Spatial and temporal patterns in the hydrology, water chemistry and algal nutrient status of Delta Marsh, as influenced by the hydrology of adjoining Lake Manitoba.

By

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Abstract

Between 2002 and 2005, I examined spatial and temporal patterns in the hydrology, water chemistry, and algal nutrient-limitation status (N and/or P) in Delta Marsh, a 18,500-ha coastal lacustrine freshwater marsh on the south shore of Lake Manitoba, to determine the influence of surface water exchange with Lake Manitoba on these properties.

Daily and annual marsh water level changes were found to be highly correlated with those of the lake, during some of the highest and lowest long-term water levels in recorded history. The average magnitude of water level changes in the marsh ranged from to a few centimeters to half a meter, which is significant in shallow coastal wetlands systems like Delta Marsh where the average depths are ≤ 1 m.

In general, marsh sites located closest to the lake were influenced to the greatest degree by the flushing and dilution effect of the lake. Spatially, in connected sections of the marsh concentrations of dissolved inorganic and total N (DIN-N and TN), total reactive and total phosphorus (TRP-P and TP), dissolved organic carbon (DOC), chloride (Cl^-), sulfate (SO_4^-), alkalinity and conductivity decreased with decreasing distance to Lake Manitoba.

Regardless of east and west location and the distance of connected marsh sites from Lake Manitoba, annual variation in water level was the most significant predictor of differences in several water chemistry characteristics between sample sites including

DIN-N, TN, TRP-P, TP, alkalinity, DOC, Cl^-, SO_4^-, and conductivity. Annually, concentrations of DIN-N, TN, alkalinity, DOC, Cl^-, SO_4^- and conductivity were negatively correlated with increasing water depth, and the spatial variation in the concentration of these water chemistry parameters also decreased with increasing water level.

Results of nutrient diffusing substrata bioassay experiments indicated that periphyton biomass in the marsh was predominately limited by N. The predominance of N limitation in Delta Marsh was found to be significantly negatively correlated with water column N concentrations, but not correlated with P concentrations. Collectively, this study illustrates the important role of lake connection and hydrological influence on the structure and function of adjoining coastal freshwater wetlands.

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Chapter 1: Introduction

Organization of thesis

This thesis is organized into eight chapters. Chapter 1 introduces the study, and study location, as well as the major objectives and hypothesis. Chapter 2 includes a review of the current and relevant literature summarizing wetland definitions, function and values, and distribution; as well as more specifically summarizing the status of knowledge surrounding the hydrology, chemistry, and algal nutrient status of coastal freshwater wetlands. Chapter 3 is an overview of the global methods used for the study, including study sites and design, field and laboratory methods, and data analysis. Chapters 4 through 6 discuss the results of this study; Chapter 4 focuses on the hydrological studies, Chapter 5 water chemistry results, and Chapter 6 the algal nutrient limitation experiments. Chapter 7 summarizes the overall major finding of the studies, revisites the hypotheses, and as well discusses resultant management implications for Delta Marsh, as well as other coastal freshwater wetlands across North America. Chapter 8 lists references for the literature cited in the thesis.

Preamble

Coastal freshwater wetlands are some of the most productive and biologically diverse ecosystems world-wide, with primary production ranges from approximately 30 to 80 metric tonnes per hectare per year (mT/ha/yr) for emergent macrophytes, 2 to 20 mT/ha/yr for submergent macrophytes, and 5 to 60 mT/ha/yr for algae (Mitsch and Gosselink 2000a). Often occupying only a small portion of the landscape, their importance is disproportionate to their size. They are well known for their ability to reduce flood and storm water flows, reduce shoreline and soil erosion, recharge groundwater, improve water quality by removing nutrients and contaminants, as well as providing critical habitat for a diverse variety of mammals, birds, reptiles, amphibians, fish, crustaceans, insects, and flora (Kadlec and Knight 1996; Mitsch and Bouchard 1998; Brazner et al. 2000; Mitsch and Gosselink 2000a & b; Tuner et al. 2000; Amezaga et al. 2002). In the prairie region of North America, wetlands are abundant, ranging in size from small potholes to large coastal marshes (National Wetlands Working Group 1988; Mitsch and Gosselink 2000b).

Despite their significance, coastal wetlands continue to be degraded and destroyed world-wide, as land is converted for agriculture, mining and urban development, hydrological alterations including dam construction and stream channelization, as well as increased sedimentation and nutrient enrichment, and the introduction of exotic species (Millar 1989; Vitousek et al. 1997; Carpenter et al. 1998; Zedler 2003). In the prairies of North America, this is especially true, as in addition to the natural fertile soils, coastal wetlands are subject to enrichment from industry, sewage, and agriculture sources, as well as land use practices that increase soil erosion and drainage (Barica 1987; Crumpton and Goldsborough 1998; Hall et al. 1999; Dixit et al. 2000; Carr et al. 2005). Many prairie

aquatic ecosystems are characterized by highly eutrophic conditions, with high total phosphorus (TP) concentrations, low nitrogen to phosphorus (N:P) molar ratios, high algal chlorophyll *a* concentrations, and the predominance of N-fixing cyanobacterial blooms (Haertel 1976; Barica 1975, 1987; Barica et al. 1980). Concurrent with wetland eutrophication and loss, the resultant ecological impacts on the biology, chemistry, and hydrology of these dynamic systems, and their ability to act as suitable habitat for many species has had a long history in aquatic ecology (Havens et al. 1999). Therefore, understanding which nutrient(s) are limiting is a key aspect of eutrophication research in aquatic ecosystems, with much research focusing on the nutrient limitation status of primary producers such as algae.

To date, the majority of studies examining algal nutrient limitation on the prairies have occurred in lakes (Haertel 1976; Allan and Kenney 1978; Barica et al. 1980; Healy and Henzel 1980; Campbell and Prepas 1986; Prepas and Trimbee 1988; Barica 1990; Waiser and Robarts 1995; Arts et al. 1997; Graham 1997). Of the few studies conducted in prairie wetlands, the majority have focused on phytoplankton (Kadlec et al. 1986; Murkin et al. 1991; Detenbeck et al. 2002), with little detailed study of benthic algae (Hooper-Reid and Robinson 1978; Murkin et al. 1991; Goldsborough and Robinson 1996; Kiers-North 2000).

Hydrology is a key factor controlling wetland structure and function (Mitsch and Gosselink 2000a & b; Grosshans 2001; Wilcox 2012). It affects many biotic and abiotic factors, including soil anaerobiosis, nutrient availability, and conductivity, which then determine biotic development, organic matter accumulation, and nutrient cycling. Due to their small volumes and shallow water depths, wetlands are dynamic environments in which small changes in water levels can result in significant biological and chemical

changes (van der Valk 2006). Water levels change both on short-term (i.e. daily and monthly) and on long-term scales (years and decades), and in the case of coastal lacustrine wetlands, water levels can be influenced by their adjoining lakes over years to decades (Maynard and Wilcox 1997, Trebitz 2002, Trebitz 2006, Morrice et al. 2011, Wilcox 2012). In these systems, long-term water levels are largely influenced by changes in the quantity of water received directly from the surrounding watershed and/or via adjoining water bodies on an annual basis. Short-term changes are influenced by storm surges and tidal and wind-induced seiche events on the adjoining lake (periodic oscillations of water level with movement of water to one side of the basin, i.e. in the direction of the wind, or pull from the moon), which can result in water exchange between the lakes and adjoining coastal wetlands, with mixing of lake and wetland water during seiche inflows and subsequent flushing during outflow. The full effect of these short-term water exchanges and changes in water level on adjoining wetlands is still uncertain, but the flow exchange appears to act similarly to tides in marine coastal estuaries, allowing exchange of water, nutrients and other materials between wetlands and the lakes, as well as impacting plant and animal species exchange, diversity, and productivity (Maynard and Wilcox 1997, Sierszen et al. 2006, Trebitz 2006, Gathman and Burton 2011, Morrice et al. 2011, Wilcox 2012). Aside from the Laurentian Great Lakes coastal wetlands, few detailed studies have examined spatial and temporal patterns in the hydrology, chemistry and algal nutrient status of coastal lacustrine wetlands, or the influence of adjoining lakes on these properties (Mitsch and Gooselink 2000a & b, Morrice et al. 2004, Sierszen et al. 2006, Trebitz 2006, Morrice et al. 2011). To obtain more knowledge on these coastal wetland properties, and the possible effect of adjoining lakes on these properities, this study was undertaken in Delta Marsh, a 18,500 ha coastal lacustrine freshwater marsh on the south shore of Lake

Manitoba, located in south-central Manitoba, Canada. Spatial and temporal patterns in the hydrology, water chemistry, and algal nutrient-limitation (N and/or P) in the marsh were examined over a four-year period during the open water season, and the influence of surface water exchange with Lake Manitoba on these properties.

This project provides a much-needed quantitative foundation of data which will lead to better understanding of the hydrological influence of Lake Manitoba on temporal and spatial patterns in hydrology, water chemistry, and algal limiting nutrients (specifically N and P) in Delta Marsh, and possibly other coastal wetlands. Understanding their sensitivity to changes in the hydrology and chemistry is a critical step in coastal wetland protection and restoration. This study focused on the influence of short-term (wind- induced) seiches and long-term (annual) hydrological variability, as well as spatial variability across the marsh.

Study Site Description

Delta Marsh

Delta Marsh (50°11'N, 98°19'W) is the largest of several marshes located on the periphery of Lake Manitoba (Figure 1.1; Watchorn et al. 2012). It is well known and highly recognized as habitat for waterfowl and other wildlife (Jones 1978, Wrubleski 1998). In 1982, it was designated as an Internationally Significant Wetland under the 1971 United Nations Convention on Wetlands of International Importance (The Ramsar Convention), and it became a Manitoba Heritage Marsh in 1988. The marsh is comprised of a network of variously-sized channels, shallow bays, and isolated small ponds. Delta Marsh is a former coastal riverine marsh, formed when the Assiniboine River flowed north into Lake Manitoba until 2000 years BCE (Teller and Last 1981). The marsh is classified as coastal lacustrine marsh attached by four channels crossing through a forested barrier beach ridge connecting to the south end of Lake Manitoba. These four channels are Cram Creek, Deep Creek, Delta Channel, and Clandeboye Channel (Figure 1.2). However, it should be noted that the number of channels that connect the marsh to the lake has varied in the past. For example, Eaglenest Creek once joined Center Marsh to Lake Manitoba until approximately the mid-1930s when it was closed off from the lake by siltation (Miller and Moore 1967).

These channels and creeks result in the hydrology of the marsh being directly and continually linked to that of the lake (De Geus 1987, Wrubleski 1998, Batt 2000). For instance, during the open-water season, storm surges and wind and wave setups (seiches) on the lake result in water exchange between the lake and marsh (Hochbaum 1944, Löve and Löve 1954, Walker 1959 and 1965, de Geus 1987, Wrubleski 1998, Batt 2000). De Geus (1987) was the first to establish a simple statistically significant relationship between

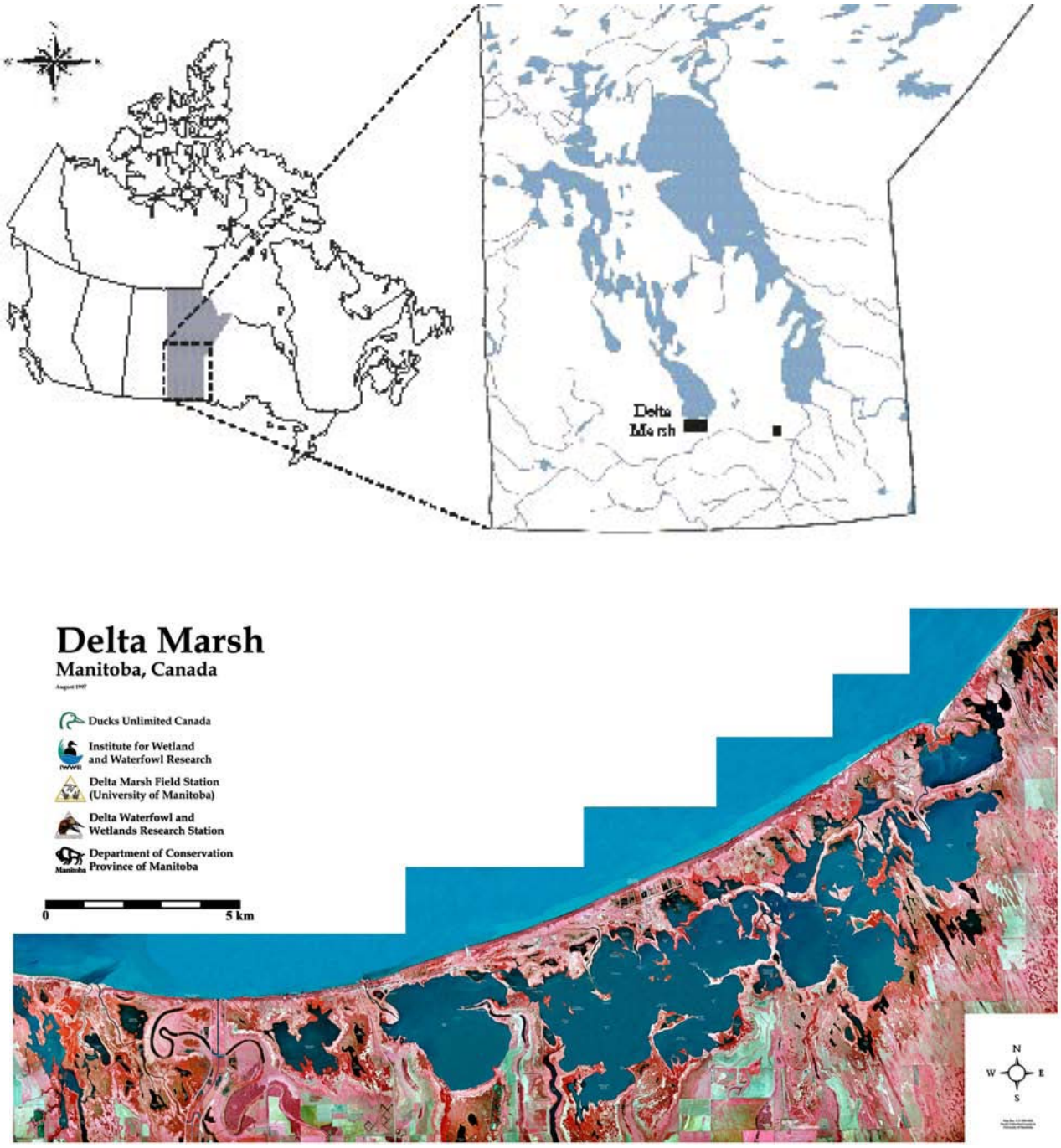

Figure 1.1 Delta Marsh located on the south shore of Lake Manitoba, in south-central Manitoba, Canada. Map courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba, 1997.

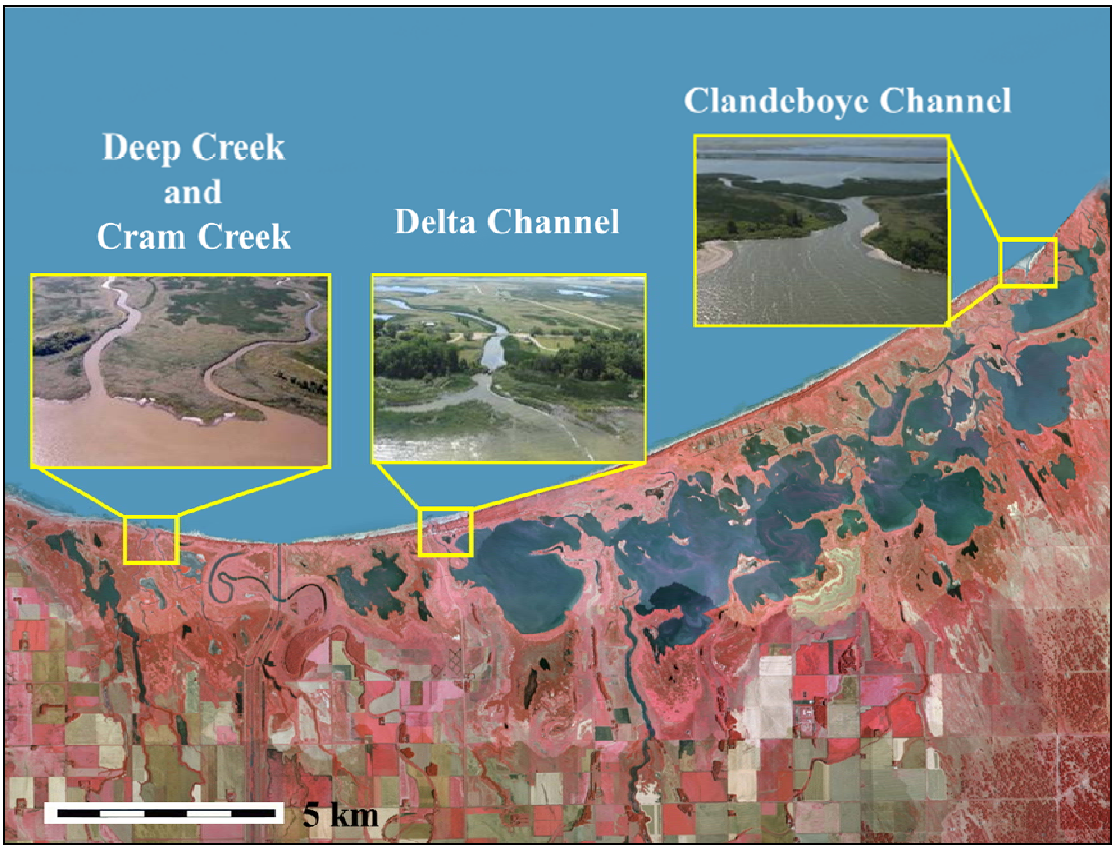

Figure 1.2 Map of the four channels connecting Delta Marsh to Lake Manitoba, located on the south shore of Lake Manitoba, in south-central Manitoba, Canada. Base photomosaic and aerial photographs courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

Lake Manitoba and Delta Marsh water levels, and concluded the lake to be the main hydrological influence on marsh water levels. Without water inflows from the lake, water levels in the marsh would decline approximately 18 cm every year (Jones 1978, De Geus 1987).

Historically until 1961, Lake Manitoba fluctuated within a range of over 1.7 m. Following a period of high lake water levels, the Fairford Dam (Fairford River Water Control Structure) was constructed in 1961 at the north end of the lake. The dam was constructed to try to control both high and low water levels on the lake, as low water levels can reduce boating navigation on the lake and wildlife availability for hunting, and high water levels flood surrounding lowlands with agricultural and cottage development (LMRRAC 2003a). Since then Lake Manitoba water level has been stabilized and regulated at a level of 247.6 m above sea level (ASL), with a range of ± 0.3 m, with the exception of 2003 and 2011 when extreme precipitation conditions resulted in lake water levels that were beyond those that could be effective managed via the Fairford Dam (Figure 1.3). In 2003, dry spring and summer conditions resulted in the lowest water level on the lake since regulation (247.0 m ASL), and in 2011 flooding in the Assiniboine River watershed, resulted in the highest water levels on the lake in recorded history (248.9 m ASL). With the exception of 2003 and 2011, regulation of the lake has reduced both the magnitude and persistence of water level fluctuations in lake, with fluctuations rarely exceeding 0.5 m, and it is assumed the timing and quantity of lake flushing flows in the marsh have been greatly altered (de Geus 1987, Kenkel 1995, LMRRAC 2003a). The reduced water level fluctuations have caused the marsh to enter a ‘lake-marsh phase’ with reduced productivity and species diversity, and has raised concerns about its long-term

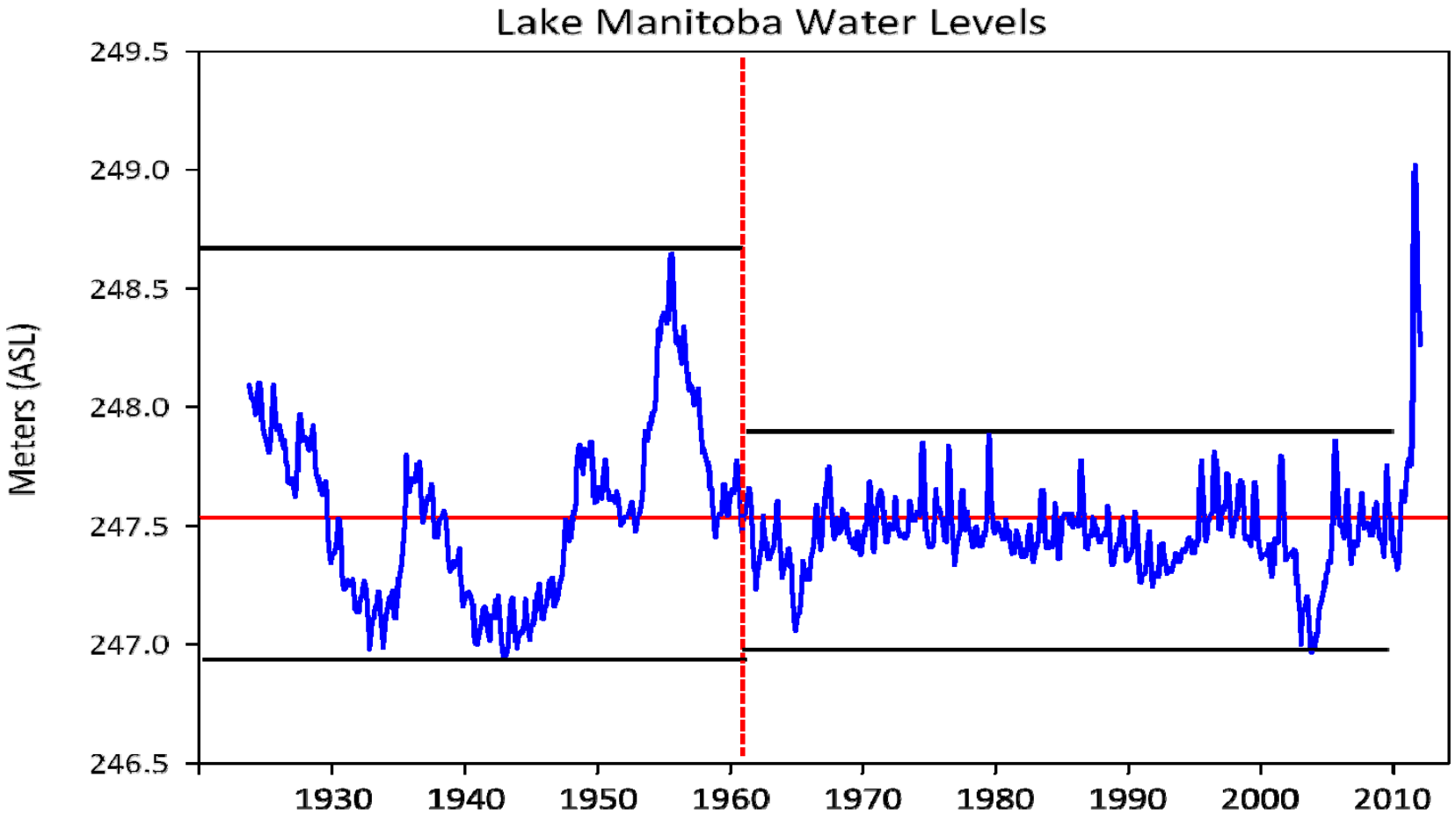

Figure 1.3 Historical water levels (meters above sea level) on Lake Manitoba recorded at Steep Rock (Water survey of Canada Station 05LK002), Manitoba, 1923 to 2011. The red dashed line represents the start of lake regulation via the Fairford Dam in 1961. Black lines are the upper and lower water level ranges pre and post regulation, with the exception of the 2011 flood.

biological health (van der Valk 1981, Kenkel 1995, Wrubleski 1998). Prolonged high water levels in the marsh have been noted to increase the impact of wave action from northerly winds on the southern shorelines of the marsh (Grosshans 2001). Previously gradual sloping shorelines have now become steeper and heavily eroded. Vegetated islands in the large bays of the marsh, including Peacock Pass, have disappeared (Grosshans 2001).The hybrid cattail (*Typha x glauca*), a hybrid of *T. latifolia* and *T. angustifolia*, has been increasingly expanding in the marsh (de Geus 1987), and Goldsborough (1987) has documented the infilling of Crescent Pond, a small pond in the west portion of marsh, by the hybrid cattail.

In 2003, the Lake Manitoba Regulation Review Advisory Committee recommended that Lake Manitoba levels be permitted to fluctuate more naturally within the range of 247.0 and 247.6 m ASL, with the expectation that further management would only occur when levels were at these extremes, so the effective range of lake level would fall between 246.9 and 247.8 m ASL level (LMRRAC 2003). This management regime would influence water levels on adjoining Delta Marsh, presumably allowing them to fluctuate more naturally, and within a larger magnitude and range in water level

Since the 1940s, there have been various water control structures constructed in the marsh, mainly to stabilize and control water levels, facilitate access, and increase wildlife production and harvest. This included dams constructed on the mouths of Deep Creek, Cram Creek and Clandeboye Channel in the 1940s (which no longer remain today), and two metal culverts under Delta Road (near the Village of Delta) which allow the flow of water from the lake into the marsh through Delta Channel (Bossenmaier 1968). The metal culverts were upgraded to a concrete structure with three flow-through channels in 1982.

The Assiniboine River Diversion (also known as the Portage Diversion) was constructed in 1969 to redirect floodwaters away from the city of Portage la Prairie, and the city of Winnipeg downstream, into Lake Manitoba. The channel, 25 km in length, diverts water from the Assiniboine River, just west of Portage la Prairie, into the south end of Lake Manitoba. Since its construction the diversion has been used more than 23 times and when used it introduces high suspended solid loads, nutrients, and dissolved ions carried by the river into Lake Manitoba (LMRRAC 2003a & b, Page 2011). The diversion channel itself is not connected directly to Delta Marsh; however, it runs through the middle of the marsh, and a portion of the channel dyke was constructed at a lower level to act as a failsafe in case high water flows exceeded the capacity of the channel. When the failsafe is used, the water overflow from the diversion is released directly into the west section of Delta Marsh (LMRRAC 2003a & b, Page 2011).

The water chemistry of Delta Marsh, like most marshes, is variable. The extent of variability is poorly documented for little water quality data exist for the marsh prior to the 1960s, when substantial changes were likely to have occurred because of the stabilization of water levels in adjoining Lake Manitoba since 1961, and the invasion of common carp to Lake Manitoba circa 1947 (Goldsborough and Wrubleski unpublished, Badiou 2005, Parks 2006, Hnatiuk 2006, Hertam 2010, Page 2011). The first limited water chemistry data available on the marsh dates back to 1936, when Hinks (1936) examined water clarity and pH, as well as the vegetation and sediments of the marsh. He noted that the marsh had excellent cover of aquatic vegetation and a great abundance of aquatic animals. The overall pH range of water was 7.6 to 8.4. More recent studies have characterized the marsh during the open water seasons as moderately brackish with conductivity values ranging from 943 to 5080 μS/cm, total alkalinity averages around 337.6 mg/L as $CaCO_3$, pH ranges from 8.0

to 9.2, total P from <0.05 to 0.39 mg/L, nitrate+nitrite-N from <0.05 to 0.34 mg/L, ammonia-N from <0.05 to 2.94 mg/L, and total phytoplankton chlorophyll a from 2.2 to 24.6 µg/L (Goldsborough 1994, Batt 2000).

There is little documented evidence of the nutrient limitation status of algae in Delta Marsh. Kiers-North (2000) used nutrient-diffusing substrata (providing inorganic N and/or P) and N debt experiments to show that periphyton and phytoplankton in the west section of Blind Channel in Delta Marsh was limited by N supply. During the ten-year Marsh Ecology Research Program (MERP), Kadlec (1986) found that TN:TP ratios indicated the likelihood of algal N-limitation in isolated ponds created in the east section of the marsh. Hooper-Reid and Robinson (1978) used several physiological indicators of algal nutrient limitation, including alkaline phosphatase activity, nitrogenase activity, ratio of protein to carbohydrate and lipid, and silica uptake rates in Crescent Pond to show that periphyton growth was most likely limited by N during the early summer. They also noted that high alkaline phosphate activity in the early summer indicated potential for P limitation and/or N+P limitation, and by late summer low Si concentrations were accompanied by reduced diatom growth.

Delta Marsh soils are a complex of gleysols and regosols (Walker 1965, Batt 2000) composed of peat deposits which overlay the lacustrine (sedimentary) clays and silts deposited in the area by Glacial Lake Agassiz in the late Quaternary (Last 1980).

Emergent macrophyte species in the marsh include hybrid cattail (*Typha x glauca*), hardstem bulrush (*Schoenoplactus acutus*), softstem bulrush (*Schoenoplectus tabernaemontani),* reed grass (*Phragmites australis*), white top (*Scholochloa festucacea*), and sedges (*Carex atherodes)*(Löve and Löve 1954, de Geus 1987, Shay 1999, Grosshans 2001). Submersed and floating aquatic vegetation is dominated by sago pondweed

(*Stuckenía pectinatus L*), coontail (*Ceratophyllum demersum)*, bladderwork (*Utricularia macrorhiza*), water milfoil (*Myriophyllum spp*.), broadleaf arrowhead (*Sagittaria latifolia*), lesser duckweed (*Lemna minor*), and ivy leaved duckweed (*Lemna trisulca*) (Löve and Löve 1954, de Geus 1987, Shay 1999, Grosshans 2001). Fauna in the marsh includes many bird species, including many migratory waterfowl and song birds, as well as species of muskrats, white-tailed deer, mink, skunk, raccoons, voles, weasels, woodchuck, squirrels, and bats (Batt 2000). Fish species found in the marsh include fathead minnow (*Pimephales promelas*), common white sucker (*Castostomus commersoni*), northern pike (*Esox lucius*), yellow perch (*Perca flavescens*), common carp (*Cyprinus carpio*), spottail shiner (*Notropis hudsonius*), emerald shiner (*Notropis atherinoides*), brook stickleback (*Culaea inconstans*), ninespine stickleback (*Pungitus pungitius*), trout-perch (*Percopsis omiscomaycus*), Johnny darter (*Etheostoma nigrum*), Iowa darter (*Etheostoma exile*), central mudminnow (*Umbra limi*), freshwater drum (*Aplodinotus grunniens*), black bullhead (*Ictalurus melas*), and brown bullhead (*Ictalurus nebulous*) (LaPointe 1986, Parks 2006).

The landscape surrounding the marsh to east, south and west is composed largely of cattle pasture land and lands cultivated with grain and forage crops such as canola, wheat, barley, flax, hay and alfalfa, as well as cottage and rural residential development (Grosshans 2001, Brown 2003). The beach ridge between the marsh and the lake is partially occupied by seasonal cottages, as well as several permanent residences.

The climate of south-central Manitoba falls within the cool to mild continental category. Typically mean temperatures range from -20°C in winter to 20°C in summer. Annual precipitation is approximately 525 mm, with the majority falling as rain during the spring and summer months, and with gross evaporation generally exceeding precipitation.

The frost-free period ranges from 90 to 100 days, and the growing season is from 170 to 180 days annually (Moulding 1979, de Geus 1987, McGinn 1992, Batt 2000).

Lake Manitoba

Lake Manitoba is the thirteenth largest lake in North America, comprising approximately 4,700 km^2 (Figure 1.1). It is divided into two basins. The south basin contains approximately 70 % of the lake's total surface area. The lake has a mean depth of 4.5 m and a maximum depth of 6.3 m (Last 1980). Lake Manitoba is a part of the Dauphin River drainage basin which comprises approximately 80,114 km^2, with the south basin comprising a drainage area of approximately 17,343 km^2 (Figure 1.4) (Last 1980; Page 2011). The main inflows to the lake are the Waterhen River, Whitemud River and, during periods of high terrestrial runoff, the Assiniboine River Diversion. Water flows out of Lake Manitoba through the Fairford River, into Lake St. Martin, and then to the Dauphin River, which empties into the north basin of Lake Winnipeg.

In 1961, the Fairford Dam was constructed on the Fairford River to stabilize water levels on Lake Manitoba. The dam was constructed to try to control both high and low water levels on the lake, as low water levels reduce boating navigation on the lake and wildlife availability for hunting, and high water levels flood surrounding lowlands with agricultural and cottage development (LMRRAC 2003a). Until this date, water levels on the lake fluctuated in a range of 1.7 m (Figure 1.3). Afterward, water levels were controlled at the target level of 247.6 MASL, operating with a range of ± 0.3 m. As a result, water levels on Lake Manitoba and Delta Marsh have been stabilized within a narrower range of variation than those that existed pre-1961.

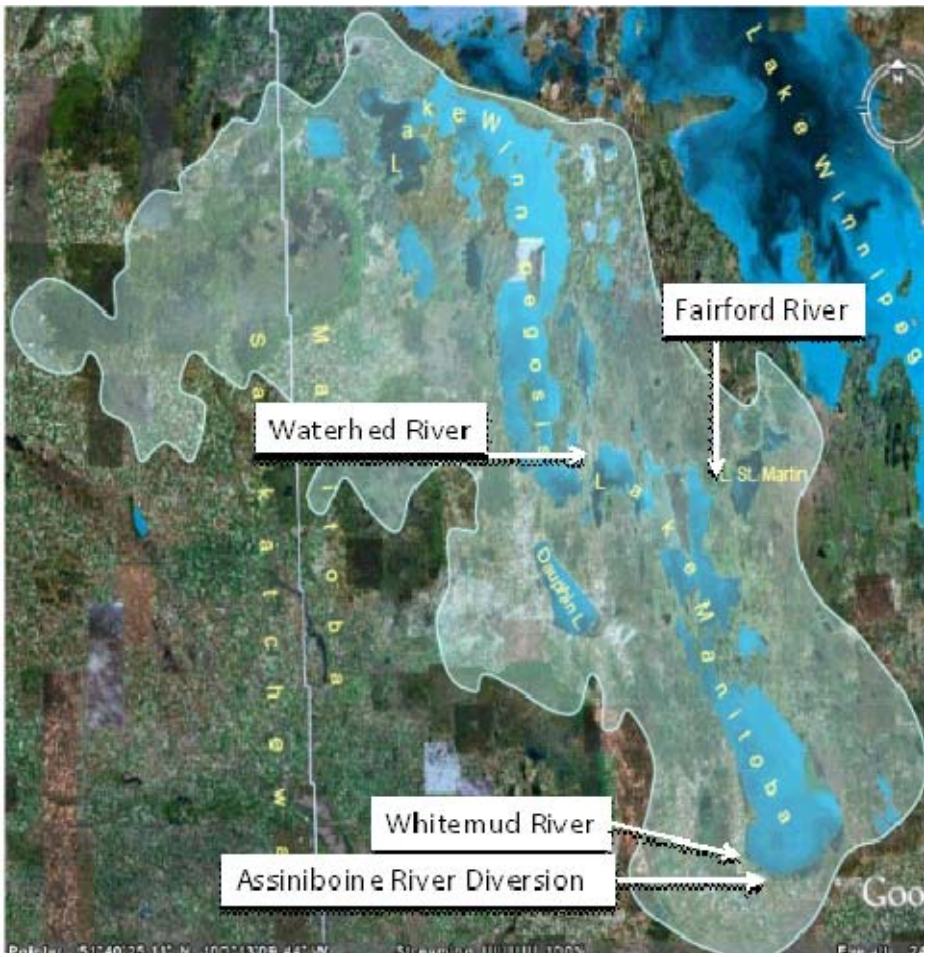

Figure 1.4 Map illustrating the Dauphin River Drainage Basin and Lake Manitoba. Source: Google.

Limited historical water quality data are available for Lake Manitoba as far back as 1926 (Bajkov 1930a). Most scientific data gathered on Lake Manitoba have been biological in nature, and mostly concerning the lake's fishery. Limited water quality data were collected in the 1950s, and during the 1960s and 1970s, there was a water quality monitoring program carried out at up to sixteen sites in the north and south basins of the lake. The program was reduced to six stations in 1973, with three in each basin, and in 1977 it was eliminated altogether. Since 1991, the Manitoba Department of Water Stewardship (now Conservation and Water Stewardship) has monitored Lake Manitoba on a monthly basis at a single monitoring station located one kilometer offshore from the Delta Marsh Field Station (University of Manitoba). The lake is characterized as slightly brackish, alkaline, and ranges from mesotrophic to eutrophic. From 2005 to 2006, the water chemistry of the lake was monitored bi-weekly at seven sites in north basin and eight sites in south basin (Page 2011). The study noted that, from 1991 to 2007, the south lake basin lake experienced statistically significant increases in total P, total N, and chlorophyll a, and decreases in dissolved solids, conductivity, sodium, and chloride. A detailed review of Lake Manitoba hydrology, morphology, and water chemistry is provided by Page (2011).

Objectives and Major Hypotheses

This study was undertaken to address an overall lack of knowledge relating to spatial and temporal patterns in the hydrology, water chemistry, and algal nutrient-limitation status of coastal freshwaters wetlands, specifically Delta Marsh, and the influence of adjoining water bodies on these properties. To obtain more knowledge on these properties, this study specifically addresses three main objectives:

- Characterize the level and flow of surface waters of Delta Marsh during the open water season, and examine any spatial and temporal patterns that are influenced by surface water connection, water level and flow hydrology of adjoining Lake Manitoba (Chapter 4).
- Characterize water chemistry of Delta Marsh during the open water season, and examine spatial and temporal patterns that are influenced by Lake Manitoba (Chapter 5).
- Determine the main nutrient(s) (N and/or P) limiting algal biomass in Delta Marsh during the open water season, and again any spatial and seasonal patterns under the influence of Lake Manitoba (Chapter 6).

Based on these objectives, the following specific hypotheses were addressed:

1. The hydrology of Lake Manitoba has a significant influence on water level and flow hydrology in Delta Marsh, and as a result water levels and flow hydrology in the Marsh will be significantly correlated spatially and temporally with water levels and wind-induced seiche set-up flow on Lake Manitoba.

2. Water levels and flow in the western compared to the eastern portion of the marsh will be more strongly correlated with the lake because of the greater degree of surface water connection, smaller open water area, and shallower depth.
3. Water levels and flow in connected compared to isolated areas of the marsh will be more strongly correlated with the lake and experience a greater degree of fluctuation because of their surface water connection with the lake.
4. Spatially, the chemical properties of marsh water will be positively correlated with increasing distance from the lake via surface water flow, because of a large source of N and P nutrients to the marsh from surrounding agricultural uplands; natural uptake of N and P nutrients in the marsh; and the dilution effect of waters entering the marsh from the lake.
5. Water chemistry in the western compared to the eastern portion of the marsh will be more strongly correlated with the lake because of the greater degree of surface water connection, smaller open water area, and shallower depth.
6. Water chemistry in connected compared to isolated areas of the marsh will be more strongly correlated with the lake due to surface water connection. Isolated areas in the marsh will have greater concentrations of N, P and major ions (Cl^-), and less variation in N, P and major ions (Cl^-), because isolated sites are less influenced by the dilution and flushing effects of the lake.
7. Algae in Delta Marsh are mainly limited by N (as indicated by algal nutrient status), and spatially that N limitation will be correlated with distance from the lake via surface water flow, because of nutrient uptake and release process in the marsh, and the dilution and flushing effects of the lake.

8. Nitrogen limitation will be greater in the western compared to eastern portion of the marsh because of the greater degree of surface water connection, smaller open water area, and shallower depth.
9. Nitrogen limitation will be greater in isolated compared to connected areas of the marsh, because isolated sites are less influenced by the dilution and flushing effect of the lake.

Chapter 2: Literature Review

Wetland definitions and distribution

Numerous attempts have been made by several organizations around the world to develop a formal definition of wetlands. The ability to define wetlands is, however, complicated by their vast diversity, as each is unique in shape, size, hydrology, soils, vegetation and position in the landscape (Mitsch and Gosselink 2000a & b, van der Valk 2006). They can range from tidal to non-tidal, saline to freshwater, lotic to lentic, and permanent to impermanent, and have vastly varying diversities of flora and fauna (Kent 2001). Wetlands are often found at the transition of aquatic and terrestrial habitats, and their boundaries can change over time with change in water inputs and outputs, and modifications in the watershed. This transitional nature makes the precise definition of boundaries difficult. Despite these difficulties, according to the National Wetlands Working Group and wetlands in North America are generally characterized by three features: (1) the presence of water, either at the surface or the root zone and which is generally less than 2 m in depth; (2) ephemerally or continuously waterlogged, anaerobic and reducing soils; and (3) hydrophytic vegetation adapted to the wet conditions (National Wetlands Working Group 1988, Goldsborough and Robinson 1996, Warner and Rubec 1997, Mitsch and Gosselink 2000a & b, van der Valk 2006).

A comprehensive inventory of world's wetlands has not been published, but some reasonable estimates are available. Wetlands are found on every continent with the exception of Antarctica, and are estimated to cover about 4 to 6% of the earth's land surface, or approximately 700 to 900 million hectares (Mitsch and Gosselink 2000a &b, van der Valk 2006; Robelo 2009). Their distribution around the globe is concentrated

around the subarctic areas of North America, Europe, Asia, as well as around river and lakes in South America and Africa (van der Valk 2006). Detailed inventories do exist for parts of Canada, the continental United States, and some countries in Western Europe (van der Valk 2006). North America contains about one-third (240 million hectares) of the world's wetlands, with more than half (130 million hectares) contained in Canada. This represents about 14% of Canada's land area. Of this total, 41% are found in Manitoba and Ontario (National Wetlands Working Group 1988). In Manitoba, wetlands cover approximately 43% of the landscape (23.3 million hectares) (Halsey et al. 1997).

Wetland function and values

Wetlands amongst the most productive, significant and unique ecosystems on Earth (Crumpton 1989, Mitsch 1996). Primary production in wetlands, in the form of macrophytes and algae ranges from approximately 30 to 80 metric tonnes per hectare per year (mT/ha/yr) for emergent macrophytes, 2 to 20 mT/ha/yr for submergent macrophytes, and 5 to 60 mT/ha/yr for algae (Mitsch and Gosselink 2000a). They perform critical ecosystem functions in the landscape including the provision of critical habitat for flora and fauna, groundwater recharge and discharge, protection of shorelines, sequestration of carbon, as well as influencing atmospheric and climate processes (Mitsch and Gosselink 2000a). Recent attempts to quantify the ecological goods and services (EG&S) benefits of wetlands have demonstrated their importance in the agricultural landscape of prairie Canada (Olewiler 2004).

Wetlands provide essential habitat, food, as well as breeding and nursing grounds to a biologically diverse array of species including microorganisms, plants, invertebrates, waterfowl, migratory songbirds, small mammals, fish, and reptiles (Smith et al. 1991;

Mitsch and Gosselink 2000b; van der Valk 2006). Found on the landscape as transition zones between terrestrial and aquatic ecosystems, the biodiversity of these systems is characterized by terrestrial and aquatic species, including endangered and rare species dependent on wetlands for survival (Smith et al. 1991; Mitsch and Gosselink 2000b). They are also highly valued for recreational activities including hunting, fishing, bird watching, and boating (Leitch and Hovde 1996).

Wetlands have been described as the "the kidneys of the landscape" (Mitsch and Gosselink 2000a) for their exceptional ability to improve water quality via unique physical and biogeochemical processes that allow them to retain sediments and act as sinks and transformers of nutrients, metals, agrochemicals and an array of other materials (Mitsch 1996, Kadlec and Knight 1996, Zedler 2003). Often located at the interface between terrestrial and aquatic ecosystems, they have been found to reduce concentrations and quantities of materials entering downstream aquatic systems (Krieger 2001; Zedler 2003). For instance, wetlands in key downstream watershed positions have been found to remove up to 80% of inflowing nitrates (Crumpton et al. 1993). These unique abilities make wetlands highly important in global and regional biogeochemical cycles (Mitsch 1996, van der Valk 2006). In the case of nutrient removal, where the physical, chemical and biological nature of wetland conditions maximize denitrification, this may act as a key permanent N loss process, due to the high organic C content and the highly reducing anaerobic conditions of water-logged sediments, facilitated by shallow waters and long water retention times that allow for increased sediment water contact (Brodrick et al. 1988; Seitzinger 1988; Neely and Baker 1989; Windolf et al., 1996; Kadlec and Knight 1996; Scheffer 1998; Saunders and Kalff 2001; Poe et al. 2003). Nitrogen is also removed via plant and algal uptake (Mitsch and Gosselink 2000a; Moss 2001). Phosphorus is removed

primarily via precipitation of organic matter and burial in the sediments, and assimilation by algae and macrophytes (Kadlec and Knight 1996; van der Valk 2006). These ecosystems can reduce concentrations of metals in surface and ground water by binding the metals to suspended clay particles or humic oxides, which are then removed from waters flowing through wetlands by wetland plants, which reduce flow velocities, and allow sediments to settle out of the water column along with attached metals and organic chemicals. Burial in the wetland substrate helps keep bound metals immobilized (Kadlec and Knight 1996; Cronk and Fennessy 2001). They have also been found to reduce concentrations of many pesticides (Krieger 2001).

Hydrologically, it is widely accepted that wetlands perform critical roles within watersheds as well as in the global water cycle (Carter 1997, Bullock and Acreman 2003). During the growing season, vegetation in wetlands removes water as well as slows the velocity of water flow, thus reducing downstream flooding and soil erosion. The significance of this process is enhanced after spring snow melt and during storm events (van der Kamp 1998, Mitsch 1998). For example, it is estimated that wetlands in the Devil's Lake region of North Dakota can store close to 72% of yearly spring runoff (Ludden et al. 1983). Wetlands also help maintain the level of the water table and serve as an important source for groundwater recharge due to their vegetation, clay and peat soils, as well as other physical features (van der Kamp 1998, Mitsch and Gosselink 2000a & b). In terms of the global hydrologic cycle, they can return over 30% of their annual water inputs to the atmosphere via evaporation and evapotranspiration as a result of their high surface to volume ratio and their vegetation (Richardson and McCarthy 1994).

Most recently, wetlands are being recognized as important global carbon dioxide (CO_2) sinks and climate stabilizers, with the vegetation sequestrating carbon from the atmosphere, and the organic soils acting as carbon sinks when they are flooded and become anaerobic (Mitsch and Gosselink 2007; Bernal and Mitsch 2012). This is an important function, particularly with concerns surrounding increasing levels of atmospheric CO_2 and associated climate change (IPCC 1998, Erwin 2009). Although wetlands cover 6 to 8% of the world's freshwater surface, they are estimated to account for one-third of the world's freshwater organic soil carbon pool (Mitsch and Wu 1995, IPCC 1996, Erwin 2009). The amount of wetland carbon storage and release depends greatly on their hydrogeochemical characteristics which, in turn, determine wetland vegetation and soil characteristics. When wetlands become degraded and drained, the soils are oxidized, increasing organic matter decomposition and the release of CO_2 to the atmosphere (Turetsky et al. 2007).

Wetland degradation and loss

There is growing awareness that wetlands are disappearing at an alarming rate. Their degradation and loss in North America is associated with multiple stressors including agricultural activities, dam construction, stream channelization, mining, filling, development, sedimentation, nutrient enrichment, and the introduction of exotic species (Millar 1989, Simestad et al. 2006, O'Connell 2003, Erwin 2009). Many of the remaining wetlands have been altered chemically, physically and biologically, compromising much of their critical functions.

Crosbie and Chow-Fraser (1999), in a large scale study examining the effects of land use on wetlands in the Canadian portion of the Laurentian Great Lakes, found that concentrations of N and P, phytoplankton chlorophyll a, and turbidity in wetlands increased

predictably with increasing dominance of agriculture in the watershed. On the prairies where intensive agricultural activities occur, wetlands are also subject to increased sedimentation and eutrophication from external sources such as agricultural fertilizers, animal manure, and domestic sewage (Turner et al. 1987).

An estimated half of the wetlands across the globe have been altered, degraded or destroyed over the last 150 years (Mitsch and Gosselink 2000b, O'Connell 2003, van der Valk 2006), with approximately 1% of the global stock of coastal marine wetlands lost per year in the late 20th century (Nicholls 2004). In North America, it has been estimated that 40% of original pre-settlement wetlands have been lost (Canadian-United States Steering Committee 1988, cited in Millar 1989). In the United States, where some of the best estimates are available, an estimated 47 million hectares or greater than 50% of wetlands were lost between 1780 and 1980. In Canada, losses are estimated at over 20 million hectares since European settlement, with 1.2 million hectares or 71% of wetlands lost in the prairie provinces alone (Environment Canada 1986, National Wetland Working Group 1988, Young 1994), and over 85% of these losses attributed to agricultural activities (Turner et al. 1987, Cox 1993) . A study by Hanuta (2001) found that a 9,400 km^2 area of wetlands in the Red River drainage basin of southern Manitoba had decreased to an area of 1,098 km^2 or 11.7% since pre-settlement (1870s), and further to only 14.5 km^2 or 0.2 % by 1995. Furthermore, wetland losses around the Laurentian Great Lakes have been estimated at 68% in Canada and 70% in the United States (Snell 1987, Mitsch and Bouchard 1998). An estimated 80% of wetlands have been lost around western Lake Ontario (Whillans 1982) and 95% of coastal and inland wetlands around the western basin of Lake Erie in Michigan and Ohio, including the 4,000 km^2 Great Black Swamp (Mitsch and Bouchard 1998), have been affected. The majority of the few remaining coastal wetlands around the

Laurentian Great Lakes have been isolated from their previously adjoining lakes and upland watersheds by dikes and dams.

Coastal freshwater lacustrine wetlands

Coastal lacustrine freshwater wetlands or marshes according to the Canadian Wetland Classification System (Warner and Rubec 1997), are wetlands found associated with the boundaries of inland freshwater lakes. These wetlands can be permanently or intermittently connected to their adjoining lakes via surface water flow, or groundwater flow. Marshes are often the most prevalent coastal wetland type, and are generally defined as periodically or continually flooded and characterized by non-woody emergent vegetation that is tolerant of fluctuating shallow waters. Other types of wetlands found associated with lakes can include swamps, fens, and bogs (Warner and Rubec 1997; Keddy and Fraser 2000; Mitsch and Gosselink 2000a; Watchorn et al. 2012).

Some of the best-known coastal freshwater marshes in North American are those found along the shores of the Laurentian Great Lakes (Mitsch and Gosselink 2000a, Ingram et al 2004). Many of the Great Lakes wetlands are riverine marshes formed from isostatic rebound, as well as lacustrine marshes found in protected shallow areas behind barrier beach ridges. Many of the Great Lakes' marshes are now managed and protected from water-level changes by artificial dikes. Estimates of coastal wetlands around the Great Lakes include 179,300 hectares in total along the Canadian and American shorelines (Brazner et al. 2000), with 63,706 hectares along the Canadian shores (Ingram et al 2004).

While the prairie region of North America is most often characterized by pothole wetlands, several large lakes located on the prairies have coastal freshwater wetlands along

their peripheries. Two of the largest coastal freshwater marshes in North America are found in southern Manitoba. Delta Marsh, comprising 18,000 hectares in size, is located at the south end of Lake Manitoba, and Netley-Libau Marsh, 25,000 hectares in size, is located at the mouth of the Red River where it empties into the south end of Lake Winnipeg (National Wetlands Working Group 1988; Grosshans et al. 2004). Lakes Winnipeg, Manitoba, and Winnipegosis together are surrounded by 270, 994 hectares of coastal wetlands (Watchorn et al. 2012).

Recent studies examining the linkages between coastal freshwater wetlands and their adjoining lakes have suggested physical, chemical and biological processes in these coastal areas influence whole lake/wetland ecosystem function. Although common in marine systems, few studies have characterized exchanges of nutrients, energy, organisms, or other materials between coastal wetlands and adjoining large lakes (Brazner et al. 2000; Trebitz 2006; Morrice 2011). It has been estimated that coastal wetlands and littoral ecosystems associated with the Laurentian Great Lakes can account for 14 to 35% of total lake primary productivity, and as much as 41% of lake-wide production occurred in these areas prior to European settlement (Brazner et al. 2000).

Role of hydrology in wetlands

Hydrology is a key factor controlling wetland structure and function (Gosselink and Turner 1978, Bedford 1992, Gilman 1994, Mitsch and Gosselink 2000a, Grosshans 2001, Wilcox et al. 2007, Wilcox 2012). Due to their small volume and shallow waters, wetlands are dynamic environments in which small changes in hydrology can result in significant changes in physical, biological and chemical characteristics such as physical habitat, water and sediment chemistry, nutrient availability and cycling, organic matter accumulation and

decomposition, and the diversity, productivity and community structure of flora and fauna (Gosselink and Turner 1978, LaBaugh et al. 1998, Keough 1999, van der Valk 2006). As a result, the hydrological regime is one of the most important characteristics in wetland classification schemes (National Wetlands Working Group 1988, Goldsborough and Robinson 1996, Warner and Rubec 1997, Mitsch and Gosselink 2000a, van der Valk 2006).

The major sources of water for wetlands are determined by basin geomorphology and local climate (van der Valk 2006). The hydroperiod or water budget (the balance between inflow and outflows water budget) can vary dramatically seasonally and inter-annually. Major constituents of a wetland's hydroperiod include precipitation, evaporation, evapotranspiration, overland flow, surface water flow and groundwater flow (Gosselink and Turner 1978, van der Valk 2006). Given such sources of variability, water levels in wetlands can be expected to exhibit considerable temporal variability.

There have been few studies of the hydrology of specific wetland types, and the resultant influence on physical, biological and chemical properties (Mitsch and Gosselink 2000a; Trebitz 2006, Morrice et al. 2011). One of the few extensive studies was the Marsh Ecology Research Program (MERP), a ten-year study conducted in ten artificially constructed experimental wetland cells within Delta Marsh. MERP examined the effects of water level manipulation, via a simulated wet-dry cycle, on ecosystem characteristics such as water chemistry, primary production, invertebrate populations and avian and mammal use (Murkin et al. 2000). Results indicated that flood depth and duration are critical determinants of spatial and temporal patterns in emergent vegetation species distribution and productivity at a habitat scale (van der Valk et al. 1994). The wet-dry cycle is essential

for both the removal and regeneration of marsh vegetation (van der Valk and Davis 1978). Prolonged high water levels are important to kill off emergent vegetation, and conversely, low water levels expose mudflats with buried seed banks that allow for plant recolonization and regeneration. Fluctuating water levels also allow wetlands to be more extensive, productive, and diverse than they would be if water levels were stable (Manard and Wilcox 1997). Keddy and Fraser (2000) suggest that productivity is higher in wetlands with high flow through, or a pulsing hydroperiod, and that prolonged periods of stabilized water levels (i.e., reduced magnitude) result in reduced habitat complexity and biodiversity.

Water levels, flow patterns and water residence time in wetlands influence many chemical, physical and biological processes including sediment biochemistry, water chemistry, decomposition, organic matter, oxygen availability, metal concentrations, nutrient availability, pH, and gas production (Gosselink and Turner 1978). The longer the water residence time, the greater the influence on these processes and characteristics. Hydrology also transports sediments, nutrients, and other materials in and out of wetlands (Crosbie and Chow-Fraser 1999). Except in nutrient-poor wetlands such as bogs, water inputs are a major source of nutrients to wetlands, and outputs remove biotic and abiotic materials such as dissolved organic C, dissolved ions, toxins, and excess sediments and detritus (Mitsch and Gosselink 2000).

In the case of coastal freshwater wetlands, their inherent hydrological complexity resulting from highly variable influences of upland water sources as well as those of adjoining lakes, has resulted in relatively few detailed studies examining hydrological interactions, and their influence on the water chemistry and algal nutrient status of adjoining coastal wetlands (de Geus 1987, Morrice et al. 2004, Trebitz 2006, Morrice et al.

2011). The majority of existing studies have focused on the coastal freshwater wetlands of the Laurentian Great Lakes, with little study of other coastal freshwater wetland systems in North America. In the case of coastal freshwater wetlands, one important characteristic feature is that water levels and flow can be influenced by water level fluctuations of an adjoining lake, both on the short term (i.e. storms and seiches) and longer term inter-annual fluctuations (Maynard and Wilcox 1997, Sierszen et al. 2006, Trebitz 2006, Helvca and Wells 2009, Gathman and Burton 2011, Morrice et al. 2011). The full effects of lake seiches on adjoining wetlands are still uncertain, but they appear to act similarly to tides in marine coastal marshes, allowing exchange of water, nutrients and other materials between wetlands and the lakes, as well as having important impacts on plant and animal species exchange, diversity, and productivity (Maynard and Wilcox 1997, Sierszen et al. 2006, Trebitz 2006, Gathman and Burton 2011, Morrice et al. 2011). The hydroperiod of coastal wetlands along the Laurentian Great Lakes varies considerably, depending on whether water levels are managed or exposed to natural levels of lake and/or river outflow and inflow (Krieger 1989, Maynard and Wilcox 1997). Of course, other water sources to coastal freshwater wetlands—surface runoff from the surrounding watershed; inputs from upland rivers and streams; groundwater discharge, and precipitation—contribute to their complexity and, as a result, coastal wetlands can be subject to frequent and often unpredictable changes in water level. These changes, as well as other natural disturbances, can lead to high biological diversity, especially if water levels on the adjoining lakes are allowed to fluctuate naturally (Keddy and Fraser 2000).

Various methods have been used to examine the hydrology of wetlands including direct measures of flow and water level meters (i.e. point measurements and real-time), as well as measures of chemical ‘tracers’ including conservative ions, dyes, and stable

isotopes (Gosselink and Turner 1978, Kadlec and Knight 1996). Gilman (1994) contains a good summary of the general wetland water balance, as well as an overview of the application of various direct measures measure, and Ward and Trimble (2004) contains and overall summary of hydrological processes in aquatic systems, and methods used for quantifying hydrologic parameters.

Chemical tracers are used in a variety of aquatic environments, such as aquifers, streams, rivers, estuaries, reservoirs, lakes, and wetlands to determine water residence times, dispersion and flow velocities, and flow paths of surface and ground-water (Kadlec and Knight 1996). Often naturally occurring salts and conservative ions are used, including chloride (Cl^-), bromide (Br^-), and magnesium (Mg^+) (Trebitz et al. 2002, Waiser 2006, Morrice et al. 2011). These major ions represent useful tracers because they are relatively inert, not easily adsorbed to surfaces and incorporated into minerals that form in soils, and they are not used to any great extent by biota. In addition, Cl^- and Br^- have been used to evaluate release and/or retention of other reactive chemical constituents (nutrients, metals, etc.) in aquatic systems in the northern prairie region of North America (Kadlec and Knight 1996; Hayashi et al. 1998; Waiser 2006) as well as wetlands of the Laurentian Great Lakes (Trebitz et al. 2002; Morrice et. al. 2011).

Naturally-occurring isotopes such as oxygen-18 (^{18}O) and deuterium (^2H, D) have been used to track global hydrological budget, as well as track the movement of ground and surface water on regional and local scales (Gibson et al. 2002; Clay et al. 2004; Gibson et al. 2005). ^{18}O and ^2D are incorporated into the water molecule and are subject to fractionation when water evaporates and is depleted of heavy isotopes, in comparison to the water source (Gonfiantini 1986). The isotopes of $^1H^2H^{16}O$ and $^1H_2{}^{18}O$ are heavier than

$^{1}H_{2}{}^{16}O$, which is more abundant globally. The physical processes that result in the fractionation of the heavier isotopes from the lighter isotopes during evaporation are complex. An excellent overview of these processes is provided in Gonfiantini (1986) and Clark and Fritz (1997). The end result of these processes gives water a characteristic ^{18}O and ^{2}H signature that denotes its origin. These characteristic signatures can be used to determine water movement and flow dynamics in both ground water and surface waters (Peters et al. 1993), including determining the inputs and outflows of water between water bodies. For instance, shallow water bodies, which are generally well mixed during the ice-free season, develop a characteristic isotopic signature which is acquired by outflowing waters and can be used in watershed studies to track movements of discrete water masses. They can also be used to study rainfall runoff from different sources (Peter et al. 1993). ^{18}O and ^{2}H are also subject to seasonal changes, as evaporation in water bodies generally peak in the summer so seasonal changes in water budgets can be examined (Clark and Fritz 1997).

Water chemistry of freshwater wetlands

The water chemistry of wetlands is determined primarily by geological location, hydrologic water balance (relative proportions of inflow, outflow, and storage), quality of inflowing water, type of soils and vegetation, and human activity within or near it (Mitsch and Gosselink 2000). Wetlands such as the prairie potholes that receive surface-water or ground-water inflow and limited outflow, that lose water primarily to evapotranspiration, tend to have high concentrations of chemical constituents rendering them brackish to saline. In contrast, wetlands that receive water mainly from precipitation and losing water

primarily via surface-water outflows and/or seepage to groundwater tend to have lower concentrations of chemicals and are less brackish.

Freshwater wetlands are generally mesotrophic to eutrophic, with high nutrient levels resulting in characteristically high macrophyte and algal productivity (Warner and Rubec 1997). The pH of these systems ranges from circum-neutral to highly alkaline due to the presence of high quantities of dissolved minerals such as calcium, potassium, carbonate and bicarbonate (Warner and Rubec 1997; Mitsch and Gosselink 2000a). The underlying soils are often composed of high amounts of minerals rather than peat, overlain with autochthonous organic matter from the decomposition of resident vegetation. The biological and chemical composition of the soils and resultant biochemical reactions influence the chemical properties of overlying waters. When soils become water-saturated, microbial respiration and biochemical reactions consume oxygen, causing the soils to become anaerobic. These anaerobic conditions, in turn, influence nutrient cycling, pH, sediment and organic matter accumulation, decomposition, and metal concentrations in the sediment and water (Kadlec and Knight 1996).

The characteristically high nutrient levels associated with wetland sediments result in high rates of bacterial activity and rapid decomposition and recycling, and rapid biomass turnover rates. Bacterial production has been noted to be 25 to 50 times greater in coastal littoral zones than in offshore waters in Lake Erie (Hwang and Heath 1997). The tight interaction between heterotrophic microbes, periphyton, and macrophytes in these systems results in high rates of nutrient transfer between these components and high productivity (Cotner et al. 2009).

Wetlands that are dominated by surface inflow and outflow from lakes and/or rivers will have chemistry similar to that of the adjoining water bodies. In most cases, wetlands receive water from more than one source, so the resultant water chemistry is a composite of the various sources. In the case of coastal wetlands, attempts to better understand the physical, and biochemical processes that occur within coastal wetlands, as well as the influence of adjoining water bodies (i.e. watershed tributaries and lakes) on these processes, has been an area of growing research over the last decade. Studies to date, however, have primarily occurred around the Laurentian Great Lakes wetlands, making it difficult to generalize and apply findings to other systems (Trebitz 2002, Morrice et al. 2004, Trebitz 2006, Sierszen 2006, Trebitz et al. 2007, Wilcox and Nichols 2008, Morrice et al. 2011, Trebitz et al. 2011). Less is known about the numerous other coastal freshwater wetlands that surround many other lakes, including Lakes Winnipeg, Manitoba and Winnipegosis in Manitoba (Watchorn et al. 2012).

Chemical processes in coastal wetlands have been shown to vary amongst different systems, and within systems, and hydrology is a key influence on these processes (Keough et al. 1999, Grosshans 2001, Trebitz et al. 2002, Morrice et al. 2004, Sierszen et al. 2009, Trebtiz 2006, Wilcox et. al. 2007, Wilcox et al. 2008, Gatham and Burton 2011, Wilcox 2012). Due to their small volumes and shallow water depths, wetlands are dynamic environments in which small changes in water levels can result in significant biological and chemical changes (Kadlec and Knight 1996, van der Valk 2006). Further, in the case of coastal freshwater wetlands, hydrology is further complicated as these systems are subject to an interplay of varying inputs of water and nutrients from the watershed by tributary flow and/or adjoining lakes by seiche activity (Wetzel 2001, Trebitz et al. 2002, Morrice et al. 2004, Lotze et al. 2006; Trebtiz 2006, Trebitz et al. 2007, Wilcox et al. 2007, Diaz and

Rosenberg 2008, Morrice et al. 2011, Trebtiz et al. 2011). The influence of these hydrological connections can vary greatly due to their own temporal variability. Changes in the magnitude of these hydrological influences may also affect water residence time which, in turn, influences water chemistry parameters, including nutrient availability, sinks and sources (Wold and Hershey 1999, Trebtiz et al. 2002, Morrice et al. 2004, Trebtiz et al. 2004, Sierszen et al. 2006, Trebitz 2006, Morrice et al. 2011), with nutrient availability and retention increasing with increased hydrological residence time. Other studies in the Laurentian Great Lakes have also attributed spatial differences in the water chemistry of coastal wetlands to hydrological influences of adjoining lake. Trebtiz et al. (2005) found that spatial differences in aquatic habitat within ten coastal marshes of Lake Superior were larger than differences amongst the marshes, and habitat patterns were strongly associated with morphology and hydrology. Further back-bay segments tended to demonstrate lower levels of seiche-induced water movement, and they were prone to high water temperatures and low dissolved oxygen levels. Increasing seiche inputs tended to homogenize habitat elements among wetland segments. Trebtiz et al. (2004) noted that hydrologic connection of Lost Creek Wetland to Lake Superior, as well as to the upland watershed resulted in large spatial and seasonal variations in the hydrology and nutrient (N and P) dynamics of the wetland.

Studies examining the influence of coastal freshwater wetlands and their adjoining lakes have suggested that physical, chemical and biological processes in these coastal areas will influence whole lake/wetland ecosystem function. Studies in coastal freshwater wetlands have found that that coastal wetlands act as transformers of nutrients (N and P) from inorganic to organic forms (Krieger 1989). This occurs primarily via uptake by algae

and macrophytes, and results in reductions of inorganic forms of nutrients entering downstream adjoining lakes.

Algae and nutrient limitation in coastal freshwater lacustrine wetlands

Several algal assemblages are found in wetlands, which differ based on their physical location and ecological requirements. There are generally four significant and somewhat distinct assemblages of algae found in prairie wetlands: phytoplankton, epiphyton, metaphyton and epipelon (Goldsborough and Robinson 1996). Phytoplankton is planktonic algae suspended in the water column of pelagic zones, or open-water areas, and may be motile or non-motile. “Periphyton” or “benthic algae” are terms used to describe algae that grow on all submerged surfaces in general (macrophytes, rocks, sands, sediment, etc.). Epiphytic algae grow attached to the surfaces of submerged or emergent vascular and nonvascular macrophytes. Metaphyton is an assemblage of unattached algae forming large, floating mats. Metaphyton originates as epiphyton which detaches and floats to the surface, due to oxygen trapped in the mat. Epipelic algae are motile algae that grow on the sediments, and can migrate in the sediment column in response to environmental conditions such as light intensity.

In fertile wetland ecosystems, algae can contribute a large portion of the total primary production of the system, in some cases surpassing that of macrophytes on a percentage area of marsh surface area basis, due to the shallow and fertile nature of wetlands, as well as the high turnover times for algae (days) compared to macrophytes (year)(Robinson et al. 1997, Mitsch and Gooselink 2000a, Murkin et al. 2000). During the Marsh Ecology Research Program (MERP) it was found that algal productivity in the MERP cells ranged from 362 to 813 g C/m^2/yr compared to 18 to 203 g C/m^2/yr (Robinson et al. 2000). There

is often a switch from benthic to pelagic algal productivity as nutrients become more abundant; presumably lack of light availability decreases benthic production in plankton-dominated systems (Vadeboncoeur et al. 2002).

Algae require many inorganic micro- and macro-nutrients to grow including carbon, oxygen, hydrogen, nitrogen, phosphorus, silicon, potassium, sulfur, calcium, iron, manganese, copper, and other trace metals (Borchardt 1996; Wetzel 2001). Sources of nutrients to wetlands are varied, with many point and non-point sources including inflows from adjoining water bodies, surface water runoff, groundwater, as well as internal decomposition and recycling. External sources continue to grow as human activities increase the availability of N and P from urban and industrial development, increasing municipal sewage disposal, regulation of wetlands and streams and more intensive agricultural and livestock farming practices (Vitousek et al. 1997, Carpenter et al. 1998, Jeppesen 1998, Schindler and Vallentyne 2008). Agricultural activities can be major sources of nitrate, which is very soluble in water (Moss 2001). Nitrogen fertilizer use has increased remarkably over the past several decades (Galloway et al. 2004). Groundwater flow usually represents small sources of P whereas surface inflow is usually a major source because of the ability of P to adsorb onto sediments and particularly clay (Moss 2001). Phosphorus is often targeted for control or removal because it enters waters from mostly point sources such as sewage treatment plants and non-point sources such as farmland, and methods for removal of P from domestic sewage and measures to protect soils from erosion are more cost-effective for P than N (Moss 2005). Furthermore, P compounds can be more readily precipitated than the most soluble N compounds. The nutrient(s) in lowest supply relative to algal physiological demands can be said to be limiting to growth (Borchardt 1996), and understanding which nutrient(s) are limiting algal biomass is a key aspect of

eutrophication research in aquatic ecosystems. Eutrophication is a major problem in a number of aquatic systems in the world, and there is a considerable financial cost associated with remediating its effects. As a result, research examining the nutrient limitation status of algae and the resultant ecological impacts has had a long history in aquatic ecology (Havens et al. 1999). Excellent reviews of algal nutrient uptake and growth kinetics, as they relate to nutrient-limitation, can be found in Borchardt (1996) and Wetzel (2001).

The few studies examining algal nutrient limitation in prairie freshwater wetlands have focused on phytoplankton (Kadlec et al. 1986, Murkin et al. 1991, Detenbeck et al. 2002) with little detailed study of other algal assemblages (Hooper-Reid and Robinson 1978, Murkin et al. 1991, Kiers-North 2000). Periphytic algae can represent a substantial component of primary production in wetland systems due to abundant available substrata (macrophytes and sediments) and high subsurface irradiance levels in these shallow systems (Robinson et al. 1997). Due to high benthic algal biomass in wetlands (Goldsborough and Robinson 1996), they can contribute significantly to key wetland functions such as nutrient cycling and trophic transfer (Lamberti 1996; Sierszen et al. 2004). Periphyton biomass has been found to be a good indicator of anthropogenic water quality degradation. Unlike aquatic plants, benthic algae obtain a great deal of their nutrients from the water column, so they are ideal organisms to monitor nutrient enrichment from land use. They respond quickly to nutrient additions due to their high productivity and rapid turnover rate (McNair and Chow-Fraser 2001, Lavoie et al. 2004).

In general, N and P are the principal macro-nutrients found to limit algal growth and production in the majority of water bodies, as they are typically the two nutrients in least

supply (Vymazal 1995; Wetzel 2001). Nitrogen and P entering wetlands are present in organic and inorganic forms, with relative proportions dependent on the source and type of water. Organic N is present in particulate and dissolved forms, while inorganic N forms (NH_4-N, NO_3-N, and NO_2-N) are dissolved. Organic P is present in particulate form (organically-bound to plant or animal tissue) and dissolved (soluble) forms, and inorganic P is present in orthophosphate and polyphosphate forms. Silicate can also be a limiting nutrient in some situations, specifically for diatoms, and perhaps occasionally iron or other trace metals.

Of the two nutrients, P has long been accepted as the key nutrient limiting to algae and plant growth in freshwater lakes due to successful experiments that have reduced eutrophication by controlling P inputs to large, deep lakes in catchments that were not intensively farmed (Vallentyne 1970, Schindler 1977, Schindler 1978, Hecky and Kilham 1988, Carpenter et al. 1992, Wetzel 2001, Howarth and Marino 2006; Schindler and Vallentyne 2008). In the last decade, however, there have been suggestions that algal P limitation in freshwaters may not be as dominant as previously thought, especially in lowland shallow lakes and wetlands located in productive watersheds (Hameed et al. 1999). Recent studies are showing that N may be as limiting to algae as P, if not more so when there is an excessive P supply (Reddy et al. 2000, Moss 2001, Moss 2005, Howarth and Marino 2006, Elser et al. 2007, Sterner 2008). Studies conducted in wetlands have indicated that algal nutrient uptake, transformation, recycling and release in these ecosystems appears to differ from that of other freshwater systems, based on key differences in their internal physical, chemical, and biological processes that influence N and P cycling, storage and removal, including water depth and residence time, water and sediment chemistry and oxygen conditions, and biological uptake and release. These key

differences have been found to create ideal conditions where N instead of P is less available to primary consumers, resulting in N-limited conditions (Moss 2001).

Phosphorus removal and storage in wetlands primarily occurs via precipitation to organic matter and burial in the sediments, and assimilation by algae, macrophytes, and bacteria (Kadlec and Knight 1996; Cronk and Fennessy 2001; Scheffer 1998; Saunders and Kalff 2001; Moss 2001; Poe et al. 2003; van derValk 2006). However, these internal storage mechanisms are not permanent as P can be released from the sediments back to the water column through a variety of means. High rates of decomposition and bacterial metabolism in the sediment consumes oxygen, and may result in anaerobic sediment conditions as the diffusion of oxygen back into the sediments cannot keep up with microbial demand (Moss 2001). Under anaerobic conditions, P bound to oxidized sediments and metals is converted to the soluble P that diffuses upward along the decreasing concentration gradient to the sediment surface. The oxidized microzone at the sediment-water surface can prevent P release, resulting in high concentrations near the sediment-water interface. However, when the oxidized microzone is disturbed or destroyed, soluble P is released into the water column. High sulfate concentrations are also common in the anaerobic sediments of wetlands, and result in the formation of hydrogen sulfide. Hydrogen sulfide can precipitate and reduce metals such as iron and form iron sulfides (FeS), resulting in the release of bound P (Caraco et al. 1989, Caraco et al. 1990, Gächter and Müller 2003). The release of P from the sediment in these shallow systems is further enhanced by frequent physical disturbance by wind and waves, bioturbation by fish and invertebrates (Riley and Prepas 1984, Scheffer 1998, Søndergaard et al. 1996), and uptake and release by rooted macrophytes (Søndergaard et al. 2003, Dunne and Reddy 2005). Phosphorus release from the sediments often occurs in the summer when water

temperatures are high, and soil oxygen conditions and water flow in wetlands is low, creating ideal conditions for phytoplankton uptake and growth (Jeppesen et al. 1997, Scheffer 1998, Moss 2001, Gächter and Müller 2003, Sødergaard et al. 2003, Dunn and Reddy 2005). The abundance of vegetation and algae results in high rates of nutrient uptake and transformation, as well as providing large amounts of organic matter and nutrient release during fall senescence, which consumes oxygen during decomposition. The high rate of microbial decomposition and recycling, coupled with anaerobic conditions, results in high rates of internal P recycling (Jeppesen et al. 1997, Scheffer 1998, Moss 2001, Gächter and Müller 2003, Sødergaard et al. 2003, Dunn and Reddy 2005). Moss (2001) found that P was unlikely the nutrient driving eutrophication in the Norfolk Broads of England, due to high rates of internal recycling and release of P from the sediments, and thus its continued availability.

Many of these same conditions in wetlands create conditions which maximize the permanent removal of N, via denitrification, with nitrate and nitrite reduced to nitrogen gas (N_2) for release to the atmosphere (Saunders and Kalff 2001). The high organic carbon content and the highly reduced anaerobic conditions of the water-logged sediments, coupled with shallow waters and long water retention times, increase sediment water contact and can promote the consumption of nitrate in the sediments by denitrifying bacteria, with nitrate used as an oxidizing agent during denitrification (Brodrick et al. 1988; Seitzinger 1988; Neely and Baker 1989; Windolf et al., 1996; Kadlec and Knight 1996; Scheffer 1998; Saunders and Kalff 2001; Poe et al. 2003). Scott et al. (2005) found that a treatment wetland complex in Waco, Texas consistently removed more N than P from the water column, largely from high rates of denitrification, resulting in decreased water column N:P ratios and increased N-limitation of periphyton at outflow sites. Productive

rooted vegetation in wetlands may readily assimilate N in many forms from the oxygen-poor sediments, further reducing oxygen conditions in the sediments and increasing denitrification rates. Some ammonium ions are available from animal excretion, but it is also consumed quickly. Any small amount of remaining N is available for ready assimilation by plants, algae, and bacteria and is eventually converted back to nitrate and N_2, which escapes to the atmosphere. Thus, while P is internally recycled and can become readily available, N becomes less available in the water column and is in greater demand throughout the growing season, resulting in undetectable N conditions in the summer months (Moss 2001). Nitrogen is often not replenished in the system until spring snow melt. While it has been argued that N limitation should not occur in aquatic systems due to N_2-fixation by some species of cyanobacteria (Schindler 1977), other studies have shown that N fixation only supplies a minute proportion (2%) of the N requirements of N_2-fixing cyanobacteria (Ferber et al. 2004). Furthermore, fixed N is often not sufficiently available to other algal species during critical times of the growing season, as it is often stored internally by cyanobacteria and not readily released until blooms collapse (Glibert and Bronk, 1994).

Several studies have noted N limitation of aquatic macrophytes in shallow systems (Van Donk et al. 1993, Meijer et al. 1994). Nitrogen-limited macrophytes often produce litter that has high C:N ratios, and high contents of lignin and other recalcitrant compounds compared with N-sufficient plants (Vitousek et al. 1991). These N-limited plant tissues decompose much more slowly, further reducing N availability for other primary producers. Moss (2001) noted low N concentrations are required for diversity of aquatic macrophytes and less dominance by species that outcompete others at high N conditions.

There is evidence for maximum P assimilation in wetlands (Vaithiyanathan and Richardson 1997, Richardson and Qian 1999) while their capacity to remove N is generally high and can be unlimited (elevated denitrification). Craft (1997), in a study examining N and P removal mechanisms in young and old estuarine marshes, found that the amount of N stored in accumulating sediment organic matter and denitrification provided reliable N sinks regardless of N loading rates, whereas in the case of P, sediments can become saturated quickly. Overall, the retention of N increased with time, whereas P retention was greatest when the marshes were young (<10 years), and decreased with time as the soils became saturated. Thus, if a maximal P assimilation capacity exists for wetlands (Richardson and Qian 1999), and denitrification can provide a permanent sink for N, wetlands with high P loads, like the study wetlands here, may be predisposed to long-term N limitation.

A number of physiological methods have been used to examine the nutrient limitation status of algae in aquatic ecosystems. These include laboratory and *in situ* bioassays, alkaline phosphatase activity, P debt, surplus P, P uptake kinetics, ammonium uptake kinetics, stimulation of C uptake, and molar ratios of water or algal cell nutrient concentrations (Healey 1975, Hooper-Reid and Robinson 1978, Goldsborough and Robinson 1996).

The most direct indicators of yield-limiting factors are the results of enrichment bioassays, using batch or continuous cultures and natural populations of test species (Elser and Kimmel 1986, Elser et al. 1990). Bioassay can occur under controlled laboratory conditions or under natural conditions *in situ*, and can be used to examine the nutrient needs of specific algal groups or species, or entire assemblages. Response to enrichment

can be measured as accumulation of biomass and assessed by chlorophyll *a* concentration. Some of the most direct and convincing evidence of nutrient limitation comes from *in-situ* nutrient enrichment studies where algal growth has been stimulated by nutrient additions via whole-system nutrient enrichment, partial system nutrient enrichment (mesocosms), and nutrient-diffusing substrata (NDS). Nutrient diffusing substrata have been used in lentic and lotic systems, and are effective nutrient bioassays as they are economical, stimulate significant algal growth, reduce sampling variability due to their uniform size, and allow for the assessment of temporal and spatial changes in nutrient conditions over long periods of time (~1 to 8 weeks). They can also be used *in situ* under natural conditions, and multiple nutrients can be manipulated at one time (Pringle 1987, Fairchild et al. 1988, Wold and Hershey 1999, Tank and Dodds 2003, Scott et al. 2005).

Measurements of water chemistry can yield information about nutrients limiting phytoplankton algae biomass in aquatic systems. Molar ratios of N:P in the water column are frequently used to predict limiting nutrient(s). The two most commonly used ratios are total N to total P, and dissolved inorganic N (DIN) to soluble reactive phosphorus (SRP; orthophosphate). Studies examining molar ratios of water column concentrations of N and P have found that TN:TP and DIN:DIP ratios <20:1 (<10:1 by weight) can indicate N limited conditions, and ratios >33:1 (>15:1 to 20:1 by weight) can indicate P-limited conditions, with ratios between the two indicative of conditions with no measurable nutrient limitation (Smith 1979, Schanz and Juon 1983, Morris and Lewis 1988, Francoeur et al. 1999, Guildford and Hecky 2000). DIN:TP is a less commonly used ratio that has been used successfully to discern nutrient limitation in lakes (Morris and Lewis 1988, Axler et al. 1994), with molar ratios > 9:1 associated with P-limitation, and ratios <3:1 associated of N-limitation. The primary assumption of DIN:TP ratios is that DIN

approximates most available N (nitrate+nitrite-N and ammonia-N), as organic N is often composed of high molecular weight compounds that are not directly available for algal uptake. Total P often best approximates available P, as P is luxury consumed by many algae, causing the quantity of inorganic P concentrations alone to underestimate available P (Morris and Lewis 1988, Scheffer 1998, Axler et. al. 1994, Francoeur et al. 2003). Other potentially useful tests of nutrient deficiency include ratios of particulate P to C, alkaline phosphatase activity, and high P debt per unit biomass. Nitrogen-deficient algae take up N compounds rapidly and uptake rates via N debt experiments can be used as an indicator of N deficiency (Healey and Hendzel, 1980, Axler et al. 1994).

Other factors that can limit algae in wetlands

Wetlands are dynamic and diverse, so nutrient limitation is only part of the story of what determines algal production. Other physical, chemical and biological factors can affect algal production and species composition, with some of the most notable including hydrology (Kadlec 1979), light availability (Hill 1996), temperature (DeNicola 1996), and grazing pressure (Steinman 1996). Many of these factors interact and influence each other, confounding algal nutrient limitation rates and biomass. Further, the degree of their importance and interaction changes spatially and temporally between water bodies.

Hydrology is one of the dominant variables that influences algae, directly and indirectly, as it affects other physical, chemical and biological properties in aquatic ecosystems (Goldsborough and Robinson 1996, Murkin et al. 2000). Annually and inter-annually water levels in wetlands can vary considerably from drought to flooded conditions (Murkin et al. 2000). During the Marsh Ecology Research Program (MERP) it was found that phytoplankton biomass increased under low waters conditions, whereas biomass of

periphyton and metaphyton increased under flooded conditions with the increased area and availability of flooded substratum, i.e. macrophytes and sediments. Water levels can also affect substrate availability for periphyton, including macrophytes and sediments (Robinson et al. 2000). The shallow water in wetlands makes the sediments susceptible to disturbance from wind and waves, which can in turn affect turbidity, nutrient release, light transmittance, and algal biomass (Hellström 1991, Scheffer 1998, Robinson et al. 2000). Water mixing and turbulence can also affect certain algal taxa. Cyanobacteria require calm water conditions to remain at the water surface, whereas others species that are subject to sinking (i.e. diatoms) do better with turbulent conditions as they require the water motion to remain suspended in the water column. (Reynolds 1984, Wetzel 2001; Wehr and Sheath 2003). Thus, different algal taxa do better at certain times of the season than others. For instance spring peaks of diatoms are often associated in past with windy periods, while cyanobacteria peaks are often associated with calmer days in the summer (Scheffer 1998).

The biomass and species composition of algal assemblages are influenced to a large degree by invertebrate and vertebrate grazing. Cladocerans, copepods, chironomids, and ostracods readily graze algae, and their abundance can increases in response to increased algal production as well as other factors (Steinman 1996, Liess and Hillebrand 2004). Characteristics of algal species influence their vulnerability to grazing. Characteristics include size, shape, motility, and toxin production (Reynolds 1984; Moss 2001). For instance, larger algae are less vulnerable to filter-feeding grazers, and algae that produce toxins, like cyanobacteria, can reduce their palatability to invertebrate grazers (Wehr and Sheath 2003). High light levels have been found to enhance effect of grazers on benthic algal biomass (Hillebrand et al. 2004, Hillebrand 2005). Grazers can improve nutrient availability to lower layers of periphyton growth on substrata by foraging on the overstory

periphyton, as well as by removing surface detritus (McCormick and Stevenson 1991). Moss (2001) noted the importance of zooplankton grazing on planktonic algae in the "Alternative Stable States" model, with both the macrophyte and the clear water state of wetlands, as well as the phytoplankton turbid state, being able to persist over a wide range of nutrient concentrations state without switching back between the two states, with the exception of the extremes of high nutrient concentrations where phytoplankton will dominant or too little nutrients and phytoplankton cannot dominant. As a result, there are other factors aside from nutrient concentrations that influence the predominance and stability of either state, with zooplankton grazing being one of the predominant factors. Zooplankters, particularly *Daphnia* and other daphnids, can graze algae heavily, particularly planktonic species. However, planktivorous fish, grazing amphibians, as well as pesticides can reduce zooplankton populations, and raise the potential for increases in phytoplankton (Moss 2001). In wetlands, aquatic vegetation provides a critical cover for zooplankton by sheltering them from grazing fish and amphibians. The lower layers of dense vegetation can also become deoxygenated due to decomposition and further reduce accessibility to fish and amphibians (Moss 2001).

Light availability in the water column is critical for algal photosynthesis (Hill 1996, Hillebrand 2005). The transmittance of light in wetlands is highly variable and can be influenced by a number of factors including turbidity, dissolved organic compounds, as well as macrophyte growth and self-shading (Hill 1996; Robinson et al. 2000). Hillebrand (2005) noted that light was of similar importance as nutrients in benthic algal biomass, and that the importance of light increased at high nutrient supplies. The relative nutrient content of algae is also affected by light, with high light levels often relating to high C: nutrient ratios, (with high ratios) resulting in poorer food for herbivores (Hillebrand et al. 2004).

Photoinhibition at the surface of waters has also been reported in a few cases, but is often species specific, and rare for periphyton (Hill 1996).

The shallow waters of wetlands and resultant wind induced mixing in wetlands generally results in little difference in temperature with depth (Robinson et al. 2000). However macrophyte and algal growth can result in significant shading of the water column and reduced temperature (McDougal 2001). McDougal (2001) found strong positive correlations between metaphyton biomass and warmer temperatures in Delta Marsh. Metaphyton mats have also been shown to significantly reduce light and temperature in the water column of wetlands (Dodds 1991). Cyanobacteria dominance in aquatic communities has also been found to be greater with warmer water temperature (Yamamoto 2009). Climate change had also been predicted to change the species composition of algae in wetlands, for instance with increased temperatures resulting in increased instances of cyanobacteria blooms (Elliot 2012, Paerl 2012).

Chapter 3: Methods

Study Sites and Design

Study sample sites in Delta Marsh were located in both sites connected to or isolated from surface water connection with Lake Manitoba from 2002 to 2005 (Figure 1.1, Table 3.1 and 3.2, Figure 3.1 and 3.2). From 2002 to 2003 water chemistry and algal nutrient limitation studies occurred in the west and east sections of the marsh, and sample sites were located along two transects that extended from the mouths of two of the main channels that connect Delta Marsh to Lake Manitoba: Deep Creek on the west side of the marsh, and Delta Channel on the east side of the marsh and Delta Channel on the east side, to sites further inland (Figure 1.2 and 3.1, Table 3.1). The sample sites along the two channels were located at increasing distances from the inlet of each to investigate any relationship between the nature and magnitude of hydrology, water chemistry, and algal nutrient limitation relative to sources of water from Lake Manitoba.

In 2002, water chemistry and algal nutrient limitation studies were conducted at 11 sample sites connected via surface water flow to Lake Manitoba were included in the study, five each along Deep Creek and Delta Channel, and one in Lake Manitoba, approximately 0.5 km off shore from the Delta Marsh Field Station (Table 3.1 and Figure 3.1). Only one sampling site could be located in Lake Manitoba because the apparatus used for the algal nutrient limitation experiments (floating frames containing Nutrient Diffusing Substrata (NDS; see section 3.4)) had to be monitored on a daily basis from the Field Station. Strong wind events were common on Lake Manitoba, and cottages along with a

Table 3.1 Description of the locations of water chemistry and algal nutrient limitation study sites, and the year(s) of use in Delta Marsh and Lake Manitoba, from 2002- 2005.

Site	Site Code	East or West Side of Marsh[1]	Isolated or connected[2]	UTM[3] Northing	UTM[3] Easting	Distance from Lake Manitoba (km)[4]	Year(s) sampled
(1) Portage Creek South	PCS	East	Connected	5554297	554157	11.5	2002 -2005
(2) Portage Creek North	PCN	East	Connected	5557861	553416	7.7	2002 -2005
(3) Naegele's Island /Simpsons Bay	SIMP	East	Connected	5559266	552981	5.7	2002 -2005
(4) Cadham Bay East	CADE	East	Connected	5558740	551718	3.8	2002 -2005
(5) Delta Channel	DCH	East	Connected	5559084	549088	0.8	2002 -2005
(6) Lake Manitoba	LK	n/a	Connected	5559576	544124	0.0	2002 -2005
(7) Deep Creek	DCRK	West	Connected	5558920	541464	0.9	2002 -2003
(8) Canvasback Bay	CANV	West	Connected	5558059	541646	2.2	2003
(9) Carp Creek	CARP	West	Connected	5556766	541527	3.8	2003
(10) Big Lake South East	BLSE	West	Connected	5557142	540971	3.6	2002-2003
(11) Short Creek	SCRK	West	Connected	5555207	541441	5.6	2002-2003
(12) Long Creek North	LCRK	West	Connected	5556238	541003	5.7	2002-2003
(13) Big Lake North West	BLNW	West	Connected	5559416	539110	8.5	2002-2003
(15) Center Marsh	CENT	East	Isolated	5558462	546276	n/a[7]	2003-2005
(16) Portage Creek Bridge	PCB	East	Connected	5550926	554053	15.1	2004-2005
(17) The Gap	GAP	East	Connected	5560016	552045	4.4	2004-2005
(19) Cadham Bay West	CADW	East	Connected	5558784	549593	1.7	2004-2005
(23) East Blind Channel	EBC	East	Isolated	5558045	545303	n/a[7]	2004-2005
(24) Naegele's Pond	NAEG	East	Isolated	5556976	552642	n/a[7]	2004-2005
(26) Crescent Pond	CRES	East	Isolated	5559374	542323	n/a[7]	2004-2005
(27) Louck's Pothole	LOUC	East	Isolated	5559447	549155	n/a[7]	2004-2005
(28) Richardson's Pothole	RICH	East	Isolated	5555133	551058	n/a[7]	2004-2005

[1] Delta Marsh is separated into the east and west portion of the marsh by the Portage Diversion
[2] Isolated or connected to Lake Manitoba via surface water flow
[3] UTM zone is 14U
[4] Distance from lake via open water

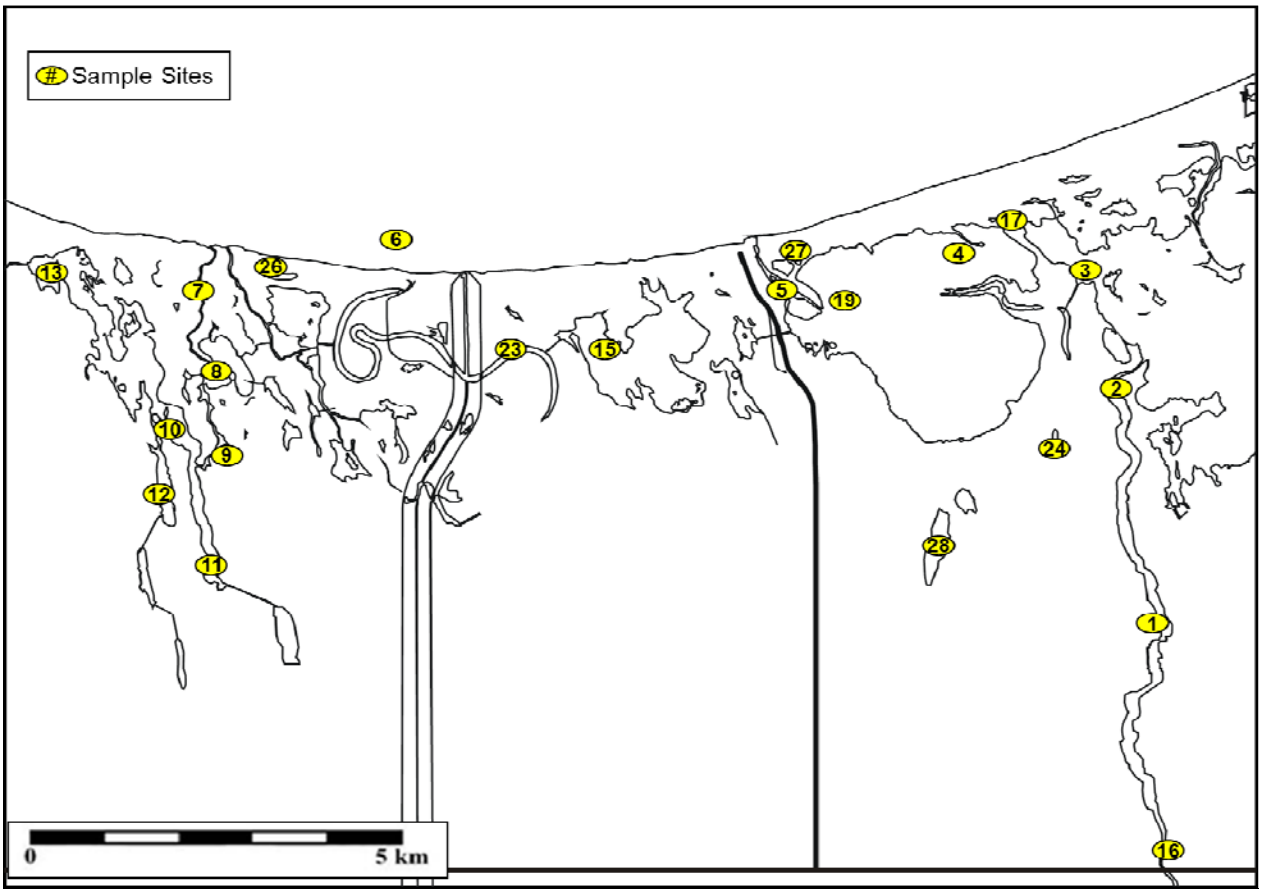

Figure 3.1 Schematic diagram of water chemistry and algal nutrient limitation sample sites in Delta Marsh and Lake Manitoba, Manitoba from 2002 to 2005. See Table 3.1 for corresponding site names and other information. Base map modified from original provided courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

public beach in the area made sampling from other locations in the lake difficult. In the summer of 2003, three additional sample sites were added to the study, to better examine spatial trends in the marsh, for a total of 15 sample sites. Two sample sites were added to the transect on the west side of the marsh (Table 3.1 and Figure 3.1). However, NDS experiments could only be performed at 14 of the 15 sample sites in 2003, as the Long Creek South sample site used in 2003 was too shallow to meet experimental parameters. The water chemistry at the site was monitored for as long as possible in 2003, until the site became too shallow for access.

In 2004, the sample sites along the transect on the west side of the marsh were dropped from the study, since water levels were too low to gain access. As a result in studies in 2004 and 2005 focused on the east side of the marsh, and several new samples sites were added to study, including several isolated sites, to allow for better comparison between the hydrology, chemistry and algal nutrient status of sites connected to, or isolated from, surface water exchange with Lake Manitoba. One of the connected sites added to study in 2004 and 2005, Portage Creek Bridge (PCB), was located at the furthest south extent of Portage Creek (15.1 km from the lake) to examine water chemistry inputs from the upland watershed during spring runoff and periodic large storm water runoff events (Figure 3.1 and Table 3.1). When present, water flow at this site was consistently to the north during the study.

From 2002 to 2005, water levels were monitored at a sub-set of the study sites, (Table 3.2, Figure 3.2 and Figure 3.3). Over the course of the study, the location of some of the water level monitors were changed to accommodate the needs of the study. In 2002 and

2003 the water level monitors were located at study sites located on both the east and west side of Delta Marsh, whereas in 2004 and 2005, the water level monitoring sites were concentrated on the east side of Delta Marsh, as use of sample sites on the west side of the marsh was discontinued due to low water levels. In 2004 and 2005 two isolated sites were also included to examine difference in the hydrology of sites connected to, or isolated from, Lake Manitoba. Due to equipment availability, the potential for vandalism, and rough conditions in the near shore environment on Lake Manitoba, the water level monitors used in the study could not be deployed in Lake Manitoba. In order to be able to compare water levels in the marsh to those in Lake Manitoba, data from one of the two water level monitoring stations on Lake Manitoba maintained by Manitoba Water Stewardship Division, near Westbourne, was used. Table 3.2 summarizes the location and the year(s) of deployment of the water level monitors in Delta Marsh, and the monitoring station in Lake Manitoba from 2002 to 2005. Figure 3.2 shows the deployment locations of the water levels monitors in Delta Marsh from 2002 to 2005, and Figure 3.3 shows the location of the monitoring station in Lake Manitoba from 2002 to 2005.

Table 3.2 Descriptions of the locations and year(s) of deployment of water level monitors at study sites in Delta Marsh and Lake Manitoba, from 2002 to 2005. Site abbreviation codes are listed in brackets following each site.

Site	East or West side of marsh [1]	Isolated or connected[2]	UTM [3] Northing	UTM [3] Easting	Distance from Lake Manitoba (km) [4]	Year(s) Monitored
(1) Big Lake North West (BLNW)	West	Connected	5559370	538785	8.5	2002 – 2003
(2) Short Creek (SCRK)	West	Connected	5555205	541416	5.7	2003
(3) DMFS Canoe dock (CAN)	West	Connected	5559179	544131	3.9	2002-2005
(4) Delta Channel (DCH)	East	Connected	5559084	549088	1.1	2004 – 2005
(5) Cadham Bay (CAD)	East	Connected	5559115	551458	3.6	2004 – 2005
(6) Simpsons Bay West (SIMP)	East	Connected	5559224	552957	5.7	2002 - 2005
(7) Portage Creek South (PCS)	East	Connected	5554288	554164	11.5	2002 – 2005
(8) Center Marsh (CENT)	East	Isolated	5558466	546277	n/a[5]	2004
(9) Richardson's Pothole (RICH)	East	Isolated	5555134	551148	n/a[5]	2005
(10) Lake Manitoba at Westbourne (LKWT)	n/a	n/a	5698836	513649	n/a	2002-2005

[1] Delta Marsh is separated into the east and west portion of the marsh by the Portage Diversion
[2] Isolated or connected to Lake Manitoba via surface water flow
[3] UTM zone is 14U
[4] Distance from lake via open water.

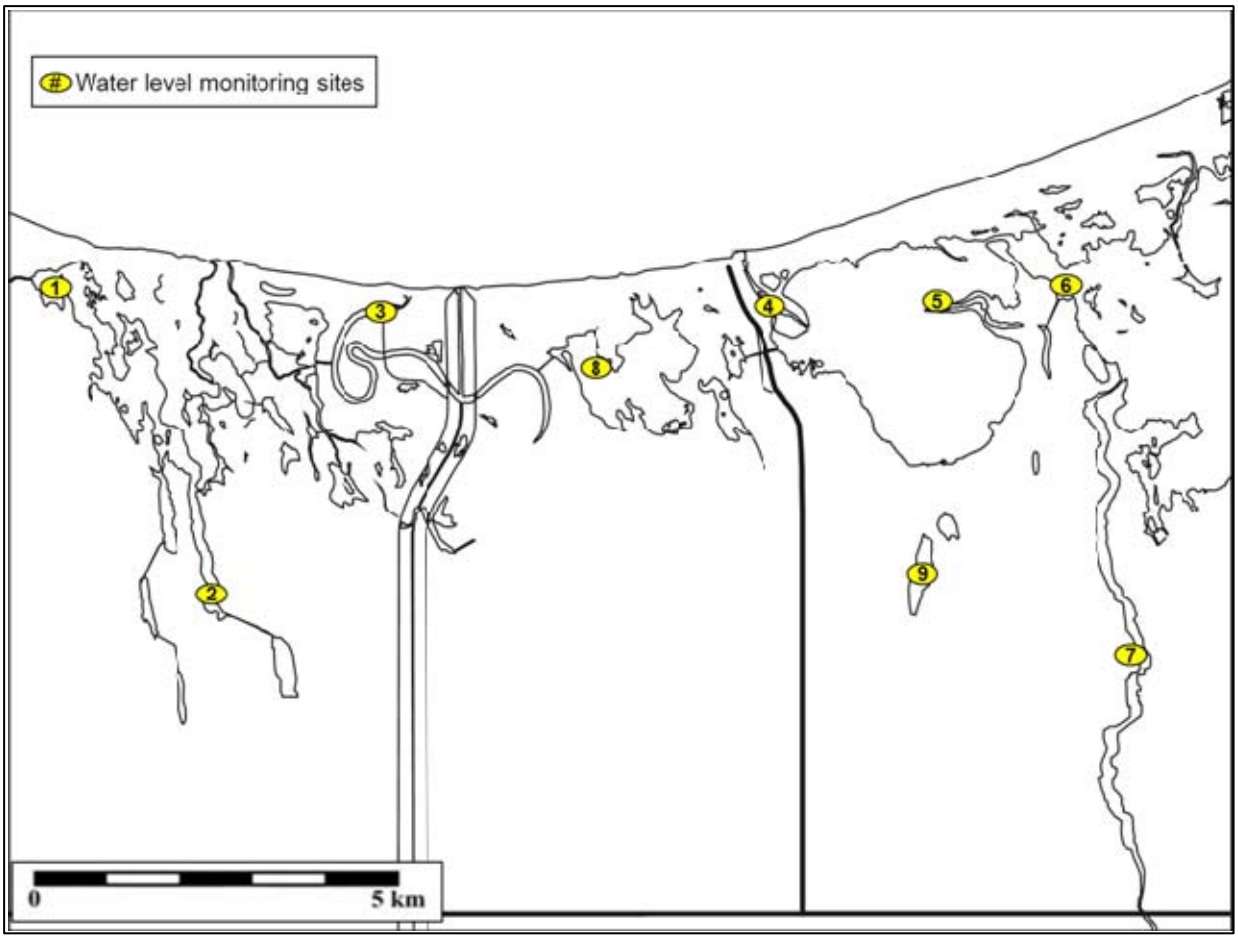

Figure 3.2 Schematic diagram of water level monitoring sites in Delta Marsh, Manitoba from 2002 to 2005. (1) Big Lake North West (BLNW), (2) Short Creek (SCRK), (3) DMFS Canoe dock (CAN), (4) Delta Channel (DCH), (5) Cadham Bay (CAD), (6) Simpsons Bay West (SIMP), (7) Portage Creek South (PCS), (8) Center Marsh (CENT), (9) Richardson's Pothole (RICH). See Table 3.2 for further site information.Base map modified from original provided courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

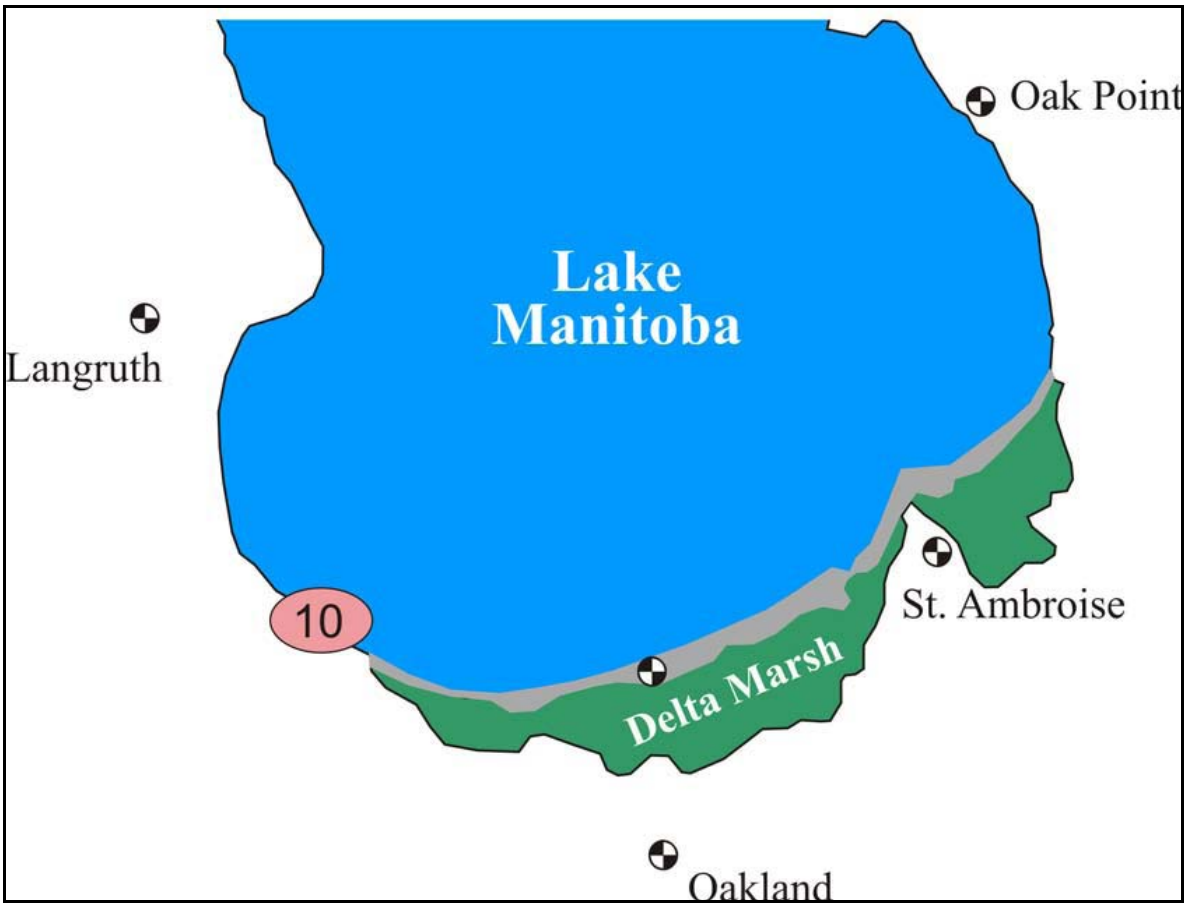

Figure 3.3 Location of water level monitoring site on Lake Manitoba, near Westbourne. (Site 10; LKWT). See Table 3.2 for further site information. Base map modified from original provided courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

Sampling and Analysis

Hydrology

To examine the influence of seiche activity and storm surges in Lake Manitoba on water levels in Delta Marsh, real-time water level monitors (Ecotone; Remote Data Systems Inc.) were deployed in the marsh from the 2002 to 2005 field seasons, to record changes in water levels as affected by wind direction and related flow patterns between the lake and the marsh (Figure 3.4). The water level monitors recorded water levels at sample locations in real-time every hour. In all years the water level recorders were deployed from mid-May to mid-October.

In 2004 and 2005, attempts were also made to examine the velocity and direction of water flow exchange between the lake and marsh during varying wind-induced seiche and storm activities on the lake. Water flow was measured via a hand-held flow probe (Global Water FP101 flow probe, Figure 3.6). The flow probe measured instantaneous flow velocity in ft/sec via a turbo-prop propeller and positive displacement electromagnetic sensor with a digital read out, and range of 0.3-25 feet/sec and accuracy of 0.1 ft/sec.

Measurements were focused in the Delta Channel, which is one of the main channels facilitating water exchange between Lake Manitoba and the east portion of Delta Marsh (Figure 3.5). The channel is relatively easy and safe to access, via a public boat launch and the Delta Channel Bridge (14U N5559553 E548709), which crosses Delta Channel approximately 0.22 km from the outlet of the channel (Figure 3.7). Measurements in the channel were concentrated off the Delta Channel Bridge, as the exact dimensions (1.28 x 1.78 m) of the three box culvert flow-through channels which concentrate flow under the

bridge were known and fixed (i.e., did not change like some areas of the channel due to high instances of sedimentation and resuspension) (Figure 3.8 and 3.9). The net result was that flow velocity (m/sec) could be converted accurately to discharge measurements (m^3/sec). Measurements could also be taken easily and safely directly off the bridge during varying directions and velocities of flow, including when flow was too great in the Delta Channel to safely make measurements across the channel from a canoe.

Figure 3.4 Photograph of the water level monitor deployed at site 7 at Portage Creek South (PCS) in 2005.

Figure 3.5 Aerial photograph looking southeast at Delta Channel, on August 29, 2005.

Figure 3.6 Photograph of the Global Water FP101 flow probe used to take flow measurements at sample sites.

Figure 3.7 Photograph of Delta Channel Bridge on Delta Channel taken in July 2005. Pictures (top left) view west of the bridge looking east at the bridge, (top right) view of the bridge looking southwest, (bottom left) view from the bridge looking north down Delta Channel, and (bottom right) view from Delta Channel looking north at the bridge.

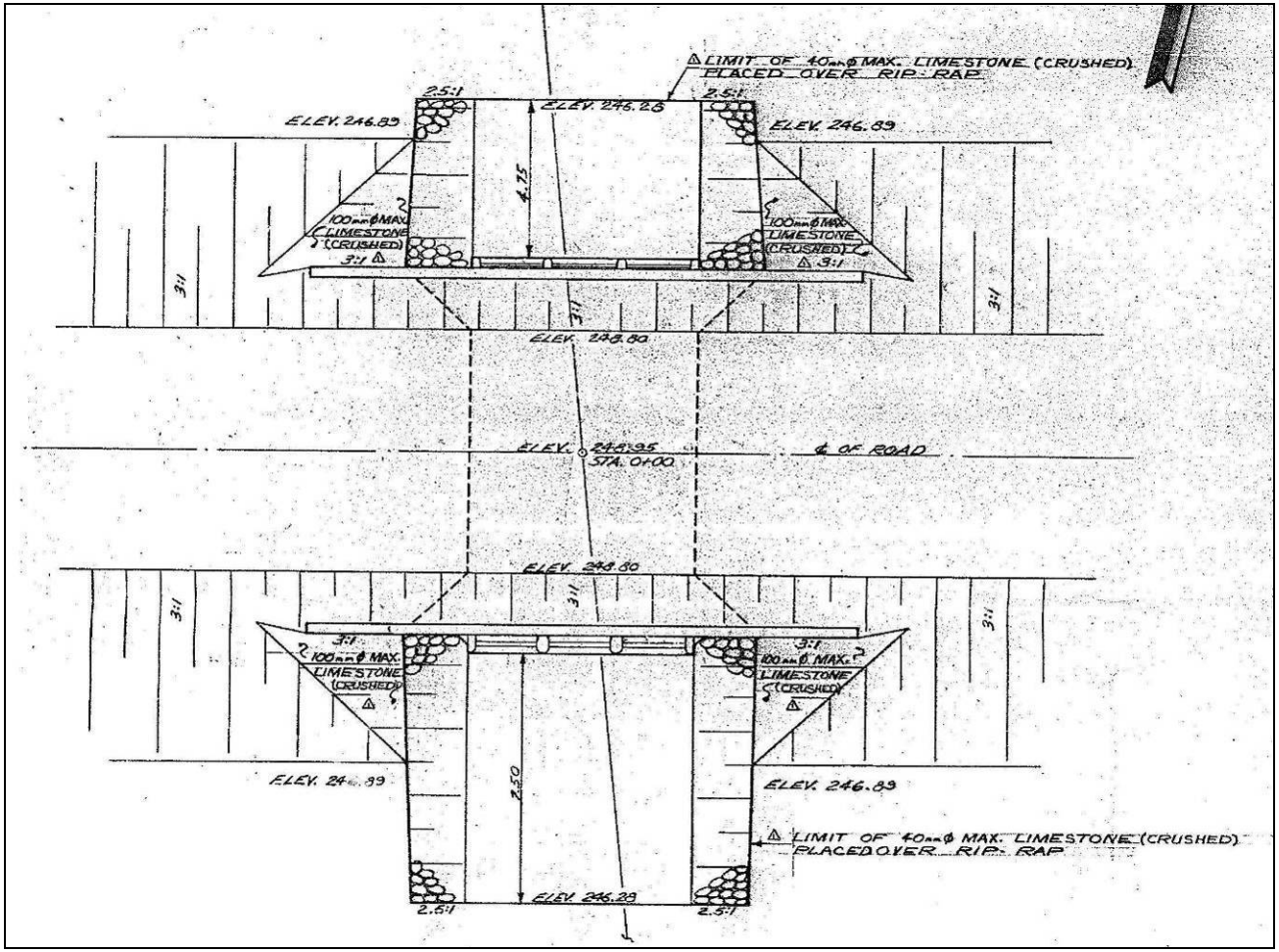

Figure 3.8 Engineered Drawings of the Delta Channel Bridge Control Structure, top view of the bridge. Elevations are in meters above sea level (ASL). Source: Province of Manitoba, Department of Natural Resources, Engineering and Construction Branch. February 1982.

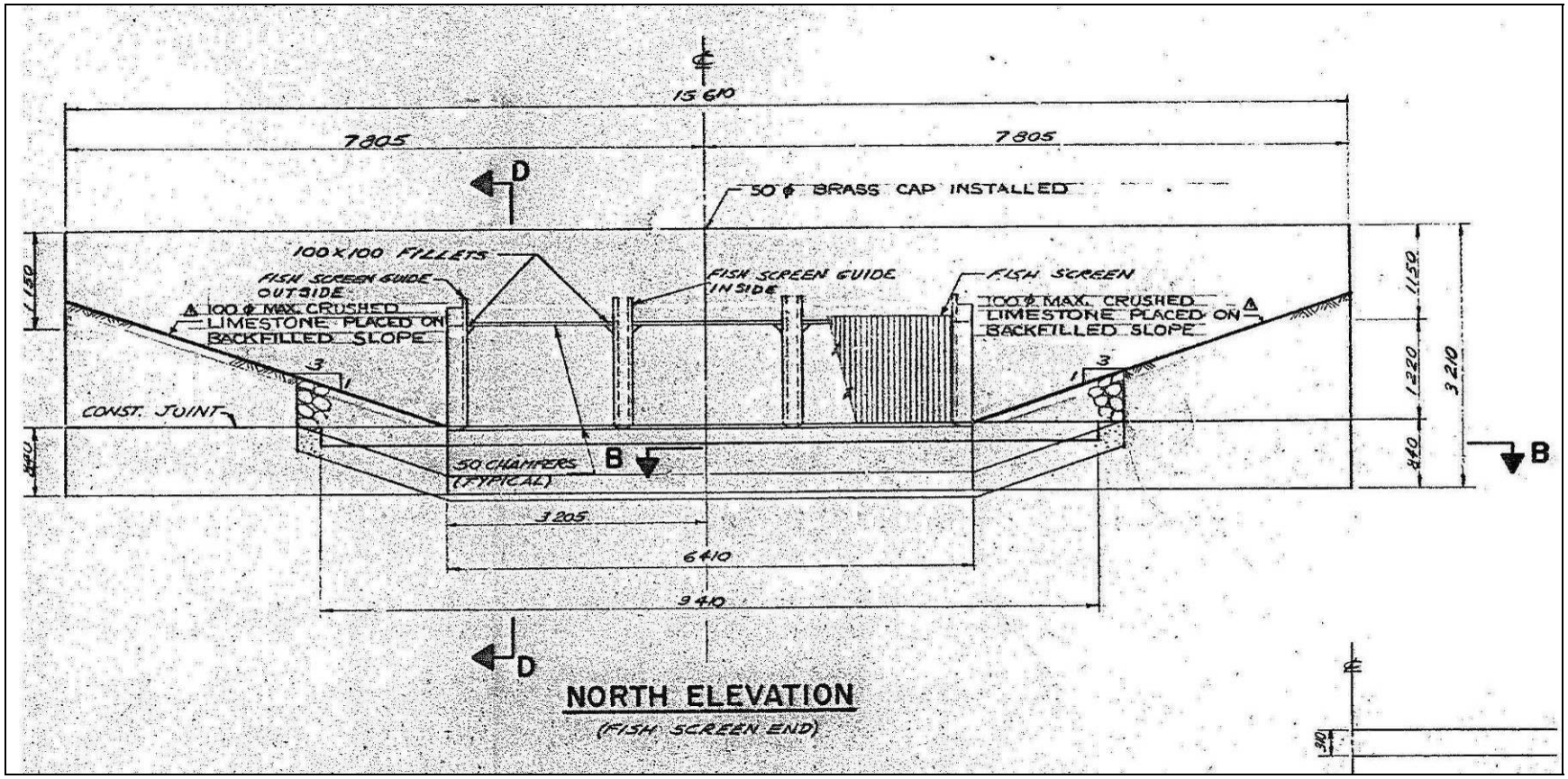

Figure 3.9 Engineered Drawings of the Delta Channel Bridge Control Structure, north side view of the bridge with the three box culvert flow through channels. Elevations are in meters above sea level (ASL). Source: Province of Manitoba, Department of Natural Resources, Engineering and Construction Branch. February 1982.

