VEGETATION AND SOIL PROPERTIES AS INDICATORS OF THE HYDROLOGY AND ECOLOGICAL HEALTH OF NORTHERN PRAIRIE WETLANDS IN NATIVE AND AGRICULTURAL LANDSCAPES

BY

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A Thesis
Submitted to the Faculty of Graduate Studies
In Partial Fulfillment of the Requirements
For the Degree of

MASTER OF SCIENCE

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FACULTY OF GRADUATE STUDIES

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Vegetation and Soil Properties as Indicators of the Hydrology and Ecological Health of Northern Prairie Wetlands in Native and Agricultural Landscapes

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ABSTRACT

Ross, Lisette C.M. M.Sc., The University of Manitoba, August, 2009. Vegetation and soil properties as indicators of the hydrology and ecological health of northern prairie wetlands in native and agricultural landscapes.
Major Professor: Dr. David A. Lobb.

Prairie wetlands are designed to withstand, if not thrive, on the climatic instability common to the prairies. It is often the human alteration of their hydrology that impacts wetland productivity. Surrounding land use such as cultivation can potentially alter the dynamics of wetland in-flows and the amount of water entering a wetland. This then alters the hydro-ecological processes of spring rise, summer drawdown, length of dry periods, and vegetative reproduction and establishment. Cultivation can also physically damage or destroy wetland vegetation and impact riparian and wetland soils. Most of the prairie wetlands that have escaped drainage in the prairie pothole region (PPR) now lie in watersheds devoted primarily to agricultural crop production. As a result, human activities have affected nearly every prairie wetland directly or indirectly.

The overall objectives of this research were to explore the influence of surrounding land-use on wetland vegetation and to better understand spring snowmelt and soil characteristics in wetlands of varying hydrological permanence. The main purpose is to improve our predictive capability for wetland restoration in prairie Canada and to provide recommendations for the future management and protection of wetlands in agricultural landscapes.

The objective of study 1 was to assess the impact of agricultural land-use on the distribution and diversity of vegetation in seasonal and semi-permanent wetlands. Vegetation surveys were conducted in 21 wetlands in Manitoba and Saskatchewan.

Agricultural croplands surrounded 18 wetlands, while native grasslands surrounded three wetlands. No significant differences in total, invasive, weed, exotic or native species were observed between seasonal and semi-permanent wetlands in agricultural landscapes. Native grassland wetlands had more total and native plant species in outer vegetation zones compared to wetlands surrounded by croplands and virtually no invasive, exotic or weed species in their shallow marsh zones. In comparison, invasive, weed and exotic species were more prevalent in wetlands surrounded by croplands, with the numbers of invasive and weed species significantly higher in shallow marsh and wet meadow zones, and exotic species higher in wet meadow zones. The loss of native species and the increase of invasive, weed and exotic species in wetlands surrounded by agriculture can be partly attributed to the loss of the low prairie and wet meadow zones. Research shows that agricultural impacts, such as excess sediment and nitrogen additions from soil erosion and changes in soil moisture and soil compaction, also favor the growth of invasive species in the outer vegetative zones of prairie wetlands. These factors help explain why the restoration of native species in the outer wetland vegetation zones is so difficult. Our ability to successfully restore and protect impacted wetlands in agricultural landscapes will be limited if these vegetative margins are not protected from further degradation.

The first objective of study 2 was to examine the accuracy and predictions of a single-basin hydrologic model designed to simulate spring snowmelt events into wetlands. The model was applied to nine seasonal and semi-permanent wetlands in Saskatchewan and Manitoba. Water level data for the ponds and the spatial distribution of the vegetation within all study wetlands indicate that 30% of the average annual snowmelt equivalent that occurs from December to April enters study wetlands. This 30% snowmelt event was consistent in all wetlands regardless of basin size, wetland class, surrounding land-use, or regional location.

The second objective of study 2 was to investigate the spatial extent of the effective transmission zone and its relationship to landscape features and wetland attributes, such as wetland water chemistry. Solute differences observed in the soils surrounding study wetlands were the result of two distinct ionic dominance patterns in wetland water chemistries. Calcium, bicarbonate and calcite precipitate reflected the influence of snowmelt chemistry in seasonal wetlands, while gypsum and other soluble salts in the soils surrounding semi-permanent wetlands reflected the influence of evapotranspiration and the concentration of solutes in more permanently flooded habitats. Results indicate that lateral soil water movement and the effective transmission zone are important in both seasonal and semi-permanent prairie wetlands. A similar spatial pattern for the effective transmission zones was displayed in both seasonal and semi-permanent ponds, with the outer extent of the zone around each wetland type located at a specific and consistent upland elevation relative to the basin itself. The outer extent of the zones were located closer to the wetland edge where slopes were steep and extended further away from the wetland edge where slopes were more gradual. Little is known about the landscape or wetland factors influencing the extent of the effective transmission zone in wetlands and further investigation is warranted.

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FOREWORD

This thesis has been prepared in manuscript format in adherence with the guidelines established by the Department of Soil Science at the University of Manitoba. A general introduction and an overall synthesis proceed and follow two manuscripts. Each manuscript consists of an abstract, introduction, methods, results, discussion and conclusion. The reference style used in this document follow the Canadian Journal of Soil Science. Chapter 2 has been submitted and accepted to Biodiversity and is slated for publication in 2009. Chapter 3 is to be submitted to a peer-reviewed journal, to be decided in the future. I will be the lead author for chapter 3 and co-authorship will be assigned accordingly. The manuscript for chapter 2 is as follows:

Ross, L.C.M., Lobb, D.A., Pennock, D.J., Goldsborough, L.G. and Armstrong, L.A. 2009. The vegetation of prairie wetlands in native and agricultural landscapes: implications for wetland health and restoration. Biodiversity *(in press).*

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1.0 INTRODUCTION

Wetlands are critical components of the freshwater resources in North America and one of the components most vulnerable to changes imposed by agriculture, industry and society (Murkin 1998). Northern prairie wetlands, in particular, are considered among the most productive ecosystems in the world (Murkin 1989). They are important for the water resources and ecology of the prairie region (Hayashi et al. 1998*a*) and provide society with critical services that include flood protection, clean water, atmospheric water, carbon sequestration, and recharge areas for groundwater (LaBaugh et al. 1998; van der Kamp and Hayashi 2009). They provide key habitat for wildlife (Kantrud et al. 1989; Puchniak 2002) and act as important depositories for plants (Cronk and Fennessey 2001; Galatowitsch 2006), invertebrates (Murkin and Ross 1999, 2000), and amphibians (Conant and Collins 1991; Lehtinen et al. 1999). Half of all breeding waterfowl in North America rely on prairie wetlands during the breeding season (Batt et al. 1989).

The Prairie Pothole Region (PPR) covers the southern portions of Manitoba, Saskatchewan and Alberta in Canada, and extends from central Iowa through western Minnesota, eastern South Dakota, and North Dakota in the United States (Luoma 1985; van der Valk and Pederson 2003). Two thirds of the PPR, or 457 736 km^2, is located in western Canada. The region formed approximately 10 000 years ago with the retreat of the glaciers during the Wisconsin period. Ice stagnations and ice thrusts during this time resulted in the formation of a diverse and hummocky landscape filled with millions of shallow depressions (van der Valk and Pederson 2003; Carlyle 2006). Referred to as marshes, potholes, or sloughs these wetlands lie at the interface between dryland upland landscapes and permanent aquatic systems. As a result, prairie wetlands often possess characteristics of both.

The PPR is a dynamic sub-humid to semi-arid region prone to periods of drought interspersed with periods of abundant precipitation (Johnson et al. 2004; Johnson et al. 2008). Annual changes in precipitation influence wetland hydrology by altering the timing and amount of atmospheric and ground-water inputs (Euliss et al. 2001; van der Kamp and Hayashi 2009). This in turn influences abiotic elements such as water depth, solute concentrations, temperature, and the drying rate and timing of exposed substrates (Euliss et al. 2001). These elements then become important determinants of the aquatic fauna and flora present in prairie wetlands (Murkin and Ross 1999, 2000; van der Kamp and Hayashi 2009).

Prairie potholes are routinely classified by the length of time they remain ponded and by the plant species present (Stewart and Kantrud 1971; Johnson et al. 2008). Extended wet and dry weather events affect the spatial and temporal dynamics of wetlands particularly their vegetative communities (Weller and Spatcher 1965; van der Valk and Davis 1978; Johnson et al. 2004). During periods of drought the seeds of perennial and annual plants deposited over time germinate on the exposed mudflats. The length and intensity of these droughts along with the seed bank present at the time and soil properties determine which species respond (van der Valk 2000). As the drought ends and standing water returns terrestrial plants die and wetland species flourish (van der Valk and Davis 1978). Which wetland species survive is mainly determined by the depth of the water and the length of time the pond remains flooded. The climate extremes that drive these vegetative changes have been described as the wet/dry cycle (van der Valk and Davis 1978) or the wetland cover cycle (Johnson et al. 2004). These wet/dry cycles drive the productivity of prairie wetlands. While semi-permanent or permanent wetlands require from 5 to 30 years to move through the wet/dry cycle (van der Valk and Davis 1978), ephemeral or temporary prairie wetlands can complete this wet/dry cycle every summer.

Many wetlands in the PPR are embedded within agricultural landscapes where they are subject to varying degrees of disturbance (Gleason and Euliss 1998) from activities such as drainage, cultivation, filling or burning. In Canada it is estimated that 85% of wetland loss is due to agricultural activities or urban development (Wiken et al. 2003). Estimates of wetland loss in the PPR due to agriculture or indirectly through the deterioration of marsh-edge vegetation are 1.2 million hectares (Glooschenko et al. 1993). A Canadian study of 10 000 prairie wetlands found that 79% of wetland margins had been degraded by agriculture (Turner et al. 1987). In addition to the physical destruction of the wetland edge by cultivation, soil compaction and vegetation removal, other impacts include large inputs of sediments, fertilizers, and other agricultural chemicals (Neely and Baker 1989). Freeland et al. (1999) found phosphorous up to 6 times higher in the outer vegetative margins of wetlands surrounded by cultivation. They also found that the dominant soil separate in these cultivated margins was silt, while sand was the dominant soil separate in margins surrounded by grasslands. Most studies focus on the consequences of anthropogenic stress from severe disruptions such as drainage. However many land-use impacts, such as the deterioration of the marsh-edge, tend to be less severe (Schindler 1987). The risk in not understanding or examining less severe impacts is not being able to predict future outcomes or long-term effects before it is too late.

This disruption of wetland ecosystems by agriculture opens up new ecological niches well-suited to plant invaders (Barrett 1989). These invasive species then out-compete native plants and disrupt native ecosystems, potentially decreasing biodiversity, changing watershed and ecosystem health, and creating a host of ecological changes with accompanying economic effects (Kaiser 2006). The invasion of non-native or alien organisms is now widely viewed as one of the major global threats to biodiversity (Anderson et al. 2006). Only a few studies on the characteristics of the invasive potential of plant species in prairie wetlands have taken place in the PPR (Johnstone 1986). None of these studies have been conducted in the Canadian portion of the PPR.

A second critical disruption by agriculture is its effect on surface water runoff. Catchment size is an important determinant of the amount of water entering a wetland in any given year (Fang et al. 2007). Primary water inputs include snowmelt runoff, wind-blown snow, and summer precipitation (van der Kamp and Hayashi 2009). Of these three it is snowmelt runoff that is considered to have the biggest influence on wetland hydrology. One third of annual precipitation in the PPR occurs as snowfall with snowmelt accounting for 80% of the annual surface runoff (Gray and Landine 1988; Pomeroy and Goodison 1997). Soils can freeze to depths of 1-m or more during the long and cold winters (van der Kamp et al. 2003). Infiltration has been shown to be limited in frozen soils where cultivation has destroyed macropores in the topsoil. Wetlands in these landscapes can receive significant snowmelt runoff in the spring as a result (van der Kamp et al. 2003). In comparison, snowmelt infiltration into frozen soils under grasslands can be high enough to absorb most or all of the snowmelt (Gray et al. 1985; van der Kamp et al. 2003). While Euliss and Mushet (1996) showed that water level fluctuations were greater in wetlands in grasslands than for wetlands in cultivated lands, van der Kamp et al. (2004) found no difference in the amount of snow trapped in grasslands versus cultivated lands. They concluded the main factor controlling the amount of snowmelt runoff into wetlands could not be snow trapping on the uplands.

A few site-specific wetland simulation models have been designed to better understand how hydrological events affect prairie wetlands in the PPR. Poiani and Johnson (2003) designed the WETSIM single-basin spatial model to predict vegetation and hydrology dynamics in semi-permanent ponds with a changing climate. Voldseth et al. (2007) added to this model by incorporating a land-use component. Su et al. (2000) modified an extant streamflow model to simulate water level variations in a prairie wetland in Saskatchewan. Fang (2007) developed a snow hydrology model to examine the modeling scale for snow accumulation and snowmelt in a prairie wetland in Saskatchewan. He concluded that while models have been developed to estimate snow accumulation, snowmelt and snowmelt runoff on the prairies their proper scale of application is still unknown in the prairie environment. Of the models discussed above, only Fang (2007) focused on snowmelt runoff in simulation models.

Four long-term, intensive study sites in the PPR have continually improved our understanding of the hydrology and geochemistry of prairie wetlands. These sites have also been central to the verification of the models discussed here. These long-term sites include: the Cottonwood Lake area in east-central North Dakota (Winter 2003), the St. Denis National Wildlife Area in south-central Saskatchewan (van der Kamp and Hayashi

2009), the Marsh Ecology Research Program site in south-central Manitoba (Murkin et al. 2000), and the Orchid Meadows site in eastern South Dakota (Voldseth et al. 2007). A number of studies from these sites have shown that inter-annual variability in wetlands is driven by snowmelt amounts (van der Kamp and Hayashi 2009). More current research now indicates that surface water chemistries of wetlands and their underlying soils are linked to the long-term transport of salts by shallow groundwater flow in the effective transmission zone (Hayashi et al. 1998*a,b*; van der Kamp and Hayashi 2009).

The overall objective of this study was to improve our understanding of the hydrological and human influences affecting wetlands and their health in the Prairie Pothole Region of southern Manitoba and Saskatchewan. The primary objective of Chapter 2 was to improve our understanding and predictive capability of the effects of surrounding land-use on native and invasive plant species in seasonal and semi-permanent wetlands. Chapter 3 examines the accuracy and predictions of a single-basin hydrologic model designed to simulate spring snowmelt events into wetlands. The model was applied to seasonal and semi-permanent wetlands surrounded by different land-uses and in different regions of the PPR. Predictions from the model allowed the examination of the relationship between snowmelt events of varying magnitudes and the spatial extent of wetland vegetation in the ponds. Additional research was conducted on study wetlands to improve our knowledge on the spatial extent of the effective transmission zone around ponds and its relationship to landscape features and wetland attributes, such as wetland water chemistry.

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2.0 THE VEGETATION OF PRAIRIE WETLANDS IN NATIVE AND AGRICULTURAL LANDSCAPES: IMPLICATIONS FOR WETLAND HEALTH AND RESTORATION

2.1 Abstract

More than 600 species and one-third of species at risk in Canada depend on wetlands during all or part of their life cycle. Prairie wetlands collectively represent a significant portion of the total inland wetland area in Canada. Eighty-five percent of prairie wetlands have been lost due to urban expansion or agriculture (Wiken et al. 2003). Seventy-nine percent of the wetlands that remain have vegetative margins that are impacted by cropping, grazing, or haying (Turner et al. 1987). We are positioned in western Canada to protect and restore thousands of impacted wetlands through voluntary programs. The success of these programs will be determined, in part, by our ability to predict desired outcomes using the science from studies on wetland restoration and conservation. Plants distribute themselves in pristine wetlands into four distinct vegetation zones based on their ability to tolerate flooding, to germinate during drought periods, and to compete with other plant species. Previous studies in the northern United States comparing natural wetlands to restored wetlands found the recovery of vegetation unsuccessful decades after restoration. Native plants in the outer vegetation zones had difficulty re-establishing while invasive species flourished. No study has compared the entire vegetation community of impacted wetlands to natural wetlands. Without this information our ability to predict or plan for successful restorations is limited. In this study, the vegetative communities of 21 wetlands surrounded by agriculture and native vegetation at six study sites in western Canada were compared. Ninety-two percent of

the wetlands surrounded by agriculture had at least one outer vegetation zone missing. This resulted in decreased biodiversity and allowed invasive and exotic species to spread to inner zones. In comparison, outer zones of native wetlands functioned as filters against the encroachment of invasive species. We propose that cropping practices influenced soil bulk densities and increased sediment and nitrogen loading to outer zones. These factors alone can reduce native species while favoring invasive species. Results indicate that maintaining intact riparian areas around wetland edges improves overall plant biodiversity, reduces invasive species in inner vegetative zones, and increases the likelihood of riparian plant recovery in future wetland restorations.

2.2 Introduction

Prairie wetlands tend to be small in size (<1 ha) even though collectively they represent a significant portion of the total inland wetland area of Canada and the United States (van der Valk and Pederson 2003). They range from permanent to ephemeral depending on their location in the landscape, their proximity to groundwater, the size of their catchment, and their connectivity with other water bodies. These factors, along with annual precipitation patterns, determine water depth and the length of time surface water is held in a wetland in any given year. Referred to as wetlands, marshes, potholes or sloughs, these areas represent portions of the landscape that are neither fully aquatic nor fully terrestrial. Consequently they often possess characteristics of both. Prairie wetlands contribute a major component of the water resources and ecology in the Prairie Pothole Region (PPR) (Hayashi et al. 1998) and provide society with an abundance of ecosystem goods and services that include flood protection, clean water, atmospheric water, and recharge areas for groundwater (LaBaugh et al. 1998; van der Kamp and

Hayashi 2009). They provide critical habitat for wildlife (Kantrud et al. 1989; Puchniak 2002) and act as important depositories for the biodiversity of plants (Cronk and Fennessey 2001; Galatowitsch 2006), invertebrates (Murkin and Ross 1999, 2000), and amphibians (Conant and Collins 1991; Lehtinen et al. 1999). Approximately one-fifth of all wader species breeding in the Nearctic region rely on prairie wetlands (Colwell 1987; Colwell and Oring 1990), as do half of all breeding waterfowl in North America (Batt et al. 1989).

Prairie wetlands in the PPR are under tremendous pressure. Human activities have affected nearly every wetland directly or indirectly (Kantrud et al. 1989) even though the exact number of prairie potholes lost or degraded in the Canadian portion of the PPR is difficult to estimate because no comprehensive study has ever been conducted (Glooschenko et al. 1993; Bedford 1999). Most of the wetlands that remain in the PPR exist on privately owned agricultural land, and many of these have been impacted by agricultural practices (Rickerl et al. 2000). Turner et al. (1993) found 79% of prairie wetlands adversely affected by agriculture with degradation at the wetland margins by activities such as clearing, haying, cultivating and grazing. In the agricultural areas of Alberta they found 93% of wetlands impacted. Ongoing work by Watmough and Schmoll (2007) indicate that wetlands continue to be lost and degraded in all ecoregions of the Canadian PPR.

In addition to wetland drainage and cultivation, other human impacts include the introduction of toxins and excess nutrients, vegetation removal, filling, and excavation (Galatowitsch et al. 1999*a*). If a wetland is not completely lost by drainage then these physical and chemical impacts are almost always accompanied by the replacement of indigenous vegetation by invasive species (Galatowitsch et al. 1999*b*). Wilcove et al. (1998) state that habitat loss and the spread of alien species are the two greatest threats to biodiversity and native species worldwide. Invasive plant species are those that aggressively spread after deliberate or inadvertent introduction (Rejmanek and Richardson 1996). They pose a significant threat not only to biodiversity (Wilcove et al. 1998; Cronk and Fennessey 2001; Chornesky and Randall 2003; Brown and Gurevitch 2004), but to native communities by altering ecosystems processes (Bart and Hartman 2000; Mack et al. 2001; Ehrenfeld 2003) and community structure (Galatowitsch et al. 1999*b*; Gratton and Denno 2005; Galatowitsch 2006).

We are positioned in western Canada to restore thousands of wetlands impacted by cultivation or those lost due to drainage or urban expansion. Throughout the 1980s the United States restored thousands of wetlands in the mid-continental United States through regulatory and voluntary programs. Restoration activity was particularly high in states such as South Dakota, North Dakota, Iowa and Minnesota. Many of these restorations were achieved by interrupting drainage lines or removing drainage tiles (Mulhouse and Galatowitsch 2003). Cropped for decades and isolated within agricultural landscapes, the revegetation of these wetlands relied mainly on the natural recolonization of plant communities. The studies that followed comparing the vegetative communities of restored wetlands to natural wetlands found that the recovery/reassembly of wetland vegetation in restored sites did not resemble that of natural wetlands even decades after post-restoration (Galatowitsch and van der Valk 1996*a*; Zedler 2000; Mulhouse and Galatowitsch 2003; Galatowitsch 2006; Aronson and Galatowitsch 2008). This was particularly true for plant species located in the outermost wetland margins, such as those plants that inhabit the wet meadow and low prairie zones. Poor success in these locations was attributed to biotic factors such as the differential in colonization efficiency between invasive and native species, and abiotic factors such as unfavorable hydrology, past cropping practices, length of disturbance and soil properties (Murphy and Lemerle 2006; Aronson and Galatowitsch 2008).

Plants in pristine wetlands are distributed into spatially distinct vegetative or concentric zones based on their ability to tolerate flooding, to germinate during drought periods, abundance in the seed bank, and capacity to compete with other species (Stewart and Kantrud 1971; van der Valk and Welling 1988; van der Valk 2000). Stewart and Kantrud (1971) identified five vegetation zones in wetlands: permanent open water (OW), deep emergent marsh (DM), shallow emergent marsh (SM), wet meadow (WM), and low prairie (LP). These zones can be used to classify wetlands based on their presence or absence at the time of classification. For example, a Class III seasonal wetland will possess a central shallow marsh zone but lack the deep emergent zone found in a Class IV wetland. The five wetland Classes described by Stewart and Kantrud (1971) are: ephemeral, temporary, seasonal, semi-permanent and permanent (Table 2.1).

Table 2.1 Classification of prairie wetlands (after Stewart and Kantrud 1971).

Wetland Class:	I	II	III	IV	V
Water Permanence:	Ephemeral	Temporary	Seasonal	Semi-permanent	Permanent
Flooding Duration: (normal conditions)	3 to 4 weeks each spring	1 to 2 months each spring	3 to 4 months each year	Remains flooded 4 out of 5 years	Remains flooded 19 out of 20 years
Vegetative Zones Present:	Low prairie	Low prairie Wet meadow	Low prairie Wet meadow Shallow marsh	Low prairie Wet meadow Shallow marsh Deep marsh	Low prairie Wet meadow Shallow marsh Deep marsh Open water

Reversing land-use degradation is often an uncertain endeavor with variable success in spite of extensive efforts to restore ecosystems worldwide (Galatowitsch 2006). Success is most challenging in extensively fragmented landscapes on highly modified sites (Hobbs 2003) and predicting which plant species are likely to become successful invaders is clearly deficient (Crawley 1987; Rejmanek and Richardson 1996). While extensive work has occurred comparing the vegetation of pristine wetlands to restored wetlands in the northern Great Plains Region of the United States, no studies to date have included a comparison between pristine wetlands and wetlands surrounded by agriculture. In this study we compare the floristic composition of 21 seasonal and semi-permanent wetlands located in Saskatchewan and Manitoba that vary in land-use from undisturbed native wetlands (native wetlands) to wetlands surrounded by agriculture (cropped wetlands). The objectives of this study were to: (1) determine if wetland vegetation differs across land-use gradients, (2) determine if wetland vegetation differs within land-use gradients but between wetland Class, and (3) determine if other wetland covariates, such as slope, wetland size, vegetation zone size or soil salinity affect the distribution or diversity of plant species in prairie wetlands.

2.3 Methods

2.3.1 Site Descriptions

Twenty-one wetlands located on six study sites in Manitoba and Saskatchewan were surveyed in the summer of 2006 (Figure 2.1). Nine wetlands were classified as Class III wetlands, while twelve were classified as Class IV wetlands (Stewart and Kantrud 1971). Fifteen of the wetlands were located on working farms that conduct zero tillage (no-till). Wetlands on these farms were surrounded by either spring sown cereals or oilseeds. All the farms had been in operation for 45 years or more. The six remaining wetlands were located on the St. Denis National Wildlife Area (SDNWA) in Saskatchewan. Three Class III wetlands at this location were surrounded by croplands, while three Class IV wetlands were bordered by native grasslands that had never been broken by cultivation (Table 2.2).

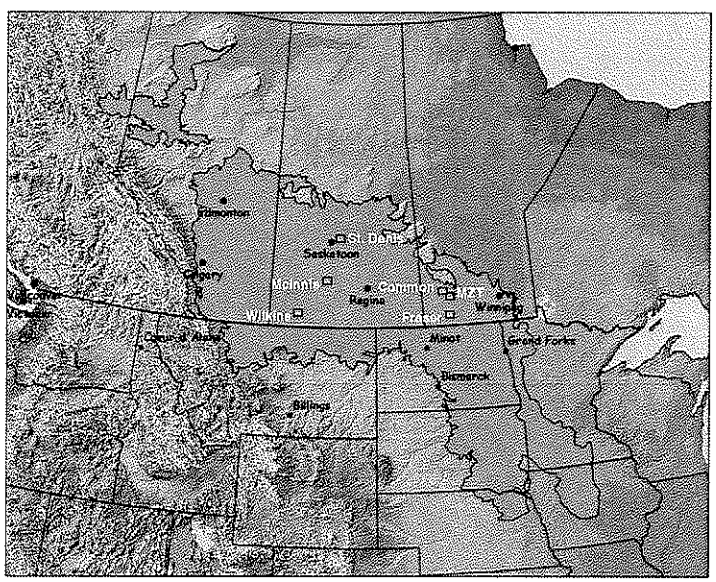

Figure 2.1 The locations of research sites in Manitoba and Saskatchewan.

Table 2.2 Summary of characteristics of wetlands including location, surrounding land-use, wetland class, wetland size, mean slope, surface water electrical conductivity and annual precipitation.

Site	Longitude Latitude	Pond #	Upland Land Use	Wetland Class	Size ha	Mean Slope %	Electrical Conductivity µS/cm	Total Annual Precipitation[1] [rainfall only] mm
Manitoba								
MZTRA:	99°56'00"W	232	Cropland	IV	0.10	2.1	288	460
	50°03'05"N	222	Cropland	IV	1.00	3.7	1322	[340]
		216	Cropland	IV	0.26	2.9	1316	
Common:	100°13'22"W	1	Cropland	III	0.23	4.7	419	441
	50°12'37"N	2	Cropland	IV	0.75	0.9	413	[337]
		3	Cropland	IV	1.10	3.9	708	
Fraser:	99°54'32"W	1	Cropland	IV	0.62	1.2	529	543
	49°21'54"N	2	Cropland	III	1.20	2.4	294	[411]
		3	Cropland	III	0.30	4.7	263	
Saskatchewan								
SDNWA:	106°05'06"W	109	Cropland	III	0.56	4.1	300	350
	52°12'38"N	117	Cropland	III	0.30	3.1	403	[265]
		120	Cropland	III	0.48	6.3	293	
		65	Native	IV	2.60	3.7	2260	
		66	Native	IV	7.50	6.1	8680	
		67	Native	IV	2.70	7.2	2350	
McInnis:	106°49'49"W	1	Cropland	IV	0.32	5.1	389	373
	50°39'16"N	2	Cropland	IV	0.85	16.5	1887	[282]
		3	Cropland	IV	0.97	8.3	2868	
Wilkins:	108°20'02"W	1	Cropland	III	0.40	4.9	195	382
	49°30'56"N	2	Cropland	III	0.44	5.0	219	[279]
		3	Cropland	III	0.50	5.1	144	

[1] Based on 30 year climate normals (Environment Canada 2009)

All three Manitoba sites are located in the Aspen Parkland area of Manitoba (Table 2.2). The Common farm consisted of one Class III and two Class IV wetlands. The landscape is hummocky with elevations ranging from 563 meters above sea level (masl) to 570 masl. Surface soil textures consists mainly of silty clay loams (SiCL) and clay loam (CL). Soils in the upland areas at the farm are predominantly Black Chernozems formed over glacial tills. Rego Black Chernozems dominate the shoulder slope positions around wetlands, while Orthic Black Chernozem and Gleyed Rego Black

Chernozem soils occur mainly in the backslope and footslope positions. Gleyed Rego Black Chernozems and Gleysols exist primarily in the riparian zones. The crops surrounding wetlands in 2006 consisted of oats (*Avena sativa*) and canola (*Brassica napus*). The electrical conductivities (EC) of water in these basins are slightly brackish.

