Prairie Pothole Drainage and Water Quality

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ABSTRACT

Pothole wetlands are ubiquitous throughout the Prairie Pothole Region and since 1900, 40-70% of potholes in the region have been drained to increase agricultural production. This thesis describes factors influencing spatial and temporal variations in wetland water quality and characteristics of drainage water. Research was conducted at Smith Creek watershed, southeastern Saskatchewan, where there has been controversy over recent renewed efforts to drain wetlands. Following snowmelt in 2009, 67 wetlands were sampled to determine whether spatial variations in wetland water quality were attributable to land cover, permanence classes, and surface drainage characteristics. Wetlands with cropped uplands had greater TP and K than wetlands with wooded and grassed uplands; TP, TDN, and DOC were higher in seasonally than permanently ponded wetlands; and salts were lower in wetlands with wooded uplands compared to wetlands with cropped and grassed uplands. Measurements of water quality of one permanently ponded wetland over a 20 week period in 2008 showed that the wetland acted as a solute trap. Variations in salts and DOC were influenced by hydrological processes such as runoff, evaporation, and shallow groundwater seepage, whereas variations in nitrogen, phosphorus, and bacteria were influenced by biotic, sorption, and hydrological processes. The experimental drainage of this wetland in November 2009 demonstrated that its water quality was an important control of drainage water quality. Further, the wetland ditch acted as a simple conduit, i.e., little solutes loss or gain occurred along it. In spring 2009, water quality along seven ditches and five natural connections that form between wetlands (termed spills) was compared. Concentrations of most solutes were similar, except TDN, DOC, HCO_3^-, K^+, and Ca^{2+} that were higher in ditches than spills. Minimal changes in water quality along ditches and spills occurred, likely due to the low temperatures occurring in spring that restrict biotic processing and sorption. Notably, because ditches connect wetlands to streams, as opposed to spills that connect adjacent wetlands, ditches have a greater potential to contribute to downstream solute loading. Wetland drainage efficiency and wetland water quality were deemed the factors critical to determining solute exports via ditches. Results of wetland water quality and drainage characteristics can be useful to future modeling exercises and could be used to inform wetland drainage practices and policies.

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ANOVA	Analysis of variance
C	Carbon
Ca^{2+}	Calcium
CCME	Canadian Council of Ministers of the Environment
Cl^{-}	Chloride
cms	Cubic metres per second
d	Depth
DEM	Digital Elevation Model
DIC	Disolved inorganic carbon
DIN	Dissolved inorganic nitrogen (NO_3^{-} + NH_4^{+})
DO	dissolved oxygen
DOC	Dissolved organic carbon
DT	Ditch
DR	Newly constructed drainage ditch
E. coli	*Escherichia coli*
F	R.A. Fisher's *F* stastistic
HCO_3^{-}	Bicarbonate
ICB	Ion charge balance
K^{+}	Potassium
LiDAR	Light Detection and Ranging
MANOVA	Multivariate analysis of variance
meq	millequivalent
Mg^{2+}	Magnesium
MPN	Most probable number
N	Nitrogen
Na^{2+}	Sodium
NH_4^{+}	Ammonium as nitrogen
NO_3^{-}	Nitrate as nitrogen
orthoP	Orthophosphate as phosphorus
P	Phosphorus
p	Statistical significance value
PO_4^{3-}	Orthophosphate
POC	Particulate organic carbon
PPR	Prairie Pothole Region
Q	Discharge
r	Pearson correlation coeffcient
SC	Specific conductivity
SO_4^{2-}	Sulfate

SP	Spill
SPOT	Earth Observation Satellite
SRC	Saskatchewan Research Council
T	Temperature
T. coli	Total coliforms
TDN	Total dissolved nitrogen
TKN	Total Kjeldahl Nitrogen
TN	Total nitrogen
TOPAZ	Topographic parameterization
TP	Total phosphorus
v	Velocity
w	Width
W1	Wetland 1
α	Probability of making a Type I error (rejecting the null hypothesis when the null hypothesis is true)

1.0 INTRODUCTION

The prairie pothole region (PPR) contains millions of pothole wetlands that generally lack surface water inflows and outflows. The ecological importance of these potholes is underscored by the fact that 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). Historically, agricultural practices have led to wetland drainage: over the past century, 40 – 70% of the wetlands located in the western prairies have been drained to increase agricultural production (Tiner, 1984; Dahl, 1990; Brinson and Malvarez, 2002; Watmough and Schmoll 2007). Recently, there have been renewed efforts to drain potholes (Watmough and Schmoll 2007), especially in the Canadian Prairie Provinces. Pothole drainage conflicts are centred around attempts to balance the private costs and social benefits associated with potholes on agricultural lands (Porter and van Kooten, 1993; Curtos et al., 2010). Costs accrued by private landowners include greater expense to farm around potholes, delayed seeding in inundated areas, and the foregone opportunity to increase agricultural production (Curtos et al., 2010). The social benefits of potholes include water storage and flood attenuation, wildlife habitat, and solute trapping (Curtos et al., 2010).

The construction of drainage ditches that connect previously isolated potholes to streams has been hypothesized to adversely affect downstream water quality (Leibowitz and Vining, 2003; Whigham and Jordan, 2003). However, to date there are no field studies to support or refute this conjecture. This thesis aims to explore the impacts of wetland drainage on prairie water quality by characterizing the spatial and temporal variation in prairie potholes water quality and assessing solute exports from naturally and artificially drained pothole wetlands. This work is a component of a larger research project that includes a comparison of stream water quality among subbasins where historical wetland distribution is similar, but recently the subbasins have been subject to differing degrees of drainage, and a study relating wetland drainage to changes in ecosystem function as determined by macroinvertebrate assemblages (Westbrook et al., 2011). Provided in the remainder of this chapter is a review of the literature describing what is known about how surrounding land use, permanence, and hydrology influence pothole water quality and insight into the possible downstream effects of their drainage.

1.1 Prairie Pothole Region

Prairie potholes are a regional type of isolated wetland found in the PPR of North America (Figure 1.1). The region is estimated to cover approximately 715 000 km^2 (Euliss et al, 1999). The prairie potholes formed during the last glacial retreat that created the hummocky, undulating terrain typical of the prairies (Tiner, 2003). About 40% of the PPR located in Canada consists of hummocky moraines which have a wetland density of 18 wetlands/km^2 and the remaining 60% of the Canadian PPR landscape is mostly lacustrine and fluvial materials which average five wetlands/km^2 (National Wetlands Working Group, 1988). Many of the wetlands located in the hummocky moraine region are isolated prairie potholes. These potholes normally do not contribute to streamflow (Stichling and Blackwell, 1957), but during very wet conditions, temporary surface connections can occur among potholes (Leibowitz and Vining, 2003; Winter and LaBaugh, 2003; Spence, 2006).

Figure 1.1 Map of the Prairie Pothole Region of North America.

The PPR represents only 10% of the continent's waterfowl breeding area; however, it produces half of North America's waterfowl in an average year (Smith et al., 1964; Batt et al., 1989). Much of the ecological importance of small prairie potholes is related to biodiversity: these wetlands often have high species richness due to moisture gradients caused by gentle slopes and varying moisture conditions (Leibowitz, 2003). Shallow wetlands thaw more quickly than deep wetlands in the spring and provide early feeding habitat for breeding ducks and other waterfowl. As well, prairie potholes and surrounding uplands provide the high habitat diversity required to support populations having broad life history requirements, such as amphibians that require aquatic habitat for breeding and larval development but later become largely terrestrial during adult life stages (Wilbur, 1984). Prairie potholes also play an important role in the life cycles of many mammals which markedly affect other components of wetland ecosystems (Fritzell, 1989). The muskrat is an example of an animal found throughout the PPR; they inhabit all types of wetlands temporarily, but they only prosper in wetlands deep enough to sustain under ice activity throughout the winter (Fritzell, 1989).

1.2 Pothole Water Budget

Many unaltered prairie potholes have no surface water connections, except perhaps in exceptionally wet years (Leibowitz and Vining, 2003). The most significant input (Figure 1.2) to the northern prairie pothole water budget is snowmelt; other large inputs include precipitation directly on the wetland and surface runoff during intense rainfall events (Woo and Roswell, 1993; Hayashi et al., 1998a; Winter et al., 2001; van der Kamp and Hayashi, 2009). The snowmelt water input is vital for the existence of wetlands because summer precipitation is exceeded by evapotranspiration in the semi-arid prairie region. Snowmelt is also important to wetland existence because windblown snow tends to be redistributed from areas of sparse vegetation and accumulates in the topographic depressions occupied by potholes (Fang and Pomeroy, 2008). The input from catchment snowmelt runoff is also generally high due to the reduced infiltration capacity of frozen soils (Gray et al., 2001; van der Kamp et al., 2003) unless macropores are abundant and soils are dry (van der Kamp et al., 2003; Bodhinayake and Si, 2004). For a typical seasonally ponded prairie pothole, snowmelt runoff transferred 30 – 60% of winter precipitation on the upland into the wetland to form the pond in the centre (Hayashi et al., 1998a). Pond refers to the variably inundated portion of the wetland (Figure 1.2). Overland flow events originating from the catchment are rare in the summer because unfrozen soils have higher infiltration and storage capacities and low moisture contents caused by large evapotranspiration demands (Hayashi et al., 1998a; van der Kamp and Hayashi, 2009; Pomeroy et al., 2010). Surface runoff is most frequently generated only in the riparian zone in summer where the water table is closer to the ground surface (van der Kamp and Hayashi, 2009).

The dominant pathways by which water leaves the wetland are evapotranspiration and infiltration driven by evapotranspiration at the willow ring (Woo and Roswell, 1993; Hayashi et al, 1998a; van der Kamp and Hayashi, 2009). For a typical seasonally ponded prairie pothole, infiltration accounted for 75% of water leaving the central pond and evapotranspiration accounted for 25% of the water level decline in the pond (Hayashi et al., 1998a) (Figure 1.2). Shallow groundwater exchanges occur readily through the relatively permeable fractured material (silty and clayey glacial till with hydraulic conductivity of ~1000 m/yr; van der Kamp and Hayashi, 2009) located a few meters beneath the wetland. Water infiltrating under the pond moves laterally to the wet margin and the uplands, and then vertically upward into the capillary fringe where it is consumed by evaporation and root uptake (Hayashi et al., 1998a). Deep groundwater flow exchanges have little effect on the water balance due to the low hydraulic conductivity (~0.1 m/year) of the deeper underlying tills (van der Kamp and Hayashi, 2009). Published estimates of groundwater recharge to regional aquifers by wetlands in the prairies range from 2 – 40 mm/yr (van der Kamp and Hayashi, 1998; Hayashi et al. 1998a). Depending on variations in climate, its position in the landscape, the configuration of associated water tables and the hydraulic conductivity of the underlying geological substrate, potholes can have a groundwater recharge, flow-through or discharge function (LaBaugh et al., 1998; van der Kamp and Hayashi, 1998; Euliss, 1999; Toth, 1999).

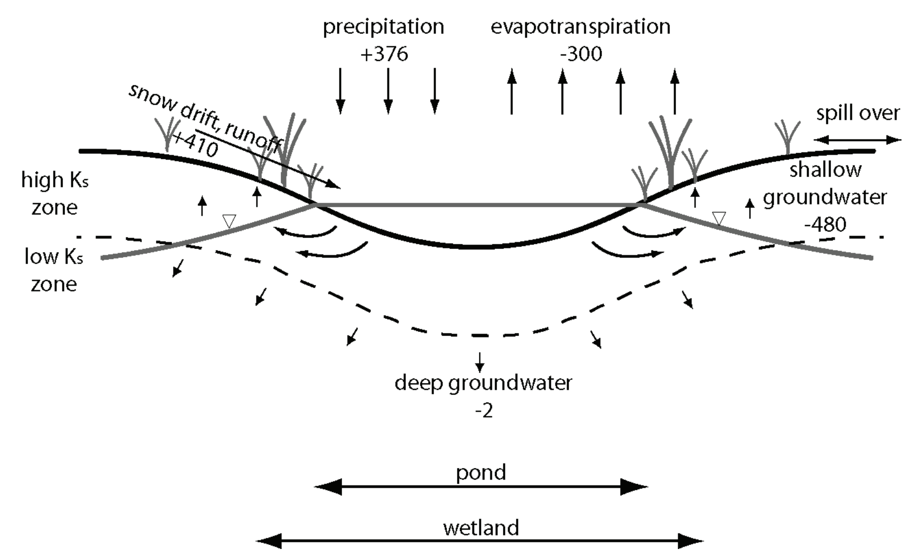


Figure 1.2 Principle prairie pothole water balance components (van der Kamp and Hayashi, 2009) and the annual water balance (mm) of a seasonally ponded wetland at St. Denis, SK (Hayashi et al., 1998a).

The magnitude of surface runoff inputs to the pothole is sensitive to changes in soil conditions and surrounding land cover (van der Kamp et al., 2003; Bodhinayake and Si, 2004). The infiltration potential of frozen soils is larger 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. Upland areas composed of grass tend to have higher macroporosity and infiltration capacity (Bodhinayake and Si, 2004), which increases infiltration of snowmelt water and rain, and decreases surface runoff (van der Kamp and Hayashi, 2009). For example, van der Kamp et al. (2003) showed that the conversion of land surrounding a pothole from cultivated to undisturbed grassland caused a pothole to dry out because grasslands more efficiently trap windblown snow, snowmelt infiltration into the frozen grassland soil was high enough to absorb most or all of the runoff in spring, and summer runoff was limited by the high infiltration potential of the grassland soil. Recent work has also shown that other changes on the land, specifically reduced stubble height in a cultivated field, increases basin snowmelt runoff by increasing blowing snow transport to and snow accumulation in the pothole (Fang and Pomeroy, 2008). Aspen forest stands have also been found to retain windblown snow (Fang et al., 2010).

Shallow groundwater exchanges are also influenced by land cover surrounding the pothole. In summer, trees around the pond seem to reduce the air flow and turbulent transport of vapor within the wetland such that evapotranspiration rates within the pond are less than potential evaporation rates of large lakes in the same region (Hayashi et al., 1998a). The transpiration of vegetation in the willow ring surrounding the pothole also drives the shallow horizontal flow of infiltrated water out of the pothole (Hayashi et al., 1998a). For example, Hayashi et al. (1998a) showed that the shallow flow direction was reversed, and was directed toward a pothole in St. Denis, Saskatchewan when vegetation was removed from a part of the upland in a year of summer fallow.

Isolated prairie potholes typically form no surface connections to the stream network at average water levels, and thus can be effective at flood attenuation (Hubbard and Linder, 1986; Murkin, 1998). A pothole will store precipitation and snowmelt up to a certain threshold beyond which it can spill over towards a down-gradient receiver (Leibowitz and Vining, 2003; Spence, 2006). The volume of storage available in prairie potholes isolated from the stream network depends on previous runoff and climactic conditions. Potholes are most effective in flood attenuation when they have a high capacity to store additional water (McAllister et al., 2000).

1.3 Pothole Permanence and Classification

The ponds at the centre of prairie pothole wetlands range in terms of water permanence from those that contain water for only a few days following spring snowmelt, to those that are continuously inundated. Shallow, local groundwater flow such as shoreline related seepage can have a critical influence on pond permanence and a pothole's hydrological function; however, deep, regional groundwater exchanges do not significantly affect pond permanence (Parsons et al., 2004; van der Kamp et al., 2003; van der Kamp and Hayashi, 2009).

Vegetation zone presence or absence, distributional pattern, and sequence are factors J. B. Millar used to create his 1976 classification system (Table 1.1). He based his system on Stewart and Kantrud's (1971) system, but made changes to reflect the specific needs of the Canadian Prairies. Wetlands in his system are named according to the type of vegetation zone dominating the central portion of the basin under normal water regimes. Salinity is included in Millar's classification system as it often reflects the involvement of groundwater in the moisture regime of a pothole (Millar, 1976; Hayashi et al., 1998b). According to Stewart and Kantrud (1971), seasonal and semi-permanent are the dominant pond permanence classes present in the prairies, in terms of surface area. Temporary ponds are numerous; however, their cumulative area is small, whereas permanent and alkali ponds are large but their quantity is limited.

Potholes of 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). A drought year was found to shift wetland time of inundation by approximately one permanence class. Johnson et al. (2004) explored differences in times of inundation during periods of drought (late 1980s) and deluge (mid 1990s) for seasonally, semi-permanently, and permanently ponded prairie wetlands in South Dakota, USA. They found seasonally ponded wetlands experienced the greatest variability in time of inundation (Table 1.2).

Table 1.1 Summary of normal vegetation patterns in stable wetlands. Images reproduced with the permission of the Minister of Public Works and Government Services Canada, 2011. (Millar, 1976, Wetland Classification in western Canada, p.13, Canadian Wildlife Service Report Series No. 37).

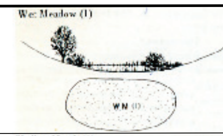
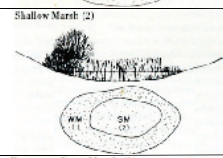
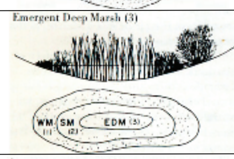
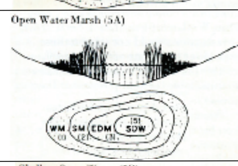
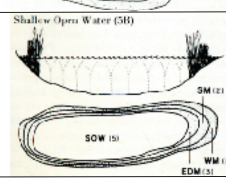
Vegetation Zone Sequence Letter abbreviations in each diagram identify vegetation zones		Description (National Wetlands Working Group, 1971)	Inundation	Vegetation Zone Type
Wet Meadow		Generally not considered true wetlands. Transition area occupying the central area of shallow depressions or peripheral bands of deeper ponds	Temporary ponds Three to four weeks in the spring, usually dry by late May.	Fine textured grasses and sedges of low stature, variety of forbs.
Shallow Marsh		Forms an inner band to the wet meadow and may occupy central areas of seasonal ponds, or marginal bands of semi-permanent to permanent ponds	Seasonal ponds Lasts until July or early August. Have 0 - 30 cm of water until midsummer (Walker and Coupland, 1970).	Coarse grasses, sedges, forbs (0.46 – 1.22 m), water tolerant herbs, some floating plants.
Emergent Deep Marsh		Depending upon water periodicity, may exhibit open water, emergent, or drawdown areas, usually with interspersed, patchy, or closed vegetation cover.	Semi-permanent ponds Ordinarily flooded to late spring or fall (1- 30 cm, (Walker and Cropland, 1970)), and occasionally throughout the winter.	Coarse and robust grass-like plants, taller than those in the shallow marsh zone.
Open Water Marsh		Stable, shallow open water subform occupying the central or deepest portion of the basin. Shallow open water zone is <75% of wetland's diameter.	Permanent ponds Flooding is permanent or occurs for several years at a time. Depth by the September exceeds 20 cm and may be >1 m.	Submergent and floating aquatic plants. Water is shallow enough to permit growth of most rooted aquatic plants.
Shallow Open Water Wetland		Stable, shallow open water subform occupying the central or deepest portion of the basin. Shallow open water zone is >75% of wetland's diameter.		

Table 1.2 Time of inundation for glaciated prairie wetlands in South Dakota, USA. Time of inundation is the percentage of sample days (~15) during the growing season when standing water was present. (Johnston et al., 2004).

Permanence Class	Dry Period (late 1980s)	Wet Period (mid 1990s)
Temporary	0%	0 – 67%
Seasonal	0 – 29%	46 – 100%
Semi-permanent	13 – 17%	100%

1.4 Pothole Water Quality

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). This thesis will focus on water quality 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 present in elevated concentrations. Isolated prairie potholes exhibit high spatial and seasonal variations in water quality (LaBaugh and Swanson, 2004). Different sources of water, land use of the catchment area, and pond permanence have been shown to regulate pothole water quality. Here I present a review of factors regulating isolated prairie pothole water quality followed by a discussion of how pothole water quality may change along man-made drainage ditches that connect isolated potholes to streams.

1.4.1 Dissolved Oxygen

Pond and lake water can be stratified into two zones separated by a transition zone, the thermocline. The thermocline is characterized by a large temperature gradient. Above the thermocline is the epilimnion, which remains in contact with the atmosphere. Below the thermocline is the hypolimnion, which is separated from the atmosphere and is non-turbulent (Kalff, 2002). Following stratification of eutrophic lakes, dissolved oxygen (DO) concentrations in the epilimnion remain close to saturation; however, DO decreases in the hypolimnion due to the decomposition of organic matter and the separation of the deep water (Kalff, 2002).

Potholes are generally assumed not to be stratified (Sloan, 1972; Barica, 1974a), owing to shallow depths (i.e., mixing depth > actual depth), wind action, and advective mixing (Sloan, 1972). However, Detenbeck et al. (2002) observed that all but one of 20 potholes (maximum depth of 125 cm) in North Dakota became stratified at least once during the growing season. Five of these remained stratified throughout the growing season, but only three approached anoxia near the bed. Even in unstratified potholes DO decreases with depth, and reaches anaerobic conditions within the sediment (Barica, 1974a). Anoxic conditions exist in pothole sediments because of microbial oxygen demand for the decomposition of organic matter and limited oxygen diffusion from the water column (Birgand et al., 2007). Most potholes develop anaerobic conditions during ice cover when aeration is prevented and decomposition processes dominate (Barica, 1974b). Large decreases in DO can also occur in midsummer following the collapse of significant algae blooms because of rapid bacterial decomposition of dead algal cells (Barica, 1974b). DO also fluctuates diurnally with productivity (Detenbeck et al., 2002).

The oxygen regime of pothole water affects a wide range of redox-sensitive biogeochemical processes (Mitsch and Gosselink, 1993). In oxygen depleted systems, iron is reduced, releasing phosphorus that was previously held as ferric phosphate compounds

(Reddy and DeLaune, 2008). Phosphorus can also be retained by wetlands as oxides and hydroxides of iron and aluminum (Mitsch and Gosselink, 1993). Higher DO tends to stimulate nitrification and inhibit denitrification (Birgand et al., 2007), which would elevate nitrate concentrations in potholes. Lower DO causes decomposition rates to be slower leading to an accumulation of organic matter on pothole beds (Neely and Baker, 1989).