At the Delta Channel Bridge, flow profiles were taken at the horizontal center of each of the three box culverts, on both the north and south sides of the bridge for a total of six flow profiles. For each of the profiles, vertical measurements were taken at 1 foot depth increments, starting just below the surface of the water to a depth 3 feet (the depth capacity of the flow probe). The average of all the measurements (± standard error) was used for data analysis. During 2004 and 2005 detailed flow velocity profiles in Delta Channel were taken at least four times per week, from the Delta Channel Bridge. Flow profiles were also measured at three other sites in the channel once a week during water sampling events (Figure 3.10). In order for the flow velocity measurements (m/sec), taken in the channel, to be converted to discharge (m^3/sec), bathymetric measurements were taken at ten transect sites along the length Delta Channel (Figure 3.11). At each transect site depth measurements were taken in approximately 1 m increments across the width of the channel from the east shoreline to the west shoreline. Effort was made to make the ten transects evenly spaced along the entire length of the channel from south to north. The GPS coordinates of each transect were recorded using a hand-held GPS unit. Surface area of the channel was determined using Geographic Information Systems (GIS) and GPS coordinate corrected satellite maps of the marsh. Depth profiles were also conducted in Cadham Bay (Figure 3.13) in 2005, so that the water volume change in the Bay could be calculated based on changes in water levels. Depth profiles in Cadham Bay were conducted similarly to those in Delta Channel, with the exception that measurements were taken from an airboat, using a rope with a weight on the end, which was marked off with depth markers, so that measurements could be taken quickly and easily along several depth transects in the bay.

In 2005, in order to gather data on flow exchange in the other channels connecting Lake Manitoba and Delta Marsh depth profile and flow measurements were conducted on Deep Creek and Cram Creek, that connect the west section of marsh to the lake (Figure 1.3). Due to the isolated location of the channels, travel to the sites, as well as measurements at the sites were conducted from canoes. Further, to also obtain a better understanding of water flow conditions within the marsh, depth profiles and flow measurements were conducted at the main water sampling sites in east section of the marsh, once a week during water sampling events. Sites examined included the Gap, Portage Creek North, Portage Creek mid, Portage Creek South, Portage Creek Bridge (Figure 3.12). However as a result of required travel time between sites, on many occasions not all sample sites could be measured in one day, which precluded comparison between multiple sites under similar flow conditions. Further some of the measurements were fraught by low flow conditions and were voided, and on other occasions unfavorable wind and flow conditions precluded access to sites via canoe. As a result the data gathered, from the sites within marsh, as well as in Deep Creek and Cram Creek, was not of sufficient quantity to warrant analysis. Depth profile data, as well as flow data, although not used directly in this study, are presented in Appendix A.

Figure 3.10 Aerial infrared photograph of Delta Channel in August 2003, with flow profile sites. Base map modified from original provided courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

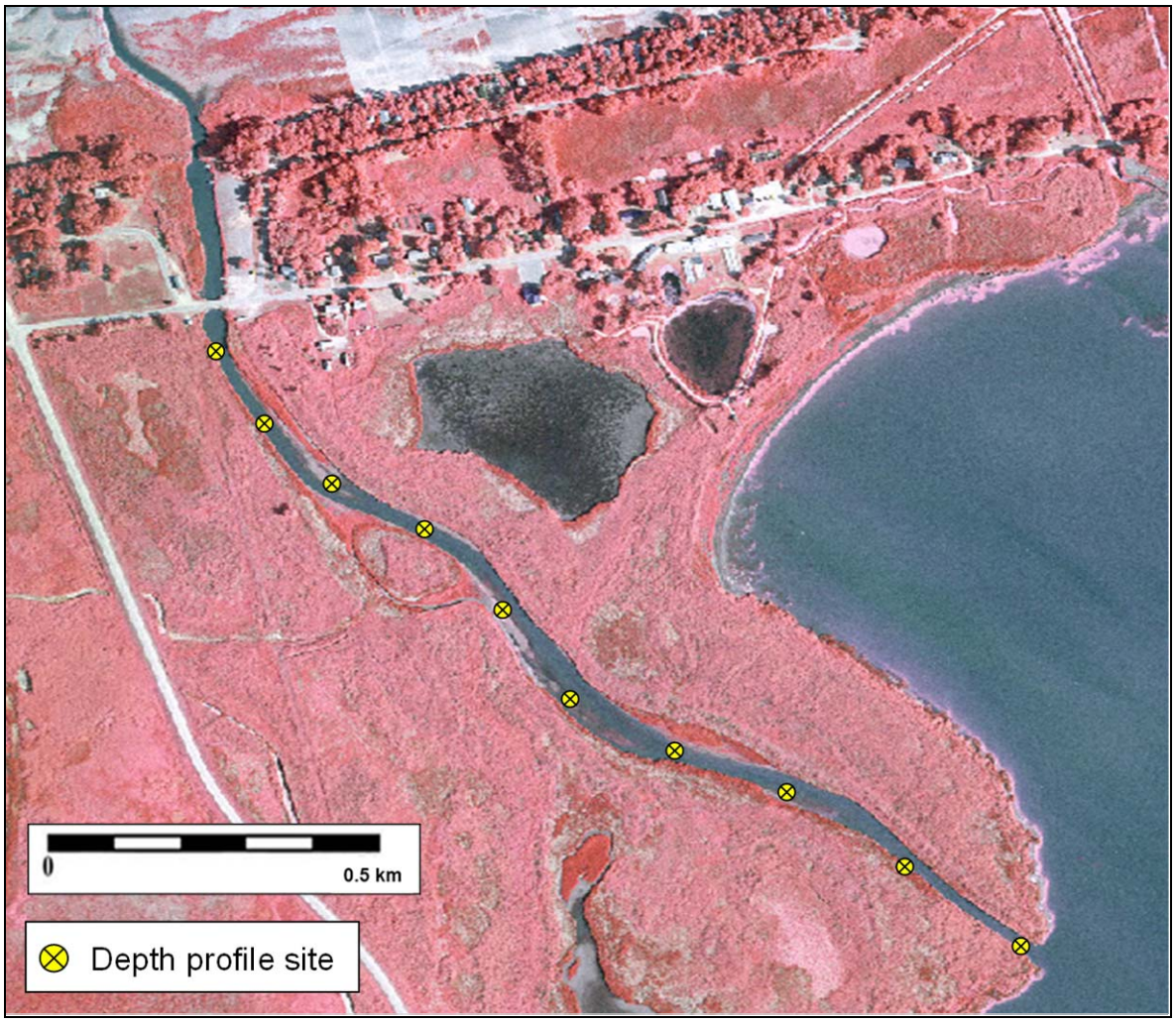

Figure 3.11 Aerial infrared photograph of Delta Channel in August 2003, with depth profile sites. Base map modified from original provided courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

Figure 3.12 Aerial infrared photograph of Delta Marsh in August 2003, with depth profile sites. (1) Deep Creek, (2) Cram Creek, (3) Delta Channel, (4) The Gap, (5) Portage Creek (north), (6) Portage Creek (mid), (7) Portage Creek (south), and (8) Portage Creek Bridge. Base map modified from original provided courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

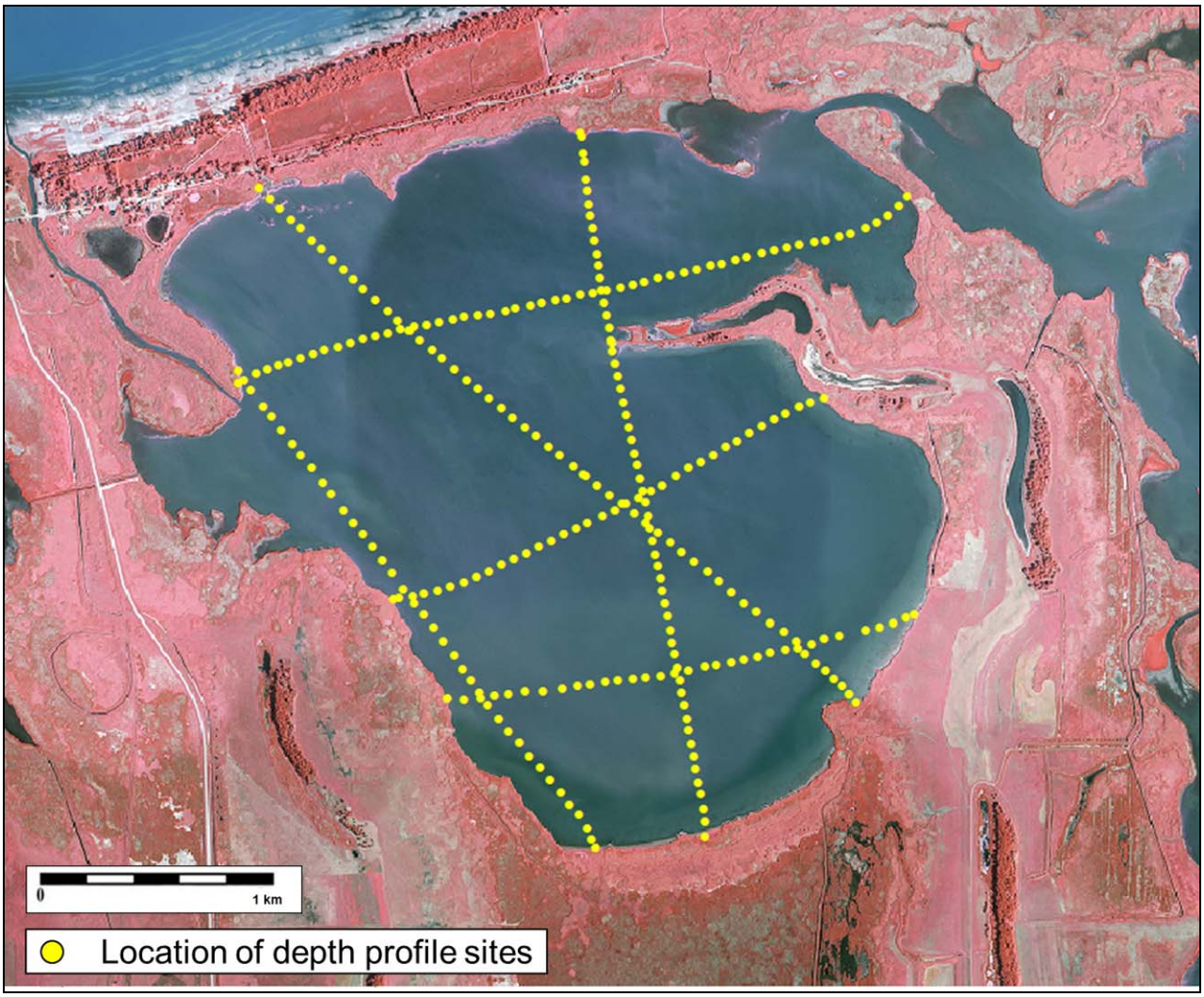

Figure 3.13 Aerial infrared photograph of Cadham Bay in August 2003, with approximate depth profile sites in 2005. Base map modified from original provided courtesy of Dr. Gordon Goldsborough, Department of Biological Sciences, University of Manitoba.

Water chemistry and clarity

Chemical and physical water quality parameters were examined at sample sites during the summers of 2002 to 2005. In 2002 this occurred every three weeks from May 22 to August 13, and from 2003 to 2005 water sampling frequency was increased to weekly. In 2003 sites were examined from May 12 to August 18, in 2004 from May 18 to August 30, and in 2005 from May 16 to August 16.

Physical and chemical water quality parameters examined included conductivity (COND), turbidity (TURB), pH, alkalinity (ALK), ammonia-N (NH_3-N), nitrate-N ($N0_3$-N), total nitrogen (TN), total reactive orthophosphate (PO_4-P), total phosphorus (TP), total suspended solids (TSS), dissolved organic carbon (DOC), chloride (Cl^-), sulfate ($SO4^-$), soluble reactive silicon (SRS), Oxygen-18 (^{18}O) and deuterium (^2H, D) stable isotopes, and total phytoplankton chlorophyll *a* (CHL).

A YSI Model 30 hand-held meter was used to measure specific conductance (COND ± 5% μS/cm) in the field. Integrated water column samples were collected using a hollow clear plastic cylinder (10 cm inner diameter, 100 cm length; submerged in the water column without disturbing the sediment surface and corked and brought up to the surface) and each sample poured into a 1 L opaque polypropylene bottle for processing in the laboratory. The sample bottles had been acid washed, and triple rinsed with site marsh water prior to sample collection. Samples were kept in the dark at approximately 5°C for transport to the laboratory.

In 2005, additional 50 ml surface water samples were collected for Oxygen-18 (^{18}O) and deuterium (^2H, D) isotope analysis from sampling sites in the east portion of the marsh

once per month from May to August, 2005. Samples were collected in 50 ml opaque plastic vials, capped underwater, with no headspace in the vials. Water samples were analyzed by at the Environmental Isotopes Laboratory, at the University of Waterloo, in Waterloo, Ontario. Oxygen-18 and deuterium were determined by mass spectrometry using a MM 903 triple Faraday bucket collector mass spectrometer.

All water samples collected for analysis were filtered, preserved, and otherwise prepared for analysis within 8 hours of collection. Analyses of samples followed recommended holding times, included the use of de-ionized water "blanks" as recommended, and followed appropriate laboratory quality control and quality assurance protocols (Stainton et al, 1977, APHA 1995). Alkalinity (ALK; mg $CaCO_3$/L) was determined by titration with 0.02 N hydrochloric acid to a clear end point (bromocresol green-methyl red indicator solution). Turbidity (NTU; TURB) was determined using a Hach turbidimeter (model 2100A). NH_3-N was determine by the hypochlorite method (detection limit 0.01 mg/L) and PO_4-P by the acid molybdate method (detection 0.01 mg/L), which measures total reactive orthophosphate P, by colorimetric analysis using an Ultrospec 400 UV/visible light spectrophotometer (Stainton et al. 1977, APHA 1998). $N0_3$+$N0_2$-N was measured by UV spectrophotometry and ion chromatography, using a Dionex DX 500 ion chromatograph (Dionex®) with a PRP-X100 column (dimensions: 150 x 4.1 mm, 10 μm). TP and TN were measured using Hach reagents and protocols (Hach Company 1997), with TN measured by the persulfate digestion method (detection limit 0.5 mg/L) and TP by the PosVer 3 with acid persulfate digestion method (detection limit 0.06 mg/L) followed by colorimetric analysis using a Ultrospec 400 UV/visible light spectrophotometer (Stainton et al. 1977, APHA 1995). All nutrients measured are reported

in milligrams per liter (mg/L). Values for ammonia-N and nitrate+nitrite-N were summed to provide dissolved inorganic nitrogen (DIN-N) values. Nitrate+Nitrite-N is here after referred to as Nitrate-N (NO_3-N). Dissolved organic carbon (DOC) was determined on filtered samples (Whatman GF/C) using scanning UV spectroscopy (Badiou et al. *in preparation*)). Total Suspended Solids (TSS) were determined by filtering subsamples (100 to 300 ml) through pre-weighted Whatman GF/C glass microfibre filters under vacuum. Filters were then dried in an oven at 100°C for a minimum of 48 hours, then weighed again, and incinerated in a muffle furnace at 550 °C for one hour and weighed again. TSS (mg/L) were calculated as the weight of the filters after drying at 100 °C, minus the tare weight of the filters, OSS were calculated as the weight of the filters after drying at 100 °C minus the weight after combustion at 550 °C, and inorganic suspended solids (ISS) as the weight after combustion minus the tare weight.

From 2003 to 2005, Cl^- and SO_4^- were also measured by the same ion chromatography method used for $N0_3$-N determination. In 2005, SRS was also examined using an acid molybdate colorimetric an Ultrospec 400 UV/visible light spectrophotometer (Stainton et al. 1977).

Phytoplankton biomass (chlorophyll *a*)

Total chlorophyll *a* (µg/L) was used as a measure of phytoplankton biomass. A subsample (100 to 300) ml of the original integrated water column samples collected for water chemistry analysis was used to determine phytoplankton biomass. The subsamples were filtered on to Whatman GF/C glass microfibre filters under vacuum, and neutralized using 2-3 drops of saturated magnesium carbonate solution. Filters were frozen for a minimum of 24 hours to lyse algal cell membranes, then placed in 5 ml 90% methanol and stored in the dark for 24 hours to extract chlorophyll pigments. Chlorophyll *a* and pheophytin concentrations were measured spectrophotometrically (Ultrospec 400 UV/visible light spectrophotometer) at 665 and 750 nm before and after acidification with HCl, then calculated using formulae of Marker et al. (1980). Total chlorophyll a (CHL; detection limit 0.1 µg/L) was calculated as the sum of chlorophyll *a* and pheophytin.

Algal nutrient limitation experiments

A micro nutrient-diffusing substratum (NDS) method (Gibeau and Miller 1989) was used to detect algal nutrient limitation at the study sites in Delta Marsh and Lake Manitoba. Each apparatus consisted of 50 ml polypropylene plastic vials (2.7 cm in diameter; Fisher Scientific cat. no.14-375-150) which were filled with 2% (w/v) granulated agar (FisherBiotech cat. no. BP1423-2) supplemented with nutrients, and sealed with a 2.6 cm diameter porous fused silica disk (Leco Corporation cat. no. 528-042) The disks were only slight smaller in diameter than the vials (0.1 cm) so they fitted snuggly inside the upper inner portion of the vials (Figure 3.14). The silica disks had been autoclaved for 20 minutes at 120 °C prior to use. Vials were filled with 2% (w/v) agar supplemented with one of four nutrient treatments, a control (C = agar only), nitrogen treatment (N = 0.5 mol/L N (7g/L NO_3-N) as $NaNO_3$), P treatment (P = 0.05 mol/L P (1.5 g/L PO_4-P) as K_2HPO_4), and a combination nitrogen plus P treatment (N+P = 0.5 mol/L and 0.05 mol/L P (1.5 g/L PO_4-P and 7 g/L NO_3-N)). All agar/nutrient mixtures were autoclaved for 30 minutes at 120 °C to ensure sterility. Once each vial had received the nutrient supplemented agar and it was capped with silica disk, it was inverted, allowing the agar mixture to solidify in contact with the silica disk. Finished vials were labeled according treatment (Figure 3.15). Four replicates of each of the four treatments were used (n=16). The NDS vials were attached to anchored floating PVC frames (91 cm long x 61 cm wide, with NDS vials spaced approximately 15 cm apart; Figure 3.16). The vials were inserted into 2.7 cm holes drilled into the PVC strips, and elastic bands were used to hold the NDS in the frames. Foam floats were attached to the frames for floatation. NDS were oriented on the frames and anchored at the sites in such a fashion that the silica disk end of each NDS was always

facing downstream, to reduce the chance of any suspended sediments in the water column accumulating on the NDS, and to reduce current drag on algal biomass growing on the NDS. To ensure the frames were able to move with changes in the current direction, circular foam floats were attached to the wooden stakes that anchored the frames, allowing the frames to move freely around the stakes (Figure 3.17) with changing wind and current direction. The NDS on the frames were suspended at a constant depth of 20 cm below the water surface, to ensure NDS were submerged at all times, and that they were receiving equal amounts of light.

In 2002, 2003 and 2005, NDS systems were incubated at sites for four consecutive three-week periods from mid-May to mid-August, and in 2004 NDS were deployed for five consecutive three-week periods from mid-May to late-August at sample sites. Incubation periods and periodicity were to permit algal growth to occur in response to specific NDS treatments and to observe seasonal changes in nutrient requirements that might be affected by temporal change in ambient nutrient conditions. After each three-week incubation period, the NDS frames were retrieved from each site, the vials were removed, and the frits removed from the vials by gently squeezing the mouth of the vials with needle nose pliers. The frits were then placed in pre-labeled 50 ml amber glass jars. Samples were then returned to the lab for analysis.

Figure 3.14 Preparation of NDS vials in laboratory. Silica frits are being added to vials containing warm agar solution.

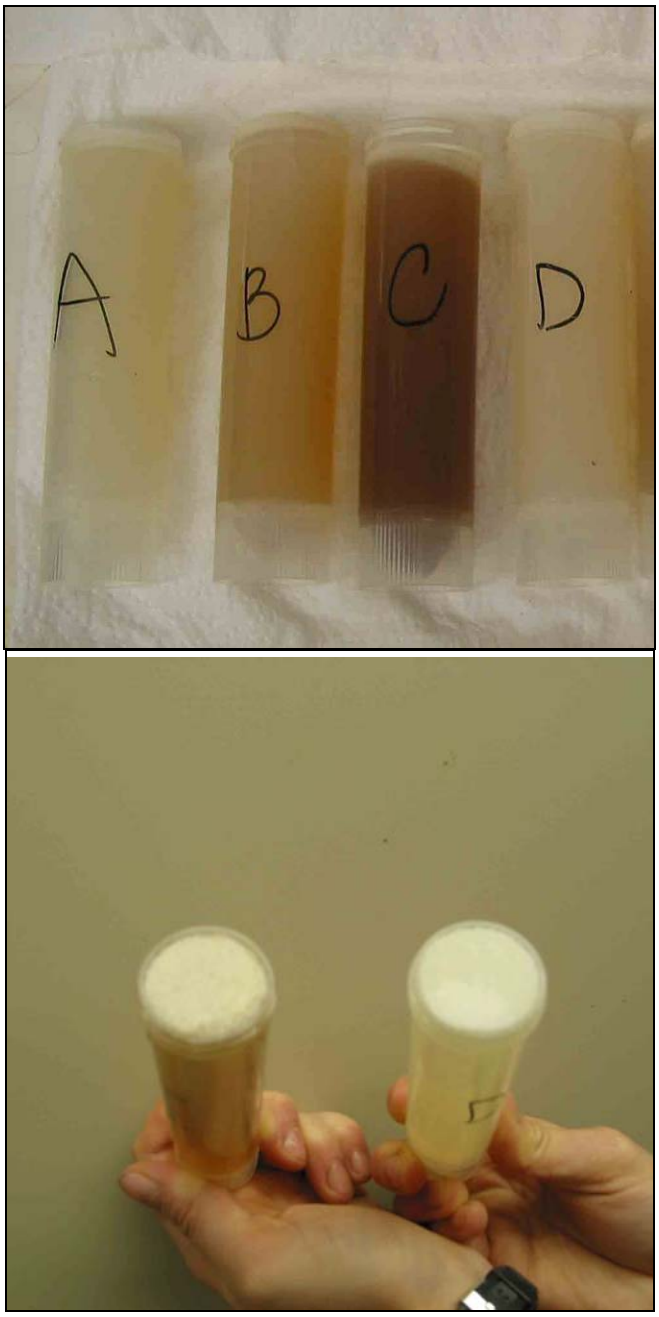

Figure 3.15 Picture of finished NDS vials ready for deployment. Top photograph: from left to right (A) control treatment, (B) N treatment, (C) P treatment, (D) N+P treatment; Bottom photograph: view of the porous silica frits used to cap the NDS vials.

Figure 3.16 Picture of NDS vials being attached to a PVC frame.

Figure 3.17 Picture of frame containing NDS anchored at sample site in Canvasback Bay in 2002. A circular float attached the frame to the stake, allowing the frame to move freely around the stake with changes in current direction.

Algal biomass on the frits was assessed as chlorophyll *a* content. In 2002, algal growth was removed from the frits (with a hard bristle tooth brush) prior to analysis, and from 2003 to 2005 algal growth on the frits was measured directly, as it was determined to be more efficient method to extract chlorophyll *a* from the samples.

For the method involving the removal of the algal growth the frits, once removed the algal growth was mixed to 300 ml of distilled de-ionized water, resulting in an algal slurry. 100 ml subsamples of the algal slurry were then filtered through glass fiber filters (Whatman GF/C) under mild vacuum. The filters were frozen for a minimum of 24 hours to lyse algal cell membranes, then placed in 5 ml 90% methanol and stored in the dark for 24 hours to extract chlorophyll pigments. Chlorophyll *a* and pheophytin concentrations were measured spectrophotometrically at 665 and 750 nm before and after acidification with HCl, then calculated using formulae of Marker et al. (1980) and summed to obtain total chlorophyll *a,* which was divided by the surface area of the frit to obtain total chlorophyll a per cm ($\mu g/cm^2$) values.

To analyze the algal biomass on the frits directly, the frits were placed in 50 ml amber jars, were frozen for a minimum of 24 hours, and then 10 ml of 90% methanol was added and they were stored in the dark for 24 hours. Chlorophyll *a* and pheophytin concentrations were measured by spectrophotometrically as in 2002, and summed to obtain total chlorophyll a per cm ($\mu g/cm^2$) values. Data comparing the two methods (scrubbing versus direct measure) are presented in Appendix B.

The NDS bioassays were also examined for diffusion rates of N and P. Since the diffusion rates could not be directly measured from the NDS in the field, laboratory experiments as described by Fairchild et al. (1985) and Gibeau and Miller (1989) were conducted to determine approximate diffusion rates of N and P from each of the NDS treatments (C, N, P, and N+P). In 2002, new NDS vials were immersed in beakers of 1L of continuously stirred distilled de-ionized water (one beaker per treatment) for 30 days. Water from each of the beakers was changed daily and samples were taken and analyzed for orthophosphate (PO_4-P) and nitrate+nitrite-N (NO_3+NO_2-N) every one to two days using standard methods as described above for water samples (Stainton *et al.* 1977, APHA 1992).

To further estimate diffusion rates in the field, and ensure a positive concentration gradient (from NDS to water column) persisted at sample sites over a three-week incubation period, concentrations of P (PO_4-P) and N (NO_3-N) remaining in the NDS following an incubation period from July 8 to July 29, 2003 was examined at three sample sites were examined. One vial of each treatment was randomly chosen from three sites (also randomly chosen) and 10 ml of agar was removed from three different sections of the agar in the vials (top section of agar next to the frit, center section, and the end section). The agar was melted down and diluted into a volume of 200 ml with distilled ionized water, and subsamples were analyzed for concentrations of TN and TP using standard techniques described above.

Other environmental data

Wind direction and velocity, and precipitation were recorded daily at the Environment Canada Meteorological Station located at the Delta Marsh Field Station (50°10'57.500" N, 98°22'56.100" W; ID: 5040764) from 2002 to 2005. Wind data were used to determine the predominant wind direction during the study, and the resultant direction of any wind-induced seiche set-up on Lake Manitoba. A hand-held Kestrel 3000 pocket weather station was also used in the field in from 2003 to 2005 to measure wind speed and air temperature at samples sites, once a week at the times of water sampling. Measurements were taken near the stake that anchored the NDS apparatus at each sample site.

Data Analysis

Hydrology

The method of Trebitz (2006) was used to calculate the magnitude in the range and the fluctuation intensity (combines magnitude and frequency) of daily water level changes from the water level time series data gathered at samples sites in the marsh and lake. This approach calculates the magnitude in the range and the fluctuation intensity of daily water level changes in a form that facilitates comparison between locations and emphasizes site-to-site and temporal variations instead of synoptic trends. This method also avoids the need to analyze the data with spectral analysis (i.e. Fourier transformation) or complex modeling, which would be exceptionally problematic given the complexity and size of the data gathered for this study (about 14,000 records per monitoring site x 10 sites). Spatial and temporal (seasonal and inter-annual) variations in daily water level fluctuations were assessed using hourly data for all the monitoring sites from May through October 2002 to 2005. All the data were examined for erroneous data and missing values prior to analysis.

The magnitude of range in daily water level changes was calculated by determining the range in daily time series for each site, i.e. the difference between the maximum and minimum water levels for each day. The distribution of daily water level ranges were found to be skewed to the right, possibly due to larger wind events, so the data were log-transformed to approximate normality. The daily (24 hour) mean (± standard error) of the log-transformed daily range data were used in statistical analysis of changes in water level range over time and between sites (Trebitz 2006).

$$\text{magnitude of range (cm)} = \sum_{t=0}^{t=1\ \text{day}} |\Delta\, z(t)|$$

z(t) is the difference in water level between two recording times.

The intensity (half the sum of water level increment changes per day) of daily water fluctuations was calculated as the sum of daily water level increments determined by a lagged version of the water level time series (each value replaced by the difference between it and the previous value), to absolute values, and then calculating the sum of those values per day. The sum is then multiplied by ½ and log-transformed. The daily mean (± standard error) of the log-transformed values were used in statistical analysis (Trebitz 2006).

$$\text{intensity (cm)} = \tfrac{1}{2} \sum_{t=0}^{t=1\ \text{day}} |\Delta z(t)|$$

Flow velocity data gathered in Delta Channel were normally distributed, and values were regressed (linear regression) against wind speed, to examine relationships between wind speed and direction and seiche-induced water exchange between the marsh and lake.

Short-term water level fluctuations in the marsh that result from water exchange with the lake can also drive flushing and residence time (i.e. the amount of time (days) it takes for a known volume of water to be replaced with the same volume of new water). Based on the average water flow velocities through Delta Channel into Cadham Bay, and the average daily intensity of water level range on Cadham Bay data gathered during this study, residence time for Cadham Bay was estimated using two calculations (1) the average volume of water in Cadham Bay (m^3), divided by the velocity of water flow (m^3/day) in Delta Channel to Cadham Bay, which equals the average number of days it takes water

flow through Delta Channel (at given velocity) to flush Cadham Bay (equation A), and (2) the average depth ($z_{mean;}$ cm) of Cadham Bay multiplied by 1 day, and divided by the average intensity of water level fluctuations(cm) on Cadham Bay (calculated above), which equals the average number of days it takes lake seiches to completely flush Cadham Bay (equation B; Trebitz 2006, Wells and Sealock 2009)). The assumption of both calculations is that the incoming lake water completely mixes with water in Cadham Bay, but this is not known with certainty.

A) $$\text{Residence time (days)} = \frac{\text{volume } (m^3)}{\text{flow velocity } (m^3/\text{day})}$$

B) $$\text{Residence time (days)} = \frac{1 \text{ day} \times z_{mean} \text{ (cm)}}{1/2 \sum |\Delta Z| \text{ (cm)}}$$

Water Quality

All environmental and chemistry data, except for pH which is already on a log scale, were converted to approximate normality using $\log_{10}(x+1)$ transformation prior to analysis. Critical p-values ≤ 0.05 were used in all analyses.

Linear regression analysis was used to examine spatial relationships between the water chemistry of marsh sample sites and their distance from the lake (via surface) water. Analysis of Variance (ANOVA) and the Tukey HSD test were used to compare water quality parameters between east/west marsh sections of the marsh, and the connection/isolation of marsh sites from the lake.

Principal Components Analysis (PCA) was used as a multivariate model to confirm temporal and spatial relationships identified by correlations and aggregate marsh water quality characteristics of sample sites between different sampling years (2002-2005), distance from the lake, east compared to west marsh sites, and connected compared to isolated marsh sites. For each PCA biplot, the PCA scores along the two primary axes for the marsh sample sites were then regressed separately against their distance from Lake Manitoba (by water distance from the lake) and water depth, as well as compared to sampling year via ANOVA to determine relationships existed between the water chemistry of sample sites and their distance from the lake, sampling year, and/or water depth.

Naturally occurring ions and stable isotopes may be used as tracers to determine different sources of water and mixing in aquatic ecosystem. The uses of naturally occurring chloride (Cl^-) concentrations and oxygen-18 (^{18}O) and deuterium (^2H, D) stable isotope were chosen as conservative hydrological tracers for this study. Cl^- represents a useful tracer, since it is relatively inert and not used by biota to a great degree and it has also been

used an effective hydrologic tracer in other wetlands in the northern prairie region of North America (Kadlec and Knight 1996; Hayashi et al. 1999; Waiser 2006), as well as the Laurentian Great Lakes Wetlands (Morrice et al. 2011). In the case of stable isotopes, individual water bodies can acquire isotope signatures reflecting their mean residence time and seasonal timing of water inputs, which can also be affected by mixing with other water sources, for example in this case the mixing of lake waters in the marsh (Gibson et al. 2002; Clay et al. 2004; Gibson et al. 2005). Evaporation also results in isotope enrichment. The oxygen-18 and deuterium isotopic signatures of adjoining water bodies receiving similar inputs of water will typically be correlated in a linear fashion (Gibson et al. 2005).

To validate the use of naturally occurring chloride concentrations as a surrogate for measuring changes in hydrological characteristics (i.e. water level change and dilution) in Delta Marsh, site water levels were regressed against chloride ion concentration at all connected samples sites from 2003 to 2005. Site water levels were used as a surrogate for volumes, since volumes could not be accurately determined at all samples sites due to the large size and complex morphology and bathymetry of study area. Assuming lake-ward decreases in chloride between sites were solely due to dilution, gradients in chloride concentration between lake and wetland waters were used to determine the extent of intrusion and dilution of lake waters in the wetland. For each site in the marsh a paired t-test was used to test for average annual (yearly) differences between wetland and lake (wetland Cl^- vs. lake Cl^-) to estimate the extent of lake water mixing in the marsh.

Similarly, to validate the use of naturally occurring oxygen-18 and deuterium to measure temporal and spatial changes in water mixing in the marsh, oxygen-18 and

deuterium isotopic signatures at sample sites were plotted and regressed against date and distance from the lake.

Algal Nutrient Limitation

Periphyton biomass response data from the nutrient-diffusing assays were log-transformed $\log_{10}(x+1)$ where necessary to stabilize the variance and to approximate a normal distribution of the errors prior to analysis. Analysis of variance (ANOVA) was performed to determine if periphyton accrual was significantly affected by nutrient treatments using SYSTAT 8.0 for Windows (SPSS 1998). If ANOVA indicated a significant effect ($P \leq 0.05$) the Tukey-Kramer pair-wise comparisons test was used to identify significant differences among periphyton accrual on treatment means. Four possible nutrient limitation conditions were defined (1) No nutrient limitation (n.s), where no treatment differed significantly from the control or other treatments, (2) N limitation, where there was a significant response to the N treatment alone, or in combination with the N+P treatment over the control (3) P limitation, where there was significant response to the P treatment alone, or in combination with the N+P treatment over the control, and (4) N+P co-limitation, where there was a significantly greater response to the N+P treatment over the control and other treatments, or all the nutrients treatments together over the control. However a response to the N+P was not considered co-limitation if one of the individual nutrient treatments (N or P) also had a significant response. In these cases, the individual nutrient treatment stimulating growth was considered to be the limiting nutrient.

Where nutrient limitation occurred (N, P, or N+P) a method similar to Scott et al. (2005) was used to calculate the nutrient limitation status index (NLSI). The mean biomass

of periphyton (chlorophyll *a*) on the control treatment was divided by the mean biomass of periphyton on the limiting nutrient treatment (N, P or N+P) at each sample site and then subtracted from 1.

NLSI = 1- (biomass on control / biomass on treatment)

Assuming that growth on the controls represents biomass under natural N and P nutrient concentrations, and accumulation on the limiting nutrient treatment represented the maximum biomass potential under optimal nutrient condition, the NLSI ratio represents the magnitude of nutrient limitation. The resulting NLSI values ranges from 0.0 to 1.0, with a value of 0.0 indicating periphyton biomass is at 100% relative to its N and P nutrient requirements and there is no nutrient limitation, whereas values near 1.0 indicate strong nutrient limitation. ANOVA was used to determine differences in NLSI values between the sites and dates.

Periphyton biomass (chlorophyll *a*) on the control treatments during each incubation period at each site were regressed against water N and P column concentrations and molar ratios to determine if relationships existed between periphyton biomass (assumed to be growth in response to natural conditions) and spatial and temporal trends in N and P nutrient concentrations and molar ratios in the marsh. When nutrient limitation occurred, the NLSI index for that treatment was regressed against water column concentrations and molar ratios of N and P to determine if relationships existed between periphyton biomass in response to the nutrient treatment and spatial and temporal trends in N and P nutrient concentrations and molar ratios.

Univariate statistical analysis was performed using JMP 10.0.1 by SAS Institute Inc. software (SAS Institute 2012). Multivariate statistical analysis was performed using CANOCO for Windows Version 4.02 (GLW-CPRO 1999) and Syn-tax 2000 (Dr. J. Podani 2000).

Chapter 4: Results – Hydrology

Introduction

Hydrology is a key factor controlling wetland structure and function (Gosselink and Turner 1978, Bedford 1992, Mitsch and Gosselink 2000a & b, Grosshans 2001; Wilcox et al. 2007, Wilcox 2012). Due to their small volume and shallow waters, wetlands are dynamic environments in which small hydrological changes can result in significant changes to many physical, biological and chemical characteristics. As a result, the hydrological regime is one of the most important characteristics in wetland classification schemes (National Wetlands Working Group 1988, Goldsborough and Robinson 1996, Warner and Rubec 1997, Mitsch and Gosselink 2000a, van der Valk 2006).

In contrast to most prairie marshes, water levels in Delta Marsh have traditionally been thought to be influenced to a great degree by adjoining Lake Manitoba; however, no detailed data have been gathered to quantify this influence. It is thought that water exchange between coastal freshwater wetlands and their adjoining lakes acts similarly to tidally-driven water fluctuations in coastal marine estuaries (Mitsch and Gosselink 2000a). The majority of hydrological studies of freshwater lacustrine coastal wetlands to date have been conducted on wetlands of the Laurentian Great Lakes, and have shown that short-term water levels and circulation in many of coastal wetlands are partially driven by storm surges and wind-driven seiche water level changes on adjoining lakes (Maynard and Wilcox 1997; Sierszen et al. 2006; Trebitz 2006; Helvca and Wells 2009; Gathman and Burton 2011; Morrice et al. 2011). Daily water level dynamics for seiche-dominated Great Lakes wetlands were found to be more spatially and temporally variable than for tide-

driven coastal marine estuaries (Trebitz 2006). Unfortunately, many of these studies have been conducted in relatively small open-bay coastal wetlands ranging from less than one hectare (ha) to 100 ha, and connected to their adjoining lakes via one main channel, making it difficult to directly apply results to Delta Marsh, which in contrast is 18,500 ha, composed of many connected and isolated bays and channels and connected to Lake Manitoba via four permanent channels. During the open-water season in southern Manitoba the wind is predominantly from the north/north-west and south/south-west (Environment Canada, 2012). As a result during strong wind events, seiche activity on the lake often occurs along the lakes longest axis (north to south, 225 km), and pushes the water level up at the down-wind end of the lake and make the level drop by a corresponding amount at the opposite end (LMRRAC 2003a). Delta Marsh located at the south-end of the lake (Figure 1.1) is anti-nodal for all north/south oscillations, making it subject to the full extent of the lake's seiche-induced water level fluctuations and water movements. Aside from its surface water connection to Lake Manitoba, the marsh does not have any permanent watershed tributaries that directly empty into it, so surface runoff (i.e. spring melt and storm water) is the only other main source of water and materials to the marsh from the surrounding watershed, along with direct precipitation and groundwater inputs.

This chapter examines short-term (daily) and long-term (annual) spatial and temporal trends in the hydrology (water level and flow) of Delta Marsh, as influenced by the hydrology of adjoining Lake Manitoba. Over the course of the study Lake Manitoba experienced its lowest and second highest water levels on record since 1944, with the lowest average water levels of 247.0 m ASL (range 247.0 to 247.6 m ASL) occurring in 2003, and the second highest water levels of 247.8 m ASL (range 247.3 to 247.8) occurring

in 2005 (Figure 1.3). These large natural differences in water levels allowed for examination of annual differences in the marsh hydrology relative to varying water levels on Lake Manitoba. Daily water level fluctuations (magnitude and frequency) in Lake Manitoba and Delta Marsh were examined to determine the importance of seiche-induced water exchange from Lake Manitoba on the water levels of Delta Marsh.

The location of this study is described in detail in Chapter 1.3, and the study sites and design are described in detail in Chapter 3.1. The detailed hydrology study methods and data analyses are presented in Chapter 3.2.1 and Chapter 3.3.1.

Results

Annually, water levels on Lake Manitoba are influenced by changes in water inputs from the surrounding watershed, and outputs via the Fairford dam on the Fairford River (Figure 1.4). During the study Lake Manitoba experienced its lowest and second highest water levels on record since 1944, with the lowest average water levels of 247.0 m ASL (range 247.0 to 247.6 m ASL) occurring in 2003, and the second highest water levels of 247.8 m ASL (range 247.3 to 247.8) occurring in 2005 (Figure 1.3). In 2002 and 2004, yearly water levels averaged 247.3 m ASL (range 247.1 to 247.6) and 247.2 m ASL (range 247.1 to 247.5), respectively. This allowed for the examination of relationships between Marsh and Lake water levels over a large range in Lake water level conditions. Figure 4.1 illustrates water levels (hourly) in Lake Manitoba from 2002 to 2005, as well as corresponding water levels (hourly) at two representative sites in Delta Marsh, one in the western portion of the marsh (CAN), and one in the eastern portion of the marsh (SIMP) during the open water period of the study, April 1 to October 31, 2002 to 2005. Since the data was recorded hourly, it also included short-term water level changes in the lake and marsh due to seiche activity. The data show that water levels in connected sites in the marsh were responsive to the long-term and short-term fluctuations in Lake Manitoba water levels. Marsh water levels generally followed the annual trends in the Lake, in 2002 and 2003 water levels were highest in the spring and decreased throughout the season, whereas in 2004 and 2005 water levels increased from spring to a mid-summer maximum followed by a gradual decrease through the late summer and fall (Figure 4.1). Linear regression of the long-term (2002-2005) water level data at the Marsh sites against those on Lake Manitoba, indicated water levels at the connected marsh

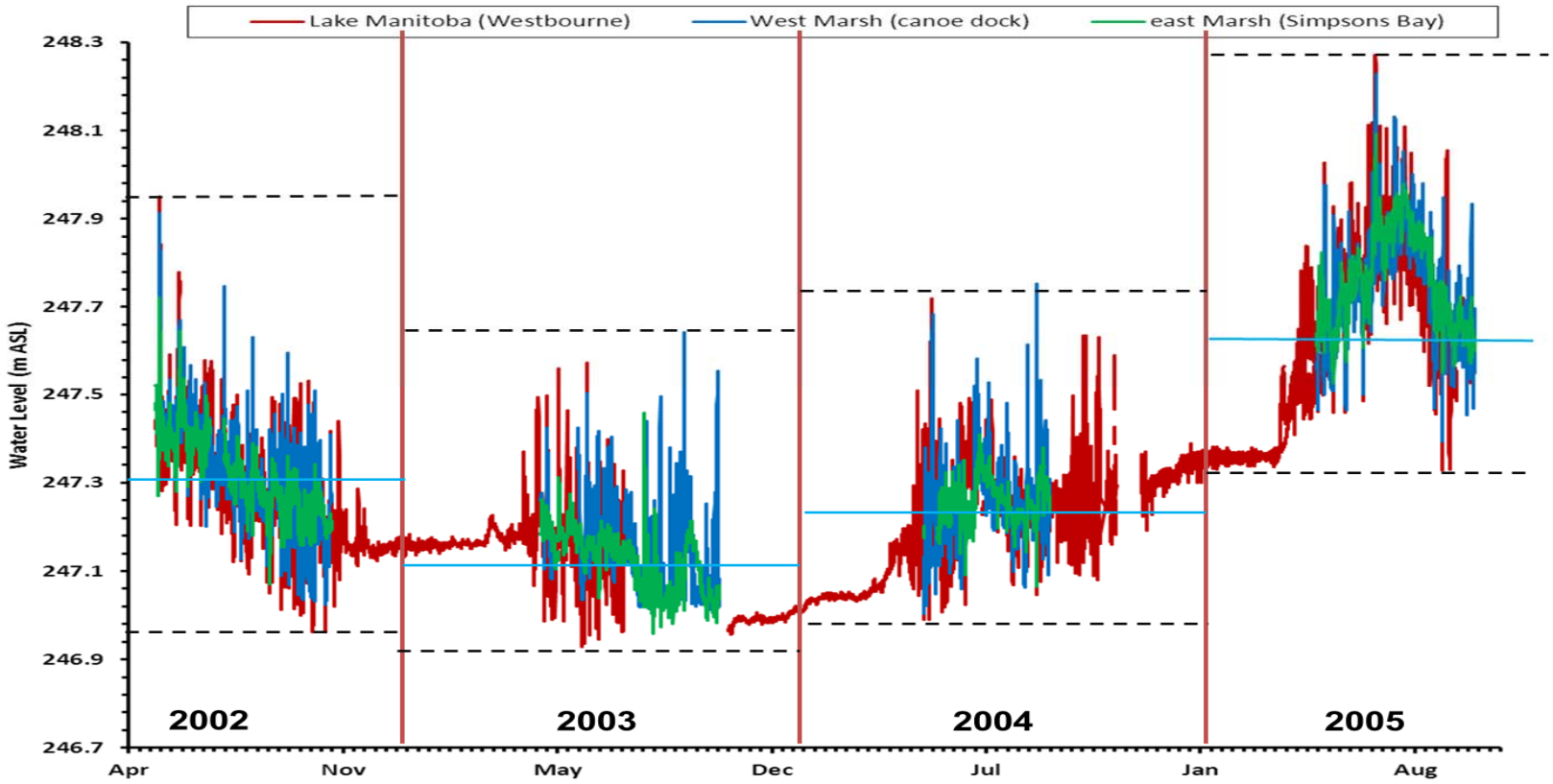

Figure 4.1 Time series of water levels in Lake Manitoba and the west and east sections of Delta Marsh during the study period from 2002 to 2005. The red lines represent the start of each calendar year. The data was recorded hourly, so include water level changes in the lake due seiche activity, and associated changes in water level in the marsh. Back dashed lines represent the upper and lower ranges and the blue lines are the average lake and marsh levels for each year. Water levels for the east portion of the marsh were recorded in Simpsons Bay and for the western portion in Blind Channel near the University of Manitoba's Delta Marsh Field Station. Water levels for Lake Manitoba were recorded at Steep Rock (Water Survey of Canada Station 05LK002). Note water level data is missing for Lake Manitoba for August to November, 2003 and November 2004.

Table 4.1 Correlations between water levels on Lake Manitoba (m ASL) and water levels at sites in Delta Marsh from 2002 to 2005. NOTE: CAN, CAD and PCS were monitored from 2002-2005, CAD and DCH were monitored 2004-2005, CENT in 2004, and RICH in 2005. For locations of sites refer to Figure 3.2 and Table 3.2.

Site	Distance from Lake Manitoba (km)	East or West side of marsh	Isolated or connected	*n*	p value	F ratio	r^2
DMFS Canoe -dock (CAN)	3.9	West	Connected	8210	<0.0001*	61573	0.88
Short Creek (SCRK)	5.7	West	Connected	847	<0.0001*	352	0.70
Big Lake NW (BLNW)	8.5	West	Connected	3921	<0.0001*	1525	0.52
Delta Channel - (DCH)	1.1	East	Connected	2284	<0.0001*	7284	0.76
Cadham Bay (CAD)	3.6	East	Connected	3565	<0.0001*	49898	0.80
Simpson's Bay (SIMP)	5.7	East	Connected	8532	<0.0001*	68462	0.89
Portage Creek South (PCS)	11.5	East	Connected	7900	<0.0001*	78334	0.91
Center Marsh (CENT)	n/a	East	Isolated	139	0.0545	76	0.35
Richardson's Pothole (RICH)	n/a	East	Isolated	2532	0.0010*	1849	0.42

sites were significantly correlated with lake water levels (Table 4.1). Water levels in one of the two isolated sites, Center Marsh (CENT), were not correlated with the Lake ($p = 0.0545$, $r^2 = 0.35$) , however water level in the other isolated site, Richardson's Pothole (RICH), were correlated with the Lake ($p = 0.0205$, $r^2 = 0.42$); Table 4.1).

Short-term water level fluctuations on Lake Manitoba and Delta Marsh are often influenced by wind-induced seiche activity on the lake. During the study, seiche induced water level changes on the lake ranged in duration from hours to days (Figure 4.2). During north/north-west wind seiche events water levels rose in the south-end of the lake, causing water to flow into the marsh and increase water levels, and during south/south-east wind seiche events water levels dropped in the south-end of the lake, causing water to flow out of the marsh and reduced marsh water levels. Water also moved into or out of the marsh, when seiche activity was followed by calm wind periods, as lake and marsh water levels equalized and returned to pre-seiche levels (personal observation). As shown in Figure 4.1, compared to Figure 1.3, seiche activity results in greater short-term variation in the daily range of water levels on Lake Manitoba (± 0.5), as well as Delta Marsh. Figure 4.2 illustrates water level changes that occurred in the Lake and Marsh during the 2005 field seasons from May 1 to August 31, and includes detailed snap-shots of the range in water level change (difference between high and low water levels) that occurred on the lake and Marsh as a result of daily short-term wind-induced seiche events. During this time typical daily water level range magnitude varied amongst the sites, and over time. For example, from May 21 to 22 and June 29 to 30, 2005 strong north/north-west wind events in the range of 25 km/hr and greater resulted in increased water levels in the magnitude of 35 to 50 cm in the south end of the lake, and 20 to 40 cm in the Marsh (Figure 4.2a and b).

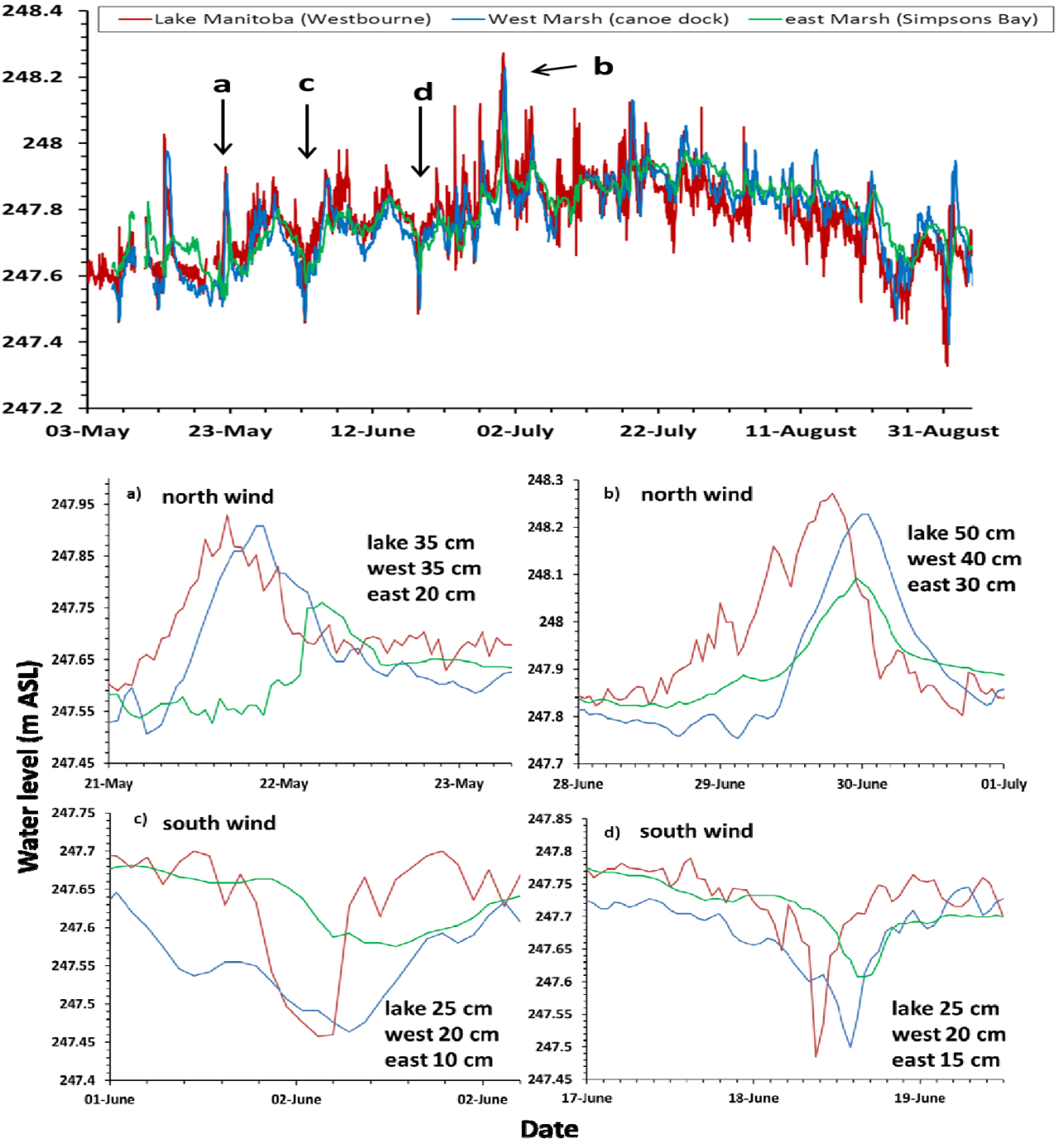

Figure 4.2 The top panel shows a time series of water level fluctuations in Lake Manitoba (Westbourne), west Delta Marsh (Canoe Dock), and east Delta Marsh (Simpsons Bay), from May 1 to August 31, 2005. The Canoe dock and Simpsons Bay sites are 3.9 and 5.7 km from the lake, respectively. Detailed plots (a-d) show water levels changes at these sites during wind-induced seiche events. Water level range values are given. Note that the axes are offset.

Similarly, from June 1 to 2 and June 18 to 19, 2005 strong south/south-east winds resulted in decreased water levels of 25 cm in the south end of the lake, and 10 to 25 cm in the Marsh (Figure 4.2c and d). In both cases the magnitude of water level change was slightly greater (± 5 to 15 cm) at the west marsh site (Canoe Dock) compared to the east marsh site (Simpsons Bay), however the overall average magnitude of daily water level change at the other west marsh sites was in the same range of magnitude as the east marsh sites (Table 4.2). The peak in Marsh water level changes also displayed a delay of approximately 4 to 5 hours compared to the lake (Figure 4.2a - d).

Over the course of the entire study from May 1 to October 31, 2002 to 2005, overall variations in short-term daily water level range in the lake and connected sites in the marsh were variable and log-normally distributed (Table 4.2; Figure 4.1). The average daily range in water level changes on Lake Manitoba was two to six times greater in magnitude than water levels changes in the marsh, with an average daily range in water level of 15 ± 0.5 cm, and minimum and maximum values of 3.5 and 59.9 cm, respectively. Marsh sites also experienced a smaller magnitude of range in water levels (~40 to 80 % attenuation) compared to the lake. In the Marsh the magnitude of daily range in water level averaged from 2.0 ± 0.2 to 9.6 ± 0.4 cm, with the full magnitude of water level change ranging from a minimum of 0.2 cm to a maximum 57.6 cm. In comparison of the east and west sections of the marsh, average daily water levels ranges where in the same magnitude, with the exception of the canoe dock site (CAN) in the west, which experience average daily values two to four times greater in magnitude. The average magnitude of daily water level range in the Marsh decreased significantly with increasing distance from Lake Manitoba in both the east ($p = 0.047$, $r^2 = 0.87$) and the west ($p = 0.047$, $r^2 = 0.91$) sections of the Marsh,

Table 4.2 Daily water level fluctuations for lake and marsh sites, as described by the back-transformed logarithmic mean ± standard error (SE) of daily range (a measure of magnitude alone), and of one-half the sum of daily water level increments (a measure of magnitude and frequency), from May to August, 2002 to 2005. For site locations refer to Figure 3.2 and Table 3.2.

Site	Code	Location[1]	Distance[2] from lake (km)	daily range (cm)			1/2 ∑ daily increments (cm)			
				Mean ± SE		min	max	Mean ± SE		min	max
Lake Manitoba at Westbourne	LKWT	Lake	n/a	15.2	± 0.5	3.5	59.9	30.9	± 1.1	7.0	120.7
DMFS Canoe dock	CAN	West	3.9	9.6	± 0.4	1.0	56.7	11.2	± 0.3	1.0	53.3
Short Creek	SCRK	West	5.7	3.4	± 0.4	0.2	27.8	3.7	± 0.4	0.2	25.0
Big Lake NW	BLNW	West	8.5	2.4	± 0.2	0.3	26.7	2.7	± 0.5	0.3	24.0
Delta Channel	DCH	East	1.1	4.0	± 0.3	0.8	27.7	4.7	± 0.2	1.4	28.5
Cadham Bay	CAD	East	3.6	3.5	± 0.2	0.5	24.1	4.0	± 0.1	0.6	22.4
Simpson's Bay	SIMP	East	5.7	3.3	± 0.2	0.2	27.6	3.7	± 0.2	0.5	23.5
Portage Creek South	PCS	East	11.5	3.0	± 0.5	0.2	33.0	3.4	± 0.2	0.4	19.8
Center Marsh	CENT	Isolated	n/a	1.3	± 0.2	0.2	4.8	1.4	± 0.3	0.1	7.5
Richardson's Pothole	RICH	Isolated	n/a	2.0	± 0.2	0.2	14.2	2.1	± 0.2	0.1	16.9

[1] Location: Lake, connected east Marsh, connected west Marsh, and isolated Marsh.
[2] Distance from Lake Manitoba via surface water flow.

however the degree of decrease was greater in the west (from 9.6± 0.4 to 2.4 ± 0.4 cm) compared to the east (from 4.0 ± 0.3 to 3.0 ± 0.5 cm, Table 4.2). The magnitude of daily water level range was smaller in isolated sites compared to the connected sites, with average values of 1.3 ± 0.2 to 2.0 ± 0.2 cm, and a smaller variation between minimum and maximum values of 0.2 to 14 cm (Table 4.2).

Intensity of water level fluctuations (one-half the sum of water level increments per day) combines both the magnitude and frequency of water level changes. The general patterns in the daily intensity of water level fluctuations in the lake and Marsh were similar to values for magnitude of water level fluctuations (Table 4.2). Lake Manitoba experienced three to ten times the intensity of water level fluctuations compared to the Marsh, with average values of 30.9 ± 0.3 cm in the lake, with a minimum and maximum of 7.0 to 120.7 cm, respectively. Average daily values in the Marsh ranged from 2.7 ± 0.4 to 11.2 ± 0.3 cm, with minimum and maximum values from 0.2 to 53.3 cm. The average intensity of daily water level range in the Marsh also decreased significantly with increasing distance from Lake Manitoba in both the east ($p = 0.0029$, $r^2 = 0.83$) and the west ($p = 0.0279$, $r^2 = 0.88$) sections of the Marsh, however the degree of decrease was greater in the west (from 11.2± 0.3 to 3.7± 0.5 cm) compared to the east (from 4.7 ± 0.2 to 3.4 ± 0.2 cm, Table 4.2). Average daily intensity in isolated Marsh sites was 30 to 50% that of connected sites, with values of 1.4 ± 0.3 to 2.1 ± 0.2 cm, and a minimum and maximum of 0.1 to 16.9 cm, respectively. In the lake the intensity of daily water level range was found to be approximately two times the magnitude of the lakes daily water level range, and in the marsh it was 1.2 times in east and west connected sites (Figure 4.2). This indicates the frequency of water level change is approximately two times greater in the Lake compared

to connected sites in the marsh. Further, the intensity and magnitude of water level range were nearly 1:1 in isolated Marsh sites, indicating changes in water level were less frequent than connected Marsh sites.

The influence of the wind direction and speed on seiche-induced water exchange between the lake and marsh was examined by measuring the volume and velocity of water flow into the marsh through Delta Channel under varying north wind event velocities in 2005. Seiche-induced water exchange flow from the lake through Delta Channel into the marsh was found to be positively correlated with the velocity of north wind events ($n = 23$; $r^2 = 0.82$, $p < 0.0001$; Figure 4.4). Based on the linear regression, at a wind speed of 15 km/hr, the estimated flow velocity through the channel was approximately 1.0 m^3/sec, and at a wind speed of 50 km/hr, it increased to approximately 4.0 m^3/sec.

Short-term water level fluctuations in the marsh that result from water exchange with the lake can also drive flushing and residence time (i.e. the amount of time (days) it takes for a known volume of water to be replaced with the same volume of new water). For Delta Marsh this calculation is complicated due to its large size and the complex morphology of numerous bays and channels of various areas and depths. However, values can be calculated for smaller sections of the marsh (i.e. Cadham Bay). For example, based on the data it can be estimated by two calculations (1) the average volume of water in Cadham Bay (m^3), divided by the velocity of water flow (m^3/day) through Delta Channel into Cadham Bay, which equals the average number of days it takes water flow through Delta Channel (at given velocity) to flush Cadham Bay (Equation A, Chapter 3.3.1), and (2) the average depth ($z_{mean;}$ cm) of Cadham Bay multiplied by 1 day, and divided by the average intensity of water level fluctuations(cm) on Cadham Bay (calculated above), which equals

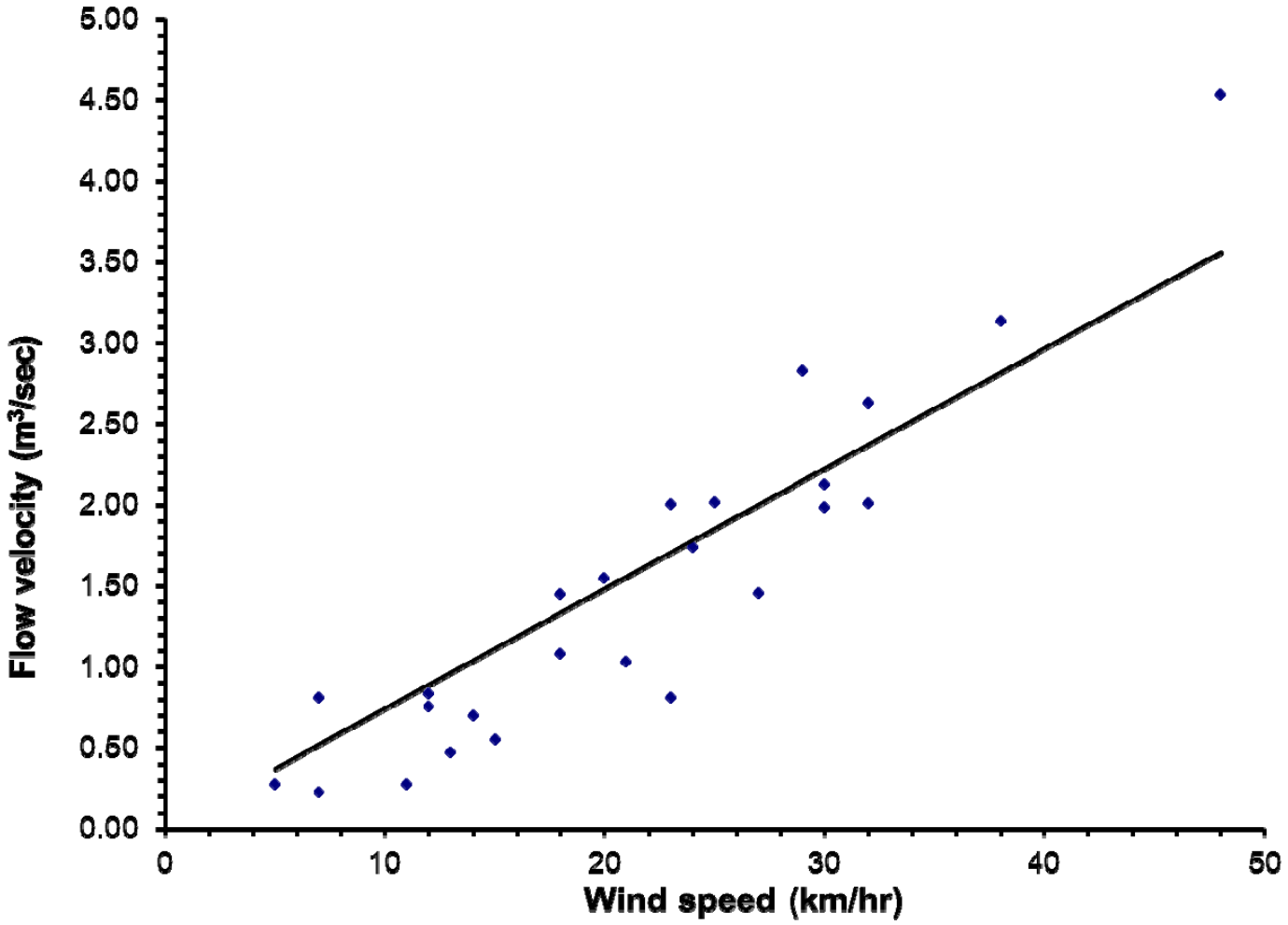

Figure 4.3 Relationship between average wind speed (km/hr) from the north and flow velocity (m^3/sec) in Delta Channel, May 1 to August 31, 2005. $r^2 = 0.82$, $p < 0.0001$. Wind speeds are average wind speeds over the previous 8 hours.

the average number of days it takes lake seiches to completely flush Cadham Bay (Equation B, Chapter 3.3.1) (Trebitz 2006, Wells and Sealock 2009). The assumption of both calculations is that the incoming lake water completely mixes with water in Cadham Bay.

Based on Equation A, using the wind and flow velocity data from Delta Channel (Figure 4.4), during north wind of 15 km/hr flow velocity was approximately 0.9 m^3/sec (77,760 m^3/day), and at a wind speed of 50 km/hr flow velocity was approximately 4.6 m^3/sec (397,440 m^3/day), and an average volume of Cadham Bay of 9,000,000 m^3 (based on average area of 9,000,000 m^2 and depth of 1 m; Appendix A bathymetric surveys of Cadham Bay), residence time due solely to water flow in Delta Channel from water level fluctuations on Lake Manitoba, ranged from approximately 22 to 115 days.

Based on Equation B, using the average intensity of water level fluctuations on Cadham Bay of 4.0 cm, and the average depth of Cadham Bay of 100 cm, residence time due solely to seiche induced water level fluctuations, would have been of 25 days, with an average range from 4.5 days (using the maximum value for intensity, 22.4 cm) to 166 days (using the minimum value for intensity, 0.6 cm). Since Equation B is based on water depth and not water volume, the resultant residence time for Cadham Bay can also be applied to the entire east section of the marsh, based on assumptions that average water depth (1 m) and average intensity of water level fluctuations in Cadham Bay are similar to those across the entire east section of the marsh.

Discussion

This study is the first to examine spatial and temporal patterns in the short-term (daily) and long-term (annual) hydrology—water levels, flow, mixing and flushing—in a lacustrine coastal freshwater wetland influenced by short-term seiche-induced water level fluctuations and yearly water level changes of the adjoining lake, outside the small coastal wetlands of the Laurentian Great Lakes. Both daily and annual water level changes in the marsh were found to be highly correlated with those of the lake, with marsh levels matching those of the lake during some of the highest and lowest long-term water levels on the latter in recorded history, as well as rapid daily (hours to days) fluctuations of lake water level. The average magnitude of water level change (a few centimeters to half a meter) may seem small, but they are significant in shallow coastal wetlands systems like Delta Marsh where the average depths are ≤ 1 m. The magnitude of daily water level changes in the marsh were 20 to 50% the values reported by Trebitz (2006) in coastal wetlands of the Laurentian Great Lakes, with the exception of Lake Ontario, which had similar values. The relationship between magnitude and intensity (½ ∑ daily increments) of water level changes in the Great Lakes marshes was also much greater at 1:3, compared to sites in Delta Marsh at 1:1.2. These differences are likely due to the larger size (~18,500 ha), volume and morphological complexity (i.e. four lake/marsh connecting channels, and internally alternating channels and bays) of Delta Marsh compared to the smaller (<1 to 100 ha) and less complex marshes (i.e. mostly open bays with one connecting channel) examined in the Laurentian Great Lakes. The greater size, volume and complexity of Delta Marsh likely resulted in it requiring a larger water level change magnitude and intensity

over a long time period to increase marsh water levels, compared to smaller and less complex Great Lakes wetlands where water level changes occur more quickly. Another source of variation is that Delta Marsh does not have permanent watershed tributaries that empty directly into the marsh, while many of the Great Lakes marshes have one to several permanent tributaries. In marshes with tributaries, water level changes due to lake seiche activity are often overshadowed during high tributary flow periods, and in some cases throughout the entire open water season. Trebitz et al. (2002) and Morrice et al. (2011) in studies of coastal wetlands on Lake Superior found that tributary flow had a greater influence on marsh hydrology than the adjoining lake, and seiche magnitudes were reduced and even negligible in wetlands with tributary flow and small wetland-lake connecting channel mouth size. In wetlands like Delta Marsh, with no upland tributary flow, seiches are the dominant source of water mixing and flushing. Sierszen et al. (2006) agreed that seiches would be the dominant form of advective water mixing and flushing in coastal wetlands with little to no tributary inputs.

Water level magnitude and intensity in Delta Marsh were likely attenuated to a large degree from those occurring on the adjacent lake (40 to 80 % in magnitude and 60% to 90 % in intensity) due to the relatively small area of the connecting channels, and further spatially within sections of the marsh due to its complex interchanging basin and constricting channel morphology. For instance the channels connecting the marsh to the lake (i.e. Deep Creek (~15.0 m wide x ~ 1.0 m deep) and Cram Creek (~ 25.0 m wide x ~ 2.0 m deep) on the west, and Delta Channel (~ 7.0 m deep x 1.0 m deep) and Clandeboye Channel (~30 m wide x 1 m deep) on the east), (Figure 1.3 and 3.1) can effect short-term

water levels fluctuations from Lake Manitoba on the marsh. Water flowing through Delta Channel is also further reduced due to Delta Bridge, which reduced flow to a width of 5.0 m and depth of 1.0 m. In the west side of the marsh, the greater attenuation of water level range at BLNW and SCRK (~80%) compared with CAN (~40%), was likely due to CAN being primarily connected to the lake via Cram Creek which has three times the area (twice as wide and deep) as Deep Creek, which connects BLNW and SCRK to the lake. Further water from the lake to BLNW and SCRK is also further constricted by another channel (Carp Creek, ~ 15.0 m wide x 0.75 m deep) before flowing into the larger bay area of the marsh (Big Lake South) that connects to these two sites. This illustrated the significant influence of the number and size of connecting channels on the magnitude and intensity of water exchange between lakes and their coastal marshes. This observed attenuation is similar to the observations of Trebitz et al. (2002) who found that degree of seiche attenuation increased with decreasing connecting channel size for several coastal wetlands on Lake Superior, where a channel with a cross-sectional area of 20-30m^2 attenuated seiche magnitude by 40-70%. Wells and Sealock (2009) found a narrow channel (30 m wide and 2 m deep) connecting Frenchman's Bay and Lake Ontario attenuated average daily water levels in the bay by 50 %, and Seldomridge and Prestegaard (2012) in a study of 267 tidal freshwater marshes found that total channel area (width and length) was more closely related to tidal water volume exchange than marsh watershed area.

According to de Geus (1987), many more channels once connected Delta Marsh to Lake Manitoba than at present but, over time, many have silted up and never reopened, indicating that it is likely that water level fluctuations, exchange, mixing, and dilution of

the marsh with lake waters have dampened over time. Since the 1940s, various man-made water control structures have been constructed on the mouths of marsh and lake connecting channels, namely Deep Creek, Cram Creek and Clandeboye Channel. While these structures no longer exist, they would have attenuated and/or impeded water exchange while they existed, and likely had effects that changed the morphology of the channels as well as other bays in the marsh. Likely reasons include increased siltation and vegetation growth that would accompany reduced water level changes, water exchange, and mixing in areas of the marsh. As noted in the results, one man-made structure still remains today, Delta Bridge, which crosses Delta Channel, and effectively reduces water flow exchange between the lake and marsh through the channel, which is easily visible during high north wind events when large amounts of water pile up on the north side of bridge. Similarly, Helvca and Wells (2009) found that man-made changes to connecting channels have the capacity to increase or reduce residence time of water in wetlands, and that a linear relationship existed between increased flushing of Freshman's Bay, an enclosed wetland on Lake Ontario, with increased cross-sectional area of a connecting channel mouth. Historically, aside from short-term seiche-induced water level changes, the magnitude of long-term water level changes on Lake Manitoba have also been dampened by the Fairford Dam control structure constructed on the north outlet of the lake in 1961, reducing the lake's long-term water level range from 2.2 m to a controlled range of 247.6 MASL ±0.3 m (de Geus 1987). As a result of the significant relationship between lake and marsh water levels, water levels in the marsh have also been dampened. Together, these man-made changes which have dampened and changed the magnitude and intensity of water level, exchange, mixing, and flushing between the marsh and lake, have likely had other

cumulative temporal and spatial effects on the structure and function of Delta Marsh that are dependent on hydrology, including marsh morphology, biochemistry, nutrient dynamics, energy transport, flora and fauna, and overall food web structure (Pringle 2001, Trebitz et al. 2002, Morrice et al. 2004, Sierszen et al. 2004, Trebitz 2006, Sierszen et al. 2006). For instance, hydrology determines water movement mixing and residence time in wetlands, which likely influences how much time nutrients are available in wetlands for uptake and other biochemical processes which, in turn, influences primary production, secondary production, and overall trophic structure.

The estimates of residence time in Cadham Bay and the entire Delta Marsh presented here are the result of several simplifying assumptions. The use of the intensity of water level change (½ ∑ increments) and/or water flow velocity in connecting channels offer much promise; however, more detailed spatial and temporal data sets are required to obtain more precise estimates of flushing time-scales in the marsh. While there are differences in the estimates of residence time for Cadham Bay between the two methods (intensity = 4.5 to 166 days versus flow velocity = 22 to 115 days), it is important to note that both estimates of residence time are of the same order of magnitude. The estimate of marsh residence time from the intensity calculation can also be directly applied to the whole marsh as it is based on water volume, whereas the flow velocity calculation would have to be multiplied by conversion factor (i.e., based on the additional area of marsh to be included) to applied to the whole marsh, as it is based on area. The differences in the results of the two calculations are likely due to limited flow data which were only collected by spot measurements during a four-month period in 2005, compared to the more extensive

water level intensity data, which were collected over a four-year period. These data would benefit from increased spatial measurements across the marsh. It would be increasingly beneficial to combine the two measures and include the measurement of water flow direction in one or several connecting channels, as flow direction can change frequently with changes in wind-induced seiche activity on the lake. They do, however, represent an important starting point for future hydrological studies in Delta Marsh. Future hydrological studies in Delta Marsh would benefit from increased spatial and temporal sampling; sampling of groundwater, precipitation, evaporation, evapotranspiration; and any surface flow from the upland in the spring or during large storms to better understand the importance of these components of the marsh water budget.

This study illustrates the intimate hydrological relationship that coastal lacustrine wetlands share with their adjoining lakes on several spatial and temporal scales. The resultant fluctuations in water level, exchange, and flushing magnitude and frequency play important roles in the ecology and associated foundational structure and function of these coastal systems. The hydrologic connectivity alone allows for the movement of organisms, water, and suspended and dissolved materials between these lakes and wetlands (LaPointe 1986, Brazner et al. 2000, Bouvier et al. 2009); the daily and long-term water level changes have important implications on flora and fauna and associated food webs (Keough et al. 1999, Brazner et al. 2000, Wilcox et al. 2002, Sierszen et al. 2004, Sierszen et al. 2006, Wilcox and Nichols 2008, Gathman and Burton 2011, Wilcox 2012); and resultant water movement and flushing, transport and stir sediment, nutrients, and small organisms, and

effect water and sediment chemistry (Trebitz et al. 2002, Morrice et al. 2004, Morrice et al. 2009, Morrice et al. 2011, Wilcox 2012).

In order to build complete functional understanding of coastal wetlands, studies and developments need to be mindful of the hydrologic connectivity between coastal wetlands and their adjoining lakes, and at a larger scale to the surrounding watersheds, to be able to distinguish between natural variability (reference conditions) and anthropogenic influences, and to assess their vulnerability to change, and establish goals for wetland restoration and protection (Keough et al. 1999, Morrice et al. 2011, Trebitz et al. 2011, Wilcox 2012). Global climate change is predicted to have the most pronounced effect on wetlands through hydrologic changes, particularly changes in hydroperiod variability and the number and severity of extreme events (IPCC 1996, Erwin 2009). The physical connectivity of the lakes and coastal marsh may be affected as lake and marsh hydrology change, which would result in cascading effects on all aspects of wetland and lake structure and function including biogeochemistry; transport of nutrients, energy, and organisms; and food webs (IPCC 1998).

Chapter 5: Results – Water Chemistry

Introduction

Situated in the landscape between upland watersheds and lakes, coastal wetlands serve as important sites that influence the exchange of water, nutrients, chemicals, biota, and other materials between their drainage areas and adjoining lakes (Kadlec and Knight 1996, Mitsch and Bouchard 1996, Brazner et al. 2000, Trebitz et al 2007, Wilcox 2008, Gathman and Burton 2011, Morrice et al 2011, Trebitz et al. 2011, Wilcox 2012), while often occupying only a small portion of the landscape. In other words, their importance may be disproportionate to their size.

Attempts to better understand the physical, and biochemical processes that occur within coastal wetlands, as well as the influence of adjoining water bodies (i.e. watershed tributaries and lakes) on these processes, has been an area of growing research over the last decade. Studies to date, however, have primarily occurred around the Laurentian Great Lakes wetlands, making it difficult to generalize and apply findings to other systems (Trebitz 2002, Morrice et al. 2004, Trebitz 2006, Sierszen 2006, Trebitz et al. 2007, Wilcox and Nichols 2008, Morrice et al. 2011, Trebitz et al. 2011). Less is known about the numerous other coastal freshwater wetlands that surround many other lakes, including Lakes Winnipeg, Manitoba and Winnipegosis in Manitoba (Watchorn et al. 2012).

Physical, chemical and biological processes in coastal wetlands have been shown to vary amongst different systems, and within systems, and hydrology is a key influence on these processes and in determining wetland structure and function (Keough et al. 1999,

Grosshans 2001, Trebitz et al. 2002, Morrice et al. 2004, Sierszen et al. 2009, Trebitz 2006, Wilcox 2007, Wilcox et al. 2008, Gatham and Burton 2011, Wilcox 2012). Due to their small volumes and shallow water depths, wetlands are dynamic environments in which small changes in water levels can result in significant biological and chemical changes (Kadlec and Knight 1996, van der Valk 2006). Further, in the case of coastal freshwater wetlands, hydrology is further complicated as these systems are subject to an interplay of varying inputs of water and nutrients from the watershed by tributary flow and/or adjoining lakes by seiche activity (Wetzel 2001, Trebitz et al. 2002, Morrice et al. 2004, Lotze et al. 2006; Trebitz 2006, Trebitz et al. 2007, Wilcox et al. 2007, Diaz and Rosenberg 2008, Morrice et al. 2011, Trebitz et al. 2011). The full effects of these hydrological connections and water and material exchange on adjoining freshwater wetlands are still unclear, but the flow exchanges appear to act similarly to tides in marine coastal estuaries, allowing exchange of water, nutrients and other materials between wetlands and the lakes, as well as important impacts on biota exchange, diversity, and productivity (Maynard and Wilcox 1997, Sierszen et al. 2006, Trebitz 2006, Wilcox et al. 2007, Trebitz et al. 2009, Gathman and Burton 2011, Morrice et al. 2011, Trebitz et al. 2011, Wilcox 2012). The influence of these hydrological connections can vary greatly due to their own temporal variability. Changes in the magnitude of these hydrological influences may also affect water residence time which, in turn, influences water chemistry parameters, including nutrient availability, sinks and sources (Wold and Hershey 1999, Trebitz et al. 2002, Morrice et al. 2004, Trebitz et al. 2004, Sierszen et al. 2006, Trebitz 2006, Morrice et al. 2011), with nutrient availability and retention increasing with increased hydrological residence time.

Increased nutrient loading and degraded water quality are significant threats to the structure and function of aquatic ecosystems, including coastal wetlands. Strategies to protect and rehabilitate these ecosystems are needed. To accomplish this, an effective understanding of the key natural factors that influence biochemical processes in these systems is needed, and in the case of coastal wetlands, understanding the hydrological relationship between adjoining water bodies and wetland water chemistry is key.

This chapter builds on the results presented in Chapter 4 (hydrology) and examines spatial and temporal patterns in the water chemistry and nutrients (specifically N and P) of Delta Marsh, as influenced by hydrological water exchange with Lake Manitoba. Over the course of the study Lake Manitoba experienced its lowest and second highest water levels on record since regulation, with the lowest average water levels of 247.0 m ASL (range 247.0 to 247.6 m ASL) occurring in 2003, and the second highest water levels of 247.8 m ASL (range 247.3 to 247.8) occurring in 2005 (Figure 1.3). These large natural differences in water levels allowed for examination of spatial and temporal differences in the water chemistry marsh relative to varying water levels on Lake Manitoba.

The use of naturally occurring chloride (Cl^-) concentrations and oxygen-18 (^{18}O) and deuterium (^2H) stable isotope signatures as conservative hydrological tracers to measure dilution and residence time of waters within the marsh, as affected by water exchange with the lake are also explored. Cl^- represents a useful tracer because it is relatively inert and not used by biota to any great degree, has also been used as an effective hydrologic tracer in other wetlands in the northern prairie region of North America (Kadlec and Knight 1996, Hayashi et al. 1999, Waiser 2006), as well as the Laurentian Great Lakes Wetlands (Morrice et al. 2011). Assuming lake-ward decreases in chloride between

connected marsh sites are solely due to dilution, gradients in chloride concentration between lake and wetland waters were used to determine the extent of intrusion and dilution of lake waters in the wetland. Oxygen-18 and deuterium stable signatures of individual water bodies can reflect their mean residence time, seasonal timing of water inputs, and mixing with other water sources, for example in this case the mixing of lake waters. Assuming lake-ward increases in ^{18}O and ^{2}H isotopic signatures between connected marsh sites are the result of decreased lake water inputs and mixing and increased residence time, gradients in their signatures between lake and wetland waters were used to determine the extent of intrusion and dilution of lake waters in the wetland. Spatial trends in nutrients (N and P), major ions (Cl^-, SO^-_4, conductivity), dissolved organic carbon (DOC), phytoplankton biomass (chlorophyll *a*), and suspended solids (TSS) with decreasing distance from Lake Manitoba were examined to determine if chemical concentrations decreased with decreasing distance from the lake via surface water flow, due to the dilution effect of waters entering the marsh from the lake, as well as the agriculturally dominated watershed having potential to be a large source of N and P nutrients to the marsh, and natural uptake process of N and P nutrients in the marsh. Water chemistry in connected and isolated areas of the marsh was also examined to determine if isolated areas had greater concentrations of nutrients (N and P) and major ions due to a lack of surface water connection to Lake Manitoba.

The location of study is described in detail in Chapter 1.3. The study sites and experimental design are described in Chapter 3.1, and the detailed sample collection and

analysis, and data analysis methods are described in Chapter 3.2 and 3.3, respectively. The results of the hydrology studies conducted in the marsh are presented Chapter 4.

Results

Chloride, and oxygen-18 and deuterium stable isotope

The regression of chloride (Cl^-) concentrations against site depth at all connected marsh samples sites (east and west) from 2003 to 2005 (n = 225), indicated chloride concentrations were inversely correlated with water depth ($r^2 = 0.59$, $p = <0.0001$) (Figure 5.1).

From 2003 to 2005, Cl^- concentrations were lower in Lake Manitoba compared to the majority of marsh sites (Figure 5.2a and b; Figure 5.2), and concentrations in the marsh decreased significantly (30 to 40%) with decreasing distance to Lake Manitoba along both the east and west transects, with the exception of Portage Creek South (PCS) and Portage Creek Bridge (PCB) (Figure 5.2a and b, Figure 5.3, Appendix D). PCB and PCS, the two furthest inland sites along the east marsh transect, did not fit the linear regression of increasing Cl^- concentration with increasing distance from the lake as both sites experienced large water inputs from the surrounding watershed via spring runoff and periodic large storm water runoff events which resulted in inputs of "fresher" water with lower Cl- concentrations compared to next closest site, Portage Creek North (PCN), and as a result they did not (Appendix E). When these sites were removed from the regressions, the significance of the trend increased significantly (Figure 5.3 without PCS and PCB; with PCS and PCS in 2003 $r^2 = 0.26$, $p = 0.0397$; in 2004 $r^2 = 0.08$, $p = 0.0300$; and in 2005 $r^2 = 0.05$, $p = 0.0675$). From 2003 to 2005, with increasing water level on Lake Manitoba and the marsh, mean Cl^- concentrations at lake and marsh sample sites and the difference in Cl^- concentrations between samples sites both decreased, indicating a greater influx and

mixing of "fresher" lake water further into the marsh in high water years and reduced marsh Cl^- concentrations (Figure 5.2a and b, Figure 5.3, Appendix D).

In comparison of study years, increasing water levels from 2003 to 2005, were associated with reduced Cl^- at all the east marsh sample sites, as well as a decrease in variation in Cl^- concentrations between sample with distance from the lake (Figure 5.3).

In comparison of connected and isolated sites, in 2004 and 2005 isolated sites had Cl^- concentrations in the same range or lower than concentrations in connected sites, with the exception of Loucks Pothole (LOUC), which had significantly greater Cl^- concentrations ($r^2 = 0.80$, $p < 0.0001$ in 2004; $r^2 = 0.57$, $p < 0.0001$ in 2005) compared to any of the connected or isolated sites (Figure 5.2b and c). Cl^- in concentrations in all isolated sites were slightly lower in 2005 compared to 2004 ($r^2 = 0.03$, $p = 0.0248$).

In the case of naturally occurring ^{18}O and ^{2}H isotope signatures in the lake and marsh, both $\delta^{18}O$ and $\delta^{2}H$ were significantly correlated ($n=40$, $r^2 = 0.96$, $p<0.0001$; Figure 5.4). As a result, δO_{18} signatures were used for comparison between sites and seasons. Spatially, mean seasonal $\delta\ ^{18}O$ signature values experienced little change with increasing distance from lake into the marsh, however the range in $\delta\ ^{18}O$ signature over the season increased with increasing distance (Figure 5.5a). The increased variation in $\delta^{18}O$ signature with increased distance from Lake Manitoba indicated a reduced influence of water exchange and mixing from Lake Manitoba with increasing distance from the lake, and resultantly less

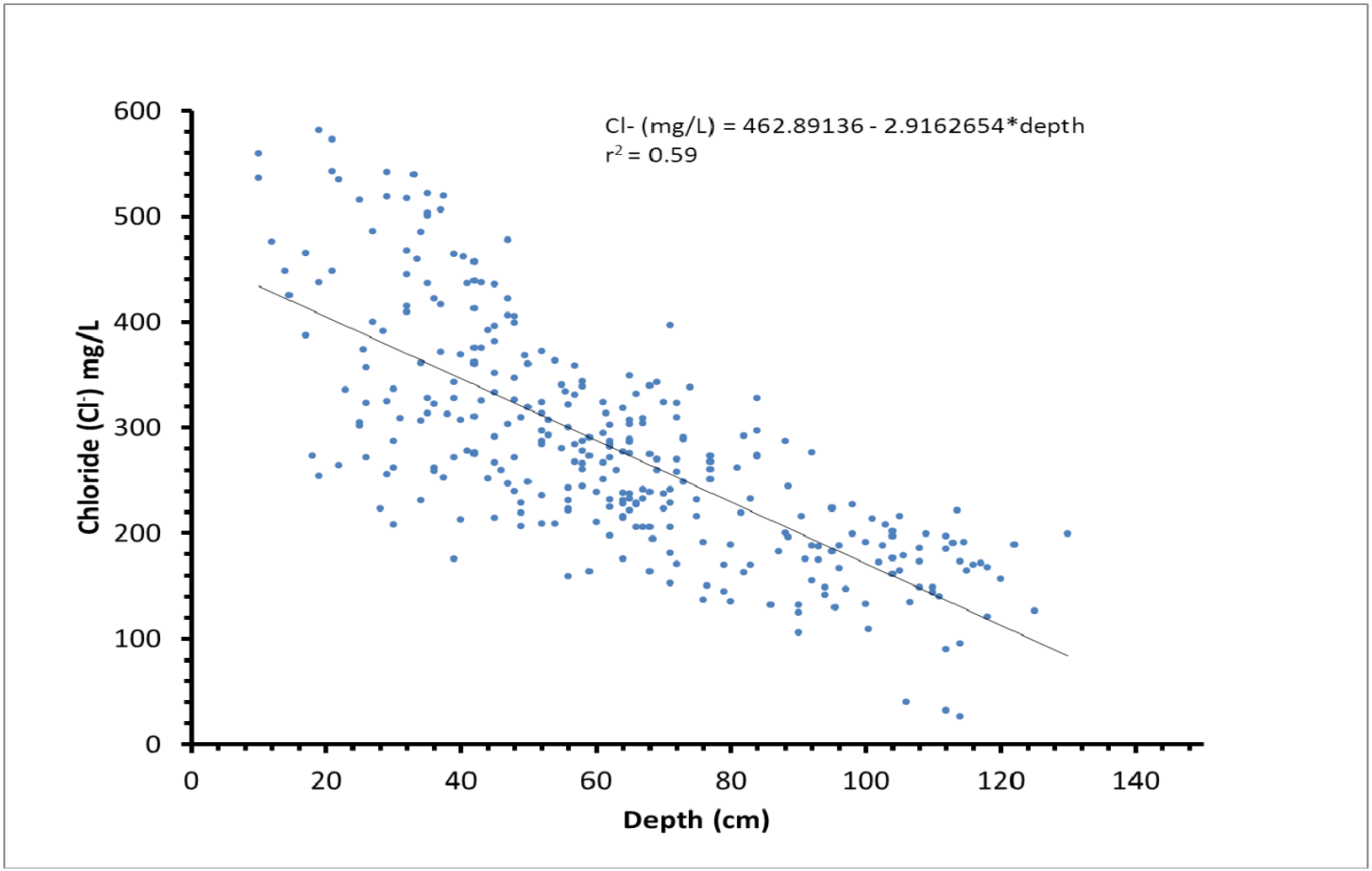

Figure 5.1 Linear regression of chloride (Cl^-) concentration (mg/L) plotted against water depth (cm) at marsh sites, from 2003 to 2005. (r^2= 0.59, $p < 0.0001$).

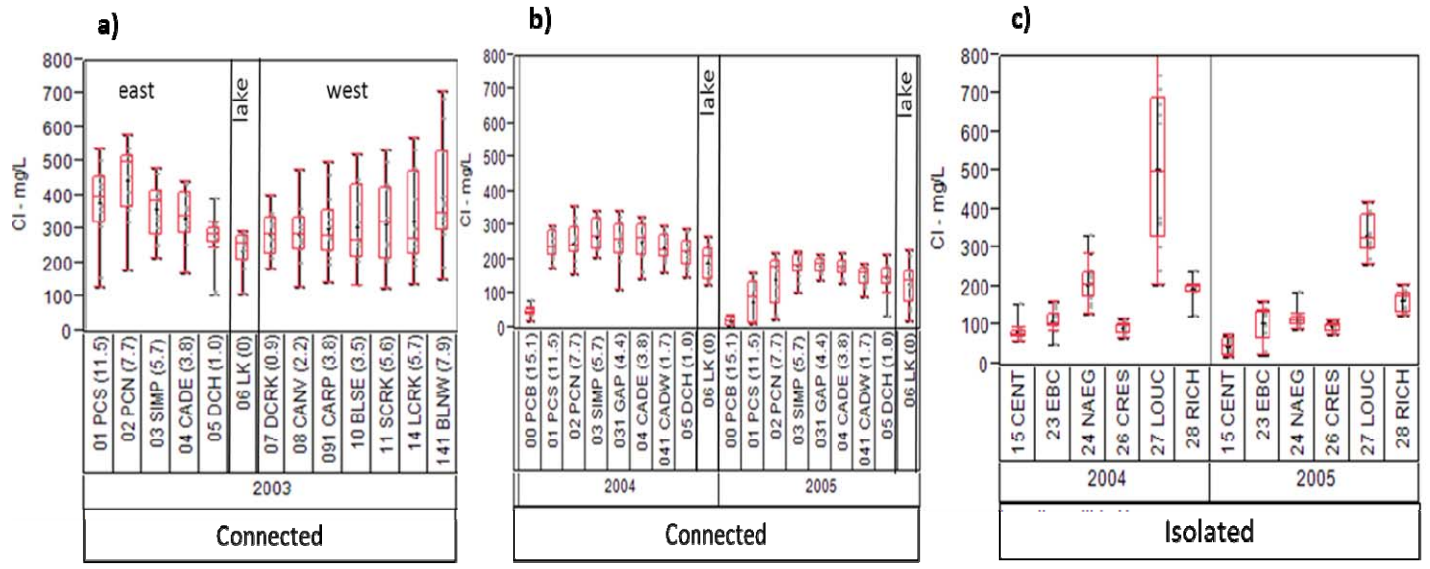

Figure 5.2 Mean and range of chloride (Cl^-) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1. Note no data are available for 2002.

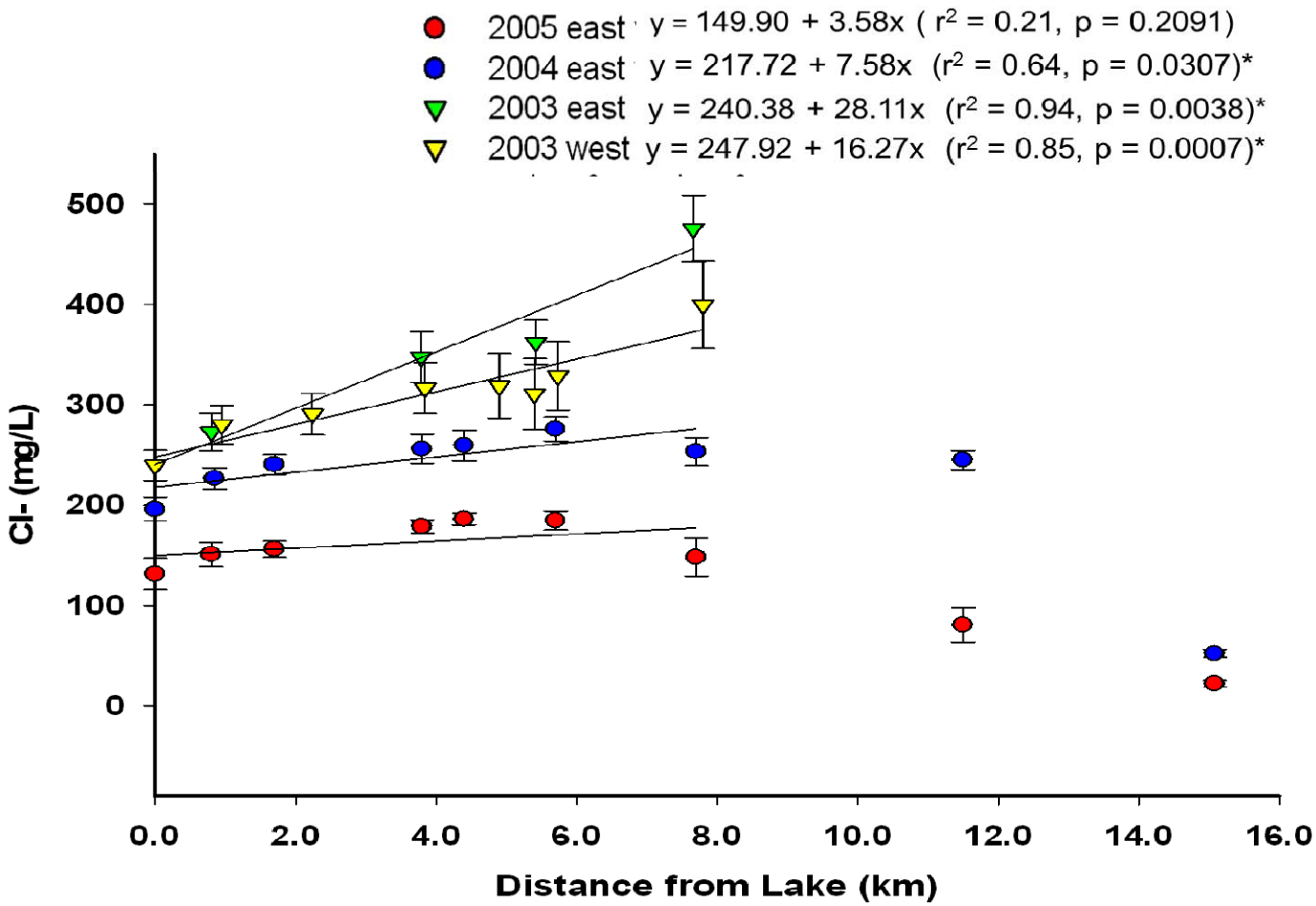

Figure 5.3 Mean chloride (Cl^-) concentrations (±SE) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km). Correlation equation, r^2 value and p value for each regression are reported. * denotes significant. Note Portage Creek South (11.5 km) and Portage Creek Bridge (15.1 km) sites along the east marsh transect have been removed from the regressions for 2004 and 2005. No data are available for 2002.

water mixing, and likely a greater influence of evaporation through the season at the far inland sites. Examination of the changes in $\delta^{18}O$ signature with distance from the lake by month (Figure 5.5c) further illustrates the large seasonal change at inland sites, with these sites becoming increasingly enriched in $\delta^{18}O$ from May to August due to reduced mixing with lake water and increased evaporation, with values increasing from -13‰ to -4 ‰. These sites also had $\delta^{18}O$ signature lower than the lake for the month of May (-13‰ to -11 ‰), indicating an increased influence of spring snowmelt and storm runoff from the upland landscape and less mixing at these sites during this time. Conversely, $\delta^{18}O$ signature in the lake, and marsh sites in close proximity to the lake (~ up to 5.7 km), experienced little change in range with distance from the lake, and with season from May to August, with values increasing only slightly from -10‰ to -8 ‰. When all marsh and lake sites were grouped (Figure 4.5b) the overall seasonal enrichment in $\delta^{18}O$ signature from May to August was apparent. These results indicated that ^{18}O and ^{2}H isotope signatures proved highly useful tracers of spatial and temporal water mass movement and mixing in the marsh. The relatively consistent oxygen isotope signature for the lake, as well as near-lake wetland sites both spatially and temporally, illustrated the high degree of lake and marsh water mass mixing at these sites. The high degree of variation in isotope signature of inland marsh sites showed reduced mixing and greater influence of precipitation via surface water and evaporation.

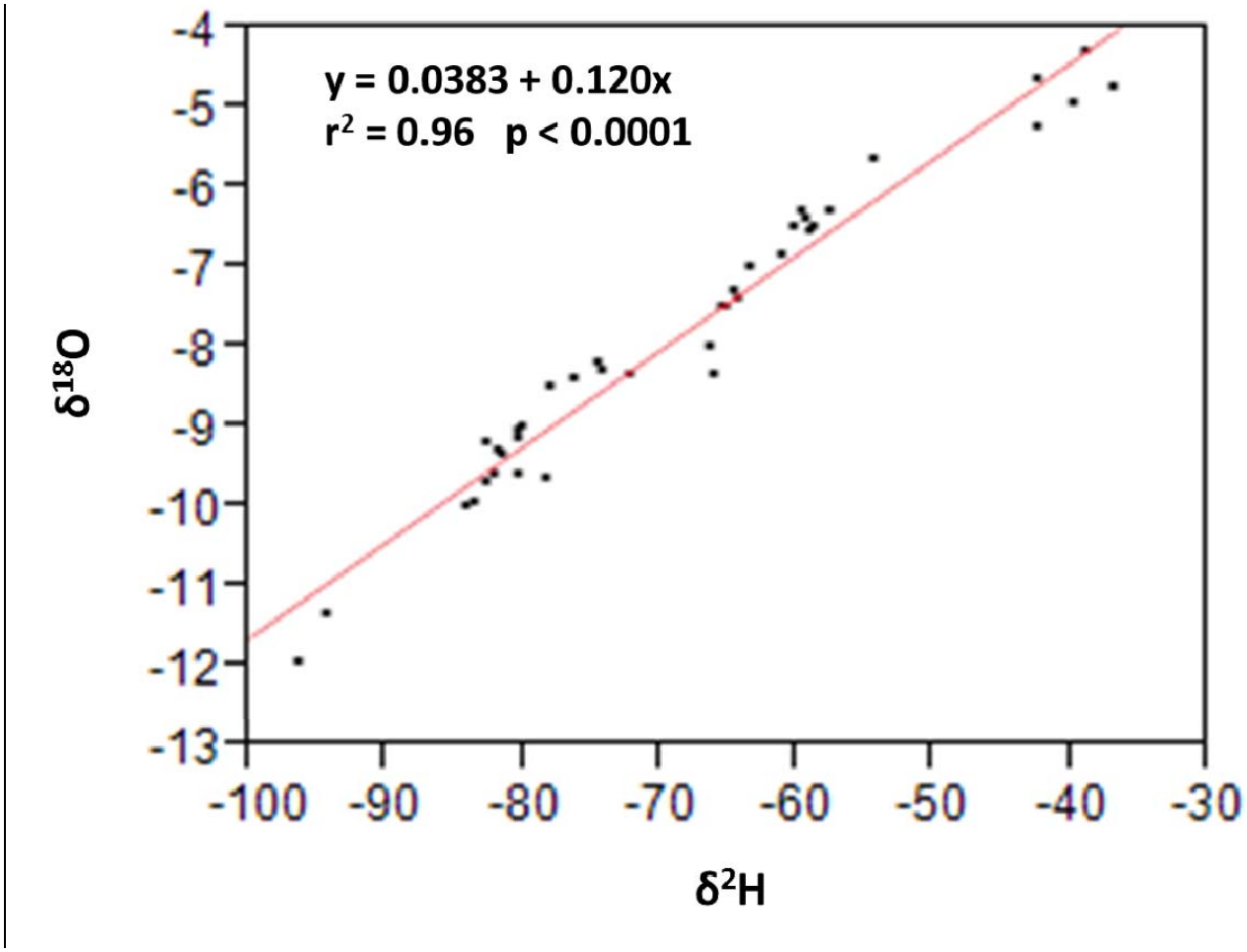

Figure 5.4 Relationship between oxygen-18 and hydrogen-2 (deuterium) isotopic signatures in Delta Marsh and Lake Manitoba, May to August 2005.

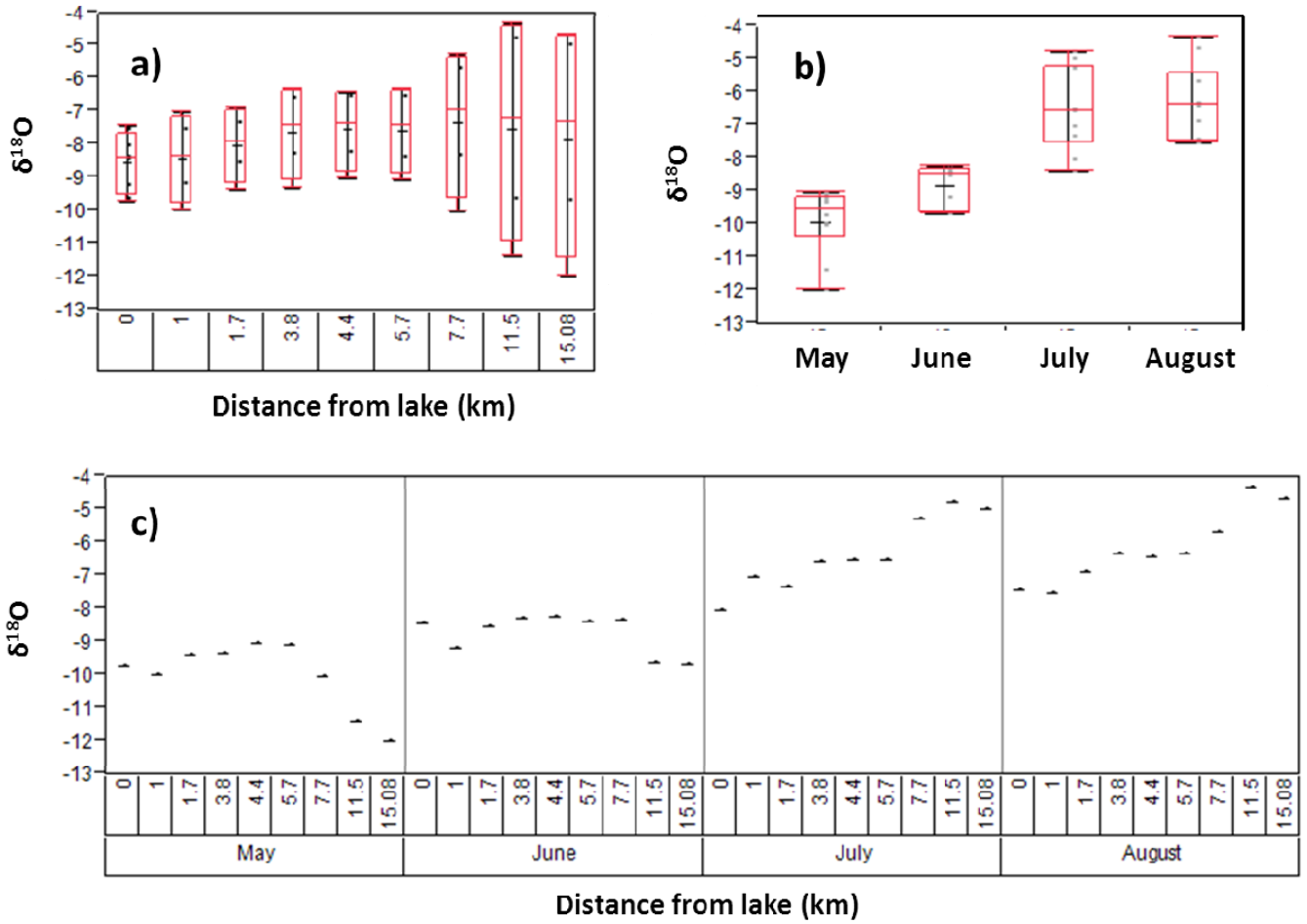

Figure 5.5 Variation in oxygen-18 isotope signatures in Delta Marsh, with (a) distance from Lake Manitoba (via surface water), (b) month, and (c) distance from the lake during each month, from May to August, 2005.

Water Chemistry

Water column N and P concentrations and molar ratios were highly variable in Delta Marsh. In the case of N concentrations (NO_3-N, NH_3-N DIN-N, TN), similar trends to those seen with Cl^- concentrations in the connected east section of the marsh in 2004 and 2005, occurred at the two most inland sites, Portage Creek South (PCS) and Portage Creek Bridge (PCB), with N concentrations at PCB and PCS lower than other nearby marsh sites, and as a result these two sites were removed from the regressions of N versus distance from the lake, resulting in improved fit (Figures 5.6 to 5.13).

In 2003, in the connected west section of the marsh mean NO_3-N concentrations did not change significantly with decreasing distance to Lake Manitoba, however in 2003 and 2004 in the east marsh concentrations of NO_3-N and decreased significantly with decreasing distance to the lake (approximately 50%; 2003: $r^2 = 0.83$, $p = 0.0110$; 2004: $r^2 = 0.80$, $p = 0.0060$). In 2005 NO_3-N concentrations did not decrease significantly with decreasing distance to the lake (Figure 5.6 and 5.7).

In 2003 in the west and east section of the marsh DIN-N concentrations decreased significantly (approximately 50%; $r^2 = 0.76$, $p = 0.0047$) with decreasing distance to the lake, and similarly in 2004 DIN-N concentrations also decreased significantly with decreasing distance to the lake in the east section of marsh (approximately 80%; $r^2 = 0.89$, $p = 0.0012$). In 2005 DIN-N concentrations did not decrease significantly with decreasing distance to the lake (Figure 5.8 and 5.9).

In 2002 NH_3-N concentrations in the west and east sections of the marsh did not change significantly with distance, however in 2003 mean NH_3-N concentrations decreased significantly in both sections with decreasing distance to the lake (approximately 80%; east: $r^2 = 0.77$, $p = 0.0213$; west: $r^2 = 0.89$, $p = 0.0012$) (Figure 5.10 and 5:11). In 2004 in the east section of the marsh, mean NH_3-N concentrations decreased significantly (approximately 80%) with decreasing distance to the lake ($r^2 = 0.97$, $p < 0.0001$). In 2005 NH_3-N concentrations did not change significantly with decreasing distance to the lake (Figures 5.10 and 5.11).

TN concentrations did not change significantly with distance to the lake in the east section of the marsh in 2002 and 2005 (Figure 5.12 and 5.13). However the trend with distance was significant in the west marsh in 2002 and 2003, and the east marsh in 2003 and 2004, with concentrations decreasing approximately 30 to 50% with decreasing distance to the lake (west 2002: $r^2 = 0.99$, $p = 0.0003$; west 2003: $r^2 = 0.89$, $p = 0.0002$; east 2003: $r^2 = 0.60$, $p = 0.0453$; east 2004: $r^2 = 0.79$, $p = 0.0044$) (Figure 5.12 to 5.13).

Annually, in the east section of the marsh, the decreases in the concentrations of NO_3-N, NH_3-N, DIN-N and TN with decreasing distance to Lake Manitoba were greatest 2003 and 2004 under lower summer water levels, compared to higher water levels in 2005. Similarly, the range in mean NO_3-N, NH_3-N, DIN-N and TN concentrations at east marsh sites were also higher in 2003 and 2004 compared to 2005, as mean NO_3-N concentrations ranged from 0.05 to 0.35 mg/L, NH_3-N from 0.05 to 0.50 mg/L, DIN-N concentrations from 0.30 to 0.80 mg/L, and TN from 8.0 to 13.9 mg/L in both 2003 and 2004, while in

2005 mean NO_3-N concentrations were below 0.20 mg/L, NH_3-N below 0.05 mg/L, DIN-N ranged from 0.21 to 0.26 mg/L, and TN from 3.3 to 5.9 mg/L in 2005 (Figure 5.8 to 5.13).

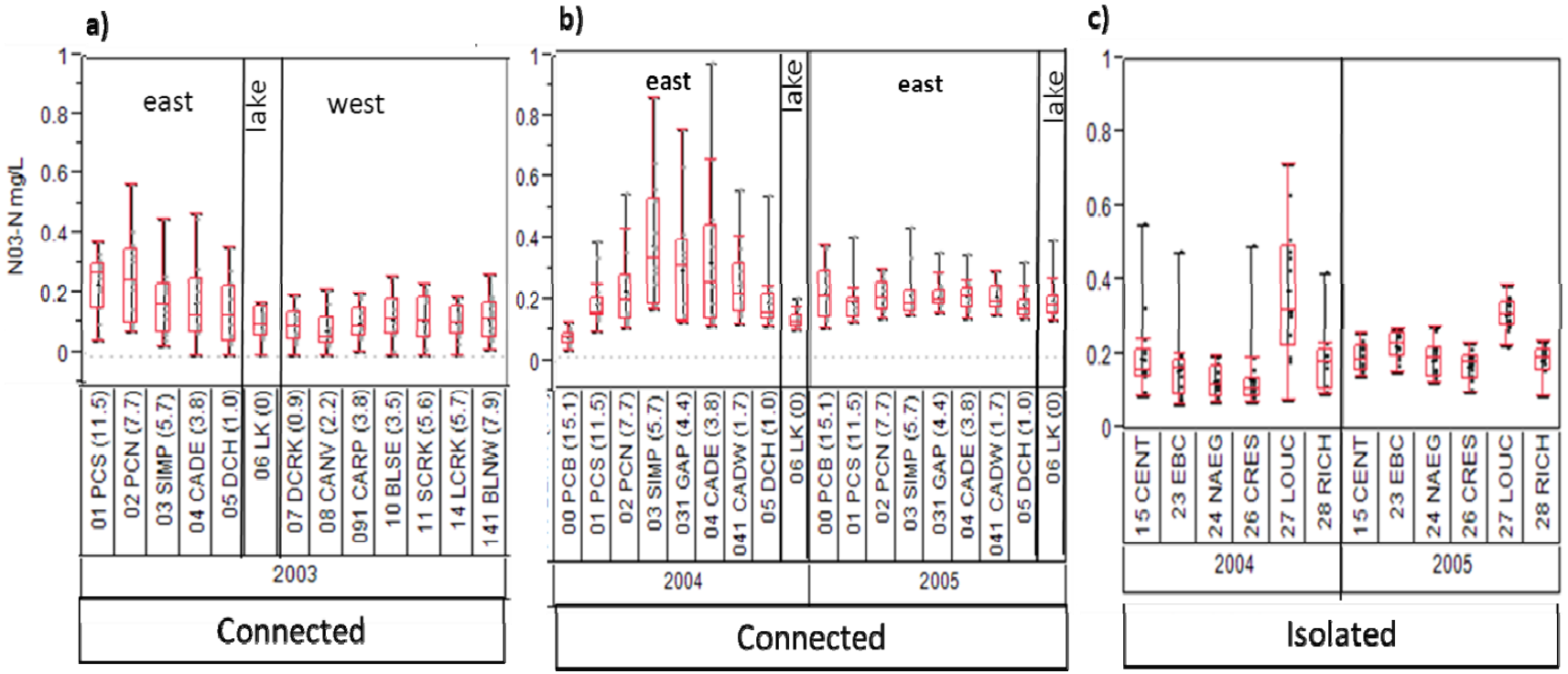

Figure 5.6 Mean and range of nitrate (NO_3-N) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1. Note no data are available for 2002.

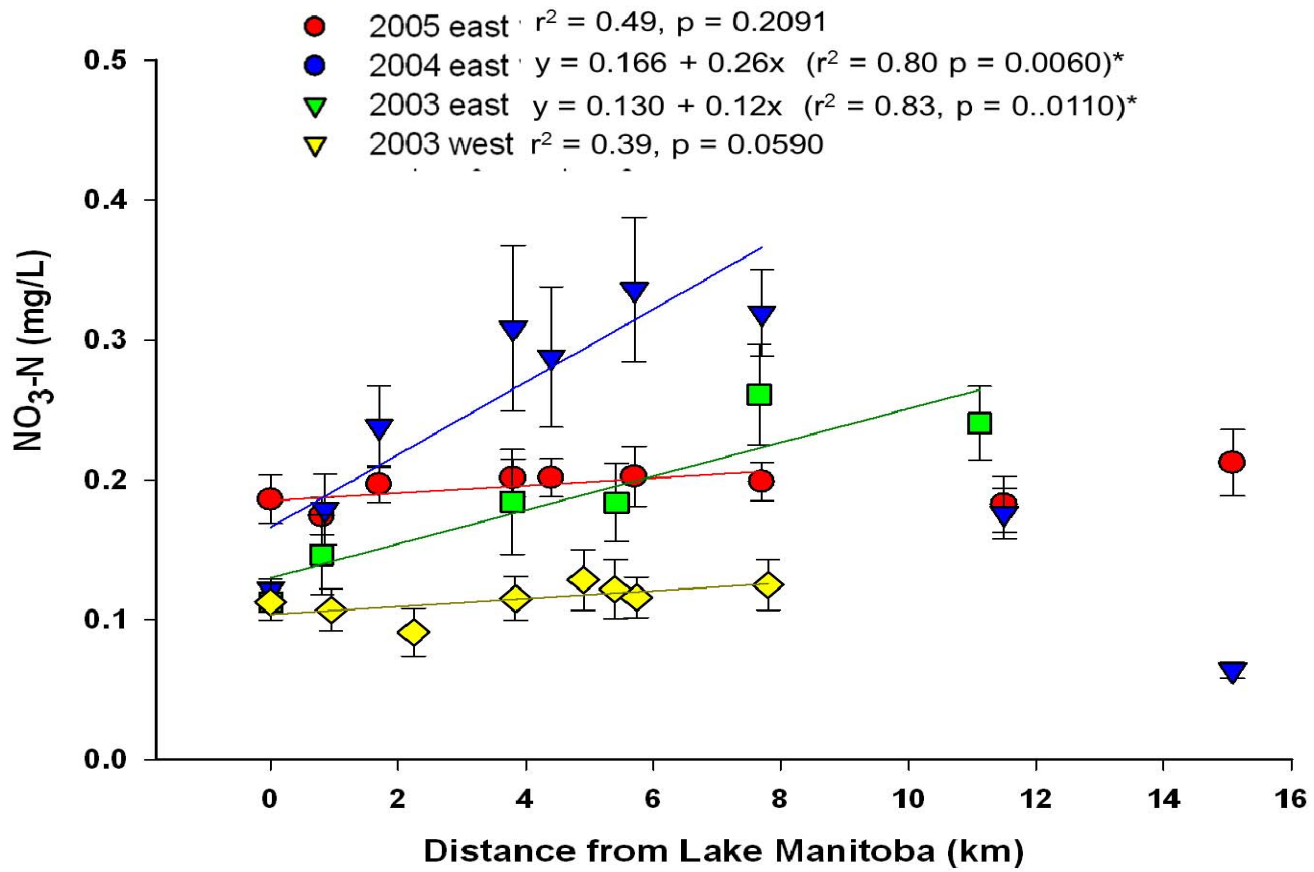

Figure 5.7 Mean ±SE NO_3-N concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant. Note Portage Creek South (11.5 km) and Portage Creek Bridge (15.1 km) sites along the east marsh transect have been removed from the regressions for 2004 and 2005. No data are available for 2002.

