University of Alberta

EFFECTS OF CATCHMENT LAND USE ON NUTRIENT EXPORT, STREAM WATER CHEMISTRY, AND MACROINVERTEBRATE ASSEMBLAGES IN BOREAL ALBERTA

by

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ABSTRACT

Conversion of catchments to agriculture can increase nutrient export to water bodies and subsequently decrease water quality and alter aquatic biota. Two agricultural and four reference watersheds were selected in the Lac La Biche region to evaluate influences of agriculture on nutrient export, stream water chemistry, and macroinvertebrate assemblages. I found that agricultural catchments exported more phosphorus and inorganic nitrogen than reference catchments. Most phosphorus was in a bioavailable form, increasing eutrophication potential. I established that beaver impoundments are common in natural streams in this region; reference streams had large open-canopy areas, large accumulations of silt, low DO and discharges, and macroinvertebrate assemblages indicative of lentic conditions. One agricultural stream lacked impoundments resulting in increased sediment and more lotic conditions and associated macroinvertebrate assemblages. Land conversions will likely result in increased phosphorus export, but effects on stream macroinvertebrate assemblages and sediment export will depend on the presence of beaver impoundments.

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Chapter 1: General Introduction

Major landscape-scale changes are currently occurring at the southern edge of Alberta's boreal region, where forest is rapidly being converted to pasture or cropland. According to census data taken from 1991 to 1996, the boreal transition zone is experiencing a moderate to large increase in the number of farms at its northern boundary (Hiley et al. 2001). Consequences of driving the boreal forest/agricultural interface further north include increasing threats to both water quality and biodiversity in lakes and streams.

It is widely accepted that replacement of forests by agriculture increases watershed nutrient export to aquatic systems (Dillon and Kirchner 1974, Omernik 1977, Beaulac and Reckhow 1982, Jordan et al. 1997, Castillo et al. 2000). Agricultural watersheds have been shown to export up to three times more total phosphorus than forested watersheds (Dillon and Kirchner 1974, Vaithiyanathan and Correll 1992, Cooke and Prepas 1998a). Total nitrogen concentrations have also been shown to be elevated in streams and rivers that drain agricultural land compared to those from forested catchments (Omernik 1977, Keeney and DeLuca 1993, Jordan et al. 1997). Increased leaching of nutrients from agricultural catchments is attributable, in part, to the combination of high levels of soil phosphorus and nitrogen with large amounts of surface runoff and erosion (Abrams and Jarrell 1995, Haygarth and Jarvis 1999, Follett and Delgado 2002, Hansen et al. 2002). Elevated levels of phosphorus and nitrogen in agricultural soils are mainly the result of inputs of synthetic fertilizer and animal waste (Sharpley and Menzel 1987, Tisdale et al. 1993). Not all agricultural land exports similar amounts of nutrients, however; export can depend on natural hydrology, soil type, catchment slope, percent of the catchment as wetland, percent of the catchment as agriculture, level of agricultural intensity, and local precipitation patterns within the watershed (Dillon and Kirchner 1974, Munn and Prepas 1986, Abrams and Jarrell 1995, Foy and Withers 1995, D'Arcy and Carignan 1997, Cooke and Prepas 1998a, Haygarth and Jarvis 1999, Reed and Carpenter 2002, Winter et al. 2002, Whitson et al. 2004).

Elevated transport of nitrogen and phosphorus from agricultural catchments can contribute to eutrophication of streams, as evidenced by increases in stream productivity of algae (often blue-green algae) and macrophytes (Foy and Withers 1995, Carpenter et al. 1998). Additional impacts of forest conversion to agricultural land on streams draining these catchments include loss of riparian streamside vegetation (Smith 1992). This can exacerbate increases in primary productivity via enhanced light availability and stream temperatures (Smith 1992, Sweeney 1992, Rutherford et al. 1997). Decreases in riparian vegetation also result in lowered allocthonous inputs to streams (Smith 1992, Richards and Host 1995, Townsend et al. 1997, Griffith et al. 2002). Further, a decrease in the permeability of terrestrial substrates associated with conversion of forested land to agricultural land has the potential to alter natural flow regimes, particularly during high precipitation events and snowmelt (Dillon and Kirchner 1974, Soranno et al. 1996, Allan et al. 1997, Jordan et al. 1997, Castillo et al. 2000, Neill et al. 2001).

In-stream physical and chemical changes resulting from agricultural activities together with increased primary productivity, have implications for organisms at higher trophic levels (Harding et al. 1998). Macroinvertebrates reflect in-stream changes through shifts in functional feeding groups, overall richness, ratios of intolerant to tolerant taxa, and abundance (Lenat and Crawford 1994, Richards et al. 1997, Townsend et al. 1997, Harding et al. 1998, Sponseller et al. 2001, Black et al. 2004). Highly productive systems often exhibit extreme fluctuations in dissolved oxygen that eliminate taxa sensitive to anoxic conditions (Lenat and Crawford 1994). Overall, agriculturally influenced streams demonstrate shifts to greater richness and abundance of tolerant groups such as Chironomidae and Oligochaeta (Rosenberg and Resh 1993, Lenat and Crawford 1994). Because macroinvertebrates are ubiquitous, respond predictably to in-stream characteristics and have wide ranges of tolerances they are useful bioindicators of effects of land use change on streams (Rosenberg and Resh 1993).

Increased nutrient and sediment export from agricultural watersheds can lead to eutrophication of downstream lakes as well (Foy and Withers 1995). In the town of Lac La Biche, Alberta, residents have voiced concerns over recent declines in Lac La Biche Lake water quality. This public concern has been the motivation for investigating the potential causes of perceived lake water quality declines, of which my research is a part. Although shallow boreal lakes in this region are naturally eutrophic, addition of nutrients has been followed with increases in blue-green algal abundance (Kotak et al. 2000) and subsequent declines in dissolved oxygen (Babin and Prepas 1985). Algal blooms contribute to summer fish kills, foul odours, unpalatability of drinking water, and overall declines in water quality (Carpenter et al. 1998). Data from sediment cores collected from Lac La Biche Lake in 2003 indicate an increase in phosphorus concentration of approximately two times above pre-settlement values, with rapid increases in the last several years (Schindler, unpublished data). Additionally, current values for sediment deposition are estimated as 5-fold higher than values in the early 20th century, again with higher accumulation in the last several years (Schindler, unpublished data). It is likely that changes in catchment land use, including conversion of forested land to agricultural land, has influenced these values, as increased nutrients and sedimentation are associated with increased agriculture at the catchment scale in other regions (Dillon and Kirchner 1974, Johnes et al. 1996, Soranno et al. 1996, Allan et al. 1997, Townsend et al. 1997, Castillo et al. 2000, Neill et al. 2001, Reed and Carpenter 2002). Increases in nutrient concentrations are likely also influenced by sewage discharge into the lake from the Lac La Biche town. Although a previous study conducted in the same region has shown that catchment conversion to agriculture is affecting stream water quality through nutrient enrichment (Cooke and Prepas 1998a), there is insufficient data outlining the effects of such land conversion on boreal stream flow patterns, water quality, and macroinvertebrate assemblages to inform management decisions.

My objectives were to determine: 1) the influence of agricultural land use on phosphorus and nitrogen export to boreal streams; 2) the influence of catchment land use on in-stream physical and chemical features and structure of macroinvertebrate assemblages; 3) the feasibility of using macroinvertebrates as bioindicators of stream water quality. These objectives are addressed in the following two chapters. In the first chapter, agricultural catchments are compared to forested reference catchments in terms of overall nutrient loading, nutrient speciation, and the timing of nutrient loading. The second chapter investigates the relationships between catchment land use, in-stream physical and chemical parameters, and macroinvertebrate assemblages. Overall conclusions summarizing this thesis are discussed in the last chapter. Results of this study are intended to inform the Lac La Biche Lake Watershed Steering Committee on the need for managing land use development and protecting water quality in Lac La Biche Lake and surrounding streams. In addition, my research will supplement current understandings of chemical and biotic responses in small boreal streams to catchment conversion from boreal forest to agricultural land.

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Chapter 2: Effects of catchment land use on phosphorus and nitrogen export in boreal streams

2.1 Introduction

Increases in nitrogen and phosphorus loading to lakes and streams has long been recognised as a threat to aquatic ecosystems (USEPA 2000). In the boreal ecoregion of Alberta, declines in lake water quality are thought to be the result, in part, of increased nutrient inputs (Mitchell and Trew 1992). Although shallow lakes in this region are naturally eutrophic, increases in nutrients have been shown to cause declines in dissolved oxygen concentrations (Babin and Prepas 1985), increases in blue-green algal abundance (Kotak et al. 2000), and declines in overall water quality. With progress on reducing point sources of nutrient pollution to aquatic systems (Sharpley et al. 1994), diffuse sources have become an increasing concern. In particular, change in catchment land use has been cited as one of the primary causes of increased phosphorus and nitrogen inputs to downstream water bodies (Omernik 1977, Beaulac and Reckhow 1982, Mitchell and Sosiak 1991, Sidle and Sharpley 1991, Mitchell and Trew 1992, Kauppi et al. 1993, Sharpley et al. 1994, Soranno et al. 1996, Jordan et al. 1997, Anderson et al. 1998a, b, Cooke and Prepas 1998, Pionke et al. 1999, Castillo et al. 2000). Agricultural watersheds can export up to three times more total phosphorus than forested watersheds (Dillon and Kirchner 1974, Vaithiyanathan and Correll 1992, Cooke and Prepas 1998). Total nitrogen concentrations are also higher in streams and rivers draining agricultural land as compared to forested catchments (Omernik 1977, Keeney and DeLuca 1993, Jordan et al. 1997).

There are several reasons why phosphorus and nitrogen export is higher from agricultural catchments than from forested catchments. Phosphorus and nitrogen are removed from agricultural land when crops are harvested and therefore, in order to sustain future crop yields, these nutrients must be continually added to the system. This renewal of nutrients is accomplished through fertilizer addition, which may be in a synthetic or animals waste form (Sharpley and Menzel 1987, Sharpley and Smith 1995, Haygarth and Jarvis 1999). Synthetic fertilizers that supply nitrogen are usually ammonia based, though most of the ammonia is converted to nitrate by nitrifying soil microbes

(Havlin et al. 2005). Most nitrogen in manure is organic; inorganic nitrogen in manure is usually dominated by ammonia but nitrogen speciation ratios are affected by soil contact time (Cabrera and Gordillo 1995, Robinson et al. 1995). Depending on the type, time, and method of nutrient additions, only minimal amounts of added phosphorus (Foy and Withers 1995) and nitrogen (Duxbury et al. 1993) may be assimilated by crops. Phosphorus, being less mobile than nitrogen, has been shown to accumulate in soil over time, and this is well documented in agricultural areas in Europe (Haygarth and Jarvis 1999) and the USA (Sims 1992, Sharpley et al. 1994). Increased nutrient concentrations in agricultural soil lead to higher nutrient concentrations in surface runoff and, along with loss of nutrients through erosion, higher overall export from the land base relative to forested areas.

Erosion and surface runoff are the main mechanisms by which phosphorus is lost from agricultural soils. Most research has shown that phosphorus exported from agricultural land is in a particulate form (phosphorus adsorbed to soil particles and bound in organic matter) that is carried to streams/rivers via surface runoff (Omernik 1977, Sharpley and Menzel 1987, Vaithiyanathan and Correll 1992). In contrast, phosphorus exported from forested land is generally in the dissolved form, which is more bioavailable (Sharpley et al. 1994). The form of phosphorus that dominates annual export can vary among watersheds, however. Vaithiyanathan and Correll (1992) found that particulate phosphorus dominated phosphorus export from both forested and agricultural subcatchments in the Rhode River watershed. Cooke and Prepas (1998) found that in agricultural catchments, most phosphorus was exported in the dissolved form whereas in forested catchments particulate phosphorus dominated the export. Differences in catchment slopes, susceptibility of the soil to erosion, crop development stage, and agricultural intensity can also account for variation in the form of phosphorus exported to water bodies (Sharpley et al. 1992, Anderson et al. 1998a, Cooke and Prepas 1998). For example, a study of 27 watersheds throughout Alberta indicated that the ratio of dissolved phosphorus to total phosphorus in streams was positively related to agricultural intensity (Anderson et al. 1998a). Also agricultural soils with more advanced crops are less susceptible to erosion and transport of phosphorus through this mechanism will likely be reduced (Sharpley et al. 1992).

The capacity of the soil to sorb phosphorus will also influence the form of phosphorus transported from the catchment (Haygarth and Jarvis 1999). Phosphorus sorbtion is highly variable and depends on pH and the clay and organic matter content of the soil; soils with higher clay and organic matter sorb more phosphorus (Abrams and Jarrell 1995), and phosphorus transport from these soils is likely to occur mainly through erosion or physical movement of the soil.

Nitrogen is more mobile than phosphorus and thus its transport within and from agricultural systems is difficult to follow (Delgado 2002, Follett and Delgado 2002). It has been estimated that in general 20% of the nitrogen applied to a crop as fertilizer is returned to the atmosphere via denitrification processes, 5% is lost in surface runoff, and 25% is lost through leaching (Duxbury et al. 1993). Nitrogen mobility depends on speciation, however. Nitrate is very mobile within the soil profile and is transported mainly through leaching (Randall and Mulla 2001, Follett and Delgado 2002); very little is lost from the soil via surface runoff (Jackson et al. 1973). Most other species of nitrogen are exported to water bodies dissolved in surface runoff water, adsorbed to eroding sediments (NH_4^+) or bound in eroding soil organic matter (organic forms of N and NH_4^+) (Follett and Delgado 2002).

Similar to phosphorus, the species of nitrogen that dominates export depends on catchment characteristics and processes operating within the catchment. This includes the type of fertilizer added and whether soil conditions are favourable for bacterial nitrification. Although nitrogen fertilizer is usually ammonia-based, in moist, well-aerated soils much of the ammonium is converted to nitrate by nitrifying bacteria (Havlin et al. 2005). Similarly, inorganic nitrogen in fresh manure is dominated by ammonia but prolonged soil contact results in bacterial conversion of ammonium to nitrate, particularly if conditions favour nitrification (Cabrera and Gordillo 1995). Increased nitrate in aquatic systems has been directly linked to intensive use of chemical fertilizers and increasing application of animal manure (Dillon et al. 1991, Sharpley et al. 1994, Van Herpe and Troch 2000). In addition, the ratio of dissolved inorganic nitrogen to total nitrogen exported has been shown to increase with increases in agricultural intensity in watersheds throughout Alberta (Anderson et al. 1998a).

Export varies among catchment land use types (forested or agricultural) and temporally within the same catchment (Sharpley et al. 1994, Johnes 1996, Haygarth and Jarvis 1999). For example, Dillon (1991) found that in forested catchments in central Ontario, mean annual export of TP ranged from 1.8 to 25.5 kg km^{-2} yr^{-1} over 12 years and Munn and Prepas (1986) found that forested catchments on the Boreal Plain of northern Alberta ranged in export from 7.5-13 kg km^{-2} yr^{-1} over 1 year. The same variability is found in agricultural catchments. Reed and Carpenter (2002) found that TP export varied from 18-69 kg km^{-2} yr^{-1} in six agricultural catchments in Wisconsin. On the Boreal Plain of Alberta, Cooke and Prepas (1998) found that high variability existed in TP export for agricultural (14-57 kg km^{-2} yr^{-1}) and forested catchments (13-22 kg km^{-2} yr^{-1}). Further, the TP export from these catchments varied among years for all watersheds. This same pattern is found with nitrogen export as well. For example, nitrate loads from an agricultural watershed of the Minnesota River varied by an order of magnitude between years (Randall and Mulla 2001). Winter et al. (2002) found that TN export from mixed agricultural subcatchments within the Lake Simcoe watershed ranged from 220 to 790 kg km^{-2} yr^{-1} while export from catchments with a high proportion of forest and scrubland ranged from 170 to 270 kg km^{-2} yr^{-1}. Factors contributing to variable nitrogen and phosphorus export from agricultural and forested catchments include year-to-year changes in climate (timing and amount of precipitation), natural hydrology, soil type, catchment slope, percent of the catchment as wetland, percent of the catchment as agriculture, and the intensity and type of agriculture within the watershed (Dillon and Kirchner 1974, Munn and Prepas 1986, Abrams and Jarrell 1995, Foy and Withers 1995, D'Arcy and Carignan 1997, Anderson et al. 1998a, Cooke and Prepas 1998, Haygarth and Jarvis 1999, Reed and Carpenter 2002, Winter et al. 2002, Whitson et al. 2004).

Seasonal variation in nutrient exports also differs between agricultural and forested reference streams and often depends on regional climate patterns. In catchments in east-central Pennsylvania, most of the phosphorus from agricultural catchments was exported during storm events (Gburek and Pionke 1995, Pionke et al. 1999). Similarly, on the Boreal Plain of Canada most phosphorus export occurred in conjunction with summer storm events when runoff was highest (Cooke and Prepas 1998). In Pistern Hill catchment in the Midlands, UK, storm-flow phosphorus concentrations were over six times greater than in baseflow (Heathwaite and Dils 2000). In contrast, delivery of phosphorus from heavily fertilized cropland to Lake Ontario showed that snowmelt during spring runoff was the most important hydrologic factor in annual phosphorus loading (Longabucco and Rafferty 1989). Additionally, the dominant export mechanism can depend on speciation; most nitrate is lost through leaching during high rainfall events whereas organic nitrogen is often transported in overland flow (Nikolaidis et al. 1998, Follett and Delgado 2002). Variability in local and seasonal climate patterns were responsible for the differences in these studies.

The low sloping topography that is characteristic of watersheds in the Boreal Plain of western Canada decreases erosion potential and impedes a main mechanism of phosphorus transport. Despite this, phosphorus loading remains a potential problem. The low leaching potential of agricultural soils combined with potentially high phosphorus concentrations compared to forested soils (Whitson 2003) may result in increased loading of dissolved phosphorus to downstream water bodies. Additionally, because snowmelt during spring thaw and rain events causes increased surface runoff (Syversen 2002), potential for increased loading during these events is high. Lower interception by vegetation and faster snowmelt also lead to increased overland flow and loading in catchments with high agricultural land use (Syversen 2002, Whitson et al. 2004). Since the main mechanism of phosphorus transport from agricultural soil is surface runoff and erosion, phosphorus loading should be higher from agricultural catchments (Sharpley et al. 1992, Sharpley et al. 1994)

More research is needed to determine if increases in agricultural land use in the catchment leads to subsequent increases in nutrient loading to streams in the Lac La Biche region. This is of particular interest because perceived declines in Lac La Biche lake water quality are thought to be, in part, the result of increased nutrient inputs. Additionally, understanding which hydrological events are responsible for the majority of nutrient export, and what the main speciation of exported nutrients is, will aid in nutrient management within agricultural catchments. My objectives were to determine if the proportion of agricultural land in the catchment affects: 1) the amount of phosphorus and nitrogen exported in the Lac La Biche region; 2) the forms of exported phosphorus and nitrogen; and 3) the timing of phosphorus and nitrogen export to streams. The Lac La

Biche region offers an excellent case study for all of my objectives due to recent public concern over degradation of lake water quality, availability of reference watersheds with little anthropogenic influences, high conversion rate of forested land to agricultural land in the watershed, and the hypothesized influence of land use change on nutrient loading to the lake.

2.3 Materials and Methods

2.3.1 Study area

The Lac La Biche watershed is within the mid boreal mixedwood ecoregion (Strong 1992) of north-central Alberta. It drains an area of 4040 km^2 (Mitchell and Prepas 1990). I selected stream catchments where the degree of agricultural land use was the primary difference between catchments. Of the nineteen streams that drain directly into Lac La Biche, only four flow continuously throughout summer, and of these only three were suitable for study. An additional three stream catchments, located just north of the Lac La Biche watershed, were included bringing the total to 6 study catchments (2 agricultural (A) and 4 reference (R); Figure 2.1). Catchments were classified as agricultural if greater than 1/3 of the catchment consisted of agricultural land. To delineate study catchment boundaries, I used a combination of a digital elevation model (DEM) and ArcGIS spatial analysis tools (ArcGIS 1999-2004). Land use in the stream catchments was quantified based on 2002 Landsat 7 EMF Imagery (Sanchez-Azofeifa 2002) and spatial analysis tools (ArcGIS 1999-2004) (Figure 2.1, Table 2.1). Land cover classifications were based on the Alberta Ground Cover Characterization (AGCC) land cover classification system (Sánchez-Azofeifa 2005). Accurate classification of land cover is estimated at 90% for the land classes used in this study (Sánchez-Azofeifa 2005). For this reason land cover data were not validated with ground data. Agricultural watersheds consisted of a mixture of non-row-cropland (hay and cereals), pasture, and small cow-calf operations. However, the exact proportion of each type of agriculture was not quantified. Intensity of agriculture in these watersheds is moderate based on total chemical expenses (26 to 39th percentile) and manure distribution (40-60th percentile) (Data from Soil Landscape of Canada 1991 Census Data).

