Impacts of Introduced Wild Rice (*Zizania palustris*) on Invertebrate Communities in Lakes in West-central Manitoba

By

Mark K. Lowdon

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In Partial Fulfillment of the
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"Impacts of Introduced Wild Rice (*Zizania palustris*) on Invertebrate Communities in Lakes in West-central Manitoba"

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Mark K. Lowdon

A Thesis/Practicum submitted to the Faculty of Graduate Studies of The University of Manitoba in partial fulfillment of the requirement of the degree

Of

MASTER OF SCIENCE

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Acknowledgements

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Abstract

In the early 1980s, wild rice (*Zizania palustris*) was introduced into a number of lakes in west-central Manitoba as a result of the expansion of the commercial wild rice industry. Concerns have been raised regarding the possible impacts wild rice may have on aquatic organisms. With one exception (Watson et al., 2001), no previous research has examined aquatic communities in littoral zones dominated by wild rice. In this study, I compared and contrasted invertebrate communities in bays with wild rice and bays with native macrophytes in three lakes, Barry, Cacholotte and Naosap, in west-central Manitoba. The first objective was to compare invertebrate diversity, abundance and composition between bays with wild rice and bays with native macrophytes. Bottle traps, emergence traps, and a bucket volume sampler were used to sample invertebrates. In addition, eighteen water quality parameters were compared among the selected sites.

A total of 11,398 invertebrates were collected from wild rice and native macrophyte bays using bottle traps, which trap nektonic invertebrates within the water column, and emergence traps which trap emerging insects primarily from the benthos. Fifty-two invertebrate taxa were collected within wild rice and native macrophyte bays. For samples collected using bottle traps, overall invertebrate abundance was not significantly different ($p > 0.05$) between wild rice and native macrophyte bays. In contrast, for samples collected using emergence traps, significantly higher numbers of emerging insects ($p = 0.03$) were collected in native macrophyte bays in June. Similar results were found in August, however differences were not significant ($p = 0.17$). No differences in overall abundance were observed among samples collected by the bucket volume sampler. Overall invertebrate diversity (absolute richness, effective richness and

evenness) was not significantly different between wild rice and native macrophyte bays for emergence or bottle trap data. Diversity of invertebrates within the bucket volume sampler was not analyzed due to large variation within and between samples.

Invertebrate composition differed between wild rice and native macrophyte bays. Amphipods, Hydracarina (water mites), ostracods, and physids were more abundant within wild rice bays. In contrast, Haliplidae, Chironomidae, and Gyrinidae were more abundant within native macrophyte bays. Wild rice possibly provides more refuge from predation and/or an additional food source for invertebrates. However, the sediment was perhaps not as stable in wild rice bays, possibly accounting for the higher abundances of benthic invertebrates within native macrophyte bays.

The second objective of the study was to use carbon and nitrogen stable isotopes to characterize trophic relationships and assess whether food webs differed in littoral areas of lakes with wild rice or native macrophytes. Results suggest that wild rice may provide an additional energy source for invertebrates and fish as their carbon signatures overlap. Wild rice carbon signatures were, however, more depleted ($\delta^{13}C$ ranged from -31.08 to -26.65 ‰) compared to the native macrophytes ($\delta^{13}C$ ranged from -30.09 to -11.42‰) and, therefore, if invertebrates or fish were feeding directly on wild rice, depletions of carbon for the invertebrates and fish within the wild rice bays would be expected. Depletions of carbon were not observed within wild rice bays (Δ ranged from 0.24 to 3.34 ‰) suggesting that wild rice does not provide the primary source of energy. $\delta^{15}N$ for invertebrate and fish were depleted in wild rice bays compared to native macrophyte bays (Δ ranged from 0 to -3.03 ‰) suggesting different nitrogen cycling of nutrients. Omnivorous fish (*Perca flavescens*) and benthivorous fish (*Catostomus* *commersoni*) in particular, had similar $\delta^{13}C$ and $\delta^{15}N$ values as invertebrates within both wild rice and native macrophyte bays. Shiners had consistently depleted $\delta^{13}C$ values within both the wild rice and native macrophyte bays on all lakes, implying a diet linked to zooplankton. Overall food chain length was slightly longer in the wild rice bays, however, the biota were found at five similar trophic levels within each bay.

Results from this study indicate that invertebrate communities in wild rice and native macrophytes bays are equally abundant, however, community composition varied between the wild rice and native macrophyte bays in this study. This study has provided important information about the relationships of wild rice with invertebrate and fish communities. Future studies are needed to determine how wild rice impacts the biota in other regions and to expand our knowledge about the complex food web interactions between invertebrate and fish communities within wild rice bays.

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Chapter 1

General introduction: Potential impacts of introduced wild rice on aquatic ecosystems

1.0 General Introduction

Wild rice is an important food source all over the world and is considered a gourmet food in North America. The demand for wild rice is on the rise (Peterson et al., 1982; Derksen, 1998). In the United States, this demand is being met by producing wild rice in paddies. In Minnesota and California, dykes have been built on farm fields, filled with water, and seeded with wild rice for commercial production (Dore, 1969; Peterson et al., 1982; Derksen, 1998). To maximize wild rice production, pesticides and fertilizers are used to control insect pests and increase yields (Archibold, 1995; Derksen, 1998). Water depth is also controlled, as fluctuating depths can destroy the plants in the early stages of their life cycle (Dore, 1969; Thomas & Stewart, 1969). In Canada, increased demand for wild rice is being met not by building wild rice paddies, but by introducing this plant into our natural lakes, streams, and rivers (Derksen, 1998). Wild rice in Canada is marketed as an organic product because no chemicals or fertilizers are used in its production, and is therefore popular with commercial growers as it provides for an increased income. Although wild rice is native to the Great Lakes region of Ontario and south-eastern Manitoba, it has been intentionally introduced in Saskatchewan, Alberta, and the northern region of Manitoba (Archibold, 1995).

Wild rice introductions in Canada have raised a number of concerns regarding possible impacts that this introduced plant may have on native aquatic communities. To protect and conserve the native fauna that inhabit these water bodies, introduced wild rice must be examined to characterize any further impacts to the aquatic primary producer, invertebrate and native fish communities.

The purpose of this study was to examine the impacts of introduced wild rice (*Zizania palustris*) on the invertebrate and fish communities in west-central Manitoba. The first chapter reviews the biology, history, and concerns regarding introduced wild rice in west-central Manitoba. The second chapter looks at the differences in the composition, abundance and diversity of the invertebrate community between bays seeded with wild rice and bays with naturally occurring aquatic plants. The third chapter examines the pathways of energy flow and trophic levels within these two types of habitats to determine whether wild rice alters food web structure in these lakes.

1.1 Biology of Northern Wild Rice

Northern wild rice is an annual, aquatic emergent macrophyte that is native to eastern Manitoba and the Great Lakes region (Dore, 1969). It is a member of the grass family Poaceae, and its physical appearance resembles wheat, oats, or barley because of its long hollow stem and blade-like leaves (Figure 1.01-D) (Archibold, 1995). The duration of the wild rice life cycle is approximately five months and consists of four distinct stages: a germination stage, a submerged stage, a floating leaf stage, and an emergent stage (Figure 1.01) (Dore, 1969; Natural Resources Institute, 1995; Archibold, 1995). After a period of dormancy during which the seed is subject to temperatures below 4°C for the winter months, germination takes place in early spring through the development of a primary root that anchors the plant into the mud substrate and protects it from currents and wave action (Figure 1.01-A) (Dore, 1969). The submerged-growth stage follows with the opening of the first shoot to produce the first leaf and stem. As more submerged leaves are produced, the lower leaves die off causing the stem be very smooth in appearance (Figure 1.01-B) (Aiken et al., 1988). Once wild rice reaches the water surface, the floating leaf stage occurs with two or three ribbon-like leaves sprawling across the water surface (Figure 1.01-C) (Thomas & Stewart, 1969; Dore, 1969). The floating leaf stage lasts only 14 days due to an increase in photosynthesis as a result of greater plant surface area capturing sunlight (Natural Resources Institute, 1995). In mid to late June the emergent stage of wild rice occurs and is clearly seen in bays (Figure 1.02); this is sometimes referred to as the wheat field phase. Flowering and pollination of wild rice occur in mid July. Wild rice seeds begin to mature in late August through September. Wild rice is commercially harvested (Figure 1.03) at this time before the seeds are dropped to the sediments to begin another cycle (Dore, 1969).

Northern wild rice grows in shallow lakes, streams, and rivers (Dixon & Derksen, 1999). The ideal water depth for wild rice growth is 0.5 to 0.6 meters (Stevenson & Lee, 1987; Dixon & Derksen, 1999), however it has been observed growing at depths ranging from 0.3 to 2.0 meters (Natural Resources Institute, 1995). The growth and spread of wild rice is also restricted by established native macrophytes which compete for light, nutrients, and space (Dore, 1969; Aiken et al., 1988; Archibold, 1995). Sediments consisting of a mix of organic matter and clay provide an ideal substrate for wild rice growth, while other sediment types may hamper wild rice development (Day & Lee, 1989). In addition, approximately 100 frost-free growing days are needed for the seeds to mature and become commercially valuable (Thorvaldson, 1995).

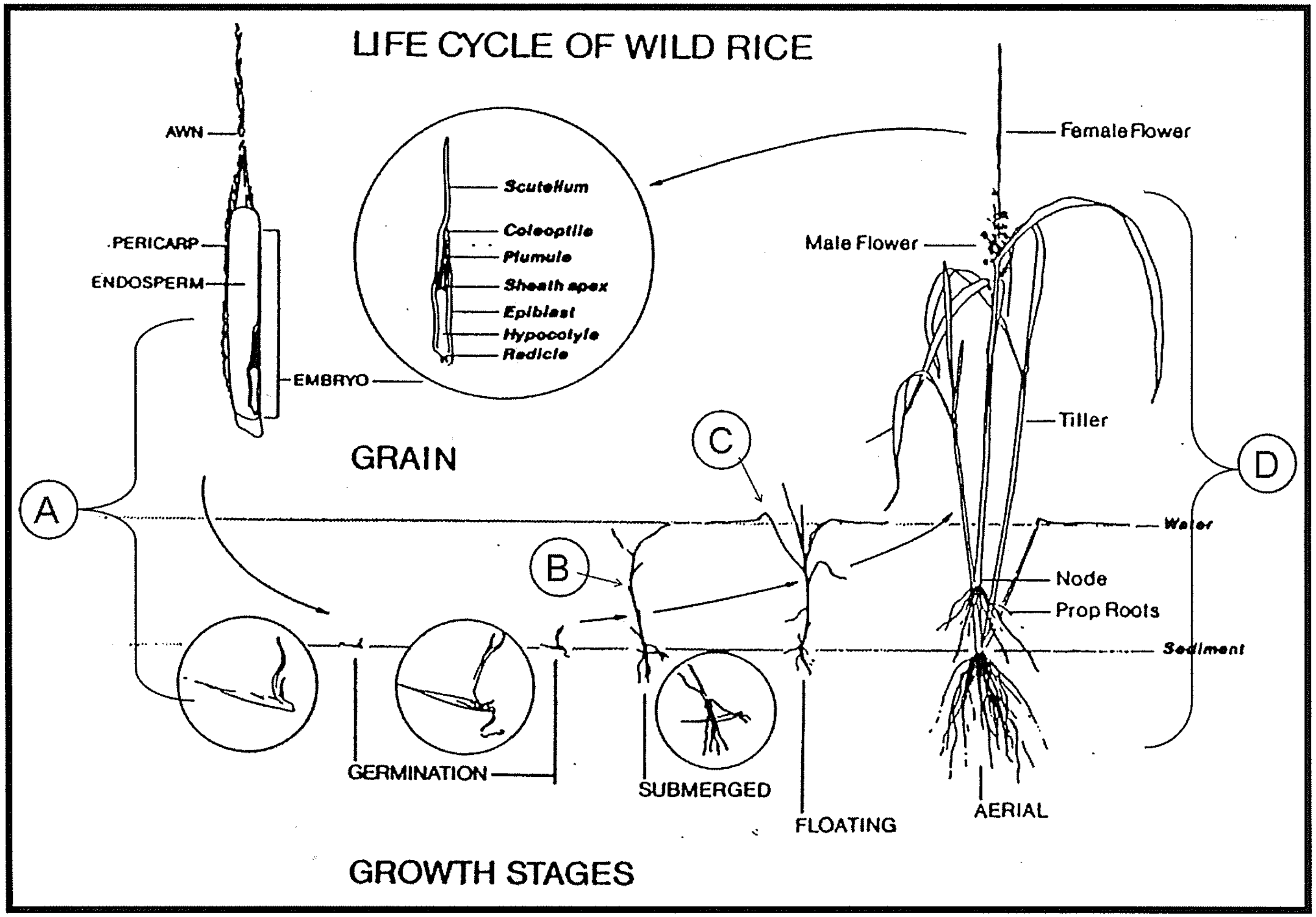


Figure 1.01. Life cycle of northern wild rice (*Zizania palustris*) illustrating the germination stage, submerged stage, floating leaf stage and the emergent or aerial stage (modified from Aiken *et al.*, 1988)

Figure 1.02. The emergent stage of wild rice in a bay on Cacholotte Lake during August of 2002. The stands grow to approximately one meter above the water surface. (photo by M. Lowdon)

Figure 1.03. An airboat used to harvest wild rice. Mature seeds fall into a trough as the boat passes over the plants. (photo by M. Lowdon)

1.2 History of Wild Rice

Northern wild rice (*Zizania palustris*) is believed to have originated within the Great Lakes region of North America (Dore, 1969). For thousands of years it has been growing in shallow lakes, streams, and rivers in northern Minnesota, Wisconsin, south-eastern Manitoba, and Ontario (Figure 1.04) (Oelke, 1993). Native people, especially the Ojibway, Menomini and Cree tribes, used wild rice as a traditional food, ceremonial gifts, and an important trade commodity (Aiken et al., 1988; Derksen, 1993; Gertzbein, 2000). This plant provides an ideal habitat and food source for waterfowl and muskrats, and was planted by native peoples to attract these animals for hunting (Archibold, 1995; Gertzbein, 2000). With all the different uses of wild rice by the various groups, many names for wild rice have been used including Canadian rice, water oats, water rice, tuscarora, folle avoine, wild oats and manomin (Dore, 1969).

In the 1930s, wild rice was introduced into a number of waterways in Saskatchewan and other parts of Canada for personal use by native peoples (Derksen, 1993; Archibold, 1995); however, it was not until the early 1970s that wild rice was intentionally introduced for commercial harvest (Archibold, 1995). Since then, wild rice has been introduced into lakes and rivers across Ontario, Manitoba, northern Saskatchewan, and Alberta. There have also been reports of wild rice production in British Columbia, Yukon, Quebec and the Maritimes (Archibold, 1995). With an increasing population and a growing market for wild rice as a nutritious cereal rich in vitamin B and low in fat (Painchaud & Archibold, 1990), harvesting wild rice has become an economic boost for those commercially producing it.

Starting in the early 1980s, northern wild rice was introduced into lakes within the west-central region of Manitoba (Figure 1.04) (Derksen, 1998). The expansion of the wild rice industry within this region has been aided by the construction of roads by the forestry industry (Dixon & Derksen, 1999). As new roads are built, lakes that otherwise would not be profitable for wild rice production can now be seeded and contribute to a growing market (Derksen, 1998). Currently, wild rice in this region covers entire bays and large areas of lakes (Derksen[1] pers. comm., 2002).

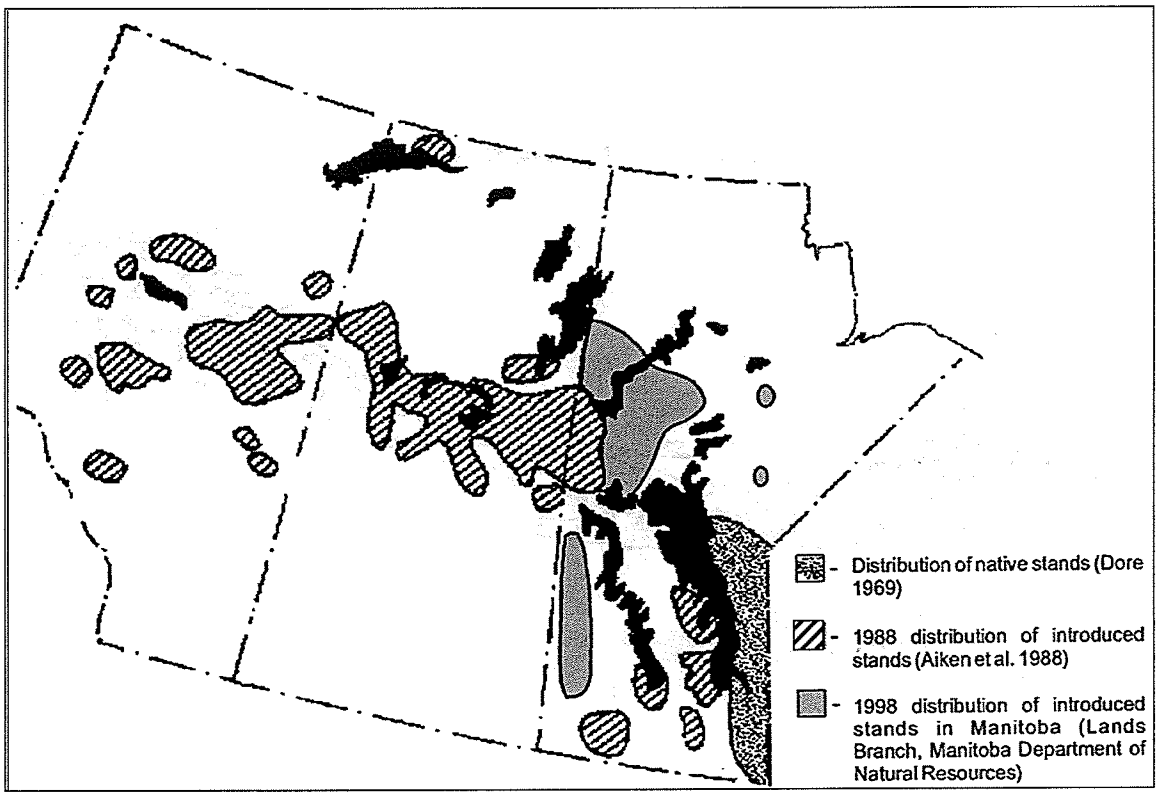


Figure 1.04. Distribution of native and introduced northern wild rice (*Zizania palustris*) in prairie Canada (reprinted with permission from Watson et al., 2001).

[1] Derksen A., Biologist, Fisheries & Oceans Canada, Winnipeg, MB.

1.3 Concerns regarding introduced wild rice

As the wild rice industry has developed in west-central Manitoba, there has been a growing concern about the potential physical and biological impacts of introduced wild rice on the native aquatic community (Derksen, 1998). After each growing season, wild rice produces a thick straw mat. There are concerns that this abundant litter may lead to decreased oxygen levels within the water column (Sain, 1983; Moyer, 2000; Watson, et al. 2001), loss of spawning habitats for fish (Derksen, 1993), and infilling of water bodies (Tattum, 1998; Derksen, 2002). Other impacts may include reduced plant diversity and the formation of monocultures (Watson et al., 2001), changes to benthic invertebrate communities (Watson et al., 2001), reduction of nutrients within lakes (Keenan & Lee, 1988), altered temperature gradients (Watson et al., 2001), and possibly a loss of fish species as a result of changing habitat (Watson et al., 2001). In addition to these concerns, the recreational use of these water bodies by boaters, canoeists and anglers may be affected as wild rice gets tangled in motors, is difficult to paddle through, and makes casting a fishing line difficult (Anon, 1993; Derksen, 1993).

