

Effects of Water Level Management on Water Chemistry and Primary  
Production of Boreal Marshes in Northern Manitoba, Canada

by

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A Thesis submitted to the Faculty of Graduate Studies of  
The University of Manitoba  
in partial fulfilment of the requirements of the degree of

MASTER OF SCIENCE

Department of Biological Sciences

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## Abstract

This experiment manipulated water levels in boreal marshes within the Saskatchewan River Delta, a 9500 km<sup>2</sup> region in northern Canada. Water levels in three wetland cells were lowered in a partial drawdown by a mean of 0.32m. Water clarity, nutrient concentrations, and periphyton nutrient limitation were measured over the summer preceding and the summer following manipulation. The water levels of three adjacent control wetlands were not manipulated. Lowering wetland water levels reduced the wind velocity necessary to resuspend bottom sediments, which led to increases in turbidity, dissolved organic carbon, and concentrations of organic and inorganic nitrogen and phosphorus. Prior to drawdown, wetland periphyton communities were limited by nitrogen or co-limited by nitrogen and phosphorus. The input of nutrients from the sediment resulted in a shift from nutrient deficiency to nutrient sufficiency. Periphyton and phytoplankton production increased in response to the nutrient input. Increased turbidity, nutrient concentrations, and algal production were correlated with depth, rather than being inherent to the drawdown condition. Other water level manipulation studies have found that a reflood after a period of total drawdown caused a pulse of nutrients leaching from decomposing litter. This work suggests that these changes may not require complete drying out of sediments, or the input of large amounts of litter from drowned annual mudflat species, but rather can occur when depths are shallow enough that sediments are more frequently resuspended by wind. These findings have implications for future management of these marshes for waterfowl and muskrat production.

## Acknowledgements

I am most grateful to my advisor, Dr Gordon Goldsborough for his wealth of knowledge, his patience, his encouragement, and his faith in me. I thank my committee members, Dr Gordon Robinson and Dr John Markham, for their advice and constructive criticism. Dr Dale Wrubleski was essentially a co-advisor and I am grateful for all his help. I would like to thank Dr Annemieke Farenhorst for lending me lab space on campus, Llewellyn Armstrong for her help with the statistics, Dr Pascal Badiou for working his DOC magic, and former aquatic ecology grad students Tara Bortoluzzi, Elaine Shipley, Scott Kolochuk, and Kasia Dyszy for passing down their valuable expertise.

Special thanks go to research assistant extraordinaire Sheila Atchison, for running the show for three years: I couldn't have done it without her. For other essential field, camp, and lab help, I am indebted to Paul "Baby Legs" Ziesmann, Nola Geard, Jared "Scootaloo" Knockaert, Martin "Moose" Blades, Mark Baschuk, Mike Ervin, John Hopkins, Mokhtar Joundi, Larkin Mosscrop, and Catherine Desrochers.

All the staff of Ducks Unlimited in The Pas contributed boats and time to this project: Robin Reader (with special thanks for his patience, wisdom, sense of local history, and general wackiness), Shaun Greer (thanks for logistics, rescue ops, and acting as town tour guide), Dave Clayton, KJ Dyrda, Dave White, Justin, and Chris Smith. Garth Ball of Manitoba Conservation was integral to the project, with his airboat, willingness to get dirty, mechanical expertise, creative cursing and advice on bush living. For the use of the Hill Island camp and Grace Lake bunkhouse I am grateful all the staff at Manitoba Conservation in The Pas, including Cam Hurst, Dale Cross, and Derek Leask. Particular thanks go to Ron Campbell for his bannock and other much needed supplies. I thank University

College of the North for the loan of a boat and lab equipment. I'm grateful to the staff at Delta Marsh Field Station for accommodating my samples, my assistants, and myself.

I am fortunate to have received input from many other people knowledgeable about the SRD: Dr Bill Clark, Dr Norm Smith, Gary Carriere, Alex Sanderson, Edwin Jebb, and all those members of Opaskwayak Cree Nation, Cormorant First Nation, Moose Lake First Nation, and Easterville who came out to our community consultation meetings to give advice and share their concerns.

Research was supported by generous funding from Manitoba Hydro, Ducks Unlimited Canada, Manitoba Conservation's Heritage Marshes program, and the Kelsey Conservation District. I was able to eat more than just Kraft Dinner thanks to the Canada Graduate Scholarship I received from the Natural Sciences and Engineering Research Council. Travel funding from the Faculty of Graduate Studies, the Faculty of Science, and the Department of Biological Sciences allowed me to present this work to the larger scientific community.

Finally, I would like to thank Dr Brian Parker for providing some much-needed motivation.

Rest in peace, Belafonte.

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# Chapter 1: Introduction

## Background

The ecology of Canada's prairie wetlands has been well described, on scales ranging from short-term descriptive studies (Weller and Fredrickson 1974; van der Valk and Davis 1978a; van der Valk and Davis 1978b) to decade-long interdisciplinary whole ecosystem manipulation projects (Murkin *et al.* 2000). It has been well-established that a fluctuating water regime is the major driver of prairie marsh dynamics. By contrast, wetland ecology in boreal Canada has been poorly studied, and the response of northern marshes to variations in water levels is thus far unknown.

## Boreal Wetlands and Deltaic Marshes

The boreal region encompasses one third of the area of Canada, and of this area, wetlands comprise approximately 20% or over 600,000 km<sup>2</sup> (National Wetlands Working Group 1988). The majority of these are peatlands, but there are significant areas of marshes along the shores of lakes and in the deltas formed as rivers discharge into large lakes. Older portions of deltas, less frequently subjected to river floods, have often developed into fens, bogs and treed swamps (Dirschl 1972b). However, in the active portions of deltas nearer to the developing margin, marshes and shallow open water can be found behind river channel levees. The larger boreal delta marshes include the Slave River Delta, the Peace-Athabasca Delta, and, largest of all, the Saskatchewan River Delta.

## Saskatchewan River Delta

The Saskatchewan River Delta (SRD) is an inland delta in the Mid-Boreal Lowlands ecoregion of the Boreal Plains ecozone. The delta has been forming

since the late Holocene with sediments deposited by the Saskatchewan River as it entered glacial Lake Agassiz and its remnant plain and lakes (Morozova and Smith 2003). The SRD now consists of over 9000 km<sup>2</sup> (National Wetlands Working Group 1888) of wetlands, shallow lakes, and active and abandoned river channels bordered by forested natural levees (Morozova and Smith 2003), in eastern central Saskatchewan and western central Manitoba (Figures 1-1 and 1-2). The larger, older Upper Delta is divided from the younger, more active Lower Delta by the moraine at The Pas, Manitoba. Together, the upper and lower portions of the SRD comprise the largest freshwater inland river delta in North America (Wrubleski 2008).

The SRD is home to 13,000 people (Smith 2008), in communities including The Pas, Opaskwayak Cree Nation, and Cumberland House, or isolated in remote areas. The delta is also home to myriad plants and animals. Fur-bearing mammals, especially beaver and muskrat, but also mink, otter, fisher, and lynx, have been and are an important resource to local trappers (McLeod *et al.* 1947; Uchtmann 2008). Large mammals, including moose, black bear, elk, wolf, and deer, provide tourism revenue from southern hunters, a source of food for local hunters, and opportunities for wildlife viewing (Smith 2008). The SRD has been designated an Important Bird Area nationally (Poston *et al.* 1990) and internationally (Partners FOR the Saskatchewan River Basin 2008). Over 120 species of birds are found in the delta (Smith 2008), including nearly 500,000 ducks (Slattery 2008). Open water areas in the delta provide habitat for 48 species of fish (Rosenberg *et al.* 2005), including commercially important species like walleye and recreationally important species like northern pike. Species important to the bait fishery, such as shiners, and species at risk, including the lake sturgeon, are also represented.

Over the last century, the SRD has been increasingly impacted by upstream and downstream development. Upstream of the delta, hydroelectric projects have combined with increased agricultural irrigation demands on the Saskatchewan

River to dramatically alter its hydrology (Figure 1-3). Nineteen dams have been constructed on the Saskatchewan River or its tributaries (Partners FOR the Saskatchewan River Basin 2008), most notably the EB Campbell, less than 100 km upstream of the SRD, and the Gardiner, on the South Saskatchewan River (Figure 1-4). These dams change downstream patterns of annual flow by retaining water during high flows and releasing it during traditionally low flow periods (Leavens 2008). Annually, 10 to 20% of the Saskatchewan River's naturalised flow is consumed upstream of the SRD, partly to support an area of 5000 km<sup>2</sup> of irrigated agriculture (Partners FOR the Saskatchewan River Basin 2008). The diminished, stabilised river flows have reduced the probability of flooding to permanently separated wetland basins from once in ten years to once in fifty years (Leavens 2008).

Changes in river water quality also have impacts on the SRD. The drainage basin of the Saskatchewan River encompasses 420,000 km<sup>2</sup> of Montana, Alberta, Saskatchewan, and Manitoba and is home to three million people (Partners FOR the Saskatchewan River Basin 2008). Agricultural activity is prevalent in much of this region – as much as 90% of land within the South Saskatchewan River sub-basin is cropland or rangeland (Saskatchewan Watershed Authority 2007), and agricultural runoff can be a non-point source of nutrients and pollutants (Cooke and Prepas 1998). The many towns and several major urban areas, including Calgary, Edmonton, Saskatoon and Lethbridge, through which the river and its tributaries pass (Figure 1-4) may be point sources of pollutants. Non-urban industrial activity within the watershed, including extraction and processing of forestry, mining, and petrochemical resources, may also change water quality in the delta (Partners FOR the Saskatchewan River Basin 2008).

Finally, the SRD has lost substantial wetland area through flooding and drainage. The hydroelectric dam at Grand Rapids, Manitoba downstream of the SRD, permanently flooded more than 1000 km<sup>2</sup> of the lower delta (Uchtmann 1983),

and the Pasquia land reclamation project drained 550 km<sup>2</sup> of wetlands in the Carrot River Valley, west of The Pas, Manitoba, for farmland in the 1950s (Partners FOR the Saskatchewan River Basin 2008). These changes to the SRD have been noted by local trappers and hunters, who have described marked declines in the resources they extract from the SRD, most particularly in muskrats (Uchtmann 2008) and ducks (Slattery 2008).

Despite the size and importance of the SRD and the threats it is facing, the ecological function of the delta has been relatively unstudied. Qualitative information is available, in the form of traditional ecological knowledge from area residents, and descriptions of annual wetland monitoring conducted by Ducks Unlimited Canada, but scientific studies have been limited. Some of the earliest descriptions were written by McLeod (1947), who, very qualitatively, described vegetation and water quality as they related to muskrat production. Dirschl and colleagues (Dirschl and Dabbs 1969; Dirschl 1970; Dirschl 1972b; Dirschl and Coupland 1972) described several vegetation assemblages within wetlands of the upper SRD, and studied vegetation succession pathways. Ducks Unlimited visits many wetlands within the SRD yearly to observe waterbirds and habitat, but data in their reports tend to be qualitative and to vary in content from year to year. More recently, Morozova and Smith (2003) have described the development of the delta by studying its progradation, avulsion and other fluvial sedimentary processes, mainly in the upper delta.

## Objectives

In response to concerns raised by local resources users over the health of the SRD ecosystem and an overall lack of knowledge relating to the ecology of boreal deltaic wetlands, a multi-faceted ecological study was conceived by Ducks Unlimited Canada scientists at the Institute for Wetlands and Waterfowl Research. This study would investigate populations of muskrats, fish, waterfowl and other waterbirds within the SRD, and, because the decline in animal

populations was suspected to be result of habitat deterioration, perhaps because of long-term water level stabilisation, it was also deemed useful to study water quality and primary production in SRD wetlands.

The study proposed to address habitat concerns and these objectives by monitoring the responses of the aforementioned populations to experimental water level manipulation, with the objective of informing future wetland management practices. These water level manipulations were designed to mimic the natural fluctuations of the SRD, which have been disrupted in recent years. The study, modeled on the Marsh Ecology Research Project (Murkin *et al.* 2000), was designed as a drawdown and reflood project, in which wetland water levels would be experimentally lowered, then, after several years, raised.

Within the context of this larger study, this research project specifically strove to achieve three objectives:

- Describe the water quality and vegetation community of the SRD, a largely unstudied northern deltaic wetland.
- Understand the effects of lowering water levels on water quality and algal primary production.
- Suggest strategies to control wetland water quality and primary production by water manipulation that may be useful for managing fish and wildlife communities.

## Hypotheses

- A. Drawdown will affect water quality by increasing water column turbidity and nutrient concentrations because shallower water allows for more sediment – water mixing by wind.

- B. Increases in nutrient concentration due to drawdown will increase algal primary production, because algal communities in deltaic wetlands, like those in many prairie wetlands, are nutrient limited.
  
- C. Turbidity, nutrient concentrations and algal primary production trends will be related to site depth because sediment resuspension is more likely to occur in shallow sites. Shallow sites in wetland basins not undergoing a drawdown should therefore be similar to drawdown sites in these parameters.
  
- D. The chemical and physical properties of wetland water and sediment will be correlated to distance from channels of the Saskatchewan River, because the river influences wetlands through flood events and seepage through levees.

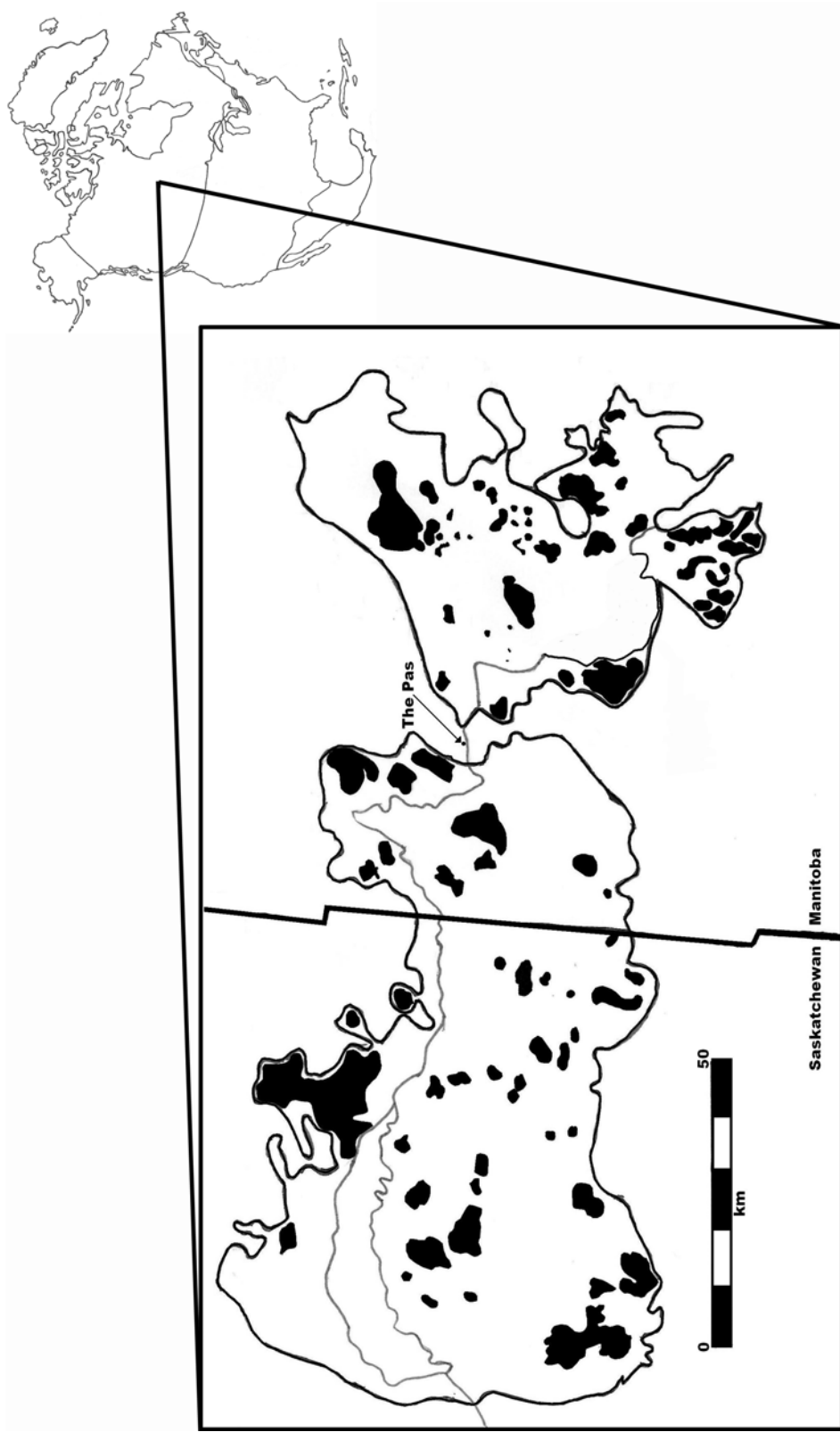


Figure 1-1: Location of the Saskatchewan River Delta in western central Saskatchewan and eastern central Manitoba, Canada. The major river channels and some large lakes are shown.



Figure 1-2: Satellite image showing SRD features including wetlands, shallow lakes, active and abandoned river channels and natural levees. Modified from Smith (2008).



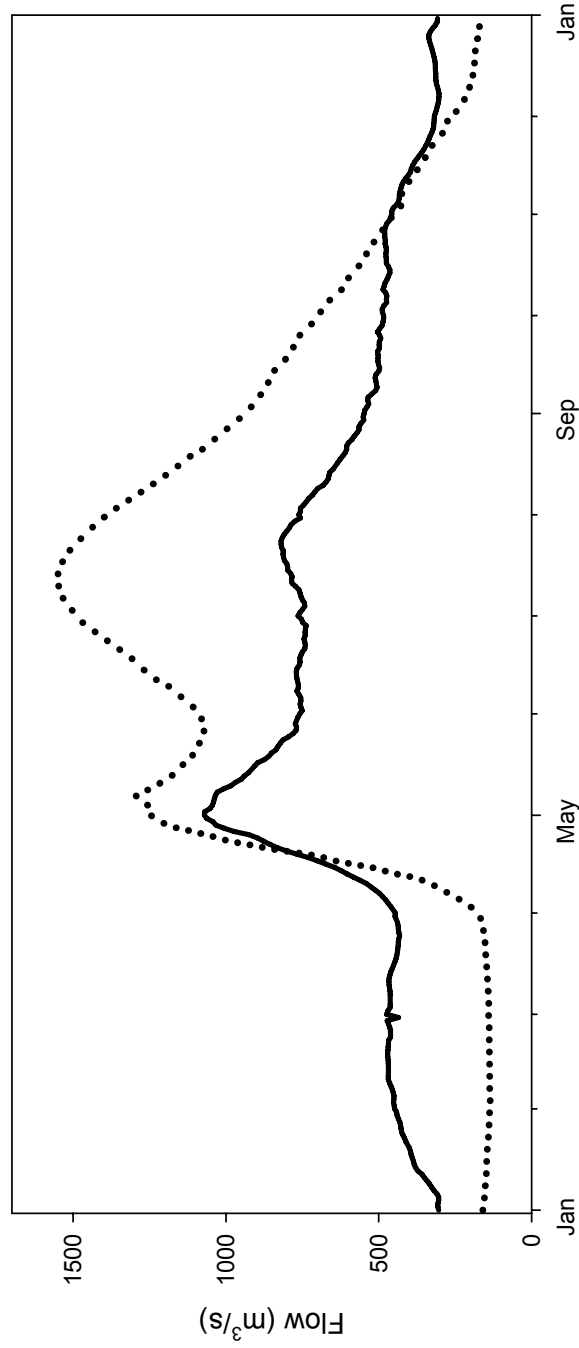


Figure 1-3: Mean daily discharge of the Saskatchewan River at the Water Survey of Canada gauging station at The Pas, Manitoba. The dashed line shows natural flows 1913–1959, before the construction of the Gardiner, EB Campbell and Grand Rapids dams (locations shown in Figure 1-4). It is characterised by low winter flows, with a spring peak from local runoff and a larger summer peak from the mountain melt. The solid line shows the altered flows since the filling of the reservoirs of those dams, 1969–2009. Winter flows are much higher due to the need for increased power generation in this season. The spring peak is smaller and the summer peak is nearly absent due to irrigation needs for water upstream. Overall, flows through the Saskatchewan River Delta are lower and more stable throughout the year. Modified from Leavens (2008) using data from <http://scitech.pyr.ec.gc.ca/waterweb/formnav.asp?lang=0>.

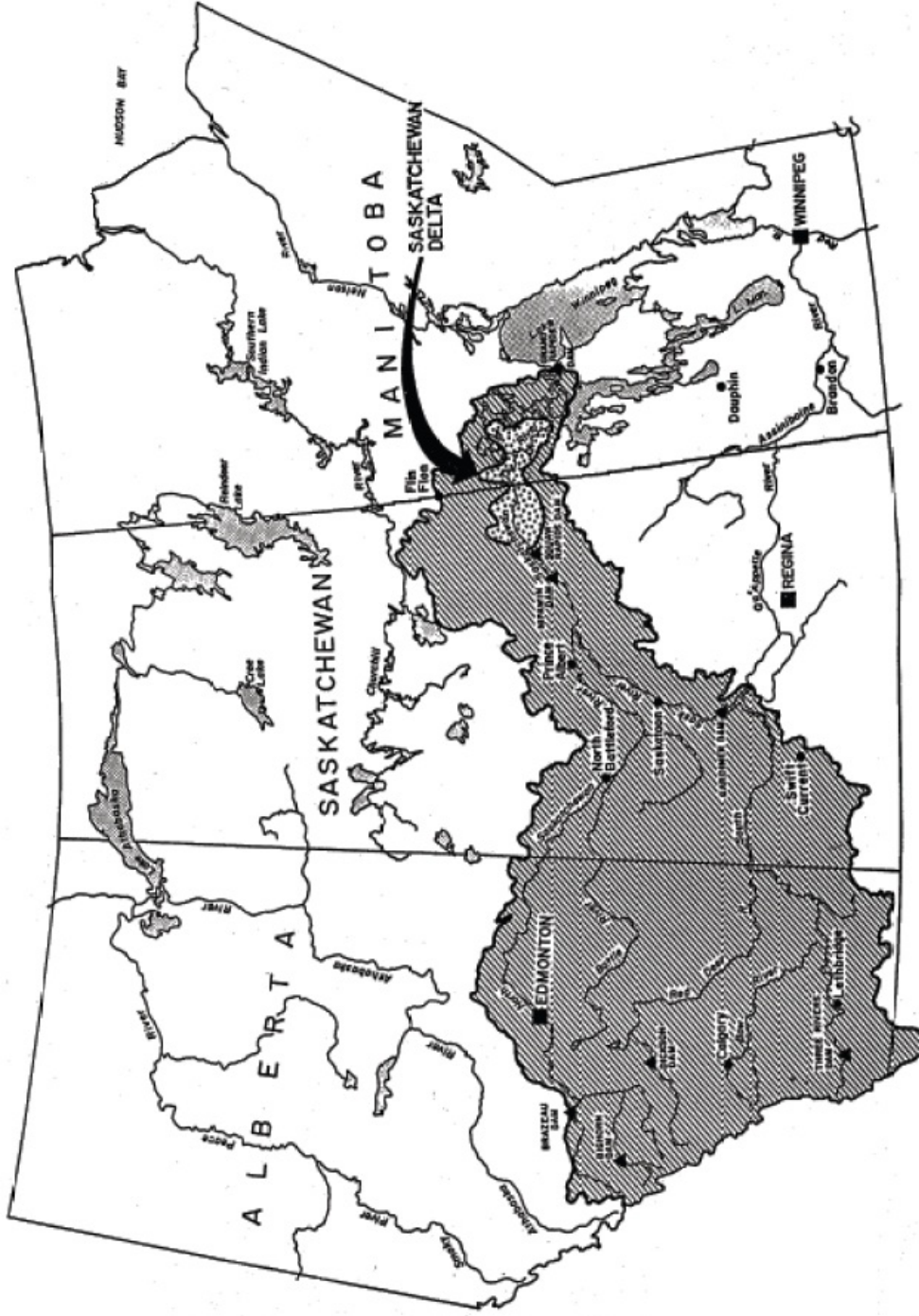


Figure 1-4: The basin of the Saskatchewan River extends across three provinces and into the United States. The location of the SRD is indicated by the arrow. Cities and major towns are indicated by squares and circles. Dams are marked by triangles. Note: Squaw Rapids Dam is currently known as the EB Campbell Dam. Modified from Mudry (nd).

## Chapter 2: Literature Review

### Effect of water level variation on nutrients, algae, and macrophytes

The drought of the 1930s contributed to a basic understanding of the effects that water level variation could have on wetland ecosystem function. A variety of descriptive research was undertaken over the next decades (Bourn and Cottam 1939; Walker 1959; Kadlec 1962; Walker 1965; Weller and Spatcher 1965; Smith 1971; Stout 1971; Millar 1973; Weller and Fredrickson 1974; van der Valk and Davis 1978a), but by the late 1970s, there were calls for long-term multidisciplinary experimentation by water level manipulation to better understand wetland ecological function (Weller 1978).

In response to this challenge, the Marsh Ecology Research Program (hereafter referred to as MERP) was initiated in 1979 by Ducks Unlimited Canada and the Delta Waterfowl and Wetlands Research Station. MERP was a ten year study conducted in ten artificially constructed experimental wetland cells within Delta Marsh, a large lacustrine wetland in Manitoba. Water chemistry, primary production, invertebrate populations, and avian and mammal use was monitored to examine changes ensuing from water level manipulation simulating a natural wet-dry cycle. Over one hundred publications resulted from MERP, including the definitive text *Prairie Wetland Ecology* (Murkin *et al.* 2000). The effects of water level manipulation on nutrient dynamics and primary production as determined through MERP and its predecessors will be summarised here.

Wetlands are dynamic ecosystems, subject to natural fluctuations of water level. Prolonged flooding of a prairie wetland results in a lake marsh state, where emergent macrophytes die back from open water bays and are unable to germinate from the seed bank (van der Valk and Davis 1978a). Without emergent macrophytes to take up nutrients from the sediment, and contribute to mineralisation

of organic matter by aerating sediments, nutrients become locked up in the sediment and porewater pools. The decline in macrophytes also leads to a decline in litter as available substrata for colonisation by periphyton, and as a source of nutrients to aboveground nutrient pools through leaching and decomposition. Phytoplankton is the dominant algal assemblage, but overall primary production from macrophytes and algae is low in the lake marsh state because nutrients are sequestered in the sediment and porewater.

A lowering of water levels, whether naturally through drought or through artificial management, is termed drawdown. Drawdown allows for aeration of the sediments, which increases microbial aerobic decomposition of organic material into inorganic nutrients. Drawdown also allows for the germination of emergent macrophytes, such as bulrush (*Scirpus acutus* and *S. validus*), cattail (*Typha latifolia*, *T. angustifolia*, and *T. (X) glauca*), phragmites (*Phragmites australis*) and whitetop (*Scholochloa fesucacea*), the growth of which is fuelled by these newly mineralised nutrients. MERP found that environmental conditions, competition, herbivory and seed dispersal are interacting factors which determine species distribution (van der Valk 2000). For example, seed germination was not greatest where seed densities for a particular species were highest, but rather where moisture, temperature, and salinity requirements were most suitable. Drawdown also allows annual terrestrial plants to become established on newly exposed mudflats, which can account for more than half of wetland above-ground biomass (van der Valk 2000).

With reflooding, the emergent vegetation again provides a pathway for taking up sediment nutrients and transferring them to the water column and other aboveground nutrient pools. Those annual plant species which cannot tolerate flooding die back and their decomposition generates a pulse of nutrients released to the water column. Epiphyton and metaphyton are the dominant algal assemblages at this stage and production is high in response to increased nutrient availability.

Some key findings of MERP were related to the elucidation of the important role

played by algae in the productivity and function of prairie wetlands. High algal productivity and rapid turnover mean that annual algal production can exceed that of macrophytes, though the standing crop at any one point in time may be lower (Robinson *et al.* 2000). MERP found that algae may play a bigger role than macrophytes or detritus in feeding secondary productivity (Neill and Cornwell 1992; Robinson *et al.* 2000), and they provide wetland invertebrates with food and structural habitat (Murkin and Ross 2000; Robinson *et al.* 2000).

Although MERP was the definitive research program on the effects of water level variation on wetland ecosystem structure and function, MERP, and indeed, the earlier contributions to understanding the wet-dry cycle (Weller and Spatcher 1965; Weller and Fredrickson 1974; van der Valk and Davis 1978a) took place in prairie wetlands. Boreal wetlands require further study to elucidate their response to water level variation.

### Boreal wetlands and water level variation

The boreal region encompasses one third of the area of Canada, and of this area, wetlands comprise approximately 20% or over 600,000 km<sup>2</sup> (National Wetlands Working Group 1988). The vast majority of these are peatlands: wetlands which have a 40cm or greater layer (Tarnocai 1980) of undecomposed organic material. Canada's boreal region supports at least ten forms (National Wetlands Working Group 1988) of fens, minerotrophic peat-producing wetlands with a groundwater connection, and bogs, ombrotrophic peat-producing wetlands receiving water only from precipitation (National Wetlands Working Group 1997). Swamps, treed wetlands not producing a peat layer, are uncommon, and marshes, non-peat-producing, untreed wetlands, and shallow open waters, with a depth less than two metres, are rare (National Wetlands Working Group 1988). Those marshes and shallow open waters that are present in boreal region are typically found either on the margins of lakes, or in the deltas of rivers discharging into lakes or lacustrine plains (National Wetlands Working Group 1988).

Because peatlands overwhelmingly dominate the boreal region – over 85% of Canada’s wetlands are peatlands (National Wetlands Working Group 1988) – it is not surprising that peatlands also dominate the literature relating to boreal wetlands (Dirschl 1972b; Zoltai and Johnson 1987; Futyma and Miller 1986; Kubiw *et al.* 1989; Miller and Futyma 1987; Nicholson 1993; Moore *et al.* 1998; Nicholson *et al.* 2006; Talbot *et al.* 2010), while marshes and shallow open waters have received less attention. Indeed, Bayley and Prather (2003) have lamented the lack of attention received by shallow boreal wetland lakes.

Peatlands and non-peat-producing wetlands are not discrete entities, however. Rather, they are different stages along successional pathways. One successional progression, from shallow open water to fen to bog has been well documented (Tallis 1983; Kratz and DeWitt 1986; National Wetlands Working Group 1988; Nicholson and Witt 1994). In the process of terrestrialisation, an open water basin develops into a fen as it becomes filled by sediments settling to the bottom and by floating mats of vegetation, including mosses, enclosing from the basin’s periphery. As more peat is deposited, the wetland surface may rise above the water table and develop into a bog.

Another potential successional pathway may be followed in regions of fluctuating water levels. Where periodic drying can prevent the colonisation of bryophytes, an open water basin may succeed to a marsh rather than a fen/bog. If drying is ongoing, trees can invade and the marsh can develop into a swamp (Glooschenko and Grondin 1988; Nicholson 1993; Nicholson and Witt 1994). If water level fluctuations continue to include periodic flooding, the wetland remains a marsh (Nicholson and Witt 1994).

Succession in deltaic boreal wetlands also follows these two successional pathways. Dirchl *et al.* (1974) studied successional trends in the Peace-Athabasca Delta. They are differentiated by successional pathways in inactive portions of the delta, where

basins receive spring floodwaters only rarely in the highest water years, and active or semi-active areas of the delta, which are directly affected by hydrological interactions with the river or lake, or which are recharged by floodwaters in most years. Autogenic succession and terrestriation was observed in inactive parts of delta, which had become bogs. Allogenic succession, where aquatic communities developed into shore communities, to meadow communities, to shrub communities, and eventually to forest communities, was observed in the active and semi-active delta.

In non-deltaic boreal peatlands, the vegetation community has been found to be most correlated with water flow and cation gradients; nitrogen and phosphorus concentrations are less important in determining species composition (Nicholson 1995). In deltaic wetlands, however, nutrients seem to be more important. Dirschl and Coupland (1972) and Dirschl (1972a) found that plant species distribution in the upper Saskatchewan River Delta depends on moisture regime, nutrient status, and pH. Nutrient availability and pH decrease with increasing distance from the river along transects from mixed forest alluvial levees, to open drainage basins with marsh or fen vegetation, to closed drainage basins with bog communities.

These studies of water level related change in deltas and other boreal wetlands have thus far been mainly concerned with successional changes over the long-term: hundreds or thousands of years. While studying the Saskatchewan River Delta, Dirschl (1970) stated that “compared to the rapid, cyclic changes evident in the wetland vegetation of the neighbouring aspen grove and grassland regions, the vegetation changes in the study area are slower, less fluctuating, and essentially unidirectional.” However, relatively recent changes to the water regime of the Peace-Athabasca Delta (PAD) demonstrated that short-term effects on water quality and vegetation can also be observed in deltaic boreal wetlands.

Prior to construction of the WAC Bennett Dam built in 1967 on the Peace River, an annual summer flood of the Peace River would normally cause a rapid rise in water

level on Lake Athabasca and connected wetlands with the delta, and flooding of isolated wetland basins (Dirschl 1972b). Since the dam interrupted this summer flood, water levels dropped substantially on Lake Athabasca and the inundated area of connected wetland lakes decreased by 38%, exposing 500 km<sup>2</sup> of mudflat (Jaques 1990). Isolated wetland basins on the floodplain have dried out partially or completely without overbanking in flood years. The wetlands changed, over a four year period, from marshes and shallow open waters, to open mudflats with scattered seedlings of emergent macrophytes, to mixed communities of sedges and grasses, to immature fens with a variety of herbaceous plants, to dense monospecies sedge meadows, and finally were colonised by willows (Dirschl *et al.* 1974). This progression was interpreted as an acceleration of the natural successional pathway of the PAD wetlands, shifting quickly from shallow marsh and wet meadow to shrub and forest communities, with an overall reduction in biodiversity and area and number of wetlands (Dirschl *et al.* 1974). However, some have cautioned against interpreting these changes as indicative of ecosystem stress or death, iterating that deltas are dynamic, hydrologically variable systems where changes in water level and plant communities are to be expected, and that current water regime trends do not fall outside the long-term normal (Timoney 2002).

More recent studies in the PAD have suggested that flooding from ice jamming on delta channels, rather than summer flooding, were generally responsible for recharging isolated wetlands (Prowse and Lalonde 1996; Northern River Basins Study Board 1996). However, neither weirs on major delta channels, nor artificial induced ice jamming, has been able to mimic natural water levels sufficiently to curb the invasion of willows and shrubs into isolated wetlands and restore pre-dam wetland conditions (Jaques 1990; Prowse and Conly 2002).

Although short-term changes resulting from water level variation have begun to be examined in the PAD, more study on these effects, here and in other locations, is necessary to contribute to an understanding of boreal wetland function and to inform



boreal wetland management.

### Algae and nutrient deficiency in wetlands

The growth of algae can be limited by physical constraints, such as temperature (DiNicola 1996), grazing pressure (Steinman 1996), and the availability of light (Hill 1996). Additionally, algae, and all plants, have certain nutritional requirements, and whichever nutrient is in lowest supply relative to algal physiological demands can be said to be limiting to growth (Borchardt 1996).

Phosphorus has long been generally accepted as the key nutrient limiting to plant growth in freshwater (Schindler 1977; Hecky and Kilham 1988; Carpenter *et al.* 1992; Lampert and Sommer 1997; Dodds 2002; Kalff 2002; Dodson 2005; Brönmark and Hansson 2005; Howarth and Marino 2006). However, recent reviews have begun to suggest that P-limitation may not be as dominant as previously thought, but rather is only limiting in certain freshwater environments and over long time scales (Elser *et al.* 2007; Sterner 2008).

The bulk of studies on algal nutrient limitation have focussed on streams (Huntsman 1948; Stockner and Shortreed 1978; Elwood *et al.* 1981; Pringle and Bowers 1984; Bothwell 1988; Winterbourn 1990; Dale and Chambers 1996; etc) and lakes (Haertel 1976; Allan and Kenney 1978; Barica *et al.* 1980; Campbell and Prepas 1986, Prepas and Trimbee 1988; Barica 1990; Waiser and Robarts 1995; Arts *et al.* 1997; Graham 1997; etc). Relatively fewer nutrient limitation studies have taken place in wetlands (Snow and Brunskill 1975; Purcell 1999; McDougal 2001; Squires and Lesack 2002; Kolochuk 2008; Hertam 2010; Bortoluzzi *et al.* in prep). Those few provide evidence that the nutrient limitation picture in wetlands differs from other freshwater environments, based on differences in depth, chemistry, and residence time.

Within wetlands, phosphorus can be highly recycled, becoming available for plants and algae. Wetlands may have long water residence times, and Barica (1987) found trends that longer residence times allow for more internal accumulation and recycling of P between sediment and water column. Unlike stratified lakes, wetlands are shallow enough to be thoroughly mixed, increasing sediment-water contact and releasing inorganic phosphorus, sorped to sediments, back into the water column (Scheffer 1998; Søndergaard *et al.* 2003; Dunne and Reddy 2005). Many wetlands have high sulphate concentrations, forming hydrogen sulphide in their anaerobic sediments. Under these conditions iron oxides may be reduced to iron sulphide, further freeing the inorganic phosphorus bound to iron oxides in the sediments (Caraco *et al.* 1989). Wetlands are highly productive and therefore also provide organic phosphorus, which is soluble and can available to algae through alkaline phosphatase activity (Morris and Lewis 1988; Vitousek *et al.* 1991; Axler *et al.* 1994).

Conversely, wetlands provide conditions where nitrogen may be less available to algae. Increased sediment-water contact does not release inorganic nitrogen, because nitrates and ammonia do not sorb to sediment particles as does phosphorus (Jensen *et al.* 1991). Although some algae can actively transport certain organic nitrogen compounds into their cells (Rees and Syrett 1979), in many wetlands, the majority of nitrogen is in complex organic forms, which, unlike organic phosphorus, may not be available for consumption by algae (Morris and Lewis 1988; Vitousek *et al.* 1991; Axler *et al.* 1994).

As well, nitrogen can be permanently lost to the atmosphere as N<sub>2</sub> gas by denitrification. Wetlands are especially prone to denitrification because their water saturated sediments have anaerobic, highly reduced conditions ideal for denitrifying bacteria, and because their shallow depths and long residence times maximise sediment-water contact (Broderick *et al.* 1988; Saunders and Kalff 2001; Poe *et al.* 2003). Craft (1997) found that, through denitrification, wetlands can be nitrogen sinks regardless of the amount of nitrogen supplied.

It has been argued that the essentially unlimited supply of atmospheric nitrogen should preclude the possibility of nitrogen limitation, because cyanobacteria can fix  $N_2$  gas into ammonia (Schindler 1977). However, Ferber et al. (2004) have demonstrated that only a tiny proportion (2%) of the nitrogen demands of nitrogen-fixing cyanobacteria in shallow lakes is met through nitrogen fixation. For the remainder of their nitrogen supply, cyanobacteria are competing for nitrogen with other freshwater flora. Additionally, both Ferber et al. (2004) and Mugidde et al. (2004) have suggested that the impression of the role of nitrogen fixation in past studies may have been exaggerated by high internal recycling of inorganic nitrogen and low allochthonous inputs. Under these conditions, nitrogen limitation is a distinct possibility.

Indeed, periphyton limitation by nitrogen has been observed in several wetland environments. Bortoluzzi *et al.* (in prep) found nitrogen to be the most common limiting nutrient in Delta Marsh, Netley-Libau Marsh, and Oak Hammock Marsh, three large wetlands in southern Manitoba of varying hydrology. Kolochuk (2008) found nitrogen limitation was nine to ten times more common than phosphorus limitation in farm pond wetlands across southern Manitoba. Nitrogen-phosphorus co-limitation was also common in farm pond wetlands (Kolochuk 2008). Squires and Lesack (2002) found that the limiting nutrient varied between nitrogen, phosphorus, and nitrogen and phosphorus together, across wetlands in the MacKenzie River Delta.

This evidence from the few nutrient deficiency studies in wetlands is supported by work conducted in lakes with wetland-like characteristics. Nitrogen limitation of algae has been seen in systems with high concentrations of total nitrogen (Campbell and Prepas 1986; Morris and Lewis 1988; Robarts *et al.* 1992). High sulphate concentrations have been linked to nitrogen deficiency due to phosphorus sufficiency (Caraco *et al.* 1989; Caraco *et al.* 1990; Robarts *et al.* 1992; Gächter and Müller 2003).

As well, nitrogen limitation by aquatic macrophytes has been observed in shallow systems (Van Donk *et al.* 1993; Meijer *et al.* 1994). Although algae and macrophytes may differ in their precise nutrient requirements, nitrogen limitation of the latter may contribute to nitrogen limitation of the former. Macrophytes growing in nitrogen deficient conditions tend to produce litter with high C:N ratios and more recalcitrant compounds, such as lignin (Vitousek *et al.* 1991). These nitrogen limited macrophytes decompose more slowly than those supplied with sufficient nitrogen, leaving nitrogen bound up in organic compounds that are unavailable for algae.

