The Effects of Increased Rainfall Precipitation on Nutrient Levels in Surface Water in Southern Manitoba

By

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Abstract

Nitrogen and phosphorus loading has become a significant political and public issue in Manitoba and has been accelerated by anthropogenic activity. These nutrients are most readily assimilated as nitrate and orthophosphate. Few data exist regarding changes in nutrient input during seasons of high precipitation. The present study compared nitrate-N, orthophosphate, total dissolved solids (TDS) and dissolved organic matter (DOM) in flooding and non-flooding seasons in surface water in southern Manitoba. Nitrate-N, orthophosphate, and TDS showed significant differences between non-flood and flood seasons. Orthophosphate and TDS were significantly affected by primary adjacent land use and water body type for the non-flood seasons and the importance of nitrate-N and TDS increased during flooding. Orthophosphate to nitrate-N ratios showed differences among land use types, that became less obvious as water flows and leaching accelerated.

Acknowledgments

I would first like to thank Dr. Eva Pip for providing me with the opportunity to make a contribution to the environment. Her dedication and enthusiasm on nutrient enrichment issues were inspiring. A sense of humour and guidance from Dr. Adkins and Dr. Blair provided reassurance throughout the process. I would like to thank Dr. Moodie for his support throughout the year. Thank you to Weldon Hiebert for assistance in producing the map of sample sites in southern Manitoba. Finally I would like to acknowledge my family and friends for their love and support.

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Introduction

Nutrient loading has become a significant political and public issue in Manitoba, particularly during the last three decades (LWSB, 2005; LWIC, 2005). The consequences of elevated nutrient loading have become increasingly visible, and have followed in the footsteps of livestock expansion, altered agricultural cropping practices, burgeoning recreational development, and effluent from industries such as pulp and paper mills, wide-spread clear-cutting of forests, and continued discharge of municipal and rural sewage effluents into surface waters (Pip, 2005; Carpenter, *et al*., 1998). Since most of southern Manitoba lies within the Lake Winnipeg watershed, much of the nutrient load imposed on streams and ditches is eventually received by Lake Winnipeg (Jones and Armstrong, 2002). This lake has shown increasing signs of eutrophication, as evidenced by large algal mats, and dense algal blooms (Appendix I-1).

Dense algal growth reduces the photic zone, and shades out macrophytes below, reducing oxygenation through photosynthesis. As a result macrophytes die, and oxygen is further depleted through decomposition. Low concentration of dissolved oxygen may cause fish and other sensitive aquatic animals to die, reducing the biodiversity (Carpenter, *et al*., 1998; Mainstone and Parr, 2002; de Jonge, *et al*., 2002; Pip, 2005). For example, five of the eleven native freshwater mussel species in Lake Winnipeg have disappeared in the last three decades (Pip, 2006). Many cyanobacteria or blue-green algae can fix nitrogen under both aerobic and anaerobic conditions. Cyanobacteria proliferate in warm water (30-35ºC) and high pH (8.0-9.5) (Prescott, *et al.*, 2002). Cyanobacteria limit the growth of eukaryotic algae and compete for nutrients by producing compounds such as siderochrome hydroxamates that bind to essential trace elements such as iron, and render them unavailable for eukaryotic algae or macrophytes (Prescott, *et al.*, 2002). Blue-green algae produce odoriferous compounds such as geosmin which may reduce the aesthetic value of a particular area and impact on property values. When cyanobacteria die, toxins are released which may be lethal to humans and livestock if ingested or may cause contact dermatitis through swimming or bathing (de Jonge, *et* al., 2002; Carpenter, *et al.*, 1998). These toxins include neurotoxins, hepatotoxins, and lipopolysaccharide toxins, all of which have no antidote. A Manitoba Government study of public water supplies found detectible levels of microcystin (a hepatotoxin) in two-thirds of supplies tested (Jones, 1999, Jones, *et al.*, 1998).

Eutrophication is a natural aging process in surface waters, but can be accelerated through the addition of nutrients by anthropogenic activities. Where there is an influx of nutrients, rapid proliferation of bacteria, algae and macrophytes occurs (Jones and Armstrong, 2001; Prescott, *et al.*, 2002). Under normal conditions, nutrients are assimilated by primary producers in the ratio of 1P: 7N: 40C per 100 grams dry weight (Vallentyne, 1974). The primary nutrients of concern with regard to eutrophication are phosphorus (P) and nitrogen (N). Both P and N occur in a variety forms in the aquatic environment, however the most readily assimilated forms are orthophosphate and nitrate respectively. While organic compounds may contain N and P, nutrients in this form are largely unavailable until they are released by decomposition. The latter process depends on pH, temperature, oxygenation and many other factors, as well as on the type of compound involved. Refractory organic matter may take decades or even centuries to decompose. Thus measurements of total N and P are poor indicators of primary production acceleration, as significant amounts of the total N and P may be unavailable for immediate assimilation. In a study of a eutrophic Canadian Shield lake, Levine *et al.* (1986) found that radio-orthophosphate was incorporated into bacteria and microphytoplankton within moments of its introduction into the water, while dissolved organic P formed only a minor component and much of it appeared to be "functionally inert".

Phosphorus is an essential element required for the synthesis of phospholipids, phosphoproteins, nucleic acids and ATP energy transformation reactions, as well as for the proper formation of bones and teeth in mammals. Orthophosphates are ionic forms of phosphate (PO_4^{3-}, $H\,PO_4^{2-}$, $H_2\,PO_4^{-}$) which are not bound to particulate matter (Jones and Armstrong, 2001). Orthophosphate is the only form of soluble inorganic phosphate that can be absorbed by plants. Some plants absorb excess phosphate through luxury consumption and store it in compounds such as inositol hexaphosphate. Phosphorus, which does not naturally circulate in the atmosphere, is returned to water or soil through excretion by animals or through the decomposition of organic matter by bacteria and fungi (Appendix I-2). Phosphorus may also enter the aquatic environment through leaching of phosphate in the soil, weathering of rocks or through run off from human activity (Campbell, *et al.*, 1999; Mainstone and Parr, 2002). Orthophosphate in drinking water has been linked to liver and kidney ailments, yet despite this the City of Winnipeg adds orthophosphate to the water distribution system to help mitigate high lead leaching. The city has no nutrient removal step in its wastewater treatment process.

Earth's atmosphere is made up of approximately 78% nitrogen gas (N_2) (Draper, 2002). Nitrification is an oxidation process which converts inert atmospheric nitrogen into ammonia (NH_3), nitrite (NO_2^-) and eventually nitrate (NO_3^-) which can be assimilated by plants (Appendix I-3) (Campbell, *et al.*, 1999; Scholz and Trepel, 2004). High levels of nitrogen can not only help to accelerate the eutrophication process, but may pose a significant health risk to humans, mammals and aquatic vertebrate and invertebrates. Nitrate and its interconversion form nitrite, may cause a potentially fatal condition known as methemoglobinemia or "blue baby syndrome" in infants under the age of 6 months. Methemoglobinemia can also cause abortions in cattle. Nitrate may be converted to nitrosamine in the presence of gastric juices in the stomach. Nitrosamine is a carcinogen, and may cause diarrhoea, kidney failure, and sudden infant death syndrome (Carpenter, *et al.* 1998).

Total dissolved solids (TDS) and dissolved organic matter (DOM) are also important indicators of water quality. TDS reflects the amount of leaching and DOM can come from natural sources such as leaf litter and water from bogs, or be associated with pollution from human activity.

Quantitative data on nutrient loading in southern Manitoba are limited (LWIC, 2005). Water quality surveys in Manitoba carried out from 1975 onwards (e.g. Pip, 1979, 1988) show a wide range of nutrients, TDS, and DOM levels in surface waters. Manitoba Conservation made public its nutrient management strategy in 2000, in an attempt to lay the ground work for an understanding of nutrient enrichment issues in Manitoba. Bourne *et al.* (2002) estimated that watershed processes (primarily agricultural activity) contributed the largest share of nutrients to Lake Winnipeg. Specifically, 71% of total N and 76 % of total P in the Assiniboine River originate in this way, while in the Red River, 59 % of total N and 73 % of P can be attributed to these sources. Trend analyses conducted by Jones and Armstrong in 2001 showed that nutrient inputs in many streams in southern Manitoba have been significantly increasing. A large study by Pip (2005) identified a number of anthropogenic nitrogen sources in Manitoba. These included municipal sewage treatment effluents, agricultural run-off from fertilizers on crop land and manure run-off and spills from intensive livestock operations, mining, recreational development, run-off from clear-cutting, and industrial effluents such as pulp and paper mills.

Algal blooms have also become more prominent in the Lake of the Woods region, which further contributes nutrients to Lake Winnipeg via the Winnipeg River that accounts for approximately 40% of the water inflow to Lake Winnipeg. Griffiths (2005) surveyed Lake of the Woods and surrounding lakes for total P between April and June, 2005 and found evidence of significant eutrophication in many areas.

Flooding commonly occurs in southern Manitoba, much of which is largely situated in the Red River floodplain. Surprisingly, however, almost no data exist regarding the effects of flooding on nutrient loadings in this region. According to LWIC (2005), "major nutrient loading events are not well measured because water flows are monitored on a pre-set schedule, and not necessarily at peak weather events." Furthermore, "understanding of phosphorus transport from land to water during snowmelt is limited" and "the processes that temporarily retain phosphorus in rivers, streams and lakes are not well understood."

Following the 1997 “Flood of the Century”, the International Joint Commission (IJC, 2000) examined chemical contamination along the mainstream of the Red River, and found “elevated levels” of nutrients, trace elements and some pesticides particularly toxaphane. Similarly Pip (2006) reported elevated nitrate, TDS, DOM and trace elements in Lake Winnipeg in 1998, following the 1997 flood.

Natural flooding may occur as a result of one or more of the following weather conditions: (1) excessive precipitation in the fall, (2) hard, penetrating frost prior to snowfall, (3) significant amounts of snowfall, (4) rapid spring thaw and (5) large amounts of precipitation during ice break-up (Simonovic, 1999). The flood-pulse concept describes the hydrological cycle in river-floodplain ecosystems (Appendix I-4). Nutrients present in the soil prior to flooding become dissolved, and are carried with the flood water. Conversely, flood water may encroach on higher land and transport dissolved nutrients from the main channel to inundated portions of the floodplain during high water. During flooding, primary production is maximized and decomposition rates increase. As water levels begin to recede, decomposition rates increase compared to production rates, and dissolved oxygen concentrations decrease. Increase in biomass occurs during drawdown due the increase of nutrients from run-off and concentration of nutrients as flood waters recede. The biota of river-floodplain ecosystems have adapted to a rapidly changing environment, which is reflected by high annual growth and mortality rates and short life cycles. Flooding is a natural process that contributes to the biodiversity of the ecosystem (Bayley, 1995; Michener and Haeuber, 1998).

The objectives of this study were to determine whether nutrient levels in southern Manitoba differed under non-flood conditions compared to flooding conditions. The study considered the effects of primary adjacent land uses and water body type on nitrate-N, orthophosphate, TDS and DOM during different precipitation conditions. Since flood conditions involve inundation of larger land areas, and flow conditions are increased, it was hypothesized that a season of higher than normal rainfall would be associated with greater levels of dissolved nutrients and salts.

Study Area

Surface water samples were collected between July 19th and September 2nd 2005. This was a season of higher than normal rainfall precipitation. The total precipitation in the Winnipeg region (49º55N/97º13W) from January 2005 until the end of the sampling season was 547.2 mm. The total average precipitation in 1998 and 2001, the non-flood comparison years used in the present study, was 465.7 mm in the same region during the same interval (Environment Canada, 2006) (Figure 1). A total of 106 samples were collected throughout southern Manitoba (N 49º-50º, W 95º-99º) (Figure 2). Sampling sites were selected randomly, and recorded using a global positioning satellite (GPS) system. The sites were classified according to water body type (Table I A), and primary adjacent land use (Table I B) (Appendix III).

Water samples for the 1998 and 2001 non-flood seasons were obtained from a frozen reference collection for a survey conducted by Dr. E. Pip. From this collection, 105 samples were selected for comparison based on similarity of GPS coordinates as closest matches to the 2005 samples (Tables I A and B)

(Appendix IV). However 37% of the sites were the same in both sets of samples. Two different non-flood years were utilized to provide a broader comparison. Primary adjacent land use categories differed slightly from the 1998 and 2001 non-flood seasons compared to the 2005 flood season. Thus the 1998 and 2001 non-flood season samples included some logging and hydro-electric categories but not mining (i.e. gravel pits) and industrial categories. The 2005 flood season samples included mining and industrial categories, but not logging and hydro-electric categories. However, each of these categories comprised less than 5% of the sampling total.

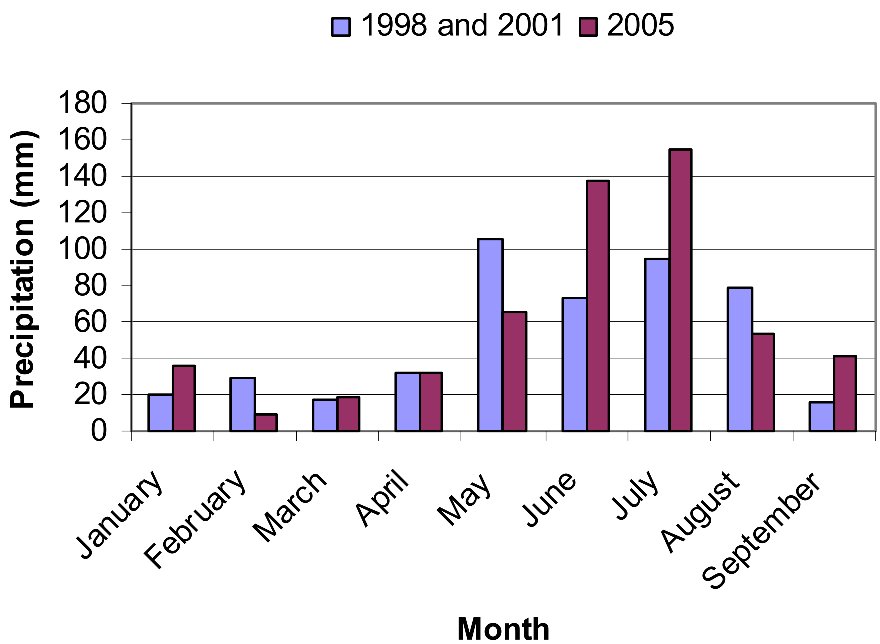


Figure 1: Precipitation data for the Winnipeg airport region (49º55N/97º13W). The precipitation during the 2005 sampling season (July-September) showed higher levels of precipitation compared to the average precipitation from the 1998 and 2001 non-flood seasons (Environment Canada, 2006).

