Experimental manipulation of ponds to determine the impact of common carp (*Cyprinus carpio* L.) in Delta Marsh, Manitoba: effects on water quality, algae, and submersed vegetation.

by

Stacy Dawn Hnatiuk

A Thesis submitted to the Faculty of Graduate Studies in partial fulfillment of the requirements for the degree of

Master of Science

Department of Botany
University of Manitoba
Winnipeg, Manitoba

THE UNIVERSITY OF MANITOBA

FACULTY OF GRADUATE STUDIES

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Experimental manipulation of ponds to determine the impact of common carp (*Cyprinus carpio* L.) in Delta Marsh, Manitoba: effects on water quality, algae, and submersed vegetation.

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Abstract

Since their introduction to Canada in the late 1800s, common carp (*Cyprinus carpio* L.) have lead to declines in the habitat quality of many aquatic systems. Carp re-suspend sediments, which leads to decreased water clarity, altered water chemistry, and shifts the ecosystem to a phytoplankton-dominated state. I monitored changes in water quality, and abundance of algae and submersed vegetation in Delta Marsh over two years. Ten ponds (1-13 ha) that had varying degrees of exposure to carp were used. Following one year of background characterization in 2001, manipulations were carried out prior to the second year of monitoring in 2002. New channels were created into two previously isolated ponds, while carp were excluded from four others ponds by screens or dikes. Four others remained unmanipulated. The results indicate that carp are contributing to the habitat degradation of Delta Marsh. Phytoplankton abundance, suspended solids, nutrients (P and Si), and turbidity increased with greater densities of young-of-the-year carp and the occurrence of large carp. Excluding large carp lead to a switch from a turbid, phytoplankton-dominated state to a clear, macrophyte-dominated state in peripheral ponds. Recommendations are made for further research and management of carp in Manitoba waters.

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Chapter 1: Introduction

Common carp (*Cyprinus carpio* L.) is an exotic species of fish in North America. Where it has been introduced it is well known to be destructive to aquatic habitats (Cahn 1929; King & Hunt 1967; Crivelli 1983). Areas where carp are present tend to be turbid, with an abundance of phytoplankton and very little submersed vegetation (Roberts et al. 1995; King et al. 1997; Lougheed et al. 1998; Zambrano et al. 1999). Through their feeding and spawning activities carp disrupt the sediments (Panek 1987). This disruption leads to the release of nutrients, the suspension of sediments, the uprooting of submersed vegetation, and the disturbance of the biological community within the upper layers of the sediment (Chow-Fraser 1998). The increase of nutrients within the water column supports an abundance of phytoplankton and periphytic algae (Chow-Fraser 1998). This abundance of phytoplankton, combined with the suspension of sediment, leads to increased turbidity and results in reduced light penetration. The lower availability of light further reduces the growth of submersed vegetation (Chow-Fraser 1998) and possibly benthic algae (Robertson et al. 1997).

Many studies have determined the impact of common carp on aquatic systems; however, most of the research has been through the use of mesocosm or enclosure experiments (King & Hunt 1967; Crivelli 1983; Winkel & Meulemans 1985; Kolterman 1990; Breukelaar et al. 1994; Roberts et al. 1995; Drenner et al. 1998; Lougheed et al. 1998; Sidorkewicj et al. 1999; Zambrano & Hinojosa 1999;

Angeler et al. 2002; Khan et al. 2003; Parkos III et al. 2003). In this type of experiment carp are confined within a defined space for the duration of the study, the system is simplified often retaining only a portion of the natural variation. Such research may exaggerate the intensity of pressure carp would apply in a natural system where the carp would be free to roam in and out of the area of interest. There is a need for research on the impact carp have on aquatic systems under more natural conditions.

1.1 Objectives

Delta Marsh is a freshwater coastal wetland in south-central Manitoba where carp are present in high abundance and have been present for more than fifty years (Wrubleski 1998). I believe that the high abundance of carp in Delta Marsh is an important contributing factor to the current turbid state of much of the marsh.

There are isolated areas of Delta Marsh where the water appears less turbid and submersed vegetation remains abundant (Goldsborough and Wrubleski *unpublished data*). These isolated sites may be different from the rest of the marsh due partially to their inaccessibility to carp. The goal of this study was to determine if the water quality, algal abundance and submersed vegetation in these areas where carp are absent are indeed different from the areas where carp are present, and to determine if the exclusion of carp from other areas of the marsh is an effective method of improving the water quality and growth of submersed vegetation.

1.2 Hypotheses

Greater turbidity in the water column will be associated with the presence of carp. Higher turbidity in these ponds will be a result of more suspended solids caused by sediment re-suspension following sediment disturbance, as well as a greater abundance of phytoplankton following increased nutrient availability.

Higher nutrient concentrations in the water column will be associated with the presence of carp. The disruption of sediments expected by the carp will release nutrients that are stored there into the water column. Nutrients such as nitrogen, phosphorus, and silica, which are present in high concentrations in the sediments, are predicted to undergo these increases.

Algal growth will be altered in the presence of carp. The higher concentrations of nutrients expected with carp will allow for the proliferation of phytoplankton and attached algae (epiphyton and periphyton). The growth of algae in the sediments (sediment associated algae; SAA) is expected to decrease due to lower light availability.

The density and diversity of submersed vegetation will be lower in the presence of carp. This reduction is expected due to the lower light availability associated with higher turbidity, and the disturbance of the sediments by carp leading to the uprooting of vegetation.

Chapter 2: Literature Review

2.1 Biology of the Common Carp

Common carp (*Cyprinus carpio* L.) is a freshwater fish belonging to the Cyprinidae or minnow family. Members of the cyprinid family most closely related to carp include koi and goldfish. Carp are an exotic fish to North-America, and its origins are in Eurasia (McCrimmon 1968).

The habitat of the common carp varies widely. Typically, carp will stay in shallow littoral areas in the spring and move to deeper waters in the winter (Garcia-Berthou 2001). However, they generally prefer shallow calm waters with adequate vegetation for cover (Panek 1987).

Common carp are dark olive-gold above with yellow-white sides and underbelly. Based on their appearance carp can be divided into three groups: scaled (fully scaled), mirrored (with patchy large scales), and leather carp (with no or almost no scales) (Sigler 1958). They have two sets of barbels on either side of the upper jaw and a mouth with large lips. Both the anal and dorsal fins of carp contain a sharp, serrated spine. The gills of carp contain pharyngeal teeth and gill rakers, which are used in feeding (Sigler 1958).

Adult carp are omnivorous. Their diet typically consists of chironomid larvae and pupae, and small crustaceans. Carp may also take in other insects, snails and small freshwater clams (Sigler 1958; Panek 1987; Garcia-Berthou 2001; Khan 2003). Adult carp will consume rooted plants, seeds, and algae, although this is believed to be due to accidental ingestion in the search for aquatic insects (Panek 1987), or to supplement the diet in a poor habitat where animal food is difficult to find (Sigler 1958). The diet of young carp is similar to the adult and will include zooplankton, small crustaceans, snails and chironomid larvae (Sigler 1958; Panek 1987; Khan 2003). Sigler (1958) notes that chironomids taken by young carp are typically from small species. Common carp feed throughout the water column, but mostly in the sediments (Panek 1987), and appear to prefer to feed in shallow water (Crivelli 1983). Carp root around in the sediments in their search for food where large carp have the ability to find food that is ten centimetres deep (Panek 1987). This feeding behaviour is known to uproot large amounts of submerged vegetation. Sigler (1958) found that sand, silt and shells could make up as much as twenty percent of the stomach volume in carp, indicating carp search for their food in the sediments and have a poor ability to separate food stuffs.

In Canada, male carp are sexually mature at three to four years, and the females at four to five years (Swee & McCrimmon 1966). When ready to spawn, as much as one third the body weight of a female carp can be egg mass (Sigler 1958). Carp spawn in late spring and early summer when water temperatures are between 16 and 28°C (Swee & McCrimmon 1966). Their preferred spawning habitat is shallow water from 30 to 60 centimetres deep in dense beds of vegetation (Panek 1987). Spawning events tend to be loud, involving vigorous activity and splashing (Sigler 1958). Multiple spawning events may occur within one season (Sigler 1958; Swee & McCrimmon 1966) and there is some evidence that carp may home to the same spawning site (Bonneau & Scarnecchia 2002). During spawning, eggs are released over the bottom, often on plant material or debris to which the sticky egg mass may attach (Swee & McCrimmon 1966). Common carp exhibit no parental care and produce a large number of eggs. Egg production is proportional to fish size and one large (85 cm) female may lay over two million eggs in a season (Swee & McCrimmon 1966).

Carp eggs will typically hatch within three to six days after fertilization (Swee & McCrimmon 1966). Upon hatching, larval carp remain attached to vegetation for the first few days, then move to the bottom when the yolk sac is depleted (Sigler 1958). Carp growth is rapid in the first few years. In a good habitat, carp can grow an average of ten centimetres in length per year for the first four years of life (Sigler 1958). This rapid increase in size quickly makes young carp difficult to handle by predators.

Besides their rapid growth rate, young carp exhibit behaviours which protect themselves from predators. Young carp will remain hidden in vegetation, virtually safe from predatory fish for most of their first summer (Sigler 1958; Swee & McCrimmon 1966). They also exhibit predator avoidance by burying themselves in mud or sand to hide from birds (Panek 1987), such as cormorants, pelicans, herons and mergansers which have the ability to eat large numbers of carp (Sigler 1958). Of the few carp predators in North America, humans are the only predator with the capacity to have any effective impact on carp populations (Sigler 1958).

Common carp are a highly robust fish. Carp can live in water temperatures ranging from 4 to 32°C. They can withstand oxygen concentrations as low as 2 mg/L. This is relatively low when compared with the tolerances of other temperate fish species (Panek 1987). Common carp also bear turbidity extremely well. Adult carp can tolerate suspended solids up to a lethal limit of 165, 000 mg/L (Sigler 1958; Panek 1987). Turbidity levels that block out all photosynthetically active radiation, killing and preventing the regeneration of plant life, have little effect on carp (Sigler 1958). Carp can gain a competitive advantage in highly turbid waters, where their keen senses of smell, taste and hearing help them to find food in the dark (Panek 1987).

There are few barriers to the distribution of common carp; however, they have a low tolerance of salinity and concentrations of 1,500 to 2,000 ppm are lethal (Panek 1987). As a result, it is uncommon to find carp in brackish water. There are many parasites that affect carp, and often an individual carp may be infected with more than one parasite (Sigler 1958). However, parasites and diseases have little impact on wild populations of carp except in high density populations, and therefore do not play an important role in population control (Panek 1987).

2.2 History: Carp and Manitoba

Carp were first imported to North America in 1877 by the United States government. Declines in the USA native fish stocks in the late 1800s caused concern for fisheries. This lead to thinking by the commercial fishing community that there was a need to stock waters with fish that would thrive in North American waters. Common carp was chosen because of the species' high fecundity, hardiness, adaptability and rapid growth, as well as because it was already a known food fish to European immigrants and would therefore be easily accepted. In 1877, three hundred forty-five carp were brought to Washington, DC from Germany to be bred and distributed for the purpose of stocking (Fritz 1987).

Around the same time, private individuals in Canada, with the support of the Canadian government, became interested in carp culture. The first record of carp being imported to Canada was in 1880, when fish from the breeding ponds in Washington were sent to two Canadians; one in Ontario and one in New Brunswick (McCrimmon 1968). In the early 1880s, the Canadian government attempted to culture carp for distribution to the public; however, the hatchery experienced problems and failed. Further stockings used fish imported from the USA (McCrimmon 1968). Manitoba saw its first recorded introduction of carp in 1885 when 100 carp were distributed between ponds in Springfield, Portage la Prairie and Minnedosa (McCrimmon 1968). Carp continued to be brought into

Canada both privately and publicly until 1897, when both the Canadian and USA governments began to discourage further introductions of the species (McCrimmon 1968).

Following its introduction to North America, carp quickly became established. By 1908, over 40 million pounds of carp were caught by commercial fishers in the USA (Fritz 1987). However, as the fish became more common in the waterways of the USA and parts of Canada throughout the 1890s, carp began to get a bad reputation. Few people wanted to eat carp due to its "coarse … texture and insipid … flavour" (Ontario Department of Game and Fisheries, 1899 *in* McCrimmon 1968). As a result, fishers were reported to have disposed of carp rather than trying to sell them. By 1899, the Ontario Department of Game and Fisheries reported that carp were "destructive of the more valuable fish"(McCrimmon 1968).

There was no indication that the introductions of carp in Manitoba resulted in permanent populations until 1938; previously, carp were unknown in the wild in Manitoba. The first recorded presence of carp was in the Red River at Lockport, soon after which the fish was seen in Lake Winnipeg (Hinks 1943; McCrimmon 1968). Carp gradually spread north through the Nelson River system and west into Saskatchewan along the Assiniboine River (Atton 1959). Carp were first documented in Lake Manitoba in 1948 (McCrimmon 1968) and subsequently Delta Marsh around 1950 (Wrubleski 1998).

2.3 *The Impact of Carp*

A 1929 paper by A. R. Cahn re-counts the story of a soft muddy-bottomed lake in southern Wisconsin that underwent dramatic changes after the accidental introduction of common carp (Cahn 1929). Within thirteen years the beautiful, previously well vegetated lake with good fishing became turbid, vegetation decreased, and fishing became poor. Similar circumstances have been seen in many wetlands, lakes, and reservoirs throughout the world following the introduction of carp (King & Hunt 1967; King et al. 1997; Chow-Fraser 1998; Zambrano et al. 1999; Garcia 2001).

Many studies have looked at the effects that carp have in their environment. The most common type of study in the literature is the enclosure experiment (Roberts et al. 1995; Drenner et al. 1998; Lougheed et al. 1998; Sidorkewicj et al. 1999; Parkos III et al. 2003, Badiou 2005), and this includes the earliest and most well known experimental studies on carp (Robel 1962; King & Hunt 1967; Crivelli 1983). Enclosure experiments are able to determine the effect of carp on specific components of the community; such as vegetation, turbidity, and nutrient dynamics, under relatively controlled conditions in small aquaria, mesocosms, or penned portions of a natural system. Gradually, as more research was conducted, the systems studied got larger and more complex. The most recent literature includes biomanipulation experiments where carp are either added (Zambrano et al. 1999, Badiou 2006) or removed (Lougheed & Chow-Fraser 2001) from an entire system and changes are observed.

Much of the impact carp has been found to have on an aquatic ecosystem are due to carp's feeding and spawning behaviours. Through both activities, they disturb the sediment, which either directly or indirectly may lead to changes in their environment.

2.3.1 Water quality

Common carp are thought to influence water quality in two ways: through sediment disturbance and excretion. The components of water quality that are most affected by these behaviours are turbidity and nutrient concentrations.

Turbidity is a measure of the clarity of a column of water and is determined by how much light is scattered as it passes through the column. The degree of light scattering is dependent on the amount and type of particles present in the water. Any particles suspended in water can scatter light (Wetzel 1983). Suspended materials can be organic, such as plankton or detritus, or inorganic in the form of suspended sediments. Carp therefore can influence turbidity by altering the abundance of plankton, detritus and/or sediment in the water column.

Carp can directly alter turbidity through their feeding and spawning behaviours which can suspend sediment and detritus. They can also affect turbidity indirectly either by increasing the availability of nutrients, resulting in increased phytoplankton growth, or through the removal of vegetation which may alter the importance of wind induced sediment suspension. There are differing opinions in the literature as to whether or not carp directly affect turbidity, a discussion of them follows.

In semi-permanent ponds in Mexico, Zambrano et al. (1999) found that Secchi depth was smaller, and the total amount of suspended solids were higher in ponds where carp were present than in ponds without carp. Drenner et al. (1998) found greater turbidity in the presence of carp, and Roberts et al. (1995) found that the increase in turbidity associated with the presence of carp, led to decreased light penetration. Lougheed and Chow-Fraser (2001) found that after excluding all large (>40 cm) carp from a large coastal wetland the turbidity throughout the marsh decreased by at least 45%. However, Angeler et al. (2002) found no change in Secchi depth with the presence of carp.

The density of carp in a system plays a role in determining the effect carp have on turbidity. Several studies have found a positive association between carp density and turbidity, or a correlated parameter such as sedimentation rate or the amount of total suspended solids. A study in The Netherlands on the impact of benthivorous fish in ponds, where little or no macrophytes were present, found that sedimentation rates and the amount of total suspended solids increased with increasing carp density (Breukelaar et al. 1994). As well, both Lougheed et al. (1998) and Badiou (2006) found a positive linear relationship with carp density and turbidity. Robertson et al. (1997) and King et al. (1997) found that sedimentation rates and turbidity were greater in areas of high carp density than in areas of low carp density.

A threshold density may exist which must be reached before carp will cause an increase in turbidity. Zambrano and Hinojosa (1999) suggest that this is when carp densities are sufficiently high, such that they lead to high intraspecific competition. This leads to increased foraging and sediment disturbance, resulting in a large amount of sediment being suspended into the water column. In their study, they found that a carp density greater than 0.8 carp/m^2 was sufficient to cause a change in turbidity.

Sediment type may also influence the impact of carp on turbidity (King et al. 1997). Crivelli (1983) observed no difference in turbidity in the presence of carp, and suggested that turbidity was dependent on the substrate type and weather conditions. Sediments that are more easily disturbed and that are retained in the water column for a longer period of time, are more likely to contribute to turbidity than sediments that are more difficult to disturb and that settle out quickly. Therefore, carp are more likely to lead to an increase in turbidity in an area where the sediments are soft and loose with a lot of organic matter, or sediments with a high clay component, than in an area with hard compact or sandy sediments (Scheffer 1998).

It should be noted that wind suspension of sediment is an important contributor to turbidity; however, if there is a sufficiently high density of carp and the sediments are of a type that is easily disturbed, the effect of carp on turbidity can take precedence over the effects of the wind (King et al. 1997). The effects of carp and wind on turbidity are additive.

The presence or absence of submersed vegetation may also influence the impact carp have on turbidity. Roberts et al. (1995) found that carp increased turbidity; however, the magnitude of the increase was reduced in the presence of vegetation. In the previously mentioned study by Breukelaar et al. (1994) where little or no macrophytes were present, a change in turbidity was easily observed, suggesting that wind was a major factor. As well, in the absence of vegetation, the removal of carp has been shown to a decrease the amount of inorganic suspended sediment in the water column (Barton et al. 2000). In an enclosure study where any effect by wind was negated, Kolterman (1990) found that common carp did not directly affect turbidity. He suggested that the influence that carp exert on turbidity may be indirect through the removal of vegetation, thereby allowing greater wind action on the sediments.

There are differing opinions in the literature as to whether or not carp directly affect turbidity. This appears to be attributable to differences in the density of carp, the type sediment, and the presence or absence of submerged vegetation in the study. However, regardless of the mechanism, the consensus is that given the right conditions, carp will tend to increase turbidity.

Common carp may impact nutrients through sediment disturbance and/or excretion. In wetlands and other aquatic environments, a large amount of nutrients is typically tied up in the sediments (Mitsch & Gosselink 2000). Carp, as benthivorous fish, are assumed to stimulate the release nutrients that are contained in the sediments into the water column. One way they are thought to do this is through sediment disturbance during feeding and spawning. Not only does this suspend sediment and therefore nutrients, but by disturbing the sediment water interface, carp are thought to disrupt any mechanisms that may be retaining the nutrients, particularly P, in the sediments and allow nutrients diffuse into the water column (Scheffer 1998). Carp may also influence nutrients dynamics through excretion. By feeding in the benthos and excreting their wastes in the water column, carp essentially act as a pump, taking nutrients from the sediments and introducing them to the water column (Scheffer 1998). However, whether or not this is a significant contribution is debatable. Lamarra (1975) determined that carp excretion can be a source of internal P loading in lakes, while Nuttall and Richardson (1991) determined that the excretion of P and N by carp can be considered a negligible contribution to the overall addition of nutrients in a natural system. However, based on nutrient excretion experiments, Badiou (2005) found that nutrient loading by carp at Delta Marsh was significant and that at a density of 200 kg/ha, nutrient excretion by carp would be similar to the external nutrient load of Lake Manitoba.

N and P are the two nutrients that are of most interest in the literature; however, whether or not carp change their dynamics seems to vary with the study. King et al. (1997) observed higher total amounts of P in the presence of higher densities of carp.

In an enclosure experiment Angeler et al. (2002) found higher concentrations of total P (TP) and total N (TN) in the presence of carp; however, they observed no correlation between the presence of carp and the concentrations of ammonia-N and soluble reactive P (SRP). They suspected that the lack of observed difference in ammonia-N and SRP was due to rapid uptake of the ammonia and the adsorption of P to suspended particles.

In a mesocosm experiment using small (< 8 cm) carp Richardson et al. (1990) observed an increase in soluble nutrients (nitrite-N and SRP) in the presence of carp. The authors proposed that the increase in nutrients was due to excretion by carp and not bioturbation as there was little sediment accumulated throughout the experiment.

Lougheed et al. (1998) found a predictable positive relationship between the density of spawning carp and nutrient levels (ammonia-N, TP and SRP) in enclosure studies; however, when large carp were excluded from the site where the experiments were conducted, no change in either total nitrate N or total P was observed (Lougheed & Chow-Fraser 2001).

At Delta Marsh, Badiou (2005) found that in a natural system the concentration of nutrients (ammonia-N and TRP) increased in the presence of carp, but only at high densities (>300kg/ha). However, he found that in enclosure experiments that ammonia-N was proportional to carp density, while TRP decreased in the presence of carp.

Contrary to the above studies, Zambrano et al. (1999) found that the presence of carp did not affect the nutrient concentrations in Mexican ponds and conclude that the carp did not influence the nutrient levels either by excretion or release from the sediments. However, the effect of carp in this case may have been insignificant due to the hypertrophic conditions, likely induced by agricultural inputs, which existed in all of the ponds.

2.3.2 Wetland Sediments

There is very little research on the impacts of carp on wetland sediments. However, Cahn (1929) mentioned that the draining of a soft-bottomed lake in which the fish population was determined to be approximately 95% carp, revealed a pock-marked bottom, presumably from where the carp had sucked-up the sediments in the search for food.

In a 1997 study, sediment levels of P, C, and N were not influenced by the density of carp (Robertson et al. 1997). However, this was a short term experiment where carp densities were reduced in part of the wetland for a season, where long term carp impact may have already altered the sediment chemistry. The authors suspected that a much longer time frame is needed to determine if carp do indeed influence the sediment chemistry of wetlands.

King et al. (1997) observed higher total amounts of P in the water column in the presence of higher densities of carp; however, in a companion study there appeared to be no differences in the nutrient concentrations in the sediments (Robertson et al. 1997).

2.3.3 Algae

By altering the availability of light, and/or nutrients, as well as the density of grazers, carp can influence algal growth. This effect of carp is indirect, rarely do carp affect algal growth directly. In order for carp to directly influence algal growth, it would be necessary for them to actively consume algae or disturb its substrata.

Many studies have found that phytoplankton biomass is higher in the presence of carp (Richardson et al. 1990; Drenner et al. 1998; Angeler et al. 2002; Khan et al. 2003, Badiou 2005), and that higher densities of carp lead to higher concentrations of phytoplankton and more intense algal blooms (King et al. 1997). However, one study found no relationship between carp density and phytoplankton growth in a Lake Ontario coastal marsh (Lougheed et al. 1998). The effect of carp on other types of algae is not as clear.

In Australian billabongs, Robertson et al. (1997) observed lower amounts of periphytic algae where higher densities of carp were present. The altered periphyton growth was believed to be due to light limitation and nutrient limitation resulting from competition with phytoplankton in high carp densities. However, in an aquarium experiment Sidokewicj et al. (1999) found that carp stimulated the growth of periphyton. In this case the increase in algal growth was facilitated by the removal of a periphyton grazer from the system, and the addition of nutrients through excretion and sediment disturbance in the presence of carp. When discussing the effect carp have on periphyton growth, it is important to consider how much of the periphyton mass is actually algal in origin. Carp may be perceived to increase periphyton growth where fine sediment conditions exist as a large component of the periphyton can be of an inorganic origin (Sidorkewicj et al. 1999).

There have been few studies to determine the effect of carp on sediment associated algae, and their findings are contradictory. Robertson et al. (1997) found that carp did not effect the growth of algae associated with the sediments. However, Zambrano et al. (1999) observed significantly lower amounts of epibenthic organisms in the presence of carp. Although they did not quantify the algal component, dietary analysis showed that the carp in the latter study were grazing on epibenthic algae.

2.3.4 Submersed vegetation

One of the most studied effects of carp is their impact on submersed vegetation. Anecdotal and experimental evidence has shown that carp have generally a negative impact on submersed vegetation.

There are many anecdotal descriptions of the impact carp have on submersed vegetation. In the earlier mentioned account by Cahn (1929), a previously well vegetated lake was absent of all vegetation within thirteen years of the introduction of carp; researchers of the impacts of carp have noted that vegetation is often seen floating on the surface of the water in areas where carp are present (Crivelli 1983; Panek 1987); and in a survey of Cootes Paradise, a coastal wetland marsh on Lake Ontario, dense, diverse stands of submersed macrophytes were always found in areas where carp were absent (Lougheed et al. 1998).

There is ample experimental evidence that carp have a negative effect on vegetation. King and Hunt (1967) concluded from an exclosure experiment in a Lake Erie marsh that carp were detrimental to submersed vegetation. They repeatedly found both a significantly greater amount and diversity of plants inside their exclosures than outside and Winkel and Meulemans (1985) presented evidence that the presence of benthivorous fish such as carp has resulted in the decline of submerged vegetation in several Dutch lakes. Robel (1962) and Crivelli (1983) found a strong negative correlation between the biomass of carp and the amount of submersed vegetation present. Drenner et al. (1998) observed an overall lower biomass and diversity of submersed vegetation when carp were present. Zambrano et al. (1999) showed that in Mexican farm ponds the density of submerged vegetation was lower in ponds that were stocked with carp than in ponds that were not used for aquaculture; they attribute the difference to direct disturbance by carp.

Carp have a direct and/or indirect negative effect on submerged vegetation. Carp may directly affect vegetation by disturbing the root zone during foraging, leading to a loss of vegetation by uprooting (Crivelli 1983); physically damaging plants through browsing (particularly on seedlings) (Sidorkewicj et al. 1999); and/or suspending sediments leading to collision injury to the plants in areas with a gravel substratum (Sidorkewicj et al. 1999). Carp may indirectly affect submersed vegetation through light limitation. As mentioned previously, the presence of carp tends to lead to higher turbidity. In turn, high turbidity leads to lower light levels, which may negatively affect plant growth and survival by a positive feedback. Direct physical disturbance and the loss of vegetation associated with carp may itself contribute to increased turbidity both by decreasing the surface area for particles to settle upon, as well as by decreasing the stability of the sediments (Scheffer 1998). Carp may also indirectly decrease the abundance of submersed vegetation, through decreased light availability, by stimulating the growth of epiphyton (Sidorkewicj et al. 1999).

The effect of common carp on submersed vegetation is greatest in the spring. Considerable destruction of the plants is believed to occur during carp's vigorous spawning activities and both directly and indirectly through feeding activities when the plants are young (King & Hunt 1967). As well, submersed vegetation will more likely be subject to destruction by carp in shallow water where carp prefer to spawn and feed (Crivelli 1983).

Some types of submersed vegetation may be more susceptible to destruction by carp depending on their anchorage (King & Hunt 1967; Crivelli 1983) and palatability (King & Hunt 1967). Plant species with a strong root structure will be more likely to withstand the influence of carp than weakly rooted types (Crivelli 1983). In a small study on the food habits of carp with respect to vegetation, King and Hunt (1967) found that carp will eat large amounts of *Chara*, whereas the largest effect on "pondweed types" is through uprooting.

Charophytes appear to be sensitive to the presence of carp. In their experiments, King and Hunt (1967) found that *Chara*, of all the species in the study, was the most affected by the presence of carp. In every replicate they found more *Chara* inside than outside their exclosures. Roberts et al. (1995) also found that while some macrophytes appeared to be unaffected, no *Chara* remained soon after carp were added to their experimental enclosures, and Drenner et al. (1998) found that *Chara* was always absent in the presence of carp.

Myriophyllum sp. (Roberts et al. 1995; Drenner et al. 1998), *Potamogeton* spp., and *Ceratophyllum* sp. (King & Hunt 1967) appear to be the species of submersed vegetation that are the least effected by the presence of carp. In areas of high carp density the only submerged or floating vegetation may be small stands of sago pondweed (King & Hunt 1967).

Vegetation can however, rapidly re-colonize an area following the removal of carp. In Cootes Paradise, a dramatic regeneration of submersed vegetation was observed, and plant species that had disappeared since the introduction of carp were present within the first year after carp removal (Lundholm & Simser 1999). King and Hunt (1967) observed a 3000 % increase in the density of *Chara* within two months of carp removal.

2.3.5 Impact Summary

Separating the individual effects from the overall impact common carp can have in a wetland is difficult. As is the typical case in an ecological system, all components of the wetland interact. Changes to one component of the system can exert a positive or negative pressure on another component(s) of the system. For example, Zimmer et al. (2003) showed that macrophyte abundance has a strong influence on phytoplankton biomass and nutrient levels in prairie wetlands. The pressures being exerting on the system can come from below such as nutrients or light availability (i.e. bottom-up effects), or pressures can come from above such as predation (i.e. top-down effects).

Common carp can exert a bottom-up effect on a system (Zambrano et al. 1999). By disturbing the sediments through feeding and spawning, carp can uproot submersed vegetation and suspend sediments and nutrients into the water column. This may lead to increased turbidity and increased phytoplankton growth. In such a system much of the vegetation may be lost, as it is successively replaced by phytoplankton.

Carp can also exert top-down pressure. Work by Khan et al. (2003) indicates that grazing pressure by juvenile carp can suppress the zooplankton community (especially the large cladocerans) such that the phytoplankton community is released. The presence of carp causes a top-down cascade effect resulting in an algal bloom. They found that the without the decreased grazing pressure the presence of higher nutrient levels that would be associated with carp excretion were not sufficient to cause a cascade alone (Khan et al. 2003).

The two examples above show how carp can have either a bottom-up or top-down effect on a wetland. However, it is important to note that the distinction between of top-down and bottom-up can be difficult to define. In similar enclosure studies Angeler et al. (2002) and Khan et al. (2003) both found that phytoplankton biomass was higher in the presence of carp. However they attributed the change to different pressures. The former suggests that the increase was due to higher nutrient levels, while the later attributes the change to a decrease in grazing pressure by zooplankton. What is interesting is that the results of both studies indicate that there were more nutrients available and a lower biomass of zooplankton, particularly large cladocerans, in the presence of carp. This situation illustrates that a wetland ecosystem is complex and that even under the controlled conditions of an enclosure experiment it is difficult to separate the mechanisms from one another. What is most likely in these cases is that the carp were exerting both bottom-up and top-down pressures.

2.4 *Alternative Stable States*

Scheffer (1990) proposed that shallow aquatic systems exist in two stable-states, a clear-vegetated state and a turbid-unvegetated state. He suggests that these states exist for three reasons: 1) in a eutrophic system macrophyte growth is typically limited by light rather than by nutrients, however algal growth will typically increase with increasing nutrients; 2) submerged vegetation has a negative effect on turbidity; and 3) the maximum depth at which submerged vegetation can grow is dependent on turbidity, therefore in a shallow system there exists a "critical turbidity" at which vegetation will not grow.

Scheffer (1990) suggested that at low nutrient levels the system always exists in the clear state, while at high nutrient levels the system will be in a perpetual turbid state. He suggests that there exists an intermediate level of nutrients at which either stable-state can exist and a switch from one state to another may occur if the "critical turbidity" is reached. In the clear state, as nutrients are added to the system, the equilibrium becomes less stable and a perturbation in the system may cause a shift to the turbid state. Once the system is in the turbid state, the same mechanisms that functioned to maintain the clear state maintain the system at a new turbid equilibrium (Scheffer 1990). The ability of two stable-states to exist under similar nutrient conditions indicates that if the system is currently in the turbid state, the switch from the clear water state was not caused only by eutrophication but by an additional cause, such as carp. Scheffer (1990) believes that if two stable-states can occur within the same system, a manipulation to remove or reduce the perturbation can be sufficient to improve or return the system to a clear water state.

2.5 *Wetlands*

Wetlands are ecosystems that elude precise definition as there are considerable variations from one specific wetland to another. Typically, wetland definitions are based on components of hydrology, soils and vegetation (Mitsch & Gosselink 2000). According to the Canadian Wetland Classification System (Warner & Rubec 1997) a wetland is a "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment." Wetlands are often considered to be ecotones; a transition zone between terrestrial and aquatic habitats. This position leads to the high biodiversity that is often associated with wetlands, as species from both habitats are borrowed by the wetland (Mitsch & Gosselink 2000).

Freshwater coastal wetlands, or lacustrine marshes (Warner & Rubec 1997), are wetlands that are associated with the boundaries of lakes. The wetland may have permanent connections with the lake that allow water and biota to be exchanged freely, or the wetland may only be influenced by the lake through the groundwater flow or flood events. Such wetlands are often used as a migration stop and breeding ground for waterfowl and other birds. As well, coastal wetlands, particularly those with connections to the lake, are an important spawning and feeding areas for many species of fish.

2.6 *Experimental scale and the need for ecosystem-level studies*

Ecological experiments can be conducted on various scales, with each scale having its advantages and disadvantages. Laboratory experiments are often used by biologists because of the ability to control many variables and the ease of replication. Such experiments are typically inexpensive and yield highly statistical results (Carpenter 1996). However, the ecologist will rarely conduct experiments in the laboratory due to the unnaturalness of the situation. Instead, mesocosm experiments are often used because they offer the researcher the benefits of control, replicability, relative low cost, and the high level of statistical confidence of a laboratory experiment under more realistic conditions (Carpenter 1996; Schindler 1998). Mesocosm experiments typically can be conducted quickly and often allow the researcher to stay out of the field (Carpenter 1996).

The disadvantages of mesocosm experiments are related to how representative they really are, and therefore their applicability to natural systems. When designing an experiment it is difficult to anticipate every variable, and therefore mesocosms are typically poor simulations of ecosystems (Schindler 1998). They usually lack all of the trophic levels and functions of a real ecosystem. Often, keystone and/or large migratory species cannot be included due to size or mobility constraints of the experimental design (Schindler 1998). They can exclude key ecosystem processes, and in some cases, experimenters have made conclusions based on results that occurred only as an artefact of the design (Schindler 1998). Mesocosms have limitations to size and time which may cause distortions in the results of the experiment (Carpenter 1996). The larger the mesocosm, the closer to the natural system it can be; however, researchers have found that the structure of the mesocosm itself can introduce changes to basic functions such as chemical and algal dynamics, and organism behaviour (Bloesch et al. 1988). A larger sized mesocosm does not guarantee that all important processes will be included in the experiment (Schindler 1998). For these reasons, Schindler (1998) cautions that larger mesocosms do not make an ecosystem level experiment. To support this view, he cites many sub-ecosystem scale experiments that he and his colleagues have conducted in which the results were very different from those in similar whole lakes experiments. The information obtained by mesocosm experiments can therefore not often be extended to predict what would happen in the natural environment (Schindler 1998). Bloesch et al. (1988) caution that biomanipulation experiments that are conducted in mesocosms should not be extrapolated to whole lakes on a quantitative basis, but they point out that useful qualitative information can be gained.

Mesocosm research has provided the basic understanding of ecosystem processes needed to conduct whole-ecosystem experiments (Drenner & Mazumder 1999). These types of experiments can be useful to our understanding of ecology, but their results must always be checked by, and considered within, the results of long-term, large-scale studies (Carpenter 1996). The researcher should remember that experiments that include more than one species are closer to natural reality than are single species experiments. As a demonstration of this, Richardson et al. (1990) found interactions between a zooplanktivore and a benthivore in a two-species mesocosm experiment that could not have been predicted based on the results of similar experiments with each species alone. Carpenter (1996) points out that while mesocosms are a important tool for ecologists because of their speed, ability for replication, statistical power, and the insight they provide into the mechanisms at work within ecosystems, they have strong limitations which make it important to also conduct field studies.

Field studies at the ecosystem-level are an important source of information due to their temporal and spatial scales (Schindler 1998). These types of experiments are truly natural studies; they study the ecosystem itself as a whole. There is no problem, as with smaller scale experiments, that some physical, chemical, or biological parameter may be excluded (Schindler 1998). Unlike small scale experiments, ecosystem-scale experiments are not limited by a time frame, and can and should be conducted for as long as possible in order to gather all important information about the natural temporal variation that exists in ecosystems (Schindler 1998). Despite the fact that ecosystem-scale experiments yield more realistic results, as with the mesocosm approach, this method also has disadvantages. Whole-ecosystem experiments are typically expensive to conduct, difficult to undertake, finding good replicates is nearly impossible, and it is often difficult to ensure other uses of the ecosystem do not influence the results (Schindler 1998).

The advantages and disadvantages of each scale has sparked much debate about the relative benefits and importance of experimental type. Carpenter (1996) presents a passionate argument for the use of field experiments. Drenner & Mazumder (1999) argue against Carpenter (1996) and for the benefits of mesocosms. They state that mesocosm research has provided the basic understanding of ecosystem processes needed to conduct whole-ecosystem experiments (Drenner & Mazumder 1999). They do agree with Carpenter that caution must always be taken in extrapolating from small scale, microcosm, studies to large natural systems that are more complex and contain a higher biodiversity. Drenner & Mazumder (1999) believe that the microcosm vs. field experiment debate continues due to a lack of experimentation comparing the relative virtues of each scale. Although both sides of the debate are strong in their opinions, it must be remembered that important knowledge of aquatic ecology has been provided from research conducted at all scales (Schindler 1998).

