Experimental Manipulation of Connectivity and Common Carp; the Effects on Native Fish, Water-Column Invertebrates, and Amphibians in Delta Marsh, Manitoba

By

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DEDICATION

This thesis is dedicated to Shawn Edinger - there *was* actually light at the end of this long tunnel!

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ABSTRACT

Common carp (*Cyprinus carpio*) have been hypothesized to contribute to declines in aquatic macrophytes, waterfowl, and water clarity in Delta Marsh, an 18,500 ha freshwater coastal wetland on Lake Manitoba, Canada. Ten ponds (1-13 ha) were chosen for a two-year experimental manipulation study. Following a year of baseline monitoring, manipulations were conducted in 2002. To facilitate access by carp into isolated ponds, channels were blasted from the main marsh into two ponds. Meanwhile, to restrict or exclude carp access into ponds, channels were either screened or diked to four ponds. Two connected and two isolated ponds functioned as controls. Although common carp were the original subject of the study, it became apparent that hydrological connection to the surrounding marsh had a paramount importance on the abundance and diversity of the fish, amphibian and water-column invertebrate communities. Connectivity, or lack of connectivity, played an important role in the distribution of the fish community, and subsequently the composition and abundance of water-column invertebrates and amphibians. Ponds with direct connection had diverse, mixed-species fish assemblages, with fewer invertebrates and amphibians. Ponds with restricted connections had fish communities composed of tolerant small-sized species and increased abundance of invertebrates and amphibians. Ponds that lacked connection could freeze and lose all fish, and had higher numbers of invertebrates and amphibians. An absence of adult common carp may have been responsible for increased amphibian numbers in the screened ponds, however more study is needed. Confounding impacts of fluctuating water levels made it impossible to implicate common carp for most changes observed within ponds in Delta Marsh.

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CHAPTER 1 – INTRODUCTION

1.0 Common Carp

Common carp (*Cyprinus carpio* L.) are the largest species in the minnow family, Cyprinidae (Figures 1-1, 1-2). Average length is reported to be between 38 - 47 cm (Scott and Crossman 1998), with some fish reaching 80 cm and weighing more than 18 kg in Canadian waters (McCrimmon 1968). These are large, deep-bodied, laterally compressed fish. Common carp are closely related to goldfish (*Carassius auratus* L.) by having a broad spine on both the dorsal and anal fins. Common carp have a protrusible mouth with two pairs of barbels; one pair on the upper lip and the other pair at the corner of the mouth (Stewart and Watkinson 2004). Body color is golden tan to olive green.

Linnaeus first described common carp in 1758. According to one interpretation, the Latin name *Cyprinus* was derived from Cyprus, referring to its fecundity, and *carpio* is Latin for carp (McCrimmon 1968; Scott and Crossman 1998). There are three subspecies of common carp, the dominant type, *C. carpio communis* (McCrimmon 1968) or *C. carpio carpio* (www.fishbase.org), with regular concentrically arranged large cycloid scales; *C. carpio specularis* or mirror carp, with abnormally large scales scattered along the body and the remaining body scaleless; and *C. carpio coriaceus, sive, nudus* or the leather carp, with the majority of the body devoid of scales (McCrimmon 1968).

Common carp prefer slow flowing warm waters of pools, lakes and marshy areas, and tend to thrive in shallow brackish, vegetated, mud-bottomed habitats (Cooper 1987; Scott and Crossman 1998). Unlike most freshwater fish, common carp have exceptional tolerances and are able to thrive in nearly anoxic, highly polluted waters. They are also well adapted to rapid temperature fluctuations, and have a growth optimum between 10°C to 25°C, with minimum and maximum lethal limits of 1.7°C and 38°C, respectively (Scott and Crossman 1998). Common carp flourish in turbid conditions, until their lethal limit of 165,000 ppm is met (Bardach et al. 1972).

Adult common carp are opportunistic omnivores consuming benthic materials such as chironomid larvae, detritus, plant remains, zooplankton, phytoplankton and small fish (Sigler 1958; Scott and Crossman 1998). They feed by sucking up a mouth full of bottom sediments, forcefully expelling it into the water while filtering out food particles. Plant material is ground using their pharyngeal teeth. Young common carp primarily feed on zooplankton (Scott and Crossman 1998).

Spawning commences in early spring and summer when water temperatures reach 17°C. Common carp can spawn multiple times in a season when temperatures are optimal (Scott and Crossman 1998). Females are extremely fecund, a 400 to 500 mm common carp can produce 50,000 to 100,000 eggs (Stewart and Watkinson 2004). Swee and McCrimmon (1966) in (Stewart and Watkinson 2004) reported a single 10.1 kg female produced 2,208,000 eggs. Adhesive eggs, measuring 1 mm in diameter, are broadcast on aquatic vegetation. Depending on water temperature, eggs hatch within three to six days (Scott and Crossman 1998). Common carp are fast growing. Young-of-the-year (YOY) fish attained lengths of 70 to 90 mm by early August in Delta Marsh (Candace Parks, personal observation).

Although native to temperate areas of Eurasia, namely the Black Sea and Aegean basins, today, common carp are one of the most widely distributed freshwater fishes in the world, largely due to human intervention (Scott and Crossman 1998). Common carp have been successfully reared since 475 B.C., which Bardach et al. (1972) speculate is the longest history of culture for any fish species. Their long history of pond culture is due to the hardiness of all life stages, and their popularity as a food fish. Common carp were successfully introduced into North America in 1877 (McCrimmon 1968; Crossman 1984), and populations have also been established in South America, New Zealand, Australia, Mexico, India, and parts of Africa and the Middle East (Roberts et al 1995; Scott and Crossman 1998; www.fishbase.org).

1.1 Introduction of Common Carp into Manitoba

As in other areas of Canada, common carp were introduced into Manitoba for commercial purposes. They were stocked in Manitoba in 1885, 1886, and 1889, however no permanent populations were established in the province until 1938 (Atton 1959; Crossman 1968; Crossman 1984). The first recorded appearance of the species was in the Red River near Lockport (Hinks 1943). Presently, common carp are widely distributed in the province. It was once thought that cold waters would prevent the northward spread of common carp, however their wide range of temperature tolerances has allowed the invasion of common carp as far north as the Hayes River Estuary on Hudson Bay (Ralley 2002).

Although introduced as a food fish, common carp have received little fanfare for recreational or commercial purposes in Manitoba and North America. Numerous attempts have been made to encourage local consumption of common carp, without much success. The Freshwater Fish Marketing Corporation, based in Winnipeg, has found a tentative market for common carp roe. This new market is said to "double" the return for common carp fishers in Manitoba (www.freshwaterfish.com/english.htm). The majority of common carp captured in Manitoba are exported to the eastern United States for the Kosher food market (Dennis Geisler, Freshwater Fish Marketing Corporation, personal communication). In the southern United States and Europe, recreational angling and/or bow fishing for common carp are quite popular. Several organizations and websites actively promote recreational carp fishing in Canada (*e.g.,* Carp Net [www.carp.net], Canadian Carp Club [www.canadiancarpclub.on.ca], Carp Anglers Group [www.carpanglersgroup]), however the majority of fishers do not regard common carp as a food or trophy fish. Local Australian governments have promoted the recreational removal of common carp, however these practices have had little effect at reducing common carp populations (Koehn et al. 2000). Seventy percent of commercially harvested common carp in Australia is sold to European and Asian markets. Suggested uses for common carp include fish oils, fodder for livestock, fertilizer and fish "leather". However, low market value, stemming from poor public perception and relatively high processing requirements, limits the prospects for expanding these markets in Australia (Koehn et al. 2000).

1.2 Impacts of Common Carp to the Aquatic Environment

Although common carp were initially introduced as a food fish, the long-term ecological impact of this exotic species was not immediately considered. At present, the majority of the scientific literature shows that high densities of common carp are detrimental to aquatic vegetation (Robel 1961; King and Hunt 1967; Crivelli 1983; Kolterman 1990;

Zambrano and Hinojosa 1999; Lougheed and Chow-Fraser 2001). In particular, soft-leaved, shallow rooted species such as sago pondweed (S*tuckenia pectinatus* [L.] Boerner) (Fletcher et al. 1985; Zambrano and Hinojosa 1999), are vulnerable through physical disturbance during feeding and spawning. Common carp also increase turbidity by stirring up flocculant bottom sediments in search of food (Robel 1961; Cooper 1987; Kolterman 1990; Drenner et al. 1997; Lougheed and Chow-Fraser 1998; Scott and Crossman 1998; Lougheed and Chow-Fraser 2001; Zambrano et al. 2001; Parkos III et al. 2003). Increased sediment suspension contributes to low light levels for aquatic macrophytes. Sediment disturbance also increases the availability of water-column nutrients, facilitating phytoplankton growth (Meijer et al. 1990; Qin and Threlkeld 1990; King et al. 1997). However, areas where bottom substrates are composed mainly of sand remain clear, even when common carp are present (Scheffer 1998).

1.3 Influence of Common Carp on Native Fish, Invertebrates and Waterfowl

Common carp are thought to directly and indirectly compete with native fauna for food and space. An overall reduction in aquatic vegetation may have cascading impacts on aquatic invertebrates, waterbirds and native fish. High densities of common carp in wetland areas have been reported to cause an indirect reduction in waterfowl use (Cahoon 1953; King and Hunt 1967). Fewer aquatic plants reduce the availability of nesting material, food resources, and plant cover for waterfowl and other waterbirds. Nesting waterfowl are attracted to areas of high aquatic vegetation and consequently high invertebrate abundance, which are often consistent with fishless wetlands (Mallory et al. 1994; Hanson and Riggs 1995).

From a fisheries perspective, the removal of aquatic vegetation disrupts fish spawning areas, egg laying sites and refuge areas for young fish (Engel 1985; Cooper 1987), and reduces the availability of aquatic invertebrate prey (Hann 1995; Sandilands 2000). Craig and Babaluk (1989) suggest that increased turbidity interferes with northern pike (*Esox lucius* L.) foraging efficiency, preventing pike from seeing their prey and thereby reducing overall growth. Additionally, the "suck, spit and pick" method of benthic feeding by common carp increases turbidity, which may effectively lessen their own risk to avian and piscivorous predation (Bruton 1985; Richardson et al. 1995). In addition, the presence of long-lived common carp is also thought to "monopolize" the food chain, locking productivity as carp biomass for extended periods of time, until the carp dies (Hanson and Butler 1994a; Ivey et al. 1998; Brazner et al. 2001).

1.4 Managing Common Carp

Control of common carp has been attempted by erecting physical barriers, water drawdown, poisoning, and harvesting (Cahoon 1953; King and Hunt 1967; Crivelli 1983; Scheffer 1998; Wilcox and Whillans 1999). In several studies, water quality, aquatic macrophytes densities and water clarity often improved after common carp removal (Tryon 1954; Robel 1961; Scheffer 1998; Lougheed et al. 2004).

Dikes, fences, or screens have been commonly used to restrict adult common carp access. In the early 1960s, an attempt was made to exclude common carp from Delta Marsh (McCrimmon 1968). Screens were erected across the channels connecting the marsh with Lake Manitoba. Although they appeared effective in reducing ingress of adult common carp, other large native species such as northern pike, which seasonally use marshes for spawning and nursery purposes, were also excluded from the marsh.

The screens became clogged with debris, required extensive maintenance and cleaning, and eventually fell into disrepair. The program was abandoned after only three years. No data from this program were available to ascertain the response of the marsh to this type of management.

In 1997, a screened fishway at the inlet of Cootes Paradise Marsh, located at the west end of Lake Ontario, became operational. The goal of the screens was to prevent mature common carp (> 5 cm wide and 30 cm in length) from accessing the marsh (Wilcox and Whillans 1999). Larger fish were instead collected in series of large baskets and manually sorted; fish other than common carp were released into the marsh, whereas common carp were released back into Hamilton Harbour. In the first year it was estimated that 97,000 common carp were excluded from the marsh and released back into the harbour (Wilcox and Whillans 1999). Within weeks, aquatic macrophyte densities increased. Species that responded most dramatically included sago pondweed (*Stuckenia pectinatus*), coontail (*Ceratophyllum demersum* L.), curly pondweed (*Potamogeton crispus* L.), and leafy pondweed (*P. foliosus* Rafinesque) (Lougheed et al. 2004). A similar structure has been constructed at Metzger Marsh, located in an embayment of Lake Erie (Wilcox and Whillans 1999).

Common carp removal programs (Cahoon 1953; Threinen and Helm 1954) which have used poisons (*i.e.,* piscicides) such as Toxaphene (Scheffer 1998) and Rotenone (Hanson and Butler 1990; Schrage and Downing 2004), have successfully reduced or eliminated overabundant common carp populations in several areas, including Lake Christina, Minnesota, and Ventura Marsh, Iowa. Seasonal drawdowns have also been used successfully to desiccate common carp eggs and to reduce adult spawning common carp populations (Kolterman 1990; Scheffer 1998).

1.5 Current Impacts to Delta Marsh

Delta Marsh is a renowned breeding and fall staging area for continental waterfowl populations (Batt 2000), however researchers and concerned citizens alike agree, that waterfowl numbers in the marsh have decreased appreciably over the last forty years (Technical Committee for Development of the Delta Marsh 1968; Ould 1980; Bond 1996). Along with waterfowl, reductions in aquatic macrophytes and inland islands have also been noted, in addition to a dramatic increase in water column turbidity and shoreline erosion (Goldsborough and Wrubleski. 2001; Gordon Goldsborough, University of Manitoba, unpublished data).

Two stable states, namely a clear water state and a turbid water state, often occur in shallow water bodies (Scheffer 1990). The clear water state is characterized by abundant aquatic macrophytes, low algal biomass, and low turbidity. In contrast, the turbid stable state is dominated by high algal biomass, high turbidity, and few aquatic macrophytes. Each stable state is resistant to change to the alternative state via a negative feedback loop. In order to switch between states, a certain threshold must be exceeded (Scheffer 1990). These conditions have been observed at Delta Marsh. Most of the marsh is turbid however a few shallow ponds around the periphery lack direct connection to the main marsh and are relatively clear. Elsewhere, aquatic ecosystems have shifted between stable states to the preferred aquatic macrophyte dominated clear state through restructuring of the fish community (*e.g.,* Lake Christina (Hanson and Butler 1994b), Cootes Paradise Marsh (Lougheed et al. 2004), Ventura Marsh (Schrage and Downing 2004), and Metzger Marsh (Wilcox and Whillans 1999; Wilcox 2003)).

At Delta Marsh, several interacting mechanisms are thought to be responsible for maintaining the turbid stable, most notably, the proliferation of common carp and the stabilization of Lake Manitoba water levels. In 1961, the Province of Manitoba constructed the Fairford Control Structure to stabilize lake levels (at approximately 247.6 m ASL). This has had the result of dampening large water level fluctuations, which in turn are critical for maintaining healthy coastal marshes (The Lake Manitoba Regulation Review Advisory Committee 2003a). It is also believed that an absence of fluctuating water levels may also contribute to the proliferation of common carp, by providing permanent refuges and ideal spawning areas. The impact of common carp on Delta Marsh's flora and fauna has never been comprehensively evaluated and their management has largely gone unchecked. Without further study, Delta Marsh may remain in a turbid state indefinitely. The goal of the experimental manipulation project described in this thesis was to examine the impact of common carp on native fish, amphibians, and water-column invertebrates in Delta Marsh.

1.6 Delta Marsh

Delta Marsh is an 18,500 ha coastal wetland on the south shore of Lake Manitoba (50° 11' N, 98° 19' W), located north of Portage la Prairie, in south-central Manitoba (Figure 1-3). As one of the most famous freshwater wetlands in North America, Delta Marsh is home to numerous species of fish, mammals, plants, songbirds and most notably waterfowl (Batt 2000). The marsh is largely surrounded by agriculture to the south and Lake Manitoba to the north. Delta Marsh was designated a "Wetland of

International Significance” under the Ramsar Convention in 1982, a Manitoba Heritage Marsh by the Manitoba Provincial Government in 1988, and an Important Bird Area (IBA) by Partners in Flight in 1999.

The marsh was formed approximately 2,500 years ago when wind and wave action formed a barrier-beach ridge from sediments deposited from the once northward flowing Assiniboine River. Remnants of the Assiniboine River (*i.e.,* Blind Channel) can be observed on the west side of marsh directly south of the Delta Marsh Field Station property. The resultant ridge separated the marsh from Lake Manitoba, sheltering it from direct physical and hydrological impacts of the lake (Teller and Last 1981).

Presently, Delta Marsh consists of a matrix of small and large shallow bays, isolated ponds and channels ranging in depth from <1 m to 3 m. Four openings (Deep Creek, Cram Creek, Delta Channel and Clandeboye Channel) in the beach ridge allow continuous water movement between the marsh and the lake. Spring run-off, and periodic wind set-ups and set-downs, ensure continual exchange of water between the lake and the marsh during the ice-free period (Wrubleski 1998; Batt 2000).

Water quality parameters collected at six sites on the west side of Delta Marsh in mid-July, August and September 1994, were summarized by Goldsborough (1994). Results indicate that the marsh is moderately brackish, with conductivity values ranging between 943 and 5080 µS/cm (mean 2198 µS/cm), pH from 8.1 to 9.2 (mean 8.7), nitrate+nitrite from <0.05 to 0.34 mg/L (mean 0.075 mg/L), ammonia from <0.005 to 2.94 mg/L (mean 0.081 mg/L), total phosphorus from <0.05 to 0.39 mg/L (mean 0.074

mg/L) and total chlorophyll from 2.2 to 24.6 µg/L (mean 11.49 µg/L). Earlier reports on Delta Marsh water quality can also be found in Anderson and Jones (1976).

Emergent vegetation composition has been described by Walker (1959; 1965), de Geus (1987), Shay et al. (1999), and summarized by Batt (2000). Dominant shoreline emergent vegetation consists of hardstem bulrush (*Schoenoplectus acutus* var. *acutus* [Muhl. *ex* Bigelow] A. and D. Löve) and cattail (*Typha* spp.; made up of *T. latifolia*, *T. angustifolia* and hybrid *T.* x *glauca*). Whitetop (*Scolochloa festucacea*), and to a lesser extent sedge species (*Carex* spp.), are found in the shallow marsh zone, and at higher elevations common reed (*Phragmites australis* (Cav.) Trin. *ex* Steud) is prevalent.

Anderson and Jones (1976) found 11 common submerged macrophyte species in the eastern portion of Delta Marsh. Dominant species, or species commonly associated with each other, included (listed in order of percent of aquatic macrophytes present); sago pondweed, a combination of sago pondweed and northern watermilfoil (*Myriophyllum sibiricum* Komarov), and sheathed pondweed (*Stuckenia vaginatus* [Turcz.] Holub). Other species in the marsh include hornwort (*Ceratophyllum demersum* L.), common bladderwort (*Utricularia macrorhiza* Le Conte.), Richardson's pondweed (*Potamogeton richardsonii* [Benn.] Rydb.), horned pondweed (*Zannichellia palustris* L.), wigeon grass (*Ruppia cirrhosa* [Petag.] Grande) flat stem pondweed (*P. zosteriformis*), and Eurasian watermilfoil (*M. spicatum*) (Anderson and Jones 1976; Sandilands 2000).

Information on the geology and soil composition of Delta Marsh can be found in Teller and Last (1981). Climate data can be found in the Delta Marsh Field Station Annual Reports (Delta Marsh Field Station 2006).

Lake Manitoba's water levels have been stabilized at 247.6 m (812.2 ft) ASL since the installation of the Fairford Control Structure in 1961 (The Lake Manitoba Regulation Review Advisory Committee 2003b) (Figure 1-4). As a consequence, water levels in the adjoining marshes, including Delta Marsh, have been stabilized.

In 1969, the Province of Manitoba constructed the Assiniboine River Diversion to divert floodwaters away from the city of Winnipeg into Lake Manitoba. The channel links the Assiniboine River, immediately west of Portage la Prairie, with Lake Manitoba, approximately 25 km to the north. The Diversion became operational in 1970. A failsafe was also built to allow excessively high flows to spill into the west side of Delta Marsh. The Diversion is responsible for adding several new fish species to Lake Manitoba and Delta Marsh, including central mudminnow (*Umbra limi* Kirkland), bigmouth buffalo (*Ictiobus cyprinellus* Valenciennes), channel catfish (*Ictalurus punctatus* Rafinesque), tadpole madtom (*Noturus gyrinus* Mitchill), brown (*Ameiurus nebulosus* Lesueur) and black bullhead (*A. melas* Rafinesque) (Stewart et al. 1985). Rock bass (*Ambloplites rupestris* Rafinesque) have recently been added to this list (Dale Wrubleski, Ducks Unlimited Canada, unpublished data). Recently, the Province of Manitoba accepted the recommendations made by the Lake Manitoba Regulation and Review Committee to partially de-regulate water levels in the lake. The Committee examined the management of Lake Manitoba water levels and the impacts of stabilization. The results of the final report concluded Lake Manitoba would benefit by increasing the amplitude of water level fluctuations, although not to the extent of pre-managed levels. It remains to be seen if these changes will allow Delta Marsh to recover without consideration of the management of common carp.

1.7 Issue Statement

Over the past forty years, Delta Marsh residents and users have observed a notable increase in water column turbidity, loss of aquatic macrophytes and a substantial decline in waterfowl use. Absences of natural wetland water level fluctuations, as a result of stabilization of Lake Manitoba, and the invasion of exotic common carp, are considered dominant forces behind the current deteriorated state of the marsh.

Therefore, it appears that this turbid ecosystem cannot again sustain levels of historic waterfowl populations with the continued absence of 1) common carp management, and 2) natural water level fluctuations which suppresses the dynamic cycle that has been integral to the overall health of Delta Marsh for thousands of years.

1.8 Objectives

The experimental manipulation project, described in this thesis, was developed to identify the mechanisms that facilitate the turbid stable state, and in turn to determine the most effective management plan for Delta Marsh. Specific objectives were as follows:

1. Determine the response of native fish, amphibians, and water-column invertebrates to experimental manipulation of common carp populations;
2. Provide a more detailed description of the fish species composition and frequency of occurrence within Delta Marsh;
3. Provide a description of the water-column invertebrate and amphibian community in a series of ponds within Delta Marsh.

1.9 Scope

The premise of this project was to characterize key differences among connected and isolated ponds, which act concurrently as surrogates for turbid and clear stable states in the larger Delta Marsh complex. Reasons for these differences are unknown since comprehensive comparisons have not been made. Hence these ponds provide a basis to identify parameters, or combination of parameters, responsible for maintaining either the clear or turbid state. Overall, the goal was to identify the factors (*i.e.,* feedback mechanism(s)) at work in order to develop a management plan to restore Delta Marsh to its former more productive, clear stable state.

In 1998 and 1999, baseline water quality data were collected every two to four weeks between May and August, from 44 sites throughout Delta Marsh (Gordon Goldsborough, University of Manitoba, unpublished data). Site selection, preparation and extensive water quality sampling began in the summer of 2000, and subsequently 10 ponds were selected for the study (Figure 1-5).

In 2001, all ponds (ranging in size from 0.8 to 12.6 ha) remained in their original state for baseline monitoring, whereby six ponds were connected (allowing water flow and common carp access) to the main marsh and four ponds were isolated (no water exchange or common carp access). In 2002, ponds were paired into the following experimental treatments; two connected and two isolated ponds remained in their natural state to act as controls, two connected ponds were screened, using either a conduit fence or culvert covers (Figures 1-6, 1-7), to allow water exchange but prevent access by larger adult fish species, two connected ponds were isolated by blocking connecting channels with sandbag dikes (Figures 1-8, 1-9, 1-10), and finally two isolated ponds had channels blasted into them to allow water exchange and access by fish, including common carp (Figures 1-11, 1-12, 1-13, 1-14). Treatments were applied in spring 2002, and sampling was carried out from early June to late August of 2001 and 2002.

Research for this thesis was a collaborative effort with a companion project undertaken by Stacy Hnatiuk, a Botany graduate student with the University of Manitoba. The objectives of her study were to determine the effects that common carp have on: (i) water quality; and (ii) algal and aquatic macrophyte productivity, abundance, and diversity. These investigations were conducted through comparisons made between and among the 10 experimental ponds with and without common carp, the same methodology employed in this study.

Although each thesis was researched and developed independently from the other, they were conducted simultaneously utilizing the same experimental ponds on both a temporal and spatial scale. The overlap in data collection periods will ultimately provide a clearer picture of the direct/indirect effect(s) introduced common carp have on the different tropic levels of Delta Marsh with respect to its floral and faunal communities.

1.10 Organization

This thesis is organized in four chapters. Chapter two and chapter three are written and organized as stand alone papers for future publication.

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TABLES AND FIGURES

Table 1-1. Location, area, and description of the 10 study ponds in Delta Marsh during the baseline (2001) and experimental years (2002).

Sites	Abb.[1]	UTM (m, Zone 14)	Open Water Area (ha)	Natural state	Manipulated State	Construction
				2001	2002	
Thompson's Bay	TB	543 273E 5556 596N	12.6	Connected	Connected	No
North School Bay	SB	548 533E 5556 895N	9.0	Connected	Connected	No
Section 5 Bay	S5	543 980E 5557 416N	5.2	Connected	Connected	Screened
Mid-Blind Channel	BC	544 293E 5558 385N	6.8	Connected	Connected	Screened
Mad Woman Bay	MW	544 063E 5556 895N	3.3	Connected	Isolated[2]	Diked
South Pitblado's Channel	PC	552 061E 5558 869N	3.0	Connected	Isolated	Diked
South Mackenzie Bay	MS	554 393E 5560 3383N	0.8	Temporarily connected	Connected	Blasted
Wye's Pond	WP	562 798E 5564 025N	3.1	Temporarily connected	Connected	Blasted
North Mackenzie Bay	MN	554 260E 5560 636N	11.9	Temporarily connected	Isolated	No
Emile's Pothole	EP	562 643E 5564 395N	2.7	Temporarily connected	Isolated	No

1-Abb. is short for abbreviation.

2-occasional wind-set up caused short-term flooding allowing fish to swim around the dike.