Soils at the Fraser farm displayed a similar distribution pattern to the Common farm. Black Chernozem soils dominate upper slope positions, while Gleysol soils dominate the riparian zones within wetlands. The two Class IIII and one Class IV wetlands at this site were bordered by wheat in 2006. Wetlands are similar in size to wetlands at the Common sites, as were the EC's. Elevations at the site ranged from 486.0 masl to 495.0 masl.

The Manitoba Zero Till Research Association farm (MZTRA) farm located in south-central Manitoba exists on an undulating to hummocky landscape that contains numerous wetlands of varying permanence. Three Class IV wetlands bordered by flax were randomly selected in 2006. Electrical conductivities were in the slightly brackish range for the smaller wetlands and moderately brackish for the largest wetland (Table 2.2). Black Chernozem soils have developed over calcareous glacial tills on the upland areas at the MZTRA farm, while mostly Gleysol soils have developed within wetlands (Podolsky and Schindler 1993). Twenty-two percent of the soils on the farm are considered moderately saline (4-8 mS cm^{-1}), with higher salinity levels occurring in soils adjacent to more permanently flooded wetlands (Podolsky and Schindler 1993). Elevations for the wetlands at the farm range from a low of 497 masl to a high of 507 masl.

The McInnis farm in Saskatchewan is located in the Missouri Coteau Upland region of the Swift Current map area (Ayres et al. 1985). The Brown Chernozem soils of this area occur on a knob and kettle morainic landscape developed from shale-modified glacial tills. Soils in the region tend to be moderately calcareous loam soils. Rego Brown Chernozem soils dominate the shoulder slope position around the study wetlands, while Orthic Brown Chernozems are mainly located in footslope positions. Soils located within, or near, the wetland riparian zones tend to be Orthic Humic Gleysols. Elevations at the site range from 724.0 masl to 737.9 masl. The two larger wetlands at this site both had moderately brackish EC's.

The Wilkins farm is located 20 km south of Shaunavon, Saskatchewan in the southwest corner of the province. The Dark Brown Chernozem soils of this area have formed on an undulating landscape characterized by a sequence of gentle slopes extending from smooth rises to gentle hollows (Saskatchewan Centre for Soil Research, 1989). Rego Brown Chernozem soils dominate the shoulder slope position around study wetlands, while Orthic Brown Chernozems have mainly developed in lower slope landscape positions. Soils located within, and nearest, the wetland riparian zones are either Humic or Luvic Gleysols. This site had the highest elevations of all the sites ranging from 1021.0 masl to 1034.0 mas. The EC's of the three wetlands at this site were all fresh.

The St. Denis National Wildlife Area (SDNWA) is located approximately 40 km east of Saskatoon (106°16'W, 51°13'N). The 385 ha site was acquired in 1968 by the Canadian Wildlife Service for the purpose of studying wildlife and wetlands within agricultural landscapes. Aerial photographs of the area show that most of the site has been under dry land agriculture consisting of a rotation of summer fallow and cereal crops since at least 1951, except for a 43 ha section of native grasslands that has never been broken by cultivation. Over 200 temporary and permanent wetlands have been classified on the SDNWA (Hogan and Conly 2002). Three Class IV wetlands surrounded by native grasslands were surveyed at St. Denis, as where three Class III wetlands surrounded by cultivation. The Class IV wetlands surrounded by native grasslands at this location were noticeably larger and their EC's higher as well.

Soils at St. Denis include thin Rego Dark Brown Chernozems on the shoulders of knolls, thicker Orthic Dark Brown Chernozems in mid-slope and toe-slope positions, and Eluviated Dark Brown Chernozems and Gleysolic Humic Luvic Chernozems in wetlands (Yates 2006). Calcareous Dark Brown Chernozem soils can also be found at shoulder and mid-slope positions as well as in some toe-slope positions (Bedard-Haughn 2001). Soil textures range from silty loam in depressions to loam at topographically high positions (Yates 2006). The topography for the area is described as moderately rolling knob-and-kettle and knoll-and-depression moraine. Elevations for the three wetlands bordered by native grasslands ranged from 543 masl to 561 masl, compared to 548 masl to 560 masl for the three wetlands surrounded by croplands.

2.3.2 Vegetation Surveys

Vegetation transects were conducted on all wetlands during late July or early August 2006. Three vegetation transects per wetland were done at the MZTRA and St. Denis farms, while one transect per wetland was conducted on the Common, Fraser, McInnis and Wilkins farms. Methods for taking vegetation surveys followed Bonham (1989). A 1-m wide belt transect was randomly placed on each wetland starting just below the lowest point of the aquatic rooted vegetation zone and extending to the upland area where terrestrial vegetation dominated. Each transect ran perpendicular to the vegetation zones of the wetland so that all the zones could be captured and determined. Dominant plant species using plant lists provided in Stewart and Kantrud (1971) were used to identify the vegetative zones present around each wetland (Table 2.3). Transect lengths depended on the extent of hydrophytic vegetation surrounding each wetland. Water depth was recorded at 1-m intervals along each transect.

Table 2.3 Key indicator species for prairie wetland vegetation zones (after Stewart and Kantrud 1971).

Wetland Zone	
Low Prairie:	**Primary Species:** *Agropyron sp., Aster ericoides, Cirsium arvense, Melilotus sp., Poa pratensis, Potentilla anserina, Sonchus arvensis, Symphoricarpos occidentalis*
	Secondary Species: *Andropogon gerardii, Elymus canadensis, Festuca sp., Stipa viridul*
Wet Meadow:	**Primary Species:** *Aster simplex, Calamagrostis inexpansa, Eleocharis sp., Hordeum jubatum, Mentha arvensis, Stachys palustris, Carex sp., Juncus balticus*
	Secondary Species: *Poa palustris, Potentilla anserina, Sonchus arvensis, Cirsium arvense*
Shallow Marsh:	**Primary Species:** *Carex antherodes, Carex aquatilis, Beckmannia syzigachne, Phalaris arundinacea, Scirpus sp., Scolochloa festucacea*
	Secondary Species: *Lemna sp.*
Deep Marsh:	**Primary Species:** *Typha latifolia, Typha augustofolia, Scirpus validus, Scirpus acutus*
	Secondary Species: *Scolochloa festucacea, Carex antherodes*

All plant species were identified and recorded in each successive 1 x 1-m quadrat along each transect. Cover data were directly assessed in the field and the percent cover-abundance of all plant taxa in each quadrat was recorded as a value from 5 to 100%, or trace (<5%). Nomenclature of plant species followed Looman and Best (1979) and the United States Department of Agriculture Natural Resources Conservation Services Plants Database (USDA 2009). All plant species identified were categorized by their life history (annual, biennial, or perennial), origin (native or exotic), designation (invasive, weed or non-weed), and growth form (tree, graminoid, forb/broadleaf or subshrub/shrub). The distinction between invasive versus weed follows the definition by Richardson et al. (2000). Invasive plants are those plants that produce reproductive offspring, often in very large numbers, at considerable distances from parent plants. They have the potential to spread over a considerable area and out-compete other species, thus threatening the native biological diversity of natural or semi-natural ecosystems (IUCN 1999). Invasive plants can be either exotic (introduced) or native (indigenous). Weeds grow in sites where they are not wanted and which usually have detectable economic or environmental effects (Richardson et al. 2000). It is important to note that not all invasive species are designated as weeds, nor are all weeds considered invasive. *Phalaris arundinacea* and *Juncus balticus* both have the potential to be invasive yet they are not recognized as weeds in wetlands. Numerous provincial, national and North American sources were used and cross-referenced to determine the designation of each species recorded in this study (Looman and Best 1979; Tannas, 1997; Royer and Dickinson 1999; Bubar et al. 2000; Canadian Botanical Conservation Network 2009; MAFRI 2009; Saskatchewan Agriculture 2009; USDA 2009).

Relative cover (RC), relative frequency (RF), and an importance value (IV) were calculated for all species on each transect using the following formulas (Mueller-Dombois and Ellenberg 1974; Doumlele 1981; Perry and Hershner 1999):

Relative Frequency (RF)

= (Species frequency/Σ frequencies for all species) *100

Relative Cover (RC)

= (Species mean/Σ means for all species) *100

Importance Value (IV)

= RC + RF

An importance value of ≥ 20 for zone indicator species was used to determine the exact start and end of all vegetation zones on each transect. This information was then used to determine and compare the effects of zone length and zone slope on plant diversity within, and between wetlands. All vegetation zones were marked in the field using either a Trimble® GeoXT™ or Sokkia Total Station to collect Global Positioning System (GPS) coordinates so that the size of each vegetation zone (m^2) and the size of each wetland (ha) could be calculated. A differential correction process improved data accuracy to the subfoot (<30-cm) level and in most cases less than 10-cm accuracy. Elevation data were collected for the McInnis, Wilkins, Common, and Fraser sites in the fall of 2007 to assess the effect of slope on species distribution and abundance. A survey grade quad-mounted Trimble® GeoXT™ collected GPS coordinates along linear track lines at 10-m increments on the uplands and at 5-m increments closer to, and within, each wetland basin. LiDAR data of the St. Denis and MZTRA sites provided elevation information for these two locations. All coordinate data was imported into ArcGIS 9.3 (ESRI Canada Limited) where spatial features were created for each wetland so that wetland size and zone length and size could be calculated. ArcGIS Spatial Analyst was used to interpolate surface information from the LiDAR and Digital Elevation Model (DEM) data. This allowed the calculation of degree of slope (%) for each zone.

2.3.3 Soil Salinity

To determine the effect of soil salinity on plant distribution and abundance a Geonics Ltd. electromagnetic meter (EM38) was used to measure the apparent soil electrical conductivity (ECa) on vegetation transects. The EM38 was calibrated according to manufacturer's directions prior to use and re-calibrated numerous times during data collection in the field. ECa readings (mS/m) in both the horizontal (EMh) and vertical (EMv) dipole positions were taken by placing the EM38 directly on the soil surface every meter along each transect. Readings could only be taken where no standing water was present. The EMh dipole position measured ECa from the soil surface to a depth of approximately 60-cm below the soil surface. The EMv dipole position measured ECa between 60-cm and 120-cm below the soil surface. A number of soil samples representing the range of ECa values recorded at the MZTRA and St. Denis sites were collected using a hand auger to a depth of 60-cm. Norman (1990) indicates that only the electrical conductivity of the soil-saturation extract (ECe) can be directly related to plant behavior. Therefore, soil samples were air dried, ground and sieved (< 2-mm) and ECe measured using the saturated paste method. Linear regression between ECa and the measured ECe resulted in an R^2 value of 70% for the St. Denis site and 99% for the MZTRA site. All EMh and EMv data from the vegetation transects were converted to ECe (dS/m) using the linear equation for each locale.

2.3.4 Statistical Analysis

SAS' PROC GLIMMIX (SAS Institute Inc. 2008) was utilized for fitting generalized linear mixed models (McCulloch and Searle 2001) to the number of species per zone. Specifically, negative binomial or Poisson models with log link functions appropriate for count data (McCulloch and Searle 2001) were utilized.

Wetland Class (III vs. IV), type (cropped vs. native), and wetland zone (DM, SM, WM, and LP) were considered as fixed categorical predictors. Wetland class and zone were combined into a single factor for purposes of model convergence when data were sparse (i.e. analyses of number of weeds and exotics). Since differences among wetland zones were of chief interest, the effect of zone was allowed to interact with all other fixed categorical effects. An interaction between wetland Class and type was not possible as there were no native Class III wetlands in this study. Site, wetland, transect, and their interactions with zone were treated as random effects, in recognition of the experimental design. Statistical contrasts of least square means (LSM) were used to explore the nature of significant fixed effects, with Bonferroni multiplicity-adjusted p-values utilized when more than two groups were compared. A significance level of ($p \leq 0.10$) was deemed significant due to greater error variability at the landscape scale (van Kessel et al. 1993; Pennock et al. 1994; Manning et al. 2001).

Continuous covariates included slope, polygon size, zone length, wetland size, EMv and EMh. The effects of these covariates were evaluated by adding them one-at-a-time to models containing the categorical predictors and random effects detailed above. Covariate*wetland zone and covariate*wetland type interactions were included to investigate wetland zone- and wetland type-specific covariate effects. Statistical contrasts of effect estimates were used to help understand significant interactions. Permutation t-tests were used to study life history trends between invasive perennials, native perennials, and annuals by wetland type and by vegetative zone. Lastly, to determine the dominant species in this study the number of zones where a species had an IV≥20 and their corresponding RC and RF values were tallied. Species were then ranked from highest to lowest based on their relative frequency of zone-dominance.

2.4 Results

2.4.1 Species Diversity and Dominance by Origin and Designation

A total of 86 plant species were observed across all wetlands (Appendix A). Sixty-three species were native, 23 were exotic, 29 were invasive, and 26 were weeds. There were 63 species identified in the 18 cropped wetlands, compared to 52 species in the 3 native wetlands at St. Denis. There were nearly two times the number of invasive and weed species in cropped versus native wetlands (25 vs. 13 and 23 vs. 12, respectively), and fewer native species (42 vs. 45). The number of exotic species in cropped wetlands was three times that found in native wetlands (21 vs. 7). Table 2.4 lists the dominant native, invasive, weed and exotics species in cropped and native wetlands.

Table 2.4 List of dominant taxa (IV≥20), by origin (native vs. exotic) and designation (invasive and/or weed), for cropped and native wetlands.

	Cropped Wetlands	%[1]	Native Wetlands	%
Native	*Carex atherodes*	26.7	*Calamagrostis inexpansa*	26.9
	Hordeum jubatum	13.3	*Scholochloa festucacea*	19.2
	Phalaris arundinacea	8.3	*Eleocharis sp.*	11.5
	Scholochloa festucacea	6.7	*Agropyron trachycaulum*	7.7
	Beckmannia syzigachne	6.7	*Stipa comata*	7.7
	Polygonum amphibium	5.0	*Juncus balticus*	3.8
	Aster ericoides	5.0	*Poa palustris*	3.8
	Juncus balticus	3.3	*Schoenplectus maritimus*	3.8
Invasive	*Bromus inermis*	26.7	*Bromus inermis*	66.7
	Hordeum jubatum	26.7	*Juncus balticus*	33.3
	Phalaris arundinacea	16.7		
	Cirsium arvense	13.3		
	Polygonum amphibium	10.0		
	Juncus balticus	6.7		
	Poa pratensis	6.7		
	Sonchus arvensis	6.7		
Weed	*Bromus inermis*	50.0	*Bromus inermis*	66.7
	Hordeum jubatum	40.0	*Achillea millefolium*	33.3
	Cirsium arvense	20.0		
	Sonchus arvensis	10.0		
	Equisetum arvense	5.0		
	Polygonum lapathifolium	5.0		
	Achillea millefolium	5.0		
Exotic	*Bromus inermis*	57.1	*Bromus inermis*	100.0
	Cirsium arvense	28.6		
	Sonchus arvensis	14.3		
	Poa pratensis	14.3		
	Festuca ovena	7.1		
	Triticum spp.	7.1		

[1] The number of zones (%) in which a species is identified as dominant.

While a plant species may often occur along a vegetation transect (RF), it may only cover a very small portion of the overall transect (RC). Therefore, the IV value for an individual plant takes into account how often it occurs as well as how much area within a transect it covers. *Carex atherodes*, *Hordeum jubatum*, and *Phalaris arundinacea* were the dominant native species in cropped wetlands, while *Calamagrostis inexpansa, Scolochloa festucacea* and *Eleocharis sp.* were dominant species in native wetlands (Table 2.4). None of the dominant species in native wetlands were invasives, while both *H. jubatum* and *P. arundinacea* in cropped wetlands were invasives. *Bromus* *inermis,* which is considered both a weed and an invasive, was the most common invasive in both wetland types. In the native wetlands only *B. inermis, Juncus balticus* and *Achillea millefolium* produced a high enough IV to be listed in the invasive, weed or exotic categories, whereas numerous species warranted a high enough IV to be listed in cropped wetlands. *Bromis enermis*, *H. jubatum*, *Cirsium arvense, Sonchus arvensis* and *P. arundinacea* were by far the most prevalent invasive, exotic and weedy species in cropped wetlands.

2.4.2 Species Diversity and Dominance within Vegetative Zones

Eleven of the 12 wetlands surveyed at the Common, Fraser, McInnis, and Wilkin farms were missing either their WM or LP zone. Six WM zones were absent, while five LP zones were missing (Table 2.5). Either the indicator plant species for these zones were missing (Stewart and Kantrud 1971) or specific species were not dominant enough to indicate that the zone was present (i.e. IV $\leq$ 20). None of wetlands at the MZTRA and St. Denis sites had missing vegetation zones.

Table 2.5 List of vegetation zones present or absent on study wetlands.

Wetland	Pond #	Class IV	Class III	Deep Marsh (DM)	Shallow Marsh (SM)	Wet Meadow (WM)	Low Prairie (LP)	Vegetative Zones Missing from Wetland
Common	1	√		x	x		x	WM
Common	2		√		x	x	x	
Common	3		√		x	x		LP
Fraser	1	√		x	x	x		LP
Fraser	2		√		x		x	WM
Fraser	3		√		x		x	WM
McInnis	1		√		x		x	WM
McInnis	2	√		x	x		x	WM
McInnis	3	√		x	x		x	WM
Wilkins	1		√		x	x		LP
Wilkins	2		√		x	x		LP
Wilkins	3		√		x	x		LP
MZTRA	232	√		x	x	x	x	
MZTRA	222	√		x	x	x	x	
MZTRA	216	√		x	x	x	x	
St. Denis	109		√		x	x	x	
St. Denis	117		√		x	x	x	
St. Denis	120		√		x	x	x	
St. Denis	65	√		x	x	x	x	
St. Denis	66	√		x	x	x	x	
St. Denis	67	√		x	x	x	x	
Total:	21	10	11	10	21	15	16	11

Results from the vegetation surveys indicate that *Bromis inermis* was the dominant species in the LP zone of Class III cropped wetlands and Class IV native wetlands, while *Hordeum jubatum* was the dominant plant in the LP zone of Class IV cropped wetlands (Table 2.6). All the dominant species in the LP zone of Class IV cropped wetlands were invasive except for *Agropyron smithii*. *Scolochloa festucacea* was a dominant species in the SM zones of all wetlands in this study, but it was most dominant in native wetlands. Approximately 40% of all SM zones in Class III and IV cropped wetlands were dominated by *Carex atherodes*. *Phalaris arundinacea*, *Polygonum amphibium,* and *Beckmannia syzigachne* were also dominant species in the SM zone of Class III and Class IV cropped wetlands. Two invasive species, *Cirsium arvense* and *H. jubatum*, dominated the WM zone in Class III and Class IV cropped wetlands respectively, while *Calamgrostis inexpensa* dominated the WM zone of native wetlands. Lastly, *C. atherodes* and *Typha glauca* were the main species occurring in the deep marsh zone of Class IV cropped wetlands compared to *Schoenoplectus maritimus* and *Scirpus validus* in native wetlands.

2.4.3 Plant Response Between Wetland Types

Statistical analyses of wetland types using all 21 wetlands found there were no significant differences in the overall number of plant, invasive, weed, exotic or native species between Class III and Class IV cropped wetlands in agricultural landscapes (Table 2.7). There were significant differences between Class IV cropped wetlands and Class IV native wetlands, with substantially more total species in the LP zone of Class IV native wetlands than in the LP zone of Class IV cropped wetlands (Figure 2.2). There was also a greater number of native plant species in the LP zone of Class IV native wetlands. In comparison, invasive, weed and exotic plant species were more prevalent in at least one vegetative zone in Class IV cropped wetlands. While the number of invasive and weed species were significantly higher in the SM and WM zones of cropped wetlands, exotic species were significantly higher in the WM zones.

Wetlands in this study were spread across a prairie region that differs markedly in annual precipitation and soil types. Consideration must be given to the effect this might have on wetland plants that are often closely linked to both soil moisture and flooding conditions. The analysis of the native and cropped wetlands at St. Denis provides a comparison of wetlands that occur on similar soils and receive similar amounts of annual precipitation. Even though we incorporated random effects of the site-to-site variability in our initial analyses, we nonetheless anticipated greater precision in the effects estimated from an analysis restricted to the St. Denis wetlands. Similar to the findings for all wetlands, total numbers of species were significantly higher in the LP zones of Class IV native wetlands. Native species were also higher in the LP and WM zones of native wetlands (Table 2.8). Invasive and weed species were higher in the SM zone of Class III cropped wetlands, with no significant differences observed for the number of exotic species in native versus cropped wetlands.

Table 2.6 List of dominant taxa (IV≥20) by wetland type in low prairie, wet meadow, shallow marsh, and deep marsh zones. An (x) indicates species that are invasive.

Zone	WETLAND TYPE Crop - Class III	%[1]	Crop - Class IV	%[1]	Native - Class IV	%[1]
Low Prairie	*Bromus inermis* (x)	54.5	*Hordeum jubatum* (x)	44.4	*Bromus inermis* (x)	22.2
	Populus tremuloides	18.2	*Agropyron smithii*	11.1	*Stipa comata*	22.2
	Festuca ovina	9.1	*Bromus inermis* (x)	11.1	*Achillea millefolium*	11.1
	Poa pratensis (x)	9.1	*Cirsium arvense* (x)	11.1	*Agropyron trachycaulum*	11.1
	Sonchus arvensis (x)	9.1	*Poa pratensis* (x)	11.1	*Calamagrostis inexpansa*	11.1
			Sonchus arvensis (x)	11.1	*Juncus balticus* (x)	11.1
					Poa palustris	11.1
Wet Meadow	*Cirsium arvense* (x)	27.3	*Hordeum jubatum* (x)	33.3	*Calamagrostis inexpansa*	66.7
	Agropyron smithii	9.1	*Aster ericoides*	22.2	*Agropyron trachycaulum*	11.1
	Agropyron trachycaulum	9.1	*Juncus balticus* (x)	22.2	*Distichlis stricta*	11.1
	Aster ericoides	9.1	*Equisetum arvense* (x)	11.1	*Eleocharis sp.*	11.1
	Bromus inermis (x)	9.1	*Poa palustris*	11.1		
	Carex atherodes	9.1				
	Hordeum jubatum (x)	9.1				
	Polygonum lapathifolium (x)	9.1				
	Triticum spp.	9.1				
Shallow Marsh	*Carex atherodes*	41.2	*Carex atherodes*	38.5	*Scolochloa festucacea*	66.7
	Scolochloa festucacea	17.6	*Phalaris arundinacea* (x)	23.1	*Eleocharis sp.*	33.3
	Lemna minor	11.8	*Beckmannia syzigachne*	15.4		
	Phalaris arundinacea (x)	11.8	*Aster simplex*	7.7		
	Polygonum amphibium (x)	11.8	*Polygonum amphibium* (x)	7.7		
	Beckmannia syzigachne	5.9	*Scolochloa festucacea*	7.7		
Deep Marsh			*Carex atherodes*	37.5	*Schoenoplectus maritimus*	25.0
			Typha glauca (x)	25.0	*Scirpus validus*	25.0
			Beckmannia syzigachne	12.5	*Scolochloa festucacea*	25.0
			Eleocharis sp.	12.5	*Triglochin maritima*	25.0
			Typha latifolia	12.5		

[1] The relative frequency of zones (%) in which each species is identified as dominant.

Table 2.7 Generalized linear mixed modeling results for the effects of wetland type, class, and zone on total species, number of invasives, number of weeds, number of exotics, and number of natives (n = 21 wetlands).

		Total Species			Invasive			Weed			Exotic			Native		
		Num df, Den df[1]	F	p-value	Num df, Den df	F	p-value	Num df, Den df	F	p-value	Num df, Den df	F	p-value	Num df, Den df	F	p-value
Fixed Effects:																
Class_zone		6, 4.5	5.26	0.054	5, 51.8	2.65	0.033	5, 9.9	2.09	0.150	5, 7.2	2.90	0.096	6, 95	5.11	0.0001
Type		1, 10.4	0.08	0.781	1, 64.9	7.56	0.008	1, 65.0	8.17	0.006	1, 6.8	6.42	0.040	1, 7.5	4.86	0.061
Class_zone*type		3, 4.0	4.83	0.080	2, 85.0	8.96	0.0003	2, 10.2	5.34	0.026	1, 5.8	4.10	0.091	3, 95	9.10	0.0001
Statistical Contrasts of Least Square Means:	**Zone, Class, and/or Type**	**F**	**p-value**	**Nature of Effect[2]**	**F**	**p-value**	**Nature of Effect**	**F**	**p-value**	**Nature of Effect**	**F**	**p-value**	**Nature of Effect**	**F**	**p-value**	**Nature of Effect**
Differences among Types (by Zone, Class IV only)[3]	DM	0.09	0.764	---	---[4]	---	---	---	---	---	---	---	---	0.04	0.840	---
	SM	2.71	0.152	---	9.04	0.004	Cr>Nat	7.36	0.008	Cr>Nat	---[5]	---	---	0.58	0.453	---
	WM	0.02	0.902	---	3.98	0.061	Cr>Nat	5.14	0.054	Cr>Nat	8.57	0.018	Cr>Nat	2.94	0.104	---
	LP	10.37	0.036	Nat>Cr	1.84	0.190	---	0.37	0.564	---	0.25	0.640		27.86	0.0001	Nat>Cr
Differences among Classes (by Zone[6], Crop only)	SM	0.01	0.930	---	0.06	0.814	---	0.005	0.944	---	0.22	0.643	---	0.19	0.668	---
	WM	2.53	0.169	---	2.07	0.164	---	1.346	0.278	---	1.39	0.279	---	1.23	0.274	---
	LP	0.98	0.352	---	1.39	0.246	---	0.608	0.453	---	0.08	0.782	---	1.85	0.178	---
Differences among Zones (by Class, Type)	Nat, Cl IV	7.81	0.002	LP>DM	7.32	0.001	LP,WM>SM	5.49	0.028	LP>SM	5.09	0.067	LP>WM	6.97	0.0003	LP, WM>DM; LP>SM
	Cr, Cl IV	7.04	0.008	WM>DM	3.33	0.041	WM>SM, LP	1.51	0.261	---	3.29	0.079	WM>SM	8.06	0.0001	SM,WM>DM, LP
	Cr, Cl III	0.36	0.718	---	1.27	0.287	---	0.75	0.499	---	2.32	0.153	---	4.14	0.019	SM>LP

[1] Denominator degrees of freedom (df) for the F-tests were determined through Satterthwaite approximations.
[2] For differences among zones, nature of effect determined from Bonferroni-adjusted multiple comparisons.
[3] There are no Native, Class III wetlands.
[4] There are no invasives, weeds, or exotics in the DM zone of native wetlands.
[5] There are no exotics in the SM zone of native wetlands.
[6] There is no DM zone in Class III wetlands.