Chapter 3: Methods

3.1 Study Site

This study was conducted at Delta Marsh in south-central Manitoba, Canada. Delta Marsh is an 18, 500 ha freshwater coastal wetland located on the south shore of Lake Manitoba (Figure 3.1). It is best classified as a lacustrine shore marsh according to the Canadian Wetland Classification System (Warner & Rubec 1997). It is protected from the lake by a beach ridge that runs along the north end of the marsh. There are four main connections between the lake and the marsh along this ridge: Deep Creek, Cram Creek, Delta Channel, and Clandeboye Channel. Through these connections, the marsh is subject to frequent changes in water levels due to seiche tides on Lake Manitoba. By this description, Delta Marsh may be more specifically classified as a barrier protected wetland, a wetland with channels according to Mooney and Goldsborough (*unpublished data*).

Historically, Delta Marsh has been an important area to many people in the area. It has been used as a place for hunting and vacationing since at least the late 1890s (Goldsborough *pers comm*). Over the last few decades the health of the marsh has been observed to be in decline (Brown 2003). This is likely due to a combination of pressures. With the completion of the Fairford Dam on Lake Manitoba in 1961, water level stabilization has changed the hydrologic regime of the marsh. This has reduced the range of the alternating wet/dry cycle that is

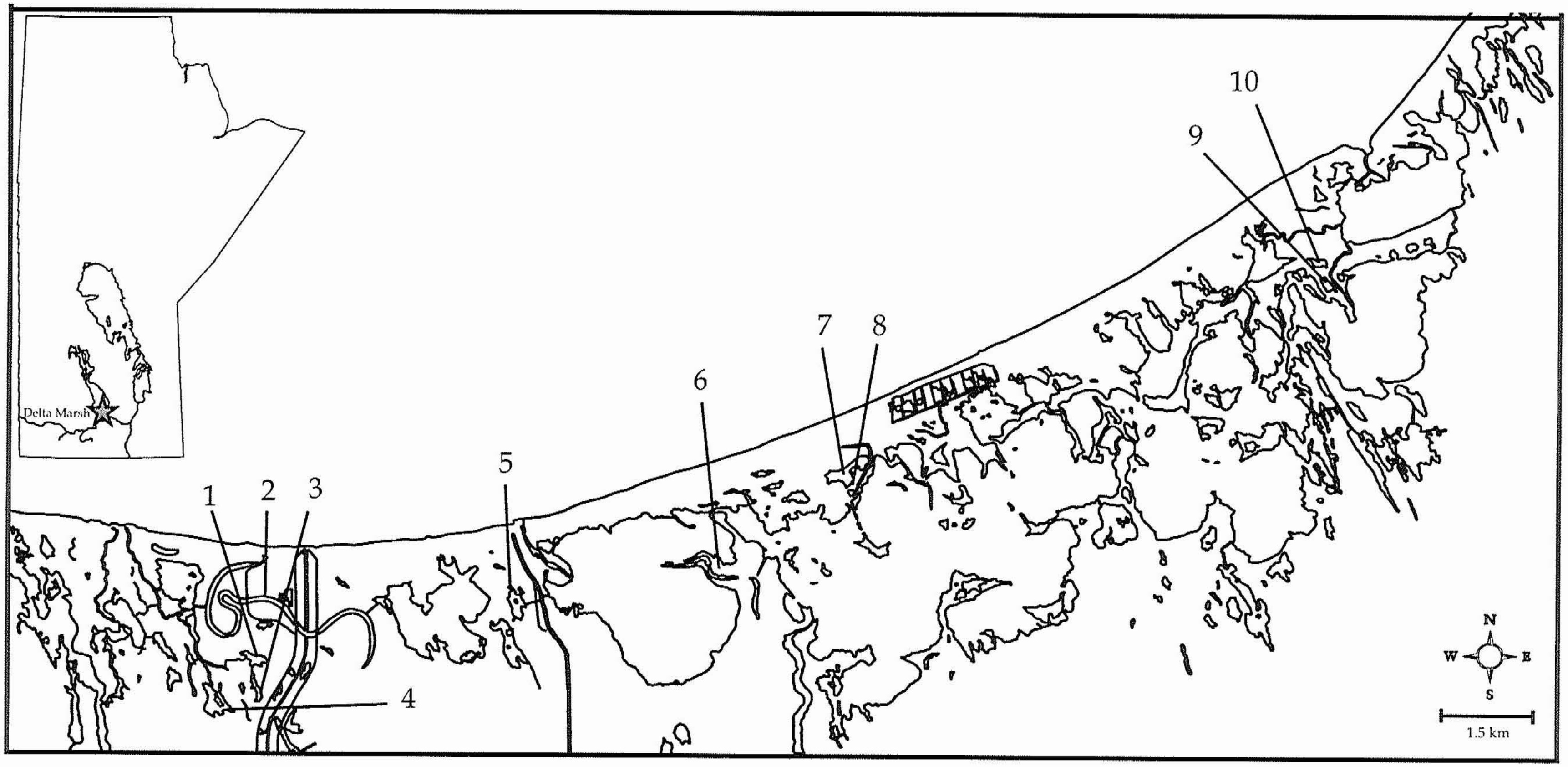


Figure 3.1: Map of Delta Marsh showing the location of the ten ponds used in the study. Inset shows the relative position of the marsh within the province of Manitoba. (C) indicates a connected pond, and (I) an isolated pond based on the condition of the pond at the start of the study. 1 - Section 5 Bay (C); 2 - Mid-Blind Channel (C); 3 - Mad Woman Bay (C); 4 - Thompson's Bay (C); 5 - North School Bay (C); 6 - South Pitblado's Channel (C); 7 - North MacKenzie Bay (I); 8 - South MacKenzie Bay (I); 9 - Wye's Pond (I); 10 - Emile's Pothole (I).

important for the health of a marsh (van der Valk 1981). Shay et al. (1999) believed that stabilized water levels have aided the rapid colonization of the marsh by *Typha X glauca,* a hybrid that does very well under the current high and stable water levels of the marsh (Waters & Shay 1992). The creation of the Portage Diversion in the late 1960s introduced water and biota from the Assiniboine River watershed into the lake and consequently the marsh (Stewart et al. 1985). These waters would have previously gone on to Lake Winnipeg and not entered Lake Manitoba directly. High density areas of residential development, in the form of year-round houses and seasonal cottages, exist along the beach ridge. Some of these residences use outhouses and/or septic systems, both of which may leak, affecting the levels of N and P in the marsh and the lake (Brown 2003). More than half of the farms in the area apply fertilizer to supplement the levels of both N and P in the soil as well as apply pesticides to their lands. Combined with the residential development, local farming practices may be impacting the marsh's water quality (Brown 2003).

Common carp were first observed in Delta Marsh around 1950 (Wrubleski 1998). Ten years later, waterfowl hunters began to notice that the amount of submersed vegetation in the marsh was declining; they attributed this loss to the abundance of carp. In response, Ducks Unlimited constructed barriers in 1964 at the entrances of Delta Channel and Clandeboye Channel in an attempt to prevent carp access (Wrubleski 1998). The barriers were made of 5.1 cm mesh (Wrubleski 1998), which prevented large fish from entering the marsh while not impeding water flow. Unfortunately, the barriers required frequent upkeep and only functioned for two seasons - 1965 and 1966. The carp barriers, though they may not have functioned fully, remained in place until 1982 when they were finally removed (Wrubleski 1998). These barriers may have been able to reduce the number of carp in the marsh, but they may also have prevented other large fish species from entering the marsh.

Delta Marsh consists of a series of open bays, surrounded by many small ponds, some that are connected and others that are isolated from the main marsh. The ponds that are connected to the main marsh presumably are accessible to carp, while the isolated ponds are not. A preliminary survey of these ponds in 2000 (Goldsborough and Wrubleski *unpublished data*) described the connected ponds as "turbid" with "carp present" while the isolated ponds were described as "very clear" and the presence of submersed vegetation was noted. The analysis of water samples collected from these sites confirmed that the connected ponds were significantly more turbid, had higher amounts of suspended solids, and greater concentrations of phytoplankton than the isolated ponds (Table 3.1). The presence of carp, or possibly the exchange of turbid waters from the large open areas of the marsh, were suspected to be the causes of these observed differences. This situation of isolated and connected ponds was seen as an excellent opportunity to determine the impact of carp on the marsh and led to the development of the current study.

Table 3.1: Selected parameters from the annual Delta Marsh water quality survey in 2000 (Goldsborough & Wrubleski, unpublished data). Shown are the means (± SE) of the connected (n=10) and isolated (n=9) ponds that were sampled. Means were based on the average value of the May, June and July samples for each pond.

Parameter	**Connected Ponds**	**Isolated Ponds**	**p-value**
Turbidity (NTU)	19.1	5.9	0.0002
	(1.9)	(2.0)	
Total Suspended Solids; TSS (mg/L)	43.5	14.9	<0.0001
	(3.8)	(4.0)	
Organic % of TSS	65.6	94.0	<0.0001
	(2.8)	(3.0)	
Total Chlorophyll *a* (µg/L)	109	22.3	<0.0001
	(7.8)	(8.2)	

3.2 Experimental Design

Ten sites were selected for the study (Table 3.2, Table 3.3, Figure 3.1). Site selection was based on a several criteria. The first criterion was accessibility of the site. Some areas of the marsh are remote and travel to the sites can be difficult. Therefore, only sites within a reasonable travel time (two hours) from the Delta Marsh Field Station (DMFS) were studied. This was done to ensure water samples could be collected and brought back to the DMFS for analysis of time sensitive parameters. The second criterion was proximity to other sites. An attempt was made to group sites geographically for ease of sampling and comparison. The third criterion was the ability to apply the treatments, described later in this section. For example, whether there was a channel leading into the pond that could be easily blocked off, or whether equipment could be transported to the site, would affect the ability to apply the treatments. Of the ten sites that were selected (Table 3.3), six were connected to the main marsh and were accessible to carp. The remaining four sites were isolated from the main marsh, and were presumed to be inaccessible to carp, and lacked overland water exchange.

The ponds were surveyed in 2001 to determine their baseline condition with respect to the parameters of interest. This was done to determine the characteristics of areas that had been exposed to carp for a long period of time

Table 3.2: General characteristics of the ponds used in the study. The data presented are a summary of the data collected throughout the present study.

Characteristic	**Description**		
Water depth	12 – 124 cm; mean = 63 cm		
Pond size	0.83 – 12.58 ha of open water; mean = 6.14 ha		
Substratum type	Saturated organic soil ("muck") with underlying glacial till.		
Surrounding cover	Consisted of emergent macrophytes: *Typha* X *glauca*, *Phragmites australis*, and rarely *Schoenoplectus* sp.		
Submerged vegetation	Density varies considerably: *Ceratophyllum demersum*, *Myriophyllum sibiricum*, *Stuckenia pectinatus*, *Utricularia macrorhiza*, *Chara* sp.		
Water quality	pH:	7.5 – 9.7	mean = 8.4
	Conductivity:	843 – 3599 µS/cm	mean = 1681
	Turbidity:	3.9 – 92 NTU	mean = 14
	Total N	3.4 – 11.4 mg/L	mean = 6.3
	Total P	0.2 – 1.9 mg/L	mean = 0.6

Table 3.3: Location (UTM coordinates for center of site; Zone 14), area of open water, and connectedness of ten ponds within Delta Marsh that were used in the study. Area of open water was determined using ArcView GIS software from orthophotos taken in 1991.

Pond	Easting (m)	Northing (m)	Open-Water Area (ha)	Relationship with main marsh prior to treatment
Mid-Blind Channel	544 293	5 558 385	6.77	Connected
Thompson's Bay	543 273	5 556 596	12.58	Connected
Section 5 Bay	543 980	5 557 416	5.21	Connected
Mad Woman Bay	544 036	5 556 895	3.31	Connected
North School Bay	548 533	5 556 895	9.01	Connected
South Pitblado's Channel	552 061	5 558 869	3.04	Connected
North MacKenzie Bay	554 260	5 560 636	11.85	Isolated
South MacKenzie Bay	554 393	5 560 338	0.83	Isolated
Emile's Pothole	562 643	5 564 395	2.69	Isolated
Wye's Pond	562 798	5 564 025	3.05	Isolated

(connected sites) versus ponds that were assumed to have not been recently exposed to carp (isolated sites).

In February of 2002, during which time the marsh is typically under snow and ice, each of the ponds was visited. If the pond was not frozen to the bottom, an under-ice water sample was collected. Samples were analysed for dissolved oxygen (Winkler titration; APHA 1998) to determine if there was a possibility of carp over-wintering in the ponds (Table 3.4).

Experimental treatments were applied to the ponds in the spring of the 2002 (Table 3.5). Four of the ten ponds remained unmanipulated. They were assumed to be subject to the same level of accessibility to carp as in the first year of the study and therefore acted as controls: two connected (Thompson's Bay and North School Bay) and two isolated (North MacKenzie Bay and Emile's Pothole). Experimental manipulations that altered carp access were carried out on the remaining six ponds.

Three experimental treatments were applied (Table 3.5). The first was the blasted treatment, which created channels from the main marsh into isolated ponds. The goal of this treatment was to permit carp access, as well as create a means for direct water exchange between the ponds and the rest of the marsh. The purpose of the remaining two treatments was to close off previously connected ponds to prevent access by large carp. These treatments included two

Table 3.4: Results of winter sampling (February 10-14, 2002) to determine if carp could have survived the winter in ten experimental ponds at Delta Marsh. Data were collected in the winter between the baseline and treatment open water seasons. Fish were considered to be unable to survive at less that 2.0 mg O_2/L (Wetzel 1983). *Indicates the number of samples collected.

Pond	**Avg. dissolved oxygen conc. (mg/L)**	**n***	**Possibility of fish survival?**	**Treatment**
Mid-Blind Chan.	7.0	2	yes	Connected - Screened
Thompson's Bay	frozen to bottom	2	no	Connected - Connected
Section 5 Bay	frozen to bottom	2	no	Connected - Screened
Mad Woman Bay	0.0	3	no	Connected - Isolated
North School Bay	frozen to bottom	2	no	Connected - Connected
S. Pitblado's Chan.	0.0	2	no	Connected - Isolated
North Mac. Bay	0.0	3	no	Isolated - Isolated
South Mac. Bay	frozen to bottom	1	no	Isolated - Connected
Emile's Pothole	9.0	2	yes	Isolated - Isolated
Wye's Pond	1.0	2	unlikely	Isolated - Connected

Table 3.5: The association of the ten ponds with the main marsh over the two years of the study.

Pond	**Status 2001**	**Status 2002**	**Treatment**	**Date of Treatment (2002)**
Thompson's Bay	Connected	Connected	None	
North School Bay	Connected	Connected	None	
South MacKenzie Bay	Isolated	Connected	Blasted	May 17
Wye's Pond	Isolated	Connected	Blasted	June 14
Mid-Blind Channel	Connected	Screened	Screened	April 22
Section 5 Bay	Connected	Screened	Screened	April 22
North MacKenzie Bay	Isolated	Isolated	None	
Emile's Pothole	Isolated	Isolated	None	
Mad Woman Bay	Connected	Isolated	Diked	May 2
South Pitblado's Channel	Connected	Isolated	Diked	April 24

levels of isolation: screened and diked. The goal of the screened treatment was to prevent large carp from entering the ponds while permitting water exchange. The second of the isolation treatments was the diked treatment. The goal of this treatment was to isolate the pond from the rest of the marsh by closing its connection. This would prevent any fish from entering the pond as well as stop direct water exchange with the marsh (i.e. surface flow). Each of the three experimental treatments was carried out on two ponds (Table 3.5).

3.2.1 Blasted Treatment (Isolated - Connected)

The blasted treatment was applied to South MacKenzie Bay and Wye's Pond. Since the four original isolated ponds from 2001 were geographically situated in pairs (see Figure 3.1), the treatment was assigned so that one from each pair would become connected. This was done so that each of the treatment ponds (T) would in be associated with its own reference pond (R): South MacKenzie Bay (T) with North MacKenzie Bay (R) (see Figure 3.2) and Wye's Pond (T) with Emile's Pothole (R) (see Figure 3.3).

The connections were made by blasting with ditching dynamite. In order to make sure the explosion would create a channel of the desired depth and width, the ground could not be frozen when the blasting was performed. In the spring of 2002 the ground at South MacKenzie Bay was thawed by the middle of May, almost a month before the ground at Wye's Pond. To help speed up the thawing time at Wye's Pond, all vegetation and litter was removed from the area of the

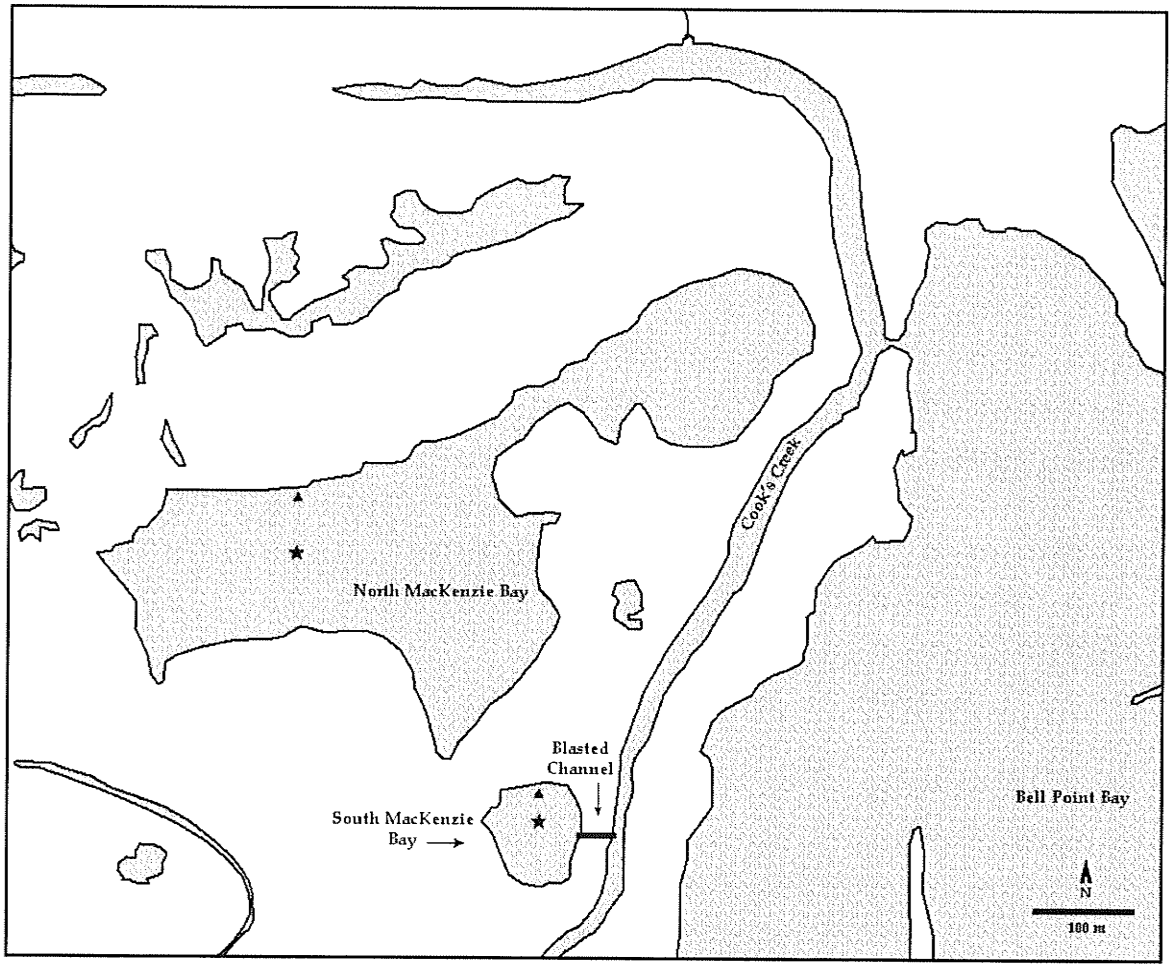


Figure 3.2: Map of the MacKenzie ponds used in the study. The blasted channel indicated was created in May of 2002 to allow fish and water to enter South MacKenzie Bay. The locations of the permanent water sampling sites (stars) as well as the periphyton enclosures (triangles) are indicated. See Figure 3.1 for the location of the ponds relative to the entire marsh.

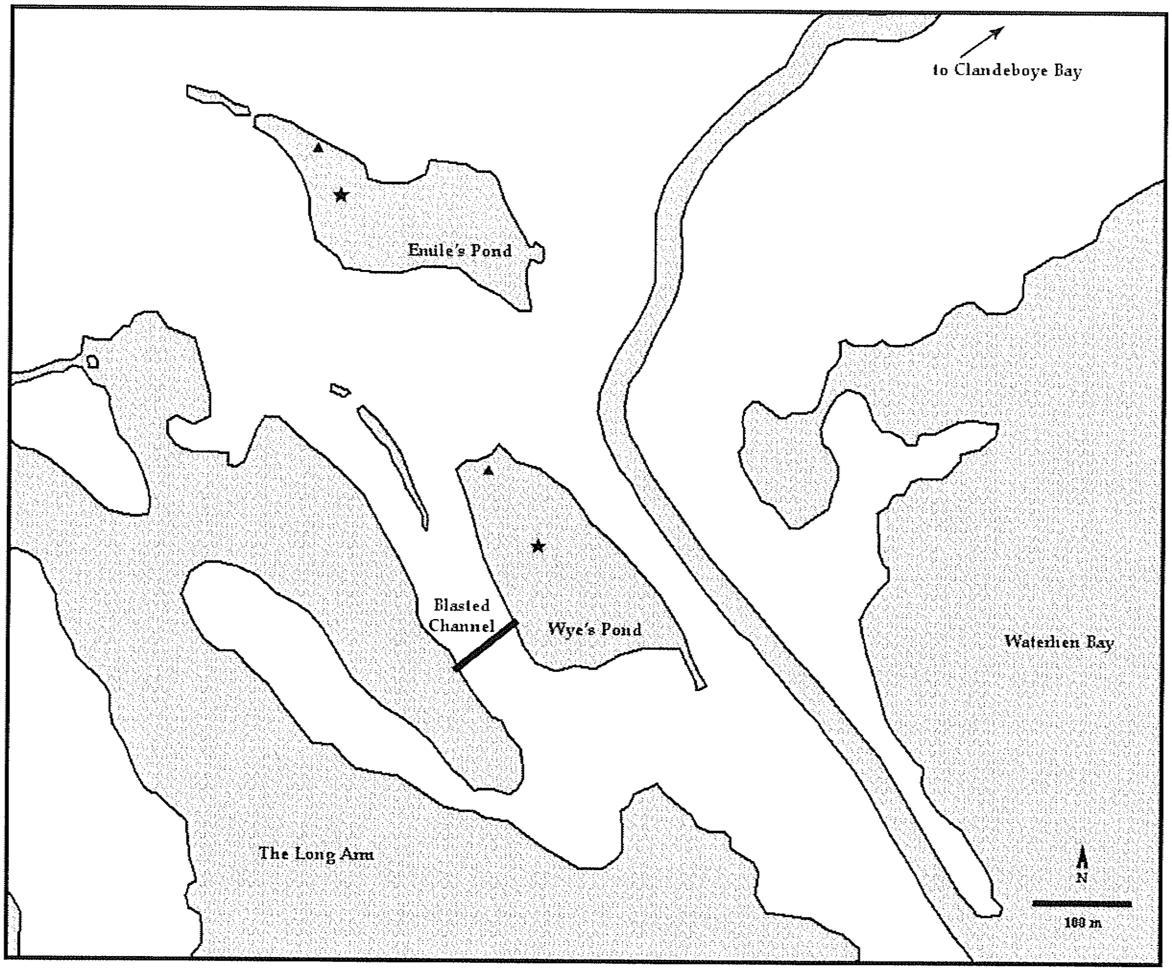


Figure 3.3: This map shows the ponds in the study that are located on the far east side of Delta Marsh. The blasted channel indicated was created into Wye's Pond in June of 2002 to allow fish and water to enter the pond. The locations of the permanent water sampling sites (stars) as well as the periphyton enclosures (triangles) are indicated. See Figure 3.1 for the location of the ponds relative to the entire marsh.

future channel. The vegetation was cut close to the ground and raked to the side along with the litter, so that bare ground was exposed. When the ground was soft enough that a stake could be easily pushed into the ground to the required depth for the explosives, experts from Manitoba Conservation did the blasting.

On May 17, 2002 the channel was created into South MacKenzie Bay from adjacent Cook's Creek (Figure 3.4). The ground at Wye's Pond was not ready for blasting until June 14, when the channel connecting it to the north-west side of Waterhen Bay was created.

The blasting was performed on days when there was little wind to ensure that minimal amounts of debris would be blown into the pond. The explosions were created using 2 kg (50 mm x 400 mm) sticks of dynamite manufactured by Powerfrac. The charges were placed approximately two meters apart and about 1 to 1.3 m in depth. The desired depth for the charges was just below the estimated ground-water level, such that the crater left by the explosion would fill with water. The blasting was done so that all debris landed on the ground next to the channel. Any large clods of soil remaining in the new channel were removed by hand and placed on the shore. The resulting channel was a similar depth (approx. 1 m) as the pond and the connected marsh. The channel at South MacKenzie Bay was approximately 1.5 m wide and 30 m long. The channel at Wye's Pond was approximately 2 m wide and 75 m long. At both sites, the water in the channel flowed out of the pond clearing away any muddy water after the blasting.

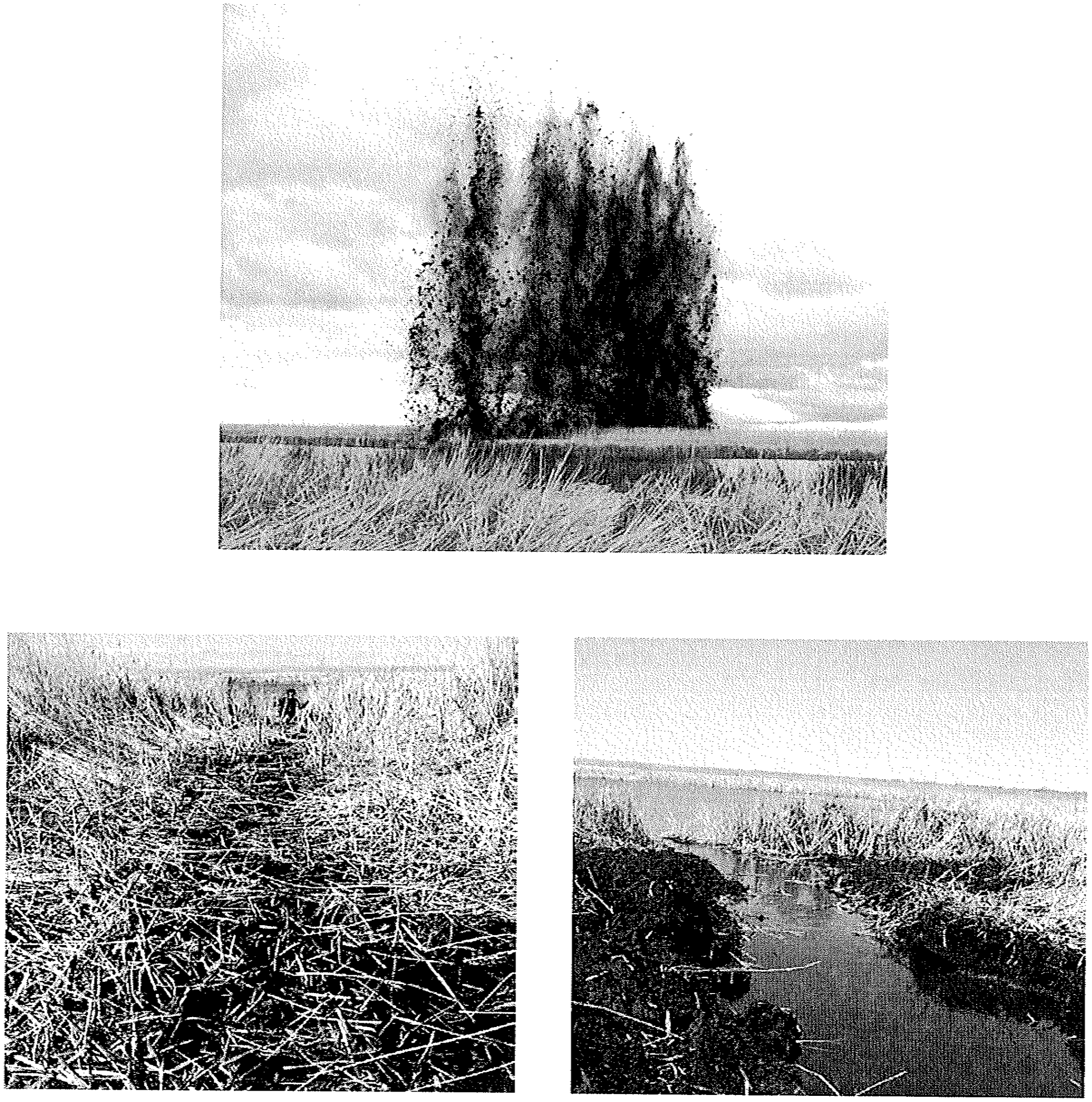


Figure 3.4: The pictures above show a time sequence of the creation of a connection between Wye's Pond and the main area of Delta Marsh. The connection was created by using dynamite to blast a channel in the upland area between the pond and adjacent Waterhen Bay on June 14, 2002. The same method was used to create a channel into South MacKenzie Bay from adjacent Cook's Creek.

3.2.2 Screened Treatment (Connected - Screened)

For the screened treatment, screens were deployed across the channel entering each site. Ponds were chosen that were most easily amenable to screen deployment. The ponds chosen were Mid-Blind Channel and Section 5 Bay. The screens that were erected were different at each of the ponds; however, they were both made such that there were 5 cm wide vertical spaces in the screen. The treatment was applied in the early spring of 2002 as soon as the ice was off the marsh.

Mid-Blind Channel is part of a larger channel that was partially isolated by a road causeway. The pond is located between the west bank of the Portage Diversion and the winter road entering Delta Mash Field Station (Figure 3.5). There are three culverts (two small at 75 cm in diameter, and one large at 90 cm in diameter) that run under the road and connect Mid-Blind Channel with the main marsh. The presence of these culverts facilitated the easy application of the screened treatment by limiting the area that needed to be screened.

Screens were placed on the culverts on April 22, 2002 (Figure 3.6). Collars supporting a steel frame (small: 75 x 82.5 cm; large: 90 x 97.5 cm) were bolted to the culverts on the west side of the road. Two steel grates of 1.5 cm diameter bars spaced 5 cm apart were placed inside each frame. The grates could be removed separately to remove accumulated debris while maintaining the integrity of the treatment.

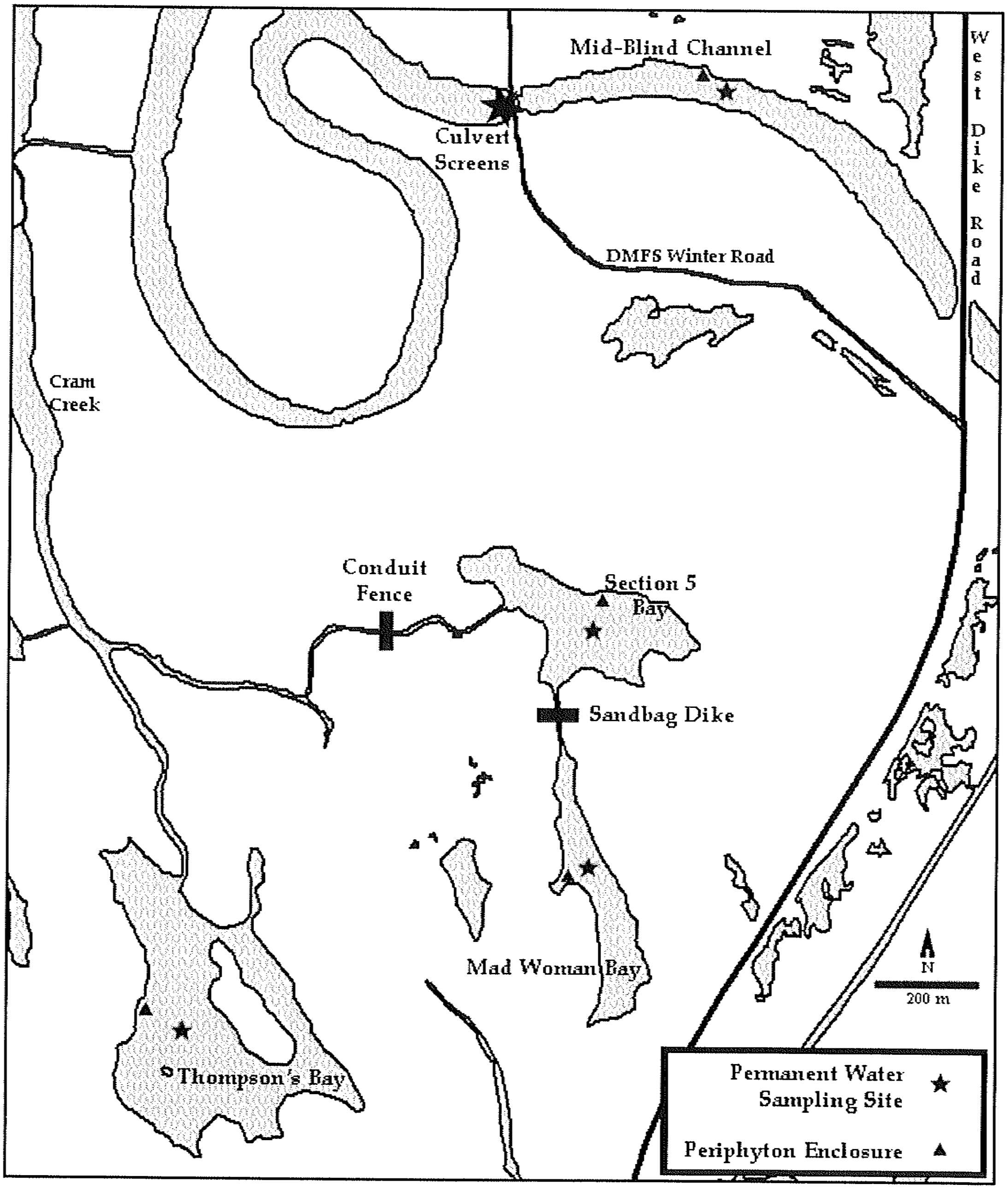


Figure 3.5: This map shows the ponds in the study that are located west of the Assiniboine River Diversion in Delta Marsh. The location of the treatments carried out prior to sampling in the spring of 2002, as well as the locations of the permanent water sampling sites and the periphyton enclosures are indicated. See Figure 3.1 for the location of the ponds relative to the entire marsh.

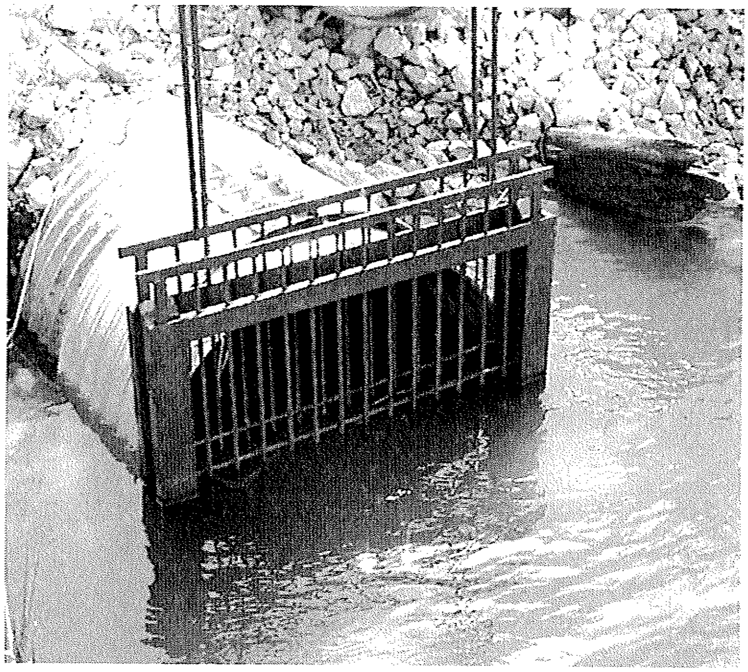


Figure 3.6: The photo shows the west-side of the culvert that runs under the road separating Mid-Blind Channel from the main area of the marsh. The removable grate was put in place for the spring of 2002 to prevent large fish from accessing Mid-Blind Channel. The pond is located on the opposite side of the road from where the photo was taken.

Section 5 Bay is located near two other connected ponds used in the study: Thompson's Bay and Mad Woman Bay (Figure 3.5). Thompson's Bay remained unmanipulated (as a reference pond) while Section 5 Bay was fenced and Mad Woman Bay was diked (see section 3.2.3).

