Wetland Drainage Effects on Prairie Water Quality

Final Report

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Submitted by:

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1 Summary

This report describes factors influencing the spatial variation in wetland water quality and how drainage of wetlands affects downstream receiving waters in terms of their water quality and biotic health. The specific objectives of this work were to: 1) characterize the spatial and temporal variation in water quality of prairie potholes after snowmelt; 2) quantify solute export along a newly constructed wetland drainage ditch; 3) characterize solute export from drained pothole wetlands; 4) determine the extent to which stream water quality is influenced by wetland drainage; 5) contribute to the understanding of how wetland drainage affects ecosystem health. The research was conducted at the Smith Creek watershed, southeastern Saskatchewan, where there has been controversy over recent renewed efforts to drain wetlands to increase agricultural production.

A total of 67 wetlands were sampled following snowmelt in 2009 to determine whether spatial variations in wetland water quality could be attributed to different land cover and permanence classes. It was found that crop wetlands had greater TP and K than wood and grass wetlands; TP, TDN, and DOC were higher in seasonal than permanent wetlands; and salts were lower in wood compared to crop and grass wetlands. Measures of water quality of one permanent wetland over a 20 week period in 2008 showed that the wetland acted as a trap for nutrients, salts and bacteria. Variations in salts and DOC loads were linked to hydrological processes whereas variations in nitrogen, phosphorus, and bacteria loads were associated with both biotic and hydrological processes.

The permanent wetland studied was experimentally drained in November 2008. The experiment demonstrated that wetland water quality was an important control of water quality in drainage water. The wetland ditch acted as a simple conduit, meaning that there was little loss or gain of solutes along the length of the ditch. Results also show that the efficiency with which a wetland is drained and the water quality characteristics of the wetland are the factors critical to determining solute exports via ditches. In spring 2009, water quality along seven ditches and five natural connections that form between wetlands (termed spills) was found to be similar, except for concentrations of TDN, DOC, HCO_3, K, and Ca that were higher in ditches than spills. As in the wetland drainage experiment, little change in the water quality along ditches and spills was found, likely due to the low temperatures occurring in spring that can restrict biotic processing. The important difference however was the physical characteristics of ditches and spills. The long ditches connect wetlands to streams whereas short spills connect wetlands to other wetlands. Thus ditches have a greater potential to contribute to downstream loading of nutrients, salts and bacteria.

Water quality of streams draining three subbasins of the Smith Creek watershed with varying wetland drainage intensities was compared during the 2009 freshet. Nearly all export coefficients for the water quality parameters studied were higher in streams draining subbasins with greater wetland drainage. Total solute export was greatest in the subbasin with medium high wetland drainage, due in part to its comparatively large size. Although water samples generally did not exceed guidelines for nitrogen and salts, the provincial objective for TP was frequently exceeded. Consequently, wetland drainage is likely to exacerbate downstream eutrophication.

The macroinvertebrate-based ecosystem health of Smith Creek watershed in stream reaches downstream of drainage activity was assessed and compared to ecosystem health at similar sites across southern Saskatchewan. Sampling was conducted through 2008 and 2009 at four and eight sites in each of these respective years. It was found that wetland drainage does not

significantly impact the ecosystem health of receiving waters, and may in fact benefit the macroinvertebrate assemblage, at least in the short term, by increasing the amount of water and habitat available. However, this project did not address the potential decline in ecosystem health in source-wetlands that are being drained, where the amount of habitat and water is decreasing.

Overall, this study represents the first field-based research in the Prairie Pothole Region on the effects of wetland drainage on downstream water quality and stream ecological health. Therefore, results should help support effective management decisions regarding future pothole drainage or restoration, balancing public and private costs and benefits. Several specific recommendations were made regarding the challenges faced by this study. It is recommended that similar studies be conducted in other geographic areas of the prairies where climate, soils, wetland configuration and drainage may produce differing results.

2 Introduction

Prairie pothole wetlands are small, shallow depressions that typically lack surface water connections and are ubiquitous across the prairie pothole region (PPR) of North America (Figure 2.1). The PPR is estimated to cover approximately 715 000 km^2 (Euliss et al., 1999). Pothole wetlands formed during the last glacial retreat that created the hummocky, undulating terrain typical of the prairies (Tiner, 2003). About 40%, or 156 000 km^2, of the PPR located in Canada consists of hummocky moraines that have a wetland density of 18 wetlands/km^2. The remaining 60% of the Canadian PPR landscape is comprised of mostly lacustrine and fluvial materials that have, on average, 5 wetlands/km^2 (National Wetlands Working Group, 1988). Pothole wetlands provide important hydrological and ecological functions. For example, the PPR represents only 10% of the continent's waterfowl breeding area but it produces half of North America's waterfowl in an average year (Smith et al., 1964; Batt et al., 1989).

Figure 2.1: Map of the prairie pothole region of North America (Euliss et al., 1999)

Many of the wetlands located in the hummocky moraine region are isolated prairie potholes. These potholes range in permanence from those that contain water for only a few days following spring snowmelt, to those that are continuously inundated. Most potholes do not normally contribute to streamflow (Stichling and Blackwell, 1957). During very wet conditions temporary surface connections can occur among them (Leibowitz and Vining, 2003; Winter and LaBaugh, 2003), which is referred to as the 'fill and spill' mechanism (Spence and Woo, 2003; Spence, 2006).

Water quality is defined differently by engineers, ecologists, hydrologists, etc., and can encompass a wide range of water quality descriptors depending on the end user, the context of interest, and natural conditions (Meybeck, 2005). Water quality, as defined in this report, is based on chemical, physical, and biological descriptors that affect the structure and function of ecosystems as well as those that negatively impact human and livestock health if in elevated concentrations. Isolated prairie potholes exhibit high spatial and seasonal variations in water quality (e.g., LaBaugh and Swanson, 2004). Previous studies have documented variations in select water quality variables in relation to individual factors such as wetland permanence, water sources, and land use of the catchment area (Rözkowska and Rözkowski, 1969; Miller et al., 1985; LaBaugh et al., 1987; Swanson et al., 1988; Neely and Baker, 1989; Detenbeck et al., 2002; Waiser, 2006). For example, Driver and Peden (1977) studied prairie pond chemistry in Saskatchewan and Manitoba and found differences among temporary, semi-permanent, and permanent ponds. Crosbie and Chow-Fraser (1999) found higher nutrient concentrations in Ontario marshes that a greater proportion of agricultural land in their watersheds. However, a concurrent assessment of nutrients and salts in potholes is needed given that different mechanisms interact to drive their concentrations (LaBaugh et al., 1987; Wetzel, 2001). As well, factors such as permanence and land cover type do not act in isolation on the landscape and could instead interact with one another to regulate wetland water quality. Studies examining the potential governing influence of the interaction of permanence and land cover type on pothole water quality are lacking.

Approximately 40-70% of the potholes in the western prairies have been drained since 1900 to increase agricultural production (Tiner, 1984; Dahl, 1990; Brinson and Malvarez, 2002). Recently, there have been renewed efforts to drain potholes (Watmough and Schmoll, 2007). Temporary wetlands are the most common permanence class in the PPR, and the most impacted by drainage and farming practices (Euliss et al., 2001). Issues surrounding pothole drainage centre around attempts to balance the social benefits and private costs associated with potholes on agricultural lands (Porter and van Kooten, 1993). The social benefits of potholes are believed to be water storage and flood attenuation, wildlife habitat, and improvements of water quality (Saskatchewan Watershed Authority, 2007a). Costs accrued by private landowners include the nuisance of farming around potholes, delayed seeding dates, and the foregone opportunity to increase agricultural production (Scarth, 1998; Brinson and Malvarez, 2002; Saskatchewan Watershed Authority, 2007a). Drainage impacts remain a critical information gap for the agricultural community, with the potential to have broad-scale economic impacts.

Drainage ditches create new surface water connections between wetlands that were previously isolated and other wetlands, roadside ditches, and streams. The new connections transform the hydrologic conditions of the prairies such that previously non-contributing areas now contribute to streamflow. Figure 2.2 illustrates the different ways that surface water can enter a stream where drainage ditches are present (McAllister et al., 2000). Isolated wetlands have the potential to intercept and store surface runoff (Figure 2.2b). Drainage ditches can transport water from one wetland to another (Figure 2.2c), which can cause local flood damage to cropland or communities surrounding the terminal wetland. Drainage ditches can also transport surface water runoff directly from a wetland to a stream (Figure 2.2d). Drainage of many potholes has the potential to significantly increase downstream flood frequencies and magnitudes (Campbell and Johnson, 1975). For example, Yang et al. (2008) used the SWAT model to show that a loss of 70% of 1968 wetlands in the Broughton's Creek watershed in western Manitoba to drainage and degradation increased the basin's contributing area by 31% (19 km^2), increased peak flows by 18%, and increased stream flow by 30%. Pomeroy et al. (2009) used the newly developed Prairie Hydrological Model to show a 117% increase in streamflow volumes with complete drainage of the wetlands in Smith Creek watershed, Saskatchewan based on the 2007-2008 meteorological conditions. Further, a Saskatchewan Watershed Authority (2008) assessment identified agricultural drainage as an important contributor to high water levels in terminal (Waldsea) and near-terminal (Deadmoose, Houghton, Fishing) lake basins in 2007.

Pothole wetlands have been shown to trap nutrients, salts and bacteria from catchment runoff (Neely and Baker, 1989; Crumpton and Goldsborough, 1998). Wetland water quality is thus expected to be an important control of water quality in drainage water. As wetland drainage connects potholes to downstream water bodies, it has the potential to generate excessive nutrients and sediment pollution that may impact the water quality, and ultimately the ecosystem health of receiving streams and rivers (Leibowitz and Vining, 2003; Whigham and Jordan, 2003). To date, however, there have not been field studies to support or refute this conjecture. Despite a lack of data, the popularly held belief that wetland drainage will impair downstream waters has influenced decision making in the courts. For example, a recent decision by the Water Appeal Board (16 August 2007) in the case of Ducks Unlimited Canada vs. Jack Kalmakoff to close a drainage ditch was due, in part, to a perceived risk of impact to downstream aquatic systems resulting from wetland drainage. While no studies of water quality in wetland drainage ditches has been conducted in the PPR, there is a body of literature describing nutrient export along upland drainage ditches located in low order agricultural watersheds, which may prove useful to understanding solute export from potholes drainage. Main findings are reviewed in the following two paragraphs.

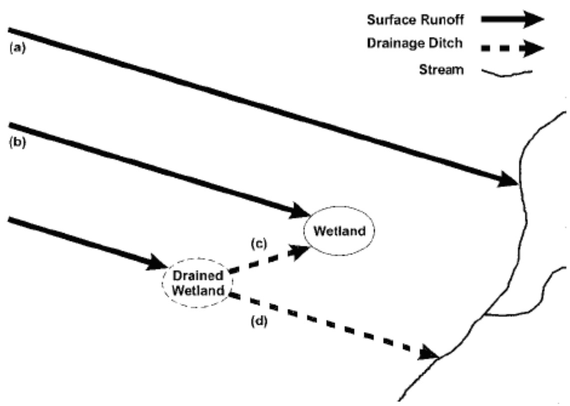


Figure 2.2: Potential pathways of surface runoff. Runoff can a) enter streams directly; b) enter and be stored in wetlands; stored runoff can be released from wetlands via drainage ditches and flow either into c) other wetlands or d) streams (modified from McAllister et al., 2000).

The water regime in upland drains is characterized by the advective flow of water downstream. Turbulent mixing encourages oxygenation of the water. For redox-sensitive water quality parameters, increased oxygenation during transport along ditches would be expected to change their concentrations. Kemp and Dodds (2001) showed stimulation of nitrification and the inhibition of denitrification in a 2nd order prairie stream with higher dissolved oxygen (DO) concentrations, which would likely result in a net reduction in N removal along the stream (Birgand et al., 2007). Stimulated nitrification rates can also lead to increased transport of nitrogen (N) to receiving streams since nitrate (NO_3^-) is quite mobile compared to ammonium (NH_4^+), which is easily adsorbed to negatively charged ditch surface particles (Strock et al., 2007). Solute sedimentation rates vary inversely with the velocity of flowing water (Julien, 2002), affecting concentrations of water quality parameters that have an affinity for the particulate phase, for example, phosphorus (P).

Recent studies have shown relatively high retention of mostly P (as well as some N) in upland ditches where soils have substantial sorption or retention capacities (Sharpley et al., 2007; Strock et al., 2007). Nguyen and Sukias (2002) showed ditch sediments in New Zealand contained 42-57% of P originating from agricultural catchments loosely bound with Al, Fe, and CO_3^-, and 6-39% of P stored more permanently in the sediment as refractory P. They also showed the proportion of P transported was governed by the form of P and the retention characteristics of the ditch sediments. Periodic high flow events that occur during snowmelt and significant rainfall events increase velocity, shear force, and scour along the ditch bottom causing the re-suspension of sediments and organic matter and consequently their downstream transport (Sharpley et al., 2007; Birgand et al., 2007). Also, macrophytes and algae living in the drains have been shown to temporarily store nutrients (Needleman et al., 2007). When they die during season changes, water level drops, or other causes, they can contribute to the organic matter content of the water column and also to the accretion of drain sediment. Macrophytes can also play an indirect role in contaminant retention by reducing flow velocities and re-suspension rates, and increasing sedimentation rates (Birgand et al., 2007).

There is very little published research on the relationship between land use and wetland drainage on stream water quality in the Canadian Prairies. Carpenter et al. (1998) is the seminal paper outlining the issue of how stream water quality is controlled by land use. Alberta Agriculture and Rural Development prepares annual reports cards for stream water quality, as measured through a water quality index, in Alberta's watersheds with varying agricultural intensities. Saskatchewan Watershed Authority makes similar reports for watersheds in Saskatchewan using a different water quality index. Stream biotic integrity is commonly assessed through measures of the aquatic macroinvertbrate community. Summaries of the ecosystem structure and function in macroinvertebrate communities are used as proxies of ecosystem health, and are frequently applied to describe impacts of human activities. In particular, specific components of the macroinvertebrate assemblage are sensitive to abiotic changes in the aquatic environment, and the expression of these changes as metrics relative to reference, or unaffected streams, is to evaluate impact. By identifying metrics that respond to specific stressor, management can be applied to the stressors in a watershed that are causing ecosystem degradation.

Leibowitz and Vining (2003) suggest that stream water quality is likely to be altered by wetland drainage as this would hydrologically connect previously isolated wetlands to streams. Similarly, how Saskatchewan macroinvertebrate assemblages respond to stressors brought about through wetland drainage (e.g., nutrients, sediment etc.) is poorly understood. Tangen et al. (2003), in their attempt to create a macroinvertebrate index of biotic integrity found only a weak correspondence of macroinvertebrate assemblages and land use in the Prairie Pothole region of North Dakota due to an inadequate number of low impact sites. Wetland drainage was not one of the land use factors considered. Further, as benthic macroinvertebrate assemblages are affected by a number of factors in addition to wetland drainage, changes in assemblage composition should currently be interpreted as a complimentary tool to primary water quality tests.

The overall objective of this research project was to determine changes in water quality of streams and impacts to stream ecosystem function associated with wetland drainage. The specific research objectives of this study were to:

1) characterize spatial and temporal variations in water quality of prairie potholes after snowmelt;
2) quantify solute export along a newly constructed wetland drainage ditch;
3) compare solute export from naturally and artificially drained pothole wetlands;
4) determine how stream water quality is influenced by wetland drainage; and
5) contribute to the understanding of how wetland drainage affects ecosystem health by quantifying benthic macroinvertebrate communities.

Saskatchewan Watershed Authority was sub-contracted to complete objective 5.

3 Study Site

The research was conducted at Smith Creek watershed (SCW; 50°50'4"N 101°34'48) in southeastern Saskatchewan. The watershed is ~400 km^2 (Pomeroy et al., 2009) with an effective contributing area of ~58 km^2 (Environment Canada, 2006). The contributing area is continuously increasing as producers drain more potholes to increase arable area. Smith Creek drains into the Assiniboine River and then Lake Winnipeg where excessive N and P loadings are causing eutrophication (Armstrong, 2002). Snowmelt is the dominant, and often only, hydrological event.

The terrain of SCW is level to undulating and rolling. SCW is located in the Aspen Parkland Continental Prairie Wetland sub-region (National Wetlands Working Group, 1988). Soils in the watershed are a mixture of Black (Oxbow) and Thick Black (Yorkton) chernozems formed in loamy glacial till (Agriculture and Agri-Food Canada, 2009).

The regional climate is semi-arid. Average seasonal temperatures are -17.9^{o}C and 17.8^{o}C for the winter and summer, respectively (as measured at the Yorkton airport, ~50 km west). The mean (1942-2009) annual precipitation is 438 mm with 28% falling as snow. Precipitation amounts prior to the 2008 study period were in the 53rd, 34th, and 26th percentiles for winter 2007, May – October 2007, and winter 2008, respectively. Summer 2008 and winter 2009 precipitation amounts were in the 60th and 66th percentiles, respectively. Total precipitation for the months of May 2008 (18 mm) and July 2008 (208 mm) were respectively in the 13th and 95th percentile of values measured at the Yorkton weather station.
Land uses were determined by Pomeroy et al. (2009) using unsupervised classification of SPOT 5 images from October 1, 2008. The dominant land use is agriculture, which occupies 54% of the watershed. Common crops include wheat, canola, and flax. Eight percent of the watershed is grassland and pasture. Wooded areas and wetlands/open water account for 23% and 11% of the watershed, respectively. Wooded stands are characterized by trembling aspen (*Populus tremuloides*) with pockets of balsam poplar (*Populus balsamifera*), together with an understory of mixed herbs and tall shrubs. Wetland vegetation is predominately willow (*Salix spp.*), cattails (*Typha latifolia L.*), sedges (*Carex spp.* and *Scirpus spp.*), duckweed (*Lemna spp.*), pondweed (*Potamogeton spp.*), and water smartweed (*Polygonum amphibium L.*). Grasslands are comprised largely of western porcupine grass (*Stipa curtiseta*), plains rough fescue (*Festuca hallii*), pasture sage (*Artemisia frigida*), and Lewis wild flax (*Linum lewisii*). The majority of the wetlands in SCW belong to the marsh and shallow open water classes (National Wetlands Working Group, 1988). The average wetland density in the basin is ~20 wetlands/km^2. Many of the wetlands in the basin are typical, isolated prairie potholes that formed in glacial depressions, which at average surface water level have no surface inflows or outflows. The wetlands range from temporary through permanent. Recently, a number of drainage ditches have been constructed to enhance agricultural production. These drainage ditches either completely eliminate the wetland or drastically lower the water levels and create downstream connections with other wetlands, roadside ditches, or streams. Between 1958 and 2001 the portion of land cover occupied by wetlands in the Smith Creek watershed decreased ~60%%. Drainage has been most prolific in the eastern and southwestern areas of the watershed (Figure 3.1). Many water control structures such as road culvert gates exist in the basin and are operated by local farmers to regulate the runoff in their cropland areas; the gates are closed during extremely high runoff periods, i.e. during fast snowmelts or intense rain storms but remain open otherwise.

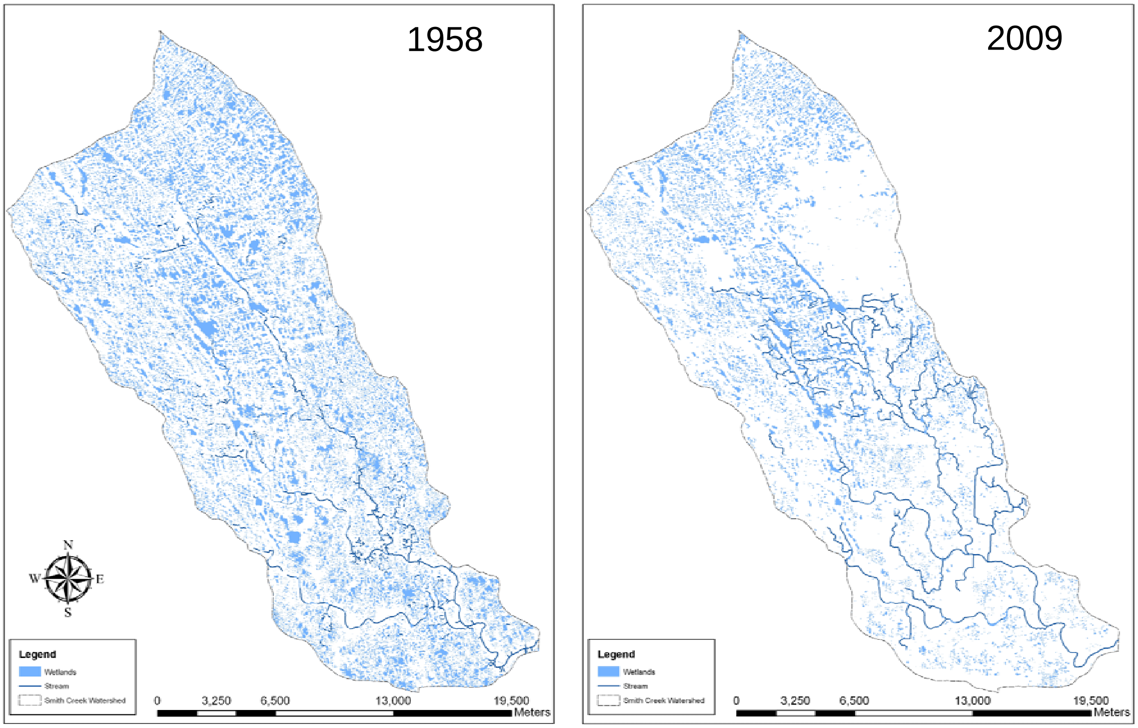


Figure 3.1: Historic and current day wetland and stream channel distribution in Smith Creek watershed, produced in conjunction with Ducks Unlimited Canada.

4 Methods

4.1 Wetland water quality characterization

Water quality in 67 wetlands (Figure 4.1) was assessed following snowmelt in 2009. Wetland selections were made based on obtaining relatively equal numbers of wetlands in the different land cover and permanence classes (Table 4.1), as well as on accessibility. Land cover classes were crop, wood, and grassland while permanence classes were seasonal, semi-permanent, and permanent. Wetland permanence classes were determined using a combination of the vegetation structure of the pothole, as per Millar (1976) and Stewart and Kantrud (1971), assessed in May and mid-August 2009, in combination with observations of the presence or absence of surface water recorded on air photos (26 October 1959 and ~31 May 2001), and SPOT 5 imagery (5 July 2007). Water samples (1 L) were collected once during 19-21 May 2009 from roughly the deepest point in the wetland, which is typically the centre of the ponds, at the midpoint in the water column.