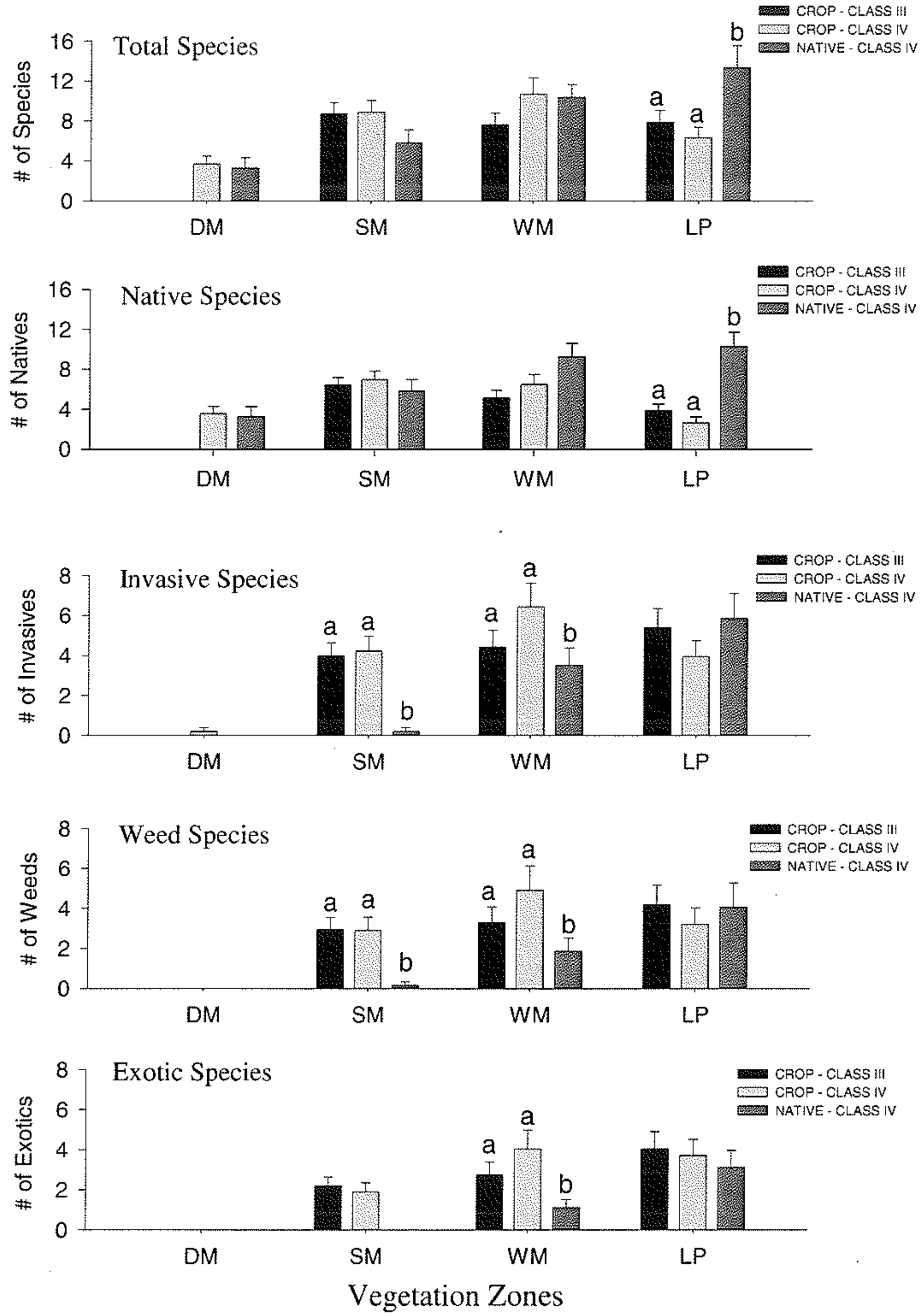

Figure 2.2 Comparisons of the least square mean numbers (±se) of total, native, invasive, weed and exotic plant species in deep marsh (DM), shallow marsh (SM), wet meadow (WM), and low prairie (LP) vegetation zones. Wetland type-means sharing a letter within each zone are not statistically different. No letters are presented when no differences are evident.

Table 2.8 Generalized linear mixed modeling results for the effects of wetland type and zone on total species, number of invasives, number of weeds, number of exotics, and number of natives in St. Denis (n = 6 wetlands).

		Total Species			Invasive			Weed			Exotic			Native		
		Num df, Den df[1]	F	p-value	Num df, Den df	F	p-value	Num df, Den df	F	p-value	Num df, Den df	F	p-value	Num df, Den df	F	p-value
Fixed Effects:																
Type[2]		1, 11.2	1.96	0.187	1, 43	5.07	0.030	1, 43	3.86	0.056	1, 14.9	2.01	0.177	1, 3.9	9.66	0.038
Zone		3, 12.0	7.86	0.004	2, 43	8.48	0.001	2, 17.6	9.50	0.002	2, 17.5	9.92	0.001	3, 46	4.14	0.011
Type*zone		2, 8.6	3.34	0.084	2, 43	3.93	0.027	2, 17.6	2.68	0.097	1, 15.2	1.12	0.307	2, 46	6.15	0.004
Statistical Contrasts of Least Square Means:	**Zone, Class, and/or Type**	**F**	**p-value**	**Nature of Effect[3]**	**F**	**p-value**	**Nature of Effect**	**F**	**p-value**	**Nature of Effect**	**F**	**p-value**	**Nature of Effect**	**F**	**p-value**	**Nature of Effect**
Differences among Types (by Zone)	SM	0.58	0.458	---	7.16	0.011	Cr>Nat	4.75	0.035	Cr>Nat	---[4]	---	---	0.01	0.925	Nat>Cr
	WM	1.94	0.189	---	0.06	0.807	---	0.78	0.389	---	2.44	0.128	---	5.92	0.029	---
	LP	6.80	0.027	Nat>Cr	0.28	0.602	---	0.02	0.897	---	0.35	0.568	---	17.88	0.001	Nat>Cr
Differences among Zones (by Type)	Nat	10.00	0.006	LP, WM>DM; LP>SM	7.33	0.002	LP, WM>SM	7.41	0.004	LP>WM>SM	7.80	0.010	LP>WM	6.95	0.001	LP>DM, SM; WM>DM
	Cr	0.06	0.940	---	1.94	0.156		3.78	0.062	LP>SM	6.03	0.015	LP>SM	2.34	0.107	

[1] Denominator degrees of freedom (df) for the F-tests were determined through Satterthwaite approximations.
[2] The effects of wetland class and type are confounded since there are only Crop, Class III and Native, Class IV wetlands at St. Denis.
[3] For differences among zones, nature of effect determined from Bonferroni-adjusted multiple comparisons.
[4] There are no exotics in the SM zone of native wetlands.

2.4.4 Plant Response Within Wetland Type

When total species were compared in Class III cropped wetlands, no evidential differences were found between the vegetation zones (Table 2.7). The DM zones of Class IV cropped wetlands had significantly fewer total species than the WM zone, while the DM zones of Class IV native wetlands had significantly fewer total species than the LP zones (Figure 2.3). For native plants the LP zones of Class IV native wetlands had the highest number of species, followed by the WM zones. Native species were low in the LP zones of cropped wetlands in this study and highest in the SM zones. Native species in the SM and WM zones of Class IV cropped wetlands were significantly higher than in the DM and LP zones, while native species in the SM zones of Class III cropped wetlands were significantly higher than the number of native species in the LP zones.

No invasive, weed or exotic species were identified in the DM zones of native wetlands in this study (Figure 2.3). *Typha glauca*, an invasive, was the only invasive present in the DM zones of MZRTA Class IV cropped wetlands. In Class IV cropped wetlands the highest numbers of invasive species occurred in the WM zones, with no significant zone differences detected for either weed or exotic species.

The results for the statistical comparisons of zone differences in Class III cropped wetlands at St. Denis were somewhat different to the findings for all Class III cropped wetlands in this study (Table 2.8). While there were no differences in the number of native species in each of the vegetative zones, there were a greater number of weed and exotic species in the LP zones compared to the SM zones. Similar to the findings for all Class III cropped wetlands there were no zonal differences for either total species or invasive species. The greatest total numbers of species and native species in Class IV native wetlands were found in the LP zones, followed by the WM and SM zones. Similar to the results for all wetlands, the greatest number of invasive, weed, and exotic species occurred in the WM zones of Class IV cropped wetlands.

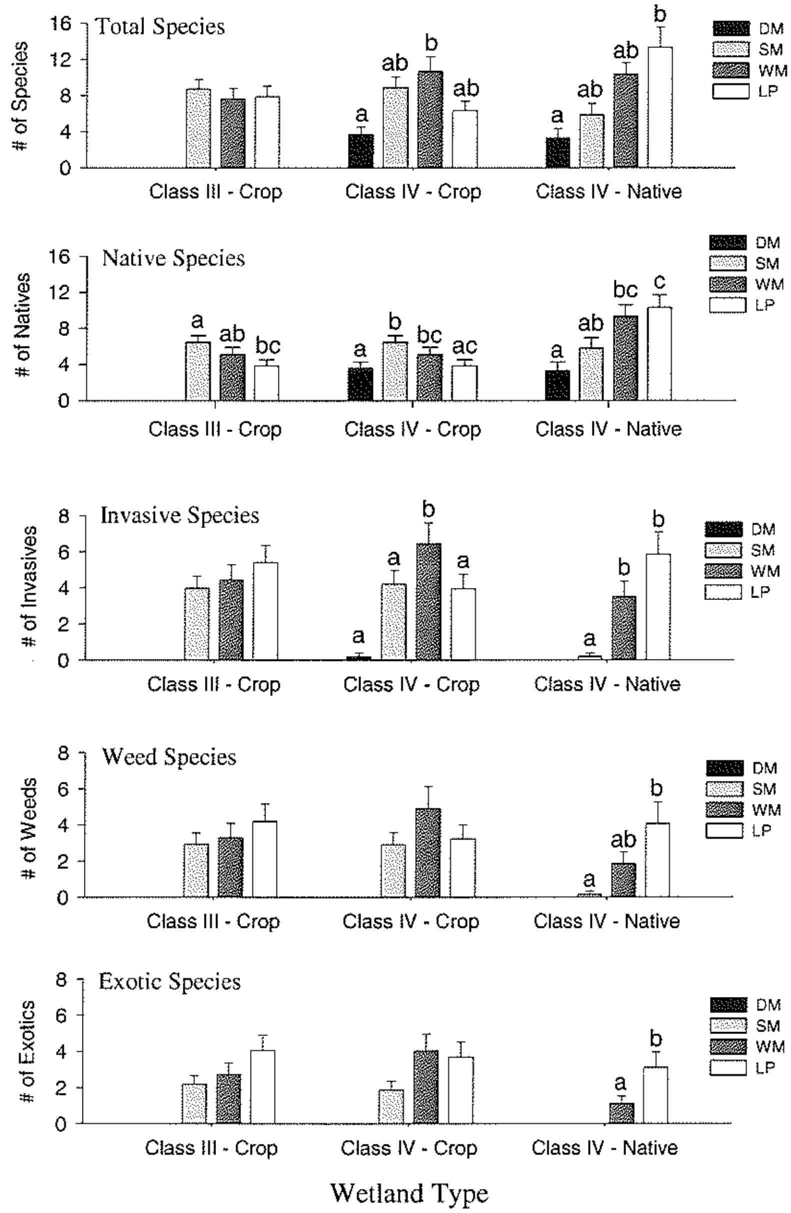

Figure 2. 3 Comparisons of the least square mean numbers (±se) of total, native, invasive, weed and exotic plant species in Class III and IV cropped wetland and Class IV native wetlands. Zone-means sharing a letter within each zone are not statistically different. No letters are presented when no differences are evident.

2.4.5 Comparison of Invasive/Native Perennials and Annuals by Wetland Type

The analysis of invasive perennial species (IPs) in native wetlands versus cropped wetlands indicate that there were more IPs in the DM ($p=0.050$) and SM ($p=0.003$) zones of cropped wetlands than in native wetlands. There was weak evidence of more IPs in the WM zones of cropped wetlands ($p=0.075$) and no significant difference in the number of IPs in LP zones. Native perennial species were higher in LP ($p<0.0001$) and WM ($p=0.003$) zones of native wetlands, while annual plant species were higher in the WM ($p=0.043$) zones of cropped wetlands.

2.4.6 Effect of Covariates on Total, Native, Invasive, Weed and Exotics

Analysis was conducted on all 21 wetlands to observe if zone length or zone slope influenced the number of species occurring in vegetative zones. Zone length, slope, and size varied across all wetland types and across all zones (Table 2.9). There was no effect of zone length on the number of total, invasive or native species within a zone, but there was a significant effect of zone length on the number of weed ($p=0.005$) and exotic ($p=0.009$) species. As zone length increased, so did the number of weed and exotic species. Zone slopes ranged from a low of -5.1%, where a slope was actually decreasing with rise, to a high of 39.2% (Table 2.9). There was no effect of slope on plants in Class III and Class IV cropped wetlands, but there was an effect of slope in the SM zones of native wetlands. As slope increased, so did the number of total ($p=0.004$), weed ($p=0.036$), exotic ($p=0.022$), and native species ($p=0.043$).

Table 2.9 Minimum, maximum, and mean values for zone length (m), slope (%), and size (m^2) for each wetland type.

			Deep Marsh	Shallow Marsh	Wet Meadow	Low Prairie
Zone Length: (m)	Crop – Class III	Min		4.0	1.0	1.0
		Max		20.0	9.0	8.0
		Mean		11.8	2.8	4.6
	Crop – Class IV	Min	2.0	2.0	2.0	1.0
		Max	2.0	23.0	17.0	6.0
		Mean	2.0	8.5	5.3	2.4
	Native – Class IV	Min	2.0	3.0	2.0	2.0
		Max	5.0	7.0	11.0	12.0
		Mean	3.5	5.1	6.8	5.3
Zone Slope: (%)	Crop – Class III	Min		0.8	0.0	0.0
		Max		4.7	16.5	4.5
		Mean		3.0	5.2	1.0
	Crop – Class IV	Min	0.0	-1.7	-5.1	0.0
		Max	5.4	14.7	6.2	39.2
		Mean	1.3	2.8	1.4	11.2
	Native – Class IV	Min	0.0	-1.3	0.0	0.0
		Max	2.2	13.6	11.8	13.0
		Mean	0.7	3.4	4.6	6.6
Zone Size: (m^2)	Crop – Class III	Min		796.6	388.2	1173.5
		Max		3916.1	1457.1	3246.6
		Mean		2042.5	1042.5	1960.2
	Crop – Class IV	Min	304.5	410.5	170.0	225.0
		Max	8387.5	4714.1	3317.9	4898.5
		Mean	2793.1	2646.2	1170.3	2350.0
	Native – Class IV	Min	16312.7	2842.8	3291.0	3554.6
		Max	47204.9	4535.2	4535.2	27417.3
		Mean	31758.8	3689.0	3913.1	11519.4

The size of a vegetative zone around a wetland had no significant effect on the number of invasive, exotic, weed or native species in this study. However, results did indicate that the larger the vegetative zone size, the fewer number of total species were present in all wetlands (p=0.043). Wetland size and soil salinity (EMh and EMv) were not found to have an effect on the numbers or types of plant species present in a wetland or their location within a basin.

2.5 Discussion

No one definition can adequately describe wetland health. One challenge is our ability to create a scientific definition that suits all wetland types. A second challenge is that most definitions focus on the interest of the source, whether that is flood control, water quality, carbon sequestration, biodiversity, waterfowl or wildlife habitat. One logical approach for determining the definition of wetland health is to compare pristine wetlands to altered or impacted wetlands, such as the wetlands surrounded by cultivated lands in this study. That is if pristine wetlands still exist? This comparison advances the science of wetland health towards a more measurable definition by focusing on features that are either lost or impacted in altered wetlands. It also enhances predictive capability when restoring wetlands. Currently, Mulhouse and Galatowitsch (2003) indicate that with few exceptions only weak predictions of restoration success can be advanced because scientific documentation is inadequate. Only one study in western Canada has documented the success of wetland restorations (Puchniak 2002). Several more studies have been conducted in the Northern Great Plains Region of the United States (Madsen 1986; LaGrange and Dinsmore 1989; Wienhold and van der Valk 1989; Galatowitsch and van der Valk 1995; Galatowitsch and van der Valk 1996a; Galatowitsch and van der Valk 1996b; Mayer and Galatowitsch 2001; Mulhouse and Galatowitsch 2003; Galatowitsch 2006; Aronson and Galatowitsch 2008). While these studies compared restored wetlands of various ages to pristine wetlands, no study included altered or impacted wetlands in their comparisons.

2.5.1 Vegetative Communities of Class III and Class IV Cropped Wetlands

An important component of this study was to determine if the plant communities of Class III and Class IV cropped wetlands differed. If differences were revealed then management strategies for their protection and restoration would also need to differ. Study results showed there were no significant differences between the vegetative communities of Class III and Class IV cropped wetlands. Invasive, weed, exotic and native plant species positioned themselves in a similar manner in and around wetlands irrespective of wetland Class. Therefore, we feel confident in providing collective recommendations for the future management and protection of both Class III and Class IV cropped wetlands in this document.

With several Canadian agencies currently working on strategies to protect existing wetlands on private lands it is important that research focus on the effect agricultural practices have on wetland plant communities. Understanding these effects will help guide management efforts. It will also help establish reasonable expectations for their restoration. Many wetlands on private lands will continue to be surrounded by croplands even though they will be protected from future drainage. In their natural state almost all Class III and Class IV wetlands possess LP, WM and SM vegetation zones (Stewart and Kantrud 1971). These zones reflect the very specific environmental and hydrologic requirements of wetland plants (van der Valk and Welling 1988; van der Valk and Pederson 1989; van der Valk et al. 1999; van der Valk 2000). If the loss of an LP or WM zone in a wetland surrounded by agriculture results in a greater risk of invasive species or a decrease in biodiversity, then recommendations are needed on how best to optimally protect these wetlands even though their surrounding land-use may not be the most favorable.

2.5.2 Vegetative Communities of Native and Cropped Wetlands

The results in this study indicate that fewer native plant species are found in wetlands surrounded by agriculture, particularly in the outermost vegetation zones. Exotic, invasive or weed plant species dominate in their absence. This not only has implications for the overall health of wetlands within agricultural landscapes, but it also provides an important insight into what is achievable before wetland restoration activities even begin. Galatowitsch (1993) and Galatowitsch and van der Valk (1994) provide a system that ranks the health of restored wetlands in the PPR. They use plant species richness to indicate whether restored wetlands are exceptional, typical, or depauperate. They do this by summing the number of plant species in each life history guild, or vegetation zone, to achieve an overall rating. Species are included in the ranking regardless of whether they are native, invasive, or a weed. A rating of less than 6 indicates a poorly restored wetland, while a ranking between 6 and 8 identifies a restored wetland as fair. A rating of 9 to 11 identifies a restored wetland as good, while a ranking between 12 and 18 classifies the wetland as excellent. Using the vegetation data in this study to rank Class IV cropped and native wetlands resulted in both receiving a "fair" value of 7 and 6, respectively. In applying this ranking we observed that no consideration is given to whether a species counted is native or invasive. As a result, the SM zones of Class IV cropped wetlands ranked higher than the SM zones of native wetlands even though the numbers of invasive species were significantly greater in cropped wetlands. Therefore, while a ranking system might reflect the species richness of a wetland it also needs to account for how many species are invasive, exotic or weedy.

2.5.3 The Role of Wet Meadow and Low Prairie Zones in Prairie Wetlands

The absence of either the LP or WM zone in all but one cropped wetland on the privately owned farm sites was an unexpected and important finding in this study. Delphey and Dinsmore (1993) also found these zones absent in restored wetlands in northern Iowa. Why these zones were missing only on privately owned farms is not clear in this study. Both the St. Denis and MZTRA sites are research farms. Modifying farm practices to help maintain wetland edge may be a priority because of their research focus. Wetland density is also much higher at both sites and this may make it too time consuming and inefficient to cultivate close to every wetland edge.

The loss of these outer margins in cropped wetlands provides a number of insights into the positioning of species and vegetative diversity in these habitats. While the numbers of native species present in SM zones were similar in native and cropped wetlands, the numbers of native species in WM and LP zones were not similar. Significantly fewer native species were present in the LP and WM zones of all cropped wetlands in this study. This suggests the WM and LP zones are not only important for a wetland's species richness, it also indicates that native plants in the WM and LP zones of cropped wetlands start from a deficit position before wetland protection or restoration is even attempted. This makes it difficult for native species to out-compete invasive species following wetland restoration. Previous studies have shown that WM and LP zones pose the biggest challenge for the successful re-establishment of native species in restored wetlands (Galatowitsch et al. 1999*b*; Mulhouse and Galatowitsch 2003). Galatowitsch and van der Valk (1996*a*) found fewer re-established wet meadow species in 3-yr old restored wetlands, while the numbers of shallow and deep emergent species were comparable to numbers present in natural wetlands. A survey of these same restored wetlands 19 years later still found that a majority of common LP and WM species never re-colonized while emergent species, such as *Typha angustifolia/x glauca* and *Phalaris arundinacea*, increased substantially in their percent cover (Aronson and Galatowitsch 2008). Galatowitsch et al. (1999*b*) state that from their experience neither active planting nor natural recolonization has been found to be successful in re-establishing the WM zones in restored wetlands in the northern United States.

The separate analysis of plant communities in Class III cropped wetlands at St. Denis added to our knowledge on the role of LP zones in maintaining vegetative health in prairie wetlands. Class III cropped wetlands at St. Denis were unique in that they all possessed WM and LP zones. Like all cropped wetlands in this study, we found significantly more invasive species in the SM zones of Class III cropped wetlands at St. Denis than in native wetlands. However, it appears that fewer invasive species are able to establish in cropped wetlands when the LP zone is present (2.5±0.5) as opposed to missing (4.0±0.6). This suggests more invasive and weed species establish in the inner vegetation zones of wetlands when the outer margins of wetlands are destroyed or degraded. The results for cropped Class IV wetlands support this finding. This has important implications for how to maintain the vegetative integrity of wetlands from this point forward. If the outer vegetation zones have the capacity to slow down the inward migration of aggressive or noxious species then land managers should strive to keep these zones intact whenever possible.

2.5.3.1 The Impact of Excess Sediments

Several factors may influence the viability of native plants in the WM and LP zones of cropped wetlands. One factor is the deposition of excess soil and nutrients into outer wetland margins as a result of surrounding land-use practices. Wind and water soil erosion can be a naturally occurring process on all land (Brady and Weil 2002). Water erosion is affected by runoff and rainfall factors, which include the amount of vegetative cover on the surface and a soil's ability to resist erosion. Wind erosion is affected by factors such as soil particle size, surface roughness, climate, vegetative cover and unsheltered distance. A third process includes tillage erosion, which is the progressive net down slope movement of soil by tillage operations (Smith et al. 2008). Tillage erosion, soil translocation, and the re-distribution of soil nutrients in agricultural fields can be substantial in hummocky landscapes (Arndt and Richardson 1988; Govers et al. 1999; Lobb et al. 1995; Pennock 2003; Li et al. 2007; Smith et al. 2008). Agricultural practices and erosive processes move soil from upper slope locations in fields to lower slope locations where wetlands are often situated. Pennock (2003) found the rate of soil loss from divergent shoulder slope positions at five cultivated till sites in Saskatchewan averaged 33 t ha^{-1} yr^{-1}, with a mean soil gain down slope of 15.2 t ha^{-1} yr^{-1}. Rates of sedimentation in wetlands have been reported to vary from 0.5 cm yr^{-1} to 3-4 cm yr^{-1} (Johnston et al. 1984, Fennessey et al. 1994). Smith et al. (2008) found that the amount of nitrate-nitrogen (NO_3^-N) in the top 15-cm of soil at convex upper slope positions was doubled when accumulated topsoil from lower slope areas was moved back to the hilltop locations. This suggests that nutrients in the soil, such as nitrogen and phosphorous, also have the potential to accumulate in wetlands margins.

While the accumulation of excess sediments has created much concern for the water quality and health of aquatic life in wetlands (Gleason and Euliss Jr. 1998), little attention has been given to the effect additional sediments have on plant communities. Evidence suggests that only small portions of incoming sediments reach the deeper areas of the wetland basin and that most sediments remain, or settle, in the outermost margins of the riparian areas. Ghaffarzadeh et al. (1992) found 85% of sediments removed within the first 3 meters of a grassed buffer. Neibling and Alberts (1979) showed a 90% reduction in sediment discharge within the first 5 meters of a grassed buffer, while Magette et al. (1989) found a 66% reduction in sediments passing through a 4.6 meter grass buffer.

Even small amounts of overlying soil can impact seed germination and species richness and diversity (Galinato and van der Valk 1986; Dittmar and Neely 1999; Werner and Zedler 2002). Galinato and van der Valk (1986) studied the germination of wetland/riparian plant seeds covered by 0, 1, 2, 3, 4, 5-cm of soil. They found that seed germination decreased from 79% to 38% for annuals and from 71% to 20% for perennials when covered by 1-cm of soil. Only *Hordeum jubatum*, an invasive perennial, was able to establish successfully under all soil depths. Mahaney et al. (2004) found all plant seeds collected from pristine wetlands in Pennsylvania impacted by 1-cm of overlying sediments while invasive species collected from impacted wetlands, such as *Phalaris arundinacea* and *Cirsium arvense,* were not. Plants belonging to the *Carex* genus, more than almost all other genera, display a marked requirement of light to germinate (Schutz and Rave 1999). This is a concern since *Carex* species are an essential plant community in the WM zones of prairie wetlands, with more than 60 species listed in the PPR

(Barkley 1986). These studies and others provide several reasons for the poor recruitment of native species (Mahaney et al. 2004). For most seeds, it is the difficulty of germination in environmental conditions that are both low in oxygen and light (Bewley and Black 1994; Baskin and Baskin 1998), resulting in the buried seeds of many plants remaining dormant (Fenner 1987). For wetland and riparian plants with small seeds, the combination of low light and oxygen can make germination even more difficult (Galinato and van der Valk 1986). Werner and Zedler (2002) found that excess sedimentation over a sedge meadow surface and the subsequent loss of tussock surface areas resulted in a loss of microhabitats, an increase in soil bulk density, and a decrease in organic matter content. They concluded that sedimentation contributed to the loss of native species in remnant wetlands and an increase in invasive species, such as *P. arundinacea*. Finally, if sedimentation is great enough that the distance between the soil surface and the water table is increased, those riparian plants that are dependent on this relationship can be adversely affected as well (Tickner et al. 2001).

2.5.3.2 Excess Nitrogen

While excess sediments in the outer margins of wetlands in this study may have created a barrier to the growth of native species, it may have also removed a barrier that once excluded invasive, exotic and weed species from establishing (Johnstone 1986). The addition of excess nitrogen deposited in and around prairie wetland basins through the application of fertilizers (Pennock 2005) and in tillage sediments (Smith et al. 2008) may have more of an effect on plant community succession than once suspected (McLendon and Redente 1992; Vasquez et al. 2008). An ample supply of N has been shown to be a key factor in the ability of invasive annuals to achieve and maintain dominance on disturbed sites (McLendon and Redente 1992). Results in this study indicate that the average number of annual species were higher in wet meadow zones of cropped wetlands than in native wetlands. Stohlgren et al. (1999) found that at plot and landscape scales areas of high soil fertility (total %N and available N) were particularly invasible by exotic species in grasslands studied in Colorado, Wyoming, South Dakota, and Minnesota. Nitrogen enrichment has also been shown to be particularly beneficial to two of the invasive species prevalent in this study: *P. arundinacea* and *C.arvense* (Mahaney et al. 2004).

2.5.3.3 Soil Moisture

Many wetland researchers focus on the effect of standing water on plant zonation and overlook that it may be soil moisture, rather than standing water, that determines vegetative growth in the outer margins of wetlands. Tickner et al. (2001) suggest riparian areas are more prone to plant invasions due to the frequency of disturbance from floods. If this is true then comparable numbers of invasive and weedy species in native and cropped wetlands at St. Denis would have been expected since the frequency of flooding events would be similar. Galatowitsch and van der Valk (1996*b*) suggest that the poor re-colonization of native species in areas above the water line may be due to an altered soil hydrology in restored wetlands. The soils in the outer margins of cropped wetlands in this study had been impacted by agricultural practices. This is evident by the absence of many of the WM and LP zones. Wetland soils that are exposed to agricultural practices are subject to oxidizing conditions, higher rates of decomposition (Dexter 1988), and increased compaction (Galatowitsch and van der Valk 1996a; Brady and Weil 2002). Even no-till management operations can cause moderate increases in soil compaction and bulk densities (Blanco-Canqui and Lal 2007). Increases in bulk density coupled with decreases in soil organic matter can result in decreases of both water infiltration (Galatowitsch and van der Valk 1996*b*; Franzluebbers 2002; Kamp et al. 2003; Bodhinayake and Cheng 2004) and plant-water availability (Naeth et al 1991). While a reduction in soil moisture has been shown to have a negative impact on the ability of native species to compete with invasive species (Wetzel and van der Valk 1998; Budelsky and Galatowitsch 1999; Fraser and Karnezis 2005; Fraser and Miletti 2008), it can also negatively influence the establishment and viability of wet meadow species such as *Carex* (van der Valk et al. 1999; Budelsky and Galatowitsch 1999). A reduction in soil moisture along with the depletion of seed bank reserves with ongoing cultivation (van der Valk and Verhoeven 1988; Wienhold and van der Valk 1989) means that the struggle to keep WM and LP areas healthy in existing prairie wetlands may be extremely difficult if current human activities around wetlands remain unchanged.