1.4.2 Nutrients

Prairie potholes may be important in maintaining stream water quality by trapping and storing nitrogen (N) and phosphorus (P) derived from agricultural runoff (Neely and Baker, 1989; Johnston, 1991; van der Valk and Jolly, 1992; Crumpton and Goldsborough, 1998; Murkin, 1998). Thus, land cover characteristics in upland areas of prairie catchments play an important role in determining nutrient concentrations in potholes. For example, Crosbie and Chow-Fraser (1999) showed an increase in N and P concentrations relative to the proportion of agriculture land in the catchments of southern Ontario marshes. Nutrient concentrations in upland runoff are expected to be higher when a manure or fertilizer application is followed by an intense rainfall (Neely and Baker, 1989; Hargrave and Shaykewich, 1997; McDowell et al., 2001). Most of the nutrient transport from catchments has been shown to correlate well with sediment loss; thus nutrient loss can be inferred from soil transport rates (Hargrave and Shaykewich, 1997). Soil erodibility in the semi-arid Brown soil zone near Swift Current, SK is lowest when soils are frozen in winter, intermediate in summer and immediately preceding snowmelt, and significantly greater during snowmelt when soils are partially frozen (McConkey et al., 1997). In Wood Mountain loams on the Canadian prairies, snowmelt runoff concentrations of N and P from a summer fallow plot were much greater than from a wheat stubble plot, due to low vegetation litter cover, thereby facilitating the suspension and movement of nutrients (Nicholaichuk and Read, 1978).

N and P in the pothole water column can be exchanged to the atmosphere (N only), sediment-interstitial water, and living and dead biomass through biogeochemical processes (Figure 1.3). Each of these storage compartments can also be a N or P source to the pothole water. Inorganic forms of N found in prairie potholes are nitrite (NO_2^-), nitrate (NO_3^-), ammonia (NH_3), and ammonium (NH_4^+) (Neely and Baker, 1989). Except in very warm and alkaline environments (temperatures > 15°C, pH > 8.5), most of the NH_3 in freshwater exists in the ionic form, NH_4^+ (Emerson et al., 1975). Organic and inorganic forms of N are considered bioavailable (Antia et al., 1991). NH_4^+ is produced upon the decomposition of organisms or excretion by animals and can be converted back to organic N by plants and microorganisms, become sorbed to soil particles, or in aerobic environments it can also be oxidized to NO_3^- through nitrification, which is easily transported by flowing water (Kalff, 2002). Nitrification also produces N_2O as a by-product. Denitrification is the most significant process for N removal (as N gases) from isolated potholes (Neely and Baker, 1989). Tracer studies suggest that up to 80% of sustained external NO_3^- loads could be lost through denitrification (Crumpton et al., 1993; Moraghan, 1993). Other tracer studies, however, have shown N_2O emissions during nitrification are larger than those during denitrification in ephemeral, cultivated wetlands (Bedard-Haughn et al., 2006). Suitable conditions for denitrification to occur include anoxic conditions, a sizable denitrifier population, a source of organic carbon, and an abundance of NO_3^- (Neely and Baker, 1989; Birgand et al., 2007). Fluctuations in water levels and the drying out and subsequent re-flooding of ephemeral potholes exposes their soils to the atmosphere, which can result in larger N losses (Neill, 1995) due to the sequential processes of nitrification while soils are exposed to the atmosphere followed by denitrification after inundation (Neill, 1995; Baldwin and Mitchell, 2000).

The important forms of P in prairie potholes are inorganic P as orthophosphate (PO_4^{-3}) and organic P (Mitsch and Gosselink, 1993). PO_4^{-3} is considered bioavailable (Reddy et al., 1999) and is often applied to agricultural fields as fertilizer while organic P is formed primarily by biological processes. Sources of organic P include manure and plant matter, but it can also be formed from PO_4^{-3} (Csuros, 1997). P retention mechanisms in potholes include biotic processes: assimilation by vegetation, plankton, periphyton, and microorganisms, and abiotic processes: adsorption by sediments, precipitation, and exchange between sediment and the overlying water column (Reddy, et al. 1999). The most significant pathway for P removal from the pothole water column is movement to the sediment-interstitial water column via sedimentation and precipitation with ions (Neely and Baker, 1989; Reddy et al., 1999). P precipitates with calcium and magnesium compounds in alkaline environments, and with aluminum and iron in acidic environments (Reddy, et al. 1999). Precipitation with calcium, for example, increases significantly with calcium concentrations >100 mg/L and pH > 9 (Diaz et al., 1994). Depending on concentration gradients between the water column and the sediment-interstitial water, P can diffuse to the underlying soil, become adsorbed and retained in the sediment, or P can return to the surface water by diffusion and mixing by bioturbation and re-suspension during turbulent water conditions (Reddy et al., 1999).

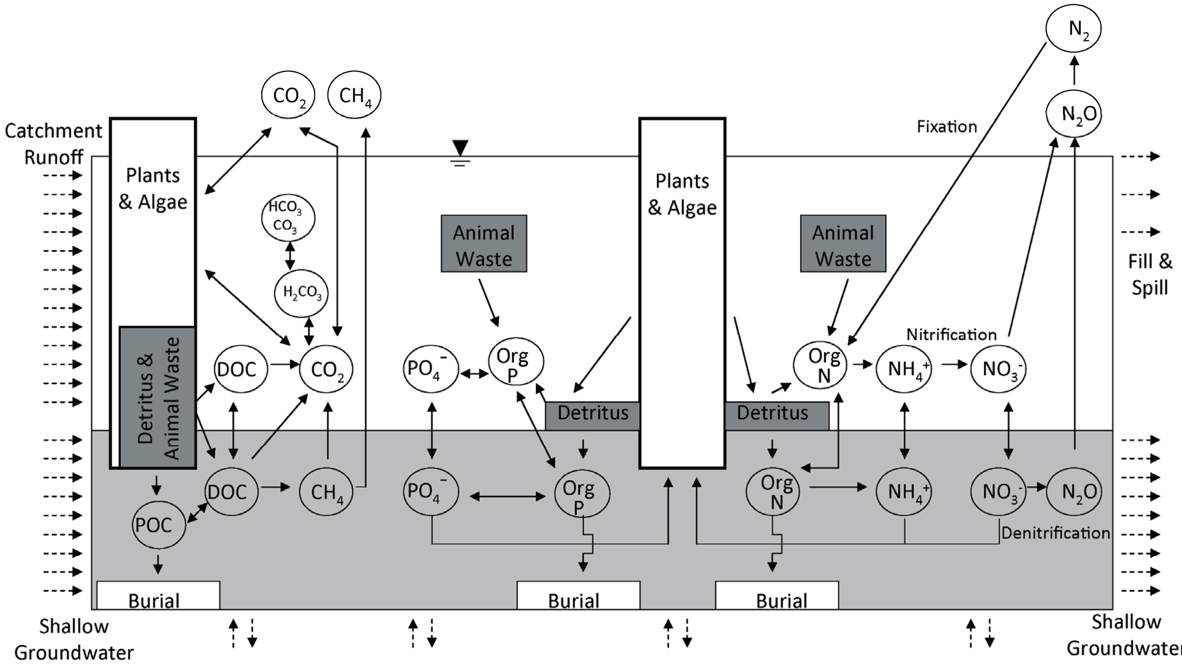


Figure 1.3 Generalized diagram of nitrogen, phosphorus, and carbon transformations and fluxes in a typical prairie wetland. Org is organic.

Microorganisms, emergent, submersed and floating macrophytes, as well as algae take up N and P from pothole waters and sediments. Nutrient concentrations in pothole vegetation tend to be highest early in the growing season and decrease as the plants mature and senesce (Johnston, 1991). The macrophyte N uptake efficiency in relatively stagnant water can be predicted as floating > submersed > floating-leaved > emergent (Birgand et al., 2007). Submersed macrophytes can obtain nutrients from the sediment in which they are rooted as well through adventitious roots growing in the water column, whereas emergent plants obtain most of their nutrients from the sediment (Kalff, 2002). Rooted macrophytes can be thought of as nutrient pumps that remove stored nutrients from sediment then return them to the water column via decomposition (Neely and Baker, 1989). During decomposition, nutrients are initially leached from macrophytes immediately following death and inundation. Up to 30% of the nutrients in litter were released by leaching within 2 days of re-flooding a Manitoba marsh after a dry period (Murkin et al., 2000). This rapid leaching can contribute to peaks in pothole nutrient concentrations. Following leaching, N and P accumulate within the litter which functions as a short term sink during microbial breakdown and long term sink as undecomposed (refractory) N and P remain in the litter after the conversion to soil (Neely and Baker, 1989; Birgand et al., 2007).

Although N and P are essential nutrients for plant production, excessive amounts in potholes can degrade water quality and lead to algae blooms. High nutrient concentrations in surface waters is one of the most serious water quality problems facing western Canada (Schindler and Donahue, 2006; Saskatchewan Watershed Authority, 2007a). Algal blooms cause decreased light penetration, reduced DO, and ultimately, eutrophication (Johnston, 1991). Many prairie lakes and potholes are eutrophic and N-limited, largely due to naturally high input from the drainage area (Barica, 1987; Anderson, 1988). Since high concentrations of N can be toxic to aquatic organisms, livestock, and people (Saskatchewan Watershed Authority, 2007b), a number of water quality guidelines at the National and Provincial level have been set. The Canadian Council of Ministers of the Environment (CCME) Canadian Environmental Quality guideline for the protection of aquatic life for nitrate is 2.9 mg NO_3^--N/L (CCME, 2003). The guideline for NH_3 + NH_4^+ is greatly affected by temperature and pH: within typical prairie pothole conditions the guideline ranges from 18.5 mg NH_3-N/L (0^oC, pH = 7.0) to 0.07 mg NH_3-N/L (15^oC, pH = 9.0) (CCME, 2003). The Saskatchewan Watershed Authority (2007b) objective for the water quality index for the Lake Stewardship Program is 0.1 mg TP/L.

1.4.3 Carbon Dynamics

Dissolved inorganic carbon (DIC) buffers freshwaters against rapid changes in pH (Figure 1.3) and is affected by photosynthetic and respiratory activity of aquatic organisms. DIC is the sum of all carbon present as carbon dioxide (CO_2), carbonic acid (H_2CO_3), bicarbonate (HCO_3^-), and carbonate (CO_3^{2-}) (Wetzel and Likens, 2000). The relative proportion of inorganic carbon (C) species in solution is related to pH such that HCO_3^- is the dominant species for pH conditions between 6.5 – 10 and the proportion of CO_3^{2-} increases for pH > 8.5 (Stumm and Morgan, 1970). Prairie potholes are typically slightly acidic to alkaline (Driver and Peden, 1977; Detenbeck et al., 2002) so most of the DIC would be expected to be in the HCO_3^- form. Respiration adds CO_2 to an aquatic system and reacts with the water to yield carbonic acid (H_2CO_3), which subsequently disassociates and produces H^+. At the same time, any CO_3^{2-} that is present consumes H^+ and yields HCO_3^-. In the absence of CO_3^{2-}, OH^- neutralizes the added H^+ (Kalff, 2002). The concentration of H^+ remains nearly constant, and thus pH remains relatively stable until the available supply of HCO_3^- and CO_3^{2-} are exhausted (Wetzel and Likens, 2000). Algae and submersed aquatic macrophytes require carbon for high sustained growth; CO_2 is the form of DIC most readily utilized (Wetzel, 2001).

Organic C in natural waters consists of dissolved organic C (DOC) and particulate organic matter (POC). Operationally, POC is defined as the amount of organic matter retained at the 0.5 μm size level (Wetzel and Likens, 2000). DOC is a highly variable and heterogeneous composition of fulvic acids, humic acids, and organic matter in various stages of decomposition (Lampert and Sommer, 2007). Organic C in the pothole water column can be exchanged to the atmosphere, sediment-interstitial water, and living and dead biomass through biogeochemical processes (Figure 1.3). Organic C is released from photosynthetically fixed C upon cell death (Bertilsson and Jones, 2003). It is supplied to aquatic ecosystems from both the surrounding terrestrial ecosystem (allochthonous) and internal, within wetland, (autochthonous) sources (Findlay and Sinsabaugh, 2003). Allochthonous inputs are generally dominated by the advective transport of surface water or groundwater travelling through or over soil and litter (Aitkenhead-Peterson et al., 2003). Factors that can increase allochthonous DOC inputs to lakes or potholes include large catchment/pond area ratio, steep catchment slopes, barriers to infiltration, high antecedent moisture conditions, increased vegetation cover, plant litter, high pH, high DOC content in the soil, and low soil retention capacity due to high clay content (Aitkenhead-Peterson et al., 2003). DOC serves as an energy source, attenuates UV solar radiation, protecting protect aquatic organisms, and alters contaminant toxicity and nutrient availability (Williamson et al., 1999). A large fraction of allochthonous C is generally less available and quite resistant to further microbial degradation due to a typically lengthy exposure to microbial decomposition and transformation on land (Kalff, 2002). Autochthonous C is more labile and originates from excretion by living organisms, cell breakdown (autolysis), herbivore grazing, and microbial decomposition of dead organisms (Lampert and Sommer, 2007).

DOC can be lost from the water column due to coagulation and flocculation, and subsequent settling (Molot and Dillon, 1996; von Wachenfeldt and Tranvik, 2008). DOC fluxes to the sediment-interstitial water column also occur by diffusion when concentrations are greater in the pothole water, which typically occurs from mid-July until pond freeze-up; however, the rate of diffusion is quite small and this process is likely relatively unimportant

(Waiser, 2006). Heterotrophic aerobic bacteria in the water column and anaerobic bacteria in pothole sediments degrade soluble organic C to CO_2 and CH_4, respectively; sedimented, relatively insoluble organic C compounds, become buried in anaerobic sediments where rates of degradations are low (Wetzel, 2001).

Allochthonous DOC (synthesized within the drainage basin) is typically more aromatic (coloured) than autochthonous DOC (synthesized in the pond) (Waiser and Robarts, 2000; Kalff, 2002). Coloured DOC attenuates light which regulates the depth at which phytoplankton photosynthesis can occur and protects aquatic organisms from the harmful effects of UV radiation (Kalff, 2002). Autochthonous C in prairie lakes and potholes undergoes significant photochemical degradation that lowers the aromaticity and molecular weight of DOC with increased residence time (Waiser and Robarts, 2000; Waiser and Robarts, 2004). In contrast with freshwater lakes in humid regions, DOC concentrations in prairie lakes and potholes tend to be high (Table 1.3) and increase with salinity, a proxy estimate of water residence time (Curtis and Adams, 1995; Waiser, 2006). For example, freshwater potholes in the St. Denis National Wildlife Area, Saskatchewan that lost most of their water via infiltration to the pothole margin had lower DOC concentrations than saline ponds that lost most of their water by evaporation (Waiser, 2006). From the same study, 68% of DOC mass present in a seasonally ponded pothole was accounted for by spring runoff, whereas no significant increase in DOC mass occurred in a permanently ponded pothole that had a small increase in volume due to spring runoff. DOC in potholes can also vary temporally. For example, Waiser (2006) found DOC concentrations increased by up to a factor of 3 from spring to fall in a St. Denis pothole that lost most of its water by evaporation.

Table 1.3 Concentrations of DOC in PPR; [1]seasonal variation and [2]multiple water bodies sampled.

Location	DOC (mg/L)	Data Source
St-Denis, SK	19.7– 02.7[1,2]	Waiser, 2006
Aspen parkland/grassland prairie transition, AB	22–330[2]	Curtis and Adams, 1995
Central Saskatchewan	4.1–156.2[2]	Arts et al., 2000
Woodworth, ND (native prairie)	25–130[1]; 14–42[1]	Detenbeck et al., 2002
Woodworth, ND (non-native prairie, tilled)	19–73[1]; 16–45[1]	Detenbeck et al., 2002

1.4.4 Major Ions

The composition of major ions, termed salts, can be used to classify wetland types (Driver and Peden, 1977), provide insight into potential sources of water (LaBaugh et al., 1987), and be used as an indicator of water quality in combination with physical, biological, and other chemical descriptors (Meybeck, 2005). Driver and Peden (1977) examined ion dominance patterns, based on equivalent concentrations, for different pothole permanence classes located in Manitoba and Saskatchewan and found ionic dominance patterns in temporarily ponded wetlands were $Ca^{2+} > K^{+} > Mg^{2+} > Na^{2+}$ / $HCO_3^{-} > Cl^{-} > SO_4^{2-}$, in semi-permanently ponded wetlands potholes were $Mg^{2+} > Ca^{2+} > Na^{+} > K^{+}$ /$SO_4^{-} > HCO_3^{-} > Cl^{-}$, while permanently ponded ones were $Ca^{2+} > Mg^{2+} > Na^{+} > K^{+}$ / $HCO_3^{-} > SO_4^{2-} > Cl^{-}$. The general pattern of ionic dominance did not change during the seasons, however salt concentrations and salinity, measured as conductivity, did vary and the greatest variation was found in semi-permanently ponded wetlands. Results from other studies are contradictory with regards to static ionic dominance patterns. For example, Detenbeck et al. (2002) observed variations in ionic dominance patterns of seasonally ponded potholes and concluded that the variations were not due to experimental catchment treatments (native prairie, restored prairie, tilled summer fallow) but instead were influenced by catchment position and the length of groundwater flowpaths. However, the trend of seasonal increases in salinity was similar. These seasonal increases were deemed to be controlled primarily by the semi-arid climate of the Canadian prairies, where evaporation exceeds summer precipitation. Seasonally, evaporation increases salinity (Table 1.4), which concentrates soluble salts (Rózkowska and Rózkowski, 1969) and causes precipitation of less soluble salts (Holland, 1978). As the concentration of salts increases, potholes become saline (with chlorides) or alkaline (with sulfates). Many semi-permanently and permanently ponded potholes are characterized by high salinity (Stewart and Kantrud, 1971; Millar, 1976; Driver and Peden; 1977). Some researchers (e.g. LaBaugh et al., 1987; van der Kamp and Hayashi, 2009) believe this indicates a transport of dissolved salts via groundwater and surface water flowpaths and their subsequent evapoconcentration. 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^{2-}/L and 1000 mg Ca^{2+}/L) is not recommended.

Table 1.4 Salinity characteristics of prairie potholes: [1]seasonal range and [2]multiple water bodies sampled.

Location	Type	Salinity	Units	Data Source
St-Denis, SK	Potholes	312 – 33,493[1,2]	µS/cm	Waiser, 2006
Woodworth, ND (native prairie)	Potholes	160 –1720[1,2]	µS/cm	Detenbeck et al., 2002
Woodworth, ND (non-native prairie, tilled)	Potholes	260 – 2710[1,2]	µS/cm	Detenbeck et al., 2002
Central SK	Potholes and lakes	270 – 74300[2]	µS/cm	Arts et al., 2000
Cotton Wood Lake area, ND	Potholes	110 – 7140[1,2]	µS/cm	Labaugh et al., 1987
Southwestern MB	Lakes	305 – 7837[1,2]	µS/cm	Barica, 1975
Moose Mountain, SK	Potholes and lakes	120 – 129144[1,2]	ppm	Rózkowska, & Rózkowski, 1969
Central & southern SK	Lakes	35 – 118000[1,2]	ppm	Rawson, 1944

1.4.5 Microbiological Parameters

All surface waters contain a variety of bacteria and most are considered benign to human health. However, some bacteria found in animal feces, such as *Escherichia coli* (*E*. coli), including the virulent 0157:H7 strain, are a concern because humans and livestock can contract diseases through direct contact with contaminated water (Miller, 2001). Coupled measures of *E*. coli and total coliforms (T. coli) serve as indicators for the presence of enteric pathogens found in fecal pollution (Federal-Provincial Working Group on Recreational Water Quality of the Federal-Provincial Advisory Committee on Environmental and Occupational Health, 1992).

A major non-point source of disease causing bacteria in agricultural landscapes is runoff containing animal wastes from pastures or fields fertilized with manure (Hyland et al., 2003). Bacterial densities in surface waters frequently peak during spring runoff (Ontkean et al., 2003) and increase following precipitation events (Hyland et al., 2003; Gannon et al., 2005) as water flows from upland areas to surface water bodies. Factors influencing bacterial contamination include topology of the landscape as well as farm specific practices such as grazing, access of livestock to streams, the amount and time of year of manure applications, and the length of time between fertilizer applications and precipitation events (Gannon et al., 2005). For example, Depoe and Westbrook (2003) found stream water coliform densities were significantly higher for Alberta watersheds with moderate or high compared to low agricultural intensity. Potholes can be further contaminated by semi-aquatic mammals and waterfowl (Hyer and Moyer, 2004). Removal mechanisms for bacteria in potholes include sediment retention and natural die-back (Hemond and Benoit, 1988; Auer and Niehaust, 1993). Fecal coliforms tend to concentrate in sediment where they survive longer, potentially due to the greater organic matter present in the sediment (Karim et al., 2004). The CCME recreational water quality guideline for indicator bacteria is 200 fecal coliforms or 200 *E*. coli per 100 ml of sample, based upon the average of at least five samples. The CCME water quality guidelines for the protection of agricultural water is 100 *E. coli* per 100 ml of sample for crop irrigation, and is animal-specific for livestock watering, but generally should be under 2 *E*. coli per 100 ml of sample.

1.4.6 Snowmelt Chemistry

As snow and snowmelt runoff account for such a large proportion of the prairie pothole water budget, their water quality characteristics have the potential to significantly influence pond water quality. Ions become entrapped in ice crystals as snow particles form and fall, and ions continue to accumulate on the snowpack due to dry deposition (DeWalle, 1989). With snowpack metamorphosis and re-freezing, ions become excluded because they lack the ability to incorporate into the crystal lattice of ice and thus become concentrated in the quasi liquid layer (Colbeck, 1981). As a result, ion concentrations in this layer often exceed concentrations in the parent snowpack (Johannessen and Henriksen, 1978) and the first pulse of snowmelt can be responsible for flushing the majority of solutes out of snowpacks (DeWalle, 1989). Lab experiments have shown that during snowmelt, there is an initial enrichment, followed by a rapid decrease in ion concentration as melt progresses, until the meltwater is depleted relative to the parent snowpack (Johannessen and Henriksen, 1978; Lilbæk and Pomeroy, 2007). Ion exclusion has also been found to take place during basal ice formation in the laboratory (Lilbæk and Pomeroy, 2008). A basal ice layer can form if the melt rate exceeds the infiltration rate, leading to ponding at the base of the snowpack, and if soils are sufficiently cold and moist (Woo and Heron, 1981); conditions typical of the prairies (Fang et al., 2007). Rain on snow events can lead to lower enrichment rates due to decreased contact time with the snow and a greater proportion of free water; whereas the refreezeing and ion exclusion that occurs during melt-freeze-cycles enhance meltwater enrichment (Colbeck, 1981; Marsh and Pomeroy, 1999).