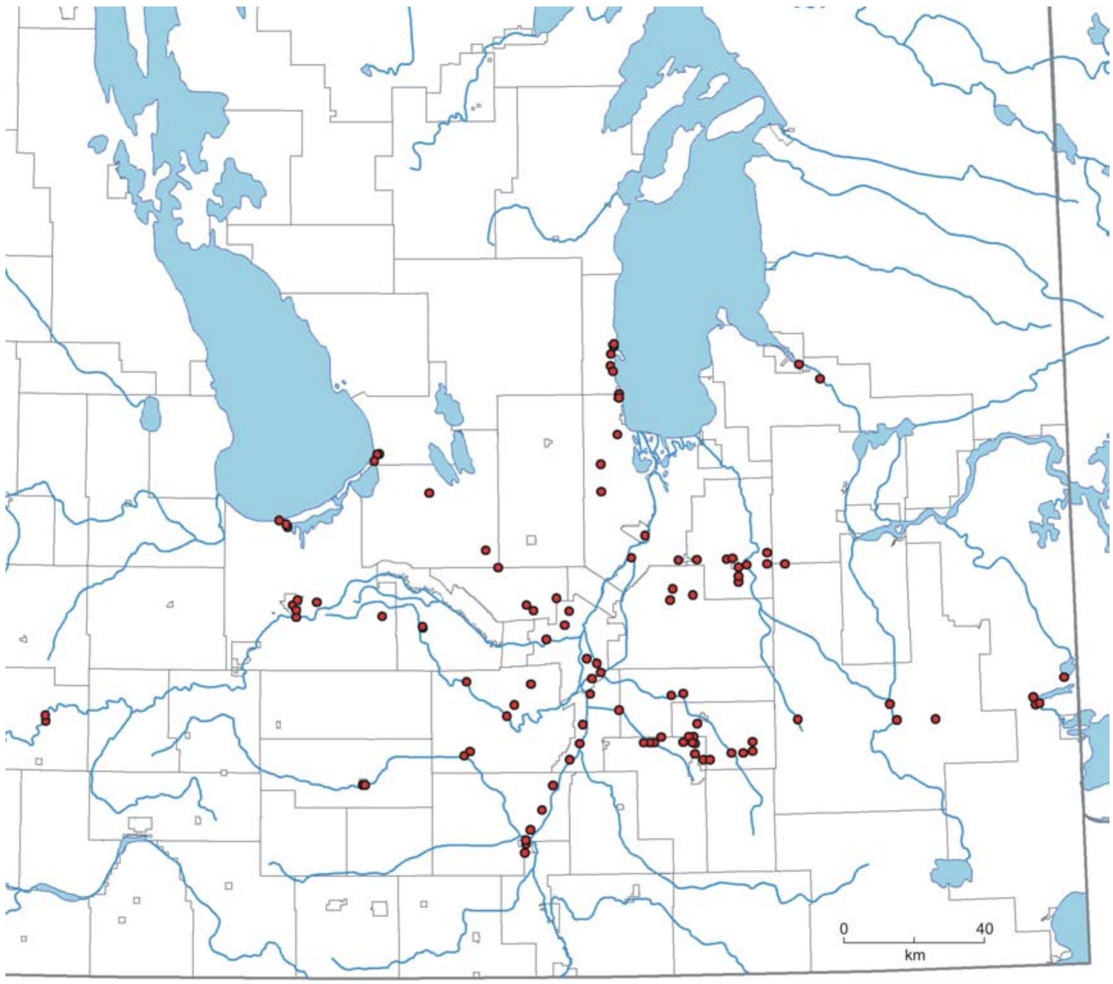


Figure 2: Sample sites for the 2005 flood season. Samples were collected in southern Manitoba (49º-50º N, 95º-99º W) and were recorded using a GPS system.

Table I A: Percentage distribution of water samples from non-flood (1998 and 2001) and flood (2005) seasons classified according to water body type.

Water Body Type	1998 and 2001 (N=105)	2005 (N=106)
Lake	23.8	10.4
River	30.5	17.9
Stream	30.5	34.0
Ditch/Pond	15.2	37.7

Table I B: Percentage distribution of water samples from non-flood (1998 and 2001) and flood (2005) seasons classified according to primary adjacent land use categories.

Adjacent Land Use	1998 and 2001 (N=105)	2005 (N=106)
Urban	13.3	18.9
Recreation/cottages	28.6	10.4
Agricultural crops	32.4	33.0
Livestock/poultry production	11.4	32.1
Mining	0	0.9
Industrial	0	2.8
Minimal	5.7	1.9
Logging	3.8	0
Hydro-electric	4.8	0

Materials and Methods

Water samples were collected 10 cm below the surface in single-use high density polypropylene bottles, and immediately placed in a cooler with ice. All of the water samples were frozen (-20ºC) within twelve hours. Samples were thawed at 4ºC, within 24 hours prior to testing.

Total dissolved solids (TDS) were measured using a portable TDS Testr 1 calibrated for 1-2000 mg/L (Oakton, Wards Natural Science, St. Catherines Ontario). Water samples exceeding this maximum value were analysed by evaporation at 80ºC
(APHA, 1995). Nitrate-N and dissolved organic matter (DOM) were measured using the ultraviolet screening method (APHA, 1995), using a Beckman DU-7 spectrophotometer (Beckman Instruments, Irvine, California). The samples were acidified with 1 N HCl (50:1) and absorbance was measured at 220 nm and 275 nm. Calibration curves yielded two linear regression models for different portions of the regression line. For absorbance values between 0 and 0.4, the model was ($r = 0.99$, $p < 0.0001$):

$$\boxed{\text{mg/L } NO_3\text{-N} = 11.0 \times (\text{absorbance}) + 0.009}$$

For samples with absorbance values greater than 0.4 the linear regression model was ($r = 0.99$, $p < 0.0001$):

$$\boxed{\text{mg/L } NO_3\text{-N} = 12.9 \times (\text{absorbance}) + 0.038}$$

Calculations for the unknowns utilized the appropriate model for maximum accuracy.

Nitrate-N, TDS and DOM values for the 1998 and 2001 non-flood seasons were obtained from the database from the survey by Pip (2005).

Orthophosphate was measured using the ascorbic acid-ammonium molybdate spectrophotometric method (APHA, 1995) using a Beckman DU-7 spectrophotometer. The linear regression model ($r = 0.99$, $p < 0.00001$) for orthophosphate was:

$$\boxed{\text{mg/L orthophosphate} = 1.61 \times (\text{absorbance}) - 0.008}$$

While molybdenum blue methods are in standard use, it is possible that some phosphorus in the colloidal fraction may inflate the dissolved orthophosphate value, yielding overestimates in organic-rich samples (Stainton, 1980). Data analysis was conducted using SPSS (Chicago, Illinois). The critical significance level for all statistical tests was $p = 0.05$. For unpaired student t-tests, group variances were pre-tested using F tests to determine whether separate or pooled variances were appropriate. Pearson correlation coefficients were calculated using untransformed data. Multivariate analysis of variance (MANOVA) utilized a non-nesting design with water chemistry parameters as dependent variables.

Results

Means, standard errors and ranges for the water chemistry parameters for the 1998 and 2001 non-flood seasons and the 2005 flood season can be found in tables II A and B respectively. Overall, higher mean concentrations were observed for water chemistry parameters in the 2005 flood season compared to the 1998 and 2001 non-flood seasons.

Unpaired t-tests indicated significant differences between non-flood and flood seasons for orthophosphate and TDS. Due to the presence of an outlier (site 55), the t-test was conducted with and without the outlier for comparison (Table III A and B). Site 55 was a ditch bordering an Agricore depot containing a large ammonia tank. When site 55 was omitted, significant differences were found between non-flooding and flood seasons for nitrate-N as well as for orthophosphate and TDS.

The water chemistry parameters showed a number of significant intercorrelations (Table IV). Nitrate-N, orthophosphate, and TDS were significantly positively correlated in both flood and non-flood seasons; however correlations were much more intense in 2005. DOM and orthophosphate were significantly correlated only in 2005.

Water Body Type

The highest mean nitrate-N concentration for the 1998 and 2001 non-flood seasons was 0.55 mg/L ± 0.17 S.E. in streams, and the lowest was 0.31 mg/L ± 0.07 S.E. in ditches and ponds (Figure 3 A). The highest mean nitrate-N concentration in the 2005 flood season was 10.38 mg/L ± 9.22 S.E. in ditches and ponds and the lowest was 0.57 mg/L ± 0.09 S.E. in lakes (Figure 3 B). However, ANOVA did not show significant differences in mean nitrate-N concentration among water types for either the 1998 and 2001 non-flood seasons or the 2005 flood season. Duncan multiple range tests did not indicate any significant differences in nitrate-N with regard to water body type for the 1998 and 2001 non-flood season nor for the 2005 flood season. T-tests showed that the 1998 and 2001 non-flood seasons and the 2005 flood season were highly significantly different ($t = 5.06$, $p < 0.0001$) in rivers with regard to nitrate-N (Appendix II A).

The highest mean orthophosphate concentration for the 1998 and 2001 non-flood seasons was 0.15 mg/L ± 0.04 S.E. in streams, and the lowest was 0.06 mg/L ± 0.03 S.E. in ditches and ponds (Figure 4 A). Duncan multiple range tests indicated that mean orthophosphate concentrations were not significantly different between water body types for the 1998 and 2001 non-flood season. The highest mean orthophosphate concentration for the 2005 flood season was 0.32 mg/L ± 0.06 S.E. in ditches and ponds, and the lowest was 0.09 mg/L ± 0.03 S.E. in lakes (Figure 4 B). Duncan multiple range tests for the 2005 flood season, indicated that the mean orthophosphate concentration in ditches and ponds was marginally different ($p = 0.06$) from the mean orthophosphate concentrations from lakes, rivers and streams. Lakes, for both the 1998 and 2001 non-flood seasons and the 2005 flood season had a mean orthophosphate concentration of 0.09 mg/L ± 0.03 S.E. ANOVA showed that, mean orthophosphate concentrations were only marginally different ($F = 2.58$, $p = 0.06$) for the 2005 flood season among water body types. T-tests indicated that the 1998 and 2001 non-flood seasons and the 2005 flood season were significantly different in rivers ($t = 2.17$, $p = 0.035$) and

Table II A: Summary of water chemistry parameters for 105 sites in southern Manitoba during non-flood seasons (1998 and 2001).

Variable	Mean ± S.E	Range
Nitrate-N (mg/L)	0.41 ± 0.06	< 0.01-6.0
Orthophosphate (mg/L)	0.10 ± 0.02	< 0.01-1.0
TDS (mg/L)	280.95 ± 35.35	10-3520
Dissolved Organic Matter (275 nm)	0.31 ± 0.03	< 0.01-2.0

Table II B: Summary of water chemistry parameters for 106 sites in southern Manitoba during the 2005 flood season.

Variable	Mean ± S.E	Range
Nitrate-N (mg/L)	4.41 ± 3.48	< 0.01-370
Orthophosphate (mg/L)	0.21 ± 0.03	< 0.01-1.0
TDS (mg/L)	532.88 ± 59.57	20-4950
Dissolved Organic Matter (275 nm)	0.36 ± 0.03	< 0.01-2.0

Table III A: T-test comparison of water chemistry parameters between 1998 and 2001 non-flood seasons (group 1) and the 2005 flood season (group 2), including the outlier site 55. NS = Mean concentration was not significantly different between groups.

Parameter	Group	Mean (±S.E)	Significance
Nitrate-N (mg/L)	1	0.41 ± 0.06	NS
	2	4.41 ± 3.48	
Orthophosphate (mg/L)	1	0.01 ± 0.02	t= 3.43, p = 0.001
	2	0.20 ± 0.03	
TDS (mg/L)	1	280.95 ± 35.35	t= 3.64, p< 0.0001
	2	532.88 ± 59.57	
Dissolved Organic Matter (275 nm)	1	0.31 ± 0.03	NS
	2	0.36 ± 0.03	

Table III B: T-test comparison of water chemistry parameters for the 1998 and 2001 non-flood seasons (group 1) and the 2005 flood season (group 2), excluding the outlier site 55.

Parameter	Group	Mean (±S.E)	Significance
Nitrate-N (mg/L)	1	0.41 ± 0.06	t = 4.03, p < 0.0001
	2	0.93 ± 0.12	
Orthophosphate (mg/L)	1	0.10 ± 0.02	t= 3.26, p = 0.001
	2	0.20 ± 0.03	
TDS (mg/L)	1	280.95 ± 35.35	t= 3.79, p< 0.0001
	2	490.81 ± 42.58	
Dissolved Organic Matter (275 nm)	1	0.31 ± 0.03	NS
	2	0.37 ± 0.03	

NS = Mean concentration was not significantly different between groups.

Table IV: Pearson correlation among chemical parameters for the 1998 and 2001 non-flood seasons (N=105) (bottom, left) and the 2005 flood season (N=106) (top, right). * Indicates significant correlation.