2.5.4 The Influence of Wetland Size, Zone Size, Slope and Soil Salinity

If a large percentage of prairie wetland margins have already experienced some level of impact (Turner et al. 1993), are there other options available for wetland protection or restoration that will maintain and encourage native plant diversity while lessening the establishment of invasive species? Decisions made by Government agencies for setting aside wetlands, either for restoration or conservation purposes, may be based on a "bigger-is-better" approach if science is not used to help guide the prioritization process (Zedler 2003). For this reason it was important to address the influence of variables such as wetland size, zone size, soil salinity, and zone slope on plant distribution. Some suggest that the number of plant species in depressional wetlands generally decrease in smaller wetlands and increase in larger wetlands (Lopez et al. 2002). Larger wetlands may favor greater colonization because they potentially offer a wider variety of habitats (i.e., hydrologic and soil conditions) and/or attract a greater variety of animal dispersers (Mulhouse and Galatowitsch 2003).

While research by Mulhouse and Galatowitsch (2003) on restored wetlands in Iowa and Minnesota did find a strong association between species richness and basin size, this study did not. Houlahan et al. (2006) also found a positive association between species richness and wetland size, but their mean wetland size was very large (66.7 ha) compared to ours. Results from this study showed no relationship between wetland size, soil salinity and the size of a vegetative zone on the number of total, native, or invasive species in either cropped or native wetlands. In fact, the total number of species present decreased as vegetative zone size increased. Except for the SM zone in native wetlands, the degree of slope had no effect on the number of species in either cropped or native wetlands. This suggests that a "bigger-is-better" approach does not necessarily preserve or encourage greater native plant diversity in prairie Canada. Seabloom and van der Valk (2003) also found no detectable relationship between wetland size and plant diversity in natural and restored wetlands in northwestern Iowa. Additional factors beyond basin size, such as a lessening of impacts on outer wetland margins, may play a more critical role in maintaining and encouraging species richness in prairie wetlands.

2.5.5 Perennials versus Annual Species

Consideration was given to perennials versus annuals in this study since evidence suggests that plant guilds may be more useful than species composition at indicating land-use impacts and for predicting the potential for restorative success (Galatowitsch et al. 2000). A study of wet meadow zones in Minnesota wetlands found clear evidence of changes among guilds along land-use gradients, with decreased importance of native perennials and increased importance of either annuals or invasive perennials with greater disturbance (Galatowitsch et al. 2000). In examining the plant guilds in all four vegetation zones, this study found that invasive perennials were more prevalent in the SM and DM zones of cropped wetlands, whereas native perennials were more prominent in the WM and LP zones of native wetlands. Similar to the findings of Galatowitsch et al. (2000), annual plant species were also more abundant in the WM zone of cropped wetlands in this study. These findings further support the importance of the WM and LP zones for native species and highlight the movement of invasive perennials into the inner vegetation zones of wetlands surrounded by agriculture. Land planners responsible for selecting wetlands, either for protection or restoration, should be aware of how wetland plant communities can be impacted by agriculture and strive to choose those wetlands that display the least degree of impact.

2.6 Conclusion

Pristine wetlands were once plentiful across the prairie landscape. New immigrants, on their way to lay claim to homesteads at the beginning of the 20th century, were forced to push through numerous sloughs on carts filled with supplies and with livestock in tow. Even though getting stuck was almost inevitable, to have skirted them would have made the trip too long (Jackson-Kennedy 1980). Cultural improvements for enhancing the prairie way of life led to an extensive network of improved roads and bigger and more productive farms (Galatowitsch et al. 1999*b*). Both of these activities resulted in a dramatic decrease of prairie wetlands and many of the wetlands that remain have not done so without impacts to their hydrology and vegetation. Agreement on a standard definition of wetland health will improve our understanding on the quality of wetlands that remain and help guide us in setting reasonable goals and outcomes for their protection and restoration. The ability to isolate wetland features that are lost or impacted in altered wetlands is a critical step to crafting a standard definition of wetland health.

The intent of this study was to assess the influence of upland land-use on the distribution and diversity of vegetation in prairie wetlands, and to improve our predictive capabilities for their restoration and protection. What is evident from this study, and other studies in the United States, is that the outermost vegetative zones of prairie wetlands are at great risk of being lost and impacted because of agriculture (Turner et al. 1987). What is also clear is that restoring native plants in these zones is extremely difficult (Galatowitsch et al. 1999*b*). Our findings indicate that these zones, when present, are critical in protecting the overall health of a wetland by decreasing the ability of invasive species to migrate inward. These zones also remain the areas where the greatest numbers of total species and native species occur. With invasive species emerging as one of the key threats to the world's biological diversity, a better scientific understanding of invasion processes and of management and policy approaches for reducing their impacts will help to form a strong foundation for taking effective action to reduce this threat (Chornesky and Randall 2003).

Few researchers provide strategies for protecting and restoring the vegetation of prairie wetlands (1999*b*). Galatowitsch and van der Valk (1994) and Puchniak (2002) suggest immediate and intensive action when aggressive weeds, such as *Phalaris* *arundinacea* and *Cirsium arvense*, are found in wetlands. If that is the case then every wetland in this study, including the native wetlands, requires immediate attention. One reasonable approach is to limit or reduce those impacts that carry with them the gravest consequences. Sediment accumulation in riparian margins has been shown to decrease the seed germination of native riparian species by reducing available light and oxygen, while at the same time favoring the growth of invasive species with increased nitrogen loads. Agricultural activities that affect soil bulk density, soil moisture and plant available water in these outer vegetative zones must be limited as well.

It is crucial that every attempt be made by land managers and policy makers to improve management activities around prairie wetlands. Our findings indicate that the outer vegetative zones of wetlands on private lands are at great risk of being lost or degraded to such a degree that successful restorations may not be possible. Encouraging agricultural activities that reduce or minimize the translocation of sediments, but maintain soil bulk density and organic matter content is crucial. Creating and maintaining wide buffer zones around wetlands that include a terrestrial vegetation zone would help ensure that excess sediments are trapped before they even reach the low prairie zone. Wetland plant communities are intricately linked to the invertebrates and vertebrates that depend and rely on these habitats. Far more than just wetland plants are at risk if no action is taken to protect these areas from further degradation.

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3.0 MODELING OF SPRING HYDROLOGY DYNAMICS IN NORTHERN PRAIRIE WETLANDS IN RELATION TO VEGETATION, SOIL PROPERTIES AND LAND-USE

3.1 Abstract

The presence or absence of wetlands in any given year is influenced by annual variations in precipitation. Little is known about the degree to which precipitation influences both wetland vegetation and the distribution of carbonation and salinity around wetlands. In prairie Canada, wetland vegetation is currently used as the only tool for delineating and classifying wetlands. However, in agricultural landscapes wetland vegetation is often impacted or missing. This study explores the spatial relationships that exist between wetland hydrology, wetland vegetation and soil salinity/carbonation. Nine wetlands on two study sites in Manitoba and Saskatchewan were studied in 2006. Three wetlands were surrounded by native vegetation while the remaining six wetlands were surrounded by croplands. Vegetation surveys were conducted on each wetland to identify their riparian zones using the Stewart and Kantrud (1971) system of wetland classification. The vegetative zones and long-term water level data were used to verify the average annual snowmelt runoff into each pond using a wetland hydrology model developed at the University of Saskatchewan. The spatial data was then compared to the extent of soil salinity or carbonates around each wetland. The findings in this study suggest that specific snowmelt runoff events at each study site predict the vegetative zones in each wetland, and that a consistent relationship occurs between each wetland and the spatial extent of soil salinity or carbonates around the basin.

3.2 Introduction

Approximately 80% of western Canada is covered by an area known as the prairie pothole region (PPR) (Batt et al. 1989). The entire PPR accounts for approximately 777 000 km^2 and covers portions of Manitoba, Saskatchewan, Alberta, North and South Dakota, Iowa, Minnesota and Montana (Luoma 1985). Glacial processes that shaped the PPR resulted in the formation of millions of variably-shaped depressions that often hold water for various periods of time in most years (Voldseth 2004). Referred to as wetlands, marshes, potholes or sloughs, these depressions represent portions of the landscape that are neither fully aquatic nor fully terrestrial. Consequently they often possess characteristics of both. Their unique vegetation and soils are not only used to identify their location, but to classify them as well (Stewart and Kantrud 1971). Prairie wetlands contribute a major component of the water resources and ecology in the PPR (Hayashi et al. 1998*a*) and provide our society with an abundance of ecosystem goods and services that include flood protection, clean water, depositories for biodiversity, and atmospheric water (LaBaugh 1998).

Very few researchers have studied the water balance of wetlands and those that have often focus on a single wetland (Poiani and Johnson 1993, Poiani et al. 1995, Hayashi et al. 1998*a*, Su 1998, Voldseth 2004). Therefore, hydrologic models that simulate water storage capacity and surface runoff across multiple wetlands are lacking (Su 1998). Since hydrology drives wetland productivity, an increase in our understanding of hydrology leads to a better appreciation of what is lost or altered when hydrology is impacted. Wetlands are designed to withstand, if not thrive, on the climatic instability common to the prairies. It is often the human alteration of their hydrology that impacts their productivity. Surrounding land use, such as cultivation versus native or managed grasslands, can potentially alter the dynamics of wetland in-flows and the amount of water reaching a wetland. This then alters the hydro-ecological processes of spring rise, summer drawdown, length of dry periods, and vegetative reproduction and establishment (Voldseth 2004).

Snowmelt constitutes the primary source of freshwater to wetlands on an annual basis (Su 1998). It is estimated that on average between 30 to 60% of winter precipitation from adjacent slopes enters prairie wetlands as snowmelt runoff (Hayashi et al. 1998*a*; Bedard-Haughn and Pennock 2002; Berthold et al. 2004). Therefore, any change in land use that alters the amount of spring runoff or snow accumulation also has important implications for the water balance of wetlands (van der Kamp et al. 1999). Both Voldseth (2004) and van der Kamp et al. (2003) found that snowmelt runoff was greater in wetlands surrounded by cultivated crops or managed grasslands compared to wetlands surrounded by smooth brome grasslands. Euliss and Mushet (1996) found that water levels fluctuated much more in wetlands surrounded by annual crops than wetlands surrounded by grasslands. Both temporary and seasonal wetlands, or those that tend to go dry every summer, showed the greatest fluctuations of water levels compared to wetlands with a more permanent hydrology. They attributed this to the influence of snowmelt as a water source for ephemeral wetlands compared to the role of groundwater as a water source for more permanent wetlands. Higher water levels and greater water level fluctuations can greatly affect the types and amounts of vegetative species in wetlands. The influence that a changing hydrology may have on the presence or extent of salinity in wetlands remains unknown.

Similar to water balance studies on wetlands, few studies have focused on the extent of soil salinity around wetlands as a potential indicator of water movement and hydrology (LaBaugh et al. 1987; Arndt and Richardson 1988; Swanson et al. 1988; Arndt and Richardson 1989; Arndt and Richardson 1993). Fewer have studied the distribution of carbonates around ephemeral wetland basins (Knuteson et al. 1989). Part of this is attributed to the challenges of modeling water movement into wetlands along with the time and effort required to assess the spatial distribution and types of salts and carbonates in and around them. Salama et al. (1999) state that the natural landscape features that determine the spatial distribution of soil salinization in catchment areas include topography and associated groundwater flow systems. For topography, landform elements such as slope, break of slope and curvature controls where groundwater discharge occurs and where water may pool (Salama et al. 1999). The magnitude of discharge or the degree to which water pools is based on the dimensions of a catchments' surface (Toth 1963; Arndt and Richardson 1988). While groundwater levels tend to be deeper in topographic highs and nearer the surface in topographic lows on the landscape (Salama et al. 1999), understanding the movement and distribution of water on hillslopes and into catchment areas is key in understanding the chemical characteristics of soils across hummocky landscapes on the prairies (Pennock et al. 1987). Therefore, a critical step in improving our understanding of the distribution of soluble salts around wetlands is our ability to identify and extrapolate landform or wetland elements that may be contributing to, or controlling, water flow and solute development.

Irrespective of the natural or human influences on wetlands, what is unique in the PPR is how hydrologically different wetlands are even though they exist in close proximity to one another. They often display surprisingly different vegetative characteristics, surface water chemistries and spatial variability in salinity or carbonates even though they are exposed to the same climatic events. It is speculated that this variability can be attributed to general hydrologic processes such as surface-water flow or precipitation-evaporation balance (Swanson et al. 1988; Richardson and Vesprakas 2001). In reality, much of the variation within and between prairie wetlands may be due to an ever-changing mosaic of surface waters interacting with the atmosphere, geologic and surface material, and groundwater (Arndt and Richardson 1989). Since mobile ions move with water through soil, the concentration, spatial distribution and types of major ions present in wetlands should provide insight into the length of time water has been held, the direction of water flow through the soil and whether hydrological interactions with nearby wetlands may be occurring. Therefore, improving our understanding of the spatial, physical, chemical and temporal processes that lead to variations in salinity and evaporitic mineralogy in prairie wetlands also provides us with a better understanding of how water is held and moved through these systems under normal hydrological conditions.

The first objective of this study was to apply a hydrological model to examine model predications of various snowmelt scenarios against the spatial positioning of wetland/riparian vegetation in wetlands that differ in hydrological permanence and surrounding land-use. The vegetation and spring water level data in each wetland were matched to the model's predictions to determine which snowmelt scenario drives the hydrology of wetlands in the spring. The second objective was to examine and compare the extent and distribution of soluble salts in soils surrounding seasonal and semi-permanent wetlands in relation to wetland attributes and landscape features.

3.3 Methods

3.3.1 Site Descriptions

Nine wetlands on two study sites in Manitoba and Saskatchewan were studied in 2006 (Figure 3.1). Six of the nine wetlands were located at the St. Denis National Wildlife Area (Figure 3.2). The St. Denis site is located approximately 40 km east of Saskatoon. The 385 ha site was acquired in 1968 by the Canadian Wildlife Service for the purpose of studying wildlife and wetlands within agricultural landscapes. Aerial photographs of the area show that most of the site has been under dry land agriculture consisting of a rotation of summer fallow and cereal crops since at least 1951, except for a 43 ha section of native grasslands that have never been broken by cultivation. Over 200 temporary and permanent wetlands have been classified on St. Denis (Hogan and Conly 2002). Ponds 65, 66 and 67 are Class IV semi-permanent wetlands surrounded by native grasslands, while ponds 109, 117 and 120 are Class III seasonal wetlands surrounded by cultivated uplands (Stewart and Kantrud 1971) (Table 3.1). It is important to note that wetlands 65, 66, and 67 were both larger and their electrical conductivity (EC) noticeably higher than wetlands 109, 117 and 120. Average annual precipitation for St. Denis is 350-mm (Environment Canada 2009). St. Denis' mean annual air temperature is 2°C, with a July mean temperature of 18°C and a January mean temperature of -19°C (Fang 2007). Elevations for the three wetlands bordered by native grasslands ranged from 543 meters above sea level (masl) to 561 masl, compared to 548 masl to 560 masl for the wetlands surrounded by crops.

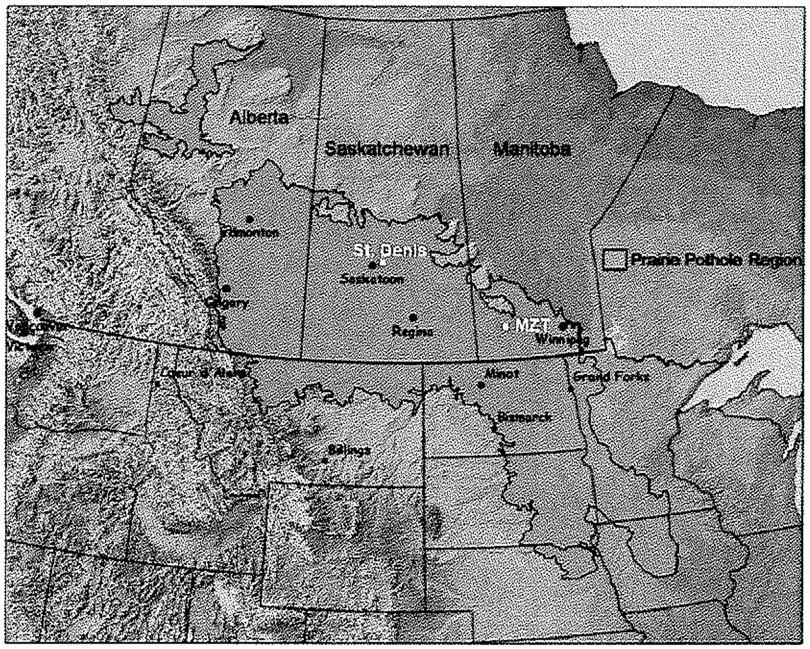


Figure 3.1 Location of the St. Denis site in Saskatchewan and the MZTRA (MZT) site in Manitoba.

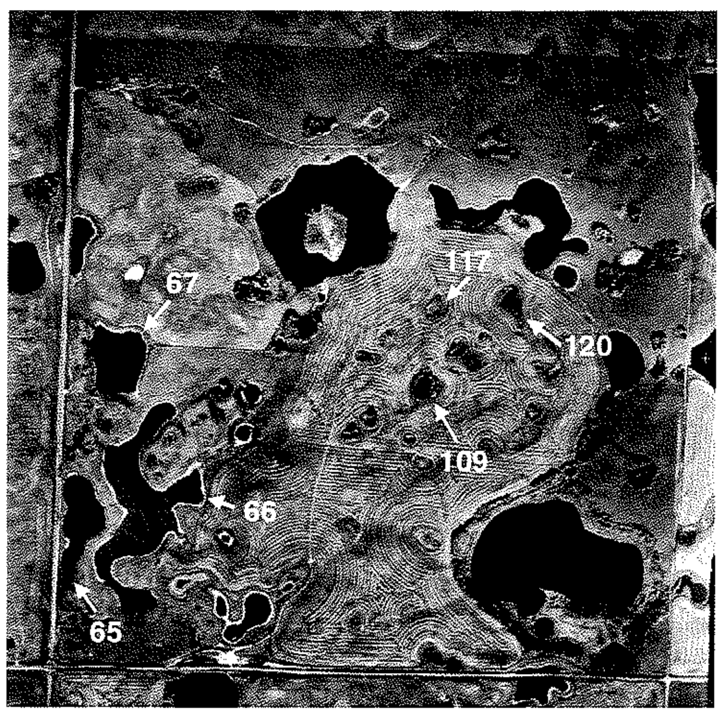


Figure 3.2 Aerial photograph of the SDNWA taken in October 1997. Study wetlands are identified by number.

Table 3.1 Summary of wetland characteristics including location, surrounding land-use, wetland class, electrical conductivity (EC), surface water salinity rating and annual precipitation.

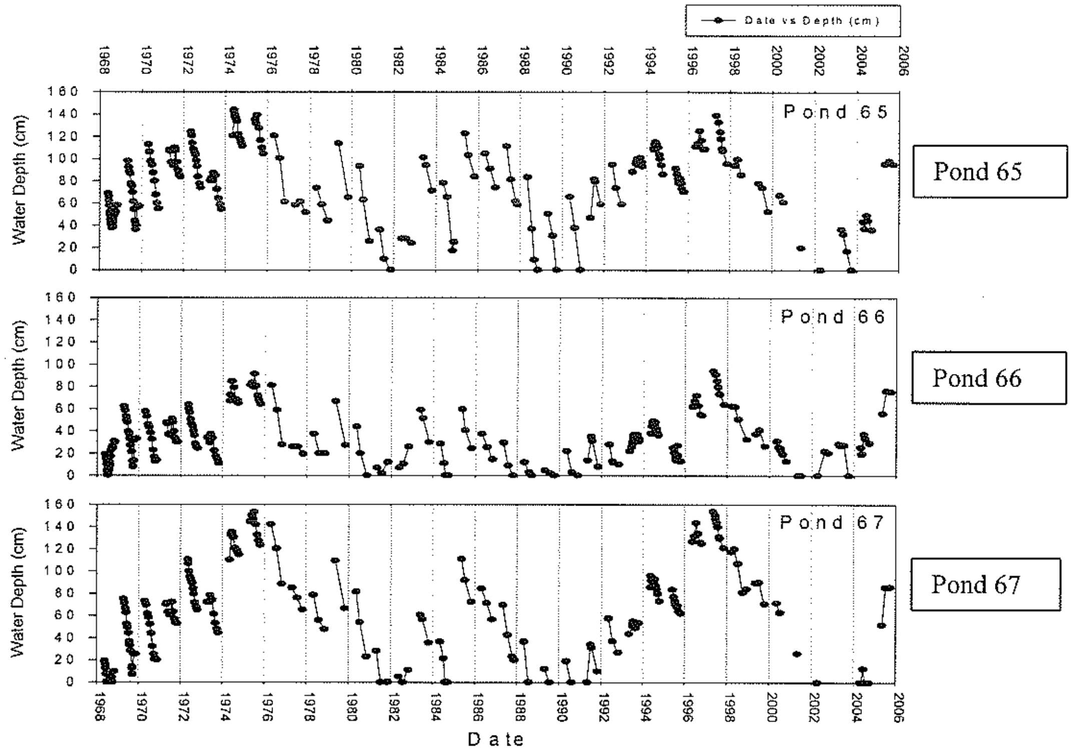
Site	Longitude Latitude	Wetland #	Surrounding Land-Use	Wetland Class	EC of Surface Water (μS/cm)	Salinity[1,2]	Annual Precipitation[3] (mm) Rainfall[4] Snowmelt[5]
Manitoba							
MZTRA:	99°56'00"W	232	Cropland	IV	288	Fresh	460[3]
	50°03'05"N	222	Cropland	IV	1322	Slightly Brackish	340[4]
		216	Cropland	IV	1316	Slightly Brackish	80[5]
Saskatchewan							
St. Denis:	106°05'06"W	109	Cropland	III	300	Fresh	350[3]
	52°12'38"N	117	Cropland	III	403	Fresh	265[4]
		120	Cropland	III	293	Fresh	104[5]
		65	Native Grassland	IV	2260	Mod. Brackish	
		66	Native Grassland	IV	8680	Brackish	
		67	Native Grassland	IV	2350	Mod. Brackish	

[1] Stewart and Kantrud (1972).
[2] Mod. brackish = moderately brackish.
[3,4] Based on 30-yr climate normals (Environment Canada 2009).
[5] Based on December to April monthly climate data for 1968 to 2008 (Environment Canada 2009).

Several research studies previously conducted on the St. Denis site provide extensive information on soils and groundwater movement. The data and findings from these studies were considered essential in providing the baseline information needed for verification of the WaterLevels hydrological model used in this study. Water level data collected monthly from the spring to the fall were available for St. Denis wetlands from 1968 to 2006 (Figure 3.3). Ponds 65, 66 and 67 wetlands were both deeper and more permanent than ponds 109, 117 and 120. Mean October water depths for ponds 65, 66, and 67 are 70.5-cm, 27.3-cm and 59.8-cm respectively, compared to 12.8-cm, 0.0-cm and 20.5-cm for ponds109, 117 and 120. General information on groundwater movement for

St. Denis has been reported by Miller (1983) and Miller et al. (1985). Soils for the site have been described by Miller (1983), Bedard-Haughn (2001) and Bedard-Haughn and Pennock (2002). Hydrological and soil characteristics for pond 117 and 120 are detailed in Bedard-Haughn et al. (2006*a,b*), while detailed hydrological characteristics of pond 109 have been described by Hayashi et al. (1998 *a,b*), van der Kamp et al. (1999), Su (1998), Conly and van der Kamp (2001), Hayashi et al. (2003), van der Kamp et al. (2003), Berthold et al. (2004), Waiser (2006) and Heagle et al. (2007).

a) Class IV Wetlands



b) Class III Wetlands

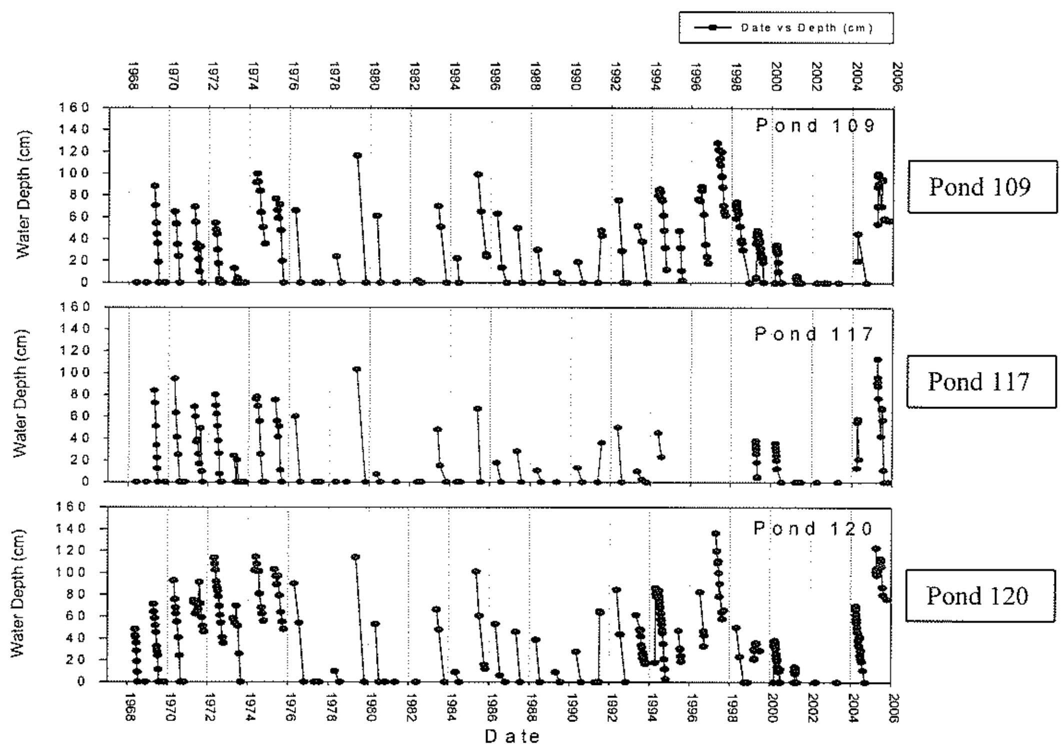


Figure 3.3 St. Denis water level data 1968-2006 for a) Class IV wetlands surrounded by native grasslands and b) Class III wetlands surrounded by croplands.

Soils at the St. Denis include thin Rego Dark Brown Chernozems on the shoulders of knolls and at mid-slope positions adjacent to seasonal ponds. Thicker Orthic Dark Brown Chernozems are found at mid-slope and toe-slope positions, while Eluviated Dark Brown Chernozems and Humic Luvic Gleysols are located in wetland basins (Yates 2006). Calcareous Dark Brown Chernozem soils can also be found at shoulder and mid-slope positions as well as in some toe-slope positions (Bedard-Haughn 2001). Soil textures range from silty loam in depressions to loam at topographically high positions (Yates 2006). The topography for the area is described as moderately rolling knob-and-kettle and knoll-and-depression moraine.

The Manitoba Zero Till Research Association (MZTRA) farm located in south-central Manitoba occurs in the Aspen Parkland area of the province on an undulating to hummocky landscape. Numerous wetlands of varying permanence occur on the site (Figure 3.4). The mean annual temperature for the area is 1.4°C (Podolsky and Schindler 1993), with a mean July temperature of 18.5°C and a mean January temperature of -17.8°C (Environment Canada 2009). Black Chernozem soils have developed over calcareous glacial tills on the upland areas of the MZTRA farm, while mostly Gleysol soils have developed in wetlands (Podolsky and Schindler 1993). Twenty-two percent of the soils on the farm are considered weakly saline (4-8 mS cm^{-1}), with higher salinity levels occurring in soils adjacent to more permanent wetlands (Podolsky and Schindler 1993). Three Class IV wetlands surrounded by flax crops were randomly selected and studied in 2006 (Table 3.1). No long-term water level information is available for wetlands at this site. Elevations for the wetlands range from a 497 masl to 507 masl. Average annual precipitation for the site is 460-mm (Environment Canada 2009).