A conduit fence was erected in the channel leading from Cram Creek to Section 5 Bay (Figure 3.7a). The fence spanned the entire width of the channel and was put in on the same day as the screens at Mid-Blind Channel, April 22. Figure 3.7b shows the basic construction of the fence. The fence was sunk into the sediments such that the lowest part of the fence rose out of the water approximately 30 cm. To ensure the integrity of the treatment was maintained in the event the water level rose greater than the height of the fence during a possible storm event, a 5 cm wire mesh was attached to the top of the fence extending its height another 30 cm. Along the edges of the fence, erosion had previously undercut the bank. To prevent the possibility of fish by passing the fence through such holes, sand bags were piled under the banks. The sand bags were further reinforced and supported by conduit pipes that were pushed through the bank down through the open space to the sediments.

3.2.3 Diked Treatment (Connected – Isolated)

Mad Woman Bay (Figure 3.5) and South Pitblado's Channel (Figure 3.8) were isolated from the surrounding marsh by sandbag dikes. The decision for the application of this treatment to Mad Woman Bay is explained previously (see Section 3.2.2). South Pitblado's Channel was chosen to be diked over the

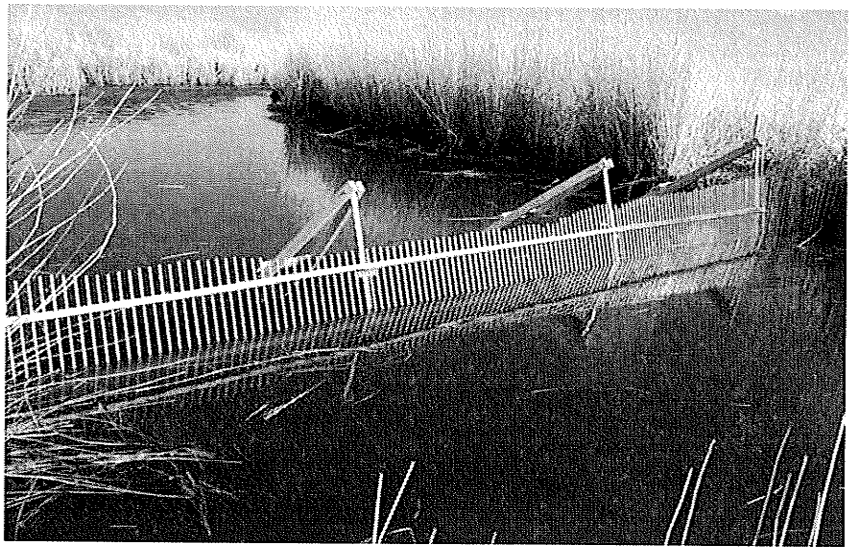


Figure 3.7a: The photo above shows the conduit fence on Cram Creek that was erected to prevent large fish from accessing Section 5 Bay during the summer of 2002 (Figure 3.5). The picture was taken shortly after construction, soon after wire mesh was added to the top of the fence to extend its height. The channel width was 10.7 cm.

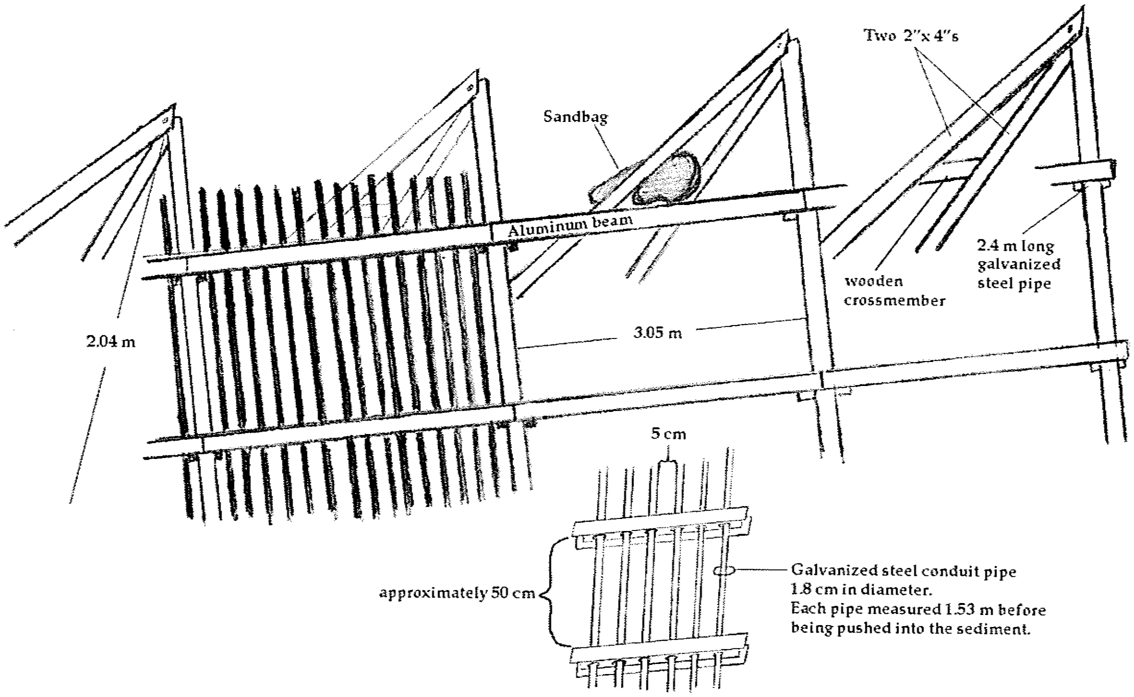


Figure 3.7b: A diagram of the fence that was constructed preventing large fish from entering Section 5 Bay.

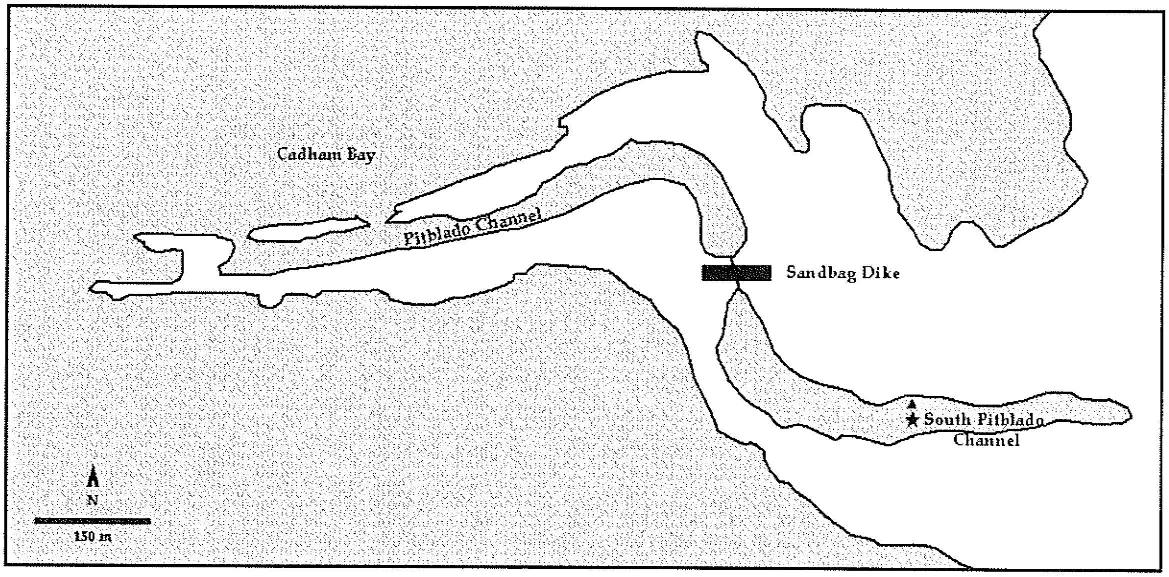


Figure 3.8: Map of South Pitblado's Channel showing the location of the permanent water sampling site (star) and the periphyton enclosure (triangle). The placement of the sandbag dike erected in the spring of 2002 is indicated. See Figure 3.1 for the location of the ponds relative to the entire marsh.

remaining connected pond, North School Bay, because it was more amenable to isolation. North School Bay (see Figure 3.9) has more than one connection with the rest of the marsh whereas South Piblado's Channel has only one entrance at its north-west end.

Sandbag dikes were built across the channels entering the ponds (Figure 3.10). The dike at South Pitblado's Channel was built on April 24, 2002. The dike at Mad Woman Bay was not put in place until May 2. Both dikes were constructed in the same manner. A dike was created by stacking 35.5 x 66 cm woven polypropylene bags filled approximately three-quarters full of sand then tied, in a pyramidal shape. A 6 mil sheet of polypropylene was placed in the center of the dike vertically between sandbags to act as a water barrier. The sheet of polypropylene stretched from the bottom to the top and past both ends of the dike. The dikes were built so that they were approximately 50 cm above the water level at the time of construction, or at least as high as the surrounding land. The dikes were at least twice as wide as they were high. Sandbags continued to be placed on the upland about a meter away from the water's edge to ensure the dike was continuous with the bank. After the dike was complete it was covered with nylon tarps held down by more sandbags to protect the sandbags from damage by sunlight.

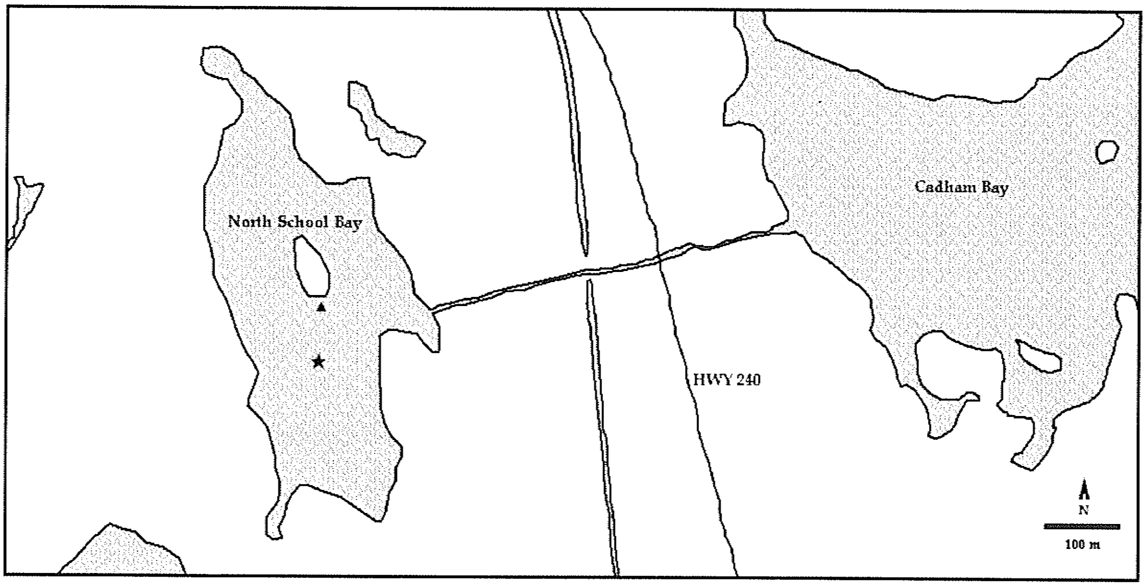


Figure 3.9: This map shows the sampling site (star) and periphyton enclosure (triangles) located in North School Bay, as well as the channel connecting the pond to Cadham Bay and the main area of the marsh through which carp could access the pond. See Figure 3.1 for the location of the ponds relative to the entire marsh.

Figure 3.10: Sandbag dike isolating South Pitblado's Channel from the main area of Delta Marsh during the summer of 2002. A similar dike was also erected to isolate Mad Woman Bay.

3.3 Sampling and Analysis Procedures

The same methods of analysis were used to evaluate the condition of the ponds in each study year. The ponds were visited for sampling every two weeks from early May to late August.

Some samples were collected from a permanent central sampling site; others were collected from random locations within the pond depending on the specific analysis. If the samples were collected from random locations, the location was always determined in the same manner: Starting at the central sampling site facing into the wind, three numbers were chosen from a random number table. The first number represented direction; odd numbers left, even numbers right. The second number represented the number of turning strokes that were taken in the chosen direction (ten strokes would turn the canoe completely around). The third number was a two digit number (0-20) and was the number of strokes that would be taken in the direction the canoe was facing after the canoe was turned. If the number of strokes indicated by the third number would place the canoe outside of the open water area, that number was re-chosen.

Some *in situ* measurements were taken during each visit, in addition to the sampling outlined below. Secchi depth, water depth, and specific conductance (YSI Model 30 conductivity meter) were determined as the average of the measurements taken at three random locations. The current wind speed was measured at the central sampling site. Wind speed at approximately 1.5 m above the water surface was taken as an average over a three minute period in the direction of the wind with a Kestrel 3000 pocket weather station. The meter was held in the hand at arms length above the head while seated in the bow seat of a canoe.

3.3.1 Water Quality

Water quality was monitored every two weeks. One water sample was collected from the permanent station every sampling period, while two additional water samples were collected from random locations throughout the pond every second sampling period (i.e. monthly). The additional samples were taken to assess spatial variation within the ponds. Integrated water column samples were obtained by lowering a clear plastic tube (6.3 cm in diameter) vertically into the water to just above the sediment. Care was taken to ensure the sediment was not disturbed, which could contaminate the water sample. A plastic ball slightly larger than the opening of the tube was placed in the top of the tube creating suction. The tube was raised and an acid washed one litre bottle, triple rinsed with site water, was placed under the tube to collect the sample. As water replaced the air in the bottle, the water in the tube was thoroughly mixed by the escaping air such that the water in the bottle represented an integration of the entire water column. The bottle was kept cool and in the dark until it could be analysed later in the laboratory. All samples were analysed as soon as possible, with the more sensitive components (i.e.

ammonia-N and total reactive phosphorous) being analysed first; usually within two hours of collection. All water quality components were analysed within the recommended time (APHA 1998).

Every two weeks, the water samples were analysed for turbidity (Hach 2100A turbidimeter), percent light transmittance by raw, unfiltered water at 450 nm, pH (Corning ion analyzer 250 pH meter), alkalinity (acid titration; (APHA 1998)), ammonia-N (hypochlorite method; (Stainton et al. 1997), total reactive phosphorous (acid molybdate-antimony method; (Stainton et al. 1997)), chlorophyll *a* concentration (APHA 1998) and organic and inorganic suspended solids (APHA 1998). Filtered samples (Whatman GF/C) were analysed for dissolved organic carbon (DOC) concentration. In 2001 the DOC samples were stored in 20 mL vials at 4°C prior to analysis at the Freshwater Institute analytical laboratory later that fall. Samples in 2002 were analysed by UV spectrometry following Badiou et al. (in prep. 2006).

In addition to the parameters mentioned above, the monthly water samples were analysed for the following: Total nitrogen (persulfate method (APHA 1998) and total phosphorous (persulfate digestion method; (APHA 1998)) were determined using analysis kits from Hach, and total reactive silica, was also measured (ammonium molybdate method; (Stainton et al. 1997). Filtered samples (Whatman GF/C) were kept refrigerated at 4°C for analysis of nitrate+nitrite-N later that fall. These samples were analysed by ion chromatography using a Dionex DX500ic system with a PRPx100 column (dimensions 150 x 4.2 mm, 10 μm) and 1mM phenol: 4mM $NaHCO_3$ buffered to pH 10.1 as the eluent. The electrochemical detector was used to determine conductivity, and peak areas were compared to external standard curves to quantify the anions.

3.3.2 Sediments

Surficial sediment cores were collected from three random locations in each pond every two weeks. Cores were collected following Bourne (2000) by lowering a clear plastic tube (6.3 cm in diameter) into the sediments. A plastic ball slightly larger than the opening of the tube was placed in the top of the tube creating suction. The tube was then removed, withdrawing an intact sediment core, which was placed on an extruder pole with a platform just smaller than the opening of the tube. The ball was removed, and the tube was slowly pushed down removing the water and leaving the sediment undisturbed sitting on top of the platform. Approximately the uppermost 3 cm of the core was sliced off and retained for analysis of sediment-associated algae (see section 3.3.3), sediment dry weight, and pore water nutrient concentration.

Sediment dry weight was determined by removing 2 cm^3 of well mixed sediment with an open-ended syringe (a 5 cc syringe with the end cut off) and placing it in a dry pre-weighed ceramic crucible. The sediment was dried for a minimum of 24 hours in an oven at 105°C, and then weighed. Percent organic matter and carbonate content were determined by incineration following Dean (1974). To determine the percent organic matter, the dry sediment was placed in an oven at 550°C for one hour, and then reweighed. The difference in weight before and after incineration was due to the loss of the organic matter fraction by ignition. The carbonate content was determined in a similar manner after incineration at 950°C for one hour.

Every second sampling period (monthly), sediment pore water concentrations of total nitrogen and total phosphorous were determined. This was done using the sediment remaining after all other analyses were performed. The remaining sediment was centrifuged at 3000 rpm for 15 minutes. The resulting supernatant was removed by pipette as a sample of the sediment pore water. Pore water total nitrogen and total phosphorous were determined using the same methods as the water samples (persulfate method and persulfate digestion method respectively, APHA 1998).

3.3.3 Algae

The abundance of phytoplankton, periphyton and sediment-associated algae (SAA) was measured in each pond every two weeks.

Phytoplankton abundance was determined as both chlorophyll *a* concentration (μg/L) and the concentration of organic suspended solids (mg/L). A known portion of the 1 L water column samples was used for the analyses.

The chlorophyll *a* concentration was determined by filtering 100 to 400 mL of the water sample through a Whatman GF/C filter. Magnesium carbonate was applied to the filter to neutralize the sample. The filter was then placed in a 7 mL vial and frozen to disrupt cell membranes and release the pigments. When the samples were to be analysed, 5 mL of 90% methanol was added to each vial. The samples were then stored in the dark. Exactly 24 hours later the methanol-pigment solution was analysed for chlorophyll *a* and corrected for pheopigments by monochromatic spectrophotometry (APHA 1998). Calculations followed Marker et al. (1980).

The concentration of organic suspended solids was determined by filtering 100 to 400 mL of the water sample through an oven dried pre-weighed glass microfibre filter (Whatman GF/C; 1.2 µm pore size). The filter was then dried in an oven at 105°C for at least 18 hours, and weighed to determine the total weight of the suspended solids in the sample. The filter was then placed in an oven at 550°C for one hour to remove all organic matter by incineration, and reweighed. The difference in weight before and after incineration was the portion of the filtered sample that was suspended organic material (APHA 1998).

Periphyton abundance was measured in two ways: 1) the algae growing on artificial substrata, henceforth called periphyton, and 2) the epiphytic algae growing on the stems of *Typha* below the water surface, henceforth called epiphyton.

Acrylic rods were used as artificial substrata for the growth of periphyton. At the start of each season, ten 0.635 cm diameter acrylic rods approximately 1 m in length were positioned vertically in each pond. The rods were placed inside a 60 x 60 cm wide, 5 cm mesh enclosure, greater than the height of the water column, in order to prevent abrasion damage to the rods by fish. The enclosures were used in all ponds, including the fishless treatments, to remove any variation between ponds due to enclosure effects. For consistency, the enclosures were placed in each pond within a few meters of the edge of the north side (see Figures 3.2, 3.3, 3.5, 3.8 and 3.9). Prior to deployment, the rods were notched with a small saw every 10 cm down the length of the rod, and then cleaned with 90% methanol. One rod was randomly chosen and removed at each sampling time. The section of rod from approximately 10-30 cm below the current water level was retained in two pre-measured sections of 10 cm in length. The sections were collected in capped tubes and then frozen to rupture the cells and release pigments. Ten millilitres of 90% methanol was added to each tube. The tubes were then placed in the dark for 24 hours and the resulting solution was centrifuged at 2000 rpm for ten minutes to remove any suspended matter, and then analysed for chlorophyll *a* (monochromatic spectrophotometry; (APHA 1998)). In 2001 there was no distinction made between the 10-20 cm and 20-30 cm sections, and all samples were analysed as pooled averages. In 2002, the sections were analysed separately to see if there was an effect of depth on periphyton chlorophyll *a* concentration.

Epiphyton was measured as the total chlorophyll *a* present in algal matter within a determined surface area of *Typha* stem growing in approximately the first 10 cm below the water surface. The samples were collected from *Typha* stands adjacent to the enclosure containing the periphyton rods in each pond. Three stems were chosen from which to collect samples. Only green stems (i.e.

from that year) that appeared to be 5-25 mm in diameter were used; otherwise stems were chosen indiscriminately. The stem was held at the surface of the water while garden shears were used to cut the stem approximately 10 cm below the water surface. The stem was removed from the water and cut into a plastic bag so that only the underwater portion of the stem was retained. In the laboratory the stems were gently scraped with an artist's brush, and rinsed with water to remove the algal material. The plastic bag was also rinsed with water and the rinse water was added to the sample. This was done to ensure any algal material that may have rubbed off onto the bag was included in the sample. The stems were then measured with a pair of callipers to determine their surface area. It was assumed that the surface area of the stem approximated that of a cylinder. Measurements were taken to the nearest millimetre in length, and the width was determined as the average of the narrowest and widest parts of the stem to the half millimetre. The samples were filtered using Whatman GF/C filters, the filters were frozen to disrupt algal membranes, and then analysed for chlorophyll *a* using the same method as for phytoplankton samples (APHA 1998).

Chlorophyll *a* concentration in sediment samples ($\mu g/cm^2$) was determined similar to the method of Bourne (2003). Three cubic centimetres of the well mixed sediment was withdrawn using an open-ended syringe, placed in a capped plastic tube and frozen to rupture the algal cells and release pigments. Fifteen millilitres of 90% methanol was later added to each tube and the tubes were shaken to distribute the methanol throughout the sample. The tubes were then placed in the dark for 24 hours where they were shaken every few hours (except at night, when the tubes were shaken before going to sleep, and then again first thing in the morning). The pigment solution was then poured off and centrifuged for ten minutes at 2000 rpm, to ensure there was no sediment in the solution. The clear solution was then analysed for chlorophyll *a* (monochromatic spectrophotometry; (APHA 1998).

3.3.4 Submersed Vegetation

Submersed vegetation was sampled once each year at the peak of the growing season (i.e. just before the onset of flowering). Samples were collected in the last week of July to the first week of August. All above-ground material within a known area (approx. 0.26 m^2) was collected from five randomly selected sites. Samples were collected by lowering a barrel into the water down to the sediment, so that the water column was enclosed. Shears were used to cut the vegetation at the sediment surface, and then all plant material within the barrel was gathered with hands, rake and a strainer. Water depth was recorded at the sample site. Samples were rinsed in clean water, and examined to remove all large algal and other non-plant material, such as invertebrates and detritus. The samples were then placed in an oven at 105°C for at least 36 hours until dry and weighed to determine the biomass. In 2002, the samples were sorted to genus while fresh, then processed as above.

3.3.5 Carp

Data on fish abundance at all sites, including young-of-the-year (YOY) carp, were collected by Parks (2006). Approximately every three weeks in 2001, and every two weeks in 2002, each pond was sampled for fish using Beamish traps (Beamish 1973). Two traps were placed around the periphery of each pond and left for approximately 24 hours. The trap was removed and the catch was counted and identified to species. All fish were released live on site.

The presence or absence of large carp in the pond was noted during each sampling visit. Carp were considered to be present if carp were seen, or if there was any visual or auditory indication that large carp were present in the pond. For example: if they could be heard spawning in the emergent vegetation surrounding the pond, or if plumes of sediment could be seen in the water indicating carp movement. This method was used to determine the relative abundance of large carp in the ponds, as Beamish traps do not adequately collect large carp, and other methods (e.g. gillnets) intended to sample large carp were ineffective (Parks 2006).

3.3.6 Statistical Analyses

Univariate statistics were performed using JMPin 3.2.1 from the SAS Institute Inc. Both analysis of variance (ANOVA) and regression analysis were used for data analysis. ANOVA's were used to determine if there was a difference in the variables based on the accessibility of large carp or treatment. A time interaction was included in the ANOVA's to determine if there were any seasonal trends present. All data were log (x+1) transformed (pH was not transformed) before they were analysed.

Multivariate analyses were performed using CANOCO for Windows 4.02. Ordination diagrams were created using SYNTAX 2000, then edited with Microsoft Office products. All variables were log (x+1) transformed (pH was not transformed) prior to use and standardized to common variance during analysis. Principal components analysis was conducted to determine the overall relationship between the ponds studied with respect to the parameters measured. This analysis was performed using the seasonal averages of 21 variables (Table 3.6) for each of the ten ponds in each of the two years. Redundancy analysis was also used to constrain the same 21 variables by factors representing the abundance of large carp.

Table 3.6: A list of the 21 variables used in the principle components analysis of the ponds studied.

Variable	Abbreviation
pH	pH
Alkalinity	ALK
Specific Conductance	COND
Dissolved organic carbon	DOC
Turbidity	NTU
Inorganic suspended solids	ISS
Organic suspended solids	OSS
Chlorophyll a in the water column	Chla
Total P	TP
Total reactive P	TRP
Total Kjeldahl N	TKN
Inorganic N	TIN
Reactive Silica	Si
Percent sediment organic matter	%OM
Percent sediment carbonate	%Carb
Sediment pore water TKN	Sed. TKN
Sediment pore water TP	Sed. TP
Macrophyte density	Macrophytes
Epiphyton abundance	Epiphyton
Periphyton abundance	Periphyton
Abundance of sediment associated algae	SAA

Chapter 4: Results

4.1 Carp

In 2001 the marsh experienced high water levels (Table 4.1.1, also see Parks 2006 Figure 2-1). As a result, the upland emergent vegetation separating the Isolated ponds from the main marsh became inundated with water for at least part of the open water season. Consequently, every pond had the ability to contain fish in 2001. In all of the Connected ponds large carp could frequently be seen throughout the summer (Table 4.1.2). In most of the Isolated ponds large carp were seen rarely, and only in May and/or June. Wye's Pond was the exception. There large carp were observed throughout the summer. Data collected by Parks (*pers comm.*) show that there were more young-of-the-year (YOY) carp present per hectare in the Isolated ponds than in most of the Connected ponds (Table 4.1.2). Wye's Pond was again the exception and few YOY carp were caught there in 2001. Emile's Pothole, an Isolated pond, contained by far the most YOY carp in 2001 of all the ponds studied. Overall in 2001, ponds with frequent observations of large carp had fewer YOY carp, than ponds with rare observations of large carp, which had higher numbers of YOY carp.

In the second year of the study, water levels in the ponds studied dropped significantly ($p < 0.0001$) from those in 2001 (Table 4.1.1). As a result, all Isolated ponds were truly Isolated from the rest of the marsh in 2002. No large carp were

Table 4.1.1: Average seasonal (May to September) depth (± SD) recorded for the ponds studied (n=7, where n is the number of measurements for each mean). Analysis of variance comparing the overall average depth in 2001 to 2002 yields a significant p-value (<0.0001) at $\alpha = 0.05$.

Pond	**2001**		**2002**	
	Depth (cm)	**SD**	**Depth (cm)**	**SD**
Mid-Blind Channel	105	13	56	5
Thompson's Bay	64	15	28	12
Section 5 Bay	58	15	23	7
Mad Woman Bay	65	15	45	7
North School Bay	99	13	47	7
South Pitblado's Channel	95	10	59	9
North MacKenzie Bay	79	13	51	8
South MacKenzie Bay	70	14	36	6
Emile's Pothole	80	9	55	4
Wye's Pond	86	5	47	10
Overall (n=10)	80	15	45	11

Table 4.1.2 : Presence of common carp in the ponds studied. The number of carp caught are based on data collected by Parks (*pers. comm.*) and include the catch of young-of-the-year. The frequency of observance of large carp is based on how often large carp were observed in each pond.

Pond	**Treatment**	**Number of carp caught per hectare**		**Frequency of observance**	
		2001	**2002**	**2001**	**2002**
Thompson's Bay	Connected	<1	432	frequently	frequently
North School Bay	Connected	<1	145	frequently	frequently
Mid-Blind Cha.	Screened	1	5	frequently	never
Section 5 Bay	Screened	5	349	frequently	never
Mad Woman Bay	Diked	2	0	frequently	never
S. Pitblado's Cha.	Diked	16	0	frequently	never
North Mackenzie	Isolated	23	0	rarely	never
Emile's Pothole	Isolated	287	0	rarely	never
South Mackenzie	Blasted	12	339	rarely	frequently
Wye's Pond	Blasted	2	271	frequently	rarely

ever observed in any of the Isolated, Diked, or Screened ponds and no YOY carp were caught in any of the Isolated or Diked ponds in 2002 (Table 4.1.2). YOY carp were caught in the Screened ponds; Mid-Blind Channel had very few YOY carp while Section 5 Bay contained levels of YOY carp comparable to the Connected and Blasted sites. All Connected and Blasted ponds contained YOY carp at levels similar to, or higher than those found in Emile's Pothole in 2001. However, large carp were rarely observed in Wye's Pond in 2002 (Table 4.1.2).

4.2 Turbidity

Turbidity is a measure of water clarity, and is related to the amount of suspended solids in the water column. In this study both inorganic suspended solids (ISS) and organic suspended solids (OSS) contributed to turbidity (Figure 4.2.1). The ISS results will be discussed here with turbidity, while OSS, which can be used as a measure of phytoplankton abundance, with be discussed later with algae (Section 4.4.1). Based on the seasonal averages from 2002, turbidity (NTU) was most closely associated with ISS (r=0.992), OSS (r=0.795), and density of YOY carp (0.771), while ISS was most closely associated with turbidity (r=0.992), density of YOY carp (r=-0.759), and water depth (r=-0.755) (Appendix 1).

Suspension of sediments by wind can alter the turbidity of a shallow system, therefore both water depth and intensity of the wind may have influenced the turbidity of the ponds in this study. Regression analysis (Table 4.2.1) showed no correlation between turbidity and wind speed or water depth in

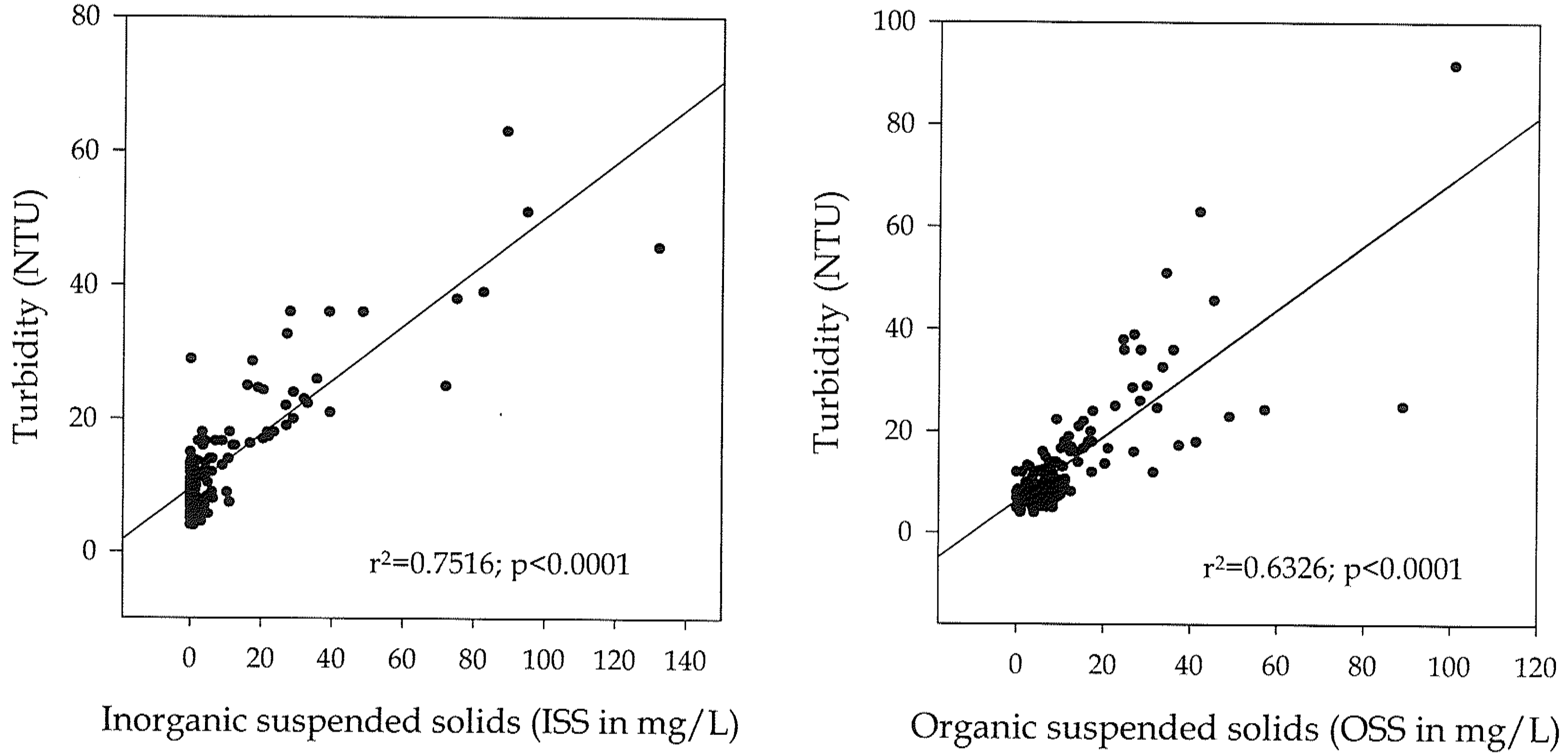


Figure 4.2.1: Simple linear regressions of suspended solids and turbidity. Individual regressions were performed for both inorganic suspended solids (ISS) and organic suspended solids (OSS). The r^2 values were calculated for each set of variables and are shown in each graph. Data includes all samples collected throughout both years of the study (n=140).

Table 4.2.1: The degree of influence of wind or water depth on turbidity. The coefficients of determination (r^2) reported are the result of simple linear regression analysis of turbidity (NTU) against the average three minute wind speed (km/h) and the average depth (cm). *Indicates a significant p-value (α=0.05) showing that a linear relationship exists between the variables. When both years are considered together similar linear regressions yield: wind r^2 = 0.020 , p-value = 0.126; and depth r^2 = 0.121, p-value < 0.0001.

			Thompson's	N. School	Mid-Blind	Section 5	Mad Woman	S. Pitblado's	Mac. North	Emile's	Mac.South	Wye's	All Ponds
wind	**2001**	**r^2**	0.001	0.525	0.224	0.192	0.105	0.926	0.466	0.002	0.169	0.130	0.001
		p-value	0.956	0.166	0.420	0.461	0.595	0.009*	0.204	0.945	0.492	0.551	0.859
	2002	**r^2**	0.474	0.071	0.355	0.099	0.143	0.014	0.152	0.001	0.003	0.023	0.028
		p-value	0.087	0.563	0.158	0.491	0.403	0.803	0.387	0.943	0.902	0.747	0.166
depth	**2001**	**r^2**	0.183	0.185	0.493	0.127	0.140	0.283	0.047	0.034	0.290	0.003	0.037
		p-value	0.339	0.335	0.079	0.434	0.408	0.219	0.641	0.692	0.212	0.913	0.111
	2002	**r^2**	0.013	0.250	0.134	0.825	0.028	0.495	0.058	0.186	0.245	0.012	0.327
		p-value	0.807	0.253	0.420	0.004*	0.720	0.078	0.605	0.287	0.259	0.062	<.0001*

2001, and wind speed in 2002. In the second year of the study there was a significant weak correlation (r=0.121, p<.0001) between depth and turbidity. On an individual pond basis, only two significant correlations were found (Table 4.2.1). In 2001, a positive correlation (r=0.926, p=0.009) was found between turbidity and wind speed in South Pitblado's Channel. There was, however, little change in NTU (Figure 4.2.2) or wind speed in S. Pitblado in 2001. Therefore the effect of wind was likely of little influence in the overall analysis. In 2002, there was a strong negative association (r=0.825, p=0.004) found between turbidity and depth in Section 5 Bay.

In 2001, the seasonal average turbidity was similar in all the ponds studied (mean NTU=10.9; Figure 4.2.2). However, this similarity may have been weak; an ANOVA resulted in a small p of 0.1128, indicating that there may have been a difference in turbidity between Connected (mean NTU=13) and Isolated (mean NTU=7.7) ponds. There was a difference (p=0.0504) in the concentration of inorganic suspended solids (ISS) between ponds. ISS was an order of magnitude greater in the Connected ponds (9.6 mg/L) than the Isolated ponds (0.8 mg/L; Figure 4.2.3).