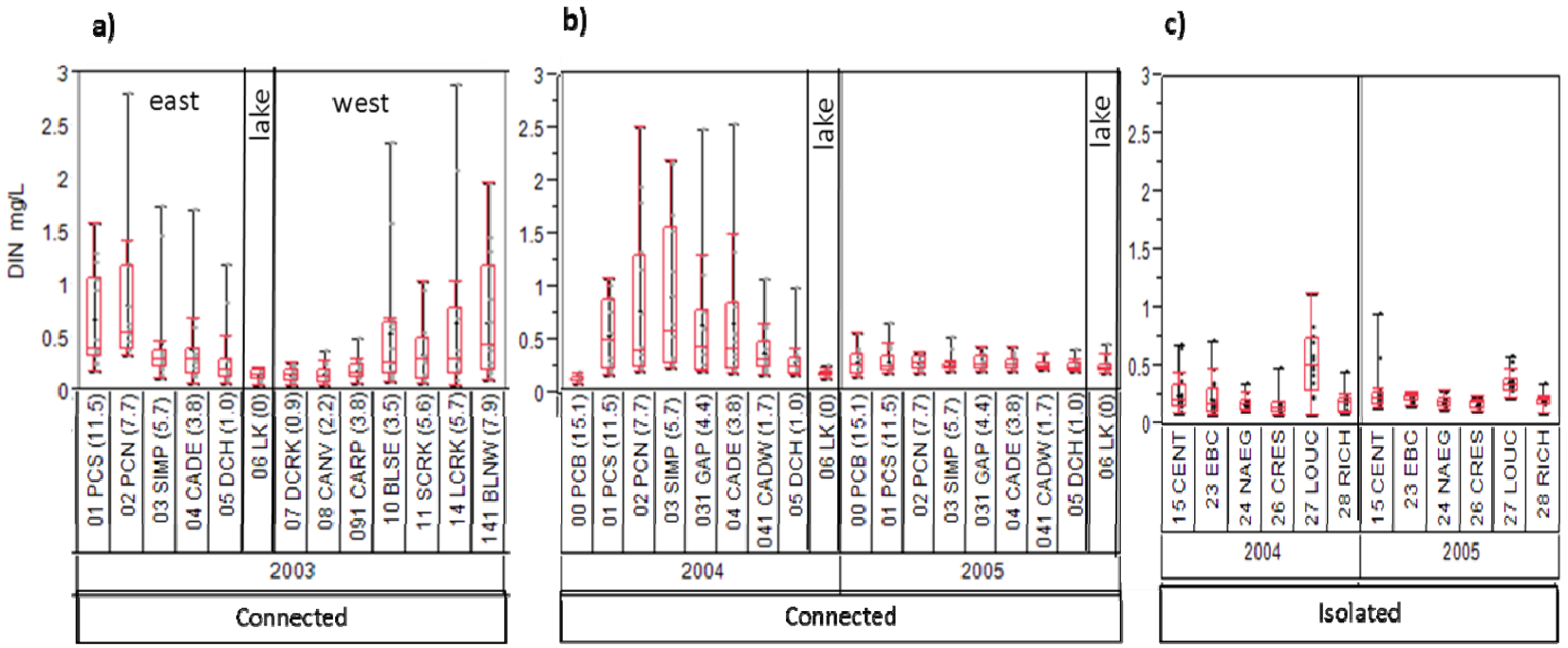

Figure 5.8 Mean and range of dissolved inorganic nitrogen (DIN) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

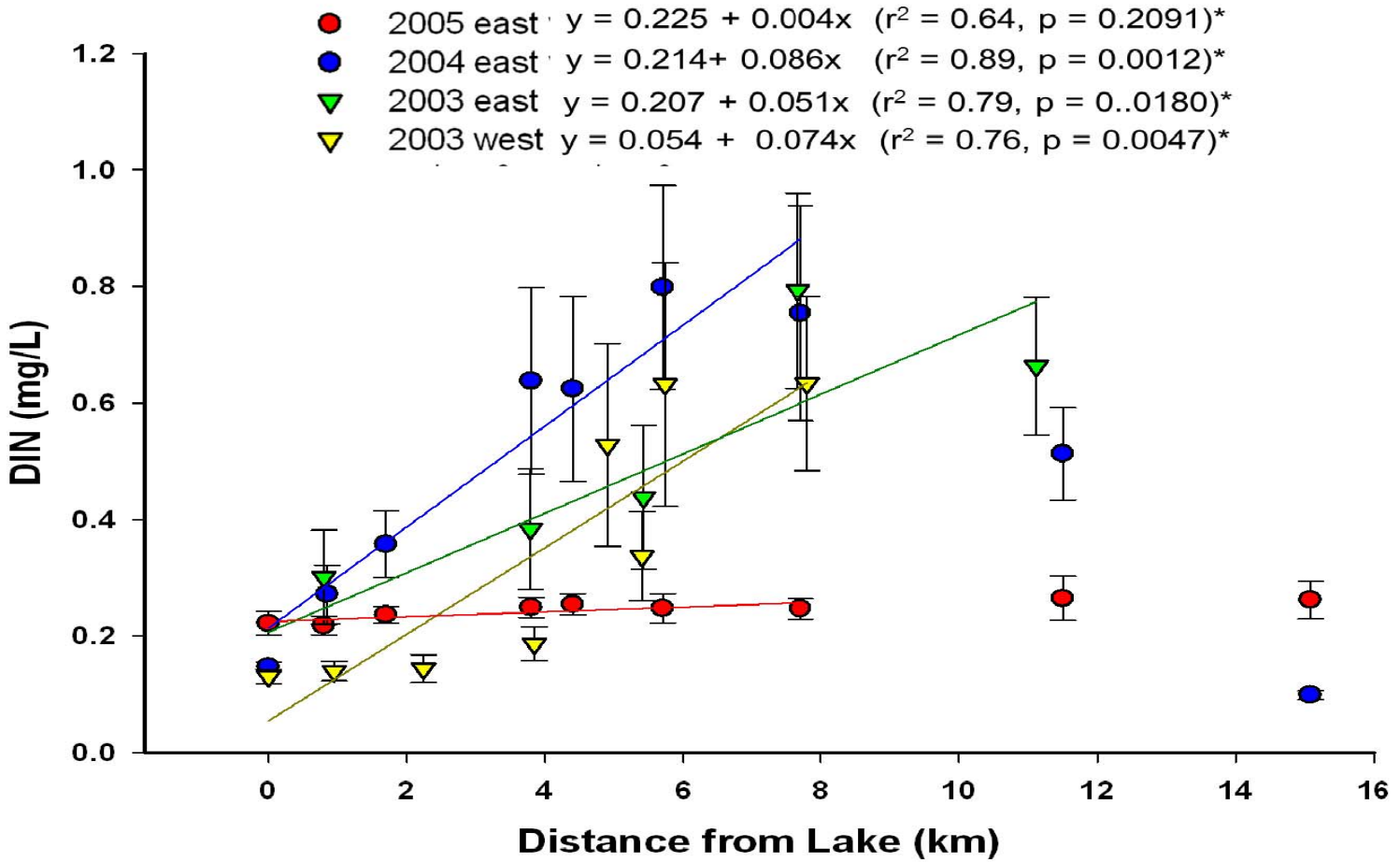

Figure 5.9 Mean ±SE dissolved inorganic nitrogen (DIN) concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant. Note Portage Creek South (11.5 km) and Portage Creek Bridge (15.1 km) sites along the east marsh transect have been removed from the regressions for 2004 and 2005. No data are available for 2002.

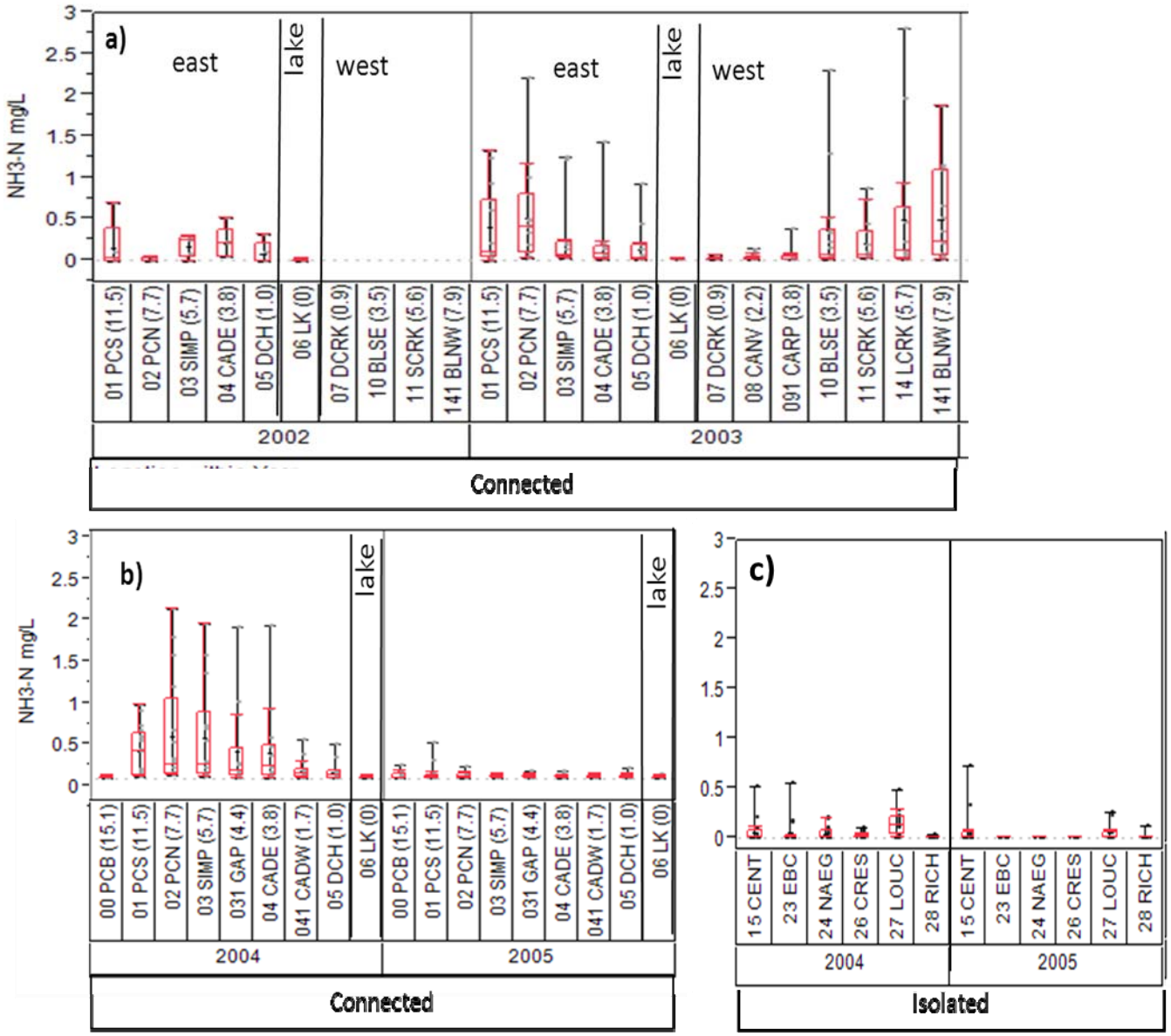

Figure 5.10 Mean and range of ammonia (NH_3-N) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1. Note no data are available for west marsh in 2002.

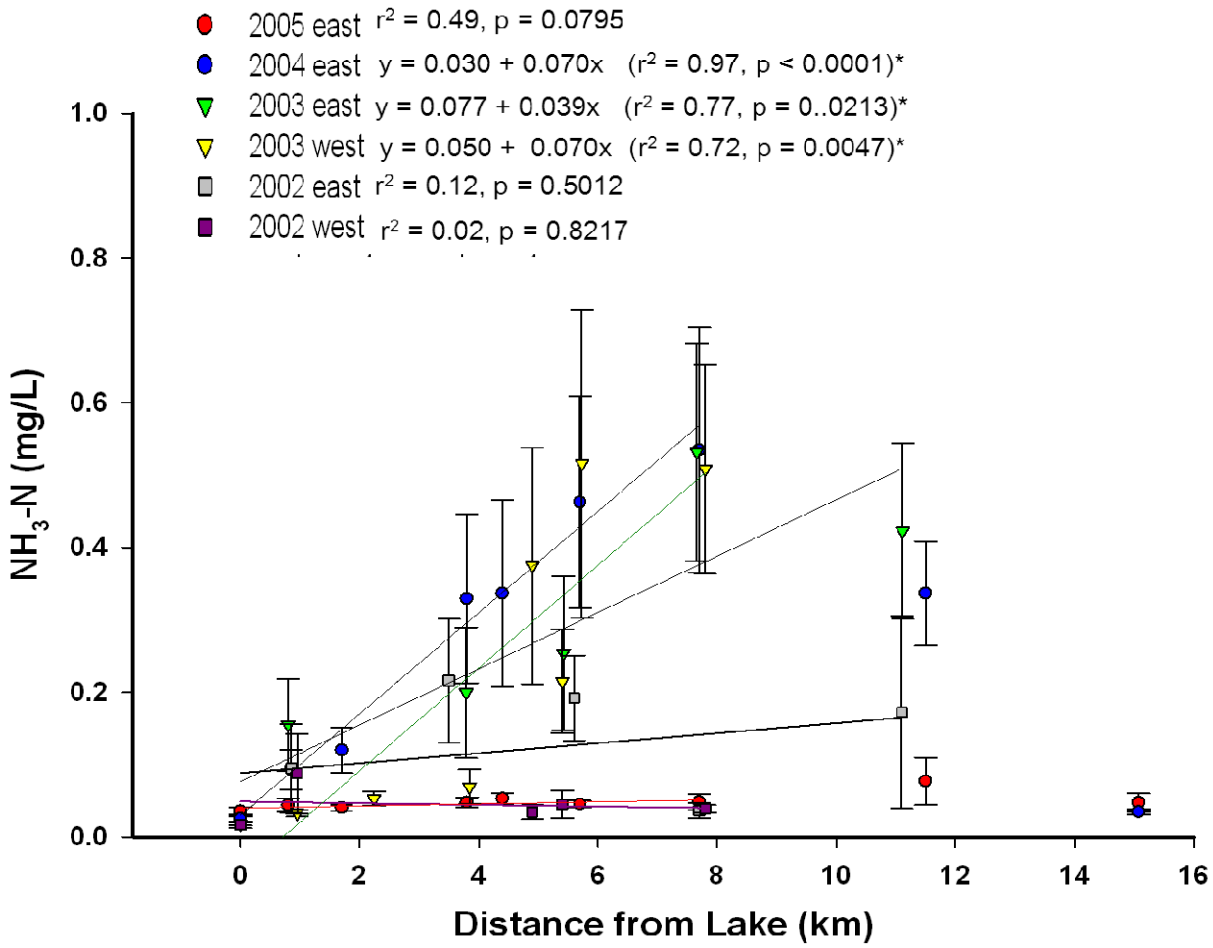

Figure 5.11 Mean ±SE ammonia (NH_3-N) concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2002 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant. Note Portage Creek South (11.5 km) and Portage Creek Bridge (15.1 km) sites along the east marsh transect have been removed from the regressions for 2004 and 2005.

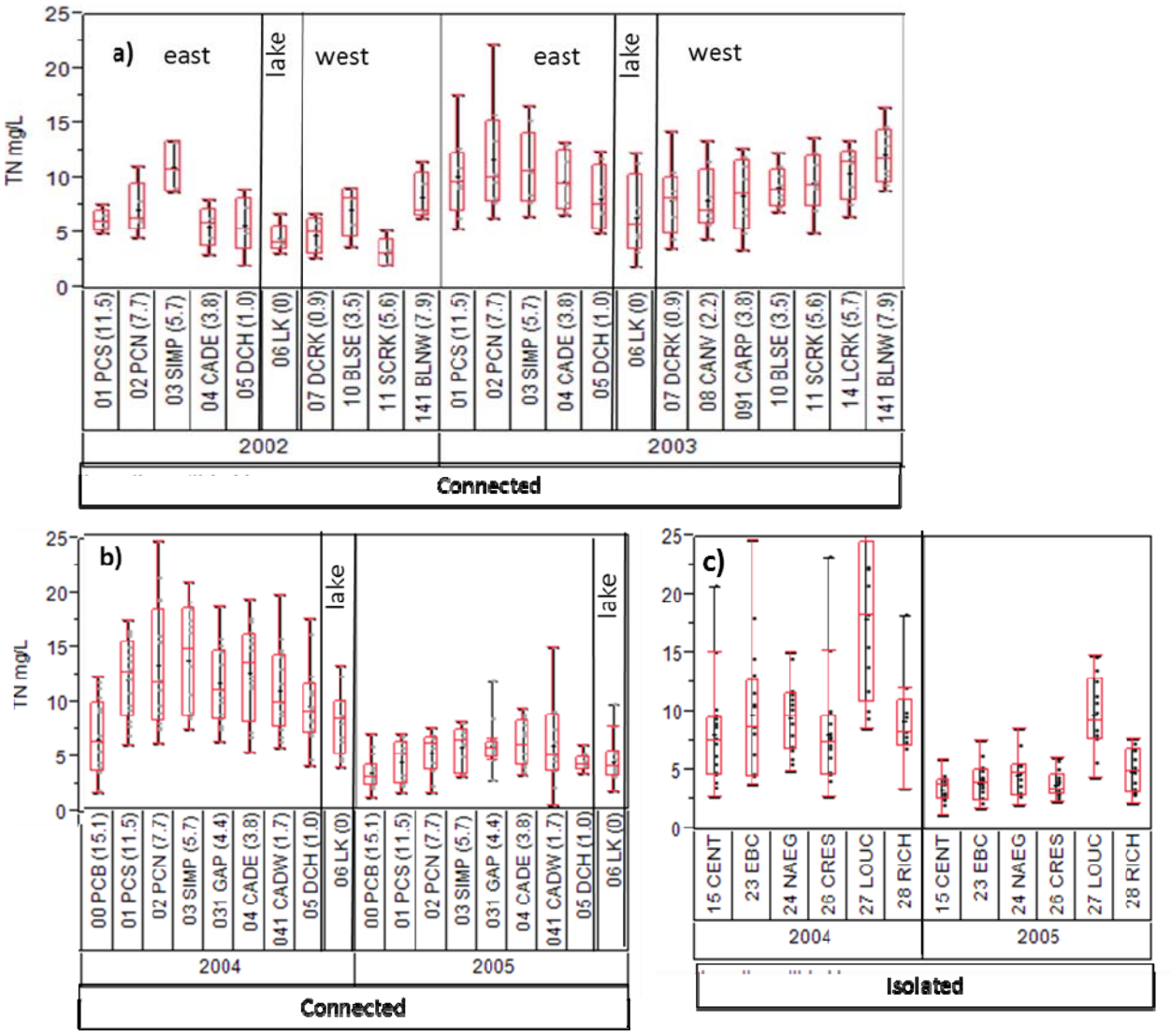

Figure 5.12 Mean and range of total nitrogen (TN) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

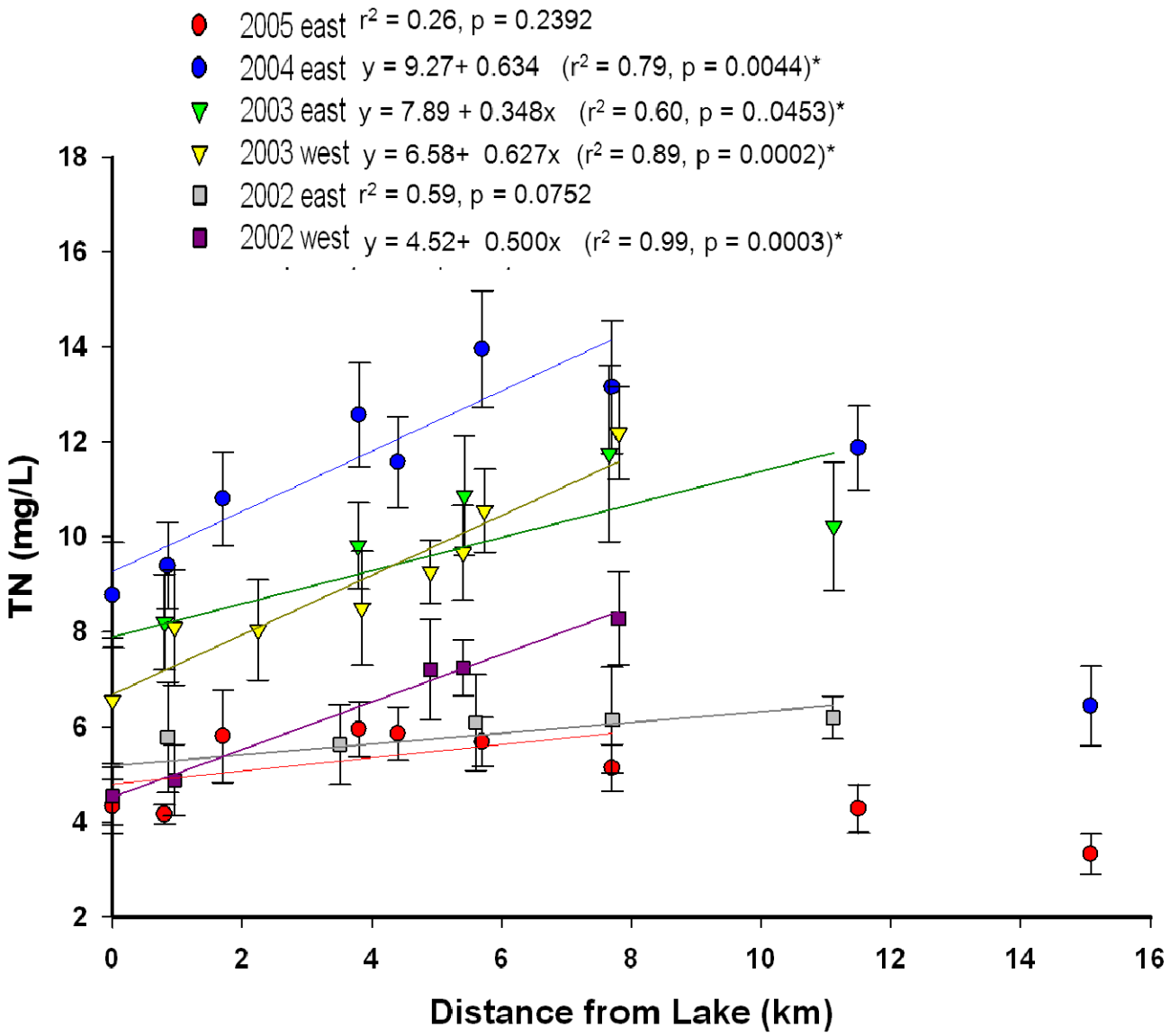

Figure 5.13 Mean ±SE total nitrogen (TN) concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant. Note Portage Creek South (11.5 km) and Portage Creek Bridge (15.1 km) sites along the east marsh transect have been removed from the regressions for 2004 and 2005.

In 2002, only TP concentrations in the east section of the marsh decrease significantly with decreasing distance to the lake (approximately 80%; $r^2 = 0.75$, $p = 0.0252$), whereas there was no significant trend in TRP-P concentrations in both the east and west, and TP concentrations in the west (Figure 5.14 to 5.17).

In 2003, TRP-P and TP concentrations decreased significantly with decreasing distance to the lake in both east and west sections of the marsh in 2003 (approximately 70 to 95%; TP east: $r^2 = 0.87$, $p = 0.0059$; TP west: $r^2 = 0.68$, $p = 0.0112$; TRP-P east: $r^2 = 0.83$, $p = 0.0109$; TRP-west: $r^2 = 0.79$, $p = 0.0074$), with the exception of the furthest site in the west section of the marsh, Big Lake north west (BLNW), where TRP-P and TP concentrations were lower than Short Creek (SCRK; Figures 5.14 to 5.17).

In 2004 and 2005, TRP-P and TP decreased significantly with decreasing distance to the lake in the east section of the marsh, with PCS and PCB include in the regressions (approximately 75 to 90%; 2004 TRP-P: $r^2 = 0.73$, $p = 0.0032$; 2004 TP: $r^2 = 0.97$, $p < 0.0001$; 2005 TRP-P: $r^2 = 0.87$, $p = 0.0002$; 2005 TP: $r^2 = 0.87$, $p = 0.0002$) (Figure 5.14 to 5.17). Both PCS and PCB were included in the regressions, as TRP-P and TP concentrations at these sites fit the regression lines and did not decrease the significance of the trends.

Annually the decrease in TRP-N and TP concentrations with decreasing distance to the lake was slightly greater in during the lower water years in 2002 and 2003 compared to higher water level conditions in 2004 and 2005. Further, the range in mean TP and TRP-P concentrations were also slightly higher in 2002 and 2003, in comparison to 2004 and 2005. Mean TRP-P concentrations ranged from 0. 01to 0.91 mg/L and TP from 0.11 to 1.2 mg/L in 2002 and 2003,

and mean TRP-P ranged from 0.01 to 0.77 mg/L and TP from 0.02 to 0.91 mg/L in 2004 and 2005 (Figure 5.14 to 5.17).

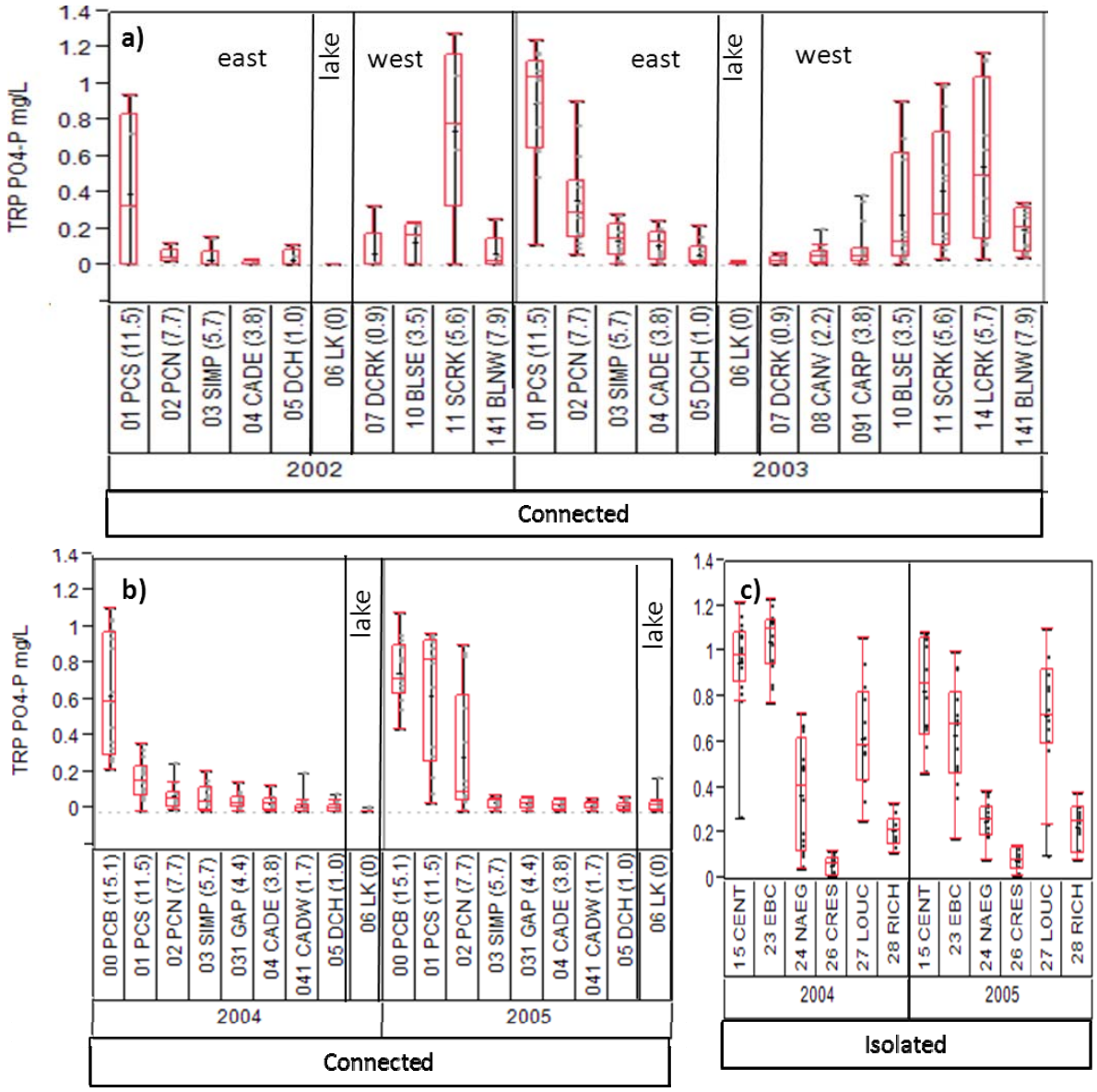

Figure 5.14 Mean and range of total reactive orthophosphate (PO_4-P) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004

and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

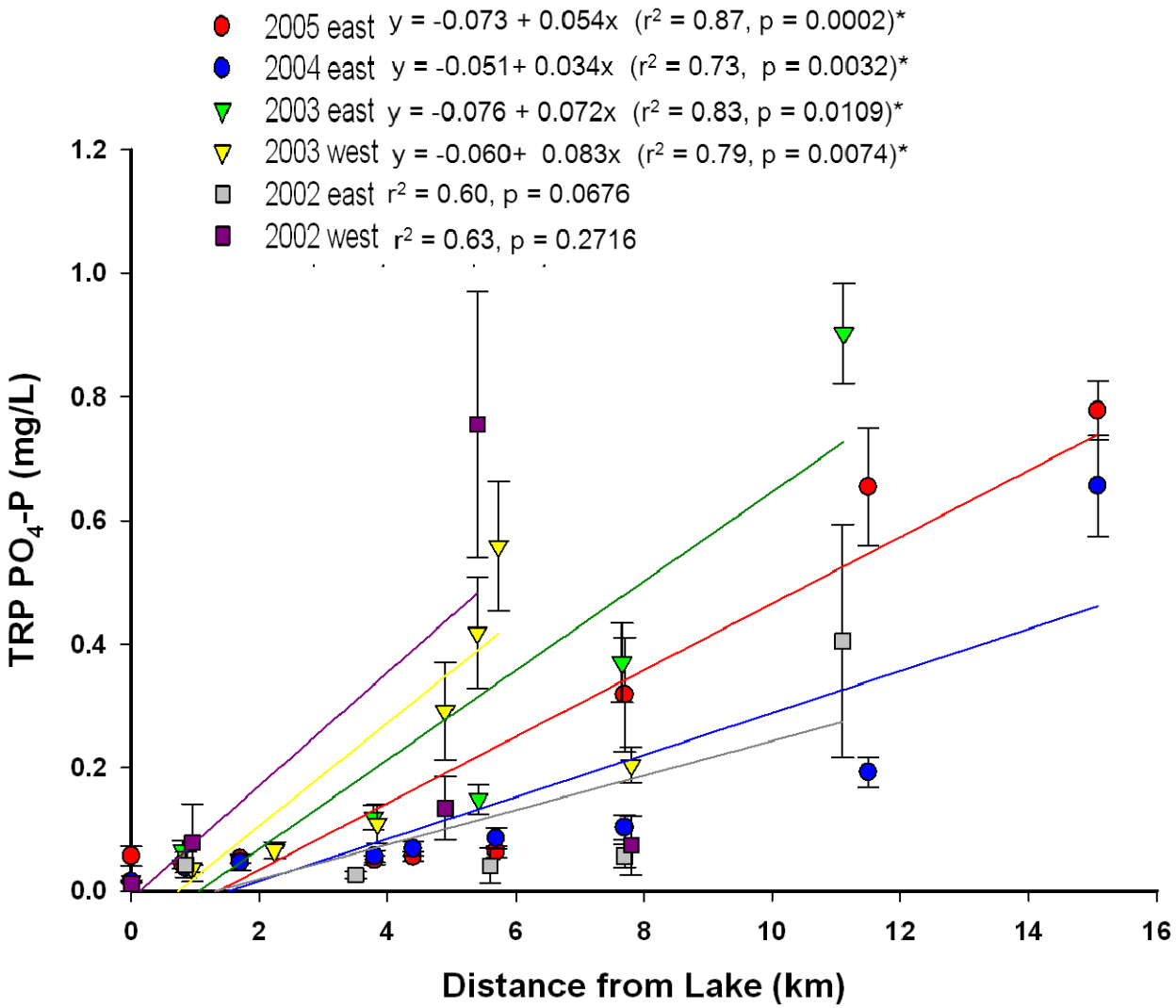

Figure 5.15 Mean ±SE total reactive orthophosphate (TRP) concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

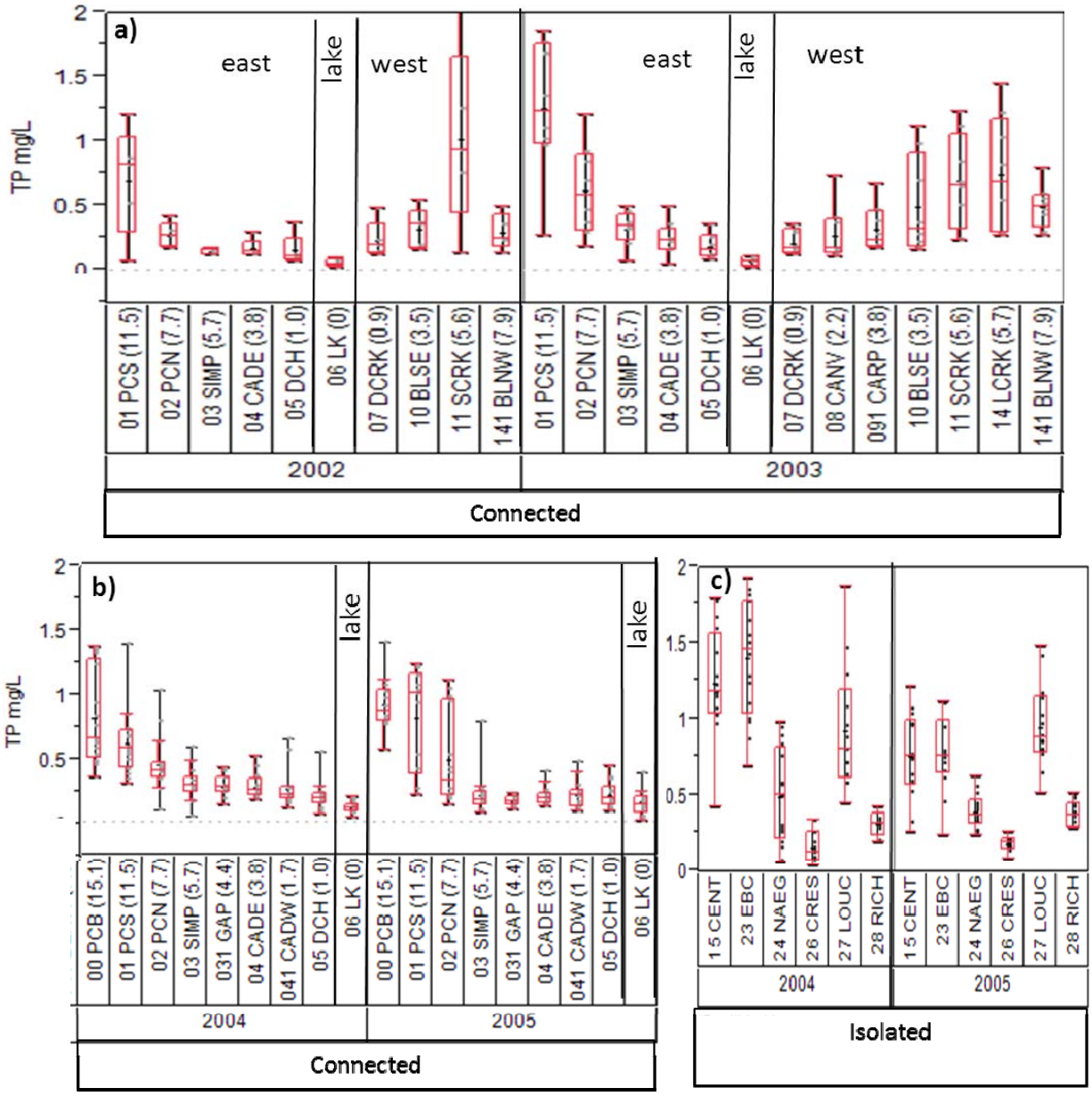

Figure 5.16 Mean and range of total phosphorus (TP) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

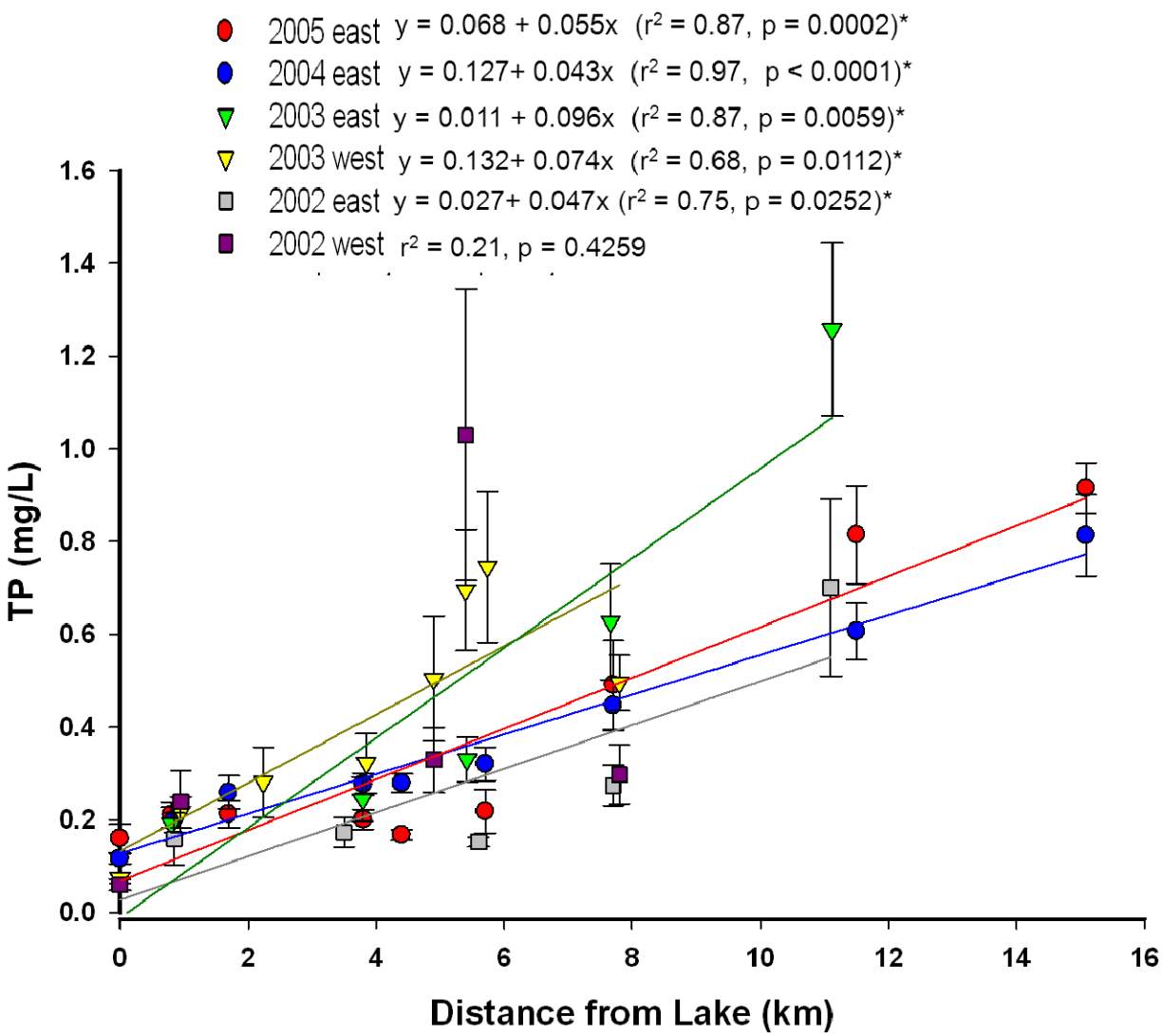

Figure 5.17 Mean ±SE total phosphorus concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

The greater decreases in TRP-P and TP concentrations compared with TN and DIN-N concentrations along the marsh transacts with decreasing distance to the lake in all study years, resulted in significant increases in the molar ratios of TN:TP, DIN:TRP, and DIN:TP along the transects with decreased distance to the lake; with ratios more than doubling in all cases (Figure 5.18 to 5.23). In the case of TN:TP, the gradient was most significant in the lower water years of 2003 and 2004, and became reduced in the higher water level of 2005. In 2004 in the east section of the marsh, and 2003 in both the east and west sections of the marsh, mean TN:TP ratios increased from 20 to 130 with decreasing distance to Lake Manitoba, whereas in 2005 in mean TN:TP ratios increased from 9 to 80 with decreasing distance to the lake. In the case of DIN:TRP the gradient was most significant in 2003 and 2004, with mean DIN:TRP ratios increasing from 0.5 to 25 with decreasing distance to the lake (Figure 5.20 and 5.21). In the case of DIN:TP the gradient was most significant in 2004 and 2005, with mean DIN:TP ratios increasing from 0.5 to 6 with decreasing distance to the lake (Figure 5.22 and 5.23). Both PCS and PCB were included in TN:TP, DIN:TRP, and DIN:TP versus distance regressions, as ratios at these sites fit the regression lines and did not decrease the significance of the trends.

Sulfate (SO_4^-), concentrations were the highest in east marsh in 2004, and concentrations decreased significantly from 567 mg/L to 195 mg/L with decreasing distance to the lake in 2004 in the east section of the marsh, and in 2003 in the west section of the marsh from 377 to 143 mg/L (Figure 5.24 and 5.25). In 2005, under high water levels, SO_4^- concentrations were lowest of all study years at all study sites, and further there was little variation in SO_4^- with distance from the lake.

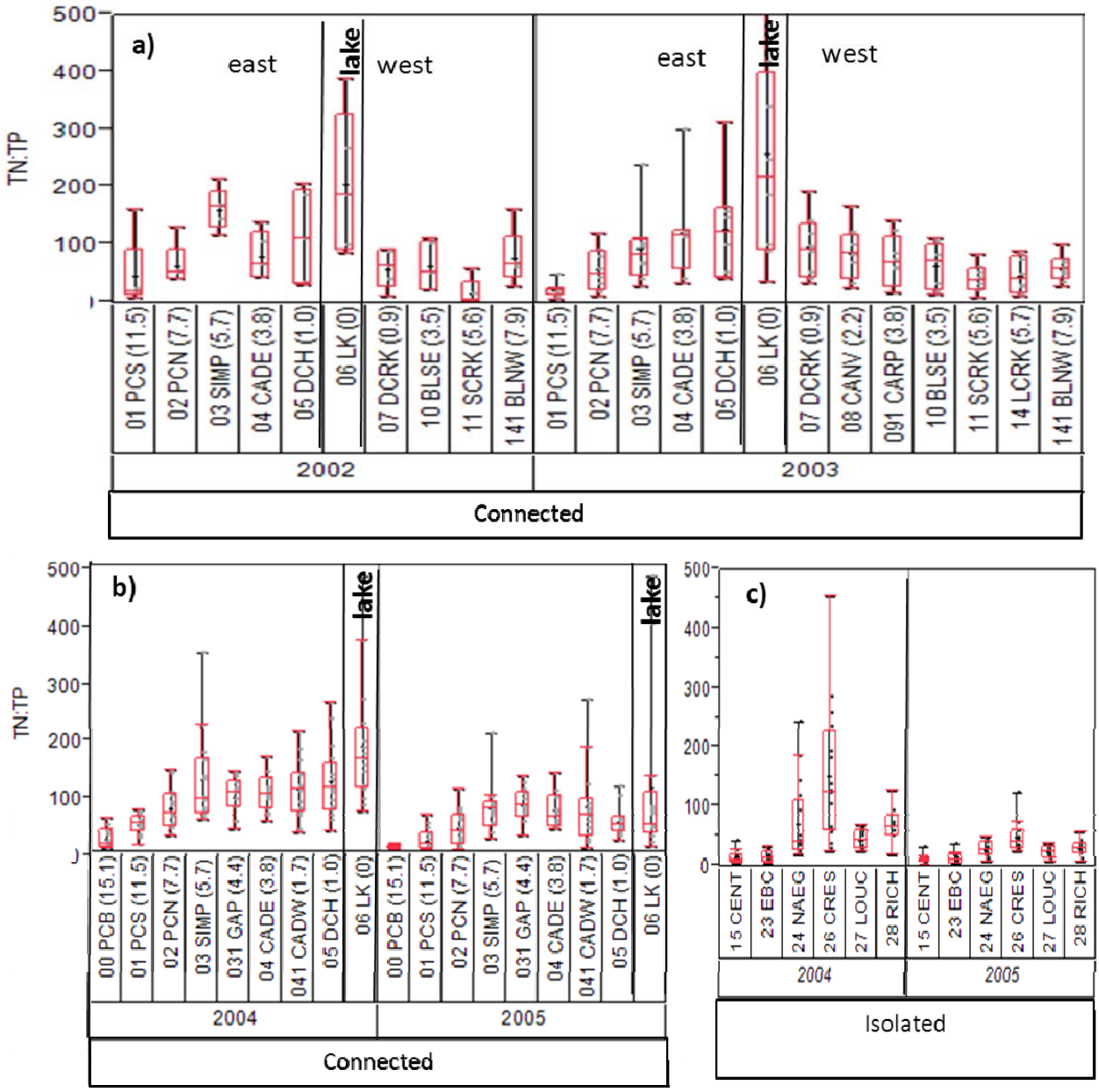

Figure 5.18 Mean and range of molar ratio of total nitrogen to total phosphorus (TN:TP) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

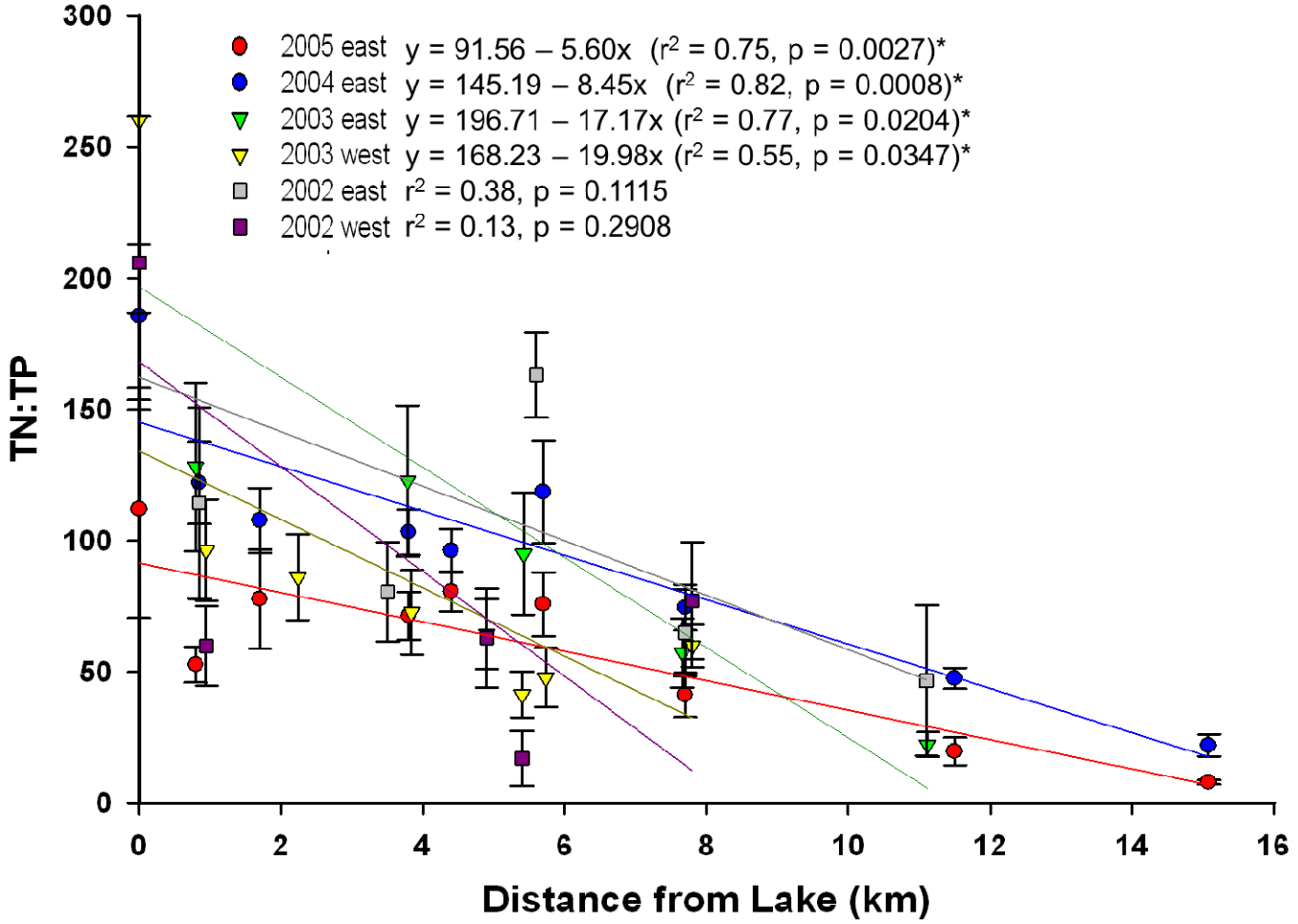

Figure 5.19 Mean ±SE total nitrogen to total phosphorus (TN:TP) molar ratios in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

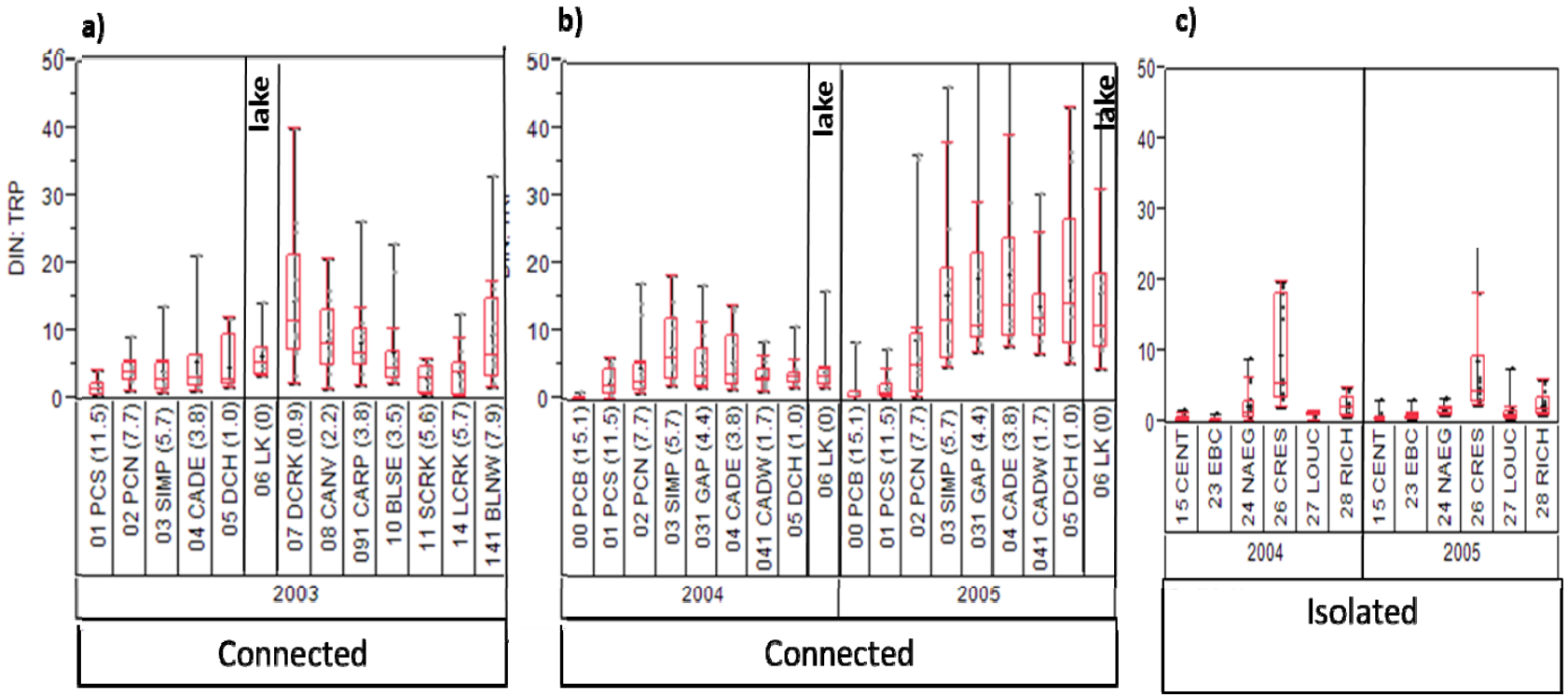

Figure 5.20 Mean and range of molar ratio of dissolved inorganic nitrogen to total reactive orthophosphate (DIN:TRP) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1. Note no data are available for 2002.

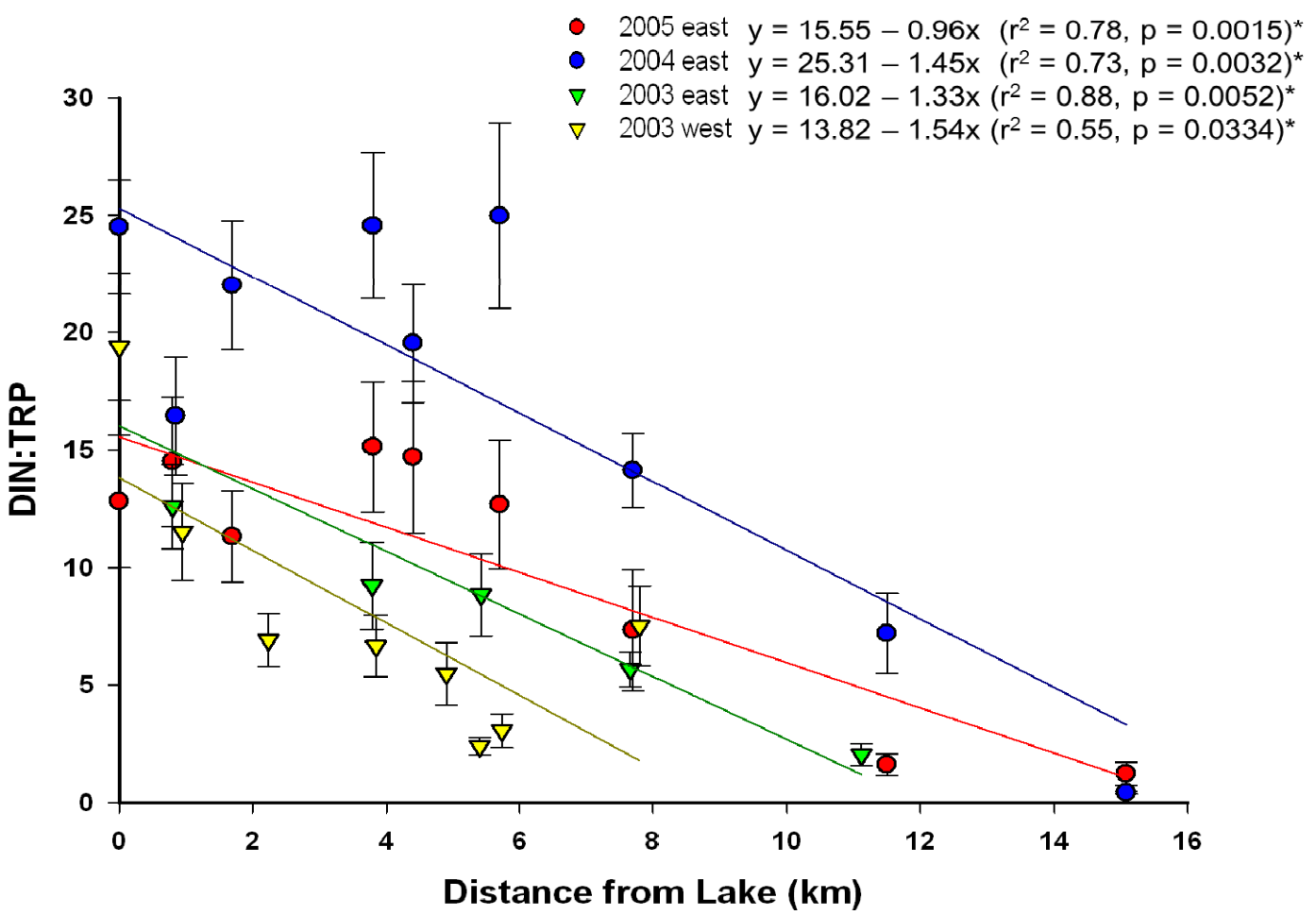

Figure 5.21 Mean ±SE dissolved inorganic nitrogen to total reactive nitrogen (DIN:TRP) molar ratios in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant. No data are available for 2002.

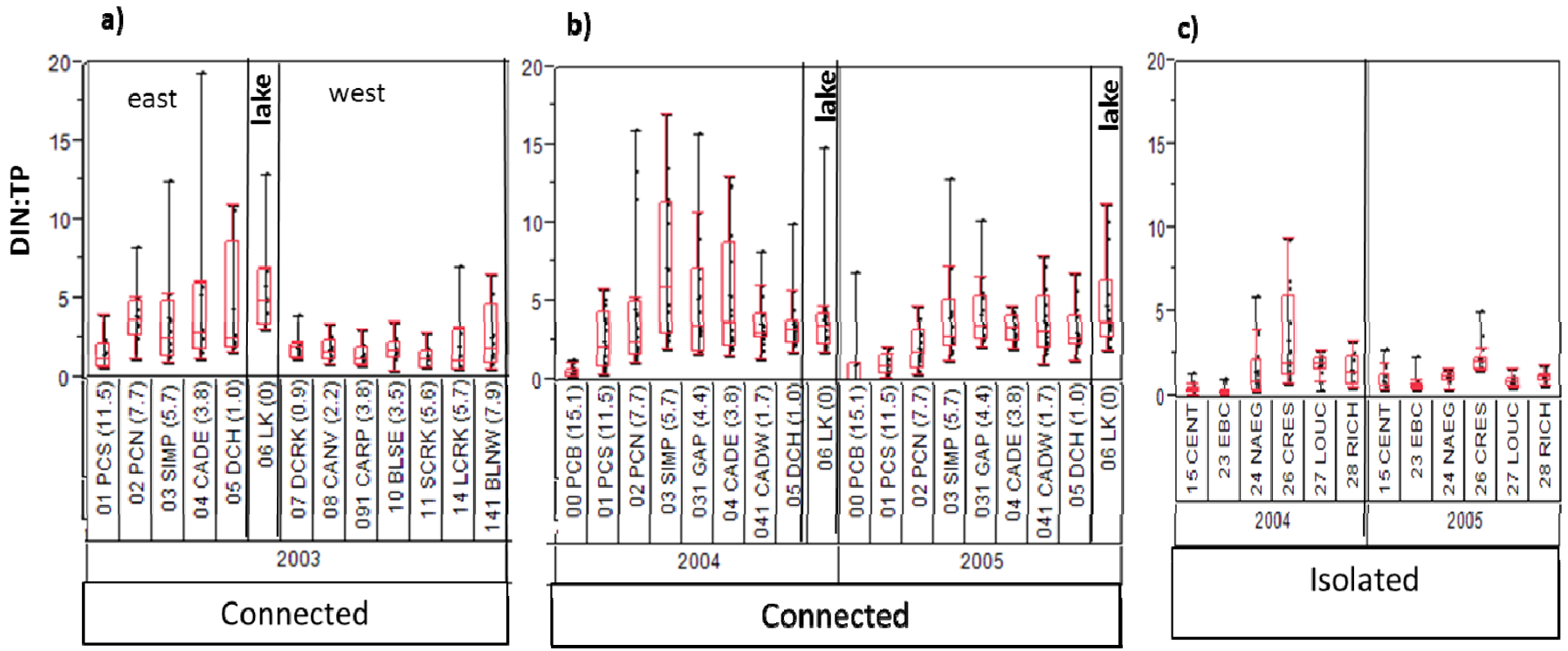

Figure 5.22 Mean and range of molar ratio of dissolved inorganic nitrogen to total phosphorus (DIN:TP) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1. Note no data are available for 2002.

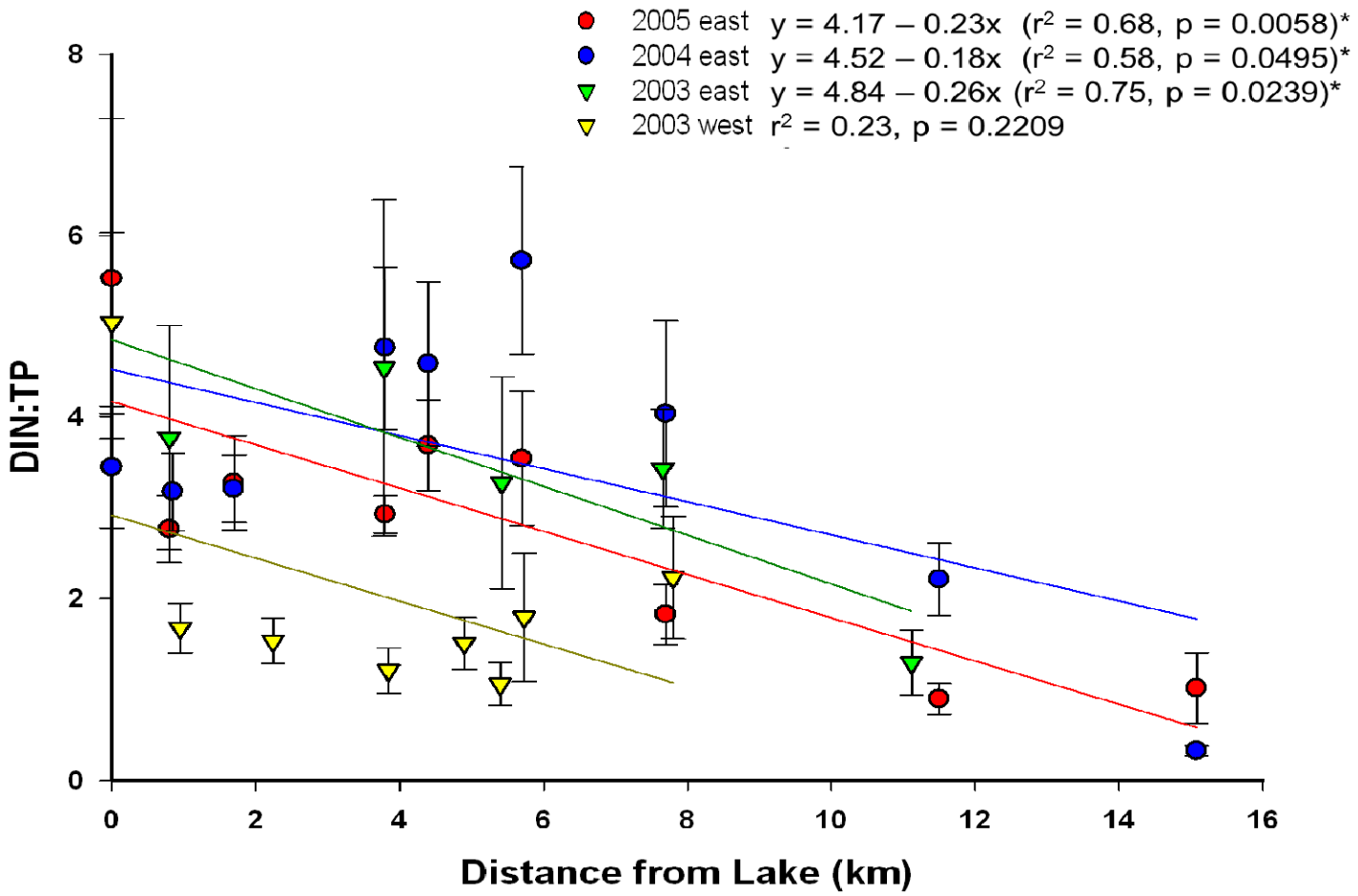

Figure 5.23 Mean ±SE dissolved inorganic nitrogen to total phosphorus (DIN:TP) molar ratios in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant. No data are available for 2002.

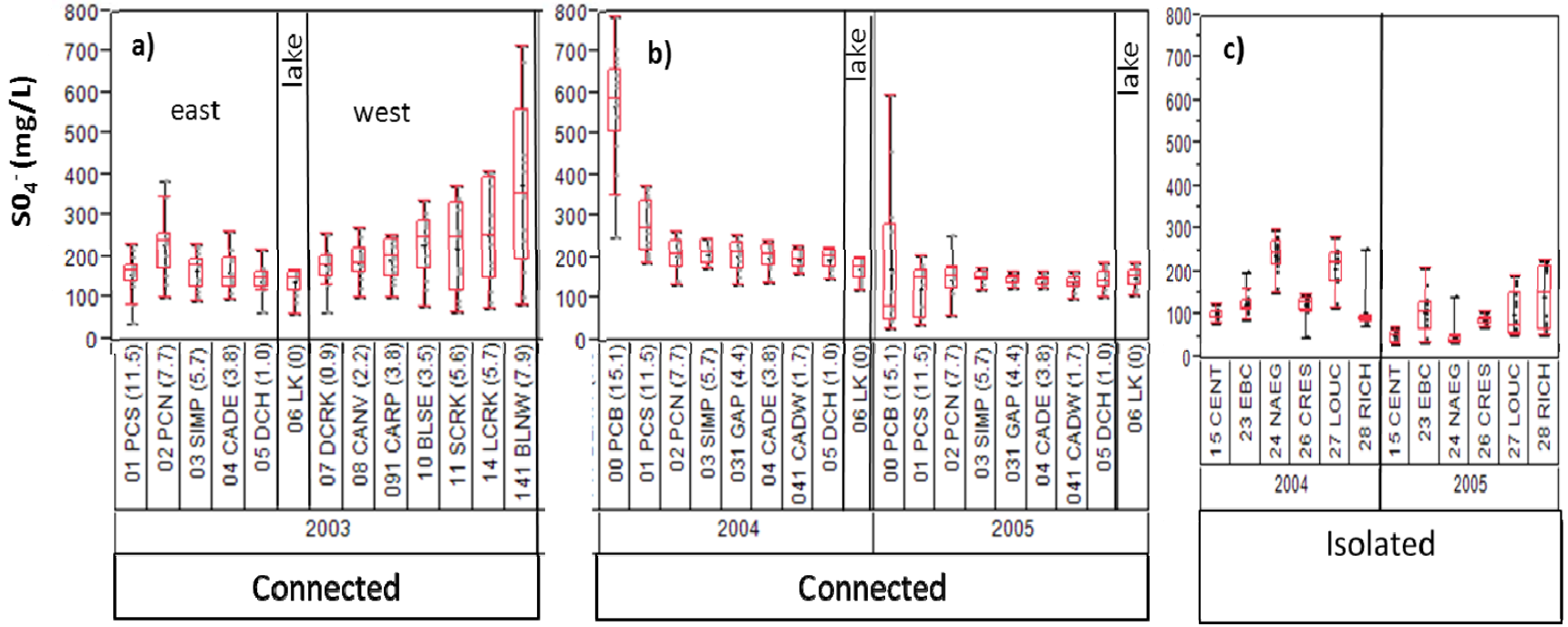

Figure 5.24 Mean and range of sulfate (SO_4^-) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1. Note no data are available for 2002.

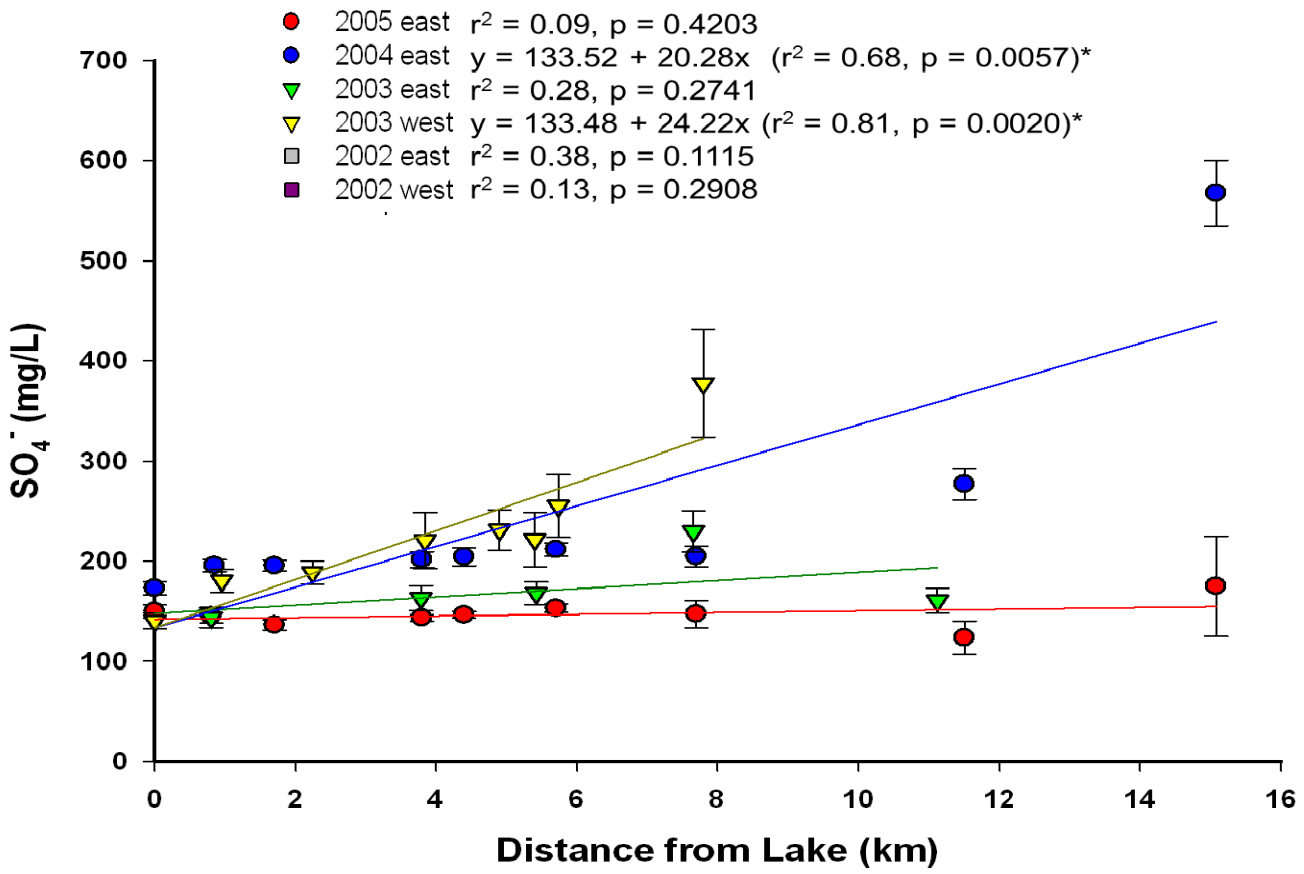

Figure 5.25 Mean ±SE Sulfate (SO_4^-) concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant. No data are available for 2002.

Dissolved organic carbon (DOC) decreased significantly with decreasing distance to the lake in all study years in the east and west sections of the marsh (Figure 5.26 and 5.27). The gradient was greatest in both the east and west sections of the marsh in during lower water levels in 2003, with concentrations decreasing approximately 70% with decreasing distance. In comparison, in 2005 under higher water levels the variation in DOC concentration with distance from the lake was much reduced. Annually, dissolved organic carbon concentrations varied between 10 and 34 mg/L in all study years (Figure 5.26 and 5.27).

Alkalinity concentrations decreased significantly 15 to 50 % with decreasing distance to the lake in the east and west sections of the marsh in 2002 and 2003, 2004, whereas it did not vary significantly with distance to the lake under high water level conditions in 2005 (Figure 5.28 and 5.29). Annually, mean alkalinity concentrations were highest in 2002 and 2003, with mean concentrations ranging from 250 to 448 mg/L, and lowest in 2005 with concentrations ranging from 225 to 250 mg/L.

Conductivity decreased most significantly (approximately 40%) with decreasing distance to the lake in the east and west sections of the marsh during low water conditions in 2003 (Figure 5.30 and 5.31). In the east in 2002 and 2004, conductivity also decreased significantly with decreased distance to the lake, with the exception of PCS PCB in 2004, were concentrations were lower than the next nearest site to the north (Portage Creek north; Figure 5.30 and 5.31). As a result PCB and PCS were not included in the regressions of conductivity versus distance to the lake in 2004 and 2005. Conductivity did not decrease significantly with decreasing distance to the lake in the west section of the marsh in 2002, and the east section in 2005. Annually, mean conductivity concentrations were highest in 2002 and 2003 (approximately 1600 to 2640 μδ/cm),

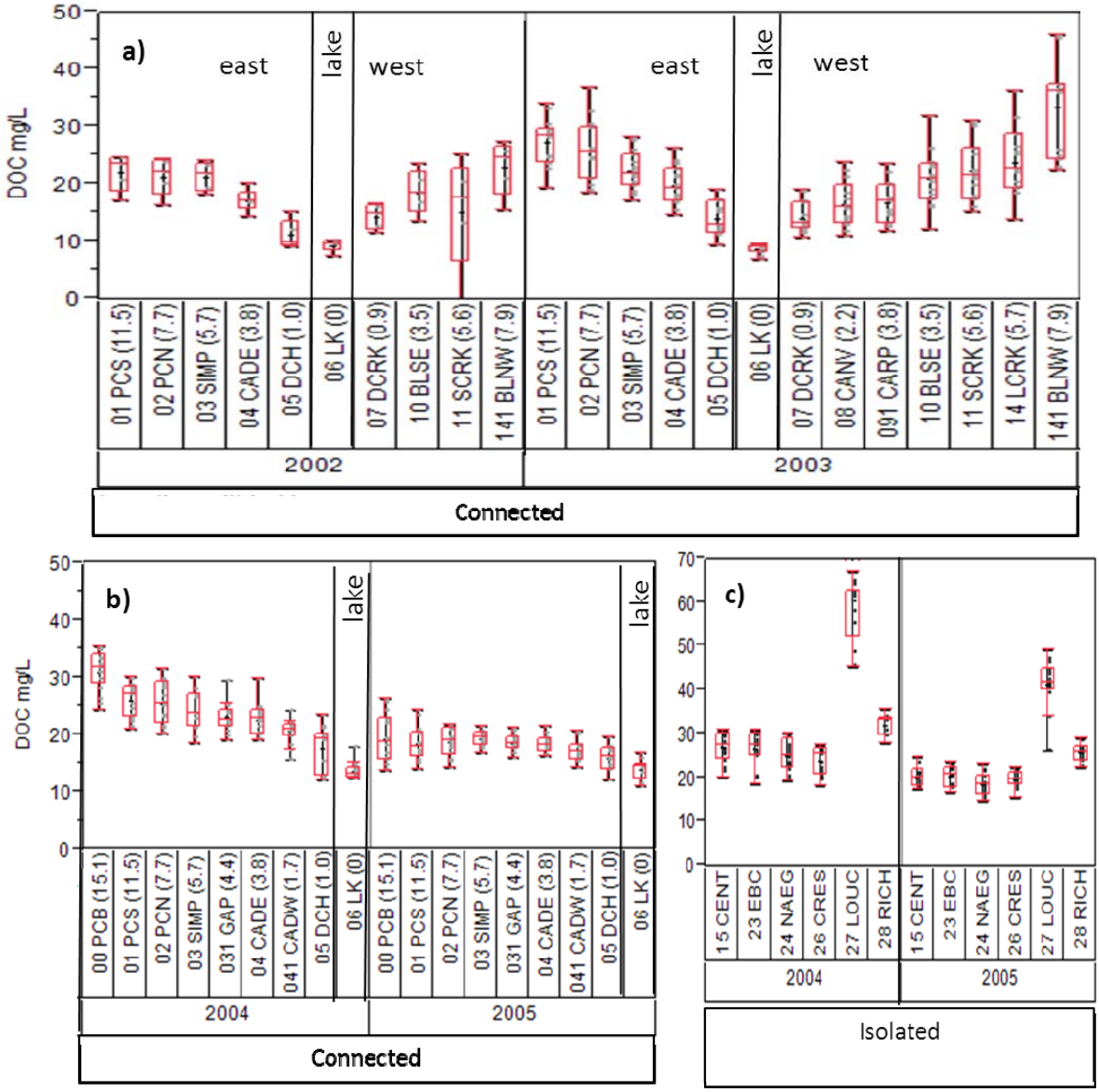

Figure 5.26 Mean and range of dissolved organic carbon (DOC) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

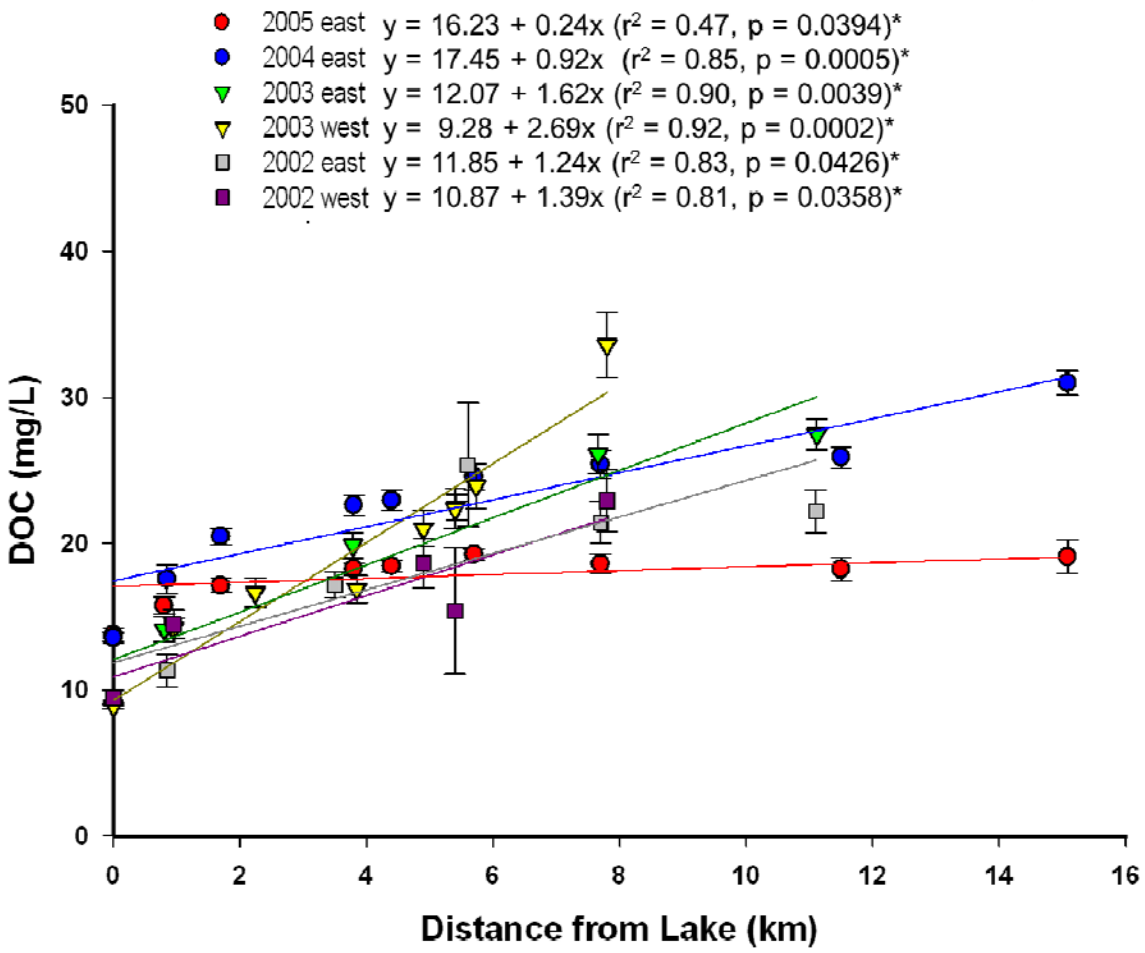

Figure 5.27 Mean ±SE of dissolved organic carbon (DOC) concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

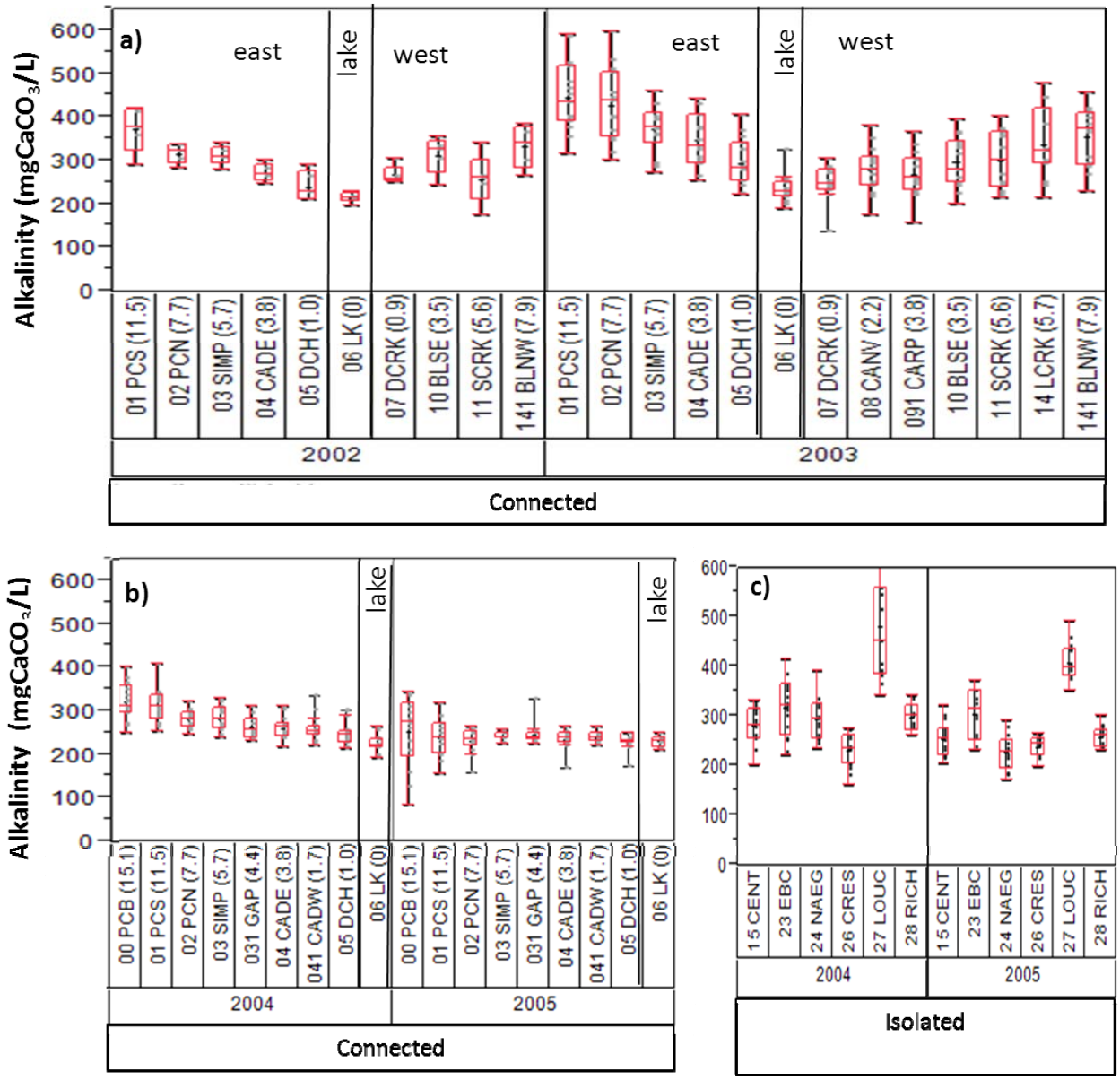

Figure 5.28 Mean and range of alkalinity concentrations ($mgCaCO_3/L$) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

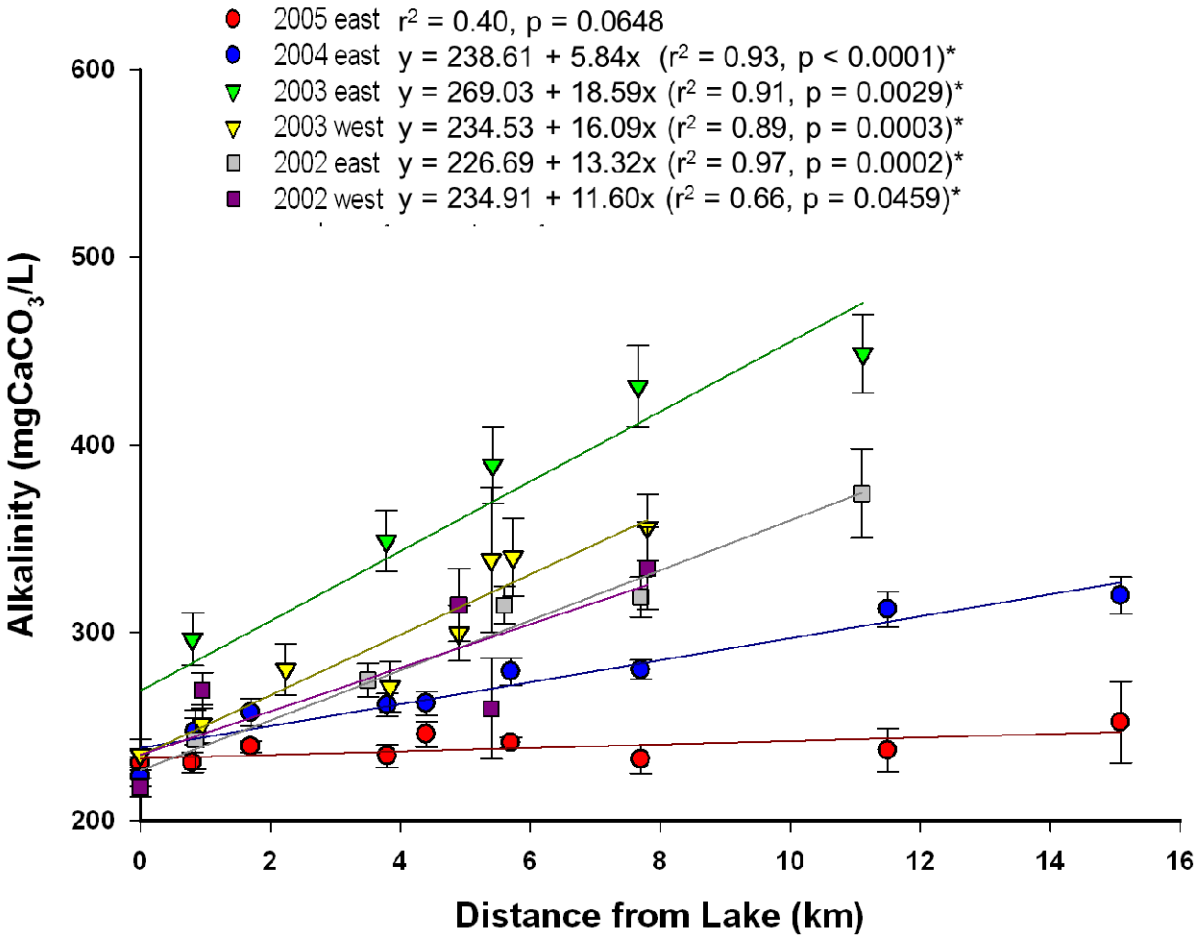

Figure 5.29 Mean ±SE of alkalinity concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

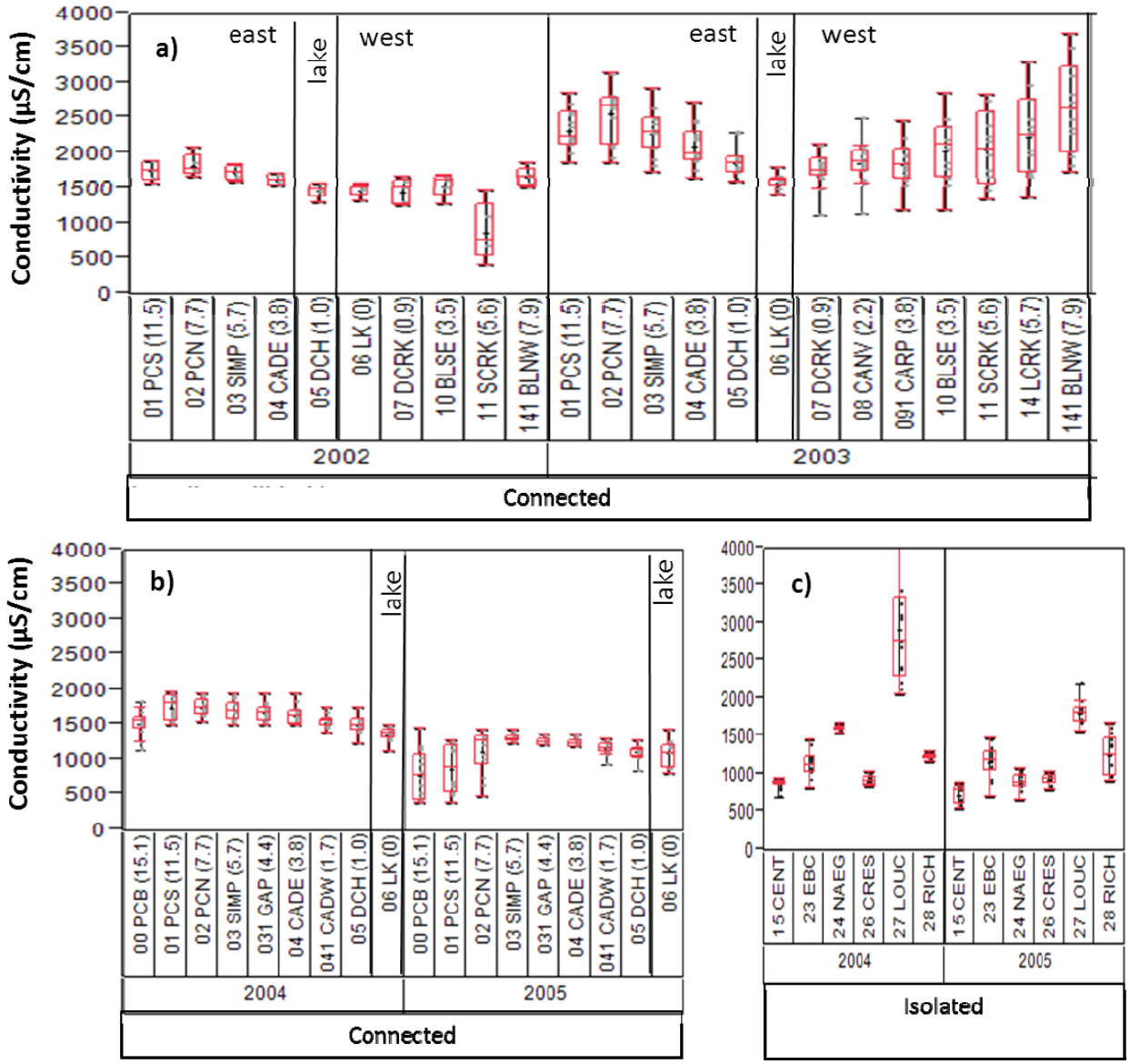

Figure 5.30 Mean and range of conductivity (µS/cm) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

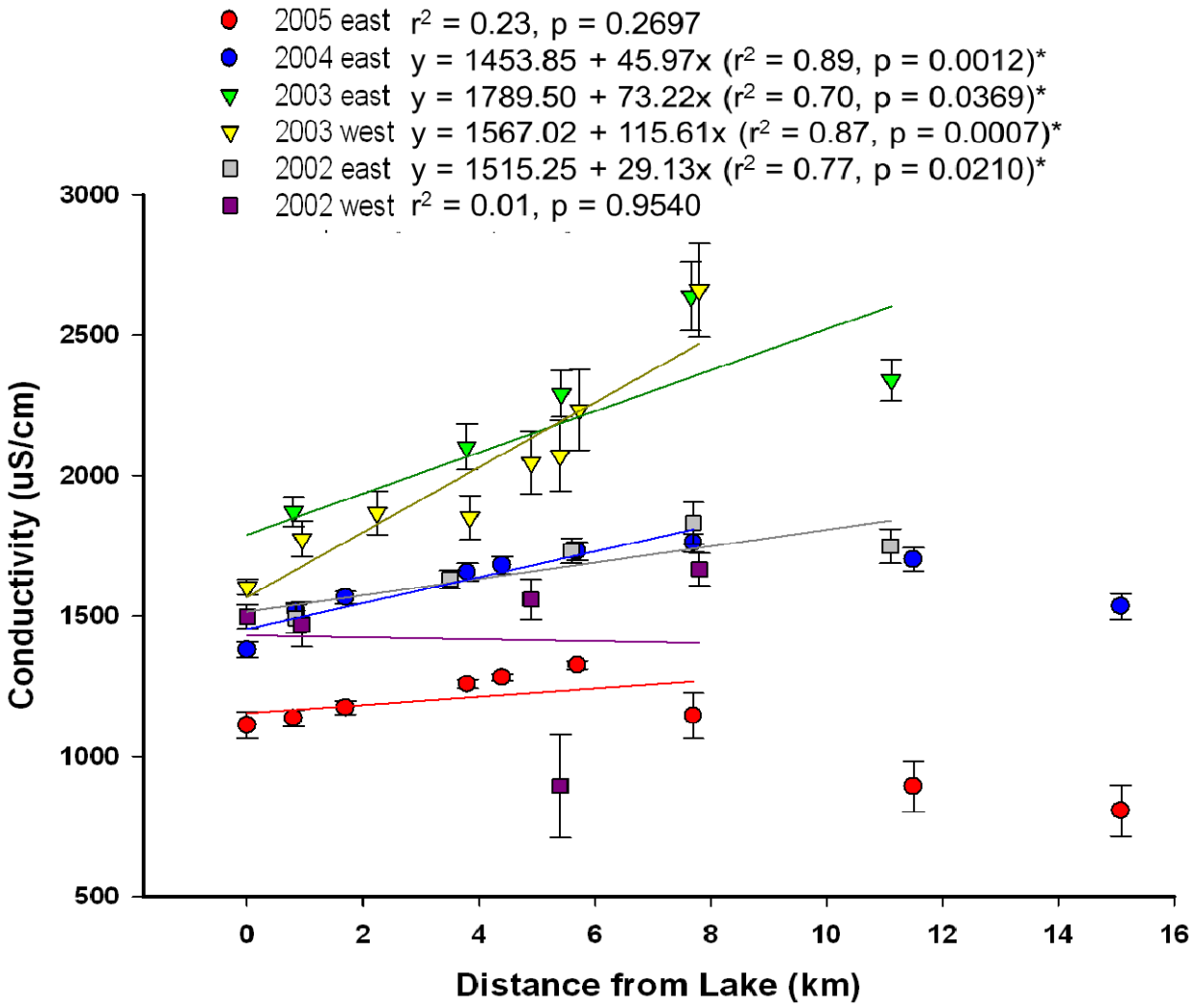

Figure 5.31 Mean ±SE of conductivity concentrations (µδ/cm) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

followed by 2004 (approximately 1500 to 1760 μδ/cm), and 2005 (approximately 800 to 1280 μδ/cm).

In 2002 and 2003, mean pH values did not vary significantly in east and west connected marsh sites, however 2004 and 2005 it did vary significantly in the east marsh (Figure 5.32 and 5.33), with pH increasing from approximately 7.2 to 8.4 with decreasing distance to the lake. Annually, mean pH values ranged between 8.0 and 8.6, with a few exceptions at the furthest inland sites in 2002, 2004 and 2005 (Figure 5.32 and 5.33).

Silica concentrations were only measured in 2005, and were high at east marsh sites and the lake (>3.0 mg/L). Mean concentrations increased from approximately 3.0 mg/L to 4.5 mg/L with decreasing distance to the lake (Figure 5.34 and 5.35).

In the case of total suspended solids (TSS) concentrations and turbidity values (NTU) in the east marsh in 2004 and 2005, PCS and PCB were removed from the regressions to improve the fit of the regressions of TSS and turbidity versus distance to the lake. The removal of PCS and PCB did improve the fit and significance of the regression of the 2004 TSS concentrations, with concentrations decreasing significantly with decreasing distance to the lake, however was still no significant trend with distance for turbidity in 2004 (Figure 5.36 to 5.39). Turbidity was only found to change significantly with distance to the lake in the east section of the marsh in under high water levels in 2005, with concentrations increasing with decreasing distance to the lake. With the removal of PCB and PCS from the regression of TSS concentrations at east marsh sites in 2004 and 2005 versus distance to the lake, concentrations of TSS significantly increased with increasing distance to the lake in 2005, whereas TSS decreased with increasing distance to the lake in 2004. The only other study year in which TSS changed significantly with distance was in

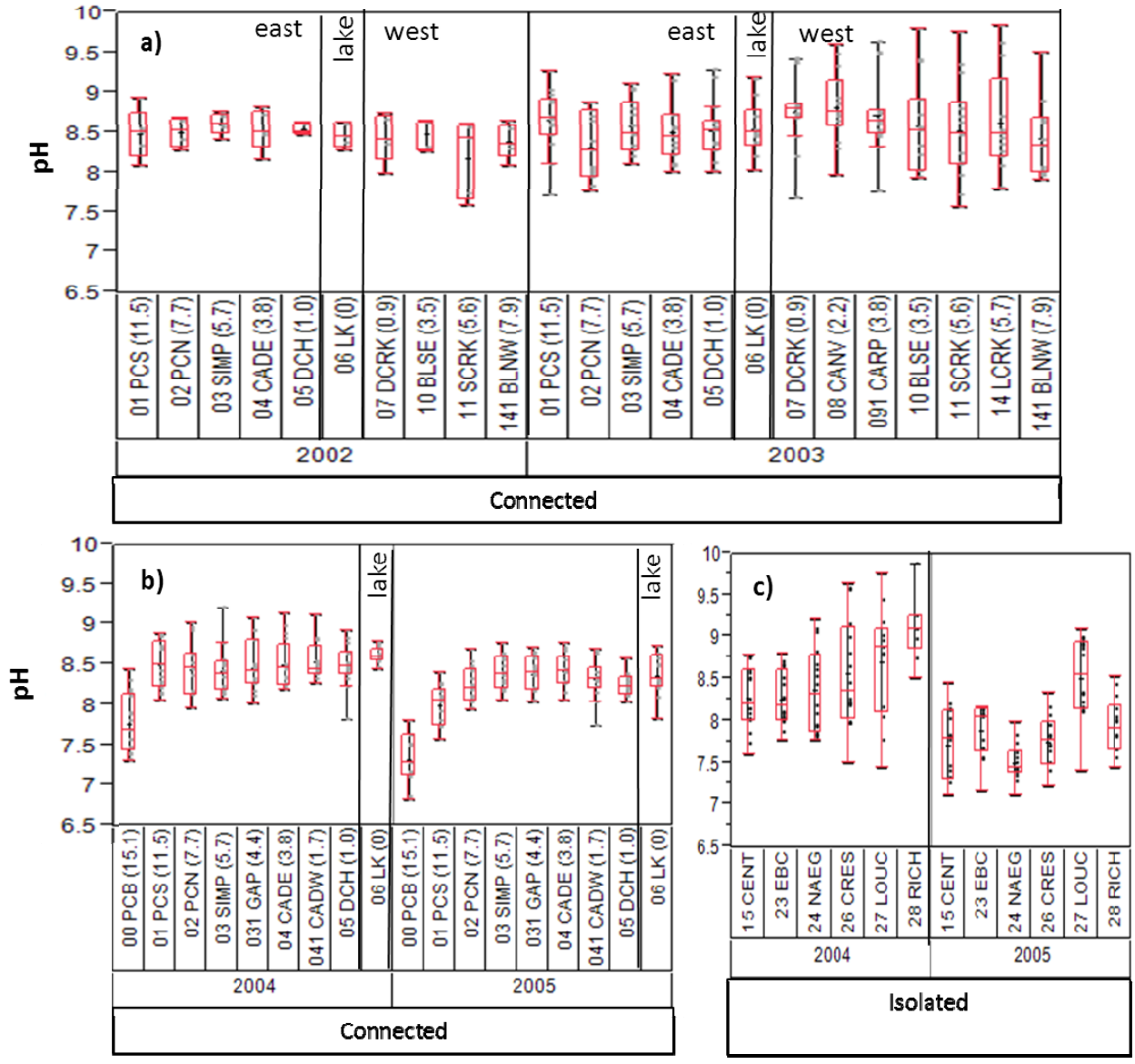

Figure 5.32 Mean and range of pH in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

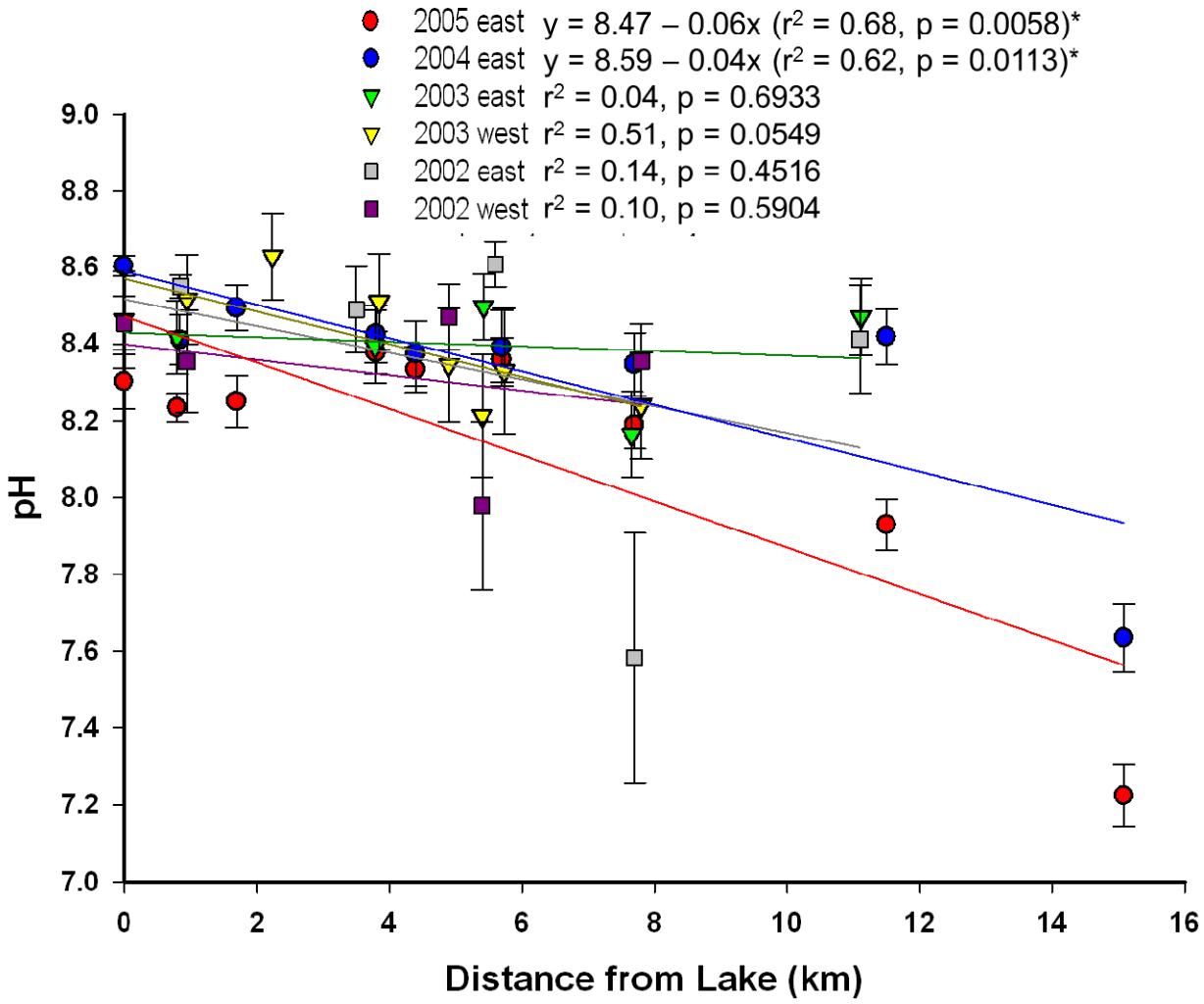

Figure 5.33 Mean ±SE of pH in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

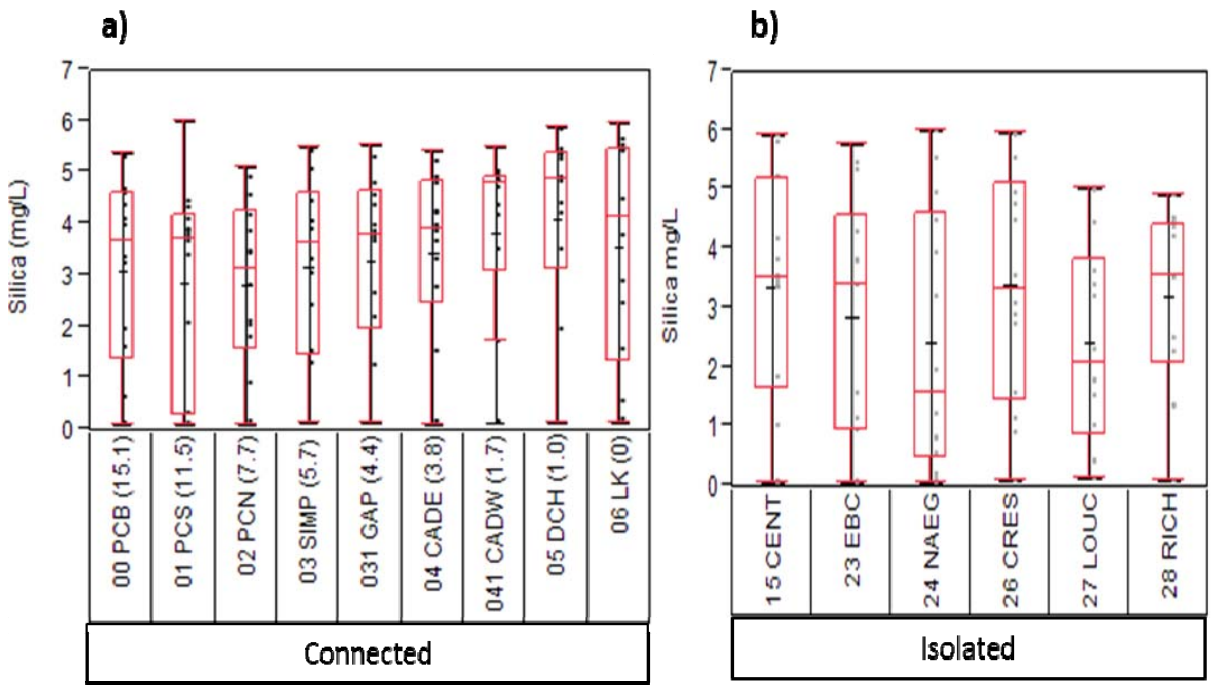

Figure 5.34 Mean and range of silica (mg/L) in (a) Lake Manitoba and the east section of Delta Marsh in 2005, with distance from the lake (km) increasing to the left in both years; and (b) in isolated sites in Delta Marsh in 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

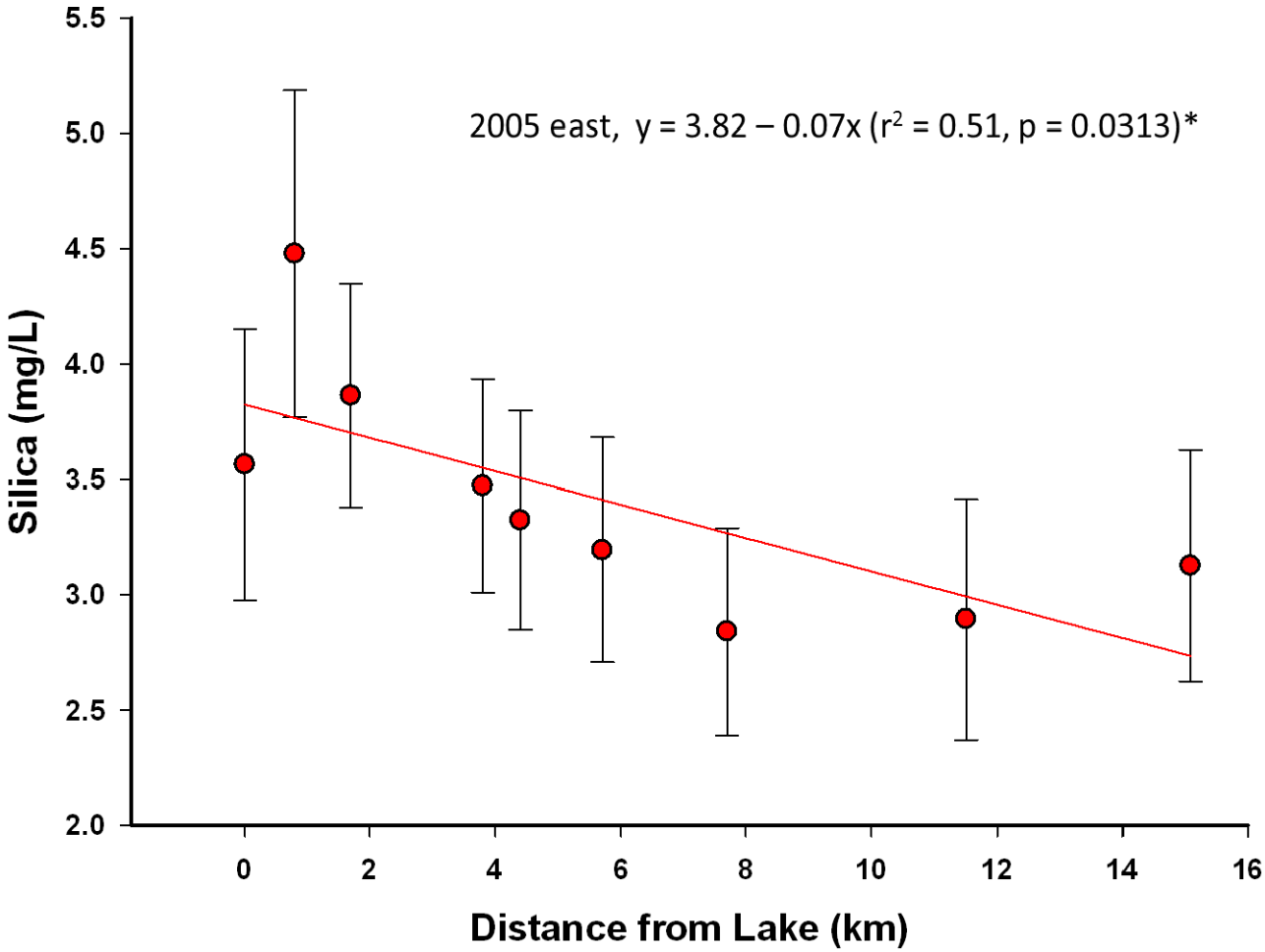

Figure 5.35 Mean ±SE of silica concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

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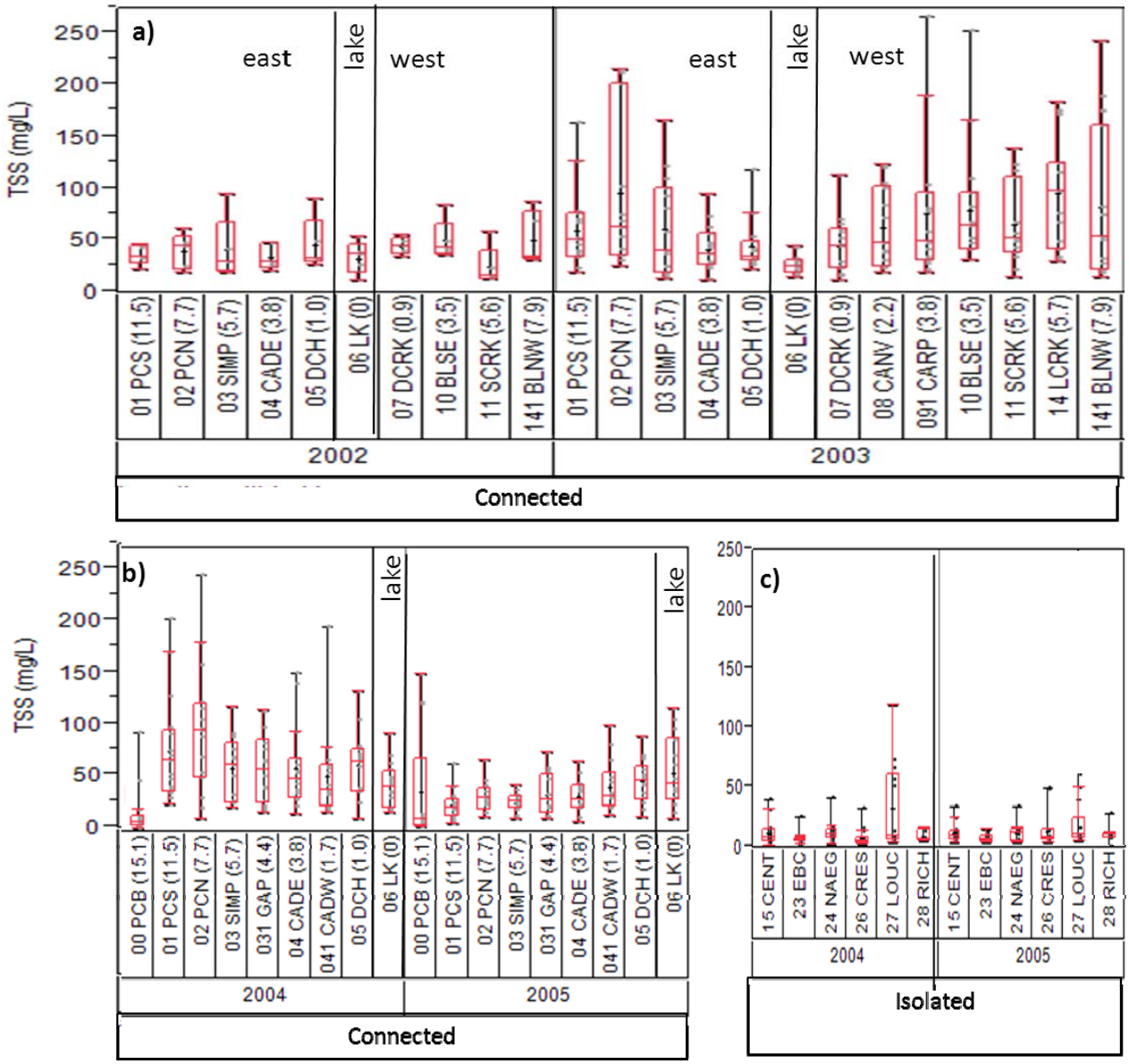


Figure 5.36 Total Suspended Solids (TSS) concentrations (mg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

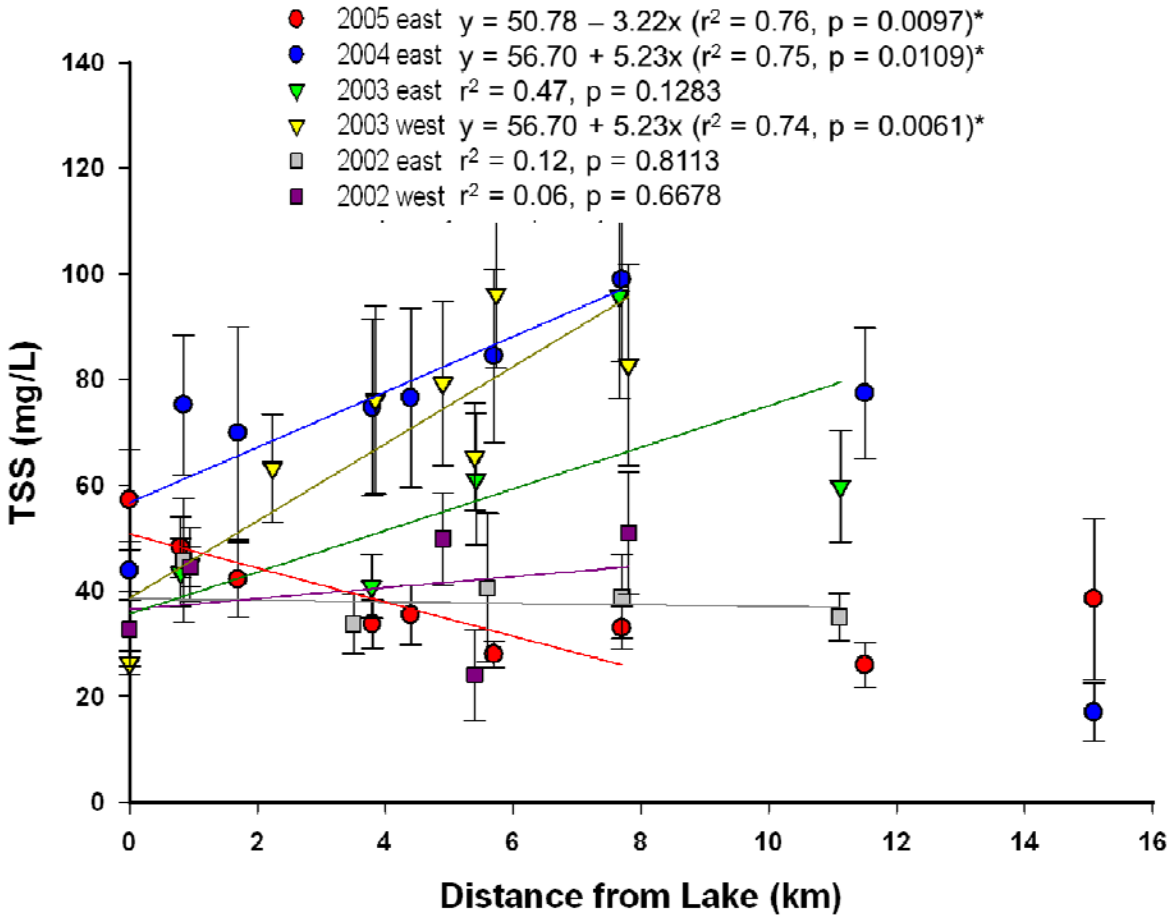


Figure 5.37 Mean TSS ±SE concentrations (mg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

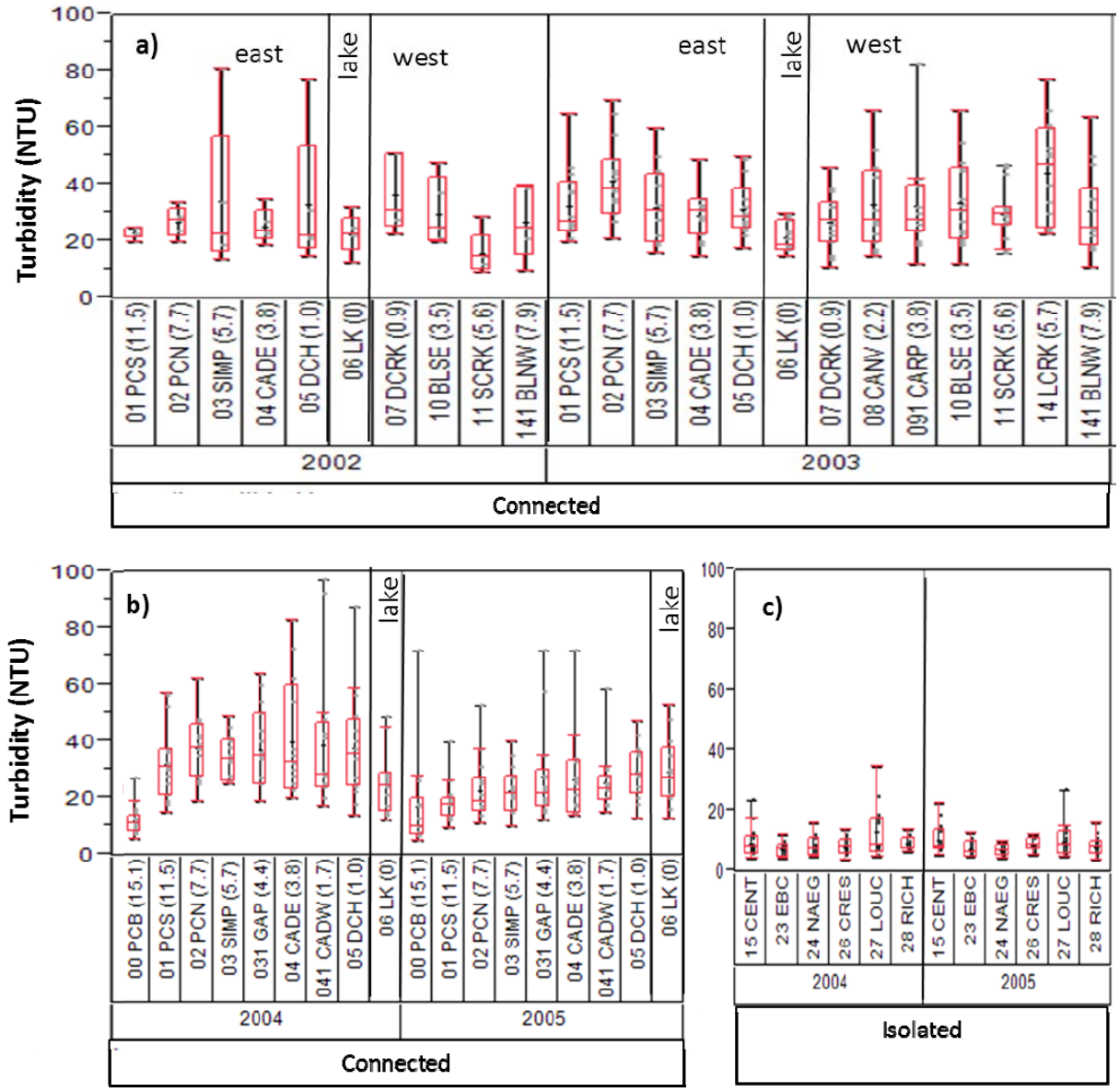


Figure 5.38 Turbidity (NTU) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

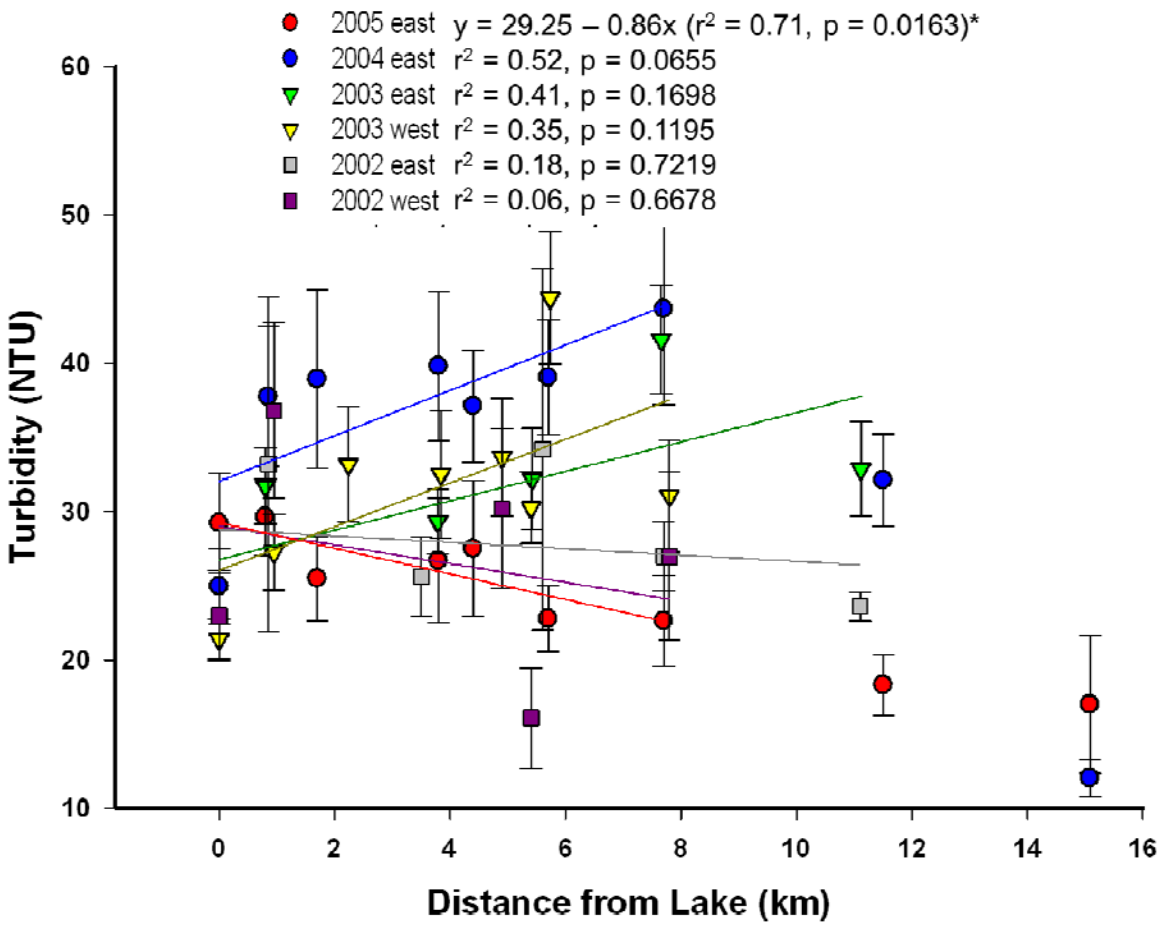


Figure 5.39 Mean turbidity ±SE in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

the west section of the marsh in 2003, with concentrations decreasing with increasing distance to the lake (Figure 5.36 and 5.37). Annually, mean TSS concentrations were higher 2003 and 2004 (30 to 100 mg/L) in comparison to 2002 and 2005 (10 to 60 mg/L). Similar to TSS concentrations, annually mean turbidity values were greater in 2003 and 2004 (12 to 45 NTU), in comparisons to 2002 and 2005 (15 to 30 NTU) (Figure 5.36 to 5.37).