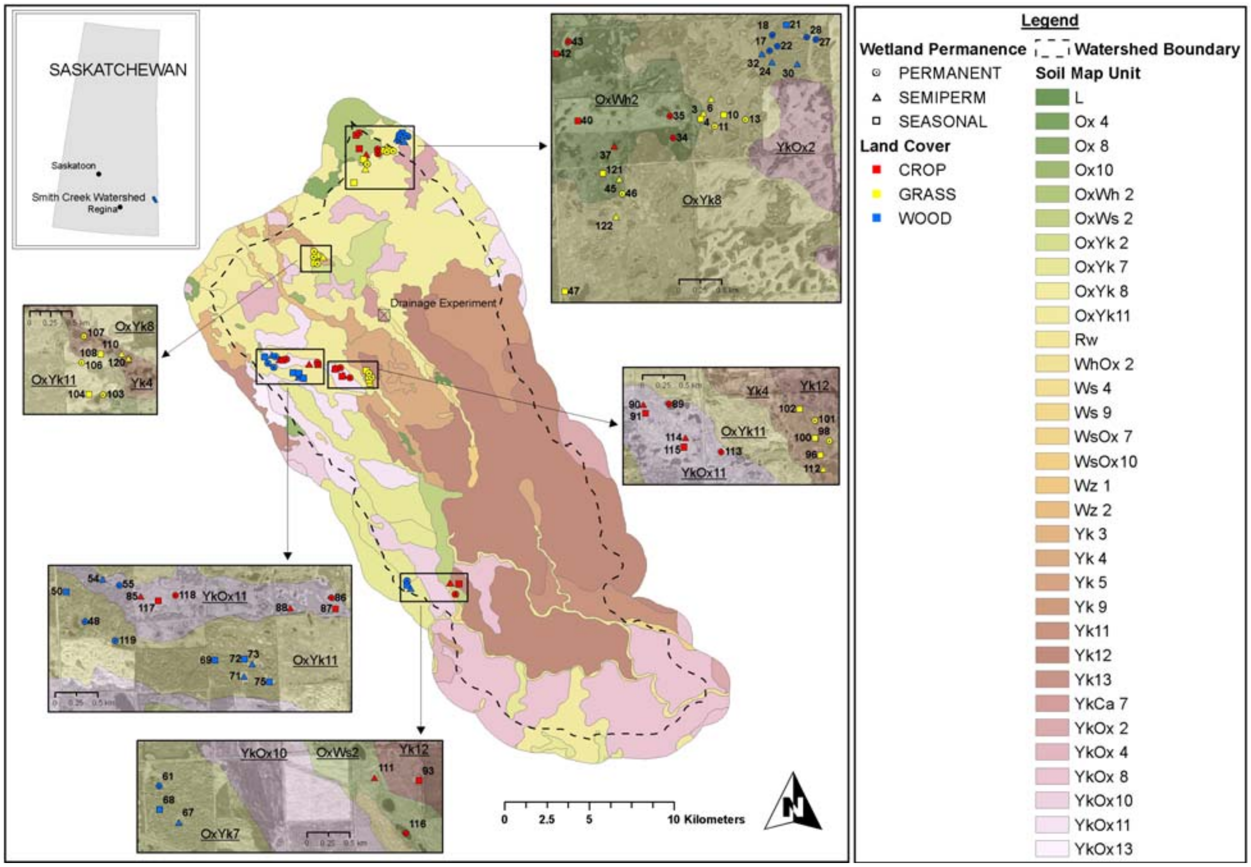


Figure 4.1: Smith Creek watershed, Saskatchewan, Canada showing the location and ID numbers of wetlands studied in relation to soil map units.

Table 4.1: Summary of number of wetlands sampled in each land cover and permanence class.

	Permanence Class			
Land Cover Type	Seasonal	Semipermanent	Permanent	Total
Crop	7	6	8	21
Grass	9	7	8	24
Wood	6	7	9	22
Total	22	20	25	67

4.2 Design of wetland drainage experiment

A drainage experiment was carried out on a permanent wetland, LR3. It was selected for study because hydrological data for it prior to drainage were available from a concurrent study (*c.f.* Minke et al., 2010). Additionally, the landowner was keen to drain it and the drain was expected to connect to Smith Creek.

Water samples were collected weekly at the center of the wetland LR3 from 18 April to 22 October 2008. Water level was measured hourly using a PT2X pressure transducer (*Northwest Instrumentation Inc*.) which was located near the edge of the wetland. Rainfall was measured nearby using a tipping bucket (*Texas Electronics Inc*., TR-525M) and a standard volumetric rain gauge. A crawler excavator and professional operator were employed to construct the artificial drainage ditch (DT6) starting 19 November 2008. Water sample collection began 20 November 2008 (1 hr) once the ditch was completed and connected to a downstream wetland. Three additional sets of samples were collected 4 hrs, 6 hrs, and 23 hrs after the start of drainage. Water samples were collected from within the wetland and at points 45 m, 70 m, 110 m, and 140 m along the ditch, measured from the wetland edge (Figure 4.2 inset) at the midpoint in the water column.

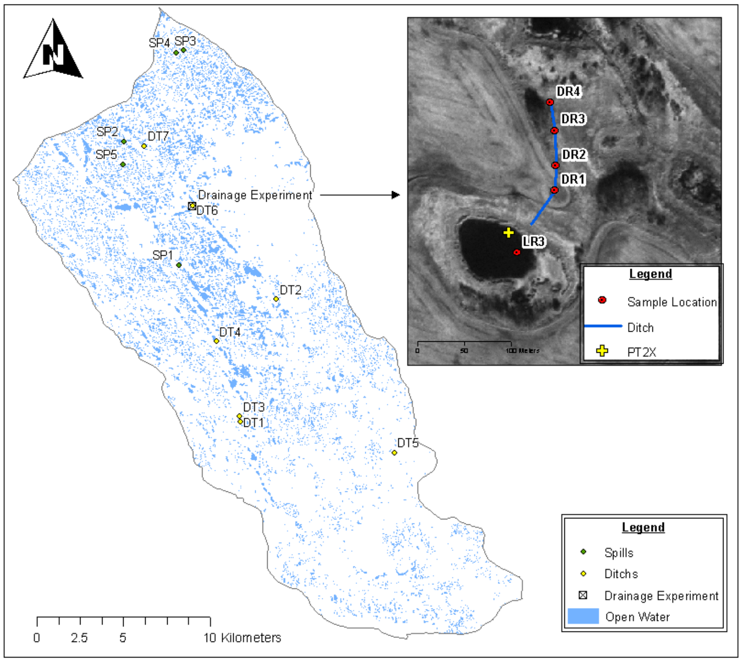
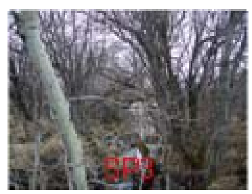


Figure 4.2: Location of the ditches (DT) and spills (SP) sampled, and the extent of open water (includes wetlands and lakes) determined by Ducks Unlimited Canada from 2005 air photos (courtesy of Lyle Boychuk). Inset: Air photo of LR3 wetland, showing the sampling sites in the wetland and along the newly constructed drainage ditch (DR). The location of the water level recorder (PT2X) is indicated by a cross.

4.3 Natural spills and artificial ditches sampling strategy

Seven artificial drainage ditches (hereafter, ditches, DT) and five natural connections (hereafter, spills, SP) (Table 4.2) were selected for study. The ditches and spills drained wetlands that did not have any surface water inflows, with the exception of DT3, which had receives inflow from DT1. However, at the time of sampling, DT1 was not flowing because it was frozen and snow covered. Due to the variation in ditch and spill length, locations for water sampling along the ditch varied; they are reported in Table 3.2. Water samples were collected 10-18 April 2009 from the thalweg of the connection, starting at the most downstream sample location. Samples were obtained from the midpoint of the water column. Manual flow gauging was carried out at the time of water quality sampling using a Marsh-McBirney Flo-mate 2000 velocity meter and wading rod. Velocity at 60% depth was measured at 20 – 40 cm intervals, so that no more than 20% of the total stream discharge was measured at each point. Ditch or spill discharge was then calculated as the sum of the product of velocity and average stream depth at each sample interval.

Table 4.2: Photographs of artificial ditches (DT) and natural connections (SP) studied that drain wetlands at Smith Creek watershed and means of physical properties measured along the connections: discharge (Q), velocity (v), depth (d), and width (w). Sample locations were measured from the wetland edge along the connection.

Sample locations (m): 0, 30, 60, 90, 120, 150 Q: 0.06 cms v: 0.04 m/s d: 0.37 m w: 3.05 m			Sample locations (m): 0, 25, 50, 75, 100 Q: 0.002 cms v: 0.009 m/s d: 0.07 m w: 2.99 m
Sample locations (m): 0, 25, 50, 75, 100 Q: 0.02 cms v: 0.02 m/s d: 0.33 cm w: 4.45 m			Sample locations (m): 0, 25, 50, 75, 100, 125 Q: 0.16 cms v: 0.11 m/s d: 0.27 m w: 5.59 m
Sample locations (m): 0, 25, 50, 75, 125, 175 Q: 0.02 cms v: 0.01m/s d: 0.22 m w: 11.0 m			Sample locations (m): 0, 25, 50, 75, 100, 125 Q: 0.06 cms v: 0.06 m/s d: 0.32 m w: 4.01 m
Sample locations (m): 0, 50, 100, 150, 200, 250 Q: 0.02 cms v: 0.05 m/s d: 0.16 m W: 3.05 m			Sample locations (m): 0, 5, 10 Q: 0.07 cms v: 0.01 m/s d: 0.28 m w: 22.3 m
Sample locations (m): 0, 15, 30, 45 Q: 0.02 cms v: 0.005 m/s d: 0.16 m W: 27.1 m			Sample locations (m): 0, 8, 15 Q: 0.006 cms v: 0.02 m/s d: 0.07 m w: 6.40 m
Sample locations (m): 0, 9, 18, 36 Q: 0.002 cms v: 0.001 m/s d: 0.18 m W: 8.49 m			Sample locations (m): 0, 10, 20 Q: 0.001 cms v: 0.001 m/s d: 0.10 m w: 4.57 m

4.4 Stream water quality sampling

Stream water samples were collected at three locations along Smith Creek in 2008, and four locations in 2009. One sample site was the watershed outlet (SC3), and the other sample sites were three sub-basins of Smith Creek (SC4, SC5, TV1; Figure 4.3) that represent areas of the watershed with differing degrees of wetland drainage. Wetland distribution across the watershed was historically (1958) even (Fang et al., 2010). Although sampling sites were chosen to reflect differing proportions of wetland loss between 1958 and 2007-2008, quantification of wetland loss was completed after site selection through the land cover classification of Fang et al. (2010). Wetland losses in the three subbasins sampled and resulting drainage classes used in this study are reported in Table 4.3. Stream water quality measurements were taken daily during peak flow (spring runoff), weekly during low flows, and following heavy summer rains (2008 only). Stream discharge at the subbasin sampling sites was estimated at the time of water quality sampling (2009 only) using a Marsh McBirney flow meter. Streamflow at the watershed outlet (Figure 4.3) was obtained from Water Survey of Canada gauge 05ME007. The water quality sampling station representing the watershed outlet was located just upstream of the town of Langenburg's drainage (SC3; Figure 4.3) as preliminary chemical analyses showed that it contributes high fecal bacteria and nutrients to Smith Creek.

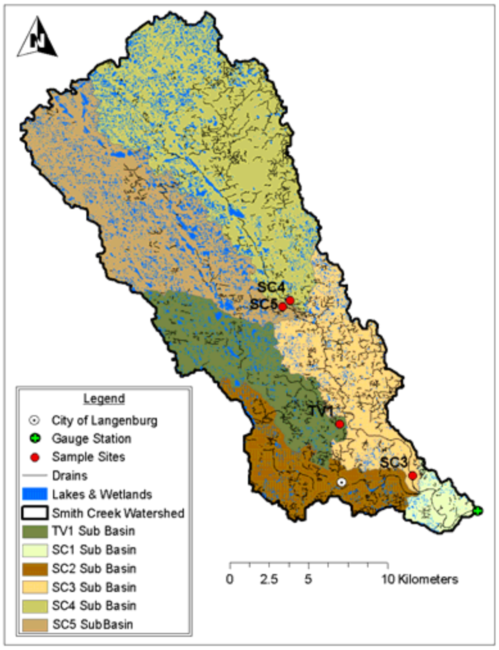


Figure 4.3: Location of stream sampling sites and the Environment Canada hydrometric gauge along Smith Creek. Sub-basins were identified by Fang et al. (2010).

Table 4.3: Wetland loss (%) and corresponding sub-basin drainage classification used in this study. The wetland loss data are derived from wetland areal extent changes between 1958 (Ducks Unlimited Canada data) and supervised land use classification data from 2007 and 2008 SPOT5 images (Fang et al., 2010).

Sample Site ID	Wetland Loss 1958-2007/8 in Sub-Basin (%)	Drainage Class
SC5	23	Low
SC4	57	Medium High
TV1	64	High

4.5 Analysis of water quality

Water samples for bacteria, i.e. *Escherichia coli* (*E.* coli) and total coliforms (T. coli), were only collected for the wetland drainage experiment and from the stream due to challenges associated with transport times to the laboratory. They were collected in 100 mL sterile bottles, preserved with $Na_2S_2O_3$, and submitted to Saskatchewan Research Council Analytical Laboratories, Saskatoon, SK (SRC) within 24 hours. They were analyzed using the chromogenic substrate method (Standard Methods part 9223).

Water samples for ion and nutrient analyses were collected in pre-rinsed 1 L polyethylene or two 350 mL glass bottles, kept on ice during the day, and then split into aliquots.

Temperature-compensated specific conductance (SC) and pH were measured in the laboratory on the day of sampling using Hach *sension*156 and Orion 3-Star hand-held meters, respectively. Two sample aliquots were preserved by the addition of HNO_3 and H_2SO_4, respectively, followed by refrigeration at 4^{o}C. A third aliquot was filtered through a 40 µm Whatman GF/C glass microfiber filter then frozen. The HNO_3 preserved sub-sample was analyzed for total phosphorus (TP) concentration at SRC within three days of sampling by inductively coupled plasma atomic emission spectroscopy (Standard Methods part 3120). Samples preserved with H_2SO_4 were only collected for the 2008 wetland drainage experiment and analyzed for total Kjeldahl nitrogen (TKN) at SRC within three days of sampling by digestion and subsequent ammonia analysis (EPA 351).

Filtered samples were analyzed for a variety of chemical parameters. Total dissolved nitrogen (TDN) and dissolved organic carbon (DOC) for samples collected in 2009 were analyzed at the University of Saskatchewan on a Shimadzu TNM-1. DOC samples from 2008 were analyzed at SRC by UV persulfate digestion and non-dispersive IR detection, (Standard Methods part 5310C). Ortho-phosphate (orthoP) was also analyzed at SRC colorimetrically (Standard Methods part 4500). A Westco SmartChem Discrete Analyzer (SmartChem 200, Method 375-100E-1) was used for analysis of ammonium (NH_4) and nitrate plus nitrite (reported as NO_3). Major ions (Cl, HCO_3, SO_4, Na, K, Mg, and Ca) were analyzed by ion chromatography with a Dionex Model ICS-2000 using potassium hydroxide and methanesulfonic acid EluGen for anion and cation analysis, respectively, at the University of Saskatchewan. Carbonate concentrations were assumed to be negligible due to high sample pH and its absence in pilot tests. One duplicate and one blank sample are analyzed for every fifteen samples collected for QA/QC. Note that charges have been left off ions throughout this report to enhance readability.

4.6 Ecosystem health sampling sites and frequency

Benthic macroinvertebrate-based biomonitoring approaches are best applied in two time windows through the course of a year – early spring shortly after ice-off, and fall shortly before ice-on (Hilsenhoff 1988). Because many Smith Creek streams cease to flow by midsummer, benthic macroinvertebrates were sampled in late April and early May in 2008 and 2009. Our 2008 study of the Smith Creek provided sampling of four sites on May 8, 2008 (Figure 4.4a; Table 4.4). However, in 2009 we sampled the stream sites on two occasions early in the spring; once on April 16, 2009 (8 sites) as the freshet commenced, and later on May 13, 2009 as water levels declined (Figure 4.4b; Table 4.4). Our first sampling event in 2009 involved collecting macroinvertebrates from eight sites comprised of four Ditch sites (DT 1, DT 2, DT 3, and DT 4) and four stream sites (SC 3, SC 4, SC 5, and TV 1). Ditch site DT 5 included in the water quality and hydrology components of this project was not included here as it was too ephemeral and observations showed that it lacked macroinvertebrates altogether. In the second sampling for 2009, however, we only repeated sampling at the four stream sites.

Table 4.4: Sites, names and coordinates included in the Smith Creek ecosystem health assessment 2008 and 2009.

Stream name	Location	SWA_Code	UTM13East	UTM13North
Smith Creek	Smith Creek North of Marchwell	SWA_2008_16	740829.6	5636736.0
Smith Creek	Smith Creek East of Langenburg	SWA_2008_17	736372.3	5638305.1
Langenburg Creek	Langenburg Creek East of Langenburg	SWA_2008_18	734626.6	5637867.9
Smith Creek	Smith Creek at Werle Farm	SWA_2008_19	732536.2	5640900.0
Smith Creek Tributary	Ditch at DT3	SWA_2009_9	723906.0	5645464.0
Smith Creek Tributary	Ditch at DT1	SWA_2009_10	723876.0	5645266.0
Smith Creek Tributary	Ditch at DT4	SWA_2009_11	723870.0	5645266.0
Smith Creek Tributary	Ditch at DT2	SWA_2009_8	725264.0	5652321.0
Smith Creek	Smith Creek at SC4	SWA_2009_42	726911.7	5648988.6
Smith Creek	Smith Creek at SC5	SWA_2009_43	727319.9	5648240.1
Thingvala Creek	Thingvala Creek at TV1	SWA_2009_6	731532.3	5641002.9
Smith Creek	Smith Creek at SC3	SWA_2009_7	736414.4	5638022.5

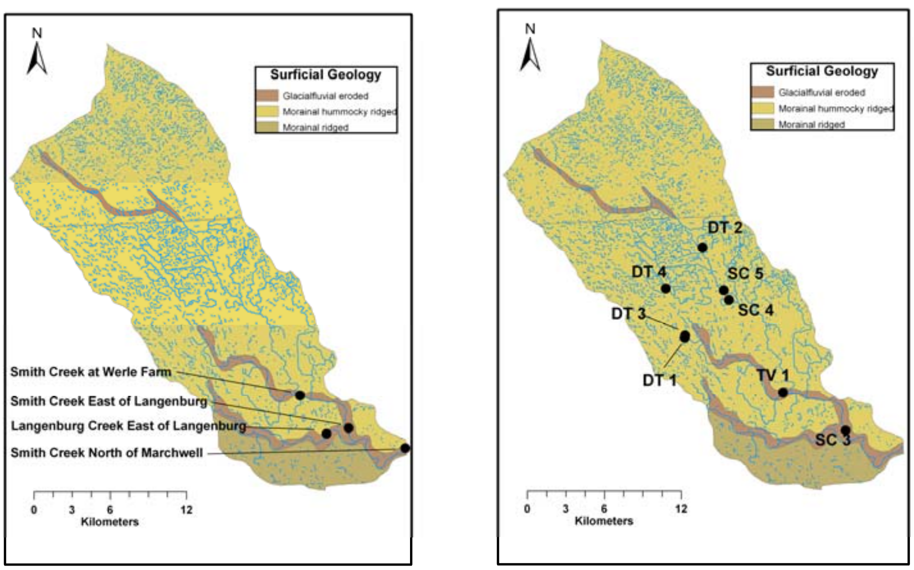


Figure 4.4: Map of Smith Creek watershed with ecosystem health assessment sampling locations from: a) 2008, and b) 2009. Note that Smith Creek at Werle Farm is TV1 and Smith Creek East of Langenburg is SC3.

4.7 Macroinvertebrate assemblage sampling methods

At each site, four qualitative 500 µm mesh D-frame net samples were collected along the length of the reach (defined as 6 times bankful width). Each sample consisted of five, 10 second, ~ 0.3 m^2, composite sweeps across the width of the stream; at the left bank, ¼ distance across, the middle, ¾ distance across, and the right bank. Samples were concentrated on a 500 µm mesh sieve and preserved in 80 % ethanol in the field. Organisms were returned to the laboratory where they were sorted from the organic material under 7 X magnification and stored in 80 % ethanol until they were identified to lowest possible taxon designation. Samples were sub-sampled where necessary. Final macroinvertebrate abundance was calculated from the subsample fraction and extrapolated to that of the original sample. Samples were required to produce a minimum of 100 specimens to be included in the data analysis.

Benthic macroinvertebrates were identified to lowest possible designation using keys for North America (Merritt and Cummins, 1996) and Western Canada (Brooks and Kelton, 1967; Dosdall and Lemkuhl, 1979; Clifford, 1991; Larson et al., 2000; and Webb, 2002). Voucher series were deposited in both the Saskatchewan Watershed Authority Invertebrate Voucher Collection (Saskatoon, Saskatchewan), and the Royal Saskatchewan Museum (Regina, Saskatchewan).

4.8 Habitat Assessment

Physical characteristics of habitat structure were collected at each site in addition to benthic macroinvertebrate sampling. Recorded were in-stream substrate composition, vegetation, algae, and detritus compositions along with notes on surrounding land use and field water quality measures at the time of sampling. These data and other more detailed GIS information are used in the test site analysis and reference condition assessment in the future.

The characteristics described in the habitat assessments are summaries and coarse descriptors of habitat condition at sites, and follow the rapid assessment protocols used by other jurisdictions in the United States (Barbour et al., 1999). Three main areas were covered in the site evaluation were substrate (% composition of material, dominant sub-dominant classification), in-stream habitat (macrophyte, algae, woody debris, and detritus use and prominence as well as characteristics of embeddedness and riparian cover), and human land use (riparian land use classification). This information will be used in discerning the condition of the sample sites when evaluating them in test site analysis in the future.

4.9 Data analyses

4.9.1 Characterization of wetland water quality

A multivariate analysis of water quality was carried out to evaluate the influence of wetland permanence and surrounding land cover type on resulting wetland water quality. Data were log transformed to correct for nonnormality and heteroscedasticity (Legendre and Legendre, 1998). Following preliminary data analysis, the water quality data set was divided into nutrient (TP, orthoP, TDN, NO_3, NH_4, DOC, and K) and salinity (SC, pH, Cl, HCO_3, SO_4, Na, Mg, K, and Ca) variable sets because the mechanisms controlling nutrient concentrations in the wetlands have been shown to be different from mechanisms controlling ionic concentrations (Labaugh, 1987). Mass per volume concentrations (i.e., mg/L) were converted to equivalence concentrations (i.e., meq/L) for direct comparison of major ions.

A two-way multivariate analysis of variance (MANOVA) was used to test the relationships among land cover and permanence for the water quality parameters measured. Post hoc comparison tests (two-way ANOVAs) were used to determine which variables contributed to the occurrences of significant differences between factors. Significant differences among land cover and permanence classes were assessed using Tukey's Honest Significant Difference (HSD) pairwise comparisons test, which is applicable to mildly unbalanced designs (Everitt and Hothorn, 2006). Statistical analyses were conducted using the R statistical language and environment (R Core Development Team, 2005). A type I error rate of 0.05 was used in significance tests unless otherwise stated.

4.9.2 Wetland drainage experiment

Wetland volumes were estimated using the full volume-area-depth method (Hayashi and van der Kamp, 2000) in combination with PT2X water level data. Coefficients required for the estimates were obtained by Minke et al. (2010) from a digital elevation model derived from total station survey data. The volume of ice water in the wetland at the time of drainage was estimated using the density of ice (920 kg/m^3) and the average ice thickness (Andres and Van Der Vinne, 2001). Constituent mass in the wetland was estimated using the concentration and the wetland volume estimate at the time of sampling. Changes in solute concentrations relative to chloride, which is impacted by evapotranspiration and dilution in the wetland but is biotically conservative, were used to indicate biotic or geochemical processing of constituents in the wetland (Duff et al., 2009).

The average concentration along the ditch multiplied by the change in estimated wetland volume at each sampling point was used to calculate total loads exported from the wetland via the newly constructed drainage ditch. Loads were normalized along the length of the new ditch by dividing the load at each sample point along the ditch by the load at the first sampling point (DR1) in the ditch. A value of one was then subtracted to set the slope intercept to zero. The statistical significance of the linear relationship with normalized load and distance along ditch was tested using the linear model (l m) function of the R statistical language and environment (R Core Development Team, 2005). The slopes of the normalized concentrations were statistically compared as per Zar (2008) to the slope of biotically conservative Cl, at each sampling instance to assess whether constituents were added or removed along the length of the ditch. Significantly different slopes indicate nutrients or salts are either abiotically or biotically removed (or added) as water travels along the ditch length.