The seasonal trend in turbidity depended on connectivity (p=0.0226). The greatest difference in turbidity between the Connected and Isolated ponds occurred in the spring when the Connected sites had their highest turbidity. In May and June both turbidity (p=0.0006) and ISS (p=0.0012) were significantly higher in the Connected ponds (mean NTU=17; mean ISS=17mg/L) than the

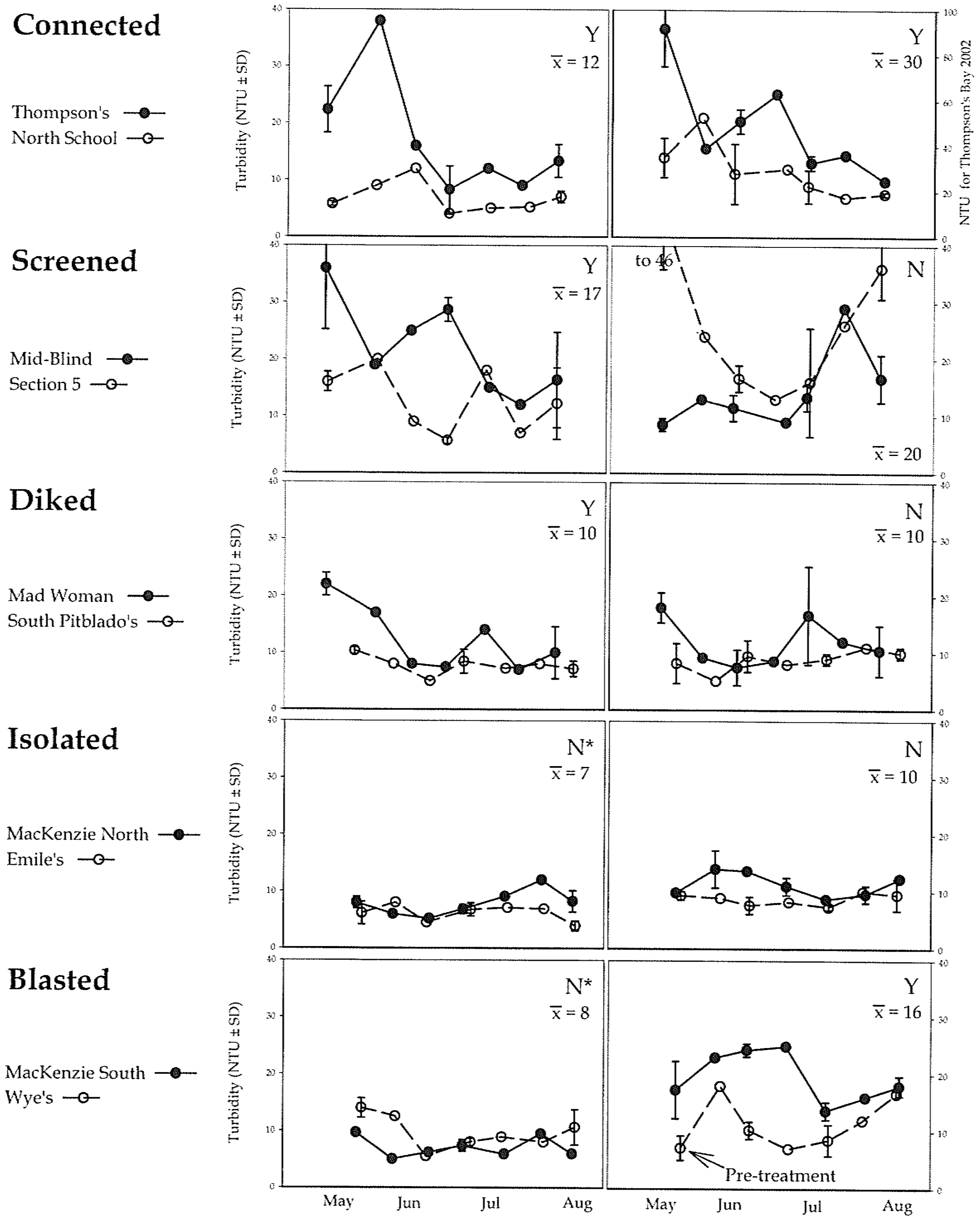


Figure 4.2.2: Seasonal variation in mean turbidity (NTU ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the growing season. X bar shows overall mean for each graph.

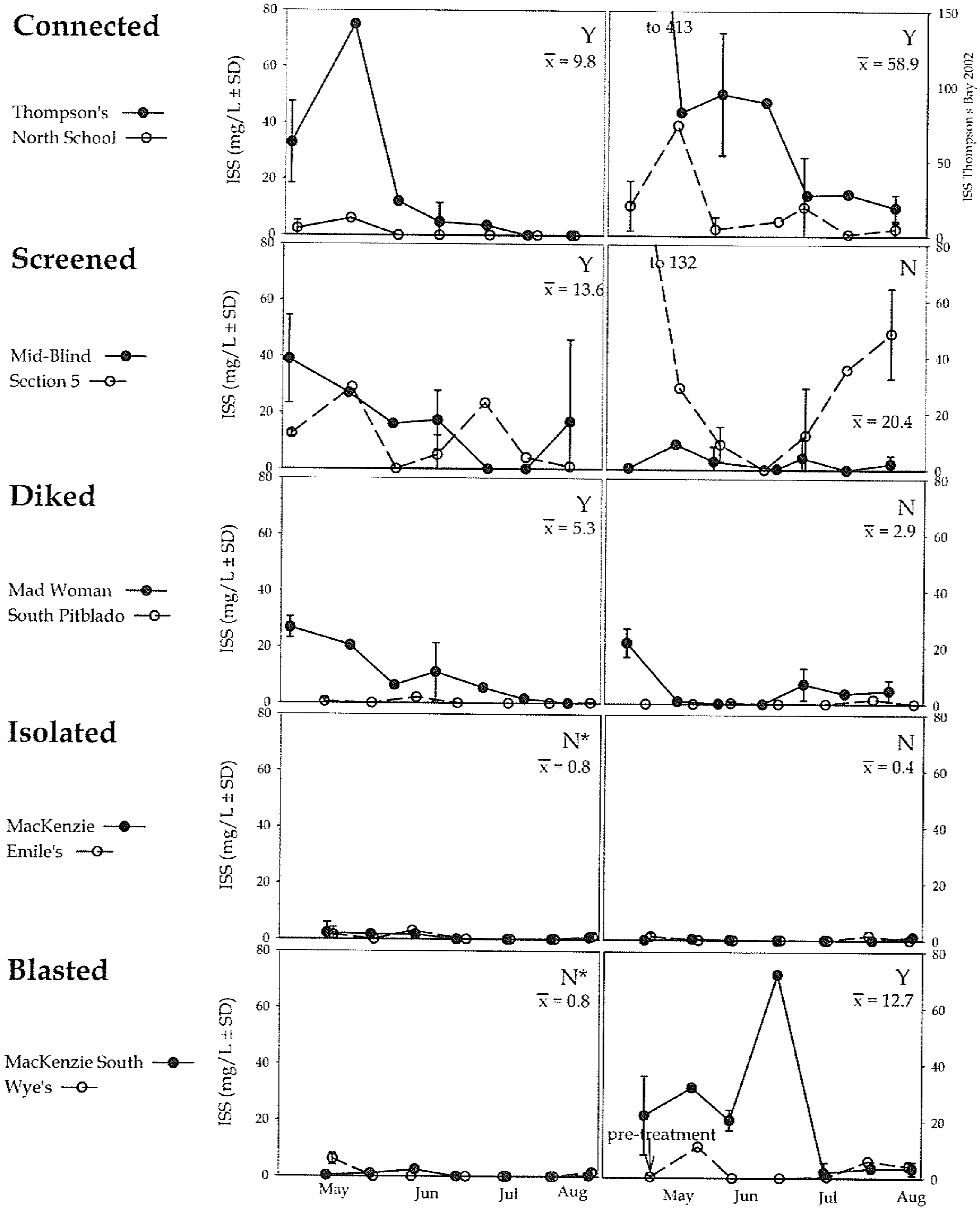


Figure 4.2.3: Shows seasonal variation in mean inorganic suspended solids (mg/L ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. * Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the growing season.

Isolated ponds (mean NTU=7.6; mean ISS=1.7 mg/L). By July, the turbidity in the Connected sites began to decrease, and by August the average turbidity in the Connected sites was similar to that of the Isolated sites. This seasonal trend was, however, only weakly significant (α=0.10) between the Connected and Isolated sites with respect to ISS (p=0.0592).

In 2001 North School Bay (NTU=6.7, ISS=1.2 mg/L) and South Pitblado's Channel (NTU=7.7, ISS=0.4 mg/L) had the lowest average turbidity of the Connected ponds while Wye's Pond (NTU=9.7, ISS=1.1 mg/L) had the highest turbidity of the Isolated ponds.

In 2002, turbidity and ISS were on average similar in all the ponds studied (NTU=17, ISS=19.0 mg/L; Figure 4.2.2 and 4.2.3). No significant differences were found based on treatment (NTU p=0.5111, ISS p=0.2466) or the presence of large carp (NTU p=0.2553, ISS p=0.1019), however a weak relationship may have occurred in the later case with respect to ISS. As was previously mentioned, in 2002 there was a strong correlation between turbidity and depth in Section 5 Bay. If this pond was removed from the analysis, the difference in turbidity between ponds that were accessible to large carp (NTU=23, ISS=35.8 mg/L) and those that were inaccessible to large carp (NTU=11 , ISS=1.8 mg/L) became greater. With respect to NTU the difference remained insignificant (p=0.1305); however, a significant difference in ISS (p=0.0281) became apparent. This situation was similar to conditions in 2001.

As was the case in 2001, the seasonal trend in turbidity in 2002 was related to the accessibility of large carp (p=0.0917; Figure 4.2.2 and 4.2.3). Ponds that were accessible to large carp in 2002 had a higher turbidity (mean NTU=27; mean ISS=61 mg/L) in the spring (May and June) than ponds that were inaccessible to large carp (mean NTU=13; ISS=11.5). This elevated spring turbidity in the accessible ponds was highly significant (NTU p<0.0001; ISS p<0.0001).

In 2002 Thompson's Bay (48 NTU, ISS=108 mg/L) and South Mackenzie Bay (20 NTU, ISS=22 mg/L) had the highest average turbidities of all the ponds; both were accessible to large carp. The turbidity was similar in the remaining ponds, and very little inorganic suspended solids were found in the Isolated and Diked sites (Figure 4.2.2 and Figure 4.2.3).

4.3 Nutrients

4.3.1 Phosphorous

Phosphorous was measured in two ways; as total reactive P (TRP; Figure 4.3.1a) and total P (TP; Figure 4.3.1b). Based on the seasonal averages from 2002, TP was most closely associated with depth (r=-0.892), OSS (r=0.726), and TRP (r=0.724), while TRP was most closely associated with TP (r=0.724), conductivity (r=-0.700), and sediment pore water TP (r=0.696) (Appendix 1).

In 2001, there were no significant differences between Connected and Isolated sites with respect to TRP (101 μg/L; p=0.1044), or TP (521 μg/L;

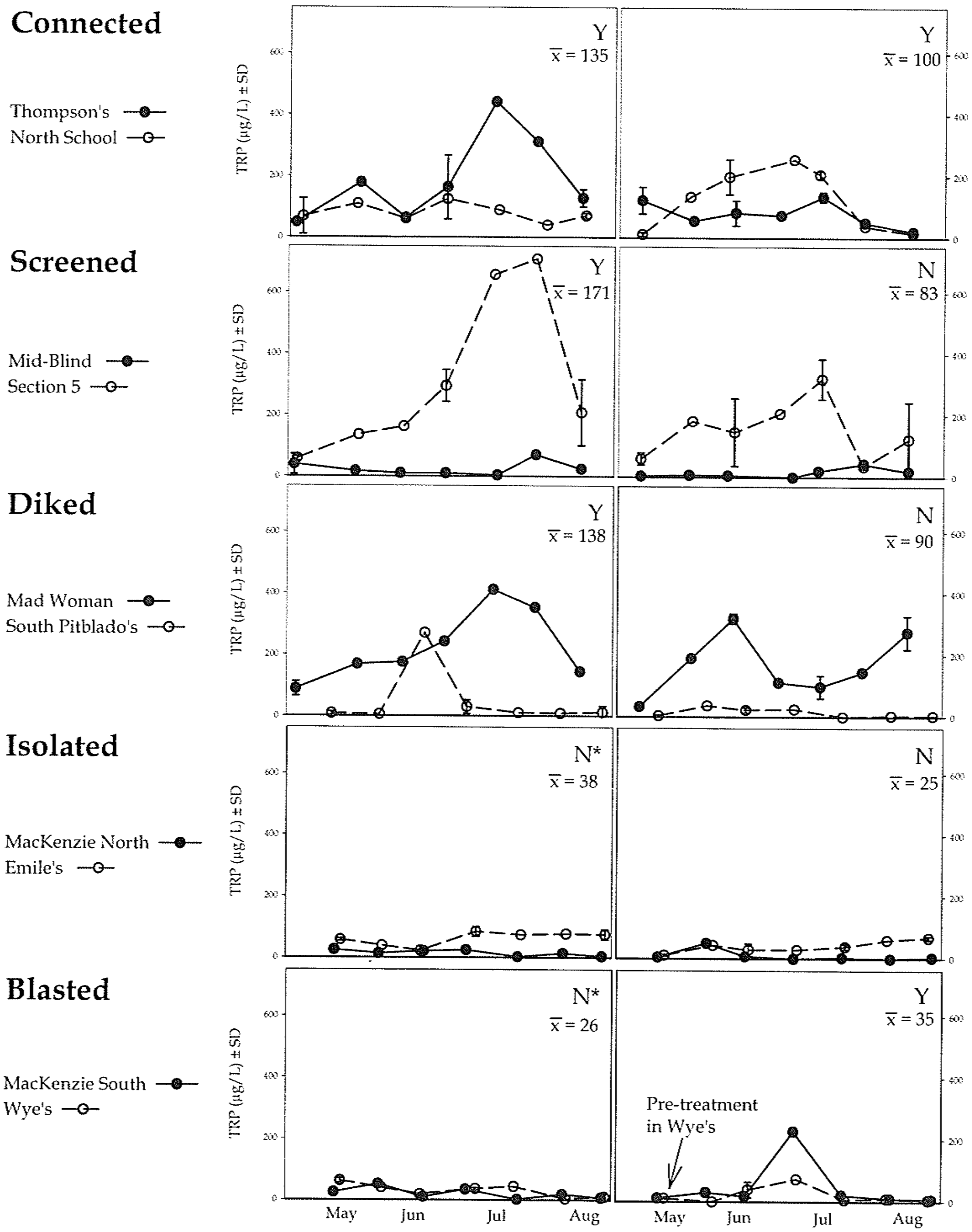


Figure 4.3.1a : Seasonal variation in mean concentration of total reactive phosphorous (TRP) (µg/L ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

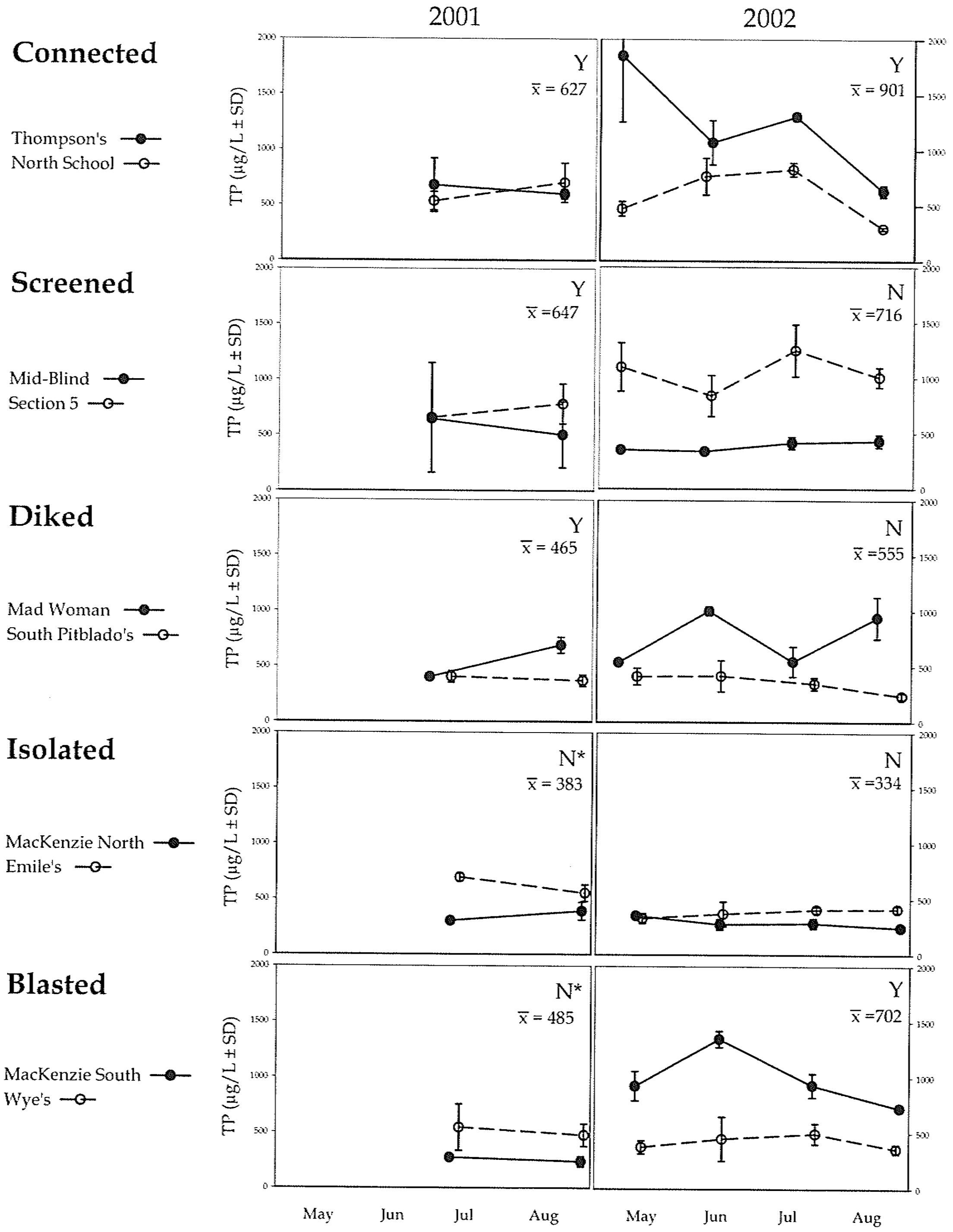


Figure 4.3.1b: Seasonal variation in mean concentration of total phosphorous (TP) (µg/L ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

p=0.1277). However, it should be noted that the highest levels of TRP were found in three of the Connected ponds: Thompson's Bay, Section 5 Bay, and Mad Woman Bay. The remaining Connected ponds had concentrations of TRP similar to the Isolated ponds (Figure 4.3.1a). There was no overall seasonal trend found for TRP (p=0.6688), or TP (p=0.8823). There was also no seasonal difference in P concentrations found between the Connected and Isolated ponds (TRP p=0.1303; TP p=0.4652).

In 2002, the P concentrations were similar in all the ponds studied (mean TRP=66 μg/L; mean TP=642 μg/L) (Figure 4.3.1a and Figure 4.3.1b). There were no significant differences found based on treatment (TRP p=0.7685; TP p=0.5914) or the accessibility of large carp (TRP p=0.7057; TP p=0.2351). Similar to the first year of the study; Thompson's Bay, Section 5 Bay and Mad Woman Bay were among the ponds with the highest concentrations of TRP, however in the second year they were joined by North School Bay (Figure 4.3.1a). In 2002, there were a few ponds that had TP concentrations that averaged above the rest: two ponds that were accessible to large carp; Thompson's Bay and South MacKenzie Bay, along with Section 5 Bay (Figure 4.3.1b). A seasonal change in average TRP concentration occurred in 2002 such that the highest average TRP was in July; however, this trend was only weakly significant (p=0.0925). When we look at the seasonal change based on the accessibility of large carp the trend improves (p=0.0199); on average, inaccessible ponds showed very little seasonal change, while the ponds that were accessible to large carp increased in TRP throughout

May and June, reached peak levels in early July, then decreased for the rest of the season. Overall, the mean concentration of TP decreased in August (p=0.0335). This seasonal trend in TP was different when the ponds are separated based on their accessibility to large carp (p=0.0598); on average, the ponds inaccessible to large carp remained relatively constant throughout the season, while the ponds that were accessible to large carp averaged higher levels of phosphorous in May, June and July than in August, when on average, their concentrations were similar to the inaccessible ponds. Mad Woman Bay and Section 5 Bay were two inaccessible ponds that were exceptions to the overall seasonal trend in P; in both of these ponds P concentrations varied seasonally.

In both years there was a large difference between the concentrations of TRP and TP, with TP being much greater. Since the forms of P accounted for by TP are mostly organic, this indicates that much of the P present in the ponds is in organic form.

4.3.2 Nitrogen

Nitrogen was measured in three ways: as ammonia and nitrate, which were combined into inorganic nitrogen (TIN) (Figure 4.3.2a) and total nitrogen (TN) (Figure 4.3.2b). As was the case with phosphorous, the majority of the nitrogen was present in organic form. Based on the seasonal averages from 2002, TN was most closely associated with organic N (r=0.999), soluble reactive silica (r=0.702), and sediment pore water N (r=0.683), while TIN was most closely

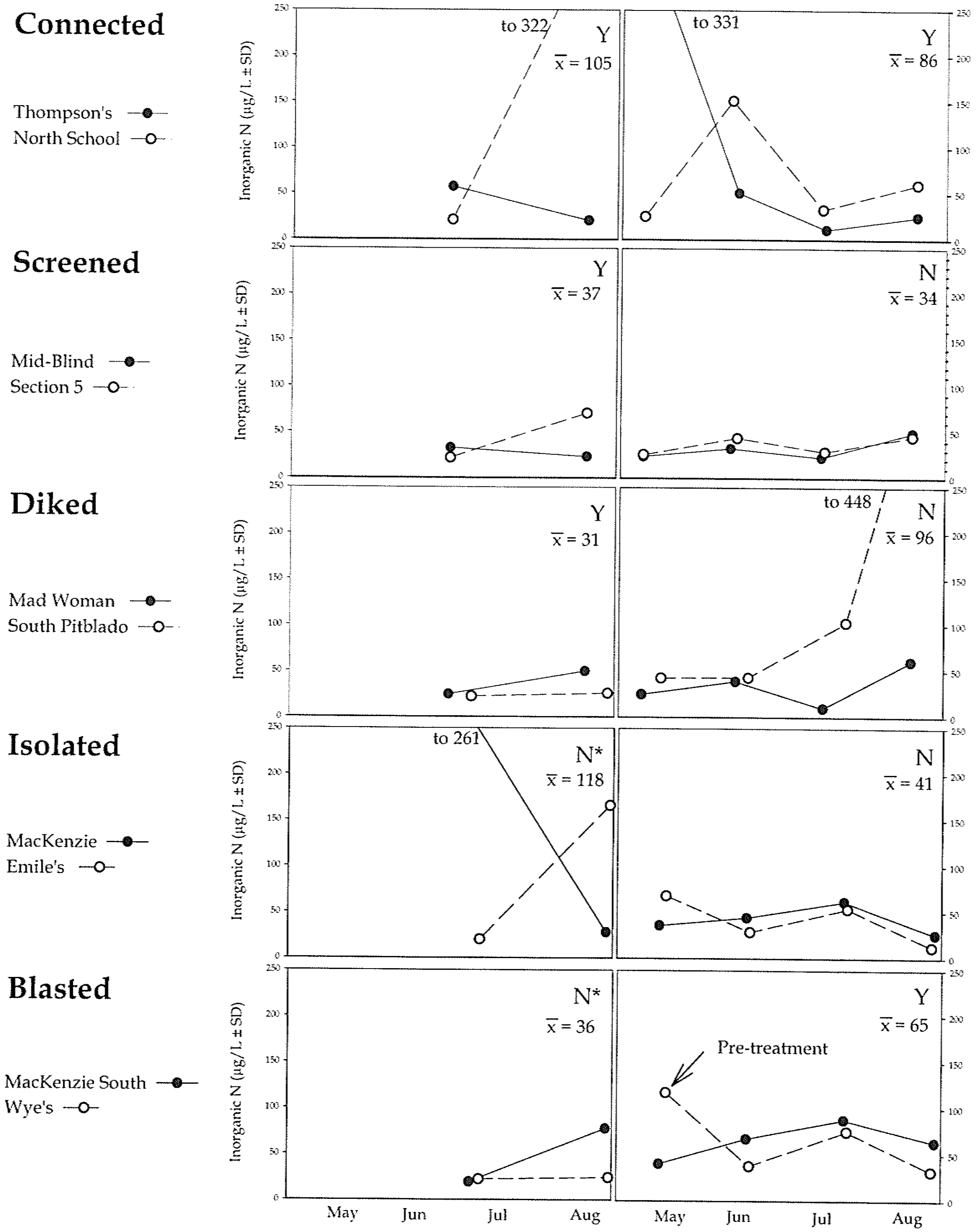


Figure 4.3.2a: Seasonal variation in inorganic nitrogen (µg/L ± SD) calculated as the sum of ammonia-N and nitrate-N. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence

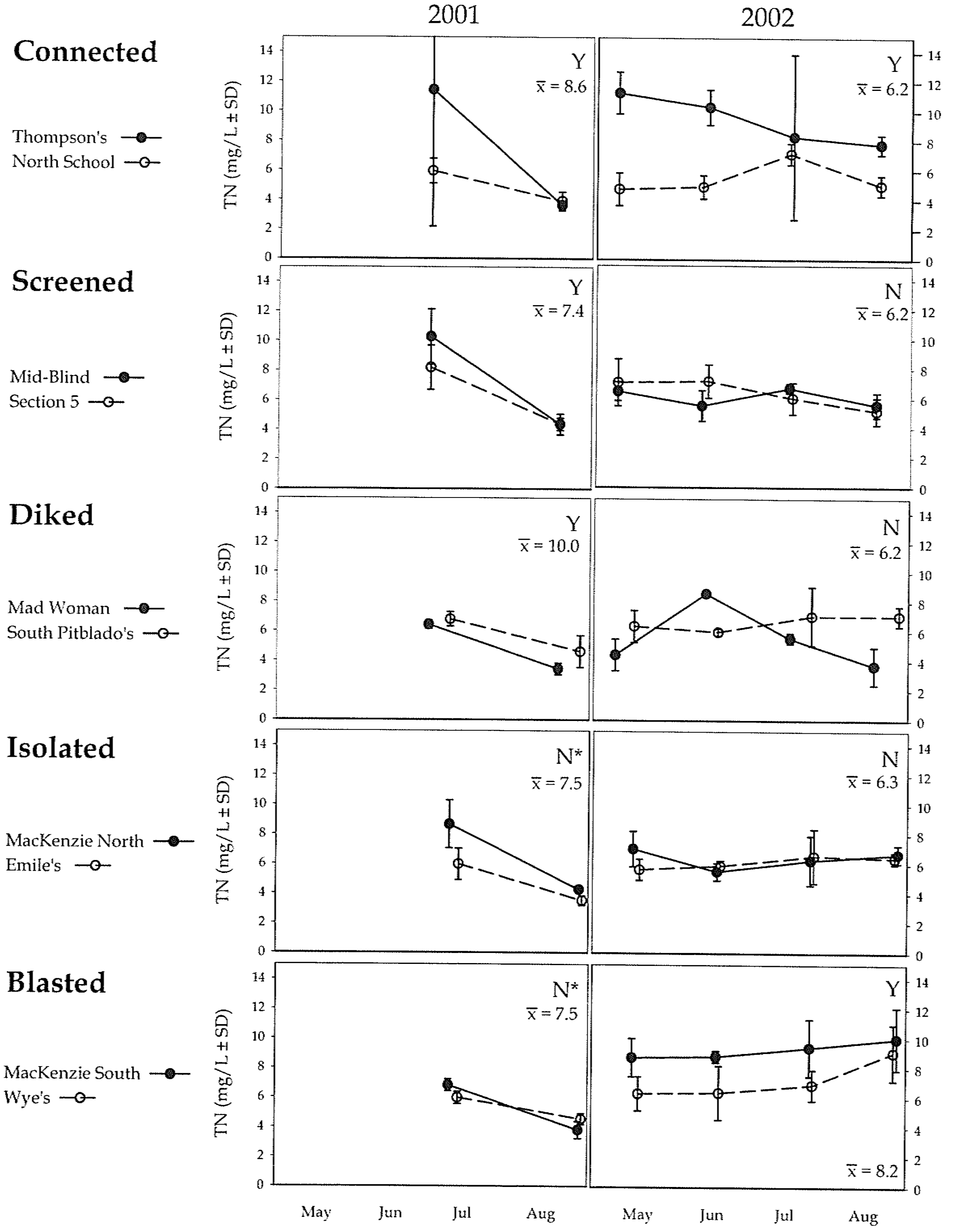


Figure 4.3.2b: Seasonal variation in total nitrogen (mg/L ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

(N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season. associated with ammonia-N (r=0.745), percent organic matter in the sediments (r=0.697), and conductivity (r=0.692) (Appendix 1).

On average, in 2001 all of the ponds in the study, regardless of connectivity, contained similar amounts of both forms of nitrogen (TIN=65 μg/L, p=0.5072; TN=5.8 mg/L, p=0.3874). There was no seasonal change in average TIN (p=0.3982), and this was the same for both the Connected and Isolated sites (p=0.8291). In 2001, TN was only measured twice, overall the July values were higher than the August values (p<0.0001). This trend occurred in both Connected and Isolated ponds to the same degree (p=0.3710).

In 2002, there was again no difference in the concentrations of nitrogen found in the ponds based on either treatment (TIN p=0.7034, TN p=0.2539) or the accessibility of carp (TIN p=0.2994, TN p=0.1756). The average concentration of TIN was 65 μg/L and of TN was 6.6 mg/L. Again, there was no seasonal change in the average TIN (p=0.8771), and this did not change based on connectivity (p=0.5655). Inorganic nitrogen concentrations were relatively stable throughout the season for most ponds, with two exceptions; Thompson's Bay had a relatively high level of TIN in May, and South Pitblado's Channel had a relatively high level in August (Figure 4.3.2a). There was also no average seasonal trend in TN (p=0.8557), and this was no different based on carp accessibility (p=0.7686). The highest average TN concentrations occurred in ponds that were accessible to large carp: two Blasted ponds; South MacKenzie Bay (9.3 mg/L), Wye's (7.2 mg/L), and a Connected pond; Thompson's Bay (6.9 mg/L).

The ratio of TN to TP averaged 11.2 in 2001 and 10.3 in 2002 (Table 4.3). These ratios are lower than the ideal Redfield ratio of 16:1. The TN:TP for most of the ponds was lower than 16, indicating that nitrogen is limiting. There were some exceptions North MacKenzie Bay (both years), South MacKenzie Bay (2001), S. Pitblado's Channel (2002), Emile's Pothole (2002), and Wye's Pond (2002). The higher ratios in these ponds is attributable to their lower P concentrations. It is interesting to note that all of the above ponds, with the exception of Wye's Pond (2002), were inaccessible to large carp; however, there was no significant difference in TN:TP found based on the accessibility to large carp (p=0.1290).

4.3.3 Silica

Silica concentrations in 2001 were similar in all the ponds studied (mean=7.3 mg/L, p=0.1720). However, the Connected sites tended to have slightly higher silica concentrations than the Isolated sites (Figure 4.3.3). The highest average silica concentrations were found in two Connected ponds; Mid-Blind Channel (10.6 mg/L) and Mad Woman Bay (10.3 mg/L), and the lowest average concentrations were found in two Isolated ponds; MacKenzie Bay's North (3.4 mg/L) and South (3.4 mg/L). There was no difference in the average concentration of silica found in the ponds in July compared to August

Table 4.3: Average seasonal ratio of total nitrogen to total phosphorous for the ponds studied. 2001 n=2 (July and August), 2002 n=4 (May, June, July, and August).

Pond	**Treatment**	**TN:TP**	
		2001	**2002**
Thompson's Bay	Connected	11.8	5.7
North School Bay	Connected	8.0	9.4
Mid-Blind Cha.	Screened	12.7	15.8
Section 5 Bay	Screened	8.7	6.1
Mad Woman Bay	Diked	9.1	7.4
S. Pitblado's Cha.	Diked	14.8	18.8
North Mackenzie	Isolated	18.7	22.1
Emile's Pothole	Isolated	7.6	16.2
South Mackenzie	Blasted	20.8	9.4
Wye's Pond	Blasted	10.3	17.0
Overall (n=10)		11.2	10.3

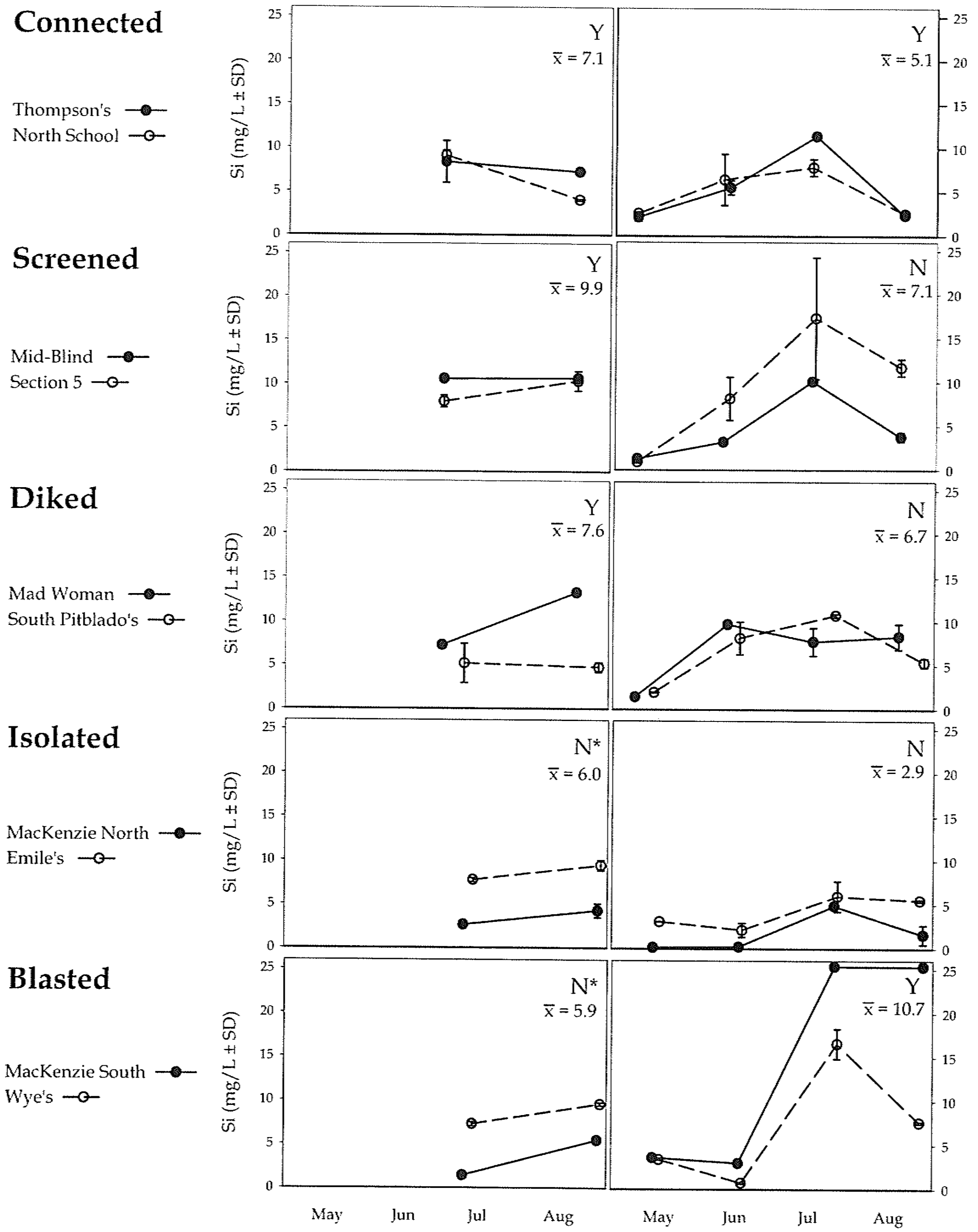


Figure 4.3.3: Seasonal variation in silica (mg/L ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

(p=0.1436). There was also no significant difference in this trend between Connected and Isolated sites (p=0.1088), however a weak relationship existed that Isolated sites had lower levels of silica in July than August unlike, Connected sites that were similar for both months.

The average seasonal silica concentration in 2002 was most closely associated with TN (r=0.702), chlorophyll *a* in the water column (r=0.652) and TP (r=0.651) (Appendix 1). In 2002, silica concentrations were again on average similar in all the study sites (mean=6.5 mg/L) regardless of treatment (p=0.2776) or accessibility of large carp (p=0.4106). The highest average concentration of silica was found in South MacKenzie Bay (14.3 mg/L), a Blasted pond, and the lowest average concentration was found in MacKenzie Bay North (1.7 mg/L), an Isolated pond. In the second year of the study, there was a strong seasonal trend in the average silica concentration (p<0.0001); the average silica concentration increased from May to July, where it peaked, and then began to decrease in August. This seasonal trend was similar for ponds that were accessible to large carp as those that were inaccessible to large carp (p=0.4632).

4.4 *Algae*

4.4.1 Phytoplankton

Based on the seasonal averages from 2002, phytoplankton abundance measured as chlorophyll *a* was most closely associated with OSS (r=0.966), density of YOY carp (r=0.680), and equally correlated with turbidity (NTU) and silica (r=0.652) (Appendix 1). Similarly organic suspended solids (OSS) was most closely associated with chl *a* in the water column (r=0.966), density of YOY carp (r=0.810), and turbidity (r=0.795).

In 2001, phytoplankton abundance was similar in all the ponds studied with an average chlorophyll *a* concentration of 22 μg/L (Figure 4.4.1a) and an average concentration of organic suspended solids of 8 mg/L (Figure 4.4.1b). There was no difference in phytoplankton abundance between Connected and Isolated ponds (chl*a* p=0.3825, OSS p=0.1349). The highest seasonal averages of phytoplankton were found in Mid-Blind Channel (chl*a* = 58 μg/L; OSS = 20 mg/L), a Connected pond, and Wye's Pond (chl*a* = 36 μg/L; OSS = 6 mg/L), an Isolated pond. Emile's Pothole, an Isolated pond, had the lowest seasonal average of phytoplankton based on both chlorophyll *a* (11 μg/L) and organic suspended solids (4 mg/L). Overall, the concentration of chlorophyll *a* in the water column did not change significantly throughout the season (p=0.1779). However, the seasonal trend was different for the Connected ponds than Isolated ponds (p<0.0001). On average, the Connected ponds had their highest levels of chlorophyll *a* in May, followed by another peak in August, while the Isolated ponds had low levels of chlorophyll *a* in the spring, increased to peak levels throughout July, followed by a bloom at the end of August (Figure 4.4.1a and Figure 4.4.1b).

