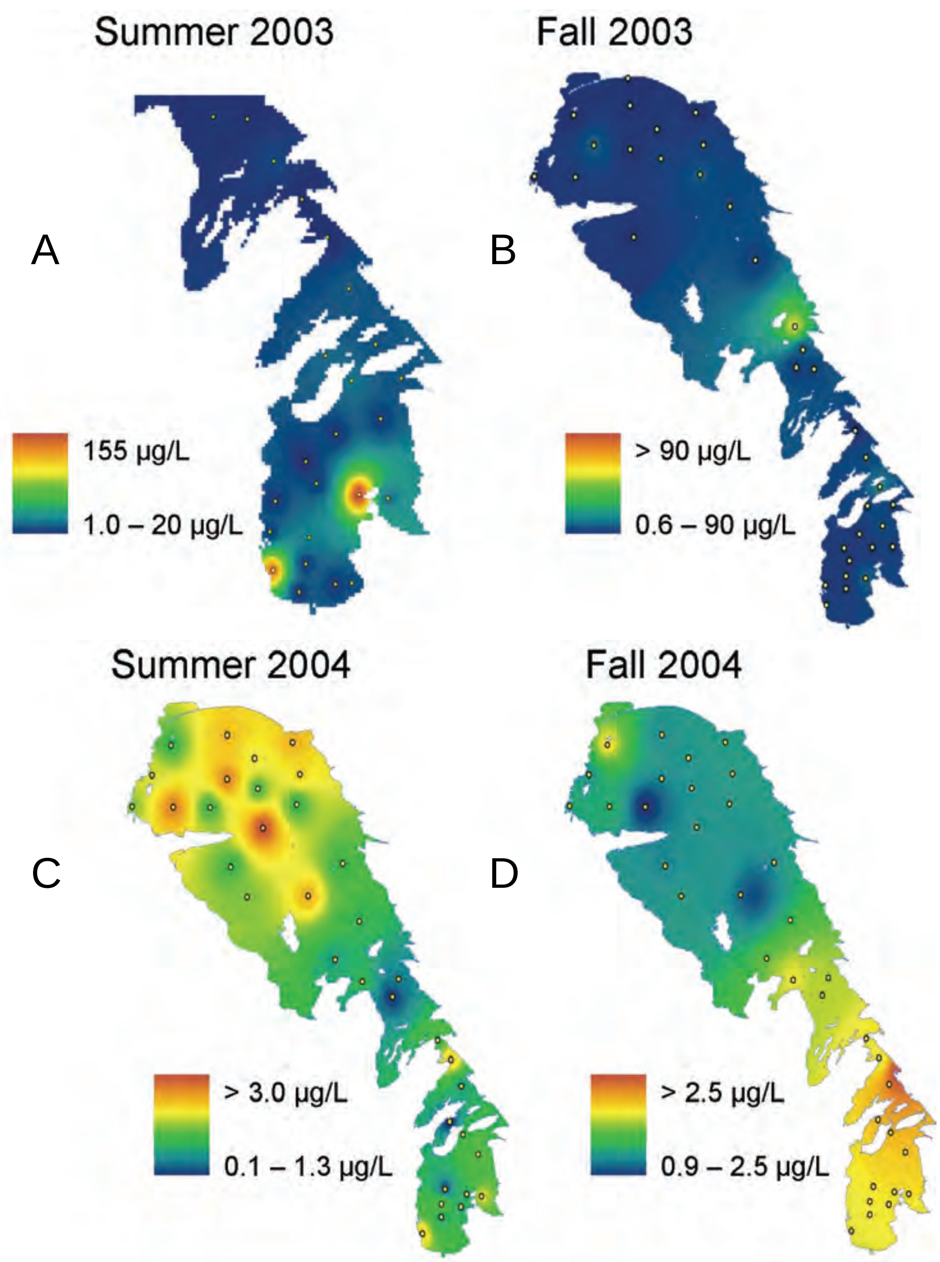


Figure 9.9: *Spatial distribution of microcystin concentration in surface water samples collected from net haul samples from Lake Winnipeg in A) July to August 2003, B) September to October 2003, C) July to August 2004, and D) September to October 2004. Concentrations represent those in the extracted net haul samples, and do not represent the concentration in the lake water (Herbert and Kling, unpublished data).*

Microcystin in Nearshore Environments

Manitoba Water Stewardship collected 269 water samples from beaches in the south basin of Lake Winnipeg from 1999 to 2007. This includes samples that were frozen and fresh (unfrozen) prior to microcystin analysis. As mentioned in Appendix 8, frozen samples tend to have higher microcystin concentrations due to a more complete lysis of cells prior to the microcystin analysis. Table 9.3 provides a summary of the concentrations of microcystin in both frozen and unfrozen samples.

The majority of samples from along beaches contained low (<1.0 µg/L) or non-detectable concentrations of microcystin. Mean and median concentrations of microcystin are far below both the recreational contact guideline values for microcystin, and therefore, observed concentrations would not be expected to represent a risk for water-related recreation.

Table 9.3: Summary statistics of nearshore water samples collected by Manitoba Water Stewardship along beaches in the south basin of Lake Winnipeg from 2000 to 2007.

	Total Microcystin (µg/L MC-LR equivalents)	
	Unfrozen Samples	**Frozen Samples**
Minimum	0.05	0.05
Maximum	331.03	17.00
Mean	1.80	0.66
Median	0.16	0.10
Number of samples	226	43

Of the 269 samples collected from 2000 to 2007, 259 contained microcystin concentrations of less than 1.0 µg/L. Of the 10 samples containing more than 1.0 µg/L of microcystin, nine of these samples ranged from 1.53 to 17.0 µg/L (which are all below draft guidelines for recreational contact of 20 µg/L). However, high concentrations can occur periodically at beaches during intensive blooms, as was observed at Sandy Hook on August 4, 2003. On this sampling date, a microcystin concentration of 331 µg/L was recorded, which represents a potentially serious health risk for recreational users. However, shoreline blooms tend to be patchy in their distribution (that is, one section of beach may have a green slick of algae, whereas another area of the same beach metres away may have none) and such blooms are often transient. The risk to humans through recreational contact is therefore intermittent and unpredictable. To complicate matters further, not all species or strains of cyanobacteria are capable of producing microcystin. A bloom along a shoreline may not contain any microcystin (or conversely, could contain a large concentration of the toxin). It is impossible to determine the risk without collecting and analyzing samples. Therefore, it is generally recommended to avoid contact with water when a heavy bloom of algae is present (that is, when the water is very green) and to prevent pets from drinking or swimming in the water.

In addition to the Manitoba Water Stewardship beach monitoring program, additional shoreline samples have been collected periodically by researchers of the Lake Winnipeg Research Consortium and also by the general public during periods of nearshore blooms. For example, a number of samples were collected at the Grand Beach area and beaches along the west shore of the south basin (Gimli, Winnipeg Beach, Matlock) in August and September of 2006 during a particularly intense algal bloom. Microcystin concentrations were generally below 10 µg/L. However, samples collected from Whytewold Beach on September 16, 2006 contained 39 to 86 µg/L of microcystin, potentially posing a human health risk for recreational contact

A non-concentrated algae sample.

(Kling, Herbert, Boyer, and Kotak, unpublished data). A sample collected during an intensive bloom off the north tip of Elk Island in August 2001 contained in excess of 200 µg/L of microcystin (Kling and Herbert, unpublished data). As mentioned previously, samples from Sandy Hook Beach on Lake Winnipeg in August 2003 had a microcystin concentration >331 µg/L.

From the nearshore and beach samples collected since 2000, it is evident that high concentrations of microcystin can occur occasionally in Lake Winnipeg, and potentially represent a health risk to recreational users. However, this risk appears to be sporadic. Not all algal bloom events along beaches and shorelines on Lake Winnipeg are sampled, so it is not possible to determine with certainty the exact level of risk to those using the lake. Each shoreline bloom should be treated with caution and recreational contact should be avoided until the bloom dissipates.

Microcystin in Other Parts of the Lake Winnipeg Aquatic Food Web

Little research has been conducted on Lake Winnipeg to determine the potential for transfer of microcystin in the aquatic food web. Studies conducted in Alberta and internationally have demonstrated that aquatic invertebrates, including zooplankton and larger macroinvertebrates (e.g., clams, snails), can accumulate microcystin by feeding on toxin-containing phytoplankton species (reviewed in Kotak and Zurawell 2007). In one study, the exposure of snails to microcystin resulted in damage to tissue, consistent with cellular damage effects observed in mammals and fish (Zurawell *et al.* 2007). As accumulation of microcystin in certain aquatic invertebrates is possible, these invertebrates can act as vectors of transfer of the toxin to higher trophic levels (fish, ducks, muskrats, otters, etc.). There have been no systematic studies of the occurrence of microcystin in the Lake Winnipeg food web. However, analysis of several dead ducks collected from Lake Winnipeg, as well as analysis of a limited number of livers from burbot, has detected the presence of microcystin at low levels in Lake Winnipeg and in clams and crayfish in Lake of the Woods portion of the watershed (Kling and Herbert, unpublished data).

Little research has been conducted on Lake Winnipeg to determine the potential for transfer of microcystin in the aquatic food web.

9.2.3 Phytoplankton Species Responsible for Microcystin Production in Lake Winnipeg

A large number of whole water and net samples have been collected and preserved to identify and estimate the biomass of phytoplankton species. These analyses are currently underway. Once complete, this information could help determine which cyanobacterial species are likely producing microcystin in Lake Winnipeg. On a more limited basis, phytoplankton species composition and biomass data were analyzed and combined with microcystin data from samples collected in 2007 (Kotak 2009). A statistical analysis to correlate microcystin concentration to phytoplankton species biomass was attempted, but due to the low range in microcystin concentrations from the samples, the analysis was not ecologically-relevant. However, an examination of those samples containing the highest microcystin concentrations did suggest that there is a limited pool of cyanobacteria that could be producing microcystin in Lake Winnipeg. The dominant species were *Anabaena cf ellipsoidea, A lemmermanii, A mendotae, A flos-aquae, A. spiroides Pseudanabaena sp.*, and *Microcystis flos-aquae*. These are all species that have been known to produce microcystin elsewhere in the world. To determine conclusively which species are producing microcystin in Lake Winnipeg, it would be necessary to culture isolates of these species in the laboratory and perhaps conduct genetic studies to identify the microcystin gene(s).

9.2.4 Cyanobacterial Neurotoxins

There has been no systematic study of cyanobacterial neurotoxins in Lake Winnipeg. While there are several species of cyanobacteria present in the lake (and at times, with significant biomass) that are capable of producing neurotoxins such as anatoxin-a and paralytic shellfish poisons, analyses for these toxins have not been done. This is in part due to the labile nature of the toxins and the difficulty in ensuring proper sample handling and storage in the field. In general, the occurrence of cyanobacterial neurotoxins in Canada appears to be less common than microcystins (Kotak and Zurawell 2007). However, anatoxin-a has been detected in bloom samples collected from Lake of the Woods and Lac Lu (Ontario) and Delta Marsh (Manitoba) at concentrations up to 5.6 µg/L (Kling and Herbert, unpublished data).

Reports of animal poisonings in western Canada from cyanobacterial neurotoxins are rare, but have been reported in Manitoba and elsewhere in the Lake Winnipeg watershed. For example, in July of 2009, a dog died within an hour of drinking lake water from Caliper Lake (south of Nestor Falls, northwestern Ontario). At the time, the shoreline and water contained a thick bloom of cyanobacteria. Symptoms of poisoning were consistent with a neurotoxin and anatoxin-a was later confirmed as the cause of death. In June 1996, 16 cattle and one dog died after drinking water from a creek impoundment near Balder, Manitoba. Microcystin concentrations were found to be low. However, neurotoxin-producing species of algae were dominant (Jones 1998). In July of that same year, three dogs died after drinking water from recreational beaches on Dauphin Lake, Manitoba. Both the symptoms and the algal species composition suggested the anatoxins were responsible for the animal deaths (Jones 1998).

The cyanobacteria suspected of producing microcystin in Lake Winnipeg include species of *Anabaena*, *Pseudanabaena* and *Microcystis,* although further research is advocated.

From the above examples, it is evident that cyanobacterial neurotoxins are produced in the Lake Winnipeg watershed. At present there are no water quality guidelines for neurotoxins. Further research on neurotoxins in Lake Winnipeg and its watershed would be desirable.

In summary, toxin-producing species of cyanobacteria commonly occur in the Lake Winnipeg phytoplankton community. The best-studied toxin, microcystin, also commonly occurs. Concentrations of microcystin in whole water and plankton net samples that have been collected from 1999 to 2007 indicated that concentrations are generally low in pelagic areas of the lake. Occasionally, these concentrations can be elevated in instances of intense offshore blooms. Nearshore samples (e.g., along beaches) demonstrate that high microcystin concentrations can occur occasionally (at times, at concentrations that represent a potential human health risk from recreational contact), but that these are highly sporadic and unpredictable. As it is impossible to determine if a bloom is toxic without sophisticated methods of analysis, people, livestock, and pets should avoid contact or consumption of waters exhibiting cyanobacterial blooms. The cyanobacteria suspected of producing microcystin in Lake Winnipeg include species of *Anabaena*, *Pseudanabaena,* and *Microcystis*, although further research is advocated. The occurrence of microcystins in other parts of the aquatic food web in Lake Winnipeg is probable, although no systematic study has been done. Little is known about other cyanobacterial toxins, including neurotoxins, in Lake Winnipeg.

9.3 Recreational Water Quality at Lake Winnipeg Beaches

(Wendy Ralley)

Lake Winnipeg beaches offer visitors a unique opportunity for numerous recreational activities from swimming and wading, to kite-boarding and windsurfing. The beaches are a strong economic driver for local communities as they attract day-users, campers, cottagers, and long-term residents. Beaches in the south basin of Lake Winnipeg are easily accessible from the populated Red River corridor including the city of Winnipeg and during the summer months, visitors to the beaches can exceed 30,000 per day. Seasonal populations in communities surrounding the south basin swell significantly, particularly those communities with large cottage developments.

It is a provincial responsibility to protect beachgoers from bather-related illnesses. Therefore, Manitoba Water Stewardship routinely monitors densities of the fecal indicator bacteria, *Escherichia coli (E. coli*). While *E. coli* are not generally disease-causing, they indicate the presence of fecal contamination and have characteristics that make them good indicators of pathogens in water bodies. It is not practical to monitor for every pathogen that may be present in surface water due to cost and the lack of reliable laboratory methods. The use of fecal indicator bacteria such as *E. coli* provides regulators and beach managers a mechanism to establish risk to bather health. Studies conducted elsewhere (United States Environmental Protection Agency 1986; Cabelli *et al.* 1983) have linked densities of fecal indicator bacteria with illness rates of bathers in recreational waters. As the density of fecal indicator bacteria increases so does the risk of contracting bather-related illnesses.

Manitoba Water Stewardship routinely monitors densities of the fecal indicator bacteria, *Escherichia coli (E. coli*). While *E. coli* are not generally disease-causing, they indicate the presence of fecal contamination and have characteristics that make them good indicators of pathogens in water bodies.

The pathogens most commonly identified and associated with waterborne diseases can be grouped into three general categories: bacteria, viruses, and protozoa. Although the most common effects of bathing in water contaminated with fecal material are illnesses affecting the gastrointestinal tract, other illnesses and conditions affecting the eye, ear, skin, and upper respiratory tract can be contracted as well. The main route of exposure of bathers to disease-causing organisms is through ingestion of contaminated water. Of the different illnesses that may be contracted during recreational activities, gastrointestinal illness occurs most frequently (The Centers for Disease Control and Prevention 2008). However, gastrointestinal illness is rarely life-threatening and symptoms include mild fever, stomach ache, nausea, vomiting, and diarrhea.

Manitoba has adopted Canadian recreational water quality guidelines that were developed for Health Canada by a committee of federal, provincial, and territorial health officials (Health and Welfare Canada 1992). The Manitoba Water Quality Objective for *E. coli* in recreational water is 200 *E. coli* per 100 mL (Williamson 2002). While national recreational water quality guidelines are currently being updated with the latest scientific knowledge (http://www.hc-sc.gc.ca/ewh-semt/consult/_2009/water_rec-eau/draft-ebauche-eng.php - f1), the recreational water quality guideline for *E. coli* in surface water is expected to remain at 200 *E. coli* per 100 mL.

As with other jurisdictions that assess risk to bathers by measuring fecal bacteria densities, culture techniques can only provide results in 18 to 24 hours, at best. By the time results from the previous sampling day are available, densities of *E. coli* would likely have changed dramatically. This lag in availability of results from when bathing areas are actually sampled is a source of frustration for beach managers and bathers. While other techniques to quantify densities of *E. coli* have been developed such as qPCR, their current reliability and practicality make them a poor choice for sampling Lake Winnipeg beaches.

9.3.1 Lake Winnipeg Beach Monitoring

Currently, the Province of Manitoba routinely monitors 19 bathing areas in the south basin of Lake Winnipeg (Figure 9.10) as part of an overall beach monitoring program. Ten beaches were monitored historically by the Province in the 1980s and 1990s. However, for the purposes of this report, only beach monitoring data collected from 2004 to 2009 are discussed (Figure 9.10, Appendix 9) as this is the most extensive data set.

Beaches on the east side of Lake Winnipeg display significantly different physical characteristics from those on the west side. Long, sandy beaches with a gently sloping depth are characteristic of the east side, while smaller rocky beaches are generally characteristic of the west side (Figure 9.11). Bather density also varies greatly among beaches. Beaches near more developed areas tend to have a higher bather density due to the availability of conveniences and the short driving distance from populated urban areas. A number of beaches are located within Manitoba's Provincial Park system while other beaches remain the responsibility of local municipal governments.

Many of the beaches are relatively isolated and not directly influenced by discharges from large urban development, although many communities have land drainage outfall culverts near or on beaches. Municipal sewage facilities are located near some beaches. However, effluent quality and timing of release is licensed by the Province to protect water-related recreation. In particular, effluent discharge must meet a Manitoba Water Quality Standard of less than 200 *E. coli* per 100 mL and discharge typically occurs outside of the swimming season.

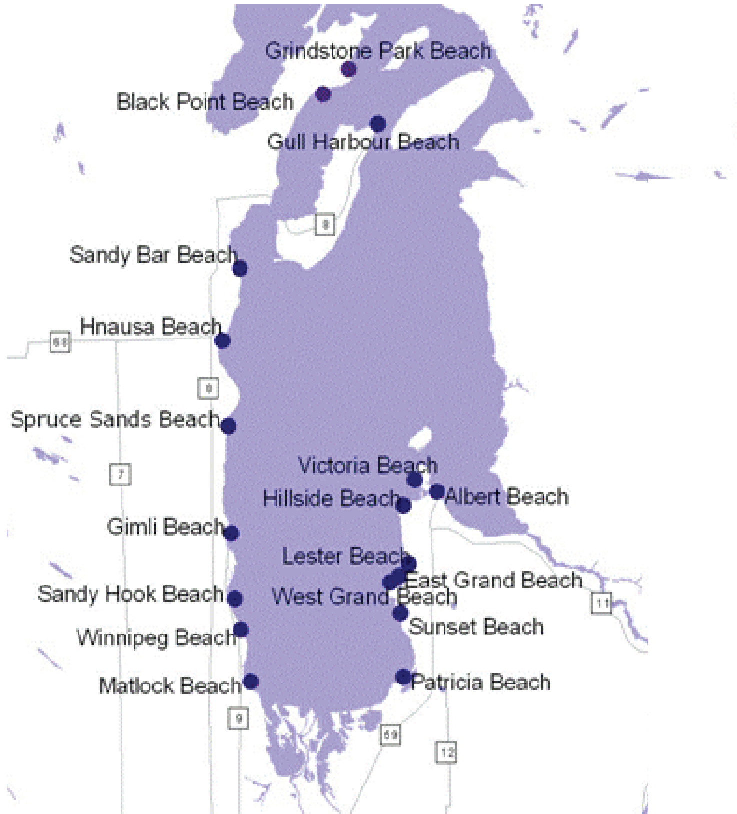


Figure 9.10: *Approximate locations of the bathing areas in the south basin of Lake Winnipeg that are monitored weekly for Fecal Indicator Bacteria (E. coli).*

Figure 9.11: *West Grand Beach (left) has a characteristically gentle sloping shoreline and shallow bathing area. The bathing area at Gimli Beach (right) has a deeper bathing area.*

9.3.2 Escherichia coli at Lake Winnipeg Beaches: 2004 to 2009

From 2004 to 2009, densities of *E. coli* bacteria at Lake Winnipeg beaches occasionally exceeded the Manitoba Water Quality Objective for recreation. However, densities usually returned to within acceptable limits within 24 hours. Generally, densities of *E. coli* were below the objective for the vast majority of samples collected from 2004 to 2009 (Table 9.4). This is consistent with historical data (prior to 2003) collected from Lake Winnipeg beaches (Williamson 1988, 1985; Manitoba Water Stewardship unpublished data). Beaches on the east side of Lake Winnipeg tended to have fewer occurrences of *E. coli* densities over the objective when compared to west side beaches. While it is uncertain why west side beaches tend to have more exceedences than east side beaches, prevailing winds, currents, and beach morphology could play a role in these differences.

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. This was specifically demonstrated by the results of daily sampling from Gimli Beach and West Grand Beach where samples were collected every two hours from 06:00 to 18:00 on two separate occasions in 2004. Generally, higher densities of *E. coli* were found in bathing water in the early morning hours when compared to densities later in the day. Whitman *et al.* (2004) were also able to demonstrate that time of sampling was a significant aspect of diurnal population changes of *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).

Results of extensive sampling in 2003 provided sufficient evidence that *E. coli* in foreshore sand is likely transferred to inshore bathing water when lake levels rise due to strong north winds.

In the open water season of 2003, Lake Winnipeg experienced the lowest lake levels across the period 1999 through 2007. A significant area of foreshore sand was created at most beaches because of the low water level. *Escherichia coli* densities exceeded the recreational water quality objective more frequently in 2003 and significant effort was dedicated to find the source of fecal contamination. Densities of *E. coli* at West Grand Beach were above the guideline for a number of consecutive readings giving the perception that densities were persistently elevated. While no single major source of contamination was discovered, extensive sampling of bathing water, sand pore water (interstitial sand water), and sand was undertaken (Williamson *et al.* 2004). Results of extensive sampling in 2003 provided sufficient evidence that *E. coli* in foreshore sand is likely transferred to inshore bathing water when lake levels rise due to strong north winds. Williamson *et al.* (2004) were able to correlate the increase in lake levels with elevated densities of *E. coli*. However, lake levels and densities of *E. coli* were not strongly correlated in subsequent years. This may be because Lake Winnipeg levels have not been as low since 2003. Low lake levels created a unique situation where exposed areas of foreshore sand were much larger than typically found in 2004 to 2009. When wind-induced lake levels inundated foreshore areas, *E. coli* may have been transferred from the sand to the bathing water, thus causing frequent and high densities of *E. coli* in 2003.

Many authors have demonstrated that densities of *E. coli* occur at levels an order of magnitude higher in sediment than in the overlying water column (Alm *et al.* 2003; Lauber *et al.* 2003; Howell *et al.* 1996; Davies *et al.* 1995). More recently, studies have shown that wet beach sand can also support significantly higher densities of *E. coli* than adjacent bathing water (Edge *et al.* 2010; Ishii *et al.* 2007; Kon *et al*. 2007;

Table 9.4: *The percentage of water samples (geometric means of at least five samples) that were below the recreational water quality objective of 200 E. coli per 100 mL and the number of sampling events (n) from each Lake Winnipeg Beach from 2004 to 2009.*

Location **Beaches on East Side of Lake Winnipeg**	2004	2005	2006	2007	2008	2009
Victoria Clubhouse Beach	95 n = 22	100 n = 21	100 n = 21	100 n = 19	95 n = 18	89 n = 18
Victoria Red Cross Dock	100 n = 17	100 n = 12	100 n = 9	100 n = 8	91 n = 11	100 n = 5
Albert Beach	95 n = 22	90 n = 21	100 n = 21	100 n = 20	100 n = 20	94 n = 17
Hillside Beach	100 n = 21	90 n = 21	100 n = 21	100 n = 20	91 n = 18	100 n = 16
Lester Beach	100 n = 20	100 n = 21	100 n = 21	100 n = 20	91 n = 20	100 n = 16
East Grand Beach	91 n = 23	100 n = 21	95 n = 22	90 n = 21	100 n = 20	89 n = 18
West Grand Beach	97 n = 99	100 n = 92	93 n = 90	99 n = 82	98 n = 80	94 n = 70
Sunset Beach	Not sampled	Not sampled	Not sampled	80 n = 15	95 n = 12	94 n = 17
Patricia Beach	92 n = 25	92 n = 22	91 n = 22	86 n = 22	100 n = 20	89 n = 18
Beaches on West Side of Lake Winnipeg						
Gull Harbour Beach	95 n = 22	83 n = 24	91 n = 23	100 n = 18	100 n = 19	100 n = 16
Grindstone Beach	Not sampled	Not sampled	95 n = 22	100 n = 19	100 n = 18	100 n = 16
Black Point Beach	Not sampled	Not sampled	90 n = 21	100 n = 18	100 n = 18	100 n = 16
Sandy Bar Beach	91 n = 11	83 n = 24	95 n = 22	90 n = 21	91 n = 22	100 n = 16
Spruce Sands Beach	86 n = 22	83 n = 242	88 n = 25	95 n = 20	80 n = 20	100 n = 16
Hnausa Beach	95 n = 21	86 n = 21	90 n = 20	94 n = 18	94 n = 18	88 n = 16
Gimli Beach	81 n = 110	84 n = 94	93 n = 90	85 n = 80	92 n = 79	76 n = 81
Sandy Hook Beach	96 n = 24	91 n = 23	91 n = 23	100 n = 18	90 n = 21	94 n = 17
Winnipeg Beach	70 n = 30	85 n = 27	80 n = 26	100 n = 19	83 n = 23	100 n = 15
Matlock Beach	96 n = 23	88 n = 25	71 n = 24	100 n = 18	95 n = 20	100 n = 16

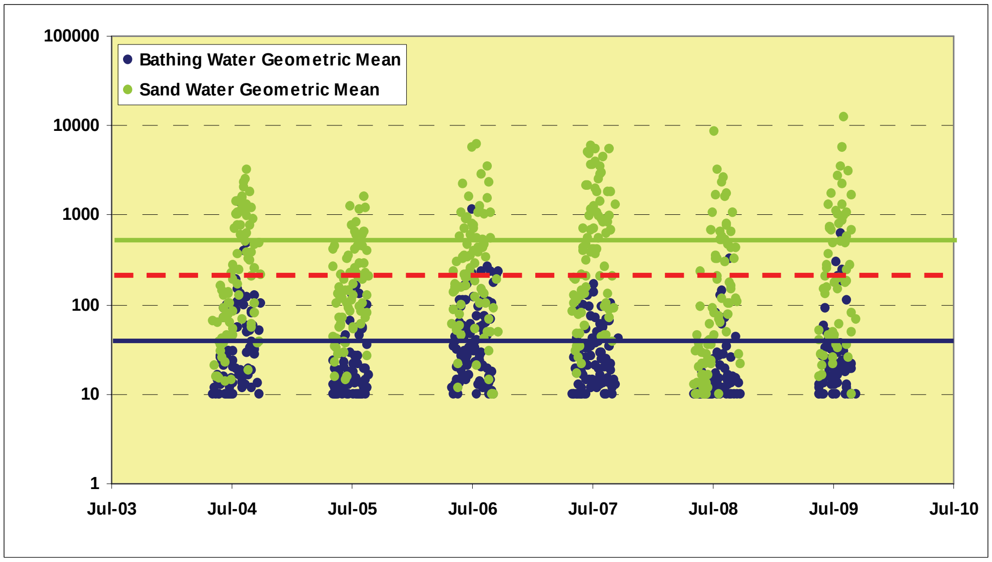


Figure 9.12: *Plot of the geometric means of E. coli densities taken from both bathing water and sand pore water at West Grand Beach during 2004 to 2009. The solid lines represent the 6-year average of geometric means for E. coli in bathing water (mean of 45 E. coli per 100 mL) and sand pore water (mean of 614 E. coli per 100 mL). The dashed line represents the recreational water quality objective of 200 E. coli per 100 mL.*

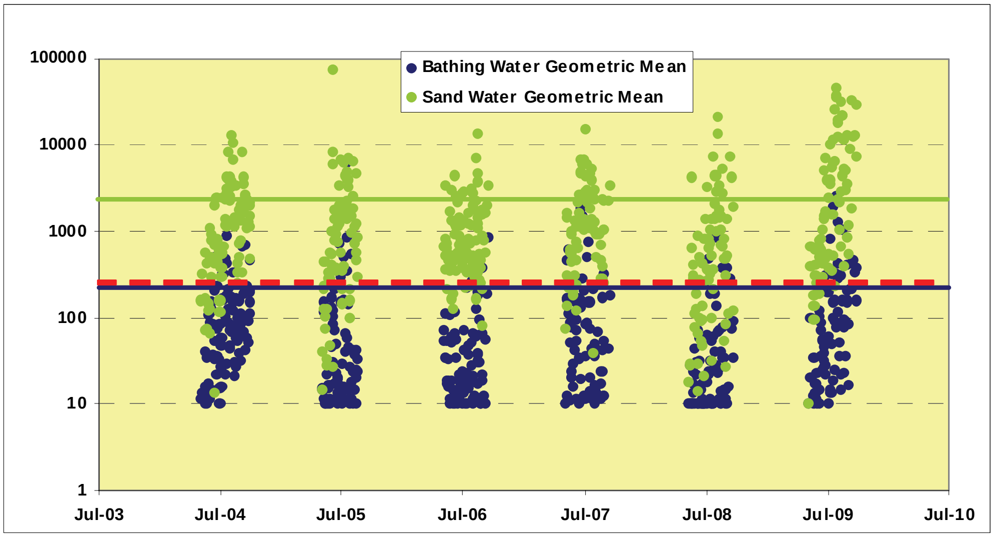


Figure 9.13: *Plot of the geometric means for E. coli densities taken from both bathing water and sand pore water at Gimli Beach during 2004 to 2009. The solid lines represent the 6-year average of geometric means for E. coli in bathing water (mean of 130 E. coli per 100 mL) and sand pore water (mean of 2564 E. coli per 100 mL). The dashed line represents the recreational water quality objective of 200 E. coli per 100 mL.*

Whitman and Nevers 2003). At Gimli and West Grand beaches, significantly higher densities of *E. coli* were found in foreshore wet sand than in bathing water for each of the sampling years of 2004 to 2009 inclusive (Figures 9.12 and 9.13).

