

**A water quality assessment of Lake Manitoba, a large shallow lake in
central Canada**

by

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ABSTRACT

Water quality of Lake Manitoba is poorly understood in comparison to other large lakes and a study was undertaken to characterize the spatial and temporal variation in water quality. To characterize the lake wide water quality conditions, samples were collected from 15 stations over a 2 year period. Geospatial mapping and principal components analysis revealed that the south basin of the lake was more turbid, nutrient rich, and more dilute in comparison to the north basin. Water samples collected daily during the operation of the Assiniboine River Diversion in 2005 and 2006 indicated that the Assiniboine River Diversion was the single largest source of phosphorus and sediment and was the second largest source of nitrogen to the lake. A non-parametric trend analysis of a 17 year historical water quality dataset from a station in the south basin indicated that Lake Manitoba has become more dilute and nutrient rich over time.

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1.0 INTRODUCTION

Manitoba is home to the three Western Great Lakes of Canada, including Lake Manitoba, the largest lake in the prairie region of North America, and the 10th largest lake in Canada. It has been recognized that large lakes are of great ecological importance and knowledge of these systems is essential to protect water quality and to sustain aquatic life. Many lakes are becoming increasingly threatened by prolific nuisance algae blooms arising from increased nitrogen and phosphorus loading arising from changing land use practices and the intensification of agricultural practices.

In 2003, a report was submitted to the Province of Manitoba by Farlinger *et al.* (2003) which highlighted the need for a better understanding of water quality in Lake Manitoba. Many large lakes of western Canada have not been studied over a continuous long term period making lake management decisions difficult owing to a lack of continuous data. Prior to 1991, the water quality monitoring studies were infrequent on Lake Manitoba making evaluation of spatial and temporal water quality difficult. The majority of the water quality monitoring studies have focused on the south basin of the lake, and prior to the present study, the water quality of the north basin had not been monitored or evaluated since the 1970s. There is limited water quality data to allow for site specific or spatial comparisons, as monitoring locations were never consistent between water quality monitoring studies. Although the current water quality record is the longest continuous water quality record for the lake, the spatial water quality information during this period is lacking given the large surface area of Lake Manitoba.

There is limited hydrological information and bathymetry making evaluation of water quality difficult given the strong correlation between water quality and hydrology in such a shallow lake with two distinctly separate basins. According to IGDSB (1976), the south basin of Lake Manitoba retains approximately 75 percent of total nitrogen inputs and 95 percent of total phosphorus inputs through biological uptake and sedimentation processes. There are great differences between the amplitude of lake levels between the pre-regulation (i.e., pre-1960) and post-regulation (1961-present) periods of Lake Manitoba. Prior to regulation, average annual variation in lake levels were as great as 1.83 meters between minimum and maximum water levels. Under the current lake regulation, lake levels are stabilized such that the annual lake level varies by a maximum of 0.61 meters (LMRRAC 2003).

Water quality data indicate that there may be deterioration of water quality (Hughes 2002; LMRRAC 2003) in Lake Manitoba from 1991 to 2007; however, the significance of the trends has yet to be evaluated, to detect the underlying trend in the data. A visual assessment of the long term water quality monitored monthly from 1991-2007 at the Delta Marsh Field Station site provided a preliminary indication that total phosphorus and chlorophyll *a* concentrations may have gradually increased, while major ions, dissolved solids, and specific conductivity have declined over the past 16 years (Hughes 2002; LMRRAC 2003). Much of the change in water quality has been attributed to the influence of the Assiniboine River Diversion, which transports Assiniboine River water to the south end of Lake Manitoba, which has significantly lower average concentrations of sodium, chloride, and dissolved solids

relative to the lake, higher total phosphorus, and may account for the gradual 'freshening' and eutrophication of Lake Manitoba.

The major objectives of the study are:

1. To determine the spatial and temporal variation of physical and chemical water quality including light, nutrients, ions, and algal biomass in Lake Manitoba (Section 4). Currently, there exist two provincial water quality monitoring stations to characterize water quality in Lake Manitoba. One station is located at the very south end of the lake offshore from the University of Manitoba Delta Marsh Field Station and another station located at the Narrows. Water quality in the north basin of Lake Manitoba has not been characterized since the 1970s. This lake wide study will provide information on whether the current monitoring infrastructure adequately characterizes the spatial water quality variation in Lake Manitoba. I hypothesized that nutrients, ion chemistry, and chlorophyll a would differ between the north and south basins of the lake because of the differences in the chemistry of the major river inflows, water residence time, and lake morphometry between the north and south basins of Lake Manitoba.
2. To characterize the nutrient and sediment loading transported by the Assiniboine River Diversion to Lake Manitoba (Section 4). Because the Assiniboine River may transport large flows to the lake in flood years, I hypothesized that the Diversion may contribute a substantial nutrient and sediment load to Lake Manitoba in wet years.

3. To determine to what extent water quality has changed from 1991 to 2007 at a single historical water quality monitoring station in the south basin of Lake Manitoba (Section 5). I hypothesized that nutrient concentrations were increasing and major ion concentrations were decreasing in Lake Manitoba.

2.0 REVIEW OF THE LAKE MANITOBA STUDY AREA

2.1 *Drainage Basin*

Lake Manitoba is a part of the Dauphin River drainage basin which occupies an area of 80 299 km², and encompasses several other large lakes including Lake Winnipegosis, Dauphin Lake, and Lake St. Martin (Figure 2.1). The headwaters of the drainage basin originate in the Pasquia Hills of eastern Saskatchewan, and Porcupine Mountain, Duck Mountain and Riding Mountain of western Manitoba (Hughes and Williamson 2002). The western edge of the Dauphin River drainage basin divide is located approximately 60 km west of Lake Manitoba and is delimited by the rise of the Manitoba Escarpment. The eastern divide of the basin is located approximately midway between Lake Manitoba and Lake Winnipeg, although drainage is minimal along the east side of the basin. To the north, the basin is bordered by Cedar Lake and Pasquia/Summerberry basins, and is bordered on the south by the Assiniboine River drainage basin (LMRRAC 2003). The topography of the drainage basin is gently sloping from west to east with the highest point at 317 meters above sea level (masl) in the west near the Manitoba Escarpment to the lowest elevation of 238 masl in south east corner of the lake near the community of Oak Point. The majority of the contributing drainage to Lake Manitoba follows the natural topography flowing from the west to east, with exception of a few small drains and creeks along the east side of the lake, which contribute only a fraction of the drainage to Lake Manitoba.

Much of the water flowing into Lake Manitoba drains from Lake Winnipegosis through the Waterhen River to the north basin of Lake Manitoba. The Whitemud River is the major drainage which is tributary to the south basin, and drains into the southwest part of the lake near the community of Westbourne at Lynchs Point. Since 1970, water also periodically drains into the south basin through the Assiniboine River Diversion (ARD), an artificial channel which diverts water from the Assiniboine River to the south basin of Lake Manitoba, and serves to prevent downstream flooding in the City of Winnipeg. When water from the Assiniboine River is diverted into Lake Manitoba, the drainage basin encompasses 121 800 km², and includes the Assiniboine River drainage basin (41 500 km²). Water drains Lake Manitoba eastward through the Fairford River in the north basin and passes through Pineimuta Lake, Lake St. Martin and the Dauphin River system into Lake Winnipeg and ultimately through the Nelson River system into Hudsons Bay (LMRRAC 2003).

2.2 Location and Climate

The study area is distinctly separated into the Prairie and the Boreal Plain ecozones. The majority of the land surrounding the north basin of Lake Manitoba is a part of the Interlake Plain ecoregion (Boreal Plain ecozone) which and extends from the Interlake region north west towards the Pasquia Hills of Saskatchewan. The south basin is a part of the Lake Manitoba Plain ecoregion (Prairie ecozone) and extends south towards the US and northwest toward Dauphin Lake. Climate in the study area is largely influenced by the topography and moderated by the large lakes. Air moves off the Manitoba Escarpment and warms as it descends in altitude towards the Lake Manitoba Plain ecoregion, the warmest and most humid ecoregion

located within the Prairie ecozone of Canada. The Lake Manitoba Plain ecoregion (recorded at Delta Marsh) is characterized by the cool winters and warm semi-arid summers with a daily average air temperature of approximately 2.1°C (Table 2.1). The extreme average daily maximum and minimum temperatures of 24.9°C and -24°C occur in July and January, respectively. Average annual total precipitation is 524.6 mm with approximately 70 percent of the total precipitation occurring from May to October (Table 2.1) (Environment Canada 2001). The Interlake Plain (recorded at Ashern) is characterized by the moist boreal ecoclimate region with a comparatively cooler average annual air temperature of 1.3°C and more extreme average daily maximum and minimum temperatures of 25.3°C and -25°C in July and January, respectively (Table 2.2) (Environment Canada 2001). The average annual precipitation in the moist boreal ecoclimate region is 499.9 mm with greater than 80 percent of the total precipitation occurring from May to October (Table 2.2). The Delta Marsh meteorological station is the only weather station which actively records wind measurements around Lake Manitoba. Historical wind records indicate that the prevailing wind direction is northwest or north with the greatest average wind speeds occurring during the month of October at 17.1 km/hr. May, November, and January are also windy with average wind speeds of 16.2, 16.5, and 16.5 km/hr, respectively. The lowest average wind speed (12.9 km/hr) occurs in the month of July (Table 2.1).

2.3 Soils and Topography

Soils overly unconsolidated surface deposits that consist of highly calcareous glacial till and some areas of consolidated bedrock. Soils are predominantly loams and clays loams and are classified as Black and Dark Grey Chernozems that are

nutrient rich (in phosphorus and ammonia), productive, and support grasses and forbs which are adapted to semi-arid to temperate soil moisture conditions. Poorly drained areas consist of Gleyosolic, Organic, and Regosolic soils that contain organic and peat matter. Soils are typically thin and rocky where they are underlain by glacial till, and are particularly shallow and thin nearing consolidated bedrock in the northern and eastern regions of the Lake Manitoba Basin (Last 1980).

The gently undulating ridge and swale topography dominates the Lake Manitoba landscape. Ridges are generally well drained and consist of coarser deposits such as gravels and cobble while swales are characterized as poorly drained areas composed of finer particulates including silts and clays (MWC 1973; LMRRAC 2003). Soil drainage has been described as imperfect based on the topography and soil type, although alluvial soils such as soils found along the Whitemud River are generally considered well-drained (MWC 1973).

2.4 Geology

Devonian limestone, dolomite bedrock, and carbonate rich deposits are prevalent along the entire length of the lakes eastern shore (Cober 1968; Last 1980). Limestone, dolomitic limestone, and shale also characterize the local geology of the north basin. Shale, sandstone, and limestone bedrock occurs along the western and southern shores of the south basin, along with some dolomitic limestone, and gypsum of the Jurassic period (Last 1980). The exposed dolomitic bedrock outcrops of the north basin give the basin its irregular shape consisting of three elongate bays, convoluted shorelines, and numerous islands. The smoothed

shorelines of the south basin is a result of the thick unconsolidated deposits of glacio-lacustrine till which overly the local bed rock (Last 1980). Glacial till is layered with lacustrine and fluvial sands and gravels in the south western part of the south basin, and lacustrine clays and silts directly south of the lake (Cober 1968; Last 1980).

Economically valuable minerals which were actively mined around the lake include anhydrite, gypsum, limestone, and salt. These minerals were deposited during the Silurian and Devonian periods when the basin was inundated by glacial Lake Agassiz (Cober 1968). Dolomite (Winnipegosis Formation) of the Devonian period and gypsum (Amaranth Formation) of the Jurassic period are located along the west shore of the lake and are associated with the brine springs, which contribute a considerable amount of dissolved ions to the lake (Last 1980).

2.5 Vegetation and Land Cover

Forested areas and grasslands constitute approximately 65 percent of the total land cover area in the south basin (AAFC-PFRA 2005), and likely more in the north basin, however, much of the discussion will cover what is known of south basin due to the lack of information for the north basin. The study area is located in a vegetation transitional zone between the Interlake Plain and the Lake Manitoba Plain. Trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), and bur oak (*Quercus macrocarpa*) are among the dominant tree species that occur interspersed with rough fescue prairie grassland in the Lake Manitoba Plain ecoregion to the south. Moving northwards, the vegetation transitions to mixed

deciduous boreal stands of trembling aspen (*P. tremuloides*), balsam poplar (*P. balsamifera*), white spruce (*Picea glauca*), black spruce (*Picea mariana*), and jack pine (*Pinus banksiana*) in the Interlake Plain ecoregion (LMRRAC 2003).

The lake is bordered by 1210 km² of coastal wetlands which comprise approximately 15 percent of the total land cover area in the south basin of the lake (AAFC-PFRA 2005). Delta Marsh is a large 180 km² coastal wetland bordering Lake Manitoba that spans much of the lakes southern shore from Lynchs Point to St. Ambroise. Other sizable coastal wetlands neighboring the lake include Marshy Point, Lynchs Point, Lake Francis, Sugar Point, Big Point, Ebb and Flow Lake, and the Sandy Bay and Langruth/Lakeview areas along the western shore of the south basin (LMRRAC 2003). Coastal wetlands of significance in the north basin include Proulx Lake and Basket Creek areas.

Approximately eight percent of the land area around the south basin is dedicated to the production of cereal grain crops, although crop production is limited to the south and south west areas surrounding the lake where soil is productive and well drained. Marginal oilseed crops and hay are more prevalent in the northern part of the south basin where soils are otherwise too thin, stony, and nutrient deficient to support cereal crops. Livestock production and forage harvesting are the predominant agricultural activities that generally occur along the shoreline of Lake Manitoba and comprise almost four percent of the total land cover around the south basin. Using satellite imagery from 1993-94 and 2000-02, Agriculture and Agri-Food Canada-Prairie Farm Rehabilitation Administration (AAFC-PFRA 2005) found that

there has been an 85 percent increase in forage cover land in the south basin of Lake Manitoba over that time period.

Waterlogged areas, consisting of flooded lands, and standing or pooled water, covers approximately six percent of the land area of the south basin. Urban areas and transportation routes and roads around the lake cover two percent of the total land area in the south basin (AAFC-PFRA 2005). Communities and cottage developments bordering the lake include, Westbourne, Langruth, Sandy Bay, and Alonsa along the southern and western shores, and St. Laurent, Oak Point, Lunder, and the Lake Manitoba First Nation along the more populated east shore of the south basin.

2.6 Lake Morphometry

Lake Manitoba has a large surface area of 4700 km² and is the largest lake in the prairie region of North America and the third largest lake in the province of Manitoba (Last 1980; Last and Teller 2002). The lake is divided into distinct northern and southern basins that are connected by the Narrows, a narrow strait that is approximately 0.8 km wide (Burrows 1970) by 4 km long, located approximately halfway along the lakes length near the community of Eddystone (Figure 2.2). The lake is elongated along a NW-SE axis (Kenney and Mollison 1980) and is 197 km in length from north to south and 47 km in breath from east to west across the widest part of the south basin (Last 1980) (Table 2.3). The total shoreline length is approximately 915 km (LMRRAC 2003).

The large wind-exposed south basin is 105 kilometers in length from north to south and is 47 km in breadth from east to west, and has a surface area of 3107 km² comprising approximately 70% of the total lake surface area (Last 1980). The pear-shaped south basin has 392 km of shoreline that is rounded at the very south end of the lake with long straight and smooth shorelines that progressively converge north towards the Narrows (Figure 2.2). The comparably smaller sheltered north basin is 92 km in length from north to south and is 22 km in breadth from east to west and has an area of 1593 km² comprising 30% of the remaining lake surface area (Last 1980). The north basin is separated into 3 smaller elongate sub-basins and is characterized by 523 km of convoluted and irregular shorelines, with many protected bays, shallow narrow straits, peninsulas, islands and islets (Figure 2.2). The north basin also has several large peninsulas including Peonan Point, a large peninsula emerging from the north shore located midway along the width of the north basin. There are 10 islands in the south basin of the lake and 12 islands in the north basin (Last 1980) (Table 2.3).

Lake Manitoba has a shallow mean depth and has been reported as ranging between 4.5 m and 5.5 m (Bajkov 1930a; Kennedy 1949; Rawson 1955; Rawson 1960; Burrows 1970; Bernard 1972; Crowe 1972; Tudorancea 1975; Tudorancea and Green 1975; Tudorancea et al. 1979; Last 1980; Lysack 1997; LMRRAC 2003). Last (1980) calculated the mean and median depths of 4.5 m and 5 m from a hypsographic curve (corrected to 247.5 m above sea level) compiled for the southern two thirds of the south basin of the lake. For the purposes of this study, and given the limited bathymetric information for the lake (see Last 1978; Last

1980), the mean depth of the south basin will be regarded as 5 m based on the median lake depth reported by Last (1980), considering the range of mean depths previously reported for the south basin (Table 2.3), and others who have reported the mean depth of the entire lake as being 5 m (Rawson 1955 & 1960; Bernard 1972; Crowe 1972; Tudorancea 1975; Tudorancea and Green 1975; Tudorancea *et al.* 1979; Lysack 1997; LMRRAC 2003). The north basin is considered to be slightly shallower in nature (Burrows 1970; IGDSB 1976; Last 1980) with reported mean depths of 4.5 m (Burrows 1970) and 3.1 m (IGDSB 1976), although the values are likely rough estimates at best, as there is no known bathymetric information compiled for the north basin. For the purposes of this study and due to the lack of bathymetric data for the north basin, the mean depth of the north basin will be regarded as 4.5 m based on the value cited by Burrows (1970), and based on the knowledge that the north basin is naturally shallower than the south basin (Burrows 1970; IGDSB 1976; Last 1980). The maximum lake depth has been reported as ranging from 6.3 m to 7.2 m by numerous studies (Bajkov 1930a; Cober 1968; Crowe 1974a; Tudorancea *et al.* 1979; Last 1980; Lysack 1997) although for this study the maximum depth will be regarded as 7 m in both basins of the lake (Bajkov 1930a; Cober 1968; Tudorancea *et al.* 1979; LMRRAC 2003) (Table 2.3).

According to the lake basin types outlined by Håkanson (1981), the south basin of Lake Manitoba basin is classified as smooth and concave. The floor of the south basin has very little slope with a relatively low mean gradient of 0.037° and is very smooth with a low bottom roughness factor of 2.92 (Last 1980) (Table 2.3). Lake Manitoba has a relatively large depth ratio of 0.71, describing the ratio of the

mean to maximum lake depth, and a large volume development index of 2.14, describing the shape and volume of the lake relative to that of a cone which has a development index of 1. Lakes with high depth ratios and volume development index values similar to Lake Manitoba are characteristically shallow lakes with gently sloping sides and with relatively flat bottoms, while lakes having steeply sloping sides and deep areas will have a comparatively smaller depth ratios and volume development indices which are generally less than 0.5 and 1.5, respectively (Kalff 2002).

The dynamic sediment ratio or dynamic ratio (DR) describes the relationship between lake bottom dynamics (i.e., erosion, transport, and accumulation) and morphometry using lake area and mean depth (Håkanson 1982; Håkanson and Jansson 1983). Deeper lakes with large mean slopes have low dynamic ratios generally less than 1.1 indicating that less than 33 percent of the lake bottom is subject to erosion and transport processes. Resuspension dominates the limnological character of shallow lakes with dynamic ratios greater than 3.8 where 100 percent of the lake bottom is subject to erosion and transportation processes (Håkanson 1982). Using Håkanson's (1982) model, large dynamic ratios of 11.1 and 8.9 were calculated for the south and north basins of Lake Manitoba indicating that the limnological character of the lake is dominated by resuspension, and according to the DR classification of Håkanson (1982), 100 percent of the lake bottom has the potential to be eroded and resuspended in both basins of Lake Manitoba (Table 2.3). It should be noted that this value is higher than reported by Last (1978), who

indicated that approximately 66 percent of the sediment has the potential to be resuspended in Lake Manitoba.

The shoreline development index (SDI), or the ratio of shoreline length relative to the length of a circle with an area equal to that of the lake (Wetzel and Likens 1991), deviates from an SDI value of one (i.e., the SDI of a circle) when shoreline becomes more irregular. The SDI is indicative of the complexity of the shoreline, the extent littoral zone, and potential for lake productivity, and is a valuable index when comparing the SDI to that of other lakes (Wetzel and Likens 1991). The high shoreline development index value for Lake Manitoba reflects a high potential for development of littoral communities and biological productivity (Table 2.3). The south basin of the lake has a SDI of 1.98 and has a lower potential for biological productivity relative to the north basin, which has a higher index value of 3.70 due to the convoluted shorelines and smaller surface area of the north basin of Lake Manitoba (Table 2.3). Relative to the shoreline development index values of other large lakes of Canada, the north basin of Lake Manitoba has a comparatively larger SDI and a greater potential for biological productivity than Lake Winnipeg, Great Slave Lake, and Lake Huron, which have SDI values of 3.4, 3.6, 3.1, respectively (Hutchison 1975).

The ratio of watershed to lake area gives an indication of the nutrient loading to a lake, with a significant amount of nutrients and sediment from tributary inflow and runoff associated with large ratios of watershed to area. The north basin of Lake Manitoba has a relatively large ratio of watershed to lake area of 40, while the south basin drains a comparatively smaller area and has a smaller ratio of 6. The ratio for

the south basin increases to 19 when the Lake Manitoba watershed area includes the 41 500 km² Assiniboine River watershed (not including the Souris River and the Qu'Appelle River basins which are generally regarded as separate watersheds) (Armstrong 2005) during operation of the Assiniboine River Diversion.

2.7 Hydrology

2.7.1 Water Budget

The north basin of Lake Manitoba receives approximately 79 percent of the average annual river inflow from the Waterhen River, which drains water from Lake Winnipegosis to the north basin of Lake Manitoba (Figure 2.3). The contribution from the Waterhen River may be as great as 90 percent in years when the Assiniboine River Diversion does not flow. The south basin receives 6 percent of the average annual river inflow from the Whitemud River, although the Assiniboine River Diversion accounts for approximately 11 percent of the average annual river inflow to the south basin during periods of operation, making it the largest inflow to the south basin of the lake (Figure 2.3). The remaining four percent of the average annual inflow enters the lake from small ungauged streams, agricultural drains, and groundwater inputs. Water drains the lake through the Fairford River on the north-east side in the north basin of the lake (LMRRAC 2003). The regulated Fairford River is the only natural outlet for water and accounts for half of the outflow (Figure 2.3). The remaining water loss from Lake Manitoba is from evaporative processes, which are estimated to account for as much as 50% of the water loss to the lake during the summer months (Last 1980; LMRRAC 2003).

2.7.2 Basin Hydrology

The north and south basins of Lake Manitoba are hydrologically distinct given their differences in lake morphology, and major drainage processes. The north basin is considered a flow through basin and with a low average water residence time of two years given the influence of the major inflowing and outflowing rivers that drive water exchange in the north basin of the lake (IGDSB 1976). Evaporative losses are also presumably lower relative to the south basin because of the smaller lake surface area. The south basin is considered a closed and evaporative basin with a comparatively high average water residence time of 20 years given the limited surface inflowing and outflowing surface water drainage and the large lake surface area. During the summer months, the south basin of the lake acts as an evaporative system with almost 50 percent of water leaving the basin through evaporation from the lake surface. The large evaporative losses from the lake surface have important implications for water quality including nutrient retention and ion concentrations in the south basin of Lake Manitoba. Water exchange between the two basins is restricted by the Lake Manitoba Narrows. Based on limited wind and water velocity measurements at the Lake Manitoba Narrows by Burrows (1970), previous studies have suggested that there is very limited net movement of water between the two basins, and at times, when water levels were low, there was no net water exchange whatsoever between the two basins (IGDSB 1976; Last 1980).

2.7.3 Water Levels

As early as 1899, water level regulation measures were employed to control lake levels and included channel dredging at the Fairford River outlet to alleviate

high water levels, and the installment of control structure in 1933 to control the low water levels of Lake Manitoba. During high water years, such as the mid 1950s when the lake level was at a high of 248.793 masl (816.25 ft.), haylands, crops, farmlands, and residences along the shoreline were flooded because of the low relief of the land surrounding the lake. Low water levels, such as those recorded at 246.864 masl (809.92 ft.) in 1943, exacerbate the winter stagnation and oxygen depletion of the lake resulting in partial winter fish kills, and also affect negatively affect navigation, marshland water levels, and hunting and trapping of wildlife in the surrounding marshlands (Figure 2.4) (Butler 1949; Pollard 1973). In response to high water levels of the 1950s and 1960s, the Fairford River Water Control Structure (FRWCS) was completed in 1961, and stabilizes both the high and low water levels by maintaining a minimum outflow of 500 cfs.

There are great differences between the amplitude of lake levels between the pre-regulation (i.e., pre-1960) and post-regulation (1961-present) periods of Lake Manitoba. Prior to regulation, average annual variation in lake levels were as great as 1.83 meters (6.33 feet) between minimum and maximum water levels (Figure 2.4). Under the current lake regulation, lake levels are stabilized such that the annual lake level only varies by a maximum of 0.61 meters (3.12 feet) (LMRRAC 2003). Upon the recommendation of the LMRRAC (2003) the provincial government now manages the lake levels within a greater range of lake levels to promote the regeneration of healthy marshlands for recreation, farming, and wildlife.

2.8 Summary of Water Quality Surveys on Lake Manitoba

2.8.1 *Pre-1990*

Major water quality investigations were conducted from 1953-1954 by the Canada Department of Mines and Technical Surveys Mines Branch (Thomas 1959), and 1966-1969 (Cober 1968; Crowe 1972; Crowe 1974a) and 1973-1977 (presented in Hughes 2002) by the Manitoba Department of Mines, Natural Resources, and Environmental Management Research Branch (Table 2.4). The first chlorophyll a data and Secchi disk measurements were collected at Twin Lakes Beach over a two year period during the open water period of 1980-1982 by the Manitoba Environmental Services Branch (Hughes 1983). Other water quality data for Lake Manitoba consisted of supplementary data collected alongside other major biological (Bajkov 1930a and 1930c; Janusz 1972; Tudorancea and Green 1975), hydrogeological (Gilliland 1965), geological (van Everdingen 1971), or lake sedimentology investigations (Last 1980). In addition to the lake monitoring, a preliminary evaluation of the effects of the Assiniboine River Diversion on the south basin of Lake Manitoba was conducted in 1974 to assess the suspended sediment loads and particle size distribution fate in Lake Manitoba (Crowe 1974b).

Prior to 1991, the water quality monitoring studies were infrequent on Lake Manitoba making evaluation of spatial and temporal water quality problematic. The majority of the water quality monitoring studies have focused on the south basin of the lake, and prior to the present study, the water quality of the north basin had not been monitored or evaluated since the 1970s (Table 2.4). There is limited water quality data to allow for site specific or spatial comparisons, as monitoring locations were never consistent between water quality monitoring studies. Until 1991, there is

also limited temporal water quality data, as many studies only monitored each site on one single occasion (Bajkov 1930a; Gilliland 1965; van Everdingen 1971; Janusz 1972) or monitored each site on a monthly or quarterly basis over a four year period (Cober 1968; Crowe 1972; Crowe 1974a). Water quality data collected during this period summarized the ion chemistry, and routine water quality of Lake Manitoba including pH, alkalinity, hardness, true colour, dissolved, suspended, and total solids, specific conductivity, and turbidity. Only cursory monitoring of nutrients was conducted during the 1966-1969 (PO_4) and 1973-1977 (TOC, TIC, TKN, NO_3 - NO_2 -N, NH_3 -N, TP) water quality monitoring studies.

2.8.2 Post-1990

Since August 1991, the Water Quality Management Section of the Manitoba Department of Water Stewardship (previously known as Manitoba Conservation and Manitoba Environment) has monitored Lake Manitoba on a monthly basis at a single monitoring station located one kilometer offshore from the Delta Marsh Field Station (University of Manitoba) in the south basin of the lake (Table 2.4). Water samples have been consistently analyzed for pH, alkalinity (total, bicarbonate, carbonate, hydroxide), specific conductivity, total dissolved solids, total suspended solids, turbidity, true colour, dissolved oxygen, carbon (total, organic, inorganic), nitrogen (total kjeldahl, nitrate-nitrite, ammonia), and phosphorus (total, particulate, dissolved), chlorophyll *a*, and Secchi disk depth (sporadic measurements since 1993). Ions (calcium, magnesium, manganese, potassium, sodium, chloride, sulphate) and iron are generally monitored four times per year. Trace elements and pesticides were monitored on a quarterly basis from 1991 to

1996 (Table 2.4).

This 16 year water quality record is the longest continuous water quality record for the lake, although the spatial water quality information during this period is lacking given the large surface area of Lake Manitoba. Future continuous long-term monitoring studies should incorporate several sites around the lake to adequately monitor variation in spatial and temporal water quality.

2.9 *Chemical Limnology of Lake Manitoba*

The current water quality and trophic status of Lake Manitoba may be evaluated using the long term water quality monitoring data collected during the open water period (May to October) from 1991 to 2005 by Manitoba Water Stewardship. Based on these data, Lake Manitoba is classified as a slightly brackish lake with large concentrations of total dissolved solids (TDS > 500 mg/L) and high specific conductivity (>200 $\mu\text{S}/\text{cm}$), reflecting the highly ionized nature of the lake. Average specific conductivity is 1544 $\mu\text{S}/\text{cm}$, although conductivity may vary considerably throughout the open water season ranging from 624 $\mu\text{S}/\text{cm}$ (in the spring) to 2070 $\mu\text{S}/\text{cm}$ (in the late summer or early fall) (Table 2.5). Similarly, total dissolved solids range from 380 mg/L (during the spring thaw period) to 1300 mg/L and has an average concentration of 909 mg/L. Collectively, chloride, sodium, and sulphate ions constitute approximately 75 percent of the total dissolved solids fraction and have average concentrations of 315 mg/L (187-454 mg/L) , 204 mg/L (17-275 mg/L), and 165 mg/L (126-206 mg/L), respectively (Table 2.5).

Lake Manitoba is alkaline (pH > 8.5) with an average pH of 8.55. The total

hardness ranges from 265 mg/L to 382 mg/L with an average value of 344 mg/L, and the magnesium always contributes more than the calcium to the total hardness. Much of the hardness in Lake Manitoba arises from non-carbonate sources associated with the saline brine springs along the southwest side of the lake (i.e., SO_4^{2-} and Cl^-), and is atypical of most alkaline prairie lakes where hardness derives largely from bicarbonate and carbonate sources. Total alkalinity (as CaCO_3) in the south basin of Lake Manitoba ranges from 135 mg/L to 250 mg/L and has an average value of 218 mg/L, indicating that the lake has a high acid neutralizing (or buffering) capacity and, according to the classification by Palmer and Trew (1987), has an extremely low sensitivity to acidification. Calcium carbonate is presumably much lower than indicated by measuring total alkalinity (as CaCO_3) in 'closed' alkaline basins such as the south basin of Lake Manitoba, as the measurement of total alkalinity (as CaCO_3) assumes that alkalinity only arises from carbonate sources (Wetzel 2001).

The south basin of Lake Manitoba is turbid and generally has low water clarity. The average Secchi disk depth for the open water period is 0.42 m and ranges from 0.1 m to 1.00 m. Total suspended solids ranges from <5 mg/L to 96 mg/L with an average value of 30 mg/L. The Water Quality Management Section (Manitoba Water Stewardship) does not differentiate between inorganic and organic suspended solids, but presumably, the low water clarity is largely attributed to the inorganic sediments that are resuspended into the water column during storms or high wind events, although, during prolific algal blooms, the organic suspended solids (i.e., algae) may comprise a significant amount of the total suspended

sediment and result in reduced water clarity.

Primary productivity of aquatic ecosystems is affected by major plant nutrients such as nitrogen and phosphorus. The average total nitrogen concentration for the south basin of Lake Manitoba is 1.25 mg/L and ranges from 0.11 mg/L to 2.51 mg/L. Much of the total nitrogen in Lake Manitoba is in the form of organic nitrogen, and dissolved inorganic forms such as nitrate-nitrite-N and ammonia-N concentrations, which algae readily utilize, have low average concentrations of 0.015 mg/L (<0.01-0.12 mg/L) and 0.032 mg/L (<0.01-0.19mg/L). The total phosphorus is relatively high in Lake Manitoba with concentrations ranging from 0.01 mg/L to 0.28 mg/L with an average concentration of 0.07 mg/L for the south basin of the lake, and, according to Williamsons (2001) total phosphorus water quality guideline for lakes (TP <0.025 mg/L), may contribute to the nuisance growth of algae. Some estimates of nutrient conditions and how they affect algal growth may be made using total nitrogen to total phosphorus ratios (N:P). According to the classification of Kalff (2002), ratios below 10 are considered nitrogen limited, ratios above 16 are phosphorus limited, and co-limitation of N and P occurs between 10 and 16. The average N:P molar ratio in the south basin of Lake Manitoba is 49 and indicative of phosphorus limiting conditions, although it appears that the N:P molar ratios may vary considerably from nitrogen limiting conditions (i.e., N:P ratio of 3) to severe phosphorus limitation (i.e., N:P ratio of 147) depending on the time of year.

Lake Manitoba has been referred to as ranging from mesotrophic, or 'moderately enriched' to eutrophic or 'enriched' based on total phosphorus, chlorophyll *a*, and Secchi disk depth measurements using Carlson's (1977) trophic

state index (TSI) (Hughes 1983; Hughes 2002). Based on the average phosphorus and chlorophyll a concentrations for Lake Manitoba (1991-2005) Carlsons index values are currently 43 and 55, respectively. Index values from 30 to 50 correspond to mesotrophic lakes and values ranging from 50 to 70 are indicative of eutrophic conditions (Carlson 1977). According to the trophic state index classification, Lake Manitoba is considered mesotrophic with respect to phosphorus and eutrophic when considering chlorophyll a values, or is considered a meso-eutrophic lake. According to the Canadian Water Quality Index, Hughes (2002) ranked Lake Manitoba water quality as ranging from 'fair' to 'good', which indicates that the lake experiences only moderate nuisance levels of algae and nutrients.

2.9.1 Spatial and Temporal Variation in Water Quality

The current water quality record for Lake Manitoba (1991-2005) precludes the analysis of current spatial trends, as only one location has been monitored since 1991; however, there are some limited data relating to spatial trends between the north and south basins from the 1966-1969 and 1973-1977 water quality surveys on Lake Manitoba. Studies initiated by the Manitoba Department of Mines, Natural Resources, and Environmental Management Research Branch in the 1960s and 1970s indicated that the south basin of the lake contained higher concentrations of ions, carbonates, and dissolved solids relative to the north basin, with exception of calcium, which appeared to be consistently lower in the south basin of the lake (Cober 1968; Cober 1972). The Manitoba Department of Mines, Resources and Environmental Management (in IGDSB 1976) and Last (1980) attributed the lower calcium concentrations in the south basin to the active precipitation of CaCO_3 , which

decreases the free calcium ions in the water. Several studies also concluded that, relative to the north basin, the south basin of Lake Manitoba is more enriched with phosphorus (Cober 1968; Cober 1972; Hughes 2002). Presently, no attempt has been made to address the spatial water quality variation within each respective basin of Lake Manitoba.

The spatial differences in water chemistry between the two basins of Lake Manitoba are largely a result of the lake hydrology and morphology. The high average water residence time of 20 years and the large evaporative losses (50 percent of total outflow) experienced in the south basin results in retention and concentration of conservative ions and nutrients in the south basin of the lake (IGDSB 1976). In years of operation, the Assiniboine River Diversion accounts for 40 to 50 percent of the total nitrogen and phosphorus loading to the south basin of Lake Manitoba, and the Waterhen River carries the greatest nutrient loads, particularly with respect to total nitrogen, to the north basin of the lake. According to IGDSB (1976), the south basin of Lake Manitoba retains approximately 75 percent of total nitrogen inputs and 95 percent of total phosphorus inputs through biological uptake and sedimentation processes, and has a high phosphorus retention coefficient of 0.95 consistent with the Dillon and Rigler (1975) equation. Differences in major ion chemistry between the basins have also been attributed to the highly concentrated saline groundwater, which is rich in Na^+ , Cl^- , and SO_4^{2-} from gypsum deposits of the Amaranth formation, located along the south west side of the lake (van Everdingen 1971; IGDSB 1976; Last 1980). Approximately 20 to 50 percent of total annual loading of Ca^{2+} , Mg^{2+} , K^+ , and HCO_3^- and 70 to 80 percent of total

annual loading of Na^+ , Cl^- , and SO_4^{2-} to the south basin are from the saline groundwater springs in the south basin, even though groundwater input may only account for 5 to 10 percent of total inflows to the south basin of Lake Manitoba. There appears to be no net retention of Mg^{2+} , Na^+ , K^+ , and Cl^- ions in the lake; however, approximately 50 to 70 percent of the total inputs of Ca^{2+} , SO_4^{2-} , and HCO_3^- are retained in the south basin of the lake. The groundwater input of ions is limited in the north basin of Lake Manitoba, and the major ion chemistry in the north basin is largely determined by the Waterhen River (IGDSB 1976).

Several studies have summarized the general seasonal trends in major ions and nutrients; however, observations only concerned water quality changes from summer to winter and do not include a detailed analysis of major nutrient or ion dynamics for the entire open water period. Crowe (1972) summarized the seasonal variation in water quality by noting that concentrations of ions and nutrients were highest in the month of March and tended to be the lowest during the month of April throughout spring breakup. Although Crowe (1972) observed that phosphate was greatest in March under ice cover, nutrients in Lake Manitoba are typically low under ice cover due to the absence of wind acting on the water surface and stirring up nutrients from the sediment. Nutrient concentrations are comparatively higher during the open water season, except under nutrient deficient conditions that may occur in the later summer and fall. Thomas (1959) noted that hardness and specific conductance were lowest during the spring breakup and highest in March when lake levels are typically at their lowest. Turbidity levels were elevated from April through July, and again in the fall, with peak values occurring in May, and were attributed to

the high winds and waves stirring up the lake sediments (Thomas 1959).

2.9.2 Long Term Change in Water Quality

Long term trend results summarize the trends between the 1953-54 (Thomas 1959), 1966-1969 (Cober 1968; Crowe 1972), 1973-1977 (unpublished data found in Hughes 2002), and 1991-2005 (Hughes 2002; Manitoba Water Stewardship 2005) open water season (May to October), though, because of differences in analytical methodologies, sampling effort, number of monitoring sites, and water levels, results should be interpreted with caution. Refer to Table 2.4 and the respective authors listed in the text for further details on each monitoring program.

Average south basin total phosphorus concentrations in were similar between the 1966-1969 and 1973-1977 open water monitoring periods (Table 2.5). It is worth noting that Hughes (2002) reported a significant increase in phosphorus between to 1960s and 1970s; however, the study averaged all data across the open water and ice cover monitoring periods. Typically the lowest nutrient concentrations occur during the ice cover period (see Section 3.8.3 for discussion on seasonal differences in water quality), and almost 75 percent of all samples for the 1960s monitoring were collected during the ice cover period. The study compared these data to the 1970s, where approximately 70 percent of all samples were collected during the open water period, explaining the 'significant increase' in total phosphorus concentrations noted by Hughes (2002). Thus the significant increasing total phosphorus trend between the 1960s and 1970s noted by Hughes (2002) may not be an actual trend, but rather, the difference in sampling effort and season between the two studies. There

appears to be an increase in average total phosphorus between the 1973-1977 monitoring period and the 1991-2005 monitoring period, although the values were not significantly different (Hughes 2002).

Average open water values for specific conductivity, total dissolved solids, and major ions including magnesium, sodium, chloride, and sulphate in the south basin were lower in 1953-1954, peaked during the 1966-1969 monitoring period, and appear to have declined appreciably in the 1973-77, and again in the 1991-2005 monitoring period (Hughes 2002) (Table 2.5). Crowe (1972) indicated that the lake has been enriched in chloride, calcium, magnesium, and sulphate from 1930 (Bajkov 1930a) to the mid 1960s. Cober (1967) indicated that Lake Manitoba was several times more saline relative to water quality data collected by Bajkov in 1928. From 1928 to the late 1960s, total solids have increased almost two-fold, chloride approximately three-fold, and sulphate increased approximately two-fold. Lake levels are often inversely related to major ions and dissolved solids concentrations, and it is worth noting that the lake levels were extremely high (approximately 815 fasl) during the mid-1950s and quite low during the mid to late 1960s (approximately 811.5 fasl). Water quality data collected in 1953-1954 by Thomas (1959) were likely significantly diluted by such high water levels, while ions and dissolved solids were much more concentrated in the mid 1960s due to the concentrating effect of low water levels. Therefore, the apparent 'peak' in ions, dissolved solids, and specific conductivity in 1966-1969 may be attributed to the extremely different water levels between the two monitoring periods and may not reflect an actual trend in the water quality of Lake Manitoba. Last (1980) indicated that the saline groundwater and the

lake level fluctuations were in large part responsible for the observed historical water quality changes.

The visual assessment of water quality trends in Lake Manitoba over the past 16 years shows there may be deterioration of water quality; however, the significance of the 1991-2007 trends will be further evaluated, accounting for changes in lake level and seasonal variability, for total phosphorus, other major nutrients, and ions in Section 5. Initial visual assessment of the long term water quality data monitored monthly from 1991-2007 at the Delta Marsh Field Station site suggests that total phosphorus and chlorophyll a concentrations may be gradually increasing over time (Figure 2.5). The increasing phosphorus trend has been attributed to the nutrient loading from the diversion of Assiniboine River water, which has an average concentration 0.306 mg/L (Hughes 2002), approximately five times greater than the average total phosphorus concentration in Lake Manitoba (Hughes 2002; LMRRAC 2003). Chlorophyll a concentrations also appear to have increased over the same time frame in Lake Manitoba, and it is likely that the increasing trend is related to the nutrient loading from the Assiniboine River Diversion. Recent monitoring also shows a sharply declining trend in specific conductivity, total dissolved solids, and major ions, including Cl^- , Na^+ , and SO_4^{2-} , which has been attributed to the dilution effect of the Assiniboine River Diversion (Hughes 2002; LMRRAC 2003). The Assiniboine River has significantly lower average concentrations of Na^+ (<50 mg/L) and Cl^- (<20 mg/L) relative to the lake, and lower average total dissolved solids (<450 mg/L), and may account for the gradual 'freshening' of Lake Manitoba.