The agricultural streams were Mission and Plamondon and the reference streams were Reutov, Deer, Cadieux, and Goldie. Blackbird Creek is the local name for the agricultural stream that I refer to as Mission. Streams were all 2nd and 3rd order depending on the time of the year. The agricultural catchment areas were 46 and 111 km^2 and the reference watershed areas were 28, 47, 48 and 109 km^2. All streams, with the exception of one of the agricultural streams (Plamondon), supported numerous beaver dams. Water sampling locations were selected based on lack of immediate cattle access and influence of beaver dams and location of adjacent roads or culverts. Precipitation data were obtained from an Environment Canada (2005) meteorological station within the town of Lac La Biche (54° 46' N, 112° 1' W). Regional long-term averages were obtained from an Athabasca meteorological station (54° 49' N, 113° 31' W) (Environment Canada, 2005).

The study duration was a calendar year starting from the beginning of May 2003 until the end of April 2004. This was due to difficulties in finding appropriate study catchments before the spring runoff period of 2003.

2.3.2 Field sampling: Summer 2003

Water samples were collected in the middle of each stream at mid depth with acid-washed polystyrene and Nalgene polyethylene containers twice monthly from May to the end of August 2003 (n= 8 samples/stream). This was supplemented by sampling during storm events (n = 3). Stream discharge was quantified weekly by measuring velocity (Swoffer 2100 current meter) and water depths at set intervals across the stream channel. In streams where velocity was too low to obtain a reading with the Swoffer 2100, surface velocity was estimated by observing the time taken for a submersed orange to travel a known distance downstream (Hauer and Lamberti 1996, USEPA 2004). Level sensors (Stevenson recorders) were set up on one of the agricultural streams (Plamondon) and all of the reference streams (Deer, Reutov, Goldie, and Cadieux) at the end of May 2003. Water levels for Mission stream were obtained from a downstream level sensor, which was set up as part of another study.

2.3.3 Field sampling: Spring runoff 2004

Streams were sampled in early April 2004 at the first sign of flow. An ice auger was used to determine if under-ice flow was occurring. Water samples were collected in the middle of each stream at mid-depth with acid-washed polystyrene and Nalgene polyethylene containers. To account for diurnal fluctuation in nutrient concentrations due to freeze and thaw cycles (Anderson et al. 1998b), all streams were sampled within a three hour period in the late afternoon. This was done once a day for 10 sampling days during the spring runoff period, which lasted from April 3-April 21 ($n = 10$). Due to the nature of the watersheds, spring runoff did not occur simultaneously for all streams and therefore sampling occurred earlier in some streams. Stream discharge was quantified using the same methodology employed during summer 2003. Level sensors were set up in the same locations as in the summer 2003 sampling period. I estimated stream discharge based on relationships derived from the depth-discharge regression equation on Plamondon. On all streams except Plamondon depth-discharge relationships were poor due to beaver activity. For these streams, I estimated discharge for the days when discharge was not measured, by interpolating between data points.

2.3.4 Water chemistry analysis

Samples were analyzed for total nitrogen (TN), total dissolved nitrogen (TDN), nitrate (NO_3^-) plus nitrite (NO_2^-), ammonium (NH_4^+), total phosphorus (TP), and total dissolved phosphorus (TDP) within 48 hours of collection. TP and NH_4^+ concentrations were determined from unfiltered stream water, whereas TDP, and $NO_3^-$and NO_2^-, were determined on water samples filtered through a 0.45-μm Millipore GF/F filter (University of Alberta Limnology Lab). I did not assess soluble reactive phosphorus (a measure of orthophosphate) because immediate analysis of samples was not possible. Nitrite is usually negligible in surface water so it was assumed that all oxidized nitrogen was nitrate (Cohn et al. 1999).

2.3.5 Calculation of P and N mass export, mass load, and FWMC

I calculated mass loads of total phosphorus and nitrogen by multiplying sample concentration by the instantaneous stream discharge at the time the sample was taken and then by the length of time that the sample represented (Cooke et al. 2000). In summer, the time intervals were generally equivalent to one-half of the time interval of the current sample and the preceding sample, and one-half of the time interval between the current sample and the following sample (Cooke et al. 2000). However if sampling occurred during a storm event when flow and concentrations were elevated I adjusted the time interval based on how long the water level (obtained from Stevenson level recorders) remained elevated. I assumed that strong depth-discharge relationships occurred only during short intervals of increased surface runoff, such as during a storm event. Although these assumptions and estimations may affect the accuracy of calculation, this approach was deemed an acceptable approximation because long-term depth discharge relationships could not be established. The total mass export (as kg km^{-2} yr^{-1}) (the amount of nutrient lost per km^2 of watershed) was calculated for TP, TDP, TN, NO_3^-, and NH_4^+ by dividing the total mass load (kg yr^{-1}) by the effective drainage area. Effective drainage area was calculated as the total watershed area minus the area of water (Cooke et al. 2000). To adjust for differences in flow volumes among streams, flow-weighted mean concentrations (FWMCs, mg/L) were calculated annually as well as separately in the summer and spring sampling periods. FWMCs were calculated by dividing total mass load estimate (kg yr^{-1}) by the total flow volume for a given time.

Accurate mass load and export estimates usually require a large number of paired concentration and discharge data points (Anderson et al. 1996, Anderson et al. 1998a). When concentration data are sparse, reasonably accurate estimates can be obtained if daily flow data are available (Mitchell and Sosiak 1991). In my study, continuous flow data were not obtained in summer and a limited number of concentration data were collected. This decreases the accuracy of my export estimates during this period. Loading estimates for the spring runoff period are more accurate because daily flow data were available and more concentration points were obtained. However, other studies quantifying nutrient export from watersheds involved daily water sampling supplemented by more frequent (often hourly) sampling during storm events combined with continuous measurement of flow (Beaulac and Reckhow 1982, Dillon et al. 1991, Vaithiyanathan and Correll 1992, Anderson et al. 1998b, Cooke and Prepas 1998). Catchment nutrient exports calculated in my study provide a relative indication of the potential for increased agricultural land to affect nutrient exports. The comparison between agricultural and reference catchments in my study was based on comparison of mass export (export coefficients). Mass load and FWMC comparisons supplemented this. Due to limitations of the study design, data were analysed using simple comparative methods. Exports, mass loads, and FWMCs for each agricultural catchment were compared to each of the reference streams and reported as the average of the differences. I also explored the relationship between instantaneous discharge and mean concentrations for each of my streams.

2.4 Results

2.4.1 Hydrology

Total annual precipitation in the study region in 2003 and 2004 was 560 mm and 500 mm, respectively, which was above or equivalent to the long-term (1970-2000) average for the region (504 mm) (Environment Canada). However, 2002 was a drought year with only 244 mm of precipitation. Precipitation for the 2003 sampling period (May-August) was 378 mm and above long term summer average (304 mm). Three storm events accounted for the majority of summer precipitation: June 2 and 3 (94 mm), June 21 and 22 (53 mm) and July 5-9 (59 mm) (Figure 2.2). I sampled during (June 3 and July 9) or within days (June 24) of these rain events. Snowfall for the winter (November to March) preceding the spring sampling period (April 2004) was 96 mm, which was slightly less than regional long-term average of 108 mm. Annual discharges for agricultural streams were 1533 x 10^3 m^3 (Plamondon) and 1010 x 10^3 m^3 (Mission). Annual discharges for reference streams ranged from 671-7627 x 10^3 m^3.

2.4.2 Total phosphorus mass export, mass load, and FWMCs

On average, agricultural catchments exported 2.5 (Plamondon) and 2.9 (Mission) times more total phosphorus than reference catchments (Figure 2.3, Table 2.1). Phosphorus export from reference catchments ranged from 2.1 to 4.1 kg km^{-2} yr^{-1} whereas export from agricultural catchments was 7.9 kg km^{-2} yr^{-1} (Plamondon) and 9.3 kg km^{-2} yr^{-1} (Mission). On average, mass load (kg yr^{-1}) of total phosphorus was 2.8 (Mission) to 5.6 (Plamondon) times higher in agricultural catchments than in reference catchments (Figure 2.4). However one reference catchment (Deer) exported a similar annual mass (~400 kg) of phosphorus as one of the agricultural streams (Mission). The remainder of the reference catchments exported only 98 to 150 kg of phosphorus annually and the other agricultural catchment (Plamondon) exported 798 kg of phosphorus annually. FWMCs of TP and TDP were consistently higher in water draining agricultural catchments compared to reference catchments. On average, TP and TDP FWMCs in agricultural catchments were 2.1-2.7 fold higher than in reference catchments. This was true both on an overall annual basis, and when considering spring and summer sampling periods separately. TP concentrations were not significantly related to instantaneous discharge and relationships were negative for all streams (linear regression, $p > 0.10$, $R^2 = -0.001$ to -0.596).

2.4.3 Total nitrogen mass export, mass load, and FWMCs

Nitrogen export from agricultural catchments was on average 1.1 times (Plamondon) to 1.7 times (Mission) higher than reference catchment exports (Figure 2.3, Table 2.1). Nitrogen export from reference catchments ranged from 25.3 to 42.9 kg km^{-2} yr^{-1}, whereas export from agricultural catchments was 35.5 (Plamondon) and 54.7 kg km^{-2} yr^{-1} (Mission). Mass load patterns for nitrogen were similar but differences were not as strong: on average, agricultural catchments exported 1.6 (Mission) to 2.4 (Plamondon) times more nitrogen than reference catchments, however one reference catchment (Deer) exported more (2519 kg) nitrogen annually than one of the agricultural catchments (Mission, 2401 kg). The other agricultural catchment (Plamondon) exported 3577 kg of nitrogen annually and the remaining reference catchments exported 1104 to 1910 kg. Total nitrogen annual FWMCs were consistently higher in water draining agricultural catchments compared to reference catchments. On average, total nitrogen FWMCs of agricultural catchments were 1.3 to 1.8 fold higher than reference catchments; this was true annually and in the summer and spring sampling periods. TN concentrations were not significantly related to instantaneous discharge for all streams and all the relationships were negative ($R^2 = -0.018$ to -0.560, $p > 0.10$).

2.4.4 Form of phosphorus and nitrogen export

All catchments exported the majority of phosphorus (>62%) as dissolved phosphorus (Figure 2.4). Although agricultural catchments exported more phosphorus overall, the percentage of phosphorus exported as dissolved or particulate was similar among land use classes. Agricultural catchments exported 23% and 25 % as particulate phosphorus and reference catchments exported 19-37 % in this form. The form of nitrogen exported differed between agricultural catchments compared to reference catchments. Specifically, on average ammonium export was 4.6 (Mission) to 11.5 (Plamondon) times greater in agricultural catchments compared to reference catchments. Similarly, on average nitrate export was 8.7 (Plamondon) to 24.5 (Mission) times greater in agricultural catchments compared to reference catchments. Both agricultural catchments exported 6.6 times more particulate nitrogen than the reference catchments.

Particulate nitrogen made 10% (Mission) and 14% (Plamondon) of the nitrogen exported from agricultural catchments compared to 0.8 to 9% of the nitrogen exported from reference catchments. Ammonium made up 4% (Mission) and 16% (Plamondon) of total nitrogen exported from agricultural catchments and 0.7 to 5% from reference catchments. Nitrate comprised a greater percentage (4 and 8%) of export from agricultural catchments than from reference catchments (< 1%). Agricultural streams exported slightly less nitrogen in dissolved form (85 and 90%) than reference streams (91-99%). Also, dissolved organic nitrogen comprised a greater percentage of the annual export from reference catchments (85-98%) than from agricultural catchments (65 and 79%).

2.4.5 General patterns in timing of nutrient export

The timing of phosphorus and nitrogen export was variable among catchments and there were no clear differences between agricultural and reference catchments (Figure 2.5). Spring runoff was a very important period for export of inorganic nitrogen from both agricultural watersheds and two of the reference watersheds. Spring runoff was particularly important in the Plamondon catchment (A) and the Deer catchment (R); at least 45% of the annual export for each nutrient form was exported in this 18-day period. To determine how important major storm events were I calculated the percent of phosphorus and nitrogen in different fractions transported during 3 major storm periods in summer 2004 (June 2-4, June 21-25, and July 5-10). The percentage of phosphorus exported in different fractions during storm events was not different between agricultural and reference catchments. These events accounted for less than 25% of the total annual export in agricultural streams. In two of the reference streams, a large precipitation event on June 2 and 3, 2004 (Figure 2.2) increased discharge and subsequently led to beaver dam "blow-outs". These "blow-outs" accounted for 9 and 33% of the annual P load for these streams. Base flow contributed 10-60% of TP load and 18-63% of TN load for streams, but again there was a clear difference between agricultural and reference streams. To assess relationships between nutrient speciation and hydrology I examined patterns of each catchment individually.

2.4.6 Timing of P export from agricultural (A) and reference (R) catchments

Plamondon (A): 50-60% of phosphorus (TP and TDP) was exported during spring runoff. Less than 20% of phosphorus (TP and TDP) export occurred during storm events.

Mission (A): Less than 25% of phosphorus (TP and TDP) was exported during spring runoff and approximately 50% of phosphorus export occurred during baseflow.

Reutov (R): Less than 30% of phosphorus (TP and TDP) export occurred during storm events. Most phosphorus was exported during baseflow.

Deer (R): Approximately 70% of phosphorus (TP and TDP) was exported during spring runoff. Less than 25% of phosphorus (TP and TDP) export occurred during storm events and during beaver dam breaks.

Cadieux (R): Less than 5% of phosphorus (TP and TDP) export occurred during spring runoff. Most phosphorus was exported during baseflow and beaver dam breaks accounted for approximately 20% of annual export.

Goldie (R): Less than 10% of phosphorus (TP and TDP) export occurred during spring runoff. 30-40% of phosphorus (TP and TDP) export occurred during storm events, the highest for all catchments.

2.4.7 Timing of N export from agricultural (A) and reference (R) catchments

Plamondon (A): 95% of ammonium, 75% of nitrate, and 50% of particulate nitrogen were exported during spring runoff. Less than 30% of export for all fractions of nitrogen occurred during storm events. Dissolved organic nitrogen was primarily exported during base-flow conditions, a pattern not observed for any other nutrient form.

Mission (A)**:** 95% of nitrate and approximately 55% of ammonium was exported during spring runoff whereas less than 25% of dissolved organic and particulate nitrogen were exported then. Less than 20% of all fractions of nitrogen were exported during storm events and no nitrate was exported during storm events. Most dissolved organic and particulate nitrogen were exported during baseflow.

Reutov (R): 75-80% of inorganic nitrogen was exported during spring runoff whereas less than 30% of other nitrogen fractions were exported during this time. Less than 25% of nitrogen export occurred during storm events. Most dissolved organic and particulate nitrogen were exported during baseflow.

Deer (R): Spring runoff accounted for over 90% of inorganic and dissolved organic nitrogen annual export. Approximately 60% of particulate nitrogen export occurred during spring runoff. Less than 10% of nitrogen export in all fractions occurred during storm events and beaver dam breaks.

Cadieux (R): Spring runoff accounted for less than 10% of nitrate and particulate nitrogen annual export. 20-35% of ammonium and dissolved organic nitrogen export occurred during spring runoff. Less than 10% of nitrogen export occurred during storm events. Beaver dam breaks accounted for 20-25% of annual particulate, nitrate, and ammonium export. Over 50% of nitrogen in all fractions was exported during baseflow.
Goldie (R): Spring runoff accounted for less than 10% of nitrate and particulate nitrogen annual export. 20-25% of ammonium and dissolved organic nitrogen export occurred during spring runoff. Approximately 30% of nitrogen export occurred during storm events for all nitrogen fractions.

2.5 Discussion

2.5.1 Total phosphorus and nitrogen export, mass load, and FWMC

Similar to previous research, phosphorus export was higher in agricultural catchments than in reference catchments (Dillon and Kirchner 1974, Vaithiyanathan and Correll 1992, Anderson et al. 1998a, Cooke and Prepas 1998). This was likely because inputs of phosphorus in agricultural systems were artificially increased through synthetic fertilizer and/or cattle manure inputs (Sharpley and Smith 1995, Follett and Delgado 2002, Hansen et al. 2002, Lemunyon and Daniel 2002, Weld et al. 2002). However, the exports that I estimated for agricultural watersheds were lower than what has been reported from studies in the Boreal Plains of Alberta. Specifically, Cooke and Prepas (1998) reported export coefficients of 14 and 57-82 kg km^{-2} yr^{-1} in 1994 and 12 and 34-57 in 1995 for two agricultural catchments. Some of these values are much higher than the exports I calculated (7.9 and 9.3 kg km^{-2} yr^{-1}). This can be partially attributed to higher percent agricultural land use in their catchments (>60% compared to <40%) and potential for point source loading by a cow-calf operation that was in close proximity to one of their streams. Furthermore, their study years (1994 and 1995) were relatively wet years, thus overland flow and nutrient leaching potential were most likely higher. In contrast, the year previous to this study (2002) was one of extreme drought. This likely resulted in a low water table and increased soil infiltration capacity, which subsequently caused decreased surface runoff in 2003. Decreased surface runoff would likely result in reduced nutrient export (Hansen et al. 2002, Lemunyon and Daniel 2002). Also, reduced water tables can cause an effective reduction in the drainage area of catchments, particularly in catchments that are characterised by low sloping topography and many depressional areas, such as my study area (Prairie and Kalff 1986). This occurs mainly because there is a loss of interconnection between pools of surface water. Reductions in effective drainage area would cause underestimations of export coefficients unless catchment size was adjusted, a difficult endeavour. Export coefficients for forested reference watersheds were also lower than what others have found. Cooke and Prepas (1998) found 5-22 kg km^{-2} yr^{-1} total phosphorus export as compared to my <4.1 kg km^{-2} yr^{-1}. This reinforces the suggestion that the low yields found in my study are attributable to climatic variation (low precipitation and subsequent reduced runoff). Likewise, the observation that discharge/instantaneous-concentration relationships were poor and negative also implies that runoff events were minimal in my study year as compared to those of other studies (see Anderson et al. 1998a). It is important to recognize, however, that differences in export coefficients between my study and other regional studies could be due to differences in sampling intensity.

In contrast to total phosphorus export, total nitrogen export coefficients were variable among watersheds and independent of catchment land use type. There are several explanations for these results. The current Alberta Agricultural Operation Practices Act (2002) identifies nitrogen as the limiting nutrient in the soil profile in Alberta, including the Boreal Plain. To meet the nitrogen requirements of typical crops grown in this region, manure is applied to agricultural soil, which has a high P:N ratio (Robinson et al. 1995, Havlin et al. 2005). This can result in the application of significantly more phosphorus than is required for crop growth (Tisdale et al. 1993, Robinson et al. 1995, Hansen et al. 2002). For this reason, excess phosphorus may be available for export while soil nitrogen would remain relatively low (Cabrera and Gordillo 1995). Census data from Soil Landscape of Canada show that manure inputs were higher than chemical expenses in my agricultural watersheds in 1991. Consistent with this, the 2001 Census of Agriculture data shows that farm type within Lakeland County is dominated by cattle operations rather than crop production. In support of the preceding argument, this suggests that manure inputs may be more important than fertilizer inputs in my study area. Further research is required to confirm this because soil phosphorus levels, types of fertilizer, application rates, and types of crop grown in agricultural watersheds were not quantified in this study.

The validity of using mass export as the sole indicator of agricultural land-use effects on surface water can is questionable. Nutrient transport may behave differently depending on watershed size and therefore estimating export can be difficult and uncertain (Beaulac and Reckhow 1982, Frink 1991, Soranno et al. 1996). Mass export can also be limited because effective catchment size can vary temporally and, depending on the dominant hydrological pathway, disproportionately among catchments (Prairie and Kalff 1986). Mass loads can aid in determining the magnitude of impact on a downstream water body but are also susceptible to catchment size. I found that agricultural catchments exported a greater mass of total phosphorus than most reference catchments (406 kg and 798 kg as compared to 98-150 kg), but the Deer (R) catchment exported a phosphorus mass load that approached the mass load exported from Mission (A) catchment (403 kg, Table 2.1). This was likely attributable to the Deer catchment being 2.3 times larger than the Mission catchment. The high influence of catchment size suggests that mass export, despite difficulties in determining active drainage area, is a more effective comparative tool in this region.

Flow-weighted mean comparisons should also be used when comparing nutrient movement among watersheds because both mass loads and mass exports can be affected by differences in discharge volume between catchments. In a study of 27 watersheds throughout Alberta, Anderson et al. (1998) found that mass load and export tended to be highest in streams with higher discharge. Similar to other studies of Albertan streams, I found that flow-weighted mean concentrations were higher in agricultural streams (Anderson et al. 1998a, Cooke and Prepas 1998). Comparisons of flow-weighted mean concentrations between agricultural and reference catchments in my study supports data that agricultural land use increases nutrient export to streams.

2.5.2 Form of phosphorus export

Unlike the majority of research that has shown that agricultural catchments export primarily particulate phosphorus (Ng et al. 1993, Tisdale et al. 1993, Hansen et al. 2002), most of the total phosphorus exported in this study was in the dissolved form. This finding is consistent with research done in watersheds throughout Alberta (Anderson et al. 1998a) and in the same region by Cooke and Prepas (1998). High proportions of dissolved phosphorus in streams draining agricultural catchments can be a result of decreased sedimentation from agricultural catchments due to the predominance of non-intensive agriculture, such as in reduced tillage systems (Sharpley et al. 2001). However, data indicate elevated sediment concentrations in the agricultural stream Plamondon (see chapter 3), yet particulate phosphorus did not make up a larger percentage of the total phosphorus load in this stream. FWMCs comparisons also show that dissolved forms dominated phosphorus fractions. There are several reasons why dissolved phosphorus may dominate export, including interactions at the soil-water interface.