Snowmelt runoff can also contain significant amounts of nutrients because microbiological activity persists beneath snowpacks at temperatures > -7 or -8 ^{o}C. This activity can lead to the mineralization of N and P under snow and a potential nutrient pulse at the onset of melt (Devito et al., 1999; Jones, 1999). However, losses of NO_3^- and NH_4^+ have also been observed in snowpack runoff relative to the parent snowpack, due to microbial utilization and their rate of uptake is increased in wet conditions (i.e. rain of snow events and ripe snowpacks). Gains also occur and are attributed to the degradation of organic matter, leaching, and/or ion exchange mechanisms (Jones, 1989). Lilbæk (2009) measured a relative enrichment of the concentrations of DOC and most major ions in snowmelt water that has been in contact with a frozen organic layer (forest detritus) in a laboratory experiment, however concentrations of NO_3^- and Mg^{2+} decreased.

The chemical composition of meltwater can also change as it flows through soils due to mixing with the soil matrix water and interaction with the soils (Quinton and Pomeroy, 2006). These changes are influenced by the chemical composition of the soil, weathering, micro-organisms, nitrogen cycling, carbonate equilibrium reactions, and redox reactions (Ollier, 1984; Anderson, 1988). Thus, snowmelt solute inputs to potholes would be expected to be greatest where soil infiltration capacity is low and there is contact with the soil.

1.5 Drainage Characteristics

Historically, agricultural practices have led to wetland drainage: over the past century, 40 – 70% of the wetlands located in the western prairies have been drained to increase agricultural production (Tiner, 1984; Dahl, 1990; Brinson and Malvarez, 2002; Watmough and Schmoll 2007). Recently, there have been renewed efforts to drain potholes (Watmough and Schmoll 2007), especially in the Canadian Prairie Provinces. Potholes are being drained to increase agricultural production and to reduce the costs of farming around potholes (Scarth, 1998; Brinson and Malvarez, 2002; Cortus et al., 2010). If wetland drainage does not completely eliminate the wetland it will largely decrease the time and depth of inundation. Temporary wetlands are the most common permanence class in the PPR, and the most impacted by drainage and farming practices (Euliss et al., 2001). Wetland loss due to land use conversion causes direct habitat loss, where the ecological functions discussed previously are lost. The magnitude of functional loss is not proportional to wetland size, meaning that the loss of area comprised of multiple small isolated wetlands may be more significant than the loss of the same area comprised of large ones (Trochlell and Bernthal, 1998). King (1998, cited in Leibowitz, 2003) illustrated that cumulative wetlands losses can affect biodiversity in a nonlinear fashion (i.e. a small regional loss of isolated wetlands caused a sharp decrease in biodiversity).

Pothole drains are typically narrow (1 to 3 m) with a rectangular cross sectional area and low sinuosity (i.e. very straight). 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 regularly contribute to streamflow. Figure 1.4 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, drains can transport water from one wetland to another, which can cause local flood damage to agricultural crops or communities surrounding the terminal wetland. A drainage ditch can also transport surface water runoff directly from a wetland to a stream. Drainage of many potholes has been found to have 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%. Similarly, a Saskatchewan Watershed Authority (2008) assessment identified that agricultural drainage contributed to the high water levels in terminal (Waldsea) and near-terminal (Deadmoose, Houghton, Fishing) lake basins in 2007 resulting from increased runoff due to increased effective drainage areas. A larger study, of which this study is part, also shows that Smith Creek subbasins experiencing greater wetland drainage indeed have poorer water quality (Westbrook et al., 2011).

It has been hypothesized that stream water quality would be adversely affected by man-made drainage ditches that connect isolated potholes to streams (Leibowitz and Vining, 2003; Whigham and Jordan, 2003). However, to date there are not field studies to support or refute this conjecture. Considerable field-based research has been directed toward the study of upland agricultural drainage ditches that are used to channel runoff away from agricultural fields (Vadas et al., 2007). However, these studies often analyze solute transport along well established ditches, as opposed to newly constructed ditches. Study of a newly constructed ditch is also important because the initial solute export may be substantial given the high solute storage potential in wetlands as well as the lack of vegetation and freshly exposed soil along the new ditch, factors key in solute uptake/release. Preceding works may prove useful to understanding solute losses from potholes drainage, with the caveat that researchers have focused primarily on nutrients and coliform rather than including other water quality descriptors in their works. A review of findings from upland drainage ditch and low order agricultural stream studies is presented in the following paragraphs.

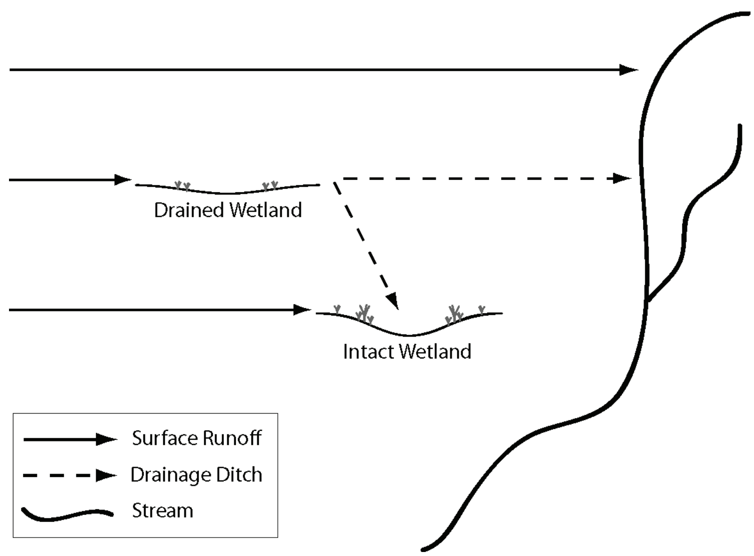


Figure 1.4 Surface runoff pathways. Runoff can enter streams directly or enter and be stored in wetlands. Stored runoff can be released from wetlands via drainage ditches and flow either into other wetlands or streams.

The water regime in pothole drains is characterized by the advective flow of water downstream and a greater potential inward flux of oxygen due to turbulent mixing. Since solute sedimentation rates vary inversely with the velocity of flowing water (Julien, 2002), relatively lower sedimentation rates are anticipated in upland drainage ditches compared to in potholes. However, 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 aluminum, iron, and carbonate, 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, 2007). Other researchers (Needleman et al., 2007) have shown macrophytes and algae can also temporarily store nutrients. However, they eventually die due to seasonality, water level drops, or other causes and can subsequently contribute organic matter to the water column and the accretion of sediment. Macrophytes can also play an indirect role in solute retention by reducing flow velocities and re-suspension rates, as well as increasing sedimentation (Birgand, 2007).

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 DO concentrations, which would be expected to 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 N to receiving streams since NO_3^- is quite mobile compared to NH_4^+, which is easily adsorbed to negatively charged ditch surface particles (Strock et al., 2007).

1.6 Current Research Gap and Thesis Objectives

Wetland water quality is expected to be an important control of that in drainage water. Previous studies have documented select water quality variables, often in relation to individual driving factors (Rözkowska & Rözkowski, 1969; Driver and Peden, 1977; Miller et al., 1985; Labaugh et al., 1987; Swanson et al., 1988; Neely and Baker, 1989; Detenbeck et al., 2002; Waiser, 2006), however, a concurrent assessment of both nutrients and salts in potholes is warranted given that differing control mechanism interact and drive nutrient and salt concentrations (LaBaugh et al., 1987; Wetzel, 2001). Factors such as permanence and land cover type also do not act in isolation and could instead interact with one another to regulate wetland water quality.

The intensification of agriculture has led to substantial wetland drainage throughout the prairies (Tiner, 1984; Dahl, 1990; Brinson and Malvarez, 2002; Watmough and Schmoll

2007). Recent studies have mainly focused on the effects of wetland drainage on downstream flooding (Saskatchewan Watershed Authority, 2008; Yang et al., 2008; Fang et al., 2010). However, potholes have been shown to act as sinks for non-point loads due to their lack of permanent surface outflow (Neely and Baker, 1989; Crumpton and Goldsborough, 1998) and although no field-based studies have been conducted, there is a popularly held belief that the creation of artificial connections (i.e. ditches) between prairie potholes and downstream surface waters negatively impacts water quality due to the export of previously stored solutes. 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 degradation of the downstream ecosystem.

Thus, the objectives of this thesis are to:

1) characterise the spatial variation in water quality of prairie potholes following snowmelt;
2) identify factors influencing temporal patterns in wetland water quality;
3) quantify solute export from a newly constructed drainage ditch; and
4) compare solute export along artificial ditches and natural spills.

To meet objective 1, water quality in 67 pothole wetlands representing different land cover types and permanence classes was measured. To meet objective 2, intensive temporal measures of water quality in one permanently ponded wetland were taken. To meet objective 3, a wetland drainage experiment was conducted whereby the wetland with intensive temporal measures of water quality was ditched and the water quality along the newly constructed ditch was characterized during the time taken to drain the wetland. Since potholes can also naturally fill and then spill (Leibowitz and Vining, 2003; Spence, 2006), comparison of water quality were made along natural and artificial pothole connections to meet objective 4.

2.0 METHODS

2.1 Study Design

2.1.1 Study Sites

Research was conducted at Smith Creek watershed (50°50'4"N 101°34'48"W, Figure 2.1 inset), which is located within the PPR in southeastern Saskatchewan. The watershed is ~445 km^2 with a highly variable effective contributing area (Pomeroy et al., 2009). The contributing area is continuously increasing as farmers drain more potholes to increase agricultural production (Figure 2.1). The terrain is level to undulating and rolling. Smith Creek watershed is located in the Aspen Parkland Continental Prairie Wetland subregion (National Wetlands Working Group, 1988). Soils in the region are a mixture of Black (Oxbow) and Thick Black (Yorkton) chernozems formed in loamy glacial till (Agriculture and Agri-Food Canada et al., 2009). Geological Survey Hydrogeochemical maps show that the groundwater in surficial deposits at Smith Creek watershed are HCO_3^-, Mg^{2+}, and Ca^{2+} dominated with specific conductivity likely below 640 μS/cm (Officers of the Geological Survey of Canada, 1967). These groundwater characteristics are similar to those noted by Barica (1978) in the Erikson lakes region, 120 km southeast of Smith Creek watershed.

The regional climate is semi-arid. The mean monthly temperatures are -17.9 °C in January and +17.8 °C in July, measured at the Yorkton airport, ~50 km west. The mean (1942 – 2009) annual precipitation is 438 mm of which 121 mm occurs mostly as snow in November to April (Environment Canada, 2009). Precipitation prior to the 2008 study period was in the 53rd, 34th, and 26th percentile for winter 2007, summer (May – October) 2007, and winter 2008, respectively. Summer 2008 and winter 2009 precipitation amounts were in the 60th and 66th percentile, respectively. 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.

Proportions of land use were determined by Fang et al. (2010) and Guo et al. (in press) using unsupervised classification of SPOT 5 images from October 1, 2008. The dominant land use is agriculture, occupying 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 the Smith Creek watershed 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, that at average surface water level have no surface inflows or outflows.

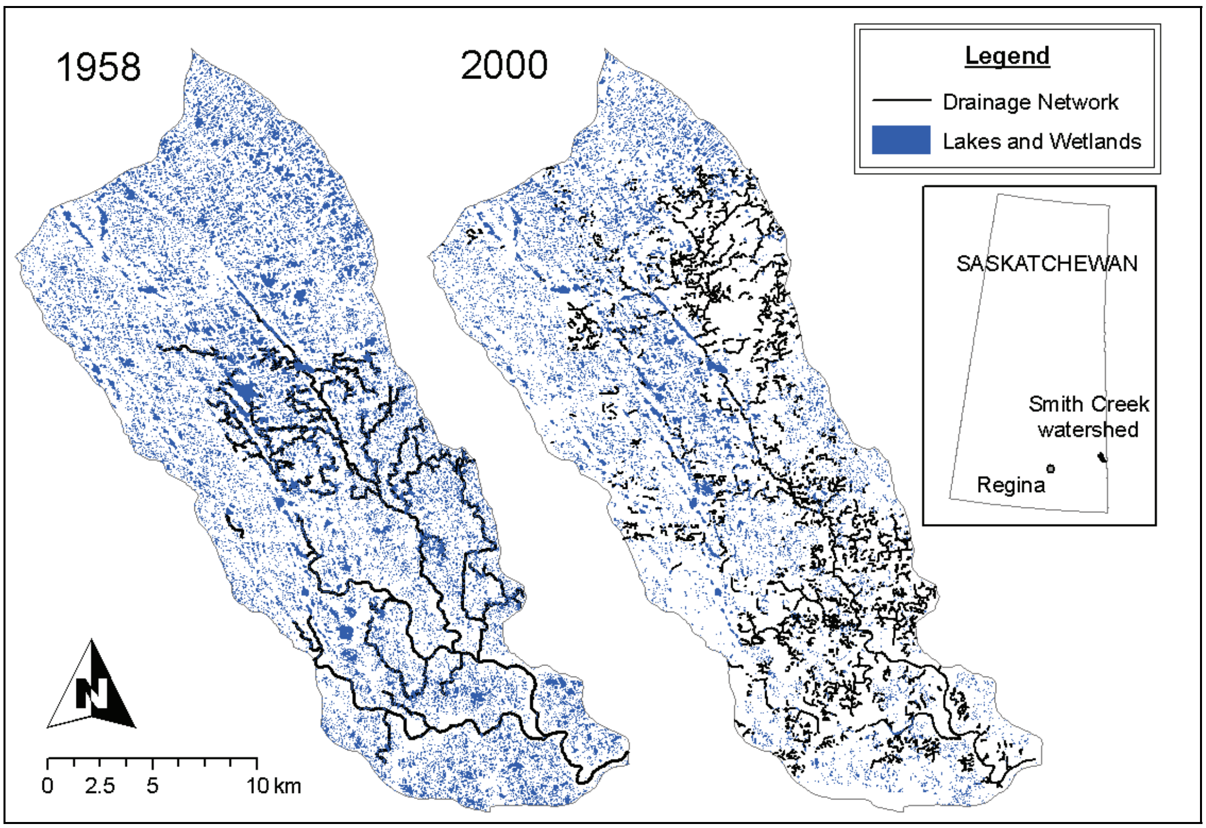


Figure 2.1 Historic (1958) and current day (2000) distribution of the drainage network, lakes, and wetlands at Smith Creek watershed. Produced by Logan Fang in conjunction with Ducks Unlimited Canada. Inset: Smith Creek watershed within Saskatchewan, Canada.

2.1.2 Spatial Variation in Wetland Water Quality

Water quality in 67 wetlands (Figure 2.2) 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 2.1), as well as on accessibility. Upland land cover classes were crop, wood, and grass while pond permanence classes were seasonal, semi-permanent, and permanent. The wetlands sampled were located in eight different soil units with various quantities of Oxbow (Ox), Yorkton (Yk), Whitewood (Wh), and Whitesand (Ws) soils (Figure 2.2) (Agricultre and Agri-Food Canada, 2009). Pond permanence classes were determined using a combination of the vegetation structure of the pothole, as per Millar (1976) and Stewart and Kantrud (1971), in combination with observations of the presence or absence of surface water recorded on air photos (October 26, 1959 and May 31, 2001), and SPOT 5 imagery (July 5, 2007. The approach used to classify pond permanence is summarized in Figure 2.3. The dominant vegetation type located in the centre of the wetlands was identified August 11-14, 2009. Sampling was conducted in spring prior to the anticipated drying of seasonally ponded wetlands. Water samples were collected once during May 19 – 21, 2009 from roughly the deepest point in the wetland, which is typically the centre of the ponds, at the midpoint in the water column.

In addition to classifications based on surrounding land cover and pond permanence, wetland locations along the Smith Creek surface drainage network were determined (Figure 2.4). Fang et al. (2010) used an automated basin delineation technique, "TOPAZ" (Garbrecht and Martz, 1993, 1997), to extract sub-basins and drainage network channels based on a LiDAR DEM of the Smith Creek drainage basin. The DEM was resampled to 50 m to provide a more computational efficient input for the TOPAZ program. TOPAZ processing parameters included a minimum 5 ha threshold for an upstream drainage area below which a source channel is initiated and maintained and a minimum of 100 m for an acceptable length for the source channel to exist. Spatial variations in wetland water quality were compared among wetlands located along a potential fill and spill sequence and in different TOPAZ sub-basins. Due to limitations caused by the extent of the LiDAR DEM only 35 wetlands were included in the TOPAZ analysis.

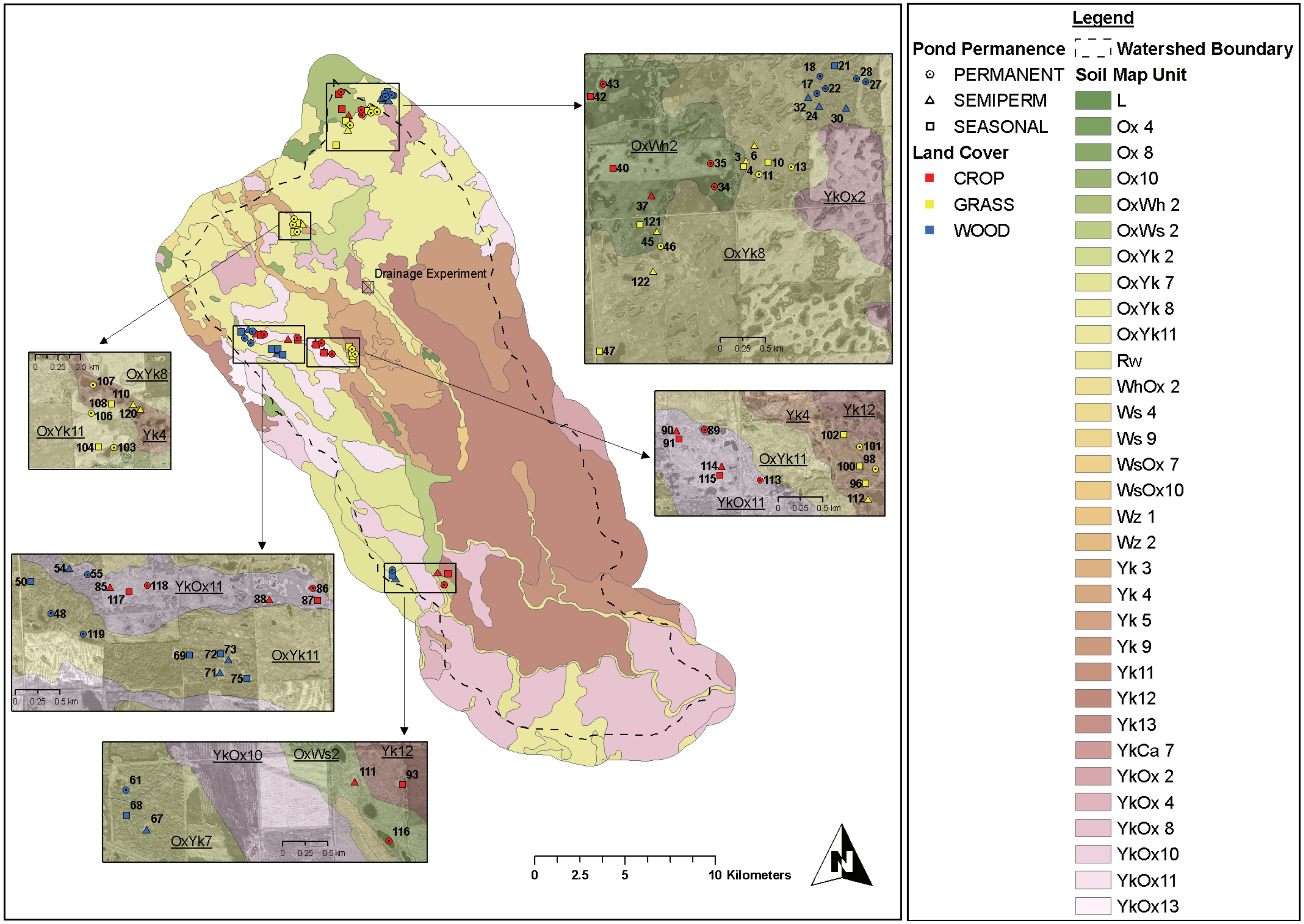


Figure 2.2 Smith Creek watershed, Saskatchewan, Canada showing the location and ID numbers of wetlands studied in relation to Agriculture and Agri-Food Canada soil map units. Semiperm is semi-permanently ponded wetlands.

Table 2.1 Number of wetlands sampled in each land cover and permanence class.