Total phytoplankton chlorophyll *a* concentrations decreased significantly 65 to 90% with decreasing distance to Lake Manitoba in the east section of the marsh in all study years (Figure 5.40 and 5.41) Removal of PCB and PCS from the 2004 and 2005 regressions resulted in significant improvement of the fit of the linear trend of decreasing chlorophyll *a* concentrations with decreasing distance to the lake, so these sites were not included in the regressions (Figure 5.41). In the east section of the marsh in 2002 and 2003, no significant trend was found between chlorophyll *a* concentrations and distance to the lake (Figure 5.41). Annually, mean concentrations of total phytoplankton chlorophyll *a* were highest in 2002 and 2003 (48 to 175 µg/L), with the range in concentrations decreasing with increasing water levels in 2004 and 2005 (47 to 118 µg/L; Figure 5.40 and 5.41).

In Lake Manitoba, throughout the study mean water column concentrations of DIN-N (0.13 to 0.17 mg/L), TN (4.3 to 8.8 mg/L), TRP-P (0.13 to 0.37 mg/L), and TP (0.06 to 0.16 mg/L) were generally below the lower range of marsh concentrations, and mean molar ratios of TN:TP (112 to 260), DIN:TRP (5 to 24) and DIN:TP (3.4 to 5.5) were higher in the lake when compared to the marsh sites (Figure 5.10 to 5.23). Mean Cl^- and SO_4^- concentrations in the lake were lower, 131 to 239 mg/L and 140 to 172 mg/L, respectively compared to the marsh sites in all study years, with the exception the first south site in the marsh, Portage Creek bridge (PCB; Figure 5.4 and 5.5, and Figure 5.24 and 5.25). DOC, alkalinity, and total chlorophyll *a*

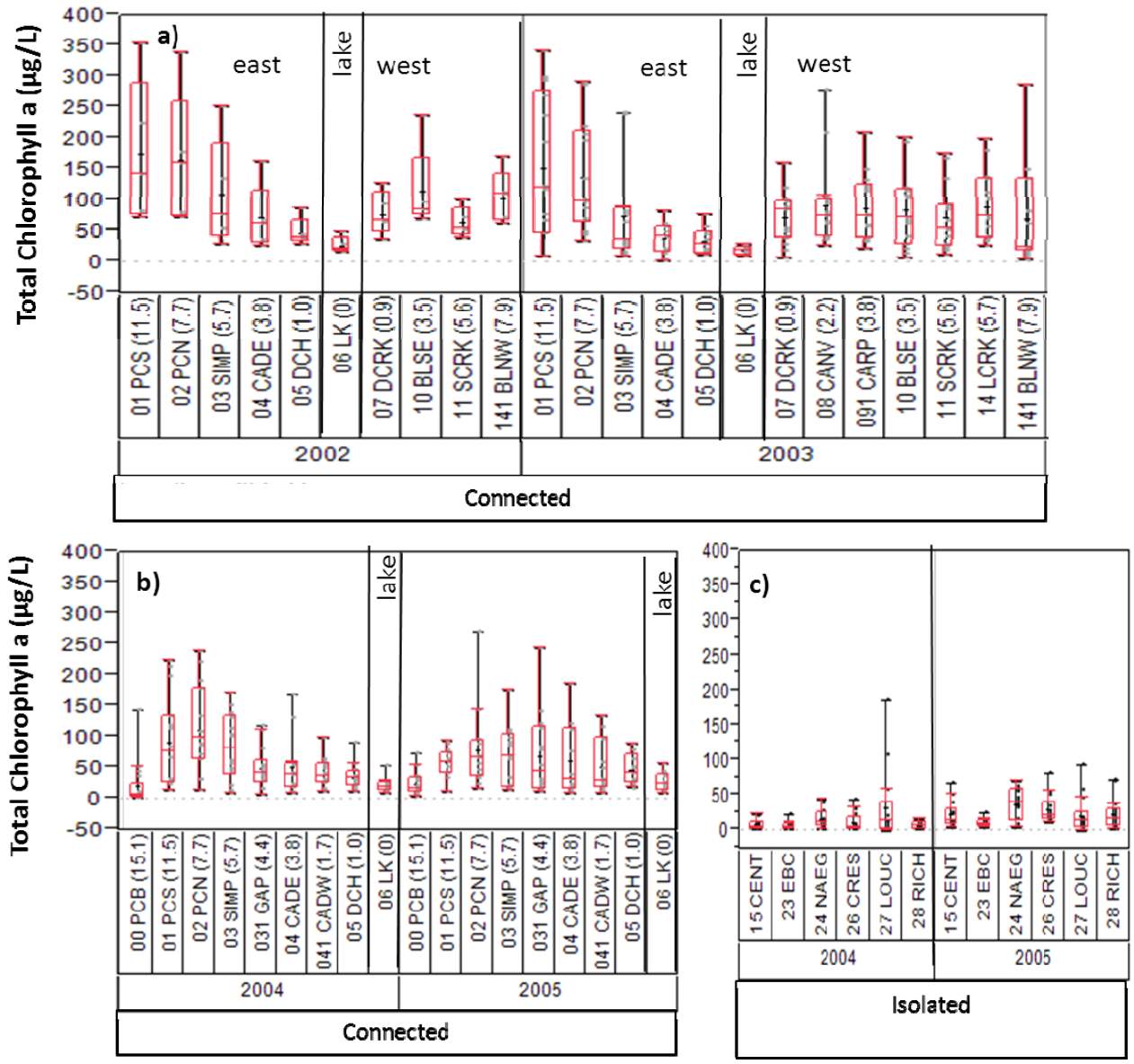


Figure 5.40 Total phytoplankton chlorophyll *a* concentrations (µg/L) in (a) Lake Manitoba (center of figure) and the connected east (left) and west (right) sections of Delta Marsh in 2002 and 2003, with distance from the lake (km) in the east and west increasing outward from center; (b) Lake Manitoba and the east section of Delta Marsh in 2004 and 2005, with distance from the lake (km) increasing to the left in both years; and (c) in isolated sites in Delta Marsh in 2004 and 2005. The distances of each site (km) from Lake Manitoba via surface water flow are noted in brackets following the site codes. The full names of the sites and additional information are provided in Table 3.1 and Figure 3.1.

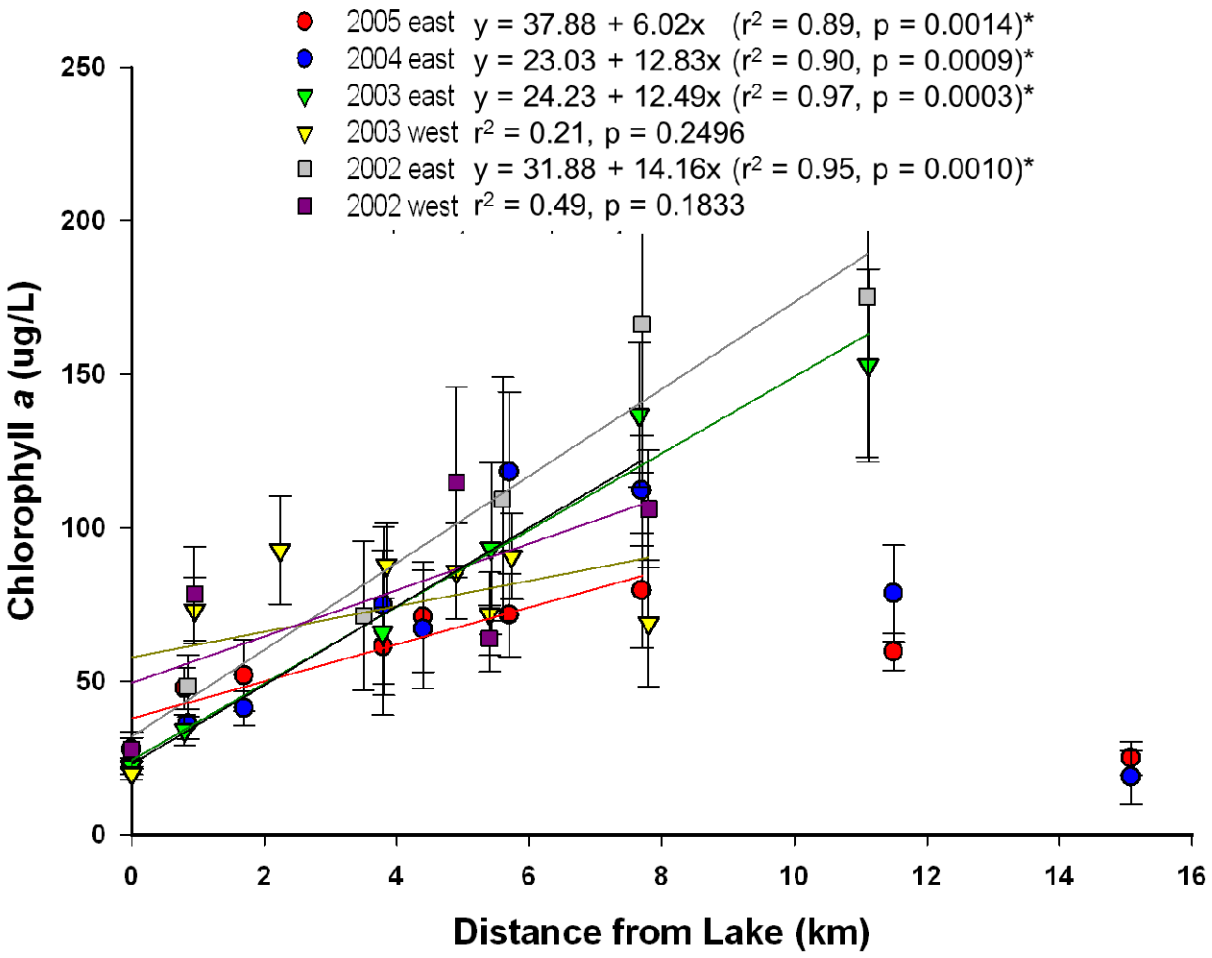


Figure 5.41 Mean phytoplankton chlorophyll *a* ±SE concentrations (µg/L) in Lake Manitoba and Delta Marsh plotted against the distance of sampling sites from Lake Manitoba (km), from 2003 to 2005. r^2 value and p value for each regression are reported, and when significant the correlation equation is also given. * denotes significant.

concentrations were also consistently lower in the lake, 10 to 14 mg/L, 217 to 234 mg/L and 19 to 27 μg/L, respectively compared to the marsh in all study years (Figure 5.26 to 5.29, and 5.40 and 5.41). In comparison to the marsh, the lake also experience less annual variation in nutrient and major ions concentrations.

Overall in comparison of sites located in the east and west sections of the marsh in 2004 and 2005, there was no significant difference in the water chemistry parameters examined, with the exception of significantly higher mean concentrations of NO_3-N ($F = 36.4072$, $p < 0.0001$), DIN:TP ($F = 12.2528$, $p = 0.0007$), and alkalinity ($F = 46.2258$, $p = <0.0001$), in the east section of the marsh in 2005 (Figures 5.6, 5.7, 5.22, 5.23, 5.28, 5.29, Table 5.1).

In comparison of connected and isolated marsh sites in 2004 and 2005, one of the isolated sites, Louck's Pothole (LOUC) was taken out of the analysis of variance (ANOVA) between the connected and isolated sites as it was determined my ANOVA (Appendix F) and PCA analysis (Figure 5.43), that majority of water chemistry parameters at LOUC were significantly different compared to the other isolated sites. Aside from LOUC, the majority of other isolated sites had significantly greater mean concentrations of TP (0.1 to 1.4 mg/L) and TRP-P (0.1 to 1.0 mg/L) than connected sites, but lower mean concentrations of DIN-N (0.17 to 0.50 mg/ L) and TN (3.5 to 9.8), as well as lower ranges in molar ratios of TN:TP (15 to 68), DIN:TRP (1 to 9), and DIN:TP (1 to 3), (Figure 5.10 to 5.23; Table 5.1). Mean concentrations of TSS (7 to 14 mg/L), turbidity (7 to 12 NTU) , SO_4^- (56 to 145 mg/L), Cl- (49 to 210 mg/L), conductivity (743 to 1621 μδ/cm) and total phytoplankton chlorophyll *a* (8 to 39 μg/L) were also significantly lower in isolated sites compared with connected sites, whereas DOC (18 to32 mg/L) and alkalinity (224 to 336 mg/L) were higher at isolated sites compared with connected sites (Figure 5.24 to 5.41; and Table 5.1).

Table 5.1 Analysis of variance (ANOVA) between the water chemistry characteristics (log-transformed (x+1)) of east and west sections of marsh in 2002 and 2003, and isolated and connected in the marsh in 2004 and 2005. * denotes significant difference. n/a = data not available.

	East/West 2002		East/West 2003		Connected/Isolated[1] 2004		Connected/Isolated[1] 2005	
Variable	F-value	p-value	F-value	p-value	F-value	p-value	F-value	p-value
	n = 45		n = 177		n = 180		n = 166	
NO_3-N (mg/L)	n/a	n/a	**36.4072**	**<0.0001***	**171.1837**	**<0.0001***	0.0010	0.9752
NH_3-N (mg/L)	n/a	n/a	0.7163	0.3985	**23.3548**	**<0.0001***	2.3392	0.1281
DIN-N (mg/L)	n/a	n/a	3.6298	0.0584	**28.5156**	**<0.0001***	1.6254	0.2042
TN (mg/L)	2.3885	0.1296	1.0165	0.3159	**17.6809**	**<0.0001***	**10.2637**	**0.0016***
TRP PO_4-P (mg/L)	2.5196	0.1198	2.6970	0.1023	**121.8119**	**<0.0001***	**22.6957**	**<0.0001***
TP (mg/L)	2.5285	0.1191	0.5778	0.4491	**48.8885**	**<0.0001***	**12.9690**	**0.0004***
TN:TP (molar ratio)	1.5189	0.2018	2.3642	0.1275	**9.3541**	**0.0026***	**34.5129**	**<0.0001***
DIN:TRP (molar ratio)	n/a	n/a	2.4552	0.1189	**107.9375**	**<0.0001***	**36.4668**	**<0.0001***
DIN:TP (molar ratio)	n/a	n/a	**12.2528**	**0.0007***	**35.1281**	**<0.0001***	**36.7772**	**<0.0001***
DOC (mg/L)	0.2600	0.6127	0.5126	0.4750	**39.5918**	**<0.0001***	**48.4180**	**<0.0001***
Cl^- (mg/L)	n/a	n/a	3.4451	0.05133	**182.4729**	**<0.0001***	**32.6122**	**<0.0001***
SO_4^- (mg/l)	n/a	n/a	2.3675	0.0915	**67.8291**	**<0.0001***	**55.1837**	**<0.0001***
Phyto total chloro (µg/L)	1.0282	0.3163	0.2929	0.5891	**69.2864**	**<0.0001***	**33.1937**	**<0.0001***
Alkalinity mg$CaCO_3$ L^{-1}	0.4333	0.5139	**46.2258**	**<0.0001***	**15.1541**	**<0.0001***	**16.4167**	**<0.0001***
pH	1.0345	0.4521	2.6967	0.1023	**60.2864**	**<0.0001***	**121.6674**	**<0.0001***
Conductivity (µS/cm)	2.3715	0.1365	3.0794	0.0713	**238.4130**	**<0.0001***	**20.1757**	**<0.0001***
Turbidity (NTU)	0.0720	0.7891	0.0277	0.8679	**206.7670**	**<0.0001***	**114.4736**	**<0.0001***
TSS (mg/L)	0.3327	0.5671	2.3623	0.1261	**117.4893**	**<0.0001***	**87.2763**	**<0.0001***

[1] Loucks Pothole (isolated site) was taken out of analysis.

Principal component analysis (PCA) was used to examine overall spatial and temporal patterns in the water chemistry of marsh sites and the lake, including patterns between (1) east and west marsh sites in 2002 and 2003, (2) connected and isolated marsh sites and the lake in 2004 and 2005, and (3) connected marsh sites and the lake from 2003 to 2005 (Figure 5.42 to 5.44). The PCA biplot of east and west marsh sites in 2002 and 2003, confirmed the results of the regression and ANOVA analysis that there was little difference in water chemistry between the two sections of the marsh, as the east and west marsh sites overlapped on the biplot, indicating none of the water chemistry parameters examined separated them out (Figure 5.42). However, the PCA biplot separated out the water chemistry of individual marsh sampling sites between sampling years (2002 compared to 2003) and distance from the lake.

The PCA of the east and west marsh sites in 2002 and 2003 is presented in Figure 4.42. The majority of the variation in the individual marsh sampling sites between 2002 and 2003 and with increased distance from the lake was summarized by Axis 1 (54%), which was dominated by increasing alkalinity, DOC, TSS, TN, conductivity, and total chlorophyll *a* concentrations (correlation coefficient with Axis 1 > 0.7). ANOVA analysis of PCA Axis 1 scores plotted against sampling year and distance from the lake, further confirmed that Axis 1 was significantly correlated with sampling year ($F = 4,4483$, $p = 0.0492$) and distance from the lake ($r^2 = 0.79$, $p < 0.0001$; Table 5.2). The regression of Axis 1 against water depth, also confirmed that Axis 1 was negatively correlated with water depth ($r^2 = 0.65$, $p < 0.0001$). The negative correlation of both year and depth with Axis 1 indicated that increases in the water chemistry variables along Axis 1 were correlated with the lower water levels in 2003 compared to 2002. Axis 2 accounted for another 22% of the variation between the sites, and graphically further separated the sites by

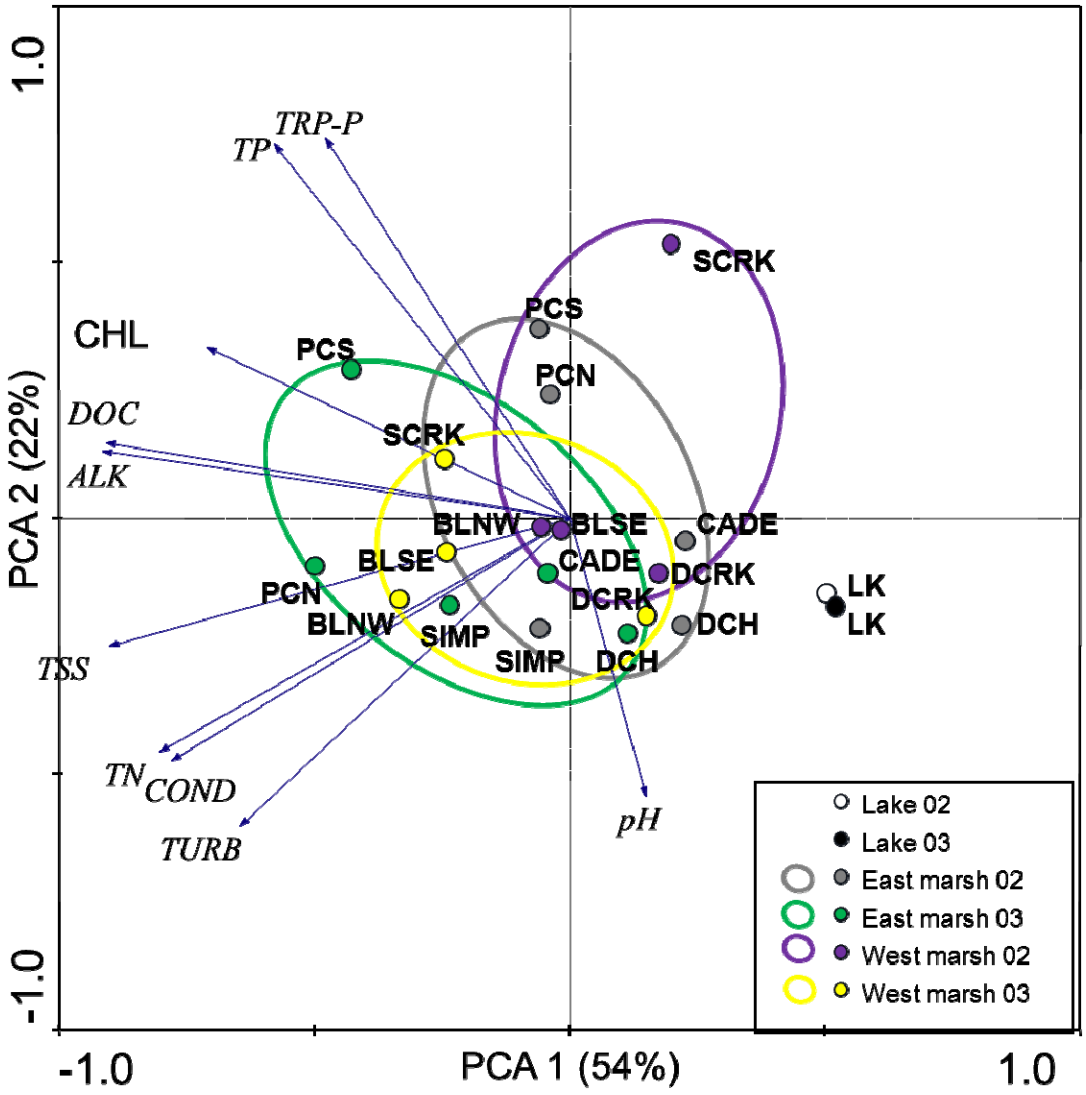


Figure 5.42 Principal component analysis ordination of Lake Manitoba sample sites, five connected sites on the east side of delta marsh, and four sites on the west side, constrained by water chemistry parameters (vectors) between years in 2002 and 2003. The first two PCA axes of the graph accounts for 76% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 54%, and PCA2 for 22%. Sites codes: (1) LK = Lake Manitoba, (2) DCH = Delta Channel, (3) CADE = Cadham Bay East, (4) SIMP = Naegele's/Simpsons Bay (5) PCN = Portage Creek North, (6) PCS = Portage Creek South, , (7) DCRK= Deep Creek, (8)BLSE = Big Lake south-east, (9)BLNW = Big Lake northwest, (10) SCRK = Short Creek, Water Chemistry Parameter Codes: pH = pH, TSS = total suspended solids, COND = specific conductance, ALK = alkalinity, TURB = Turbidity (NTU), TRP = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, CHL= total chlorophyll *a*.

Table 5.2 PCA1 and PCA2 and the environmental variables associated with the PCA axes, plotted against distance from the lake, sampling year and water depth. * Denotes significant difference.

PCA Figure		Distance[1,2]		Year[3]		Depth[2]	
		r^2	*p* value	F value	*p* value	r^2	*p* value
Fig 5.42- 2002 & 2003 East and west connected and lake	PCA1	**0.79**	**<0.0001***	**4.4483**	**0.0492***	**0.65**	**<0.0001***
	PCA2	**0.46**	**0.0414***	1.6140	0.2201	0.12	0.1247
Fig 5.43- 2004 & 2005 East connected, lake and isolated sites	PCA1	0.03	0.4755	**13.5300**	**0.0020***	**0.60**	**0.0002***
	PCA2	**0.72**	**0.0008***	**4.7803**	**0.0440***	0.02	0.4178
Fig 5.44- 2003 to 2005 East connected and lake	PCA1	**0.44**	**0.0037***	**9.6417**	**0.0016***	**0.33**	**0.0009***
	PCA2	**0.30**	**0.0101***	**35.7361**	**<0.0001***	**0.73**	**<0.0001***

[1] Isolated sites are not connected to lake via surface water course, and therefore distance via surface water flow
[2] Regression analysis
[3] Analysis of variance (ANOVA)

distance from Lake Manitoba, with TRP and TP increasing with increasing distance from the lake ($r^2 = 0.46$, $p = 0.0414$; Table 5.2).

The PCA biplot comparing the east connected and isolated marsh sites and the lake in 2004 and 2005 (Figure 5.43), also supported the ANOVA results and clearly separated out the isolated and connected marsh sites, as well as both groups of marsh and the lake sites by year (2004 and 2005; Figure 5.43). The first axis accounted for 42% of the variation and was dominated by conductivity, TSS, TN, DIN, turbidity, Cl^- and total chlorophyll *a*. Axis 2 counted for another 28% of the variation and was dominated by alkalinity, DOC, TP, and TRP. Axis 1 separated isolated and connected marsh sites, as we as marsh sites by sampling year (2004 compared to 2005), connected sites had greater conductivity, TSS, TN, DIN, turbidity, Cl^- and total chlorophyll *a* concentrations compared to isolated sites, and both connected and isolated marsh sites and the lake had greater conductivity, TSS, TN, DIN, turbidity, Cl^- and total chlorophyll *a*. concentrations in 2004 compared to 2005. ANOVA analysis of Axis 1 against year and water depth, confirmed that Axis 1 was correlated with year ($F = 13.5300$, $p = 0.0020$) and depth ($r^2 = 0.60$, $p = 0.0002$; Table 5.1). Axis 2 separated the isolated and connected marsh sites, as well as the connected sites by their distance from the lake, with higher concentrations of TP, TRP, DOC and alkalinity in isolated compared to connected sites and in connected sites with increasing distance from the lake. Regression analysis of Axis 2 against distance from the lake ($r^2 = 0.72$, $p = 0.0008$; Table 5.2). Similar to the ANOVA analyses (Appendix F) Axis 2 also clearly separated Loucks Pothole (LOUC) from the other isolated sites, as it had higher DOC, alkalinity, TN and conductivity concentrations compared to the other isolated sites. Similar to the results of the regression analysis, PCA Axis 2 also separated Portage Creek bridge (PCB) from the rest of the connected marsh sites in 2004, as well as PCB and Portage Creek south (PCS) from the rest

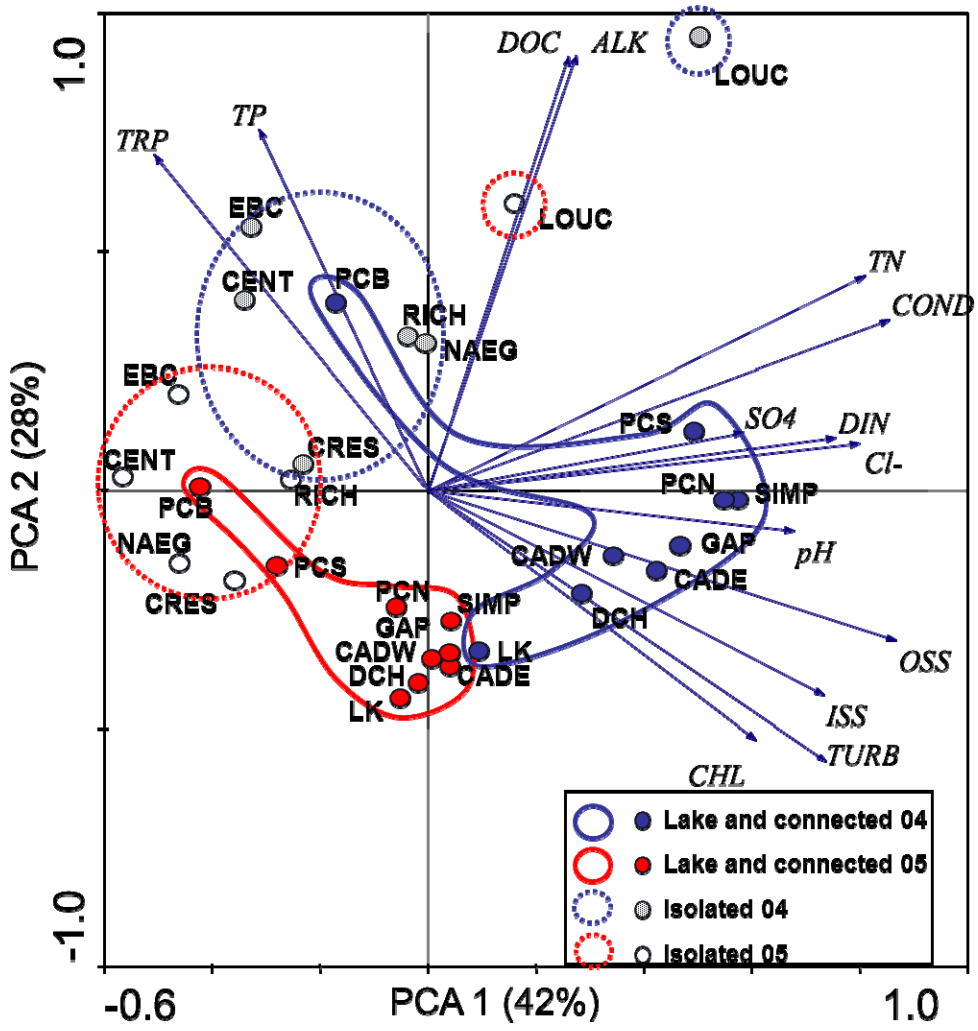


Figure 5.43 Principal component analysis ordination of Lake Manitoba sample sites, eight connected sites on the east side of delta marsh, constrained by water chemistry parameters (vectors) between years in 2004 and 2005. The first two PCA axes of the graph accounts for 89% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 50%, and PCA2 for 29%. Sites codes: (1) LK = Lake Manitoba, (2) DCH = Delta Channel, (3) CADE = Cadham Bay East, (4) CADW = Cadham Bay West, (5) GAP = The Gap, (6) SIMP = Naegele's/Simpsons Bay (7) PCN = Portage Creek North, (8) PCS = Portage Creek South, (9) PCB = Portage Creek Bridge. Water Chemistry Parameter Codes: pH = pH, TSS = total suspended solids, COND= specific conductance, ALK = alkalinity, TURB = Turbidity (NTU), TRP = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulfate, Cl^-= chloride, and CHL = total chlorophyll *a*.

of the connected marsh sites in 2005. According to the PCA biplot in the noted years these sites had water chemistry characteristics that were more similar to inland isolated sites compared to the connected sites (i.e. lower Cl^-, conductivity, N (DIN, and TN), and chlorophyll *a* concentrations, and higher TP and TRP concentrations), indicating these sites were more likely influenced by water inputs from upland watershed than Lake Manitoba.

The PCA biplot of the east connected sites and the lake from 2003 to 2005, clearly separated the sites by sampling year (Figure 5.44). The first axis accounted for 48% of the variation and was dominated by conductivity, Cl^-, TSS, DIN, SO4-, turbidity, and total chlorophyll *a* (Figure 5.44). Axis 2 accounted for another 26% of the variation and was dominated by alkalinity, TN, DOC, TP and TRP. Graphically, Axis1 and Axis 2 together separated the sites by year (2003 to 2005), and distance from the lake. Along Axis 1, marsh and lake sites had greater conductivity, Cl^-, TSS, DIN, $SO4^-$, turbidity, and total chlorophyll a concentrations in 2004 compared to 2005 and concentrations decreased with decreasing distance to the lake, and along Axis 2 alkalinity, TN, DOC, TP and TRP concentrations were higher in 2004 compared to 2005 and in the marsh decreased with decreasing distance to the lake. ANOVA and regression of Axis 1 and Axis 2 against distance from the lake, year and water depth confirmed that Axis1 was significantly correlated with distance from the lake ($r^2 = 0.44$, $p = 0.0037$), year ($F = 9.6417$, $p= 0.0016$) and depth ($r^2 = 0.33$, $p = 0.0009$; Table 5.2), and Axis 2 was correlated with distance to the lake ($r^2 = 0.30$, $p = 0.0101$), year ($F = 35.7361$, $p < 0.0001$), depth ($r^2 = 0.73$, $p < 0.0001$). Overall, the biplot indicated that all the individual connected marsh sites became more closely grouped to each other, as well as the lake from 2003 to 2005. This indicated that the mean water chemistry characteristics became increasing similar from 2003 to 2005, which also corresponded with increasing water levels in both the lake and marsh from 2003 to 2005.

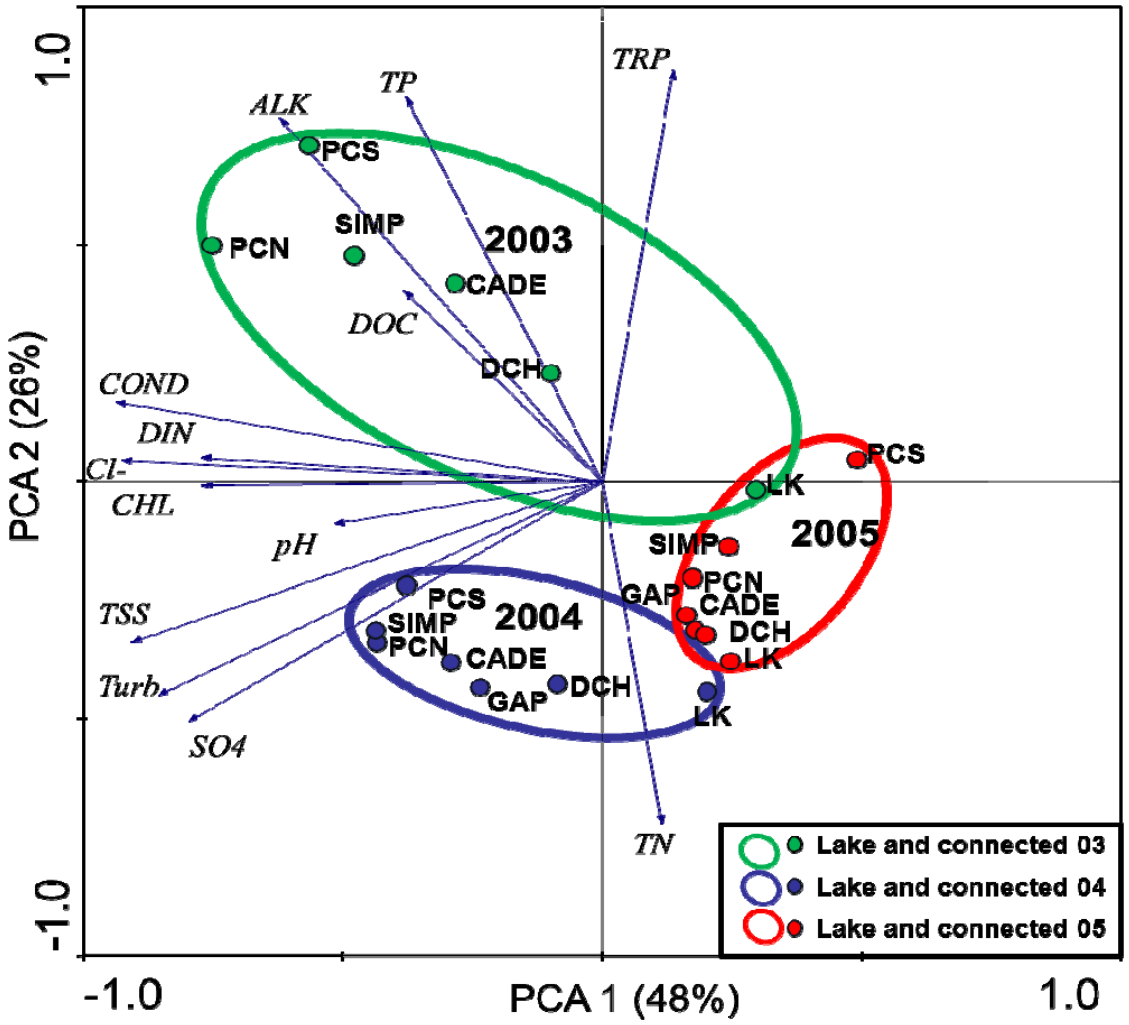


Figure 5.44 Principal component analysis ordination of Lake Manitoba sample sites, six connected sites on the east side of delta marsh, and one isolated site (Center Marsh), constrained by water chemistry parameters (vectors) from 2003 to 2005. The first two PCA axes of the graph accounts for 74% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 48%, and PCA2 for 26%. Sites codes: (1) LK = Lake Manitoba, (2) DCH = Delta Channel, (3) CADE = Cadham Bay East, (4) GAP = The Gap, (5) SIMP= Naegele's/Simpsons Bay (6) PCN = Portage Creek North, (7) PCS = Portage Creek South. Water Chemistry Parameter Codes: pH = pH, TSS = total suspended solids, Cond = specific conductance, ALK = alkalinity, TURB = Turbidity (NTU), TRP = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, DIN = dissolved inorganic nitrogen, Cl- = chloride, SO4 = sulfate, and CHL = total chlorophyll *a*.

In summary the primary findings of the PCA analyses were that year, water depth, and distance from the lake explained a significant amount of the variation in the water chemistry characteristics of the sample sites. In comparison, differences between study years explained more of the variation in the water chemistry data, than distance from the lake. Concentrations of nitrogen (DIN and TN), phosphorus (TP and TRP), alkalinity, DOC, Cl-, SO4 and conductivity all decreased with increased water depth, as well as with decreasing distance from the lake. These trends indicate that connection to the lake, as well as distance from the lake, were important predictors of water chemistry in both the east and west sections of the marsh. Additional PCA biplots that separated the water chemistry of the sites into greater spatial and temporal detail are presented in Appendix G.

Discussion

The significant spatial and seasonal trends in Cl^- concentrations observed in the connected marsh sites, coupled with the proven use of Cl^- in other studies as good hydrological tracer of water mixing and flow due to its relatively inert nature (Kadlec and Knight 1996, Hayashi et al. 1999, Waiser 2006, Morrice et al. 2011), supports the assumption that the decreased Cl^- concentrations in the marsh with decreasing distance to the lake, were mainly the result of spatial differences in the inflow and mixing of relatively dilute lake waters with marsh waters. This is further supported as Lake Manitoba is the permanent year-round source of surface water flow to the marsh.

Results of this study indicated that water column concentrations of most measured water chemistry variables and ions decreased in the marsh with decreasing distance to the lake along both east and west transects in most study years, with the exception of 2005, as higher water levels resulted in reduced annual concentrations and spatial variation in many of the water chemistry parameters examined, namely NO_3-N, NH_3-N, DIN-N, TN, SO_4^-, DOC, Cl^-, conductivity, and alkalinity. Water inflows to Delta Marsh from Lake Manitoba were not a significant source of nutrients (N and P) to Delta Marsh, as mean concentrations were consistently lower in the lake than in marsh sites. The exception was the two furthest inland sites in the east marsh in 2004 and 2005, PCB and PCS, which were influenced by water and nutrient inputs from the surrounding inland watershed than the lake. Compared to the rest of the east marsh, PCB and PCS were characterized by lower concentrations of NO_3-N, NH_3-N, DIN-N and TN, and higher concentrations of TRP-P and TP. This also indicates that the upland watershed maybe a more significant of water inputs higher in P compared N to the marsh, with resultantly

lower N:P ratios. Aside from PCB and PCS, the other furthest inland marsh sites, on the both east and west sides, were characterized by relatively high N (NO_3-N, NH_3-N, DIN-N, TN) and P (TP and TRP), poorer water clarity (high TSS and turbidity), and high DOC, conductivity, alkalinity, phytoplankton chlorophyll *a*, chloride and sulfate concentrations. In comparison at the other end of the gradient, connected sites located in closer proximity to Lake Manitoba exhibited lower nutrient concentrations (N and P), higher water clarity, and lower conductivity, phytoplankton chlorophyll *a*, Cl^- and SO_4^- concentrations.

PCA ordination analysis of the marsh and lake sample sites further summarized these trends and better illustrated the complex spatial and temporal trends in the water chemistry of marsh. Regardless of east and west location in the marsh, and annual variation in water level, regression of the PCA scores showed that relative distance from Lake Manitoba was a significant predictor of differences in several water chemistry characteristics between marsh sample sites. The PCA ordinations showed that annually, concentrations of several of the water chemistry parameters were significantly negatively correlated with increasing water depth from 2003 to 2005, including N (NO_3-N, NH_3-N, DIN-N, TN), conductivity, SO_4^-, DOC, alkalinity, conductivity, total chlorophyll *a*, water clarity (TSS and turbidity). However, concentrations of P (TP and TRP) did not appear to be negatively affected by increasing water depth. Overall, increasing water depth resulted in reduced variation in the majority of water chemistry characteristics between connected marsh sites, as well as between connected marsh sites and the lake.

The consistently higher inland concentrations of P and major ions indicated the surrounding watershed may be the predominant source of nutrients, ions and suspended solids to the marsh during spring snowmelt and periodic storm surface runoff events. The landscape surrounding Delta Marsh is composed largely of agricultural lands, livestock and grain crops, as well as

cottage and recreational development on the beach ridge (Grosshans 2001, Brown 2003). Agriculture has been noted to be one of the largest sources of non-point pollution to aquatic systems in Canada, specifically by livestock wastes, fertilizers (commercial and manure), and sediments from erosion (SCE 2001). When fertilizers and manure are applied in excess of plant requirements, nutrients can build-up in the soil and lead to a loss of these nutrients to surface and ground water (SCE 2001). Livestock can impact water quality directly when they are allowed direct access to water sources, as they defecate in the waters, and destabilize shorelines creating erosion and damaging shoreline vegetation. The residential and cottage developments in the area can also be sources of nutrients from animal wastes, human sewage wastes, and lawn fertilizers (Brown 2003). Studies in other coastal wetlands in North American have also found that the percentage of agriculture in the watershed has a significant effect on the water chemistry of these systems. Crosbie and Chow-Fraser (1999), in a large scale study examining the effects of land use on wetlands in the Laurentian Great Lakes, found that concentrations of N and P, phytoplankton chlorophyll a, and turbidity in wetlands increased predictably with increasing dominance of agriculture in the watershed. Morrice et al. (2008) found that the proportion of agriculture in the watershed, both the proportion of cultivated land and intensity of agricultural chemical use, were strongly related to Great Lakes coastal wetland water quality, with increased concentrations of TP, inorganic forms of N and P, TSS, phytoplankton chlorophyll *a* and Cl^- associated with increased proportions of agricultural land. Trebitz et al. (2007) studied water quality in 58 coastal wetlands surrounding the Great Lakes and found that concentrations of total N and P, water clarity, Cl^- were strongly associated with agricultural intensity in the watershed, with increased agricultural intensity associated with higher N and P chlorophyll *a*, $SO_4^-$and Cl^- concentrations, and lower water clarity. Further, the high $SO4^-$concentrations experienced at the

inland connected wetland sites in this study, have also been noted in other studies to be associated with the proportion of agricultural activity in the watershed (Trebitz et al. 2007).

The spatial decreases in Cl^- concentrations in the marsh with decreasing distance to Lake Manitoba, as well decreases in annual Cl^- concentrations and spatial variation between sites with increasing water levels from 2003 to 2005, along with the oxygen-18 and hydrogen-2 stable isotope data, illustrated the strong influence that seiche-induced water inflows from the lake and associated mixing, had on marsh hydrology and water chemistry, by periodically increasing marsh water level and flushing the marsh with relatively 'fresher' water with lower low nutrient and ion concentrations. As a result, the reductions in nutrients (N and P) and dissolved ions observed in the marsh with decreasing distance to the lake can, in part, be contributed to dilution by 'fresher' water inputs from the lake. The increasing degree of variation in Cl^- concentrations, oxygen-18 and hydrogen-2 stable isotope signatures, nutrients (N and P), and ion concentrations also illustrate the reduced influence of the lake on marsh water chemistry with increasing distance. Moreover, the lakes dilution and flushing effect appeared to increase to a greater degree spatially in the marsh with increasing lake water levels from 2003 to 2005. Although the isotope results presented here are limited, in that only a small number of samples have been analyzed over a short time period of four months, the results did demonstrate how water source and mixing can vary over relatively small temporal and spatial scales in coastal wetlands. The relatively consistent oxygen isotope signature for the lake, as well as near-lake wetland sites both spatially and temporally, illustrated the high degree of lake and marsh water mass mixing at these sites, and the high degree of variation in isotope signature of inland marsh sites illustrated that reduced mixing occurs at these sites, and they are influenced to greater degree by surface water runoff and evaporation. The data would also further benefit from examination of spatial

and temporal patterns in chloride (Cl^-) concentrations in the lake and marsh before, during, and after the course of various magnitudes of lake seiche events.

Other studies in the Laurentian Great Lakes have also attributed spatial differences in the water chemistry of coastal wetlands to hydrological influences of adjoining lake. Trebitz et al. (2005) found that spatial differences in aquatic habitat within ten coastal marshes of Lake Superior were larger than differences amongst the marshes, and habitat patterns were strongly associated with morphology and hydrology. Further back-bay segments tended to demonstrate lower levels of seiche-induced water movement, and they were prone to high water temperatures and low dissolved oxygen levels. Increasing seiche inputs tended to homogenize habitat elements among wetland segments. Trebitz et al. (2004) noted that hydrologic connection of Lost Creek Wetland to Lake Superior, as well as to the upland watershed resulted in large spatial and seasonal variations in the hydrology and nutrient (N and P) dynamics of the wetland.

It is important to note that the larger deceases in N and P concentrations (50 to 90%) with decreasing distance from the lake, compared to Cl^- concentrations (30 to 40%), also indicates that other internal biochemical processing of N and P is occurring in the marsh, helping the marsh to act as a sink rather than source of these nutrients to the adjoining lake. While this points to N and P retention in the marsh, the data do not enable conclusions to be drawn on the processes of retention. In general, nutrient retention can result from many processes in wetlands including uptake by primary producers (algae and macrophytes), adsorption and burial in the sediments, and, in the case of N, bacterial denitrification.

The differences in the water chemistry between the connected and isolated sites could be attributed to several factors. For instance, Goldsborough and Wrubleski (unpublished) noted that

isolated ponds in Delta Marsh had clear water, more submerged vegetation, and invertebrates compared to the rest of the marsh, which was largely attributed to a lack of surface water connection and access of Common Carp (*Cyprinus carpio* L.) to these sites. These differences could also be the result of the influence of surface water connection and lack thereof on internal biochemical processes. For instance, the lower phytoplankton chlorophyll *a* concentrations and higher water clarity in the isolated sites compared to the connected sites could associated with the higher biomass of submerged macrophytes in the isolated sites (Appendix C), which can compete with phytoplankton for light and nutrients, as well as decreased turbidity levels from reduced wind and wave resuspension of sediments (Jeppesen et al. 1997, Scheffer 1998). The lower N and higher P concentrations in the isolated marsh sites compared with the connected sites could also be associated with high rates of N uptake my algae and vegetation as well as permanent losses to the atmosphere by denitrification, coupled with high rates of microbial decomposition and internal recycling of P, and a general lack of a permanent sink for P (Jeppesen et al. 1997, Scheffer 1998, Moss 2001, Gächter and Müller 2003, Sødergaard et al. 2003, Poe et al, Dunn and Reddy 2005).

Overall, the data show that surface water connection of Delta Marsh to Lake Manitoba, the spatial location of marsh sites relative to distance from the lake via water course, and annual changes in lake and marsh water levels interact to influence the hydrology and water quality of the marsh. Water chemistry in the marsh varied predictably with the relative extent of surface water connection and distance to Lake Manitoba, as well as inter-annual changes in water level and hydrology. This appears to be primarily an effect of water exchange between Lake Manitoba and Delta Marsh, as the data trends show that surface water connection to the lake and resulting inflow of water from Lake Manitoba had an important dilution and flushing effect on the marsh,

as the concentrations of nutrient (N and P) and dissolved ions (conductivity) decreased with decreasing distance to Lake Manitoba. Although not examined in this study, these seiche-induced water movements may also play an important role in reducing water stagnation (Trebitz et al. 2002). In general, the sites located closest to the lake were influenced to the greatest degree by the flushing and dilution effect of the lake, and this effect increased in spatial extent in the marsh with increasing water levels.

Lake Manitoba is the only year-round permanent source of water to Delta Marsh, and collectively this study illustrates the important influence the lake has on the structure and function of Delta Marsh. Goldsborough and Wrubleski (unpublished) concluded that a number of complex interacting variables were resulting in the degradation of Delta Marsh, including nutrient enrichment from the surrounding landscape and stabilized hydrology of Lake Manitoba. Due to the intimate link between lake and adjoining marsh hydrology and water chemistry, it can be expected that when the natural hydrological regime of a lake is altered (i.e. hydroelectric development, flood control, etc.) the hydrology of the adjoining coastal wetland(s) are also affected. This hydrological connection can also be altered when natural surface water connections between coastal wetlands and their adjoining lakes are altered by shoreline development. These types of hydrological alterations have been shown to effect the hydraulic residence time and biochemical processing in coastal wetlands of the Laurentian Great Lakes (Carter 1997, Keough et al. 1999, Trebitz et al. 2002, Morrice et al 2004). The natural hydrological connectivity of coastal wetlands to their adjoining lakes therefore needs to be considered when evaluating the vulnerability of coastal wetlands to anthropogenic hydrological changes on adjoining lakes, as well as during efforts to restore coastal wetlands following hydrological alterations.

Chapter 6: Results –Algal Nutrient Limitation

Introduction

Physical, chemical and biological processes in coastal freshwater wetlands have been shown to vary amongst different systems, and within systems (Keough et al. 1999, Grosshans 2001, Trebitz et al. 2002, Morrice et al. 2004, Sierszen et al. 2009, Trebitz 2006, Wilcox 2007, Wilcox et al. 2008, Gatham and Burton 2011, Wilcox 2012), with their water chemistry further complicated by varying inputs of water and nutrients from the watershed and/or adjoining lakes (Wetzel 2001, Trebitz et al. 2002, Morrice et al. 2004, Lotze et al. 2006; Trebitz 2006, Trebitz et al. 2007, Wilcox et al. 2007, Diaz and Rosenberg 2008, Morrice et al. 2011, Trebitz et al. 2011). Changes in the magnitude of water inputs may also affect water residence time which, in turn, influences water chemistry parameters, including nutrient cycling and availability (Wold and Hershey 1999, Trebitz et al. 2002, Morrice et al. 2004, Trebitz et al. 2004, Sierszen et al. 2006, Trebitz 2006, Morrice et al. 2011), with nutrient availability and retention increasing with increased hydrological residence time.

To date, the majority of studies examining the effect of water column nutrient concentrations on algal nutrient limitation (N and/or P) and biomass in shallow prairie wetlands have focused on phytoplankton (Allan and Kenney 1978, Barica et al. 1980, Kadlec et al. 1986, Barica 1990, Murkin et al. 1991, Waiser and Robarts 1995, Detenbeck et al. 2002), with relatively little study of periphytic (benthic) algae (Hooper-Reid and Robinson 1978, Murkin et al. 1991, Goldsborough and Robinson 1996, Kiers-North 2000), despite their important role in wetland ecosystems. Periphytic algae can represent a substantial component of primary

production in wetland systems due to abundant colonizable substrata (i.e., macrophytes and sediments), and high subsurface irradiance levels in these shallow systems (Robinson et al. 1997). Due to their often high biomass in wetlands (Goldsborough and Robinson 1996), periphytic algae can contribute significantly to key wetland functions such as nutrient cycling, while representing a key food source for consumers in wetland systems (Lamberti 1996; Sierszen et al. 2004). Periphyton biomass, often measured by its chlorophyll *a* content, has been found by previous studies to be a good bioindicator of human-induced water quality degradation (Fairchild et al. 1985, Gibeau and Miller 1989, Scrimgeour and Chambers 1997, McNair and Chow-Fraser 2003). McNair and Chow-Fraser (2003) concluded that variations in benthic and plankton algae biomass amongst 24 coastal wetlands, located in the Laurentian Great Lakes, were good indicators of varying water quality and environmental conditions, ranging from nutrient-poor marshes with clear water and abundant submerged macrophytes to eutrophic marshes with turbid water and scarce submerged macrophytes. Unlike aquatic macrophytes, benthic algae obtain a great deal of their nutrients from the water column, so they are ideal organisms with which to monitor nutrient enrichment from watershed land use, as they respond quickly to nutrient additions due to their high productivity and rapid turnover rates (McNair and Chow-Fraser 2001, Lavoie et al. 2004). As a result, they are also commonly used in *in situ* bioassays to determine the nutrient limitation status of aquatic systems, including tributaries (Gibeau and Miller 1989, Scrimgeour and Chambers 1997, Wold and Hershey 1999), lakes (Fairchild et al. 1985, Fairchild et al. 1989) and wetlands (McCormick et al. 1996, McNair Chow-Fraser 2003, Scott et al. 2005).

This chapter builds on the results presented in Chapter 4 (hydrology) and Chapter 5 (water chemistry) and examines spatial and temporal patterns in the nutrient limitation of algae (specifically N and P, and periphyton) in Delta Marsh, as influenced by spatial and temporal

patterns in marsh water chemistry and hydrology. Variations in nutrient limitation status (N and/or P) of periphytic algae were assessed by the response of periphyton growth on *in situ* nutrient (N and P) diffusing substrata (NDS) (Fairchild et al. 1985). Nutrient (N and P) water column concentrations and molar ratios in the study area, presented in Chapter 5, were also examined for comparison with the NDS results. The NDS release nutrient(s) over a given time period, and the resultant periphyton biomass on the NDS can be measured as the response. Nutrient-diffusing substrata have been used in a number of lentic and lotic systems, and have been found to provide economical and effective bioassays of the algal response to nutrients; they stimulate significant algal growth and reduce sampling variability due to their uniform size (Pringle 1987, Fairchild et al. 1988, Wold and Hershey 1999, Tank and Dodds 2003, Scott et al. 2005). Multiple nutrients can be manipulated at one time and they allow for the assessment of temporal and spatial changes in nutrient conditions over long periods of time (~1 to 8 weeks).

Molar ratios of N:P in the water column have also been used to predict limiting nutrient(s) in aquatic systems. The most commonly used ratios are total nitrogen (TN) to total phosphorus (TP), dissolved inorganic N (DIN) to TRP (total reactive P, or orthophosphate PO_4), and DIN:TP. When examining N:P ratios in aquatic systems, the traditional Redfield (1958) N:P molar ratio of 16:1 has long been used as the indicator of ideal cellular N and P concentrations, with P being interpreted as being limiting above 16:1, and N limiting below 16:1. Other more recent studies have found that molar TN:TP and DIN:TRP ratios of <20:1 (<10:1 by weight) better indicate N-limited conditions, with molar ratios >33:1 (>15:1 to 20:1 by weight) indicating P-limited conditions, with ratios between the two indicative of conditions with no nutrient limitation (Sakamoto 1966, Smith 1979, Schanz and Juon 1983, Morris and Lewis 1988, Axler et

al. 1994, Francoeur et al. 1999, Guildford and Hecky 2000). For DIN:TP, molar ratios >9:1 are associated with P limitation, and ratios <3:1 are associated with N limitation.

Over the course of the study Lake Manitoba experienced its lowest and second highest water levels on the record since regulation, with the lowest average water levels of 247.0 m ASL (range 247.0 to 247.6 m ASL) occurring in 2003, and the second highest water levels of 247.8 m ASL (range 247.3 to 247.8) occurring in 2005 (Figure 1.3). These large natural differences in water levels allowed for examination of spatial and temporal differences in the algal nutrient limitation in the marsh relative to varying water levels on Lake Manitoba.

The location of study is described in detail in Chapter 1.3. The study sites and experimental design are described in Chapter 3.1, and the detailed sample collection and analysis, and data analysis methods are described in Chapter 3.2 and 3.3, respectively. The results of the hydrology studies conducted in the marsh are presented Chapter 4, and the results of the water chemistry studies are presented in Chapter 5.

Results

NDS diffusion rate experiments

The laboratory diffusion rate experiments indicated that all the NDS treatments diffused nutrients throughout a 30-day period, and diffusion rate decreased at a log-linear rate. N release rates declined from approximately 65 μmol/cm^2/day on day 1 to 30 μmol/cm^2/day on Day 30, and P from 7.0 μmol/cm^2/day on day 1 to 1.9 μmol/cm^2/day on Day 30. Similar results were obtained by Pringle and Bowers (1984), Fairchild et al. (1985), and Gibeau and Miller (1989). Neither N nor P was detected diffusing from the control treatment NDS. While diffusion gradients between the NDS and the water column at the sample sites would be less than the laboratory experiments with distilled deionized water (reduced gradient of N and P concentrations from NDS to water column at sites, as N and P present in water column but not in de-ionized water), the laboratory experiments provide important estimates of upper level diffusion rates of N and P from the treatment NDS over a 30-day period. Examination of the NDS following a three-week incubation period at select sites confirmed that measurable concentrations of N and P still remained in the NDS, and that N and P concentrations decreased over the incubation period. Concentrations of N were reduced 60 to 70% and P from 70 to 80% over the 21-day period, with 2 to 3 g/L of NO_3-N and 0.3 to 0.4 g/L of PO_4-P remaining in the NDS. This corresponded to diffusion rates of approximately 22 μmol/cm^2/day for N and 1.7 μmol/cm^2/day for P. In comparison to water column concentrations of P and N at sample sites during the study (0.02 to 1.2 mg/L TP, and 0.20 to 14 mg/L TN; data presented in Chapter 5), newly prepared NDS contained at least 500x N and 1200x P, and used NDS following the three week incubation contained at least 150x N and 250x P. This indicates a strong positive

concentration gradient (from NDS to water column) which was likely continually present at the sample sites throughout the NDS experiment. Data are presented in Appendix B.

Spatial and temporal trends in N:P water column ratios

As shown in Chapter 5, water column N and P concentrations and molar ratios were highly variable in Delta Marsh (Figure 5.10 to 5.23). Annually, mean TN:TP values ranged from 47 to 133 in 2002, 2003 and 2004, and from 40 to 53 in 2005, indicative of P limitation in all years, with the exception of Portage Creek South which had a ratio of 19 in 2005, indicating N limitation (Figure 5.18 to 5.23). Mean molar ratios of DIN:TRP had the highest range in values, of 1 to 25 in 2004 (indicating N-limitation and no limitation), followed by 1 to 15 in 2005 (indicating N-limitation), and 1to 5 in 2003 (indicating N-limitation). Mean DIN:TP values were highest in 2003, ranging from 1 to 12, and lower in both 2004 and 2005 ranging from 1 to 5 (indicating N-limitation to no limitation). The gradient in decreasing N:P ratios with decreasing distance to the lake in all study years, indicated a general gradient transitioning from N limitation to P limitation with decreasing distance to the lake. Mid-marsh sites had the greatest potential to experience high annual and inter-annual variability amongst P and N limitation, and even lack of nutrient limitation.

Compared to connected marsh sites, isolated sites had lower molar ratios of TN:TP (15 to 68; indicative of N to P limitation), DIN:SRP (1 to 9; indicative of N limitation), and DIN:TP (1 to 3; indicative of N limitation; Figure 5.18 to 5.23). As at connected sites, higher mean TN:TP, DIN:SRP, and DIN:TP ratios were observed in isolated sites in 2004 compared to 2005.

In Lake Manitoba, mean molar ratios of TN:TP (112 to 260), DIN:TRP (5 to 24) and DIN:TP (3.4 to 5.5) were higher in the lake compared to the marsh (Figure 5.18 to 5.23). The

high TN:TP ratio values consistently indicated the potential for P limitation in the lake annually and inter-annually, whereas DIN:SRP indicated greater potential for variable N limitation and no nutrient limitation, and DIN:TP for no nutrient limitation. Compared to the marsh, there was less annual variation in N:P molar ratios in the lake.

Algal response to nutrient limitation bioassays (NDS)

From 2002 to 2005 the NDS experiments conducted in Delta Marsh indicated that periphyton limitation by N occurred most often (71%) in the marsh, followed by no nutrient limitation (24%), and co-limitation by N and P (5%); Figure 6.1 to 6.16; Appendix H). In the marsh, periphyton biomass was never stimulated by the addition of P alone. In isolated marsh sites, periphyton biomass was consistently stimulated by N alone in all sites and in all study years (Figure 6.13 to 6.16). In connected marsh sites periphyton biomass was N limited at the majority of samples sites in 2002 and 2003, and 2005, whereas in 2004 periphyton biomass at the majority of sites experienced a lack of nutrient limitation (i.e. biomass on all three of the nutrient treatments (N, P, and N+P) was not significantly greater from the control). Seasonally in 2002, the majority of connected sites in the east section of the marsh were stimulated by N in May and August, with stimulation by both N+P in June and July at sites in closest proximity to the Lake (Simpsons Bay, Cadham Bay East, Delta Channel), and a lack of nutrient limitation occurred at the two furthest sites (Portage Creek North and Portage Creek South). In the west section of the

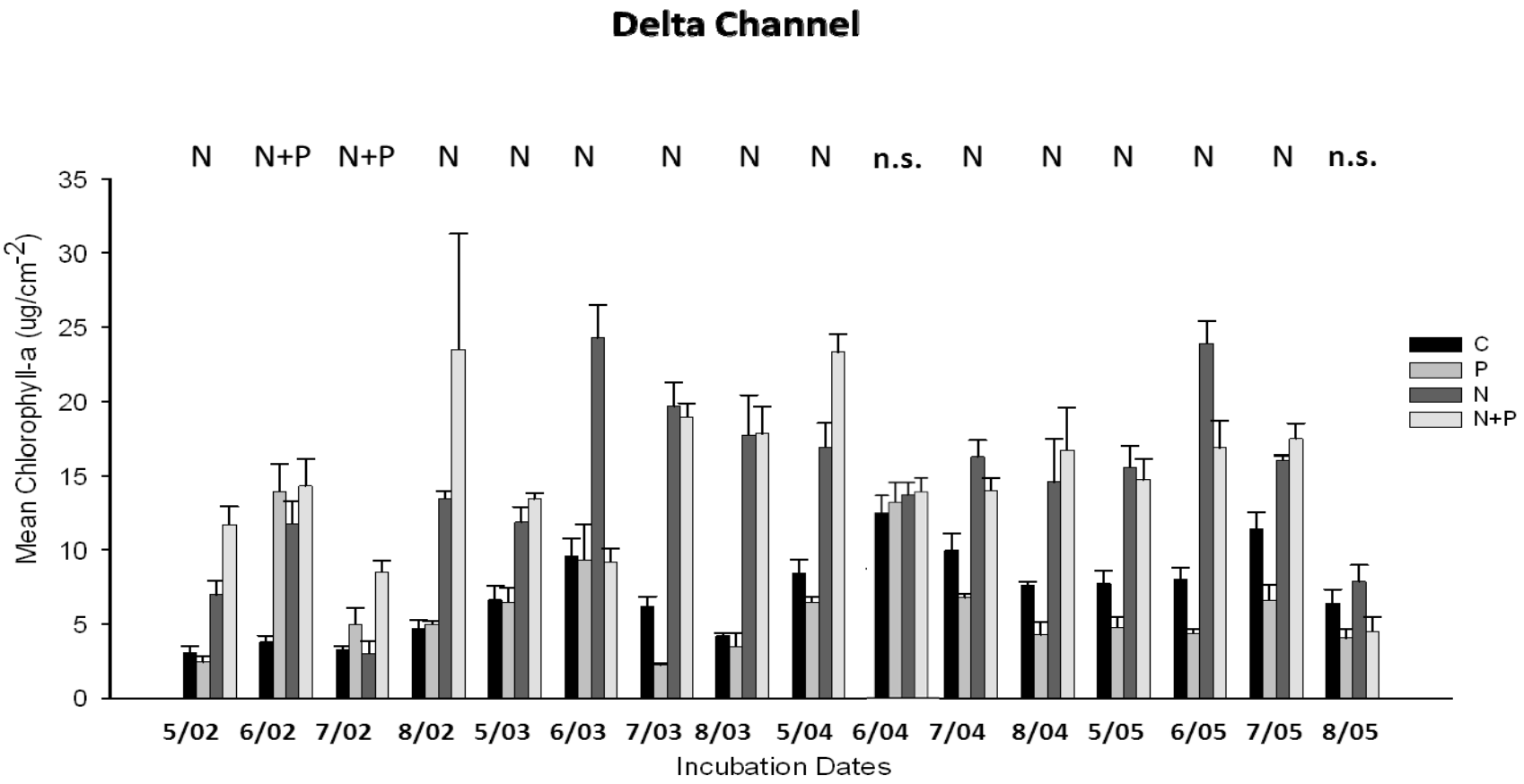


Figure 6.1 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Delta Channel, from 2002 to 2005. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

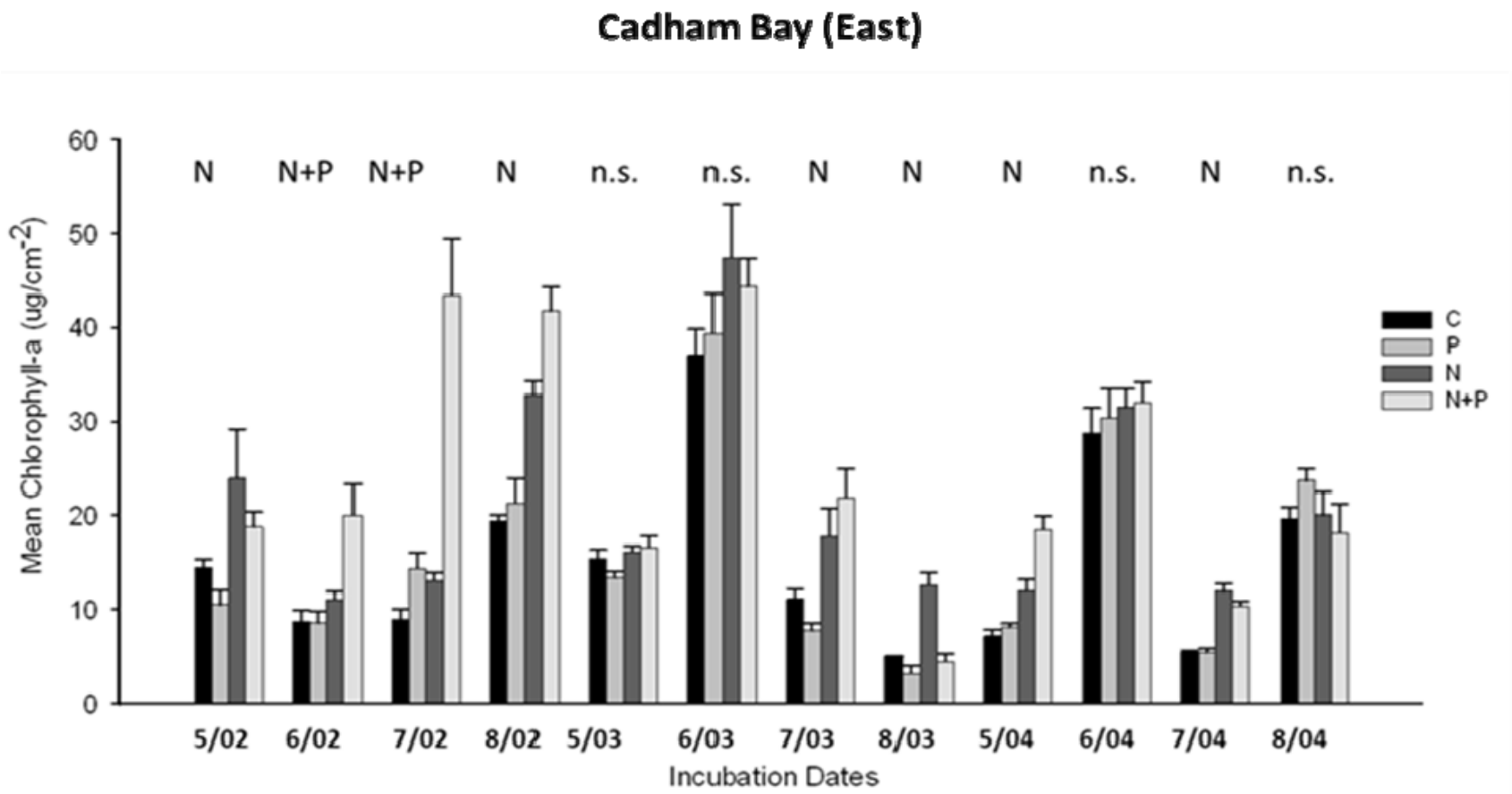


Figure 6.2 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Cadham Bay east, from 2002 to 2004. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine significant differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

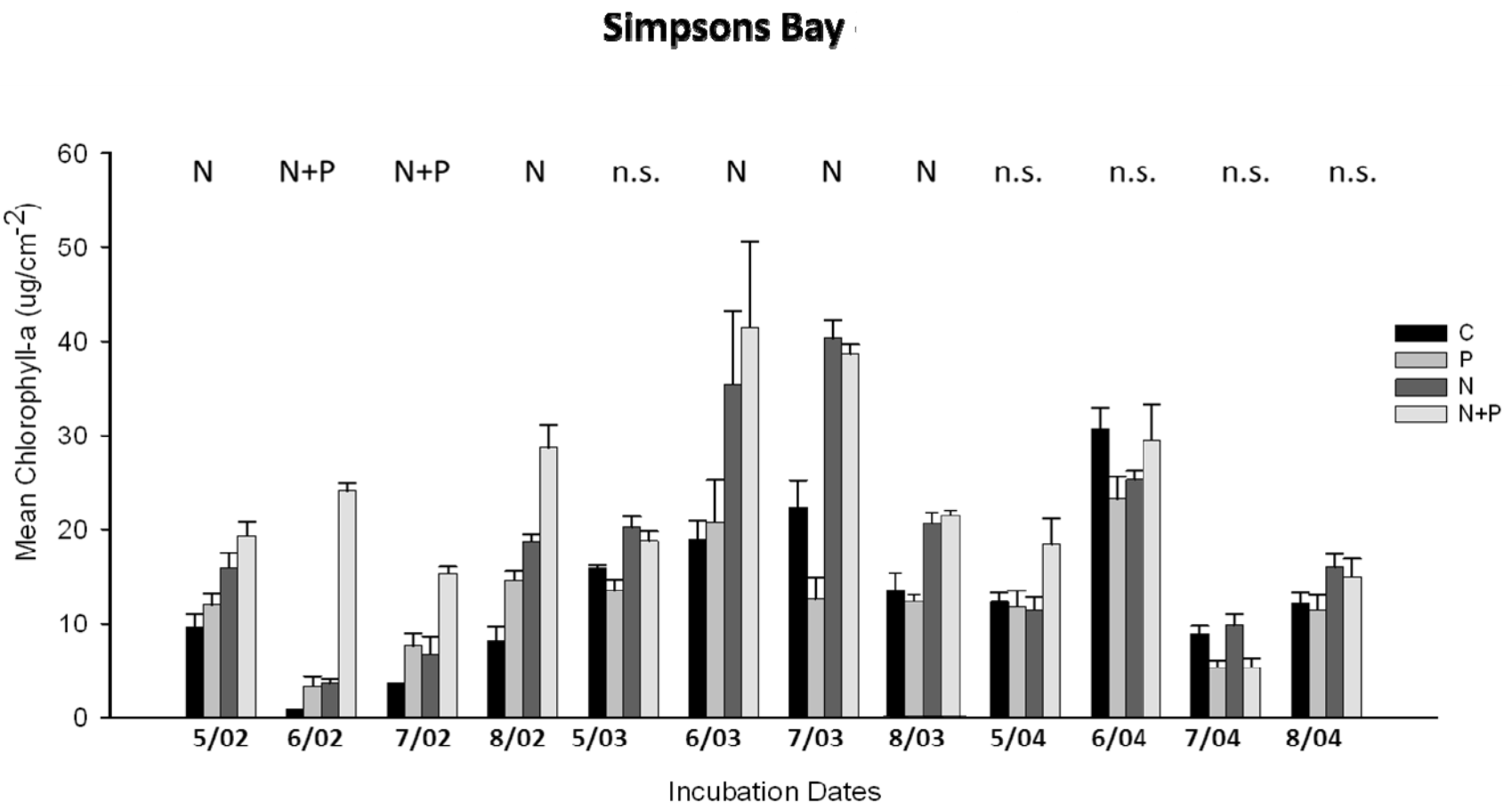


Figure 6.3 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Simpsons Bay, from 2002 to 2004. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

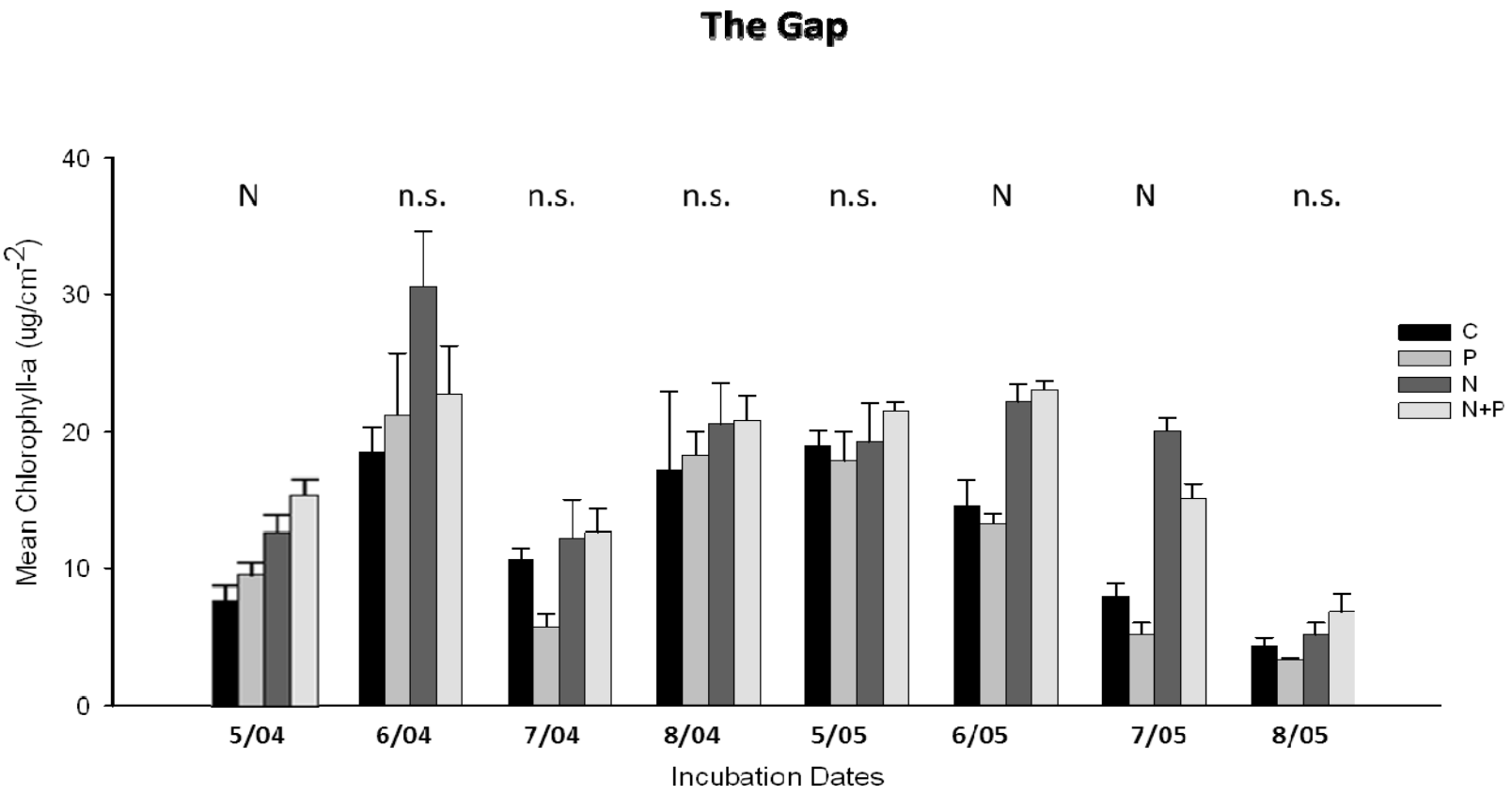


Figure 6.4 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in the Gap, from 2004 to 2005. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N= nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

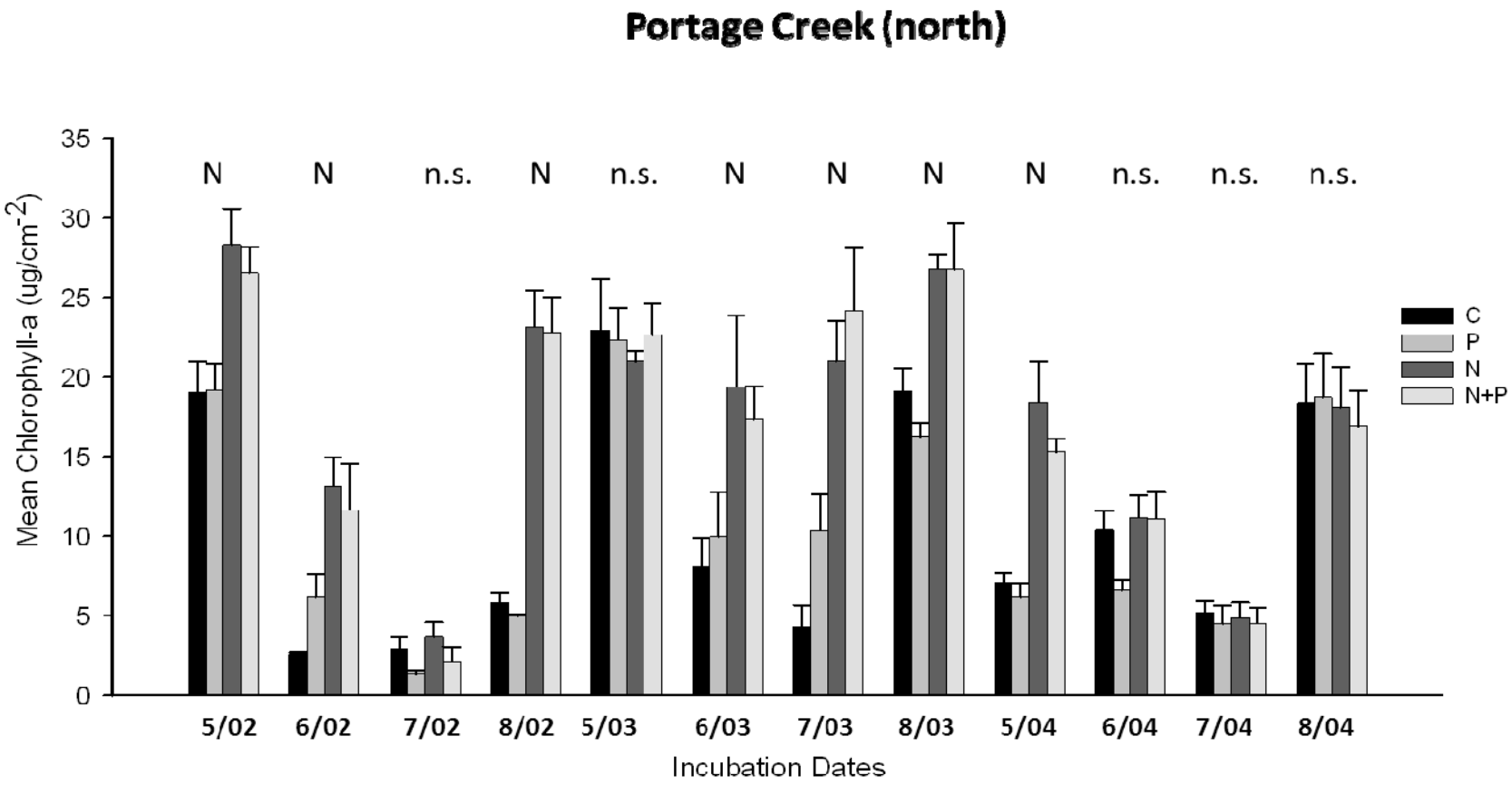


Figure 6.5 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Portage Creek north, from 2002 to 2004. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

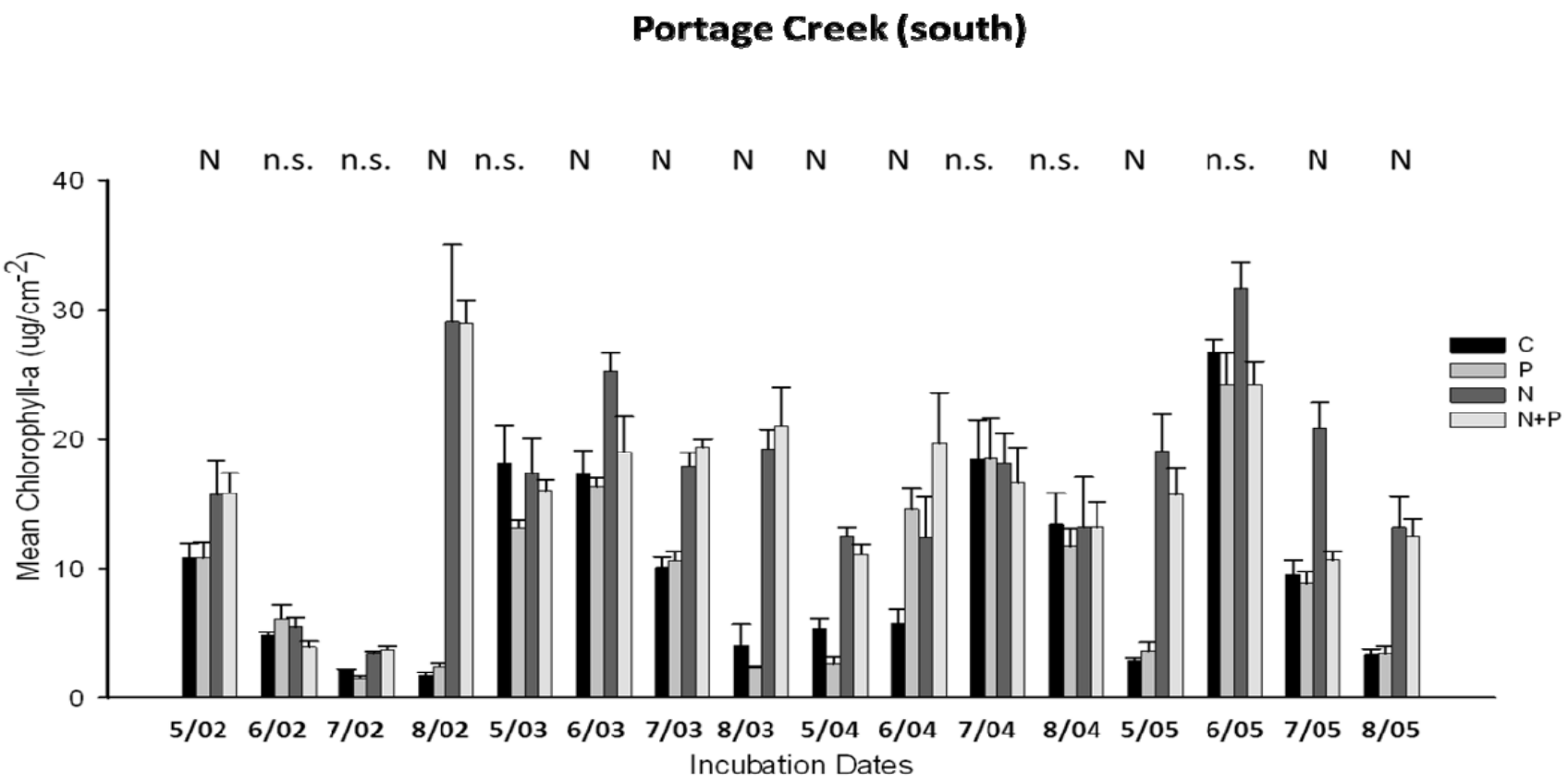


Figure 6.6 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Portage Creek south, from 2002 to 2005. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

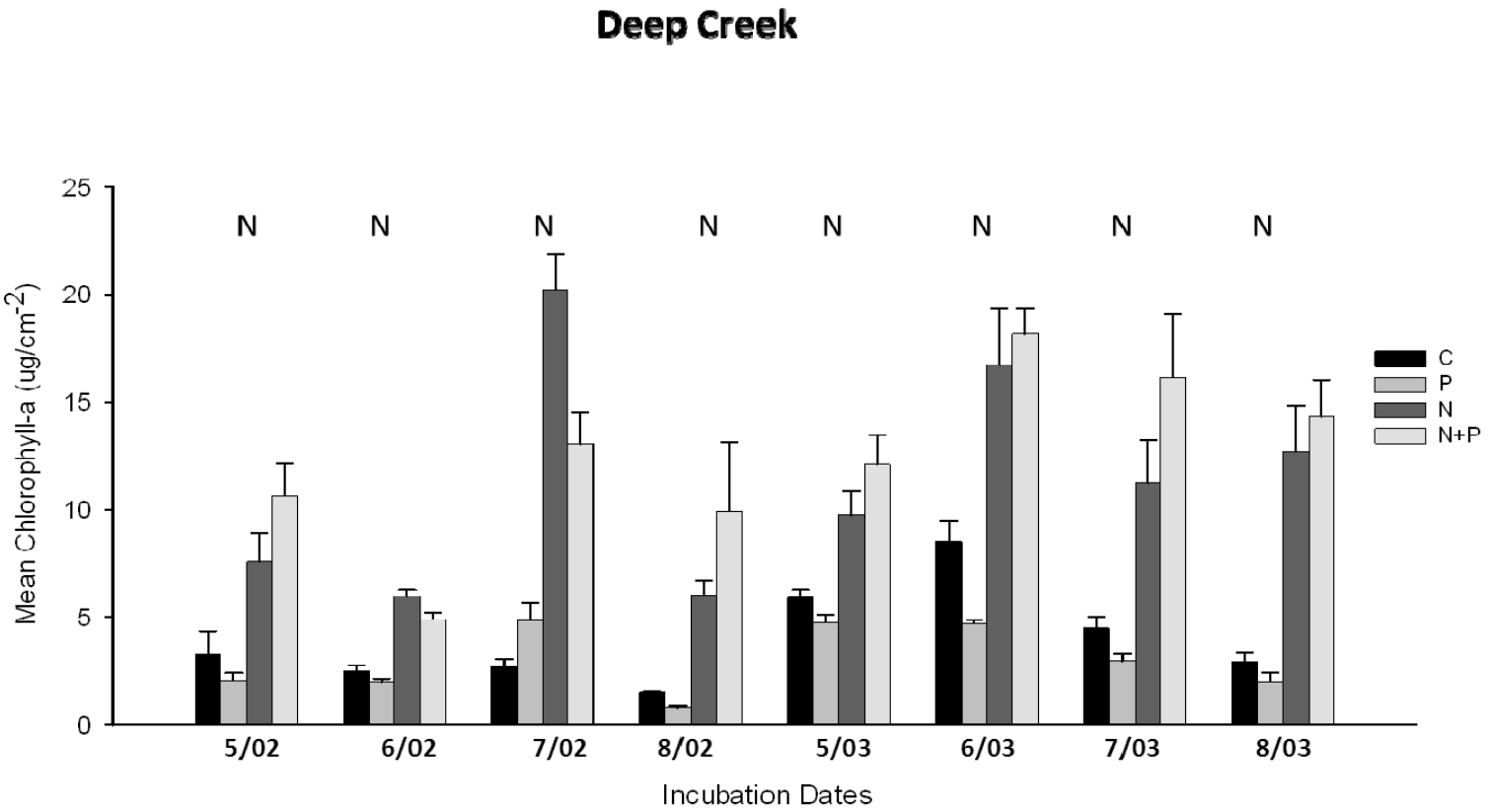


Figure 6.7 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Deep Creek, from 2002 to 2003. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

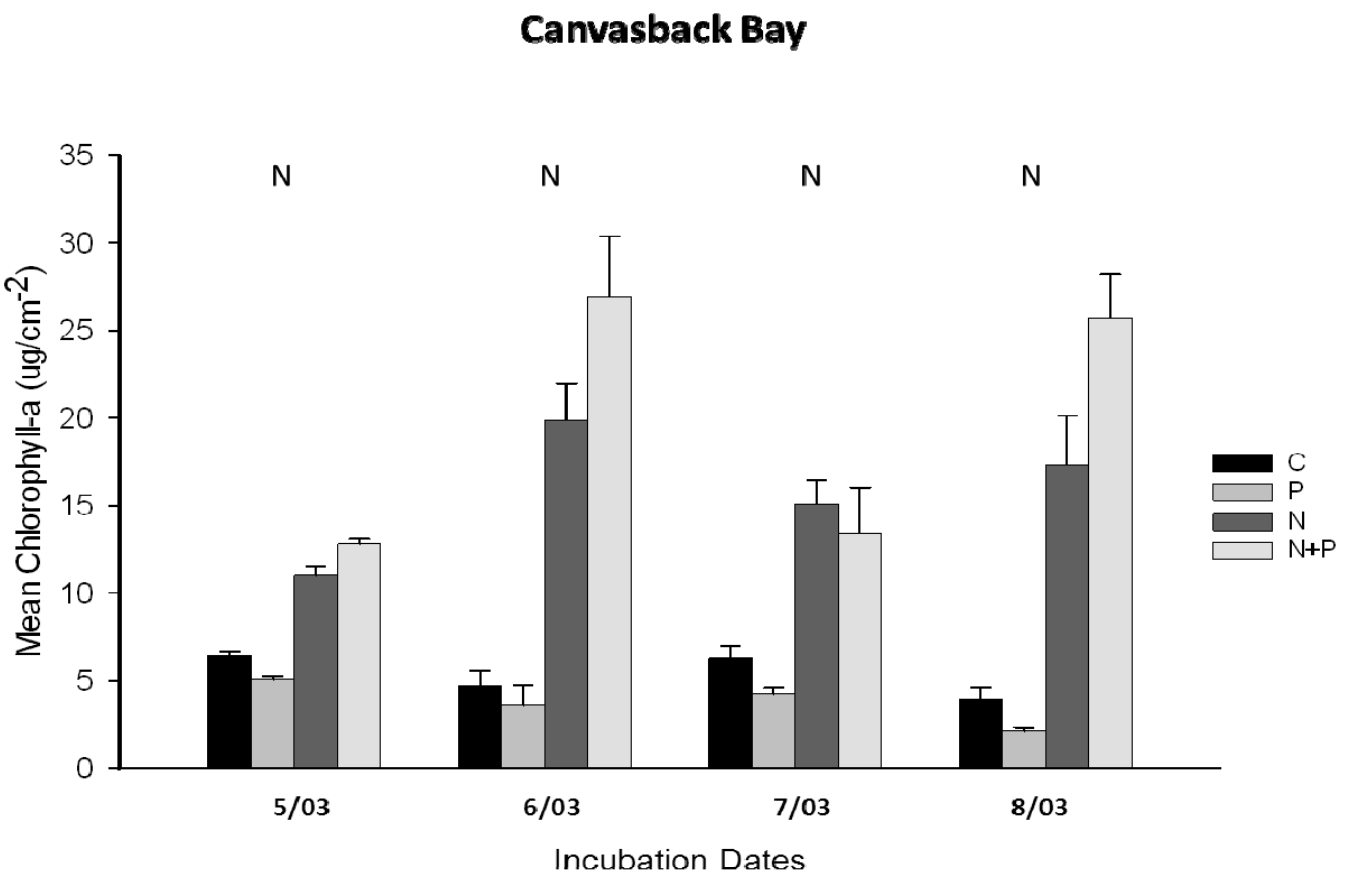


Figure 6.8 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Canvasback Bay in 2003. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

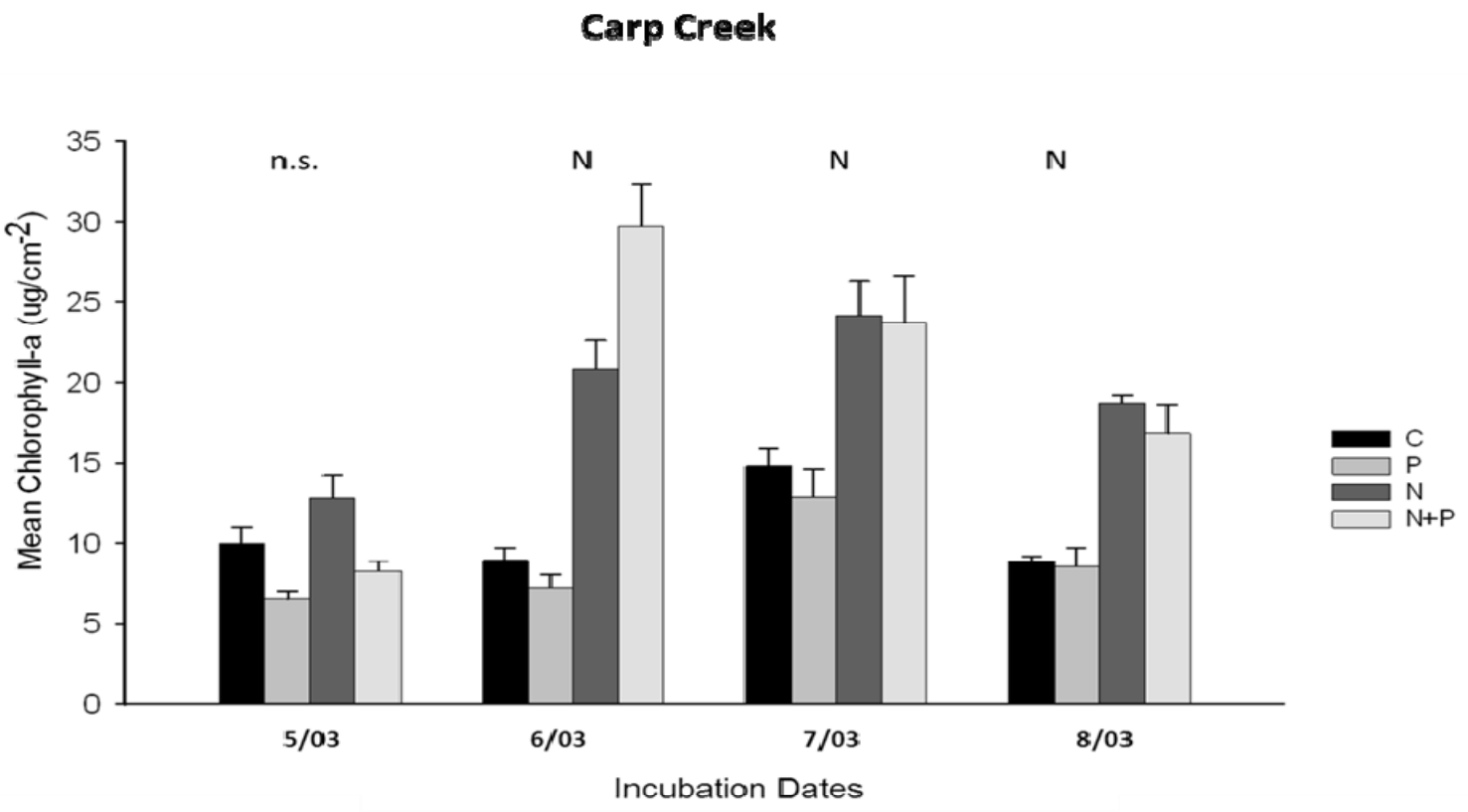


Figure 6.9 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Carp Creek, in 2003. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

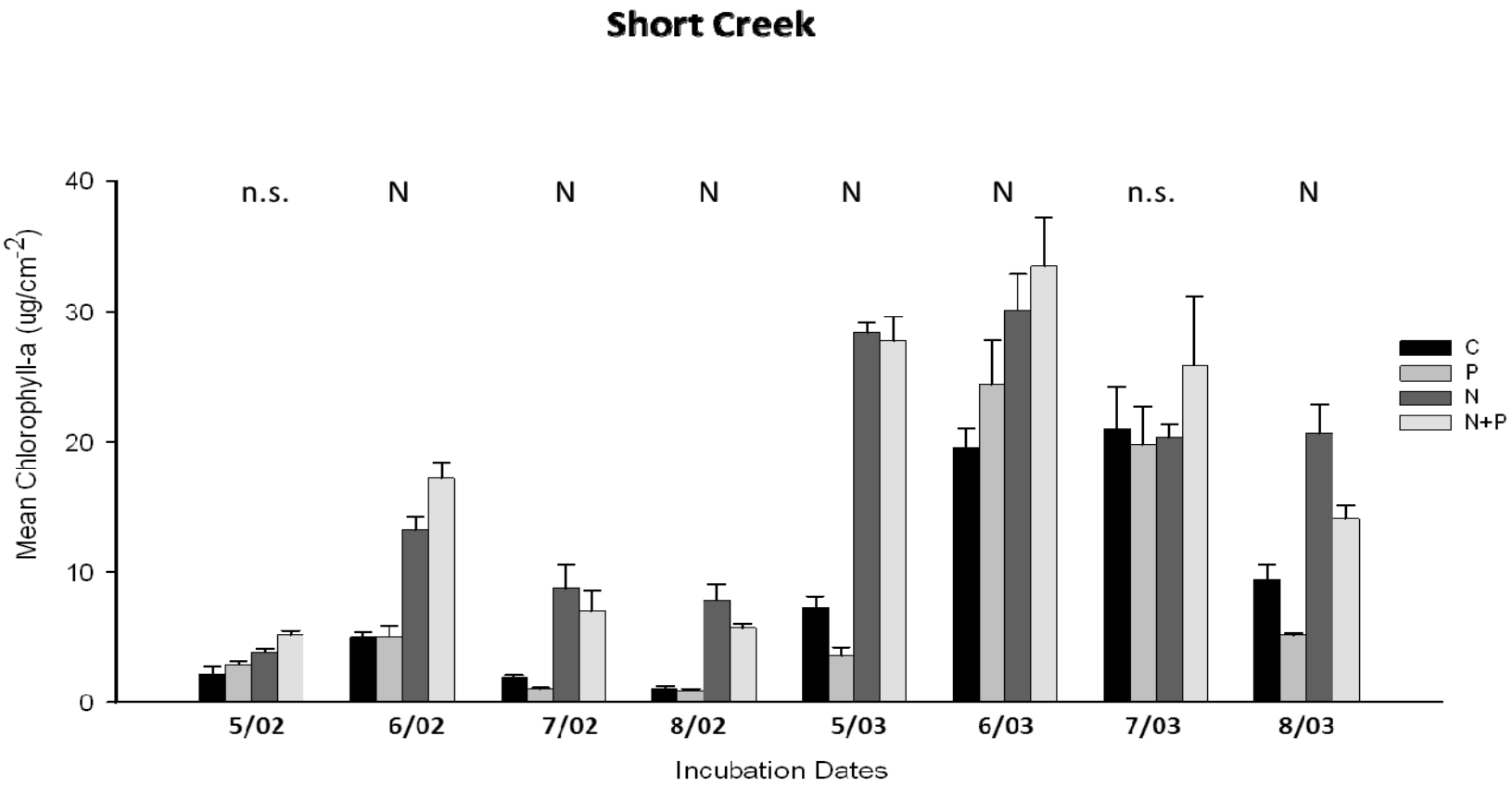


Figure 6.10 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Short Creek, from 2002 to 2003. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

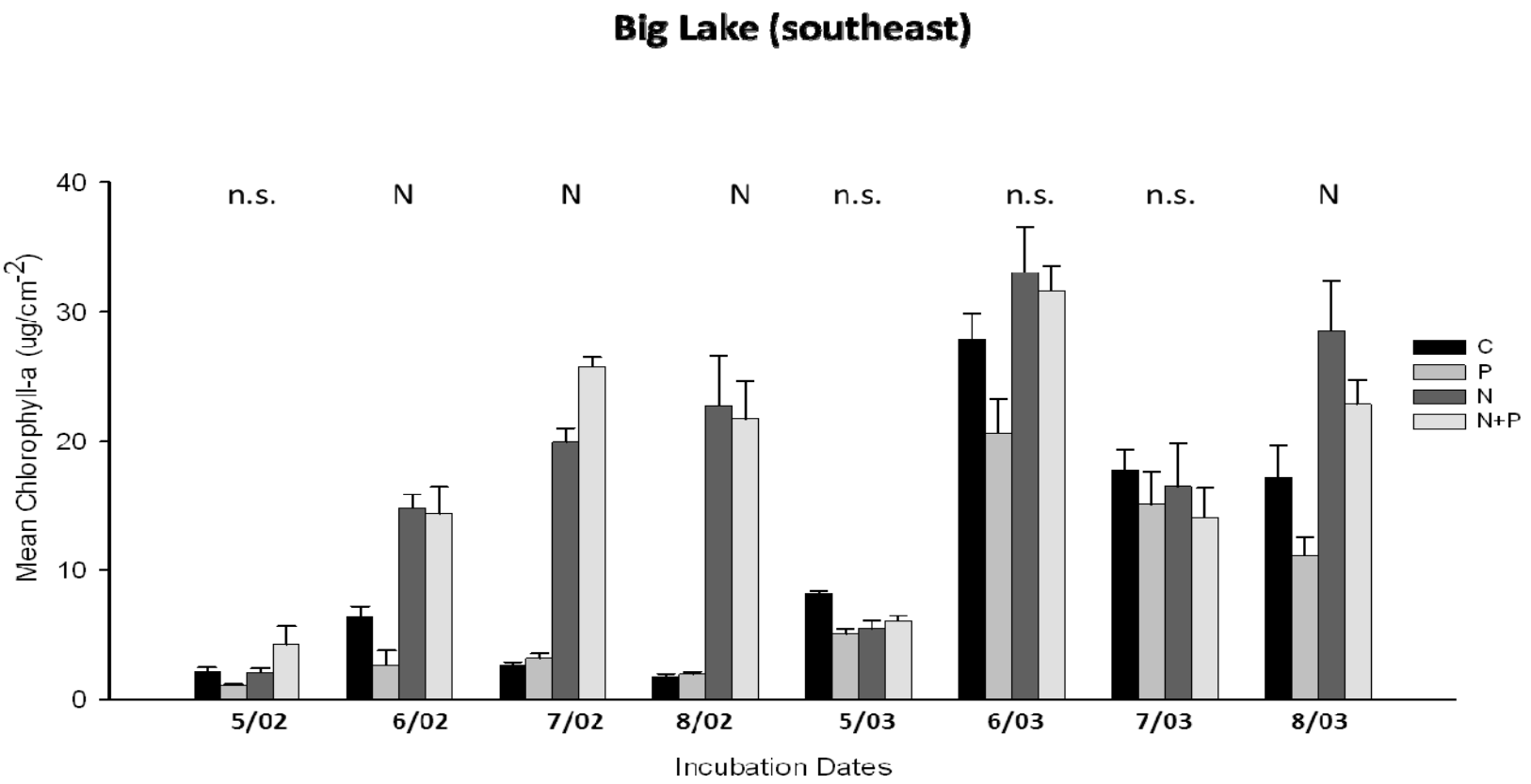


Figure 6.11 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Big Lake southeast, from 2002 to 2003. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

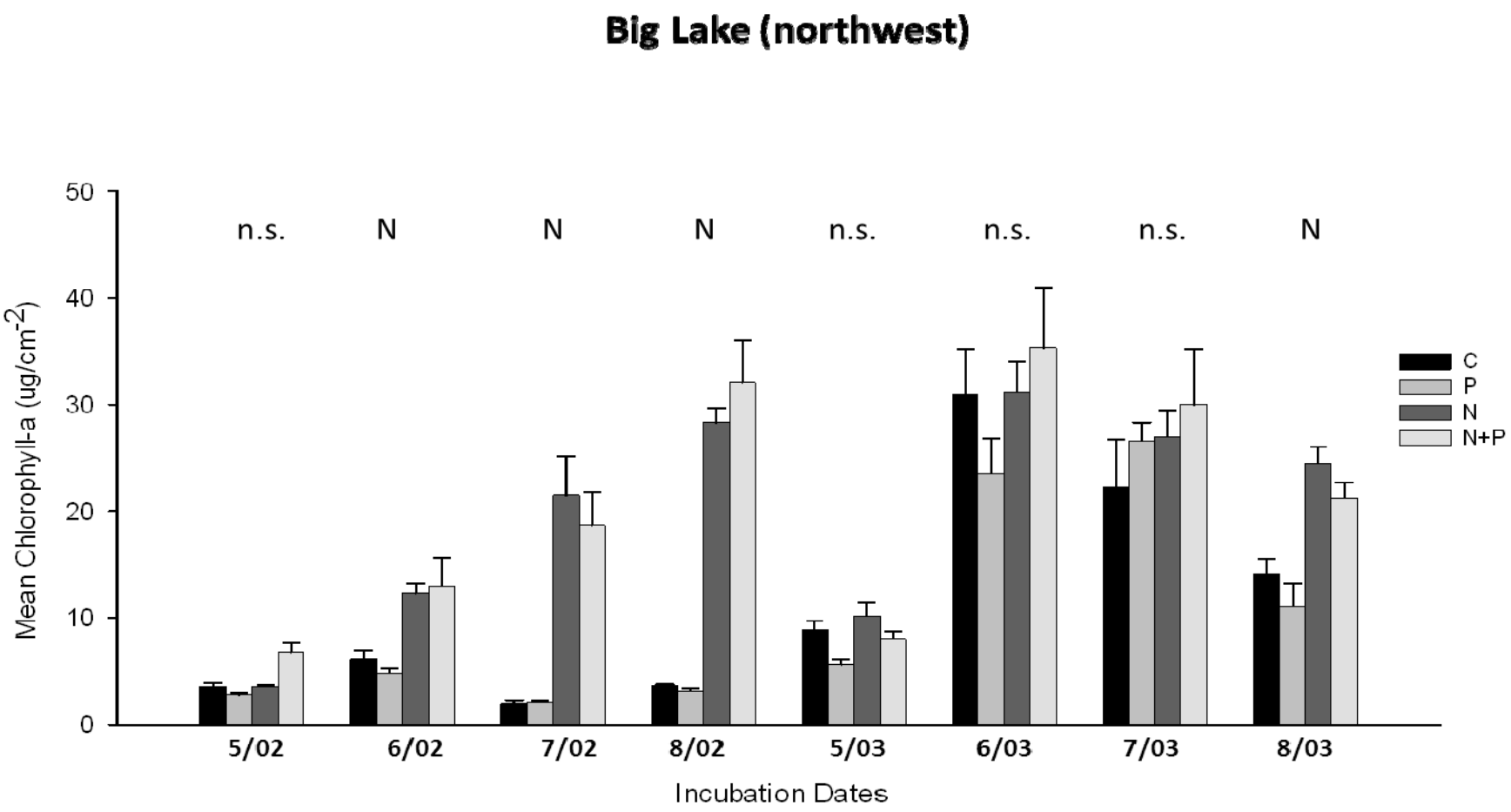


Figure 6.12 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Big Lake northwest, from 2002 to 2003. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

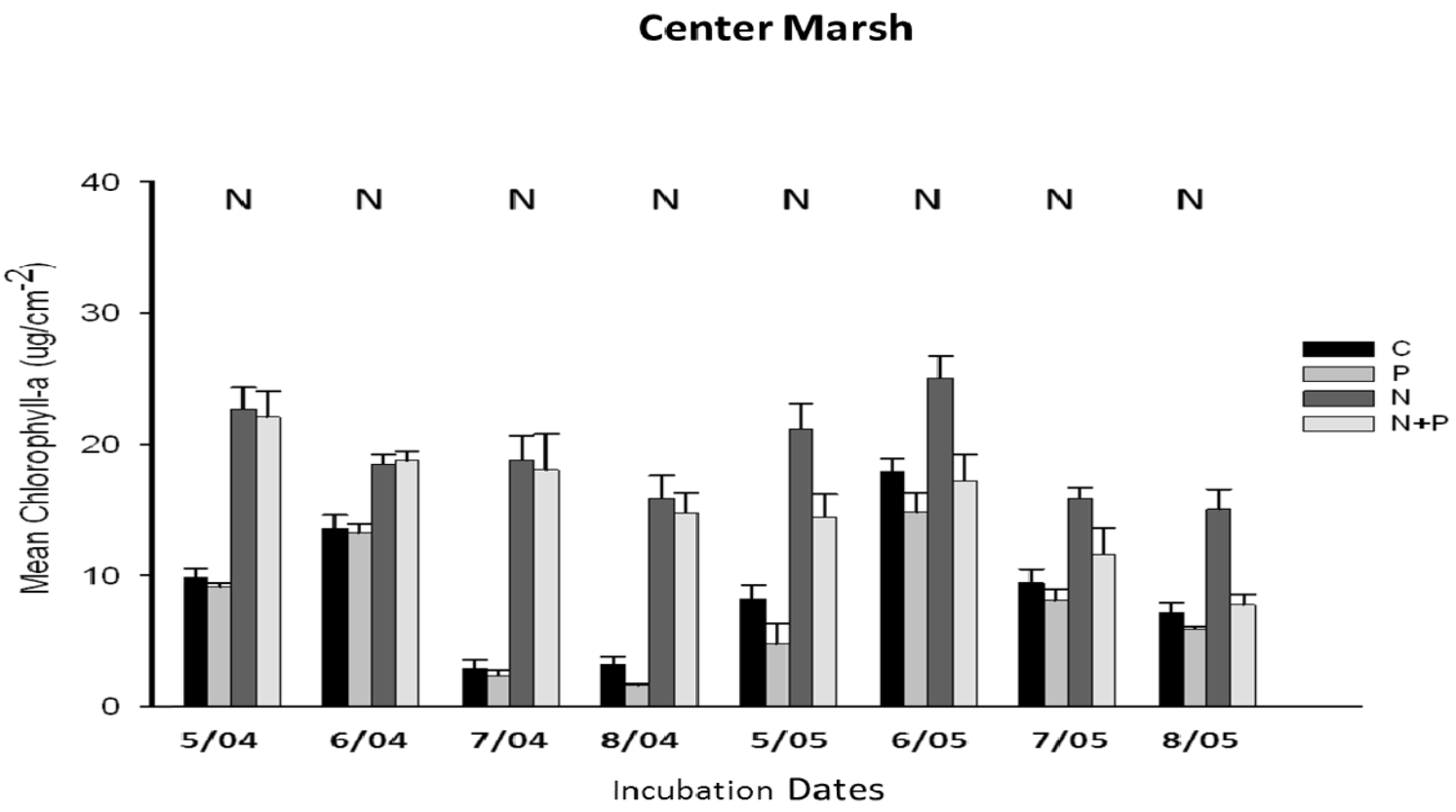


Figure 6.13 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Center Marsh, from 2004 to 2005. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

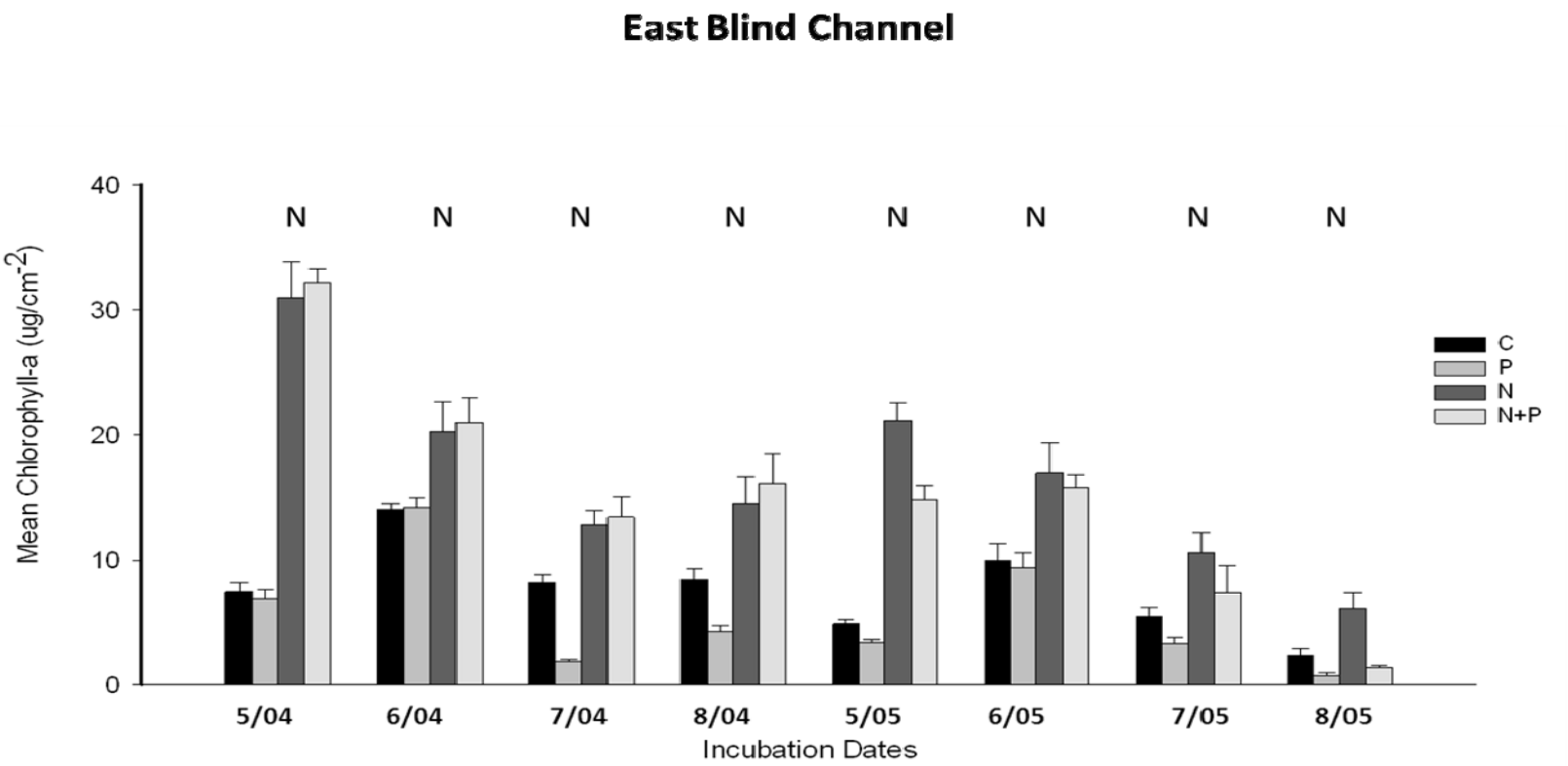


Figure 6.14 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in East Blind Channel, from 2004 to 2005. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

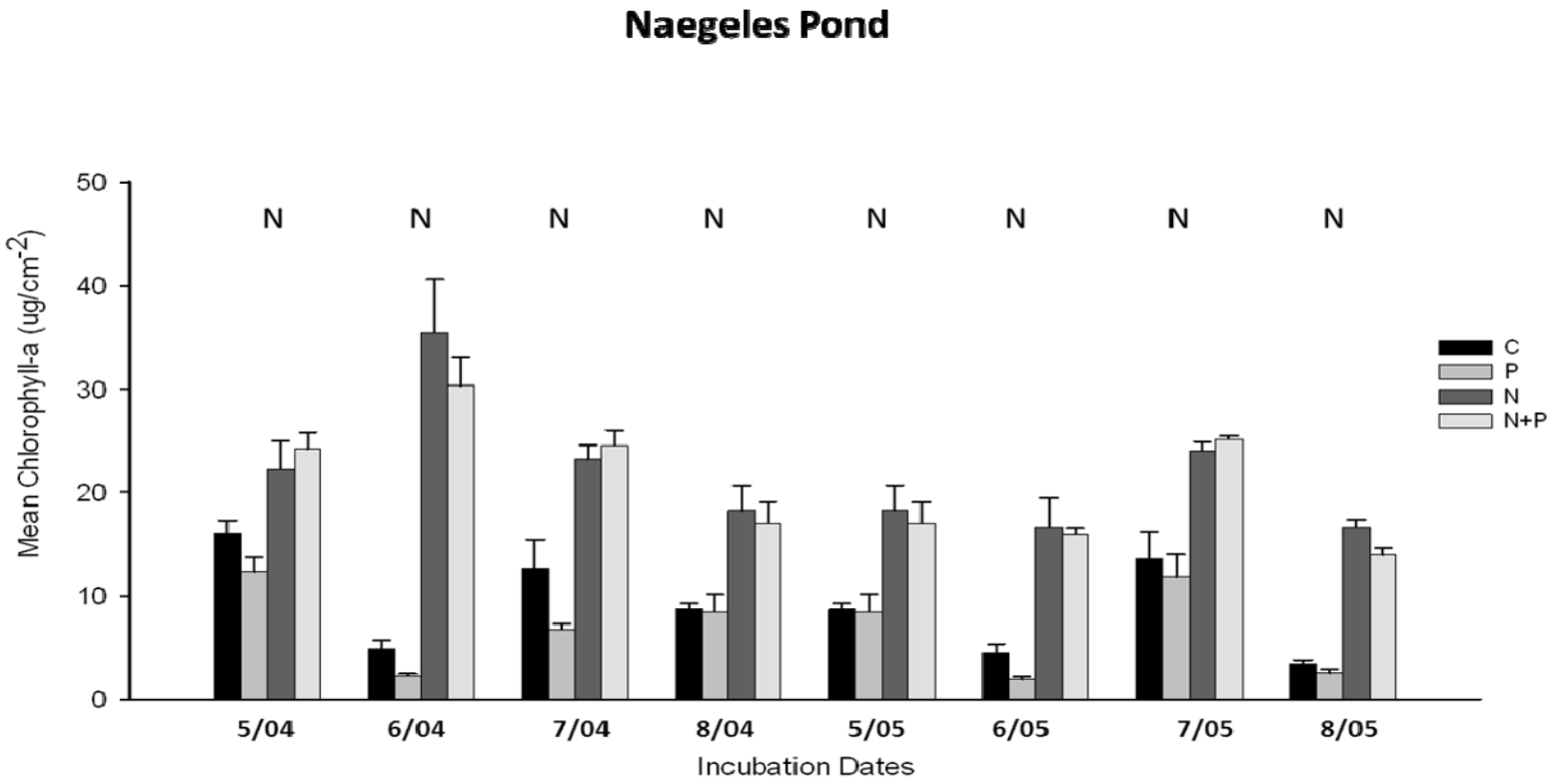


Figure 6.15 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Naegeles Pond, from 2004 to 2005. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

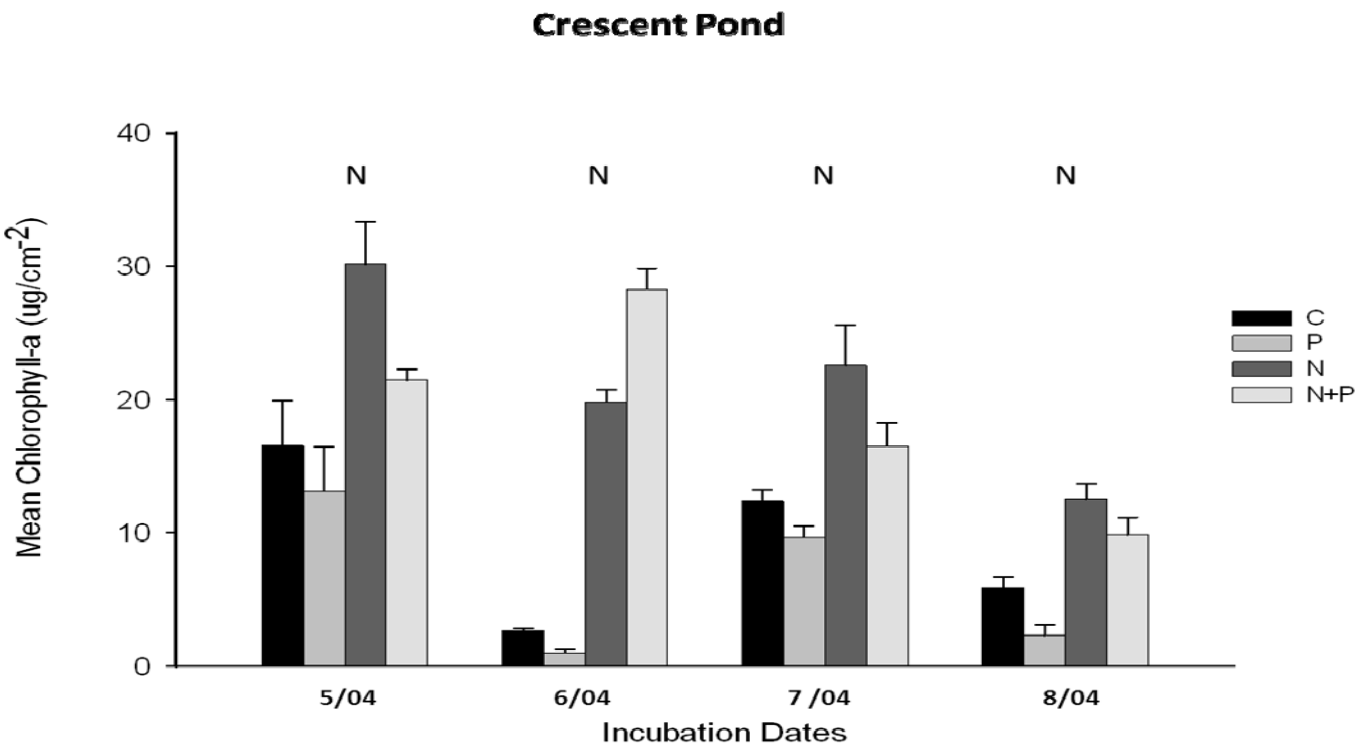


Figure 6.16 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Crescent Pond, in 2004. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

marsh, N limitation predominated from May to August, with the exception of the sites furthest from the lake (i.e. Short Creek, Big Lake North West, and Big lake South East) in May 2002, which exhibited no nutrient limitation. In 2003, periphyton was not limited by N or P in May, but were limited by N in June through August 2003, with the exception of sites in the west section of the Marsh and furthest from the lake (i.e. Big Lake South, Big Lake North West and Short Creek), which experienced a lack of nutrient limitation from May to July 2003. The opposite was true in 2004, as most connected sites were stimulated by N in May, with no nutrient limitation in June through August 2002. In 2005, the connected marsh sites were predominately limited by N alone in May through August. In Lake Manitoba, periphyton biomass was variably stimulated by N+P, and N and P alone between study months and years, with predominant limitation by N+P in 2002, P in 2003, and N in 2004 (Figure 6.17, Appendix H).