Figure 1-1. **Young-of-the-year common carp (*Cyprinus carpio*), approximately 7 cm in length (Photo by Candace Parks).**

Figure 1-2. **Size comparison of an adult and a young-of-the-year (YOY) common carp (Photo by Susan Hertam).**

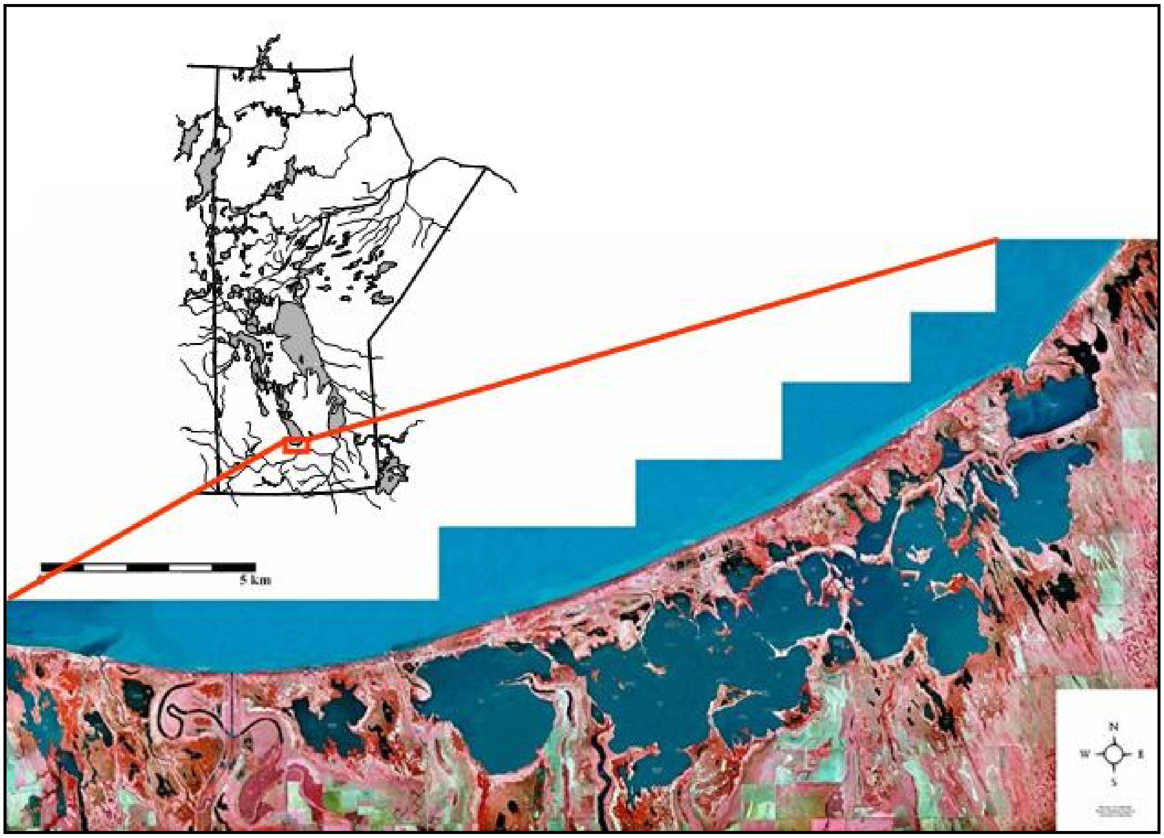

Figure 1-3. Delta Marsh at the south end of Lake Manitoba.

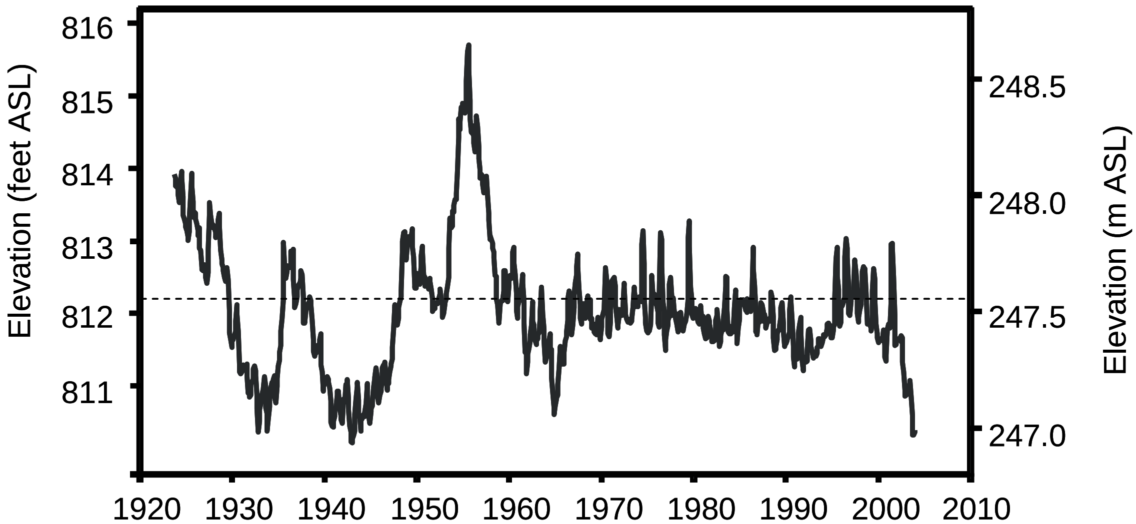

Figure 1-4. **Monthly mean water levels for Lake Manitoba, September 1923 to December 2003 (Steep Rock gauging station 05LK002, Environment Canada website: www.wsc.ec.gc.ca/hydat/H2O/index_e.cfm?cname=WEBfrmMeanReport_e.cfm).**

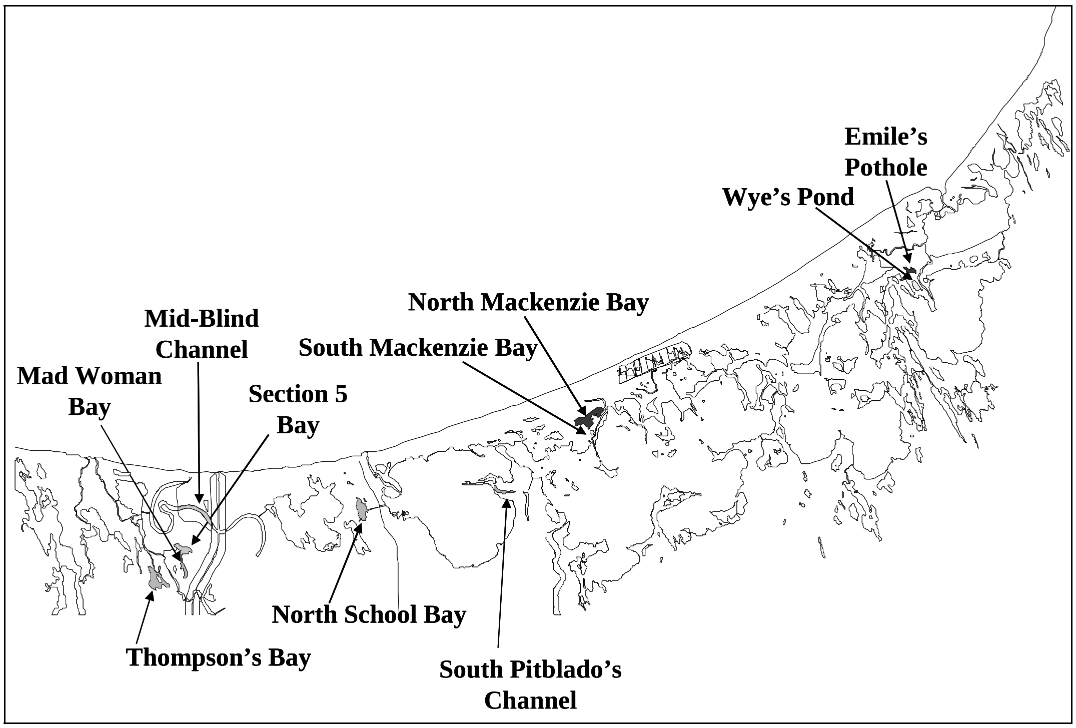

Figure 1-5. Location of study ponds in Delta Marsh, Manitoba.

Figure 1-6. Conduit fence constructed in channel upstream of Section 5 Bay on April 22, 2002. Stucco fencing was added to the top of the conduit fence to prevent fish like common carp from leaping over during periods of wind set-up (Photo by Dale Wrubleski).

Figure 1-7. Three screened culvert covers under the Delta Marsh Field Station winter road prevented large common carp from entering Mid-Blind Channel from West Blind Channel (Photo by Dale Wrubleski).

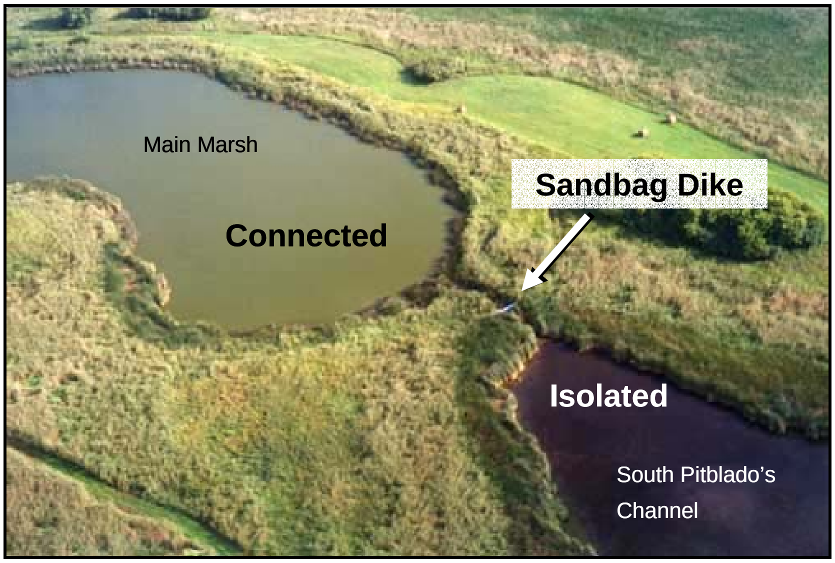

Figure 1-8. Sandbag dike isolating South Pitblado's Channel from Cherry Ridge and the main marsh (Photo by Gordon Goldsborough).

Figure 1-9. Sandbag dike constructed at entrance of South Pitblado's Channel on April 24, 2002 (Photo by Dale Wrubleski).

Figure 1-10. Sandbag dike constructed on May 2, 2002 at the channel entrance of Mad Woman Bay, to isolate the pond from Section 5 Bay and the main marsh (Photo by Dale Wrubleski).

Figure 1-11. Channel created by a dynamite blast connecting South Mackenzie Bay to Cook's Creek and the main marsh (Photo by Candace Parks).

Figure 1-12. Aerial photograph of South Mackenzie Bay with blasted channel, summer 2002 (Photo by Gordon Goldsborough).

Figure 1-13. Blasted channel connecting Wye's Pond to the main marsh (Photo by Dale Wrubleski).

Figure 1-14. Aerial photograph of Wye's Pond with newly created channel connecting the pond to the main marsh, summer 2002 (Photo by Gordon Goldsborough).

CHAPTER 2 – THE FISH COMMUNITY OF DELTA MARSH, MANITOBA; A LARGE COASTAL WETLAND

2.0 Introduction

Freshwater coastal wetlands across North America are being lost at an alarming rate largely due to human alterations. Along the lower Laurentian Great Lakes-St. Lawrence River for example, it has been estimated that 70% of the shoreline wetlands have been lost following European settlement (Moser et al. 1996). Although coastal wetlands are increasingly being regarded as highly productive and important ecosystems, they continue to be threatened by drainage, dredging, shoreline development, water level stabilization, isolation from lake waters, nutrient enrichment, pollution, increased turbidity, and the introduction of exotic species (Whillans 1979; Kriger et al. 1992; Mitsch and Gosselink 2000).

The ecological importance of coastal wetlands for supporting fish and wildlife has been described by Jaworski and Raphael (1978), Herdendorf et al. (1986), Herdendorf (1987, 1992), and others. Research in the Laurentian Great Lakes has revealed that coastal wetlands are crucial spawning, nursery, migration, feeding, and over-wintering habitat for many fish (Chubb and Liston 1986; Herdendorf 1987; Stephenson 1990; Jude and Pappas 1992; Brazner 1997), including commercial, recreational and forage species (Herdendorf 1992). For example, of 43 fish species inhabiting coastal wetlands of Lake Erie, 26 are considered valuable recreationally, commercially or as prey (Herdendorf 1992). The high recreational fish value of the Great Lakes is largely attributed to neighbouring coastal wetland habitats. Jaworski and Raphael (1978) estimated that Lake

Michigan's coastal wetlands are worth $116/wetland hectare to the sport fish industry per year.

A combination of high nutrients, shallow warm waters, and dense aquatic vegetation form an optimal foodchain base attracting numerous fish and wildlife to coastal wetlands (Herdendorf 1987; Stephenson 1988). Use of coastal wetlands by fish is presumed to be due to the high levels of primary productivity, abundant invertebrate prey, and the diversity of habitats that wetlands offer (Stephenson 1990; Jude and Pappas 1992). Despite numerous anthropogentic pressures, the remaining coastal wetlands in the Laurentian Great Lakes and elsewhere continue to perform essential fisheries functions.

Numerous authors (*e.g.,* Herdendorf et al. 1986; Herdendorf 1987; Stephenson 1990; Kriger et al. 1992) stress the need for additional research on fish community structure, habitat utilization, migratory and feeding function of Great Lakes coastal wetlands. Unfortunately, similar information on the fisheries of coastal wetlands bordering other North American lakes is very limited. The majority of published coastal wetland studies outside the Laurentian Great Lakes have focused on lower trophic levels and waterfowl production, resulting in large knowledge gaps pertaining to fish community structure and utilization of these habitats.

Examination of freshwater coastal wetland functions is timely, given that these wetlands have been disappearing, and those remaining are under constant threat of environmental degradation. In this paper, I provide a description of the fish community in a freshwater coastal wetland outside the Laurentian Great Lakes. The objectives of this chapter were,

1) to gain insight into the composition and abundance of the small-sized fish community within Delta Marsh, a large coastal wetland, in south-central Manitoba, 2) to compare my results with previously unpublished fish studies of the marsh, 3) compare the fish community of Delta Marsh with other coastal marshes, and 4) to gain a better understanding of the importance of the marsh to Lake Manitoba's commercial fisheries. This study was part of a larger collaborative effort that examined coastal wetland ecosystem responses to the experimental manipulation of the presence/absence of common carp (*Cyprinus carpio*).

2.1 Study Site

2.1.1 Delta Marsh

This study took place in Delta Marsh, an 18,500 ha barrier-beach protected marsh located on the south shore of Lake Manitoba (50° 11' N, 98° 19' W) (Figure 1-3). Delta Marsh is a slightly brackish, permanent lacustrine wetland (Batt 2000) connected to Lake Manitoba by four channels. Water quality has been summarized by Anderson and Jones (1976) and Goldsborough (1994), and emergent and submergent vegetation are described by Batt (2000). Aquatic vegetation varies spatially across the marsh with areas dominated by either submersed macrophytes (*e.g., Stuckenia pectinatus* (L.) Boerner, *Myriophyllum sibiricum* Komarov, *Utricularia macrorhiza* Le Conte) in open water, or emergent macrophytes (*e.g*., *Schoenoplectus acutus* var. *acutus* (Muhl. *ex* Bigelow) A. & D. Löve, *Typha* X *glauca* Godr., *Phragmites australis* (Cav.) Trin. *ex* Steud). Delta Marsh is shallow, on average less than 1 m deep (Macaulay 1973). At such shallow depths, fish are susceptible to summer and winter kills from low dissolved oxygen concentrations (Suthers 1982).

Along with large, wind-swept bays and a network of connecting channels, numerous small water bodies exist around the periphery of the main marsh. These water bodies range from small isolated or connected ponds, to channel remnants of the Assiniboine River that once flowed into Lake Manitoba (Teller and Last 1981). Ponds vary from turbid waters with few aquatic macrophytes to clear waters with abundant aquatic macrophytes (Evelsizer 2001).

2.1.2 Study Ponds

Information on the location, area and experimental treatment of the 10 study ponds is presented in Table 1-1 and Figure 1-5. Ponds were chosen based on several factors, including location, size, ownership (*i.e.,* private or Crown land), presence or absence of a connection to the main marsh, logistics, and accessibility. In 2001, four ponds were isolated from the main marsh, while the remaining six were connected by natural or man-made channels. Although four of the ponds are described as isolated, high water levels on Lake Manitoba in the spring, along with several storm events in 2001, significantly raised water levels in the marsh (Figure 2-1) and resulted in these isolated ponds being connected to the main marsh for varying periods of time. To reduce confusion, isolated ponds with varying connection to the main marsh in 2001 will be referred to as “temporarily connected ponds” for the remainder of this paper.

In 2002, several ponds were altered as part of the common carp manipulation study. Two connected (Thompson’s Bay, North School Bay) and two isolated ponds (North Mackenzie Bay, Emile’s Pothole) were maintained in their natural state (Table 1-1). Four connected ponds were selectively isolated from the main marsh by either the installation of a fish screen or construction of a sandbag dike. The screens [a conduit fence across a channel (Section 5 Bay), and a series of three culvert screens (Mid-Blind Channel)] allowed water flow between the main marsh and the ponds, yet prevented access by adult common carp (Figures 1-6, 1-7). The conduit fence consisted of four A-frame stands that supported two aluminum beams positioned perpendicular to shore, one above the sediment and the other just above the water surface. Aluminum conduit pipes were inserted vertically into holes that had been drilled into both beams, thus creating a sloped fence with 5 cm gaps (Figure 1-6). The second screened pond (Mid-Blind Channel) was separated from the main marsh by a road crossing with three culverts under the road. Each culvert was fitted with a frame that held two removable steel grates with vertical steel bars 5 cm apart (Figure 1-7). Each grate could be individually removed for cleaning, thus preventing adult common carp from entering the pond. This gap width was selected as the optimum width to prevent access by common carp larger than 34 cm total length, yet would allow passage of native[1] fish, including northern pike (*Esox lucius*) up to 49.4 to 70.0 cm total length (French III et al. 1999). The screens were similar to those used on fishways at Cootes Paradise Marsh and Metzger Marsh. Both coastal marshes have screens present to prevent access by large common carp (Wilcox and Whillans 1999).

Sandbag dikes were constructed on the connecting channels into two ponds (South Pitblado's Channel and Mad Woman Bay) (Figures 1-8, 1-9, 1-10) to completely isolate each pond, and prevent water and fish exchange with the main marsh. Dikes were

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[1] In this context, native refers to fish species either naturally found in Delta Marsh, or naturally occurring in adjacent waterbodies (*e.g.*, black bullheads (*Ictalurus melas*), rock bass (*Ambloplites rupestris*), tadpole madtom (*Noturus gyrinus*) among others) and have entered the marsh due to connection of the Assiniboine River Diversion to Lake Manitoba.

constructed on April 24 (South Pitblado's Channel) and May 2 (Mad Woman Bay). Dikes were constructed of polypropylene sandbags, five sandbags wide at the base and two sandbags wide at the top. A 6 mil polyethylene sheet was positioned within each dike to provide additional water proofing, and each dike was covered with a polyethylene tarp to provide long-term protection of the sandbags from UV damage. Each dike extended laterally approximately one meter on either side of the channel to prevent water from flowing around the dike during wind set-up or storm events. Ice had already started to melt on Delta Marsh and some fish movement into the marsh and study ponds had already taken place prior to construction of the dikes. In addition, the dike at Mad Woman Bay was unable to completely block rising waters due to wind set-up on Delta Marsh (Figure 1-10). Spawning fish (*e.g.*, northern pike) were observed swimming around the dike during these events (Candace Parks, personal observation).

Two ponds, previously not connected to the main marsh, were connected via new channels (approximately 2 m wide and 1 m deep) created by blasting with ditching dynamite (Powerfrac, 50 x 400 mm sticks, 1250 g/stick). Prior to blasting, terrestrial vegetation was cleared along a 4 m wide path between the main marsh and the ponds to facilitate ground thaw. Charges were positioned approximately 2 m apart and buried 0.5 m in the ground. Charges were connected by a fuse, which was triggered sequentially to ensure the channel would remain clear of debris. Channels were blasted on May 17 (South Mackenzie Bay) and June 14 (Wye's Pothole) (Figures 1-11, 1-12, 1-13, 1-14). These newly constructed channels allowed water exchange and permitted access by fish, including common carp.

2.2 Methods and Materials

2.2.1 Field Sampling Methods

Sampling of the fish community in the ponds was conducted using Beamish traps, a passive trapping method similar to Fyke nets (Beamish 1973). This trap design was selected because it did not have rigid hoops or frames, could be modified to sample in very shallow water, and was relatively easy to transport and set up. Importantly, this trap allowed sampling of the entire water column, capturing both pelagic and benthic fish species. Both young-of-the-year (YOY) and adult fish were captured, although the majority of the catch was YOY fish (Table 2-1). A few adult fish of larger species [*e.g.*, freshwater drum (*Aplodinotus grunniens*), northern pike, yellow perch (*Perca flavescens*), white sucker (*Catostomus commersoni*), bullheads (*Ameiurus* spp.) and common carp] were caught but are not reported in this paper.

In 2001, un-baited minnow traps were also used to sample fish. Traps were set mid-water column at six locations in each pond, four times throughout the summer. Unfortunately, minnow traps selectively captured pelagic species such as fathead minnow (*Pimephales promelas*) and brook stickleback (*Culaea inconstans*), which composed 76.9% and 19.2% of the total catch, respectively. Placement of the traps mid-water column likely precluded the capture of benthic species such as darters *Etheostoma* spp. and mudminnows *Umbra* sp. In comparison, Beamish traps captured a more diverse fish fauna comprised of a wider variety of pelagic and benthic species. In many cases, minnow traps may actively attract species such as fathead minnows and brook sticklebacks, which use traps as nesting sites leading to biases in species abundances

(Suthers 1982). Also, the opening of the minnow trap precluded the capture of larger fish species. As a result, the use of minnow traps was discontinued the following year.

Two slightly different versions of the Beamish trap were used in this study. In the first year of sampling, a trap was used that was similar to the original description (Beamish 1973). This trap (referred to as the 'older' trap) was constructed of mesh with 1.5 mm openings, and a 15 x 15 cm square funnel opening in the box. The second trap (referred to as the 'newer' trap) was similar in design, but had mesh with 3 mm openings and incorporated a rigid 15 cm diameter plastic ring in the inner funnel opening. This ring was added to ensure that the funnel would remain open with fluctuating water levels. Wind set-up and set-down on Lake Manitoba and Delta Marsh resulted in fluctuating water levels, and I was concerned that during periods of low water levels, the trap funnel would partially collapse, preventing fish capture. The rigid ring in the funnel helped prevent this from happening.

In the first year of sampling, a single older Beamish trap was set randomly every three weeks in each pond from June to August, 2001. In the second year of sampling, an older trap and a newer trap were used on an approximate three-week rotation in each pond from May to August, 2002. The first trap was set randomly in the pond and the second trap was set on the opposite shore from the first trap. Traps were secured with three or four bamboo stakes and retrieved after 24 hours.

Trap catches were emptied into a large holding tub, identified, counted, and released immediately. To avoid unnecessary fish mortality, larger catches (> 1000 fish) were enumerated using a sub-sampling technique. Randomly, fish were removed from the trap and placed into a smaller 5-L collecting bucket. The captured fish within the bucket were then identified and counted. This was done twice in order to get a more accurate representation of the fish composition. The same 5-L bucket was then used to estimate the number of fish remaining in the Beamish trap by counting the successive buckets until the trap was empty. Presence of rare fish species in the trap was also recorded. Estimated catches were determined by taking the average count for the two sub-sampled buckets, multiplied by the total number of buckets.

Low water levels, coupled with periodic wind set-downs in 2002, prevented the use of older Beamish traps in two of the shallowest ponds. Older traps lacked the rigid ring in the funnel opening and thus collapsed, preventing access by fish. Instead, a single newer Beamish trap was set in Thompson's Bay and Section 5 Bay throughout the summer. As the summer progressed, each pond became shallower. By mid-summer, traps could only be positioned in the deepest part of each pond. In Section 5 Bay, the Beamish trap was set beside the channel connecting Section 5 Bay to Mad Woman Bay. At Thompson's Bay, the trap was set in the channel leading into the pond with the funnel facing directly into the pond. The position of the trap at Thompson's Bay meant that most fish leaving the pond were funneled directly into the trap, and this resulted in highly inflated numbers of fish captured for this pond.

2.2.2 Biomass Determination

In addition to providing abundance estimates based on counts, I also estimated abundance based on biomass (fresh weight). To determine approximate biomass, 11 fish species, including white sucker, black bullhead (*Ameiurus melas*), Iowa darter (*Etheostoma exile*), brook sticklebacks, ninespine sticklebacks (*Pungitius pungitius*), fathead minnows, spottail shiner (*Notropis hudsonius*), emerald shiner (*Notropis atherinoides*), yellow perch, common carp and northern pike were collected in 2004 (Appendix A). Each fish was weighed and fork length measured. Based on all data obtained, an overall mean weight for each species was determined. The seasonal mean weight for each species was then multiplied by the seasonal mean number to determine an approximate biomass of fish captured in the 10 study ponds. Due to low numbers in the ponds, lengths and weights for some species were not obtained [*i.e.*, trout-perch (*Percopsis omiscomaycus*) central mudminnow, brown bullhead (*Ameiurus nebulosus*), freshwater drum and johnny darter (*Etheostoma nigrum*)], and biomass estimates were not determined. Since Iowa darters are approximately the same shape and size as johnny darters, the calculated mean weight of Iowa darters was applied to johnny darters to estimate biomass.

2.2.3 Data Summation and Analyses

A two-factor without replication ANOVA analysis revealed no significant difference ($F_{1, 15} = 0.03$, $p > 0.05$) in species composition between older and newer Beamish traps. Thus, each sample date in each pond, an average of the two traps was calculated for 2002. To account for unequal sampling effort and timing differences in the collection of Beamish traps among the 10 ponds and between the two years, an overall mean trap catch for each pond was calculated. This was done to compare fish abundance trends among the 10 ponds and between the two sampling years.

Trap data are presented in this paper as seasonal mean trap counts, as well as relative proportions. Proportions of trap catch represented by each fish species are provided to compare the relative fish distributions in each pond and among the study ponds.

Standardizing to a sum of 100% minimizes the influence of the inflated numbers (*i.e.*, Thompson's Bay). The data obtained from Thompson's Bay is 2002 are still useful to determine what fish species utilized this pond. However, when comparing the seasonal means and biomass between the two years, values for Thompson's Bay in 2002 were not included to eliminate the influence of the inflated counts (*e.g.*, yellow perch), likely attributed to trap placement rather than actual fish numbers in the pond.

The number of species captured in an area, or absolute richness, s, is a common indicator of diversity. For example, an area with $s = 10$ is more diverse than an area where $s = 4$. Effective richness (N_2 or Hill's N_2) is similar to absolute richness yet it considers the relative proportion of individuals within each species. Therefore if all species have equal numbers of individuals, effective richness would be maximized.

The formula based on the Simpson Index (D) is as follows;

$$D = \frac{1}{\sum_{i=1}^{s} p_i^2}$$

Where:

p_i = Proportion of taxa $_i$ in the community (Krebs 1999).

The range for effective richness is between 1 and s, where the closer to s the greater the diversity (Krebs 1999). N_2 was chosen as an appropriate diversity measure since it is less sensitive to rare species and thus allowed comparison between and among the study ponds. Evenness (E = N_2/s), expressed as a percent, describes the distribution of taxonomic groups within each sample ranging between zero and one. The higher the proportion of a single species, the lower the overall evenness (Legendre and Legendre 1998).

2.3 Results

2.3.1 Delta Marsh Fish Community

Sixteen fish species, representing nine families were collected during the current study (Table 2-1). Based on numbers of fish present, cyprinids were the most abundant family captured, followed by ictalurids, percids and catostomids in 2001 (Table 2-2). Again in 2002, cyprinids were most abundant, followed by gasterosteids and ictalurids. Numerically, fathead minnows were the most abundant fish species in the study ponds in both years (Table 2-2). In 2001, fatheads composed 46.9% of the average catch (Table 2-2). Black bullhead (22.3%), yellow perch (15.5%), white sucker (6.4%), brook stickleback (5.7%), and common carp (1.9%) were also commonly captured (Table 2-2). Fathead minnows accounted for 46.1% of the average Beamish trap catch in 2002 (Table 2-2). Common carp (35.0%), brook stickleback (6.3%), black bullhead (4.9%), spottail shiner (3.5%), and yellow perch (3.4%), were also caught frequently (Table 2-2).