Land Cover Class	Permanence Class: Seasonal	Semi-permanent	Permanent	Total
Crop	7	6	8	21
Grass	9	7	8	24
Wood	6	7	9	22
Total	22	20	25	67

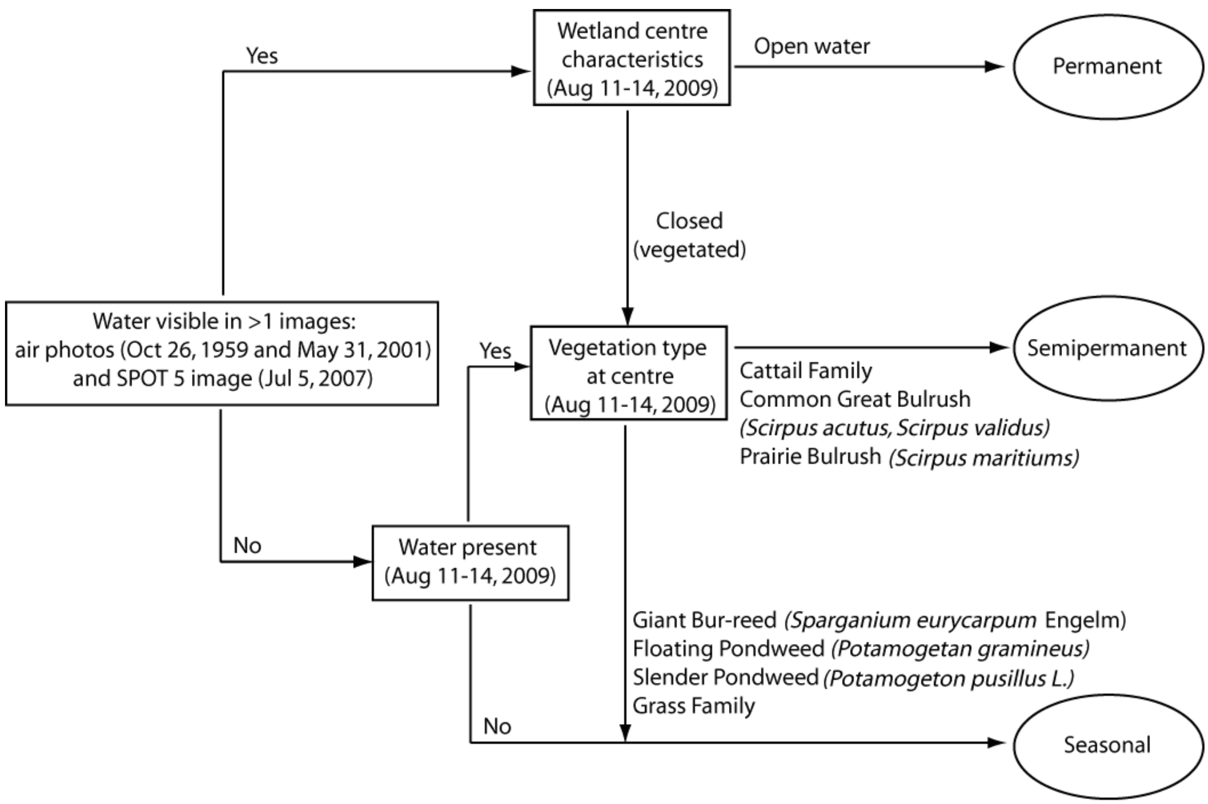


Figure 2.3 Dichotomous key used to determine wetland pond permanence. Plant species listed indicate dominant vegetation at the wetland centre.

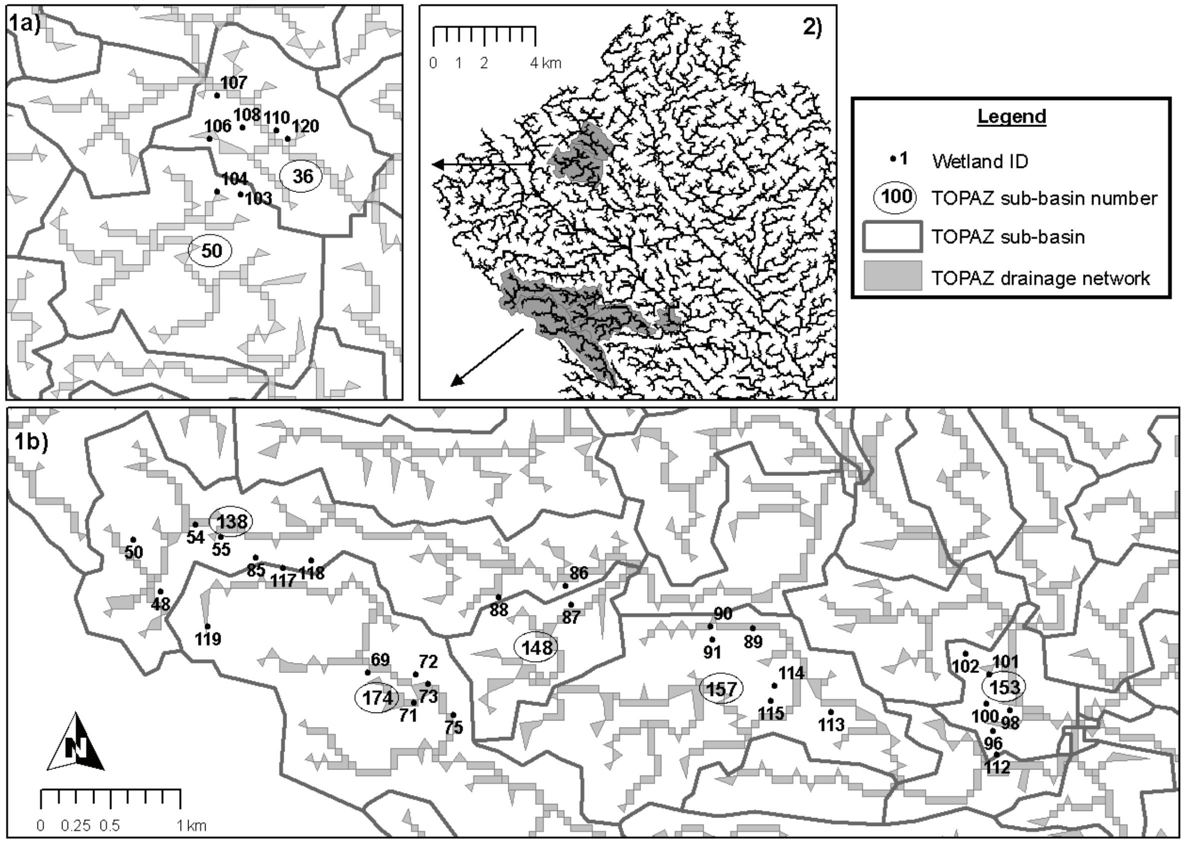


Figure 2.4 TOPAZ drainage network, TOPAZ sub-basins, and wetlands sampled in 1a) sub-basins 50 and 36, and 1b) sub-basins 138, 148, 153, 157, and 174; 2) Smith Creek TOPAZ drainage network and TOPAZ sub-basins containing wetlands sampled. A potential fill and spill sequence is located in sub-basin 138 and low-gradient connections may form in sub-basins 153 and 157.

2.1.3 Wetland Drainage Experiment

The permanently ponded wetland (LR3) selected for the drainage experiment was an ideal choice as there was detailed hydrological information for it prior to its drainage (Minke et al. 2010). Also, the landowner was keen to drain it and the drain was expected to connect to Smith Creek. Water samples were collected weekly at the centre of wetland LR3 from April 18 to October 22, 2008, prior to its drainage. Water level was measured hourly using a PT2X pressure transducer (*Northwest Instrumentation Inc*.) located near the wetland perimeter. 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) and connect it to a down-gradient wetland, November 19 – 20, 2008. At the time of drainage wetland LR3 was covered with ~8 cm of ice. Water sample collection began November 20, 2008 one hour after the ditch was completed. The initial sampling instance was delayed to maintain a safe distance from the excavator and due to attempts to remedy nonfunctional flow gauging equipment afflicted by the cold air temperature (-15 ^{o}C). Three additional sets of samples were collected 4 hr, 6 hr, and 23 hr 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, as measured from the wetland edge (Figure 2.5) at the midpoint in the water column.

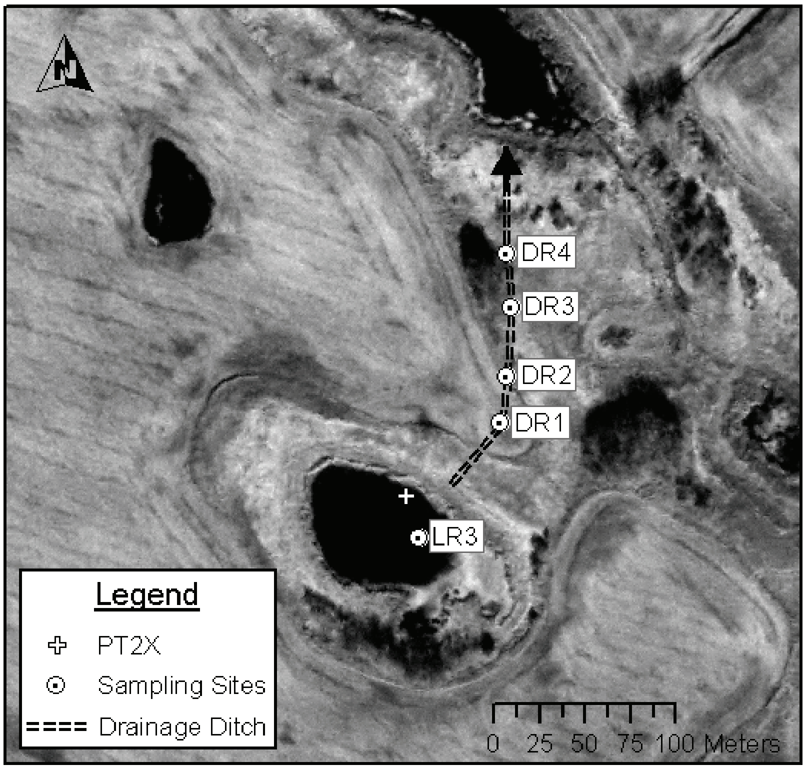
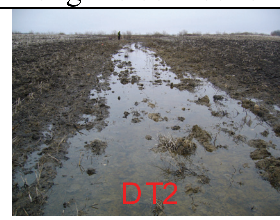


Figure 2.5 The wetland drainage experiment sample site in the LR3 wetland and sites along the newly constructed drainage ditch (DR) as well as the location of the water level recorder (PT2X).

2.1.4 Comparing Artificial Ditches to Natural Spills

Water quality along seven artificial drainage ditches (i.e. ditches, abbreviated DT) and five natural connections (i.e. spills, abbreviated SP) (Table 2.2) was compared. The ditches and spills selected for the study drained wetlands that did not have any surface water inflows, with the exception of DT3. DT1 drains into the wetland drained by DT3, however, at the time of sampling, DT1 was snow covered and not flowing. Due to the variation in ditch and spill length, locations for water sampling along the ditch varied (Table 2.2). Samples were collected April 10 – 18, 2009. Water samples were collected from the thalweg of the connection, starting at the most downstream sample location at one-half water depth. 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 calculated as the sum of the product of velocity and average stream depth at each sample interval.

Table 2.2 Photographs of artificial ditches (DT) and natural spills (SP) studied that drain wetlands at Smith Creek watershed and means of physical properties measured along the connections: discharge (Q), velocity (v), depth (d), width (w), water temperature (T). 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 T: 6.5 ^{o}C			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 T: 6.7 ^{o}C
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 T: 4.9 ^{o}C			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 T: 4.1 ^{o}C
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 T: 7.2 ^{o}C			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 T: 5.1 ^{o}C
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 T: 2.9 ^{o}C			Sample locations (m): 0, 5, 10 Q: 0.07 cms v: 0.01 m/s d: 0.28 m w: 22.3 m T: 5.3 ^{o}C
Sample locations (m): 0, 15, 30, 45 Q: 0.02 cms v: 0.005 m/s d: 0.16 m w: 27.1 m T: 2.8 ^{o}C			Sample locations (m): 0, 8, 15 Q: 0.006 cms v: 0.02 m/s d: 0.07 m w: 6.40 m T: 1.9 ^{o}C
Sample locations (m): 0, 9, 18, 36 Q: 0.002 cms v: 0.001 m/s d: 0.18 m w: 8.49 m T: 6.5 ^{o}C			Sample locations (m): 0, 10, 20 Q: 0.001 cms v: 0.001 m/s d: 0.10 m w: 4.57 m T: 4.4 ^{o}C

2.2 Water Sample Collection and Chemical Analysis

Water samples for salt and nutrient analysis were collected in pre-rinsed 1 L polyethylene or two 350 mL glass bottles, kept on ice during the day, and then split. Temperature-compensated specific conductance (SC) and pH were measured in the field 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°C. A third aliquot was filtered through a 0.45 μm Whatman GF/C glass microfiber filter then frozen. The HNO_3 preserved sub-sample was analyzed for total phosphorus (TP) concentration at Saskatchewan Research Council Analytical Laboratories, Saskatoon, SK (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 the following 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). Orthophosphate as phosphorus (orthoP) was analyzed at SRC colorimetrically (Standard Methods part 4500-P part E). A Westco SmartChem Discrete Analyzer (SmartChem 200, Method 375-100E-1) was used for analysis of ammonium nitrogen (NH_4^+) and nitrate plus nitrite nitrogen, reported as NO_3^-. Major ions (Cl^-, HCO_3^-, SO_4^{2-}, Na^+, K^+, Mg^{2+}, and Ca^{2+}) 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.

Water samples for coliforms, i.e. *Escherichia coli* (*E*. coli) and total coliforms (T. coli), were only collected for the wetland drainage experiment 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 SRC within 24 hours for analysis by chromogenic substrate method (Standard Methods part 9223).

The following terms are used throughout this thesis: nutrient refers to all forms of nitrogen and phosphorus; salt refers to Cl^-, HCO_3^-, SO_4^{2-}, Na^+, K^+, Mg^{2+}, and Ca^{2+}; coliforms refers to *E*. coli and T. coli; and solute refers to all parameters measured.

2.3 Data Analyses

2.3.1 Spatial Variation in Wetland Water Quality

A multivariate analysis of water quality measurements was carried out to evaluate the possible effects of pond permanence and surrounding land cover type. 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^{2-}, Na^+, Mg^{2+}, K^+, and Ca^{2+}) variable sets because the mechanisms controlling nutrient concentrations in the wetlands have been shown to be different from mechanisms controlling salt concentrations (Wetzel, 2001). Mass per volume concentrations (i.e. mg/L) were converted to milliequivalent concentrations (i.e. meq/L) for analysis of major ions. The ion charge balance (ICB) of samples was calculated as

$$ICB = \frac{\sum cations - \sum anions}{\sum cations + \sum anions}$$

The acceptable error depends on the total ion concentration of the sample and increases for low concentrations. Fishman and Friedman (1989) suggest an acceptable error of 2%, 3%, and 12% for samples with a total (cation plus anions) milliequivalent per litre value of 20, 7, and 0.9, respectively. Ion charge balances ranged from -9.8% to 25.2% and averaged 9.9%. Ion charge balance values for each wetland are included in Appendix B.

A two-way multivariate analysis of variance (MANOVA) was used to test the relationships among land cover and permanence classes for the nutrient and salinity data sets. Post hoc comparison tests (two-way ANOVAs) were used to determine which variables contributed to the occurrences of significant differences among factor classes. Significant differences among land cover and permanence classes were assessed using Tukey's Honest Significant Difference pairwise comparisons test, which is applicable to mildly unbalanced designs (Everitt and Hothorn, 2006). These 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.

Trilinear or piper plot diagrams were created using the Piper Plot macro of SigmaPlot v9.01 for the wetlands located within the TOPAZ drainage network. Piper plots are used to indicate geochemical facies, hydrogeochemical evolutions along a flow path, and mixing trends (Fetter, 2001).

2.3.2 Wetland Drainage Experiment

Wetland volumes were estimated by inputting water levels into the full volume-area-depth equation (Hayashi and van der Kamp, 2000). 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). Solute mass in the wetland was estimated by multiplying solute concentration by wetland volume estimate at the time of sampling. Changes in solute mass relative to chloride, which is impacted by hydrological processes such as evapotranspiration and dilution in the wetland but is biotically conservative, were used to indicate biotic or geochemical processing of solutes in the wetland (Heagle et al., 2007; Duff et al., 2009). Data were normalized to their May 1, 2008 mass for the comparison with chloride; at this date, nutrient concentrations in the wetland had stabilized and provided an adequate reference point.

Total loads exported from wetland LR3 along the newly constructed ditch were calculated by multiplying average solute concentration along the ditch by the change in estimated wetland volume at each sampling point. 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. For display purposes, a value of one was subtracted from these values to set the slope intercept to zero. The statistical significance of the linear relationship between normalized load and distance along the ditch was tested using the linear model (lm) function of the R statistical language and environment (R Core Development Team, 2005). The slopes of the normalized concentrations that were determined to be significantly linear were statistically compared, in a manner analogous to a *t*-test, as per Zar (2000) to the slope of biotically conservative Cl^-. Slopes were compared at each sampling instance to assess whether solutes 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.

2.3.3 Comparing Artificial Ditches to Natural Spills

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

3.0 RESULTS

3.1 Spatial Variations in Wetland Water Quality

Nutrient concentrations in the pothole wetlands studied ranged widely (APPENDIX A). TP ranged from 0.02 to 2.8 mg/L. Based on the trophic classification presented in Wetzel (2001), most wetlands studied could be classified as eutrophic, with the exception of six which could be classified as hypereutrophic (i.e. TP > 0.6 mg/L),. The majority of P was typically in the organic form, with the exception of six wetlands in cropped areas and one wetland in a wooded area 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 mass ratios of dissolved inorganic nitrogen (NO_3^- + NH_4^+) to dissolved inorganic phosphorus (orthoP) (i.e. DIN:orthoP) to the Redfield Ratio provides insight into nutrient limitations that may restrict algal growth (Rhee and Gotham, 1980; Wetzel, 2001). Based on this ratio, eight wetlands may be P limited (DIN:orthoP>12:1), 46 may be N limited (DIN:orthoP<7:1), and 13 may be limited by neither N nor P (i.e., DIN:orthoP 7 – 12:1). However, many sites had elevated nutrient concentrations and may not be limited by N or P (Dodds, 2003). DOC ranged from 19 to 55 mg/L.

Salt concentrations in the pothole wetlands studied also varied greatly (APPENDIX B). The wetlands were neutral to slightly basic (pH of 6.6 – 8.6) and ranged from fresh to brackish (SC of 57 – 1780 μS/cm). Maximum concentrations of salts were HCO_3^- = 3.8 meq/L, SO_4^{2-} = 15.5 meq/L, Mg^{2+} = 13.7 meq/L, and Ca^{2+} = 6.8 meq/L. Distinct patterns of ion dominance groups were apparent (Figure 3.1). 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^{2+} > Ca^{2+} > K^+ > Na^+. All but two of these wetlands were classed as crop or grassland; 41% and 34% of these were classed as permanently and semi-permanently ponded, respectively. The other anion dominance pattern observed was HCO_3^- > SO_4^{2-} > Cl^-, and 50% of these wetlands were also characterized by Mg^{2+} > Ca^{2+} > K^+ > Na^+. The second most common (21%) cation dominance pattern for these wetlands was Ca^{2+} > Mg^{2+} > K^+ > Na^+.

The two-way MANOVA for the nutrient variable set indicated significant differences existed among land cover types (p = 0.01) and permanence classes (p = 7×10^{-5}). However, there was no significant interaction between land cover type and pond 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 pond permanence class ($p = 0.46$) were not significant.

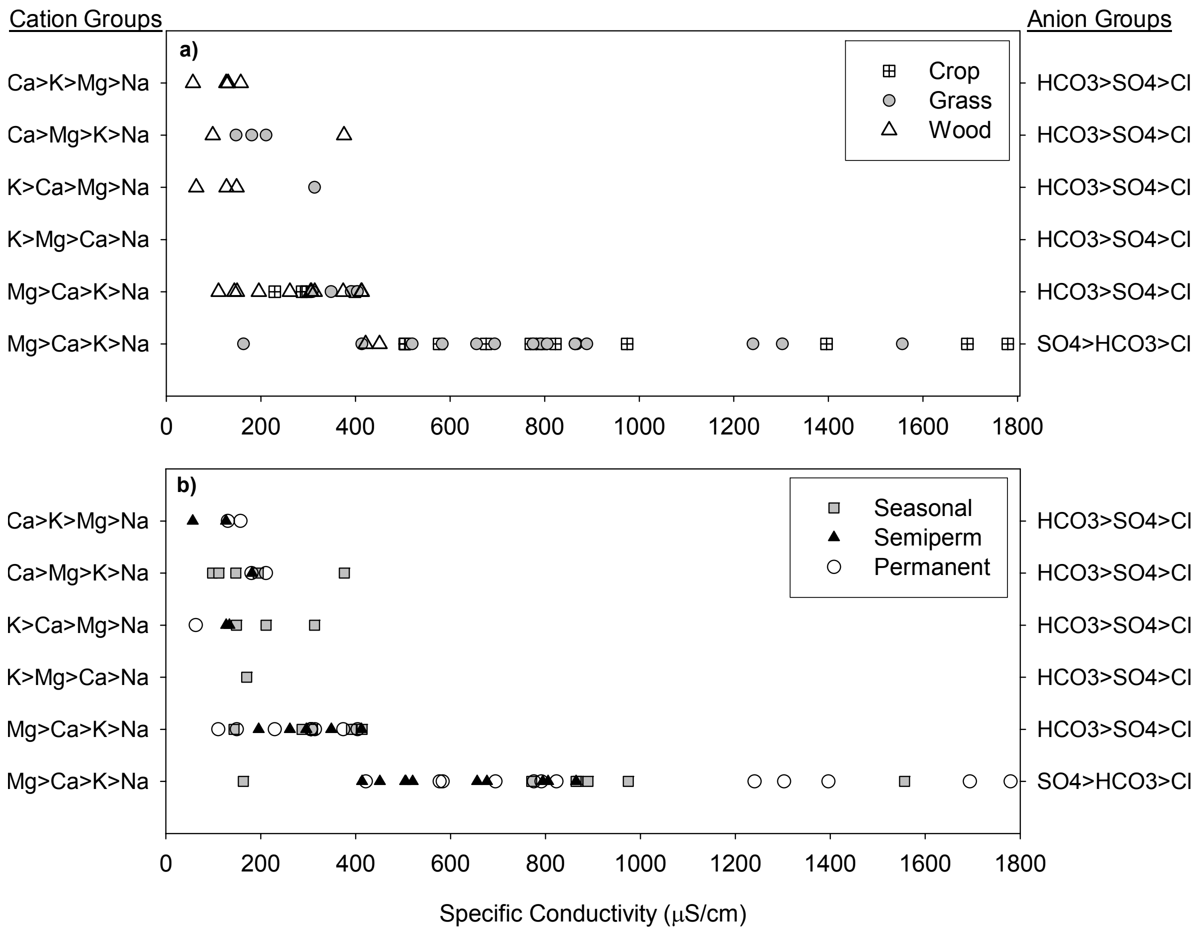


Figure 3.1 Distribution of cation and anion dominance groups as a function of specific conductivity for the 67 wetlands studied grouped by a) land cover and b) permanence classes. Semiperm is semi-permanently ponded wetlands.