There seems, therefore, to be some initial support to the hypothesis that nitrogen may be a limiting nutrient to wetland algal and macrophyte communities. More research in a variety of wetland environments is necessary.

## Chapter 3: Site Description and Study Design

### Summerberry Marshes Site Description

The Summerberry Marshes comprise a 150 km<sup>2</sup> region located in the lower SRD, approximately 25 km southeast of The Pas (Figure 3-1). The area takes its name from the Summerberry River, a distributary of the Saskatchewan River located to the east and north of the main river channel. The Summerberry complex consists of approximately 40 discrete wetland basins between the Summerberry and Saskatchewan Rivers and just south of the Saskatchewan, most of which are 5 km<sup>2</sup> in size or smaller (Clay 1978). The wetlands are of similar size, age, and vegetation community, making Summerberry an ideal region for controlled experimentation. The basins are small enough to be manipulated with minimal difficulty, and many have water control structures already in place. Additionally, Manitoba Conservation has a patrol cabin in Summerberry at Hill Island, which provided a convenient base of operations for field research, including accommodations and a laboratory tent.

### Water Level Control in Summerberry

The Wildlife Branch of the Government of Manitoba first began manipulating water levels at Summerberry, with the goal of improving wildlife habitat, in the late 1930s with control structures built in 1936-37 (Ramsay et al. 1955). In the late 1970s and early 1980s, these structures were upgraded to fixed crest weirs and gated culverts by Ducks Unlimited Canada (DUC), who undertook responsibility for water level control in the area (Smith and Jones 1982). Infrastructure at Summerberry currently consists of 29 control structures which allow water level manipulation to 26 wetland basins (Reader 2008).

Throughout the 1980s, drawdowns were periodically implemented in many

Summerberry basins, with the goal of improving wildlife habitat (Smith 1986; Smith 1987). This practice was based upon wetland dynamic models developed in prairie wetland ecosystems (van der Valk & Davis 1978). The practice of drawdown in the SRD was discontinued in the late 1980s as per the recommendations of Smith (1987) when plant communities failed to respond as predicted by these models. Specifically, DUC monitoring noted a decrease in diversity of emergent vegetation and an increased density within sedge stands (Smith 1986; Smith 1987).

No drawdowns had taken place within Summerberry since the late 1980s (Smith 1987). In the years preceding this study, most Summerberry control structures had been holding high levels of water in the wetlands, leading to even greater water level stabilisation than that seen by the SRD as a whole (Baschuk 2010).

### Study Design

Two experimental treatments were applied to six study wetlands. A control treatment involved no water level manipulation. A drawdown treatment involved draining water from the wetland into the Saskatchewan River or, in the case of one wetland, to a smaller distributary known as the back channel.

The six study wetlands were chosen for this study in three pairs, with each pair consisting of a drawdown and a control wetland. These pairs were 33/35HI (drawdown) with 34HI (control), 34/37C (drawdown) with 32C (control), and 14R (drawdown) with 21C (control), which can be seen in the site map (Figure 3-2). Basic characteristics of each study wetland are given in Table 3-1. Wetlands were paired based on similar depth, water quality and vegetation communities, as determined by water sampling and visual inspection during a preliminary visit in August 2006. The drawdown wetlands were selected from amongst those with newer, weir-type control structures, rather than the older culvert-type control structures which can be more difficult to operate (Figure 3-3). A final criterion for

wetland selection was proximity to the other wetland within the pair, to other pairs, and to the Hill Island base camp. This minimised travel time and allowed for the control and drawdown wetland of a pair to be sampled on the same day and under similar weather conditions.

Manipulated wetlands underwent a partial drawdown because of concerns that a complete drawdown would kill most or all of a wetland's muskrat population. Because the rationale of the greater study was to improve habitat for wetland wildlife, including muskrat, it was decided to leave approximately 60 cm of water depth in the deepest places of each wetland basin. It was hypothesized that this depth would be sufficient for muskrat to travel below the ice, but potentially shallow enough to see a response in wetland vegetation. A target level for drawdown based on this criterion was determined for each wetland. These targets, which are listed in Table 3-2, were a mean of 32 cm below pre-drawdown levels.

The use of pumps to remove water from the drawdown wetlands, which would have allowed for fine precision and bi-directional movement of water regardless of river water levels, was not deemed financially feasible, either for this study or for the future management the study was designed to guide. Therefore a gravity drawdown was achieved simply by opening water control structures (Figure 3-4).

Some areas defined as a single wetland for the purpose of this study were in fact two basins with an open water channel connection and without obvious levee separation. 33HI and 35HI were connected, and were therefore treated as a single wetland because the combination more closely matched the size and other characteristics of its paired control, 34HI. 34C and 37 were similarly connected, and were therefore treated as a single wetland in 2007. However, when drawdown was initiated in fall 2007, beaver damming on the channel between 34C and 37C prevented water levels from dropping in 34C. Dam removal and efforts to deter beaver activity through the use of electric fences and live traps consistently failed throughout summer 2008. Therefore any comparison of drawdown with pre-

drawdown conditions involves fewer measurements from 34/37C than from other drawdown wetlands, because 34C sites could not be included as true drawdown sites and only 37C sites were included.

Sampling of water quality, algal production and macrophyte vegetation began in June 2007. Drawdown commenced in late August 2007. Sampling in summer 2007 provided data on the baseline, reference condition of the study wetlands, and summer 2008's water quality and algal production data measured the response to the drawdown treatment. Studies of the muskrat population and waterbird and fish communities began in 2008. Climate data for 2007 and 2008 are presented in Table 3-3, with monthly trends shown in Figures 3-5, 3-6 and 3-7.



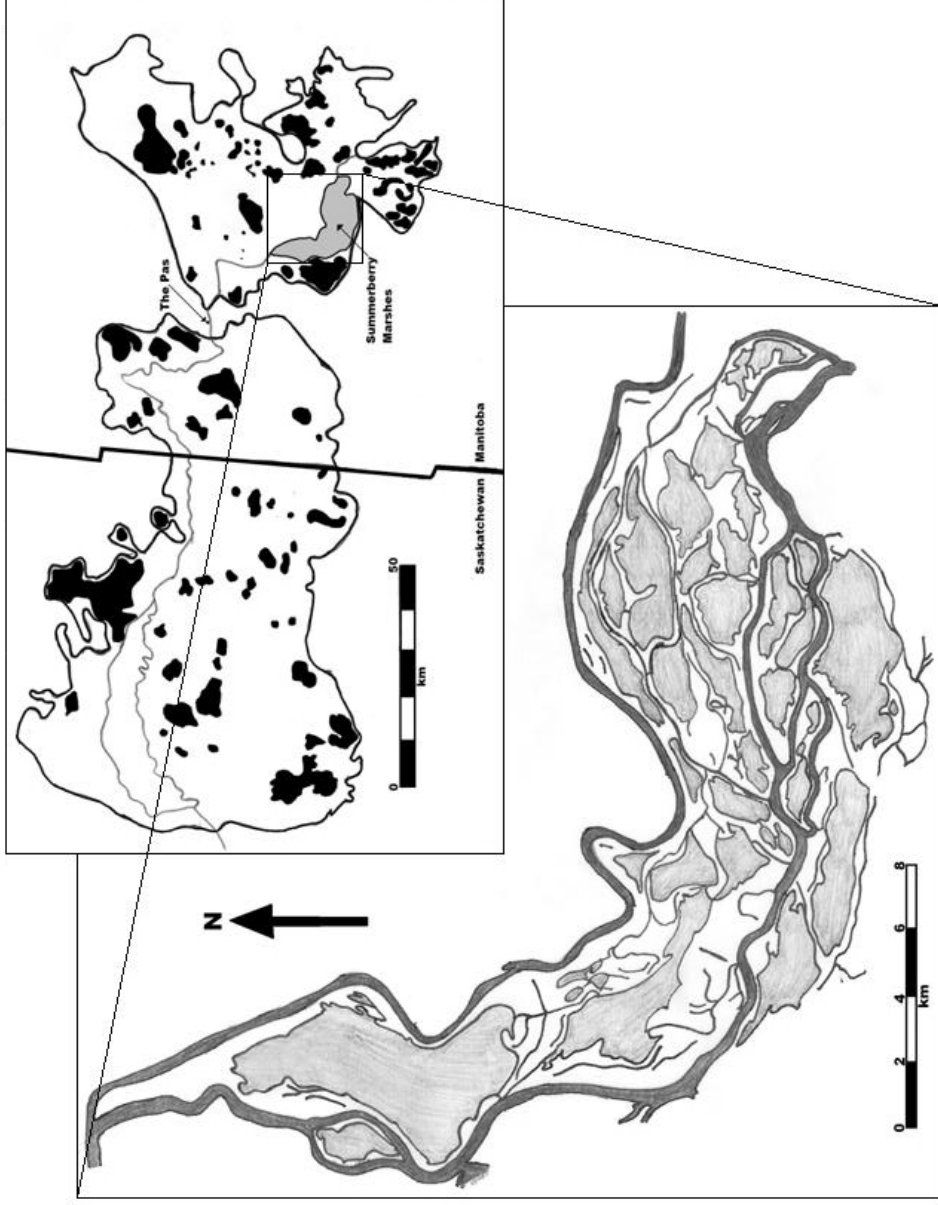


Figure 3-1: Location of the Summerberry Marshes within the Saskatchewan River Delta. The Saskatchewan River flows through the region; the Summerberry River is shown to the north. Study wetlands are identified in Figure 3-2.

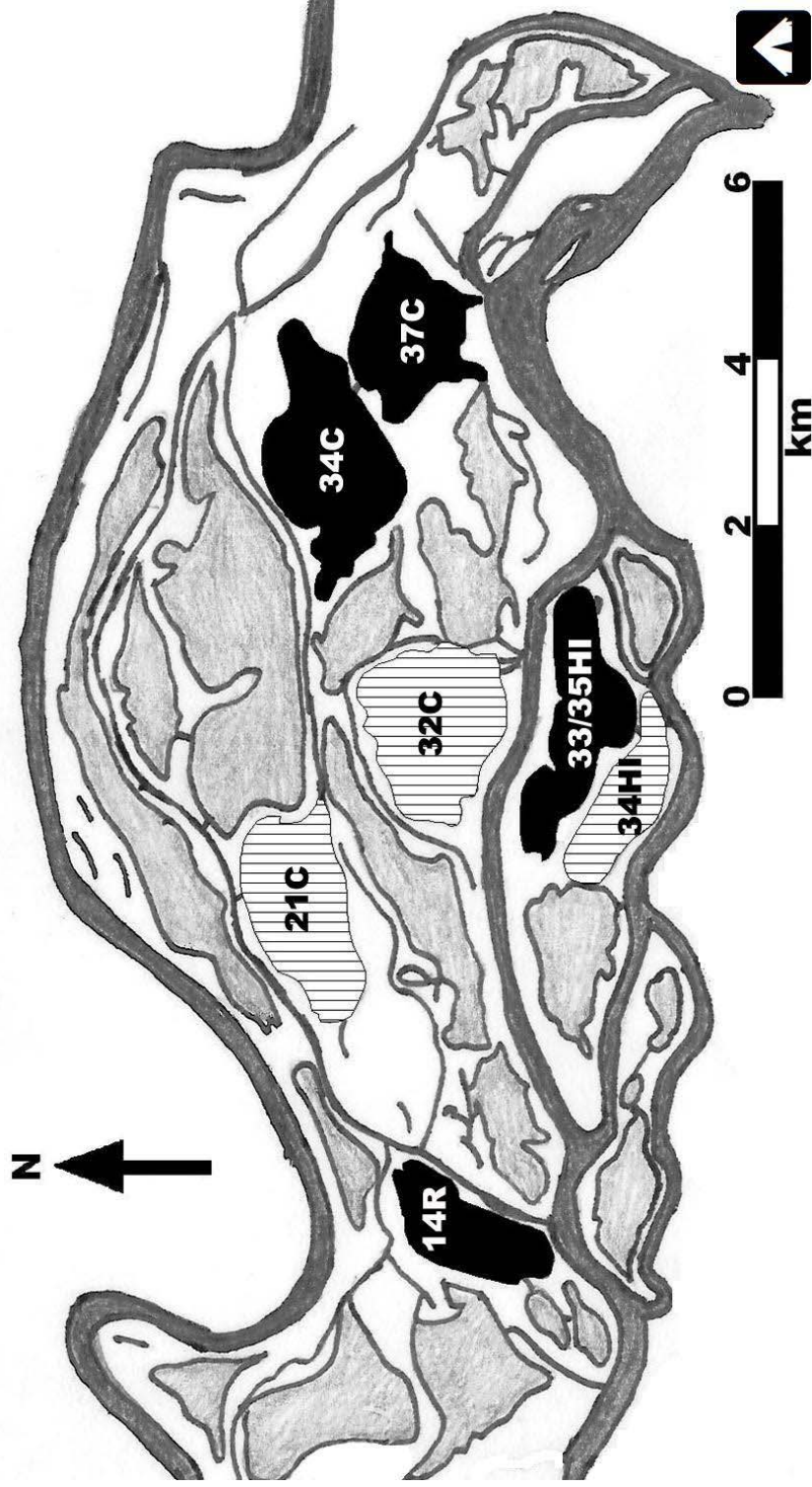


Figure 3-2: A map of the Summerberry Marshes, with study wetlands highlighted. Drawdown wetlands 33/35HI, 34/37C and 14R are marked in solid black. Control wetlands 34HI, 32C, and 21C are shown with black and white hatching. Hill Island camp is represented by the tent symbol.



Figure 3-3: Summerberry water control structures. Top left: 35HI weir, viewed from the wetland. Top right: 37C weir, viewed from the Saskatchewan River, releasing water. Bottom left: 14R weir, viewed from the back channel. Bottom right: 32C gated culvert control structure. Other control wetlands had similar gated culvert controls, which were not operated during 2007 and 2008. Photography by Dale Wrubleski.

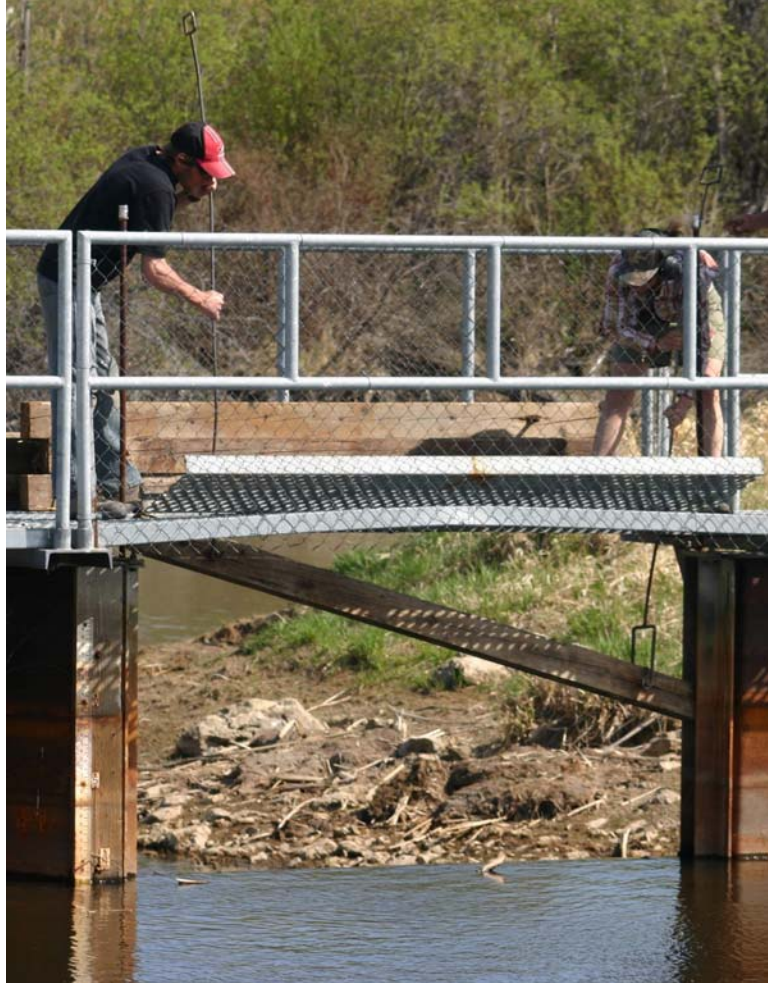


Figure 3-4: Research assistants Jared Knockaert and Sheila Atchison initiate drawdown by removing stoplogs from a Summerberry weir.

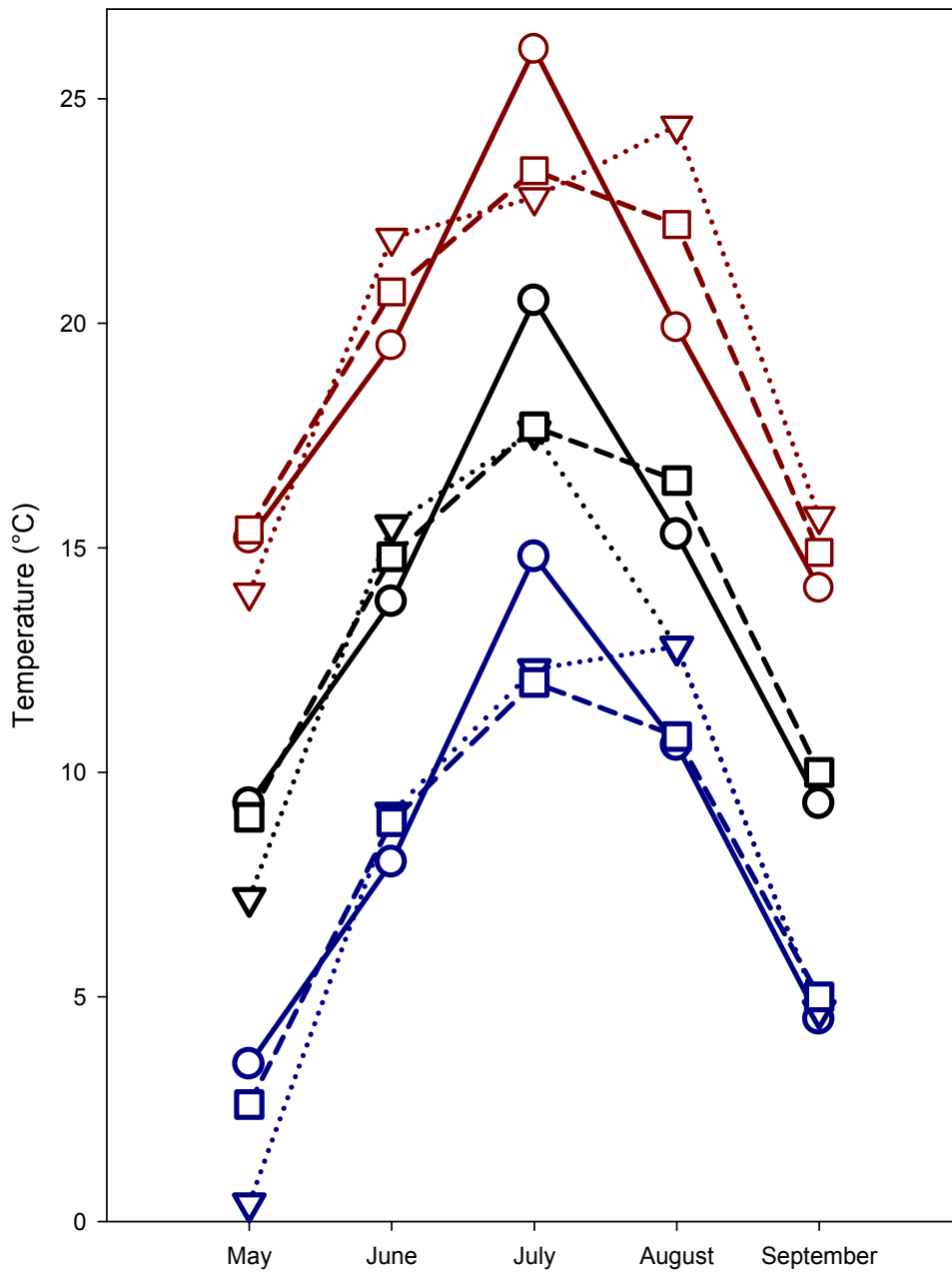


Figure 3-5: Monthly trends in mean maximum temperature (red), mean temperature (black), and mean minimum temperature (blue) for summer months at The Pas, Manitoba. Circles and solid lines represent 2007; triangles and dotted lines represent 2008; squares and dashed lines represent norms for the period 1971 – 2000. Data from the Canadian Meteorological Service.

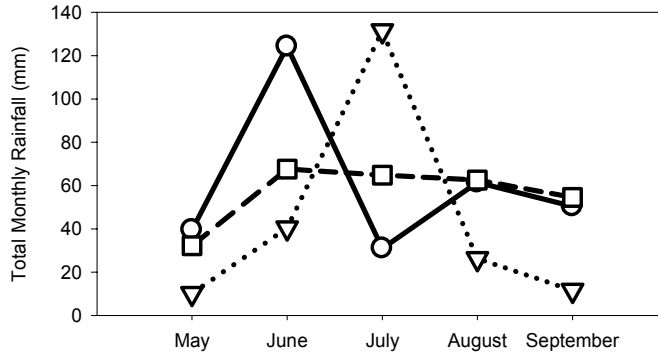


Figure 3-6: Total monthly rainfall for summer months at The Pas, Manitoba. Circles and solid lines represent 2007; triangles and dotted lines represent 2008; squares and dashed lines represent norms for the period 1971 – 2000. Data from the Canadian Meteorological Service.

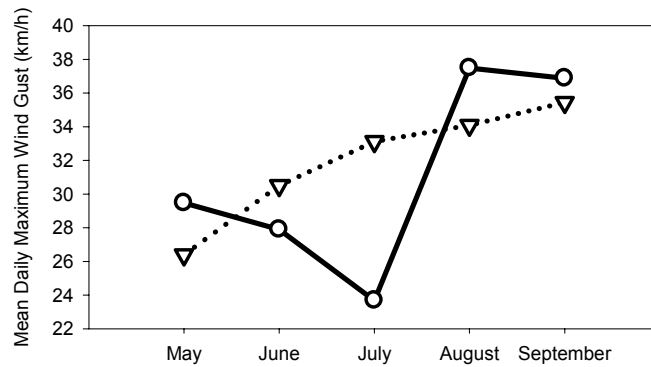


Figure 3-7: Monthly trends in mean daily maximum wind gust for summer months at The Pas, Manitoba. See Figure 3-6 for legend.

Table 3-1: Characteristics describing Summerberry study wetlands. Basin areas from Clay (1978).

Wetland	Location	Basin Area (ha)	Treatment	Control Structure	Nearest Channel
14R	53° 37' 32" N 100° 53' 21" W	126.4	drawdown	weir	back channel
21C	53° 38' 30" N 100° 50' 21" W	176.2	control	gated culvert	back channel
33/35HI	53° 37' 00" N 100° 48' 48" W	171.5	drawdown	weir	SK river
34HI	53° 36' 45" N 100° 49' 26" W	101.3	control	gated culvert	SK river
34C	53° 38' 18" N 100° 46' 31" W	277.2	control	beaver dam	SK river
37C	53° 37' 59" N 100° 45' 21" W	314.3	drawdown	weir	SK river
32C	53° 37' 50" N 100° 48' 49" W	207.8	control	gated culvert	SK river

Table 3-2: Original and target water levels for drawdown wetlands, and the date drawdown commenced and completed. The largest wetland, 37C, did not fully drain to target level in early fall 2007, so drawdown was reinitiated in spring 2008. 34C never reached the target water level.

<b>Drawdown Basin</b>	<b>Water Level Prior to Drawdown (masl)</b>	<b>Target Drawdown Water Level (masl)</b>	<b>Date Drawdown Commenced</b>	<b>Date Drawdown Completed</b>
14R	257.75	257.25	August 10, 2007	September 2007
33/35HI	257.1	256.95	August 10, 2007	September 2007
34/37C	256.95	256.65	August 9, 2007	37C – June 2008 34C – not achieved

Table 3-3: Climate parameters for June – July – August 2007 and 2008 at Environment Canada’s The Pas airport weather station. Normals for the period 1971 – 2000 are also presented. 2008 was the warmer, drier, windier year. Both 2007 and 2008 were warmer and much drier than the normals.

<b>Parameter</b>	<b>June – Aug 2007</b>	<b>June – Aug 2008</b>	<b>June – Aug 1971-2000</b>
Highest Extreme Temp (°C)	34.3	32.3	-
Mean Maximum Daily Temp (°C)	21.6	24.4	19.7
Mean Temp (°C)	16.4	17.6	13.7
Mean Minimum Daily Temp (°C)	11.0	12.8	7.8
Lowest Extreme Temp (°C)	8.3	6.8	-
Total Rainfall (mm)	216.6	131.4	281.6
Highest Extreme Wind Gust (km/h)	69	76	-
Mean Maximum Daily Wind Gust(km/h)	29.4	34.1	-



## Chapter 4: Water Quality Response to Drawdown

### Introduction

This study examined the effect of drawdown on water quality in the Summerberry wetlands. Water column and sediment nutrient concentrations, water clarity, ion concentrations, sediment composition, and algal primary production were measured in drawdown wetlands and were compared with control wetlands and with the pre-drawdown state.

To examine whether water quality changes due to drawdown were specifically due to applying the drawdown treatment to an entire wetland, or were more general trends based on depth applicable across the region, relationships between water quality parameters and depth were studied.

To understand the influence of the Saskatchewan River on the Summerberry Marshes, wetland water quality was also compared with that in nearby river channels, and related to distance from the river.

### Methods

#### Sampling Sites

There were 36 wetland sites where water samples were collected: six in each of the three manipulated and three control wetlands (Figure 4-1). Sites were chosen to represent a variety of water depths, and degree of shelter from emergent plants. All water sampling sites were approximately 5m from the edge

of emergent vegetation islands or shorelines because airboats, which are very difficult to stop on open water, were used for transportation. Because emergent vegetation may protect water from wind effects, which could affect water chemistry, sites were chosen with this potentially sheltering emergent vegetation in a variety of directions. Approximately half the sites were classed as open water sites, near the edge of large open bays; the remainder were classed as vegetated sites, with emergent vegetation within 15m in all directions.

Because flooding from the Saskatchewan River may supply these wetlands with water, a comparison between study wetlands and the river channels was also undertaken. Two sampling sites were established in channels of the Saskatchewan River near study wetlands: one site in the main channel and one in a smaller back channel which passes close to two of the study wetlands (Figure 4-1).

Each of these 38 water sampling sites was visited by airboat approximately biweekly throughout the two field seasons: five times during summer 2007, and seven times in 2008. Drawdown wetlands were sampled on the same day as the most similar control wetland, except in the event of extreme weather or airboat breakdown. Dates of sampling rounds are listed in Table 4-1.

Sites were geo-referenced with the use of a handheld Garmin 76CSx GPS receiver (Appendix I). Distance between each site and the main channel of the Saskatchewan River was measured using ArcMap 9.3. Distance from each site to the nearest smaller river channel was also measured.

### Water Quality

*In situ* water measurements made at each site included mid-column specific conductance, light extinction profile (using a Li-Cor LI1000 data logger with LI193

spherical underwater and LI200 flat aerial sensors), water depth, and Secchi depth.

Two integrated water column samples were collected from each site, using a clear acrylic tube 6.3cm in diameter and one meter in length. The column was oriented vertically and lowered until the top of the tube was just above the surface of the water, or, for shallower sites, until the bottom of the tube was 10cm above the sediment. A plastic ball capping the tub created the suction necessary to lift the sample above the wetland surface, and 1L of the water obtained was collected in a twice-rinsed 1L polypropylene bottle. The samples were kept in coolers with ice packs for a maximum of six hours until transported back to the laboratory where they were kept at 4°C until analysed. All chemical analysis took place as soon as possible within 24 hours of sample collection, and in compliance with the APHA (1998) recommended times.

The water samples were analysed for pH, alkalinity, total reactive phosphorus, ammonia-nitrogen (total ammonia nitrogen or TAN, hereafter referred to as 'ammonia'), and turbidity following the methods listed in Table 4-2. A 20mL subsample of water was refrigerated and later analysed for total phosphorus and total nitrogen (Table 4-2). A 20mL vial of Whatman GF/C filtrate was refrigerated for analysis of dissolved organic carbon by UV spectroscopy (Table 4-2).

Subsamples of water were also sent to CanTest Laboratories in Winnipeg for analysis of major cations ( $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Ca}^{2+}$  and  $\text{Mg}^{2+}$ ) and chloride. These subsamples were collected during the first, third, and final sampling round in 2007 and the final sampling round in 2008 (see Table 4-1 for dates).

Mean nutrient concentrations, ion concentrations, and physical parameters were plotted against date, using SigmaPlot 10.0, to elucidate seasonal or yearly trends. Water quality parameters were compared between drawdown versus control sites, and open water versus vegetated sites, by one-way ANOVA using

JMP 8 software. Relationships between water quality parameters and site depth, and between water quality and distance from the Saskatchewan River, were explored with regression analyses using JMP 8 software.

Turbidity was the only measure of water clarity included in comparisons between wetland sites. Although field measurements for the calculation of euphotic depth were made where possible, the nature of some wetland sites frequently precluded these measurements. Euphotic depth can be calculated as the depth which receives 1% of surface light. However, the underwater light sensor required a site depth of at least 50cm to accurately measure light extinction, so no readings could be made at shallow sites. Euphotic depth can also be approximated using the Secchi disk depth and a calibration factor (Tyler 1968), but no Secchi readings could be taken at sites where high water clarity allowed the disk to be seen on the wetland bottom. These constraints resulted in a high proportion of missing datapoints in which clear shallow sites could have been over-represented. Therefore euphotic depth is not included in wetland comparisons between drawdown and control sites. The greater depths and lower water clarity at river channel sites allowed light extinction and Secchi readings to be taken reliably, and therefore euphotic depth, calculated based on light readings and supplemented by Secchi-calculated approximates where light data were unavailable, is presented for sites in the Saskatchewan River and back channel.

### Algae

Biomass of phytoplankton and periphyton was approximated by concentration of chlorophyll  $\alpha$  as per the method in Table 4-2. Phytoplankton was collected as the retentate from passing a known volume of water through a Whatman GF/C filter. Periphyton was collected by suspending sterile silica substrata of known surface area (Leco Instruments 5288-042) 10cm below the water surface for a period of

two weeks. A detailed description including photographs of the periphyton sampling apparatus can be found in the next chapter (Figure 5-1).

### Sediments

Surface sediment samples were obtained at every sampling site using a Wildco aquatic sediment corer during the first, third and fifth sampling round of 2007, and the final sampling round of 2008 each year. The top 5 cm of each sediment core was retained and refrigerated. Loss on evaporation and ignition methodology (Heiri *et al.* 2001) was used to determine sediment wet density, percent water content, percent organic content and percent carbonate. Calculations are shown in Table 4-3. Hach persulfate digestion kits were used to measure sediment total nitrogen and phosphorus as per the methods for analysis of water (Table 4-2).

Sediment composition and nutrient parameters were compared between drawdown and control sites by one-way ANOVA using JMP 8 software. Relationships between sediment parameters and depth, and between sediment parameters and distance from the Saskatchewan River were explored with regression analysis. Linear regression analysis was also used to examine correlations between water column nutrient concentrations and sediment nutrient concentrations.



Figure 4-1: The Summerberry Marshes, with the locations of this study's 36 wetland and two river channel sampling sites marked by white circles.

Table 4-1: Dates of the five rounds of water sampling in 2007 and the seven rounds in 2008.

2007	2008
	May 25 - 29
	June 6 - 10
June 20 - 24	Jun 19 - 23
July 4 - 10	July 3 - 6
July 18 - 23	July 16 - 21
Aug 1 - 5	July 29 - Aug 2
Aug 14 - 22	Aug 21 - 25

Table 4-2: Methods used for water chemistry analyses.

Parameter	Method	Reference	Minimum Detectable
<b>Physical Parameters</b>			
Specific Conductance	YSI model 30 meter		± 5% µS/cm
pH	Hach portable meter		0.1
Turbidity	nephelometry; Hach model number 2100 turbidimeter		± 2% NTU
<b>Nutrients</b>			
<b>Carbon</b>			
Dissolved Inorganic C	colourimetric; alkalinity acid titration	APHA 1998	20 mg/L
Dissolved Organic C	UV spectrophotometry	Badiou <i>et al.</i> (see Appendix II)	
<b>Nitrogen</b>			
Ammonia-N	colourimetric; phenolhypochlorite	Stainton <i>et al.</i> 1977	0.01 mg/L
Total N	colourimetric; Hach persulfate digestion kit	APHA 1998	0.5 mg/L
<b>Phosphorus</b>			
Total P	colourimetric; Hach persulfate digestion kit	APHA 1998	0.06 mg/L
Total Reactive P	colourimetric; acid molybdate	Stainton <i>et al.</i> 1977	0.01 mg/L
<b>Ions</b>			
Total Calcium	inductively couple		0.05 mg/L
Total Magnesium	plasma optical emission		0.05 mg/L
Total Potassium	spectroscopic		0.25 mg/L
Total Sodium			0.1 mg/L
Total Chloride	ion chromatographic	APHA 1998	0.2 mg/L
<b>Biological Parameters</b>			
Chlorophyll α	spectrophotometry; 90% methanol extraction	Marker <i>et al.</i> 1980	0.1 µg/L

## Results

### **Wetland Water Quality**

Total nitrogen (TN) concentrations in study wetlands ranged from undetectable levels ( $< 0.25$  mg/L) to 113mg/L, with a mean value of 3.52 mg/L and a median of 2.12 mg/L. Maximum TN concentrations were reached in July or August, with lower values in spring and fall (Figure 4-2). Drawdown wetlands (2008) were 1.7 higher in total nitrogen than control wetlands (2008) ( $p = 0.04$ ; Figure 4-3). There was no significant difference between those wetlands in 2007. Total nitrogen concentrations were significantly higher in 2008 than in 2007 in both drawdown (2.5 times;  $p = 0.01$ ) and control (1.7 times;  $p = 0.005$ ) wetlands.

Ammonia concentrations were below the minimum detection threshold of 25  $\mu\text{g/L}$  through most of summers 2007 and 2008 (Figure 4-4), but reached higher levels in August, to a maximum of 768.4  $\mu\text{g/L}$ . The mean wetland ammonia concentration was 19.9  $\mu\text{g/L}$  while the median value was undetectable. Drawdown wetlands (2008) were 3.0 times higher in ammonia than control wetlands (2008) ( $p = 0.02$ ; Figure 4-5). There was no significant difference between those wetlands in 2007. The drawdown wetlands were 4.2 times higher in ammonia in 2008 under drawdown conditions than in 2007 under high water conditions. There was no significant difference in ammonia concentrations in the control wetlands between 2007 and 2008.

Total phosphorus (TP) concentrations were highly variable, especially at vegetated sites and in drawdown wetlands (Figure 4-6). Values ranged from undetectable ( $< 60$   $\mu\text{g/L}$ ) to 3197  $\mu\text{g/L}$ , with a mean of 116  $\mu\text{g/L}$ . The median TP value was undetectable. There was no difference in TP concentrations in 2008 between drawdown wetlands and control wetlands, nor did TP differ in the drawdown wetlands between 2007 and 2008. TP concentrations were 1.8 times



higher in 2008 than in 2007 ( $p = 0.04$ ; Figure 4-7).

Total reactive phosphorus (TRP) concentrations ranged from undetectable levels ( $< 25 \mu\text{g/L}$ ) to  $1760 \mu\text{g/L}$ , with a mean of  $70 \mu\text{g/L}$  and median of  $40 \mu\text{g/L}$ . Concentrations were highly variable, especially in summer 2008 (Figure 4-8). Drawdown wetlands (2008) were 2.5 times higher in TRP than control wetlands (2008) ( $p = 0.0005$ ; Figure 4-9). There was no significant difference between those wetlands in 2007. The drawdown wetlands were 2.8 times higher in TRP in 2008 under drawdown conditions than in 2007 under high water conditions ( $p = 0.006$ ). There was no significant difference in TRP concentrations in the control wetlands between 2007 and 2008.

Dissolved organic carbon (DOC) concentrations ranged from  $9.3$  to  $32.5 \text{ mg/L}$ , with a mean concentration of  $14.7 \text{ mg/L}$  and a median value of  $14.0$ . There was a general trend of increasing DOC concentrations over the course of each summer, reaching a peak in late August or September (Figure 4-10). Drawdown wetlands (2008) were 1.4 times higher in DOC than control wetlands (2008) ( $p < 0.0001$ ; Figure 4-11). There was no significant difference between those wetlands in 2007. The drawdown wetlands were 1.3 times higher in DOC in 2008 under drawdown conditions than in 2007 under high water conditions. There was no significant difference in DOC concentrations in the control wetlands between 2007 and 2008. DOC in all wetlands was 1.1 times higher at vegetated sites than at open water sites ( $p < 0.0001$ ; Figure 4-12).

Concentrations of dissolved inorganic carbon (DIC) concentrations ranged from  $7.0$  to  $62.7 \text{ mg/L}$ . Mean and median concentrations were  $23.5 \text{ mg/L}$  and  $22.8 \text{ mg/L}$ , respectively. No seasonal trend in DIC was observed (Figure 4-13). There was no significant difference in DIC concentrations between drawdown and control wetlands in 2008 (Figure 4-14). Both control wetlands and drawdown wetlands were 1.1 times higher in DIC in 2008 than in 2007 ( $p < 0.0001$  and  $p = 0.002$ , respectively). DIC concentrations were 1.2 times higher at vegetated sites

than at open water sites ( $p < 0.0001$ ; Figure 4-15).

Specific conductance in study wetlands ranged from 51 to 987  $\mu\text{S}/\text{cm}$ , with a mean of 389  $\mu\text{S}/\text{cm}$  and median value of 391  $\mu\text{S}/\text{cm}$ . Many open water sites showed a trend of increasing specific conductance throughout each summer, while vegetated sites were quite variable (Figure 4-16). Drawdown wetlands (2008) had 1.1 times higher specific conductance than control wetlands (2008) ( $p = 0.001$ ; Figure 4-17). In 2007, these drawdown wetlands were significantly lower in specific conductance than control wetlands ( $p = 0.002$ ). The specific conductance of drawdown wetlands was 1.3 times higher in 2008 under drawdown conditions than in 2007 under high water conditions ( $p < 0.0001$ ). There was no significant difference in specific conductance in the control wetlands between 2007 and 2008. Specific conductance in all wetlands was 1.1 times higher at vegetated water sites than at open water sites ( $p = 0.001$ ; Figure 4-18).

Calcium was the most abundant ion measured, followed by magnesium, sodium, chloride, and potassium. Sodium concentrations ranged from 7.65 mg/L to 26.7 mg/L, with a mean of 14.8 mg/L. Potassium concentrations ranged from 0.7 mg/L to 21.1 mg/L, with a mean of 4.1 mg/L. Calcium concentrations ranged from 12.1 mg/L to 71.8 mg/L, with a mean of 39.3 mg/L. Magnesium ranged in concentration from 9.9 mg/L to 25.6 mg/L, with a mean of 14.9 mg/L. Chloride concentrations ranged from 0.9 mg/L to 49.4 mg/L, with a mean of 13.4 mg/L. Mean concentrations of sodium, potassium, calcium, magnesium and chloride in each study wetland are shown in Tables 4-4 through 4-8.

Drawdown wetlands in 2008 were significantly higher in chloride (1.9 times,  $p = 0.01$ ; Figure 4-19), sodium (1.5 times,  $p = 0.0005$ ), and potassium (2.1 times,  $p < 0.0001$ ) than control wetlands in 2008. There was no significant difference in the concentration of chloride, sodium, or potassium ions between those drawdown wetlands in 2007.

The drawdown wetlands were significantly higher in chloride (2.4 times,  $p < 0.0001$ ; Figure 4-19), sodium (2.0 times,  $p < 0.0001$ ), and potassium (2.0 times,  $p < 0.0001$ ) in 2008 under drawdown conditions than in 2007 under high water conditions. There was no significant difference in chloride or potassium concentrations in the control wetlands between 2007 and 2008. There was a significant difference in sodium concentration in the control wetlands between 2007 and 2008 ( $p = 0.007$ ): in 2008 control wetlands had 24% higher sodium concentrations than in 2007, whereas the sodium concentration in drawdown wetlands doubled over this same period.

Magnesium and calcium concentrations were not significantly different between drawdown and control wetlands in 2008, nor did these concentrations differ in the drawdown wetlands between 2007 and 2008. Magnesium and calcium concentrations in all wetlands were significantly higher in 2008 than in 2007 (1.3 times,  $p < 0.0001$  and 1.2 times,  $p = 0.005$ , respectively; Figure 4-20).

Calcium concentrations in all wetlands were 1.2 times higher at vegetated water sites than at open water sites ( $p = 0.0005$ ; Figure 4-21). Other ion concentrations were not significantly different between open water and vegetated sites.