Correlations	***Nitrate-N***	***Orthophosphate***	***TDS***	***DOM***
Nitrate-N		r= 0.3385 (p< 0.0001)*	r= 0.7165 (p< 0.0001)*	r= -0.1116 (p= 0.127)
Orthophosphate	r= 0.2505 (p= 0.005)*		r= 0.4173 (p<0.0001)*	r= 0.4344 (p<0.0001)*
TDS	r= 0.2147 (p=0.014)*	r= 0.3717 (p<0.0001)*		r= 0.0433 (p=0.330)
DOM	r= -0.092 (p=0.175)	r= -0.0135 (p=0.446)	r= -0.0996 (p=0.156)	

1. Lake
2. River
3. Stream
4. Ditch/Pond

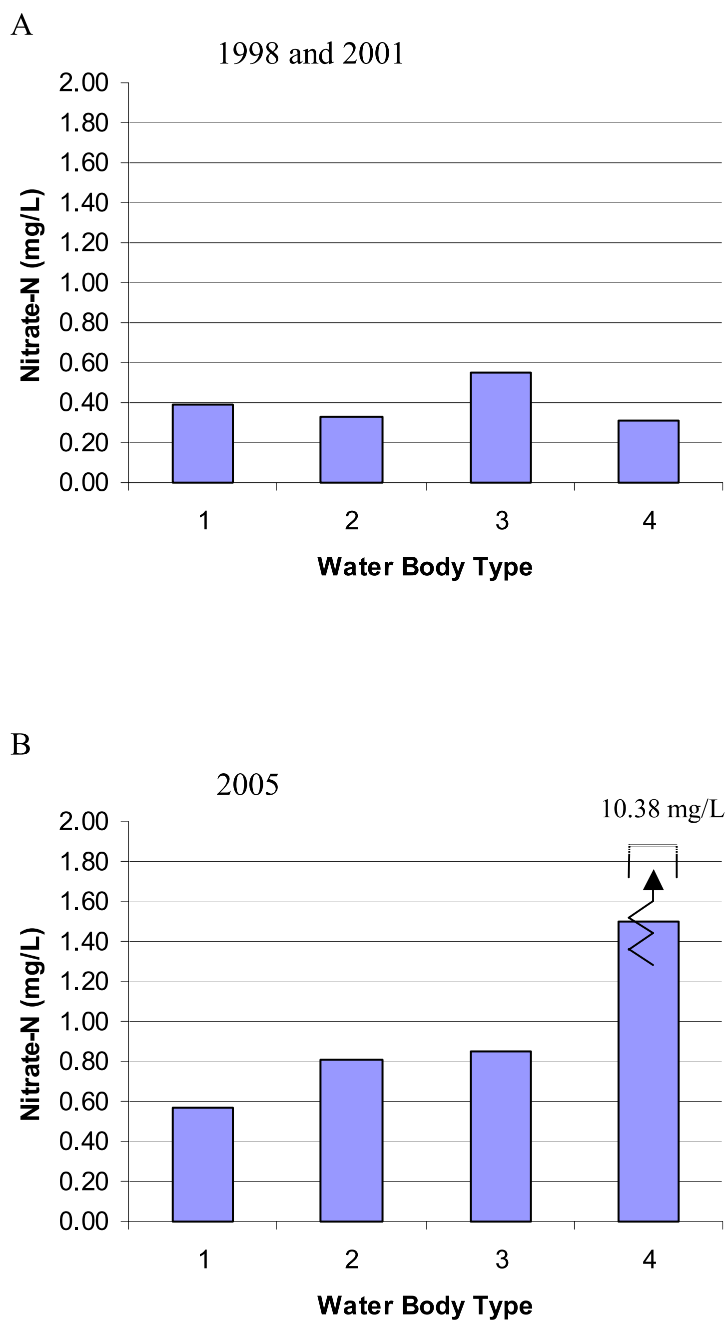
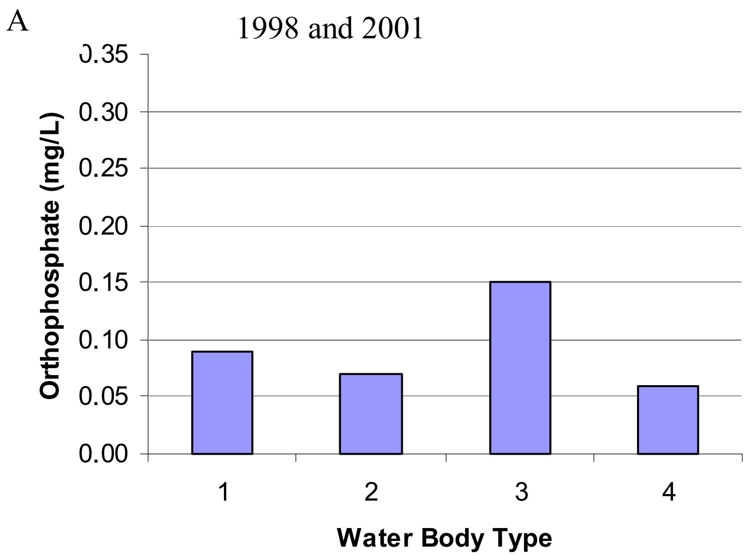


Figure 3: Mean nitrate-N concentration (mg/L) for each water body type for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

1. Lake
2. River
3. Stream
4. Ditch/Pond



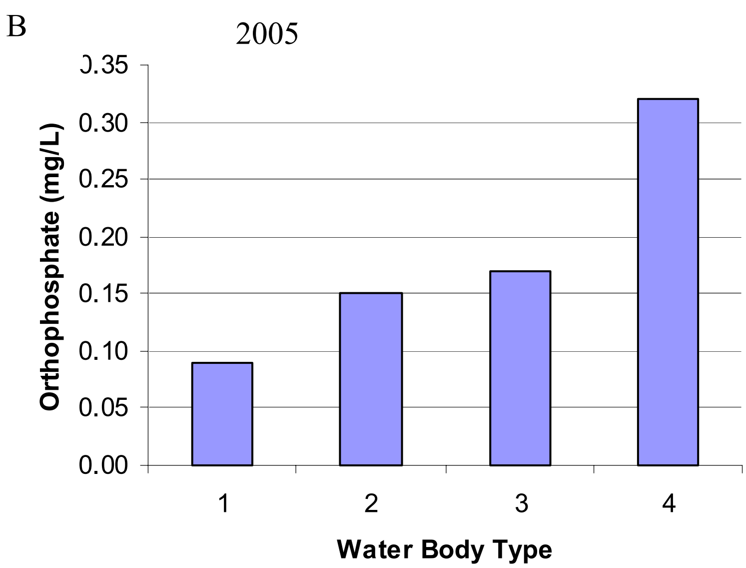


Figure 4: Mean orthophosphate concentration (mg/L) for each water body type for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

ditches and ponds ($t = 2.51$, $p = 0.015$) (Appendix II A) with regard to orthophosphate.

The highest TDS concentration for the 1998 and 2001 non-flood seasons was 396 mg/L ± 136 S.E. in lakes and the lowest was 214 mg/L ± 33 S.E. in rivers (Figure 5 A). Duncan multiple range tests did not indicate any significant differences between water body types for the 1998 and 2001 non-flood season. The highest mean TDS concentration for the 2005 flood season was 786 mg/L ± 136 S.E. in ditches and ponds, and the lowest mean TDS concentration was 288 mg/L ± 73 S.E. in lakes (Figure 5 B). For the 2005 flood-season, Duncan multiple range tests indicated that the mean TDS concentration of 786 mg/L ± 136 S.E. for ditches and ponds was significantly different ($p = 0.007$) from lakes, rivers and streams. ANOVA showed that, mean TDS concentrations were significantly different ($F = 4.26$, $p = 0.007$) among water body types for the 2005 flood season. T-tests indicated that mean concentrations were significantly different between non-flood and flood seasons for streams ($t = 2.37$, $p = 0.021$) (Appendix II A).

The highest mean DOM index for the 1998 and 2001 non-flood seasons was 0.58 ± 0.18 S.E. at 275 nm in ditches and ponds, and the lowest mean DOM index was 0.18 ± 0.02 S.E. at 275 nm in lakes (Figure 6 A). Duncan multiple range tests indicated that for the 1998 and 2001 non-flood seasons, the DOM index in ditches and ponds were significantly different from all other water body types ($p = 0.0011$). The highest mean DOM index for the 2005 flood season was 0.43 ± 0.06 S.E. at 275 nm in ditches and ponds, and the lowest mean DOM index was 0.21 ± 0.03 S.E. at 275 nm in lakes (Figure 6 B). For 2005, Duncan multiple

1. Lake
2. River
3. Stream
4. Ditch/Pond

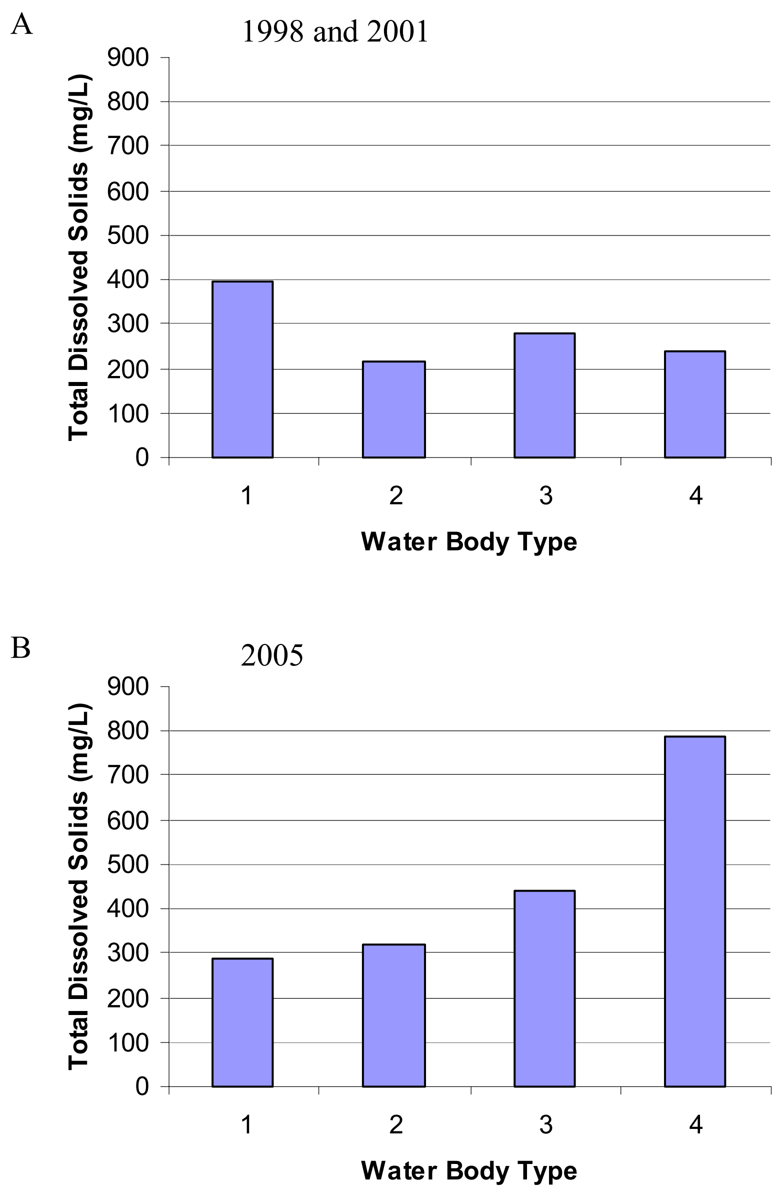


Figure 5: Mean total dissolved solids concentrations (mg/L) for each water body type for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

1. Lake
2. River
3. Stream
4. Ditch/Pond

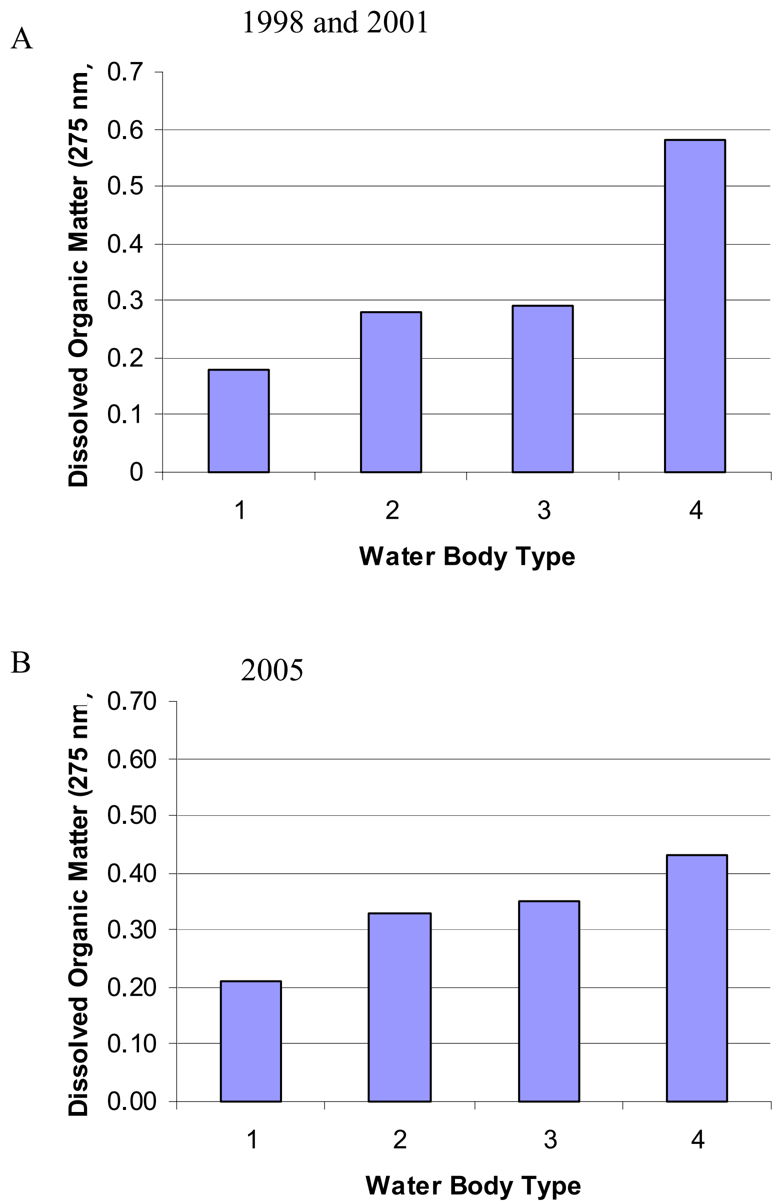


Figure 6: Mean dissolved organic matter index (275 nm) for each water body type for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

range tests showed that the mean DOM index for ditches and ponds was almost significantly different from lakes ($p = 0.0694$). ANOVA showed that mean DOM indices were significantly different among water body type ($F = 5.76$, $p = 0.001$) in the 1998 and 2001 non-flood seasons and only marginally significant ($F = 2.43$, $p = 0.069$) for the 2005 flood season. T-tests did not indicate significant differences between the 1998 and 2001 non-flood seasons and the 2005 flood seasons among water body type with regard to DOM (Appendix II A).