In both east and west connected marsh sites when nutrient limitation was present, seasonal trends in the Nutrient Limitation Status Index (NLSI) values followed a similar pattern (Figure 6.18 and 6.19). In 2002 and 2003, NLSI values generally increased through the season (May to August) at the sites closet to the lake (i.e. Delta Channel, Cadham Bay, Deep Creek, Canvasback Bay and Carp Creek). In the west section of the marsh in 2002, NLSI values at the other sites (i.e. Big Lake southeast, Short Creek, and Big Lake northwest) followed a similar pattern and generally increased through the season, however in 2003, Short Creek experienced high NLSI values in May, June, and August, with no nutrient limitation in July; and Big Lake southeast and Big Lake northwest only experienced nutrient limitation in August. In 2002, at the rest of the sites in the east section of the marsh, NLSI values were more variable, with values peaking in May and August at Portage Creek south, and peaking in June followed by a slight decrease through to August at Portage Creek south and Simpson's Bay; and in 2003, NLSI

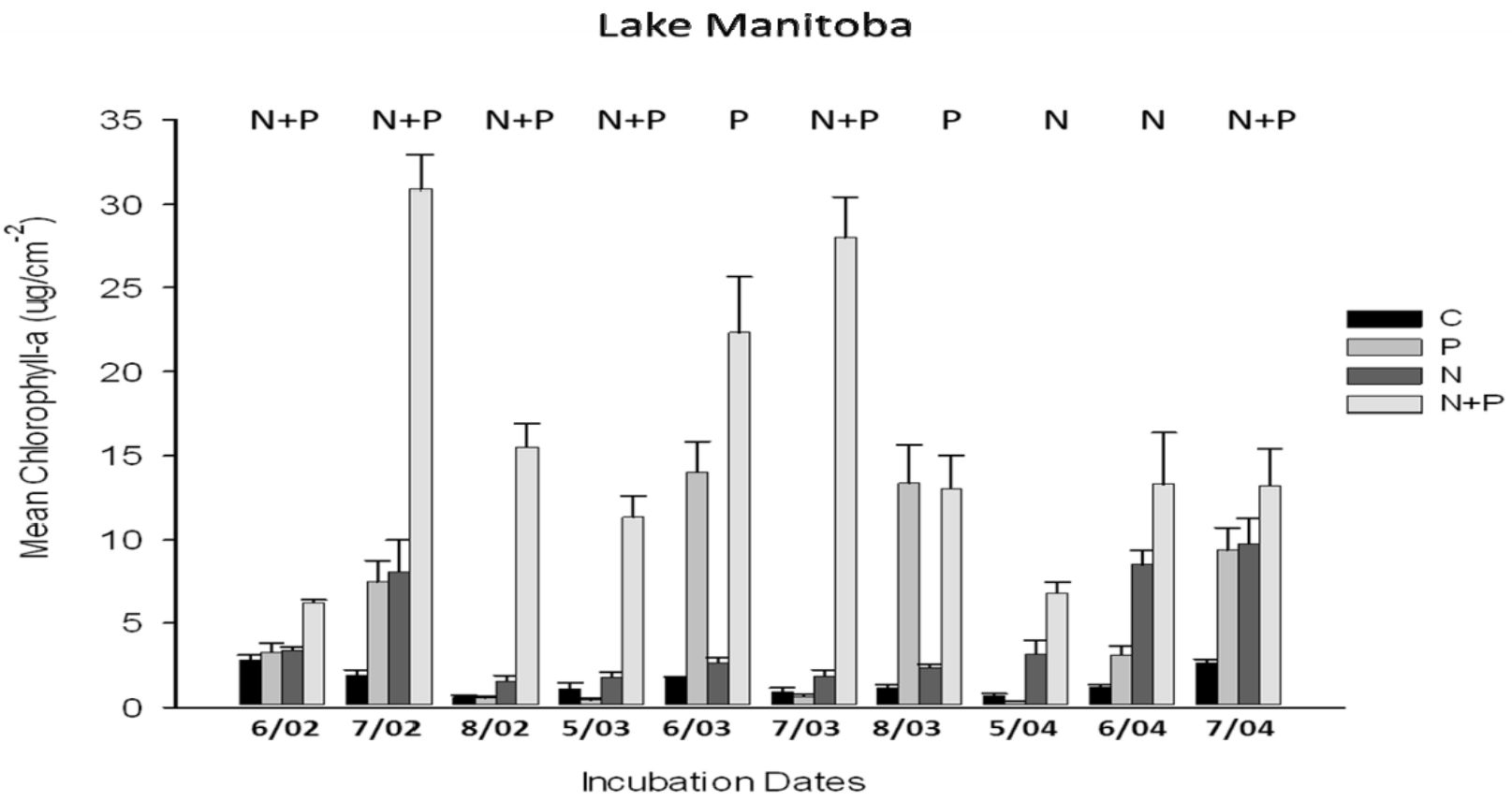


Figure 6.17 Response of mean periphyton biomass (total chlorophyll *a* ± SE) to NDS nutrient treatments in Lake Manitoba, from 2002 to 2004. ANOVA and Tukey-Kramer pairwise multiple comparison tests ($p \leq 0.05$) were used to determine differences among means (log transformed values), which was interpreted as a significant response to nutrient limitation. Letter above the bars indicates the nutrient limitation (N = nitrogen, P = phosphorus, N+P = nitrogen and phosphorus, n.s. = no nutrient limitation). More detailed results of the statistical analyses are presented in Appendix H. See Table 3.1 and Figure 3.1 for sample site locations and descriptions.

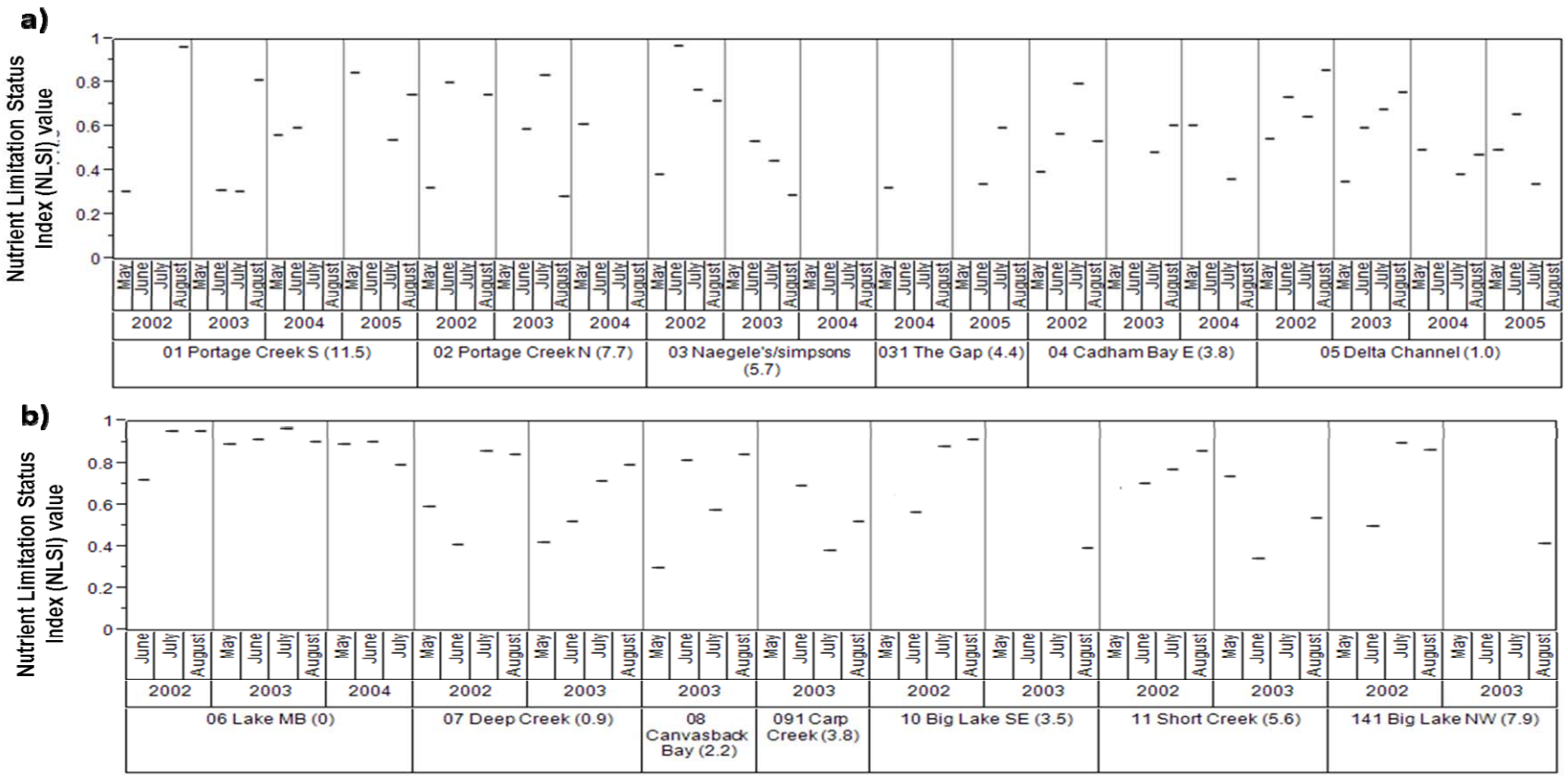


Figure 6.18 Nutrient Limitation Status Index (NLSI) values at sample sites in Delta Marsh and Lake Manitoba, from 2002 to 2005. The higher the value (0.0 to 1.0 scale), the greater the degree of nutrient limitation. Sites are plotted by distance from the lake via surface water flow (km), year and month; a) east section of connected Marsh, b) Lake Manitoba and west section of connected marsh. See Table 3.1 and Figure 3.1 for site descriptions.

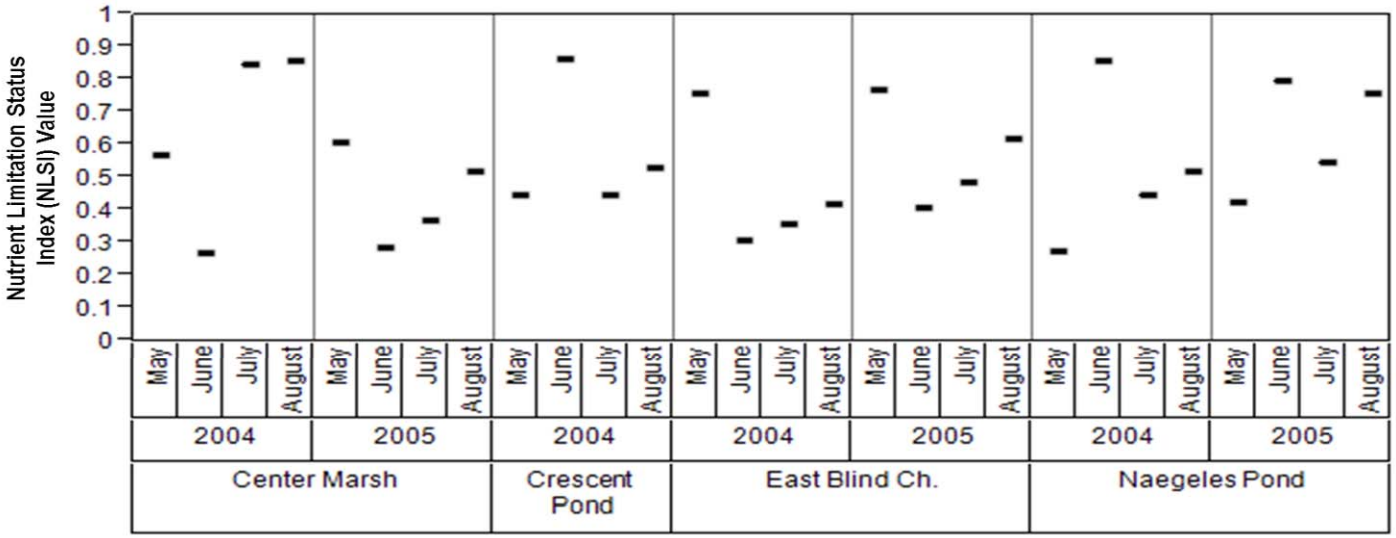


Figure 6.19 Nutrient Limitation Status Index (NLSI) values in isolated sample sites in Delta Marsh, from 2004 to 2005. The higher the value (0.0 to 1.0 scale) the greater the degree of nutrient limitation. Sites are plotted by site, year and month. See Table 3.1 and Figure 3.1 for site descriptions.

values generally peaked in June and July at Portage Creek south, Portage Creek north and Simpsons Bay. In 2004 in the east section of the marsh, NLSI values at all sites generally peaked in the first half of the season (i.e. May and June), with the exception of the Simpsons Bay site, which did not experience any limitation. In 2005, NLSI values were highest in May and August at Portage Creek south, whereas the opposite was true at The Gap and Delta Channel sites, where NLSI values peaked in June and July, respectively.

In 2004 and 2005, seasonal trends in NLSI values were also evident in the isolated marsh sites. Center Marsh and East Blind Channel experienced similar seasonal trends with generally higher NLSI ratios in May compared to June, followed by a step-wise increase from June to August. In comparison, Naegeles Pond experienced the lowest ratios in May and the highest ratios in June, followed by a decrease to mid-range ratios in July, and an increase August. NLSI in Crescent Pond, examined in 2004 only, followed the same seasonal trend as Naegeles Pond, with the lowest ratio in May and highest ratio in June, followed by a decrease to mid-range ratios in July, a modest increase in August.

Mean periphyton biomass (chlorophyll *a*) on the control NDS treatment, a measure of background periphyton biomass conditions in the marsh, generally increased with increasing distance from Lake Manitoba in the east section of the marsh in 2002 and 2003 (note only significant in 2003; Figure 6.20). Although not significant, periphyton biomass generally decreased in the west section in 2003 and the east section in 2002 with increasing distance from the lake (Figure 6.20). In the connected west section of the marsh, annual mean periphyton chlorophyll *a* on the control NDS was higher in 2003 compared with 2002, and in the east annual variations varied by site (Figure 6.20).

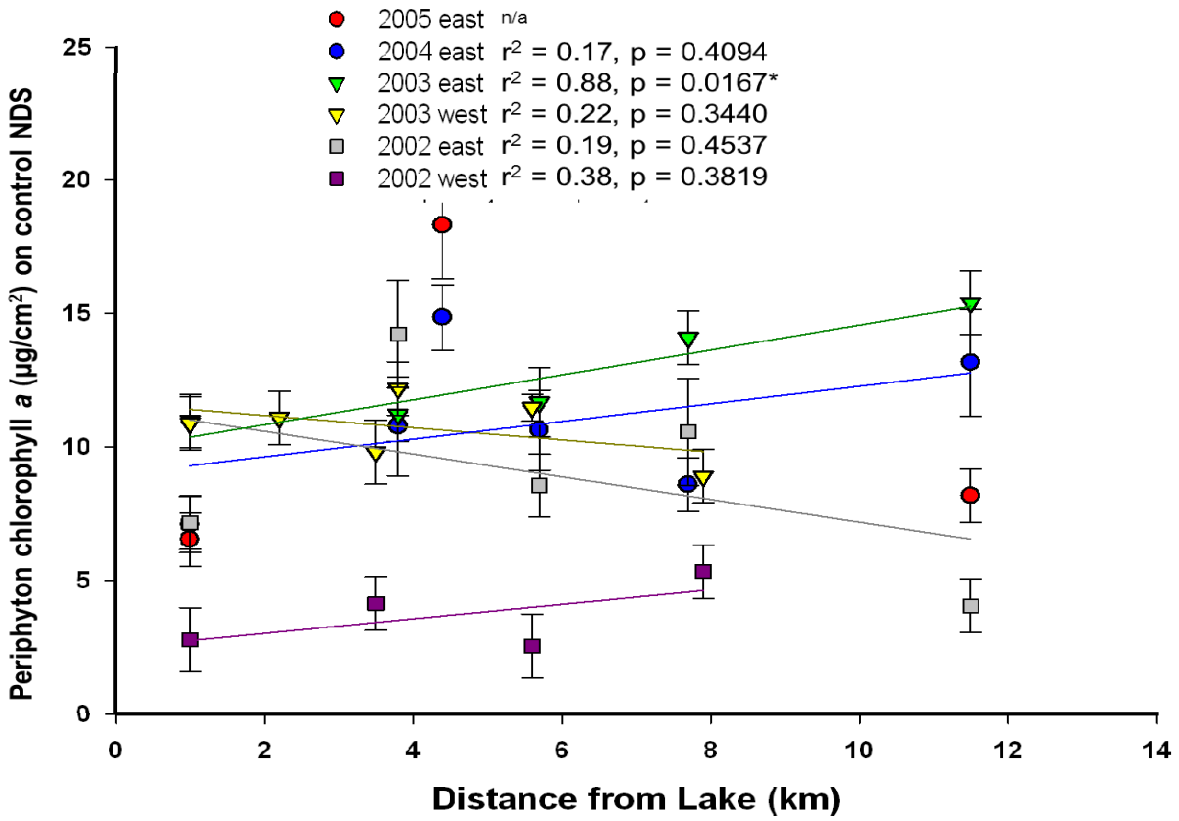


Figure 6.20 Mean (±SE) periphyton biomass (chlorophyll a µg/cm^2) on the control NDS treatment at connected sample sites in Delta Marsh plotted against distance from the lake (km), from 2002 to 2005. *Denotes a significant. 1 Data for 2005 only includes three points for significance of quadratic regression cannot be determined.

In the isolated sites, yearly differences in periphyton biomass on the control NDS also varied by site (Figure 6.13 to 6.15). In Center Marsh, mean periphyton growth was slightly higher in 2005 (7.3 µg/L) compared to 2004 (10.7 µg/L). The opposite was true in East Blind Channel and Naegeles Pond with mean values lower in 2005 (5.6 µg/L in both) compared to 2004 (9.5 µg/L and 10.6 µg/L, respectively).

No correlation was found between N and P water column concentrations and molar ratios and the results of the NDS bioassays. When nutrient limitation was present, resultant nutrient limitation status index (NLSI) values were regressed against water column concentrations of N (NH_4, NO_3, DIN, TN) and P (SRP-P, TRP) as well as molar ratios of TN:TP, DIN:TRP, and DIN:TP (Table 6.1, Figure 6.21 to 6.23). The number of significant responses of periphyton biomass to the N+P (5) and P (1) treatments in the marsh (2003 to 2005) were too low, so the resultant NLSI values for these treatments could not be compared statistically to water column concentrations and ratios of N and P. However, there were 103 significant responses of periphyton biomass to the N treatment, which allowed for comparison of the resulting NLSI values for N. The nitrogen NLSI values in connected sections of the marsh were found to be negatively correlated with water column DIN ($r^2 = 0.26$, $p = 0.0012$) and TN ($r^2 = 0.32$, $p = 0.0007$; Table 6.1, Figure 6.21), indicating increased growth on N-enriched treatments compared to the control substrata with decreasing water column N concentrations. Interestingly, periphyton growth on the control treatments at marsh sites showed the inverse trend and were positively correlated with water column DIN ($r^2 = 0.48$, $p < 0.0001$) and TN ($r^2 = 0.29$, $p<0.0001$), as well as weakly correlated with molar ratios of DIN:TP ($r^2 = 0.12$, $p < 0.0088$; Table 6.1 and Figure 6.22), indicating increased periphyton biomass on the controls with increasing N concentration and decreasing DIN:TP ratios. However, neither the response of

Table 6.1 Pairwise correlations (P≤0.05) between algal biomass response (as chlorophyll-a) on Log Control CHL (log chlorophyll *a* on control NDS), Log NLSI, Log Phyto CHL (log phytoplankton chlorophyll *a*), and water column nutrient concentrations (N and P) and nutrient molar ratios at connected sample sites in Delta Marsh, from 2003 to 2005. Significant correlations are bolded.

Dependant Variable	Independent Variable	Pairwise Correlation Coefficient	r^2	F-value	p-value
Log NLSI	**Log NO3-N**	**-0.43**	**0.17**	**7.5260**	**0.0095**
n = 45	**Log NH3-N**	**-0.47**	**0.23**	**10.3575**	**0.0028**
	Log DIN-N	**-0.51**	**0.26**	**12.4077**	**0.0012**
	Log TN	**-0.56**	**0.32**	**13.7343**	**0.0007**
	Log TRP-P	0.02	0.01	0.2381	0.2769
	Log TP	0.08	0.01	0.0478	0.8282
	Log TN:TP	-0.20	0.05	1.7684	0.2568
	Log DIN:TRP	0.17	0.02	3.0288	0.3221
	Log DIN:TP	-0.29	0.09	3.5944	0.0929
Log Control CHL	**Log NO3-N**	**0.31**	**0.13**	**9.7895**	**0.0026**
n = 66	**Log NH3-N**	**0.65**	**0.43**	**46.6938**	**<0.0001**
	Log DIN-N	**0.73**	**0.48**	**60.0448**	**<0.0001**
	Log TN	**0.53**	**0.29**	**25.0118**	**<0.0001**
	Log TRP-P	0.20	0.03	1.9251	0.1706
	Log TP	0.07	0.02	0.2881	0.5935
	Log TN:TP	0.17	0.09	1.2714	0.2017
	Log DIN:TRP	0.06	0.12	0.2640	0.0695
	Log DIN:TP	**0.34**	**0.13**	**7.3783**	**0.0088**
Log Phyto CHL	Log NO3-N	-0.11	0.01	0.0035	0.9531
n =68	Log NH3-N	-0.15	0.02	1.6337	0.2057
	Log DIN-N	-0.13	0.09	1.3771	0.2478
	Log TN	-0.12	0.01	0.9431	0.3352
	Log TRP-P	**0.29**	**0.09**	**5.9222**	**0.0179**
	Log TP	**0.57**	**0.36**	**30.2202**	**<0.0001**
	Log TN:TP	**-0.45**	**0.20**	**13.4889**	**0.0005**
	Log DIN:TRP	**-0.31**	**0.10**	**6.0654**	**0.0169**
	Log DIN:TP	**-0.26**	**0.09**	**5.5226**	**0.0474**

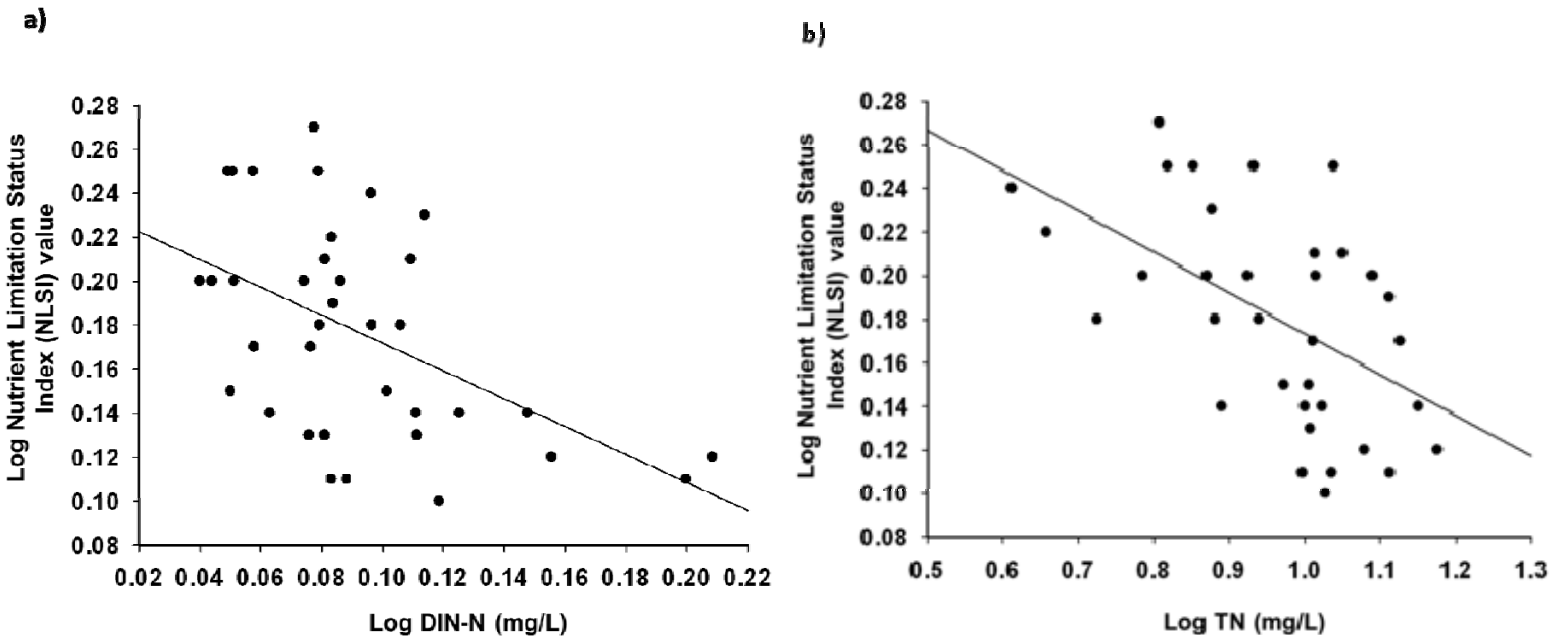
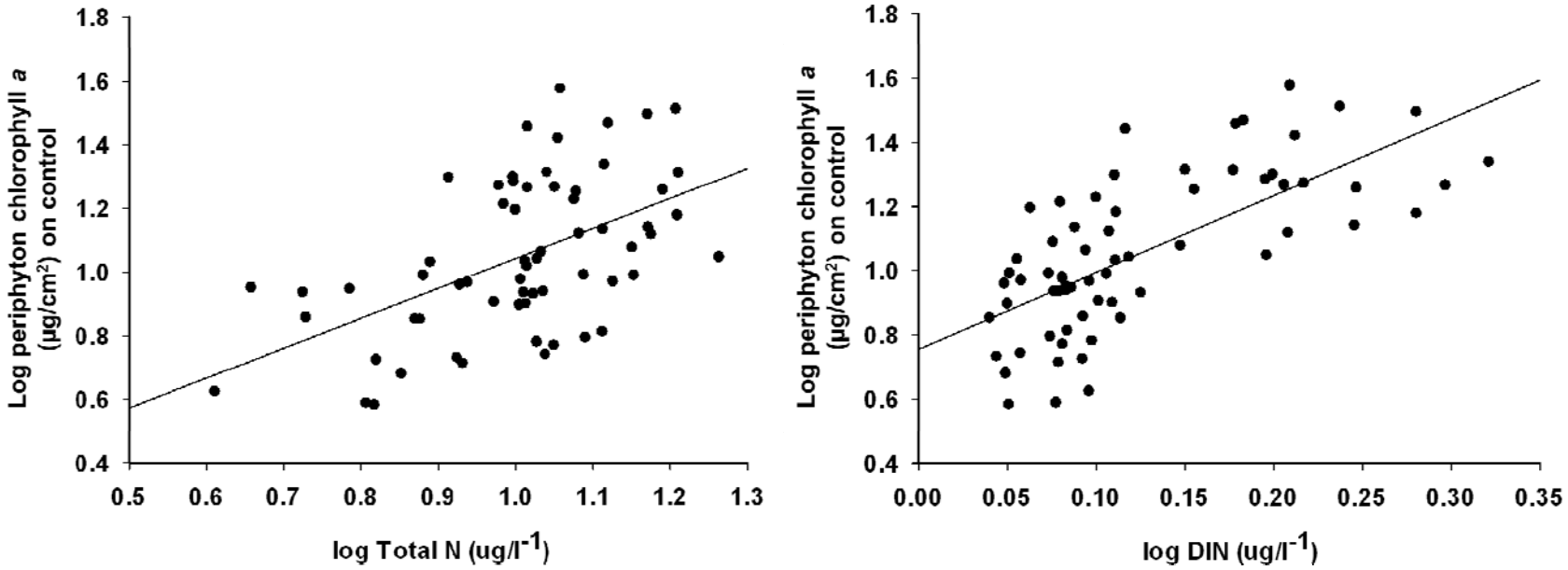


Figure 6.21 Log (x+1) of Nutrient Limitation Status Index (NLSI) values plotted against water column concentrations of dissolved inorganic nitrogen (DIN-N) and total nitrogen (TN), at connected sample sites in Delta Marsh, from 2003 to 2005. There was a significant negative relationship between algal response versus (a) and DIN ($r^2 = 0.26$, $p = 0.0012$), and (b) TN ($r^2 = 0.32$, $p = 0.0007$)).



\Figure 6.22 Log (x+1) of periphyton chlorophyll *a* (µg/L) on the control treatments plotted against water column concentrations of dissolved inorganic nitrogen (DIN-N) and total nitrogen (TN) at connected sample sites in Delta Marsh from 2003 to 2005. There was a significant positive relationship between algal control response and a) TN ($r^2 = 0.29$, $p < 0.0001$)), and b) DIN ($r^2 = 0.48$, $p < 0.0001$).

periphyton biomass on the control treatment or the NLSI value for the N treatment were significantly correlated with water column concentrations of TRP-P and TP, or water column molar ratios of TN:TP and DIN:TRP (Table 6.1). Interestingly, the opposite trend occurred between mean phytoplankton biomass in the marsh (as measured by water column chlorophyll *a*; presented in Chapter 5 Figure 5.40 and 5.41) and water column N and P concentrations and molar ratios, as phytoplankton chlorophyll *a* concentrations were found to be positively correlated with water column concentrations of TP ($r^2 = 0.36$, $p < 0.0001$), and negatively with molar ratios of TN:TP ($r^2 = -0.20$, $p = 0.0005$; Table 6.1, Figure 6.23). Phytoplankton biomass was not found to be correlated significantly with water column concentrations of DIN or TN, or molar ratios of DIN:SRP and DIN:TP (Table 6.1).

At the majority of marsh sites, molar ratios of TN: TP indicated P limitation, whereas the NDS bioassays indicated N limitation (Appendix H). Although not statistically significant, DIN:TRP and DIN:TP ratios most often agreed with the NDS results, predicting N limitation or no limitation at most marsh samples sites; however, this agreement was highly unpredictable spatially and inter-annually (Appendix H). While not significantly correlated with nitrogen NLSI values, the ratio of DIN:TP was found to be significantly positively correlated with periphyton biomass on the control NDS (Table 6.1).

The relationship between algal biomass response on control and N treatments with other physiochemical variables (i.e. depth, light and turbidity) could not be examined as the NDS bioassays were suspended at equal depths just below the water surface (10 cm) at all sample sites, and thus not influenced by the full effect of the varying water column depths and light conditions at sample sites.

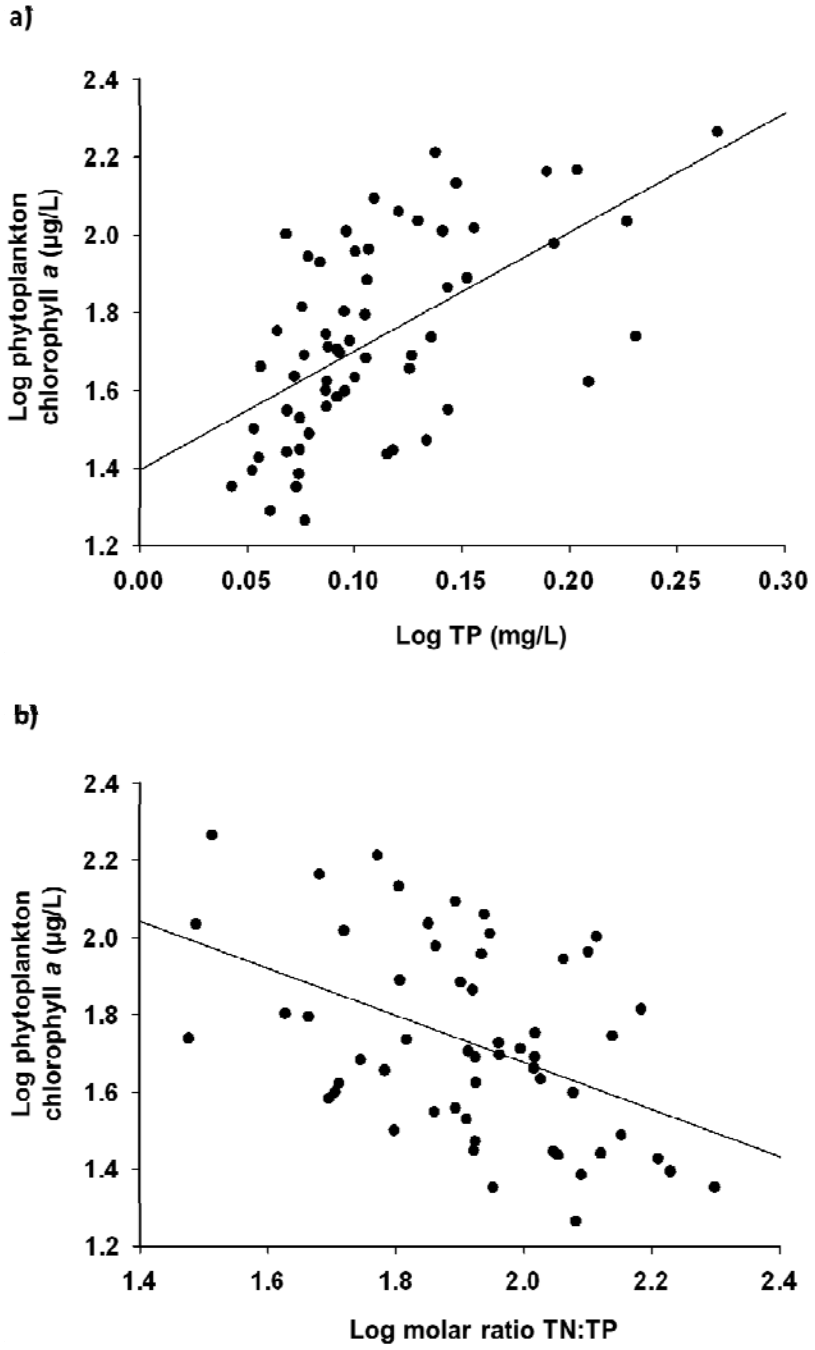


Figure 6.23 Log of phytoplankton chlorophyll *a* (µg/L) plotted against the log of water column concentrations of total phosphorus (TP) and the log of the molar ratio of total nitrogen (TN) to TP, at connected sample sites in Delta Marsh from 2003 to 2005. There was a significant positive relationship between algal response and a) TP ($r^2 = 0.36$, $p < 0.0001$), and b) significant negative relationship between TN:TP ($r^2 = 0.20$, $p = 0.0005$)).

Discussion

My finding that periphyton growth in Delta Marsh was most frequently stimulated by N is supported by previous studies in Delta Marsh. Kiers-North (2000) used nutrient-diffusing substrata (N and/or P) and N debt experiments to conclude that periphyton and phytoplankton growth in the west section of Blind Channel was limited by N. During the Marsh Ecology Research Program (MERP), Kadlec (1986) found that TN:TP ratios indicated the likelihood of phytoplankton N limitation in isolated ponds created in the east section of Delta Marsh. Hooper-Reid and Robinson (1978) used several physiological indicators of algal nutrient limitation, including alkaline phosphatase activity, nitrogenase activity, ratio of protein to carbohydrate and lipid, and silica uptake rates in Crescent Pond to conclude that periphyton growth was most likely limited by N during the early summer. They also noted that high alkaline phosphate activity in the early summer, indicative of potential P limitation and/or N+P limitation, and low Si concentrations in late summer, was accompanied by reduced diatom growth.

As noted in Chapter 5, a pronounced P and N nutrient-depletion gradient was evident in the east and west connected sections of the marsh, with nutrient concentrations decreasing as a function of decreasing distance to Lake Manitoba. Internally, the spatial concentration gradients of TRP-P and TP decreased to a greater degree with decreasing distance to lake Manitoba compared to DIN-N and TN, resulting in increasing N:P molar ratios, and the potential for a transitional gradient from N limitation to P limitation with decreasing distance to Lake Manitoba. While the NDS data also supported a generally higher potential for N limitation, there was no consistently significant relationship between the N:P ratios examined and NDS results. Furthermore, the NDS bioassays indicated N limitation, or a lack of nutrient limitation, at sites

across the marsh often regardless of distance to Lake Manitoba. When NDS bioassays indicated N limitation in the marsh, no single-limitation pattern described all sites in the marsh, as nutrient limitation of periphyton biomass was highly variable, both spatially and temporally. Results of the bioassays at the two sites in closest proximity to the Lake (Deep Creek in west section, and Delta Channel in the east) were also contradictory to N and P levels and N:P ratios in the water column, which predicted P limitation, while Deep Creek and Delta Channel consistently exhibited N limitation in all study years. (The exception was Delta Channel in June and July of 2002, where NDS bioassays indicated co-limitation by N+P.) It has been found that one of main drawback of using N:P ratios to predict nutrient limitation conditions is they only provide static snapshots of water column nutrient concentrations at fixed moments in time, and as a result they are often not good indicators of long-term nutrient concentrations, especially in highly dynamic and productive systems such as wetlands and other shallow water bodies. Weithoff and Walz (1999) used carbon to nitrogen (C:N) ratios and ammonia enhancement growth response after nutrient additions to measure phytoplankton N limitation in three shallow eutrophic lakes in Germany, however, the rapidly changing nutrient conditions in the lakes resulted in high variability in the indicators used resulting in contradictory results. Dodds (2003) noted that inorganic concentrations of N (DIN) and P (TRP) in the water column should not be used to determine nutrient limitation status as they are static measures, and cannot be used with certainty to estimate nutrient supply over time (i.e. turnover rate and uptake kinetics) Further, it must be recognized that algal species will vary in nutrient requirements. As a result, even though water column ratios of N:P might indicate one conclusion, different algal species may demonstrate different types and degrees of nutrient limitation.

NDS results indicated stimulation by both N+P in Delta Channel, Cadham Bay and Simpsons Bay, the first three sites along the east transect, in June and July 2002. Similarly, during this time bioassay results indicated that periphyton in Lake Manitoba was limited by N+P, possibly indicating an increased hydrological interaction and nutrient exchange between Lake Manitoba and the marsh. The occurrence of co-limitation can be caused by several factors including low water column nutrient concentrations and increasing nutrient cycling within the periphyton matrix (Mulholland et al. 1998). In all cases where co-limitation occurred in the marsh, periphyton biomass was stimulated by all three of the individual treatments. This suggests a portion of the periphyton biomass at these sites may have been limited by N, and another portion by P, and possibly another portion limited by both N and P. The increased response to the N+P treatment was potentially a result of components of complex algal assemblages fulfilling their needs for a primary limiting nutrient, and needing to meet a requirement for a secondary nutrient. For example, the N-limited portion of the periphyton biomass may have been stimulated by N to the point that P became limiting to growth, or vice versa.

NDS bioassays indicated that periphyton stimulation by N was highly variable annually and inter-annually in the marsh. However, some inter-annual and annual trends were present in connected sections, and occurrences of N limitation or a lack of nutrient limitation did generally coincide with water column concentrations of N. Seasonally, N limitation occurred in May and August in most study years, with variations between N limitation and a lack of nutrient limitation in July and August. Annually, bioassays indicated N limitation was most predominant in both 2003 and 2005, compared to 2002 and 2004, corresponding to higher concentrations of TP, and lower TN concentrations and N:P ratios.

The significant correlation of reduced periphyton biomass on the N bioassays with increasing water column concentrations of N, coupled with the positive correlation between periphyton biomass on control NDS and water column N concentrations, indicate that periphyton in Delta Marsh was predominantly N-limited during the study. The lack of significant correlation between the response of periphyton biomass to the N NDS treatment and the control NDS to water column P concentrations, also indicates that at times of N limitation, P concentrations were typically high and likely sufficient for periphyton needs. Further, the positive correlation between phytoplankton biomass and P concentrations in the marsh, and lack of correlation with N concentrations, indicates phytoplankton biomass in the marsh was likely more dependent on P concentrations, and independent of variations in N concentration during the study. This could indicate that the phytoplankton is the marsh was composed of N-fixing cyanobacteria, as they have been shown to be superior competitor in high P, low N, and low TN:TP conditions (Barica 1980, Sommers 1989, Kann 1997). This could also possibly explain why in 2004, the NDS experiments indicated a general lack of periphyton nutrient limitation by N or P at the majority of connected marsh sites, which coincided with higher TN concentrations and lower molar ratio of N:P, indicating that the higher concentrations of N relative to P were possibly allowing periphyton biomass to increase, and outcompete phytoplankton biomass. Correspondingly, phytoplankton biomass was also lower in 2004, compared to 2002 and 2003, at the majority of marsh sites. However, analysis of the species composition of phytoplankton and periphyton at the marsh sites during the study is required before any conclusion can be drawn.

In comparison to the connected sites, isolated marsh sites exhibited N limitation annually and inter-annually, regardless of physical location. These sites correspondingly also exhibited generally higher TP concentrations, lower N concentrations, and lower N:P molar ratios

compared connected sites. Goldsborough and Wrubleski (*personal communication*) noted that isolated ponds in Delta Marsh generally have clear water, more submerged vegetation, and invertebrates compared to the rest of the marsh. Several factors could contribute to N deficiency and P sufficiency in these sites, including the possibility that the high macrophyte biomass can result in increased competition with algae for light and nutrients (Scheffer 1998). Macrophytes may create ideal environments for the permanent removal of N to the atmosphere by denitrification (Brodrick et al. 1988, Seitzinger 1988, Neely and Baker 1989, Windolf et al., 1996, Kadlec and Knight 1996, Scheffer 1998, Saunders and Kalff 2001, Poe et al. 2003). Macrophyte decomposition also results in oxygen consumption in sediments causing anoxic conditions resulting in the release of iron-bound P from the sediments (Meiger et al. 1994, Van den Berg et al. 1997). The isolation of these sites from surface water connection with the rest of the marsh and Lake Manitoba, likely also influenced internal biochemical processes. Morrice et al. (2004) found that seasonal variations in coastal wetland hydrology can regulate internal nutrient dynamics, with hydraulic residence time positively related to the retention of inorganic N, and negatively to sediment retention of total and inorganic P, resulting in greater potential for N limitation. Work in the Norfolk Broads wetlands in England have shown that even when not severely P-polluted, the sediments can be significant P sources, released under aerobic as well as anaerobic conditions, with internal cycling providing sufficient supplies of P to maintain high algal growth (Moss 2001). The release of P from the sediment in these shallow systems can be further enhanced by frequent physical (i.e. wind and waves) and biological perturbation by fish and invertebrates (Riley and Prepas 1984, Søndergaard et al. 1992, Scheffer 1998), and uptake and release by rooted macrophytes (Søndergaard et al. 2003, Dunne and Reddy 2005). Thus,

while P is recycled internally and can become readily available, N becomes less available and is in greater demand throughout the growing season.

It has been shown that N limitation is becoming increasing common in shallow water bodies in watersheds dominated by agriculture and urban development (Downing and McCauley1992, Arbuckle and Downing 2001, Moss 2001, Munn et al. 2002, Hill et al. 2006). While agricultural sources of N are more generally diffuse across the land, cultivated/farmed soils are becoming saturated with P (Moss 2001). Manure from animal agriculture is also P-rich, and can enter aquatic systems through uncontrolled runoff, erosion, and leaching resulting in lowered N:P ratio and greater instance of N-limitation in receiving waters (Downing and McCauley 1992, Arbuckle and Downing 2001). Hill et al. (2006) found that low N:P ratios in the water and sediments, and high microbial enzyme activity in several coastal wetlands of the Laurentian Great Lakes, were positively correlated with an increasing agricultural stress in the watershed, and indicated a predominance of N limitation, rather than P limitation. Arbuckle and Downing (2001) observed that lakes in a highly agricultural watershed with large areas of pasturelands have high TP concentrations and low N:P. Billen and Garnier (1997) concluded historical changes in land use have resulted in shifts in N and/or P limitation in coastal zones, with increasing predominance of N limitation since the onset of industrialization. Kolochuk (2008) found that periphyton growth was stimulated by N in one-fourth of 59 farm pond across southern Manitoba. Carpenter et al. (1998) noted that 80% of the P load to Lake Sempach, in Switzerland, was the result of intensive agricultural activities in the lake's watershed.

Results of this study suggest that algal nutrient limitation can be highly variable both spatially and temporally within a relatively small geographic area in coastal wetlands, both in severity and type. The predominance of N limitation in Delta Marsh was found to be negatively

correlated with water column N concentrations but not P concentrations. This was likely the results of high P concentrations that were in excess of periphyton requirements, relative to N requirements. This indicates that other coastal wetlands across the prairies may likewise experience predominance of N limitation, from a combination of possibly high P relative to N loading from agricultural sources in the watershed, and internal wetland process that can result in high rates of permanent N loss to the atmosphere via denitrification, coupled with high rates of internal P recycling from the sediments. Coastal wetlands can also be affected to varying degrees by hydrological influence and nutrient input, as well as internal biotic and abiotic factors resulting in large seasonal and spatial differences between different wetlands, as well as within individual wetlands. This was evident by the greater spatial and temporal variability in periphyton nutrient limitation in connected compared isolated sites in the marsh, indicating that open water connection to Lake Manitoba, coupled with varying hydrological and nutrient inputs from the surrounding watershed, may have profound influences on nutrient inputs as well as internal cycling, increasing the spatial and temporal variability in nutrient concentrations and algal nutrient limitation. It is also important to note that overall algal nutrient limitation conditions in the marsh cannot be inferred solely from water column N:P ratios or phytoplankton chlorophyll *a* concentrations, but instead is best determined by a combination of bioassays as well as other *in situ* physiological tests (i.e., alkaline phosphatase activity, nitrogenase activity, and nutrient debt experiments).

Chapter 7: Discussion and Conclusions

Hypotheses revisited

1. **My hypothesis that water level and flow hydrology in Delta marsh will be correlated spatially and temporally with water levels and wind-induced seiche set-up flow on Lake Manitoba was supported.** Water levels in the marsh were highly correlated with those of the lake, particularly during times when lake levels were at some of the highest and lowest values in recorded history. Short-term water level change in the marsh was caused by wind-induced seiche setup events on Lake Manitoba, with the speed of northerly wind events positively correlated with increasing volume of water exchange from the lake into the marsh. Correspondingly, marsh water levels decreased during southerly wind-induced seiche events, as well as during calm periods following seiche events as lake level returned to equilibrium. Resulting water changes up to 56 cm over a 24-hour period were recorded in the marsh. The magnitude and intensity of seiche-induced water level changes in Delta Marsh were attenuated 40 to 80 % in comparison to lake water levels changes, with the degree of attenuation within the marsh increasing with increasing distance from Lake Manitoba.

2. **My hypothesis that water levels and flow in the western versus the eastern portion of Delta Marsh will be more strongly correlated with those of Lake Manitoba because of the greater degree of surface water connection, smaller open water area, and shallower depth was not supported.** With the exception of the Canoe Dock (CAN) site in the east section of the marsh, sites in both east and west sections of the marsh experienced similar long-term and daily water level changes, and there was no significant difference between the

magnitude and intensity of daily water level changes. In the west side of the marsh, the difference in the attenuation of water level range at Big Lake North West (BLNW) and Short Creek (SCRK) compared to Canoe Dock (CAN) was likely the result of these sites having different connecting channels with different dimensions that connected them to the lake, with Canoe Dock being primarily connected to the lake via Cram Creek which has three times the area (twice as wide and deep) as Deep Creek, which connects Big Lake North West and SCRK to the lake. Furthermore, water passing from the lake to Big Lake North West and Short Creek is also constricted along its water course by another channel (Carp Creek, ~ 15.0 m wide x 0.75 m deep). The overall lack of difference between the magnitude and intensity of water levels change at sites in east and west sections of the marsh is potentially the result of the shorter and straighter length of the connecting channel (Delta Channel, ~1.2 km) in the east section of the marsh compared to the channel in the west section (Deep Creek, ~ 2 km). Even though Delta Channel is narrower (~7.0 m wide x 1.0 m deep; reduced to 5.0 m wide x 1.0 m deep at Delta Bridge) than Deep Creek, the short channel length maybe resulting in a shorter amount of time that water is spatially constricted before entering the larger open area of the east marsh. The additional constriction of water flow to Big Lake North West and Short Creek at Carp Creek (~ 15.0 m wide x 0.75 m deep) is likely adding to the reduced changes in magnitude of water level changes in the west section. The significant influence of the morphology and dimension of connecting channels (i.e. number, size, and length) on the magnitude and intensity of water exchange between lakes and their coastal marshes has been observed in other coastal wetlands (Trebitz et al. 2002, Wells and Sealock 2009, Seldomridge and Prestegaard 2012).

3. **My hypothesis that water levels and flow in connected versus isolated areas of the marsh will be more strongly correlated with the lake and experience a greater degree of fluctuation because of their surface water connection with the lake was supported.** Long-term water levels in connected sections of the marsh were significantly correlated with water levels of Lake Manitoba, whereas water level changes in isolated sites of marsh were weakly correlated with Lake water level. The range, magnitude and intensity of daily water level changes was much reduced in isolated sites in comparison to connected sites, as they were not directly subject to the influence of daily surface water level changes on Lake Manitoba.

4. **My hypothesis that the chemical properties of marsh water will be positively correlated with increasing distance from the lake—because of a large source of N and P from surrounding agricultural uplands, natural uptake of N and P in the marsh, and the dilution effect of waters entering from the lake—was supported.** Significant correlations were found between the distance from Lake Manitoba (via surface water connection) and the majority of water quality parameters. Concentrations of dissolved inorganic N (nitrate and ammonia), total N, total P, soluble reactive P, dissolved organic C, chloride, sulfate, alkalinity, conductivity, suspended solids, and phytoplankton chlorophyll *a* all decreased significantly with decreasing distance from Lake Manitoba. The only two parameters examined that did not vary with distance were pH and silica concentrations. Silica was high at all sites (> 2.4 mg/L) thus any small changes in concentration may not have been apparent in the relatively limited data collected in a single year (2005). The decrease in chloride concentrations in the marsh with decreasing distance to the lake indicated that hydrological exchange with the lake resulted in periodic flushing and dilution of marsh waters with

'freshwater' lake waters. This dilution was found to increase spatially to a greater distance and in magnitude in the marsh with increasing lake water levels. Concentrations of N, P, major ions, suspended solids, and chlorophyll a were all highly variable with annual changes in water level, with increased water levels resulting in a significant decrease in concentrations. The increased water levels, and magnitude and spatial extent of the dilution also resulted in the overall water chemistry (dissolved inorganic N, total N, total P, soluble reactive P, dissolved organic carbon, chloride, sulfate, alkalinity , conductivity, suspended solids, and phytoplankton chlorophyll *a*) of marsh sites being less spatially variable with distance in 2005.

5. **My hypothesis that water chemistry in the western versus the eastern portion of the marsh will be more strongly correlated with the lake because of the greater degree of surface water connection, smaller open water area, and shallower depth was not supported.** Spatially there was greater variation between sites within the west and east sections of the marsh than between the two sections as a whole. In both sections of the marsh, the majority of water quality parameters decreased significantly with decreasing distance to Lake Manitoba (i.e. dissolved inorganic N, total N, total P, soluble reactive P, dissolved organic C, chloride, sulfate, alkalinity, conductivity, suspended solids, and phytoplankton chlorophyll *a*). This finding was probably indicative of the predominant agricultural sources of nutrients to the marsh in the surrounding watershed, and the influence of Lake Manitoba on the magnitude of water level changes and dilution in the marsh regardless of location. While the west section of Delta Marsh is shallower and smaller compared to the east section, it has longer and more complex channels that connect it to the

lake. This likely slowed water flows into it, resulting in similar effects of dilution on water chemistry in both sections.

6. **My hypothesis that water chemistry in connected versus isolated areas of the marsh will be more strongly correlated with the lake due to surface water connection and that isolated areas in the marsh will have greater concentrations of N, P and major ions (Cl^-), and less variation in N, P and major ions (Cl^-), because isolated sites are less influenced by the dilution and flushing effects of the lake, was partially supported.** In comparison to connected sites, most isolated sites experienced greater concentrations of total and soluble P; however, concentrations of total N, dissolved inorganic N and chloride were lower in isolated sites. The lower concentration of N in these sites probably resulted from the significantly higher biomass of macrophytes in isolated compared to connected sites. Macrophytes can compete effectively with algae for nutrients (Ciurli et al. 2009). Other studies have also shown N limitation of aquatic macrophytes in wetlands and other shallow systems (Van Donk et al. 1993, Meijer et al. 1994). Rooted macrophytes can readily assimilate N in many forms from the oxygen-poor sediments, further reducing oxygen conditions in the sediments and increasing denitrification, which is a permanent loss process removing N from these sites as N_2 (Kadlec and Knight 1996; Scheffer 1998; Saunders and Kalff 2001). Thus, while P is internally recycled and can become readily available, N becomes less available in the water column and is in greater demand throughout the growing season (Moss 2001). Replanting of aquatic macrophytes had been shown to be a useful tool for the restoration of eutrophic aquatic systems that are in a 'turbid water state' (Scheffer 1998, Moss 2001). The lower chloride concentrations are likely the result of isolated sites receiving water primarily by precipitation and groundwater. Both could result in a dilution

effect. The connected sites, on the other hand, received a larger amount of water from the surrounding watershed which can contain high concentrations of ions from agricultural activities (Trebitz et al. 2007, Morrice et al. 2011, Trebitz et al. 2012).

7. **My hypothesis that Delta Marsh is mainly limited by N (as indicated by algal nutrient status), and that the extent of N limitation will be correlated with distance from the lake via surface water flow, because of nutrient uptake and release process in the marsh and the dilution and flushing effects of the lake, was partially supported.** The results of the NDS experiments supported my hypothesis, as periphyton in Delta Marsh were most frequently stimulated by N. However, there were instances where this was not true. N limitation and a lack of limitation in the marsh were highly variable, both spatially and temporally. There was no trend in N limitation with marsh distance to Lake Manitoba. Some annual trends were present, as bioassays indicated N limitation was most prevalent in 2003 and 2005, compared to 2002 and 2004, corresponding to higher TP, and lower TN and N:P. The significant correlation of reduced periphyton biomass on the N bioassays with increasing water column concentrations of N, coupled with the positive correlation between periphyton biomass on control NDS and water column N concentrations, also supports a view that periphyton in Delta Marsh was predominantly N limited during the study. The lack of significant correlation between the response of periphyton biomass to the N NDS treatment and the control NDS to water column P concentrations, also indicates that at times of N limitation, P concentrations were typically high and likely sufficient for periphyton needs. The lack of significant correlation between the response of periphyton biomass to the N NDS treatment and the control NDS to water column P concentrations, also indicates that at times of N limitation, P concentrations were typically high and likely sufficient for

periphyton needs. In 2004, the general lack of periphyton nutrient limitation by N or P at the majority of connected marsh sites, also coincided with higher TN concentrations and lower molar ratio of N:P, indicating that the higher concentrations of N relative to P.

8. **My hypothesis that N limitation will be greater in the western versus the eastern portion of the marsh, because of the greater degree of surface water connection, smaller open water area, and shallower depth, was not supported.** As noted above, no significant spatial patterns in N limitation were found in connected sites; there was no significant difference between the east and the west sections neither of the marsh, nor with distance from the lake in both sections. Algal nutrient limitation was highly variable, spatially and temporally, in the marsh. This is likely the result of the high seasonal and spatial variability in N and P concentrations. Variations in other factors that can affect algal biomass (i.e., light, hydrology, herbivory, and temperature) also occur and can interact, confounding algal nutrient limitation and biomass (Goldsborough and Robinson 1996, Scheffer 1998, DiNicola 1996, Steinman 1996, Moss 2001).
9. **My hypothesis that N limitation will be greater in isolated versus connected areas of the marsh, because isolated sites are less influenced by the dilution and flushing effect of the lake, was supported.** In comparison to the connected sites, all isolated sites exhibited N limitation through the study, both seasonally and inter-annually. These sites exhibited higher TP, and lower N, than connected sites. The lack of connection to Lake Manitoba, as well as connected sections of the marsh via surface water, both resulted in less variation in daily water levels and less seasonal and spatial variation in nutrient concentrations. The isolation of these sites from surface water connection also likely influenced internal biochemical processes (Moss 2001, Morrice et al. 2004) and excluded Common Carp, which

have been shown to have negative effects on macrophyte biomass, sediment resuspension and water clarity in connected sections of the Marsh (Badiou 2005, Park 2006, Hnatiuk 2006, Hertam 2010).

Significance of study

This study illustrates the importance of the hydrological connectivity and the associated intimate hydrological relationship shared between Delta Marsh and Lake Manitoba, on several spatial and temporal scales, and the resultant significant influences on the associated water chemistry, and algal biomass and nutrient limitation in the marsh. Both daily and annual water level changes in the marsh were found to be highly correlated with those of the lake, with marsh levels matching those of the lake during the lowest (2003) and second highest (2005) water levels in recorded history, as well as rapid daily (hours to days) fluctuations of lake water level. This inter-annual variability in coastal wetland hydrology, and associated water residence time, is critical to coastal wetland structure and function, by improving water mixing and circulation (Trebitz et al. 2002, Morrice et al. 2004, Trebitz 2006, Wells and Sealock 2009, Morrice et al. 2011), increasing species and habitat diversity (Wilcox et al. 2002, Sierszen et al. 2006, Wilcox et al. 2007, Wilcox and Nichols 2008, Gathman and Burton 2011, Wilcox 2012), as well as affecting many internal biological and chemical processes (Gosselink and Turner 1978, Mitsch and Gosselink 2000a, Morrice et al. 2004, Knuth and Kelly 2011, Wilcox 2012).

During this study short- and long-term water level changes were associated with marked spatial and temporal variability in the water chemistry and algal nutrient status of the marsh. The large variations in annual water levels resulted in greater annual variation in the water chemistry of study sites between years than variation between sites and distance from Lake Manitoba. Significant changes in chloride concentrations and oxygen-18 and deuterium stable isotope signature indicated that wind seiches on Lake Manitoba have important dilution and flushing effects on the marsh and may play an important role in increasing water mixing and reducing

water stagnation, with greater mixing and flushing in high water years (Trebitz et al. 2002). Annual changes in lake water levels also influenced the spatial extent of dilution and flushing, resulting in large spatial and temporal differences in marsh water chemistry and algal nutrient status between years. Higher water levels in 2005 resulted greater water dilution and mixing, and significantly lower nutrients (N and P), major ions, and suspended solids (turbidity) concentrations and reduced the spatially variability of water chemistry between connected sites in the marsh, as well as between the marsh and the lake. The differences in water chemistry with increasing water levels, namely reduced nutrient concentrations, was likely the result of dilution as well as confounding internal wetlands biological processes affected by water level changes. Thus the inter-annual variability in wetland water levels is an important driver of spatial and temporal variability in wetland water chemistry, algal biomass and nutrient limitation, as well as associated wetland structure and function. Increased water levels can reduce instances of sediment suspension from wind and waves, and result in cooler water temperature (Kadlec and Knight 1997). Declining water levels have the opposite effect, resulting in higher water temperature and concentration of dissolved nutrients in the water column by mobilization from shoreline sediments and exposed plant litter (Schoenberg and Oliver 1988). Hydrologic conditions also affect microbial denitrification although the full effects of changing water levels in coastal wetlands on denitrification are still poorly understood (Knuth and Kelly 2011).

It is also important to note that historically since 1961, water level on Lake Manitoba, and in turn Delta Marsh, have been regulated via the Fairford Dam. Since regulation the magnitude and intensity of water level fluctuations on Lake Manitoba have been reduced with fluctuations rarely exceeding 0.5 m (range of over 1.7 m prior to regulation). As a result, it is assumed the timing and quantity of wind-induced seiche lake flushing flows in the marsh have also been

greatly altered (de Geus 1987, Kenkel 1995, LMRRAC 2003a). Along with the reduced magnitude and intensity of natural annual water level fluctuations and wind-induced seiche inflows into the marsh, it is likely that their influence on annual variations in marsh water chemistry, and algal biomass and nutrient limitation have also been reduced compared to pre-regulation. The reduced water level fluctuations have caused the marsh to enter a 'lake-marsh phase' with reduced water quality, productivity, and species diversity, and has raised concerns about its long-term biological health (van der Valk 1981, de Geus 1987, Kenkel 1995, Wrubleski 1998, Grosshans 2001, and Goldsborough and Wrubleski unpublished). Shay (1999) also found that regulation of water levels on Lake Manitoba has resulted in reduced vegetation diversity in Delta Marsh and the dominance of hybrid *Typha x glauca,* and suggested if stable water levels continued that *Typha* would continue to expand and eventually fill in the marsh. These changes in marsh vegetation are important to note since macrophytes have been shown to have an important influence on wetland water chemistry, algal biomass and algal nutrient limitation (van Donk et al. 1993, Meijer et al. 1994, Kadlec and Knight 1996, Scheffer 1998, Saunders and Kalff 2001, Ciurli et al. 2009). In the Laurentian Great Lakes wetlands, Wilcox (2012) noted similar changes in some of the coastal wetlands of Lakes Superior, which has been regulated since 1914, resulting in the elimination of low water levels, and the dominance of shrub and woody vegetation on the shorelines that have not been affected by flooding. The wetland interiors become dominated by submerged macrophytes which have not been affected by water drawdown events (Wilcox 2012). Similarly on Lake Ontario, water level have been regulated since 1960, and as a result the range in lake water level fluctuation have been reduced from 2 m (prior to regulation), to 1.3 m (following regulation; Wilcox 2012). The reduced magnitude of flooding, and especially reduced dewatering conditions, in the adjoining coastal wetlands have resulted in

the expansion of Typha spp. to the detriment of other vegetation types and diversity (Wilcox et al. 2008, Wilcox 2012).

Trebitz (2006) found that the magnitude of seiches on Lake Superior, as well as upland tributary inputs, were key factors influencing habitat patterns and water chemistry in coastal wetlands. Increasing seiche activity on the lake and tributary inputs increased wetland water mixing and moderated temperature fluctuation and oxygen depletion. Suzuki et al. (1995) and Cardinale et al. (1998) reported decreasing dissolved oxygen, turbidity, and phytoplankton chlorophyll a concentration with decreasing distance to Lake Huron in coastal wetlands fringing Saginaw Bay. Millie et al. (2006), in a study of phytoplankton abundance in Saginaw Bay, also found spatial and temporal gradients in the wetland, including increasing phytoplankton chlorophyll a, TP, TRP, silica, chloride, total suspended solids, particulate organic carbon and temperature with increasing distance from Lake Huron. Moss (2001) noted that flushing in shallow productive systems is important to reduce the predominance of cyanobacteria, which under calm conditions can use gas vacuoles to float to the water surface to photosynthesize as well as fix N_2. Flushing helps other species of algae, including green algae and diatoms, to outcompete cyanobacteria. The occurrence of cyanobacteria blooms is concerning due to their negative effects on oxygen levels when large blooms collapse, resulting in fish kills, and their production of hepato- and neuro-toxins, including microcystins (Kotak et al. 1995, Wood et al. 2012).

The finding of this study that periphyton growth is most frequently stimulated by N in Delta Marsh is supported by results obtained from previous studies in the marsh, as well as other prairie wetlands. Kiers-North (2000) observed that periphyton and phytoplankton in the west section of Blind Channel in Delta Marsh limited by N. Hooper-Reid and Robinson (1978) found

that periphyton growth was most likely limited by N during the early summer in Crescent Pond, an isolated pond in Delta Marsh. Bortoluzzi et al. (in prep) found that algal N limitation was predominant in other wetlands in the prairies region of southern Manitoba; Oak Hammock Marsh, an isolated basin marsh located in south-central Manitoba, as well as Netley-Libau coastal, coastal riverine-lacustrine marsh located at the confluence of the Red River and the south basin of Lake Winnipeg. The predominance of N limitation by periphyton in Delta Marsh is likely the result of a combination of increased P relative to N inputs from agricultural development in the watershed (which is likely the predominant source of nutrients, ions and suspended solids to Delta Marsh, and not Lake Manitoba as shown by this study) coupled with internal wetlands processes that can result in greater reductions in N compared to P, including denitrification, which is permanent N loss process (McCauley 1992, Robarts et al. 1992, Carpenter et al. 1998, Reinhardt et al. 2005). Downing and McCauley (1992) found waters draining disturbed urban and agricultural watersheds experienced high TP, low N:P ratios, and greater instance of N limitation as algae become P sufficient, compared to water bodies receiving water from undisturbed watersheds, which often have low TP and higher N:P ratios. Robarts et al. (1992) found that a shallow lake in south-central Saskatchewan, which received high nutrient loads from treated sewage, cattle feedlot drainage, and agricultural runoff, had high average TP concentrations (0.30 mg/L) relative to DIN (0.18 mg/L), resulting in low TN:TP ratios (1:1). Other shallow prairie lakes and wetlands have also been characterized by highly eutrophic conditions, with high TP and chlorophyll a concentrations, low N:P ratios (Haertel 1976; Barica 1975, 1987; Barica et al. 1980), indicating they may also experience N limitation. Collectively, this indicates that other pothole and coastal wetlands in the prairie region are may also be predominately limited by N, and not P.

The likelihood that the surrounding agriculturally dominated watershed is a predominant source of nutrients to Delta Marsh, coupled with the larger spatial decreases in N and P nutrient concentrations relative to chloride concentrations with decreasing distance to the lake, indicated that other internal wetland biochemical process (aside from dilution from the lake) reduced nutrient concentrations and improved the quality of water entering the marsh from the surrounding watershed, before it entered Lake Manitoba. Many natural processes in coastal wetlands have been noted to result in improved water quality, including denitrification, sedimentation, filtration, absorption, and biological uptake (Kadlec and Knight 1997). Hydrology, specifically water levels and residence time, have been noted as dominant influences on these processes and the resultant treatment capacities of coastal wetlands (Nichols 1983, Clausen and Johnson 1990, Carter 1997, Keddy and Frazer 2000, Morrice et al. 2004, Hill 2006). Notably, increasing water mixing and longer water residence time in marshes have been shown to increase N and P retention and uptake (van der Valk 1978, Saunders and Kalff 2001, Reinhardt et al. 2005). Thus, through Delta Marsh's hydrological connection to Lake Manitoba, as well as its location in the landscape between the lake and surrounding upland watershed to the south, the marsh is likely performing an important ecological function by reducing inputs of N and P, as well as improving the overall quality of water entering the lake from the upland watershed surrounding the marsh. A number of studies have found that other coastal wetlands can significantly improve the quality of water entering adjoining lakes from the watershed. The majority of studies have occurred in the Laurentian Great Lakes coastal wetlands, which have been shown to reduce the concentrations of N and P from inflowing tributary waters. For instance, studies in Old Woman Creek, on Lake Erie, have indicated the wetland can annually retained 33-36% of incoming TP, 21 to 81% of TRP, 18% of DIN, and 15% of TN from the

upland watershed before it enters Lake Erie (Klarer and Millie 1989, Mitsch and Reeder 1992, Krieger 2003). Studies in Lost Creek wetland, on Lake Superior, indicate the wetlands TP retention ranged from 4 to 24% and up 76% for TRP, and DIN retention ranged from 11% to 94% (Morrice et al. 2004). Sierszen et al. (2012) estimated that the U.S. Laurentian Great Lakes coastal wetlands alone retain nearly half of nutrient exported from the surrounding U.S. watershed into the Great Lakes. Clausen and Johnson (1990) in a study of Stevens Brook Wetland, on Lake Champlain, in northwest Vermont found that the wetland was a net sink for total suspended solids, TP and total Kjeldahl nitrogen (TKN) entering the wetland from upland watershed.

The NDS experiments proved useful to examine *in situ* periphyton response to nutrient stimulation under natural conditions in Delta Marsh, including variations in hydrology and water chemistry, and the interacting effects of light intensity, temperature change, and herbivory. They also allowed for the simultaneous examination of natural algal stimulation by nutrients within the large geographical extent of the marsh. While the NDS are not natural substrata for periphytic algal growth, they enabled easier direct comparison and measurement of response to nutrient stimulation between sites versus manipulation and sampling of natural substrata (rocks, macrophytes, sediment, etc.). Like any index used to measure algal nutrient limitation in highly dynamic and variable wetland systems, caution should be exercised when interpreting NDS bioassay results. For instance, equal growth on all treatments can be interpreted as a lack of nutrient limitation. This would be the case if periphyton biomass was equally high on all treatments. However, if biomass is equally low on all treatments, some other factor may be limiting algal growth, such as light, grazing, or temperature. In order to differentiate what constitutes ‘high’ and ‘low’ periphyton biomass response at each site, experiments over several

years at varying locations may be required, as was the case during this study. The results of my NDS experiments would benefit from examination of the algal composition in the various treatments, as well as natural substrates at each deployment site. Different algal species have varying requirements for nutrients, light and temperature (Healey and Henzel 1979). This can result in a species-specific response to the NDS bioassay (Healey and Hendzel 1979, Rhee and Gotham 1980, Fairchild et al. 1985, Francoeur 2001, Scott et al. 2009). The use of other indices to measure algal nutrient limitation, for example alkaline phosphatase enzyme production, nitrogenase activity, and P and N debt (Healey and Henzel 1979, Rhee and Gotham 1980), in combination with future NDS experiments, would also help better establish the accuracy of the NDS results and thereby validate the determination of algal nutrient limitation in wetlands.

Wetland conservation, restoration and management recommendations

The regulation of water levels on Lake Manitoba, and in turn Delta Marsh, since 1961, has resulted in the reduced magnitude and frequency of water level changes in the lake and marsh, as well as reduced species diversity and water quality in Delta Marsh (Kenkel 1995, Shay et al. 1999). Coastal wetlands are naturally dynamic, with water level variation essential for overall marsh health, productivity and species diversity. Further, large and persistent changes in water level and flow conditions are essential for increased species diversity, water quality improvement, and habitat diversity (Clausen and Johnson 1990). To build a complete functional understanding of coastal wetlands like Delta Marsh, scientific studies and management plans need to take into account the hydrologic connectivity between the wetlands and their adjoining lakes, as well as the surrounding watershed, to be able to distinguish between natural variability (reference conditions) and anthropogenic influences, as well as quantify their vulnerability to

change, and establish goals for conservation, restoration and/or management. However, in complex dynamic systems, such as the Delta Marsh, constant, and likely unpredictable change, may well be the rule and thus make the notion of establishing "reference conditions" impractical.

In 2003, the Lake Manitoba Regulation Review Advisory Committee recommended that Lake Manitoba levels be permitted to fluctuate more or less naturally within the range of 246.9 and 247.6 m above sea level, with the expectation that further management would only occur when levels were at these extremes, so the effective range of lake level would fall between 246.8 and 247.8 m above sea level) (LMRRAC 2003a). This management regime would influence water levels on adjoining Delta Marsh, presumably allowing them to fluctuate more naturally. I would recommended that the management plan for Lake Manitoba and Delta Marsh include several years of low water levels, to reduce the persistence of *Typha x glauca,* and reestablish the seed bank and increase the diversity and biomass of other macrophytes in connected sections of the marsh (Kenkel 1995, Wilcox et al. 2008, Wilcox 2012), as macrophytes have been shown to have an important role in reduce water column nutrient concentrations and phytoplankton biomass, as well as increase water clarity (van Donk et al. 1993, Meijer et al. 1994, Kadlec and Knight 1996, Scheffer 1998, Saunders and Kalff 2001, Ciurli et al. 2009). Following the dewatering period, the management regime should allow for greater persistence of natural water level fluctuations, and as much as possible the regulation of Lake Manitoba should allow for greater magnitude and intensity of water level variations to ensure the natural hydrology of the marsh is maintained. In the case of this study, high water levels have also been shown in perform an important dilution and flushing effect on the marsh, reducing overall nutrient concentrations and the spatial variation in water chemistry.

The lake and marsh management strategies also need to address the reduction of nutrient inputs to the marsh by the possible provision for vegetative buffer areas between the marsh and surrounding agricultural, urban and cottage lands, and reductions of nutrient inputs at point sources. To enable the restoration of aquatic vegetation in Delta Marsh, it will be essential to promote a return to a clear-water state. This may require isolation of the marsh to Common Carp and reductions to N concentrations to increase macrophyte diversity (Moss 2001).

Recommendations for future work

A study of the hydrological and nutrient budget of Delta Marsh needs to be undertaken to better understand marsh hydrology, as well as to determine a nutrient mass balance. Specifically, the volume and direction of hydrological flow exchange between the marsh, via its four connecting channels and the lake, needs to be quantified during varying degrees of wind-induced seiche set-up on Lake Manitoba, to be able to develop a relationship between seiche set-up on the lake, and resultant direction and volume of flow exchange and water level change in various sections of the marsh. This information is needed to determine the spatial and seasonal extent of mixing and flushing in the marsh and change in water chemistry. Measurement of N and P nutrient biological and chemical sinks, sources and transformation processes within the marsh are needed, including denitrification rates, P sedimentation and resuspension, and N and P uptake by macrophytes and algae, to better quantify the effect of lake dilution and flushing compared to the impacts of other biological process on the nutrient budget of the marsh.

Similar studies of other coastal wetlands around Lakes Winnipeg, Winnipegosis, and Manitoba should be conducted to better understand spatial and temporal differences in hydrological water exchange of these wetlands with adjoining lakes, and effects on marsh water

chemistry and algal nutrient status. Delta Marsh is one of the largest coastal wetlands in the prairie region. It has no upstream tributaries and it has four channels connecting it to Lake Manitoba, whereas other coastal wetlands will have differences in these variables which, in turn, will influence the degree of hydrological water exchange between them and their lakes. This information is needed to be able to compare between, as well as collectively determine effects of, varying degrees of water exchange, and associated changes in water chemistry of the wetlands, as well as their structure and function.

Future research in these coastal wetlands should have a watershed perspective and consider the impacts of the watershed and adjoining lakes on receiving wetlands. For instance, the presence of agricultural production in watersheds has been associated with increased loading of N and P to downstream water bodies; however, there are few quantitative data of loading rates under varying extent and form of agriculture in the watershed, as well as other land uses. This information is needed to be able to quantify the rate of nutrient loading to coastal wetlands and determine the effects of different land uses and associated nutrient loading rates on wetland water chemistry and algal nutrient status.

The 2011 flood on Lake Manitoba and the surrounding area, including Delta Marsh, presents an exceptionally unique opportunity to examine the effect of flooding and prolonged high water levels (one-in-400-year flood) on a large coastal marsh ecosystem. These high water level events may serve as ‘reset’ mechanisms in coastal marshes, analogous to forest fires, and may help to reduce water stagnation and mono-species dominance, and increase diversity. A multi-year ecosystem level study examining changes to the physical, chemical, and biological environment of marsh would be useful.

Identification and enumeration of the phytoplankton and periphyton taxa in water and NDS samples collected during my study would allow for a better understanding of species- and population-level responses to the nutrient treatments, as well spatial and temporal variations in species composition in areas of the marsh with varying water chemistry and hydrology.

A final word

"Some time before the year 2000, unless something is done to avert the situation, we shall find ourselves living in the middle of an Algal Bowl, with effects on water comparable to those on land during the great American Dust Bowl of the 1930s." (John R. Vallentyne 1974)

In the 21st century, cultural eutrophication continues to hold mysteries that baffle aquatic ecologists, with no one boilerplate solution fitting all water bodies. In the highly productive prairies of North America, increasing livestock production, fertilizer use, and urban and industrial sewage generation are changing the availability of N and P in the environment. The resultant effects on nutrient availability in coastal wetlands are still largely unknown. Wetlands ecosystems are naturally dynamic with both biotic and abiotic factors continuously interacting and changing seasonally and temporally from one year to the next. Within all the unpredictability, one common denominator remains central to wetland function, diversity and health, and that is the fundamental importance of hydrologic variability to the maintenance of the wetland ecosystem.

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Appendix A: Depth profile data for Delta Marsh

Location:	Delta Channel						
Site	Site_description	Date_time	EASTING	NORTHING	Depth_cm	Depth_m	elevation_masl_bottom_sediments
Delta Channel	stake to S of NDS stake (tracer site 5)	14/07/2005 10:15	549285.47	5558996.44	69	0.69	247.20
Delta Channel		14/07/2005 10:16	549281.96	5558988.99	97.5	0.98	246.92
Delta Channel		14/07/2005 10:20	549280.81	5558985.28	102	1.02	246.87
Delta Channel		14/07/2005 10:23	549278.50	5558977.84	139.5	1.40	246.50
Delta Channel		14/07/2005 10:24	549272.62	5558970.38	72	0.72	247.17
Delta Channel	between NDS and last stake	14/07/2005 10:28	549182.05	5559026.46	47	0.47	247.42
Delta Channel		14/07/2005 10:30	549179.32	5559021.40	98.5	0.99	246.91
Delta Channel		14/07/2005 10:32	549178.17	5559017.69	93	0.93	246.96
Delta Channel		14/07/2005 10:33	549174.65	5559012.10	105	1.05	246.84
Delta Channel		14/07/2005 10:34	549171.15	5559004.65	106	1.06	246.83
Delta Channel	NDS site (tracer site 4) site w flow	14/07/2005 10:37	549094.17	5559092.90	60	0.60	247.29
Delta Channel		14/07/2005 10:40	549089.48	5559085.44	99	0.99	246.90
Delta Channel	site main center flow measurments	14/07/2005 10:42	549082.37	5559081.67	107	1.07	246.82
Delta Channel	site east flow	14/07/2005 10:43	549076.49	5559074.20	117	1.17	246.72
Delta Channel		14/07/2005 10:46	549065.06	5559069.67	84.5	0.85	247.05
Delta Channel	slightly N of NDS	14/07/2005 10:49	549054.39	5559148.13	70	0.70	247.19
Delta Channel		14/07/2005 10:50	549045.73	5559142.75	115	1.15	246.74
Delta Channel		14/07/2005 10:51	549036.64	5559136.85	79	0.79	247.10
Delta Channel		14/07/2005 10:53	549030.49	5559131.62	70	0.70	247.19
Delta Channel		14/07/2005 10:55	549023.64	5559127.46	75.5	0.76	247.14
Delta Channel	more N of NDS	14/07/2005 11:03	549012.16	5559210.75	124	1.24	246.65
Delta Channel		14/07/2005 11:05	549007.44	5559207.00	116	1.16	246.73
Delta Channel		14/07/2005 11:07	549002.67	5559202.65	85	0.85	247.04
Delta Channel		14/07/2005 11:08	548996.81	5559197.64	6935	69.35	178.54
Delta Channel		14/07/2005 11:10	548988.55	5559190.15	76	0.76	247.13
Delta Channel	stake N of NDS (tracer site 3)	14/07/2005 11:13	548953.08	5559278.60	46	0.46	247.43
Delta Channel		14/07/2005 11:14	548949.71	5559273.19	100	1.00	246.89
Delta Channel		14/07/2005 11:16	548947.35	5559267.99	117.5	1.18	246.72
Delta Channel		14/07/2005 11:18	548944.84	5559263.60	103	1.03	246.86
Delta Channel		14/07/2005 11:21	548941.20	5559256.93	79	0.79	247.10
Delta Channel	between stakes	14/07/2005 11:24	548892.59	5559307.95	82	0.82	247.07
Delta Channel		14/07/2005 11:26	548889.93	5559304.14	104	1.04	246.85
Delta Channel		14/07/2005 11:26	548888.20	5559300.63	104	1.04	246.85
Delta Channel		14/07/2005 11:29	548886.44	5559295.36	105	1.05	246.84
Delta Channel		14/07/2005 11:30	548884.10	5559291.12	70	0.70	247.19
Delta Channel	next stake N (tracer site 2)	14/07/2005 11:33	548838.29	5559355.56	78.5	0.79	247.11
Delta Channel		14/07/2005 11:34	548830.06	5559344.37	60	0.60	247.29
Delta Channel		14/07/2005 11:39	548821.80	5559336.88	70	0.70	247.19
Delta Channel		14/07/2005 11:43	548817.09	5559331.27	101.5	1.02	246.88
Delta Channel		14/07/2005 11:46	548813.57	5559325.68	74	0.74	247.15
Delta Channel	between stakes	14/07/2005 11:49	548778.30	5559408.76	34	0.34	247.55
Delta Channel		14/07/2005 11:50	548775.97	5559403.18	84	0.84	247.05
Delta Channel		14/07/2005 11:52	548773.61	5559401.30	100	1.00	246.89
Delta Channel		14/07/2005 11:53	548768.87	5559399.40	118.5	1.19	246.71
Delta Channel		14/07/2005 11:54	548762.95	5559395.64	110	1.10	246.79
Delta Channel	stakes closet to bridge (tracers site 1)	14/07/2005 11:56	548745.73	5559456.64	52	0.52	247.37
Delta Channel		14/07/2005 11:58	548740.99	5559454.75	89.5	0.90	247.00
Delta Channel		14/07/2005 11:59	548738.63	5559452.87	106.5	1.07	246.83
Delta Channel		14/07/2005 12:02	548734.09	5559450.64	145	1.45	246.44
Delta Channel		14/07/2005 12:05	548731.52	5559449.10	154	1.54	246.35

Location:	Cadham Bay					
Northing	Easting	Date	Time	Depth cm	DEPTH m	DEPTH at bottom sediments masl
+5558851.411	+549553.8612	29/08/2005	14:05	60	0.60	247.15
+5558894.326	+549624.8661	29/08/2005		120	1.20	246.55
+5558911.477	+549675.4031	29/08/2005		140	1.40	246.35
+5558925.212	+549717.4032	29/08/2005		150	1.50	246.25
+5558941.185	+549760.8102	29/08/2005		160	1.60	246.15
+5558956.087	+549808.5113	29/08/2005		155	1.55	246.20
+5558971.015	+549859.0679	29/08/2005		155	1.55	246.20
+5558988.175	+549910.3174	29/08/2005		160	1.60	246.15
+5558999.789	+549963.0467	29/08/2005		160	1.60	246.15
+5559018.081	+550016.4270	29/08/2005		160	1.60	246.15
+5559034.144	+550069.1139	29/08/2005		180	1.80	245.95
+5559047.963	+550119.6796	29/08/2005		165	1.65	246.10
+5559061.810	+550173.1009	29/08/2005		165	1.65	246.10
+5559075.643	+550225.0939	29/08/2005		165	1.65	246.10
+5559085.050	+550279.2706	29/08/2005		160	1.60	246.15
+5559095.536	+550329.8668	29/08/2005		160	1.60	246.15
+5559102.720	+550384.0641	29/08/2005		160	1.60	246.15
+5559114.352	+550438.2192	29/08/2005		175	1.75	246.00
+5559127.056	+550488.0798	29/08/2005		160	1.60	246.15
+5559135.341	+550540.8382	29/08/2005		160	1.60	246.15
+5559140.290	+550593.6280	29/08/2005		160	1.60	246.15
+5559153.104	+550654.9112	29/08/2005		160	1.60	246.15
+5559162.427	+550699.8051	29/08/2005		170	1.70	246.05
+5559173.005	+550759.6810	29/08/2005		160	1.60	246.15
+5559186.790	+550805.9601	29/08/2005		160	1.60	246.15
+5559196.230	+550862.9904	29/08/2005		170	1.70	246.05
+5559204.484	+550912.1779	29/08/2005		170	1.70	246.05
+5559217.240	+550967.0341	29/08/2005		165	1.65	246.10
+5559226.614	+551016.9245	29/08/2005		160	1.60	246.15
+5559236.063	+551074.6680	29/08/2005		160	1.60	246.15
+5559248.828	+551130.2375	29/08/2005		150	1.50	246.25
+5559260.454	+551182.9617	29/08/2005		170	1.70	246.05
+5559273.165	+551232.8193	29/08/2005		160	1.60	246.15
+5559287.043	+551288.3773	29/08/2005		160	1.60	246.15
+5559299.776	+551340.3762	29/08/2005		150	1.50	246.25
+5559310.306	+551394.5380	29/08/2005		160	1.60	246.15
+5559321.928	+551446.5472	29/08/2005		160	1.60	246.15
+5559336.928	+551502.8074	29/08/2005		155	1.55	246.20
+5559347.411	+551551.9711	29/08/2005		150	1.50	246.25
+5559361.300	+551608.2415	29/08/2005		150	1.50	246.25
+5559373.994	+551655.9554	29/08/2005		140	1.40	246.35
+5559389.016	+551714.3562	29/08/2005		130	1.30	246.45
+5559402.872	+551767.0562	29/08/2005		130	1.30	246.45
+5559412.288	+551820.5132	29/08/2005		130	1.30	246.45
+5559422.802	+551872.5313	29/08/2005		145	1.45	246.30
+5559434.498	+551931.6774	29/08/2005		145	1.45	246.30
+5559443.860	+551979.4225	29/08/2005		145	1.45	246.30
+5559458.893	+552038.5356	29/08/2005		140	1.40	246.35
+5559473.801	+552084.7981	29/08/2005		130	1.30	246.45
+5559493.289	+552144.5809	29/08/2005		130	1.30	246.45
+5559515.981	+552190.7666	29/08/2005		120	1.20	246.55
+5559547.589	+552239.0063	29/08/2005		115	1.15	246.60
+5559588.037	+552281.4469	29/08/2005		100	1.00	246.75
+5559626.191	+552316.7699	29/08/2005		70	0.70	247.05

Northing	Easting	Date	Time	Depth cm	DEPTH m	DEPTH at bottom sediments masl
+5558787.727	+551972.2598	29/08/2005		40	0.40	247.35
+5558774.940	+551915.2593	29/08/2005		90	0.90	246.85
+5558755.586	+551869.0336	29/08/2005		110	1.10	246.65
+5558727.275	+551816.4680	29/08/2005		140	1.40	246.35
+5558706.789	+551768.1102	29/08/2005		140	1.40	246.35
+5558674.025	+551714.8728	29/08/2005		150	1.50	246.25
+5558644.625	+551664.4584	29/08/2005		150	1.50	246.25
+5558613.001	+551614.0651	29/08/2005		150	1.50	246.25
+5558588.076	+551566.4626	29/08/2005		160	1.60	246.15
+5558556.447	+551515.3541	29/08/2005		165	1.65	246.10
+5558529.265	+551464.2020	29/08/2005		170	1.70	246.05
+5558503.203	+551413.7527	29/08/2005		160	1.60	246.15
+5558471.581	+551363.3565	29/08/2005		160	1.60	246.15
+5558432.088	+551303.7522	29/08/2005		170	1.70	246.05
+5558403.831	+551256.1788	29/08/2005		170	1.70	246.05
+5558377.770	+551205.7273	29/08/2005		160	1.60	246.15
+5558352.836	+551156.6927	29/08/2005		160	1.60	246.15
+5558327.895	+551106.9437	29/08/2005		170	1.70	246.05
+5558305.172	+551056.4588	29/08/2005		170	1.70	246.05
+5558275.696	+550997.4686	29/08/2005		170	1.70	246.05
+5558251.889	+550949.8496	29/08/2005		170	1.70	246.05
+5558226.916	+550896.5283	29/08/2005		170	1.70	246.05
+5558203.096	+550847.4804	29/08/2005		170	1.70	246.05
+5558178.138	+550795.5863	29/08/2005		170	1.70	246.05
+5558150.963	+550744.4269	29/08/2005		165	1.65	246.10
+5558124.901	+550693.2565	29/08/2005		165	1.65	246.10
+5558101.049	+550640.6363	29/08/2005		165	1.65	246.10
+5558077.225	+550590.8719	29/08/2005		170	1.70	246.05
+5558051.144	+550537.5578	29/08/2005		170	1.70	246.05
+5558028.405	+550484.9256	29/08/2005		170	1.70	246.05
+5558008.976	+550429.4050	29/08/2005		165	1.65	246.10
+5557995.134	+550376.6879	29/08/2005		170	1.70	246.05
+5557982.390	+550322.5318	29/08/2005		160	1.60	246.15
+5557973.037	+550274.0567	29/08/2005		130	1.30	246.45
+5557965.841	+550218.4195	29/08/2005		120	1.20	246.55
+5557958.914	+550191.3464	29/08/2005		80	0.80	246.95
+5557551.818	+550414.4483	29/08/2005		20	0.20	247.55
+5557549.172	+550487.3243	29/08/2005		115	1.15	246.60
+5557550.920	+550554.4448	29/08/2005		130	1.30	246.45
+5557562.609	+550614.3287	29/08/2005		140	1.40	246.35
+5557570.921	+550669.9590	29/08/2005		160	1.60	246.15
+5557579.329	+550735.5874	29/08/2005		160	1.60	246.15
+5557582.302	+550814.1235	29/08/2005		155	1.55	246.20
+5557595.289	+550893.2780	29/08/2005		160	1.60	246.15
+5557602.567	+550956.7740	29/08/2005	15:30:00	160	1.60	246.15

Northing	Easting	Date	Time	Depth cm	DEPTH m	DEPTH at bottom sediments masl
+5557610.856	+551009.5468	29/08/2005		155	1.55	246.20
+5557620.339	+551070.8787	29/08/2005		150	1.50	246.25
+5557627.510	+551122.9478	29/08/2005		160	1.60	246.15
+5557633.653	+551183.5972	29/08/2005		160	1.60	246.15
+5557640.845	+551237.8084	29/08/2005		155	1.55	246.20
+5557646.982	+551297.7434	29/08/2005		160	1.60	246.15
+5557656.427	+551354.7895	29/08/2005		150	1.50	246.25
+5557666.992	+551412.5389	29/08/2005		155	1.55	246.20
+5557684.332	+551480.9357	29/08/2005		160	1.60	246.15
+5557690.333	+551526.5871	29/08/2005		150	1.50	246.25
+5557702.074	+551590.7523	29/08/2005		160	1.60	246.15
+5557711.460	+551641.3701	29/08/2005		165	1.65	246.10
+5557726.496	+551701.2176	29/08/2005		160	1.60	246.15
+5557742.561	+551752.4843	29/08/2005		160	1.60	246.15
+5557752.067	+551815.2417	29/08/2005		150	1.50	246.25
+5557760.315	+551863.0130	29/08/2005		155	1.55	246.20
+5557777.591	+551924.2659	29/08/2005		150	1.50	246.25
+5557797.001	+551976.2131	29/08/2005		150	1.50	246.25
+5557813.167	+552037.4761	29/08/2005		150	1.50	246.25
+5557834.235	+552145.1134	29/08/2005		120	1.20	246.55
+5557845.906	+552201.4202	29/08/2005		110	1.10	246.65
+5557865.361	+552257.6505	29/08/2005		115	1.15	246.60
+5557888.151	+552313.8476	29/08/2005		90	0.90	246.85
+5557898.502	+552348.7410	29/08/2005		50	0.50	247.25
+5557536.976	+552108.0257	29/08/2005		45	0.45	247.30
+5557571.172	+552079.8364	29/08/2005		80	0.80	246.95
+5557605.360	+552050.9334	29/08/2005	15:50	95	0.95	246.80
+5557636.151	+552015.6359	29/08/2005		105	1.05	246.70
+5557665.850	+551982.4921	29/08/2005		135	1.35	246.40
+5557697.683	+551940.0431	29/08/2005		135	1.35	246.40
+5557738.405	+551896.7939	29/08/2005		140	1.40	246.35
+5557786.986	+551861.3250	29/08/2005		150	1.50	246.25
+5557826.506	+551808.8042	29/08/2005		150	1.50	246.25
+5557860.411	+551750.6253	29/08/2005		150	1.50	246.25
+5557908.779	+551693.0211	29/08/2005		150	1.50	246.25
+5557945.006	+551644.8198	29/08/2005		155	1.55	246.20
+5557986.759	+551592.9948	29/08/2005	16:00	150	1.50	246.25

Northing	Easting	Date	Time	Depth cm	DEPTH m	DEPTH at bottom sediments masl
+5557986.759	+551592.9948	29/08/2005	16:00	150	1.50	246.25
+5558031.847	+551541.1383	29/08/2005		150	1.50	246.25
+5558073.615	+551490.7432	29/08/2005		165	1.65	246.10
+5558114.243	+551437.5032	29/08/2005		170	1.70	246.05
+5558153.788	+551387.1311	29/08/2005		160	1.60	246.15
+5558194.445	+551336.7490	29/08/2005		160	1.60	246.15
+5558235.109	+551287.0818	29/08/2005		160	1.60	246.15
+5558274.649	+551235.9979	29/08/2005		160	1.60	246.15
+5558312.013	+551189.9345	29/08/2005		170	1.70	246.05
+5558352.658	+551138.1273	29/08/2005		170	1.70	246.05
+5558391.073	+551085.6282	29/08/2005		170	1.70	246.05
+5558430.641	+551037.4035	29/08/2005		175	1.75	246.00
+5558469.098	+550989.1901	29/08/2005		170	1.70	246.05
+5558476.800	+550980.5472	29/08/2005		170	1.70	246.05
+5558511.969	+550937.3647	29/08/2005		180	1.80	245.95
+5558554.820	+550883.3981	29/08/2005		175	1.75	246.00
+5558588.845	+550836.6574	29/08/2005		170	1.70	246.05
+5558632.809	+550782.6818	29/08/2005		170	1.70	246.05
+5558670.176	+550736.6247	29/08/2005		170	1.70	246.05
+5558710.853	+550687.6804	29/08/2005		170	1.70	246.05
+5558749.327	+550640.9001	29/08/2005		170	1.70	246.05
+5558789.984	+550589.8154	29/08/2005		170	1.70	246.05
+5558827.354	+550543.7610	29/08/2005		170	1.70	246.05
+5558865.829	+550496.9827	29/08/2005		175	1.75	246.00
+5558907.606	+550446.6036	29/08/2005		170	1.70	246.05
+5558953.791	+550391.8994	29/08/2005		170	1.70	246.05
+5558988.992	+550351.5806	29/08/2005		165	1.65	246.10
+5559031.896	+550302.6212	29/08/2005		170	1.70	246.05
+5559069.329	+550262.9966	29/08/2005		165	1.65	246.10
+5559071.519	+550259.4058	29/08/2005		160	1.60	246.15
+5559113.325	+550211.8863	29/08/2005		160	1.60	246.15
+5559151.850	+550170.1106	29/08/2005		160	1.60	246.15
+5559194.749	+550120.4402	29/08/2005		150	1.50	246.25
+5559242.090	+550070.0149	29/08/2005		150	1.50	246.25
+5559280.622	+550028.9551	29/08/2005		150	1.50	246.25
+5559322.471	+549985.7228	29/08/2005		150	1.50	246.25
+5559364.327	+549943.2052	29/08/2005		140	1.40	246.35
+5559406.156	+549897.8325	29/08/2005		140	1.40	246.35
+5559444.684	+549856.0613	29/08/2005		130	1.30	246.45
+5559493.158	+549807.7720	29/08/2005		130	1.30	246.45
+5559530.628	+549771.7237	29/08/2005		125	1.25	246.50
+5559574.649	+549722.7636	29/08/2005		110	1.10	246.65
+5559614.296	+549681.6985	29/08/2005		100	1.00	246.75
+5559660.589	+549637.7163	29/08/2005	16:30	50	0.50	247.25
+5558904.793	+549554.7926	29/08/2005	17:00	70	0.70	247.05

Northing	Easting	Date	Time	Depth cm	DEPTH m	DEPTH at bottom sediments masl
+5558866.064	+549575.1462	29/08/2005		110	1.10	246.65
+5558821.882	+549606.9757	29/08/2005		120	1.20	246.55
+5558756.728	+549655.4246	29/08/2005		120	1.20	246.55
+5558714.777	+549687.9487	29/08/2005		150	1.50	246.25
+5558659.556	+549728.4519	29/08/2005		145	1.45	246.30
+5558599.949	+549775.4237	29/08/2005		150	1.50	246.25
+5558554.703	+549812.2650	29/08/2005		160	1.60	246.15
+5558501.714	+549853.4639	29/08/2005		170	1.70	246.05
+5558443.220	+549900.4281	29/08/2005		165	1.65	246.10
+5558393.507	+549935.1711	29/08/2005		160	1.60	246.15
+5558327.271	+549986.4944	29/08/2005		165	1.65	246.10
+5558277.613	+550026.9514	29/08/2005		170	1.70	246.05
+5558227.954	+550067.4092	29/08/2005		160	1.60	246.15
+5558174.954	+550107.1850	29/08/2005		165	1.65	246.10
+5558118.645	+550149.8493	29/08/2005		160	1.60	246.15
+5558070.073	+550187.4425	29/08/2005		150	1.50	246.25
+5558002.707	+550236.6403	29/08/2005		150	1.50	246.25
+5557945.281	+550278.6037	29/08/2005		130	1.30	246.45
+5557895.625	+550319.0663	29/08/2005		130	1.30	246.45
+5557842.660	+550362.4176	29/08/2005		130	1.30	246.45
+5557790.727	+550397.1897	29/08/2005		140	1.40	246.35
+5557730.027	+550445.6153	29/08/2005		140	1.40	246.35
+5557674.772	+550481.8489	29/08/2005		150	1.50	246.25
+5557627.266	+550514.4386	29/08/2005		145	1.45	246.30
+5557577.531	+550546.3359	29/08/2005		140	1.40	246.35
+5557523.415	+550585.4176	29/08/2005		140	1.40	246.35
+5557469.327	+550627.3568	29/08/2005		130	1.30	246.45
+5557415.233	+550668.5826	29/08/2005		130	1.30	246.45
+5557370.075	+550714.0097	29/08/2005		140	1.40	246.35
+5557314.869	+550755.2478	29/08/2005		140	1.40	246.35
+5557265.257	+550800.0047	29/08/2005		140	1.40	246.35
+5557213.395	+550841.9268	29/08/2005		130	1.30	246.45
+5557163.791	+550887.3996	29/08/2005		135	1.35	246.40
+5557115.196	+550922.1493	29/08/2005		140	1.40	246.35
+5557069.985	+550961.8673	29/08/2005		110	1.10	246.65
+5557018.049	+550995.9359	29/08/2005		110	1.10	246.65
+5556969.298	+551014.2601	29/08/2005		105	1.05	246.70
+5556935.001	+551032.4461	29/08/2005	17:20	50	0.50	247.25
+5556983.824	+551484.1324	29/08/2005	17:25	30	0.30	247.45
+5557042.760	+551484.2774	29/08/2005		90	0.90	246.85
+5557098.263	+551474.4554	29/08/2005		100	1.00	246.75
+5557155.997	+551465.3263	29/08/2005		110	1.10	246.65
+5557204.823	+551454.8549	29/08/2005		130	1.30	246.45
+5557260.312	+551443.6051	29/08/2005		140	1.40	246.35
+5557310.256	+551433.8376	29/08/2005		145	1.45	246.30
+5557364.648	+551424.0273	29/08/2005		140	1.40	246.35

Northing	Easting	Date	Time	Depth cm	DEPTH m	DEPTH at bottom sediments masl
+5557419.033	+551413.5030	29/08/2005		140	1.40	246.35
+5557468.998	+551405.8785	29/08/2005		150	1.50	246.25
+5557524.487	+551394.6297	29/08/2005		140	1.40	246.35
+5557576.669	+551386.2699	29/08/2005		140	1.40	246.35
+5557627.725	+551376.4927	29/08/2005		145	1.45	246.30
+5557681.012	+551367.4083	29/08/2005	17:35:00	150	1.50	246.25
+5557734.285	+551356.8959	29/08/2005		150	1.50	246.25
+5557784.229	+551347.1299	29/08/2005		150	1.50	246.25
+5557836.418	+551339.4850	29/08/2005		160	1.60	246.15
+5557890.803	+551328.9624	29/08/2005		160	1.60	246.15
+5557941.846	+551317.7580	29/08/2005		160	1.60	246.15
+5557996.224	+551306.5217	29/08/2005		160	1.60	246.15
+5558045.050	+551296.0532	29/08/2005		160	1.60	246.15
+5558099.441	+551286.2455	29/08/2005		160	1.60	246.15
+5558144.938	+551276.5236	29/08/2005		160	1.60	246.15
+5558202.639	+551263.8278	29/08/2005	17:40	160	1.60	246.15
+5558251.478	+551254.7882	29/08/2005		160	1.60	246.15
+5558304.758	+551244.9919	29/08/2005		160	1.60	246.15
+5558358.032	+551234.4818	29/08/2005		160	1.60	246.15
+5558409.061	+551221.8511	29/08/2005		160	1.60	246.15
+5558458.991	+551210.6594	29/08/2005		180	1.80	245.95
+5558513.370	+551199.4251	29/08/2005		170	1.70	246.05
+5558563.321	+551190.3759	29/08/2005		175	1.75	246.00
+5558618.798	+551177.7033	29/08/2005		180	1.80	245.95
+5558669.841	+551166.5017	29/08/2005		170	1.70	246.05
+5558720.891	+551156.0143	29/08/2005		165	1.65	246.10
+5558773.052	+551145.5164	29/08/2005		160	1.60	246.15
+5558829.648	+551133.5481	29/08/2005		160	1.60	246.15
+5558879.572	+551121.6440	29/08/2005		150	1.50	246.25
+5558932.825	+551108.9941	29/08/2005		150	1.50	246.25
+5558983.854	+551096.3658	29/08/2005		120	1.20	246.55
+5559021.685	+551098.8590	29/08/2005		55	0.55	247.20
+5559032.763	+551094.4685	29/08/2005		60	0.60	247.15
+5559080.484	+551084.7284	29/08/2005	17:55	130	1.30	246.45
+5559132.639	+551073.5179	29/08/2005		150	1.50	246.25
+5559183.682	+551062.3183	29/08/2005		160	1.60	246.15
+5559235.851	+551052.5361	29/08/2005		160	1.60	246.15
+5559286.935	+551045.6204	29/08/2005		170	1.70	246.05
+5559339.138	+551039.4081	29/08/2005		160	1.60	246.15
+5559387.998	+551032.5140	29/08/2005		160	1.60	246.15
+5559442.411	+551024.8528	29/08/2005		155	1.55	246.20
+5559492.377	+551017.2343	29/08/2005		150	1.50	246.25
+5559545.671	+551008.8702	29/08/2005		160	1.60	246.15
+5559591.196	+551002.0085	29/08/2005		150	1.50	246.25
+5559648.938	+550993.6020	29/08/2005		160	1.60	246.15
+5559695.602	+550989.5854	29/08/2005		145	1.45	246.30
+5559766.707	+550983.1931	29/08/2005		150	1.50	246.25
+5559804.462	+550977.8341	29/08/2005		130	1.30	246.45
+5559872.232	+550971.4739	29/08/2005		110	1.10	246.65
+5559891.113	+550969.1514	29/08/2005	18:00	70	0.70	247.05