Turbidity was highly variable, especially in drawdown wetlands; no seasonal trends were observed (Figure 4-22). Values for turbidity ranged from 0.08 NTU to 324 NTU, with a mean of 12.96 NTU and a median of 4.00 NTU. Drawdown wetlands (2008) were 3.5 times higher in turbidity than control wetlands (2008) ( $p < 0.0001$ ; Figure 4-23). There was no significant difference between those wetlands in 2007. The turbidity of drawdown wetlands increased more than fourfold from 2007 to 2008 ( $p < 0.0001$ ). Control wetlands also increased in turbidity in this period ( $p = 0.02$ ), but by only 95%.

Phytoplankton ranged from 0.3 µg/L chlorophyll α to 2943 µg/L chlorophyll α, with a mean value of 61.3 µg/L chlorophyll α and a median of 0.7 µg/L chlorophyll α. Study wetlands at each sampling period are shown in Figure 4-24. Phytoplankton chlorophyll was variable, especially in drawdown wetlands and at vegetated sites, but most wetlands reached a peak in late July or early August. There was no significant difference in phytoplankton biomass (measured as µg/L chlorophyll α) between drawdown wetlands and control wetlands, either in 2007 or 2008. Phytoplankton biomass was 5.4 times higher at vegetated water sites than at open water sites ( $p < 0.0001$ ).

There was significantly more periphyton on artificial surfaces in the drawdown wetlands in 2008 as compared with those same wetlands in 2007 ( $p < 0.0001$ ; Figure 4-26), and as compared with the control wetlands in 2008 ( $p = 0.0008$ ; Figure 4-27). However, there was also significantly more algal growth in the control wetlands in 2008 as compared with the same wetlands in 2007 ( $p = 0.04$ ). Growth in the control wetlands increased by only 0.71 µg/cm<sup>2</sup>, or 87%, while growth in the drawdown wetlands increased by 1.45 µg/cm<sup>2</sup> or nearly two and a half times. Overall there was 2.2 times more periphyton on artificial surfaces in low water wetlands than in high water wetlands ( $p < 0.0001$ ; Figure 4-27). Periphyton chlorophyll did not show a significant monthly trend.

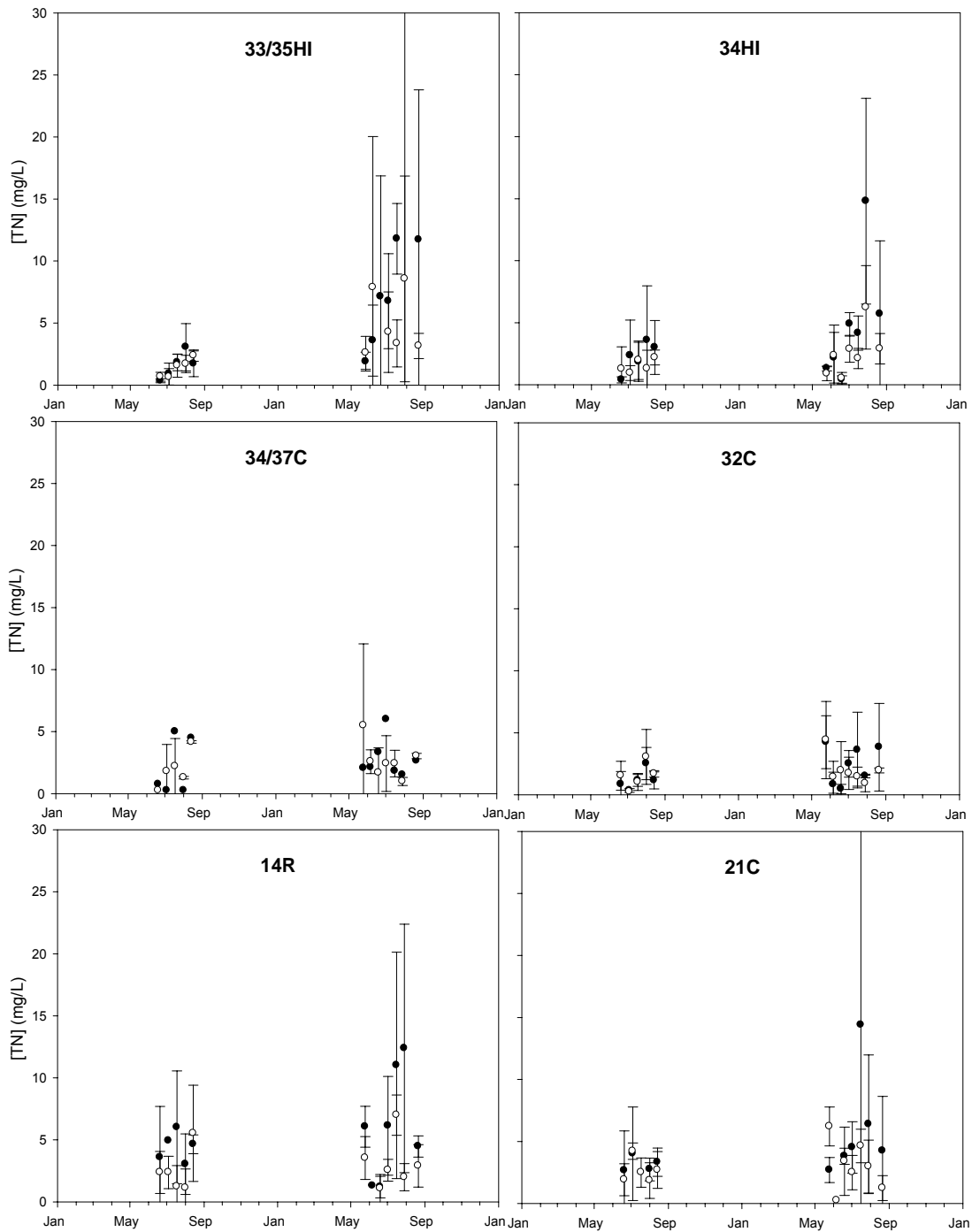


Figure 4-2: Total nitrogen concentrations (mg/L) in each of the six study wetlands over the summers of 2007 and 2008. Each white circle represents the mean of open water sites (n = 3); each black circle represents the mean of vegetated / sheltered sites (n = 3). Error bars show standard deviation. Drawdown wetlands are on the left; control wetlands, on the right.

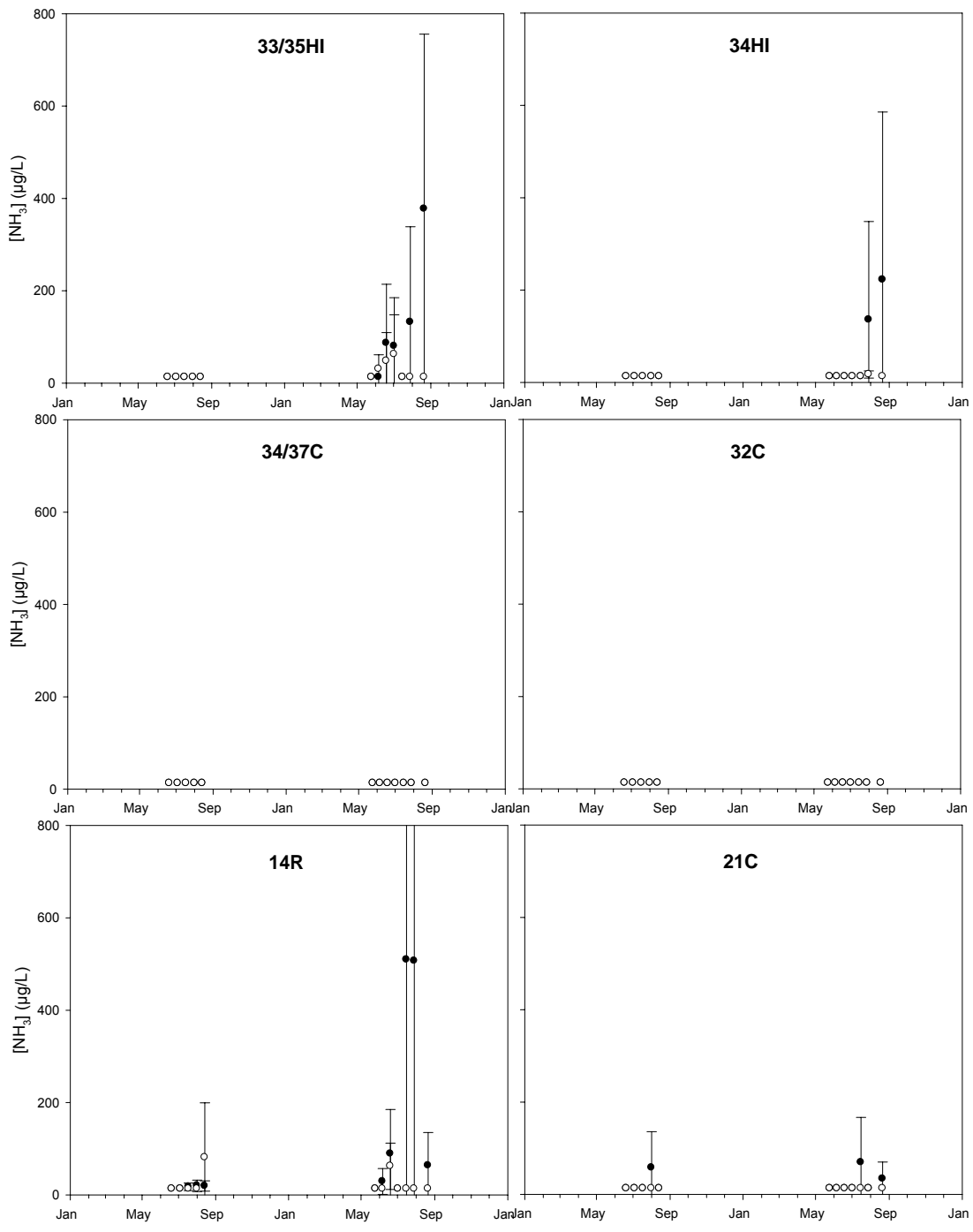


Figure 4-4: Ammonia concentrations ( $\mu\text{g/L}$ ) in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.

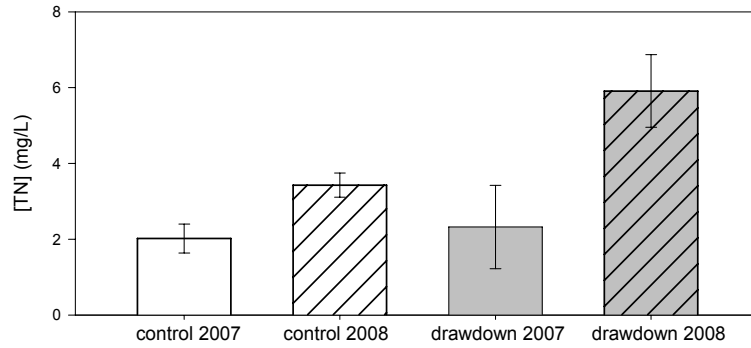


Figure 4-3: Total nitrogen concentrations (mg/L) in control wetlands in 2007 (n = 85), control wetlands in 2008 (n = 122), drawdown wetlands in 2007 (n = 72) and drawdown wetlands in 2008 (n = 95). Error bars show standard error.

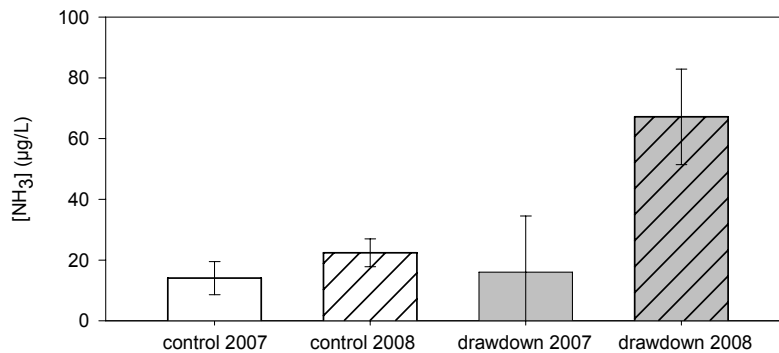


Figure 4-5: Ammonia concentrations (µg/L) in control wetlands in 2007 (n = 90), control wetlands in 2008 (n = 126), drawdown wetlands in 2007 (n = 75) and drawdown wetlands in 2008 (n = 104). Error bars show standard error.

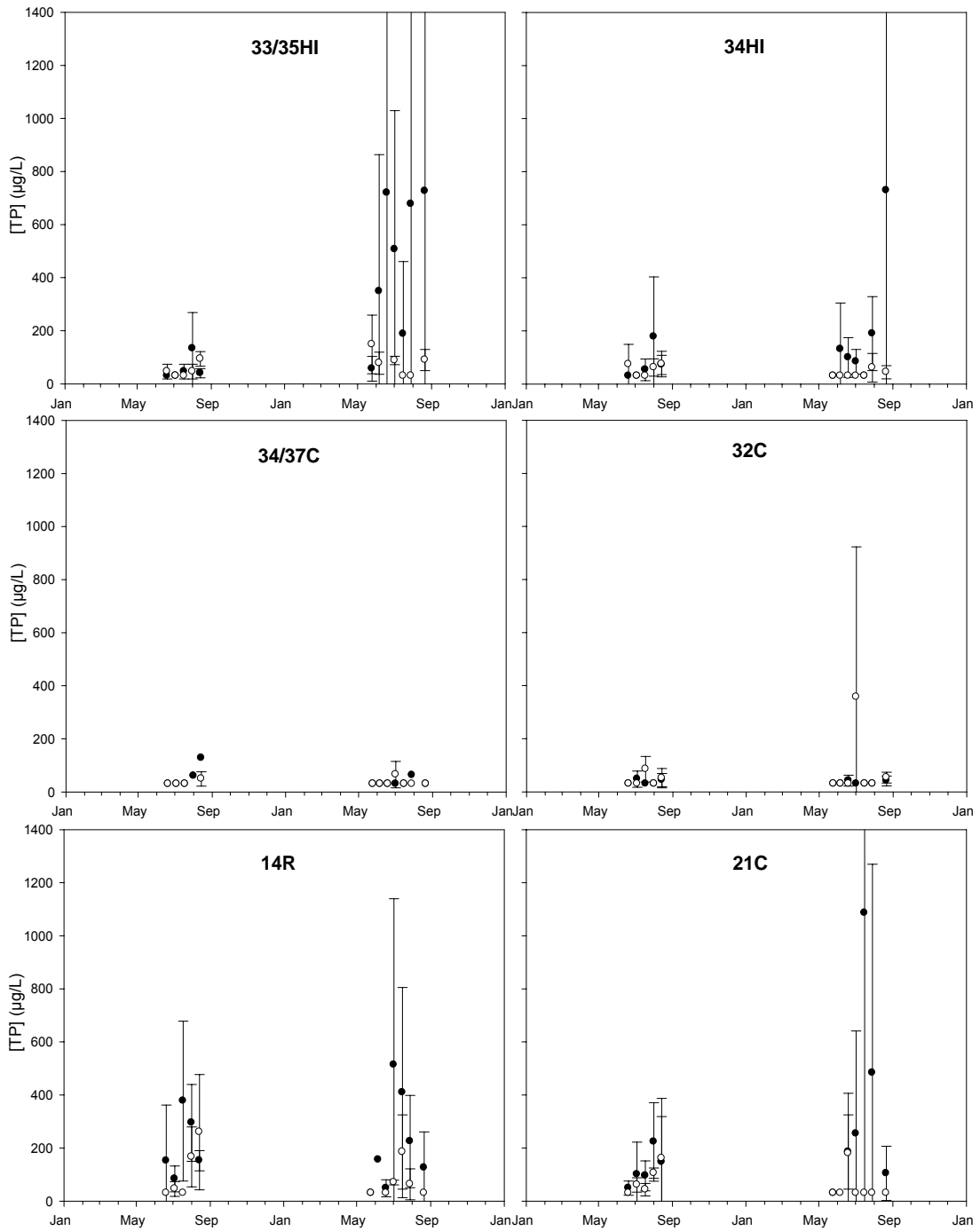


Figure 4-6: Total phosphorus concentrations ( $\mu\text{g/L}$ ) in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.



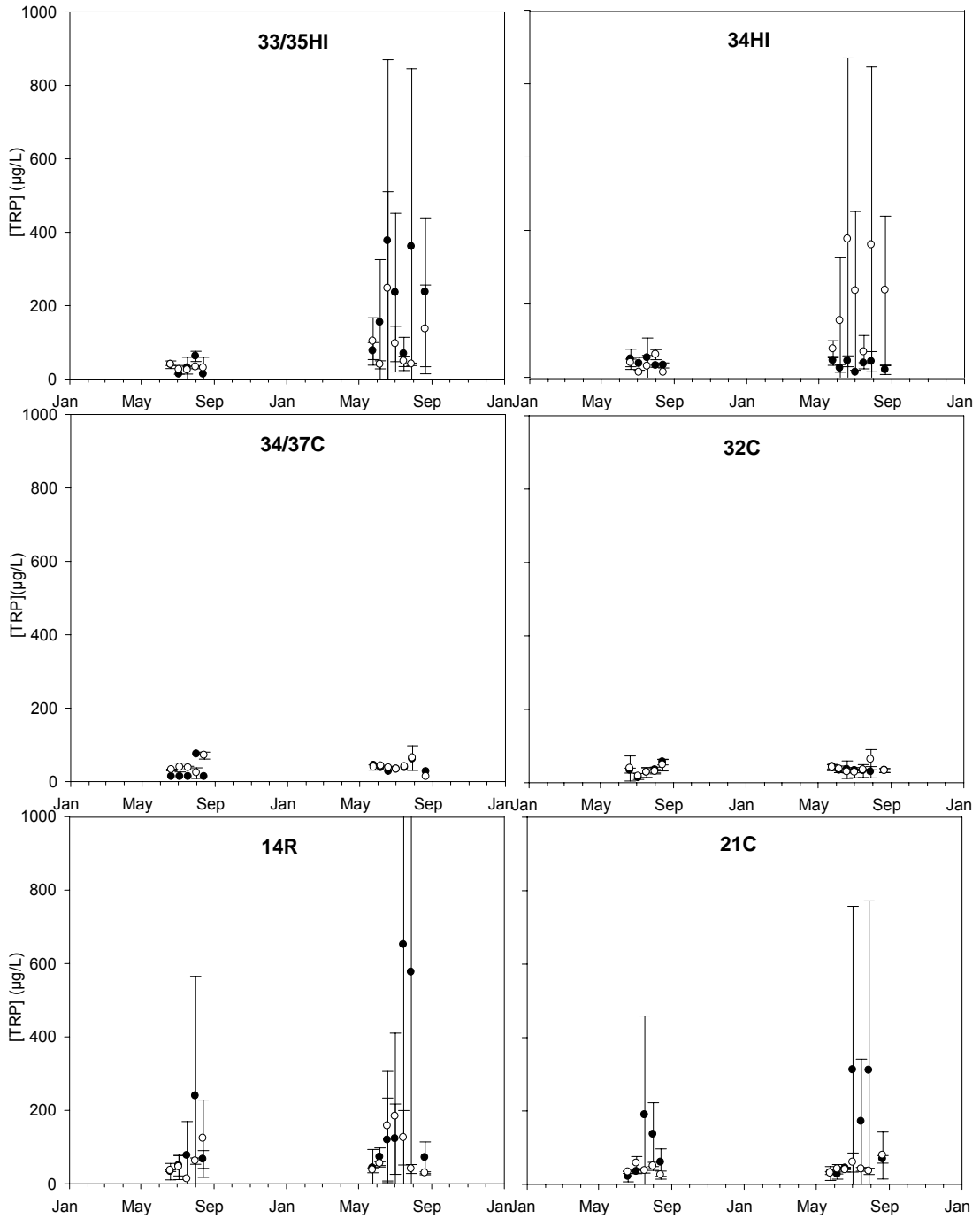


Figure 4-8: Total reactive phosphorus concentrations ( $\mu\text{g/L}$ ) in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.

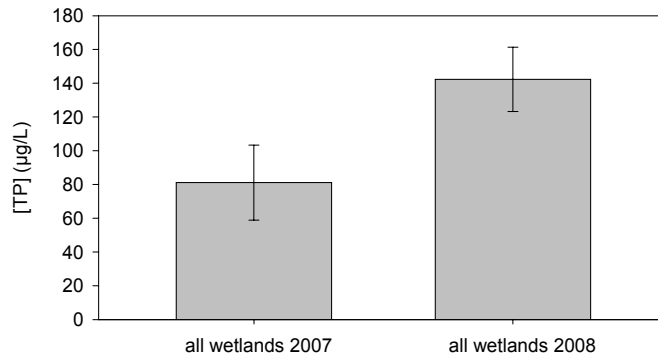


Figure 4-7: Total phosphorus concentrations ( $\mu\text{g/L}$ ) in all wetlands in 2007 ( $n = 160$ ) and 2008 ( $n = 217$ ). Error bars show standard error.

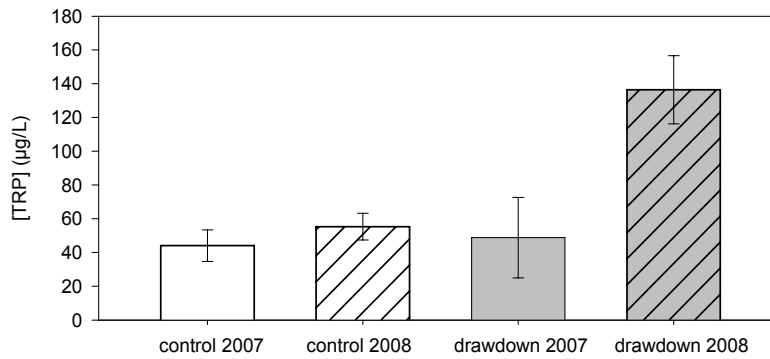


Figure 4-9: Total reactive phosphorus concentrations ( $\mu\text{g/L}$ ) in control wetlands in 2007 ( $n = 90$ ), control wetlands in 2008 ( $n = 126$ ), drawdown wetlands in 2007 ( $n = 75$ ) and drawdown wetlands in 2008 ( $n = 104$ ). Error bars show standard error.

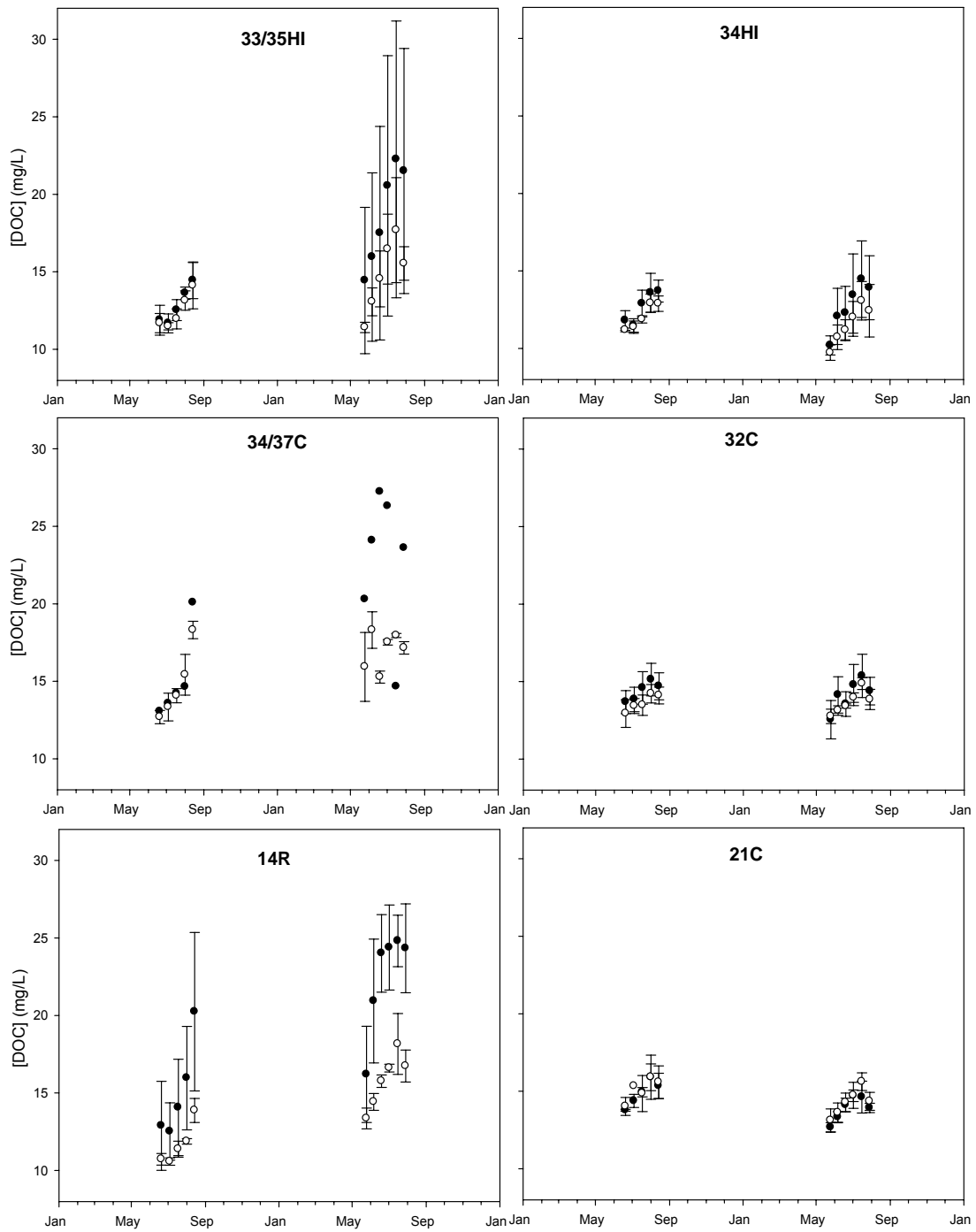


Figure 4-10: Dissolved organic carbon (mg/L) in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.

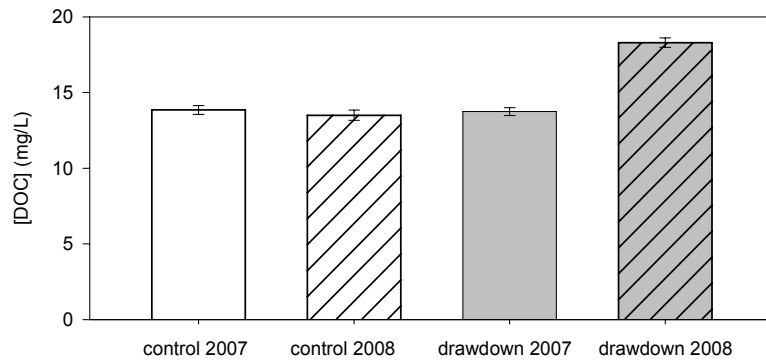


Figure 4-11: Dissolved organic carbon concentrations (mg/L) in control wetlands in 2007 (n = 102), control wetlands in 2008 (n = 125), drawdown wetlands in 2007 (n = 75) and drawdown wetlands in 2008 (n = 87). Error bars show standard error.

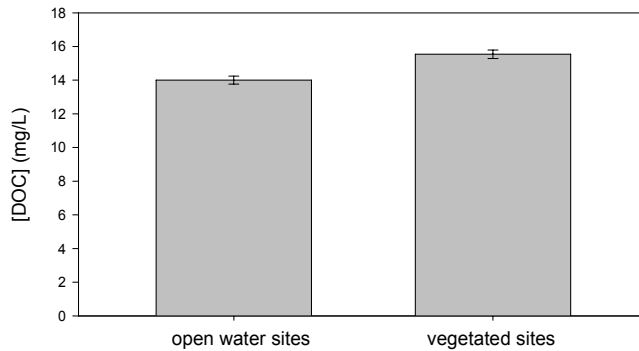


Figure 4-12: Dissolved organic carbon concentrations (mg/L) at open water sites (n = 204) and vegetated sites (n = 185). Error bars show standard error.

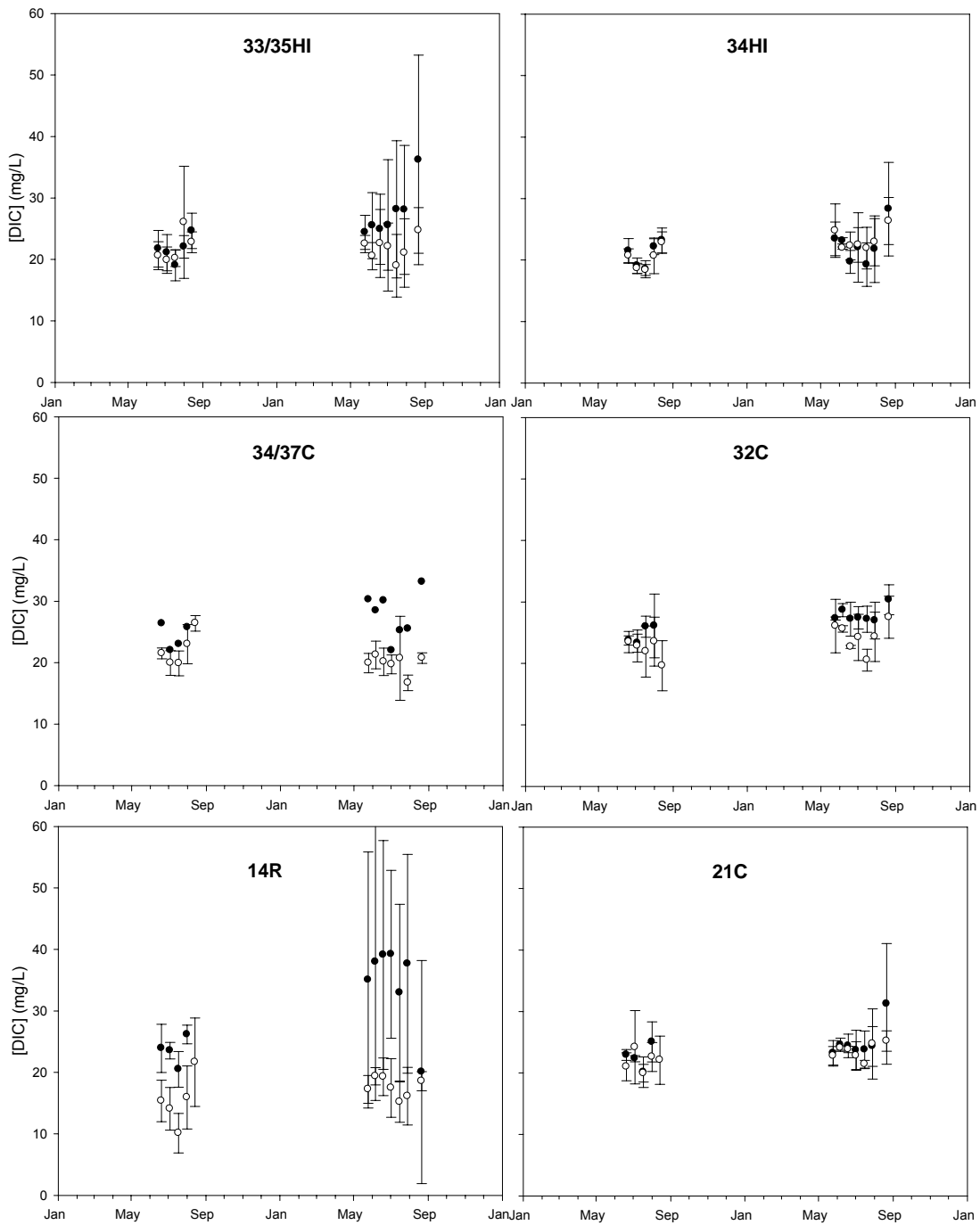


Figure 4-13: Dissolved inorganic carbon concentrations (mg/L) in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.

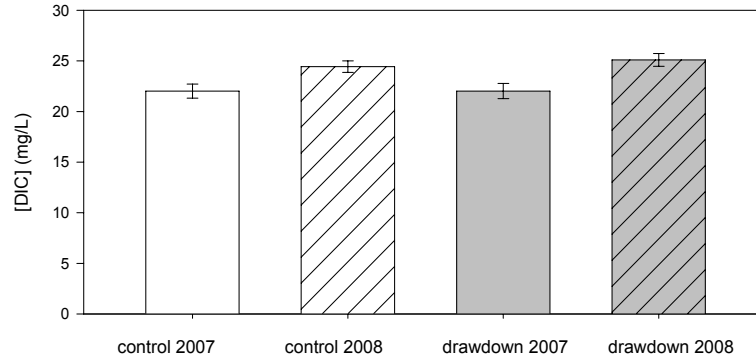


Figure 4-14: Dissolved organic carbon concentrations (mg/L) in control wetlands in 2007 (n = 84), control wetlands in 2008 (n = 126), drawdown wetlands in 2007 (n = 71) and drawdown wetlands in 2008 (n = 104). Error bars show standard error.

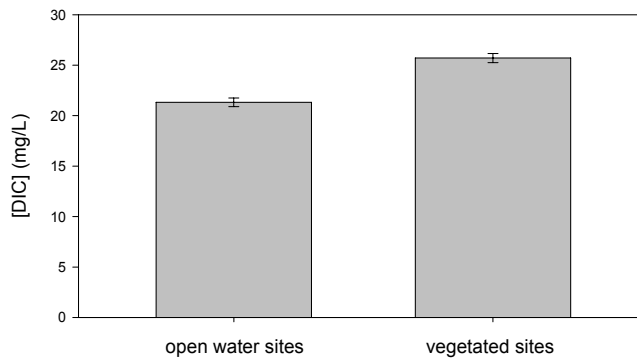


Figure 4-15: Dissolved inorganic carbon concentrations at open water sites (n = 205) and vegetated sites (n = 184) in all wetlands in 2007 and 2008. Error bars show standard error.

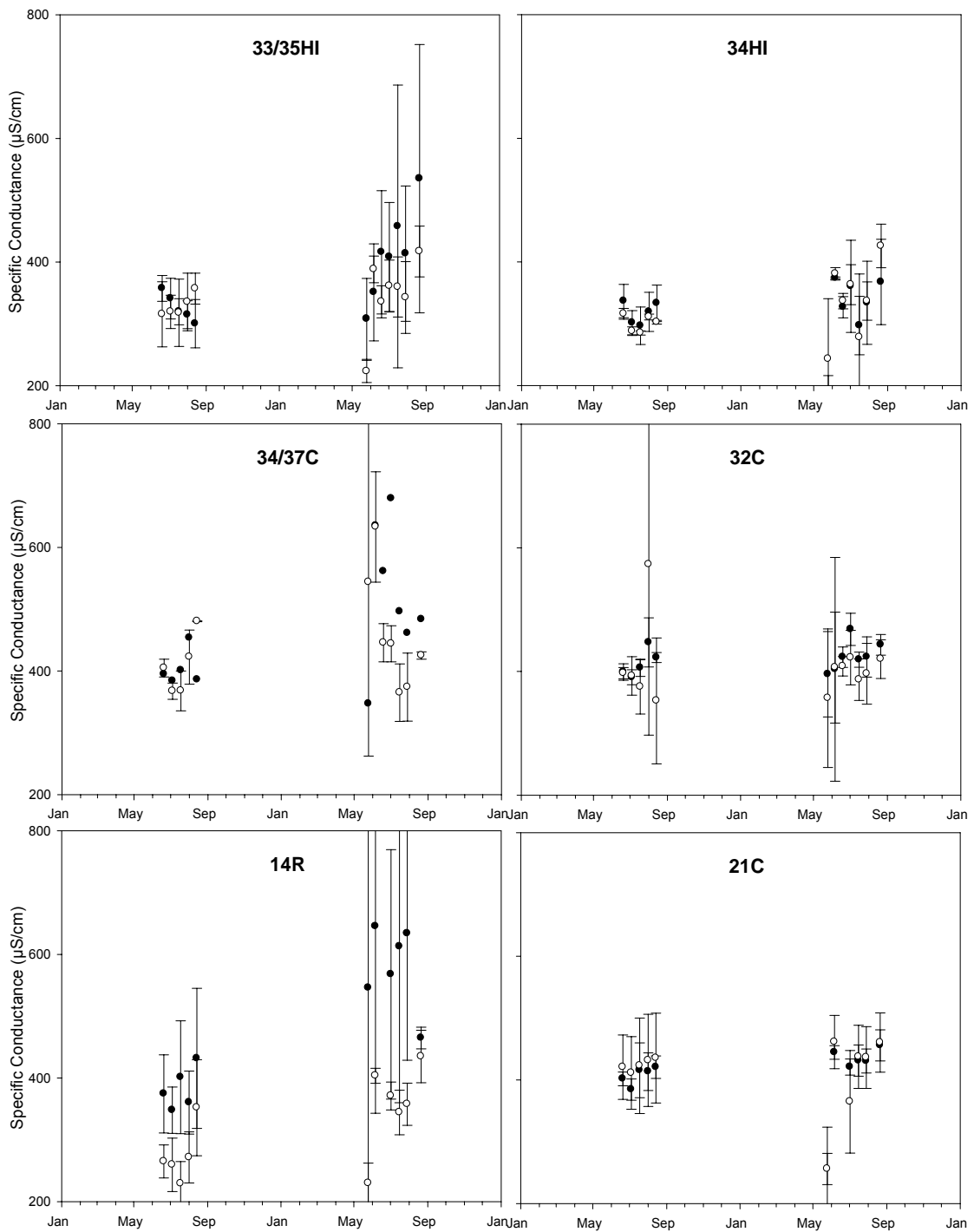


Figure 4-16: Specific conductance in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.

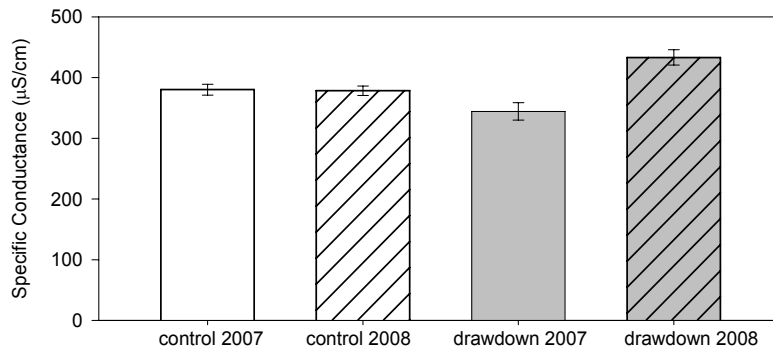


Figure 4-17: Specific conductance in control wetlands ( $\mu\text{S}/\text{cm}$ ) in 2007 ( $n = 90$ ), control wetlands in 2008 ( $n = 120$ ), drawdown wetlands in 2007 ( $n = 75$ ) and drawdown wetlands in 2008 ( $n = 98$ ). Error bars show standard error.

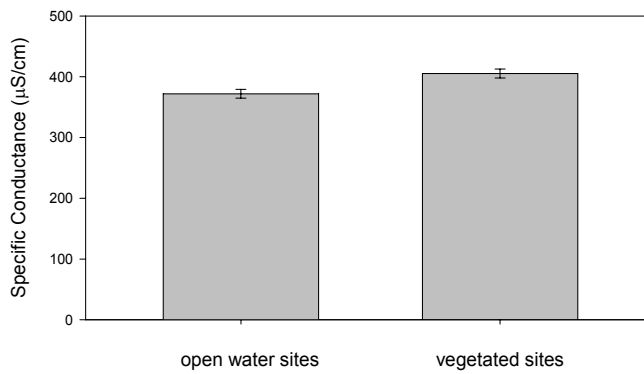


Figure 4-18: Specific conductance ( $\mu\text{S}/\text{cm}$ ) at open water sites ( $n = 209$ ) and vegetated sites ( $n = 207$ ) in all wetlands in 2007 and 2008. Error bars show standard error.



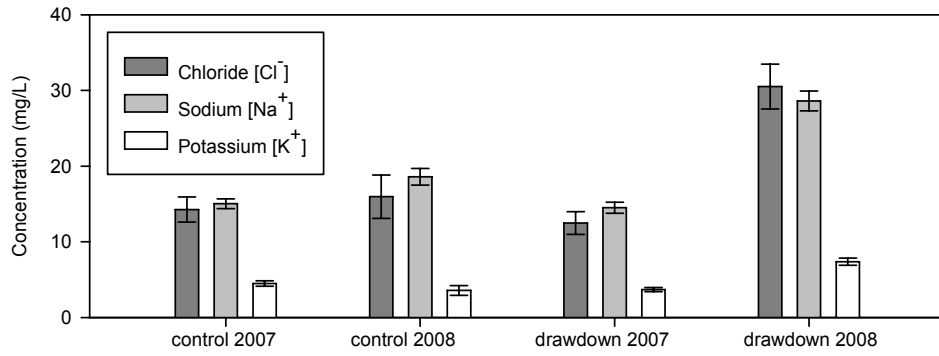


Figure 4-19: Chloride, sodium and potassium concentrations (mg/L) in control wetlands in 2007 (n = 54), control wetlands in 2008 (n = 18), drawdown wetlands in 2007 (n = 45) and drawdown wetlands in 2008 (n = 14). Error bars show standard error.