Although many concerns have been raised, little scientific research has been conducted on the impacts that introduced wild rice may have on aquatic communities. There have been a number of studies and anecdotal literature examining the biology and economics of wild rice production (Lee & Stewart, 1984; Lee, 1986; Keenan & Lee, 1988; Day & Lee, 1989; Painchaud & Archibold, 1990; Natural Resources Institute, 1995); however, in west-central Manitoba there has been only one study that examined the impacts of introduced wild rice on native aquatic organisms (Watson et al., 2001).

Straw

One of the main concerns with the introduction of wild rice is the formation of large straw mats after each growing season (Figure 1.05) (Tattum, 1998; Derksen, 2002). Decomposition of the straw mats may create anoxic conditions within the water column and result in intolerable or lethal conditions for fish and invertebrates both under the ice in the winter and during the summer months (Sain, 1983 & 1984; Archibold, 1989; Derksen, 1993; Natural Resources Institute, 1995; Derksen, 1998; Watson et al., 2001; Derksen, 2002). During the winter months in 2000, dissolved oxygen concentrations were measured at twenty-five stations within wild rice bays and non-wild rice areas in Kisseynew Lake, near Flin Flon, Manitoba. There were consistently lower concentrations of dissolved oxygen at sites with wild rice as compared to sites without wild rice (Moyer, 2000, unpublished data).

Wild rice straw takes approximately three years to completely decompose (Archibold, 1990; Tattum, 1998; Derksen, 2002). These straw mats may settle onto the substrate and smother not only the wild rice seeds, but seedlings of natural vegetation. This organic matter may also accumulate on the shoreline and cover important fish spawning sites (Derksen, 1993). After the introduction of wild rice in Kisseynew Lake in the 1980s, an accumulation of approximately one meter of organic material was observed in core samples (Derksen, 2002; Watson et al., 2001). With this build up of organic matter, *Phragmites* sp. a semi-aquatic plant, may become established and alter the shorelines (Derksen, 2002). *Phragmites* sp. grows well in marshes and swamps, along the shoreline of streams, lakes, ponds, and in ditches. It is an invasive plant that is difficult to eradicate (Haslam, 1972; Galatowitsh et al., 1999; Saltonstall, 2002). Therefore, accumulation of organic material on the shoreline within wild rice bays in the west-central region of Manitoba may lead to the spread of this invasive plant species (Derksen, 2002).

Figure 1.05. Thick straw mats that form after each wild rice growing season in Naosap Lake in June of 2003. (Photos by M. Lowdon)

Nutrient Removal

Another potential impact associated with the introduction of wild rice is related to the removal of nutrients from the water column and sediment and the subsequent effects of this removal on other primary producers and the entire food web. Straw mats drifting down stream and harvesters removing the seed from lakes may also decrease the amount of nutrients available for native plant species (Keenan & Lee, 1988). After several growing seasons, significant decreases in manganese, zinc, copper, magnesium, potassium, and especially nitrogen were observed in the sediment (Keenan & Lee, 1988). Archibold (1990) discovered that when straw from the previous year was mulched and returned to the lake higher yields of wild rice were produced, suggesting that available nutrients may limit wild rice and possibly native plant growth. The removal of nutrients by wild rice plants may also decrease the abundance and diversity of native plant species, eventually leading to bays with wild rice monocultures (Derksen, 2002). Similarly, nutrient removal within a wild rice bay may also decrease periphyton production (Derksen, 1998). In contrast, the removal of nutrients by wild rice may be beneficial in eutrophic lakes where excess nutrients may contribute to the formation of algal blooms (Kennedy et al., 1994; Tattum, 1998).

Physical Effects of Wild Rice

The greater abundance of emergent plants and plant litter in wild rice bays may impede water flow and therefore affect water chemistry and temperature gradients (Watson et al., 2001; Derksen, 2002). If surface water temperature increases and water movement within littoral zones is restricted by the presence of wild rice stems, stratification of the water column within littoral zones is possible. If this occurs, oxygen diffusion may be restricted to the upper layers leading to increased anoxia in the deeper waters and in the sediments. Also, temperatures within littoral zones where water movement is impeded may reach as high as 30°C which may be lethal for invertebrates that are unable to move to cooler zones (Natural Resources Institute, 1995).

Fish and Invertebrate Interactions

Because fish and invertebrates depend upon macrophytes for food, shelter and refuge any alteration of the native macrophytes in the littoral zones of lakes may have impacts on these biota. Invertebrates are an important component of aquatic food webs because they provide a food source for most fish species during one or more stages of their life cycle (Gerking, 1962; Fairchild, 1982; Mittelbach, 1984; Carpenter & Lodge, 1986). With the introduction of wild rice, diverse native plant communities that provided a more complex microhabitat and surface area for invertebrates to colonize (Figure 1.06a) were replaced with a plant community dominated by dense stands of single stems reaching from the sediment to the water surface (Figure 1.06b) (M. Lowdon, personal observation). Wild rice stands in Saskatchewan and Manitoba waterways have been observed with densities greater than one hundred and fifty stems per square meter (Lee, 1986). These alterations may decrease predator-prey interactions between fish and invertebrates within littoral zones which may lead to decreased complexity of the entire aquatic food web (Connell, 1975). Fish movement and foraging efficiencies may be hindered within these habitats, resulting in decreased fecundities, growth, or productivity (Derksen, 2002). Aquatic macrophytes do provide refuges for both invertebrates and forage fish communities (Crowder & Cooper, 1982), however, at extremely high or low densities, the influence of predator-prey interactions may benefit only one group, either fish or invertebrates. Crowder and Cooper (1982) found that intermediate macrophyte growth provided ample refuge for invertebrates while not impeding fish foraging efficiencies. The energy flow and food web structure in lakes may also be altered with the introductions of wild rice (Derksen, 2002). In addition, these changes in the local habitat may directly impact both the invertebrate and fish communities if there are any adverse effects on their food supply (Derksen, 1993, 2002).

Dispersal

The unintended expansion of wild rice into new lakes is another concern. As wild rice has been observed growing to maximum water depths of up to two meters, the spread of wild rice within lakes is mainly restricted to the littoral zones of those lakes. However, migrating birds have been observed to transport wild rice grains caught within their feathers (Natural Resources Institute, 1995). In addition, drifting straw mats or water currents may carry dormant seeds to new locations (Natural Resources Institute, 1995). However, it has also been proposed that the wild rice seeds fall only in close proximity to the parent plant in the water column and, for this reason, there is limited spread of the plant (Dore, 1969). At present it is unclear how extensive the spread of wild rice is from areas in which it has been purposely introduced.

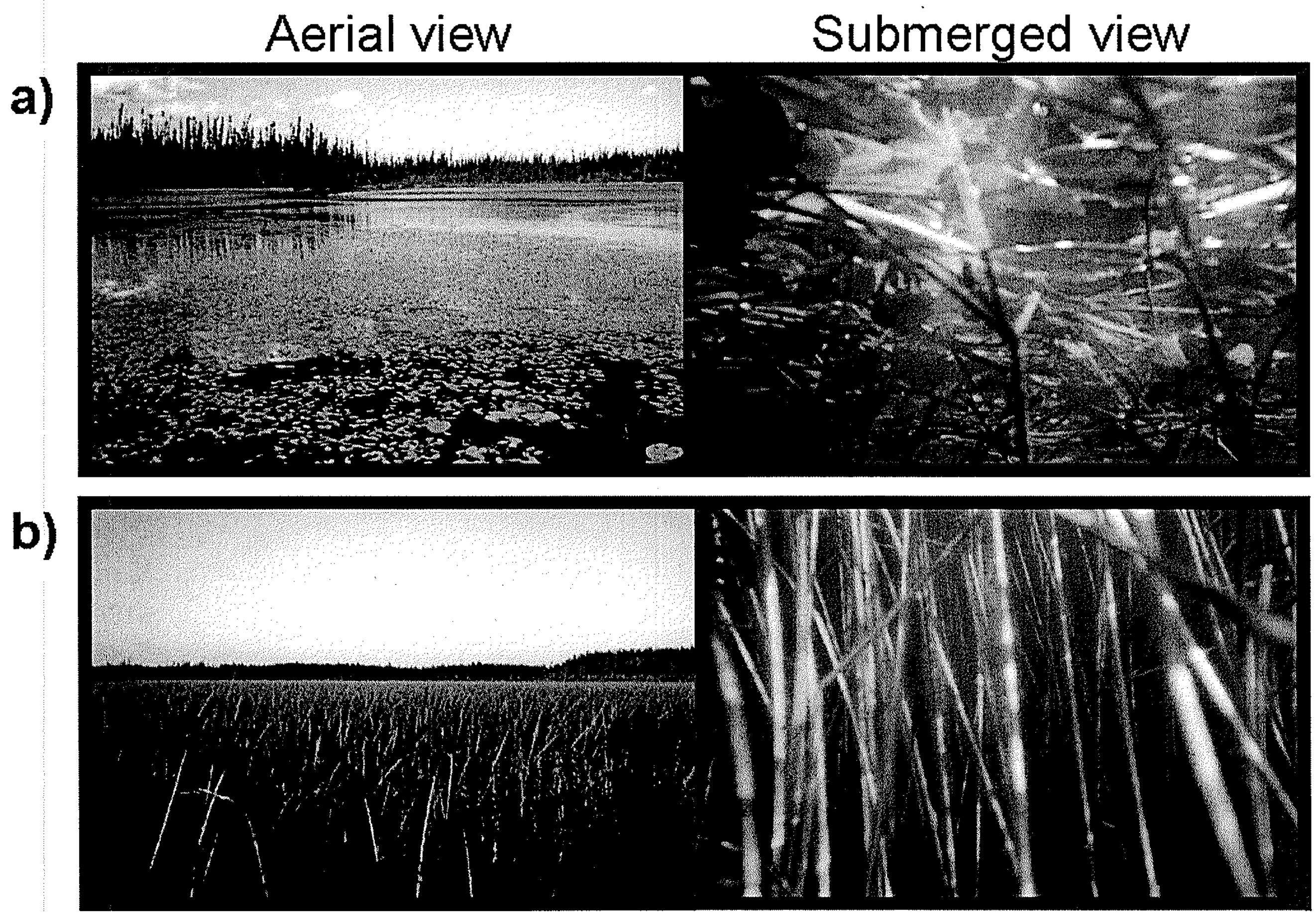


Figure 1.06. Aerial and submerged view of the physical structure of a) native macrophytes and b) wild rice on Cacholotte Lake in west-central Manitoba. (photo by M. Lowdon)

1.4 Issue Statement

Fisheries and Oceans Canada conducted a pilot study in 1998 on the ecology and community structure of fish and invertebrates within a wild rice bay (Watson et al., 2001). The objective of the project was to compare aquatic communities within a bay dominated by wild rice to an adjacent open water site. Kisseynew Lake ($54^0 58$'N, $101^0 35$'W), situated approximately 16 km north of Flin Flon, Manitoba, was chosen for the study as it was the first lake in the region to be seeded with wild rice (1980) (Watson et al., 2001). The pilot study (Watson et al., 2001) showed that there were differences in fish population, water chemistry, macroinvertebrates, phytoplankton, algae communities, and sediment profiles between sites with wild rice and those in an open water area. Dissolved organic carbon (DOC), total dissolved nitrogen (TDN), and soluble reactive silica (SRS) within wild rice stands were higher than in the open water site in September of 1998, with concentrations that were 30, 50, and 200% higher, respectively. Dissolved oxygen concentrations in September and August within the wild rice stand was 20 and 25% lower, respectively, compared to the open water. Surface temperatures averaged 1.27°C warmer than bottom temperatures in the wild rice stand between June and September, while an average difference of only 0.17°C was observed between the surface and bottom temperatures at the open water site. These results suggest water circulation within the wild rice bed was impeded, thereby resulting in the formation of thermal and chemical gradients. Watson et al. (2001) found that invertebrates were 1.5 and 2.2 times more abundant within the wild rice stand in June and August, respectively. In September, invertebrate abundance within the wild rice stand was 6 times higher when compared to the open water site. Fish communities were also noticeably different between the wild rice stand and the open water site. More northern pike greater than 2 years were found in the open water when compared to the wild rice stand. In contrast, there were more juvenile northern pike within wild rice habitats than in the open water. Increased numbers of white suckers (*Catostomus commersoni*) within the wild rice stand were also observed (Watson et al., 2001). However, during that study (Watson et al., 2001) there was no attempt to compare water quality, temperature and oxygen profiles, and invertebrate and fish communities between bays seeded within wild rice and bays that were still dominated by native macrophyte communities. Comparisons were made between wild rice and open water sites in one lake only. Many of the effects reported for the wild rice bay (Watson et al., 2001) are consistent with the known impacts/benefits of aquatic macrophytes on littoral food webs (Crowder & Cooper, 1982; Dvořak & Best, 1982; Wiley et al., 1984; Keast, 1984; Engel, 1990; Jones et al, 1998), and may not be unique to wild rice in lakes where native macrophytes may also dominate littoral zones.

1.5 Thesis Objectives

This thesis examines the impacts that introduced wild rice (*Zizania palustris* L.) had on the invertebrate communities and the energy flow within three lakes: Barry Lake, Cacholotte Lake, and Naosap Lake in west-central Manitoba, Canada. The second chapter examines the diversity, abundance, and composition of the invertebrate community within wild rice and native macrophyte habitats of these three study lakes. Differences in water quality, temperature profiles, and dissolved oxygen are also discussed. The third chapter focuses on energy flow and food web structure within wild rice and native macrophyte habitats using carbon and nitrogen stable isotopes.

1.6 Related Research

In conjunction with my research describing potential impacts of wild rice on macroinvertebrates and energy flow, Lavergne (2005) examined the potential impacts of wild rice on the littoral fish communities between wild rice and native macrophyte bays. He also examined the physical and chemical characteristics of these littoral areas. These two studies were conducted concurrently and will increase our understanding of the potential impacts of wild rice on aquatic communities in this region.

1.7 Study Area & Design

Study Area

Fieldwork was conducted in August of 2002 and June and August of 2003 in west-central Manitoba near Flin Flon (Figure 1.07). In July of 2002, fifteen lakes were surveyed to find those that had both bays with wild rice and native aquatic vegetation. In August of 2002, preliminary studies were conducted to test sampling protocols and to better understand the biology and life cycle of wild rice in this region. During the winter of 2002 the preliminary data were examined and a new study design was prepared for 2003.

Three lakes were chosen for study in 2003: an unnamed lake locally called Barry Lake (54°57' N, 101°21' W), Cacholotte Lake (54°59' W, 101°38' W), and Naosap Lake (54°51' N, 101°24' W) (Figure 1.07). These lakes had a wild rice and native macrophyte bay of similar size and water depth, were accessible by boat, and had little recreational or commercial fishing pressure. Barry, Cacholotte, and Naosap lakes are oligotrophic and situated in the boreal forest of the Canadian Shield (Figure 1.07). Barry Lake is the smallest lake with a surface area of 1.5 km^2 (Figure 1.08). Cacholotte Lake is larger than Barry Lake with a surface area of 3.5 km^2 (Figure 1.09), and Naosap Lake is the largest of the three systems with a surface area of 24 km^2 (Figure 1.10). Kisseynew Lake (54°58' N, 101°35' W) examined in Watson et al. (2001) was also situated near the study area.

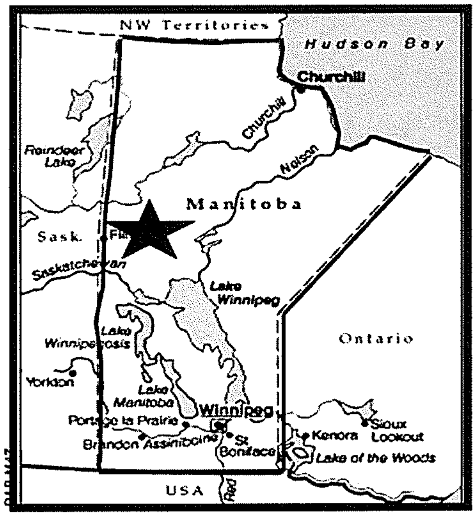
The geology of the region is Precambrian Shield, ranging from volcanic gneissic, quartzite and granite bedrock (Rowe, 1972). The vegetation varies depending on the amount of drainage. Black spruce (*Picea marina*) and jack pine (*Pinus banksiana*) are dominant species on thin soils, while tamarack (*Larix laricina*) and black spruce are found in intervening swamps, and meadows in poorly drained areas. White spruce (*Picea glauca*), trembling aspen (*Populus tremuloides*), and balsam fir (*Abies balsamea*) are found in the well drained areas (Rowe, 1972; Dixon & Derksen, 1999).

The climate in the region is characterized by short, cool summers, although warm spells are common. Winters are long and cold with average temperatures in January of -21.7 °C. The mean annual temperature is -0.9 °C. The average growing season is 159 days with around 1250 growing degree days. The precipitation varies from year to year with the highest precipitation occurring during the summer months. Average precipitation is 70 mm per year (Smith et al., 1998).

Study Design

The study was designed to compare the invertebrate communities in littoral bays of three lakes with either wild rice or native macrophytes as the predominant vegetation. The two main assumptions of the research project were 1) that each bay with wild rice was similar to the bay with native macrophytes before it was seeded and 2) that each native macrophyte bay would be capable of supporting wild rice if it were seeded. Mean differences in abundance, diversity, and composition of invertebrate communities were compared between wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap lakes in June and August of 2003. Lakes were compared in June, during the floating leaf stage and in August, during the emergent stage of wild rice. These two periods were examined to determine if wild rice had different impacts during these unique stages of the wild rice life cycle. In addition, stable carbon and nitrogen isotopes were used to determine if the pathways of the energy flow and trophic levels of aquatic organisms differed between bays with and without wild rice. This study will contribute much needed information on differences in the littoral communities in bays with and without wild rice.

a)



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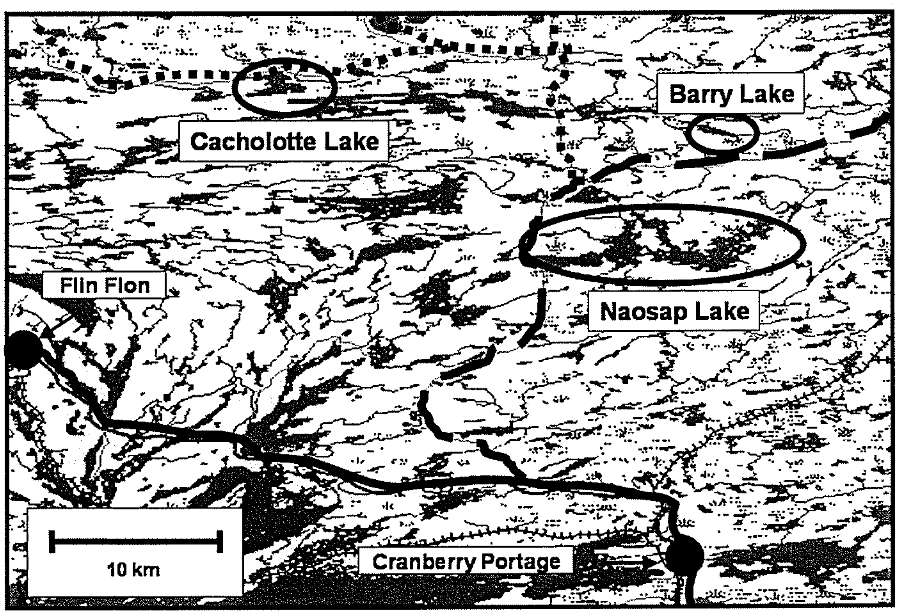


Figure 1.07. Location of A) the study area in Manitoba and B) the three study lakes: Barry, Cacholotte and Naosap. On the map of Manitoba, location is shown by a star between 54°59' and 54°51' North latitudes and between 101°21' and 101°38' West longitudes. On lower map B) Solid line = #10 highway, large dash line = Sherridon Road, small dash line = logging roads.