Subsequent pairwise comparison tests elucidated differences among surrounding land cover types (Table 3.1) and permanence classes (Table 3.2) for the nutrient variable set and land cover types for the salinity variable set There were significant differences among land cover types for TP and K^+, i.e., wetlands with cropped uplands had greater TP and K^+ than wetlands with wooded or grassed uplands. Significant differences among permanence classes for TP, TDN, and DOC were also found. Permanently ponded wetlands had lower TP than seasonally and semi-permanently ponded wetlands. In addition, TDN and DOC were higher in seasonally ponded wetlands compared to semi-permanently and permanently ponded wetlands. SC, Cl^-, HCO_3^-, SO_4^{2-}, Na^+, Mg^{2+}, and Ca^{2+} concentrations were significantly lower in the wetlands with wooded compared to the cropped and grassed uplands.

Ionic proportions of wetlands included in the TOPAZ drainage network analysis are summarized in Figure 3.2. Ionic proportions of wetlands in sub-basins 153 and 174 were similar within each sub-basin, and these wetlands occupied similar topographic positions whereas wetlands located in sub-basin 138 showed a hydrochemical evolution from HCO_3^- to SO_4^{2-} dominated water (Figure 3.3). These ionic proportions and their relative position along the TOPAZ drainage network (Figure 3.4) suggests that wetlands in sub-basin 138 may connect along a fill and spill sequence. Wetlands located near the top of this TOPAZ drainage network (W48 and W50) were dominated by HCO_3^- and there is a general trend of increasing proportions of SO_4^{2-} along the flow path from W55 to W86 and W88.

Table 3.1 Mean and standard error of wetlands within land cover classes and a summary of results for two-way ANOVAs. Differing letter subscripts indicate significantly different (α = 0.05) Tukey's pairwise comparisons. * and ** denote a statistically significant difference at α = 0.05 and α = 0.01, respectively.

		Land Cover Class			Two-way ANOVA	
Variable	Unit	Crop	Grass	Wood	F	p-value
pH		7.33^c (0.08)	7.42^c (0.09)	7.18^d (0.12)	1.96	0.151
SC	µS/cm	633^c (104)	649^c (75)	194^d (23)	22.86	0.000**
Cl^-	meq/L	0.19^c (0.02)	0.23^c (0.05)	0.05^d (0.00)	19.77	0.000**
HCO_3^-	meq/L	1.89^c (0.15)	1.88^c (0.13)	1.18^d (0.11)	10.22	0.000**
SO_4^-	meq/L	3.24^c (0.81)	3.77^c (0.75)	0.33^d (0.13)	22.07	0.000**
Na^+	meq/L	0.57^c (0.16)	1.39^c (0.31)	0.07^d (0.02)	31.60	0.000**
Ca^{2+}	meq/L	1.65^c (0.23)	1.59^c (0.27)	0.64^d (0.06)	13.15	0.000**
Mg^{2+}	meq/L	3.63^c (0.83)	4.13^c (0.67)	0.78^d (0.16)	20.72	0.000**
K^+	meq/L	0.63^c (0.07)	0.43^d (0.03)	0.43^d (0.04)	4.17	0.020*
TP	mg/L	0.46^c (0.14)	0.18^d (0.04)	0.15^d (0.03)	4.68	0.013*
orthoP	mg/L	0.10 (0.03)	0.02 (0.01)	0.07 (0.03)	1.98	0.148
TDN	mg/L	1.40 (0.08)	1.23 (0.10)	1.27 (0.06)	2.62	0.082
NO_3^-	mg/L	0.03 (0.01)	0.02 (0.01)	0.02 (0.01)	0.54	0.585
NH_4^+	mg/L	0.026 (0.004)	0.027 (0.003)	0.026 (0.026)	0.56	0.572
DOC	mg/L	34.0 (2.1)	32.3 (1.6)	31.7 (1.6)	0.39	0.682
DIN:orthoP		4.3 (1.2)	6.7 (2.0)	5.3 (1.4)	1.75	0.183

Table 3.2 Mean and standard error of wetlands within permanence classes and a summary of results for two-way ANOVAs. Semiperm is semi-permanently ponded wetlands. Differing letter subscripts 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.

		Permanence Class			Two-way ANOVA	
Variable	Unit	Seasonal	Semiperm	Permanent	F	p-value
pH		7.18^{a} (0.09)	$7.21^{a,b}$ (0.09)	7.52^{b} (0.10)	4.26	0.019*
SC	µS/cm	446 (81)	417 (56)	601 (102)	1.68	0.195
Cl^-	meq/L	0.21 (0.06)	0.14 (0.03)	0.13 (0.02)	0.67	0.516
HCO_3^-	meq/L	1.66 (0.15)	1.57 (0.12)	1.71 (0.16)	0.17	0.842
SO_4^-	meq/L	2.06 (0.79)	1.95 (0.43)	3.25 (0.78)	1.56	0.219
Na^+	meq/L	0.62 (0.25)	0.50 (0.16)	0.93 (0.27)	2.15	0.126
Ca^{2+}	meq/L	1.42 (0.32)	1.17 (0.15)	1.29 (0.19)	0.08	0.921
Mg^{2+}	meq/L	2.33 (0.66)	2.20 (0.40)	3.89 (0.81)	2.54	0.088
K^+	meq/L	0.55 (0.05)	0.51 (0.07)	0.43 (0.03)	1.44	0.246
TP	mg/L	0.36^{a} (0.08)	0.34^{a} (0.14)	0.11^{b} (0.02)	8.73	0.001**
orthoP	mg/L	0.06 (0.02)	0.09 (0.03)	0.04 (0.01)	0.20	0.821
TDN	mg/L	1.58^{a} (0.10)	1.21^{b} (0.07)	1.12^{b} (0.05)	11.43	0.000**
NO_3^-	mg/L	0.02 (0.00)	0.04 (0.01)	0.02 (0.01)	0.16	0.851
NH_4^+	mg/L	0.026 (0.003)	0.024 (0.003)	0.030 (0.003)	1.02	0.366
DOC	mg/L	38.0^{a} (2.0)	31.5^{b} (1.2)	28.8^{b} (1.4)	8.96	0.000**
DIN:orthoP		4.3 (1.0)	6.8 (2.4)	5.6 (1.3)	0.30	0.742

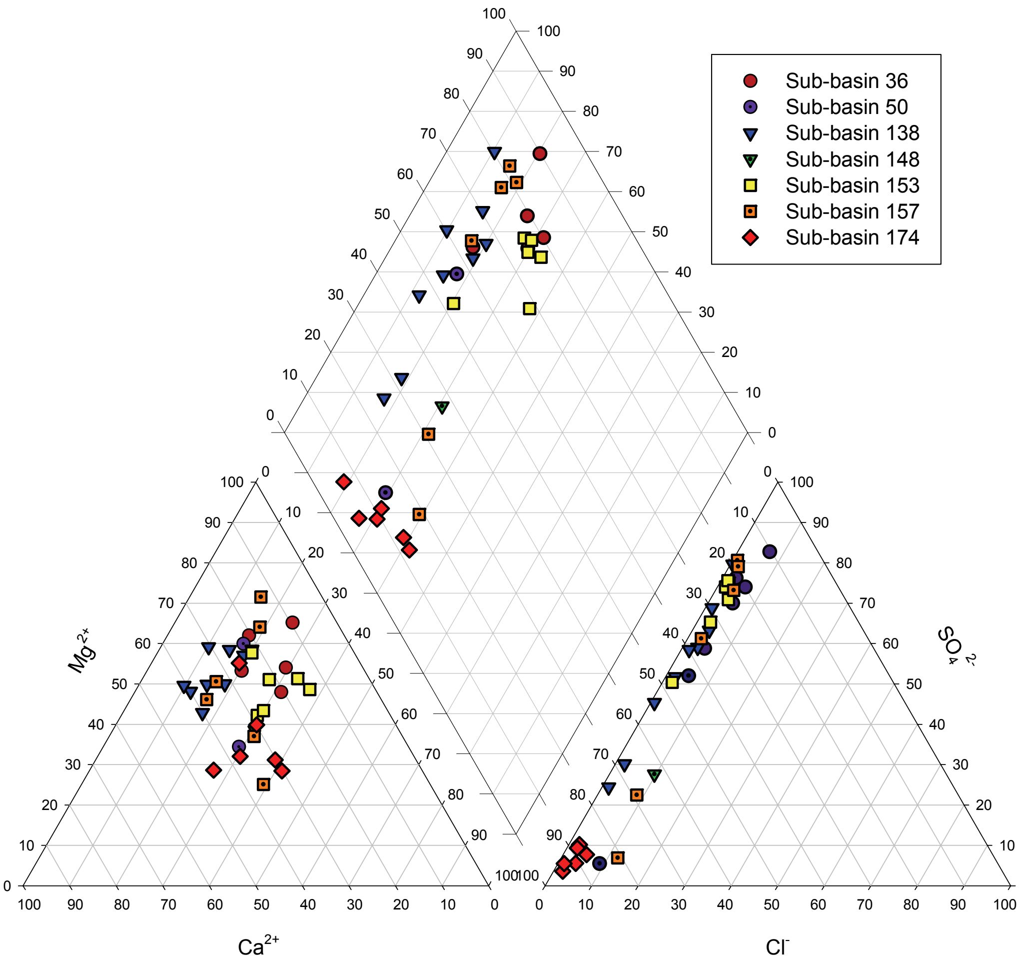


Figure 3.2 Piper plot of wetlands grouped by TOPAZ subbasins shown in Figure 2.4.

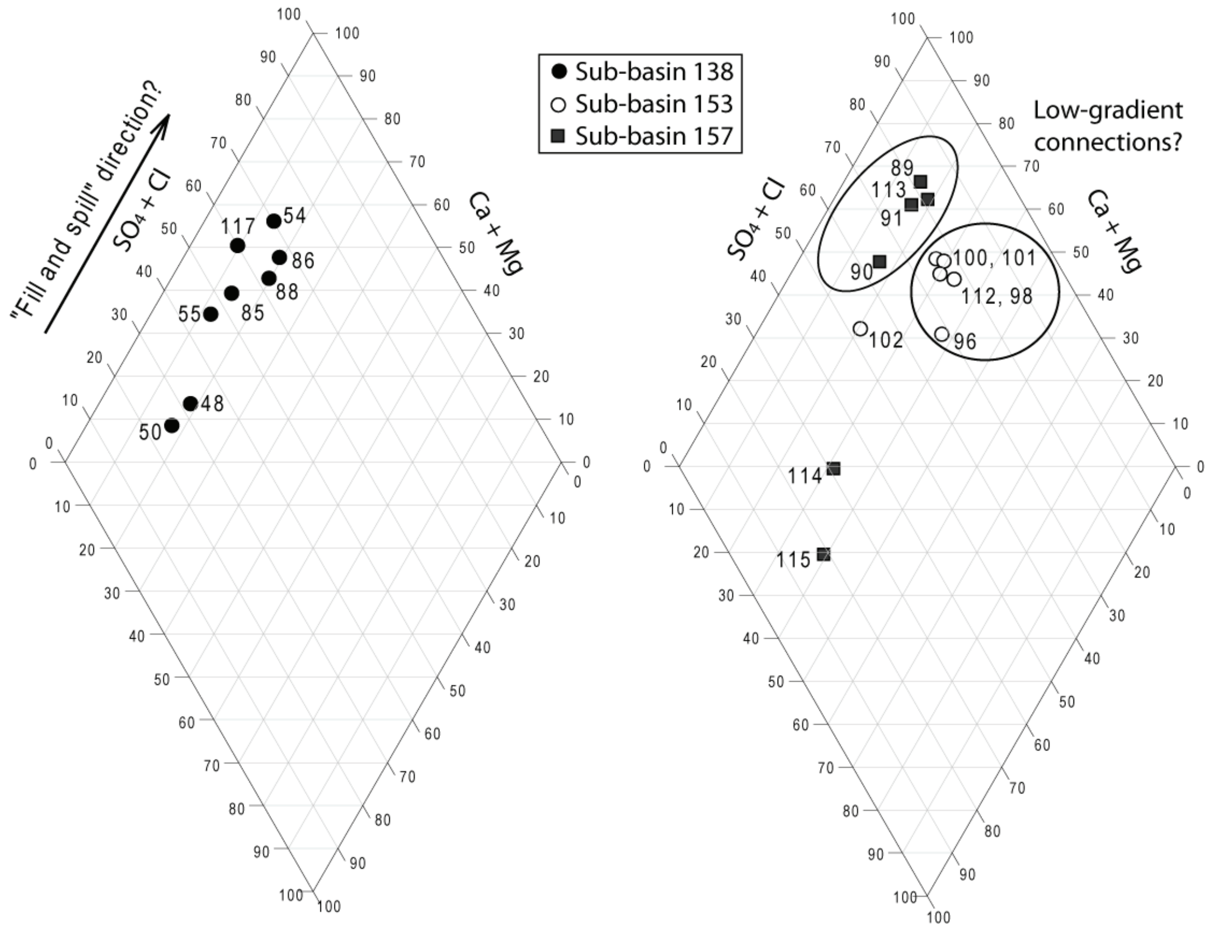


Figure 3.3 Central diamond shape of piper plot for wetlands located along a potential fill and spill sequence in sub-basin 138 (Figure 3.4) as identified from the TOPAZ drainage network analysis and topographic (LiDAR) position (left). Wetlands may also form low-gradient connections in wet years (sub-basins 153 and 157; right). Arrow depicts the general direction of a potential fill and spill sequence within the sub-basin. Circles encompass clusters of wetlands that may form low-gradient surface water connections.

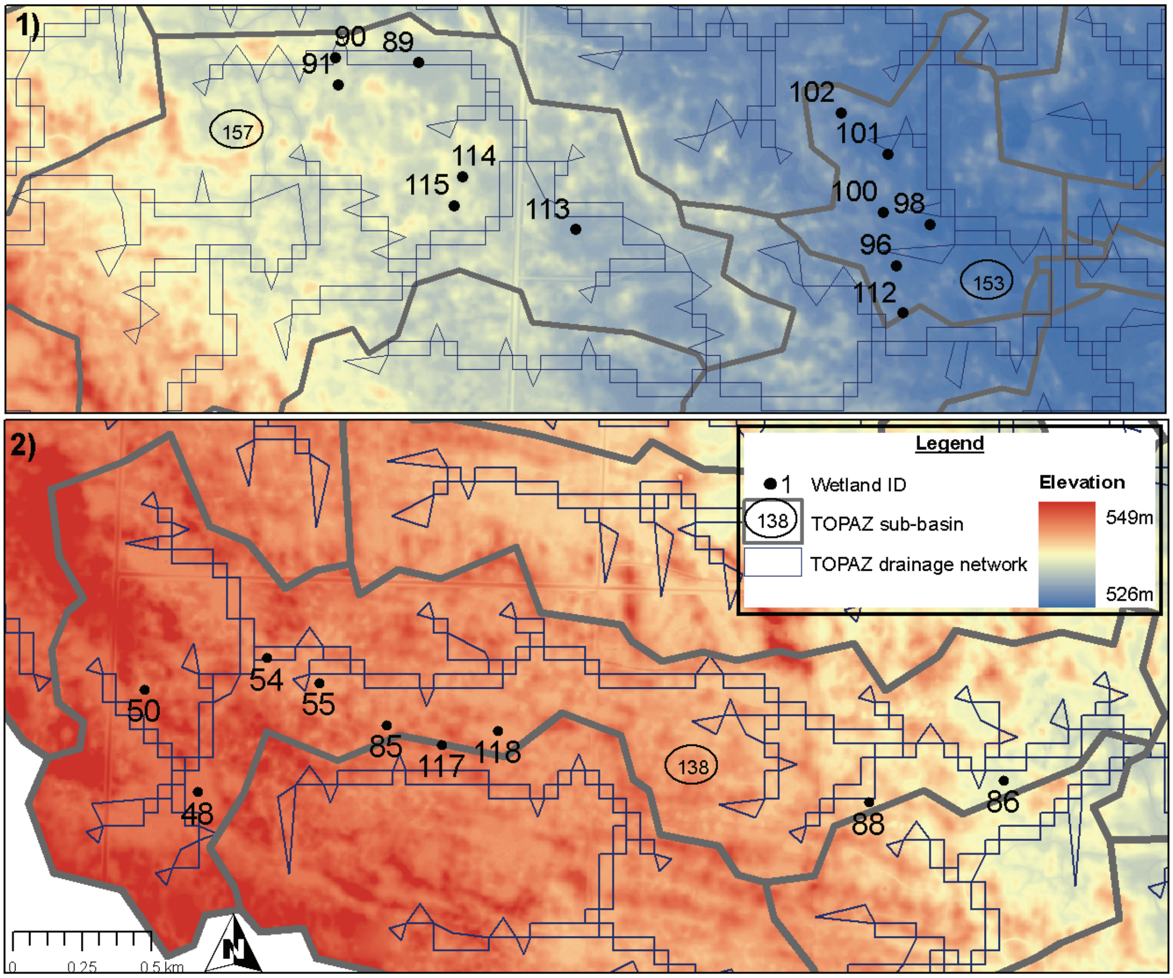


Figure 3.4 TOPAZ drainage network and wetland locations within TOPAZ sub-basin 1) 157 and 153, and 2) 138. Wetlands in sub-basins 153 and 157 potentially form low-gradient surface water connections. Wetlands in sub-basin 138 form a potential fill and spill sequence.

3.2. Solute Export during Wetland Drainage

3.2.1 Hydrological Characteristics of the LR3 Wetland Prior to Drainage

The volume of the LR3 wetland increased from 776 m^3 on October 23, 2007 to 2703 m^3 on April 18, 2008 following snowmelt. The volume tended to decrease during rain free periods and increase following rain events throughout the 2008 open water season (Figure 3.5). Following the snowmelt period, volume decreased ~60% throughout spring and early summer, reaching a minimum on July 7. Frequent and large rain events (termed midsummer rain events, Figure 3.5) occurred between July 7 and August 14, which caused the volume in LR3 to increase above the spring volume and remain high until the drainage experiment in November. Based on daily wetland water level fluctuations and precipitation data, ~6 mm/day of water was estimated to be lost from the wetland between May 1 and October 22, 2008. 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.

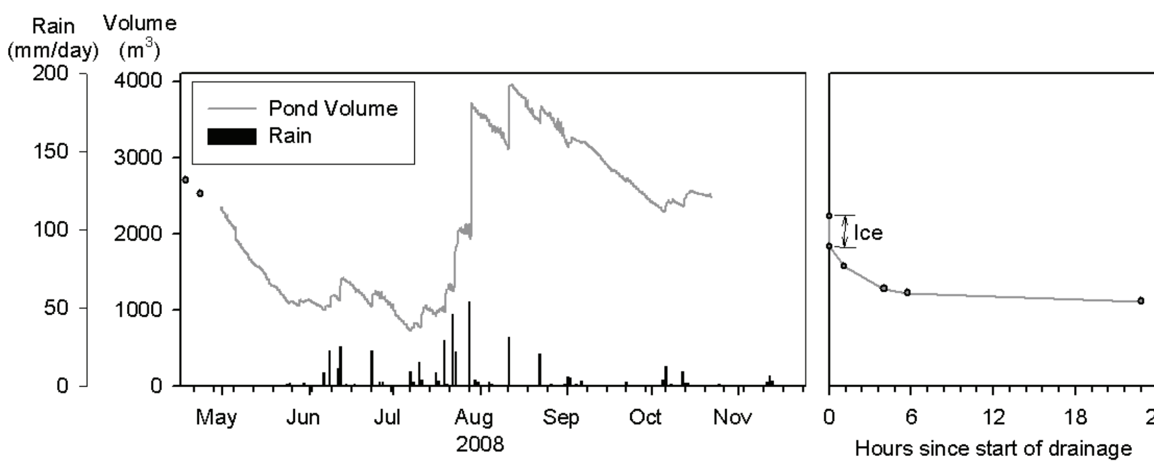


Figure 3.5 Daily rainfall and volume in the wetland prior to drainage and during the drainage experiment.

3.2.2 Water Quality Characteristics of the LR3 Wetland Prior to Drainage

Concentration (Figure 3.6) and mass (Figure 3.7) of forms of N and P were highly seasonal variable. Seasonal 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 midsummer 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 TP. TP ($p = 0.42$) and orthoP ($p = 0.29$) were not significantly correlated with Cl^-. The normalized mass data (Figure 3.8) show that TP and orthoP were both added to the wetland relative to Cl^- during the growing season.

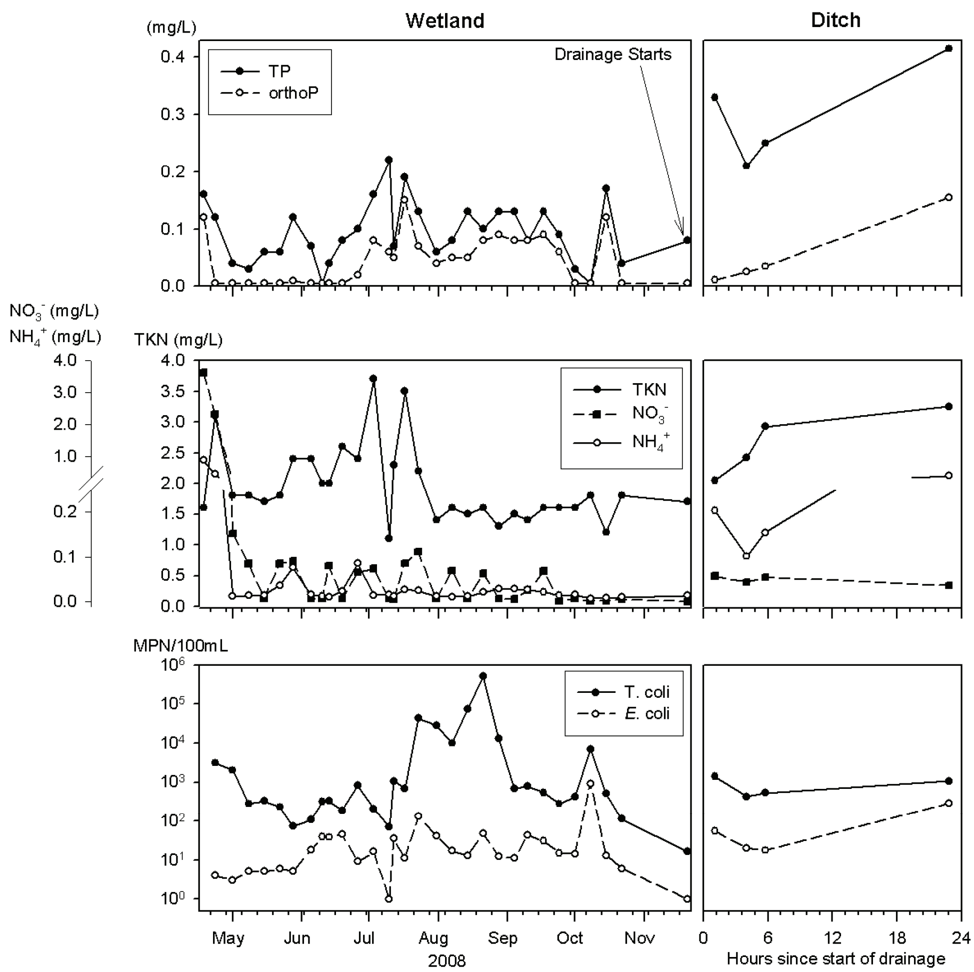


Figure 3.6 Concentration of nitrogen, phosphorus, and coliforms measured in the wetland prior to the drainage experiment and in the newly constructed ditch.