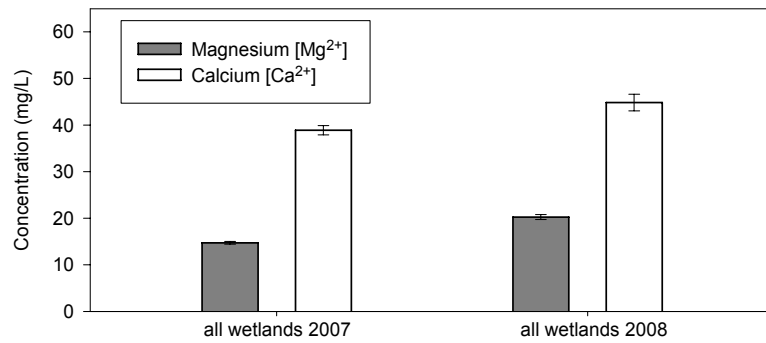


Figure 4-20: Magnesium and calcium concentrations (mg/L) in all wetlands in 2007 (n = 99) and 2008 (n = 32). Error bars show standard error.

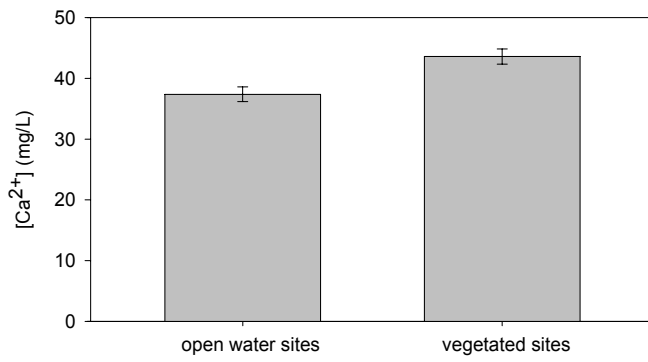


Figure 4-21: Calcium concentrations (mg/L) at open water sites (n = 68) and vegetated sites (n = 63) in all wetlands in 2007 and 2008. Error bars show standard error.

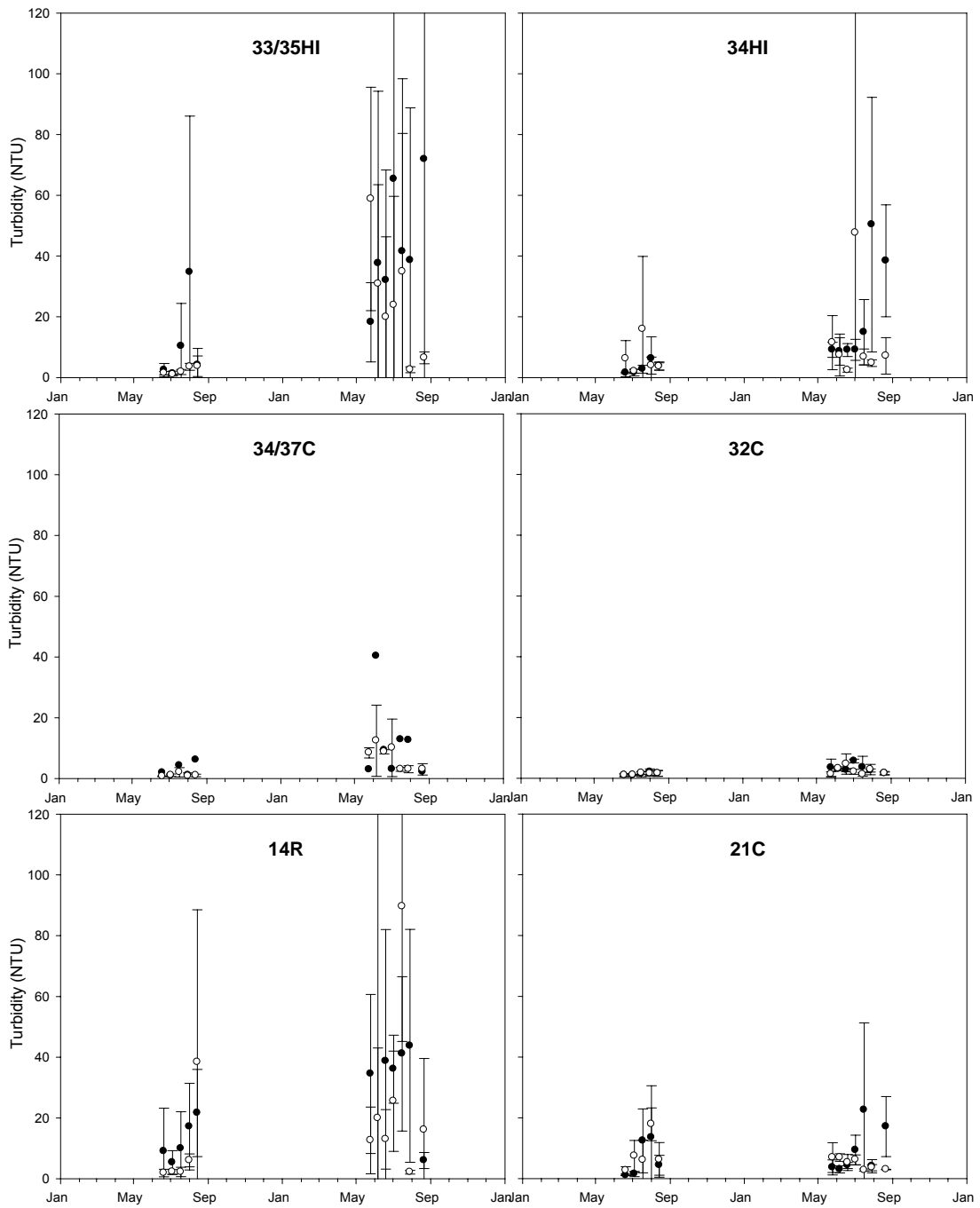


Figure 4-22: Turbidity (NTU) in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.

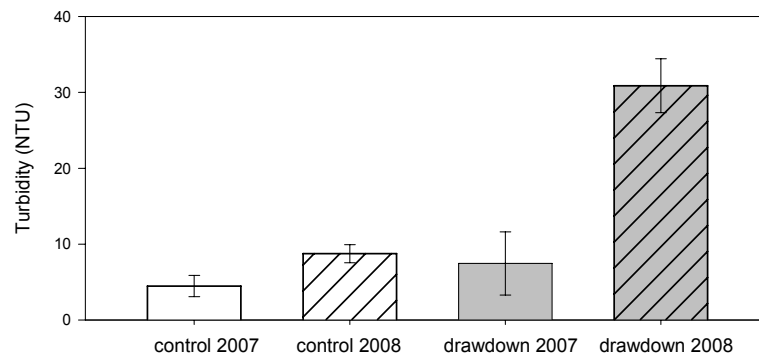


Figure 4-23: Turbidity (NTU) in control wetlands in 2007 (n = 90), control wetlands in 2008 (n = 125), drawdown wetlands in 2007 (n = 75) and drawdown wetlands in 2008 (n = 104). Error bars show standard error.

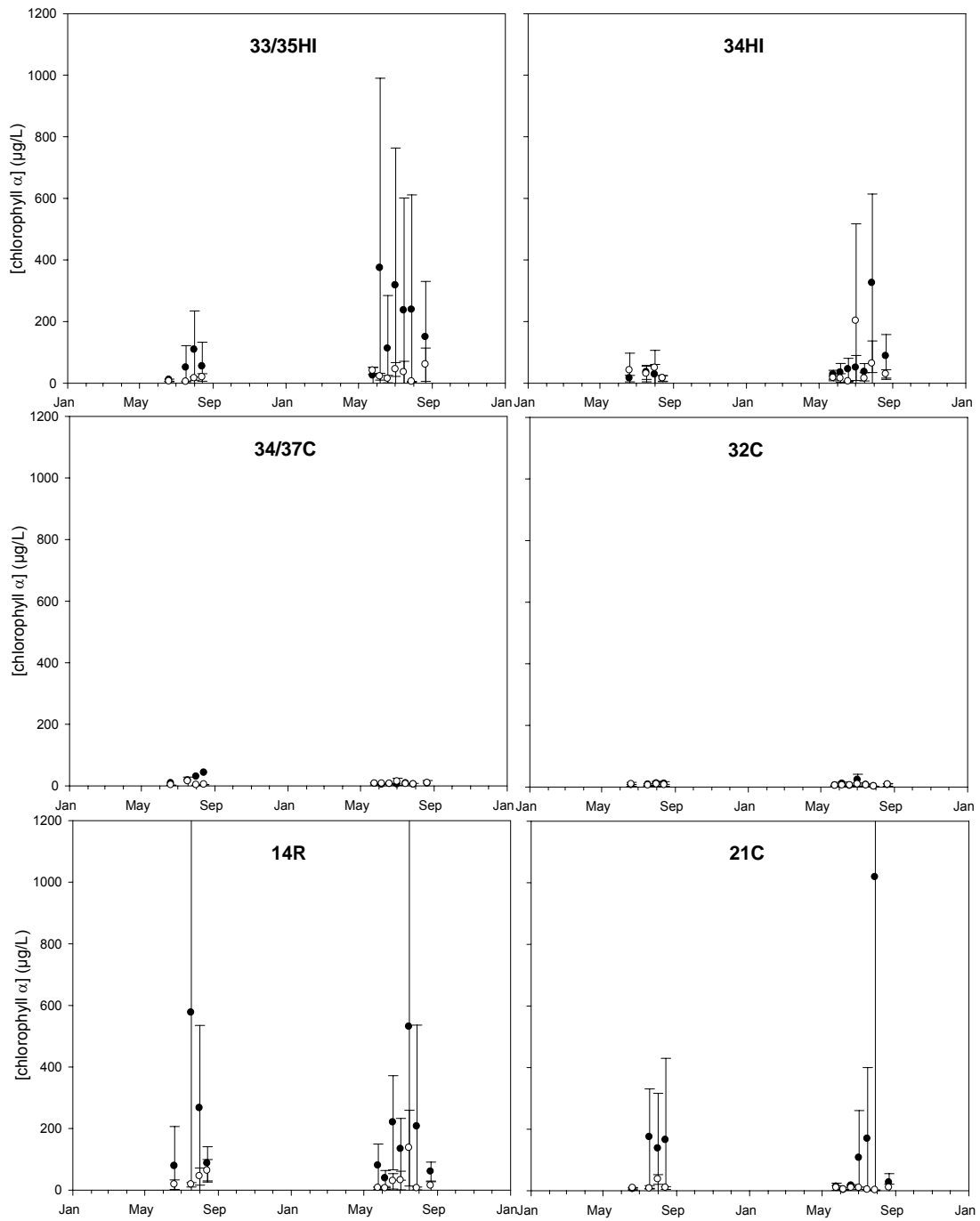


Figure 4-24: Phytoplankton biomass, expressed as µg/L chlorophyll α, in each of the six study wetlands over the summers of 2007 and 2008. See Figure 4-3 for legend.

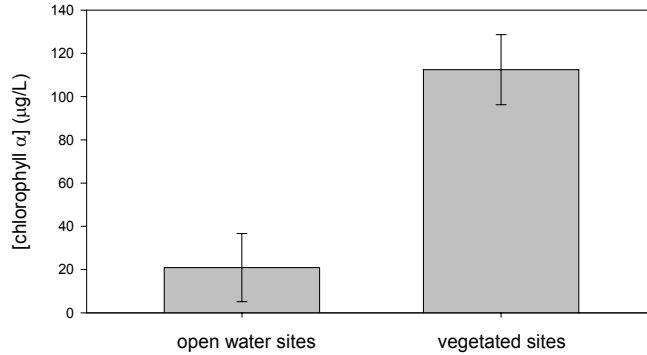


Figure 4-25: Phytoplankton biomass, represented by chlorophyll  $\alpha$  concentration ( $\mu\text{g/L}$ ), at open water sites ( $n = 186$ ) and vegetated sites ( $n = 175$ ) in all wetlands in 2007 and 2008. Error bars show standard error.

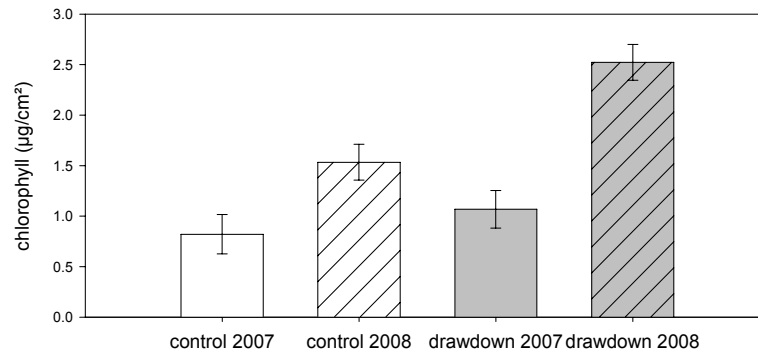


Figure 4-26: Periphyton biomass, expressed as chlorophyll ( $\mu\text{g/cm}^2$ ), in control wetlands in 2007 ( $n = 8$ ), control wetlands in 2008 ( $n = 9$ ), drawdown wetlands in 2007 ( $n = 9$ ) and drawdown wetlands in 2008 ( $n = 9$ ), over a two-week duration, on surfaces not enriched by nutrients. Error bars show standard error.

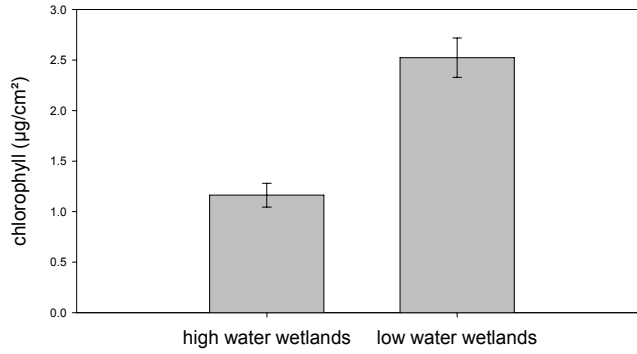


Figure 4-27: Periphyton biomass, expressed as chlorophyll ( $\mu\text{g}/\text{cm}^2$ ), in high water and low water wetlands over a two-week duration. Error bars show standard error.

Table 4-3: Formulae for calculation sediment composition parameters by loss on ignition.

Parameter	Formula
Wet Density	$\frac{\text{wet weight}}{\text{wet volume}}$
% Water Content	$\frac{\text{wet weight} - \text{weight after drying at } 100^{\circ}\text{C}}{\text{wet weight}}$
% Organic Content	$\frac{\text{weight after drying at } 100^{\circ}\text{C} - \text{weight after heating to } 550^{\circ}\text{C}}{\text{weight after drying at } 100^{\circ}\text{C}}$
% Carbonate Content	$\frac{\text{weight after heating to } 550^{\circ}\text{C} - \text{weight after heating to } 900^{\circ}\text{C}}{\text{weight after heating to } 550^{\circ}\text{C}}$

Table 4-4: Mean sodium concentrations (mg/L) in open water sites and vegetated / sheltered sites in summers 2007 and 2008  $\pm$  standard deviation.

	2007		2008	
	Open Water	Vegetated	Open Water	Vegetated
33/35HI	12.8 $\pm$ 1.3	13.3 $\pm$ 1.2	20.1 $\pm$ 2.1	31.5 $\pm$ 18.9
34HI	9.1 $\pm$ 1.5	10.7 $\pm$ 1.6	11.6 $\pm$ 0.8	14.1 $\pm$ 3.6
34/37C	17.2 $\pm$ 2.8	15.8 $\pm$ 1.1	26.7 $\pm$ 7.3	21.4 $\pm$ 2.6
32C	15.8 $\pm$ 1.2	16.4 $\pm$ 1.2	20.4 $\pm$ 2.8	20.7 $\pm$ 0.8
14R	14.7 $\pm$ 0.6	13.9 $\pm$ 2.0	29.6 $\pm$ 1.0	35.5 $\pm$ 2.1
21C	20.3 $\pm$ 4.6	18.0 $\pm$ 0.7	23.6 $\pm$ 6.0	21.0 $\pm$ 0.8



Table 4-5: Mean potassium concentrations (mg/L) in open water sites and vegetated / sheltered sites in summers 2007 and 2008.

	2007		2008	
	Open Water	Vegetated	Open Water	Vegetated
33/35HI	3.5 ± 1.4	3.6 ± 1.1	7.9 ± 3.6	6.6 ± 3.2
34HI	1.9 ± 1.0	2.2 ± 1.0	1.5 ± 0.8	2.0 ± 1.0
34/37C	4.3 ± 0.5	3.9 ± 0.5	4.5 ± 0.8	5.8 ± 1.9
32C	5.3 ± 0.5	5.6 ± 1.0	4.1 ± 0.3	4.1 ± 0.6
14R	3.3 ± 0.9	3.8 ± 1.7	9.6 ± 0.3	9.0 ± 4.7
21C	4.7 ± 0.4	7.2 ± 5.7	4.6 ± 1.1	5.2 ± 0.7

Table 4-6: Mean calcium concentrations (mg/L) in open water sites and vegetated / sheltered sites in summers 2007 and 2008.

	2007		2008	
	Open Water	Vegetated	Open Water	Vegetated
33/35HI	35.7 ± 6.5	35.9 ± 7.1	42.5 ± 10.9	56.4 ± 22.6
34HI	36.9 ± 3.5	37.4 ± 5.7	52.2 ± 5.4	53.3 ± 18.7
34/37C	41.3 ± 6.9	44.5 ± 3.3	34.7 ± 9.5	46.9 ± 4.2
32C	38.5 ± 8.1	43.2 ± 5.9	43.1 ± 10.0	47.2 ± 6.2
14R	25.8 ± 13.9	47.1 ± 13.1	25.6 ± 4.9	47.8 ± 2.1
21C	42.4 ± 4.9	43.2 ± 3.9	43.7 ± 4.4	47.5 ± 4.9

Table 4-7: Mean magnesium concentrations (mg/L) in open water sites and vegetated / sheltered sites in summers 2007 and 2008.

	2007		2008	
	Open Water	Vegetated	Open Water	Vegetated
33/35HI	13.9 ± 1.0	14.1 ± 1.1	19.9 ± 0.6	26.1 ± 9.9
34HI	11.9 ± 1.0	12.6 ± 1.1	16.1 ± 1.0	18.2 ± 4.1
34/37C	16.6 ± 1.5	16.4 ± 0.8	21.2 ± 1.8	20.6 ± 2.2
32C	16.1 ± 0.4	16.7 ± 1.1	21.0 ± 1.2	21.5 ± 0.7
14R	11.7 ± 1.3	15.4 ± 4.6	16.4 ± 0.8	23.4 ± 1.8
21C	16.7 ± 1.0	16.3 ± 0.6	19.7 ± 1.1	19.6 ± 0.5

Table 4-8: Mean chloride concentrations (mg/L) in open water sites and vegetated / sheltered sites in summers 2007 and 2008.

	2007		2008	
	Open Water	Vegetated	Open Water	Vegetated
33/35HI	6.3 ± 1.2	6.5 ± 1.2	12.0 ± 1.0	21.3 ± 20.4
34HI	2.4 ± 1.5	3.1 ± 1.5	2.5 ± 0.6	3.4 ± 0.5
34/37C	17.8 ± 6.7	14.8 ± 0.9	30.4 ± 17.3	21.1 ± 6.5
32C	12.0 ± 0.9	12.4 ± 1.0	13.1 ± 1.6	13.0 ± 1.2
14R	14.4 ± 1.3	15.4 ± 3.3	40.9 ± 0.3	48.2 ± 31.1
21C	30.5 ± 12.6	25.1 ± 2.2	35.1 ± 14.3	28.6 ± 1.7

## Depth Relationships

Most water quality parameters – all measured nutrients, turbidity, phytoplankton, conductivity, and the ions calcium, magnesium, and sodium – were correlated with depth by an exponential relationship described  $y = ax^{-b}$  (Figures 4-28 through 4-30). Shallow depths corresponded to the highest concentrations and turbidities. Values decreased sharply with initial increases in depth, but after a certain threshold was reached, further increases in depth had little or no effect on these variables. The threshold level for nutrients (carbon, nitrogen, phosphorus) and turbidity appeared by visual inspection to be approximately 40cm. The threshold for ions, conductivity and sodium appeared to be smaller, at approximately 20cm. There was no significant relationship between depth and chloride or potassium.

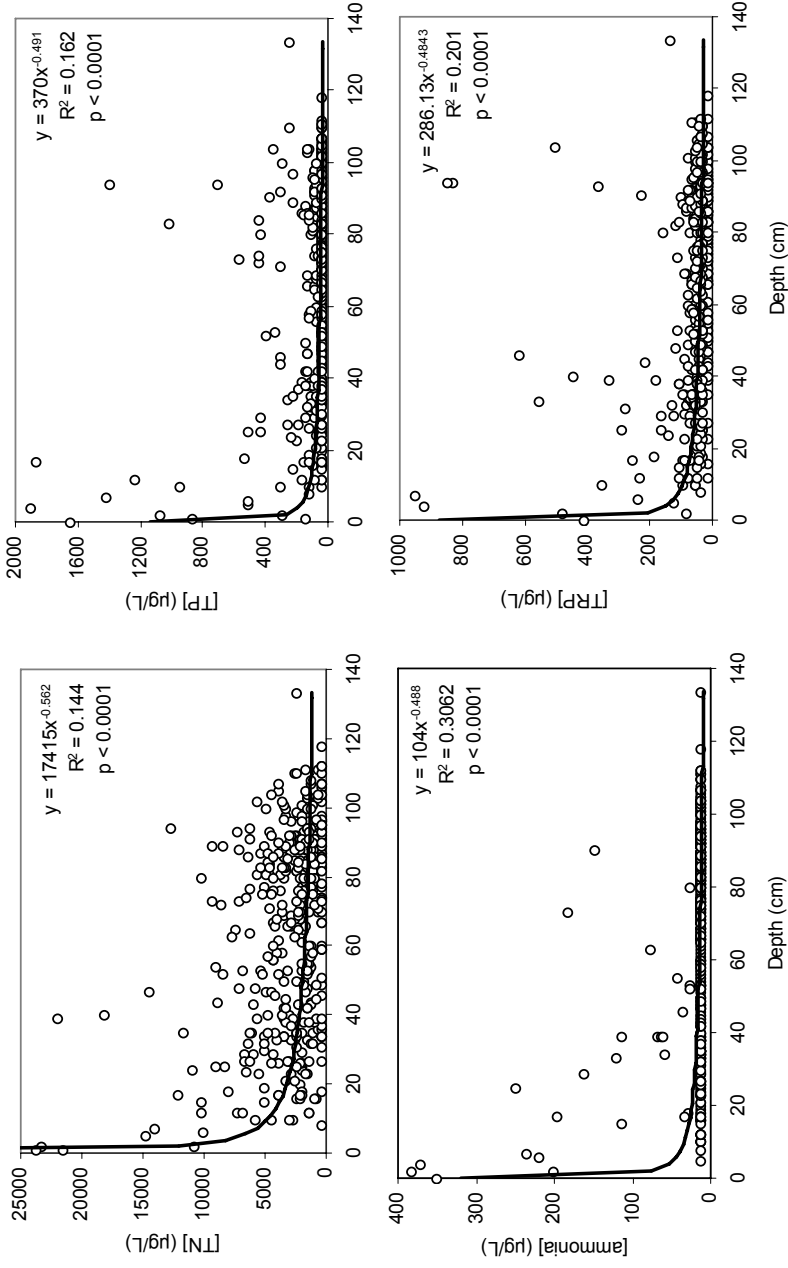


Figure 4-28: Relationships between nutrient concentration and site depth. Clockwise from top left: total nitrogen (n = 403), total phosphorus (n = 406), total reactive phosphorus (n = 414), and ammonia (n = 424). A minimum of 98% of points are displayed (some high concentrations at shallow depths were outside the range of the y axis).

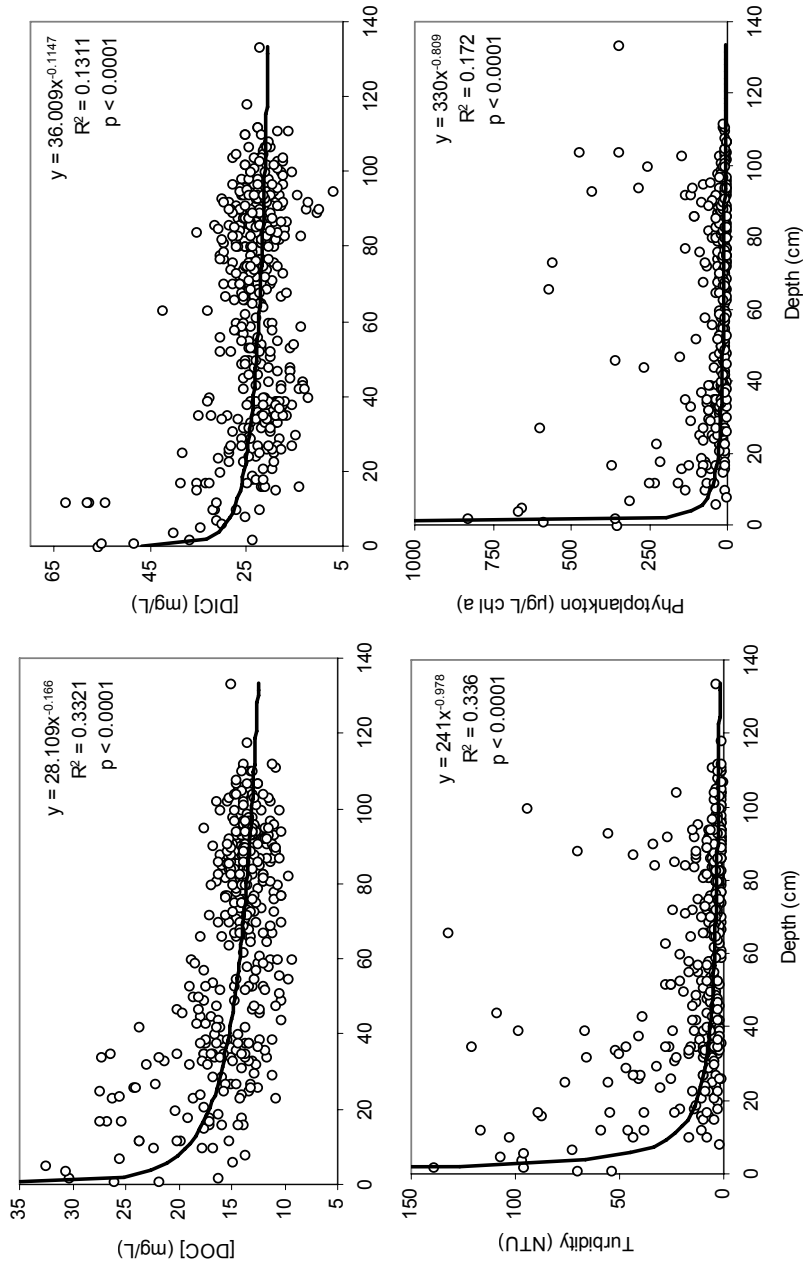


Figure 4-29: Relationships between water quality parameters and site depth. Clockwise from top left: dissolved organic carbon (n = 383), dissolved inorganic carbon (n = 414), phytoplankton chlorophyll (n = 388), and turbidity (n = 424). A minimum of 98% of points are displayed (some high concentrations at shallow depths were outside the range of the y axis).

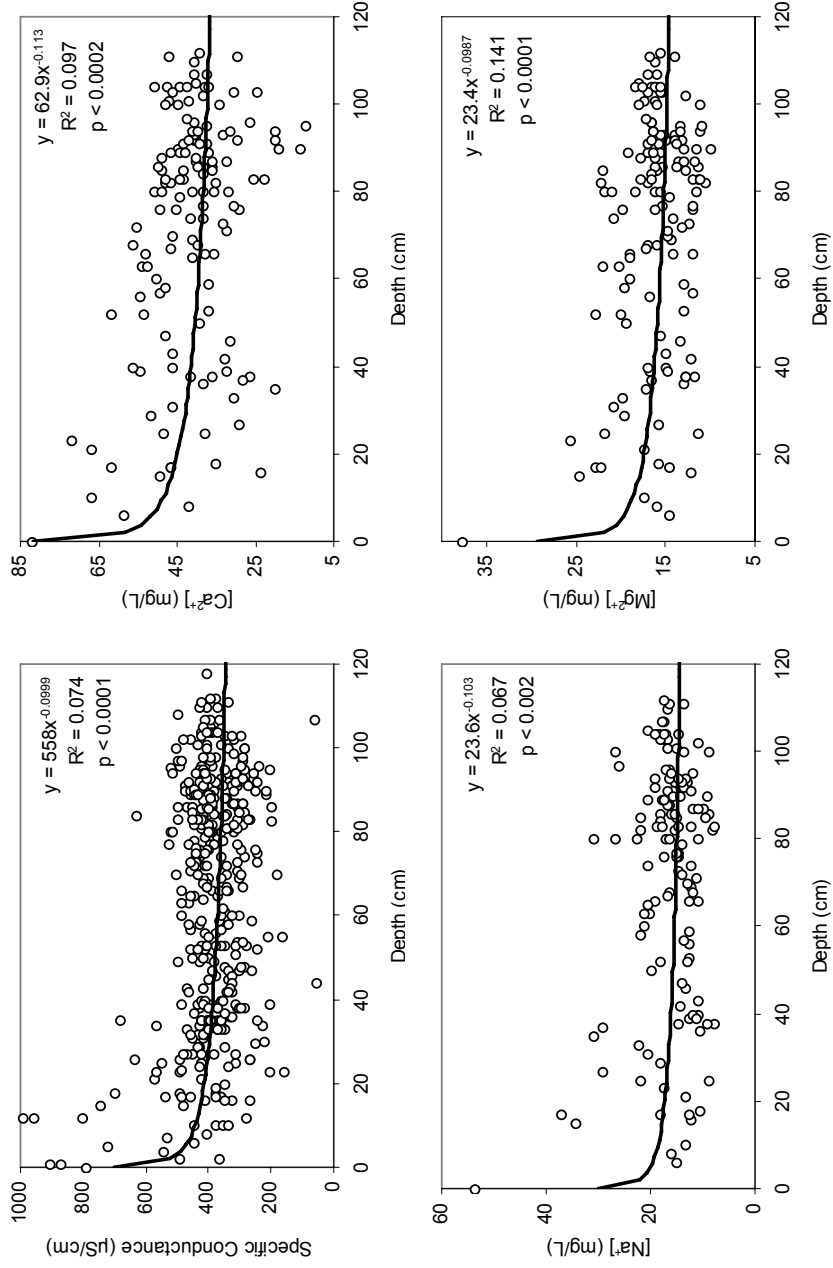


Figure 4-30: Relationships between water quality parameters and site depth. Clockwise from top left: specific conductance (n = 412), calcium ion concentration (n = 139), magnesium ion concentration (n = 139), and sodium ion concentration (n = 139). A minimum of 98% of datapoints are displayed (some high concentrations at shallow depths were outside the range of the y axis).



### **River and Back Channel Water Quality**

Mean values for all water quality parameters measured in the Saskatchewan River and back channel are presented in Table 4-9. The concentrations of many nutrients appeared higher in the river than in the back channel. Some nutrient concentrations were higher in 2007 than in 2008. Water clarity was greater in the back channel and especially in the river in 2008 as compared with 2007. The concentration of many ions was lower in 2008 than in 2007.

There was no significant difference between wetland sites and channel sites in TN concentration. The river was significantly higher in TN than the back channel ( $p = 0.03$ ; Figure 4-31). There was no significant difference in TN at pooled channel sites between 2007 and 2008. The back channel, however, was significantly higher in TN in 2007 than in 2008 ( $p = 0.04$ ; Figure 4-32).

Ammonia concentration was not significantly different between wetland sites and channel sites, between the river and back channel, or in channel sites between 2007 and 2008 (Table 4-9).

There was no significant difference between wetland sites and channel sites in TP concentration. The river was significantly higher in TP than the back channel ( $p < 0.05$ ; Figure 4-33). There was no significant difference in TP at pooled channel sites between 2007 and 2008. The river, however, was lower in TP in 2008 than in 2007 ( $p = 0.03$ ; Figure 4-34).

There was no significant difference in TRP concentration between wetland sites and channel sites. The river was significantly higher in TRP than the back channel ( $p = 0.01$ ; Figure 4-35). There was no significant difference in TRP at channel sites between 2007 and 2008.

Channel sites were significantly lower than wetlands sites in DOC concentration ( $p < 0.0001$ ; Figure 4-36). There was no significant difference in DOC between the river and back channel. DOC at channel sites was significantly higher in 2007 than in 2008 ( $p < 0.0001$ ; Figure 4-37).

Channel sites were significantly higher than wetlands sites in DIC concentration ( $p < 0.0001$ ; Figure 4-38). There was no significant difference in DIC between the river and back channel, or at channel sites between 2007 and 2008.

Specific conductance was not significantly different between wetland sites and channel sites, or between the river and back channel. There was no significant difference in specific conductance at channel sites between 2007 and 2008 (Table 4-9).

There was no significant difference between wetland sites and channel sites, or between the river and back channel, in any ion concentrations. The channel sites were significantly lower in chloride ( $p < 0.0001$ ), sodium ( $p = 0.04$ ) and potassium ( $p = 0.002$ ) in 2008 than in 2007 (Figure 4-39). There were no significant differences at channel sites between 2007 and 2008 for calcium or magnesium.

Turbidity was significantly higher at channel sites than at wetlands sites ( $p = 0.005$ ; Figure 4-40). The river was significantly higher in turbidity than the back channel ( $p < 0.0001$ ; Figure 4-41). There was no significant difference in turbidity at channel sites between 2007 and 2008.

The euphotic depth was significantly higher in the back channel than in the river ( $p = 0.0007$ ; Figure 4-42). Euphotic depth at channel sites did not differ significantly between 2007 and 2008. The euphotic depth in the back channel increased significantly between 2007 and 2008 ( $p = 0.02$ ; Figure 4-43).

Phytoplankton chlorophyll did not differ significantly between wetland sites and

channel sites, between the river and back channel, or at channel sites between 2007 and 2008 (Table 4-9).

There was more abundant periphyton growth in channels than in wetlands ( $p = 0.01$ ; Figure 4-44). There was significantly less periphyton growth in the Saskatchewan River than in the back channel ( $p = 0.01$ ; Figure 4-45).

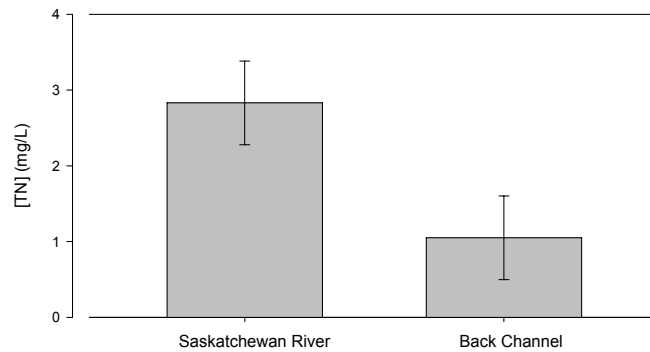


Figure 4-31: Total nitrogen concentrations (mg/L) in the Saskatchewan River ( $n = 11$ ) and the back channel ( $n = 11$ ) in 2007 and 2008. Error bars show standard error.

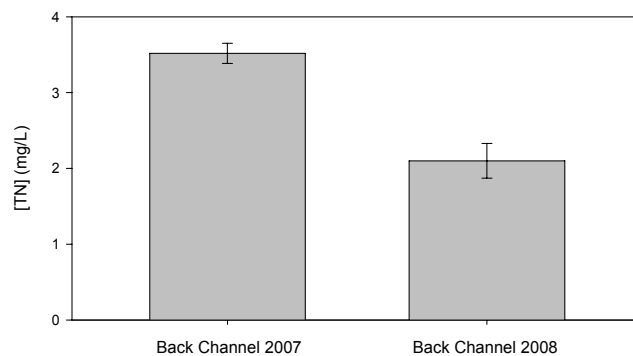


Figure 4-32: Total nitrogen concentrations (mg/L) in the back channel in 2007 ( $n = 5$ ) and 2008 ( $n = 6$ ). Error bars show standard error.

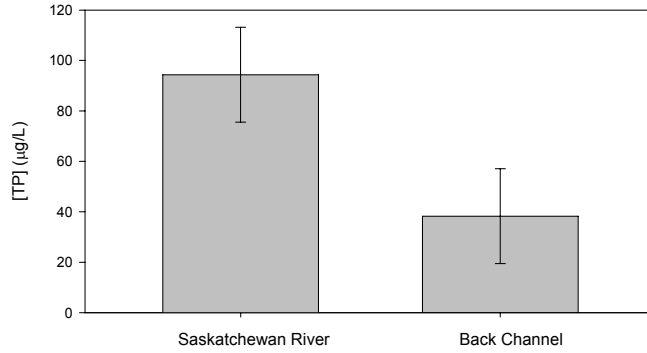


Figure 4-33: Total phosphorus concentrations ( $\mu\text{g/L}$ ) in the Saskatchewan River ( $n = 11$ ) and the back channel ( $n = 11$ ) in 2007 and 2008. Error bars show standard error.

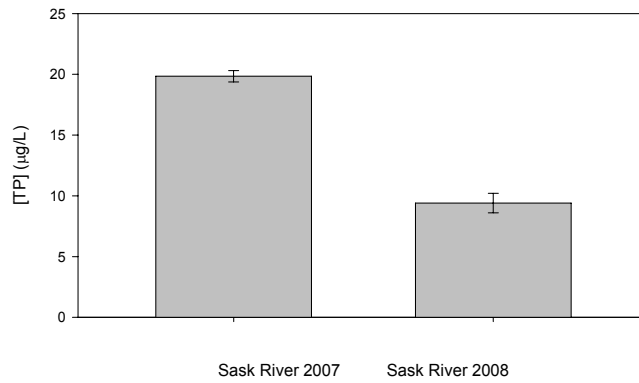


Figure 4-34: Total phosphorus concentrations ( $\mu\text{g/L}$ ) in the Saskatchewan River in 2007( $n = 5$ ) and 2008 ( $n = 6$ ). Error bars show standard error.

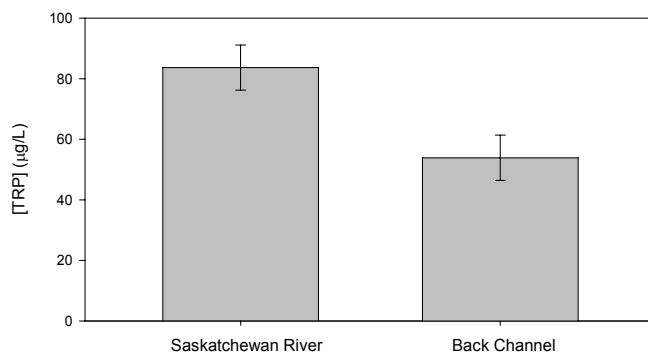


Figure 4-35: Total reactive phosphorus concentrations ( $\mu\text{g/L}$ ) in the Saskatchewan River ( $n = 12$ ) and the back channel ( $n = 12$ ) in 2007 and 2008. Error bars show standard error.

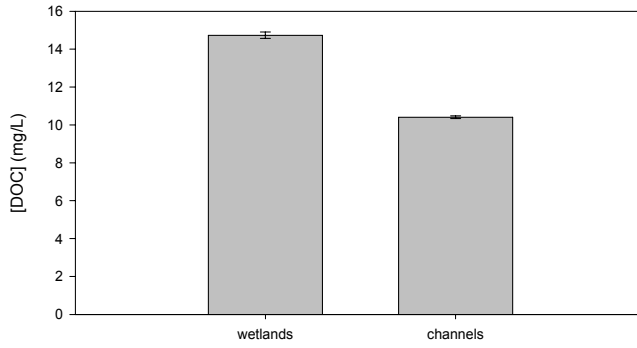


Figure 4-36: Dissolved organic carbon concentrations (mg/L) in all wetland sites (n = 389) and the channel sites (Saskatchewan River and back channel), (n = 21) in 2007 and 2008. Error bars show standard error.

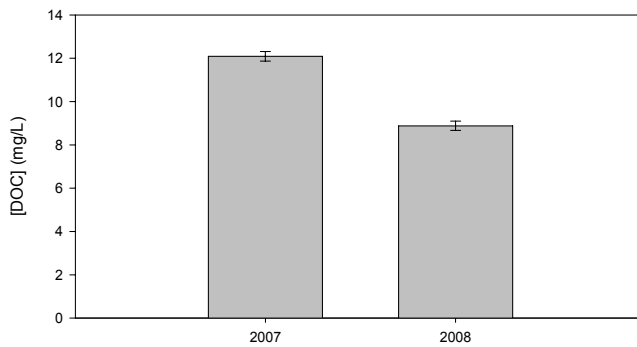


Figure 4-37: Dissolved organic carbon concentrations (mg/L) at channel sites in 2007 (n = 10) and 2008 (n = 11). Error bars show standard error.