Land Use

The highest nitrate-N concentration for the 1998 and 2001 non-flood seasons was 0.85 mg/L ± 0.37 S.E. in the urban category, and the lowest was 0.07 mg/L ± 0.05 S.E. at minimally impacted sites (Figure 7 A). In 1998 and 2001, Duncan multiple range tests showed that mean nitrate-N concentration of 0.85 mg/L ± 0.37 S.E. for the urban impacted sites was significantly different ($p = 0.04$) from the sites that were minimally impacted, or affected by recreation/cottages, and agricultural crops. The highest mean nitrate-N concentration for the 2005 flood season was 123.8 mg/L ± 3.48 S.E. in the industrial category, and the lowest was 0.42 mg/L ± 0.26 S.E. in the minimally impacted category (Figure 7 B). For 2005, Duncan multiple range tests indicated that the industrial category was highly significantly different ($p < 0.0001$) from all of the other land uses. ANOVA, showed significant differences in mean nitrate-N concentration among adjacent land use categories for the 1998 and 2001 non-flood seasons ($F = 2.27$, $p = 0.043$), but these differences intensified significantly for the 2005 flood season samples ($F = 7.99$, $p < 0.0001$). T-tests showed significant differences in nitrate-N concentrations between 1998 and 2001 non-

1. Urban
2. Recreation/Cottages
3. Agricultural Crops
4. Livestock/Poultry Production
5. Mining
6. Industrial
7. Minimal
8. Logging
9. Hydro-electric

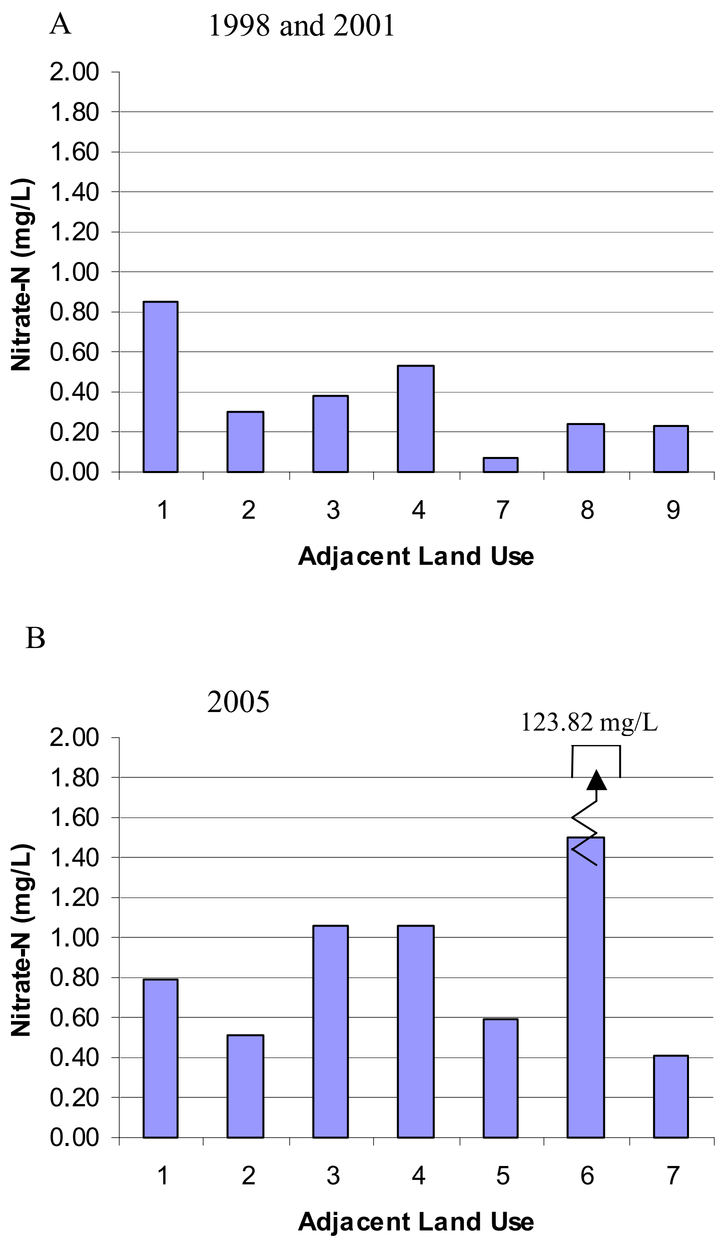


Figure 7: Mean nitrate-N concentrations (mg/L) for each primary adjacent land use for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

flood seasons and the 2005 flood season for recreation/cottages (t = 2.22, p = 0.031) and agricultural crops (t = 2.94, p = 0.005) (Appendix II B).

The highest mean orthophosphate concentration for the 1998 and 2001 non-flood seasons was 0.15 mg/L ± 0.22 S.E. in the urban category, and the lowest was 0.01 mg/L ± 0.02 S.E. in both the logging and hydroelectric categories (Figure 8 A). For the 1998 and 2001 non-flood seasons, Duncan multiple range tests showed that mean orthophosphate concentrations for urban and agricultural crops categories approached a significant different significantly different (p = 0.0511) from recreation/cottages. The highest mean orthophosphate concentration for the 2005 flood season was 0.45 mg/L ± 0.03 S.E. in sites affected by industry, and the lowest was 0.01 ± 0.03 S.E. in sites affected by mining (Figure 8 B). Duncan multiple range tests did not indicate significant differences among land uses for the 2005 flood season. ANOVA found mean orthophosphate concentrations to differ significantly (F = 2.18, p = 0.051) among land uses for the 1998 and 2001 non-flood seasons, but not for the 2005 flood season. T-tests showed no significant differences between the 1998 and 2001 non-flood seasons and the 2005 flood season among primary adjacent land use with regard to orthophosphate (Appendix II B).

The highest TDS concentration for the 1998 and 2001 non-flood seasons was 417 mg/L ± 98 S.E. in sites adjacent to agricultural crops and the lowest was 118 mg/L ± 49 S.E. in the minimally impacted sites (Figure 9 A). Duncan multiple range tests showed no significant differences among land uses for TDS concentrations for the 1998 and 2001 non-flood seasons. The highest mean TDS concentration for the 2005 flood season was 1890 mg/L ± 60 S.E. in the industrial

1. Urban
2. Recreation/Cottages
3. Agricultural Crops
4. Livestock/Poultry Production
5. Mining
6. Industrial
7. Minimal
8. Logging
9. Hydro-electric

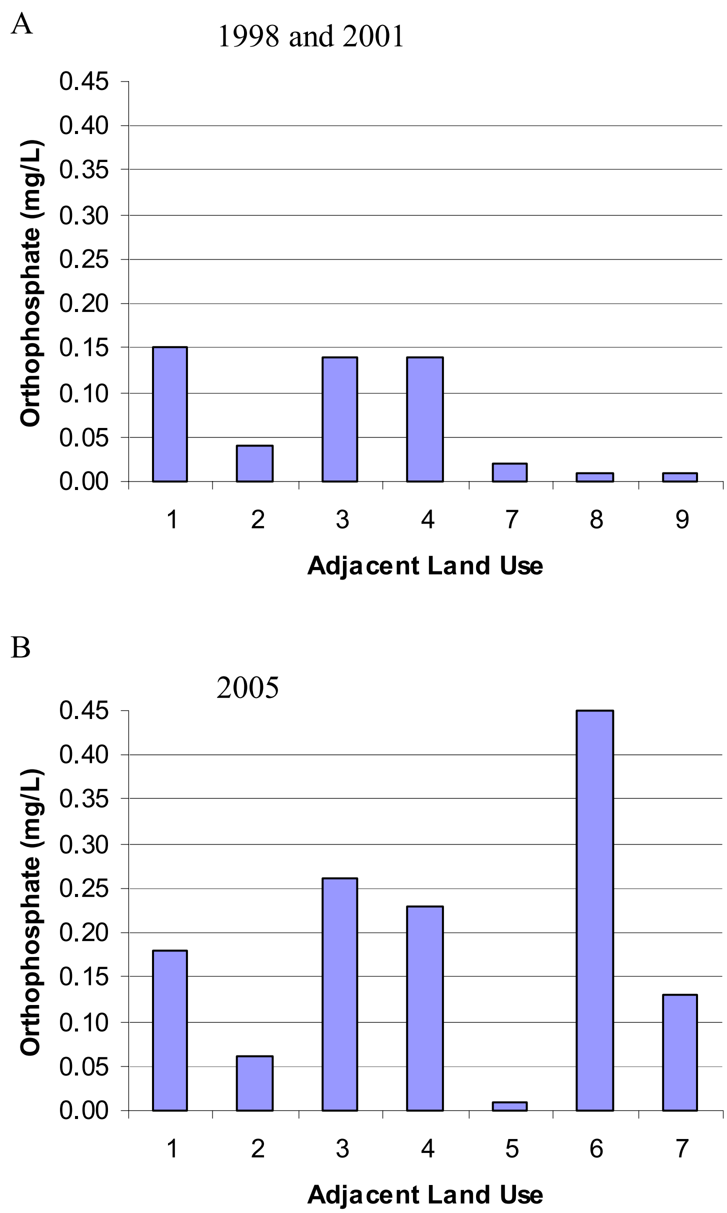
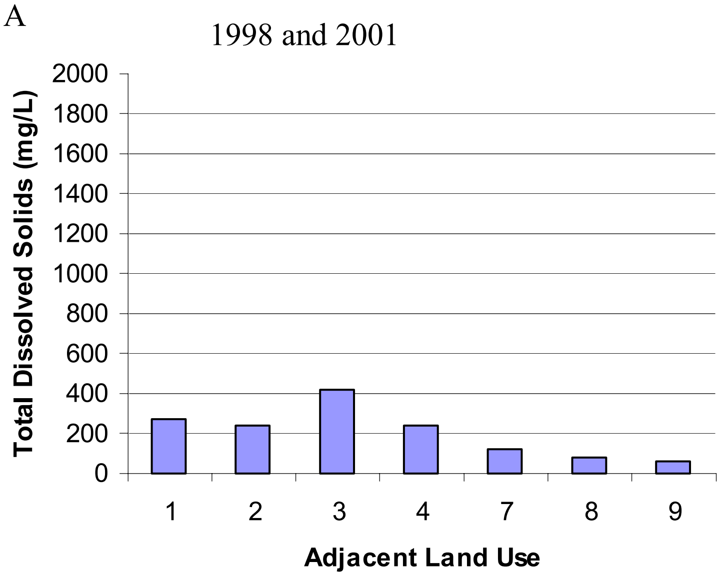


Figure 8: Mean orthophosphate concentrations (mg/L) for each adjacent land use for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

1. Urban
2. Recreation/Cottages
3. Agricultural Crops
4. Livestock/Poultry Production
5. Mining
6. Industrial
7. Minimal
8. Logging
9. Hydro-electric

A



B

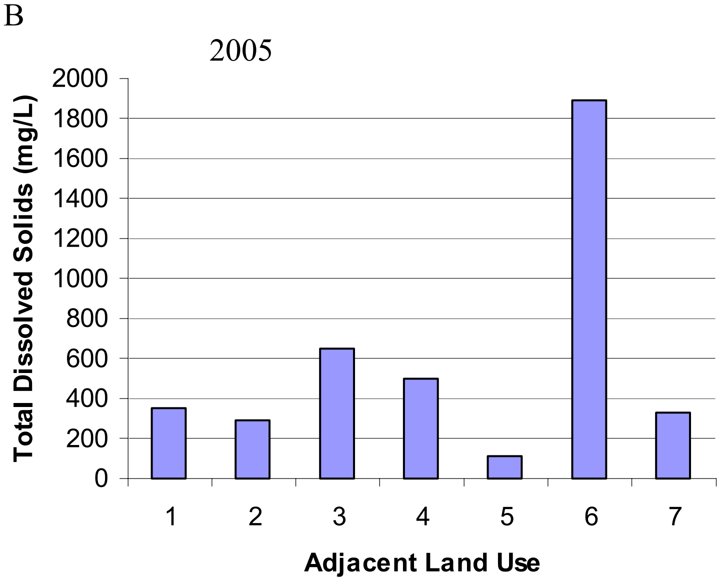


Figure 9: Mean total dissolved solids concentration (mg/L) for each adjacent land use for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

category, and the lowest was 110 mg/L ± 60 S.E. in sites affected by mining (Figure 9 B). Similarly, Duncan multiple range tests indicated that the mean TDS concentration in the industrial category was significantly higher ($p = 0.0013$) than mean TDS concentrations for all other adjacent land uses. ANOVA showed that TDS concentrations were significantly different among land uses ($F = 3.97$, $p = 0.001$) for the 2005 flood season samples. T-tests showed no significant differences in TDS concentrations between non-flood and flood seasons for each primary adjacent land use (Appendix II B).

The highest mean DOM index for the 1998 and 2001 non-flood seasons were 1.3 ± 0.30 S.E. at 275 nm in the minimally impacted sites, and the lowest was 0.14 ± 0.03 S.E. at 275 nm in the hydro-electric category (Figure 10 A). Similarly, Duncan multiple range tests indicated that in the 1998 and 2001 non-flood seasons, DOM for the minimally impacted sites was highly significantly different ($p < 0.0001$) all other primary adjacent land uses, and sites affected by logging were significantly different from sites affected by hydro-electric, urban, recreation/cottages, and agricultural crops activity. The highest mean DOM index in the 2005 flood season was 0.44 ± 0.06 S.E. at 275 nm in the agricultural crops category, and the lowest was 0.13 ± 0.03 S.E. at 275 nm in the mining category (Figure 10 B). For the 2005 flood season, Duncan multiple range tests indicated that surface water affected by livestock/poultry production was marginally different ($p = 0.07$) from urban sites. ANOVA showed that mean DOM indices were highly significantly different among land uses for the 1998 and 2001 non-flood season ($F = 25.67$, $p < 0.0001$). T-tests showed significant differences in DOM indices between non-flood and flood seasons in the urban ($t = 3.62$, $p =$

1. Urban
2. Recreation/Cottages
3. Agricultural Crops
4. Livestock/Poultry Production
5. Mining
6. Industrial
7. Minimal
8. Logging
9. Hydro-electric

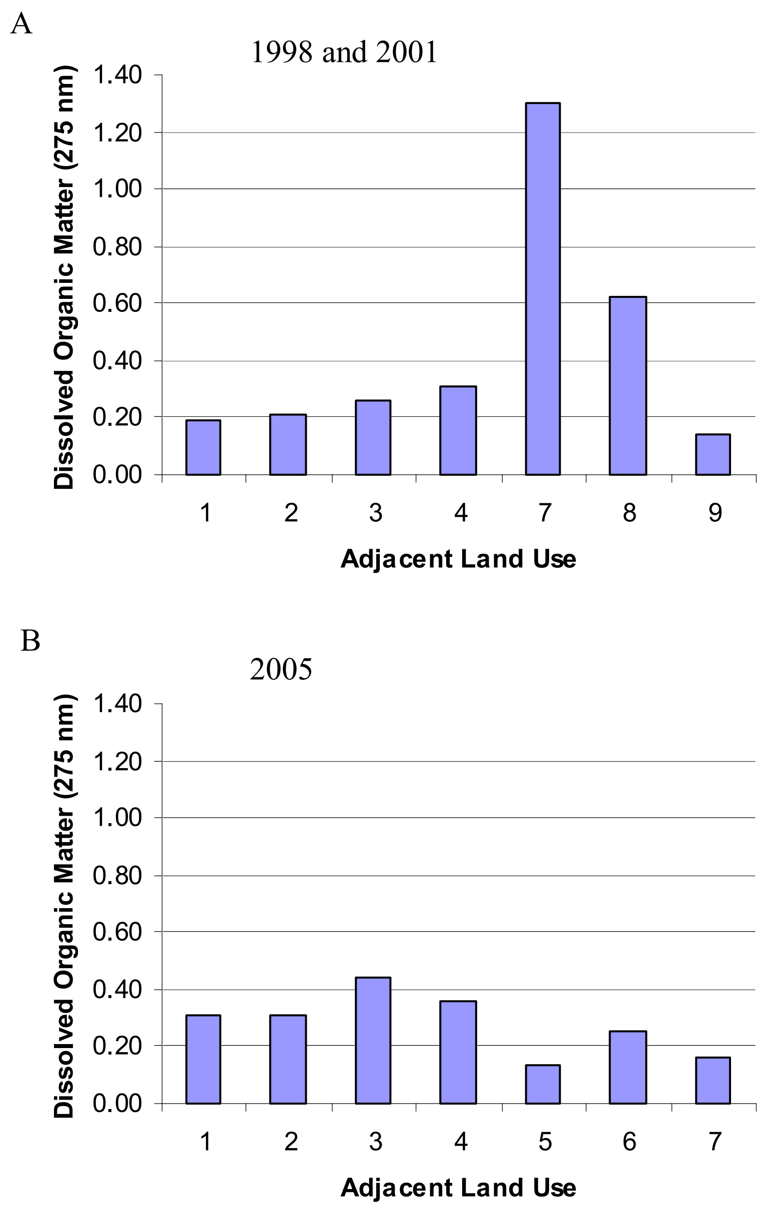


Figure 10: Mean dissolved organic matter index (275 nm) for each primary adjacent land use for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

0.001), minimally impacted ($t = -3.62$, $p = 0.013$), and agricultural crops sites ($t = 2.64$, $p = 0.010$) (Appendix II B).