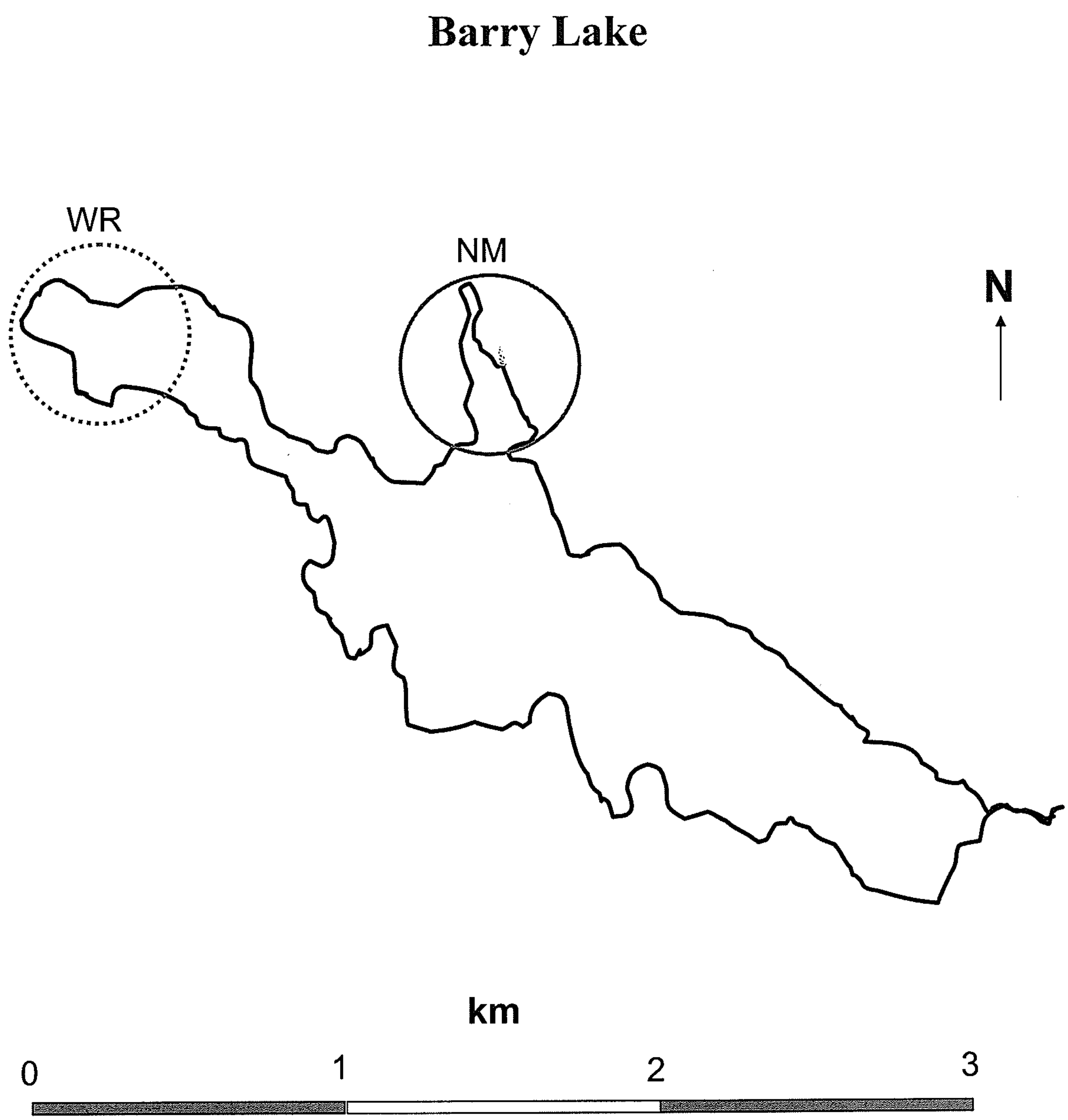


Figure 1.08. Map of Barry Lake in west-central Manitoba showing the locations of the bays with wild rice (WR) and native macrophytes (NM) sampled in this study.

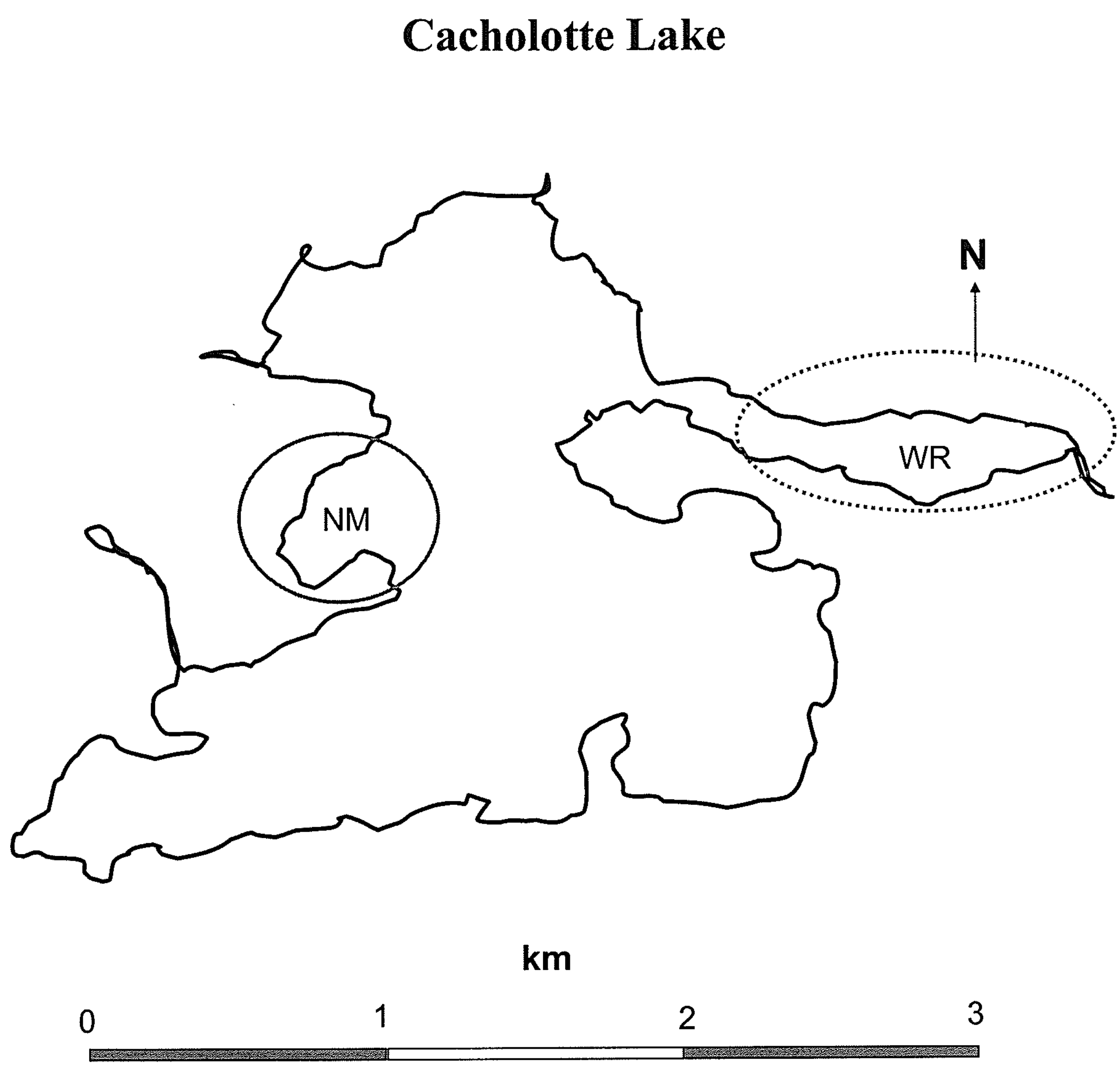


Figure 1.09. Map of Cacholotte Lake in west-central Manitoba showing the locations of the bays with wild rice (WR) and native macrophytes (NM) sampled in this study.

Naosap Lake

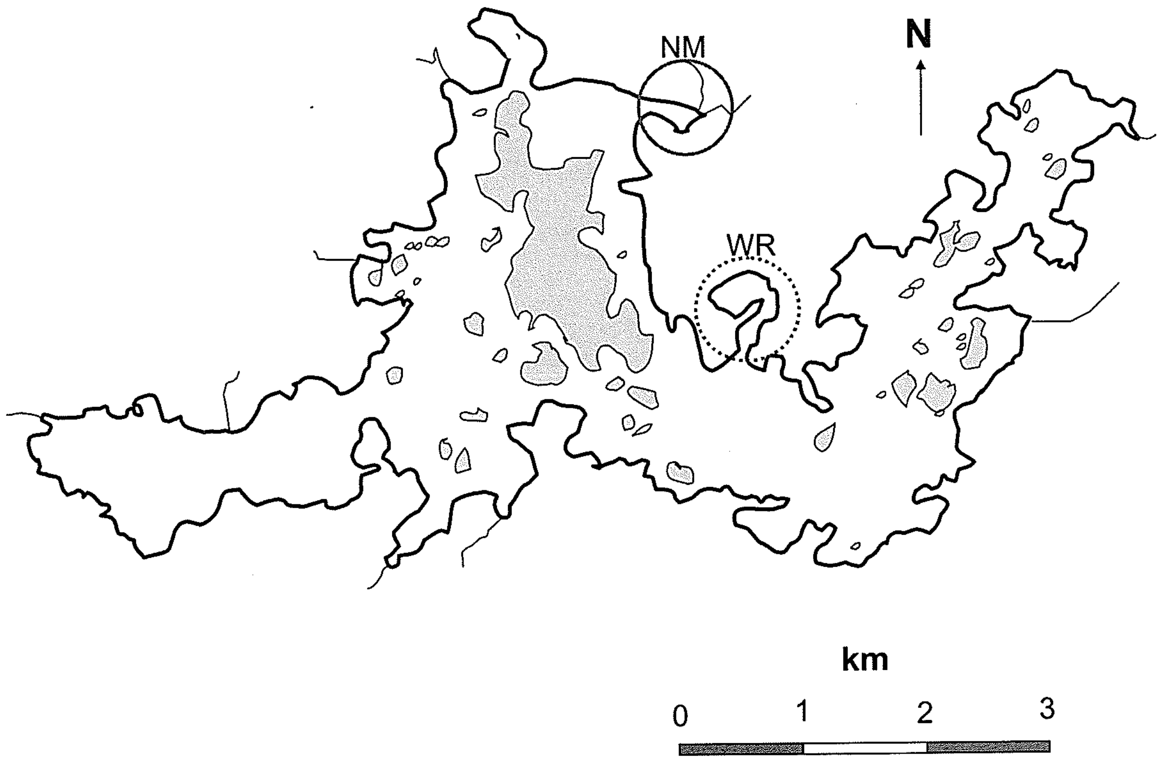


Figure 1.10. Map of Naosap Lake in west-central Manitoba showing the locations of the bays with wild rice (WR) and native macrophytes (NM) sampled in this study. Shaded areas are islands.

Chapter 2

Diversity, abundance, and composition of the macroinvertebrate community within wild rice and native macrophyte bays of lakes in west-central Manitoba

Abstract

With the intentional introductions of wild rice to the west-central region of Manitoba there have been many concerns about the impacts this introduced species will have on invertebrate and fish communities. To examine this I compared the invertebrate abundance, diversity, and community composition between bays with wild rice and bays with native macrophytes in three lakes (Barry, Cacholotte and Naosap) in west-central Manitoba. Environmental variables, including water quality, temperature and dissolved oxygen concentrations, were also compared. Bottle trap, emergence trap, and bucket volume samplers were used to capture invertebrates within both habitats. Paired *t*-test results showed that the abundance and diversity of the invertebrate community was not significantly different between the wild rice and native macrophyte bays ($P>0.05$). However, invertebrate community composition differed between wild rice and native macrophyte bays. Principal Component Analysis (PCA) showed that Amphipoda, Physidae, and Ostracoda were more abundant within the wild rice bays while Haliplidae, Chironomidae, and Gyrinidae were more abundant within native macrophyte bays. Dissolved oxygen concentrations were consistently lower in the wild rice bays of two of the three lakes when compared to native macrophyte bays, whereas water quality was similar between bays. Water temperatures were variable across bays and were not consistently higher or lower in wild rice bays. Although invertebrate communities in wild rice and native macrophytes bays appeared to be equally abundant, community composition varied between the wild rice and non wild rice bays in this study.

2.0 Introduction

In the 1980s and 1990s, wild rice (*Zizania palustris*), an aquatic emergent macrophyte, was intentionally introduced into a number of lakes near Flin Flon, in the west-central region of Manitoba. These introductions were part of the expansion of the commercial wild rice industry in the province (Derksen, 2002). The littoral zones of lakes, which have the highest level of primary productivity and greatest diversity and biomass of macroinvertebrate and fish communities (Werner et al. 1977; Keast et al., 1978), are being transformed from native macrophyte plant communities to wild rice monocultures within this region (Derksen, 2002).

Very little is currently known about the possible impacts these introductions may have on native fish and aquatic invertebrate communities. Previous studies were focused on wild rice biology and on the economics of wild rice production, not on the relationships between wild rice and aquatic communities (Gertzbein, 2000). Only one pilot study conducted by Watson et al. (2001) contrasted a wild rice bay with an open water area to assess chemical, physical and biological differences and to identify areas where future research may be warranted. With the paucity of studies, more research was needed to understand whether the introductions of wild rice have had any adverse affects on native invertebrate and fish communities, as well as the physical and chemical characteristics of these water bodies.

2.0.1. Macrophyte and Invertebrate Interactions

Even though the relationships between wild rice and the invertebrate and fish communities have not been studied, the importance of aquatic macrophytes to the dynamics of an aquatic ecosystem is well documented (Dvořak & Best, 1982; Downing & Cyr, 1985; Carpenter & Lodge, 1986; Engel, 1988; Engel, 1990; Diehl, 1992; Lodge, 1991; Engel, 1995; Hann, 1995). Aquatic macrophytes within littoral zones add physical complexity (Crowder et al., 1998), and provide habitat, food, and shelter for bacteria, algae, protozoans, rotifers, and large invertebrates (Wiley et al., 1984; Engel, 1990; Diehl & Kornijow, 1998; Jones et al., 1998). Moreover, they play a key role in invertebrate production by increasing habitat diversity within aquatic habitats. Macrophytes impede water movement which increases silt sedimentation and creates conditions that are favourable to detritus feeders (Petr, 1968; Engel, 1988; Engel, 1990) and decreases wave action detrimental to epiphytic invertebrates (Cyr & Downing, 1988). Aquatic macrophytes also alter temperature and oxygen gradients (Dale & Gillespie, 1977), reduce light penetration and increase areas of shade (Engel, 1990), act as nutrient sink or source, and alter sediment and water chemistry (Engel, 1990). Submerged macrophytes also substantially increase the surface area available for invertebrates, resulting in large increases in benthic macroinvertebrate productivity and carrying capacity (Wiley et al., 1984). In addition, macrophytes also increase periphyton production by providing additional substrate for colonization and, therefore, an additional food source for epifauna (Cyr & Downing, 1988).

Engel (1990) found approximately 60% more midges (Chironomidae) and 90% more snails (Gastropoda), fingernail clams (Sphaeriidae), larvae of caddisflies (Trichoptera), and damselflies and dragonflies (Odonata) within littoral zones containing macrophytes compared to open water sites. Density and diversity of aquatic invertebrates are also known to be positively correlated with complexity of the macrophyte community (Diehl, 1992). In addition, the unique characteristics of different macrophyte species within a community have allowed invertebrates to specialize in their feeding strategies and to coexist in these microhabitats in close proximity to one another (Diehl, 1992). Some invertebrates feed on the macrophytes and chew or shred living tissue, others feed on the roots of the living macrophytes, and others feed on the decaying plant matter within the water column or on the surface of the sediments. Some invertebrates consume the algae growing on the plants surfaces (Dvořak & Best, 1982; Engel, 1990).

Aquatic macrophytes also influence predator-prey interactions between fish and invertebrates within littoral zones which may lead to increased complexity of the entire aquatic food web (Connell, 1975). Fish movement and foraging efficiencies are hindered within these habitats, resulting in refuges for both invertebrates and forage fish communities (Crowder & Cooper, 1982). In addition, rare species are able to co-exist with their dominant competitors because of the increased habitat complexity that macrophytes provide, resulting in higher species richness and overall diversity (Cyr & Downing, 1988). Aquatic macrophytes are, therefore, an important component to the dynamics and stability of an aquatic habitat (Engel, 1990).

2.0.2. Invertebrates and Food Web Interactions

Aquatic invertebrates provide essential ecosystem functions such as nutrient cycling and the transport and mixing of materials (Downing, 1986; Wallace & Webster, 1996). These primary and secondary consumers are eaten by most fish during one or more stages of their life cycle (Gerking, 1962; Fairchild, 1982; Mittelbach, 1988; Carpenter & Lodge, 1986), and are therefore the major energy pathway connecting the primary producers at the base of the food web to the top predators (Ware & Gasaway, 1978; Teels et al., 1978; Wetzel, 1979; Colle & Shireman, 1980; Mittelbach, 1981; Keast, 1984).

2.0.3. Research Needs and Objectives

The overall objective of this chapter was to determine whether aquatic invertebrate diversity, abundance, and composition differed in bays with wild rice compared to bays with native macrophytes within the littoral zone of lakes in west-central Manitoba. Environmental parameters such as water quality (i.e. pH, alkalinity, nitrate etc.), temperature, and dissolved oxygen were also compared between wild rice and native macrophyte bays. The null hypothesis was that the diversity, abundance and composition of the invertebrate communities would not differ between wild rice and native macrophyte bays of the three lakes examined.

2.1 Materials & Methods

Lakes Sampled

In June and August of 2003, three lakes (Barry, Cacholotte, and Naosap) in west-central Manitoba were studied to contrast invertebrate communities and water quality variables in bays with wild rice (*Zizania palustris*) and bays dominated by the native macrophyte plant communities. These oligotrophic lakes were seeded with wild rice in the early 1980s and were chosen for the present study because they each had a bay with wild rice and a comparable native macrophyte bay that was assumed to be capable of supporting wild rice if it were seeded. Other characteristics used to select the study lakes included accessibility, low fishing pressure, and similar fish communities across lakes. As described below, water quality, plant community composition and % coverage, temperature and dissolved oxygen profiles, and the invertebrate and fish communities in these bays were examined. The field work was conducted from the 10-29 June, 2003 during the floating leaf stage, and 13 August – 1 September 2003 during the emergent stage of the wild rice life cycle.

Percentage of plant cover

Aerial cover was determined for the plant species in the wild rice and native macrophyte bays using the following methods. A 50 m rope marked at 5 m intervals was tied to a stake on the shoreline and stretched to a post in the middle of the bay. With both ends of the rope tightly fastened, a $1m^2$ floating quadrat was positioned at 5 m intervals along the rope and a visual estimate of the percentage of plant cover was recorded. This process was repeated at five locations within each of the wild rice and native macrophyte bays for each of the lakes sampled (Lavergne, 2005).