4.9.3 Comparison of ditches and spills

Statistically significant differences in water quality constituents (concentrations and loads) and physical properties between ditches and spills were assessed with *t*-tests computed using SPSS (version 14.0). Data used were log transformed, means measured along each ditch or spill. Differences between loads at the connection inlet and outlet were also compared using *t*-tests to assess whether transformations of water quality constituents occurred along the length of the connection. A type I error rate of 0.05 was used unless otherwise stated.

4.9.4 Stream water quality

To account for differences in sub-basin size and water flows, export coefficients (E_{ij}) for nutrients and salts were calculated by

$$E_{ij} = \frac{C_{ij} Q_{ij}}{A_i} \qquad (1)$$

where C and Q are concentration and discharge measured on day j, and A is the area of subbasin i. This permitted direct statistical comparison of findings among subbasins characterized with differing wetland drainage. E_{ij} values were compared during the 2009 snowmelt period among subbasins using a one-way ANOVA and Sidak post-hoc comparisons in SPSS v.14.0. Additionally, total water and solute exports for spring melt in 2009 were computed by summing daily values. Solute data from 2008 are not reported because of lack of discharge measurements at all but site SC3 due to equipment failure. However, trends in water quality concentrations in 2008 were similar to those reported for 2009.

4.9.5 Macroinvertebrate analysis

For a complete description of the construction of SWA's RCA and Test Site Analysis (TSA) based aquatic ecosystem health tool please refer to the Aquatic Health Feasibility study (Phillips and M^c^Master, 2010). A summary of tool construction is provided below.

Reference sites included in this study are a subset of 100 sites collected in 2006 and 2007 as part of the construction of SWA's aquatic health feasibility study (Phillips and McMaster, 2010). Reference sites were separated from a larger set of sites based on the amount of human land-use in a buffer area 10 km long and 1 km wide on either side of the stream, upstream of the site. Because of the predominance of agricultural development in southern Saskatchewan watersheds, reference condition is not identified as pristine land-cover with the absence of human activity, but more practically at that condition available that has the 'least' amount of human activity (Stoddard et al., 2006). Specifically, reference site candidacy was restricted to the following criteria: < 50% cropland, < 5% urban land-use, < 50% pasture, < 80% total land under human influence (combination of cropland, urban, and pasture), ≤ 2 landfills, oil wells, bridges or road crossings upstream, and no reservoirs within 10 km. Groups of reference sites with similar community compositions were identified using cluster analysis, and used to define what underlying physiochemical conditions structure macroinvertebrate communities in southern Saskatchewan. In these analyses, it was found that surficial geology composition in the 10 km buffer upstream of the site, and the site ecoregion membership best explained the composition of the macroinvertebrate community. From this step, test sites (potentially disturbed sites) were matched to the appropriate reference group based on similarities in surficial geology and ecoregion. The test site was then compared with the reference grouping for which it had the highest probability of membership, determined using discriminate functions analysis. All Smith Creek Test Sites included in this study were identified as belonging to the Biological Grouping 2, and thus further analyses comparing the test sites to reference sites were conducted against the pool of Biological Grouping 2 reference sites (*c.f.* Phillips and McMaster, 2010).

Next, attributes of the benthic macroinvertebrate assemblage were selected for comparison between test and reference sites using those adopted as part of the aquatic health feasibility study (Phillips and McMaster, 2010) and 2010 SWA State of the Watershed Reporting (SWA 2010). Specifically, the biological endpoints, or metrics, of abundance, taxa richness, Shannon's Diversity, and community structure were summarized using Correspondence Analysis (CA) ordination axes. The measure of taxa richness used was produced as Margalef's species richness for each sample, and was a measure of the number of species making some allowance for the number of individuals. Both taxa richness and Shannon's Diversity were calculated using PRIMER version 6 (Plymouth Marine Labs, Plymouth, UK; Clarke and Warwick, 2001). We conducted our CAs using the Biplot add-in for Excel 2009. Separate CA analyses were conducted for each Smith Creek Test Site with the combination of Reference Sites. The resulting axis 1 score was used in further test site analysis (Michelle Bowman, personal communication).

Finally, TSA was used to formally evaluate the magnitude of difference between each test site and the appropriate reference sites (e.g., Bowman and Somers, 2006) based on the metrics mentioned above. TSA based on individual metrics is equivalent to evaluating whether a metric value is outside the normal range of variation in the reference sites. A non-central alpha value (*ncP*) > 0.95 indicates a site is significantly within reference condition and healthy, a value between 0.95 and 0.05 indicates the site is outside the normal range and is stressed, while a *ncP* value ≤ 0.05 indicates that the site is significantly outside the normal range of reference variation and is thus impaired.

5 Results

5.1 Chemical characterization of wetlands

Nutrient concentrations in the wetlands studied ranged widely (Appendix A; Table 5.1). TP ranged from 0.02 to 2.8 mg/L. Most wetlands studied were classified as eutrophic, with the exception of six which were hypereutrophic (i.e. TP > 0.6 mg/L), based on trophic classification presented in Wetzel (2001). The majority of P was typically in the organic form, with the exception of six cropped wetlands and one wood wetland that were characterized by greater proportions of orthoP. TDN in the wetlands ranged from 0.8 to 2.8 mg/L. N was predominantly present in the organic form with DON making up 96 % on average of TDN. Comparing DIN:DIP to the Redfield Ratio allows for determination of the limiting nutrient on algae growth (Rhee and Gotham, 1980; Wetzel, 2001). Eight wetlands were P limited (DIN:DIP>12:1), 46 were N limited (DIN:DIP<7), and 13 were limited by neither N nor P (i.e., DIN:DIN 7-12). DOC ranged from 19 to 55 mg/L.

Salt concentrations in the wetlands studied were also greatly varied (Appendix B; Table 5.1). The wetlands were neutral to slightly basic (pH of 6.6-8.6) and ranged from fresh to brackish (SC varied from 57 to 1780 µS/cm). Maximum concentrations of major ions were HCO_3 =3.8 meq/L, SO_4=15.5 meq/L, Mg =13.7 meq/L, and Ca =6.8 meq/L. Distinct patterns of ion dominance groups were apparent (Figure 5.1). With the exception of one wetland (W3), all 29 wetlands with SC > 413 µS/cm were characterized by SO_4>HCO_3>Cl and Mg>Ca>K>Na. All but two of these wetlands were classed as crop or grassland and 41% and 34% were classed as permanent and semi-permanent, respectively. The other anion dominance pattern observed was HCO_3> SO_4>Cl, and 50% of these wetlands were also characterized by Mg>Ca>K>Na. The second most common (21%) cation dominance pattern for these wetlands was Ca>Mg>K>Na.

The two-way MANOVA for the nutrient variable set indicated significant differences existed among land cover types ($p = 0.009$) and permanence classes ($p = 7x10^{-5}$). However, there was no significant interaction between land cover type and wetland permanence class ($p = 0.23$). For the salinity variable set, there was a significant difference among land cover types ($p = 5x10^{-10}$). However, differences among permanence classes ($p = 0.11$) and the interaction between land cover type and wetland permanence class ($p = 0.46$) were not significant.

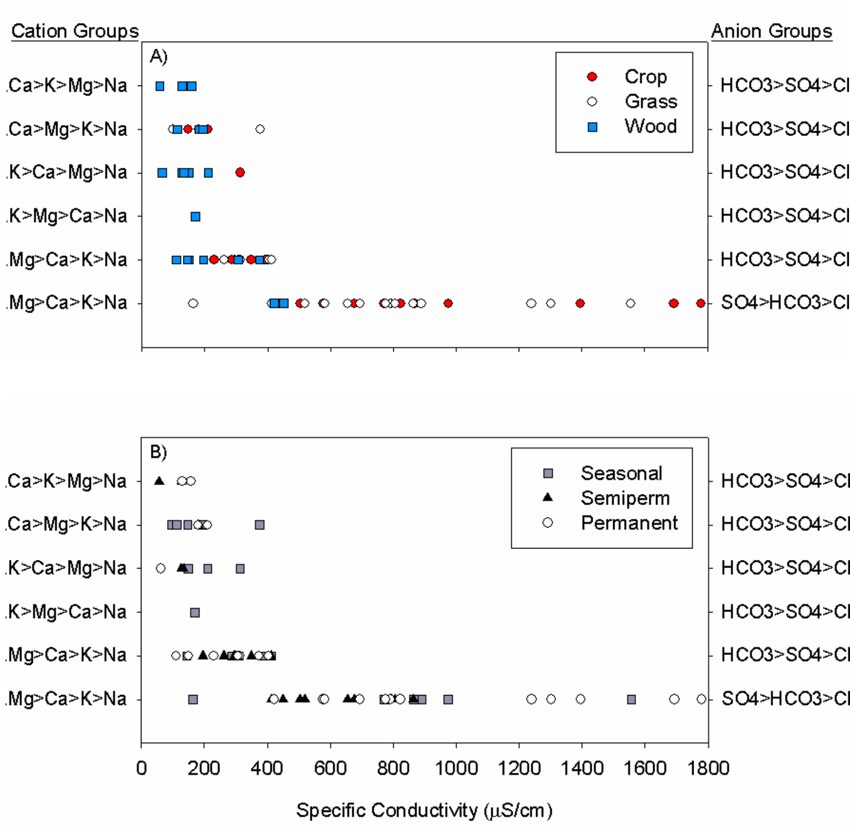


Figure 5.1: Distribution of cation and anion dominance groups as a function of specific conductivity for the 67 wetlands studied grouped by A) land cover and B) permanence classes. Semiperm is semipermanent wetlands.

Subsequent pairwise comparison tests elucidated differences among land cover types and permanence classes for the nutrient variable set and land cover types for the salinity variable set (Table 5.1). There were significant differences among land cover types for TP and K, i.e., crop wetlands had greater TP and K than wood or grass wetlands. Significant differences among permanence classes for TP, TDN, and DOC were also found. Permanent wetlands had lower TP than seasonal and semi-permanent wetlands. In addition, TDN and DOC were higher in seasonal wetlands compared to semi-permanent and permanent wetlands. SC, Cl, HCO_3, SO_4, Na, Mg, and Ca concentrations were significantly lower in the wood wetlands compared to the crop and grass wetlands.

Table 5.1: Mean (standard error) of wetlands within each permanence and land cover class, as well as a summary of results for two-way ANOVAs. Semiperm is semipermanent wetlands. Differing letter superscripts indicate significantly different ($\alpha = 0.05$) Tukey's pairwise comparisons; * and ** denote a statistically significant difference at $\alpha = 0.05$ and $\alpha = 0.01$, respectively.

Variable	Unit	Permanence					Land Cover				
		Permanence Class			Two-way ANOVA		Land Cover Class			Two-way ANOVA	
		Seasonal	Semiperm	Permanent	F	*p*	Crop	Grass	Wood	F	*p*
pH		7.18^{a}	7.21^{a,b}	7.52^{b}	4.26	0.0188*	7.33^{c}	7.42^{c}	7.18^{d}	1.96	0.1507
		(0.09)	(0.09)	(0.10)			(0.08)	(0.09)	(0.12)		
SC	uS/cm	446	417	601	1.68	0.1952	633^{c}	649^{c}	194^{d}	22.86	0.0000**
		(81)	(56)	(102)			(104)	(75)	(23)		
Cl	meq/L	7.4	5.0	4.5	0.67	0.5158	6.7^{c}	8.1^{c}	1.8^{d}	19.77	0.0000**
		(2.0)	(1.1)	(0.7)			(0.8)	(1.9)	(0.2)		
HCO_3	meq/L	101.3	96.0	104.6	0.17	0.8423	115.4^{c}	114.5^{c}	72.3^{d}	10.22	0.0002**
		(9.0)	(7.4)	(9.5)			(9.0)	(7.7)	(6.5)		
SO_4	meq/L	99.1	93.6	156.2	1.56	0.2187	155.5^{c}	180.9^{c}	15.9^{d}	22.07	0.0000**
		(37.9)	(20.8)	(37.5)			(39.0)	(36.0)	(6.3)		
Na	meq/L	14.3	11.5	21.4	2.15	0.1260	13.2^{c}	32.0^{c}	1.6^{d}	31.60	0.0000**
		(5.8)	(3.6)	(6.1)			(3.8)	(7.0)	(0.5)		
Ca	meq/L	28.5	23.4	25.9	0.08	0.9206	33.2^{c}	31.9^{c}	12.7^{d}	13.15	0.0000**
		(6.5)	(3.1)	(3.7)			(4.7)	(5.4)	(1.2)		
Mg	meq/L	28.3	26.7	47.3	2.54	0.0880	44.1^{c}	50.2^{c}	9.4^{d}	20.72	0.0000**
		(8.1)	(4.9)	(9.8)			(10.1)	(8.1)	(2.0)		
K	meq/L	21.7	19.9	16.7	1.44	0.2464	24.8^{c}	16.7^{d}	16.9^{d}	4.17	0.0203*
		(2.1)	(2.7)	(1.1)			(2.7)	(1.2)	(1.4)		
TP	mg/L	0.36^{a}	0.34^{a}	0.11^{b}	8.73	0.0005**	0.46^{c}	0.18^{d}	0.15^{d}	4.68	0.0131*
		(0.08)	(0.14)	(0.02)			(0.14)	(0.04)	(0.03)		
orthoP	mg/L	0.06	0.09	0.04	0.20	0.8214	0.10	0.02	0.07	1.98	0.1479
		(0.02)	(0.03)	(0.01)			(0.03)	(0.01)	(0.03)		
TDN	mg/L	1.58^{a}	1.21^{b}	1.12^{b}	11.43	0.0001**	1.40	1.23	1.27	2.62	0.0817
		(0.10)	(0.07)	(0.05)			(0.08)	(0.10)	(0.06)		
NO_3	mg/L	0.02	0.04	0.02	0.16	0.8513	0.03	0.02	0.02	0.54	0.5853
		(0.00)	(0.01)	(0.01)			(0.01)	(0.01)	(0.01)		
NH_4	mg/L	0.026	0.024	0.030	1.02	0.3661	0.026	0.027	0.026	0.56	0.5720
		(0.003)	(0.003)	(0.003)			(0.004)	(0.003)	(0.026)		
DOC	mg/L	38.0^{a}	31.5^{b}	28.8^{b}	8.96	0.0004**	34.0	32.3	31.7	0.39	0.6822
		(2.0)	(1.2)	(1.4)			(2.1)	(1.6)	(1.6)		
DIN:DIP	--	4.3	6.8	5.6	0.30	0.7422	4.3	6.7	5.3	1.75	0.1833
		(1.0)	(2.4)	(1.3)			(1.2)	(2.0)	(1.4)		

5.2 Wetland drainage experiment

The water stored in wetland LR3 increased following snowmelt from 776 m^3 on October 23, 2007 to 2703 m^3 on April 18, 2008. Water volume tended to decrease during rain free periods and increase following rain events throughout the 2008 open water season (Figure 5.2). Following the snowmelt period, water volume decreased ~60% throughout spring and early summer, reaching a minimum on July 7. Frequent and large rain events occurred between July 7 and August 14, which caused water volume in LR3 to increase above that in spring and remain high until the drainage experiment in November. Based on daily wetland water level fluctuations (data not shown) and precipitation data, ~6 mm/day of water was estimated to be lost from the wetland between May 1 and October 22, 2008.

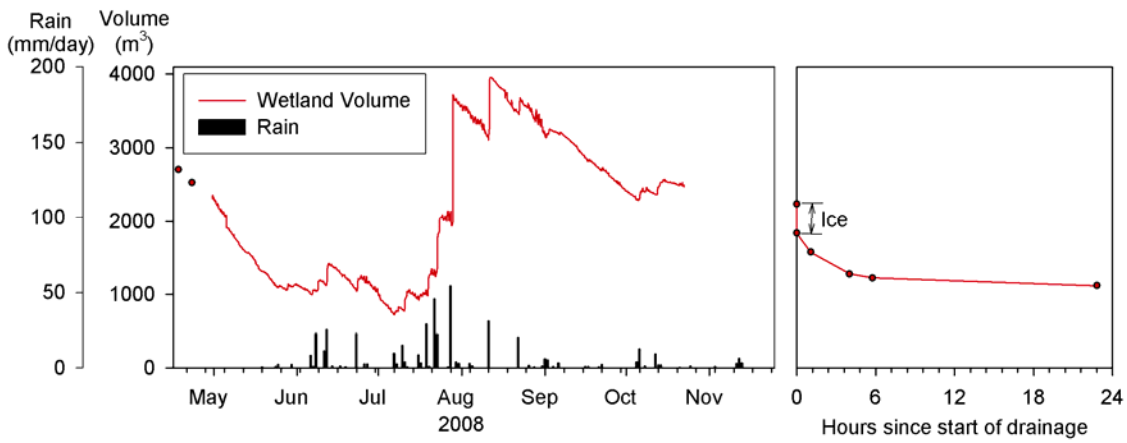


Figure 5.2: Daily rainfall and volume in wetland LR3 prior to and during drainage.

Concentration (Figure 5.3) and mass (Figure 5.4) of forms of nitrogen and phosphorus were highly seasonal variable. TP concentrations in LR3 ranged from 0.22 mg/L to below analytical detection limits (i.e., 0.01 mg/L). TP mass and concentration were both elevated April 18 and decreased sharply to lows on May 8. During the relatively rain free spring and early summer, TP concentration increased, peaking July 10. TP mass also increased during the relatively rain free period, however, it continued to increase during the July and August rain events whereas TP concentration declined in late July and increased in early August. TP concentrations and mass stabilized throughout much of August and September and began to decline in late September. TP mass and concentration spiked October 15 following a 15.8 mm rain event on October 13-14. OrthoP concentrations in the wetland ranged from 0.15 mg/L to below analytical detection limits (i.e., 0.01 mg/L). Seasonal variations in orthoP concentration and mass were similar to those for TP. TP ($p = 0.42$) and orthoP ($p = 0.29$) were not significantly correlated with Cl. The normalized mass data (Figure 5.5) show that TP and orthoP were both added to the wetland relative to Cl during the growing season.

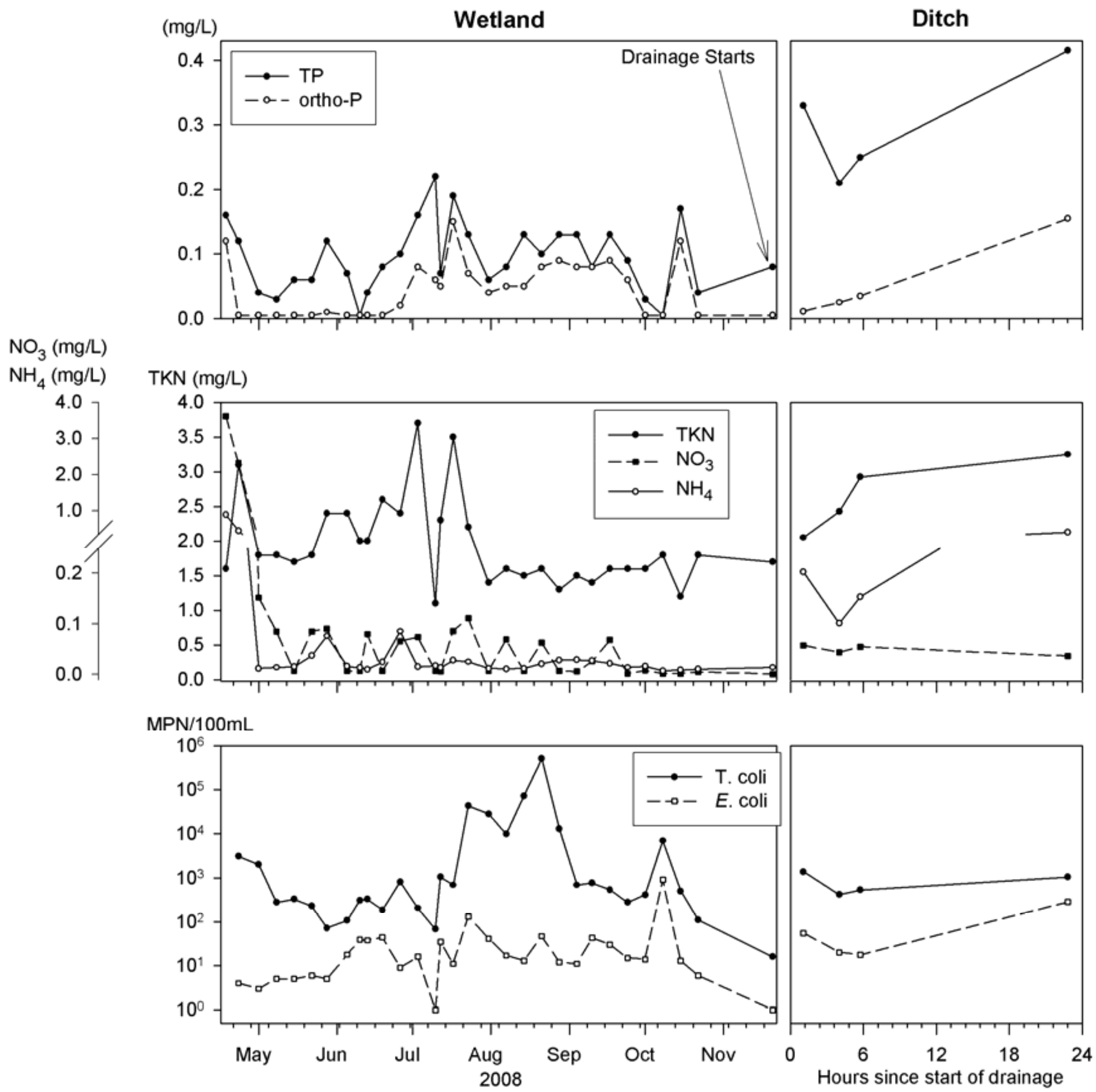


Figure 5.3: Concentration of nitrogen, phosphorus, and bacteria measured in the LR3 wetland prior to the drainage experiment and in the newly constructed ditch.

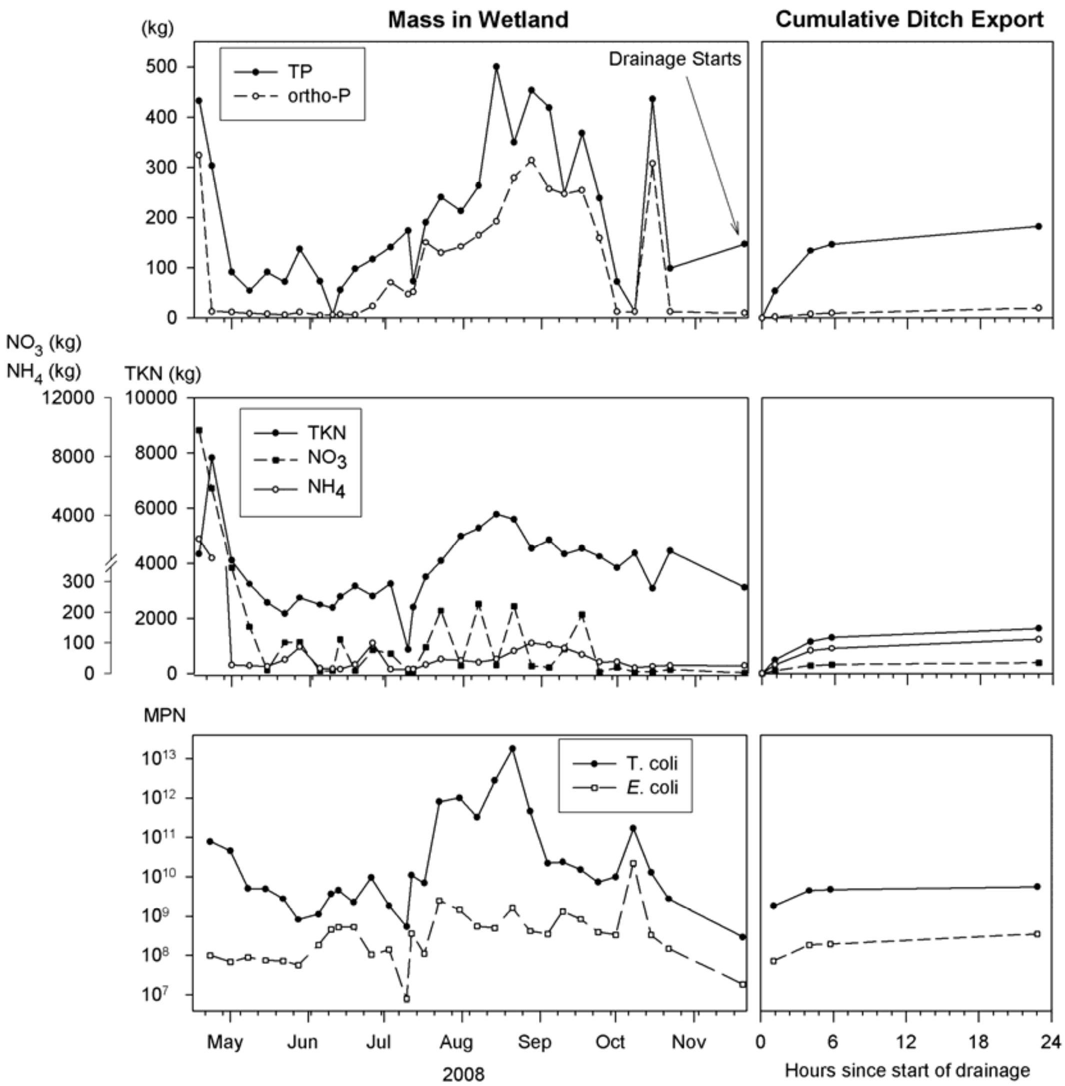


Figure 5.4: Total mass of nitrogen, phosphorus, and bacteria measured in the LR3 wetland prior to the drainage experiment and the cumulative mass exported via the newly constructed ditch.