In 2001, fathead minnows accounted for the largest proportion of fish biomass representing 33.8% of the average catch, followed by black bullheads (23.7%), yellow perch (22.3%), white sucker (11.9%), common carp (3.0%), northern pike (2.8%), and spottail shiner (1.6%), with the remaining nine species accounting for 1.0% of the biomass (Table 2-2). Common carp had the highest biomass in 2002, accounting for 49.8% of the average catch, followed by fathead minnow (29.3%), spottail shiner

(7.3%), black bullhead (4.6%), yellow perch (4.3%), and northern pike (3.4%). The remaining 10 species represented 1.3% of the biomass (Table 2-2).

Overall, nearly seven times as many fish were captured in 2001 compared to 2002 (Table 2-2). Most species declined in abundance in 2002. Common carp was the only species to show a significant increase between years, with nearly three times more caught in 2002 compared to 2001 (Table 2-2).

In 2001, species richness was high with values ranging from 5 – 11 species in each pond (Table 2-3). However, effective richness (N_2) was low, ranging from 1.1 (South Mackenzie Bay) to 3.0 (Wye's Pond). Evenness (expressed as a percent) was also relatively low, ranging from 12.2 in South Pitblado's Channel to 32.6 in Emile's Pothole (Table 2-3). In 2002, species richness, effective richness and evenness showed a wider range of values compared to 2001. Number of species per pond ranged from a high of 12 (Thompson's Bay, Mid-Blind Channel, South Mackenzie Bay) to no fish present (North Mackenzie Bay). Effective richness varied between 0 and 3.3 (Thompson's Bay), and evenness ranged from 0 to 65.0 (South Pitblado's Channel) (Table 2-3).

2.3.2 Fish Communities in Study Pond

Fish community composition and abundance varied both spatially and temporally among the 10 study ponds sampled in Delta Marsh. Ponds with a direct connection (*i.e.*, channel) to the main marsh had a more diverse fish community compared to ponds lacking a connection. High water levels on Lake Manitoba in the spring of 2001 led to correspondingly high water levels in Delta Marsh, flooding upland areas and permitting fish access to the four isolated ponds. As a result, these temporarily connected ponds had fish communities resembling ponds with direct connections to the main marsh (Tables 2-4, 2-5). However, declining water levels in 2002, along with pond manipulations, contributed to different fish communities among ponds. On average in 2002, isolated ponds had only two fish species present, diked ponds had six, screened ponds had nine, and blasted ponds and connected ponds had 11 species present (Tables 2-6, 2-7).

Average Beamish trap catch in 2001 ranged from a high of 8,170 fish in North Mackenzie Bay to a low of 8 fish in North School Bay (Table 2-4). In 2002, average trap catch ranged from a high of 6,760 in Thompson's Bay to zero fish in North Mackenzie Bay (Table 2-6). For most ponds, fewer fish were caught in 2002 compared to 2001. Exceptions were Thompson's Bay, North School Bay and Section 5 Bay. As noted previously, fish numbers from Thompson's Bay were inflated due to trap position in 2002.

In 2001, fatheads were the most abundant fish species in four study ponds (Tables 2-4, 2-5), including three temporarily connected ponds (Wye's Pond, North Mackenzie Bay and Emile's Pothole), and one connected pond (South Pitblado's Channel). Yellow perch were the dominant species in three connected ponds (Thompson's Bay, Mid-Blind Channel and Mad Woman Bay). Brook sticklebacks were the most abundant species in two study ponds, Section 5 Bay and South Mackenzie Bay (Tables 2-4, 2-5). In North School Bay, northern pike were the most abundant species. Overall, lower abundances of yellow perch and northern pike corresponded to ponds with higher fathead minnow and/or brook stickleback abundances. In 2002, fathead minnows were the most commonly captured species in half of the study ponds (North School Bay, Mid-Blind

Channel, Mad Woman Bay, South Pitblado's Channel, and South Mackenzie Bay) (Tables 2-6 and 2-7). Common carp were the dominant species in one screened pond (Section 5 Bay) and one blasted pond (Wye's Pond). Yellow perch dominated the catch at Thompson's Bay, and brook stickleback were the most abundant fish in Emile's Pothole. No fish were captured in North Mackenzie Bay in 2002 (Tables 2-6, 2-7).

In 2001, estimated biomass (wet weight) values ranged from a high of 16,832 g in North Mackenzie Bay to 90 g in Section 5 Bay (Table 2-8). Overall, with the exception of South Pitblado's Channel and South Mackenzie Bay, YOY of larger-sized fish species (*i.e.*, northern pike, yellow perch, black bullhead, common carp, and white sucker) accounted for the greatest proportion of the fish biomass of the average catch in 2001 (Tables 2-8, 2-9). Yellow perch accounted for the largest biomass in three connected ponds (Thompson's Bay, Mid-Blind Channel and Mad Woman Bay). Northern pike dominated the fish biomass in two connected ponds, accounting for 99.3% of the average catch in North School Bay and 57.1% in Section 5 Bay (Table 2-9). In Emile's Pothole, common carp represented 46.8% of biomass, and in North Mackenzie Bay black bullheads accounted for half of the fish biomass. In temporary connected Wye's Pond, white sucker had the greatest proportion of fish biomass accounting for 55.2% of the average catch (Table 2-9). Smaller species such as fathead minnows had the greatest biomass in a single connected pond (South Pitblado's Channel), while brook stickleback dominated the biomass in temporarily connected South Mackenzie Bay (Tables 2-8, 2-9).

Total fish biomass ranged from 19,790 g in Thompson's Bay to 0 g in North Mackenzie Bay in 2002 (Table 2-10). Fathead minnows dominated the fish biomass in half of the study ponds in 2002, including one connected pond (North School Bay), both diked ponds (Mad Woman Bay and South Pitblado's Channel), one blasted pond (South Mackenzie Bay), and one isolated pond (Emile's Pothole) (Table 2-11). In the remaining five ponds (with fish present), fathead minnows had the second greatest proportion of fish biomass in Section 5 Bay and Mid Blind Channel. Common carp dominated the biomass in one screened pond (Section 5 Bay) and one blasted pond (Wye's Pond). Spottail shiners represented the largest proportion of fish biomass in Mid-Blind Channel (Table 2-11). Yellow perch were prominent in a single connected pond, accounting for 48.2% of the fish biomass in Thompson's Bay (Tables 2-10, 2-11).

2.4 Discussion

2.4.1 The Fish Community of Peripheral Ponds in Delta Marsh

Connectedness is a key factor determining fish utilization of many aquatic habitats, including coastal wetlands (Herdendorf et al. 1986; Jude and Pappas 1992; Lougheed and Chow-Fraser 2001), small lakes (Tonn and Magnuson 1982), swamps (Carlson and Duever 1978 in: Clark 1978), mangroves and salt marshes (Clark 1978). Channels and temporary surface water connections link water bodies and function as corridors for movement (Clark 1978; Herdendorf 1992; Jude and Pappas 1992; Brazner et al. 2000, 2001; Evelsizer 2001; Lougheed and Chow-Fraser 2001). For example, small isolated northern Minnesota lakes commonly experience hypoxia during winter. Lakes without direct connection to a larger permanent water body have low fish diversity and a community composed of tolerant smaller species, such as mudminnows and certain cyprinids. Alternatively, the presence of "centrarchid-*Esox*" assemblages in small lakes is dependent upon the degree of connection to a larger water body, where fish can use adjoining streams as conduits to avoid low oxygen levels (Tonn and Magnuson 1982).

Duration and timing of connections (*i.e.,* flooding) (Clark 1978; Liston and Chubb 1985; Snodgrass et al. 1996; Poizat and Crivelli 1997; Baber et al. 2002) are also critical to fish movement. The longer and more frequent the connection to an adjacent water body, the greater the opportunity for fish access (Clark 1978; Poizat and Crivelli 1997). Fish in coastal wetland habitats have ready access to an adjoining larger water body and can adjust their use of coastal wetlands to avoid detrimental environmental conditions such as increasing temperatures, low dissolved oxygen, and decreasing water levels (Baber et al. 2002).

Connectivity within coastal wetlands is dictated to a large degree by hydrological factors of adjoining lakes (Whillans 1996; Keough et al. 1999). Water levels on large lakes fluctuate among years and seasonally, and also fluctuate daily due to seiches and wind set-up (Einarsson and Lowe 1968; Keough et al. 1999). When strong winds blow from the north, water levels rise at the south end of Lake Manitoba and within Delta Marsh. High spring water levels or wind set-up can result in higher water levels in the marsh. High water levels in the marsh result in overland flooding and provide opportunities for fish to gain access to previously inaccessible habitats. As a consequence, ponds that would not normally support fish, or only a limited number of species, were able to support a diverse community. Such was the case in 2001, when temporarily connected ponds had high numbers and diversity of YOY fish. However, as water levels fell in the summer of 2001, these ponds became isolated again. Samples from 2002 indicated that most fish species did not survive winter conditions within the ponds, and in the case of North Mackenzie Bay, no fish survived the winter.

My studies demonstrate a gradient in fish habitat and communities within the peripheral ponds of Delta Marsh. This gradient is controlled by the degree of connectivity with the main marsh, which is determined by the presence or absence of a physical connection and water levels. Connected ponds, those with a permanent connection to the main marsh (*i.e.*, naturally connected ponds or ponds with newly formed channels), generally had a diverse, mixed-species assemblage, including yellow perch, northern pike, bullheads, and many cyprinids. These fish can move freely back and forth between these ponds and the marsh, and use these habitats on a seasonal basis to feed, spawn and/or as a nursery. At Delta Marsh, the entire marsh either freezes to the bottom or becomes anoxic in winter (Gordon Goldsborough, University of Manitoba, unpublished data). Fish in ponds with permanent channels must migrate in the fall to neighbouring Lake Manitoba to overwinter if they are to survive (Gee 1975; Lapointe 1986).

When water levels rise in the marsh, upland areas are flooded and isolated ponds without a physical connection become connected to the main marsh. These ponds then have fish communities that are similar to ponds with permanent connections (Figure 2-2). This was the case in my temporarily connected ponds in 2001. However, as water levels fell through the summer of 2001, these ponds again became isolated. Most fish species were unable to survive the winter period. In those ponds with sufficient water, a reduced fish assemblage of tolerant species was found. Fathead minnows and brook sticklebacks have morphological, physiological and behavioural adaptations (Klinger et al. 1982) to allow these species to survive in near shore areas during winter (Suthers

1982; Candace Parks, personal observation). Other species such as central mudminnows may also be present in low numbers. These species are commonly found in isolated ponds of Delta Marsh, and are also characteristic of the prairie potholes (Peterka 1989) and winterkill lakes (Magnuson et al. 1985).

In those ponds where the water column becomes anoxic or freezes completely, no fish will be found. This was likely the situation in North Mackenzie Bay in 2002, when no fish were present within the pond. Fish will only become re-established in these ponds when water levels increase or artificial channels are created.

Numbers of fish captured were much lower in 2002 compared to 2001, and almost all species showed a decline in abundance. Reasons for this difference are not known, but may be related to changing water levels on the marsh. As noted above, water levels were higher in 2001 compared to 2002 and flooded upland areas around the marsh. Rising water levels have been reported to contribute to increased spawning for many fish species (Clark 1978), including northern pike (Howard and Thomas 1970). Liston and Chubb (1985) describe how high spring water levels contributed to increased spawning and nursery success of cyprinids, escocids, centrarchids, and percids in various aquatic habitats. Similarly, in Delta Marsh high spring water levels flooded terrestrial vegetation and likely led to increased spawning for many species, and subsequent increases in YOY. Low water levels in 2002 did not flood surrounding uplands, reduced spawning activity and the production of YOY.

Fish screens are currently being used on several Great Lakes coastal wetlands to prevent access by adult common carp, yet still allow access by most other fish species (Wilcox and Whillans 1999). However, there are concerns that screens reduce access by larger fish species, such northern pike (Johnson et al. 1997). The effect of screens on fish passage is dependent on the space or gap that is provided between bars; the smaller the space, the smaller the fish that are allowed through. French et al. (1999) reported that pike measuring 49.4 to 70.0 cm total length could readily pass through vertical bar screens with a 5 cm gap width. Usage of screens with this gap width have been found to successfully prevent access by adult common carp and aid in the restoration of several coastal marshes, such as Cootes Paradise Marsh and Metzger Marsh (Wilcox and Whillans 1999). At Cootes Paradise Marsh, fish screens have successfully excluded large common carp (>30 cm) and resulted in increased water clarity and aquatic macrophytes (Wilcox and Whillans 1999). However, the impact on overall fish community composition and abundance is not well understood. In both Cootes Paradise Marsh and Metzger Marsh, larger fish, unable to pass through the screens, were manually sorted. Adult common carp are returned to the lake and other species are allowed to pass into the marsh. In the case of a much larger coastal wetland such as Delta Marsh, with four connections to Lake Manitoba, manual sorting of adult fish would be logistically and economically impractical. The effect of screens, without manually sorting adult fish, has not been assessed.

In Delta Marsh, fish screens installed at Mid-Blind Channel and Section 5 Bay attempted to exclude adult common carp, while permitting entrance to native fish species. At Mid-Blind Channel, screens successfully excluded adult common carp and at the same time resulted in increased native fish diversity compared to the previous year. As expected, there were improvements in aquatic macrophyte growth and water clarity

(Stacy Hnatiuk, unpublished data; Candace Parks, personal observation). The number of fish species in Mid-Blind Channel increased in 2002, from nine to 12 species, but there was an overall reduction in the total number of fish captured, with major reductions in black bullheads, yellow perch and white suckers. Reasons for these reductions are not known, but may be related to lower water levels and reduced spawning effort in 2002 as noted above. Also, whenever the current was flowing out of Mid-Blind Channel into the main marsh, large numbers of adult common carp congregated in front of the screened culverts, and this may have created a biological barrier, preventing other fish species from moving into the pond (Figures 2-3, 2-4). Overall, the screen itself did not seem to hinder migration patterns of native fish into or out of Mid-Blind Channel and resulted in an overall improvement in marsh habitat for native fish species, aquatic invertebrates and waterbirds (Candace Parks, personal observation).

At Section 5 Bay, the number of fish species captured following installation of the conduit fence decreased from nine in 2001 to six in 2002. However, the abundance of most species increased in 2002, and the average number of fish per Beamish trap increased by almost 30 times. Unlike Mid-Blind Channel, the dominant fish species was YOY common carp. Although no adult common carp were directly observed within the pond, several may have breached the conduit fence and this may explain the large number of YOY in this pond. Two dead common carp were found on top of the screen, indicating that fish tried to jump over the fence during wind set-ups or storm events. It is unclear why a reduction in the fish diversity was observed in Section 5 Bay. Dewatering from mid-July to the end of August may have led to increased water temperatures, lowering oxygen levels and an increase in avian predation, which may not have been advantageous for many fish species (Candace Parks, personal observation).

2.4.2 The Fish Community of Delta Marsh

This study captured 16 fish species. Fathead minnows were numerically dominant and also represented a significant part of the overall biomass of the fish community of Delta Marsh. Past studies confirm fathead minnows are abundant in the marsh (Kiers and Hann 1995; Goodyear 1996; Evelsizer 2001) (Table 2-12), yet the current study is the first to estimate approximate biomass of Delta Marsh's small-sized fish community. Approximate biomass is an important indicator of fish abundance. For example, nearly three times as many common carp were captured in 2002 than in 2001. Although the majority of larger-growing species (*i.e.*, white sucker, yellow perch, common carp, northern pike and black bullheads) were captured in lower frequencies, they contributed to a large proportion of the overall fish biomass in ponds in 2002 (Table 2-2).

Overall, Delta Marsh supports numerous rough, prey and commercially important fish species. Sticklebacks, white suckers, freshwater drum, common carp, and emerald and spottail shiners (Schneider 1983; Kiers and Hann 1995; Goodyear 1996) (Candace Parks, personal observation) are vital prey items for predatory lake fish (Scott and Crossman 1998), and are commercially profitable as bait fish (Janusz and O'Connor 1985). Although common carp are not usually considered valuable in North America, their roe is gaining popularity for human consumption (Dennis Geisler, Freshwater Fish Marketing Corporation, personal communication, 2005).

Delta Marsh provides spawning and nursery habitat for several commercially important fish species, including yellow perch, northern pike and white sucker. This is contrary to earlier reports that Delta Marsh did not provide habitat for economically important species (Technical Committee for Development of the Delta Marsh 1968). Yellow perch are a dominant species in almost all studies listed in Table 2-12, and they were the third most abundant species captured entering the marsh by Lapointe (1986) (Table 2-13). Lake Manitoba has a significant yellow perch fishery (Dennis Geisler, Freshwater Fish Marketing Corporation, personal communication, 2005) and further comprehensive studies are needed to determine the relative importance of Delta Marsh as a source of commercially important species, such as yellow perch.

Overall, the small fish community of Delta Marsh appears to have remained relatively unchanged since 1982 (Table 2-12). Schneider (1983) captured a total of 15 species using Beamish traps, whereas the current study captured 16 species. The species list for both studies are similar, with trout perch and freshwater drum missing from Schneider's list, and log perch missing from the current study

Although 17 species were captured in the studies listed in Table 2-12, there are an additional 13 species present in Delta Marsh that were not caught in the study. These include: goldeye (*Hiodon alosoides*), mooneye (*Hiodon tergisus*), quillback (*Carpiodes cyprinus*), bigmouth buffalo (*Ictiobus cyprinellus*), silver redhorse sucker (*Moxostoma anisurum*), shorthead redhorse sucker (*Moxostoma macrolepidotum*), channel catfish (*Ictalurus punctatus*), tadpole madtom (*Noturus gyrinus*), cisco (*Coregonus artedi*), sauger (*Sander canadense*), walleye (*Sander vitreum*), burbot (*Lota lota*), and rock bass (*Ambloplites rupestris*) (Suthers 1982, Stewart et al. 1985, Lapointe 1986, Dale

Wrubleski, Ducks Unlimited Canada, personal communication, 2005). These species may not have been captured for several reasons. First, many are rare in the marsh (*e.g.*, goldeye, mooneye, channel catfish, tadpole madtom, silver redhorse, shorthead redhorse) (Dale Wrubleski, unpublished data) and are unlikely to have been captured in studies that sampled a relatively small portion of the marsh over a short time period. Second, many of these species may not spawn in Delta Marsh (*e.g.*, cisco, sauger, walleye), so it is unlikely that YOY would have been caught in Beamish and/or minnow traps. Third, several fish species (*e.g.*, burbot) are likely using the marsh early in the year (Lapointe 1986), prior to the start of the studies listed in Table 2-12. Fourth, while some of these species may be using the marsh, they may be using the larger bays of Delta Marsh, particularly on the east side, and may not be using smaller, shallower habitats, such as peripheral ponds. And fifth, several of the studies listed in Table 2-12 sampled with minnow traps only. These traps tend to be size selective and have other sampling issues (Suthers 1982; Layman and Smith 2001; Candace Parks personal observation). For example, those studies that used minnow traps tended to capture more brook sticklebacks than did Beamish traps (Table 2-12), suggesting that this sampling method may inflate their abundance.

2.4.3 The Fish Communities of Coastal Wetlands

The Delta Marsh fish assemblage closely resembled the 25 species inhabiting neighbouring Netley-Libau Marsh (Janusz and O'Connor 1985). Netley-Libau is the largest freshwater coastal wetland in Manitoba, representing a transition zone between Lake Winnipeg and the Red River. The fish communities of the two marshes are comparable, despite the notable absence of fathead minnows in Netley-Libau Marsh.

The absence of such a prominent species is though to be due to an oversight in data recording or a sampling error (Ken Stewart, University of Manitoba, personal communication, 2004). Otherwise species such as northern pike, yellow perch, sauger, and various shiners can be found in Delta Marsh (Tables 2-12, 2-13) and Netley-Libau Marsh (Janusz and O'Connor 1985) in spring and summer.

Jude and Pappas (1992) summarized the results of numerous fisheries investigations and found a total of 47 species (including 15 exotics) from 14 families inhabiting nine Great Lakes coastal wetlands. Cyprinids were the dominant family representing 15 species, followed by centrarchids, percids, catostomids, and ictalurids (Jude and Pappas 1992). In comparison to Great Lake coastal wetlands, Delta Marsh has lower species richness but a much smaller incidence of exotic species. As with many Great Lakes coastal wetlands, Delta Marsh is also dominated by pollution tolerant benthic species (*e.g.*, bullheads, white sucker, and common carp). Yellow perch, spottail shiners, white sucker, common carp and emerald shiners were commonly shared species between many coastal wetlands of the Laurentian Great Lake and Delta Marsh.

Fathead minnows are abundant in Delta Marsh, but are not an important component to the fish community in the Laurentian Great Lakes or their adjacent coastal wetlands. As summarized by Jude and Pappas (1992), fathead minnows were ranked rare or absent in 81.3% of the examined areas, common in 12.5% of the areas and considered abundant in only three marshes around Toronto. The predominance of fathead minnows in Delta Marsh and the surrounding prairie pothole region is likely attributable to their ability to tolerate harsh winter conditions. Fathead minnows are tolerant to low dissolved oxygen concentrations (0.5 mg/l), high salinity concentrations and can spawn several times a year (Peterka 1989).

Among the fisheries studies done on Laurentian Great Lakes coastal wetlands, Delta Marsh compared favourably to Allouez Bay, a 148 ha coastal wetland of Lake Superior (Tanner et al. 2004). Eleven of the 16 species identified from Allouez Bay, were captured in Delta Marsh during the current study or are known to occur in Delta Marsh. Like Allouez Bay, Delta Marsh is outside the geographic range of many salmonids, clupeids, and some centrarchids species [*e.g.*, pumpkinseed (*Lepomis gibbosus*) and bluegill (*Lepomis macrochirus*)] which are characteristically found in more southerly coastal marshes of the Laurentian Great Lakes.

In spite of the fact that fish communities in Laurentian Great Lakes coastal wetlands have been studied more intensively, there are apparently no previous studies that have observed comparable relationships between fish compositions in isolated, connected and temporarily connected ponds within these coastal wetlands. The majority of fisheries studies in the Laurentian Great Lakes have focused their attention on coastal wetland–lake dynamics, with little attention paid to examining the importance of internal areas within larger coastal wetlands, and their implications for fish inhabiting the larger coastal wetland and/or the adjacent larger lake habitats. Consequently, the current study is the only study that has examined these internal relationships between peripheral ponds and the adjacent coastal wetland.

2.4.4 Summary and Future Research Needs

Hydrological connectedness is important to sustaining fish communities in freshwater coastal wetlands. Connectedness is either a function of natural or artificial channels, or is determined by water levels, and can have a pronounced impact on fish community structure of Delta Marsh and other coastal wetlands. Increased water levels caused temporary connections, creating opportunities for fish passage into previously isolated ponds. Water level changes at critical periods can either increase or decrease accessibility to spawning, nursery and feeding sites (Liston and Chubb 1985; Poizat and Crivelli 1997). The longer and more frequent the connection, the greater the chance fish will populate a habitat and contribute to increased species richness (Poizat and Crivelli 1997). At Delta Marsh, increased water levels in 2001 had a pronounced effect on fish numbers and diversity. Flooding of isolated ponds (temporarily connected ponds) provided fish with access to habitats with abundant prey items (*e.g.*, invertebrates and tadpoles). A combination of these factors could have attributed to increased reproductive success of numerous species.

Alternatively, declining water levels or diking provided a barrier to fish movement, and exposed the resident fish to unfavourable environmental conditions. Like shallow winter-kill lakes, isolated ponds had simple communities, composed of species that are able to survive severe conditions such as low dissolved oxygen levels. These communities were composed of brook sticklebacks, fathead minnows, and at times central mudminnows. These species have numerous behavioural, physiological and morphological adaptations that enable them to survive in dissolved oxygen levels less than 1.0 ppm (Klinger et al. 1982).

Overall, the knowledge of Delta Marsh's fish assemblage remains low. Although many species move into the marsh in the spring (Lapointe 1986), little information is available on their spawning and production within the marsh. The majority of past studies have been conducted in easily accessible, site-specific areas. Additional sampling that encompasses a larger area of the marsh is required to obtain a more thorough picture of the fish community of the marsh. For example, the composition of the fish fauna inhabiting large wind-swept bays, typically considered harsh for fish due to their less protective nature and lower abundance of aquatic vegetation, is unknown. Comprehensive fisheries studies are required throughout the year to gain an understanding of what fish species utilize the marsh in winter and early spring. Past fish studies have largely focused on sampling in summer, biasing results for winter spawning fish such as burbot and early spawning species such as walleye, sauger, northern pike, and white sucker, which may utilize the marsh during or immediately after ice-out (Lapointe 1986). Expansion of studies monitoring fish movements between Lake Manitoba and Delta Marsh would be beneficial in determining the seasonal and temporal movements of fish. This would also provide better information on the contribution of Delta Marsh to the recreational and commercial fisheries of Lake Manitoba. It is also imperative to use a variety of sampling gear types to obtain a more comprehensive description of the fish community. Minnow traps were ineffective, thus a combination of sampling gears similar to Beamish traps, which can be modified for shallow water conditions, are preferred.

Overall, Delta Marsh and other neighbouring coastal wetlands have been under-valued for fish spawning, feeding and nursery areas. Past reports have claimed Delta Marsh is not an important fish habitat, however this study showed large numbers of fish, including economically important and forage fish species, opportunistically use Delta Marsh during the summer months. Delta Marsh has received international, provincial and regional recognition because of is importance as waterbird habitat. My work would suggest that Delta Marsh also deserves high priority for protection and rehabilitation because of its importance as fish habitat.

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TABLES AND FIGURES

Table 2-1. Families and species of fish captured in Beamish traps set in Delta Marsh, summer 2001 and 2002.

Scientific Name (Authority)		Common name
CYPRINIDAE		
	Pimephales promelas Rafinesque	Fathead minnow
	Cyprinus carpio L.	Common carp
	Notropis hudsonius (Clinton)	Spottail shiner
	Notropis atherinoides Rafinesque	Emerald shiner
GASTEROSTEIDAE		
	Culaea inconstans (Kirtland)	Brook stickleback
	Pungitius pungitius L.	Ninespine stickleback
ICTALURIDAE		
	Ictalurus melas (Rafinesque)	Black bullhead
	Ictalurus nebulosus (Lesueur)	Brown bullhead
ESOCIDAE		
	Esox lucius L.	Northern pike
PERCOPSIDAE		
	Percopsis omiscomaycus (Walbaum)	Trout-perch
PERCIDAE		
	Perca flavescens (Mitchill)	Yellow perch
	Etheostoma nigrum Rafinesque	Johnny darter
	Etheostoma exile (Girard)	Iowa darter
CATOSTOMIDAE		
	Catostomus commersoni (Lacepède)	White sucker
UMBRIDAE		
	Umbra limi (Kirtland)	Central mudminnow
SCIAENIDAE		
	Aplodinotus grunniens Rafinesque	Freshwater drum

Table 2-2. Mean number (number/trap) and biomass (g/trap) of fish captured per Beamish trap in 10 study ponds in Delta Marsh between early June and late August in 2001 and 2002.