Other studies have shown that densities of *E. coli* are higher in inshore bathing areas when compared to deeper depths (McLellan and Salmore 2003; Whitman and Nevers 2003). Whitman and Nevers (2004) demonstrated significant spatial differences in densities of *E. coli* in areas near and far from the shoreline. Manitoba Water Stewardship also found that densities of *E. coli* decreased with increased water depth. Densities of *E. coli* in bathing water at Lake Winnipeg beaches are an inshore phenomenon that appears to be influenced by interactions of bathing water with the foreshore beach sand.

In addition to monitoring densities of *E. coli*, efforts were aimed at establishing the original source of *E. coli* contamination. Ribotyping *E. coli* is a technique of microbial source tracking where the DNA of *E. coli* from known sources is compared to the DNA of *E. coli* found in the environment. From 2004 to 2007, weekly samples of sand, pore water, and bathing water containing *E. coli* were submitted for DNA ribotyping along with fecal material from likely sources including livestock and wildlife. Approximately 2,600 isolates of *E. coli* from both Gimli and West Grand beaches were analyzed for DNA ribotyping. Results over the past several years indicate that approximately 63 % of the *E. coli* isolates were due to animals and 5 % of the isolates originated from humans (Figure 9.14). Approximately 32 % of the isolates could not be attributed to a particular source. Of the animal component, the greatest identifiable sources of *E. coli* were shorebirds and geese. The importance of shorebirds as a source of contamination to bathing water was also found in similar studies (Edge *et al.* 2010; Edge and Hill 2007; Fogarty *et al.* 2003; Whitman and Nevers 2003; Feare *et al.* 1999).

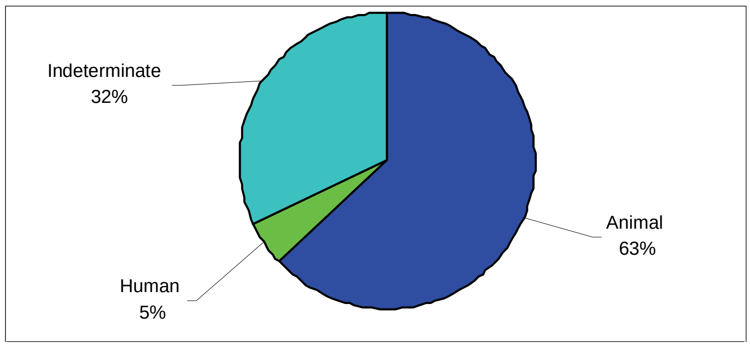


Figure 9.14: *Pie chart representing the percentage of total DNA isolates (n = 2,600) that could be discriminated into major groupings of animal, human, or indeterminate.*

Escherichia coli is an indicator of the possible presence of human pathogens. Since extensive sampling of West Grand and Gimli beaches demonstrated that a significant fecal load was present in wet foreshore sand, samples were analyzed for human pathogens. Weekly samples were collected during 2005 and 2006 and analyzed for *Campylobacter jejuni, Shigella sp., Samonella sp., Pseudamonas aerugenosa,* and *Vibrio parahaemolyticus.* In 2005, 40 samples of sand water and sand were analyzed for pathogens from Gimli and West Grand beaches. No isolates were found of *Campylobacter jejuni, Shigella sp., Samonella sp.* and one sand sample from West Grand Beach contained *Vibrio parahaemolyticus* (11 MPN/g). Approximately one-half of the sand water and sand samples from both West Grand and Gimli beaches contained *Pseudamonas aerugenosa* that ranged in density from 6 to 290 MPN/g (Table 9.5 on the following page). While little information is available about densities of specific pathogens in sand, Mendes *et al.* (1993) found densities of *Pseudamonas aerugenosa* in sand from various Italian beaches ranged from <3 to 2.4×10^7 organisms/g. There did not appear to be a difference in the number of pathogens on the east versus the west side of Lake Winnipeg, or between sand water and sand.

Table 9.5: *The ranges of densities or number of isloates for human pathogens found in sand, pore water, and bathing water from West Grand Beach and Gimli Beach during 2005 and 2006.*

Beach	Campylobacter jejeuni	Shigella sp.	Salmonella sp.	Pseudamonas aerugenosa	Vibrio parahaemolyticus
West Grand 2005					
Sand	No isolates	No isolates	No isolates	<3 to 210 MPN/gm	No isolates
Sand Water	No isolates	No isolates	No isolates	<3 to 20 MPN/mL	No isolates
West Grand 2006					
Sand	No isolates	No isolates	1 isolate	<3 to 9 MPN/gm	No isolates
Sand Water	No isolates	No isolates	2 isolates	<3 to 4 MPN/mL	No isolates
Bathing Water	No isolates	No isolates	1 isolate	<3 to 4 MPN/mL	No isolates
Gimli 2005					
Sand	No isolates	No isolates	No isolates	<3 to 290 MPN/gm	No isolates
Sand Water	No isolates	No isolates	No isolates	<3 to 240 MPN/mL	No isolates
Gimli 2006					
Sand	No isolates	No isolates	No isolates	<3 to 9 MPN/gm	No isolates
Sand Water	No isolates	No isolates	No isolates	<3 to 21 MPN/mL	No isolates
Bathing Water	No isolates	No isolates	No isolates	no isolates	No isolates

In 2006, 96 samples of sand water, sand, and bathing water were analyzed for pathogens from the two beaches. No isolates of *Campylobacter jejuni, Shigella sp.*, nor *Vibrio parahaemolyticus* were found in any samples. *Pseudamonas aerugenosa* was found at very low densities in occasional samples, and *Samonella sp.* was found in four samples (Table 9.5).

Given the high densities of fecal indicator bacteria in sand water and sand, the number of human pathogens is relatively small by comparison. Manitoba has no specific epidemiological data related to bather illness, but Colford *et al.* 2007 speculated that beaches without point sources of human sewage have lower bather illness rates compared to beaches that are directly impacted by point source discharge of human sewage. Lake Winnipeg beaches are in relatively isolated areas and seemingly unaffected by direct point sources of human fecal pollution.

However, studies conducted in Manitoba and elsewhere have shown that *E. coli* can survive and persist for long periods of time in bottom sediment and steam banks (Ishii *et al.* 2006, 2007). Other studies have demonstrated that continuous backstream loading of *E. coli* can influence downstream beach water quality (Whiteman *et al.* 2006; Byanppanahalli *et al.* 2003; Jamieson *et al.* 2003). Substantial densities of fecal indicator bacteria in stream sediments can become resuspended by wave action or increased stream flow causing elevated densities of indicator bacteria in recreational waters (Jamieson *et al.* 2003). Whiteman *et al.* (2006) speculated that examining the complicated variability of sources of fecal indicator bacteria and influencing factors should be conducted on a 'beachshed' bases.

9.4 Aquatic Invasive Species (AIS) in Lake Winnipeg

(Candace Parks, Wendy Ralley, Lucie Lévesque)

9.4.1 What Are Aquatic Invasive Species?

Aquatic invasive species (AIS) are non-native plants, animals, or micro-organisms that have been intentionally or unintentionally introduced by humans into areas outside of their natural ranges. Aquatic invasive species can significantly alter ecological relationships among native species and can affect ecosystem function, economic value of ecosystems, and human health. Because invasive species evolved elsewhere, they lack their natural predators and diseases which keep their populations in check. In many examples aquatic invasive species can out-compete native species for food and other resources.

Introductions of aquatic invasive species can result from accidental or unintentional actions such as unknowingly launching watercraft with aquatic invasive species attached, or deploying contaminated water-based equipment. Aquatic invasive species can be introduced by: discharging ballast water from ocean-going vessels, escape from fish farms, illegal use of live bait, opening of canals and waterways, release or escape of organisms from the aquarium or water garden trade, unauthorized fish transfers or stockings, releases of live food fish trade, recreational and commercial watercraft users moving watercraft, and gear (Ontario Ministry of Natural Resources webpage Jan 25, 2010). Unintentional introductions can have far-reaching impacts that cause irreversible ecological damage and cost economies millions, if not billions, of dollars to control. For example, the U.S. Fish and Wildlife Service estimates the economic cost of zebra mussels to U.S. and Canadian water users within the Laurentian Great Lakes region from 2000 to 2010 at $5 billion (Great Lakes Science Centre 2010).

Unintentional introductions (of aquatic invasive species) can have far-reaching impacts that cause irreversible ecological damage and cost economies millions, if not billions, of dollars to control.

9.4.2 Aquatic Invasive Species in Manitoba and the Susceptibility of Lake Winnipeg

In comparison to nearly 200 aquatic invaders in the Laurentian Great Lakes (Ontario Ministry of Natural Resources Jan 25, 2010), Manitoba has relatively few aquatic invasive species. To date, 14 aquatic invasive species, three of which are emergent plants have been introduced in Manitoba. Common Carp (*Cyprinus carpio* Linnaeus), Rainbow Smelt (*Osmerus mordax* Mitchill), White Bass (*Morone chrysops* Rafinesque), Feral Goldfish (*Carassius auratus* Linnaeus), Rusty Crayfish (*Orconectes rusticus* Girard), spiny waterflea (*Bythotrephes cederstroemi* Schödler) (*Eubosmina coregoni* Baird), freshwater jellies (*Craspedacusta sowerbyi*), black algae (*Lyngbya wollei* Farlow ex Gomont), purple loosestrife (*Lythrum salicaria* Linnaeus), flowering rush (*Butomus umbellatus* Linnaeus), invasive common reed (*Phragmites australis* subsp. *australis*), Asian tapeworm (*Bothriocephalus acheilognathi* Yamaguti) and Koi Herpes Virus (KHV) have been introduced into Manitoba's waters.

9.4.3 Aquatic Invasive Species in Lake Winnipeg

Common Carp *(Cyprinus carpio)*

Established populations of Common Carp were found in the Red River near Lockport in 1938 (Hinks 1943). Their tolerance to a wide range of environmental conditions has resulted in the wide distribution of Common Carp through the waters of Manitoba, including Lake Winnipeg and as far north as the Hayes River on Hudson Bay (A. Derksen, formally of Manitoba Conservation, pers.comm.). The impacts of Common Carp on aquatic environments may be substantial, resulting in alteration of the ecology of water bodies through disturbance of aquatic vegetation, (King and Hunt 1967; Robel 1961), and reduction in water clarity. Common Carp has been blamed for reduced abundance of waterfowl, native fish including visual predators such as northern pike, and other aquatic organisms (Parks 2006; K. Stewart pers. comm. in Ralley 2002; Swain 1979). The effects of Common Carp have been studied in Delta Marsh (Shay *et al.* 1999; Wrubleski and Anderson 1999), a large coastal wetland along the southern shore of Lake Manitoba. These findings are likely comparable to impacts sustained in Lake Winnipeg and its coastal wetlands. Common Carp are the dominant, large-sized fish comprising the majority of the fish biomass in the Delta Marsh (Wrubleski 1998). In-depth studies have not been conducted on Lake Winnipeg.

Rainbow Smelt *(Osmerus mordax)*
Rainbow Smelt were introduced into the Lake Winnipeg watershed by unauthorized, illegal introductions into lakes in the Rainy and English/Wabigoon river systems in Minnesota and northwestern Ontario (Stewart and Watkinson 2004). Their presence was subsequently noted in the south basin of Lake Winnipeg in 1991 (Campbell *et al.* 1991). It is suspected that Rainbow Smelt were introduced into Lake Winnipeg through unauthorized releases by humans directly into the lake (Campbell *et al.* 1991). Rainbow Smelt multiply rapidly, are a voracious predator of early year classes of native fish species, and compete with native fish species for food resources. In other areas where Rainbow Smelt have been established, the fish have been implicated in the decline of native species such as Cisco, Lake Whitefish and Emerald Shiners (Franzin *et al.* 1994). Rainbow Smelt appear to be partially filling a similar trophic niche as prey of larger species in the north basin of Lake Winnipeg, similar to the way Emerald Shiners do in the southern portion of the lake. As such, the presence of Rainbow Smelt in Lake Winnipeg may contribute to a decline in the success of commercial fisheries, and cause a deleterious effect on Lake Winnipeg aquatic communities (Franzin *et al.* 1994; Wain 1993; Remnant 1991). Preferential feeding on Smelt versus native forage species will likely increase growth rate and fat content of Lake Winnipeg Walleye and Pike (Stewart 2000), potentially contributing to a reduction in palatability of these species.

Rainbow Smelt in Lake Winnipeg may contribute to a decline in the success of commercial fisheries, and cause a deleterious effect on Lake Winnipeg aquatic communities.

White Bass *(Morone chrysops)*
White Bass are voracious, predatory fish that were introduced into Lake Ashtabula, North Dakota in 1953 (Stewart and Watkinson 2004). They were first reported in commercial catches in the south basin of Lake Winnipeg in 1963. By the mid-1990s they were found throughout the lake, and were the most abundant spiny-rayed fish in the south basin (Stewart and Watkinson 2004). Though White Bass have not been determined to have a detrimental effect on the ecological characteristics and functioning of Lake Winnipeg, they are thought to displace Yellow Perch (*Perca flavescens* Mitchell) in the south basin (K. Stewart pers. comm. in Ralley 2002). The population of Emerald Shiners, the primary food source for White Bass, apparently has not changed in Lake Winnipeg. The long-term trends and impacts of White Bass on Lake Winnipeg ecology and fisheries are not conclusive (M. Erickson pers. comm. 2009).

Cladoceran *(Eubosmina coregoni)*
This cladoceran, native to Eurasia, was introduced to the Laurentian Great Lakes region in the mid-1960s (Lieder 1991). It was likely first introduced in the Lake of the Woods area in the early 1990s and then entered Manitoba through the Winnipeg and Rainy rivers (Suchy and Hann 2007). Transport of *E. coregoni* in the digestive tract of Rainbow Smelt is hypothesized to be responsible for its introduction to Lake of the Woods (Suchy and Hann 2007). *E. coregoni* were first observed in Lake

Winnipeg in 1999 (A. Salki pers. comm. in Ralley 2002) and quickly became the most dominant cladoceran in the north basin. The impacts of this alien invasive species on the Lake Winnipeg ecosystem, including trophic status and food web are unknown and are currently under investigation by Fisheries and Oceans Canada.

Asian tapeworm *(Bothriocephalus achelignoathi)*
The Asian tapeworm was first discovered in White Bass in the south basin of Lake Winnipeg in 2003 (Choudhury *et al.* 2006). Subsequently, the tapeworm was found in Emerald Shiners from the Red River near its entrance to Lake Winnipeg in 2006 (International Red River Board 2008). Although it is not known to infect humans, the tapeworm can cause weight loss, anemia, and mortality of young fish (Marcogliese 2008). It is unknown how the parasite first infected Lake Winnipeg fish species. To date, there have been no noticeable impacts of the Asian tapeworm on fish abundances in Lake Winnipeg. Its presence in Lake Winnipeg Emerald Shiner populations raises serious concerns, however, as spread of the infestation could increase fish mortality or pathogenic effects in other predatory fish populations such as Walleye, Sauger, or Goldeye.

9.4.4 Pathways for AIS Spread into Manitoba and Lake Winnipeg
Accurately predicting the pattern and rate of invasion of aquatic invasive species into Manitoba and Lake Winnipeg is difficult given that dispersal of AIS is dependent on numerous factors including the interprovincial and international nature of water within watersheds draining to Manitoba and Lake Winnipeg and the relative proximity of other watersheds and basins to Lake Winnipeg. Additional pathways of concern to Manitoba include:

- diversions and canals that connect water basins and watersheds,
- recreational or commercial boating watercraft or other water-based equipment on which AIS can attach,
- bait fish release, and
- aquarium and water garden trade and the illegal release of non-native aquatic species.

Inter-basin transfers and water diversions into the Red River drainage area pose significant threats of biota transfer. In addition to inter-provincial and international waters, the relatively small size of land barriers separating Manitoba's waters from other watersheds may also contribute to the introduction of invasive species to Lake Winnipeg. Round Goby, Ruffe, Eurasian Water Milfoil, and a variety of other aquatic invasive species are found in adjacent watersheds. The movement of watercraft by national and international recreational boaters between water bodies is largely responsible for the range expansion of AIS, such as zebra and quagga mussels to inland water bodies and isolated lakes across North America. Once these species invade a waterbody, passive dispersal by water currents contribute further to their ability in establishing downstream.

Inter-basin transfers and water diversions into the Red River drainage area pose significant threats of biota transfer.

Bait fish releases remain a concern for AIS introductions into Manitoba. Anglers can accidentally spread non-native species and diseases by releasing unwanted bait and water. Releasing plants and animals purchased through the aquarium and water garden trade pose threats to AIS introduction into Manitoba. Although many consider this action to be harmless, diseases and parasites can be introduced unknowingly. Koi Herpes Virus was positively identified in Lake Manitoba in 2008 and it is suspected diseased pet goldfish were released into the lake resulting in hundreds of dead, mature Common Carp washing up on the shores of Lake Manitoba.

Unintentional releases of purple loosestrife seeds from a local garden retailer resulted in the blanketing of the Assiniboine River and eventually the Red River and Netley-Libau Marsh shorelines with the invasive species. Purple loosestrife is a noxious weed and is listed as such under Manitoba's *Noxious Weed Act* thereby prohibiting the sale or possession of the plant in the province.

9.4.5 Potential Threats to Lake Winnipeg

Spiny Water Flea *(Bythotrephes longimanus)*
Spiny water flea is a rapidly reproducing crustacean native to Europe and the Caspian Sea that was accidentally introduced into the Laurentian Great Lakes. Spiny water fleas were found in Lake of the Woods in 2007, entering through the Rainy River system. Spiny water fleas compete with small fish and other invertebrates for food. Due to their long, spiky tail, spiny water fleas are not readily consumed by small-sized predators and therefore their numbers quickly grow to dominate the water body that they have invaded. Spiny water fleas are a nuisance to commercial fishers and anglers, they entangle fishing lines and nets and impair downrigger cables. Spiny water fleas were identified downstream from Pointe du Bois Dam on the Winnipeg River in 2009 and upstream of Powerview Dam in August 2010. Two samples were collected from Traverse Bay in Lake Winnipeg from the *MV Namao*, the research ship on Lake Winnipeg, on October 19, 2010. However, no spiny water fleas were found. Further sampling in Lake Winnipeg for spiny water flea will occur in 2011. It is likely they are already present in Lake Winnipeg. Manitoba Water Stewardship will continue to work with neighbouring jurisdictions such as the Province of Ontario on tracking the distribution and movement of spiny water flea.

Spiny water fleas are a nuisance to commercial fishers and anglers, they entangle fishing lines and nets and impair downrigger cables.

Rusty Crayfish *(Orconectes rusticus)*
Rusty Crayfish is an aggressive, invasive species not native to Canada. This species first became established in Lake of the Woods, Ontario in the early 1960s. In 2005, they were discovered in the Ontario portion of the Winnipeg River. In July 2007, Rusty Crayfish were discovered in Falcon Lake, in Whiteshell Provincial Park. Best estimates indicate that Rusty Crayfish were likely introduced into Falcon Lake by anglers using them as bait. Populations of Rusty Crayfish have the potential to dramatically disrupt the aquatic ecosystem. These hardy, aggressive invaders threaten native crayfish and fish species by competing for aquatic invertebrate food resources and shelter. They eliminate fish spawning and nursery habitats by cutting aquatic vegetation with their powerful claws. Once introduced, they expand their range through connected waterways or by illegal bait transfers. To prevent further spread, the Province of Manitoba banned the possession and transport of all species of crayfish in 2007, including those used for human consumption. Additional monitoring for Rusty Crayfish in Whiteshell lakes and other lakes in central and western Manitoba was conducted in late summer 2008 and 2009 (M. Erickson pers. comm. 2009). Falcon Lake is the only lake in Manitoba to have a known population of Rusty Crayfish.

Falcon Lake is the only lake in Manitoba to have a known population of Rusty Crayfish.

Black Algae *(Lyngbya wollei)*
Black algae is an invasive and exotic blue-green algae that grows predominantly in the southeastern United States, most abundantly in Florida. Manitoba's Whiteshell Provincial Park (White Lake and Betula Lake) is the only known location in Canada to have black algae. This algae forms dense mats on the lake bottom or floats in areas near the shore. Profuse mats of black algae are aesthetically unpleasant for swimmers and boaters. Black algae has not been recorded in Lake Winnipeg to date.

Koi Herpes Virus
The first occurrence of Koi Herpes Virus was found in Common Carp from Lake Manitoba in August 2008. The virus likely entered Manitoba waters by the unauthorized release of infected pet Koi or Goldfish (Manitoba Water Stewardship 2009a). This is a new and exotic disease that only affects Common Carp, Koi, and Goldfish, none of which are native to Manitoba. Koi Herpes Virus is not harmful to humans. The virus is found world-wide and was first detected in North America in 1999 and in Canada in 2007 (Manitoba Water Stewardship 2009a). The virus is likely impossible to treat and is transmitted by direct contact with infected fish. At present, Koi Herpes Virus has not been recorded in Lake Winnipeg.

9.4.6 Zebra and Quagga Mussels Threat to Manitoba Waters
Zebra (*Dreissena polymorpha* Pallas) and quagga mussels (*Dreissena rostriformis bugensis* Andrusov) are native to the Black, Caspian, and Azov seas of eastern Europe and western Asia. During the last century, zebra mussels have migrated throughout much of central and western Europe due to increased shipping through extensive networks of constructed inland waterways and canals. They were first found in North America in Lake St. Clair in the late 1980s. It is thought that zebra mussels were accidentally released in Lake St. Clair through discharged freshwater ballast from an ocean-going ship.

Zebra mussels were found in (Big) Pelican Lake, Minnesota in September 2009, which was the first occurrence of established colonies in the Lake Winnipeg watershed.

Zebra mussels have since invaded all of the Laurentian Great Lakes and many adjoining tributaries including the St. Lawrence Seaway, the Trent-Severn system, and the Ottawa, Mississippi, Missouri, Hudson, Illinois, Ohio, Arkansas, and Tennessee rivers. Zebra mussels were found in (Big) Pelican Lake, Minnesota in September 2009, which was the first occurrence of established colonies in the Lake Winnipeg watershed (Manitoba Water Stewardship 2009b). This discovery is particularly important to Manitoba since it demonstrates how a costly invader can be accidentally transported overland on recreational watercraft to establish in a separate drainage basin. On July 1, 2010, a zebra mussel veliger was subsequently found on a dam in the Red River at Wahpeton, North Dakota and downstream from Pelican Lake, Minnesota. It is suspected veligers, or the larval stage of zebra mussels, were swept downstream into the Red River. Dams and weirs act as recruitment sites where zebra mussels can reproduce. Passive dispersal of larval, free floating zebra mussel veligers is thought to be the predominant means of rapid dispersal of this species throughout the Laurentian Great Lakes (Benson and Raikow 2009). The Red River was identified as a high risk waterbody for zebra mussel colonization based on certain water quality characteristics (Sorba and Williamson 1997). The Winnipeg River, among several other water bodies in the Precambrian Shield of south-eastern Manitoba, was evaluated as having a lower potential due to lower calcium concentration, which limits the ability of zebra mussels to create a shell (Ralley 2002).

Zebra Mussel

Quagga mussels are a close relative to zebra mussel and in the late 2000s they significantly increased their range in North America largely due to the movement of trailered watercraft. In 2007, quagga mussels established in the lower Colorado River and have recently established in the Columbia River. Of concern to Manitoba is that quagga mussels can establish on softer substrates which is a condition offered in large areas of the Red River and Lake Winnipeg. Both adult zebra and quagga mussels can be easily transported overland on watercraft, trailers, aquatic plants, float-plane pontoons, and related equipment. In cool, damp conditions, adults can survive upwards of 14 days out of water. To date, live zebra or quagga mussels have not been recorded in Manitoba's waters.

9.4.7 Activities to Prevent Further Spread of Aquatic Invasive Species

Fisheries and Oceans Canada and Manitoba Water Stewardship work together to prevent and manage introductions of aquatic invasive species to Manitoba. The Province's efforts to prevent introductions include public outreach programs, monitoring, developing strong policy, and provincial preparedness activities. For the past 20 years, Manitoba has aimed its efforts at preventing and delaying invasive species introductions into provincial waters.

The *Manitoba Fisheries Regulation* under the federal *Fisheries Act* was amended in 2008 to significantly increase the number of prohibited species anglers are allowed to possess or transport. Manitoba legislation also prohibits the transportation of live bait into the province, as well as fishing with live bait in some areas. The positive identification of Rusty Crayfish in Falcon Lake Manitoba in 2007 prompted the Province to prohibit the possession of all species of crayfish. In 2003, all provinces and territories became signatory to the *National Code on Introductions and Transfers of Aquatic Organisms*. This code sets a framework to assess risk to the aquatic environment should a specific organism be introduced or released. Additional gaps to current legislation include limiting introduction of aquatic organisms by the aquarium trade, live food trade, and water garden trade.

Since the Lake Winnipeg watershed is shared between four Canadian provinces and four U.S. states, communication about activities and newly established invaders outside of Manitoba's borders is key to preparedness.

Efforts in Manitoba concerning aquatic invasive species have been aimed at public outreach and education including development and distribution of written material, presentations, surveys, inspections, boat launch signs, and others. Since the Lake Winnipeg watershed is shared between four Canadian provinces and four U.S. states, communication about activities and newly established invaders outside of Manitoba's borders is key to preparedness. Manitoba participates on the Western Regional Panel on Aquatic Nuisance Species, a panel with membership from the 17 western U.S. states, four western Canadian provinces, and the Upper Mississippi Basin Panel. Communication and coordination is also provided through Manitoba's participation on the National Aquatic Invasive Species Committee under the direction of the Canadian Council of Fisheries and Aquaculture Ministers. Manitoba has worked to prevent the introduction of zebra mussels across the 100th meridian.

Each summer since 2000, the Province of Manitoba has had summer students conducting watercraft and trailer inspections at the main two Canada/U.S. border crossings in Sprague and Emerson, as well as at points within Whiteshell Provincial Park. Trailered watercraft are inspected and travelers are surveyed about their general knowledge of aquatic invasive species. On an annual basis in late September and early October, Manitoba Water Stewardship staff inspects navigational buoys, anchors, and chains pulled from the Red River and the south and north basins of Lake Winnipeg to determine whether there is evidence of the presence of zebra mussels or other invasive species. Swimming buoys, ropes, and water level recorders are inspected at several lakes in Whiteshell Provincial Park that correspond to heavy recreational boating areas from both inside and outside Manitoba's borders. The Province, in cooperation with Environment Canada, began monitoring weekly for zebra mussel veligers from the Red River at Emerson.