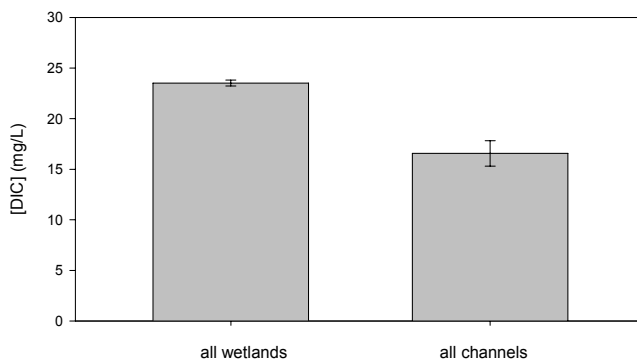


Figure 4-38: Dissolved inorganic carbon concentrations (mg/L) in all wetland sites (n = 419) and the channel sites (Saskatchewan River and back channel), (n = 24) in 2007 and 2008. Error bars show standard error.

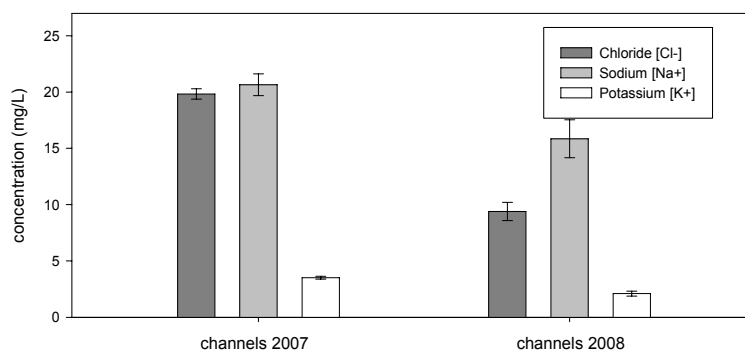


Figure 4-39: Concentrations of chloride, sodium and potassium (mg/L) at all channel sites (Saskatchewan River and back channel) in 2007 (n =6) and 2008 (n = 2). Error bars show standard error.

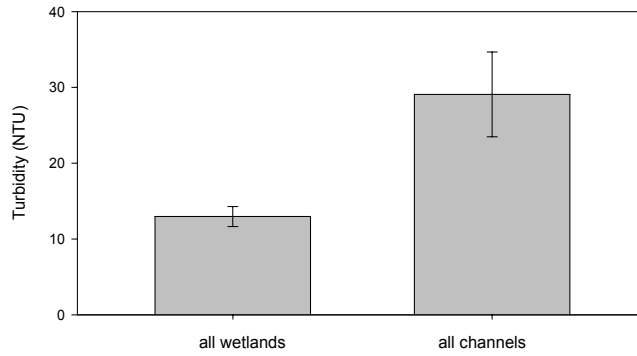


Figure 4-40: Turbidity (NTU) in all wetland sites (n = 430) and the channel sites (Saskatchewan River and back channel, (n = 24) in 2007 and 2008. Error bars show standard error.

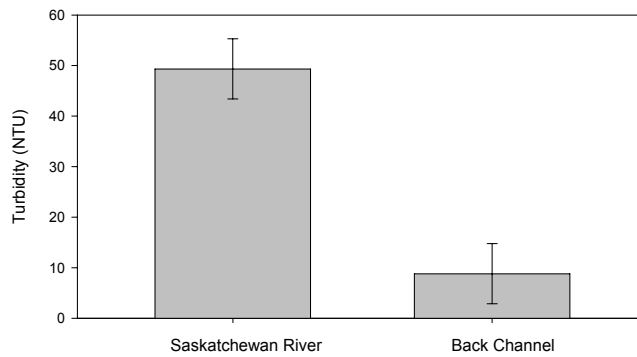


Figure 4-41: Turbidity (NTU) in the Saskatchewan River (n = 12) and the back channel (n = 12) in 2007 and 2008. Error bars show standard error.



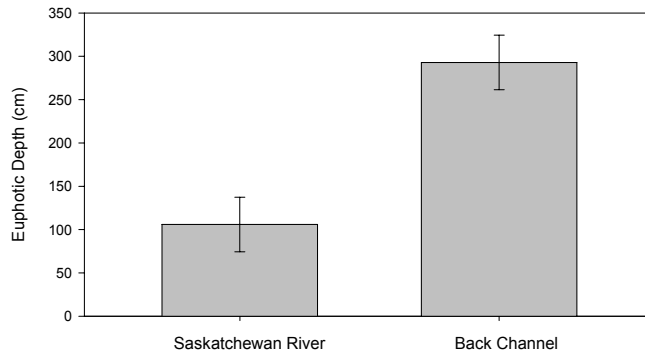


Figure 4-42: Euphotic depth (cm), in the Saskatchewan River (n = 9) and the back channel (n = 9) in 2007 and 2008. Error bars show standard error.

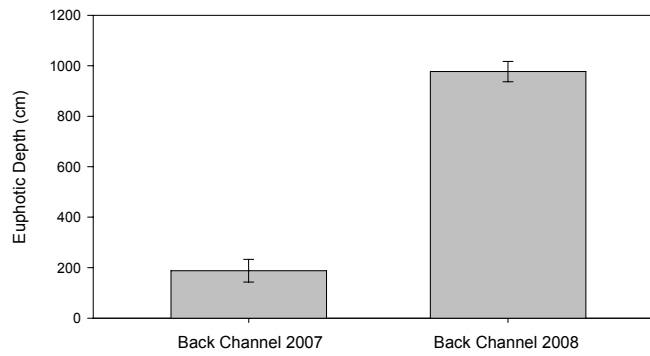


Figure 4-43: Euphotic depths (cm) in the back channel in 2007(n = 4) and 2008 (n = 5). Error bars show standard error.

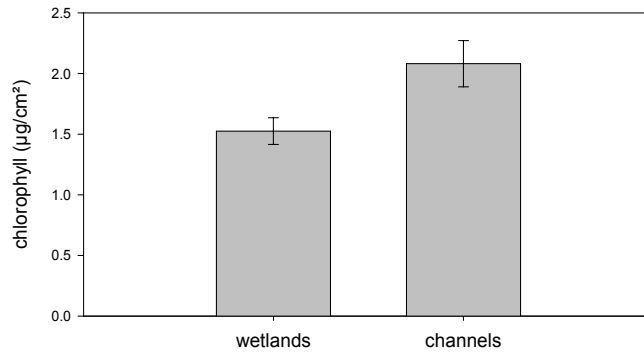


Figure 4-44: Periphyton biomass, expressed as chlorophyll ( $\mu\text{g}/\text{cm}^2$ ), in wetlands ( $n = 35$ ) and channels ( $n = 12$ ), over two-week durations in 2007 and 2008. Error bars show standard error.

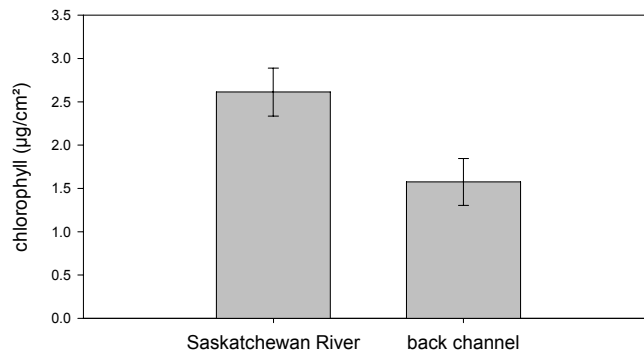


Figure 4-45: Periphyton biomass, expressed as chlorophyll ( $\mu\text{g}/\text{cm}^2$ ), in the Saskatchewan River ( $n = 6$ ) and the back channel ( $n = 6$ ), over two-week durations in 2007 and 2008. Error bars show standard error.

Table 4-9: Water quality in the Saskatchewan River and Back Channel. Mean values for each parameter are presented, with the range from minimum and maximum shown below.

	2007				2008			
	River		Back Ch		River		Back Ch	
TP ( $\mu\text{g/L}$ )	154.4		37.6		44.3		38.8	
	70.4	316.9	30.0	67.9	30.0	80.4	30.0	83.0
TRP ( $\mu\text{g/L}$ )	68.6		46.8		94.4		58.9	
	33.1	114.9	36.5	57.0	60.4	121.7	29.7	77.4
TN (mg/L)	3.7		2.1		2.1		1.9	
	1.1	6.1	0.3	4.2	0.3	5.1	0.3	4.7
$\text{NH}_3$ ( $\mu\text{g/L}$ )	12.5		12.5		15.4		21.7	
	12.5	12.5	12.5	12.5	12.5	32.9	12.5	62.7
DIC (mg/L)	16.4		16.8		16.1		16.9	
	13.2	19.9	14.3	18.4	362.9	14.8	15.0	16.8
DOC (mg/L)	11.9		12.3		9.0		8.7	
	10.9	12.6	11.5	13.2	8.3	9.6	7.9	9.9
Phytoplankton chlorophyll $\alpha$ ( $\mu\text{g/L}$ )	9.9		7.5		24.3		6.4	
	7.6	11.5	3.0	10.2	3.5	566.4	1.9	12.4
Euphotic Depth (cm)	86.4		181.6		111.0		348.4	
	73.0	101.5	108.7	285.7	91.7	140.8	206.2	487.8
Turbidity (NTU)	39.4		6.6		56.5		10.3	
	17.1	83.1	2.9	9.5	31.8	118.0	4.9	26.7
Specific Conductance ( $\mu\text{S/cm}$ )	409.9		410.3		380.5		392.9	
	380.0	425.8	375.0	426.3	249.3	442.8	362.9	441.8
$[\text{Na}^+]$ (mg/L)	20.3		21.0		14.9		16.8	
	17.9	22.5	18.0	24.4				
$[\text{K}^+]$ (mg/L)	3.6		3.5		2.1		2.1	
	3.3	3.9	3.2	4.0				
$[\text{Ca}^{2+}]$ (mg/L)	40.8		38.9		39.0		38.0	
	38.4	45.2	38.0	39.7				
$[\text{Mg}^{2+}]$ (mg/L)	15.8		15.1		14.8		15.1	
	14.6	16.9	14.3	15.9				
$[\text{Cl}^-]$ (mg/L)	19.7		20.0		8.4		10.4	
	18.0	20.3	19.4	20.9				

### **Water Quality Related to Distance from River**

Ammonia was significantly higher at wetland sites closer to a channel ( $p = 0.04$ ; Figure 4-46). Dissolved organic carbon was significantly lower at sites close to a channel ( $p = 0.02$ ; Figure 4-47). However, these trends explained only approximately 1% of the variability in the ammonia and DOC concentrations. No other nutrient concentrations were correlated to distance from the river or nearest channel.

Conductivity was correlated with distance from the nearest channel (Table 4-10). Additionally, the concentrations of all measured ions were directly proportional to distance to the nearest channel. An example of this trend can be seen in Figure 4-48, and parameters relating to each relationship are given in Table 4-10. (Some ions also correlated significantly with distance to the river, but these trends are not shown because they were consistently less significant than trends involving the nearest channel).

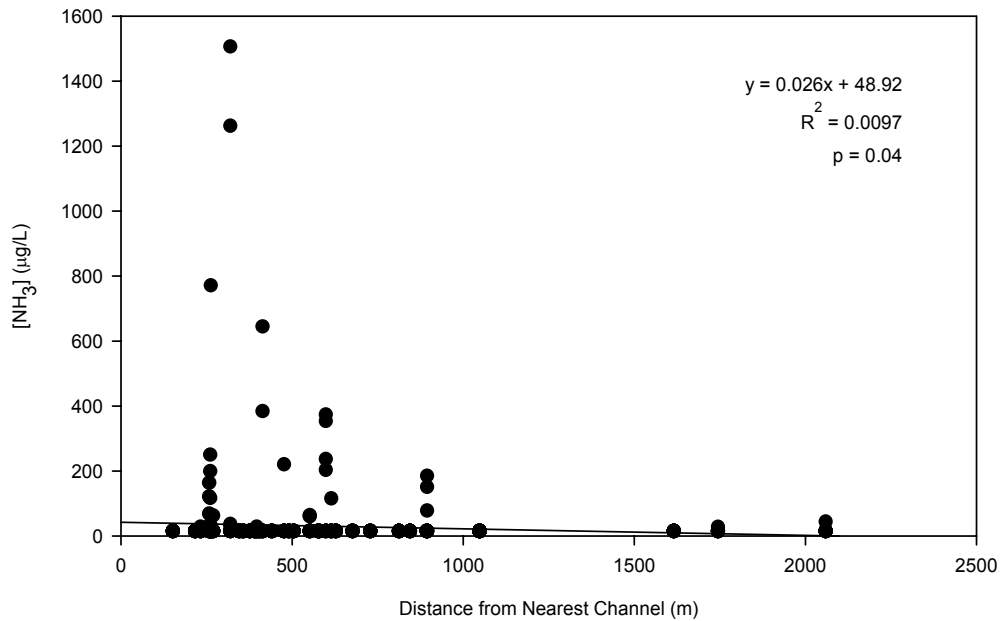


Figure 4-46: The relationship between wetland water column ammonia concentration ( $\mu\text{g/L}$ ) and distance from the nearest channel (n = 431).

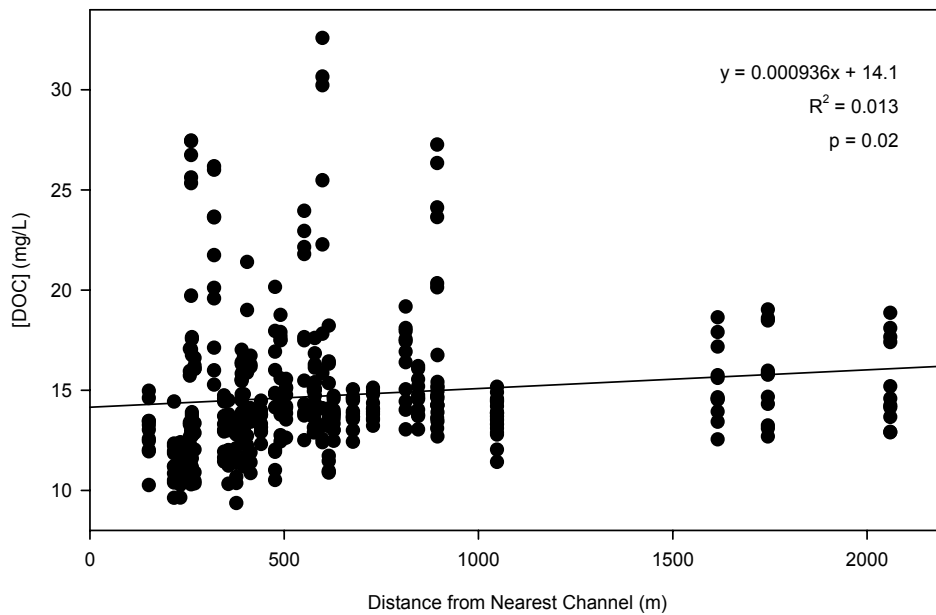


Figure 4-47: The relationship between dissolved organic carbon concentration ( $\text{mg/L}$ ) and distance from the nearest channel (n = 389).

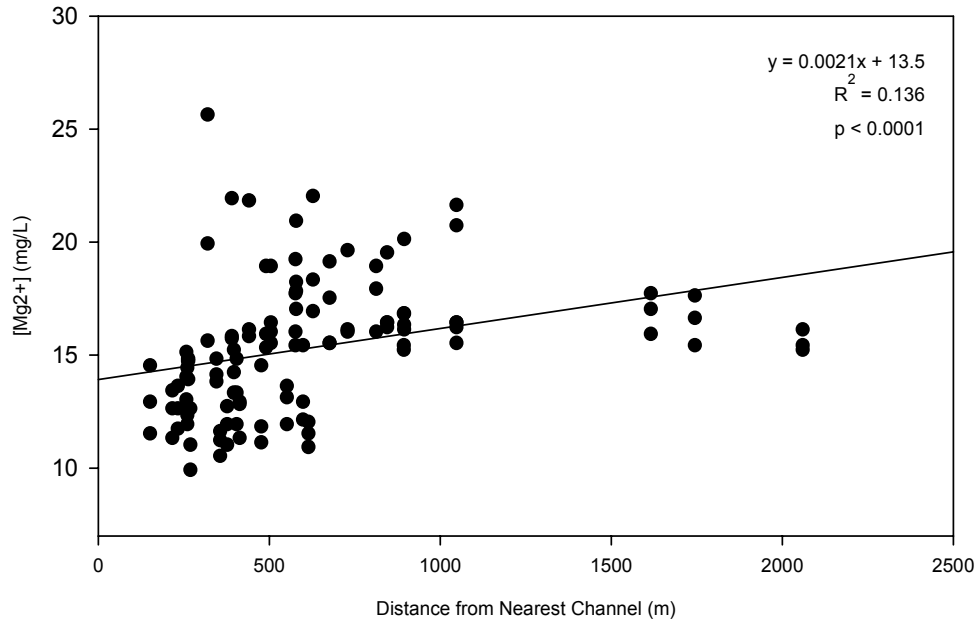


Figure 4-48: The relationship between magnesium concentration (mg/L) and distance from the nearest channel (n = 108).

Table 4-10: Parameters describing the relationship between major ions and distance from the nearest channel (n = 108). The relationship between conductivity and distance from the nearest channel is also included (n = 105).

Ion	Slope	Intercept	R <sup>2</sup> Value	p Value
Na <sup>+</sup>	0.0024	13.3	0.085	0.002
K <sup>+</sup>	0.001	3.5	0.037	0.04
Mg <sup>2+</sup>	0.0021	13.5	0.135	< 0.0001
Ca <sup>2+</sup>	0.0041	36.7	0.039	0.04
Cl <sup>-</sup>	0.0047	10.4	0.049	0.02
Conductivity	0.053	333.6	0.095	0.001

## **Sediment Chemistry**

Wetland sediments were variable in composition, ranging from clay to peat. River and back channel sediments were dense clay-like materials with low water and organic contents. A summary of sediment characteristics is presented in Table 4-11.

Not surprisingly, wet sediment density was directly proportional to percent water weight (Figure 4-49). Percent water weight and percent organic weight were correlated by a logarithmic fit (Figure 4-50): water content increased sharply with organic content up to approximately 20% organic content, then increased more gradually between 20% and 50% organic content. After sediment organic content reached a threshold of approximately 50%, there was very little corresponding increase in water content.

Sediment water content, organic content (Figure 4-51), and density (Figure 5-52) increased significantly with distance from the Saskatchewan River. Distance from the river had no effect on carbonate content. There was no relationship between any sediment composition parameter and distance to the nearest channel.

Sediment nutrient content was highly variable (Table 4-11). Sediment TP was inversely proportional to distance from the river (Figure 4-53). Sediment TN was not correlated with distance to the river. There was no relationship between sediment nitrogen or phosphorus and site depth, and no significant differences related to drawdown. Differences in sediment nutrient concentrations between wetlands and the Saskatchewan River and back channel were not significant.

## Relationships Between Sediment Chemistry and Water Quality

Sites with wetter sediment had significantly higher concentrations of total nitrogen, total phosphorus, and dissolved inorganic carbon in the water column (Figures 4-54 through 4-56). Total reactive phosphorus and ammonia concentrations, however, were not affected by sediment water content.

Nutrient concentrations in the water column were not affected by sediment density, organic content, or carbonate content. Phytoplankton chlorophyll was not affected by sediment density or water, organic, or carbonate content. There was no significant relationship between ratios of inorganic to organic nitrogen and phosphorus and any sediment composition parameter.

Water column concentrations of nitrogen (TN and ammonia) and phosphorus (TP and TRP) were not correlated with sediment concentrations of total nitrogen and phosphorus. Dissolved inorganic carbon concentration and phytoplankton biomass were proportional to sediment total nitrogen (Figures 4-57 and 4-58), but were not related to sediment total phosphorus.

Sites with wetter sediments had significantly higher calcium and magnesium concentrations in the water column (Figure 4-59). Specific conductance also increased significantly with sediment water content (Figure 4-60). Sodium, potassium and chloride concentrations were not significantly affected by sediment water content. There was no significant relationship between the concentration of any ion and sediment density, or sediment organic or carbonate content.



Table 4-11: Mean values of sediment composition parameters  $\pm$  standard deviation. nm = not measured.

Location	% Water	% Organic	% Carbonate	[TN] (mg/gww)	[TP] ( $\mu$ g/gww)
Wetlands	81.0% $\pm$ 11.8%	36.6% $\pm$ 20.2%	3.8% $\pm$ 2.6%	54.1 $\pm$ 42.7	132.3 119.4
Saskatchewan River	37.3% $\pm$ 2.6%	4.0% $\pm$ 0.7%	5.4% $\pm$ 0.1%	70.9 $\pm$ 6.3	nm
Back Channel	45.7% $\pm$ 2.0%	8.3% $\pm$ 0.5%	5.2% $\pm$ 0.4%	83.6 $\pm$ 66.4	nm

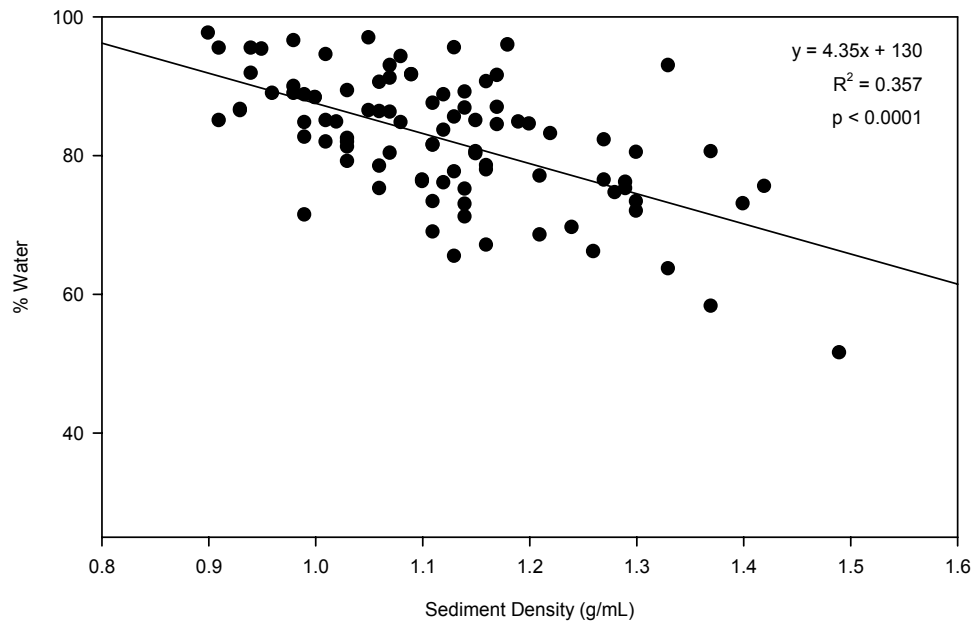


Figure 4-49: The relationship between sediment water content and sediment density.

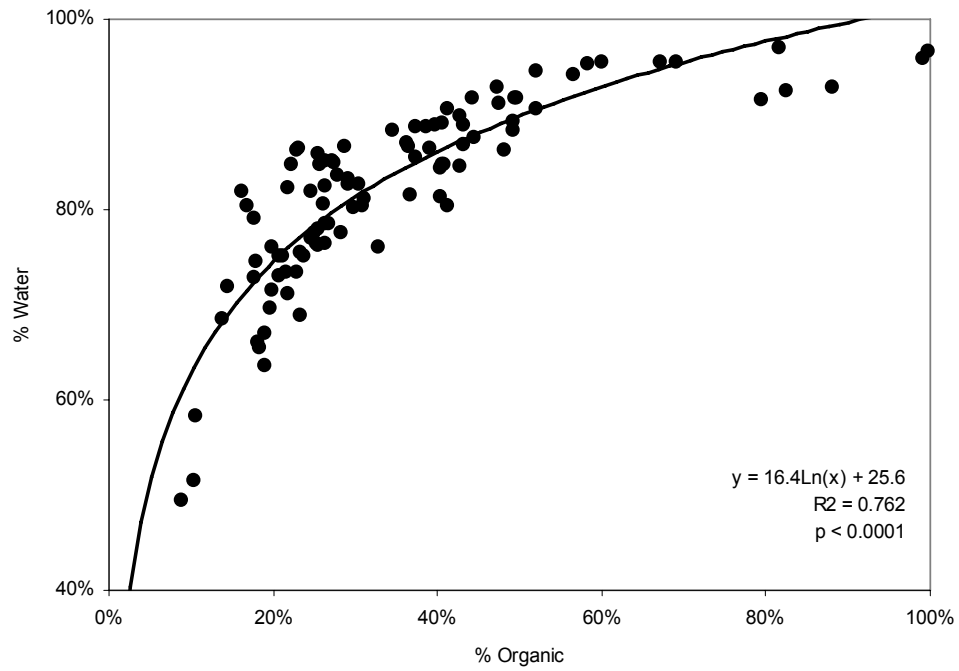


Figure 4-50: The relationship between sediment water content and sediment organic content.

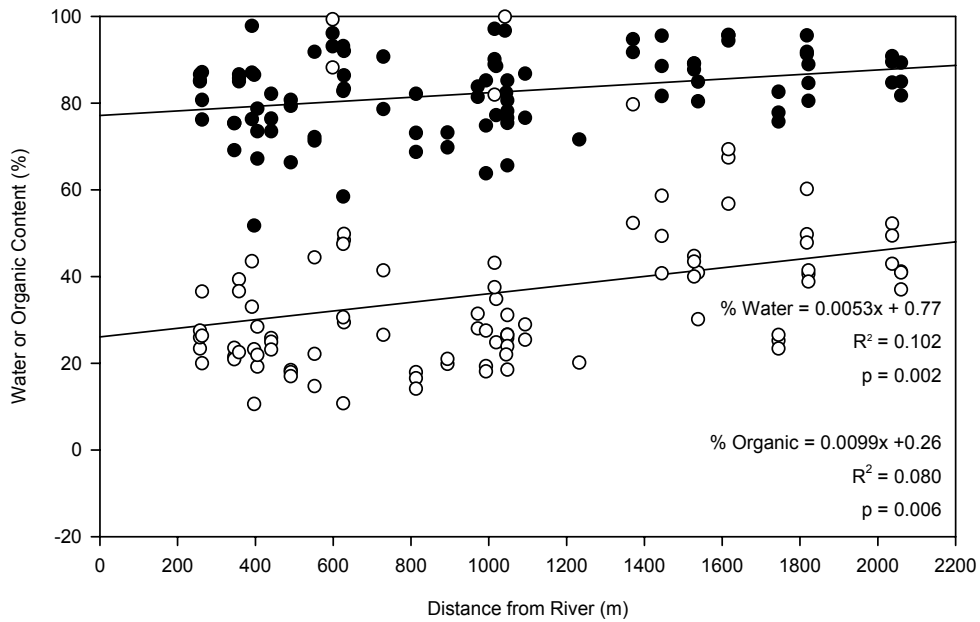


Figure 4-51: The relationships between water content and organic content of wetland sediment, and the distance from the Saskatchewan River (n = 96). Filled circles represent water content;

open circles represent organic content.

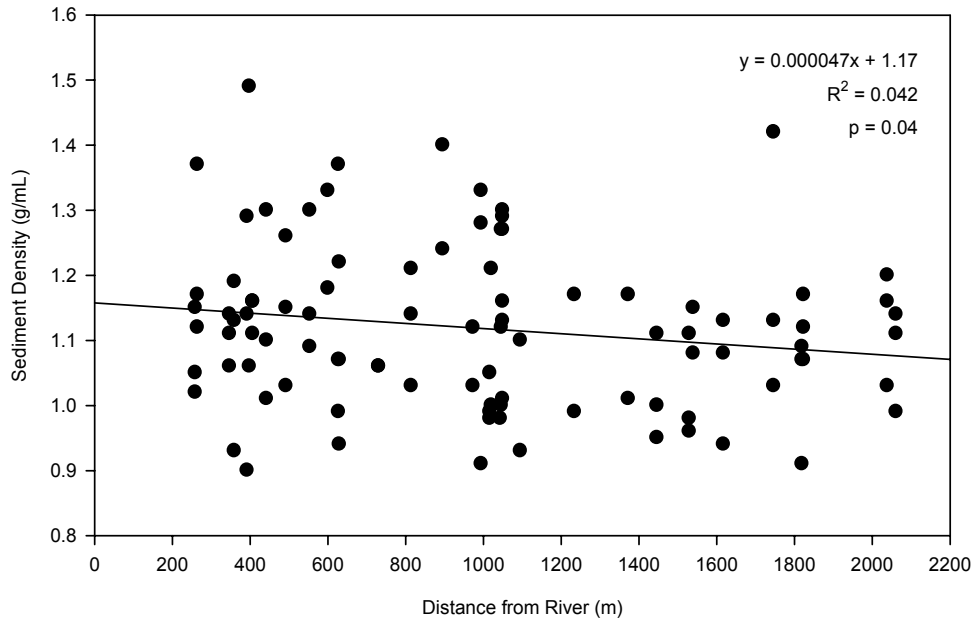


Figure 4-52: The relationship between wetland wet sediment density and the distance from the Saskatchewan River (n = 102).

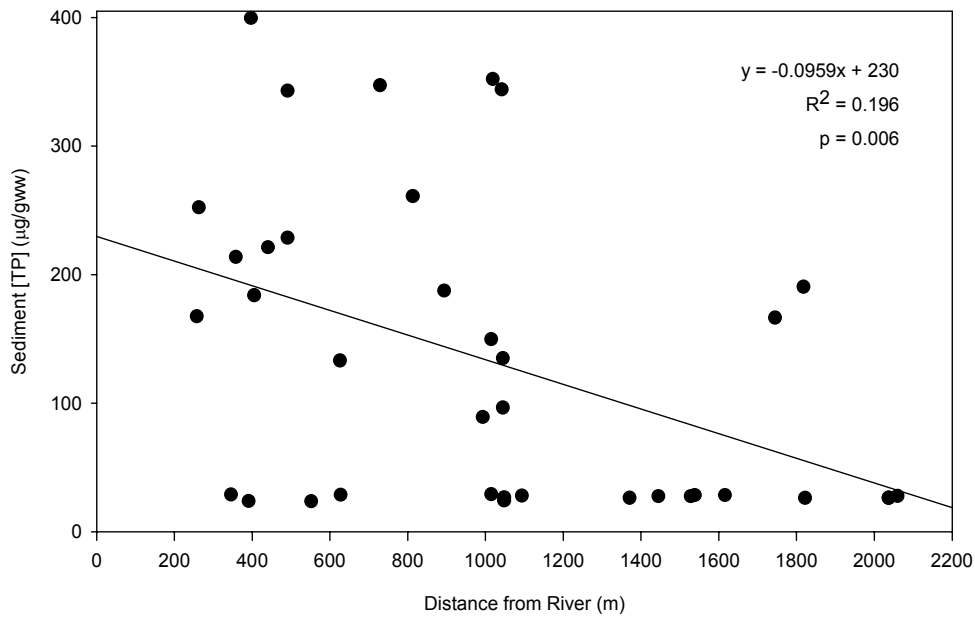


Figure 4-53: The relationship between total phosphorus concentration ( $\mu\text{g/gww}$ ) in wetland sediment and distance from the Saskatchewan River (n = 37).

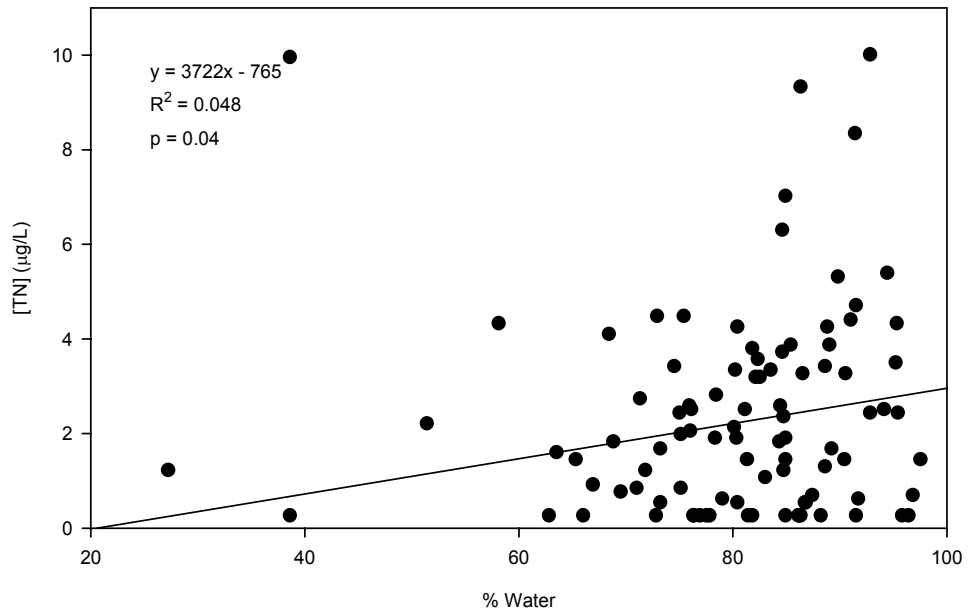


Figure 4-54: The relationship between the concentration of total nitrogen (mg/L) in the wetland water column and the sediment water content at corresponding sites (n = 93).

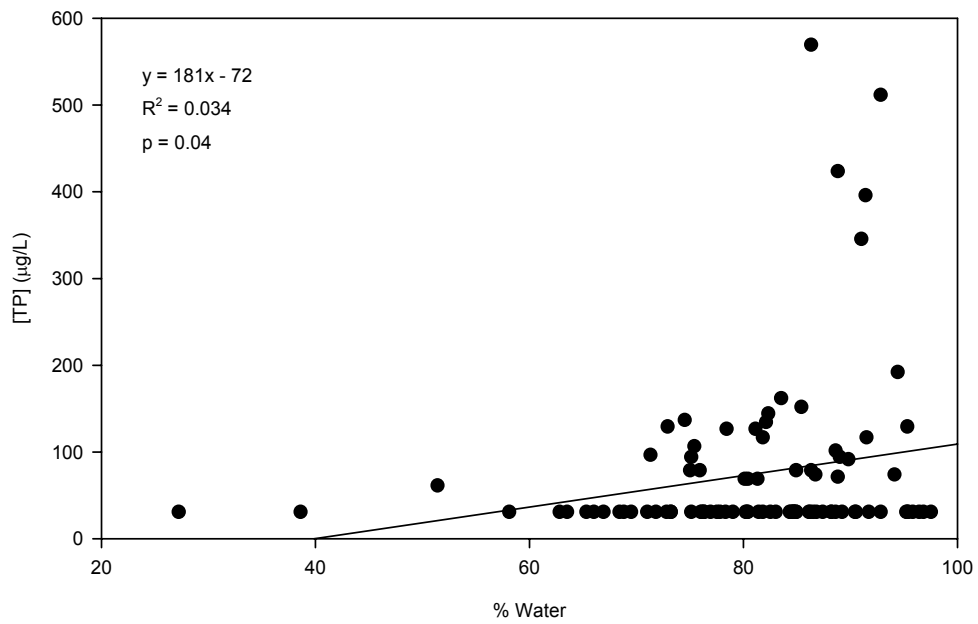


Figure 4-55: The relationship between the concentration of total phosphorus ( $\mu\text{g/L}$ ) in the wetland water column and the sediment water content at corresponding sites ( $n = 95$ ).

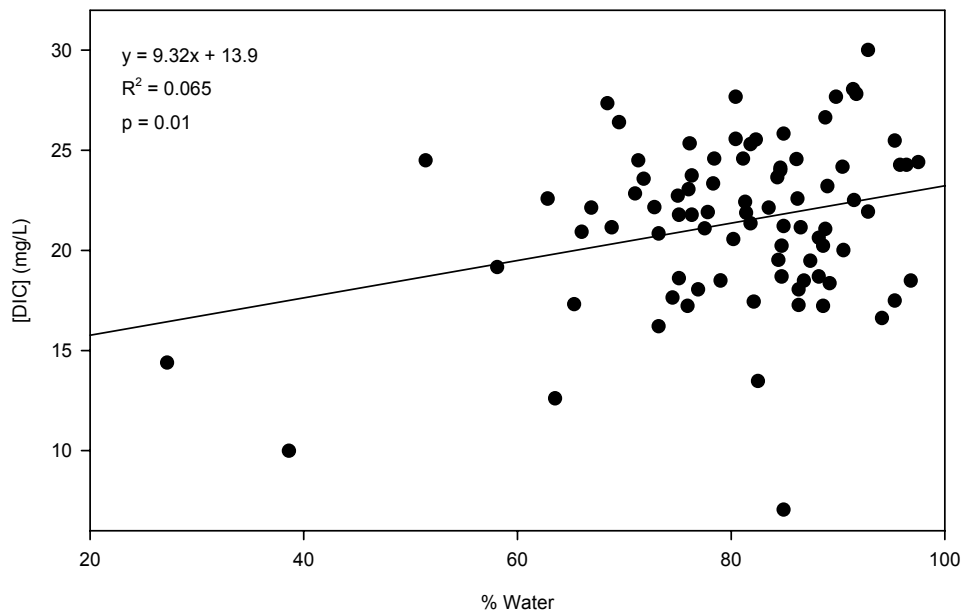


Figure 4-56: The relationship between the concentration of dissolved inorganic carbon ( $\text{mg/L}$ ) in the wetland water column and the sediment water content at corresponding sites ( $n = 84$ ).

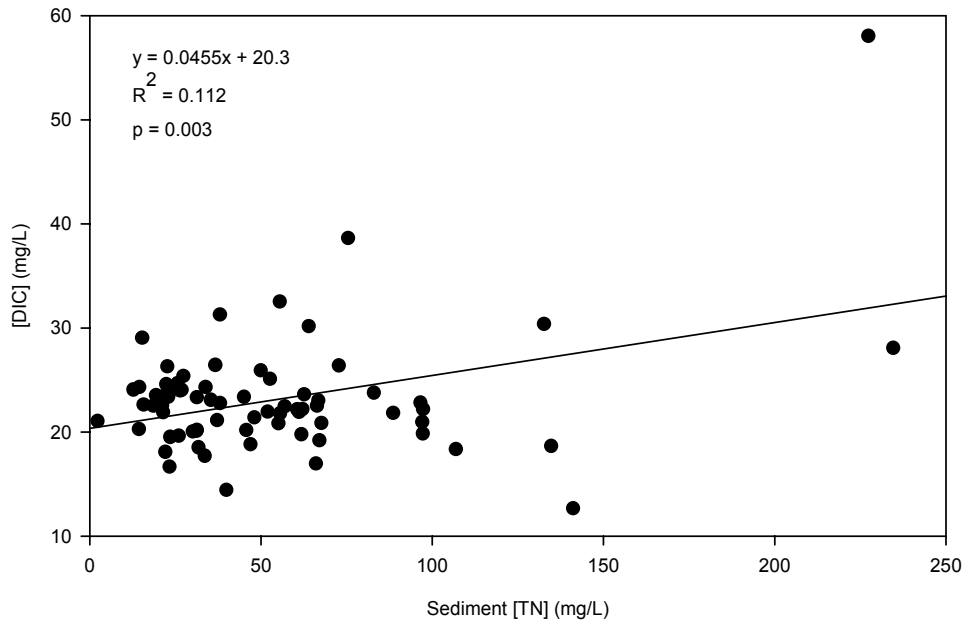


Figure 4-57: The relationship between dissolved inorganic carbon (mg/L) in the water column and sediment total nitrogen concentrations (mg/L) (n = 76).

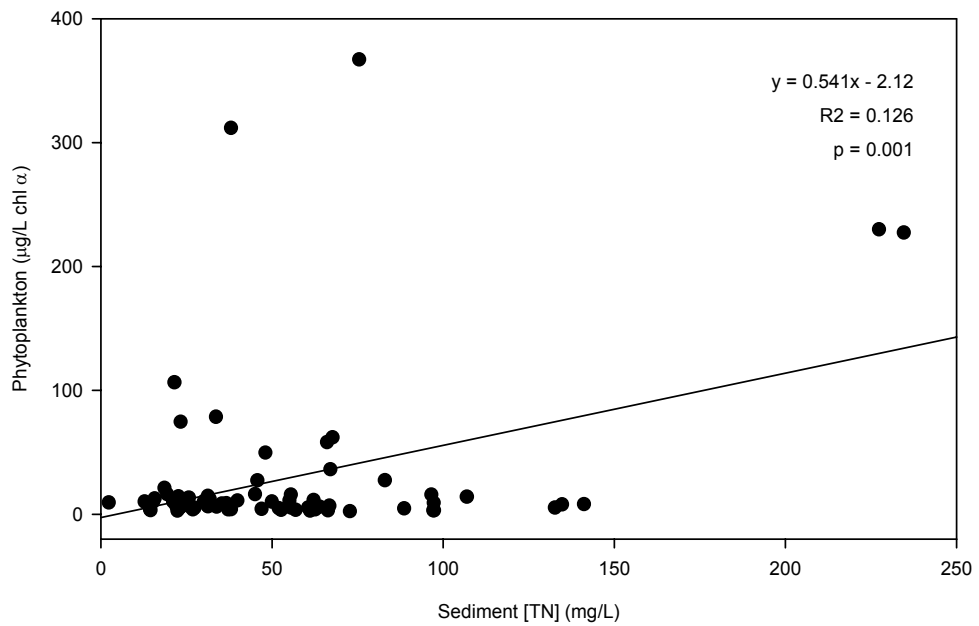


Figure 4-58: The relationship between phytoplankton biomass, as approximated by chlorophyll  $\alpha$  concentration ( $\mu\text{g/L}$ ) in the water column and sediment total nitrogen concentrations (n = 76).