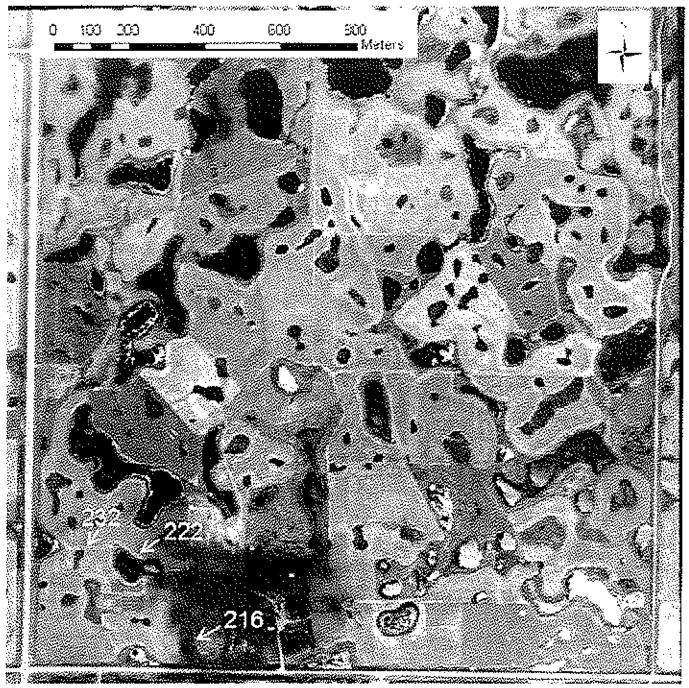


Figure 3.4 Aerial photograph of the MZTRA farm taken in 2001. Wetlands are identified by number.

3.3.2 Vegetation Surveys

The vegetation was surveyed on all wetlands in July and August of 2006. Three vegetation transects per wetland were surveyed on each wetland using methods outlined by Bonham (1989). A 1-m wide belt transect was randomly placed on each wetland starting just below the lowest point of the aquatic rooted vegetation zone and extending to the upland area where terrestrial vegetation dominated. Each transect ran perpendicular to the vegetation zones of the wetland so that all the zones could be captured and determined (Figure 3.5). Dominant plant species using plant lists provided by Stewart and Kantrud (1971) were used to identify the vegetative zones present around each wetland (Table 3.2). Transect lengths depended on the extent of hydrophytic vegetation surrounding each wetland. Water depth was recorded at 1-m intervals along each transect.

Figure 3.5 Vegetation survey of MZTRA wetland 222 using a 1-m wide belt transect.

Table 3.2 Classification of prairie wetlands using vegetative zones (after Stewart and Kantrud 1971).

Wetland Class:	**I**	**II**	**III**	**IV**	**V**
Water Permanence:	Ephemeral	Temporary	Seasonal	Semi-permanent	Permanent
Vegetative Zones Present:	Low prairie	Low prairie Wet meadow	Low prairie Wet meadow Shallow marsh	Low prairie Wet meadow Shallow marsh Deep marsh	Low prairie Wet meadow Shallow marsh Deep marsh Open water

The % cover-abundance of all plant taxa was assessed in each successive 1-m x 1-m quadrat along each transect. Cover data was directly assessed in the field and the percent cover-abundance of all plant taxa in each quadrat was recorded as a value from 5 to 100%, or trace (<5%). Relative cover (RC), relative frequency (RF), and an importance value (IV) were calculated for all species on each transect using the following formulas (Mueller-Dombois and Ellenberg 1974; Doumlele 1981; Perry and Hershner 1999):

Relative Frequency (RF)
= (Species frequency/Σ frequencies for all species)*100

Relative Cover (RC)
= (Species mean/Σ means for all species)*100

Importance Value (IV)
= RC + RF

An importance value of $\geq$ 20 for species indicative of each zone was used to determine the exact start and end of the vegetation zones on each transect. The outer extents of these zones were marked in the field using either a Trimble® GeoXT™ or Sokkia Total Station to collect Global Positioning System (GPS) coordinates. A differential correction process improved data accuracy to the subfoot (<30-cm) level and in most cases less than 10-cm accuracy. All coordinate data was imported into ArcGIS 9.3 (ESRI Canada Limited) where spatial features could be created for each wetland.

3.3.3 Soil Salinity and Soil Carbonate Sampling

To determine the extent of soil salinity around wetlands a Geonics Ltd. electromagnetic meter (EM38) was used to measure the apparent soil electrical conductivity (ECa). The EM38 was calibrated according to manufacturer's directions prior to use and re-calibrated numerous times during data collection in the field. EM38 readings were taken at each 1-m interval on vegetation transects and on the surrounding upland soils around each wetland. ECa readings ($mS\ m^{-1}$) in both the horizontal (EMh) and vertical (EMv) dipole positions were taken by placing the EM38 directly on the soil surface. The EM38 could only be used in locations where no standing water was present. The EMh dipole position measures ECa from the soil surface to a depth of approximately 60-cm below the soil surface. The EMv dipole position measures ECa between approximately 60-cm and 120-cm below the soil surface. The location at which EMh readings fell below 40 mS m^{-1} in upslope areas around a wetland was considered to be point where the effective transmission zone (ETZ) of the wetland ended. The outer extent of the ETZ was marked in the field using the same method of collection as the vegetation zones. Soil samples representing the range of ECa values recorded by the EM38 were collected at each site using a hand auger to a depth of 60-cm. Samples were air dried, ground and sieved (< 2 mm). Electrical conductivity extract (ECe) was determined using the saturated paste method and the correlation coefficient determined via linear regression for ECa to ECe.

The ETZ is considered that area where groundwater flows readily between the water table and the bottom of the fractured till (van der Kamp and Hayashi 2009). It is important in the exchange of groundwater between the pond and the wet margin surrounding the basin. It tends to expand as the water table rises during wet periods and contract as the water table lowers during dry periods (van der Kamp and Hayashi 2009).

If EMh readings within and around a wetland never exceeded 40 mS m^{-1} then soils were considered non-saline and surrounding soils were sampled using a hand corer for the presence of secondary carbonate enrichment near the soil surface. These soils, when present, would be considered Rego Dark Brown Chernozems in the Canadian System of Soil Classification (Soil Classification Working Group 1998). The goal was to determine if locations existed around non-saline wetlands where the process of carbonate-rich soil water moving upwards through the soil profile, rather than downward, had resulted in the formation of a calcic horizon near the soil surface (Knuteson et al. 1989; Bedard-Haughn 2001). This process of water movement leads to the formation of a unique ring of discharge soils around the basin that can be used to indicate the extent of the ETZ around the wetland (Figure 3.6) (van der Kamp and Hayashi 2009). A 10% hydrochloric acid (HCl) solution was dripped onto soil core samples to check for the degree of effervescence. Soils containing carbonates effervesce when treated with dilute HCl due to the production of CO_2 gas (Equation 3.1). An extremely strong effervescence reaction at the soil surface indicated an enrichment of secondary carbonates. GPS coordinates of these soils were then collected in a similar fashion to the collection of GPS coordinates for soil salinity.

$$CaCO_3 + 2HCl = H_20 + CO_2 + Ca^{2+} + 2Cl^- \qquad (3.1)$$

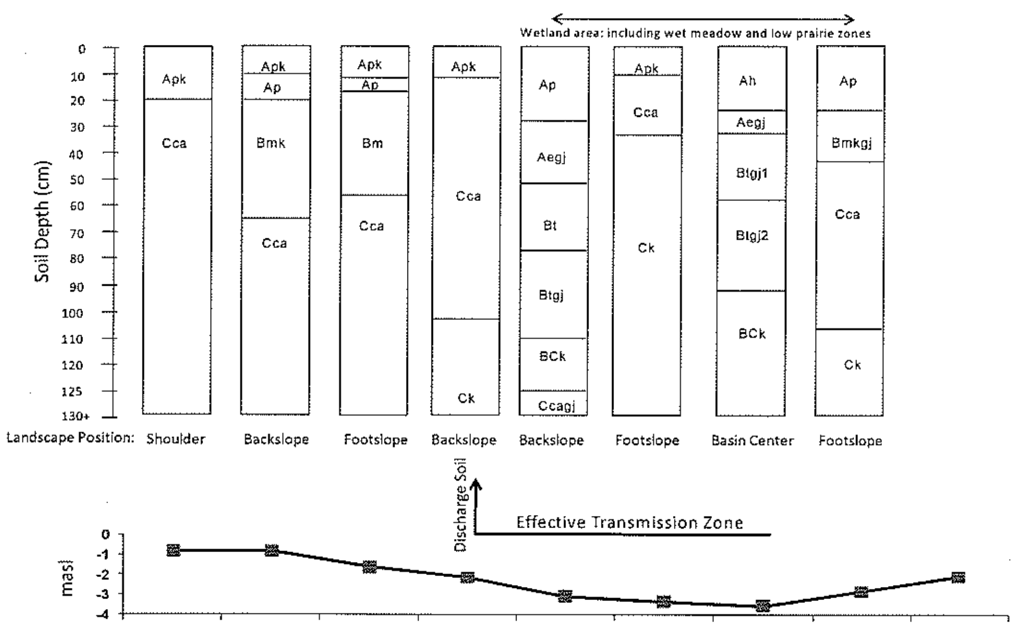


Figure 3.6 Example of a carbonate discharge soil located upslope of a Class III wetland at St. Denis.

3.3.4 Model Description

A series of landform segmentation programs described by Pennock et al. (1987), Martz and de Jong (1988), Bedard-Haughn and Pennock (2002) and Pennock (2003) were used to subdivide the digital elevation model (DEM) for each site into discrete landform elements based on landscape elements such as planned curvature, profile curvature, gradient and aspect. Kriging with the default linear variogram using Surfer© (Golden Software, Golden, CO) was used to generate a DEM with 10-m spacing from sub 1-m Light Detecting and Ranging (LiDAR) data available for both study sites. The LiDAR data collected in August of 2005 at St. Denis is described by Fang (2007). LiDAR at the MZTRA site were collected using a 5-cm vertical resolution by Aeroscan International Inc. in October 2002. A mathematical filter was used to remove elevation outliers >2-m due to vegetation and buildings. The LiDAR data were interpolated with kriging to a 5.0 m grid and filtered with a 3 by 3 smoothing filter. Chaubey et al. (2005) recommend that input DEM data match, as close as possible, the scale of the other data sets it is combined with, such as spatial vegetation data. Therefore, the DEMs in this study were generated to match the scale of the vegetation data which was collected at 10-m increments in the field. The DEMON program by Costa-Cabral and Burges' (1994) was used to determine and calculate the specific catchment area (SCA) and specific dispersal areas (SDA) for each wetland. One of the challenges with flow models is their inappropriate one-dimensional down slope projection of flow. DEMON offers the main advantage of contour-based models with flow width represented as a function of local topography, allowing for the computation of both SCAs and SDAs. Calculating the SDA is critical for delineating a wetland's basin and for determining its potential water storage. Two additional programs were used to conduct further landform segmentation on the data. LANDFM assigns the landform element complex information to the data (Pennock et al. 1994), while LEClump groups the contiguous landform element complexes into clusters and then assigns a cluster maximum to each SCA and SDA value (Bedard-Haughn and Pennock 2002, D. Pennock unpublished data).

WaterLevels is a single-basin hydrologic program (Appendix B) designed to model spring runoff events in increments of 1-cm using the true bottom bathymetry of the ponds. Since WaterLevels models the runoff as a spring snowmelt event, it assumes that an ice seal exists on the soil surface. Therefore, the entire runoff volume enters the basin. LiDAR data cannot accurately collect elevation information through standing water. As a result, the bottom elevations for ponds 109, 120, 65, 66, 67, 216, 222 and 232 were recorded in the field using the Sokkia Total Station. These new data points replaced the LiDAR data points in the generated DEM's for these wetlands. Data input requirements for WaterLevels include: the runoff event (mm), DEM's for each wetland and its catchment watershed, grid size of the DEM, the spill-over point for the wetland (masl), the bottom elevation of each wetland (masl), and the average water level for each wetland in the fall (masl). Data outputs include: water levels in the wetland before and after the event (masl), water level rise in the wetland (m), pond area (m^2), pond volume before and after the event (m^3), runoff volume (m^3), and catchment area (m^3). The spill-over volume (m^3) from the wetland is also provided if the precipitation event was great enough to cause the wetland to spill-over into adjacent locations downslope.

3.3.5 Model Runs

To estimate reasonable spring runoff scenarios for the model we calculated winter precipitation values for each study site. These calculations combined rainfall and snowmelt equivalent values for December to April using monthly weather data provided by Environment Canada (2009). Climate data from the Diefenbaker Airport in Saskatoon were used for the St. Denis wetlands, while climate data from the Brandon Airport were used for the MZTRA wetlands in Manitoba. These data sets were considered the most reliable long-term data sets for each site. The average winter precipitation from 1971 to 2000 was 80-mm for Saskatoon and 104-mm for Brandon. It is estimated that between 30% and 60% of spring snowmelt on the prairies flows into wetlands in a given year (Hayashi et al. 1998*a*; Bedard-Haughn 2001; Berthold et al. 2004). Much of the loss that occurs can be linked to sublimation. Both Pomeroy et al. (1993) and Burford and Stewart (1998) found that sublimation snow loss accounted for 40% to 70% of annual snow loss. Therefore, three initial model runs using 30%, 60% and 100% of the total winter precipitation were performed for each wetland at each location. This meant 24-mm, 48-mm, and 80-mm spring runoff events were modeled for St. Denis wetlands, and 30-mm, 60-mm, and 104-mm spring runoff events were modeled for MZTRA wetlands.

3.3.6 Assessing Model Accuracy

The WaterLevels program is designed to allow the programmer to map specific water depths for each wetland using the output data produced by the program because the true bottom bathymetry of the wetland is used in the model. The areal extent of these water depths can then be compared to the spatial distribution of the vegetation data collected in the field to determine if the modeled versus actual data sets align. Vegetation communities in prairie wetlands require specific water depths in order to survive and thrive (Stewart and Kantrud 1971; van der Valk 2000). Although specific plant species may change within vegetative zones in wet versus dry years on the prairies, wetland vegetation zones tend to be static in the spatial locations of their outer margins, particularly the shallow marsh and wet meadow zones. Therefore, the spatial location of vegetation zones in wetlands can be used to predict the locations of specific water depths within the ponds (van der Valk 2000).

In addition to the vegetation data, spring water depths data for St. Denis wetlands was valuable for assessing the accuracy of the model and for determining which snowmelt scenario best matched the long-term spring water level depths in each pond.

3.3.7 Statistical Analysis

Yearly precipitation totals and water levels were assessed for normality using Shapiro-Wilk tests (Shapiro and Wilk 1965). We studied two time periods of winter precipitation to look for common patterns and to help us better interpret any commonalties we might observe in our application of the WaterLevels model at the two sites. These two periods included the long-term average precipitation that accumulates from December to April and the precipitation that falls only in March and April. For those two periods we also looked at the probability of 24, 48, and 80-mm of precipitation falling in Saskatoon, and 30, 60, and 104-mm of precipitation falling in Brandon. We calculated the probabilities of attaining various precipitation or water level thresholds using normal distribution approximations where appropriate. The empirical distribution function (Conover 1980) was used to estimate specified thresholds for data not well approximated by a normal distribution. T-tests comparing September, October, and November average water levels from 1968 to 2006 were used to determine which month(s) best represented the average long-term water depths of wetlands at St. Denis in the fall. This fall water depth was then used as the input into the WaterLevels program.

3.4 Results

3.4.1 Wetland Characteristics and Vegetation Distribution

The Class IV wetlands in this study were all noticeably larger wetlands except for ponds 216 and 232 at the MZTRA site (Table 3.1). With the exception of pond 232, their surface water conductivities were also higher. All Class IV wetlands in this study had a deep marsh, shallow marsh, wet meadow and low prairie zone (Table 3.3). The deep marsh zone was not present in Class III wetlands at St. Denis. This zone is normally absent from Class III wetlands (Stewart and Kantrud 1971). The shallow marsh, wet meadow and low prairie vegetation zones were used to indicate the spatial extent of water depths in each wetland. Figure 3.7 provides a visual perspective on the location of these zones in six of the nine wetlands. Water depths become shallower and the vegetation zones change as one moves from the center of the basin towards the basin's outer edge. Therefore, shallow marsh vegetation is usually located in deeper water than either wet meadow or low prairie vegetation. The mean water depth of the shallow marsh zone in July and August for wetlands in this study was 15.7-cm, compared to 4.9-cm for the wet meadow zones, and 1.8-cm for the low prairie zones.

Table 3.3 Vegetation zones present in each study wetland.

Wetland	**Pond #**	**Class IV**	**Class III**	**Deep Marsh (DM)**	**Shallow Marsh (SM)**	**Wet Meadow (WM)**	**Low Prairie (LP)**
MZTRA	232	√		x	x	x	x
MZTRA	222	√		x	x	x	x
MZTRA	216	√		x	x	x	x
St. Denis	109		√		x	x	x
St. Denis	117		√		x	x	x
St. Denis	120		√		x	x	x
St. Denis	65	√		x	x	x	x
St. Denis	66	√		x	x	x	x
St. Denis	67	√		x	x	x	x
Total:	9	6	3	6	9	9	9

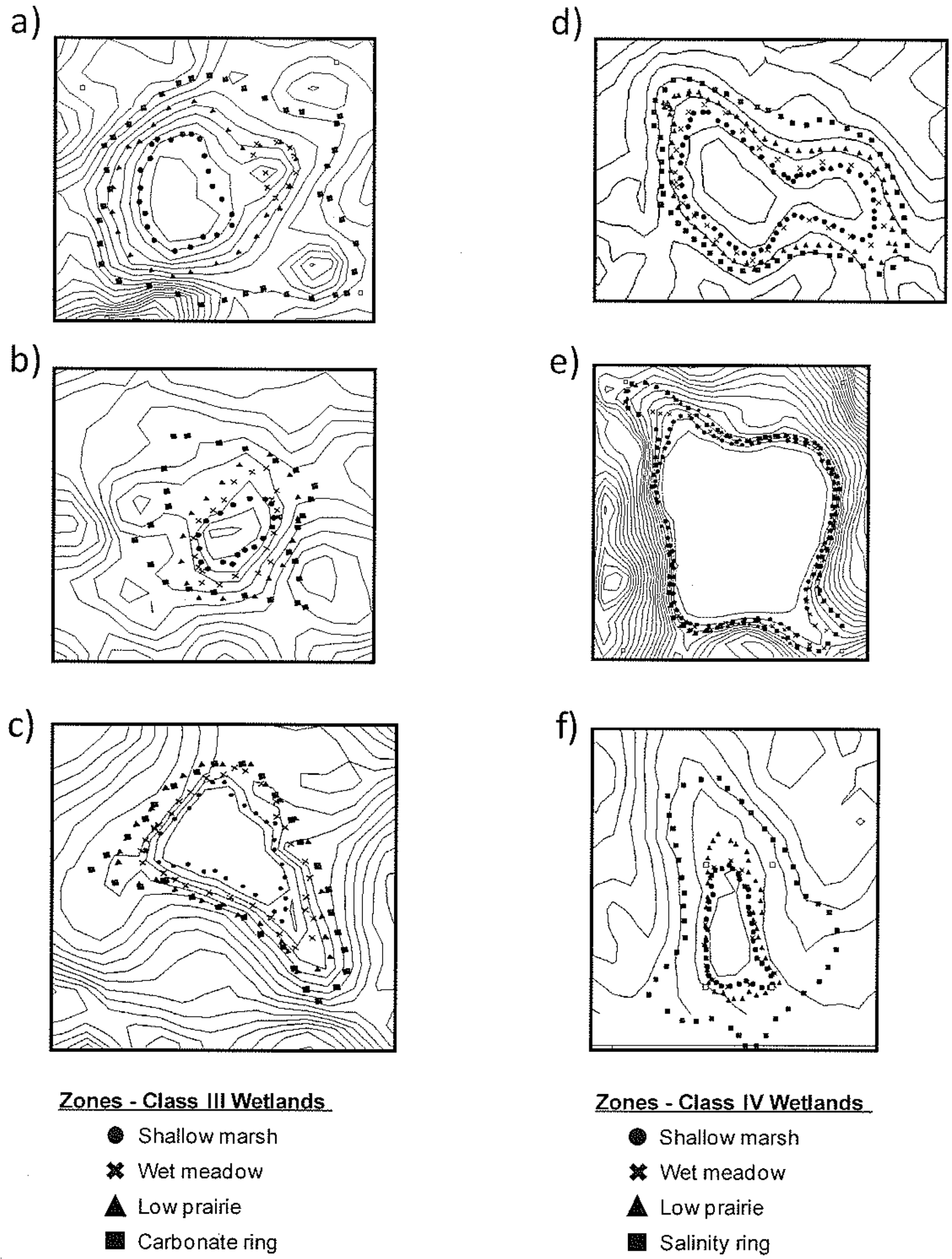


Figure 3.7 Vegetation zones and the extent of the effective transmission zones (carbonate or salinity ring) in: Class III wetlands a) 109 b) 117 c) 120, and Class IV wetlands d) 222 e) 67 f) 216.

3.4.2 Soil Salinity and Soil Carbonate Distribution around Wetlands

Soil salinity never exceeded 40 mS m^{-1} around Class III wetlands in this study. As a result, upland soils were sampled for the presence of concentrated secondary carbonates near the soil surface. Figure 3.7 shows the boundaries of these carbonate soils around ponds 109, 117 and 120. Our results indicate that discharge soils were spatially located at consistent elevations around each Class III pond. Discharge soils were located closer to the wetland edge in landscape locations where slopes were steeper and further away from the wetland edge in locations where the surrounding slopes were more gradual (Figure 3.7).

Soils with conductivities exceeding 40 mS m^{-1} were present around all Class IV wetlands in this study except wetland 232 at the MZTRA site. The EM38 performed well at predicting changes in soil salinity at both sites. The R^2 values for ECa versus ECe was 0.99 and 0.70 for MZTRA and St. Denis soils, respectively. An ECa of 40 mS m^{-1} represented an ECe reading of approximately 1 dS m^{-1} for both locations, whereas a field reading of 100 mS m^{-1} represented a soil ECe of 3.1 dS m^{-1} at the MZTRA site and 4.3 dS m^{-1} at the St. Denis site. Soils with an ECe of 3.0 dS m^{-1} are considered slightly saline, while soils with an ECe of 4.0 to 8.0 dS m^{-1} are considered moderately saline (Henry 2003). The distribution of saline soils around Class IV wetlands followed a similar pattern to the distribution of discharge carbonate soils. Saline soils extended further away from the wetland edge in locations where slopes were gradual and pulled in closer to the edge where slopes were steeper (Figure 3.7). While seasonal wetlands possessed a distinct ring of discharge soils around them, semi-permanent wetlands possessed an entire band of saline soils around them. Saline soils, like carbonate discharge soils, appear to be associated with a consistent elevation around each wetland.

3.4.3 Model Simulations

3.4.3.1 Snowmelt Scenarios

Over 300 model simulations were completed with the WaterLevels program. Three initial snowmelt events of 30, 60 and 100% were run to determine which event best predicted the spatial distribution of the vegetation in each wetland. If none of these events had done well at predicting the distribution of vegetation then further runoff scenarios would have been modeled. Study findings indicate that the model's predictions for the 24-mm runoff event at St. Denis and the 30-mm runoff at the MZTRA site fit best with both the spatial distribution of vegetation in the ponds and the water depth requirements of vegetative zones in the spring. These findings were consistent across all wetlands regardless of wetland Class or surrounding land-use. Figure 3.8 provides a comparison of the model's predictions for 30% and 100% snowmelt event for wetlands 67 and 117 at St. Denis and wetland 222 at the MZTRA site. The figure shows the spatial extents of water depths predicted by the model to the spatial locations of the vegetative zones. Model predictions for wetlands 67 and 117 also include additional spillover amounts from nearby upslope wetlands (i.e. 67a, 117a, 117b). We found that the water depth predications for the 30% snowmelt scenario coincided with the spatial locations of both the wet meadow and shallow marsh zones. In most years one would not expect the low prairie zone to be flooded in the spring. This was not the case for the 80-mm scenario at St. Denis or the 104-mm scenario at the MZTRA site (Figure 3.8).

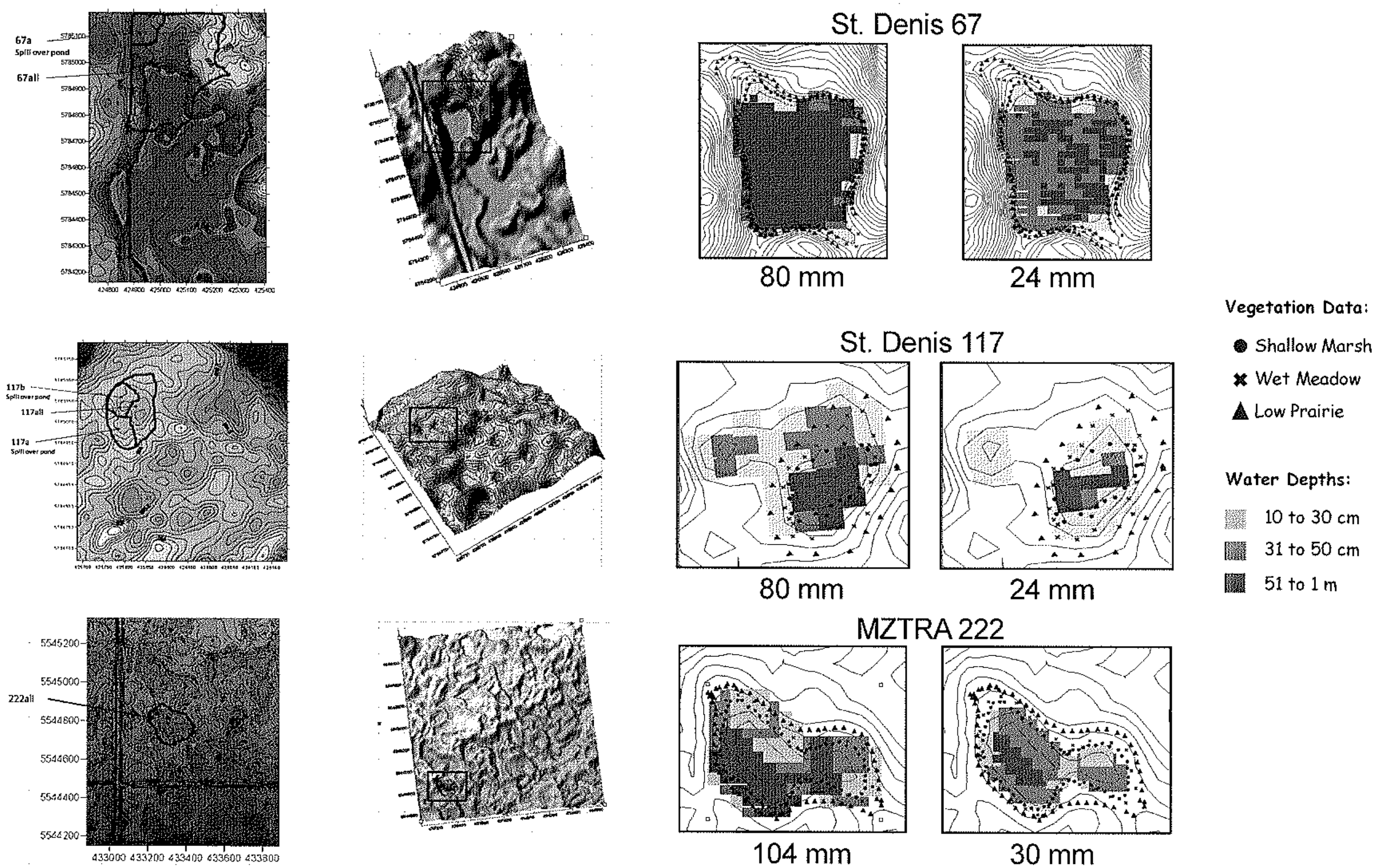


Figure 3.8 Water depth predictions by WaterLevels for a 30% and 100% snowmelt runoff event for wetlands 67, 117 and 222.