2.10 Physical Limnology of Lake Manitoba

2.10.1 Thermal Mixing Properties

The polymictic nature and the absence of thermal stratification is an important physical property governing physical, chemical, and biological processes in Lake Manitoba (Last 1978; Kenney 1985). Only very weak thermal stratification lasting for several hours (Bajkov 1930a,c; Tudorancea 1975; Kenney 1985) and development of a temporary microthermocline (Tudorancea *et al.* 1979) during successively calm and hot days for the open water season have been noted. When the water column is well-mixed by high winds, differences of no more than 0.5°C were observed from top to bottom (Kenney 1985). Bajkov (1930a,c) was the first to conduct temperature and dissolved oxygen profiles for Lake Manitoba, which included a site located near Fairford in the near shore and offshore areas, and noted that the surface water temperatures never differed from temperatures at depth by more than three degrees. Tudorancea *et al.* (1979) indicated that constant winds and mixing during the open water period precluded thermal stratification and, similar to Bajkov, only observed up to 3°C differences between surface and bottom temperatures. On one occasion, Tudorancea *et al.* (1979) observed the development of a 3°C microthermocline over a depth of 1 m (from 1 to 2 meters in the water column) after successively calm (average wind velocity < 5 km/hr) hot summer days. This differs from a true thermocline in that the microthermocline only temporarily occurs at shallow lake depths given the warm daily air temperatures and low wind velocities. Kenney (1985) measured water temperature at 5 depths while studying resuspension and currents in Lake Manitoba and described the thermal stability as

being neutral or near-neutral during high winds (22 km/hr) and noted no more than a 0.4°C difference over 4.2 m of the water column.

Inverse thermal stratification has been previously documented at 16 sites located 0.5 km to 24 km offshore in the southern two thirds of the south basin during the ice cover period in Lake Manitoba (Last 1980) with temperatures of approximately 1°C at the surface and 4°C near the bottom of the lake. Tudorancea et al. (1979) noted bottom temperatures of 0.5°C at a site near the confluence with the Whitemud River, and 2.5°C at six other sampling sites (located 0.5 to 7 km offshore) in the south basin during the ice cover period.

2.10.2 Sediment Resuspension and Lake Currents

Suspended sediment can be quite variable in Lake Manitoba due to wind and wave action experienced in the littoral zone of the lake (Thomas 1959), and wind generated waves may be able to resuspend approximately 66 percent of the lake bottom (Last 1978). Although not quantified for Lake Manitoba, turbulent surface waves will greatly influence the resuspension of sediment during high wind events given the shallow nature of the lake, and may contribute significantly to nutrient recycling processes in Lake Manitoba (Kenney 1985).

To assess the vertical particle distribution, particle size, and chemistry of resuspended sediments in the water column and water currents of Lake Manitoba, a suspended sediment sampler instrument consisting of 9 vertically spaced sediment traps was deployed in the south basin (Kenney 1985). Total resuspended sediment mass increased vertically with depth, and sand contributed the greatest mass to

suspended sediment samples in the upper 2.7m of the water column; however, at depths greater than 2.7 m, there was a marked shift in particle size distribution with silt comprising large fraction of the resuspended sediments. Interestingly, Kenney (1985) attributed the vertical shift in particle size to two distinct water currents moving at differing speeds and directions. The alongshore currents, moving in an easterly or south easterly direction at a 90 degree angle to the direction of surface flow (i.e., mainly N, NW, or NE), were found at water depths greater than 2.7m, were 1.5 to 2 times slower than average surface current speeds, and were largely determined by the lake circulation pattern (i.e., counterclockwise barotropic circulation). The faster onshore currents comprised the top half of the water column, and moved in the same direction as the prevailing wind, and were driven by wind shear stress.

2.11 *Biological Characteristics of Lake Manitoba*

2.11.1 Plankton

Information on the phytoplankton and zooplankton of Lake Manitoba is lacking, and unfortunately, samples collected in 1928 by Bajkov (1930c) are currently the only documented characterization of the Lake Manitoba plankton community composition. Bajkov (1930c) provided the first investigation of the Lake Manitoba phytoplankton community, and included a listing of species and a descriptive assessment of phytoplankton abundance (i.e., abundant, very common, common, less common or rare). Generally, green and blue-green algae dominated the phytoplankton species composition; however several diatom species were

considered to be 'very common' to Lake Manitoba. *Scenedesmus quadricauda* (Breb.), *Rivularia pisum* (Ag.), *Distyosphaerium pulchellum* (Wood), *Pediastrum* sp., *Fragiliaria crotonensis* (Kitton), and *Melosira* sp. were among the most 'abundant' or 'very common' algal species in Lake Manitoba. *Daphnia longispina* (var. *hyaline*), *Polyarthra platyptera*, and *Triarthra* sp. were among the most abundant zooplankton species occurring in Lake Manitoba (Bajkov 1930c).

2.11.2 Benthic Invertebrates

Crowe (1974a) assessed the productivity of the benthos of Lake Manitoba and indicated that Lake Manitoba is highly productive. The aerial benthic productivity of Lake Manitoba was 1320 organisms/km² and had a dry weight standing crop of 48.1 kg/ha, while the south basin of neighboring Lake Winnipeg, had an aerial benthic productivity of 776 organisms/km² and a dry weight standing crop of 6.57 kg/ha. Midge larvae comprised almost 60 percent of the benthos of Lake Manitoba, while snails and fingernail clams comprised 24 and 14 percent, respectively (Crowe 1974a). The large percentage of shelled organisms (38 percent) was a unique feature of Lake Manitoba, as mollusks and shelled organisms comprised only 19 percent of the total composition of Lake Winnipeg. Bajkov (1930b) examined the benthos of Lake Manitoba and indicated that the mollusks and chironomid larvae were the main food items of fish in the lake with the latter being found in whitefish stomach towards the end of the ice cover season.

2.11.3 Fish

The earliest record of fish species occurring within Lake Manitoba was

documented in a report by Bajkov (1928) that provided a complete listing of 95 species known to occur in the Hudson Bay drainage system. The Lake Manitoba sustains a large winter commercial fishery yielding 1 to 2 million pounds per year, with walleye, sauger, yellow perch, and whitefish being the most profitable commercial species (Lysack 1997). Lake Manitoba's fishery has the largest annual production per area of any great lake in Manitoba at 5.01 pounds per acre, which is comparatively higher than Lake Winnipegosis and Lake Winnipeg that have average annual productions of 4.4 and 2.9 pounds per acre, respectively (Butler 1949). Greater production in Lake Manitoba has been attributed to higher concentrations of nitrogen relative to concentrations in Lake Winnipegosis and Cedar Lake (Elkins *et al.* 1948).

Annual fish production in Lake Manitoba has declined drastically since the 1950s and is largely attributed to the regulation of water levels (Pollard 1973; Derksen 1978; Lysack 1997; Wrubleski 1998). The drastic decline in production has also been attributed to the loss of spawning habitat from accelerated siltation, and the drainage of wetlands, which help to moderate the extreme fluctuations in water levels observed on Lake Manitoba. There have also been several reports of 'death waves' resulting in a large number of dead fish below the ice surface, along with rusty red coloured water and a strong odour. These waves have been reported to move from shallow littoral areas and move offshore towards the center of the lake at 0.1 to 1.6 kilometers per week although at 12 to 16 kilometers offshore the death waves dissipate (Anon 1961). Water samples indicated that oxygen was depleted to 20 percent saturation near the lake bottom while the water directly below the ice was

well saturated with 80 to 90 percent saturation (Anon 1961). Doan (1971) reported that the death waves are likely from the severe oxygen depletion experienced below the ice from low water levels and red colouration from red algae below the ice surface (Hinks 1938; Anon 1961; Doan 1971; Nero 1971).

Table 2.1. Climate normals for the Lake Manitoba Plain Ecoregion from the Environment Canada meteorological station located at Delta Marsh CS (5040764, 50° 10' N, 98° 22' W, 248.40 m), 1971 to 2000. Data are provided by Environment Canada.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
Daily Average (°C)	-18	-14	-7	2.2	11.1	16.8	19.1	17.8	11.9	5	-5.2	-14	2.1
Daily Maximum (°C)	-12	-8.3	-1.3	8	17.1	22.5	24.9	23.9	17.8	10.6	-0.5	-9.3	7.7
Daily Minimum (°C)	-24	-20	-13	-3.5	5.1	11.2	13.3	11.6	6	-0.5	-9.8	-19	-3.5
Rainfall (mm)	0.2	1.8	7	24.4	53	81.4	77.1	64.3	52.5	33.6	5.8	0.8	402
Snowfall (cm)	23.9	18.4	19	10	1.6	0	0	0	0.3	4.3	23	25.4	126
Total Precipitation (mm)	23	18.5	26	34.6	54.6	81.4	77.1	64.3	52.9	37.8	28.9	25.6	525
Wind Speed (km/h)	16.5	15.9	16.1	16.1	16.2	14.8	12.9	13	15.2	17.1	16.5	16	15.5
Most Frequent Wind Direction	NW	NW	N	N	N	NE	NW	NW	NW	NW	NW	NW	NW
Lake Evaporation (mm)						4	4	3.2	2				

Table 2.2. Climate normals for the Boreal Plain Ecoregion from the Environment Canada meteorological station located at Ashern (5040120, 51° 7' N, 98° 22' W, 263.00 m), 1971 to 2000. Data are provided by Environment Canada.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
Daily Average (°C)	-19	-16	-7.8	3	11	15.7	18.4	16.9	10.8	4.5	-6	-17	1.3
Daily Maximum (°C)	-13	-9.4	-1.6	9.6	18.1	22.5	25.3	23.8	17	9.8	-1.6	-11	7.4
Daily Minimum (°C)	-25	-22	-14	-3.6	4	8.8	11.5	10	4.6	-0.8	-10	-22	-4.9
Rainfall (mm)	0.2	0	6.2	17.9	46.9	87.4	61.1	80.7	59.8	30.9	3.7	1.2	396
Snowfall (cm)	16.9	12.8	17.4	11	3.3	0	0	0	1.1	7.2	18.1	16.4	104
Total Precipitation (mm)	17.1	12.8	23.5	28.9	50.2	87.4	61.1	80.7	60.8	38.2	21.7	17.5	500

Table 2.3. The physical quantitative and qualitative morphometry of Lake Manitoba.

Lake Morphometry	Units	South Basin ¹	North Basin ¹	Whole Lake ¹
<i>Quantitative Morphometry</i>				
Watershed Area	km ²	17343	62956	80299
Watershed Area (with ARD)	km ²	58843	62956	121799
Lake Area	km ²	3107	1593	4700
Maximum Length	km	105	92	197
Maximum Breadth	km	47	22	47
Shoreline Length	km	392	523	915
Number of Islands (>0.01 km ²)	-	10	12	22
Mean Depth ²	m	5	4.5	-
Max Depth ²	m	7	7	-
Mean Basin Slope	°	0.037	-	-
Lake Bottom Roughness Factor	-	2.92	-	-
<i>Qualitative Morphometry</i>				
Depth Ratio (z:z _m)	-	0.71	0.64	-
Volume Development Index	-	-	-	2.14
Dynamic Ratio (DR)	-	11.1	8.9	-
Erosion and Transportation Area	%	100	100	-
Shoreline Development Index	-	1.98	3.70	-
Watershed : Lake Area Ratio	-	6	40	-
Watershed : Lake Area Ratio (with Assiniboine River)	-	19	40	-
Estimated Volume	km ³	15.5	7.2	22.7

¹ Lake morphology of the south basin and the entire lake was obtained from Last (1980). North basin morphology was measured (length, breadth) or determined mathematically (watershed area, lake area). Total lake shoreline length was reported by LMRRAC (2003) and the south basin shoreline length was reported by Last (1980).

² Mean and maximum depths are reported estimates (refer to Section 2.6 for further discussion).

Table 2.4. Summary of water quality sampling on Lake Manitoba (1928-2007) including year of water sample collection, sampling interval, time of year, number of sites, total number of samples, and the type of water chemistry analysis.

Year(s) of Water Sample Collection	Sampling Interval ¹	Time of Year ²	Number of Sites			Number of Samples	Type of Analyses ⁴	Source
			Total	South Basin	North Basin ³			
<i>Water Quality Surveys</i>								
1953-1954	M	S,W	2	2	0	13	R, I, NO ₃	Thomas (1959)
1966-1969	M	S,W	7	5	2	179	R, I, PO ₄	Cober (1968); Crowe (1972)
1973-1977	Q	S,W	6	3	3	87	R, I, N, B	Published in Hughes (2002)
1991-Present	M	S,W	1	1	0	191	R, I, N, TE, P, Chl a	Hughes (2002); MWS (2007)
2005-2007	B	S,W	22	13	9	294	R, I, N, Chl a	Page (2011)
<i>Other Water Quality Data</i>								
1928	O	S	6	2	4	6	I, TDS, alkalinity	Bajkov (1930a)
1963	O	W	1	1	0	1	R, I, NO ₃	Gilliland (1965)
1968-1969	O?	S	17	8	9	17	I, alkalinity	van Everdingen (1971)
1971	O	S	2	2	0	2	R	Janusz (1972)
1973-1974	Q	S,W	7	7	0	35?	R, I	Tudorancea and Green (1975)
1978	?	W	6	6	0	?	I, TDS, alkalinity	Last (1980)
1980-1982	M or B	S	1	1	0	20	Chl a	Hughes (1983)
1997-2000	M	S	3	3	0	77	B	in Hughes (2002)

¹Sampling intervals are classified as biweekly (B), monthly (M), three to five times per year or quarterly (Q), and a single sampling event (O).

²Summer (S) and/or winter (W). A bold S or W indicates a large proportion of the monitoring was either during the ice cover or open water periods.

³The Narrows is considered a north basin site.

⁴Type of analysis grouped as routine water chemistry (R), ions (I), and nutrients (N), bacteriological (B), trace elements (TE), and pesticides (P). Routine water chemistry refers to any measurements of TSS, Specific Conductivity, Turbidity, True Colour, and pH. Trace elements and pesticides were measured by the province from 1991-1996 (Hughes 2002).

Table 2.5. Summary historical water quality data (mean, standard error, min, max, n) for the south basin of Lake Manitoba during the open water period (May to October) of 1953-54 (Thomas 1959), 1966-1969 (Cober 1968; Crowe 1972), 1973-1977 (Manitoba Department of Mines, Resources and Environmental Management), and 1991-2005 (Manitoba Water Stewardship 2007).

Variable	1953-1954					1966-1969					1973-1977					1991-2005				
	mean	se	min	max	n	mean	se	min	max	n	mean	se	min	max	n	mean	se	min	max	n
Specific Conductivity (µS/cm)	1974	36.54	1781	2152	9	2173	47.16	780	2750	43	1762	46.29	1166	2210	28	1544	31.18	624	2070	88
Total Dissolved Solids(mg/L)	1188	28.27	1098	1257	5	1360	23.69	530	1730	44	1060	40.15	717	1650	28	897.6	20.11	110	1300	86
Chloride (mg/L)	413.9	8.829	354	438	9	469.2	12.46	105	590	44	353.6	14.99	200	480	28	314.7	10.55	187	454	39
Sodium, Extractable (mg/L)	279	6.637	238	303	9	329.2	5.585	268	527	44	234.6	7.918	150	310	28	203.6	9.123	17	275	32
Sulphate (mg/L)	170	2.606	158	179	9	183.4	3.962	98.6	212	44	156.8	3.14	110	180	28	164.5	3.581	126	206	35
pH (pH units)	8.47	0.053	8.30	8.80	9	8.322	0.047	7.80	8.75	33	8.62	0.022	8.30	8.85	28	8.552	0.021	7.92	8.93	88
Hardness (CaCO ₃) (mg/L)	361.9	4.736	338	386	9	417.1	5.462	308	494	44	356.4	8.307	253	450	27	344.2	8.758	265	382	14
Total Alkalinity	-	-	-	-	-	229.7	2.309	168	264	44	209.1	3.554	135	244	28	217.5	1.986	135	250	86
Total Suspended Solids (mg/L)	35	10.98	15.9	62	5	-	-	-	-	-	33	7.253	<5	148	28	30	2.148	<5	96	87
Secchi Disc Depth (m)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.415	0.028	0.10	1.00	41
Nitrate-Nitrite-N (mg/L)	1.189	0.326	0	3	9	-	-	-	-	-	0.134	0.038	<0.01	0.83	28	0.015	0.002	<0.01	0.12	87
Total Ammonia-N (mg/L)	-	-	-	-	-	-	-	-	-	-	0.062	0.019	<0.02	0.4	27	0.032	0.004	<0.01	0.19	88
Total Nitrogen	-	-	-	-	-	-	-	-	-	-	1.227	0.056	0.91	2.13	26	1.251	0.047	0.11	2.51	87
Total Phosphorus (mg/L) ¹	-	-	-	-	-	0.049	0.018	0.0	0.24	13	0.046	0.007	<0.02	0.16	28	0.068	0.004	0.01	0.28	87
TN:TP ²	-	-	-	-	-	-	-	-	-	-	69	8.475	15	181	22	49	2.738	2.9	147	87
Chlorophyll a (µg/L)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.0	1.24	<1	49.9	69

¹Hughes (2002) estimated total phosphorus values for the 1966-1969 period based on total phosphate values reported by Crowe (1972).

²TP values below the detection limit were not included in the TN:TP ratio calculations.



Figure 2.1. A schematic map delineating the Dauphin River Drainage Basin and Lake Manitoba. Base Map Source: Google Earth © 2007.

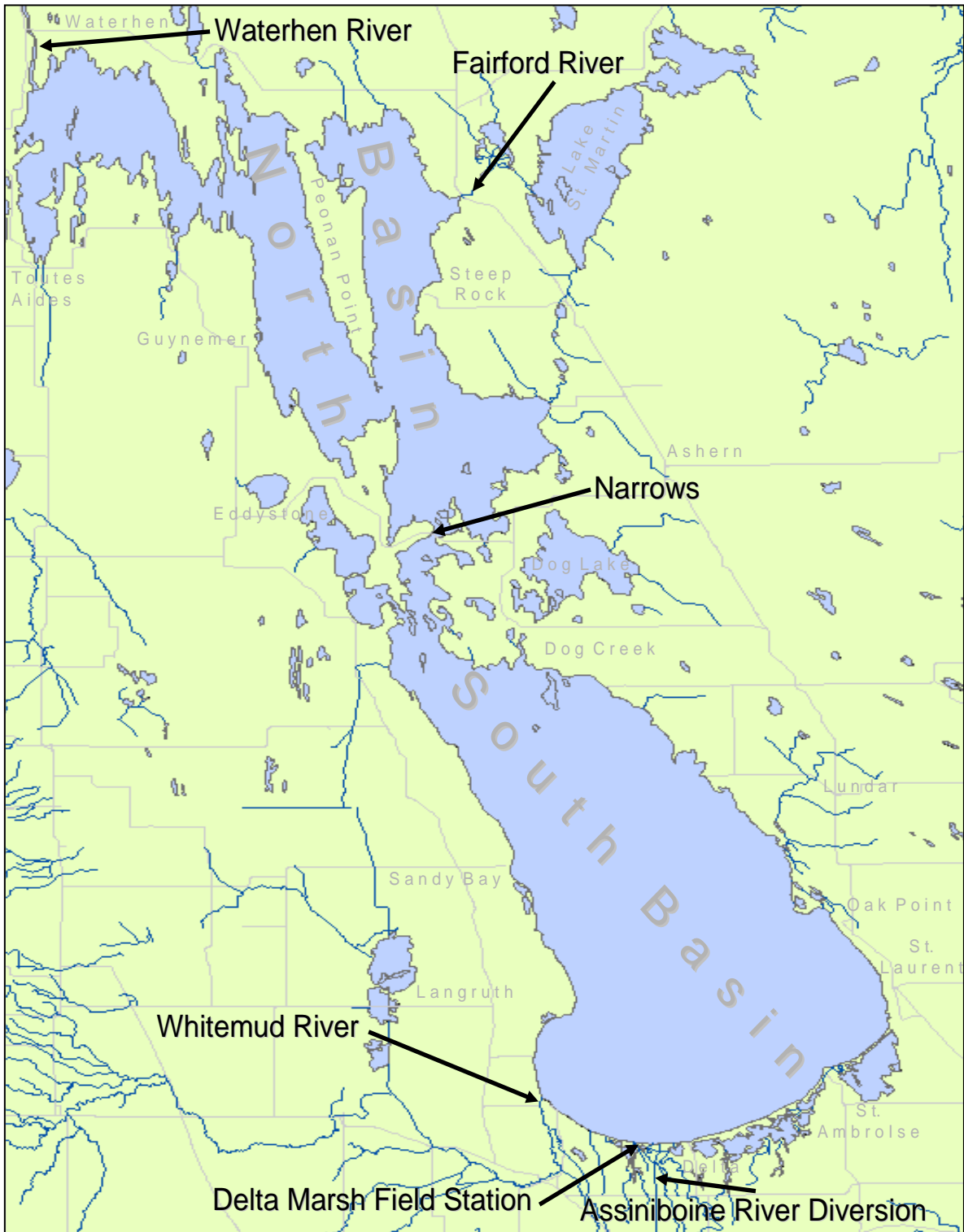


Figure 2.2. A map of Lake Manitoba showing the major morphological features of the lake, the surrounding water bodies, and the communities located around the lake.

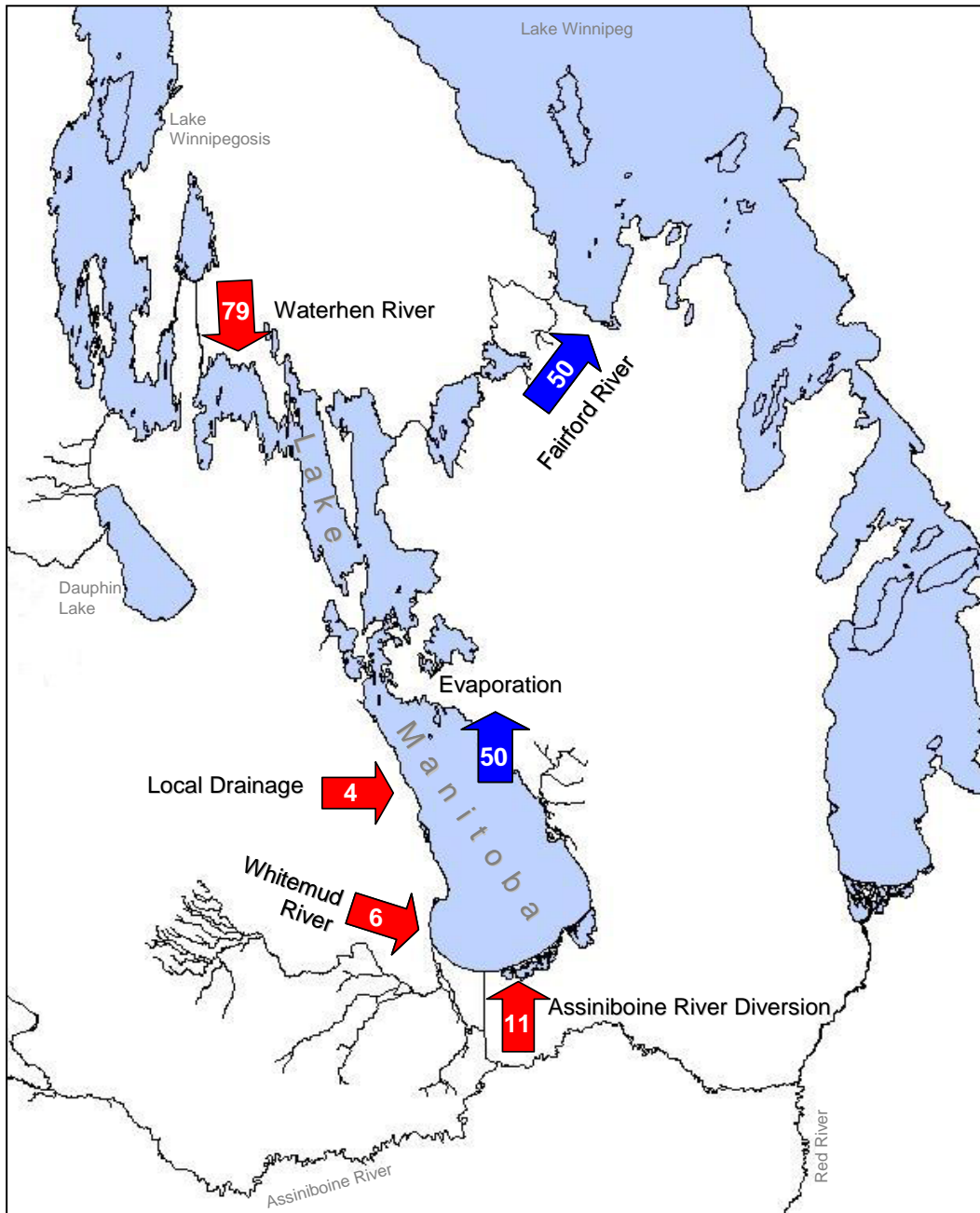


Figure 2.3. A schematic diagram of the average annual contribution (%) of surface inflows and outflows to Lake Manitoba, 1972-2001. Red arrows signify inflows and blue arrows signify outflows. The water budget accounts for the average annual contribution of the Assiniboine River Diversion, 1970-2001. Local drainage includes small ungauged streams, drains, and groundwater inputs. Data Source: LMRRAC (2003).

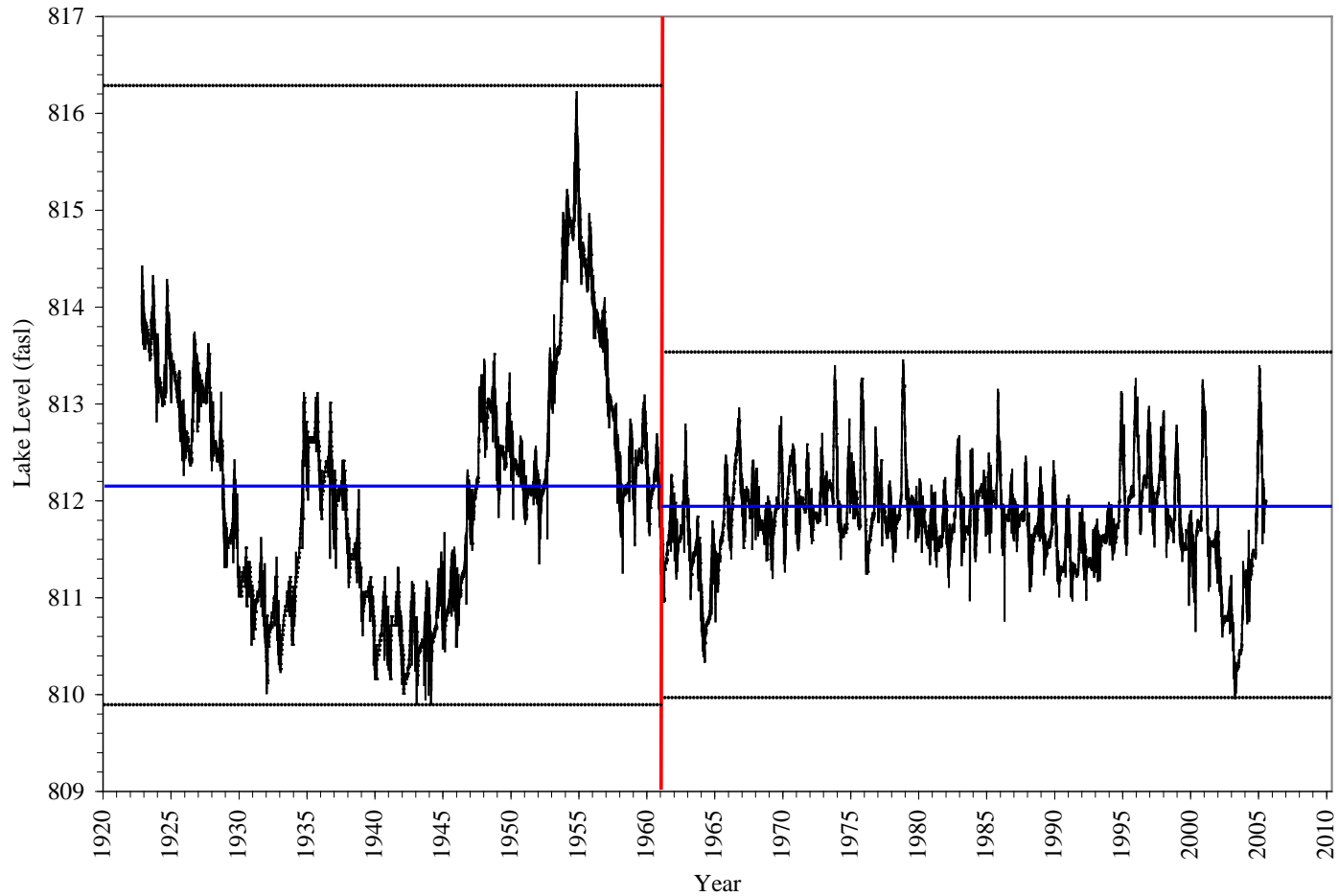


Figure 2.4. Historical Lake Manitoba water levels recorded at Steep Rock (Water Survey of Canada Station 05LK002), Manitoba, 1923-2006. The red line represents the start of lake regulation in 1961. Black dashed lines are the upper and lower ranges and the blue lines are the average lake levels for pre and post regulation periods. Data source: Water Survey of Canada.

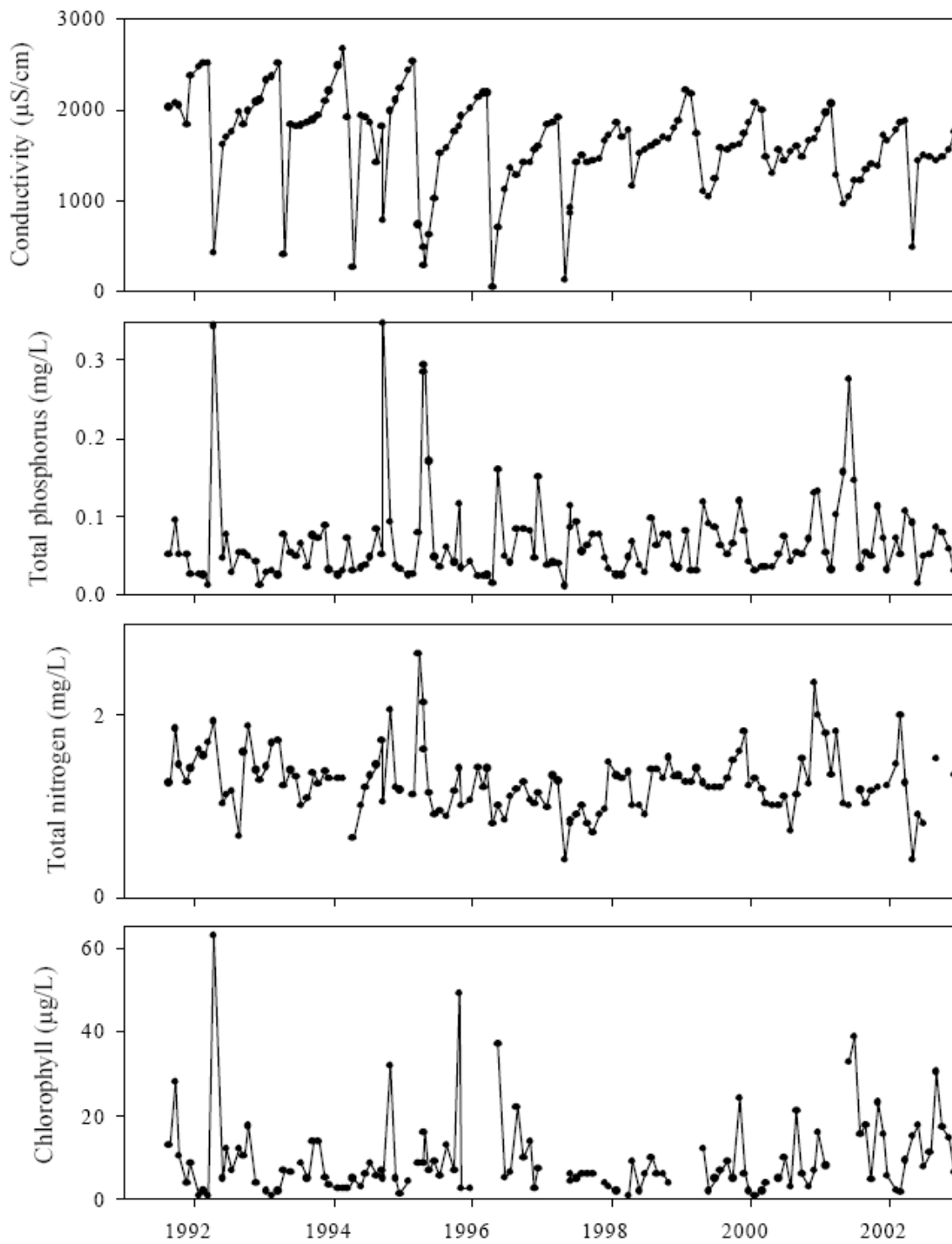


Figure 2.5. Historical water quality trends in specific conductivity, total phosphorus, total nitrogen, and chlorophyll a monitored in Lake Manitoba, 1991-2003.

3.0 METHODS

3.1 Lake Manitoba Sampling (Objective 1)

3.1.1 Open-Water Sample Collection

Surface water samples (0.1 m) were collected from 7 lake sites in the north basin (including the Narrows) and 8 lake sites in the south basin of Lake Manitoba every two weeks from mid May to the end of August or September in 2005 and 2006 for water chemistry analysis (Figure 3.1, Tables 3.1 and 3.2). Water samples were also collected biweekly from the Whitemud River, the Waterhen River, and the Fairford River, and daily or twice daily from the Assiniboine River Diversion (see Section 3.5) during periods operation in 2005 and 2006. Water sampling locations were selected on the basis of spatial uniformity across each basin and accessibility to the lake. Wherever possible, sampling locations were chosen to correspond with previously established provincial water quality monitoring stations (Table 3.1), although new sites were established to maximize the spatial water quality information. Water sampling locations were also established in close proximity to Water Survey of Canada lake level gauging stations near Westbourne (Station 05LL012) and Steep Rock (Station 05LK002) and the Environment Canada meteorological stations near Delta Marsh Field Station (Station 5040764) and Oak Point (Station 504K0NM) to characterize lake level and local climate. River sampling locations were selected on the basis of bridge accessibility and proximity to Water Survey of Canada continuous flow gauging stations located on the Whitemud River (Station 05LL002 near Westbourne), Waterhen River (Station 05LH005 near

Waterhen), and the Fairford River (Station 05LM001 near Fairford) to characterize nutrient and particulate matter loading to the lake.

In addition to the biweekly sampling at the 15 lake sites, concurrent water samples were collected during 8 sampling periods in 2005, 2006, and ice-cover 2007 (see Table 3.2), for Manitoba Water Stewardship (Water Quality Management Section) at existing provincial sites and newly established sites to augment the spatial and temporal water quality data for the provincial Lake Manitoba water quality database.

Water samples were collected from an anchored canoe or by wading (during inclement weather) at sites that were established within the littoral zone, approximately 200 meters from shore at each lake site (Plate 3.1). Where samples were collected by wading during inclement weather, care was taken to minimize disturbance of sediments and samples were only collected several minutes after arrival at the site to allow any disturbed sediments to settle. The coordinates of each site was recorded using a Garmin GPS 72 to ensure the location of the site was consistent between sampling periods. In some cases, it was not possible to reach the true coordinates of the site when wading, although the coordinates of the new location were recorded to account for the change in location. At each site, the canoe anchor was lowered slowly to the lake bottom to minimize sediment disturbance during water sampling. A one liter opaque polypropylene bottle was triple rinsed with lake water from each site before collecting the final water sample. Water samples were always collected at the bow of the canoe with the bottle oriented into the waves (Plate 3.1). Water samples were collected by inverting the sampling bottle and

placing the bottle approximately 0.1 m below the water surface and slowly re-inverting to completely fill the bottle leaving no headspace under the cap. Surface samples (0.1 m) for cyanobacterial toxins and phytoplankton taxonomy were also collected in 150 mL polyvinyl chloride amber bottles using the same collection method as the raw water samples.

For the Whitemud, Waterhen, and Fairford rivers and the Lake Manitoba Narrows site, a 0.5 L high density polyethylene Nalgene® bottle, located inside a grab sampler that consisted of a cement weighted plastic polyvinyl chloride tube, was triple rinsed with the water from each site prior to the collection of the final water sample. The sample was collected on the upstream side of the bridge (with exception of the Fairford River which was collected on the downstream side) from center channel using a rope to lower the weighted Nalgene® grab sampler to the water and submersing it until the bottle was full. The water collected in the weighted Nalgene® grab sampler was transferred into a one litre polypropylene sample bottle for water quality analysis and the 150 mL amber bottle for toxin and phytoplankton analysis.

Water temperature, specific conductance, salinity, dissolved oxygen (YSI 85 multi-parameter probe) and pH (Hanna model HI 9026), were measured *in situ* at each sampling location during every sampling period. Probe measurements were taken 0.1 m below the water surface at the stern of the canoe several minutes after anchoring to prevent inaccurate measurements arising from resuspended sediment. Vertical depth profiles of water temperature, specific conductance, and dissolved oxygen, were also measured at 0.5 m depth intervals at a number of sampling

locations during calm and well mixed lake conditions to effectively characterize the mixing properties of the lake. Additional *in situ* measurements included water depth using a weighted metered rope, water clarity was determined using a 22 cm diameter Secchi disk, and light attenuation was measured with a LiCor LI-1000 datalogger, an LI-190 quantum sensor, and a LI-192SA spherical PAR sensor. Air temperature, average and maximum wind speed, and wind direction were also noted at each site using a Kestrel 3000 portable weather meter. The meter was held 2 meters above the ground and oriented facing into the prevailing wind direction for a period of 2 minutes before recording final wind and temperature readings.

3.1.2 *Ice-Cover Sample Collection*

Surface water samples were collected from sites during the ice cover period from 2005 to 2007 (Figure 3.1, Tables 3.1 and 3.2). Sites further offshore were accessed using a Bombardier and littoral sites around the lake were accessed by walking offshore or using a snowmobile and sled (Plate 3.2). At each site, a power ice auger was used to drill a 0.3 m diameter hole through the ice surface and the hole was cleared of ice using an ice scoop. Surface water samples were collected by submersing the bottle just below the water surface until the bottle was full. Temperature, specific conductance, salinity, and dissolved oxygen were measured *in situ* approximately 0.1 m below the water surface at each sampling site using a YSI 85 multi-parameter probe, and at 1 m depth intervals for vertical depth profiles. Additional *in situ* measurements included water depth using a weighted metered line, water clarity using Secchi disk measurements, air temperature, average and maximum wind speed, and wind direction.

3.2 Water Chemistry Analyses

Water samples were stored in dark coolers and kept cool using ice packs and were analyzed within 48 hours of collection at the Delta Marsh Field Station (University of Manitoba). Unfiltered water samples were analyzed colorimetrically for total phosphorus (TP; acid persulfate digestion; Hach 1997), total reactive phosphorus (TRP; acid molybdate method; modified from Stainton *et al.* 1977), total nitrogen (TN; persulfate digestion; Hach 1997), ammonium nitrogen (NH₃-N) (hypochlorite method; Stainton *et al.* 1977), and soluble reactive silicon (SRS; acid molybdate method; Stainton *et al.* 1977) using an Ultrospec 400 UV/Visible light spectrophotometer. Water samples were also analyzed for turbidity (Hach model 2100A Turbidimeter), percent transmittance of unfiltered water at 450nm, and pH (Corning ion analyzer 250). Alkalinity was determined by acid titration using 0.02 M HCl and 3 drops of bromocresol green-methyl red indicator to a clear endpoint and calculated according to APHA (1998). Alkalinity and pH values were used to calculate the dissolved inorganic carbon concentrations according to formulae of Stainton *et al.* (1977). Unfiltered water samples were also analyzed for potassium (mV; Ion selective electrode No. 300744 with 1M NaCl ionic strength adjuster) and sodium using ion selective electrodes (mV; Ion selective electrode No. 300741 with 1M NH₄OH ionic strength adjuster).

Filtered water samples (1.2 µm glass microfibre Whatman GF/C filters) were subsequently frozen (excluding dissolved organic carbon and total dissolved solids samples) and analyzed a few months later at the University of Manitoba Biogeochemistry Laboratory for nitrate (NO₃-N), chloride (Cl⁻), and sulphate (SO₄⁻),

by ion chromatography using a Dionex[®] DX 500 IC auto analyzer with a PRP-X100 column (dimensions: 150 x 4.1 mm, 10 µm), and a phenol carbonate eluent (2 mM C₆H₅OH, 1.8 mM Na₂NO₃, and 1.7 mM NaHCO₃) adjusted to a pH of 10.10 with a 0.1M NaOH buffer solution.