Dominance of dissolved phosphorus in export may be because soil properties in this region are not conducive to adsorption of phosphorus. Phosphorus adsorption increases with concentrations of clay, organic matter, Al, Fe, and Ca in the soil (Tisdale et al. 1993, Whitson 2003). Of these, clay is the most important in Gray Luvisols, which dominate in the study region (Whitson et al. 2004). Because clay in Gray Luvisols is found low in the soil horizon, sorbtion capacity of soils in the upper horizons is limited (Whitson 2003). In addition to this, dissolved forms of phosphorus dominate at neutral soil pH levels (Ng et al. 1993) and the pH of the stream water for all catchments were 6.2-7.1 (see Chapter 3). Along with these explanations, agricultural practices in this area may increase soluble and reactive phosphorus concentrations in the uppermost part of the soil (e.g., long-term inputs of manure and fertilizer, (Hansen et al. 2002). The increased concentration of soluble and reactive phosphorus at the soil water interface (1-5 cm), combined with low adsorption potential of the soil at this depth (Whitson 2003), would increase the concentration of dissolved phosphorus in runoff (Pote et al. 1996, Sharpley et al. 1996). In addition to influences of soil properties on the fractionation of phosphorus in streams draining agricultural catchments, high fractions of dissolved phosphorus may be due to a combination of low runoff and longer leaching times because of low sloping topography (Cooke and Prepas 1998). Low discharges and flow velocities can limit re-suspension of particulate matter from the streambed and erosion from stream banks.

It is not clear if dominance of dissolved phosphorus in export from reference catchments is a result of similar mechanisms that occur in agricultural catchments. The proportion of dissolved phosphorus in the four forested streams in my study was higher than that found by two previous studies on the Boreal Plain (Munn and Prepas 1986, Cooke and Prepas 1998). Differences may be due to reduced runoff and discharge during the study period, low leaching potential of the soils, high proportion of organic soils, or differences in vegetation types. It is also not clear if dissolved phosphorus was dominated by reactive phosphorus or organic phosphorus and if this dominance differed between reference and agricultural catchments. In study of nearby catchments, however, Prepas and Cooke (1998) found that dissolved phosphorus consisted mainly of reactive phosphorus in both forested and agricultural streams. Further study would be needed for more definitive answers on why dissolved forms of phosphorus dominate in both agricultural and forested references catchments in this region.

2.5.3 Form of nitrogen export

Although average nitrogen export to streams was not greater in agricultural watersheds, they exported more inorganic nitrogen (NO_3^- and NH_4^+) than reference systems. This suggests that conversion of catchments to agriculture influences nitrogen speciation, increasing the pool of bioavailable nitrogen. Similarly, Kemp and Dobbs (2001) found increasing nitrate concentrations along a gradient from pristine upland prairie streams to downstream, agriculturally-impacted reaches. However, inorganic nitrogen export was dominated by different species among agricultural watersheds in this study (4:1 NH_4^+:NO_3^- for Plamondon and 1:2 NH_4^+: NO_3^- for Mission).

Differences in ratios of ammonium and total nitrate that make up the inorganic nitrogen pool are likely due to catchment differences in agricultural management practices and within-stream processing. Cooke and Prepas (1998) found that nitrate was the dominant inorganic nitrogen species from cropland runoff whereas ammonium was the dominant species from mixed agricultural watersheds. Bacterial nitrification of anhydrous ammonia, a common fertilizer used in this region, has been cited as a reason for high nitrate export from row-crop dominated catchments (Lucey and Goolsby 1993). Speciation may also be attributed to differences in within-stream transformation of nitrogen (Heathwaite and Johnes 1996). Mission supports numerous beaver dams, which increase retention time and likely increase the potential for bacterial oxidization of ammonium to nitrate (see Chapter 3).

Ammonium is likely to dominate annual export when most export occurs during spring runoff from catchments where manure has accumulated over the winter (Trew et al. 1987, Anderson et al. 1998b, Cooke and Prepas 1998). This is because low temperatures in early spring and winter are likely to inhibit nitrification of ammonium to nitrate (Anderson et al. 1998b, Cooke and Prepas 1998). This is consistent with my data as over 95% of the ammonium exported from Plamondon, the stream where 79% of inorganic N was ammonium, was during spring runoff. Additionally, high proportions of ammonium in inorganic nitrogen export can result when runoff events occur soon after application of manure to crops (Kirchmann 1994). Fresh manure has high ammonium content (Cabrera and Gordillo 1995) but increased soil-manure contact results in bacterial conversion of ammonia to nitrate (unless the soil is frozen). Heathwaite and Johnes (1996) found that over 90% of nitrogen export from heavily grazed systems was ammonium. This was largely because low infiltration capacity of the soil led to high surface runoff and transport of recently deposited urine and faeces. Rapid transport of manure into streams from my catchments was unlikely during summer because of low surface runoff but contributions during spring flow were possible and could account for differences in inorganic fractions.

2.5.4 Timing of export

ROLE OF SUMMER STORMS IN NUTRIENT EXPORT FROM CATCHMENTS

Contrary to previous research, nutrient transport from agricultural streams during summer storm events was minimal in this study. This is likely due to decreased overland flow (also accounting for decreased overall export) because of prolonged regional drought conditions in the years before the study. Generally the soil needs to be saturated for overland runoff to occur (Chang 2003). When water tables are low, soil takes longer to saturate and the precipitation rate rarely exceeds the infiltration rate. It is thus not surprising that nutrient transport from reference streams during summer storm events was low. Also, most precipitation events were not overly high, excepting the June 2-3 rain event. Forested systems have litter layers, which maintain an absorbent soil surface and high water infiltration rates, and precipitation is also intercepted by vegetation; consequently, overland flow is minimized in these systems and summer storm events are not expected to result in greatly increased nutrient transport (Chang 2003). Lack of difference between contribution of summer storms to nutrient export among agricultural and reference watersheds was likely because infiltration capacities were not challenged; subterranean water storage capacity for both agricultural and reference catchments was high. In years when summer precipitation and water tables are higher, summer storms are likely to be more important nutrient transport period, perhaps disproportionably so in agricultural systems because of lower infiltration rates.

ROLE OF SPRING RUNOFF IN NUTRIENT EXPORT FROM WATERSHEDS AND THE ROLE OF BEAVER IMPOUNDMENTS

Snowmelt is more rapid in agricultural catchments because there is decreased forest cover and increased light penetration. This results in a large volume of runoff occurring in a shorter period (Whitson et al. 2004) and explains why agricultural catchments may export a higher proportion of nutrients during spring runoff than reference catchments. I found that spring runoff was an important period for delivery of nutrients to agricultural streams, but it was not of greater importance than for some reference catchments. This can be partially attributed to retention of snowmelt water by beaver impoundments in the Mission (A) catchment; such impoundments were not present in Plamondon.

As a general rule, snowmelt is an important contributor to nutrient export from catchments in northern temperate areas (Anderson et al. 1998a, Syversen 2002). In contrast to this, I found spring runoff played a very minimal role in overall nutrient export from two reference catchments; overall spring runoff was an important period of nutrient export from only four catchments. Variability in beaver pond distribution and retention capabilities among catchments can account for this. For example, the Deer catchment, which exported more than 50% of all nutrients during spring runoff, had been subjected to numerous human-induced beaver dam removals throughout the year. This likely led to decreased stability and retention capabilities of the remaining (or newly built) impoundments; high inputs of snowmelt may have lead to burst dams and high spring discharges.

Spring runoff was an important period for export of inorganic nitrogen for both agricultural watersheds and for two of the reference watersheds. During spring snowmelt, eluviation is low and highly soluble inorganic forms of nitrogen are readily transported in surface runoff (Syversen 2002). Ammonium export is likely to be highest in spring runoff because of reduced bacterial processing during lower temperature periods (Trew et al. 1987, Anderson et al. 1998b, Cooke and Prepas 1998).

VARIABILITY IN EXPORT PATTERN IN NUTRIENT SPECIES AMONG CATCHMENTS

In general, there were no clear differences in timing of export of nutrient species between agricultural and reference catchments in this study. Variability of nutrient export over time and space can be partially attributed to variation in precipitation patterns. Investigations of spring snowmelt of central Russian rivers showed that an increase in forest within the catchment results in a reduction in spring runoff (Shiklomanov and Krestovsky 1988). Discharge from field and forest areas during spring runoff were dependent on the region and annual climate patterns (Shiklomanov and Krestovsky 1988). This and my data suggest that the relative importance of spring runoff, storm events, and baseflow depend on summer precipitation patterns, snowmelt patterns, catchment infiltration capacity, vegetation types, and beaver dam dynamics. It also varies among nutrient species. Because these parameters, as well as concentration of nutrient species within the soil, likely vary among catchments, each catchment will be affected differently though not predictably within the scope of this study.

2.6 Conclusions, implications for regional lakes, and recommendations for watershed management

The results of this research indicate that phosphorus export in boreal Alberta is higher from agricultural catchments than reference catchments. Therefore, continuing conversion of forested land to agricultural land within this region will likely increase phosphorus loading to streams. Further, potential for eutrophication of downstream aquatic systems is enhanced because most phosphorus is exported in a dissolved form and is therefore more bioavailable (Cooke and Prepas 1998, Hansen et al. 2002). Cooke and Prepas (1998) found that most (>75%) of the dissolved phosphorus exported was reactive species and also the proportion was highest in stream water draining agricultural catchments. Because the region, topography, and geology were similar to my study, I assumed dissolved phosphorus in this study followed the same speciation pattern.

While phosphorus is the limiting nutrient in most Boreal Plain lakes, nitrogen can also play an important role in lake productivity. Although agricultural catchments exported more inorganic nitrogen than reference catchments, the ratio of dissolved nitrogen to phosphorus (expressed as mass) was close to one. Because at least an order of magnitude more nitrogen is needed than phosphorus to produce a unit of plant mass, there is subsequently more effective phosphorus loading occurring. This is important because increased phosphorus to nitrogen rations can help shift phytoplankton communities toward cyanobacterial domination (Schindler 1977, Trimbee and Prepas 1987), which may lead to poor water quality.

Attempts to reduce phosphorus loads to aquatic systems in this region should focus on improved land management practices, including the reduction of pools of soluble and reactive phosphorus within the soil (Sharpley et al. 2001, Weld et al. 2002). Typical mitigation methods aimed at reducing erosion, such as riparian grass-strips, would not reduce the majority of phosphorus loading. Management of agricultural land within the Lac La Biche watershed should move to a phosphorus-based limit in the soil profile in order to control excess phosphorus and reduce phosphorus losses to surface waters (Sharpley et al. 1994). This can be done by applying phosphorus only to fields that have an agronomic need for phosphorus, applying fertilizers based on the ratio equivalent to soil needs, and reducing the amount of annual runoff from agricultural fields through crop selection and soil conservation practices. Beaver impoundments can also be important in reducing phosphorus transport, at least temporarily, and maintenance of beaver habitat should be encouraged. However, effective land management and maintenance of beaver impoundments can only go so far in reducing nutrient loading. It may be necessary to limit conversion of forested land to agricultural land within watersheds where eutrophication of surface waters is a concern.

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Table 2.1. Watershed characteristics for six streams in the Lac La Biche region. Agricultural streams are represented by (A) and reference streams by (R). Exports are in kg km^{-2} yr^{-1}, catchment area is in km^2, mass loads are kg yr^{-1}, FWMC (flow weighted mean concentrations are in mg L^{-1} and annual discharge is in m^3 x 10^3.

	Plamondon (A)	Mission (A)	Reutov (R)	Deer (R)	Cadieux (R)	Goldie (R)
Catchment Area	111.4	46.1	48.0	109.0	28.4	46.6
% Urban	7.9	10.1	4.9	6.5	5.8	3.6
% Agricultural	42.9	38.9	10.4	4.3	3.4	0.0
% Forest	34.1	42.5	70.5	77.7	79.5	82.0
% Wetland	4.4	2.4	9.0	2.2	2.5	7.5
TP Export	7.9	9.3	2.1	4.1	4.4	3.4
PP Export	2.1	2.3	0.8	0.8	1.0	0.9
TDP Export	5.8	6.9	1.3	3.3	3.4	2.5
TN Export	35.5	54.7	25.9	25.3	41.8	42.9
PN Export	5.1	5.2	2.0	2.3	0.3	1.1
TDN Export	30.4	49.4	23.9	23.1	41.4	41.9
NH_4^+ Export	5.7	2.1	0.5	1.3	0.3	0.5
NO_3^- Export	1.5	4.2	0.2	0.3	0.1	0.2
DON Export	23.2	43.2	23.2	21.4	41.0	41.2
Mass Load TP	797.5	406.2	97.6	403.9	116.2	150.0
Mass Load TDP	585.0	305.1	61.1	327.8	89.3	111.7
Mass Load TN	3576.8	2400.9	1210.3	2519.3	1103.6	1909.5
Annual FWMC TP	0.520	0.402	0.128	0.308	0.173	0.093
Summer FWMC TP	0.360	0.363	0.096	0.268	0.171	0.089
Spring FWMC TP	0.687	0.554	0.326	0.331	0.295	0.235
Annual FWMC TDP	0.382	0.302	0.080	0.250	0.133	0.070
Summer FWMC TDP	0.328	0.295	0.067	0.200	0.132	0.068
Spring FWMC TDP	0.437	0.329	0.160	0.278	0.182	0.119
Annual FWMC TN	2.333	2.376	1.587	1.924	1.643	1.189
Summer FWMC TN	1.878	2.265	1.548	1.829	1.653	1.185
Spring FWMC TN	2.805	2.807	1.828	1.976	1.150	1.311
Annual Discharge	1533	1010	763	1309	671	1606

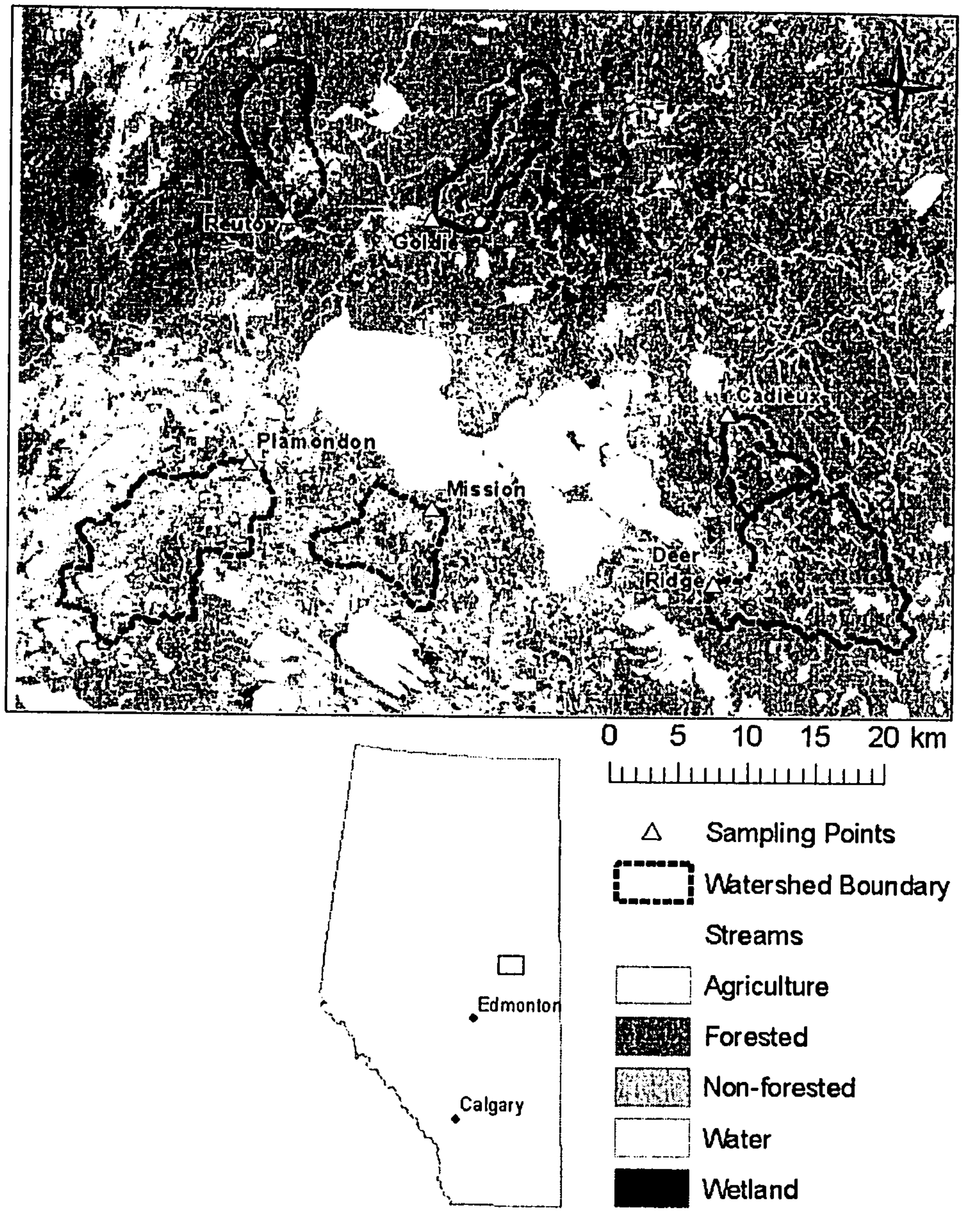

Figure 2.1. Land use in the 6 study catchments. Agriculture sites are Plamondon and Mission (= Blackbird Creek). Reference sites are Reutov, Deer, Cadieux, and Goldie. The large lake in the center of the diagram is Lac La Biche.

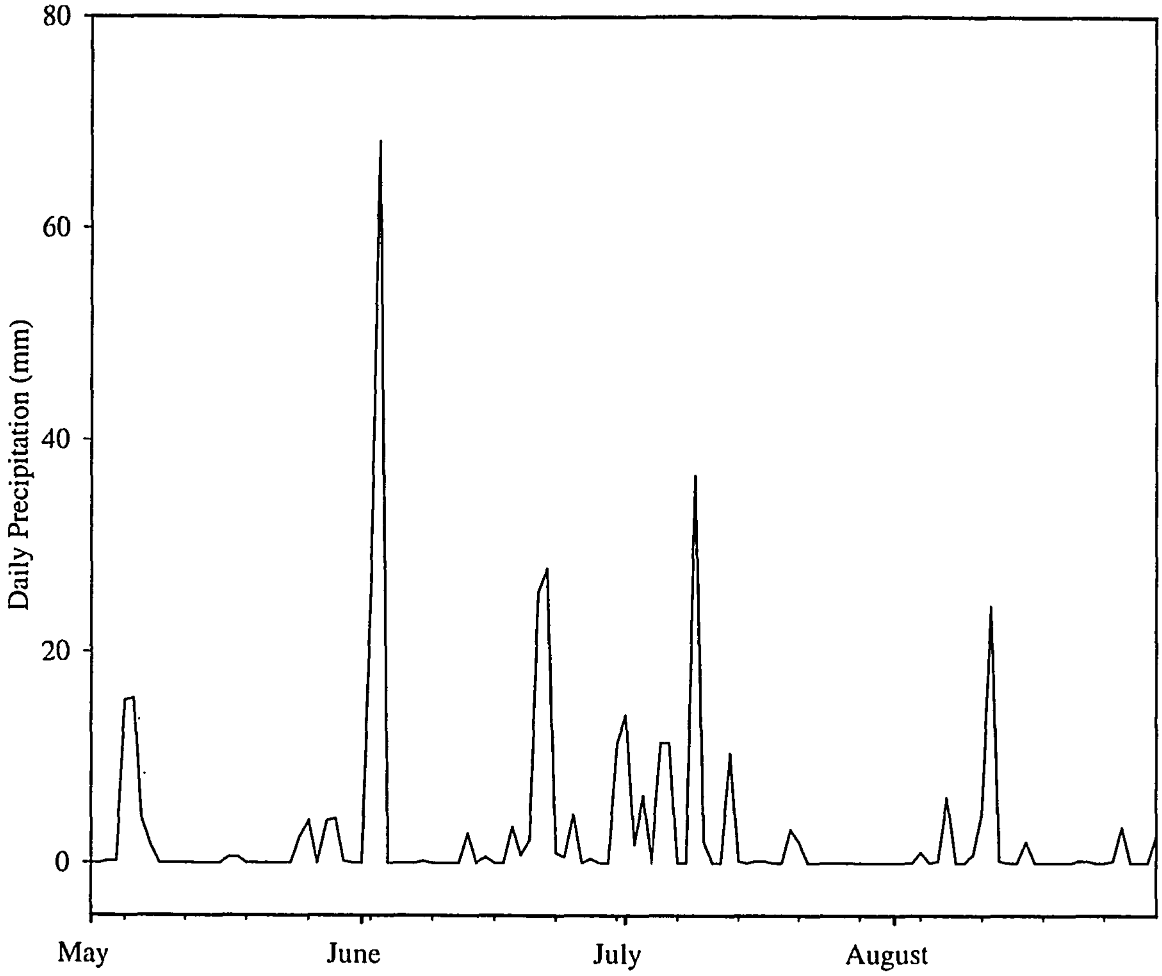

Figure 2.2. Daily precipitation in May to August 2004 for the Lac La Biche region taken from a single Alberta Environment gauging station at Lac La Biche airport.