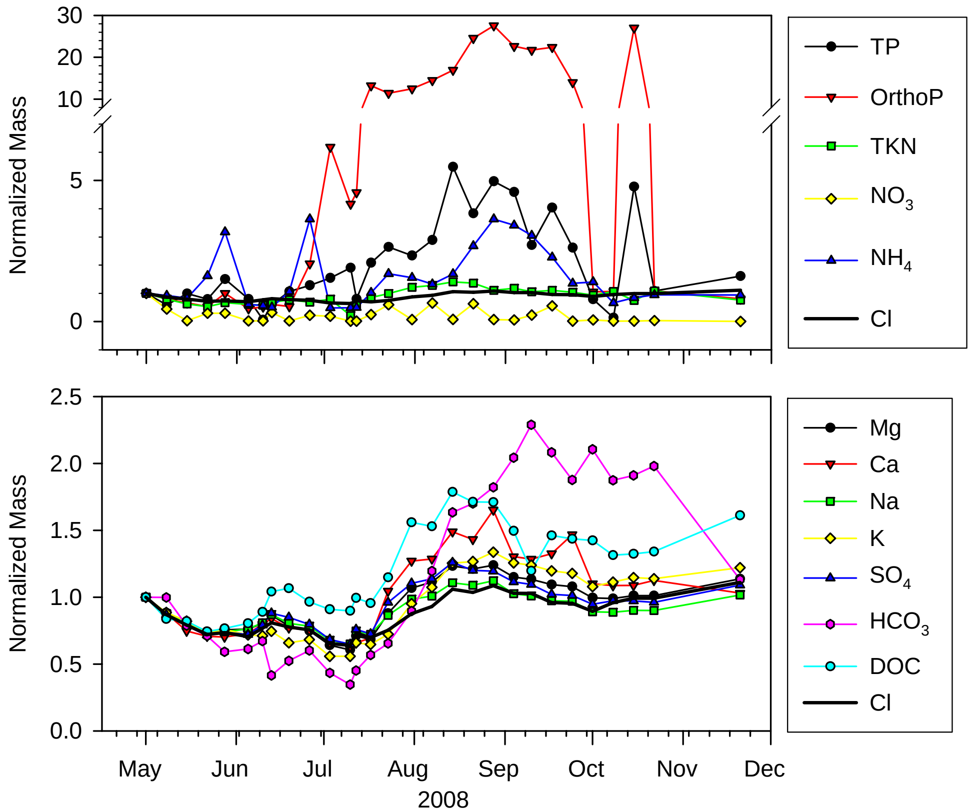


Figure 5.5: Mass of various water quality parameters, normalized to May 1, 2008, measured in wetland LR3.

TKN concentrations in the wetland ranged from 3.7 mg/L to 1.1 mg/L. NO_3 and NH_4 concentrations ranged from below analytical detection limits (i.e., 0.01 mg/L) to 3.6 mg/L and 0.9 mg/L, respectively. Average concentrations of TKN, NO_3, and NH_4 were respectively 1.9 mg/L, 0.23 mg/L, and 0.06 mg/L. TKN, NO_3, and NH_4 concentrations were elevated following snowmelt and masses decreased to lows May 22, May 15 and May 1, respectively. During the relatively rain free period of April 18 to June 6, TKN concentration increased, peaking July 3. NH_4 and NO_3 concentrations were quite variable throughout the study period. At the beginning of the rain events (July 10) TKN and NO_3 decreased substantially while NH_4 increased slightly. Peaks in concentrations were reached July 17 (TKN and NH4) and July 23 (NO_3). Minimum NH_4 and NO_3 concentrations occurred October 8 and 22, respectively. Mass of TKN and NH_4 increased steadily from July 10 until August 21 and August 28, and then tended to decrease until the start of the drainage experiment. NO_3 mass was highly variable during the same time period, but remained constant and low after September 24. Comparing mass ratios of dissolved inorganic nitrogen ($NO_3 + NH_4$) to dissolved inorganic phosphorus (PO_4) (i.e. DIN:DIP) to the Redfield Ratio indicates that the limiting nutrient in the wetland was P in spring (Figure 5.6; Wetzel, 2001). After June 5, the limiting nutrient in the wetland was predominantly N for the remainder of the study. TKN was positively correlated with Cl ($r = 0.56$, $p = 1x10^{-3}$), however, NH_4 ($p = 0.72$) and NO_3 ($p = 0.64$) were not correlated with Cl. The normalized mass data (Figure 5.5) show that over the course of the study period, NH_4 was added to the wetland, NO_3 was removed, and TKN was neither greatly added nor removed relative to Cl.

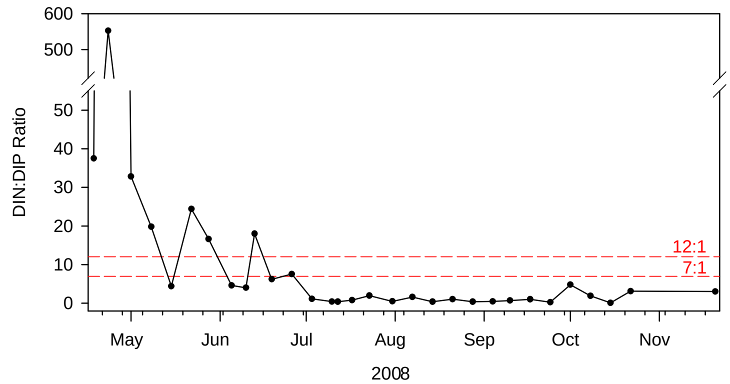


Figure 5.6: Mass ratio of DIN:DIP (NO_3+NH_4:PO_4) measured in the LR3 wetland prior to its drainage. Values below seven indicate N limiting conditions and values above 12 indicate P limiting conditions (Wetzel, 2001).

Variations in density (Figure 5.3) and loads (Figure 5.4) of both T. coli and *E*. coli were very similar throughout 2008, and as such only trends in loads are summarized. Following the snowmelt period, loads of T. coli and *E*. coli in LR3 decreased until May 28, then increased until June 26 and June 19, respectively. Minimums were reached July 10. Loads of T. coli and *E*. coli then increased during the mid summer rain events. T. coli and *E*. coli loads then generally decreased until the start of the drainage experiment. A secondary peak in *E*. coli and T. coli occurred October 8. Neither T. coli ($p = 0.12$) nor *E*. coli ($p = 0.50$) were significantly correlated with Cl.

Seasonal variations in major ion and DOC concentrations were similar throughout summer 2008, with the exception of HCO_3 (Figure 5.7). DOC ($r = 0.88$, $p = 3x10^{-11}$) and all major ions ($r = 0.93$-0.98, $p = 3x10^{-25}$-$7x10^{-14}$), with the exception of HCO_3 ($p = 0.32$), were significantly correlated with Cl. Following snowmelt and during the relatively rain free spring and early summer, ion and DOC concentrations generally increased, peaking around July 10. Ion concentrations then reached minimums on July 30 during rain events and then increased until the start of the drainage experiment. In contrast, HCO_3 concentration did not noticeably increase between the snowmelt and the end of July. Steady increases in HCO_3 concentrations were observed between August and November. Trends in major ion and DOC mass (Figure 5.8) differed from trends in concentration. DOC and major ion mass decreased in the wetland during the relatively rain free spring and early summer until July 10 then increased on average by a factor of 2.4 and peaked ~August 28. Major ion and DOC mass mostly decreased in September and increased in October. The ion dominance pattern in the LR3 wetland was SO_4>HCO_3>Cl and Mg>Ca>Na>K, which remained constant throughout the study period in 2008. Seasonal variation in pH (Figure 5.7) was similar to that observed for concentrations of major ions. However, pH peaked at 9.5 on July 3 and reached a minimum value of 7.4 August 14. The normalized mass data (Figure 5.5) show that DOC, HCO_3, and Ca were added to the wetland relative to Cl.

At the time of the drainage experiment, the LR3 wetland was covered by ~8 cm of ice. As a result, only 81% of the water was in liquid form. The wetland volume decreased rapidly when the drainage ditch was completed (Figure 5.2). After 4 hrs of drainage, 30% of the water (total) had exited the wetland via the ditch. Concentrations of N, P, bacteria, and major ions were greater in the drainage ditch than those measured in the wetland at the start of drainage (Figures 5.3 and 5.4). DOC concentrations in the ditch were less than that measured in the wetland at the start of drainage, not including the final sampling instance, 23 hr after the start of drainage. The pH was consistently lower in the newly constructed drainage ditch than in the wetland at the start of drainage (Figure 5.7).

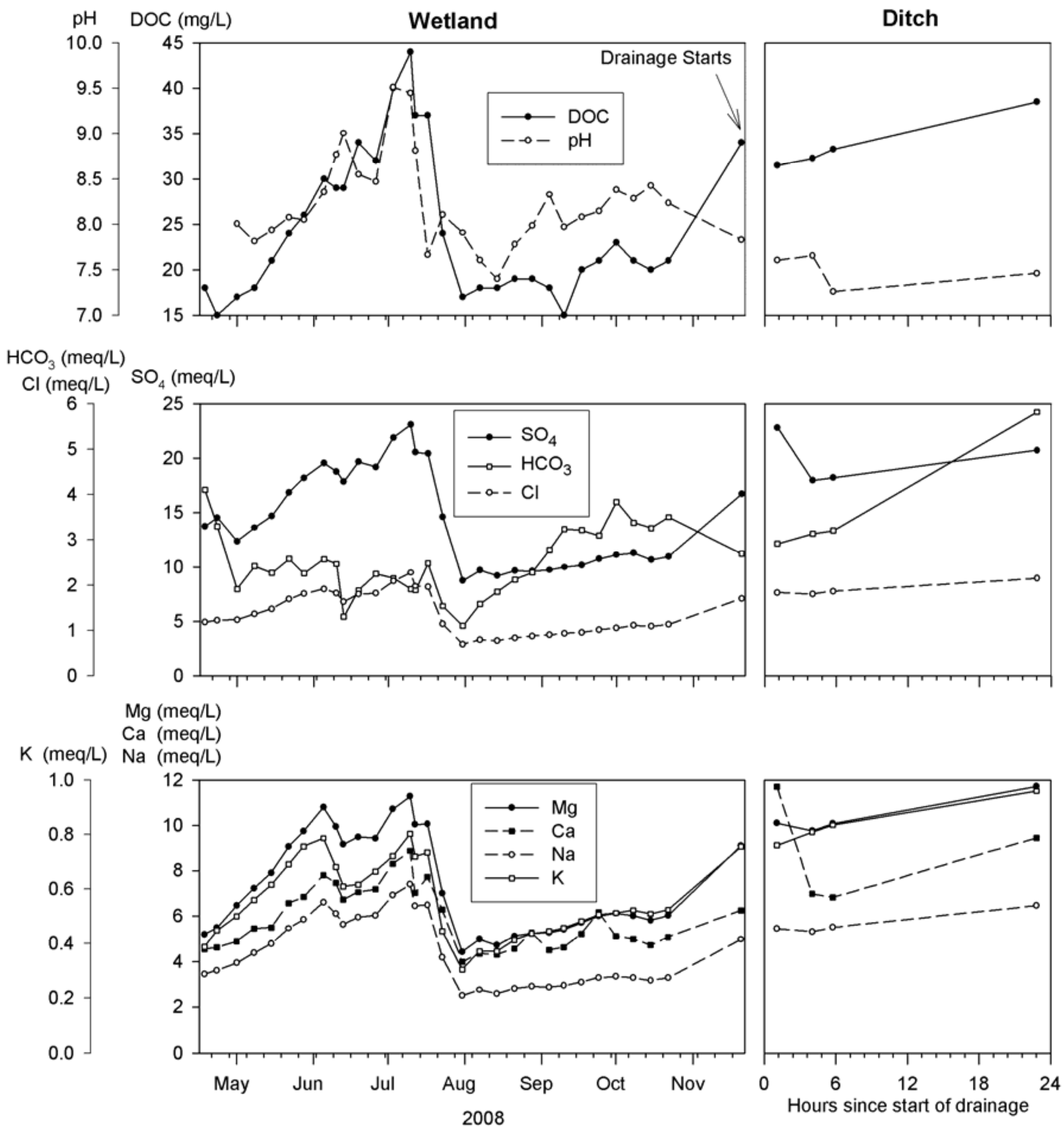


Figure 5.7: pH and concentration of DOC and major ions measured in the LR3 wetland prior to the drainage experiment and in the newly constructed ditch.

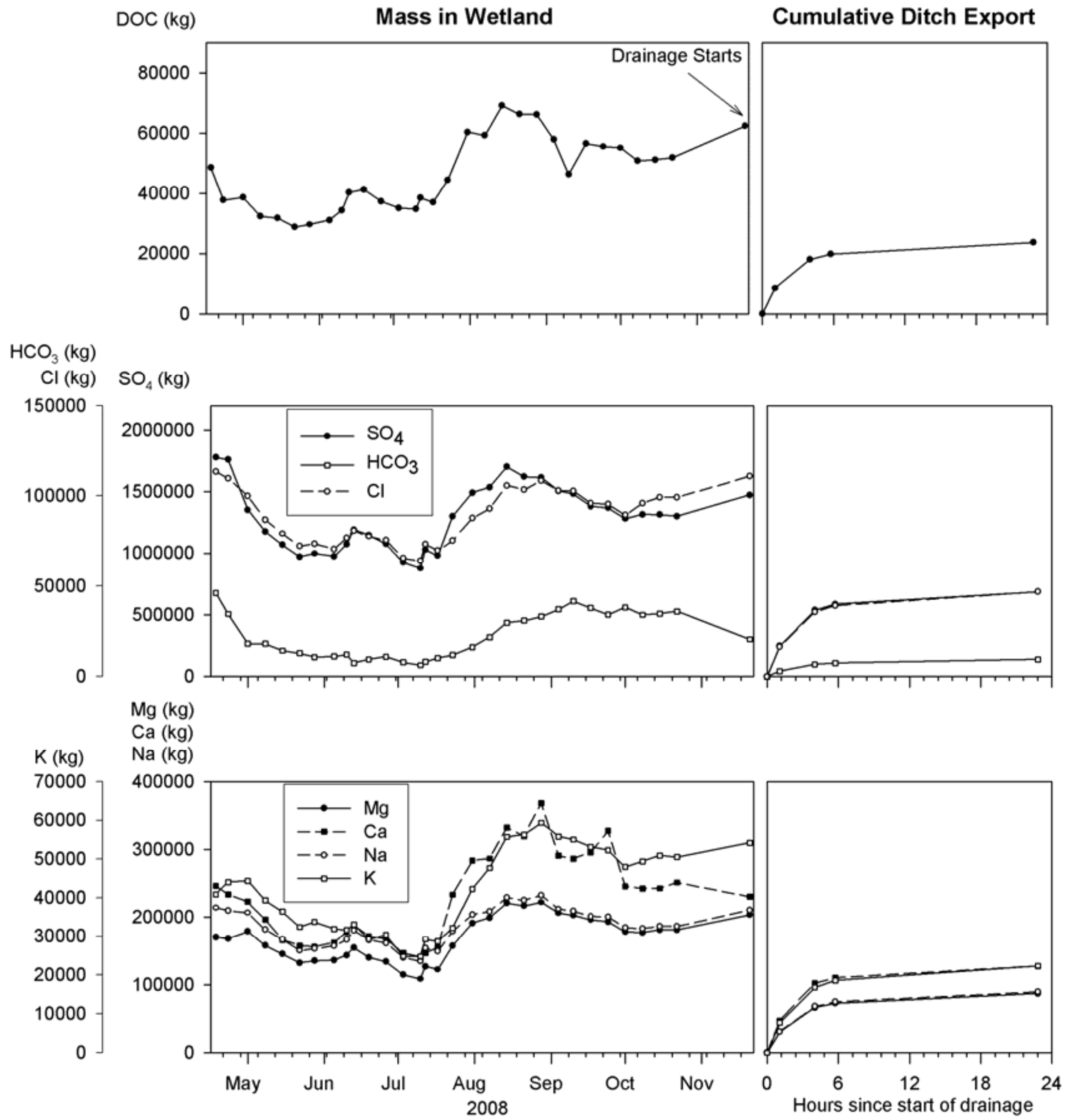


Figure 5.8: Total mass of DOC and major ions measured in wetland LR3 prior to the drainage experiment and the cumulative mass exported via the newly constructed ditch.

Over the course of the drainage experiment, concentrations of N, P, and DOC generally increased in the drainage ditch. However, concentrations of TP, NO_3, and NH_4 were greater 1 hr after the start of drainage compared to concentrations measured 4 and 6 hr after the start of drainage. Total mass exported from LR3 for each constituent during the experiment is summarized in Figures 5.4 and 5.8. The total masses of TP, orthoP, NO_3, and NH_4, and the total number of *E*. coli and T. coli colonies exported via the drainage ditch exceeded those estimated in the wetland at the start of the drainage experiment by a factor of 1.2, 2.1, 19.0, 4.3, 19.4, and 18.9, respectively. Total masses of TKN, DOC, and major ions exported via the drainage ditch were less than those estimated in the wetland at the start of the drainage experiment by factors ranging from 0.4 to 0.6.

Results for solutes that were significantly correlated ($\alpha = 0.05$) with distance along the drainage ditch and that also had at least marginally significantly different ($\alpha = 0.10$) slopes than

Cl are shown in Figure 5.9. Slopes that differ from Cl suggest that the nutrient or ion experienced biotic processing, sorption or release along the ditch length. Slopes of DOC (1 hr), orthoP (1 hr, 4 hr, and 23 hr), T. coli (4 hr), HCO_3 (6 hr), and NH_4 (6 hr and 23 hr) were less steep than the Cl slope. Slopes of HCO_3 (4 hr), T. coli (23 hr), and NO_3 (23 hr) were steeper than the Cl slope. Normalized Cl concentrations were not significantly ($\alpha = 0.05$) correlated with distance 4 hr and 6 hr after the start of drainage.

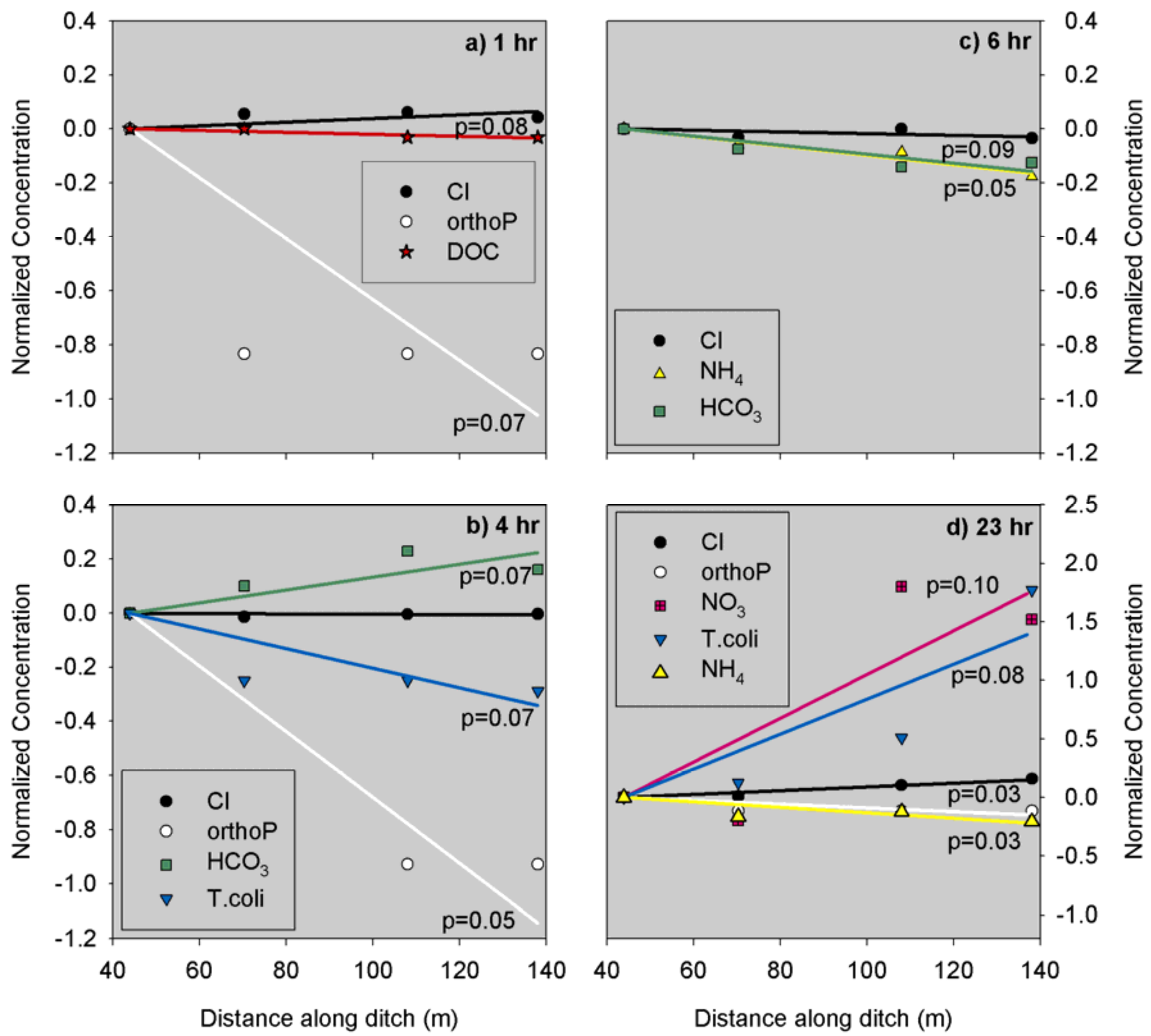


Figure 5.9: Slopes of normalized concentrations measured along the newly constructed ditch draining the LR3 wetland at: a) 1 hr, b) 4 hr, c) 6 hr, and d) 23 hr after the start of the drainage experiment. Concentrations were normalized by dividing the concentration at each sample point along the ditch by the concentration at the first sampling point (DR1) in the ditch. A value of one was then subtracted to set the intercept to zero. Only constituent slopes that were at least marginally different ($\alpha = 0.10$) from the chloride slope and that had significant ($\alpha = 0.05$) linear relationships between normalized concentration and distance are shown. P values indicate the level of significance for slopes differing from chloride, suggesting that a portion of the variability is due to biotic processes or sorption.