For the determination of total dissolved solids (TDS), approximately 100 to 200 milliliters of a water sample was vacuum filtered through a 1.2 µm glass microfibre filter (Whatman GF/C) and the filtrate was then passed through a smaller 0.45 µm filter (Whatman Cellulose) to remove any remaining particulate matter. Twenty milliliters of the final filtrate was transferred into a pre-weighed 20 mL scintillation vial and dried in an oven at 103°C for a period of 24 hours and re-weighed. TDS was calculated as the difference between the tare weight and the final weight divided by the sample volume (modified from Stainton *et al.* 1977).

Total suspended solids (TSS) were analyzed by vacuum filtration of approximately 100 to 200 milliliters of a water sample through a pre-dried and weighed 1.2 µm glass microfibre filter (Whatman GF/C). Filters were retained and dried in an oven at 103°C for 24 hours and weighed, and subsequently incinerated in a muffle furnace at 550°C for one hour and re-weighed. Total suspended solids were calculated as the difference between the tare weight and the dry weight at 100°C divided by the volume of sample filtered. Organic suspended solids were calculated from the difference between the dry weight 550°C and the dry weight at 100°C divided by the sample volume. Inorganic suspended solids were calculated as the difference between the TSS and the OSS.

To obtain an estimate of algal biomass as chlorophyll a, approximately 100 to 200 milliliters of a whole water sample was filtered through 1.2 μm glass microfibre filters (Whatman GF/C). Filters were retained and frozen for a minimum of 24 hours to facilitate the lysis of the algal cell membranes. Chlorophyll pigments were extracted from the ruptured cells using 7 ml of 90% methanol and kept in a cool dark place for a period of 24 hours. Following extraction, absorbance of chlorophyll samples were measured using an Ultrospec 400 UV/Visible spectrophotometer at 665nm and 750nm before and after acidification with 0.1 M HCl. Concentrations of chlorophyll and phaeophytin were calculated using the formulae of Marker *et al.* (1980), and total chlorophyll was estimated as the sum of chlorophyll and phaeophytin. Cyanobacterial toxin samples were frozen and submitted to AlgalTox International (Pine Falls, Manitoba) for analysis. Samples were analyzed for microcystin-LR using a colorimetric protein phosphatase assay. Samples collected for algal identification were preserved immediately after collection to a final concentration of 1 percent Lugol's iodine and stored in a cool, dark place. Time constraints precluded analysis of samples collected for algal identification.

Additional water samples that were collected concurrently for Manitoba Water Stewardship (Water Quality Management Section) were submitted to Cantest for analysis of nutrients (total kjeldahl nitrogen (TKN), ammonia nitrogen ($\text{NH}_3\text{-N}$), nitrate and nitrite nitrogen ($\text{NO}_3+\text{NO}_2\text{-N}$), total phosphorus (TP), particulate phosphorus (PP), soluble phosphorus (SP)), and routine water chemistry parameters (pH, alkalinity, conductivity, TDS) in 2005, 2006 and the ice-cover

season of 2007. Raw water quality data for the open water monitoring period is provided in Appendix 1 and ice cover monitoring data is summarized in Appendix 2.

3.3 Sediment Sampling

Sediment samples were collected from 13 water sampling locations over a three day period from 8 August to 10 August, 2005 (Table 3.2, Appendix 3). Larger cobbles and gravel precluded sediment sample collection at Manipogo, and sediment samples were not collected at the Narrows site due to the current experienced that made collection problematic. Triplicate sediment cores were obtained from each site using a polyethylene coring tube (6.3 cm in diameter). The coring tube was lowered through the water column and pushed into the sediment to obtain a sample of the upper 15 centimeters of sediment. Once the tube was in place, a rubber stopper was placed over the open end to create suction, and the sample was brought to the surface where the suction was released and the sediment core was transferred into a plastic basin. The sediment sample was completely homogenized in a plastic basin using a spatula and a sub-sample was collected for analysis. Two replicate sediment samples were collected within several meters of the first sediment sampling location following the same collection and sub-sampling methods.

3.4 Sediment Analyses

Water, organic matter, and carbonate mineral content was analyzed following the method of Dean (1974). Clay crucibles were dried in an oven for 24 hours at 103°C and tare weights of each crucible was recorded. Sediment from each site was

homogenized using a, and a 2cm³ representative sub-sample was placed into each crucible using an open-ended syringe. Crucibles with sediment were dried for a period of 24 hours and weighed to determine the water content as wet weight of each sample. The dry sample was incinerated in a muffle furnace at 550°C for 1 hour causing the combustion of all organic matter. Percent organic matter content as dry weight was calculated as the difference between the weight of the sample dried at 100°C and the weight after incineration of organic matter at 550°C. The final step involved incinerating the sample at 900°C for 1 hour to combust the carbonate minerals. Percent carbonate content as dry weight was calculated as the difference between the weight of the sample at 550°C and the weight after incineration at 900°C.

Sediment samples were analyzed for particle size distribution of sand, silt, and clay using a modified ASTM hydrometer method (ASTM 1985). Samples were passed through a No. 10 sieve (2mm) to remove any large cobble, gravel, or coarse particle fractions, dried at 103°C for 24 hours to remove moisture, and were crushed to a fine powder using a mortar and pestle until no clumping was evident. Approximately 30 to 60 grams of each sample was weighed out depending on sediment composition, and samples were incinerated in a muffle furnace at 550°C for 1 hour to remove organic matter. Samples were soaked in 125 mL of sodium hexametaphosphate dispersing agent (40g Calgon® powder in 1L distilled water) for approximately 16 hours and diluted with distilled water to a final volume of 1000 ml in volumetric cylinders. The soil water slurry was homogenized for a period of exactly 1 minute, and 40 seconds later, a hydrometer reading was recorded by

slowly lowering a 152H hydrometer into the solution and reading its meniscus. A hydrometer reading was also recorded after 2 hours to account for the clay that remained in solution. Percentage sand, silt, and clay were calculated using ASTM formulae (ASTM 1985).

Soil texture for each sampling location was described on the basis of percent sand, silt and clay with the U.S. Department of Agriculture soil textural classification system (USDA 1993).

3.5 *Assiniboine River Diversion Sampling (Objective 2)*

During the two study years, the Assiniboine River Diversion operated from 5-24 April and 13 June to 2 August in 2005 and from 3 April to 28 May in 2006 which provided an opportunity to monitor the inputs of nutrients and particulate matter to Lake Manitoba (Table 3.2). Surface water samples were collected twice daily (morning and evening) from the west side of the diversion channel near the outlet into Lake Manitoba using a grab sampler during the first few days of the peak spring flow, and on a daily basis once flows had subsided. Samples were kept cool and were analyzed as soon as possible for TP, TRP, TN, NH₃ -N, nitrate+nitrite-N, chlorophyll a, TSS, turbidity, percent transmittance at 450 nm, pH, alkalinity, and specific conductance according to the same analytical methods for lake and river water samples (refer to Section 3.2). Average daily flows (cfs) required to calculate mass nutrient and particulate loading to the lake were recorded at a Water Survey of Canada continuous flow gauging station located west of the city of Portage la Prairie

(station: 05LL09). Data were provided by Manitoba Water Stewardship (Water Branch) and Water Survey of Canada.

3.6 *Statistical Analyses – Objective 1*

The data were tested for normality and homogeneity of variance by conducting a distribution analysis and generating summary statistics for each water quality parameter using JMP IN 5.1[®] statistical software (SAS Institute Inc.). The Wilcoxon Rank sum test was used to test for differences in water chemistry parameters between basins and years. Differences were considered significant at an alpha level 0.05. Pearson's correlation coefficients were calculated using JMP IN 5.1[®] to analyze relationships between nutrients (nitrogen, phosphorus) and measures of water clarity (TSS, percent transmittance, turbidity) with algal biomass (chlorophyll a), algae toxins (microcystin-LR), and wind velocity (Water Survey of Canada). A principal components analysis (PCA) was conducted on log (x+1) transformed data using PCORD5[®] to summarize the major environmental components separating lake sites.

3.7 *Statistical Analyses – Objective 3*

3.7.1 *Data Treatment and Screening*

Lake Manitoba water quality has been monitored collaboratively by the Province of Manitoba and the University of Manitoba Delta Marsh field station on a monthly basis since August 1991 at a station located approximately one kilometre offshore from the field station in south basin of the lake. Historical water quality data

for Lake Manitoba were obtained from the Province of Manitoba for the period of 1991 to 2007 for trend analysis (Manitoba Water Stewardship 2007). Data were initially screened for duplicate records, and proportion of censored data. Where samples were collected twice in one month, data were adjusted to one value per month by averaging over the records to create a single record. The database was also screened for the proportion of censored data for each variable. Variables with greater than 50 percent censored data including nitrate + nitrite nitrogen and all metals were excluded from the trend analysis. Censored data were replaced with half the reported detection limit. Following the recommendations of Hirsch et al. (1982), in cases where more than one detection limit existed for a single water quality variable, the maximum detection limit was applied to all values below the maximum detection limit.

Time series data were initially plotted for each parameter to look at the data structure and to determine whether a step trend analysis or a monotonic trend analysis was warranted. Based on the initial visual screening, all water quality variables were analyzed for monotonic trends given that the water quality record was continuous and that there were no major break points or steps in the data. A locally weighted scatterplot smooth (LOWESS) with a tension of 0.5 was then fit for each variable through time to visually examine the changes in trend over time. The LOWESS time series plot assisted in identifying outlying values, seasonality, inter-annual variation, autocorrelation, and gave an indication of the overall direction of the trend (i.e, upwards, downwards, or stationary).

3.7.2 *Volume Adjustment*

Correlation analysis was used to evaluate whether lake hydraulics covaried with water quality data collected in Lake Manitoba and whether it was necessary for data adjustment prior to trend analysis. Monthly flow data were obtained from Water Survey of Canada and the monthly inflow volume to the south basin of Lake Manitoba was calculated from 1991 to 2007 using the monthly flow records of the Assiniboine River Diversion near Portage la Prairie (Station 05LL019) and the Whitemud River at Westbourne (Station 05LL002). The monthly inflow to the south basin was calculated as the sum of the monthly discharge for the Whitemud River and the Assiniboine River Diversion and converted to total monthly inflow volume. Monthly inflow volume to the south basin explained a significant amount of the variability in the lake water chemistry data. Where total inflow volume was significantly correlated ($p < 0.05$) to a water quality variable, the relationship between each water quality variable and inflow volume was modelled using a LOWESS smooth (tension of 0.5) to remove the effects of fluctuations in inflow on lake water chemistry. Water quality data were then de-trended for water volume by subtracting the measured concentrations from the LOWESS smooth concentrations to yield the residual values. The residual values from the series are considered to be the concentrations that cannot be explained by the fluctuating inflow volumes to the south basin of Lake Manitoba.

3.7.3 *Trend Analysis*

Water quality data were analyzed using the seasonal Kendall test, a robust non-parametric method that is most commonly used in water quality trend detection

(Helsel and Hirsch 1992). The seasonal Kendall test is applicable to water quality records that are positively skewed, non-normally distributed, and is robust to missing values, censored data, and can account for the effects of seasonality, features that are all common to water quality records. The seasonal Kendall trend analysis was carried out on both the residual concentrations and the original concentrations for comparison. Trends were considered statistically significant if $p < 0.05$. The magnitude or slope of statistically significant trends was estimated using the seasonal Kendall slope estimator (Hirsch *et al.* 1982). The percent change in a statistically significant trend was calculated as the product of the seasonal Kendall slope estimator and the period of record divided by the median value for a given variable multiplied by 100. All statistical data analyses were performed using SYSTAT 12.

Table 3.1. Locations of the 15 open-water (O) and ice-cover (I) water quality sampling locations on Lake Manitoba and rivers sites including site numbers, site names, coordinates, in 2005, 2006, and ice-cover 2007.

Site ID	Provincial EMS Site Number	Site Name	Site Description	Latitude	Longitude
<i>South Basin</i>					
DELT	MB05LLS013	Delta	Lake Manitoba at Delta Marsh Field Station	50.18529	98.38195
LYPT	MB05LLS047	Lynchs Point	Lake Manitoba at Lynchs Point Beach Campground	50.24929	98.57416
AMAR	MB05LLS053	Amaranth	Lake Manitoba at Sandy Bay (Amaranth, P.R. 261)	50.13323	98.59093
MARG	MB05LKS013	Margaret Bruce	Lake Manitoba at Margaret Bruce Beach, NE of Silver Ridge	50.81770	98.79517
ELMP	MB05LNS012	Elm Point	Lake Manitoba at Elm Point	50.87514	98.52400
OAKP	MB05LNS014	Oak Point	Lake Manitoba at Oak Point (Weather Station Rd.)	50.49929	98.03867
LUND	MB05LNS011	Lundar	Lake Manitoba at Lundar Beach Provincial Park	50.72092	98.27494
STAB	MB05LLS045	St. Ambroise	Lake Manitoba at St. Ambroise Beach Provincial Park	50.27103	98.08083
<i>North Basin</i>					
NARR	MB05LKS009	Narrows	Lake Manitoba at the Narrows (P.R. 68 Bridge)	51.08451	98.78410
HORN	MB05LKS012	Watchorn	Lake Manitoba at Watchorn Beach Provincial Park	51.27347	98.56349
STPR	MB05LNS010	Steep Rock	Lake Manitoba at Steep Rock Beach Campground	51.46158	98.76693
POGO	MB05LKS014	Manipogo	Lake Manitoba at Manipogo Provincial Park, Toutes Aides	51.51731	99.54762
GUYN	MB05LKS007	Guynemer	Lake Manitoba at Guynemer, North of Point Asham	51.36778	99.10792
SPEN	MB05LKS015	Spence Bay	Lake Manitoba at Spence Bay, west of Bretechers Point	51.76913	99.35612
<i>River Sites</i>					
WMUD	MB05LLS001	Whitemud River	Whitemud River at Westbourne (Hwy. 16 Bridge)	50.13323	98.59093
WHEN	MB05LHS002	Waterhen River	Waterhen River at Waterhen (P.R. 238 Bridge)	51.82996	99.54740
FAIR	MB05LMS001	Fairford River	Fairford River at Fairford (P.T.H. 6)	51.58815	98.72810
ARD	-	Assiniboine Diversion	Assiniboine River Diversion at Lake Manitoba	50.18110	98.37862
See Table 3.2 for a detailed water sampling schedule.					

Table 3.2. A list of all water quality sampling sites and corresponding sampling periods on Lake Manitoba, 2005 to 2007.

Site ID	Site Name	2005										2006							2007	
		18 Feb	16 May	30 May	13 Jun ¹	27 Jun	11 Jul	25 Jul	8 Aug	22 Aug ^{1,2}	4 Sep	24 Mar ¹	15 May	29 May ¹	12 Jun	26 Jun	11 Jul ¹	24 Jul ¹	7 Aug	17 Aug ¹
<i>South Basin</i>																				
DELT	Delta	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
LYPT	Lynchs Point		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
AMAR	Amaranth	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
MARG	Margaret Bruce			x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
ELMP	Elm Point			x	x	x	x	x	x	x		x	x	x	x	x	x	x	x	
OAKP	Oak Point			x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
LUND	Lundar		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
STAB	St. Ambrose		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
<i>North Basin</i>																				
NARR	Narrows	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
HORN	Watchorn		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
STPR	Steep Rock		x	x	x	x	x	x	x	x		x	x	x	x	x	x	x	x	
DAVB	Davis Bay			x	x	x	x	x	x	x		x	x	x	x	x	x	x	x	
POGO	Manipogo		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
GUYN	Guynemer	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
SPEN	Spence Bay				x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
<i>River Sites</i>																				
WMUD	Whitemud River		x	x	x	x	x	x	x	x		x	x	x	x	x	x	x	x	
WHEN	Waterhen River		x	x	x	x	x	x	x	x		x	x	x	x	x	x	x	x	
FAIR	Fairford River	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
ARD	Assiniboine Diversion ³																			

¹ Additional water samples collected for Manitoba Water Stewardship (Water Quality Management Section) and submitted to Cantest for analysis.

² Sediment samples collected for particle size, organics, and carbonate analysis.

³ Assiniboine River Diversion sampled daily or twice daily (during high flow conditions) from 4-25 April and 13 June to 2 August in 2005 and from 3 April to 28 May in 2006.

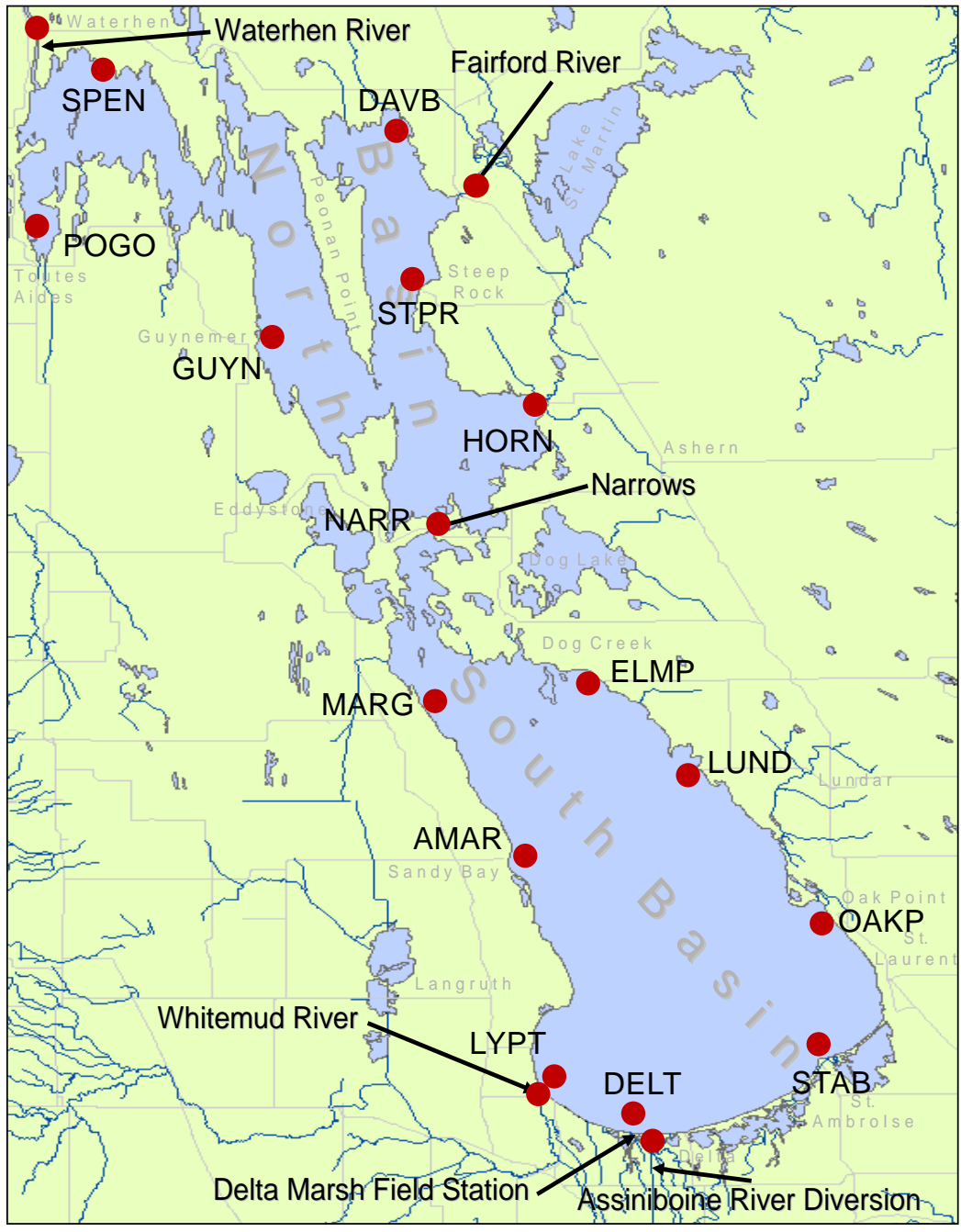


Figure 3.1. Schematic diagram of the Lake Manitoba study area showing the water quality locations in the open water and ice cover seasons in 2005, 2006, and 2007. Refer to Table 3.2 for further details on frequency, season, and location of sample collection.



Plate 3.1. Elaine Page collecting a water sample at Spence Bay located in the north basin of Lake Manitoba, June 12th, 2006.



Plate 3.2. Winter water quality sampling using a Bombardier to access pelagic sites on Lake Manitoba, March 24th 2006.

4.0 RESULTS & DISCUSSION: SPATIAL AND TEMPORAL VARIATION IN WATER QUALITY AND NUTRIENT LOADING TO LAKE MANITOBA

4.1 *Climatological Conditions and Flows*

Temperature and precipitation differed markedly between the 2005 and 2006 water quality monitoring periods at the Delta Marsh meteorological station. In 2005, the climate was characterized by above normal precipitation through the spring and mid-summer and again through the late fall and early winter (Figure 4.1). Precipitation in June was nearly double the climate normal because of heavy rain events (>15 mm). Early fall was comparatively dry and precipitation was approximately half of the climate normal. Cool conditions predominated in 2005 with air temperature above normal for April and below normal or near normal from May to September (Figure 4.2). The climate in 2006 was comparatively dry and precipitation was approximately half of the climate normal throughout spring and summer. Temperature was above normal in 2006 from April to September (Figure 4.2).

In both study years, flows in the Waterhen River increased from April to June and declined in August (Figure 4.3). Flows for the Waterhen River were highest in 2006 for the period of April to September and were significantly different between years ($p < 0.0001$). In 2005, the Whitemud River had two periods of peak flow (Figure 4.3). Flows peaked in April ($108 \text{ m}^3/\text{s}$) during the spring freshet and again in July ($90 \text{ m}^3/\text{s}$) coincident with intense summer precipitation events. Peak flow during the spring freshet was higher in 2006, but flows quickly declined to below $15 \text{ m}^3/\text{s}$ by mid-June. In 2005, flows were higher in late spring and summer and remained above $15 \text{ m}^3/\text{s}$ from April to September. Flows were

considered significantly different between years for the Whitemud River ($p < 0.0001$).

The Assiniboine River Diversion operated for a total of 72 days in 2005 and for 58 days in 2006. The length of operation in 2005 was only second to the longest operation of the Diversion in 1999, when water flowed for a total of 87 days. In 2005, flows in the Diversion began on April 4th, peaked at 500 m³/s, and by April 25th there was no flow in the Diversion channel (Figure 4.3). Historically, peak spring flows in the Diversion channel have only ever been higher in 1976 (736 m³/s), 1996 (581 m³/s), and 1974 (515 m³/s). Because of heavy rain events in May and June of 2005, the Diversion was re-opened on June 13th. The peak flow during summer operation was low in comparison to the spring (305 m³/s), although the summer period of operation was much longer (51 days) as compared to the spring operation (21 days). In 2006, flow in the Diversion channel began on April 4th, peaked at 347 m³/s, and water continued to flow for nearly 2 months. Overall, the total water volume delivered from the Diversion to Lake Manitoba was approximately 40 percent greater in 2005 (11109 m³) in comparison to 2006 (6584 m³), and flow in 2005 was the fourth greatest since the operation of the Diversion in 1970. Total water volume was only ever higher in 1976 (20279 m³) in 1995 (15918 m³), and 1999 (13051 m³).

4.2 Total Nitrogen

Over the entire open-water (May to October) study period in 2005 and 2006, the average total nitrogen concentration for the whole lake was 4.06 mg/L (± 0.16) and individual total nitrogen concentrations ranged from 0.001 to 20.1 mg/L (Table 4.1). The average nitrate-nitrite nitrogen and dissolved ammonia concentrations were 0.141 mg/L and 0.013 mg/L, respectively. On average, dissolved inorganic nitrogen comprised a relatively small fraction of the total nitrogen pool (4 %) in Lake Manitoba, while organic

nitrogen comprised approximately 96 % of the total nitrogen pool.

Total nitrogen concentrations did not differ between years in the north basin ($p = 0.133$) or the south basin of Lake Manitoba ($p = 0.176$; Table 4.2). Seasonal patterns in total nitrogen were relatively similar in the north and south basins in 2005 and 2006, but opposite seasonal patterns occurred for total nitrogen in the two study years (Figure 4.4). In 2005, total nitrogen was highest in May and declined throughout the open water season and was lowest in the fall (Figure 4.4). In 2006, total nitrogen was lowest in July and increased to a maximum total nitrogen concentration in August (Figure 4.4). In June and July of 2005, two large peaks of total nitrogen in the north basin were attributed to elevated nitrogen concentrations at a few stations located in isolated bays of the north basin (i.e., Spence Bay, Davies Bay).

In comparison to the south basin, total nitrogen concentrations were 22 and 25 % greater in the north basin of Lake Manitoba in 2005 ($p = 0.021$) and 2006 ($p = 0.009$), respectively (Table 4.3). The lowest total nitrogen concentrations were generally found in the south western area of the south basin at the stations located near the inflow of the Whitemud River (Lynch's Point, Figure 4.5). Total nitrogen concentrations in the south basin were generally lower (< 2.60 to 3.84), while concentrations at the Narrows and at stations in the north basin ranged from 3.84 to 5.24 mg/L. Nitrogen concentrations were variable in the north basin with nearly a two fold difference among stations. The lowest total nitrogen concentration occurred in at the very western edge of the north basin in an isolated bay (in Toutes Aides Bay at the Manipogo station), while the highest concentrations were found east of the inflow of the Waterhen River (at Spence Bay and Guynemer stations). Nitrogen was similar among sites in the eastern arm of the north basin

but concentrations were slightly lower than concentrations found in the west arm of the north basin. Total nitrogen was weakly correlated to nitrate-nitrite nitrogen ($r = 0.173$, $p = 0.047$) and dissolved ammonia ($r = -0.214$, $p = 0.013$) in the south basin, but no significant correlation was found for the north basin.

4.3 Nitrate

Nitrate comprised more than 80 percent of the dissolved inorganic nitrogen fraction and concentrations ranged from 0.005 to 1.23 mg/L in Lake Manitoba (Table 4.1). Nitrate concentrations were almost two times higher in 2005 as compared to 2006 and differences were highly significant between years in both basins of Lake Manitoba ($p < 0.0001$, Table 4.2). Seasonal patterns in nitrate were generally similar between north and south basins of the lake (Figure 4.4). Overall, nitrate was generally the lowest at the end of June or beginning of July in both years, with exception of the south basin in 2005 when concentrations were lowest in May and September. The north basin had the greatest seasonal range in nitrate concentrations and was 4 times greater in 2005 (0.120 to 0.200 mg/L) in comparison to 2006 (0.08 to 0.10 mg/L).

Nitrate was 13 % greater in the south basin in 2005 ($p = 0.004$), although there was no significant difference in nitrate concentrations between basins in 2006 ($p = 0.625$; Table 4.3). Average spatial patterns in nitrate indicated that concentrations were highest at the very south end of the lake (0.140 to 0.219 mg/L), particularly in the south east area of the lake (near the St. Ambrose station), along the eastern shoreline of the south basin, and also near the inflow of the Whitemud River (Figure 4.5). Nitrate concentrations were quite variable within the south basin of Lake Manitoba with nearly a two fold difference between the Elm Point and St. Ambrose stations. Nitrate concentrations in the north basin were

generally lower than those in the south basin and concentrations ranged from between 0.126 and 0.140 mg/L with exception of one station in the north west arm of the lake (Spence Bay) which was < 0.123 mg/L.

In the north basin, nitrate was correlated to specific conductivity ($r = 0.667$, $p < 0.0001$), turbidity ($r = 0.607$, $p < 0.0001$), DIN:TRP ($r = 0.546$, $p < 0.0001$), specific conductivity ($r = 0.382$, $p = 0.003$), sodium ($r = -0.625$, $p < 0.0001$), and water temperature ($r = -0.311$, $p = 0.0007$). Nitrate was moderately correlated to silica ($r = 0.485$, $p < 0.0001$), DIN:TRP ($r = 0.380$, $p < 0.0001$) alkalinity ($r = 0.346$, $p < 0.0001$) in the south basin of the Lake Manitoba, and was also significantly correlated ($p < 0.05$) with other variables (chlorophyll a, TSS, ISS, TN, TDS, conductivity, sodium, DIC) although the correlation was weak ($r < 0.3$).

4.4 Ammonia

Dissolved ammonia comprised the smallest fraction of the dissolved inorganic nitrogen pool in Lake Manitoba (< 20 %) and concentrations ranged from 0.005 to 0.150 mg/L (Table 4.1). Ammonia did not differ between study years in the north basin ($p = 0.584$). However, dissolved ammonia concentrations were 33 percent greater in 2005 as compared to 2006 in the south basin of Lake Manitoba ($p = 0.047$; Table 4.2). In the south basin, ammonia concentrations were greatest in May and mid-July in both years, while concentrations were greatest in either June or July the north basin (Figure 4.4). The seasonal range in dissolved ammonia concentrations in the north basin of Lake Manitoba was similar in both study years (0.005 to ca. 0.015 mg/L). In the south basin of Lake Manitoba, there was a slightly greater seasonal range in dissolved ammonia in 2006 (0.010 to 0.029 mg/L) compared to that of 2005 (0.013 to 0.025 mg/L). However, concentrations in

2005 were generally higher than in 2006 (Figure 4.4).

Dissolved ammonia concentrations were 2 to 3 times greater in the south basin of Lake Manitoba in both study years ($p < 0.0001$, Table 4.3). Greatest concentrations of dissolved ammonia were found in the south west area of the south basin near the inflow of the Whitemud River (> 0.019 mg/L, Figure 4.5). Relatively high ammonia concentrations also occurred at several other stations in the south basin (ca. 0.016 to 0.019 mg/L at Delta, Oak Point, and Margaret Bruce stations). However, there was no clear latitudinal or longitudinal spatial gradient within the south basin. Ammonia was quite low at all stations in the eastern arm of the north basin (< 0.006 mg/L) and only slightly higher in the western arm at two stations closest to the inflow of the Waterhen River (0.006 to 0.010 mg/L).

Dissolved ammonia was weakly correlated to nitrate ($r = -0.226$, $p = 0.001$), pH ($r = -0.221$, $p = 0.017$), and specific conductivity ($r = -0.193$, $p = 0.039$) in the north basin of Lake Manitoba. A weak positive correlation existed between dissolved ammonia and microcystin-LR in the north basin of Lake Manitoba ($r = 0.249$) although the relationship was not considered significant ($p = 0.051$) at an alpha level of 0.05. In the south basin, turbidity ($r = 0.570$, $p < 0.0001$), total phosphorus ($r = 0.401$, $p < 0.0001$), total reactive phosphorus ($r = 0.358$, $p < 0.0001$), organic suspended solids ($r = 0.368$, $p < 0.0001$), and dissolved inorganic carbon ($r = 0.370$, $p < 0.0001$) were moderately correlated to dissolved ammonia. Total nitrogen, DIN:TRP, water temperature, sodium, chloride, pH, chlorophyll a, and alkalinity were also significantly correlated with dissolved ammonia ($p < 0.05$) in the south basin of Lake Manitoba but correlations were weak ($r < 0.3$).

4.5 Total Phosphorus

Over the entire open-water (May to October) study period in 2005 and 2006, the

average total phosphorus concentration for the whole lake was 0.079 mg/L (± 0.0005) and individual total phosphorus concentrations ranged from 0.005 to 0.453 mg/L (Table 4.1). Lake Manitoba is considered eutrophic according to the Organization for Economic Cooperation and Development's trophic classification of lakes (OECD 1982). Almost 80 % of the total phosphorus pool in Lake Manitoba was in the form of organic phosphorus.

In 2005, total phosphorus concentrations were higher in the north basin (13 %) and the south basin (19%) of Lake Manitoba, but differences were not significant between years ($p = 0.534$ and 0.528 , respectively, Table 4.2). Seasonal patterns in total phosphorus differed between years in the south basin (Figure 4.6). In 2005, total phosphorus concentrations were lowest in the south basin in late spring (0.038 mg/L) and increased over the open water season to the highest total phosphorus concentration in September (0.208 mg/L). Greatest increases in total phosphorus were evident between the months of June and July where concentrations increased nearly 3 fold in the south basin. In contrast, total phosphorus concentrations in the south basin exhibited an opposite seasonal pattern in 2006. Phosphorus concentrations were greatest in May (0.153 mg/L) and declined to the lowest average concentration in early August (0.026 mg/L). At the end of August, there was a sharp increase in total phosphorus increased to nearly 0.090 mg/L in the south basin of the lake.

Seasonal patterns in total phosphorus were generally similar between years in the north basin (Figure 4.6). In 2005 and 2006, total phosphorus concentrations were highest in the spring, lowest in the summer, and increased in the fall. In comparison to 2005, the spring peak in total phosphorus was greater in 2006 and the decline in the summer was much more gradual. In 2006, there was a sharp rise in fall total phosphorus concentrations

(concentrations increased from 0.018 mg/L to 0.063 mg/L over 2 week period), while the increase was much more gradual (concentrations increased from 0.051 mg/L to 0.064 mg/L).

Total phosphorus concentrations were almost two times greater in the south basin as compared to the north basin in 2005 ($p = 0.0007$) and 2006 ($p < 0.0001$; Table 4.3). Higher phosphorus concentrations typically occurred in the southern two thirds of the south basin of Lake Manitoba and concentrations generally declined moving northwards in the lake (Figure 4.5). The average open water concentration was greatest in the south eastern area of the south basin near the inflow of the Whitemud River where total phosphorus exceeded 0.1 mg/L. South basin stations that were located further northward (e.g., Margaret Bruce, Elm Point) were more similar to phosphorus concentrations at the Narrows and Watchorn Bay. Total phosphorus was typically lowest (< 0.05 mg/L) in the north western area of the north basin (in Toutes Aides Bay) and at the very northern end of the lake in the east arm of the north basin. Other stations in the north basin had relatively low average open water phosphorus concentrations (< 0.055 mg/L).

In the north basin, total phosphorus was positively correlated to total reactive phosphorus ($r = 0.220$, $p = 0.018$), inorganic suspended solids ($r = 0.266$, $p = 0.004$), and negatively correlated to microcystin-LR ($r = -0.262$, $p = 0.039$). Total phosphorus was moderately correlated to total nitrogen ($r = -0.459$, $p < 0.0001$), dissolved ammonia ($r = 0.401$, $p < 0.0001$), organic suspended solids ($r = 0.332$, $p < 0.0001$), inorganic suspended solids ($r = 0.318$, $p = 0.0002$), and total suspended solids ($r = 0.313$, $p = 0.0002$) in the south basin of Lake Manitoba. Total phosphorus also exhibited weak but significant positive correlations with turbidity ($r = 0.200$) and chlorophyll a ($r = 0.190$) in the south basin.

4.6 Total Reactive Phosphorus

The pooled 2005 and 2006 average total reactive phosphorus concentration for Lake Manitoba (0.018 mg/L) comprised a relatively small fraction of the total phosphorus pool in the lake (23 %), and individual measurements of total reactive phosphorus concentrations ranged from 0.005 to 0.287 mg/L (Table 4.1).

Total reactive phosphorus did not differ between years in the north basin of Lake Manitoba ($p = 0.536$), and concentrations were relatively low in both years (0.011 mg/L in 2005 and 2006; Table 4.2). In the south basin, total reactive phosphorus concentrations were two times greater in 2005 (0.032 mg/L) as compared to 2006 (0.015 mg/L) although differences were not significant ($p = 0.309$; Table 4.2). There appeared to be no consistent seasonal trend in total reactive phosphorus in the north basin of Lake Manitoba, as concentrations remained relatively low (generally <0.020 mg/L) throughout the open water season (Figure 4.6). In the south basin, seasonal concentrations of total reactive phosphorus were lowest in early June, peaked in the summer, and declined in the fall in both years. However, the seasonal range in total reactive phosphorus concentrations in the south basin was much greater in 2005 (0.020 to 0.060 mg/L) as compared to 2006 (0.015 to 0.025 mg/L). Large variability was associated with the marked increase in total reactive phosphorus concentrations during June and July of 2005 in the south basin of Lake Manitoba (Figure 4.6). This was mainly attributed to high concentrations at a station near the inflow of the Whitemud River (> 0.2 mg/L at Lynch's Point) and at the lake station near the inflow of the Assiniboine River Diversion (> 0.1 mg/L at Delta station). Reactive phosphorus concentrations were comparatively low (<0.025 mg/L) at all other south basin sites during this period.

Total reactive phosphorus was 65 % and 27 % greater in the south basin of Lake Manitoba in 2005 and 2006, respectively ($p < 0.0001$ and $p = 0.002$, respectively, Table 4.3). Total reactive phosphorus concentrations were greatest (> 0.018 mg/L) near the inflow the Whitemud River and at the station located offshore from the Delta Marsh Field Station (Figure 4.7). Total reactive phosphorus concentrations were similar at all other stations in the south basin (0.014 to 0.018 mg/L), with exception of a single station (at the very north eastern area of the south basin) which was comparable to concentrations in the narrows and the north basin. In comparison to the south basin, total reactive phosphorus was relatively low at all stations in the north basin (0.007 to 0.014 mg/L; Figure 4.7), and was lowest at the two stations located just north of the narrows (at Guynemer and Watchorn stations) where average concentrations did not exceed 0.008 mg/L.

There was a strong negative correlation between total reactive phosphorus and DIN:TRP in the north basin of Lake Manitoba ($r = -0.839$, $p < 0.0001$). Weak ($r < 0.3$) but significant ($p < 0.05$) positive correlations existed for silica and total phosphorus, and total reactive phosphorus was negatively correlated to total nitrogen in the north basin. Total reactive phosphorus was strongly correlated to DIN:TRP ($r = -0.858$, $p < 0.0001$), moderately correlated to total suspended solids ($r = 0.365$, $p < 0.0001$), dissolved ammonia ($r = 0.358$, $p < 0.0001$), and water transparency ($r = -0.389$, $p < 0.0001$), and weakly correlated to ($r < 0.3$, $p < 0.05$) chlorophyll a, total phosphorus, organic suspended solids, alkalinity, turbidity, dissolved inorganic carbon, and pH in the south basin of Lake Manitoba.

4.7 Soluble Reactive Silica

In 2005 and 2006, soluble reactive silica concentrations ranged from 0.46 mg/L to 7.96 mg/L in Lake Manitoba (mean = 3.05 mg/L; Table 4.1). Silica concentrations differed

between years in Lake Manitoba. In the north basin, soluble reactive silica concentrations were 12 % greater in 2006 ($p = 0.025$), while in the south basin, silica concentrations were 35 % greater in 2005 ($p < 0.0001$; Table 4.2).

Soluble reactive silica concentrations were lowest in the spring and generally increased over the open water season reaching highest concentrations in August and September (Figure 4.6). This is in exception to the south basin in 2005, when silica concentrations peaked early in July (ca. 6.0 mg/L), had already started to decline by mid-July, and continued to decline through August (to 4.5 mg/L). In 2005, the range in silica concentrations was much greater in both the north basin (1 to 3.7 mg/L) and the south basin (1.5 mg/L to 6 mg/L) of Lake Manitoba. Silica concentrations in 2006 did not vary more than 1.0 and 3.5 mg/L in the north basin and south basin of Lake Manitoba, respectively.

In 2005, soluble reactive silica concentrations were two times higher ($p < 0.0001$) in the south basin in comparison to the north basin, although there was no difference between basins in 2006 ($p = 0.203$; Table 4.3). Silica concentrations were generally greater than 3.0 mg/L at all stations in the south basin of Lake Manitoba, and concentrations were greatest (>3.7 mg/L) in the south west area near the inflow of the Whitemud River and in the north east area of the south basin at the station closest to the narrows. On average, silica concentrations were nearly 25 % lower at stations in the north basin with the lowest concentrations (<1.85 mg/L) at two stations in the north basin of the lake (at Guynemer and Davies Bay stations). It is interesting to note that other dissolved nutrients (e.g., dissolved ammonia and total reactive phosphorus) were also quite low at the Guynemer station in the north basin of Lake Manitoba.

Silica was moderately correlated to pH ($r = 0.387$, $p < 0.0001$), chlorophyll a ($r = 0.363$, $p < 0.0001$), water temperature ($r = 0.326$, $p = 0.0003$), DIN:TRP ($r = -0.350$, $p = 0.0001$), and TN ($r = -0.309$, $p = 0.0007$) in the north basin of Lake Manitoba. Nitrate ($r = 0.485$, $p < 0.0001$), pH ($r = 0.485$, $p < 0.0001$), chlorophyll a ($r = 0.365$, $p < 0.0001$) were among the variables correlated to soluble reactive silica in the south basin.

4.8 Chlorophyll a

Based on the mean whole lake chlorophyll a concentration ($25 \mu\text{g/L}$), Lake Manitoba is considered eutrophic according to the OECD trophic classification of lakes (OECD 1982). In 2005 and 2006, chlorophyll a concentrations for Lake Manitoba ranged from $2.8 \mu\text{g/L}$ to $326 \mu\text{g/L}$ (Table 4.1). Chlorophyll a did not differ between years in the north basin of Lake Manitoba ($p = 0.547$; Table 4.2). In the south basin, chlorophyll a concentrations were 35 % greater in 2005, although concentrations were not considered significantly different between years at an alpha level of 0.05 ($p = 0.068$; Table 4.2).