Water Quality

Four litres of subsurface (0.5 m) water were collected at each of five locations in each lake in both June and August. Two random samples were taken in the native macrophyte bay, two random samples in the wild rice bay and one random sample in the open water. All samples were collected at 4:00 pm. In the field, approximately 150 mL of water from each sample was filtered through three individual GF/C filters for analysis of total carbon, total phosphorus, and chlorophyll *a*. The chlorophyll *a* and the carbon filters were frozen immediately whereas the phosphorus filters were placed in a dark container at room temperature for subsequent analysis. In addition, 350 mL of water from each sample was filtered through an ashed GF/C filter with a nominal pore size of 1.2 μm, collected in a Nalgene ® container, and frozen for later analysis of dissolved nutrients. Water samples and GF/C filters were analyzed by the chemistry lab at the Freshwater Institute in Winnipeg, Manitoba for nitrite, nitrate, ammonia, total dissolved nitrogen, total dissolved phosphorus, soluble reactive phosphorus, dissolved inorganic carbon, dissolved organic carbon, suspended phosphorus, suspended carbon, suspended nitrogen, chlorophyll *a*, pH, and alkalinity using the methods detailed on this website (http://www.umanitoba.ca/institutes/fisheries/Chemistry.html).

Oxygen and Temperature Profiles

Dissolved oxygen and temperature profiles were determined at three random locations within each of the native macrophyte and wild rice bays. Measurements were taken at the surface, at mid-depth, and at 5 cm above the sediment-water interface in water depths ranging from 0.9 to 1.2 m using a calibrated YSI Model 59 Dissolved Oxygen Meter (Lavergne, 2005).

Invertebrate Sample Collection

Because of the variety of habitats available for aquatic invertebrates (Murkin et al., 1994), three sampling devices were used to characterize the invertebrate communities within these bays. Bottle traps, modified from Murkin et al. (1983), were used to capture active invertebrates swimming (nektonic) within the water column (Figure 2.01). These traps allow invertebrates to swim into the bottles through the funnel opening (outer diameter =10 cm; inner diameter = 2.4 cm) located on one side of the trap. These traps were built out of two transparent two litre plastic pop bottles with the tops removed and fastened together (Figure 2.01). Since invertebrates swim in many different directions within the water column, each sample consisted of one bottle set in a horizontal position and one bottle set in a vertical position (Figure 2.01). Invertebrates collected in both bottles were pooled to make one sample. Twenty bottle traps were placed approximately 0.3 meters above the bottom sediments at ten random locations within each of the bays for a 48-hour period once during both the June and August sampling trips (Appendices 1.01 – 1.06). To minimize the consumption of invertebrates by predators, the traps were emptied after 24-hours and then re-set at the same locations for another 24-hour period. Invertebrates collected in traps were sieved using a 250 μm (# 60) mesh screen and the invertebrates collected were immediately placed into 10% formalin for fixation and then transferred to 70% ethanol for preservation prior to identification and counting.

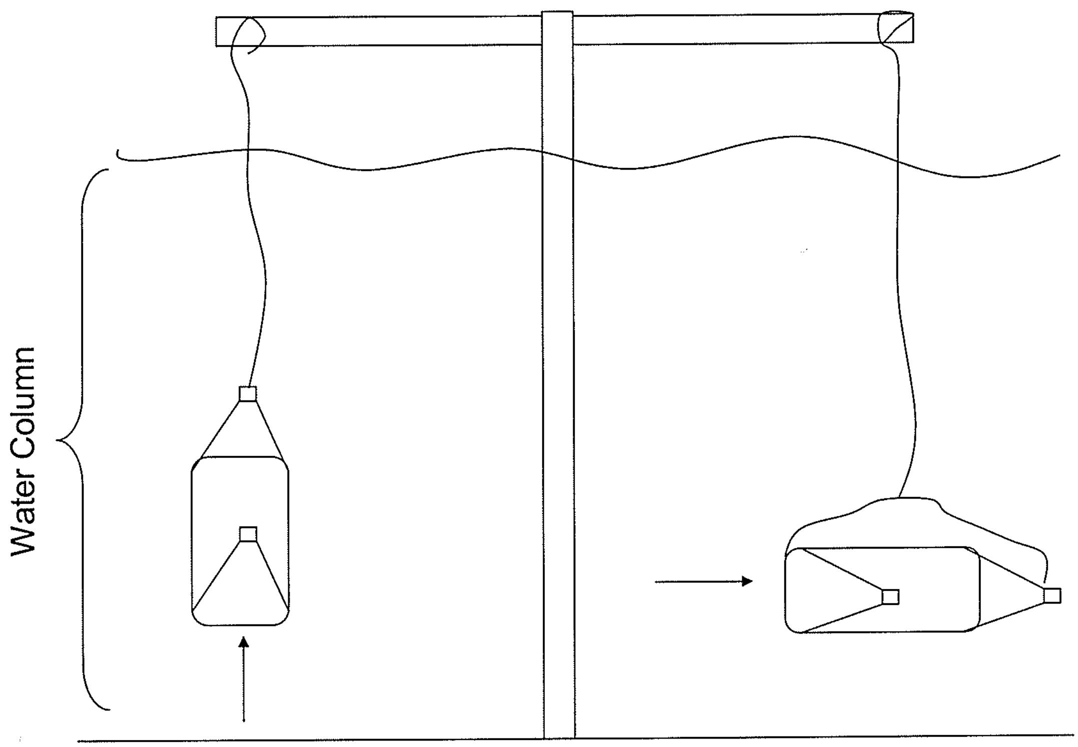


Figure 2.01. Diagram of the bottle trap sampler used to collect active invertebrates in the water column. The diameter of each trap opening was 0.1 m.

Submersed emergence traps, designed by Davies (1984), were used to capture insects emerging from the bottom sediments and water column (Figure 2.02). As the insects metamorphose from an aquatic to a terrestrial stage in their life cycle, they swim up through the water column, break the water surface/air interface and become trapped within the air bubble inside the glass jar located at the top of the trap (Figure 2.02). The funnel opening of each trap was 0.1 m^2 and the traps were made of clear plastic to prevent the insect from seeing and avoiding them during emergence. Insects were removed by covering the base of the trap with a 250 μm (# 60) sieve before lifting the trap out of the water. Ten traps were placed approximately 0.5 m above the bottom sediments for 96-hours at random locations within both the wild rice and native macrophyte bays once during both the June and August sampling trips (Appendix 1.01-1.06). To prevent the adult insects from decomposing while in the trap, the traps were emptied after 48-hours and then reset in the same location for another 48-hour period. All the invertebrates collected were immediately fixed in 10% formalin and then transferred to 70% ethanol for storage prior to identification and counting.

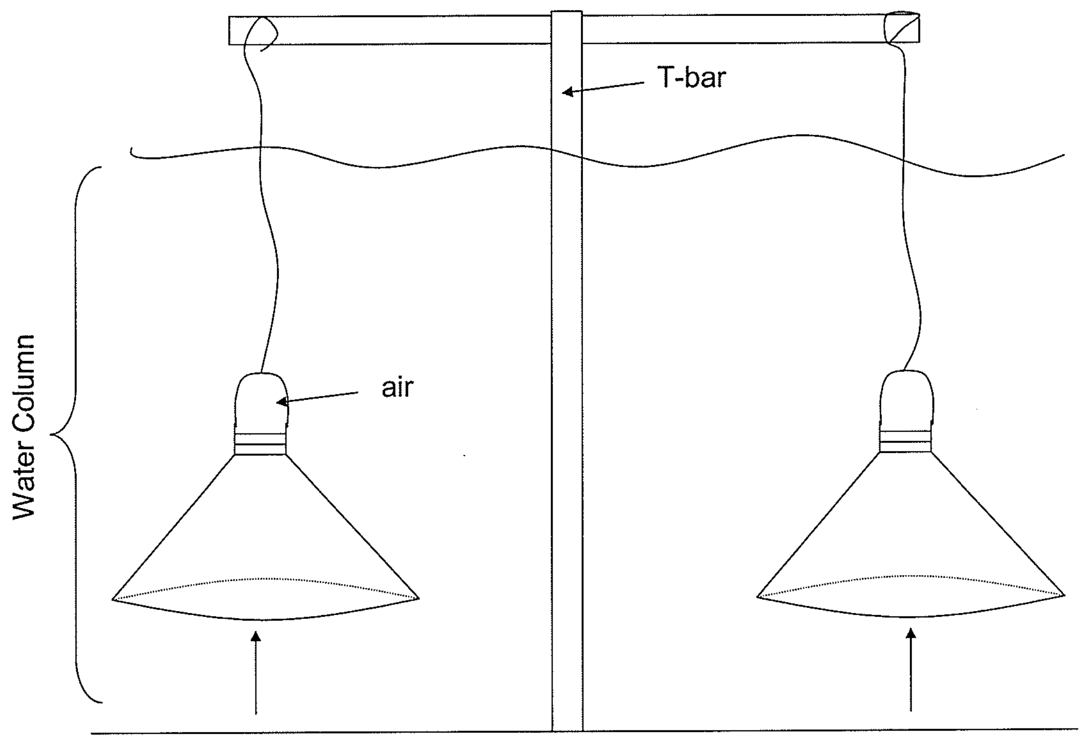


Figure 2.02. Two emergence traps, used to collect emerging invertebrates, suspended from a T-bar anchored in the sediments. The funnel opening of each trap was 0.1 m^2 and the jar was 250 mL in volume.

The final method used to capture invertebrates was a two-meter tall bucket volume sampler modified from the stovepipe sampler designed by Merritt et al. (1984) (Figure 2.03). This sampler was designed to trap all invertebrates on the plants, the surrounding water column, and within the upper sediment layer within a given volume of water. This open-ended bucket (0.55 m diameter) was placed over the plant material and pushed into the sediment at three random, one-meter deep, locations in both the wild rice and native macrophyte bays. Once securely placed in the sediment, a D-frame sweep net with a mesh size of 250 μm was used to recover all invertebrates within the cylinder. The net was swept repeatedly though the contained water volume and the upper sediment layer for fifteen minutes to standardize effort between sampling sites. The plant matter, detritus and debris collected were sieved through a 0.5 mm sieve, and the total mass of the sample was determined. Due to the large amount of organic material collected, twenty thirty-gram subsamples were removed from each sample. The plant material collected within each sample was cut up into small pieces and mixed evenly in each bulk sample so that it could be included in the subsamples. Samples were fixed in 10% formalin. Rose Bengal was added to the subsamples to increase the effectiveness of sorting the invertebrates in the lab because organisms stained pink were more visible within the samples. Borax, a cleaning agent with a basic pH, was also added to each sample to prevent formalin from dissolving the shells of molluscs collected within the bucket volume sampler. In the lab, five subsamples were randomly selected and transferred to 70% ethanol for sorting and identification. The subsamples were placed in sorting trays, and a microscope with 10X magnification was used to process the samples.

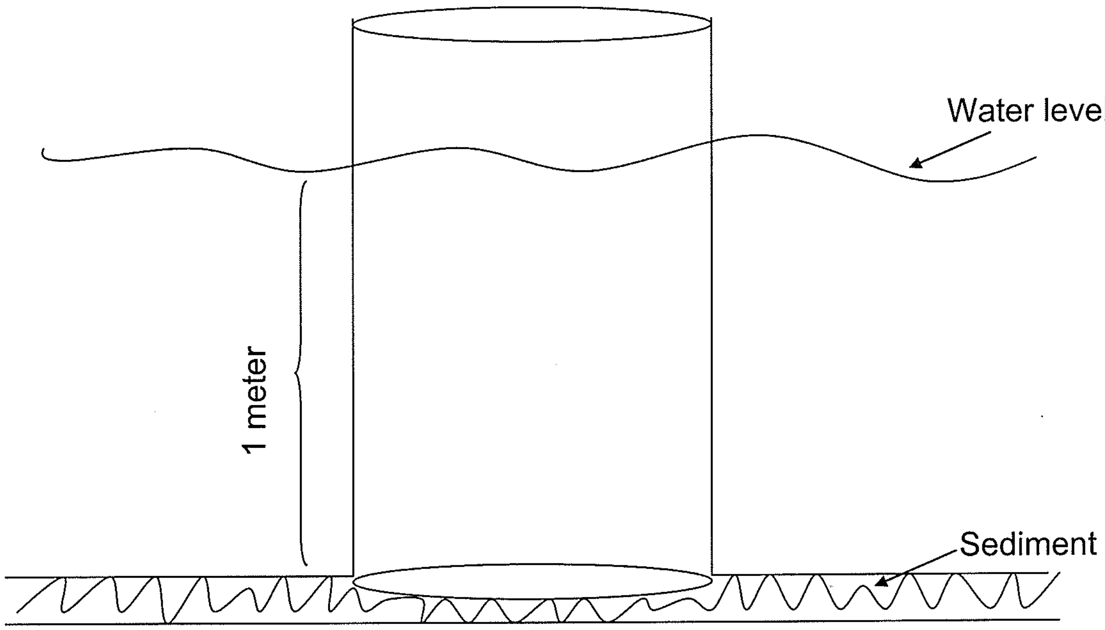


Figure 2.03. A bucket sampler used to collect invertebrates within the sediment, on the plants and in the surrounding water column in wild rice and native macrophyte bays. The bucket was 0.55m in diameter by 2m in height.

Invertebrates were identified to class for Oligochaeta, Nematoda, Amphipoda, Microturbellaria, to order for Arachnidae and to family for the Insecta, Mollusca, and Hirudinea using Merritt and Cummins (1996) and Clifford (1991). Invertebrates less than 250 μm were not included in the comparisons between the wild rice and native macrophyte bays. In addition, Cladocera and Copepoda were not identified or counted.

Data Analysis

1) Environmental variables

Plant community data were obtained from Lavergne (2005). Dissolved oxygen concentrations and temperatures in the surface, middle, and bottom waters were analyzed using a *t*-test comparison for each lake (Minitab statistical software, version 12.1). Water quality data were interpreted without statistical analysis because only two samples were collected from each bay.

2) Invertebrate Identification and Classification

Invertebrates were identified to the lowest taxon as described above and also grouped into functional feeding groups (FFG) which characterize invertebrates based on the morphological and behavioural mechanisms for exploiting foods (Dvořak & Best, 1982; Merritt & Cummins, 1996; Wallace & Webster, 1996; Gullan & Cranston, 2000; Kedzierski & Smock, 2001; Gerino et al., 2003; Varga, 2003). FFG included 1) scrapers that graze algae, periphyton, and epiphytic algae from mineral and organic substrates; 2) shredders that chew or shred plant material either living or decomposing; 3) collectors/gatherers feed primarily on detritus and debris within the water column, occasionally on live organisms, and on fine particulate organic matter (FPOM <1mm diameter); 4) predators consume prey items by either piercing, sucking or fully engulfing; 5) omnivores are generalists feeding on vegetation and prey items; and 6) parasites/parasitoids which obtain their food directly from another organism but do not necessarily cause death to the host (Merritt and Cummins, 1996; Wallace & Webster, 1996). These taxa and FFGs were then compared between the wild rice and native macrophyte bays within and across lakes. In addition, the combined total number of the invertebrates collected in each type of trap was compared between wild rice and native macrophyte bays for all of the lakes examined.

3) Invertebrate Diversity

Effective richness, absolute richness, evenness and abundance were examined for the invertebrate taxa and functional feeding groups collected within the wild rice and native macrophyte bays using data from each of the different samplers. Abundance (standardized by area) is the total number of invertebrates collected. Absolute richness (S) is the total number of taxonomic units or functional feeding groups present. Effective richness ($N2 = \sum_{i=1}^{s} 1/p_i^2$) is the relative proportion of individuals within the community for each taxonomic group; if all groups have equal numbers of individuals, then effective richness is maximized. The range of effective richness is therefore from 1 to S ($N2 = 1 / D$, where D = Simpson's index). Finally, evenness is the ratio between 0 and 1 ($E = N2 / S$) which describes how evenly distributed the taxonomic units or FFGs are within each sample. A *t*-test was used to assess significant differences between the diversity measures in the wild rice and native macrophyte bays for each sampling method and within each of the three lakes (Minitab statistical software, version 12.1).

4) Among Lake Comparisons

Paired *t*-tests were used to determine whether the differences in means for invertebrate abundance and diversity between the native macrophyte and wild rice bays, which are unlikely to equal zero due to sampling variability (Steel et al., 1997), were significantly different across lakes. The main assumption made when using a paired *t*-test is that there is a normal distribution for the individual communities examined. Advantages of using paired *t*-tests are that: 1) they target a common variable between the paired samples, for instance the presence or absence of wild rice, and 2) there is a smaller within-site variance due to the fact that other variance is controlled, e.g. among-lake differences in productivity, increasing the power of detecting differences across paired observations. In addition, once the mean, standard deviation and confidence limits are known it is easy to calculate the *t*-test by hand, noting that the hypothesized difference is zero (Smith, 1999).

Paired *t*-tests were used to examine the differences in effective richness, absolute richness, evenness and abundance of the invertebrate communities between wild rice and native macrophyte bays across lakes. In addition, a paired *t*-test was performed on the abundances of invertebrate functional feeding groups to determine whether there were significant differences between wild rice and native macrophyte bays. Paired *t*-test was also performed on the eleven most abundant invertebrates captured (equal to 95% of total invertebrate abundance) between wild rice and native macrophyte bays.

For a two-tailed test, the confidence interval was calculated as follows

$$d - t_{\alpha/2}\ (sd/n) \text{ to } d + t_{\alpha/2}\ (sd/n)$$

where: d = sd/n, where d = x1 - x2, and x1 and x2 are means of observations from populations 1 and 2, respectively

$t_{\alpha}/2$ = the value from a t-distribution where α is 1 - confidence level / 100

sd = the standard deviation of the differences

n = number of pairs of values

(Minitab statistical software, version 12.1)

5) Principal Component Analysis

Ordination was used to examine trends in the invertebrate communities in wild rice and native macrophyte bays. Ordination is a method used to simplify biological data into multi-dimensional space allowing one to observe trends that otherwise would be difficult to observe within a data set (Kenkel et al., 2002). In an ordination diagram, the distance between two points represents the similarity between variables; data points close together share similar characteristics while points with a large separation are dissimilar. The axes of the ordination diagram explain the amount of variation described within the data with the first axis explaining the greatest amount of variation followed by the second, third, and so on. Each axis has an eigenvalue that symbolizes the variation explained by that particular axis. The combined total value of the eigenvalues within a diagram equals the total amount of variation explained by the individual graph. Although many eigenvalues may be present within a data set, in an ordination diagram, only the first two are usually shown on the first and second axes. Plots with individual samples can be plotted against the species and environmental variables simultaneously to reveal trends within the data (Kenkel et al., 2002).

Principal Component Analysis (PCA) is an ordination technique used to describe the relationship between two quantitative variables, such as invertebrate abundance and sample sites (Kenkel et al., 2002). PCA can be used to separate plots based on attributes such as species abundance and habitat characteristics (wild rice vs. native macrophytes). SYN-TAX 2000 was used for data analysis and the data were expressed in graphical form as a scattergram (SYN-TAX 2000, version 5.1).

In this study PCA was used to reveal patterns between invertebrate taxa or functional feeding group (FFG) abundances in wild rice and native macrophyte bays. These data were transformed by taking the log of the raw data (Log $n + 1$) to prevent the data from being skewed by large differences in abundances of the taxa or FFGs.

6) Multiple Discriminant Analysis

Multiple Discriminant Analysis (MDA) was used to compare statistically the PCA results and to examine whether invertebrate taxa and functional feeding data of the predetermined groups (wild rice and native macrophytes) were significantly different in the bottle trap samples. MDA is similar to PCA, however, the objective is to find best linear fit that maximally discriminates among the groups (wild rice or native macrophytes). In MDA, there is an additional constraint that each individual has been assigned a priori to one group. The component scores obtained from the PCA for the taxa and functional feeding group data were used for MDA. It was essential to use the component scores of the PCA to reduce the axes for MDA as 10, 000 individuals in the 52 invertebrate taxa would add excessive dimensionality to the MDA, leading to uninterruptible results. Since only two groups were compared (wild rice and native macrophytes), the output from the MDA resulted in a one dimension scattergram, and therefore, frequency graphs were used to illustrate the position on the axis where the replicates for each group lie. The chi^2 values and the degree of freedom obtained from the MDA (frequency graphs) were examined on a chi^2 table to obtain *P* values and identify significant differences between the two groups (wild rice vs. native macrophytes). A 95% confidence interval was used to interpret significant differences ($\alpha = 0.05$). SYN-TAX 2000 (version 5.1) was used for data analysis and the data were expressed as frequency graphs created with Data Desk (version 6.1).