5.3 Comparison of ditches and spills

Ditches were on average 71 % longer ($p = 1x10^{-4}$), 12 % narrower ($p = 0.035$), and tended to have 33% higher flow velocities ($p = 0.016$) than spills (Figure 5.10). Constituent concentrations also differed between ditches and spills (Figure 5.11). Specifically, TDN ($p = 0.003$), DOC ($p = 0.007$), HCO_3 ($p = 0.023$), K ($p = 0.001$), and Ca ($p = 0.010$) concentrations were greater in ditches than spills. NO_3 ($p = 0.058$) and NH_4 ($p = 0.055$) concentrations tended to be higher in ditches than spills. Loads of TDN ($p = 0.038$), NO_3 ($p = 0.034$), and K ($p = 0.048$) were significantly greater in ditches than spills (Figure 5.12). In contrast, loads and concentrations of TP and orthoP were not significantly different between ditches and spills. T-test comparisons of inlet and outlet loads showed that solute loads did not change along the length of ditches or spills ($0.29 < p < 0.97$), with the exception of orthoP which tended to be greater ($p = 0.058$) at spill outlets than inlets.

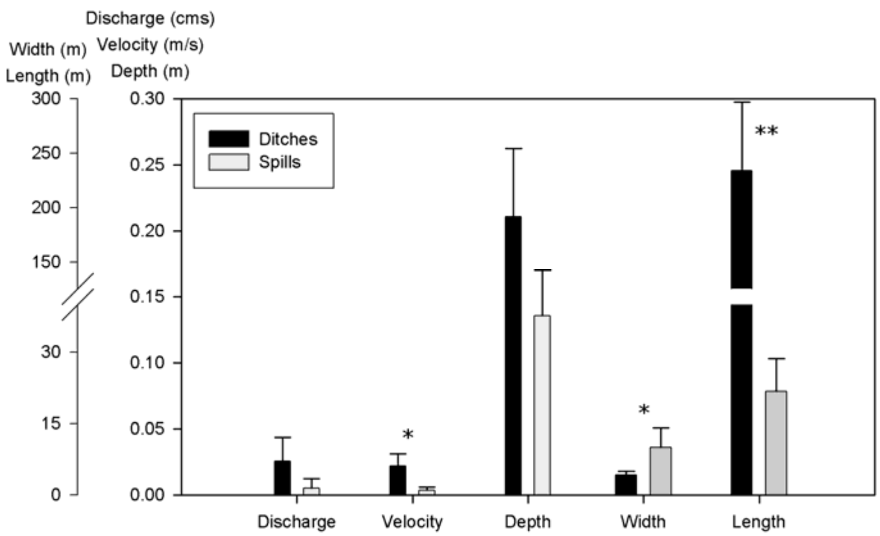


Figure 5.10: Mean and standard error of ditch and spill physical properties where * and ** denote statistically significant differences at α = 0.05 and 0.01, respectively.

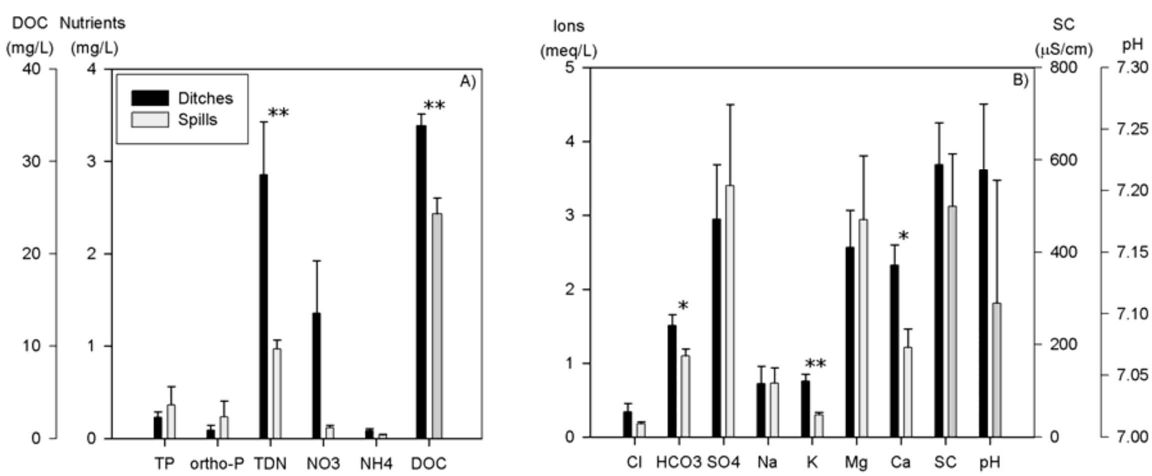


Figure 5.11: Mean and standard error of A) nutrients and DOC, and B) major ions, SC, and pH in ditches and spills where * and ** denote statistically significant differences at α = 0.05 and 0.01, respectively.

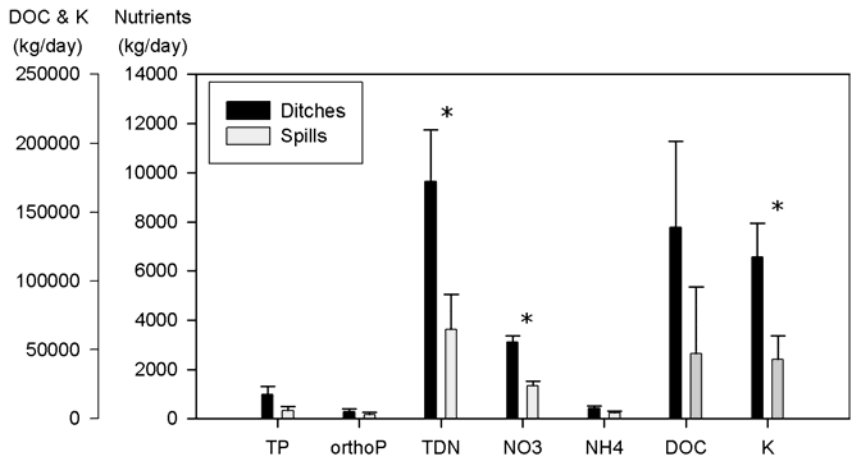


Figure 5.12: Mean and standard error of ditch and spill nutrient and DOC loads where *denotes a statistically significant difference at $\alpha = 0.05$.

5.4 Stream water quality

Typical of the northern prairies, the 2009 snowmelt period was short, lasting 28 days. Thaw (and subsequent flow) at the low drainage subbasin was delayed by about a week compared to that at the Smith Creek outlet and its other subbasins. Maximum stream discharge and concentrations of the water quality parameters measured, with the exception of SC and E. coli, occurred during onset of the spring freshet (Figure 5.13). Ratios of DIN ($NO_3 + NH_4$) to DIP (PO_4) measured near the Smith Creek outlet indicate N limiting conditions on nearly all sampling dates in 2009 (Figure 5.14). During the recession limb of the hydrograph, the dominant anion switched from HCO_3 to SO_4 (Figure 5.15), coincident with an increase in SC to >520 mS/cm (Figure 5.13). This switch was observed in each of the studied subbasins.

There was a trend toward higher discharge, and nutrient and salt concentrations at the subbasins with medium high and high wetland drainage compared to the subbasin with low wetland drainage (Figure 5.13). After normalizing concentration data for variations in streamflows and basin areas, differences in export coefficients among subbasins with variable wetland drainage were found for the 2009 freshet (Table 5.2). Subbasins characterized by medium high and high wetland drainage had significantly higher mean export coefficients of TP, orthoP, HCO_3, SO_4, Mg, and Ca than the subbasin with low wetland drainage. Because of its large size, the subbasin characterized by medium high wetland drainage had the highest total export of nutrient, salts and bacteria (Figure 5.16). However, total export of these water quality constituents at Smith Creek outlet, which is located just upstream of the town drainage (Langenburg Creek), was higher than the sum of the three studied subbasins.

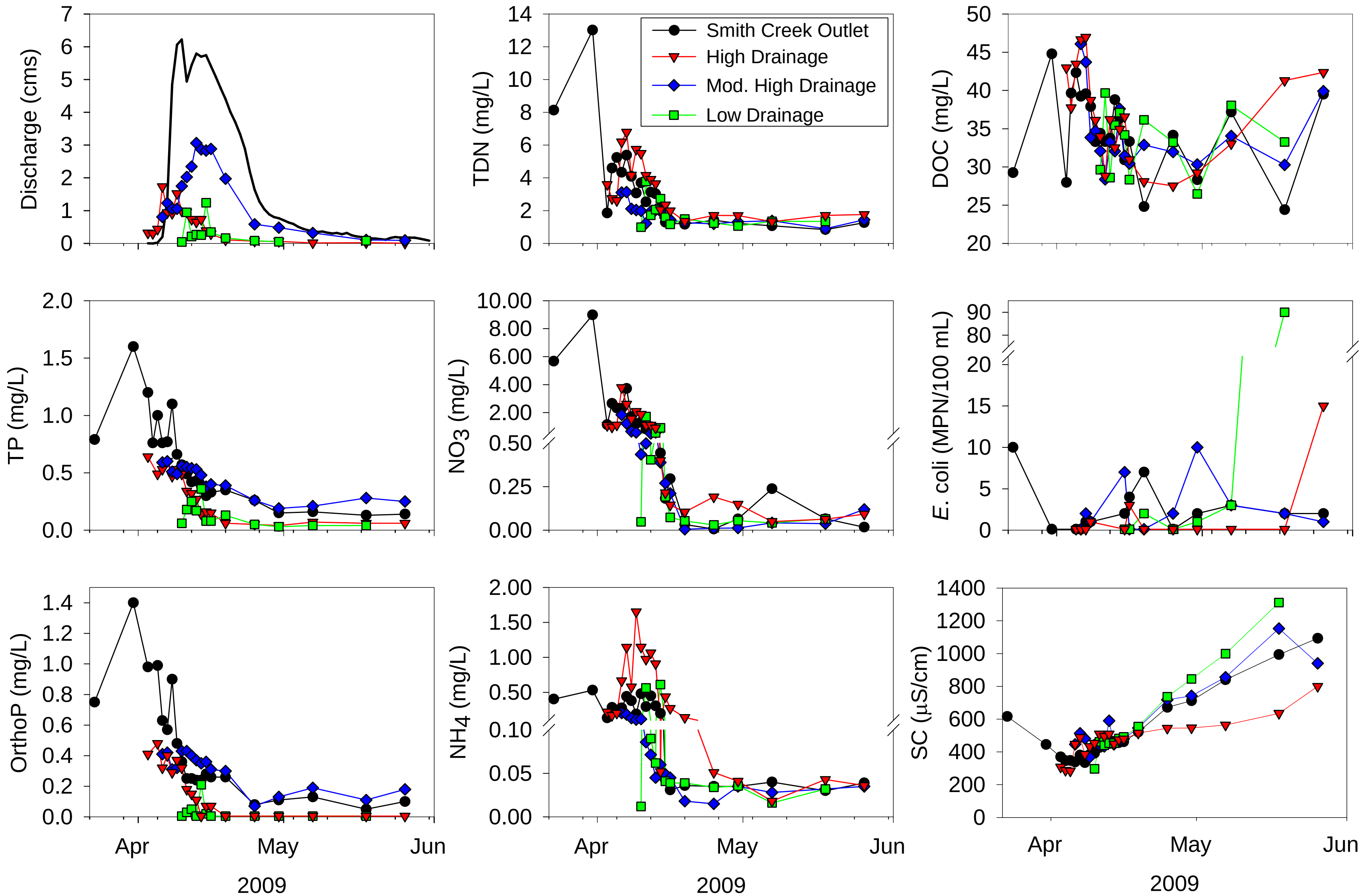


Figure 5.13: Discharge and water quality concentrations during the 2009 freshet at sites characterized by differing watershed drainage: high (TV1), medium high (SC4), and low (SC5). No data are presented prior to April 8, 2009 due to ice at the gauge station.

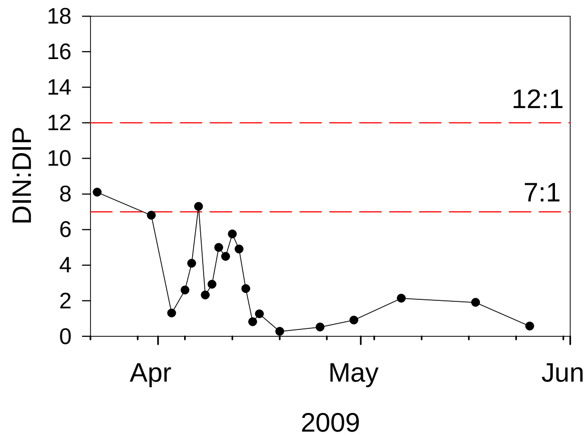


Figure 5.14: Ratios of DIN (NO_3 + NH_4) to DIP (PO_4) measured near the Smith Creek watershed outlet (SC3) during 2008 and 2009. Values below seven indicate N limiting conditions while those above 12 indicate P limiting conditions (Wetzel, 2001).

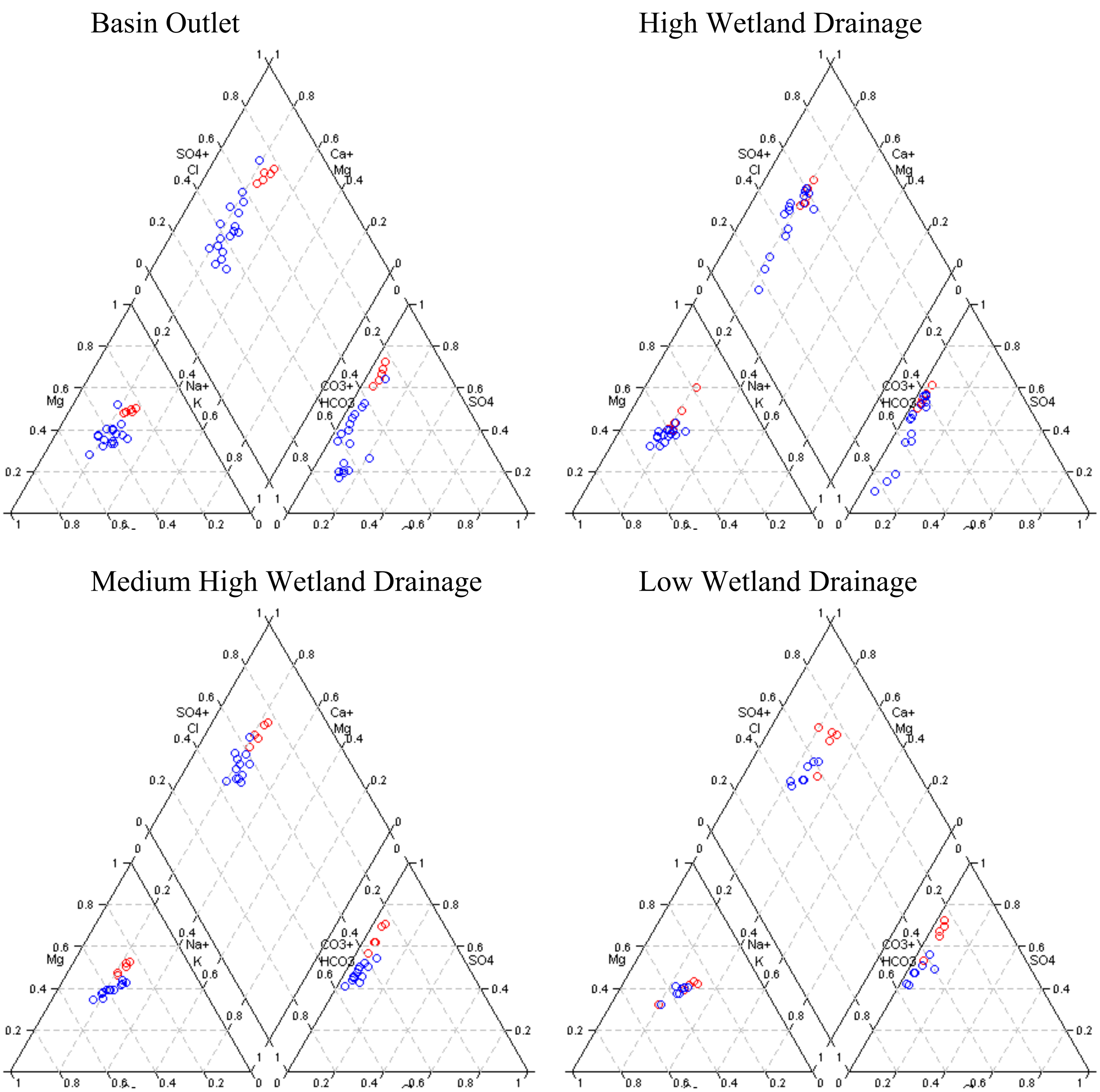


Figure 5.15: Piper plots for: basin outlet (SC3) and sub-basins characterized by high (TV1), medium high (SC4), and low (SC5) wetland drainage. Blue and red circles indicate samples obtained during the snowmelt period (March 23 – April 19, 2009) and following snowmelt (April 25 – May 26, 2009), respectively.

Table 5.2: Mean (standard error) stream export coefficients for the 2009 freshet by site. Differing superscripts indicate significant difference at α = 0.05, assessed using a one-way ANOVA and the Holm-Sidak method for multiple pair-wise comparisons.

Variable	Units	Wetland Drainage Intensity			ANOVA results	
		Low	Med. High	High	F	*P*
Q	$m^3/day/km^2$	392^a (151)	1212^b (154)	1293^b (208)	5.24	0.011
TP	$kg/day/km^2$	0.06^a (0.02)	0.60^b (0.07)	0.55^b (0.13)	5.81	0.008
OrthoP	$kg/day/km^2$	0.01^a (0.01)	0.45^b (0.06)	0.34^b (0.09)	6.94	0.003
DOC	$kg/day/km^2$	14.1^a (5.53)	$41.2^{a,b}$ (4.83)	49.6^b (9.08)	4.59	0.019
TDN	$kg/day/km^2$	0.93^a (0.42)	$2.31^{a,b}$ (0.27)	6.02^b (1.45)	5.49	0.010
NO_3	$kg/day/km^2$	0.29^a (0.19)	0.63^a (0.1)	2.34^a (0.79)	3.37	0.048
NH_4	$kg/day/km^2$	0.1^a (0.06)	0.1^a (0.01)	1.1^b (0.31)	6.38	0.005
Cl	$kg/day/km^2$	1.7^a (0.4)	8.5^a (0.7)	8.0^a (2.2)	3.54	0.042
HCO_3	$kg/day/km^2$	34.0^a (11.2)	123.7^b (17.3)	120.9^b (24.2)	4.28	0.024
SO_4	$kg/day/km^2$	26.8^a (7.1)	95.0^b (11.2)	82.1^b (18.6)	3.92	0.031
Na	$kg/day/km^2$	3.0^a (1.1)	10.4^b (1.3)	$6.3^{a,b}$ (1.3)	6.56	0.004
K	$kg/day/km^2$	6.2^a (2)	$24.7^{a,b}$ (3.5)	29.3^b (5.7)	4.98	0.014
Mg	$kg/day/km^2$	5.4^a (1.8)	23.1^b (2.7)	21.7^b (3.8)	6.43	0.005
Ca	$kg/day/km^2$	9.5^a (3.5)	36.0^b (3.4)	40.4^b (7.8)	5.36	0.010

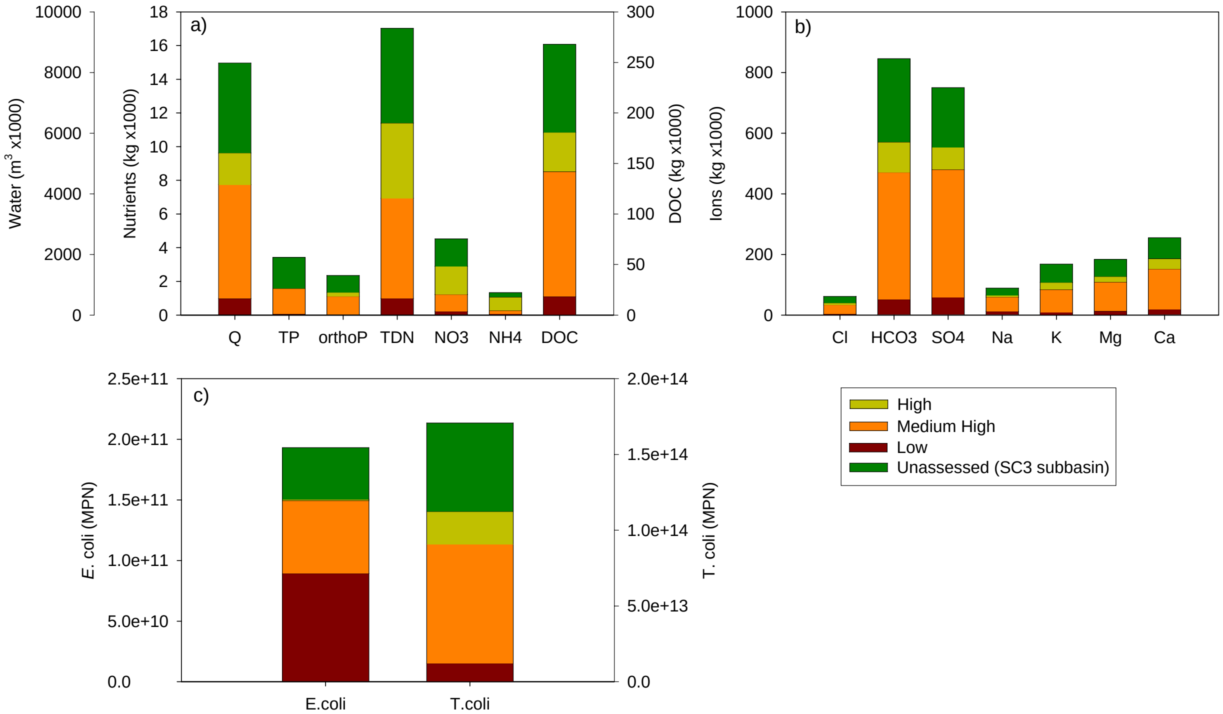


Figure 5.16: Total exports of: a) water (Q), nitrogen (TDN, NO3, and NH4) and phosphorus (TP and orthoP); b) major ions; and c) bacteria (*E.* coli and T. coli) during the 2009 freshet (3 April - 26 May) at sub-basins characterized by low, medium high, and high wetland drainage. Exports for the SC3 subbasin (*c.f.* Figure 4.3) were calculated by subtraction of exports for the three subbasins from that for the whole watershed. Total height of the bars indicates export for the whole Smith Creek watershed.

5.5 Description of macroinvertebrate assemblages

Acquired were 17 samples representing eight sites in 2008, and 15 samples representing eight sites across two time periods in 2009. Because we were unable to find suitable sampling sites through much of the upstream Smith Creek watershed, we focused sampling efforts in 2008 on replication at the site level. Specifically, six samples were produced from the Smith Creek at Marchwell (SWA_2008_16) site, eight samples were produced from the Smith Creek East of Langenburg site (SWA_2008_17), three from the Langenburg Creek East of Langenburg Site (SWA_2008_18), and a single sample from the Smith Creek at Werle Farm Site (SWA_2008_19). Sample number 3 from the Langenburg Creek East of Langenburg and samples 2, 3, and 4 from the Smith Creek at Werle Farm all contained too few macroinvertebrates for further analysis.

Based on the consistency of the results at the Marchwell and Langenburg sites (SWA_2008_16 and SWA_2008_17 respectively, Table 5.3) in 2008, and constraints of budgeted sorting and identification costs, only a single sample from each of the sites monitored was analyzed in 2009. However, the remaining three samples from each of the sites have been retained for future analysis.