Chlorophyll a exhibited different seasonal patterns in 2005 and 2006 (Figure 4.8). However, the overall pattern in chlorophyll a concentrations between basins in each year was similar. In 2005, chlorophyll a was generally lowest in the May or early June and greatest in August in both basins. In contrast, chlorophyll a in 2006 exhibited an initial spring peak in May, declined in early June, and was followed by an increase through the second half of June. Chlorophyll a concentrations then remained relatively stable through July and August in both basins of Lake Manitoba. In 2005, the seasonal range in chlorophyll a concentrations was much greater for the south basin (20 to $60 \mu\text{g/L}$) as compared to the north basin (8 to $15 \mu\text{g/L}$; Figure 4.8). In 2006, concentrations in the south basin ranged from 15 to $37 \mu\text{g/L}$ (with spring and summer peaks being approximately

equal). The peak spring chlorophyll a concentration in the north basin (20 µg/L) was 25 % higher than the chlorophyll a concentration in late summer (15 µg/L). Organic suspended solids was moderately correlated to chlorophyll a concentrations in the north ($r = 0.328$, $p = 0.003$) and south basins ($r = 0.340$, $p < 0.0001$) of Lake Manitoba, and seasonal patterns in organic suspended solids were generally similar seasonal patterns in chlorophyll a (Figure 4.8).

Chlorophyll a concentrations were two to three times greater in the south basin as compared to the north basin in both study years and differences were considered significant ($p < 0.0001$; Table 4.3). In 2005, concentrations were highest in the south and south eastern area of the lake (at Delta, St. Ambroise, and Oak Point stations) where average open water concentrations were greater than 27 µg/L (Figure 4.7). Average chlorophyll a was greatest at the Delta station (> 39 µg/L) at the very south end of Lake Manitoba. Chlorophyll a declined moving northwards in the lake and was lowest in isolated bays of the north basin (Manipogo, Davies Bay, and Watchorn stations) where average chlorophyll (May to September) was less than 9 µg/L. Chlorophyll a ranged from 9 to 13 µg/L at all other stations in the north basin of Lake Manitoba (including the narrows station).

In addition to organic suspended solids, chlorophyll a was also correlated to silica and water transparency in the north (Silica: $r = 0.363$, $p < 0.0001$, Water Transparency: $r = -0.225$, $p = 0.015$) and south basins (Silica: $r = 0.365$, $p < 0.0001$, Water Transparency: $r = -0.240$, $p = 0.006$) of Lake Manitoba. No significant relationships existed between chlorophyll a and nitrogen or phosphorus in the north basin. In the south basin, chlorophyll a was weakly ($r < 0.3$) but significantly correlated ($p < 0.05$) to total phosphorus, total reactive phosphorus, dissolved ammonia, and nitrate. A weak correlation also existed

between water temperature and chlorophyll a in the south basin ($r = 0.171$, $p = 0.049$).

4.9 DIN:TRP

Lake Manitoba was considered phosphorus limited over the open water season (37:1, Table 4.1). However, phytoplankton nutrient status ranged from severe nitrogen limitation (1:1) to severe phosphorus limitation (261:1). Although Lake Manitoba was phosphorus limited in both study years, the lake was nearly twice as phosphorus limited in 2005, and differences were considered significant in the north basin and the south basin of the lake ($p < 0.0001$ and $p = 0.004$, respectively, Table 4.2).

There was no clear seasonal coherence in the DIN:TRP ratios between basins or within or between years in Lake Manitoba (Figure 4.8). In 2005, peaks in the DIN:TRP ratio (ca. 60:1) in the south basin coincided with chlorophyll a peaks in early July and late August. The DIN:TRP ratio was greatest in the north basin in early June (100:1) when chlorophyll a concentrations were lowest in the north basin. Through June and July, the DIN:TRP ratio declined sharply in the north basin (from 100:1 to 20:1) and fluctuated between ratios of 20:1 and 60:1 during July and August. Higher chlorophyll a concentrations occurred in the north basin during July and August when the north basin became less phosphorus limited. In 2006, there was little variation in the DIN:TRP ratios in the north basin (range = 22:1 to 30:1). The south basin was considered nitrogen limited in late June and early July (ca. 12:1 to 14:1) when algal biomass increased to 37 $\mu\text{g/L}$, and was the most phosphorus limited in early June (39:1) in association with the lowest seasonal algal biomass (15 $\mu\text{g/L}$).

The north basin of Lake Manitoba was significantly more phosphorus limited than the south basin. DIN:TRP ratios were approximately 20 % higher in the north basin in 2005 ($p =$

0.0061) and 2006 ($p = 0.050$; Table 4.3). Nutrient limitation was highly variable within the south basin of Lake Manitoba (Figure 4.7). The DIN:TRP ratio was lowest in the south west area of the lake (<15:1) near the inflow of the Whitemud River. However, the ratio increased from <15:1 (or nitrogen limitation) to >39:1 (phosphorus limitation) moving eastward along the shoreline of the south basin (to the St. Ambroise station). Moving northwards, the two stations located closest to the narrows were more phosphorus limited (32 to 39:1) relative to most of the other stations in the south basin (with exception of the Lundar and St. Ambroise stations which were more phosphorus limited). The north basin was generally more phosphorus limited (39:1 to 56:1), particularly along the east arm of the north basin (at Watchorn and Davies Bay stations), in the lower part of the west arm (at Guynemer), and at the Narrows (Figure 4.7). It is interesting to note that the north basin was less phosphorus limited in the west arm at the two stations closest to the inflow of the Waterhen River (at Manipogo and Spence Bay).

In the north basin, the DIN:TRP ratio was strongly correlated to total reactive phosphorus ($r = -0.839$, $p < 0.0001$) and moderately correlated to silica ($r = -0.350$, $p < 0.0001$), nitrate ($r = 0.546$, $p < 0.0001$), and turbidity ($r = 0.410$, $p < 0.0001$). Both nitrate (0.380, $p < 0.0001$) and total reactive phosphorus ($r = -0.858$, $p < 0.0001$) were significantly correlated to the DIN:TRP ratio in the south basin of Lake Manitoba. No significant correlations existed between chlorophyll a and the DIN:TRP ratio or organic suspended solids in either basin of Lake Manitoba.

4.10 *Microcystin-LR*

Microcystin-LR was analyzed on a limited number of samples ($n = 165$) in 2005 and 2006 in an effort to assess the range of microcystin concentrations in Lake Manitoba. The

average microcystin concentration (0.15 µg/L; Table 4.1) and the range of concentrations (<0.1 to 1.41 µg/L) in individual samples collected from Lake Manitoba was below the proposed recreational water quality guideline for microcystin (20 µg/L), and the Canadian drinking water guideline (1.5 µg/L;).

Microcystin concentrations were almost 4 times greater in the south basin (0.21 mg/L ± 0.02) as compared to the north basin (0.06 mg/L ± 0.004) of Lake Manitoba. Only 3 % of samples collected in the north basin were above the analytical detection limit for microcystin (0.1 µg/L), whereas 75 % of samples collected in the south basin exceeded the detection limit. Microcystin was negatively correlated to total phosphorus in the north basin of Lake Manitoba ($r = -0.262$, $p = 0.039$). In the south basin, microcystin was negatively correlated to nitrate ($r = -0.209$, $p = 0.035$) and total nitrogen ($r = -0.201$, $p = 0.043$).

4.11 Water Clarity

Water clarity was quite variable in Lake Manitoba and ranged from highly turbid (Turbidity >100 NTU, TSS >300 mg/L) to relatively clear conditions (TSS and Turbidity <1 mg/L and <1 NTU; Table 4.1). Turbidity, total suspended solids, inorganic suspended solids, and % water transparency were all significantly correlated in both basins of Lake Manitoba ($p < 0.05$). Inorganic suspended solids comprised approximately 50 % of the TSS in the south basin. In contrast, much of the TSS in the north basin (75%) of Lake Manitoba was present as OSS. All measures of water clarity indicated that the south basin of Lake Manitoba was significantly more turbid in 2005 ($p < 0.05$) in comparison to 2006 (Table 4.2). In the north basin, turbidity was nearly two times greater in 2005 ($p < 0.0001$). However, there was no significant difference between years for TSS ($p = 0.780$) and ISS ($p = 0.391$) in the north basin of Lake Manitoba.

In 2005, water in the south basin was generally most turbid in June and July (Figure 4.9). Both total and inorganic suspended solids increased more than 4 fold throughout August. Despite this, there was no corresponding increase in turbidity. In 2005, turbidity (and variance) was greatest in the south basin at the end of June because of elevated turbidity readings (>50 NTU) at several stations in the south basin (Delta, Lynch's Point, and Amaranth). In 2005, there was no apparent seasonal variation in turbidity, TSS or ISS in the north basin of Lake Manitoba. Turbidity did not follow the same seasonal pattern in 2006 (Figure 4.9). Turbidity, TSS and ISS were greatest in the in May and lowest in August in both basins of Lake Manitoba. In the south basin, the large spring peak in turbidity was attributed to the high readings (>100 NTU) at both the Delta and Lynch's Point stations.

The south basin of the lake was turbid than the north basin of the lake in 2005 and 2006 ($p < 0.0001$ for all measures of water clarity; Table 4.3). Turbidity was 2 to 3 times greater in the south basin in 2005 and 2006, respectively. Turbidity was generally higher in the southern half of the lake's south basin (> 20 NTU) with the greatest turbidity (>27 NTU) at the very southern end of the lake (at Lynch's Point and Delta stations; Figure 4.10). The two stations in the south basin located closest to the narrows were notably less turbid (than those stations located further south in the lake. In the north basin, turbidity was generally < 13 NTU with the greatest water clarity in the north west arm (<10 NTU) near the inflow of the Waterhen River. While there were some differences, the overall spatial pattern of other measures of water clarity (TSS, ISS, % water transparency) was similar to that of turbidity (Figure 4.10).

4.12 Major Ions and General Chemistry

Lake Manitoba was slightly brackish in 2005 and 2006 (TDS: 725 mg/L, specific

conductivity: 1283 $\mu\text{S}/\text{cm}$), although dissolved solids and conductivity were highly variable in the lake and were as low as 110 mg/L and 424 $\mu\text{S}/\text{cm}$, respectively (Table 4.1). Total dissolved solids and specific conductivity were significantly correlated in both the north and south basins of Lake Manitoba ($p < 0.05$). Chloride, sodium and sulphate were the dominant ions in Lake Manitoba with average concentrations of 226, 128, and 114 mg/L, respectively. In the south basin, significant positive correlations existed between chloride, sulphate, and sodium ($r = 0.29$ to 0.410 , $p < 0.05$). However, these ions were not correlated in the north basin of Lake Manitoba. Conductivity was significantly different between years in the north and south basins of Lake Manitoba ($p < 0.0001$), although there was no significant difference between years for dissolved solids ($p > 0.05$). Chloride, sodium, and sulphate concentrations in the south basin were significantly lower in 2005 ($p < 0.0001$ to 0.035 ; Table 4.2). In the north basin, only sodium differed between years ($p < 0.0001$) with lower concentrations in 2005.

There was no clear seasonal pattern in total dissolved solids and specific conductivity in Lake Manitoba (Figure 4.11). Of note was the lower total dissolved solids concentrations in the south basin through June and the beginning of July of 2005. Chloride peaked in June in the south basin of Lake Manitoba, declined in June and July, and then remained relatively stable through August and September (Figure 4.12). In the north basin, chloride concentrations in June and July were approximately 50 percent lower than concentrations in the spring in later summer. Chloride and sodium concentrations were much less variable in 2006 and were generally lowest in spring and increased over the open water season. No seasonal trend in sulphate was observed in 2005 or 2006 (Figure 4.12).

Total dissolved solids and conductivity were 11 and 15 % higher in the north basin of Lake Manitoba in 2005 ($p = 0.007$ and <0.0001 , respectively; Table 4.3). However, differences between basins were not considered significant in 2006 ($p>0.05$). Chloride and sodium were approximately 22 and 26 % greater in the north basin in both study years, respectively ($p <0.0001$ for chloride and sodium in 2005 and 2006). Total dissolved solids, conductivity, chloride, and sodium were lowest along the southern beach ridge of Lake Manitoba (at Delta, St. Ambroise and Lynch's Point stations) and were lowest near the inflow of the Whitemud River (Figure 4.13). These same parameters were typically highest along the eastern arm of the north basin and were similar or slightly lower in the western arm of the north basin. Sulphate was approximately 30 % higher in the south basin of Lake Manitoba in both study years ($p <0.0001$ in 2005 and 2006; Table 4.3). Sulphate was lowest at all three stations the western arm of the north basin (< 81 mg/L) and slightly higher in the eastern arm (81 to 98 mg/L; Figure 4.14). Sulphate concentrations were greatest along the east side of the south basin in Lake Manitoba (>134 mg/L). Stations in the south and south western areas of the lake (Lynch's Point, Delta, St. Ambroise, and Amaranth) had moderate sulphate concentrations (98 to 134 mg/L; Figure 4.14).

Lake Manitoba was alkaline, well buffered, and dissolved inorganic carbon (DIC) was sufficient for primary production (Table 4.1). Alkalinity, pH, and DIC did not differ between years in the north basin of Lake Manitoba ($p = 0.092$ to 0.116) but differed between years in the south basin ($p <0.0001$ to 0.018 ; Table 4.2). Dissolved inorganic carbon and alkalinity were strongly correlated in the north basin ($r = 0.987$, $p <0.0001$) and the south basin ($r = 0.886$, $p <0.0001$) of Lake Manitoba. A negative correlation existed between DIC and pH in the south basin ($r = -0.297$, $p <0.0001$) but not for the north basin.

No significant correlation was detected between pH and alkalinity in Lake Manitoba. Alkalinity and DIC were greater in the south basin of the lake in both study years ($p < 0.0001$; Table 4.3). pH was higher in the south basin as compared to the north basin in 2005 ($p = 0.02$). However, there was no difference between basins in 2006 ($p = 0.085$). Spatial patterns in alkalinity and DIC concentrations were similar in Lake Manitoba (Figure 4.14). Alkalinity and DIC were highest in the south basin (218 to 233 mg/L and 47 to 56 mg/L, respectively) with the greatest concentrations at the Lynch's Point and Oak Point stations. Concentrations of alkalinity and DIC were slightly lower in the eastern arm of the north basin and lowest in the western arm at stations closest to the Waterhen River.

4.13 Multivariate Analysis

Principal components analysis was used to describe water quality conditions at 15 stations in Lake Manitoba. In 2005, the first and second axes were considered significant, and explained 84 % of the total cumulative variance in the data (Figure 4.15). The first axis was negatively correlated with water clarity, nutrients (TP, TRP, Si), and algal biomass. Major ions (chloride, sodium), conductivity, total nitrogen, and the DIN:TRP ratio was positively correlated to the first axis. The second axis exhibited a relatively strong negative correlation with sulphate and was also negatively correlated with DIN:TRP, alkalinity, and conductivity. On a broad scale, the PCA separated the 15 Lake Manitoba stations according to lake basin along the first axis, with the north basin sites on the right side of the ordination biplot and the south basin sites on the left side of the ordination biplot (Figure 4.15). The Lake Manitoba Narrows station was grouped on the right side of the biplot indicating that the water chemistry at the narrows was similar to that of the stations in the north basin of the lake.

At a finer scale, the PCA grouped the 15 stations into three distinct geographic areas: north basin, south basin (central), south basin (south west). The north basin stations were generally characterized by higher concentrations of chloride, sodium, and higher conductivity. The north basin stations were also grouped along the first axis according to nutrient concentrations, algal biomass, and water clarity. These stations were characterized by higher total nitrogen concentrations, lower total phosphorus concentrations and correspondingly higher TN: TP ratios. Chlorophyll a concentrations were also lowest in the north basin, particularly at stations in the west arm of the north basin closest to the inflow of the Waterhen River (e.g., Manipogo, Spence Bay). North basin sites were also characterized by high water clarity particularly at sites located in the eastern arm of the north basin or Portage Bay (e.g., Davies Bay, Steep Rock) and lower concentrations dissolved nutrients (soluble reactive Si, DIN, TRP), alkalinity, and sulphate (Figure 4.15).

The south basin sites were characterized by high total phosphorus, lower N:P ratios, and high turbidity, particularly at sites located near the very south end of the lake (e.g., Delta, Lynch's Point; Figure 4.15). These sites also had low conductivity and low concentrations of dissolved sodium and dissolved chloride. Lynch's Point, located near the inflow of the Whitemud River, had the lowest conductivity, lowest concentrations of dissolved sodium and chloride. Of all 15 lake stations, total phosphorus and total reactive phosphorus was highest at Lynch's Point and had the lowest overall water clarity. Relative to the sites located at the southern area of the south basin, sites located the in the southeast area of the lake (e.g., St. Ambrose, Oak Point) and stations located at higher latitudes in the south basin (i.e., Lundar, Margaret Bruce) were not as high in total

phosphorus and total reactive phosphorus concentrations and consequently higher N:P ratios, higher water clarity, and greater algal biomass. These sites were also characterized by high concentrations of sulphate, with the greatest sulphate concentrations recorded at St. Ambroise, Oak Point, and Lundar in the south east corner and along the east side of the of the lake.

A principal components analysis was run on the 2006 data using the same 13 variables. In 2006, the first two axes were considered significant and explained approximately 72 percent of the total cumulative variance in the data (Figure 4.16). The first axis was correlated with nutrients (TP, TRP), algal biomass, water clarity, and alkalinity. Major ions (chloride, sodium), conductivity, total nitrogen, and the N:P ratio were all negatively correlated to the first axis. The second axis showed a relatively strong positive correlation with sulphate and conductivity and also had a moderate association with alkalinity.

Similar to 2005, the PCA separated the 15 Lake Manitoba stations according to lake basin along the first axis, with the north basin sites (including the narrows) on the right side of the ordination biplot and the south basin sites on the left side of the ordination biplot (Figure 4.16). The PCA biplot grouped the 15 stations into four distinct geographic areas in 2006: north basin (east arm), north basin (west arm), south basin (central), south basin (south). Stations located in the east arm of the north basin were characterized by greater conductivity, were more alkaline, and had higher concentrations of sulphate relative to the west arm. Stations located at the very south end of Lake Manitoba (e.g., Delta, St. Ambroise) were characterized by lower conductivity, sulphate, chloride, and sodium and higher concentrations of total phosphorus, total reactive phosphorus, and chlorophyll a

relative to stations located further northwards in the south basin.

4.14 Assiniboine River Diversion - Chemistry

Total nitrogen was slightly higher in Lake Manitoba relative to concentrations in the Diversion, although total phosphorus concentrations were 6 times greater in Diversion (Table 4.4). In 2005, both total nitrogen and phosphorus were greatest at the beginning of Diversion operation during peak flows on April 7th (500 m³/s) and decreased in association with declining flows (Figure 4.17). During the summer operation in 2005, nitrogen concentrations peaked at the beginning of operation in June (ca. 9.5 mg/L), again in early July, and subsequently declined to approximately 3 mg/L (Figure 4.17). There was also a notable rise in phosphorus at the beginning of summer operation, though unlike total nitrogen, phosphorus concentrations continued to rise throughout the summer operation period. Phosphorus concentrations were much greater during the spring (0.060 to 5.5 mg/L) in comparison to the summer (0.030 to 0.550 mg/L) diversion operation in 2005. Patterns in nitrogen and phosphorus concentrations were similar in 2006, with peak concentrations at the beginning of Diversion operation (Figure 4.18).

The Diversion was considerably more turbid than Lake Manitoba with total suspended solids concentrations nearly 4 times greater than suspended solids in the lake (Table 4.4). Temporal patterns in suspended solids was very similar to total nitrogen and phosphorus concentrations with peak concentrations during high flow events (Figure 4.17 and 18).

Lake Manitoba was more alkaline (pH = 8.56) than the Diversion (pH = 7.83), and both waterbodies were well buffered (Table 4.4). Sulphate dominated the ion chemistry in the Diversion channel and concentrations were slightly higher (186 mg/L) in comparison to

concentrations in Lake Manitoba (114 mg/L; Table 4.4). The average sodium and chloride concentrations in the Assiniboine River Diversion were 4 and 11 times lower as compared to concentrations in the lake, respectively.

The average conductivity in the Diversion for 2005 and 2006 indicated that the diversion was relatively dilute (618 $\mu\text{S}/\text{cm}$) in comparison to Lake Manitoba (1272 $\mu\text{S}/\text{cm}$; Table 4.4). Conductivity was lowest during peak flows in the spring of 2005 and 2006 and gradually increased over the period of operation in both years (Figure 4.17 and 4.18). Conductivity was greater during the summer Diversion operation in 2005, and was lowest in June (ca. 500 $\mu\text{S}/\text{cm}$) and increased through June and July (to 750 $\mu\text{S}/\text{cm}$; Figure 4.17).

4.15 Nutrient Loading to Lake Manitoba

The total nitrogen load to Lake Manitoba was greater in 2006 (10,660 t/yr) as compared to 2005 (9532 t/yr). The Waterhen River contributed to 48 % (4621 t/yr) and 77 % (7387 t/yr) of the total nitrogen load to Lake Manitoba in 2005 and 2006, respectively (Figure 4.19). The Assiniboine River was the second largest source of total nitrogen to Lake Manitoba, and contributed to 40 % (3767 t/yr) and 28 % (2709 t/yr) of the total load to the lake in 2005 and 2006, respectively. The Whitemud River was a comparatively small source of nitrogen to the lake and comprised less than 12 % in both years (Figure 4.18). In 2005, the Assiniboine River Diversion and the Whitemud River together contributed a greater load of nitrogen to Lake Manitoba than the Waterhen River.

In contrast, total phosphorus loading to Lake Manitoba was greatest in 2005 (780 t/yr) in comparison to 2006 (533 t/yr). The Assiniboine River Diversion transported the greatest phosphorus load to Lake Manitoba in 2005 (585 t/yr) and 2006 (358 t/yr) comprising 75 % and 67 % of the total phosphorus load, respectively (Figure 4.18). The

Whitemud River transported the second greatest phosphorus load to the lake in 2005 comprising 14 % (106 t/yr) of the total load. However, phosphorus loading from the Whitemud River was lowest in 2006 and comprised only 4% (21 t/yr) of the total load to Lake Manitoba. The Waterhen River contributed to 11 % (89 t/yr) and 29 % (154 t/yr) of the phosphorus load to the lake in 2005 and 2006, respectively and represented an important contribution to the lake phosphorus load in 2006 (Figure 4.18).

Similar to phosphorus, the total suspended solids load to Lake Manitoba was higher in 2005 (185,552 t/yr) relative to 2006 (170,308 t/yr). The Assiniboine River Diversion transported the majority of the total suspended solids load (74 %) to Lake Manitoba in 2005 (136,738 t/yr) and 2006 (126,445 t/yr; Figure 4.18). In 2005, the total suspended solids load from the Whitemud River (23,536 t/yr) and Waterhen River (25,278 t/yr) were similar and comprised 13 % and 14 % of the load, respectively. In contrast, the suspended solids load transported by the Waterhen River in 2006 was much more considerable (24 % or 41,335 t/yr) relative to the Whitemud River, which only comprised 1 % (2528 t/yr) of the suspended solids load to Lake Manitoba (Figure 4.18).

4.16 Discussion

In Lake Manitoba, nitrogen was not largely bioavailable with much occurring as organic nitrogen. Seasonal patterns in total nitrogen were not consistent between the two study years. In 2005, total nitrogen was highest in the spring and declined to the lowest concentrations at the end of the growing season. This seasonal pattern in total nitrogen is consistent with patterns observed for the large shallow lakes Taihu (China) and Okeechobee (USA; James *et al.* 2009). A study of the nitrogen dynamics in Lake Taihu attributed the seasonal decline and the subsequent low summer concentrations to high

rates of denitrification (McCarthy *et al.* 2007). However, total nitrogen concentrations in Lake Manitoba were greatest in late summer the following year. This indicates that rates of denitrification were quite low during the late summer of 2006 or the growth of nitrogen fixing cyanobacteria were responsible for elevated total nitrogen concentrations at the end of the growing season in the fall of 2006. In the late summer of 2006, chlorophyll a concentrations were highest, algae were nitrogen limited, and large blooms of cyanobacteria were observed in the south basin of the lake. Given this, it is possible that the late summer increases in total nitrogen concentrations in Lake Manitoba were related to the fixation of atmospheric nitrogen by cyanobacteria. Studies on nitrogen in Dutch lakes have indicated that nitrogen fixation may be an important source of nitrogen when phosphorus, light, and metals are sufficient for growth (van der Molen 1998). Nitrogen fixation may also help to explain the two large peaks in total nitrogen in the north basin of the lake in July and August of 2005.

Total nitrogen concentrations were significantly greater in the north basin of Lake Manitoba in both study years. This finding was particularly unexpected given the nutrient rich nature of the south basin. Elevated total nitrogen concentrations in the north basin may be related to the influence of the Waterhen River which was the single largest source of total nitrogen to Lake Manitoba in both study years. The average (2005 to 2006) total nitrogen concentration in the Waterhen River (3.89 mg/L) was nearly two times greater than that of the Whitemud River (1.94 mg/L) and was comparable to concentrations in the Assiniboine River Diversion (3.52 mg/L). Despite similar average total nitrogen concentrations in the Waterhen River and the Diversion, nitrogen loading from the Waterhen River was much greater relative to the Diversion because of the high flows on the

Waterhen River. For instance, the total flow from the Waterhen was approximately 3 times and 8 times greater than flow from the Diversion in 2005 and 2006, respectively. Higher total nitrogen in the north basin of Lake Manitoba may arise from the groundwater springs along the west side of the lake that contain elevated concentrations of ammonia and nitrogen (G. Goldsborough, pers. comm.). Wetlands surrounding the north basin of the lake may also be another source of organic nitrogen to the lake. Lowest total nitrogen concentrations were observed at the station nearest the inflow of the Whitemud River. Nitrogen concentrations at these two stations were similar and highlight the influence of the Whitemud River on lake chemistry at this particular station.

Total nitrogen concentrations were highly spatially variable in the north basin of Lake Manitoba and may be partly explained by the influence of the Waterhen River. Water from the Waterhen River ultimately flows from west to east and highest concentrations were evident at the two stations located just east of the inflow of Waterhen River in the centre arm of the north basin. Total nitrogen concentrations at these stations (ca. 5 mg/L) were slightly higher than the mean concentration in the Waterhen River. Total nitrogen concentrations also appeared to vary according to the 3 distinct arms of the north basin with greatest concentrations in the centre arm and lowest in the west arm. This spatial distinction highlights how lake morphometry may affect water circulation and mixing and water chemistry in the north basin of Lake Manitoba.

Nitrate concentrations in Lake Manitoba were nearly two times higher in 2005 as compared to 2006. Given that total nitrogen concentrations did not differ between study years, this finding was unexpected. Two different ion chromatographic columns were used for 2005 and 2006 and may be one reason for the large inter-annual differences observed.

Samples from a concurrent study were analyzed using both columns for comparison and found no clear correlation between the two datasets (Kolochnik 2008). It is unclear which year was incorrect, although nitrate concentrations are approximately an order of magnitude higher in 2005 than has been recorded historically in Lake Manitoba.

Seasonal nitrate concentrations were quite variable and patterns were difficult to recognize. Nitrate concentrations were slightly lower in the mid-summer months and may be related to periods of biological uptake by phytoplankton. The seasonal nitrate patterns in Lake Manitoba differ from other large shallow lakes. For example, in Lake Okeechobee, there was consistent spring peak in nitrate followed by a strong summer decline (James *et al.* 2009). Although nitrate concentrations in Lake Manitoba tend to be lowest in the summer, the seasonal pattern in nitrate did not vary as greatly through the growing season. This perhaps suggests that nitrate was not generally limiting to phytoplankton in Lake Manitoba. Studies of four large Swedish lakes indicated that nitrogen limitation generally occurred at dissolved inorganic nitrogen concentrations of less than 0.03 mg/L (Wilander and Persson 2001), although others have suggested a higher threshold of 0.1 mg/L (Horne and Goldman 1994). Most dissolved inorganic nitrogen concentrations in Lake Manitoba were well above these thresholds and the DIN:TRP ratios indicated that most areas in the lake were nitrogen sufficient.

Nitrate was spatially variable in the south basin with greatest concentrations at the very southern end of the lake and also along the eastern shoreline. Nitrate dynamics in lakes are regulated by nitrification and denitrification processes and may also be related to point sources and diffuse sources of nitrate. Plausible sources of nitrate in the south basin of Lake Manitoba include the Whitemud River, the Assiniboine River Diversion, adjacent

coastal marshes (e.g., Delta Marsh), and overland runoff .

Ammonia concentrations were significantly higher in the south basin of Lake Manitoba, particularly near the inflow of the Whitemud River. The river appeared to affect the spatial variation in ammonia in the south basin of the lake given the higher ammonia concentrations detected at the station nearest the inflow which were comparable to ammonia concentrations in the Whitemud River (0.026 mg/L). Ammonia was lowest at many stations in the north basin, particularly in the eastern arm of the north basin.

Total phosphorus concentrations were higher in 2005 relative to 2006 although differences were not significant. Phosphorus loading to Lake Manitoba was approximately 30 % higher in 2005 because of wet conditions and may help to explain the slightly higher phosphorus concentrations observed in Lake Manitoba in 2005. There were clear basin differences in the seasonal total phosphorus dynamics in the north and south basins of Lake Manitoba. In the north basin, seasonal total phosphorus dynamics in both study years were similar to that of shallow mesotrophic Danish lakes with greater concentrations in the spring and fall and lowest concentrations during the summer (Jeppesen *et al.* 1997). Seasonal total phosphorus patterns in the north basin of Lake Manitoba were also similar to patterns observed in the isolated eastern and northwestern areas of Lake of the Woods (Pla *et al.* 2005). Seasonal declines in lake phosphorus concentrations is characteristic of less productive Precambrian shield lakes where seasonal phosphorus dynamics are regulated by phytoplankton settling out of the epilimnion in the summer (Scheffer 1998).

In the south basin of Lake Manitoba, seasonal phosphorus dynamics resembled patterns typical of shallow eutrophic lakes (in 2005) with phosphorus increasing from spring through the summer (Riley and Prepas 1985, Jeppesen *et al.* 1997, Scheffer 1998, Haldna

et al. 2008, James *et al.* 2009). Seasonal increases in total phosphorus concentrations in shallow lakes are mostly attributed to the internal phosphorus load from sediment resuspension and mineralization processes (Jeppesen *et al.* 1997, Søndergaard *et al.* 2003). While it is possible that the seasonal increase in phosphorus concentrations in the south basin of Lake Manitoba in 2005 may be partly explained by internal nutrient loading, seasonal increases in total phosphorus were also related to the influence of phosphorus loads from the Assiniboine River Diversion. In 2005, the Assiniboine River Diversion contributed 75 % of the phosphorus load to Lake Manitoba and large increases in total phosphorus in the south basin of Lake Manitoba coincided with the summer operation of the Assiniboine River Diversion. Total phosphorus concentrations at monitoring stations nearest the inflows of the Assiniboine River Diversion and the Whitemud Rivers were generally an order of magnitude higher than most other stations in the south basin in July. However, phosphorus concentrations continued to increase in the south basin nearly a month after the end of diversion operation perhaps indicating that the late summer increases in phosphorus were a result of internal loading. In 2006, total phosphorus exhibited an opposite seasonal pattern in the south basin with peak concentrations in the spring and an overall decline through the summer. The relatively dry climate and lower external phosphorus loading in 2006 may help to explain the interannual differences in the seasonal total phosphorus concentrations in Lake Manitoba.

Spatial differences in total phosphorus concentrations were apparent within and between basins in Lake Manitoba. The south basin was much more enriched in phosphorus, particularly at the very south end of the lake and may reflect the large external phosphorus load that is transported to the south basin of Lake Manitoba. For example, in

2005, nearly 90 % of the external phosphorus load was transported into the south basin of the lake via the Assiniboine River Diversion and the Whitemud River. The higher population density and agriculture surrounding the extreme south end of the south basin are also likely sources of phosphorus to the south basin. However, it is unknown to what extent agriculture practices (land clearing, wetland drainage, and fertilizer application) and human impacts (e.g., contribution from malfunctioning septic fields and removal of shoreline vegetation) contribute to the nutrient budget of Lake Manitoba.

Lower phosphorus concentrations in the north basin may be attributed to the comparatively smaller total phosphorus loads from the Waterhen River, lower population density (e.g., less human impact) and different agricultural practices in the north basin. Spatial water quality differences may also be affected by the distinct lake morphometry and hydrology in the north and south basins of the lake. The south basin of the lake acts as an evaporative basin with limited water inflows and large evaporative losses resulting in long water residence times in the south basin of the lake (20 years). Because of the relatively long water residence time in the south basin nutrients are potentially retained to a greater degree. However, nutrients entering the north basin of Lake Manitoba may be flushed through the north basin more rapidly because of the large water volume and short water residence time. Morphometry may also help to explain the spatial differences between the north and south basins given the larger more open and wind exposed south basin.

Phosphorus is considered the primary nutrient that regulates algal growth in lakes (Schindler 2008) and is positively correlated to chlorophyll a (Dillon and Rigler 1974). There was an apparent de-coupling of the phosphorus chlorophyll a relationship in Lake Manitoba. Only very weak correlations existed between phosphorus and chlorophyll a in the

south basin, and no significant correlations in the north basin of the lake. This highlights the importance of other factors regulating algal biomass in Lake Manitoba. In other large lakes, the relationship between phosphorus and chlorophyll is weak or nonexistent in certain seasons and has been attributed to the effects of light and temperature (e.g., Pla *et al.* 2005). Similarly, Canfield and Hoyer (1988) found no significant relationship between phosphorus and chlorophyll a in the large shallow polymictic Lake Okeechobee.

Soluble reactive silica concentrations were lowest in spring and generally increased through the open water season in both basins of Lake Manitoba. This seasonal pattern is typical of other large shallow lakes of central Canada (Hecky *et al.* 1986). Silica flux is largely regulated by diatom uptake and sedimentation processes, and regeneration from sediments (Parker *et al.* 1977). Diatoms require silica for the synthesis of their frustules and concentrations are typically depleted in the spring in association with the spring dominance of diatoms and associated uptake processes. The spring decline in silica concentrations may also be associated with diatom sedimentation from the water column. In Lake Manitoba, increases in silica through the summer and early fall may be partially explained by regeneration from sediments. Hecky *et al.* (1986) found that late summer increases in silica concentrations in Southern Indian Lake and the north basin of Lake Winnipeg were associated with diatom dissolution and silica regeneration from the sediments. In the south basin of Lake Manitoba, silica began to decline in the late summer of 2005 which may be related to a shift in phytoplankton species composition and increases in fall diatom biomass. Diatom growth is generally limited by silica at concentrations of 0.5 mg/L (Wetzel 1983). Diatom biomass may have been limited by silica in the north basin during the spring of 2005 when silica concentrations were slightly above or below 0.5 mg/L at several

stations.

Silica concentrations were twice as high in the south basin of Lake Manitoba and may be related to the underlying geology and the silica rich inflows of the south basin. Sediment collected from the nearshore sites in the south basin were predominately comprised of sand and may help to explain these large basin differences in silica (Appendix 3). In addition, the two sites in the north basin with the lowest silica concentrations had the lowest fractions of sand in sediment samples collected from these sites. A major source of silica in lakes originates from the breakdown of aluminosilicate minerals (Wetzel 1983) which includes feldspars and quartz. It is possible that differences in the underlying geology between the north and south basins of Lake Manitoba may help to explain the distinct geospatial basin wide differences in silica. Both the mean and the range in silica concentrations were similar at Whitemud River (mean = 4.36 mg/L, range = 1.12 to 8.20 mg/L) and the Lynch's Point (mean = 4.02 mg/L, range = 1.10 to 7.31 mg/L) sites suggesting that the Whitemud River may be an important riverine source of silica to the south basin of Lake Manitoba.

Seasonal patterns in chlorophyll a in Lake Manitoba were generally lowest in the late spring or early summer and increased through the summer. Eutrophic temperate lakes typically exhibit a consistent chlorophyll phenology consisting of a spring bloom followed by a decline in the early summer and a subsequent fall bloom (Marshall and Peters 1989, Talling 1993). The timing of the spring bloom varies from year to year and is negatively correlated to trophic state and mean annual air temperature (Marshall and Peters 1989). In Lake Manitoba, the spring bloom was not apparent in 2005 but appeared in May of the following year. Because monitoring did not begin until mid May when the lake was

completely ice free, it is possible that the spring bloom was not captured in 2005 depending on the timing of the spring bloom. In the south basin, the summer decline in chlorophyll a in 2006 appeared in early June while in 2005 the chlorophyll minimum was observed several weeks earlier. According to Talling (1993), early summer declines in chlorophyll a are often associated with diatom sedimentation, species replacement and zooplankton grazing.

Chlorophyll a concentrations were two to three times greater in the south basin than those in the north basin of Lake Manitoba and may be partly explained by the differences in nutrient availability between basins. Nutrients were generally much higher and more available for growth in the south basin in comparison to the north basin. Dissolved nutrient fractions remained low or below analytical detection limits in the north basin and may be one of the factors responsible for constraining algal growth in the north basin. Dissolved nutrient ratios also indicated that the north basin was much more phosphorus limited at most sites. Basin differences in chlorophyll cannot be reasonably explained by light limitation. Chlorophyll was much higher in the south basin despite much lower water clarity, while chlorophyll was lowest in the north basin where water was much clearer.

Basin differences in chlorophyll a may also be partly explained by the differences in the lake morphometry. The south basin consists of one large open basin that is presumably affected by wind to a greater extent than the smaller more isolated bays of the north basin. Higher chlorophyll a in the south basin may be partly explained by the direct wind induced resuspension of chlorophyll a or from the secondary effects effect of resuspended nutrients.

Carrick *et al.* (1993) found that increases in algal biomass in the large shallow Lake Apopka were related to the vertical resuspension of meroplankton during high wind events. Phytoplankton and diatom biomass doubled during these high wind events in Lake Apopka.

Similarly, Schallenberg and Burns (2004) examined the effects of resuspension on primary production and found that entrainment of meroplankton may contribute significantly to the overall primary production of shallow lakes. Other studies of large shallow lakes have found that increases in algal biomass were a result of the uptake of the newly resuspended nutrients that became available for uptake (Canfield and Hoyer 1988).

The highest concentrations of chlorophyll in the south and south east areas of Lake Manitoba may also be partly explained by the prevailing north-westerly wind direction and the horizontal dispersion of phytoplankton. During north-westerly wind events surface blooms of buoyant phytoplankton such as cyanobacteria or phytoplankton with low sinking rates may accumulate at the downwind end of the lake. Webster (1990) developed an empirical model to characterize the effect of wind on the horizontal distribution of phytoplankton in lakes and found that buoyant phytoplankton with low sinking velocities would aggregate at the downwind end of a polymictic lake. Wind speed, water depth, and phytoplankton rising and sinking velocities were used to model the rate of chlorophyll increase at the downwind end of the lake. This geospatial pattern of chlorophyll accumulation at the downwind end of lakes has been observed in other north temperate lakes (George and Edwards 1976, Stauffer 1982).

Higher dissolved nutrient ratios indicated that the north basin of Lake Manitoba was more phosphorus limited than the south basin. These high ratios are a result of the relatively low concentrations of total reactive phosphorus (mean = 0.011 mg/L) in the north basin of Lake Manitoba. Phosphorus limitation was somewhat alleviated in the south basin because of the high phosphorus concentrations transported by the Whitemud River and the Assiniboine River Diversion. The two sites located at the very northern part of the south

basin were more phosphorus limited because of declining phosphorus concentrations moving northwards. The southwestern area of the south basin was nitrogen limited and largely influenced by the nutrient regime of the Whitemud River (e.g., high phosphorus and low nitrogen concentrations). Elemental nutrient ratios are useful tools for a preliminary assessment of phytoplankton nutrient limitation. Future studies should consider using *in situ* or laboratory enrichment bioassays and other physiological indicators (nitrogen debt assays, alkaline phosphatase activity) to provide a more comprehensive assessment of the factors affecting phytoplankton growth in Lake Manitoba.

Water clarity was significantly greater in the north basin of Lake Manitoba while the south basin was considerably more turbid. Organic suspended solids comprised half of the total suspended solids in the south basin while the organic fraction contributed 75 % of the total suspended solids in the north basin. This suggests that phytoplankton was largely responsible for explaining turbidity in the north basin while both phytoplankton and sediment were responsible for the turbid nature of the south basin.

Spatial differences in water clarity in Lake Manitoba may be partly explained by the influence of tributary rivers. High sediment concentrations are transported by the Assiniboine River Diversion and the Whitemud River to the south basin of Lake Manitoba while sediment concentrations are relatively low in the Waterhen River. Basin differences in water clarity may also be related to the potential for resuspension in each basin. Philips *et al.* (1993) noted that shallow polymictic lakes may be subject to resuspension of sediments from wind action and contain a large amount of non-algal suspended sediments. Positive correlations between maximum wind speed and water clarity measures indicated that wind induced sediment resuspension was significant in the north and south basins of Lake

Manitoba. However, a stronger correlation existed between wind speed and suspended sediment in the south basin. The large open south basin has a large fetch distance and may be more readily affected by wind induced sediment resuspension. Although wind induced resuspension contributed to turbidity in north basin, it is possible that resuspension is dampened by the smaller fetch distance across the more isolated bays in the north basin.

Turbidity was significantly greater in the south basin of the lake in 2005 and may be related to the higher chlorophyll and greater sediment loads transported to the lake in that year. Based on measurements of turbidity, the north basin was significantly more turbid in 2005 despite no inter-annual differences in water transparency and total, inorganic, or organic suspended solids. It is unknown why turbidity showed differences between years in the north basin while other measures of water clarity did not. Although higher precipitation was recorded in 2005, greater turbidity in north basin was somewhat unexpected given that a greater load of suspended solids was transported by the Waterhen River in 2006 when river flows were higher.