2.2 Results

Plant Communities

Plant communities in bays with wild rice were clearly different from bays dominated by native macrophytes (Figure 2.04). In all bays with introduced wild rice, this plant made up between 89 and 93% of the plant cover. Floating-leaved pondweed (*Potamogeton natans*) was the dominant plant within the native macrophyte bays of Barry and Cacholotte lakes in June and August of 2003 (Figure 2.04), and represented between 64 and 78 % of the cover. On Naosap Lake in the native macrophyte bay, thread-leaved pondweed (*Stuckenia filiformis*) and linear-leaved pondweed (*Potamogeton sp.*) were the dominant plant species, representing 85% of the cover. In addition, species richness was consistently higher within native macrophyte bays, with 2-6 more species occurring at these sites (Figure 2.04).

Mean aerial plant cover was higher in wild rice bays compared to native macrophyte bays in Barry and Cacholotte lakes in June. In Naosap Lake, plant cover was similar in the wild rice and native macrophyte bays (Figure 2.04). Plant cover in native macrophyte bays on all three lakes increased from June to August as the macrophytes developed over the season. In contrast, the aerial cover of wild rice did not change from June to August. Wild rice stands grew vertically between June and August whereas native macrophytes grew both horizontally and vertically within the water column (Figure 2.04). However, no significant differences in mean cover of macrophytes were found between wild rice and native macrophyte bays for Barry, Cacholotte, and Naosap lakes as there were large variations in cover within each bay (Lavergne, 2005).

The architecture of the plants also differed in wild rice and native macrophyte habitats. Wild rice stands consisted of single stems reaching from the sediment to the water surface, while native macrophytes were broad-leaved, narrow-leaved, or needled, and were either floating on the water surface or submerged within the water column (M. Lowdon, personal observation). Individual native macrophyte plants were more complex and likely had a greater surface area than an individual wild rice stem, although this was not examined in this study (Figure 1.06 – Chapter 1). Overall, plant community abundance, physical appearance, surface area, and complexity were noticeably different between wild rice and native macrophyte bays.

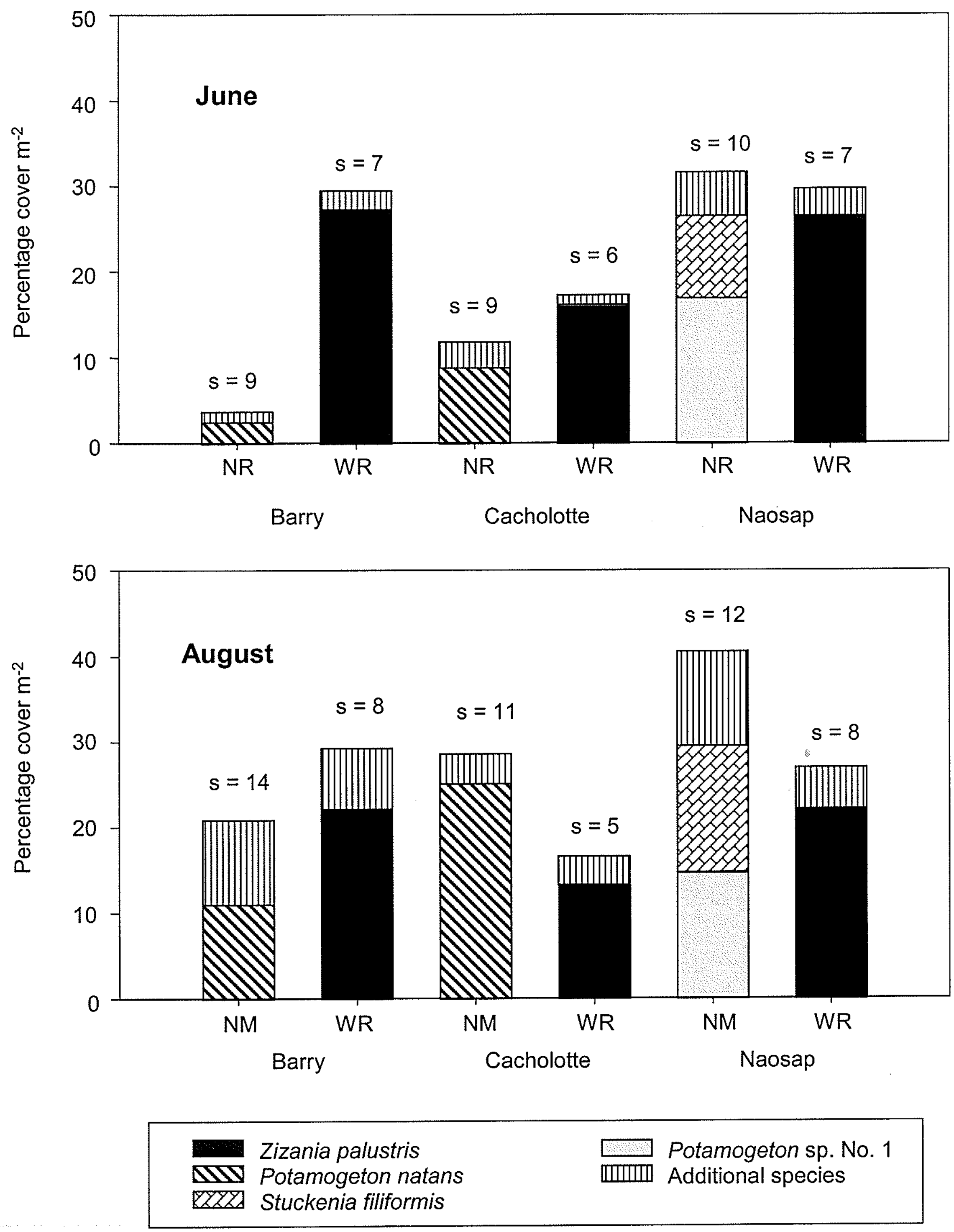


Figure 2.04. Mean percentage of plant cover (m^{-2}) for species found in native macrophyte (NM) and wild rice (WR) bays of three west-central Manitoba lakes during June (top) and August (bottom), 2003 (results obtained from Lavergne, 2005). s = species richness.

Water Quality

Fifteen different water quality parameters were measured in bays with wild rice and bays with natural vegetation. No consistent differences were observed between wild rice and native macrophyte bays in nitrate (NO_3-N), nitrite (NO_2-N), soluble reactive phosphorus (SRP), total dissolved phosphorus (TDP), suspended phosphorus (SUSP P), chlorophyll *a* (CHL-a), conductivity (COND), pH, and alkalinity (ALK) in June (Table 2.01). Ammonia (NH_4-N), dissolved organic carbon (DOC) and total dissolved nitrogen (TDN) were consistently higher in wild rice bays compared to native macrophyte bays in all lakes in June. In contrast, suspended nitrogen (SUSP N) and suspended carbon (SUSP C) were slightly lower within the wild rice bays on all lakes in June. In August, dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC) were consistently higher in wild rice bays compared to native macrophyte bays. For all other water quality parameters, no noticeable differences were observed between bays within each lake (Table 2.01).

Table 2.01. Mean water quality values (n=2) for littoral bays with wild rice and with native macrophytes on three lakes within west-central Manitoba in June and August, 2003. Symbol; NM = native macrophyte bays; WR = wild rice bays.

	Water quality parameters	Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
	NO3-N (μg/l)	0.75	0.5	0.5	1.5	1	1
	NO2-N (μg/l)	1	1	2	1.5	0.5	1
	NH4-N (μg/l)	10.5	16	9	12.5	6	16
	TDN (μg/l)	590	648	618	675	470	563
	SRP (μg/l)	1.5	1.5	3	1.5	1	1.5
	TDP (μg/l)	9	10.5	18	15.5	6	8.5
	DIC (μm/l)	419	426	398	397	749	745
June	DOC (μm/l)	1329	1418	1416	1596	970	1010
	SUSP N (μg/l)	125.5	115.5	156.5	85	140.5	120.5
	SUSP P (μg/l)	9	11	14.5	10	8.5	22.5
	SUSP C (μg/l)	980	945	1080	570	1070	880
	CHL-a (μg/l)	2.4	2.82	3.98	2.21	6.42	1.45
	COND (μS/cm)	58.5	60	54	55	85.5	85
	pH	7.86	7.89	7.82	7.82	8.11	8.10
	ALK (μeq/l)	455	469	431	432	733	710
	NO3-N (μg/l)	0.5	0.5	3	1.3	0.5	0.5
	NO2-N (μg/l)	1	2	2	2	1	1
	NH4-N (μg/l)	9	11.5	63.5	14	9	13.5
	TDN (μg/l)	560	625	730	727.5	487.5	755
	SRP (μg/l)	0.5	0.5	5	0.5	0.5	0.5
	TDP (μg/l)	10.5	11.5	20	10.5	6	8.5
	DIC (μm/l)	413.5	461	437.5	472	778.5	798
August	DOC (μm/l)	1431	1453	1510	1771	1028	1430
	SUSP N (μg/l)	74	42.5	389	77.5	120	197.5
	SUSP P (μg/l)	20	5.5	38.5	6.5	10.5	12
	SUSP C (μg/l)	705	475	2110	765	1040	2009
	CHL-a (μg/l)	2.12	1.69	5.36	2.95	4.65	7.13
	COND (μS/cm)	58.5	61	56.5	59	89	91.5
	pH	7.82	7.71	7.68	7.68	8.02	8.05
	ALK (μeq/l)	453	466	444	488	754	802

Dissolved Oxygen

In August, dissolved oxygen concentrations were significantly lower at all three depths examined in the wild rice bays when compared to the native macrophyte bays of Naosap Lake and Barry Lake. August measurements in Cacholotte Lake revealed significantly lower oxygen concentrations in the middle of the water column within the wild rice bay when contrasted with the native macrophyte bay (Table 2.02). In June, there were no significant differences in dissolved oxygen concentrations in the wild rice and native macrophyte bays in Cacholotte Lake (Table 2.02; data not available for Barry Lake). Significantly lower surface oxygen concentrations ($p = 0.022$) were observed in the wild rice bay on Naosap Lake in June of 2003. Overall, dissolved oxygen concentrations were generally lower in wild rice bays than in native macrophyte bays.

Table 2.02. Mean dissolved oxygen concentration (p.p.m) (n=3) for littoral bays with wild rice (WR) and with native macrophytes (NM) on three lakes within west-central Manitoba in June and August, 2003. Symbol: nd = no data; ± = standard deviation (data from Lavergne, 2005).

		Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
June	10 cm below surface	nd	nd	7.7 ± 0.36	8.3 ± 0.10	8.6 ± 0.17	7.8 ± 0.26 ★
	Mid-depth	nd	nd	7.5 ± 0.30	8.1 ± 0.20	8.3 ± 0.20	7.6 ± 0.40
	5 cm above sediment	nd	nd	7.3 ± 0.17	7.7 ± 0.20	8.2 ± 0.17	7.7 ± 0.30
August	10 cm below surface	6.7 ± 0.44	5.1 ± 0.20 ★	5.7 ± 0.89	3.6 ± 0.46	7.5 ± 0.26	4.6 ± 0.35 ★
	Mid-depth	6.5 ± 0.30	4.8 ± 0.10 ★	5.6 ± 0.53	3.3 ± 0.20 ★	6.8 ± 0.08	4.5 ± 0.35 ★
	5 cm above sediment	6.2 ± 0.46	4.6 ± 0.10 ★	3.8 ± 1.48	3.2 ± 0.26	6.3 ± 0.50	4.0 ± 0.17 ★

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$, no ★ = no significant difference at $p < 0.05$. Results obtain from a t-test.

Water Temperature Profiles

Temperatures within wild rice and native macrophyte bays were more variable than dissolved oxygen concentrations. On Cacholotte Lake, significantly higher temperatures were recorded at the surface and at mid-depth in the native macrophyte bay in June ($p = 0.006$ and 0.0001, respectively) and August ($p = 0.028$ and 0.008, respectively) of 2003 (Table 2.03) than in the wild rice bay. On Barry Lake in August, the water temperatures at the surface and at mid-depth were not significantly different between the wild rice and native macrophyte bays. However, the water temperature at the sediment-water interface was significantly higher in the native macrophyte bay. In contrast, in Naosap Lake in August, significantly higher water temperatures were found at all three depths within the wild rice bay. In June in Naosap Lake, there were no significant differences observed across bays (Table 2.03). Water temperatures did not show the consistent between-bay trends that the oxygen data did.

Table 2.03. Mean water temperatures (°C) (n=3) in littoral bays with wild rice (WR) and with native macrophytes (NM) on three lakes within west-central Manitoba in June and August, 2003. Symbol nd = no data, ± = standard deviation (data from Lavergne, 2005).

		Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
June	Surface-water interface	nd	nd	20.5 ± 0.5	18.0 ± 0.0 ★	19.8 ± 0.29	19.8 ± 0.29
	Mid-depth	nd	nd	20.2 ± 0.29	17.8 ± 0.29 ★	19.7 ± 0.29	19.3 ± 0.29
	Sediment-water interface	nd	nd	20.2 ± 0.29	17.5 ± 0.0	18.5 ± 1.32	19.2 ± 0.29
August	Surface-water interface	24.5 ± 0.0	24.8 ± 0.29	23.8 ± 0.28	21.7 ± 0.58 ★	17.2 ± 0.29	18.8 ± 0.29 ★
	Mid-depth	24.5 ± 0.0	24.5 ± 0.0	23.3 ± 0.25	21.8 ± 0.29 ★	17.0 ± 0.0	18.8 ± 0.29 ★
	Sediment-water interface	24.5 ± 0.5	23.2 ± 0.29 ★	22.5 ± 0.50	21.7 ± 0.29	17.0 ± 0.0	18.5 ± 0.50 ★

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$. no ★ = no significant difference at $p < 0.05$. Results obtained from t-test.

Invertebrate Community Composition and Abundance

Fifty-two invertebrate taxonomic groups and seven functional feeding groups (FFGs) were sampled in the wild rice and native macrophyte bays using bottle traps, emergence traps, and bucket volume sampler (Tables 2.04 - 2.07). Because invertebrates have higher abundance at the beginning of the season, the total number of invertebrates sampled in June for both the wild rice and native macrophytes bays from all three lakes were approximately two times higher than those in August.

Bottle Traps

In the bottle trap samples the most abundant invertebrate groups were Hydracarina (water mites), Ostracoda, Oligochaeta, Amphipoda, Chironomidae, Corixidae, Hydrobiidae, Microturbellaria, Haliplidae, Tanypodinae (Chironomidae predators), and Physidae. Combined, these groups accounted for 95% of the invertebrates collected in bottle trap samples (Tables 2.04). The remaining 41 invertebrate taxa made up 5% of all invertebrates collected in the wild rice and native macrophyte bays. Of these 41 taxa, 26 were found in low numbers and together represented only 1% of the total abundance of the invertebrate community in either the wild rice or native macrophyte bays (Table 2.04).

Predators, collectors/gatherers, and omnivores constituted 58, 19, and 14%, respectively, of the total numbers of invertebrates captured with bottle trap samples in both wild rice and native macrophyte bays (Table 2.05). In June, collectors/gatherers were 2.7 times more abundant, omnivores were 2.6 times more abundant and predators were 1.8 times more abundant in wild rice bays compared to native macrophyte bays. In August, the abundances of these FFGs were similar between wild rice and native macrophyte bays (Table 2.05). The remaining four FFGs, scrapers, shredders, parasites/parasitoids, and filter-feeders contributed 4.0, 2.2, 2.1, and 0.1%, respectively, of the total numbers of invertebrates captured with bottle traps. Of these groups, parasites/parasitoids were nine and two times more abundant in wild rice bays compared to native macrophyte bays in June and August, respectively. Similar results were found with the scrapers being 4.5 times more abundant in June and 1.15 times more abundant in August in wild rice than native macrophyte habitats. In contrast, the shredders were 14 and 3.4 times more abundant within native macrophyte bays compared to wild rice bays in both June and August, respectively.

Mean abundance of the most dominant individual invertebrates (95% of sample) were compared between wild rice and native macrophyte bays for the three lakes using *t-tests*. In June, Oligochaeta were significantly more abundant in wild rice bays in all lakes compared to native macrophyte bays. Hydracarina, Ostracoda, and Physidae were also significantly more abundant in the wild rice bays in Barry and Naosap lakes compared to native macrophyte bays. However, in Cacholotte Lake, Ostracoda were significantly more abundant within the native macrophyte bay compared to the wild rice bay in June. Corixidae abundance varied from lake to lake and was significantly higher in the wild rice bay in Barry, and significantly lower in the wild rice bay on Cacholotte compared to the native macrophyte bays. Amphipoda, Microturbellaria and Tanypodinae were significantly more abundant in the wild rice bay on Naosap Lake compared to the native macrophyte bay. Haliplidae was the only taxon with significantly higher abundances in the native macrophyte bays when compared to the wild rice bays in more than one lake (Cacholotte and Naosap) (Figure 2.06). Chironomidae and Hydrobiidae collected with this trap had similar abundances within wild rice and native macrophyte bays on all lakes in June.

In August, fewer among-bay differences were observed in mean abundances of the most dominant invertebrates collected with the bottle traps. Hydracarina, Oligochaeta, Amphipoda, Hydrobiidae, Haliplidae, Tanypodinae and Physidae had similar abundances between the wild rice and native macrophyte habitats in Barry, Cacholotte, and Naosap lakes (Figure 2.07). Ostracoda had significantly higher abundances in the wild rice bays in Cacholotte and Naosap lakes. Corixidae and Microturbellaria were also significantly more abundant within the wild rice bay on Naosap Lake (Figure 2.07). In contrast, Chironomidae was significantly more abundant in the native macrophyte bay on Cacholotte Lake (Figure 2.07). In summary, differences between bays were more apparent in June, with wild rice bays typically having significantly higher abundances of each taxa than was found in the native macrophyte bays during this sampling period.

Table 2.04. Total numbers of invertebrates captured using bottle traps within native macrophyte and wild rice bays in Barry, Cacholotte and Naosap lakes 2003. FFG = Functional Feeding Groups, Pred = predators, Omn = omnivores, Col/Gath = collectors/gatherers, Scrap = scrapers, Para = parasites/parasitoids, Shre = shredders, Filt = filter-feeders, herb = herbivores.