Each year new proactive efforts are developed to prevent the intentional and accidental introduction of AIS into Manitoba. These include efforts aimed at heightening public awareness to AIS and reminding the public of their responsibility to clean, drain, and dry their aquatic equipment and to take other precautionary steps to prevent the spread of AIS in Manitoba. The Province put into place an aquatic invasive species hotline (1-87-STOP AIS-0 or 1-877-867-2470) that is a means for the general public to report AIS sightings and to obtain information. Partnerships with neighbouring provincial and state jurisdictions and long-running education and outreach programs in Manitoba have resulted in a high level of public awareness which speaks to the success of keeping AIS such as zebra mussels out of Manitoba waters.

9.5 Climate Change

(Greg McCullough)

Atmospheric climate governs the thermal regime of lakes, forcing the development and decay of winter ice cover and the warming and possible stratification of the water column through the spring and early summer, and cooling through autumn. Primary productivity, and cyanobacterial productivity in particular, has been shown to be positively correlated with summer temperature in numerous multi-year studies (e.g. Hamilton *et al.* 2005; Milius *et al.* 2005; Park *et al.* 2004). Consequently, the nature and extent of eutrophication and associated effects on Lake Winnipeg may be affected by rising air temperatures.

Analysis of data in satellite images has been used to show that, given sufficiently high nutrient concentrations, surface blooms of phytoplankton in the lake tend to be larger and more intense in warmer years (McCullough *et al.* 2006). If greater summer warming leads to more frequent stratification, then dissolved oxygen concentration in bottom waters may be reduced, possibly to the point of anoxia in the hypolimnion or surface sediments (Section 5.2) and internal nutrient loading may be increased (Text Box 6; Section 7.0). This in turn has implications for primary productivity in Lake Winnipeg. Any changes to water chemistry or primary production would have consequences at higher tropic levels. Moreover, increased temperature may affect fish directly. In a recent study of potential climate effects on the Lake Winnipeg fishery, Franzin *et al.* (2005) pointed to summer temperature as one determinant of the species composition in a lake, and to the importance of the length of the growing season (usually defined as the period with water temperature >10^{o}C) to over-winter survival of young-of-the-year fish.

More specific discussion of potential effects of climate change on the ecology and biology of Lake Winnipeg depends on quantitative estimates of the effects of potential climate change on future summer water temperatures and the length of the open water season on the lake. To this end, results of a simple model of future thermal regimes is presented below. The model is described in Appendix 6. Briefly, it employs empirical relationships between monthly water temperature, spring ice melt/break-up, and fall freeze-up for the north and south basins, and monthly air temperature, and is forced by air temperature predictions from Version 3 of the Canadian Global Climate Model (CGCM3).

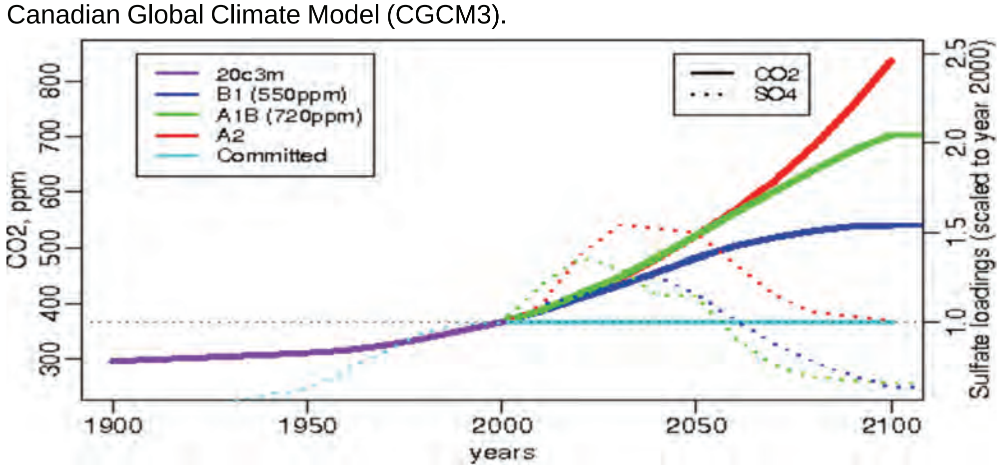


Figure 9.15: *Time evolution of global carbon dioxide (CO^2) concentrations (solid lines) and sulphate aerosol loadings (dotted lines) as predicted in several scenarios developed by the Intergovernmental Panel on Climate Change. "20c3m" describes historic increases. "Committed" describes GHGs and aerosols fixed at year 2000 levels. This study predicts future temperature and ice break-up and freeze-up periods based on the three scenarios B1, A1B, and A2.*

The CGCM3 predicts potential climate effects of potential 21st century changes in atmospheric concentrations of both greenhouse gases and aerosol given various forcing emission scenarios developed by the Intergovernmental Panel on Climate Change. Three scenarios that predict a broad range of future climates are modelled here. The scenarios describe evolution of greenhouse gas concentrations and aerosol loadings in response to varying population growth, economic development, and technological change over the next century. In order of increasing greenhouse gas concentrations developed over the next century, the scenarios are referred to as B1, A1B, and A2 (Figure 9.15). As with all global climate models, projected temperature tends to be highly variable among models. The degree to which these climate models are useful in assessing potential future conditions also depends on the scale at which they are applied. Consequently, interpretation of projected changes for small regions (such as that for Lake Winnipeg and area) should be made conservatively.

Under the strongest A2 forcing scenario, the CGCM3 predicts that mean mid-summer air temperature over Lake Winnipeg will increase by 2.4°C during the first half of the 21st century and another 2.0°C during the second half of the century (Figure 9.16). Under the weakest forcing scenario, B1, a mean mid-summer increase almost as great in the first half of the century, 2.0°C is predicted, but that increase is followed by only 0.3°C more in the second half of the 21st century. Every scenario predicts mid-winter (December to February) air temperature increases that are 60 to 120 % greater than for mid-summer months. Modelled air temperature increases are least in the spring and late autumn months (April, May, and November) although even in those months, the CGCM3 predicts increases of 1.1 to 2.1°C by mid-century, with an additional 0.1 to 1.8°C by the end of the century. Spring and autumn air temperature increases are the main drivers of potential changes in the period of spring break-up and autumn freeze-up of the ice cover on the lake.

Over the period 1970 to 1992, mid-summer (July and August) surface water temperature averaged 18.5°C in the north basin, and 21.0°C in the south basin. Under any of the three modelled forcing scenarios, mean mid-summer surface water temperatures are predicted to rise by 1.9 to 2.5°C in both basins through the

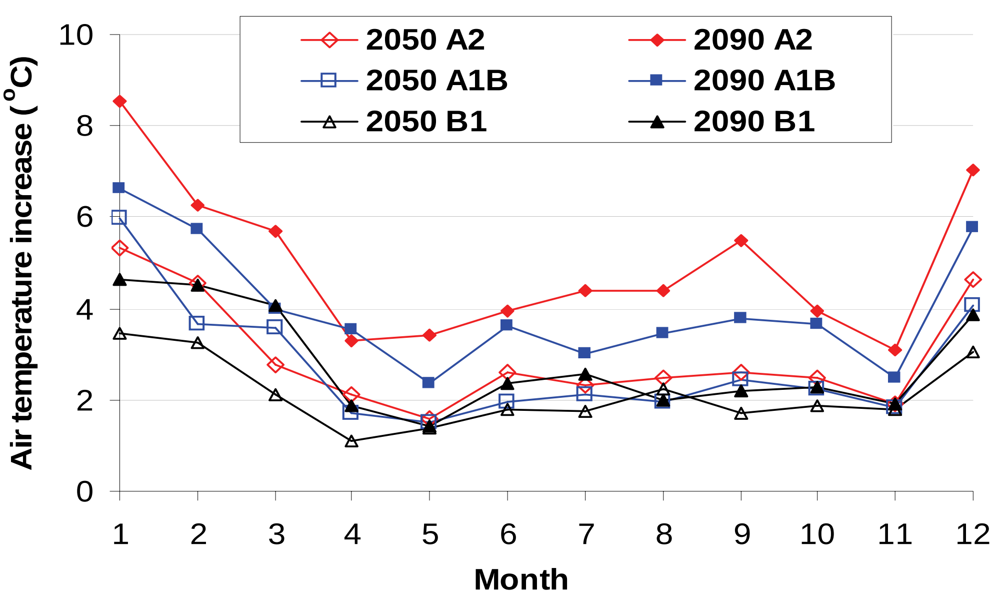


Figure 9.16: *Increases in air temperature predicted for the Lake Winnipeg region for two 20-year periods, 2040 to 2059 (labelled 2050) and 2080 to 2099 (labelled 2090) using CGCM3 with 3 forcing emissions scenarios, B1, A1B, and A2. Each point is the mean of 100 predicted values, i.e. 5 model runs X 20 years. Predicted 21st century temperatures are compared to temperatures modelled using CGCM3 over the base period from 1970 to 1992 (excluding the years 1976, 1980, and 1981 for which considerable data is missing in the observed record).*

first half of this century. Under scenario A2, surface water temperatures are predicted to rise by about a further 2.0°C by the end of the 21st century; under scenario B1, only a further 0.5°C, approximately. That is, under forcing scenario A2, mean mid-summer temperatures in the north basin would be 21.0°C by 2050 and 22.9°C by 2090, and in the south basin, 23.5°C by 2050 and 25.5°C by 2090 (Figure 9.17). Under scenario B2, with lower greenhouse gas increases, mid-summer warming through the first period would be much the same as under A2 (21.0°C in north basin and 23.5°C in south basin by 2050), but by 2090, the increase would be more modest. Under B2, by 2090 the median mid-summer surface temperature in the north basin would be 20.9°C, and in the south basin, 23.4°C. Under the intermediate forcing scenario A1B, increases would again be the same as for scenarios A2 and B1, but would continue to rise through the second half of the century, although at only about one-half the rate of the strongest forcing scenario A2.

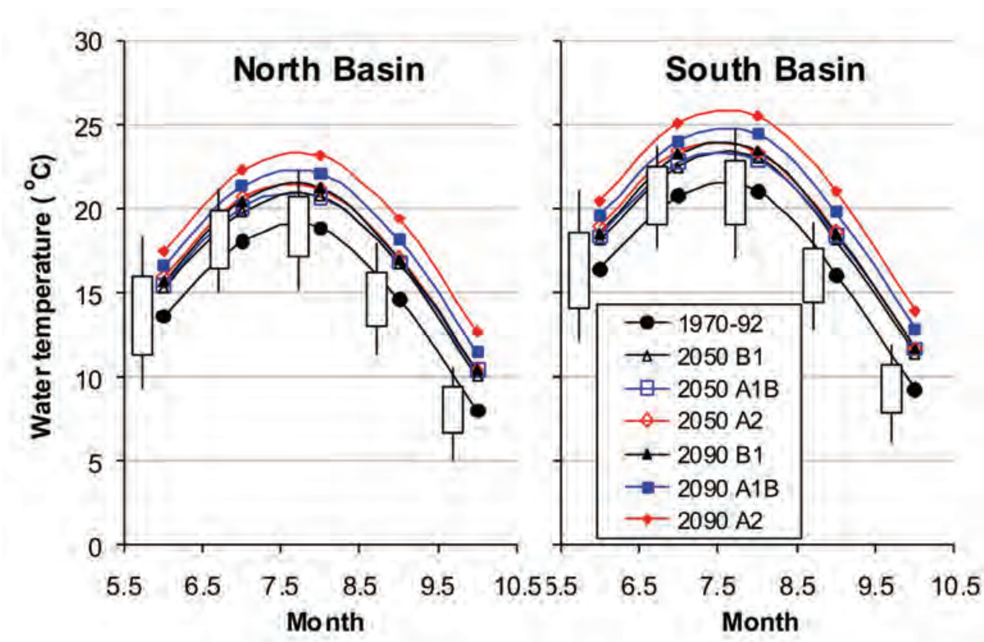


Figure 9.17: *Historic and predicted 20-year averages of open water season monthly mean surface water temperatures in the north and south basins of Lake Winnipeg. Offset box and whisker plots indicate one standard deviation and range of the 1970 to 1992 data. Air temperature used to estimate water temperature for the 1970 to 1992 period (n = 20 with 1976, 1980, and 1981 excluded due to missing data) is from the observational record as described in the text. In the legend, 2050 refers to the 20-year period 2040 to 2059; 2090 refers to 2080 to 2099. Air temperature data for 21st century periods were retrieved from CGCM3 model runs, with 3 forcing emissions scenarios: B1, A1B and A2. For these data sets, n = 100 (i.e. 5 model runs X 20 years).*

The standard deviations of mid-summer mean surface water temperatures over the period 1970 to 1992 (estimated from the historical air temperature record) have ranged from 1.6°C for July data to 2.0°C for August data. This implies the probability that for roughly one year in six, the peak monthly mean mid-summer surface water temperature would be at least 1.6°C warmer than predicted under each scenario. That is, under the strongest forcing scenario A2, by the end of the 21st century the July or August mean temperature of the north basin would be over 22.6°C, and of the south basin, over 24.5°C, about one year in six. Both values are as high (for the south basin) and higher (for the north basin) than the highest monthly mean temperatures in the last 60 years (based on monthly mean temperature records modelled from the historic air temperature record). On the other hand, the whole water column is unlikely to be heated to these predicted surface temperatures. In the great majority of recorded thermal profiles, the temperature range through the water column has been less than one or two degrees (Figure 5.8). However, in some years the water column has stratified to form a hypolimnetic layer which has remained several degrees cooler than at the surface through the warmest period of the year (Figure 5.9). It may be that the tendency to stratify during the summer months will increase with greater warming of surface waters but this possibility has not been explored.

Figure 9.18 shows historic and predicted dates of mid-break-up (defined here as the date when the basin is half clear of ice in the spring) estimated from air temperature recorded through the historic period 1970 to 1992 and from adjusted CGCM3 daily air temperature predicted for the periods 2040 to 2059 and 2080 to 2099. Under any of the three modelled forcing scenarios, by the middle of the 21st century mid-break-up would occur on average four to six days earlier in the south basin and about a week earlier in the north basin. Under the weakest forcing scenario, B1, there would be little further change over the next half-century. Under the strongest forcing scenario, A2, the timing of break-up would continue to advance, so that by the end of the 21st century it would be a week and one-half earlier on the south basin, and nearly two weeks earlier in the north basin.

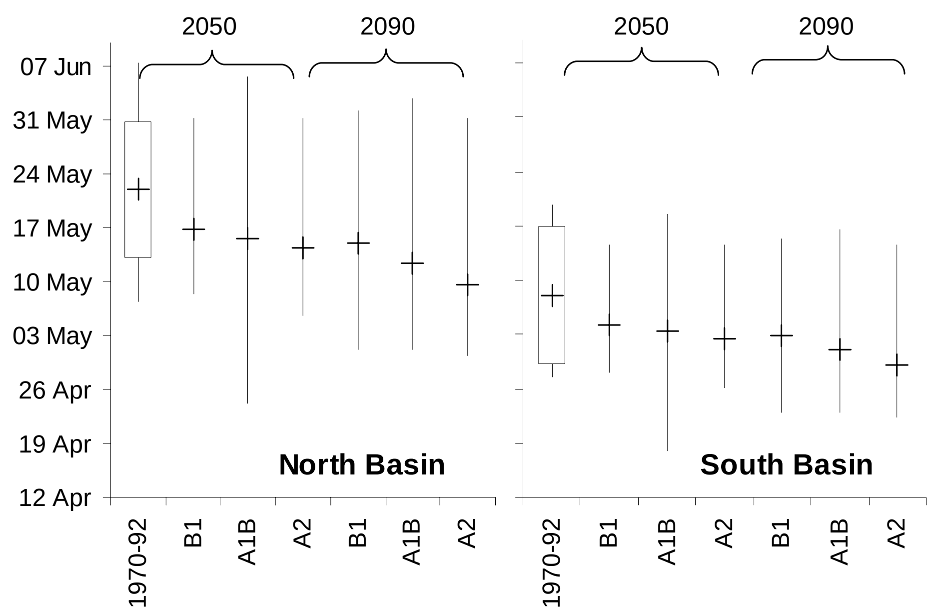


Figure 9.18: *Historic and predicted dates of mid-break-up (date when the basin becomes half clear of ice) on the north and south basins of Lake Winnipeg. Crosses indicate means. For 1970 to 1992 data, boxes indicate 1 standard deviation and whiskers indicate range of values (n = 20). For predicted 21st century values, only the means and ranges are shown (n = 99 - 100; i.e. 5 model runs X 20 years, with a single early April outlier removed from each of A1B 2050 and A2 2090. Note that for the historic data, the standard deviation is reasonably representative of inter-annual variability - roughly two and one half weeks - whereas for the predicted data, it would represent some combination of inter-annual variability and cumulative differences among 5 model runs). In the legend, 2050 refers to the 20-year period 2040 to 2059; 2090 refers to 2080 to 2099.*

Figure 9.19 shows historic and predicted dates of mid-freeze-up on the south basin (*i.e.* the date when the basin is half covered by ice in the autumn). By the middle of the 21st century, under any of the three modelled forcing scenarios, freeze-up will be on average almost a week later than at the end of the 20th century. As with break-up, under the weakest forcing scenario, B1, there would be little further change over the next half-century. Under the strongest forcing scenario, A2, the timing of freeze-up would be further delayed, so that by the end of the 21st century it would be two weeks later. Under the intermediate forcing scenario, A1B, it would be a week and one-half later. Freeze-up of the north basin was not modelled. In those years when the process could be studied using satellite data, it has followed freeze-up on the south basin within a few days (Figures 5.3, 5.7). It is unlikely that this small lag will be changed significantly under any of the scenarios considered, since the waters in the north basin are deeper than in the south basin and heat loss will be slower in the former.

From 1946 to 1991, when records were kept by the weather observer at Gimli, the average ice-free season in the south basin of Lake Winnipeg (from last observed ice in spring to first observed ice in autumn) was 180 days. The ice-free period in the north basin is estimated to be roughly two weeks less (Section 5.1). From the advances in break-up and retardation of freeze-up just described, under any of the modelled warming scenarios, it is likely that by the middle of the 21st century the ice-free season will have been increased by two weeks in each basin. Under the weakest warming scenario, B1, there will be little further change over the next 50 years. However, under the strongest warming scenario, A2, the ice-free season will be very nearly a month longer - roughly 194 days in the north basin and 204 days in the south basin - by the end of the 21st century.

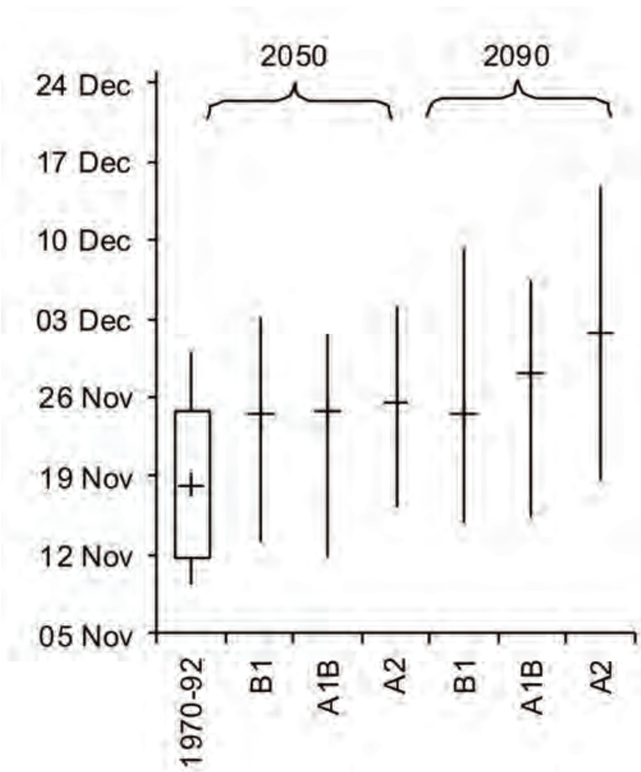


Figure 9.19: *Historic and predicted dates of mid-freeze-up (date when the basin becomes half covered by ice, estimated using the 40-day moving average air temperature as described in the text) on the south basin of Lake Winnipeg. The north basin typically freezes over within a few days of the south basin. Other information are as in Figure 9.18, except n = 98 - 100 for predicted data (with 1 October outlier removed from B1, 2050 and two removed from each of B1 2090, A1B 2050 and A2 2050).*

The conclusion that freeze-up will happen later in the season, given altered 21st century climate regimes, differs from the result reported in an earlier study (McCullough 2005) where an earlier version of the Canadian Global Climate Model (CGCM2) indicated only small increases in autumn air temperature, and therefore negligible change in the mean timing of freeze-up. The more recent version of the model employs improved spatial resolution of the atmospheric computations and improved treatment of land surface processes. Therefore, it is presumed to provide improved predictions of future changes in air temperature under the Intergovernmental Panel on Climate Change scenarios tested.

Spring melt and break-up are dynamic processes forced not only by atmospheric warming, but more particularly by radiative heating and, in the later stages, winds. The timing must also be an inverse function of ice thickness at the onset of melt. Of the forcing variables, over the long term, wind effects may be considered random. It is likely that radiative heating is sufficiently covariant with warming that potential future change will not alter the relationship.

However, ice thickness at the onset of melt is independent of atmospheric warming in the spring, and may be affected by climate change. A thicker ice cover will develop given colder weather early in the freeze-up period. On the other hand, snow accumulation early in the freeze-up period provides insulation from the atmosphere, and is at least as important as air temperature to the ultimate thickness of the ice cover. Global climate models do not provide a consistent guide to changes in precipitation in the region, but global climate models in general and the CGCM2 in particular, do predict warmer winters in the region. Thus, ice thickness at the onset of melt seems more likely to decrease rather than increase given climate warming over the remainder of this century. Consequently, the estimated advance of break-up presented here should be considered conservative.

The conclusion that the mean time of freeze-up will be retarded is not constrained by these considerations. Cooling of the lake is indeed more a function of radiative and evaporative cooling than it is a function of physical transfer of heat into a cooling atmosphere. Nonetheless, the strong correlation with atmospheric cooling suggests that other factors are on average covariant with atmospheric cooling. In any case, other possible forcing variables are not predicted with sufficient certainty by current global climate models to warrant further investigation here.

In a study by Franzin *et al.* (2005), the consequences of these potential changes to the thermal regime of Lake Winnipeg have been extended only to the fishery and not to other trophic levels in the ecosystem. In general, Franzin *et al.* (2005) noted that the increases in peak mid-summer temperature described above should be positive for species near the northern limit of their distribution, but detrimental for those near their southern limit. In particular, Franzin *et al.* (2005) concluded that a 2.0°C increase in mid-summer monthly mean temperature by the middle of the 21st century, could limit the success of as many as 12 fish species common to Lake Winnipeg, and that a further 2.0°C warming (CGCM3, Scenario A1 only) could affect another 15 fish species by the end of the century. Prominent among these are Lake Sturgeon, Lake Trout, and Burbot, which could be limited out as early as 2050, and Lake Cisco and Lake Whitefish, by the end of the century. On the other hand, warm water species, including Walleye, Yellow Perch, and White Bass, would be expected to benefit under any climate-warming scenario from more growth during the longer, warmer open water seasons, leading to better first-winter survival through the shorter winters predicted. For Walleye, at least, this expectation is supported by a statistical study by Lysack (2005) of factors influencing fish populations in Lake Winnipeg. By regression analysis employing multiple environmental variables, Lysack (2005) reported that either vernal warming (approximately equivalent to the date of ice break-up) or heating degree days >10°C improved the prediction of stock recruitment of Walleye (using the Gimli record of ice observations and daily temperature from 1972 to 1991).

Franzin *et al.* (2005) concluded that a 2.0°C increase in mid-summer monthly mean temperature by the middle of the 21st century could limit the success of as many as 12 fish species common to Lake Winnipeg, and that a further 2.0°C warming could affect another 15 fish species by the end of the century.

Unfortunately, Franzin *et al.* (2005) were not able to evaluate indirect ecosystem effects (e.g. due to effects of changes in the thermal regime on benthos and plankton) on fish in Lake Winnipeg. The preceding quantitative analysis should provide a useful basis for such an evaluation. On the other hand, it does not extend to possible effects of climate change on the Lake Winnipeg watershed. These may include altered runoff quantity and seasonality, and total nutrient loading (Schindler 2009) with potentially greater effects on the lake than direct temperature increases.

10. KNOWLEDGE GAPS

(Lucie Lévesque, Elaine Page)

Eutrophication of Lake Winnipeg has become increasingly pronounced over the past decade and has become the prime concern regarding the health of the lake. Periodic research and monitoring from the 1930s to the 1990s provides some indication as to past conditions in Lake Winnipeg, supported by paleolimnological research which has pointed to deteriorating water quality in Lake Winnipeg since the early 1900s. More rigorous and continuous monitoring on the lake in response to the increase in the intensity and frequency of algal blooms in Lake Winnipeg over the past 10 years has provided information on the chemical, physical, and biological characteristics of Lake Winnipeg, and has highlighted the highly variable nature of water quality in the lake.

Further information is needed for a more complete understanding of the lake, the sources and nature of stressors, and the response of the lake to these stresses.

Knowledge of Lake Winnipeg has been further enhanced by focussed research that has provided crucial information on nutrient dynamics and balance, trophic structure and interactions, responses of biological communities to nutrients, range and nature of physical and hydrodynamic conditions in the lake, and nutrient sources. These studies and recent examinations of change over the long term have provided new insights into the structure and function of the lake. Further information is needed, however, for a more complete understanding of the lake, the sources and nature of stressors, and the response of the lake to these stresses, thereby facilitating the development of management objectives for Lake Winnipeg. A brief summary of existing knowledge gaps as described through the work presented in this report is discussed below. Other gaps such as in emerging compounds, contaminants, etc. are beyond the scope of this report.

Hydrology

Several components of the lake water balance remain unresolved.

Long-term hydrometric monitoring in and near Lake Winnipeg has provided a reliable historical record of lake water levels and major tributary discharges into and from the lake. However, several components of the lake water balance remain unresolved, including estimates of evaporation, groundwater inflow, and contributions from remote ungauged rivers flowing into the east side of the lake. Currently, evaporation from Lake Winnipeg has been estimated using Dauphin Lake as a proxy. Additionally, the nature of freeze-up and break-up in Lake Winnipeg, which is of importance to the hydrology and critical to the ecology of the lake, requires further consideration. Information on the length of the open water season and inter-annual variation of break-up and freeze-up dates is limited for the north basin, while some information on the south basin is available. These aspects of Lake Winnipeg's hydrology are essential to the development of complete water budgets and ultimately to continued water quality and food web modelling, which are key in supporting future water resources management decisions in the lake and watershed.

Physical and Chemical Characteristics of Water

The physical and chemical characteristics of Lake Winnipeg's waters have been measured over recent decades by monitoring programs as well as through research studies. Although the record of physical and chemical characteristics of Lake Winnipeg from recent decades is relatively continuous, gaps in knowledge and information still remain with respect to under-ice conditions, littoral areas of Lake Winnipeg, and the range and variability in water characteristics intra- and inter-annually, as well as spatially through the lake.

Monitoring and research on Lake Winnipeg takes place for the most part over the open water season. The sampling network for Lake Winnipeg is limited to pelagic areas where the research vessel *MV Namao* is able to navigate on its three seasonal cruises. Information on physical and chemical characteristics of inflows to the lake from tributaries in the more remote eastern side of the lake is also limited. Environment Canada is currently conducting a network optimization study to determine the number and location of sampling stations, and the frequency of sampling, that most effectively represent the characteristics of Lake Winnipeg.

The physical and chemical nature of Lake Winnipeg waters has been observed to be highly heterogeneous inter- and intra-annually, and spatially. However, the nature and degree of this variability (e.g., with depth and season) is not well defined. In addition to limited information on the nearshore areas of the lake, information on the characteristics of the lake through the water column is of particular interest given recent observations of thermal stratification and potential for dissolved oxygen depletion. Environment Canada is conducting a study examining the physical and ecological characteristics of the littoral zone in Lake Winnipeg and continues its spatially- and temporally-rigorous study of thermal stratification and dissolved oxygen in the lake. This includes further investigation into the duration and spatial extent of thermocline development in the north basin during the summer, and the spatial and temporal extent of bottom water dissolved oxygen levels in the north basin in winter. Manitoba Water Stewardship continues to monitor ice-free and under-ice conditions at long-term monitoring stations across the lake including assessment of physical, chemical, and biological parameters. A more thorough understanding of the physical characteristics of Lake Winnipeg will enhance understanding of the structure and function of the lake, the health of its ecosystem, and its response to increases and reductions in nutrient loading.