Water in the south basin was more dilute than the north basin of Lake Manitoba with significantly lower concentrations of both sodium and chloride. This particular finding was unexpected given that samples collected from Lake Manitoba in the 1960s indicated that the north basin was generally more dilute (Hughes and Williamson 2002, Last 1980, Crowe 1972, Cober 1968). The apparent reversal in spatial patterns of conductivity, sodium, and chloride observed in the present study may be partly explained by influence of the Assiniboine River Diversion. Since 1970, the Diversion has been used to mitigate downstream flooding by diverting Assiniboine River water into the south basin of Lake Manitoba. Comparison of water quality data collected from the Diversion and Lake

Manitoba in 2005 and 2006 indicated that lake water had significantly higher chloride, sodium, and conductivity while water from the Diversion was quite dilute in comparison. It is clear that the south basin of Lake Manitoba is diluted by the inflow of Assiniboine River water and may explain the spatial differences between the lake surveys of the 1960s and the present study.

Major ions were lower in the south basin during 2005 owing to the wet conditions and high inflows from the much more dilute waters of the Whitemud River and the Assiniboine River Diversion. The influence of the Diversion on the south basin was evident in the summer of 2005 with lower total dissolved solids and chloride through the summer. In the north basin, ions were generally lowest in the spring from the freshet and increased over the summer. There was no clear seasonal pattern in major ions concentrations for the south basin and may be attributed to the confounding effects of the Diversion.

Sulphate concentrations were greatest along the east side of the south basin. Differences in major ion chemistry in Lake Manitoba have been attributed to the highly concentrated saline groundwater rich in sodium, chloride, and sulphate arising from gypsum deposits (van Everdingen 1971; IGDSB 1976; Last 1980). Studies in the 1970s indicated that 70 to 80 percent of total annual sodium, chloride, and sulphate loads transported to the south basin originated from saline groundwater sources despite relatively small flow contributions (IGDSB 1976). However, these saline springs have only been documented in the south western area of the lake. Given this, it is not clear why sulphate concentrations are higher along the east side of the south basin. There was an apparent distinction in sulphate and total dissolved solids concentrations between the eastern arm and the western arm of the north basin. The sites along the west arm were more dilute relative to the

east arm and may be related to the inflow of the dilute water from the Waterhen River.

Highest concentrations of nitrogen, phosphorus, and suspended sediment occurred during the initial spring peak flow of the Diversion in 2005 and 2006. However, nutrient concentrations quickly declined in association with flow. This suggests that elevated nutrient and sediment concentrations in the Diversion were a consequence of the high flows in the Diversion channel. Conductivity was generally lowest at the beginning of operation in spring and gradually increased with declining flows. Spring conductivity was likely diluted by the initial high flows and the concurrent spring snowmelt.

The Diversion was the primary source of sediment and phosphorus to Lake Manitoba in both study years because of high flows and elevated concentrations of phosphorus and sediment, particularly when the Diversion started to flow. The Waterhen River was the largest source of total nitrogen to Lake Manitoba. However, in 2005, nitrogen contributions from the Waterhen River were only slightly higher than what was contributed by the Diversion. In 2006, the Waterhen River transported much higher nitrogen and phosphorus loads because of higher flows that year. The Whitemud River was the generally smallest source of nitrogen and phosphorus to the lake, although it contributed slightly more phosphorus than the Waterhen River in 2005 because of higher flow conditions.

Table 4.1. Descriptive statistics for water quality variables measured from May to September in 2005 and 2006 in Lake Manitoba.

Variable	Units	Average	Min	Max	SE	n
Total Nitrogen	mg·L ⁻¹	4.05	<0.025	20.7	0.160	250
Nitrate-Nitrite Nitrogen	mg·L ⁻¹	0.141	<0.01	1.23	0.006	250
Dissolved Ammonia	mg·L ⁻¹	0.013	<0.01	0.150	0.001	250
Total Phosphorus	mg·L ⁻¹	0.079	<0.01	0.453	0.005	250
Total Reactive Phosphorus	mg·L ⁻¹	0.018	<0.01	0.287	0.002	250
Soluble Reactive Silica	mg·L ⁻¹	3.05	0.46	7.97	0.09	249
Chlorophyll a	µg·L ⁻¹	25.2	2.84	326	1.77	250
DIN:TRP	molar ratio	37	0.7	261	2.0	250
Microcystin-LR	µg·L ⁻¹	0.15	<0.1	1.41	0.01	165
Water Transparency (450 nm)	%	86	22	97	0.6	249
Turbidity	NTU	19	1.4	150	1.1	249
Total Suspended Solids	mg·L ⁻¹	35	<1	380	2.9	249
Inorganic Suspended Solids	mg·L ⁻¹	16	<1	292	2.1	249
Organic Suspended Solids	mg·L ⁻¹	19	<1	120	1.0	249
Total Dissolved Solids	mg·L ⁻¹	752	110	2705	12.2	249
Specific Conductivity	µS·cm ⁻¹	1272	12.35	1702	14.97	249
Dissolved Chloride	mg·L ⁻¹	226	5.69	324	4.57	250
Sodium	mg·L ⁻¹	128	0.664	232	3.13	250
Sulphate	mg·L ⁻¹	114	14.0	178	2.00	250
Lab pH	pH units	8.56	7.86	9.06	0.02	249
Alkalinity (as CaCO ₃)	mg·L ⁻¹	203	124	290	2.03	249
Dissolved Inorganic Carbon	mg·L ⁻¹	48	30	70	0.5	249

Table 4.2. Comparison of average open-water (May to October) conditions in 2005 and 2006 for Lake Manitoba.

Variable	Units	2005		2006	
		North Basin	South Basin	North Basin	South Basin
Total Nitrogen	mg·L ⁻¹	5.13	3.84	4.14	3.24
Nitrate-Nitrite Nitrogen	mg·L ⁻¹	0.170	0.196	0.091	0.096
Dissolved Ammonia	mg·L ⁻¹	<0.010	0.021	<0.010	0.014
Total Phosphorus	mg·L ⁻¹	0.058	0.110	0.051	0.089
Total Reactive Phosphorus	mg·L ⁻¹	0.011	0.032	0.011	0.015
Soluble Reactive Silica	mg·L ⁻¹	2.19	4.41	2.48	2.87
Chlorophyll a	µg·L ⁻¹	11	34	9	22
DIN:TRP	molar ratio	56	43	28	22
Water Transparency (450 nm)	%	93	79	90	85
Turbidity	NTU	14	29	8	25
Total Suspended Solids	mg·L ⁻¹	14	65	15	39
Inorganic Suspended Solids	mg·L ⁻¹	2.4	37	3.9	18
Organic Suspended Solids	mg·L ⁻¹	11	29	11	21
Total Dissolved Solids	mg·L ⁻¹	779	697	755	786
Specific Conductivity	µS·cm ⁻¹	1475	1259	1200	1163
Dissolved Chloride	mg·L ⁻¹	244	190	268	211
Sodium	mg·L ⁻¹	123	92	179	130
Sulphate	mg·L ⁻¹	92	129	94	138
Lab pH	pH units	8.55	8.63	8.49	8.56
Alkalinity (as CaCO ₃)	mg·L ⁻¹	181	234	175	216
Dissolved Inorganic Carbon	mg·L ⁻¹	43	55	42	51
Water Temperature	°C	19.2	19.5	20.5	20.7

Table 4.3. Comparison of average open-water (May to October) conditions in the north basin and south basin of Lake Manitoba.

Variable	Units	North Basin		South Basin	
		2005	2006	2005	2006
Total Nitrogen	mg·L ⁻¹	5.13	4.14	3.84	3.24
Nitrate-Nitrite Nitrogen	mg·L ⁻¹	0.170	0.091	0.196	0.096
Dissolved Ammonia	mg·L ⁻¹	<0.010	<0.010	0.021	0.014
Total Phosphorus	mg·L ⁻¹	0.058	0.051	0.110	0.089
Total Reactive Phosphorus	mg·L ⁻¹	0.011	0.011	0.032	0.015
Soluble Reactive Silica	mg·L ⁻¹	2.19	2.48	4.41	2.87
Chlorophyll a	µg·L ⁻¹	11	9	34	22
DIN:TRP	molar ratio	56	28	43	22
Water Transparency (450 nm)	%	93	90	79	85
Turbidity	NTU	14	8	29	25
Total Suspended Solids	mg·L ⁻¹	14	15	65	39
Inorganic Suspended Solids	mg·L ⁻¹	2.4	3.9	37	18
Organic Suspended Solids	mg·L ⁻¹	11	11	29	21
Total Dissolved Solids	mg·L ⁻¹	779	755	697	786
Specific Conductivity	µS·cm ⁻¹	1475	1200	1259	1163
Dissolved Chloride	mg·L ⁻¹	244	268	190	211
Sodium	mg·L ⁻¹	123	179	92	130
Sulphate	mg·L ⁻¹	92	94	129	138
Lab pH	pH units	8.55	8.49	8.63	8.56
Alkalinity (as CaCO ₃)	mg·L ⁻¹	181	175	234	216
Dissolved Inorganic Carbon	mg·L ⁻¹	43	42	55	51
Water Temperature	°C	19.2	20.5	19.5	20.7

Table 4.4. Comparison of water quality at the mouth of the Assiniboine River Diversion (n = 119) and Lake Manitoba (n = 149 to 150) for 2005 and 2006 (pooled). Samples were collected on a daily basis from the Diversion during operation and on a biweekly basis from 15 stations from May to September in 2005 and 2006.

Variable	Units	Diversion		Lake Manitoba	
		Mean	Range	Mean	Range
Total Nitrogen	mg·L ⁻¹	3.52	0.600 - 11.5	4.05	<0.025 - 20.7
Total Phosphorus	mg·L ⁻¹	0.520	0.015 - 5.35	0.079	<0.01 - 0.453
Total Suspended Solids	mg·L ⁻¹	137	<1 - 1362	35	<1 - 380
Specific Conductivity	µS·cm ⁻¹	618	344 - 921	1272	12 - 1702
Dissolved Chloride	mg·L ⁻¹	19	6 - 30	226	6 - 324
Sodium	mg·L ⁻¹	24	12 - 31	128	<1 - 232
Sulphate	mg·L ⁻¹	183	59 - 329	114	14 - 178
pH	pH units	7.83	7.28 - 8.72	8.56	7.86 - 9.06
Alkalinity	mg·L ⁻¹	211	120 - 304	203	124 - 290

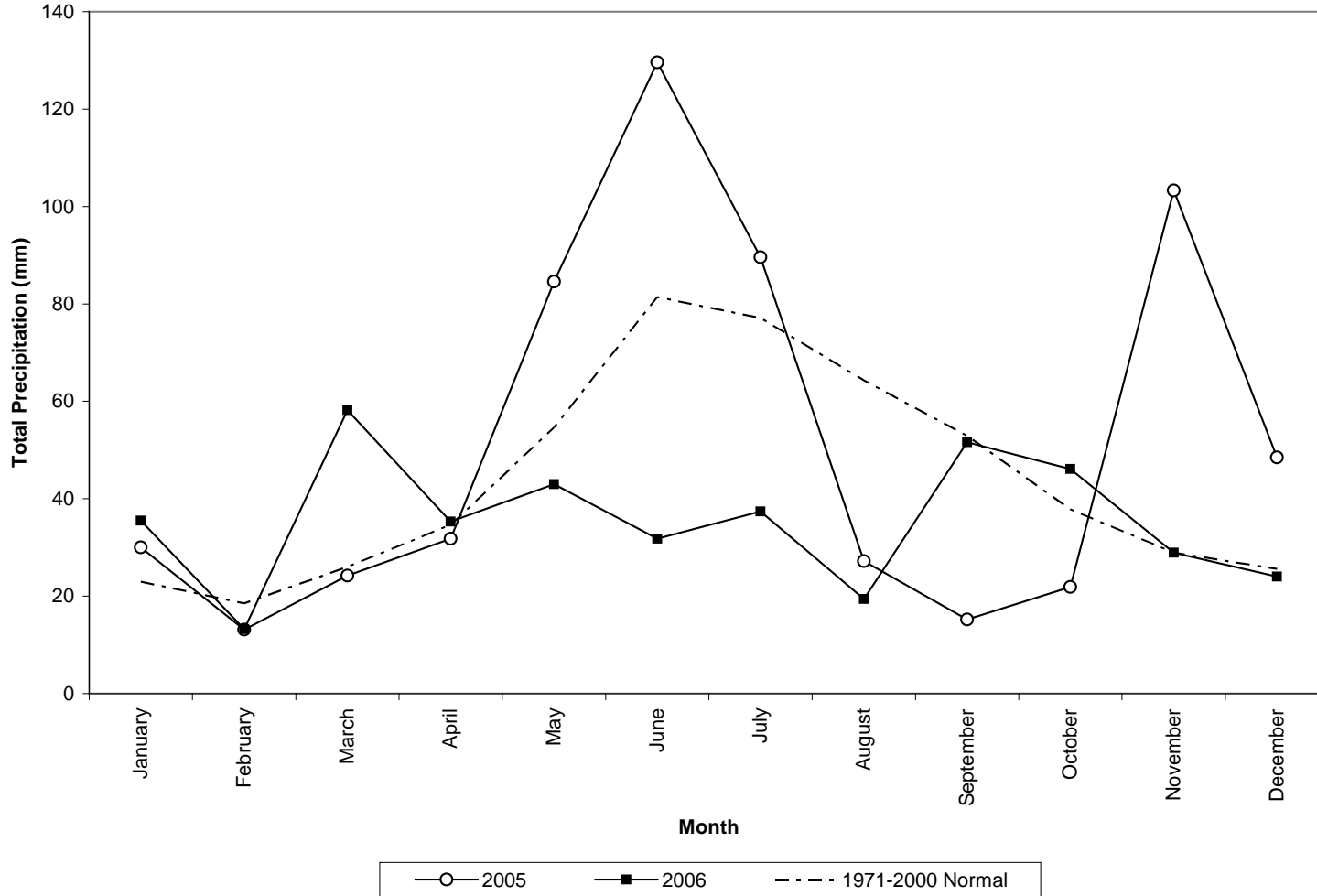


Figure 4.1. Total monthly precipitation (2005-2006) and precipitation normals (1971-2000) recorded at the Delta Marsh, MB meteorological station. (Data Source: Environment Canada).

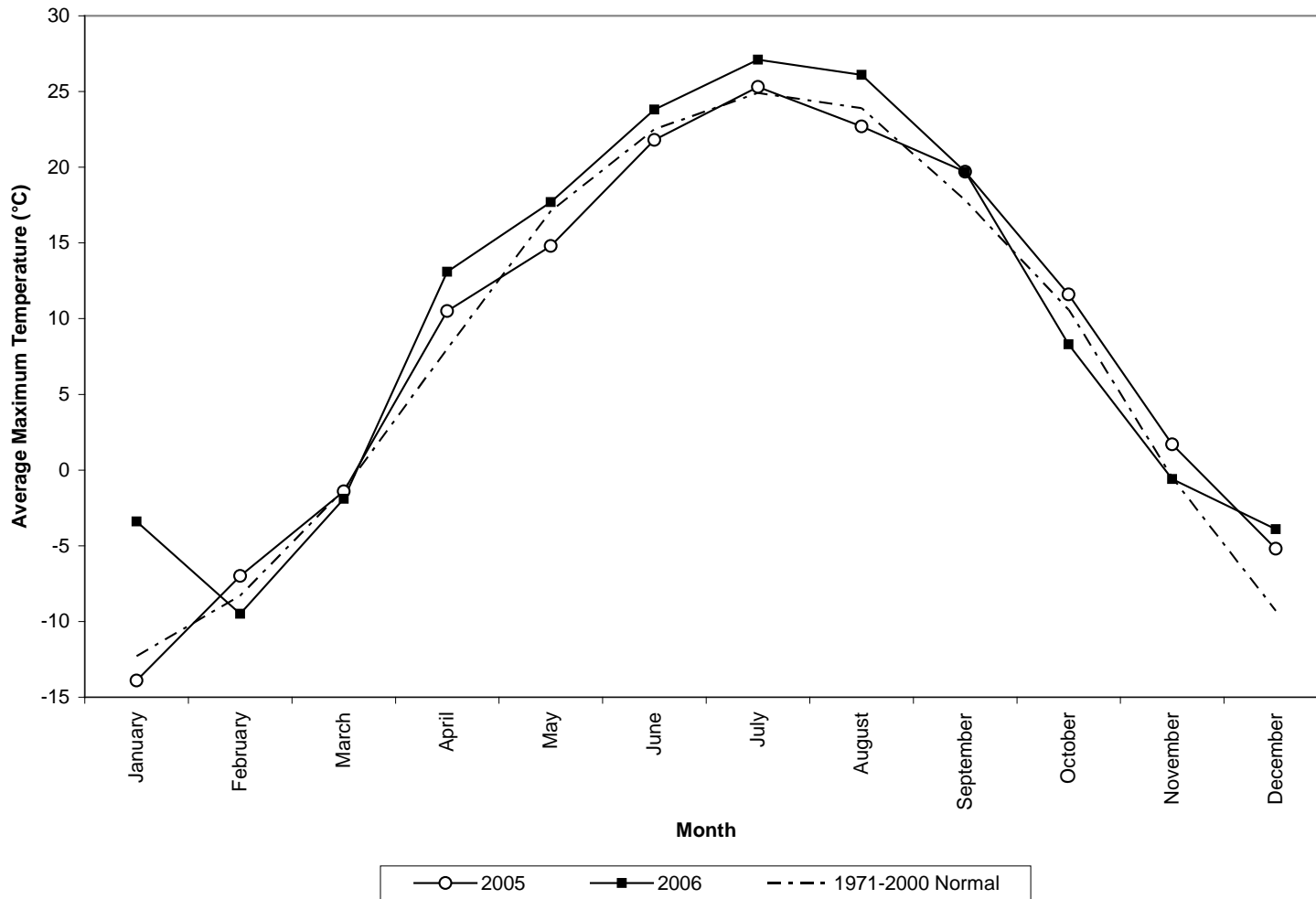


Figure 4.2. Average maximum temperature (2005-2006) and temperature normals (1971-2000) recorded at the Environment Canada meteorological station located at Delta Marsh. (Data Source: Environment Canada).

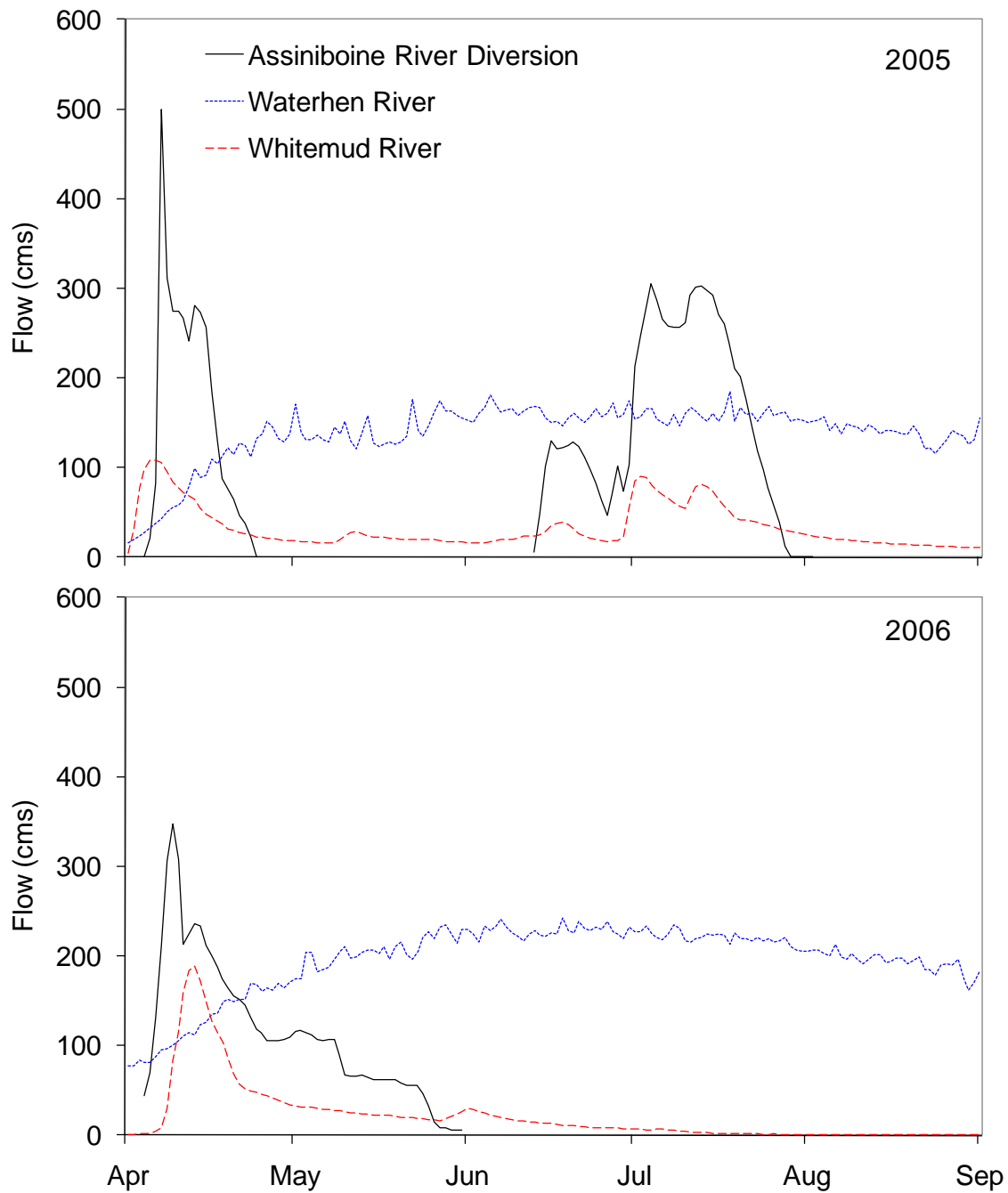


Figure 4.3. Daily flows (cms) for the Waterhen River, Whitemud River, and the Assiniboine River Diversion in 2005 and 2006. (Data Source: Water Survey of Canada).

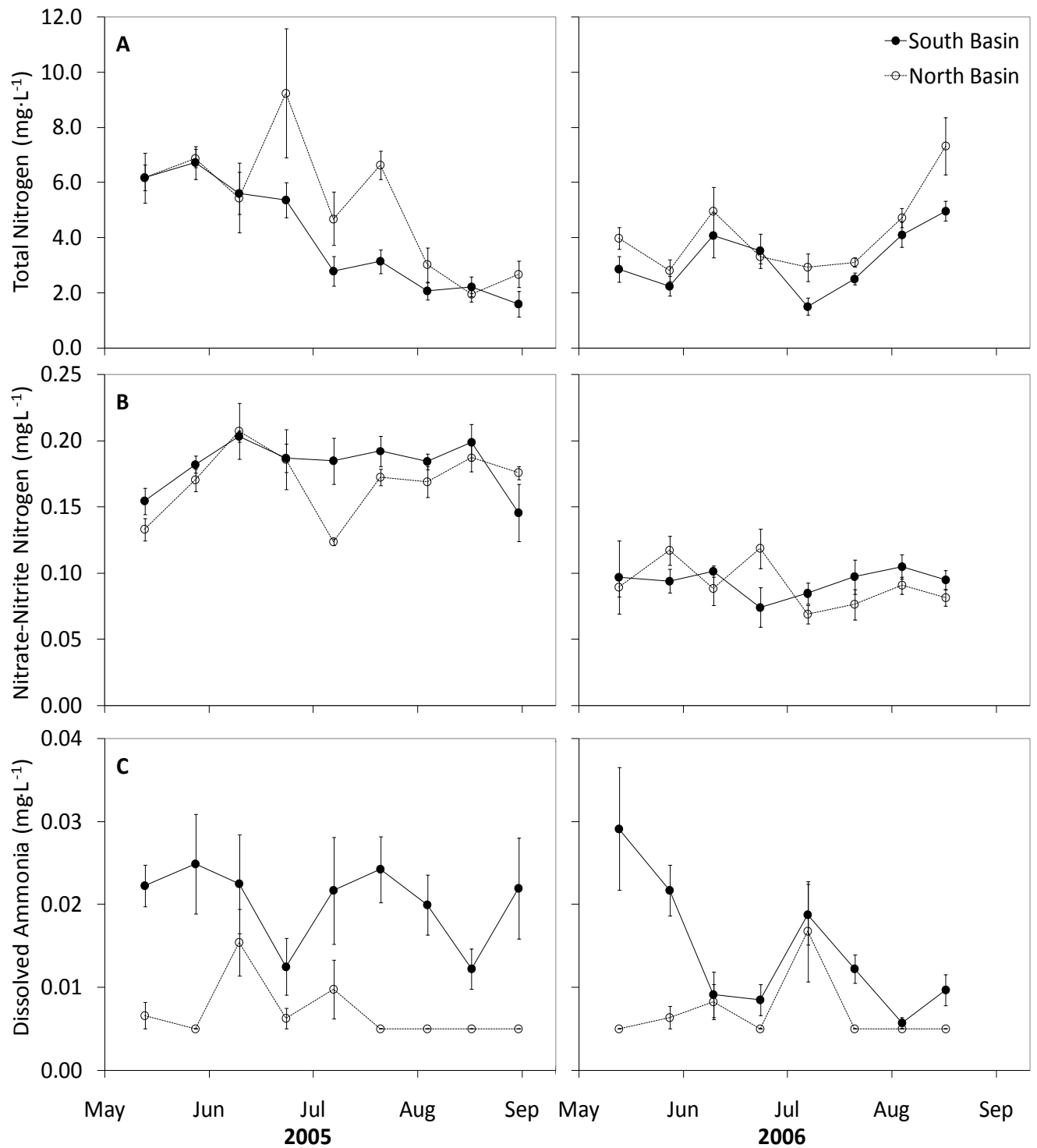


Figure 4.4. Total nitrogen (A), nitrate-nitrite nitrogen (B), and dissolved ammonia (C) concentrations (mg/L) in the north and south basins of Lake Manitoba from May to September in 2005 and 2006. Points represent the biweekly mean of stations located in each respective basin of the lake (\pm SE; $n=6$ to 8).

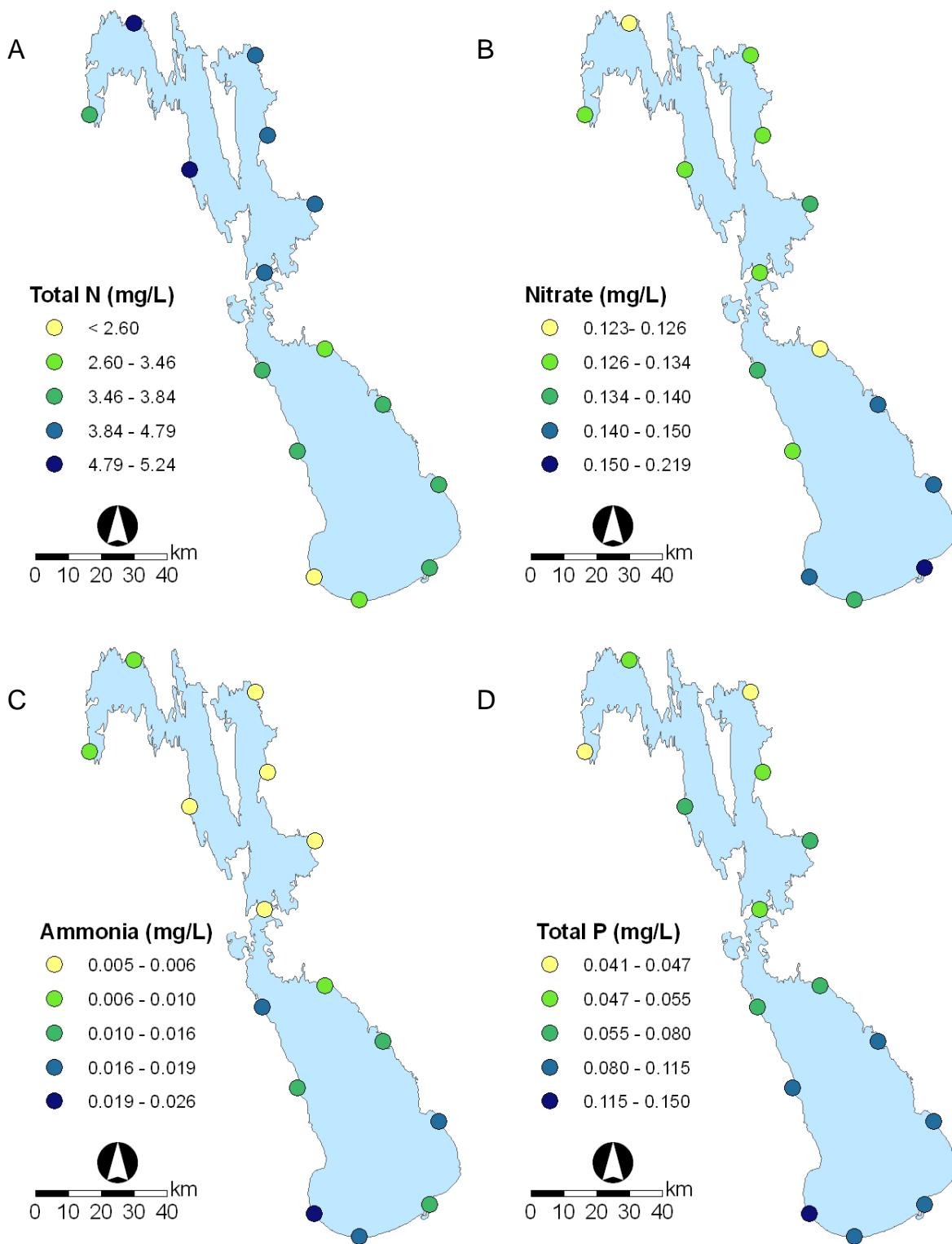


Figure 4.5. Spatial patterns in total nitrogen (A), nitrate-nitrite nitrogen (B), dissolved ammonia (C), and total phosphorus (D) concentrations (mg/L) in Lake Manitoba. Values represent the open-water (May to September) average of each station for 2005 and 2006 (n=17).

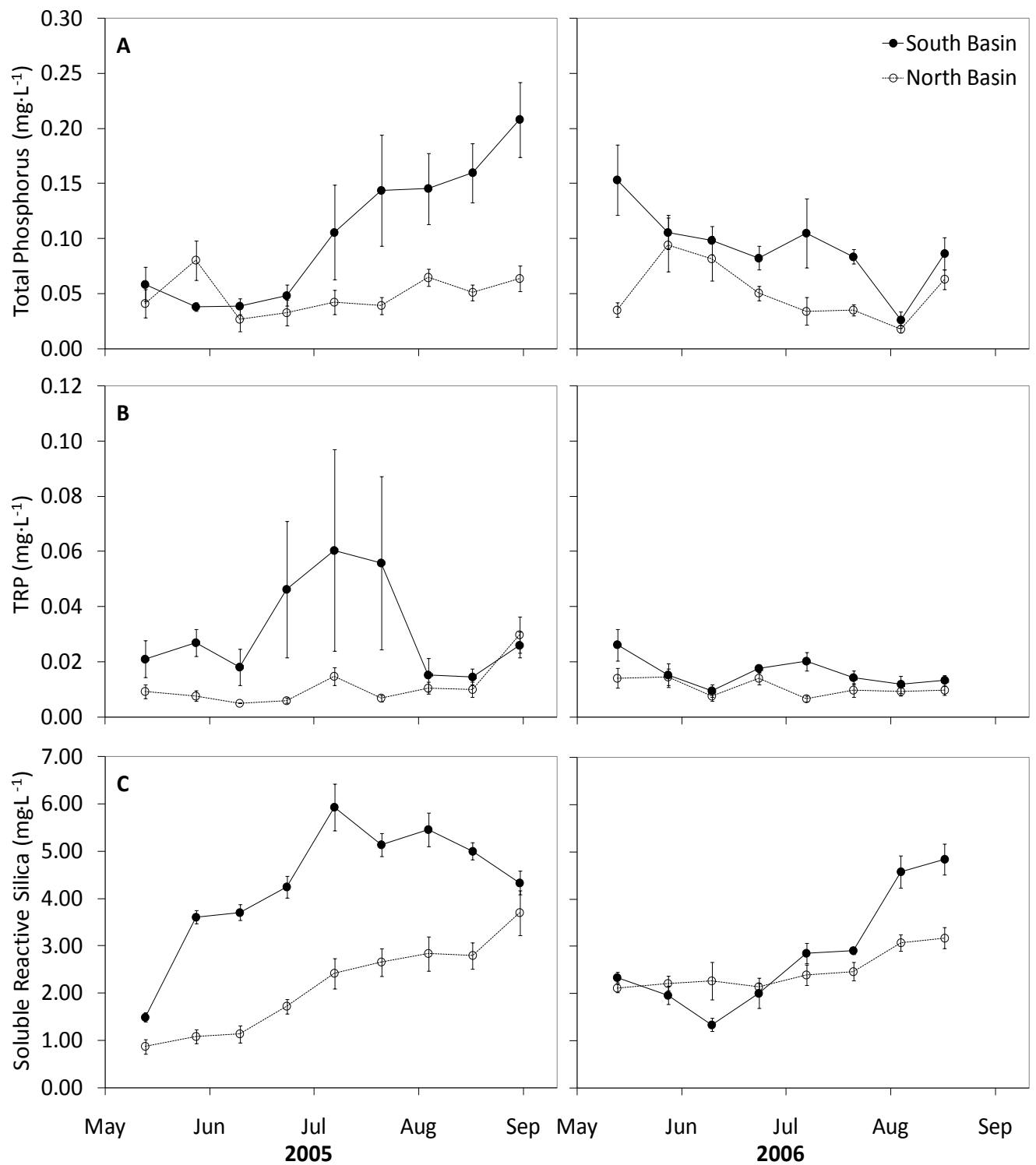


Figure 4.6. Total phosphorus (A), total reactive phosphorus (B), and soluble reactive silica (C) concentrations (mg/L) in the north and south basins of Lake Manitoba from May to September in 2005 and 2006. Points represent the biweekly mean of stations located in each respective basin of the lake (\pm SE; $n=6$ to 8).

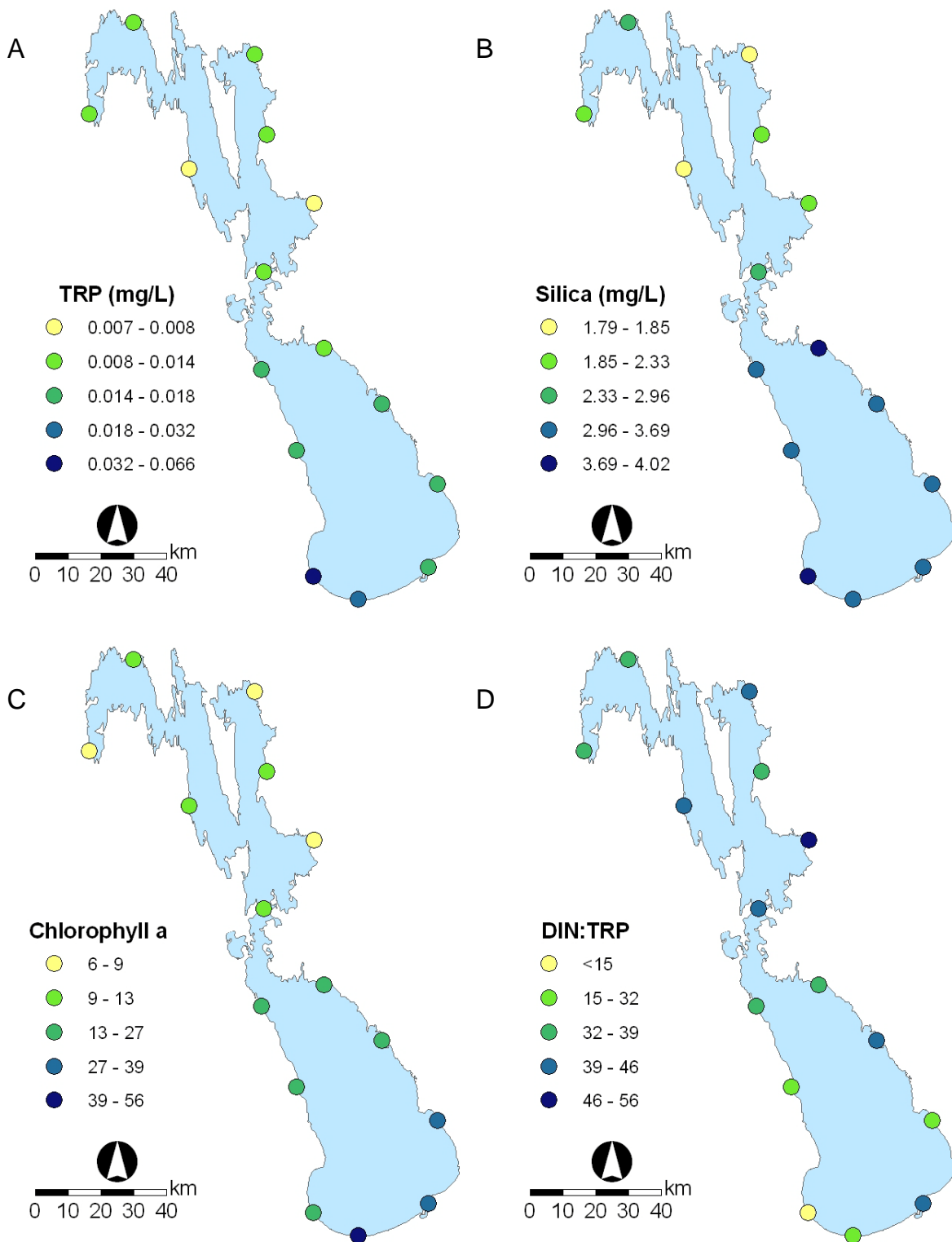


Figure 4.7. Spatial patterns in total reactive phosphorus (TRP; A; mg/L), soluble reactive silica (B; mg/L), chlorophyll a (C; $\mu\text{g/L}$) concentrations, and the ratio of dissolved inorganic nitrogen (DIN) to total reactive phosphorus (TRP; D) in Lake Manitoba. Values represent the open-water (May to September) average of each station in 2005 and 2006 (n=17).

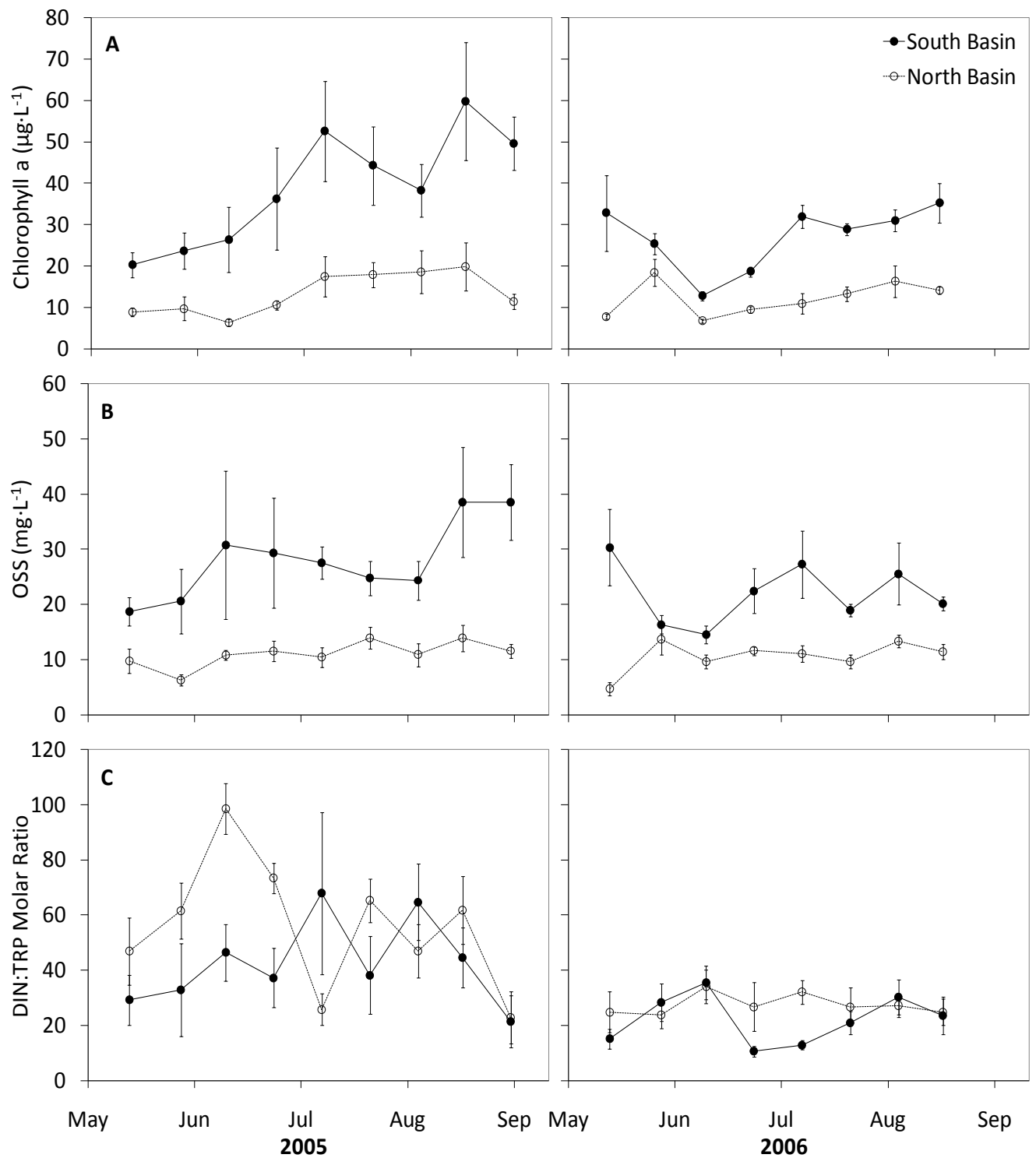


Figure 4.8. Chlorophyll a (A; $\mu\text{g/L}$) and organic suspended solids (OSS; B; mg/L) concentrations, and the dissolved inorganic nitrogen (DIN) to total reactive phosphorus (TRP) molar ratio (C) in the north and south basins of Lake Manitoba from May to September in 2005 and 2006. Points represent the biweekly mean of stations located in each respective basin of the lake (\pm SE; $n=6$ to 8).