MANOVA analysis was conducted using the first four categories of land uses, i.e. urban, recreation, agricultural crops and livestock/poultry production as these data cells contained adequate sample numbers for comparison. For the 1998 and 2001 non-flood seasons, MANOVA showed that mean orthophosphate and TDS concentrations were highly significantly affected ($F = 4.17$, $p = 0.001$ and $F = 8.82$, $p < 0.0001$ respectively) by both water body type and type of adjacent land use. Univariate F-tests indicated that water body type was relatively more important than land use in terms of vulnerability to TDS ($F = 21.0$, $p <<< 0.0001$ vs. $F = 14.4$ ($p <<< 0.0001$ (type vs. use)). But for orthophosphate, land use was more important than water body type ($F = 5.2$, $p = 0.003$ vs. $F = 8.4$, $p <<< 0.0001$(type vs. use)). Conversely, during the 2005 flood season, MANOVA showed that mean concentrations of nitrate-N and TDS were highly significantly affected ($F = 4976.83$, $p < 0.0001$ and $F = 6.07$, $p < 0.0001$ respectively) by water body type and land use. However univariate F-tests indicated that water body type was relatively more important than land use in terms of vulnerability to nitrate-N and TDS input ($F = 18928$, $p < 0.0001$ vs. $F = 11903$, $p < 0.0001$ (type vs. use for nitrate-N)) ($F = 25.6$, $p <<< 0.0001$ vs. $F = 15.4$, $p < 0.0001$ (type vs. use for TDS). MANOVA showed that water body type and land use were not significantly interrelated therefore were not interdependent.

Thus distinctions among different land uses became obliterated during flooding for phosphorus. However water body type remained relatively more important than type of land use throughout both flooding and non-flooding seasons. The importance of nitrate-N and its impact on receiving waters became more marked during flooding.

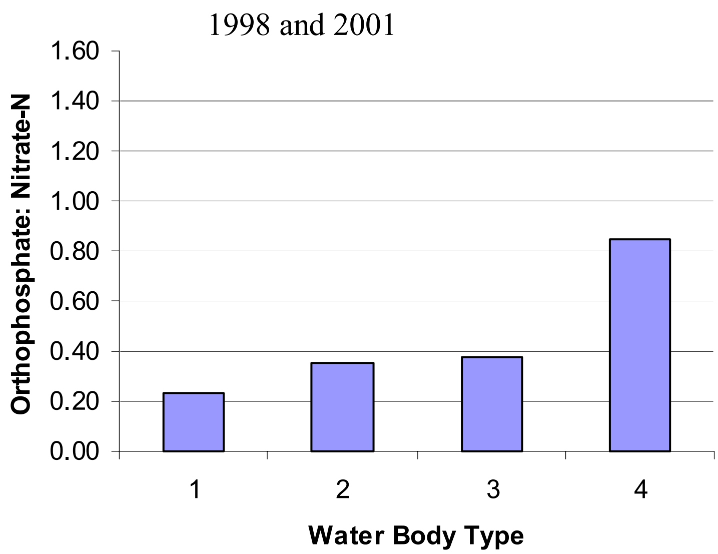
Phosphorus/Nitrogen Ratios

The mean orthophosphate to nitrate-N (P/N) ratios for the 1998 and 2001 non-flood seasons and 2005 flood seasons were examined with respect to water body type (Figure 11 A and B). For the 1998 and 2001 non-flood seasons, the highest P/N ratio was 0.85 in ditches and ponds, and the lowest was 0.23 in lakes. The highest P/N ratio for the 2005 flood season was 1.49 in streams and the lowest was 0.14 in the lakes. ANOVA showed that the P/N ratio was only somewhat different ($F = 2.24$, $p = 0.088$) among water body types for the 1998 and 2001 non-flood seasons. There were no significant differences in P/N ratios observed among water body types for the 2005 flood season.

Mean P/N ratios for the 1998 and 2001 non-flood seasons and the 2005 flood season were examined with respect to primary adjacent land use (Figure 12 A and B). For the 1998 and 2001 non-flood seasons the highest P/N ratio was 1.87 observed in the minimally impacted sites, and the lowest was 0.05 in the hydro-electric sites. For the 2005 flood season, the highest P/N ratio was 1.61 sites affected by urban effluent, and the lowest was 0.02 in the mining sites. ANOVA showed that mean P/N ratios were highly significantly different ($F = 4.72$, $p < 0.0001$) among land uses for the 1998 and 2001 non-flood seasons. The P/N ratio was not found to be significantly different among land uses for the 2005 flood season.

1. Lake
2. River
3. Stream
4. Ditch/Pond

A



B

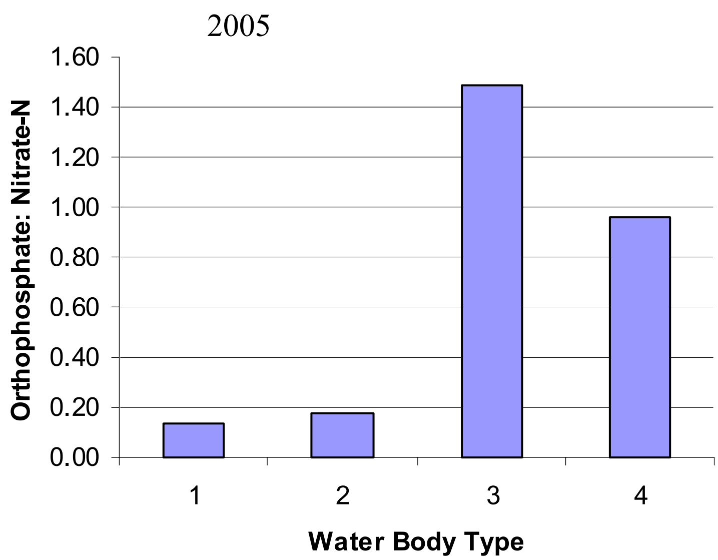


Figure 11: A comparison of mean orthophosphate to nitrate-N ratios for each water body type for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

5. Urban
6. Recreation/Cottages
7. Agricultural Crops
8. Livestock/Poultry Production

5. Mining
6. Industrial
7. Minimal
8. Logging
9. Hydro-electric

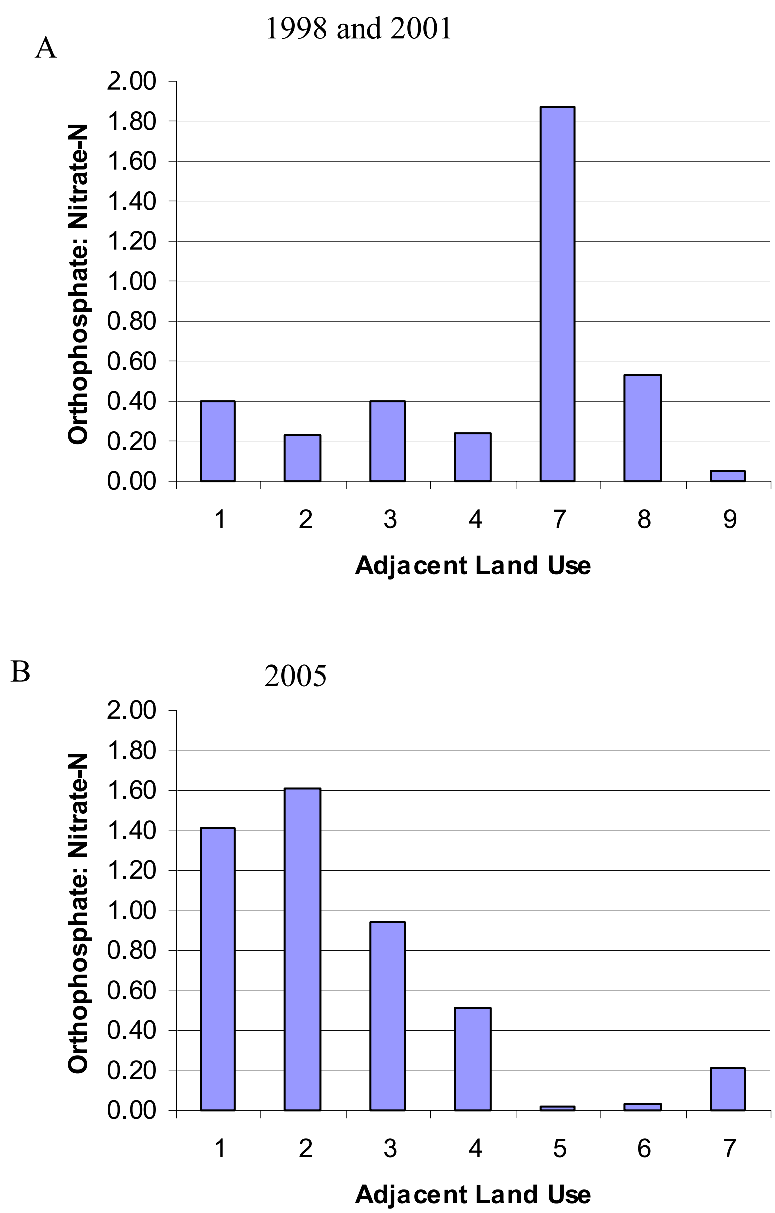


Figure 12: A comparison of mean orthophosphate to nitrate-N ratios for each primary adjacent land use for the (A) 1998 and 2001 non-flood seasons and (B) 2005 flood season.

Discussion

Overall, higher means were observed for all water chemistry parameters during flooding, which causes nutrients to be leached from the soil at higher rates (Bayley, 1995). When the outlier sample (site 55) was omitted, t-tests showed significant increases in nitrate-N, orthophosphate and TDS concentrations during flooding. Salvia-Castellvi, *et al.* (2005) showed that nitrate-N concentrations were leached at much higher rates during wet periods, and total phosphorus loading to streams was increased during high precipitation. According to Gardi (2001), flooding is heavily influenced by precipitation, and therefore nitrate concentrations in run-off are related to rainfall. Gardi (2001) also showed that highly permeable soils resulted in less dilution in receiving water than less permeable soil, and therefore higher nitrate concentrations. However, the type of substrate comprising bottom sediments in the receiving water body did not influence nitrate-N concentrations in the overlying water.

Intercorrelations among nitrate-N, orthophosphate and TDS were more profound during flooding, and DOM was correlated with orthophosphate during flooding only. This suggested that during flooding, leaching increased with respect to all of these parameters. In the non-flood seasons, orthophosphate was correlated only with TDS and nitrate-N. DOM and nitrate-N were not significantly correlated, although Pip (2005) did find a positive correlation in lotic waters.

The non-flood seasons showed marginally higher mean nitrate-N concentrations in streams, which agreed with findings of Pip (2005) for streams on the southern Manitoba floodplain. Total nitrogen loading to streams has been significantly increasing (Jones and Armstrong, 2001; and Bourne, *et al.* 2002). For the 2005 flood season ditches and ponds showed very high values of nitrate-N as a result of flooding, although considerable variation existed as some of the high values came from industrial sites. Ditches and ponds are small water bodies which are less dilute, have fewer sustained outflows and experience proportionately higher rates of decomposition compared to larger water bodies due to their shallowness and abundance of vegetation.

Nitrate-N concentrations increased significantly in rivers during flooding. Flooding inundates nutrient-containing soils, flushes animal and septic waste, and accelerates decomposition of terrestrial and aquatic plants. Decomposition products such as nitrate are carried by rivers to larger water bodies such as lakes. This has major implications on the effect of flooding on nutrient input to Lake Winnipeg.

The highest mean orthophosphate concentrations for the non-flood seasons occurred in streams which were the most immediate receiving waters, yet their lower flow rates and volumes allowed more nutrient assimilation and retention before they reached larger waters downstream. Total phosphorus loading to streams has been significantly increasing in Manitoba (Jones and Armstrong, 2001; Bourne, *et al.* 2002). However ditches and ponds showed the highest mean orthophosphate concentrations in 2005 because of their contained volumes and probable saturation of primary production. The current study found that lakes showed similar mean orthophosphate concentrations during flooding and non-flooding. Because of the lag between watershed leaching and the eventual entry of floodwaters to the receiving lake, this result was expected, and nutrient increases would not have been apparent until later in the season after sampling had ceased, or as found by Pip (2006) in Lake Winnipeg: after the 1997 flood, increases were detected the following year. In general the lag period was greater with increased distance between the source and the receiving water body (Pip, 2006).

Orthophosphate concentrations increased significantly during flooding in rivers and ponds, which are more immediately affected by flooding. Ditches and ponds approached a significant different significantly different from all other water body types during flooding, where their small size and retention resulted in the least dilution. High orthophosphate concentrations in this category were influenced by samples with high values from the industrial category. Since under normal conditions, orthophosphate is usually in short supply, a rapid increase in orthophosphate in these small water bodies may result in accelerated primary production, disrupting the natural balance of the ecosystem (Mainstone and Parr, 2002). However the small size of these water bodies eventually limits further production and uptake of all available nutrients. Carrying capacity is reached due to restrictions such as physical space, light availability, and progressive encroachment of the carbonate-bicarbonate buffer system towards carbonate and excessively high pH. As algae out compete macrophytes for the nutrients, lower strata are shaded out with increasing algal density at the top, and the photic zone becomes increasingly compressed, causing anoxia and decomposition below.

While mean TDS concentrations were highest in lakes during non-flood seasons, lakes had the lowest mean TDS concentrations during flooding. This suggests that, due to high precipitation directly in the lake and increased shoreline run-off, TDS concentration in lakes was diluted more than in other water body types. Ditches and ponds were significantly higher with respect to TDS in the flood season, and once again showed the highest mean values among water body type. TDS concentrations in streams were significantly higher in the flood season.

DOM indices among water body types showed the same trend in the 1998 and 2001 non-flood season as in the 2005 flood season. Ditches and ponds showed the highest DOM indices, and the values were progressively lower for streams, rivers and lakes respectively. Since DOM can be an indication of decomposition and productivity, smaller water bodies with few outflows, would be assumed to have the highest DOM indices due to higher concentration of nutrients and more primary production compared to larger water bodies.