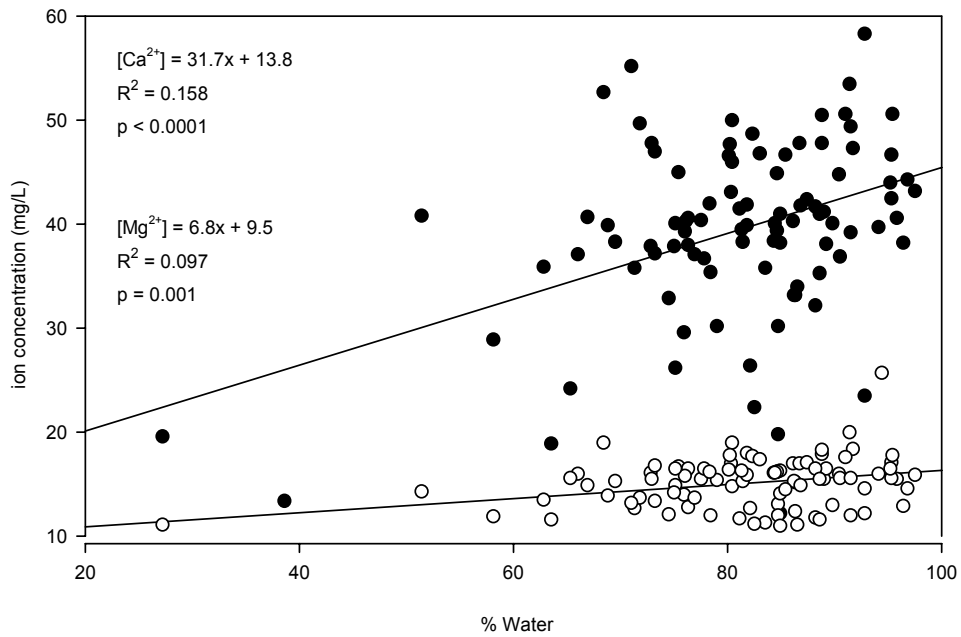


Figure 4-59: The relationship between calcium and magnesium concentrations (mg/L) and sediment water content. Filled circles represent calcium; open circles represent magnesium (n = 96).

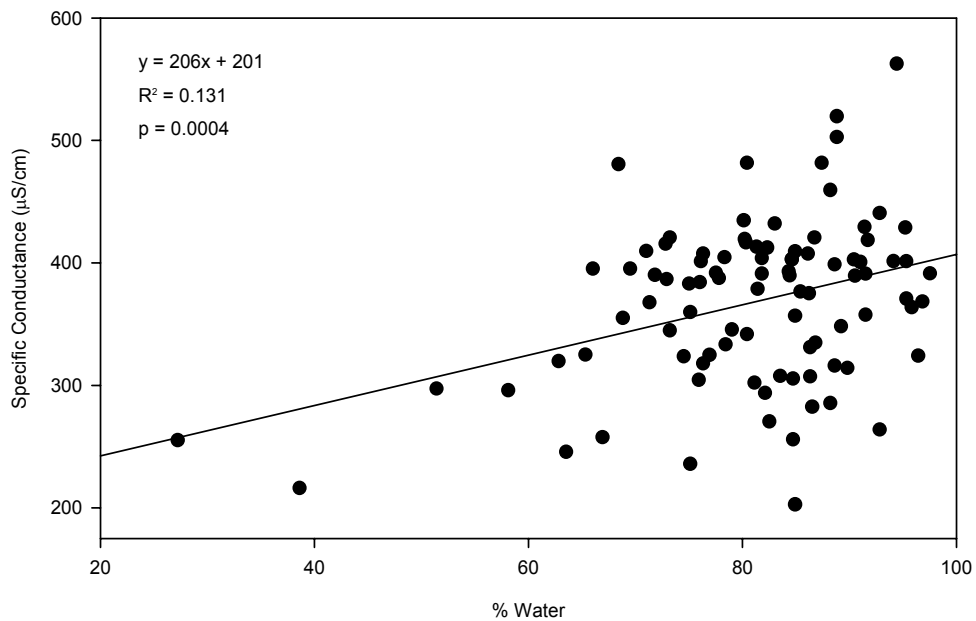


Figure 4-60: The relationship between specific conductance ( $\mu\text{S}/\text{cm}$ ) and sediment water content (n = 93).

## Discussion

### **Wetland water quality**

Turbidity and the concentrations of most nutrients (TN, ammonia, TRP) increased when study wetlands underwent drawdown, as did the variability of these parameters. There is some evidence that drawdown had greater effect on the concentrations of inorganic forms of nutrients than on organic nutrients. For example, total reactive phosphorus was higher in drawdown wetlands than in control wetlands, but total phosphorus was not significantly different. Perhaps sediments became more aerobic when water level was lowered, allowing for more decomposition of organic compounds, but this hypothesis cannot be addressed here because dissolved oxygen at the bottom of the water column was not measured.

Although certain nutrient concentrations (TP, DIC) were not significantly higher in drawdown wetlands, all measured nutrients, as well as turbidity, showed trends that increased sharply at shallow depths. These depth relationships involved a threshold point or maximum depth above which increasing depth had little correlation to an increase in the parameter in question. This threshold point was not precise, but by visual inspection of graphs could be estimated at 40cm for most parameters.

The threshold hypothesis to explain the relationship between depth and nutrient concentration and turbidity may be justified by models of wave dynamics. Surface waves occurring where the water depth is more than twice the wavelength are termed deepwater waves (US Coastal Engineering Research Center 1977), or short waves. Movement of water below short waves is circular, with each successively deeper circle exponentially smaller than that above it. Thus, only a small area of the bottom is affected by wave energy. Surface waves



occurring where depth is less than twice the wavelength are termed long waves. The movement of water below long waves becomes more elliptical: narrower in height but not narrower in width. Thus, the entire area of bottom under the wave is subject to water oscillating back and forth over the sediment. Given sufficient energy, this oscillation can re-suspend sediments into the water column (Carper and Bachmann 1984, Gons *et al.* 1986).

The threshold at which depth begins to influence turbidity and nutrient concentrations could be a transition point between short and long waves. This accounts for the less-than-perfect fit of the depth relationships, because wavelength (and therefore the depth at which waves transition from short to long) is influenced by wind velocity, which varied over the sampling period.

The speed of wind necessary to resuspend sediments at given depths can be calculated with equations developed by the US Army Coastal Engineering and Research Center (1977)

$$Z = 2L$$

$$L = \frac{g T^2}{2 \pi}$$

$$\frac{g T}{2 \pi U} = 1.20 \tanh \left[ 0.077 \left( \frac{g F}{U^2} \right)^{0.25} \right]$$

Where  $Z$  is the water depth (m),  $L$  is the wavelength (m),  $g$  is the gravitational constant  $9.8 \text{ m/s}^2$ ,  $T$  is the wave period (s),  $U$  is the wind velocity (m/s), and  $F$  is the effective fetch (m). With maximum daily wind gust data available from the Meteorological Service of Canada, it was estimated that resuspension would have occurred on at least 38% of summer days at a typical Summerberry site if site depth was 30 cm (Table 4-12), slightly below the apparent threshold of

40cm. If site depth was 50 cm, however, resuspension would have occurred on only 5% of days.

The correlation between sediment water content and water column nutrient concentrations can also be explained by wind action. Shallower sites had sediments more mixed by the wind, and a less sharp interface between water and substratum.

Wind-driven resuspension of sediments and its effect on nutrient flux has been well-studied in lakes (Carper and Bachmann 1984, Gons et al. 1986, Simon 1988, Reddy et al. 1996) but information relating to wetlands is not available (Reddy and Dalaune 2008). This study may be therefore one of the first on this process in a wetland ecosystem.

Specific conductance and the concentrations of the major ions sodium, potassium and chloride were higher in drawdown wetlands. Increased conductivity and higher concentrations of several major ions were correlated with shallower depths. These trends can be explained by the greater impact of evaporative concentration of ions where water levels are low.

Interestingly, although calcium, magnesium and dissolved inorganic carbon concentrations were related to depth, there was no corresponding difference in concentrations between drawdown and control wetlands.

A possible explanation is that a better growing season in 2008, for algae and potentially submersed vegetation, masked the effect of drawdown. Better growing conditions would have caused an increase in photosynthesis which, in turn, would have consumed hydrogen ions and shifted the carbonate buffering system such that the concentrations of dissolved inorganic carbon, calcium, and magnesium increased. Higher concentrations in all wetlands in 2008 than in 2007 would have diluted perceived effects due to drawdown. There are three

lines of evidence supporting the hypothesis of an improved growing season. Firstly, the summer of 2008 was warmer than the summer of 2007. Secondly, periphyton was measurably more abundant in 2008 than in 2007. Finally, qualitative field observations noted more submersed vegetation in 2008 (discussed further in Chapter 6). However, there was no difference in phytoplankton growth between 2007 and 2008.

Algal primary production was increased in shallow water. Phytoplankton biomass increased as depth decreased, and although the sample size of the periphyton experiment was too small to examine depth relationships, periphyton was more abundant in drawdown wetlands. This may be related to higher levels of nutrients, liberated from the sediment via mixing, in the water column. However, since the concentration of every nutrient was higher at shallower depths, it is impossible to determine which nutrient most affected algae. The role of nutrients in limiting algal primary production is explored in the next chapter.

Vegetated sites had more phytoplankton, higher conductivity, and higher concentrations of dissolved inorganic carbon and calcium. The latter three trends might be attributable to the effects of increased photosynthesis from the surrounding emergent plants. However, vegetated sites were significantly shallower than open water sites, so these perceived trends at vegetated sites may in fact be a function of the relationships with depth. The increase in phytoplankton at vegetated sites seems to refute the idea of emergent plant effects and support the idea of depth effects. If emergent vegetation affected phytoplankton biomass, one would expect this effect to be negative: emergent plants might shade phytoplankton or compete with them for nutrients. The fact that phytoplankton instead increases at vegetated sites suggests that site depth is a more important factor than proximity of emergent plants. DIC and calcium may indeed increase at vegetated sites due to photosynthesis, but the driver of this increase may be the algae that proliferate at shallower depths, not the emergent vegetation present. An alternate explanation could be that periphyton

detaching from macrophytes contributed to the perceived increase in phytoplankton chlorophyll at vegetated sites.

Table 4-12: The minimum wind speed required for wave action to resuspend sediments at a hypothetical but typical Summerberry site (fetch = 150 metres across a circular pond; effective fetch = 54.4m), given various water depths. The percentage of days from May to September 2007 and 2008 on which that minimum wind velocity was reached is also listed.

Site Depth (cm)	Necessary Wind Speed (km/h)	Percentage Days With Resuspension
30	36	38%
40	47.5	18%
50	59.5	5%
60	71	1%
70	83	0%

### **River Channel Water Quality**

The studied channels of the Saskatchewan River were higher in turbidity and carbon concentrations than Summerberry wetlands. These differences can be attributed to the dynamic nature of a river system as compared with relatively stationary wetland water.

Periphyton was more abundant in channels than in wetlands. In wetlands, algae may deplete their microhabitat of nutrients, whereas in the river and back channel, the supply nutrients is constantly refreshed.

The back channel differed from the main river channel, with lower nutrient concentration, higher water clarity, and less abundant periphyton. The back channel is influenced by both the Saskatchewan River and Summerberry

wetlands, and can be considered intermediate between the two extremes. Water quality in this channel will vary depending on the proportions of its flow coming from the Saskatchewan River through its narrow mouth versus from wetland spillage.

### River Effects on Wetlands

The Saskatchewan River plays a major role in determining wetland sediment composition at nearby sites. As distance from the river increased, wetland sedimentation processes dominated: sediments tended to be wetter, less dense, and more organic (peats). Closer to the river, however, dense, less wet, and less organic sediments (clays) were prevalent. This trend is due to overbanking events: brief and localized flooding from the Saskatchewan River, probably due to ice jams in spring. The river sediments from overbanking events are a significant source of phosphorus to the wetlands. This phosphorus likely arrives sorbed to clay particles, a common method of phosphorus transport in floodplain wetlands (Mitch et al. 1979; Mitsch and Gosselink 2000).

Phosphorus inputs to the wetlands from overbanking events cannot be explained by the influx of water, because channel concentrations of phosphorus were no higher than wetland water column phosphorus concentrations. Clay wetland sediments, similar to and in close proximity to river sediment, were higher in phosphorus than peat-like wetland sediments, implying that the river sediment is the source of phosphorus to Summerberry wetlands.

The distance from river parameter is only a rough approximation of the extent to which sites are influenced by overbanking events. The path length of the route taken by floodwater, and the amount of open water (versus densely vegetated zones) along that path should more accurately correlate with sediment composition. However, without observing precisely at which locations

overbanking events occur, the less-than-ideal distance from river parameter is useful.

It is evident that the back channel does not overbank into nearby wetlands with the same frequency as the main channel of the Saskatchewan River, because distance from the river was a better indicator of sediment composition than distance from the nearest channel. The sediments in the main river channel and back channel were similar in terms of composition so their effect on wetland sediments would be similar.

Seepage through levees was an important source of water to the wetlands, which is demonstrated by the fact that ion concentrations were lower at sites near channels, during periods when there was no direct surface flow from these channels. This effect is not specific to the river, but rather due to the proximity of any source of flowing surface water, because ion concentrations correlated better with distance from the nearest channel than with distance to the river. Groundwater had a greater influence farther from the channels, which is reflected in higher conductivities and ion concentrations at these sites. Seepage from the river does not appear to be a source of nutrients to wetlands, as nutrient concentrations are not correlated to distance from the nearest channel.

### **Comparison with water quality of nearby wetlands**

The Summerberry Marshes were generally less eutrophic, fresher, and more acidic than other large Manitoba wetlands such as Delta, Netley-Libau, and Oak Hammock Marshes (Table 4-13), and were more similar to boreal wetland sites, including hydrologically-restricted basins in the Peace-Athabasca Delta, and several shallow wetland lakes in northern Alberta. Interestingly, chlorophyll was higher at Summerberry than at other Manitoba or boreal wetlands.

Table 4-13: A comparison of water quality parameters between the Summerberry Marshes and other Manitoba and boreal wetlands. Summerberry values are 2007 only. Delta Marsh values from Goldsborough (unpublished data). Netley-Libau and Oak Hammock Marshes (1997) values from Bortoluzzi *et al.* (in prep). PAD values estimated from figures in Wolfe *et al.* (2007). Northern Alberta boreal lakes values from Bayley and Prather (2003).

	<b>Summerberry Marshes, SRD, Manitoba</b>	<b>Delta Marsh, Manitoba</b>	<b>Netley-Libau Marsh, Manitoba</b>	<b>Oak Hammock Marsh, Manitoba</b>	<b>restricted basins, Peace- Athabasca Delta, Alberta</b>	<b>boreal wetland lakes, northern Alberta</b>
Depth (cm)	84		75	49		
TN (mg/L)	2.2		7.9		1.6	1.3
Ammonia (µg/L)	14.7	109	37.5	20		15.9
TP (µg/L)	79.8	140	450			123.2
TRP (µg/L)	45.7		212.5	220	22	28.8
DOC (mg/L)	13.7					
DIC (mg/L)	21.7	67	55	92.7		
pH	7.5	8.8	7.8	8.7		7.6
Specific Conductance (µS/cm)	367.9	2189			350	206.2
Sodium (mg/L)	14.8	274			12	
Potassium (mg/L)	4.1	21.4			6	4.2
Calcium (mg/L)	39.2	42.6			33	
Magnesium (mg/L)	14.8	101.5			12	9.2
Chloride (mg/L)	13.4	443				1.4
Turbidity (NTU)	6.2	9.8				3.8
Chlorophyll (µg/L)	49.5	11.5			2.8	34.1

## Conclusion

This study provides one of the first characterisations of water quality and algal production in a boreal marsh and therefore contributes to the collective understanding of boreal wetland ecology. The description of water quality is also of value as a reference condition of wetlands in the SRD prior to further change or degradation.

This work comprises the first comprehensive study of water quality changes in response to water level manipulation in boreal wetlands. This provides valuable information for managers attempting to use water level manipulation as a technique to manage secondary production by managing water quality and algal production. Specific recommendations to managers regarding artificial drawdown can be found in Chapter 8.

In addition to its utility as a management technique, artificial drawdown may also be a model for forecasted changes to water supply for wetlands in the SRD. Increasing temperatures and decreasing streamflows are predicted for the Canadian prairies over the next decades (Yulianti and Burn 1998; Mehdi *et al.* 2002). As climate change reduces available water across the Saskatchewan River basin, wetlands in the SRD may face similar conditions to those of an artificial drawdown. Additionally, upstream demand on water for power generation, irrigation, and industry is likely to increase. Alberta and Saskatchewan are not yet consuming the full portion of water from Saskatchewan River tributaries to which they are entitled under Prairie Provinces Water Board water sharing agreements (Partners FOR The Saskatchewan River Basin 2008). The response of water quality and algal primary production to artificial drawdown elucidated from this study may help predict future wetland changes due to further decreased flows on the Saskatchewan River.



## Chapter 5: Algal Response – Nutrient Diffusing Substrata

### Introduction

The nutrient concentrations presented in the previous chapter are important in quantifying the water chemistry of Summerberry. However, these values represent only the portion of the nutrient pool present in the water column, not that which is actually available to biota. An in situ bioassay can be useful because it can directly show which nutrient, if any, is actually limiting algal growth.

This bioassay measured the response of the periphyton community to nutrient supplementation. Much research has focused on nutrient limitation of phytoplankton (Glooschenko and Alvis 1973; Gerhardt and Likens 1975; Schindler 1975; Healey and Hendzel 1980; Hardy *et al.* 1986; Welch *et al.* 1989; Elser *et al.* 1990). However, because phytoplankton is mobile, experiments must be physically isolated in a micro- or mesocosm. Periphyton is stationary and can therefore be studied in the natural environment by providing only an artificial surface and nutrient source. Periphyton bioassays of nutrient limitation work have been successfully used in a variety of aquatic environments (Wuhrmann and Eichenberger 1975; Stockner and Shortreed 1978; Sladeckova 1979; Marcus 1980; Krewer and Holm 1982; Peterson *et al.* 1983; Fairchild *et al.* 1985; Francouer *et al.* 1999; McDougal 2001; Scott *et al.* 2005; Kolochuk 2008; Bortoluzzi *et al.* in prep).

Redfield (1958) noted that the molar ratio of C:N:P was 106:16:1 in both marine phytoplankton and in ocean water, and suggested that deviations from the ratio can predict which nutrient limits algal growth. Although the C:N and N:P ratios tend to be greater in freshwater lakes than in marine environments (Healey and

Hendzel 1980; Kilham 1990; Hecky *et al.* 1993), a relationship still exists between nutrient ratios and algal nutrient limitation. I therefore anticipated that nutrient limitation of periphyton production at Summerberry could be predicted based on several ratios calculated from water column nutrient concentrations. For two nitrogen to phosphorus ratios, TN:TP and NH<sub>3</sub>:TRP, I expected that N:P > 20 would predict phosphorus limitation and N:P < 10 would predict nitrogen limitation. For two carbon to nitrogen ratios, TC:TN and DIC:NH<sub>3</sub>, I expected that C:N > 106 would predict nitrogen limitation and C:N < 106 would predict carbon limitation.

## Methods

Nutrient-diffusing substrata (NDS) were employed at one open water site per wetland in June, July and August of 2007 and 2008. NDS were also used at the sampling sites in the Saskatchewan River and the back channel.

The NDS, modeled after those used by Kolochuk (2008) and Bortoluzzi *et al.* (in prep), consisted of 50mL plastic centrifuge vials filled with a 2% agar solution, capped with a porous silica frit (Leco Instruments 528-042) (Figure 5-1). Four vial treatments were used: agar enriched with 0.05 M phosphorus (in the form of Na<sub>2</sub>HPO<sub>4</sub>), agar enriched with 0.05 M nitrogen (in the form of NaNO<sub>3</sub>, with 0.05 M NaCl added to balance sodium concentrations), agar containing 0.05 M of both nitrogen and phosphorus, and control agar with no added nutrient.

Four replicates of each treatment were deployed in floating 60cm by 30cm PVC frames (Figure 5-2), which held the vials horizontally 10cm below the water surface, a depth determined by Kolochuk (2008) to show little light limitation of algal growth. Frames were affixed to a stake by means of a circular float, which allowed rotation with the wind (Figure 5-3), such that frits were always pointing

downwind and periphyton was therefore less likely to be sloughed off by wave action. The frames remained in the water for two weeks, which is within the period during which stable concentrations of nutrients diffuse from the agar (see Appendix I). After this time, vials were removed from frames and the silica frits were removed (Figure 5-1) and frozen. Periphyton chlorophyll  $\alpha$  per unit surface area was determined via a 90% methanol extraction, as described in Table 4-2.

Differences in algal chlorophyll per unit surface area between nutrient and control treatments were analysed using a one-way ANOVA with JMP 8 software. The nutrient treatment that was significantly higher in algal chlorophyll by Tukey-Kramer HSD was deemed to be the limiting nutrient. The possible outcomes, described further in Figure 5-4, were N limitation (denoted by N), P limitation (P), N and P co-limitation (NP), and no significant treatment effect (ns). An example the periphyton response for one outcome is shown in Figure 5-5.

The relative magnitude of the treatment effect was obtained by dividing the mean algal chlorophyll of the limiting nutrient treatment by the mean of the control treatment.

The frequency of N, P, NP, and ns limitation was compared between different environments, for example between wetlands and channels, and between drawdown and high-water wetlands, using a ChiSquare analysis and likelihood ratio. Sites limited by the same nutrient were grouped and their mean water column nutrient concentrations, nutrient ratios, and water clarity variables were analysed via one-way ANOVA.



Figure 5-1: Nutrient diffusing substratum consisting of a plastic tube filled with nutrient enriched agar and capped with a silica frit. The frit is being removed, to be frozen for a methanol extraction of periphyton chlorophyll.



Figure 5-2: The author removing a floating NDS frame, holding 16 randomly arranged vials of four nutrient agar treatments, after its deployment in the marsh for a period of two weeks.



Figure 5-3: A floating frame containing NDS vials, deployed at a marsh site.

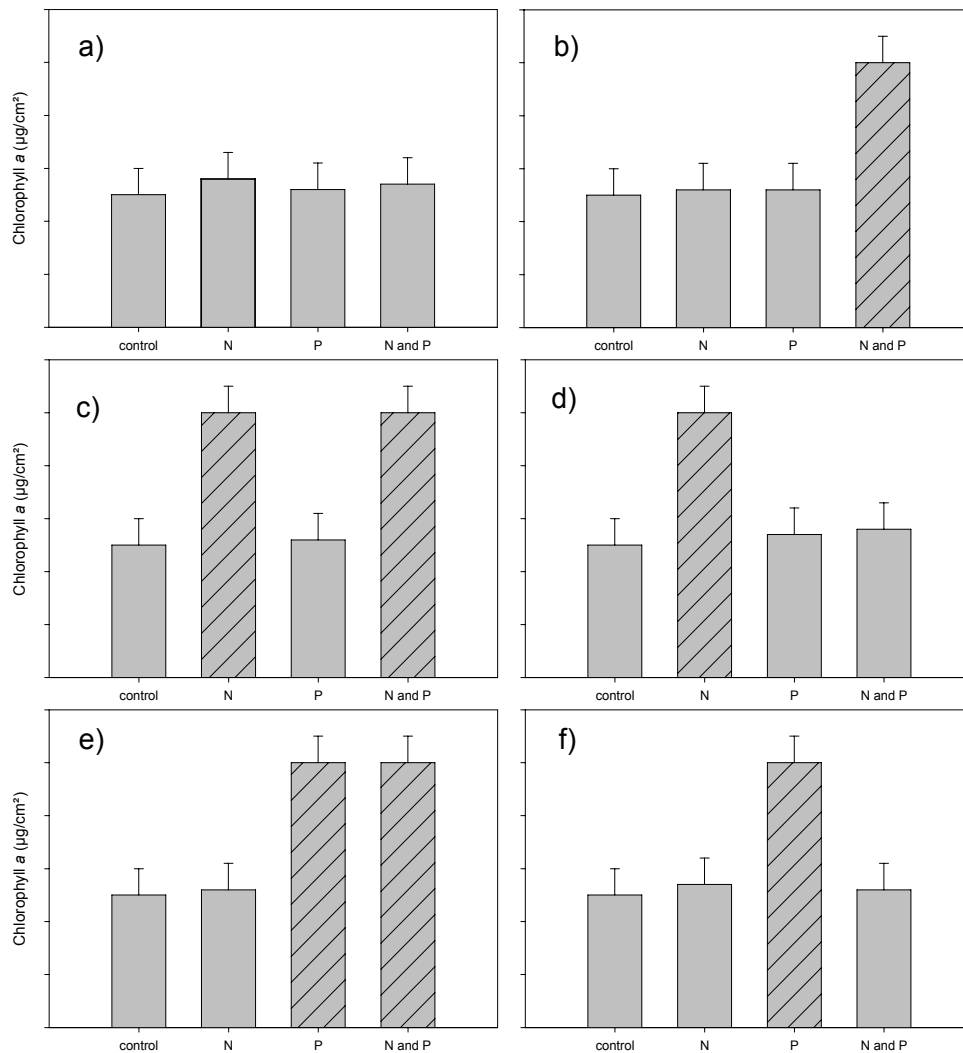


Figure 5-4: Six potential NDS outcomes: a) ns: no significant treatment effect (no treatment has significantly more algal growth than any other), b) NP: nitrogen and phosphorus co-limitation (the N and P combined treatment has significantly more growth), c) N: nitrogen limitation (the N alone and NP treatments have significantly more growth), d) N: nitrogen limitation (the N alone treatment has significantly more growth), e) P: phosphorus limitation (the P alone and NP treatments have significantly more growth), or f) P: phosphorus limitation (the P alone treatment has significantly more growth).



Figure 5-5: A sample periphyton growth response to the four NDS treatments. Clockwise from top left: control agar, agar enriched with N, agar enriched with N and P, agar enriched with P.



## Results

Wetlands were most commonly NP co-limited (Figure 5-6), but sometimes N-limited or not significantly limited by either nutrient (Figure 5-7). P-limitation was not observed at any site in either year of study.

The Saskatchewan River was limited by neither N nor P toward the end of each summer, but was N-limited in June 2007 and NP co-limited in June and July 2008. The back channel was nearly always NP-co-limited. The one deviation from NP co-limitation occurred in August 2008.

The relative magnitude of the algal response to enrichment by the limiting nutrient ranged from 2.2 to 11.9; the mean was 4.9. There was no significant difference between the relative magnitude of the N and NP effect. The back channel had a larger response than the river ( $p = 0.02$ ), but neither was significantly different than the wetlands (Figure 5-8).

There was more NP-co-limitation in 2007 than in 2008 (Figure 5-9), and more limitation by neither nutrient in 2008. N-limitation was only observed in 2007, and it was generally seen early on in the season. There was no difference in the magnitude of the algal response to the limiting nutrient between 2007 and 2008.

Drawdown wetlands most frequently showed limitation by neither nutrient (Figure 5-10). When there was nutrient limitation, it was always NP co-limitation. High-water wetlands (all wetlands in 2007 plus control wetlands in 2008) were co-limited by N and P, or, less frequently, limited by N (Figure 5-10). A result of ns, or no significant limitation by either nutrient, in high-water wetlands was rare and usually was close to significant ( $p = 0.054$ ) with N-limitation or NP-co-limitation. There was no difference in the magnitude of the algal response to the limiting nutrient based on drawdown.

No wetlands were limited by N in 2008. Figure 5-11 shows that in 2008, drawdown wetlands were most frequently limited by neither N nor P, whereas control wetlands were most frequently NP co-limited.

The experimental wetlands were all N-limited or NP co-limited in the pre-drawdown summer of 2007 (Figure 5-12). During the drawdown of 2008, the majority these wetlands became limited by neither nutrient. The remainder were co-limited by NP.

Total nitrogen concentrations were significantly higher at sites not limited by nutrients, as compared to nutrient-limited sites ( $p = 0.02$ ; Figure 5-13), but there was no difference between N-limited and NP-co-limited sites. Dissolved organic carbon concentrations were also significantly higher at sites not limited by nutrients, with no difference between N-limited and NP co-limited sites ( $p = 0.02$ ; Figure 5-14). There was no significant difference in total phosphorus, TRP,  $\text{NH}_3$ , DIC or TC concentrations between differently limited sites.

In situ nitrogen to phosphorus molar ratios were not good indicators of nutrient limitation as determined by NDS. The TN:TP molar ratio most commonly predicted P-limitation and was not correlated with nutrient limitation status. The  $\text{NH}_3$ :TRP molar ratio predicted N-limitation in every instance which did not correlate with limitation status.

In situ carbon to nitrogen molar ratios were also poor predictors of biologically determined nutrient limitation status. The DIC: $\text{NH}_3$  ratio predicted N-limitation in every case and was not correlated to limitation status. The TC:TN ratio predicted N-limitation at every wetland site but one. However, N-limited sites had significantly higher TC:TN molar ratios than NP-co-limited sites or sites not limited by nutrients ( $p = 0.006$ ; Figure 5-15).

Turbidity was not a good indicator of nutrient limitation status. The euphotic depth, when calculated as the depth which receives 1% of surface light, could not predict nutrient limitation. Neither could euphotic depth, when calculated by multiplying the Secchi depth by a factor of three, significantly predict nutrient limitation. However, few Secchi measurements were available because the Secchi depth was frequently greater than the site's depth. Light sensor readings were also frequently unavailable because many sites were too shallow for the light meter apparatus. When light-sensor-calculated euphotic depths were used to supplement unavailable Secchi-calculated euphotic depths, it was found that nutrient-limited sites in the wetlands and river had significantly larger euphotic depths than sites not limited by either nutrient ( $p = 0.01$ ; Figure 5-16). This trend did not hold true in the back channel.

Sites limited by neither N nor P had significantly more periphyton growth (on surfaces not supplemented by nutrients) than sites limited by N ( $p < 0.0001$ ) or co-limited by NP ( $p < 0.0001$ ) (Figure 5-17). There was no difference in algal growth between N- and NP-limited sites. There was no significant difference in phytoplankton chlorophyll between N-limited sites, NP-co-limited sites, and sites limited by neither nutrient.



Figure 5-6: The most common response to NDS was NP-colimitation. Clockwise from top left: control treatment; N treatment; N + P treatment, showing a dramatic periphyton response; P treatment.

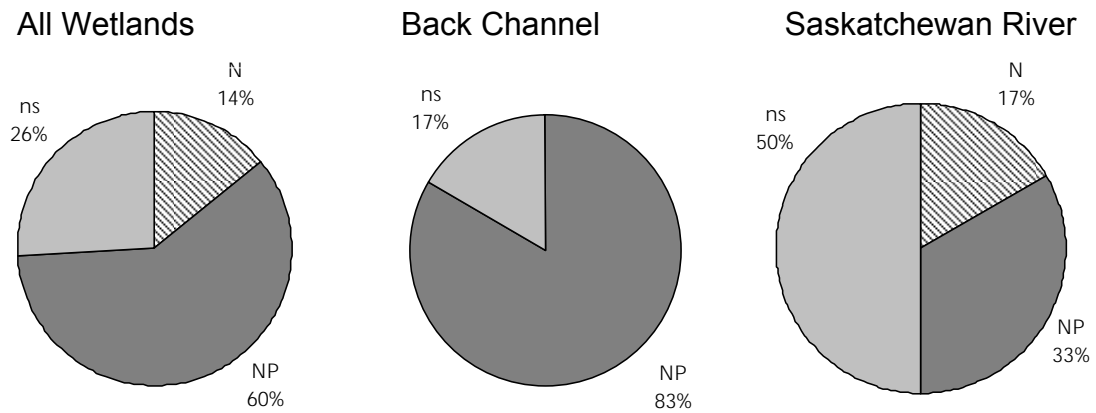


Figure 5-7: Distribution of the limitation status of all sites in which NDS experiments were conducted over 2007 and 2008. Wetlands sites ( $n = 35$ ) are shown separately from the back channel ( $n = 6$ ) and Saskatchewan River sites ( $n = 6$ ). N indicates N-limitation, NP indicates NP co-limitation, and ns indicates no significant limitation by either nutrient ( $p > 0.05$ ). Results are presented although there was no statistically significant difference between wetlands, back channel and river by ChiSquare analysis.

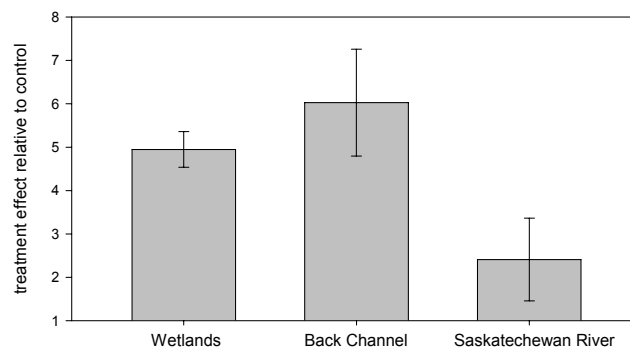


Figure 5-8: The magnitude of the treatment effect of nutrient enrichment on periphyton, relative to the control, in Summerberry wetlands ( $n = 27$ ), in the back channel ( $n = 5$ ) and in the Saskatchewan River ( $n = 3$ ), including both N- and NP-limited sites. The relative treatment effect in the river is significantly lower than that in the back channel ( $p = 0.02$ ), but neither differs significantly from the wetlands.

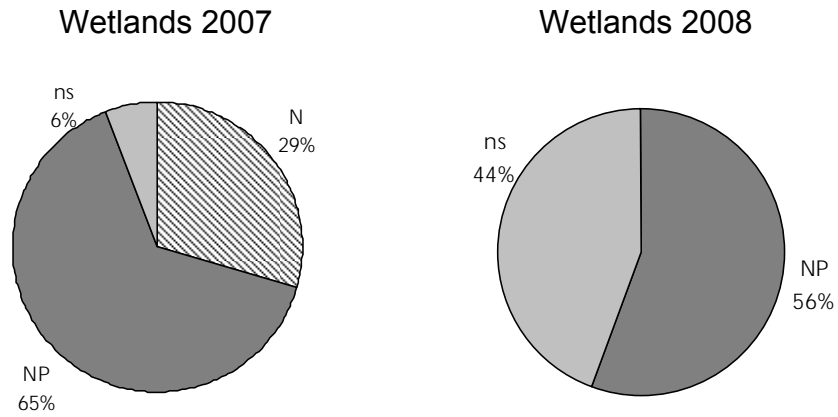


Figure 5-9: Distribution of the limitation status of wetlands in 2007 (n = 17) and 2008 (n = 18) was significantly different ( $p = 0.002$ ;  $RSquare = 0.201$ ). See Figure 5-7 for legend.

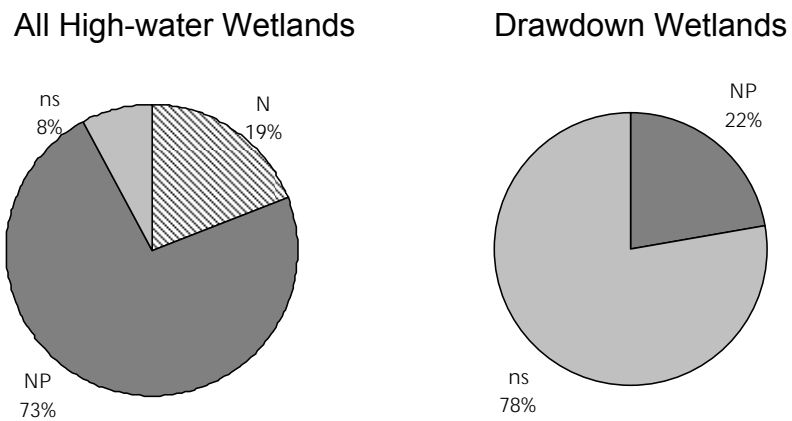
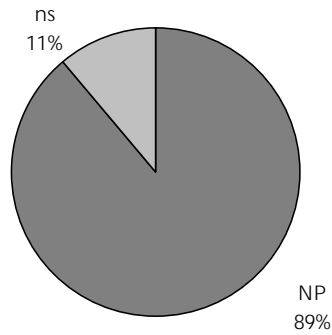


Figure 5-10: Distribution of the limitation status of high-water wetlands (all wetlands in 2007 plus control wetlands in 2008; n = 24) and drawdown wetlands (in 2008; n = 9) was significantly different ( $p = 0.0002$ ;  $RSquare = 0.263$ ). See Figure 5-7 for legend.

Control Wetlands 2008



Drawdown Wetlands 2008

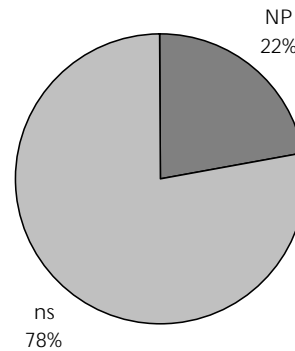
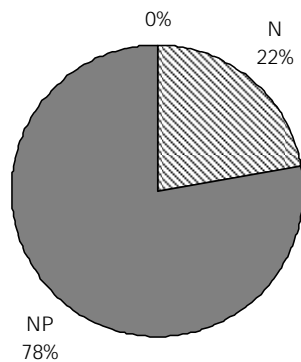


Figure 5-11: Distribution of the limitation status of control wetlands (n = 9) and drawdown (n = 9) wetlands in 2008 was significantly different (p = 0.003; RSquare = 0.361). See Figure 5-7 for legend.

Pre-Drawdown Wetlands 2007



Drawdown Wetlands 2008

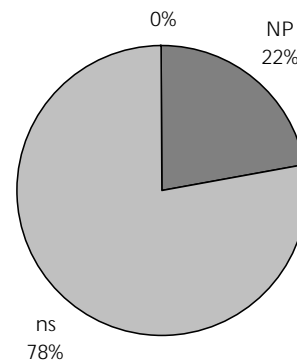


Figure 5-12: Distribution of the limitation status of the experimental wetlands in the pre-drawdown season of 2007 (n = 9) and the same wetlands during drawdown in 2008 (n = 9) was significantly different (p = 0.0004; Rsquare = 0.447). See Figure 5-7 for legend.

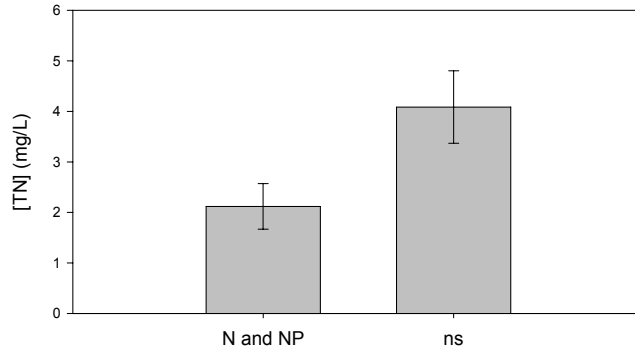


Figure 5-13: Total nitrogen concentrations (mg/L) at nutrient-limited sites (n = 26) and sites limited by neither nitrogen nor phosphorus (n = 9). Error bars represent standard error.

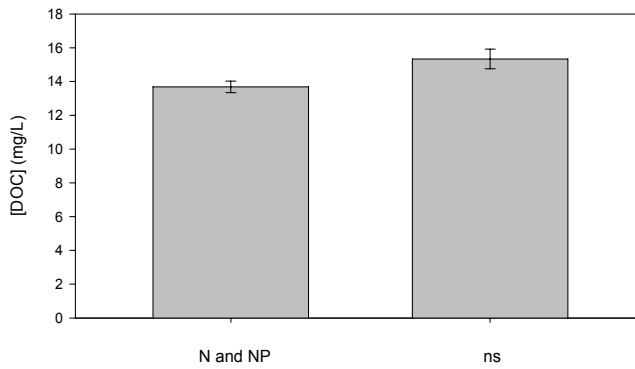


Figure 5-14: Dissolved organic carbon (mg/L) at nutrient-limited sites (n = 26) and sites limited by neither nitrogen nor phosphorus (n = 9). Error bars represent standard error.