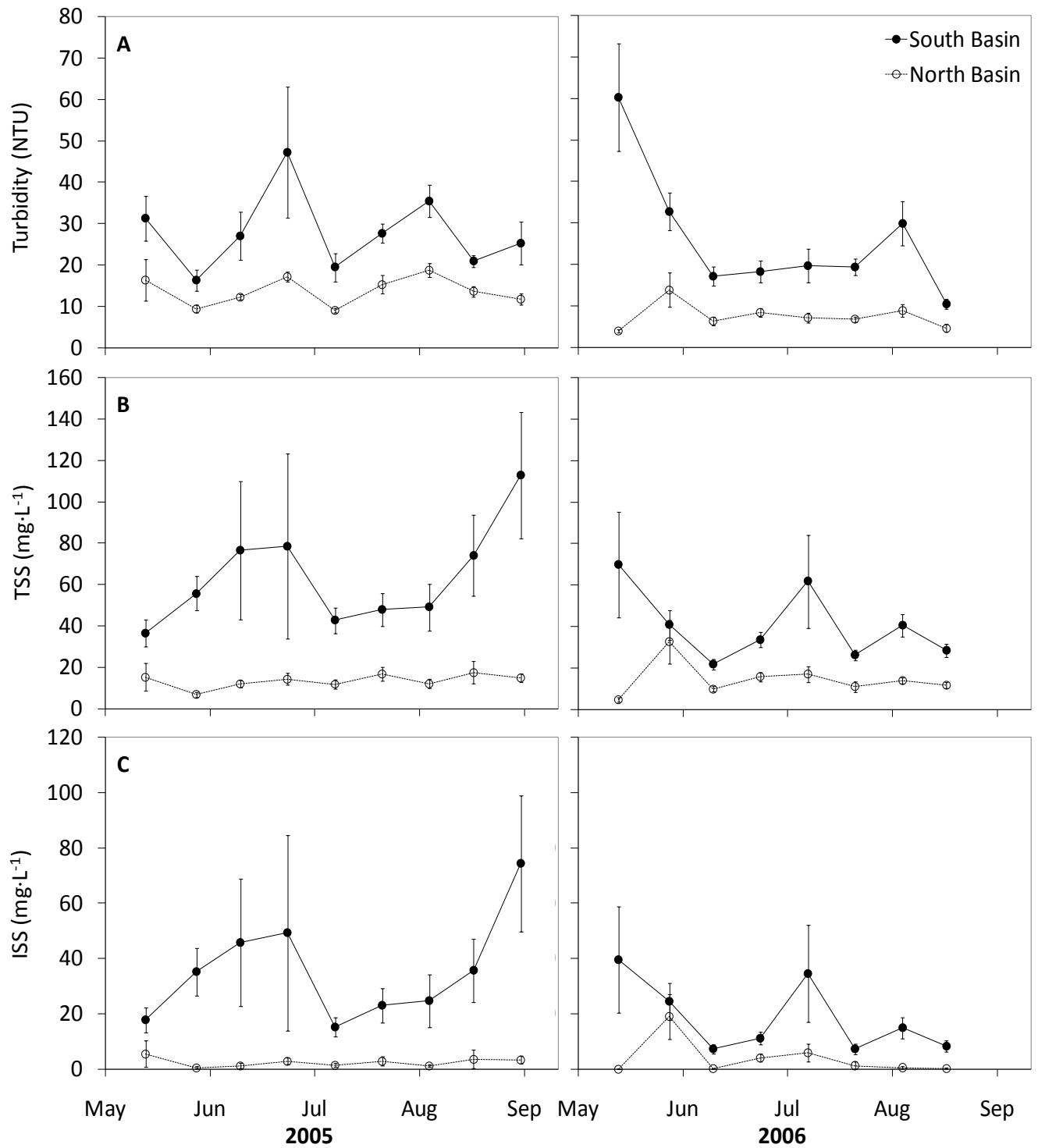


Figure 4.9. Turbidity (A; NTU), and total suspended solids (TSS; B) and inorganic suspended solids (ISS; C) concentrations in the north and south basins of Lake Manitoba from May to September in 2005 and 2006. Points represent the biweekly mean of stations located in each respective basin of the lake (\pm SE; $n=6$ to 8).

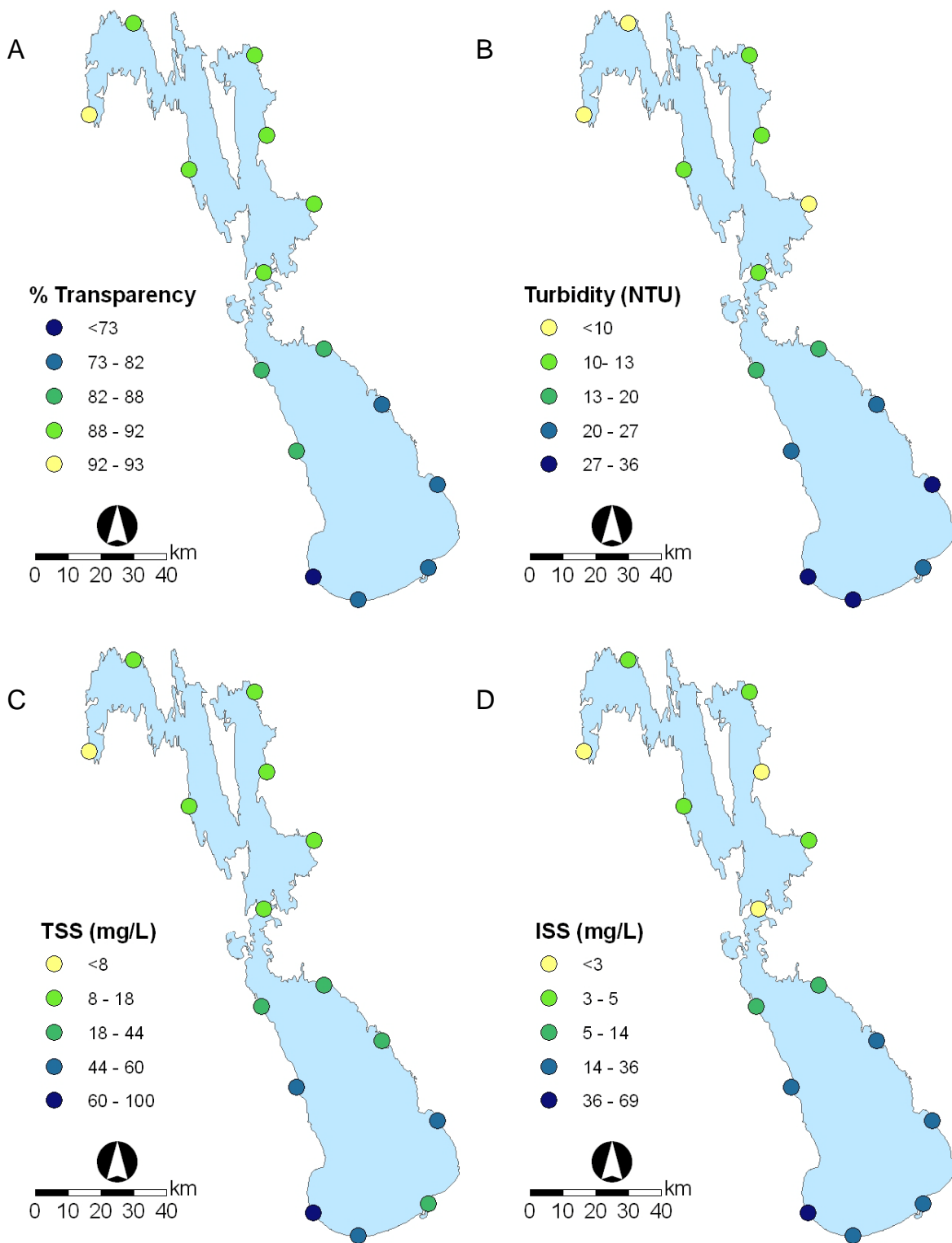


Figure 4.10. Spatial patterns in water transparency (450 nm; A; %), turbidity (B; NTU), total suspended solids (TSS; C; mg/L) and inorganic suspended solids (ISS; D; mg/L) in Lake Manitoba. Values represent the open-water (May to September) average of each station in 2005 and 2006 (n=17).

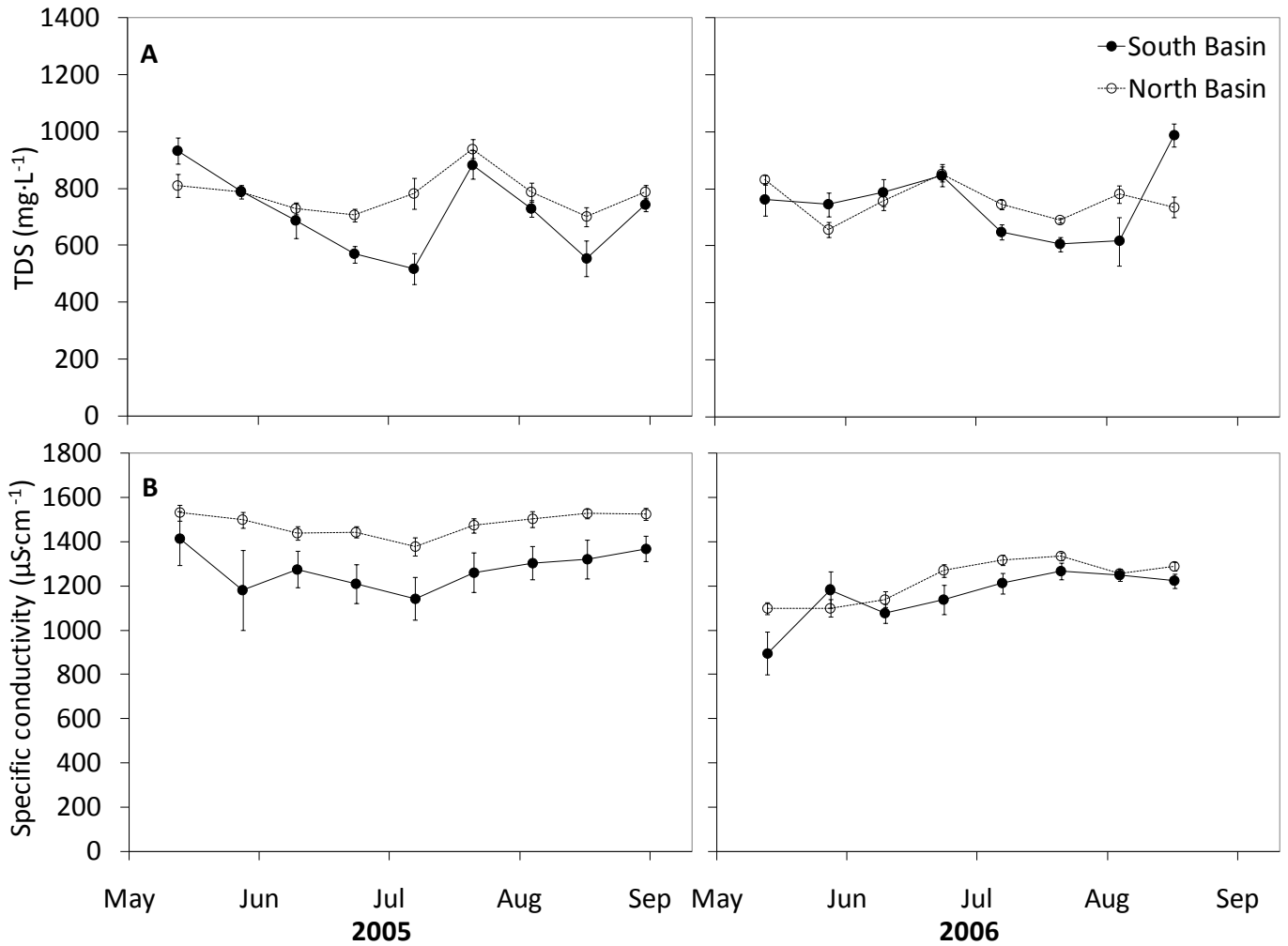


Figure 4.11. Total dissolved solids concentration (TDS; A; mg/L) and specific conductivity ($\mu\text{S}\cdot\text{cm}^{-1}$; B) in the north and south basins of Lake Manitoba from May to September in 2005 and 2006. Points represent the biweekly mean of stations located in each respective basin of the lake (\pm SE; $n=6$ to 8).

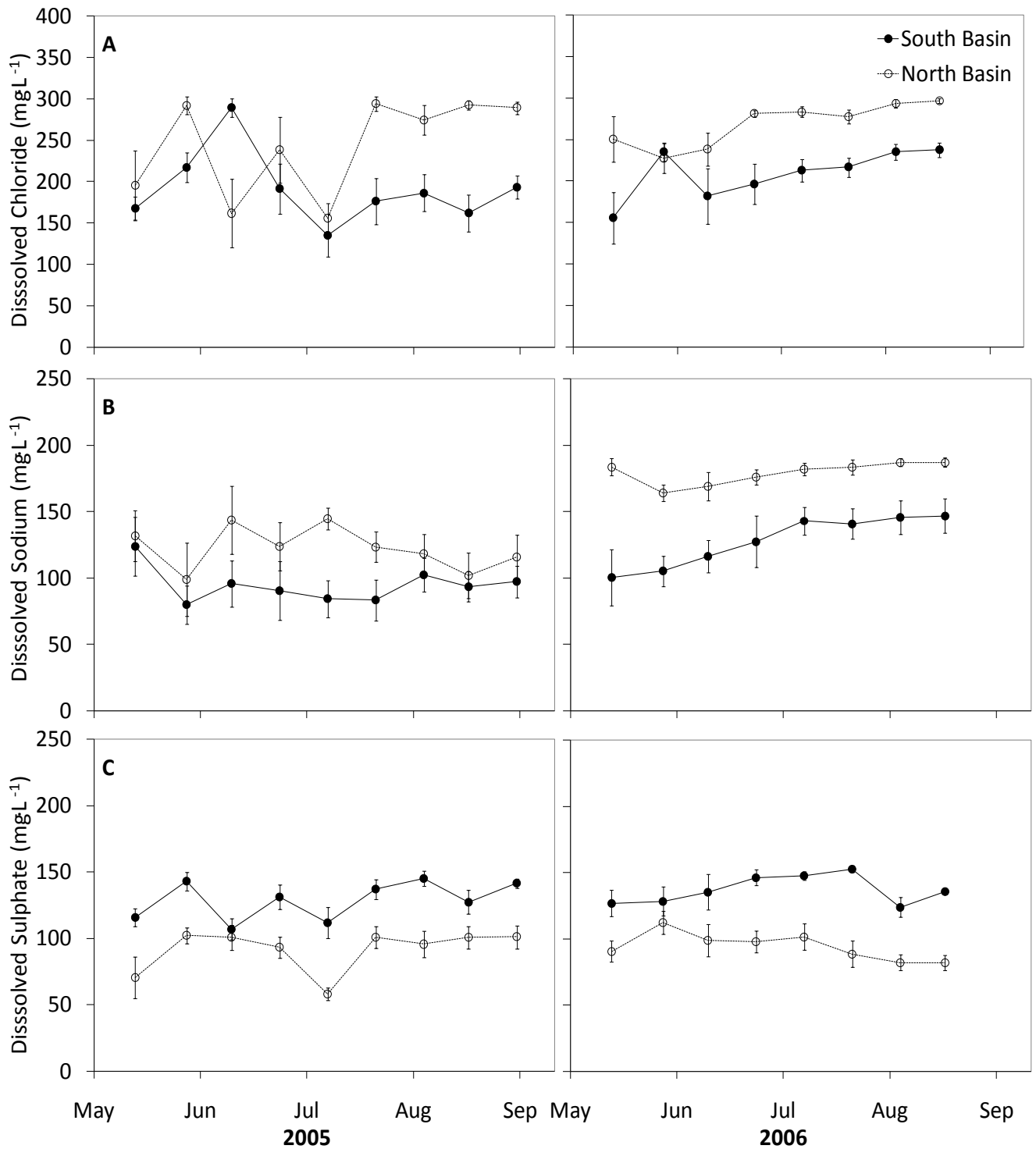


Figure 4.12. Dissolved chloride (A), dissolved sodium (B), and dissolved sulphate (C) concentrations (mg/L) in the north and south basins of Lake Manitoba from May to September in 2005 and 2006. Points represent the biweekly mean of stations located in each respective basin of the lake (\pm SE; n=6 to 8).

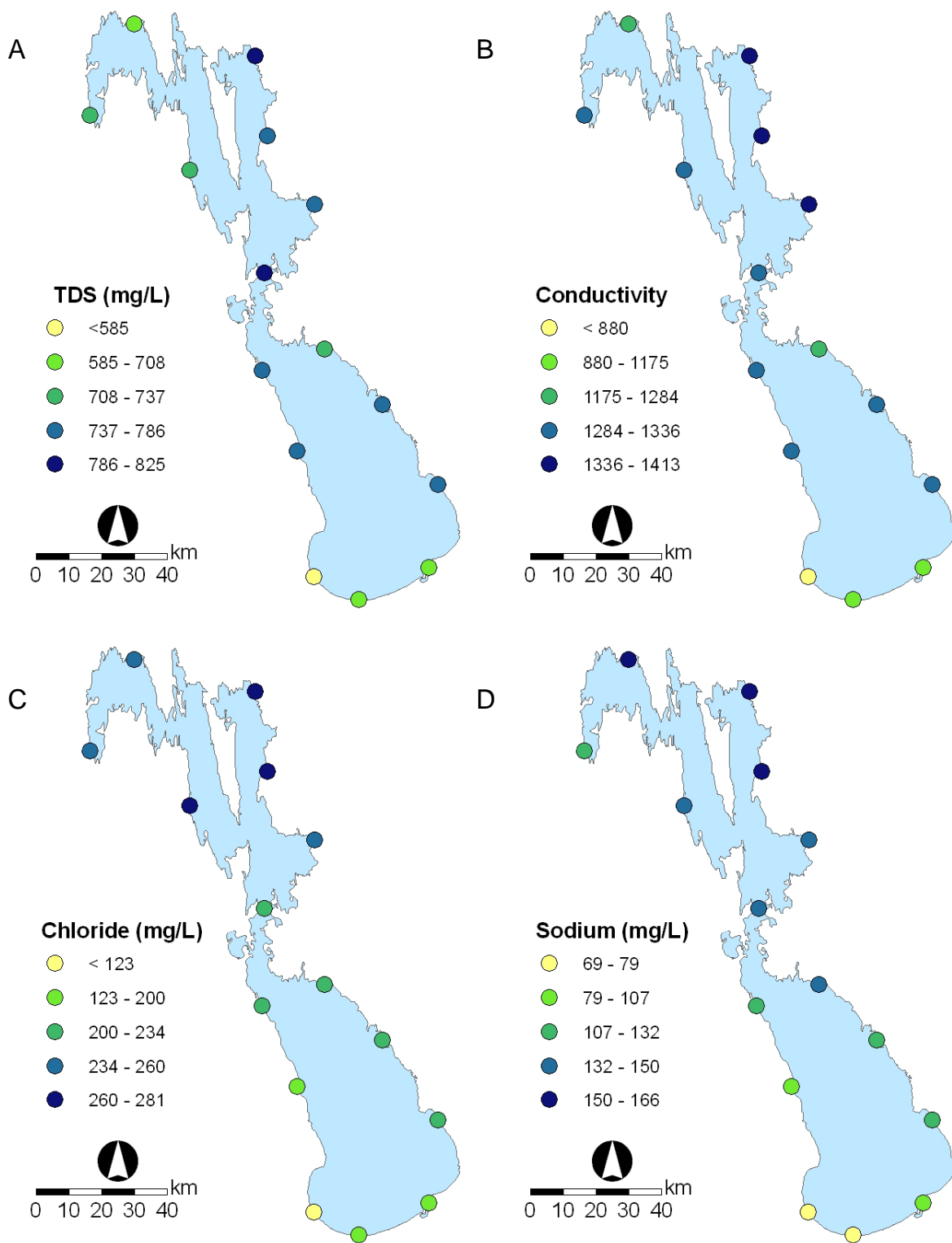


Figure 4.13. Spatial patterns in total dissolved solids (TDS; A; mg/L), specific conductivity (B; $\mu\text{S}/\text{cm}$), dissolved chloride (C; mg/L) and sodium (D; mg/L) in Lake Manitoba. Values represent the open-water (May to September) average of each station in 2005 and 2006 (n=17).

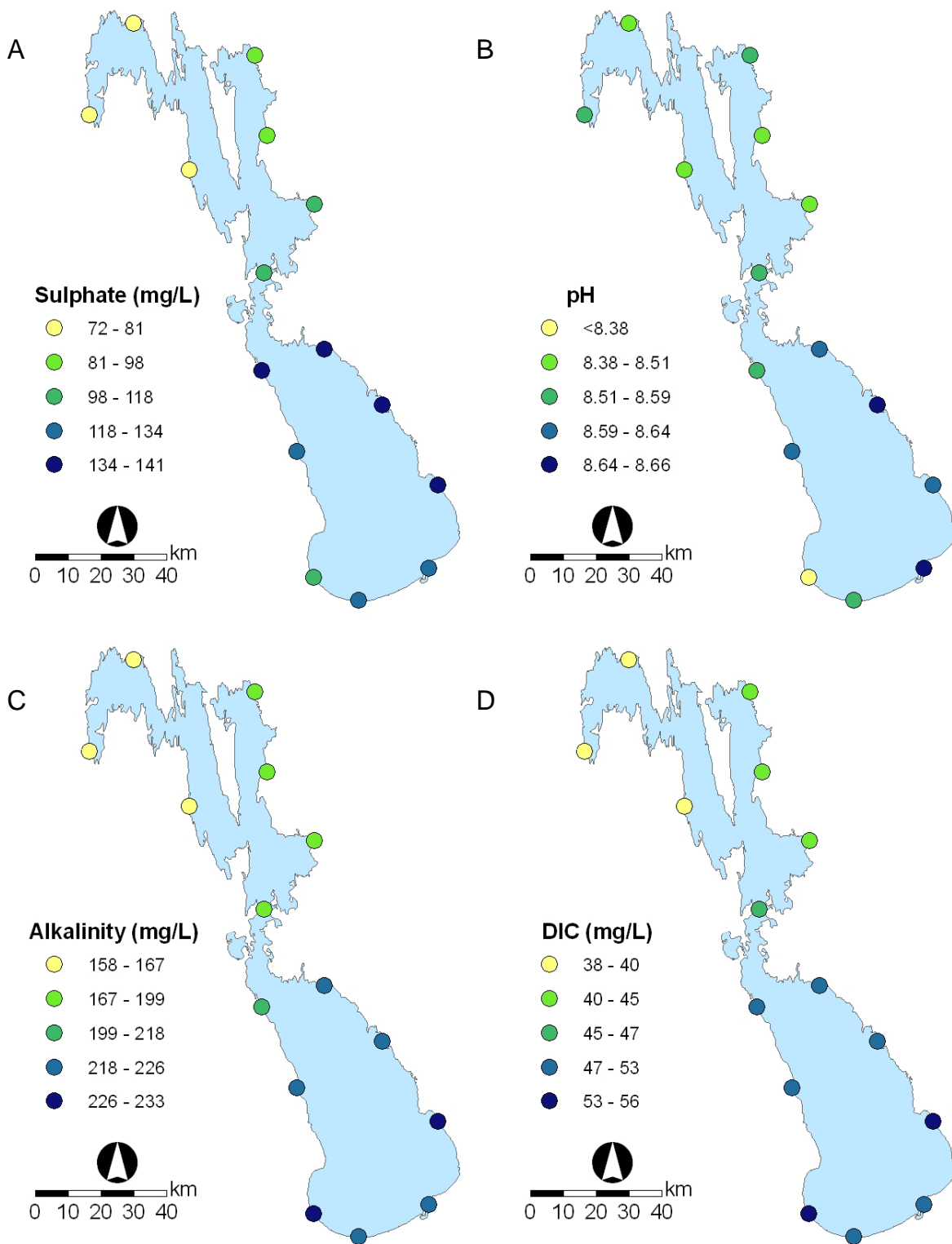


Figure 4.14. Spatial patterns in sulphate (A; mg/L), pH (B), alkalinity (as CaCO₃; C; mg/L) and dissolved inorganic carbon (DIC; D; mg/L) in Lake Manitoba. Values represent the open-water (May to September) average of each station in 2005 and 2006 (n=17).

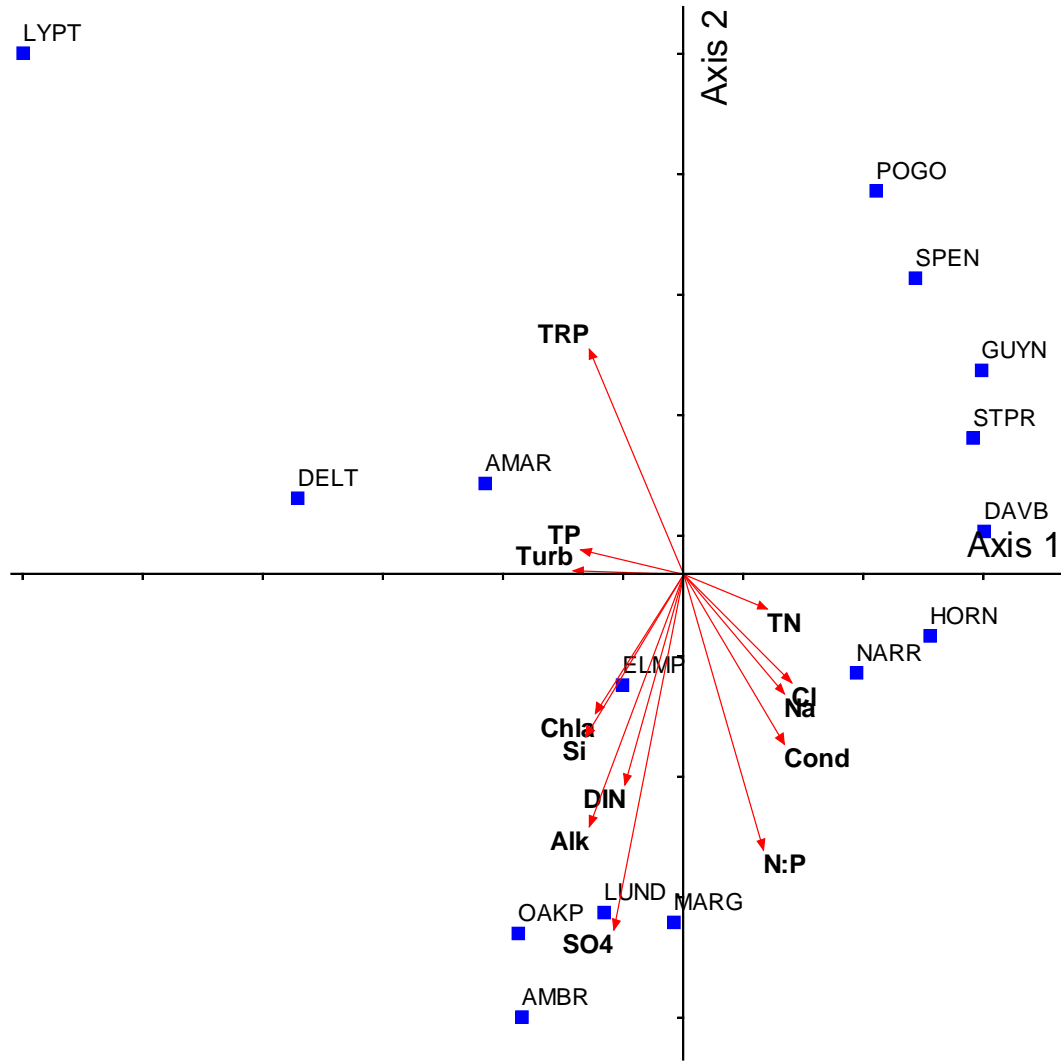


Figure 4.15. Principal components analysis biplot of mean open water (May to September) chemistry from 2005. Axis 1 and 2 accounted for 63 and 21 percent of the variation, respectively. Station codes: Amaranth (AMAR), St. Ambrose (AMBR), Delta (DELTA), Elm Point (ELMP), Davies Bay (DAVB), Guynemer (GUYN), Watchorn (HORN), Lundar (LUND), Lynch's Point (LYPT), Margaret Bruce (MARG), Narrows (NARR), Oak Point (OAKP), Manipogo (POGO), Spence Bay (SPEN), and Steep Rock (STPR).

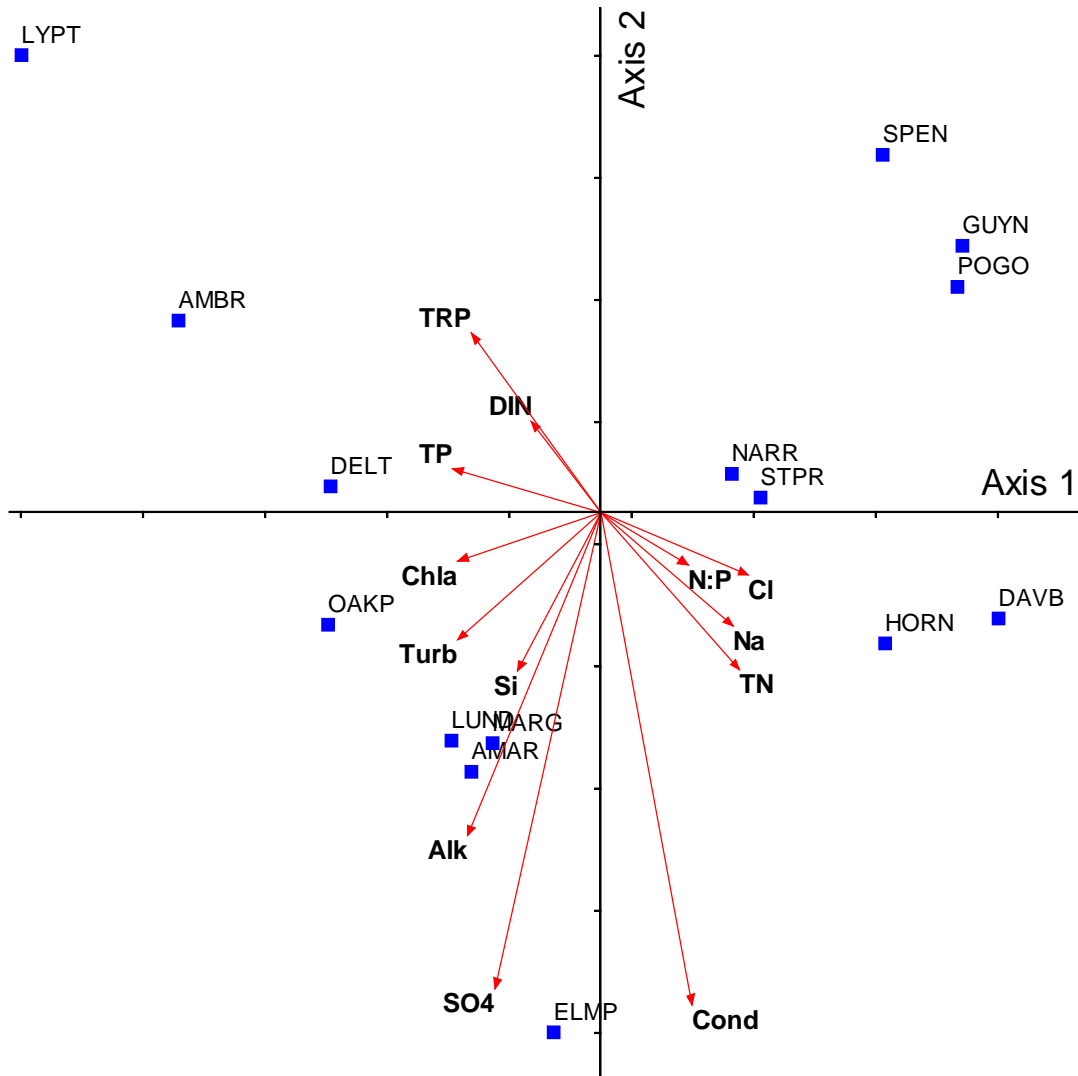


Figure 4.16. Principal components analysis biplot of mean open water (May to September) chemistry from 2006. Axis 1 and 2 accounted for 60 and 12 percent of the variation, respectively. Station codes: Amaranth (AMAR), St. Ambrose (AMBR), Delta (DELTA), Elm Point (ELMP), Davies Bay (DAVB), Guynemer (GUYN), Watchorn (HORN), Lundar (LUND), Lynch’s Point (LYPT), Margaret Bruce (MARG), Narrows (NARR), Oak Point (OAKP), Manipogo (POGO), Spence Bay (SPEN), and Steep Rock (STPR).

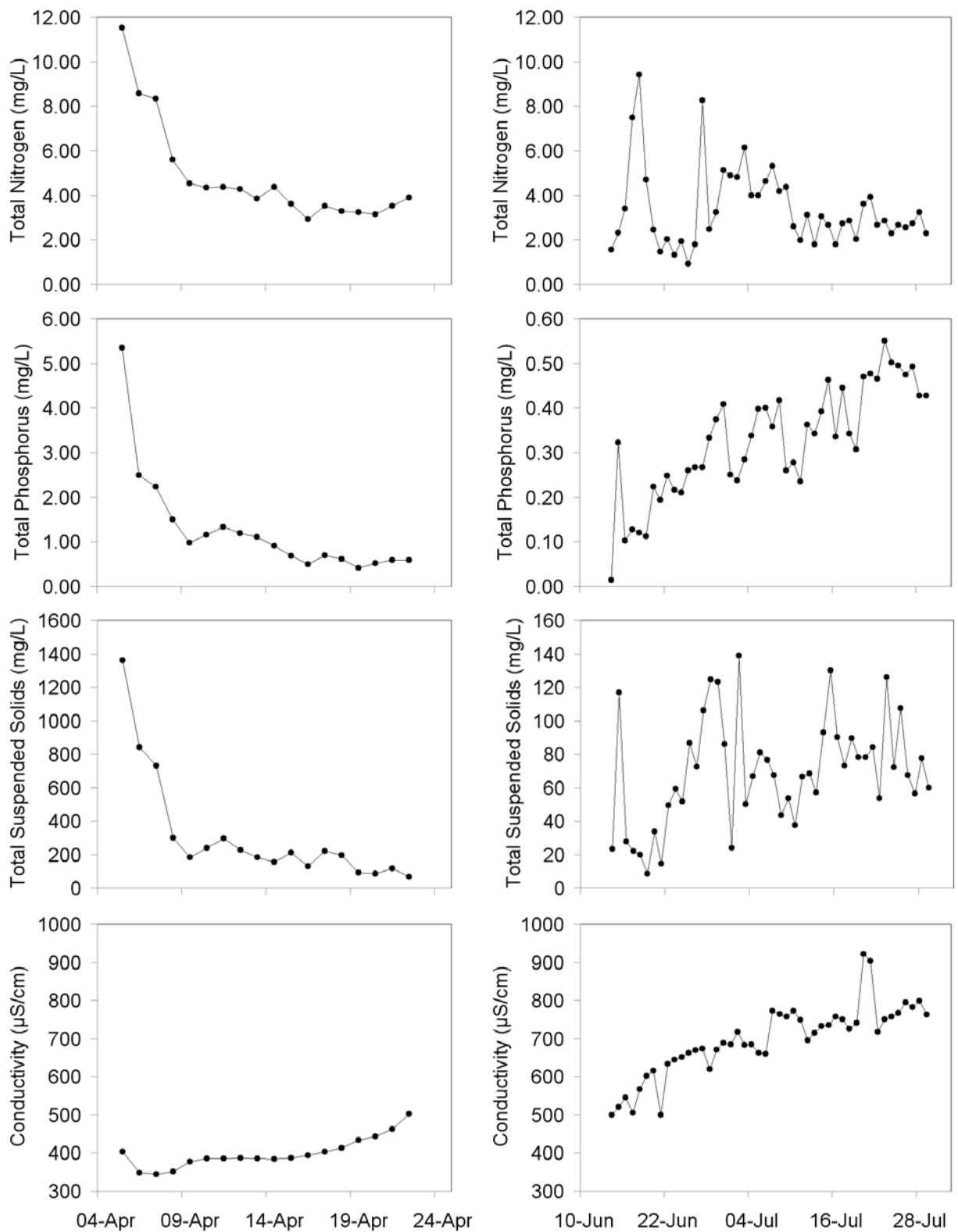


Figure 4.17. Trends in water quality during the spring (5 - 22 April) and summer periods (14 June – 29 July) of flow at the mouth of the Portage Diversion into Lake Manitoba in 2005. Total phosphorus and total suspended solids were an order of magnitude lower during summer operation.

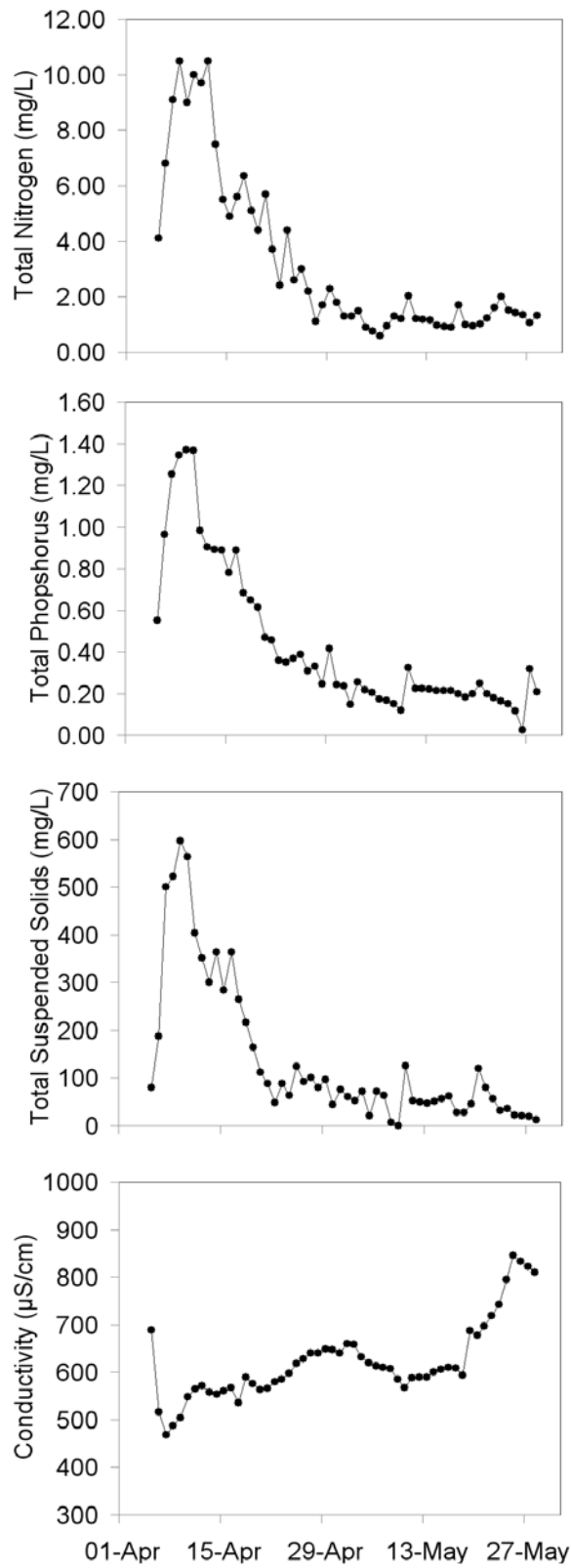


Figure 4.18. Trends in water quality during the spring period of flow at the mouth of the Portage Diversion into Lake Manitoba, 5 April – 28 May, 2006.