Gaps in knowledge and information still remain with respect to under-ice conditions, littoral areas of Lake Winnipeg, and the range and variability in water characteristics intra- and inter-annually, as well as spatially through the lake.

Nutrients

Detailed information on nutrients over recent decades has been acquired from monitoring nutrient concentrations in the lake, river inflows, and at the outflow of Lake Winnipeg. This includes information on the sources of nutrients to Lake Winnipeg and their relative contributions (Bourne *et al.* 2002). However, several parts of the nutrient balance have yet to be refined for Lake Winnipeg. Atmospheric loading of nutrients, which is currently estimated using a single deposition rate for nitrogen and phosphorus over the period of 1999 to 2007, deserves further attention. In comparison to atmospheric nutrient deposition rates for other lakes, loading estimates for Lake Winnipeg are considered conservative. It has been estimated that a considerable amount of nitrogen is fixed by cyanobacteria in Lake Winnipeg (9,300 tonnes; Hendzel 2006). However, this single value has been used as an estimate of atmospheric nitrogen fixation for the entire period of record (1999 to 2007). It is unknown how nitrogen fixation in Lake Winnipeg would vary inter-annually given a range of bloom conditions. This requires further investigation. Losses due to de-nitrification have not been quantified and would also help to resolve the nitrogen budget for the lake.

Several parts of the nutrient balance have yet to be refined for Lake Winnipeg.

Knowledge on the role of internal nutrient loading and nutrient resuspension in Lake Winnipeg is limited, with little information on the duration, frequency, magnitude, and extent of these events. The internal loading component is likely to be of considerable importance to the nutrient balance given the potential for nutrient release during hypoxic conditions, and the potential for resuspension of nutrients in the relatively shallow waters of Lake Winnipeg. This is especially true for the shallower south basin. The information will be important for modelling the response of the lake to various nutrient management strategies, as lakes with high internal loading are known to respond more slowly to reductions in external nutrient loads.

The role of flood events on nutrient concentrations and residence times in the lake is not well understood. A more thorough understanding of potential lags from substantial flood-associated nutrient loads is necessary to understand the physical, chemical, and biological nature of the lake and its response to nutrient loading and management strategies. The highly heterogeneous nature of Lake Winnipeg, as well as limited knowledge on its structure and function, complicate this understanding and any efforts to close the nutrient balance. This includes thresholds beyond which there may be irreversible change in the physical, chemical, and biological compartments of the lake.

The role of flood events on nutrient concentrations and storage in the lake, as well as the residence times of nutrients, is not well understood.

Biology

Historical and more recent phytoplankton, zooplankton, benthic invertebrate, and fish information has been collected to characterize the biological community structure of Lake Winnipeg. However, the biological component is perhaps the least understood in terms of trophic structure, food web linkages, and top-down and bottom-up interactions. For instance, algal blooms have become more frequent and intense in Lake Winnipeg, although it is not fully understood how nutrients, light, water temperature, physical processes (water column stability, mixing), and food web structure have affected phytoplankton biomass and species composition in the lake. Nutrient stoichiometry has suggested that phytoplankton in Lake Winnipeg may be either nitrogen-limited or phosphorous-limited, depending on the basin and season. Results from research studies currently underway with respect to algal-nutrient limitation (protein phosphatase, N debt assays, bottle bioassays) will provide further insight into the nutrient status of the phytoplankton over the open water season. Light limitation is also a key factor controlling bloom development in Lake Winnipeg, particularly in the south basin and the narrows where high sediment loads from the Red River may limit primary production. Focussed studies evaluating phytoplankton responses along light and nutrient gradients could provide key information on physical and chemical interactions and how they affect phytoplankton biomass and composition in Lake Winnipeg. Complex deterministic water quality models, such as those currently being developed by Environment Canada, will be one of the other key tools to support lake management strategies to understand bloom dynamics and to reduce the frequency and severity of algal blooms in the lake.

The biological component (of Lake Winnipeg) is perhaps the least understood in terms of trophic structure, food web linkages, and top-down and bottom-up interactions.

Since its introduction to Lake Winnipeg in the early 1990s, the Rainbow Smelt has become a dominant component of the Lake Winnipeg food web, particularly in the north basin of the lake, where fish trawls can be almost entirely comprised of Rainbow Smelt. It is currently unknown what impact the Rainbow Smelt has on the Lake Winnipeg food web. Rainbow Smelt are small planktivorous fish that graze on zooplankton. With such a large biomass of Rainbow Smelt in the north basin, it is unknown how this top-down pressure has affected zooplankton biomass and structure. Reduced grazing pressure from zooplankton may be affecting phytoplankton biomass and composition in the north basin of the lake. Zooplankton has been collected historically and more recently from Lake Winnipeg, but it is currently unknown how they have been affected by lake eutrophication.

Walleye production in the lake has increased dramatically since the 1990s and the Walleye fishery is much more productive than observed historically. It is unknown why production has increased, although it is thought that increases may be related to an increase in nutrients and ultimately an increase in food source for Walleye. However, other factors may be affecting Walleye production in Lake Winnipeg. For instance, the timing of the invasion of Rainbow Smelt was coincident with increases in Walleye production. It is unknown to what extent Rainbow Smelt has affected the Walleye production in the lake and whether Rainbow Smelt now comprises a significant part of the Walleye diet in the north basin. Research is currently underway to address the role of invasive Rainbow Smelt in Lake Winnipeg food web and the impact on the commercial fishery (University of Manitoba).

It is unknown why (Walleye) production has increased, although it is thought that increases may be related to an increase in nutrients and ultimately an increase in food source for Walleye.

Research by Environment Canada and the University of Saskatchewan will also define the spatial-temporal food web relationships for fish and invertebrates in Lake Winnipeg using stable isotope analyses. Food web models, such as the one currently being developed by the University of Toronto, will also be useful in addressing these research questions moving forward.

Algae Toxins

Microcystin has generally remained low in the offshore areas of Lake Winnipeg during the seasonal surveys of the lake. However, concentrations in the nearshore have been high on occasion, and in some instances, have exceeded the recreational water quality guideline. It is not understood why blooms in the nearshore zone of the lake have produced toxins, while large prolific blooms in the offshore have not. Further research will be needed to understand the cyanotoxin dynamics in Lake Winnipeg.

Little work has been done to evaluate the potential for transfer of microcystin to higher trophic levels in the Lake Winnipeg food web. Other studies have shown bioaccumulation in lower trophic levels and thus the potential for transfer to commercially important fish species. It is also currently unknown what species of cyanobacteria are producing microcystin in Lake Winnipeg. While there is some phytoplankton species composition information, a more focussed study would be required to isolate the gene that is responsible for microcystin production to determine which species is producing the toxin.

A more focussed study would be required to isolate the gene that is responsible for microcystin production to determine which species (of cyanobacteria) is producing the toxin.

Currently, there is no systematic investigation of neurotoxins in Lake Winnipeg. Although neurotoxins have not been detected in Lake Winnipeg, they have been found throughout the Lake Winnipeg watershed. Neurotoxins are generally more potent than hepatotoxins and effects of exposure can be much more acute in nature. However, there are currently no water quality guidelines for neurotoxins and due to the unstable nature of these toxins, sampling and laboratory analyses procedures are not readily available.

11.0 CONCLUSION

Eutrophication of Lake Winnipeg has become increasingly pronounced over recent years and is the main concern regarding the health of the lake. Lake conditions from 1999 to 2007 illustrate the eutrophic characteristics of Lake Winnipeg (e.g. elevated nutrient levels, turbid waters, potential for oxygen depletion, increasing occurrence of algal blooms). Periodic research and monitoring on the lake from the 1930s to the 1990s provides some indication as to past conditions in the lake and this is supported by paleolimnological research which indicates deteriorating water quality in Lake Winnipeg since the early 1900s. More rigorous and continuous monitoring on the lake in response to the increase in the intensity and frequency of algal blooms in Lake Winnipeg over the past 10 years has provided information on the chemical, physical, and biological characteristics of Lake Winnipeg, and has highlighted the highly variable nature of water quality in the lake.

Knowledge of Lake Winnipeg has been further enhanced by the focussed research of the recent Lake Winnipeg Basin Initiative. This initiative has provided crucial information on topics such as nutrient dynamics and balance, trophic structure and interactions, responses of biological communities to nutrients, range and nature of physical and hydrodynamic conditions in the lake, nutrient sources, and other stressors to the lake. Recent examinations of change in the lake over the long term have provided new insights into its structure and function. This cumulative knowledge base on Lake Winnipeg will help to facilitate the development of management objectives and management options for the lake.

Environment Canada and Manitoba Water Stewardship are committed to continue working collaboratively with others involved in research on Lake Winnipeg including the Lake Winnipeg Research Consortium, universities and their students, private consultants, and others, to better understand Lake Winnipeg and to monitor long-term changes in its physical, biological, and chemical condition.

In September of 2010, Environment Canada and Manitoba Water Stewardship signed a Memorandum of Understanding Respecting Lake Winnipeg and its Basin. The Memorandum provides a framework for:

- Identification of joint priorities for coordinated scientific activities;
- Coordination of information sharing between federal and provincial programs and activities; and
- Providing a forum for communication, consensus building and coordination of activities.

The Memorandum of Understanding will help to guide future research and monitoring on Lake Winnipeg and its large watershed and will also help to identify and implement the solutions required to reduce nutrient loading and halt eutrophication.

Methodological Appendix

Appendix 1. Reporting on Recent Conditions (1999 to 2007)

Chemical and biological samples collected as a part of the Province of Manitoba's long-term water quality monitoring program from 1999 to 2007 were summarized to describe the temporal variation in physical, chemical, and biological characteristics of Lake Winnipeg. Unless otherwise specified, these data represent the surface and euphotic zone (Appendix 1c). Surface samples were collected immediately below the water surface. The euphotic zone was defined at each station as one per cent of surface irradiance. Samples were depth-integrated. The spatial and seasonal coverage of the water quality record varied over the nine-year period depending on logistics, resources, program requirements, and research objectives. Samples were generally collected during the spring, summer, and fall from a network of 14 long-term stations and up to 50 additional stations in the Lake Winnipeg monitoring network (Figure 2.1). Sampling effort varied from year to year. For instance, during the first year of study in 1999, samples were only collected during the summer. In 2005, when the research vessel *MV Namao* was in dry dock for repairs and was only on the lake in the fall, sampling in spring and summer of 2005 was conducted using smaller vessels. However, the selection of monitoring stations was often restricted by weather conditions and access to the stations. Given this variability in the frequency and timing of sampling, consistency in intra- and inter-annual data from 1999 to 2007 has some limitations. Under-ice sampling was also conducted, but with less intensity than that during the open water season.

Water samples were also collected intensively by Fisheries and Oceans Canada from the network of approximately 65 monitoring stations. Chemical samples were collected during the spring, summer, and fall. These data were used in this report to describe seasonal and spatial variation in physical and chemical characteristics of Lake Winnipeg in 2002, depicting ‘average conditions’. That year was selected to depict 'average conditions' given that precipitation, air temperature, and water levels were near normal in 2002 (Section 4.0). Data were interpolated and plotted geospatially (Appendix 1b).

Other agencies and individuals have also conducted sampling on Lake Winnipeg in recent years. For detail as to sample collection, handling, and analytical methods see Appendix 1c.

Appendix 1a. Data Analyses: 1999 to 2007

Data analyses for 1999 to 2007 were conducted using samples collected by Manitoba Water Stewardship during the open water season (May to October). Descriptive statistics (number of samples, mean, standard error, standard deviation, minimum, maximum) were calculated for the north basin and the south basin, including the narrows. Samples from the narrows were included with those of the south basin due to the low sample frequency in the narrows. Seasonal statistics were calculated for the spring (May to June), summer (July to August), and fall (September to October). Data for a given season were excluded when there were fewer than three samples; but this varied inter-annually and between parameters. Annual statistics were calculated using all samples from any given year. Annual mean was season-weighted in recognition of inter-annual discrepancies in seasonal data. Where large standard errors existed and values lay outside the range typically observed for a given season over the period under consideration, outliers were verified against field and lab records and excluded if deemed erroneous. Extreme values were excluded from the calculation of descriptive statistics for 1999 to 2007. This step was taken to allow for inter-annual and inter-seasonal comparisons of means in the absence of extreme values.

Appendix 1b. GIS mapping

Mapping was completed for physical, chemical, and biological data to display spatial and seasonal variability in Lake Winnipeg. The data were manipulated using ESRI's software ArcInfo 9.3 and interpolated using the advanced surface modelling ESRI extension Geostatistical Analyst. The kriging prediction method was used to produce a surface of predicted values for each parameter on the lake, assuming an unknown constant mean. Once the interpolation was derived, a manual classification was used to display the results. Kriging is part of a group of geo-statistical techniques that are used to interpolate the value of a random field at an unobserved location from observations at nearby locations. It weighs the surrounding measured values to derive a prediction for each location. However, the weights are based not only on the distance between the measured points and the prediction location, but also on the overall spatial arrangement among the measured points. This method is built on the assumption of spatial auto-correlation. Once the surfaces were derived, they were reclassified manually to best illustrate trends specific to each parameter.

Appendix 1c. Inter-agency Sampling and Analysis

(MWS - Manitoba Water Stewardship; EC - Environment Canada; DFO - Fisheries and Oceans Canada).

Physical	Agency	Sample Depth	Sampling Device	Filtration, Preservation	Analysis
Dissolved Oxygen	MWS	Surface: 0 m	Weighted DO sampler	Reagents added on board	Winkler titration of iodine with thiosulphate. APHA 4500 O-C.
		Euphotic: integrated over the photic zone	Integrated tube sampler		
		Bottom: 0.5 m from lake bottom	Kemmerer		
	EC	Realtime depth profiles at 1 m intervals from surface to bottom (2006-2007)	YSI 6000 series sonde with optical DO sensor, manually lowered	n/a	Manual recording of concentrations and % saturation
		Realtime continuous depth profiling from surface to bottom, data acquired on the downcast (2008-2009)	Seabird SBE43 DO sensor on automated CTD profiler		Raw high frequency data processed with SeaTerm software to 10 and 100 cm depth intervals
	DFO	Various	Weighted bottle, Van Dorn or pumped from depth	For Winkler method, reagents added immediately and titrated on board	Winkler titration of iodine with thiosulphate. Some direct measurement with Orion sensors.
TSS	MWS	Surface: 0 m	Weighted Nalgene bottle	Whatman GF/C	APHA 2540B
		Euphotic: integrated over the photic zone	Integrated tube sampler		
		Bottom: 0.5 m from lake bottom	Kemmerer		
	DFO	0 to 0.5 m	Weighted 2 litre polyethylene bottle	Whatman GF/C	Filter tare and filter + particulate dried at 104 ^{0}C and weighed to constant weight (<0.04 mg difference on sequential weighings)
Turbidity	MWS	Surface: 0 m	Weighted Nalgene bottle	Lab measurement	HACH Model 2100 AN Ratio Turbidimeter. Light scattered at 90°
		Euphotic: integrated over the photic zone	Integrated tube sampler		
		Bottom: 0.5 m from lake bottom	Kemmerer		
	EC	Profile	Turbidity sensor: Turner Cyclops calibrated to report NTU (0.05 NTU is the minimum detection limit). Light scattered at 90°.	*In Situ* measurement	Calibrated at Environment Canada, NWRI Burlington
	DFO	Profiles	Branker CTD	*In Situ* measurement	Branker CTD calibrated against TSS analysis

Continued...

Appendix 1c. Inter-agency Sampling and Analysis (Continued)

	Agency	Sample Depth	Sampling Device	Filtration, Preservation	Analysis
PAR	EC	Profile	PAR sensor (surface reference): QCR-2200 cosine-corrected irradiance PAR sensor (scalar collector): QSP-2300 calibrated to report quanta/cm^2/sec or µE/cm^2/sec	*In Situ* measurement	Calibrated at EC NWRI Burlington
	DFO	Profile	DFO sensor on EC Seabird profiler	*In Situ* measurement	Collection of PAR data (Licor sensor) on all CTD casts with simultaneous correction for incident radiation with deck PAR sensor
Nutrients					
TN	MWS	Surface: 0 m Euphotic: integrated over the photic zone Bottom: 0.5 m from lake bottom	Weighted Nalgene bottle Integrated tube sampler Kemmerer	H_2SO_4 preserved	Calculated as the sum of TKN and NO_3-NO_2-N. TKN measured colorimetrically on acid digested samples using an auto-analyzer. See DIN for a description of NO_3-NO_2-N analyical methods.
	DFO	Various depths	Weighted bottle, Van Dorn or pumped from depth.	Filtered on board all cruises 1969 to present	TN is a calculated value from the sum of TDN (total dissolved N) and Particulate N where dissolved and particulate is determined by the pore size of a Whatman GF/C. TDN is measured quantified as NH_4 following UV digestion (convert organic N to inorganic N) and conversion of inorganic N produced by digestion (NH_4, NO_3, NO_2) to NH_4 (reduction using Zn and acid matrix)
DIN	MWS	Surface: 0 m Euphotic: integrated over the photic zone Bottom: 0.5 m from lake bottom	Weighted Nalgene bottle Integrated tube sampler Kemmerer	H_2SO_4 preserved	Calculated from the sum of NH_3 and NO_3-NO_2-N. Both NO_3-NO_2-N and NH_3 were measured colorimetrically using an auto-analyzer.
	DFO	n/a	n/a	n/a	Not measured but the sum of measured NO_3, NO_2 and NH_4. All measured colourimetrically using conventional auto-analyzer technology

Continued...

Appendix 1c. Inter-agency Sampling and Analysis (Continued)

	Agency	Sample Depth	Sampling Device	Filtration, Preservation	Analysis
TP	MWS	Surface: 0 m Euphotic: integrated over the photic zone Bottom: 0.5 m from lake bottom	Weighted Nalgene bottle Integrated tube sampler Kemmerer	H_2SO_4 preserved	APHA 4500-P
	DFO	n/a	n/a	n/a	Same as for TN - a calculated value from the sum of measured TDP (again UV digestion of organic P) and particulate P (on a GF/C filter)
DP	MWS	Surface: 0 m Euphotic: integrated over the photic zone Bottom: 0.5 m from lake bottom	Weighted Nalgene bottle Integrated tube sampler Kemmerer	H_2SO_4 preserved	APHA 4500-P
	DFO	Various Depths	Weighted bottle, Van Dorn, or pumped from depth	Filtered on board	UV digestion to yield TDP. Organic P converted to PO_4-P then same method as for SRP
PP	MWS	Surface: 0 m Euphotic: integrated over the photic zone Bottom: 0.5 m from lake bottom	Weighted Nalgene bottle Integrated tube sampler Kemmerer	H_2SO_4 preserved	Calculated as the difference between TP and DP
	DFO	Various Depths	Weighted bottle, Van Dorn, or pumped from depth	Filtered on board	Particulates retained by a GF/C filter. Filter combusted at 500C. Ash acid hydrolyzed to PO_4-P. Same detection as for SRP
Biological					
Phyto-plankton	MWS	Surface: 0 m Euphotic: integrated over the photic zone	Weighted Nalgene bottle Integrated tube sampler	Unfiltered, 2% Lugol's solution	Inverted phase contrast compound microscope. Identified to lowest level possible.
	Kling	ALL years: surface (0-0.5m consistent with early DFO samples in whole lake 1969 survey) Note[1]	Van Dorn or Nisken bottles	Unfiltered, 2% Lugol's solution	Inverted Microscope method founded by Utermohl (1958) modified by Nauwerk (1963) and considering also Lund *et al.* (1958) and Findlay *et al.* (1998) Inverted Microscope method founded by Utermohl (1958) modified by Nauwerk (1963) and considering also Lund *et al.* (1958) and Findlay *et al.* (1998) Note[2]

Continued...

Appendix 1c. Inter-agency Sampling and Analysis (Continued)

	Agency	Sample Depth	Sampling Device	Filtration, Preservation	Analysis
Benthic Invert-ebrates	MWS	Bottom sediments	Standard Ponar dredge (9"x9")	Sample washed through a 0.4 mm nylon net; On board preservation with 70% ethanol	Samples sieved (250µm), sorted, and picked using a stereoscopic and/or compound microscope. Identification to family level.
	Hann	Bottom sediments	1 Ekman dredge (6" x 6") sample at ~50 of the routine sampling stations (spring, summer, autumn) throughout the lake	On board preservation with full strength formalin; within a month, transfer to 70% ethanol in the University of Manitoba laboratory	Sorting, picking of specimens using dissecting microscope (10X), transfer to labeled vials, preserved in 70% ethanol. Identification to family level.
Fish	MWS/ DFO	Trawl towed along side the ship randomly at one of three depths: surface, midwater, or deepwater (due to differences in depths of different areas of the lake, deepwater trawls not conducted in the south basin)	Beam trawl (3m x 3m) 10.8-m, mesh sized graduated from 76.2 mm stretch measure at the mouth to 19.1 mm attached to the cod end.	Fish sorted by species and frozen in labelled bags for further processing in the lab (incidental catch of conspicuously larger fish (roughly >200 mm) identified, measured and returned to the lake)	Fish thawed, identification confirmed, and fish sorted into size groups (age-0, small, medium, large) to ensure representative sample measured. For each species and size group from a trawl sample, fork length (FL, mm) and weight (Wt, g) of the first 100 fish of a species individually measured and weighed, remaining fish counted and bulk weighed.

Note[1] Some sites/years augmented with depth profiles at one metre intervals. Profile data give measure of variability over the water column. Other samples such as integrated euphotic zone have also been compared to discrete depth data in some cases.

Note[2] A brief description of methods: the inverted microscope standard method which was used for Lake Winnipeg included sediment between two millilitre (mL) sub-samples of whole water lake samples previously killed with 1 mL of Lugol's solution per 100 mL of sample which were 20x pre-concentrated by sedimentation or 2-10mL of whole lake water depending on the density of organisms. Analysis consisted of enumerating the large <20 µm LD and usually less numerous taxa at low magnification (235x) over one-half of the chamber. More numerous and/or small taxa were enumerated at high magnification 625x or 925x.

At least 10 or more individual specific taxa were measured and if fewer organisms were encountered all were measured. Biomass expressed as µg/L or mg/m^3 was calculated from the linear measurements and appropriate volume formulae (eg., excluding chrysophyte lorica, gelatinous mucilage, long extensions not containing protoplasm) based on Rott (1981). In most cases under general analysis 200 to 400 organisms were enumerated and consideration was given to procedures given in Lund et al. (1958). A list of keys used in analysis is available on request. A species list of phytoplankton found in Lake Winnipeg from 1924 to 2000 is currently being revised and will be in the Fisheries and Oceans Canada biodiversity database.

In the late 1980s and early 1990s, scanning electron microscopy was used to identify the major planktonic diatoms in the lake because this was deemed important for their use as indicator organisms for paleo studies and also as indicators of current environmental conditions. Stephanodiscus agassizensis (Hakanson and Kling 1989), described initially from the Red River and south basin of the lake, has recently become widespread and numerous in the plankton. Several chain-forming centric diatoms had formerly been called Melosira: 1) Aulacoseira islandica, a *common component of the under-ice and spring bloom community, 2) Aulacoseira granulata, a warm eutrophic riverine species, and 3) Aulacoseira ambigua meso-eutrophic low-light lake species. Stephanodiscus binderanus, a chain-forming diatom which looks similar to Aulacoseira but belongs to the genus Stephanodiscus, has been present in the lake for a long time but not abundant, though recently it has been forming open water blooms. This diatom is important as it is an indicator of eutrophication. It became very abundant in inshore southern regions of Lake Michigan during increased eutrophication and also during the highly eutrophic areas of Lake Ontario and Lake Erie. There is also a true freshwater species of Melosira in Lake Winnipeg which is usually found in the areas of higher conductivity. The centric diatom Cyclostephanos dubius is also now an important component in the plankton near the Winnipeg River and is a signature of this river's water plume. This species is not widely spread in either the north or south basin but it is very common now in Lake of the Woods.*

When it is possible to identify major taxa in a lake to species within a reasonable time frame using this standard inverted light microscopic method and available literature, it is necessary to do so. There are many possible sources of error to be considered such as sampling methods (from the lake to the bottle), preservation and preservation vessel, microscope method, enumeration of major species vs genera, years of experience of analysts, and literature used in identification. The algal species are important in biodiversity and food web interactions, and water quality issues such as toxin production, as many problems related to water management are caused by the presence of individual algal species or mass development (e.g., fish kills by Prymensium parvum in east and west coastal high conductivity reservoirs). These cannot be resolved by the algal classes or total biomass and chlorophyll. The analysis of species is especially important when it involves blue green algae and toxin production, taste and odour problems, the clogging of fisher's nets or water intake pipes, and determining a healthy food web structure in a lake.

Appendix 2. Precipitation and Evaporation

Fisheries and Oceans Canada and Environment Canada compiled precipitation data from Meteorological Service of Canada climate stations in the immediate vicinity of Lake Winnipeg to describe the water balance, to simulate the nutrient balance of Lake Winnipeg (1907 to 2006; Fisheries and Oceans Canada; Stainton; McCullough *et al.*), to model eutrophication and circulation of the lake (2002 to 2006; Environment Canada; Yerubandi; Zhang *et al.*), and to estimate evaporation. Total annual precipitation over the north and south basins of the lake was calculated by averaging data from stations located around the perimeter of each of the basins (Table A1). Precipitation over the whole lake was calculated as the basin-area-weighted average of the north and south basins (area north of narrows 19,768 km^2, south of narrows 3,982 km^2, as per G. McCullough, Fisheries and Oceans Canada, unpublished data). Mean monthly evaporation from the lake was derived from relative monthly evaporation off Dauphin Lake (from Morton 1985), which is located at approximately the same latitude and in the vicinity of Lake Winnipeg. Evaporation was then corrected from Dauphin Lake total annual evaporation of 665 mm to Lake Winnipeg total annual evaporation of 689 mm (estimated as the mean water balance residual from 1975 to 2004, G. McCullough, pers. comm.) and area-weighted to determine monthly evaporation for each of the north and south basins (W. Zhang, pers. comm.).

Table A1. *Environment Canada climate stations used in calculating precipitation on Lake Winnipeg from 1999 to 2007. (http://carnak.wxe.sk.ec.gc.ca)*

Station Name	Station ID	Latitude (N)	Longitude (W)
The Pas A	5052880	53 58	101 06
Indian Bay	5031320	49 37	95 12
Norway House	506B047	53 57	97 51
Norway House (A)	5062045	53 58	97 51
Grand Rapids	5031111	53 09	99 17
George Island (Aut)	5030984	52 49	97 37
Berens River CS	5030203	52 22	97 01
Pine Dock	5032163	51 38	96 48
Bissett	5030282	51 02	95 42
Vogar	5043020	50 55	98 45
Arborg	5030080	50 56	97 05
Gimli	5031039	50 38	97 01
Pine Falls	5032164	50 34	96 13
Great Falls	5031200	50 28	96 00
Great Falls Climate	5031201	50 31	95 59

Appendix 3. Tributary Discharges

Mean monthly discharges to and from Lake Winnipeg from 1999 to 2007 were determined for large rivers flowing into Lake Winnipeg, and for unmetered streams draining to Lake Winnipeg (as per G. McCullough). Mean monthly discharges to and from the lake were calculated from daily or monthly data at Water Survey of Canada hydrometric stations near the outlet of the Saskatchewan, Dauphin (Fairford), Red-Assiniboine, and Winnipeg rivers, and at the mouth of the Nelson River (east and west channel). Unmetered flows were estimated as the flow from that portion of the terrestrial watershed of the major tributaries that was unmetered, at a rate of runoff per unit area assumed to be the same as that for the metered portion of the watershed (Table A2).