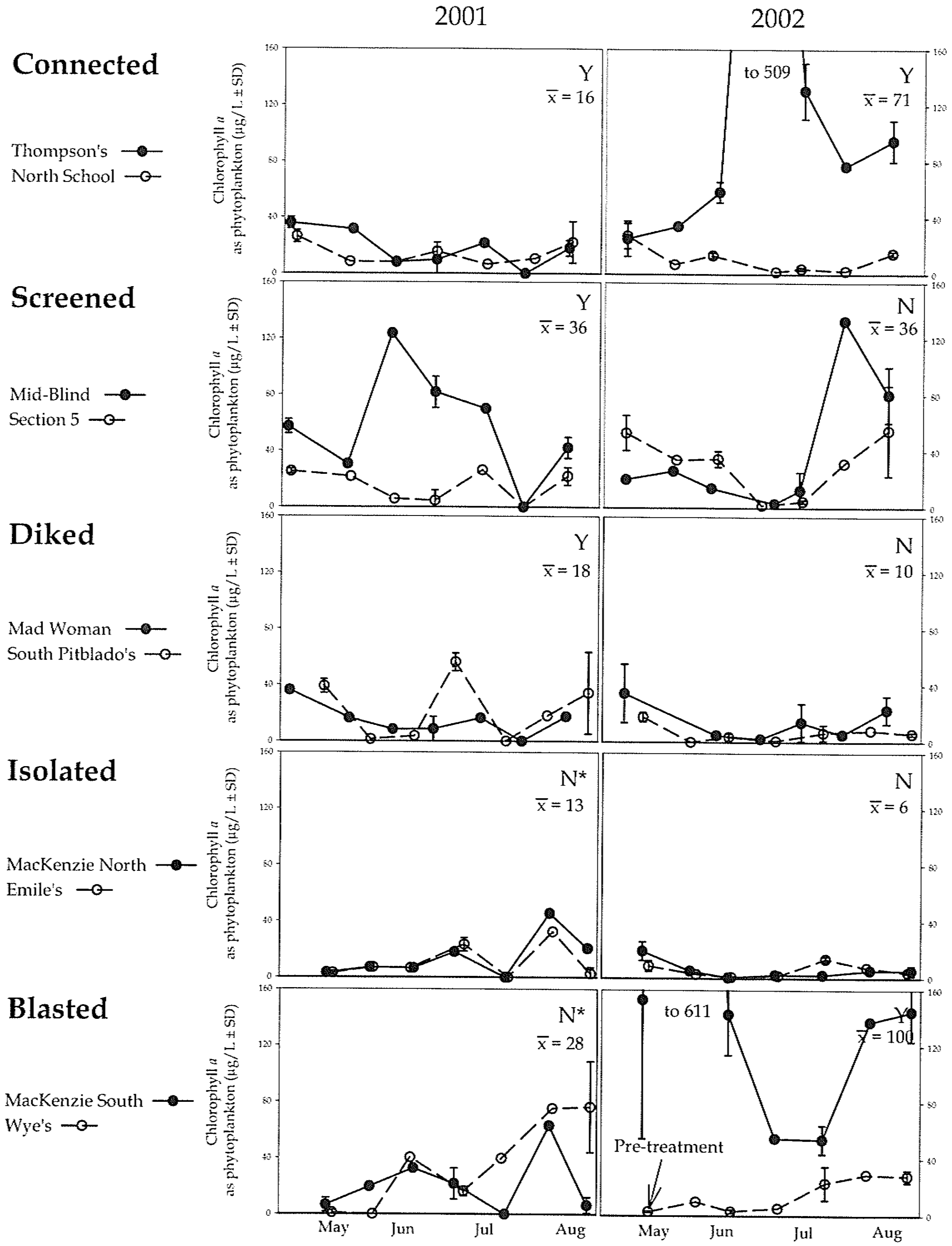


Figure 4.4.1a: Seasonal variation in mean concentration of chlorophyll *a* (µg/L ± SD) present as phytoplankton for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

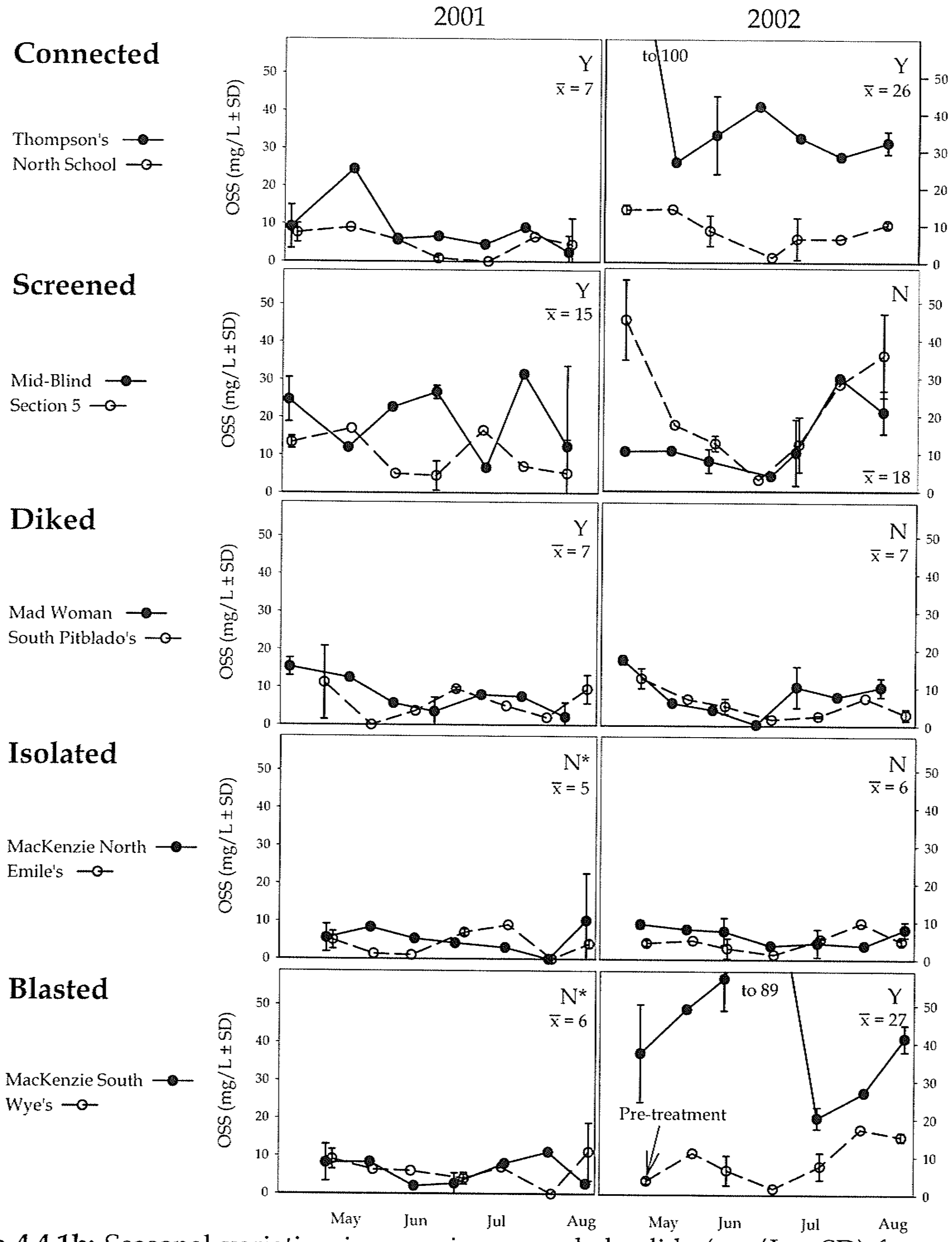


Figure 4.4.1b: Seasonal variation in organic suspended solids (mg/L ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as In 2002, a difference in phytoplankton abundance was observed between the ponds.

On average, the highest densities of phytoplankton were found in Thompson's Bay (chla = 132 µg/L; OSS = 43 mg/L) and South MacKenzie Bay well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

(chl*a* = 185 µg/L; OSS = 46 mg/L), Connected and Blasted ponds respectively, while the lowest densities were found in the Diked (chl*a* = 10 µg/L; OSS = 7 mg/L) and Isolated (chl*a* = 6 µg/L; OSS = 6 mg/L) ponds (Figure 4.4.1a and Figure 4.4.1b). The Screened ponds (chl*a* = 36 µg/L; OSS = 18 mg/L); Mid-Blind Channel and Section 5 Bay had mean phytoplankton densities slightly higher than the Isolated and Diked ponds. As was the case with turbidity, the phytoplankton abundance in Section 5 Bay was again correlated to changes in depth (Figure 4.4.1c). North-School Bay (Connected) and Wye's Pond (Blasted) differed from the other ponds which were accessible to large carp. They had lower phytoplankton densities which were similar to the Diked ponds; chl*a* = 10 µg/L, OSS = 9 mg/L and chl*a* = 15 µg/L, OSS = 9 mg/L respectively. This high degree of variation amongst the Connected and Blasted ponds lead to differing results when ANOVAs were performed (Section 5 Bay was removed from the analyses due to the high correlation between depth and phytoplankton abundance). Based on the accessibility of the ponds to large carp there was a weak significant difference between ponds that were accessible and ponds that were inaccessible to large carp (chl*a* p=0.0961, OSS p=0.0654; α=0.10). However,

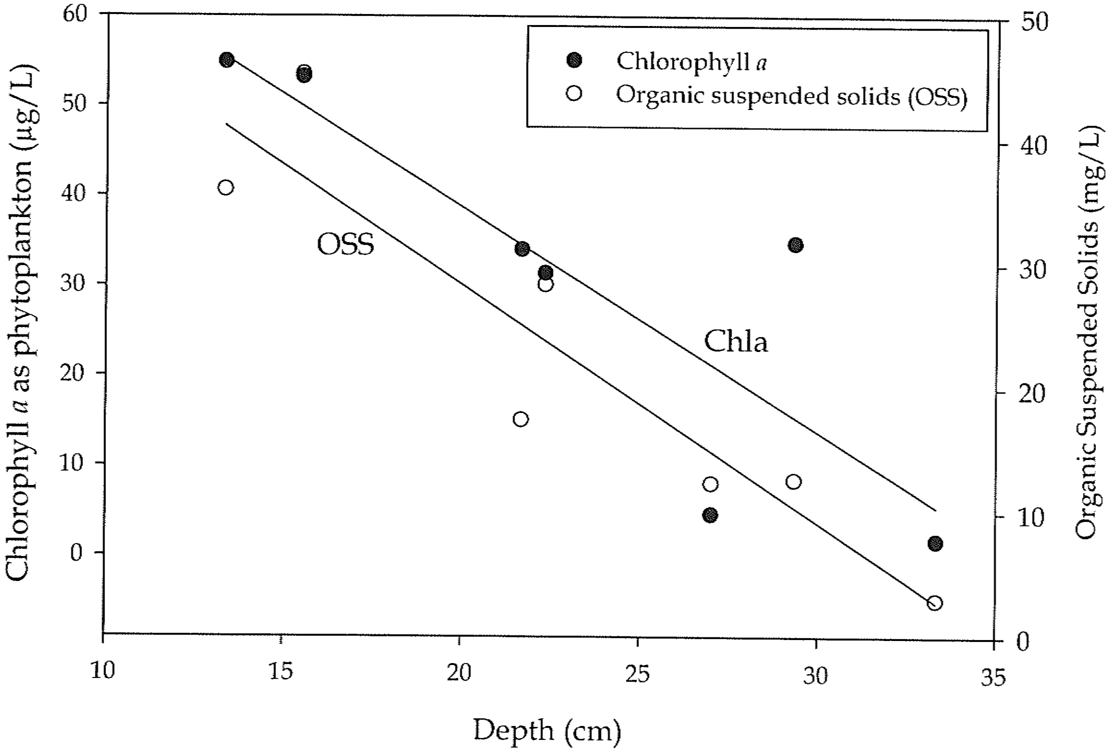


Figure 4.4.1c: Linear regression of phytoplankton density versus depth for Section 5 Bay in 2002. A significant linear relationship was found for both chlorophyll *a* (r^2=0.75; p=0.0120) and organic suspended solids (r^2=0.86; p=0.0028).

when the ponds were grouped by treatment, no significant difference was found between treatments (chl*a* p=0.4895, OSS p=0.7704).

4.4.2 Epiphyton and Periphyton

Based on the seasonal averages from 2002, epiphyton chlorophyll *a* was most closely associated with alkalinity (r=0.512), sediment chl *a* (r=0.477), andpH (r=-0.459) (Appendix 1). Periphyton chlorophyll *a* was most closely associated with percent carbonate in the sediments (r=0.744), sediment pore water TN (r=0.604), and TP (r=-0.598).

In 2001, epiphyton abundance was similar in all the ponds studied with an average chlorophyll *a* concentration of 0.31 μg/cm^2 (Figure 4.4.2a). There was no difference found between Connected and Isolated sites (p=0.5079). In all the ponds the epiphyton abundance changed throughout the sampling season with the lowest densities occurring in August (p=0.0336). However, this seasonal trend was different for the Isolated sites than for the Connected sites (p=0.0089). In the Isolated ponds the epiphyton abundance was constant throughout the season with one high peak in the first week of June. This high June value was due solely to higher epiphyton densities in Emile's Pothole and Wye's Pond. The Connected sites, on the other hand, increased throughout June, remained higher than the Isolated sites in July and then dropped to chlorophyll values similar to the Isolated sites in August (Figure 4.4.2a).

Periphyton abundance was also similar in all the ponds in 2001 (Figure 4.4.2b). On average, periphyton chlorophyll *a* was 2.56 μg/cm^2 regardless of

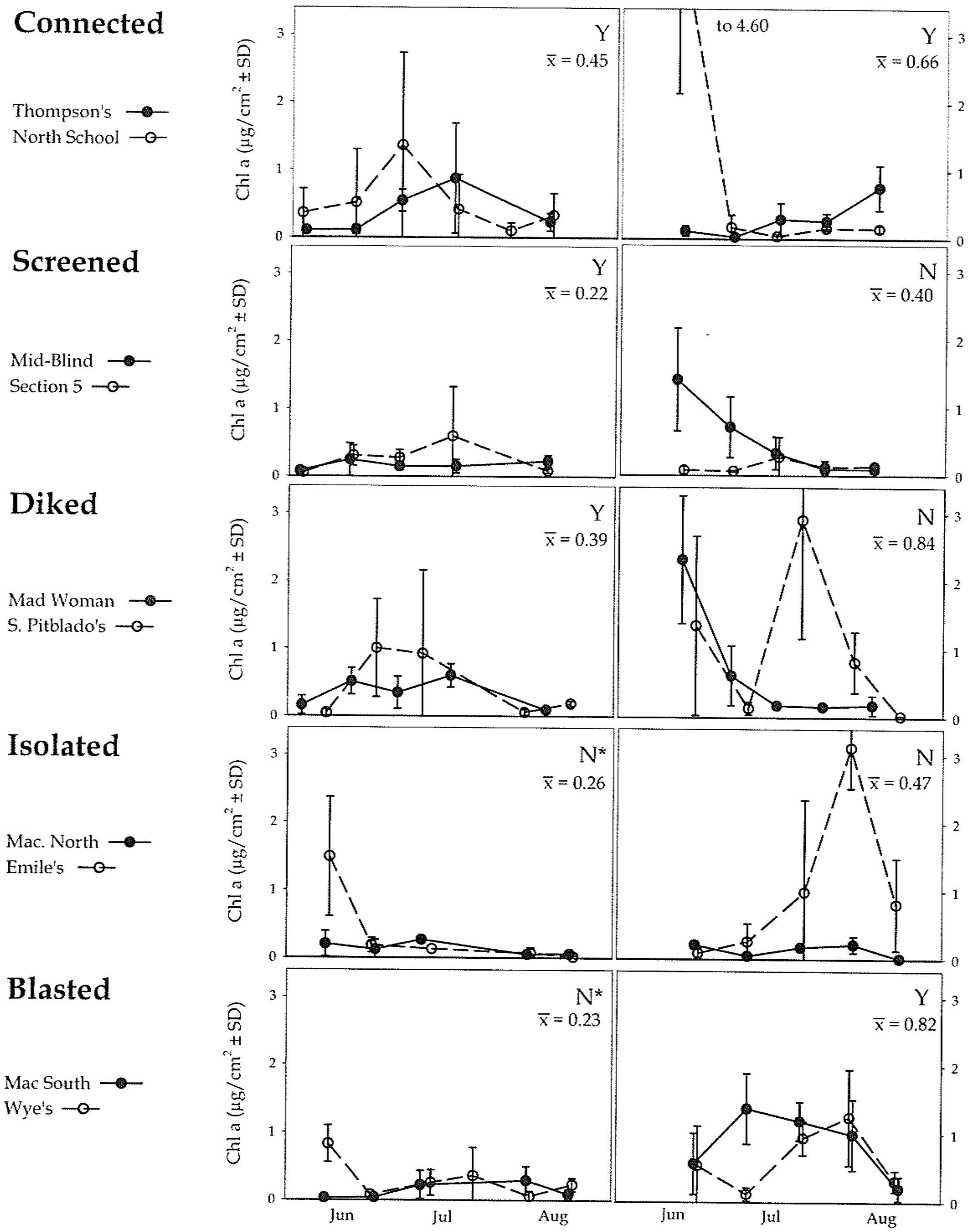


Figure 4.4.2a: Seasonal variation in mean density of epiphytic algae (µg/cm² ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

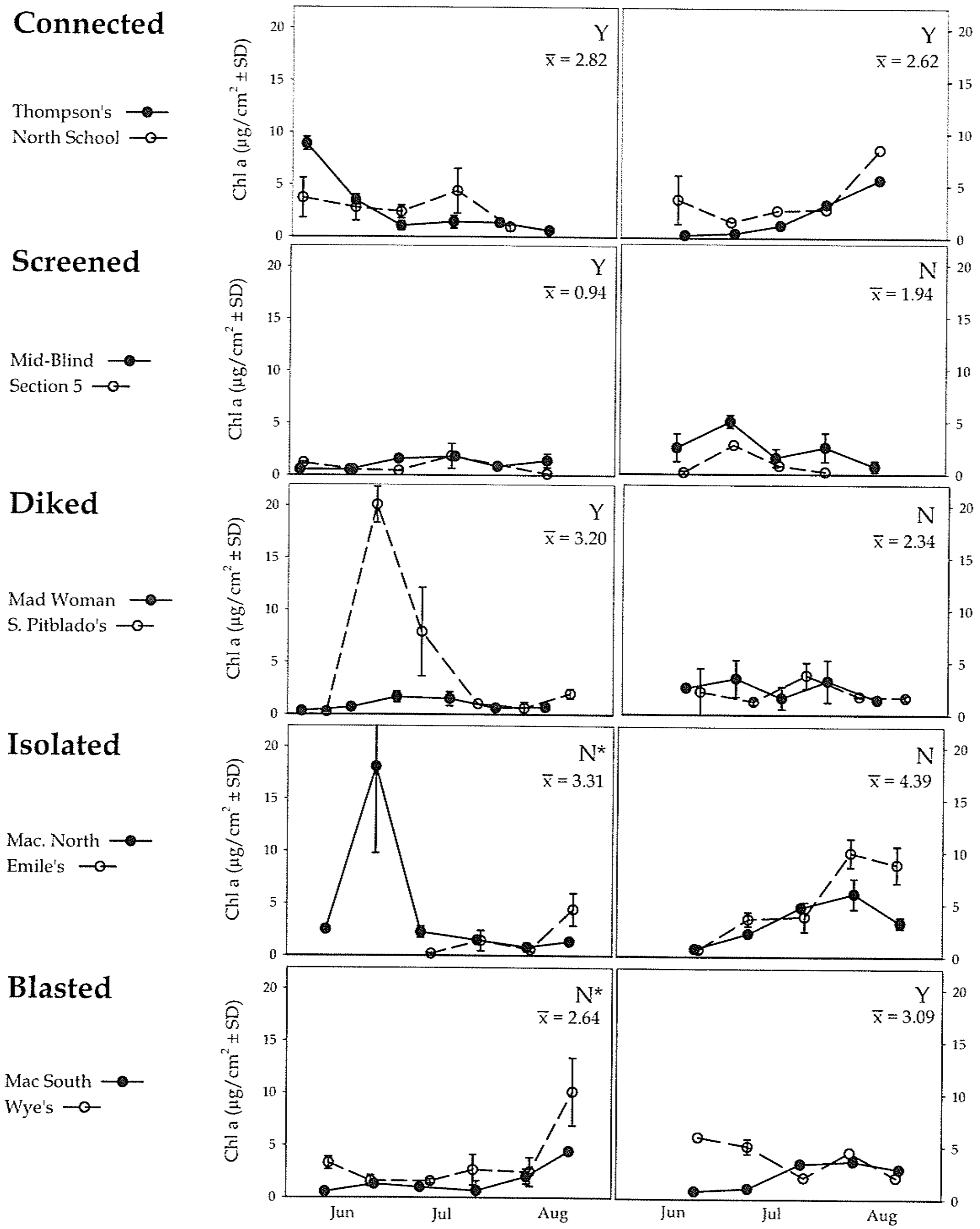


Figure 4.4.2b: Seasonal variation in mean density of periphytic algae (μg/cm² ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

connectivity (p=0.3784). Unlike epiphyton, there was no seasonal periphyton trend (p=0.3299), and no combined treatment and time effect (p=0.2937).

In 2002, the average epiphyton abundance in the ponds studied was 0.65 μg/cm^2 (Figure 4.4.2a). No significant differences were found in epiphyton abundance based on the accessibility to large carp (p=0.4597) or treatment (p=0.6053) and no seasonal trends were found.

The average abundance of periphyton in 2002 was 2.94 μg/cm^2 (Figure 4.4.2b). Periphyton abundance, like epiphyton abundance, did not differ with accessibility of large carp (p=0.7153) or treatment (p=0.5740).

In 2002 an attempt was made to determine if there was a difference in periphyton growth based on depth from the surface. However, because the water in many of the ponds was too shallow, it was often not possible to collect a sample at the lower depth. As a result there were only four ponds, all of which were accessible to large carp, in which comparisons could be made (Figure 4.4.2c). In these ponds the periphyton abundance appeared to be the same regardless of depth from the surface (means for each depth where both segments were available: 10-20 cm: 2.89 μg/cm^2 and 20-30 cm: 2.99 μg/cm^2).

4.4.3 Sediment Associated Algae (SAA)

Based on the seasonal averages from 2002, SAA was most closely associated with frequency of observance of large carp (r=0.568), pH (r=-0.499), and epiphyton chl *a* (r=-0.477) (Appendix 1).

In 2001, there was an overall seasonal trend in which SAA decreased throughout July followed by an increase in SAA at the end of August ($p<0.0001$) (Figure 4.4.3a). The mean seasonal abundance of sediment chlorophyll was higher in the Connected ponds (15.1 $\mu g/cm^2$) than in the Isolated ponds (12.3 $\mu g/cm^2$); however, this was not a statistically significant difference ($p=0.1882$). It is important to note that the mean seasonal abundance of SAA in Emile's Pothole, an Isolated pond, was the second highest mean abundance of SAA in 2001 of all the ponds (18.2 $\mu g/cm^2$). If Emile's Pothole was not considered, the average abundance of SAA in 2001 for the Isolated ponds was much lower (10.4 $\mu g/cm^2$). By removing Emile's Pothole from the analysis the difference in abundance of SAA between Isolated and Connected ponds became significant ($p=0.035$).

In 2002, seasonal trends in SAA were again present (Figure 4.4.3a). In the second year of the study there was a decrease in SAA chlorophyll from June to September ($p=0.0001$). Isolated and Diked ponds had the least change in SAA over the season, while the Blasted ponds and North School Bay (Connected) showed the greatest change (Figure 4.4.3a); however, this trend was not significant ($p=0.2866$). Sediment chlorophyll was lowest in the Isolated and Diked ponds (8.4 and 8.9 $\mu g/cm^2$ respectively) but not significantly different from other treatment means ($p=0.1913$).

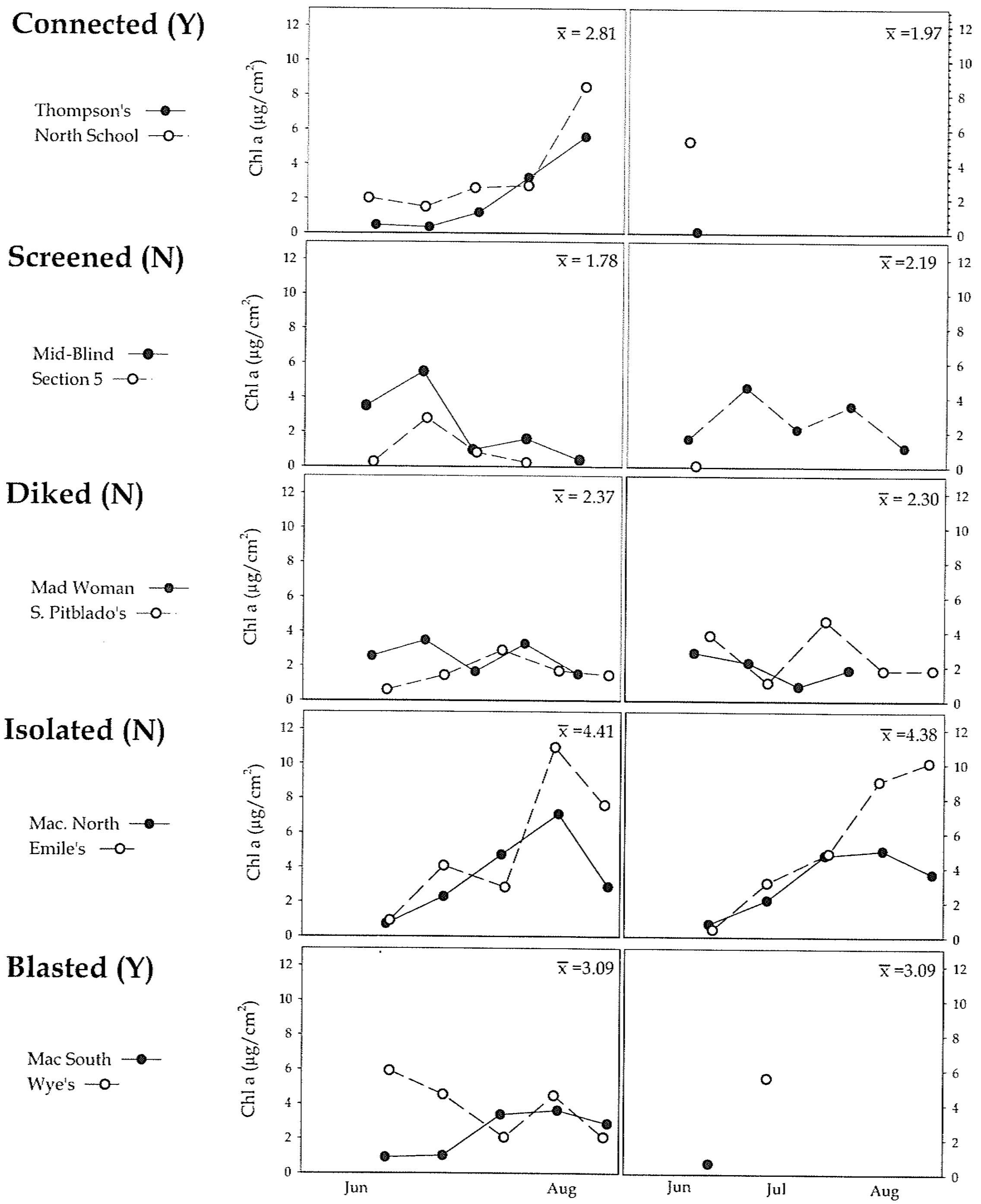


Figure 4.4.2c : Seasonal variation in mean density of periphytic algae (µg/cm²) found at two depth ranges for each pond in 2002. There are many missing data points for the 20 to 30 cm depth because the water depth at the collection site was shallower than 30 cm. Ponds are grouped by treatment to show the effect of the manipulation as well as the presence (Y) or absence (N) of large carp.

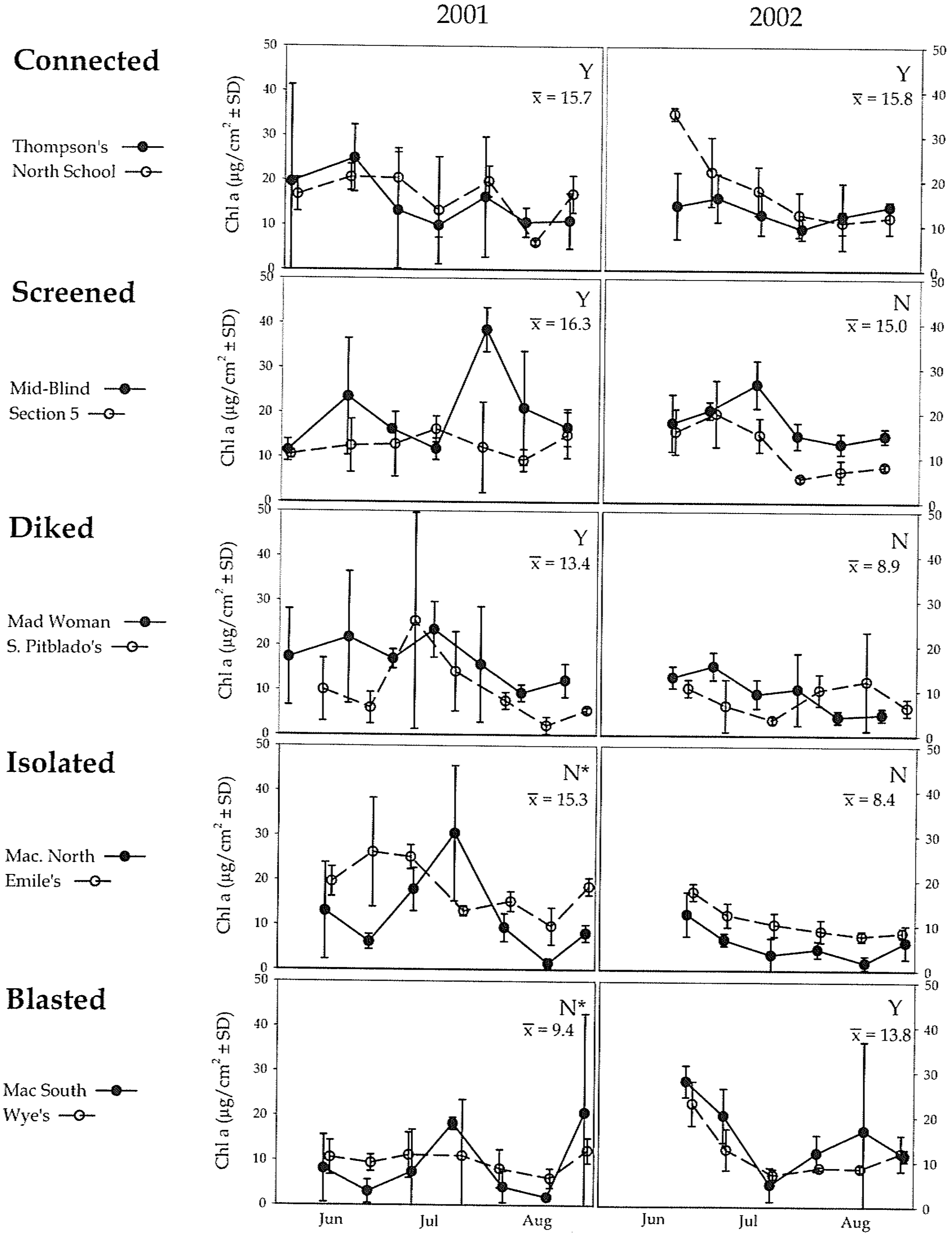


Figure 4.4.3a: Seasonal variation in mean density of sediment associated algae (SAA) (µg/cm^2 ± SD) for each pond over the two years of the study. Ponds are grouped by treatment to show the effect of the manipulation as well as the effect of the presence (Y) or absence (N) of large carp. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the sampling season.

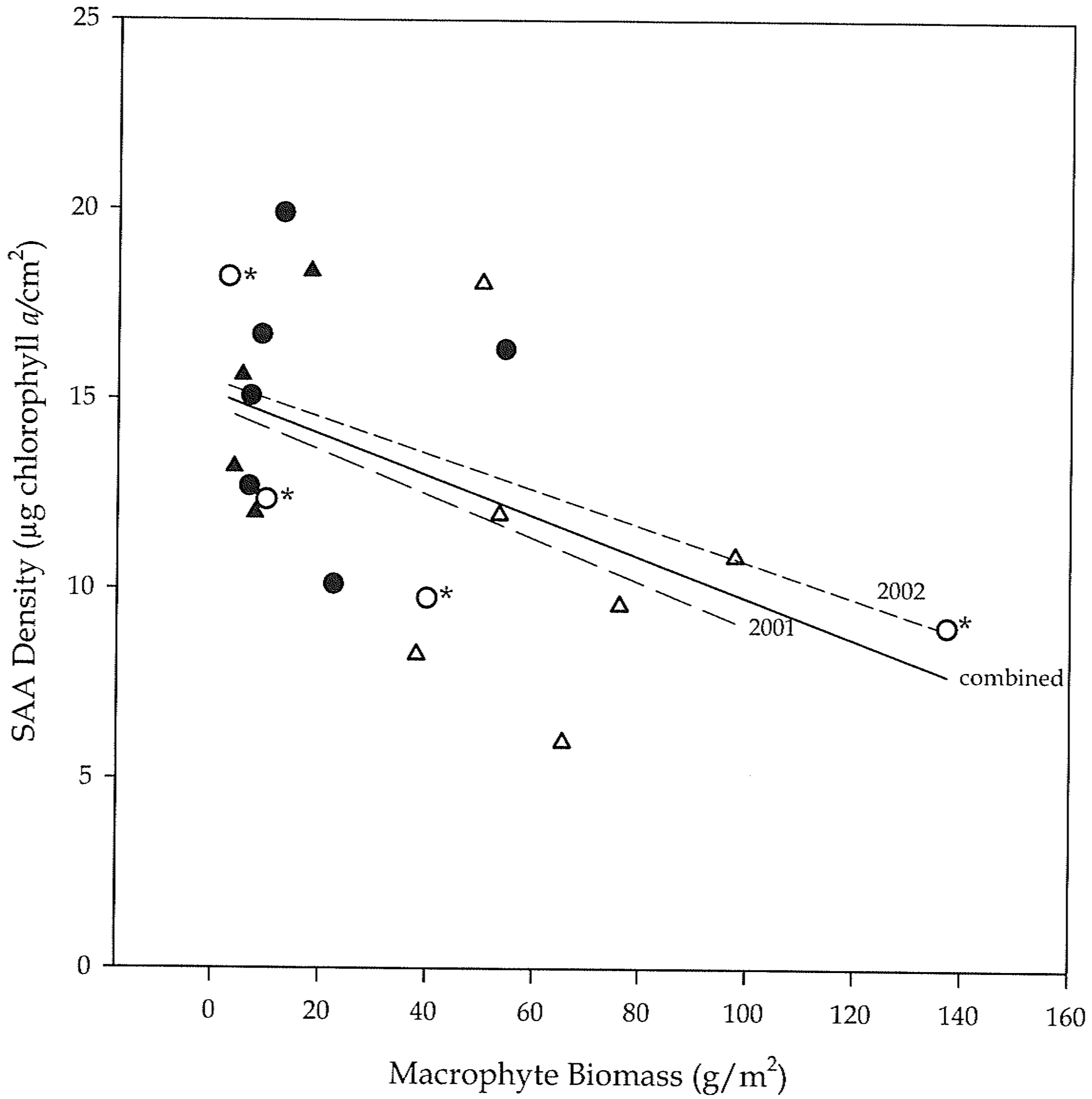


Figure 4.4.3b : Simple-linear regression of SAA and submersed macrophytes: combined years (r^2 =0.257; p-value = 0.023), 2001 (r^2=0.267) and 2002 (r^2 = 0.217). The data presented are the annual mean densities for 2001 (circles) and 2002 (triangles) for each of the ten ponds studied. Closed symbols indicate that carp were present, open symbols indicate ponds in which large carp were absent. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the season.

Using data from both years, a slight inverse correlation was found between the mean seasonal abundance of SAA and macrophyte biomass (Figure 4.4.3b; $r^2=0.257$, $p=0.023$).

4.5 Submersed vegetation

Throughout both years of the study, ponds that were accessible to large carp had, on average, less submerged vegetation than ponds that were inaccessible to large carp (Figure 4.5.1). Based on the seasonal averages from 2002, macrophyte biomass was most closely associated with pH ($r=0.792$), density of YOY carp ($r=-0.714$), and OSS ($r=-0.603$) (Appendix 1, Appendix 2).