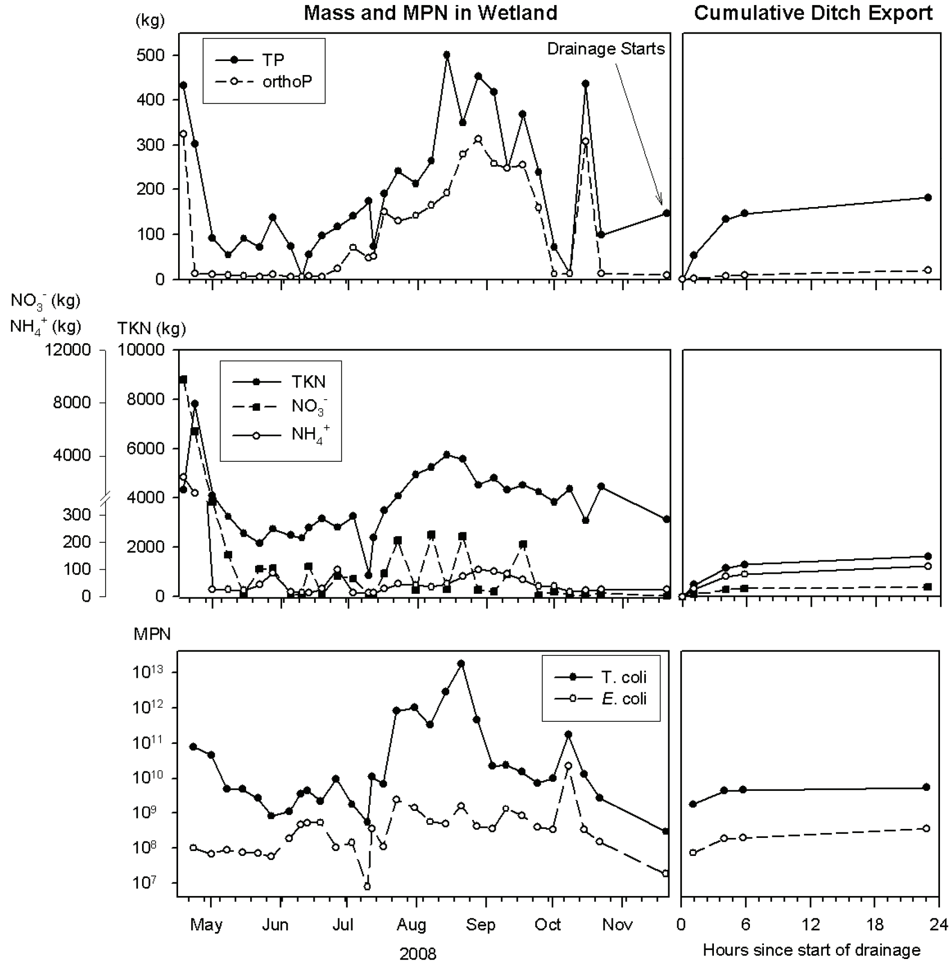


Figure 3.7 Total mass of nitrogen and phosphorus, and most probable number (MPN) of coliforms measured in the wetland prior to the drainage experiment and the cumulative amount exported via the newly constructed ditch.

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 and peaked July 3. NH_4^+ and NO_3^- concentrations were quite variable throughout the study period. Near the start of the midsummer rain events (July 10) TKN and NO_3^- decreased significantly while NH_4^+ increased slightly. Peaks in concentrations were reached July 17 (TKN and NH_4^+) 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 nutrient ratios (i.e. DIN:orthoP) to the Redfield Ratio indicates that the limiting nutrient in the wetland was potentially P in spring (Figure 3.8; Rhee and Gotham, 1980; Wetzel, 2001). After June 5, the limiting nutrient in the wetland was most likely predominantly N. 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 3.9) show that over the course of the study period, NH_4^+ was added to the wetland, NO_3^- was removed, and TKN was neither largely added nor removed relative to Cl^-.

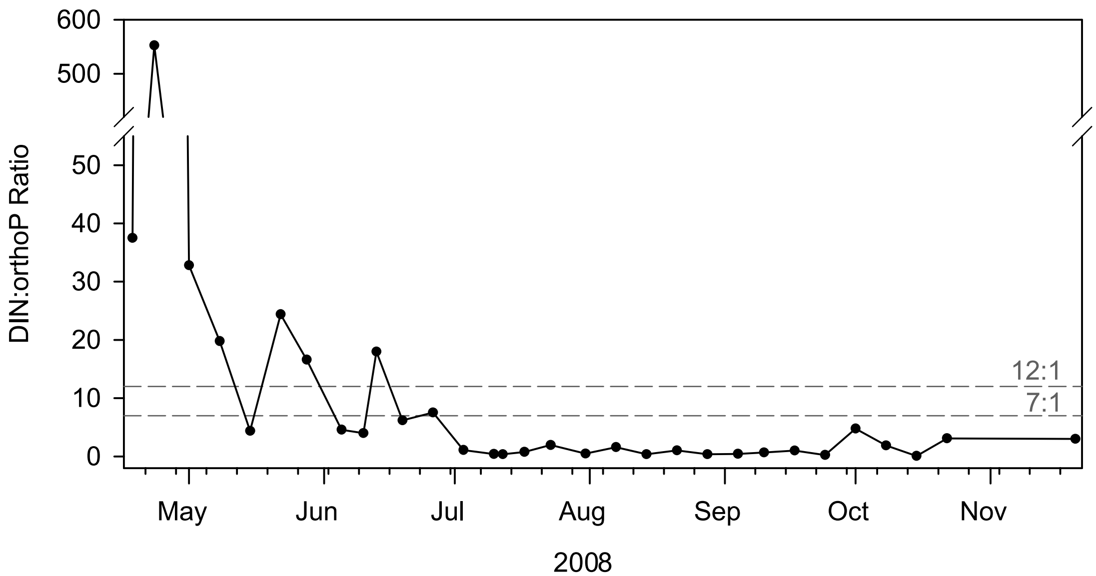


Figure 3.8 Mass ratio of dissolved inorganic nitrogen (NO_3^- + NH_4^+) to orthoP measured in the wetland prior to drainage. Ratio values below seven indicate that algal production may have been limited by nitrogen.

Variations in density (Figure 3.6) and most probable number (MPN), an estimate of total number, (Figure 3.7) of both T. coli and *E*. coli were very similar throughout 2008, and as such only trends in MPN are summarized. Following the snowmelt period, T. coli and *E*. coli total number in LR3 decreased until May 28, then increased until June 26 and June 19, respectively. Minimums were reached July 10. T. coli and *E*. coli total numbers increased during the midsummer rain events. T. coli and *E*. coli total numbers 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^-.

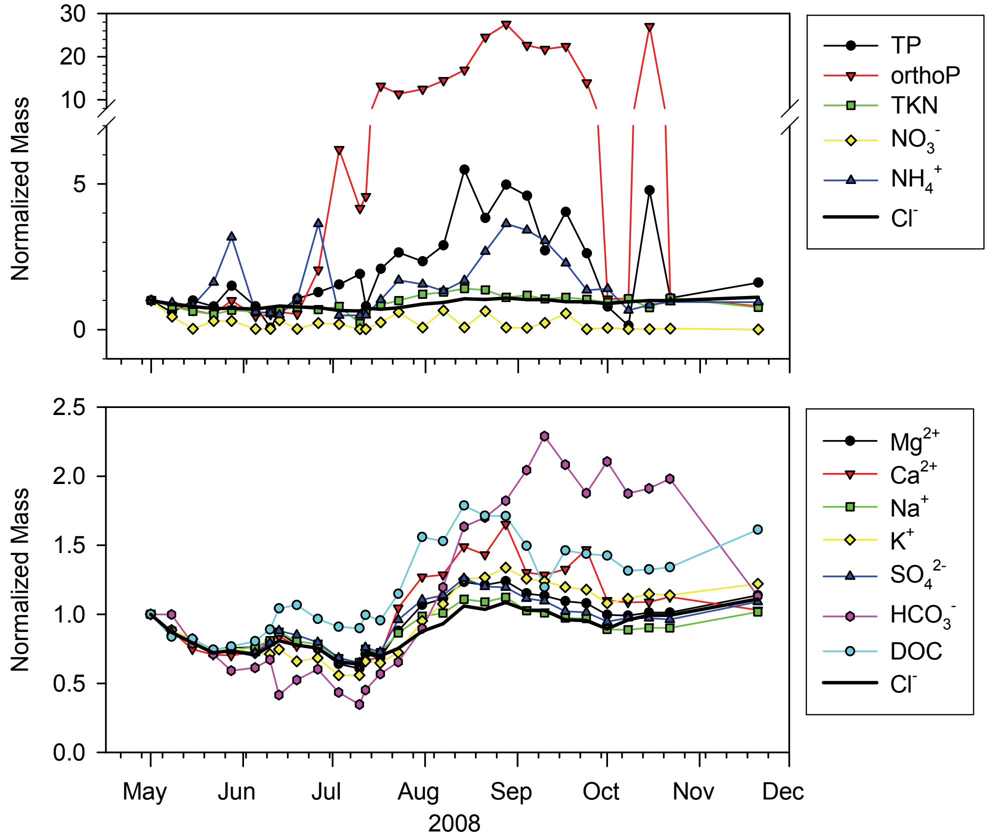


Figure 3.9 Normalized mass measured in the wetland during 2008. Data were normalized to May 1, 2008.

Seasonal variations in salts and DOC concentrations were similar throughout summer 2008, with the exception of HCO_3^- (Figure 3.10). DOC ($r = 0.88$, $p = 3x10^{-11}$) and all salts ($r = 0.93 - 0.98$, $p = 3x10^{-25} - 7x10^{-14}$), 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, salt and DOC concentrations generally increased, peaking around July 10. Salt concentrations then reached minimums on July 30 during the midsummer 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 October. Trends in salts and DOC mass (Figure 3.11) differed from trends in concentration. Salt and DOC 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. Salt and DOC mass mostly decreased in September and increased in October. The ion dominance pattern in the LR3 wetland was $SO_4^{2-} > HCO_3^- > Cl^-$ and $Mg^{2+} > Ca^{2+} > Na^+ > K^+$, which remained constant throughout the study period in 2008. Seasonal variation in pH (Figure 3.10) was similar to those observed for salt concentrations. However, pH peaked at 9.5 on July 3 and reached a minimum value of 7.4 on August 14. The normalized mass data (Figure 3.8) show that DOC, HCO_3^-, and Ca^{2+} were added to the wetland relative to Cl^-.

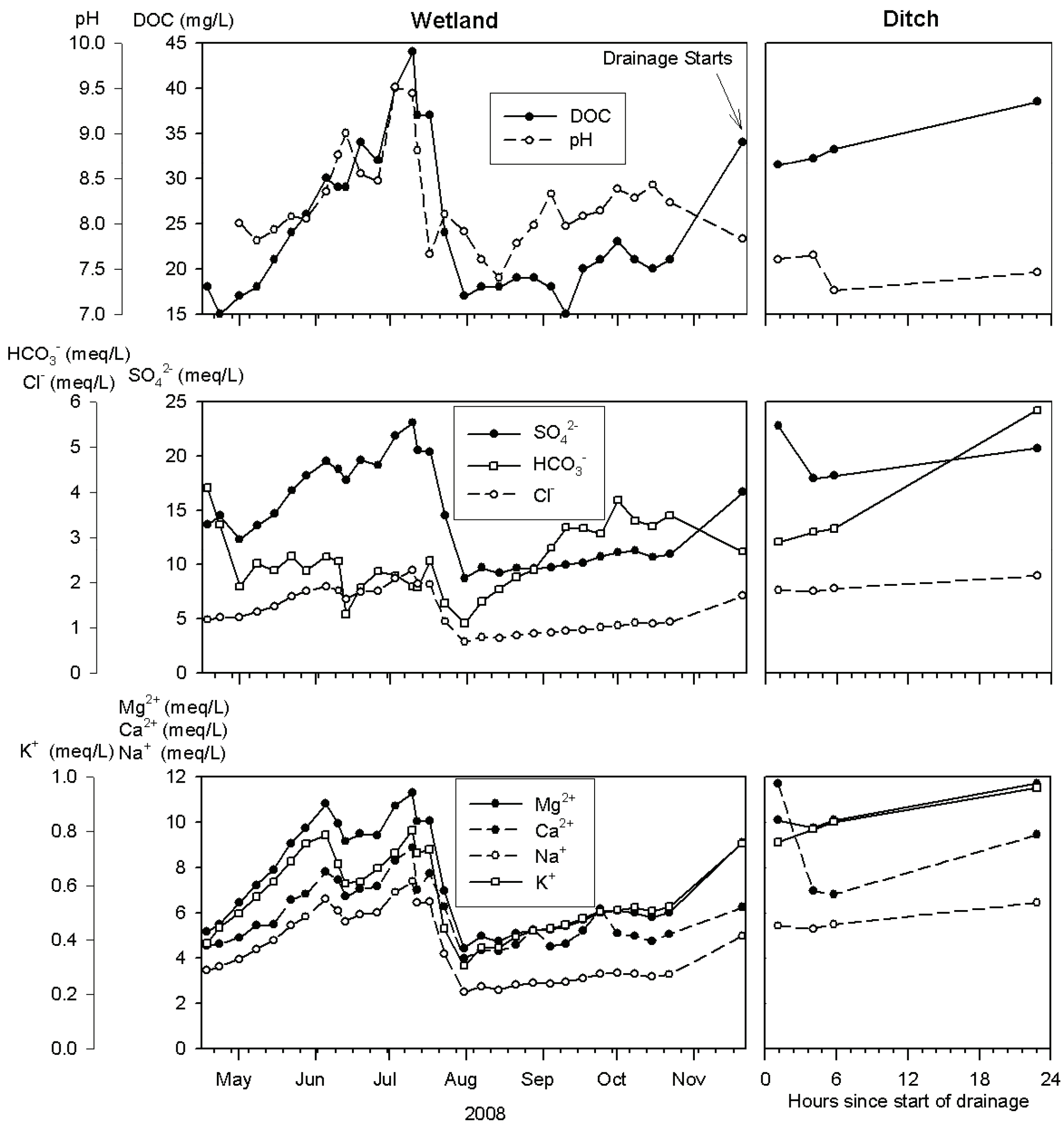


Figure 3.10 pH and concentration of DOC and salts measured in the wetland prior to the drainage experiment and in the newly constructed ditch.

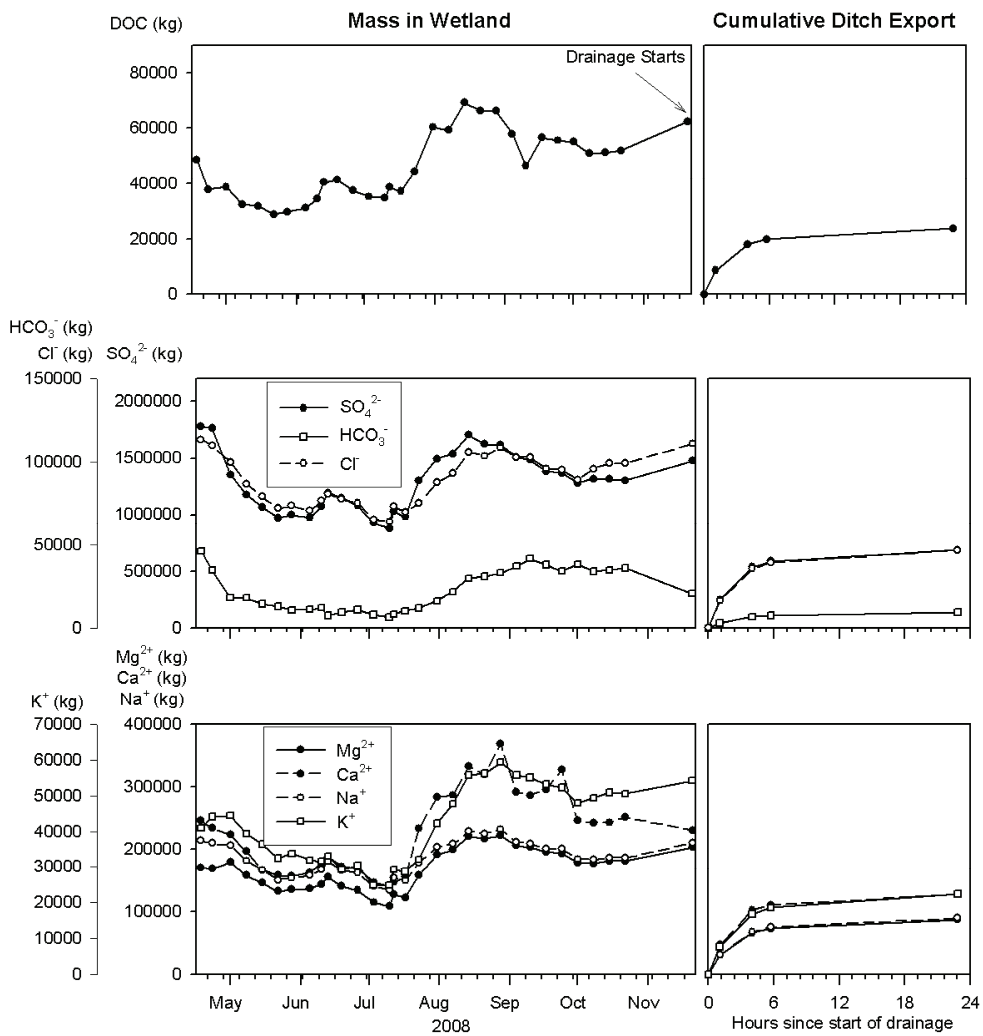


Figure 3.11 Total mass of DOC and salts measured in the wetland prior to the drainage experiment and the cumulative mass exported via the newly constructed ditch.

3.2.3 Water Quality Trends during Experimental Wetland Drainage

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 3.5). After 4 hr of drainage, 30% of the water and ~80% of solutes exported during the experiment had exited the wetland via the ditch. The average ditch water temperature was 0.6 ^{o}C, 0.4 ^{o}C, 0.3 ^{o}C, and 0.1 ^{o}C after 1 hr, 4 hr, 6 hr, and 23 hr of drainage. Following a preliminary analysis, the samples collected from within the wetland (LR3) during the experiment were not included in future analysis because the sampling point became too shallow and the samples were deemed unrepresentative of the water exiting the wetland at the time. Concentrations of N, P, coliforms, and salts in the drainage ditch exceeded those measured in the wetland at the start of drainage (Figures 3.6

and 3.10). 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.

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 solute mass or number exported from LR3 during the experiment is summarized in Figures 3.7 and 3.11. The total masses of TP, orthoP, NO_3^-, and NH_4^+, and the total number of *E*. coli and T. coli 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 salts 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 3.12. 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.

3.3 Ditch and Spill Solute Exports

Ditches were on average 71 % longer ($p = 1 \times 10^{-4}$), 12 % narrower ($p = 0.035$), and tended to have 33% higher flow velocities ($p = 0.016$) than spills (Figure 3.13). Solute concentrations also differed between ditches and spills (Figure 3.14). Specifically, TDN ($p = 0.003$), DOC ($p = 0.007$), HCO_3^- ($p = 0.023$), K^+ ($p = 0.001$), and Ca^{2+} ($p = 0.010$) 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 also significantly greater in ditches than spills (Figure 3.15). 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 was marginally greater ($p = 0.058$) at spill outlets than inlets.

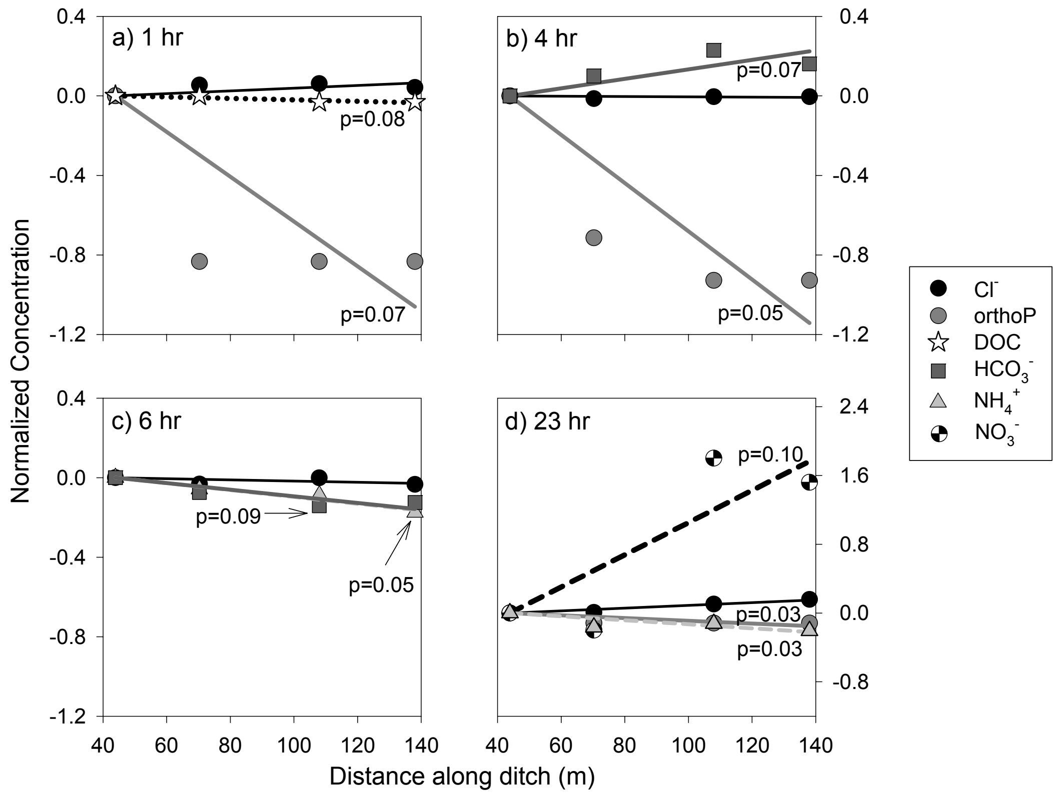


Figure 3.12 Slopes of normalized concentrations measured along the newly constructed drainage ditch a) 1 hr, b) 4 hr, c) 6 hr, and d) 23 hr after the start of the drainage experiment. Concentrations were normalized by dividing the concentration at each sample point along the ditch by the concentration at the first sampling point (DR1) in the ditch. A value of one was then subtracted to set the intercept to zero. Only solute 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.