Model output data were used to determine which runoff event best predicted the actual mean pond depth of St. Denis wetlands in April and May. Similar to the findings for wetland water depths and vegetation distribution, our results indicate that the 24-mm snowmelt event best predicted the long-term mean April/May pond depths of wetlands at St. Denis (Table 3.4). All water depth predictions by the model for the 24-mm event were within 13% of the actual pond depths except for pond 117 where the prediction was more than double the mean value. Both the 48-mm and 80-mm model predictions overestimated the spring pond depths for all wetlands.

Table 3.4 Spring pond depth predictions from the WaterLevels model for a 24, 48 and 80-mm snowmelt event at St. Denis.

	Pond #	Mean Fall[1] Pond Depth (cm)	Mean[2] Spring Pond Depth (cm)	Predicted Pond Depth (cm)	Predicted Pond Depth (cm)	Predicted Pond Depth (cm)
Runoff Event:				**24-mm**	**48-mm**	**80-mm**
Class IV Wetlands	**65**	63.2	85.6	83.0	121.0	134.0
	66	27.0	40.4	37.0	48.0	61.0
	67	56.0	71.1	70.0	85.0	1.02
Class III Wetlands	**109**	10.7	47.9	49.0	79.0	104.0
	117	0.0	31.6	70.0	91.0	97.0
	120	18.6	53.2	60.0	78.0	78.0

[1] Fall water depths used as inputs into the model were calculated from t-tests comparing September, October, and November average water levels from 1968 to 2006.
[2] Actual mean April/May water depths were calculated from field data collected from 1968 to 2006.

3.4.3.2 Spillover Scenarios

A series of additional snowmelt scenarios, using the mean fall water depth as input into the model, were run to determine how much snowmelt is required for a pond to exceed its maximum storage capacity and spillover. Additional spillover from upslope wetlands were included in the snowmelt amounts for ponds 67, 109, 117, and 232 (i.e. 67a, 109a,b, 117a,b, and 232a). Table 3.5 shows the model estimates for each wetland along with the snowmelt runoff required to cause the spillover of a pond. Very high snowmelt events were required to cause spillover of ponds 65, 66, 109, 222, and 232 (116-mm to 335-mm), compared to ponds 67, 117, 120, and 216 (36-mm to 76-mm). What was evident from our modeling efforts was how uncorrected LiDAR data affected spillover estimates. Only 55-mm of snowmelt was needed to cause pond 222 to spillover when uncorrected bottom elevations were used in the model. This compares to 355-mm using corrected bottom elevations. The runoff amount required to cause the spillover of pond 232 dropped to 14-mm using the uncorrected elevations from 116-mm with corrected elevations. Using uncorrected bottom elevations also decreased the volume capacities of ponds 232 and 222 from 1168-m^3 to 70-m^3 and 12197-m^3 to 3012-m^3, respectively.

Table 3.5 Model estimates of catchment area, pond area and volume capacity for each wetland and their associated spillover wetlands using a 24-mm (St. Denis) or 30-mm (MZT=MZTRA) snowmelt scenario in the model.

Site	Pond #	Catchment Area (ha)	Wetland Area (ha)	Catchment: Wetland Ratio	Potential Wetland Volume (m^3)	Wetland Volume before Event (m^3)	Runoff needed for Spillover to occur (mm)	Runoff Volume produced by a 24-mm/30-mm Event (m^3)
St. Denis	65	4.9	2.3	2.1	8967.0	803.5	167	3787.9
	66	24.0	7.9	3.0	57295.3	1613.5	226	15567.0
	67	13.0	2.2	5.9	15395.5	6083.0	76	7975.3
	67a	1.2	0.4	3.0	441.2	0.0	38	935.0
	109	3.3	1.2	2.8	6708.8	58.6	164	2549.3
	109a	0.6	0.06	10.0	204.6	0.0	33	503.5
	109b	0.4	0.03	13.3	105.4	0.0	30	279.7
	117	1.7	0.4	4.3	1303.5	0.0	56	1318.6
	117a	0.4	0.05	8.0	16.1	0.0	5	319.7
	117b	0.4	0.3	1.3	42.0	0.0	11	319.7
	120	3.5	0.3	13.0	1411.0	199.0	36	2733.1
MZT	216	3.5	0.5	7.0	1237.6	0.0	36	1035.0
	222	3.2	0.8	4.0	12196.7	992.5	355	840.6
	232	0.5	0.1	5.0	1168.2	0.0	116	158.5
	232a	0.7	0.13	5.4	214.8	0.0	33	200.4

3.4.4 Spring Water Depths

The relative frequency histograms of April/May mean water depths for St. Denis wetlands indicate a wide range of water depths in Class IV wetlands (Figure 3.9) compared to Class III wetlands at St. Denis (Figure 3.10). While the distributions of water depths in Class IV wetlands tend to be more symmetric about their means, water depths in Class III wetlands are skewed towards shallower water depths in the spring, particularly in ponds 109 and 117. Pond 117 is unique in that it displays a bimodal distribution, with spring water depths distributed more often at shallower (5-cm) and deeper water depths (70-cm) and less often around its mean April/May water depth of 31.6-cm.

Water depths were the deepest in ponds 65 and 67 and the shallowest in 117. Although pond 66 is the largest wetland in this study, it is not that different from ponds 109 and 120 in terms of its April/May mean water depth or in its 20th, 50th and 80th percentile values. What is different is the increased frequency with which ponds 109 and 120 hold very little, or no, water in the spring. Pond 117 is unique compared to the other wetlands at St. Denis in the number of years it holds very little water in the spring. The percentiles for this pond indicate that spring water depths in this wetland are 5-cm or less approximately 30% of the time.

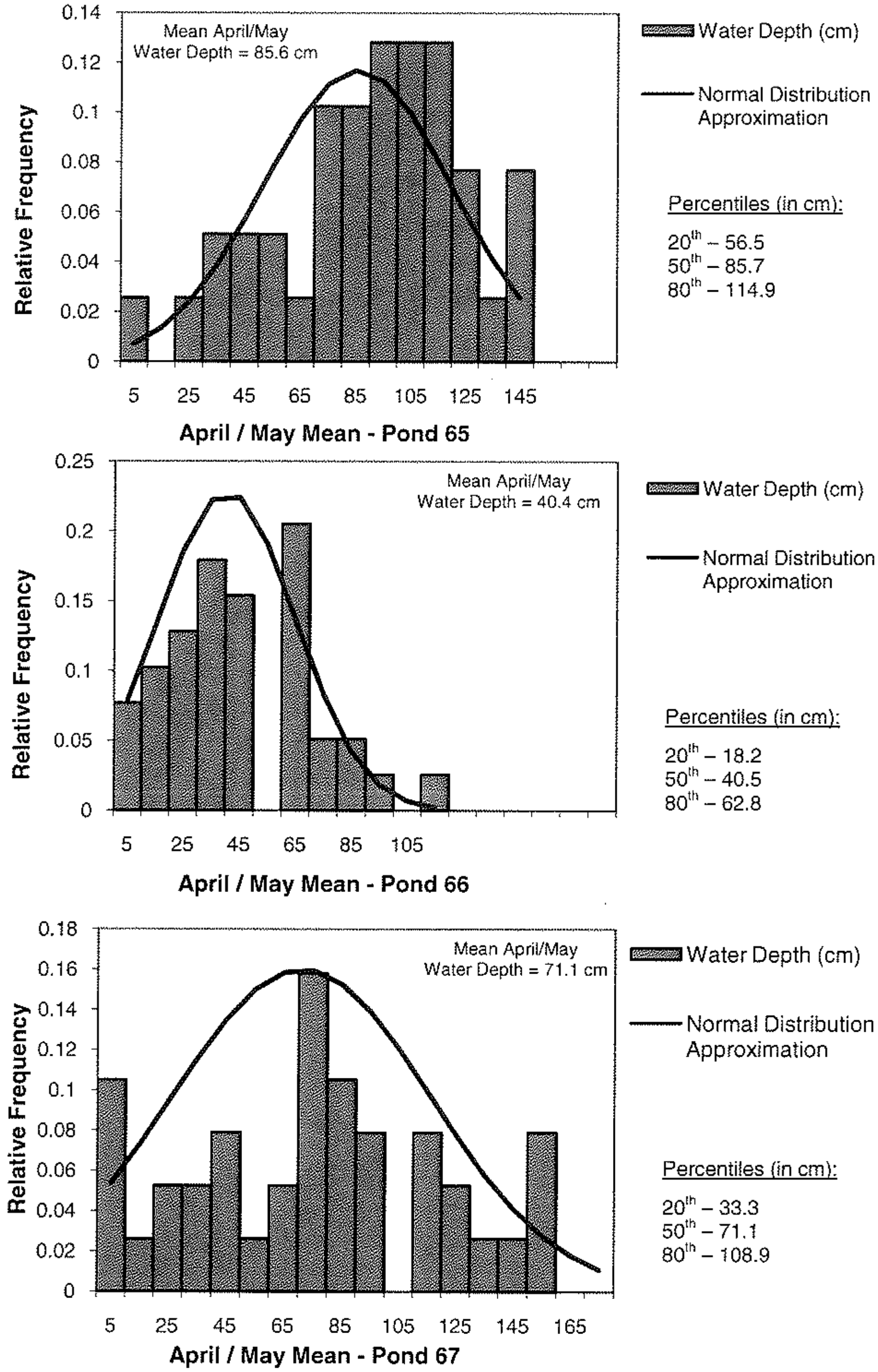


Figure 3.9 Relative frequency histograms and the 20th, 50th and 80th percentile values of combined mean April/May water depths for Class IV wetlands at St. Denis, SK, from 1968 to 2006.

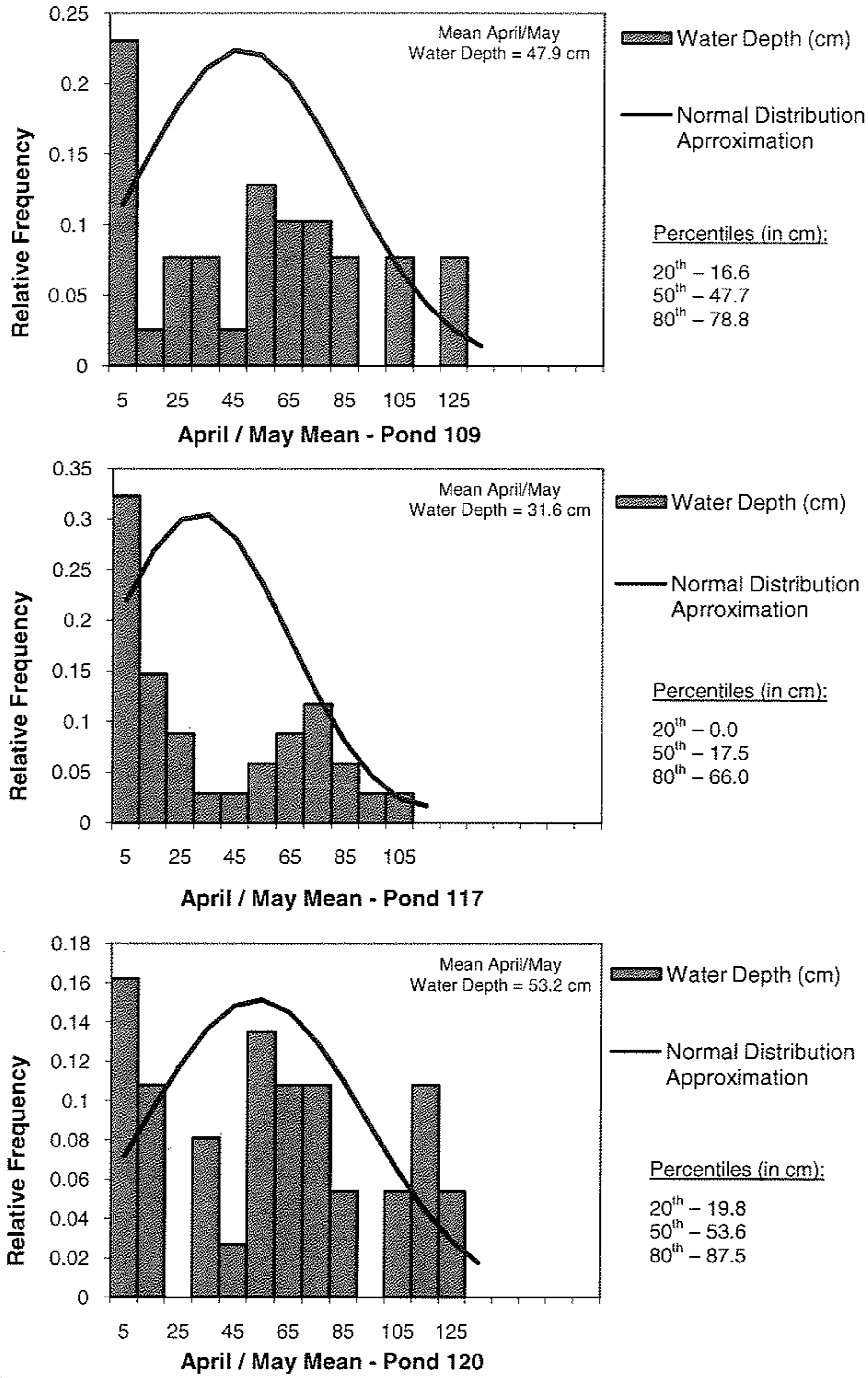


Figure 3.10 Relative frequency histograms and the 20th, 50th and 80th percentile values of the mean April/May water depths for Class III wetlands at St. Denis, SK, from 1968 to 2006.

3.4.5 Winter Precipitation Patterns

The MZTRA site in Manitoba receives approximately 110-mm more annual precipitation than St. Denis. Some of that difference can be attributed to the additional 24-mm of precipitation Manitoba receives from December through April. The mean winter precipitation totals for December to April at the Saskatoon and Brandon Airports are 80-mm and 104-mm from December to April and 38.5 and 53.5 in March and April. Relative frequency histograms of precipitation for these two time periods are shown in Figure 3.11. A more diffuse distribution of winter precipitation values occurs during the December to April time period at both the Brandon and Saskatoon Airports. This indicates that there is more variation in precipitation when more months are included in the analysis. The probability that we observe 80-mm of precipitation in Saskatoon and 104-mm of precipitation in Brandon from December until April is approximately 50%, or every second winter. The probability that only 30% of the mean winter precipitation falls in any given year from December until April is only 1 to 2 % for either site, or about a 1 in 100 year event (Figure 3.11). This probability increases to 21%, or about 1 in every 5 years, when the precipitation amounts during only March and April are considered.

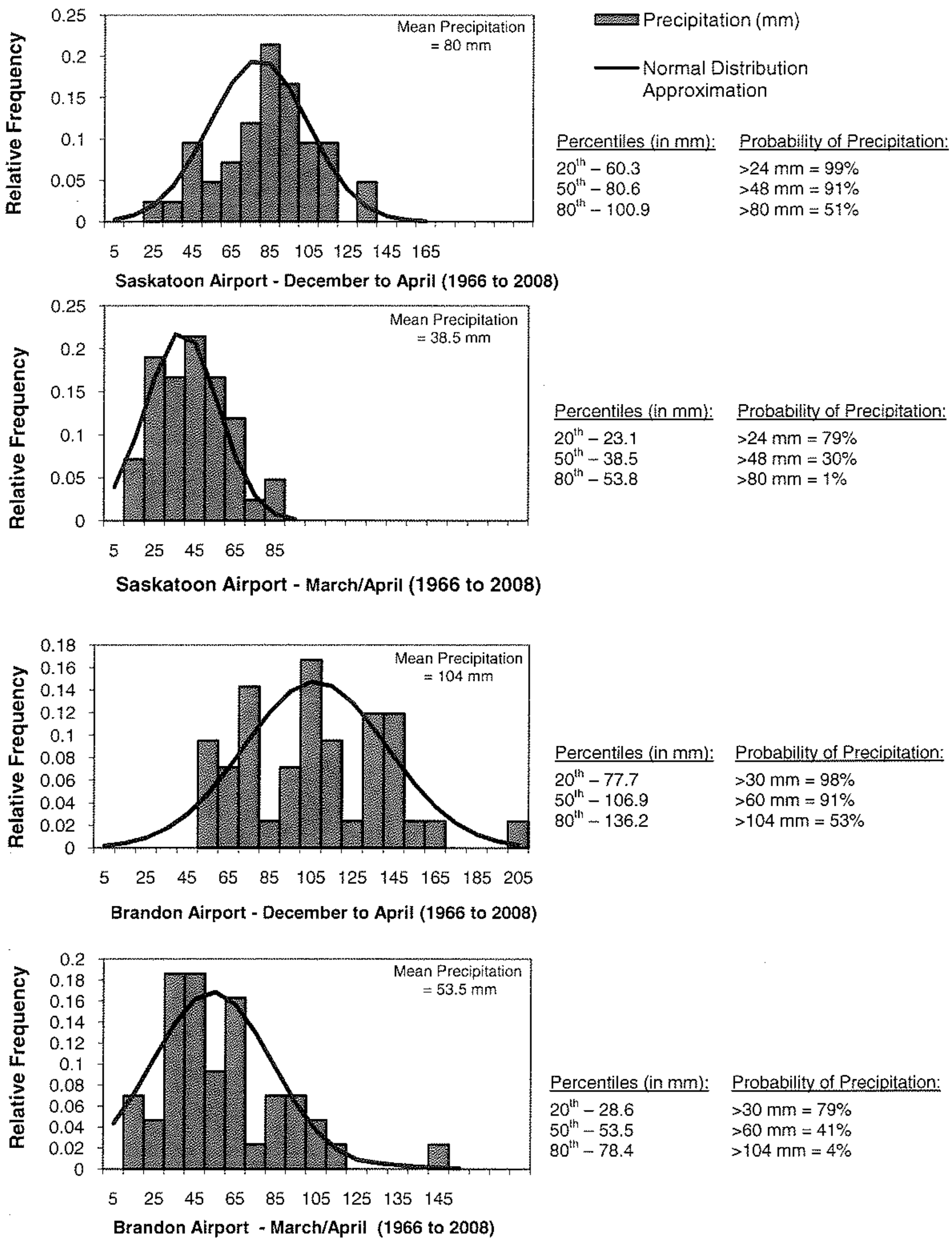


Figure 3.11 Relative frequency histogram for precipitation from December to April and from March to April at the Saskatoon and Brandon Airports from 1966 to 2008.

3.5 Discussion

3.5.1 Snowmelt Runoff and Model Predictions

Prairie wetlands can receive runoff during the year either through snowmelt in the spring or from heavy rains in summer. Research indicates that it is spring snowmelt that supports their hydrology in most years (Woo and Rowsell 1993; Woo and Winter 1993; Hayashi et al. 1998*a*; Su et al. 2000; Berthold et al. 2004). Only exceptionally heavy storms result in overland flow during the summer months (Hayashi et al. 2003). Some snowmelt in wetlands accumulates through wind-driven snowdrifts (Pomeroy et al. 1998), but most snowmelt comes from overland flow in the spring over frozen soils (Hayashi et al. 2003; Voldseth et al. 2007). The permeability of clay-rich soils can be extremely low when soil is frozen in the winter (Berthold et al. 2004). Hayashi et al. (2003) looked at the infiltration of snowmelt in frozen soils at St. Denis and found the infiltration rate unmeasurable (= 0) in 50% of their samples.

Of the few hydrological models that exist for prairie wetlands most consider a number of hydrologic inputs in their models such as evapotranspiration, soil moisture capacity, subsurface flow, snowpack accumulations and watershed-soil surface processes (Poiani and Johnson 1993; Poiani et al. 1996; Su et al. 2000; Voldseth et al. 2007). These models strive to simulate annual variations in wetland water levels to better understand the effects of groundwater interactions (Su et al. 2000), surrounding upland use (Voldseth et al. 2007), or long-term changes in climate (Poiani et al. 1996). Many of these studies focus on a single wetland because of the detailed knowledge required to populate the data requirements of the model. For this reason, it is difficult to look for common trends across multiple wetlands. Johnson et al. (2004) note that no existing models of prairie wetlands have been tested outside the data sets or wetlands under which they were parameterized. The WaterLevels program was used to model two study locations. In contrast to other wetland models, WaterLevels uses the simplified approach of treating frozen soil as one of the main parameters in the model. Therefore soil infiltrability is considered restricted (Gray et al. 2001) and total runoff input into the model enters the wetland. If the WaterLevels model is able to predict the distribution of vegetation and spring water depths in wetlands with reasonable accuracy, then the model can be used to study hydrological events that drive wetland productivity. A better understanding of these processes will lead to more informed discussions on topics such as the impact of climate change on wetlands and the role of wetland drainage on spring runoff.

The nine wetlands used in this study provide a good representation of the types of wetlands found in southern Manitoba and Saskatchewan. They varied in catchment size and water-holding capacity, surrounding land-use, and regional location. One key advantage of the WaterLevels model is its ability to predict spring water-level rise based on total runoff entered into the model. This allowed comparisons of the model's predictions for different runoff events to the actual mean water depths of St. Denis wetlands in April and May. A second advantage of the model was the ability to use the output data to specify and map different water depths using the true bottom bathymetry of the ponds. This allows the user to select and map water depth ranges that coincide with the water depths required by wetland vegetation in the spring. The findings in this study indicate that the spatial distribution of the vegetation in these ponds best matched the 30% snowmelt event in all wetlands regardless of wetland size, wetland class, regional location or surrounding land-use. April/May mean water depths for the St. Denis wetlands were also best predicted by the 30% snowmelt, except for pond 117.

Hydrological models that are designed to use wetland values derived from averages may produce unrealistic results when the model is applied to extreme conditions (Poiani et al. 1996). Our results suggest that assessing the model's accuracy using only the mean water depth of the pond may not always be appropriate for ephemeral wetlands, since these wetlands tend to be the most hydrologically variable (Johnson et al. 2004). The frequency distributions of spring water depths for Class III wetlands at St. Denis were skewed towards 0-cm indicating that there is a greater probability of encountering a spring when very little water is present in a pond. The WaterLevels model is not designed to capture a 0-cm spring pond depth unless both the fall pond depth and the runoff amount entered into the model equal zero. It is also important to note that the frequency distribution of spring pond depths for wetland 117 was bimodal. Therefore, the mean of 31.6-cm is not particularly representative of the spring depth of pond 117 in most years. While lower spring pond depths ($\leq$ 5-cm) for pond 117 likely represent years when very little snowmelt occurs and none of the upper contributing areas provide additional inflow into pond 117, springs when pond depths are higher ($\approx$ 70-cm) likely represent those years when the entire catchment of 117 contributes runoff into the pond.

What was unanticipated in this study was the consistent relationship we observed between almost all average spring water depths and the 30% snowmelt scenario. This occurred even though December to April precipitation values for both sites differs. March and April precipitation amounts combined alone do not account for the patterns we observed. However, the frequency with which the December to April precipitation amounts occurs in any given year is approximately 50% for both sites. Estimating spring runoff is challenging. Several unique factors can affect the amount of snow that accumulates, the extent of snowmelt that occurs, and the degree to which the snowmelt infiltrates frozen soil (Fang 2007). Snow loss to sublimation may also play a key role in the 30% snowmelt value we observed in this study. Both Pomeroy et al. (1993) and Burford and Stewart (1998) attributed 40% to 70% of annual snow loss in their studies to sublimation. Hayashi et al. (2003) looked at snowmelt runoff for pond 109 and three smaller adjacent basins in 1999, 2000, and 2001. In 1999 and 2000 they found very similar snowmelt amounts entering pond 109 as our own 30% value of 24-mm. In 2001 snowmelt runoff dropped to only 3-mm and they could provide no reason why the runoff had dropped so dramatically. Fang (2007) examined the proper scale for pre-melt snow accumulation as snow water equivalent (SWE) and snowmelt for pond 109 with the objective of deriving a snow hydrology model for the prairies. He used the snowmelt values determined by Hayashi et al. (2003) in the model and therefore their study provides no new additional comparative data for our own study. Poiani and Johnson's (2004) WETSIM 1.0 and 2.0 used snowmelt runoff in models designed to predict vegetative changes with climate change, but snowmelt was only used in those years when which amounts were considered significant or when snow depth exceeded 30-cm.

More focus has been devoted to the effect of climate change on evapotranspiration rates in prairie wetlands (Martin et al. 1989; Rosenberg et al. 1989; McKenney and Rosenberg 1993), as it is considered the main driver of water loss in prairie wetlands during the growing season (Shjeflo 1968; Woo and Rowsell 1993, Poiani and Johnson 2004). In comparison, little deliberation has been given to the impact of a changing climate on snowmelt runoff estimates, even though the melting of seasonal snowcover is considered one of the most important hydrological events of the water year (Zhao and Gray 1999). Part of this may relate to the lack of information on snowmelt runoff values for wetlands. Fang and Pomeroy (2007) found that a decrease in precipitation of 15% combined with a rise in air temperature of 2.5°C resulted in the cessation of snowmelt runoff in their models. Global Climate Models (GCMs) projected to the mid-21st century on prairie spring runoff show an initial increase in spring runoff of 24% by 2050 followed by a decrease of 37% by 2080 (Fang and Pomeroy 2007). This substantial decrease in snowmelt runoff by the end of this century could have serious implications for prairie wetland hydrology and productivity, since our model indicates that our wetlands are already at the lower end of the snowmelt scale at the 30% scenario.

It is difficult to explain why the 30% snowmelt event fit well with both native grassland and cropland wetlands. Cultivated lands and grasslands should have considerably different responses to year-to-year variations in snowfall and rainfall because of their different surface structures. However, van der Kamp and Hayashi (2009) point out that these differences are still not well understood. While studies have shown that snowmelt runoff is greater in cultivated versus grassland landscapes (van der Kamp et al. 2003; Voldseth 2004) and that water levels fluctuate much more in wetlands surrounded by annual crops than wetlands surrounded by grasslands (Euliss and Mushet 1996), this study found that over a 38-yr period the dominant snowmelt event in both grassland and cultivated landscapes remained the same, even across provinces.

3.5.2 Spillover Events

Surface connectivity, or episodic spillovers, between prairie wetlands has received less consideration in the scientific literature than subsurface connectivity (Leibowitz and Vining 2003). Characteristics of ground-water flow, atmospheric-water flow and catchment elements dictate the degree to which wetlands remain isolated or connected. Isolation in any given year is related to the height to a spill elevation and the recurrence interval of various magnitudes of precipitation for wetlands that spillover their surface watersheds (Winter and LaBaugh 2003). Isolated wetlands are those whose drainage divides are too high. These wetlands will neither contribute surface water to wetlands downslope nor receive spillover from wetlands located upslope. Discussions in the United States pertaining to the protection of prairie wetlands has lead to much debate in both the legal and regulatory arenas on the degree to which prairie wetlands remain isolated from one another (Nadeau and Leibowitz 2003). Unfortunately little documentation is available to help guide these discussions.

Four of the nine wetlands in this study (ponds 67, 117, 120, and 216) have the potential to spillover their surface watersheds. Conceptual prairie pothole groundwater models have relied on groundwater connectivity to explain inter-wetland differences in salinity, duration of inundation, and vegetation structure (Cook and Hauer 2007). However, it may be hypothesized that surface connectivity is as important a factor in explaining inter-wetland differences. Both Cook and Hauer (2007) and Leibowitz and Vining (2003) suggest that surface water connections have the potential to affect wetland hydrology, water chemistry, and the abundance and distribution of organisms. Leibowitz and Vining (2003) found the specific conductance of surface waters in two surface-connected North Dakota wetlands similar in the spring, but very different at other times of the year. Eisenlohr and Sloan (1968) also report a similar finding between two surface connected wetlands in North Dakota's Missouri Coteau. Cook and Hauer (2007) studied the effects of surface connectivity on the vegetation and water chemistry of 34 wetlands in Montana. They found the mean specific conductance of surface waters three times higher in connected wetlands than in isolated wetlands and the catchment areas of connected wetlands proportionally smaller than the catchment areas of isolated wetlands. We similarly found that the wetlands with the greatest potential to spillover also had the largest catchment size to wetland size ratio.