Overall, 27,985 benthic macroinvertebrates were collected, representing 106 unique taxa (Appendix C). This study was dominated by the fairy shrimp (Anostraca), the dipteran midge larvae (Chironomidae), freshwater worms (Oligochaeta), nematodes (Nematoda), one genera of mayfly (*Caenis*) and blackfly larvae (Simuliidae). There were no surprises or occurrences of unexpected or rare taxa based on previous understanding of prairie streams in the area (see Phillips et al., 2008).

The 2008 results indicate that the Smith Creek at Marchwell is a healthy site, and not significantly different from reference conditions (Table 5.3). However, the Smith Creek east of Langenburg produced three of seven sites which were stressed, the Langenburg Creek was overall severely impaired, and the Smith Creek at Werle Farm was stressed and close to the impairment threshold (Table 5.3). A closer look at the nature of the ecosystem health conclusions based on the individual metrics that make up this TSA identifies that abundance is consistently within reference condition (Table 5.4).

Table 5.3: Sites sampled for ecosystem health in 2008.

Smith Creek Site Code	Smith Creek Watershed Site	TSA Results D	ncP	Overall Health
SWA_2008_16_1	Smith Creek North of Marchwell	1.86	1.00	Healthy
SWA_2008_16_2	Smith Creek North of Marchwell	1.50	1.00	Healthy
SWA_2008_16_3	Smith Creek North of Marchwell	0.89	1.00	Healthy
SWA_2008_16_4	Smith Creek North of Marchwell	1.28	1.00	Healthy
SWA_2008_16_5	Smith Creek North of Marchwell	0.87	1.00	Healthy
SWA_2008_16_6	Smith Creek North of Marchwell	1.19	1.00	Healthy
SWA_2008_17_1	Smith Creek East of Langenburg	2.39	0.99	Healthy
SWA_2008_17_2	Smith Creek East of Langenburg	1.09	1.00	Healthy
SWA_2008_17_3	Smith Creek East of Langenburg	3.18	0.62	Stressed
SWA_2008_17_4	Smith Creek East of Langenburg	2.21	1.00	Healthy
SWA_2008_17_6	Smith Creek East of Langenburg	4.03	0.12	Stressed
SWA_2008_17_7	Smith Creek East of Langenburg	2.76	0.89	Stressed
SWA_2008_17_8	Smith Creek East of Langenburg	0.93	1.00	Healthy
SWA_2008_18_1	Langenburg Creek East of Langenburg	7.75	0.00	Impaired
SWA_2008_18_2	Langenburg Creek East of Langenburg	33.02	0.00	Impaired
SWA_2008_18_4	Langenburg Creek East of Langenburg	4.09	0.11	Stressed
SWA_2008_19_1	Smith Creek at Werle Farm	4.17	0.09	Stressed

Table 5.4: Metric values and their probability of difference from reference (Metric Significance)

SWA Site	Metric Values: Abundance	Metric Values: Taxa Richness	Metric Values: Shannon's Diversity	Metric Values: CA Axis 1	Metric Significance: Abundance	Metric Significance: Taxa Richness	Metric Significance: Shannon's Diversity	Metric Significance: CA Axis 1
SWA_2008_16_1	453	4.42	2.56	0.87	1.00	0.85	1.00	1.00
SWA_2008_16_2	235	4.21	2.46	1.02	1.00	0.97	1.00	1.00
SWA_2008_16_3	120	3.55	2.24	1.11	1.00	1.00	1.00	1.00
SWA_2008_16_4	396	4.01	2.36	0.98	1.00	1.00	1.00	1.00
SWA_2008_16_7	58	3.20	2.37	1.15	1.00	1.00	1.00	1.00
SWA_2008_16_8	365	3.73	2.08	0.85	1.00	1.00	1.00	1.00
SWA_2008_17_1	1295	4.88	2.15	0.80	1.00	0.30	1.00	1.00
SWA_2008_17_2	142	3.63	2.01	1.02	1.00	1.00	1.00	1.00
SWA_2008_17_3	844	4.75	2.28	0.38	1.00	0.45	1.00	1.00
SWA_2008_17_4	483	4.69	2.25	0.79	1.00	0.52	1.00	1.00
SWA_2008_17_6	635	4.49	2.28	0.15	1.00	0.77	1.00	0.89
SWA_2008_17_7	311	3.14	2.32	0.49	1.00	1.00	1.00	1.00
SWA_2008_17_8	496	2.74	1.78	0.75	1.00	1.00	1.00	1.00
SWA_2008_18_1	634	2.33	1.54	2.57	1.00	1.00	1.00	0.00
SWA_2008_18_2	732	1.21	0.28	-6.63	1.00	0.82	0.01	0.00
SWA_2008_18_4	1634	1.89	1.48	1.64	1.00	1.00	1.00	0.86
SWA_2008_19_1	28	0.00	0.00	0.87	1.00	0.01	0.00	1.00
SWA_2009_10_1	196	1.14	0.84	3.65	1.00	0.74	0.43	1.00
SWA_2009_11_1	224	1.29	0.54	0.88	1.00	0.89	0.05	1.00
SWA_2009_4_1_May	1102	1.86	1.12	-0.43	1.00	1.00	0.95	0.02
SWA_2009_4_3_April	1068	4.45	1.75	0.85	1.00	0.82	1.00	1.00
SWA_2009_5_1_April	192	3.23	2.22	1.00	1.00	1.00	1.00	1.00
SWA_2009_5_1_May	337	6.36	2.76	0.82	1.00	0.00	0.92	1.00
SWA_2009_6_1_May	523	2.56	1.21	0.93	1.00	1.00	0.99	1.00
SWA_2009_6_2_April	740	2.88	1.39	0.90	1.00	1.00	1.00	1.00
SWA_2009_7_1_May	424	3.31	1.66	0.91	1.00	1.00	1.00	1.00
SWA_2009_7_2_April	546	3.12	1.54	0.89	1.00	1.00	1.00	1.00
SWA_2009_8_1	70	0.71	0.39	0.86	1.00	0.24	0.02	1.00
SWA_2009_9_1	576	3.46	2.58	0.82	1.00	1.00	1.00	1.00

Early 2009 spring sampling results indicate that all stream sites in the Smith Creek were healthy regardless of drainage stress (Table 5.5). However, May samples show that SC 4 had become impaired and SC 5 had become stressed (Table 5.6). Ditches in the 2009 study ranged from healthy at DT 3, stressed in DT 4, DT 2 and DT 1 (Table 5.6).

Table 5.5: Test Site Analysis results for sites from April 13, 2009 in the Smith Creek Watershed. *D* indicated the magnitude of effect or difference from reference, while *ncP* is the measure of significance.

Smith Creek Watershed Site	Drainage Stress	TSA Results D	TSA Results ncP	Overall Health
Smith Creek at SC4	Medium High Drainage	2.46	0.98	Healthy
Smith Creek at SC5	Low drainage	0.66	1.00	Healthy
Thingvala Creek at TV1	High Drainage	1.51	1.00	Healthy
Smith Creek at SC3	Very High Drainage	1.23	1.00	Healthy
Ditch at DT3	Ditch	1.76	1.00	Healthy
Ditch at DT1	Ditch	3.00	0.75	Stressed
Ditch at DT4	Ditch	2.99	0.75	Stressed
Ditch at DT2	Ditch	3.30	0.52	Stressed

Table 5.6: Test Site Analysis results for sites from April 13, 2009 in the Smith Creek Watershed. *D* indicated the magnitude of effect or difference from reference, while *ncP* is the measure of significance.

Smith Creek Watershed Site	Drainage Stress	TSA Results D	TSA Results ncP	Overall Health
Smith Creek at SC4	Medium High Drainage	5.54	0.00	Impaired
Smith Creek at SC5	Low drainage	3.88	0.18	Stressed
Thingvala Creek at TV1	High Drainage	1.82	1.00	Healthy
Smith Creek at SC3	Very High Drainage	1.28	1.00	Healthy

6 Discussion

6.1 Wetland water quality characterization

Researchers have previously used ion dominance patterns and SC (as a proxy for net groundwater seepage rates) as indicators of wetland permanence (Sloan, 1972; Millar, 1976; Driver and Peden, 1979). Although exchanges between the wetland and deep groundwater have only a minimal influence on the water balance, their effect on salinity can be important as the direction of flow determines whether salts accumulate in the wetland due to upward flow or are leached out of the wetland by outward flows (Hayashi et al., 1998b; van der Kamp and Hayashi, 2009). SC has thus been used as an indicator of relative position of wetlands along regional groundwater flow paths (LaBaugh and Swanson, 2004). However, results presented herein show that neither SC nor ion dominance can be used to distinguish among wetland permanence classes at Smith Creek watershed. Although five of the six highest SC measurements were obtained from permanent wetlands, there was no significant difference among permanence classes for SC or solute concentrations. The high within permanence class variation likely occurred because localized shallow groundwater flow systems dominate the groundwater hydrology of prairie potholes (Hayashi et al. 1998a; van der Kamp et al., 2008). Thus, linkages between SC and permanence may only be appropriate for comparisons made at the regional scale, and the focus of this study was at a smaller spatial scale. Furthermore, solute concentration and SC were likely diluted during the sampling period by snowmelt runoff, which would mask differences between permanence classes. Differences among permanence classes may however become more pronounced later in the season due to evapoconcentration (Rózkowska and Rózkowski, 1969; LaBaugh et al., 1987). Other studies have mostly conducted their sampling campaigns mid-summer (e.g. Arts et al., 2000); however, wetlands included in this study were not sampled later in the season because seasonal wetlands typically become dry. The lack of difference among permanence classes may also be attributed to the fact that classes are not necessarily strictly distinct and static. Instead, potholes within each permanence class can show sharp differences in time of inundation and in water depth between periods of drought and deluge (Johnson et al., 2004), conditions common in the prairies.

The wetlands sampled grouped distinctly into low SC and HCO_3 dominated groups, largely typical of the wood wetlands, and relatively high SC and SO_4 dominated groups, largely typical of grass and crop wetlands. An association between SC and anion dominance has been previously observed in prairie lakes and wetlands (Rawson, 1944; Rózkowska and Rózkowski, 1969; Barica, 1975; LaBaugh et al., 1987, Swanson et al., 1988). As solutes become increasingly concentrated by evaporation in closed basins, saturation levels for calcium and magnesium carbonates are reached first, and then of calcium sulfate, causing the minerals to precipitate in that order (Holland, 1978). The significantly lower SC and solute concentrations measured in the wooded areas may be a result of a high order topographic position of these wetlands in the Smith Creek watershed and thus their likely position near the top of regional groundwater flow paths (Tóth, 1963; Rózkowski, 1969; Devito et al., 2000).

Topography, however, is not always a good indicator of water table elevations in the prairies (van der Kamp, pers. comm.). Transects studies by Barica (1978) indicate that closed basin lake salinity varied considerably regardless of the general topographic slope. Also, wood wetlands may, in part, have lower Ca, K, and Mg concentrations than grass or crop wetlands because they uplands are comprised primarily of aspen (*Populus tremuloides*), which are known to store these solutes in their standing biomass (Wang et al., 1995).

SC and solute concentrations measured throughout Smith Creek were lower than maximum values measured at other lakes and wetlands in the prairie pothole region (Rózkowska and Rózkowski, 1969; Barica, 1975; LaBaugh et al., 1987; Detenbeck et al., 2002; Waiser, 2006) and were comparable to values observed by Nicholson (1995) in a northern Alberta transition zone between semi-arid prairie and moister boreal forest. These lower concentrations may be the result of dilution caused by the relatively higher annual precipitation at Smith Creek watershed compared to other parts of the prairie pothole region (Millet et al., 2009), which would minimize the precipitation to evaporation ratio and the effects of evapoconcentration. Additionally, the mid-summer precipitation events that occurred in the region during the 2008 summer prior to wetland sampling caused many potholes to fill with dilute rain water and remain relatively full at the time of freeze-up. SC was also likely low because samples were collected in late spring shortly after the wetlands have filled with dilute snowmelt runoff and before solutes have become concentrated by evaporation. Dilute salt concentrations in precipitation have been measured 220 km southwest of Smith Creek at Bratt's Lake station, which is part of the Canadian Air and Precipitation Monitoring Network (CAPMoN, 2007). Although not measured, it is likely that some of the HCO_3 dominated wetlands with SC near 400 μS/cm became SO_4

dominated later in summer. This inference is based on the work of LaBaugh et al. (1987) and Detenbeck et al. (2002), which showed some temporal progression in seasonal wetland ion dominance patterns.

Some of the spatial variation in some wetland nutrients can be attributed to variations in land cover and permanence classes. The higher concentrations of TP, TDN and DOC in seasonal wetlands likely results from the pronounced periods of flooding and drying that affect them. Mineralization of organic matter is enhanced during dry periods, resulting in leaching of nutrients from standing dead litter and sediment when the next inundation occurs (Bärlocher et al. 1978; Neill, 1995; Baldwin and Mitchell, 2000; Aldous et al., 2005). In contrast, the lack of vegetation and continuous flooding of sediments located within the open-water zones of permanent wetlands and some semi-permanent wetlands would not lead to the same annual release of nutrients (Brinson et al., 1981). The near continuous flooding of semipermanent and permanent wetlands suggests that conditions may be sufficiently reduced for denitrification to occur, also leading to comparatively lower total N concentrations (Neely and Baker, 1989; Crumpton and Goldsborough, 1998). Seasonal wetlands have been shown to have higher TP concentrations compared to semi-permanent ones due to leaching from standing dead plants in the marsh zones surrounding them (LaBaugh and Swanson 2004). Further, seasonal wetlands are often cropped when they are dry, thus they can have higher N and P concentrations than semipermanent or permanent wetlands because of direct fertilizer application (Cowardin et al., 1981).

In addition to differences in wetland nutrients among wetland permanence classes, differences in some nutrients were found among wetland land cover types. The greater TP and K concentrations in crop than wood and grass wetlands may have resulted from fertilizer inputs being transported to wetlands from their cropped watersheds (Hansen et al., 2002; Little et al., 2007; Tiessen et al., 2010) during snowmelt. The finding that N and DOC did not differ significantly among land cover types was surprising given that other researchers have shown that wood litter contains fewer nutrients and DOC than grass litter (Fuller and Anderson, 1993; Köchy and Wilson, 1997), and that nutrients stay stockpiled in the woody biomass located throughout wood wetland watersheds for extended periods of time (Wang et al., 1995). The lack of significant difference in N among land cover types may be attributed to the fact that the majority of wetlands sampled, similar to shallow prairie lakes, were eutrophic and characterized by low DIN:DIP ratios and were thus N limited (Barica, 1990; Hall et al., 1999).

The greater TP and K concentrations measured in crop than wood and grass wetlands may also be the result of varying amounts of surface runoff from different land cover types. Surface runoff and the nutrients transported with it are likely greatest in cropped areas partly because of higher overland flows during spring (van der Kamp et al., 2003; Bodhinayake and Si, 2004). The infiltration potential of frozen soils is higher if the soils are dry and have a well developed macropore structure compared to saturated soils with poor macropore development, which have very low infiltration potential (Gray et al., 2001). Cultivation reduces macroporosity and infiltration capacity (Bodhinayake and Si, 2004), which increases infiltration of snowmelt water and rain, and decreases surface runoff to depressions (van der Kamp and Hayashi, 2009). Further, snow and rain interception by forest canopies reduce snowpacks and throughfall (Pomeroy et al., 1997). Consequently runoff in wood areas is likely lower than grass and crop areas.

It is unlikely that the observed differences in wetland salinity and nutrient concentrations among land cover types were attributed to differences in soil characteristics. While spatial variation in soil types occur across the Smith Creek watershed (Figure 4.1), the majority of wetlands resided in soil units characterized by orthic or calcareous soils on lower slopes or poorly drained depressions. In all, 100%, 46%, and 50% of crop, grass, and wood wetlands, respectively, were located in soil units with orthic soils in depression, 17% and 36% of grass and wood wetlands were located in soil units with calcareous soils in depression, and 38% of grass wetlands and 14% of wood wetlands were located in soil units with carbonated and mixed (orthic and calcareous) soils in depressions. There was a trend, however, between SC-SO_4 dominance and soil type. Forty one percent of wetlands characterized by high SC and SO_4 dominance were located in soil units characterized by slightly larger proportions of sandy loams (OxWh2, OxWs2, and Yk12). These soils are expected to have a relatively higher hydraulic conductivity which means that the wetlands would likely experience greater shallow groundwater and salt fluxes.

Results also show that the interaction between land cover and permanence classes did not significantly influence nutrient or salt water quality parameters. This result is not surprising given that the control mechanisms on water quality parameters attributable to wetland permanence and land cover type are likely additive. The lack of significant interaction may have also resulted from the relatively high variability of wetland water quality measured. Other factors not considered in this study that could also influence prairie wetland water quality include grazing, cultivation, and tillage practices, varying groundwater fluxes, and the presence of willow rings surrounding wetlands.

6.2 Factors controlling temporal patterns in wetland water quality

The LR3 wetland effectively trapped N, P, DOC, coliforms, and major ions during runoff events and exchanged them with the surrounding uplands between events prior to the construction of the drainage ditch. Intensive temporal measures at the LR3 wetland suggest hydrological processes are the dominant control on all the studied ionic water quality parameters, except for HCO_3. However, the lack of significant correlations with Cl for most nutrient variables and differing seasonal dynamics of concentrations, masses and normalized masses between major ions and DOC compared to N, P, and bacteria suggest there are other processes important in determining wetland nutrients. This concept of differing control mechanisms for major ions and nutrients was hypothesized by LaBaugh et al. (1987). Seasonal fluctuations of ions and DOC appear to be primarily linked to hydrological processes due to the significant correlations with Cl. Instead, variable seasonal dynamics observed between major hydrological events and the lack of significant correlations between Cl and TP, orthoP, NH_4, NO_3, and bacteria suggests that wetland N, P, and bacteria are linked to both hydrological processes and biotic/sorption processes. For example, sequences of algae and plant uptake and decay, microbial processing (i.e. mineralization, nitrification, and denitrification), sedimentation, and waste from waterfowl and semi-aquatic mammals have been all shown to influence wetland nutrient availability (Barica, 1974; Neely and Baker, 1989; Neill, 1995; LaBaugh and Swanson, 2004). Removal mechanisms for bacteria in wetlands include sediment retention and natural die-back (Hemond and Benoit, 1988; Auer and Niehaus, 1993).

Temporal variations in solute concentrations and masses in the wetland can largely be attributed to the hydrologically isolated nature of the wetland. The estimate of ~6 mm/day of water lost from the wetland is within the range of those for other isolated prairie wetlands (1.7 – 7.4 m) via shallow groundwater infiltration and evapotranspiration (Shjeflo, 1968; Millar 1971; Woo and Roswell, 1993; Hayashi et al. 1998a; Su et al., 2000). Shallow, lateral flows, driven by hydraulic gradients and transpirative demand by upland plants, occur at wetland margins within the top 5-6 m of till in this landscape where the hydraulic conductivity is relatively high due to fractures (Hayashi et al., 1998a). These flows transport solutes with them (Hayashi et al. 1998b; Parsons et al., 2004), explaining the reduction in mass of most water quality constituents in the wetland between rain events.

Transport with lateral flows can effectively concentrate solutes in uplands (Arndt and Richardson, 1993; Winter and Rosenberry, 1995). They are then likely to be leached and cycled back to the wetland during snowmelt and rainfall runoff events. Precipitation events can also cause the water table beneath the wetland margin to rise above the pond level causing a reversal in shallow groundwater flow toward the pond (Gerla 1992; Winter and Rosenberry, 1995; Hayashi et al., 1998b; Parsons et al., 2004), which would also transport solutes back to the wetland. Daily water level increases in the LR3 wetland exceeded daily precipitation by >5 mm on 14 days during the study period, indicating that surface and/or subsurface runoff likely contributed to the increase in wetland water storage. Specifically, runoff contributions to the wetland likely occurred on at least nine days during the midsummer rain events as well as on May 28, June 12, June 23, October 6 and October 12. Surface and/or subsurface runoff are thus likely responsible for the high solute masses and bacteria loads observed during the snowmelt period and the increases that occurred following the rain events.

Concentrations of DOC and major ions increased during rain free periods likely due to evapoconcentration and mass decreased likely due to the transport with water out of the wetland by means of shallow groundwater seepage (Hayashi et al. 1998b; Waiser, 2006), as is described above. Data from Bratt's Lake station of the Canadian Air and Precipitation Monitoring Network (CAPMoN), which is 220 km southwest of the Smith Creek watershed, show precipitation is dilute with regards to major ions (CAPMoN, 2007). As a result, inputs during the midsummer rain events probably led to DOC and major ion dilution. It is thus likely that the increases of DOC and major ions mass measured immediately following rain events were caused primarily by runoff and shallow groundwater inputs.

Following snowmelt, the wetland had high nutrient concentrations. While transient, elevated N and P concentrations immediately following snowmelt are important to note because ditched wetlands drain at this time. Similar to results from Batt et al. (1989) and LaBaugh and Swanson (2004), these elevated N and P concentrations coincided with periods of use by breeding waterfowl. They also are coincident with a 2.5-fold increase in wetland volume, relative to the previous fall. Freshly flooded above ground litter leaches stored N and P to the water column (Reddy and Patrick, 1975; Davis and van der Valk, 1978; Neill, 1995) and nutrients accumulated in soils of the wetland periphery are released upon re-wetting (Reddy and Patrick, 1975; Murkin et al., 2000; Aldous et al., 2005). Soils in the region are naturally nutrient rich, especially in P (Anderson, 1988). In addition, nutrients were likely released from the cropped areas of the catchment that received fertilizer in fall 2007 and May 2008, as has been shown to occur in other agricultural systems (Hansen et al., 2002; Little et al., 2007).

The midsummer runoff events led to increased TKN and TP mass in the LR3 wetland. Prior to the rain events orthoP represented 13% on average of TP and increased to 60% following the rain events. This increase in the proportion of TP as orthoP may have resulted from the transport of orthoP with sediment (Neely and Baker, 1989) during overland flows. Although NH_4 and NO_3 are predominantly transported with surface runoff and subsurface flows, respectively (Neely and Baker, 1989), their percentages of TN were similar before and during the midsummer rains. Average concentrations of NO_3 and NH_4 in precipitation at the CAPMoN

Bratt's Lake station (2007) are 2.1 mg/L and 0.95 mg/L, respectively. These concentrations are much greater than those measured in the wetland throughout the study period. Thus wet deposition in addition to upland runoff and leaching from soil and plants located within the freshly flooded wetland periphery could have contributed to the increase in nutrient concentrations and mass loads measured in midsummer. The normalized mass data (Figure 4.4) suggest that reductions in TP, orthoP, and NH_4 mass following the midsummer rain events were due to biotic uptake and/or biogeochemical reactions rather than hydrological processes. Furthermore, these data also suggest that NO_3 was removed from the wetland relative to Cl, likely by biotic uptake and denitrification. Although measurements of nitrogen cycling in prairie wetlands are limited (e.g. Moraghan, 1993), researchers such as Neely and Baker (1989) have noted that conditions are likely suitable for denitrification to occur: anaerobic conditions and the presence of a large organic carbon stock.

Temporal variations in HCO_3 were likely influenced by carbonate equilibrium relationships as Heagle et al. (2007) identified carbonate mineral dissolution to be an important geochemical reaction in a recharge prairie wetland. Normalized mass data showed that proportions of HCO_3 and Ca became elevated relative to Cl. This result suggests that the increase in these ions was not attributable to water inputs, marked by Cl variations, alone. Sulfate reduction was also identified as a key geochemical reaction by Heagle et al. (2007). The proportional mass data did not indicate that SO_4 was removed from the wetland differently than was Cl, meaning sulfate reduction was likely not an important driver of SO_4 concentrations at this site.