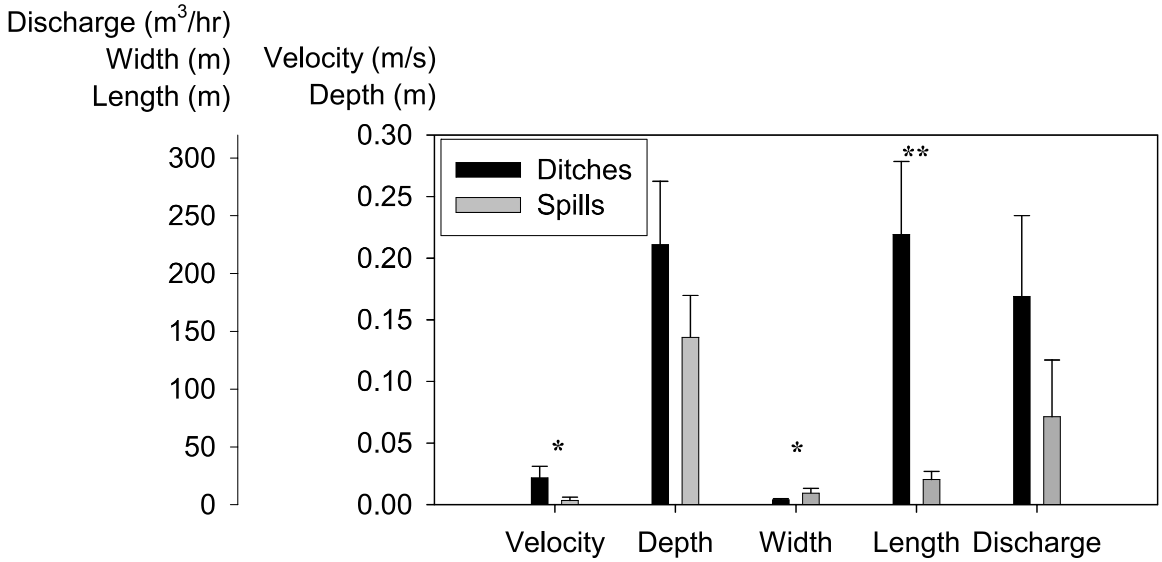


Figure 3.13 Mean and standard error of ditch and spill physical properties. *denotes a statistically significant difference at α = 0.05 and **denotes a statistically significant difference at α = 0.01.

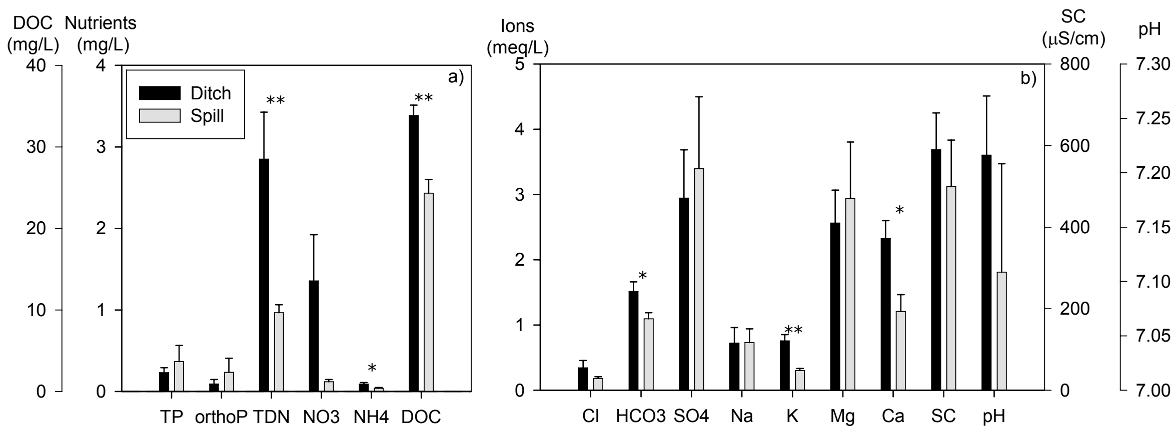


Figure 3.14 Mean and standard error of a) nutrients and DOC concentrations, and b) salt concentrations, SC, and pH in ditches and spills. *denotes a statistically significant difference at α = 0.05, and **denotes a statistically significant difference at α = 0.01.

3.4 Exceedance of Federal and Provincial Water Quality Guidelines

TP concentrations in the newly constructed ditch, 6 of 7 ditches, and 3 of 5 spills exceeded the Saskatchewan Watershed Authority (2007b) objective 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 wetlands with cropped, grassed, and wooded uplands, and 77%, 65%, and 40% of seasonally, semi-permanently, and permanently ponded wetlands. The TP objective was also exceeded for 45% of the LR3 wetland sampling instances.

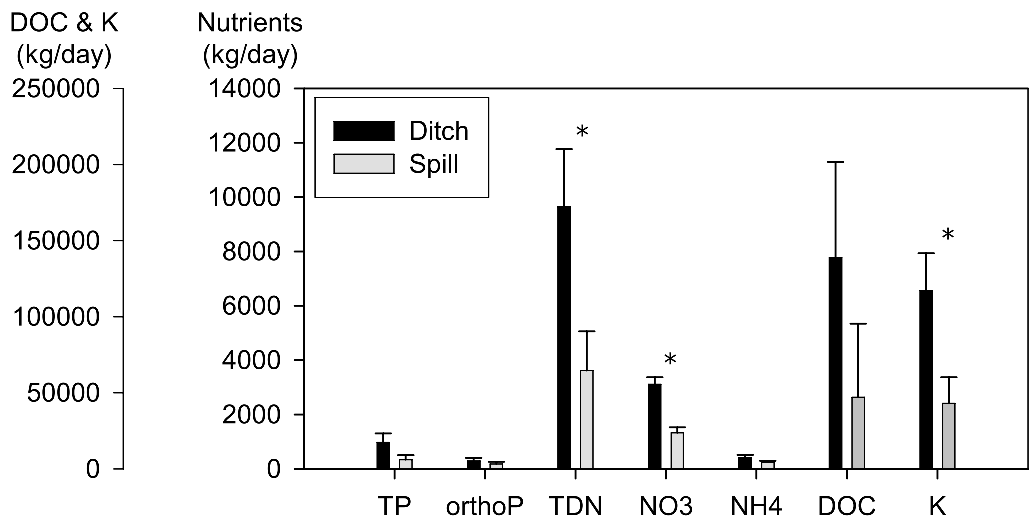


Figure 3.15 Mean and standard error of ditch and spill nutrient, K^+, and DOC loads. *denotes a statistically significant difference at $\alpha = 0.05$.

The majority of NO_3^- concentrations measured in the wetlands, newly constructed ditch, ditches, and spills did not exceed the CCME guideline for the protection of aquatic life (2003). However, this guideline was exceeded twice in the LR3 wetland following snowmelt and in 2 of 7 ditches. The CCME guideline for $NH_3 + NH_4^+$ was only exceeded by the first two samples obtained following snowmelt in the LR3 wetland.

The use of water containing salt concentrations that exceeded CCME guidelines for water used for irrigation and for livestock is not recommended. Concentrations of Cl^-, SO_4^{2-}, and Ca^{2+} measured in the 67 wetlands, ditches, and spills sampled in 2009 were below the CCME guidelines. The SO_4^{2-} guideline for livestock watering 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 was only exceeded by the samples collected in the newly constructed ditch 23 hours after the start of drainage. The CCME indicator bacteria guideline for the protection of agricultural water for crop irrigation 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 indicator bacteria CCME guideline for livestock watering.

4.0 DISCUSSION

4.1 Spatial Variation in Wetland Water Quality

Researchers have previously used ion dominance patterns and SC (as a proxy for net groundwater seepage rates) as indicators of pond permanence (Sloan, 1972; Millar, 1976; Driver and Peden, 1977). 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 local groundwater flow paths (LaBaugh and Swanson, 2004). However, results presented herein show that neither SC nor ion dominance can be used to distinguish among pond permanence classes at Smith Creek watershed. Although all but one of the six highest SC measurements were obtained from permanently ponded wetlands, there was no significant difference among permanence classes for SC or salt concentrations. This lack of difference among groups likely occurred due to the high variation within groups and because permanence classes are not 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 lack of difference among permanence classes may also be attributed to the fact that salt concentrations 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 midsummer (e.g. Arts et al., 2000); however, wetlands included in this study were not sampled later in the season because seasonally ponded wetlands typically become dry.

Variations in wetland water quality may instead be related to localized shallow groundwater (Hayashi et al. 1998a; van der Kamp and Hayashi 2009) and ephemeral surface connections, whereby wetlands fill and spill towards down-gradient receivers (Spence, 2006). Leibowitz and Vining (2003) used variations in SC as evidence for a wetland filling and spilling into a down gradient receiver wetland. These associations may serve as indicators for a wetland's position within a fill and spill sequence; Wetland located at the top of a fill and spill sequence are more likely to have lower SC and be HCO_3^- dominated, compared to terminal or down gradient wetlands in the fill and spill sequence that would likely have higher SC and be SO_4^{2-} dominated due to the accumulation and concentration of solutes. For example, an association between SC and anion dominance has been previously observed in prairie lakes and wetlands (Rawson, 1944; Rózkowska, & Rózkowski, 1969; Barica, 1975; Gorham et al., 1983; LaBaugh et al., 1987, Swanson, 1988). As salts 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).

Ionic proportions and topographic positions of wetlands in sub-basins 153 and 174 were similar, suggesting that these wetlands transiently form low-gradient connections and mix during very wet periods. Such conditions occurred in Smith Creek in spring 1995 when the highest stream discharge values over the 32 year period of record were observed (Environment Canada, 2011; Figure 4.1). The legacy of these flood conditions may explain the similarities in ionic proportions among wetlands in sub-basins 153 and 157, respectively.

Figure 4.1 Aerial photo of Smith Creek basin during maximum discharge observed over 32 period of record (spring, 1995). Photo courtesy of Don Werle.

Ionic proportions of wetlands located in sub-basin 138 showed a hydrochemical evolution from HCO_3^- to SO_4^{2-} dominated water and the relative position of these wetlands along the TOPAZ drainage network suggests that these wetlands may form a fill and spill sequence. Wetlands located near the top of the TOPAZ drainage network (W48 and W50) were dominated by HCO_3^- and during wet periods, these wetlands may fill and spill into down gradient receivers such as W54 and W55. W54 also likely receives water from wetlands located in the northwest portion of the basin during wet periods. W86 and W88, located near the bottom of the TOPAZ drainage network, were SO_4^{2-} dominated. Although W85 and W117 are not located along the TOPAZ drainage network, the LiDAR DEM and their ionic proportions suggest that they form low-gradient surface water connections with adjacent wetlands that are connected to the fill and spill flow path during wet periods. Future field testing of geochemical evolution along fill and spill pathways and transient low-gradient wetland connections is needed.

The 67 wetlands sampled also grouped distinctly into low SC and HCO_3^- dominated groups, largely typical of the wetlands with wooded uplands, and relatively high SC and SO_4^{2-} dominated groups. A relational shift in ion dominance with increasing SC has been previously noted by Gorham et al. (1983) in a study of lakes in north-central United States. This shift was reflected a westward increase in climactic aridity and a sequencing of glacial drift from noncalcareous, to calcareous, and to calcareous with abundant sulfur-bearing minerals. It is thus possible that the observed differences in wetland salinity at Smith Creek among land cover types may be attributed to differences in soil characteristics and a detailed analysis of soil characteristics in relation to wetland pond water quality may provide insight into spatial variations. Significantly lower SC and salt concentrations measured in the wooded areas could also result from their likely position near the top of a fill and spill sequence, rather than characteristics of the surrounding uplands.

Snowmelt characteristics (i.e. slow/delayed melt and high infiltration) may also partially explain the low SC observed in wetlands with wooded uplands. Laboratory experiments have shown that early snowmelt runoff can contain elevated ion concentrations as a result of ion exclusion, followed by a rapid decrease in ion concentration as melt progresses (Johannessen and Henriksen, 1978; Lilbæk and Pomeroy, 2007). Thus, wetlands with grassed and copped uplands receiving early snowmelt runoff may also receive a significant pulse of salts with snowmelt runoff. However, the magnitude of snowmelt water inputs will depend on the infiltration capacity of the surrounding uplands. The infiltration potential of frozen soils is higher if the soils are dry and have a well-developed macropore structure (Gray et al., 2001; Bodhinayake and Si, 2004). Cultivation typically reduces macroporosity and infiltration capacity (Bodhinayake and Si, 2004), which decreases infiltration of snowmelt water and rain, and increases surface runoff to depressions (van der Kamp and Hayashi, 2009). The higher infiltration capacity of wooded soils (Gray et al., 2001), suggests that fewer solutes are transported by snowmelt runoff from wooded uplands than from grassed or cropped uplands. Volumetric soil moisture, measured October 22, 2008 was 53%, 33% and 21% in representative grass, crop, and wood areas at Smith Creek watershed (Fang et al., 2010). Although volumetric soil moistures at the onset of melt likely differed, SC was not significantly different between wetlands with cropped and grassed uplands. SC could also have been lower in wetlands with wooded uplands because the rate of snowmelt is slower in wooded areas compared to open areas since trees emit long wave radiation, which contributes to melting snow, however trees also absorb shortwave radiation and reduce turbulent transfers of heat to the snow surface, which reduces snow melt rates (Suzuki et al., 2003). A longer snowmelt period can further increase infiltration and also increases the probability that rain on snow events will occur. Rain on snow events can lead to lower ion enrichment rates due to decreased contact time of precipitation with the snow and a greater proportion of free water (Colbeck, 1981; Marsh and Pomeroy, 1999). Rain on snow events did not occur during the 2008 melt period, however these events were frequent during the 2009 melt period.

Wetlands with wooded uplands may also, in part, have lower Ca^{2+}, K^{+}, and Mg^{2+} concentrations than wetlands with grassed or cropped uplands because their uplands are comprised primarily of aspen (*Populus tremuloides*), which are known to store these solutes in their standing biomass (Wang et al., 1995). Aspen have the ability to cycle Ca^{2+} from soluble sources at depths (or from within adjacent wetlands) to the soil surface, leading to secondary calcite precipitation in forest soils (Fuller et al., 1999). Similar processes likely also affect other solutes.

SC and salt concentrations measured throughout the Smith Creek watershed were lower than maximum values measured at other lakes and wetlands in the PPR (Rózkowska and Rózkowski, 1969; Barica, 1975; LaBaugh et al., 1987; Detenbeck, 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 PPR (Millet et al., 2009), which would increase the precipitation to evaporation ratio and minimize the effects of evapoconcentration. Additionally, the midsummer rain events that occurred in the region during the 2008 summer prior to wetland sampling caused many wetlands 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 2009, shortly after the wetlands have filled with dilute snowmelt runoff, and before salts 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^{2-} dominated later in summer, based on the work of LaBaugh et al. (1987) and Detenbeck et al. (2002), which showed some temporal progression in a seasonally ponded wetland ion dominance patterns.

Some of the spatial variation in wetland nutrients can be attributed to variations in land cover and permanence classes. The greater TP and K^+ concentrations measured in wetlands with cropped than wooded and grassed uplands may be the result of varying amounts of surface runoff influenced by infiltration capacity from different land cover types, as described above. The higher concentrations of TP, TDN and DOC in seasonally ponded 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 permanently ponded wetlands and some semi-permanently ponded wetlands would not lead to the same enhanced mineralization and nutrients released (Brinson et al. 1981). LaBaugh and Swanson (2004) suggested that this vegetation zonation explained the occurrence of higher TP concentrations in seasonally ponded wetlands compared to semi-permanently ponded ones. Permanently ponded wetlands are also typically deeper, and thus less prone to sediment resuspension and the associated resuspension of nutrients. Nutrient concentrations may also be lower in permanently ponded wetlands due to the presence of floating and submersed vegetation that more efficiently remove nutrients from the water and sediment compared to emergent plants of seasonally ponded wetlands that obtain most of their nutrients from the sediment alone (Birgand et al., 2007). The near continuous flooding of semi-permanently and permanently ponded wetlands also 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). Further, seasonally ponded wetlands are often cropped when they are dry, thus they can have higher N and P concentrations than semi-permanently or permanently ponded wetlands because of direct fertilizer application (Cowardin et al., 1981).

In addition to differences in nutrients among pond permanence classes, differences in nutrients were also found among surrounding land cover types. The greater TP and K^+ concentrations in wetlands with cropped than wooded and grassed uplands may have resulted from fertilizer inputs being transported to wetlands from their cropped uplands during runoff events (Hansen et al., 2002; Little et al., 2007; Tiessen et al., 2010). The finding that N and DOC did not differ significantly among land cover types was surprising given that other researchers have shown that apsen litter contains fewer nutrients and DOC than grass litter (Fuller and Anderson, 1993; Köchy and Wilson, 1997), and that nutrients stay stockpiled in woody biomass for extended periods of time (Wang et al., 1995); thus fewer nutrients and DOC would be expected to be available for leaching to wetland water columns in wooded areas. The lack of significant difference in N among land cover types may also be attributed to the fact that the majority of wetlands sampled, similar to shallow prairie lakes, were likely eutrophic and characterized by low DIN:orthoP ratios, and were thus potentially N limited (Barica, 1990; Hall et al., 1999); meaning that excess N could be readily taken up.

The analysis of dissolved nutrient ratios should however only serve as a rough guide of nutrient limitation because ratios of DIN to soluble reactive phosphorus (SRP; an estimate of orthoP) are a weak surrogate for TN:TP that represent the total nutrient content actually in biomass or available for incorporation into active biomass (Dodds, 2003). Additional problems with using this ratio to indicate nutrient limitation may also arise because concentrations of DIN and SRP are not necessarily indicative of supply or nutrient turnover rates (Dodds, 2003). Furthermore, aquatic systems can become nutrient saturated and insensitive to changes in nutrients concentrations. In freshwater lakes, a threshold value for SRP has been suggested to be to be 8 μg/L by Cooke et al. (1993) and Marsden (1989), and 10 μg/L by Prepas (1983), Auer et al. (1986), and Interlandi and Kilham (2001). Davies et al.

(2004) found that plankton communities were not limited by phosphorus when TP exceeded 11 μg/L. However, in a prairie lake study by Waiser and Robarts (1995), a higher threshold value of 30 μg/L SRP was suggested because in this highly saline system, all P may not be available for microbiological growth. Thus, a threshold value between 10 μg/L and 30 μg/L orthoP is likely for moderately saline Smith Creek wetlands, given that Dodds (2003) reports that measures of SRP often underestimate the amount of orthoP. Percentages of Smith Creek wetlands sampled with orthoP concentrations < 10 μg/L and > 30 μg/L were 54% and 25%, respectively. Studies in freshwater lakes of DIN have suggested that at concentrations above 15 – 50 μg/L, nitrogen is not limiting (Davies et al, 2004), while Interlandi and Kilham (2001) suggested that this threshold is 100 μg/L. Percentages of Smith Creek wetlands with DIN < 50 μg/L and > 100 μg/L were 61% and 5%, respectively; average DIN was 52 μg/L.

Results also show that the interaction between land cover and permanence classes did not significantly influence nutrient or salt water quality parameters. Thus, control mechanisms on water quality parameters attributable to pond permanence and land cover type are likely additive. The lack of significant interaction and extensive differences among permanence or land cover classes 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 pothole water quality include grazing, cultivation, and tillage practices, soil characteristics, shallow groundwater fluxes, and the presence of willow rings surrounding wetlands.

The hypothetical drainage of these wetlands would decrease the amount of water stored on the landscape and would likely increase streamflow of Smith Creek during spring freshets (Campbell and Johnson, 1975; Saskatchewan Watershed Authority, 2008; Yang et al., 2008). Permanently ponded wetlands are generally larger and contain more water; however, TP and TDN concentrations were more dilute in these wetlands. Thus, downstream loadings could comparatively be greater due to the drainage of an equivalent volume of smaller, seasonally ponded wetlands with elevated TP and TDN. Although water in the wetlands did not exceed guidelines for NO_3^-, NH_4^+, Cl^-, SO_4^{2-}, and Ca^{2+} the provincial objective for TP was exceeded and these exceedances occurred most frequently in seasonally ponded wetlands with cropped uplands. Consequently, wetland drainage would be expected to degrade downstream water quality with respect to P. As the research site is a sub-basin of the Assiniboine River, nutrient loadings from drained wetlands could facilitate further eutrophication of Lake Winnipeg, into which the Assiniboine River drains (Schindler, 1977; Armstrong, 2002).

4.2 Factors Influencing Temporal Patterns in Wetland Water Quality

The LR3 wetland effectively trapped N, P, DOC, coliforms, and salts 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 show hydrological processes, such as runoff, evaporation, and shallow groundwater seepage, are the dominant control on all solutes studied, 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 among salts and DOC compared to N, P, and coliforms suggest differing control mechanisms in the wetland. Differing control mechanism for salts, nutrients, and coliforms are recognized (Wetzel, 2001) and LaBaugh et al. (1987) suggested that differing control mechanisms were responsible for variations between salt and nutrient concentrations in prairie wetlands. Seasonal fluctuations of salts and DOC appear to be primarily linked to hydrological processes due to the significant correlations with Cl^-. Whereas variable seasonal dynamics observed between major snowmelt and precipitation events, and the lack of significant correlations between Cl^- and TP, orthoP, NH_4^+, NO_3^-, and coliforms suggests that wetland N, P, and coliforms are linked to both hydrological processes and biotic/sorption processes. For example, wetland nutrients have been shown to be influenced by 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 (Barica, 1974b; Neely and Baker, 1989; Neill, 1995; Labaugh and Swanson, 2004).