Groups	# of individuals	FFG	Total # of individuals per taxonomic group in each zone			
			NM June	WR June	NM August	WR August
Hydracarina	4981	Pred	1214	2169	754	844
Ostracoda	1389	Omn	204	529	230	426
Oligochaeta	659	Col/Gath	70	417	121	51
Amphipoda	603	Col/Gath	59	200	151	193
Chironomidae (herb)	544	Col/Gath	148	201	121	74
Corixidae	406	Pred	153	156	50	47
Hydrobiidae	237	Scrap	83	96	38	20
Microturbellaria	208	Para	15	148	15	30
Haliplidae	204	Shre	155	8	34	7
Tanypodinae	148	Pred	17	83	23	25
Physidae	130	Scrap	6	91	16	17
Caenidae	53	Col/Gath	23	13	5	12
Chaoboridae	49	Pred	3	26	7	13
Glossiphonidae	41	Pred	13	18	8	2
Planorbidae	35	Scrap	6	14	5	10
Dytiscidae	34	Pred	10	23	0	1
Baetidae	32	Col/Gath	6	10	9	7
Leptoceridae	32	Omn	7	11	5	9
Lymnaeidae	24	Scrap	2	5	10	7
Gyrinidae	21	Pred	11	5	5	0
Coenagrionidae	20	Pred	1	12	4	3
Valvatidae	20	Scrap	1	4	5	10
Ceratopogonidae	13	Pred	3	6	3	1
Polycentripodidae	12	Pred	1	1	5	5
Araneae	11	Pred	1	3	5	2
Siphlonuridae	11	Col/Gath	4	7	0	0
Leptophlebidae	9	Col/Gath	0	0	9	0
Notonectidae	9	Pred	0	2	1	6
Scelionidae	9	Col/Gath	0	0	2	7
Erpobdellidae	8	Pred	1	5	2	0
Libellulidae	6	Pred	0	1	4	1
Ephydridae	5	Col/Gath	0	2	1	2
Pyralidae	5	Shre	1	1	0	3
Aeshnidae	4	Pred	0	3	0	1
Eulophidae	3	Para	0	1	0	2
Limnephilidae	3	Col/Gath	1	0	1	1
Sphaeridae	3	Filt	0	2	1	0
Braconidae	2	Para	0	2	0	0
Chrysomelidae	2	Shre	0	2	0	0
Culicidae	2	Filt	0	1	1	0
Ephemerellidae	2	Col/Gath	0	0	1	1
Nematoda	2	Para	0	0	1	1
Nematomorpha	2	Para	1	1	0	0
Plecoptera	2	Col/Gath	1	1	0	0
Ancylidae	1	Filt	0	0	0	1
Collembola	1	Col/Gath	0	0	0	1
Curculionidae	1	Col/Gath	0	0	1	0
Cyclorrhaphous-Brachycera	1	Col/Gath	0	0	0	1
Muscidae	1	Col/Gath	0	1	0	0
Piscicolidae	1	Para	0	1	0	0
Sciomyzidae	1	Pred	0	1	0	0
Tabanidae	1	Pred	0	1	0	0
Total # of Individuals	10003		2221	4284	1654	1844

Table 2.05. Total numbers of invertebrates for each functional feeding group captured using bottle trap samples in the native macrophyte and wild rice bays in Barry, Cacholotte and Naosap lakes in June and August, 2003.

Groups	# of individuals	Total # of individuals per taxonomic group in each zone			
		NM June	WR June	NM August	WR August
Predators	5765	1428	2515	871	951
Collectors & Gatherers	1927	312	852	420	343
Omnivores	1421	211	540	235	435
Scrapers	446	98	210	74	64
Parasites & Parasitoids	227	16	153	18	40
Shredders	211	156	11	34	10
Filter-feeders	6	0	3	2	1
Total # of Individuals	10003	2319	4497	1730	1909

Table 2.06. Mean abundance (± standard deviation) of dominant invertebrate taxa captured with bottle trap samples in native macrophyte (NM) and wild rice (WR) bays in Barry, Cacholotte and Naosap lakes, June, 2003.

	Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
Hydracarina	22.0 ± 10.6	**69.2 ± 33.7** *	74.8 ± 44.6	76.4 ± 43.8	24.6 ± 24.2	**71.3 ± 46.6** *
Ostracoda	2.5 ± 1.96	**7.3 ± 5.01** *	17.6 ± 11.3	**3.70 ± 3.74** *	0.30 ± 0.48	**41.9 ± 45.9** *
Oligochatea	0.10 ± 0.32	**1.70 ± 1.16** *	4.60 ± 5.50	**15.70 ± 12.60** *	2.40 ± 4.25	**24.3 ± 28.0** *
Amphipoda	2.30 ± 4.62	5.20 ± 5.57	2.50 ± 2.55	5.10 ± 3.60	1.10 ± 1.66	**9.7 ± 11.3** *
Chironomidae (herbivores)	3.10 ± 1.91	9.30 ± 9.45	8.10 ± 5.15	5.00 ± 4.24	3.60 ± 3.20	5.80 ± 4.82
Corixidae	0.80 ± 1.03	**4.50 ± 3.41** *	13.50 ± 9.31	**4.50 ± 3.24** *	1.00 ± 1.49	6.60 ± 9.08
Hydrobiidae	3.50 ± 3.69	3.00 ± 2.40	4.60 ± 4.86	5.90 ± 5.49	0.20 ± 0.42	0.70 ± 1.34
Microturbellaria	1.20 ± 1.40	2.30 ± 2.50	0.10 ± 0.32	9.2 ± 13.4	0.30 ± 0.95	**3.30 ± 3.47** *
Haliplidae	5.20 ± 7.76	0.10 ± 0.32	4.50 ± 2.46	**0.60 ± 1.90** *	5.80 ± 6.30	**0.10 ± 0.32** *
Tanypodinae	0.60 ± 0.97	2.20 ± 2.04	0.80 ± 1.03	1.70 ± 1.89	0.30 ± 0.48	**4.40 ± 2.95** *
Physidae	0.20 ± 0.42	**2.30 ± 2.36** *	0.20 ± 0.42	2.10 ± 3.18	0.20 ± 0.42	**4.70 ± 3.47** *

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$, no ★ = no significant difference at $p < 0.05$. Results obtained from a *t-test*.

Table 2.07. Mean abundance (± standard deviation) of dominant invertebrate taxa captured with bottle trap samples in native macrophyte (NM) and wild rice (WR) bays in Barry, Cacholotte and Naosap lakes, August, 2003.

	Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
Hydracarina	40.5 ± 21.2	47.8 ± 29.9	27.2 ± 17.6	24.50 ± 8.50	7.70 ± 6.06	12.10 ± 5.76
Ostracoda	17.20 ± 9.67	20.2 ± 10.6	3.90 ± 2.38	**11.70 ± 6.83** *	1.90 ± 2.13	**10.70 ± 7.33** *
Oligochatea	3.10 ± 3.54	0.80 ± 1.32	6.50 ± 6.67	2.30 ± 1.77	2.50 ± 1.84	2.00 ± 2.16
Amphipoda	6.50 ± 3.34	6.50 ± 4.48	2.60 ± 4.12	3.50 ± 3.92	6.00 ± 4.92	9.30 ± 7.85
Chironomidae (herbivores)	3.70 ± 1.83	4.00 ± 2.83	7.20 ± 6.16	**1.70 ± 1.70** *	1.20 ± 1.32	1.70 ± 1.49
Corixidae	3.20 ± 2.66	2.70 ± 2.83	1.80 ±2.82	1.00 ± 1.05	0.10 ± 0.32	**1.00 ± 0.67** *
Hydrobiidae	2.40 ± 2.12	1.70 ± 1.49	1.40 ± 2.46	0.30 ± 0.48	N/A	N/A
Microturbellaria	1.00 ±1.25	1.60 ± 1.51	0.50 ± 0.85	0.10 ± 0.32	0.10 ± 0.32	**1.40 ± 1.78** *
Haliplidae	0.100 ± 0.32	0.40 ± 1.26	2.70 ± 4.14	0.20 ± 0.63	0.70 ± 0.82	0.10 ± 0.32
Tanypodinae	1.30 ± 1.89	1.30 ± 1.49	0.90 ± 1.29	0.60 ± 0.84	0.10 ± 0.32	0.60 ± 0.70
Physidae	1.00 ± 0.94	0.90 ± 0.88	0.30 ± 0.68	0.70 ± 0.82	0.30 ± 0.68	0.10 ± 0.32

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$, no ★ = no significant difference at $p < 0.05$. Results obtained from a *t-test*.

Emergence Traps

For the emergence trap samples, 100% of invertebrates captured were insects and approximately 80% were from the family Chironomidae (Table 2.08). Therefore, data concerning only the abundance of the invertebrate taxa were compared between the wild rice and native macrophyte bays because the functional feeding group composition was essentially the same (Tables 2.08 and 2.09). Although Chironomidae (herbivores) made up 80% of the invertebrates captured in the emergence traps, there were differences in abundances between wild rice and native macrophyte bays. Chironomidae were 1.3 and 1.6 times more abundant within native macrophyte bays during June and August, respectively. Caenidae, which made up 9% of the total numbers of emerging insects captured, were two and three times more abundant within the native macrophyte bays compared to the wild rice bays in June and August, respectively. Tanypodinae (predacious chironomids) made up only 7% of the invertebrate numbers but were two times more abundant in the native macrophyte bays compared to the wild rice bays in June. In August, there were no observed differences between the abundance of Tanypodinae within the wild rice and native macrophyte bays (Table 2.08). In general, in both June and August there were approximately 50 % more individuals caught in emergence traps in native macrophyte than wild rice bays.

Table 2.08. Total numbers of invertebrates captured using emergence traps within native macrophyte and wild rice bays in Barry, Cacholotte, and Naosap lakes in June and August of 2003. FFG = Functional Feeding Groups, Pred = predators, Omn = omnivores, Col/Gath = collectors/gatherers, Scrap = scrapers, Para = parasites/parasitoids, Shre = shredders, Filt = filter-feeders.

Groups	# of individuals	FFG	Total # of individuals per taxonomic group in each zone			
			NM June	WR June	NM August	WR August
Chironomidae (herb)	1119	Col/Gath	446	355	196	122
Caenidae	125	Col/Gath	82	39	3	1
Chironomidae (pred)	107	Pred	69	33	2	3
Limnephilidae	12	Col/Gath	5	6	1	0
Coenagrionidae	8	Pred	4	3	0	1
Hydroptilidae	6	Col/Gath	3	0	0	3
Braconidae	5	Para	0	3	0	2
Ceratopogonidae	4	Pred	4	0	0	0
Cyclorrhaphous-Brachycera	3	Col/Gath	0	1	0	2
Baetidae	2	Col/Gath	0	2	0	0
Ephydridae	2	Col/Gath	0	0	0	2
Leptophlebidae	1	Col/Gath	1	0	0	0
Culicidae	1	Filt	0	1	0	0
Simulidae	1	Filt	1	0	0	0
Phyganeidae	1	Omni	0	1	0	0
Tipulidae	1	Pred	1	0	0	0
Total # of Individuals	1398		616	444	202	136

Table 2.09. Total numbers of invertebrates for each functional feeding group captured using emergence traps in native macrophyte and wild rice bays in Barry, Cacholotte and Naosap lakes in June and August of 2003.

Groups	# of individuals	Total # of individuals per taxonomic group in each zone			
		NM June	WR June	NM August	WR August
Collectors & Gatherers	1270	537	403	200	130
Predators	120	78	36	2	4
Parasites & Parasitoids	5	0	3	0	2
Filter-feeders	2	1	1	0	0
Omnivores	1	0	1	0	0
Scrapers	0	0	0	0	0
Shredders	0	0	0	0	0
Total # of Individuals	1398	616	444	202	136

Bucket Sampler

The invertebrate taxa and functional feeding groups collected with the bucket volume sampler were similar to the taxa collected with both the bottle and emergence traps. Chironomids, Amphipoda, Oligochaeta, Caenidae, Sphaeridae, Tanypodinae, Nematoda, Hydracarina, Planorbidae, Hydrobiidae, Ostracoda, Ceratopogonidae, Leptoceridae, Coenagrionidae, Valvatidae, and Polycentripodidae were the most abundant taxa collected with this device within all lakes and constituted over 95% of the total abundance in August (Table 2.10). However, there were no significant differences for any taxa between the wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap lakes.

Table 2.10. Mean abundance (± standard deviation) of dominant invertebrate taxa captured (95% of sample) with bucket volume trap samples in native macrophyte (NM) and wild rice (WR) bays in Barry, Cacholotte and Naosap lakes, August, 2003.

	Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
Chironomidae (herbivores)	2301 ± 1872	1298 ± 211	1328 ± 517	1266 ± 683	1457 ± 154	776 ± 355
Amphipoda	1552 ± 882	1070 ± 1186	1217 ± 353	255 ± 56	474 ± 730	321 ± 294
Oligochatea	319 ± 88	532 ± 365	1217 ± 353	745 ± 381	771 ± 264	2785 ± 1601
Caenidae	239 ± 266	230 ± 180	334 ± 110	79 ± 81	685 ± 126	1128 ± 680
Sphaeridae	68 ± 76	651 ± 357	661 ± 390	322 ± 286	290 ± 164	229 ± 162
Tanypodinae	186 ± 58	374 ± 283	479 ± 144	396 ± 272	263 ± 201	304 ± 107
Nematoda	697 ± 962	167 ± 112	199 ± 119	72 ± 38	477 ± 148	268 ± 103
Hydracarina	126 ± 130	86 ± 35	114 ± 49	257 ± 141	445 ± 387	238 ± 51
Planorbidae	104 ± 148	94 ± 82	65 ± 57	183 ± 95	106 ± 113	231 ± 140
Hydrobiidae	0	172 ± 165	0	163 ± 50	0	52 ± 65
Ostracoda	50 ± 12	34 ± 33	188 ± 40	138 ± 112	10 ± 18	43 ± 74
Ceratopogonidae	45 ± 39	120 ± 99	170 ± 40	139 ± 66	109 ± 97	22 ± 37
Leptoceridae	55 ± 95	0	14 ± 24	13 ± 22	207 ± 191	21 ± 36
Coenagrionidae	32 ± 30	50 ± 86	17 ± 15	77 ± 103	41 ± 10	66 ± 35
Valvatidae	12 ± 22	0	7 ± 13	102 ± 146	17 ± 29	54 ± 20
Polycentripodidae	0	45 ± 50	17 ± 15	57 ± 54	0	165 ± 157

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$, no ★ = no significant difference at $p < 0.05$. Results obtained from a *t-test*.

Total Invertebrate Abundance and Diversity

Bottle Traps

Total invertebrate abundance was compared between the wild rice and native macrophyte bays within each lake using *t*-tests. For the bottle trap samples, significantly higher numbers of invertebrates were captured in the wild rice bays on Barry Lake in June ($p = 0.001$) and on Naosap Lake in both June ($p = 0.0008$) and August ($p = 0.0063$) of 2003 (Table 2.11). There were no significant differences in total invertebrate abundance between wild rice and native macrophyte bays for Cacholotte Lake in both June ($p = 0.99$) and August ($p = 0.29$), and for Barry Lake in August ($p = 0.73$) (Table 2.11).

Absolute richness (s) of the taxa collected within wild rice and native macrophyte bays showed similar trends to those discussed above. Absolute richness within wild rice bays was significantly higher when compared to native macrophyte bays in June and August with the exceptions of Cacholotte Lake in June ($p = 0.49$) and August ($p = 0.56$), which had no differences between bays, and Barry Lake in August, which had significantly higher numbers within the native macrophyte bay ($p = 0.0019$) (Table 2.11).

Absolute richness of FFGs in the bottle trap samples were not significantly different between wild rice and native macrophyte bays for any of the lakes sampled, with one exception. Naosap Lake in June of 2003 had significantly higher numbers of FFGs within the wild rice bay than the native macrophyte bay ($p = 0.0002$) (Table 2.12).

The final two diversity measures used to compare invertebrate communities between the wild rice and native macrophyte bays were effective richness (N2) and evenness. No significant differences were observed in the N2 or evenness of the invertebrate taxa between wild rice and native macrophyte bays for all three lakes, with two exceptions (Table 2.11). June samples from Naosap Lake had a significantly higher evenness ratio in the native macrophyte bay than the wild rice bay ($p = 0.049$). Significant differences in the effective richness of the invertebrate functional feeding groups were found on Naosap Lake in August with higher measures within the wild rice bay when compared to the native macrophyte bay with values of 3.20 ± 0.67 and 2.37 ± 0.57 ($p = 0.0086$), respectively (Table 2.12). Therefore, the functional feeding groups within the wild rice bay on Naosap Lake were essentially dominated by three equal groups while only two groups were dominant within the native macrophyte bay. Overall, few differences in the effective richness or evenness were observed between the wild rice and native macrophyte bays.

Table 2.11. Abundance and diversity of the invertebrate community in samples taken with bottle traps in native macrophyte (NM) and wild rice (WR) bays on Barry, Cacholotte and Naosap lake in June and August 2003. Abundance = total number of invertebrates collected; Absolute richness (s) = total number of taxa collected; Effective Richness (n) = reciprocal of Simpson's diversity index; Evenness = s/n, ± = standard deviation.

		Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
June	Total Invertebrate Abundance	42.80 ± 14.82	**111.60 ± 32.74** *	136.3 ± 49.01	136.8 ± 72.42	43.20 ± 33.22	**180.10 ± 88.65** *
	Absolute Richness	7.5 ± 1.78	**11.7 ± 2.98** *	11.6 ± 2.76	12.4 ± 2.32	7.9 ± 2.28	**12.5 ± 2.07** *
	Effective Richness	2.86 ± 0.80	3.23 ± 2.51	3.22 ± 1.19	3.00 ± 0.97	3.21 ± 1.11	3.57 ± 0.89
	Evenness	0.40 ± 0.14	0.26 ± 0.16	0.30 ± 0.15	0.25 ± 0.08	0.45 ± 0.22	**0.29 ± 0.06** *
August	Total Invertebrate Abundance	85.90 ± 32.62	91.90 ± 41.80	56.80 ± 15.77	49.90 ± 12.31	22.70 ± 11.93	**42.60 ± 16.08** *
	Absolute Richness	13.40 ± 2.41	**11.20 ± 1.99** *	8.60 ± 1.71	9.10 ± 2.02	6.10 ± 1.97	**9.10 ± 2.08** *
	Effective Richness	3.72 ± 0.88	3.28 ± 1.06	3.72 ± 1.52	3.10 ± 0.63	3.55 ± 0.93	4.53 ± 1.31
	Evenness	0.29 ± 0.11	0.30 ± 0.11	0.44 ± 0.18	0.35 ± 0.09	0.60 ± 0.12	0.51 ± 0.15

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$, no ★ = no significant difference at $p < 0.05$. Results obtained from a *t-test*.

Table 2.12. Diversity of the invertebrate community (as indicated by number of functional feeding groups) in samples taken with bottle traps in native macrophyte (NM) and wild rice (WR) bays on Barry, Cacholotte and Naosap lake in June and August 2003. Absolute richness (s) = total number of functional feeding groups collected; Effective Richness (n) = reciprocal of Simpson's diversity index; Evenness = s/n, ± = standard deviation.

		Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
June	Absolute Richness	4.8 ± 0.63	5.0 ± 0.82	5.10 ± 0.32	5.0 ± 0.67	3.8 ± 0.63	**5.0 ± 0.47** *
	Effective Richness	2.43 ± 0.69	2.07 ± 0.73	2.17 ± 0.66	2.18 ± 0.54	2.27 ± 0.46	2.55 ± 0.38
	Evenness	0.51 ± 0.13	0.42 ± 0.15	0.43 ± 0.14	0.44 ± 0.11	0.62 ± 0.19	0.51 ± 0.05
August	Absolute Richness	4.90 ± 0.74	4.90 ± 0.57	4.60 ± 0.70	4.20 ± 0.92	3.60 ± 0.97	4.30 ± 0.67
	Effective Richness	2.64 ± 0.46	2.62 ± 0.64	2.52 ± 0.97	2.38 ± 0.45	2.37 ± 0.57	**3.20 ± 0.67** *
	Evenness	0.56 ± 0.16	0.54 ± 0.15	0.54 ± 0.16	0.58 ± 0.13	0.69 ± 0.17	0.75 ± 0.15

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$, no ★ = no significant difference at $p < 0.05$. Results obtained from a *t-test*.

Emergence Traps

For the emergence trap samples, significantly more invertebrates were found in the native macrophyte bays compared to the wild rice bays in Naosap Lake in June ($p = 0.04$) and in Cacholotte Lake in August ($p = 0.032$). In contrast, Barry Lake had significantly higher invertebrate abundances in the traps set in June ($p = 0.0045$) in the wild rice bay. No significant differences were found for all other samples (Table 2.13).