Table A2. *Watershed areas for tributaries to Lake Winnipeg. River basin area was defined as the area upstream from Water Survey of Canada hydrometric gauging stations at the downstream end of each river. Terrestrial watershed sub-area was the total drainage area for a river, accounting for watershed areas that were not included in the Water Survey of Canada measure. These measures of watershed and lake basin area were applied in developing a water balance of Lake Winnipeg (McCullough, unpublished data) and have been incorporated into nutrient balance (McCullough et al. submitted, Zhang and Yerubandi, 2009) and circulation (Yerubandi and Zhang 2009) models for Lake Winnipeg.*

River	Terrestrial watershed sub-area km^2	River basin area km^2	Watershed area factor*
Manigotagan	31,692	1,830	17.318
Winnipeg	163,912	134,050	1.223
Saskatchewan	364,000	364,000	1.000
Dauphin	101,603	82,400	1.233
Red	291,794	287,000	1.050
Terrestrial drainage area	953,000	869,280	
North basin	20,519		
South basin (and narrows)	3,981		
Lake area	24,500		
Total drainage area (at outlet of Lake Winnipeg)	977,500		
Total drainage area (*at metering stations at Jenpeg Dam/Sea River)	980,000		

Appendix 4. Water Level

Water levels for Lake Winnipeg were calculated using data from Water Survey of Canada hydrometric stations located in the north and south basins, with missing values in-filled through interpolation (as per McCullough, Table A3). Mean monthly water levels were calculated as the five-day daily averages at the end of each month for each station in order to reduce the effects of secche due to wind on lake levels (McCullough, pers. comm.). At least one station at the north end and one station at the south end of each basin was selected for inclusion in mean water levels for each basin.

Table A3. *Water Survey of Canada hydrometric stations used to calculate 1999 to 2007 mean monthly water levels and mean monthly discharges for Lake Winnipeg.*

Station Name Lake Winnipeg at:	Station ID	Latitude (N)	Longitude (W)
Montreal Point	05RF001	53.6250	-97.844440
Mission Point	05SG001	53.189440	-99.211940
Berens River	05RD005	52.353310	-97.022140
Anama Bay	05SF001	51.996670	-98.057220
George Island	05RE003	52.818420	-97.619560
Pine Dock	05SD001	51.641670	-96.795830
Victoria Beach	05SA003	50.695110	-96.562170
Gimli	05SB006	50.6310	-96.981830
Winnipeg Beach	05SB001	50.506110	-96.965280

Appendix 5. Water Residence Time

The morphologic and hydrological conditions of Lake Winnipeg were examined for the period of 2002 to 2006 to develop the eutrophication model for Lake Winnipeg; this examination was extended to include all years from 1999 to 2007 for the purposes of this report. Eutrophication modelling included the calculation of hydraulic retention time for Lake Winnipeg. The morphology of the north and south basins were characterized according to Brunskill *et al.* (1980, Table A4) and the water balance was modelled, including the exchange of flows between basins, inflows, outflows, precipitation, and evaporation. Dominant in-flowing tributaries included in the model were the Winnipeg and Red rivers to the south basin, and the Saskatchewan and Dauphin rivers to the north basin, with average monthly flow of 1,562 m^3/s between basins.

Table A4. *Morphometric parameters for Lake Winnipeg as per Brunskill et al. South basin defined as "south basin and narrows" for the purposes of water balance modelling. (Yerubandi and Zhang, pers.comm.)*

Parameters	South basin and narrows	North basin	Total
Surface Area (m^2)	6.230×10^{9}	1.752×10^{10}	2.375×10^{10}
Volume (m^3)	5.160×10^{10}	2.324×10^{11}	2.840×10^{11}
Mean Depth (m)	8.3	13.3	12

Appendix 6. Procedures Used to Predict the Thermal Regime of Lake Winnipeg Under a Changing Climate

The procedures described below were first reported by McCullough (2005) who estimated future water temperature, and ice break-up and freeze-up periods in Lake Winnipeg from air temperature predictions using Version 2 of the Canadian Global Climate Model (CGCM2). Here, air temperature is predicted using Version 3 of the CGCM (CGCM3). Consequently, some results in this report differ slightly from results reported by McCullough (2005). In particular, CGCM3 predicts greater autumn warming than CGCM2, with the result that by mid-21st century, freeze-up is predicted to occur a week later than reported by McCullough (2005).

Surface water temperature data recorded by Fisheries and Oceans Canada staff on six whole-lake cruises in 1969 were used to develop a statistical model predicting water temperature in the north and south basins of Lake Winnipeg as functions of air temperature. Specifically, polynomial regressions were calculated predicting monthly mean temperature in the upper metre of the water column of each basin, as functions of current and antecedent monthly mean air temperature recorded at a centrally-located meteorological station at Berens River. Model output was validated using an independent data set, i.e. hourly surface temperature data recorded at instrumented moorings deployed by Environment Canada in the north and south basins of the lake from 1999 to 2006 (Figure A1). The model predicts monthly June to October monthly mean surface water temperatures recorded at these moorings with root mean square errors of 0.7°C (n = 8) and 0.9°C (n = 25) in the north and south basins respectively.

The air temperature record at Berens River was also used to develop statistical models of the timing of the spring ice-melt/break-up process, and the fall freeze-up process in each basin of the lake. Both processes are spread over several weeks each year. The first autumn observation of ice in shallow bays, for instance, may be followed by a warming period without further growth, or even with retreat of the ice cover, or conversely, may be followed by a cold, calm period ideal for further ice development. The record of ice observations by satellite allows identification, in many years, of the dates when the ice cover has developed (in autumn) or decayed (in spring) until it covers roughly one-half of the area of each basin. These mid-process dates tend to be easier to define (if satellite data is used) and better correlated with antecedent air temperature than either the beginning or the end of each process. The 40-day moving average air temperature provides a robust predictor of these mid-break-up and mid-freeze-up dates.

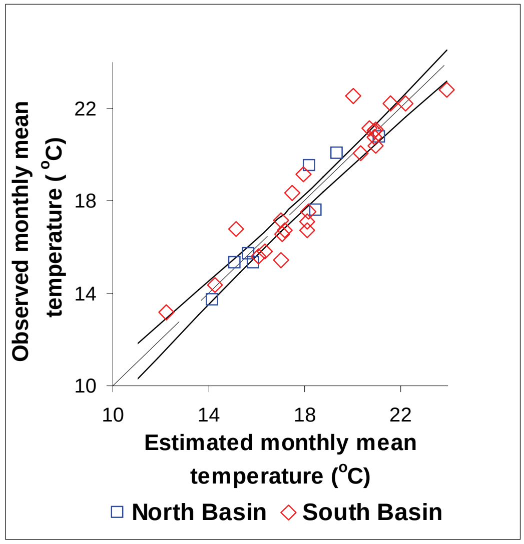


Figure A1. *Comparison of modelled with observed monthly mean surface water temperature, where observed monthly means were calculated from the 10-minute interval record at moorings in the north and south basins. Solid lines indicate 95 % confidence limits for the mean prediction; dashed line indicates target 1:1 prediction of observed temperature. Model-building data were obtained from ship-board measurements made on six whole-lake cruises in 1969. Validation data are from the period 1999 to 2006.*

The day of the year when the antecedent 40-day moving average rises to +5.0°C (the "warming index" in Figure A2) is very highly correlated with mid-break-up as observed in the 1987 to 2009 satellite record (r^2 = 0.72 and 0.49 in the north and south basins respectively, $P < 0.000$ and n = 19 in both cases). Because cloud cover frequently obscured the lake in autumn, the satellite record of freeze-up is too sparse to provide a basis for a similar analysis. Consequently, the Environment Canada record of freeze-up observations in the south basin is substituted here to develop a model of freeze-up as a function of antecedent air temperature. Specifically, the median date between the first observation of ice and the last observation of open water is treated as an index of the mid-freeze-up date.

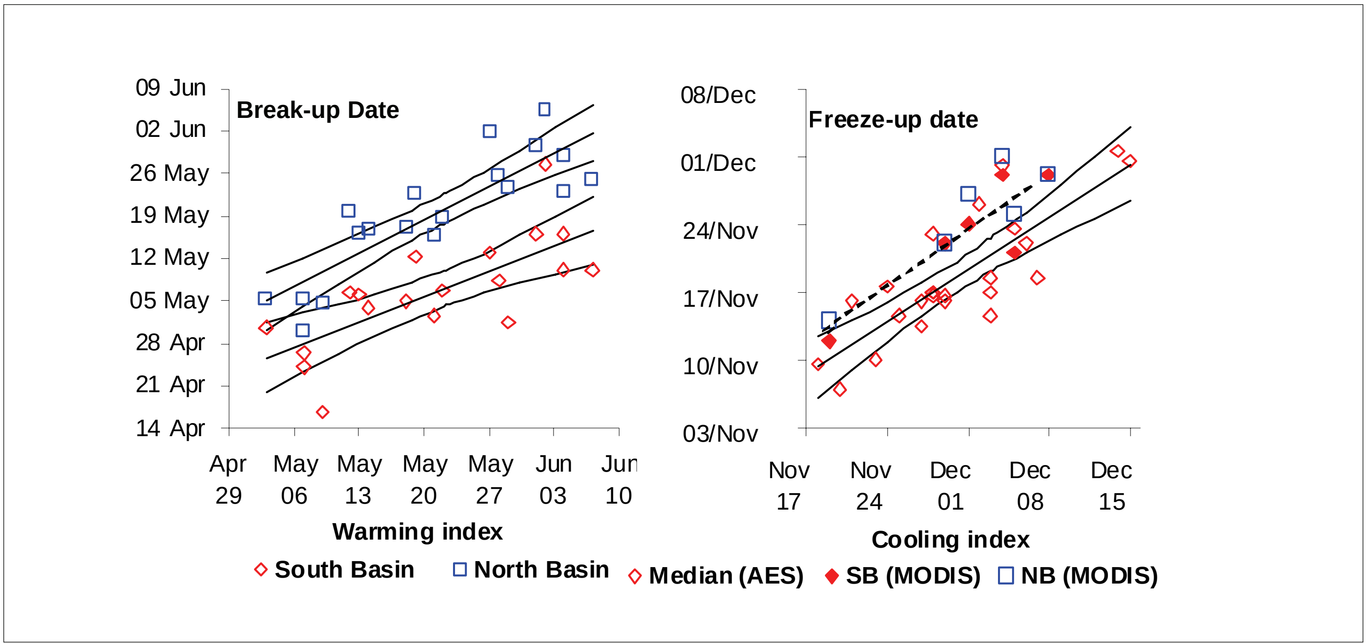


***Figure A2**. Mid-break-up and mid-freeze-up dates plotted as functions of air temperature indices. Mid-break-up dates (left panel) were determined from AVHRR and MODIS satellite data (1987 to 2009). Mid-freeze-up dates (right panel) were determined from the Atmospheric Environment Service weather observer's record ("Median AES", 1966 to 1991 data for the south basin only) or from MODIS satellite data ["SB (MODIS)" and "NB (MODIS)"]. Warming and cooling indices were calculated from antecedent 40-day moving average air temperature, as described in the text. Straight lines mark best-fit regressions; curves indicate 95 % confidence limits for the mean prediction. A dashed line shows the best fit regression predicting mid-freeze-up in the south basin as determined by analysis of MODIS data, for comparison with the best fit regression using the weather observer's record.*

From the few satellite-derived records available, it appears that this median date between first ice and last open water precedes the satellite-derived mid-freeze-up date by only a few days (Figure A2). It is very highly correlated with the day of the year when the antecedent 40-day moving average falls to -5.0°C (the "cooling index" in Figure A2; $r^2 = 0.66$, P <0.000, n = 24). Because the mid-freeze-up date in the north basin followed that in the south basin by only 0 to 4 days, it was not considered necessary to develop an independent model for freeze-up in the north basin. Although both relationships with the 40-day moving average air temperature are very highly significant, they are only very rough predictors of either process in any given year. Nonetheless, both are useful predictors of the mean response. The 95 % confidence interval for the mean prediction of median break-up in the north basin is two to three and one-half days, and in the south basin, three to five days (Figure A2). For median freeze-up in the south basin, the 95 % confidence interval for prediction of the mean is roughly two days (Figure A2). Consequently, only mean predictions of mid- and late 21st century water temperature, calculated from multiple climate model runs over 20-year periods, will be discussed below.

To use these relationships for predicting potential climate-driven changes to the thermal regime of Lake Winnipeg over the next century, local air temperature series predicted by CGCM3 were acquired from the Canadian Centre for Climate Modeling and Analysis (at http://www.cccma.ec.gc.ca/models/cgcm3.shtml). The model is global, with grid cells 3.7° longitude by 3.7° latitude. Isothermals in the Lake Winnipeg region generally follow a northwestern-to-southeastern diagonal. Consequently, this analysis employs the mean temperature predicted for four grid cells crossing Lake Winnipeg and following that diagonal (Table A5, on the following page). The mean location of these grid cells is about 30 km southwest of Berens River, near the geographic centre of Lake Winnipeg.

Table A5. *Identifiers and locations of four CGCM3 grid cells averaged to represent air temperature over central Lake Winnipeg.*

x	y	Long.	Lat	x	y	Long.	Lat.	x	y	Long.	Lat.
70	39	101.25	53.81	71	39	95.5	53.81				
				71	38	97.5	50.1	72	38	93.75	50.1

The CGCM3 predicts climate effects of potential 21st century changes in atmospheric concentrations of both greenhouse gases and aerosol given various forcing emission scenarios developed by the Intergovernmental Panel on Climate Change. Three scenarios which predict a broad range of future climates are modelled in this study. In order of increasing greenhouse gas concentrations developed over the next century, they are referred to as B1, A1B and A2. The three scenarios describe evolution of greenhouse gas concentrations and aerosol loadings in response to varying population growth, economic development, and technological change over the next century. A description of Intergovernmental Panel on Climate Change scenarios can be viewed at http://www.ipcc.ch/pdf/special-reports/spm/sres-en.pdf.

Modelled daily mean temperatures were downloaded for five model-runs each for three 20-year periods: 1970 to 1992, 2040 to 2059, and 2080 to 2099. (For the historic period, 20 years were selected from 1970 to 1992 i.e. excluding the three years 1976, 1980, and 1981 which are incomplete in the observed historic record of air temperatures used to model water temperature and ice processes as described above). In order to correct for any bias in the modelled data, modelled daily temperatures for the historic period (1970 to 1992) were subtracted from modelled daily temperatures for two 21st century periods (2040 to 2059, and 2080 to 2099), then the difference was added to the observed daily temperatures for the historic period to create sets of 20-year runs of unbiased daily mean temperatures for the two 21st century periods. These adjusted, unbiased estimates of mid- and late 21st century air temperature over Lake Winnipeg were then used to model 21st century open water season monthly mean surface water temperatures, and break-up and freeze-up dates for the two major basins of the lake, using the relationships described above.

Appendix 7. Nutrient Loading

7.1 Methods

The study period for the description of nutrient loading in this report covered the period 1994 through 2007. Nutrient loads were calculated for major and minor tributaries to Lake Winnipeg including the Red, Winnipeg, Saskatchewan, Dauphin, Fisher, Icelandic, and Brokenhead rivers. Nutrient loads were also calculated for main tributaries to the Red and Assiniboine rivers. Where data were available, nutrient loads were calculated at the most downstream station as well as at any transboundary stations between Manitoba and upstream jurisdictions. Water quality data were obtained from Manitoba Water Stewardship's long-term water quality monitoring network and Environment Canada's long-term water quality monitoring network, and stream flow data were collected through the Canada-Manitoba Hydrometric Agreement (distributed by Water Survey of Canada 2010, Table A6 on the following page). For rivers monitored by Manitoba Water Stewardship, a change in the analytical laboratory took place in April of 2001 and unfortunately few data are available for direct comparison between laboratories although both laboratories were accredited by the Canadian Association for Laboratory Accreditation and use standard methods. In some instances, water quality and stream flow stations are monitored in partnership with Manitoba Hydro or the East Interlake Conservation District. While in general, methods of calculating nutrient loads are consistent with those described in Bourne *et al.* (2002), in some cases, methods of analysis and data sources have been updated. Therefore, some annual estimates of total nitrogen and total phosphorous loads for the period 1994 through 2001 differ as compared to Bourne *et al.* (2002).

For rivers with routine monthly water quality monitoring (Red, Assiniboine, Souris, Qu'Appelle, and Pembina rivers, Winnipeg River at Point du Bois, Saskatchewan River above the Carrot River), nutrient loads were calculated for each month by multiplying mean monthly flow by the measured nutrient concentration. Monthly nutrient loads were summed to determine annual nutrient loads.

Missing data points were estimated from the previous and subsequent month when only one or two consecutive data points were missing. When more than three months of data were missing, mean monthly nutrient concentrations (January, February, March, etc.) were calculated from the entire data set and used to replace missing data. Values below the detection limit were replaced with a value equal to the detection limit.
For rivers with routine quarterly water quality monitoring (La Salle, Seine, Roseau, Rat, and Brokenhead), nutrient loads were calculated for each year by multiplying mean annual nutrient concentration by mean annual flow. Where only four water quality samples were collected per year (January, April, July, and October), nutrient loads can be significantly underestimated or overestimated, depending in particular on the timing of sampling during spring freshet (see Appendix 7.2). Quarterly sampling can also miss important summer rainfall events that may flush nutrients into rivers and streams.

The Seine River Diversion diverts flows from the Seine River near Ste. Anne straight west to the Red River south of the station at St. Norbert. Water quality monitoring is conducted on the Seine River downstream from the diversion near Winnipeg and the stream flow for calculating loads was obtained from the Seine River near Prairie Grove; this does not include flows diverted to the Red River as these are accounted for at St. Norbert. On average, flows in the Seine River Diversion near its confluence with the Red River are 57 % of the flow in the Seine River near the city of Winnipeg.

For the Dauphin River, water quality monitoring was conducted by Environment Canada until the end of 1996. Water quality monitoring was re-initiated by Manitoba Water Stewardship in 2004. For years with water quality data, nutrient loads were calculated each year by multiplying mean annual nutrient concentration by mean annual flow. The relationship between nutrient loads and flow for periods with water quality data was used to estimate nutrient loads for the period 1997 through 2003 (Total nitrogen $r^2 = 0.97$, $p<0.0001$; Total phosphorus $r^2 = 0.72$, $p = 0.0002$).

For the Fisher and Icelandic rivers, water quality monitoring was initiated by Manitoba Water Stewardship in 2006. Average total nitrogen and total phosphorous concentrations were calculated with data from 2006 to 2008 and were used along with annual average flows to estimate nutrient loading to Lake Winnipeg for the period 1994 through 2007.

Table A6. *Water quality and stream flow stations. (all stream flow data collected through the Canada-Manitoba Hydrometric Agreement and/or estimated by Manitoba Water Stewardship)*
(Manitoba Water Stewardship - MWS; Environment Canada - EC)

River	Water Quality Data		Stream Flow Data
	Monitoring Agency	**Station**	**Station**
Red River			
At Emerson	EC	MA05OC0001	05OC001
At St. Norbert	MWS	MB05OCS004	05OC012 + 05OE001
At Selkirk	MWS	MB05OJS074	05OJ010
Red River Tributaries			
LaSalle River	MWS	MB05OCS026/ MB05OGS039	05OG001
Roseau River	MWS	MB05ODS032	05OD001
Rat River	MWS	MB05OES026	05OE001
Pembina River	EC	MA05OB0001	05OB007
Seine River	MWS	MB05OHS003	05OH009
Assinboine River			
At Kamsack	EC	SA05MD0002	05MD004
At Brandon	MWS	MB05MHS021	05MH013
At Treesbank	MWS	MB05MHS006	Estimated from 05MH013, 05MH005 and 05NG001
At Headingly	MWS and EC	MB05MJS053, MA05MJ0006	05MJ001
Assiniboine River Tributaries			
Qu'Appelle River	EC	SA05JM0014	05JM001
Souris River, Westhope	EC	US05NF0001	05NF012
Souris River, Treesbank	MWS	MB05NGS003	05NG001
Other Lake Winnipeg Tributaries			
Brokenhead River	MWS	MB05SAS038	05SA002
Dauphin River	EC(until December 1996), MWS starting in 2004	MA05LM0005, MB05LMS002/03	05LM006
Fisher River	MWS in Partnership with East Interlake Conservation District	MB05SDS004	05SD003
Icelandic River	MWS in Partnership with East Interlake Conservation District	MB05SCS017	05SC002
Saskatchewan River above Carrot River	EC	MA05KH0001	05KJ001 - 05KH007
Saskatchewan River at Grand Rapids	EC(until December 1996), MWS in Partnership with Manitoba Hydro	MA05SH0001, MB05SHS015	05KL001
Winnipeg River at Point du Bois	EC	MA05PF0022	05PF063
Winnipeg River at Pine Falls	MWS in Partnership with Manitoba Hydro	MB05PFS069	05PF069
Lake Winnipeg East Side rivers	Estimated by MWS	Estimated with MA05PFO022	Estimated
Nelson River	MWS in Partnership with Manitoba Hydro	MB05UBS002, MB05UDS014	05UB001, 05UB009

For the Winnipeg River at Pine Falls and the Saskatchewan River at Grand Rapids (post 1996), water quality sampling was initiated in May 2001 through a partnership between Manitoba Hydro and Manitoba Water Stewardship. Sample collection was sporadic with an average of 10 (Saskatchewan) and 11 (Winnipeg) samples collected per year. Monthly mean nutrient concentrations were calculated for all available months and loads were calculated by multiplying by mean monthly flow. Monthly nutrient loads were summed to determine annual nutrient loads. As described above, missing data points were estimated from the previous and the subsequent month when only one or two consecutive data points were missing and were estimated based on monthly means calculated from the entire data set when more than three months of data were missing.

For the years with water quality and stream flow data from the Saskatchewan River at Grand Rapids (pre-1997 and 2001 onward), there was an expected relationship between total nitrogen and phosphorous loads and flow (Total nitrogen $r^2 = 0.71$, $p<0.0001$; Total phosphorus $r^2 = 0.82$, $p<0.0001$). The relationships were used to predict nutrient loads during years where water quality data were missing but stream flow data were available.

For the Winnipeg River at Pine Falls, no water quality data were available prior to 2001. However, where data existed for the period 2002 through 2007, there was a strong relationship between nutrient loads on the Winnipeg River at Pine Falls and the Winnipeg River at Point du Bois (2002 through 2007) (Total nitrogen $r^2 = 0.97$, $p = 0.0004$; Total phosphorus $r^2 = 0.92$, $p = 0.0026$). Therefore, total nitrogen and total phosphorous loads from the Winnipeg River at Point du Bois were used to estimate loads at Pine Falls for 1994 through 2001. McCullough (2001) estimated that nutrient loads from the Winnipeg River were underestimated by about 30 % due to use of data from the Point du Bois water quality station. When estimated data for 1994 through 2001 and actual data for 2002 through 2007 for Pine Falls, and actual data for Point du Bois for 1994 through 2007 were compared, total phosphorous and total nitrogen loads were 24 % and 35 % higher, respectively at Pine Falls as compared to Point du Bois.

Historical stream flows for the Red River were as distributed by Water Survey of Canada (2010), or for the period 1913 to 1961 as estimated by the Surface Water Management Section, Manitoba Water Stewardship (2010). Historical Winnipeg River stream flows were as distributed by Water Survey of Canada (2010). Where available, historical stream flows from the Saskatchewan River at Grand Rapids were as distributed by Water Survey of Canada (2010). Missing years (all prior to construction of the Grand Rapids Generating Station) were estimated with the relationship between stream flows on the Saskatchewan River at The Pas and at Grand Rapids for years with data (23 years).

The rivers on the east side of Lake Winnipeg have not been monitored routinely for water quality and many have not been monitored for stream flow since 1996. However, information from years when stream flow was gauged (1980 to 1996) indicates that the rivers on the east side of Lake Winnipeg such as the Black, Manigotogan, Bloodvein, Pigeon, Berens, and Poplar rivers contribute significantly (11 % on average) to flow in Lake Winnipeg. A relationship between the cumulative, annual average flow from the Black, Manigotogan, Bloodvein, Pigeon, Berens, and Poplar rivers and the annual average flow in the Winnipeg River between 1980 and 1996 was used to predict annual average flow from the six east side rivers ($r^2 = 0.60$, $p = 0.0002$). Since water quality data were not available for these six east side rivers, annual average total nitrogen and total phosphorous concentrations from the Winnipeg River at Point du Bois were used as a surrogate. The use of these Winnipeg River water quality data likely resulted in a slight over-estimation of nutrient loading from the six east side rivers as development (urban, cottage, agricultural, industrial) in the Winnipeg River watershed upstream from Point du Bois exceeds that in the relatively remote watersheds of the Black, Manigotogan, Bloodvein, Pigeon, Berens, and Poplar rivers (Water Survey of Canada 05RA, 05RB, 05RC, 05RD, 05RE, 05RF).

Inflow to Lake Winnipeg from the west side was considerably less than that estimated for the east side of the lake. Most of the flow from western tributaries was contributed by two large rivers: the Saskatchewan and the Dauphin. The seven small watersheds directly adjacent to the west side of Lake Winnipeg (Water Survey of Canada 05SB, 05SC, 05SD, 05SE, 05SF, 05SG, 05SH) make up about three percent of the western portion of the gross watershed area in Canada and generally contribute only small local flows. The Icelandic (05SC) and Fisher rivers (05SD) are the only rivers from this small watershed area that contribute measurable flow to the lake and both are gauged. Therefore, no additional inflow into the west side of Lake Winnipeg was estimated.

As in Bourne *et al.* (2002), total nitrogen and total phosphorous loads to Lake Winnipeg from atmospheric deposition were estimated with surface area (23,750 km^2) and atmospheric deposition (0.20 kilograms total phosphorus per hectare per year and 4.0 kilograms total nitrogen per hectare per year [kg/ha/y]) as reported by Chambers and Dale (1997). Rates of nitrogen deposition from Chambers and Dale (1997) are similar to those calculated with data from Ruddy *et al.* (2006) for the period 1994 through 2000 for the U.S. states of Minnesota and North Dakota in the southern portion of the Lake Winnipeg watershed (4.1 kg/ha/y). More recent data from the Experimental Lakes Area in northwestern Ontario (1993 to 2004) suggests that nitrogen deposition was on average 7.4 kg/ha/y (M.P. Stainton, unpublished data), in the range reported by Lorenz *et al.* (2009) for heavily industrialized areas of the Great Lakes, Mississippi, and Ohio River watersheds. Overall, a rate of 4.0 kg/ha/y can be considered conservative. For phosphorus, a review by Flaten *et al.* (2003) suggested that 0.20 kg/ha/y might be conservative given Manitoba estimated rates of dry deposition by Bennet (1985, 0.82 kg/ha/y) and Beck (1985, 0.41 kg/ha/y), and rates of wet and dry deposition at the Experimental Lakes Area of northwestern Ontario (Schindler 1977, 0.24 to 0.53 kg/ha/y). More recently (1993 to 2004) phosphorous deposition has been on average 0.30 kg/ha/y at the Experimental Lakes Area (M.P. Stainton, unpublished data). A rate of 0.20 kg of total phosphorus per hectare can therefore be considered conservative. Estimates of nitrogen generated through nitrogen-fixing species of algae are from Len Hendzel (unpublished data 2006) as reported by the Lake Winnipeg Stewardship Board (2006).

Outflow from Lake Winnipeg was calculated by summing the flow in the east and west channels of the Nelson River downstream from Lake Winnipeg. On average between 1994 and 2007, 14 % of the flow was carried in the east channel of the Nelson River where the water quality and hydrometric stations are about 50 river kilometres from the outlet of Lake Winnipeg. Approximately 85 % of the flow was carried in the west channel of the Nelson River and through the hydro-electric dam at Jenpeg. The water quality and hydrometric stations on the west channel are about 120 river kilometres from the outlet of Lake Winnipeg. Prior to 2001, water was sampled for nutrients only three times per year (June, July, and August) in the east channel of the Nelson River near the community of Norway House. Starting in 2001, water quality samples were also collected in cooperation with Manitoba Hydro in the forebay behind the Jenpeg hydro-electric dam. Samples were collected as often as once every two weeks, thereby providing considerably more information on water quality in the Nelson River. Throughout the study period, annual average total nitrogen and total phosphorous concentrations were calculated for each channel, where available, and used to estimate nutrient loads in the Nelson River. Efforts to better characterize water quality flowing from Lake Winnipeg are underway.

Nutrient export was calculated by dividing the mean annual load by the gross watershed area of the basin.