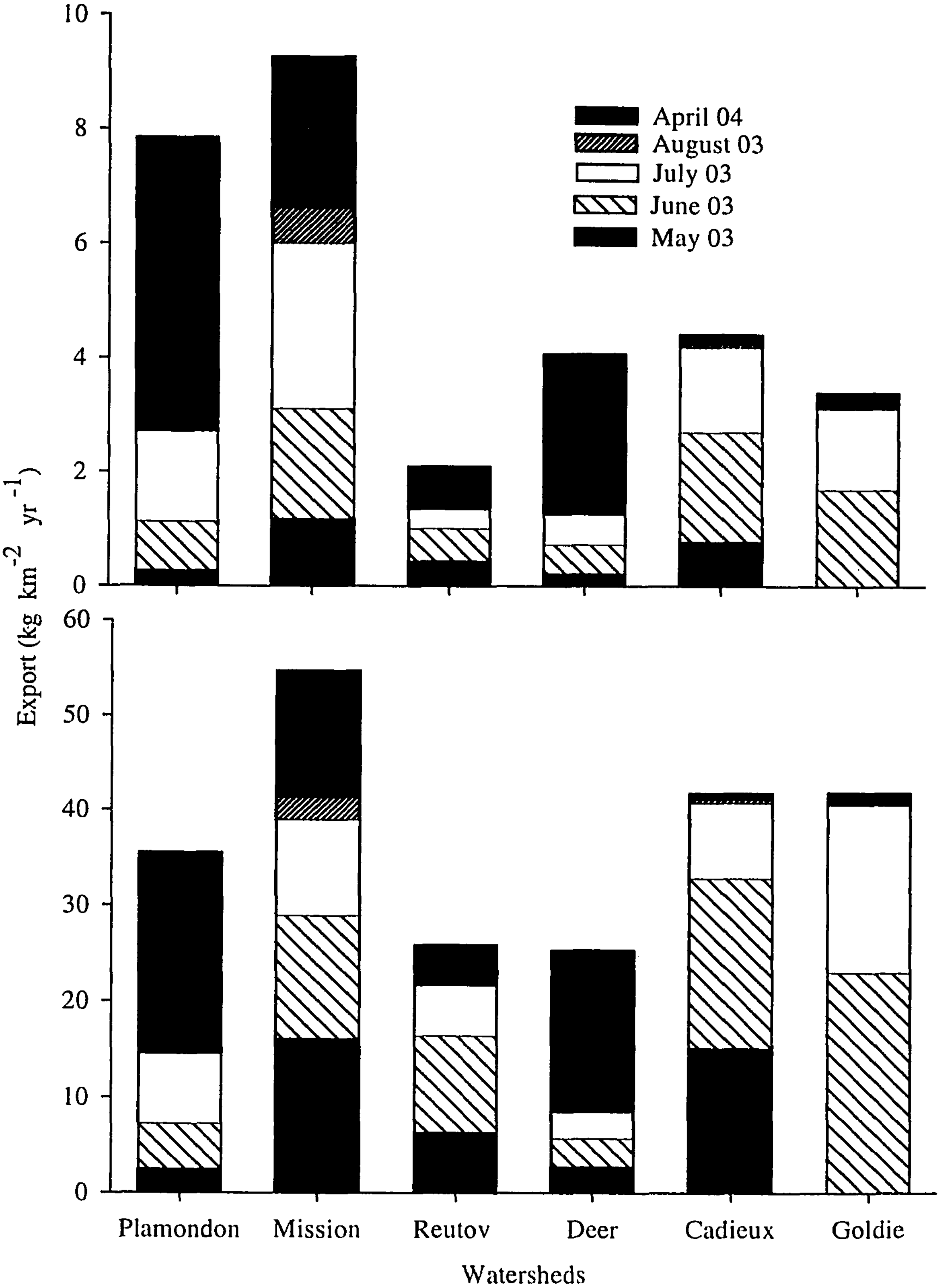

Figure 2.3. Total phosphorus (top) and total nitrogen (bottom) export for May 2003 to April 2004 for six watersheds in the Lac La Biche region. From September 2003 to the end of March 2004 export from all watersheds was negligible.

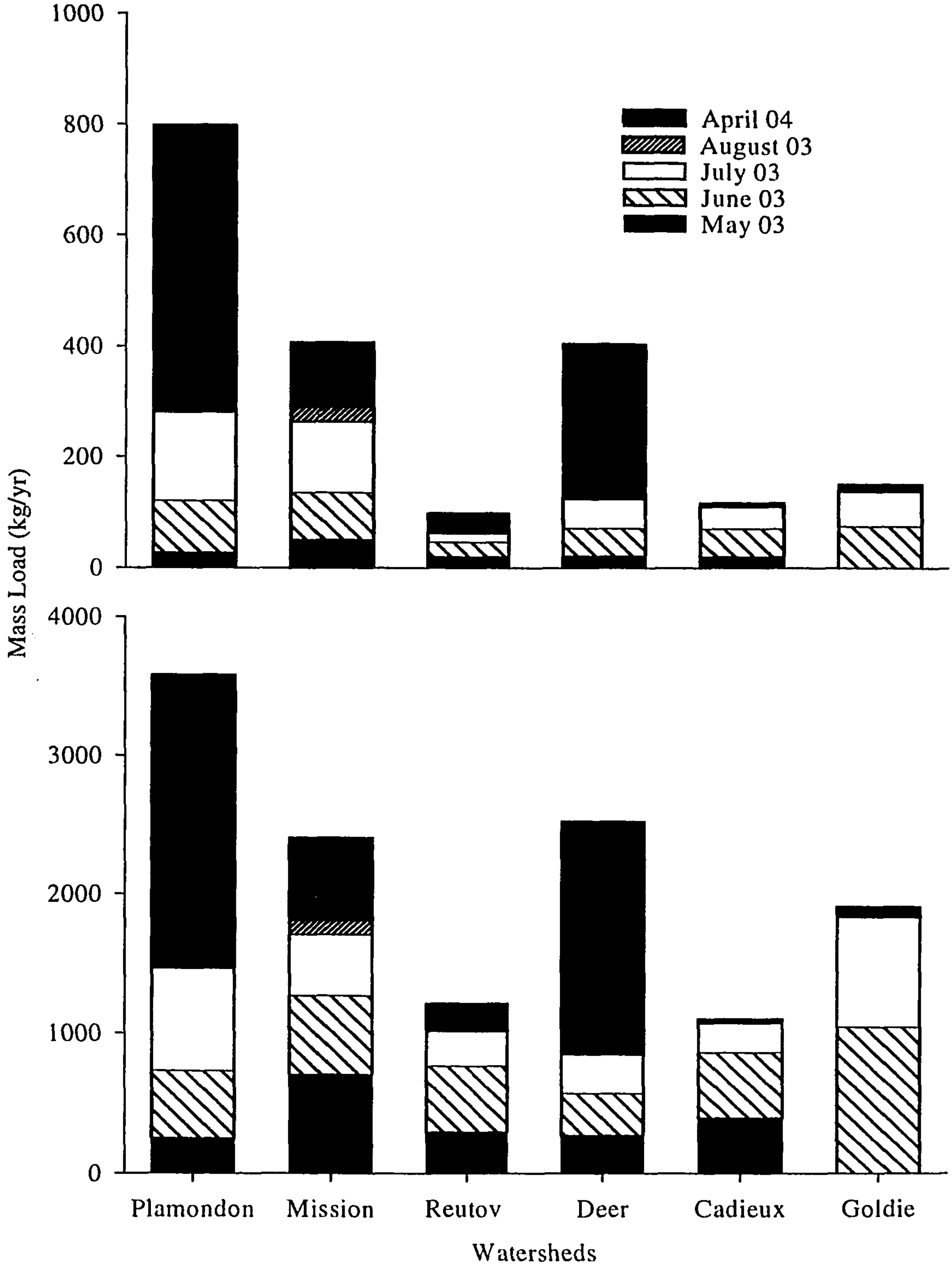

Figure 2.4. Total phosphorus (top) and total nitrogen (bottom) mass load for May-August 2003 and April 2004 for six watersheds in the Lac La Biche region. From September 2003 to the end of March 2004 export from all watersheds was negligible.

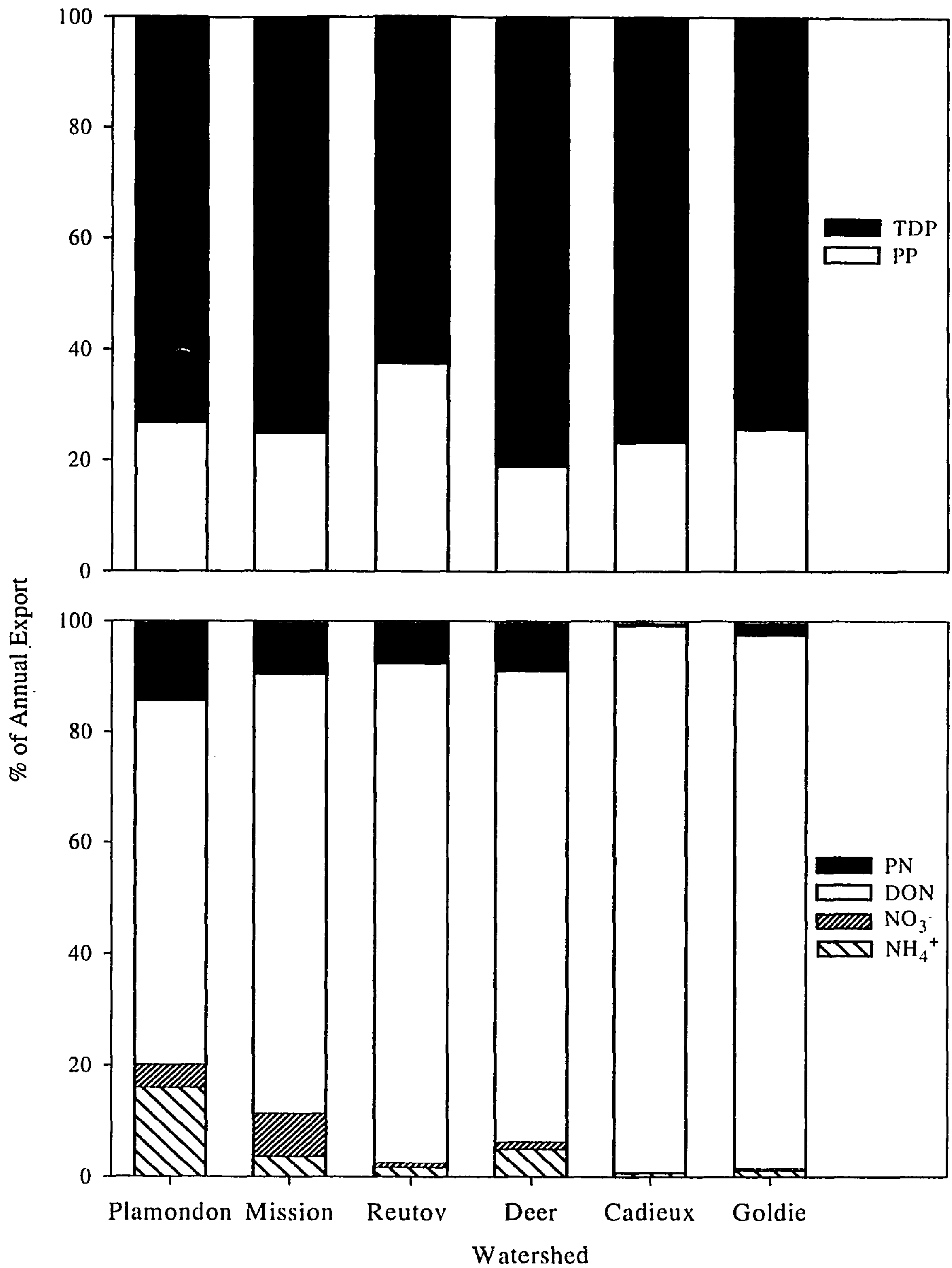

Figure 2.5. Percentage of annual export of phosphorus as dissolved or particulate forms, and nitrogen as particulate nitrogen, dissolved organic nitrogen, nitrate or ammonium for six watersheds in the Lac La Biche region.

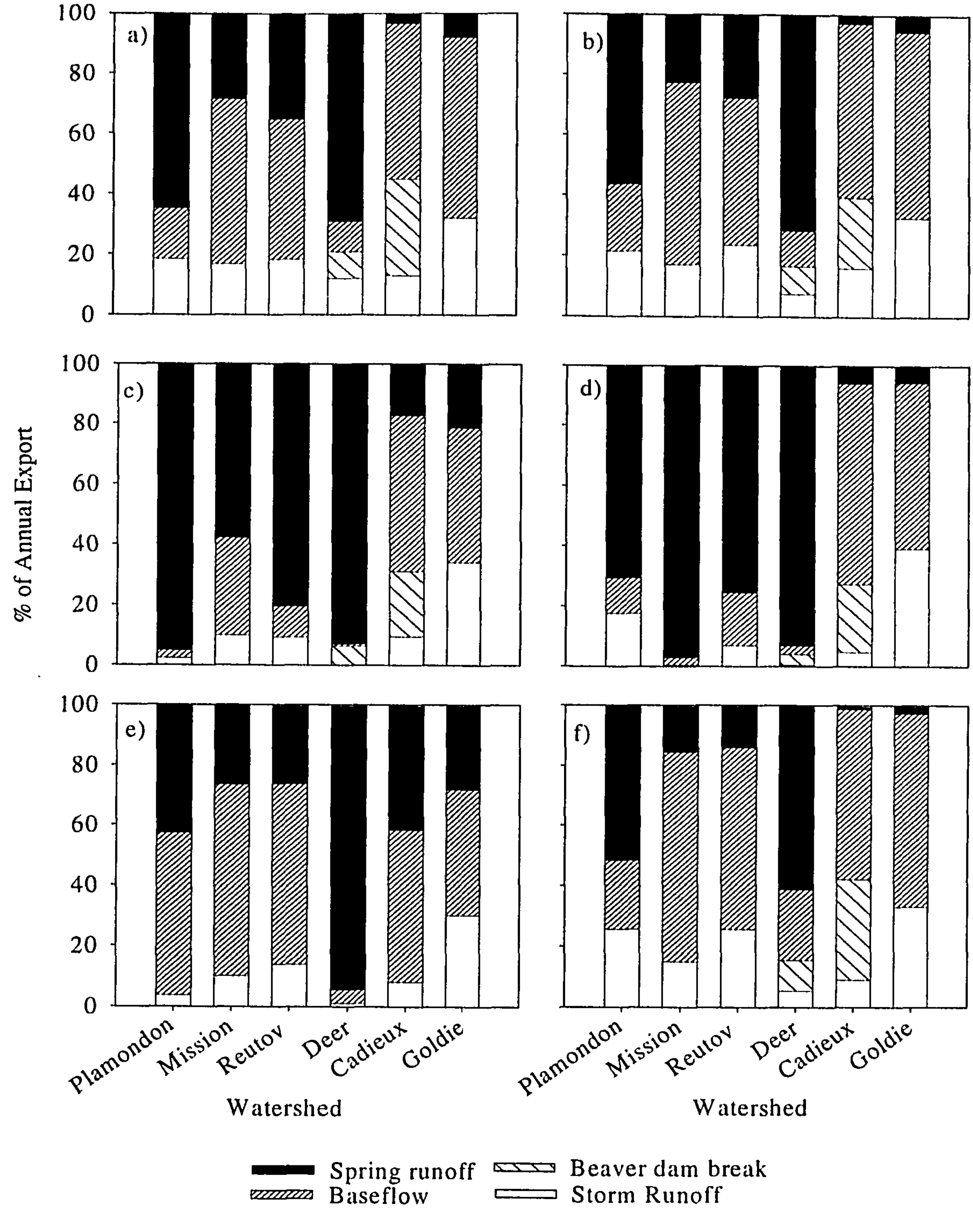

Figure 2.6. Percentage of TP (a), TDP (b), NH_4^+(c), NO_3^- (d), DON (e), and PN (f) exported during baseflow, storm runoff events, beaver dam breaks, and spring runoff for six watersheds in the Lac La Biche region.

Chapter 3: Effects of catchment land use on water chemistry and macroinvertebrate assemblages in beaver-impacted streams in boreal Alberta

3.1 Introduction

Inclusion of a broad landscape perspective is integral to effective watershed management and protection of aquatic systems (Sidle and Sharpley 1991, Johnson and Gage 1997, O'Neill et al. 1997) because processes and disturbances that occur at large spatial scales influence aquatic systems (Richards and Host 1995, Richards et al. 1996, Johnson and Gage 1997, Townsend et al. 1997, Sponseller et al. 2001). For this reason, investigations of stream water quality and biota have shifted focus away from investigating only stream-reach parameters and now include analyses at multiple spatial scales, including land use in the catchment (Allan et al. 1997, Johnson and Gage 1997, Gergle et al. 2002).

The US Environmental Protection Agency regards agriculture as the leading source of impairment of the United States' rivers (USEPA 2000). Increased nutrients and sedimentation are the two most important problems associated with increased agriculture at the catchment scale (Dillon and Kirchner 1974, Johnes et al. 1996, Soranno et al. 1996, Allan et al. 1997, Townsend et al. 1997, Neill et al. 2001, Townsend et al. 2004). Increased nutrient inputs often lead to increases in productivity of algae and macrophytes and overall stream eutrophication (Carpenter et al. 1998). Additional impacts of agricultural land on streams draining these catchments include loss of riparian streamside vegetation (Smith 1992). This can exacerbate increases in primary productivity via enhanced light availability and stream temperatures (Smith 1992, Sweeney 1992, Rutherford et al. 1997). Higher levels of sedimentation in streams draining agricultural catchments can lead to physical modification of the streambed, which affects water channelisation and increases flooding frequencies. Further, a decrease in the permeability of terrestrial substrates associated with conversion of forested land to agricultural land has the potential to alter natural flow regimes, particularly during high precipitation events and snowmelt (Dillon and Kirchner 1974, Soranno et al. 1996, Allan et al. 1997, Jordan et al. 1997, Castillo et al. 2000, Neill et al. 2001). In addition to influences of increased nutrient and sediment inputs, agriculturally facilitated decreases in riparian vegetation also result in lowered allocthonous inputs (e.g. coarse woody debris or leaf litter) to streams (Smith 1992, Richards and Host 1995, Townsend et al. 1997, Griffith et al. 2002).

In-stream physical and chemical changes associated with increased agriculture in the catchment have predictable effects on stream biota. As agricultural land use increases, indices of biological condition decrease across a wide range of taxa (macroinvertebrates, fish, algae) (Cuffney et al. 2000). Macroinvertebrates reflect in-stream changes through shifts in functional feeding groups, overall richness, ratios of intolerant to tolerant taxa, and abundance (Lenat and Crawford 1994, Richards et al. 1997, Black et al. 2004, Townsend et al. 2004). Stream nutrient enrichment and associated increases in primary productivity are typically associated with increased biomass and richness of scrapers and decreases in other taxa (Rosenberg and Resh 1993). This is because such highly productive systems often exhibit extreme fluctuations in dissolved oxygen that eliminate taxa sensitive to anoxic conditions, such as Plecoptera and many Ephemeroptera (Lenat and Crawford 1994). Changes in hydrological regimes influence the distribution of fine and coarse particulate organic matter (FPOM and CPOM), which also affect ratios of functional feeding groups among macroinvertebrates. For example, declines in CPOM due to loss of riparian inputs are correlated with reduced abundance of shredders (Fore et al. 1996). Decreases in coarse woody debris (CWD) inputs will likely result in decreases in both wood-associated taxa and potentially overall taxon richness, because woody debris is an important resource for invertebrates (Braccia and Batzer 2001). Increased sedimentation decreases species richness and density of benthic macroinvertebrates and increases burrowing chironomid larvae and oligochaetes predominance (Hellawell 1986). Overall, agriculturally-influenced streams shift to greater richness and abundance of groups tolerant to organic pollution, low DO, and high sediment, such as Chironomidae and Oligochaeta (Hogg and Norris 1991, Lenat and Crawford 1994).

Because there are predictable species changes in composition, macroinvertebrates are useful in assessment of stream water quality (Rosenberg and Resh 1993, Karr 1999, Cuffney et al. 2000, Karr and Chu 2000). Assessment can be based on subsets of or entire assemblages. Effective members of an "indictor subset" generally have low ranges of tolerance to environmental perturbations, and abundances that reflect adequate physical, chemical and nutritional conditions (Rosenberg and Resh 1993). Taxa that are poor indicators of disturbance are not affected by substantial changes in environmental quality. To determine indicator taxa for a particular perturbation it is important to identify which taxa respond to the perturbation. Likewise, indicators often vary regionally because local characteristics affect abundances and distribution. For this reason, assessment in areas where biomonitoring programs have not been previously established should be informed by examination of assemblages that occur naturally in the region (Resh et al. 1995). This lesson has been demonstrated through many failed attempts to replicate in places other than eastern North America Vannote's (1980) "River Continuum Concept" prediction of a consistent pattern of changes in invertebrate feeding groups from headwaters to high-order streams (Lake 1985, Marchant 1985).