The absolute richness (s) of the emerging insect taxa collected within wild rice and native macrophyte bays were not significantly different for any of the lakes in both June and August, 2003 (Table 2.13). Also, no significant differences were observed in the N2 or evenness of the invertebrate taxa between wild rice and native macrophyte bays for all lakes (Tables 2.13).

The absolute richness, evenness, and effective richness for the functional feeding groups collected in the emergence trap samples were not analyzed since over 80% of the invertebrates sampled were collectors/gatherers and, therefore, results were essentially the same as was observed for the taxonomic data (Table 2.13).

Table 2.13. Abundance and diversity of the invertebrate community in samples taken with emergence traps in native macrophyte (NM) and wild rice (WR) bays on Barry, Cacholotte and Naosap lake in June and August 2003. Abundance = total number of invertebrates collected; Absolute richness (s) = total number of taxa collected; Effective Richness (n) = reciprocal of Simpson's diversity index; Evenness = s/n, ± = standard deviation.

	Diversity Index	Barry NM	Barry WR	Cacholotte NM	Cacholotte WR	Naosap NM	Naosap WR
June	Total Invertebrate Abundance	7.70 ± 4.32	**15.10 ± 9.50** *	17.40 ± 9.34	11.20 ± 7.04	36.50 ± 23.30	**18.10 ± 9.76** *
	Absolute Richness	1.90 ± 0.57	2.20 ± 0.79	2.50 ± 0.85	2.40 ± 0.97	1.80 ± 0.97	1.80 ± 1.26
	Effective Richness	1.55 ± 0.39	1.75 ± 0.49	1.55 ± 0.92	1.27 ± 0.78	1.90 ± 0.80	1.80 ± 0.33
	Evenness	0.84 ± 0.16	0.82 ± 0.16	0.33 ± 0.38	0.59 ± 0.36	0.54 ± 0.20	0.53 ± 0.12
August	Abundance	8.80 ± 5.88	8.50 ± 7.50	4.10 ± 2.88	**1.50 ± 1.96** *	7.40 ± 5.72	3.60 ± 3.20
	Absolute Richness	1.30 ± 0.67	1.80 ± 0.79	1.00 ± 0.47	0.90 ± 1.20	1.10 ± 0.57	1.00 ± 0.47
	Effective Richness	0.99 ± 0.37	1.21 ± 0.54	0.92 ± 0.33	0.86 ± 1.08	0.83 ± 0.49	0.70 ± 0.50
	Evenness	0.74 ± 0.33	0.71 ± 0.34	0.66 ± 0.47	0.59 ± 0.51	0.72 ± 0.41	0.70 ± 0.48

Note: ★ = significant difference from the native macrophyte bay at $p < 0.05$, no ★ = no significant difference at $p < 0.05$. Results obtained from a *t-test*.

Bucket Volume Sampler

The final method used to compare invertebrate abundance between the wild rice and native macrophyte bays was the bucket volume sampler. Using this sampling device, no significant differences in the estimated invertebrate abundance within the bucket volume sampler were observed between the wild rice and native macrophyte bays for the three lakes sampled (Figure 2.05).

Absolute richness, effective richness, and evenness of the invertebrate community were not examined for samples taken with the bucket volume sampler because abundances of invertebrates had large variations between buckets within and across bays for each lake. Significant differences among bays were unlikely given the high variability observed among individual samples.

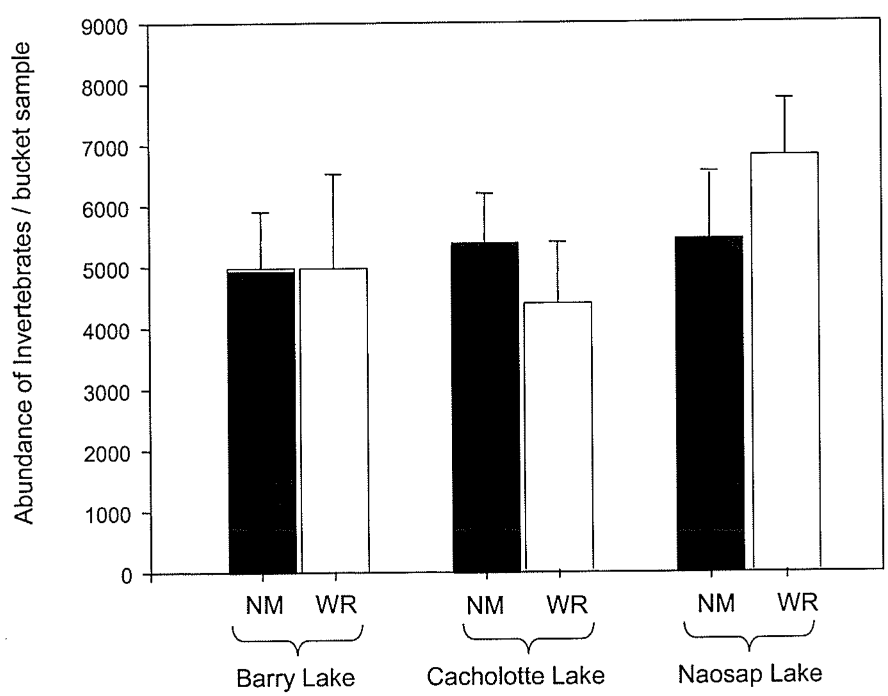


Figure 2.05. Mean (± standard deviation) abundance of invertebrates captured with bucket volume samplers for the native macrophyte and wild rice bays on Barry, Cacholotte, and Naosap lakes in August, 2003.

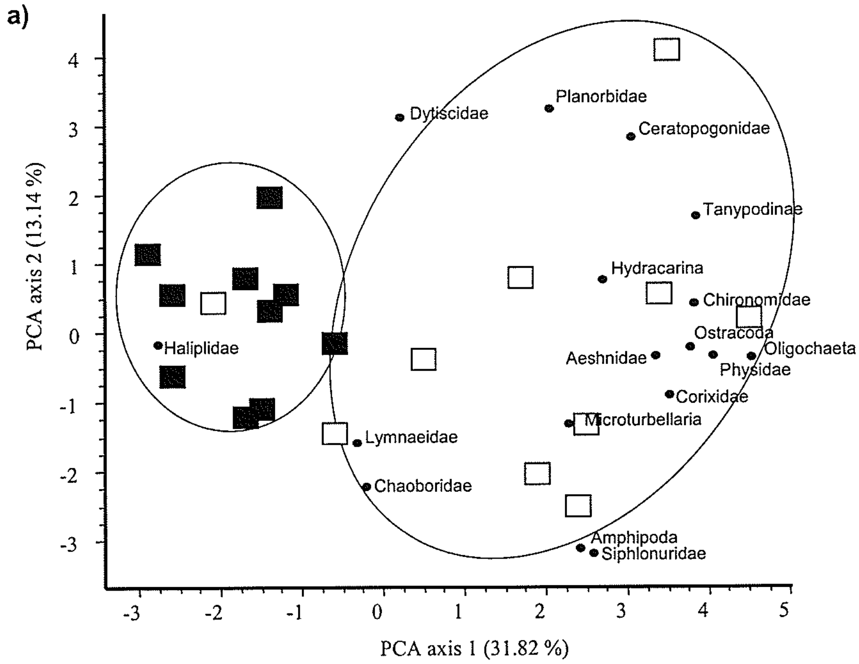
Principal Component Analysis of Invertebrate Community Composition

Principal Component Analysis (PCA) was used to analyze the composition of the community using taxa or functional feeding group data from bottle traps set in the wild rice or native macrophyte bays. Results of these analyses showed similar trends within Barry, Cacholotte and Naosap lakes for both June and August (Figures 2.06 – 2.11). The first PCA axis separated the samples (i.e. 10 native macrophyte and 10 wild rice samples) based on the abundances of the different invertebrate taxa within each sample. For example, Haliplidae were found in higher numbers in the native macrophyte samples compared to the wild rice samples and haliplids were positioned on the scattergrams with similar values (positive or negative) of the first axis as the individual native macrophyte samples (see Figure 2.06). The second PCA axis for each of the lakes also separated the invertebrates based on abundance, however, the wild rice and native macrophyte samples were not separated on this axis. All invertebrate taxa and wild rice and native macrophyte samples were therefore plotted simultaneously on the scattergram (Figures 2.06 – 2.11).

For Barry Lake in June, and for Cacholotte and Naosap lakes in June and August, each PCA scattergram showed that samples taken from wild rice and native macrophyte bays were separated based on the abundances of the different invertebrate taxa within each lake. Amphipoda, Physidae, Oligochaeta, Tanypodinae, Hydracarina, Planorbidae, Hydrobiidae and Microturbellaria were found in higher numbers and had similar values on the first axis as the ten wild rice samples suggesting that these taxa were more abundant in wild rice habitats (Figures 2.06a – 2.08a). In contrast, Haliplidae, Caenidae, and Gyrinidae had similar values on the first axis as the native macrophyte samples implying these taxa were more abundant in native macrophyte habitats. In August, the ten wild rice and ten native macrophyte samples did not separate out on the first axis on

Barry Lake, implying that there were no differences in the abundances of the individual invertebrate taxa captured (Figure 2.06b). In contrast, Haliplidae and Oligochaeta were situated with the native macrophyte samples while Amphipoda, Hydracarina, and Ostracoda were situated with the wild rice samples on the first axis in Cacholotte and Naosap lakes (Figures 2.07 - 2.08).

PCA of the functional feeding groups showed similar trends to those described previously for the taxonomic data with collectors/gatherers, predators, scrapers, and parasites/parasitoids grouping with the wild rice samples, and the shredders grouping with the native macrophyte samples on the first axis for all lakes in June (Figures 2.09 – 2.11). During this period, omnivores grouped with the native macrophyte samples in Cacholotte Lake and with the wild rice samples in Barry and Naosap lakes. In August, the native macrophyte and wild rice samples were separated on the first axis in Cacholotte and Naosap lakes but not in Barry Lake. In Naosap Lake the same trends were observed as in June; however, in Cacholotte Lake collector/gatherers, scrapers, parasites/parasitoids, and shredders grouped with the native macrophyte samples on the positive side of the first axis. Omnivores and predators were found on the negative side of axis one with the wild rice samples (Figure 2.10). These results corresponded with results obtained using paired *t*-tests for the individual functional feeding groups (see Figure 2.16).



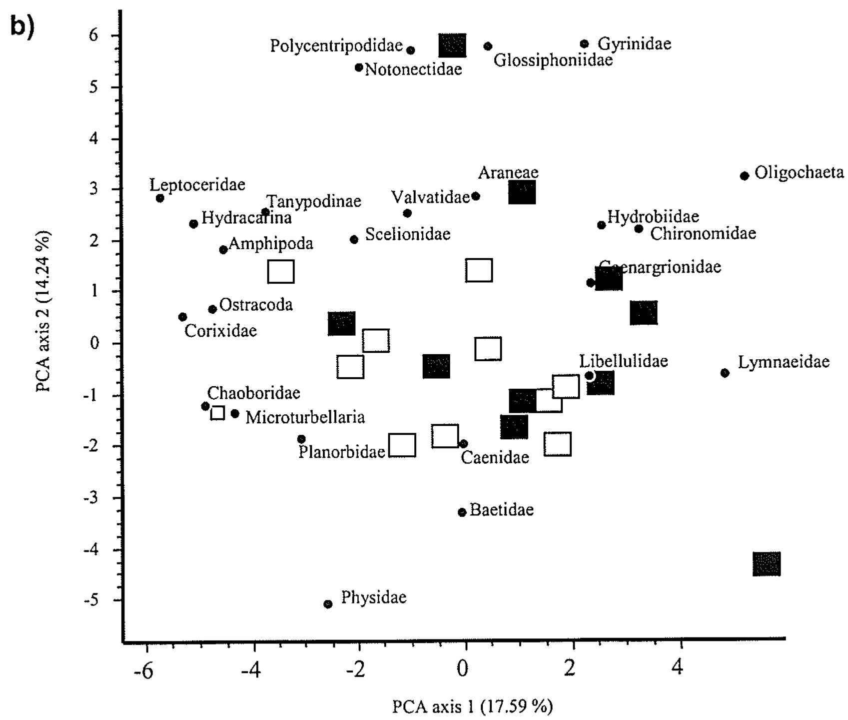


Figure 2.06. Principal Component Analysis (PCA) of the abundance of invertebrate taxa collected with bottle trap samples on Barry Lake during a) June and b) August, 2003. Data were log transformed [log (n+1)]. Symbols: dots represent the invertebrate taxa; black squares = 10 samples collected in the native macrophyte bay; white squares = 10 samples collected in the wild rice bay. Large elliptical circles represent visual groupings of the native macrophyte and wild rice samples.

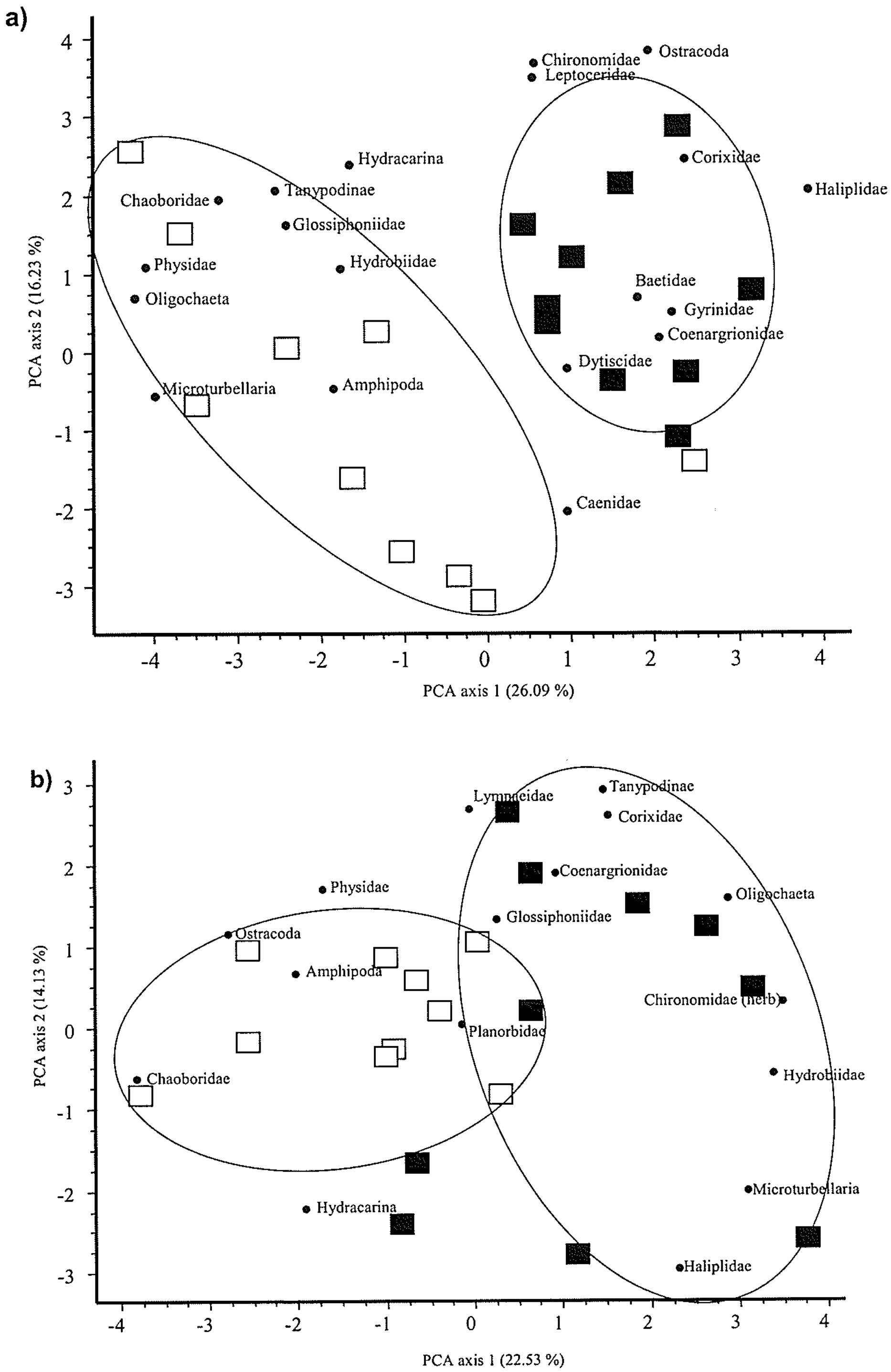


Figure 2.07. Principal Component Analysis (PCA) of the abundance data of invertebrate taxa collected with bottle trap samples on Cacholotte Lake during a) June and b) August, 2003. Data were log transformed [log (n+1)]. Symbols: dots represent invertebrate taxa; black squares = 10 samples collected in the native macrophyte bay; white squares = 10 samples collected in the wild rice bay. Large elliptical circles represent visual groupings of the native macrophyte and wild rice samples.