7.2 Analysis of the Impact of Sampling Frequency (Rivers)

To better understand the impact of frequency and timing of water quality sampling on calculation of nutrient loads, some weekly water quality sampling was conducted in the La Salle River in 2007. In total, 13 samples were collected throughout 2007 instead of the four that would have normally been collected. Figure A3 indicates how sample collection was distributed across the hydrograph. Even with the increased frequency of sampling, the peak in the hydrograph on April 15, 2007 and a summer rainstorm in late June were not sampled. Total phosphorous and total nitrogen loads were calculated with and without the additional nine water quality samples. To calculate annual nutrient loads with 13 samples, total phosphorous and total nitrogen concentrations were interpolated between sampling periods such that daily nutrient loads were calculated and summed to provide an annual nutrient load. For total phosphorus, concentrations varied from 0.282 mg/L in May to 0.929 mg/L in August. Total annual phosphorous load was estimated at 52 tonnes per year with four samples and with 13 samples. In contrast, total nitrogen concentrations ranged from 1.1 mg/L in early October to 6.03 mg/L during spring freshet. Since nitrogen concentrations were relatively high during spring freshet, calculation of daily rather than annual average nutrient loads produced more than double the estimate of total annual nitrogen load (394 tonnes per year as compared to 195 tonnes per year). Work to better estimate nutrient loads continues and calculation of loads in future years will benefit from improved sampling frequency.

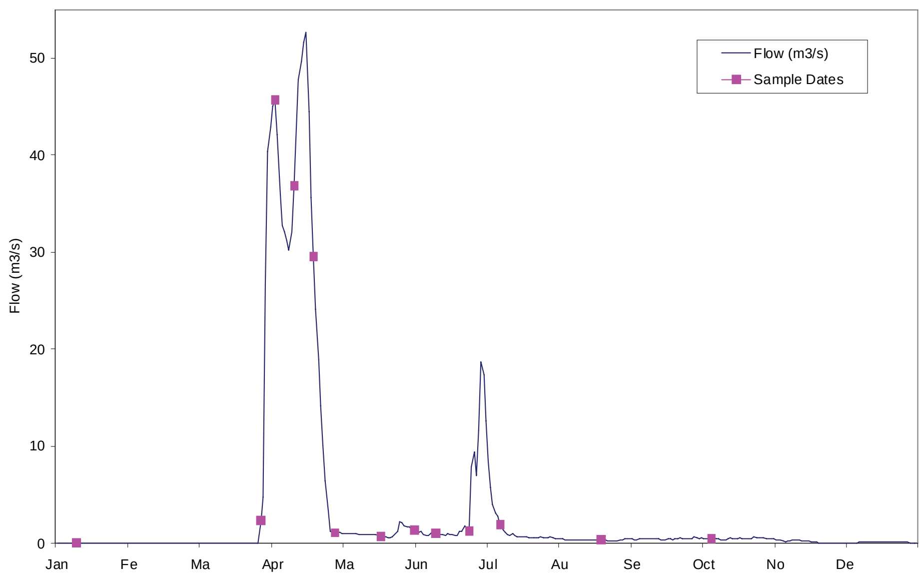


Figure A3. *Water quality sampling dates and flow in the La Salle River in 2007 (m^3/s).*

Appendix 8. Collection and Analytical Methods for Toxins

All offshore (pelagic) samples were collected from the Lake Winnipeg Research Consortium research vessel *MV Namao* (http://www.lakewinnipegresearch.org/Namao.html). For analytical purposes, values reported at the detection limit for samples collected by Manitoba Water Stewardship (1999 to 2007) and AlgalTox International (2006 and 2007), were assumed to be 50 % of the detection limit. For the purpose of this report, the samples have been categorized as either nearshore or offshore (pelagic).

8.1 Data Collected by Manitoba Water Stewardship (1999 to 2007)

Samples were collected on Lake Winnipeg during the period of 1999 to 2007. During these years, samples were collected from both long-term water quality monitoring stations, and also at various offshore locations where evidence of blooms were present. In addition, as part of Manitoba Water Stewardship's beach monitoring programs, algal samples were collected when significant blooms along the Lake Winnipeg beaches were evident. A limited number of samples was also collected by scientists associated with the Lake Winnipeg Research Consortium and the public during bloom events at various locations along beaches and shorelines on Lake Winnipeg.

Samples collected by Manitoba Water Stewardship for microcystin analysis were submitted to Dr. Charles Holmes laboratory at the University of Alberta for the period of 2000 and 2001. HydroQual Laboratories in Alberta and ALS Laboratories Group in Winnipeg analyzed the samples for microcystin for the periods 2002 to 2003, and 2004 to 2007 respectively. Prior to 2004, samples were analyzed using the protein phosphatase assay, whereas samples collected post 2003 were analyzed using the ELISA Kit (Enzyme-Linked ImmunoSorbent Assay) (http://www.envirologix.com/library/ep022insert.pdf) method supplied by Envirologix. For samples collected by Manitoba Water Stewardship between August 1999 and June 2007, microcystin was analyzed on unfrozen samples. Subsequent to June 2007, samples were analyzed after the samples had been frozen. A comparison of results from frozen and unfrozen paired samples was also completed in August and September of 2006. Frozen samples resulted in higher toxin levels, as the freezing causes the cells to lyse, releasing the cellular toxin, which is measured during the analysis. Therefore, for the purposes of this report it is not possible to evaluate trends in the data over time. However, since the results from both frozen and unfrozen samples were generally at a very low concentration and generally not at environmentally significant levels, the Manitoba Water Stewardship data have been combined for this report.

Beach Monitoring Protocol

Algal blooms are reported to Manitoba Water Stewardship by the public, regional staff, or from beach monitoring staff responsible for collecting microbiological samples from beaches across the province. The Province of Manitoba responds to a report of an algal bloom by collecting water samples in the area where the bloom was observed and filling two bottles. Samples are collected at mid-thigh depth with a sweeping motion through the water in order to simulate the level to which swimmers would be exposed. One sample bottle is analyzed by a local laboratory (ALS Laboratories) for a count of cyanobacteria cells (preserved with Lugol's solution) and one for analysis of microcystin-LR (ELISA method). Results are interpreted using the guidance in the proposed Canadian Recreational Water Quality Guidelines. If Manitoba Water Stewardship finds that the guideline of 100,000 cells of cyanobacteria per 1 mL is exceeded, but are below the 20 µg/L microcystin-LR, then the Medical Officer of Health posts an algal advisory sign (http://www.gov.mb.ca/waterstewardship/quality/images/Algae%20Advisory%20Signs%20for%20Web.jpg). The advisory sign doesn't close the beach but offers advice on additional precautions that should be taken when algae blooms are present. A second level of signage is sometimes posted by the Medical Officer of Health if toxin concentrations exceed 20 µg/L (http://www.gov.mb.ca/waterstewardship/quality/Second%20Level.Toxic%20Algae.jpg). This sign recommends not drinking, swimming, or having other contact with the water. These signs have been posted several times along Lake Winnipeg beaches over the past several years. Re-sampling of these beaches is conducted if either the cell counts or the microcystin-LR are above guidelines, and sampling is continued until results dropped below the guidelines. Algal blooms can appear and disappear quickly depending on weather conditions and water currents.

8.2 Data Collected by AlgalTox International (2006 and 2007)

In both 2006 and 2007, the Lake Winnipeg Research Consortium vessel, the *MV Namao*, was employed as the sampling platform. Whole water samples were collected for microcystin analysis and for phytoplankton enumeration by lowering a tube fitted with a one-way foot valve over the side of the ship to a depth corresponding to the euphotic zone. The euphotic zone was determined as twice the Secchi depth at each site. The water therefore represented an integrated water column sample from the surface of the water to the bottom of the euphotic zone. One 60 mL subsample was transferred to an opaque plastic vial and frozen for subsequent microcystin analysis.

In 2006, 93 pelagic samples were collected on the lake in the north basin, narrows, and south basin in June and then again in September/October. In 2007, 86 pelagic samples were collected from July 24 to August 15 and from September 17 to October 5. All sampling sites corresponded to the established Lake Winnipeg Research Consortium monitoring sites on the lake in the north basin, narrows, and south basin.

In addition to whole water euphotic zone samples, a 64 µm plankton net was also used to collect plankton for both microcystin analysis and phytoplankton enumeration in 2007. The main purpose for collecting these additional samples from the euphotic zone was to compare microcystin concentrations in whole water samples and in net samples, as well as to determine if cyanobacteria present in the lake at the time contained microcystin, even when the toxin may have not been measurable in whole water samples.

Net samples were collected by lowering the net vertically down to the euphotic zone depth and raising the net to the surface. Lake water was used to spray the outside of the net so that the plankton was washed into the net bucket. Water was then removed from the sample before transferring it to an opaque plastic vial. A small amount of lake water was used to transfer the plankton sample from the net bucket to the vial and the vial was topped up with lake water in order to ensure that a standard volume of sample was present in the vial. In some cases, when very little phytoplankton was present, the net was lowered several times until enough biomass was collected. In all cases, care was taken to de-water the sample before each subsequent lowering of the net. This may have caused a slight loss of sample. In other cases, when a large amount of phytoplankton was present at the sampling site, one vertical haul of the net produced more than a sufficient sample. In such cases, the sample was split among several vials, and these samples were later combined before microcystin analysis. The number of vertical hauls and the depth of the euphotic zone were recorded, so that an estimate of the volume of water passed through the net could be calculated. This allowed for a comparison between microcystin concentrations in whole water and net samples, taking into account that each method of sampling has its error.

Microcystin analysis was conducted by AlgalTox International (a division of Miette Environmental Consulting Inc.). Frozen whole water and plankton samples were thawed and ultrasonicated for 45 seconds to disrupt phytoplankton cells and release cellular microcystin. The samples were then filtered through 0.45 µm nitrocellulose filters to remove cellular debris. Microcystin analysis was conducted on the filtrate, and the reported concentrations represented the sum of microcystin that was dissolved in the lake water, as well as that found originally in the phytoplankton cells. Quantification of microcystin used protein phosphatase inhibition assay (after the methods of An and Carmichael, 1994) and the reported concentrations represented total microcystin, expressed in microcystin-LR equivalents. The assay is based on the following principles. Protein phosphatase 1c (PP1c) catalyzes the removal of phosphate from colourless p-nitrophenyl phosphate (p-NPP). This reaction yields a yellow endpoint through the production of p-nitrophenol (p-NP). As microcystin inhibits the activity of PP1c, the conversion of p-NPP to p-NP is also inhibited. Thus, increasing concentration of microcystin corresponds to a decrease in colour. Samples were analyzed in triplicate, with duplicate negative control samples to correct for the optical density due to the sample itself. A 100 % enzyme activity control (in triplicate) was also used to determine the colour generated by an uninhibited reaction of PP1c enzyme with p-NPP, as this reaction is dependent on the enzyme activity and concentration, reaction time, and incubation temperature. A series of positive controls (microcystin standards) were analyzed in triplicate to generate a calibration curve. Duplicate positive reaction controls were used to account for the optical density of the positive control buffer used in both the 100 % enzyme activity controls and the Microcystin standards. Detection limit of the assay was approximately 0.100 µg/L.

8.3 Data Collected from Algal Taxonomy and Ecology Inc and Lake Winnipeg Research Consortium Inc by Kling and Herbert

Samples collected in 2003 and 2004 were obtained using a plankton net towed behind the *MV Namao* for 30 minutes. Samples were frozen and then freeze-dried prior to analysis. All samples were prepared using a partially modified method developed by Carmichael and Brittain (unpublished data), which was adapted from Meriluoto *et al.* (Meriluoto and Eriksson 1988). For every gram of sample biomass, 50 mL of 80 % ACS grade methanol was used in extraction. One hundred twenty-five to 700 mg of lyophilized plankton haul sample was weighed out, extracted, shaken overnight, and centrifuged for 15 minutes at 25,000 rpm. The supernatant was removed and injected into a glass vial. This process was repeated twice more with the sample being shaken for 15 minutes the second and third times. The volume of supernatant was recorded each time, and the final sample was dried in a fume hood under an air stream, and then stored at -25°C until analysis. All samples were pretested utilizing the Microcystin ELISA Tube kits produced by Envirologix, Portland, Maine (QualiTube Kit for Microcystins - ET 022) with a limit of detection of 0.3 µg/L.

Protein Phosphatase Inhibition Assay was conducted at the lab of Greg Boyer in Syracuse, New York (Boyer, unpublished data footnote: www.esf.edu/chemistry/faculty/boyer.htm). Most samples were analyzed in duplicate wells of a 96-well plate, except Station 31 where there was only a trace amount of toxin present. Any sample that tested greater than 0.5 µg/L was confirmed by HPLC using a PDA detector. In order to process the samples with the HPLC, they were freeze-dried and then extracted onto columns in 50 % methanol acidified with 1 % acetic acid. Microcystin values were expressed as µg of Microcystin-LR per litre.

LC-ESI-MS-MS analysis was conducted by J. Dahlmann at the University of Jena, Jena, Germany (Dahlmann and Luckas 2005). Plankton hauls were placed in 2 mL Eppendorf cups and extracted twice with a mixture of water/methanol (50/50; v/v) by 10-minute sonication in an ultrasonic bath. The extracts were centrifuged (14,000 rpm) and the supernatant was directly subjected to LC-ESI-MS-MS analysis. Microcystin values are expressed as µg/g dry weight. The limit of detection (LOD) of all toxin variants was determined to be 0.5 ng for each individual microcystin, absolute. Table A7 provides an overview of the number of pelagic samples that were analyzed for microcystin from 1999 to 2007 for all three data sets.

Year	# of Samples
1999	49
2000	3
2001	0
2002	0
2003	76
2004	57
2005	15
2006	135
2007	101
Total	436

Table A7: *Number of pelagic whole water and plankton net samples analyzed for microcystin from 1999 to 2007. Samples include those collected by Manitoba Water Stewardship and other researchers of the Lake Winnipeg Research Consortium.*

Appendix 9. Beach Sampling

The number of Lake Winnipeg beaches that are sampled for fecal indicator bacteria has steadily increased from 10 beaches monitored in the 1980s and 1990s to 19 swimming areas that are currently sampled. The Province of Manitoba continues to follow the sampling procedure guidance outlined in Health Canada's Guidelines for Canadian Recreational Water Quality (Health and Welfare Canada 1992), but the Province exceeds the recommended sampling frequency of individual beaches. A minimum of five samples per beach were collected every two weeks prior to 2003 but this frequency was increased to at least weekly sampling events during the bathing seasons of 2004 to present. The geometric mean for each beach was calculated and compared to the Manitoba Water Quality Objective for *E. coli* in recreational water. Prior to 1999, beaches were monitored for fecal coliform as the indicator of fecal contamination, but as laboratory methods changed and new growth media became more reliable, the Province switched to monitoring densities of the indicator bacteria, *E. coli.* With guidance from Manitoba Health, a posting notification protocol was developed to advise the bathing public when densities of *E. coli* exceeded the recreational guideline.

Table A8. *Lake Winnipeg south basin beaches monitoring schedule and collection matrix from 2004 to 2009.*

Beach	Monitoring Frequency	Matrix
Grindstone Park Beach	Weekly	Lake Water
Black Point Beach	Weekly	Lake Water
Gull Harbour Beach	Weekly	Lake Water
Sandy Bar Beach	Weekly	Lake Water
Spruce Sands Beach	Weekly	Lake Water
Gimli Beach	Daily (Monday to Friday)	Lake Water, Sand, Sand Pore Water
Sandy Hook Beach	Weekly	Lake Water
Winnipeg Beach	Weekly	Lake Water
Matlock Beach	Weekly	Lake Water
Patricia Beach	Weekly	Lake Water
Sunset Beach	Weekly	Lake Water
West Grand Beach	Daily (Monday to Friday)	Lake Water, Sand, Sand Pore Water
East Grand Beach	Weekly	Lake Water
Lester Beach	Weekly	Lake Water
Hillside Beach	Weekly	Lake Water
Victoria Beach	Weekly	Lake Water
Albert Beach	Weekly	Lake Water

All samples were collected according to established sampling protocols (Health and Welfare Canada 1992; Whitman and Nevers 2003), stored in coolers with ice, and transported to the laboratory as quickly as possible.

Samples were filtered, plated, and placed in the incubator on the same day they were sampled. Sampling began as early as possible in the season, frequently well before bathing season which usually begins in mid-June. Densities of *E. coli* were analyzed by a contracted laboratory accredited under the Canadian Analytical Laboratory Association.

During the open water seasons of 2004 to 2009, all of the beaches were routinely monitored once per week, with five individual samples collected at each beach. Two beaches, West Grand Beach on the east side of the south basin, and Gimli Beach on the west side of the south basin were monitored daily (Table A8). Additionally, five sand samples and five sand pore water samples were collected daily from West Grand Beach and Gimli Beach, respectively. Sand and sand pore water were analyzed to provide additional information about the densities of *E. coli* in these areas, and to provide supporting evidence that *E. coli* from beach sand could be transferred to bathing water.

References

Adjusted and Homogenized Canadian Climate Data (AHCCD). 2010. http://ec.gc.ca/dccha-ahccd/Default.asp?lang=En&n=B1F8423A-1

Ahl, T. 1988. Background yield of phosphorus from drainage area and atmosphere: An empirical approach. Hydrobiologia 170: 35-44.

Alberta Environment. 1999. Surface water quality guideline for use in Alberta. Environmental Assurance Division, Science and Standards Branch, November, 1999. 20 pp.

Alm, E.W., J. Burke, and A. Spain. 2003. Fecal indicator bacteria are abundant in wet sand at freshwater beaches. Water Research 37: 3978-3982.

An, J-S. and W.W. Carmichael. 1994. Use of a colorimetric protein phosphatase inhibition assay and enzyme linked immunosorbent assay for the study of microcystins and nodularins. Toxicon 32: 1495-1507.

Anderson, F.O. and P. Ring. 1999. Comparison on phosphorus release from littoral and profundal sediments in a shallow, eutrophic lake. Hydrobiologia 408-409: 175-183.

Arnott, D.L. and M. J. Vanni. 1996. Nitrogen and phosphorus recycling by the zebra mussel (*Dreissena polymorpha*) in the western basin of Lake Erie. Canadian Journal of Fisheries and Aquatic Sciences 53: 646-659.

Ayles, G.B., and D.M. Rosenberg (editors). 2005. Lake Winnipeg Science Workshop, November 29-30, 2004. Canadian Manuscripts Reports of Fisheries and Aquatic Sciences 2732: xii + 123 pp.

Backer, L.C. 2002. Cyanobacterial harmful algal blooms (CyanoHABs): Developing a public health response. Lake and Reservoir Management 18(1): 20-31.

Baird, W.F. & Associates Coastal Engineers Ltd. and Stantec Consulting Ltd. 2000. Lake Winnipeg shoreline erosion study. W.F. Baird & Associates Coastal Engineers Ltd., Oakville, Ontario.

Bajkov, A. 1930. Biological conditions of Manitoban Lakes. Contributions to Canadian Biology and Fisheries 5(12): 383-421.

Bajkov, A. 1934. The plankton of Lake Winnipeg drainage system. Internationale Revue der gesamten Hydrobiologie und Hydrographie 31(3/4): 239-272.

Barton, B.A. and B.R. Taylor. 1996. Oxygen requirements of fishes in northern Alberta rivers with a general review of the adverse effects of low dissolved oxygen. Water Quality Research Journal of Canada 31(2): 361-409.

Bayley, S.E., D.W. Schindler, K.C. Beaty, B.R. Parker, and M.P. Stainton. 1992. Effects of multiple forest fires on nutrient yields from streams draining boreal forest and fen watersheds: nitrogen and phosphorus. Canadian Journal of Fisheries and Aquatic Sciences 49: 584-596.

B.C. Ministry of Environment, Lands and Parks. 1998. Guidelines for Interpreting Water Quality Data. Prepared by Ministry of Environment, Lands and Parks for the Land Use Task Force Resources Inventory Committee. Version 1.0. http://www.ilmb.gov.bc.ca/risc/pubs/aquatic/interp/index.htm.

Beck, A.E. 1985. Recreational development capacity study of twelve lakes in the south Riding Mountain Planning District. Department of Environment and Workplace Safety and Health. Water Standards and Studies Report 87-5.

Benson, A.J. and D. Raikow. 2009. *Dreissena polymorpha*. USGS Nonindigenous Aquatic Species Database, Gainesville, FL. Revision Date: 10/31/2008.

Bernhardt, E.S., G.E. Likens, D.C. Busco, and C.T. Driscoll. 2003. In-stream uptake dampens effects of major forest disturbance on watershed nitrogen export. Proceedings of the National Academy of Sciences 100: 10304-10308.

Bertram, P.E. 1993. Total phosphorus and dissolved oxygen trends in the central basin of Lake Erie, 1970-1991. Journal of Great Lakes Research 19: 224-236.

Bourne, A., N. Armstrong, and G. Jones. 2002. A preliminary estimate of total nitrogen and total phosphorus loading to streams in Manitoba, Canada. Water Quality Management Section, Water Branch, Winnipeg, Manitoba. Manitoba Conservation, Report No. 2002-04.

Brittain, S.M., J. Wang, L. Babcock-Jackson, W.W. Carmicheal, K.L. Rinehart, and D.A. Culver. 2000. Isolation and characterization of microcystins, cyclic heptapeptides hepatotoxins from a Lake Erie strain of *Microcystis aeruginosa*. Journal of Great Lakes Research 2693: 241-249.

Brunskill, G.J. 1973. Rates of supply of nitrogen and phosphorus to Lake Winnipeg, Manitoba, Canada. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie 18(3): 1755-1759.

Brunskill, G.J. and B.W. Graham. 1979. The offshore sediments of Lake Winnipeg. Department of Fisheries and Oceans, Winnipeg, Manitoba. Canadian Fisheries & Marine Service Manuscript Report No. 1540, 75 pp.

Brunskill, G.J., P. Campbell, and S.E.M. Elliott. 1979a. Temperature, oxygen, conductance and dissolved major elements in Lake Winnipeg. Fisheries and Marine Service, Manuscript Report No. 1526, 127 pp.

Brunskill, G.J., D.W. Schindler, S.E.M. Elliott, and P. Campbell. 1979b. The attenuation of light in Lake Winnipeg waters. Department of Fisheries and the Environment, Winnipeg, Manitoba. Fisheries & Marine Service Manuscript Report No. 1522.

Brunskill, G.J., S.E.M. Elliott, and P. Campbell. 1980. Morphometry, hydrology, and watershed data pertinent to the limnology of Lake Winnipeg. Department of Fisheries and Oceans, Winnipeg, Manitoba. Canadian Manuscript Report of Fisheries & Aquatic Science No. 1556.

Bukata, R.P., J.H. Jerome, K.Y. Kondratyev, and D.V. Pozdnyakov. 1995. Optical properties and remote sensing of inland and coastal waters. CRC Press, Boca Raton, Fla. 362 pp.

Burn, D.H. and N.K. Goel. 2001. Flood frequency analysis for the Red River at Winnipeg. Canadian Journal of Civil Engineering 28: 355-362.

Byappanahalli M., M. Fowler, D. Shively, and R. Whitman. 2003. Ubiquity and persistance of *Escherichia coli* in a midwestern coastal stream. Appl. Environ. Microbiol. 69(8): 4549-4555.

Cabelli, V.J., A.P. Dufour, L.J. McCabe, and M.A. Levin. 1983. A marine recreational water quality criterion consistent with indicator concepts and risk analysis. Journal Water Pollution Control Federation 55: 1306-1314.

Campbell, K.B., A.J. Derksen, R.A. Remnant, and K.W. Stewart. 1991. First specimens of the rainbow smelt, *Osmerus mordax*, from Lake Winnipeg, Manitoba. Canadian Field-Naturalist 105(4): 568-570.

Canadian Council of Ministers of the Environment (CCME). 2003. Climate, Nature, People: Indicators of Canada's Changing Climate. http://www.ccme.ca/assets/pdf/cc_ind_people_drought_e.pdf

Canadian Council of Ministers of the Environment (CCME). 2005. Canadian Environmental Quality Guidelines. Update 5.0. Canadian Council of Ministers of the Environment, Winnipeg, Manitoba. http://www.ccme.ca/publications/ceqg_rcqe.html

Carignan, R., D. Planas, and C. Vis. 2000. Planktonic production and respiration in oligotrophic Shield lakes. Limnology and Oceanography 45(1): 189-199.

Carmichael, W.W. 1992. A Status Report on Planktonic Cyanobacteria (Blue Green Algae) and their Toxins. EPA/600/R-92/079, Environmental Monitoring Systems Laboratory, Office of Research and Development, US Environmental Protection Agency, Cincinnati, Ohio.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8: 559-568.

Carrik, H.J. 2004. Algal distribution patterns in Lake Erie: Implications for oxygen balances in the eastern basin. Journal of Great Lakes Research 30: 133-147.

Casselman, J.M., K.A. Scott, D.M. Brown, and C.J. Robinson. 1999. Changes in relative abundance, variability, and stability of fish assemblages of Eastern Lake Ontario and the Bay of Quinte - the value of long-term community sampling. Aquatic Ecosystem Health 2: 255-269.

Chambers, P.A. and A.R. Dale. 1997. Contribution of industrial, municipal, agricultural and groundwater sources to nutrient export, Athabasca and Wapiti-Smokey Rivers, 1989-1993. Northern River Basins Study Project Report No. 110.

Chambers, P.A., M. Guy, E.S. Roberts, M.N. Charlton, R. Kent, C. Gagnon, G. Grove, and N. Foster. 2001. Nutrients and their impact on the Canadian environment. Agriculture and Agri-Food Canada, Environment Canada, Fisheries and Oceans Canada, Health Canada and Natural Resources Canada 241 pp.

Chang, P.S.S., D.G. Cobb, J.F. Flannagan, and O.E. Saether. 1992. The oligochaetes from the 1969 Lake Winnipeg baseline survey. Department of Fisheries and Oceans, Winnipeg, Manitoba. Canadian Data Report of Fisheries and Aquatic Sciences 869.

Chang, P.S.S., D.G. Cobb, J.F. Flannagan, and O.E. Saether. 1993. The aquatic insects from the 1969 Lake Winnipeg, baseline survey. Department of Fisheries and Oceans, Winnipeg, Manitoba. Canadian Data Report of Fisheries and Aquatic Sciences 897.

Chang, P.S.S., D.G. Cobb, J.F. Flannagan, and O.E. Saether. 1994. Light trap collections of mayflies, caddisflies and chironomids from Lake Winnipeg during 1969 and 1971. Department of Fisheries and Oceans, Winnipeg, Manitoba. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2223.

Chapman, G. 1986. Ambient water quality criteria for dissolved oxygen. U.S. Environmental Protection Agency, W., DC. (Editor).

Charlton, M.N. 1987. Lake Erie oxygen revisited. Journal of Great Lakes Research 13(4): 697-708.

Chen, H., J.M. Burke, T. Mosindy, P.M. Fedorak, and E.E. Prepas. 2009. Cyanobacteria and microcystin-LR in a complex lake system representing a range in trophic status: Lake of the Woods, Ontario, Canada. Journal of Plankton Research 31: 993-1008.

Chesapeake Bay Program. 2009. Chesapeake Bay Program - A Watershed Partnership. http://www.chesapeakebay.net/index.aspx?menuitem=13853

Chorus, I. and J. Bartram. 1999. Toxic cyanobacteria in water: a guide to their public health consequences, monitoring and management. London: E & F Spon. 416 pp.

Choudhury, A., E. Charipar, P. Nelson, J.R. Hodgson, S. Bonar, R.A., and R. Cole. 2006. Comparative Parasitology 73(2): 269-273.

Colby, P.J., G.R. Spangler, D.A. Hurley, and A.M. McCombie. 1972. Effects of eutrophication on salmonid communities in oligotrophic lakes. Journal of the Fisheries Research Board of Canada 29: 975-983.

Colford, J.M., T.J. Wade, K.C. Schiff, C.C. Wright, J.F. Griffith, S.K. Sandhu, S. Burns, M. Sobsey, G. Lovelace, and S.B. Weisberg. 2007. Water quality indicators and the risk of illness at ceaches with nonpoint sources of fecal contamination. Epidemiology 18: 27-35.

Conroy, J.D., D.A. Culver, W.J. Edwards, D. Kane, R. Pontius, J. Richey, J. Shea, and H. Zhang. 2005a. Differential excretion by dreissenid taxa: implications for western Lake Erie. Freshwater Biology 50: 1146-1162.

Conroy, J.D., D.D. Kane, D.M. Dolan, W.J. Edwards, M.N. Charlton, and D.A. Culver. 2005b. Temporal trends in Lake Erie plankton biomass: roles of external phosphorus loading and dreissenid mussels. Journal of Great Lakes Research 31(2): 89-110.