Ditches and ponds were significantly different from lakes, rivers, and streams during non-flooding, but were only significantly different from lakes during flooding. This implies that during flooding, the rate of decomposition and the amount of pollution in these water bodies may have been more similar due to high volumes of water and mixing from accelerated flows. Also, since ditches and ponds showed the highest values for DOM in both non-flood and flood seasons, it suggests that decomposition rates and vulnerability to pollution is greater in smaller water bodies.

With regard to primary adjacent land use, urban sites showed significantly higher nitrate-N concentrations in non-flood seasons than most other land uses. Pip (2005) found that urban effluent had the most noticeable effect on nitrate-N concentrations in the southern Manitoba floodplain. During flooding, industrial sites showed markedly higher nitrate-N concentrations compared to most other land uses. The industrial category contained the extreme outlier, site 55, which was a ditch bordering a depot with a large ammonia tank. Ammonia is used in the manufacturing of ammonium and nitrate fertilizers. Ammonia also occurs naturally through the decomposition of plants and animals and aquatic invertebrates and fish excrete nitrogen waste in this form. In aerobic environments, ammonia undergoes microbial nitrification to form nitrite and nitrate, which is the form of nitrogen that is most readily assimilated by plants (Scholz and Trepel, 2004).

Nitrate-N concentrations increased significantly during flooding in sites affected by recreation and agricultural crops. Cottage development accompanied by leaching of septic fields, holding tank spills, pet waste and application of fertilizers to waterfront and near shore properties. Orthophosphate concentrations were significantly higher in the non-flood seasons for sites affected by urban effluent and agricultural crops compared to recreational development. Thus input of both nitrate-N and orthophosphate was elevated during high precipitation due to greater run-off from fields treated with inorganic fertilizers and manure.

Salvia-Castellvi (2005) showed that soluble phosphorus was high in surface waters from urban and agricultural effluent. Mainstone and Parr (2002) found that the main source of phosphorus in rivers was urban sewage effluent. Bourne *et al.* (2002) estimated that 8% of total phosphorus transported by the Red River was contributed by Winnipeg's three wastewater treatment facilities, and watershed processes, mainly agricultural activities contributed the largest share of nutrients to Lake Winnipeg.

According to Carpenter, *et al*. (1998) 3-20% of phosphorus applied as fertilizers enters surface waters through leaching or erosion. In the United States and Europe, only about 30% of the total phosphorus applied as fertilizers are utilized by crops. Therefore there is an excess of phosphorus being applied, which ultimately contributes to eutrophication. Since phosphorus has a tendency to remain in soils, controlling the amount of erosion through modified tillage practices, or preservation of permanent vegetation may help to reduce phosphorus loading (Mainstone and Parr, 2002). However tillage practices and tile fields primarily reduce phosphorus that is carried with soil particles; soluble phosphorus is less affected, unless escape of water from fields is restricted also. But with current trends to increase drainage in most Manitoba municipalities, water from cropland runs off more, rather than less, quickly resulting in more loss of soluble phosphorus.

When the industrial category was omitted, the highest concentrations of TDS in the non-flood and flood seasons were in sites affected by agricultural crops. Pip (2005) found that in southern Manitoba, TDS was elevated in surface water associated with logging/clear-cutting. However both clear-cutting of forests and agricultural cropping practices involve land clearing and the disruption of soils, and increase the amount of erosion, promoting leaching of dissolves substances into surface water. According to Pip (2005), flood waters cause increased leaching and soil erosion as well as decomposition of plant material, resulting in high TDS concentrations.

The highest DOM indices for the non-flood seasons were in minimally impacted sites. Surface water collected from these sites was frequently from bogs or marshes, which were associated with smaller water volumes, high productivity in the case of marshes, and less water flow during normal precipitation levels. Pip (2005) found that sites adjacent to logging, agricultural crops and livestock operations showed significantly higher DOM indices. Clear-cutting disrupts the natural vegetation, and contributes high amounts of DOM to surface waters (Pip, 2005).

During flooding, the highest DOM index was at agricultural crops sites, and significantly higher DOM indices were identified in sites affected by urban effluent and agricultural crops. Accelerated leaching of DOM from agricultural activities including crops and livestock operations may be due to more intensive tillage practices and farming of marginal soils, combined with larger-scale mechanization and disappearance of small family farms (deJonge, *et al.*, 2002).

Since during non-flood seasons, orthophosphate and TDS concentrations were significantly affected by both water body type and type of primary adjacent land use this implies that nutrient management strategies for surface waters should take into account the type of receiving waters before permitting certain types of land use. During flooding, nitrate-N and TDS concentrations were significantly affected by water body type and type of primary adjacent land use. Nitrate-N is normally present in much greater concentrations than orthophosphate in surface waters, and flooding seems to magnify these differences. Therefore nutrient budgets for watersheds should not only consider nutrient input during non-flood seasons with regard to orthophosphate, but also nitrate-N and the effects of flooding.

P/N ratios increased in the small water bodies, streams and ponds during flooding. Phosphorus is normally in much shorter supply than nitrogen. During flooding, orthophosphate leaching relative to nitrate-N increased. During non-flood seasons, minimally impacted sites showed significantly higher P/N ratios than all sites impacted by human activity. These results suggest that all types of human activity were associated with proportionally more nitrate-N than orthophosphate input, compared to more natural nutrient input regimes. However during flooding, urban effluent, recreation/cottages, agricultural crops and livestock operations assumed the highest P/N ratios, again indicating that orthophosphate was leached at much greater rates than nitrate-N.

Conclusion

(1) The present study found that nitrate-N, orthophosphate and TDS increased significantly during flooding.

(2) During flooding due to higher rainfall, nitrate-N concentrations increased significantly in rivers, orthophosphate was significantly higher in rivers and ponds, and TDS was significantly higher in streams and ponds. This suggests that smaller water bodies are most vulnerable to nutrient loading during flooding.

(3) With respect to primary adjacent land use, nitrate-N increased significantly in sites impacted by recreation, and agricultural crops during flooding. DOM increased significantly in the urban and agricultural crops sites. Thus effects of municipal effluents and agricultural activity became magnified during flooding.

(4) Both water body type and land use significantly affected orthophosphate and TDS during non-flood seasons, and nitrate-N and TDS during the flood season, suggesting that both the characteristics of receiving waters and the type of primary adjacent land use must be taken into account when formulating nutrient management strategies.

(5) These conclusions must be interpreted with caution as year to year variation within flood events is unknown and non-flood events were pooled. In addition, samples were not all collected at the same time and place.

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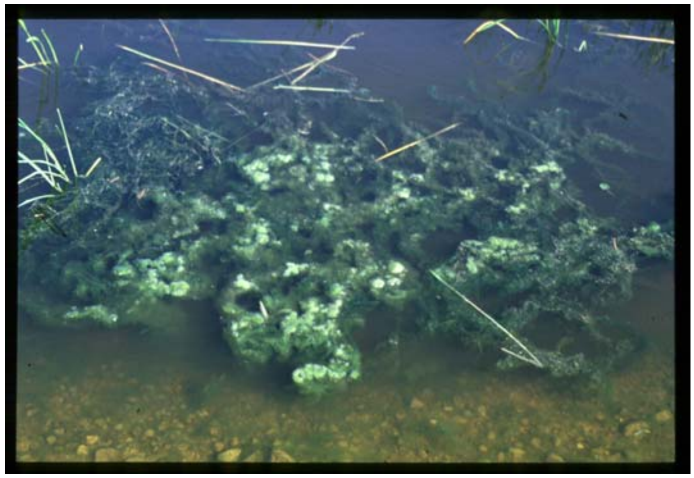
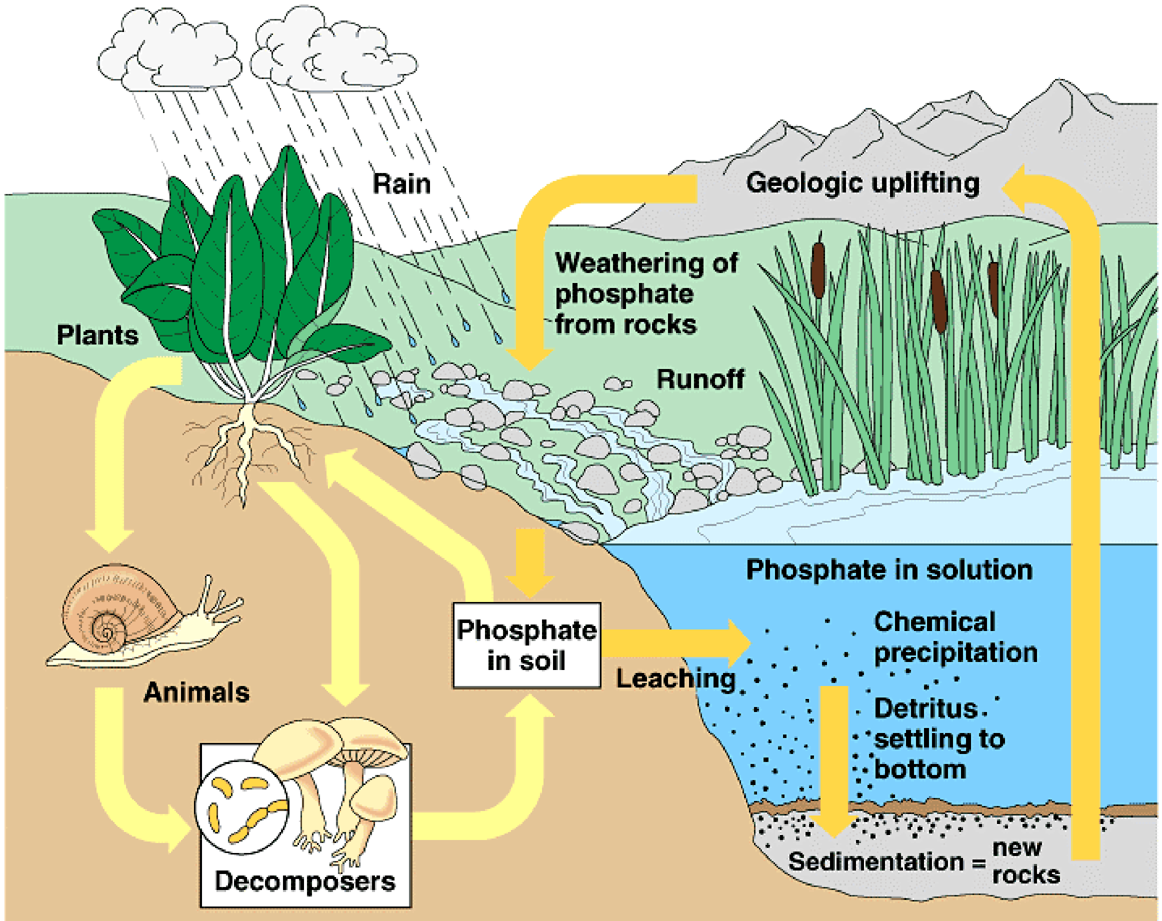
A