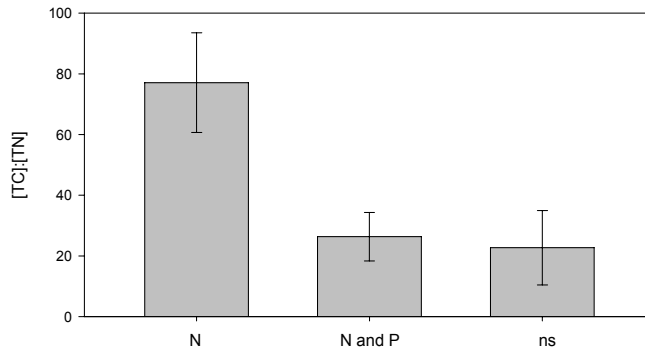


Figure 5-15: Total carbon (mg/L) to total nitrogen molar ratios at N-limited sites (n = 5), NP-co-limited sites (n = 21) and sites limited by neither nitrogen nor phosphorus (n = 9). Error bars represent standard error.

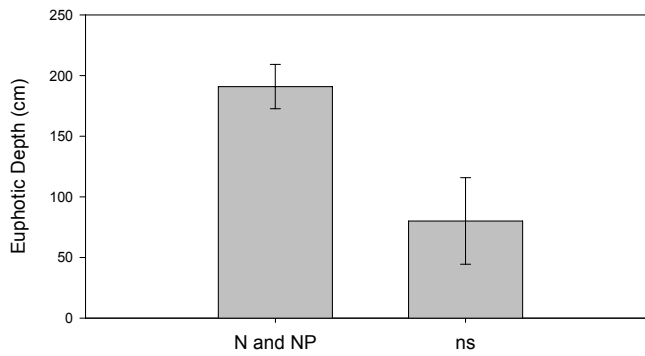


Figure 5-16: Mean euphotic depths (m) at nutrient-limited (N-limited or NP-co-limited; n = 20) and non nutrient-limited sites (n = 3) in the wetlands and river.

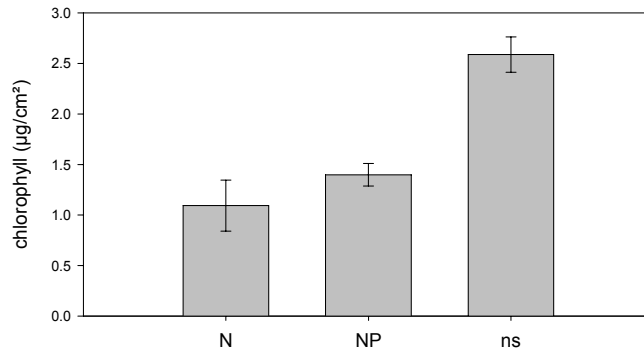


Figure 5-17: Periphyton biomass, expressed as chlorophyll ( $\mu\text{g}/\text{cm}^2$ ), in N-limited wetlands ( $n = 22$ ), NP-co-limited wetlands ( $n = 112$ ), and wetlands limited by neither N nor P ( $n = 46$ ), over two-week durations in 2007 and 2008, on surfaces not enriched by nutrients.

## Discussion

The nutrient limitation observed in the NDS experiments was overwhelmingly NP co-limitation, with some N-limitation. Phosphorus was never limiting. This is similar to the situation in other Manitoba wetlands, including Delta Marsh (Hertam 2010; Bortoluzzi et al. in prep), Netley-Libau Marsh, and Oak Hammock Marsh (Bortoluzzi et al. in prep), and smaller prairie ponds (Kolochuk 2008). However, it goes against the prevailing dogma that freshwater is P-limited (Schindler 1977; Hecky and Kilham 1988; Carpenter *et al.* 1992; Lampert and Sommer 1997; Dodds 2002; Kalff 2002; Dodson 2005; Brönmark and Hansson 2005; Howarth and Marino 2006). This phosphorus paradigm is based on experimentation in deep, oligotrophic systems over annual or multi-annual time scales (Sterner 2008). It should not be surprising that in wetland environments, which can have higher overall nutrient concentrations, shallower depths, allowing for greater phosphorus release from the sediments (Scheffer 1998; Søndergaard *et al.* 2003; Dunne and Reddy 2005) and higher rates of denitrification (Broderick *et al.* 1988; Saunders and Kalff 2001; Poe *et al.* 2003), and long residence times, allowing for phosphorus to be highly recycled (Barica 1987; Bortoluzzi *et al.* in prep) would be differently limited. These NDS experiments support the notion of nitrogen deficiency in wetlands.

Only one nutrient can be limiting to an organism at one given time (Borchardt 1996), but this does not preclude the possibility of co-limitation. There are at least two scenarios in which this NDS bioassay could indicate NP co-limitation: either certain algal species were limited by nitrogen and others by phosphorus, or the entire algal assemblage was first limited by one nutrient, and then, after it was provided in excess, the second nutrient became limiting. The former does not seem likely, because if phosphorus limited species were present, one would expect to observe them thriving on the phosphorus-only treatment. The latter scenario is supported by the fact that NP-co-limited site frequently had elevated

algal growth on nitrogen-only treatment (but higher-still growth on the NP treatment). This suggests nitrogen was limiting until it was supplied in excess by the assay, at which point phosphorus became limiting.

Although NP co-limitation was more common than N-limitation, there was some evidence that nitrogen played a more important role. At sites with higher water column concentrations of total nitrogen, interestingly, phosphorus did not become the limiting nutrient. Rather, there was no significant nutrient limitation seen at these sites.

The presence of certain submersed macrophyte species supports the prevalence of N limitation in these wetlands. Specifically, bladderwort (*Utricularia* sp.), a carnivorous plant with the ability to obtain nitrogen by digesting animal protein, was common or ubiquitous (see Chapter 6; Table 6-?). *Utricularia* has a competitive advantage where nitrogen is limiting (Ulanowicz 1995), so, although precise nutrient requirements differ between macrophytes and algae, its presence suggests overall nitrogen deficiency in Summerberry wetlands.

There was no difference in any N:P nutrient ratio between N-limited and NP limited sites. However, wetlands can be dynamic with respect to nutrient fluxes from the sediment and atmosphere, and so the one-time sample on which the nutrient ratios were based may not have represented average nutrient conditions over the full two-week NDS deployment period. The TC:TN molar ratio was useful in separating N-limited sites from NP-co-limited or non-nutrient-limited sites.

Caution should be taken when using nutrient ratios to predict nutrient limitation. Different ratios (TN:TP, NH<sub>3</sub>:TRP, DIC:TRP, TC:TN) produced very different nutrient limitation status predictions. Predictions from any of the aforementioned ratios correlated poorly with the observed response by algae. Predictions based on ratios involving inorganic nutrients tended to be somewhat

more accurate than those based on ratios involving organic nutrients. The values on which these predictions were based was developed on research done on the open ocean by Redfield (1958) and have been subsequently modified based on research in deepwater lakes (Healey and Hendzel 1980; Kilham 1990; Hecky *et al.* 1993), where inorganic portions can make up nearly the entire pool of total nitrogen and phosphorus. The poor decomposition conditions of a wetland environment, however, can result in the bulk of nitrogen and phosphorus being found in organic form. At Summerberry, ammonia (for the purposes of approximation, assumed to comprise the majority of inorganic nitrogen) represented only 1% of TN, while TRP made up 44% of TP. Organic nitrogen in particular can be biologically unavailable to algae which require inorganic nutrition. So although TN:TP ratios were high, predicting P-limitation, much of the nitrogen was in organic forms potentially unavailable to plant life. Therefore ratios of inorganic nutrients could be more appropriate, as they represent the portion of nutrients which are available to algae.

Shallow wetlands sites were more likely to be limited by neither nitrogen nor phosphorus. The supply of nitrogen at these non-nutrient-limited sites was higher than at sites where nutrients were limiting. Light, then, may have become limiting to plant growth at sites where water was low and nitrogen was not insufficient. Shallow water is more easily mixed with sediment by wind, decreasing water clarity and thereby decreasing the euphotic depth. Indeed, sites showing no nutrient limitation had significantly shallower euphotic depths.

Euphotic depth as calculated with Secchi depth was a better indicator of nutrient limitation than euphotic depth as measured by a light sensor. This may be due in part to the effect of submersed plants shading the meter, whereas these plants are moved aside by a Secchi disc. Also, calculating the euphotic depth as 1% of light incident to the water's surface has been argued to be an arbitrary value, in that the light requirements of plants do not change when the sun is lower in the sky or is covered by cloud.

Interestingly, turbidity was not a good indicator of nutrient limitation or non-limitation. It may be that turbidity in shallow wetlands changes rapidly, and an instantaneous biweekly snapshot did not represent the average turbidity at a site over two weeks. Indeed, dissolved organic carbon concentration was a good predictor of nutrient limitation. Increased DOC, like increased turbidity, could be a result of sediment resuspension, but this parameter may drop more slowly after wind events than turbidity and therefore be a better indicator of light limitation. However, this does not explain why the combined Secchi – light sensor euphotic depth measurement, taken at the same interval as turbidity, was able to predict nutrient limitation.

The back channel was a site not limited by either nutrient, yet the euphotic depth was calculated as over four meters. In this case, light was unlikely to have been limiting, since flows from the more turbid Saskatchewan River into the channel were lowest at this time. However, another nutrient besides N or P could have been limiting. More experimentation, involving NDS with other nutrients, would have to be conducted to understand what might be occurring here.

The growth of periphyton in the controls was higher in wetlands not limited by N or P than in nutrient-limited wetlands. This was unexpected, considering there was no significant difference in nutrient ratios, and the water clarity was lower than at nutrient-limited sites. However, the absolute concentrations of total nitrogen were higher at these sites, leading to a periphyton response of increased growth.

It is surprising that while the growth of periphyton increased at sites not limited by N or P, the growth of phytoplankton was unchanged. Why was there no effect on phytoplankton at sites where periphyton was responding to an increase in available nitrogen? Perhaps the effect of the increase in nitrogen was counteracted by the decrease in available light. The periphyton, growing only

10 cm below the surface, would have been little affected by decreased light levels, whereas phytoplankton would be present throughout the water column and those lower down would be limited by shading.

## Conclusion

Based on these data, periphyton growth in the Summerberry wetlands was mostly limited by nitrogen supply. Light may have become limiting where water column concentrations of nitrogen were higher. Phosphorus was not a limiter of the periphyton community in this environment.

I suggest that NDS results should be taken as a better indicator of nutrient limitation than the nutrient ratios. Firstly, NDS represent a practical, biological test of what nutrients are available to algae. Nutrient pools in the water may not reflect what is actually available to growing algae, but rather what remains after some has been taken up. Secondly, nutrient ratios are based on instantaneous measurements, whereas NDS, on the other hand, are integrated over a longer period of time. Wetlands are dynamic systems where nutrients can change form and location quickly, and the snapshot of nutrient levels provided by a water sample may not adequately explain what occurs over a longer interval.

## Chapter 6: Vegetation

### Introduction

The vegetation of northern deltaic wetlands has not been well described. The Canadian Wildlife Service produced a comprehensive series of reports on the more northerly Peace Athabaska Delta (Dirschl 1972; Dirschl *et al.* 1974). However, descriptions of wetland vegetation in the southern boreal plains region are lacking. Dirschl and Coupland (1970) described vegetation in the upper Saskatchewan River Delta, above The Pas moraine, but there has been no description of vegetation in the wetlands of the younger, lower SRD where deltaic processes are ongoing. This chapter will address these gaps by describing major species and quantifying above- and below-ground wetland plant biomass in the Summerberry Marshes.

Describing the current state of the vegetation of the SRD as a reference condition will be valuable in case of further changes due to anthropogenic modifications to flows on the Saskatchewan or due to climate change. Additionally, should studies relating to drawdown continue at Summerberry, this description will provide a pre-manipulation vegetation condition.

### Methods

Above-ground vegetation sampling to determine maximum standing crop took place during mid-August 2007. Within each wetland, two stands of each of four stand types (cattail, bulrush, phragmites, and mixed whitetop / sedge / horsetail) were randomly selected based on four band multispectral QuickBird satellite imagery (2m resolution) obtained in late June / early July of each year. Within



each chosen stand, three randomly selected sites were sampled. At every site, all vegetation within a known area ( $1\text{m}^2$  quadrat for cattail and  $0.25\text{m}^2$  quadrat for other stand types) was clipped at the ground. The sample was sorted between live and dead material and the latter was discarded. The remaining material was sorted between the dominant vegetation type of the stand, and other species of plants. Both portions were stored in paper bags to be oven-dried at  $100^\circ\text{C}$  and weighed.

A fifth stand type consisted of beds of submersed vegetation in open water bays. Above-sediment portions of plants were raked up and gathered by hand from within a 0.5m diameter bottomless barrel (Figure 6-2). The samples obtained were rinsed to remove non-plant material, identified and sorted to the genus level, dried at  $100^\circ\text{C}$ , and weighed.

Below-ground biomass sampling of vegetation took place when at its seasonal maximum (van der Valk and Davis 1978b; Murkin and Murkin 1989) in late September 2007. The same emergent stands were sampled as with aboveground sampling, but submersed vegetation was not sampled. Rather than using quadrats, a root coring device (Figure 6-3) was employed. The root corer consisted of a steel cylinder with a 15cm diameter and a length of approximately 0.5m. Along the bottom edge of the cylinder, a hacksaw blade was welded, which cut through roots as the device was twisted into the ground. The clay below the vegetative layer formed a seal which held the root core in the cylinder as it was lifted out of the ground, and a plunger assisted in removing the core from the corer. Three cores were obtained from each stand.

Root cores were washed, using an automatic root washing machine (Figure 6-4), to remove sediment and clay and leave only the vegetative material. The root washer consisted of four cylindrical wire mesh cages rotating within a tub of continuously refreshed river water. Root cores were placed in the mesh cage and rinsed in this water for five to twenty minutes. The cleaned cores were then

frozen until such time as the live roots could be sorted from the dead plant material, dried at 100°C, and weighed.



Figure 6-1: Research assistant Sheila Atchison harvesting aboveground vegetation from a quadrat in a whitetop / sedge / horsetail stand.



Figure 6-2: Research assistants Martin Blades and Jared Knockaert sampling submersed vegetation from a barrel in a drawdown wetland.



Figure 6-3: The root coring device with a below-ground biomass sample. Photography by Dale Wrubleski.



Figure 6-4: The automatic root washing machine. A submersible pump in the river forced water into the grey tub through the white hose in the foreground. An electric motor (right) caused the wire mesh cylinder to rotate. Root cores were placed in the four separate cages within the cylinder. Dirty water drained through the hose at the rear.

## Results

Table 6-1 lists all plants species identified in aboveground vegetation samples and elsewhere at Summerberry. Plants are divided into emergent and transitional species, and submersed and floating-leaved species. Because some algae, including the large algae stonewort (*Chara* sp.) and filamentous metaphytic algae, were sampled along with submersed vascular plants, a category for large non-vascular plants was included.

Both the native common cattail (*Typha latifolia*) and the introduced narrow-leaved cattail (*T. angustifolia*) were present at Summerberry (Figure 6-5). *T. latifolia* was seen comprising large stands, extending far inshore from open water areas into shallow or nearly dry zones. *T. angustifolia* was typically observed directly abutting open water bays. *T. glauca*, the hybrid formed from these species, was also observed but was least common. Many cattail stands were “floating”: their root mats were detached from the sediments at the bottom of the marsh.

In addition to those plant species identified in vegetation samples, other species were noted at Summerberry. Several species of willows (*Salix* spp.) bordered each wetland, often encroaching into the sedge band surrounding most wetlands. Reed canary grass (*Phalaris arundinacea*) grew near the riverbank on the wetland side. Channel banks and dikes between wetlands were dominated by deciduous trees and shrubs.

Crowfoot (*Ranunculus* sp.) and water marigold (*Megalodonta* sp.) were observed in the back channel. Water smartweed (*Polygonum* sp.) was seen along the edges of the back channel and river. White waterlily (*Nymphaea* sp.) and wild rice (*Zizania aquatica*) were seen in neighbouring wetlands but not in sampled wetlands. Floating aquatic liverworts were observed sporadically but were not present in samples.

The means of the aboveground biomass (Figure 6-6) for each of the five stand types were shown to be significantly different via Van der Waerden chi square ( $p < 0.0001$ ). Stands of phragmites had the highest aboveground biomass, and submersed vegetation, the lowest.

Bulrush, cattail, and phragmites stands were predominantly composed of that type vegetation (Figure 6-7). Stands ranged from 100% type vegetation to less than half of the biomass being of the type genus, in the case of bulrush and phragmites, and less than one quarter, in the case of cattail. The percentage of total biomass which was the main stand vegetation is shown in Table 6-2. There was no significant difference between stand types in terms of percentage of total biomass which was main stand vegetation ( $p = 0.22$ ).

Sedge stands were dominated by sedge: as seen in Table 6-3, sedge comprised nearly 60% of the overall biomass of sedge stands. These stands ranged from 100% sedge, to sedge with horsetail, sedge with whitetop, and sedge with horsetail and whitetop. There were even two sites where horsetail and whitetop were present together and sedge was absent. Because this variability occurred within a quadrat's throw and without a visible barrier, all vegetation combinations described above are all treated as one stand type referred to as sedge. Examples of the variability within sedge stands can be seen in Figure 6-8.

The composition of beds of submersed vegetation was highly variable; examples of the variability can be seen in Figure 6-9. Table 6-4 shows average and maximum biomass of submersed plant genera, ranked in order from most to least ubiquitous. The small emergents class included *Sparganium natans* and *Sagittaria* spp. which have the majority of their biomass below the water's surface and only a small emergent section. The duckweed class included *Spirodella polyrhiza*, *Lemna trisulca*, and *Wolffia globosa*. There was frequently

an appreciable amount of filamentous metaphytic algae entwined with the duckweed which could not be separated.

*Potamogeton*, *Ceratophyllum*, and *Stuckenia* were most frequently present at sampled sites. The submersed plants with the highest biomass across the wetlands were *Chara*, *Ceratophyllum*, *Potamogeton*, and *Stuckenia*. The highest biomasses at a single site belonged to *Chara*, *Ceratophyllum*, duckweed / metaphyton, and *Myriophyllum*. *Naja*, *Hippuris*, and *Nuphar* were the least abundant submersed plants by any measure.

Using the sum of mean above- and mean belowground biomass, the percentage of biomass that was aboveground was calculated. Bulrush stands were unique in having only a quarter of total biomass aboveground; Phragmites, cattail and sedge stands had just over 40% biomass aboveground (Table 6-5).

#### Qualitative Observations on the Effects of Drawdown

This investigation did not include a description of the vegetation condition during drawdown, so no quantitative comparisons can be made between high and low water wetlands. However, some general qualitative observations were made on the effects of drawdown on wetland macrophytes.

These observations suggested that a single year of partial drawdown was insufficient to affect the distribution of emergent vegetation. Germination of emergent plants in open water areas was not observed at any time during drawdown, and there was no increased interspersion of water and vegetation. The partial drawdown at Summerberry did not expose mudflats.

Qualitative observations also suggested that biomass in emergent vegetation stands is unaffected by a single-year partial drawdown. However, an increase in



primary production of submersed macrophytes was noted in drawdown wetlands. There may have been an overall increase in submersed vegetation in 2008 as compared to 2007, but increases in drawdown wetland submersed macrophyte production seemed greater than increases in control wetlands.

Mean belowground biomass was significantly different across the four stand types ( $p = 0.001$ ; Figure 6-10). Bulrush had the highest belowground biomass and sedge the lowest.



Figure 6-5: Cattails in the Summerberry Marshes. From left: *Typha latifolia*, native common cattail; *T. X glauca*, hybrid cattail, exhibiting characteristics of both parents; *T. angustifolia*, introduced narrow-leaved cattail.

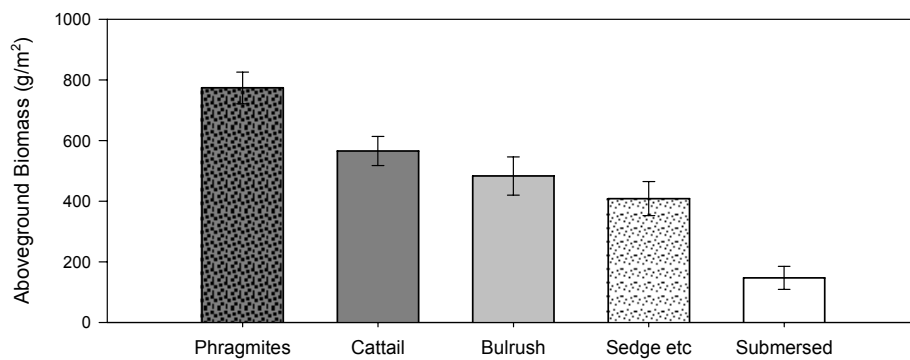


Figure 6-6: Mean aboveground biomass in stands of phragmites (n = 28), cattail (n = 33), bulrush (n = 19), sedge (n = 24), and submersed vegetation (n = 53). Error bars show standard error (n = 157).



Figure 6-7: Phragmites stands (top) and bulrush stands (bottom) in the Summerberry Marshes exhibited monodominance.



Figure 6-8: Some examples of the variability of composition of sedge stands, including (clockwise from top left) sedge dominant with horsetail present; horsetail dominant with sedge present; sedge, horsetail and whitetop grass present; whitetop grass dominant with sedge present.

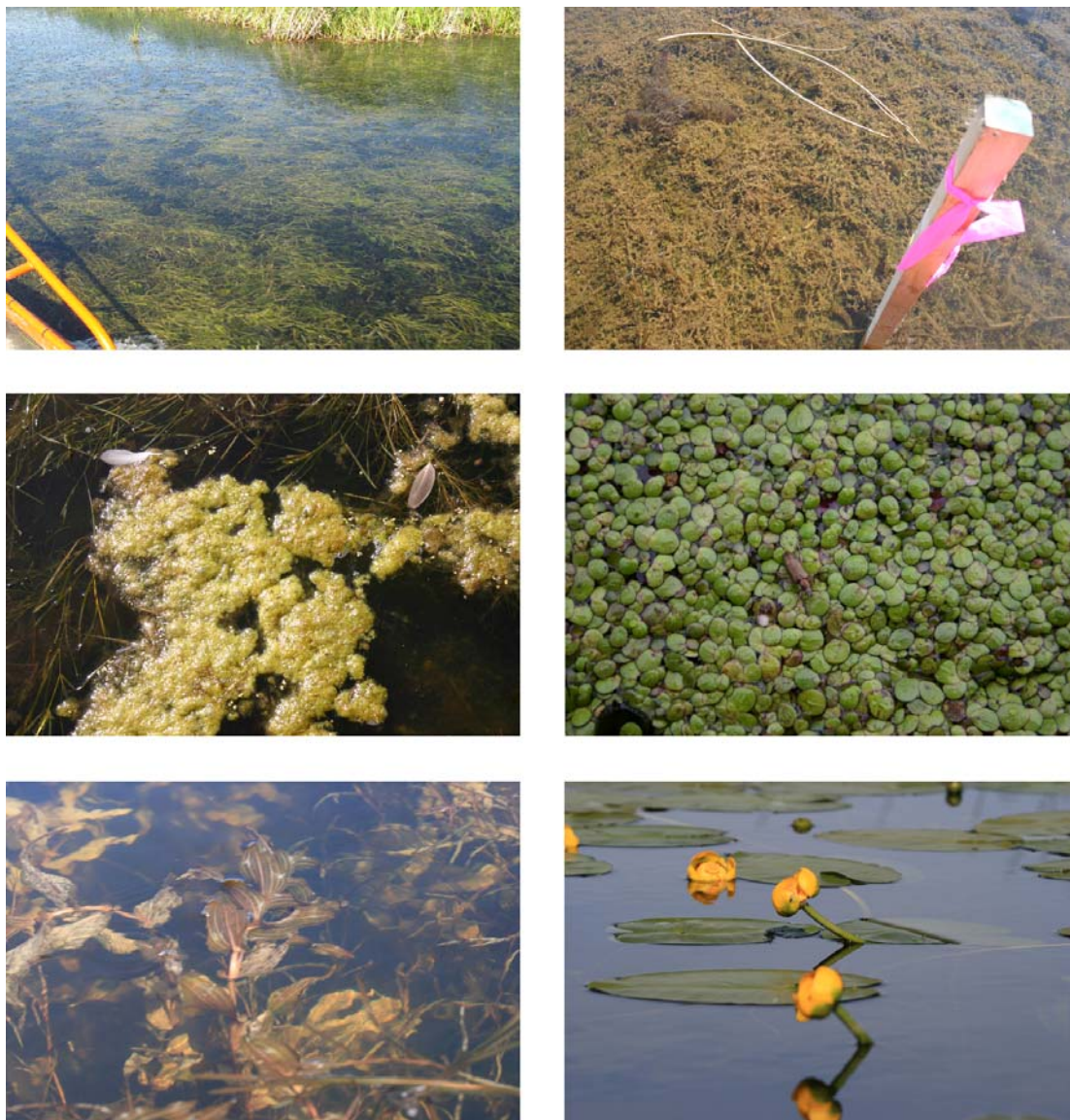


Figure 6-9: Several examples of the variability of species composition in stands of submersed and floating-leaved plants.

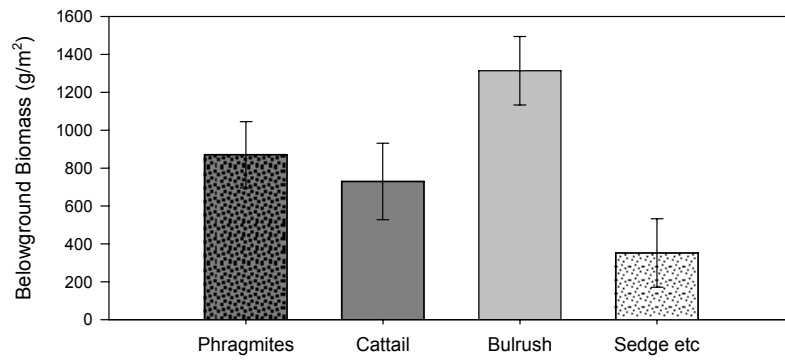


Figure 6-10: Mean belowground biomass in stands of Phragmites (n = 32), cattail (n = 24), bulrush (n = 30), and sedge (n = 30). Error bars show standard error (n = 116).

Table 6-1: Major plant species observed in the Summerberry region, with scientific and common nomenclature according to Laring (2003). A cross (†) precedes those species which were present in vegetation samples. An asterisk (\*) precedes the most abundant member of a genus where more than one species was present.

Emergent and Transitional		Submersed and Floating-leaved	
†	<i>Acorus americanus</i>	sweet flag	hornwort
†	<i>Bidens cernua</i>	nodding beggarticks	Canada waterweed
†	<i>Carex aquatilis</i>	water sedge	mare's-tail
† *	<i>Carex lacustris</i>	lakeshore sedge	ivy-leaved duckweed
†	<i>Equisetum fluviatile</i>	swamp horsetail	water marigold
†	<i>Gallium triflorum</i>	sweet-scented bedstraw	little water-milfoil
†	<i>Phalaris arundinacea</i>	reed canary grass	slender naiad
†	<i>Phragmites australis</i>	giant reed grass	yellow pondlily
†	<i>Polygonum amphibium</i>	water smartweed	Leiberg's waterlily
†	<i>Potentilla palustris</i>	marsh cinquefoil	floating-leaf pondweed
†	<i>Sagittaria cuneata</i>	arrow-leaved arrowhead	Richardson's pondweed
† *	<i>Sagittaria latifolia</i>	broad-leaved arrowhead	flat-stemmed pondweed
†	<i>Scholochloa festuacea</i>	whitetop grass	crowfoot
† *	<i>Scirpus acutus</i>	hardstem bulrush	larger duckweed
†	<i>Scirpus validus</i>	softstem bulrush	sago pondweed
†	<i>Scutellaria galericulata</i>	marsh scullcap	flat-leaved bladderwort
†	<i>Sparganium eurycarpum</i>	giant bur-reed	small bladderwort
† *	<i>Sparganium natas var minimum</i>	slender bur-reed	common bladderwort
† *	<i>Typha latifolia</i>	common cattail	globose water-meal
†	<i>Typha angustifolia</i>	narrow-leaved cattail	wild rice
†	<i>Typha (X) glauca</i>	hybrid cattail	
<b>Large Non-vascular</b>			<b>Trees and Shrubs</b>
†	<i>Chara</i> sp	stonewort	balsam poplar
†	<i>Bryophyta</i> sp	mosses	sandbar willow
	<i>Marchantiophyta</i> sp	liverworts	beaked willow
			low-bush cranberry



Table 6-2: The percentage of the total biomass which was comprised of the stand's type vegetation. The means, standard errors, and minimums are presented for each stand type: bulrush (n = 19), cattail (n = 33) and phragmites (n = 28).

<b>Stand Type</b>	<b>Mean Percentage Of Total Biomass Which Is Type</b>	<b>Standard Error</b>	<b>Minimum</b>
Bulrush	93.5%	4.3%	46.3%
Cattail	84.2%	3.3%	24.0%
Phragmites	86.5%	3.5%	46.2%

Table 6-3: The makeup of a sedge stand: biomass of sedge, horsetail and whitetop grass within Sedge etc stands.

<b>Plant Type</b>	<b>Biomass (g/m<sup>2</sup>)</b>	<b>Percent of Total Biomass</b>	<b>Percent of Sites Present</b>
Sedge	230.7	58.8%	92%
Horsetail	76.2	19.4%	64%
Whitetop	53.4	13.6%	64%
Other	31.8	8.1%	56%

Table 6-4: Biomass of submersed plants and the percentage of sites where genera was present.

<b>Plant Type</b>	<b>Mean Biomass across open water (g/m<sup>2</sup>)</b>	<b>Maximum Biomass (g/m<sup>2</sup>)</b>	<b>Percentage Sites Where Present</b>
<i>Potamogeton</i>	26.6	140.5	89.6%
<i>Ceratophyllum</i>	24.6	236.6	62.5%
<i>Stuckenia</i>	22.0	119.7	58.3%
<i>Utricularia</i>	4.8	89.4	47.9%
<i>Myriophyllum</i>	13.8	208.7	35.4%
small emergents	8.0	171.6	31.3%
<i>Elodea</i>	10.1	153.6	27.1%
duckweed and metaphyton	20.7	226.4	25.0%
<i>Chara</i>	28.6	493.8	16.7%
<i>Nuphar</i>	0.9	35.7	8.3%
<i>Hippuris</i>	0.3	13.0	4.2%
<i>Najas</i>	0.2	1.8	2.1%

Table 6-5: Percentage of the total (above- plus below-ground) biomass of emergent vegetation that is aboveground, or the shoot to root ratio.

<b>Stand Type</b>	<b>Percentage Of Biomass Which Is Aboveground</b>
Bulrush	24.1%
Cattail	41.5%
Phragmites	43.0%
Sedge	43.6%

## Discussion

Bladderwort (*Utricularia* spp) was common in the wetlands. However, it was most frequently observed in small, less than ~10 m<sup>2</sup> patches of open water amongst emergent vegetation, not in the larger open water bays where the sites for sampling submersed vegetation were located (Figure 6-11). Mare's-tail (*Hippuris vulgaris*), though not so ubiquitous as bladderwort, followed the same pattern. Both these groups are therefore likely underrepresented in the submersed data. The alga *Chara* is likely overrepresented; its biomass is misleadingly large. *Chara* excretes calcium and magnesium carbonate which are deposited on the cell wall, making it much heavier than a similar volume of other submersed plants species (Lee 1989).

It is important to remember that belowground sampling did not occur simultaneously with aboveground sampling, but rather up to a month later, in order to sample each section at its seasonal maximum standing crop (Murkin and Murkin 1989). During this later sampling period, it can be expected that aboveground biomass would have been smaller than the measured values, because starches and other nutrients would have been transported from shoots to roots. The cited percentages would therefore likely have been higher had all biomass sampling been conducted in September and lower had all been conducted in August.

The timing of belowground sampling may explain the difference seen in shoot to root ratio of bulrush as compared to the other types of emergent vegetation. Cattail, phragmites and sedge stands had lost their green colour and aboveground vegetation appeared dead or dying. Bulrush culms, conversely, still appeared dark green and healthy; photosynthesis may have been ongoing at this time and the reliance on belowground stores of energy could have been lower. Because of differences between species, there is no ideal time to sample

just as the belowground biomass is at its highest: each species reaches its maximum at slightly different times.

Submersed macrophyte biomass at Summerberry was consistent with other published values (Table 6-6). The percentage of emergent macrophyte biomass which is above-ground was within or near published ranges. Bulrush, however, had higher, and cattail and phragmites had lower above-ground biomass than other North American wetlands. There is a dearth of published macrophyte biomass data, especially for boreal wetlands; in the absence of more data, it is difficult to be sure whether these wetlands do in fact differ from more southerly, prairie wetlands.



Figure 6-11: *Utricularia vulgaris*. High densities of *Utricularia* spp. were observed in small patches of open water amongst emergent vegetation; lower densities were sampled from large open bays.

Table 6-6: A comparison of above-ground macrophyte biomass ( $\text{g/m}^2$ , and percent of biomass which is above-ground) between the Summerberry Marshes, Delta Marsh (Shay and Shay 1986) and Eagle Lake Marsh (van der Valk and Davis 1978b).

Macrophyte	Summerberry Marshes, SRD, Manitoba		Delta Marsh, Manitoba		Eagle Lake Marsh, Iowa
	$\text{g/m}^2$	%	$\text{g/m}^2$	%	$\text{g/m}^2$
<i>Typha</i> sp	566	42	781 - 1754	41 - 57	758 - 1549
<i>Scirpus</i> sp	482	24	290 - 365	18 - 47	
<i>Phragmites australis</i>	774	43	812	30 - 40	
Submersed	146				91 - 260

## Conclusion

This study is the first quantitative characterisation of marsh vegetation communities in the lower SRD. It contributes to the collective knowledge of boreal marshes and shallow open waters which have been, thus far, poorly described in comparison with boreal peatlands.

This description of the current state of the vegetation at Summerberry provides a pre-manipulation condition for studies on the effects of drawdown on vegetation in the same wetlands, which is examined by Baschuk (2010).

Finally, this work provides a valuable reference condition prior to changes facing SRD. As mentioned previously, the SRD is likely to face continued drying through climate change (Yulianti and Burns 1998; Mehdi *et al.* 2002), increased upstream demands for water (Partners FOR the Saskatchewan River Basin 2008), and natural deltaic successional processes (Dirschl 1972b). Wetlands in the SRD also face the threat of invasive species, such as narrow-leaved and hybrid cattail. Although these species are currently present at Summerberry, they do not yet dominate as they do in other Manitoba wetlands including Delta Marsh (de Geus 1987) and Netley-Libau Marsh (Grosshans 2004).

## Chapter 7: Research Synthesis

*Hypothesis A: Drawdown will affect water quality by increasing water column turbidity and nutrient concentrations because shallower water allows for more sediment – water mixing by wind.*

Hypothesis A was supported. Lowering wetland water levels reduced the wind velocity necessary to provide sufficient energy to resuspend bottom sediments. Sediment resuspension therefore became more frequent, and this resuspension caused an increase in water column turbidity and disturbance of the sediment – water interface such that the water content of surface sediments increased. Mixing of sediments and water caused an increase in turbidity, and an increase in the eflux of organic carbon from the sediment into the water column. Sediment mixing also increased water column concentrations of nitrogen and phosphorus, in both organic and inorganic forms.

Algal photosynthesis probably increased in response to increased nutrient availability, which consumed hydrogen ions and shifted the carbonate buffering system such that the concentrations of dissolved inorganic carbon, calcium, and magnesium increased. Phytoplankton proliferation may have further contributed to turbidity.

Conductivity and the concentrations of major ions increased during drawdown due to the increased effects of evaporative concentration at shallower depths.

*Hypothesis B: Increases in nutrient concentration due to drawdown will increase algal primary production, because algal communities in deltaic wetlands, like those in many prairie wetlands, are nutrient limited.*

Hypothesis B was supported. Prior to drawdown, wetland periphyton communities were N-limited or NP-co-limited. The input of nitrogen from the

sediment to the water column in combination with an increase in turbidity resulted in a shift from nutrient limitation to light limitation of periphyton growth.

The correlation between concentrations of total nitrogen and total phosphorus in the water column with sediment water content, when paired with the lack of correlation between concentrations of inorganic forms of nitrogen and phosphorus with sediment water content, supports the NP-limitation. Although organic forms of nitrogen and phosphorus increased as the sediment–water interface was disrupted by wind and wave action, inorganic forms were taken up by primary producers as soon as they were available.

*Hypothesis C: Turbidity, nutrient concentrations and algal primary production trends will be related to site depth because sediment resuspension is more likely to occur in shallow sites. Shallow sites in wetland basins not undergoing a drawdown should therefore be similar to drawdown sites in these parameters.*

Hypothesis C was supported. Although drawdown wetlands had significantly increased nutrient concentration, ion concentration, conductivity, turbidity, and algal production than control wetlands and the control season, this work showed that these trends were correlated with depth, rather than being inherent to the drawdown condition.

*Hypothesis D: The chemical and physical properties of wetland water and sediment will be correlated to distance from channels of the Saskatchewan River, because the river influences wetlands through flood events and seepage through levees.*

Hypothesis D was supported in part. Ion concentrations and specific conductance increased with distance from the nearest channel, indicating that ion-dilute water was seeping through levees to sites at the wetland peripheries, whereas sites in basin centres were more influenced by groundwater or the



effects of evaporative concentration. Sediment composition and nutrient concentrations were related to distance from the river, with highly inorganic and nutrient-rich clays at sites closest to the river, and organic peats farther from the river. This suggested that flooding from the Saskatchewan River supplied riverine sediments to wetlands. However, water column nutrient concentrations were not especially related to distance from the river, suggesting that the input of river waters through flood events to wetlands do not have a localised effect on water chemistry throughout the summer or over several years.

#### Contribution to the understanding of effects of water level variation on wetlands

Some obvious parallels exist between this project and the Marsh Ecology Research Program (MERP; Murkin *et al.* 2000) which took place at Delta Marsh, Manitoba through the 1980s. Indeed, both studies involved artificial water level manipulation, including drawdown, in replicated control and treatment basins.

However, comparisons between the two studies are compounded by several factors. Firstly, MERP experimentation took place over ten years, simulating an entire wet-dry cycle; while research at Summerberry was conducted in only two field seasons. Additionally, MERP was based around the key stage of a complete drawdown: a total dewatering of wetlands and exposure of wetland sediments, involving water level changes of up to 91 cm. Summerberry water level manipulation was limited to a partial drawdown, involving a mean water level change of only 32 cm, and sediments were not exposed.

During the drawdown years of MERP, the four dominant emergent macrophytes – bulrush, cattail, *Phragmites* and whitetop – germinated from the seedbank. Several annual species also became established on mudflats and accounted for more than half of wetland aboveground biomass. Though the same emergent species were present at Summerberry, colonisation of new areas by emergent plants germinating from the seed bank was not observed, in drawdown wetlands

or elsewhere. Mudflats were not exposed and therefore there was no colonisation by mudflat annuals.

When MERP wetlands were reflooded after a period of total drawdown, a pulse of nutrients was released from the sediments, as a result of increased decomposition due to aeration of exposed sediments, and leaching from the litter of the annual mudflat vegetation killed by high water levels. Summerberry wetlands also had increased water column concentrations of most nutrients after partial drawdown, but this cannot be explained by inputs from drowned annuals or by aeration of exposed and dried sediments. Rather these nutrient increases occurred where depths were shallow enough to allow sediment resuspension by wind. It may be that wind and wave action did aerate sediments and contribute to increased decomposition, but this was not measured.

MERP found no relationship between site depth and phytoplankton per unit of volume, whereas this study found a highly significant relationship. In MERP, nutrients from organic litter were already decomposed aerobically during drawdown, and then released from the sediment across the entire wetland during reflooding. At Summerberry, there may or may not have been increased aerobic decomposition at shallow sites, but regardless, nutrients were not flushed from the sediments, basin-wide, by an influx of reflooding water. Rather, the sediment-water interface was disturbed by wave action at shallow sites only, and therefore nutrient-deficient algae were supplemented with nitrogen at these sites only. Wind resuspension of sediments was likely less important in MERP cells than in the study wetlands at Summerberry as the areas of the former were 20 to 60 times smaller than those of the latter and a correspondingly smaller effective fetch.

MERP vegetation studies focused mainly on emergent macrophytes and annual mudflat species, and little work was done on the effect of drawdown on submersed vegetation. van der Valk (2000) did note that submersed aquatic

plants were adversely affected by a decrease in water level, whereas my qualitative observations suggested an increase in submersed vegetation in drawdown wetlands. The total drawdown in MERP left very few and very shallow open water areas which were poorly suited for submersed vegetation. The partial drawdown at Summerberry maintained depth suitable for submersed vegetation, and production may have been increased by nitrogen concentrations in the water column and sediment.

## Chapter 8: Recommendations

### Recommendations for management

The findings of this work have implications for future management of boreal marshes for waterfowl and fur-bearing mammal production.

Partial wetland drawdown led to a shift from nutrient deficiency to nutrient sufficiency, which increased algal primary production. Because algae can be the base of wetland food webs (Neill and Cornwell 1992; Robinson et al. 2000), it is expected that this increase in primary production should lead to an increase in secondary production. The aquatic invertebrate community may be augmented by greater production at lower trophic levels, providing greater food availability to waterfowl and other vertebrates. Further research on the effects of drawdown on secondary production at Summerberry (Baschuk 2010) examines these trends.