Crowe, J.M.E. 1969. A preliminary report on the benthic organisms of Lake Winnipeg during September, 1964. Manitoba Department of Mines and Natural Resources, Fisheries Branch, Winnipeg, Manitoba. Manuscript Report No. 69-6.

Crowe, J.M.E. 1972a. Pollution monitoring in Lake Winnipeg, South Basin, 1967. Manitoba Department of Mines, Resources, and Environmental Management, Research and Development Branch, Winnipeg, Manitoba. Manuscript Report 71-11.

Crowe, J.M.E. 1972b. The south basin of Lake Winnipeg, 1970. Manitoba Department of Mines, Resources, and Environmental Management, Research and Development Branch, Winnipeg, Manitoba. Manuscript Report No. 71-13.

Crowe, J.M.E. 1972c. The south basin of Lake Winnipeg - an assessment of pollution. Manitoba Department of Mines, Resources, and Environmental Management, Winnipeg, Manitoba. Report No. 72-14.

Crowe, J.M.E. 1973a. Limnology of the south basin of Lake Winnipeg, March 1965 to March 1969. Manitoba Department of Mines, Resources and Environment Management, Research Branch, Winnipeg, Manitoba. Manuscript Report No. 73-24, pp. 37.

Crowe, J.M.E. 1973b. The benthic fauna of the south basin of Lake Winnipeg, March 1965 to March 1967. Manitoba Department of Mines, Resources, and Environmental Management, Research Branch, Winnipeg, Manitoba. Manuscript Report No. 73-27.

Dahlman, J. and B. Luckas. 2005. Microcystin analysis in drinking and surface water using a highly sensitive LC/MS/MS direct injection technique. Applied Biosystems/MDS SCIEX www.appliedbiosystems.com

David, M.B. and L.E. Gentry. 2000. Anthropogenic inputs of nitrogen and phosphorus and riverine export for Illinois, USA. Journal of Environmental Quality 29: 494-508.

Davies, C.M., J.A.H. Long, M. Donald, and N.J. Ashbolt. 1995. Survival of fecal microorganisms in marine and freshwater sediments. Applied Environmental Microbiology 61(5): 1888-1896.

del Giorgio, P.A. and R.H. Peters. 1994. Patterns in planktonic P:R ratios in lakes: Influence of lake trophy and dissolved carbon. Limnology and Oceanography 39(4): 772-787.

DePinto, J.V., T.C. Young, and S.C. Martin. 1981. Algal-available phosphorus in suspended sediments from lower Great Lakes tributaries. Journal of Great Lakes Research 7: 311-325.

Derksen, A.J. and E.I. Hangasjarvi. 1979. A survey of net plankton in the North Basin of Lake Winnipeg, July 1974. Manitoba Department of Mines, Natural Resources and Environment, Winnipeg, Manitoba. Manuscript Report No. 79-46.

Diaz, R.J. and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321(5891): 926-929.

Dillon, P.J. and F.H. Rigler. 1974. The phosphorus-chlorophyll relationship in lakes. Limnology and Oceanography 19: 767-773.

Dorich, R.A., D.W. Nelson, and L.E. Sommers. 1980. Algal availability of sediment phosphorus in drainage water of the Black Creek watershed. Journal of Environmental Quality 9: 557-563.

Dubois, K., R. Carignan, and J. Veizer. 2009. Can pelagic net heterotrophy account for carbon fluxes from eastern Canadian lakes? Applied Geochemistry 24(5): 988-998.

East Interlake Conservation District. 2007. Know Your Watershed. East Interlake Conservation District, Gimli, Manitoba.

Eckmann, R. and R. Rösch. 1998. Lake Constance fisheries and fish ecology. Advances in Limnology 53: 285-301.

Edge, T.A. and S. Hill. 2007. Multiple lines of evidence to identify the sources of fecal pollution at a freshwater beach in Hamilton Harbour, Lake Ontario. Water Research 41: 3585-3594.

Edge, T.A., S. Hill, P. Seto, and J. Marsalek. 2010. Library-dependent and library-independent microbial source tracking to identify spatial variation in faecal contamination sources along a Lake Ontario beach (Ontario, Canada). Water Science & Technology 62(3): 719-727.

Edwards, W.J., J.D. Conroy, and D.A. Culver. 2005. Hypolimnetic oxygen depletion dynamics in the central basin of Lake Erie. Journal of Great Lakes Research 31(Sup. 2): 262-271.

Eimers, M.C. and Winter, J.G. 2005. Lake Simcoe Water Quality Update: Lake Simcoe Water Quality Update 2000–2003. Technical Report Imp. B.20. 22p.

Environment Canada. 2004. National Guidelines and Standards Office. Water Policy and Coordination Directorate. Canadian Guidance Framework for the Management of Phosphorus in Freshwater Systems. Report No. 1-18.

Environment Canada and the U.S. Environmental Protection Agency. 2009. State of the Great Lakes 2009. Binational Progams December 2009; http://binational.net/solec/sogl2009_e.html

Environment Canada Atmospheric Environment Service. http://www.climate.weatheroffice.ec.gc.ca

Evans, D.O., B.A. Henderson, N.J. Bax, T.R. Marshall, R.T. Oglesby, and W.J. Christie. 1987. Concepts and methods of community ecology applied to freshwater fisheries management. Canadian Journal of Fisheries and Aquatic Sciences 44 (Suppl. 2): 448-470.

Falkowski, P.G. and J.A. Raven. 1997. Aquatic photosynthesis. Blackwell Science 375 pp.

Feare, C.J., M.F. Sanders, R. Blasco, and J.D. Bishop. 1999. Canada goose (*Branta canadensis*) droppings as a potential source of pathogenic bacteria. Journal of the Royal Society for the Promotion of Health 119: 146-155.

Federal-Provincial-Territorial Committee on Drinking Water. 2002. December 2009; http://www.hc-sc.gc.ca/ewh-semt/alt_formats/hecs-sesc/pdf/pubs/water-eau/cyanobacterial_toxins/cyanobacterial_toxins-eng.pdf

Ferber, L.R., S.N. Levine, A. Lini, and G.P. Livingstone. 2004. Do cyanobacteria dominate in eutrophic lakes because they fix atmospheric nitrogen? Freshwater Biology 49: 690-708.

Figuerola, J., A.J. Green, and T.C. Michot. 2005. Invertebrate eggs can fly: evidence of waterfowl-mediated gene flow in aquatic invertebrates. The American Naturalist 165: 274-280.

Findlay, D.L. and H.J. Kling. 1998. Protocols for Measuring Biodiversity: Phytoplankton in Freshwater. www.eman-rese.ca/eman/ecotools/protocols/freshwater/phytoplankton/intro.html

Flannagan, J.F. and D.G. Cobb. 1981. Changes in the profundal Trichoptera of Lake Winnipeg 1928-32 to 1969. G.P. Moretti, Proceedings of 3rd International Symposium on Trichoptera, W. Junk, The Hague, 67-74 pp.

Flannagan, J.F. and D.G. Cobb. 1984. Production of *Hexagenia limbata* and *H. rigida* in Lake Winnipeg, Manitoba. V. Landa, T. Soldan, and M. Tonner, Proceedings of the 4th International Conference on Ephemeroptera, Czechoslovakia Academy of Sciences, v. 2127, 35 pp.

Flannagan, J.F. and D.G. Cobb. 1991. The molluscs from the 1969 Lake Winnipeg baseline survey. Department of Fisheries and Oceans, Winnipeg, Manitoba. Canadian Manuscript Report of Fisheries and Aquatic Sciences No. 2127.

Flannagan, J.F. and D.G. Cobb. 1994. The benthic crustaceans from the 1969 Lake Winnipeg baseline survey. Department of Fisheries and Oceans, Winnipeg, Manitoba. Canadian Data Report of Fisheries and Aquatic Sciences 928.

Flannagan, J.F., D.G. Cobb, and P.M. Flannagan. 1994. A review of the research on the benthos of Lake Winnipeg. Canadian Technical Report of Fisheries and Aquatic Sciences 2261: i-17.

Flaten, D., K. Snelgrove, I. Halket, K. Buckley, G. Penn, W. Akinremi, B. Wiebe, and E. Tyrchniewicz. 2003. Acceptable phosphorus concentrations in soils and impact on the risk of phosphorus transfer from manure amended soils to surface waters. A Review of Literature for the Manitoba Livestock Manure Management Initiative. Phase 1 of MLMMI Project #02-HERS-01. 216 pp.

Fogarty, L.R., S.K. Haack, M.J. Wolcott, and R.L. Whitman. 2003. Abundance and characteristics of the recreational water quality indicator bacteria *Escherichia coli* and enterococci in gull faeces. Applied Environmental Microbiology 95(5): 865-878.

Franzin, W.G., B.A. Barton, R.A. Remnant, D.B. Wain, and S.J. Pagel. 1994. Range extension, present and potential distribution and possible effects of rainbow smelt in Hudson Bay Drainage waters of Northwestern Ontario, Manitoba, and Minnesota. North American Journal of Fisheries Management 14: 65-76.

Franzin, W.G., K.W. Stewart, G.F. Hanke, and L. Heuring. 2003. The fish and fisheries of Lake Winnipeg: the first 100 years. Canadian Technical Report of Fisheries and Aquatic Sciences 2398: v + 53 pp.

Franzin, W.G., D.A. Watkinson, and S. Backhouse. 2005. Fishes of the Lake Winnipeg Basin, their temperature tolerances and potential effects of climate warming on populations. Final report to the Canadian Climate Action Fund Project A499. Winnipeg, Manitoba.

Giani, A., D.F. Bird, Y.T. Prairie, and J.F. Lawrence. 2005. Empirical study of cyanobacterial toxicity along a trophic gradient of lakes. Canadian Journal of Fisheries and Aquatic Sciences 62: 2100-2109.

Goldsborough, G. 1999. What is happening to the Delta Marsh and Why? Manitoba Environment, Winnipeg, Manitoba, February 25, 1999.

Golterman, H.L., C.C. Bakels, and J. Jakobsmogelin. 1969. Availability of mud phosphates for the growth of algae. Int. Ver. Theor. Angew, Limnol. Verh. 17: 467-479.

Great Lakes Science Center. 2010. USGS: Zebra mussels cause economic and ecological problems in the Great Lakes GLSC Fact Sheet 2000-6.

Guildford, S.J., A.E. Poste, S.J. Yakobowski, A. Chhun, L.A. Chiavaroli, K. Muller, and H.J. Kling. 2008. Microcystin Research in Southern Ontario: Some Recent Results. 2008 International Lake of the Woods Water Quality Forum on March 12th and 13th 2008, International Falls, Minnesota.

Håkansson, H. and H.J. Kling. 1989. A light and electron microscope study of previously described and new *Stephanodiscus* species (Bacillariophyceae) from central and northern Canadian Lakes with ecological notes on the species. Dia. Res. 4, 269-288

Hamilton, P.B., L.M. Ley, S. Dean, and F.R. Pick. 2005. The occurrence of the cyanobacterium *Cylindrospermopsis raciborskii* in Constance Lake: an exotic cyanoprokaryote new to Canada. Phycologia 44(1): 17-25.

Hann, B.J. and S.-L. Kowalshuk. 2004. Zoobenthos and meiobenthos communities in Lake Winnipeg. June 11, 1010; http://www.lakewinnipegresearch.org/pdf%20files/ZOOBENTHOS%20and%20MEIOBENTHOS.pdf

Hann, B.J. and S.-L. Kowalshuk. 2005. Investigations of zoobenthos and meiobenthos communities in Lake Winnipeg. In Lake Winnipeg Research 2003: Report to Manitoba Hydro. The Lake Winnipeg Research Consortium Inc., Winnipeg, Manitoba.

Hanson, J.M. and W.C. Leggett. 1982. Empirical prediction of fish biomass and yield. Canadian Journal of Fisheries and Aquatic Sciences 39: 257-263.

Havel, J.E. and J. Stelzleni-Schwent. 2000. Zooplankton community structure: the role of dispersal. Verhandlungen des Internationalen Verein Limnologie. Limnology 27: 3264-3268.

Healney, C.D., E. Sams, S. Wing, S. Marshall, K. Brenner, A.P. Dufour, and T.J. Wade. 2009. Contact with beach sand among beachgoers and risk of illness. American Journal of Epidemiology 170: 164-172.

Health and Welfare Canada. 1992. Guidelines for Canadian Recreational Water Quality. Prepared by the Federal-Provincial Working Group on Recreational Water Quality of the Federal-Provincial Advisory Committee on Environmental and Occupational Health. 100 pp.

Health Canada. 2002. Guidelines for Canadian Drinking Water Quality: Federal-Provincial-Territorial Committee on Drinking Water. Cyanobacterial Toxins - Microcystin-LR.

Health Canada. 2009. Guidelines for Canadian Recreational Water Quality. http://www.hc-sc.gc.ca/ewh-semt/alt_formats/hecs-sesc/pdf/consult/_2009/water_rec-eau/water_rec-eau-eng.pdf

Hellawell, J.M. 1986. Biological indicators of freshwater pollution and environmental management. Elsevier, London.

Hendzel, L. 2006. Cyanobacterial Nitrogen Fixation: A Significant Source of Nitrogen to Lake Winnipeg. Presentation at the Lake Winnipeg Research Consortium Annual Workshop. March 14, 2006.

Hendzel, L., R. Hesslein, A. Salki, M. Stainton, H.J. Kling, G.K. McCullough, and B.G. Kotak. 2006. Modelling Lake Winnipeg nutrient loading and concentration. Canadian Water Resources Association, Winnipeg. 17 Oct 2006.

Hesslein, R.H., M.P. Stainton, S.J. Page, and G.K. McCullough. 2007. Modelling Lake Winnipeg Nutrient Concentrations. International Water Conference Proceedings 2007, International Water Institute website December 2009; http://www.internationalwaterinstitute.org/forms/IWI2007/1B2%20Climate%20Change,%20Runoff,%20and%20Lake%20Winnipeg,%20Eutrophication%20Part%202.pdf

Heuring, L. 1993. A historical assessment of the commercial and subsistence fish harvests of Lake Winnipeg. Master of Resource Management Practicum, University of Manitoba. 103 pp.

Hinks, D. 1943. The fishes of Manitoba. Department of Mines and Natural Resources, Province of Manitoba. Winnipeg, Manitoba.

Horgan, M.J. and E.L. Mills. 1997. Clearance rates and filtering activity of zebra mussel (*Dreissena polymorpha*): implications for freshwater lakes. Canadian Journal of Fisheries and Aquatic Sciences 54: 249-255.

House, W.A., D. Leach, M.S. Warwick, B.A. Whitton, S.N. Pattinson, G. Ryland, A. Pinder, J. Ingram, J.P. Lishman, S.M. Smith, E. Rigg, and F.H. Denison. 1997. Nutrient transport in the Humber rivers. The Science of the Total Environment 194/195: 303-320.

Howell, J.M, M.B. Coyne, and P.L. Cornelius. 1996. Effect of sediment particle size and temperature on fecal bacteria mortality rates and the fecal coliform/fecal Streptococci ratio. Journal of Environmental Quality 25: 1216-1220.

Hughes, C. 1983. A report on the trophic status of the South Basin of Lake Winnipeg (1980-1982). Water Standards and Studies Section, Environmental Management Services Branch, Environmental Management Division, Winnipeg, Manitoba. Water Standards and Studies 83-10.

International Red River Board. 2008. Ninth Annual Progress Report October 2008. Report to the International Joint Commission. 50 pp.

Ishii, S.D., W.B. Winfried, R.E. Hicks, and M.J. Sandowsky. 2006. Presence and growth of naturalized *Escherichia coli* in temperate soils from Lake Superior watersheds. 2006. Appl. Environ. Microbiol.72(1):612-621.

Ishii, S., D.L. Hansen, R.E. Hicks, and M.J. Sadowsky. 2007. Beach sand and sediments are temporal sinks and sources of *Escherichia coli* in Lake Superior. Environmental Science and Technology 41: 2203-2209.

Jamieson R.C., R.J. Gordon, S.C. Tattrie, and G.W. Stratton. 2003. Sources and persistance of fecal coliform bacteria in a rural watershed. Water Qual. Res. J. Canada. 38(1):33-47.

Jarnagin, S.T., B.K. Swan, and W.C. Kerfoot. 2000. Fish as vectors in the dispersal of *Bythotrephes cederstroemi*: diapausing eggs survive passage through the gut. Freshwater Biology 43: 579-589.

Jarvie, H.P., C. Neal, and A.D. Tappin. 1997. European land-based pollutant loads to the North Sea: an analysis of the Paris Commission data and review of monitoring strategies. The Science of the Total Environment 1994/1995: 39-58.

Jeppesen, E., J.P. Jensen, M. Søndergaard, T. Lauridsen, and F. Landkildehus. 2000. Trophic structure, species richness and biodiversity in Danish lakes: changes along a phosphorus gradient. Freshwater Biology 45: 201-218.

Johnston, T.A., W. Lysack, and W.C. Leggett. In review. Abundance, growth and life history characteristics of sympatric walleye (*Sander vitreus*) and sauger (*S. canadensis*) in Lake Winnipeg, Manitoba. Journal of Great Lakes Research.

Jones, G. 1998. Blue-green algae and microcystin-LR in Surface Water Supplies of Southwestern Manitoba. Manitoba Environment, Water Quality Management Section, Winnipeg, Manitoba. CA2 ME r 98-06.
Jones, G. 1999. Microcystin-LR in Municipal Surface Water Supplies of Southern Manitoba, June 1996-February 1999. Manitoba Environment, Water Quality Management Section, Winnipeg, Manitoba. CA2 ME r 99-08.

Jones, G. 2002. Phytoplankton and microcystin-LR in the Assiniboine River at Portage la Prairie, Manitoba 1997-2001. Manitoba Conservation, Water Quality Management Section. Winnipeg, Manitoba. CA2 MC r 2002-06.

Jones, G. and N. Armstrong. 2001. Long-term trends in total nitrogen and total phosphorus concentrations in Manitoba streams. Water Quality Management Section, Water Branch, Manitoba Conservation, Winnipeg, MB. Manitoba Conservation Report No. 2001-07. 154 pp.

Kalff, J. 2002. Limnology. Prentice-Hall Inc., Englewood Cliffs, New Jersey.

Kane, D.D., J.E. Gannon, and D.A. Culver. 2004. The Status of *Limnocalanus macrurus* (Copepoda: Calanoida: Centropagidae) in Lake Erie. Journal of Great Lakes Research 30: 22-30.

Keleher, J.J. 1970. The contamination of Canadian fish by mercury. A Bulletin for Commercial Fishermen, Fishing, Manitoba Department of Mines and Natural Resources 4: 14-20.

King, D.R. and G.S. Hunt. 1967. Effect of carp on vegetation in a Lake Erie marsh. Journal of Wildlife Management 31(1): 181-188.

Kling, H.J. 1996. Fossil and modern phytoplankton from Lake Winnipeg. In: Lake Winnipeg Project Cruise Report and scientific results. (Eds.) B.J. Todd, M. Lewis, L.H. Thorleifson, and E. Neilsen. Geological Survey of Canada Open file 3113. 283-310 pp.

Kling, H.J. 1998. A summary of past and recent plankton of Lake Winnipeg, Canada using algal fossil remains. Journal of Paleolimnology 19(3): 297-307.

Kling, H.J., C. Herbert, M. Stainton, L. Hendzel, G. McCullough, M. Satchwell, G. Boyer, and A. Salki. 2007. Phytoplankton changes in Lake Winnipeg: cyanobacteria, nutrients and toxin issues. IAGLR, Montreal 12-18 Aug. 2007.

Kon, T., S.C. Weir, E.T. Howell, H. Lee, and J.T. Trevors. 2007. Genetic relatedness of *Escherichia coli* isolates in interstitial water from a Lake Huron (Canada) beach. Applied and Environmental Microbiology 73: 1961-1967.

Kotak, B.G., A.K-Y. Lam, E.E. Prepas, S.L. Kenefick, and S.E. Hrudey. 1995. Variability of the hepatotoxin microcystin-LR in hypereutrophic drinking water lakes. Journal of Phycology 31: 248-263.

Kotak, B.G., R.W. Zurawell, E.E. Prepas, and C.F.B. Holmes. 1996a. Microcystin-LR concentration in aquatic food web compartments from lakes of varying trophic status. Canadian Journal of Fisheries and Aquatic Sciences 53: 1974-1985.

Kotak, B.G., S. Semalulu, D.L. Fritz, E.E. Prepas, and R.W. Coppock. 1996b. Hepatic and renal pathology of intraperitoneally administered microcystin-LR in rainbow trout (*Oncorhynchus mykiss*). Toxicon 34: 517-525.

Kotak, B.G., A.K-Y. Lam, E.E. Prepas, and S.E. Hrudey. 2000. Role of chemical and physical variables in regulating microcystin-LR concentration in phytoplankton of eutrophic lakes. Canadian Journal of Fisheries and Aquatic Sciences 57: 1584-1593.

Kotak, B.G. and R.W. Zurawell. 2007. Cyanobacterial toxins in Canadian freshwaters: A review. Lake and Reservoir Management. 23: 109-122.

Kotak, B.G. 2009. Occurrence of the cyanobacterial toxin, microcystin, in Lake Winnipeg in 2007. Manitoba Water Stewardship Project 26037 Report. 24 pp.

Krieger, K.A., M.T. Bur, J.J.H. Ciborowski, D.R. Barton, and D.W. Schloesser. 2007. Distribution and abundance of burrowing mayflies (*Hexagenia spp*.) in Lake Erie, 1997-2005. Journal of Great Lakes Research 33: 20-33.

Kristofferson, A.H., D.R. Toews, and A.J. Derksen. 1975. Limnological Study of the North Basin of Lake Winnipeg, 1974. Manitoba Department of Mines Resources and Environment Management, Research Branch, Manuscript Report No. 75-5, 54 pp.

Lake of the Woods Control Board. 2002. Managing the water resources of the Winnipeg River drainage basin. Lake of the Woods Control Board, Ottawa, Canada. 20 pp.

Lake Simcoe Region Conservation Authority. 2009. Report on the Phosphorus Loads to Lake Simcoe. http://www.ene.gov.on.ca/en/water/lakesimcoe/docs/LSRCAPhosphorusReport.pdf

Lake Superior Centre. 1998. More Than Just a Lake. Lake Effects - The Lake Superior Curriculum Guide. http://www.coseegreatlakes.net/downloads/more_than_just_a_lake.pdf

Lake Winnipeg Implementation Committee (LWIC). 2005. Restoring the health of Lake Winnipeg. Lake Winnipeg Implementation Committee, Winnipeg, Manitoba.

Lake Winnipeg Stewardship Board. 2006. Reducing Nutrient Loading to Lake Winnipeg and its Watershed. Our Collective Responsibility and Commitment to Action. Report to the Minister of Water Stewardship. 78 pp.

Larsen, D.P. and H.T. Mercier. 1976. Phosphorus retention capacity of lakes. Journal of Fisheries Research Board of Canada 33: 1742-1750.

Lau, S.S.S. and S.N. Lane. 2002. Biological and chemical factors influencing shallow lake eutrophication: A long-term study. Science of the Total Environment 288(3): 167-181.

Lauber, C.L., L. Glatzer, and R.L. Sinsabaugh. 2003. Prevalence of pathogenic *Escherichia coli* in recreational waters. Journal of Great Lakes Research 29(2): 301-306.

Laugaste, R., V.V. Jastremskij, and I. Ott. 1996. Phytoplankton of Lake Peipsi-Pihkva: species composition, biomass and seasonal dynamics. Hydrobiologia 338: 49-62.

Leach, J.H., M.G. Johnson, J.R.M. Kelso, J. Hartmann, W. Nümann, and B. Entz. 1977. Response of percid fishes and their habitats to eutrophication. Journal of Fisheries Research Board of Canada 34: 1964-1971.

Leach, J.H., L.M. Dickie, B.J. Shuter, U. Borgmann, J. Hyman, and W. Lysack. 1987. A review of methods for prediction of potential fish production with application to the Great Lakes and Lake Winnipeg. Canadian Journal of Fisheries and Aquatic Sciences 44(Suppl. 2): 471-485.

Lester, N.P., A.J. Dextrase, R.S. Kushneriuk, M.R. Rawson, and P.A. Ryan. 2004. Light and temperature: key factors affecting walleye abundance and production. Transactions of the American Fisheries Society 133: 588-605.

Levine, S.N. and W.M. Lewis. 1984. Diel variation of nitrogen fixation in Lake Valencia, Venezuela. Limnology and Oceanography 29: 887-893.

Lewis, W.M. 1983. A revised classification of lakes based on mixing. Canadian Journal of Fisheries and Aquatic Sciences 40: 1779-1787.

Lieder, U. 1983. Revision of the genus Bosmina BAIRD, 1845 (Crustacea, Cladocera). Int. Revue ges. Hydrobiology 68: 121-139.

Lieder, U. 1991. The *Bosmina kessleri*-like morphotype of *Eubosmina* in Lake Muskoka, Ontario, Canada, as putative interspecific hybrids. Hydrobiologia 225:71-80.

Lindon, M. and S. Heiskary. 2009. Blue-green algal toxin (microcystin) levels in Minnesota lakes. Lake and Reservoir Management 25: 240-252.

Lorenz, D.L., D.M. Robertson, D.W. Hall, and D.A. Saad. 2009. Trends in streamflow and nutrient and suspended-sediment concentrations and loads in the Upper Mississippi, Ohio, Red, and Great Lakes River Basins, 1975-2004. U.S. Geological Survey Scientific Investigations Report 2008-5213, 81 pp.

Lowe, C.W. 1924. The freshwater algae of Canada. Transactions of the Royal Society of Canada Section V. II pg 19-47 + plates I - IV.

Ludsin, S.A., M.W. Kershner, K.A. Blocksom, R.L. Knight, and R.A. Stein. 2001. Life after death in Lake Erie: nutrient controls drive fish species richness, rehabilitation. Ecological Applications 11: 731-746.

Lund, J.W.G., C. Kipling, and E.D. Le Cren. 1958. The inverted microscope method of estimating algal numbers and the statistical basis of estimations by counting. Hydrobiologia 11: 143-170.

Lysack, W. 2005. The Lake Winnipeg fisheries. Report to the Canadian Climate Action Fund Project A499. 51 pp.

Mackie, G.L. and R. Claudi. 2009. Monitoring and control of macrofouling mollusks in fresh water systems, Second Edition. CRC Press, Boca Raton, FL. 488 pp.

Magnuson, J.J., D.M. Robertson, B.J. Benson, R.H. Wynne, D.M. Livingstone, T. Arai, R.A. Assel, R.G. Barry, V. Card, E. Kuusisto, N.G. Granin, T.D. Prowse, K.M. Stewart, and V.S. Vuglinski. Historical trends in lake and river ice cover in the northern Hemisphere. Science 289: 1743-1746.

Makarewicz, J.C. 1993. Phytoplankton biomass and species composition in Lake Erie, 1970 to 1987. Journal of Great Lakes Research 19: 258-274.

Manitoba Department of Mines, Resources and Environmental Management (MMREM). 1973. Water quality study, south portion of Lake Winnipeg, 1973. Manitoba Department of Mines, Resources, and Environmental Management, Environmental Management Division, Environmental Protection Branch, Report No. 24.

Manitoba Hydro-electric Board. 2006.

Manitoba Water Stewardship 2008. Manitoba Water Stewardship 2007-2008 Annual Report, Winnipeg. 84-85 pp.

Manitoba Conservation. 2001. Long-term phosphorous trends in total nitrogen and total phosphorus concentrations in Manitoba streams. Water Quality Management Section, Water Branch, Manitoba. 154 pp.

Manitoba Water Stewardship A. March 2009. Koi Herpes virus confirmed in Lake Manitoba carp. Information fact sheet.

Manitoba Water Stewardship B. September 2009. Koi Herpes virus confirmed in Lake Manitoba carp. Press Release.

Manitoba Water Stewardship, Surface Water Management Section. 2010. Estimation of Red River at Lockport stream flow.

Marcogliese, D.J. 2008. First report of the Asian fish tapeworm in the Great Lakes. Journal of Great Lakes Research 34(3): 566-569.

Marshall, C.T. and R.H. Peters. 1989. General patterns in the seasonal development of chlorophyll a for temperate lakes. Limnology and Oceanography 34(5):856-867.

Mayer, T., S.L. Simpson, L.H. Thorleifson, W.L. Lockhart, and P. Wilkinson. 2006. Phosphorus geochemistry of recent sediments in the South Basin of Lake Winnipeg. Aquatic Ecosystem Health and Management. 9 (2006) 307-318.