Studies investigating increased agricultural catchment conversion on stream water quality and/or macroinvertebrate assemblages have taken place in Central Canada (Barton 1996), Midwestern (Johnson et al. 1997), Mid-Atlantic (Herlihy et al. 1998) and Central Plains USA (Griffith et al. 2002), Brazil (Neill et al. 2001), New Zealand (Townsend et al. 1997, Riley et al. 2003), and Australia (Smith et al. 1999). Differences in forest composition, climate, geology, and topography among regions mean that results are generally applicable only to each particular study region (Barbour et al. 1999). To establish effective regional biomonitoring programs, an understanding of the natural variability in local stream macroinvertebrate assemblages and physicochemical parameters is needed.

Currently, major landscape-scale changes are occurring at the southern edge of Alberta's boreal zone as forest is rapidly converted to pasture or cropland (Hiley et al. 2001). Although some research has shown that these catchment alterations are affecting stream water quality through nutrient enrichment (Anderson et al. 1998, Cooke and Prepas 1998), there is inadequate data outlining the effects of such land conversion on boreal stream flow patterns, water quality, and macroinvertebrate assemblages to support sound management decisions. Recent concerns with poor water quality in streams and lakes of Alberta (Anderson et al. 1998) further demonstrate the need for more research. I investigated the effect of agricultural land use on small boreal streams in the Lac La

Biche region of northcentral Alberta. My main research objectives were to: 1) investigate the influence of agricultural land use on in-stream physical and chemical features and structure of macroinvertebrate assemblages; and 2) investigate the potential use of macroinvertebrates as bioindicators of boreal stream water quality. Because beavers (*Castor canadensis*) frequently alter the physical structure of streams in boreal Alberta, I also included estimates of beaver activity as part of the study. Beavers have been shown to significantly alter ecosystem processes in streams through damming (Naiman et al. 1986), which results in decreases in stream current velocity, retention of sediment and organic matter, anoxic conditions, build-up of sediment on stream floors, and concurrent reduction of macroinvertebrate fauna associated with faster flowing streams (Naiman et al. 1986, McDowell and Naiman 1989, Collen and Gibson 2001). Additionally, beaver impoundments alter nutrient cycles (Naiman et al. 1991, Yavitt et al. 1992), availability (Johnston and Naiman 1990, Pinay and Naiman 1991), and transport (Naiman et al. 1986, McDowell and Naiman 1989).

3.2 Materials and Methods

3.3.1 Study area

The Lac La Biche watershed is part of the mid-boreal Mixedwood ecoregion (Strong 1992) in north-central Alberta, and drains an area of 4040 km^2 (Mitchell and Prepas 1990). I selected streams where the percentage of agricultural land use was the primary difference between catchments. Of the nineteen streams that drain directly into Lac La Biche, only four flow continuously throughout summer, and of these only three were suitable for study (Red Deer Brook was downstream of Field Lake to which the town of Lac La Biche sewage is discharged). An additional three stream catchments, located just north of the Lac La Biche watershed, were included bringing the total to 6 study catchments (2 agricultural (A) and 4 reference (R); Figure 2.1). Catchments were classified as agricultural if greater than 1/3 of the catchment consisted of agricultural land. To delineate study catchments, I used a combination of a digital elevation model (DEM) and ArcGIS spatial analyst tools (ArcGIS 1999-2004). Land use in the stream catchments was quantified based on 2002 Landsat 7 EMF Imagery (Sanchez-Azofeifa 2002) and spatial analyst tools (ArcGIS 1999-2004) (Figure 2.1, Table 2.1). Land cover classifications were based on the Alberta Ground Cover Characterization (AGCC) land cover classification system (Sánchez-Azofeifa 2005). Accurate classification of land cover is estimated at 90% for the land classes used in this study (Sánchez-Azofeifa 2005). Because of this high accuracy, there was no strong need to validate land cover data were not validated with ground data. Agricultural watersheds consisted of a mixture of nonrow cropland (hay and cereals), pasture and small cow-calf operations. The exact proportion of each type of agriculture was not quantified, however. Compared to other areas of Alberta, intensity of agriculture in these watersheds is moderate based on total chemical expenses (26 to 39th percentile) and manure distribution (40-60th percentile) (Data from Soil Landscape of Canada Census Data, 1991).

The agricultural streams were Mission and Plamondon and the reference streams were Reutov, Deer, Cadieux, and Goldie. Blackbird Creek is the local name for the agricultural stream that I refer to as Mission. Streams were all 2nd and 3rd order depending on the time of the year. The agricultural catchment areas were 46 and 111 km^2 and the reference watershed areas were 28, 47, 48 and 109 km^2. All streams, with the exception of one of the agricultural streams (Plamondon), supported numerous beaver dams. Water sampling locations were selected based on lack of immediate cattle access and influence of beaver dams and location of adjacent roads or culverts. Precipitation data were obtained from an Environment Canada (2005) meteorological station within the town of Lac La Biche (54° 46' N, 112° 1' W). Regional long-term averages were obtained from an Athabasca meteorological station (54° 49' N, 113° 31' W) (Environment Canada, 2005).

3.3.2 Macroinvertebrates

Macroinvertebrates were sampled at three equidistant points along a 50 m stream reach in May, June and July 2003 (N = 9 samples/stream). Samples were obtained by sweeping the entire width of the stream for 45 seconds with a trapezoidal-framed dipnet (250 μm mesh, 424 cm^2 opening). Pre-sampling of streams indicated 45 seconds would provide and appropriate estimate of density and diversity. Macroinvertebrates were sorted live in the field until the majority of organisms were removed. The residual sample was preserved and sorted with the use of a dissecting microscope in the laboratory. All organisms were preserved in alcohol/formaldehyde with the exception of water mites (Acari: Hydracarina), which were preserved in Koenike's solution (45% H_2O, 45%

glycerol, 10% glacial acetic acid). Macroinvertebrates were enumerated and identified to genera in most cases. Chironomidae, Oligochaeta, Cladocera, and Copepoda were not identified beyond the taxonomic levels indicated and only presence or absence of Cladocera and Copepoda in the sample was recorded. All samples were kept separate.

3.3.3 Water chemistry and physical characteristics

Water samples were collected in the middle of each stream at mid depth with acid-washed polystyrene and Nalgene polyethylene containers, twice monthly from May to the end of August 2003 (n= 8 samples/stream). Samples were taken within a 3 hour period in the morning. This was supplemented by sampling during storm events (n = 3). Samples were analyzed for total nitrogen (TN), total dissolved nitrogen (TDN), nitrate (NO_3^-) plus nitrite (NO_2^-), ammonium (NH_4^+), total phosphorus (TP), and total dissolved phosphorus (TDP) within 48 hours of collection. TP and NH_4^+ concentrations were determined from unfiltered stream water, whereas TDP, $NO_3^-$and NO_2^-, were determined on water samples filtered through a 0.45-µm Millipore GF/F filter (University of Alberta Limnology Lab). Nitrite is usually negligible in surface water so it was assumed that all oxidized nitrogen was nitrate (Cohn et al. 1999). Stream discharge was quantified weekly by measuring velocity (Swoffer 2100 current meter) and water depths at set intervals across the stream channel. In streams where velocity was too low to obtain a reading with the Swoffer 2100, surface velocity was estimated by observing the time taken for a submersed orange to travel a known distance downstream (Hauer and Lamberti 1996, USEPA 2004). Level sensors (Stevenson recorders) were set up on one of the agricultural streams (Plamondon) and all of the reference streams (Deer, Reutov, Goldie, and Cadieux) at the end of May 2003. Dissolved oxygen (DO), temperature, and conductivity were recorded weekly using aYSI meter. DO readings from the YSI meter were verified with Winkler's analyses monthly. pH was determined weekly using a Beckman 32 pH meter in the field as well as a Fisher Accumet model 925 in the lab twice per month. Inorganic substrate concentrations were obtained by filtering 400-500 ml of water (collected during chemistry sampling) through a pre-weighed Whatman GF/F glass fibre filter, oven drying for 24 hours at 550° C, and weighing.

3.3.4 Statistical methodology

Stream catchments were divided into two categories based on dominant land use (Figure 3. 1,Table 3. 1): 38-43% agricultural land use (Mission and Plamondon) and less than 10% agricultural land use (Goldie, Reutov, Cadieux, and Deer). Because the data were not normally distributed and sample sizes were small, I used non-parametric statistics. Mann-Whitney U tests were used to compare in-stream physical and chemical characteristics for agricultural (n=2) and non-agricultural reference streams (n=4) using Systat 10.2 software. I chose a significance level of 0.10 to avoid type II error due to low replication. Correspondence analysis (CA), an unconstrained ordination technique, in Version 4.5 of CANOCO (ter Braak and Smilauer 1997-2002) was used to determine the variability of macroinvertebrate taxa in multi-dimensional space. Canonical correspondence analysis (CCA) in Version 4.5 of CANOCO (ter Braak and Smilauer 1997-2002) was then used to determine which environmental variables best explained the abundance of macroinvertebrate taxa in multidimensional space. Raw untransformed data (counts per taxonomic unit) were used in both ordinations because ordination with log transformed data did not lead to altered interpretation. Rare taxa (found at a single sampling location) were not included in the analyses. CCA is a direct gradient method that derives a linear combination of the environmental variables. I assumed that macroinvertebrate species have a limited range of environmental conditions and that there is an optimum where they demonstrate maximum abundance. Canonical correspondence analysis determined which environmental variables were the most important in shaping macroinvertebrate community composition. Variables were selected based on their effectiveness in explaining the variation in the macroinvertebrate data as well as their interpretable ecological significance, through forward stepwise selection. The significance of the CCA axes was tested using an unrestricted Monte Carlo permutation test (999 permutations, $p = 0.10$; (ter Braak and Verdonschot 1995). I employed multivariate community analyses because the regional biology of macroinvertebrates was not known with certainty and therefore taxa could not be delineated into functional feeding groups. Further, taxa tolerant to environmental perturbations have yet to be established in this region.

3.3 Results

3.3.1 Physical and chemical responses

Mean TP, TDP, TN, and TDN were significantly higher in agricultural streams than reference streams (p = 0.064; Figure 3. 2, Table 3. 2). Mean TDP, NO_3^-, sediment and DO concentrations, as well as conductivity and velocity, were noticeably higher in agricultural streams, though none of these differences were statistically significant (Figures 3. 2-3. 4, Table 3. 2). However, the appearance of higher mean sediment concentrations, NO_3^- concentrations, and velocity in the agricultural streams was the result of data from a single agricultural stream (Plamondon). Mean sediment concentration was 4-6 times higher in Plamondon than other streams (Figure 3.4), and mean velocity was 2-13 times higher than in the four reference streams and 66 times higher than the other agricultural stream (Mission). Further, Plamondon had 62% of dissolved inorganic nitrogen (DIN) in the form $NO_3^- + NO_2^-$ compared to 8-20% in all other streams. Ammonia made up 38% of DIN in Plamondon compared to 80-91% of DIN in other streams. Mean temperature, NH_4^+, and pH did not differ between agricultural and reference streams.

3.3.2 Macroinvertebrate assemblages

Canonical correspondence analysis, with forward stepwise selection, indicated that total phosphorus, percentage of native grassland, percentage of agriculture, total dissolved phosphorus, and velocity were significant descriptors of the variation in macroinvertebrate composition (Table 3. 3). These environmental variables explained 65.5% of the variation in macroinvertebrate assemblages. CCA produced an overall significant ordination (p= 0.016) in addition to a highly significant first axis (p= 0.004). The first two axes collectively explained 48.7% of the variation within the macroinvertebrate-environment relationship (λ_1 = 0.525, 27.7%; λ_2 = 0.400, 21.0%). Similarly, the first two CA axes collectively explained 51.3% of the variation within the macroinvertebrate assemblage data, (λ_1 = 0.554, 29.1%; λ_2 = 0.420, 22.2%, Figure 3.5). In the biplot diagrams of CA and CCA both axes are similar for species and stream distributions (Figure 3.5-3.8). This suggests that the measured environmental variables are those responsible for variation in taxon assemblage variation and that CCA is an appropriate analyses for the data. The species-environment correlation for CCA axis 1

(0.979), confirms a strong relationship between the macroinvertebrate assemblages and the selected environmental variables. TP, TDP, percentage of agricultural land, and velocity were associated with the CCA axis 1 and also positively correlated with each other. I found that *Valvata* (Gastropoda), Oligochaeta, and *Simulium* (Diptera) were positively associated with TP, TDP, agricultural land use, and velocity (Figure 3. 7, Table 3.4, Appendix A). Additionally I found three groupings of taxa negatively associated with the above parameters but also separated from each other along CCA axis II. The leeches *Nephelopsis* and *Glossiphonia*, the mayfly *Siphlonurus*, the damselflies *Enallagma* and *Lestes* and the beetle *Laccophilus* were opposite to the taxa associated with high agricultural land use but were not distributed on CCA axis II. The two additional groupings of taxa were opposite to each other on CCA axis II. Associated with native grassland were the amphipod *Gammarus lacustris*, the beetle *Graphoderus*, the gastropods *Helisoma* and *Lymnaea stagnalis*, and the dragonfly *Aeshna*; opposite to this group were the water mite *Piona*, phantom midge *Chaoborus*, mayfly *Caenis*, amphipod *Hyallela azteca*, tricophteran *Banksiola* and whirligig beetle *Gyrinus*.

In terms of stream distribution on the ordination biplot, reference streams Goldie, Deer and Reutov were opposite agricultural streams Plamondon and Mission along axis I (Figure 3.8). However, macroinvertebrate assemblages in these three streams were very different and there was separation along CCA axis II. As expected Plamondon and Mission were related to agricultural land use and correlated parameters, but reference stream Cadieux was also grouped with these agricultural streams (Figure 3. 8). After removing the variability explained by velocity (31.5%), I found that together total phosphorus, native grassland, agriculture, total dissolved phosphorus significantly explain the remaining variation in macroinvertebrate composition (p= 0.012 with 499 permutations).

To reduce the influence of velocity and other characteristics associated with beaver impoundments, I preformed a CCA excluding Plamondon (the only stream that lacked beavers). Canonical correspondence analysis with forward stepwise selection indicated that total dissolved phosphorus, percentage of native grassland, percentage of agricultural land, and deciduous forest were significant descriptors of the variation in macroinvertebrate composition (Table 3.5). These environmental variables could explain

68.5% of the variation in macroinvertebrate assemblages. CCA produced an overall significant ordination ($p = 0.012$) in addition to a significant first axis ($p = 0.01$). The first two axes collectively explained 56.5% of the variation within the macroinvertebrate assemblage data, ($\lambda_1 = 0.456$, 29.1%; $\lambda_2 = 0.430$, 27.4%). The species-environment correlations for CCA axis 1 (0.988) and CCA axis 2 (0.980), confirmed a strong relationship between the macroinvertebrate assemblages and the selected environmental variables along each axis. I found that Oligochaeta was still positively associated with TDP and agricultural land use (Figure 3. 9, Table 3.4), but *Simulium* (Diptera) and *Valvata* (Gastropoda) were not. Additionally I found that the true flies *Aedes*, *Anopheles*, and Chironomidae, the beetles *Hydrobius* and *Laccobius*, the snail *Gyraulus*, and the water mite *Eylias* were associated with TDP and agricultural land use. Among stream and stream-taxa relations remained similar to the first ordination biplot (Figure 3.10).

3.4 Discussion

3.4.1 Effects of agricultural land use on stream water chemistry

Nutrient concentrations (mean TP, TN and TDN) were significantly higher in agricultural streams than in reference streams. Studies in other regions and in boreal Alberta have shown elevated nutrient concentrations in streams draining agricultural land (Dillon and Kirchner 1974, Johnes et al. 1996, Soranno et al. 1996, Allan et al. 1997, Townsend et al. 1997, Anderson et al. 1998, Cooke and Prepas 1998, Neill et al. 2001, Townsend et al. 2004). Elevated nutrient concentrations in agricultural catchments were likely due to increased inputs of nutrients either through addition of synthetic fertilizer and animal wastes, and cultivation and other physical disturbance of the top soil leading to increased nutrient mobility, or a combination thereof (Anderson et al. 1998, Carpenter et al. 1998, Cooke and Prepas 1998, Haygarth and Jarvis 1999).

Consistent with research done in watersheds throughout Alberta (Anderson et al. 1998), dissolved phosphorus was the primary species of phosphorus in all streams. Reasons for this include: decreased sedimentation from agricultural catchments due to the predominance of non-intensive agriculture (Sharpley et al. 2001); soil properties that encourage dissolved forms to dominate; low runoff during the study period; and longer leaching times because of low sloping topography (Cooke and Prepas 1998). Because most phosphorus is exported in a dissolved form and is therefore more bioavailable, potential for increased primary production in streams is enhanced (Cooke and Prepas 1998, Hansen et al. 2002).

Inconsistent with other research that showed higher inorganic nitrogen in streams draining agricultural land (Lenat and Crawford 1994, Gburek and Folmar 1999, Randall and Mulla 2001, Follett and Delgado 2002), mean concentrations of inorganic nitrogen were not higher in agricultural streams compared to reference streams. This can be partially attributed to low runoff and precipitation during the study period. Runoff and leaching are the most important pathways for transport of inorganic nitrogen to streams (Jackson et al. 1973, Randall and Mulla 2001). The year prior to our study was one of extreme drought, resulting low water tables and decreased runoff during storm events (see Chapter 2). In addition a large portion of inorganic nitrogen is often transported during spring runoff, particularly in agricultural catchments (Cooke and Prepas (1998), Anderson et al. 1998). Sampling in my study occurred in May to August, and most inorganic nitrate may have already been assimilated by periphyton in agricultural streams (Duff et al. 1984).

Interestingly, Plamondon (A) exhibited nitrate concentrations 10 times higher than the other agricultural stream Mission and 5-25 higher than all reference streams. Differences in inorganic nitrogen speciation among agricultural streams could be a result of differences in land use practices in the catchments (see Chapter 2). For example, Cooke and Prepas (1998) found that nitrate was the dominant inorganic nitrogen species from cropland runoff whereas ammonium was the dominant species from mixed agricultural watersheds. Inorganic nitrogen speciation is often dependent on its source. Runoff from land where manure was applied will tend to have inorganic forms that are dominated by ammonia (Kirchmann 1994). In contrast runoff from land where synthetic fertilizers were applied often results in nitrate domination (Lucey and Goolsby 1993). Nitrogen speciation may also be affected by within-stream transformation of nitrogen, for example through alterations of nitrogen cycling and processing due to beaver impoundments (see below).

3.4.2 Influence of beaver impoundments on stream physical and chemical characteristics

Stream water temperatures were not different in streams draining agricultural catchments compared to those draining reference catchments. In previous studies, higher in-stream temperatures in agricultural catchments were attributed to lack of shading due to reduction in riparian vegetation and reduced current speed (Sweeney 1992, Rutherford et al. 1997, Storey and Cowley 1997, Sponseller et al. 2001). In my study region, however, beaver activity has effectively removed much of the riparian vegetation in non-agricultural catchments and stream temperatures are often higher in water draining from beaver impoundments than in unimpounded streams (Margolis et al. 2001b).

Increased agriculture in the catchment also did not lead to decreases in dissolved oxygen in streams. Depletion of oxygen in streams draining agricultural land was expected, and usually results from bacterial breakdown of accumulated organic matter. There are often increased amounts of organic matter in agricultural streams because of increased direct inputs through runoff or indirect inputs through the breakdown of decomposing algae (whose growth is enhanced by nutrient enrichment (Munn et al. 2002)) (Foy and Withers 1995). In my study, however, dissolved oxygen appeared to be primarily influenced by beaver-induced reductions in current velocity. I suggest that increases in stream temperature and decreases in stream velocity and DO concentrations that are often associated with agricultural catchment land use, are not apparent in this region because 'natural' processes (i.e. beaver impoundments) in reference streams also lead to these conditions.

Increased sedimentation has been described as one of the most important problems associated with increased agriculture at the catchment scale (Dillon and Kirchner 1974, Johnes et al. 1996, Soranno et al. 1996, Allan et al. 1997, Townsend et al. 1997, Neill et al. 2001, Townsend et al. 2004). Consistent with this, I found that agricultural land use in the catchment increased sediment concentrations in some streams. However, I also found that beaver impoundments were effective at reducing downstream sediment concentrations. Sediment concentrations were over four times greater in the agricultural stream without beaver impoundments than all other streams. I suggest that maintenance of beaver impoundments in agricultural catchments could potentially mitigate sediment loading to downstream water bodies. This is consistent with other research in the boreal region that showed beaver impoundments are effective at filtering sediment from water (Naiman et al. 1988).

Although impoundments may be effective at reducing downstream sediment export, they could have a negligible role in reducing downstream phosphorus export in the Lac La Biche region. Impoundments have been shown to act as sinks of phosphorus mainly through sedimentation processes (Maret et al. 1987, Naiman et al. 1988, Correll et al. 2000). This is because phosphorus is typically transported as particulate phosphorus, particularly from agricultural catchments (Omernik 1977, Sharpley and Menzel 1987, Vaithiyanathan and Correll 1992). Because most phosphorus in my streams was in a dissolved form, however, impoundments may play a minimal role in preventing downstream export of nutrients from agricultural land. This may also be true for other areas of Alberta where most phosphorus is exported in dissolved forms (Anderson et al. 1998, Cooke and Prepas 1998). The role of beaver impoundments in reducing nutrient loading will depend on runoff, however. Runoff (a main mechanism of sediment transport) was likely very low during my study period due to drought conditions in the previous year (see Chapter 2). In years when runoff from summer storms is higher, beaver impoundments may be more important in mitigating nutrient export. Impoundments could reduce phosphorus transport by increasing residence time and thereby potential for biological uptake. Higher algae and macrophyte populations in beaver impoundments would further enhance uptake of dissolved phosphorus (Naiman, 1988).