Although this investigation did not include a study into the effects of drawdown on wetland macrophytes, general observations suggest that primary production of submersed vegetation also increases when water levels are lowered. An increase in the density of submersed plants would further improve the suitability of marshes as waterfowl feeding grounds, both by providing habitat for the invertebrates eaten by waterfowl (Krull 1970; Longcore et al 2006) and by directly providing food resources to waterfowl in the form of seeds, vegetative materials and tubers (Murkin and Cadwell 2000). From a food availability perspective, it can therefore be recommended that partial drawdown may be an effective management strategy to increase the suitability of wetland habitat for waterfowl.

Qualitative observations suggested that a single year of partial drawdown was insufficient to affect the distribution of emergent vegetation. Germination of emergent plants in open water areas was not observed at any time during

drawdown, and there was no improved interspersion of water and vegetation. The hemi-marsh condition that provides best cover for duck production (Murkin and Caldwell 2000) did not appear within a year of lowered water levels, which suggests that single-year partial drawdown is not an effective management technique for improving waterfowl brood-rearing habitat. This trend should be monitored over future years of drawdown, and the effects of a total drawdown should also be studied. Other facets of waterfowl habitat suitability, such as nesting grounds and potential for predation, which are outside the scope of this study, should be investigated as well.

General observations showed that the biomass in emergent vegetation stands is also unaffected by single-year partial drawdown. Although more work must be conducted to confirm the validity of this observation, it suggests that, from a food availability perspective, partial drawdown would not be an effective management tool to increase wetland production of muskrat. Ongoing research at Summerberry and throughout the SRD (Ervin in prep) will provide further insight into the effects of drawdown on muskrat populations, predation and food availability.

## Recommendations for future research

### Drawdown and water and sediment chemistry

There is some evidence that drawdown had greater effect on the concentrations of inorganic forms of nutrients than on organic nutrients. For example, total reactive phosphorus was higher in drawdown wetlands than in control wetlands, but total phosphorus was not significantly different. Perhaps sediments became more aerobic when water levels were lowered, allowing for more decomposition of organic compounds. This hypothesis could be tested by measuring dissolved

oxygen at the bottom of the water column and into the sediment, or by decomposition experiments modeled after Kadlec *et al.* (2000).

A general trend that sediment concentration total nitrogen may increase at shallower depths was noted, although it was not significant at the 0.05 probability level. This trend might prove to be significant with a larger sample size of sites or a higher frequency of sampling, and should be further investigated.

### Nutrient limitation in boreal wetlands

The nutrient diffusing substrata experiments conducted at Summerberry highlighted the difficulty of predicting nutrient sufficiency or deficiency based on water column nutrient ratios. The results of the NDS bioassays frequently conflicted with information about nutrient concentrations. This work should demonstrate to future researchers of wetland nutrient limitation the paramount importance of exercising caution when making assumptions about how biota will respond based on nutrient ratios. Bioassays together with nutrient availability information can provide more useful information than either alone. The analysis of elemental ratios within the phytoplankton and periphyton assemblages may also prove more beneficial than water column nutrient ratio values.

For a more complete picture of nutrient limitation in the boreal marshes wetlands, it would be interesting to examine and identify members of the periphyton algal assemblage. What algae are present on control substrata? How do these communities on artificial media compare with those on natural surfaces? When substrata are supplemented with a nutrient like nitrogen, which community members are being selected for? Do these taxa actually comprise an appreciable fraction of the “normal” native wetland flora? What are nutrient requirements of the groups present?

### Drawdown and macrophyte production

The description of submersed and emergent macrophyte production at Summerberry provided a clear snapshot of these communities prior to drawdown. As drawdown research in these wetlands continues, further vegetation sampling, modeled after the techniques in this study, could be conducted to provide a second picture of the biomass and composition of macrophyte communities several years after water levels were first lowered. Distribution of macrophyte species could be mapped before and after manipulation using remotely-sensed data. This information would provide another useful tool for wetland managers aiming to improve the wildlife habitat potential of these and other boreal marshes.

## References

- Allan RJ, Kenney BC. 1978. *Rehabilitation of eutrophic prairie lakes in Canada*. Verhandlungen Internationale Vereinigung für theoretische und angewandte Limnologie 20: 214-224.
- American Public Health Association (APHA). 1998. *Standard methods for the analysis of water and waste water*. Washington, DC.
- Arts MT, Robarts RD, Evans MS. 1997. *Seasonal changes in particulate and dissolved lipids in an eutrophic lake*. Freshwater Biology 38: 525-537.
- Axler RP, Rose C, Tikkanen CA. 1994. *Phytoplankton nutrient deficiency as related to atmospheric nitrogen deposition in north Minnesota acid-sensitive lakes*. Canadian Journal of Fisheries and Aquatic Sciences 51: 1281-1296.
- Barica J. 1987. *Water quality problems associated with high productivity of prairie lakes in Canada: a review*. Water Quality Bulletin 12: 107-114.
- Barica J. 1990. *Seasonal variability of N:P ratios in eutrophic lakes*. Hydrobiologia 191: 97-103.
- Barica J, Kling H, Gibson J. 1980. *Experimental manipulation of algal bloom composition by nitrogen additions*. Canadian Journal of Fisheries and Aquatic Sciences 37: 1175 -1183.
- Baschuk M. 2010. *Effects of water-level management on the abundance and habitat use of waterfowl and marsh birds in the Saskatchewan River Delta, Manitoba, Canada*. MSc Thesis. University of Manitoba.
- Bayley SE, Prather CM. 2003. *Do wetland lakes exhibit stable states?*



- Submersed aquatic vegetation and chlorophyll in western boreal shallow lakes.* Limnology and Oceanography 48: 2335-2345.
- Borchardt MA. 1996. *Nutrients.* In Algal Ecology: Freshwater Benthic Ecosystems. Stevenson RJ, Bothwell ML, Lowe RL, eds. Academic Press.
- Bortoluzzi T, Goldsborough G, McDougal R. *Inorganic nitrogen limits periphyton growth in three large prairie wetlands of south-central Canada.* Unpublished manuscript.
- Bothwell ML. 1988. *Growth rate responses of lotic periphytic diatoms to experimental phosphorus enrichment: the influence of temperature and light.* Canadian Journal of Fisheries and Aquatic Sciences 46: 1293-1301.
- Bourn WS, Cottam C. 1939. *The effect of lowering marsh levels on marsh wildlife.* Transactions of the North American Wildlife and Natural Resources Conference 4: 343-350.
- Brodrick SJ, Cullen P, Maher W. 1988. *Denitrification in a natural wetland receiving secondary treatment effluent.* Waters Research 22: 431-439.
- Brönmark C, Hansson LA. 2005. *The Biology of Lakes and Ponds.* Oxford University Press.
- Campbell CE, Prepas EE. 1986. *Evaluation of factors related to the unusually low chlorophyll level in prairie saline lakes.* Canadian Journal of Fisheries and Aquatic Sciences 43: 846-854.
- Caraco NF, Cole JJ, Likens GE. 1989. *Evidence for sulfate controlled phosphorus release from sediments of aquatic systems.* Nature 341: 316-318.

- Caraco N, Cole JJ, Likens GE. 1990. *A comparison of phosphorus immobilization in sediments of freshwater and coastal marine systems*. Biogeochemistry 9: 277-290.
- Carper GL, Bachmann RW. 1984. *Wind resuspension of sediments in a prairie lake*. Canadian Journal of Fisheries and Aquatic Sciences 41: 1763-1767.
- Clay RT. 1978. *Report on the prevailing conditions and waterfowl potential of the Summerberry Marsh Complex*.
- Cooke SE, Prepas EE. 1998. *Stream phosphorous and nitrogen export from agricultural and forested watersheds on the Boreal Plain*. Canadian Journal of Fisheries and Aquatic Sciences 55: 2292-2299.
- Craft CB. 1997. *Dynamics of nitrogen and phosphorus retention during wetland ecosystem succession*. Wetlands Ecology and Management 4: 177-187.
- Dale AR, Chambers PA. 1996. *Growth rate and biomass responses of periphytic algae to phosphorus enrichment in experimental flumes, Athabasca Rivre, seasonal variation, 1993 and 1994*. Northern River Basins Study Project Report No 68.
- de Geus PM. 1987. *Vegetation changes in the Delta Marsh, Manitoba between 1948 - 1980*. MSc Thesis, University of Manitoba.
- DiNicola DM. 1996. *Periphyton responses to temperature at different ecological levels*. In Algal Ecology: Freshwater Benthic Ecosystems. Stevenson RJ, Bothwell ML, Lowe RL, eds. Academic Press.
- Dirschl HJ. 1970. *Ecology of the vegetation of the Saskatchewan River Delta*. PhD thesis, University of Saskatchewan.

- Dirschl HJ. 1972a. *Evaluation of ecological effects of recent low water levels in the Peace-Athabasca Delta*. Canadian Wildlife Service Occasional Paper No 13.
- Dirschl HJ. 1972b. *Geobotanical processes in the Saskatchewan River Delta*. Canadian Journal of Earth Sciences 9: 1529-1549.
- Dirschl HJ and Coupland RT. 1972. *Vegetation patterns and site relationships in the Saskatchewan River delta*. Canadian Journal of Botany 50: 647-675
- Dirschl HJ and Dabbs DL 1969. *A contribution to the flora of the Saskatchewan River Delta*. Canadian Field Naturalist 83:212-228.
- Dirschl HJ, Dabbs DL, Gentle GC. 1974. *Landscape classification and plant successional trends: Peace-Athabasca Delta*. Canadian Wildlife Service Report Series No 30.
- Dodds WK. 2002. *Freshwater Ecology: Concepts and Applications*. Academic Press.
- Dodson SI. 2005. *Introduction to Limnology*. McGraw Hill.
- Dunne EJ, Reddy KR. 2005. *Phosphorus biochemistry of wetlands in agricultural watersheds*. In *Nutrient Management in Agricultural Watersheds: A Wetlands Solution*. Dunne EJ, Reddy KR, Carton OT, eds. Wageningen Academic Publishers, Wageningen, Netherlands, 105-120.
- Elser JJ, Bracken MES, Cleland EE, Gruner DS, Harpole WS, Hellebrand H, Ngai JT, Seabloom EW, Shurin JB, Smith JE. 2007. *Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine*

*and terrestrial ecosystems.* Ecology Letters 10: 1135-1142.

Elser JJ, Marzolf ER, Goldman CR. 1990. *Phosphorus and nitrogen limitation of phytoplankton growth in the freshwaters of North America: a review and critique of experimental enrichments.* Canadian Journal of Fisheries and Aquatic Sciences 47: 1468-1477.

Elwood JW, Newbold JD, Trimble AF, Stark RW. 1981. *The limiting role of phosphorus in a woodland stream ecosystem: effects of P enrichment on leaf decomposition and primary producers.* Ecology 62: 146-158.

Ervin M. *Effects of a Drawdown Treatment on Habitat and Muskrat Populations in Saskatchewan River Delta Wetlands.* Unpublished MSc thesis. Iowa State University.

Fairchild GW, Lowe RL, Richardson WB. 1985. *Algal periphyton growth on nutrient-diffusing substrates: an in situ bioassay.* Ecology 66: 465-472.

Ferber LR, Levine SN, Lini A, Livingston GP. 2004. *Do cyanobacteria dominate in eutrophic lakes because they fix atmospheric nitrogen?* Freshwater Biology 49: 690-708.

Francoeur, SN, Biggs DJF, Smith RA, Lowe RL. 1999. *Nutrient limitation of algal biomass accrual in streams: seasonal patterns and a comparison of methods.* Journal of the North American Benthological Society 18: 242-260.

Futyma RP, Miller NG. 1986. *Stratigraphy and genesis of the Lake Sixteen peatland, northern Michigan.* Canadian Journal of Botany 64: 3008-3019.

- Gächter R, Müller B. 2003. *Why the phosphorus retention of lakes does not necessarily depend on oxygen supply to their sediment surface*. Limnology and Oceanography 48: 929-933.
- Gerhardt DZ, Likens GE. 1975. *Enrichment experiments for determining nutrient limitation: four methods compared*. Limnology and Oceanography 20: 649-653.
- Gibeau GC, Miller MC. 1989. *A microassay for epilithon using nutrient-diffusing artificial substrates*. Freshwater Ecology 5: 172-176.
- Glooschenko WA, Alvis C. 1973. *Changes in species composition of phytoplankton due to enrichment by N, P, and Si of water from a North Florida lake*. Hydrobiologia 42: 285-294.
- Glooschenko V, Grondin P. 1988. *Wetlands of eastern temperate Canada*. In Wetlands of Canada. Ecological Land Classification Series No 24. Sustainable Development Branch, Environment Canada, Ottawa, Ontario, and Polyscience Publications, Montreal, Quebec.
- Gons HJ, Veeningen R, Van Keulen RV. 1986. *Effects of wind on a shallow lake ecosystem: resuspension of particles in the Loosdrecht Lakes*. Aquatic Ecology 20: 109-120
- Graham MD. 1997. *Omnivory and selective feeding by zooplankton along a lake production gradient: complementary <sup>15</sup>N isotope and gut pigment analysis*. MSc Thesis, University of Regina.
- Grosshans RE, Wrubleski DA, Goldsborough LG. 2004. *Changes in the emergent plant community of Netley-Libau Marsh between 1979-2001*. Delta Marsh Field Station, University of Manitoba, Occasional Publication No 4.

- Haertel L. 1976. *Nutrient limitation of algal standing crops in shallow prairie lakes*. Ecology 57: 664-678.
- Hardy FJ, Shortreed KS, Stockner JG. 1986. *Bacterioplankton, phytoplankton, and zooplankton communities in a British Columbia coastal lake before and after nutrient reduction*. Canadian Journal of Fisheries and Aquatic Sciences 43: 1504-1514.
- Harris SW, Marshall WH. 1963. *Ecology of water level manipulation on a northern marsh*. Ecology 44: 331-343.
- Healey FP, Hendzel LL. 1980. *Physiological indicators of nutrient deficiency in lake phytoplankton*. Canadian Journal of Fisheries and Aquatic Sciences 37: 442-453.
- Hecky RE, Kilham P. 1988. *Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidence on the effects of enrichment*. Limnology and Oceanography 33: 796-822.
- Hecky RE, Campbell P, Hendzil LL. 1993. *The stoichiometry of carbon, nitrogen, and phosphorus in particulate matter of lakes and oceans*. Limnology and Oceanography 34: 704-724.
- Hertam S. 2010. *The effects of common carp (Cyprinus carpio L.) on water quality, algae and submerged vegetation in Delta Marsh, Manitoba*. MSc thesis. University of Manitoba.
- Hill WR. 1996. *Effects of light*. In *Algal Ecology: Freshwater Benthic Ecosystems*. Stevenson RJ, Bothwell ML, Lowe RL, eds. Academic Press.

- Howarth RW, Marino R. 2006. *Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over three decades*. *Limnology and Oceanography* 51: 364-376.
- Huntsman AG. 1948. *Fertility and fertilization of streams*. *Journal of Fisheries Research Board of Canada* 7: 248-253.
- Jaques DR. 1990. *Vegetation habitat types of the Peace–Athabasca Delta: 1976–1989*. Prepared by Ecosat Geobotanical Surveys Inc. for Parks Canada, Wood Buffalo National Park: Fort Smith, Canada.
- Jensen JP, Kristensen P, Jeppeson E. 1991. *Relationship between N loading and in-lake N concentrations in shallow Danish lakes*. *Verhandlungen Internationale Vereinigung Theoretisch Angewandte Limnologie* 24: 201-204.
- Kadlec JA. 1962. *The effects of a drawdown on a waterfowl impoundment*. *Ecology* 43: 267-281.
- Kadlec JA, van der Valk AG, Murkin HR. 2000. *The MERP nutrient budgets*. In *Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program*. Murkin HR, van der Valk AG, Clark WR, eds. Iowa State University Press, 37-54.
- Kalff J. 2002. *Limnology*. Prentice Hall.
- Kilham SS. 1990. *Relationship of phytoplankton and nutrients to stoichiometric measures*. In *Large lakes: ecosystem structure and function*. Tilzer MM, Serruya C, eds. Springer, 403-414.
- Kolochuk JS. 2008. *Landscape and land use impacts on farm pond water quality in the Portage Plains of south-central Manitoba*. MSc Thesis.

University of Manitoba.

- Kratz TK, DeWitt CB. 1986. *Internal factors controlling peatland-lake development*. Ecology 67: 100-107.
- Krewer JA, Holm HW. 1982. *The phosphorus-chlorophyll a relationship in periphytic communities in a controlled ecosystem*. Hydrobiologia 94: 173-176.
- Krull JN. 1970. *Aquatic plant-macroinvertebrate associations and waterfowl*. Journal of Wildlife Management 34: 707-718.
- Kubiw H, Hickman M, Vitt DH. 1989. *The developmental history of peatlands at Muskiki and Marguerite lakes, Alberta*. Canadian Journal of Botany 67: 3534-3544.
- Laring H. 2003. *Water and wetland plants of the prairie provinces: a field guide for Alberta, Saskatchewan, Manitoba, and the northern United States*. Canadian Plains Research Center, University of Regina.
- Leavens L. 2008. *Saskatchewan River Delta hydrology*. Proceedings of the Saskatchewan River Delta Symposium. Saskatoon, Saskatchewan.
- Lee RE. 1989. *Phycology* 2<sup>nd</sup> ed. Cambridge University Press.
- Longcore JR, McAuley DG, Pendelton GW, Reid-Bennatti C, Mingo TM, Stromborg KL. 2006. *Macroinvertebrate abundance, water chemistry, and wetland characteristics affect use of wetlands by avian species in Maine*. Special Issue of Hydrobiologia 567: 143-167.
- Marcus MD. 1980. *Periphyton community response to chronic nutrient enrichment by a reservoir discharge*. Ecology 61: 387-399.



- Marker AFH, Crowther CA, Gunn RJM. 1980. *Methanol and acetone solvents for estimating chlorophyll  $\alpha$  and phæopigments by spectrophotometry*. Archiv für Hydrobiologie Beihefte 14: 52-69.
- McDougal RL. 2001. *Algal primary production in prairie wetlands: the effects of nutrients, irradiance, temperature and aquatic macrophytes*. PhD Thesis, University of Manitoba.
- McLeod JA, Baldwin S, Levin M. 1947. *A biological investigation of the Saskatchewan River Delta with special reference to muskrat production*. Unpublished report in Conservation and Environment Library, Winnipeg, Manitoba.
- Mehdi BB, Hovda J, Madramootoo CA. 2002. *Impacts of climate change on Canadian water resources*. Canadian Climate Impacts and Adaptation Research Network, Natural Resources Canada, McGill University.
- Meijer ML, Van Nes EH, Van Donk E, Moss B. 1994. *Long-term response to fish-stock reduction in small lakes – Interpretation of five-year results of four biomanipulation cases in the Netherlands and Denmark*. Hydrobiologia 276: 457-466.
- Millar JB. 1973. *Vegetation changes in shallow marshwetlands under improving moisture regime*. Canadian Journal of Botany 51: 144-1457.
- Miller NG, Futyma RP. 1987. *Paleohydrological implications of Holocene peatland development in northern Michigan*. Quaternary Research 27: 297-311.

- Mitsch WJ. 1979. *Interactions between a riparian swamp and a river in southern Illinois*. In: Johnson RR, McCormick JF. Strategies for the Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. Proceedings of the Symposium, Calaway Gardens, Georgia, December 11-13 1978. General Technical Report WO-12, US Forest Service, Washington. 63-72.
- Mitsch WJ, Gosselink JG. 2000. *Wetlands*, 3<sup>rd</sup> edition. John Wiley & Sons.
- Moore TR, Roulet NT, Watddington JM. 1998. *Uncertainty in predicting the effect of climate change on the carbon cycling of Canadian peatlands*. Climatic Change 40 : 229-245.
- Morozova GS, Smith ND. 2003. *Organic-rich deposition in the Saskatchewan River floodplain (Cumberland Marshes, Canada): effects of progradational avulsions*. Sedimentary Geology 157: 15-29.
- Morris DP, Lewis WM. 1988. *Phytoplankton nutrient limitation in Colorado mountain lakes*. Freshwater Biology 20: 315-327.
- Mudry N. no date. *Evolution of agricultural reclamation in the Saskatchewan River Delta*. Unpublished report in Conservation and Environment Library, Winnipeg, Manitoba
- Mugidde R, Hecky RE, Hendzel LL, Taylor WD. 2004. *Pelagic nitrogen fixation in Lake Victoria (East Africa)*. Journal of Great Lakes Research 29: 76-88.
- Murkin HR, Caldwell PJ. 2000. *Avian use of prairie wetlands*. In Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program. Murkin HR, van der Valk AG, Clark WR, eds. Iowa State University Press, 249- 286.

- Murkin EJ, Murkin HR. 1989. *Marsh ecology research program: long-term monitoring procedures manual*. Delta Waterfowl and Wetlands Research Station Technical Bulletin 2.
- Murkin HR, Ross LCM. 2000. *Invertebrates in prairie wetlands*. In *Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program*. Murkin HR, van der Valk AG, Clark WR, eds. Iowa State University Press, 201-234.
- Murkin HR, van der Valk AG, Clark WR. 2000. *Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program*. Iowa State University Press.
- National Wetlands Working Group. 1988. *Wetlands of Canada*. Ecological Land Classification Series No 24. Sustainable Development Branch, Environment Canada, Ottawa, Ontario, and Polyscience Publications, Montreal, Quebec.
- National Wetlands Working Group. 1997. *The Canadian wetland classification system, 2<sup>nd</sup> ed*. Warner BG and Rubec CDA, eds. Wetlands Research Centre, University of Waterloo.
- Neill C, Cornwell JC. 1992. *Stable carbon, nitrogen, and sulfur isotopes in a prairie marsh food web*. *Wetlands* 12: 217-224.
- Nicholson BJ. 1993. *The wetlands of Elk Island National Park: Vegetation, development and chemistry*. PhD Thesis, University of Alberta.
- Nicholson BJ. 1995. *The wetlands of Elk Island National Park: vegetation classification, water chemistry, and hydrotopographic relationships*. *Wetlands* 15: 119-133.

- Nicholson BJ and Witt DH. 1994. *Wetland development at Elk Island National Park, Alberta, Canada*. Journal of Paleolimnology 12: 19-34.
- Nicholson B, Bayley SE, Whitehouse EH. 2006. *Inferred history of a boreal pond from sediment and vegetation characteristics*. Canadian Journal of Soil Science 86: 335-347.
- Northern River Basins Study Board. 1996. *Northern river basins study report to the ministers*. Nautilus Publications, Edmonton, Alberta.
- Partners FOR the Saskatchewan River Basin. 2008. *From the mountains to the sea: the state of the Saskatchewan River basin*.
- Peterson BJ, Deegan L, Helfrich J, Hobbie JE, Hullar MAJ, Moller B, Ford TE, Hershey A, Hiltner A, Kipphut G, Lock MA, Fiebig DM, McKinley V, Miller MC, Vestal JR, Ventullo RM, Volk GS. 1993. *Biological responses of a tundra river to fertilization*. Ecology 74: 53-672.
- Peterson BJ, Hobbie JE, Corliss TL, Kriet K. 1983. *A continuous-flow periphyton bioassay: tests of nutrient limitation in a tundra stream*. Limnology and Oceanography 28: 583-591.
- Poe AC, Piehler MF, Thompson SP, Paerl HW. 2003. *Denitrification in a constructed wetland receiving agricultural runoff*. Wetlands 23: 817-826.
- Poston B, Ealey D, Taylor P, McKeating GB. 1990. *Priority migratory bird habitats of Canada's prairie provinces*. Environment Canada, Edmonton, Alberta.

- Prepas EE, Trimbee AM. 1988. *Evaluation of indicators of nitrogen limitation in deep prairie lakes with laboratory bioassay and limnocorrals*. *Hydrobiologia* 159: 269-276.
- Pringle CM, Bowers JA. 1984. *An in situ substratum fertilization technique: diatom colonization on nutrient-enriched, sand substrata*. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 1247-1251.
- Prowse TD, Conly FM. 2002. *A review of hydroecological results of the Northern River Basins Study, Canada. Part 2. Peace-Athabasca Delta*. *River Research and Applications* 18: 447-460.
- Prowse TD, Lalonde V. 1996. *Open-water and ice-jam flooding of a northern delta*. *Nordic Hydrology* 27: 85-100.
- Purcell SL. 1999. *The significance of waterfowl feces as a source of nutrients to algae in a prairie wetland*. MSc Thesis, University of Manitoba.
- Ramsay WW, Clark J, Moffat ME. 1955. *History of engineering development in the Summerberry fur development project*. Unpublished report in Conservation and Environment Library, Winnipeg, Manitoba.
- Reader R. 2008. *Historical perspective of water controls*. Proceedings of the Saskatchewan River Delta Symposium. Saskatoon, Saskatchewan.
- Reddy RK, Delaune RD. 2008. *Biogeochemistry of wetlands: science and applications*. Taylor and Francis Group.
- Reddy KR, Fisher MM, Ivanoff D. 1996. *Resuspension and diffusive flux of nitrogen and phosphorus in a hypereutrophic lake*. *Journal of Environmental Quality* 25: 363-371.

- Redfield AC. 1958. *The biological control of chemical factors in the environment*. American Scientist 46: 205-222.
- Rees TAV, Syrett PJ. 1979. *The uptake of urea by the diatom Phaeodactylum*. New Phytologist 82: 169-178.
- Robarts RD, Evans MS, Arts MT. 1992. *Light, nutrients, and water temperature as determinants of phytoplankton production in two saline, prairie lakes with high sulfate concentrations*. Canadian Journal of Fisheries and Aquatic Sciences 49: 2281-2290.
- Robinson GGC, Gurney SE, Goldsborough LG. 2000. *Algae in prairie wetlands*. In Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program. Murkin HR, van der Valk AG, Clark WR, eds. Iowa State University Press, 163-199.
- Rosenberg DM, Chambers PA, Culp JM, Franzin WG, Nelson PA, Salki AG, Stainton MP, Bodaly RA, Newbury RW. 2005. *Nelson and Churchill River Basins*. Chapter 19 in *Rivers of North America*. Benke AC and Cushing CE (eds). Elsevier Academic Press.
- Saskatchewan Watershed Authority. 2007. *Background report, South Saskatchewan River watershed*. Regina, Saskatchewan.
- Saunders DL, Kalff J. 2001. *Nitrogen retention in wetland, lakes and rivers*. Hydrobiologia 443: 205-212.
- Scheffer M. 1998. *Ecology of Shallow Lakes*. Chapman & Hall, London.
- Schindler D. 1977. *Evolution of phosphorus limitation in lakes*. Science 195:

260-262.

Scott JT, Doyle RD, Filstrup CT. 2005. *Periphyton nutrient limitation and nitrogen fixation potential along a wetland nutrient depletion gradient*. *Wetlands* 25: 439-448.

Shay JM, Shay CT. 1986. *Prairie marshes in western Canada, with specific reference to the ecology of five emergent macrophytes*. *Canadian Journal of Botany* 64: 443-454.

Simon NS. 1988. *Nitrogen cycling between sediment and the shallow-water column in the transition zone of the Potomac River and estuary: nitrate and ammonium fluxes*. *Estuarine Coastal and Shelf Science* 26: 483-497.

Sladeckova A. 1979. *Periphyton assays in situ*. In *Algal assays and monitoring eutrophication*. Marvan P, Pribil S, Lhotsky O, eds. Schweizerbat'sche Verlagsbuchhandlung, Stuttgart, Germany, 205-209.

Slattery S. 2008. *Status of waterfowl in SRD*. Proceedings of the Saskatchewan River Delta Symposium. Saskatoon, Saskatchewan.

Smith AG. 1971. *Ecological factors affecting waterfowl production in the Alberta parklands*. US Bureau of Sport Fisheries and Wildlife Resource Publication 98.

Smith CE. 1986. *Drawdown monitoring in the Summerberry Marsh Complex: 1985*. Ducks Unlimited Canada report.

Smith CE. 1987. *Drawdown monitoring in the Summerberry Marsh Complex: 1986*. Ducks Unlimited Canada report.

- Smitch CE, Jones CK. 1982. *Summerberry Marsh Complex annual report no 1: 1981 activities and future management recommendations.*
- Smith N. 2008. *What is the Saskatchewan River Delta?* Proceedings of the Saskatchewan River Delta Symposium. Saskatoon, Saskatchewan.
- Snow NB, Brunskill GJ. 1975. *Crude oil and nutrient enrichment studies in a MacKenzie Delta lake.* Environment Canada, Fisheries and Marine Service Technical Report No 553.
- Søndergaard M, Jenseon JP, Jeppesen E. 2003. *Role of sediment and internal loading of phosphorus in shallow lakes.* Hydrobiologia 506: 135-145.
- Squires MM, Lesack LFW. 2002. *Water transparency and nutrients as controls on phytoplankton along a flood-frequency gradient among lakes of the Mackenzie Delta, western Canadian Arctic.* Canadian Journal of Fisheries and Aquatic Sciences 59: 1339-1349.
- Stainton MP, Capel MJ, Armstrong FA. 1977. *The chemical analysis of freshwater.* 2<sup>nd</sup> Edition. Fisheries and Environment Canada. Miscellaneous Special Publication No 25. Ottawa, Ontario.
- Steinman AD. 1996. *Effects of grazers on freshwater benthic algae.* In Algal Ecology: Freshwater Benthic Ecosystems. Stevenson RJ, Bothwell ML, Lowe RL, eds. Academic Press.
- Sterner RW. 2008. *On the phosphorus limitation paradigm for lakes.* International Review of Hydrobiology 93: 433-445.
- Stockner JG, Shortreed KRS. 1978. *Enhancement of autotrophic production by nutrient addition in a coastal rainforest stream on Vancouver Island.* Journal of



Fisheries Research Board of Canada 35: 28-34.

Stout JH. 1971. *Ecological factors affecting waterfowl production in the Saskatchewan parklands*. US Bureau of Sport Fisheries and Wildlife Resource Publication 99.

Talbot JT, Richard JH, Roulet NT, Booth RK. 2010. *Assessing long-term hydrological and ecological responses to drainage in a raised bog using paleoecology and a hydrosequence*. *Journal of Vegetation Science* 21: 143-156.

Tallis JH. 1983. *Changes in wetland communities*. In *Ecosystems of the World: Mires: Swamp, Bog, Fen and Moor, Volume 4A*. Gore AJP, ed. Elsevier Scientific Publishing Co, 311-349.

Tarnocai C. 1980. *Canadian wetland registry*. Proceedings, workshop on Canadian Wetlands. CDA Rubec and FC Pollett, eds. Lands Directorate, Environment Canada. Ecological Land Classification Series No 12. Ottawa, Ontario.

Timoney K. 2002. *A dying delta? A case study of a wetland paradigm*. *Wetlands* 22: 282-300.

Tyler JE. 1968. *The Secchi disc*. *Limnology and Oceanography* 13: 1-6.

Uchtmann RH. 1983. *Summary of resource use in the Saskatchewan River Delta system*. Wildlife and Ecosystem Protection Branch. Unpublished report in Conservation and Environment Library, Winnipeg, Manitoba.

Uchtmann RH. 2008. Proceedings of the Saskatchewan River Delta Symposium. Saskatoon, Saskatchewan.

- Ulanowicz RE. 1995. *Utricularia's secret: the advantage of positive feedback in oligotrophic environments*. Ecological Modelling 79: 49-57.
- US Army Coastal Engineering Research Center. 1977. *Shore protection manual Volume I*. US Army Coastal Engineering Research Center, Fort Belvoir, Virginia.
- van der Valk AG. 2000. *Vegetation dynamics and models*. In Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program. Murkin HR, van der Valk AG, Clark WR, eds. Iowa State University Press, 125-161.
- van der Valk AG, Davis CB. 1978a. *Role of seed banks in the vegetation dynamics of prairie glacial marshes*. Ecology 59:322-335.
- van der Valk AG, Davis CB. 1978b. *Primary production in prairie glacial marshes*. In Freshwater Wetlands: Ecological Processes and Management Potential. Good RE, Whigham DF and Simpson RL, eds. Page 21-37. Academic Press, New York.
- Van Donk E, Gulati RD, Iedema A, Meulemans JT. 1993. *Macrophyte-related shifts in the nitrogen and phosphorus contents of different trophic levels in a biomanipulated shallow lake*. Hydrobiologia 251: 19-26.
- Vitousek PM, Aber JD, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH, Tilman DG. 1997. *Technical report: human alteration of the global nitrogen cycle: sources and consequences*. Ecological Applications 7: 737-750.
- Waiser MJ, Roberts RD. 1995. *Microbial nutrient limitation in prairie saline lakes with high sulfate concentrations*. Limnology and Oceanography 40: 566-574.

- Walker JM. 1959. *Vegetation studies in the Delta Marsh, Delta, Manitoba*. MSc Thesis, University of Manitoba.
- Walker JM. 1965. *Vegetation changes with falling water levels in the Delta Marsh, Manitoba*. PhD Dissertation, University of Manitoba.
- Welch HE, Legault JA, Kling HJ. 1989. *Phytoplankton, nutrients, and primary production in fertilized and natural lakes at Saqvaquac, NWT*. Canadian Journal of Fisheries and Aquatic Sciences 46: 90-107.
- Weller MW. 1978. *Management of freshwater marshes of wildlife*. In Freshwater Wetlands: Ecological Processes and Management Potential. Good RE, Whigham DF and Simpson RL, eds. Pages 267-284. Academic Press, New York.
- Weller MW, Fredrickson LH. 1974. *Avian ecology of a managed glacial marsh*. Living Bird 12: 269-291.
- Weller MW, Spatcher CE. 1965. *Role of habitat in the distribution and abundance of marsh birds*. Department of Zoology and Entomology Special Report No 43. Agricultural and Home Economics Experiment Station, Iowa State University.
- Winterbourn MJ. 1990. *Interactions among nutrients algae and invertebrates in a New Zealand mountain stream*. Freshwater Biology 23: 463-474.
- Wolfe BB, Karst-Riddoch TL, Hall RI, Edwards TWD, English MC, Palmini R, McGowan S, Leavitt PR, Vardy SR. 2007. *Classification of hydrological regimes of northern floodplain basins (Peace-Athabasca Delta, Canada) from analysis of stable isotopes ( $\delta^{18}\text{O}$ ,  $\delta^2\text{H}$ ) and water chemistry*. Hydrological

Processes 21: 151-168.

Wuhrmann K, Eichenberger E. 1975. *Experiments on the effects of inorganic enrichment of rivers on periphyton primary production.* Verhandlungen Internationale Vereinigung für Theoretische und Angewandte Limnologie 19: 2028-2034.

Wrubleski D. 2008. *Inland deltas, the Saskatchewan River Delta, and does size really matter?* Proceedings of the Saskatchewan River Delta Symposium. Saskatoon, Saskatchewan.

Yulianti JS, Burn DH. 1998. *Investigating links between climate warming and low streamflow in the Prairies region of Canada.* Canadian Water Resources Journal 23: 45-60.

Zoltai SC, Johnson JD. 1987. *Relationships between nutrients and vegetation in peatlands of the prairie provinces.* Proceedings of the Symposium'87 Wetlands / Peatlands, Edmonton, 535-543.

## Appendix I: NDS Time Lapse Experiment

(The following explanation was provided by: Tara Bortoluzzi, PhD Candidate, University of Manitoba)

Nutrient diffusion rates from silica disk micro-NDS were determined via laboratory experiments as described by Fairchild *et al.* (1985). NDS vials were immersed in 500 ml of distilled water in the laboratory. Samples were taken at the start of the experiment, once every 24 hours for 4 days, and then every second day for 45 days. The samples were analyzed for TRP ( $\text{PO}_4\text{-P}$ ) and nitrate+nitrite ( $\text{NO}_3\text{+NO}_2\text{-N}$ ) using standard methods for water samples (Stainton *et al.* 1977, APHA 1992). The experiments indicated that all NDS treatments diffused nutrients throughout a 45-day period, and the diffusion rate decreased as a negative exponential as found similarly by Pringle and Bowers (1984), Fairchild *et al.* (1985), and Gibeau and Miller (1989). The mean diffusion rates from the NDS were  $33 \mu\text{mol}/\text{cm}^2/\text{day}$  for  $\text{NO}_3$ , and  $2.2 \mu\text{mol}/\text{cm}^2/\text{day}$  for  $\text{PO}_4$ . No nitrogen or phosphorus was detected diffusing from the control substrata.

## Appendix II: Assessment of Dissolved Organic Carbon (DOC) using scanning UV spectroscopy

(The following explanation was provided by: Pascal Badiou, Ducks Unlimited Canada, Institute for Wetland and Waterfowl; Gordon Goldsborough, University of Manitoba; Diane Malley, PDK Projects Inc)

Dissolved organic carbon (DOC) concentrations were measured on 192 water samples taken from lakes and wetlands across the prairie provinces of Canada and several northern US states between 2001 and 2005. These DOC measurements were conducted at the Freshwater Institute in Winnipeg using an O.I. Corporation model 700 TOC Analyzer. Filtered water samples (Whatman GF/C filters) were acidified with phosphoric acid and sparged with carbon free nitrogen to remove inorganic carbon. Samples were then heated (100°C) and treated with acid persulphate to oxidize DOC to CO<sub>2</sub>. Resulting CO<sub>2</sub> from the digestion of DOC in the samples was then stripped with carbon free nitrogen and trapped on a Molecular Sieve column at ambient temperature. At the completion of sample digestion, the accumulated CO<sub>2</sub> was released from the Molecular Sieve column by heating to 200°C and then carried to an infra-red (IR) detector for quantification.

Filtered aliquots of the same water samples (Whatman GF/C filters) were scanned in a 1-cm quartz cell with an Ultrospec 4000 scanning UV/Vis spectrophotometer. UV/Vis spectra were collected between 190 and 400 nm. All scans were corrected using distilled water as a blank. The resulting spectra were sorted based on the measured DOC concentrations in the samples from lowest to highest. At this point the spectra were divided into two new sets, termed 'Cal' and 'Pred'. The Cal set contained every spectra from the original dataset starting with the lowest DOC concentration while the Pred set contained every second spectra beginning from the second lowest DOC concentration. Predicted

(Cal/Pred) DOC was significantly correlated to measured (Reference) DOC with a conversion of 0.99 (Figure II-1).

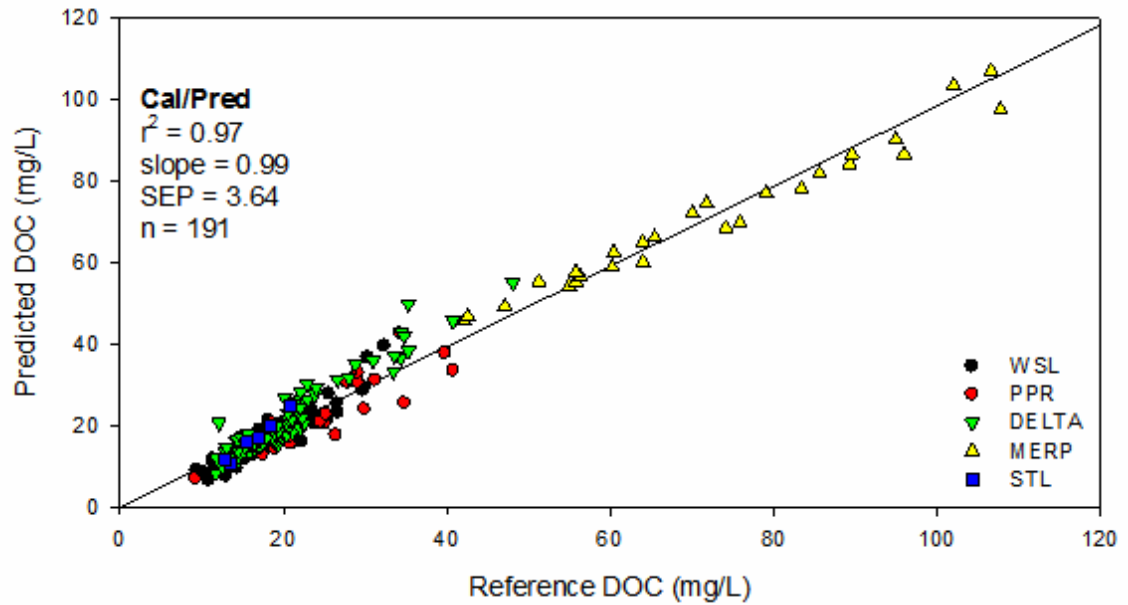


Figure II-1: Relationship between measured and UVR-predicted dissolved organic carbon concentrations (mg/L). WSL – Whiteshell/Boreal Lakes, Manitoba; PPR – Prairie Pothole Region wetlands, North Dakota, Manitoba, Saskatchewan, and Alberta; Delta – Delta Marsh, Manitoba; MERP – Marsh Ecology Research Program experimental wetland cells, Delta Marsh, Manitoba; STL – Stephens Lake, Manitoba.