Temporal variations in solute concentrations and masses in the wetland can be largely 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 solutes 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. Surface and/or subsurface runoff, indicated by daily water level increases that exceeded 5 mm, are likely responsible for the high solute masses and coliform loads observed during the snowmelt period and the increases that occurred following the rain events.

Concentrations of DOC and salts 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 evaporation driven 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 Cl^-, Na^+, Ca^{2+}, Mg^{2+} and K^+ (CAPMoN, 2007). As a result, inputs during the midsummer rain events lead to a decrease in salt concentrations. It is thus unlikely that the large increases of salt mass measured immediately following rain events were primarily caused by inputs from direct precipitation on the wetland, but rather the increases were caused by solutes transported by runoff and potentially by shallow groundwater inputs. Although increases in DOC concentrations have been associated with high rates of summer precipitation in boreal lakes (e.g. Hudson et al., 2003; Zhang et al., 2010), DOC concentration in the LR3 wetland decreased following the mid-summer rain events; however DOC mass did increase during that time period.

Following snowmelt, the wetland had high nutrient concentrations. While transient, elevated N and P concentrations that exceeded water quality guidelines immediately following snowmelt are important to note because ditched wetlands drain at this time. Since both NH_4^+ and NO_3^- were elevated in the wetland following snowmelt, flushing of DIN from soils insulated by snow is likely, as has been shown for boreal forest systems (Devito et al. 1999; Jones, 1999). Soil (0 – 20 cm) temperatures at Smith Creek during the winter prior to the sampling of the wetland were above -6 °C (Fang et al. 2010), and thus nutrient mineralization was possible beneath the snowpack.

Elevated nutrient concentrations also coincided with a 2.5-fold increase in wetland volume, relative to the previous fall. Upon re-wetting, 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 also released (Reddy and Patrick, 1975; Murkin et al., 2000; Aldous et al., 2005). As has been shown to occur in agricultural systems, runoff likely transported nutrients from the cropped areas of the catchment that received fertilizer in fall 2007 and May 2008 (Hansen et al., 2002; Little et al., 2007) or from soils in the region that are naturally nutrient rich, especially in P (Anderson, 1988). Similar to results from Batt et al. (1989) and LaBaugh and Swanson (2004), these elevated N and P concentrations also coincided with periods of use by breeding waterfowl.

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 P 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 0.5 mg/L and 0.8 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 suggest that reductions in TP, orthoP, and NH_4^+ mass following the midsummer rain events were due to biotic uptake and/or biogeochemical reactions. Furthermore, the normalized mass data also suggest that NO_3^- was effectively removed from the wetland system relative to Cl^-, likely by biotic uptake and denitrification. Although measurements of nitrogen cycling in prairie wetlands are limited (e.g. Moraghan, 1993), and some of the following factors were not measured for LR3 wetland, authors such as Neely and Baker (1989) have noted that conditions are likely suitable in prairie wetlands for denitrification to occur: anaerobic conditions, the presence of a large organic carbon base, an abundance of NO_3^-, and sizable denitrifier population.

Temporal variations in Ca^{2+} and 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^{2+} 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). However, the proportional mass data did not indicate that SO_4^{2-} was removed from the wetland differently than was Cl^-, meaning sulfate reduction was likely not an important driver of SO_4^{2-} 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 or grazed by cattle 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 lodges constructed sometime between June 25 and September 24, 2008. Given that all but one sample collected in the LR3 wetland exceeded the animal-specific CCME indicator bacteria guideline for livestock watering, livestock should be prevented from accessing similar wetlands.

4.3 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. However, because the wetland was not fully drained, significant proportions of solutes remained in the wetland at the end of the drainage experiment. 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). Fall ditch construction following harvest, when water temperatures are lowest, 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. The drainage conditions studied here are common in the prairies, and thus results herein should be transferable to other watersheds in the PPR, although rigorous quantitative testing and monitoring 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 solutes concentrations. Barica (1974), Fisher and Reddy (2001), and Barker et al. (2010) have 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 amount of organic matter present in the sediment than water column (Karim et al., 2004). A vertical gradient in concentration may also explain why the masses and total numbers 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 centre of the wetland at half the water depth and thus the amount of mass calculated from these samples would be underestimated if a vertical concentration gradient existed. Solutes may also become stockpiled by the undrained portion of wetland since at the end of the drainage experiment ~50% of the water, TKN, DOC, and salts estimated remained in the wetland.

Another explanation for the increase in concentration of water quality parameters is that many wetlands in the prairies have salt rings around them (Arndt and Richardson, 1993). 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.

Because snowmelt runoff typically dominates prairie wetland water inputs, proportions of nutrients transported by snowmelt runoff generally exceed amounts transported during rainfall events (Timmons and Holt, 1977; Tiessen et al., 2010). Thus, had it not been for the uncharacteristically high midsummer rainfall, solute concentrations leaving the wetland would probably have been much higher and solute mass exported lower at the time of drainage. However, it is likely that the guideline for TP would remain the only nutrient guideline exceeded. Decreases in SC and salt concentrations, have also been attributed to rainfall and decreased evaporation (Barica 1978; LaBaugh and Swanson, 2004). The increase in concentrations of salts 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 and Gallup, 1978; Lilbæk and Pomeroy, 2008). Whereas low coliform numbers at the start of the drainage experiment were likely the results of sedimentation on the wetland bed (Auer and Niehaus; 1993; Wang and Doyle 1998).

4.4 Comparing Artificial Ditches to 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^{2+} observed in ditches than spills were most likely due to differences in wetland water quality rather than differing physical attributes of drains and spills. Ditches often drain seasonally ponded wetlands because these wetlands are frequently the easiest to drain and because ditching permanently and semi-permanently ponded wetlands effectively turns them into seasonal ones because they are drained each year. Ditched wetlands are also predominantly located in cropped areas, a situation typical across the PPR (Gunterspergen et al., 2002). In contrast, spills flowed largely from permanently ponded wetlands and were located only in grass and wooded areas. As outlined in section 3.1 of this thesis, seasonally ponded wetlands are characterized by greater concentrations of TP, TDN and DOC; and TP and K^+ concentrations are greatest in wetlands with cropped uplands. As a result, ditches draining this type of wetland have the potential to contribute higher concentrations of these nutrients to downstream ecosystems.

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 the 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 are often transiently connecting wetlands to other wetlands. These different connection characteristics suggest that wetland drainage has a higher likelihood of enhancing downstream nutrient, salt and coliform loading than spills.

To date, one of 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, they 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. Results presented herein suggest that a nutrient export coefficient other than 1 is not warranted. This result is likely attributable to generally low temperatures during the snowmelt period when ditches and spills flow and when new ditch construction occurs. Instead, the volume of water and solute mass exported from a drained wetland will depend on: i) how effectively the ditch drains the wetland; and ii) the water quality characteristics of the wetland that are influenced by the surrounding land cover and permanence classes.

Increased salinity in downstream water and in soils adjacent to waterways can have negative impacts on downstream water quality and agricultural production decreasing soil fertility and impeding the growth of crops in soils adjacent to ditches (Skarie et al., 1986; Steppuhn et al., 2001). However, the provincial guidelines for Cl^-, SO_4^{2-}, and Ca^{2+} were not exceeded in the ditches and spills, and thus wetland drainage is not expected to degrade soil productivity as a result of direct salinisation. The more frequent exceedance of NO_3^- and TP guidelines in ditches than spills further suggests that ditches will more negatively impact downstream water bodies. Moreover, the impacts of wetland drainage on water quality will likely be intensified where wetland ditches drain to terminal and near terminal basins, such as the nearby Waldsea, Deadmoose, Houghton, and Fishing lake basins, as nutrients can become increasingly concentrated by evaporation.

5.0 CONCLUSIONS

This thesis has shown that spatial variations in wetland water quality can be attributed in part to different land cover and permanence classes. Unexpectedly, there was no interactive (i.e. non-additive) effect of land cover and permanence classes on wetland solute chemistry. It was also shown that neither SC nor ion dominance can be used to distinguish among pond 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 pond permanence. Results also indicate that position within a fill and spill flow pathway, and whether wetlands form low-gradient surface water connections during wet conditions may also partially explain observed spatial variations in wetland pond water quality. Factors not explicitly addressed in this thesis, such as agricultural practices, soil characteristics, landscape position, and drought/deluge cycles, can also affect water quality and thus complicate prediction of wetland water quality at the watershed scale. Future comprehensive investigation of these factors could be informative. Overall though, 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. Testing of the usefulness of a wetlands position within a fill and spill sequence across the PPR is recommended.

In addition to the high spatial variation in wetland water quality documented, this thesis also provides a temporally intensive investigation of the water quality of one permanently ponded wetland (LR3) that was experimentally drained in late fall following freeze-up. Variations in salts and DOC prior to drainage were linked to hydrological processes, such as runoff, evaporation, and shallow groundwater seepage, whereas variations in N, P, and coliforms seemed to be regulated by biotic and sorption processes in addition to hydrological processes. The wetland acted as a sink and appeared to exchange solutes with the surrounding uplands. This study did not analyze specific constituent pathways and a comprehensive understanding of water quality processes and pathways occurring in isolated prairie wetlands and its uplands is currently lacking and would be beneficial.

Wetland water quality was found to be an important control of water quality in drainage water. Thus, the occurrence of high nutrient concentrations measured in the intact 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 loading. Not assessed in the wetland drainage experiment was how changing the permanently ponded wetland to a temporarily or seasonally ponded 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 seasonally ponded 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?

Different physical characteristics between ditches and spills suggest that potential impacts on downstream solute loadings will be greater for wetlands drained by ditches than spills because of the increased degree of connectivity to streams. Transport of water along ditch and spill connections was shown here not to alter water quality characteristics along the new ditch and during snowmelt period, when hydrologic connectivity in the watershed is maximized. Because the ditches acted as simple conduits, an export coefficient of 1, signifying no change in concentration or mass, is recommended for use in future wetland drainage modeling exercises. Ditch age (as a proxy for ditch condition) was not controlled for in this study. While the body of scholarly work on the study of upland agricultural drainage ditches shows that ditch condition (e.g. grassed vs. bare) can influence nutrient exports, it is unknown whether the same effects would be found for wetland ditches located in the PPR. The use of flow control structures that delay drainage in spring could serve to reduce flood impacts due to wetland drainage and could increase nutrient retention by vegetation during the growing season.

Physical characteristics of drained wetlands were most similar to those of seasonally ponded wetlands with cropped uplands. This suggests that significant amounts of TP, TDN, DOC, and K^+ may be transported downstream when wetlands are drained. Results also suggest that the efficiency with which a wetland is drained is an important factor in quantifying downstream exports. Thus, the proportion of water volume lost along with knowledge of the water quality characteristics of the wetland, although challenging to characterize, appear to be the most crucial characteristics for the accurate prediction of drainage exports through modeling. The temporally intensive measures of water quality during drainage suggested that a vertical concentration gradient existed in the LR3 wetland such that the water closest to the sediment and in the sediment porewater had the highest solute concentrations. As a result, wetland water samples obtained from the centre of the wetland at half the water depth are likely to provide an underestimate of the total solute mass in the wetland if calculated from these samples. The use of composite samples, comprised of samples collected at different locations and depths within the pond, may serve to better characterize total solutes contained in wetlands. An investigation of the spatial variability of water quality in the wetland at the time of drainage may also be useful for estimating solute exports, especially if the wetland is stratified and not completely drained.

Although water in the wetlands, ditches, spills, and drainage experiment generally did not exceed guidelines for NO_3^-, NH_4^+, and salts, the provincial objective for TP was frequently exceeded. Consequently, wetland drainage is likely to degrade downstream water quality with respect to P. As well, stream water quality may be degraded by the export of TDN along wetland drains if the receiving streams are N-limited. As the research site is a sub-basin of the Assiniboine River, nutrient loadings from drained wetlands could facilitate further eutrophication of Lake Winnipeg, into which the Assiniboine River drains. The larger study, of which this thesis is part, shows that Smith Creek subbasins experiencing greater wetland drainage indeed have poorer water quality (Westbrook et al., 2011). The study of water quality along a ditch drainage network containing multiple drained wetlands, combined with a measure of wetland connectivity, may also prove useful for better quantifying exports due to wetland drainage.

Overall, results presented herein should be useful for water resource managers and landowners in Smith Creek watershed to make informed decisions with regards to better understanding solute loading due to wetland drainage. Thesis results could also be used in the future to inform model creation suitable for simulating downstream impacts of wetland drainage.

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APPENDIX A

Nutrient variable set water quality measurements of 67 wetlands located at Smith Creek basin SK, sampled May 19 – 21, 2009. Notes indicate if the area surrounding the wetland was tilled, used for grazing, and the type of crop harvested in 2008. Semiperm is semi-permanently ponded wetlands.

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:orthoP (mass)
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 1: *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:orthoP (mass)
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

Salinity variable set water quality measurements of 67 wetlands located at Smith Creek basin, sampled May 19 – 21, 2009. Semiperm is semi-permanently ponded wetlands.

ID	Land Cover	Permanence	pH	SC (μS/cm)	Cl^- (mg/L)	HCO_3^- (mg/L)	SO_4^{2-} (mg/L)	Na^+ (mg/L)	Mg^{2+} (mg/L)	Ca^{2+} (mg/L)	K^+ (mg/L)	ICB %
W3	Grass	Seasonal	6.75	163	2.1	89.8	0.8	1.3	9.7	8.6	16.0	4.5
W4	Grass	Semiperm	7.24	414	1.8	120.5	95.7	8.9	28.2	21.1	10.0	0.0
W6	Grass	Semiperm	7.31	656	1.7	124.7	210.0	19.3	68.5	39.2	11.9	14.9
W10	Grass	Seasonal	7.44	309	3.8	80.2	22.0	2.6	15.3	12.7	14.0	11.3
W11	Grass	Permanent	7.68	695	1.8	127.8	168.4	21.0	74.2	35.6	16.0	23.9
W13	Grass	Permanent	7.24	404	1.3	79.3	49.3	5.7	20.3	9.7	7.9	4.8
W17	Wood	Permanent	6.92	158	2.0	63.0	0.6	0.3	6.2	13.8	20.3	22.4
W18	Wood	Permanent	6.55	63	1.5	25.7	0.1	0.2	1.6	4.6	11.3	17.0
W21	Wood	Seasonal	6.78	149	2.3	54.4	6.3	0.3	4.8	11.3	23.4	18.5
W22	Wood	Permanent	6.86	131	1.9	62.8	0.5	0.2	4.8	10.0	18.9	11.8
W24	Wood	Semiperm	6.77	127	1.4	63.7	0.1	0.2	3.8	11.0	19.4	11.6
W27	Wood	Permanent	6.74	111	2.4	48.2	2.3	0.5	4.3	8.7	12.7	11.3
W28	Wood	Permanent	6.76	150	2.0	70.0	7.8	0.6	6.7	11.3	13.4	4.2
W30	Wood	Semiperm	6.83	127	1.7	64.7	0.1	0.2	3.8	9.8	21.5	10.3
W32	Wood	Semiperm	6.84	57	0.0	18.3	0.1	0.1	1.2	4.9	4.6	21.0
W34	Crop	Permanent	7.19	181	2.0	72.6	9.1	2.0	7.8	13.1	8.3	5.4
W35	Crop	Permanent	7.19	211	4.9	69.1	18.3	1.9	7.7	14.1	18.1	6.5
W37	Crop	Semiperm	7.26	296	3.3	95.3	40.0	3.8	10.0	16.3	9.6	-9.8
W40	Crop	Seasonal	7.38	287	4.6	98.4	6.0	5.4	11.8	19.3	18.7	17.3
W42	Crop	Seasonal	6.80	147	9.5	56.2	1.0	2.3	4.8	11.5	13.4	7.7
W43	Crop	Permanent	7.59	230	2.9	66.9	2.7	1.7	6.6	10.4	11.2	7.2
W45	Grass	Semiperm	7.28	413	3.2	98.5	70.2	7.5	29.1	14.5	14.2	9.2
W46	Grass	Permanent	7.47	314	5.9	77.4	50.7	6.7	22.5	13.9	13.5	12.1
W47	Grass	Seasonal	7.10	414	12.4	103.4	52.1	6.9	25.4	22.3	12.8	10.1
W48	Wood	Permanent	7.75	306	2.0	113.2	39.7	4.5	17.5	18.3	12.9	2.5
W50	Wood	Seasonal	7.19	144	0.7	53.9	14.0	0.5	7.8	12.0	9.1	11.3
W54	Wood	Semiperm	8.62	451	1.9	62.7	114.6	9.9	41.2	30.7	17.3	25.2
W55	Wood	Permanent	8.42	374	0.7	66.7	44.3	2.4	14.3	19.6	7.3	9.3
W61	Wood	Permanent	7.74	422	3.9	94.6	81.7	6.8	24.5	18.2	18.2	4.6
W67	Wood	Semiperm	6.93	134	2.8	79.9	0.7	0.6	5.0	9.0	19.3	-1.0
W68	Wood	Seasonal	6.88	112	1.2	55.5	1.3	0.4	4.7	10.8	12.6	13.6
W69	Wood	Seasonal	7.06	211	1.4	87.9	8.1	0.9	7.6	13.3	34.1	14.4
W71	Wood	Semiperm	7.09	182	2.0	80.4	3.9	0.8	5.9	11.3	16.7	1.8
W72	Wood	Seasonal	6.90	170	2.7	77.7	5.4	0.9	6.9	11.0	25.8	10.8
W73	Wood	Semiperm	6.94	196	1.7	110.9	9.2	1.5	10.3	12.7	22.5	1.4
W75	Wood	Seasonal	7.24	194	0.9	70.2	2.1	1.1	4.9	12.7	13.0	7.7
W85	Crop	Semiperm	7.07	504	3.5	125.5	111.0	4.2	27.7	32.6	18.9	1.2
W86	Crop	Permanent	7.91	577	6.1	93.3	140.8	10.6	38.4	23.3	23.9	7.6
W87	Crop	Seasonal	6.84	399	9.0	99.1	34.5	4.7	16.4	21.7	33.9	15.0
W88	Crop	Semiperm	7.19	677	7.0	135.4	167.4	11.0	57.8	40.3	42.6	17.1
W89	Crop	Permanent	8.16	1694	5.1	145.4	505.4	44.0	133.4	40.4	17.0	8.1
W90	Crop	Semiperm	7.51	506	3.4	68.7	92.6	6.1	20.1	27.0	12.6	6.6
W91	Crop	Seasonal	6.96	975	13.3	130.0	329.3	18.6	68.7	74.5	39.2	8.9
W93	Crop	Seasonal	7.09	391	6.8	115.1	31.9	3.2	16.1	24.0	33.2	12.2
W96	Grass	Seasonal	7.15	868	8.5	157.8	256.1	70.2	55.0	26.3	16.1	6.6
W98	Grass	Permanent	7.56	1241	8.6	161.6	417.9	120.1	107.8	53.0	20.9	19.7
W100	Grass	Seasonal	7.27	890	5.2	117.4	284.0	60.8	68.0	47.3	13.7	15.6
W101	Grass	Permanent	7.06	792	3.7	95.1	248.2	54.7	47.5	53.1	12.7	15.1
W102	Grass	Seasonal	7.80	865	5.8	212.1	177.6	24.8	54.5	34.3	19.6	2.9

APPENDIX 2: *continued*

ID	Land Cover	Permanence	pH	SC (μS/cm)	Cl^- (mg/L)	HCO_3^- (mg/L)	SO_4^{2-} (mg/L)	Na^{++} (mg/L)	Mg^{2+} (mg/L)	Ca^{2+} (mg/L)	K^+ (mg/L)	ICB %
W103	Grass	Permanent	7.98	584	7.6	114.1	108.7	12.7	43.5	27.1	18.9	15.8
W104	Grass	Seasonal	7.94	376	7.7	124.8	6.3	1.2	10.3	18.0	25.9	1.4
W106	Grass	Permanent	7.86	776	11.2	141.6	180.9	21.9	67.6	36.8	24.1	16.7
W107	Grass	Permanent	8.12	1303	13.6	160.1	464.5	80.7	128.9	31.1	23.5	12.4
W108	Grass	Seasonal	8.54	1557	46.3	116.8	742.6	100.9	166.1	135.9	32.2	15.7
W110	Grass	Semiperm	7.17	520	9.5	53.6	156.8	27.8	28.9	20.4	13.6	5.7
W111	Crop	Semiperm	7.37	795	15.1	139.5	191.4	12.1	47.0	49.8	55.8	10.7
W112	Grass	Semiperm	7.00	806	11.5	122.8	273.6	51.2	49.2	49.9	22.0	7.5
W113	Crop	Permanent	7.58	1396	10.1	163.1	538.6	61.7	137.1	60.3	24.3	10.8
W114	Crop	Semiperm	7.13	349	8.6	118.8	30.5	4.6	15.1	21.4	33.1	8.7
W115	Crop	Seasonal	7.08	314	11.6	131.6	8.8	1.5	8.3	19.6	39.1	1.1
W116	Crop	Permanent	8.05	1780	8.2	232.9	488.9	50.3	149.4	28.7	29.6	7.9
W117	Crop	Seasonal	7.23	771	4.2	162.0	187.6	6.6	52.9	71.4	22.2	13.5
W118	Crop	Permanent	7.43	823	1.7	103.6	329.0	20.0	89.9	76.5	16.2	18.5
W119	Wood	Permanent	8.24	307	1.5	165.5	7.7	2.9	19.5	15.2	16.5	-0.1
W120	Grass	Semiperm	7.73	865	16.0	127.0	284.3	57.5	67.6	34.0	20.5	9.8
W121	Grass	Seasonal	6.59	98	2.0	33.7	2.9	0.6	3.7	8.3	9.0	18.5
W122	Grass	Semiperm	6.95	262	3.1	108.6	18.9	1.9	13.5	12.8	11.2	-3.1