Inorganic forms of nitrogen were dominated by ammonium in beaver impounded streams and nitrate in streams without impoundments (Plamondon). Potential for beaver impoundments to alter nitrogen speciation in streams is high, because beaver impoundments create sites for denitrification of nitrate to nitrogen gas and ammonification (Maret et al. 1987, Margolis et al. 2001a). Plamondon exhibited nitrate concentrations 10 times higher than the other agricultural stream Mission, which had similar nitrate concentration as reference streams. Because nitrate concentrations are typically elevated in streams draining agricultural watersheds (Randall and Mulla 2001) high concentration were also expected in Mission. Nitrate may have been denitrified to nitrogen gas within the impoundment and lost to the atmosphere. Impoundments can also result in increased habitat for aquatic macrophytes and periphyton that assimilate bioavailable forms of nitrogen (Duff et al. 1984, Naiman, 1988, Cushing and Allan 2001). Increases in water residence time could also lead to greater potential for biological uptake. Depending on the size of the pond the maintenance of beaver impoundments could reduce export of bioavailable nitrogen in this region, but this is speculative.

3.4.3 Effects of catchment land use and stream physical and chemical characteristics on macroinvertebrate assemblages

My data support previous research that macroinvertebrate assemblages are influenced by catchment land use. In particular, I found that percent agricultural land use in the catchment, and associated increases in nutrients (TP and TDP), were significant descriptors of macroinvertebrate assemblages. It is not surprising that nutrients and agricultural land use are significant descriptors of macroinvertebrate assemblages; others have found similar relationships (Richards et al. 1996, Allan et al. 1997, Sponseller et al. 2001, Black et al. 2004). Effects of land use and subsequent nutrient enrichment on macroinvertebrate assemblages were largely masked in my study, however, by the influences of beaver impoundments. This occurred in part because one agricultural stream lacked beaver impoundments and as such had higher current speed; current speed is a very important determinate of macroinvertebrate assemblages in streams (Cushing and Allan 2001, Sandin 2003). Simuliidae, for example, was not associated with agricultural landuse after removal of Plamondon (thereby the influence of velocity) from the ordination likely because velocity was a limiting factor for Simuliidae persistence. This is further supported by the fact that there were no Simullidae found in the agricultural stream Mission, which had the lowest velocity of all streams (Appendix A and B). Velocity could not account for all differences between macroinvertebrate assemblages in agricultural streams. *Valvata,* for example, was found only in Plamondon but the biological reasoning for this is uncertain.

Despite this, agricultural streams had macroinvertebrate assemblages that were similarly influenced by increased nutrients. Ordination with the removal of Plamondon resulted in similar taxa being associated with agricultural land use as when Plamondon was included. Specifically, CCA identified Oligochaeta and *Anopheles* as being associated with agricultural land use and increased total dissolved phosphorus. This suggests that these taxa are potential indicators of increased agriculture in the catchment and related nutrient enrichment. Consistent with this, research has shown that at low to medium levels of nutrient enrichment Oligochaeta increase in abundance (Hellawell 1986). Though taxa richness between streams was variable, higher numbers of unique taxa in agricultural streams support suggestions that agricultural land use influences macroinvertebrate communities (Appendix A).

In other regions chironomids are typically indicators of low dissolved oxygen, organic pollution, high sediment (Rosenberg and Resh 1993) and, indirectly, agriculture. They were poor indicators in this region, however. Their strength as indicators of high nutrients and agricultural land use increased substantially when only impounded streams were included in the ordination. This was also true for *Hydrobius*, *Laccobius* and *Gyraulus* and implies that chironomids and these other taxa may be suitable indicators for nutrient enrichment and agricultural land use when all beaver impoundments remain intact. Low taxonomic resolution of chironomids may have also decreased their indicative power. Chironomidae is a large family with species varying in requirements with some species intolerant of low DO and sediment (Rosenberg and Resh 1993).

One of the reference streams (Cadieux) was associated with the same taxa as the impounded agricultural stream Mission. Although current land use classifications deemed this site non-agricultural and therefore unimpacted, the stream was dredged by the Canadian Pacific Railway in the early 1960's, possibly leading to atypical in-stream conditions. Although most physical and chemical parameters were not different from other reference streams, Cadieux did have higher concentrations of total dissolved phosphorus than other reference streams. This, combined with compromised habitat, could have led to similar macroinvertebrate species as an agriculturally impacted stream. It is also possible that Cadieux is simply exhibiting variability in reference streams conditions. Further study would be required to resolve this.

3.4.4 Variability in the macroinvertebrate assemblages of reference streams

Although there was low variability in macroinvertebrate assemblages within reference streams, there was high variability among reference streams. Percentage of the catchment as native grassland was one important determinant of macroinvertebrate assemblages in these streams. In this region, natural grassland areas occurred when beaver ponds were abandoned and the nutrient-rich flood plain was re-colonized by vegetation (Naiman et al. 1986, Hammerson 1994). Since the cycle of beaver colonisation, persistence, abandonment, and vegetation establishment is a lengthy process, increased grassland cover in the catchment may indicate a longer period of beaver influence on the stream. This may cause in-stream characteristics to differ from more recently colonized streams. Beaver induced alterations include increased submerged macrophyte abundance, greater sediment accumulation, and higher amounts of woody debris, all which could affect macroinvertebrate composition. Others have found that beaver ponds in this region are very variable in terms of stability (frequency of blowouts), age, and pond size (Martell 2004) and therefore likely show high variability in in-stream channel characteristics (Naiman et al. 1986, McDowell and Naiman 1989). This is consistent with my ordination analysis results, which show high variation among reference streams in terms of macroinvertebrate assemblages. Sampling location in regards to the impoundment may also account for some variation among reference sites as others have found different macroinvertebrate assemblages upstream of, within, and downstream of impoundments (Clifford et al. 1993, Rolauffs et al. 2001).

3.4.5 Importance of regional reference streams in the assessment of impacts of agricultural landuse on stream water quality and macroinvertebrate assemblages

Reference streams in published studies show regional variability in structure and function. Most studies investigating relationships between land use and streams, however, have target streams that are characterized by narrow and heavily shaded channels, rocky or coarse substrata, allochthonous energy inputs mainly from the surrounding forests, and minimal beaver impoundment (Lenat and Crawford 1994, Johnson et al. 1997, Townsend et al. 1997, Ometo et al. 2000, Huryn et al. 2002). These similarities among streams have allowed generalities to be made, such that deviation from 'natural' states result in poor water quality evidenced by low dissolved oxygen, high nutrient concentrations, increased water temperature, increased algal production, and high conductivity. In contrast, our reference sites did not adhere to standards of high stream water quality set for other areas despite being relatively unimpacted by human activity. For example, mean dissolved oxygen levels were below the impairment level of 6 mg/L suggested by Lenat and Crawford (1994) in all streams. In addition, my study streams were naturally higher in nutrients than those in other regions (Lenat and Crawford 1994), and therefore comparisons between absolute values were not appropriate.

Current Alberta Provincial Guidelines for the Protection of Freshwater Aquatic Life (Alberta Environment 1999) guidelines suggest that total nitrogen should not consistently exceed 1.0 mg/L. All my reference and agricultural streams were higher than this value. These guidelines are more meaningful when standards are based on deviation from natural conditions in the study region. For example, to protect aquatic life, long-term it is suggested that phosphorus concentrations be within 0.05 mg/L of background levels. According to this, agricultural streams were on average five times higher than reference streams, suggesting nutrient enrichment. Similarly, the minimum acceptable level for dissolved oxygen is within 90% of the natural concentrations. Based on this, all the agricultural streams were within 90% of the mean dissolved oxygen of the reference streams; moreover the lowest mean DO was found in one of the reference streams. Increased DO in one of the agricultural streams, however, was likely the result of increased velocity due to loss of beaver impoundments. This suggests that general water quality guidelines should be supplemented with other assessment criteria because impacts can lead to compliance despite stream conditions being vastly altered from reference conditions. Ideally, in areas where there are specific water quality concerns, regional or site-specific objectives should be established based on natural conditions.

My data support suggestions that effective bioassessment of land use conversion on stream using macroinvertebrates requires comparison to assemblages that occur naturally in the region (Resh et al. 1995). Natural streams in this region are beaver-impounded and as such supported lentic taxa such as Odonata, Gastropoda, Amphipoda, and Hirudinea (Appendix A). Others have found that these taxa as well as chironomids and oligochaetes increase in abundance after formation of beaver impoundments (Naiman et al. 1986, Clifford et al. 1993, Collen and Gibson 2001, Margolis et al. 2001b) because slow-flows effectively change invertebrate dominance from lotic organisms to lentic organisms (Naiman et al. 1986, Naiman et al. 1988, Clifford et al. 1993, Collen and Gibson 2001, Cushing and Allan 2001, Margolis et al. 2001b). Because many of these taxa are commonly associated with disturbed or impacted streams in other regions

(Barbour et al. 1999, Smith et al. 1999), it is of particular importance that assessment in this area be based on regional assemblages.

3.5 Conclusions and potential for stream biomonitoring

This study indicated that beaver impoundments are very common in natural streams in the Lac La Biche region and as such, most natural streams are characterized by large open canopy areas, increased wetland area, large accumulations of silt, low dissolved oxygen concentrations, low discharges, and variable macroinvertebrate assemblages. Because of this, effects of agricultural land on other in-stream parameters such as velocity, temperature, and hydrology were not consistent with the majority of findings in other regions (Sweeney 1992, Rutherford et al. 1997, Storey and Cowley 1997, Sponseller et al. 2001).

Similar to Margolis et al. (2001a), beaver impoundments proved to be important in reducing sediment concentrations in streams. In contrast, lack of impoundment in agricultural streams led to more lotic conditions and higher sediment concentrations and macroinvertebrate assemblages reflected this. In the Lac La Biche region, effects of conversion of forest to agricultural land will likely result in increased nutrient concentrations and export. Export of sediment from agricultural land to downstream water bodies will likely depend on whether beaver impoundments remain intact.

The effectiveness of regional biomonitoring using macroinvertebrates is dependent on recognising all assemblages naturally found in reference streams and identifying deviations from that condition (Rosenberg and Resh 1993). I found that reference conditions in the Lac La Biche region were highly variable, likely more than other regions because of the additional influence of beaver impoundments. Effects of conversion of forest to agricultural land on macroinvertebrates largely depend on whether beaver impoundments remain. This is because current speed is a significant determinant of macroinvertebrate assemblages. Because macroinvertebrate assemblages also responded to nutrient enrichment in a predictable way there is potential for using macroinvertebrates to indicate effects of conversion of forest to agricultural land. Further understanding of how local conditions and specific agricultural practices affect individual macroinvertebrate taxa in this region is needed, however, before indicators can be used effectively.

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Table 3.1. Percentage of watershed in various land use cover based on 2002 Landsat 7 imagery. Land cover classifications were based on the Alberta Ground Cover Characterization (AGCC) land cover classification system.

	Plamondon	Mission	Reutov	Deer	Cadieux	Goldie
Urban	7.9	10.1	4.9	6.5	5.8	3.6
Agriculture	42.9	38.9	10.4	4.3	3.4	0.0
Conifer	11.8	11.3	48.2	28.0	52.4	43.9
Deciduous	18.9	26.5	18.4	46.4	24.3	32.9
Mixed	2.8	3.7	2.0	1.5	1.4	2.5
Shrub	0.7	1.0	2.0	1.8	1.4	2.7
Grassland	0.2	0.1	1.7	0.3	1.0	0.8
Wetland	4.4	2.4	9.0	2.2	2.5	7.5
Bog	1.0	1.4	1.1	0.3	1.0	1.5
Water	9.6	4.7	2.4	8.8	6.9	4.7

Table 3.2. Physical, chemical and biological variables at all study streams. Data are mean ± SE derived from weekly (pH, temperature, conductivity, DO; n=16), monthly (P, N, sediment; n=8), or single (width, depth) samples taken through May, June, July and August 2003. (A) refers to catchments classified as agricultural and (R) refers to catchments classified as reference.

Parameter	Plamondon (A)	Mission (A)	Reutov (R)	Deer (R)	Cadieux (R)	Goldie (R)
WS Area (km^2)	111.4	46.1	48.0	109.0	28.4	46.6
Width (m)	2.66 ± 0.36	6.67 ± 0.42	5.79 ± 0.44	5.10 ± 0.22	3.49 ± 0.26	4.42 ± 0.28
Depth (m)	0.27 ± 0.02	0.95 ± 0.05	0.73 ± 0.05	0.79 ± 0.07	0.22 ± 0.02	0.92 ± 0.07
Temperature (°C)	15.69 ± 1.12	15.24 ± 0.83	15.55 ± 0.83	16.66 ± 1.04	14.55 ± 0.84	15.46 ± 0.81
Maximum Temperature	20.90	18.20	18.90	21.20	18.60	18.80
PH	7.14 ± 0.19	6.53 ± 0.14	6.65 ± 0.24	6.93 ± 0.17	6.23 ± 0.17	6.83 ± 0.15
Conductivity	445.8 ± 32.7	355.8 ± 40.3	178.3 ± 4.4	445.0 ± 71.69	118.9 ± 10.7	235.1 ± 9.9
Velocity	0.215 ± 0.052	0.003 ± 0.002	0.016 ± 0.007	0.094 ± 0.029	0.060 ± 0.023	0.055 ± 0.019
DO (mg/L)	5.99 ± 0.58	3.56 ± 0.86	3.34 ± 0.59	5.35 ± 0.44	2.86 ± 0.54	4.34 ± 0.73
Sediment (mg/L)	1.02^{E-01} ± 1.04^{E-01}	2.28^{E-03} ± 6.12^{E-04}	1.93^{E-03} ± 4.89^{E-04}	2.67^{E-03} ± 7.03^{E-04}	2.91^{E-03} ± 4.35^{E-04}	2.86^{E-03} ± 1.48^{E-03}
TP (mg/L)	0.458 ± .063	0.497 ± .100	0.140 ± .024	0.264 ± .035	0.260 ± .054	0.126 ± .019
TDP (mg/L)	0.280 ± .054	0.423 ± .096	0.084 ± .009	0.183 ± .029	0.231 ± .048	0.080 ± .012
TN (mg/L)	1.99 ± 0.119	2.46 ± 0.159	1.66 ± 0.089	1.92 ± 0.053	1.86 ± 0.102	1.40 ± 0.093
TDN (mg/L)	1.87 ± 0.095	2.29 ± 0.073	1.52 ± 0.060	1.82 ± 0.071	1.74 ± 0.099	1.31 ± 0.090
NO_3^- (mg/L)	0.051 ± 0.038	0.005 ± 0.002	0.002 ± 0.001	0.010 ± 0.004	0.004 ± 0.001	0.003 ± 0.001
NH_4^+ (mg/L)	0.032 ± 0.010	0.049 ± 0.012	0.020 ± 0.011	0.074 ± 0.038	0.041 ± 0.018	0.013 ± 0.003

Table 3.3. Marginal and conditional effects of significant parameters selected from forward step-wise selection for CCA ordination of all streams.

Parameter	Conditional Effects	Significance Value (p)	Marginal Effects
Total Phosphorus (TP)	0.458	0.002	0.458
% Native Grassland in Catchment	0.386	0.012	0.427
Total Dissolved Phosphorus (TDP)	0.263	0.006	0.410
Mean Velocity	0.129	0.004	0.400
% Agriculture in Catchment	0.061	0.010	0.381

Table 3.4. Abbreviations used for various taxa in CCA ordination biplots.

Taxa	Abbreviation	Taxa	Abbreviation
Aeshna	AESH	*Hyalella azteca*	HYAZ
Agabus	AGAB	*Hydrobius*	HYDB
Anopheles	ANPH	*Hydaticus*	HYDT
Arrenurus	ARRE	*Hydroporus*	HYDR
Bakerilymnaea	BAKL	*Laccobius*	LCBS
Banksiola	BANK	*Laccophilus*	LCPH
Callicorixa	CACX	*Lestes*	LEST
Caenis	CAEN	*Limnephilus* 3	LIM3
Ceratopogoninae	CERT	*Lymnaea stagnalis*	LYST
Chaoborus	CHAO	*Nephelopsis obscura*	NEOB
Chironmidae	CHRM	*Notonecta*	NOTO
Dina	DINA	Oligochaeta	OLIG
Dytiscus	DYTS	*Physa*	PHYS
Enallagma	ENAL	*Piona*	PION
Eylais	EYLA	*Promenetus exacuous*	PREX
Gammarus lacustris	GALA	*Prionocera*	PRIC
Glossiphonia complanata	GLOS	*Rhantus*	RHNT
Graphoderus	GRPH	*Simulium*	SIMU
Gyraulus	GYRL	*Siphlonurus*	SIPH
Gyrinidae	GYRN	*Somatochlora*	SOMA
Helisoma	HELI	*Stagnicola*	STAG
Helophorus	HELO	*Stratiomyidae*	STRT
Helobodella stagnalis	HEST	*Sympetrum*	SYMP
Haliplus	HLPL	*Valvata sincera helicoidea*	VALV

Table 3.5. Marginal and conditional effects of significant parameters selected from forward step-wise selection for CCA ordination of all beaver impounded streams.

Parameter	Conditional Effects	Significance Value (p)	Marginal Effects
Total Dissolved Phosphorus (TDP)	0.390	0.086	0.390
% Native Grassland in Catchment	0.390	0.024	0.350
% Deciduous Forest in Catchment	0.200	0.012	0.320
% Agriculture in Catchment	0.090	0.014	0.250

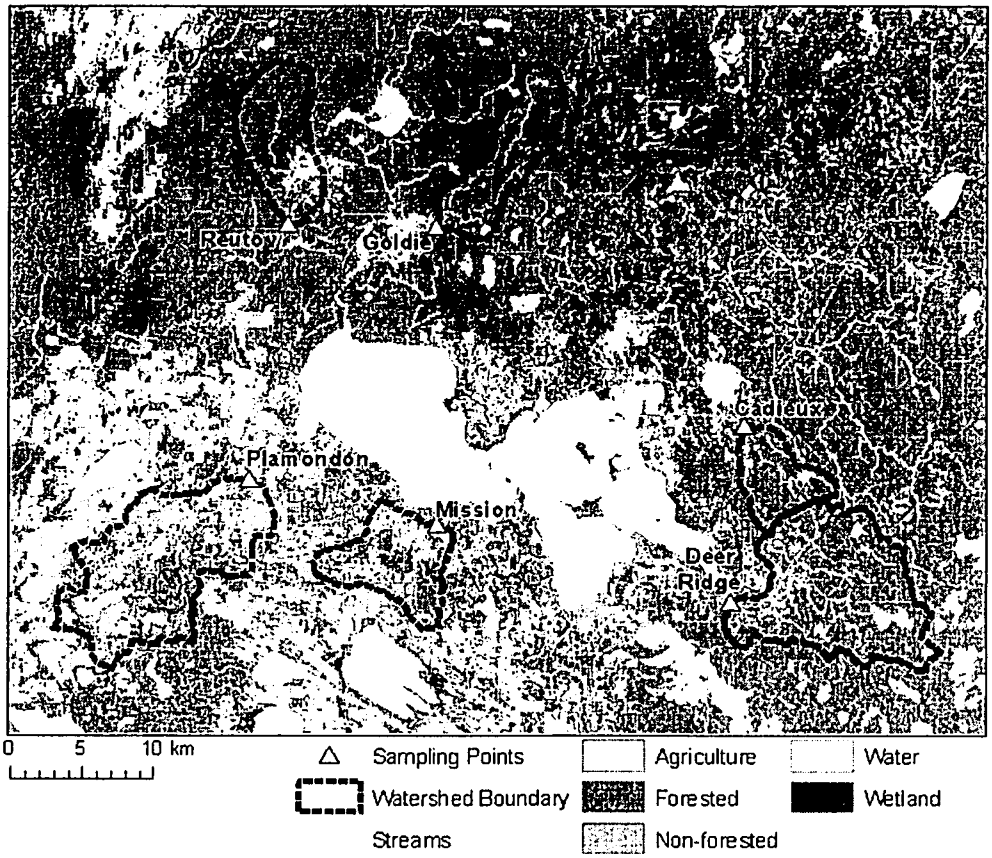

Figure 3.1. Land use in the 6 study catchments. Agricultural sites are Plamondon and Mission. Reference sites are Reutov, Deer, Cadieux, and Goldie. The large lake in the center is Lac La Biche.