The specific conductance of surface waters in this study appear to be better linked to wetland Class and the length of time water is held in a pond than to the surface connectivity between ponds. However, surface connectivity was not the primary focus of this study and it is possible that not enough wetlands were studied to get a true perspective of the influence of surface connectivity on water chemistry. Our data indicates that pond 117 has the potential to spill over into 120 yet their surface water conductivities are not that different from pond 109, which we consider an isolated wetland. Similarly, pond 67 has the potential to spill over into pond 66 yet its surface water conductivity is similar to pond 65. Water in ephemeral ponds such as 109, 117, 120, and 232 is primarily derived from snowmelt. Therefore their chemistries tend to reflect that of precipitation (Driver and Peden 1977; Swanson et al. 1988; Arndt and Richardson 1989). Higher surface water conductivities in ponds 65, 66, 67, 216, and 222 are most likely linked to the substantial loss of surface water through evapotranspiration

(Arndt and Richardson 1989; Waiser 2006) resulting in the concentration of solutes in the water due to evaporative concentration.

3.5.3 The Effective Transmission Zone and the Movement of Solutes

Since the late 1980's, many of the dominant prairie wetland hydrological models have focused on the link between wetland hydrology and local groundwater movement (Richardson and Arndt 1989 ref 271; Richardson and Vepraskas 2001; Winter and LaBaugh 2003). From these models three types of prairie wetlands have been described (Lissey 1971; Arndt and Richardson 1989; Richardson and Arndt 1989; Winter 1989; Winter and Rosenberry 1995). Recharge wetlands, such as ponds 109, 117, and 120, receive a majority of their water from depression-focused overland flow and snowmelt. Flow-through wetlands are those ponds that receive water from, and yield water to, the groundwater system. Discharge wetlands, such as 65, 66, 67 and 222 are those ponds that receive most of their water through groundwater discharge with no losses except through evapotranspiration.

Recent findings have brought into question how much connection there is between the surface waters of wetlands and the groundwater below, with the degree of connection controlled to a large extent by the hydraulic conductivity of the geologic materials through which groundwater flows (Winter and LaBaugh 2003). The hydrogeology of the northern prairie wetland region is dominated by glacial deposits that are tens of meters to hundreds of meters thick. These deposits consist largely of clay-rich glacial tills interspersed with deposits of glaciolacustrine clay and silt, and glaciofluvial sand and gravel (van der Kamp and Hayashi 2009). The hydraulic conductivity of these tills depends on the degree to which they are fractured, with most fractures occurring in the zone of oxidation (van der Kamp and Hayashi 2009). Research at St. Denis on the hydraulic conductivity of soils has shown that conductivity generally decreases with depth below the pond, ranging from values as high as 1,000 m yr^{-1} just below the ground surface to less than 0.1-m yr^{-1} at depths below 4 to 5-m and even smaller values at greater depths (van der Kamp and Hayashi 2009). Hayashi and van der Kamp (1998*a*) found that the net recharge rate of the aquifer underlying pond 109 was only 1 to 3-mm $year^{-1}$. Similar findings of low conductivities have been found by Shaw and Hendry (1998) and Zebarth et al. (1989) in Saskatchewan and MacLean and Pawluk (1975) in Alberta.

One key mechanism of soil-water movement in this study involved the lateral movement of water through the effective transmission zones (ETZ) around wetlands and into the surrounding upland soils (van der Kamp and Hayashi 2009). The ETZ is particularly important around wetlands positioned over soils of low hydraulic conductivity. It is also essential in the transmission of solutes from the wetland to the surrounding uplands (van der Kamp and Hayashi 2009). Research on pond 109 found that 75 to 80% of water lost from the pond was due to the lateral flow of water away from the pond margins and into the ETZ (Hayashi and van der Kamp 1998*a,b*). Our own research using soil resistivity and carbonate distribution to detect the extent of the ETZ revealed that water has the potential to move great distances laterally through the soil in both seasonal and semi-permanent wetlands. What is also evident from this study is that the solutes being transported and deposited in the soils around seasonal and semi-permanent ponds are not the same even though the mechanisms of water movement may be similar.

Solute differences observed in the soils surrounding Class III and IV wetlands at St. Denis and the MZTRA were the result of two distinct ionic dominance patterns in water chemistries. The main cations/anions present in seasonal Class III ponds are calcium (Ca^{2+}), potassium (K^+), and bicarbonate (HCO_3^-), compared to magnesium (Mg^{2+}), sodium (Na^+), and sulfate (SO_4^-) in semi-permanent Class IV ponds (Dan Pennock, pers. comm.). Similar examples of differences in pond chemistry and by wetland permanence have been observed in studies by Driver and Peden (1977), LaBaugh et al. (1987), Swanson et al. (1988), and Arndt and Richardson (1989). Hardie and Eugster (1970) were among the first to propose the mineral dissolution/precipitation reaction, or evaporative pathway, that results in the chemical differences observed in St. Denis and MZTRA ponds. Gorham et al. (1983) and Arndt and Richardson (1989) added to this knowledge by linking the evaporative pathway to increases in specific conductivity. Calcite is the first to precipitate at an electrical conductivity (EC) of 1 dS m^{-1} (Gorham et al. 1983; Arndt and Richardson 1989). Magnesium, sodium and sulfate become more concentrated as water evaporates from the surface of more permanently flooded wetlands (Hardie and Eugster 1970). Gypsum occurs as the second precipitate at an EC of 3.7 dS m^{-1} depending on the amount of calcium present in the pond after calcite precipitation (Arndt and Richardson 1989; Steinwand and Richardson 1989). Sodium, magnesium and sulfate increase at a linear rate with continuing increases in EC above 3.7 dS m^{-1} (Arndt and Richardson 1989). This process of evaporitic concentration results in water dominated by Mg-Na-SO_4-Cl or the group "D" pathway of brine evolution described by Hardie and Eugster (1970).

Calcium, bicarbonate and calcite precipitate reflect the influence of snowmelt chemistry in Class III wetlands at St. Denis and the importance of atmospheric precipitation and the dissolution and transport of carbonate minerals from the surrounding tills (Hayashi et al. 1998*b*; Heagle et al. 2007). The concentration of carbonates in the upper soil horizons around seasonal ponds was essential in indicating the location of discharge soils and the extent of the ETZ in this study. Knuteson et al. (1989) observed similar discharge soils with upper calcic horizons (Bearden soil) adjacent to ephemeral wetlands (Lindaas soil) in eastern North Dakota. They found that an argillic horizon of low permeability in the Lindaas soil caused waters with high levels of carbonate and bicarbonate to move laterally away from the wetland edge and vertically upward by evaporative discharge as $Ca(HCO_3)_2$ where it precipitated as $CaCO_3$ when the solution reached the solubility limits of calcite (Richardson et al. 1992). They also found that calcite remained fixed in upper soil horizons even when sufficient downward leaching of water occurred from the soil surface. They estimate it took between 5000 and 6000 years to deposit the amount of $CaCO_3$ they observed in calcic horizons. Miller et al. (1985) also describe "rego ring" soils, or Rego Dark Brown Chernozems, adjacent to wetlands at St. Denis. These soils had thin solas, low concentrations of soluble salts, with carbonates to the surface associated with well developed Cca horizons. They attributed low soil EC in these soils to downward leaching during spring snowmelt.

Gypsum and other soluble salt accumulations around Class IV wetlands at St. Denis and the MZTRA site reflect the influence of Hardie and Eugster's (1970) evaporative pathway through the evapotranspiration and concentration of solutes in more permanently flooded wetland systems. Steinwand and Richardson (1989) mapped gypsum concentrations along the edges of semi-permanent wetlands using an EM38 and found that gypsum formation, as with our own study, was strongly correlated to the hydrological permanence and chemistry of the wetland. They concluded that gypsum migrates in solution to the wetland edge by saturated water flow and then moves by unsaturated flow to upslope areas of lower water potential where it precipitates as the soil solution becomes concentrated by evapotranspiration. They also found that gypsum accumulations were transient and that frequent inundations with fresh pond water from spring snowmelt and intense summer and fall storms restored a leaching regime in the coarser upper portions of the pedons surrounding the ponds. Knuteson et al. (1989) also found that gypsum was short-lived in the upper horizons of Beardon soils because of its solubility.

What is evident in Knuteson et al.'s (1989) paper, but not discussed by the authors, is the spatial relationship that appears to exist between calcic soils and the wetlands. As in this study, discharge soils around a basin tend to be located at a specific and consistent elevation relative to the wetland itself. Discharge soils are found closer to the wetland edge where slopes are steep and extend further away from the wetland edge where slopes are more gradual. While the approach for delineating the edge of the ETZ differs for Class III versus Class IV wetlands, the spatial relationship we observe between the wetland itself and the specific elevation at which the outer edge of the ETZ occurs remains the same. Little is known about the factors influencing the extent of the ETZ around ponds. Hayashi et al. (1998*a*) found that the magnitude of horizontal flow under pond 109 was approximately equal to that of effective snowmelt runoff and that the complex variations they observed in subsurface flow between the pond and the surrounding uplands was likely due to changes in the horizontal hydraulic gradient. Thompson et al. (1998) suggest that variations occur because of a combination of surface topography and subsurface stratigraphy that preferentially direct subsurface waters to different areas of the landscape. They found that terrain attributes alone could not adequately explain the flow of water over and through the soils in their study.

Changes in the horizontal hydraulic gradient alone cannot completely explain the relationship we observed between the gradient of slope and the extent of the ETZ. Soil textures differ between wetland types, as do soil horizon development in Class III and IV ponds. One hypothesis that has been proposed is that the spatial extent of the EZT in Class III wetlands is determined by the amount of snowmelt runoff entering the wetland as well as the formation of an argillic horizon within and around the margins of a basin (Knuteson et al. 1989). The argillic horizon forms over many years by downward, saturated flow in recharge areas within the pond. The spatial extent of the ETZ in Class IV wetlands would also, in part, be determined by the amount of snowmelt entering the pond each year. However, it is possible that the spatial extent of the ETZ in Class IV wetlands may also determined by a number of factors unique to Class IV wetlands. Arndt and Richardson (1988) found reduced clay translocation in semi-permanent ponds, greater mechanical sorting of the shoreline due to increased wave action, and negligible sediment inputs because of the low catchment size to basin size ratio. They also observed poor horizon development in the wet meadow and shallow marsh zones because of the high water table. These soil characteristics may explain why soluble salts in Class IV wetlands are more likely to rise and fall within the soil profile depending on the time of the year and surface moisture. This would also help explain why salts form a continuous band around Class IV wetlands as opposed to the very abrupt ring of discharge carbonate soils around seasonal wetlands in this study.

Understanding near-shore environments is an important step toward understanding the role of hydrologic margins in facilitating biogeochemical transformations and transport of nutrient elements (Barrett et al. 2007). Gooseff et al. (2007) tested a hypothesis that the wicking of water from aquatic environments into lateral margins is controlled by capillary action determined by the physical properties of near-shore sediments, including particle size distribution and slope in arctic lakes. As in this study they found that hydrological margins extended farther in locations where surrounding slopes were gradual. Particle size distribution was poorly linked to lateral flow in their study, whereas slope explained 73% of the variance in the dimensions of the hydrologic zone. They felt that lake levels also influenced the hydrologic exchange between aquatic and terrestrial domains. They are optimistic that with more research the relationship between slope and water levels may be used as an important predictive tool for these hydrologic zones using remote sensing products.

3.6 Conclusion

The spatial distribution of vegetative communities in wetlands is relatively static, particularly those communities positioned in the outer margins of a basin. Consequently, they can be valuable for verifying wetland hydrological models. Jolly et al. (2008) state there is a clear need to develop hydrological models for the movement of salt to, from, and within wetlands so that temporal predictions of wetland salinity can be used to assess ecosystem health. We would also argue that by improving our modeling capabilities, we also improve our knowledge of the influence of snowmelt and rainfall on wetland hydrology and improve our understanding on the connectivity between basins. Many models apply fixed volume equations to predict the average volume of wetlands rather than taking into account the true basin topography or existing spring water levels. It is difficult to verify these models since most wetlands do not possess a spigot through which we can measure outflow. This study demonstrates the importance of using existing spring water depths and true bottom bathymetry in models. Long-term water level data at St. Denis demonstrated that many of these wetlands enter the spring holding water in their basins. Using the true bottom bathymetry of the ponds in the model allowed a comparison of different snowmelt scenarios to existing wetland vegetation in the field.

The WaterLevels model indicated that 30% of the average annual snowmelt equivalent at both the St. Denis and MZTRA sites enters study wetlands. This amount of snowmelt resulted in a good match between spring pond depths and the location of the vegetation communities within each wetland, regardless of basin size, wetland class, or surrounding land-use. The model also indicated that four of the nine wetlands in this study have the potential to spill over into adjacent landscapes. Very few authors discuss the role of snowmelt in their models even though snowmelt runoff is considered to be the most important hydrological event in prairie wetlands. With GCMs predicting snowmelt decreases by the end of this century there is a real scientific imperative to better understand snowmelt runoff in the prairies recognizing that water is the main driver of flora and fauna in prairie wetlands.

Very few researchers have documented the extent of carbonate discharge soils around seasonal wetlands. More emphasis has been placed on studying the extent of salinity around semi-permanent or permanent wetlands because of its affect on agricultural crops. In fact, the soil forming processes that create the Rego Dark Brown Chernozems used to verify the location of carbonate discharge in this study are poorly described in the Canadian System of Soil Classification (Soil Classification Working Group 1998). Part of this relates to the fact that much of the soil surveying completed in Prairie Canada has focused on upland soils for agricultural purposes. Soil surveying has often ended where wetland soils begin. The results in this study indicate that lateral soil water movement and the ETZ is important around both seasonal and semi-permanent wetlands. The type of solute found in adjacent soils differs depending on the water chemistry of the pond, its hydrological permanence, and the mineralogy of the soils. Certain factors, such as the argillic horizon described by Knuteson et al. (1989) may play an important role in the way water distributes itself laterally around seasonal wetlands. What was evident from this work is the extent of the ETZ is somehow connected to slope gradients around the ponds and further work is needed to better understand how wetland and landscape features influence the pattern we observed.

Two observations became apparent during this study. First, access to long-term wetland data is crucial in verifying hydrological models. At least 82 publications have been produced from research at St. Denis. Much of this research has focused on wetlands, soils, hydrology and the wildlife associated with them. It would have been difficult to predict or interpret snowmelt runoff or to verify the accuracy of the WaterLevels model without having these former studies to refer to or having access to the long-term water level data. Secondly, it is crucial that future models are tested on wetlands in other regions. Part of what makes this challenging is the lack of long-term wetland data in other parts of the prairies. By testing models across regions we gain better insight into common factors influencing all wetlands and improve our wetland predictive capabilities to support resolution of key issues such as climate change, greenhouse gas emissions, carbon sequestration, wetland drainage and overall landscape health.

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4.0 OVERALL SYNTHESIS AND STUDY RECOMMENDATIONS

The overall intent of this thesis was to improve our understanding of prairie wetland ecosystems in agricultural landscapes, with a focus on the effect of surrounding land-use on wetland vegetative communities and snowmelt hydrology. Understanding the influence of surrounding land-use not only aids us in taking action to minimize detrimental effects, it also improves our predictive capability of what may happen to prairie wetlands in the coming decades if agricultural practices remain unchanged or as climate change occurs. Several stressors are imposed on wetlands by surrounding land-use (Galatowitsch et al. 2000). These include nutrient and pollutant additions, altered hydrology, excess sediment, and physical destruction of the vegetation and soil. Land-use impacts may appear to be mild when only small portions of the Prairie Pothole Region (PPR) are assessed. However, cumulatively the effect of wetland degradation may be much greater than we think. Denny (1994) states the extinction of species in wetlands is higher than for any other ecosystem.

An estimated 127 million ha of wetlands exist in Canada (Environment Canada 1991). Although the lack of a national wetland inventory makes it difficult to estimate wetland loss since the time of settlement (Watmough and Schmoll 2007), it is estimated that 20 million ha have been lost since the 1800's (Environment Canada 1991). Most prairie wetlands that have escaped drainage in the PPR now lie in watersheds devoted primarily to agricultural crop production (Kantrud et al. 1989). Therefore, almost all PPR wetlands exist in agricultural landscapes that are in private ownership (Rickerl et al. 2000). Human activities have affected nearly every wetland directly or indirectly as a result (Kantrud et al. 1989; Watmough and Schmoll 2007).

The main objective of the vegetation component of this study was to improve our predictive capability for wetland restoration in prairie Canada and to provide recommendations for the future protection of wetlands in agricultural landscapes. This was accomplished by assessing the influence of surrounding land-use on the distribution and diversity of vegetation in 21 prairie wetlands in Manitoba and Saskatchewan. No vegetative study we know of in prairie Canada has included wetlands in cultivated lands as part of its research. What became evident during this study is that the outermost vegetative zones of prairie wetlands are at great risk of being impacted and lost because of agriculture activities. In fact, almost all wetlands on private lands were missing either their wet meadow or low prairie zone. This is a concern since our results indicate these outer zones support the greatest numbers of native species and discourage the establishment and advancement of invasive and weed species.

Studies that have followed plant succession in restored wetlands have consistently found the restoration of native plant species in outer vegetation zones extremely difficult (Galatowitsch et al. 1999). Our research provides insight into why this may be. Wet meadow and low prairie zones start with a deficit of native plant species before restoration is even attempted. Fewer native species in these zones must then out-compete invasive species once restoration takes place. Agricultural impacts that have occurred over decades of farming may make it even more difficult for native species to succeed. Research indicates that impacts such as excess sediment and nitrogen additions from soil erosion along with changes in soil moisture and soil compaction in agricultural landscapes can favor the growth of invasive and weed plant species.

What is evident from this research is that the protection and retention of wet meadow and low prairie zones in prairie wetlands is crucial for both the viability of native species in prairie landscapes and as a deterrent against the spread of invasive species. Any conservation program that focuses on protecting wetlands in agricultural landscapes must consider protecting all vegetative zones of the wetland. Setbacks or buffer zones are often placed around protected wetlands to buffer them against human disturbances. While many papers encourage both private landowners and policymakers to consider preserving wetlands and adjacent land in cropped fields, many do not go so far as to provide guidance on how to implement this practice (Rickerl et al. 2000). For those that do provide guidance the suggested buffer widths almost always focuses on improving water quality (Lowrance et al. 2002) not vegetative quality. As a consequence, the outer vegetation zones of ponds become those locations where sediments settle out first.

One challenge with assigning specific buffer widths is that we assume one width suits all purposes. It also generally disregards the vegetative zonation that occurs around basins and the influence these zones have in maintaining vegetative integrity. An additional problem is that many buffer widths are recommended based on minimal scientific evidence (Lowrance et al. 2002). For example, the Saskatchewan Wetland Conservation Corporation (SWCC) recommends a 10-m buffer width for vegetation surrounding a wetland in cropland landscapes. We found the average vegetative buffer width around cropland wetlands in this study to be 19.2-m for Class III wetlands and 18.2-m for Class IV wetlands. These widths were clearly larger than the 10-m specified by SWCC, but not nearly wide enough to stop the negative impacts we observed. Most buffer research to date has been limited to only a few sites in the United States, with data from those sites being extrapolated to much of the country (Lowrance et al. 2002). With most buffer research focusing on the water quality guidelines established by federal and provincial governments, there is now a real need to expand our focus beyond this one measure and recognize the importance of ecosystem health, rather than only waterway health.

Field studies are needed that test various buffer widths composed of different plant communities for their efficacy in trapping water and sediment runoff, reducing nonpoint-source pollutants while still enhancing the aquatic and vegetative ecosystem. Comparisons need to be made between wetlands protected by surrounding upland buffer strips to wetlands whose buffers only extend up to, and including, the low prairie zone. Our study also suggests that focus needs to be placed on those factors that pose threats to riparian vegetation in agricultural landscapes. These include soil translocation via tillage erosion, the deposition of excess nitrogen with sediments, soil integrity (i.e. soil structure and compaction), soil organic content, and subsurface hydrology. Studies also need to consider and address the range of soils, climates and landscapes that exist across the PPR. It is important to keep in mind that without protecting both the wetland and essential portions of the upland we are not providing full protection to the wetland itself.

One question that arose during this study was if invasive and weed species provide the same biological function as native species in prairie wetlands, should we be concerned? While there is concern about the general loss of biodiversity worldwide with the loss of native species, what is often overlooked is the potential impact invasive and weed species have on agriculture. Farmers have been fighting weeds since the very beginnings of agriculture (McNeely 2006). It is estimated that weeds alone cause an overall reduction of 12% in crop yields in the United States, representing a $32 billion loss in crop production annually (Pimentel et al. 2000). Pimentel (1997) goes on to estimate that for every 1% decrease in crop yield there is on average a 4.5% increase in crop price value at the farm gate. This results in an increase in the price of food for the consumer. Asking producers to shoulder the cost of wetland protection is not appropriate. Cropland hectares can be reduced by 20% in agricultural landscapes where wetlands are buffered (Rickerl et al. 2000). So, while the producer may decrease their costs for weed control in healthier landscapes, they also carry the burden of lost hectares in crop production when buffers are in place on their lands.

It is clear that the adoption of wetland buffers will be limited without added economic incentives (Rickerl et al. 2000). The United States has provided the Wetland Reserve Program (WRP) and the Conservation Reserve Program (CRP) as financial incentives for setting aside and protecting wetlands in the U.S. portion of the PPR. The implementation of these two programs has resulted in the restoration of approximately 2,200,000 ha (5,436,200 acres) of wetland and grassland habitats in the PPR since 1989. While two thirds of the PPR exists in Canada, wetland protection incentives across the Canadian portion of the region are piecemeal at best. For example, Manitoba recently announced their Wetland Restoration Incentive Program (WRIP). The program works with private landowners to restore and protect wetlands on their lands in perpetuity. It is estimated that over the next 3 years the program will restore and protect 1012 ha (2,500 acres) in Manitoba. While the Manitoba government is to be commended for their efforts in protecting wetland habitat, it demonstrates how difficult it is to have any real landscape impact if efforts are not coordinated at all levels of government. With no coordinated effort wetlands will continue to be lost or degraded steadily over the next 60 years as they have been over the last 60 years (Watmough and Schmoll 2007). There is absolutely no indication that agricultural practices will change in the PPR if governments at all levels in Canada do not work together to reverse the ongoing degradation of wetland habitats.

A second objective of this thesis was to examine the accuracy and predictions of a single-basin hydrologic model designed to simulate spring snowmelt events into wetlands in native and agricultural landscapes in Manitoba and Saskatchewan. Snowmelt runoff is recognized as the single biggest hydrological event of the year for most prairie wetlands (Zhao and Gray 1999), yet very little consideration has been given to the influence of snowmelt in wetland models. For the few wetland models that exist none have been tested outside the data sets for which they were parameterized (Johnson et al. 2004). Spring snowmelt is generally the most important input into watershed-level hydrological models for the prairies. With millions of small wetlands still existing in the PPR, the divide between what we know about snowmelt events at the basin level and what we are trying to achieve at the watershed level is very apparent. One also wonders what variables are being used in watershed models to test whether the wetlands within these larger landscapes are being modeled correctly. Our results indicate that 30% of the average annual snowmelt equivalent that falls from December to April enters our wetlands in Manitoba and Saskatchewan. This scenario was consistent regardless of basin size, wetland class, regional location or surrounding land-use. While research suggests that snowmelt and snow catch should be different for wetlands in native versus cropland landscapes, we found no evidence to support that. Our snowmelt findings also relate nicely to the level of snow loss predicted by sublimation each year, which is estimated to be between 40 to 70% (Pomeroy et al. 1993; Burford and Stewart 1998).

Most wetland protection programs in the PPR of the United States depend on our ability to prove that surface connectivity between wetlands exists. Surface connectivity is also an important component of watershed modeling efforts. More research is needed before we understand the connectivity of wetlands in the PPR. This is particularly important in areas where high densities of wetlands exist but where nearby stream networks are minimal. Currently, most water modeling in the PPR is linked to simulations for wetlands being drained into stream networks. Unfortunately this scenario only represents one portion of wetlands in the PPR. Future research efforts must also focus on verifying wetland hydrological models within the PPR beyond the individual locations for which they were parameterized. These models also need to study the potential connectivity of these basins and the influence this has spring flooding events, on wetland water chemistry, and on the abundance and distribution of organisms.

One key mechanism of soil-water movement examined in this study involved the lateral movement of water through the effective transmission zone (ETZ) around wetlands and into the surrounding upland soils. Little is known about the landscape or wetland factors influencing the extent of the EZT around ponds and aquatic systems (Gooseff et al. 2007). A growing body of literature illustrates the importance of hydrologic margins and terrestrial–aquatic connectivity as a major control over biogeochemical cycling aquatic systems (Barrett et al. 2007). Just as surface water is important in influencing the types of vegetation present in a pond, so too is this lateral movement of soil water for plants growing in the wetland fringes. Our study found that solute differences observed in the soils surrounding Class III and IV wetlands were the result of two distinct ionic dominance patterns in a pond's water chemistry. Calcium, bicarbonate and calcite precipitate reflect the influence of snowmelt chemistry in Class III wetlands, while gypsum and salt deposits in soils surrounding Class IV wetlands reflect the influence of the evaporative pathway described by Hardie and Eugster (1970) and further detailed by Arndt and Richardson (1989). Our results also showed that the outer extent of the ETZ around each wetland was located at a specific and consistent upland elevation relative to the basin itself regardless of basin Class. Few researchers have examined this aspect in wetlands. We would suggest that the relationship between slope and the average spring water levels of wetlands requires further investigation.

Two national issues became apparent during this study. The first is Canada's lack of a completed national wetland inventory. While efforts have been made by provincial and federal agencies to rectify this, only New Brunswick and Prince Edward Island have completed inventories. This makes it extremely difficult to put research findings into perspective at either the regional or national level. A second issue is the lack of long-term wetland research sites in Canada. The St. Denis National Wildlife Management Area in Saskatchewan is the only site we know off in the country. Long-term research sites have the capacity to provide insight into the similarities and differences of wetlands across the PPR along with important information on their hydrological cycles over many years. Single-basin hydrological models, such as the WaterLevels model tested in this study, could not have been verified without the data available from St. Denis. The absence of research sites makes it difficult to test and verify the accuracy of wetland models, whether these focus on hydrology, biodiversity, or carbon sequestration.

In addition to the issues mentioned above, wetland soils remain poorly understood in Canada and are poorly reflected in the Canadian System of Soil Classification (Soil Classification Working Group 1998). Part of this relates to the fact that soil surveying in prairie Canada has mainly focused on upland soils for agricultural purposes. Our ability to predict which soils types can be found in different Classes of prairie wetlands is lacking. While many of the features unique to prairie wetlands can be quickly lost through the physical destruction of surface vegetation or wetland drainage, wetland soils are much more resistant to change. This means that soils can act as good indicators of the location of wetlands in disturbed landscapes, particularly agricultural landscapes. Future wetland field studies that include descriptions of soil types found in various classes of wetlands would greatly advance our knowledge in this area.

Lastly, this study was part of a larger study funded by Ducks Unlimited Canada and Agriculture and Agri-Food Canada under the Advancing Canadian Agriculture and Agri-Food (ACAAF) Program. The primary goal of this larger project was to enhance the sustainability of the agriculture industry by addressing gaps in our understanding of the environmental role and economic value of wetlands and riparian zones in agricultural landscapes with respect to green house gas (GHG) fluctuations and carbon sequestration potential. Understanding the interactions between agricultural fields and other landscape components such as wetlands is crucial for comprehensive, whole-landscape accounting of soil organic carbon (SOC) (Bedard-Haughn et al. 2006) and GHG emissions. Landform, land-use, and hydrology have been shown to have either a direct or indirect influence on carbon sequestration and emissions (Yates 2006*a*). The potential for SOC gain under altered management may correspond to landform position-specific differences in cultivation-induced soil changes, such as the translocation of SOC from upperslope areas to footslope or depressional areas via tillage erosion (Pennock 2005). Additional losses of SOC can occur with the decomposition of newly exposed SOC due to the cultivation of the soil (Bedard-Haughn et al. 2006). Nitrous oxide (N_2O) emissions in hummocky landscapes appear to be triggered by precipitation events and the recession of water from wetlands following snowmelt (Yates 2006*a*). This ultimately influences the amount of water-filled pore space in the soil (Corre et al. 1996; Corre et al. 1999; Pennock 2005). N_2O emissions in cultivated versus uncultivated wetlands have been shown to occur over longer periods of time in the spring because the soils in these wetlands remain wetter for longer (Yates 2006*b*). Nitrogen fertilization during spring pulse of activity and high-rainfall summer months have also been shown to contribute to annual N_2O emissions (Corre et al. 1996).