A major non-point source of disease causing coliforms and indicator coliforms in agricultural landscapes is runoff containing animal wastes from pastures or fields fertilized with manure (Hyland et al., 2003). While the wetland catchment was not fertilized with manure over the course of the study period, the high coliform densities measured have been observed in comparable agricultural systems following snowmelt and large precipitation events (Ontkean et al., 2003). Semi-aquatic mammals and waterfowl that commonly occupy the wetland can also contaminate wetland water and uplands with fecal matter (Hyer and Moyer, 2004; Kadlec et al., 2007). Muskrats occupied the wetland as evidenced from newly constructed lodges some time between June 25 and September 24, 2008.

6.3 Water quality characteristics of a newly constructed drainage ditch

Construction of the drainage ditch at LR3 transported solutes previously stored in the wetland downstream. Wetland water storage decreased exponentially when the drain was completed. Within four hours ~80% of solutes and 30% of water exported had exited the wetland. The newly constructed ditch acted primarily as a conduit, transporting solutes downstream directly from the wetland. Although some solute concentrations were significantly correlated with distance along the newly constructed ditch and had slopes that differed from the Cl slope, these relationships were not consistent across time. For example, concentrations of HCO_3 were higher along the ditch length 4 hr and lower 6 hr since the start of the drainage experiment and concentrations of orthoP were often constant along the length of the ditch with the exception of the first point that differed. These data thus suggest that no consistent biotic or abiotic processing occurred along the length of the new ditch. This result is in direct contrast with previous studies that have shown that agricultural drainage ditches can act as solute sources and/or sinks where changes in solute concentrations are attributed to sedimentation/resuspension, adsorption/desorption, biotic uptake/release, and microbial mediated reactions such as mineralization and nitrification (Skaggs et al., 1994; Sharpley et al., 2007; Strock et al., 2007). However, the conditions reported in this study differed from existing studies by: i) the experimental drainage ditch was shorter than ditches typically studied, such that the residence time may be too short for processing to occur; ii) the ditch was new and lacked well established aquatic vegetation and microbial communities; and iii) drainage occurred over a very short period in late fall when water temperature was near freezing. Cold temperatures have been shown by others to restrict nutrient uptake by vegetation, microbially mediated reactions, as well as sorption and diffusion rates (Kadlec and Reddy, 2001). The drainage conditions studied here are common in the prairies, and thus results given herein should be transferable to other watersheds in the PPR, although rigorous quantitative testing is recommended.

Although there was no change in concentration along the length of the new ditch, there was an increase in concentration of most solutes studied with time since the start of drainage. This trend suggests that a vertical concentration gradient existed in the wetland such that the water closest to the sediment and in the sediment porewater had the highest constituent concentrations. Barica (1974), Fisher and Reddy (2001), and Barker et al. (2010) have all found vertical concentration gradients and elevated concentrations in sediment pore water in similar marshes and shallow lakes. Fecal coliforms also tend to concentrate in sediment where they survive longer, potentially due to the greater organic matter present in the sediment than water column (Karim et al., 2004). A vertical gradient in concentration may also explain why the masses of TP, orthoP, NO_3, NH_4, E. coli and T. coli exported via the drainage ditch exceeded estimates of those in the wetland. Wetland water samples were obtained from the center of the wetland at half the water depth and thus the masses calculated from these samples would be underestimated if a vertical concentration gradient existed.

Another explanation for the increase in concentration of water quality parameters is that many wetlands in the prairies have salt rings around their perimeter. Salts are likely concentrated by shallow groundwater fluxes at this transition zone between wetlands and their uplands. Constructing a ditch that traverses this ring may have also contributed to the elevated concentrations and mass exceedances observed at LR3. Further research is needed to determine the cause of the mass exceedances because accurately estimating the amount of solutes leaving the wetland is the most important factor for predicting downstream export and associated ecological consequences, given that the ditch acted as a simple conduit.

Had it not been for the uncharacteristically high midsummer rainfall, constituent concentrations leaving the wetland would probably have been higher and constituent mass exported lower at the time of drainage. Proportions of nutrients transported by snowmelt runoff typically exceed exports during rainfall events given that snowmelt runoff typically dominates prairie hydrology (Timmons and Holt, 1977; Tiessen et al., 2010). Decreases in specific conductivity and by association ion concentrations, have been attributed to rainfall and decreased evaporation (Barica 1978; LaBaugh and Swanson, 2004). The increases in ion concentrations between October 22 and the time of drainage were likely a result of their exclusion from the overlying ice and their freezing-out into a reduced water volume (Barica, 1975; Schwartz et al., 1978; Lilbaek and Pomeroy, 2008). In comparison, low coliform densities at the start of the drainage experiment were likely the result of sedimentation on the wetland bed (Auer and Niehaus, 1993; Wang and Doyle, 1998).

6.4 Comparing artificial ditches and natural spills

Similar to the results for the newly constructed drainage ditch, water quality was not altered during transport along ditch or spill connections. Consequently, the significantly greater concentrations of TDN, DOC, HCO_3, K, and Ca observed in ditches than spills were most likely due to differences in wetland water quality rather than differing physical attributes of drains and spills. Ditching permanent and semipermanent wetlands effectively turns them into seasonal ones because they are drained each year. Further, drained wetlands are predominantly located in cropped areas, a situation typical across the PPR (Guntersspergen et al., 2002). In contrast, spills flowed predominantly from permanent wetlands and were located only in grassland and wooded areas. As outlined in section 4.1 of this report, seasonal wetlands are characterized by greater concentrations of TDN and DOC, and K concentrations are greater in cropped wetlands than grassland or wooded wetlands.

The ditches and spills also had significantly different physical characteristics. The ditches were more channelized, longer, and had higher flow velocities. Although this had no apparent direct influence on the quality of water moving along them, the differences are likely to influence the impacts on downstream water bodies. Wetland drainage ditches are created to connect wetlands to the watershed drainage network. In contrast, the short length of spills means they often are transiently connecting wetlands to other wetlands. These different connection characteristics suggest that wetland drainage has a higher likelihood of potentially enhancing downstream nutrient, salt and bacteria loading than spills.

To date, the only study of prairie pothole drainage effects on downstream water quality (nutrients only) was a modeling exercise that compared different scenarios of wetland restoration at Broughton Creek watershed, Manitoba (Yang et al. 2008). Using the SWAT model, the researchers ran a wetland restoration scenario in which the 2005 wetland area was increased to match 1968 conditions. They predicted a 23% reduction in TN and TP loads to the stream using an empirical nutrient export coefficient. In contrast, results presented herein suggest that a nutrient export coefficient other than 1 is not warranted. This is likely the result of generally low temperatures during the snowmelt period when ditches and spills flow and when new ditch construction occurs. Fall ditch construction following harvest is more common than spring ditch construction because wetter soils in spring impede access to wetlands by heavy machinery and farmers are not preoccupied with seeding. Instead, the volume of water and mass of constituents exported from wetland drainage will depend on: i) how effectively the ditch drains the wetland; and ii) the water quality characteristics of the wetland that is influences by the combined permanence and land use setting.

6.5 Stream water quality

There was a trend toward higher discharge for the subbasins with medium high and high wetland drainage compared to the one with low wetland drainage. Higher discharge is comparable to other studies that show wetland drainage increases effective contributing area and stream discharge (Campbell and Johnson, 1975; Saskatchewan Watershed Authority, 2008; Yang et al., 2008). Peak flows may have been delayed in the subbasin with low wetland drainage because of increased water storage potential (Pomeroy et al., 2009; Fang et al., 2010).

The dominant anion switch from HCO_3 to SO_4 on the falling limb of the freshet hydrograph coincident with an increase in SC to >520 μS/cm was apparent in all subbasins. Therefore, it was not associated with wetland drainage. Instead, it likely occurred as proportion of input from dilute snowmelt runoff decreased relative to that of saltier groundwater and more permanent surface water bodies.

Subbasins characterized by moderately high and high wetland drainage had significantly higher mean export coefficients of TP, orthoP, HCO_3, SO_4, Mg, and Ca than the low drainage subbasin. These nutrients and salts have been shown in this report and by other researchers (Waiser and Robarts, 2004) to be concentrated in isolated prairie potholes located in cropped watersheds. The potholes most frequently drained in Smith Creek watershed are semi permanent to permanent ones located in cropped areas.

The differences in water quality found among subbasins may have been partly attributed to variations in the land use distribution. Effects of nonpoint source pollution on stream water quality, particularly its eutrophic effect on downstream water bodies, have been widely documented; see Carpenter et al. (1998) for a comprehensive review. However, it is more likely that the differing mean export coefficients of TP, orthoP, HCO_3, SO_4, Mg, and Ca among subbasins was related to wetland drainage degree rather than land cover variations. Guo et al. (in press) report land covers in Smith Creek watershed as extracted from 2007 and 2008 classified SPOT5 images. Their data show land covers were very similar among subbasins; albeit they lumped subbasins SC4 and SC5 rather than reporting them separately (see Figure 4.3 for areas covered by these basins).

Generally, no differences in the water quality parameters studied were found for subbasins characterized by medium high and high wetland drainage. This is not surprising given the relatively small difference (7%) in the proportion of wetlands lost for the two subbasins. Unfortunately, site selection preceded wetland loss determination from remote sensing imagery. It is recommended that future studies examine water quality in relation to stream flows across a wider range of wetland drainage scenarios. Some subbasins of Smith Creek watershed have much higher wetland loss than those studied here; for example, SC3 experienced ~81% wetland loss between 1958 and 2007-2008. Total export of the various water quality constituents at Smith Creek outlet, which is upstream of the town drainage (Langenburg Creek), was higher than the sum of the three studied subbasins. This suggests the lower part of the watershed, where many wetlands have been lost, is a significant source of nutrients, salts and bacteria to Smith Creek. Future examination of water quality in the subbasins that make up the lower part of the Smith Creek watershed is thus warranted.

It has been shown in other agricultural systems outside of the PPR that wetland restoration reduces downstream nutrient loading (Woltemade, 2000). Water quality improvements are not necessary linearly related to numbers or areas of wetlands gained as the cumulative effects of wetland loss on downstream water quality are not usually additive and instead tend to be nonlinear (Whigham et al., 1988). Stream flows are also important in determining how effective wetlands are in maintaining good quality water downstream. For example, Johnston et al. (1990) reported that wetlands removed more TP, ammonia and suspended solids at high flows but removed more NO_3 at lows. It would be very interesting to compare water quality in Smith Creek under different wetland restoration scenarios.

6.6 Exceedance of Federal and Provincial water quality guidelines for wetlands, ditches, spills and streams

TP concentrations in the newly constructed ditch, 6 of 7 ditches, and 3 of 5 spills exceeded the Saskatchewan Watershed Authority (2007b) objective of 0.1 mg/L for the Lake Stewardship Program water quality index. This objective was also exceeded in 40 of 67 wetlands sampled. Exceedances occurred in 71%, 50%, and 59% of crop, grass, and wood wetlands, and 77%, 65%, and 40% of seasonal, semi-permanent, and permanent wetlands. The TP objective was also exceeded for 45% of the LR3 wetland sampling instances. The TP objective was exceeded for all samples from the streams draining the subbasins with medium high and high wetland drainage and 63% of stream samples from the subbasin with low wetland drainage.

The majority of NO_3 concentrations measured in the wetlands, newly constructed ditch, ditches and spills, and streams did not exceed the Canadian Council of Ministers of the Environment (CCME) Canadian Environmental Quality guideline for the protection of aquatic life (2.9 mg/L; Canadian Council of Ministers of the Environment, 2003). However, this guideline was exceeded twice in the LR3 wetland following snowmelt, in 2 of 7 ditches, and in 1 stream sample from the high wetland drainage subbasin. The CCME guideline for $NH_3 + NH_4$ is greatly affected by temperature and pH; for typical conditions the guideline ranges from 18.5 mg/L (0^oC, pH = 7.0) to 0.07 mg/L (15^oC, pH = 9.0) (Canadian Council of Ministers of the Environment, 2003). Concentrations of NH_4 did not exceed these guidelines, with the exception of the first two samples obtained following snowmelt in the LR3 wetland.

The use of water containing salt concentrations that exceed CCME guidelines for water used for irrigation (100 – 700 mg Cl/L) and for livestock (1000 mg SO_4/L and 1000 mg Ca/L) is not recommended. Concentrations of Cl, SO_4, and Ca measured in the 67 wetlands, ditches and spills, and streams sampled in 2009 were below the CCME guidelines; thus drainage water would not be anticipated to degrade water quality with regards to these parameters. The guideline for SO_4 was exceeded twice in the LR3 wetland when the wetland volume was lowest and in the newly constructed drainage ditch 1 hour after the start of drainage.

The CCME indicator bacteria guideline for recreational water quality (200 E. coli per 100 ml) was only exceeded by the samples collected in the newly constructed ditch 23 hours after the start of drainage. The CCME guideline for the protection of agricultural water for crop irrigation (100 E. coli per 100 ml) was exceeded July 23, 2008 in the LR3 wetland. With the exception of the November 20, 2008 sample in the LR3 wetland, all samples collected in the LR3 wetland and the new ditch exceeded the animal-specific CCME guideline for livestock watering (2 E. coli per 100 ml of sample). E. coli densities exceeded the livestock watering guideline for 29%, 27% and 17% of the stream samples taken from subbasins with low, medium high, and high wetland drainage, respectively

6.7 Stream ecosystem health

6.7.1 Effects of wetland drainage stress levels on receiving lotic ecosystem health

Data from 2008 and 2009 show that wetland drainage stress does not have a significant effect on the ecosystem health of the Smith Creek. However, the Langenburg Creek East of Langenburg is significantly impaired relative to reference, as two of the three samples from this site reveal significantly different assemblages (through CA Axis 1, Table 4.3) and one with significantly impacted diversity. This impairment may be the result of stressors from the town of Langenburg rather than wetland drainage as the samples are dominated by taxa such as Nematoda, Ostracoda, Oligochaeta, and Chironomidae, which typically reflect high organic pollution (Barbour et al., 1999). However, these results should merely be taken as an alarm that warrants further study into the how drainage from the town of Langenburg may influence the ecosystem health of Smith Creek.

The ditches included in this study did support aquatic benthic macroinvertebrate assemblages, and in the instance of ditch DT 3 it even had healthy assemblage. The other three ditches included in this study revealed stressed ecosystem health, particularly due to the richness and diversity measures of their condition. The assemblages in these three stressed ditches were dominated by the midges Chironomidae, the freshwater worms Oligochaeta and the biting midge (or no-see-ums) Ceratopogonidae with few other species. These are all organisms tolerant to environmental extremes and capable of multiple generations per year. Further, as the ditches dry these organisms either die and leave resting eggs, burrow deep into wet areas, or emerge and disperse as in the case of Chironomidae and Ceratopogonidae.

6.7.2 Findings on the influence of time on ecosystem health

Reassuringly, sampling efforts from 2008 whereby multiple samples from Smith Creek North of Marchwell, Smith Creek East of Langenburg, and Langenburg Creek East of Langenburg were taken all provided relatively consistent precision in conclusions about the ecological condition at each site. As such, we are confident that single samples from each site in this watershed are relatively accurate in estimating condition, and that am certain that comparisons can be made through time both between years and between sampling periods within a year. In particular, the site Smith Creek East of Langenburg from 2008 and SC 3 from 2009 are overlapping sites and can be compared between the two years of study. The condition of this site in the river did not change between years despite a few samples appearing “stressed” in 2008; closer inspection shows that they are stressed because their diversity is higher than reference thus can be interpreted as a positive measure of impact. Therefore, this mainstem Smith Creek site did not vary in its ecosystem health between years.

The second temporal component of this study, comparing between collection periods in the spring in 2009, produced results suggesting that ecosystem health may degrade as the stream slows and dries. Specifically, our two most northerly sites, and thus lower order and ergo earlier to cease flowing in the spring, dropped from a healthy assemblage of macroinvertebrates in the early spring to impaired at SC 4 and stressed at SC5. Interestingly, SC 4 has medium high drainage, while SC5 has low drainage. A potential explanation for the difference in impact between these two sites is that although they are lower order, the medium high drainage of the SC 4 site may have lead to a more flashy flow this spring while the low drainage of SC5 may have provided for a more prolonged and sustained flow into the later spring. The two remaining stream sites, TV1 and SC 3 are further downstream in the watershed and may have still had adequate flows to support this healthy macroinvertebrate assemblage through into mid May. Further investigation into the relationship between the flow in this watershed through time and the timing of ecosystem health degradation is warranted based on these preliminary observations.

6.7.3 Unanswered aspects of drainage-biotic relationships and future directions

The lack of baseline data makes it difficult to place sites such as this in perspective, and such information would allow the monitoring of ecosystem health through time as drainage practices in the watershed are either increased or decreased. Further, lacking is any information on the effect drainage may have had on the wetland ecosystem health in the Smith Creek watershed. Certainly completely draining a wetland and ploughing it into terrestrial agricultural land will result in a complete loss of aquatic ecosystem health, but intermediate drainage to control the expansion or merely to reduce the size of wetlands may have negative effects on the benthic macroinvertebrates due to loss of habitat complexity, and loss of habitat quantity. Developing a reference condition approach for wetlands would be extremely useful in setting ecosystem health objectives for wetlands, and assessing the impacts of activities such as drainage and land-use.

7 Conclusions and recommendations

7.1 Conclusions

Project results show that prairie wetlands act as traps for nutrients, ions and bacteria. Temporally intensive measures of ions and DOC in one, permanent wetland suggest that hydrology is a strongly regulator of their storage. In contrast, nutrient and bacteria storage in the wetland appeared to be regulated by hydrological and biotic processes. Spatial variations in wetland water quality can be attributed in part to different land cover and permanence classes. Unexpectedly, there was no interactive effect of land cover and permanence classes on wetland solute chemistry. Also demonstrated was that neither SC nor ion dominance can be used to distinguish among wetland permanence classes at Smith Creek watershed. This lack of association is in contrast with previous studies that have linked ion dominance patterns and SC (as a proxy for net groundwater seepage rates) to wetland permanence. Overall, the results mean knowledge of land cover and/or permanence class can be used to provide a reasonable estimate of the water quality of a wetland.

Wetland water quality was found to be an important control of water quality in drainage water. Thus, the occurrence of high nutrient concentrations (which were sometimes above federal or provincial water quality guidelines) measured in the LR3 wetland at the onset of the spring freshet has important implications because drained wetlands typically connect with streams or other downstream water bodies at this time. Hence, wetland drainage may augment downstream nutrient loads. Not assessed in the wetland drainage experiment was how changing the permanent wetland to a temporary one will influence nutrient exports in future spring freshet events. For instance, will nutrient concentrations remain high in the wetland from year to year, and similar to those observed in seasonal wetlands at Smith Creek? Or, will nutrients become progressively flushed from the stockpiles established in the drained wetland soils leading to reduced nutrient exports over time?

Results also suggest that the efficiency with which a wetland is drained is an important factor in quantifying downstream exports. The temporally intensive measures of drainage water quality suggest that a vertical concentration gradient existed in the LR3 wetland such that the water closest to the sediment and in the sediment porewater of the wetland had the highest constituent concentrations. Thus wetland water samples obtained from the center of the wetland at half the water depth are likely to provide an underestimate of the total solute mass stored in the wetland if calculated from these samples.

Ditches had higher TDN, DOC, HCO_3, K, and Ca than spills. Similar to the results for the newly constructed drainage ditch, water quality was not altered during transport along ditch or spill connections. Since water quality was not altered during transport along the length of ditch or spill connections, differences in wetland water quality likely controlled ditch chemistry rather than differing physical attributes of drains. Drained wetlands are predominantly located in cropped areas, a situation typical across the PPR, whereas natural connections among wetlands tended to be located in the grassland and wooded areas of Smith Creek watershed.

Nearly all water quality parameters studied were higher in streams draining subbasins with greater wetland drainage. The provincial TP objective for protection of The low ratios of DIN to DIP that indicate Smith Creek is usually N-limited at its outlet combined with the finding that ditches export greater TDN than natural spills suggests that even small future increases in wetland drainage in the watershed could lead to enhanced stream algal growth. As Smith Creek is a sub-basin of the Assiniboine River, excess nutrient loadings from draining wetlands could facilitate further eutrophication of Lake Winnipeg, into which the Assiniboine River drains. It is important to remember that wetland drainage is not the only landscape stressor in watersheds located in the Prairie Pothole Region that can impact downstream water quality.

This is the first to identify wetland drainage impacts on the instream ecosystem health. Overall, these preliminary results indicate that wetland drainage does not have a dramatic impact on the benthic macroinvertebrate assemblage. Due to the small number of sampling sites, results of this work are best used to inform future hypotheses about the interaction between wetland drainage and instream ecosystem health rather than being treated as a definitive conclusion with which to base management of wetland drainage. Aspects of the hydrology such as reduced permanence of wetlands produced by drainage could impact stream ecosystem health negatively as flashier, less stable freshets may result in quicker drying and less diverse assemblages of macroinvertebrates. Research is needed on this topic. Also lacking is any information about how changes in the water level of the drained wetlands may have altered their benthic macroinvertebrate assemblage or ecosystem health, and comparisons need to be made between the positive gains in ecosystem health downstream receiving waters get due to drainage compared to what may be lost from the wetlands being drained. It is hypothesized that the increase in amount of water in a lotic site may be the cause of increased ecosystem health. Conversely, decreasing water in a wetland via drainage could lead to declining wetland ecosystem health if the benthic macroinvertebrates respond similarly (but inversely) in streams and wetlands.

7.2 Recommendations

Recommendation 1:

Water quality along wetland drains was found to be similar to that in natural spills. Exceptions are that TDN, DOC, HCO_3, K, and Ca concentrations were found to be higher along wetland drains. The important difference, however, is that spills typically connect wetlands to one another whereas drains connect wetlands to streams. One option to consider that could reduce downstream export of solutes and bacteria during wetland drainage is having wetland drains emulate spills. Drains could be constructed in such a manner as to connect wetlands to other wetlands so that solutes and bacteria remain stored in the watershed. It would be ideal to consider locating these storage zones in areas of the watershed where land productivity is low.

Recommendation 2:

Little change in water quality along the length of wetland drains was found. This means that the use of an empirical nutrient export coefficient is not required to be included in a model suited to simulating spring freshet nutrient exports due to wetland drainage at Smith Creek. However, knowing wetland water quality and how effectively the ditch drains the wetland are important to predicting solute exports to streams. Future models should thus include representations of wetland land use and permanence setting, and wetland drainage efficacy.

Recommendation 3:
Many wetland drains connect directly to roadside ditches that then empty into Smith Creek. An evaluation of solutes losses or gains during transport along roadside ditches was not investigated and future study is warranted.

Recommendation 4:
At Smith Creek, wetland drainage appears to strongly influence stream water quality, but not stream biotic integrity. This conclusion has strong policy implications but has been determined for only a small number of sites in one watershed in the prairies over a limited time period. Basins in other parts of the prairies with differing wetland and drainage configurations, soils and climate should be investigated to see if this conclusion holds elsewhere or is specific to Smith Creek.

Recommendation 5:
The wetland drainage experiment performed showed that considerable solute and bacteria exports occur during drainage of a permanent wetland. Needed is an examination of solute release upon re-wetting of the wetland bed and subsequent export along drains for wetlands that have experienced repeated draining.

Recommendation 6:
This study did not examine benthic macroinvertebrate assemblages in wetlands being drained. As drainage reduces the permanence of a wetland, investigation into linkages between wetland water availability and biotic health is needed.

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10 Other aspects

10.1 Communications

The research group utilizes several methods to communicate our project and its findings with other researchers, the people living in the watersheds we study, and society. These include conference talks, academic publications, public open houses and website development (www.usask.ca/hydrology). Journal publications are forthcoming and we are currently preparing an open house for the residents of Smith Creek watershed which is expected to be held in early spring of 2011. Ms. Brunet will defend her thesis in March 2011. Below is a list of all scholarly communications of this project thus far. CBC Saskatchewan Radio and Ducks Unlimited Canada have asked for a copy of this report.