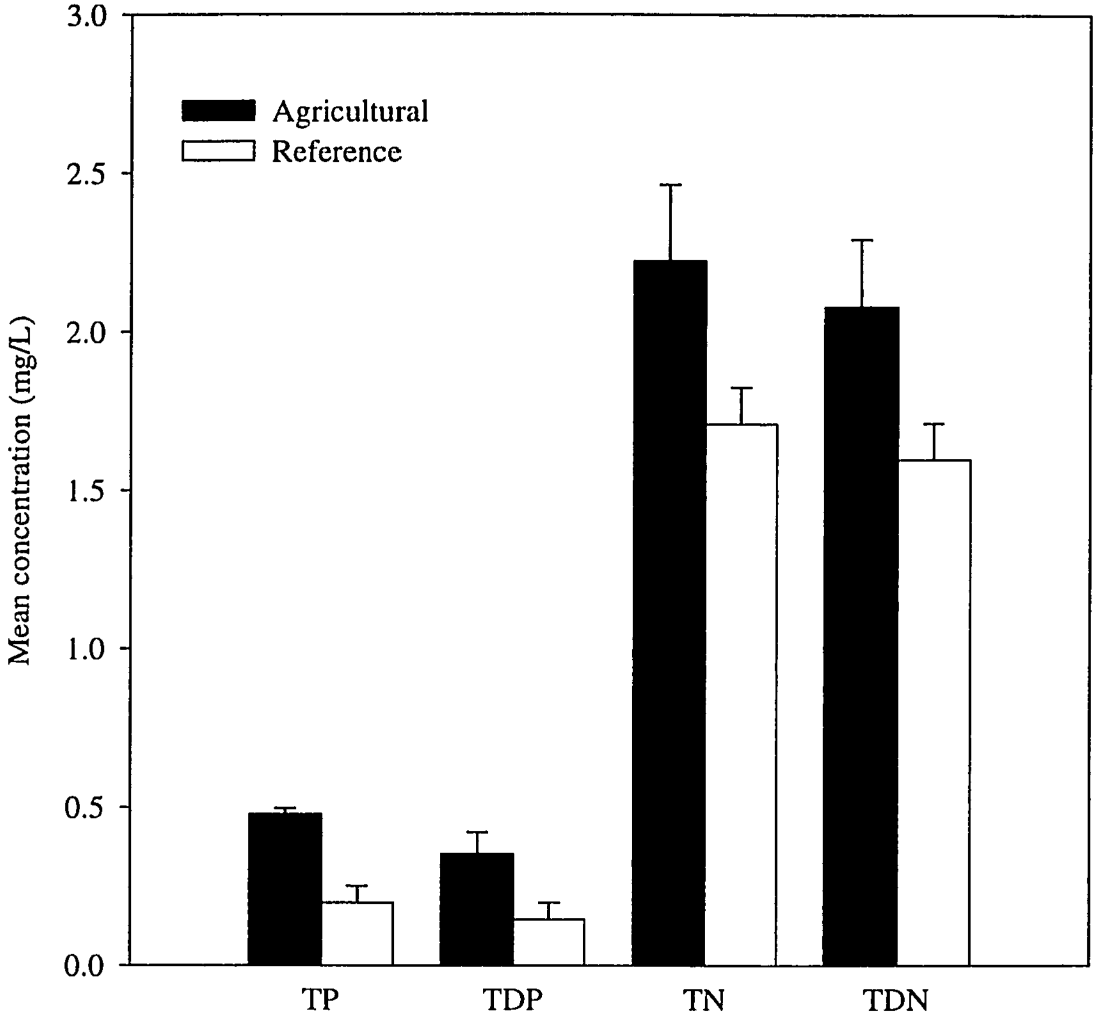

Figure 3.2. Mean total phosphorus (TP), total dissolved phosphorus (TDP), total nitrogen (TN), and total dissolved nitrogen (TDN) concentrations (mg/L) in agricultural streams (n=2) compared to reference streams (n = 4). Error bars represent the standard error.

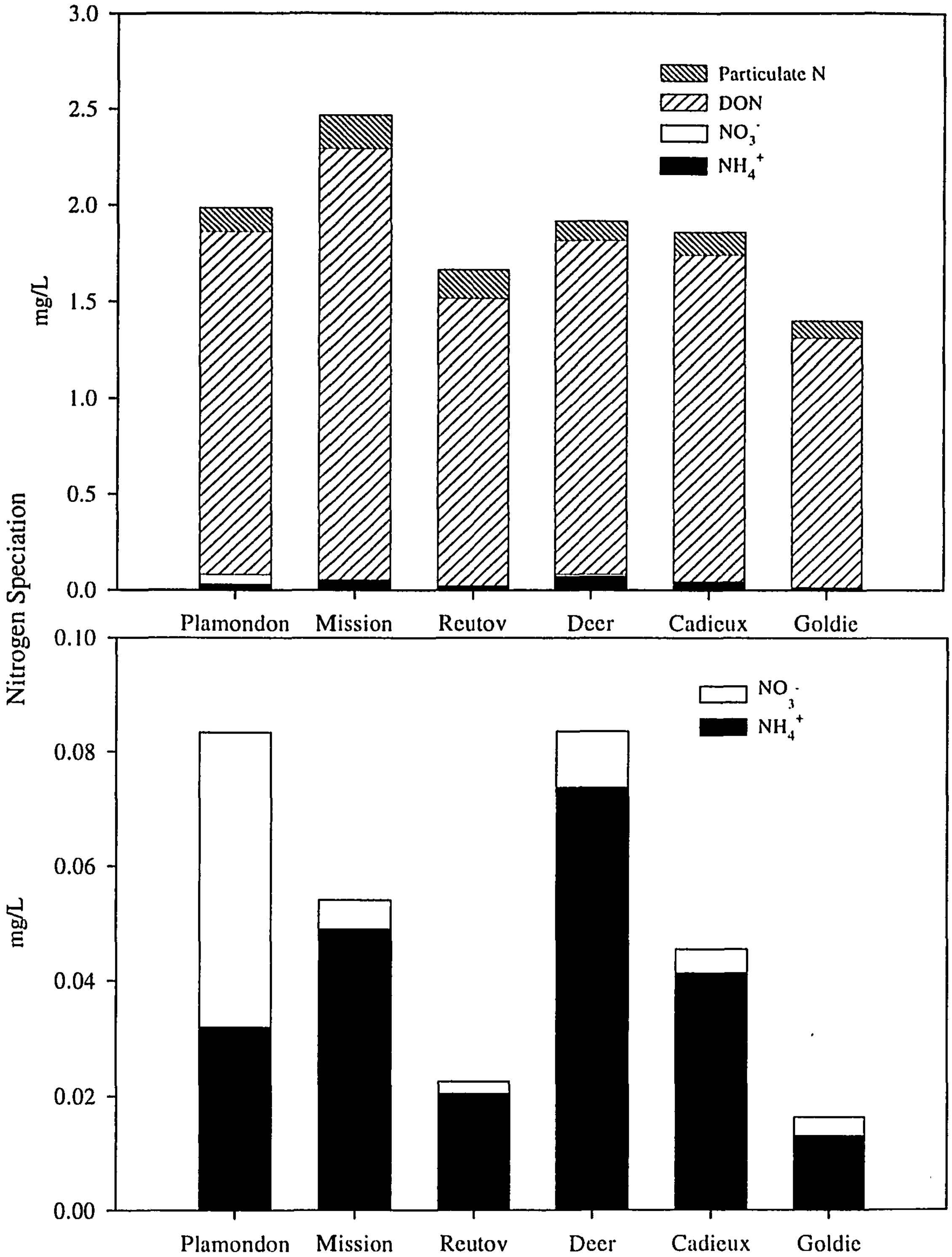

Figure 3.3. Mean concentrations (mg/L) of ammonium, nitrate, dissolved organic nitrogen, and suspended nitrogen in six boreal streams.

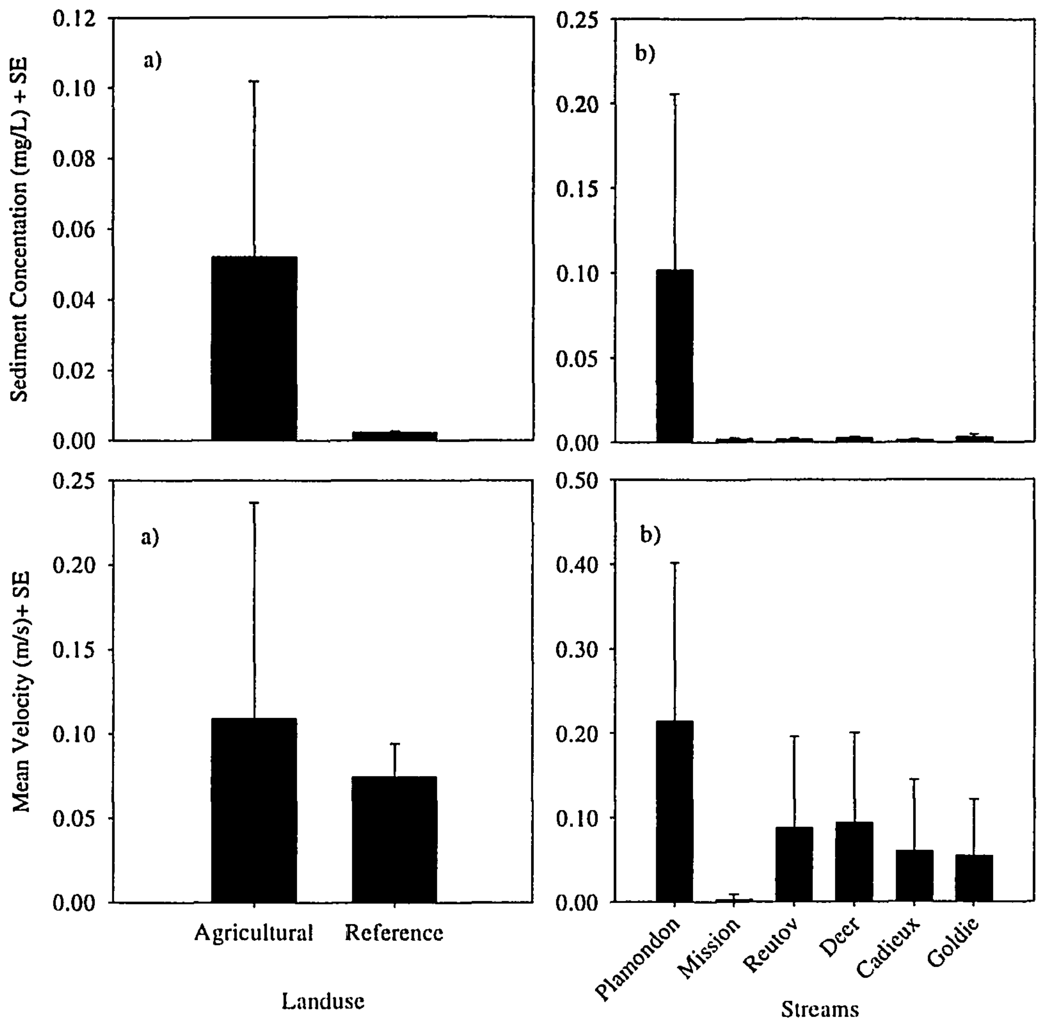

Figure 3.4. Mean sediment concentrations (mg/L) ± SE (top) and velocity (m/s) ± SE (bottom) in reference and agricultural streams (a) and individual streams (b).

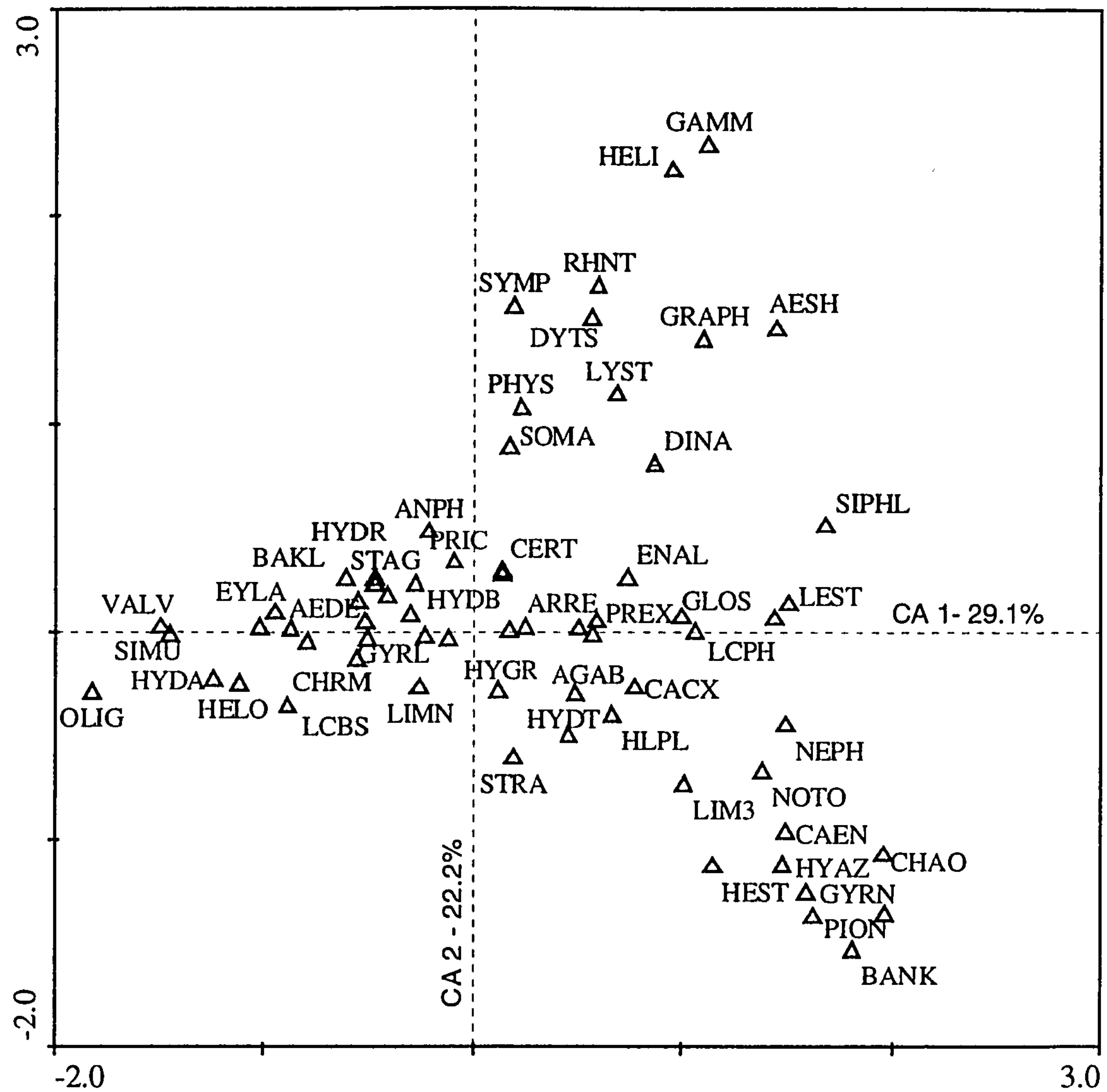

Figure 3.5. CA ordination biplot of taxa collected in June, July and August 2003 in six boreal streams.

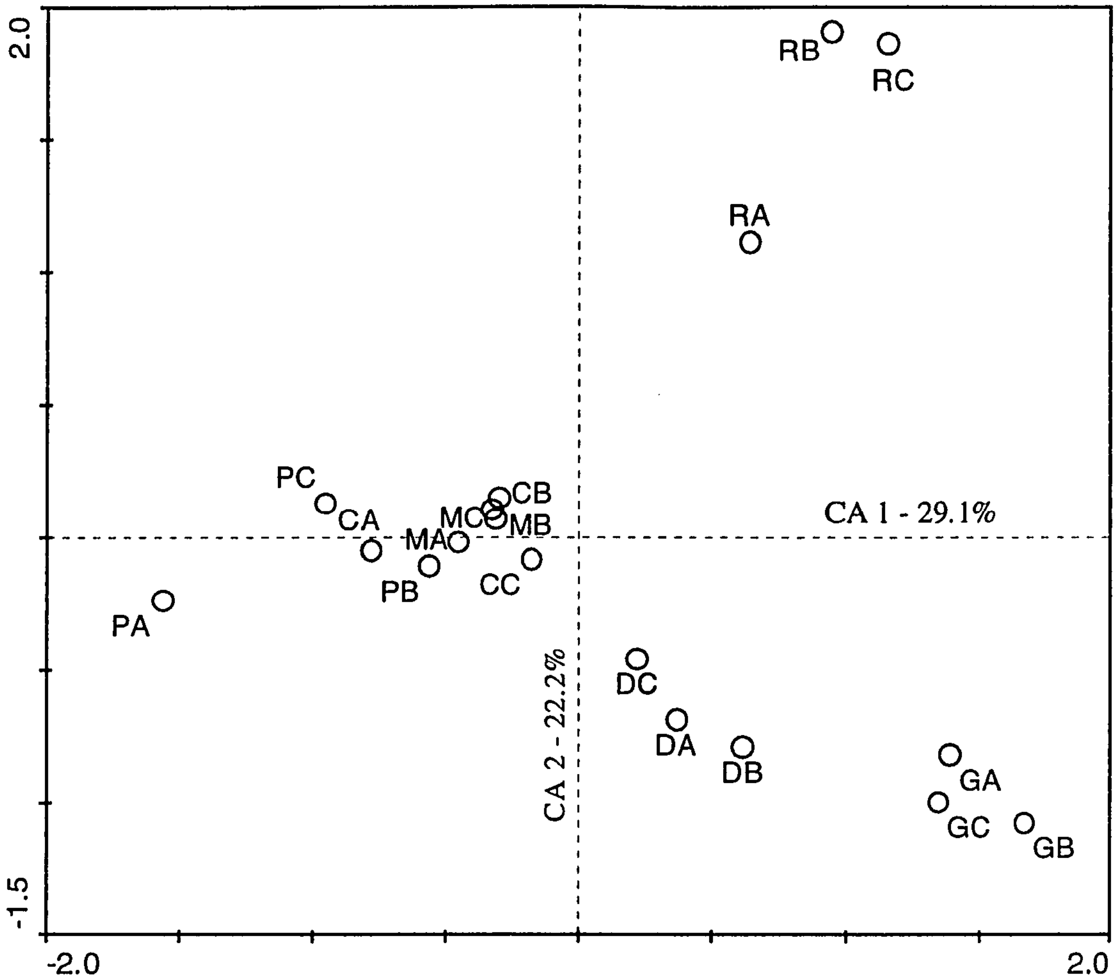

Figure 3.6. CA ordination biplot of 6 streams with three samples for each stream based on taxa data collected in June, July and August 2003.

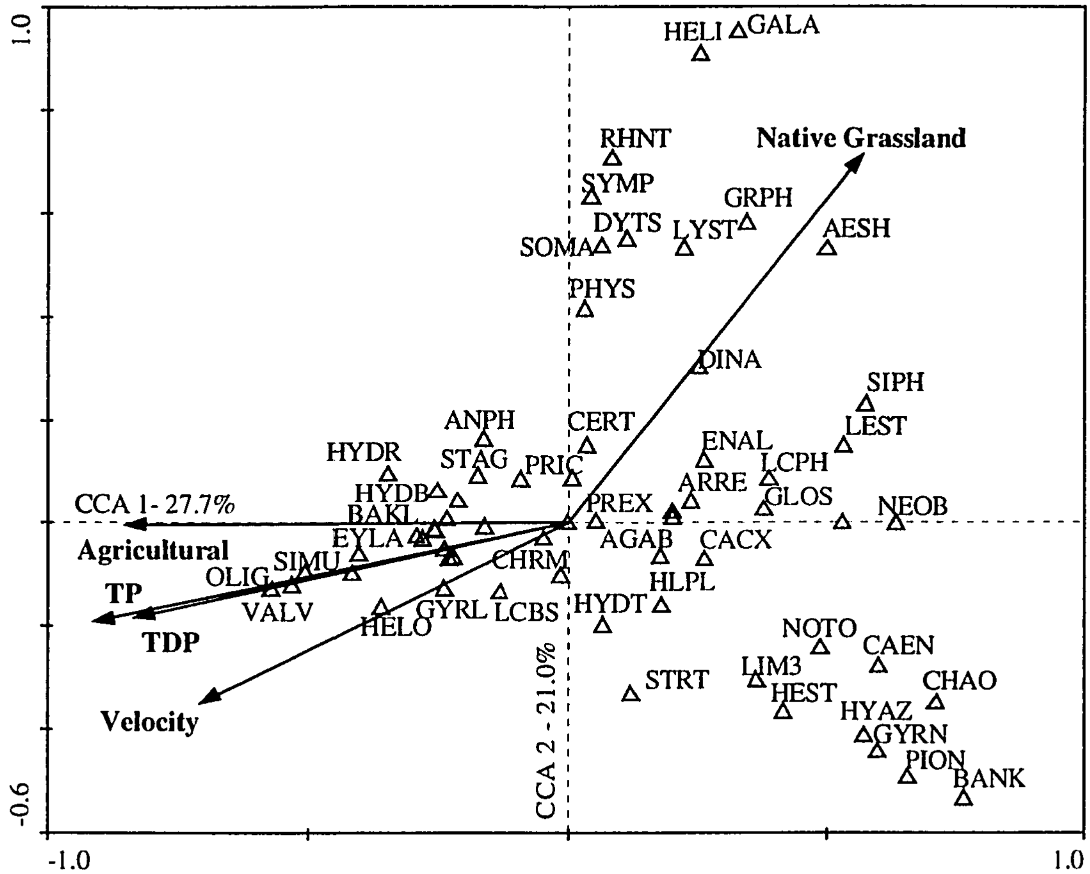

Figure 3.7. CCA ordination biplot of environmental variables with taxa collect in May, June and July 2003 for six boreal streams (see Table 3.4 for explanation of abbreviations).