In the first year of the study the Connected ponds contained a lower biomass of submerged macrophytes than did the Isolated ponds. However, in 2001 there was a great degree of variation within the Isolated ponds, such that both the highest biomass (South MacKenzie Bay, 137 g/m^2) and lowest biomass (Emile's Pothole, 3 g/m^2) of vegetation were found in Isolated ponds. This high degree of variation within the Isolated ponds resulted in no significant difference in the average biomass of submerged macrophytes between treatments ($p=0.617$). Although there was variation within the Isolated sites, the Connected sites were more similar to each, other with average densities near 10 g/m^2. Two Connected ponds were higher: South Pitblado's Channel (22 g/m^2) and North School Bay (54 g/m^2) (Figure 4.5.1). Although in 2001 submerged vegetation taxa

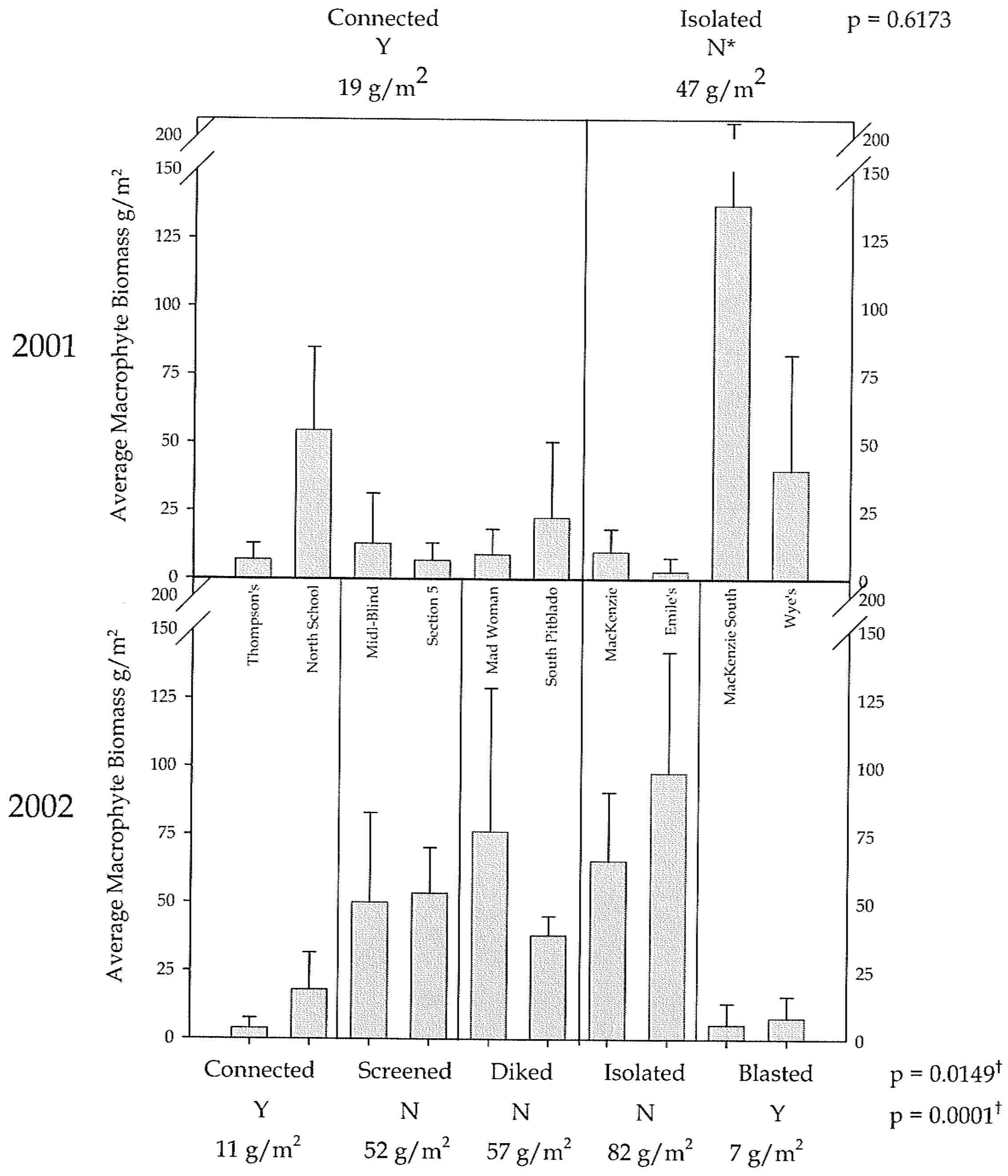


Figure 4.5.1: Mean biomass (g/m² ± SD) of submerged macrophytes in 2001 (pre-treatment) and 2002 (post-treatment) for the ten ponds in the study. The ponds are grouped by treatment and by presence (Y) and absence (N) of large carp. The numbers included with each section are the mean biomass for that treatment. P-values reported are the results of an analysis of variance comparing the group means for that year. *Unusually high water levels in 2001 may have allowed large carp to access the ponds at least for part of the growing season. †P-value is significant at $\alpha=0.05$.

were not identified, field observations indicated that *Ceratophyllum demersum* was present in these two Connected ponds.

The trends were clearer in 2002. On average, ponds that were inaccessible to large carp had a higher biomass of submerged vegetation than ponds that were accessible to large carp (p=0.0001). Post-manipulation, ponds that were Screened or Diked contained more vegetation than they did in the pre-manipulation year. The reverse was true for Blasted ponds (Figure 4.5.1). An ANOVA comparing the change in biomass of submersed macrophytes from 2001 to 2002 indicated that there was a significant treatment effect (p=0.0131).

Plant identification in 2002 revealed six submerged vegetation taxa growing in the study ponds (Table 4.5.1). *Stuckenia* sp. was the only macrophyte found in all ten ponds. As was observed in the first year of the study, North School Bay had a higher biomass of submerged vegetation than the other Connected ponds (Figure 4.5.1). Identification of the samples collected in 2002 revealed that most of the plant biomass in North School Bay was from the unattached species *Ceratophyllum demersum* (Table 4.5.1). This was consistent with field observations from 2001. *Chara* sp. was only found in one pond in 2002, Mackenzie Bay North, a long-term Isolated pond. Field observations from 2001 indicate that *Chara* sp. was present in South MacKenzie Bay; however, there is no indication as to whether or not *Chara* sp. was present in North MacKenzie Bay in the first year.

Table 4.5.1: Biomass (g/m^2) ± SD (indicated in brackets) of submerged vegetation in 2002 broken down by species. Ponds are grouped by treatment and the presence (Y) or absence (N) of large carp. *Taxon was present in the pond but was not found in the quadrats sampled.

	Connected (Y)		**Screened (N)**		**Diked (N)**		**Isolated (N)**		**Blasted (Y)**		**Mean**
	Thom.	**N.School**	**M. Blind**	**Section 5**	**Mad W.**	**S.Pitbld.**	**Mac. N**	**Emile's**	**Mac. S.**	**Wye's**	
Stuckenia sp.	3.89 (3.71)	0.09 (0.21)	37.44 (29.25)	53.45 (16.69)	59.93 (19.81)	10.66 (19.81)	14.98 (25.98)	97.61 (44.45)	1.65 (3.69)	6.86 (6.31)	**28.66 (32.61)**
Ceratophyllum demersum		15.95 (12.52)	12.62 (27.43)			3.09 (5.24)		0.09 (0.21)		0.04 (0.06)	**6.36 (7.43)**
Utricularia macrorhiza		0.19 (0.43)			0.11 (0.24)	24.28 (13.61)				0.94 (2.07)	**6.38 (11.94)**
Bryophyte		0.25 (0.16)							3.69 (5.06)	0.01 (0.02)	**1.32 (2.06)**
Myriophyllum sibiricum		1.51 (1.92)			16.09 (24.40)		*Present				**8.80 (10.31)**
Chara sp.							50.50 (5.50)				**50.50 (5.50)**
All Taxa	**3.89 (3.71)**	**18.00 (13.59)**	**50.06 (32.88)**	**53.45 (16.69)**	**76.13 (52.66)**	**38.02 (7.05)**	**65.48 (25.11)**	**97.71 (44.37)**	**5.34 (7.85)**	**7.84 (7.96)**	**41.59 (32.61)**

4.6 Multivariate analysis

Principal components analysis (PCA) revealed that there was a strong separation of ponds based on their location in Delta Marsh and sampling year(Figure 4.6.1). Ponds that were located on the east side of the Assiniboine diversion tended to be located toward the left side of axis 1, while ponds on the west side of the diversion tended toward the right side of axis 1. Axis 2 appeared to be related to sampling year, separating 2001 and 2002. Redundancy analysis confirmed that there was a strong relationship between the 21 variables and geographic location and year. A combination of these variables accounted for 39 percent of the variation in the data.

A second PCA was performed on the same set of variables as the first PCA and included UTM easting, UTM northing and year as covariables; this removed the variation due to location and year from the analysis (Figure 4.6.2). The resulting ordination accounted for 68 percent of the remaining variation in the first two axes. PCA axis 1 (PCA 1) accounted for 49 percent of the remaining variation and corresponded to macrophyte biomass, total phosphorous (TP), inorganic suspended solids (ISS), chlorophyll *a*, organic suspended solids (OSS) and silica. PCA axis 2 (PCA 2) accounted for 19 percent of the remaining variation, and was most strongly associated with total reactive phosphorous (TRP) and chlorophyll *a*. When the pond's accessibility to large carp was superimposed on the ordination diagram, PCA 1 roughly divided the ponds. The

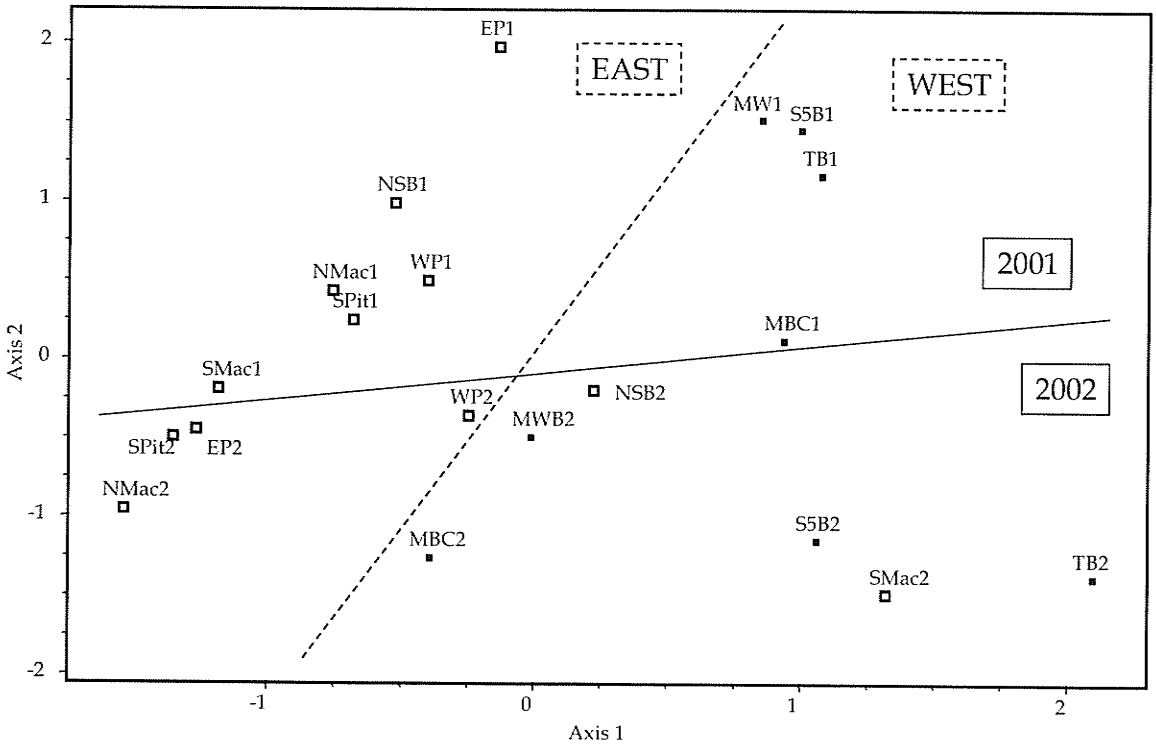


Figure 4.6.1: Ordination diagram of 20 samples; the 10 ponds in each of the 2 years in the study. Ponds are labelled by name, and year. Open and closed symbols indicate ponds on the east and west sides of the Assiniboine diversion respectively. The ordination diagram was the result of a principal components analysis of the seasonal averages of 21 variables. Axis 1 and axis 2 summarize 45.9 and 19.9 percent of the variation between the samples respectively. The lines superimposed on the ordination have been placed to indicate how Axis 1 shows an east to west gradient (dashed line), and axis 2 separates 2001 from 2002 (solid line). Redundancy analysis confirmed that location in Delta Marsh (UTM easting and northing) as well as year accounted for 39.1 percent of the variation in the data.

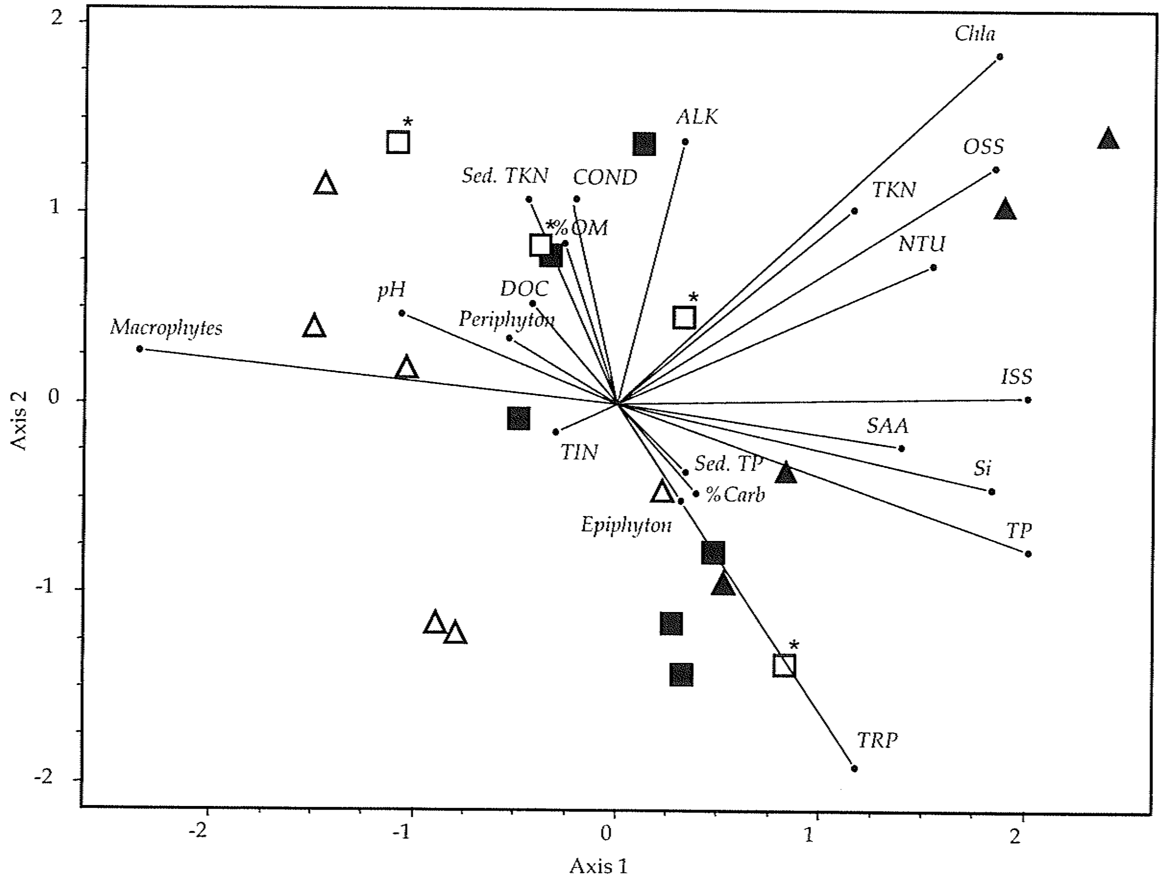


Figure 4.6.2: The ordination biplot resulting from a principal components analysis (PCA) of 20 samples and 21 variables with year, UTM northing and UTM easting used as covariables. The seasonal averages were used to represent the 21 variables. Axis 1 and Axis 2 accounted for 49.1 and 18.9 percent of the remaining variation in the data, respectively. Closed symbols represent ponds that were accessible to large carp; open symbols represent ponds that were inaccessible. 2001 is indicated by a square, 2002 by a triangle. *Unusually high water levels in 2001 may have allowed carp to enter the pond for a least part of the sampling season.

ponds that were accessible to large carp tended towards the right side of PCA 1. These ponds had a low abundance of macrophytes and high concentrations of nutrients (TP and Si), suspended solids, and phytoplankton. The ponds that were inaccessible to large carp tended towards the left side of PCA 1 and had a high macrophyte biomass and low concentrations of nutrients (TP and Si), suspended solids, and phytoplankton. There were a few exceptions; three ponds that were inaccessible to large carp were located further left in the ordination space than the other ponds that were inaccessible to large carp. These ponds were Wye's Pond and Emile's Pothole in 2001 and Section 5 Bay in 2002.

From 2001 to 2002, there was a change in PCA ordination space for each of the ponds (Figure 4.6.3). All of the ponds that were inaccessible to large carp in the 2002 were located further to the left side on PCA 1 in 2002 than in 2001. This included the Isolated, Diked, and Screened ponds. The reverse was true of the ponds that were accessible to large carp in 2002; the Connected and Blasted ponds moved to a position further to the right side on PCA 1. The greatest overall change occurred in a South MacKenzie Bay (Blasted). The smallest changes occurred in Wye's Pond (Blasted) and Section 5 Bay (Screened).

Redundancy analysis was used to confirm if the presence of common carp could explain the variation in the data (Figure 4.6.4). With location and year as covariables, the same previous 21 variables were constrained by two carp factors: the observed frequency of large carp and the density of YOY carp (Table 4.1.1). A combination of these two carp variables accounted for 48 percent of the

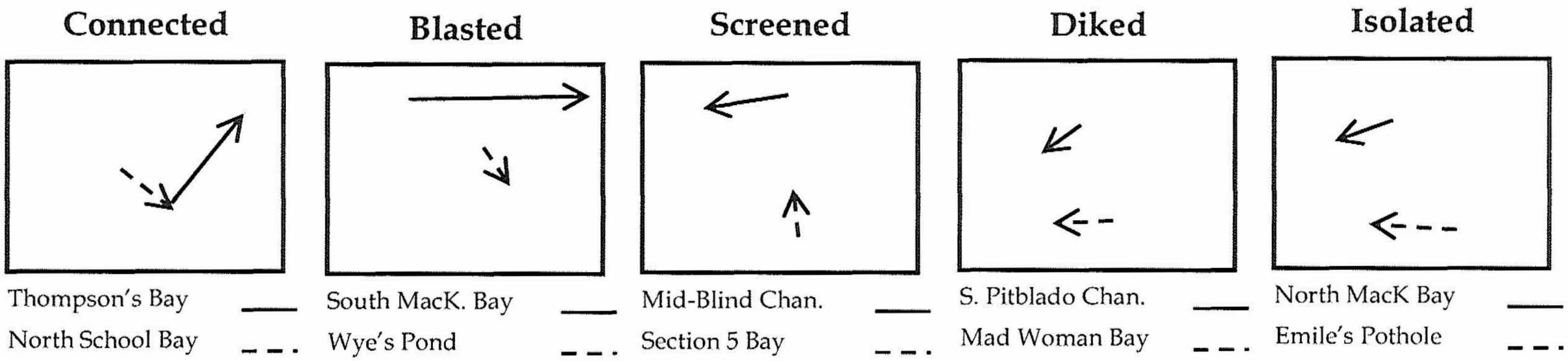


Figure 4.6.3: Change in PCA ordination space between years due to treatment. PCA 1 (horizontal axis) corresponded to macrophyte density (left) and nutrient, ISS, and phytoplankton (right). PCA 2 (vertical axis) corresponded to phytoplankton (up) and TRP (down). Connected and Blasted ponds were accessible to large carp in the second year, while Screened, Diked and Isolated ponds were inaccessible to large carp in the second year. Unusually high water levels in the first year may have allowed carp into the Blasted and Isolated sites.

remaining variation in the data. Most of this variation, 45 percent, could be explained by the first axis. This RDA axis was similar to PCA axis 1. Both frequency and density of carp were in the direction of chlorophyll *a*, ISS, OSS, TP, NTU, and Si, and opposite to macrophyte biomass.

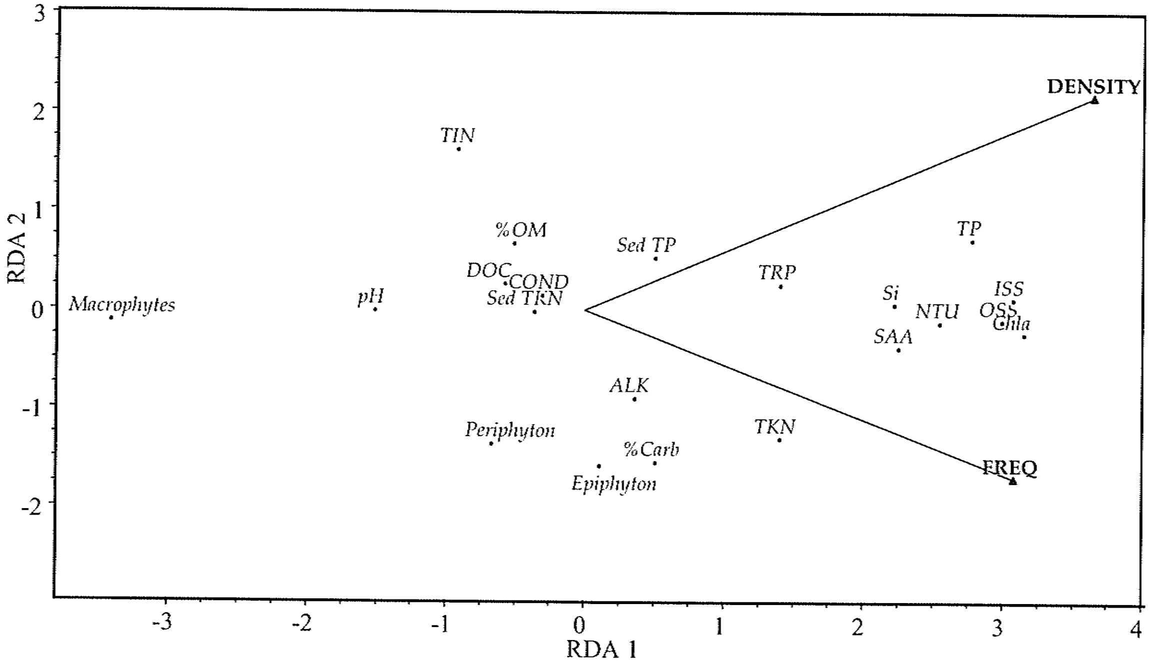


Figure 4.6.4: RDA ordination biplot of 21 variables (points) constrained by two carp factors (vectors): density of young of the year carp, and the frequency of observation of large carp. The analysis included only the information retained after removing the effects of location and sampling year; UTM easting, UTM northing, and year were included in the analysis as covariables. RDA 1 summarized 45% of the remaining variation in the data, a small amount of variation was summarized in RDA 2. Redundancy is 47.6%.

Chapter 5: Discussion

5.1 Impact of Carp on Delta Marsh

The impact carp can have on a wetland occurs by two methods; mostly through their activity during spawning and feeding, but also through waste excretion (Figure 5.1). During spawning and feeding carp can actively disrupt the sediments, and physically damage vegetation. Their excretion of wastes adds nutrients into the water column. I hypothesised that by these means carp cause changes in the water quality, algae and submersed vegetation of Delta Marsh.

5.1.1 Effects on Turbidity

I hypothesised that greater turbidity in the water column would be associated with the presence of carp. The increase in turbidity would be caused by the carp spawning and feeding activity, as well as through waste excretion (Figure 5.1). During spawning and feeding, carp would disturb the sediments leading to sediment resuspension and the release of nutrients from the sediment. Sediment resuspension would increase the amount of suspended solids, and thereby increase turbidity. As well, the nutrients added to the water through the disruption of sediments and waste excretion would promote the growth of phytoplankton, which would also contribute to higher turbidity. The results of this study partially support this hypothesis.

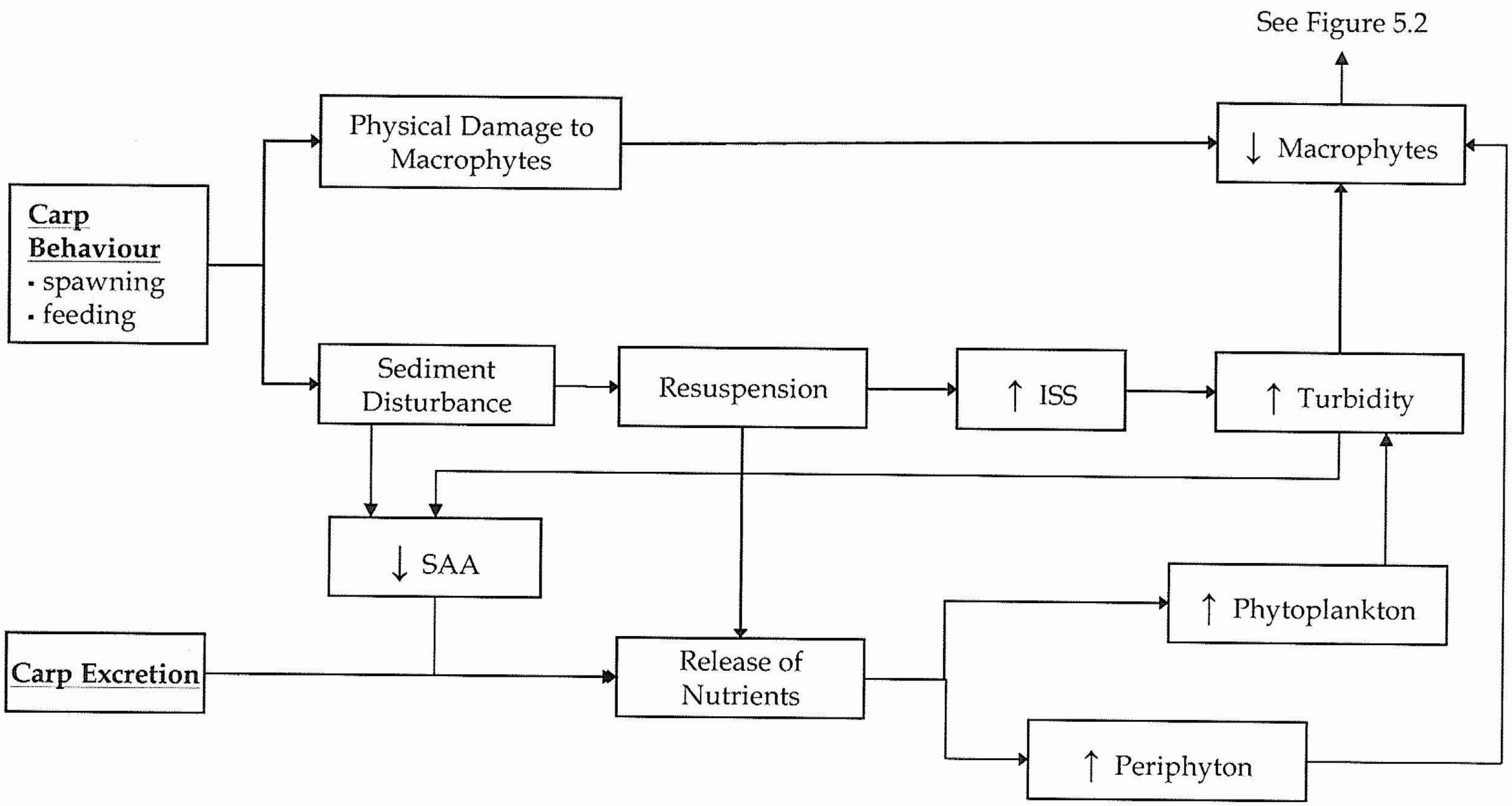


Figure 5.1: Hypothesized model of the impact of common carp at Delta Marsh. Carp impact the system through their spawning and feeding behaviour and excretion of wastes.

There is some indication that there was a positive relationship between the accessibility of large carp and turbidity in Delta Marsh. Although there was no significant difference in the average seasonal turbidity between sites that were accessible or inaccessible to large carp, most accessible sites tended to be turbid, while inaccessible sites tended to be clear. Even though there was no significant effect of carp on turbidity found, accessible ponds had on average significantly more inorganic solids suspended in the water column than did inaccessible ponds throughout both years of the study. This is similar to studies elsewhere, which have found that suspended solids increased in the presence of carp (Breukelaar et al. 1994; Zambrano et al. 1999; Angeler et al. 2002). As well, the seasonal trend in turbidity of a pond depended on its accessibility to large carp. On average, accessible ponds showed the most change in turbidity throughout the season, with their greatest turbidity occurring in the spring. This is the time of the year when the majority of carp spawning occurs (Swee & McCrimmon 1966). My results therefore indicate that carp resuspend sediment thereby increasing inorganic suspended solids, which may affect the overall turbidity in some cases, and that the greatest effect that carp had on the turbidity was in the spring when they were actively spawning.

There were some exceptions to this general trend in turbidity. In 2001, North School Bay and South Pitblado's Channel, two connected ponds, were not turbid like the other connected ponds, and they did not show the high spring turbidity of other sites accessible to large carp. The relative abundance of submersed vegetation may account for these differences. In 2001, both of these ponds contained an abundance of macrophytes and this may have contributed to reducing the effect of carp on turbidity by reducing the amount of suspended sediment and phytoplankton (Roberts et al. 1995) (see Section 5.4 and Figure 5.2 for an explanation). Badiou (2005) found that a similar situation occurred in experimental wetlands at Delta Marsh; where macrophytes continued to grow in the presence of carp, their effect on turbidity was minimized.

In both 2001 and 2002 Wye's Pond (Blasted) was also an exception to the trend in turbidity that sites accessible to large carp were more turbid than sites that were inaccessible. In the first year of the study, the turbidity in Wye's Pond was the highest of all the isolated ponds; however, this pond was not truly isolated and large carp were frequently seen. The presence of carp may have lead to higher turbidity here than would otherwise have been expected. In the second year, after a channel was blasted into Wye's Pond, there was only a small increase in turbidity and suspended solids, unlike the other Blasted pond, South MacKenzie Bay, that showed large increases in both parameters. This was possibly due to an ineffective application of the treatment at Wye's Pond; the treatment was applied late (June 14), and the channel was not as open or as deep as intended. As a result, carp were not present in the pond during the peak in their spawning season, and large carp were seen only rarely in the pond in 2002. The lack of intended situations in Wye's Pond in both 2001 and 2002 resulted in

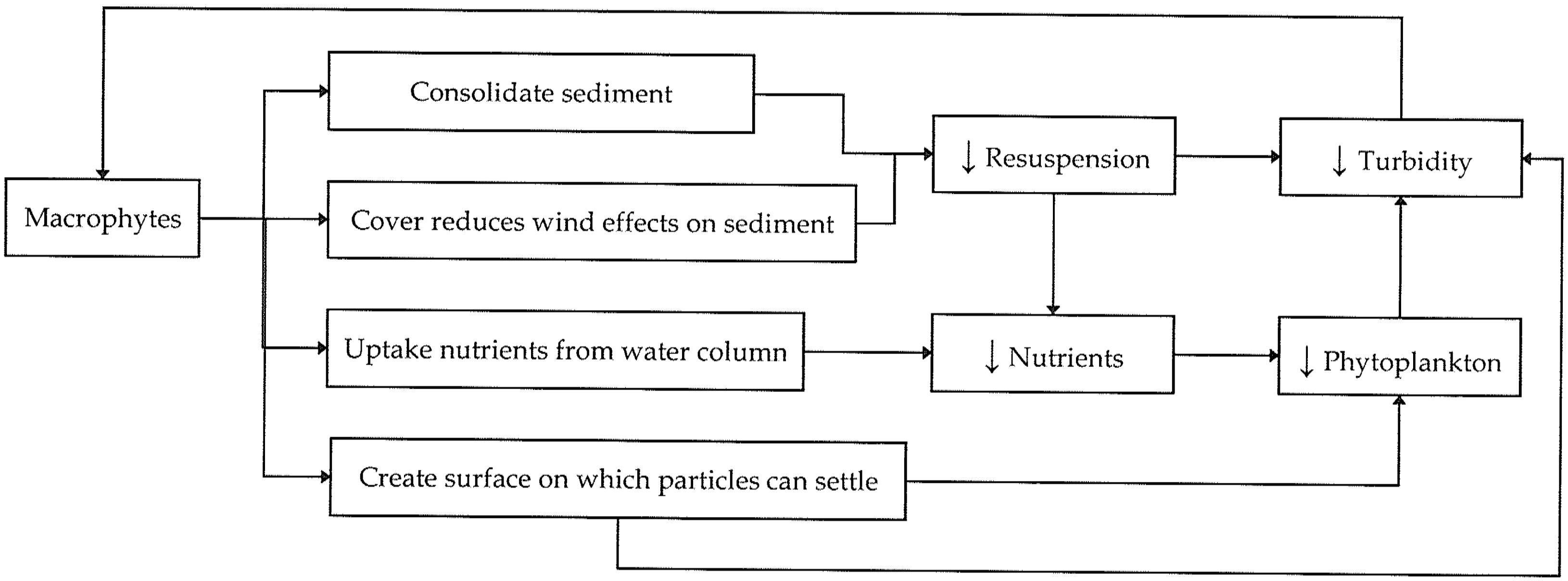


Figure 5.2: A feedback loop shows theoretically how the presence of submerged vegetation can promote its own growth. This process is fundamental to maintaining the clear water stable state.

turbidity levels that were different than would be expected based on the pond's intended accessibility to large carp.

In shallow water systems, turbidity can be influenced by wind through sediment resuspension. Two factors which determine whether or not wind resuspension will occur are wind speed and water depth (Scheffer 1998). In this study, both of these factors were compared against turbidity. *In situ* wind-speed was measured and did not show a connection with turbidity; however, this does not conclusively show that wind-speed had no effect on turbidity. A better way of determining if wind-speed was influencing turbidity, would have been to compare turbidity to the *in situ* wind-speed of a pond relative to the current wind-speed at a set location and averaged over a longer period of time than the *in situ* measurements. This was my initial plan for the study, however, the anemometer at the nearest weather station (Delta Marsh Field Station) was not functioning properly for much of the study period. As a result, a comparison of relative wind-speed was not possible. Water depth, on the other hand, was associated with turbidity. No correlation was found in the first year of the study. However, in the second year when water-levels were much lower, a weak linear relationship was found, which indicated that turbidity increased with decreasing water depth. In particular, the turbidity of Section 5 Bay (Screened) was strongly influenced by water depth. In 2002, this pond became more turbid when water levels dropped. The resuspension of sediments by wind is the probable cause of changes in turbidity there in 2002, and may have been a source of turbidity at shallow sites throughout the marsh.

The results of this study partially support the hypothesis that turbidity will be higher in the presence of carp. Carp tended to increase turbidity in the spring, and the average seasonal concentration of ISS was higher in ponds that were accessible to carp. However, the fact that carp had access to a pond did not mean that the pond was turbid. Abundant submersed vegetation may have reduced the impact carp had on turbidity in some sites. As well, wind resuspension occurred where water levels were shallow, and the overall contribution of wind to turbidity is unknown.

5.1.2 Effects on Nutrients

I hypothesised that higher nutrient concentrations in the water column would be associated with the presence of carp. The sediment disturbance associated with carp activity, as well as their excretion of wastes, would release additional nutrients into the water column (Figure 5.1). The nutrients that would most likely be effected would be P, N, and Si. The results of this study did not support this hypothesis.

The majority of the N (99%) and P (2001:84%, 2002:91%) in the ponds was present in organic form, and therefore unavailable to most autotrophic organisms. The ponds tended to have N:P molar ratios less than 16, indicating nitrogen limitation. In some ponds (Mid-Blind Channel 2002 and Emile's Pothole 2002) that were inaccessible to large carp, phosphorous was present in lower concentrations, and the N:P ratio increased to approximately 16 such that nitrogen was no longer limiting, while in other ponds (North MacKenzie Bay, South MacKenzie Bay 2001, and South Pitblado's Channel 2002) the ratio was between 18 and 23 indicating phosphorous limitation.

Overall, the accessibility of a pond to large carp was not found to increase the amount of P in the water column. All ponds that were accessible to large carp varied in P concentration throughout the season, with the highest concentrations occurring in July; however, some ponds which were inaccessible to carp (Section 5 Bay 2002 and Mad Woman Bay 2002) also had P concentrations that varied seasonally.

The water concentration of N in a pond was not affected by the pond's accessibility to large carp. Nitrogen was not found to be in significantly higher amounts in ponds that could contain carp; however, the highest N concentrations in this study were found in carp accessible ponds.

The average concentration of silica in a pond was also not affected by its accessibility to large carp. Although I did not find a significant difference between accessible and inaccessible ponds, the highest Si concentrations were found in ponds accessible to large carp, and the lowest concentrations occurred in ponds inaccessible to large carp. South MacKenzie Bay (Isolated-Blasted) was among the ponds with the lowest concentration of Si in the base-line year, and this pond had the highest post-manipulation concentration of Si of all the ponds.