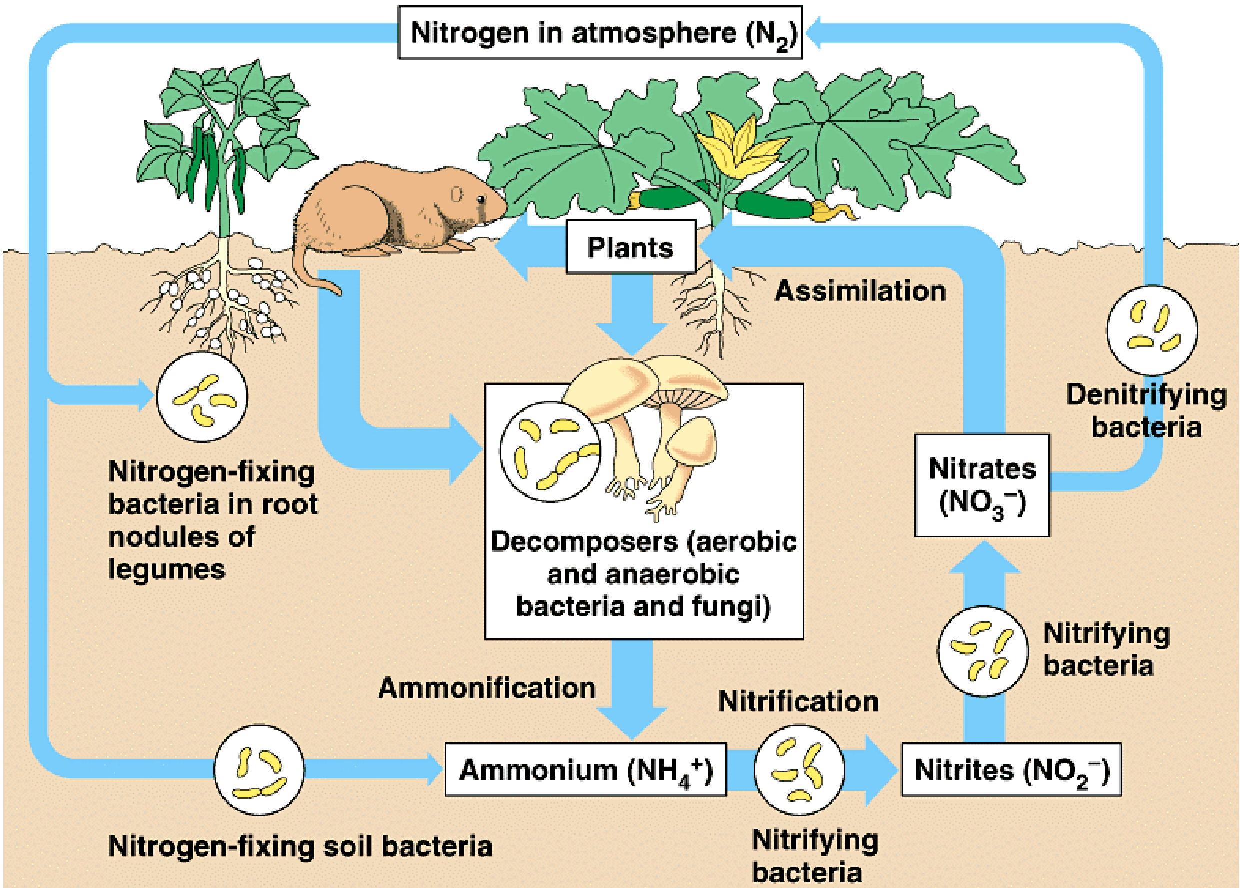


B

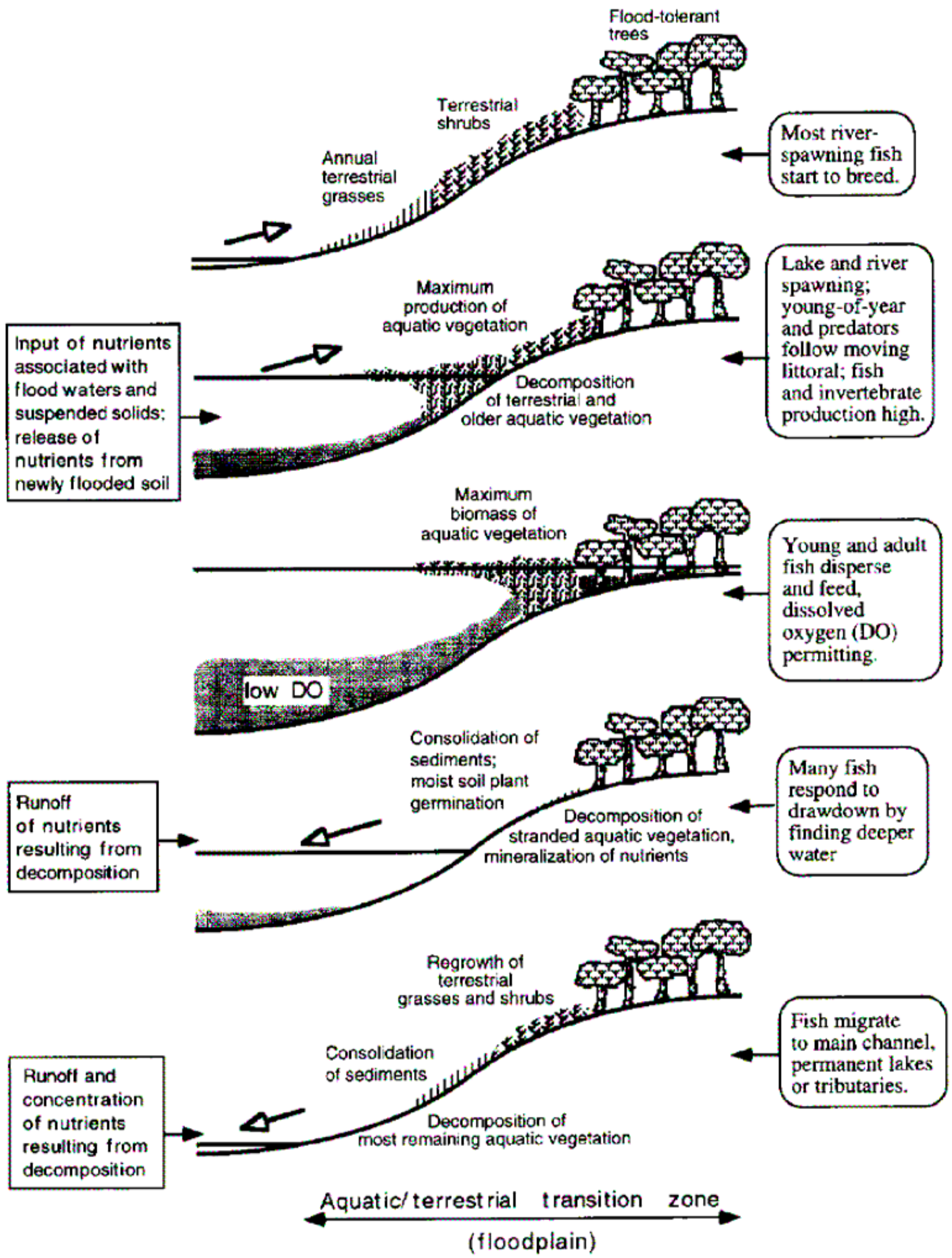
Appendix I-1: Prominent indicators of eutrophication include the presence of (A) large algal mats and (B) dense algal blooms (E. Pip, University of Winnipeg).



Appendix I-2: The phosphorus cycle. Phosphorus is assimilated by plants in the soluble inorganic form, orthophosphate. Phosphorus can be returned to the water through excretion, decomposition, weathering of rocks and through run-off. The eventual phosphorus sink is sedimentation (Campbell, *et al.*, 1999).



Appendix I-3: Nitrogen cycle. Atmospheric nitrogen is converted to ammonium through the action of nitrogen fixing bacteria. Ammonium is utilized by plants as well as aerobic bacteria which oxidize ammonium to nitrite and nitrate through their metabolic activity. Nitrate, is assimilated by plants and converted to organic nitrogen which is used by animals. Dinitrogen is returned to the atmosphere through the anaerobic reduction of nitrate. Nitrogen is cycled back to the soil in the form of ammonium through mineralization of organic nitrogen by decomposers (Campbell, *et al.*, 1999).



Appendix I-4: Flood-pulse concept describes the annual hydrological cycle for a river floodplain ecosystem (Bayley, 1995). DO= Dissolved Oxygen.

Appendix II-A

Differences in water chemistry for the 1998 and 2001 non-flood seasons (Group 1) and the 2005 flood season (Group 2) for each water body type category.
*** Indicates significant difference**

	Parameter	Group	Mean (±S.E)	Significance
Lake	**Nitrate-N**	1	0.39 ± 0.05	t = 1.76, p = 0.096
		2	0.57 ± 0.09	
	Orthophosphate	1	0.09 ± 0.03	t = 0.08, p = 0.936
		2	0.09 ± 0.03	
	TDS	1	396 ± 136	t = -0.51, p = 0.615
		2	288 ± 73	
	DOM	1	0.18 ± 0.02	t = 0.92, p = 0.373
		2	0.21 ± 0.03	
	Parameter	Group	Mean (±S.E)	Significance
River	**Nitrate-N**	1	0.33 ± 0.04	t = 5.06, p < 0.0001 *
		2	0.81 ± 0.10	
	Orthophosphate	1	0.07 ± 0.02	t = 2.17, p = 0.035 *
		2	0.15 ± 0.04	
	TDS	1	214 ± 33	t = 1.56, p =0.129
		2	319 ± 59	
	DOM	1	0.28 ± 0.03	t = 0.99, p = 0.326
		2	0.33 ± 0.03	
	Parameter	Group	Mean (±S.E)	Significance
Stream	**Nitrate-N**	1	0.55 ± 0.17	t = 1.10, p = 0.275
		2	0.85 ± 0.22	
	Orthophosphate	1	0.15 ± 0.04	t = 0.33, p = 0.740
		2	0.17 ± 0.05	
	TDS	1	279 ± 27	t = 2.37, p =0.021 *
		2	440 ± 59	
	DOM	1	0.29 ± 0.04	t = 1.25, p = 0.216
		2	0.35 ± 0.03	
	Parameter	Group	Mean (±S.E)	Significance
Ditch/pond	**Nitrate-N**	1	0.31 ± 0.07	t = 0.69, p = 0.495
		2	10.38 ± 9.22	
	Orthophosphate	1	0.06 ± 0.03	t = 2.51, p = 0.015 *
		2	0.32 ± 0.06	
	TDS	1	239 ± 35	t = 2.51, p = 0.015 *
		2	786 ± 136	
	DOM	1	0.58 ± 0.18	t = -1.07, p = 0.291
		2	0.43 ± 0.06	

Appendix II-B

Differences in water chemistry for the 1998 and 2001 non-flood seasons (Group 1) and the 2005 flood season (Group 2) for each primary adjacent land use category.
*** Indicates significant difference.**

	Parameter	Group	Mean (±S.E)	Significance
Urban	**Nitrate-N**	1	0.85 ± 0.37	t = -0.18, p = 0.862
		2	0.79 ± 0.11	
	Orthophosphate	1	0.15 ± 0.22	t = 0.39, p = 0.701
		2	0.18 ± 0.20	
	TDS	1	269 ± 43	t = 1.38, p = 0.177
		2	350 ± 39	
	DOM	1	0.19 ± 0.02	t = 3.62, p = 0.001 *
		2	0.31 ± 0.03	
	Parameter	Group	Mean (±S.E)	Significance
Recreation/Cottages	**Nitrate-N**	1	0.30 ± 0.03	t = 2.22, p = 0.031 *
		2	0.51 ± 0.14	
	Orthophosphate	1	0.04 ± 0.01	t = 1.28, p = 0.209
		2	0.06 ± 0.02	
	TDS	1	2434± 31	t = 0.62, p =0.539
		2	287 ± 82	
	DOM	1	0.21 ± 0.03	t = 1.66, p = 0.120
		2	0.31 ± 0.06	
	Parameter	Group	Mean (±S.E)	Significance
Agricultural Crops	**Nitrate-N**	1	0.38 ± 0.03	t = 2.94, p = 0.005 *
		2	1.06 ± 0.23	
	Orthophosphate	1	0.14 ± 0.03	t = 1.87, p = 0.065
		2	0.26 ± 0.05	
	TDS	1	417 ± 98	t = 1.84, p =0.071
		2	653 ± 84	
	DOM	1	0.26 ± 0.02	t = 2.64, p = 0.010 *
		2	0.44 ± 0.06	
	Parameter	Group	Mean (±S.E)	Significance
Livestock/Poultry Production	**Nitrate-N**	1	0.53 ± 0.14	t = 1.16, p = 0.250
		2	1.06 ± 0.26	
	Orthophosphate	1	0.14 ± 0.09	t = 0.82, p = 0.419
		2	0.23 ± 0.06	
	TDS	1	243 ± 53	t = 1.77, p =0.084
		2	500 ± 84	
	DOM	1	0.31 ± 0.04	t = 1.10, p = 0.280
		2	0.36 ± 0.04	
	Parameter	Group	Mean (±S.E)	Significance
Minimal Impact	**Nitrate-N**	1	0.07 ± 0.05	t = 1.34, p = 0.398
		2	0.42 ± 0.26	
	Orthophosphate	1	0.02 ± 0.01	t = 1.81, p = 0.120
		2	0.13 ± 0.12	
	TDS	1	118 ± 49	t = 1.08, p =0.46
		2	335 ± 48	
	DOM	1	1.30 ± 0.30	t = -3.62, p = 0.013 *
		2	0.16 ± 0.08	

Appendix III

Location of sampling sites during the 2005 flood season.

Sample # and Date	Site Description	GPS Coordinates (Latitude/Longitude)
19-Jul-05		
1	Wavey Creek, at Hwy. 8, cattle	50°15.364N/ 097°02.670W
2	Netley Creek, at Hwy. 8, urban	50°19.624N/ 097°02.665W
3	Boundary Creek, recreation	50°30.456N/ 096°57.980W
4	Winnipeg Beach, Lake Winnipeg	50°29.900N/ 096°58.040W
5	Willow Creek, cattle	50°34.853N/ 096°59.901W
6	Gimli Harbour, Lake Winnipeg, urban	50°37.880N/ 096°59.067W
7	Gimli Beach, Lake Winnipeg, urban	50°38.208N/ 096°59.109W
8	Ditch, at Minerva Rd., Crops	50°36.637N/ 096°59.811W
9	Lake Winnipeg, at Husavik Bay, Urban	50°33.961N/ 096°59.369W
10	Ditch, crops	50°24.182N/ 096°58.530W
11	Red River, at Eveline St., Selkirk, urban	50°08.565N/ 096°52.124W
12	Red River Floodway, at Lockport, urban	50°05.126N/ 096°55.568W
21-Jul-05		
13	Assiniboine River, Beaudry Provincial Park, crops	49°58.983N/ 097°13.761W
28-Jul-05		
14	Red River at St. Agathe, Samoiset Ave.& Pembina Trail, urban	49°33.898N/ 097°11.061W
15	Ditch at Hwy. 75 South of Morris, crops	49°19.509N/ 097°21.944W
16	Ditch south of Morris stampede grounds and north of residential	49°20.846N/ 097°21600W
17	Ditch south of Morris River at	49°21.468N/ 097°21.703W

		Mary St.	
	18	Stream at Hwy. 75, urban	49°23.038N/ 097°20.533W
	19	Ditch at Hwy. 75, crops	49°26.074N/ 097°17.737W
	20	Ditch at Hwy. 75, crops	49°29.934N/ 097°15.138W
	21	Stream at Hwy. 75, livestock	49°36.337N/ 097°08.611W
	22	Ditch at Hwy. 75, livestock	49°39.342N/ 097°07.798W
31-Jul-05			
	23	Winnipeg River, downstream of Pine Falls pulp and paper mill	50°34.437N/ 096°13.980W
	24	Winnipeg River at St. George, urban	50°32.142N/ 096°08.910W
10-Aug-05			
	25	Assiniboine River, Spirit Sands, Spruce Woods Provincial Park, minimal impact	49°39666N/ 099°16.015W
	26	Marsh's Lake, Wayside and Interpretive Trail, Spruce Woods Provincial Park, minimal	49°40507N/ 099°16.113W
16-Aug-05			
	27	Pond at Paul Blvd., Canada Geese, urban	49°44.012N/ 097°05.879W
	28	Ditch at Paul Blvd., livestock	49°46.413N/ 097°05.502W
	29	Red River Floodway, south of St. Anne's Rd., urban	49°47.329N/ 097°03.429W
	30	Seine River at Creek Bend, urban	49°48.819N/ 097°04.308W
19-Aug-05			
	31	Ditch, Lorne Hill Road, cattle	49°49.536N/ 097°06.660W
	32	Cook's Creek, Zora Rd. and Road 68, urban	50°00.211N/ 096°45.779W
	33	Ditch at Hwy. 213 and Hwy. 212, crops	49°58.463N/ 096°46.413W
	34	Omand's Creek at St. James street, residential	49°54.761N/ 097°11.932W

	35	Sturgeon Creek at Woodhaven Park, Portage Ave., urban	49°52.619N/ 097°16.352W
22-Aug-05			
	36	Stream at Hwy. 44; West of Garson, crops	50°04.613N/ 096°44.296W
	37	Stream at Hwy. 44; Tyndall, crops	50°04.594N/ 096°39.831W
	38	Ditch at Road 40 E and Hwy. 44, crops	50°04.583N/ 096°32.632W
	39	Ditch at Road 71 N and Hwy. 302 South, crops	50°03.294N/ 096°29.871W
	40	Brokenhead River at Hwy. 44, crops	50°03.685N/ 096°27.858W
	41	Stream at Hwy. 44 and Road 47 E, cattle, horses	50°03.695N/ 096°22.891W
	42	Stream at Hwy. 44 and Road 47 E, crops	50°05.474N/ 096°22.895W
	43	Stream at Hwy. 44; west of Seddons Corner, horses	50°03.693N/ 096°18.649W
	44	Ditch at Road 42 E and Mile G9, livestock	50°01.046N/ 096°29.915W
	45	Ditch at Road 42 E and Mile 70 N, cattle, horses	50°01.930N/ 096°29.881W
	46	Ditch Hwy. 44 and Road 41 E, cattle	50°04.750N/ 096°31.249W
	47	Ditch at Brier Cliff Road, Heartland Hutterite colony, livestock	49°59.182N/ 096°40.939W
28-Aug-05			
	48	Stream from Lake Manitoba at Twin Beach Rd., cottages	50°21.452N/ 097°56.452W
	49	Lake Manitoba at Twin Beach Rd., cottages	50°21.391N/ 097°56.900W
	50	Marsh at Twin Beach Rd., cottages	50°20.352N/ 097°57.550W
	51	Ditch Hwy. #6 North of	50°15.355N/ 097°44.288W