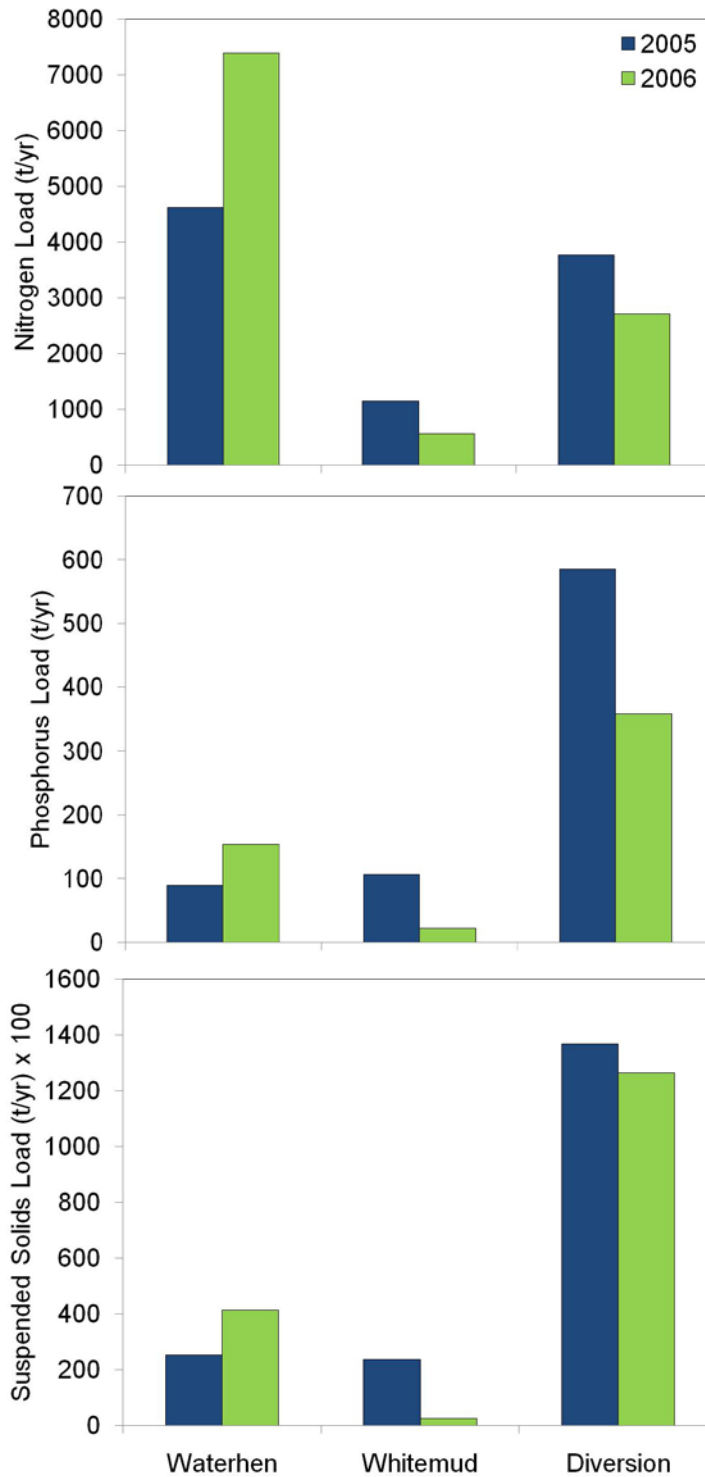


Figure 4.19. Nutrient and sediment loading (t/yr) to Lake Manitoba from the Waterhen River, the Whitemud River, and the Assiniboine River Diversion in 2005 and 2006.

5.0 RESULTS & DISCUSSION: WATER QUALITY TREND ANALYSIS

5.1 *Nutrients*

Based on the visual inspection of the total phosphorus time series record, concentrations appear to have increased at the long-term water quality sites in Lake Manitoba (Figure 5.1) from 1991 to 2007 with increasing variance in the series over time (Figure 5.2). Large peaks in the total phosphorus record were evident in 1992, 1995, 1996, 2001, 2004, and 2005, and occurred between April and May of each year. Peaks in total phosphorus generally followed patterns in high annual river inflow volume to the south basin, with exception of high phosphorus concentrations in 1992 and 2004 that occurred during comparatively lower river inflow volume years (Figure 5.3). Correlation analysis confirmed that inflow volume to the south basin of Lake Manitoba was positively correlated to total phosphorus concentrations in the lake (0.4377, $p < 0.0001$, $n = 184$) (Table 5.1), and concentrations were volume adjusted prior to trend analysis. Results of the trend analysis on volume adjusted total phosphorus concentrations revealed that there has been a significant increase ($+0.016 \text{ mg}\cdot\text{L}^{-1}$, $p = 0.045$) in total phosphorus concentration ($0.05 \text{ mg}\cdot\text{L}^{-1}$) in the south basin of Lake Manitoba from 1991 to 2007 (Table 5.1; Figure 5.2). The increase in total phosphorus over the 16 year record represents a 32 percent increase in phosphorus concentration in Lake Manitoba or a 2 percent annual increase in phosphorus concentration. Volume adjustment did not change the magnitude of the phosphorus trend, however, the significance of the trend changed with adjusted ($p = 0.045$) and unadjusted data ($p = 0.008$) (Table 5.2).

Dissolved phosphorus concentrations appear to be declining in Lake

Manitoba from the 1996 to 2007 time series, however there is no apparent change in the particulate phosphorus time series during the same period (Figure 5.3). From 1996 to 2007, there were noticeable peaks in dissolved phosphorus in 2001 (0.139 mg·L⁻¹), 2003 (0.144 mg·L⁻¹), 2004 (0.351 mg·L⁻¹), 2005 (0.180 mg·L⁻¹), and 2007 (0.180 mg·L⁻¹). Similar to patterns in total phosphorus, peak values typically occurred between April and May in Lake Manitoba with exception of the peak dissolved phosphorus value in 2003 which occurred in October (Figure 5.3). Apart from the peaks in the dissolved phosphorus time series, there appears to be an overall declining trend in dissolved phosphorus time series with slightly higher concentrations occurring between 1996 to 2001 and lower concentrations from 2002 onwards (Figure 5.3). Particulate phosphorus data were highly seasonal with noticeably lower concentrations during the winter months under ice cover and higher concentrations generally during the spring and fall months (Figure 5.3). Over the 11 year period of record, the highest particulate phosphorus concentration (0.347 mg · L⁻¹) was recorded in April 2005 during high spring flows and the operation of the Assiniboine River Diversion (Figure 5.3, Figure 5.4). Similar to total phosphorus, both dissolved phosphorus and particulate phosphorus concentrations were well correlated to inflow volume to the south basin (0.2704, p=0.0018, n=129 for dissolved phosphorus, 0.3847, p=<0.0001, n=131 for particulate phosphorus) and were volume weighted prior to trend analysis. The trend analysis on the unadjusted dissolved phosphorus data revealed that there was a significant downward trend in dissolved phosphorus from 1996 to 2007 (-0.001 mg · L⁻¹yr⁻¹, p=0.022), however with volume adjustment there was a no detectable change in dissolved phosphorus and

the trend was insignificant ($0.000 \text{ mg}\cdot\text{L}^{-1}\text{yr}^{-1}$, $p=0.105$) (Table 5.2). Trend analysis revealed that there was no significant trend in both the adjusted ($p=0.333$) and unadjusted ($p=0.392$) particulate phosphorus concentrations from 1996 to 2007 (Table 5.2).

Concomitant with increases in total phosphorus, the total nitrogen time series also showed that nitrogen has increased in the south basin of Lake Manitoba (Figure 5.5). From 1991 to 1997, total nitrogen appears to be decreasing while the last five years of the time series (i.e., from 2002 to 2007) appears to be higher than concentrations earlier in the time series. From 1991 to 2007, total nitrogen concentrations ranged from $0.180 \text{ mg}\cdot\text{L}^{-1}$ (in August 2001) to $2.94 \text{ mg}\cdot\text{L}^{-1}$ (in August 2004). Peaks in nitrogen concentration are also evident in 1995 ($2.67 \text{ mg}\cdot\text{L}^{-1}$), 2000 ($2.35 \text{ mg}\cdot\text{L}^{-1}$), 2003 ($2.74 \text{ mg}\cdot\text{L}^{-1}$). Total nitrogen concentrations were highly variable however the high concentrations did not consistently occur during the spring months as observed with total phosphorus. Large inter-annual and intra-annual variance was also evident in the time series with large variance occurring between 2004 and 2007 (Figure 5.5). Trend analysis was carried out on unadjusted total nitrogen concentrations, as the monthly concentrations were not significantly correlated to average monthly inflow volume to the south basin of Lake Manitoba (-0.0052 , $p=0.9444$, $n=184$) (Table 5.2). Results of the seasonal Kendall test for trend on unadjusted total nitrogen concentrations confirmed that there has been a significant increase ($+0.176 \text{ mg}\cdot\text{L}^{-1}$, $p=0.038$) in total nitrogen from 1991 to 2007 at station LKS013 in the south basin of Lake Manitoba (Figure 5.5). Overall, total nitrogen has increased by approximately 16 percent from 1991 to 2007 in Lake Manitoba which is

equivalent to a 0.8 percent annual increase in nitrogen ($+0.011 \text{ mg}\cdot\text{L}^{-1} \text{ yr}^{-1}$) (Table 5.2).

From visual inspection of the ammonia time series it is difficult to visually detect a trend in the data set due to the large inter-annual and intra-annual variance in ammonia (Figure 5.6). However, ammonia concentrations appear to be stable from 1991 to 2000, and increasing concentrations are first apparent in 2000 and 2001. The greatest increases in ammonia concentrations appear to occur between 1999 and 2007 (Figure 5.6). From 1991 to 2007 dissolved ammonia concentrations ranged from $0.01 \text{ mg}\cdot\text{L}^{-1}$ to a maximum concentration of $0.892 \text{ mg}\cdot\text{L}^{-1}$ in March 1995. Other large peaks in dissolved ammonia concentration were evident in the spring of 1994 ($0.654 \text{ mg}\cdot\text{L}^{-1}$), 1996 ($0.334 \text{ mg}\cdot\text{L}^{-1}$), 2004 ($0.530 \text{ mg}\cdot\text{L}^{-1}$), 2005 ($0.330 \text{ mg}\cdot\text{L}^{-1}$), and 2007 ($0.270 \text{ mg}\cdot\text{L}^{-1}$) (Figure 5.6A). Similar to total nitrogen, dissolved ammonia concentrations were also not well correlated with inflow volume to the south basin of the lake and did not warrant volume adjustment prior to trend analysis (0.0463 , $p=0.5322$) (Table 5.1). Results of the seasonal Kendall test for trend on ammonia concentrations revealed that there has been a highly significant increase ($+0.001 \text{ mg}\cdot\text{L}^{-1} \text{ yr}^{-1}$, $p<0.001$, $n=184$) in ammonia concentrations in Lake Manitoba from 1991 to 2007 (Table 5.2, Figure 5.6). Overall, the trend analysis revealed that dissolved ammonia concentrations have increased by 53 percent over 16 years in Lake Manitoba ($+3.3 \text{ percent}\cdot\text{yr}^{-1}$) (Table 5.2).

5.2 Chlorophyll a and Algal Nutrient Limitation (TN:TP)

Between 1991 and 2007, chlorophyll a in Lake Manitoba ranged from $0.5 \mu\text{g}\cdot\text{L}^{-1}$ (which generally occurred on several occasions between January and April) to

a peak algal biomass of $85 \mu\text{g}\cdot\text{L}^{-1}$ in August 2005. According to the OECD (1982) trophic classification based on mean chlorophyll a concentration, Lake Manitoba is considered eutrophic (10.5 ± 0.93 , $n=172$). Chlorophyll a concentrations exceeded $40 \mu\text{g}\cdot\text{L}^{-1}$ on six occasions between 1991 and 2007 with four of the six peak concentrations occurring between 2005 and 2006. Similar to nutrient trends, there was also an apparent increasing trend in the chlorophyll a time series from 1991 to 2007 with most noticeable increases after 2001 (Figure 5.7). Chlorophyll was positively correlated to inflow volume (0.2477 , $p=0.001$, $n=172$) (Table 5.1), and the trend analysis was carried out on adjusted and unadjusted data for comparison (Table 5.2). The seasonal Kendall test and Kendall slope estimator revealed that there was a highly significant upward trend in chlorophyll a ($+0.280 \mu\text{g}^{-1} \text{yr}^{-1}$, $p<0.001$) from 1991 to 2007 in Lake Manitoba (Table 5.2, Figure 5.7). Overall, there has been a 72 percent increase ($+4.5 \text{ percent yr}^{-1}$) in chlorophyll a concentrations ($8.3 \mu\text{g}\cdot\text{L}^{-1}$) from 1991 to 2007 in the south basin of Lake Manitoba (Figure 5.7). The magnitude of the trend declined from $0.313 \mu\text{g}^{-1} \text{yr}^{-1}$ to $0.218 \mu\text{g}\cdot\text{L}^{-1} \text{yr}^{-1}$ with volume adjustment of chlorophyll a concentrations, however, the significance of the trend remained unchanged between adjusted and unadjusted data ($p=<0.0001$).

Based on data from 1991 to 2007, the TN:TP molar ratio in Lake Manitoba ranged from 2 (in June 2001) to 396 (in February 2006) with a median that was indicative of phosphorus limitation (56). Ratios less than or equal to 10 (i.e., indicative of nitrogen limitation) were only detected in the lake between 2001 and 2005, and 75 percent of the ratios below 20 also occurred between 2001 and 2005 (Figure 5.8). Based on visual examination of the monthly time series, the ratio

appeared to be declining from 1991 to 2000 with an evident drop in the ratio in 2001. However, from 2001 to 2007, the TN:TP ratio appeared to be increasing in Lake Manitoba. In particular, the ratio was increasing in Lake Manitoba in the last two years of the series (Figure 5.8). The TN:TP molar ratio was also positively correlated to the inflow volume to the south basin of Lake Manitoba (0.3549, $p < 0.0001$) and the trend analysis was performed in both adjusted and unadjusted data (Table 5.2). Trend analysis performed on the unadjusted data revealed that the TN:TP ratio had declined by 32 percent in Lake Manitoba from 1991 to 2007 (-1.03 yr^{-1} , -16.5 from 1991 to 2007, $p = 0.02$). With volume adjustment, there was also a declining trend in the TN:TP series, and the estimated slope of the trend was much lower (-0.335 yr^{-1} , -5.3 from 1991 to 2007), however the adjusted trend was not statistically significant ($p = 0.295$) (Table 5.2).

5.3 Dissolved Solids, Conductivity, Major Ions

Lake Manitoba appears to be becoming less brackish, as total dissolved solids and conductivity appear to have decreased considerably from 1991 to 2007 (Figure 5.9, Figure 5.10). Evident in the time series is the highly seasonal nature of the data, with lowest conductivity measurements (mean=615, $n=15$) and total dissolved solids concentrations (mean = 344, $n=15$) in April and peak dissolved solids concentrations (mean=1210, $n=16$) and conductivity measurements (mean=2074, $n=16$) under ice cover in February. There was also comparatively large range between minimum and maximum conductivity and total dissolved solids values from 1991 to 1998 relative to the range in values after 1999 (Figure 5.9, Figure 5.10). Of all the conductivity measurements above $2000 \mu\text{S}\cdot\text{cm}^{-1}$ ($n=30$), a

majority of the values ($n=26$) above $2000 \mu\text{S} \cdot \text{cm}^{-1}$ occurred between 1991 and 1999 in Lake Manitoba (Figure 5.10). Total dissolved solids followed the same pattern with higher dissolved solids concentrations ($> 1200 \text{ mg} \cdot \text{L}^{-1}$) occurring earlier in the series (Figure 5.9). Correlation analysis with the river inflow volume to the south basin revealed significant negative correlations with total dissolved solids (-0.5833 , $p < 0.0001$) and conductivity (-0.6538 , $p < 0.0001$) (Table 5.1). Volume-adjustment of both the total dissolved solids and conductivity time series appears to have reduced the high degree of variability in the data owing to the seasonal nature of the data. The volume adjusted data for total dissolved solids and conductivity showed a similar oscillating pattern with lower values occurring from 1997 to 1998 and from 2002 to 2003 and peak values occurring in 1994, 1999, 2001, and in 2005 (Figure 5.9, Figure 5.10).

Trend analysis was performed on both the adjusted and unadjusted dissolved solids and conductivity data. The seasonal Kendall test for trend on the volume-adjusted data revealed that there has been a highly significant decreasing trend in total dissolved solids in Lake Manitoba from 1991 to 2007 (-347 , $p < 0.001$) (Table 5.2). Based on the seasonal Kendall slope for total dissolved solids ($-13/\text{year}$), there has been a 1.4 percent annual declining trend, and an overall 22 percent decrease in total dissolved solids concentrations from 1991 to 2007 at station LKS013 in the south basin of Lake Manitoba (Figure 5.9). The seasonal Kendall test for trend on volume-adjusted conductivity values indicated that there has been a highly significant decreasing trend in conductivity from 1991 to 2007 (-373 , $p < 0.001$) (Table 5.2). The seasonal Kendall slope for conductivity was -23.7 per year meaning that

there has been a 1.5 percent annual decrease in median conductivity and a 24 percent decrease in median conductivity from 1991 to 2007 in the south basin Lake Manitoba (Table 5.2; Figure 5.10).

Chloride and sodium make up the dominant ion chemistry in Lake Manitoba and showed strong downward tendencies from 1991 to 2007 in the lake concurrent decreases in conductivity and dissolved solids (Figure 5.11, Figure 5.12). Dissolved chloride ranged from $5 \text{ mg} \cdot \text{L}^{-1}$ (in April 1996) to $502 \text{ mg} \cdot \text{L}^{-1}$ (January 1992) and has a median concentration of $291 \text{ mg} \cdot \text{L}^{-1}$. Chloride is the dominant anion in Lake Manitoba with a median concentration of 285 mg/L. Concurrent with trends in total dissolved solids and conductivity, chloride concentrations also appear to be decreasing in Lake Manitoba from 1996 to 2007, however the large seasonal fluctuations in chloride make it difficult to distinguish a trend in the data (Figure 5.10). A seasonal Kendall trend test on the volume-adjusted chloride values revealed a significant decreasing trend in dissolved chloride from 1996 to 2007 (-52, $p=0.002$) (Table 5.1). Dissolved chloride has decreased approximately 3 percent per year (annual slope of -8.5) and has decreased by 29 percent from 1996 to 2007 (Table 5.1; Figure 5.10).

Sodium is the dominant cation in Lake Manitoba with a median concentration of 193 mg/L. Sodium concentrations appear to have a strong declining trend over the 16 year period of record (Figure 5.11). The seasonal Kendall test for trend detected a significant decrease in sodium concentrations in Lake Manitoba from 1991 to 2007 (-80, $p=0.013$). Based on the seasonal Kendall slope for sodium from 1991 to 2007 (annual slope of -4.05), there has been a 34 percent decrease in

sodium concentrations, which is equivalent to a 2 percent annual decrease in the median sodium concentration in Lake Manitoba (Table 5.1, Figure 5.11).

Sulphate is also a major anion in Lake Manitoba with a median concentration of 154 mg/L. Sulphate concentrations also appear to be declining with time in Lake Manitoba (Figure 5.12). However, trends in the volume-adjusted dissolved sulphate values in Lake Manitoba were not consistent with decreases in chloride and sodium. The seasonal Kendall test for trend determined that there was no significant trend in volume-adjusted dissolved sulphate values ($p=0.352$) (Table 5.1, Figure 5.12). Although insignificant, the seasonal Kendall statistic for dissolved sulphate was -6 meaning that there may be a negative trend in the data, however, the sulphate data are currently not sufficient to detect a statistically significant trend (Table 5.1).

5.4 Discussion

It is clear that the south basin of Lake Manitoba experienced significant changes in water quality from 1991 to 2007. Increases in total phosphorus, total nitrogen, and chlorophyll indicated that the very south end of Lake Manitoba has become more eutrophic. Trend analysis also indicated that the lake water in the south basin is becoming fresher evidenced by clear declines in dissolved solids, conductivity, sodium, and chloride.

One of the reasons for the observed water quality changes may be related to the effects of the Assiniboine River Diversion. Data collected in the present study indicated that the Diversion was much more phosphorus rich than the lake and that the Diversion was the greatest source of phosphorus and the second largest source of nitrogen to Lake Manitoba. Conductivity, sodium, and chloride were also

significantly lower in water from the Assiniboine River Diversion in comparison to water from Lake Manitoba. Analysis of the historical flow record for the Diversion indicated that average flows and the average length of operation from 1990 to 2009 was twice as great as compared to the 1970 to 1989 period. During the period of trend analysis (1991 to 2007) the Diversion operated every year with exception of 2000. It is possible that greater Diversion inflows and longer periods of Diversion operation in recent years may be contributing greater nutrient loads to the lake and causing the lake to become less saline.

It is very likely that the Diversion also has the most immediate effect on water quality samples collected at the long-term University of Manitoba Delta Marsh Field Station given the proximity of the station to the outflow of the Diversion channel. Highly significant correlations between monthly river inflow volume to the south basin of the lake and total phosphorus, particulate phosphorus, conductivity, total dissolved solids, sodium, and chloride suggests that the Diversion affects water quality at the long-term monitoring site. While the inflow volume to the south basin includes the Whitemud River, much of the water volume is transported by the Diversion during periods of operation.

Changes in the water chemistry of the Whitemud River also may also partly explain increased nutrient concentrations in Lake Manitoba. Jones and Armstrong (2001) conducted a trend analysis on major rivers and streams in Manitoba to characterize changes in total phosphorus and total nitrogen concentrations from the mid 1970s to the late 1990s. The study found that total nitrogen increased by 37 % and total phosphorus increased by 64 % over the 30 year period of record. Although

flow in the Whitemud River generally determines the total nutrient load transported to the lake, concentrations may also be important during low flow conditions and may help to explain nutrient increases in Lake Manitoba. Results of the trend analysis on the Waterhen River indicated that total nitrogen concentrations had increased while no change was detected for total phosphorus. However, it seems unlikely that increases in total nitrogen concentrations in the Waterhen River would affect water quality at the very south end of Lake Manitoba given the convoluted morphometry of the north basin. It is unknown to what extent water is exchanged between basins, although the spatially heterogeneous water chemistry suggests limited mixing in Lake Manitoba.

Precipitation may also account for changes observed in the water quality record for Lake Manitoba. Precipitation accounts for a substantial fraction of the total water budget for Lake Manitoba (43 %) because of the large surface area of the lake (LMRRAC 2003). Precipitation is a source of nutrients to waterbodies and may be affecting nutrient concentrations in Lake Manitoba. Precipitation accounts for approximately 11 % of the total nitrogen load and 8 % of the total phosphorus load to Lake Winnipeg (Bourne *et al.* 2002). It is possible that the relative nutrient loading from precipitation may be greater for Lake Manitoba given the smaller lake volume and lower nutrient loads from tributary rivers. Precipitation may also dilute water from Lake Manitoba and may also be one of the factors explaining the decline in dissolved solids. However, visual inspection of the total annual precipitation record at Delta Marsh does not seem to indicate that precipitation has changed over the 1991 to 2007 period.

Changes in land use may also partly explain the water quality changes observed in Lake Manitoba. Land surrounding the lake is predominantly pastureland for cattle. Analysis of land cover changes using satellite imagery between 1993-94 and 2000-02 indicated that there has been an 85 percent increase in forage cover land in the south basin of Lake Manitoba (AAFC-PFRA 2005). Cattle access to the shoreline may represent a nutrient source to Lake Manitoba although it is unknown to what extent cattle contribute to the nutrient budget of Lake Manitoba. Crop agriculture is also prevalent to the south and west of the south basin where fertile soils exist and may also be another important diffuse source of nutrients to the lake. It is also unknown how drainage surrounding Lake Manitoba has changed and how it may be affecting the water quality of Lake Manitoba.

Increases in chlorophyll a may be related to the parallel increases in both nitrogen and phosphorus and the change in the N:P ratio in Lake Manitoba. Phosphorus increases were greater than observed for nitrogen in the south basin, and although the trend was not significant, the N:P ratio appeared to decline over the same period in the lake. Phytoplankton community composition and algal biomass are affected by changes in the N:P ratio (Smith 1982). Nitrogen is limiting at N:P ratios lower than 10:1 and ratios greater than 20:1 are generally indicative of phosphorus limitation. Cyanobacteria typically dominate the algal community when ratios are low (<10:1) because of their ability to fix atmospheric nitrogen. It is likely that the decline in the N:P ratio is explained by the high phosphorus loads transported by the Assiniboine River Diversion and the Whitemud River to the south basin of the lake. Further declines in the ratio will most certainly affect the

phytoplankton community composition and favour cyanobacteria dominance in Lake
Manitoba.

Table 5.1. Correlation analysis (Pearsons r) between monthly river inflow volume to the south basin of Lake Manitoba and nutrients, algal biomass, ions, and water chemistry, 1991-2007.

Variable	r	R^2	p -value
Alkalinity as HCO_3	-0.4415	0.1949	0.0000
Calcium	0.3942	0.1554	0.0013
Chloride	-0.7825	0.6123	0.0000
Chlorophyll a	0.2477	0.0614	0.0010
Conductivity	-0.6538	0.4274	0.0000
Dissolved Ammonia	0.0463	0.0021	0.5322
Dissolved Phosphorus	0.2704	0.0731	0.0018
Magnesium	-0.6774	0.4588	0.0000
Particulate Phosphorus	0.3847	0.1480	0.0000
pH	-0.0287	0.0008	0.6988
Potassium	-0.6606	0.4364	0.0000
Sodium	-0.7179	0.5154	0.0000
Sulphate	-0.6330	0.4007	0.0000
TN:TP	0.3549	0.1260	0.0000
Total Dissolved Solids	-0.5833	0.3401	0.0000
Total Nitrogen	-0.0052	0.0000	0.9444
Total Phosphorus	0.4377	0.1916	0.0000
Total Suspended Solids	0.3105	0.0964	0.0000

Table 5.2. Summary of the seasonal Kendall trend test and seasonal Kendall slope results for volume-adjusted residual values in the south basin of Lake Manitoba. Trends significant at $p \leq 0.05$ are highlighted in bold.

Variable	Unit	Median	n	Period of Record	Unadjusted Data			Volume-Adjusted Data		
					S Statistic	p-value	Slope	S Statistic	p-value	Slope
Total Phosphorus	mg/L	0.050	184	1991-2007	174	0.008*	0.001	121	0.047*	0.001
Dissolved Phosphorus	mg/L	0.035	129	1996-2007	-87	0.022*	-0.001	-54	0.105	0.000
Particulate Phosphorus	mg/L	0.017	131	1996-2007	13	0.392	0.000	20	0.333	0.000
Total Nitrogen	mg/L	1.30	184	1991-2007	128	0.038*	0.011			
Dissolved Ammonia	mg/L	0.030	184	1991-2007	257	<0.0001**	0.001			
Chlorophyll <i>a</i>	µg/L	6.25	172	1991-2007	358	<0.0001**	0.313	257	<0.0001**	0.280
TN:TP	molar ratio	56	184	1991-2007	-149	0.02**	-1.031	-39	0.295	-0.335
Total Suspended Solids	mg/L	18	184	1991-2007	-26	0.358	0.000	-107	0.070	-0.286
Total Dissolved Solids	mg/L	947	181	1991-2007	-509	<0.0001**	-18.5	-347	<0.0001**	-13.0
Conductivity	µS/cm	1605	184	1991-2007	-535	<0.0001**	-35.71	-373	<0.0001**	-23.9
Chloride	mg/L	291	72	1991-2007	-173	<0.0001**	-9.071	-101	0.004**	-4.04
Sodium	mg/L	191	61	1991-2007	-127	<0.0001**	-6.000	-80	0.013*	-4.05
Sulphate	mg/L	160	67	1991-2007	-86	0.011*	-1.708	-30	0.213	-0.523
Magnesium	mg/L	56.4	62	1991-2007	-45	0.122	-0.400	9	0.408	0.000
Calcium	mg/L	41.5	63	1991-2007	72	0.028*	0.000	23	0.270	0.000
Potassium	mg/L	16.3	63	1991-2007	-41	0.143	-0.055	39	0.156	0.016
pH	pH units	8.39	181	1991-2007	-474	<0.0001**	-0.016			
Alkalinity as HCO ₃	mg/L	260	184	1991-2007	313	<0.0001**	2.34	218	0.001**	1.25

* $p \leq 0.05$ ** $p \leq 0.005$

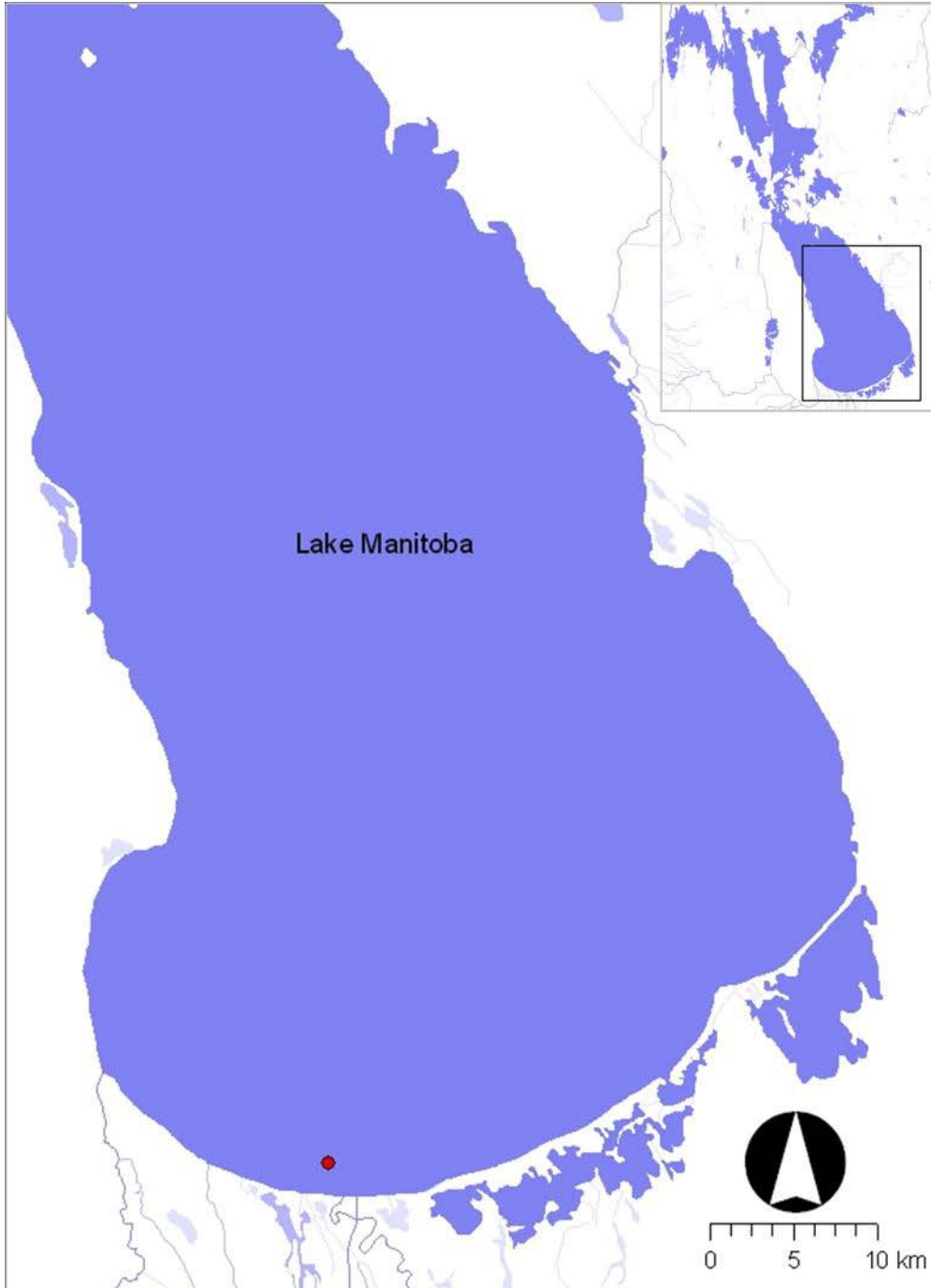


Figure 5.1. Location of the provincial long-term monitoring station in the south basin of Lake Manitoba.

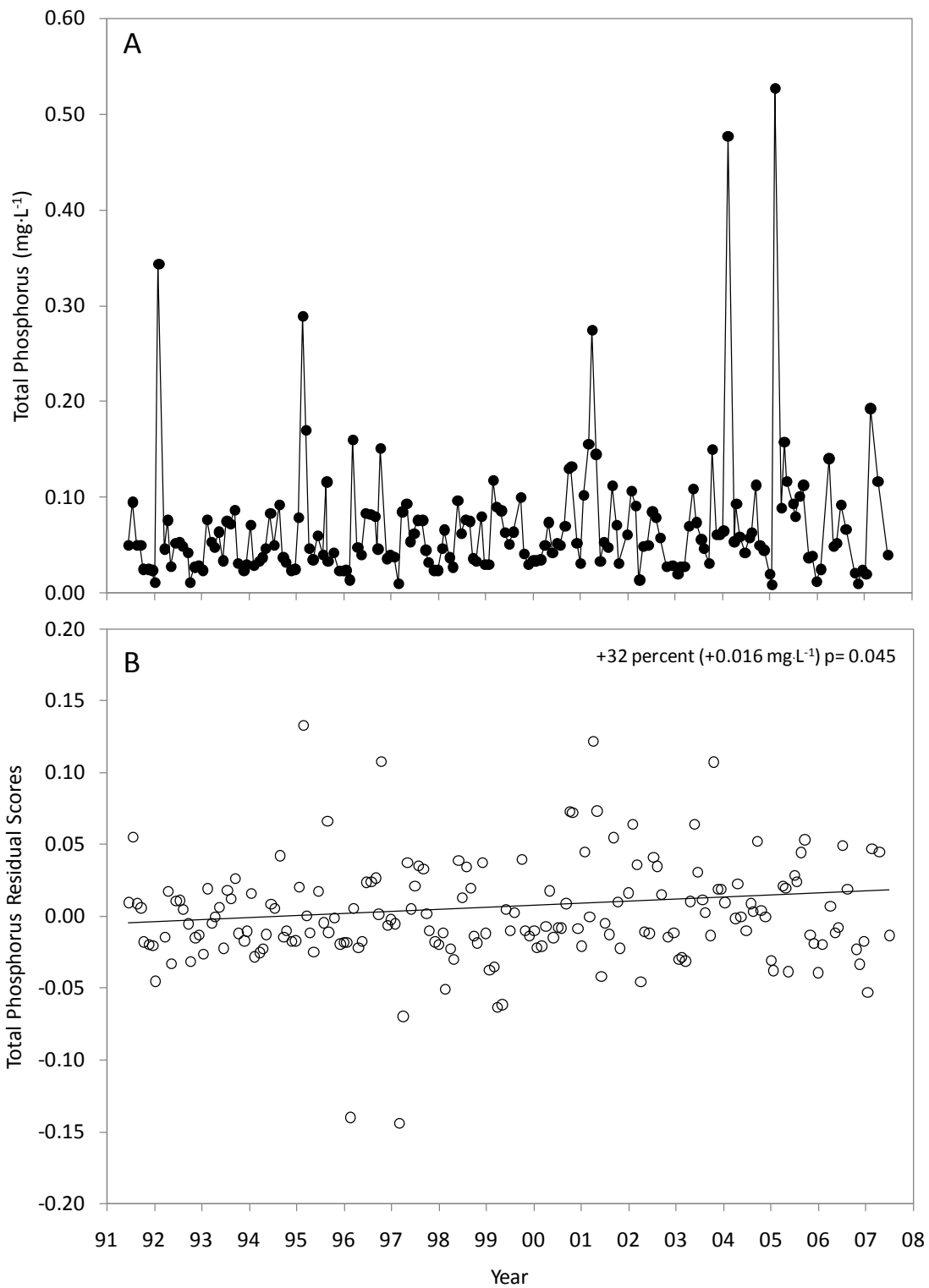


Figure 5.2. Time series of monthly total phosphorus concentrations (A) and trend in volume-adjusted total phosphorus residuals (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

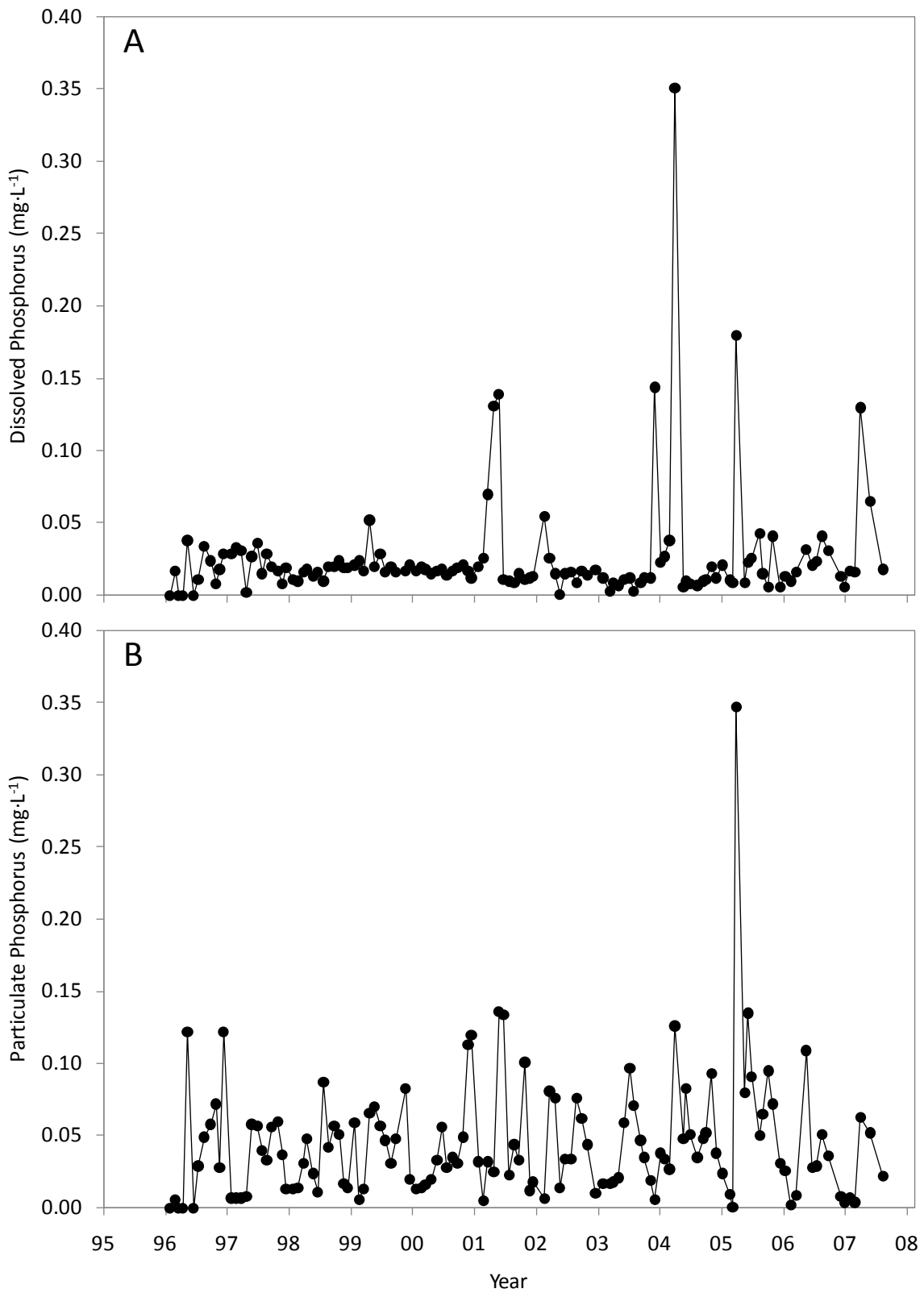


Figure 5.3. Time series of monthly dissolved phosphorus (A) and particulate phosphorus (B) concentrations in the south basin of Lake Manitoba from 1996 to 2007.

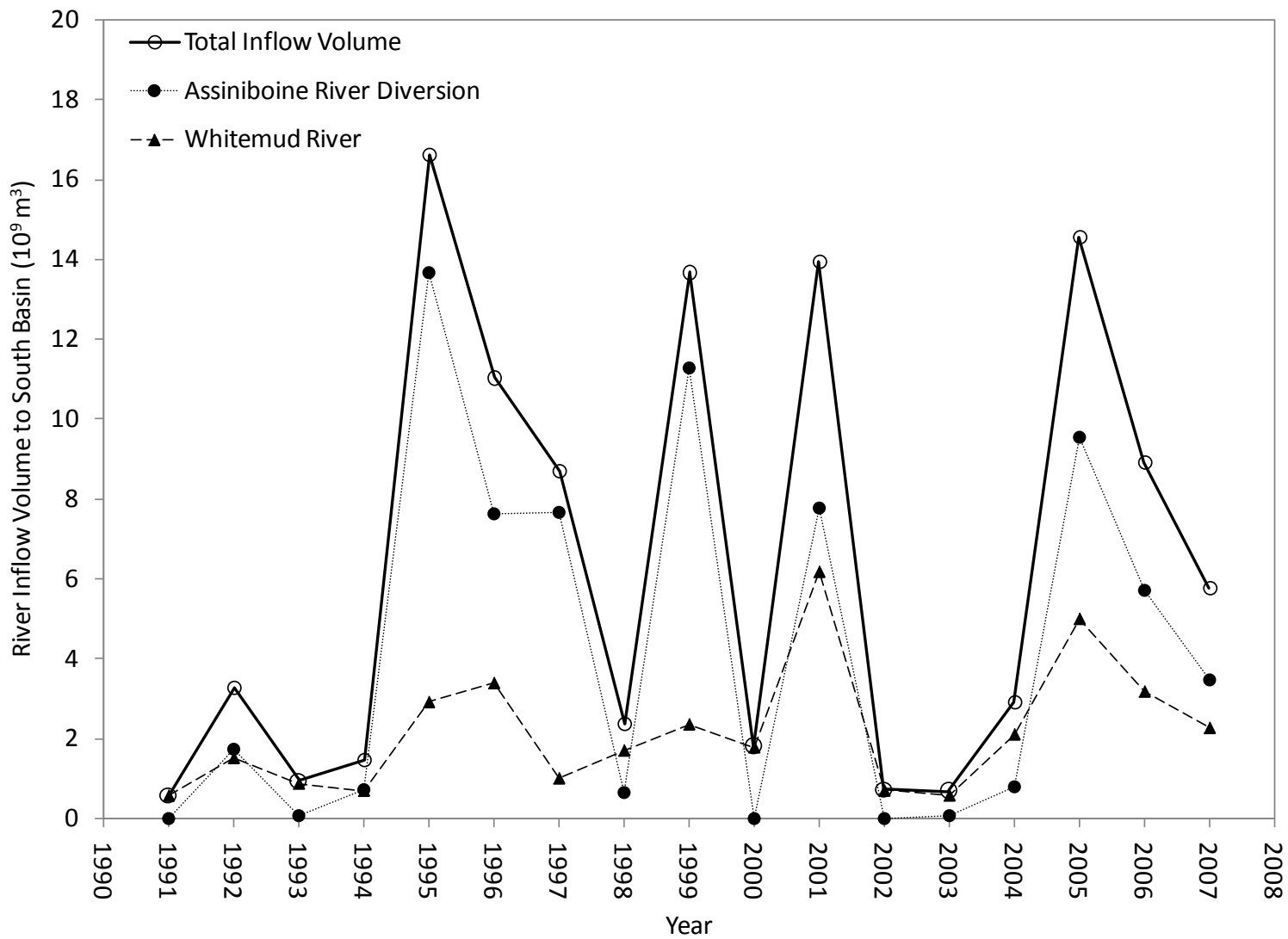


Figure 5.4. Annual river inflow volume to the south basin of Lake Manitoba and the relative contribution of the Assiniboine River Diversion and the Whitemud River, 1991 to 2007.