While the findings of the larger study are still pending, it appears that many of the direct and indirect factors influencing SOC sequestration and GHG emissions in and around wetlands in agricultural landscapes also influence the overall productivity of prairie wetlands and the health of their plant communities. The WaterLevels model in this study demonstrated the indirect influence and importance of snowmelt on a series of wetlands that varied in size, location, surrounding land-use and catchment area. Snowmelt runoff in most years determines the depth of water held in a wetland and the length of time it will be held. N_2O emissions, in turn, may be determined to a certain degree by the plant communities that respond to this flooding regime and the influence these communities have on soil moisture. Direct factors we find influencing GHG emissions and SOC sequestration in these landscapes are the same factors we suggest impact wetland plant communities in this study. These factors include soil/tillage erosion, changes in SOC, soil nitrogen and soil-water availability. Therefore, agricultural practices that benefit SOC sequestration and discourage GHG emissions also benefit the long-term hydrological and plant health of prairie wetlands. Therefore, these practices need to be encouraged whenever possible.

While the overall intent of this thesis was to improve our understanding of prairie wetland ecosystems within agricultural landscapes, none of the work in either chapter could have been accomplished without recognizing and understanding the role wetland vegetation plays in acknowledging and interpreting long-term environmental trends. Typical wetland plant species do not occur randomly mixed together in prairie wetlands. Each species has its preferred habitat, often occurring in rough zones on flooding gradients within the basin (Mitsch and Gosselink 2000). Our current understanding of these zones plays a crucial role in studies such as this one, whether we are attempting to decipher the changes that occur in wetland plant communities surrounded by agriculture or trying to improve our understanding of wetland snowmelt hydrology across decades. Wetland plants, in many instances, will remain our only option for verifying the extent of anthropogenic impacts. This will be particularly important for research on the impacts of surrounding land-use, for verifying wetland hydrological models, and it will be particularly important for research on the long-term impacts of climate change.

In conclusion, Allan et al. (2005) state the threats to freshwater systems worldwide include nutrient enrichment, hydrological modifications, habitat loss and degradation, pollution, and the spread of invasive species. What is evident from this study is almost all of the wetlands that remain in prairie Canada experience each one of these threats. A changing climate now poses additional risks upon these existing threats (Allan et al. 2005). The luxury of not acting to minimize or reverse these threats to wetlands is no longer an option.

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5.0 APPENDICES

5.1 Appendix A - List of Plant Species Surveyed

Scientific Name	Authority	Common Name	Origin	Life History	Growth Form	Indicator Status (USA)[1]	Zone Designation[2]	Invasive	Weed
Achillea millefolium	L.	Common yarrow	native	perennial	forb	FACU	UPLA	N	Y
Agropyron smithii	Rydb.	Western wheat grass	native	perennial	graminoid	FACU	LP	N	N
Agropyron trachycaulum	(Link) Malte	Slender wheat grass	native	perennial	graminoid	FACU	LP	N	N
Amelanchier alnifolia	Nutt. ex M. Roemer	Saskatoon serviceberry	native	perennial	shrub	FACU	UPLA	N	N
Anemone canadensis	L.	Canadian anemone	native	perennial	forb	FACW	LP	N	N
Artemisia ludoviciana	Nutt.	Prairie sage	native	perennial	forb	FACU	UPLA	N	N
Antennaria neglecta	Greene	Pussy toes	native	perennial	forb	NI	UPLA	N	N
Aster ericoides	L.	Many flowered aster	native	perennial	forb	FAC	LP	N	N
Aster simplex	Willd.	Small blue aster	native	perennial	forb	FACW	WM	N	N
Avena fatua	L.	Oats - wild	exotic	annual	forb	UPLA	UPLA	Y	Y
Avena sativa	L.	Oats - crop	exotic	annual	graminoid	UPLA	UPLA	N	N
Beckmannia syzigachne	(Steud.) Fernald	Slough grass	native	annual	graminoid	OBL	SM	N	N
Bidens cernua	L.	Nodding beggartick	native	annual	forb	OBL	UPLA	Y	Y
Brassica kaber	(DC.) L.C. Wheeler	Wild mustard	exotic	annual	forb	UPLA	UPLA	Y	Y
Bromus inermis	Leyss.	Smooth brome	exotic	perennial	graminoid	FACU	UPLA	Y	Y
Calamagrostis inexpansa	A. Gray	Northern reed grass	native	perennial	graminoid	FACW	WM		
Caltha palustris	L.	Yellow marsh marigold	native	perennial	forb	OBL	DM	N	N
Campanula rotundifolia	L.	Bluebell bellflower	native	perennial	forb	FAC	LP	N	N
Carduus nutans	L.	Nodding plumeless thistle	exotic	biennial	forb	FAC	LP	Y	Y
Carex aquatilis	Wahl.	Water sedge	native	perennial	graminoid	OBL	SM	N	N

Scientific Name	Authority	Common Name	Origin	Life History	Growth Form	Indicator Status (USA)	Zone Designation	Invasive	Weed
Carex atherodes	Spreng.	Awned sedge	native	perennial	graminoid	OBL	SM	N	N
Ceratophyllum demersum	L.	Coonstail	native	perennial	forb	OBL	DM	N	N
Chenopodium rubrum	L.	Red goosefoot	native	annual	forb	OBL	SM	N	N
Chenopodium album	L.	Lamb's quarters	exotic	annual	forb	FAC	LP	Y	Y
Cicuta maculata	L.	Water hemlock	native	biennial/perennial	forb	OBL	SM / fen	N	Y
Cirsium arvense	(L.) Scop.	Canada thistle	exotic	perennial	forb	FACU	WM	Y	Y
Descurainia sophia	(L.) Webb. Ex Prantl	Flixweed	exotic	annual/biennial	forb	UPLA	UPLA	Y	Y
Distichlis stricta	(L.) Greene	Saltgrass	native	perennial	grass	FACW	WM	N	N
Festuca ovina	L.	Sheep fescue	exotic	perennial	graminoid	UPLA	UPLA	N	N
Elaeagnus commutata	Bernh. ex Rydb.	Silverberry	native	perennial	shrub	FAC	LP	N	N
Eleocharis sp.	R. Br.	Spikerush	native	perennial	graminoid	OBL	WM	N	N
Equisetum arvense	L.	Field horsetail	native	perennial	forb	FAC	WM	Y	Y
Galium boreale	L.	Northern bedstraw	native	perennial	forb	FACU	LP	N	N
Glyceria grandis	S. Wats.	Manna grass	native	perennial	graminoid	OBL	SM	N	N
Grindelia squarrosa	(Pursh) Dun.	Curlycup gumweed	native	perennial	forb	UPLA	UPLA	Y	N
Helianthus sp.	L.	Sunflower	native	perennial	forb	FACU	LP	Y	N
Hordeum jubatum	L.	Foxtail barley	native	perennial	graminoid	FACW	WM	Y	Y
Juncus arcticus	Willd. ssp. *littoralis*	Baltic rush	native	perennial	graminoid	OBL	WM	Y	N
Lactuca tatarica var. pulchella	*Lactuca tatarica* (L.) C.A. Mey. var. *pulchella* (Pursh) Breitung	Common blue lettuce	native	perennial	forb	FACU	WM	N	N
Lemna minor	L.	Lesser duckweed	native	perennial	forb	OBL	DM	N	N
Lemna triscula	L.	Ivy-leaved duckweed	native	perennial	forb	OBL	DM	N	N
Liatris pycnostachya	Michx.	Prairie blazing star	native	perennial	forb	FAC	NI	N	N
Linum usitatissimum	L.	Common flax	exotic	annual	forb	UPLA	UPLA	N	N

Scientific Name	Authority	Common Name	Origin	Life History	Growth Form	Indicator Status (USA)	Zone Designation	Invasive	Weed
Lycopus asper	Greene	Western water horehound	native	perennial	forb	OBL	WM	N	N
Maianthemum trifolium	(L.) Sloboda	Three-leaved Solomon's-seal	native	perennial	forb	OBL	NI	N	N
Melilotus alba	Medic.	White sweet clover	exotic	annual/biennial	forb	FACU	NI	Y	Y
Mentha arvensis	L. var. *villosa* (Benth.)	Field mint	native	perennial	forb	FACW	WM	N	N
Pisum sativum	L.	Peas	exotic	annual	forb	UPLA	NI	N	N
Phalaris arundinacea	L.	Reed canary grass	native	perennial	graminoid	FACW	WM	Y	N
Plantago major	L.	Common plaintain	exotic	annual	forb	FAC	WM	Y	Y
Poa palustris	L.	Fowl bluegrass	native	perennial	graminoid	FACW	WM	N	N
Poa pratensis	L.	Kentucky blue grass	exotic	perennial	graminoid	FACU	LP	Y	N
Polygonum amphibium	L. var. *stipulaceum* Coleman	Water smartweed	native	perennial	forb	OBL	SM	Y	N
Polygonum convolvulus	L.	Wild buckwheat	exotic	annual	forb	FAC	LP	Y	Y
Polygonum lapathifolium	L.	Curlytop knotweed	native	annual	forb	OBL	WM	Y	Y
Populus tremuloides	Michx.	Quaking aspen	native	perennial	tree	FACU	NI	N	N
Potentilla anserina	L.	Silverweed	native	perennial	forb	OBL	WM	N	N
Potentilla norvegica	L.	Norwegian cinquefoil	exotic	annual/biennial	forb	FAC	WM	Y	Y
Stuckenia pectinata	L.	Sago pondweed	native	perennial	forb	OBL	DM	N	N
Potamogeton pusillus	L.	Small pondweed	native	perennial	forb	OBL	DM	N	N
Primula sp.	L.	Primrose	native	perennial	forb	UPLA	NI	N	N
Ranunculus cymbalaria	Pursh	Seaside buttercup	native	perennial	forb	OBL	SM	N	N
Ranunculus gmelinii	DC.	Gmelin's buttercup	native	perennial	forb	FACW	WM	N	N
Ranunculus macounii	Britt.	Macoun's buttercup	native	perennial	forb	OBL	WM	N	N
Ribes sp.	L.	Currant	native	perennial	shrub	UPLA	NI	Y	Y
Rosa acicularis	Lindl.	Prickly rose	native	perennial	shrub	FACU	LP	N	N
Rumex crispus	L.	Curled dock	exotic	perennial	forb	FACW	WM	Y	Y

Scientific Name	Authority	Common Name	Origin	Life History	Growth Form	Indicator Status (USA)	Zone Designation	Invasive	Weed
Salix sp.		Willow	native	perennial	shrub	FACW	NI	N	N
Scirpus validus	Vahl.	Softstem bulrush	native	perennial	graminoid	OBL	DM	N	N
Scolochloa festucacea	(Willd.) Link	Whitetop	native	perennial	graminoid	OBL	SM	N	N
Schoenoplectus maritimus	L. (*Scirpus maritimua*)	Alkali bulrush	native	perennial	graminoid	OBL	SM	N	N
Solidago canadensis	L.	Canada goldenrod	native	perennial	forb	FACU	LP	Y	Y
Sonchus arvensis	L.	Perennial sow thistle	exotic	perennial	forb	FAC	WM	Y	Y
Sparganium eurycarpum	Engelm.	Broadfruit bur-reed	native	perennial	forb	OBL	SM	N	N
Spartina gracilis	Trin.	Alkali cordgrass	native	perrenial	grass	FACW	WM	N	N
Stachys palustris	L.	Marsh hedgenettle	native	perennial	forb	OBL	WM	Y	Y
Stipa comata	Trin. & Rupr.	Needle-and-thread grass	native	perennial	graminoid	UPLA	NI	N	N
Symphoricarpos albus	(L.) Blake	Common snowberry	native	perennial	shrub	FACU	LP	N	N
Symphoricarpos occidentalis	Hook. (weed in MB only)	Buckbrush	native	perennial	shrub	NI	LP	N	N
Taraxacum officinale	F.H. Wigg.	Common dandelion	exotic	perennial	forb	FACU	LP	Y	Y
Thlaspi arvense	L.	Stinkweed	exotic	annual	forb	FACU	LP	Y	Y
Triglochin maritima	L. (weed in MB only)	Sea-side arrowgrass	native	perennial	graminoid	OBL	WM	N	N
Triticum spp.	L.	Wheat	exotic	annual	graminoid	UPLA	NI	N	N
Typha glauca	Godr. (pro sp.) [*angustifolia* or domingensis × *latifolia*]	Hybrid cattail	native	perennial	forb	OBL	DM	Y	N
Typha latifolia	L.	Broadleaf cattail	native	perennial	forb	OBL	DM	N	N
Urtica dioica	L. (weed in MB only)	Stinging nettle	exotic	perennial	forb	FACW	NI	N	Y
Zizia aptera	(Gray) Fern.	Heart-leaved alexander	native	perennial	forb	FACW	LP	N	N

[1] United States wetland indicator status: OBL = obligate wetland, FACW = facultative wetland, FAC = facultative, FACU = facultative upland, UPLA = upland.

[2] Wetland vegetation zones: DM = deep marsh, SM = shallow marsh, WM = wet meadow, LP = low prairie NI = not indicated.

5.2 Appendix B - Fortran Program for WaterLevels Model

```
! Program WaterLevels
        ! E.N. van den Bos and D. Pennock, written in October 2005
        ! The purpose of program WaterLevels is to calculate water level after a
        ! rainevent based on the (measured) waterlevel before the rain event, the amount of rainfall and
        ! the duration of the rain event. The infiltrationcapacity/field saturated conductivity should
        ! be defined before running this program.
        ! Revised February 03 2006 to use a single user-specified Kfs value (Pennock)
        ! Revised February 13 2006 to correct errors in calculation of spillover volume and volume
        ! of water in pond after a spillover event
        !Declaration Statements
        DIMENSION X(100000), Y(100000), Z(100000), PONDB(100000), PONDA(100000),
VOLPCELLB(100000), OZ(100000), DEPTH (100000)
        DIMENSION RCELL(100000), VOLCELLA(100000), Kfs(100000), VOLPACELL(10000),
POTVOL (100000),SPILLVOL(100000)
        REAL X, Y, Z, Kfs, GRIDSIZE, GRIDAREA, H, P, D, SPILLZ, PONDBCNT, PONDAREAB,
CATCH, CATCHAREA
        REAL VOLB, SUMR, VOLPOND, VOLUME, ZMAX, SUMVOLCELLA, SPILLDIFF,
SPILLVOL, PONDACNT, PONDAAREA
        REAL DELTAH, SPILLVOLSUBCATCH, Kfsvalue, FVOLUME, POTVOL,
TOTPOTVOL,WMAX, DEPTH
        INTEGER NROW, NCOL, TOTNUM, SPILL
        ! Variable Definitions
        CHARACTER*30 FILNM,COMFILE,OUTFILE
        CHARACTER    ANSWER*1 ! HEADERJUNK*135

        ! DEFINITION OF VARIABLES
        ! INTERACTIVE INPUT
        WRITE(*,*)'************************* February 2006 *************'
  WRITE(*,*)'*  PROGRAM => WaterLevels                  *'
  WRITE(*,*)'*            E.N. van den Bos and D. Pennock *'
  WRITE(*,*)'*            Dept. of Soil Science         *'
  WRITE(*,*)'*            University of Saskatchewan    *'
  WRITE(*,*)'******************************************************'
9191 CONTINUE
        WRITE (*,*) 'ENTER NAME OF Basin FILE (created in Surfer)'
        WRITE (*,*) 'INCLUDING THE SUFFIX (e.g.Pond117.dat)'
        READ(*,*) FILNM
        COMFILE=FILNM

        WRITE (*,*)'INPUT A NAME TO BE USED FOR Waterlevel OUTPUT FILES?'
        WRITE (*,*)'  (file name no longer than 23 characters)'
        WRITE (*,*)'(ie. testsite; name appended with OUT.DAT)'
        READ (*,*)FILNM

        !       ** CHECKING THE STRING LENGTH OF INPUT FILE NAME **
        ! IN ORDER TO CREATE THE NEW OUTPUT FILES
                CHECK2=SCAN(FILNM,' ')
                OUTFILE=FILNM(1:CHECK2-1)//'OUT.DAT'
        WRITE (*,*)'*************** Waterlevel CALCULATES THE FOLLOWING
****************'
        WRITE (*,*)'ARRAY OUPUT FILE =>',OUTFILE
        WRITE (*,*)'*****************************************************************'
```

```
        WRITE (*,*) 'ENTER THE NUMBER OF ROWS'
        READ(*,*) NROW

        WRITE (*,*) 'ENTER THE NUMBER OF COLUMNS'
        READ(*,*) NCOL
          TOTNUM = NCOL*NROW

        WRITE (*,*) 'Enter grid spacing'
        READ (*,*) GRIDSIZE
          GRIDAREA = GRIDSIZE*GRIDSIZE

        WRITE (*,*) 'Enter the (measured) waterlevel of the pond before the rain event'
        Write (*,*) 'If pond is dry enter minimum elevation in pond'
        READ (*,*) H

        WRITE (*,*) 'Enter the amount of rain fall in mm'
        READ (*,*) P

        WRITE (*,*) 'Enter the duration of the rain event in hours'
        READ (*,*) D

        WRITE(*,*) 'Enter spillover elevation'
        READ(*,*) SPILLZ

        WRITE(*,*) 'Enter 1 if cultivated or 2 if grassed or 3 if ice seal'
        READ(*,*) Kfsvalue

        WRITE (*,*) 'Enter the total volume in m3 that enters the catchment due to'
        WRITE (*,*) 'spillover of other (sub)catchments'
        READ (*,*) SPILLVOLSUBCATCH

WRITE (*,*)'***** VARIABLE DATA INPUT IS NOW COMPLETE *****'
WRITE (*,*)'   Please review and make any corrections'
WRITE (*,*)'            before proceeding'
WRITE (*,*)'   ** If correct type Y, if not type N **'
READ (*,*) ANSWER
              IF ((ANSWER.EQ.'N').or.(ANSWER.EQ.'n')) THEN
              GOTO 9191
              END IF
!************************* INPUT COMPLETE ****************************

        WRITE (*,*) '*** DATA BEING READ FROM INPUT FILES ***'
        WRITE (*,*) 'Basin FILE =>',COMFILE
        WRITE (*,*) '****************************************'

        ! INPUT *.LFM FILE
        OPEN (55,FILE=COMFILE, STATUS='OLD')
! READ(55,*)HEADERJUNK ! use if there is a headerline
        READ(55,*)(X(I),Y(I),Z(I), I=1, TOTNUM)
        PONDAREA = 0
        VOLB = 0
        CATCHAREA = 0
        ZMAX = 0
        DELTAH = 0
        PONDAREA = 0
        VOLPOND = 0
```

```
SUMR = 0
VOLUME = 0
SPILLVOL = 0
OZMAX = 0
POTVOL=0
TOTPOTVOL=0
SPILL=0
!  Set POND = 0 (non-pond, dry)
DO 100 I=1, TOTNUM
        PONDB(I)=0
        PONDA(I)=0
        DEPTH(I)=0.0
        OZ(I)=Z(I)
        If (Kfsvalue.EQ.1) Then
                Kfs(I)=2.6E-06
        END IF
        If (Kfsvalue.EQ.2) Then
        Kfs(I)=1.4E-05
        END IF
        IF (Kfsvalue.EQ.3) Then
        Kfs(I)=1.0E-09
        END IF
        100     CONTINUE

! Set SURFER Missing Value code to -9999.0
DO 101 I=1, TOTNUM
        IF (Z(I).GT.100000.0)THEN
                        Z(I)=-9999.0
                        PONDB(I)=2.0
                        PONDA(I)=2.0
        END IF
101     CONTINUE
        ! Calculated pond area
PONDBCNT=0
CATCH=0
! Land is 0, water is 1, blanked is 2
DO 102 I=1, TOTNUM
        IF (Z(I).GE.H)THEN
                        PONDB(I)=0.0
        END IF
                IF ((Z(I).LT.H).AND.(Z(I).GT.-9999))THEN
                        PONDB(I)=1.0
                        PONDBCNT=PONDBCNT+1
        END IF
                IF ((PONDB(I).LT.2.0))THEN
                CATCH=CATCH+1
        END IF
        IF ((PONDB(I).LT.2.0).AND.(Z(I).LT.SPILLZ))THEN
                POTVOL(I) = (SPILLZ - Z(I))*GRIDAREA
        END IF
        TOTPOTVOL = TOTPOTVOL + POTVOL(I)
        102 CONTINUE
PONDAREAB=0
CATCHAREA=0
IF (PONDBCNT.GT.0) THEN
PONDAREAB=PONDBCNT*GRIDAREA
```

```
END IF
CATCHAREA=CATCH*GRIDAREA
! Calculate the volume of the pond before the rain event
VOLB=0.00
DO 104 I=1, TOTNUM
        VOLPCELLB(I)=0.00
104     CONTINUE
    DO 105 I=1, TOTNUM
        IF((H.GT.Z(I)).AND.(PONDB(I).LT.2.0)) THEN
            VOLPCELLB(I)= (H-Z(I))*GRIDAREA
        END IF
        VOLB = VOLB + VOLPCELLB(I)
105 CONTINUE
!Calculate the runoff
DO 106 I=1,TOTNUM
        RCELL(I)=0
106 CONTINUE
SUMR=0
DO 107 I=1,TOTNUM
        IF ((Z(I).GT.H).AND.((P/(D*3600*1000)).GT.Kfs(I))) THEN
            RCELL(I)=((P/(3600*1000*D))-Kfs(I))*GRIDAREA*3600*D
        END IF
SUMR=SUMR+RCELL(I)
107     CONTINUE
! Calculate the volume added to the pond by direct rainfall on the pond
DO 108 I=1,TOTNUM
    VOLPACELL(I)=0
108 CONTINUE
VOLPOND=0
DO 109 I=1, TOTNUM
IF (PONDB(I).EQ.1.0) THEN
    VOLPACELL(I)=(P/1000)*GRIDAREA
    END IF
     VOLPOND=VOLPOND+VOLPACELL(I)
    109 CONTINUE
    ! Calculate the total volume of water in the pond after the rainfall
VOLUME=0
VOLUME=SUMR+VOLPOND+VOLB+SPILLVOLSUBCATCH
    IF (VOLUME.EQ.0.0) THEN
        GOTO 500
    END IF
! Calculate the waterlevel in the pond after the rain event
! Assess maximum Z value
ZMAX=-9999
DO 110 I=1,TOTNUM
    IF(Z(I).GT.ZMAX)THEN
        ZMAX=Z(I)
    END IF
! wmax is max water level
WMAX = ZMAX
110 CONTINUE
IF (VOLUME.GT.TOTPOTVOL)THEN
    TOTSPILLVOL=VOLUME-TOTPOTVOL
    VOLUME=TOTPOTVOL
    WMAX=SPILLZ
END IF
```

```
! Calculate the volume for different elevations of water added to each grid cell.
! If the total volume of water is greater than the amount of water input, then substract
! 1 mm from the elevation of water.
! Start of volume calculation
SUMVOLCELLA=0.00
DO 111 I=1,TOTNUM
        VOLCELLA(I)=0
111 CONTINUE
200 CONTINUE
DO 112 I=1,TOTNUM
        IF((Z(I).LT.WMAX).AND.(PONDA(I).LT.2.0))THEN
                VOLCELLA(I)=(WMAX-Z(I))*GRIDAREA
        END IF
112 CONTINUE
DO 213 I=1, TOTNUM
SUMVOLCELLA=SUMVOLCELLA+VOLCELLA(I)
213 CONTINUE
IF(SUMVOLCELLA.GT.VOLUME)THEN
        WMAX=WMAX-0.001
        SUMVOLCELLA=0
        GOTO 200
END IF
! IF (WMAX.GT.SPILLZ) THEN
!       GOTO 600
!       END IF
DO 214 I=1,TOTNUM
        IF((Z(I).LT.WMAX).AND.(PONDA(I).LT.2.0))THEN
        Z(I)=WMAX
        PONDA(I)=1.0
        PONDACNT=PONDACNT+1
        DEPTH(I)= Z(I)-OZ(I)
        END IF
214 CONTINUE
PONDAAREA=PONDACNT*GRIDAREA
DELTAH=WMAX-H
 GOTO 500
! 600 CONTINUE
! Check to see if WMAX is greater than spillover Z; if so set ZMAX to
! spillover elevation
!SPILLDIFF=0.00
!SPILLVOL=0.00
!PONDACNT=0
!PONDAAREA=0
!SPILL = 0

! fEB 06: sPILLOVER ELEVATION SET TO SPILLOVER - 1 CM
    !       SPILLDIF=WMAX-SPILLZ
    !       OWMAX=WMAX
    !       WMAX=SPILLZ-0.1
    !       SPILL=1
    !DO 114 I=1,TOTNUM
    !       IF((Z(I).LT.WMAX).AND.(PONDA(I).LT.2.0))THEN
    !               Z(I)=WMAX
    !               PONDA(I)=1.0
    !               PONDACNT=PONDACNT+1
    !       END IF
```

```
!114 CONTINUE
!IF(PONDACNT.EQ.0)THEN
!       GOTO 500
!END IF
!PONDAAREA=PONDACNT*GRIDAREA
! calculate spillover volume
!DO 115 I=1,TOTNUM
!       IF (PONDA(I).EQ.1.0)THEN
!               SPILLVOL(I)=(GRIDAREA*SPILLDIFF)
!END IF
!115 CONTINUE
!DO 116 I=1, TOTNUM
!       IF ((PONDA(I).EQ.0.0).AND.(Z(I).LT.OWMAX)) THEN
!       SPILLVOL(I) = ((OWMAX-Z(I))*GRIDAREA)
!       END IF
!116 CONTINUE
!TOTSPILLVOL=0.0
!DO 117 I=1, TOTNUM
!        TOTSPILLVOL=TOTSPILLVOL+SPILLVOL(I)
!117 CONTINUE
! Calculate the difference in waterlevel
!IF (SPILL.EQ.1) THEN
!       DELTAH=WMAX-H
!       END IF
! Calculate the final volume of water in the pond after the rainfall and spillover
500 CONTINUE
FVOLUME=0
FVOLUME=SUMR+VOLPOND+VOLB+SPILLVOLSUBCATCH-TOTSPILLVOL
!****************** ARRAY (ARR) FILE OUTPUT ***********************
        OPEN (6, file=OUTFILE, STATUS='unknown')
        WRITE (*,*) 'OUTPUT FILE =>',OUTFILE
        WRITE (*,*) ' '
        WRITE (*,*) 'Before rain event:'
        WRITE (*,*) 'Spillover elevation=',SPILLZ
        WRITE (*,*) 'Total potential volume before spillover=', TOTPOTVOL
        WRITE (*,*) 'Input water level= ', H
        WRITE (*,*) 'Pond Area before rain event= ', PONDAREAB
        WRITE (*,*) 'Volume of the pond before rain event= ',VOLB
        WRITE (*,*) 'Catchment area=',CATCHAREA
        WRITE (*,*) ' '
        WRITE (*,*) 'After rain event:'
        WRITE (*,*) 'Runoff volume=',SUMR
        WRITE (*,*) 'Volume of water added to the pond by direct rainfall=', VOLPOND
        WRITE (*,*) 'Water level after rain event=',WMAX
        WRITE (*,*) 'Waterlevel rise=',DELTAH
        WRITE (*,*) 'Pond Area after rain event=',PONDAAREA
        WRITE (*,*) 'Volume of the pond after rain event=',FVOLUME
        WRITE (*,*) 'Spillover volume=',TOTSPILLVOL
        WRITE (6, 900)
        WRITE (6, 901) (X(i), Y(i), Z(i), PONDB(i), PONDA(i),DEPTH(I), i=1, TOTNUM)
900  FORMAT (9X, 'X,',13X,'Y,',13X,'Z,',15X,'PONDB,',9X,'PONDA,',9X, 'DEPTH,' 9X)
901  FORMAT (1X,2F13.2,F15.3,F5.0,F5.0,F9.3)
!       WRITE(*,*)'***Press Enter Key to Return To Main Menu***'
        PAUSE
        stop
        END
```