Site	Site_description	Date_time	EASTING	NORTHING	Depth_cm	Depth_m	elevation_masl_bottom_sediments
Portage Creek South	west shore	8/17/06 14:45	553471.25	5555909.29	40	0.40	247.44
Portage Creek South	flow w side	8/17/06 14:45	553488.45	5555909.29	142	1.42	246.42
Portage Creek South	flow/sampling site (center)	8/17/06 14:45	553547.80	5555908.25	160	1.60	246.24
Portage Creek South	flow e side	8/17/06 14:45	553583.21	5555910.00	149	1.49	246.35
Portage Creek South	east shore	8/17/06 14:45	553593.10	5555908.77	30	0.30	247.54
Portage Creek mid	west shore	8/2/06 12:50	553471.25	5555909.29	20	0.20	247.65
Portage Creek mid		8/2/06 12:50	553488.45	5555909.29	90	0.90	246.95
Portage Creek mid		8/2/06 12:50	553501.45	5555907.73	164	1.64	246.21
Portage Creek mid		8/2/06 12:50	553524.88	5555907.70	170	1.70	246.15
Portage Creek mid	flow/sampling site (center)	8/2/06 12:50	553547.80	5555908.25	177	1.77	246.08
Portage Creek mid		8/2/06 12:50	553561.30	5555908.70	170	1.70	246.15
Portage Creek mid		8/2/06 12:50	553576.00	5555909.80	147	1.47	246.38
Portage Creek mid		8/2/06 12:50	553583.21	5555910.00	70	0.70	247.15
Portage Creek mid	east shore	8/2/06 12:50	553593.10	5555908.77	25	0.25	247.60
Portage Creek N	transect (1) at mouth of simpsons	12/07/2005 14:18	553825.01	5558106.33	93	0.93	246.97
Portage Creek N		12/07/2005 14:22	553821.30	5558110.70	108	1.08	246.82
Portage Creek N		12/07/2005 14:27	553817.79	5558113.67	99	0.99	246.91
Portage Creek N		12/07/2005 14:31	553815.80	5558119.30	100	1.00	246.90
Portage Creek N		12/07/2005 14:33	553812.94	5558122.88	113.5	1.14	246.76
Portage Creek N		12/07/2005 14:35	553805.76	5558126.52	114	1.14	246.76
Portage Creek N		12/07/2005 14:37	553798.56	5558132.00	117	1.17	246.73
Portage Creek N		12/07/2005 14:39	553792.59	5558133.80	100	1.00	246.90
Portage Creek N		12/07/2005 14:40	553788.99	5558137.47	98.5	0.99	246.91
Portage Creek N	transect (2) at stake #1 (from simpson's)	12/07/2005 14:44	553784.21	5558139.27	81	0.81	247.09
Portage Creek N		12/07/2005 14:46	553772.74	5558096.53	100	1.00	246.90
Portage Creek N		12/07/2005 14:48	553775.15	5558092.85	83.5	0.84	247.06
Portage Creek N		12/07/2005 14:50	553778.76	5558089.18	90	0.90	247.00
Portage Creek N		12/07/2005 14:51	553781.00	5558085.00	106	1.06	246.84
Portage Creek N		12/07/2005 14:53	553784.81	5558079.97	107.5	1.08	246.82
Portage Creek N		12/07/2005 14:54	553789.62	5558074.46	106	1.06	246.84
Portage Creek N	transect (3) b/w stake 1 and 2	12/07/2005 14:59	553684.33	5558010.39	78	0.78	247.12
Portage Creek N		12/07/2005 15:00	553686.79	5558003.00	113.5	1.14	246.76
Portage Creek N		12/07/2005 15:02	553689.21	5557999.32	89.5	0.90	247.00
Portage Creek N		12/07/2005 15:04	553691.50	5557996.01	113	1.13	246.77
Portage Creek N		12/07/2005 15:05	553695.25	5557990.11	111	1.11	246.79
Portage Creek N	transect (4) at stake #2	12/07/2005 15:07	553624.10	5557963.45	90	0.90	247.00
Portage Creek N		12/07/2005 15:08	553625.45	5557961.00	114.6	1.15	246.75
Portage Creek N		12/07/2005 15:10	553627.75	5557956.07	105	1.05	246.85
Portage Creek N		12/07/2005 15:12	553629.05	5557954.00	118.7	1.19	246.71
Portage Creek N		12/07/2005 15:15	553630.18	5557950.54	134.3	1.34	246.55
Portage Creek N		12/07/2005 15:17	553634.98	5557946.88	129.5	1.30	246.60

Site	Site_description	Date_time	EASTING	NORTHING	Depth_cm	Depth_m	elevation_masl_bottom_sediments
Portage Creek N	transect (5) 1/2 b/w stake 2 and 3	12/07/2005 15:20	553579.06	5557944.46	89.5	0.90	247.00
Portage Creek N		12/07/2005 15:21	553581.46	5557942.63	111	1.11	246.79
Portage Creek N		12/07/2005 15:23	553581.50	5557938.93	108.5	1.09	246.81
Portage Creek N		12/07/2005 15:25	553582.72	5557935.23	115.2	1.15	246.74
Portage Creek N		12/07/2005 15:27	553584.40	5557932.17	125.5	1.26	246.64
Portage Creek N		12/07/2005 15:28	553585.18	5557927.84	122	1.22	246.68
Portage Creek N	transect (6) 2/3 b/w stake 2 and 3	12/07/2005 15:31	553498.39	5557917.71	48.5	0.49	247.41
Portage Creek N		12/07/2005 15:33	553499.63	5557912.16	103	1.03	246.87
Portage Creek N		12/07/2005 15:34	553503.24	5557908.49	11	0.11	247.79
Portage Creek N		12/07/2005 15:36	553504.47	5557904.79	112	1.12	246.78
Portage Creek N		12/07/2005 15:37	553505.76	5557902.95	119.5	1.20	246.70
Portage Creek N		12/07/2005 15:39	553506.92	5557897.41	118.5	1.19	246.71
Portage Creek N	transect (7) at stake #3	12/07/2005 15:41	553415.58	5557866.84	72	0.72	247.18
Portage Creek N	sample site stake	12/07/2005 15:43	553418.10	5557864.30	91	0.91	246.99
Portage Creek N		12/07/2005 15:45	553422.77	5557861.35	108.5	1.09	246.81
Portage Creek N		12/07/2005 15:46	553425.19	5557857.67	115.5	1.16	246.74
Portage Creek N		12/07/2005 15:48	553426.42	5557853.97	105.6	1.06	246.84
Portage Creek N		12/07/2005 15:49	553430.40	5557852.09	102.3	1.02	246.87
Portage Creek N		12/07/2005 15:50	553434.81	5557848.50	93.5	0.94	246.96
Portage Creek N	transect (8) b/w stake 3 and 4	12/07/2005 15:53	553374.61	5557797.85	74.5	0.75	247.15
Portage Creek N		12/07/2005 15:54	553380.58	5557796.06	78.8	0.79	247.11
Portage Creek N		12/07/2005 15:56	553386.55	5557794.27	81.5	0.82	247.08
Portage Creek N		12/07/2005 15:57	553390.12	5557794.30	91.6	0.92	246.98
Portage Creek N		12/07/2005 15:59	553393.69	5557794.34	103.3	1.03	246.86
Portage Creek N		12/07/2005 16:01	553398.45	5557794.39	101.7	1.02	246.88
Portage Creek N		12/07/2005 16:03	553402.20	5557795.10	94.5	0.95	246.95
Portage Creek N		12/07/2005 16:05	553406.93	5557795.46	95.5	0.96	246.94
Portage Creek N		12/07/2005 16:07	553411.54	5557794.52	89.7	0.90	247.00
Portage Creek N	transect (9) at stake #4	12/07/2005 16:09	553386.27	5557703.45	76	0.76	247.14
Portage Creek N		12/07/2005 16:12	553394.62	5557701.68	120	1.20	246.70
Portage Creek N		12/07/2005 16:14	553398.19	5557701.72	79	0.79	247.11
Portage Creek N		12/07/2005 16:15	553402.93	5557703.62	83.2	0.83	247.06
Portage Creek N		12/07/2005 16:18	553407.69	5557703.67	89	0.89	247.01
Portage Creek N		12/07/2005 16:19	553412.45	5557703.72	100	1.00	246.90
Portage Creek N		12/07/2005 16:21	553414.81	5557705.59	104	1.04	246.86
Portage Creek N		12/07/2005 16:24	553419.53	5557705.84	109.6	1.10	246.80
Portage Creek N		12/07/2005 16:25	553425.54	5557703.85	114.5	1.15	246.75

Site	Site_description	Date_time	EASTING	NORTHING	Depth_cm	Depth_m	elevation_masl_bottom_sediments
Deep Creek	4th most S stake	13/07/2005 13:52	541421.64	5559188.88	118	1.18	246.72
Deep Creek		13/07/2005 13:58	541417.88	5559190.74	150	1.50	246.40
Deep Creek		13/07/2005 14:01	541414.73	5559191.66	169	1.69	246.21
Deep Creek		13/07/2005 14:02	541412.00	5559192.10	150	1.50	246.40
Deep Creek		13/07/2005 14:05	541407.17	5559192.52	89	0.89	247.01
Deep Creek	1/2 way between 4th stake and i	13/07/2005 14:07	541443.03	5559246.87	117	1.17	246.73
Deep Creek		13/07/2005 14:11	541440.06	5559248.37	120	1.20	246.70
Deep Creek		13/07/2005 14:12	541437.36	5559248.99	156.5	1.57	246.33
Deep Creek		13/07/2005 14:15	541434.11	5559248.33	160	1.60	246.30
Deep Creek		13/07/2005 14:16	541430.54	5559248.30	168	1.68	246.22
Deep Creek		13/07/2005 14:18	541438.34	5559337.68	36	0.36	247.54
Deep Creek		13/07/2005 14:20	541442.94	5559337.35	190	1.90	246.00
Deep Creek		13/07/2005 14:22	541446.51	5559337.38	177	1.77	246.13
Deep Creek		13/07/2005 14:23	541450.08	5559337.40	110	1.10	246.80
Deep Creek	just S of island (3rd stake at isla	13/07/2005 14:25	541454.84	5559337.44	30	0.30	247.60
Deep Creek	just N of island	13/07/2005 14:28	541455.66	5559383.78	37	0.37	247.53
Deep Creek		13/07/2005 14:30	541452.09	5559383.75	88	0.88	247.02
Deep Creek		13/07/2005 14:33	541447.32	5559385.57	179	1.79	246.11
Deep Creek		13/07/2005 14:36	541442.55	5559387.38	190	1.90	246.00
Deep Creek		13/07/2005 14:37	541438.95	5559391.06	179	1.79	246.11
Deep Creek		13/07/2005 14:40	541487.84	5559438.44	72	0.72	247.18
Deep Creek		13/07/2005 14:41	541488.59	5559434.07	110	1.10	246.80
Deep Creek		13/07/2005 14:43	541489.00	5559431.70	168	1.68	246.22
Deep Creek		13/07/2005 14:44	541489.82	5559428.52	159	1.59	246.31
Deep Creek	between island and 2nd stake	13/07/2005 14:46	541492.23	5559424.84	121	1.21	246.69
Deep Creek	3rd stake (this stake came out)	13/07/2005 14:50	541568.25	5559442.11	108	1.08	246.82
Deep Creek		13/07/2005 14:53	541564.41	5559445.47	139	1.39	246.51
Deep Creek		13/07/2005 14:54	541563.82	5559449.28	160	1.60	246.30
Deep Creek		13/07/2005 14:57	541562.23	5559451.33	168	1.68	246.22
Deep Creek		13/07/2005 14:58	541559.84	5559453.16	68	0.68	247.22
Deep Creek	half way to 1st stake	13/07/2005 15:01	541627.06	5559529.67	70	0.70	247.20
Deep Creek		13/07/2005 15:02	541623.52	5559532.32	17	0.17	247.73
Deep Creek		13/07/2005 15:04	541621.08	5559533.33	180	1.80	246.10
Deep Creek		13/07/2005 15:06	541617.66	5559536.13	139	1.39	246.51
Deep Creek		13/07/2005 15:07	541615.11	5559536.99	58	0.58	247.32
Deep Creek	1st stake (underwater and where	13/07/2005 15:17	541642.86	5559557.40	144	1.44	246.46
Deep Creek		13/07/2005 15:20	541640.27	5559558.38	180	1.80	246.10
Deep Creek	where flow is done (2st site S)	13/07/2005 15:24	541636.33	5559561.25	180	1.80	246.10
Deep Creek		13/07/2005 15:28	541633.93	5559564.93	139	1.39	246.51
Deep Creek		13/07/2005 15:33	541638.73	5559559.41	197	1.97	245.93
Deep Creek	half way to mouth	13/07/2005 15:36	541644.05	5559620.17	69	0.69	247.21
Deep Creek		13/07/2005 15:39	541641.45	5559617.70	170	1.70	246.20
Deep Creek	flow done (1st site S of lake)	13/07/2005 15:42	541639.61	5559616.83	168	1.68	246.22
Deep Creek		13/07/2005 15:43	541636.26	5559614.86	128	1.28	246.62
Deep Creek		13/07/2005 15:44	541633.55	5559613.11	28	0.28	247.62
Deep Creek		13/07/2005 15:48	541582.93	5559648.65	25	0.25	247.65
Deep Creek		13/07/2005 15:49	541585.89	5559651.99	176	1.76	246.14
Deep Creek		13/07/2005 15:51	541588.00	5559654.96	170	1.70	246.20
Deep Creek		13/07/2005 15:54	541590.22	5559658.05	132	1.32	246.58
Deep Creek	mouth of channel into lake	13/07/2005 15:55	541591.53	5559660.97	60	0.60	247.30

Site	Site_description	Date_time	EASTING	NORTHING	Depth_cm	Depth_m	elevation_masl_bottom_sediments
Cram	west of PCC dock (double docks	13/07/2005 9:42	542291.74	5558577.10	140	1.40	246.49
Cram		13/07/2005 9:45	542283.41	5558574.33	227.5	2.28	245.61
Cram		13/07/2005 9:48	542278.41	5558569.27	235	2.35	245.54
Cram		13/07/2005 9:51	542272.49	5558565.51	204.5	2.05	245.84
Cram		13/07/2005 9:53	542266.05	5558562.52	121	1.21	246.68
Cram	1st point N of PCC dock -8th sta	13/07/2005 9:56	542229.26	5558677.09	98	0.98	246.91
Cram		13/07/2005 9:59	542225.23	5558672.62	209	2.09	245.80
Cram		13/07/2005 10:00	542220.50	5558668.88	224.5	2.25	245.64
Cram		13/07/2005 10:02	542215.75	5558666.99	227	2.27	245.62
Cram		13/07/2005 10:04	542208.66	5558661.37	144	1.44	246.45
Cram	2nd point N of PCC dock - 7th st	13/07/2005 10:07	542123.41	5558757.07	160	1.60	246.29
Cram		13/07/2005 10:09	542117.47	5558754.85	225	2.25	245.64
Cram		13/07/2005 10:10	542114.00	5558750.68	252	2.52	245.37
Cram		13/07/2005 10:13	542111.61	5558744.00	218	2.18	245.71
Cram		13/07/2005 10:14	542110.48	5558736.58	145.5	1.46	246.43
Cram	3rd point N of N PCC dock - 6th	13/07/2005 10:18	542076.37	5558836.38	97	0.97	246.92
Cram		13/07/2005 10:19	542068.87	5558835.39	171.5	1.72	246.17
Cram		13/07/2005 10:20	542063.31	5558832.57	214.5	2.15	245.74
Cram		13/07/2005 10:23	542057.39	5558828.82	237	2.37	245.52
Cram		13/07/2005 10:24	542050.28	5558825.06	138	1.38	246.51
Cram	(center of ch site for flow 5th sta	13/07/2005 10:27	542032.63	5558957.24	169	1.69	246.20
Cram		13/07/2005 10:29	542027.83	5558954.61	198	1.98	245.91
Cram		13/07/2005 10:30	542023.09	5558952.72	219	2.19	245.70
Cram		13/07/2005 10:32	542017.17	5558948.96	200	2.00	245.89
Cram		13/07/2005 10:33	542011.22	5558948.92	118	1.18	246.71
Cram	5th point N of PCC dock - 4th st	13/07/2005 10:36	542027.19	5559036.14	90	0.90	246.99
Cram		13/07/2005 10:38	542020.82	5559034.31	209	2.09	245.80
Cram		13/07/2005 10:39	542015.32	5559032.34	240	2.40	245.49
Cram		13/07/2005 10:40	542009.02	5559030.84	224	2.24	245.65
Cram		13/07/2005 10:42	542005.84	5559026.71	110	1.10	246.79
Cram	6th point N of PCC dock - 3rd st	13/07/2005 10:44	541973.22	5559089.46	140	1.40	246.49
Cram		13/07/2005 10:46	541969.44	5559086.38	229	2.29	245.60
Cram		13/07/2005 10:47	541964.96	5559080.13	240	2.40	245.49
Cram		13/07/2005 10:48	541962.50	5559074.58	185	1.85	246.04
Cram		13/07/2005 10:51	541960.29	5559068.98	120	1.20	246.69
Cram	(2nd stake S of canoe launch for	13/07/2005 10:54	541917.58	5559203.93	129	1.29	246.60
Cram		13/07/2005 10:56	541911.81	5559202.33	207	2.07	245.82
Cram		13/07/2005 10:58	541906.90	5559200.14	230	2.30	245.59
Cram		13/07/2005 11:00	541902.14	5559200.10	221	2.21	245.68
Cram		13/07/2005 11:01	541895.01	5559198.19	109	1.09	246.80
Cram	(1st stake S of canoe launch for	13/07/2005 11:05	541838.77	5559390.48	157	1.57	246.32
Cram		13/07/2005 11:06	541834.02	5559388.59	157	1.57	246.32
Cram		13/07/2005 11:08	541825.72	5559390.15	254	2.54	245.35
Cram		13/07/2005 11:16	541818.08	5559389.45	236	2.36	245.53
Cram		13/07/2005 11:18	541811.40	5559390.27	157	1.57	246.32
Cram	9th site N of PCC dock	13/07/2005 11:24	541862.52	5559488.04	116.5	1.17	246.72
Cram		13/07/2005 11:27	541854.67	5559486.98	201	2.01	245.88
Cram		13/07/2005 11:30	541847.53	5559486.92	258	2.58	245.31
Cram		13/07/2005 11:32	541841.55	5559490.58	226	2.26	245.63
Cram		13/07/2005 11:33	541835.59	5559492.38	72	0.72	247.17

Appendix B: Comparison of methods used to measure chlorophyll a on NDS

Table B.1: Comparison of straight extraction and scrubbing and filter methods used to measure chlorophyll a on NDS.

			Delta Channel								Cadham Bay east						
			straight extraction			scrub and filter					straight ex			scrub and filter			
Vial ID	N conc	P conc	Total chlorophyll a (ug/cm2)	Average Total chlorophyll (ug/cm2)	Standard Deviation (+/- ug/cm2)	Total chlorophyll a (ug/cm2)	Average Total chlorophyll (ug/cm2)	Standard Deviation (+/- ug/cm2)	factor		Total chlorophyll a (ug/cm2)	Average Total chlorophyll (ug/cm2)	Standard Deviation (+/- ug/cm2)	Total chlorophyll a (ug/cm2)	Average Total chlorophyll (ug/cm2)	Standard Deviation (+/- ug/cm2)	factor
A1	0	0	1.95	2.26	0.28	2.87	2.09	0.65	**1.08**		3.84	3.54	0.28	lost	3.13	0.29	**1.13**
A2	0	0	2.20			2.01					3.49			2.81			
A3	0	0	2.26			1.28					3.66			3.35			
A4	0	0	2.62			2.18					3.18			3.23			
B1	0	0.05	1.32	1.50	0.46	lost	1.41	0.41	**1.06**		3.19	3.54	0.45	2.81	2.87	0.29	**1.23**
B2	0	0.05	2.14			1.83					3.89			2.56			
B3	0	0.05	1.07			1.02					3.12			3.25			
B4	0	0.05	1.45			1.38					3.97			2.87			
G1	0.5	0	6.65	6.95	1.48	5.72	6.47	2.60	**1.08**		10.57	10.64	1.66	9.76	9.31	2.15	**1.14**
G2	0.5	0	8.72			7.64					9.28			7.08			
G3	0.5	0	7.27			5.12					9.70			8.30			
G4	0.5	0	5.15			7.38					12.99			12.08			
H1	0.5	0.05	11.45	11.58	0.24	9.70	10.75	2.36	**1.16**		19.33	19.01	0.85	18.48	16.28	1.83	**1.17**
H2	0.5	0.05	11.92			9.82					17.74			16.60			
H3	0.5	0.05	11.38			13.11					19.46			15.97			
H4	0.5	0.05	11.58			10.38					19.51			14.06			

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Table B.2: Concentrations of N and P in new and used NDS, and calculated diffusion rates from used NDS following 21 day incubation period at sites in Delta Marsh.

Date sample analyzed	Site	NDS treatment	part of agar	P- PO4 ug/L	P-PO4 mg/L	P-PO4 mg/L in agar	P-PO4 g/L in agar	diffusion rate umol/cm2/day		N03-N ug/L	N mg/L	N mg/L in agar	N g/L in agar	diffusion rate umol/cm2/day
12-Jul-05	NDS new	B	Front	3853.67	3.85	541.47	1.54							
12-Jul-05	NDS new	B	Middle	4012.00	4.01	604.80	1.60							
12-Jul-05	NDS new	B	End	3850.33	3.85	540.13	1.54							
12-Jul-05	NDS new	G	Front							9842.50	9.84	9842.50	9.84	
12-Jul-05	NDS new	G	Middle							6992.50	6.99	6992.50		
12-Jul-05	NDS new	G	End							9152.50	9.15	9152.50		
12-Jul-05	NDS new	H	Front	3868.67	3.87	366.87	0.39	1.71		8427.50	8.43	8427.50		
12-Jul-05	NDS new	H	Middle	4502.00	4.50	450.20	0.45	1.62		9487.50	9.49	9487.50		
12-Jul-05	NDS new	H	End	4752.00	4.75	475.20	0.48	1.58						
18-Jul-05	Gap	B	Front	3710.33	3.71	371.03	0.37	1.74						
18-Jul-05	Gap	B	Middle	3953.67	3.95	395.37	0.40	1.70						
18-Jul-05	Gap	B	End	3743.67	3.74	374.37	0.37	1.73						
18-Jul-05	Gap	G	Front											
18-Jul-05	Gap	G	Middle											
18-Jul-05	Gap	G	End											
18-Jul-05	Gap	H	Front	4070.33	4.07	407.03	0.41	1.68						
18-Jul-05	Gap	H	Middle	4057.00	4.06	405.70	0.41	1.68						
18-Jul-05	Gap	H	End	3915.33	3.92	391.53	0.39	1.71						
18-Jul-05	Delta Channe	B	Front	3173.67	3.17	317.37	0.32	1.82						
18-Jul-05	Delta Channe	B	Middle	3222.00	3.22	322.20	0.32	1.81						
18-Jul-05	Delta Channe	B	End	3368.67	3.37	336.87	0.34	1.79						
18-Jul-05	Delta Channe	G	Front							3897.50	3.90	3897.50	3.90	389.75
18-Jul-05	Delta Channe	G	Middle							4092.50	4.09	4092.50		409.25
18-Jul-05	Delta Channe	G	End							3517.50	3.52	3517.50		351.75
18-Jul-05	Delta Channe	H	Front	3432.00	3.43	343.20	0.34	1.78		2977.50	2.98	2977.50		297.75
18-Jul-05	Delta Channe	H	Middle	3360.33	3.36	336.03	0.34	1.79		3967.50	3.97	3967.50		321.75
18-Jul-05	Delta Channe	H	End	3532.00	3.53	353.20	0.35	1.77		3215.50	3.15	3215.50		330.75
07-Jul-05	Center Marsh	B	Front	2785.33	2.79	278.53	0.28	1.88						
07-Jul-05	Center Marsh	B	Middle	2880.33	2.88	288.03	0.29	1.87						
07-Jul-05	Center Marsh	B	End	3222.00	3.22	322.20	0.32	1.81						
07-Jul-05	Center Marsh	G	Front							3967.50	3.97	3967.50	3.97	396.75
07-Jul-05	Center Marsh	G	Middle							3217.50	3.22	3217.50		321.75
07-Jul-05	Center Marsh	G	End							3377.50	3.38	3377.50		337.75
07-Jul-05	Center Marsh	H	Front	3588.67	3.59	358.87	0.36	1.76		5977.50	5.98	5977.50		597.75
07-Jul-05	Center Marsh	H	Middle	3922.00	3.92	392.20	0.39	1.70		2282.50	2.28	2282.50		228.25
07-Jul-05	Center Marsh	H	End	3832.00	3.83	383.20	0.38	1.72		2292.50	2.29	2292.50		229.25
11-Jul-06	Portage Cree	B	Front	3173.67	3.17	317.37	0.32	1.82						
11-Jul-06	Portage Cree	B	Middle	3493.67	3.49	349.37	0.35	1.77						
11-Jul-06	Portage Cree	B	End	3458.67	3.46	345.87	0.35	1.78						
11-Jul-06	Portage Cree	G	Front							2957.50	2.96	2957.50	2.96	295.75
11-Jul-06	Portage Cree	G	Middle							3412.50	3.41	3412.50		341.25
11-Jul-06	Portage Cree	G	End							3537.50	3.54	3537.50		353.75
11-Jul-06	Portage Cree	H	Front	3473.67	3.47	347.37	0.35	1.77		4222.50	4.22	4222.50		422.25
11-Jul-06	Portage Cree	H	Middle	3628.67	3.63	362.87	0.36	1.75		4237.50	4.24	4237.50		423.75
11-Jul-06	Portage Cree	H	End	3630.33	3.63	363.03	0.36	1.75		4092.50	4.09	4092.50		409.25

Table B.3: N and P diffusion rates from new NDS in laboratory experiments.

Day	Treatment	NO3-N							PO4-P					
		conc NO3-N µg/L	ug/day	umol/day	umol/cm/day	log(umol/day)	log(umol/cm/day)	average	PO4-P ug/L	ug/day	umol/day	umol/cm/day	log10(umol/day)	log(umol/cm/day)
1	B								1594.33	1255.38	40.53	7.078	1.61	0.85
	C								1761.83	1387.27	44.79	7.822	1.65	0.89
	D	2606.250	2052.165	146.583	25.600	2.166	1.408		-					
	E	3691.250	2906.496	207.607	36.257	2.317	1.559		1696.00	1335.43	43.11	7.530	1.63	0.88
	F	2336.250	1839.567	131.398	22.948	2.119	1.361		1990.17	1567.06	50.59	8.836	1.70	0.96
	G	6993.750	5506.890	393.349	68.695	2.595	1.837	1.835	-					
	H	7018.750	5526.575	394.755	68.941	2.596	1.838		1300.17	1023.75	33.05	5.772	1.52	0.76
	I	6891.250	5426.181	387.584	67.689	2.588	1.831		1953.50	1538.19	49.66	8.673	1.70	0.94
2	B								726.83	1038.333	33.52	5.854	1.53	0.77
	C								1060.17	1514.524	48.90	8.539	1.69	0.93
	D	1031.250	1473.214	105.230	18.378	2.022	1.264		-					
	E	1283.750	1833.929	130.995	22.877	2.117	1.359		726.00	1037.143	33.48	5.848	1.52	0.77
	F	688.750	983.929	70.281	12.274	1.847	1.089		1066.00	1522.857	49.17	8.586	1.69	0.93
	G	3868.750	5512.500	393.750	68.765	2.595	1.837	1.835	-					
	H	3876.250	5537.500	395.536	69.077	2.597	1.839		541.00	772.857	24.95	4.358	1.40	0.64
	I	3773.750	5391.071	385.077	67.251	2.586	1.828		909.33	1299.048	41.94	7.324	1.62	0.86
4	B								2190.17	1095.083	35.35	6.174	1.55	0.79
	C								2106.83	1053.417	34.01	5.940	1.53	0.77
	D	4876.250	2868.382	204.884	35.781	2.312	1.554		-					
	E	5528.750	3252.206	232.300	40.569	2.366	1.608		2261.00	1130.500	36.50	6.374	1.56	0.80
	F	3593.750	2113.971	150.998	26.371	2.179	1.421		2424.33	1212.167	39.13	6.835	1.59	0.83
	G	7243.750	4261.029	304.359	53.154	2.483	1.726	1.727	-					
	H	7323.750	4308.088	307.721	53.741	2.488	1.730		1289.33	644.667	20.81	3.635	1.32	0.56
	I	7213.750	4243.382	303.099	52.934	2.482	1.724		2456.83	1228.417	39.66	6.926	1.60	0.84
5	B								956.83	966.905	31.22	5.452	1.49	0.74
	C								2100.17	2122.274	68.52	11.966	1.84	1.08
	D	866.250	866.263	61.805	10.794	1.791	1.033		-					
	E	1276.250	1289.684	92.120	16.088	1.964	1.207		1036.83	1047.747	33.83	5.908	1.53	0.77
	F	906.250	915.789	65.414	11.424	1.816	1.058		2375.17	2400.168	77.49	13.533	1.89	1.13
	G	4706.250	4756.789	339.699	59.326	2.531	1.773	1.745	-					
	H	4726.250	4776.000	341.143	59.578	2.533	1.775		596.00	602.274	19.44	3.396	1.29	0.53
	I	3861.250	3901.789	277.985	48.548	2.444	1.686		1385.17	1399.747	45.19	7.892	1.66	0.90
6	B								840.17	849.011	27.41	4.787	1.44	0.68
	C								2398.50	2423.747	78.25	13.666	1.89	1.14
	D	836.250	845.053	60.361	10.542	1.781	1.023		-					
	E	1426.250	1441.263	102.947	17.979	2.013	1.255		1074.33	1085.642	35.05	6.121	1.54	0.79
	F	556.250	562.105	40.150	7.012	1.604	0.846		1795.17	1814.063	58.57	10.228	1.77	1.01
	G	4458.750	4505.684	321.835	56.206	2.508	1.750	1.711	-					
	H	4061.250	4104.000	293.143	51.195	2.467	1.709		352.67	356.379	11.51	2.009	1.06	0.30
	I	3751.250	3790.737	270.767	47.287	2.433	1.675		1423.50	1438.484	46.44	8.111	1.67	0.91
7	B								619.33	774.167	24.99	4.365	1.40	0.64
	C								1386.83	1733.542	55.97	9.774	1.75	0.99
	D	753.750	942.188	67.299	11.753	1.828	1.070		-					
	E	601.250	751.563	53.683	9.375	1.730	0.972		558.50	698.125	22.54	3.936	1.35	0.60
	F	361.250	451.563	32.254	5.633	1.509	0.751		968.50	1210.625	39.09	6.826	1.59	0.83
	G	3028.750	3785.938	270.424	47.227	2.432	1.674	1.677	-					
	H	3013.750	3767.188	269.085	46.994	2.430	1.672		391.00	488.750	15.78	2.756	1.20	0.44
	I	3103.750	3879.688	277.121	48.397	2.443	1.685		1281.00	1601.250	51.70	9.028	1.71	0.96
9	B								1076.83	633.431	20.45	3.572	1.31	0.55
	C								1569.33	923.137	29.80	5.205	1.47	0.72
	D	2126.250	1250.735	89.338	15.602	1.951	1.193		-					
	E	1276.250	750.735	53.624	9.365	1.729	0.972		877.67	516.275	16.67	2.911	1.22	0.46
	F	1171.250	688.971	49.212	8.595	1.692	0.934		1666.00	980.000	31.64	5.526	1.50	0.74
	G	6171.250	3630.147	259.296	45.284	2.414	1.656	1.664	-					
	H	6028.750	3546.324	253.309	44.238	2.404	1.646		571.83	336.373	10.86	1.897	1.04	0.28
	I	6668.750	3996.912	279.779	48.861	2.447	1.689		1676.83	986.373	31.85	5.562	1.50	0.75
10	B								683.50	683.500	22.07	3.854	1.34	0.59
	C								1395.17	1395.167	45.04	7.866	1.65	0.90
	D	831.250	831.250	59.375	10.369	1.774	1.016		-					
	E	918.750	918.750	65.625	11.461	1.817	1.059		782.67	782.667	25.27	4.413	1.40	0.64
	F	466.250	466.250	33.304	5.816	1.522	0.765		1176.83	1176.833	37.99	6.635	1.58	0.82
	G	3423.750	3423.750	244.554	42.709	2.388	1.631	1.581	-					
	H	3413.750	3413.750	243.839	42.595	2.387	1.629		399.50	399.500	12.87	2.247	1.11	0.35
	I	2433.750	2433.750	173.839	30.360	2.240	1.482		970.17	970.167	31.32	5.470	1.50	0.74

Day	Treatment	NO3-N							PO4-P					
		conc NO3-N µg/L	ug/day	umol/day	umol/cm/day	log(umol/day)	log(umol/cm/day)	average	PO4-P ug/L	ug/day	umol/day	umol/cm/day	log10(umol/day)	log(umol/cm/day)
12	B								1041.00	533.846	17.24	3.010	1.24	0.48
	C								1659.33	850.940	27.47	4.798	1.44	0.68
	D	1376.250	705.769	50.412	8.804	1.703	0.945		-					
	E	1723.750	883.974	63.141	11.027	1.800	1.042		1325.17	679.573	21.94	3.832	1.34	0.58
	F	993.750	509.615	36.401	6.357	1.561	0.803		1903.50	976.154	31.52	5.504	1.50	0.74
	G	6313.750	3237.821	231.273	40.390	2.364	1.606	1.593	-					
	H	6743.750	3458.333	247.024	43.141	2.393	1.635		738.50	378.718	12.23	2.135	1.09	0.33
	I	5381.250	2759.615	197.115	34.425	2.295	1.537		1611.83	826.581	26.69	4.661	1.43	0.67
13	B								572.67	636.296	20.54	3.588	1.31	0.55
	C								1395.17	1550.185	50.05	8.740	1.70	0.94
	D	596.250	662.500	47.321	8.264	1.675	0.917		-					
	E	771.250	856.944	61.210	10.690	1.787	1.029		793.50	881.667	28.46	4.971	1.45	0.70
	F	513.750	570.833	40.774	7.121	1.610	0.853		1700.17	1889.074	60.99	10.651	1.79	1.03
	G	3203.750	3559.722	254.266	44.405	2.405	1.647	1.647	-					
	H	3203.750	3559.722	254.266	44.405	2.405	1.647		576.00	640.000	20.66	3.609	1.32	0.56
	I	3203.750	3559.722	254.266	44.405	2.405	1.647		1397.67	1552.963	50.14	8.756	1.70	0.94
15	B								1018.50	509.250	16.44	2.871	1.22	0.46
	C								1289.33	644.667	20.81	3.635	1.32	0.56
	D	1566.250	783.125	55.938	9.769	1.748	0.990		-					
	E	1521.250	760.625	54.330	9.488	1.735	0.977		816.83	408.417	13.19	2.303	1.12	0.36
	F	986.250	493.125	35.223	6.151	1.547	0.789		1255.17	627.583	20.26	3.539	1.31	0.55
	G	5916.250	2958.125	211.295	36.901	2.325	1.567	1.549	-					
	H	5701.250	2850.625	203.616	35.560	2.309	1.551		571.83	285.917	9.23	1.612	0.97	0.21
	I	5436.250	2718.125	194.152	33.907	2.288	1.530		1105.17	552.583	17.84	3.116	1.25	0.49
17	B								1002.67	500.464	16.16	2.822	1.21	0.45
	C								2475.17	1235.438	39.89	6.966	1.60	0.84
	D	1028.750	513.484	36.677	6.405	1.564	0.807		-					
	E	526.250	262.669	18.762	3.277	1.273	0.515		611.83	305.386	9.86	1.722	0.99	0.24
	F	651.250	325.061	23.219	4.055	1.366	0.608		2466.00	1230.863	39.74	6.940	1.60	0.84
	G	5726.250	2858.163	204.154	35.654	2.310	1.552	1.529	-					
	H	5686.250	2838.198	202.728	35.405	2.307	1.549		627.67	313.289	10.11	1.766	1.00	0.25
	I	4916.250	2453.865	175.276	30.611	2.244	1.486		1872.67	934.711	30.18	5.270	1.48	0.72
18	B								511.75	467.886	15.11	2.638	1.18	0.42
	C								1022.17	934.552	30.17	5.269	1.48	0.72
	D	618.125	565.143	40.367	7.050	1.606	0.848		-					
	E	1086.875	993.714	70.980	12.396	1.851	1.093		837.58	765.790	24.72	4.318	1.39	0.64
	F	591.875	541.143	38.653	6.750	1.587	0.829		1038.83	949.790	30.66	5.355	1.49	0.73
	G	3461.875	3165.143	226.082	39.483	2.354	1.596	1.597	-					
	H	3444.375	3149.143	224.939	39.284	2.352	1.594		524.25	479.314	15.47	2.703	1.19	0.43
	I	3493.125	3193.714	228.122	39.840	2.356	1.600		1083.83	990.933	31.99	5.587	1.51	0.75
20	B								1423.50	474.500	15.32	2.675	1.19	0.43
	C								1685.17	561.722	18.14	3.167	1.26	0.50
	D	2126.250	708.750	50.625	8.841	1.704	0.947		-					
	E	1938.750	923.214	65.944	11.517	1.819	1.061		1435.17	478.389	15.44	2.697	1.19	0.43
	F	1473.750	701.786	50.128	8.754	1.700	0.942		1713.50	571.167	18.44	3.220	1.27	0.51
	G	6851.250	3262.500	233.036	40.698	2.367	1.610	1.612	-					
	H	6903.750	3287.500	234.821	41.010	2.371	1.613		929.33	309.778	10.00	1.747	1.00	0.24
	I	6903.750	3287.500	234.821	41.010	2.371	1.613		1748.50	582.833	18.82	3.286	1.27	0.52

Day	Treatment	NO3-N							PO4-P					
		conc NO3-N µg/L	ug/day	umol/day	umol/cm/day	log(umol/day)	log(umol/cm/day)	average	PO4-Pug/L	ug/day	umol/day	umol/cm/day	log10(umol/day)	log(umol/cm/day)
22	B								825.17	347.947	11.23	1.962	1.05	0.29
	C								1471.00	620.275	20.03	3.497	1.30	0.54
	D	1393.750	587.701	41.979	7.331	1.623	0.865		-					
	E	1201.250	506.530	36.181	6.319	1.558	0.801		819.33	345.483	11.15	1.948	1.05	0.29
	F	1286.250	542.372	38.741	6.766	1.588	0.830		1366.00	576.000	18.60	3.248	1.27	0.51
	G	8121.250	3424.480	244.606	42.718	2.388	1.631	1.632	-					
	H	8171.250	3445.564	246.112	42.981	2.391	1.633		775.17	326.864	10.55	1.843	1.02	0.27
	I	8126.250	3426.589	244.756	42.745	2.389	1.631		1398.50	589.704	19.04	3.325	1.28	0.52
24	B								996.00	343.448	11.09	1.936	1.04	0.29
	C								1711.00	590.003	19.05	3.327	1.28	0.52
	D	1591.250	548.707	39.193	6.845	1.693	0.835		-					
	E	1168.750	403.017	28.787	5.027	1.459	0.701		1070.17	369.023	11.91	2.081	1.08	0.32
	F	908.750	313.362	22.383	3.909	1.350	0.592		1677.67	578.508	18.68	3.262	1.27	0.51
	G	6853.750	2363.362	168.812	29.482	2.227	1.470	1.464	-					
	H	6853.750	2363.362	168.812	29.482	2.227	1.470		695.17	239.713	7.74	1.352	0.89	0.13
	I	6601.250	2276.293	162.592	28.395	2.211	1.453		1376.00	474.483	15.32	2.675	1.19	0.43
26	B								747.67	383.419	12.38	2.162	1.09	0.33
	C								2142.67	1098.803	35.48	6.196	1.55	0.79
	D	861.250	441.667	31.548	5.510	1.499	0.741		-					
	E	636.250	326.282	23.306	4.070	1.367	0.610		887.67	455.214	14.70	2.567	1.17	0.41
	F	501.250	257.051	18.361	3.207	1.264	0.506		2249.33	1153.504	37.24	6.504	1.57	0.81
	G	5811.250	2980.128	212.866	37.175	2.328	1.570	1.525	-					
	H	6143.750	3150.641	225.046	39.302	2.352	1.594		747.67	383.419	12.38	2.162	1.09	0.33
	I	4026.250	2064.744	147.482	25.756	2.169	1.411		1961.83	1006.068	32.48	5.673	1.51	0.75
27	B								518.50	432.083	13.95	2.436	1.14	0.39
	C								994.33	828.611	26.75	4.672	1.43	0.67
	D	521.250	521.250	37.232	6.502	1.671	0.813		-					
	E	336.250	336.250	24.018	4.195	1.381	0.623		546.00	455.000	14.69	2.565	1.17	0.41
	F	321.250	321.250	22.946	4.007	1.361	0.603		1230.17	1025.139	33.10	5.780	1.52	0.76
	G	2726.250	2726.250	194.732	34.008	2.289	1.532	1.557	-					
	H	3141.250	3141.250	224.375	39.185	2.351	1.593		329.33	274.444	8.86	1.547	0.95	0.19
	I	2826.250	2826.250	201.875	35.256	2.305	1.547		1065.17	887.639	28.66	5.005	1.46	0.70
29	B								834.33	333.733	10.77	1.882	1.03	0.27
	C								1873.50	749.400	24.19	4.225	1.38	0.63
	D	1446.250	492.083	34.435	6.014	1.537	0.779		-					
	E	516.250	172.083	12.292	2.147	1.090	0.332		832.67	333.067	10.75	1.878	1.03	0.27
	F	761.250	253.750	18.125	3.166	1.258	0.500		2065.17	826.067	26.67	4.658	1.43	0.67
	G	6641.250	2213.750	158.125	27.615	2.199	1.441	1.434	-					
	H	6528.750	2176.250	155.446	27.147	2.192	1.434		652.67	261.067	8.43	1.472	0.93	0.17
	I	6418.750	2139.583	152.827	26.690	2.184	1.426		2074.33	829.733	26.79	4.678	1.43	0.67
31	B								1021.00	340.333	10.99	1.919	1.04	0.28
	C								2830.17	943.389	30.46	5.319	1.48	0.73
	D	1078.750	359.583	25.685	4.486	1.410	0.652		-					
	E	793.750	264.583	18.899	3.301	1.276	0.519		1466.83	488.944	15.79	2.757	1.20	0.44
	F	576.750	192.917	13.780	2.407	1.139	0.381		3456.00	1152.000	37.19	6.496	1.57	0.81
	G	6718.750	2239.583	159.970	27.938	2.204	1.446	1.432	-					
	H	6221.250	2073.750	148.125	25.869	2.171	1.413		1023.50	341.167	11.01	1.924	1.04	0.28
	I	6581.250	2193.750	156.696	27.366	2.195	1.437		3106.00	1035.333	33.43	5.838	1.52	0.77
	mean				42.976							3.500		
					33.3333333							2.173913043		

Appendix C: Summary of sediment and macrophyte sampling, analysis and results at sample sites, from 2003 to 2005.

Methods

Macrophyte biomass

From 2003 to 2005, submerged open-water macrophytes were also sampled at study sites. In 2003 and 2004 submerged macrophytes were sampled once during August when biomass was maximal. Submerged macrophytes were collected from three random sampling locations at each study site in 2003, and at five random sampling locations in 2004, using an open ended plastic barrel cylinder to delineate a sample area (0.24 m^2). All above-sediment macrophyte biomass was harvested using a hand-held rake, gently rinsed to remove epiphytes and macroinvertebrates, dried at 105 °C for at least 48 hours, and weighed to determine macrophyte biomass per square meter (g/m^2). Samples were also collected from each sample site for identification to genus. In 2005, due to high water levels, the previous method could not be used to sample submerged macrophytes, because the water was too deep for the sampling barrel. As result, in 2005 macrophyte percentage cover was determined at sample sites by canoeing along random transects at the sample site and visually estimating percentage cover and percentage species composition. Samples were also collected for identification.

Sediment grain size and chemistry

Surficial sediment samples were also collected at sample sites from 2003 to 2005 for determination of organic matter content and grain size. During 2003 sediments were sampled and examined three times over the field seasons, on June 23/24, July 14/15, and August 5/6, 2003. In 2004 samples were collected fours times over the field season, on June 14/15, July 5, July 19-21, and August 12, 2004. In 2005 sediment samples were collected on five occasions, on May 16, June 6, June 27, July 18 and August 8, 2005. Sediment samples were collected using a plastic tube (10 cm diameter) which was embedded in the surface of the sediments to at least 20 cm. A plastic ball was used to create suction in the tube so the tube containing the sediment sample could be brought up to the surface, and before it was pulled above the water surface, a core extruder was placed up the bottom of the tube. The tube was then slowly lowered down on the extruder causing the extrusion of the sediment core at the barrel top. Approximately 2 cm of uppermost surface sediment was then collected in a 250 ml sample container, and taken back to the lab for analysis. Two sediment cores where collected from each site in order to obtain at least 200 ml of sample. For determination of organic and inorganic matter three 2 ml subsamples were placed in pre-weighed crucibles. The samples were weighed, and dried at 100 °C for a minimum of 48 hours and reweighed. Samples were then incinerated at 550 °C for one hour. The percentage of sand, silt and clay in the sediment samples was determined by hydrometric analysis using the ASTM D422 standard method for particle-size analysis of soils (ASTM International, 1963).

In 2004 sediment samples were also examined for total P content, and 2005 sediment pore-water from the sediment samples was examined for total P and total N content. In 2004 sediments were prepared for total P by an ignition method described by Anderson (1976) and analyzed via the colorimetric acid molybdate method described above. In 2005 to separate the sediment-pore water from the sediment, samples were centrifuged at 3000 RPM for 10 minutes (IEC Centra CL2 benchtop centrifuge). Pore-water samples were then analyzed for the total N and total P as described above for water samples.

Results

In 2002 and 2003, aquatic macrophyte biomass was greatest in isolated sites compared to connected sites, and greater on the west section of the marsh than the east (Table 5.1 to 5.3). Biomass in isolated sites ranged from 17.5 to 144.4 g/m^2, and from 0 to 22.0 g/m^2 and 8.0 to 86.4 g/m^2, respectively in both east and west connected sites. The sparse macrophytes stands in the connected sites were composed primarily of *Stuckenia* sp. whereas macrophytes in isolated sites were more diverse and composed of *Ceratophyllum* sp., *Myriophyllum* sp., *Potamogeton* spp., *Stuckenia* sp., *Utricularia* sp. and *Najas* sp.

Surficial sediments were predominately sandy loam with the exception of some locations in the east connected section of the marsh, which were largely sand (Table 5.1 to 5.3). Sediment organic matter content increased significantly with increasing distance from Lake Manitoba in the east and west connected sites (r^2 =0.62, p = 0.0003) and was significantly higher in the west side of the marsh ($F_{4,12}$ =14.45, p = 0.0003) (Table 5.1 and 5.2). Sediment organic matter was significantly greater in isolated compared to connected sites in both 2004 and 2005 ($F_{4,12}$ =189.78, p = <0.0001)) (Table 5.1 and 5.3). Sediment

TP, measured in 2005 only, did not vary significantly with distance from Lake Manitoba in the east (r^2 =0.19, p = 0.27), but was significantly higher in isolated sites ($F_{4,12}$ =10.26, p = 0.0022). Conversely, sediment TN increased significantly inland (r^2 =0.78, p = 0.0036), and was also significantly higher in isolated sites F (4,12) =4.74, p =0.0336) (Table 5.1 and 5.2)

Table C1 Yearly mean ± SE of sediment chemistry and macrophyte biomass at sample sites in Lake Manitoba and Delta Marsh from 2003 to 2005 Refer to Figure 3.1 and Table 3.1 for site descriptions.

Site Code	Year	Sed Org %	Sed TP mg/L	Sed TN mg/L	Sed Grain Class	Macrophyte Biomass g / m^2
LK	2003	5.5 ± 3.4	N/A	N/A	sand	0.0
DCh	2003	12.2 ± 0.7	N/A	N/A	sandy loam	18.0 ± 8.4
CadE	2003	12.0 ± 0.8	N/A	N/A	sandy loam	0.0
Simp	2003	21.9 ± 0.7	N/A	N/A	sandy loam	4.5 ± 3.9
PCN	2003	15.7 ± 2.1	N/A	N/A	sandy loam	1.2 ± 0.9
PCS	2003	1.8 ± 0.4	N/A	N/A	sand	16.7 ± 13.1
DCrk	2003	7.1 ± 0.5	N/A	N/A	sandy loam	26.6 ± 12.5
Canv	2003	9.1 ± 0.5	N/A	N/A	sandy loam	86.4 ± 43.3
Carp	2003	10.9 ± 1.4	N/A	N/A	sandy loam	8.1 ± 5.5
BLSE	2003	20.3 ± 2.1	N/A	N/A	sandy loam	62.7 ± 33.4
SCrk	2003	17.6 ± 0.8	N/A	N/A	sandy loam	61.7 ± 27.4
LCrk	2003	17.8 ± 5.3	N/A	N/A	sandy loam	24.6 ± 14.8
BLNW	2003	24.7 ± 0.6	N/A	N/A	sandy loam	53.7 ± 43.5
Center	2003	21.4 ± 0.7	N/A	N/A	sandy loam	171.0 ± 52.3
LK	2004	4.1 ± 2.3	N/A	N/A	sand	0.0
DelCh	2004	12.6 ± 0.7	N/A	N/A	sandy loam	3.1 ± 1.4
CadW	2004	7.9 ± 0.1	N/A	N/A	sand	0.0
CadE	2004	12.9 ± 0.5	N/A	N/A	sandy loam	2.8 ± 1.5
Gap	2004	9.9 ± 2.6	N/A	N/A	sandy loam	0.0
Simp	2004	15.9 ± 0.8	N/A	N/A	sandy loam	3.1 ± 2.0
PCS	2004	1.6 ± 0.1	N/A	N/A	sand	3.9 ± 1.5
PCB	2004	N/A*	N/A	N/A	rock*	86.3 ± 17.2
Center	2004	24.2 ± 1.0	N/A	N/A	sandy loam	113.0 ± 14.1
EBC	2004	22.8 ± 0.3	N/A	N/A	sandy loam	17.5 ± 7.5
NaeP	2004	44.3 ± 2.1	N/A	N/A	sandy loam	144.4 ± 28.8
CresP	2004	20.2 ± 0.2	N/A	N/A	sandy loam	31.8 ± 13.4
LouP	2004	37.0 ± 2.6	N/A	N/A	sandy loam	59.7 ± 25.2
RichP	2004	13.6 ± 2.3	N/A	N/A	sandy loam	78.9 ± 34.0
LK	2005	0.6 ± 0.2	7.804 ± 0.761	0.230 ± 0.054	sand	N/A
DCh	2005	11.5 ± 1.6	15.107 ± 1.065	0.425 ± 0.057	sandy loam	N/A
CadW	2005	6.3 ± 2.1	8.094 ± 2.731	0.785 ± 0.384	sand	N/A
CadE	2005	10.8 ± 0.4	13.115 ± 2.100	0.657 ± 0.077	sandy loam	N/A
Gap	2005	15.5 ± 0.2	11.725 ± 0.795	1.891 ± 0.226	sandy loam	N/A
Simp	2005	15.1 ± 0.4	11.566 ± 1.971	1.110 ± 0.082	sandy loam	N/A
PCN	2005	20.4 ± 0.5	12.485 ± 2.051	2.457 ± 0.516	sandy loam	N/A
PCS	2005	2.7 ± 0.7	11.855 ± 1.331	1.998 ± 0.455	sand	N/A
PCB	2005	N/A*	N/A*	N/A*	rock*	N/A
Center	2005	19.2 ± 0.4	10.868 ± 1.595	2.620 ± 0.384	sandy loam	N/A
EBC	2005	23.9 ± 0.4	13.575 ± 1.488	2.436 ± 0.255	sandy loam	N/A
NaeP	2005	41.8 ± 0.9	13.251 ± 2.137	1.442 ± 0.251	sandy loam	N/A
CresP	2005	20.1 ± 0.4	18.698 ± 4.281	2.025 ± 0.461	sandy loam	N/A
LouP	2005	24.3 ± 0.9	25.371 ± 9.263	2.100 ± 0.279	sandy loam	N/A
RichP	2005	13.7 ± 1.0	12.638 ± 1.257	1.450 ± 0.277	sandy loam	N/A

Appendix D: Tables of average yearly water chemistry at sample sites

Yearly mean ± SE of water chemistry and stoichiometry at sample sites in Lake Manitoba and Delta Marsh. from 2002 to 2005.

Site Code	Year	NO3+N02-N mg L-1	NH3+NH4-N mg L-1	DIN-N mg L-1	TN mg L-1	PO4-P (TRP-P) mg L-1	TP mg L-1
LK	2002	n/a	0.017 ± 0.004	n/a	4.539 ± 0.604	0.013 ± 0.000	0.060 ± 0.013
DCh	2002	n/a	0.095 ± 0.062	n/a	5.779 ± 1.159	0.043 ± 0.020	0.159 ± 0.056
CadE	2002	n/a	0.216 ± 0.085	n/a	5.629 ± 0.838	0.026 ± 0.006	0.173 ± 0.032
Simp	2002	n/a	0.192 ± 0.059	n/a	10.089 ± 1.009	0.041 ± 0.029	0.152 ± 0.011
PCN	2002	n/a	0.037 ± 0.010	n/a	7.249 ± 1.111	0.057 ± 0.018	0.274 ± 0.044
PCS	2002	n/a	0.172 ± 0.133	n/a	6.201 ± 0.447	0.405 ± 0.188	0.701 ± 0.191
DCrk	2002	n/a	0.088 ± 0.055	n/a	4.880 ± 0.749	0.079 ± 0.063	0.240 ± 0.066
BLSE	2002	n/a	0.035 ± 0.010	n/a	7.208 ± 1.051	0.135 ± 0.051	0.329 ± 0.070
S Crk	2002	n/a	0.046 ± 0.019	n/a	3.246 ± 0.589	0.756 ± 0.216	0.851 ± 0.315
BLNW	2002	n/a	0.039 ± 0.005	n/a	8.284 ± 0.977	0.074 ± 0.047	0.298 ± 0.064
LK	2003	0.113 ± 0.013	0.018 ± 0.002	0.131 ± 0.012	6.553 ± 1.314	0.016 ± 0.002	0.074 ± 0.012
DelCh	2003	0.146 ± 0.028	0.155 ± 0.064	0.301 ± 0.081	8.204 ± 1.005	0.065 ± 0.016	0.191 ± 0.036
CadE	2003	0.184 ± 0.038	0.200 ± 0.091	0.384 ± 0.103	9.813 ± 0.910	0.119 ± 0.020	0.245 ± 0.048
Simp	2003	0.184 ± 0.028	0.254 ± 0.106	0.438 ± 0.124	10.868 ± 1.251	0.149 ± 0.024	0.330 ± 0.048
PCN	2003	0.261 ± 0.036	0.532 ± 0.151	0.793 ± 0.167	0.253 ± 1.857	0.370 ± 0.064	0.627 ± 0.125
PCS	2003	0.241 ± 0.026	0.423 ± 0.121	0.663 ± 0.118	10.221 ± 1.351	0.902 ± 0.081	1.257 ± 0.187
DCrk	2003	0.107 ± 0.015	0.033 ± 0.005	0.140 ± 0.017	8.094 ± 1.213	0.037 ± 0.006	0.217 ± 0.034
Canv	2003	0.091 ± 0.017	0.053 ± 0.010	0.145 ± 0.024	8.042 ± 1.062	0.066 ± 0.013	0.281 ± 0.076
Carp	2003	0.115 ± 0.015	0.070 ± 0.024	0.187 ± 0.029	8.502 ± 1.202	0.109 ± 0.032	0.322 ± 0.064
BLSE	2003	0.129 ± 0.022	0.375 ± 0.163	0.528 ± 0.174	9.268 ± 0.665	0.292 ± 0.079	0.504 ± 0.134
SCrk	2003	0.122 ± 0.021	0.215 ± 0.072	0.337 ± 0.076	9.668 ± 0.998	0.418 ± 0.090	0.696 ± 0.131
LCrk	2003	0.116 ± 0.015	0.516 ± 0.213	0.632 ± 0.210	10.553 ± 0.874	0.558 ± 0.105	0.745 ± 0.163
BLNW	2003	0.125 ± 0.018	0.509 ± 0.145	0.634 ± 0.149	12.187 ± 0.968	0.204 ± 0.029	0.495 ± 0.059
Center	2003	0.158 ± 0.021	0.069 ± 0.022	0.227 ± 0.028	6.573 ± 0.813	0.646 ± 0.078	0.881 ±

Site Code	Year	NO3−N02-N mg L-1	NH3−NH4-N mg L-1	DIN-N mg L-1	TN mg L-1	PO4-P (TRP-P) mg L-1	TP mg L-1
LK	2004	0.122 = 0.007	0.025 = 0.004	0.147 = 0.008	8.776 = 1.106	0.016 = 0.002	0.116 = 0.011
DCh	2004	0.179 = 0.025	0.093 = 0.028	0.272 = 0.050	9.391 = 0.910	0.038 = 0.007	0.200 = 0.027
CadW	2004	0.238 = 0.029	0.120 = 0.031	0.358 = 0.057	10.803 = 0.988	0.047 = 0.013	0.259 = 0.036
CadE	2004	0.309 = 0.059	0.329 = 0.117	0.638 = 0.160	12.567 = 1.093	0.056 = 0.010	0.277 = 0.023
Gap	2004	0.288 = 0.050	0.337 = 0.129	0.625 = 0.159	11.574 = 0.959	0.069 = 0.012	0.279 = 0.020
Simp	2004	0.336 = 0.052	0.463 = 0.147	0.799 = 0.175	13.956 = 1.229	0.085 = 0.017	0.320 = 0.036
PCN	2004	0.219 = 0.031	0.535 = 0.169	0.754 = 0.184	13.154 = 1.399	0.103 = 0.020	0.447 = 0.054
PCS	2004	0.176 = 0.018	0.337 = 0.072	0.513 = 0.080	11.874 = 0.884	0.192 = 0.025	0.607 = 0.061
PCB	2004	0.064 = 0.006	0.035 = 0.004	0.099 = 0.008	6.439 = 0.839	0.656 = 0.081	0.813 = 0.088
Center	2004	0.196 = 0.028	0.076 = 0.035	0.272 = 0.046	8.134 = 1.154	0.955 = 0.055	1.248 = 0.088
EBC	2004	0.163 = 0.024	0.074 = 0.036	0.237 = 0.043	9.810 = 1.441	1.051 = 0.034	1.410 = 0.097
NaeP	2004	0.131 = 0.011	0.058 = 0.018	0.189 = 0.016	9.582 = 0.759	0.370 = 0.062	0.502 = 0.079
CresP	2004	0.138 = 0.025	0.032 = 0.009	0.170 = 0.025	8.198 = 1.278	0.061 = 0.011	0.165 = 0.026
LouP	2004	0.351 = 0.047	0.148 = 0.033	0.499 = 0.074	9.243 = 1.813	0.588 = 0.063	0.875 = 0.106
RichP	2004	0.191 = 0.033	0.020 = 0.004	0.211 = 0.037	9.367 = 1.370	0.216 = 0.023	0.319 = 0.027
LK	2005	0.156 = 0.017	0.036 = 0.005	0.175 = 0.020	4.329 = 0.572	0.037 = 0.016	0.160 = 0.031
DCh	2005	0.174 = 0.013	0.044 = 0.009	0.218 = 0.016	4.959 = 0.202	0.044 = 0.007	0.210 = 0.028
CadW	2005	0.197 = 0.013	0.040 = 0.004	0.237 = 0.014	5.803 = 0.975	0.052 = 0.006	0.212 = 0.029
CadE	2005	0.201 = 0.014	0.048 = 0.007	0.249 = 0.017	5.949 = 0.581	0.051 = 0.007	0.201 = 0.021
Gap	2005	0.202 = 0.014	0.053 = 0.007	0.255 = 0.018	5.856 = 0.556	0.056 = 0.008	0.168 = 0.011
Simp	2005	0.203 = 0.021	0.046 = 0.005	0.248 = 0.025	5.686 = 0.517	0.064 = 0.008	0.218 = 0.047
PCN	2005	0.199 = 0.014	0.048 = 0.011	0.247 = 0.018	5.137 = 0.493	0.318 = 0.093	0.491 = 0.096
PCS	2005	0.182 = 0.020	0.077 = 0.033	0.265 = 0.038	4.281 = 0.509	0.654 = 0.096	0.815 = 0.106
PCB	2005	0.212 = 0.024	0.048 = 0.012	0.262 = 0.033	3.327 = 0.426	0.778 = 0.048	0.915 = 0.054
Center	2005	0.194 = 0.010	0.103 = 0.055	0.298 = 0.059	3.527 = 0.322	0.832 = 0.064	0.766 = 0.074
EBC	2005	0.226 = 0.010	0.013 = 0.000	0.239 = 0.010	4.081 = 0.432	0.640 = 0.063	0.775 = 0.067
NaeP	2005	0.190 = 0.013	0.014 = 0.001	0.204 = 0.013	4.675 = 0.496	0.255 = 0.022	0.401 = 0.031
CresP	2005	0.173 = 0.011	0.013 = 0.000	0.185 = 0.011	3.848 = 0.310	0.087 = 0.013	0.189 = 0.012
LouP	2005	0.310 = 0.011	0.066 = 0.024	0.376 = 0.028	9.874 = 0.870	0.720 = 0.077	0.960 = 0.073
RichP	2005	0.190 = 0.010	0.023 = 0.009	0.213 = 0.016	5.059 = 0.507	0.234 = 0.028	0.384 = 0.022

Site Code	Year	TN:TP	DIN:TRP	DIN:TP
LK	2002	205.8 ± 55.9	n/a	n/a
DCh	2002	114.4 ± 36.4	n/a	n/a
CadE	2002	80.5 ± 18.8	n/a	n/a
Simp	2002	133.2 ± 16.2	n/a	n/a
PCN	2002	65.0 ± 16.5	n/a	n/a
PCS	2002	46.8 ± 28.7	n/a	n/a
DCrk	2002	60.0 ± 15.2	n/a	n/a
BLSE	2002	62.9 ± 19.0	n/a	n/a
S Crk	2002	57.1 ± 10.5	n/a	n/a
BLNW	2002	77.2 ± 22.2	n/a	n/a
LK	2003	260.3 ± 73.3	19.4 ± 2.3	5.0 ± 1.0
DelCh	2003	128.1 ± 32.1	12.6 ± 1.8	3.8 ± 1.2
CadE	2003	122.8 ± 28.8	9.2 ± 1.8	4.5 ± 1.9
Simp	2003	95.1 ± 23.2	8.8 ± 1.7	3.3 ± 1.2
PCN	2003	57.3 ± 13.2	5.7 ± 0.7	3.4 ± 0.7
PCS	2003	22.4 ± 4.7	2.0 ± 0.5	1.3 ± 0.4
DCrk	2003	96.5 ± 19.3	11.5 ± 2.1	1.7 ± 0.3
Canv	2003	86.1 ± 16.3	6.9 ± 1.1	1.5 ± 0.3
Carp	2003	72.8 ± 16.1	6.7 ± 1.3	1.2 ± 0.2
BLSE	2003	64.5 ± 13.4	5.5 ± 1.3	1.5 ± 0.3
SCrk	2003	41.5 ± 8.7	2.4 ± 0.4	2.4 ± 0.4
LCrk	2003	47.9 ± 11.3	3.1 ± 0.7	3.1 ± 0.7
BLNW	2003	60.0 ± 8.1	7.5 ± 1.7	7.5 ± 1.7
Center	2003	20.7 ± 4.3	1.0 ± 0.2	1.0 ± 0.2

Site Code	Year	TN:TP	DIN:TRP	DIN:TP
LK	2004	185.6 ± 27.2	24.5 ± 2.0	3.4 ± 0.7
DCh	2004	122.1 ± 15.6	1.4 ± 2.5	3.2 ± 0.4
CadW	2004	107.8 ± 12.4	22.0 ± 2.7	3.2 ± 0.4
CadE	2004	103.3 ± 8.7	24.6 ± 3.1	4.8 ± 0.9
Gap	2004	96.3 ± 8.3	19.5 ± 2.5	4.6 ± 0.9
Simp	2004	118.7 ± 19.5	25.0 ± 3.9	5.7 ± 1.0
PCN	2004	74.6 ± 8.7	14.1 ± 1.6	4.0 ± 1.0
PCS	2004	47.7 ± 4.0	7.2 ± 1.7	2.2 ± 0.4
PCB	2004	22.1 ± 4.2	0.4 ± 0.1	0.3 ± 0.1
Center	2004	15.4 ± 2.4	0.7 ± 0.1	0.5 ± 0.1
EBC	2004	16.4 ± 2.4	0.5 ± 0.1	0.4 ± 0.1
NaeP	2004	72.9 ± 16.8	2.4 ± 0.6	1.6 ± 0.4
CresP	2004	53.9 ± 28.6	9.7 ± 1.9	3.5 ± 0.7
LouP	2004	46.4 ± 4.0	1.8 ± 0.2	1.2 ± 0.1
RichP	2004	68.5 ± 10.1	2.4 ± 0.5	1.6 ± 0.3
LK	2005	112.2 ± 41.6	12.8 ± 2.8	5.5 ± 1.8
DCh	2005	52.8 ± 6.8	14.5 ± 2.8	2.8 ± 0.4
CadW	2005	77.8 ± 18.9	11.3 ± 1.9	3.3 ± 0.5
CadE	2005	71.4 ± 9.1	15.1 ± 2.8	2.9 ± 0.2
Gap	2005	80.7 ± 7.5	14.7 ± 3.2	3.7 ± 0.5
Simp	2005	75.9 ± 12.2	12.7 ± 2.7	3.5 ± 0.7
PCN	2005	41.3 ± 8.4	7.3 ± 2.6	1.8 ± 0.3
PCS	2005	19.8 ± 5.3	1.6 ± 0.5	0.9 ± 0.2
PCB	2005	8.1 ± 0.8	1.2 ± 0.5	1.0 ± 0.4
Center	2005	12.0 ± 1.9	0.9 ± 0.2	1.0 ± 0.2
EBC	2005	13.7 ± 2.4	1.0 ± 0.2	0.8 ± 0.1
NaeP	2005	27.8 ± 3.4	1.9 ± 0.2	1.2 ± 0.1
CresP	2005	49.6 ± 7.1	9.0 ± 2.9	2.3 ± 0.3
LouP	2005	24.8 ± 2.9	1.7 ± 0.5	0.9 ± 0.1
RichP	2005	30.7 ± 3.5	2.6 ± 0.4	1.2 ± 0.1

Site Code	Year	DOC mg L-1	Cl- mg L-1	SO4 mg L-1	Si mg L-1	Total Chloro a µg L-1	Alkalinity mgCaCO3 L-1	pH
LK	2002	9.45 ± 0.49	n/a	n/a	n/a	27.82 ± 5.65	217.6 ± 5.1	8.5 ± 0.1
DelCh	2002	11.30 ± 1.11	n/a	n/a	n/a	48.41 ± 10.06	243.0 ± 15.9	8.5 ± 0.0
CadE	2002	17.18 ± 0.89	n/a	n/a	n/a	71.31 ± 24.24	274.7 ± 8.9	8.5 ± 0.1
Simp	2002	25.38 ± 4.24	n/a	n/a	n/a	109.36 ± 39.58	314.6 ± 9.9	8.6 ± 0.1
PCN	2002	21.44 ± 1.44	n/a	n/a	n/a	166.21 ± 48.40	319.0 ± 10.6	7.6 ± 0.3
PCS	2002	22.21 ± 1.46	n/a	n/a	n/a	175.25 ± 52.44	374.1 ± 23.7	8.4 ± 0.1
DCrk	2002	14.47 ± 0.97	n/a	n/a	n/a	78.40 ± 15.31	269.2 ± 9.5	8.4 ± 0.1
BLSE	2002	18.66 ± 1.70	n/a	n/a	n/a	114.82 ± 31.02	314.8 ± 19.4	8.5 ± 0.1
S Crk	2002	15.39 ± 4.30	n/a	n/a	n/a	63.91 ± 10.81	259.6 ± 26.7	8.0 ± 0.2
BLNW	2002	22.94 ± 2.14	n/a	n/a	n/a	106.14 ± 18.99	334.4 ± 21.8	8.4 ± 0.1
LK	2003	8.89 ± 0.21	239.75 ± 15.22	140.29 ± 8.12	n/a	19.75 ± 1.73	234.9 ± 8.4	8.5 ± 0.1
DelCh	2003	14.13 ± 0.84	273.00 ± 18.78	143.67 ± 10.01	n/a	34.09 ± 5.13	296.7 ± 14.1	8.4 ± 0.1
CadE	2003	19.87 ± 0.87	347.75 ± 24.91	162.98 ± 12.65	n/a	65.81 ± 26.86	348.8 ± 16.2	8.4 ± 0.1
Simp	2003	22.50 ± 0.85	362.44 ± 21.84	167.89 ± 11.20	n/a	93.21 ± 27.99	389.2 ± 20.2	8.5 ± 0.1
PCN	2003	26.10 ± 1.35	475.71 ± 33.01	229.64 ± 20.30	n/a	136.72 ± 23.56	431.2 ± 21.7	8.2 ± 0.1
PCS	2003	27.47 ± 1.05	380.11 ± 29.31	160.51 ± 12.48	n/a	152.84 ± 31.37	448.5 ± 21.1	8.5 ± 0.1
DCrk	2003	14.19 ± 0.68	280.10 ± 19.31	180.40 ± 11.77	n/a	73.08 ± 10.81	251.4 ± 10.3	8.5 ± 0.1
CanCrk	2003	16.65 ± 1.01	290.78 ± 20.81	188.67 ± 11.45	n/a	92.68 ± 17.56	280.5 ± 13.7	8.6 ± 0.1
CarCrk	2003	16.88 ± 0.91	316.60 ± 25.31	220.78 ± 28.02	n/a	87.81 ± 13.59	271.2 ± 13.5	8.5 ± 0.1
BLSE	2003	21.03 ± 1.24	319.04 ± 32.20	231.24 ± 20.24	n/a	85.77 ± 15.58	299.7 ± 14.7	8.3 ± 0.1
SCrk	2003	22.37 ± 1.37	310.84 ± 35.27	221.45 ± 27.27	n/a	71.93 ± 13.56	338.8 ± 38.7	8.2 ± 0.2
LCrk	2003	23.94 ± 1.53	328.82 ± 34.17	254.98 ± 31.30	n/a	90.61 ± 13.98	340.3 ± 20.7	8.3 ± 0.2
BLNW	2003	33.57 ± 2.22	399.77 ± 43.53	377.58 ± 53.74	n/a	68.85 ± 20.58	356.0 ± 17.5	8.2 ± 0.1
Center	2003	29.98 ± 1.37	195.91 ± 17.87	81.34 ± 18.14	n/a	48.95 ± 18.12	386.4 ± 19.1	8.8 ± 0.1

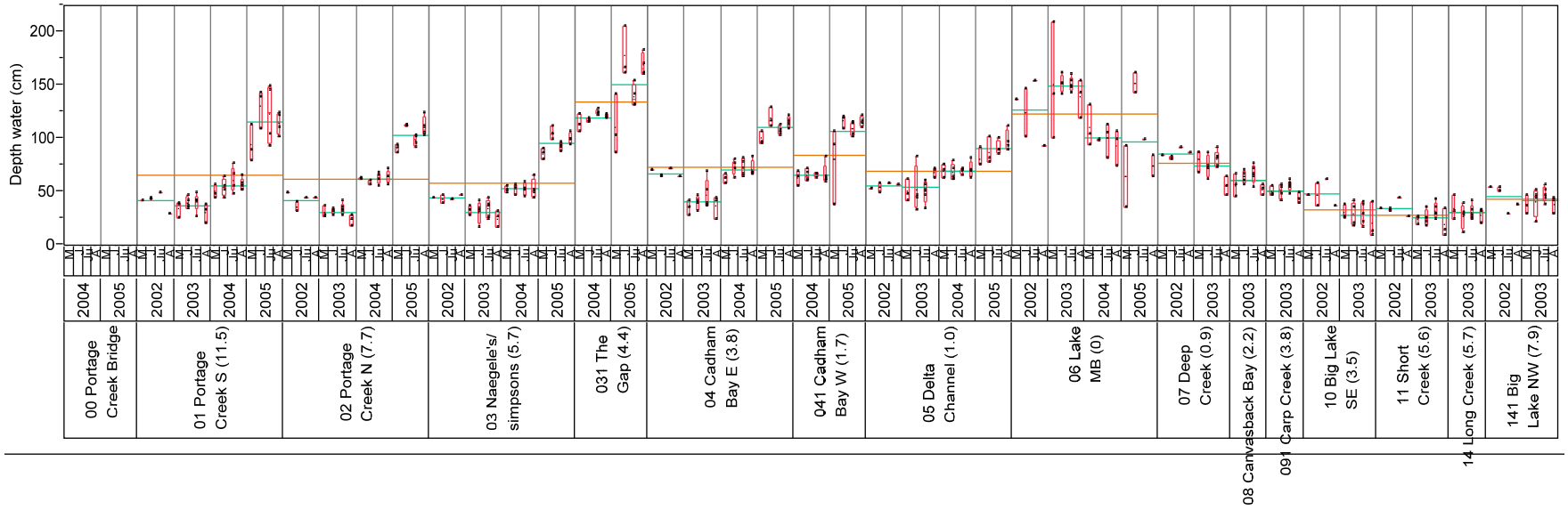
Site Code	Year	DOC mg L-1	Cl- mg L-1	SO4 mg L-1	Si mg L-1	Total Chloro a µg L-1	Alkalinity mgCaCO3 L-1	pH
LK	2004	13.54 ± 0.35	195.71 ± 11.70	172.92 ± 6.73	n/a	22.25 ± 2.64	223.3 ± 4.8	8.6 ± 0.0
DelCh	2004	17.56 ± 0.96	226.57 ± 10.77	195.63 ± 5.89	n/a	36.06 ± 4.90	247.5 ± 7.0	8.4 ± 0.1
CadW	2004	20.47 ± 0.53	240.50 ± 9.81	195.47 ± 5.56	n/a	41.22 ± 5.66	257.5 ± 7.3	8.5 ± 0.1
CadE	2004	22.63 ± 0.68	256.01 ± 14.51	201.58 ± 7.84	n/a	74.71 ± 25.61	261.5 ± 6.1	8.4 ± 0.1
GAP	2004	22.94 ± 0.67	259.42 ± 15.28	204.27 ± 8.97	n/a	66.90 ± 19.13	262.4 ± 6.3	8.4 ± 0.1
Simp	2004	24.55 ± 0.91	275.73 ± 12.07	211.55 ± 6.74	n/a	118.08 ± 26.11	279.4 ± 7.4	8.4 ± 0.1
PCN	2004	25.39 ± 0.93	253.37 ± 13.91	204.76 ± 10.16	n/a	112.03 ± 17.87	280.3 ± 5.2	8.3 ± 0.1
PCS	2004	25.89 ± 0.72	245.10 ± 9.39	276.88 ± 15.59	n/a	91.86 ± 17.21	312.5 ± 9.2	8.4 ± 0.1
PCB	2004	30.98 ± 0.82	52.00 ± 3.60	567.55 ± 32.95	n/a	21.32 ± 8.62	319.8 ± 9.9	7.6 ± 0.1
Center	2004	27.17 ± 0.88	86.99 ± 5.63	101.74 ± 3.17	n/a	9.49 ± 1.85	281.8 ± 9.4	8.1 ± 0.1
EBC	2004	26.73 ± 1.01	114.57 ± 7.48	129.04 ± 6.82	n/a	8.10 ± 1.49	319.9 ± 14.7	8.2 ± 0.1
NaeP	2004	25.27 ± 0.93	210.43 ± 13.05	242.97 ± 10.38	n/a	17.76 ± 3.49	295.0 ± 10.0	8.2 ± 0.1
CresP	2004	24.11 ± 0.80	97.51 ± 3.58	126.95 ± 6.29	n/a	12.92 ± 3.33	233.3 ± 8.4	8.2 ± 0.2
LouP	2004	62.56 ± 4.31	488.36 ± 53.78	206.53 ± 12.25	n/a	30.96 ± 13.46	464.3 ± 30.3	8.2 ± 0.2
RichP	2004	32.15 ± 0.84	197.82 ± 10.38	112.37 ± 18.12	n/a	8.02 ± 1.85	299.1 ± 9.5	9.0 ± 0.1
LK	2005	13.78 ± 0.46	131.39 ± 15.74	149.73 ± 6.39	3.56 ± 0.59	27.73 ± 3.84	230.5 ± 3.6	8.3 ± 0.1
DelCh	2005	15.75 ± 0.60	150.84 ± 12.04	145.18 ± 6.75	4.48 ± 0.71	47.64 ± 6.70	230.9 ± 5.3	8.2 ± 0.0
CadW	2005	17.12 ± 0.50	156.13 ± 8.05	136.24 ± 5.02	3.86 ± 0.48	51.83 ± 11.51	239.4 ± 3.1	8.2 ± 0.1
CadE	2005	18.25 ± 0.41	178.69 ± 6.49	143.38 ± 3.25	3.47 ± 0.46	61.28 ± 15.84	234.3 ± 6.0	8.4 ± 0.1
Gap	2005	18.45 ± 0.38	186.04 ± 5.77	146.34 ± 3.12	3.32 ± 0.48	70.91 ± 17.89	246.0 ± 6.8	8.3 ± 0.1
Simp	2005	19.26 ± 0.38	184.47 ± 9.14	152.95 ± 4.09	3.19 ± 0.49	71.54 ± 13.51	241.4 ± 2.8	8.4 ± 0.1
PCN	2005	18.60 ± 0.66	147.98 ± 19.07	147.15 ± 13.37	2.84 ± 0.45	79.52 ± 18.65	232.6 ± 7.6	8.2 ± 0.1
PCS	2005	18.24 ± 0.80	80.50 ± 17.37	123.54 ± 16.62	2.89 ± 0.52	59.54 ± 5.93	237.4 ± 11.5	7.9 ± 0.1
PCB	2005	19.07 ± 1.15	21.75 ± 3.21	174.96 ± 49.97	3.13 ± 0.50	24.92 ± 5.40	252.3 ± 21.7	7.2 ± 0.1
Center	2005	20.34 ± 0.66	49.17 ± 6.11	56.02 ± 4.17	3.38 ± 0.52	23.46 ± 5.10	256.0 ± 9.4	7.5 ± 0.1
EBC	2005	20.50 ± 0.64	111.62 ± 12.64	109.94 ± 12.98	2.88 ± 0.54	11.68 ± 1.64	306.9 ± 13.9	7.8 ± 0.1
NaeP	2005	18.71 ± 0.70	119.59 ± 6.10	52.81 ± 7.23	2.44 ± 0.59	39.52 ± 6.18	230.3 ± 9.6	7.5 ± 0.1
CresP	2005	19.73 ± 0.54	100.00 ± 2.90	91.69 ± 2.71	3.43 ± 0.52	30.66 ± 5.45	241.1 ± 5.1	7.6 ± 0.1
LouP	2005	41.60 ± 1.53	339.95 ± 13.62	107.82 ± 13.72	2.46 ± 0.45	23.28 ± 7.13	410.3 ± 10.1	8.2 ± 0.1
RichP	2005	25.91 ± 0.56	166.80 ± 7.16	145.19 ± 20.24	3.24 ± 0.39	24.46 ± 6.09	261.0 ± 5.6	7.8 ± 0.1

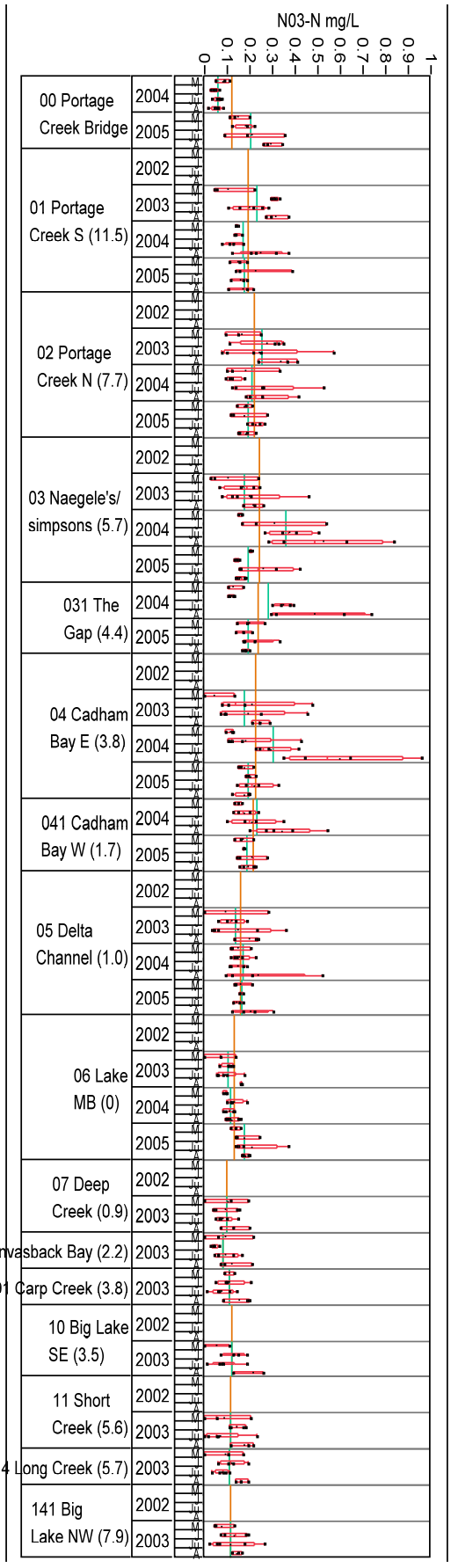
Site Code	Year	Specific Conductivity µS cm-1	Turbidity NTU	OSS mg L-1	TSS mg L-1	Depth cm
LK	2002	1497 ± 43	23.0 ± 3.0	17.07 ± 2.08	32.73 ± 7.06	127.4 ± 12.2
DelCh	2002	1491 ± 50	33.2 ± 1.2	22.60 ± 3.41	45.77 ± 11.73	56.0 ± 1.8
CadE	2002	1630 ± 31	25.6 ± 2.7	17.33 ± 1.96	33.73 ± 5.64	67.8 ± 1.5
Simp	2002	1732 ± 45	34.2 ± 2.2	24.63 ± 5.02	40.63 ± 14.13	44.8 ± 1.4
PCN	2002	1831 ± 74	27.0 ± 2.4	26.65 ± 3.01	38.97 ± 7.90	42.6 ± 3.0
PCS	2002	1748 ± 61	23.6 ± 1.0	22.43 ± 2.78	35.10 ± 4.42	42.4 ± 3.3
DCrk	2002	1469 ± 79	36.8 ± 5.9	21.53 ± 1.36	44.50 ± 3.69	86.1 ± 1.9
BLSE	2002	1559 ± 70	30.2 ± 5.4	28.80 ± 2.44	49.90 ± 8.67	48.6 ± 5.3
S Crk	2002	894 ± 184	16.1 ± 3.4	14.53 ± 2.00	24.13 ± 8.66	34.9 ± 2.7
BLNW	2002	1665 ± 58	27.0 ± 5.7	27.97 ± 2.60	50.90 ± 11.53	46.1 ± 4.9
LK	2003	1603 ± 26	21.4 ± 1.4	18.10 ± 1.79	26.33 ± 2.32	149.5 ± 5.9
DelCh	2003	1870 ± 53	31.7 ± 2.6	21.72 ± 2.47	43.47 ± 6.37	55.1 ± 3.7
CadE	2003	2102 ± 81	29.3 ± 2.2	22.08 ± 2.67	40.86 ± 6.03	41.2 ± 2.7
Simp	2003	2291 ± 83	32.2 ± 3.4	28.65 ± 4.46	61.18 ± 12.41	31.8 ± 2.1
PCN	2003	2639 ± 123	41.6 ± 3.7	41.56 ± 6.00	95.80 ± 19.41	31.5 ± 1.4
PCS	2003	2340 ± 73	32.9 ± 3.2	32.17 ± 4.37	59.82 ± 10.53	37.7 ± 2.1
DCrk	2003	1774 ± 62	27.3 ± 2.6	23.54 ± 2.58	45.01 ± 6.91	74.6 ± 3.4
CanCrk	2003	1867 ± 77	33.2 ± 3.9	34.51 ± 6.44	63.30 ± 10.20	61.7 ± 2.4
CarCrk	2003	1850 ± 78	32.5 ± 4.3	31.93 ± 4.87	76.15 ± 17.78	51.6 ± 1.7
BLSE	2003	2046 ± 111	33.7 ± 4.0	37.68 ± 5.48	79.32 ± 15.53	28.4 ± 2.8
SCrk	2003	2071 ± 127	30.3 ± 2.4	30.41 ± 3.47	65.43 ± 10.07	26.8 ± 2.3
LCrk	2003	2234 ± 143	44.4 ± 4.5	38.80 ± 3.99	96.23 ± 14.03	30.7 ± 2.3
BLNW	2003	2660 ± 167	31.1 ± 3.8	34.81 ± 7.01	82.78 ± 19.03	42.7 ± 2.5
Center	2003	1431 ± 41	14.7 ± 1.3	13.36 ± 2.19	16.32 ± 3.14	49.1 ± 2.4

Site Code	Year	Specific Conductivity µS cm-1	Turbidity NTU	OSS mg L-1	ISS mg L-1	TSS mg L-1	Depth cm
LK	2004	1379 ± 28	25.0 ± 2.6	20.07 ± 2.01	23.74 ± 3.75	43.81 ± 5.51	130.4 ± 4.6
DCh	2004	1517 ± 28	37.8 ± 4.7	29.26 ± 4.47	45.95 ± 9.02	75.21 ± 13.25	70.0 ± 1.5
CadW	2004	1565 ± 21	38.9 ± 6.0	28.87 ± 5.95	41.02 ± 14.32	69.89 ± 20.12	66.1 ± 1.6
CadE	2004	1655 ± 32	39.8 ± 5.1	34.43 ± 6.78	40.22 ± 10.42	74.65 ± 16.78	71.4 ± 1.9
Gap	2004	1680 ± 31	37.1 ± 3.8	35.66 ± 6.44	40.85 ± 10.88	76.51 ± 16.98	120.3 ± 1.4
Simp	2004	1730 ± 31	39.1 ± 3.9	41.81 ± 6.54	42.65 ± 10.49	84.46 ± 16.53	52.4 ± 1.6
PCN	2004	1760 ± 32	43.7 ± 6.4	44.59 ± 5.35	54.34 ± 10.78	98.93 ± 15.49	62.6 ± 1.1
PCS	2004	1761 ± 42	32.1 ± 3.1	37.85 ± 5.00	39.57 ± 7.97	77.42 ± 12.44	56.6 ± 2.1
PCB	2004	1534 ± 45	12.1 ± 1.3	10.77 ± 1.80	6.24 ± 4.04	17.01 ± 5.53	52.3 ± 2.4
Center	2004	886 ± 16	9.7 ± 1.3	9.33 ± 1.35	3.50 ± 1.73	12.83 ± 2.87	67.7 ± 1.2
EBC	2004	1145 ± 40	7.6 ± 0.6	7.53 ± 1.40	3.00 ± 1.50	7.53 ± 1.40	90.9 ± 1.1
NaeP	2004	1621 ± 10	8.5 ± 0.8	11.48 ± 1.31	1.46 ± 1.08	12.94 ± 2.21	76.3 ± 1.9
CresP	2004	918 ± 16	8.5 ± 0.7	7.98 ± 1.38	0.75 ± 0.68	8.73 ± 1.93	66.9 ± 1.5
LouP	2004	2801 ± 200	12.7 ± 2.2	21.16 ± 5.59	8.63 ± 3.58	29.79 ± 9.10	26.4 ± 3.4
RichP	2004	1244 ± 12	9.2 ± 0.8	9.31 ± 1.38	0.30 ± 0.23	9.61 ± 1.46	100.1 ± 0.4
LK	2005	1110 ± 47	29.2 ± 3.3	19.80 ± 2.05	37.43 ± 7.79	57.23 ± 9.42	141.0 ± 16.4
DCh	2005	1134 ± 29	29.6 ± 2.6	20.50 ± 2.18	27.80 ± 4.27	48.30 ± 5.77	90.7 ± 2.8
CadW	2005	1173 ± 24	25.5 ± 2.9	22.65 ± 6.23	19.51 ± 4.08	42.17 ± 7.07	107.0 ± 5.6
CadE	2005	1256 ± 16	26.7 ± 4.2	21.01 ± 2.83	12.48 ± 2.38	33.75 ± 4.56	111.4 ± 2.4
Gap	2005	1281 ± 13	27.5 ± 4.6	19.46 ± 2.54	15.10 ± 3.98	35.45 ± 5.68	150.7 ± 8.2
Simp	2005	1325 ± 14	22.8 ± 2.2	17.77 ± 1.91	11.54 ± 1.85	27.96 ± 2.51	96.7 ± 2.2
PCN	2005	1144 ± 81	22.6 ± 3.1	19.16 ± 2.02	13.49 ± 2.35	33.00 ± 4.05	104.0 ± 2.8
PCS	2005	892 ± 91	18.3 ± 2.0	16.96 ± 2.13	8.75 ± 2.27	25.96 ± 4.17	106.7 ± 5.9
PCB	2005	806 ± 90	17.0 ± 4.6	17.65 ± 5.19	20.85 ± 10.66	38.50 ± 15.22	77.4 ± 2.5
Center	2005	743 ± 32	10.7 ± 1.4	10.44 ± 1.75	1.39 ± 0.69	11.83 ± 2.30	101.1 ± 3.4
EBC	2005	1175 ± 57	7.6 ± 0.7	7.48 ± 0.91	0.54 ± 0.41	8.01 ± 0.89	115.4 ± 2.8
NaeP	2005	903 ± 30	7.1 ± 0.5	10.86 ± 1.49	1.11 ± 1.00	11.96 ± 2.20	113.3 ± 3.0
CresP	2005	944 ± 19	9.1 ± 0.6	10.43 ± 1.65	4.14 ± 2.44	14.57 ± 3.97	82.7 ± 1.6
LouP	2005	1822 ± 46	10.2 ± 1.6	22.56 ± 9.25	18.19 ± 14.79	40.75 ± 23.90	51.4 ± 5.3
RichP	2005	1261 ± 72	8.5 ± 0.9	10.50 ± 1.19	2.00 ± 1.00	12.50 ± 1.92	115.5 ± 3.7

Appendix E: Variability Charts of average monthly water chemistry at marsh and lake sites.

Monthly variations in depth and water chemistry at connected sites in the east and west sections of Delta Marsh and Lake Manitoba from 2002 to 2005. The distances of each site (km) via surface water flow from Lake Manitoba are noted in brackets following the site name.

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0.4
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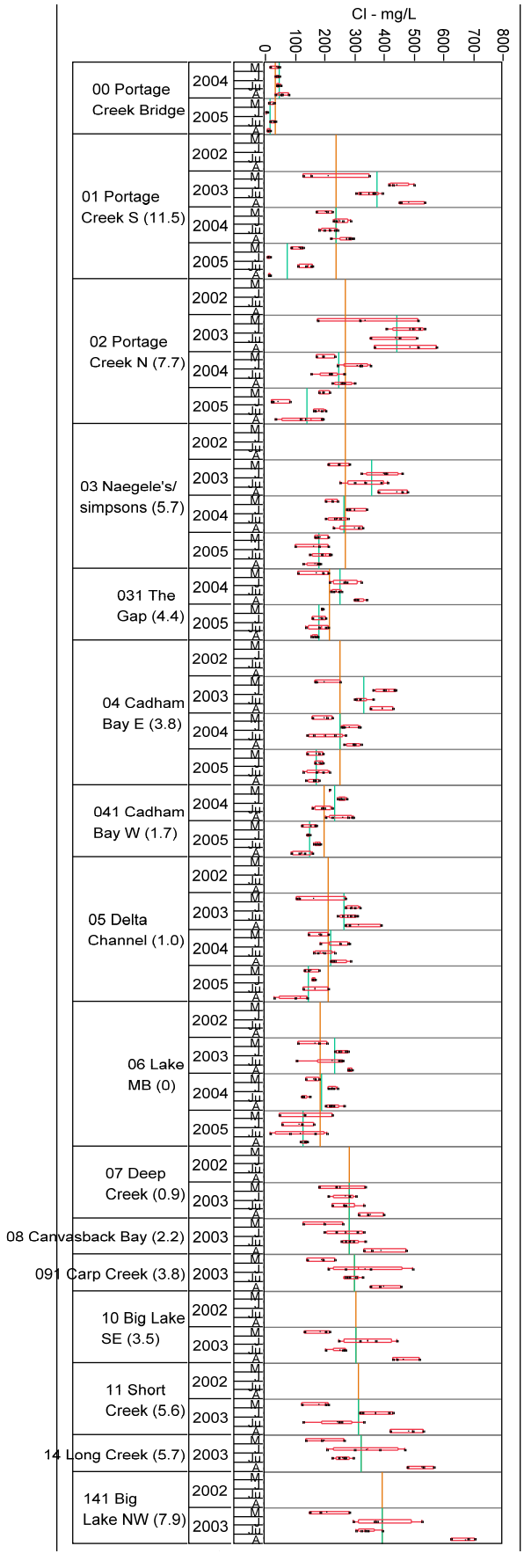
0.6
0.7
0.8
0.9
1
00 Portage Creek Bridge
2004
2005
01 Portage Creek S (11.5)
2002
2003
2004
2005
02 Portage Creek N (7.7)
2002
2003
2004
2005
03 Naegele's/ simpsons (5.7)
2002
2003
2004
2005
031 The Gap (4.4)
2004
2005
04 Cadham Bay E (3.8)
2002
2003
2004
2005
041 Cadham Bay W (1.7)
2004
2005
05 Delta Channel (1.0)
2002
2003
2004
2005
06 Lake MB (0)
2002
2003
2004
2005
07 Deep Creek (0.9)
2002
2003
08 Canvasback Bay (2.2)
2003
091 Carp Creek (3.8)
2003
10 Big Lake SE (3.5)
2002
2003
11 Short Creek (5.6)
2002
2003
14 Long Creek (5.7)
2003
141 Big Lake NW (7.9)
2002
2003


Cl - mg/L
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200
300
400
500
600
700
800
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2004
2005
01 Portage Creek S (11.5)
2002
2003
2004
2005
02 Portage Creek N (7.7)
2002
2003
2004
2005
03 Naegele's/ simpsons (5.7)
2002
2003
2004
2005
031 The Gap (4.4)
2004
2005
04 Cadham Bay E (3.8)
2002
2003
2004
2005
041 Cadham Bay W (1.7)
2004
2005
05 Delta Channel (1.0)
2002
2003
2004
2005
06 Lake MB (0)
2002
2003
2004
2005
07 Deep Creek (0.9)
2002
2003
08 Canvasback Bay (2.2)
2003
091 Carp Creek (3.8)
2003
10 Big Lake SE (3.5)
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2002
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14 Long Creek (5.7)
2003
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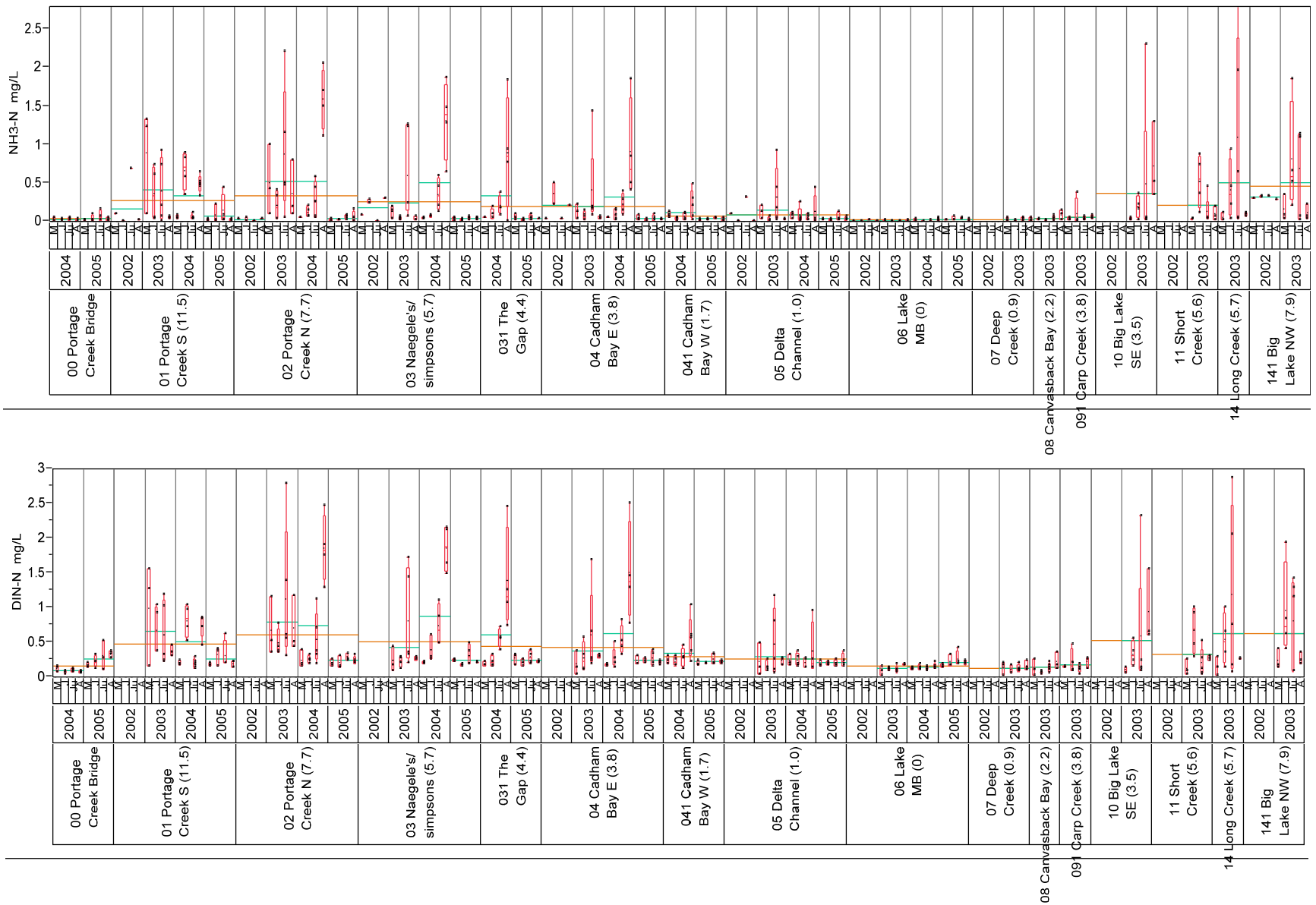
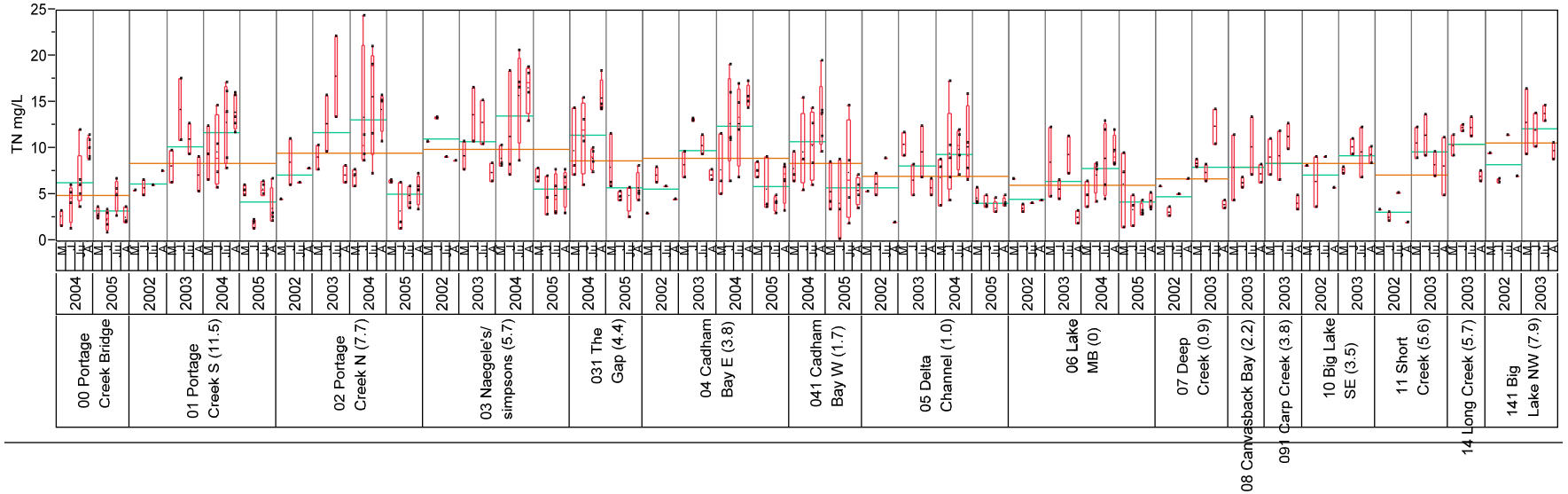
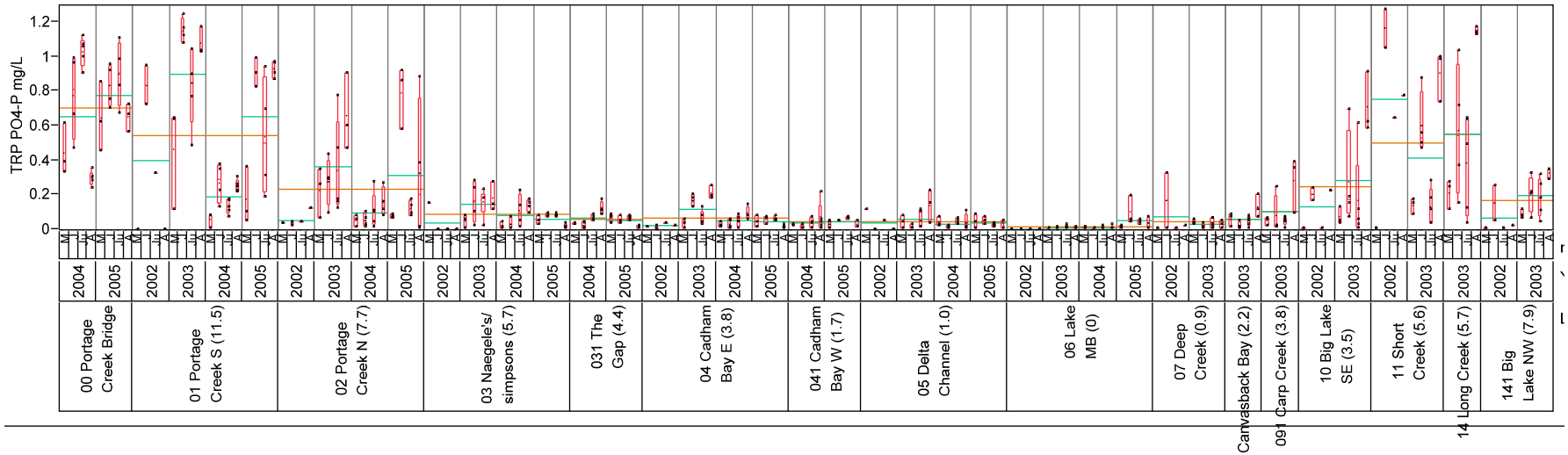
2002
2003


NH3-N mg/L
DIN-N mg/L
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
08 Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)


TN mg/L
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
08 Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)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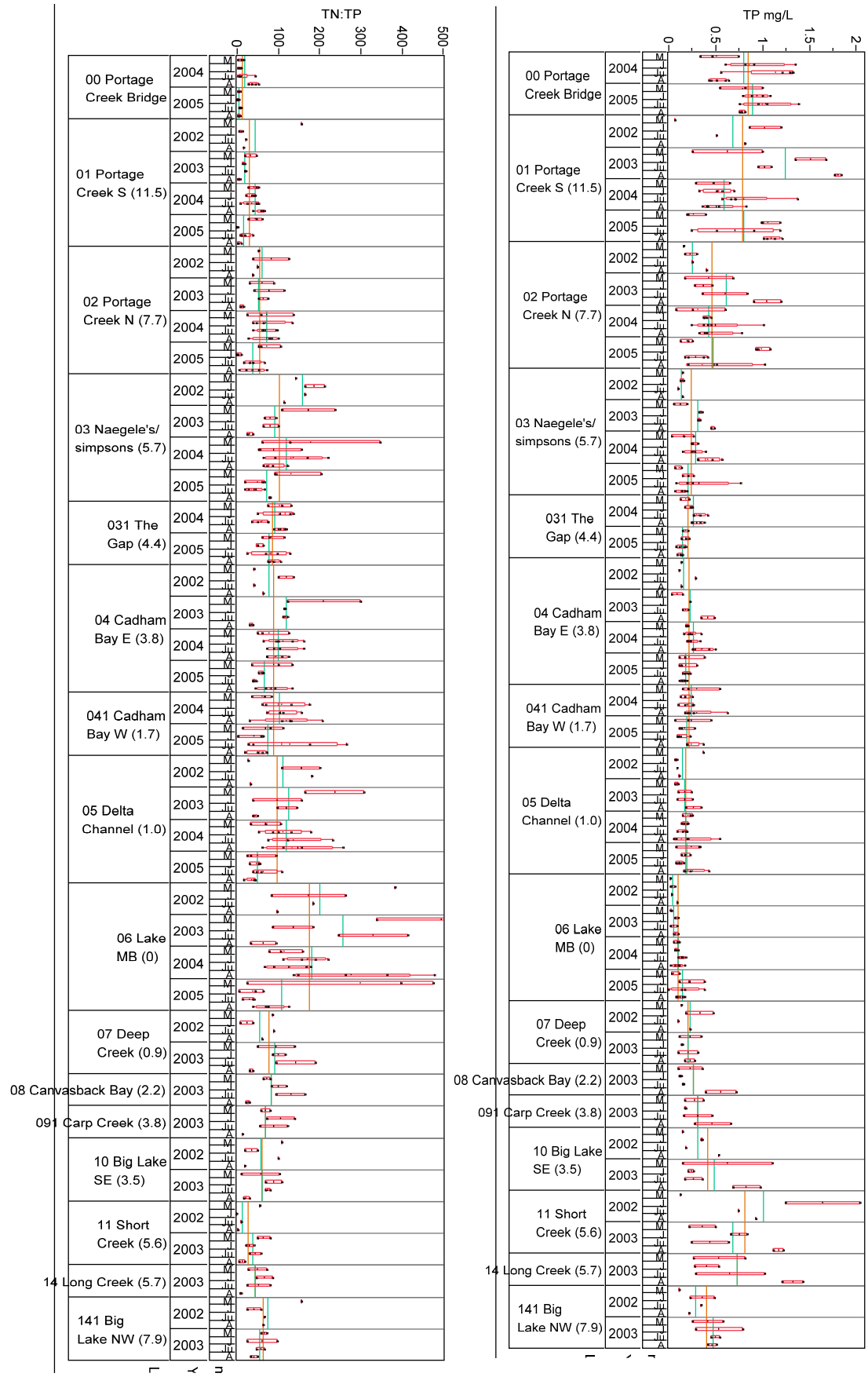
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)


TN:TP
0
100
200
300
400
500
TP mg/L
0
0.5
1
1.5
2
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
08 Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)
2002
2003
2004
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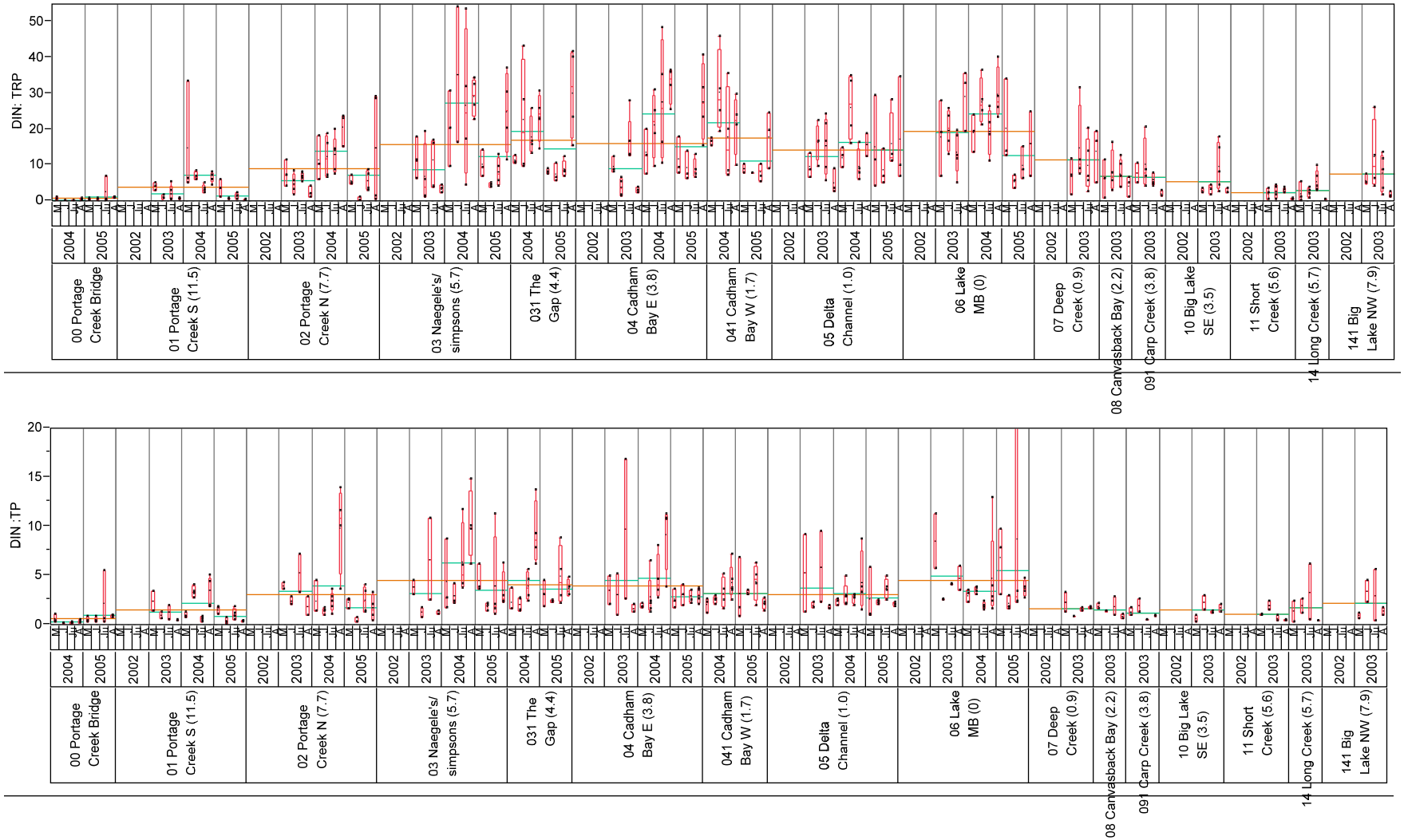
M
J
J
A
S
O
N
D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DIN:TP
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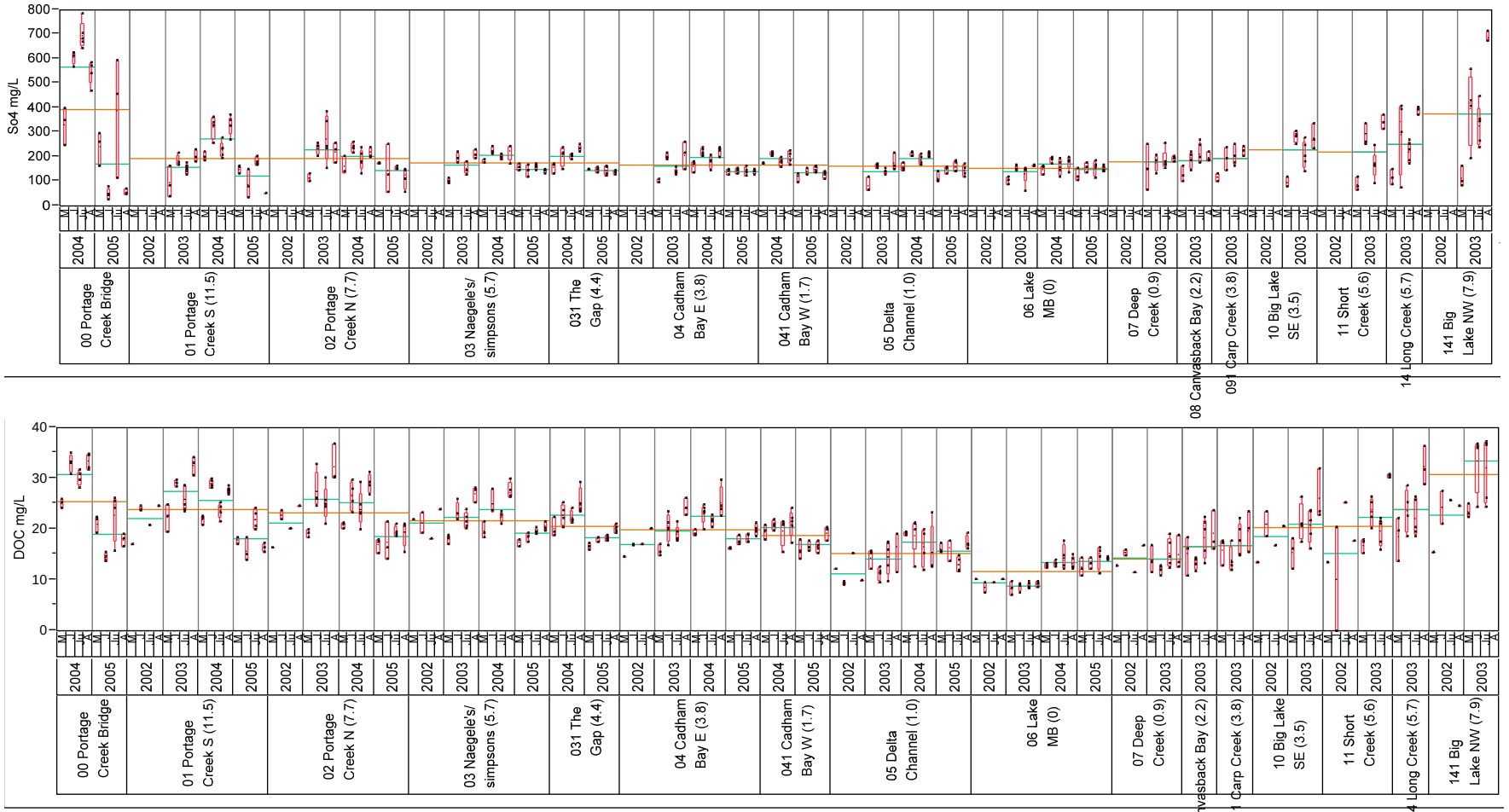
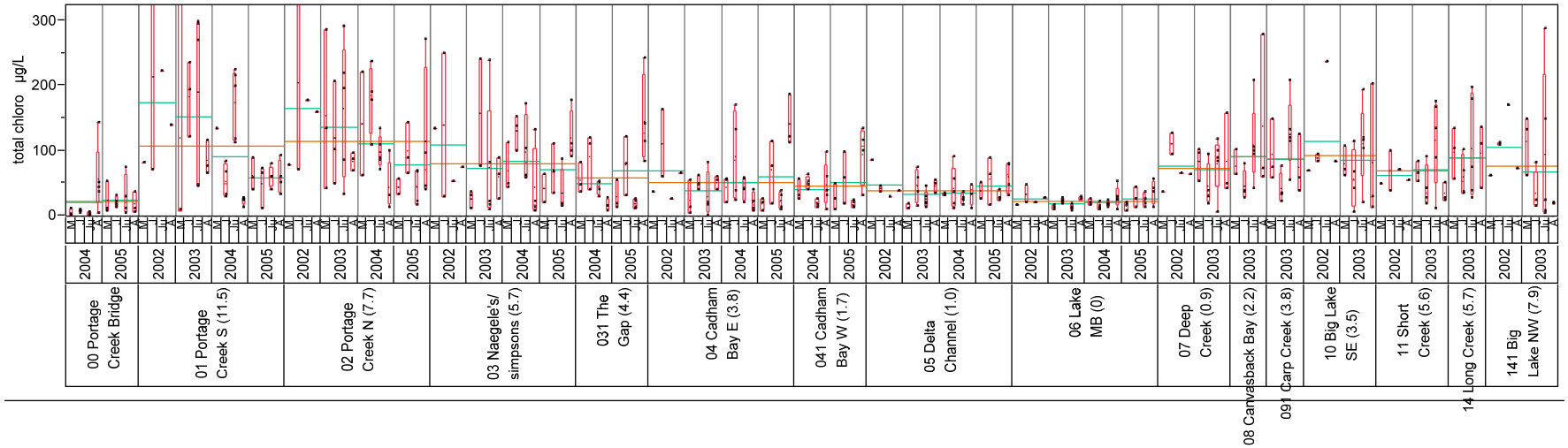
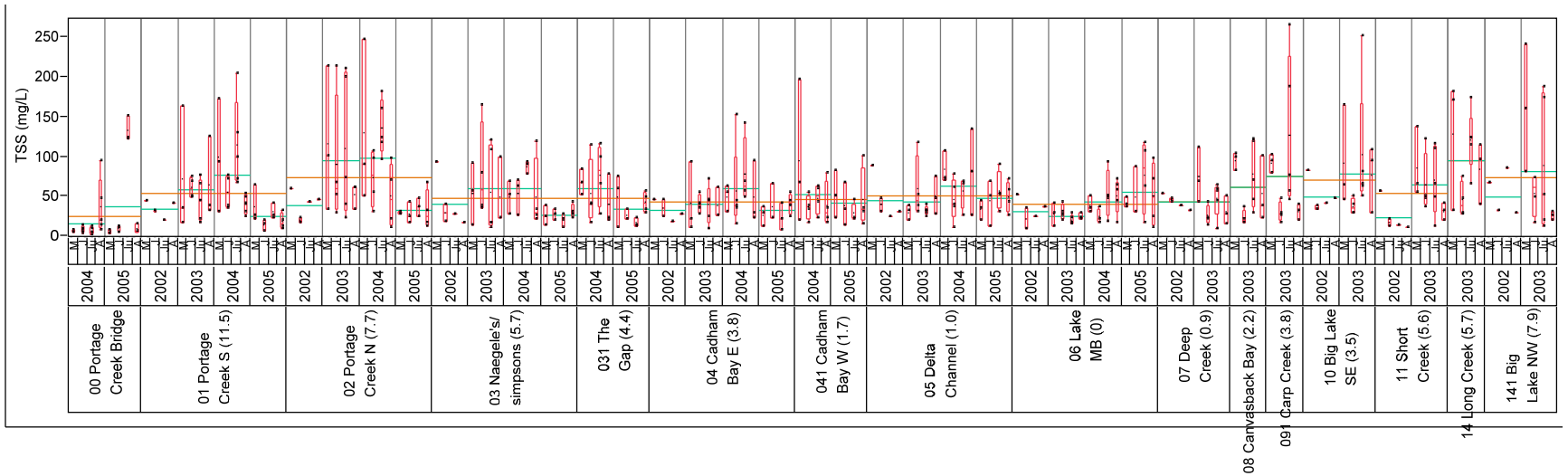
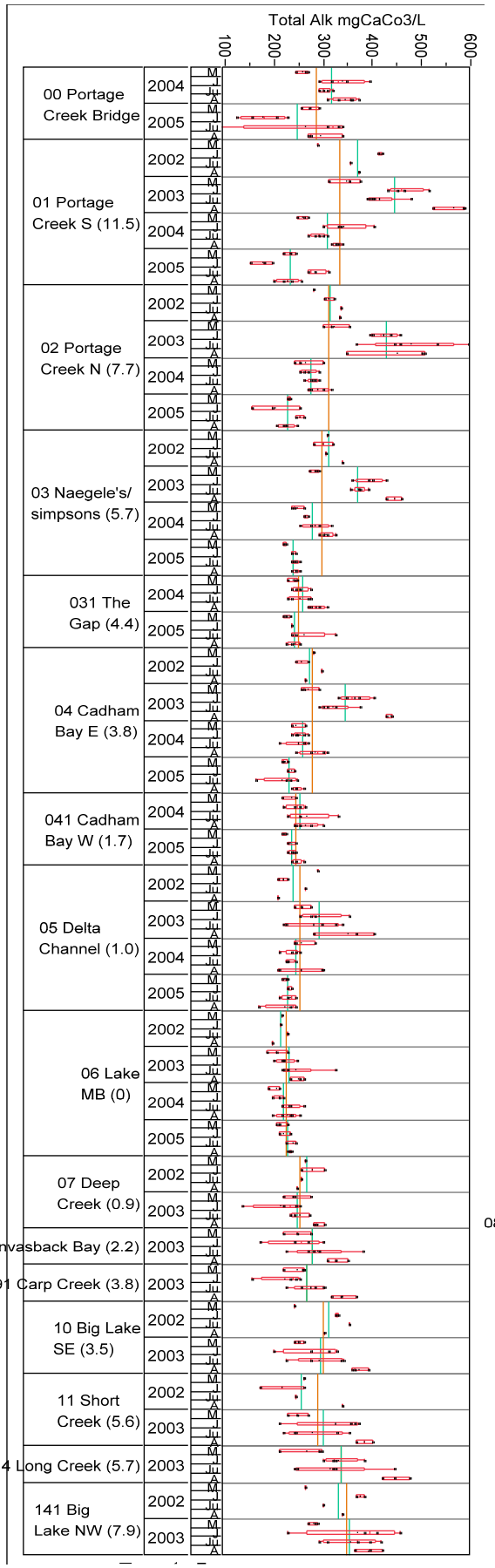
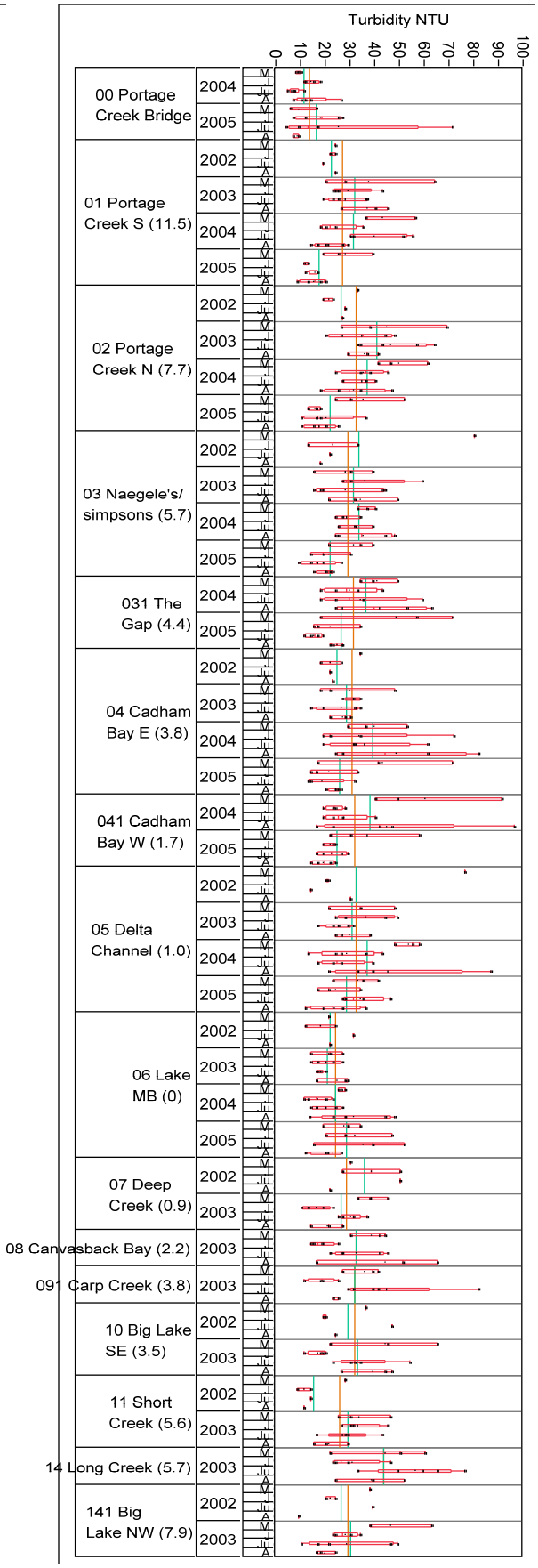
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
08 Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)


So4 mg/L
DOC mg/L
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
08 Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)


total chloro µg/L
300
200
100
0
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
08 Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)


TSS (mg/L)
250
200
150
100
50
0
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
08 Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)


Total Alk mgCaCo3/L
100
200
300
400
500
600
00 Portage Creek Bridge
2004
2005
01 Portage Creek S (11.5)
2002
2003
2004
2005
02 Portage Creek N (7.7)
2002
2003
2004
2005
03 Naegele's/ simpsons (5.7)
2002
2003
2004
2005
031 The Gap (4.4)
2004
2005
04 Cadham Bay E (3.8)
2002
2003
2004
2005
041 Cadham Bay W (1.7)
2004
2005
05 Delta Channel (1.0)
2002
2003
2004
2005
06 Lake MB (0)
2002
2003
2004
2005
07 Deep Creek (0.9)
2002
2003
08 Canvasback Bay (2.2)
2003
091 Carp Creek (3.8)
2003
10 Big Lake SE (3.5)
2002
2003
11 Short Creek (5.6)
2002
2003
14 Long Creek (5.7)
2003
141 Big Lake NW (7.9)
2002
2003

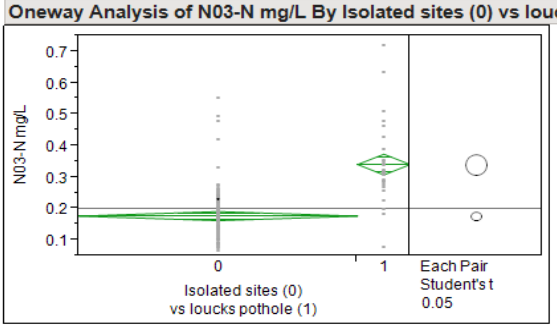

Turbidity NTU
0
10
20
30
40
50
60
70
80
90
100
00 Portage Creek Bridge
2004
2005
01 Portage Creek S (11.5)
2002
2003
2004
2005
02 Portage Creek N (7.7)
2002
2003
2004
2005
03 Naegele's/ simpsons (5.7)
2002
2003
2004
2005
031 The Gap (4.4)
2004
2005
04 Cadham Bay E (3.8)
2002
2003
2004
2005
041 Cadham Bay W (1.7)
2004
2005
05 Delta Channel (1.0)
2002
2003
2004
2005
06 Lake MB (0)
2002
2003
2004
2005
07 Deep Creek (0.9)
2002
2003
08 Canvasback Bay (2.2)
2003
091 Carp Creek (3.8)
2003
10 Big Lake SE (3.5)
2002
2003
11 Short Creek (5.6)
2002
2003
14 Long Creek (5.7)
2003
141 Big Lake NW (7.9)
2002
2003

Specific conductance (Us/cm2)
3500
3000
2500
2000
1500
1000
500
0
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)

pH
10
9.5
9
8.5
8
7.5
7
6.5
00 Portage Creek Bridge
01 Portage Creek S (11.5)
02 Portage Creek N (7.7)
03 Naegele's/ simpsons (5.7)
031 The Gap (4.4)
04 Cadham Bay E (3.8)
041 Cadham Bay W (1.7)
05 Delta Channel (1.0)
06 Lake MB (0)
07 Deep Creek (0.9)
Canvasback Bay (2.2)
091 Carp Creek (3.8)
10 Big Lake SE (3.5)
11 Short Creek (5.6)
14 Long Creek (5.7)
141 Big Lake NW (7.9)

Appendix F: ANOVAs on water chemistry parameters comparing Loucks Pothole to other isolated sites, in 2004 and 2005.