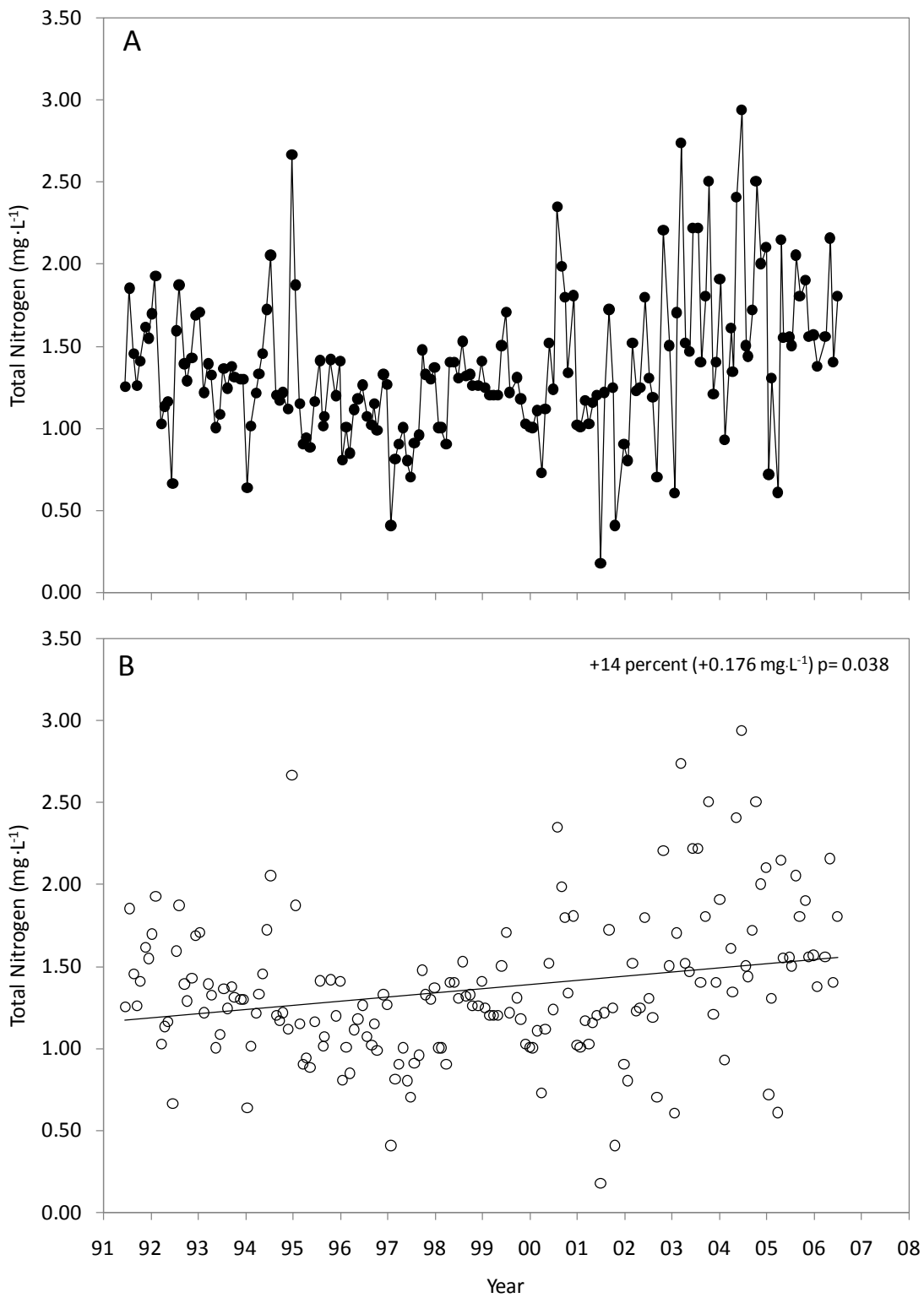


Figure 5.5. Time series of monthly total nitrogen concentrations (A) and trend in total nitrogen concentrations (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

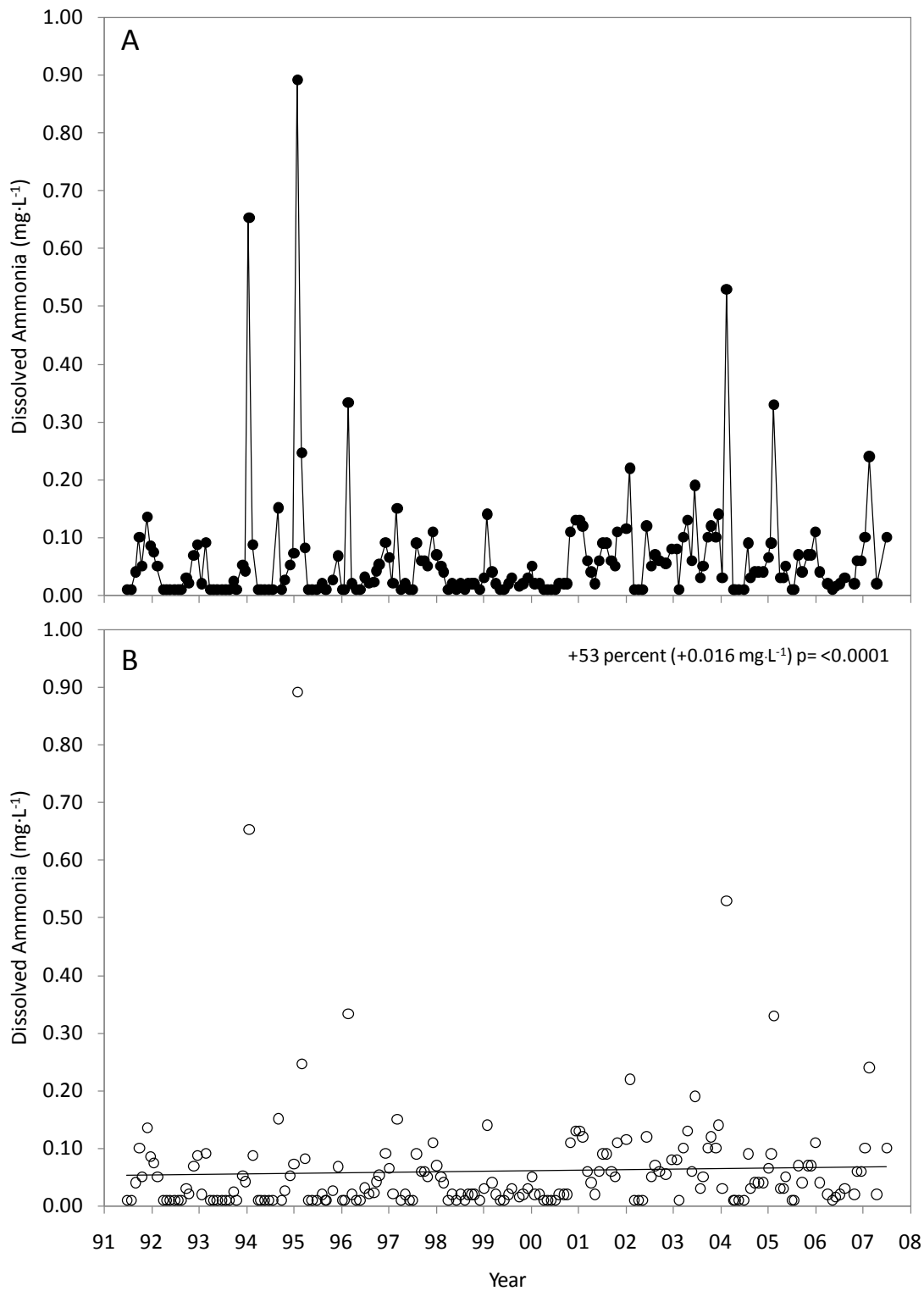


Figure 5.6. Time series of monthly dissolved ammonia concentrations (A) and trend in dissolved ammonia residuals (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

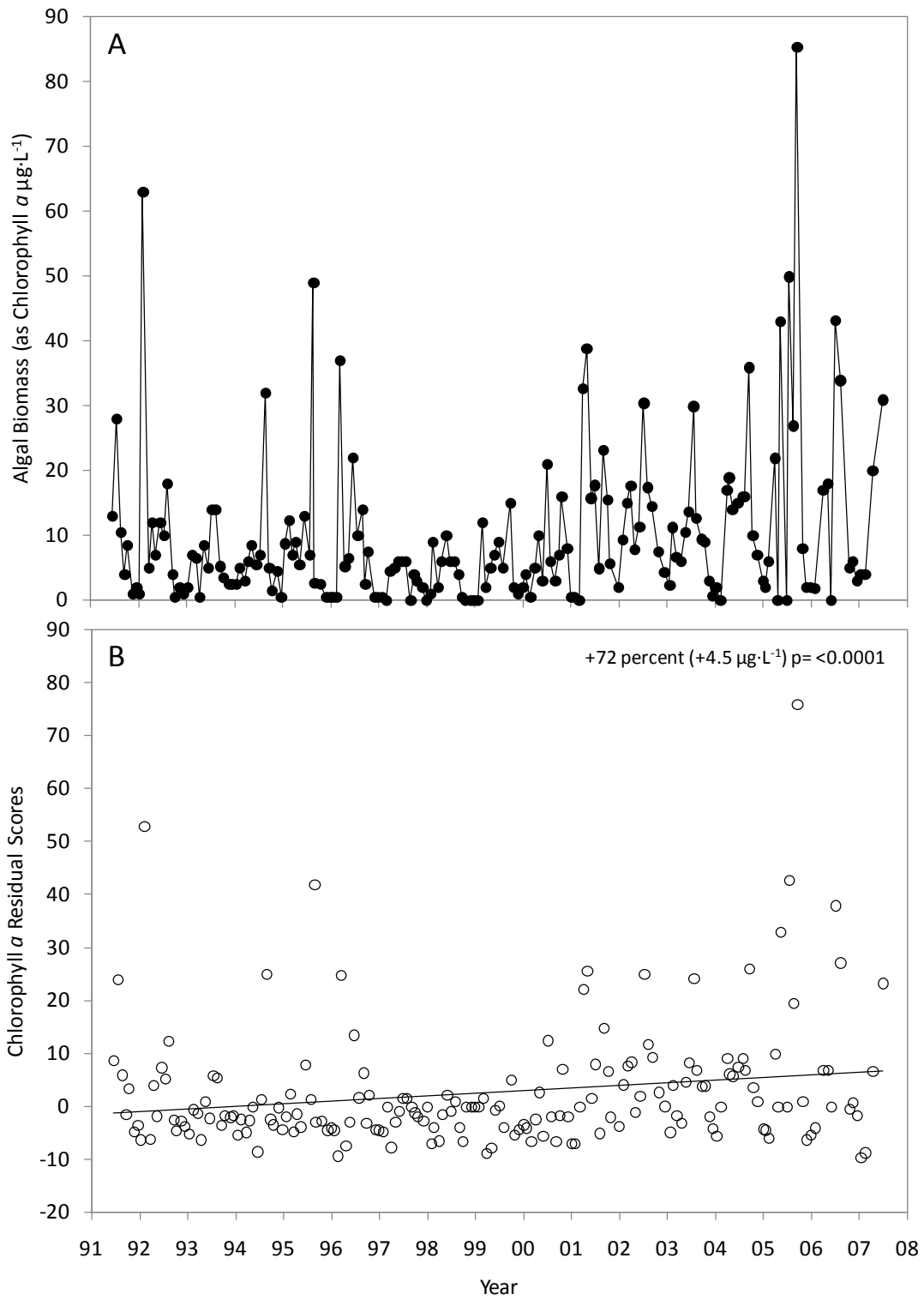


Figure 5.7. Time series of monthly algal biomass (estimated as chlorophyll a) (A) and trend in volume-adjusted chlorophyll a residuals (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

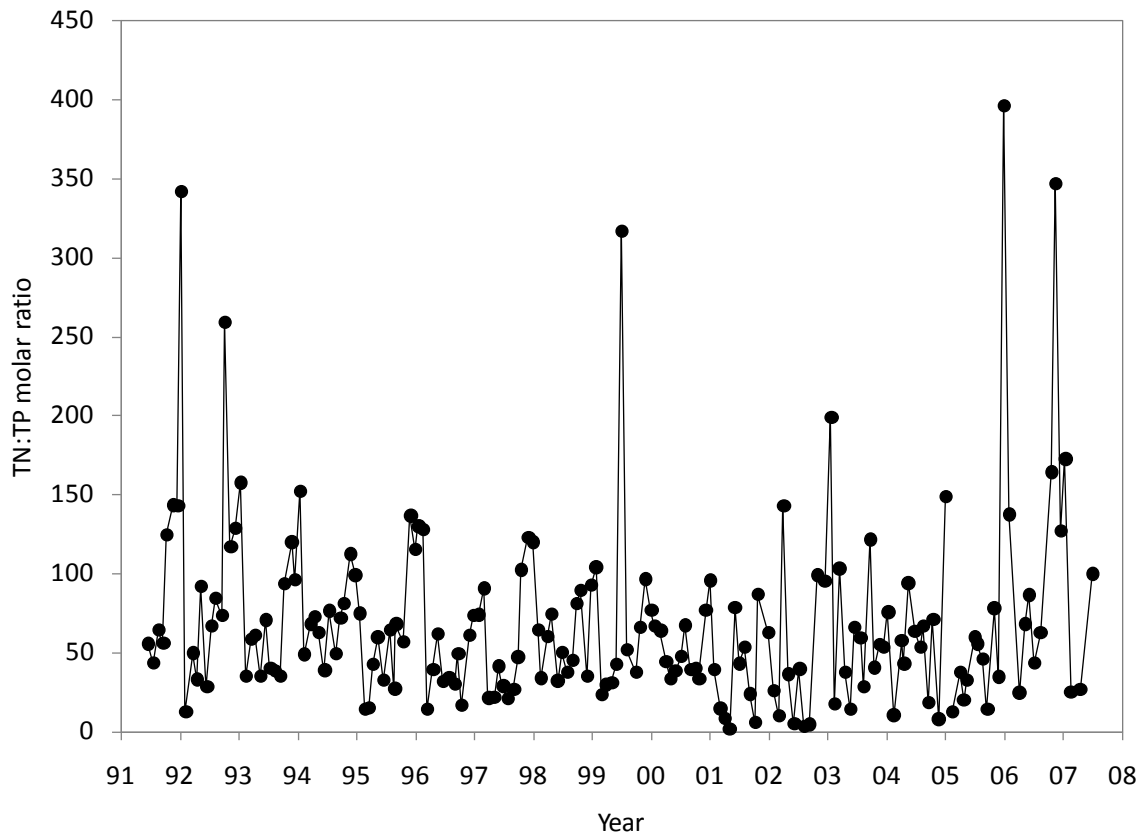


Figure 5.8. Time series of the monthly TN:TP molar ratio in the south basin of Lake Manitoba from 1991 to 2007.

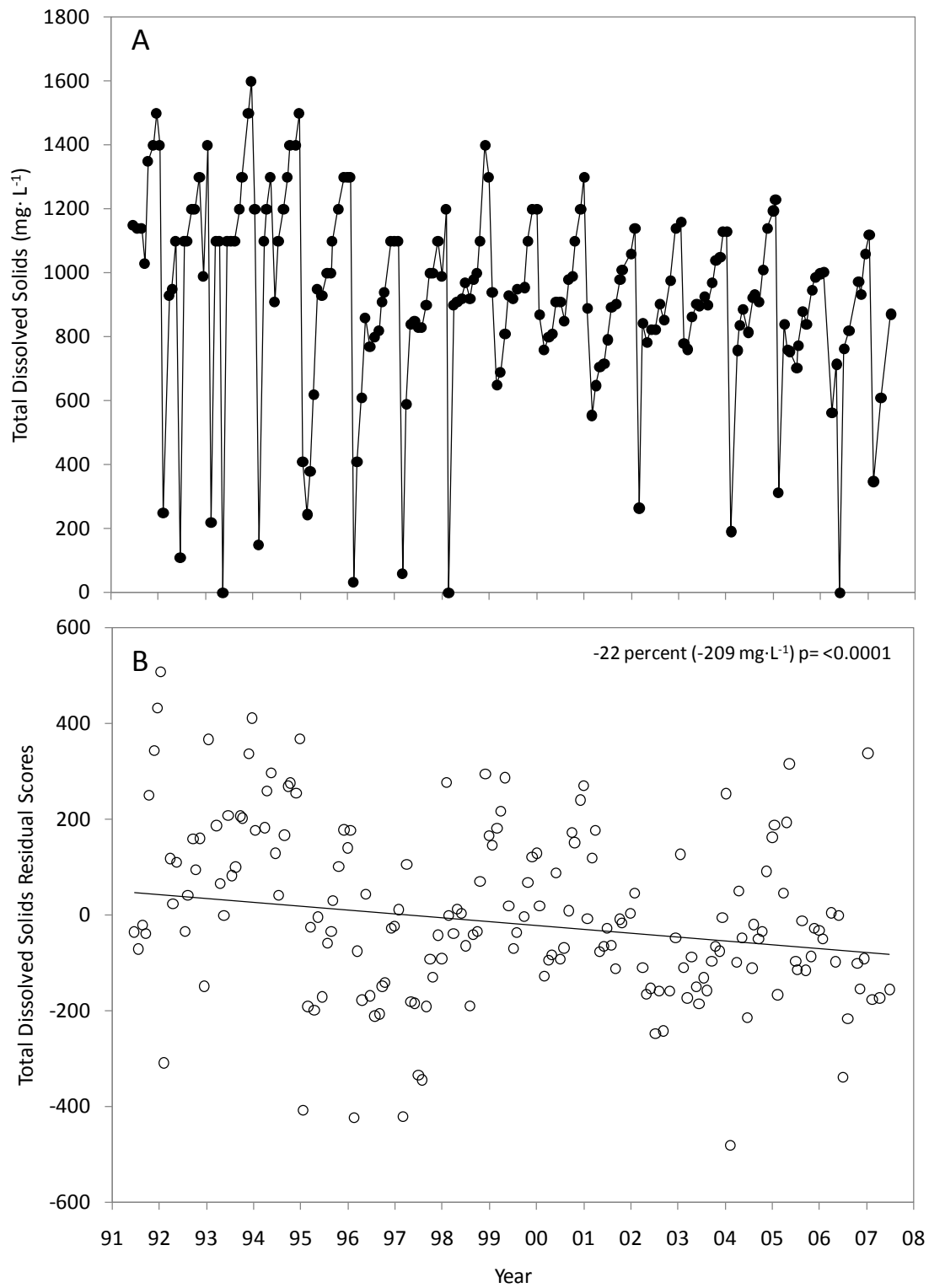


Figure 5.9. Time series of monthly total dissolved solids concentrations (A) and trend in volume-adjusted total dissolved solids residuals (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

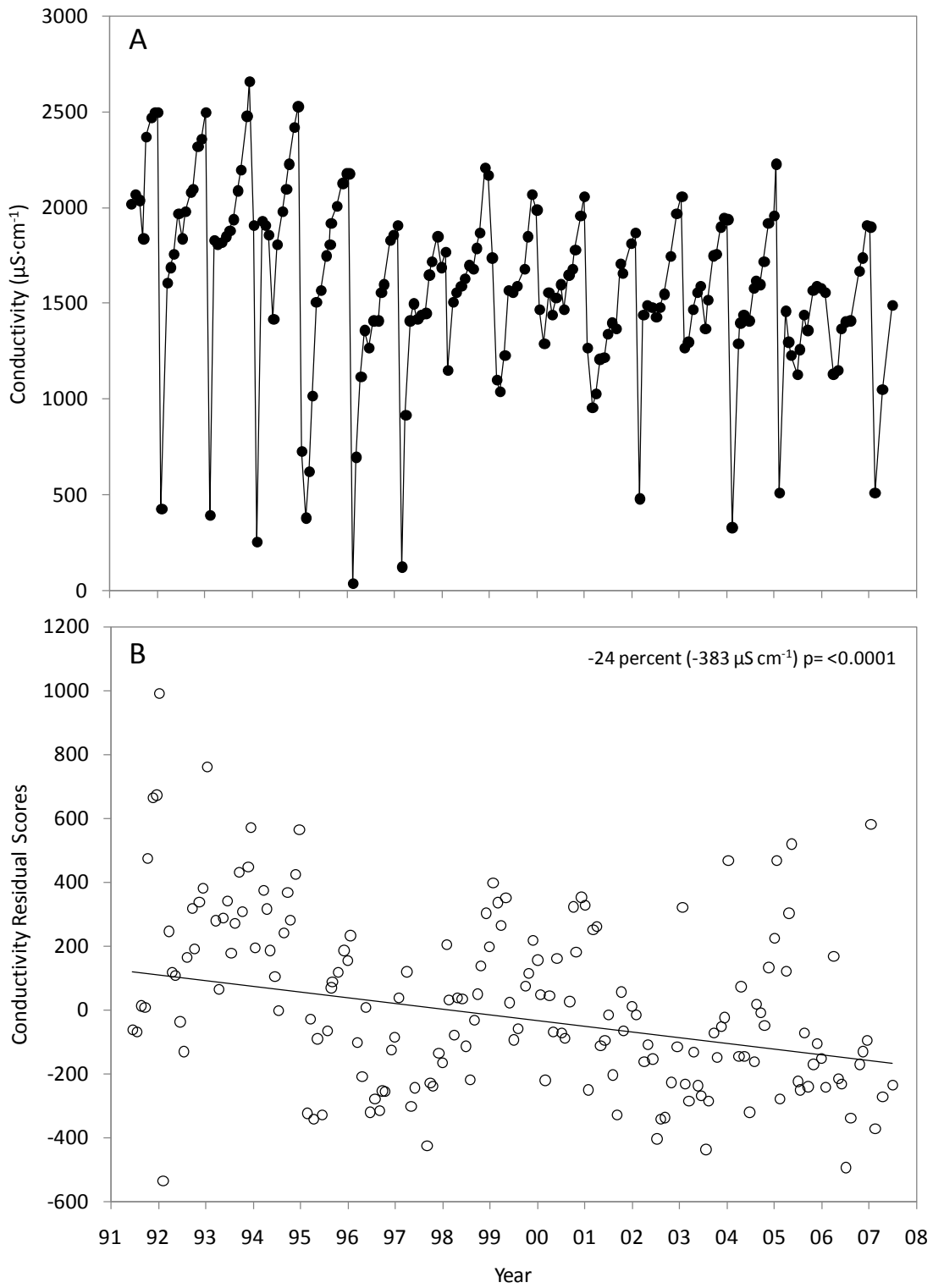


Figure 5.10. Time series of monthly conductivity measurements (A) and trend in volume-adjusted conductivity residuals (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

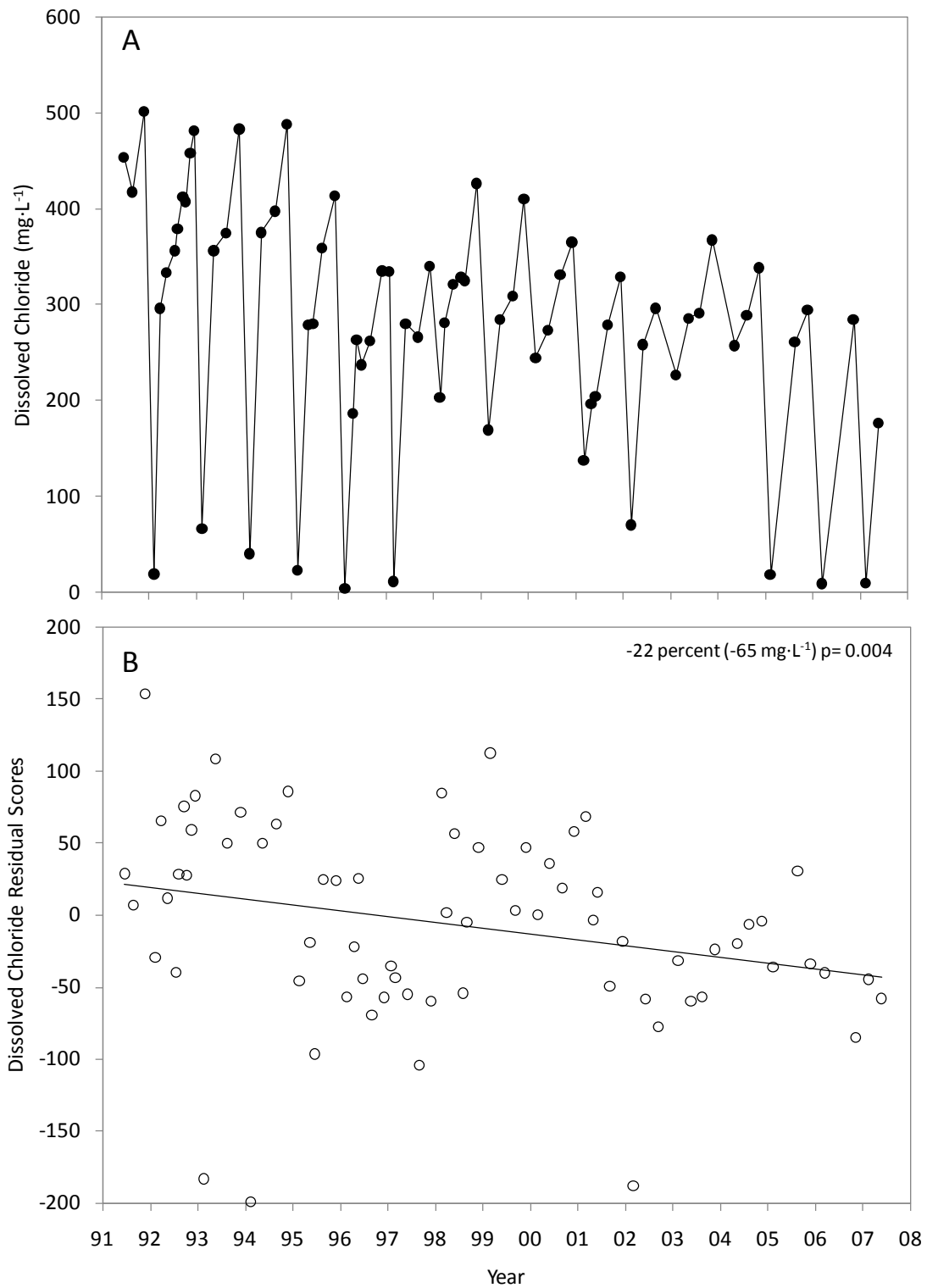


Figure 5.11. Time series of monthly chloride concentrations (A) and trend in volume-adjusted chloride residuals (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

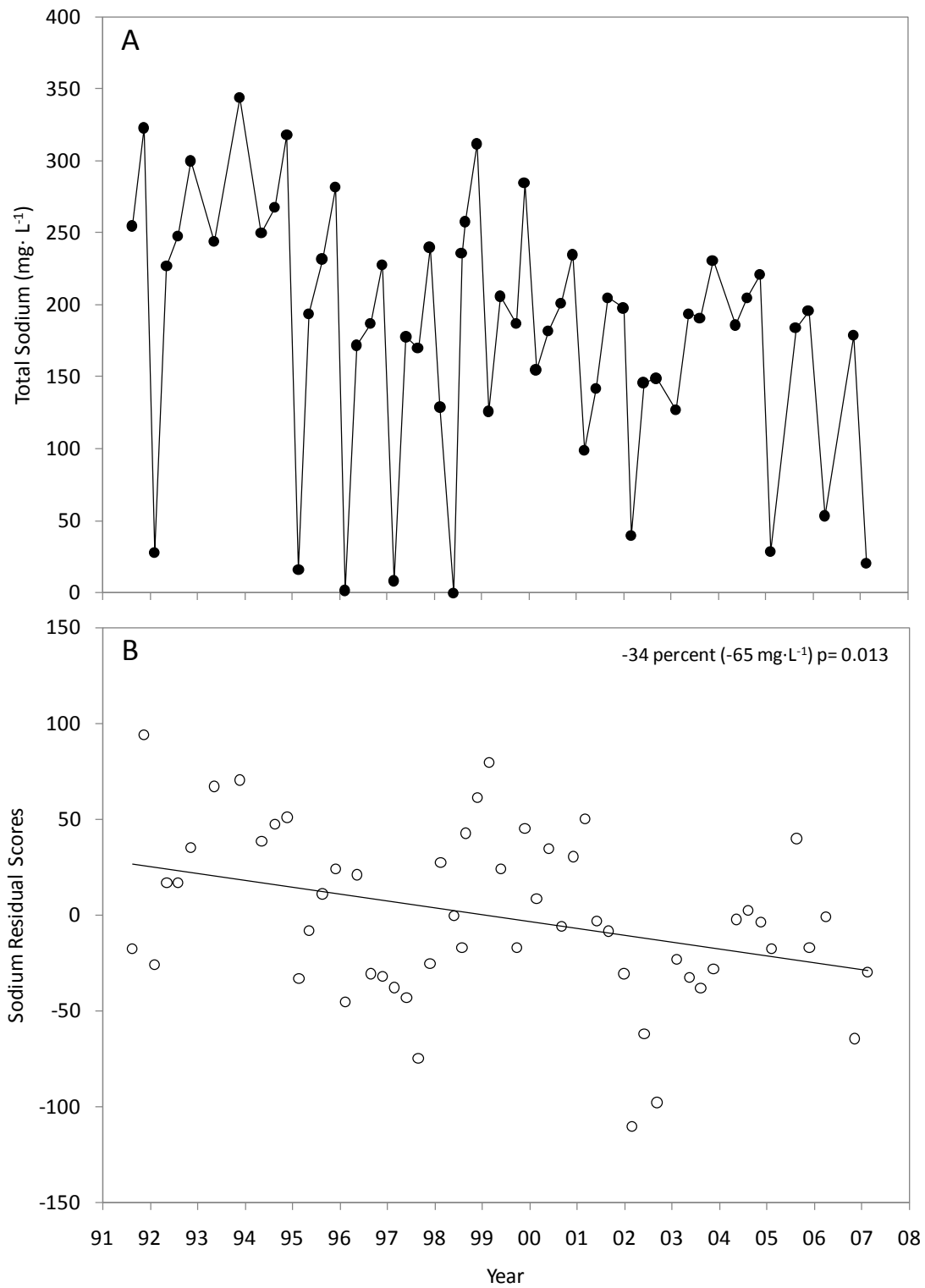


Figure 5.12. Time series of monthly total sodium concentrations (A) and trend in volume-adjusted sodium residuals (B) for station MB05LKS013 in the south basin of Lake Manitoba from 1991 to 2007. Solid trend line represents the seasonal Kendall slope.

6.0 RECOMMENDATIONS

6.1 *Research & Science Needs*

Detailed nitrogen and phosphorus budgets for Lake Manitoba are required to fully understand nutrient dynamics in Lake Manitoba. Nutrient loads have been calculated for the Assiniboine River Diversion, the Whitemud River, and the Waterhen River and loads exported from Lake Manitoba via the Fairford River may be calculated using available flow and water chemistry data. However, several key components of the nutrient budget remain unresolved. It is unknown to what extent internal nutrient loading contributes to the overall nutrient budget of Lake Manitoba. Internal loading may represent a considerable nutrient source in shallow lakes that are well mixed by wind. The atmospheric nutrient contribution is also unknown and may also contribute substantially to the nutrient budget given the large surface area of the lake. Nutrient inputs from groundwater and nitrogen inputs from nitrogen fixation also need to be determined. A complete nutrient budget for Lake Manitoba will identify the greatest sources of nutrients to the lake and help identify where load reductions will have the greatest impacts.

Analysis of historical changes in water quality using satellite imagery and paleolimnological techniques should be undertaken. There currently only exists one continuous long-term water quality record for Lake Manitoba and it is unknown as to how water quality has changed in other areas of the lake. It would be valuable to develop numerical algorithms for chlorophyll a using satellite imagery. Satellite imagery would extend our spatial and temporal understanding of water quality changes in Lake Manitoba. Analysis of several cores throughout the lake would give

an indication as to how water quality has changed over longer time scales across the lake.

Analysis of the factors affecting algal biomass in Lake Manitoba should be undertaken. While nutrient ratios indicate that the lake is generally phosphorus limited nutrient enrichment bioassays would provide information on which nutrient (N, P treatments added alone) or combinations of nutrients (N+P combined) stimulates algal growth in Lake Manitoba. This should be carried out at many locations across all seasons to gain a better understanding of the full spatial and temporal variation in nutrient limitation. A more comprehensive analysis of nutrient limitation in Lake Manitoba would benefit from measurements of nitrogen debt and alkaline phosphatase activity in association with bioassay experiments.

6.2 Recommendations for Management

Establish long-term water quality monitoring stations at several locations to better characterize water quality changes in Lake Manitoba. Current monitoring infrastructure does not capture the heterogeneous water quality conditions in Lake Manitoba. Currently, the lake is monitored on a monthly basis at Delta and on a quarterly basis at the Narrows. One additional station should be added halfway up the south basin in the vicinity of the Lunder or Amaranth stations given the clear north-south gradients in light, chlorophyll, and phosphorus in the south basin. Two stations should be added in the north basin based on the distinct water chemistry in the west and east arms of the basin. Because of the challenges associated with working on such a large windswept lake, a coordinated monitoring program with dedicated resources will be required to maintain these stations. The Province and

the University of Manitoba should continue to work cooperatively to maintain the water quality station at the University of Manitoba Delta Marsh Field Station. This water quality record is very important because it represents the most temporally intensive and the longest continuous water quality record of any lake in the Province of Manitoba.

Review the current operation of the Assiniboine River Diversion and explore ways to minimize the impacts on Lake Manitoba. The Assiniboine River Diversion contributed significantly to the phosphorus, sediment, and to a lesser extent, nitrogen loads transported to Lake Manitoba and is also lowering the salinity of the lake. A review of the operating rules is warranted and it is recommended that a review is undertaken to determine if there are ways to minimize nutrient and sediment loading to the lake. Resources should also be dedicated to strengthening and building new dykes along flood prone areas of the Assiniboine River in effort to minimize the use of the Diversion.

A targeted wetland restoration program in the Assiniboine River watershed is recommended. Land use has changed considerably since widespread European settlement and a significant percentage of wetlands have been drained in Manitoba. These wetlands would have otherwise stored water and nutrients during floods but now water moves quickly downstream to major waterways such as the Assiniboine River. Wetland restoration in the Assiniboine River watershed could reduce downstream flooding and potentially minimize the operation of the Assiniboine River Diversion. Hydrological models could be used to optimize wetland restoration by identifying areas that would maximize water and nutrient retention potential. Such an

initiative would require significant financial incentives to make considerable progress. However, the benefits of flood protection and the added value of nutrient sequestration would be beneficial to the whole watershed and the Province of Manitoba.

7.0 CONCLUSIONS

Objective 1: To determine the spatial and temporal variation of physical and chemical water quality including water clarity, nutrients, ions, and chlorophyll a in Lake Manitoba. These data represent the first spatially intensive assessment of water quality in Lake Manitoba. Research indicated that the south basin of the lake was generally more productive, turbid, less phosphorus limited, and more dilute in comparison to the north basin of the lake. The north basin was less productive and clearer, highly phosphorus limited and more saline. Although the lake is shallow and generally well mixed, water quality was very spatially heterogeneous in the lake. Clear latitudinal gradients existed for many water quality variables in the south basin. It is clear that these gradients were related to the nutrient rich and less saline inflows to the south basin. Resuspension during high wind events also likely explains the spatial heterogeneity in the south basin. North basin chemistry appeared to be affected by the convoluted basin morphometry. The east arm and the west arm of the north basin showed clear differences for certain water quality parameters suggesting that mixing between the basins may be limited.

Objective 2: To characterize the nutrient and sediment loading transported by the Assiniboine River Diversion to Lake Manitoba. Water transported by the Assiniboine River Diversion was much more enriched in nitrogen, phosphorus, and sediment in comparison to Lake Manitoba. Diversion water was also considerably more dilute with lower conductivity, and very low sodium and chloride concentrations as compared to the lake. Nutrient and sediment loads calculated for the Waterhen, Whitemud, and Assiniboine River Diversion indicated that the Diversion contributed

the greatest amount of phosphorus and sediment to the lake in 2005 and 2006. Nitrogen loading from the Diversion was also considerable although the Waterhen River was the greatest source of nitrogen to Lake Manitoba.

Objective 3: To determine to what extent water quality has changed from 1991 to 2007 at a single historical water quality monitoring station in the south basin of Lake Manitoba. Trend analysis of the long-term water quality record in the south basin indicated that Lake Manitoba has experienced considerable water quality changes over the 16 year record. Increases in total nitrogen, total phosphorus, and chlorophyll suggest that the south basin of Lake Manitoba is becoming increasingly eutrophic. Major ion chemistry and salinity changes have also occurred over the same period and the south basin of the lake is becoming less saline. These water quality changes are partly explained by the high nutrient and sediment loads and dilute waters transported by Assiniboine River Diversion into the south basin of Lake Manitoba. It is likely that the Diversion has the greatest effect on water quality given that the Diversion is the greatest riverine source of nutrients to Lake Manitoba and considering the proximity of the water quality site to the outflow of the Diversion. Nutrients have also increased the Whitemud River, and may also partly explain nutrient increases in the lake. The N:P ratio appears to be declining in the lake, although the trend was not significant. Further increases in phosphorus loading to the south basin may lower the N:P ratio and favour cyanobacteria dominance in Lake Manitoba.

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Appendix 1. DVD with raw water quality data collected from Lake Manitoba in 2005 and 2006.

Appendix 2. Summary statistics of water chemistry collected during the ice cover season of 2005, 2006, and 2007 from water quality sites (n= 9 to 11) the north and south basins of Lake Manitoba (Data Source: Manitoba Water Stewardship).

Variable	Units	Mean	Min	Max	SD	SE	n
Conductivity	µS/cm	1729	788	2180	272	51	29
Dissolved Ammonia	mg/L	0.148	0.040	0.430	0.112	0.021	29
Nitrogen Dissolved NO ₃ & NO ₂	mg/L	0.093	0.010	0.540	0.123	0.023	29
Total Nitrogen	mg/L	1.70	0.78	2.73	0.44	0.08	29
Particulate Phosphorus	mg/L	0.012	0.004	0.022	0.005	0.001	29
Total Phosphorus	mg/L	0.025	0.014	0.070	0.010	0.002	29
Dissolved Phosphorus	mg/L	0.012	0.004	0.048	0.008	0.001	29
TN:TP	molar ratio	178	41	377	87	16	29

Appendix 3. Results of sediment particle size analysis for Lake Manitoba, summer 2005.

Site	Weight of Sample (g)	40 Sec. 152H Hydrometer Reading (Mg m ⁻³)	40 Sec. 151H Hydrometer Reading (Mg m ⁻³)	2 hr. 152H Hydrometer Reading (Mg m ⁻³)	2 hr. 151H Hydrometer Reading (Mg m ⁻³)	Silt and Clay (g)	Sand (g)	Silt (g)	Clay (g)	% Sand	% Silt	% Clay	Classification
Delta	55.868	6.5	1.0065	5	1.005	10.439	45.429	2.4091	8.0303	81	4	14	Sandy Loam
Lynch's	56.235	5.5	1.0055	5	1.005	8.8333	47.402	0.803	8.0303	84	1	14	Sandy Loam
Amaranth	69.7	6	1.006	5	1.005	9.6364	60.064	1.6061	8.0303	86	2	12	Loamy Sand
Marg. Bruce	63.09	7	1.007	4.5	1.0045	11.242	51.848	4.0152	7.2273	82	6	11	Loamy Sand
Elm Point	58.766	7	1.007	5	1.005	11.242	47.524	3.2121	8.0303	81	5	14	Sandy Loam
Oak Point	63.12	23	1.023	7	1.007	36.939	26.181	25.697	11.242	41	41	18	Loam
Lundar	54.975	13.5	1.0135	6.5	1.0065	21.682	33.293	11.242	10.439	61	20	19	Sandy Loam
St. Ambrose	32.145	13	1.013	5.5	1.0055	20.879	11.266	12.045	8.8333	35	37	27	Clay Loam
Watchorn	60.68	10	1.01	5.5	1.0055	16.061	44.619	7.2273	8.8333	74	12	15	Sandy Loam
Steep Rock	64.39	6	1.006	5.5	1.0055	9.6364	54.754	0.803	8.8333	85	1	14	Loamy Sand
Davies Bay	52.062	20	1.02	8	1.008	32.121	19.941	19.273	12.848	38	37	25	Loam
Guynemer	55.328	14.5	1.0145	5.5	1.0055	23.288	32.04	14.455	8.8333	58	26	16	Sandy Loam
Spence	50.221	18	1.018	7	1.007	28.909	21.312	17.667	11.242	42	35	22	Loam