	Numbers				Biomass			
	2001		2002		2001		2002	
Species	**Mean**	**%**	**Mean**	**%**	**Mean**	**%**	**Mean**	**%**
fathead minnow	725.4	46.9	103.9	46.1	1,233.2	33.8	176.6	29.3
common carp	28.7	1.9	78.9	35.0	109.1	3.0	299.8	49.8
spottail shiner	10.4	0.7	8.0	3.5	57.2	1.6	44.0	7.3
emerald shiner	2.4	0.2	0.4	0.2	2.9	0.1	0.5	0.1
brook stickleback	88.8	5.7	14.2	6.3	26.6	0.7	4.3	0.7
ninespine stickleback	0.0	0.0	0.1	t	0.0	0.0	0.1	t
black bullhead	345.7	22.3	11.0	4.9	864.3	23.7	27.5	4.6
brown bullhead	0.0	0.0	0.0	0.0	0.0	0.0	t	t
northern pike	1.0	0.1	0.2	0.1	102.5	2.8	20.5	3.4
trout-perch	0.3	t	t	t	t	t	t	t
yellow perch	239.3	15.5	7.6	3.4	813.6	22.3	25.8	4.3
johnny darter1	5.8	0.4	0.4	0.2	5.2	0.1	0.4	0.1
Iowa darter	1.2	0.1	0.3	0.1	1.1	t	0.3	t
white sucker	98.9	6.4	0.4	0.2	435.2	11.9	1.8	0.3
central mudminnow	t2	t	0.1	t	0.0	0.0	0.0	0.0
freshwater drum	0.1	t	0.2	0.1	0.3	t	0.5	0.1
Totals	1,548.0	100.0	225.6	100.0	3,651.0	100.0	602.0	100.0

1 - weights for lengths of johnny darter captured at Delta Marsh were calculated using the Iowa darter length/weight relationship from Delta Marsh
2 - t = trace (<0.1 or 0.1%)
Note that 2002 data do not include Thompson's Bay. Due to sampling problems with this pond, numbers of fish captured were highly inflated and were not used to determine annual means (see text for further explanation).

Table 2-3. Species richness, effective richness and evenness of fish communities sampled in 10 study ponds in Delta Marsh between early June and late August in 2001 and 2002.

	2001				2002			
Ponds	**Treatment**	**Species Richness** ***s***	**Effective Richness** N_2	**Evenness (ER/s) %** E_3	**Treatment**	**Species Richness** ***s***	**Effective Richness** N_2	**Evenness (ER/s) %** E_3
Thompson's Bay	connected	9	1.4	15.1	connected[1]	12	3.3	27.8
North School Bay	connected	5	1.5	30.2	connected	11	1.8	16.4
Section 5 Bay	connected	9	2.9	31.8	screened	6	1.6	26.9
Mid-Blind Channel	connected	9	1.6	17.4	screened	12	2.5	21.2
Mad Woman Bay	connected	8	2.1	25.8	diked	9	1.3	14.0
South Pitblado's Channel	connected	10	1.2	12.2	diked	3	1.9	65.0
South Mackenzie Bay	temporarily connected	5	1.1	21.1	blasted	12	2.0	16.9
Wye's Pond	temporarily connected	11	3.0	27.2	blasted	10	3.1	31.3
North Mackenzie Bay	temporarily connected	7	2.0	28.9	isolated	0	0.0	0.0
Emile's Pothole	temporarily connected	9	2.9	32.6	isolated	4	2.0	50.2

1 - due to low water levels, this pond could not representatively sampled (see text for details)

Table 2-4. Mean number (number/trap) of fish captured in Beamish traps set in 10 study ponds in Delta Marsh between early June and late August, 2001.

	CONNECTED						TEMPORARILY CONNECTED			
Species	**TB**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	16.0	0.0	0.3	180.0	0.5	1,045.8	7.8	975.1	4,656.3	372.5
common carp	0.5	0.8	6.8	1.8	1.8	12.3	2.5	1.5	66.8	192.9
spottail shiner	34.3	0.0	0.0	24.8	6.3	2.0	0.0	21.5	0.0	15.3
emerald shiner	4.3	0.0	0.0	0.0	0.8	0.0	0.0	19.3	0.0	0.0
brook stickleback	6.8	0.3	10.0	10.8	1.0	58.8	454.8	83.0	65.3	197.0
ninespine stickleback	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
black bullhead	7.5	0.3	0.5	29.0	19.3	10.5	0.0	18.3	3,370.8	0.8
brown bullhead	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
northern pike	0.3	6.3	0.5	0.0	0.0	0.5	0.8	0.0	2.0	0.0
trout-perch	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0	0.0	0.0
yellow perch	428.5	0.3	2.0	1628.4	56.3	13.8	0.0	251.3	1.3	11.0
johnny darter	0.0	0.0	0.0	0.0	0.0	0.3	1.5	48.4	7.8	0.5
Iowa darter	0.0	0.0	0.3	0.8	0.0	0.5	0.0	10.0	0.0	0.8
white sucker	4.0	0.0	0.3	187.8	0.3	12.3	0.0	780.5	0.0	3.8
central mudminnow	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
freshwater drum	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0
Totals	502.0	7.8	20.8	2,063.6	86.0	1,156.5	467.3	2,211.8	8,170.1	794.4

Table 2-5. Relative frequency of occurrence (expressed as a percent) of fish (number/trap) captured in Beamish traps set in 10 study ponds in Delta Marsh between early June and late August, 2001.

	CONNECTED						TEMPORARILY CONNECTED			
Species	**TB**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	3.2	0.0	1.2	8.7	0.6	90.4	1.7	44.1	57.0	46.9
common carp	0.1	9.7	32.5	0.1	2.0	1.1	0.5	0.1	0.8	24.3
spottail shiner	6.8	0.0	0.0	1.2	7.3	0.2	0.0	1.0	0.0	1.9
emerald shiner	0.8	0.0	0.0	0.0	0.9	0.0	0.0	0.9	0.0	0.0
brook stickleback	1.3	3.2	48.2	0.5	1.2	5.1	97.3	3.8	0.8	24.8
ninespine stickleback	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
black bullhead	1.5	3.2	2.4	1.4	22.4	0.9	0.0	0.8	41.3	0.1
brown bullhead	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
northern pike	0.0	80.6	2.4	0.0	0.0	t[1]	0.2	0.0	t	0.0
trout-perch	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
yellow perch	85.4	3.2	9.6	78.9	65.4	1.2	0.0	11.4	t	1.4
johnny darter	0.0	0.0	0.0	0.0	0.0	t	0.3	2.2	0.1	0.1
Iowa darter	0.0	0.0	1.2	0.0	t	t	0.0	0.5	0.0	0.1
white sucker	0.8	0.0	1.2	9.1	0.3	1.1	0.0	35.3	0.0	0.5
central mudminnow	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
freshwater drum	0.0	0.0	0.0	0.0	t	0.0	0.0	0.0	0.0	0.0
Totals	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

1 - t = trace 0.1%

Table 2-6. Mean number (number/trap) of fish captured in Beamish traps set in 10 study ponds in Delta Marsh between early June and late August, 2002.

	CONNECTED		SCREENED		DIKED		BLASTED		ISOLATED	
Species	**TB[1]**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	2,096.8	359.9	98.3	134.8	66.6	94.7	105.5	63.3	0.0	12.2
common carp	1,084.8	109.1	455.0	3.3	0.0	0.0	28.1	114.4	0.0	0.0
spottail shiner	245.2	17.2	0.5	53.7	0.2	0.0	0.4	0.1	0.0	0.0
emerald shiner	108.8	0.4	0.0	0.7	0.1	0.0	2.1	0.3	0.0	0.0
brook stickleback	0.6	10.3	16.3	7.7	5.7	64.5	7.2	0.0	0.0	15.8
ninespine stickleback	0.0	0.4	0.0	0.5	0.0	0.0	0.3	0.0	0.0	0.0
black bullhead	417.8	2.4	0.0	2.1	0.8	0.0	9.1	84.6	0.0	0.0
brown bullhead	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
northern pike	0.4	0.1	0.0	0.1	0.5	0.8	0.2	0.0	0.0	0.0
trout-perch	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
yellow perch	2,804.4	5.1	19.5	33.4	0.5	0.0	0.5	9.3	0.0	0.0
johnny darter	0.0	t[2]	0.0	1.3	0.0	0.0	1.0	1.0	0.0	0.0
Iowa darter	0.0	0.0	0.0	0.2	0.0	0.0	0.0	3.0	0.0	0.0
white sucker	0.6	0.4	2.3	0.0	0.1	0.0	0.7	0.1	0.0	0.0
central mudminnow	0.2	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.2
freshwater drum	0.4	0.0	0.0	0.0	0.0	0.0	1.3	t	0.0	0.1
Totals	6,760.2	505.3	591.8	238.0	75.1	159.9	156.4	275.9	0.0	28.3

1 - due to low water levels, this pond could not representatively sampled (see text for details)
2 - t - < 0.1

Table 2-7. Relative frequency of occurrence (expressed as a percent) of fish (number/trap) captured in Beamish traps set in 10 study ponds in Delta Marsh between early June and late August, 2002.

	CONNECTED		SCREENED		DIKED		BLASTED		ISOLATED	
Species	**TB[1]**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	31.0	71.2	16.6	56.6	88.7	59.2	67.5	22.9	0.0	43.1
common carp	16.0	21.6	76.9	1.4	0.0	0.0	18.0	41.5	0.0	0.0
spottail shiner	3.6	3.4	0.1	22.6	0.3	0.0	0.3	t[2]	0.0	0.0
emerald shiner	1.6	0.1	0.0	0.3	0.1	0.0	1.3	0.1	0.0	0.0
brook stickleback	t	2.0	2.7	3.2	7.6	40.3	4.6	0.0	0.0	55.8
ninespine stickleback	0.0	0.1	0.0	0.2	0.0	0.0	0.2	0.0	0.0	0.0
black bullhead	6.2	0.5	0.0	0.9	1.1	0.0	5.8	30.7	0.0	0.0
brown bullhead	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
northern pike	t	t	0.0	t	0.7	0.5	0.1	0.0	0.0	0.0
trout-perch	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
yellow perch	41.5	1.0	3.3	14.0	0.7	0.0	0.3	3.4	0.0	0.0
johnny darter	0.0	t	0.0	0.5	0.0	0.0	0.6	0.4	0.0	0.0
Iowa darter	0.0	0.0	0.0	0.1	0.0	0.0	0.0	1.1	0.0	0.0
white sucker	t	0.1	0.4	0.0	0.1	0.0	0.4	t	0.0	0.0
central mudminnow	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.7
freshwater drum	t	0.0	0.0	0.0	0.0	0.0	0.8	t	0.0	0.4
Totals	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	0.0	100.0

1 - due to low water levels, this pond could not representatively sampled (see text for details).
2 - t - < 0.1%

Table 2-8. Approximate biomass (g/trap) calculated from an average weight for each fish species multiplied by the seasonal mean for each species captured in study ponds in Delta Marsh between early June and late August, 2001.

	CONNECTED						TEMPORARILY CONNECTED			
Species	**TB**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	27.2	0.0	0.4	306.0	0.9	1,777.8	13.2	1,657.7	7,915.6	633.3
common carp	1.9	2.9	25.7	6.7	6.7	46.6	9.5	5.7	253.7	732.9
spottail shiner	188.4	0.0	0.0	136.1	34.4	11.0	0.0	118.3	0.0	83.9
emerald shiner	5.1	0.0	0.0	0.0	0.9	0.0	0.0	23.1	0.0	0.0
brook stickleback	2.0	0.1	3.0	3.2	0.3	17.6	136.4	24.9	19.6	59.1
ninespine stickleback	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
black bullhead	18.8	0.6	1.3	72.5	48.1	26.3	0.0	45.6	8,427.1	1.9
brown bullhead	-	-	-	-	-	-	-	-	-	-
northern pike	25.6	640.6	51.3	0.0	0.0	51.3	76.9	0.0	205.0	0.0
trout-perch	-	-	-	-	-	-	-	-	-	-
yellow perch	1,456.9	0.9	6.8	5,536.5	191.3	46.8	0.0	854.3	4.3	37.4
johnny darter	0.0	0.0	0.0	0.0	0.0	0.2	1.4	43.5	7.0	0.5
Iowa darter	0.0	0.0	0.2	0.7	0.0	0.5	0.0	9.0	0.0	0.7
white sucker	17.6	0.0	1.1	826.1	1.1	53.9	0.0	3,434.2	0.0	16.5
central mudminnow	-	-	-	-	-	-	-	-	-	-
freshwater drum	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0
Totals	1,743.5	645.0	89.7	6,889.0	283.6	2,031.8	237.3	6,216.3	16,832.2	1,566.1

Table 2-9. Approximate biomass (g/trap; expressed as a percent) calculated from an average weight for each fish species multiplied by the seasonal mean for each species captured in study ponds in Delta Marsh between early June and late August, 2001.

	CONNECTED						TEMPORARILY CONNECTED			
Species	**TB**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	1.6	0.0	0.5	4.4	0.3	87.5	5.6	26.7	47.0	40.4
common carp	0.1	0.4	28.6	0.1	2.3	2.3	4.0	0.1	1.5	46.8
spottail shiner	10.8	0.0	0.0	2.0	12.1	0.5	0.0	1.9	0.0	5.4
emerald shiner	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.4	0.0	0.0
brook stickleback	0.1	0.0	3.3	0.0	0.1	0.9	57.5	0.4	0.1	3.8
ninespine stickleback	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
black bullhead	1.1	0.1	1.4	1.1	17.0	1.3	0.0	0.7	50.1	0.1
brown bullhead	-	-	-	-	-	-	-	-	-	-
northern pike	1.5	99.3	57.1	0.0	0.0	2.5	32.4	0.0	1.2	0.0
trout-perch	-	-	-	-	-	-	-	-	-	-
yellow perch	83.6	0.1	7.6	80.4	67.4	2.3	0.0	13.7	0.0	2.4
johnny darter	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.7	0.0	0.0
Iowa darter	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.1	0.0	0.0
white sucker	1.0	0.0	1.2	12.0	0.4	2.7	0.0	55.2	0.0	1.1
central mudminnow	-	-	-	-	-	-	-	-	-	-
freshwater drum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Totals	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

Table 2-10. Approximate biomass (g/trap) calculated from an average weight for each fish species multiplied by the seasonal mean for each species captured in study ponds in Delta Marsh between early June and late August, 2002.

	CONNECTED		SCREENED		DIKED		BLASTED		ISOLATED	
Species	**TB[1]**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	3,564.6	611.9	167.0	229.2	113.2	160.9	179.4	107.5	0.0	20.7
common carp	4,122.2	414.5	1,729.0	12.5	0.0	0.0	106.8	434.8	0.0	0.0
spottail shiner	1,348.6	94.4	2.8	295.4	1.1	0.0	2.2	0.5	0.0	0.0
emerald shiner	130.6	0.5	0.0	0.8	0.1	0.0	2.5	0.3	0.0	0.0
brook stickleback	0.2	3.1	4.9	2.3	1.7	19.3	2.2	0.0	0.0	4.7
ninespine stickleback	0.0	0.2	0.0	0.3	0.0	0.0	0.2	0.0	0.0	0.0
black bullhead	1,044.5	6.0	0.0	5.3	2.0	0.0	22.8	211.5	0.0	0.0
brown bullhead	-	-	-	-	-	-	-	-	-	-
northern pike	41.0	8.5	0.0	10.3	51.3	76.9	20.5	0.0	0.0	0.0
trout-perch	-	-	-	-	-	-	-	-	-	-
yellow perch	9,535.0	17.3	66.3	113.6	1.7	0.0	1.7	31.5	0.0	0.0
johnny darter	0.0	0.0	0.0	1.2	0.0	0.0	0.9	0.9	0.0	0.0
Iowa darter	0.0	0.0	0.0	0.2	0.0	0.0	0.0	2.6	0.0	0.0
white sucker	2.6	1.8	9.9	0.0	0.4	0.0	3.1	0.4	0.0	0.0
central mudminnow	-	-	-	-	-	-	-	-	-	-
freshwater drum	1.0	0.0	0.0	0.0	0.0	0.0	3.3	0.1	0.0	0.3
Totals	19,790.2	1,158.3	1,979.9	670.9	171.5	257.1	345.3	789.9	0.0	25.7

1 - due to low water levels, this pond could not representatively sampled (see text for details)

Table 2-11. Approximate biomass (g/trap; expressed as a percent) calculated from an average weight for each fish species multiplied by the seasonal mean for each species captured in study ponds in Delta Marsh between early June and late August, 2002.

	CONNECTED		SCREENED		DIKED		BLASTED		ISOLATED	
Species	**TB**[1]	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	18.0	52.8	8.4	34.2	66.0	62.6	51.9	13.6	0.0	80.6
common carp	20.8	35.8	87.3	1.9	0.0	0.0	30.9	55.0	0.0	0.0
spottail shiner	6.8	8.2	0.1	44.0	0.6	0.0	0.6	0.1	0.0	0.0
emerald shiner	0.7	0.0	0.0	0.1	0.1	0.0	0.7	0.0	0.0	0.0
brook stickleback	0.0	0.3	0.2	0.3	1.0	7.5	0.6	0.0	0.0	18.4
ninespine stickleback	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
black bullhead	5.3	0.5	0.0	0.8	1.2	0.0	6.6	26.8	0.0	0.0
brown bullhead	-	-	-	-	-	-	-	-	0.0	-
northern pike	0.2	0.7	0.0	1.5	29.9	29.9	5.9	0.0	0.0	0.0
trout-perch	-	-	-	-	-	-	-	-	0.0	-
yellow perch	48.2	1.5	3.3	16.9	1.0	0.0	0.5	4.0	0.0	0.0
johnny darter	0.0	0.0	0.0	0.2	0.0	0.0	0.3	0.1	0.0	0.0
Iowa darter	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
white sucker	0.0	0.2	0.5	0.0	0.3	0.0	0.9	0.0	0.0	0.0
central mudminnow	-	-	-	-	-	-	-	-	0.0	-
freshwater drum	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	1.0
Totals	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	0.0	100.0

1 - due to low water levels, this pond could not representatively sampled (see text for details)

Table 2-12. Summary of fisheries studies conducted at Delta Marsh, Manitoba between 1983 and 2002. Studies are grouped by sampling method. Fish species are listed in frequency of occurrence.

Sampling Method	Beamish traps			Beamish & minnow traps	
Study	Schneider	Parks		Evelsizer	
Site(s)	Blind Channel	10 ponds across the marsh		connected ponds	isolated ponds
Year	1983	2001	2002	1999	1999
	black & brown bullheads[1]	fathead minnow	fathead minnow	spottail shiner	brook stickleback
	spottail shiner	black bullhead	yellow perch	emerald shiners	fathead minnow
	fathead minnow	yellow perch	common carp	fathead minnow	ninespine stickleback
	yellow perch	brook stickleback	black bullhead	black bullhead	
	ninespine stickleback	white sucker	brook stickleback	Iowa darter	
	emerald shiners	common carp	spottail shiner	yellow perch	
	brook stickleback	spottail shiner	emerald shiners	brown bullhead	
	common carp	johnny darter	Iowa darter	brook stickleback	
	white sucker	emerald shiners	johnny darter	common carp	
	Iowa darter	Iowa darter	white sucker		
	johnny darter	northern pike	northern pike		
	trout perch	trout perch	freshwater drum		
	log perch	freshwater drum	ninespine stickleback		
	northern pike		central mudminnow		
	central mudminnow		trout perch		

1 - no distinction was made between black and brown bullheads
2 - species captured but frequency of occurrence was not recorded
3 - species only captured once or twice during sampling

Table 2-12. Continued.

Minnow traps			
Suthers	Kiers & Hann[3]	Goodyear	Parks
Mid-Blind Channel	Mid-Blind Channel	Blind Channel & Crescent Pond	10 ponds across the marsh
1982	1995	1996	2001
yellow perch	fathead minnow	fathead minnow	fathead minnow
fathead minnow	brook stickleback	yellow perch	brook stickleback
brown bullhead	yellow perch	brook stickleback	common carp
brook stickleback	ninespine stickleback	ninespine stickleback	yellow perch
spottail shiner	spottail shiner[2]	spottail shiner[3]	black bullhead
common carp	white sucker[2]	emerald shiners[3]	white sucker
northern pike	common carp[2]	common carp[3]	central mudminnow
ninespine stickleback	black & brown bullheads[1,2]	white sucker[3]	
	Iowa darter[2]		

1 - no distinction was made between black and brown bullheads
2 - species captured but frequency of occurrence was not recorded
3 - species only captured once or twice during sampling

Table 2-13. Fish caught entering and leaving Delta Marsh through Cram Creek, May 3 to August 27, 1983 (from Lapointe 1986).

Species	Entering	Leaving
white sucker	576	877
northern pike	248	317
yellow perch	163	105
common carp	91	289
brown bullhead	54	19
sauger	18	13
burbot	14	18
walleye	12	12
quillback	9	43
lake herring, cisco	3	37
bigmouth buffalo	2	3
silver redhorse	1	5
channel catfish	1	3
freshwater drum	0	12
mooneye	0	6

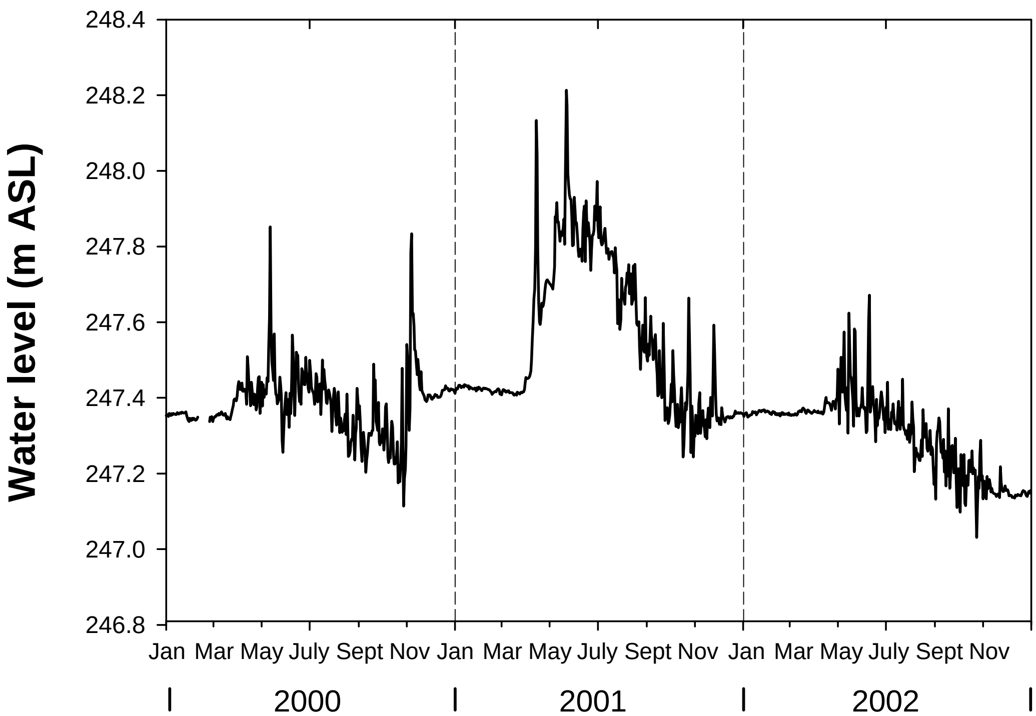

Figure 2-1. Daily water levels on Lake Manitoba at Westbourne, Manitoba (hydrographic station 05LL012). Data from Environment Canada website (www.wsc.ec.gc.ca/hydat/H2O/index_e.cfm).

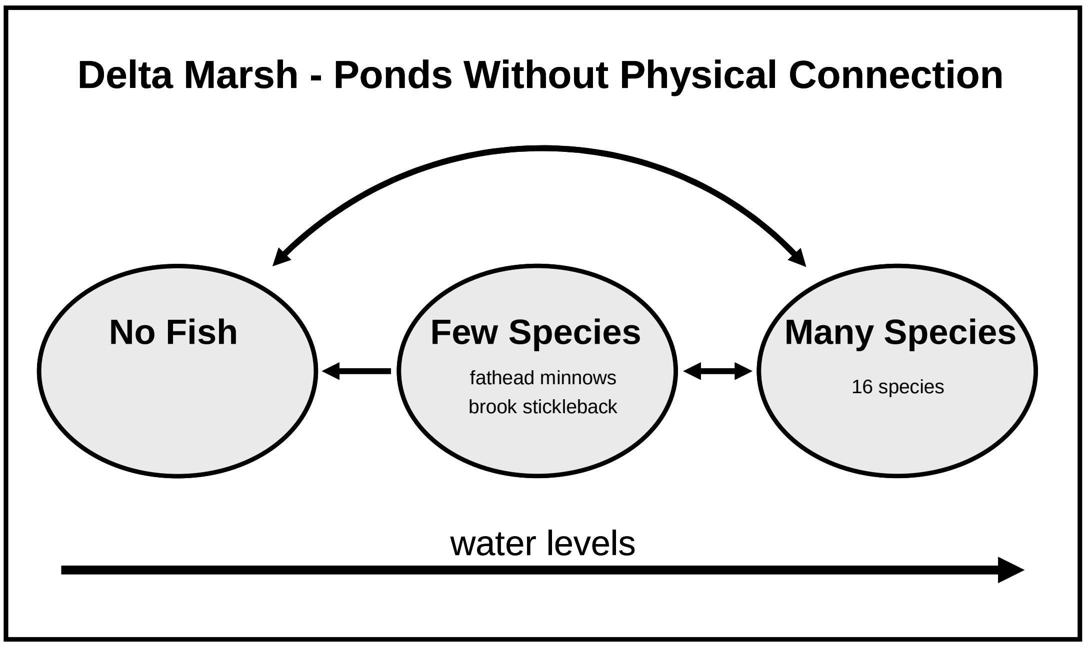

Figure 2-2. The fish communities in isolated peripheral ponds of Delta Marsh.

Figure 2-3. Common carp congregating in front of the screened culvert opening at Mid-Blind Channel (Photo by Dale Wrubleski).

Figure 2-4. Group of large common carp at culverts, forming a biological barrier, potentially blocking entrance by other fish into Mid-Blind Channel (Photo by Candace Parks).

APPENDIX

Appendix A. Mean fork length (cm) and wet weight (g) by species, for fish captured using Beamish traps set in study ponds between early June and late August, 2004.