Oneway Analysis of N03-N mg/L By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

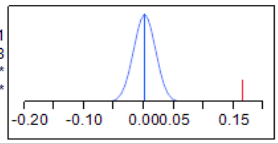
Summary of Fit

Rsquare	0.32639
Adj Rsquare	0.322381
Root Mean Square Error	0.086577
Mean of Response	0.203571
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	0.163906	t Ratio	9.022341
Std Err Dif	0.018167	DF	168
Upper CL Dif	0.199770	Prob > |t|	<.0001*
Lower CL Dif	0.128042	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	0.6101554	0.610155	81.4026	<.0001*
Error	168	1.2592482	0.007496		
C. Total	169	1.8694037			

Means for Oneway Anova

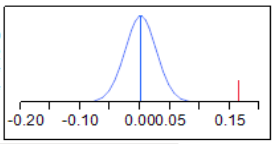
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	0.177538	0.00724	0.16325	0.19183
1	27	0.341444	0.01666	0.30855	0.37434

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	0.163906	t Ratio	6.224899
Std Err Dif	0.026331	DF	29.20882
Upper CL Dif	0.217742	Prob > |t|	<.0001*
Lower CL Dif	0.110070	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

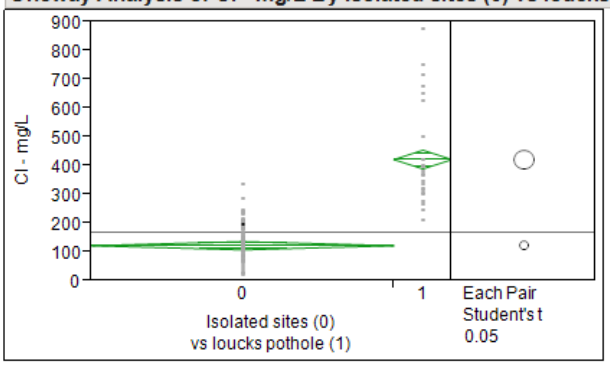
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-0.04652	0.12804
0	0.12804	-0.02021

Positive values show pairs of means that are significantly different.

Oneway Analysis of Cl - mg/L By Isolated sites (0) vs loucks pothole (1)



Missing Rows 3

Oneway Anova

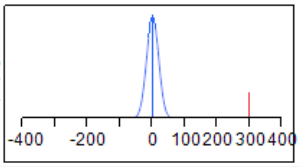
Summary of Fit

Rsquare	0.624089
Adj Rsquare	0.62181
Root Mean Square Error	86.09999
Mean of Response	171.9532
Observations (or Sum Wgts)	167

t Test

1-0
Assuming equal variances

Difference	299.528	t Ratio	16.55093
Std Err Dif	18.097	DF	165
Upper CL Dif	335.260	Prob > |t|	<.0001*
Lower CL Dif	263.796	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	2030723.4	2030723	273.9332	<.0001*
Error	165	1223179.2	7413		
C. Total	166	3253902.6			

Means for Oneway Anova

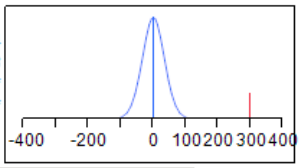
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	140	123.526	7.277	109.16	137.89
1	27	423.055	16.570	390.34	455.77

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	299.528	t Ratio	8.870774
Std Err Dif	33.766	DF	27.0573
Upper CL Dif	368.803	Prob > |t|	<.0001*
Lower CL Dif	230.254	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97445	0.05

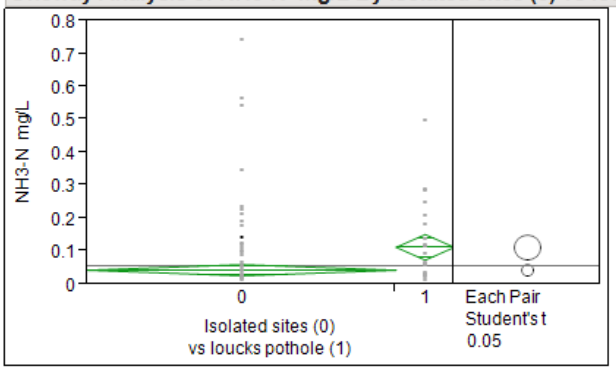
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-46.27	263.80
0	263.80	-20.32

Positive values show pairs of means that are significantly different.

Oneway Analysis of NH3-N mg/L By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

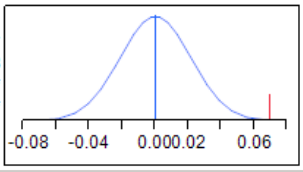
Summary of Fit

Rsquare	0.058699
Adj Rsquare	0.053096
Root Mean Square Error	0.102342
Mean of Response	0.055606
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	0.069508	t Ratio	3.236723
Std Err Dif	0.021475	DF	168
Upper CL Dif	0.111903	Prob > |t|	0.0015*
Lower CL Dif	0.027113	Prob > t	0.0007*
Confidence	0.95	Prob < t	0.9993



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	0.1097276	0.109728	10.4764	0.0015*
Error	168	1.7596010	0.010474		
C. Total	169	1.8693286			

Means for Oneway Anova

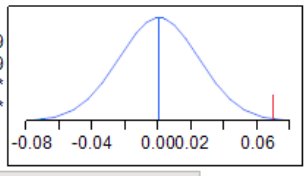
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	0.044566	0.00856	0.02767	0.06146
1	27	0.114074	0.01970	0.07519	0.15296

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	0.069508	t Ratio	2.838839
Std Err Dif	0.024485	DF	33.01479
Upper CL Dif	0.119321	Prob > |t|	0.0077*
Lower CL Dif	0.019694	Prob > t	0.0038*
Confidence	0.95	Prob < t	0.9962



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

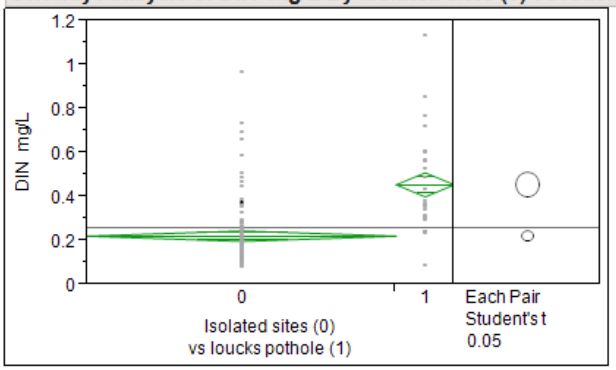
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-0.05499	0.02711
0	0.02711	-0.02389

Positive values show pairs of means that are significantly different.

Oneway Analysis of DIN mg/L By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

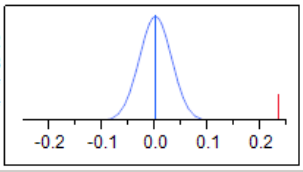
Summary of Fit

Rsquare	0.261173
Adj Rsquare	0.256775
Root Mean Square Error	0.144483
Mean of Response	0.258953
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	0.233635	t Ratio	7.706322
Std Err Dif	0.030317	DF	168
Upper CL Dif	0.293487	Prob > |t|	<.0001*
Lower CL Dif	0.173783	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	1.2397323	1.23973	59.3874	<.0001*
Error	168	3.5070574	0.02088		
C. Total	169	4.7467896			

Means for Oneway Anova

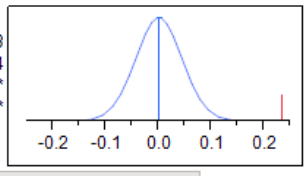
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	0.221846	0.01208	0.19799	0.24570
1	27	0.455481	0.02781	0.40059	0.51038

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	0.233635	t Ratio	5.292893
Std Err Dif	0.044141	DF	29.15774
Upper CL Dif	0.323893	Prob > |t|	<.0001*
Lower CL Dif	0.143377	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

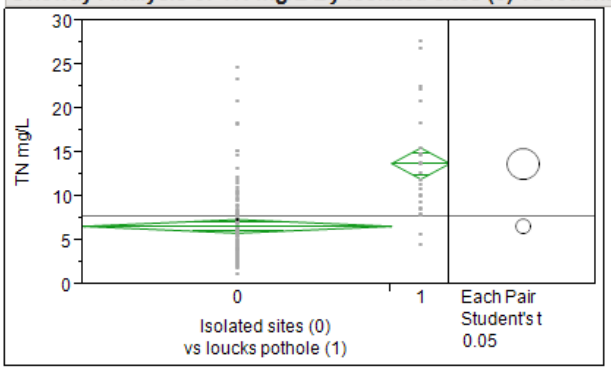
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-0.07763	0.17378
0	0.17378	-0.03373

Positive values show pairs of means that are significantly different.

Oneway Analysis of TN mg/L By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

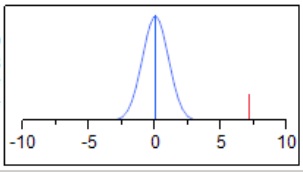
Summary of Fit

Rsquare	0.239517
Adj Rsquare	0.23499
Root Mean Square Error	4.679819
Mean of Response	7.7955
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	7.14302	t Ratio	7.274079
Std Err Dif	0.98198	DF	168
Upper CL Dif	9.08163	Prob > |t|	<.0001*
Lower CL Dif	5.20440	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	1158.8151	1158.82	52.9122	<.0001*
Error	168	3679.3188	21.90		
C. Total	169	4838.1339			

Means for Oneway Anova

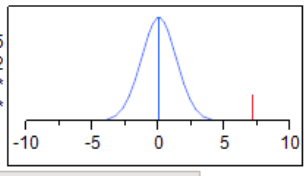
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	6.6610	0.39135	5.888	7.434
1	27	13.8040	0.90063	12.026	15.582

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	7.14302	t Ratio	5.343795
Std Err Dif	1.33669	DF	29.98132
Upper CL Dif	9.87298	Prob > |t|	<.0001*
Lower CL Dif	4.41305	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

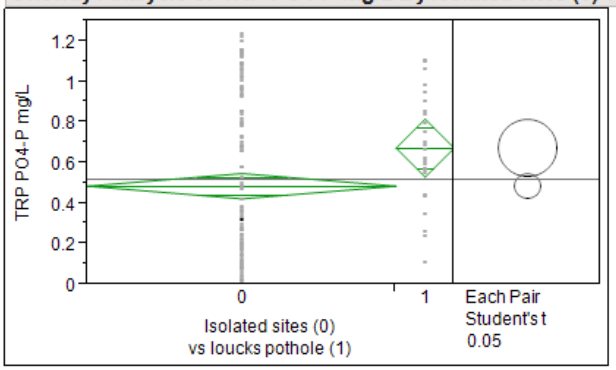
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-2.5145	5.2044
0	5.2044	-1.0926

Positive values show pairs of means that are significantly different.

Oneway Analysis of TRP PO4-P mg/L By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

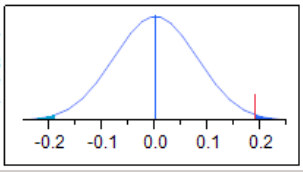
Summary of Fit

Rsquare	0.032518
Adj Rsquare	0.026759
Root Mean Square Error	0.376447
Mean of Response	0.516553
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	0.187703	t Ratio	2.376257
Std Err Dif	0.078991	DF	168
Upper CL Dif	0.343646	Prob > |t|	0.0186*
Lower CL Dif	0.031760	Prob > t	0.0093*
Confidence	0.95	Prob < t	0.9907



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	0.800192	0.800192	5.6466	0.0186*
Error	168	23.807654	0.141712		
C. Total	169	24.607846			

Means for Oneway Anova

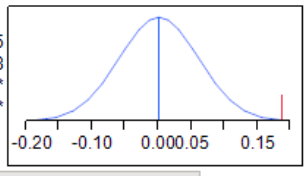
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	0.486741	0.03148	0.42459	0.54889
1	27	0.674444	0.07245	0.53142	0.81747

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	0.187703	t Ratio	3.099985
Std Err Dif	0.060550	DF	50.74228
Upper CL Dif	0.309277	Prob > |t|	0.0032*
Lower CL Dif	0.066130	Prob > t	0.0016*
Confidence	0.95	Prob < t	0.9984



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

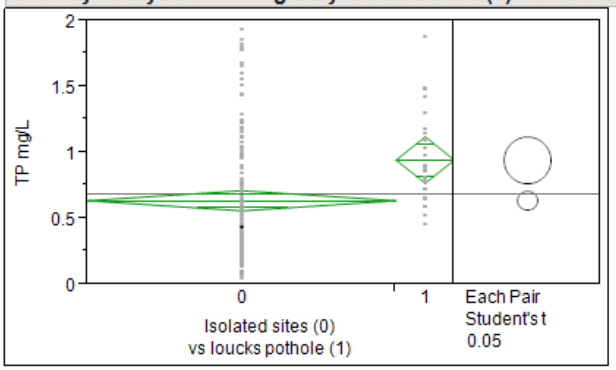
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-0.20227	0.03176
0	0.03176	-0.08789

Positive values show pairs of means that are significantly different.

Oneway Analysis of TP mg/L By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

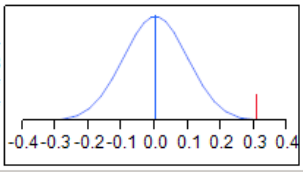
Summary of Fit

Rsquare	0.056387
Adj Rsquare	0.05077
Root Mean Square Error	0.464043
Mean of Response	0.687335
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	0.308516	t Ratio	3.16844
Std Err Dif	0.097372	DF	168
Upper CL Dif	0.500746	Prob > |t|	0.0018*
Lower CL Dif	0.116287	Prob > t	0.0009*
Confidence	0.95	Prob < t	0.9991



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	2.161757	2.16176	10.0390	0.0018*
Error	168	36.176373	0.21534		
C. Total	169	38.338130			

Means for Oneway Anova

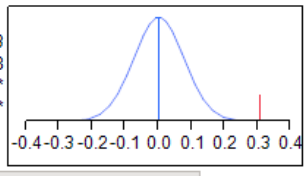
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	0.638336	0.03881	0.56173	0.7149
1	27	0.946852	0.08931	0.77055	1.1232

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	0.308516	t Ratio	4.030583
Std Err Dif	0.076544	DF	48.69948
Upper CL Dif	0.462361	Prob > |t|	0.0002*
Lower CL Dif	0.154672	Prob > t	<.0001*
Confidence	0.95	Prob < t	0.9999



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

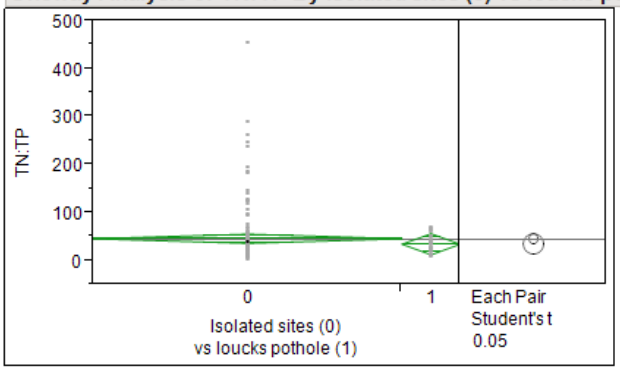
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-0.24933	0.11629
0	0.11629	-0.10834

Positive values show pairs of means that are significantly different.

Oneway Analysis of TN:TP By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

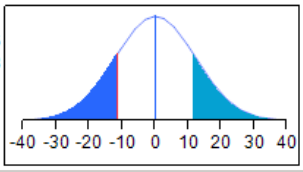
Summary of Fit

Rsquare	0.005264
Adj Rsquare	-0.00066
Root Mean Square Error	58.12641
Mean of Response	44.51751
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	-11.500	t Ratio	-0.94286
Std Err Dif	12.197	DF	168
Upper CL Dif	12.579	Prob > |t|	0.3471
Lower CL Dif	-35.579	Prob > t	0.8264
Confidence	0.95	Prob < t	0.1736



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	3003.62	3003.62	0.8890	0.3471
Error	168	567618.12	3378.68		
C. Total	169	570621.74			

Means for Oneway Anova

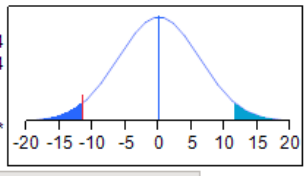
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	46.3440	4.861	36.748	55.940
1	27	34.8440	11.186	12.760	56.928

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	-11.500	t Ratio	-1.85624
Std Err Dif	6.195	DF	149.644
Upper CL Dif	0.742	Prob > |t|	0.0654
Lower CL Dif	-23.742	Prob > t	0.9673
Confidence	0.95	Prob < t	0.0327*



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

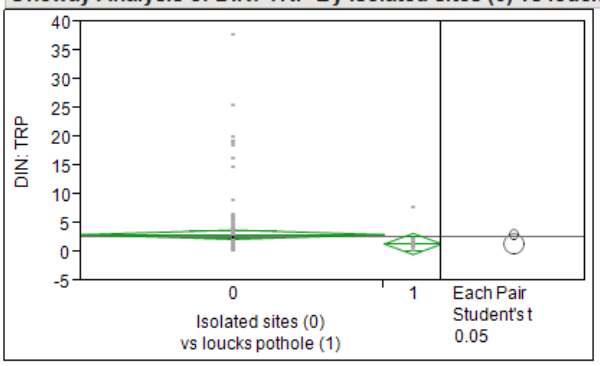
LSD Threshold Matrix

Abs(Dif)-LSD

	0	1
0	-13.571	-12.579
1	-12.579	-31.232

Positive values show pairs of means that are significantly different.

Oneway Analysis of DIN: TRP By Isolated sites (0) vs loucks pothole (1)



Missing Rows 1

Oneway Anova

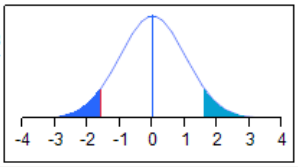
Summary of Fit

Rsquare	0.014927
Adj Rsquare	0.009028
Root Mean Square Error	4.85177
Mean of Response	2.835799
Observations (or Sum Wgts)	169

t Test

1-0
Assuming equal variances

Difference	-1.6204	t Ratio	-1.59078
Std Err Dif	1.0186	DF	167
Upper CL Dif	0.3906	Prob > |t|	0.1135
Lower CL Dif	-3.6315	Prob > t	0.9432
Confidence	0.95	Prob < t	0.0568



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	59.5693	59.5693	2.5306	0.1135
Error	167	3931.1255	23.5397		
C. Total	168	3990.6949			

Means for Oneway Anova

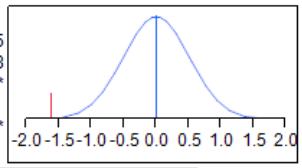
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	142	3.09468	0.40715	2.291	3.8985
1	27	1.47426	0.93372	-0.369	3.3177

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	-1.6204	t Ratio	-3.17835
Std Err Dif	0.5098	DF	155.6648
Upper CL Dif	-0.6133	Prob > |t|	0.0018*
Lower CL Dif	-2.6275	Prob > t	0.9991
Confidence	0.95	Prob < t	0.0009*



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97427	0.05

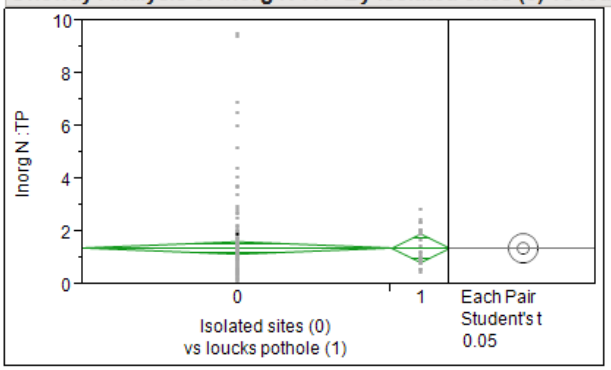
LSD Threshold Matrix

Abs(Dif)-LSD

	0	1
0	-1.1368	-0.3906
1	-0.3906	-2.6070

Positive values show pairs of means that are significantly different.

Oneway Analysis of Inorg N :TP By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

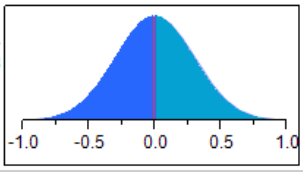
Summary of Fit

Rsquare	1.328e-5
Adj Rsquare	-0.00594
Root Mean Square Error	1.422886
Mean of Response	1.399012
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	-0.01410	t Ratio	-0.04724
Std Err Dif	0.29857	DF	168
Upper CL Dif	0.57533	Prob > |t|	0.9624
Lower CL Dif	-0.60353	Prob > t	0.5188
Confidence	0.95	Prob < t	0.4812



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	0.00452	0.00452	0.0022	0.9624
Error	168	340.13348	2.02460		
C. Total	169	340.13799			

Means for Oneway Anova

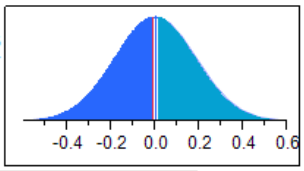
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	1.40125	0.11899	1.1663	1.6362
1	27	1.38715	0.27383	0.8465	1.9277

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	-0.01410	t Ratio	-0.07636
Std Err Dif	0.18471	DF	81.5247
Upper CL Dif	0.35337	Prob > |t|	0.9393
Lower CL Dif	-0.38158	Prob > t	0.5303
Confidence	0.95	Prob < t	0.4697



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

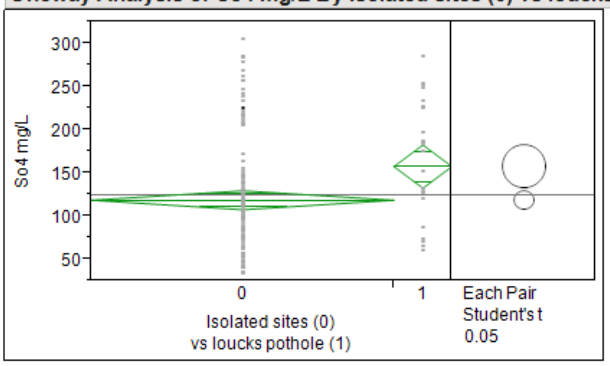
LSD Threshold Matrix

Abs(Dif)-LSD

	0	1
0	-0.33220	-0.57533
1	-0.57533	-0.76452

Positive values show pairs of means that are significantly different.

Oneway Analysis of So4 mg/L By Isolated sites (0) vs loucks pothole (1)



Missing Rows 3

Oneway Anova

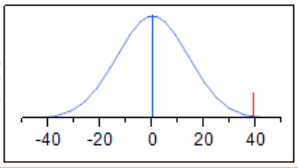
Summary of Fit

Rsquare	0.045844
Adj Rsquare	0.040061
Root Mean Square Error	65.87938
Mean of Response	125.1895
Observations (or Sum Wgts)	167

t Test

1-0
Assuming equal variances

Difference	38.9883	t Ratio	2.81561
Std Err Dif	13.8472	DF	165
Upper CL Dif	66.3288	Prob > |t|	0.0055*
Lower CL Dif	11.6478	Prob > t	0.0027*
Confidence	0.95	Prob < t	0.9973



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	34406.78	34406.8	7.9277	0.0055*
Error	165	716115.37	4340.1		
C. Total	166	750522.15			

Means for Oneway Anova

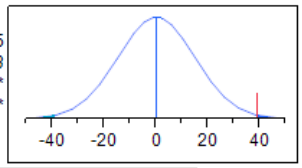
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	140	118.886	5.568	107.89	129.88
1	27	157.874	12.678	132.84	182.91

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	38.9883	t Ratio	2.608405
Std Err Dif	14.9472	DF	34.47183
Upper CL Dif	69.3493	Prob > |t|	0.0134*
Lower CL Dif	8.6273	Prob > t	0.0067*
Confidence	0.95	Prob < t	0.9933



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97445	0.05

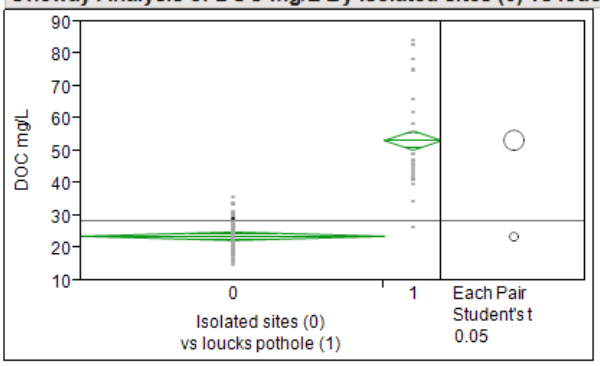
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-35.402	11.648
0	11.648	-15.547

Positive values show pairs of means that are significantly different.

Oneway Analysis of DOC mg/L By Isolated sites (0) vs loucks pothole (1)



Missing Rows 1

Oneway Anova

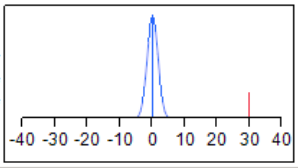
Summary of Fit

Rsquare	0.660712
Adj Rsquare	0.65868
Root Mean Square Error	7.813674
Mean of Response	28.63379
Observations (or Sum Wgts)	169

t Test

1-0
Assuming equal variances

Difference	29.5837	t Ratio	18.03351
Std Err Dif	1.6405	DF	167
Upper CL Dif	32.8225	Prob > |t|	<.0001*
Lower CL Dif	26.3450	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	19855.045	19855.0	325.2073	<.0001*
Error	167	10195.934	61.1		
C. Total	168	30050.979			

Means for Oneway Anova

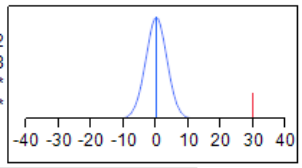
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	142	23.9074	0.6557	22.613	25.202
1	27	53.4911	1.5037	50.522	56.460

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	29.5837	t Ratio	9.251632
Std Err Dif	3.1977	DF	26.81178
Upper CL Dif	36.1470	Prob > |t|	<.0001*
Lower CL Dif	23.0205	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97427	0.05

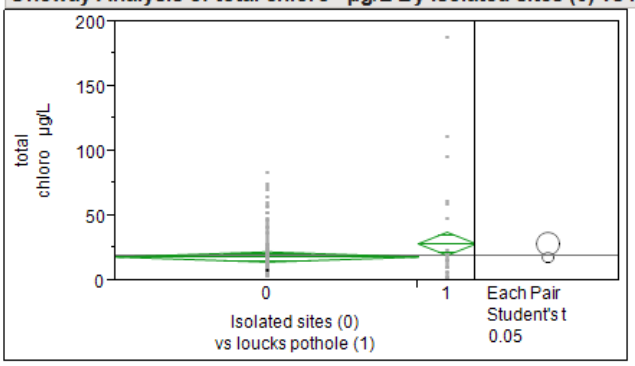
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-4.199	26.345
0	26.345	-1.831

Positive values show pairs of means that are significantly different.

Oneway Analysis of total chloro µg/L By Isolated sites (0) vs loucks pothole (1)



Missing Rows 2

Oneway Anova

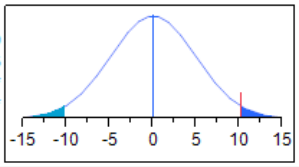
Summary of Fit

Rsquare	0.024998
Adj Rsquare	0.019125
Root Mean Square Error	23.55481
Mean of Response	20.37571
Observations (or Sum Wgts)	168

t Test

1-0
Assuming equal variances

Difference	10.2082	t Ratio	2.063039
Std Err Dif	4.9482	DF	166
Upper CL Dif	19.9776	Prob > |t|	0.0407*
Lower CL Dif	0.4388	Prob > t	0.0203*
Confidence	0.95	Prob < t	0.9797



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	2361.425	2361.43	4.2561	0.0407*
Error	166	92101.647	554.83		
C. Total	167	94463.073			

Means for Oneway Anova

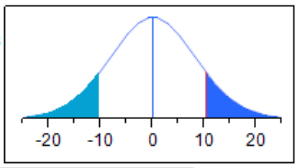
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	141	18.7351	1.9837	14.819	22.652
1	27	28.9433	4.5331	19.993	37.893

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	10.208	t Ratio	1.231808
Std Err Dif	8.287	DF	27.83131
Upper CL Dif	27.188	Prob > |t|	0.2283
Lower CL Dif	-6.772	Prob > t	0.1142
Confidence	0.95	Prob < t	0.8858



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97436	0.05

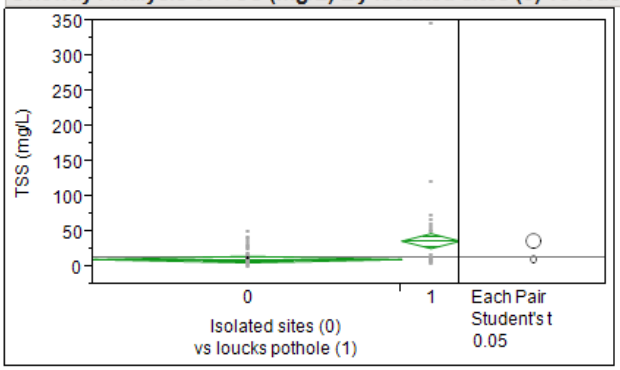
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-12.657	0.439
0	0.439	-5.539

Positive values show pairs of means that are significantly different.

Oneway Analysis of TSS (mg/L) By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

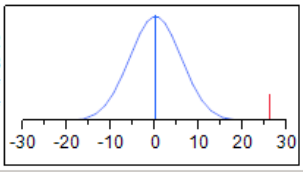
Summary of Fit

Rsquare	0.105448
Adj Rsquare	0.100124
Root Mean Square Error	27.97603
Mean of Response	15.22176
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	26.1236	t Ratio	4.450122
Std Err Dif	5.8703	DF	168
Upper CL Dif	37.7126	Prob > |t|	<.0001*
Lower CL Dif	14.5345	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	15499.44	15499.4	19.8036	<.0001*
Error	168	131486.55	782.7		
C. Total	169	146985.99			

Means for Oneway Anova

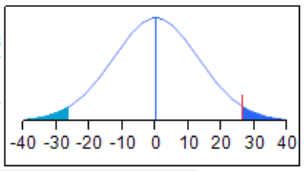
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	11.0727	2.3395	6.454	15.691
1	27	37.1963	5.3840	26.567	47.825

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	26.124	t Ratio	1.990618
Std Err Dif	13.123	DF	26.16377
Upper CL Dif	53.091	Prob > |t|	0.0571
Lower CL Dif	-0.844	Prob > t	0.0285*
Confidence	0.95	Prob < t	0.9715



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

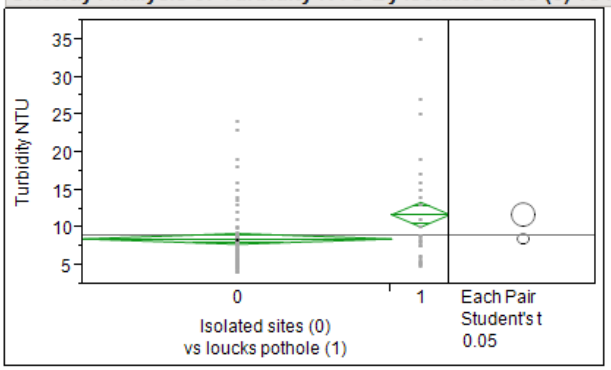
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-15.032	14.535
0	14.535	-6.532

Positive values show pairs of means that are significantly different.

Oneway Analysis of Turbidity NTU By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

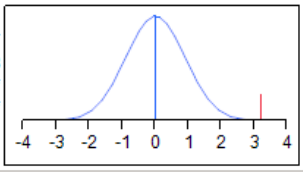
Summary of Fit

Rsquare	0.069828
Adj Rsquare	0.064292
Root Mean Square Error	4.304073
Mean of Response	9.120588
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	3.20733	t Ratio	3.551317
Std Err Dif	0.90314	DF	168
Upper CL Dif	4.99029	Prob > |t|	0.0005*
Lower CL Dif	1.42437	Prob > t	0.0002*
Confidence	0.95	Prob < t	0.9998



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	233.6351	233.635	12.6118	0.0005*
Error	168	3112.2078	18.525		
C. Total	169	3345.8429			

Means for Oneway Anova

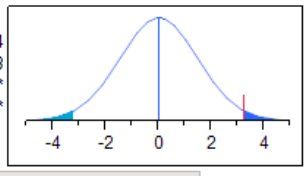
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	8.6112	0.35992	7.901	9.322
1	27	11.8185	0.82832	10.183	13.454

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	3.20733	t Ratio	2.158564
Std Err Dif	1.48586	DF	27.97663
Upper CL Dif	6.25110	Prob > |t|	0.0396*
Lower CL Dif	0.16356	Prob > t	0.0198*
Confidence	0.95	Prob < t	0.9802



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

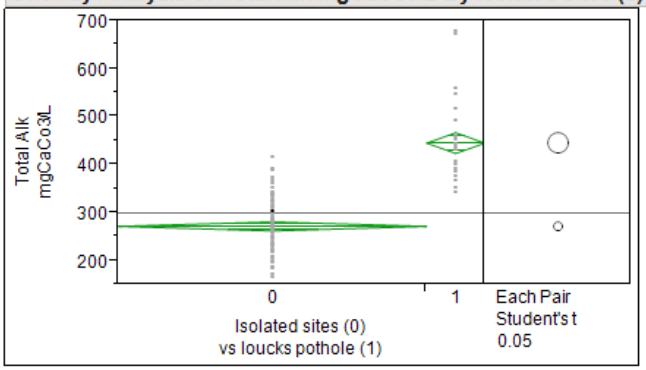
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-2.3126	1.4244
0	1.4244	-1.0049

Positive values show pairs of means that are significantly different.

Oneway Analysis of Total Alk mgCaCo3/L By Isolated sites (0) vs loucks pothole (1)



Oneway Anova

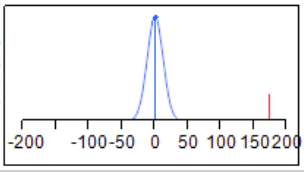
Summary of Fit

Rsquare	0.556552
Adj Rsquare	0.553913
Root Mean Square Error	57.01557
Mean of Response	299.6471
Observations (or Sum Wgts)	170

t Test

1-0
Assuming equal variances

Difference	173.722	t Ratio	14.52066
Std Err Dif	11.964	DF	168
Upper CL Dif	197.341	Prob > |t|	<.0001*
Lower CL Dif	150.103	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	685424.6	685425	210.8496	<.0001*
Error	168	546130.2	3251		
C. Total	169	1231554.8			

Means for Oneway Anova

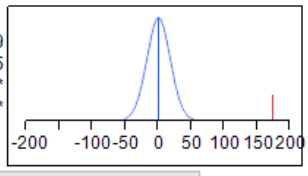
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	272.056	4.768	262.64	281.47
1	27	445.778	10.973	424.12	467.44

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	173.722	t Ratio	9.782169
Std Err Dif	17.759	DF	28.94615
Upper CL Dif	210.046	Prob > |t|	<.0001*
Lower CL Dif	137.398	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97419	0.05

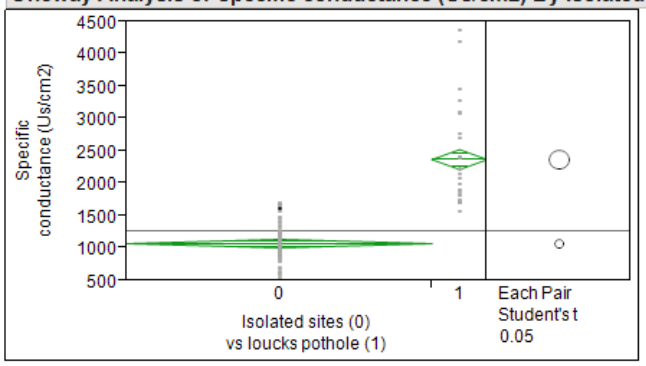
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-30.63	150.10
0	150.10	-13.31

Positive values show pairs of means that are significantly different.

Oneway Analysis of Specific conductance (Us/cm2) By Isolated sites (0) vs loucks pothole (1)



Missing Rows 1

Oneway Anova

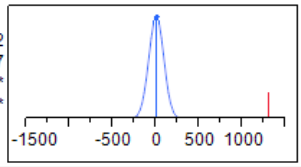
Summary of Fit

Rsquare	0.583553
Adj Rsquare	0.581059
Root Mean Square Error	397.8671
Mean of Response	1281.29
Observations (or Sum Wgts)	169

t Test

1-0
Assuming equal variances

Difference	1297.61	t Ratio	15.29742
Std Err Dif	84.83	DF	167
Upper CL Dif	1465.08	Prob > |t|	<.0001*
Lower CL Dif	1130.14	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	37043525	37043525	234.0110	<.0001*
Error	167	26435807	158298.25		
C. Total	168	63479333			

Means for Oneway Anova

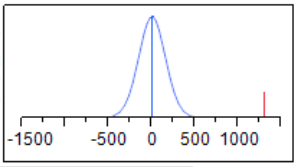
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	143	1081.66	33.271	1016.0	1147.3
1	26	2379.27	78.028	2225.2	2533.3

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	1297.61	t Ratio	8.428517
Std Err Dif	153.95	DF	26.22438
Upper CL Dif	1613.94	Prob > |t|	<.0001*
Lower CL Dif	981.28	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97427	0.05

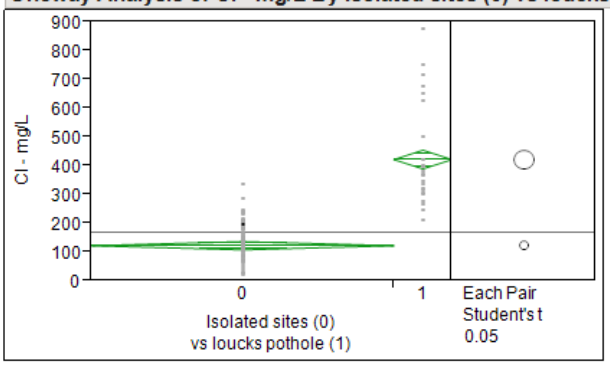
LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-217.9	1130.1
0	1130.1	-92.9

Positive values show pairs of means that are significantly different.

Oneway Analysis of Cl - mg/L By Isolated sites (0) vs loucks pothole (1)



Missing Rows 3

Oneway Anova

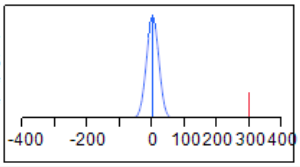
Summary of Fit

Rsquare	0.624089
Adj Rsquare	0.62181
Root Mean Square Error	86.09999
Mean of Response	171.9532
Observations (or Sum Wgts)	167

t Test

1-0
Assuming equal variances

Difference	299.528	t Ratio	16.55093
Std Err Dif	18.097	DF	165
Upper CL Dif	335.260	Prob > |t|	<.0001*
Lower CL Dif	263.796	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Isolated sites (0) vs loucks pothole (1)	1	2030723.4	2030723	273.9332	<.0001*
Error	165	1223179.2	7413		
C. Total	166	3253902.6			

Means for Oneway Anova

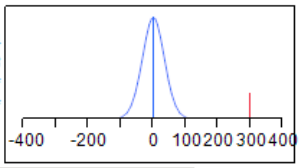
Level	Number	Mean	Std Error	Lower 95%	Upper 95%
0	140	123.526	7.277	109.16	137.89
1	27	423.055	16.570	390.34	455.77

Std Error uses a pooled estimate of error variance

t Test

1-0
Assuming unequal variances

Difference	299.528	t Ratio	8.870774
Std Err Dif	33.766	DF	27.0573
Upper CL Dif	368.803	Prob > |t|	<.0001*
Lower CL Dif	230.254	Prob > t	<.0001*
Confidence	0.95	Prob < t	1.0000



Means Comparisons

Comparisons for each pair using Student's t

Confidence Quantile

t	Alpha
1.97445	0.05

LSD Threshold Matrix

Abs(Dif)-LSD

	1	0
1	-46.27	263.80
0	263.80	-20.32

Positive values show pairs of means that are significantly different.

Appendix G: Additional Principal Component Analysis (PCA) on water chemistry parameters

Table G.1: Results of principal components analysis (PCA) of 2002 environmental data for samples sites in west Delta Marsh (n= 4), east Delta Marsh (n = 5), and Lake Manitoba (n = 1) and correlations between the environmental variables and principal components axes. Three separate PCAs were performed (1) All samples (east, west and lake), (2) East sample sites and lake, (3) west samples sites and lake. Table does not show inclusion of L. Manitoba.

PCA	Principal Component Axis	Variance explained %	Environmental Variable	Correlation Coefficient
(1) East, west, and Lake MB.	PCA1 (eigenvalue = 0.48)	48	Conductivity	-0.90
			Turbidity	-0.87
			TRP-P	0.86
			TN	-0.76
			TP	0.75
			OSS	-0.72
			pH	-0.63
			ISS	-0.53
	PCA2 (eigenvalue = 0.30)	30	DOC	-0.96
			Alkalinity	-0.90
			TP	-0.54
			TN	-0.53
			OSS	-0.48
			ISS	0.45
(2) East Only	PCA1 (eigenvalue = 0.53)	53	Conductivity	-0.95
			DOC	-0.94
			Alkalinity	-0.94
			OSS	-0.80
			TN	-0.76
			ISS	0.69
			TP	-0.65
			TRP-P	-0.51
	PCA2 (eigenvalue = 0.23)	23	TRP-P	0.82
			TP	0.74
			Turbidity	-0.70
			TN	-0.50
			ISS	0.46
(3) West Only	PCA1 (eigenvalue = 0.66)	66	Conductivity	-0.98
			ISS	-0.95
			TRP-P	0.92
			pH	-0.92
			Turbidity	-0.87
			OSS	-0.83
			TN	-0.82
			TP	0.82
	PCA2 (eigenvalue = 0.28)	28	DOC	-0.99
			Alkalinity	-0.90
			TP	-0.55
			OSS	-0.54
			TN	-0.48

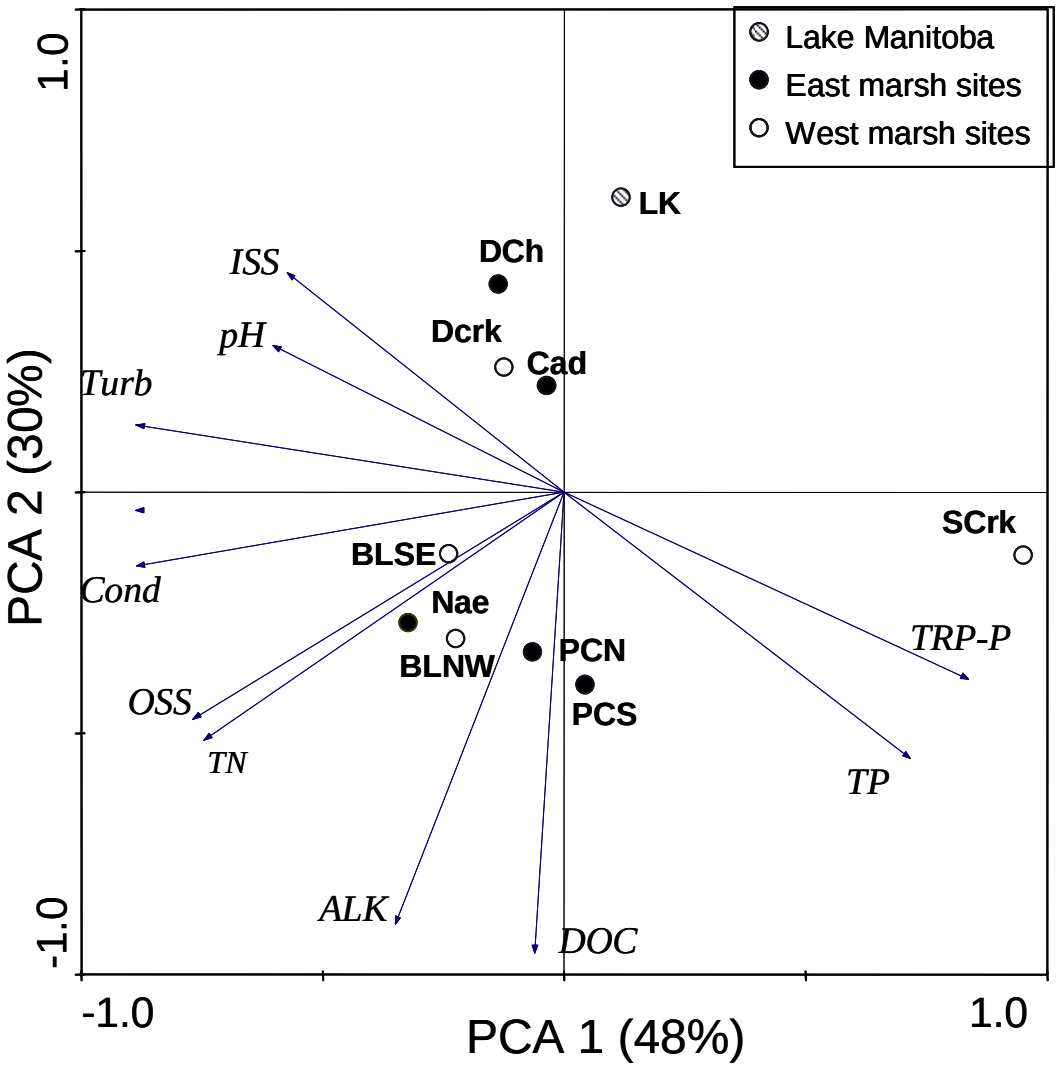


Figure G.1: Principal component analysis ordination of Lake Manitoba and nine samples sites on the east and west sides of Delta Marsh constrained by water chemistry parameters (vectors) in 2002. The first two PCA axes of the graph accounts for 78% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 48%, and PCA2 for 30%. Sites codes: (1) LK = Lake Manitoba, (2) DCh = Delta Channel, (3) CadE = Cadham Bay East, (4) Nae = Naegele's/Simpsons Bay (5) PCN = Portage Creek North, (6) PCS = Portage Creek South, (7) DCrk = Deep Creek, (8) BLSE = Big Lake south-east, (9) BLNW = Big Lake northwest, (10) SCrk = Short Creek. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TN = total nitrogen, TRP-P = total reactive phosphorus (PO4-P), TP= total phosphorus.

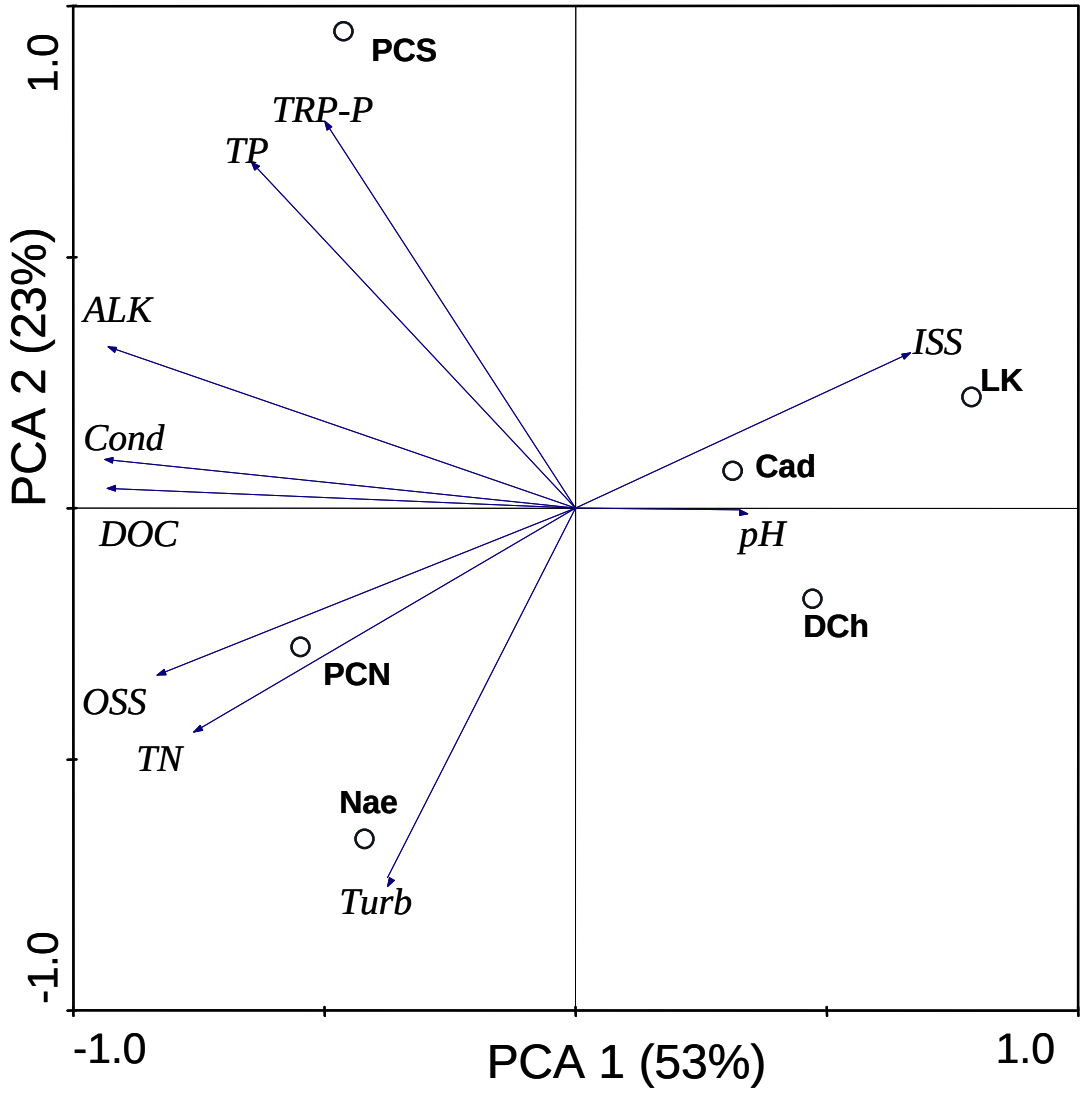


Figure G.2: Principal component analysis ordination of Lake Manitoba and five samples sites on the east side of Delta Marsh constrained by water chemistry parameters (vectors) in 2002. The first two PCA axes of the graph accounts for 76% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 53%, and PCA2 for 23%. Sites codes: (1) LK = Lake Manitoba, (2) DCh = Delta Channel, (3) CadE = Cadham Bay East, (4) Nae = Naegele's/Simpsons Bay (5) PCN = Portage Creek North, (6) PCS = Portage Creek South. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TN = total nitrogen, TRP-P = total reactive phosphorus (PO4-P), TP= total phosphorus.

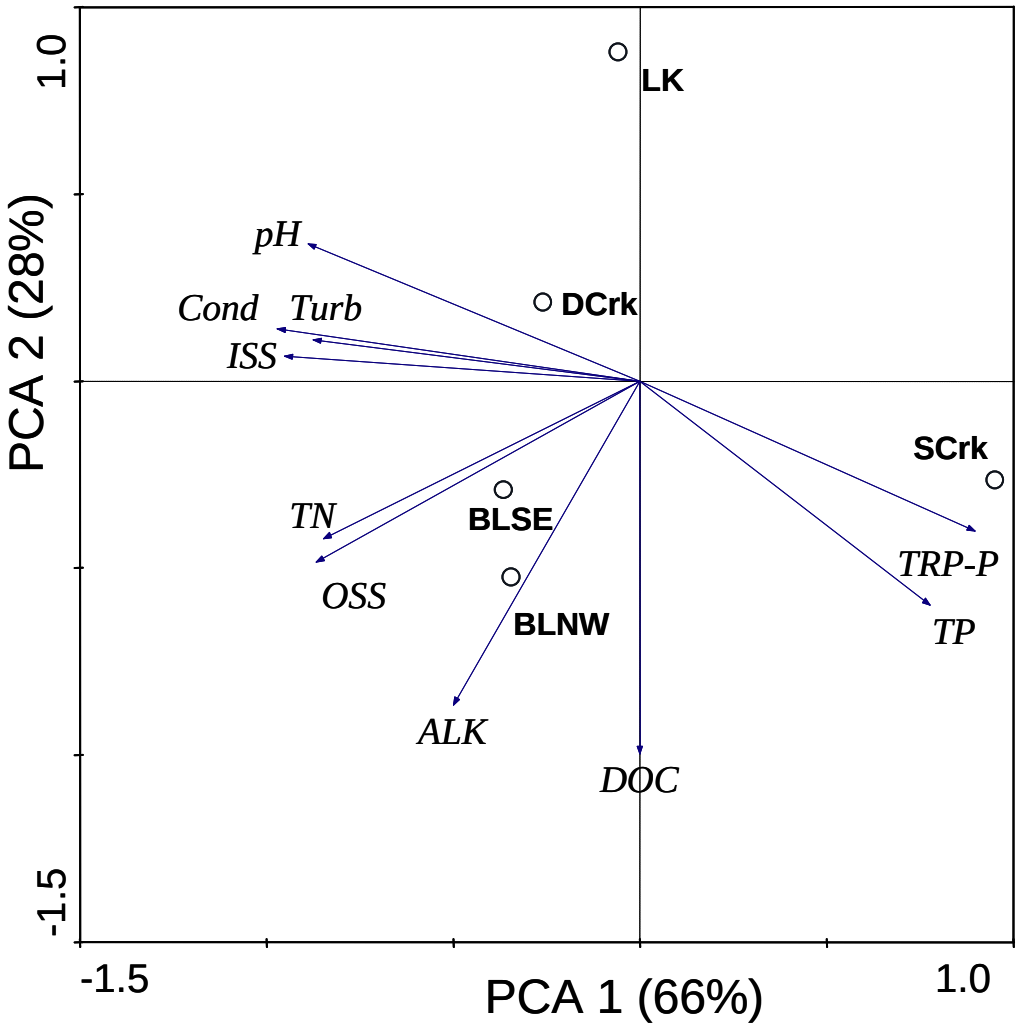


Figure G.3: Principal component analysis ordination of Lake Manitoba and four sample sites on the west side of Delta Marsh constrained by water chemistry parameters (vectors) in 2002. The first two PCA axes of the graph accounts for 94% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 66%, and PCA2 for 28%. Sites codes: (1) LK = Lake Manitoba, (2) DCrk = Deep Creek, (3)BLSE = Big Lake south-east, (4) BLNW = Big Lake northwest, (5) SCrk = Short Creek. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TN = total nitrogen, TRP = total reactive phosphorus (PO4-P), TP= total phosphorus.

Table G.2: Results of principal components analysis (PCA) of 2004 environmental data for connected sites in east delta marsh (n =7), isolated sites (n=6), and Lake Manitoba (n=1) and correlations between the environmental variables and principal components axes. Two separate PCAs were performed (1) All sites (connected east, isolated, and lake), and (2) east sample sites and lake.

PCA	Principal Component Axis	Variance explained %	Environmental Variable	Correlation Coefficient
(4) All sites	PCA1 (eigenvalue = 0.47)	47	OSS	-0.95
			Turbidity	-0.87
			Cl-	-0.85
			ISS	-0.84
			DIN	-0.80
			TN	-0.77
			TRP-P	0.73
			Conductivity	-0.68
			TP	0.56
			K+	-0.52
	PCA2 (eigenvalue = 0.30)	30	Alkalinity	0.98
			DOC	0.93
			TP	0.64
			TRP-P	0.57
			Conductivity	0.56
			TN	0.55
	PCA3 (eigenvalue = 0.13)	13	SO4	0.82
			pH	-0.76
(5) East Only	PCA1 (eigenvalue = 0.77)	77	Conductivity	-0.99
			DOC	-0.99
			OSS	-0.98
			K+	-0.95
			Alkalinity	-0.92
			TN	-0.89
			DIN	-0.89
			Cl-	-0.88
			pH	0.87
			TP	-0.86
			ISS	-0.78
			TRP-P	-0.78
			SO4	-0.73
			Turbidity	-0.70
	PCA2 (eigenvalue = 0.15)	17	Turbidity	-0.65
			SO4	0.63
			TRP-P	0.62
	PCA3 (eigenvalue = 0.05)	5	ISS	0.46

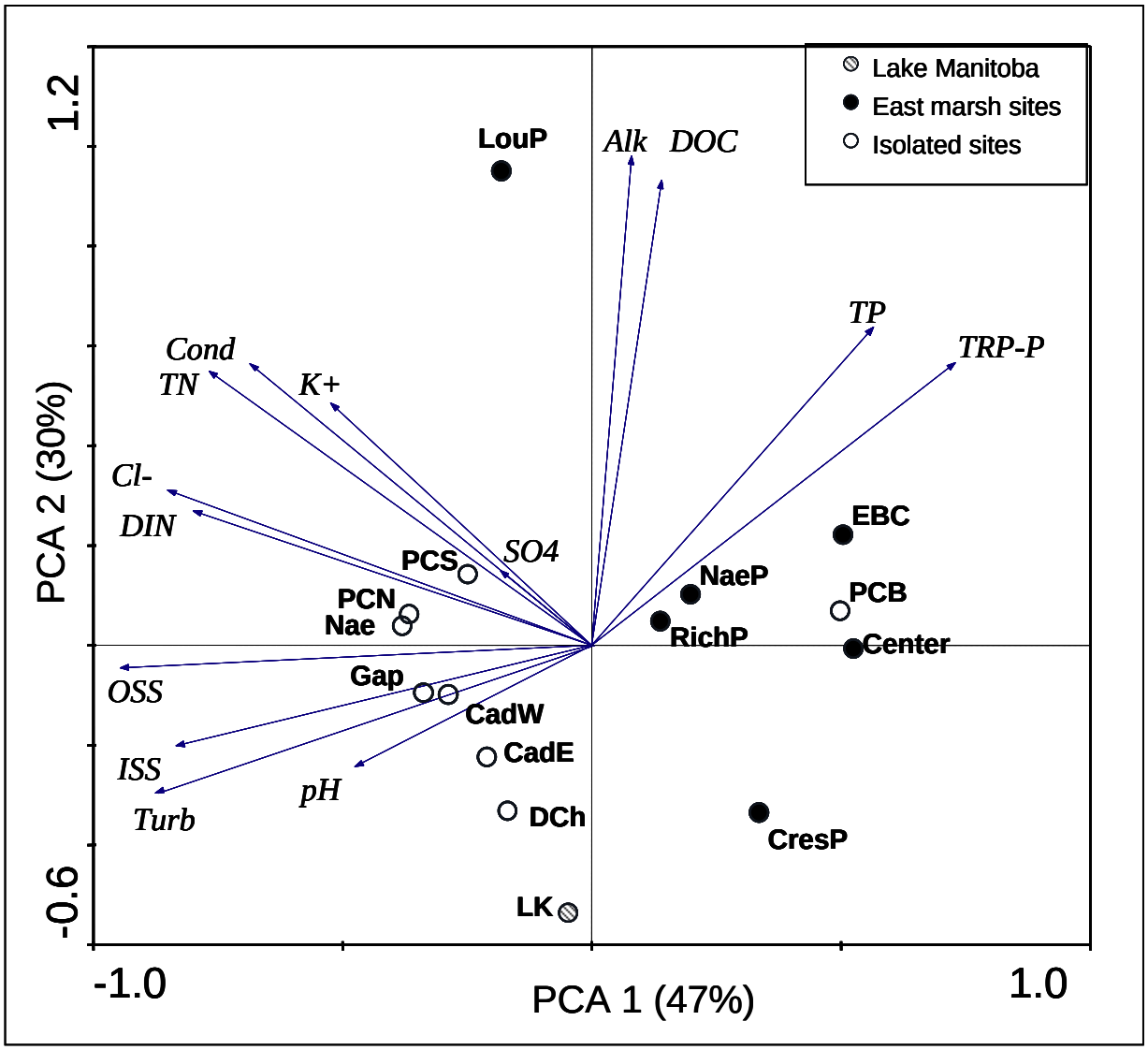


Figure G.4: Principal component analysis ordination of Lake Manitoba, eight connected sites on the east side of Delta Marsh, and six isolated sites in the marsh constrained by water chemistry parameters (vectors) in 2004. The first two PCA axes of the graph accounts for 77% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 47%, and PCA2 for 30%. Sites codes: (1) LK = Lake Manitoba, (2) DCh = Delta Channel, (3) CadE = Cadham Bay East, (4) CadW = Cadham Bay West, (5) Gap = The Gap, (6) Nae = Naegele's/Simpsons Bay (7) PCN = Portage Creek North, (8) PCS = Portage Creek South, (9) PCB = Portage Creek Bridge, (10) Center = Center Marsh, (11) ECB = East Blind Channel, (12) CresP = Crescent Pond, (13) NaeP = Naegele's Pond, (14) RichP = Richardson's Pond, (15) Louck's Pond. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TRP-P = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulfate, Cl = chloride, and K+= potassium.

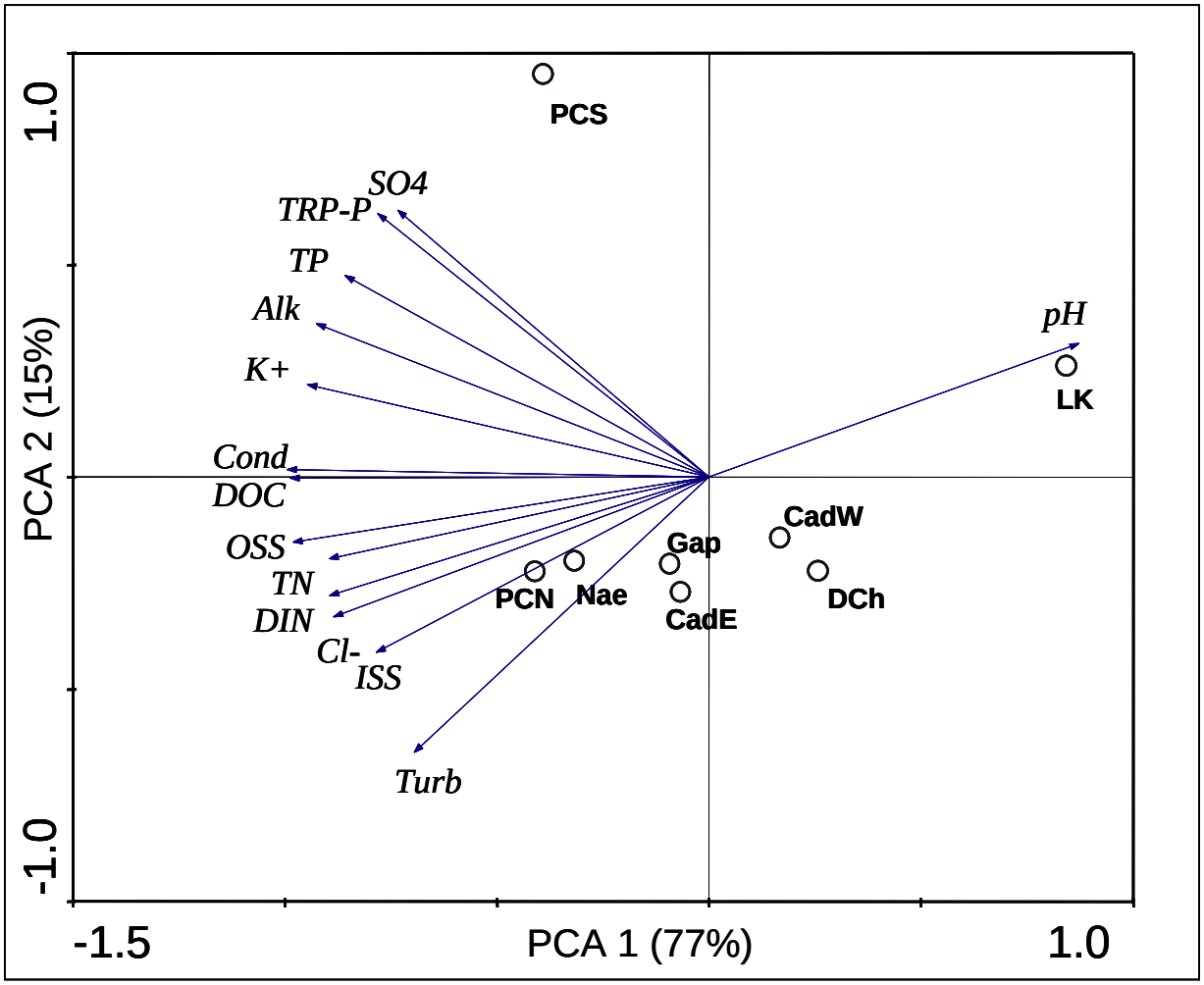


Figure G.5: Principal component analysis ordination of Lake Manitoba and seven connected sites on the east side of Delta Marsh constrained by water chemistry parameters (vectors) in 2004. The first two PCA axes of the graph accounts for 92% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 77%, and PCA2 for 15%. Sites codes: (1) LK = Lake Manitoba, (2) DCh = Delta Channel, (3) CadE = Cadham Bay East, (4) CadW = Cadham Bay West, (5) Gap = The Gap, (6) Nae = Naegele's/Simpsons Bay (7) PCN = Portage Creek North, and (8) PCS = Portage Creek South. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TRP-P = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulfate, Cl = chloride, and K+= potassium.

Table G.3: Results of principal components analysis (PCA) of 2005 environmental data for connected sample sites in east delta marsh (n=5), isolated sites (n=6), and Lake Manitoba (n =1) and correlations between the environmental variables and principal components axes. Two separate PCAs were performed (1) All sites (connected east, isolated, and lake), (2) connected east sample sites and lake.

PCA	Principal Component Axis	Variance explained %	Environmental Variable	Correlation Coefficient
(6) All sites	PCA1 (eigenvalue = 0.40)	40	pH	-0.94
			OSS	-0.81
			Conductivity	-0.79
			Cl-	-0.76
			K+	-0.75
			TN	-0.73
			Turbidity	-0.69
			ISS	-0.73
			TRP-P	0.60
			SO4	-0.59
			TP	0.51
	PCA2 (eigenvalue = 0.34)	34	DOC	0.94
			Alkalinity	0.92
			DIN	0.77
			Silica	-0.71
			TP	0.66
			TN	0.62
			TRP-P	0.61
			K+	0.59
			Turbidity	-0.54
(7) East Only	PCA1 (eigenvalue = 0.57)	57	TRP-P	0.97
			TP	0.96
			Turbidity	-0.94
			pH	-0.93
			Cl-	-0.91
			Conductivity	-0.87
			K+	-0.81
			OSS	-0.74
			DIN	0.72
			TN	-0.67
			Alkalinity	0.62
			Silica	-0.61
	PCA2 (eigenvalue = 0.24)	24	ISS	0.92
			DOC	-0.76
			Alkalinity	-0.65
			TN	-0.67
			DIN	-0.65

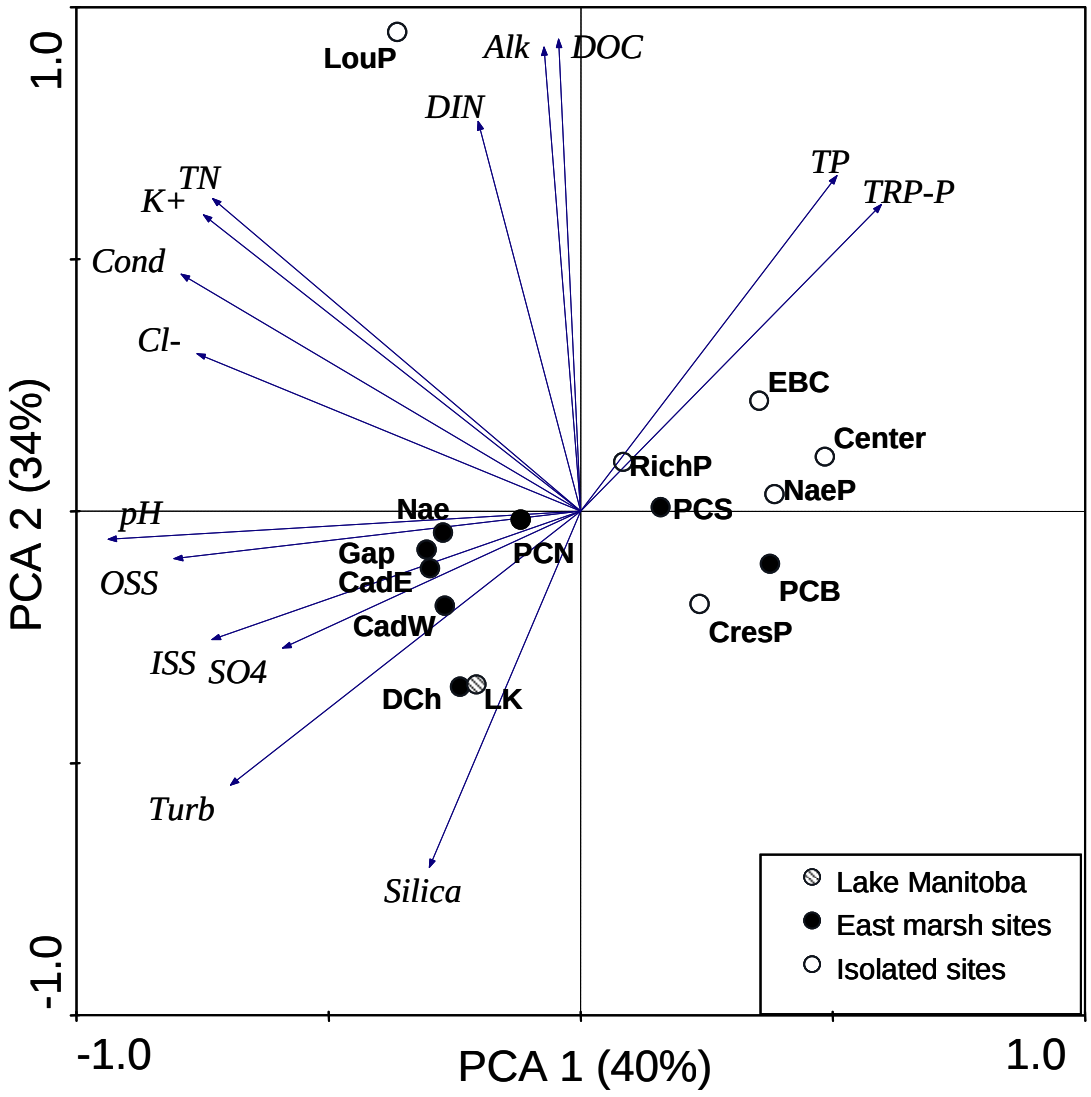


Figure G.6: Principal component analysis ordination of Lake Manitoba, eight connected sites on the east side of Delta Marsh, and six isolated sites in the marsh constrained by water chemistry parameters (vectors) in 2005. The first two PCA axes of the graph accounts for 74% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 40%, and PCA2 for 34%. Sites codes: (1) LK = Lake Manitoba, (2) DCh = Delta Channel, (3) CadE = Cadham Bay East, (4) CadW = Cadham Bay West, (5) Gap = The Gap, (6) Nae = Naegele's/Simpsons Bay (7) PCN = Portage Creek North, (8) PCS = Portage Creek South, (9) PCB = Portage Creek Bridge, (10) Center = Center Marsh, (11) ECB = East Blind Channel, (12) CresP = Crescent Pond, (13) NaeP = Naegele's Pond, (14) RichP = Richardson's Pond, (15) Louck's Pond. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TRP-P = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulfate, Cl-= chloride, K+= potassium, and silica = silica (Si).

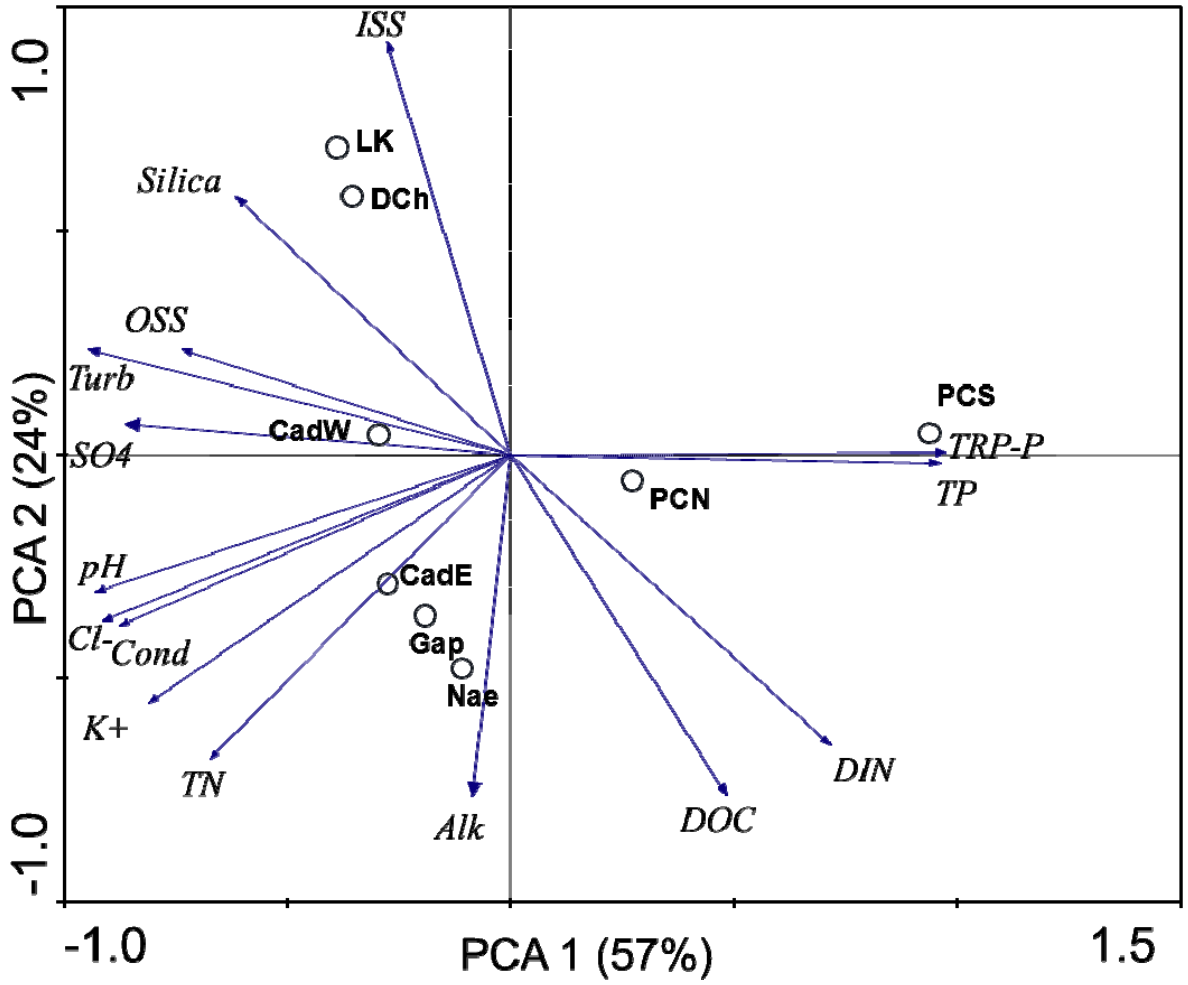


Figure G.7: Principal component analysis ordination of Lake Manitoba and eight connected sites on the east side of Delta Marsh constrained by water chemistry parameters (vectors) in 2005. The first two PCA axes of the graph accounts for 81% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 57%, and PCA2 for 24%. Sites codes: (1) LK = Lake Manitoba, (2) DCh = Delta Channel, (3) CadE = Cadham Bay East, (4) CadW = Cadham Bay West, (5) Gap = The Gap, (6) Nae = Naegele's/Simpsons Bay (7) PCN = Portage Creek North, (8) PCS = Portage Creek South, and (9) PCB = Portage Creek Bridge. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TRP-P = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulfate, Cl-= chloride, K+= potassium, and silica = silica (Si).

Table G.4 Results of principal components analysis (PCA) of 2003 environmental data for samples sites in west Delta Marsh (n= 7), east Delta Marsh (n = 5), center marsh (n=1), and Lake Manitoba (n = 1) and correlations between the environmental variables and principal components axes. Three separate PCAs were performed (1) All samples (east, west, center and lake), (2) east sample sites and lake, (3) west samples sites and lake.

PCA	Principal Component Axis	Variance explained %	Environmental Variable	Correlation Coefficient
(8) East & West	PCA1 (eigenvalue = 0.57)	57	TN	-0.94
			DIN	-0.89
			Cl-	-0.88
			Conductivity	-0.86
			ISS	-0.84
			OSS	-0.82
			Turbidity	-0.79
			SO4	-0.70
			DOC	-0.65
	PCA2 (eigenvalue = 0.26)	26	TRP-P	0.84
			TP	0.78
			ALK	0.68
			DOC	0.64
(9) East Only	PCA1 (eigenvalue = 0.17)	75	DIN	-0.97
			Alkalinity	-0.96
			Cl-	-0.96
			OSS	-0.96
			DOC	-0.95
			Turbidity	-0.92
			TN	-0.92
			ISS	-0.90
			Conductivity	-0.89
			SO4	-0.80
			TP	-0.78
			TRP-P	-0.71
	PCA2 (eigenvalue = 0.17)	17	pH	0.89
(10) West Only	PCA1 (eigenvalue = 0.78)	78	TN	-0.98
			DOC	-0.97
			Alkalinity	0.97
			ISS	-0.96
			TP	-0.92
			Conductivity	-0.91
			DIN	-0.91
			SO4	-0.90
			Cl-	-0.87
			OSS	-0.84
			TRP-P	-0.81
			Turbidity	-0.78
	PCA2 (eigenvalue = 0.10)	10	pH	-0.66

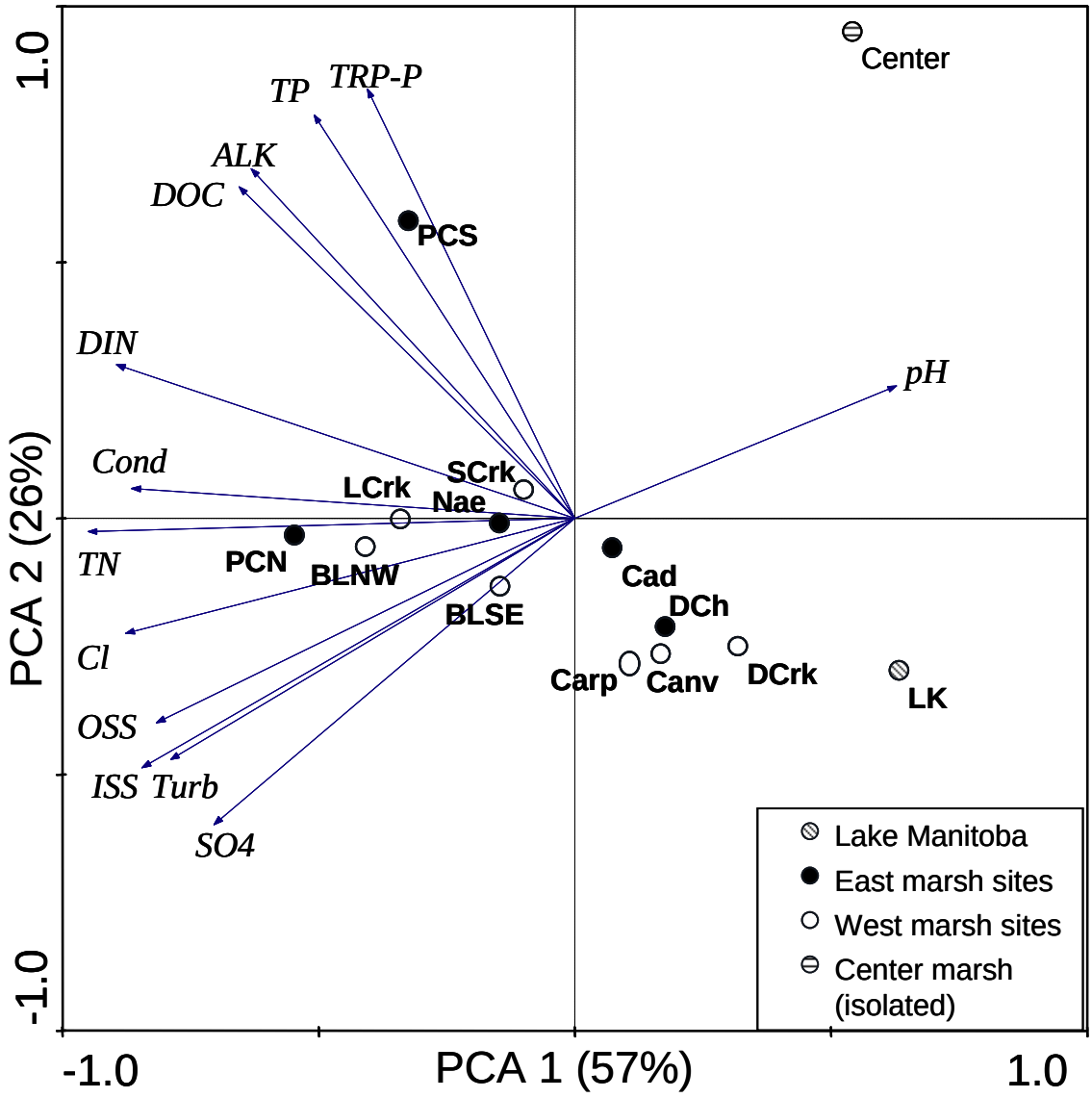


Figure G.8: Principal component analysis ordination of Lake Manitoba, seven sample sites on the west side of Delta Marsh, five on the east, and center marsh (isolated site) constrained by water chemistry parameters (vectors) in 2003. The first two PCA axes of the graph accounts for 83% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 57%, and PCA2 for 26%. Sites codes: (1) LK = Lake Manitoba, (1) LK = Lake Manitoba, (2)DCh = Delta Channel, (3) CadE = Cadham Bay East, (4) Nae = Naegele's/Simpsons Bay (5)PCN = Portage Creek North, (6) PCS = Portage Creek South, (7) DCrk = Deep Creek, (8) BLSE = Big Lake south-east, (9)BLNW = Big Lake northwest, (10) SCrk = Short Creek, (11) LCrk = Long Creek, (12) Canv = Canvasback Bay, (13) Carp = Carp Creek, (14) Center = Center Marsh. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TRP-P = total reactive phosphorus (PO4-P), TP= total phosphorus, TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulphate, and Cl = chloride.

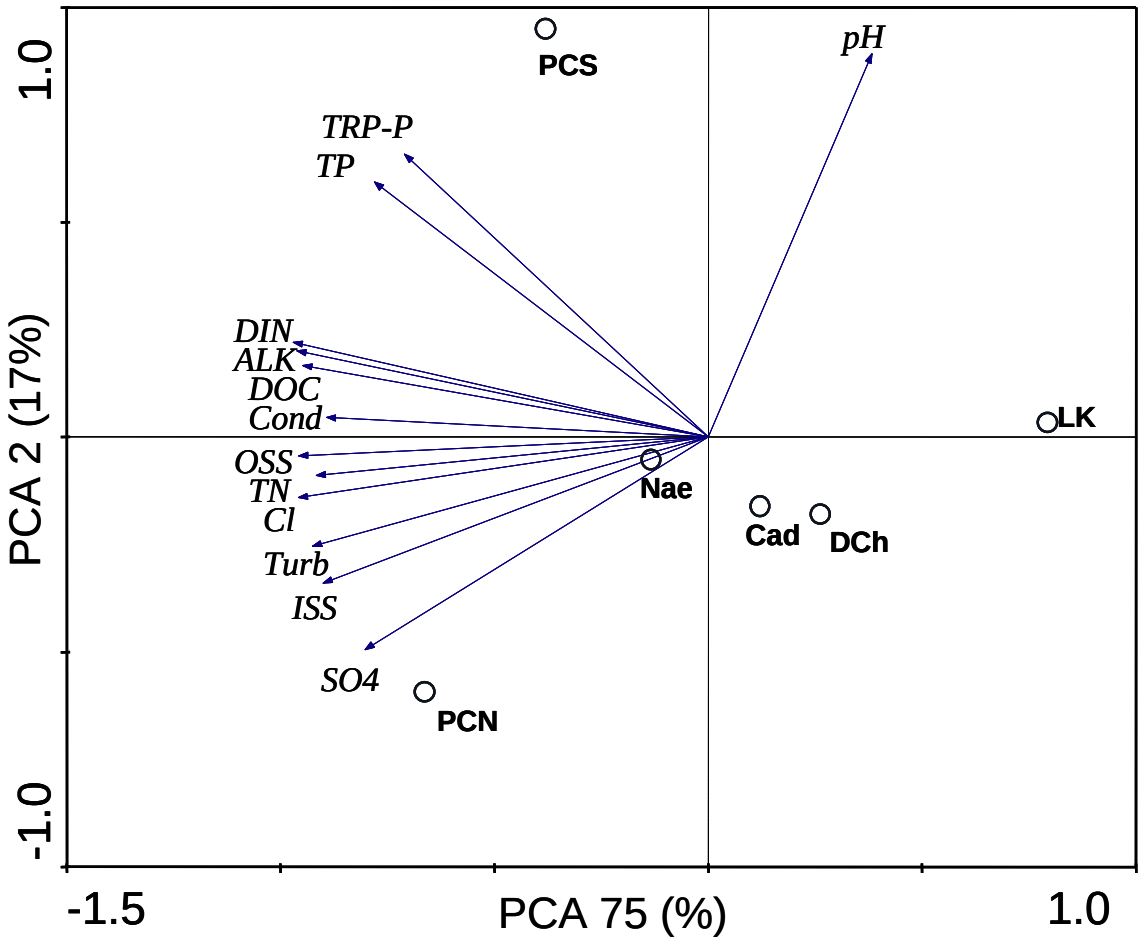


Figure G.9: Principal component analysis ordination of Lake Manitoba and five sample sites on the east side of Delta Marsh constrained by water chemistry parameters (vectors) in 2003. The first two PCA axes of the graph accounts for 94% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 92%, and PCA2 for 17%. Sites codes: (1) LK = Lake Manitoba, (2)DCh = Delta Channel, (3) CadE= Cadham Bay East, (4) Nae = Naegele's/Simpsons Bay (5) PCN = Portage Creek North, (6) PCS = Portage Creek South. Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TRP = total reactive phosphorus (PO4-P), TP= total phosphorus. TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulfate, and Cl = chloride.

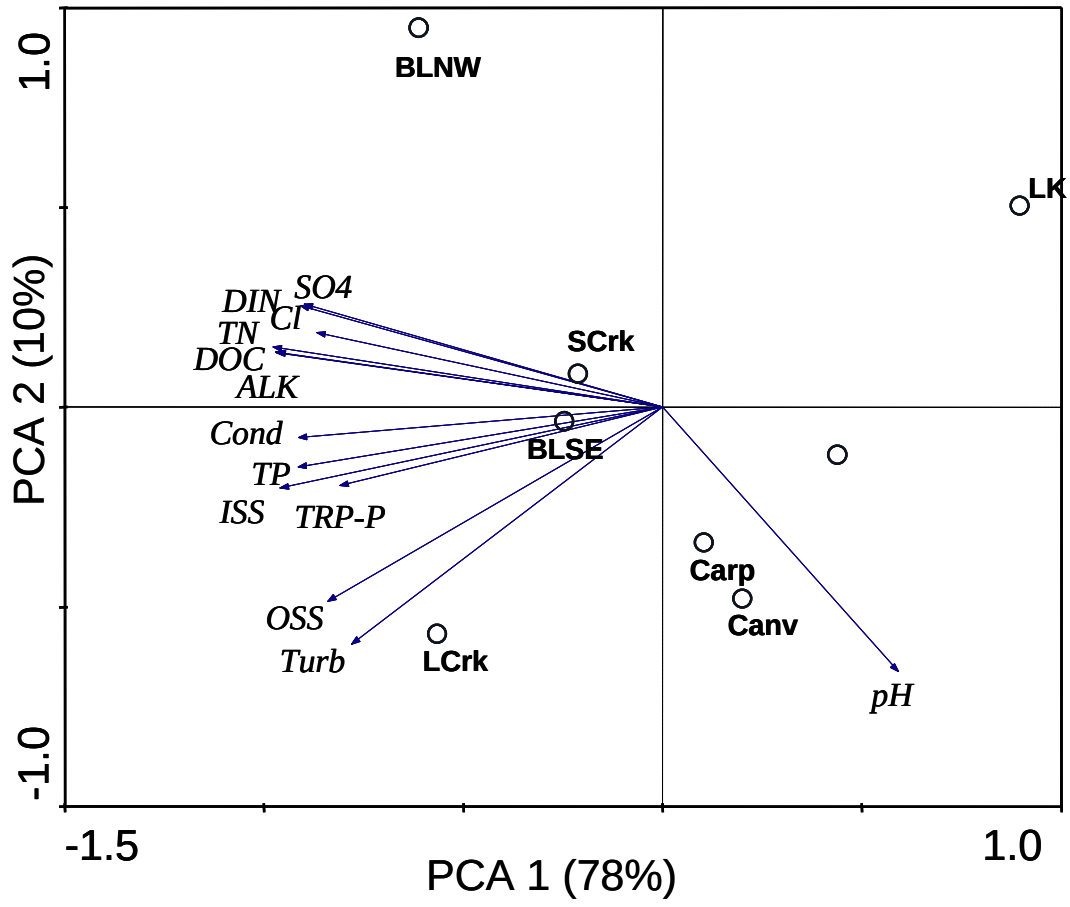


Figure G.10: Principal component analysis ordination of Lake Manitoba and seven sample sites on the west side of Delta Marsh constrained by water chemistry parameters (vectors) in 2003. The first two PCA axes of the graph accounts for 88% of the variation between the water chemistry of the samples sites, with PCA1 accounting for 78%, and PCA2 for 10%. Sites codes: (1) LK = Lake Manitoba, (2) DCrk = Deep Creek, (3)BLSE = Big Lake south-east, (4)BLNW = Big Lake northwest, (5) SCrk = Short Creek, (6) LCrk = Long Creek, (7) Canv = Canvasback Bay, (8) Carp = Carp Creek, Water Chemistry Parameter Codes: pH = pH, OSS = organic suspended solids, ISS = inorganic suspended solids, Cond = specific conductance, Alk = alkalinity, Turb = Turbidity (NTU), TRP = total reactive phosphorus (PO4-P), TP= total phosphorus. TN = total nitrogen, DIN = dissolve inorganic nitrogen, SO4 = sulfate, and Cl = chloride.

Appendix H: Summary table of NDS data and water column N:P molar ratios

Response of periphyton to NDS (mean ± SE) experiments using ANOVA and Tukey-Kramer Pair-wise HSD multiple comparison tests ($p \leq 0.05$), as well as predicted nutrient limitation based on molar ratios of water column N:P during the experiments at sample sites in Delta Marsh and Lake Manitoba, from 2002 to 2005. Note for NDS, means with different grouping letter are significantly different, and were interpreted as a response to nutrient additions. Results of statistics are given only for statistically significant positive responses to nutrient additions, n.s. = not significant, and N/A = not available. For molar ratios TN= total nitrogen, TP=total phosphorus, TRP=total reactive phosphorus, and DIN= dissolved inorganic nitrogen.