Brunet, N. and Westbrook, C.J. 2010. Prairie stream water quality in sub-basins characterized by differing degrees of wetland drainage. American Geophysical Union 2010 Fall Meeting, San Francisco, USA, December 13-17.

Brunet, N. and Westbrook, C.J. 2010. Prairie wetland drainage effects on water quality. Canadian Soil Science Society Annual Meeting, Saskatoon, Canada, June 20-24.

Brunet, N. and Westbrook, C.J. 2010. Prairie wetland drainage effects on water quality. Joint Assembly of the Canadian Geophysical Union and Canadian Meteorological and Oceanographic Society, Ottawa, Canada, May 31 to June 4.

Brunet, N., and Westbrook C.J. 2010. Characterization of spatial variations in prairie wetland water quality. CGU-HS Student Meeting, Edmonton, Canada, January 30.

Brunet, N. and Westbrook, C.J. 2009. Characterization of spatial variations in prairie wetland water quality. Drought Research Initiative Conference, Saskatoon, Canada, November 18.

Westbrook, C.J. Pomeroy, J., Fang, X., Guo, X., Minke, A., Brunet, N., Shook, K., Brown, T., 2009. Smith Creek and the importance of hydrometric data in modeling and water quality research. Water Survey of Canada, Saskatchewan Division Staff Workshop, December, Saskatoon, Saskatchewan.

10.2 Personnel involved

Cherie J. Westbrook – Principle Investigator, Assistant Professor, University of Saskatchewan
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11 Appendices

Appendix A: Nutrient concentrations (May 19-21, 2009) for 67 wetlands in Smith Creek watershed, SK. Notes indicate whether the upland area surrounding the wetland was tilled, used for grazing, and the type of crop harvested in 2008, as reported by land owners.

ID	Land Cover	Notes	Permanence	Max Depth (cm)	orthoP (mg/L)	TP (mg/L)	TDN (mg/L)	NO_3 (mg/L)	NH_4 (mg/L)	DOC (mg/L)	DIN:DIP --
W3	Grass	Ungrazed	Seasonal	44	0.005	0.09	1.13	0.018	0.011	33.0	5.8
W4	Grass	Ungrazed	Semiperm	79	0.005	0.09	0.91	0.002	0.016	28.9	3.6
W6	Grass	Ungrazed	Semiperm	63	0.005	0.13	0.90	0.021	0.032	30.3	10.6
W10	Grass	Ungrazed	Seasonal	62	0.03	0.17	1.08	0.018	0.036	28.4	1.8
W11	Grass	Ungrazed	Permanent	102	0.01	0.12	1.39	0.013	0.038	36.5	5.1
W13	Grass	Ungrazed	Permanent	87	0.01	0.07	1.01	0.007	0.010	33.1	1.7
W17	Wood	Ungrazed	Permanent	54	0.14	0.24	1.17	0.045	0.038	33.1	0.6
W18	Wood	Ungrazed	Permanent	60	0.06	0.13	0.94	0.009	0.018	19.2	0.5
W21	Wood	Ungrazed	Seasonal	44	0.12	0.22	1.50	0.014	0.041	38.6	0.5
W22	Wood	Ungrazed	Permanent	58	0.18	0.29	1.35	0.006	0.041	32.6	0.3
W24	Wood	Ungrazed	Semiperm	56	0.35	0.43	1.09	0.013	0.014	33.9	0.1
W27	Wood	Ungrazed	Permanent	100	0.03	0.16	1.05	0.017	0.044	21.6	2.0
W28	Wood	Ungrazed	Permanent	96	0.01	0.11	1.12	0.008	0.014	27.8	2.2
W30	Wood	Ungrazed	Semiperm	59	0.47	0.53	1.30	0.007	0.017	35.0	0.1
W32	Wood	Ungrazed	Semiperm	52	0.02	0.22	0.99	0.002	0.018	24.2	1.0
W34	Crop	Canola/Tilled	Permanent	67	0.03	0.08	1.16	0.027	0.040	22.3	2.2
W35	Crop	Canola/Tilled	Permanent	76	0.1	0.18	1.29	0.035	0.025	26.2	0.6
W37	Crop	Canola/Tilled	Semiperm	78	0.03	0.17	1.31	0.025	0.014	30.9	1.3
W40	Crop	Canola/Tilled	Seasonal	34	0.005	0.19	1.49	0.002	0.014	37.1	3.2
W42	Crop	Canola/Tilled	Seasonal	48	0.005	0.55	0.88	0.006	0.014	26.8	4.0
W43	Crop	Canola/Tilled	Permanent	109	0.005	0.05	0.77	0.006	0.036	21.4	8.4
W45	Grass	Grazed	Semiperm	74	0.03	0.11	1.41	0.008	0.053	31.3	2.0
W46	Grass	Grazed	Permanent	98	0.03	0.08	1.10	0.008	0.035	26.3	1.4
W47	Grass	Grazed	Seasonal	36	0.04	0.18	1.13	0.023	0.012	30.9	0.9
W48	Wood	Grazed	Permanent	90	0.005	0.02	0.81	0.006	0.012	22.5	3.6
W50	Wood	Grazed	Seasonal	34	0.005	0.10	1.48	0.010	0.042	30.1	10.4
W54	Wood	Grazed	Semiperm	44	0.005	0.09	1.05	0.012	0.035	29.8	9.3
W55	Wood	Grazed	Permanent	70	0.005	0.02	0.88	0.007	0.013	24.0	3.9
W61	Wood	Grazed	Permanent	80	0.005	0.02	1.14	0.005	0.014	29.5	3.8
W67	Wood	Grazed	Semiperm	65	0.005	0.04	1.15	0.004	0.012	31.0	3.2
W68	Wood	Grazed	Seasonal	20	0.005	0.06	1.61	0.002	0.013	31.7	3.0
W69	Wood	Ungrazed	Seasonal	22	0.005	0.11	1.82	0.040	0.041	52.8	16.2
W71	Wood	Ungrazed	Semiperm	66	0.005	0.17	1.40	0.037	0.040	30.1	15.4
W72	Wood	Ungrazed	Seasonal	61	0.03	0.18	1.48	0.023	0.041	35.5	2.1
W73	Wood	Ungrazed	Semiperm	64	0.02	0.05	1.77	0.034	0.015	42.4	2.4
W75	Wood	Ungrazed	Seasonal	28	0.005	0.11	1.66	0.022	0.042	38.3	12.8
W85	Crop	Wheat/Tilled	Semiperm	62	0.01	0.19	1.23	0.005	0.024	30.2	2.9
W86	Crop	Wheat/Tilled	Permanent	102	0.03	0.09	1.86	0.015	0.014	51.5	1.0
W87	Crop	Wheat/Tilled	Seasonal	26	0.26	0.61	1.98	0.027	0.039	50.7	0.3
W88	Crop	Wheat/Tilled	Semiperm	70	0.3	0.93	0.95	0.222	0.029	24.2	0.8
W89	Crop	Wheat	Permanent	102	0.005	0.06	0.83	0.007	0.031	20.7	7.6
W90	Crop	Wheat	Semiperm	87	0.005	0.13	1.64	0.016	0.020	38.8	7.2
W91	Crop	Wheat	Seasonal	50	0.43	1.10	1.65	0.027	0.020	44.2	0.1
W93	Crop	Canola	Seasonal	74	0.02	0.11	1.34	0.036	0.002	33.0	1.9
W96	Grass	Ungrazed	Seasonal	44	0.005	0.10	1.74	0.022	0.013	44.9	7.0
W98	Grass	Ungrazed	Permanent	90	0.01	0.03	1.15	0.022	0.033	33.2	5.5
W100	Grass	Ungrazed	Seasonal	32	0.005	0.05	1.25	0.028	0.034	35.3	12.4
W101	Grass	Ungrazed	Permanent	66	0.005	0.03	0.87	0.004	0.034	27.5	7.6
W102	Grass	Ungrazed	Seasonal	22	0.03	0.87	2.49	0.007	0.025	55.3	1.1

Appendix A: *continued*

ID	Land Cover	Notes	Permanence	Max Depth (cm)	orthoP (mg/L)	TP (mg/L)	TDN (mg/L)	NO_3 (mg/L)	NH_4 (mg/L)	DOC (mg/L)	DIN:DIP --
W103	Grass	Grazed	Permanent	97	0.12	0.29	1.00	0.002	0.045	29.7	0.4
W104	Grass	Grazed	Seasonal	26	0.03	0.22	1.41	0.005	0.009	33.6	0.5
W106	Grass	Grazed	Permanent	70	0.14	0.37	1.26	0.014	0.012	31.6	0.2
W107	Grass	Grazed	Permanent	98	0.005	0.07	0.79	0.019	0.023	24.8	8.4
W108	Grass	Grazed	Seasonal	20	0.01	0.42	2.77	0.026	0.048	49.6	7.4
W110	Grass	Grazed	Semiperm	46	0.005	0.12	0.85	0.226	0.015	24.2	48.1
W111	Crop	Canola	Semiperm	69	0.2	0.34	1.70	0.028	0.002	39.7	0.1
W112	Grass	Ungrazed	Semiperm	31	0.005	0.10	1.09	0.011	0.035	35.5	9.2
W113	Crop	Wheat	Permanent	91	0.005	0.05	1.06	0.045	0.035	27.5	16.0
W114	Crop	Wheat	Semiperm	84	0.38	2.80	1.60	0.008	0.039	37.3	0.1
W115	Crop	Wheat	Seasonal	34	0.13	1.30	1.83	0.022	0.016	42.0	0.3
W116	Crop	Canola	Permanent	80	0.005	0.12	1.34	0.016	0.084	37.5	20.0
W117	Crop	Wheat/Tilled	Seasonal	38	0.05	0.53	2.17	0.009	0.018	45.2	0.5
W118	Crop	Wheat/Tilled	Permanent	87	0.005	0.09	1.34	0.024	0.036	26.5	12.0
W119	Wood	Grazed	Permanent	70	0.01	0.08	1.31	0.226	0.015	34.7	24.1
W120	Grass	Grazed	Semiperm	74	0.005	0.07	0.90	0.014	0.015	28.1	5.8
W121	Grass	Grazed	Seasonal	55	0.02	0.57	0.90	0.025	0.036	19.6	3.0
W122	Grass	Grazed	Semiperm	60	0.005	0.08	0.92	0.026	0.033	24.2	11.8

Appendix B: Salt concentrations (May 19-21, 2009) for 67 wetlands in Smith Creek watershed.

ID	Land Cover	Permanence	pH	SC (μS/cm)	Cl (mg/L)	HCO_3 (mg/L)	SO_4 (mg/L)	Na (mg/L)	Mg (mg/L)	Ca (mg/L)	K (mg/L)
W3	Grass	Seasonal	6.75	163	2.1	89.8	0.8	1.3	9.7	8.6	16.0
W4	Grass	Semiperm	7.24	414	1.8	120.5	95.7	8.9	28.2	21.1	10.0
W6	Grass	Semiperm	7.31	656	1.7	124.7	210.0	19.3	68.5	39.2	11.9
W10	Grass	Seasonal	7.44	309	3.8	80.2	22.0	2.6	15.3	12.7	14.0
W11	Grass	Permanent	7.68	695	1.8	127.8	168.4	21.0	74.2	35.6	16.0
W13	Grass	Permanent	7.24	404	1.3	79.3	49.3	5.7	20.3	9.7	7.9
W17	Wood	Permanent	6.92	158	2.0	63.0	0.6	0.3	6.2	13.8	20.3
W18	Wood	Permanent	6.55	63	1.5	25.7	0.1	0.2	1.6	4.6	11.3
W21	Wood	Seasonal	6.78	149	2.3	54.4	6.3	0.3	4.8	11.3	23.4
W22	Wood	Permanent	6.86	131	1.9	62.8	0.5	0.2	4.8	10.0	18.9
W24	Wood	Semiperm	6.77	127	1.4	63.7	0.1	0.2	3.8	11.0	19.4
W27	Wood	Permanent	6.74	111	2.4	48.2	2.3	0.5	4.3	8.7	12.7
W28	Wood	Permanent	6.76	150	2.0	70.0	7.8	0.6	6.7	11.3	13.4
W30	Wood	Semiperm	6.83	127	1.7	64.7	0.1	0.2	3.8	9.8	21.5
W32	Wood	Semiperm	6.84	57	0.0	18.3	0.1	0.1	1.2	4.9	4.6
W34	Crop	Permanent	7.19	181	2.0	72.6	9.1	2.0	7.8	13.1	8.3
W35	Crop	Permanent	7.19	211	4.9	69.1	18.3	1.9	7.7	14.1	18.1
W37	Crop	Semiperm	7.26	296	3.3	95.3	40.0	3.8	10.0	16.3	9.6
W40	Crop	Seasonal	7.38	287	4.6	98.4	6.0	5.4	11.8	19.3	18.7
W42	Crop	Seasonal	6.80	147	9.5	56.2	1.0	2.3	4.8	11.5	13.4
W43	Crop	Permanent	7.59	230	2.9	66.9	2.7	1.7	6.6	10.4	11.2
W45	Grass	Semiperm	7.28	413	3.2	98.5	70.2	7.5	29.1	14.5	14.2
W46	Grass	Permanent	7.47	314	5.9	77.4	50.7	6.7	22.5	13.9	13.5
W47	Grass	Seasonal	7.10	414	12.4	103.4	52.1	6.9	25.4	22.3	12.8
W48	Wood	Permanent	7.75	306	2.0	113.2	39.7	4.5	17.5	18.3	12.9
W50	Wood	Seasonal	7.19	144	0.7	53.9	14.0	0.5	7.8	12.0	9.1
W54	Wood	Semiperm	8.62	451	1.9	62.7	114.6	9.9	41.2	30.7	17.3
W55	Wood	Permanent	8.42	374	0.7	66.7	44.3	2.4	14.3	19.6	7.3
W61	Wood	Permanent	7.74	422	3.9	94.6	81.7	6.8	24.5	18.2	18.2
W67	Wood	Semiperm	6.93	134	2.8	79.9	0.7	0.6	5.0	9.0	19.3
W68	Wood	Seasonal	6.88	112	1.2	55.5	1.3	0.4	4.7	10.8	12.6
W69	Wood	Seasonal	7.06	211	1.4	87.9	8.1	0.9	7.6	13.3	34.1
W71	Wood	Semiperm	7.09	182	2.0	80.4	3.9	0.8	5.9	11.3	16.7
W72	Wood	Seasonal	6.90	170	2.7	77.7	5.4	0.9	6.9	11.0	25.8
W73	Wood	Semiperm	6.94	196	1.7	110.9	9.2	1.5	10.3	12.7	22.5
W75	Wood	Seasonal	7.24	194	0.9	70.2	2.1	1.1	4.9	12.7	13.0
W85	Crop	Semiperm	7.07	504	3.5	125.5	111.0	4.2	27.7	32.6	18.9
W86	Crop	Permanent	7.91	577	6.1	93.3	140.8	10.6	38.4	23.3	23.9
W87	Crop	Seasonal	6.84	399	9.0	99.1	34.5	4.7	16.4	21.7	33.9
W88	Crop	Semiperm	7.19	677	7.0	135.4	167.4	11.0	57.8	40.3	42.6
W89	Crop	Permanent	8.16	1694	5.1	145.4	505.4	44.0	133.4	40.4	17.0
W90	Crop	Semiperm	7.51	506	3.4	68.7	92.6	6.1	20.1	27.0	12.6
W91	Crop	Seasonal	6.96	975	13.3	130.0	329.3	18.6	68.7	74.5	39.2
W93	Crop	Seasonal	7.09	391	6.8	115.1	31.9	3.2	16.1	24.0	33.2
W96	Grass	Seasonal	7.15	868	8.5	157.8	256.1	70.2	55.0	26.3	16.1
W98	Grass	Permanent	7.56	1241	8.6	161.6	417.9	120.1	107.8	53.0	20.9
W100	Grass	Seasonal	7.27	890	5.2	117.4	284.0	60.8	68.0	47.3	13.7
W101	Grass	Permanent	7.06	792	3.7	95.1	248.2	54.7	47.5	53.1	12.7
W102	Grass	Seasonal	7.80	865	5.8	212.1	177.6	24.8	54.5	34.3	19.6

Appendix B: *continued*

ID	Land Cover	Permanence	pH	SC (μS/cm)	Cl (mg/L)	HCO_3 (mg/L)	SO_4 (mg/L)	Na (mg/L)	Mg (mg/L)	Ca (mg/L)	K (mg/L)
W103	Grass	Permanent	7.98	584	7.6	114.1	108.7	12.7	43.5	27.1	18.9
W104	Grass	Seasonal	7.94	376	7.7	124.8	6.3	1.2	10.3	18.0	25.9
W106	Grass	Permanent	7.86	776	11.2	141.6	180.9	21.9	67.6	36.8	24.1
W107	Grass	Permanent	8.12	1303	13.6	160.1	464.5	80.7	128.9	31.1	23.5
W108	Grass	Seasonal	8.54	1557	46.3	116.8	742.6	100.9	166.1	135.9	32.2
W110	Grass	Semiperm	7.17	520	9.5	53.6	156.8	27.8	28.9	20.4	13.6
W111	Crop	Semiperm	7.37	795	15.1	139.5	191.4	12.1	47.0	49.8	55.8
W112	Grass	Semiperm	7.00	806	11.5	122.8	273.6	51.2	49.2	49.9	22.0
W113	Crop	Permanent	7.58	1396	10.1	163.1	538.6	61.7	137.1	60.3	24.3
W114	Crop	Semiperm	7.13	349	8.6	118.8	30.5	4.6	15.1	21.4	33.1
W115	Crop	Seasonal	7.08	314	11.6	131.6	8.8	1.5	8.3	19.6	39.1
W116	Crop	Permanent	8.05	1780	8.2	232.9	488.9	50.3	149.4	28.7	29.6
W117	Crop	Seasonal	7.23	771	4.2	162.0	187.6	6.6	52.9	71.4	22.2
W118	Crop	Permanent	7.43	823	1.7	103.6	329.0	20.0	89.9	76.5	16.2
W119	Wood	Permanent	8.24	307	1.5	165.5	7.7	2.9	19.5	15.2	16.5
W120	Grass	Semiperm	7.73	865	16.0	127.0	284.3	57.5	67.6	34.0	20.5
W121	Grass	Seasonal	6.59	98	2.0	33.7	2.9	0.6	3.7	8.3	9.0
W122	Grass	Semiperm	6.95	262	3.1	108.6	18.9	1.9	13.5	12.8	11.2

Appendix C: Taxa List Continued (Part 1 of 4)

	Smith Creek Watershed Site												
Major Taxa	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	
To Family	**2008**	**2008**	**2008**	**2008**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	
To Lowest Identification	**16**	**17**	**18**	**19**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Total**
Amphipoda													
Gammaridae													
Gammarus lacustris		3											3
Taltridae													
Hyalella azteca			5		3	402	4						414
Hydrachnidia	92	14	5		4	4031	24				4	2	4174
Platyhelminthes		11											11
Rhynchobdellida													
Glossiphoniidae						1							1
Helobdella elongata			2										2
Anostraca						5368	2				146	48	5516
Collembola	6	16	8										30
Nematoda	52	83	1642		67	2	17	5		8	6		1882
Oligochaeta	334	517	168	7	262	59	730	20	64	28	102	111	2402
Ostracoda	185	103	901		1		4	5					1199
Pelecypoda													
Sphaeriidae	2			182		45				4			233
Pisidium	2												2
Gastropoda	15	11	2	1		4		9		4	1		47
Lymnaeidae	106	238	14		38	17	10	14		80		2	519
Aplexa hypnorum		1											1
Fossaria	7	3											10
Fossaria/Stagnicola					3	4	2	1		12			22
Pseudosuccinea		45											45
Stagnicola		9			21	8							38
Stagnicola elodes					1					44			45
Physidae	30	12	2		16	2							62
Aplexa hypnorum					2	6							8
Physa	1												1
Physa skinneri	1	35	9		5			5		52			107

	Smith Creek Watershed Site												
Major Taxa	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	
To Family	**2008**	**2008**	**2008**	**2008**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	
To Lowest Ientification	**16**	**17**	**18**	**19**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Total**
Planorbidae		10	13		27	3		2		24	4		83
Gyraulus		2	13		4								19
Promenetus exacuous						2							2
Coleoptera													
Dytiscidae	4	6			11	1	5						27
Agabus	6	13			1	30							50
Colymbetes						1							1
Colymbetes exaratus		2											2
Hydroporinae						26						1	26
Hygrotus		3					2						5
Hygrotus sellatus							2						2
Ilybius					6								6
Liodessus			2										2
Neoporus						26							26
Oreodytes					2								2
Rhantus	1	2											3
Rhantus sericans		3											3
Haliplidae													
Haliplus	9	5	21		2			1					38
Haliplus immaculicollis	3				1								4
Hydraenidae						2				4			6
Hydrophilidae					2								2
Berosus	4	49	5	1	2		10						71
Helophorus		9			1	1	3	1					15
Paracymus		3											3
Diptera													
Ceratopogonidae	150	110	105	8	26	109	26	9	2	76	38	39	698
Atrichopogon		2			10	14							26
Pericoma/Telmatoscopus					24	16				104			144
Chaoboridae													

	Smith Creek Watershed Site												
Major Taxa	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	
To Family	**2008**	**2008**	**2008**	**2008**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	
To Lowest Ientification	**16**	**17**	**18**	**19**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Total**
Chaoborus					1	26	2	1					30
Chironomidae	497	1415	383	115	9	1159	164	338	221	2	56	24	4383
Culicidae	114	26		3		1	2	2					148
Aedes						17	23						40
Dolichopodidae	9	6			28	15	14		2	24			98
Empididae					4		2						6
Muscidae	1	7			7								15
Psychodidae	10	2	6		8						2	1	28
Sciomyzidae	10	16	2			1		3					32
Simuliidae	3	135					1						139
Simulium		279		40	532	14		2					867
Simulium venustum/verecundum					6								6
Simulium vittatum		120		55	94	4							273
Stratiomyidae		17	8		2								27
Nemotelus							2			8			10
Odontomyia	58					2							60
Stratiomys	10	56	2	1		6				4			79
Syrphidae	1												1
Tabanidae	2	1				2							5
Chrysops		1											1
Tipulidae	2	6	2			1	2			4			17
Tipula		203				1				8			212
Ephemeroptera													
Baetidae		1											1
Caenidae													
Caenis					6	2136	23	17					2182
Hemiptera													
Corixidae	41	17				17				4			79
Callicorixa audeni	1	2				2							5
Cenocorixa dakotensis								1					1

	Smith Creek Watershed Site												
Major Taxa	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	**SWA**	
To Family	**2008**	**2008**	**2008**	**2008**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	**2009**	
To Lowest Ientification	**16**	**17**	**18**	**19**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Total**
Hesperocorixa	3					2							5
Hesperocorixa atopodonta	1					2							3
Hesperocorixa laevigata	1												1
Hesperocorixa michiganensis	1	2											3
Hesperocorixa vulgaris						2							2
Sigara bicoloripennis		3			1	5							9
Sigara conocephela	1				1								2
Sigara decoratella	2	2				2							6
Sigara solensis	11					2							13
Lepidoptera	1						4			12			17
Odonata													
Anisoptera		4			2	26		6					38
Libellulidae	4	15	1										20
Zygoptera		1		1		27	4	2		8			43
Coenagrionidae													
Enallagma/Coenagrion						27	1				2	2	30
Lestidae													
Lestes	67	46	2							4			119
Trichoptera	46	2	4	1	6	2				4	1		66
Leptoceridae	2					1							3
Oecetis						4							4
Limnephilidae	64	577		2		14	3	100					760
Anabolia bimaculata	2	2					1						5
Asynarchus		34											34
Limnephilus		4											4
Philarctus/Limnephilus				15									15
Grand Total	1993	4349	3327	433	2413	12712	1263	428	70	576	421		27985