	Length (cm)						Weight (g)					
Species	**n[1]**	**Mean**	**Std[2]**	**Range**			**n**	**Mean**	**Std**	**Range**		
black bullhead	15	5.2	1.5	3.0	-	8.2	15	2.5	1.4	1.0	-	5.4
brook stickleback	157	2.8	1.0	1.4	-	7.5	155	0.3	0.6	0.0	-	3.6
common carp	724	4.8	2.1	1.3	-	18.0	718	3.8	7.0	0.0	-	127.5
emerald shiner	52	4.8	1.1	3.2	-	8.1	52	1.2	1.0	0.4	-	4.5
fathead minnow	329	4.7	0.9	2.2	-	6.7	326	1.7	0.8	0.1	-	4.2
freshwater drum	3	5.1	0.2	5.0	-	5.4	3	2.5	0.5	2.1	-	3.1
Iowa darter	43	4.5	0.4	3.2	-	5.5	43	0.9	0.3	0.4	-	1.5
northern pike	48	25.4	12.0	7.7	-	52.0	48	102.5	91.2	4.7	-	400.3
ninespine stickleback	29	3.3	1.7	1.6	-	7.2	29	0.5	0.9	0.0	-	2.7
spottail shiner	209	7.3	1.5	2.0	-	10.0	207	5.5	2.7	0.1	-	13.5
trout-perch	2	4.5	2.8	2.5	-	6.5	2	1.4	1.7	0.1	-	2.6
white sucker	102	6.3	2.3	2.1	-	15.5	98	4.4	7.1	0.1	-	41.4
yellow perch	497	5.9	1.9	2.5	-	12.5	496	3.4	3.4	0.2	-	22.7

1 - n = number measured (may not correspond to the number of fish captured)

2 - Std = standard deviation

CHAPTER 3 – THE IMPACT OF POND CONNECTIVITY AND COMMON CARP ON NATIVE FISH, AMPHIBIANS AND WATER-COLUMN INVERTEBRATES IN A COASTAL MARSH IN SOUTH-CENTRAL MANITOBA, CANADA

3.0 Introduction

Coastal wetlands are among the most productive, ecologically diverse ecosystems on earth (Constanza et al 1997), and they provide numerous practical and economic benefits (Mitsch and Gosselink 2000). They offer shoreline protection, buffer wave energy from storms and hurricanes (Bourne 2005), dampen short-term seiches and storm surges (Herdendorf 1992), provide flood reduction, water retention, groundwater recharge (Mitsch and Gosselink 2000), and remove nutrients from surface waters (Murkin 1998). Coastal wetlands provide important habitat for many commercially and recreationally valuable taxa (Clark 1978; Jaworski and Raphael 1978; Herdendorf et al. 1986; Herdendorf 1987; Stephenson 1990; Herdendorf 1992; Jude and Pappas 1992). Louisiana coastal wetlands are estimated to be worth over US$300 million annually in seafood revenue, and are second to Alaska in commercial fish landings (Bourne 2005). Coastal wetlands are also home to numerous insects, crustaceans, fish, birds, amphibians, reptiles and mammals (Weller 1978a; Clark 1978; Herdendorf 1992), and are recognized as crucial habitat for many threatened or endangered species (Mitsch and Gosselink 2000; Young et al. 2002).

Despite their importance, coastal wetlands are disappearing at alarming rates. In the past, significant losses occurred as a result of drainage for agricultural purposes (Jaworski and Raphael 1978; Murkin 1998), dredging, filling and canal building (Milano 1999; Bourne 2005). More recently, urbanization, residential encroachment, oil and gas exploration (Bourne 2005), water-level regulation (Wilcox and Whillans 1999), and dike construction (Young et al. 2002) have interrupted hydrological function and flow regimes of many coastal wetlands (Meador et al. 2003). Fertilizers and pesticides from agricultural and urban runoff degrade water quality (Jaworski and Raphael 1978). Of increasing concern is climate change, adding further stress to wetlands already impacted by human alterations (Mortch 1998). Changes in land use have made native species more susceptible to competition from invasive species, which may be more adaptable to degraded environments (Meador et al. 2003).

After habitat alteration, invasive species are the most common cause of species decline and extinction (Wilcove et al. 1998), and have been estimated to cost the United States economy US$137 billion annually (Perrings 2000). Nutria (*Myocastor coypus*) are destroying large expanses of wetlands across Florida and the Gulf States (The Louisiana Department of Wildlife and Fisheries 2003). Zebra mussels (*Dreissena polymorpha*) have drastically upset the Great Lakes ecosystem (Ralley 2002) and are now widely distributed across all five lakes, and the Mississippi, Hudson, Tennessee, and Ohio river basins (USGS 2000). As successful invaders, zebra mussels form dense colonies on boats, water-intake pipes, docks, aquatic plants and on slow moving crayfish, clams, and turtles. The US Fish and Wildlife Service has estimated that the potential economic damage caused by zebra mussels to water users of the Laurentian Great Lakes will reach $5 billion dollars over the next ten years (USGS 2004).

In recent years, most non-indigenous introductions have been accidental. Organisms such as zebra mussels, spiny water flea (*Bythotrephes cedertroemi*), and ruffe (*Gymnocephalus cernuus*) were inadvertently transported in the ballast water of transoceanic ships (Great Lakes Information Network 2005), while nutria (The Louisiana Department of Wildlife and Fisheries 2003), bighead carp [*Aristichthys nobilis* (Richardson)] and silver carp [*Hypophthalmichthys molitrix* (Valenciennes)] (Great Lakes Fish Commission nd. accessed November 2005) escaped from rearing areas. Other invasive taxa were intentionally released. The common carp (*Cyprinus carpio*) was introduced into North American waters in the late 1880s (McCrimmon 1968). Initially introduced as an inexpensive protein source (Li & Moyle 1999), common carp have proliferated across much of the continent (Cooper 1987; Scott & Crossman 1998). Their success is credited to their high fecundity, hardiness, and their unprecedented tolerance to a wide range of environmental conditions (Bardach et al. 1972; Scott and Crossman 1998). Unfortunately, little thought was given to the ecological implications of their introduction (Zambrano et al. 2001).

Despite their initial perception as an excellent food fish, the introduction of common carp received little fanfare and has instead been considered a nuisance by fisheries and wildlife managers (McCrimmon 1968; Stewart and Watkinson 2004). Numerous eradication programs have been implemented across North America (Hanson and Butler 1994; Zambrano et al. 1999; Zambrano et al. 2001; Lougheed et al. 2004), Europe (Scheffer 1998) and Australia (Koehn et al. 2000). Whether common carp have taken advantage of poor habitat conditions attributed to increased human development, environmental degradation, and overexploitation of native species, or whether they are the reason of decline of aquatic ecosystems is subject to debate (Whillans 1996; Koehn et al. 2000).

In several Great Lakes coastal wetlands, barriers, such as dikes or dams, have been used to artificially manage water levels to improve wetlands for waterfowl and prevent access by common carp (Sigler 1958; Bookhout et al. 1989). However, these physical barriers impede the exchange of water and native fish species between coastal wetlands and their adjoining lakes (*e.g.*, Johnson et al. 1997). To mitigate these impacts, screened fishways have been constructed on several Great Lakes coastal wetlands, including Cootes Paradise Marsh (Lougheed et al. 2004) and Metzger Marsh (Wilcox and Whillans 1999). These control structures allow hydrologic connection (Wilcox and Whillans 1999), but impede access by adult common carp. Small fish pass through the screens (5 cm gaps between bars), but large fish are funneled into cages that can be brought to the surface. Personnel manually sort captured fish, with common carp released back into the lake and native fish released into the marsh.

The aim of common carp removal programs has been focused on habitat improvement. However, this management has been largely based on the results of small enclosure experiments, which have reported correlations between common carp biomass and increased turbidity (Roberts et al. 1995), aquatic macrophyte destruction (Threinen and Helm 1954; Tryon 1954; Robel 1961; King and Hunt 1967) and reductions in large-bodied zooplankton numbers (Lougheed et al. 1998). Results from more natural studies have been variable. In Delta Marsh, a study evaluating varying densities of common carp found total suspended solids (TSS) and sedimentation rates were two to four times greater in small mesocosms compared to larger wetland cells (Badiou 2005). Common carp had little effect on aquatic macrophytes in the wetland cells when exposed to varying densities of common carp. Impacts of common carp can also vary among aquatic systems. For example, the effects of common carp on water-column turbidity may be linked to substrate texture; clearer water is often found over sandy substrates, whereas more turbid waters are found over silty substrates (Scheffer 1998; Parkos III et al. 2003). Other factors such as water depth and wind may make it difficult to isolate the impacts of common carp on aquatic habitats. In a study that monitored ten water bodies in northern Australia exposed to varying densities of common carp, no association was found between common carp density and water clarity (Fletcher et al. 1985). However, weakly rooted and soft-leaved aquatic vegetation were damaged by common carp spawning and foraging activities. Results of common carp control programs that have removed the entire fish fauna with piscicides, such as Rotenone (*i.e.*, Lake Christina, (Hanson and Butler 1994)), may be based on incomplete or unrealistic data. In particular, there are few studies that have looked at the effects of native benthivores on vegetation growth (Kolterman 1990) and aquatic invertebrate abundance (Parkos III et al. 2003). The effects of native benthivores may be underestimated and instead attributed to the removal of common carp. In an enclosure study in North Dakota, black bullhead were shown to negatively impact aquatic vegetation growth (Kolterman 1990). Unfortunately, the effects of other native benthic fish have rarely been experimentally tested.

The current project examined the effects of pond connectivity and common carp within Delta Marsh, a large coastal wetland on the south shore of Lake Manitoba. Ten study ponds around the periphery of the main marsh were modified with respect to their connection to the main marsh. These modifications altered water and fish movement between the ponds and the main marsh. I was particularly interested in common carp and their effects within the study ponds. Common carp are a dominant fish species within the marsh (Dale Wrubleski, Ducks Unlimited Canada, personal communication), and as noted above, may have a significant impact on this wetland.

3.1 Study Site – Delta Marsh

This study took place in Delta Marsh (50° 11' N, 98° 19' W), situated on the south shore of Lake Manitoba. This 18,500 ha lacustrine freshwater marsh is protected by a narrow forested barrier beach ridge (Figure 1-3). As one of the largest marshes in the province, Delta Marsh is considered important habitat for numerous fish, mammals, plants, songbirds, and especially waterfowl (Batt 2000). Delta Marsh is a permanent, slightly brackish, wetland made up of a network of large and small bays, smaller ponds and a matrix of winding channels. The marsh is relatively shallow, averaging about 1 m in depth (Macaulay 1973). As a consequence of the four connecting channels between the marsh and Lake Manitoba, water levels on the marsh are dictated to a large extent by the lake. Similar to Lake Winnipeg (Einarsson and Lowe 1968), water levels on Lake Manitoba fluctuate frequently due to wind set-up and set-down, however historic wide-ranging variations have been dampened by stabilization of the lake (de Geus 1987). This has also resulted in a long-term stabilization of water levels on Delta Marsh.

Aquatic vegetation varies spatially across the marsh and has been described by Löve and Löve (1954), Walker (1959; 1965) and Anderson and Jones (1976) and summarized by Shay (1999). Changes in emergent vegetation have been described by de Geus (1987) and summarized by Shay et al (1999) and Batt (2000). The dominant emergent vegetation fringing the open water habitat includes *Typha* X *glauca* Godr., mixed with *T. latifolia* L. and *T. angustifolia* L., *Schoenoplectus acutus* (Muhl. *ex* Bigelow) A. & D.

Löve, and at higher elevations *Phragmites australis* (Cav.) Trin. *ex* Steud (Shay et al 1999). Small stands of *Schoenoplectus tabernaemontani* (K.C. Gmel.) Palla can also be found infrequently along shorelines in shallow areas (Shay et al. 1999). Aquatic macrophytes have been surveyed by Anderson and Jones (1976) and described by Batt (2000). Three main species or species-assemblages of submersed macrophytes exist in the marsh, and include *Stuckenia pectinatus* (L.) Boerner, *Myriophyllum sibiricum* Komarov, *Utricularia macrorhiza* Le Conte and *Ceratophyllum demersum* (L.) (Anderson and Jones 1976). Detailed information of the hydrology and water chemistry can be obtained from Anderson and Jones (1976) and Goldsborough (1994).

3.2 Study Ponds

Ten ponds, situated around large bays of the main marsh, were selected for the manipulation study (Table 1-1). Ponds were chosen based on numerous factors such as location, size, ownership (*i.e.,* private or Crown land), presence or absence of a connection to the main marsh, logistics, and feasibility.

Four ponds were naturally isolated from the surrounding marsh, whereas six were connected to the main marsh by way of a natural or artificial channel (Table 1-1). In 2001, all ponds remained in their natural state to provide baseline information. Although four ponds are described as isolated, high water levels on Lake Manitoba in the spring of 2001, along with several storm events (see Figure 2-1), raised water levels in the marsh. As a consequence, these ponds became temporarily connected to the main marsh. To avoid confusion, these ponds will be referred to as "temporarily connected" for the remainder of the paper (Table 1-1).

The manipulation part of the study began in the spring of 2002, when six ponds were altered with respect to their connection to the main marsh. Two connected and two isolated ponds functioned as controls and remained in their natural state. Four ponds were selectively isolated from the main marsh; two ponds by sandbag dikes and two ponds by fish screens. Dikes were constructed at the entrances of South Pitblado's Channel (Figures 1-8, 1-9), and Mad Woman Bay (Figure 1-10) on April 24 and May 2, respectively (see Chapter 2). However, in both cases, dikes were not completely effective at preventing fish access and water exchange between the pond and the marsh. Spawning northern pike gained access to South Pitblado's Channel, and subsequently their young-of-the-year (YOY) became trapped inside the diked pond. Adult northern pike were not captured during the study and likely emigrated prior to construction of the dike. At Mad Woman Bay, wind set-up caused temporary flooding of the area immediately surrounding the dike, facilitating movement of fish, such as northern pike (Candace Parks, personal observation), into the pond (Figure 1-10).

Fish screens were installed at Mid-Blind Channel (Figure 1-7) and Section 5 Bay (Figure 1-6) (see Chapter 2). The purpose of the screens was to allow the exchange of water and small fish between the main marsh and ponds while excluding adult common carp. A similar screening structure with a 5 cm gap width has successfully been used to restrict common carp larger than 34 cm from entering Cootes Paradise Marsh (Wilcox and Whillans 1999). This width was chosen because it still allowed for free passage of northern pike ranging from 49.4 to 70.0 cm in total length (French III et al. 1999). On April 22, the three culvert openings separating Mid-Blind Channel from the main marsh were outfitted with barred culvert covers (Figure 1-7). Meanwhile, a conduit fence was constructed across the channel leading into Section 5 Bay (Figure 1-6). Screens were removed in late August allow free passage of fish between the ponds and the main marsh.

In contrast, Mackenzie Bay South and Wye's Pond had no prior permanent connection to the main marsh. On May 17 and June 14, 2002 respectively, connections were created by blasting with ditching dynamite (see Chapter 2). The resultant channels, less than 2 m wide and approximately 1 m deep, allowed the exchange of water between the main marsh and the ponds, and also functioned as access routes for fish (Figures 1-11, 1-12, 1-13, 1-14). For the remainder of this paper, these two newly connected ponds will be referred to as "blasted" ponds.

3.3 Field Sampling Methods

3.3.1 Water- Column Invertebrates

Sampling water-column invertebrates within each pond was done along two transects arranged perpendicular to the shore. Each transect consisted of three sampling sites, each marked with a single wooden post: site 1 was approximately two metres from shore, site 3 was approximately at the centre of the pond, and site 2 was equidistance between sites 1 and 3. Transects were placed several metres apart to accommodate canoe movement among the posts. Each post was fitted with a crossbar, mounted approximately 0.5 m above the water surface, on which were hung the activity traps.

Activity traps, modified from Murkin et al. (1983), with 1-L glass bottles and an inverted funnel with an opening of 2 cm, were used to sample aquatic nektonic and free-floating invertebrates within study ponds (Figure 3-1). Activity traps were chosen because they are easy to setup and retrieve, and incorporate variations in horizontal and vertical diel movements of many taxa (Murkin et al. 1983; Murkin et al. 1993). Since activity traps are stationary, they do not disturb aquatic macrophytes, unlike other methods such as sweep nets. Also, activity traps are effective in collecting highly mobile nektonic taxa, as well as free-floating invertebrates (Murkin et al. 1983), collectively referred to as water-column invertebrates for the remainder to this paper. However, this type of trap precludes the capture of larger predaceous invertebrates (*i.e.,* larger than 2 cm in width), and only provides relative estimates of abundance, since the actual volume of water sampled is unknown. Additional advantages and disadvantages of activity traps for sampling water-column invertebrates in freshwater aquatic habitats are discussed in Murkin et al. (1983), Murkin et al. (1993), and Hyvonen and Nummi (2000). Although horizontal activity traps have been predominantly used in wetland invertebrate studies, Muscha et al. (2001) determined that vertical traps produced more diverse and abundant samples for most taxa, and captured more small-sized zooplankton than horizontal activity traps. Consequently, a total of 12 activity traps, six oriented horizontally and six oriented vertically, were hung mid-water column with one horizontal and one vertical activity trap/post. Vertical traps were set with the funnel oriented downward toward the sediments.

In 2001 and 2002, sampling took place every three to four weeks. Traps were retrieved after 24 h and the contents were poured through a 500 µm sieve in the field. Fish and larval frogs that could be readily identified were counted and immediately released. Invertebrates collected on the sieve were washed into labeled 250 ml plastic jars, and a small volume of 90% ethyl alcohol was added to ensure all organisms were killed immediately. In the laboratory, samples were filtered again using a fine mesh < 0.5 mm sieve, and contents were placed in storage vials with 70% ethyl alcohol.

Due to the large number of samples collected, and high invertebrate diversity and abundance within each sample, samples collected from one of the two transects were identified for this study. Invertebrates from each sample were sorted according to taxa, counted and identified to at least order. Activity trap samples were pooled according to the trap orientation among the three sampling sites. This was done for several reasons. First, the main goal was to compare among the treatments (*e.g.*, blasted ponds versus diked ponds) investigating the differences based upon connectivity and the presence or absence of common carp, rather than focusing on subtle individual site-to-site differences in a single study pond. Quantitative information on aquatic macrophytes (*i.e.,* structure, diversity and density) immediately around transects was absent, making it difficult to rationalize specific taxa distributions for each individual trap based on vegetation presence or absence. An overall seasonal average was calculated for water-column invertebrates within each pond, making individual trap counts unnecessary. Aquatic insect stages including adults, nymphs and larvae were identified to family using Brooks and Kelton (1967), Merritt and Cummins (1996) and Clifford (1991), and were not differentiated in the analyses.

Sub-sampling was used to count large samples of copepods (Copepoda) and cladocerans (Cladocera) (where $n \geq 500$) using methods defined by Edmondson (1984). Although populations of small-sized organisms are often distributed randomly, verification was done by counting several sub-samples to calculate variance to mean ratios (CV) to see whether the sub-samples conformed to a random distribution for each taxa (McCauley

1984). Once a random distribution was verified, sub-sampling was deemed an appropriate method. The total volume of each preserved sample was standardized to 20 ml (60 ml for very large samples) by adding water to the sample, which was then thoroughly shaken and successive 1 ml sub-samples were removed using a wide bore pipette. Six to ten sub-samples were counted from each standardized sample. An average of the sub-samples was calculated for each vial and used to extrapolate the total number of zooplankton captured in one activity trap. To further ensure accuracy of sub-samples, total counts were randomly done on several vials to compare between the actual counts and the calculated values; results between the two methods were very similar.

As the season progressed in 2002, ponds become shallower, and on occasion periodic wind set-downs lowered water levels even further. As a consequence, activity traps set in Thompson's Bay and Section 5 Bay became inaccessible by canoe and remained in the pond for longer than 24 h. A few vertical traps were observed to be no longer submerged and the contents were lost. To account for this variability, a seasonal average was determined for both 2001 and 2002 to allow for direct comparison among ponds, and between years. Invertebrates such as *Hydra* (Hydrozoa), oligochaetes (Oligochaeta), veiilids (Veliidae), chironomids (Chironomidae), and belostomatids (Belostomatidae) were not effectively sampled by activity traps, or occurred in too low frequencies to permit quantitative comparisons, and were omitted from further analyses.

3.3.2 Fish and Amphibians

Beamish traps were used to sample fish and amphibians (Beamish 1973). This trap design was chosen because it was relatively easy to transport, set-up, and most importantly, captured both pelagic and benthic fish species.

These traps resemble Fyke nets, yet are able to be used for shallow habitats (Figure 3-2). A ring (15 cm diameter) was built into the funnel to ensure that it would remain open during periods of low water levels (provided enough water was available to ensure the trap was submerged). Fish larger than 15 cm in diameter were excluded from capture by the ring. Alternatively, a trap without the rigid ring at the opening of the funnel would collapse on itself at shallow depths preventing fish access. Minnow traps (Hubert 1996) were also used in 2001, but were discontinued in 2002. Traps selectively captured pelagic fathead minnows and brook sticklebacks and precluded the capture of benthic and larger-sized fish species (see Chapter 2) (Figure 3-3).

In 2001, a single Beamish trap (1.5 mm stretched mesh, with a 15 cm by 15 cm inner funnel opening lacking the rigid plastic ring) was set randomly along the shore in each study pond. Traps were set parallel to shore, with one wing extending perpendicular into the emergent vegetation edge and the other towards the center of the pond. The following year, additional Beamish traps with the rigid plastic ring in the funnel opening were obtained, and subsequently two Beamish traps were set in each pond as opposed to a single trap as in 2001. One Beamish trap without the ring and one trap with the rigid plastic ring were set in each pond in 2002. The dimensions of the traps were identical, although the traps with the rigid ring had a larger mesh size (3 mm stretched mesh). The first trap was set randomly within the pond, while the second was set on the opposite shoreline. Beamish traps are set in each pond for 24 h, on an approximate three week rotation between June and August of both years. The majority of catches were composed of YOY or small-sized fish species. Several fish species for example, common carp, can grow larger than the opening of the Beamish trap (15 cm diameter) thus could not be effectively sampled. As a result, the few adults that were captured were noted but excluded from analyses.

The contents of each Beamish trap were emptied into a large holding tub, identified to species, enumerated, and released immediately. To reduce fish mortality, large catches (> 1000 fish) were sub-sampled using a 5–L bucket. Fish were randomly placed into the bucket, identified, counted and released. Two successive bucket sub-samples were counted. The number of buckets required to remove all fish from the holding tub was then determined and capture of rare species was also recorded. Total trap catch was estimated by taking an average of the two sub-sampling buckets and multiplying by the number of buckets required to empty the holding tub.

Older Beamish traps lacked a rigid ring in the funnel, thus could easily collapse if water levels declined, preventing access by fish and amphibians. As a result, reductions in water levels and sporadic wind set-downs in 2002 precluded the use of the older Beamish traps. As an alternative, a single newer Beamish trap was set in Thompson's Bay and Section 5 Bay for the remainder of the summer. Despite this attempt, water levels in these ponds continued to drop and by mid-summer ponds became too shallow to sample randomly. As a compromise, the single Beamish trap was set in the deepest part of each pond. In Section 5 Bay, the Beamish trap was set beside the channel connecting Section 5 Bay to Mad Woman Bay. At Thompson's Bay, the trap was set in the only channel leading into the pond, with the funnel oriented directly into the pond. Positioning of the trap at Thompson's Bay meant that most fish exiting the pond were funneled directly into the trap, and this resulted in highly inflated numbers of fish captured for this pond.

3.4 Data Analyses

3.4.1 Species Composition and Diversity Measures

Absolute richness, effective richness, and evenness were determined for each pond. Absolute richness (s) describes the number of taxa captured in an area. The more taxa present in a pond, the more "rich" or specious the pond (Krebs 1999). Effective richness (N_2 or Hill's N_2) expressed as a percent, considers the number of taxa within a community and the equitability among the taxa. These particular measures are based on the Simpson's index, where s is the total number of taxa in each pond (Begon et al. 1996). Values range from 1 to s, the closer the value is to s, the greater the diversity (Krebs 1999). Because taxa are ranked, results are less influenced by uneven sample sizes and by the presence rare taxa, allowing comparison among the study ponds. If all taxonomic groupings have equal numbers of individuals, effective richness is maximized. The formula is as follows;

$$D = \frac{1}{\sum_{i=1}^{s} p_i^2}$$

Where:

p_i = Proportion of taxa $_i$ in the community (Krebs 1999).

Evenness (E = N_2/s), expressed as a percent, represents the equitability of taxa albeit families, species etc. within a sample. Here, it represents the relative abundance of different taxa within a community and ranges between zero and one (or when expressed as a percent, 0 and 100%). For instance, an area dominated by a single taxa would have a low evenness value close to zero (Legendre and Legendre 1998).

3.4.2 Multivariate Analysis

Multivariate analysis is a descriptive modeling technique which considers many variables simultaneously reducing them into a two-dimensional scatterplot that can be readily interpreted (Kenkel et al. 2002; Shaw 2003). Initial Principal Component Analyses (PCA) were performed on all data to check for outliers and to reveal underlying data structure. All multivariate analyses were performed using CANOCO version 4 (ter Braak and Smilauer 1998) and subsequent results were plotted using SYN-TAX 5.0 (Podani 1994) and CorelDraw 12 computer programs.

3.4.2.1 Water-Column Invertebrates

Multivariate analysis is a beneficial tool to examine ecological trends. Preliminary analyses revealed the overall data structure was linear, therefore PCA was deemed an appropriate multivariate modeling technique for the activity trap data. The main goal of these methods is to simplify data by reducing the dimensions in data space. This is done by rotating additional axis about their origin to give a least squares best fit of the data where residuals are fitted orthogonally to give the lowest possible sum of squared residuals (Shaw 2003). The second axis passes through the mean of the data at a 90 degrees angle to the first axis and is rotated to find the angle that explains the greatest degree of variation (Shaw 2003). The resultant ordination diagram places ponds with similar taxa composition closer together, and ponds with different taxa are placed further apart. Values furthest away from the origin (0,0) indicate stronger correlations between the vectors and the axes (Shaw 2003).

Taxa captured in low abundances, *i.e.* captured once or twice, such as conchostracans (Conchostraca) and larval gyrinids (Gyrinidae), were removed from subsequent analyses. The remaining water-column invertebrate data were standardized by multiplying by a factor of 100 and normalized to log (n+1). This was done prior to analyses to prevent extreme values from swamping the data (Kenkel et al. 2002; Shaw 2003).

3.4.2.2 Fish

Ecological data, composed of many poorly represented taxa and a few very abundant ones, are generally non-linear (Kenkel et al. 2002). Consequently, Correspondence Analysis (CA) is the most appropriate ordination technique for datasets which have a large number of zero values (Kenkel et al. 2002; Shaw 2003). My original fish data were comprised of 44% zero values. However, this ordination technique has shortcomings. CA is based upon an eigenanalysis technique, where dissimilarities in the data are measured using χ^2 differences. As a result, it responds overly well to noisy data, thereby focusing on outliers in the data set (Kenkel et al. 2002). To avoid this problem, the taxa abundance data were standardized by multiplying all values by a factor of 100 to make certain all values were greater than one prior to performing the CA multivariate model. A log (n+1) transformation was then applied to stabilize variances (Townend 2002). This modification places marginal presence of all taxa in each pond, in order to downweight the influence of rare taxa, or hotspots in the data. This meets the assumption that all variables have the same underlying data structure.

Similar to other multivariate methods, CA produces an ordination diagram in which ponds with similar taxa composition are plotted close together, in contrast to ponds with different taxa composition which lie further apart (Shaw 2003). CA was performed on the seasonal data, or the average of all the sampling dates over the entire season, to examine fish community structure in each pond in the two years. Ponds which relate highly to the first few ordination axes account for a large proportion of the variation observed in the taxa data (Shaw 2003).

The results from the CA were compared with those of a Detrended Correspondence Analysis (DCA) in order to eliminate a potential arching effect commonly occurring with CA ordination plots (Shaw 2003). The CA and DCA plots were very similar, so only the results of the CA will be presented here.