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN: TP	DIN: TRP	DIN: TP
Lake Manitoba	2002	June	C	A	9.0525	0.0021	N+P	p	N/A	N/A
			P	A						
			N	A						
			N+P	B						
Lake Manitoba	2002	July	C	A	143.994	<0.0001	N+P	P	N/A	N/A
			P	B						
			N	B						
			N+P	C						
Lake Manitoba	2002	August	C	A	104.607	<0.0001	N+P	P	N/A	N/A
			P	A						
			N	A						
			N+P	B						
Delta Channel	2002	May	C	A	26.5383	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	C						
Delta Channel	2002	June	C	A	10.3568	0.0012	N+P	P	N/A	N/A
			P	B						
			N	B						
			N+P	B						
Delta Channel	2002	July	C	A	10.3685	0.0012	N+P	P	N/A	N/A
			P	A						
			N	A						
			N+P	B						
Delta Channel	2002	August	C	A	14.5217	0.0003	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	C						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	IN:SRP	DIN:TP
Cadham Bay	2002	May	C	A	4.0655	0.033	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	A, B						
Cadham Bay	2002	June	C	A	8.0226	0.0034	N+P	P	N/A	N/A
			P	A						
			N	A						
			N+P	B						
Cadham Bay	2002	July	C	A	24.9932	<0.0001	N+P	P	N/A	N/A
			P	A						
			N	A						
			N+P	B						
Cadham Bay	2002	August	C	A	27.8627	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Simpsons Bay	2002	May	C	A	9.7459	0.0015	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Simpsons Bay	2002	June	C	A	263.078	<0.0001	N+P	P	N/A	N/A
			P	A, B						
			N	B						
			N+P	C						
Simpsons Bay	2002	July	C	A	18.1236	<0.0001	N+P	P	N/A	N/A
			P	A						
			N	A						
			N+P	B						
Simpsons Bay	2002	August	C	A	17.9712	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	C						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Portage Creek S	2002	May	C	A	4.9225	0.0187*	N	None	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Portage Creek S	2002	June	C	A	n.s.	n.s.	None	None	N/A	N/A
			P	A						
			N	A						
			N+P	A						
Portage Creek S	2002	July	C	A	n.s.	n.s.	none	None	N/A	N/A
			P	A						
			N	A						
			N+P	A						
Portage Creek S	2002	August	C	A	77.2291	<0.0001	N	N	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Deep Creek	2002	May	C	A	11.0573	0.0009	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Deep Creek	2002	June	C	A	41.1345	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Deep Creek	2002	July	C	A	45.7146	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Deep Creek	2002	August	C	A	6.4852	0.0074	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Big Lake SE	2002	May	C	A	n.s.	n.s.	n.s.	P	N/A	N/A
			P	A						
			N	A						
			N+P	A						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Big Lake SE	2002	June	C	A	19.5965	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Big Lake SE	2002	July	C	A	326.179	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	C						
Big Lake SE	2002	August	C	A	23.9554	<0.0001	N	None	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Short Creek	2002	May	C	A	n.s.	n.s.	n.s.	P	N/A	N/A
			P	A						
			N	A						
			N+P	A						
Short Creek	2002	June	C	A	46.6697	<0.0001	N	None	N/A	N/A
			P	A						
			N	B						
			N+P	C						
Short Creek	2002	July	C	A	101.152	0.0013	N	N	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Short Creek	2002	August	C	A	30.7387	<0.0001	N	N	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Big Lake NW	2002	May	C	A	n.s.	n.s.	n.s.	P	N/A	N/A
			P	A						
			N	A						
			N+P	A						
Big Lake NW	2002	June	C	A	8.1665	0.0031	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Big Lake NW	2002	July	C	A	18.9585	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Big Lake NW	2002	August	C	A	53.7211	<0.0001	N	P	N/A	N/A
			P	A						
			N	B						
			N+P	B						
Lake Manitoba	2003	May	C	A	42.2212	<0.0001	N+P	P	P	P
			P	A						
			N	A						
			N+P	B						
Lake Manitoba	2003	June	C	A	59.1165	<0.0001	P	P	N	None
			P	B						
			N	A						
			N+P	C						
Lake Manitoba	2003	July	C	A	117.1023	<0.0001	N+P	P	N	None
			P	A						
			N	A						
			N+P	B						
Lake Manitoba	2003	August	C	A	18.7558	<0.0001	P	P	P	None
			P	B						
			N	A						
			N+P	B						
Delta Channel	2003	May	C	A	12.9186	0.0005	N	P	None	P
			P	A						
			N	B						
			N+P	B						
Delta Channel	2003	June	C	A	17.6705	0.0001	N	P	None	N
			P	A						
			N	B						
			N+P	A						
Delta Channel	2003	July	C	A	81.7162	<0.0001	N	P	None	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Delta Channel	2003	August	C	A	22.2135	<0.0001	N	P	None	N
			P	A						
			N	B						
			N+P	B						
Cadham Bay	2003	May	C	A	n.s.	n.s.	None	P	N	P
			P	A						
			N	A						
			N+P	A						
Cadham Bay	2003	June	C	A	n.s.	n.s.	None	P	N	None
			P	A						
			N	A						
			N+P	A						
Cadham Bay	2003	July	C	A	7.9907	0.0034	N	P	N	N
			P	A						
			N	B						
			N+P	C						
Cadham Bay	2003	August	C	A	21.2984	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	A						
Simpsons Bay	2003	May	C	A	n.s.	n.s.	None	P	None	N
			P	A						
			N	A						
			N+P	A						
Simpsons Bay	2003	June	C	A	10.5741	0.0011	N	P	None	P
			P	A						
			N	A						
			N+P	A				P	None	N
Simpsons Bay	2003	July	C	A	39.9892	<0.0001	N			
			P	A						
			N	B						
			N+P	B						
Simpsons Bay	2003	August	C	A	4.7178	0.0213	N	P	None	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Portage Creek N	2003	May	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Portage Creek N	2003	June	C	A	4.3192	0.0305	N	P	N	None
			P	A						
			N	B						
			N+P	B						
Portage Creek N	2003	July	C	A	8.1331	0.0032	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Portage Creek N	2003	August	C	A	8.3656	0.0035	N	P	N	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Portage Creek S	2003	May	C	A	n.s.	n.s.	None	N	N	N
			P	A						
			N	A						
			N+P	A						
Portage Creek S	2003	June	C	A	4.8778	0.0192	N	N	N	N
			P	A						
			N	B						
			N+P	A, B						
Portage Creek S	2003	July	C	A	22.5163	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Portage Creek S	2003	August	C	A	27.8161	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Deep Creek	2003	May	C	A	11.6471	0.0007	N	P	N	N
			P	A						
			N	B						
			N+P	C						
Deep Creek	2003	June	C	A	17.885	0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Deep Creek	2003	July	C	A	11.356	0.0008	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Deep Creek	2003	August	C	A	21.0025	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Canvasback Bay	2003	May	C	A	31.2832	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Canvasback Bay	2003	June	C	A	28.0153	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Canvasback Bay	2003	July	C	A	15.9498	0.0002	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Canvasback Bay	2003	August	C	A	33.8158	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	C						
Carp Creek	2003	May	C	A	n.s.	n.s	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Carp Creek	2003	June	C	A	41.6799	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	C						
Carp Creek	2003	July	C	A	8.1978	0.0031	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Carp Creek	2003	August	C	A	23.8837	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Big Lake SE	2003	May	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Big Lake SE	2003	June	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Big Lake SE	2003	July	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Big Lake SE	2003	August	C	A,B	10.3643	0.0012	N	P	N	N
			P	A						
			N	C						
			N+P	B						
Short Creek	2003	May	C	A	133.7314	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Short Creek	2003	June	C	A	4.3037	0.028	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Short Creek	2003	July	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Short Creek	2003	August	C	A	24.8244	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	C						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Big Lake NW	2003	May	C	A	n.s.	n.s.	None	P	N	None
			P	A						
			N	A						
			N+P	A						
Big Lake NW	2003	June	C	A	n.s.	n.s.	None	P	N	None
			P	A						
			N	A						
			N+P	A						
Big Lake NW	2003	July	C	A	n.s.	n.s.	None	P	N	None
			P	A						
			N	A						
			N+P	A						
Big Lake NW	2003	August	C	A	13.8314	0.0003	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Center Marsh	2003	May	C	A	38.9563	<0.0001	N	None	N	N
			P	A						
			N	B						
			N+P	C						
Center Marsh	2003	June	C	A	21.3744	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Center Marsh	2003	July	C	A	4.275	0.0119	N	None	N	N
			P	A						
			N	B						
			N+P	B						
Center Marsh	2003	August	C	A	13.692	0.0004	N	N	N	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Lake Manitoba	2004	May	C	A	32.0807	<0.0001	N	P	None	None
			P	A						
			N	B						
			N+P	C						
Lake Manitoba	2004	June	C	A	11.0077	0.0009	N	P	None	N
			P	A						
			N	B						
			N+P	B						
Lake Manitoba	2004	July	C	A	8.7423	0.0024	N+P	P	None	N
			P	B						
			N	B						
			N+P	B						
Lake Manitoba	2004	August	C	N/A	N/A	N/A	N/A	P	None	N
			P							
			N							
			N+P							
Delta Channel	2004	May	C	A	47.6246	<0.0001	N	P	None	N
			P	A						
			N	B						
			N+P	C						
Delta Channel	2004	June	C	A	3.7745	0.0510	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Delta Channel	2004	July	C	A	21.826	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Delta Channel	2004	August	C	A	7.7954	0.0038	N	P	None	None
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Cadham E	2004	May	C	A	24.4255	<0.0001*	N	P	None	N
			P	A						
			N	B						
			N+P	C						
Cadham E	2004	June	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						
Cadham E	2004	July	C	A	35.1561	<0.0001*	N	P	None	None
			P	A						
			N	B						
			N+P	B						
Cadham E	2004	August	C	A	n.s.	n.s.	n.s.	P	P	None
			P	A						
			N	A						
			N+P	A						
The Gap	2004	May	C	A	9.2329	0.0019	N	P	None	N
			P	A,B						
			N	B,C						
			N+P	C						
The Gap	2004	June	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						
The Gap	2004	July	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						
The Gap	2004	August	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						
Simpsons Bay	2004	May	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						
Simpsons Bay	2004	June	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Simpsons Bay	2004	July	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						
Simpsons Bay	2004	August	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						
Portage Creek N	2004	May	C	A	18.4123	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Portage Creek N	2004	June	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Portage Creek N	2004	July	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Portage Creek N	2004	August	C	A	n.s.	n.s.	None	P	None	P
			P	A						
			N	A						
			N+P	A						
Portage Creek S	2004	May	C	A	43.8342	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Portage Creek S	2004	June	C	A	4.6397	0.0224	N	P	N	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
Portage Creek S	2004	July	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
Portage Creek S	2004	August	C	A	n.s.	n.s.	None	P	N	None
			P	A						
			N	A						
			N+P	A						
Center Marsh	2004	May	C	A	30.9666	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Center Marsh	2004	June	C	A	8.1913	0.0031	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Center Marsh	2004	July	C	A	29.318	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Center Marsh	2004	August	C	A	37.3006	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						
EBC	2004	May	C	A	75.0865	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	B						
EBC	2004	June	C	A	4.1437	0.0313*	N	N	N	N
			P	A						
			N	B						
			N+P	B						
EBC	2004	July	C	A	26.2809	<0.0001*	N	N	N	N
			P	B						
			N	C						
			N+P	C						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
EBC	2004	August	C	A	10.7421	0.001	N	P	None	N
			P	A						
			N	B						
			N+P	B						
Crescent Pond	2004	May	C	A	9.6132	0.0016	N	P	N	None
			P	A						
			N	B						
			N+P	A						
Crescent Pond	2004	June	C	A	202.3828	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	C						
Crescent Pond	2004	July	C	A	8.2687	0.003	N	P	N	N
			P	A						
			N	B						
			N+P	A,B						
Crescent Pond	2004	August	C	A	17.2584	0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	A, B						
Neageles Pond	2004	May	C	A	9.3442	0.0018	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Neageles Pond	2004	June	C	A	34.1447	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Neageles Pond	2004	July	C	A	25.4818	<0.0001	N	None	N	N
			P	A						
			N	B						
			N+P	B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN: TP	DIN: SRP	DIN: TP
Neageles Pond	2004	August	C	A	8.5593	0.0026	N	p	N	N
			P	A						
			N	B						
			N+P	B						
Delta Channel	2005	May	C	A	21.7927	<0.0001	N	p	N	N
			P	A						
			N	B						
			N+P	B						
Delta Channel	2005	June	C	A	48.2272	<0.0001	N	p	N	N
			P	A						
			N	B						
			N+P	C						
Delta Channel	2005	July	C	A	23.7831	<0.0001	N+P	p	None	N
			P	A						
			N	B						
			N+P	C						
Delta Channel	2005	August	C	A	n.s.	n.s.	None	P	N	None
			P	A						
			N	A						
			N+P	A						
The Gap	2005	May	C	A	n.s.	n.s.	None	P	N	N
			P	A						
			N	A						
			N+P	A						
The Gap	2005	June	C	A	14.9315	0.0003	N	P	N	None
			P	A						
			N	B						
			N+P	B						
The Gap	2005	July	C	A	53.8613	<0.0001	N	P	N	N
			P	A						
			N	B						
			N+P	C						
The Gap	2005	August	C	A	n.s.	n.s.	None	P	None	None
			P	A						
			N	A						
			N+P	A						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN: TP	DIN: SRP	DIN: TP
Portage Creek S	2005	May	C	A	21.3106	<0.0001	N	p	N	N
			P	A						
			N	B						
			N+P	B						
Portage Creek S	2005	June	C	A	n.s.	n.s.	None	N	N	N
			P	A						
			N	A						
			N+P	A						
Portage Creek S	2005	July	C	A	29.5181	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	A						
Portage Creek S	2005	August	C	A	15.4839	0.0002	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Center Marsh	2005	May	C	A	19.9376	0.0005	N	N	N	N
			P	A						
			N	B						
			N+P	A,B						
Center Marsh	2005	June	C	A	7.9328	0.0035	N	N	N	N
			P	A						
			N	B						
			N+P	A,B						
Center Marsh	2005	July	C	A	5.4726	0.0133	N	N	N	N
			P	A						
			N	B						
			N+P	A,B						
Center Marsh	2005	August	C	A	7.9328	0.0035	N	N	N	N
			P	A						
			N	B						
			N+P	A,B						

			Nutrient Diffusing Substrata					Molar Ratios		
Site	Year	Batch	Treatment	Mean Grouping	F(3,12)	P	Limitation Status	TN:TP	DIN:SRP	DIN:TP
EBC	2005	May	C	A	78.4577	<0.0001	N	N	N	N
			P	A						
			N	B						
			N+P	C						
EBC	2005	June	C	A	6.078	0.0093	N	N	N	N
			P	A						
			N	B						
			N+P	B						
EBC	2005	July	C	A	5.0147	0.0176	N	N	N	N
			P	A						
			N	B						
			N+P	A						
EBC	2005	August	C	A	9.4817	0.0017	N	N	N	N
			P	A						
			N	A						
			N+P	B						
Naegeles Pond	2005	May	C	A	15.8662	0.0002	N	P	N	N
			P	A						
			N	B						
			N+P	B						
Naegeles Pond	2005	June	C	A	181.1636	<0.0001	N	None	N	N
			P	A						
			N	B						
			N+P	C						
Naegeles Pond	2005	July	C	A	11.8862	0.0007	N	N	N	N
			P	A						
			N	B						
			N+P	B						
Naegeles Pond	2005	August	C	A	11.8367	0.0007	N	None	N	N
			P	A						
			N	B						
			N+P	A,B						

Appendix I: Total microcystin data

Analysis performed by: AlgalTox International A Division of Miette Environmental Consulting Inc.
Type of Analysis: Total Microcystin
Method of Analysis: Protein Phosphatase Inhibition (based on An and Carmichael, 1994)
Calibration Curve: r^2 0.95 Detection Limit (ug/L): 0.10

Site ID	Site number	Date (2005)	Result (µg/L MCLR-eq)
Portage Creek S	1	27-Jun	0.26
Portage Creek S	1	05-Jul	<0.10
Portage Creek S	1	11-Jul	<0.10
Portage Creek S	1	18-Jul	<0.10
Portage Creek S	1	25-Jul	<0.10
Portage Creek S	1	02-Aug	<0.10
Portage Creek S	1	08-Aug	<0.10
Portage Creek S	1	15-Aug	<0.10
Portage Creek N	2	27-Jun	2.93
Portage Creek N	2	27-Jun	1.01
Portage Creek N	2	05-Jul	0.30
Portage Creek N	2	11-Jul	0.25
Portage Creek N	2	18-Jul	<0.10
Portage Creek N	2	25-Jul	<0.10
Portage Creek N	2	02-Aug	<0.10
Portage Creek N	2	08-Aug	0.10
Portage Creek N	2	15-Aug	<0.10
Simpson Bay	3	27-Jun	3.17
Simpson Bay	3	05-Jul	0.55
Simpson Bay	3	11-Jul	0.75
Simpson Bay	3	18-Jul	0.33
Simpson Bay	3	25-Jul	<0.10
Simpson Bay	3	02-Aug	<0.10
Simpson Bay	3	08-Aug	<0.10
Simpson Bay	3	15-Aug	<0.10
Cadham Bay E	4	27-Jun	<0.10
Cadham Bay E	4	05-Jul	0.54
Cadham Bay E	4	11-Jul	1.55
Cadham Bay E	4	18-Jul	0.37
Cadham Bay E	4	25-Jul	0.14
Cadham Bay E	4	02-Aug	<0.10
Cadham Bay E	4	08-Aug	<0.10
Cadham Bay E	4	15-Aug	<0.10
Delta Channel	5	05-Jul	0.22
Delta Channel	5	11-Jul	0.61
Delta Channel	5	18-Jul	0.21
Delta Channel	5	25-Jul	<0.10
Delta Channel	5	02-Aug	<0.10
Delta Channel	5	08-Aug	<0.10
Delta Channel	5	15-Aug	<0.10

Site ID	Site number	Date	Result (µg/L MCLR-eq)
Lake Manitoba	6	27-Jun	<0.10
Lake Manitoba	6	05-Jul	<0.10
Lake Manitoba	6	11-Jul	0.81
Lake Manitoba	6	18-Jul	<0.10
Lake Manitoba	6	25-Jul	0.10
Lake Manitoba	6	02-Aug	<0.10
Lake Manitoba	6	08-Aug	<0.10
Lake Manitoba	6	15-Aug	0.60
Center Marsh	15	27-Jun	0.14
Center Marsh	15	05-Jul	0.51
Center Marsh	15	11-Jul	<0.10
Center Marsh	15	18-Jul	<0.10
Center Marsh	15	25-Jul	<0.10
Center Marsh	15	02-Aug	<0.10
Center Marsh	15	08-Aug	<0.10
Center Marsh	15	15-Aug	<0.10
Portage Creek Bridge	16	27-Jun	<0.10
Portage Creek Bridge	16	05-Jul	<0.10
Portage Creek Bridge	16	11-Jul	<0.10
Portage Creek Bridge	16	18-Jul	<0.10
Portage Creek Bridge	16	25-Jul	<0.10
Portage Creek Bridge	16	02-Aug	<0.10
Portage Creek Bridge	16	08-Aug	0.17
Portage Creek Bridge	16	15-Aug	0.15
East Blind Channel	23	05-Jul	<0.10
East Blind Channel	23	11-Jul	<0.10
East Blind Channel	23	18-Jul	<0.10
East Blind Channel	23	25-Jul	<0.10
East Blind Channel	23	02-Aug	<0.10
East Blind Channel	23	08-Aug	<0.10
East Blind Channel	23	15-Aug	<0.10
Crescent Pond	26	27-Jun	<0.10
Crescent Pond	26	05-Jul	<0.10
Crescent Pond	26	11-Jul	<0.10
Crescent Pond	26	18-Jul	<0.10
Crescent Pond	26	25-Jul	<0.10
Crescent Pond	26	02-Aug	<0.10
Crescent Pond	26	08-Aug	<0.10
Crescent Pond	26	15-Aug	<0.10
Richardson’s Pothole	28	05-Jul	<0.10
Richardson’s Pothole	28	11-Jul	<0.10
Richardson’s Pothole	28	18-Jul	<0.10
Richardson’s Pothole	28	25-Jul	<0.10
Richardson’s Pothole	28	02-Aug	<0.10
Richardson’s Pothole	28	08-Aug	0.18
Richardson’s Pothole	28	15-Aug	<0.10

Appendix J: NDS light limitation experiment data

Site: Delta Channel Date NDS in: 5/17/05 11:30 Date NDS out: 6/6/05 10:45

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
Sand 1	10	0	0	5.43	0.00	5.43	0.81
Sand 2	10	0	0	3.23	0.03	3.26	0.63
Sand 3	10	0	0	6.83	0.00	6.83	0.89
Sand 4	10	0	0	3.84	0.23	4.07	0.71
A1	10	0	0	7.44	0.00	7.44	0.93
A2	10	0	0	10.00	0.00	10.00	1.04
A3	10	0	0	7.81	0.00	7.81	0.94
A4	10	0	0	5.18	0.61	5.80	0.83
B1	10	0	0.05	3.48	0.38	3.85	0.69
B2	10	0	0.05	2.32	1.99	4.30	0.72
B3	10	0	0.05	6.83	0.00	6.83	0.89
B4	10	0	0.05	3.96	0.14	4.11	0.71
D1	10	0.05	0	9.51	3.57	13.09	1.15
D2	10	0.05	0	18.66	0.00	18.66	1.29
D3	10	0.05	0	11.75	1.65	13.40	1.16
D4	10	0.05	0	16.77	0.48	17.26	1.26
E1	10	0.05	0.05	17.32	0.00	17.32	1.26
E2	10	0.05	0.05	11.22	0.00	11.22	1.09
E3	10	0.05	0.05	2.87	13.86	16.73	1.25
E4	10	0.05	0.05	13.78	0.00	13.78	1.17
Sand 1	30	0	0	3.78	0.29	4.07	0.70
Sand 2	30	0	0	2.13	0.48	2.61	0.56
Sand 3	30	0	0	3.90	0.31	4.21	0.72
Sand 4	30	0	0	2.26	0.22	2.48	0.54
A1	30	0	0	2.07	0.09	2.16	0.50
A2	30	0	0	3.05	0.00	3.05	0.61
A3	30	0	0	2.38	0.00	2.38	0.53
A4	30	0	0	5.31	0.00	5.31	0.80
B1	30	0	0.05	2.20	0.04	2.23	0.51
B2	30	0	0.05	1.71	0.20	1.91	0.46
B3	30	0	0.05	1.77	0.38	2.15	0.50
B4	30	0	0.05	1.52	0.14	1.66	0.43
D1	30	0.05	0	3.90	0.41	4.31	0.73
D2	30	0.05	0	8.78	0.00	8.78	0.99
D3	30	0.05	0	6.59	0.00	6.59	0.88
D4	30	0.05	0	3.35	0.00	3.35	0.64
E1	30	0.05	0.05	0.91	4.39	5.30	0.80
E2	30	0.05	0.05	5.31	0.00	5.31	0.80
E3	30	0.05	0.05	3.84	0.47	4.31	0.73
E4	30	0.05	0.05	4.33	0.00	4.33	0.73
Sand 1	50	0	0	0.67	0.31	0.98	0.30
Sand 2	50	0	0	1.95	0.34	2.30	0.52
Sand 3	50	0	0	1.46	0.44	1.90	0.46
Sand 4	50	0	0	1.71	0.48	2.18	0.50
A1	50	0	0	1.10	0.07	1.17	0.34
A2	50	0	0	1.46	1.20	2.66	0.56
A3	50	0	0	0.91	0.49	1.40	0.38
A4	50	0	0	1.04	0.65	1.68	0.43
B1	50	0	0.05	1.40	0.00	1.40	0.38
B2	50	0	0.05	0.73	0.05	0.78	0.25
B3	50	0	0.05	1.46	0.00	1.46	0.39
B4	50	0	0.05	1.22	0.06	1.28	0.36
D1	50	0.05	0	1.28	0.00	1.28	0.36
D2	50	0.05	0	1.89	0.00	1.89	0.46
D3	50	0.05	0	0.55	0.47	1.02	0.30
D4	50	0.05	0	lost	lost	lost	lost
E1	50	0.05	0.05	1.04	0.16	1.20	0.34
E2	50	0.05	0.05	1.71	0.00	1.71	0.43
E3	50	0.05	0.05	1.28	0.03	1.31	0.36
E4	50	0.05	0.05	lost	lost	lost	lost

Site: Delta Channel Date NDS in: 6/6/05 10:45 Date NDS out: 6/28/05 8:45

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	8.42	0.00	8.42	0.97
A2	10	0	0	7.99	0.00	7.99	0.95
A3	10	0	0	9.82	0.00	9.82	1.03
A4	10	0	0	6.04	0.00	6.04	0.85
A5	10	0	0	5.06	0.00	5.06	0.78
B1	10	0	0.05	4.64	0.00	4.64	0.75
B2	10	0	0.05	4.70	0.00	4.70	0.76
B3	10	0	0.05	2.38	0.00	2.38	0.53
B4	10	0	0.05	3.54	0.00	3.54	0.66
B5	10	0	0.05	4.64	0.00	4.64	0.75
D1	10	0.05	0	23.18	0.00	23.18	1.38
D2	10	0.05	0	22.26	0.00	22.26	1.37
D3	10	0.05	0	21.84	0.00	21.84	1.36
D4	10	0.05	0	28.36	0.00	28.36	1.47
D5	10	0.05	0	20.19	0.00	20.19	1.33
E1	10	0.05	0.05	9.33	0.00	9.33	1.01
E2	10	0.05	0.05	15.37	0.00	15.37	1.21
E3	10	0.05	0.05	12.44	0.00	12.44	1.13
E4	10	0.05	0.05	18.79	0.00	18.79	1.30
E5	10	0.05	0.05	20.86	0.00	20.86	1.34
A1	30	0	0	8.72	0.00	8.72	0.99
A2	30	0	0	8.30	0.00	8.30	0.97
A3	30	0	0	9.58	0.00	9.58	1.02
A4	30	0	0	5.92	0.00	5.92	0.84
A5	30	0	0	8.54	0.00	8.54	0.98
B1	30	0	0.05	7.56	0.00	7.56	0.93
B2	30	0	0.05	6.77	0.00	6.77	0.89
B3	30	0	0.05	3.78	0.00	3.78	0.68
B4	30	0	0.05	3.60	0.00	3.60	0.66
B5	30	0	0.05	3.90	0.00	3.90	0.69
D1	30	0.05	0	22.20	0.00	22.20	1.37
D2	30	0.05	0	21.53	0.00	21.53	1.35
D3	30	0.05	0	24.28	0.00	24.28	1.40
D4	30	0.05	0	20.49	0.00	20.49	1.33
D5	30	0.05	0				
E1	30	0.05	0.05	19.64	0.00	19.64	1.31
E2	30	0.05	0.05				
E3	30	0.05	0.05	16.47	0.00	16.47	1.24
E4	30	0.05	0.05	18.48	0.00	18.48	1.29
E5	30	0.05	0.05	20.31	0.00	20.31	1.33
A1	50	0	0	4.27	0.00	4.27	0.72
A2	50	0	0	4.76	0.00	4.76	0.76
A3	50	0	0	6.71	0.00	6.71	0.89
A4	50	0	0	5.79	0.00	5.79	0.83
A5	50	0	0	4.15	0.00	4.15	0.71
B1	50	0	0.05	2.44	0.00	2.44	0.54
B2	50	0	0.05	2.38	0.00	2.38	0.53
B3	50	0	0.05	1.16	0.00	1.16	0.33
B4	50	0	0.05	2.50	0.00	2.50	0.54
B5	50	0	0.05	1.89	0.00	1.89	0.46
D1	50	0.05	0	6.47	0.00	6.47	0.87
D2	50	0.05	0	3.66	0.00	3.66	0.67
D3	50	0.05	0	9.45	0.00	9.45	1.02
D4	50	0.05	0	9.21	0.00	9.21	1.01
D5	50	0.05	0	7.50	0.00	7.50	0.93
E1	50	0.05	0.05	6.04	0.00	6.04	0.85
E2	50	0.05	0.05	3.23	0.00	3.23	0.63
E3	50	0.05	0.05	2.56	0.00	2.56	0.55
E4	50	0.05	0.05	4.57	0.00	4.57	0.75
E5	50	0.05	0.05	2.81	0.00	2.81	0.58

Site: Delta Channel Date NDS in: 6/28/05 8:45 Date NDS out: 7/19/05 16:00

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	13.91	0.00	13.91	1.17
A2	10	0	0	10.61	0.00	10.61	1.06
A3	10	0	0	12.32	0.00	12.32	1.12
A4	10	0	0	8.91	0.00	8.91	1.00
B1	10	0	0.05	5.06	0.00	5.06	0.78
B2	10	0	0.05	5.55	0.00	5.55	0.82
B3	10	0	0.05	9.58	0.00	9.58	1.02
B4	10	0	0.05	6.34	0.00	6.34	0.87
D1	10	0.05	0	13.48	0.00	13.48	1.16
D2	10	0.05	0				
D3	10	0.05	0	13.60	0.00	13.60	1.16
D4	10	0.05	0	12.14	0.00	12.14	1.12
E1	10	0.05	0.05	18.79	0.00	18.79	1.30
E2	10	0.05	0.05				
E3	10	0.05	0.05	10.06	4.49	14.55	1.19
E4	10	0.05	0.05	19.09	0.00	19.09	1.30
B(circled)	10	0	0.05	2.81	0.00	2.81	0.58
B(circled)	10	0	0.05	5.43	0.00	5.43	0.81
B(circled)	10	0	0.05	5.06	0.00	5.06	0.78
E (circled)	10	0.05	0.05	4.88	9.74	14.61	1.19
E (circled)	10	0.05	0.05	10.19	0.00	10.19	1.05
E (circled)	10	0.05	0.05	3.84	10.67	14.52	1.19
A1	30	0	0	7.20	0.00	7.20	0.91
A2	30	0	0	7.38	0.00	7.38	0.92
A3	30	0	0	10.06	0.00	10.06	1.04
A4	30	0	0	7.99	0.00	7.99	0.95
B1	30	0	0.05	7.20	1.48	8.68	0.99
B2	30	0	0.05				
B3	30	0	0.05	9.27	0.00	9.27	1.01
B4	30	0	0.05	6.53	0.00	6.53	0.88
D1	30	0.05	0	6.95	2.13	9.09	1.00
D2	30	0.05	0	6.04	3.47	9.51	1.02
D3	30	0.05	0	6.89	0.00	6.89	0.90
D4	30	0.05	0	11.71	0.00	11.71	1.10
E1	30	0.05	0.05	5.55	4.98	10.53	1.06
E2	30	0.05	0.05	4.88	4.50	9.37	1.02
E3	30	0.05	0.05	2.32	8.43	10.75	1.07
E4	30	0.05	0.05	5.31	6.42	11.73	1.10
A1	50	0	0	2.20	0.21	2.41	0.53
A2	50	0	0	6.04	4.85	10.89	1.08
A3	50	0	0	10.00	0.38	10.38	1.06
A4	50	0	0	2.26	0.00	2.26	0.51
B1	50	0	0.05	3.35	0.00	3.35	0.64
B2	50	0	0.05	6.34	0.76	7.11	0.91
B3	50	0	0.05	8.72	0.00	8.72	0.99
B4	50	0	0.05	4.82	0.45	5.27	0.80
D1	50	0.05	0	3.05	2.21	5.26	0.80
D2	50	0.05	0	2.20	5.00	7.20	0.91
D3	50	0.05	0	3.29	0.35	3.64	0.67
D4	50	0.05	0	2.99	4.58	7.56	0.93
E1	50	0.05	0.05	9.15	0.00	9.15	1.01
E2	50	0.05	0.05	1.10	2.17	3.27	0.63
E3	50	0.05	0.05	0.79	7.40	8.19	0.96
E4	50	0.05	0.05	1.04	4.89	5.92	0.84

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	3.84	0.00	3.84	0.69
A2	10	0	0	7.26	0.00	7.26	0.92
A3	10	0	0	8.17	0.00	8.17	0.96
A4	10	0	0	6.40	0.00	6.40	0.87
B1	10	0	0.05	2.68	0.00	2.68	0.57
B2	10	0	0.05	3.35	0.00	3.35	0.64
B3	10	0	0.05	5.43	0.00	5.43	0.81
B4	10	0	0.05	4.70	0.00	4.70	0.76
D1	10	0.05	0	5.73	0.00	5.73	0.83
D2	10	0.05	0	6.47	0.00	6.47	0.87
D3	10	0.05	0	10.67	0.00	10.67	1.07
D4	10	0.05	0	8.72	0.00	8.72	0.99
E1	10	0.05	0.05	1.65	3.33	4.98	0.78
E2	10	0.05	0.05	2.01	0.00	2.01	0.48
E3	10	0.05	0.05	1.10	5.79	6.89	0.90
E4	10	0.05	0.05	1.46	2.61	4.07	0.71
A1	30	0	0	3.54	0.00	3.54	0.66
A2	30	0	0				
A3	30	0	0	8.05	0.00	8.05	0.96
A4	30	0	0	4.21	0.00	4.21	0.72
B1	30	0	0.05	4.39	0.18	4.57	0.75
B2	30	0	0.05	3.84	0.00	3.84	0.69
B3	30	0	0.05	3.84	0.00	3.84	0.69
B4	30	0	0.05	2.01	0.00	2.01	0.48
D1	30	0.05	0				
D2	30	0.05	0	11.59	0.00	11.59	1.10
D3	30	0.05	0	11.95	0.00	11.95	1.11
D4	30	0.05	0	17.14	0.00	17.14	1.26
E1	30	0.05	0.05	8.66	0.00	8.66	0.99
E2	30	0.05	0.05	2.62	0.35	2.97	0.60
E3	30	0.05	0.05	2.56	0.00	2.56	0.55
E4	30	0.05	0.05	4.88	0.94	5.82	0.83
A1	50	0	0	0.91	0.01	0.92	0.28
A2	50	0	0	1.22	0.12	1.34	0.37
A3	50	0	0	0.98	0.00	0.98	0.30
A4	50	0	0	1.34	0.00	1.34	0.37
B1	50	0	0.05	1.40	0.00	1.40	0.38
B2	50	0	0.05	0.43	0.21	0.63	0.21
B3	50	0	0.05	1.52	0.00	1.52	0.40
B4	50	0	0.05	0.55	0.00	0.55	0.19
D1	50	0.05	0	1.22	0.23	1.45	0.39
D2	50	0.05	0	3.42	0.00	3.42	0.64
D3	50	0.05	0	0.67	0.49	1.16	0.33
D4	50	0.05	0	0.91	0.39	1.30	0.36
E1	50	0.05	0.05	1.04	0.23	1.27	0.36
E2	50	0.05	0.05	0.91	1.45	2.37	0.53
E3	50	0.05	0.05	0.73	0.18	0.92	0.28
E4	50	0.05	0.05	0.30	0.15	0.46	0.16

Site: The Gap

Date NDS in: 5/16/05 15:45

Date NDS out: 6/6/05 12:45

Vial ID	Depth from surface (cm)	N conc	P conc	Volume Methanol	abs 665 nm A	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	10	0.759	20.98	0.00	20.98	1.34
A2	10	0	0	10	1.537				
A3	10	0	0	10	0.587	15.80	0.00	15.80	1.23
A4	10	0	0	10	0.853	20.13	0.00	20.13	1.32
B1	10	0	0.05	10	0.873	20.80	0.00	20.80	1.34
B2	10	0	0.05	10	0.801	22.26	0.00	22.26	1.37
B3	10	0	0.05	10	0.524	14.94	0.00	14.94	1.20
B4	10	0	0.05	10	0.489	13.42	0.00	13.42	1.16
D1	10	0.05	0	10	0.902	23.91	0.00	23.91	1.40
D2	10	0.05	0	10	0.577	11.83	0.94	12.77	1.14
D3	10	0.05	0	10	0.745	15.43	0.89	16.32	1.24
D4	10	0.05	0	10	1.001	23.97	0.00	23.97	1.40
E1	10	0.05	0.05	10	0.483				
E2	10	0.05	0.05	10	0.837	20.68	0.00	20.68	1.34
E3	10	0.05	0.05	10	0.892	17.44	2.90	20.34	1.33
E4	10	0.05	0.05	10	1.016	19.76	3.64	23.41	1.39
A1	30	0	0	10	0.354	9.94	0.00	9.94	1.04
A2	30	0	0	10	0.611	15.07	0.00	15.07	1.21
A3	30	0	0	10	0.464	12.87	0.00	12.87	1.14
A4	30	0	0	10	0.334	5.79	2.43	8.23	0.97
B1	30	0	0.05	10	0.357	7.44	0.35	7.79	0.94
B2	30	0	0.05	10	0.288	6.83	0.00	6.83	0.89
B3	30	0	0.05	10	0.302	9.15	0.00	9.15	1.01
B4	30	0	0.05	10	0.36	10.55	0.00	10.55	1.06
D1	30	0.05	0	10	0.487	11.28	0.00	11.28	1.09
D2	30	0.05	0	10	0.39				
D3	30	0.05	0	10	0.678	14.27	1.16	15.43	1.22
D4	30	0.05	0	10	0.712	20.49	0.00	20.49	1.33
E1	30	0.05	0.05	10	0.586	16.65	0.00	16.65	1.25
E2	30	0.05	0.05	10	0.42	12.38	0.00	12.38	1.13
E3	30	0.05	0.05	10	0.659	15.00	0.00	15.00	1.20
E4	30	0.05	0.05	10	0.496	15.92	0.00	15.92	1.23
A1	50	0	0	10	0.121	2.44	0.25	2.69	0.57
A2	50	0	0	10	0.182	4.21	0.00	4.21	0.72
A3	50	0	0	10	0.163	3.78	0.00	3.78	0.68
A4	50	0	0	10	0.206	5.25	0.00	5.25	0.80
B1	50	0	0.05	10	0.087	1.59	0.22	1.80	0.45
B2	50	0	0.05	10	0.062	0.06	1.70	1.76	0.44
B3	50	0	0.05	10	0.103	1.89	0.40	2.29	0.52
B4	50	0	0.05	10	0.153	3.35	0.00	3.35	0.64
D1	50	0.05	0	10	0.121	2.74	0.00	2.74	0.57
D2	50	0.05	0	10	0.097	1.46	0.82	2.28	0.52
D3	50	0.05	0	10	0.116	2.68	0.00	2.68	0.57
D4	50	0.05	0	10	0.126	2.93	0.00	2.93	0.59
E1	50	0.05	0.05	10	0.12	2.93	0.00	2.93	0.59
E2	50	0.05	0.05	10	0.071	1.34	0.07	1.42	0.38
E3	50	0.05	0.05	10	0.111	2.81	0.00	2.81	0.58
E4	50	0.05	0.05	10	0.101	2.32	0.00	2.32	0.52

Site: The Gap Date NDS in: 6/6/05 12:45 Date NDS out: 6/28/05 10:45

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	11.77	0.00	11.77	1.11
A2	10	0	0	19.94	0.00	19.94	1.32
A3	10	0	0	11.65	0.00	11.65	1.10
A4	10	0	0				
A5	10	0	0	29.15	5.72	34.87	1.55
B1	10	0	0.05	11.71	0.00	11.71	1.10
B2	10	0	0.05	15.07	0.00	15.07	1.21
B3	10	0	0.05				
B4	10	0	0.05	12.99	0.00	12.99	1.15
B5	10	0	0.05				
D1	10	0.05	0	16.41	3.35	19.76	1.32
D2	10	0.05	0				
D3	10	0.05	0	25.07	0.00	25.07	1.42
D4	10	0.05	0	23.73	0.00	23.73	1.39
D5	10	0.05	0	20.01	0.00	20.01	1.32
E1	10	0.05	0.05	24.52	0.00	24.52	1.41
E2	10	0.05	0.05	32.39	0.00	32.39	1.52
E3	10	0.05	0.05				
E4	10	0.05	0.05	23.60	0.00	23.60	1.39
E5	10	0.05	0.05	21.65	0.00	21.65	1.36
A1	30	0	0	11.71	11.89	23.60	1.39
A2	30	0	0	24.09	0.00	24.09	1.40
A3	30	0	0	20.80	0.00	20.80	1.34
A4	30	0	0	18.79	0.00	18.79	1.30
A5	30	0	0	lost	lost	lost	lost
B1	30	0	0.05	6.28	0.00	6.28	0.86
B2	30	0	0.05	9.39	0.00	9.39	1.02
B3	30	0	0.05	9.76	0.00	9.76	1.03
B4	30	0	0.05				
B5	30	0	0.05	11.65	0.00	11.65	1.10
D1	30	0.05	0	26.04	0.00	26.04	1.43
D2	30	0.05	0	15.68	0.00	15.68	1.22
D3	30	0.05	0	23.42	0.00	23.42	1.39
D4	30	0.05	0	19.76	0.00	19.76	1.32
D5	30	0.05	0	20.19	0.00	20.19	1.33
E1	30	0.05	0.05	21.53	0.00	21.53	1.35
E2	30	0.05	0.05	21.29	0.00	21.29	1.35
E3	30	0.05	0.05	18.18	0.00	18.18	1.28
E4	30	0.05	0.05	19.09	0.00	19.09	1.30
E5	30	0.05	0.05	18.60	0.00	18.60	1.29
A1	50	0	0	12.93	0.00	12.93	1.14
A2	50	0	0	9.82	0.00	9.82	1.03
A3	50	0	0	12.26	0.00	12.26	1.12
A4	50	0	0	9.09	0.00	9.09	1.00
A5	50	0	0				
B1	50	0	0.05	13.05	0.00	13.05	1.15
B2	50	0	0.05	11.28	0.00	11.28	1.09
B3	50	0	0.05	8.54	0.00	8.54	0.98
B4	50	0	0.05				
B5	50	0	0.05	6.59	0.00	6.59	0.88
D1	50	0.05	0	11.95	3.10	15.05	1.21
D2	50	0.05	0	17.57	0.00	17.57	1.27
D3	50	0.05	0				
D4	50	0.05	0	14.52	0.00	14.52	1.19
D5	50	0.05	0	12.02	0.00	12.02	1.11
E1	50	0.05	0.05	10.67	3.41	14.09	1.18
E2	50	0.05	0.05	8.97	0.00	8.97	1.00
E3	50	0.05	0.05	10.67	0.00	10.67	1.07
E4	50	0.05	0.05	14.15	0.00	14.15	1.18
E5	50	0.05	0.05				

Site: The Gap | *Date NDS in:* 6/28/05 10:45 | Date NDS out: 7/19/05 12:30

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	6.34	0.00	6.34	0.87
A2	10	0	0				
A3	10	0	0	7.26	0.00	7.26	0.92
A4	10	0	0	10.37	0.00	10.37	1.06
B1	10	0	0.05	5.86	0.00	5.86	0.84
B2	10	0	0.05	3.54	0.00	3.54	0.66
B3	10	0	0.05	7.26	0.00	7.26	0.92
B4	10	0	0.05	3.90	0.17	4.07	0.71
D1	10	0.05	0	19.40	0.00	19.40	1.31
D2	10	0.05	0	18.66	0.00	18.66	1.29
D3	10	0.05	0	10.55	8.94	19.50	1.31
D4	10	0.05	0	22.75	0.00	22.75	1.38
E1	10	0.05	0.05	13.17	0.00	13.17	1.15
E2	10	0.05	0.05	7.01	10.28	17.29	1.26
E3	10	0.05	0.05	5.98	10.60	16.58	1.24
E4	10	0.05	0.05	13.48	0.00	13.48	1.16
A1	30	0	0	7.81	0.00	7.81	0.94
A2	30	0	0	11.89	0.00	11.89	1.11
A3	30	0	0	8.78	0.00	8.78	0.99
A4	30	0	0	7.75	0.00	7.75	0.94
B1	30	0	0.05	4.51	0.00	4.51	0.74
B2	30	0	0.05				
B3	30	0	0.05	5.43	0.00	5.43	0.81
B4	30	0	0.05	4.27	0.00	4.27	0.72
D1	30	0.05	0	8.78	0.00	8.78	0.99
D2	30	0.05	0	7.08	0.00	7.08	0.91
D3	30	0.05	0	3.29	3.49	6.78	0.89
D4	30	0.05	0	10.12	0.00	10.12	1.05
E1	30	0.05	0.05	8.23	0.00	8.23	0.97
E2	30	0.05	0.05	8.66	0.00	8.66	0.99
E3	30	0.05	0.05	6.34	0.00	6.34	0.87
E4	30	0.05	0.05	lost	lost	lost	lost
A1	50	0	0				
A2	50	0	0	3.78	0.94	4.72	0.76
A3	50	0	0	lost	lost	lost	lost
A4	50	0	0	6.95	2.55	9.50	1.02
B1	50	0	0.05	4.39	0.04	4.43	0.73
B2	50	0	0.05	2.44	0.01	2.45	0.54
B3	50	0	0.05	1.71	0.30	2.01	0.48
B4	50	0	0.05	1.77	0.28	2.05	0.48
D1	50	0.05	0	3.84	1.44	5.28	0.80
D2	50	0.05	0	5.18	0.79	5.97	0.84
D3	50	0.05	0	1.22	3.50	4.72	0.76
D4	50	0.05	0	9.45	0.00	9.45	1.02
E1	50	0.05	0.05	2.56	0.44	3.00	0.60
E2	50	0.05	0.05	1.65	3.67	5.32	0.80
E3	50	0.05	0.05	0.61	2.44	3.05	0.61
E4	50	0.05	0.05	0.67	4.14	4.81	0.76

Site: The Gap Date NDS in: 7/19/05 12:30 Date NDS out: 8/9/05 10:00

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	2.38	3.41	5.79	0.83
A2	10	0	0	lost	lost	lost	lost
A3	10	0	0	4.03	0.00	4.03	0.70
A4	10	0	0	3.35	0.00	3.35	0.64
B1	10	0	0.05	3.60	0.00	3.60	0.66
B2	10	0	0.05	3.11	0.00	3.11	0.61
B3	10	0	0.05	3.54	0.00	3.54	0.66
B4	10	0	0.05	2.62	0.55	3.18	0.62
D1	10	0.05	0	7.44	0.00	7.44	0.93
D2	10	0.05	0	3.35	0.00	3.35	0.64
D3	10	0.05	0	4.27	0.00	4.27	0.72
D4	10	0.05	0	5.43	0.00	5.43	0.81
E1	10	0.05	0.05	1.52	4.00	5.52	0.81
E2	10	0.05	0.05	0.67	4.76	5.43	0.81
E3	10	0.05	0.05	2.56	3.09	5.66	0.82
E4	10	0.05	0.05	7.99	2.71	10.70	1.07
A1	30	0	0	9.39	0.00	9.39	1.02
A2	30	0	0	8.48	0.00	8.48	0.98
A3	30	0	0	6.10	0.00	6.10	0.85
A4	30	0	0	6.40	0.00	6.40	0.87
B1	30	0	0.05	4.03	0.00	4.03	0.70
B2	30	0	0.05	8.42	0.00	8.42	0.97
B3	30	0	0.05	4.57	0.17	4.75	0.76
B4	30	0	0.05	3.29	0.38	3.68	0.67
D1	30	0.05	0	13.85	0.00	13.85	1.17
D2	30	0.05	0	11.10	0.00	11.10	1.08
D3	30	0.05	0	8.72	0.00	8.72	0.99
D4	30	0.05	0	6.10	0.00	6.10	0.85
E1	30	0.05	0.05	4.64	1.39	6.02	0.85
E2	30	0.05	0.05	1.52	9.72	11.25	1.09
E3	30	0.05	0.05	9.51	3.13	12.64	1.13
E4	30	0.05	0.05	5.61	1.51	7.12	0.91
A1	50	0	0	9.76	0.00	9.76	1.03
A2	50	0	0	6.16	0.00	6.16	0.85
A3	50	0	0	7.44	0.01	7.45	0.93
A4	50	0	0	5.61	0.00	5.61	0.82
B1	50	0	0.05	2.38	0.31	2.69	0.57
B2	50	0	0.05	3.35	0.22	3.58	0.66
B3	50	0	0.05	2.68	0.18	2.87	0.59
B4	50	0	0.05	2.50	0.15	2.66	0.56
D1	50	0.05	0	6.95	0.00	6.95	0.90
D2	50	0.05	0	2.26	0.00	2.26	0.51
D3	50	0.05	0	3.11	0.01	3.12	0.61
D4	50	0.05	0	5.37	0.00	5.37	0.80
E1	50	0.05	0.05	0.43	4.14	4.56	0.75
E2	50	0.05	0.05	0.61	2.61	3.22	0.63
E3	50	0.05	0.05	3.96	3.01	6.97	0.90
E4	50	0.05	0.05	1.04	9.27	10.30	1.05

Site:	Portage Creek S	Date NDS in:	5/16/05 12:00		Date NDS out:	6/6/05 14:15	
Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	2.81	0.00	2.81	0.58
A2	10	0	0	3.42	0.00	3.42	0.64
A3	10	0	0	2.50	0.00	2.50	0.54
A4	10	0	0	2.32	0.54	2.86	0.59
B1	10	0	0.05	2.38	0.00	2.38	0.53
B2	10	0	0.05	2.87	0.00	2.87	0.59
B3	10	0	0.05	4.09	0.00	4.09	0.71
B4	10	0	0.05	5.25	0.00	5.25	0.80
D1	10	0.05	0	27.02	0.00	27.02	1.45
D2	10	0.05	0	13.42	0.00	13.42	1.16
D3	10	0.05	0	18.05	0.00	18.05	1.28
D4	10	0.05	0	17.87	0.00	17.87	1.28
E1	10	0.05	0.05	18.54	0.00	18.54	1.29
E2	10	0.05	0.05	15.13	0.00	15.13	1.21
E3	10	0.05	0.05	19.03	0.00	19.03	1.30
E4	10	0.05	0.05	10.06	0.00	10.06	1.04
A1	30	0	0	2.87	0.00	2.87	0.59
A2	30	0	0	2.87	0.07	2.94	0.60
A3	30	0	0	3.11	0.00	3.11	0.61
A4	30	0	0	3.60	0.00	3.60	0.66
B1	30	0	0.05	0.91	0.01	0.92	0.28
B2	30	0	0.05	1.40	0.00	1.40	0.38
B3	30	0	0.05	1.65	0.00	1.65	0.42
B4	30	0	0.05	1.77	0.00	1.77	0.44
D1	30	0.05	0	9.09	0.00	9.09	1.00
D2	30	0.05	0	6.47	0.00	6.47	0.87
D3	30	0.05	0	6.22	0.00	6.22	0.86
D4	30	0.05	0	10.43	0.00	10.43	1.06
E1	30	0.05	0.05	9.33	0.00	9.33	1.01
E2	30	0.05	0.05	0.67	5.52	6.19	0.86
E3	30	0.05	0.05	8.17	0.00	8.17	0.96
E4	30	0.05	0.05	8.05	0.00	8.05	0.96
A1	50	0	0	0.30	2.77	3.08	0.61
A2	50	0	0	0.06	2.73	2.79	0.58
A3	50	0	0	0.00	2.14	2.14	0.50
A4	50	0	0	0.18	4.30	4.49	0.74
B1	50	0	0.05	0.61	0.54	1.15	0.33
B2	50	0	0.05	0.79	0.23	1.02	0.31
B3	50	0	0.05	0.79	0.13	0.92	0.28
B4	50	0	0.05	1.52	0.17	1.70	0.43
D1	50	0.05	0	10.80	0.00	10.80	1.07
D2	50	0.05	0	5.79	0.00	5.79	0.83
D3	50	0.05	0	4.51	1.40	5.92	0.84
D4	50	0.05	0	8.66	0.00	8.66	0.99
E1	50	0.05	0.05	5.55	0.00	5.55	0.82
E2	50	0.05	0.05	4.39	0.00	4.39	0.73
E3	50	0.05	0.05	2.26	1.36	3.61	0.66
E4	50	0.05	0.05	8.05	0.00	8.05	0.96

Site: Portage Creek S Date NDS in: 6/6/05 14:15 Date NDS out: 6/28/05 13:00

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	26.41	0.00	26.41	1.44
A2	10	0	0	29.22	0.00	29.22	1.48
A3	10	0	0	20.62	6.05	26.67	1.44
A4	10	0	0	19.88	4.56	24.44	1.41
A5	10	0	0	21.53	0.00	21.53	1.35
B1	10	0	0.05	19.88	0.00	19.88	1.32
B2	10	0	0.05	19.03	0.00	19.03	1.30
B3	10	0	0.05	25.01	0.00	25.01	1.42
B4	10	0	0.05	13.05	0.00	13.05	1.15
B5	10	0	0.05				
D1	10	0.05	0	35.99	0.00	35.99	1.57
D2	10	0.05	0	19.27	7.70	26.98	1.45
D3	10	0.05	0	20.92	12.58	33.50	1.54
D4	10	0.05	0	28.30	1.96	30.26	1.50
D5	10	0.05	0	24.46	0.00	24.46	1.41
E1	10	0.05	0.05	19.46	0.00	19.46	1.31
E2	10	0.05	0.05	23.48	0.00	23.48	1.39
E3	10	0.05	0.05	26.72	0.00	26.72	1.44
E4	10	0.05	0.05	32.02	7.00	39.02	1.60
E5	10	0.05	0.05	19.94	7.26	27.20	1.45
A1	30	0	0	18.30	0.00	18.30	1.29
A2	30	0	0	17.26	0.00	17.26	1.26
A3	30	0	0	22.08	0.00	22.08	1.36
A4	30	0	0	19.40	0.00	19.40	1.31
A5	30	0	0	26.53	0.00	26.53	1.44
B1	30	0	0.05	16.10	0.00	16.10	1.23
B2	30	0	0.05	9.27	0.00	9.27	1.01
B3	30	0	0.05				
B4	30	0	0.05	9.70	0.00	9.70	1.03
B5	30	0	0.05	13.17	0.00	13.17	1.15
D1	30	0.05	0	25.31	0.00	25.31	1.42
D2	30	0.05	0	36.84	0.00	36.84	1.58
D3	30	0.05	0	20.62	0.00	20.62	1.33
D4	30	0.05	0	27.20	0.00	27.20	1.45
D5	30	0.05	0	24.89	0.00	24.89	1.41
E1	30	0.05	0.05	25.37	0.00	25.37	1.42
E2	30	0.05	0.05	27.63	0.00	27.63	1.46
E3	30	0.05	0.05	14.58	0.45	15.02	1.20
E4	30	0.05	0.05	20.37	0.00	20.37	1.33
E5	30	0.05	0.05	lost	lost	lost	lost
A1	50	0	0	18.66	0.00	18.66	1.29
A2	50	0	0	14.09	2.26	16.35	1.24
A3	50	0	0	20.31	0.00	20.31	1.33
A4	50	0	0	lost	lost	lost	lost
A5	50	0	0	15.37	0.00	15.37	1.21
B1	50	0	0.05	11.83	0.00	11.83	1.11
B2	50	0	0.05	11.41	0.00	11.41	1.09
B3	50	0	0.05	8.42	0.00	8.42	0.97
B4	50	0	0.05	8.23	0.00	8.23	0.97
B5	50	0	0.05	8.17	0.00	8.17	0.96
D1	50	0.05	0	4.33	10.85	15.19	1.21
D2	50	0.05	0	14.39	0.00	14.39	1.19
D3	50	0.05	0	11.83	0.00	11.83	1.11
D4	50	0.05	0	18.60	0.00	18.60	1.29
D5	50	0.05	0	10.67	4.72	15.40	1.21
E1	50	0.05	0.05	15.07	0.00	15.07	1.21
E2	50	0.05	0.05	21.04	0.00	21.04	1.34
E3	50	0.05	0.05	13.97	0.00	13.97	1.18
E4	50	0.05	0.05	12.38	0.00	12.38	1.13
E5	50	0.05	0.05	19.03	0.00	19.03	1.30

Site:	Portage Creek S	Date NDS in:	5/16/05 12:00	Date NDS out:	6/6/05 14:15

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	8.84	0.00	8.84	0.99
A2	10	0	0	6.83	0.00	6.83	0.89
A3	10	0	0	10.12	0.00	10.12	1.05
A4	10	0	0	12.20	0.00	12.20	1.12
B1	10	0	0.05	2.38	4.31	6.69	0.89
B2	10	0	0.05	3.78	0.01	3.79	0.68
B3	10	0	0.05	1.89	1.06	2.95	0.60
B4	10	0	0.05	5.98	0.00	5.98	0.84
D1	10	0.05	0	lost	lost	lost	lost
D2	10	0.05	0	24.82	0.00	24.82	1.41
D3	10	0.05	0	22.20	0.00	22.20	1.37
D4	10	0.05	0	15.68	0.00	15.68	1.22
E1	10	0.05	0.05	5.73	4.87	10.61	1.06
E2	10	0.05	0.05	11.77	0.00	11.77	1.11
E3	10	0.05	0.05	2.68	8.70	11.38	1.09
E4	10	0.05	0.05	3.05	5.69	8.74	0.99
A1	30	0	0				
A2	30	0	0	12.50	0.00	12.50	1.13
A3	30	0	0	9.64	0.00	9.64	1.03
A4	30	0	0	15.68	0.00	15.68	1.22
B1	30	0	0.05	5.98	0.00	5.98	0.84
B2	30	0	0.05	6.16	0.00	6.16	0.85
B3	30	0	0.05	5.37	0.00	5.37	0.80
B4	30	0	0.05	7.32	0.00	7.32	0.92
D1	30	0.05	0	11.47	0.00	11.47	1.10
D2	30	0.05	0	14.94	0.00	14.94	1.20
D3	30	0.05	0				
D4	30	0.05	0	9.51	0.00	9.51	1.02
E1	30	0.05	0.05	13.54	0.00	13.54	1.16
E2	30	0.05	0.05	13.72	0.00	13.72	1.17
E3	30	0.05	0.05	3.23	6.30	9.54	1.02
E4	30	0.05	0.05				
A1	50	0	0	4.03	0.00	4.03	0.70
A2	50	0	0	3.23	0.00	3.23	0.63
A3	50	0	0	4.76	0.00	4.76	0.76
A4	50	0	0	2.68	0.00	2.68	0.57
B1	50	0	0.05	0.98	0.60	1.58	0.41
B2	50	0	0.05	2.68	0.00	2.68	0.57
B3	50	0	0.05	1.10	3.97	5.06	0.78
B4	50	0	0.05	1.89	3.09	4.98	0.78
D1	50	0.05	0	21.35	0.00	21.35	1.35
D2	50	0.05	0	5.00	0.03	5.03	0.78
D3	50	0.05	0	2.56	1.44	4.00	0.70
D4	50	0.05	0	16.53	0.00	16.53	1.24
E1	50	0.05	0.05	8.78	0.00	8.78	0.99
E2	50	0.05	0.05	6.47	0.00	6.47	0.87
E3	50	0.05	0.05	2.99	0.71	3.70	0.67
E4	50	0.05	0.05	2.32	0.44	2.75	0.57

Site:	Portage Creek S	Date NDS in:	6/28/05 8:45	Date NDS out:	7/19/05 16:00

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	1.16	0.91	2.07	0.49
A2	10	0	0	4.21	0.00	4.21	0.72
A3	10	0	0	3.90	0.00	3.90	0.69
A4	10	0	0	3.05	0.00	3.05	0.61
B1	10	0	0.05	2.44	0.00	2.44	0.54
B2	10	0	0.05	2.56	0.00	2.56	0.55
B3	10	0	0.05	3.90	0.00	3.90	0.69
B4	10	0	0.05	4.76	0.00	4.76	0.76
D1	10	0.05	0	11.10	0.00	11.10	1.08
D2	10	0.05	0	18.60	0.00	18.60	1.29
D3	10	0.05	0	15.19	0.00	15.19	1.21
D4	10	0.05	0	7.87	0.00	7.87	0.95
E1	10	0.05	0.05	9.88	0.00	9.88	1.04
E2	10	0.05	0.05	11.47	0.00	11.47	1.10
E3	10	0.05	0.05				
E4	10	0.05	0.05	16.04	0.00	16.04	1.23
A1	30	0	0	1.71	1.68	3.39	0.64
A2	30	0	0				
A3	30	0	0	3.60	0.00	3.60	0.66
A4	30	0	0	5.00	0.00	5.00	0.78
B1	30	0	0.05	1.40	1.98	3.38	0.64
B2	30	0	0.05	2.01	0.73	2.74	0.57
B3	30	0	0.05	4.70	0.00	4.70	0.76
B4	30	0	0.05	1.46	0.99	2.45	0.54
D1	30	0.05	0	17.14	0.00	17.14	1.26
D2	30	0.05	0	21.35	0.00	21.35	1.35
D3	30	0.05	0	19.09	0.00	19.09	1.30
D4	30	0.05	0	23.79	0.00	23.79	1.39
E1	30	0.05	0.05	11.34	0.00	11.34	1.09
E2	30	0.05	0.05	5.37	0.92	6.29	0.86
E3	30	0.05	0.05	5.98	0.00	5.98	0.84
E4	30	0.05	0.05	10.73	0.00	10.73	1.07
A1	50	0	0	0.73	0.01	0.74	0.24
A2	50	0	0	0.79	0.40	1.19	0.34
A3	50	0	0	1.28	0.00	1.28	0.36
A4	50	0	0	2.07	0.16	2.23	0.51
B1	50	0	0.05	0.61	0.34	0.95	0.29
B2	50	0	0.05	0.37	0.13	0.49	0.17
B3	50	0	0.05	0.24	0.38	0.63	0.21
B4	50	0	0.05	0.24	0.21	0.45	0.16
D1	50	0.05	0	0.43	0.83	1.25	0.35
D2	50	0.05	0	1.28	0.34	1.62	0.42
D3	50	0.05	0	4.33	0.00	4.33	0.73
D4	50	0.05	0	1.89	0.95	2.85	0.58
E1	50	0.05	0.05	0.30	1.94	2.25	0.51
E2	50	0.05	0.05	1.46	0.30	1.76	0.44
E3	50	0.05	0.05	0.73	0.70	1.43	0.39
E4	50	0.05	0.05	2.44	0.90	3.34	0.64

Site: Center Marsh Date NDS in: 5/16/05 17:15 Date NDS out: 6/7/05 11:50

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	10.80	0.00	10.80	1.07
A2	10	0	0	6.59	0.00	6.59	0.88
A3	10	0	0	6.34	0.00	6.34	0.87
A4	10	0	0	9.03	0.00	9.03	1.00
B1	10	0	0.05	2.20	0.00	2.20	0.50
B2	10	0	0.05	5.92	0.00	5.92	0.84
B3	10	0	0.05	2.13	0.00	2.13	0.50
B4	10	0	0.05	8.54	0.00	8.54	0.98
D1	10	0.05	0	lost	lost	lost	lost
D2	10	0.05	0	22.02	0.00	22.02	1.36
D3	10	0.05	0	25.31	0.00	25.31	1.42
D4	10	0.05	0	16.04	0.00	16.04	1.23
E1	10	0.05	0.05	17.93	0.00	17.93	1.28
E2	10	0.05	0.05	10.12	0.00	10.12	1.05
E3	10	0.05	0.05	16.77	0.00	16.77	1.25
E4	10	0.05	0.05	12.75	0.00	12.75	1.14
A1	30	0	0	3.23	0.00	3.23	0.63
A2	30	0	0	3.11	0.60	3.71	0.67
A3	30	0	0	3.72	0.00	3.72	0.67
A4	30	0	0				
B1	30	0	0.05	1.22	0.26	1.48	0.39
B2	30	0	0.05	1.34	0.01	1.35	0.37
B3	30	0	0.05	1.59	0.22	1.80	0.45
B4	30	0	0.05	1.22	0.33	1.55	0.41
D1	30	0.05	0	18.30	0.00	18.30	1.29
D2	30	0.05	0	21.90	0.00	21.90	1.36
D3	30	0.05	0	12.93	0.00	12.93	1.14
D4	30	0.05	0	15.43	0.00	15.43	1.22
E1	30	0.05	0.05	11.71	0.00	11.71	1.10
E2	30	0.05	0.05	15.25	0.00	15.25	1.21
E3	30	0.05	0.05	10.25	0.00	10.25	1.05
E4	30	0.05	0.05	10.37	0.00	10.37	1.06
A1	50	0	0	3.96	0.08	4.04	0.70
A2	50	0	0	4.57	0.17	4.75	0.76
A3	50	0	0	3.42	0.06	3.47	0.65
A4	50	0	0	4.27	0.00	4.27	0.72
B1	50	0	0.05	1.52	0.00	1.52	0.40
B2	50	0	0.05	1.34	0.21	1.55	0.41
B3	50	0	0.05				
B4	50	0	0.05	1.65	0.33	1.98	0.47
D1	50	0.05	0	13.42	0.00	13.42	1.16
D2	50	0.05	0	7.14	0.00	7.14	0.91
D3	50	0.05	0	9.82	0.00	9.82	1.03
D4	50	0.05	0	9.58	0.00	9.58	1.02
E1	50	0.05	0.05	3.23	0.00	3.23	0.63
E2	50	0.05	0.05	3.66	0.00	3.66	0.67
E3	50	0.05	0.05	3.60	0.02	3.62	0.66
E4	50	0.05	0.05	5.98	0.00	5.98	0.84

Site: Center Marsh | Date NDS in: 6/7/05 11:50 | Date NDS out: 6/27/05 10:30

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	17.99	0.00	17.99	1.28
A2	10	0	0	18.24	0.00	18.24	1.28
A3	10	0	0	20.01	0.00	20.01	1.32
A4	10	0	0				
A5	10	0	0	15.31	0.00	15.31	1.21
B1	10	0	0.05	17.57	0.00	17.57	1.27
B2	10	0	0.05	16.29	0.00	16.29	1.24
B3	10	0	0.05				
B4	10	0	0.05	10.61	0.00	10.61	1.06
B5	10	0	0.05	14.52	0.00	14.52	1.19
D1	10	0.05	0	22.02	0.00	22.02	1.36
D2	10	0.05	0	1.46	28.36	29.83	1.49
D3	10	0.05	0	23.97	0.00	23.97	1.40
D4	10	0.05	0				
D5	10	0.05	0	24.34	0.00	24.34	1.40
E1	10	0.05	0.05	21.53	0.00	21.53	1.35
E2	10	0.05	0.05	13.17	0.00	13.17	1.15
E3	10	0.05	0.05	14.70	0.00	14.70	1.20
E4	10	0.05	0.05	19.40	0.00	19.40	1.31
E5	10	0.05	0.05	19.94	0.00	19.94	1.32
A1	30	0	0	16.18	0.00	16.18	1.24
A2	30	0	0	19.40	0.00	19.40	1.31
A3	30	0	0	19.88	0.00	19.88	1.32
A4	30	0	0	19.94	0.00	19.94	1.32
A5	30	0	0	14.39	0.00	14.39	1.19
B1	30	0	0.05	13.78	0.00	13.78	1.17
B2	30	0	0.05	12.20	0.00	12.20	1.12
B3	30	0	0.05	9.88	0.00	9.88	1.04
B4	30	0	0.05	8.60	0.00	8.60	0.98
B5	30	0	0.05	10.55	0.00	10.55	1.06
D1	30	0.05	0	10.12	0.00	10.12	1.05
D2	30	0.05	0	19.52	0.00	19.52	1.31
D3	30	0.05	0	25.86	0.00	25.86	1.43
D4	30	0.05	0	14.46	0.00	14.46	1.19
D5	30	0.05	0	19.09	0.00	19.09	1.30
E1	30	0.05	0.05	13.24	0.00	13.24	1.15
E2	30	0.05	0.05	10.67	0.00	10.67	1.07
E3	30	0.05	0.05	14.46	0.00	14.46	1.19
E4	30	0.05	0.05				
E5	30	0.05	0.05	16.96	0.00	16.96	1.25
A1	50	0	0	9.27	0.00	9.27	1.01
A2	50	0	0	9.58	0.00	9.58	1.02
A3	50	0	0	9.58	0.00	9.58	1.02
A4	50	0	0	12.44	0.00	12.44	1.13
A5	50	0	0	11.16	0.00	11.16	1.08
B1	50	0	0.05	5.00	0.00	5.00	0.78
B2	50	0	0.05	4.88	0.00	4.88	0.77
B3	50	0	0.05	4.03	0.00	4.03	0.70
B4	50	0	0.05	4.33	0.00	4.33	0.73
B5	50	0	0.05	4.45	0.00	4.45	0.74
D1	50	0.05	0	7.81	0.00	7.81	0.94
D2	50	0.05	0	0.43	19.44	19.87	1.32
D3	50	0.05	0	0.06	11.97	12.03	1.12
D4	50	0.05	0	0.06	20.49	20.55	1.33
D5	50	0.05	0	0.00	12.76	12.76	1.14
E1	50	0.05	0.05	9.33	0.00	9.33	1.01
E2	50	0.05	0.05	6.28	4.03	10.31	1.05
E3	50	0.05	0.05	4.70	0.00	4.70	0.76
E4	50	0.05	0.05	9.21	0.00	9.21	1.01
E5	50	0.05	0.05	3.23	0.00	3.23	0.63

Site: Center Marsh | Date NDS in: 6/27/05 10:30 | Date NDS out: 7/19/05 12:15

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	10.55	0.00	10.55	1.06
A2	10	0	0	6.47	0.00	6.47	0.87
A3	10	0	0	9.88	0.00	9.88	1.04
A4	10	0	0	10.80	0.00	10.80	1.07
B1	10	0	0.05	6.28	0.00	6.28	0.86
B2	10	0	0.05	9.45	0.00	9.45	1.02
B3	10	0	0.05	3.29	6.42	9.71	1.03
B4	10	0	0.05	6.16	0.73	6.89	0.90
D1	10	0.05	0	13.17	2.77	15.95	1.23
D2	10	0.05	0				
D3	10	0.05	0	16.04	0.00	16.04	1.23
D4	10	0.05	0	12.63	0.00	12.63	1.13
E1	10	0.05	0.05	4.39	3.73	8.12	0.96
E2	10	0.05	0.05	17.14	0.00	17.14	1.26
E3	10	0.05	0.05	9.03	2.57	11.59	1.10
E4	10	0.05	0.05	9.33	0.00	9.33	1.01
A1	30	0	0	5.18	0.00	5.18	0.79
A2	30	0	0	2.99	0.00	2.99	0.60
A3	30	0	0	7.87	0.00	7.87	0.95
A4	30	0	0	8.17	0.00	8.17	0.96
B1	30	0	0.05				
B2	30	0	0.05	5.73	0.00	5.73	0.83
B3	30	0	0.05	6.89	0.00	6.89	0.90
B4	30	0	0.05	4.15	0.00	4.15	0.71
D1	30	0.05	0	7.14	0.96	8.09	0.96
D2	30	0.05	0	2.56	4.96	7.52	0.93
D3	30	0.05	0	11.65	0.00	11.65	1.10
D4	30	0.05	0	2.44	5.73	8.17	0.96
E1	30	0.05	0.05	7.99	0.00	7.99	0.95
E2	30	0.05	0.05	9.15	0.00	9.15	1.01
E3	30	0.05	0.05	9.03	0.00	9.03	1.00
E4	30	0.05	0.05	5.18	0.00	5.18	0.79
A1	50	0	0	0.67	0.31	0.98	0.30
A2	50	0	0	0.55	0.22	0.77	0.25
A3	50	0	0	1.04	0.34	1.37	0.38
A4	50	0	0	0.49	0.04	0.53	0.18
B1	50	0	0.05	0.55	0.29	0.84	0.27
B2	50	0	0.05	lost	lost	lost	lost
B3	50	0	0.05	0.24	0.21	0.45	0.16
B4	50	0	0.05	0.18	0.48	0.66	0.22
D1	50	0.05	0	0.49	0.42	0.91	0.28
D2	50	0.05	0	0.06	0.49	0.55	0.19
D3	50	0.05	0	0.61	0.61	1.22	0.35
D4	50	0.05	0	0.12	0.47	0.59	0.20
E1	50	0.05	0.05	0.24	0.66	0.90	0.28
E2	50	0.05	0.05	0.12	0.50	0.62	0.21
E3	50	0.05	0.05	0.43	0.27	0.70	0.23
E4	50	0.05	0.05	0.91	0.59	1.51	0.40

Site: Center Marsh | *Date NDS in:* 7/19/05 12:15 | Date NDS out: 8/10/05 9:10

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	8.54	0.00	8.54	0.98
A2	10	0	0	7.56	0.00	7.56	0.93
A3	10	0	0	5.06	0.00	5.06	0.78
A4	10	0	0	7.56	0.00	7.56	0.93
B1	10	0	0.05	6.10	0.00	6.10	0.85
B2	10	0	0.05	4.45	1.64	6.09	0.85
B3	10	0	0.05	5.31	0.00	5.31	0.80
B4	10	0	0.05				
D1	10	0.05	0	1.40	2.95	4.35	0.73
D2	10	0.05	0	18.36	0.00	18.36	1.29
D3	10	0.05	0	5.98	0.95	6.92	0.90
D4	10	0.05	0	4.39	1.87	6.26	0.86
E1	10	0.05	0.05	9.09	0.00	9.09	1.00
E2	10	0.05	0.05	3.66	4.06	7.72	0.94
E3	10	0.05	0.05	5.61	0.00	5.61	0.82
E4	10	0.05	0.05	1.65	6.98	8.63	0.98
A1	30	0	0	4.09	0.00	4.09	0.71
A2	30	0	0	3.11	0.00	3.11	0.61
A3	30	0	0	4.03	0.00	4.03	0.70
A4	30	0	0	6.04	0.00	6.04	0.85
B1	30	0	0.05	4.88	0.00	4.88	0.77
B2	30	0	0.05	3.17	0.00	3.17	0.62
B3	30	0	0.05	2.56	0.00	2.56	0.55
B4	30	0	0.05	2.26	0.00	2.26	0.51
D1	30	0.05	0	1.59	1.63	3.22	0.62
D2	30	0.05	0	14.64	0.00	14.64	1.19
D3	30	0.05	0	2.32	2.33	4.65	0.75
D4	30	0.05	0	lost	lost	lost	lost
E1	30	0.05	0.05	5.00	0.00	5.00	0.78
E2	30	0.05	0.05	5.25	0.00	5.25	0.80
E3	30	0.05	0.05	5.18	0.00	5.18	0.79
E4	30	0.05	0.05	8.30	0.00	8.30	0.97
A1	50	0	0	3.05	0.00	3.05	0.61
A2	50	0	0	2.20	0.00	2.20	0.50
A3	50	0	0	2.13	0.00	2.13	0.50
A4	50	0	0	3.05	0.00	3.05	0.61
B1	50	0	0.05	1.77	0.00	1.77	0.44
B2	50	0	0.05	0.98	0.12	1.10	0.32
B3	50	0	0.05	2.13	0.00	2.13	0.50
B4	50	0	0.05	1.46	0.00	1.46	0.39
D1	50	0.05	0	13.48	0.00	13.48	1.16
D2	50	0.05	0	5.25	0.00	5.25	0.80
D3	50	0.05	0	6.10	0.00	6.10	0.85
D4	50	0.05	0	9.70	0.00	9.70	1.03
E1	50	0.05	0.05	4.39	0.00	4.39	0.73
E2	50	0.05	0.05	4.64	0.00	4.64	0.75
E3	50	0.05	0.05				
E4	50	0.05	0.05	3.90	0.00	3.90	0.69

Site:	East Blind Channel	Date NDS in:	5/16/05 18:00	Date NDS out:	6/7/05 10:30

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	3.66	0.30	3.96	0.70
A2	10	0	0	5.43	0.00	5.43	0.81
A3	10	0	0	5.31	0.00	5.31	0.80
A4	10	0	0	3.27	1.48	4.74	0.76
B1	10	0	0.05	2.56	0.16	2.73	0.57
B2	10	0	0.05	0.61	2.79	3.40	0.64
B3	10	0	0.05	2.93	0.50	3.43	0.65
B4	10	0	0.05	3.29	0.52	3.81	0.68
D1	10	0.05	0	18.12	0.00	18.12	1.28
D2	10	0.05	0	19.94	0.00	19.94	1.32
D3	10	0.05	0	21.77	0.00	21.77	1.36
D4	10	0.05	0	24.89	0.00	24.89	1.41
E1	10	0.05	0.05	15.55	0.00	15.55	1.22
E2	10	0.05	0.05	15.00	0.00	15.00	1.20
E3	10	0.05	0.05	11.53	0.00	11.53	1.10
E4	10	0.05	0.05	17.14	0.00	17.14	1.26
A1	30	0	0	3.54	0.28	3.82	0.68
A2	30	0	0	2.32	0.16	2.48	0.54
A3	30	0	0	4.64	0.08	4.71	0.76
A4	30	0	0	3.66	9.56	5.90	0.84
B1	30	0	0.05	2.13	0.13	2.27	0.51
B2	30	0	0.05	2.99	0.37	3.36	0.64
B3	30	0	0.05	2.26	0.36	2.61	0.56
B4	30	0	0.05	2.50	1.88	4.38	0.73
D1	30	0.05	0	15.07	0.00	15.07	1.21
D2	30	0.05	0	18.85	0.00	18.85	1.30
D3	30	0.05	0	24.28	0.00	24.28	1.40
D4	30	0.05	0	23.97	0.00	23.97	1.40
E1	30	0.05	0.05	18.60	0.00	18.60	1.29
E2	30	0.05	0.05	12.75	0.00	12.75	1.14
E3	30	0.05	0.05	12.93	0.00	12.93	1.14
E4	30	0.05	0.05	14.76	0.00	14.76	1.20
A1	50	0	0	4.39	0.00	4.39	0.73
A2	50	0	0	4.15	1.10	5.25	0.80
A3	50	0	0	3.78	2.25	6.03	0.85
A4	50	0	0	3.29	0.00	3.29	0.63
B1	50	0	0.05	2.68	0.05	2.73	0.57
B2	50	0	0.05	3.42	0.00	3.42	0.64
B3	50	0	0.05	1.71	0.34	2.05	0.48
B4	50	0	0.05	LOST	LOST	LOST	LOST
D1	50	0.05	0	18.05	0.00	18.05	1.28
D2	50	0.05	0				
D3	50	0.05	0	4.51	14.16	18.67	1.29
D4	50	0.05	0	LOST	LOST	LOST	LOST
E1	50	0.05	0.05	1.34	9.62	10.97	1.08
E2	50	0.05	0.05	2.07	7.43	9.50	1.02
E3	50	0.05	0.05	18.72	0.00	18.72	1.30
E4	50	0.05	0.05	9.45	0.00	9.45	1.02

Site:	East Blind Channel	*Date NDS in:*	6/7/05 10:30	Date NDS out:	6/28/05 10:15

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	13.54	0.00	13.54	1.16
A2	10	0	0				
A3	10	0	0	10.43	0.00	10.43	1.06
A4	10	0	0	8.36	0.00	8.36	0.97
A5	10	0	0	7.50	0.00	7.50	0.93
B1	10	0	0.05	7.44	0.00	7.44	0.93
B2	10	0	0.05	7.99	0.00	7.99	0.95
B3	10	0	0.05	12.81	0.00	12.81	1.14
B4	10	0	0.05				
B5	10	0	0.05	9.21	0.00	9.21	1.01
D1	10	0.05	0	23.32	0.00	23.32	1.39
D2	10	0.05	0	13.78	0.00	13.78	1.17
D3	10	0.05	0	18.12	0.00	18.12	1.28
D4	10	0.05	0	12.75	0.00	12.75	1.14
D5	10	0.05	0	15.61	0.00	15.61	1.22
E1	10	0.05	0.05	13.60	0.00	13.60	1.16
E2	10	0.05	0.05	14.58	0.00	14.58	1.19
E3	10	0.05	0.05				
E4	10	0.05	0.05	18.18	0.00	18.18	1.28
E5	10	0.05	0.05	16.96	0.00	16.96	1.25
A1	30	0	0	8.17	0.00	8.17	0.96
A2	30	0	0	4.70	0.00	4.70	0.76
A3	30	0	0	8.23	0.00	8.23	0.97
A4	30	0	0	6.04	0.00	6.04	0.85
A5	30	0	0				
B1	30	0	0.05	7.50	0.00	7.50	0.93
B2	30	0	0.05	6.47	0.00	6.47	0.87
B3	30	0	0.05	4.64	0.00	4.64	0.75
B4	30	0	0.05	3.23	0.00	3.23	0.63
B5	30	0	0.05	5.12	0.00	5.12	0.79
D1	30	0.05	0	11.77	0.00	11.77	1.11
D2	30	0.05	0	8.30	5.17	13.47	1.16
D3	30	0.05	0	11.95	1.31	13.26	1.15
D4	30	0.05	0	12.26	0.00	12.26	1.12
D5	30	0.05	0				
E1	30	0.05	0.05	3.35	3.63	6.99	0.90
E2	30	0.05	0.05	2.56	3.68	6.24	0.86
E3	30	0.05	0.05	5.73	0.00	5.73	0.83
E4	30	0.05	0.05				
E5	30	0.05	0.05	2.32	4.81	7.13	0.91
A1	50	0	0	6.53	0.00	6.53	0.88
A2	50	0	0	6.16	0.00	6.16	0.85
A3	50	0	0	6.28	0.00	6.28	0.86
A4	50	0	0	2.74	0.00	2.74	0.57
A5	50	0	0	3.96	0.87	4.83	0.77
B1	50	0	0.05	4.21	0.00	4.21	0.72
B2	50	0	0.05	3.84	0.00	3.84	0.69
B3	50	0	0.05	4.51	0.00	4.51	0.74
B4	50	0	0.05	2.99	0.00	2.99	0.60
B5	50	0	0.05	1.59	0.32	1.91	0.46
D1	50	0.05	0	16.65	0.00	16.65	1.25
D2	50	0.05	0	13.17	0.00	13.17	1.15
D3	50	0.05	0	4.27	6.85	11.12	1.08
D4	50	0.05	0	3.23	7.30	10.54	1.06
D5	50	0.05	0	lost	lost	lost	lost
E1	50	0.05	0.05	2.38	4.44	6.82	0.89
E2	50	0.05	0.05	lost	lost	lost	lost
E3	50	0.05	0.05	0.91	4.52	5.44	0.81
E4	50	0.05	0.05	1.65	3.60	5.25	0.80
E5	50	0.05	0.05	2.99	2.06	5.05	0.78

Site:	East Blind Channel	*Date NDS in:*	6/28/05 10:15	Date NDS out:	7/19/05 10:20

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	4.27	0.00	4.27	0.72
A2	10	0	0				
A3	10	0	0	7.38	0.00	7.38	0.92
A4	10	0	0	4.64	0.00	4.64	0.75
B1	10	0	0.05	3.23	0.00	3.23	0.63
B2	10	0	0.05	2.07	0.00	2.07	0.49
B3	10	0	0.05	4.39	0.00	4.39	0.73
B4	10	0	0.05				
D1	10	0.05	0	24.03	16.70	40.73	1.62
D2	10	0.05	0	7.62	0.00	7.62	0.94
D3	10	0.05	0	3.60	5.05	8.65	0.98
D4	10	0.05	0	11.41	0.00	11.41	1.09
E1	10	0.05	0.05	17.32	5.95	23.27	1.39
E2	10	0.05	0.05	6.16	0.00	6.16	0.85
E3	10	0.05	0.05	7.20	0.00	7.20	0.91
E4	10	0.05	0.05	3.05	0.00	3.05	0.61
A1	30	0	0	9.70	0.00	9.70	1.03
A2	30	0	0	9.09	0.00	9.09	1.00
A3	30	0	0	4.27	0.00	4.27	0.72
A4	30	0	0	5.67	0.00	5.67	0.82
B1	30	0	0.05	11.53	0.00	11.53	1.10
B2	30	0	0.05	19.76	5.44	25.20	1.42
B3	30	0	0.05	25.25	5.38	30.63	1.50
B4	30	0	0.05	11.59	0.00	11.59	1.10
D1	30	0.05	0	8.84	7.67	16.52	1.24
D2	30	0.05	0	11.83	0.00	11.83	1.11
D3	30	0.05	0				
D4	30	0.05	0	12.50	7.46	19.96	1.32
E1	30	0.05	0.05	11.83	0.00	11.83	1.11
E2	30	0.05	0.05	18.30	0.00	18.30	1.29
E3	30	0.05	0.05	3.29	2.45	5.75	0.83
E4	30	0.05	0.05	13.17	28.84	42.01	1.63
A1	50	0	0				
A2	50	0	0	12.63	0.00	12.63	1.13
A3	50	0	0	14.39	0.00	14.39	1.19
A4	50	0	0	19.33	0.23	19.57	1.31
B1	50	0	0.05	6.28	0.00	6.28	0.86
B2	50	0	0.05	14.70	0.00	14.70	1.20
B3	50	0	0.05	12.87	0.00	12.87	1.14
B4	50	0	0.05	5.98	0.00	5.98	0.84
D1	50	0.05	0	14.58	2.34	16.92	1.25
D2	50	0.05	0	7.99	0.00	7.99	0.95
D3	50	0.05	0	10.61	0.00	10.61	1.06
D4	50	0.05	0	7.26	5.80	13.06	1.15
E1	50	0.05	0.05	9.82	10.69	20.51	1.33
E2	50	0.05	0.05	20.86	1.65	22.51	1.37
E3	50	0.05	0.05	9.03	0.00	9.03	1.00
E4	50	0.05	0.05	14.03	0.00	14.03	1.18

Site:	East Blind Channel	*Date NDS in:*	7/19/05 10:20	Date NDS out:	8/10/05 9:55

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	2.20	0.00	2.20	0.50
A2	10	0	0	3.84	0.00	3.84	0.69
A3	10	0	0	1.52	0.00	1.52	0.40
A4	10	0	0	1.71	0.00	1.71	0.43
B1	10	0	0.05	0.43	0.00	0.43	0.15
B2	10	0	0.05	0.67	0.00	0.67	0.22
B3	10	0	0.05	1.40	0.02	1.42	0.38
B4	10	0	0.05	0.37	0.02	0.39	0.14
D1	10	0.05	0				
D2	10	0.05	0	4.82	0.00	4.82	0.76
D3	10	0.05	0	1.34	1.97	3.31	0.63
D4	10	0.05	0	9.94	0.00	9.94	1.04
E1	10	0.05	0.05	1.04	0.13	1.17	0.34
E2	10	0.05	0.05	1.46	0.00	1.46	0.39
E3	10	0.05	0.05	1.10	0.17	1.27	0.36
E4	10	0.05	0.05	1.65	0.00	1.65	0.42
A1	30	0	0	0.79	0.06	0.85	0.27
A2	30	0	0	1.04	0.00	1.04	0.31
A3	30	0	0	2.20	0.00	2.20	0.50
A4	30	0	0	1.22	0.00	1.22	0.35
B1	30	0	0.05	1.04	0.00	1.04	0.31
B2	30	0	0.05	0.79	0.13	0.92	0.28
B3	30	0	0.05	0.49	0.08	0.57	0.19
B4	30	0	0.05	0.73	0.08	0.81	0.26
D1	30	0.05	0	13.36	0.00	13.36	1.16
D2	30	0.05	0				
D3	30	0.05	0	6.59	0.00	6.59	0.88
D4	30	0.05	0	10.43	0.00	10.43	1.06
E1	30	0.05	0.05	1.65	0.61	2.25	0.51
E2	30	0.05	0.05	7.56	0.00	7.56	0.93
E3	30	0.05	0.05	0.67	1.21	1.88	0.46
E4	30	0.05	0.05	0.73	0.56	1.30	0.36
A1	50	0	0	1.52	0.00	1.52	0.40
A2	50	0	0	1.16	0.00	1.16	0.33
A3	50	0	0	1.59	0.00	1.59	0.41
A4	50	0	0	1.28	0.00	1.28	0.36
B1	50	0	0.05	1.16	0.08	1.24	0.35
B2	50	0	0.05	1.04	0.00	1.04	0.31
B3	50	0	0.05	1.16	0.00	1.16	0.33
B4	50	0	0.05	0.98	0.12	1.10	0.32
D1	50	0.05	0	0.67	0.31	0.98	0.30
D2	50	0.05	0	5.37	0.00	5.37	0.80
D3	50	0.05	0	7.01	0.00	7.01	0.90
D4	50	0.05	0	1.28	0.72	2.00	0.48
E1	50	0.05	0.05	0.43	1.07	1.49	0.40
E2	50	0.05	0.05	2.32	0.00	2.32	0.52
E3	50	0.05	0.05	2.50	1.40	3.90	0.69
E4	50	0.05	0.05	2.32	0.33	2.65	0.56

Site:	Neagele's Pond	*Date NDS in:*	5/17/05 8:45	Date NDS out:	6/8/05 11:45

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	8.17	0.00	8.17	0.96
A2	10	0	0	15.31	0.00	15.31	1.21
A3	10	0	0	20.01	0.00	20.01	1.32
A4	10	0	0	10.92	0.00	10.92	1.08
B1	10	0	0.05	12.63	0.00	12.63	1.13
B2	10	0	0.05	7.32	0.00	7.32	0.92
B3	10	0	0.05	10.06	0.00	10.06	1.04
B4	10	0	0.05	17.32	0.00	17.32	1.26
D1	10	0.05	0	21.65	0.00	21.65	1.36
D2	10	0.05	0	25.68	0.00	25.68	1.43
D3	10	0.05	0	24.64	0.00	24.64	1.41
D4	10	0.05	0	24.21	0.00	24.21	1.40
E1	10	0.05	0.05	24.40	0.00	24.40	1.40
E2	10	0.05	0.05				
E3	10	0.05	0.05	25.92	0.00	25.92	1.43
E4	10	0.05	0.05	25.19	0.00	25.19	1.42
A1	30	0	0	7.87	0.00	7.87	0.95
A2	30	0	0	5.86	0.00	5.86	0.84
A3	30	0	0	16.29	0.00	16.29	1.24
A4	30	0	0	15.86	0.00	15.86	1.23
B1	30	0	0.05	7.44	0.00	7.44	0.93
B2	30	0	0.05	5.43	0.00	5.43	0.81
B3	30	0	0.05	0.91	3.18	4.09	0.71
B4	30	0	0.05	7.99	0.00	7.99	0.95
D1	30	0.05	0	20.62	0.00	20.62	1.33
D2	30	0.05	0	15.37	0.00	15.37	1.21
D3	30	0.05	0	22.81	0.00	22.81	1.38
D4	30	0.05	0	11.47	0.00	11.47	1.10
E1	30	0.05	0.05	19.94	0.00	19.94	1.32
E2	30	0.05	0.05	13.24	0.00	13.24	1.15
E3	30	0.05	0.05	18.30	0.00	18.30	1.29
E4	30	0.05	0.05	22.38	0.00	22.38	1.37
A1	50	0	0	15.43	0.00	15.43	1.22
A2	50	0	0	10.92	0.00	10.92	1.08
A3	50	0	0	13.48	0.00	13.48	1.16
A4	50	0	0	4.51	0.00	4.51	0.74
B1	50	0	0.05	0.24	2.51	2.27	0.51
B2	50	0	0.05	0.24	1.07	1.32	0.36
B3	50	0	0.05	0.24	5.17	4.92	0.77
B4	50	0	0.05	3.54	0.00	3.54	0.66
D1	50	0.05	0	16.65	0.00	16.65	1.25
D2	50	0.05	0	20.68	0.00	20.68	1.34
D3	50	0.05	0	22.45	0.00	22.45	1.37
D4	50	0.05	0	22.81	0.00	22.81	1.38
E1	50	0.05	0.05	10.67	0.00	10.67	1.07
E2	50	0.05	0.05	12.93	0.00	12.93	1.14
E3	50	0.05	0.05	14.88	0.00	14.88	1.20
E4	50	0.05	0.05	14.39	0.00	14.39	1.19

Site:	Neagele's Pond	*Date NDS in:*	6/8/05 11:45	Date NDS out:	6/28/05 11:30

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	0.43	3.76	4.18	0.71
A2	10	0	0	0.61	3.13	3.74	0.68
A3	10	0	0	0.37	2.68	3.04	0.61
A4	10	0	0	1.04	1.65	2.68	0.57
A5	10	0	0	0.24	3.18	3.42	0.65
B1	10	0	0.05	0.37	2.30	2.66	0.56
B2	10	0	0.05	0.49	0.94	1.43	0.39
B3	10	0	0.05	lost	lost	lost	lost
B4	10	0	0.05	0.67	2.48	3.16	0.62
B5	10	0	0.05	0.18	2.51	2.69	0.57
D1	10	0.05	0	1.16	13.66	14.82	1.20
D2	10	0.05	0	1.16	16.90	18.06	1.28
D3	10	0.05	0	2.26	12.53	14.78	1.20
D4	10	0.05	0	1.46	15.50	16.97	1.25
D5	10	0.05	0	1.71	15.13	16.84	1.25
E1	10	0.05	0.05	1.77	11.07	12.84	1.14
E2	10	0.05	0.05				
E3	10	0.05	0.05	2.07	13.67	15.74	1.22
E4	10	0.05	0.05	2.01	11.07	13.09	1.15
E5	10	0.05	0.05	2.38	11.68	14.06	1.18
A1	30	0	0	0.85	3.34	4.20	0.72
A2	30	0	0	0.85	3.44	4.30	0.72
A3	30	0	0	1.40	1.81	3.21	0.62
A4	30	0	0	1.40	1.53	2.93	0.59
A5	30	0	0	lost	lost	lost	lost
B1	30	0	0.05				
B2	30	0	0.05	0.37	2.26	2.63	0.56
B3	30	0	0.05	0.73	4.98	5.71	0.83
B4	30	0	0.05	0.61	3.27	3.88	0.69
B5	30	0	0.05	0.37	3.92	4.28	0.72
D1	30	0.05	0	1.40	12.87	14.28	1.18
D2	30	0.05	0	2.38	9.37	11.75	1.11
D3	30	0.05	0	7.56	0.00	7.56	0.93
D4	30	0.05	0	2.20	15.07	17.26	1.26
D5	30	0.05	0	1.83	14.11	15.94	1.23
E1	30	0.05	0.05	3.35	4.25	7.61	0.93
E2	30	0.05	0.05	1.28	8.89	10.17	1.05
E3	30	0.05	0.05	0.61	4.92	5.53	0.82
E4	30	0.05	0.05	1.65	13.39	15.04	1.21
E5	30	0.05	0.05	1.71	8.10	9.80	1.03
A1	50	0	0	0.49	2.46	2.94	0.60
A2	50	0	0	0.73	2.05	2.78	0.58
A3	50	0	0	0.73	1.91	2.64	0.56
A4	50	0	0	0.79	2.12	2.92	0.59
A5	50	0	0	0.73	2.74	3.47	0.65
B1	50	0	0.05	0.43	0.65	1.08	0.32
B2	50	0	0.05	0.79	0.37	1.16	0.33
B3	50	0	0.05	0.49	0.59	1.08	0.32
B4	50	0	0.05	0.00	1.10	1.10	0.32
B5	50	0	0.05	0.30	2.29	2.59	0.56
D1	50	0.05	0	1.34	3.97	5.31	0.80
D2	50	0.05	0	2.01	3.87	5.88	0.84
D3	50	0.05	0	0.24	3.55	3.80	0.68
D4	50	0.05	0	1.65	3.05	4.70	0.76
D5	50	0.05	0	lost	lost	lost	lost
E1	50	0.05	0.05	0.91	6.80	7.71	0.94
E2	50	0.05	0.05				
E3	50	0.05	0.05	0.00	2.96	2.96	0.60
E4	50	0.05	0.05	0.37	2.51	2.87	0.59
E5	50	0.05	0.05	1.28	5.37	6.65	0.88

Site: Neagele's Pond *Date NDS in:* 6/28/05 11:30 Date NDS out: 7/18/05 14:45

Vial ID	Depth from surface (cm)	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	Log (x+1) Total chlor a
A1	10	0	0	3.78	0.00	3.78	0.68
A2	10	0	0	2.32	0.16	2.48	0.54
A3	10	0	0	3.72	0.00	3.72	0.67
A4	10	0	0	3.90	0.00	3.90	0.69
B1	10	0	0.05	0.91	0.28	1.20	0.34
B2	10	0	0.05	0.73	0.46	1.19	0.34
B3	10	0	0.05	0.79	0.23	1.02	0.31
B4	10	0	0.05	0.61	0.17	0.78	0.25
D1	10	0.05	0	5.79	0.00	5.79	0.83
D2	10	0.05	0				
D3	10	0.05	0	6.28	0.00	6.28	0.86
D4	10	0.05	0	3.72	7.07	10.79	1.07
E1	10	0.05	0.05	7.20	2.38	9.58	1.02
E2	10	0.05	0.05	6.34	0.00	6.34	
E3	10	0.05	0.05	13.48	0.00	13.48	
E4	10	0.05	0.05	4.51	1.44	5.95	0.84
A1	30	0	0	2.38	0.24	2.62	0.56
A2	30	0	0	2.81	0.00	2.81	0.58
A3	30	0	0	1.04	0.34	1.37	0.38
A4	30	0	0	1.04	0.10	1.13	0.33
B1	30	0	0.05	0.37	0.61	0.98	0.30
B2	30	0	0.05	0.85	0.48	1.33	0.37
B3	30	0	0.05	0.43	0.48	0.91	0.28
B4	30	0	0.05	0.73	0.32	1.05	0.31
D1	30	0.05	0	8.91	0.00	8.91	1.00
D2	30	0.05	0	18.79	0.00	18.79	1.30
D3	30	0.05	0	1.40	0.26	1.66	0.42
D4	30	0.05	0	8.54	1.04	9.58	1.02
E1	30	0.05	0.05				
E2	30	0.05	0.05	1.34	1.63	2.97	0.60
E3	30	0.05	0.05	2.56	16.57	19.14	1.30
E4	30	0.05	0.05	8.17	0.00	8.17	0.96
A1	50	0	0	0.37	0.16	0.53	0.18
A2	50	0	0	0.49	0.15	0.63	0.21
A3	50	0	0	0.79	0.06	0.85	0.27
A4	50	0	0	0.30	0.15	0.46	0.16
B1	50	0	0.05	0.61	0.23	0.84	0.27
B2	50	0	0.05	0.24	0.11	0.35	0.13
B3	50	0	0.05	0.49	0.18	0.67	0.22
B4	50	0	0.05	0.43	0.24	0.67	0.22
D1	50	0.05	0	0.37	0.71	1.08	0.32
D2	50	0.05	0	0.30	0.50	0.80	0.26
D3	50	0.05	0	0.24	0.42	0.66	0.22
D4	50	0.05	0	1.04	0.34	1.37	0.38
E1	50	0.05	0.05	0.18	1.24	1.42	0.38
E2	50	0.05	0.05	0.12	0.88	1.00	0.30
E3	50	0.05	0.05	0.18	1.27	1.45	0.39
E4	50	0.05	0.05	0.49	0.32	0.81	0.26

Appendix K: NDS grazing experiments

Site:	Portage Creek S		Date Floater in:	6/28/04 16:00		Date Floater out:	7/21/04 15:10	
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	chlorophyll a (ug/cm2)
A1	0	0	14.27	0.00	14.27	24.58	0.00	24.58
A2	0	0	12.32	0.00	12.32	23.73	0.00	23.73
A3	0	0	23.18	0.00	23.18	14.82	0.00	14.82
A4	0	0	24.09	0.00	24.09	19.46	0.00	19.46
B1	0	0.05	3.72	0.00	3.72	7.81	0.00	7.81
B2	0	0.05	14.76	0.00	14.76	9.33	0.13	9.46
B3	0	0.05	4.64	0.00	4.64	13.17	0.00	13.17
B4	0	0.05	14.88	0.00	14.88	14.82	0.00	14.82
C1	0	0.5	2.56	12.75	15.31	1.04	30.50	31.54
C2	0	0.5	1.22	21.05	22.27	3.23	28.81	32.05
C3	0	0.5	2.50	26.29	28.79	2.26	26.83	29.09
C4	0	0.5	1.46	16.74	18.21	5.43	47.12	52.55
D1	0.05	0	18.79	0.00	18.79	13.97	0.00	13.97
D2	0.05	0	12.20	0.00	12.20	18.30	0.00	18.30
D3	0.05	0	11.04	0.00	11.04	13.85	0.00	13.85
D4	0.05	0	lost	lost	lost	30.62	0.00	30.62
E1	0.05	0.05	7.93	0.00	7.93	13.85	3.67	17.52
E2	0.05	0.05	4.70	0.00	4.70	15.13	2.36	17.49
E3	0.05	0.05	13.85	0.00	13.85	13.11	10.45	23.57
E4	0.05	0.05	12.87	0.00	12.87	21.59	0.00	21.59
F1	0.05	0.5	0.30	33.80	34.10	1.95	22.58	24.53
F2	0.05	0.5	0.49	10.83	11.32	2.26	25.01	27.26
F3	0.05	0.5	4.09	36.81	40.90	3.29	34.44	37.74
F4	0.05	0.5	lost	lost	lost	1.77	18.76	20.53
G1	0.5	0	19.27	0.00	19.27	12.08	3.26	15.33
G2	0.5	0	18.05	0.00	18.05	9.94	0.00	9.94
G3	0.5	0	23.24	0.00	23.24	17.57	0.00	17.57
G4	0.5	0	12.02	0.00	12.02	19.64	2.14	21.78
H1	0.5	0.05	16.41	0.00	16.41	13.66	0.00	13.66
H2	0.5	0.05	24.28	0.00	24.28	22.99	0.00	22.99
H3	0.5	0.05	13.30	0.00	13.30	lost	lost	lost
H4	0.5	0.05	12.63	0.00	12.63	19.46	0.00	19.46
I1	0.5	0.5	1.16	11.70	12.86	2.07	35.25	37.32
I2	0.5	0.5	1.59	17.83	19.42	2.50	25.49	27.99
I3	0.5	0.5	1.83	20.53	22.36	5.18	48.84	54.03
I4	0.5	0.5	1.16	13.87	15.03	1.71	38.64	40.35

Site:	Portage Creek S		Date Floater in: 7/21/04 15:10			Date Floater out: 8/10/04 11:50		
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyl
A1	0	0	8.54	0.00	8.54	6.53	0.00	6.53
A2	0	0	10.55	0.00	10.55	13.72	0.00	13.72
A3	0	0	19.09	0.00	19.09	16.96	0.00	16.96
A4	0	0	15.55	0.00	15.55	13.17	0.00	13.17
B1	0	0.05	8.11	0.00	8.11	12.26	0.00	12.26
B2	0	0.05	13.30	0.00	13.30	14.64	0.00	14.64
B3	0	0.05	11.04	0.00	11.04	11.71	0.00	11.71
B4	0	0.05	14.52	0.00	14.52	lost	lost	lost
C1	0	0.5	0.85	8.75	9.61	0.61	5.47	6.08
C2	0	0.5	0.12	1.16	1.28	0.18	2.30	2.49
C3	0	0.5	0.06	2.46	2.52	1.40	7.94	9.35
C4	0	0.5	1.83	14.77	16.60	1.04	8.47	9.51
D1	0.05	0	21.10	0.00	21.10	9.33	0.00	9.33
D2	0.05	0	12.02	0.00	12.02			
D3	0.05	0	2.74	0.30	3.04	4.76	0.00	4.76
D4	0.05	0	16.65	0.00	16.65	9.09	0.00	9.09
E1	0.05	0.05	9.88	0.00	9.88	6.16	0.00	6.16
E2	0.05	0.05	18.54	0.00	18.54	8.66	0.00	8.66
E3	0.05	0.05				7.08	0.00	7.08
E4	0.05	0.05	11.22	0.00	11.22	5.12	0.00	5.12
F1	0.05	0.5	0.61	5.75	6.36	0.37	2.09	2.46
F2	0.05	0.5	0.79	5.99	6.78	0.43	5.17	5.60
F3	0.05	0.5	0.37	6.95	7.32	0.85	5.38	6.23
F4	0.05	0.5	0.06	3.01	3.07	0.61	5.03	5.64
G1	0.5	0	9.27	1.26	10.53	8.66	0.00	8.66
G2	0.5	0	lost	lost	lost	15.43	0.00	15.43
G3	0.5	0	14.09	0.00	14.09	lost	lost	lost
G4	0.5	0	9.58	1.31	10.88	9.82	0.00	9.82
H1	0.5	0.05	6.83	0.00	6.83	4.51	0.00	4.51
H2	0.5	0.05	6.47	0.00	6.47	4.15	0.00	4.15
H3	0.5	0.05	10.49	0.00	10.49	8.60	0.00	8.60
H4	0.5	0.05	7.87	0.00	7.87	11.65	0.00	11.65
I1	0.5	0.5	0.73	7.67	8.40	0.49	2.39	2.87
I2	0.5	0.5	0.24	2.24	2.49	0.43	4.10	4.53
I3	0.5	0.5	0.37	4.40	4.77	0.61	4.51	5.12
I4	0.5	0.5	lost	lost	lost	0.43	3.51	3.94

Site:	Cadham Bay east		Date Floater in: 7/21/04 15:10			Date Floater out: 8/10/04 11:50		
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0				4.64	0.00	4.64
A2	0	0	7.14	0.00	7.14	5.79	0.00	5.79
A3	0	0	7.62	0.00	7.62	4.82	0.00	4.82
A4	0	0	7.93	0.00	7.93	4.51	0.00	4.51
B1	0	0.05	6.22	0.00	6.22	2.32	0.00	2.32
B2	0	0.05	5.12	0.00	5.12	2.50	0.50	3.00
B3	0	0.05	6.16	0.00	6.16	lost	lost	lost
B4	0	0.05	4.39	0.00	4.39	2.07	0.26	2.33
C1	0	0.5	0.55	4.78	5.32	0.79	3.33	4.12
C2	0	0.5	0.43	3.24	3.67	0.43	1.79	2.22
C3	0	0.5	lost	lost	lost	0.61	2.03	2.64
C4	0	0.5	0.67	5.42	6.09	0.67	3.52	4.19
D1	0.05	0	16.10	0.00	16.10	8.48	0.00	8.48
D2	0.05	0	8.23	0.00	8.23	11.28	0.00	11.28
D3	0.05	0	10.37	0.00	10.37	10.67	0.00	10.67
D4	0.05	0	8.05	0.00	8.05	5.73	0.00	5.73
E1	0.05	0.05	6.77	0.00	6.77	9.03	0.00	9.03
E2	0.05	0.05	10.80	0.00	10.80	5.43	0.00	5.43
E3	0.05	0.05	7.20	0.00	7.20	5.18	0.00	5.18
E4	0.05	0.05	8.36	0.00	8.36			
F1	0.05	0.5	0.49	6.04	6.53	lost	lost	lost
F2	0.05	0.5	0.37	3.61	3.97	0.37	2.85	3.22
F3	0.05	0.5	0.55	6.43	6.98	0.37	5.19	5.56
F4	0.05	0.5	1.28	10.72	12.00	0.37	3.16	3.53
G1	0.5	0	13.54	0.00	13.54			
G2	0.5	0				4.03	0.00	4.03
G3	0.5	0	10.00	0.00	10.00	7.08	0.00	7.08
G4	0.5	0	12.44	0.00	12.44	8.66	0.00	8.66
H1	0.5	0.05	11.53	0.00	11.53	2.01	0.15	2.16
H2	0.5	0.05	10.55	0.00	10.55	2.20	0.00	2.20
H3	0.5	0.05	9.58	0.00	9.58	3.17	0.00	3.17
H4	0.5	0.05	9.88	0.00	9.88	3.54	0.00	3.54
I1	0.5	0.5	1.04	10.51	11.54	0.30	2.56	2.87
I2	0.5	0.5	0.43	4.00	4.42	0.55	4.26	4.81
I3	0.5	0.5	0.91	7.69	8.61	0.12	2.60	2.73
I4	0.5	0.5	0.73	8.42	9.16	0.49	5.25	5.74

Site:	Cadham Bay east		Date Floater in: 7/21/04 15:10			Date Floater out: 8/10/04 11:50		
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0	23.30	0.00	23.30	17.75	0.00	17.75
A2	0	0	17.63	0.00	17.63	17.26	0.00	17.26
A3	0	0	19.46	0.00	19.46	10.25	0.00	10.25
A4	0	0	18.24	0.00	18.24	18.79	0.00	18.79
B1	0	0.05				24.82	0.00	24.82
B2	0	0.05	23.48	0.00	23.48	12.81	0.00	12.81
B3	0	0.05	27.33	0.00	27.33	10.00	0.00	10.00
B4	0	0.05	23.06	0.00	23.06	11.47	0.00	11.47
C1	0	0.5	0.67	5.55	6.22	0.37	3.51	3.87
C2	0	0.5	0.37	3.16	3.53	0.91	7.94	8.85
C3	0	0.5	1.40	11.29	12.69	0.67	8.28	8.95
C4	0	0.5	2.81	23.33	26.14	0.18	1.89	2.07
D1	0.05	0	13.78	0.00	13.78	23.24	0.00	23.24
D2	0.05	0	18.30	0.00	18.30	12.99	0.00	12.99
D3	0.05	0	15.68	0.00	15.68	15.74	0.00	15.74
D4	0.05	0	15.61	0.00	15.61	15.98	0.00	15.98
E1	0.05	0.05	15.07	0.00	15.07	15.61	0.00	15.61
E2	0.05	0.05	21.10	0.00	21.10	3.54	19.62	23.16
E3	0.05	0.05	10.61	0.00	10.61	12.50	0.00	12.50
E4	0.05	0.05	13.72	0.00	13.72	5.43	18.23	23.66
F1	0.05	0.5	0.85	9.31	10.16	0.73	11.80	12.53
F2	0.05	0.5	0.73	7.32	8.05	1.10	14.96	16.06
F3	0.05	0.5	0.91	8.38	9.30	0.55	7.88	8.43
F4	0.05	0.5	lost	lost	lost	0.49	5.59	6.08
G1	0.5	0	25.62	0.00	25.62	17.63	0.00	17.63
G2	0.5	0	15.31	0.00	15.31	16.90	0.00	16.90
G3	0.5	0	16.53	0.00	16.53	19.09	0.00	19.09
G4	0.5	0	22.93	0.00	22.93	19.09	0.00	19.09
H1	0.5	0.05	12.02	0.00	12.02	12.14	0.00	12.14
H2	0.5	0.05	13.85	0.00	13.85	6.77	0.00	6.77
H3	0.5	0.05	25.13	0.00	25.13	5.73	0.00	5.73
H4	0.5	0.05	21.59	0.00	21.59	22.75	0.00	22.75
I1	0.5	0.5				0.24	1.97	2.21
I2	0.5	0.5	1.22	10.78	12.00	0.43	2.83	3.25
I3	0.5	0.5	1.04	8.68	9.72	1.10	13.41	14.51
I4	0.5	0.5	1.71	15.27	16.97	0.12	2.78	2.90

Site:	Delta Channel		Date Floater in: ##########			Date Floater out: 8/10/04 11:50		
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0	8.48	0.00	8.48	6.28	0.17	6.45
A2	0	0	13.24	0.00	13.24	7.38	0.55	7.93
A3	0	0	8.17	0.00	8.17	9.03	0.00	9.03
A4	0	0				7.44	0.00	7.44
B1	0	0.05	7.38	0.00	7.38	6.59	1.70	8.28
B2	0	0.05	6.95	0.00	6.95	3.96	0.83	4.80
B3	0	0.05	6.04	0.00	6.04	7.44	1.25	8.69
B4	0	0.05	6.83	0.00	6.83	6.40	0.00	6.40
C1	0	0.5	0.24	2.28	2.52	0.18	4.68	4.87
C2	0	0.5	0.43	2.55	2.98	0.43	5.20	5.63
C3	0	0.5	0.24	4.14	4.38	0.24	4.83	5.07
C4	0	0.5	0.43	3.27	3.70	0.37	7.02	7.39
D1	0.05	0	11.41	0.00	11.41	11.41	5.53	16.93
D2	0.05	0				11.41	6.11	17.52
D3	0.05	0	12.20	0.00	12.20	11.59	3.35	14.94
D4	0.05	0	17.32	0.00	17.32	8.48	3.69	12.16
E1	0.05	0.05	9.94	0.00	9.94	15.25	3.90	19.14
E2	0.05	0.05	8.54	0.00	8.54	7.75	5.43	13.18
E3	0.05	0.05	12.93	0.00	12.93	9.03	3.95	12.97
E4	0.05	0.05	13.78	0.00	13.78	8.42	7.40	15.82
F1	0.05	0.5	0.73	5.91	6.64	0.73	13.80	14.53
F2	0.05	0.5	0.49	3.49	3.98	0.79	13.47	14.26
F3	0.05	0.5	0.24	2.42	2.66	0.55	10.64	11.18
F4	0.05	0.5	0.61	5.20	5.81	0.73	10.08	10.81
G1	0.5	0	18.54	0.00	18.54	6.16	4.80	10.96
G2	0.5	0	17.32	0.00	17.32	11.34	2.35	13.69
G3	0.5	0	13.60	0.00	13.60	16.53	6.44	22.97
G4	0.5	0	15.86	0.00	15.86	6.28	16.71	23.00
H1	0.5	0.05	13.30	0.00	13.30	10.37	7.02	17.39
H2	0.5	0.05	12.38	0.00	12.38	11.83	9.04	20.88
H3	0.5	0.05	16.41	0.00	16.41	10.98	3.74	14.72
H4	0.5	0.05				12.99	4.33	17.32
I1	0.5	0.5	0.30	2.88	3.18	1.28	9.54	10.83
I2	0.5	0.5	0.61	5.72	6.33	0.49	3.87	4.36
I3	0.5	0.5	1.04	9.16	10.20	1.04	9.68	10.72
I4	0.5	0.5	0.61	4.06	4.67	2.32	16.78	19.09

Site:	Delta Channel		Date Floater in: ##########			Date Floater out: 8/10/04 11:50		
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0	7.75	0.00	7.75	8.54	0.00	8.54
A2	0	0	7.69	0.00	7.69	7.69	0.00	7.69
A3	0	0	7.01	0.00	7.01	5.86	1.06	6.92
A4	0	0	8.11	0.00	8.11	5.73	1.39	7.12
B1	0	0.05	4.57	0.00	4.57	4.94	0.00	4.94
B2	0	0.05	6.65	0.00	6.65	2.74	0.00	2.74
B3	0	0.05	2.50	0.00	2.50	3.11	0.00	3.11
B4	0	0.05	3.29	0.00	3.29	3.23	0.00	3.23
C1	0	0.5	0.43	2.69	3.11	0.12	3.91	4.04
C2	0	0.5	1.04	4.40	5.44	0.43	2.48	2.91
C3	0	0.5	0.49	2.84	3.32	0.61	4.30	4.91
C4	0	0.5	0.49	4.90	5.39	0.37	2.88	3.25
D1	0.05	0				8.30	0.00	8.30
D2	0.05	0	12.56	0.00	12.56	16.35	0.00	16.35
D3	0.05	0	13.78	0.00	13.78	14.27	0.00	14.27
D4	0.05	0	11.47	0.00	11.47	11.16	0.00	11.16
E1	0.05	0.05	15.25	0.00	15.25	lost	lost	lost
E2	0.05	0.05	13.30	0.00	13.30	11.10	0.00	11.10
E3	0.05	0.05	16.16	0.00	16.16	6.77	1.17	7.95
E4	0.05	0.05	12.44	0.00	12.44	10.49	1.28	11.77
F1	0.05	0.5	0.79	8.54	9.33	0.49	6.49	6.98
F2	0.05	0.5	1.34	10.52	11.86	0.61	7.92	8.53
F3	0.05	0.5	0.85	8.37	9.23	0.43	7.69	8.11
F4	0.05	0.5	1.10	11.76	12.85	0.73	7.42	8.16
G1	0.5	0	22.45	0.00	22.45	16.65	0.00	16.65
G2	0.5	0	14.21	0.00	14.21	7.20	0.00	7.20
G3	0.5	0	13.11	0.00	13.11	11.16	0.00	11.16
G4	0.5	0	8.66	0.00	8.66	lost	lost	lost
H1	0.5	0.05	20.07	0.00	20.07	4.64	3.77	8.40
H2	0.5	0.05	10.25	0.00	10.25	11.71	0.00	11.71
H3	0.5	0.05	22.87	0.00	22.87	4.15	4.10	8.25
H4	0.5	0.05	13.66	0.00	13.66	lost	lost	lost
I1	0.5	0.5	1.71	7.48	9.18	0.49	4.32	4.81
I2	0.5	0.5	0.91	10.28	11.19	0.61	8.54	9.15
I3	0.5	0.5	0.85	9.17	10.02	1.46	11.33	12.80
I4	0.5	0.5	1.04	6.85	7.89	0.91	6.66	7.58

Site:	Center Marsh		Date Floater in:	7/21/04 15:10		Date Floater out:	8/10/04 11:50	
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0	1.59	0.60	2.18	6.71	0.00	6.71
A2	0	0	1.34	0.32	1.66	4.03	0.00	4.03
A3	0	0	3.05	0.00	3.05	3.54	0.00	3.54
A4	0	0	4.57	0.00	4.57	5.37	0.00	5.37
B1	0	0.05	3.35	0.00	3.35	2.32	0.00	2.32
B2	0	0.05	1.52	0.17	1.70	1.89	0.00	1.89
B3	0	0.05	2.74	0.00	2.74	2.68	0.00	2.68
B4	0	0.05	1.46	0.00	1.46	2.13	0.00	2.13
C1	0	0.5	0.06	0.73	0.79	0.00	0.97	0.97
C2	0	0.5	0.00	1.17	1.17	0.12	1.57	1.69
C3	0	0.5	lost	lost	lost	0.06	1.01	1.07
C4	0	0.5	0.24	0.97	1.21	0.06	1.25	1.31
D1	0.05	0	15.80	0.00	15.80	18.18	0.00	18.18
D2	0.05	0	17.02	0.00	17.02	13.97	3.83	17.80
D3	0.05	0	18.18	0.00	18.18	17.02	2.80	19.81
D4	0.05	0	24.09	0.00	24.09	23.60	0.00	23.60
E1	0.05	0.05	17.51	0.00	17.51	12.20	0.00	12.20
E2	0.05	0.05	13.66	0.00	13.66	14.46	0.32	14.78
E3	0.05	0.05	14.09	0.00	14.09	13.54	0.00	13.54
E4	0.05	0.05	14.39	2.55	16.95	8.36	0.87	9.23
F1	0.05	0.5	0.55	5.77	6.32	0.85	8.93	9.78
F2	0.05	0.5	1.52	7.27	8.80	0.79	8.05	8.85
F3	0.05	0.5	0.49	6.80	7.29	0.79	6.54	7.33
F4	0.05	0.5	0.98	8.53	9.51	0.24	5.52	5.76
G1	0.5	0	20.19	0.00	20.19	24.34	0.00	24.34
G2	0.5	0	10.80	2.50	13.30			
G3	0.5	0	11.83	6.73	18.57	11.59	9.35	20.94
G4	0.5	0	15.07	0.00	15.07	23.85	0.00	23.85
H1	0.5	0.05	14.27	0.00	14.27	18.72	0.00	18.72
H2	0.5	0.05	23.60	0.00	23.60	15.43	0.00	15.43
H3	0.5	0.05	21.71	0.00	21.71	15.80	0.09	15.88
H4	0.5	0.05	12.32	0.23	12.55	17.51	0.00	17.51
I1	0.5	0.5	0.79	7.95	8.74	1.16	12.42	13.58
I2	0.5	0.5	1.16	11.04	12.20	0.79	8.47	9.26
I3	0.5	0.5	0.79	10.02	10.81	0.85	10.10	10.95
I4	0.5	0.5	0.67	6.79	7.46	1.65	13.81	15.46

Site:	Center Marsh		Date Floater in:	7/21/04 15:10		Date Floater out:	8/10/04 11:50	
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0	3.66	0.00	3.66	3.23	0.00	3.23
A2	0	0	2.62	0.10	2.73	5.12	0.00	5.12
A3	0	0	1.65	0.00	1.65	4.82	0.00	4.82
A4	0	0	4.57	0.00	4.57	4.64	0.00	4.64
B1	0	0.05	1.89	0.00	1.89	1.77	0.21	1.98
B2	0	0.05	1.16	0.15	1.31	1.34	0.00	1.34
B3	0	0.05	1.52	0.00	1.52	1.83	0.00	1.83
B4	0	0.05	1.59	0.00	1.59	1.34	0.00	1.34
C1	0	0.5	0.18	1.27	1.45	0.24	2.07	2.32
C2	0	0.5	0.24	1.42	1.66	0.06	1.35	1.42
C3	0	0.5	0.12	1.05	1.18	0.06	1.39	1.45
C4	0	0.5	0.12	1.02	1.14	0.00	1.03	1.03
D1	0.05	0	21.16	0.00	21.16	19.33	0.00	19.33
D2	0.05	0	23.12	0.00	23.12	22.20	0.00	22.20
D3	0.05	0	13.05	0.00	13.05			
D4	0.05	0	24.28	0.00	24.28	19.52	0.00	19.52
E1	0.05	0.05	11.89	0.00	11.89	9.82	1.90	11.72
E2	0.05	0.05	12.87	0.00	12.87	7.99	0.00	7.99
E3	0.05	0.05	6.65	0.00	6.65	7.20	3.07	10.27
E4	0.05	0.05	13.92	0.00	13.92	15.00	0.00	15.00
F1	0.05	0.5	0.67	5.69	6.36	17.26	0.00	17.26
F2	0.05	0.5	0.91	9.97	10.88	20.68	0.00	20.68
F3	0.05	0.5	0.98	9.60	10.58	16.35	1.17	17.52
F4	0.05	0.5	0.67	7.17	7.84	24.15	0.00	24.15
G1	0.5	0	12.44	0.00	12.44	24.58	0.00	24.58
G2	0.5	0	19.46	0.00	19.46	29.64	0.00	29.64
G3	0.5	0	13.42	0.00	13.42	24.76	0.00	24.76
G4	0.5	0	18.18	0.00	18.18	30.50	0.00	30.50
H1	0.5	0.05	15.92	0.00	15.92	10.49	0.00	10.49
H2	0.5	0.05	18.54	0.00	18.54	14.21	0.00	14.21
H3	0.5	0.05	13.30	0.00	13.30	7.01	0.00	7.01
H4	0.5	0.05	11.04	0.00	11.04	9.21	0.00	9.21
I1	0.5	0.5	0.55	10.81	11.36	1.52	6.00	7.52
I2	0.5	0.5	6.40	2.36	8.76	0.61	6.41	7.02
I3	0.5	0.5	0.55	8.19	8.74	0.73	9.11	9.85
I4	0.5	0.5	1.52	6.86	8.39	0.85	8.65	9.50

Site:	Crescent Pond		Date Floater in: 7/21/04 15:10			Date Floater out: 8/10/04 11:50		
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0	13.78	0.00	13.78	10.92	0.00	10.92
A2	0	0	11.59	0.00	11.59	13.11	0.00	13.11
A3	0	0	15.61	0.00	15.61	7.01	0.00	7.01
A4	0	0	8.60	0.00	8.60	16.83	0.00	16.83
B1	0	0.05	7.99	0.00	7.99	7.44	0.00	7.44
B2	0	0.05	11.28	0.00	11.28	5.49	0.00	5.49
B3	0	0.05	10.86	0.00	10.86	5.92	0.00	5.92
B4	0	0.05	8.54	0.00	8.54	5.25	0.00	5.25
C1	0	0.5	0.43	7.38	7.80	0.37	3.51	3.87
C2	0	0.5	0.49	7.28	7.77	0.30	2.84	3.15
C3	0	0.5	0.18	3.61	3.80	1.28	5.62	6.90
C4	0	0.5	0.43	5.72	6.15	0.30	3.22	3.52
D1	0.05	0	17.02	1.35	18.37	19.21	0.00	19.21
D2	0.05	0	21.71	0.00	21.71	21.96	0.00	21.96
D3	0.05	0	24.28	0.00	24.28	29.64	0.00	29.64
D4	0.05	0	26.17	0.00	26.17	23.91	0.00	23.91
E1	0.05	0.05	24.40	0.00	24.40	21.47	0.00	21.47
E2	0.05	0.05	14.27	1.60	15.88	12.75	0.00	12.75
E3	0.05	0.05	10.55	2.64	13.19	16.65	0.00	16.65
E4	0.05	0.05	12.08	0.19	12.26	24.09	0.00	24.09
F1	0.05	0.5	0.55	8.43	8.98	0.85	8.13	8.99
F2	0.05	0.5	1.04	11.54	12.58	0.67	7.83	8.50
F3	0.05	0.5	1.46	11.57	13.04	0.85	9.79	10.64
F4	0.05	0.5	1.10	12.45	13.54	1.10	10.89	11.99
G1	0.5	0	30.68	0.00	30.68	11.95	0.00	11.95
G2	0.5	0	16.59	0.00	16.59	19.70	0.00	19.70
G3	0.5	0	23.48	0.00	23.48	34.95	0.00	34.95
G4	0.5	0	19.52	0.00	19.52	19.27	0.00	19.27
H1	0.5	0.05	18.24	0.00	18.24	26.17	0.00	26.17
H2	0.5	0.05	13.42	0.00	13.42	13.85	0.00	13.85
H3	0.5	0.05	20.55	0.00	20.55	14.39	0.00	14.39
H4	0.5	0.05	13.91	0.00	13.91	19.64	0.00	19.64
I1	0.5	0.5	1.04	11.40	12.44	0.85	8.96	9.81
I2	0.5	0.5	0.79	9.26	10.05	1.04	9.09	10.13
I3	0.5	0.5	1.10	13.14	14.23	2.62	17.82	20.45
I4	0.5	0.5	0.61	5.92	6.53	0.98	10.88	11.85

Site:	Crescent Pond		Date Floater in: 7/21/04 15:10			Date Floater out: 8/10/04 11:50		
			No malathion			malathion treatment		
Vial ID	N conc	P conc	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)	chlorophyll a (ug/cm2)	pheophytin a (ug/cm2)	Total chlorophyll a (ug/cm2)
A1	0	0	6.40	0.00	6.40	2.13	0.00	2.13
A2	0	0	8.23	0.00	8.23	3.17	0.00	3.17
A3	0	0	5.18	0.00	5.18	3.05	0.00	3.05
A4	0	0	3.66	0.00	3.66	2.81	0.00	2.81
B1	0	0.05	1.28	0.00	1.28	1.10	0.00	1.10
B2	0	0.05	1.65	0.00	1.65	0.73	0.05	0.78
B3	0	0.05	1.40	0.08	1.49	0.98	0.19	1.16
B4	0	0.05	4.74	0.00	4.74	0.85	0.00	0.85
C1	0	0.5	0.06	1.42	1.48	0.24	0.69	0.94
C2	0	0.5	0.18	1.61	1.80	0.12	0.85	0.97
C3	0	0.5	0.06	0.77	0.83	0.00	0.90	0.90
C4	0	0.5	0.18	1.20	1.38	0.00	0.59	0.59
D1	0.05	0	15.19	0.00	15.19	8.17	0.00	8.17
D2	0.05	0	4.70	0.00	4.70	8.84	0.00	8.84
D3	0.05	0	8.48	0.00	8.48	13.72	0.00	13.72
D4	0.05	0	10.00	0.00	10.00	9.82	0.00	9.82
E1	0.05	0.05	7.81	2.44	10.25	3.48	0.00	3.48
E2	0.05	0.05	1.34	1.56	2.90	6.95	1.17	8.12
E3	0.05	0.05	11.53	0.00	11.53	4.64	0.84	5.47
E4	0.05	0.05	4.03	2.77	6.80	5.73	0.00	5.73
F1	0.05	0.5	0.30	4.94	5.25	0.18	4.13	4.31
F2	0.05	0.5	0.43	3.62	4.05	0.18	1.72	1.90
F3	0.05	0.5	0.55	3.74	4.29	0.24	2.93	3.18
F4	0.05	0.5	0.49	4.87	5.36	0.37	3.02	3.39
G1	0.5	0	12.75	0.00	12.75	5.67	0.00	5.67
G2	0.5	0	13.91	0.00	13.91	7.62	0.00	7.62
G3	0.5	0	9.21	0.00	9.21	9.94	0.00	9.94
G4	0.5	0	14.27	0.00	14.27	lost	lost	lost
H1	0.5	0.05	9.94	0.00	9.94	3.60	1.26	4.86
H2	0.5	0.05	12.02	0.00	12.02	1.40	3.26	4.66
H3	0.5	0.05	11.28	0.00	11.28	4.64	0.00	4.64
H4	0.5	0.05	6.10	0.00	6.10	5.98	0.00	5.98
I1	0.5	0.5	0.18	1.79	1.97	0.24	4.69	4.94
I2	0.5	0.5	0.30	2.67	2.97	0.55	3.81	4.36
I3	0.5	0.5	0.24	2.62	2.87	0.24	2.18	2.42
I4	0.5	0.5	0.49	4.04	4.53	0.49	3.80	4.29