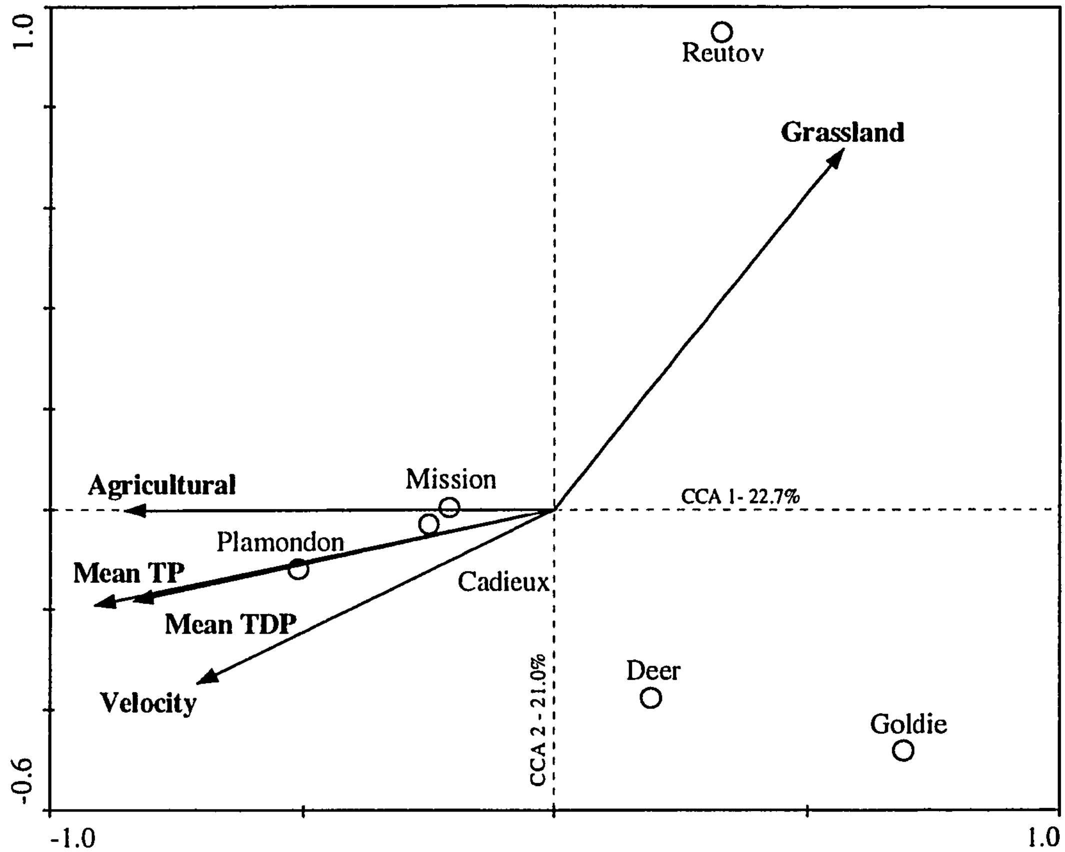

Figure 3.8. CCA ordination biplot of environmental variables with 6 boreal streams based on data collected in May, June and July 2003.

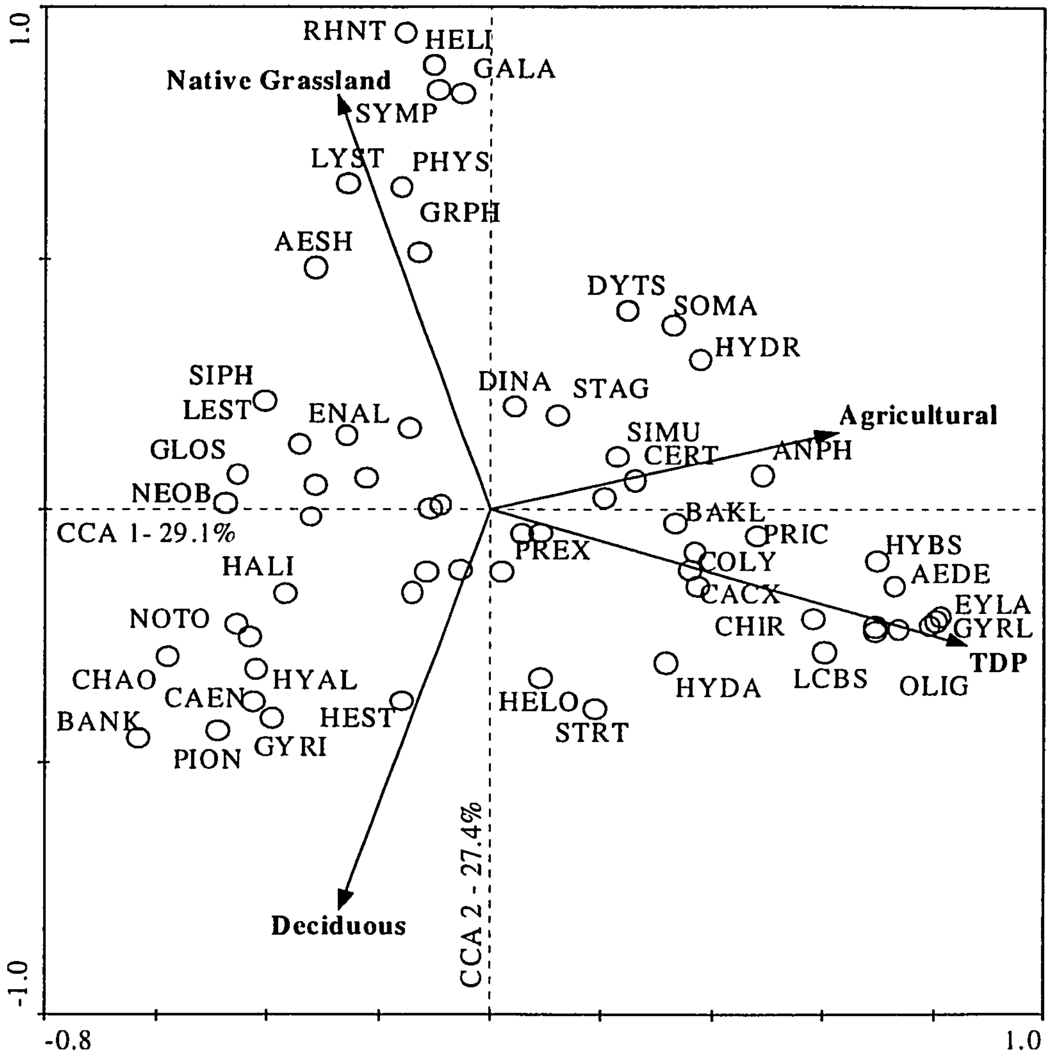

Figure 3.9. CCA ordination biplot of environmental variables with taxa collected in May, June and July for five impounded streams (see Table 3.4 for explanation of abbreviations).

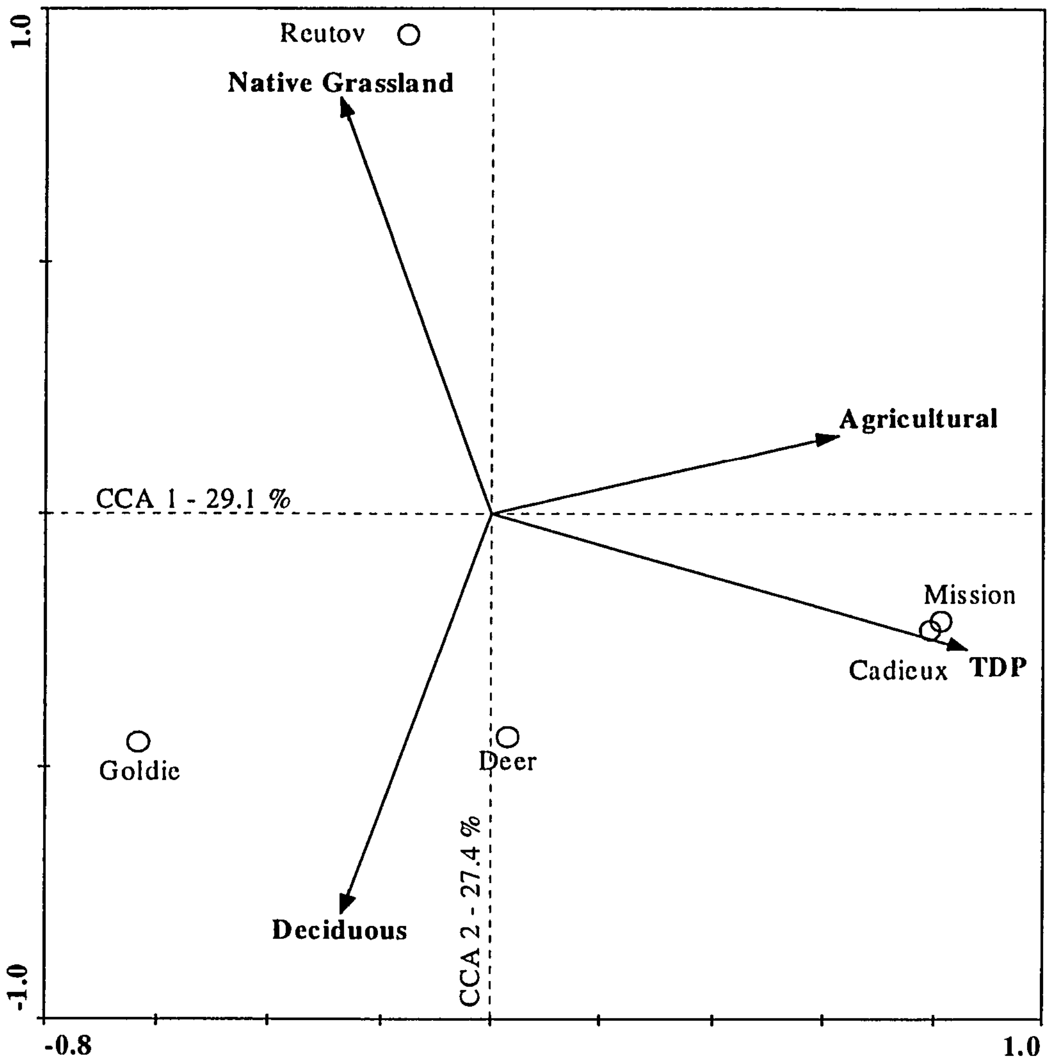

Figure 3.10. Canonical correspondence analysis ordination biplot of environmental variables with five impounded streams based on data collected in May, June and July.

Chapter 4: General Conclusions

Consistent with previous research (Dillon and Kirchner 1974, Omernik 1977, Vaithiyanathan and Correll 1992, Cooke and Prepas 1998), the results of this study indicate that agricultural catchments export more phosphorus than reference catchments. Therefore, continuing conversion of forested land to agricultural land within the Lac La Biche region will likely increase phosphorus loading to streams and downstream water bodies. Potential for eutrophication of downstream water bodies is of particular concern in this region because most phosphorus is exported in a dissolved form and is therefore bioavailable (Trimbee and Prepas 1987, Sharpley et al. 1992, Cooke and Prepas 1998). Although the pattern of speciation of exported phosphorus in my study is consistent with previous studies conducted in the same region (Cooke and Prepas 1998) and in other areas of Alberta (Anderson et al. 1998), studies conducted in other regions have found that phosphorus is mainly exported as particulate species (Omernik 1977, Sharpley and Menzel 1987, Vaithiyanathan and Correll 1992). This indicates that regional soil and catchment characteristics have considerable impact on phosphorus speciation and ultimately, on bioavailability of phosphorus in downstream water bodies. Predictions of land use change on nutrient loading and export should therefore consider regional soil and catchment characteristics.

Consistent with previous research, agricultural catchments exported more inorganic nitrogen annually than reference catchments (Anderson et al. 1998, Cooke and Prepas 1998, Gburek and Folmar 1999, Randall and Mulla 2001, Follett and Delgado 2002). This occurred despite the fact that total nitrogen was not higher in streams draining catchments with higher agricultural land use. Among-stream differences in speciation patterns of organic nitrogen exported from agricultural catchments were likely due to differences in in-stream biogeochemical processing (as influenced by beaver impoundments) and agricultural land use practices.

The influence of agricultural land on other in-stream parameters such as velocity, temperature, and hydrology were not consistent with the majority of findings in other regions (Sweeney 1992, Rutherford et al. 1997, Storey and Cowley 1997, Sponseller et al. 2001). This was most likely because reference streams were heavily influenced by beaver impoundments and these impoundments affect velocity, temperature, and hydrology. Beaver impoundments are very common in natural streams in the Lac La Biche region and as such, most natural streams are characterized by large open canopy areas, increased wetland area, large accumulations of silt, low dissolved oxygen conditions, low discharges, and variable macroinvertebrate assemblages. In contrast, some agricultural catchments have no beaver impoundments. Lack of beaver impoundments lead to more lotic conditions and macroinvertebrate assemblages reflected that. Similar to other areas (Naiman et al. 1988) beaver impoundments were important in reducing sediment export to down stream water bodies. In the Lac La Biche region, effects of conversion of forest to agricultural land will likely result in increased nutrient concentrations and export. Nutrient enrichment will thereby influence macroinvertebrate assemblages. However effects on slow-flowing macroinvertebrate assemblages and sediment export will also depend on whether beaver impoundments remain intact after conversion. High variability in macroinvertebrate assemblages suggests that further investigation into the influence of local factors on assemblage structure is needed before they can be used as bioindicators (Resh et al. 1996).

Ratios of bioavailable phosphorus to nitrogen suggest that effective phosphorus loading is greater than effective nitrogen loading. To reduce impacts on in-stream macroinvertebrate communities and downstream water bodies, land use management in the Lac la Biche region should focus on the reduction of soil phosphorus concentrations. Effective land management is only a partial solution, however. It may be necessary to limit conversion of forested land to agricultural land within watersheds where eutrophication of surface waters is a concern.

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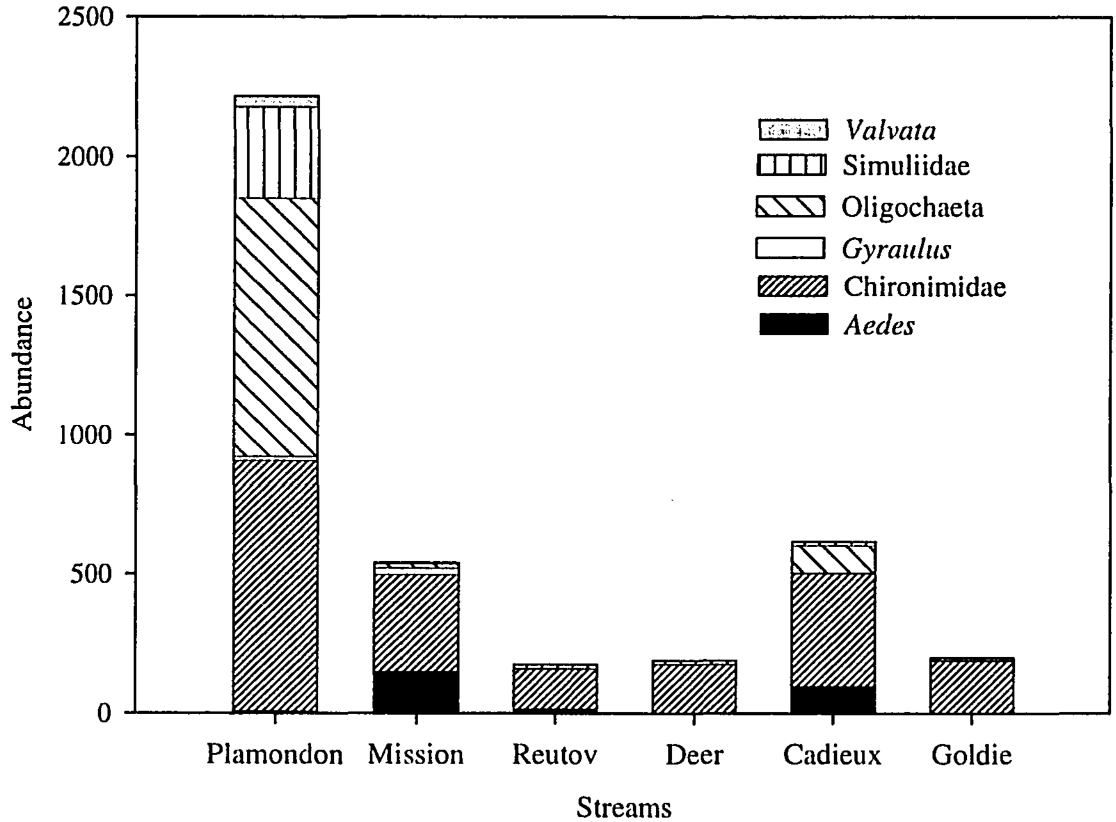
Appendix A: Summary of macroinvertebrate data

A.1. Bar graph of taxa abundance for taxa that appeared to be indicators of agricultural land use and nutrient enrichment.



A.2. Summary table of taxa raw abundance in each stream. Abundances are combined totals from May, June, and July samples. Cladocera and Copepoda were not enumerated and were present in all streams.

Taxa/Stream	Plamondon	Mission	Reutov	Deer	Cadieux	Goldie
Aedes	8	150	16	1	97	3
Aeshna	0	0	2	0	0	1
Agabus	5	9	6	3	3	13
Anopheles	10	12	10	3	13	0
Arrenurus	1	0	2	0	2	3
Bakerilymnaea	24	4	4	1	7	3
Banksiola	0	0	0	0	0	3
Caenis	1	7	12	4	0	69
Callibaetis	35	0	7	1	3	6
Callicorixa	7	19	16	13	3	32
Ceratopogonidae	15	46	34	13	11	16
Chaoborus	2	0	41	0	0	314
Chironomidae	901	349	144	176	406	186
Colymbetes	46	32	15	6	25	15
Corisella	0	1	0	0	1	0
Corixidae	12	48	20	10	43	27
Culex	0	0	0	1	1	0
Culiseta	0	0	0	0	1	0
Dina	0	1	5	0	3	3
Dixella	0	0	0	0	0	3
Dolichopodidae	0	2	0	0	0	0
Dytiscus	0	10	16	0	3	1
Enallagma	104	33	170	14	11	184
Ephydridae	0	1	0	0	1	0
Erpobdella punctata	0	0	0	0	0	4
Eylais	2	5	0	0	1	0
Gammarus lacustris	0	0	455	0	36	10
Glossiphonia complanata	2	0	3	0	0	5
Graphoderus	1	4	26	0	1	8
Gyraulus	14	24	1	1	3	4
Gyrinidae	0	1	0	0	0	5
Haliplus	26	2	10	5	2	33
Helisoma	14	1	141	0	7	0
Helobdella stagnalis	0	0	0	0	1	2
Helophorus	6	1	0	0	0	1
Hyalella azteca	22	1	39	150	51	681
Hydaticus	0	0	0	0	2	1

Hydrachna	0	0	6	2	1	0
Hydrobiidae	31	14	0	1	3	1
Hydrobius	3	4	2	0	13	0
Hydrochara	0	2	0	0	0	0
Hydroporus	15	3	4	0	2	0
Hygrotus	7	2	2	2	0	4
Ilybius	0	0	0	0	0	2
Laccobius	1	0	0	1	3	0
Laccophilus	2	1	11	1	4	15
Lestes	0	1	9	1	0	11
Leucorrhinia	0	0	4	0	0	0
Libellula	2	0	0	0	0	0
Limnephilus1	0	0	0	0	24	1
Limnephilus2	46	4	39	1	8	62
Limnephilus3	0	2	0	0	0	0
Limnesia	3	0	1	1	1	9
Limnoporus	31	0	5	10	3	9
Limoniinae	0	0	0	0	0	2
Lymnaea stagnalis	4	0	11	0	0	3
Nephelopsis obscura	0	0	1	0	0	2
Notonectidae	8	2	9	1	2	46
Oligochaeta	929	94	1	0	98	7
Physa	117	10	154	3	3	30
Piona	0	1	0	3	1	29
Placobdella papillifera	0	0	1	0	0	0
Prionocera	3	15	4	3	0	1
Promenetus exacuous	2	5	6	0	6	10
Pyrrhalta	1	0	0	0	0	0
Rhantus	3	0	7	0	0	0
Sciomyzidae	0	0	0	1	0	0
Scirtidae	0	1	0	0	1	0
Simulium	327	0	12	11	12	0
Siphlonurus	0	0	11	0	0	12
Somatochlora	0	2	2	0	0	0
Stagnicola	36	4	15	0	11	8
Stratiomyidae	0	1	0	4	0	0
Sympetrum	8	2	17	0	0	0
Tabanidae	0	2	0	0	0	0
Theromyzon	18	0	8	14	0	3
Valvata tricarinata	38	0	0	0	0	0

A.3. Table of macroinvertebrate taxa diversity (Simpson index and richness) and number of unique species for each stream based on data collected in May, June and July 2003. (A) represents agricultural streams and (R) represents reference streams.

Parameter	**Plamondon (A)**	**Mission (A)**	**Reutov (R)**	**Deer (R)**	**Cadieux (R)**	**Goldie (R)**
Simpson's Diversity Index	0.78	0.79	0.87	0.67	0.80	0.83
Total Richness	50	49	40	38	40	53
Number of Unique Species	3	4	3	3	0	2