In addition to PCA and CA, additional exploratory analyses were done using Redundancy Analysis (RDA) to see whether water-column invertebrate distributions or a limited number of environmental variables (see below) explained any additional variation in fish distributions in the study ponds. Alternatively, a CCA was used to determine whether fish distributions explained additional variation in the water-column invertebrates. Canonical Correspondence Analysis (CCA) is a direct gradient analysis method incorporating the response of environmental variables on the data matrix. CAA, like CA, places samples with similar species compositions and distributions closer together and those with dissimilar compositions and distributions further apart. This method tries to find the combination of environmental variables that maximizes the separation of species (Shaw 2003). In 2002, I measured a limited number of environmental variables (water temperature, water depth, percent submerged vegetation cover and approximate photic depth) within each pond and averaged them across the sampling season to obtain a single value for each study pond. Environmental variables were measured to see if they could explain any additional variation in the fish and/or water-column invertebrate distributions. Environmental variables are represented on the CCA triplot as vectors, which increase in value from the origin. The longer the vector, the more important the variable is in explaining variation of the plot. The averaged data were transformed in the same manner as in CA, by multiplying by a factor of 100, then applying a log (n+1) transformation.

3.6 Results

3.6.1 Water-Column Invertebrates

3.6.1.1 Community Structure, Abundance and Diversity

Seventeen invertebrate taxa were collected during this study (Table 3-1). In both years, small-sized zooplankton (*i.e.*, Copepoda, Cladocera) were dominant (Table 3-1). Copepods were the most abundant taxa in 2001 and accounted for 45.5% of the average catch, while cladocerans represented 34.1%. Other dominant taxa in 2001 included the Corixidae (10.3%), Hydracarina (7.9%), Gastropoda (0.7%) and Ostracoda (0.7%). The remaining 11 taxa combined accounted for 0.8% of the average catch (Table 3-1). In 2002, Cladocera were the most abundant invertebrate taxa captured in activity traps and represented 75.3% of the average catch (Table 3-1). Copepoda (14.1%), Corixidae (5.2%), Hydracarina (1.7%), Gastropoda (1.5%), and Amphipoda (0.8%) were also commonly encountered taxa. The remaining 11 taxa collectively represented 1.5% of the average catch (Table 3-1).

Invertebrate abundances varied between years. Nearly 14 times more invertebrates were captured per activity trap in 2002 compared to 2001 (Table 3-1). Except for the larval gyrinids, all invertebrate taxa were more abundant in 2002 than in 2001. Thirty times more Cladocera were caught in 2002, while Dytiscidae and Notonectidae abundances greatly improved in the second year (Table 3-1).

In 2001, aquatic invertebrate taxa richness ranged from 10 to 16 taxa per pond (Table 3-2). Despite flooding, taxa richness was higher in temporarily connected ponds (13 to 16 taxa) compared to permanently connected ponds (10 to 13 taxa). In general, effective richness (N_2) was low across all ten ponds and ranged from 1.4 in Wye's Pond to 4.5 in Thompson's Bay. Evenness was also low, varying from 10.3 in Emile's Pothole to 45.3 in Thompson's Bay (Table 3-2). In 2002, reduced water levels ensured that isolated ponds became disconnected from the main marsh. For most ponds, the number of water-column invertebrate taxa increased, ranging from an increase of one taxa in Wye's Pond, to six taxa in Mid-Blind Channel (Table 3-2). Thompson's Bay had the same number of taxa in both years, and only South Mackenzie Bay showed a decline, from 16 to 12 taxa. Compared to the baseline year, effective richness and evenness declined, values ranged from 1.2 (Mad Woman Bay and South Pitblado's Channel) to 4.9 (Mid-Bind Channel), and 8.1 (Mad Woman Bay) to 36.9 (Thompson's Bay), respectively (Table 3-2).

Average activity trap catch (number/trap) in 2001 ranged from a low of 8 in Thompson's Bay, to a high of 357 invertebrates in Wye's Pond (Table 3-3). Activity traps set in temporarily connected ponds had a mean of 262 invertebrates, while traps set in permanently connected ponds had on average 82 invertebrates (Table 3-3). In 2002, average trap catch (number/trap) ranged from a low of 200 in Thompson’s Bay, to 5,307 invertebrates in South Pitblado’s Channel (Table 3-4).

3.6.1.2 Pond Treatments and Water-Column Invertebrates

Overall, invertebrate abundances were lower in all the study ponds in 2001 than in 2002. With the exception of three ponds (Mid-Blind Channel, Wye’s Pond and North Mackenzie Bay), Copepoda and Cladocera were the dominant taxa captured in 2001 with corixids and water mites being the third and fourth most abundant water-column invertebrates (Table 3-3). In Mid-Blind Channel and North Mackenzie Bay, copepods were the most common followed by water mites. Cladocera and corixids were the third or forth most abundant invertebrates captured. In Wye’s Pond, copepods, corixids and Cladocera ranked as the first, second and third most common water-column invertebrates captured in activity traps (Table 3-3).

In 2002, Cladocera were the dominant water-column invertebrate in seven study ponds (North School Bay, Section 5 Bay, Mad Woman Bay, South Pitblado’s Channel, Wye’s Pond, North Mackenzie Bay, and Emile’s Pothole), whereas copepods were the dominant taxa in the remaining three ponds (Table 3-4). Corixids and water mites ranked as the third and fourth abundant water-column invertebrates respectively in activity traps in the experimental year (Table 3-4). Abundances of other invertebrate taxa captured in activity traps in both the baseline and experimental year will be discussed in more detail below.

Invertebrate abundances varied greatly between the two connected ponds, Thompson's Bay and North School Bay, and between both years of the study. Across all ten study ponds and in both years of sampling, Thompson's Bay had the lowest number of water-column invertebrates per activity trap (Tables 3-3, 3-4). North School Bay on the other hand had the highest number of water-column invertebrates of all the permanently connected ponds in 2001 (Table 3-3), and had the second highest number of invertebrates of all ponds in 2002 (Table 3-4). Nearly 26 times more invertebrates were collected per activity trap in North School Bay, as opposed to Thompson's Bay in both years (Tables 3-3, 3-4). In both ponds, the number of aquatic invertebrates increased approximately 25 times in 2002, when water levels were lower, compared to 2001 when water levels were higher (Tables 3-3, 3-4). Cladocera were the most abundant invertebrate taxa in both ponds in 2001, and also were dominant in North School Bay in 2002 (Tables 3-3, 3-4). Copepods were more abundant in Thompson's Bay in 2002. North School Bay had the highest number of Cladocera of all ponds sampled in 2001, and had the second most abundant Cladocera in 2002. Also, the highest ostracod abundance of all the study ponds was recorded in North School Bay in 2002 (Table 3-4). Thompson's Bay had 10 invertebrate taxa in both the baseline and experimental years, while North School Bay taxa richness increased from 11 to 16 (Table 3-2).

North Mackenzie Bay and Emile's Pothole, temporarily connected in 2001 but isolated in 2002, had diverse invertebrate assemblages. In 2001, North Mackenzie Bay had the second highest number of water-column invertebrates captured in activity traps

(Table 3-3). The following year, invertebrate numbers increased nearly 16 times (Table 3-4). Emile's Pothole had comparable numbers of invertebrates as North Mackenzie Bay in 2001, but in 2002, water-column invertebrate numbers only increased by six times. In both ponds, copepods were the dominant taxa in 2001, but Cladocera dominated in 2002. Of interest, North Mackenzie Bay had the highest number of amphipods, conchostracans, dytiscids, Hydracarina, and gastropods of all ponds in 2002 (Table 3-3). Ephemeroptera were also abundant in both isolated ponds in 2002 (Table 3-4). North Mackenzie Bay and Emile's Pothole had 14 and 13 invertebrate taxa, respectively, in 2001 (Table 3-2). In 2002, North Mackenzie Bay shared the largest number of invertebrate taxa (17) with Section 5 Bay, whereas 15 invertebrate taxa were found in Emile's Pothole (Table 3-2).

In 2001, when Mad Woman Bay and South Pitblado's Channel were connected to the main marsh, invertebrate numbers were lower than compared to when the ponds were isolated from the main marsh in 2002 (Tables 3-3, 3-4). When these ponds were connected to the marsh in 2001, copepods were the dominant taxa, followed by Cladocera and corixids in Mad Woman Bay whereas Cladocera were more abundant that copepods in South Pitblado's Channel (Table 3-3). Water-column invertebrates such as amphipods, gastropods, and most insects (*i.e.*, dytiscids and notonectids) were also greatly reduced or absent (Table 3-3). In comparison, when Mad Woman Bay and South Pitblado's Channel were isolated from the main marsh in 2002, invertebrate abundance increased greatly (Tables 3-3, 3-4). In fact, South Pitblado's Channel had the highest water-column invertebrate abundance of all study ponds. Besides the numerically dominant Cladocera and corixids, both ponds had high abundances of amphipods, as well as high occurrences of aquatic insects (Table 3-4). Species richness of the water-column invertebrate community increased from 13 to 15 in Mad Woman Bay, and from 12 to 15 in South Pitblado's Channel between the baseline and experimental years (Table 3-2).

As was found in all other ponds in 2002, numbers of water-column invertebrates increased in the two screened ponds, relative to 2001 numbers (Tables 3-3, 3-4). However, the size of the increase was much lower than that found in other ponds. Numbers only increased by eight and 11 times in Section 5 Bay and Mid-Blind Channel, respectively. Section 5 Bay had consistently more water-column invertebrates than Mid-Blind Channel in both years. In 2001, aquatic insects were nearly absent from both ponds (Table 3-3), and when the ponds were screened in 2002, abundances of larger, more mobile taxa, such as amphipods and aquatic insects, increased (Table 3-4). Cladocera were the dominant taxa in Section 5 Bay, whereas Copepoda were the dominant taxa in Mid-Blind Channel in both years. In Mid-Blind Channel, species richness improved from 10 invertebrate taxa in 2001 to 16 in 2002 (Table 3-2). Likewise, Section 5 Bay showed an increase from 13 to 17 taxa when screened (Table 3-2).

Lastly, blasted ponds (South Mackenzie Bay and Wye's Pond) showed variable results. On average, 219 invertebrates per trap were captured in Mackenzie Bay South and 357 in Wye's Pond when ponds were temporarily connected to the main marsh in 2001 (Table 3-3). The following year, when the ponds became connected to the main marsh, invertebrate abundances increased eight times in South Mackenzie Bay and five times in Wye's Pothole (Tables 3-3, 3-4). Several invertebrate taxa, including Amphipoda, Conchostraca, Ostracoda, Notonectidae, Dytiscidae, Hydracarina and Gastropoda increased in abundance in Wye's Pond in 2002, whereas the invertebrate response in South Mackenzie Bay was mixed; notonecids and dytiscids increased, yet numbers of amphipods and gastropods declined (Table 3-4). Interestingly, South Mackenzie Bay was the only study pond that showed a decline in species richness (from 16 to 12 taxa) between the baseline and experimental years (Table 3-2). Wye's Pond increased marginally from 14 to 15 taxa (Table 3-2).

3.6.2 Amphibian Communities

Larval leopard frogs (*Rana pipiens*) were the only amphibian taxa captured in Beamish traps in 2001 (Table 3-5). They were present in the four temporarily connected ponds, as well a single connected pond (South Pitblado's Channel). In contrast, three amphibian species [leopard frog, Canadian toad (*Bufo hemiophrys*) and tiger salamander (*Ambystoma tigrinum*)] were captured in 2002 (Table 3-6). Leopard frog larvae were found in nine of 10 ponds, and were most abundant in isolated, diked and screened ponds (Table 3-6). Larval Canadian toads were found in the two isolated ponds (North Mackenzie Bay and Emile's Pothole), but were most abundant in a blasted pond (Wye's Pond). Tiger salamanders were only captured in a single isolated pond (North Mackenzie Bay) (Table 3-6). Compared to the previous year, ponds that became connected (blasted) to the main marsh in 2002 had reduced amphibian abundances, although Wye's Pond still maintained relatively high numbers in 2002 (Tables 3-5, 3-6). Isolated, screened and diked ponds demonstrated an increase in larval amphibians in 2002. The two connected ponds showed consistently low numbers of amphibians in both years of the study (Tables 3-5, 3-6).

3.6.3 Fish

3.6.3.1 Community Structure, Abundance and Diversity

During the current study, 16 fish species from nine families were captured (Table 2-1). In 2001, cyprinids were the most commonly encountered family, followed by ictalurids, percids and catostomids (Tables 2-1, 3-7). Again in 2002, cyprinids were most abundant, followed by gasterosteids and ictalurids. Fathead minnows were the most numerically abundant fish species captured in ponds in both study years (Table 3-7). In 2001, fathead minnows accounted for 46.9% of the average Beamish trap catch (Table 3-7), followed by black bullhead [*Ictalurus melas* (22.3%)], yellow perch (15.5%), white sucker (6.4%), brook stickleback [*Culaea inconstans* (5.7%)], and common carp (1.9%) (Table 3-7). In 2002, fathead minnows composed 46.1% of the average catch (Table 3-7). Common carp (35.0%), brook stickleback (6.3%), black bullhead (4.9%), spottail shiner [*Notropis hudsonius* (3.5%)], and yellow perch (3.4%) were also commonly captured species (Table 3-7).

On average, 1,548 fish (number/trap) were captured per Beamish trap between early June and late August in 2001, compared to 226 fish (number/trap) in 2002 (Table 3-7). Most species were more abundant in 2001, compared to 2002 (Table 3-7). Of the three species that actually increased in abundance in 2002, common carp showed the greatest increase between years. On average, 29 common carp were captured per Beamish trap in 2001 compared to the 79 in 2002 (Table 3-7).

In 2001, species richness ranged from five (North School Bay and South Mackenzie Bay) to 11 (Wye's Pond) species (Table 3-8). Effective richness was low, ranging from 1.1 (South Mackenzie Bay) to 3.0 (Wye's Pond). High dominance by one or two species results in low evenness, thus when expressed as a percent, evenness was low in the ponds and ranged from 12.2% in South Pitblado's Channel to 32.6% in Emile's Pothole (Table 3-8). In 2002, distinct differences in species richness, effective richness and evenness were apparent compared to 2001. Species richness ranged from zero [fishless North Mackenzie Bay] to 12 species (Thompson's Bay, Mid-Blind Channel, and South Mackenzie Bay). Effective richness varied from zero to 3.3, and evenness values were wide ranging from zero (North Mackenzie Bay) to 65.0 (South Pitblado's Channel) (Table 3-8).

Composition and abundance of the fish community in Delta Marsh varied spatially and temporally among the ten study ponds. Overall, ponds with a permanent direct connection (*i.e.*, channel) to the main marsh had a more diverse fish community compared to ponds lacking a connection. On average in 2002, both connected and blasted ponds had 11 fish species, screened ponds had nine, diked ponds had six, and isolated ponds had only two fish species present (Table 3-8). Conversely, this was not the case in 2001. High water levels on Lake Manitoba in the spring of 2001, led to correspondingly high water levels in Delta Marsh. This in turn flooded upland areas and permitted fish access to the four previously isolated ponds. As a result, these temporarily connected ponds had fish communities similar to ponds with direct connections to the main marsh (Table 3-8).

In 2001, average Beamish trap catches (number/trap) varied from a high of 8,170 fish in North Mackenzie Bay, to a low of 8 fish in North School Bay (Table 3-9). The following year, average trap catches (number/trap) ranged from a high of 592 (Section 5 Bay) to zero fish (North Mackenzie Bay) (Table 3-10). As previously mentioned, fish numbers from Thompson's Bay were highly inflated due to trap location in 2002, and were not considered meaningful for comparison among ponds. For most ponds, fewer fish were caught in 2002 than in 2001 (Table 3-7), however, North School Bay and Section 5 Bay were exceptions; very low numbers in 2001 (Table 3-9) compared to much higher numbers in 2002 (Table 3-10).

In 2001, fathead minnows were the most abundant fish species in four study ponds (Table 3-9), including temporarily connected Wye's Pond, North Mackenzie Bay and Emile's Pothole, and naturally connected South Pitblado's Channel. Yellow perch were the most abundant species in three connected ponds (Thompson's Bay, Mid-Blind Channel and Mad Woman Bay). Brook sticklebacks were the dominant species in two study ponds; Section 5 Bay and South Mackenzie Bay (Table 3-9). Northern pike were the most abundant species in North School Bay. In general, ponds with higher fathead minnow and/or brook stickleback abundances corresponded to a lower proportion of yellow perch and northern pike in the population (Table 3-9). In 2002, fathead minnows were the most commonly captured species in North School Bay, Mid-Blind Channel, Mad Woman Bay, South Pitblado's Channel, and South Mackenzie Bay (Table 3-10). Common carp dominated the catch in a single screened (Section 5 Bay) and blasted pond (Wye's Pond). Yellow perch were most abundant in Thompson's Bay, and brook stickleback dominated the catch in Emile's Pothole. It is important to note, no fish were captured in North Mackenzie Bay in 2002 (Table 3-10).

3.6.3.2 Pond Treatments and Fish

In 2001, high water levels on Delta Marsh resulted in all study ponds having some water connection to the main marsh, and fish communities were relatively similar regardless of the permanence of the connection (Tables 3-9, 3-10). Numbers of species collected ranged from five to 11 across the 10 ponds, with the lowest number of species in one permanently connected pond (North School Bay) and one temporarily connected pond (South Mackenzie Bay) (Table 3-9). Highest number of species was found in a temporarily connected pond (Wye's Pond) (Table 3-8). Average catch in Beamish traps was higher in temporarily connected ponds (mean = 2,911) compared to permanently connected ponds (mean = 640), but variation was high across all ponds (Table 3-9).

In 2002, the pond manipulations began. Ponds which were temporarily connected to the main marsh in 2001 became naturally isolated in 2002 due to lower water levels in Lake Manitoba. Generally, decreased connection to the main marsh resulted in lower fish species diversity and abundance. Across the 10 ponds, the numbers of species collected ranged from zero to 12, with the lowest number of species found in isolated North Mackenzie Bay (Table 3-8). The highest number of species were found in Thompson's Bay (connected pond), Mid-Blind Channel (screened pond), and South Mackenzie Bay (blasted pond) (Table 3-8). Average catch (number/trap) in Beamish traps was highest in connected ponds (mean = 3,633), followed by screened ponds (mean = 415), blasted ponds (mean = 216), diked ponds (mean = 155) and isolated ponds (mean = 14) (Table 3-10).

The dominant species in Thompson's Bay in 2001 was yellow perch (Table 3-9). Other commonly captured species included spottail shiners and fathead minnows. Yellow perch were also dominant in 2002, but fathead minnows and common carp were also abundant (Table 3-10). In North School Bay, the other permanently connected pond that was not altered, northern pike were the dominant species in 2001, and this was the only pond in which northern pike was an important part of the fish community in either year. In 2002, fathead minnows and common carp were abundant species in this pond (Table 3-10). Number of fish species increased in both ponds in 2002. Thompson’s Bay increased from nine to 12 species, and North School Bay increased from five to 11 species (Table 3-8). Although I cannot say anything about the numbers of fish captured in Thompson’s Bay in 2002, due to problems related to trap position, numbers of fish captured in North School Bay showed a 65 times increase between years.

North Mackenzie Bay and Emile's Pothole were temporarily connected in 2001, but isolated in 2002. The flooding in 2001 had allowed fish access to these ponds, resulting in a diverse fish fauna characteristic of connected ponds (Table 3-9). North Mackenzie Bay had the highest number of fish of any pond sampled in 2001 (8,170 fish/trap on average) (Table 3-9). Fathead minnows and black bullheads composed 98% of the total catch. Common carp, brook stickleback, johnny darter, northern pike and yellow perch were also captured (Table 3-9). Emile's Pothole had less than one tenth the number of fish found in North Mackenzie Bay. Fathead minnows were the dominant species, followed by brook stickleback and common carp. With the decline in water levels in 2002, North Mackenzie Bay and Emile's Pothole were isolated from the main marsh, which resulted in a marked change in the fish community of these ponds (Table 3-10). Emile’s Pothole had a much simpler community, dominated mostly by brook sticklebacks and fathead minnows. North Mackenzie Bay had no fish present 2002 (Table 3-10).

South Mackenzie Bay and Wye's Pond were temporarily connected in 2001, but permanently connected with new channels in 2002. Even temporary connections resulted in an abundant and diverse fish community being present in 2001. Wye's Pond had the second highest number of fish captured in 2001, and the highest abundance of white sucker of all ponds sampled that year (Table 3-9). Combined, fathead minnows, white sucker and yellow perch accounted for nearly 91% of the total number of fish (Table 3-9). In 2001, South Mackenzie Bay had only a fifth the number of fish found in Wyes' Pond. Brook sticklebacks were the dominant species, although fathead minnows, common carp, yellow perch, and northern pike were also captured (Table 3-9). In 2002, soon after new channels were created by blasting, South Mackenzie Bay and Wye's Pond quickly developed fish assemblages characteristic of naturally connected ponds (Table 3-10). As previously mentioned, blasted ponds had similar numbers of fish species as connected ponds (Table 3-8). Surprisingly, numbers of fish per Beamish trap actually declined in both ponds after they became permanently connected (Table 3-9), compared to when they were temporarily connected the previous year (Table 3-10). Similar to other connected ponds, fathead minnows were the dominant species, accounting for 67.5% of the average Beamish trap catch in South Mackenzie Bay (Table 3-10). Common carp, black bullheads and brook sticklebacks were also frequently encountered species. In Wye's Pond, common carp was the dominant species accounting for 41.5% of the total catch, followed by black bullheads and fathead minnows (Table 3-10).

In 2001, when South Pitblado's Channel was naturally connected to the main marsh it had a total of 10 species present, whereas Mad Woman Bay had eight (Table 3-8). In South Pitblado's Channel, fathead minnows composed 90% of the average Beamish trap catch (Table 3-9). Other commonly captured species included brook stickleback, yellow perch, white sucker and common carp. Yellow perch and black bullhead were the dominant species caught in Mad Woman Bay in 2001 (Table 3-9). Sandbag dikes, constructed in early spring of 2002 at Mad Woman Bay and South Pitblado's Channel, were an attempt to isolate these ponds from the main marsh. As noted above, fish were already moving into the South Pitblado's Channel before the dike was completed, and the dike at Mad Woman Bay was only partially effective at isolating that pond. Despite the presence of northern pike, the fish community of South Pitblado's Channel resembled that of naturally isolated ponds, with the only other fish species present being fathead minnows and brook sticklebacks (Table 3-10). In addition, numbers of fish in South Pitblado's Channel also declined from 2001 to 2002 (Table 3-10). Mad Woman Bay had a more diverse fish community in 2002, with nine species present, likely a result of the poorly functioning dike. Similar to South Pitblado's Channel, fathead minnows and brook sticklebacks were the most abundant species, however northern pike, spottail shiner, emerald shiner (*Notropis atherinoides*), black bullhead, yellow perch, white sucker and central mudminnow were also captured in Mad Woman Bay in 2002 (Table 3-10).

The two screened ponds, Section 5 Bay and Mid-Blind Channel, had different fish assemblages and abundances in 2001 and 2002 (Tables 3-9, 3-10). While connected to the main marsh, Mid-Blind Channel had an average 2,064 fish (average number/trap) in Beamish traps, compared to only 21 fish in Section 5 Bay (Table 3-9). In Mid-Blind Channel, yellow perch were the most frequently encountered species, followed by white sucker and fathead minnows. Brook stickleback, common carp and yellow perch dominated the fish community in Section 5 Bay (Table 3-9). In 2002, screens were effective for the most part at preventing access by adult common carp, although a few may have breached the conduit fence at Section 5 Bay (Candace Parks, personal observation). In 2002, both ponds responded differently to screening. Average Beamish trap catch increased by 28 times in Section 5 Bay, whereas Mid-Blind Channel showed a 90% decline in numbers (Table 3-10). Section 5 Bay had more than twice as many fish per Beamish trap than Mid-Blind Channel. The increase in fish numbers in Section 5 Bay was largely due to increased numbers of common carp and fathead minnows. In Mid-Blind Channel, yellow perch, and to a lesser extent white sucker, were responsible for the significant decline in fish numbers in 2002 (Table 3-10).

3.6.4. Multivariate Statistics

3.6.4.1 Invertebrates Principal Component Analysis (PCA)

The principal component analysis (PCA) shows the distribution of water-column invertebrate taxa in the ten study ponds in 2001 and 2002, with 66.8% of the variation in invertebrate community structure explained by the first axis (Figure 3-4). The primary axis showed a trend based on connectivity with the main marsh. Ponds with a connection (on the right of the bi-plot) had a reduced invertebrate assemblage while ponds on the left corresponded to a more diverse community. With a lack of environmental data, it is difficult to determine what factors were affecting the second axis. In Figure 3-4, all 17 invertebrate taxa are located on the far left of the ordination (negative association), with the most influential taxa being the Dytiscidae. Meanwhile, phantom midges (Chaoboridae) were most influential on the second axis of the ordination.

A preliminary RDA using the same invertebrate/pond data set with limited environmental data (*i.e.*, average pond depth, percent cover of aquatic macrophytes, water temperature and photic depth) were analyzed. Water temperature was not an important factor in describing additional variability in the triplot. Percent aquatic macrophyte cover and average water depth were highly correlated with each other, and positively correlated with the primary axis. However, they did not help explain any additional variation already accounted for in the PCA biplot. Thus the data were omitted from further analyses. As a result, factors controlling the placement of points along the second axis in the ordination diagram were unknown.

In Figure 3-4, all 17 invertebrate taxa were located on the far left of the ordination (negative association), with the most influential taxa being the Dytiscidae. Meanwhile, phantom midges (Chaoboridae) were most influential on the second axis of the ordination.

Four groups of study ponds were apparent on the bi-plot. The first group corresponds to ponds closely associated with the water-column invertebrates (Figure 3-4). They include ponds which were naturally isolated [North Mackenzie Bay (2002), Emile's Pothole (2002)], diked [South Pitblado's Channel (2002), Mad Woman Bay (2002)], or screened [Mid-Blind Channel 2002] from the main marsh The majority of ponds with a close association to water-column invertebrates appear to have either no direct connection, or a reduced (screened) connection, to the main marsh. However, North School Bay (2001), Section 5 Bay (2001) and Wye's Pond (2002) were exceptions. Despite being connected to the main marsh, the ordination places these ponds in closer association with diked, fenced and isolated ponds. South Mackenzie Bay (2001) lies somewhat in equidistant from the two main groups of ponds showing signs that this pond was intermediate between isolated and connected ponds, and thus grouped separately (Figure 3-4). South Mackenzie Bay (2002) forms a third group. Again, this pond is equidistant between the two main groups, but is plotted higher than the other ponds, due in part to higher abundances of Chaoboridae in 2002 (Figure 3-4). The last group, placed on the right of the ordination diagram, characteristically had lower numbers of water-column taxa, and included ponds connected to the main marsh naturally [Thompson's Bay 2001, 2002, Mid-Blind Channel 2001, South Pitblado's Channel 2001, Mad Woman Bay 2001, North School Bay 2002, and Section 5 Bay 2002], or temporarily by flooding [Wye's Pond 2001, Emile's Pothole 2001 and North Mackenzie Bay 2001] (Figure 3-4). With a few exceptions, the PCA also partitioned sampling years into separate groups. The majority of ponds in 2001 are positively associated with the first axis, whereas 2002 ponds are negatively associated with the first axis. Without environmental data, it is difficult to determine why certain ponds were grouped the way they were.