The results of this study failed to support the hypothesis that nutrient concentrations would be higher in the presence of large carp. The P, N, and Si concentrations in a pond was independent of its accessibility to large carp. These results do not correspond to the much of the literature, in which, an increase of nutrients either N and/or P, occurs in the presence of carp (Richardson et al. 1990; Lougheed et al. 1998; Angeler et al. 2002; Khan et al 2003). However, other studies in natural systems have also found no change in the concentration of either N or P occurred with the removal (Lougheed & Chow-Fraser 2001) or introduction (King et al. 1997; Drenner et al. 1998) of carp.

I do not believe this hypothesis, that nutrients would be increased in the presence of carp, should be rejected on the basis of this study. The majority of the nutrients present in the ponds were in organic form, indicating that inorganic nutrient uptake would be rapid. In his study, Badiou (2005) reports that there were indications that TRP was rapidly taken up by phytoplankton. In my study, the levels of nutrients in a pond were measured biweekly or monthly, so this provided only a "snapshot" of nutrient conditions. Any differences between ponds due to carp activity may have been missed. A better way of comparing the nutrient conditions of the ponds would be to use a method that integrates nutrient availability over a longer period of time, such as nutrient diffusing substrata (NDS). NDS are left *in situ* for a set period of time, during which time a known amount of N and P are released from a substrate, alone and in combination (Gibeau & Miller 1989). The difference in algal growth on the various substrata over the sampling period (usually a few weeks) can then be used to determine what nutrients were limiting at the site. One would predict that if carp were contributing to higher nutrient concentrations, the nutrients released by the NDS would stimulate algal growth on the substrata to a lesser extent in a pond with carp than in a pond without carp.

5.1.3 Effects on Algae

I hypothesised that planktonic and benthic algal abundance would be affected by the presence of carp. The higher nutrient levels occurring due to sediment disturbance and waste excretion were expected to promote the growth of phytoplankton and periphyton (Figure 5.1). SAA abundance was predicted to decrease due to physical disturbance and lower light availability associated with the predicted higher turbidity in the presence of carp. Based on the results of this study, support for these hypotheses was equivocal.

The presence of carp may have increased phytoplankton abundance in the marsh, although there was a high degree of variation throughout the study. In the first year of the study, the ability of carp to access a pond was not shown to affect the pond's biomass of phytoplankton. However, this may have been due to the lack of true isolation of the Isolated ponds. Since the Isolated ponds were accessible to large carp for at least part of the sampling season, carp may have affected the growth of phytoplankton in the Isolated ponds, as well as the Connected ponds, therefore resulting in no significant difference between pond types. Conversely, in the second year, phytoplankton densities were highest in ponds that were accessible to large carp. However, there were two exceptions. In 2002, Wye's Pond (Blasted) and North School Bay (Connected) did not have as much phytoplankton as the other sites that were accessible to large carp. This may be a result of the late application of the treatment in the former, resulting in fewer carp entering the pond, and the presence of macrophytes in the later, which are known to decrease the abundance of phytoplankton (Scheffer 1998).

Spring phytoplankton blooms occurred in large carp accessible ponds, that did not occur in ponds inaccessible to carp. This was similar to the findings for turbidity. Ponds that were accessible to large carp followed a different seasonal trend with regards to phytoplankton and turbidity than those that were inaccessible to large carp. This suggests that in the spring, spawning carp may have been releasing nutrients into the water column, facilitating the growth of phytoplankton. Many studies have found higher abundances of phytoplankton in the presence of carp (Richardson et al. 1990; Drenner et al. 1998; Sidorkewicj et al. 1999; Angeler et al. 2002; Khan et al 2003) and that phytoplankton abundance increased with increasing carp density (King et al. 1997; Badiou 2005); however, none of the studies that I read on the effects of carp indicated the occurrence of any seasonal effect on the abundance of phytoplankton or turbidity.

The abundance of epiphyton and periphyton was not affected by the connectivity of a pond and therefore the accessibility of large carp. The presence of large carp did not increase the amount of nutrients in the water column. Therefore, it follows that without the expected increase in nutrients, an increase in algal growth would not occur. It is not surprising that no difference in the biomass of epiphyton and periphyton was found between ponds that were accessible and ponds that were inaccessible to large carp. However, as previously discussed, due to the methods used to sample nutrients in the ponds, I cannot be certain that carp did not affect nutrient levels. Some studies have found that nutrient increases in the presence of carp lead to marked increases in periphyton growth (Sidorkewicj et al. 1999), so I feel the effect of carp on epiphyton/periphyton at Delta Marsh requires further investigation.

In addition to nutrient availability, the availability of substrata is important contributing factor to epiphyton abundance. The amount of surface area available for colonization by epiphyton should therefore be considered. An estimate of the biomass of epiphyton growing on submersed vegetation (Table 5.1) in the ponds in 2002, shows that large carp may have affected the abundance of epiphyton after all. Most ponds that were accessible to large carp were estimated to have a lower biomass of epiphyton than ponds that were inaccessible to large carp.

SAA abundance was not directly influenced by the presence of common carp in Delta Marsh. Some ponds accessible to large carp had a lower abundance of SAA than inaccessible ponds; however, I found no significant difference in SAA between ponds. Similarly, Robertson et al. (1997) found no effect of carp on sediment chlorophyll *a*. I believe that any effect on SAA was indirect through carp effects on the abundance of submersed vegetation (Section 5.1.4).

Table 5.1: Estimation of epiphyton biomass expressed as dry weight per area of marsh bottom for the ten ponds in 2002. This estimation assumes that the measured abundance of epiphyton on *Typha* stems (µg chl *a*/cm^2) approximated the abundance of epiphyton growing on submersed macrophytes. Epiphyton abundance as µg chl *a*/cm^2 of surface area was converted to area of marsh bottom following the relationships between macrophyte dry weight and surface area created by McDougall (2001). Macrophyte species for which no relationship was available, were assumed to follow the same relationship as the species with the most similar growth form. Algal biomass was then estimated using a chlorophyll conversion factor of 0.25% (Goldsborough 2001).

Pond	Accessible to large carp? (Y/N)	Treatment	Estimated epiphyton biomass (g/m^2)
Thompson's Bay	Y	Connected	0.14
North School Bay	Y	Connected	0.63
South MacKenzie Bay	Y	Blasted	1.15
Wye's Pond	Y	Blasted	0.53
Mid-Blind Channel	N	Screened	2.81
Section 5 Bay	N	Screened	0.76
Mad Woman Bay	N	Diked	4.17
S. Pitblado's Channel	N	Diked	1.96
North MacKenzie Bay	N	Isolated	1.39
Emile's Pothole	N	Isolated	5.90

Submersed macrophytes played a important role in the growth of SAA in this study. SAA was inversely correlated to macrophyte biomass. In both years, SAA decreased in abundance throughout the open water season, corresponding to the seasonal expansion of macrophytes. This indicates that macrophytes were out competing SAA, most likely by limiting the amount of light available to the algae at the sediment surface.

Overall, the presence of carp did not lead to a higher seasonal biomass of phytoplankton and epiphyton/periphyton, or a lower abundance of SAA as was predicted. However, spring phytoplankton blooms did occur in ponds accessible to carp, which may have been facilitated by nutrients released from the sediments during carp spawning. These spring blooms of phytoplankton did not occur in ponds that were inaccessible to carp. The results also indicated that carp increased SAA biomass indirectly. When macrophyte abundance was reduced in the presence of carp, more light was available at the sediments, allowing for greater growth of SAA.

5.1.4 Effects on Submersed Vegetation

I hypothesised that the density and diversity of submersed vegetation would be lower in the presence of carp (Figure 5.1). Feeding and spawning carp would physically disrupt macrophytes, damaging the foliage and uprooting the plants. Sediment resuspension and the release of nutrients would lead to higher turbidity and algal growth, both of which would result in less light being available to macrophytes. Some species were expected to be more tolerant than others leading to differences in the composition of submersed vegetation. This hypothesis was supported by the results of this study.

There was strong evidence that carp influenced the submersed vegetation community of Delta Marsh. The biomass of macrophytes was lower in ponds that were accessible to large carp. When carp were prevented from entering a pond, the amount of submersed vegetation increased, and it decreased when carp were allowed access into a previously inaccessible pond. *Stuckenia* sp. was the most commonly found macrophyte species, and therefore presumably the most tolerant to the presence of carp. These plants are able to withstand the turbid conditions typically found in the presence of carp (King & Hunt 1967; Lougheed et al. 2001) and it was the only submersed vegetation found in the most turbid sites. In some instances (e.g. North School Bay 2001) the unattached species *Ceratophyllum demersum* was able to persist in the presence of carp, and may have helped to improve the water clarity, facilitating the growth of other macrophyte species.

When macrophytes are present in high abundance in a shallow system, they help to stabilize the system in a clear state (Scheffer 1998). Macrophytes reduce resuspension and compete with phytoplankton for water column nutrients thereby lowering turbidity, which in turn benefits the macrophytes. Thus a positive feedback loop is created where the presence of macrophytes encourages their own growth (Figure 5.2). As an example of this, Badiou (2005) found that the presence of macrophytes, in experimental wetlands at Delta Marsh, prevented a state-shift from occurring in following the introduction of carp.

Similar to other studies (King & Hunt 1967; Roberts et al. 1995; Drenner et al. 1998) charophytes were only present where large carp did not have access to the pond, suggesting they are especially sensitive. Charophytes may therefore be an important indicator species, when assessing the impact of carp at Delta Marsh.

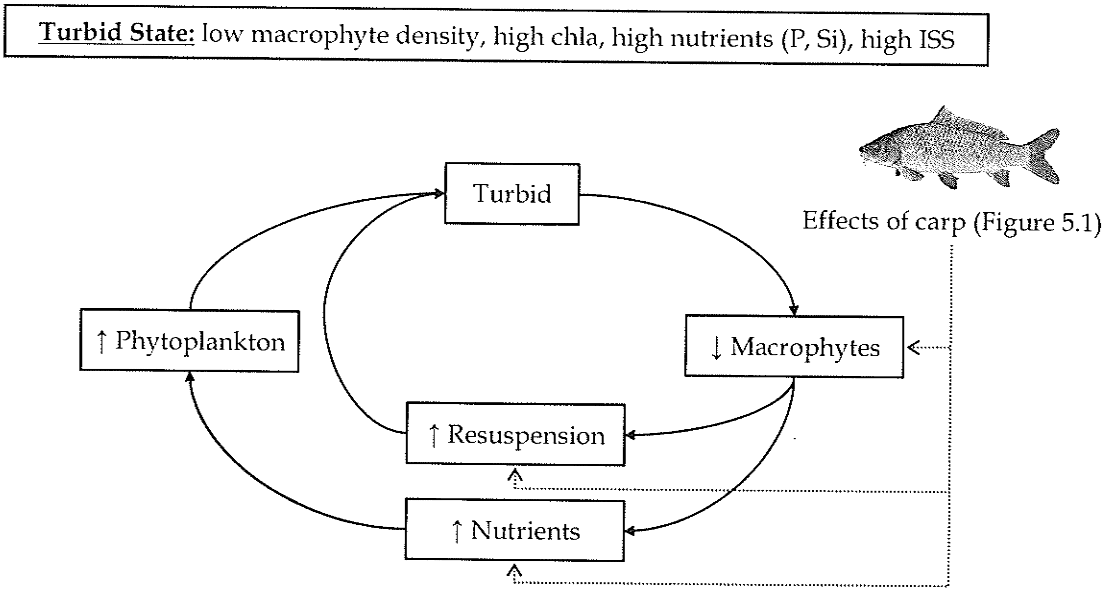
5.1.5 Overall impact and the presence of two states

Water quality, and the abundance of algae and macrophytes, was highly variable during the study. Differences in the overall state of ponds occurred between years and locations in the marsh. In the first-year of the study, conditions within the ponds tended to be more similar to each other than in the second-year when there was greater separation between clear and turbid sites. The location of a pond within the marsh was also important. Ponds on the west side of the Assiniboine Diversion tended to be more turbid, had higher levels of TP, and seemed to be more susceptible to the impact of carp than those on the east side of the Diversion. Once the variation due to location and sampling year was removed, the differences between ponds that were accessible to large carp and those that were inaccessible became evident, although some variation remained unexplained.

The ponds separated out based on their biomass of submersed vegetation, P concentration, amount of suspended solids, phytoplankton abundance, and silica concentration. Ponds that were accessible to large carp were characterised by a low biomass of macrophytes, high nutrient concentrations (P and Si), high amounts of suspended solids, and high phytoplankton abundance. The reverse was true for ponds that were inaccessible to large carp.

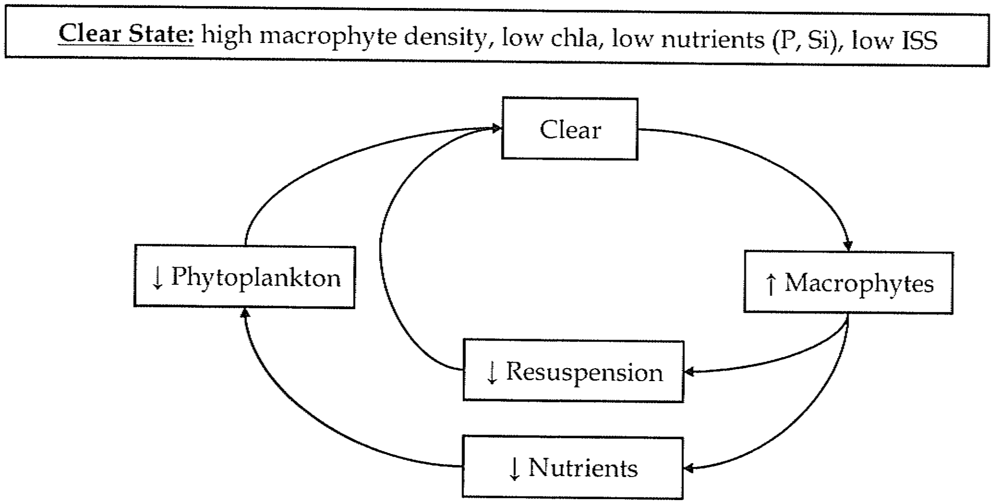
The differing conditions observed between the ponds that were accessible and those that were inaccessible to large carp correspond to the stable states suggested by Scheffer (1990). A clear state is dominated by macrophytes, and a turbid state is dominated by phytoplankton. Ponds that were accessible to large carp demonstrate the turbid state, while ponds that were inaccessible to large carp demonstrate the clear state. Ponds in the turbid state were characterised by a low macrophyte biomass, high phytoplankton chlorophyll *a*, high nutrients (P, Si), and high ISS, while ponds in the clear state had a high macrophyte biomass, low phytoplankton chlorophyll *a*, low nutrients, and a low ISS (Figure 5.3). The two states appeared to be maintained by internal mechanisms controlling the relative abundance of macrophytes, phytoplankton and suspended solids. Scheffer (1990) proposed that if two stable states can occur within the ecosystem, the removal or introduction of a perturbation such as carp, can be sufficient to cause the system to switch states.

Generally, in this study, the effects of carp appeared to lead to a turbid state, and the removal of carp appeared to allow a pond to return to the clear state (Figure 5.3). Post-manipulation, ponds that were inaccessible to large carp became clearer, had lower levels of phytoplankton and P, and their abundance of



Self-maintaining turbid state

When carp were removed from the pond it returned to the clear state



Clear state maintained by macrophyte feedback loop (Figure 5.2)

Figure 5.3: Conceptual model of clear and turbid states seen at Delta Marsh. Model shows the mechanisms working to maintain the states as well as the effect of carp on the state of a pond.

including the Isolated ponds, since in 2001 carp were present in all the submersed vegetation increased from the previous year. These ponds switched from a turbid state to a clear state. Improvement was likely seen in all the ponds ponds thus pushing the Isolated sites closer to the turbid state in 2001 than would otherwise be expected. Conversely, the ponds that were accessible to carp in 2002 moved toward the turbid state. This was expected for the Blasted sites, as the introduction of carp was expected to cause a shift to the turbid state (Figure 5.3). The further movement toward the turbid condition that occurred in the Connected sites is likely attributable to the lower water levels in 2002 compared to 2001. Carp will have a greater impact in shallower sites (Scheffer 1998) and the lower water levels in 2002 may have allowed for greater wind-resuspension of the sediments to occur.

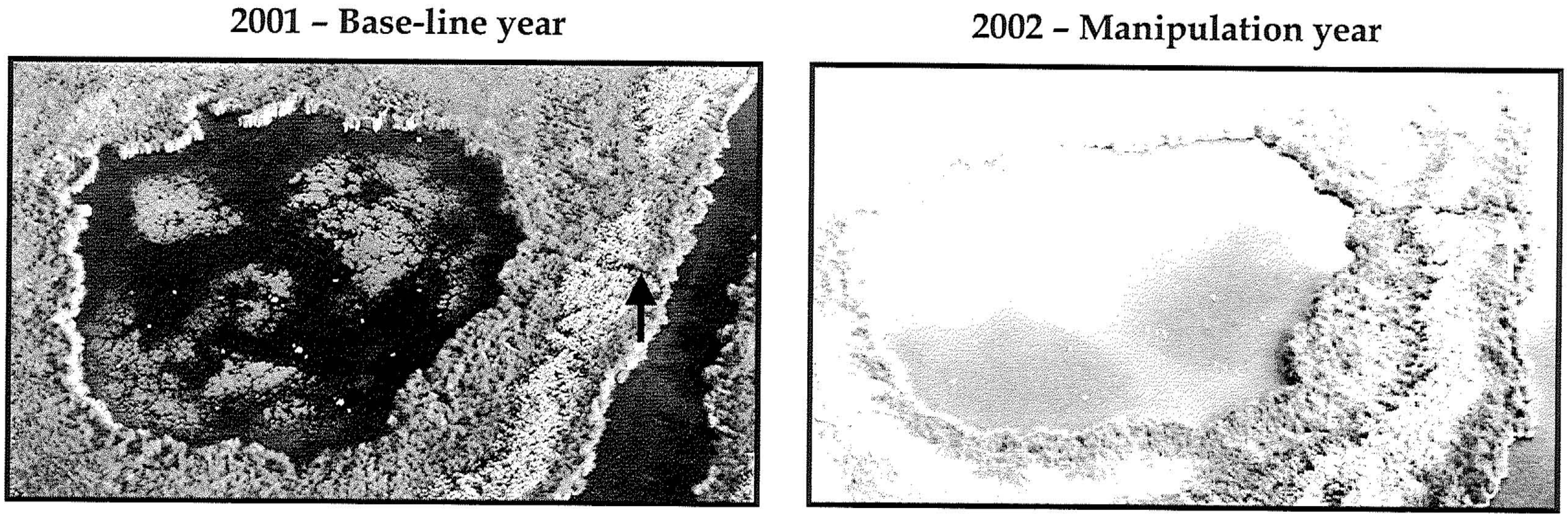
The state-shifts that occurred in this study, were contrary to what Badiou (2005) found in the Marsh Ecology Research Program (MERP) cells at Delta Marsh. He found that after the introduction of carp, even at high densities of carp, a state-shift did not occur. Badiou (2005) hypothesised that the high levels of DOC, which occurred in the MERP cells, were limiting to the growth of phytoplankton. He suggests, this combined with the continued growth of macrophytes, prevented a state-shift from occurring. However, he did observe a state-shift in his mesocosm experiments following the introduction of carp. The mesocosm experiments were conducted in the West-Blind Channel (near the Mid-Blind Channel in the current study) the levels of DOC in the Blind Channel were much lower, and fewer macrophytes were present, here Badiou (2005) found that a state-shift occurred. The levels of DOC in the ten ponds in the current study were much lower than those found in the MERP cells by Badiou (2005) (Table 5.2). I suggest that it may be the difference in the levels of DOC, that explains the conflict between the current study and Badiou's (2005) study in the MERP cells.

South MacKenzie Bay (Blasted) was the pond in this study that showed the most clear example of a shift in stable states (Figure 5.4). In the baseline year, this pond was clear, had low levels of nutrients (P and Si), low phytoplankton abundance, and the highest biomass of macrophytes of all the ponds in the study. In the manipulation year, a channel was created, granting carp access to the pond. The pond switched from its former clear state to the turbid state. In 2002, South MacKenzie Bay had high turbidity, high levels of nutrients (P, Si, and N), large blooms of phytoplankton, and the second lowest biomass of macrophytes of all the ponds. The introduction of carp to South MacKenzie Bay is likely what caused the pond to switch from its former clear state to the turbid state.

Wye's Pond (Blasted) and Section 5 Bay (Screened) showed the least change of all the ponds following the manipulation of carp accessibility. Wye's Pond showed little change following the introduction of carp, while Section 5 Bay failed to clear as would be expected with the removal of carp. The small

Table 5.2: Mean and range of concentrations of dissolved organic carbon (DOC) found in 2001 and 2002 in the ten ponds studied. Also included is the range of DOC concentrations found by Badiou (2005) in the MERP cells. All concentrations are given in mg/L.

2001 (current study)	**2002 (current study)**	**MERP cells (Badiou 2005)**
Mean: 16.4 Range: 8 - 26	Mean: 29.92 Range: 13 - 55	Range: 58 - 91



2001 – Base-line year

2002 – Manipulation year

Figure 5.4: Photographs showing the switch from the clear-state to the turbid-state that occurred in South MacKenzie Bay following the creation of a channel into the pond from the adjacent Cooks creek. In the base-line year (2001), when the pond was inaccessible to common carp, the water in the pond appeared clear, and an abundance of submersed vegetation was visible. Post-manipulation (2002), when the pond was accessible to carp, the water was visibly turbid and the abundance of submersed vegetation, seen in the previous year, was absent. A path through the upland emergent vegetation was present in the 2001 photo. This was the location where the subsequent channel was created. The path and subsequent channel are indicated with an arrow.

difference between years in Wye's Pond can be explained by two experimental problems. In 2001, the pond was not truly isolated, and large carp were frequently seen throughout the sampling season. In 2002, the treatment was applied late, and the blasting was not as effective as it could have been. The channel created by the blast was not as clear of debris or as deep as intended, so carp were rarely seen in Wye's Pond in 2002. The lack of improvement in Section 5 Bay following screening in 2002 can be attributed to factors other than the effects of carp. This pond remained in the turbid state post-manipulation. The extremely shallow conditions of Section 5 Bay in 2002 may have allowed wind-resuspension of the sediments to occur frequently, preventing the pond from clearing after the removal of large carp. This illustrates that simply removing the perturbation from the system will not necessarily cause a state shift, and that other factors can play a role (Scheffer 1998).

Due to the design of the study and the fact that the density of large carp in the ponds was not determined, I can not definitively conclude that large carp were the cause of the changes observed. I can only link the variation observed, and the changes that occurred, to the differences in accessibility of large carp. However, the density of YOY carp, and the frequency with which large carp were seen in the ponds is known. Assuming that the effects of large carp were proportional to a combination of these two factors the impact of large carp at Delta Marsh can be inferred. After the variation due to location and year were removed, the presence of large carp correlated with changes in water quality, and the abundance of algae and submersed vegetation of Delta Marsh. The abundance of phytoplankton, concentration of suspended solids, concentration of phosphorous, water turbidity and silica concentration increased with the density and frequency of carp, while macrophyte biomass decreased.

In a study conducted concurrently with this one, Parks (2006) states that it was difficult to determine the relative contribution of common carp to the changes that were observed in the invertebrate, amphibian and fish communities in the ponds studied. She suspected the exclusion of adult carp was what led to an increase in the number of amphibians in the Screened ponds in 2002; however, with respect to invertebrates and fish, Parks (2006) deemed it difficult to separate out the effects of carp with those of connectivity. As a result of this, she concluded that her findings "imply that common carp are a symptom of poor habitat conditions within Delta Marsh, but are not a cause." This statement is contrary to my findings, and is likely a result of the lack of habitat data available to Parks (2006), combined with the complexity of the communities she was investigating. It therefore critical that the data from both studies be incorporated together, so that a much clearer and more reliable picture of the relative contribution of common carp to the conditions observed in each of the ponds studied may be reached.

5.2 *Review of the study: the ecosystem-level approach*

Research at the ecosystem-level is important to the understanding of how a natural system will react under anthropogenic or natural perturbation. These types of studies are critical because of the information they provide at various temporal and spatial scales (Schindler 1998). In addition to the ecosystem-level approach, mesocosm studies are also used in ecological research. These types of studies are beneficial because they provide the basic understanding needed to conduct ecosystem-level experiments and allow one to isolate and study individual factors which are impossible to do in a larger ecosystem study (Drenner & Mazumder 1999). However, mesocosms represent a simplification of reality, and can therefore not be used to predict the outcome of a natural system (Schindler 1998). It is therefore important to conduct studies at the ecosystem level to understand how the system will react to change when developing a management plan to improve a system that has been degraded by a perturbation, such as the introduction of common carp.

This study was conducted at the ecosystem-level, and is therefore important to the understanding of the natural impact of carp in Delta Marsh, and is a vital step towards managing the carp problem. It is an example of how spatial and temporal variation occurs within an ecosystem and why it is important to our understanding of the impact of common carp. This study also demonstrates the reasons why mesocosm experiments are often more appealing to researchers due to the greater amount of control they provide.

There was a large difference in water depth between the two years of the study; 2001 was a high water year and 2002 was a low water year (see figure 2-1 in Parks 2006). This difference shows the temporal variation that can occur, and lead to complications within the study. In 2001, because of the high water levels, the areas of upland vegetation separating the Isolated ponds from the main marsh were flooded. This allowed carp and other fish to gain access to these ponds for at least part of the sampling season (Parks 2006). Since the amount of flooding differed for each of the ponds, I could not be certain if large carp accessed all or only some of the Isolated ponds, or to what degree. The presence of YOY carp in the Isolated ponds indicated that large carp were present; however, they may have not actually entered the pond, but merely spawned in flooded upland vegetation. It is also unclear whether the large carp remained in the Isolated ponds after spawning. Therefore, I can not be certain how much, or if any, impact carp had in the Isolated ponds in the baseline year. Due to the uncertainties caused by flooding in 2001, three Isolated ponds (Crescent Pond, Garnham's Pothole, Pull-through Pond) that were originally part of the study were dropped. These ponds were flooded to the greatest degree, and could be considered Connected for most of the sampling season. This situation demonstrates how the researcher must be adaptable with ecosystem level studies, as one does not have the same control that they would with a mesocosm experiment.

Temporal variation in water quality was also demonstrated within the sampling seasons. Carp were found to have a greater impact in the spring, presumably due to their spawning activities. A mesocosm experiment may not have shown this seasonal effect because of the size limitations that are usually associated with this type of study. Mesocosms are comparatively small and typically contain only a few fish (Drenner et al. 1998; Lougheed et al. 1998; Sidorkewicj et al. 1999; Badiou 2005). In such an experiment, carp would be held in within a small area, and not be permitted to come and go as they would naturally. During spawning, carp gather in groups. Most mesocosm experiments would prohibit such behaviour. As well, in order to observe the effects of a spawning event in a mesocosm study, it would be necessary for researchers to ensure the male to female ratio is representative of that in a natural system. No such restraints occurred in this study, and I was able to see how the natural behaviour of the carp lead to seasonal variation in water quality.

The geographic differences between ponds in this study demonstrate the spatial variation that is included in an ecosystem-level experiment. I found that ponds on the west side of the Assiniboine Diversion tended to be more turbid, had higher levels of TP and seemed to be more susceptible to the impact of carp than those on the east side of the Diversion. These differences may have been attributable to differences in the land use of the surrounding areas or sediment characteristics of the ponds. East of the Diversion, a large number of cottages are located along the beach ridge that divides the marsh from Lake Manitoba, whereas there are few residences on the west side. The differences in land use surrounding the ponds on either side of the Diversion may lead to different types and levels of external nutrient loading to the ponds. As well, I noticed that the sediments on the west side of the Diversion were looser than those on the east side; the sediments become more packed and sandier in the east and south parts of the marsh. Softer sediments are more easily suspended than hard packed sediment, and sand settles out of suspension rapidly, while fine textured sediments will stay suspended in the water column for a much longer period of time. The soft, fine-textured sediments observed on the west side of the Diversion likely contributes to the higher turbidity seen there. Had a mesocosm experiment been conducted, unless several mesocosms were deployed in various areas of the marsh, the geographic difference within the marsh would have been unknown.

Spatial variation in water quality and submersed vegetation also occurred between and within ponds. I noticed that carp did not distribute themselves evenly throughout the marsh or the ponds; their distribution was patchy. Just because a pond was accessible to carp did not mean that carp would enter the pond. Throughout the study large groups of carp were frequently seen in Cooks Creek (Figure 5.5), Cram Creek, Thompson's Bay, The Maze, The Long Arm, Cherry Ridge, Blind Channel, First Lead, the entrances between Lake Manitoba

Figure 5.5: Photograph of "boiling" water, resulting from a large group of carp in Cooks Creek in mid-July. The water in the pond was calm until the carp were disturbed by an approaching canoe. In the background, a calm area is visible where the carp were inactive.

and Clandeboye Bay, and 22 Bay. These places were calm, shallow and warm compared to nearby areas in the marsh. Carp were less abundant in the large, deeper, open water areas that occurred on the east side of the Diversion. As well, there tended to be certain "hotspots" within a pond where carp could typically be seen. For example, carp were frequently seen in 2002 in North School Bay; however, they were most often seen on the east side of the pond near the channel into Cadham Bay, or on the far south end of the pond. These two areas were the shallowest areas. The situation described here differs from the situation that occurs in smaller scale mesocosm studies where carp are penned within a small area, and are therefore constantly present within the study site. In such experiments carp are not given a "choice" of where they are, their natural behaviours are limited by the conditions of the experiment. I believe that conditions within mesocosms studies are unrealistic and exaggerate the impact that carp have in a natural system. For example, Badiou (2005) observed that as the size of the experimental system decreased, there was an increase in the amount of resuspension by carp, indicating that the results of mesocosm studies with carp are compounded by their small size. In ecosystem level studies, the spatial variation that is included is important to determining the true response of the system to a perturbation.

All of the above mentioned situations demonstrate why ecosystem-level studies should be undertaken. They demonstrate that the "real-world" has variation. This study shows that not every part of a system will react to a stressor in the same way, and reaffirms that ecosystem-level experiments are necessary before realistic management programs can be developed.

5.3 Recommendations

5.3.1 Recommendations for Research

1) Continue long-term monitoring

Monitoring of the ponds should continue for a minimum of two additional years. More would be ideal. By collecting data for a longer period of time a couple of questions could be answered. There was a large degree of variation between 2001 and 2002. Continuing to monitor the ponds for additional sampling seasons may make any differences between treatments clearer by minimizing the effects of interannual variation. This will also allow us to see if the ponds that were made inaccessible to carp will continue to improve, and remain in their current clear state. Other studies indicate that the switch may be temporary (Lougheed & Chow-Fraser 2001). If wind action is strong enough, sediment resuspension may prohibit the lower levels of turbidity required to maintain the macrophyte dominated state. This situation is most likely to occur in open areas where there is a lack of emergent vegetation (Lougheed et al. 2004). The longer monitoring continues, the more information we will have, and the better we will be able to understand and predict the effects that carp have on Delta Marsh. However, a balance will need to be made between financial feasibility and research goals. Nevertheless, if a management plan to control the impact of carp on the marsh is developed, it will be necessary to include some type of monitoring program to determine if it is effective, and so that adaptations can be made if required.

2) Improve methods of evaluating water quality, particularly nutrient conditions

The methods I used resulted in uncertain findings about the nutrient dynamics in the ponds studied. Biweekly and monthly sampling periods were not frequent enough to monitor changes, and resulted in large variations between sampling points. There was some indication of differences between large carp accessible and inaccessible ponds, however they were not clear. This was particularly true for Si and N. I recommend that the sampling frequency be increased to weekly, triplicate samples be collected each time (ponds with carp were observed to have patchy conditions), and TN and TP be determined for each sample instead of monthly. As mentioned previously, the use of nutrient diffusing substrata may be helpful in determining differences in nutrient limitation between sites with and without large carp.

3) Improve treatments

If this study is continued, or a similar one was to be conducted, I recommend that there be changes to the treatments. This includes improvement to the Blasted treatment, as well as the inclusion of additional. The Blasted treatment was very effective in South MacKenzie Bay; however, this was not true for Wye's Pond. We were unable to create the channel into Wye's Pond until mid-June of 2002 when the ground thawed. Consequently, carp did not have access to this pond during their peak spawning period. I believe that this resulted in carp having very little impact in Wye's Pond, and was a strong contributor to the variation between the two ponds in the Blasted treatment. I recommend that if this treatment is to be used again, the channel into the pond should be created late in the fall of the previous year. This would ensure that carp had access to the pond at the same time as they would to other areas of the marsh, minimizing the variation within carp accessible ponds.

Another treatment that could be added later in the study would be to Reopen a previously Diked pond. This treatment would have two levels: Reopen completely (i.e. similar to the Blasted treatment), Reopen-Screened. The former would allow researchers to determine if an improved site would be able to maintain its clear state if carp were reintroduced, and the latter would show if the exchange of turbid water with the main marsh would be enough on its own to cause a state shift in an improved site.

4) Integrate ecosystem-level results

The results presented here should be combined with those of Parks (2006) to determine the effect of carp on the whole-ecosystem. In a companion study, Parks (2006) collected information on the fish, invertebrate, and amphibian communities. Her data was gathered in the same ponds, during the same time period as the data in this study. Combining our results would paint a clearer picture of what was going on in the ponds, by allowing us to determine if any top-down effects were occurring. Such effects may account for unexplained differences that were observed in the water quality, algae, and submersed vegetation of the ponds. For example, the abundance of periphyton and epiphyton was found to be independent of the accessibility of carp. However, by combining data with Parks (2006), it may be found that more grazers were present in some ponds, and that this may have reduced the observed abundance of attached algae. Also, combining data from both studies may help to explain why not all Connected sites were impacted by carp to the same degree. In 2001, North School Bay contained an abundance of northern pike (*Esox lucius*) (Candace Parks, *pers. com.*). Pike are piscivores; they are predators of small carp as well as planktivorous fish. They are known to cause cascading effects on a system by reducing the population of benthivores and releasing the grazing pressure on large zooplankton allowing them to control phytoplankton biomass (Grimm & Backx 1990). I believe the large population of pike is likely to have contributed to the low turbidity and the presence of submersed vegetation in North School Bay in 2001. This collaboration will also help to determine if other benthivorous fish, such as black bullheads (*Ameiurus* sp.), are making an impact, or if the effects seen can be attributed solely to common carp. This study was initially developed to understand the impact of carp on the whole-ecosystem of Delta Marsh, combining data from both components of the study is an important next step.

5.3.2 Recommendations for Management

1) Create and maintain barriers to prevent large carp from accessing the marsh, particularly areas that are most susceptible to their impact.

This study has shown that preventing large carp from entering areas of Delta Marsh results in improved water quality and the recovery of submersed vegetation. Therefore it follows that preventing carp from accessing the marsh could be a useful management strategy for Delta Marsh. Barriers to prevent carp from entering the marsh have been used in the past and were successful; however, they also prevented other fish including some economically valuable species from entering the marsh (Wrubleski 1998).

A solution to the problem of such a barrier impeding all fish was addressed in the restoration of Cootes Paradise, a coastal wetland of Lake Erie (Lougheed et al. 2004). In an attempt to prevent large carp from entering this wetland, a fishway was installed. The fishway was constructed such that small fish were able to pass through, but larger fish were collected, and then hand sorted to exclude carp while allowing in the desirable species. The results of this restoration have been favourable; turbidity has decreased and submersed vegetation has recovered (Lougheed & Chow-Fraser 2001). A similar method could be employed at Delta Marsh; however, since there are multiple channels connecting the marsh to the lake, this could be an enormous undertaking. Erecting simple screens at all but two of the channels (Cram Creek and Delta Channel) at which fishways could be installed, would decrease the amount of intensive sorting that would be required. Still this approach may require too much manpower and be too costly to implement.

An alternative approach would be to install barriers, similar in dimension to the screens used in this study, at strategic locations throughout the marsh. The new barriers would still permit all fish, including carp to enter the marsh; however, they would prevent larger fish from accessing key areas. The results of this study indicate that shallow, soft-bottomed areas, particularly on the west-side of the Assiniboine Diversion are the most susceptible (Table 5.3). Preventing

carp from entering these areas would allow for the recovery of submersed vegetation, and thereby provide important refugia for zooplankton and small fish. These refugia are important to the restoration of shallow systems (Perrow et al. 1997).

2) Further develop the carp fishery on Lake Manitoba and Delta Marsh, and promote the use of carp by industry and the public.

Further developing the carp fishery on Lake Manitoba and Delta Marsh may be critical to reducing and controlling the biomass of carp throughout Delta Marsh and the other coastal wetlands of Lake Manitoba. Young carp quickly become too large to be handled by predatory fish and birds, so human removal of the species may be the only way of reducing the number of carp (Sigler 1958). A fishery aimed at intentionally selecting carp will remove them from the system without detrimental effects to non-target fish species. Such fisheries in some

Table 5.3: Characteristics that determine the susceptibility of areas of Delta Marsh to the impact of carp, based on the results of the study as well as personal observations.

High Risk	**Low Risk**
Soft sediment with high percent organic matter	Hard, packed mineral sediment
Fine textured sediment: clay, silt	Course textured sediment: gravel, sand
Open water areas lacking emergents , large fetch	Areas with interspersed emergent vegetation, small fetch
Shallow water; < 50 cm	Areas where the water is unlikely to drop below 50 cm.
Examples: Cooks Creek, Cram Creek, The Maze, The Long Arm, Cherry Ridge, Blind Channel, First Lead, the entrance between Lake Manitoba and Clandeboye Bay, 22 Bay.	Examples: Clandeboye Bay, The Gap, Waterhen Bay

European lakes have been successful in controlling the population of carp (Arlinghaus & Mehner 2003).