		Woodlands, crops	
	52	Ditch at Road 75 NW and Hwy. 6, RG Hamilton Farm (crops)	50°06.490N/ 097°30.607W
	53	Ditch, Meridian Road and Road 72N, Mixed farming; Rock Lake Hutterite Colony	50°03.829N/ 097°27.736W
	54	Stream at West Perimeter Hwy. and Inkster Blvd., crops	49°57.082N/ 097°19.265W
	55	Ditch, at Road 5E and Inkster Blvd., Agricore, ammonia tanks; Industrial	49°57.940N/ 097°20.974W
30-Aug-05			
	56	Ditch at Hwy. 1 East and Murdock Road, crops	49°56.979N/ 097°10.716W
	57	Seine River at Mun 30 E, horses	49°43.641N/ 096°46.561W
	58	Stream at Road 207 E, sheep	49°43.873N/ 096°43.737W
	59	Seine River Diversion, at Hwy. 12, hogs	49°39.166N/ 096°40.471W
	60	Ditch at Penner Rd. livestock	49°37.149N/ 096°41.311W
	61	Ditch at Penner Rd. livestock	49°37.139N/ 096°42.573W
	62	Ditch at Penner Rd. livestock	49°37.139N/ 096°42.573W
	63	Ditch at Hwy. 311 W and Twin Creek Road, hogs	49°36.305N/ 096°43.950W
	64	Stream at Hwy. 12 and Hwy. 311 E, hogs	49°34.437N/ 096°41.174W
	65	Ditch at Hwy. 311, Blumenort, Granny's Poultry	49°36.031N/ 096°41.229W
	66	Ditch at Hwy. 311, Steve's Livestock Transport (Feed Lot)	49°36.246N/ 096°41.731W
	67	Ditch at Hwy. 206, livestock	49°37.191N/096°49.232W
	68	Manning Canal, at Hwy. 311, feed lot	49°36.320N/ 096°50.939W
	69	Stream, Hwy. 311 and Road 26 E	49°36.314N/ 096°51.989W
	70	Stream, Hwy. 311, north of New Bothwell, poultry and hogs	49°36.338N/ 096°53.320W

31-Aug-05			
	71	Stream, Hwy. 1, crops	49°41.537N/ 096°59.147W
	72	La Salle River, Hwy. 1, crops	49°54.501N/ 097°45.848W
	73	Ditch, Hwy. 1, crops	49°54.712N/ 097°45.988W
	74	Stream, Hwy. 1, crops	49°56.320N/ 097°55.661W
	75	Stream, Hwy. 1, crops	49°58.476N/ 098°11.447W
	76	Crescent Lake, Crescent Rd. and 3rd Street, urban	49°58.052N/ 098°17.200W
	77	Stream, Westco and 6th Ave. south of McCaine's food plant, industrial	49°58.898N/ 098°16.058W
	78	Marsh, Hwy. 240	50°10.162N/ 098°18.542W
	79	Delta Marsh, Lake Manitoba, Hwy. 240	50°10.572N/ 098°18.803W
	80	Delta Beach, Lake Manitoba, cottages	50°11.101N/ 098°20.521W
	81	Assiniboine River, Hwy. 240, potatoes	49°56.142N/ 098°16.384W
01-Sep-05			
	82	Ditch at Hwy. 311 E and Ridgewood Rd., cattle	49°57.229N/ 098°16.371W
	83	Stream at Mun Rd. 38 N, cattle	49°33.589N/ 096°39.260W
	84	Ditch, at Mun Rd. 38 N, cattle	49°33.575N/ 096°37.679W
	85	Stream, Hwy. 311 E, corn	49°34.474N/ 096°32.502W
	86	Ditch, Hwy. 302 N, crops	49°34.460N/ 096°29.742W
	87	Marsh, Hwy. 302 N, gravel pit (mining)	49°34.714N/ 096°27.505W
	88	Stream, Hwy. 302 N, cattle	49°36.188N/ 096°27.414W
	89	Whitemouth River, Hwy. 11 N, crops	49°41.497N/ 095°54.451W
	90	Whitemouth River, Hwy. 1 E, east of nursery	49°38.971N/ 095°52.824W
	91	Birch River, at Hwy. 1 E	49°38.891N/ 095°43.542W
	92	Marsh, Falcon Lake, Whiteshell Provincial Park, recreation	49°40.590N/ 095°19.706W
	93	Falcon Lake, Whiteshell	49°40.816N/ 095°18.609W

		Provincial Park, recreation	
	94	West Hawk Lake, beach, Whiteshell Provincial Park, recreation	49°44.637N/ 095°12.565W
	95	Pond, Falcon Beach Riding Stables	49°41.803N/ 095°20.115W
	96	Brokenhead River at Hwy. 1 E	49°39.460N/ 096°16.450W
02-Sep-05			
	97	Ditch, Hwy. 3, hogs	49°45.689N/ 097°20.095W
	98	Stream, Hwy. 3, bison and free range chickens	49°42.486N/ 097°24.075W
	99	La Salle River at Hwy. 247, Sanford, urban	49°40.792N/ 097°25.855W
	100	Morris River at Hwy. 3, south of Brunkild, crops	49°35.300N/ 097°34.860W
	101	Norquay Chanel, at Hwy. 3, poultry	49°34.596N/ 097°36.136W
	102	Boyne River, at Hwy. 245, Carmen, urban	49°30.127N/ 098°00.409W
	103	Red River Floodway at Hwy. 13, urban	49°30.131N/ 097°59.845W
	104	Elm Creek, at Hwy. 13 and Hwy. 2, horses	49°30.131N/ 097°59.845W
	105	Stream, Hwy. 2, north of Fannystelle, crops, south of Starbuck	49°30.131N/ 097°59.845W
	106	La Salle River, Hwy. 2, Starbuck, urban	49°46.104N/ 097°35.531W

Appendix IV

Location of sampling sites for the 1998 and 2001 non-flood seasons.

Sample # and Date	Site Description	GPS Coordinates (Latitude/Longitude)
22-Apr-98		
13	Oak Point, L. Manitoba	50°30.396 N/ 98°02.566 W
14	Laurentia Beach, St. Laurant, L. Manitoba	50°25.101N/ 97°57.556W
12-Jun-98		
15	Whitemouth River at Hwy. 307	50°06.338N/ 96°01.955W
16	roadside ditch	50°06.337N/ 95°54.295W
33	Rennie River at Hwy. 307	49°57.171N/ 95°30.330W
34	Brereton Lake at Inverness Falls	49°55.754N/ 95°32.751W
37	Brokenhead River at Hwy. 44	50°03.761N/ 96°28.109W
16-Jun-98		
38	Seine River diversion at Hwy. 44	49°39.171N/ 96°40.452W
39	Manning Canal at Hwy. 311	49°36.337N/ 96°50.951W
40	Stream at Hwy. 311	49°36.335N/ 97°07.041W
41	Rat River at Hwy. 200	49°35.181N/ 97°08.325W
42	Red River at St. Agathe	49°34.041N/ 97°10.852W
43	La Salle River at Hwy. 344	49°37.272N/ 97°22.578W
44	Morris River at Brunkild	49°35.256N/97°34.936W
45	Norquay Channel at Hwy. 3	49°34.549N/97°36.126W
46	Boyne River at Carmen	49°30.157N/ 98°00.450W
47	Lake Stephenfield Provincial Park	49°31.372N/ 98°17.583W
48	Cypress River at Hwy. 34	49°27.934N/ 98°50.384W
50	Pond at Hwy. 3	49°13.282N/ 98°43.792W
55	Colent Beach, Lake Minnewasta, Morden	49°11.026N/ 98°08.283W
56	River at Plum Coulee at Hwy. 14	49°11.133N/ 97°45.840W
57	Plum River at Hwy. 14	49°11.448N/ 97°24.118W
58	Mom's River at Hwy. 75	49°21.592N/ 97°21.849W
59	La Salle River at Hwy. 75 (St. Norbert)	49°45.862N/ 97°09.103W
60	Seine River at Winnipeg south perimeter Hwy	49°48.515N/ 97°03.950W
23-Jun-98		
62	roadside ditch	50°45.814N/ 96°10.255W
30-Jun-98		
95	Cook's Creek at Hwy. 44	50°04.653N/ 96°40.194W

96	Red River at Lockport	50°05.053N/ 96°56.591W
97	Oak Hammock Marsh	50°10.315N/ 97°08.054W
98	Portage Creek at Hwy. 227	50°06.489N/ 98°14.677W
99	Lake Manitoba at U of M field station	50°11.006N/ 98°22.966W
100	Blind Channel at U of M field station	50°10.370N/ 98°22.412W
103	Lynch Point, Lake Manitoba	50°14.848N/ 98°34.568W
113	Creek at Hwy. 34	50°05.419N/ 98°56.967W
115	Assiniboine River at Hwy. 34	49°41.971N/ 98°53.995W
116	Boyne River at Hwy. 2	49°39.016N/ 98°30.477W
117	La Salle River at Hwy. 2	49°46.082N/ 97°35.489W
09-Jul-98		
118	Marsh	49°43.098N/ 96°27.357W
120	small stream, cattle	49°33.301N/ 96°30.211W
121	Stream, cattle	49°31.945N/ 96°30.443W
122	marsh, hogs	49°22.639N/ 96°30.725W
123	Rat River	49°13.742N/ 96°30.631W
124	Drainage channel	49°06.704N/ 96°24.716W
131	Birch Point, Buffalo Bay, Lake of the Woods	49°09.895N/ 95°14.076W
132	stream at Hwy. 308	49°15.466N/ 95°16.904W
133	stream at Hwy. 308	49°20.617N/ 95°20.706W
134	roadside ditch at Hwy. 308	49°26.511N/ 95°26.881W
135	Boggy River at east Braintree	49°37.144N/ 95°37.477W
136	Falcon Lake	49°40.747N/ 95°18.873W
138	West Hawk Lake	49°44.694N/ 95°12.557W
140	Caddy Lake, east shore	49°48.509N/ 95°11.644W
141	Lily Pond	49°48.927N/ 95°16.007W
142	River at Hwy. 44, north of Lily Pond	49°49.852N/ 95°20.114W
143	Bog River at Hwy. 44	49°55.674N/ 95°51.846W
144	Whitemouth River at Elma	49°52.516N/ 95°54.456W
21-Jul-98		
145	La Salle River at Elie	49°54.098N/ 97°45.732W
146	Squirrel Creek at Hwy. 1	49°56.847N/ 98°59.063W
148	River at Hwy. 1 east of Douglas	49°54.099N/ 99°41.503W
151	Assiniboine River at Hwy. 340	49°41.652N/ 99°39.511W
154	Marsh at Hwy. 10	49°18.166N/ 100°00.763W
158	Lake at Hwy. 5	49°25.199N/ 99°18.919W
159	Lake Kiche Maniton, Spruce Woods Provincial Park	49°39.043N/ 99°15.339W
160	Marsh Lake, Spruce Woods Provincial Park	49°40.521N/ 99°16.237W

161	Pond at Hwy. 351	49°52.313N/ 99°13.564W
30-Jul-98		
162	Cook's Creek at Hwy. 59	50°07.313N/ 96°49.802W
163	Devil's Creek at Hwy. 59	50°14.212N/ 96°44.730W
165	Gull Lake, north shore	50°25.039N/ 96°31.016W
168	Marsh, Grand Beach, L. Winnipeg	50°33.097N/ 96°36.691W
169	Grand Beach Lagoon, Grand Beach Provincial Park	50°33.138N/ 96°36.251W
170	east shore, Grand Beach, Grand Beach Provincial Park	50°34.145N/ 96°35.813W
04-Aug-98		
177	Wavy Creek at Hwy. 9	50°16.033N/ 96°58.444W
178	Netley Creek at Petersfield	50°18.043N/ 96°57.773W
188	roadside ditch at Hwy. 8	57°07.857N/ 96°46.270W
13-Aug-98		
203	Grassmere Creek at Hwy. 9	49°58.669N/ 97°04.183W
204	Rat Creek at Hwy. 16	49°59.088N/ 98°28.150W
21-Aug-98		
225	Miami Beach, off Hwy. 6 near Woodlands	50°12.614N/ 97°42.737W
244	Winnipeg River at Hwy. 211	50°08.518N/ 96°02.936W
245	Old Pinawa Dam, Winnipeg River	50°13.102N/ 95°55.742W
246	Pinawa Channel at Hwy. 313	50°16.249N/ 95°52.547W
248	Point du Bois, Winnipeg River	50°18.024N/ 95°33.024W
249	pond at Hwy. 313	50°18.852N/ 95°35.017W
263	Lee River at Camp Hide-A-Way	50°26.474N/ 95°13.773W
264	Winnipeg River at Hwy. 313	50°17.036N/ 95°59.887W
268	St. George, Winnipeg River	50°32.138N/ 96°08.937W
269	Brokenhead River at Hwy. 12	50°17.440N/ 96°29.352W
02-Sep-98		
272	Bigelows Slough at Hwy. 254	49°40.786N/ 100°41.365W
273	Stony Creek, draining Maple Lake at Hwy. 83	49°27.232N/ 100°57.778W
274	Creek at Hwy. 256	49°21.951N/ 101°15.414W
276	Creek south of Lyleton	49°00.505N/ 101°10.996W
04-Sep-98		
293	Drainage ditch at Hwy. 12 and Settler's Road	49°48.527N/ 96°38.306W
294	Seine River Diversion at Hwy. 12	49°39.199N/ 96°40.448W
295	River at Hwy. 216, south of Kleefeld	49°28.843N/ 96°52.323W
296	Gravel Pit at Grunthal	49°24.216N/ 96°50.954W
06-Sep-98		
301	Winnipeg River. 1/4 mile downstream from	50°34.385N/ 96°13.965W

	Pine Falls Pulp and paper discharge	
08-Sep-98		
302	Brokenhead River at Hwy. 15	49°53.099N/ 96°21.964W
06-Jun-01		
312	Jet Whitetail Trail and James Lane, Petersfield area	50°17.07N/ 96°55.75W
313	Netley Marsh, Petersfield area	50°17.25N/ 96°55.79W
314	L. Winnipeg at Dunnutar, east end of Matlock Road	50°25.95N/ 96°57.10W
12-Jun-01		
338	Sandy Hook	50°32.69N/ 96°58.84W
339	east end of Husavik Road	50°33.97N/ 96°59.37W
340	Miklavik, Marsh Channel	50°35.01N/ 96°59.77W
341	Gimli	50°37.78N/ 96°59.24W
343	Gimli	50°38.42N/ 96°58.98W
345	Camp Rusalka	50°40.05N/ 96°59.40W
4-Sept-01		
425	Traverse Bay off Hwy. 11	50°34.16N/ 96°12.88W