Changes in water-column invertebrate composition between 2001 and 2002 can be compared by connecting the points corresponding to each individual pond (Figure 3-4). For example, isolated ponds (North Mackenzie Bay, Emile's Pothole), which were temporarily connected to the main marsh for part of 2001, showed signs of a reduced invertebrate assemblage as indicated by their placement on the right-hand side of the ordination. In 2002, these ponds had no connection to the main marsh and were now plotted on the left side of the ordination. The same pattern is also apparent for both diked ponds.

However, two ponds did not follow this overall pattern. North School Bay, a connected pond in both years, had relatively high numbers of water-column invertebrates in 2001 but was placed on the left side of the ordination with ponds that were isolated (Figure 3-4). In 2002, it was plotted on the right side with the other connected ponds. Fewer planktivores (*e.g.*, small fish species such as fathead minnows) in this pond may be partly responsible for the high numbers of invertebrates in 2001, compared to the other connected ponds. Wye's Pond was temporarily connected in 2001, and fully opened to the main marsh by blasting in 2002. Rather than being plotted on the right side for both years, it showed a shift to the left in the second year, opposite to the pattern seen in most other ponds (Figure 3-4). Although this pond is described as opened to the main marsh in 2002, the channel was actually blasted in late June. Wye's Pond would have been an isolated pond for the first part of the field season, and the invertebrate community responded as found in the other isolated ponds.

3.6.4.2 Fish Correspondence Analysis (CA)

The correspondence analysis (CA) depicted a gradient partitioning fish species based on connectivity to the main marsh (Figure 3-5). The first ordination axis accounted for 28.1% of the variance in the data, while the second axis 13.8%. Brook sticklebacks, central mudminnows and northern pike were closely coupled with ponds positively associated with the primary axis (Figure 3-5). These fish were typically found in North Mackenzie Bay (2001), Section 5 Bay (2001), Mad Woman Bay (2002), South Mackenzie Bay (2001), Emile's Pothole (2002), South Pitblado's Channel (2002), and North School Bay (2001). The remaining ponds [Mad Woman Bay (2001), Thompson's Bay (2001 and 2002), Wye's Pond (2001 and 2002), Blind Channel (2001 and 2002), North School Bay (2002), South Mackenzie Bay (2002), Section 5 Bay (2002), and South Pitblado's Channel (2001)] had a more seasonally diverse fish community (Figure 3-5). Fish species found near the origin of the bi-plot (*i.e.*, common carp and fathead minnows) were ubiquitous and found within most study ponds. North Mackenzie (2002) does not appear on the CA bi-plot given that the pond was fishless that year (Figure 3-5).

The ponds separated in two main groups; the group on the right has a reduced fish assemblage averaging three to nine species, which is more indicative of isolated ponds while the group on the left has a more diverse fish assemblage ranging from 6 to 12 species, which is characteristic of connected ponds (Figure 3-5). Of the ponds in the right-hand side, North Mackenzie Bay (2001), Section 5 Bay (2001), and Mad Woman Bay (2002), have a higher number of species (seven to nine species) within this group and an abundance of fathead minnows and brook sticklebacks (Figure 3-5). Section 5 Bay (2001) is closely associated with Mad Woman Bay (2002) due to the presence central mudminnows. South Mackenzie Bay (2001), Emile's Pothole (2002) and South Pitblado's Channel (2002) were grouped in the lower right corner, due to their reduced fish assemblage of three to five species (Figure 3-5). Brook sticklebacks were the dominant species in these ponds, however central mudminnows and northern pike were captured in low abundances in Emile's Pothole (2002) and South Pitblado's Channel (2002), respectively. Lastly, North School Bay (2001) had a total of five species, yet unlike any other pond, northern pike were the dominant species. Consequently it was placed alone in the upper right corner of the ordination (Figure 3-5).

Ponds with greater fish diversity, ranging from six to 12 species, were tightly grouped on the left side of the ordination, with the exception of Wye's Pond, which is pulled away from the group by the presence of trout-perch (Figure 3-5). When points on the CA were linked between successive sampling years, the majority of the patterns align with the primary axis representing various stages of connectivity to the main marsh (Figure 3-5). Emile's Pothole, South Pitblado's Channel's, Mad Woman Bay and Mid-Blind Channel shifted from left to right on the ordination, corresponding to a reduction in connection to the main marsh. The length of each vector linking consecutive years indicates the degree of change in fish abundance and diversity between the two years. Thus Mid-Blind Channel, which was naturally connected in 2001 and then screened 2002, had a minimal change, compared to South Pitblado's Channel, Mad Woman Bay and Emile's Pothole. These ponds demonstrated a larger shift in species composition. For example, South Pitblado's Channel had a total of 10 species when connected to the main marsh in 2001, but was subsequently reduced to three species when the pond was isolated (diked) the following year. Species such as fathead minnows, brook sticklebacks and central mudminnows were captured in 2002 (Table 3-10)

In contrast, ponds such as Section 5 Bay, South Mackenzie Bay, and North School Bay shifted from the less diverse group on the right in 2001, to the more diverse group on the left in 2002 (Figure 3-5). In South Mackenzie Bay, this increase in number and diversity of species was expected, since the pond went from being temporarily connected in 2001, to being permanently connected in 2002. North School Bay was somewhat of an anomaly; unlike other connected ponds in the marsh, the water in the pond was relatively clear and it had a greater abundance of aquatic macrophytes (Candace Parks, personal observation). This pond remained in its natural state for the entire study. However in 2001, it had a reduced number of small fish species, yet had the largest abundance of YOY northern pike than any other pond (Table 3-9). The following year, species richness increased dramatically from five species to 11 in 2002 (Table 3-8). Finally, the shift of Section 5 Bay on the ordination is attributed to an increased abundance of common carp and yellow perch when the pond was screened, relative to the previous year (Tables 3-8, 3-19, 3-10 and Figure 3-5).

Wye's Pond and Thompson's Bay did not align as well to the primary axis. Vertical displacement of the vectors of these ponds indicates that the species composition did not change dramatically between the two years (Figure 3-5). Specifically, the negative displacement of Wye's Pond in 2001 is attributed to the abundance of rare trout-perch, relative to 2002 (Table 3-10). Since this pond was temporally connected to the main marsh in 2001, the newly created channel created in 2002 did not greatly change the overall fish composition (Figure 3-5). Similarly, little change was observed in Thompson's Bay between years (Figure 3-5). This pond had a high number of species in both years (Table 3-8).

3.6.4.3 Effects of Environmental Variables

When using water-column invertebrates a biotic variable to explain variation in the fish data, the sum of the eigenvalues was 0.88, and the correspondence correlations were 0.22 for the first axis and 0.15 for the second axis. When these values were compared to the eigenvalues from the fish only CA (Figure 3-5), the values were similar, indicating water-column invertebrates did not help explain additional variation in the fish ordination. The same was true when fish were used as a biotic variable on the invertebrate PCA. A Redundancy Analysis (RDA) of the invertebrates, with fish as a biotic variable, did not provide any additional explanation of the variation of the water-column invertebrates. This project was a collaborative effort and additional environmental data were collected by a colleague. Future analyses with this additional information may help to explain additional variation in water-column invertebrates and fish distributions in the study ponds.

It is interesting to note, with the numerous multivariate analyses performed on the data (*i.e.*, PCA and RDA versus CA and CAA), results in the response variables were not significantly different among the analyses, suggesting the initial data matrix was robust and not affected greatly by different ordination methods.

3.7 Discussion

3.7.1 Pond Connectivity, Water Levels and the Fish Community of Delta Marsh

The fish community of Delta Marsh is characterized by a diversity of seasonal fish species (see Chapter 2), some of which opportunistically utilized smaller peripheral ponds when physical connection and/or increased water levels permitted. Increased connection to a larger water body has been shown to be an important factor in structuring the fish communities in numerous habitats, including coastal wetlands (Herdendorf et al. 1986; Jude and Pappas 1992; Lougheed and Chow-Fraser 2001), small lakes (Tonn and Magnuson 1982), swamps (Carlson and Duever 1978 In: (Clark 1978), temporary ponds (Baber et al. 2002), mangroves and salt marshes (Clark 1978). Delta Marsh is subjected to both long-term (*i.e.*, flooding, drought) and short-term (*i.e.,* wind driven set-up or set-down events) water level fluctuations (Batt 2000). These fluctuations affect connectivity of the many channels, ponds and bays within the marsh, and this in turn impacts fish movement and habitat use.

At Delta Marsh, connectivity is important at several different scales. For the entire marsh, the four channels between the marsh and Lake Manitoba enable many fish to use the marsh that would otherwise not survive the harsh winter conditions that occur in shallow wetlands at northern latitudes (Tonn and Magnuson 1982). Delta Marsh supports 30 species of fish (see Chapter 2), but most are unable to overwinter in the marsh, and must return to Lake Manitoba in the fall prior to winter freeze-up (Gee 1975; Lapointe 1986). Without these connections, the marsh would have a very limited fish community of tolerant smaller-sized species (*i.e.*, mudminnows, sticklebacks, and certain cyprinids).

On a smaller scale, connectivity impacts the fish community of the individual ponds and bays of Delta Marsh. Isolated ponds with no connection to the main marsh have either a limited assemblage of smaller-sized species such as fathead minnows and brook sticklebacks, which are resilient to high water temperatures and low oxygen levels (Suthers and Gee 1986; Peterka 1989), or no fish (see Chapter 2). Ponds with permanent connections, either natural or man-made, have a much more diverse fish community (up to 12 species). When water levels rise in the marsh, isolated ponds may become connected to the main marsh and fish have access to these newly connected habitats. In these ponds, fish populations can vary considerably from year to year, and these changes are largely dependent upon water levels (Kushlan 1976; Weller 1978; Kushlan 1980).

In northerly latitudes, fish opportunistically use coastal wetlands due to the increased food resources, attributed to the high levels of productivity, and increased abundance and diversity of habitats (Stephenson 1988). High productivity and habitat diversity are maintained by fluctuations in water levels (Murkin and Ross 2000). Thus, water level fluctuations are extremely important in coastal wetland habitats. Not only do these fluctuations help maintain overall wetland productivity, they also are important in determining fish use of these habitats.

Seasonal movements of fish are largely based on the timing and duration of connections (Clark 1978; Liston and Chubb 1985; Snodgrass et al. 1996; Poizat and Crivelli 1997; Baber et al. 2002). In the spring, fish move from Lake Manitoba into Delta Marsh and the smaller ponds when conditions are optimal. Lapointe (1986) found many fish species routinely travel back and forth between the marsh and Lake Manitoba during the open water season. Thus inside the marsh, fish use connections to move back and forth between the ponds and the larger bays when conditions deteriorate. Subsequently, fish move into Lake Manitoba to overwinter to avoid severe marsh conditions (Gee 1975; Lapointe 1986).

High water levels in the marsh in spring 2001 likely contributed to the higher numbers of YOY fish found that year. High water flooded new habitat and provided connections to other habitats that otherwise were unavailable. These new habitats provided additional spawning areas and resulted in an increased spawning effort by fish within the marsh (Kushlan 1980; Liston and Chubb 1985; Jude and Pappas 1992).

North School Bay was unusual among study ponds because of the high abundance of northern pike in 2001. North School Bay was the only study pond that was part of a larger drainage system south of Delta Marsh. There was considerable flow through the pond for much of the open water season in 2001. The increased outflow, a result of snow melt and rainfall within the drainage basin, likely attracted spawning northern pike, more so than in other ponds. North School Bay did not experience similar outflow in 2002, nor did the other nine ponds at any time during the study. Surprisingly, fathead minnows were absent in Beamish traps in North School Bay in 2001, and increased predation by northern pike may have caused this.

During flooding, fish can move into new habitats from adjacent permanent habitats by overland flow (Hanson and Butler 1990). Such was the case with the four isolated ponds that became temporarily connected in 2001. In 2002, high water levels caused water to flow around the dike at Mad Woman Bay allowing movement of fish into the pond. This led to unexpectedly high fish diversity (nine species) compared to other isolated ponds. Alternatively, delays in establishing connections may have lessened the chance for certain species to access seasonal habitats. In Manitoba, the majority of fish spawn between early April and late June (Stewart and Watkinson 2004). In our blasted study ponds, the timing of new channels into South Mackenzie Bay and Wye's Pond likely influenced how fish used these ponds in 2002. South Mackenzie Bay's channel was created on May 17 but Wye's Pond's channel was not blasted until nearly a month (28 days) later. Considering the numbers and diversity of species that were found in these ponds in 2001, when they were temporarily connected to the main marsh by flooding, many fish species that opportunistically use connected ponds as spawning sites could have already spawned elsewhere.

Of interest, I found high inter-annual variation in the fish assemblages among the ponds regardless of the experimental treatments. Increases in spring and summer water levels resulted in increased YOY fish abundances of all study ponds in 2001. Elsewhere, increased water levels has shown to lead to improved spawning success and stronger year classes of many species including cyprinids, yellow perch, northern pike (Liston and Chubb 1985) and largemouth bass (*Micropterus salmoides*) (Miranda et al. 1984). My results support these findings. On average, nearly seven times more fish were captured in Beamish traps in 2001 when water levels were higher compared to 2002 when levels were lower. Thus increased water levels in Delta Marsh may have led to improved spawning habitat in terms of flooded terrestrial habitat around the marsh (*e.g.*, northern pike) and access to previously isolated habitats (*i.e.*, isolated ponds) which resulted in increased spawning effort and greater fish numbers in 2001.

3.7.2 Invertebrates, Pond Connectivity and Fish

At Delta Marsh, high water levels in 2001 resulted in fish access to all study ponds, including the four isolated ponds. High numbers of YOY fish were found, likely due to increased spawning effort by fish throughout the marsh (see above). The high numbers of fish in 2001 may be responsible for the lower numbers of water-column invertebrates found that year. In 2002, when water levels receded, fish numbers declined in almost all ponds, and invertebrate numbers increased in all study ponds, even those with no experimental manipulations. The most diverse invertebrate assemblages were found in ponds with either no fish, or those with a reduced fish community of fathead minnows, brook sticklebacks, and to a lesser extent, central mudminnows. In addition, higher numbers of large mobile invertebrate taxa (*e.g.*, Amphipoda, Dytiscidae) were found in 2002, compared to 2001.

Invertebrate composition can be affected by fish (Clark 1978; Liston and Chubb 1985; Jude and Pappas 1992; Mallory et al. 1994; Hanson & Riggs 1995; Zimmer et al. 2000; Zimmer et al. 2001; Laurich et al. 2003). This may well be the case in our study ponds at Delta Marsh. High numbers of fish in 2001 resulted in increased predation, and reduced numbers and diversity of the water-column invertebrate community. Reduced fish numbers in 2002, resulted in an increase in the invertebrate community. Lapointe (1986) and Hann (1999) also found significant reductions in abundance of several water-column invertebrate taxa in the presence of fish during experiments conducted in Delta Marsh.

In addition, higher numbers of larger, more mobile water-column invertebrate taxa were found in ponds without fish. In 2002, large-bodied Amphipoda, Conchostraca, and Dytiscidae were most abundant in fishless North Mackenzie Bay and yet were nearly absent from this pond when it was temporarily connected in 2001. This suggests that fish are size-selective predators and that the physical size of the invertebrates were impacted by the presence of fish. For example, Pont et al. (1991) found three-spined sticklebacks, which colonized formerly fish-free environments, preferentially fed on more conspicuous, larger-sized zooplankton and later switched to smaller prey items in temporary shallow marshes. An overall decrease in fish abundance in study ponds in 2001 likely coincided with an invertebrate community composed of larger-sized taxa (*i.e.*, Amphipoda).

In this study, pond connectivity and fish communities were affected by water levels. Although changes found in the invertebrate community may be in response to changes in the fish community, they may also be in response to other factors that were affected by water levels. For example, water levels may have affected the growth and abundance of aquatic macrophytes, since deeper water may limit the amount of light reaching the plants. Aquatic macrophytes play a crucial role in regulating predator-prey interactions (Gilinsky 1984; McPeek 1990) and invertebrate habitat (Voigts 1976; Hann 1999). This study was a part of a larger collaborative effort with a companion project undertaken by Stacy Hnatiuk, a Botany graduate student with the University of Manitoba. She concurrently examined water quality, and algal and aquatic macrophyte productivity, abundance and diversity within the study ponds. When published, this information will hopefully provide further insight into the factors that affected the aquatic invertebrate community.

Although most ponds responded as discussed above, several connected ponds did not. Water-column invertebrate communities in North School Bay, with a permanent channel in 2001, and Wye’s Pond with a newly created channel in 2002, were similar to those found in isolated ponds. In North School Bay, the high prevalence of northern pike in 2001 may have reduced the number of small-sized planktivores (*e.g.*, fathead minnows) (Table 3-9), leading to higher invertebrates numbers (Table 3-3). In Wye's Pond in 2002, the delay in blasting the channel likely lessened the opportunity for spawning fish to enter the pond. By June 14, the majority of marsh fish species had spawned, lessening the impact of adult or juvenile fish predation on invertebrate taxa.

3.7.3 Effects of Pond Connectivity and Fish on Amphibians

In my study ponds, increased connectivity was associated with lower abundances of amphibians. In 2001, when water levels were high and all ponds had some connection to the main marsh, only northern leopard frogs were found, and predominantly in the four temporarily connected ponds. In 2002, with declining water levels and the experimental manipulations, three species of amphibians were present, and in much higher numbers, particularly in the isolated, diked and screened treatments.

The link between amphibian numbers and pond connectivity was probably due to increased access by fish. Fish are known predators of amphibians, consuming eggs, larvae and adults (Semlitsch 1988; Skelly 1996; Smith et al. 1999), and can significantly impact amphibian populations (Merrell 1977; Teplitsky et al. 2003; Hecnar 2004). Fish can be especially detrimental when introduced into previously fish-free habitats during periods of flooding, overland flow (Hecnar 2004) or by human activities. However, not all fish are predators of amphibians. High numbers of amphibians can be found in habitats with communities of small fish, such as planktivorous fathead minnows and brook sticklebacks (Lehtinen et al. 1999). Gape size prevents these two species from becoming predators of anuran tadpoles (Held and Peterka 1974) or on salamanders and their eggs (Deutschman and Peterka 1988). Such was the case at Delta Marsh, where high numbers of amphibians were present in ponds with abundant fathead minnow and brook stickleback populations.

There are also several indirect ways in which fish can impact amphibian populations. Fish consume large-bodied, predaceous invertebrates such as dytiscids, odonate nymphs, notonectids, belostomids and larval caddisflies (Phryganeidae). These invertebrates are known to feed on larval amphibians (Merrell 1977; Smith et al. 1999). By reducing predaceous invertebrates, fish may in turn reduce predation pressures on larval amphibians. In my study ponds, increased amphibian numbers were also associated with higher numbers of predaceous invertebrates, and are therefore not thought to be important predators of amphibians in Delta Marsh.

Fish can also indirectly impact amphibians through alterations to their habitat. Common carp, for example, are known to eliminate submersed aquatic vegetation through their feeding and spawning activity (Cahn 1929; Threinen and Helm 1954; Robel 1961; King and Hunt 1967; Crivelli 1983). Numbers of amphibians increased in the screened ponds although not to the extent found in the isolated ponds in 2002. The higher numbers in the isolated ponds are likely due to reduced predation as noted above. Increased numbers of leopard frog tadpoles in the screened ponds may be due to either reduced numbers of large predacious fish, such as large northern pike that were unable to pass through the screens, or to exclusion of adult common carp and the greater abundance of aquatic macrophytes within these ponds.

Wye's Pond was an exception to my overall observations. Amphibian numbers were high in this pond, even though it was connected to the main marsh. This unusual result may be a function of the timing of blasting and channel creation. Wye's Pond was not connected to the main marsh until mid-June, when the soil was thawed enough to permit blasting. This later date likely allowed amphibians to spawn in the pond prior to its connection to the main marsh. Consequently, tadpoles were already present in the pond when fish were allowed access.

3.7.4 Common Carp

This study was initially undertaken to determine the effects of common carp on native fish, amphibians and water-column invertebrates in Delta Marsh. Common carp have been implicated as contributing to habitat deterioration by increasing water-column turbidity (Cahn 1929; Hanson and Butler 1990; Meijer et al. 1990; Chow-Fraser et al. 1998), uprooting of aquatic macrophytes (Cahn 1929; Threinen and Helm 1954; Robel 1961; King and Hunt 1967; Crivelli 1983), reducing large-bodied zooplankton and benthic invertebrates (Scheffer 1998), decreasing native fish diversity and abundances (Craig and Babaluk 1989) and competing with waterfowl for food resources (Mallory et al. 1994; Bouffard and Hanson 1997). However, it became apparent that during the experimental manipulations, altered pond connectivity had a larger influence on the study ponds, more so than simply altering the presence or absence of common carp. My results implicate pond connectivity as having a significant impact on native fish, amphibians and water-column invertebrates, but the relative contribution of common carp to these changes were difficult to determine.

I found that common carp were an important part of the fish community of peripheral ponds of Delta Marsh. Common carp accounted for 35% of the overall catch in 2002. In half of the study ponds, they ranked as either the first, second or third most abundant species. In a single pond (Section 5 Bay), common carp represented 76.9% of the average fish catch in 2002. This is consistent with what we know of the larger fish community in the main marsh, where common carp are a dominant species (Dale Wrubleski, Ducks Unlimited Canada, personal communication). Overall, common carp comprise a large proportion of the overall fish community of Delta Marsh. Unlike most other fish species in the study ponds, however, common carp were more abundant in 2002, when water levels were lower, than in 2001 when water levels were higher. Common carp can spawn more than once in a season when conditions are optimal, and preferentially use shallow waters that warm quickly (Sigler 1958; McCrimmon 1968). Therefore, low water levels may benefit common carp in Delta Marsh. In Australia, reductions in water level fluctuations have been shown to reduce spawning by native fish species but benefit common carp (Harris 1996).

In this study, changes observed in diked, blasted, connected and isolated ponds are most likely explained by pond connectivity. The screened ponds, however, are most likely to show how exclusion of adult common carp might impact Delta Marsh. Mid-Blind Channel and Section 5 Bay showed improved water clarity and increased aquatic macrophyte abundances (Stacy Hnatiuk, unpublished data; Candace Parks, personal observation), and also showed increases in water-column invertebrate and amphibian (leopard frog tadpole) numbers. However, fish numbers actually declined after screening in Mid-Blind Channel, but fish numbers also declined in most study ponds. Section 5 Bay was an exception, showing an increase in fish numbers in 2002, primarily due to increased numbers of YOY common carp. Although invertebrate numbers increased in both ponds following screening, invertebrate abundances increased in all ponds in 2002, even those that were not manipulated. Amphibian numbers increased in both screened ponds, but they also increased in both isolated and diked ponds, as well as one blasted pond. Increased numbers in the blasted pond (Wye's Pond) may be due to timing of connection, as explained above. Amphibian numbers increased slightly in one permanently connected pond and remained absent in the other. Amphibian responses in the isolated and diked ponds may be explained by lower numbers of predaceous fish. In the screened ponds, smaller predaceous fish (*i.e.*, those that could pass through the 5 cm gaps in the screens) would have been present, but do not appear to have had an impact on amphibian numbers. The amphibian response in the two screened ponds can only have occurred as a result of the exclusion of large predatory fish (*e.g.,* large northern pike) or the exclusion of adult common carp. Smaller adult northern pike and other predaceous fish would have passed through the screens (French III et al. 1999), and are likely to have been present in the ponds. So their presence had little or no impact on amphibians. Instead, it would seem most likely that the exclusion of adult common carp contributed to the increased numbers of amphibians in the screened ponds in 2002.

My results provide partial support for the effects of common carp on amphibians, and little evidence for their effects on native fish and water-column invertebrates due to confounding influence of connectivity. On the whole, my study has shown that annual variation in water levels may be a stronger driving force than common carp. Large-scale changes in water levels over the entire marsh may have confounded the small-scale changes in the study ponds. Furthermore, I monitored only the first year of manipulations and it may take more than one summer to show effects. At Cootes Paradise Marsh, climatic variations initially confounded the results of a common carp exclusion project (Lougheed et al. 2004).

To further examine the impacts of common carp at Delta Marsh, I would suggest continuing with natural field studies, rather than proceed with studies that are under artificial conditions (*i.e.*, small mesocosms or enclosures). Experimental enclosures and exclosures have been designed specifically to determine the degree of damage to aquatic vegetation by common carp (see Tryon 1954; Robel 1961; Crivelli 1983). However my results, and those of Badiou (2006) and Lougheed et al. (2004), conducted in natural ponds, large wetland cells, or small wetlands respectively, used a study design that more closely resembled natural wetland conditions and the true effects of common carp.

The role of common carp in the decline of native fish is not fully understood (Koehn et al. 2000). Despite much attention, there is little scientific evidence common carp cause reductions in native fish populations (Harris 1994). In Delta Marsh and elsewhere, it is inconclusive as to whether common carp are a major cause of fish declines. Several studies in Australia have concluded that reductions of native fish species occurred before increases in the catch of common carp (see Derwent 1994, Clunie and Koehn 1998).

I was also unable to attribute any effects of common carp on water-column invertebrates. As omnivores, common carp may have less of an effect because of their ability to shift to alternate prey species or food items, thereby enabling them to adapt to a variety of habitats. Diehl (1992) suggests that the effects of omnivorous fish on invertebrates is complex, and their effects may be difficult to identify due to the potential variety of prey taxa involved. Numbers of invertebrates increased in all ponds in 2002 when common carp numbers also increased. These findings further imply that common carp are a symptom of poor habitat conditions within Delta Marsh, but are not a cause.

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TABLES AND FIGURES

Table 3-1. Mean number (number/trap) of invertebrates captured per activity trap in Delta Marsh between early June and late August in 2001 and 2002.

	Numbers			
	2001		2002	
Species	**Mean**	**%**	**Mean**	**%**
Amphipoda	0.8	0.5	16.7	0.8
Conchostraca	0.0	0.0	2.2	0.1
Ostracoda	1.1	0.7	11.9	0.5
Cladocera	54.3	34.1	1,639.9	75.3
Copepoda	72.4	45.5	306.2	14.1
Corixidae	16.4	10.3	113.2	5.2
Notonectidae	t[1]	t	3.7	0.2
Dytiscidae	0.1	0.0	8.7	0.4
Gyrinidae	0.1	0.1	0.1	t
Haliplidae	t	t	0.5	t
Ephemeroptera	0.1	0.1	1.7	0.1
Trichoptera	t	t	1.5	0.1
Chaoboridae	t	t	0.5	t
Odonata	0.1	0.1	0.7	t
Hydracarina	12.5	7.9	37.2	1.7
Gastropoda	1.2	0.7	32.7	1.5
Hirudinea	t	t	1.4	0.1
Totals	159.3	100.0	2,178.6	100.0

1 - t = trace (<0.1 or 0.1%)

Table 3-2. Taxa richness, effective richness and evenness of water-column invertebrate communities sampled in 10 study ponds in Delta Marsh, 2001 and 2002.