A small carp fishery currently exists on Lake Manitoba. From 1992-2002, the carp fishery accounted for just over four percent of the total commercial fishing production in Manitoba with thirteen percent of the catch coming from Lake Manitoba (Manitoba Conservation Fisheries Branch 2003). In 2001, 206,200 kg of carp were harvested from Lake Manitoba, which had an approximate value of $94,000 (calculated from (Manitoba Conservation Fisheries Branch 2003)). Given the large biomass of carp thought to be present in Delta Marsh, they represent a valuable resource that has not been exploited to its full potential.

Although the resource is present, such a fishery will not be feasible unless the market for carp can be improved. Carp is no longer a fish common on the Canadian table; therefore, marketers will need to reintroduce the fish to the public through advertising and the development of innovative uses of carp products. Recently, one such innovation has been initiated by the Freshwater Fish Marketing Corporation (FFMC) who has begun processing carp roe for sale as an alternative to traditional caviar. Another novel idea, which has been developed and marketed by a company in Australia (Charlie Carp Ltd.), is to make carp into liquid fertiliser. Carp has also been used as a substitute for leather in decorative items, made into fish meal and added to human and pet foods as a protein source. If a carp fishery on Lake Manitoba is to be viable, new ideas will need to be explored.

3) Develop management strategies to reduce the impact of other factors contributing to the degradation of the marsh: loss of emergent vegetation and external nutrient loading.

It is unknown whether or not the clear-state that has been achieved with the exclusion of carp from ponds in this study will be maintained. It is therefore necessary to consider what may contribute to this possible failure, as lack of recovery following the removal of carp has occurred in other coastal wetlands in Canada (Lougheed et al. 2004). At Delta Marsh, the loss of emergent vegetation and external nutrient loading are other factors contributing to the poor health of the marsh, and should be addressed along with the presence of carp.

Emergent vegetation is an important factor in reducing the resuspension of sediments in wetlands (Dieter 1990). Section 5 Bay in this study, as well as shallow open-water areas in Cootes Paradise (Lougheed et al. 2004), failed to recover after carp exclusion as well as areas with emergent vegetation. Therefore, re-establishing islands of emergent vegetation throughout the open-water areas of the marsh will be important to reducing turbidity. However, in order for this to occur, the conditions in the marsh must allow for the regeneration of emergents. Emergent vegetation requires periodic drawdown conditions for new stands to develop (van der Valk & Davis 1980). Without these drawdown conditions the emergent vegetation can be flooded out. This is the current condition in much of Delta Marsh. Water levels within the marsh have been relatively stable since the construction of the Fairford Dam on Lake Manitoba.

The operation of the dam should be adjusted to allow the water level of lake, and therefore Delta Marsh to fluctuate to a degree that would permit for periodic drawdown conditions. This would allow stands of emergent vegetation to be re-established throughout the marsh and thereby help reduce sediment resuspension.

High nutrient levels may contribute to returning the marsh to the turbid state (Perrow et al. 1997; Janse 1997). Therefore, steps should be taken to decrease the external nutrient loading to the marsh (Scheffer 1998; Hansson et al. 1998). Residential development and local farming practices have been suggested to contribute to the nutrient load of Delta Marsh (Brown 2003). Brown (2003) has recommended that residents in the area stop using septic fields and outhouses in favour of holding tanks, minimize their use of fertilizer, and cleanup after their pets. Farmers should conduct regular soil testing to determine nutrient levels to ensure that they fertilize only when necessary, use injection methods of fertilizing to prevent runoff, leave a vegetation buffer zone around water bodies, and prevent livestock from accessing surface water. Instituting regulations that ensure these recommendations are followed will be important to reducing the external nutrient load on the marsh.

By addressing all factors that are thought to contribute to the current turbid condition of Delta Marsh, we will more likely have a favourable outcome. Therefore, any management program intended to reduce the impact of carp on the marsh, will necessarily include the management of emergent vegetation and nutrient loading.

Summary

Common carp (*Cyprinus carpio* L.) are contributing to the habitat degradation of Delta Marsh. I found that there was a great degree of geographic and interannual variation amongst the 10 ponds studied, but the presence of carp at selected sites was correlated with changes in water quality, and the abundance of algae and submersed vegetation. Generally, ponds that were accessible to large carp existed in a turbid, phytoplankton-dominated state, while ponds that were inaccessible to large carp existed in a clear, macrophyte-dominated state.

The buffering effect of submersed macrophytes may prevent a pond from becoming as turbid as it would otherwise in the presence of carp. Depth was also a factor influencing turbidity. Wind resuspension of sediment occurred in shallow sites, leading to increases in ISS and phytoplankton with the release of nutrients from the sediments. Carp prefer shallow ponds, so these sites are likely to be the most impacted with the combined effects of carp and wind resuspension.

Redundancy analysis showed that the phytoplankton, suspended solids, P, water turbidity and silica concentration increased with the density of YOY carp and the frequency of occurrence of large carp, while macrophyte abundance decreased. The impact of carp was the greatest during spring spawning, as this was when the greatest difference in turbidity and phytoplankton occurred between carp accessible and carp inaccessible ponds. When large carp were prevented from accessing a pond, a shift from a turbid to a clear state occurred. The reverse was true when carp were given to access a pond.

I recommend that the research started here be continued and expanded to determine if the observed changes will continue, and if they are stable, as well as to gain a better understanding of the nutrient dynamics. I recommend a management plan should be developed for Delta Marsh that includes, the use of carp barriers and the development of a carp fishery. I suggest that the successful restoration of Delta Marsh will take more than reducing carp abundance; the loss of emergent and submersed vegetation as well as reduction of external nutrient loading will need to be addressed.

References

American Public Health Association (APHA) 1998. Standard methods for the examination of water and waste water. Washington, D.C.

Angeler, D.G., Álvarez-Cobelas, M., Sánchez-Carrillo, S., and Rodigo, M.A. 2002. Assessment of exotic fish impacts on water quality in a degraded semi-arid floodplain wetland. Aquatic Sciences **64:** 76-86.

Arlinghaus, R. and Mehner, T. 2003. Socio-economic characterisation of specialised common carp (*Cyprinus carpio* L.) anglers in Germany, and implications for inland fisheries management and eutrophication control. Fisheries Research **61:** 19-33.

Atton, F.M. 1959. The invasion of Manitoba and Saskatchewan by carp. Transcripts of the American Fisheries Society **88:** 203-205.

Badiou, P.H.J. 2005. Ecological impacts of an exotic benthivorous fish in wetlands: a comparison between common carp (*Cyprinus carpio* L.) additions in large experimental wetlands and small mesocosms in Delta Marsh, Manitoba. Ph.D. Thesis. University of Manitoba. Winnipeg, Manitoba.

Barton, D.R., Kelton, N., and Eedy, R.I. 2000. The effects of carp (*Cyprinus carpio* L.) on sediment export from a small urban impoundment. Journal of Aquatic Ecosystem Stress and Recovery **8:** 155-159.

Beamish, R.J. 1973. Design of a trapnet with interchangeable parts for the capture of large and small fishes from varying depths. Journal of the Fisheries Research Board of Canada **30:** 587-590.

Bloesch, J., Bossard, P., Buhrer, H., Burgi, R., and Uehlinger, U. 1988. Can results from limnocorral experiments be transferred to in situ conditions? Hydrobiologia **159:** 297-308.

Bonneau, J.L. and Scarnecchia, D.L. 2002. Spawning-season homing of common carp and river carpsucker. Prairie Naturalist **34:** 13-20.

Bourne, A. 2000. Factors influencing the abundance of sediment associated algae in two isolated ponds and a turbid channel of Delta Marsh, Manitoba. M.Sc. Thesis. University of Manitoba. Winnipeg, Manitoba.

Breukelaar, A.W., Lammens, E.H.R.R., Klein Breteller, J.G.P., and Tatrai, I. 1994. Effects of benthivorous bream (*Abramis brama*) and carp (*Cyrinus carpio*) on sediment resuspension and concentrations of nutrient and chlorophyll *a*. Freshwater Biology **32:** 113-121.

Brown, S.D. 2003. Land use practices in the vicinity of Delta Marsh as they may be affecting water quality. Masters thesis. University of Calgary. Calgary, Alberta

Cahn, A.R. 1929. The effect of carp on a small lake: the carp as a dominant. Ecology **10:** 271-274.

Carpenter, S.R. 1996. Microcosm experiments have limited relevance for community and ecosystem ecology. Ecology **77:** 677-680.

Chow-Fraser, P. 1998. A conceptual ecological model to aid restoration of Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario, Canada. Wetlands Ecology and Management **6:** 46-57.

Crivelli, A.J. 1983. The destruction of aquatic vegetation by carp. Hydrobiologia **106:** 37-41.

Dieter, C.D. 1990. The importance of emergent vegetation in reducing sediment resuspension in wetlands. Journal of Freshwater ecology **5:** 467-473.

Drenner, R.W., Gallo, K.L., Baca, R.M., and Smith, D.J. 1998. Synergistic effects of nutrient loading and omnivorous fish on phytoplankton biomass. Canadian Journal of Fisheries and Aquatic Sciences **55:** 2087-2096.

Drenner, R.W. and Mazumder, A. 1999. Microcosm experiments have limited relevance for community and ecosystem ecology: comment. Ecology **80:** 1081-1085.

Fritz, A.W. 1987. Commercial fishing for carp. *In* Cooper, E.L., ed. Carp in North America. American Fisheries Society, Bethesda, Maryland, USA, pp 17-30.

Garcia-Berthou, E. 2001. Size- and depth-dependent variation in habitat and diet of the common carp (*Cyprinus carpio*). Aquatic Sciences **63:** 466-476.

Garcia, P.A. 2001. Competition with carp may limit white-headed duck populations in Spain. TWSG News **13:** feature 8.

Gibeau, G.G.Jr. and Miller, M.C. 1989. A micro-bioassay for epilithon using nutrient-diffusing artificial substrata. Journal of Freshwater Ecology **5:** 171-176.

Goldsborough, L.G. 2001. Sampling algae in wetlands. *In* Biomonitoring and Management of North American Freshwater Wetlands. *Eds:* Rader, R.B., Batzer, D.P. and Wissinger, S. Academic Press. pp 263-295.

Grimm, M.P. and Backx, J.J.G.M. 1990. The restoration of shallow eutrophic lakes, and the role of northern pike, aquatic vegetation and nutrient concentration. Hydrobiologia **200/201:** 557-566.

Hansson, L.-A., Annadotter, H., Bergman, E., Hamrin, S.F., Jeppesen, E., Kairesalo, T., Luokkanen, E., Nilsson, P.-A., Søndergaard, M., and Strand, J. 1998. Biomanipulation as an application of food-chain theory: constraints, synthesis and recommendations for temperate lakes. Ecosystems **1:** 558-574.

Hinks, D.C.C. 1943. The fishes of Manitoba. Province of Manitoba, Department of mines and natural resources. Winnipeg, Manitoba.

Janse, J.H. 1997. A model of nutrient dynamics in shallow lakes in relation to multiple stable states. Hydrobiologia **342/343:** 1-8.

Khan, T.A. 2003. Dietary studies on exotic carp (*Cyprinus carpio* L.) from two lakes of western Victoria, Australia. Aquatic Sciences **65:** 272-286.

Khan, T.A., Wilson, M.E., and Khan, M.T. 2003. Evidence for invasive carp mediated trophic cascade in shallow lakes of western Victoria, Australia. Hydrobiologia **506-509:** 465-472.

King, A.J., Robertson, A.I., and Healy, M.R. 1997. Experimental manipulations of the biomass of introduced carp (*Cyprinus carpio*) in billabongs. I. impacts on water-column properties. Marine and Freshwater Research **48:** 435-443.

King, D.R. and Hunt, G.S. 1967. Effect of carp on vegetation in a Lake Erie marsh. Journal of Wildlife Management **31:** 181-188.

Kolterman, B.F. 1990. Effects of common carp and black bullheads on sago pondweed. M.Sc. Thesis. South Dakota State University. Brookings, South Dakota.

Lougheed, V.L. and Chow-Fraser, P. 2001. Spatial variability in the response of lower trophic levels after carp exclusion from a freshwater marsh. Journal of Aquatic Ecosystem Stress and Recovery **9:** 21-34.

Lougheed, V.L., Crosbie, B., and Chow-Fraser, P. 1998. Predictions on the effect of common carp (*Cyprinus carpio*) exclusion on water quality, zooplankton, and submergent macrophytes in a Great Lakes wetland. Canadian Journal of Fisheries and Aquatic Sciences **55:** 1189-1197.

Lougheed, V.L., Crosbie, B., and Chow-Fraser, P. 2001. Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes basin: latitude, land use, and water quality effects. Journal of Fisheries and Aquatic Sciences **58**: 1603-1612.

Lougheed, V.L., Theysmeyer, T., Smith, T., and Chow-Fraser, P. 2004. Carp exclusion, food-web interactions, and the restoration of Cootes Paradise Marsh. Journal of Great Lakes Research **30:** 44-57.

Lundholm, J.T. and Simser, L.W. 1999. Regeneration of submersed macrophyte populations in a disturbed Lake Ontario coastal marsh. Journal of Great Lakes Research **25:** 395-400.

Manitoba Conservation Fisheries Branch. 2003. A profile of Manitoba's commercial fishery. Manitoba Conservation. Winnipeg, Manitoba.

McCrimmon, H.R. 1968. Carp in Canada. Bulletin of the Fisheries Research Board of Canada **165**.

Mitsch, W.J. and Gosselink, J.G. 2000. Wetlands. John Wiley and Sons, Inc., United States of America.

Panek, F.M. 1987. Biology and ecology of carp. *In* Cooper, E.L., ed. Carp in North America. American Fisheries Society, Bethesda, Maryland, USA, pp 1-16.

Parkos III, J.J., Santucci, V.J.Jr., and Wahl, D.H. 2003. Effects of adult common carp (*Cyprinus carpio*) on multiple trophic levels in shallow mesocosms. Canadian Journal of Fisheries and Aquatic Sciences **60:** 182-192.

Parks, C.R. 2006. Experimental manipulation of connectivity and common carp; the effects on native fish, water-column invertebrates, and amphibians in Delta Marsh, Manitoba. M. Sc. Thesis. University of Manitoba. Winnipeg, Manitoba.

Perrow, M.R., Meijer, M.-L., Dawidowicz, P., and Coops, H. 1997. Biomanipulation in shallow lakes: state of the art. Hydrobiologia **342/343:** 355-365.

Richardson, W.B., Wickham, S.A., and Threlkeld, S.T. 1990. Foodweb response to the experimental manipulation of a benthivore (*Cyprinus carpio*), zooplanktivore (*Menidia beryllina*) and benthic insects. Archiv für Hydrobiologie **119:** 143-165.

Robel, R.J. 1962. The relationship of carp to waterfowl food plants on a western marsh. Ph.D. Thesis. Utah State University. Logan, Utah.

Roberts, J., Chick, A., Oswald, L., and Thompson, P. 1995. Effect of carp, *Cyprinus carpio* L., an exotic benthivorous fish, on aquatic plants and water quality in experimental ponds. Marine and Freshwater Research **46:** 1171-80.

Robertson, A.I., Healey, M.R., and King, A.J. 1997. Experimental manipulations of the biomass of introduced carp (*Cyprinus carpio*) in billabongs. II. Impacts on benthic properties and processes. Marine and Freshwater Research **48:** 435-443.

Scheffer, M. 1990. Multiplicity of stable states in freshwater systems. Hydrobiologia **200/201:** 475-486.

Scheffer, M. 1998. Ecology of Shallow Lakes. Chapman & Hall, London, England.

Schindler, D.W. 1998. Replication versus realism: the need for ecosystem-scale experiments. Ecosystems **1:** 323-334.

Shay, J.M., de Gues, P.M.J., and Kapinga, M.R.M. 1999. Changes in shoreline vegetation over a 50-year period in the Delta Marsh, Manitoba in response to water levels. Wetlands **19**(2): 413-425.

Sidorkewicj, N.S., Lopez Cazorla, A.C., Fernandez, O.A., Mockel, G.C., and Burgos, M.A. 1999. Effects of *Cyprinus carpio* on *Potamogeton pectinatus* in experimental culture: the incidence of the periphyton. Hydrobiologia **415:** 13-19.

Sigler, W.F. 1958. The ecology and use of carp in Utah. Utah State University, Logan Agricultural Experiment Station, Bulletin 405. Logan, Utah.

Stainton, M.P., Capel, M.J., and Armstrong, F.A. 1977. The chemical analysis of freshwater. 2nd Edition. Fisheries and Environment Canada. Miscellaneous Special Publication No. 25. Ottawa, Ontario.

Stewart, K.W., Suthers, I.M., and Leavesley, K. 1985. New fish distribution records in Manitoba and the role of a man-made interconnection between two drainages as an avenue of dispersal. The Canadian Field-Naturalist **99:** 317-326.

Swee, U.B. and McCrimmon, H.R. 1966. Reproductive biology of the carp, *Cyprinus carpio* L., in Lake St. Lawrence, Ontario. Transactions of the American Fisheries Society **95:** 372-387.

van der Valk, A.G. 1981. Succession in wetlands: A Gleasonian approach. Ecology **62:** 688-698.

van der Valk, A.G. and Davis, C.B. 1980. The impact of a natural drawdown on the growth of four emergent species in a prairie glacial marsh. Aquatic Botany **9:** 301-322.

Warner, B.G. and Rubec, C.D.A. 1997. The Canadian Wetland Classification System. 2nd Edition. National Wetlands Working Group, University of Waterloo. Waterloo, Ontario.

Waters, I. and Shay, J.M. 1992. Effect of water depth on population parameters of a *Typha glauca* stand. Canadian Journal of Botany **70:** 349-351.

Wetzel, R.G. 1983. Limnology. Saunders College Publishing, United States of America.

Winkel, E.H. and Meulemans, J.T. 1985. Effects of fish upon submerged vegetation. Hydrobiological bulletin **18:** 157-158.

Wrubleski, D.A. 1998. The fish community of Delta Marsh: a review. Institute for Wetland and Waterfowl Research, Ducks Unlimited Canada.

Zambrano, L. and Hinojosa, D. 1999. Direct and indirect effects of carp (*Cyprinus carpio* L.) on macrophyte and benthic communities in experimental shallow ponds in central Mexico. Hydrobiologia **408/409:** 131-138.

Zambrano, L., Perrow, M.R., Macias-Garcia, C., and Aquirre-Hidal, V. 1999. Impact of introduced carp (*Cyprinus carpio*) in subtropical shallow ponds in Central Mexico. Journal of Aquatic Ecosystem Stress and Recovery **6:** 281-288.

Appendix 1: Correlation matrices of the of the major parameters measured in 2002. Included are matrices for the seasonal averages (A), monthly averages (B), and the biweekly water column samples (C).

A1) Seasonal averages – water quality

Variable	Depth	NTU	pH	Cond	Alk	Chla	OSS	ISS	NH3	TRP	DOC	TN	IN	ON	TP	Si
Depth	1.000	-0.777	0.448	0.366	0.153	-0.555	-0.733	-0.755	0.212	-0.581	0.256	-0.108	0.395	-0.130	-0.892	-0.491
NTU	-0.777	1.000	-0.318	-0.325	-0.158	0.653	0.795	0.992	0.045	0.243	-0.495	0.007	-0.287	0.023	0.654	0.140
pH	0.448	-0.318	1.000	-0.170	-0.445	-0.461	-0.507	-0.346	-0.340	-0.212	-0.145	-0.450	-0.313	-0.435	-0.431	-0.565
Cond	0.366	-0.325	-0.170	1.000	0.854	0.116	0.006	-0.358	0.573	-0.700	0.501	0.613	0.692	0.579	-0.404	0.302
Alk	0.153	-0.158	-0.445	0.854	1.000	0.419	0.295	-0.188	0.656	-0.405	0.506	0.673	0.680	0.639	-0.073	0.543
Chla	-0.555	0.653	-0.461	0.116	0.419	1.000	0.966	0.605	0.189	-0.036	0.108	0.633	-0.158	0.645	0.609	0.652
OSS	-0.733	0.795	-0.507	0.006	0.295	0.966	1.000	0.752	0.126	0.103	-0.038	0.512	-0.211	0.527	0.726	0.624
ISS	-0.755	0.992	-0.346	-0.358	-0.188	0.605	0.752	1.000	0.099	0.264	-0.486	-0.025	-0.248	-0.011	0.627	0.089
NH3	0.212	0.045	-0.340	0.573	0.656	0.189	0.126	0.099	1.000	-0.407	0.397	0.317	0.745	0.277	-0.272	-0.012
TRP	-0.581	0.243	-0.212	-0.700	-0.405	-0.036	0.103	0.264	-0.407	1.000	-0.386	-0.448	-0.257	-0.436	0.724	0.212
DOC	0.256	-0.495	-0.145	0.501	0.506	0.108	-0.038	-0.486	0.397	-0.386	1.000	0.683	0.193	0.676	-0.269	0.354
TN	-0.108	0.007	-0.450	0.613	0.673	0.633	0.512	-0.025	0.317	-0.448	0.683	1.000	0.140	0.999	0.101	0.702
IN	0.395	-0.287	-0.313	0.692	0.680	-0.158	-0.211	-0.248	0.745	-0.257	0.193	0.140	1.000	0.085	-0.314	0.077
ON	-0.130	0.023	-0.435	0.579	0.639	0.645	0.527	-0.011	0.277	-0.436	0.676	0.999	0.085	1.000	0.119	0.703
TP	-0.892	0.654	-0.431	-0.404	-0.073	0.609	0.726	0.627	-0.272	0.724	-0.269	0.101	-0.314	0.119	1.000	0.651
Si	-0.491	0.140	-0.565	0.302	0.543	0.652	0.624	0.089	-0.012	0.212	0.354	0.702	0.077	0.703	0.651	1.000
SedChl	-0.162	0.194	-0.499	-0.187	0.074	0.382	0.365	0.163	-0.241	0.139	-0.175	0.082	-0.236	0.096	0.255	0.237
%OM	0.411	-0.361	-0.153	0.857	0.773	0.159	0.008	-0.357	0.671	-0.605	0.690	0.656	0.697	0.621	-0.337	0.352
%Carb	0.271	-0.338	0.233	0.067	-0.226	-0.185	-0.259	-0.330	-0.191	-0.503	0.496	0.362	-0.388	0.386	-0.470	-0.180
SedTN	0.434	-0.505	-0.039	0.562	0.435	0.042	-0.123	-0.497	0.379	-0.665	0.925	0.683	0.154	0.678	-0.515	0.140
SedTP	-0.551	0.537	-0.392	-0.734	-0.451	0.228	0.332	0.608	-0.083	0.696	-0.283	-0.216	-0.268	-0.202	0.624	0.076
Macrophytes	0.378	-0.462	0.792	-0.247	-0.462	-0.589	-0.603	-0.460	-0.334	0.108	-0.007	-0.524	-0.149	-0.519	-0.258	-0.363
EpiChla	0.372	-0.425	-0.459	0.205	0.512	0.082	-0.068	-0.393	0.363	0.023	0.437	0.234	0.426	0.212	-0.121	0.251
PeriChla	0.567	-0.483	0.281	-0.174	-0.329	-0.365	-0.484	-0.425	-0.028	-0.344	0.444	0.013	-0.232	0.026	-0.598	-0.450
PondSize	-0.179	0.480	0.200	-0.358	-0.334	-0.025	0.083	0.491	0.116	0.005	-0.438	-0.486	-0.280	-0.473	-0.088	-0.636
YOY density	-0.871	0.771	-0.689	-0.122	0.050	0.680	0.810	0.759	-0.017	0.247	-0.101	0.414	-0.265	0.432	0.703	0.521
Carp freq.	-0.426	0.501	-0.735	-0.128	0.235	0.670	0.661	0.525	0.287	0.096	0.134	0.433	-0.125	0.442	0.379	0.310

A2) Seasonal averages – sediments, algae and submersed vegetation

Variable	SedChl	%OM	%Carb	SedTN	SedTP	Macrophytes	EpiChla	PeriChla
Depth	-0.162	0.411	0.271	0.434	-0.551	0.378	0.372	0.567
NTU	0.194	-0.361	-0.338	-0.505	0.537	-0.462	-0.425	-0.483
pH	-0.499	-0.153	0.233	-0.039	-0.392	0.792	-0.459	0.281
Cond	-0.187	0.857	0.067	0.562	-0.734	-0.247	0.205	-0.174
Alk	0.074	0.773	-0.226	0.435	-0.451	-0.462	0.512	-0.329
Chla	0.382	0.159	-0.185	0.042	0.228	-0.589	0.082	-0.365
OSS	0.365	0.008	-0.259	-0.123	0.332	-0.603	-0.068	-0.484
ISS	0.163	-0.357	-0.330	-0.497	0.608	-0.460	-0.393	-0.425
NH3	-0.241	0.671	-0.191	0.379	-0.083	-0.334	0.363	-0.028
TRP	0.139	-0.605	-0.503	-0.665	0.696	0.108	0.023	-0.344
DOC	-0.175	0.690	0.496	0.925	-0.283	-0.007	0.437	0.444
TN	0.082	0.656	0.362	0.683	-0.216	-0.524	0.234	0.013
IN	-0.236	0.697	-0.388	0.154	-0.268	-0.149	0.426	-0.232
ON	0.096	0.621	0.386	0.678	-0.202	-0.519	0.212	0.026
TP	0.255	-0.337	-0.470	-0.515	0.624	-0.258	-0.121	-0.598
Si	0.237	0.352	-0.180	0.140	0.076	-0.363	0.251	-0.450
SedChl	1.000	-0.300	-0.169	-0.142	0.195	-0.466	0.477	-0.034
%OM	-0.300	1.000	0.106	0.688	-0.447	-0.051	0.360	0.075
%Carb	-0.169	0.106	1.000	0.697	-0.238	0.055	-0.184	0.744
SedTN	-0.142	0.688	0.697	1.000	-0.420	-0.051	0.333	0.604
SedTP	0.195	-0.447	-0.238	-0.420	1.000	-0.150	0.062	0.025
Macrophytes	-0.466	-0.051	0.055	-0.051	-0.150	1.000	-0.212	0.296
EpiChla	0.477	0.360	-0.184	0.333	0.062	-0.212	1.000	0.237
PeriChla	-0.034	0.075	0.744	0.604	0.025	0.296	0.237	1.000
PondSize	-0.002	-0.549	-0.116	-0.347	0.113	-0.136	-0.372	-0.066
YOY density	0.324	-0.199	-0.042	-0.157	0.469	-0.714	-0.224	-0.402
Carp freq.	0.568	-0.092	-0.008	0.107	0.500	-0.822	0.386	0.011

B1) Monthly averages – water quality

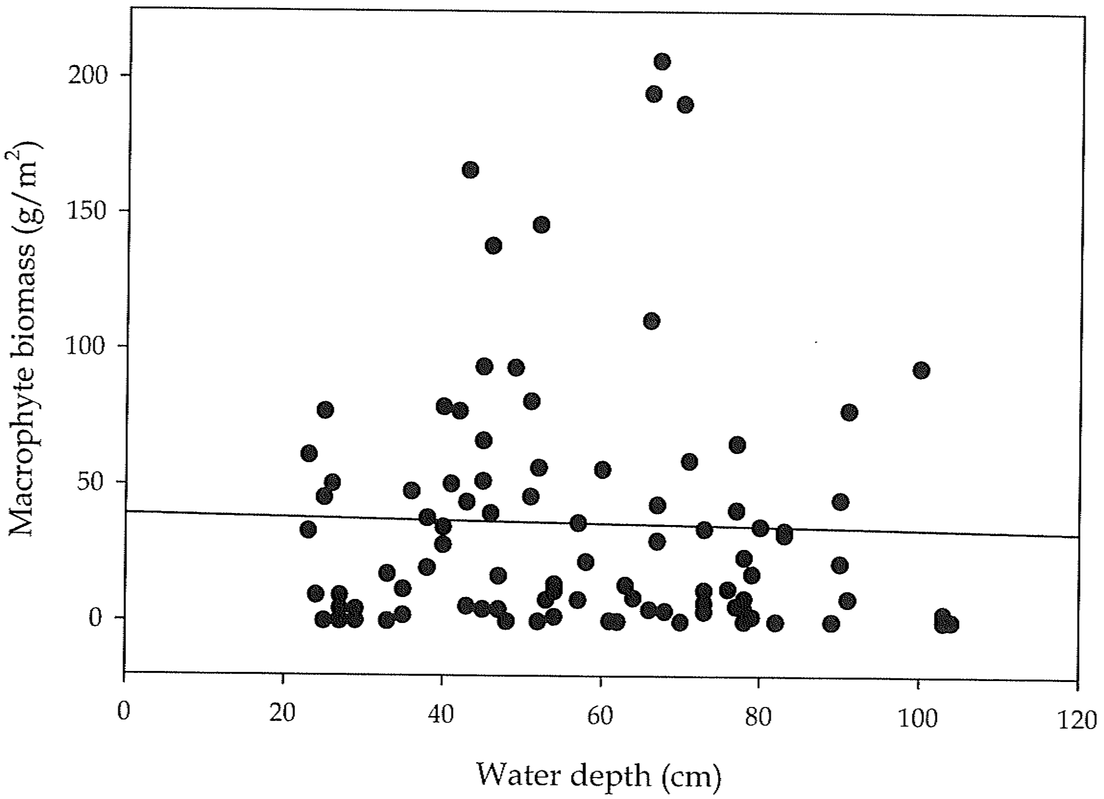
Variable	NTU	pH	Cond.	Alkalinity	Chl a	OSS	ISS	TN	NO3	NH3	TP	TRP	DOC	Si
NTU	1.000	-0.083	-0.286	-0.219	0.658	0.857	0.904	0.171	-0.261	0.179	0.709	0.110	-0.459	0.033
pH	-0.083	1.000	-0.137	-0.192	-0.158	-0.196	-0.211	-0.080	-0.054	-0.168	-0.219	-0.014	0.274	-0.074
Conductivity	-0.286	-0.137	1.000	0.747	-0.076	-0.118	-0.388	0.497	0.609	-0.211	-0.373	0.045	0.357	0.078
Alkalinity	-0.219	-0.192	0.747	1.000	0.108	0.056	-0.207	0.405	0.461	-0.215	-0.073	0.031	0.130	0.169
Chl a	0.658	-0.158	-0.076	0.108	1.000	0.841	0.618	0.258	-0.079	-0.168	0.535	-0.193	-0.444	0.106
OSS	0.857	-0.196	-0.118	0.056	0.841	1.000	0.823	0.298	-0.146	0.044	0.672	-0.011	-0.529	0.027
ISS	0.904	-0.211	-0.388	-0.207	0.618	0.823	1.000	0.135	-0.272	0.178	0.790	0.163	-0.575	0.151
TN	0.171	-0.080	0.497	0.405	0.258	0.298	0.135	1.000	0.247	-0.113	0.241	-0.053	0.131	0.151
NO3	-0.261	-0.054	0.609	0.461	-0.079	-0.146	-0.272	0.247	1.000	-0.079	-0.231	0.179	0.249	0.114
NH3	0.179	-0.168	-0.211	-0.215	-0.168	0.044	0.178	-0.113	-0.079	1.000	0.229	0.411	-0.182	-0.001
TP	0.709	-0.219	-0.373	-0.073	0.535	0.672	0.790	0.241	-0.231	0.229	1.000	0.205	-0.417	0.328
TRP	0.110	-0.014	0.045	0.031	-0.193	-0.011	0.163	-0.053	0.179	0.411	0.205	1.000	-0.100	0.195
DOC	-0.459	0.274	0.357	0.130	-0.444	-0.529	-0.575	0.131	0.249	-0.182	-0.417	-0.100	1.000	-0.003
Si	0.033	-0.074	0.078	0.169	0.106	0.027	0.151	0.151	0.114	-0.001	0.328	0.195	-0.003	1.000
Epi Chla	-0.155	-0.174	0.264	0.265	-0.108	-0.102	-0.079	0.137	0.078	-0.070	0.025	0.072	0.217	0.316
Peri Chla	-0.320	0.298	0.165	0.106	-0.304	-0.341	-0.314	0.007	0.092	-0.089	-0.257	0.062	0.448	0.335
Sed Chl a	-0.031	0.183	0.169	0.146	-0.044	-0.103	-0.048	0.138	0.054	-0.014	0.027	0.168	0.323	0.409
%OM	-0.132	0.282	0.318	0.253	-0.159	-0.212	-0.192	0.113	0.282	-0.033	-0.072	0.302	0.470	0.460
%Carbonate	-0.140	0.317	0.192	0.090	-0.174	-0.205	-0.209	0.094	0.103	-0.044	-0.130	0.183	0.541	0.314
Sed TKN	-0.136	0.241	0.333	0.231	-0.155	-0.173	-0.201	0.108	0.290	0.001	-0.144	0.261	0.506	0.319
Sed TP	0.036	0.245	0.048	0.101	-0.005	-0.073	-0.009	0.020	0.052	0.059	0.095	0.143	0.346	0.468
Macrophytes	-0.206	0.516	-0.035	-0.034	-0.300	-0.390	-0.197	0.046	-0.009	-0.173	-0.016	0.132	0.230	0.378

B2) Monthly averages – sediments, algae, and submersed vegetation

Variable	Epi Chla	Peri Chla	Sed Chl a	%OM	%Carb.	Sed TN	Sed TP	Macrophytes
NTU	-0.155	-0.320	-0.031	-0.132	-0.140	-0.136	0.036	-0.206
pH	-0.174	0.298	0.183	0.282	0.317	0.241	0.245	0.516
Conductivity	0.264	0.165	0.169	0.318	0.192	0.333	0.048	-0.035
Alkalinity	0.265	0.106	0.146	0.253	0.090	0.231	0.101	-0.034
Chl a	-0.108	-0.304	-0.044	-0.159	-0.174	-0.155	-0.005	-0.300
OSS	-0.102	-0.341	-0.103	-0.212	-0.205	-0.173	-0.073	-0.390
ISS	-0.079	-0.314	-0.048	-0.192	-0.209	-0.201	-0.009	-0.197
TN	0.137	0.007	0.138	0.113	0.094	0.108	0.020	0.046
NO3	0.078	0.092	0.054	0.282	0.103	0.290	0.052	-0.009
NH3	-0.070	-0.089	-0.014	-0.033	-0.044	0.001	0.059	-0.173
TP	0.025	-0.257	0.027	-0.072	-0.130	-0.144	0.095	-0.016
TRP	0.072	0.062	0.168	0.302	0.183	0.261	0.143	0.132
DOC	0.217	0.448	0.323	0.470	0.541	0.506	0.346	0.230
Si	0.316	0.335	0.409	0.460	0.314	0.319	0.468	0.378
Epi Chla	1.000	0.556	0.547	0.494	0.393	0.461	0.353	0.023
Peri Chla	0.556	1.000	0.711	0.749	0.783	0.752	0.697	0.320
Sed Chl a	0.547	0.711	1.000	0.897	0.884	0.891	0.862	0.181
%OM	0.494	0.749	0.897	1.000	0.912	0.959	0.855	0.315
%Carbonate	0.393	0.783	0.884	0.912	1.000	0.935	0.800	0.253
Sed TN	0.461	0.752	0.891	0.959	0.935	1.000	0.848	0.145
Sed TP	0.353	0.697	0.862	0.855	0.800	0.848	1.000	0.250
Macrophytes	0.023	0.320	0.181	0.315	0.253	0.145	0.250	1.000

C) Biweekly samples - water quality

Variable	Depth	NTU	pH	Cond	Alk	DIC	CHla	OSS	ISS	NH3	TRP	DOC
Depth	1.000	-0.572	0.061	0.191	0.074	0.054	-0.157	-0.567	-0.480	0.108	-0.315	0.027
NTU	-0.572	1.000	-0.135	-0.237	-0.184	-0.143	0.403	0.789	0.878	0.146	0.129	-0.410
pH	0.061	-0.135	1.000	-0.078	-0.236	-0.402	-0.123	-0.212	-0.130	-0.272	-0.199	0.124
Cond	0.191	-0.237	-0.078	1.000	0.701	0.681	-0.026	-0.063	-0.202	-0.020	-0.410	0.432
Alk	0.074	-0.184	-0.236	0.701	1.000	0.983	0.142	0.093	-0.178	0.238	-0.105	0.419
CHla	-0.157	0.403	-0.123	-0.026	0.142	0.159	1.000	0.475	0.157	-0.054	-0.072	-0.090
OSS	-0.567	0.789	-0.212	-0.063	0.093	0.133	0.475	1.000	0.745	0.269	0.107	-0.110
ISS	-0.480	0.878	-0.130	-0.202	-0.178	-0.141	0.157	0.745	1.000	0.278	0.150	-0.334
NH3	0.108	0.146	-0.272	-0.020	0.238	0.271	-0.054	0.269	0.278	1.000	0.126	-0.057
TRP	-0.315	0.129	-0.199	-0.410	-0.105	-0.070	-0.072	0.107	0.150	0.126	1.000	-0.067
DOC	0.027	-0.410	0.124	0.432	0.419	0.376	-0.090	-0.110	-0.334	-0.057	-0.067	1.000



Appendix 2: Relationship between macrophyte biomass and water depth throughout the study. Presented are the data for each of 5 replicates for all 10 ponds from both 2001 and 2002; n=100. A simple linear regression was performed (r^2=0.00058; p=0.8182), and showed that no linear relationship was present between water depth and macrophyte biomass.