McCambridge, J. and T.A. McMeekin. 1981. Effect of solar radiation and predacious microorganisms on survival of fecal and other bacteria. Applied Environmental Microbiology 41(5): 1083-1087.

McCauley, E., J.A. Downing, and S. Watson. 1989. Sigmoid relationships between nutrients and chlorophyll among lakes. Canadian Journal of Fisheries and Aquatic Sciences 46: 1171-1175.

McCormick, M.J. and G.L. Fahnenstiel. 1999. Recent climatic trends in nearshore water temperatures in the St. Laurence Great Lakes. Limnology and Oceanography 44(3): 530-540.

McCullough, G. 2001. Organic carbon, nitrogen and phosphorous fluxes in rivers flowing into and out of Lake Winnipeg. Final Report. Prepared for the Canada Department of Fisheries and Oceans, Winnipeg, Manitoba. 46 pp. plus appendices.

McCullough, G. 2009. Satellite images of Manitoba's great lakes. Available from http://home.cc.umanitoba.ca/~gmccullo/LWsat.htm [accessed 29 May 2009].

McCullough, G.K., K. Hocheim, and P.M. Cooley. 2001. Retrospective study of suspended sediment patterns on Lake Winnipeg using NOAA AVHRR Satellite Imagery. Report to the Department of Fisheries and Oceans, Winnipeg, Canada. v+72 pp.+4 appendices.

McCullough, G.K., M.P. Stainton, and H.J. Kling. 2004. Environmental controls of algal blooms in Lake Winnipeg. 47th Ann. International Association of Great Lakes Research Conference. Waterloo, Ontario. 24-28 May 2004.

McCullough, G.K. 2005. Surface water temperature and break-up and freeze-up of the ice cover on Lake Winnipeg. Report to the Canadian Climate Action Fund Project A499. Canadian Department of Fisheries and Oceans. v + 42 pp.

McCullough, G.K., M. Stainton, and H.J. Kling. 2006. Environmental controls of algal blooms in Lake Winnipeg. Canada Water Resources Forum: Managing Nutrients in the Lake Winnipeg Watershed. Winnipeg, Canada.18, 19 October 2006.

McCullough, G.K. 2007. MERIS/MODIS prediction of chlorophyll in Lake Winnipeg. Report to the Canadian Department of Fisheries and Oceans, Institute of Ocean Sciences, Nanaimo, British Columbia (Dr. Jim Gower, Contract Scientific Authority). iv + 45 pp.

McLellan, S.L. and A.K. Salmore. 2003. Evidence for localized bacterial loading as the cause of chronic beach closings in a freshwater marina. Water Research 37: 2700-2708.

McLeod, S. 2007. Characterization of the urban runoff quality from the City of Saskatoon to the South Saskatchewan River. Masters Thesis. Department of Civil & Geological Engineering, University of Saskatchewan, Saskatoon.

Mendes, B., M.J. Nascimento, and J.S. Oliveira. 1993. Preliminary characterisation and proposal of microbiological quality standard of sand beaches. Water Science and Technology 27(3-4): 453-456.

Meriluoto, J.A.O. and J.E. Eriksson. 1988. Rapid analysis of peptide toxins in cyanobacteria. Journal of Chromatography 438: 93-99.

Milius, A., R. Laugaste, T. Möls, M. Haldna, and K. Kangur. 2005. Water level and water temperature as factors determining phytoplankton biomass and nutrient content in Lake Peipsi. Proceedings of the Estonian Academy of Sciences 54(1): 5-17.

Millard, S. and T. Howell. 2009. Phosphorus concentrations and loadings: Indicator #111. In Environment Canada and the United States Environmental Protection Agency. State of the Great Lakes 2009. Toronto and Chicago. 443 pp.

Mitch, A.A, K.C. Gasner, and W.A. Mitch. 2010. Fecal coliform accumulation within a river subject to seasonally-disinfected wastewater discharges. Water Res. 44:4776-4782.

Mohamed, A., W.W. Carmichael, and A.A. Hussein. 2003. Estimation of microcystins in the freshwater fish *Oreochromis niloticus* in an Egyptian fish farm containing a Microcystis bloom Zakaria. Environmental Toxicology 18: 137-141.

Munawar, M. and I.F. Munawar. 1982. Phycological studies in Lakes Ontario, Erie, Huron, and Superior. Canadian Journal of Botany 60: 1837-1858.

Nalepa, T.F., D.L. Fanslow, M.B. Lansing, and G.A. Lang. 2003. Trends in the benthic macroinvertebrate community of Saginaw Bay, Lake Huron, 1987 to 1996: Responses to phosphorus abatement and the zebra mussel, *Dreissena polymorpha*. Journal of Great Lakes Research 29: 14-33.

Nalepa T.F., D.L. Fanslow, S.A. Pothoven, A. J. Foley, and G. A. Lang. 2007. Long-term trends in benthic macroinvertebrate populations in Lake Huron over the past four decades. Journal of Great Lakes Research 33: 421-436.

Neave, F. 1932. A study of the mayflies (Hexagenia) of Lake Winnipeg. Contributions to Canadian Biology and Fisheries 7(15): 177-201.

Neave, F. 1933. Ecology of two species of Trichoptera in Lake Winnipeg. International Revue der gesamten Hydrobiologie und Hydrographie 31: 17-28.

Neave, F. 1934. A contribution to the aquatic fauna of Lake Winnipeg. International Revue der gesamten Hydrobiologie und Hydrographie 31: 157-170.

Neilson, M., S. L'Italien, V. Glumac, D. Williams, and P. Bertram. 1995. Nutrients: Trends and system response. State of the Lakes Ecosystem Conference Background Paper. Environment Canada and U.S. Environmental Protection Agency. EPA 905-R95-015. 34 pp.

Noges, T., L. Tuvikene, and P. Noges. 2010. Contemporary trends of temperature, nutrient loading, and water quality in large Lakes Peipsi and Võrtsjärv, Estonia. Aquatic Ecosystem Health and Management 13: 143-153.

North/South Consultants Inc. 2006. Literature review related to setting nutrient objectives for Lake Winnipeg. Report prepared for Manitoba Water Stewardship. 186 pp.

Odum, H.T. 1956. Primary production in flowing waters. Limnology and Oceanography 1(2): 102-117.

Ontario Ministry of Environment. 2010.

Ontario Ministry of Natural Resources. 2005.

Ontkean, G.R., D.S. Chanasyk, and D.R. Bennett. 2005. Snowmelt and growing season phosphorus flux in an agricultural watershed in south-central Alberta, Canada. Water Quality Research Journal of Canada 40(4): 402-417.

Organisation for Economic Cooperation and Development. 1982. Eutrophication of waters. Monitoring, assessment and control. OECD, Paris, 154 pp.

Park, S., M.T. Brett, A. Müller-Solger, and C.R. Goldman. 2004. Climatic forcing and primary productivity in a subalpine lake: Interannual variability as a natural experiment Limnology and Oceanography 49(2): 614-619.

Parks, C.R. 2006. Experimental manipulation of connectivity and common carp; the effects on native fish, water-column invertebrates and amphibians in Delta Marsh, Manitoba. Department of Environment and Geography. University of Manitoba, Winnipeg, Manitoba.

Partners for the Saskatchewan River Basin. 2009. From the Mountains to the Sea. Summary of the State of the Saskatchewan River Basin. 16 pp.

Patalas, K. 1981. Spatial structure of the crustacean planktonic community in Lake Winnipeg, Canada. Verhandlungen Internationale Vereinigung Limnologie 21: 305-311.

Patalas, K. and A. Salki. 1992. Crustacean plankton in Lake Winnipeg: variation in space and time as a function of lake morphology, geology, and climate. Canadian Journal of Fisheries and Aquatic Science 49: 1035-1059.

Patterson, J.C., B.R. Allanson, and G.N. Ivey. 1985. A dissolved oxygen budget model for Lake Erie in summer. Freshwater Biology 15(6): 683-694.

Phillips, K.A. and M.W. Fawley. 2002. Winter phytoplankton community structure in three shallow temperate lakes during ice cover. Hydrobiologia 470: 97-113.

Pick, F.R. and D.R.S. Lean. 1987. The role of macronutrients (C,N,P) in controlling cyanobacterial dominance in temperate lakes. New Zealand Journal of Marine and Freshwater Research 21: 425-434.

Pollard, W.R. 1973. Limnological investigations of the north basin of Lake Winnipeg - summer, 1968. Manitoba Department of Mines, Resources, and Environmental Management, Research Branch, Environmental Management Division, Winnipeg, Manitoba. Manuscript Report No. 73-28.

Prairie Farm Rehabilitation Administration. 2008. PFRA Watershed Project Version 8.

Prepas, E.E., B.G. Kotak, L.M. Campbell, J.C. Evans, S.E. Hrudey, and C.F.B. Holmes. 1997. Accumulation and elimination of cyanobacterial hepatotoxins by the freshwater clam, *Anodonta grandis simpsoniana*. Canadian Journal of Fisheries and Aquatic Sciences 54: 41-46.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman. 2002. Gulf of Mexico hypoxia, aka "The dead zone". Annual Review of Ecology and Systematics 33: 235-263.

Ralley, Wendy. 2002. Alien aquatic species in Manitoba: present and threatening. Claudi, R., P. Nantel, E. Muckle-Jeffs (eds). Alien Invaders in Canada's Waters, Wetlands and Forests. Canadian Forest Service, Natural Resources Canada, Ottawa, ON. 93-102 pp.

Rannie, W.F. 1983. Breakup and freezeup of the Red River near Winnipeg, Manitoba in the 19th century and some climatic implications. Climatic Change: 283-296.

Rathke, D.E. and C.J. Edwards (Ed.) 1985. A Review of Trends in Lake Erie Water Quality with Emphasis on the 1978-1979 Intensive Survey. Report to the International Joint Commission. http://gis.lrs.uoguelph.ca/AgriEnvArchives/download/L-Erie_wat_qual_1978-79.pdf

Redfield, A.C., B.H. Ketchum, and F.A. Richards. 1963. The influence of organisms on the composition of sea-water. In: Hill, N. (Ed.), In: The Sea, 2nd edition. Wiley, New York, USA, 26-77 pp.

Remnant, R.A. 1991. An assessment of the potential impact of rainbow smelt on the fishery resources of Lake Winnipeg. MNRM Practicum. University of Manitoba, Winnipeg, Manitoba. 170 pp.

Rengefors, K. and D.M. Anderson. 1998. Environmental and endogenous regulation of cyst germination in two freshwater dinoflagellates. Journal of Phycology 34: 568-577.

Resh, V.M. and D.M. Rosenberg. 2010. Recent trends in life-history research on benthic macroinvertebrates. Journal of the North American Benthological Society 29: 207-219.

Reynolds, C.S. 1984. The ecology of freshwater phytoplankton. Cambridge University Press, Cambridge and New York. 384 pp.

Reynolds, C.S. 1998. What factors influence the species composition of phytoplankton in lakes of different trophic status? Hydrobiologia 369/370: 11-26.

Robel, R.J. 1961. The effects of carp populations on the production of waterfowl food plants on a western waterfowl marsh. Transactions of the North American Wildlife Conference 26: 147-159.

Robillard, M.M. and M.G. Fox. 2006. Historical changes in abundance and community structure of warmwater piscivore communities associated with changes in water clarity, nutrients, and temperature. Canadian Journal of Fisheries and Aquatic Sciences 63: 798-809.

Rockwell, D.C., J.W. Glenn, P.E. Bertram, D.K. Salisbury, and N.M. Burns. 2005. The U.S. EPA Lake Erie indicators monitoring program 1983-2002: Trends in phosphorus, silica, and chlorophyll a in the central basin. Journal of Great Lakes Research 31(Suppl. 2): 23-34.

Rosa, F. and N.M. Burns. 1987. Lake Erie central basin oxygen depletion changes from 1929-1980. Journal of Great Lakes Research 13(4): 684-696.

Rott, E. 1981. Some results from phytoplankton counting intercalibrations. Schweizerische Zeitschrift fur Hydologie. 43: 43-62.

Ruddy, B.C., D.L. Lorenz, and D.K. Mueller. 2006. County-level estimates of nutrient inputs to the land surface of the conterminous United States, 1982-2001. USGS Scientific Investigations Report 2006-5012, Reston, VA.

Rybicki, R.W. 1966. Limnological survey of the north basin of Lake Winnipeg 1963 and 1964. Resources, M.D.o.M.a.N. (Editor), Winnipeg.

Ryder, R.A. 1977. Effects of ambient light variations on behavior of yearling, subadult, and adult walleyes (*Stizostedion vitreum vitreum*). Journal of the Fisheries Research Board of Canada 34: 1481-1491.

Sagher, A., R.F. Harris, and D.E. Armstrong. 1975. Availability of sediment phosphorus to microorganisms. Technical Report WIS WRC 75-01, Water Resources Center, Uni. Of Wisconsin, Madison, USA.

Sagher, A. 1976. Availability of Soil Runoff Phosphorus to Algae. Ph.D. Thesis, Uni. Of Wisconsin, Madison, USA.

Sakamoto, M. 1966. Primary production by phytoplankton community in some Japanese lakes and its dependence on lake depth. Archiv fur Hydrobiologie 62: 1-28.

Salki, A., G.K. McCullough, M.P. Stainton, W. Franzin, and K. Patalas. 2006. Response of Lake Winnipeg zooplankton to habitat change. Lake Winnipeg Research Consortium Science Seminar, Freshwater Institute Winnipeg.

Scheffer, M., S. Rinaldi, A. Gragnani, L.R. Mur, and E.H. van Nes. 1997. On the dominance of filamentous Cyanobacteria in shallow, turbid lakes. Ecology 78(1): 272-282.

Schindler, D.W. 2009. Lakes as sentinels and integrators for the effects of climate change on watersheds, airsheds, and landscapes. Limnology and Oceanography 54(6, part 2): 2349-2358.

Scott, L.D., J.G. Winter, and R.E. Girard. 2006. Annual water balances, total phosphorus budgets and total nitrogen and chloride loads for Lake Simcoe (1998-2004). Lake Simcoe Environmental Management Strategy Implementation Phase III. Technical Report. Ontario. 141 pp.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Bulletin of the Fisheries Research Board of Canada. 184, 966 pp.

Shay, J.M., P.M.J. de Geus, and M.R.M. Kapinga. 1999. Changes in shoreline vegetation over a 50-year period in the Delta Marsh, Manitoba in response to water levels. Wetlands 19(2): 413-435.

Simpson, S.L., L.H. Thorleifson, C.F.M. Lewis, and J.W. King. 2003. 1999 Lake Winnipeg Project: Cruise report and scientific results. Natural Resources Canada, Ottawa, Ontario. Geological Survey of Canada: Open File 4196.

Smith, V.H. 1983. Low nitrogen to phosphorus ratios favour dominance by blue-green algae in lake phytoplankton. Science 221: 669-671.

Sondergaard, M., J.P. Jensen, and E. Jeppesen. 2001. Retention and internal loading of phosphorus in shallow, eutrophic lakes. Scientific World 1: 427-442.

Sondergaard, M., E. Jeppesen, and J.P. Jensen. 2003. Internal phosphorus loading and the resilience of Danish Lakes. LakeLine 23(1): 17-20.

Sorba, E.A. and D.A. Williamson. 1997. Zebra mussel colonization potential in Manitoba, Canada. Water Quality Management Section. Manitoba Environment. Report No. 97-07.

St. George, S. 2006. Hydrological dynamics in the Winnipeg River basin, Manitoba. In Report on Activities 2006, Manitoba Science, Technology, Energy and Mines, Manitoba Geological Survey. 226-230 pp.

Stainton, M. 2005. Lake Winnipeg 2003 chemistry assessment. Lake Winnipeg Research 2003: Report to Manitoba Hydro. The Lake Winnipeg Research Consortium Inc, Winnipeg.

Stainton, M.P. and G.K. McCullough. 2004. Investigations of water chemistry in Lake Winnipeg. In Lake Winnipeg Research 2003: Report to Manitoba Hydro. The Lake Winnipeg Research Consortium Inc., Winnipeg, Manitoba.

Stainton, M.P., G.K. McCullough, R.H. Hesslein, and S.J. Page. 2008. Effects of climate change on phosphorous and nitrogen loading to Lake Winnipeg. IAGLR, Peterborough. 19-23 May 2008.

State of the Lakes Ecosystem Conference 1998 (SOLEC). 1999. Selection of Indicators for Great Lakes Basin Ecosystem Health, Version 3 Draft for Review, Prepared by: Paul Bertram and Nancy Stadler-Salt. United States Environmental Protection Agency. December 2009; http://www.epa.gov/solec/solec_1998/Selection_of_Indicators_for_Great_Lakes_Ecosystem_Health_Version_3_%28FULL%29.pdf

State of the Lakes Ecosystem Conference 2008 (SOLEC). 2008. Nearshore Areas of The Great Lakes background report. http://www.solecregistration.ca/en/nearshore.asp

Statistics Canada. 2007. Canadian Environmental Sustainability Indicators: Socio-economic Information. Statistics Canada. http://www.statcan.gc.ca/pub/16-253-x/16-253-x2007000-eng.pdf

Stewart, A.R., G.A. Stern, A. Salki, M.P. Stainton, W.L. Lockhart, B.N. Billeck, R. Danell, J. Delaronde, N.P. Grift, T. Halldorson, K. Koczanski, A. MacHutcheon, B. Rosenberg, D. Savoie, D. Tenkula, G. Tomy, and A. Yarchewski. 2000. Influence of the 1997 Red River flood on contaminant transport and fate in southern Lake Winnipeg. International Red River Basin Task Force, Winnipeg, Manitoba. 190 pp.

Stewart, K.W. 2000. Rainbow smelt has negative effects. The Winnipeg Free Press.

Stewart, K. and D. Watkinson. 2004. Freshwater Fishes of Manitoba. University of Manitoba Press, Winnipeg, MB. 276 pp.

Suchy, K.D. and B.J. Hann. 2007. Using microfossil remains in lake sediments to examine the invasion of *Eubosmina coregoni* (Cladocera, Bosminidae) in Lake of the Woods, Ontario, Canada. Journal of Great Lakes Research 33(4): 867-874.

Swain, D.P. 1979. Biology of the carp (*Cyprinus carpio L.*) in North America and its distribution in Manitoba, North Dakota and Neighbouring U.S. waters. Manitoba Dept. Mines, Natural Resources and Environment. Manuscript Report No. 79-73. 36 pp.

The Centers for Disease Control and Prevention (CDC). 2008. Surveillance for Waterborne Disease and Outbreaks Associated with Recreational Water Use and Other Aquatic Facility-Associated Health Events - United States, 2005 to 2006. Morbidity and Mortalities Weekly Report 57(SS09): 1-29.

The Lake Manitoba Regulation Review Advisory Committee. 2003. Regulation of water levels on Lake Manitoba and along the Fairford River, Pineimuta Lake, Lake St. Martin and Dauphin River and related issues. A report to the Manitoba Minister of Conservation. Volume 2: Main Report. 99 pp.

Todd, B.J., C.F.M. Lewis, L.H. Thorleifson, and E. Nielsen. 1996. Lake Winnipeg project: Cruise report and scientific results. Natural Resources Canada, Ottawa, Ontario. Geological Survey of Canada: Open File No. 3113.

Todd, B.J., C.F.M.Lewis, D.L. Forbes, L.H. Thorleifson, and E. Nielsen. 2000. 1996 Lake Winnipeg Project: Cruise report and scientific results. Natural Resources Canada, Ottawa, Ontario. Geological Survey of Canada: Open File 3470.

Ueno, Y., S. Nagata, T. Tsutsumi, A. Hasegawa, M.F. Watanabe, H.D. Park, G.C. Chen, G. Chen, and S.Z. Yu. 1996. Detection of microcystins, a blue-green algal hepatotoxin, in drinking water sampled in Haimen and Fusui, endemic areas of primary liver cancer in China, by highly sensitive immunoassay. Carcinogenesis 17: 1317-1321.

US Environmental Protection Agency. 1986. Ambient Water Quality Criteria for Bacteria 1986. Office of Water. EPA440/5-84-002. 18 pp.

Vanderploeg, H.A., J.R. Liebig, W.W. Carmicheal, M.A. Agy, T.H. Johengen, G.F. Fahnenstiel, and T.F. Nalepa. 2001. Zebra mussel (*Dreissena polymorpha*) selective filtration promoted toxic microcystis blooms in Saginaw Bay (Lake Huron) and Lake Erie. Canadian Journal of Fisheries and Aquatic Sciences 58: 1208-1221.

Vincent, L.A., X. Zhang, B.R. Bonsal, and W.D. Hogg. 2002. Homogenization of daily temperatures over Canada. Journal of Climate 15: 1322-1334.

Wahl, E.W. and T.L. Lawson. 1970. The climate of the mid-nineteenth century United States compared to the current normals. Monthly Weather Review 98: 259-265.

Wain, D.B. 1993. The effects of introduced rainbow smelt (*Osmerus mordax*) on the indigenous pelagic fish community of an oligotrophic lake. M.Sc. Thesis. University of Manitoba, Winnipeg, Manitoba. 131 pp.

Wang, S.H., A.R. Dzialowski, J.O. Meyer, F. deNoyelles, N.C. Lim, W.W. Spotts, and D.H. Huggins. 2005. Relationships between cyanobacterial production and the physical and chemical properties of a Midwestern Reservoir, USA. Hydrobiologia 541: 29-43.

Water Survey of Canada. 2010. Hydat. Online database. http://www.wsc.ec.gc.ca/hydat/H2O/index_e.cfm?cname=main_e.cfm

Webster, K.E., P.A. Soranno, K.S. Cheruvelil, M.T. Bremigan, J.A. Downing, P.D. Vaux, T.R. Asplund, L.C. Bacon, and J.A. Connor. 2008. An empirical evaluation of the nutrient-color paradigm for lakes. Limnology and Oceanography 53: 1137-1148.

Westmacott, J.R. and D.H. Burn. 1997. Climate change effects on the hydrologic regime within the Churchill-Nelson River Basin. Journal of Hydrology 202: 263-279.

Wetzel, R.G. 2001. Limnology: Lake and river ecosystems. Academic Press, San Diego, U.S.A. 3rd edition. xvi + 1006 pp.

Whiteman, R. L., M.B. Nevers, and M. Byappanahalli. 2006. Examination of the watershed-wide distribution of *Escherichia coli* along southern Lake Michigan: an integrated approach. Applied Environmental Microbiology. 72(11):7301-7310.

Whitman, R.L. and M.B. Nevers. 2003. Foreshore sand as a source of *Escherichia coli* in nearshore water of a Lake Michigan beach. Applied Environmental Microbiology 69(9): 5555-5562.

Whitman, R.L. and M.B. Nevers. 2004. *Escherichia coli* sampling reliability at a frequently closed Chicago beach: Monitoring and management implications. Environmental Science and Technology 38(16): 4241-4246.

Whitman, R.L., M.B. Nevers, G.C. Korinek, and M.N. Byappanahalli. 2004. Solar and temporal effects on *Escherichia coli* concentration at a Lake Michigan swimming beach. Applied Environmental Microbiology 70(7): 4276-4285.

Williamson, D.A. 1985. Bacteriological characteristics of twenty recreational beaches Manitoba, Canada 1984. Manitoba Environment and Workplace Safety and Health. Water Standards and Studies Report No. 85-3.

Williamson, D.A. 1988. Data Supplement: A four year study of recreational beaches, Manitoba, Canada. Manitoba Environment and Workplace Safety and Health. Water Standards and Studies Report No. 88-3.

Williamson, D.A. 2002. Manitoba water quality standards, objectives, and guidelines. Manitoba Conservation Report 2002-11. 76 pp.

Williamson, D.A., W.E. Ralley, A. Bourne, N. Armstrong, R. Fortin, and C.E. Hughes. 2004. Principal factors affecting *Escherichia coli* densities at Lake Winnipeg beaches. Water Quality Management Section. Manitoba Water Stewardship. Manitoba Water Stewardship Report No. 2004-01.

Wilson, K.A., J.J. Magnuson, D.M. Lodge, A.M. Hill, T.K. Kratz, W.L. Perry, and T.V. Willis. 2004. A long-term rusty crayfish *(Orconectes rusticus*) invasion: dispersal patterns and community change in a north temperate lake. Canadian Journal of Fisheries and Aquatic Sciences 61(11): 2255-2266.

Wilson, A.E., D.C. Gossiaux, T.O. Hook, J.P. Berry, P.F. Landrum, J. Dyble, and S.J. Guildford. 2008. Evaluation of the human health threat associated with the hepatotoxin microcystin in the muscle and liver tissues of yellow perch *(Perca flavescens*) Canadian Journal of Fisheries and Aquatic Sciences 65: 1487-1497.

Winter, J.G., M.C. Eimers, P.J. Dillon, L.D. Scott, W.A. Scheider, and C.C. Willox. 2007. Phosphorus inputs to Lake Simcoe from 1990 to 2003: declines in tributary loads and observations on lake water quality. Journal of Great Lakes Research 33: 381-396.

Wrubleski, D.A. 1998. The fish community of Delta Marsh: a review. Institute for Wetland and Waterfowl Research. Ducks Unlimited Canada. Stonewall, Manitoba. 48 pp.

Wrubleski, D.A. and M.G. Anderson. 1999. The submersed aquatic macrophytes of east Delta Marsh, 1974 and 1997. University of Manitoba Field Station (Delta Marsh) Annual Report No.33. In press.

Wulf, A. 2002. Is there a coupling between increased UVB radiation and toxic algal blooms? SIL 2001 XXVIII Congress, Melbourne, Australia, abstract.

Yerubandi, R.R., N. Hawley, M.N. Charlton, and W.M. Schertzer. 2008. Physical processes and hypoxia in the central basin of Lake Erie. Limnology and Oceanography 53(5(2)): 2007-2020.

Young, J., A. Landre, J. Winter, H. Jarjanazi, and J. Kingston. 2010. Lake Simcoe water quality update, May 2010. Ontario Ministry of the Environment, Water Monitoring and Reporting Section, Environmental Monitoring and Reporting Branch. 56 pp.

Young, T.C., J.V. DePinto, S.C. Martin, and J.S. Bonner. 1985. Algal-available particulate phosphorus in the Great Lakes basin. Journal of Great Lakes Research. 11:434-447.

Zhang, W. and Y.R. Rao. 2011. Application of a eutrophication model for assessing water quality in Lake Winnipeg. Journal of Great Lakes Research (In Press).

Zhao, J., Y.R. Rao, and L.I. Wassenaar. 2011. Numerical Modeling of Hydrodynamics and Tracer Dispersion During Ice-free Period in Lake Winnipeg. Journal of Great Lakes Research (Accepted).

Zhou, L., H. Yu, and K. Chen. 2002. Relationship between microcystin in drinking water and colorectal cancer. Biomedical and Environmental Sciences 15: 166-171.

Zimba, P.V., L. Khoo, P.S. Gaunt, S. Brittain, and W. Carmichael. 2001. Confirmation of catfish, *Ictalurus punctatus* (Rafinesque), mortality from Microcystis toxins. Journal of Fish Diseases 24: 41-47.

Zrum, L. 1999. Abundance and species composition of zooplankton in the Nelson River estuary: Baseline Monitoring Program 1998 - Year III. A report prepared for Manitoba Hydro. North/South Consultants Inc.

Zurawell, R.W., B.G. Kotak, and E.E. Prepas. 1999. Influence of lake trophic status on the occurrence of microcystin-LR in the tissue of pulmonate snails. Freshwater Biology 42: 707-718.

Zurawell, R.W., H. Chen, J.M. Burke, and E.E. Prepas. 2005. Hepatotoxin cyanobacteria: A review of the biological importance of microcystins in freshwater environments. Journal of Toxicology and Environmental Health, Part B 8: 1-37.

Zurawell, R.W., C.F.B. Holmes, and E.E. Prepas. 2006. Elimination of the cyanobacterial hepatotoxin microcystin from the freshwater pulmonate snail *Lymnaea stagnalis jugularis* (Say). Journal of Toxicology and Environmental Health, Part A 69: 303-318.

Zurawell, R.W., J.I. Goldberg, C.F.B. Holmes, and E.E. Prepas. 2007. Tissue distribution and oral dose effects of microcystin in freshwater pulmonate snail *Lymnaea stagnalis jugularis* (Say). Journal of Toxicology and Environmental Health, Part A 70: 620-626.