	2001				2002			
Experimental Ponds	**Treatment**	**Taxa Richness** ***s***	**Effective Richness** N_2	**Evenness %** E_3	**Treatment**	**Taxa Richness** ***s***	**Effective Richness** N_2	**Evenness %** E_3
Thompson's Bay	connected	10	4.5	45.3	connected	10	3.7	36.9
North School Bay	connected	11	1.7	15.2	connected	16	1.4	8.7
Section 5 Bay	connected	13	3.1	24.1	screened	17	3.1	18.2
Mid-Blind Channel	connected	10	3.0	29.7	screened	16	4.9	30.7
Mad Woman Bay	connected	13	3.6	27.6	diked	15	1.2	8.1
South Pitblado's Channel	connected	12	3.1	25.5	diked	15	1.2	8.2
South Mackenzie Bay	temporarily connected	16	3.0	18.6	blasted	12	2.7	22.6
Wye's Pond	temporarily connected	14	1.4	10.3	blasted	15	2.0	13.4
North Mackenzie Bay	temporarily connected	14	3.0	21.4	isolated	17	1.5	9.1
Emile's Pothole	temporarily connected	13	2.3	17.6	isolated	15	1.8	12.3

Table 3-3. Mean number (number/trap) of water-column invertebrates captured in activity traps set in 10 study ponds in Delta Marsh between early June and late August, 2001.

Taxonomic group	Connected						Temporarily Connected			
	TB	SB	S5	BC	MW	PC	MS	WP	MN	EP
Amphipoda	0.0	0.8	0.3	0.7	0.1	0.8	3.3	1.3	0.4	0.4
Conchostraca	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ostracoda	0.2	1.7	2.4	1.3	0.7	0.3	3.3	0.5	0.5	0.2
Cladocera	2.6	156.9	36.9	1.7	9.0	61.9	114.5	24.5	17.2	108.3
Copepoda	2.3	30.7	25.2	13.0	11.2	49.8	40.8	295.5	115.6	111.5
Corixidae	1.0	14.2	8.2	0.8	6.5	23.6	20.0	27.3	45.4	10.8
Notonectidae	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.1
Dytiscidae	0.0	0.1	0.0	0.0	0.0	0.2	0.1	0.2	0.0	0.0
Gyrinidae	0.1	0.0	0.4	0.0	0.3	0.1	0.0	0.0	0.0	0.0
Haliplidae	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1
Ephemeroptera	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.3	0.1	0.0
Trichoptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Chaoboridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Odonata	0.0	0.0	0.1	0.0	0.0	0.1	0.8	0.1	0.0	0.0
Hydracarina	1.4	1.2	5.6	5.1	1.0	7.8	30.6	6.6	57.8	2.3
Gastropoda	0.2	0.3	1.2	1.4	0.3	0.4	4.9	0.8	1.5	0.3
Hirudinea	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.1
Totals	8.0	205.8	80.4	24.0	29.1	144.9	219.0	357.4	238.7	234.2

Table 3-4. Mean number (number/trap) of water-column invertebrates captured in activity traps set in 10 study ponds in Delta Marsh between early June and late August, 2002.

Taxonomic group	Connected		Screened		Diked		Blasted		Isolated	
	TB	SB	S5	BC	MW	PC	MS	WP	MN	EP
Amphipoda	1.0	20.0	3.3	15.6	21.0	28.4	0.4	13.4	47.0	17.0
Conchostraca	0.0	0.0	0.0	0.0	0.3	0.0	0.0	4.8	16.8	0.0
Ostracoda	1.0	46.4	9.3	8.0	3.3	5.4	3.2	13.2	22.0	7.0
Cladocera	35.0	4,299.6	347.0	65.0	1305.0	4798.0	279.2	1269.4	3025.8	974.8
Copepoda	88.3	569.2	204.3	89.2	24.0	154.4	923.0	432.8	293.0	283.8
Corixidae	41.3	90.8	43.8	38.8	27.5	162.0	477.4	120.4	101.8	28.4
Notonectidae	0.0	6.6	7.5	0.2	1.0	4.6	6.2	1.8	2.2	7.0
Dytiscidae	0.3	8.8	11.0	0.6	8.8	5.6	4.6	7.0	20.6	19.4
Gyrinidae	0.0	0.0	0.5	0.2	0.0	0.0	0.0	0.0	0.0	0.0
Haliplidae	0.0	1.6	0.5	0.2	0.5	1.0	0.0	0.4	0.2	0.2
Ephemeroptera	0.3	2.8	2.3	3.6	1.3	1.0	0.0	1.0	2.6	2.6
Trichoptera	0.0	0.2	0.5	0.4	0.0	12.2	0.0	0.0	1.4	0.0
Chaoboridae	0.0	0.2	1.0	0.0	0.0	1.0	1.6	0.0	0.0	1.0
Odonata	0.0	0.4	3.3	1.0	0.5	0.6	0.0	0.4	0.4	0.6
Hydracarina	32.0	50.4	17.3	43.2	16.8	23.4	66.6	25.8	79.6	17.0
Gastropoda	0.3	8.6	8.0	6.2	13.8	109.2	1.8	9.6	159.8	9.4
Hirudinea	0.0	0.6	1.3	2.4	0.0	0.0	0.2	0.8	2.2	6.4
Totals	199.7	5,106.2	660.5	274.6	1,423.5	5,306.8	1,764.2	1,900.8	3,775.4	1,374.6

Table 3-5. Mean number (number/trap) of amphibians captured in Beamish traps in study ponds in Delta Marsh between early June to late August, 2001.

	CONNECTED						TEMPORARILY CONNECTED			
Species	**TB**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
leopard frog	0.0	0.0	0.0	0.0	0.0	1.0	27.0	177.0	5.0	51.0
Canadian toad	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
tiger salamander	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Totals	0.0	0.0	0.0	0.0	0.0	1.0	27.0	177.0	5.0	51.0

Table 3-6. Mean number (number/trap) of amphibians captured in Beamish traps in study ponds in Delta Marsh between early June and late August, 2002.

	CONNECTED		SCREENED		DIKED		BLASTED		ISOLATED	
Species	**TB**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
leopard frog	0.0	2.0	36.0	61.0	92.0	39.0	1.0	40.0	78.0	314.0
Canadian toad	0.0	0.0	0.0	0.0	0.0	0.0	0.0	22.0	2.0	1.0
tiger salamander	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0
Totals	0.0	2.0	36.0	61.0	92.0	39.0	1.0	62.0	81.0	315.0

Table 3-7. Mean number (number/trap) of fish captured per Beamish trap in 10 study ponds in Delta Marsh between early June and late August in 2001 and 2002.

	Numbers			
	2001		2002	
Species	**Mean**	**%**	**Mean**	**%**
fathead minnow	725.4	46.9	103.9	46.1
common carp	28.7	1.9	78.9	35.0
spottail shiner	10.4	0.7	8.0	3.5
emerald shiner	2.4	0.2	0.4	0.2
brook stickleback	88.8	5.7	14.2	6.3
ninespine stickleback	0.0	0.0	0.1	t[1]
black bullhead	345.7	22.3	11.0	4.9
brown bullhead	0.0	0.0	t	t
northern pike	1.0	0.1	0.2	0.1
trout-perch	0.3	t	t	t
yellow perch	239.3	15.5	7.6	3.4
johnny darter	5.8	0.4	0.4	0.2
Iowa darter	1.2	0.1	0.3	0.1
white sucker	98.9	6.4	0.4	0.2
central mudminnow	t	t	0.1	t
freshwater drum	0.1	t	0.2	0.1
Totals	1,548.0	100.0	225.6	100.0

1 - t = trace (<0.1 or 0.1%)

Note that 2002 data do not include Thompson's Bay. Due to sampling problems with this pond, numbers of fish captured were highly inflated and were not used to determine annual means (see text for further explanation).

Table 3-8. Species richness, effective richness and evenness of fish communities sampled in 10 study ponds in Delta Marsh between early June and late August, 2001 and 2002.

	2001				2002			
Ponds	Treatment	Species Richness *s*	Effective Richness N_2	Evenness (ER/s) % E_3	Treatment	Species Richness *s*	Effective Richness N_2	Evenness (ER/s) % E_3
Thompson's Bay	connected	9	1.4	15.1	connected[1]	12	3.3	27.8
North School Bay	connected	5	1.5	30.2	connected	11	1.8	16.4
Section 5 Bay	connected	9	2.9	31.8	screened	6	1.6	26.9
Mid-Blind Channel	connected	9	1.6	17.4	screened	12	2.5	21.2
Mad Woman Bay	connected	8	2.1	25.8	diked	9	1.3	14.0
South Pitblado's Channel	connected	10	1.2	12.2	diked	3	1.9	65.0
South Mackenzie Bay	temporarily connected	5	1.1	21.1	blasted	12	2.0	16.9
Wye's Pond	temporarily connected	11	3.0	27.2	blasted	10	3.1	31.3
North Mackenzie Bay	temporarily connected	7	2.0	28.9	isolated	0	0.0	0.0
Emile's Pothole	temporarily connected	9	2.9	32.6	isolated	4	2.0	50.2

1 - due to low water levels, this pond could not representatively sampled (see text for details).

Table 3-9. Mean number (number/trap) of fish captured in Beamish traps set in 10 study ponds in Delta Marsh between early June and late August, 2001.

	CONNECTED						TEMPORARILY CONNECTED			
Species	**TB**	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	16.0	0.0	0.3	180.0	0.5	1,045.8	7.8	975.1	4,656.3	372.5
common carp	0.5	0.8	6.8	1.8	1.8	12.3	2.5	1.5	66.8	192.9
spottail shiner	34.3	0.0	0.0	24.8	6.3	2.0	0.0	21.5	0.0	15.3
emerald shiner	4.3	0.0	0.0	0.0	0.8	0.0	0.0	19.3	0.0	0.0
brook stickleback	6.8	0.3	10.0	10.8	1.0	58.8	454.8	83.0	65.3	197.0
ninespine stickleback	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
black bullhead	7.5	0.3	0.5	29.0	19.3	10.5	0.0	18.3	3,370.8	0.8
brown bullhead	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
northern pike	0.3	6.3	0.5	0.0	0.0	0.5	0.8	0.0	2.0	0.0
trout-perch	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0	0.0	0.0
yellow perch	428.5	0.3	2.0	1628.4	56.3	13.8	0.0	251.3	1.3	11.0
johnny darter	0.0	0.0	0.0	0.0	0.0	0.3	1.5	48.4	7.8	0.5
Iowa darter	0.0	0.0	0.3	0.8	0.0	0.5	0.0	10.0	0.0	0.8
white sucker	4.0	0.0	0.3	187.8	0.3	12.3	0.0	780.5	0.0	3.8
central mudminnow	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
freshwater drum	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0
Totals	502.0	7.8	20.8	2,063.6	86.0	1,156.5	467.3	2,211.8	8,170.1	794.4

Table 3-10. Mean number (number/trap) of fish captured in Beamish traps set in 10 study ponds in Delta Marsh between early June and late August, 2002.

	CONNECTED		SCREENED		DIKED		BLASTED		ISOLATED	
Species	**TB**[1]	**SB**	**S5**	**BC**	**MW**	**PC**	**MS**	**WP**	**MN**	**EP**
fathead minnow	2,096.8	359.9	98.3	134.8	66.6	94.7	105.5	63.3	0.0	12.2
common carp	1,084.8	109.1	455.0	3.3	0.0	0.0	28.1	114.4	0.0	0.0
spottail shiner	245.2	17.2	0.5	53.7	0.2	0.0	0.4	0.1	0.0	0.0
emerald shiner	108.8	0.4	0.0	0.7	0.1	0.0	2.1	0.3	0.0	0.0
brook stickleback	0.6	10.3	16.3	7.7	5.7	64.5	7.2	0.0	0.0	15.8
ninespine stickleback	0.0	0.4	0.0	0.5	0.0	0.0	0.3	0.0	0.0	0.0
black bullhead	417.8	2.4	0.0	2.1	0.8	0.0	9.1	84.6	0.0	0.0
brown bullhead	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
northern pike	0.4	0.1	0.0	0.1	0.5	0.8	0.2	0.0	0.0	0.0
trout-perch	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
yellow perch	2,804.4	5.1	19.5	33.4	0.5	0.0	0.5	9.3	0.0	0.0
johnny darter	0.0	t	0.0	1.3	0.0	0.0	1.0	1.0	0.0	0.0
Iowa darter	0.0	0.0	0.0	0.2	0.0	0.0	0.0	3.0	0.0	0.0
white sucker	0.6	0.4	2.3	0.0	0.1	0.0	0.7	0.1	0.0	0.0
central mudminnow	0.2	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.2
freshwater drum	0.4	0.0	0.0	0.0	0.0	0.0	1.3	t	0.0	0.1
Totals	6,760.2	505.3	591.8	238.0	75.1	159.9	156.4	275.9	0.0	28.3

[1] - due to low water levels, this pond could not representatively sampled (see text for details).

Figure 3-1. Activity trap used to sample water-column invertebrates. Traps were set in 10 study ponds in Delta Marsh between early June and late August, 2001 and 2002.

Figure 3-2. **Beamish trap with a modified 15 cm opening for use in shallow water. Traps were set in 10 study ponds in Delta Marsh between early June and late August, 2001 and 2002.**

Figure 3-3. **Minnow trap. Traps were set in 10 study ponds in Delta Marsh between early June and late August 2001.**

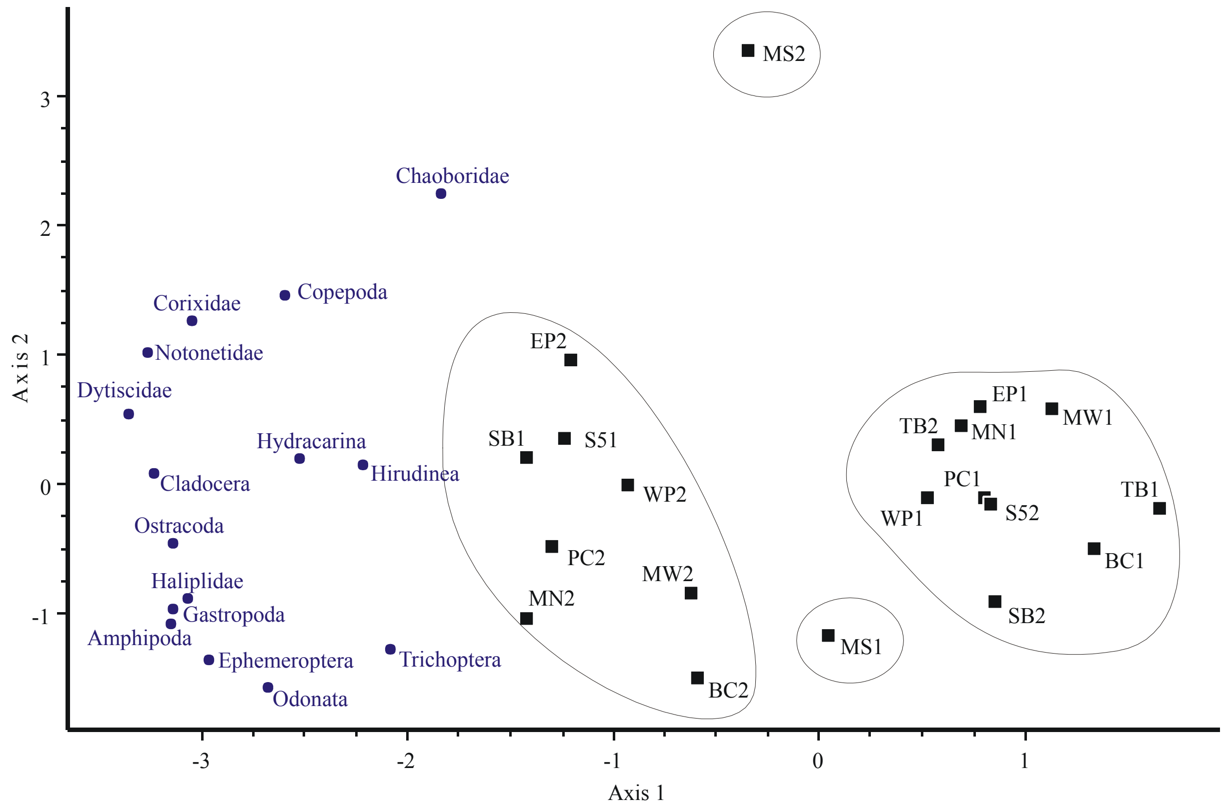

Figure 3-4. **PCA ordination bi-plot of the seasonally averaged invertebrate families (number/trap) constrained by study ponds in Delta Marsh, 2001 and 2002. A total of 66.8% of the variation in invertebrate structure was explained by the first axis.**

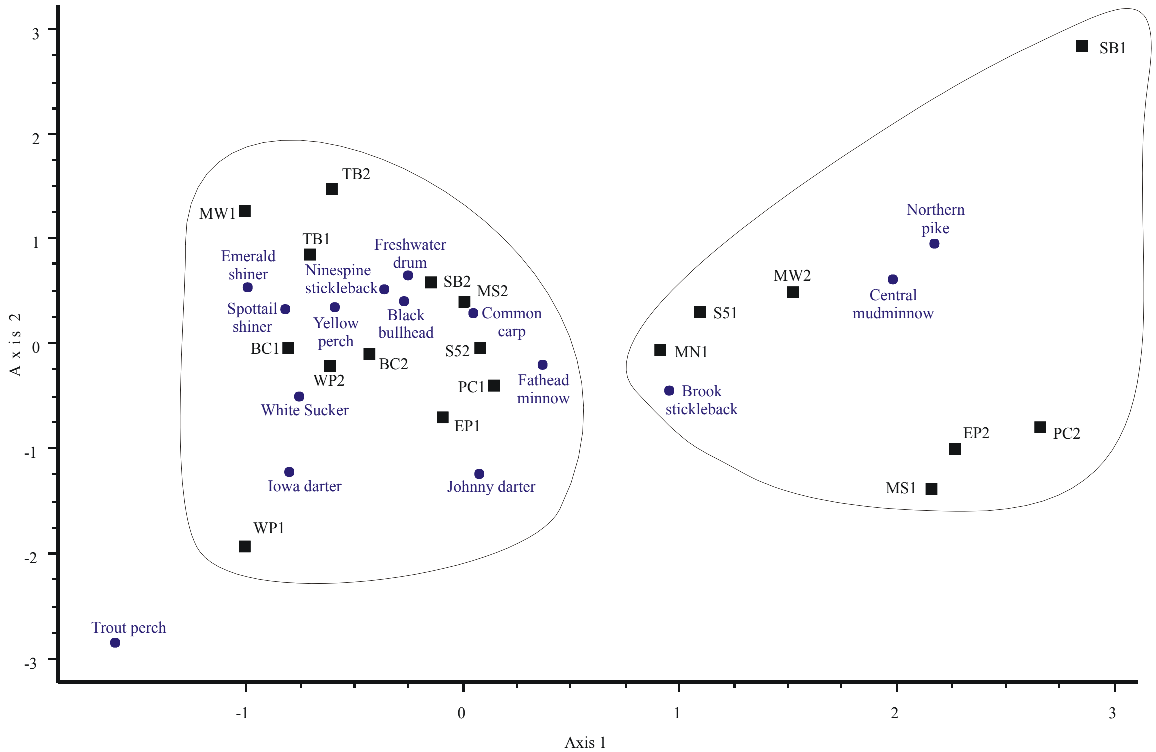

Figure 3-5. **Correspondence analysis (CA) ordination biplot of the seasonally averaged fish species (number/trap) constrained by study ponds. Correspondence correlations are 0.28 for the first axis and 0.14 for the second axis. Axes 1 and 2 composed of 41.9% of the cumulative variance. Total interia 0.67.**

CHAPTER 4 – SUMMARY AND RECOMMENDATIONS

In the past four decades, concerns have been raised about the health of Delta Marsh. Most apparent is an increase in water-column turbidity, disappearance of aquatic macrophytes and decline in waterfowl numbers (Goldsborough and Wrubleski 2001; Gordon Goldsborough, unpublished data). One of the main contributing factors suggested for this decline is the presence of introduced common carp (*Cyprinus carpio*). Since their arrival in the province in 1938 (Hinks 1943; Atton 1959; Crossman 1968), the impacts of common carp in the three large lakes of Manitoba and their peripheral coastal wetlands, including Delta Marsh, is largely unknown. Consequently, a collaborative manipulation study was designed and implemented to determine the effects of common carp on both lower and higher trophic levels. Experimental manipulations of 10 peripheral ponds were intended to facilitate (connected and blasted ponds), restrict (screened ponds) or exclude (isolated or diked ponds) access by common carp in order to compare spatial and temporal differences among treatments.

Although common carp were the original subject of the study, once the project began, it became apparent that hydrological connection to the surrounding marsh had a paramount importance on the structure and diversity of the fish, amphibian and water-column invertebrate communities. Unanticipated high water levels in 2001 temporarily connected all study ponds to the main marsh, permitting access by common carp, as well as most other native fish species. In general, an increased abundance of fish correlated with lower numbers and diversity of water-column invertebrates and amphibians. As a consequence, it was not possible to implicate common carp for changes observed within the study ponds. Furthermore, in 2002, when the manipulations were initiated, connectivity, or lack of connectivity, continued to play an important role in the distribution of the fish community, and subsequently the composition and distribution of water-column invertebrates and amphibians.

I would recommend the long-term continuation of the pond study at Delta Marsh, with some alternative pond treatments which may allow for a better understanding of common carp impacts on the marsh. To elicit a stronger common carp response, I would include additional treatments, such as immediately screening a newly blasted pond and a previously diked pond. These manipulations would facilitate hydrological connection and movement of native fish species, without allowing access by adult common carp. I believe that this experimental design would be better at isolating the impacts of adult common carp on the Delta Marsh ecosystem. Whillans (1996) hypothesized, that once a habitat has been restored, common carp should not be able to destroy it. Therefore, it would be of interest to see the long-term response of a pond that was initially diked (restored), then screened, and then subsequently opened to the main marsh to allow access by adult common carp. This setup would gradually introduce common carp impacts to the pond and potentially help to isolated common carp influences from other confounding factors.

Besides a handful of unpublished reports and theses, knowledge of the overall fish, amphibian and water-column invertebrate community of Delta Marsh is sparse. First, it is essential to gather basic ecological information on their role in coastal wetlands in order to accurately determine the structure and function of these systems, as well as to gain a better understanding of the predator-prey relationships. Data from a few unpublished reports and theses on Delta Marsh is too fragmentary for piecing together the complex community interactions, therefore long-term data collection is strongly recommended. It is important to gain information on the structure of the seasonal fish community, especially during the winter and early spring months. My results show Delta Marsh is under appreciated as fish habitat for numerous commercial species, consequently without supplemental data this critical fish habitat will continue to be degraded. Of greater importance is the need to determine year-round movements, spawning patterns of fish and whether or not they utilize the deeper areas of the main marsh during winter. To date, the majority of fisheries studies are conducted in the summer months leaving large knowledge gaps for winter spawning species, such as burbot, which are one of the top predators in Lake Manitoba, could potentially use Delta Marsh for spawning.

Furthermore, additional study is also needed to determine the effects of hydrological changes to Lake Manitoba, and its impact on Delta Marsh, its native fish communities and common carp populations. Stabilized water levels may provide optimal habitat for common carp, leading to increased abundance of this species (Harris 1996). It is possible that by restoring a more dynamic water regime over the long and short-term at Delta Marsh, this may help reduce common carp numbers, either by reducing spawning habitat, desiccating eggs during low water periods, or enhancing predatory northern pike abundances during high water periods. A return to a more dynamic water regime in the marsh may not only reduce common carp numbers, but would also benefit the plant communities within the marsh as well (Batt 2000). Finally, it is important to recognize that under the current stabilized water level regime on Lake Manitoba, isolated ponds within Delta Marsh function solely as temporary habitat for most fish species. Allowing more dynamic water levels will facilitate movement of fish species between the main marsh and its peripheral ponds, increasing fish production in Delta Marsh which could potentially benefit Lake Manitoba commercial fishers.

Results from the current study suggest common carp may have some influence on amphibian populations within Delta Marsh, but no clear impacts on native fish or aquatic invertebrates were apparent. The results of studies on common carp impacts on the lower trophic levels that were undertaken as part of a larger investigation are yet to be completed. The lack of impacts on the parameters that I examined may be due to several things. First, variable water levels between years confounded pond manipulations and my results. Second, I only monitored the first year of manipulations, and it may well take more time for common carp impacts to be fully apparent on higher trophic levels. An alternative hypothesis might be that common carp are not the driving force behind the deterioration of Delta Marsh as believed. Instead, common carp may simply be better adapted to survive in poorer habitat conditions. A combination of stabilized water levels, changes to surrounding land use and poor water quality may be more responsible for the overall decline in water quality and loss of submersed vegetation, but there are no data to support this theory. Further work is needed to determine the influences of surrounding land use on the overall health of Delta Marsh.

Complete elimination of common carp at Delta Marsh is highly unlikely. Piscicides, such as Rotenone, have been used to completely eliminate fish communities (*e.g.,* Lake Christina (Hanson & Butler 1990), but are highly regulated substances in Canada. Rotenone has not been used in Manitoba for the past 25 years (Joel Hunt, Manitoba Conservation, personal communication). In addition, these poisons are not species specific and would be logistically difficult to apply to a large coastal wetland like Delta Marsh, without allowing re-colonization of the common carp from adjacent Lake Manitoba. Screening the four channels that connect Delta Marsh to Lake Manitoba would likely prevent access by common carp in the spring. When this was done in the early 1960s (McCrimmon 1968) common carp were prevented from entering the marsh, however extensive maintenance was required to clean and maintain the screens. In addition, the screens also prevented other large fish species from accessing the marsh. Incorporation of fishways, such as those used at Cootes Paradise Marsh would be effective, but require manual sorting of large fish in order to facilitate their entrance into the marsh. These costs would likely be too great to justify at Delta Marsh. Removal of common carp by trapping in the marsh would be physically demanding, time consuming and expensive. Commercial harvesting of common carp may be the most promising solution for reducing population numbers. Recent development of a rough fish initiative in Manitoba has increased the commercial popularity of common carp, resulting in a doubling of profits for fishers. Returns to fishers capturing common carp and their roe reached a ten-year high of $310,000 to $410,000 in the 2002/2003 winter/spring seasons (Freshwater Fish Marketing Corporation, unpublished data). Further development of international markets for common carp roe would be advantageous for both wetland managers and fishers in Lake Manitoba and surrounding large lakes within the province.

It is imperative that the Province of Manitoba take the necessary steps to stop additional invasions of other carp species, such as bighead carp [*Hypophthalmichthys nobilis* (Richardson)], silver carp [*Hypophthalmichthys molitrix* (Valenciennes)] and grass carp [*Ctenopharyngodon idella* (Valenciennes in Cuvier and Valenciennes)], as well as strengthen current legislation, or devise new legislation, and increase educational campaigns to reduce the risk of other invasive exotics [*e.g.*, zebra mussels (*Dresissena polymorpha* (Pallas))] from getting into the province. Currently, managers are erecting electrical barriers to prevent the migration of bighead carp into the Laurentian Great Lakes (Environmental Protection Agency 2004). Combined with common carp, these new species have the potential to negatively impact the entire aquatic ecosystem due to their diversity of foraging strategies and invasive nature. The further introduction of new carp and other invasive fish species into the province would devastate Manitoba's commercial and recreational fisheries.

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