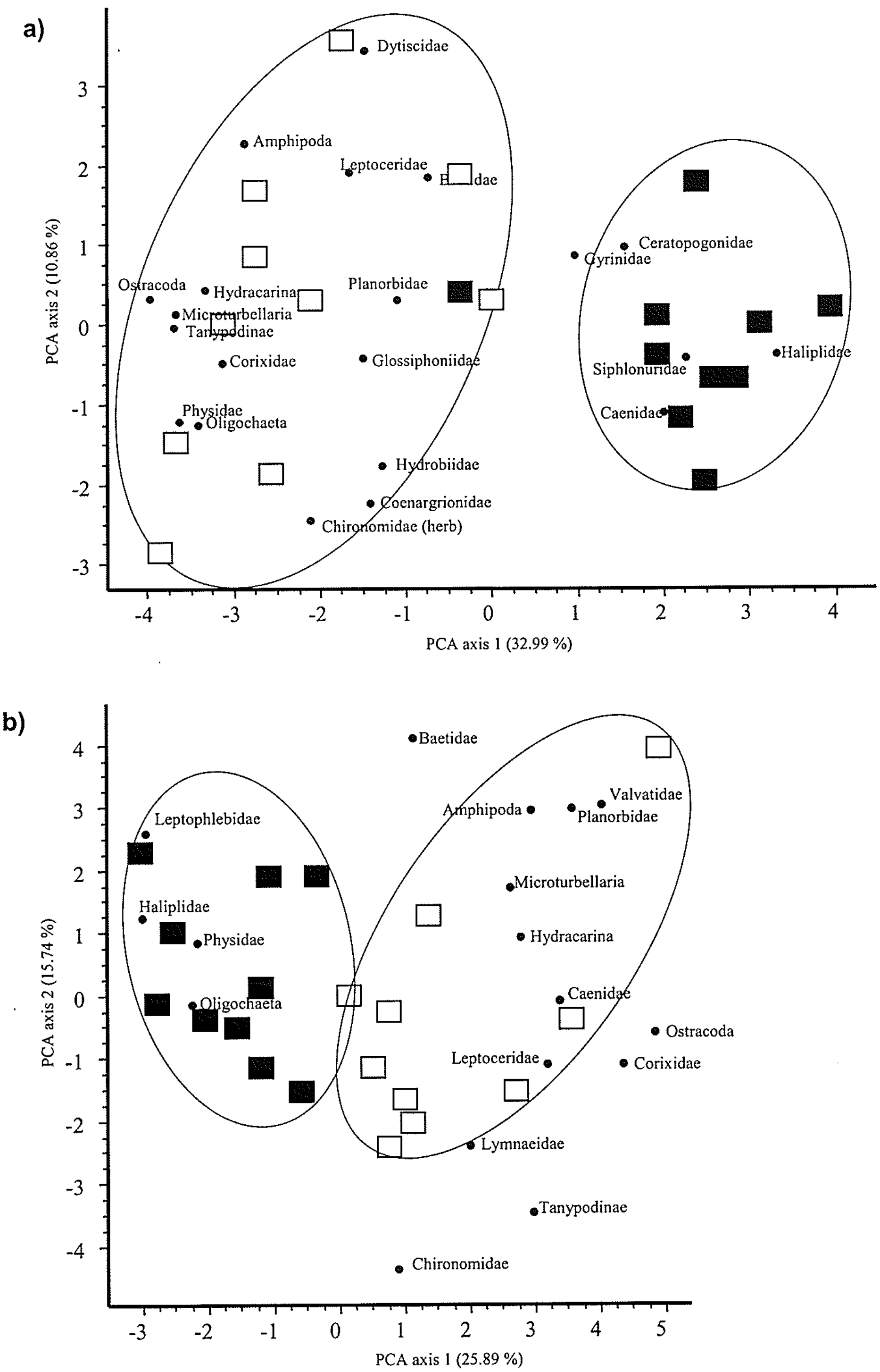
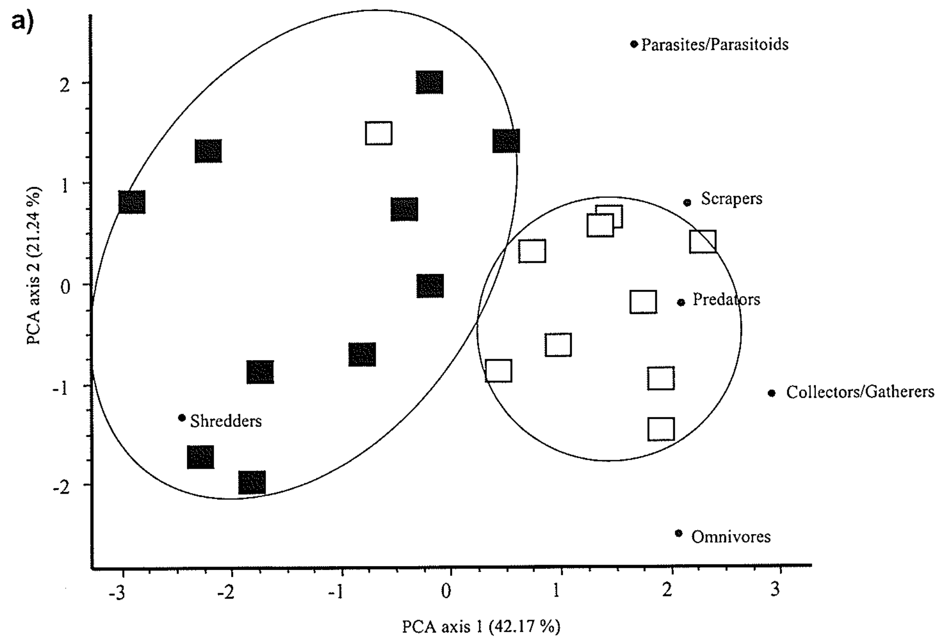


Figure 2.08. Principal Component Analysis (PCA) of the abundance data of invertebrate taxa collected with bottle trap samples on Naosap Lake during a) June and b) August, 2003. Data were log transformed [log (n+1)]. Symbols: dots represent invertebrate taxa; black squares = 10 samples collected in the native macrophyte bay; white squares = 10 samples collected in the wild rice bay. Large elliptical circles represent visual groupings of the native macrophyte and wild rice samples.



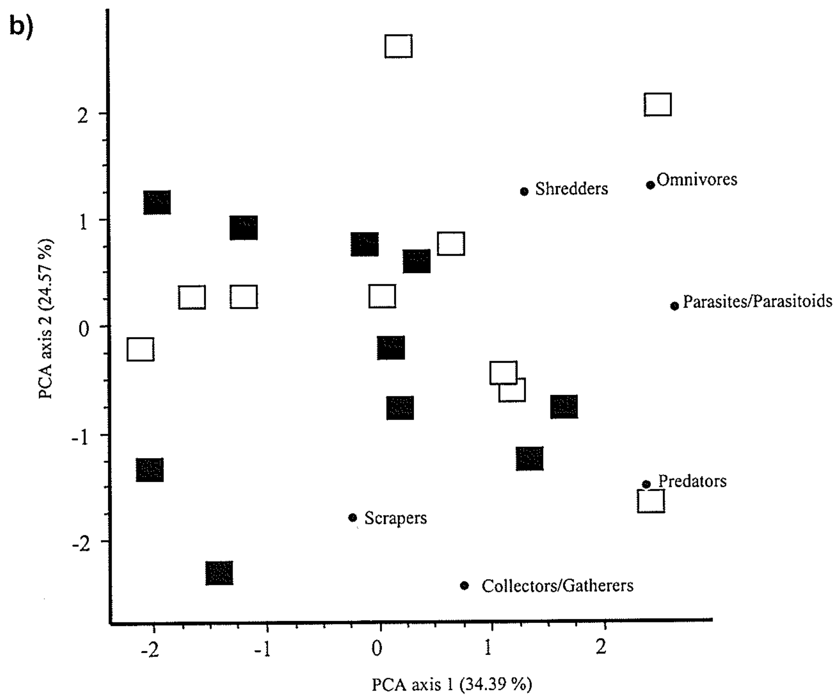


Figure 2.09. Principal Component Analysis (PCA) of the abundance data for invertebrate functional feeding groups collected with bottle trap samples on Barry Lake during a) June and b) August, 2003. Data were log transformed [log (n+1)]. Symbols: dots represent invertebrate FFGs; black squares = 10 samples collected in the native macrophyte bay; white squares = 10 samples collected in the wild rice bay. Large elliptical circles represent visual groupings of the native macrophyte and wild rice samples.

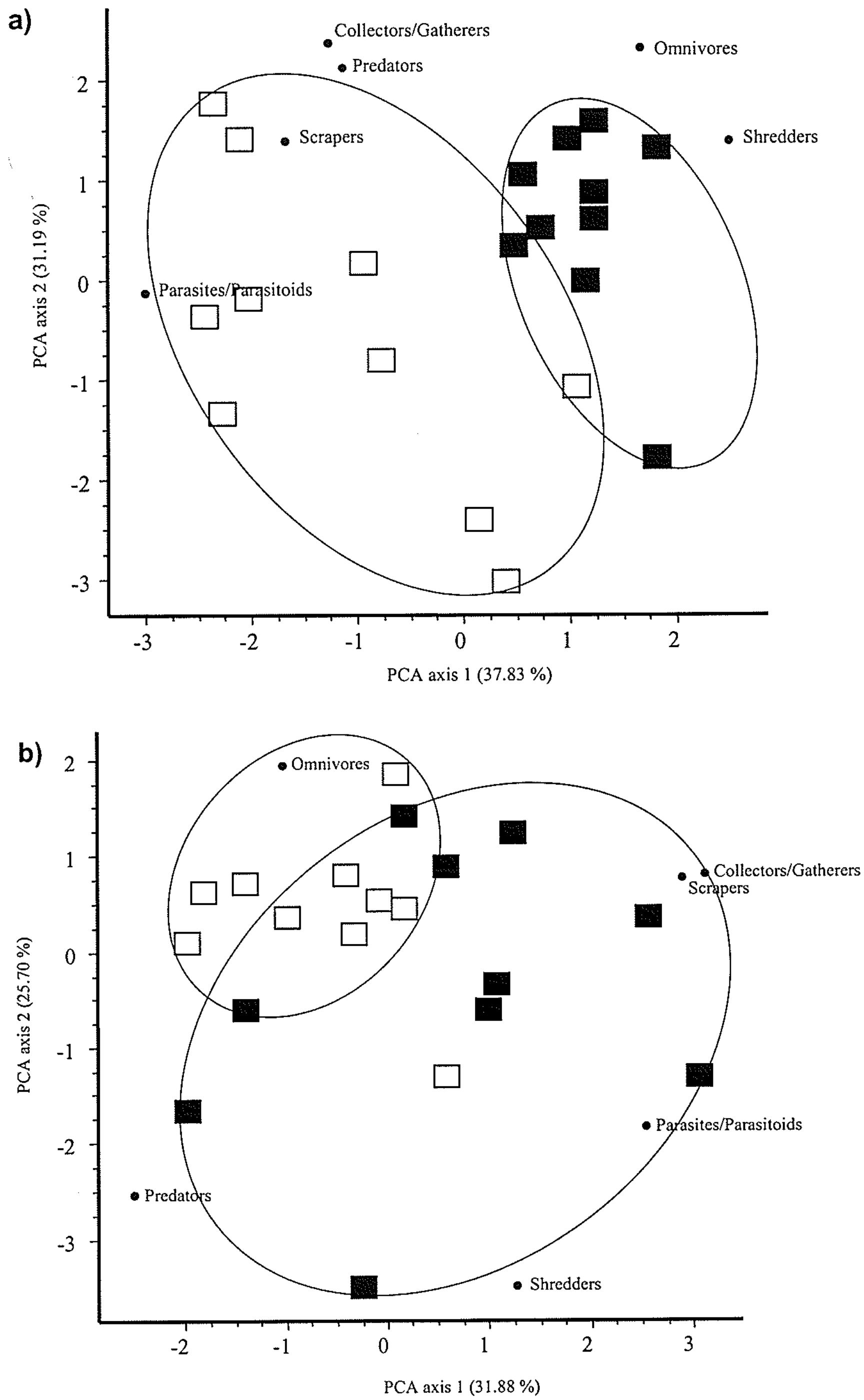
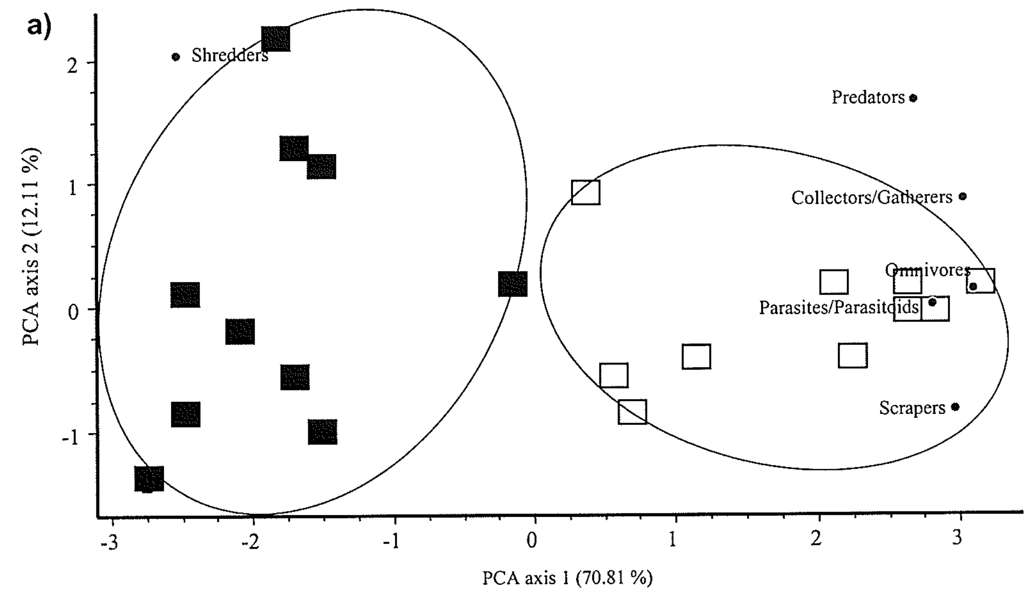


Figure 2.10. Principal Component Analysis (PCA) of the abundance data for invertebrate functional feeding groups collected with bottle trap samples on Cacholotte Lake during a) June and b) August. 2003. Data were log transformed [log (n+1)]. Symbols: dots represent invertebrate FFGs; black squares = 10 samples collected in the native macrophyte bay; white squares = 10 samples collected in the wild rice bay. Large elliptical circles represent visual groupings of the native macrophyte and wild rice samples.



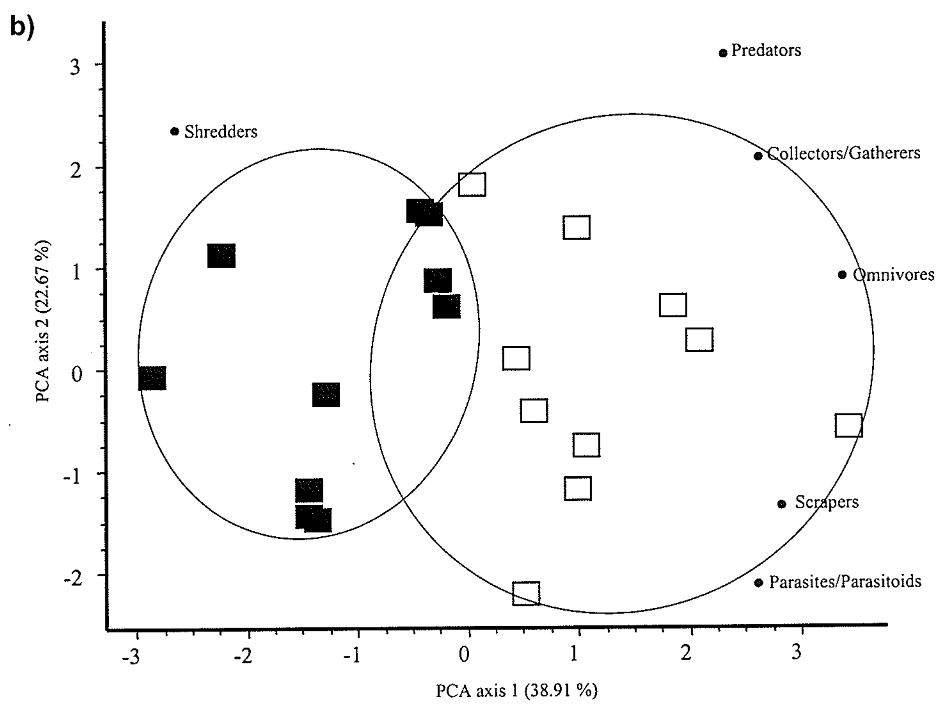


Figure 2.11. Principal Component Analysis (PCA) of the abundance data for invertebrate functional feeding groups collected with bottle trap samples on Naosap Lake during a) June and b) August, 2003. Data were log transformed [log (n+1)]. Symbols: dots represent invertebrate FFGs; black squares = 10 samples collected in the native macrophyte bay; white squares = 10 samples collected in the wild rice bay. Large elliptical circles represent visual groupings of the native macrophyte and wild rice samples.

Multiple Discriminant Analysis of Invertebrate Community Composition

Multiple Discriminant Analysis (MDA) was used to compare statistically the PCA results and to examine whether invertebrate taxa and functional feeding data of the predetermined groups (wild rice and native macrophytes) were significantly different in the bottle trap samples (Figures 2.12 - 2.13). In June, invertebrate community composition was significantly different between wild rice and native macrophyte bays in all three lakes. In August, similar trends were found in Cacholotte and Naosap lakes when compared to the results for the June samples; however, the invertebrate community composition was not significantly different between wild rice and native macrophyte bays on Barry Lake (Figure 2.12).

Functional feeding group community composition was also analyzed using MDA and significant differences were also observed between wild rice and native macrophyte bays in June for all three lakes. In August, both Barry and Cacholotte lakes did not have significantly different functional feeding groups between wild rice and native macrophyte bays. In contrast, Naosap Lake had significantly different communities in August when comparing the wild rice and native macrophyte bays (Figure 2.13). The MDA results generally indicated that that the invertebrate community composition differed between wild rice and native macrophyte bays, especially during the floating leaf stage for the wild rice.

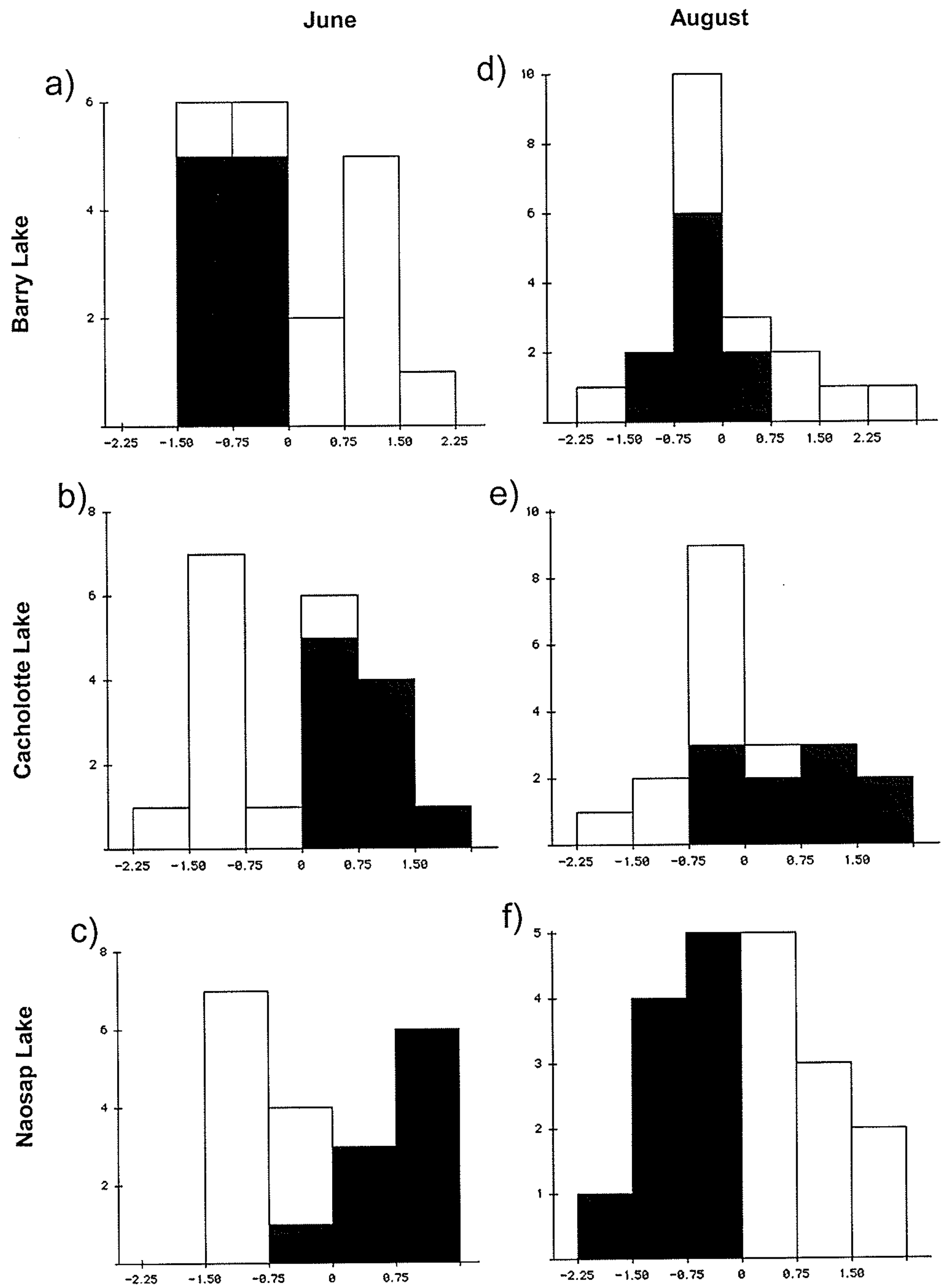


Figure 2.12 Frequencies from the MDA (based on PCA component scores) of invertebrate taxa composition for the bottle trap samples collected in native macrophyte and wild rice bays in Barry, Cacholotte, and Naosap lakes in June and August, 2003. Black areas represent the 10 samples collected in native macrophyte bays; white areas represent the 10 samples collected in wild rice bays. *P* values were < 0.05 for plots a, b, c, e and f, and > 0.05 for plot d.

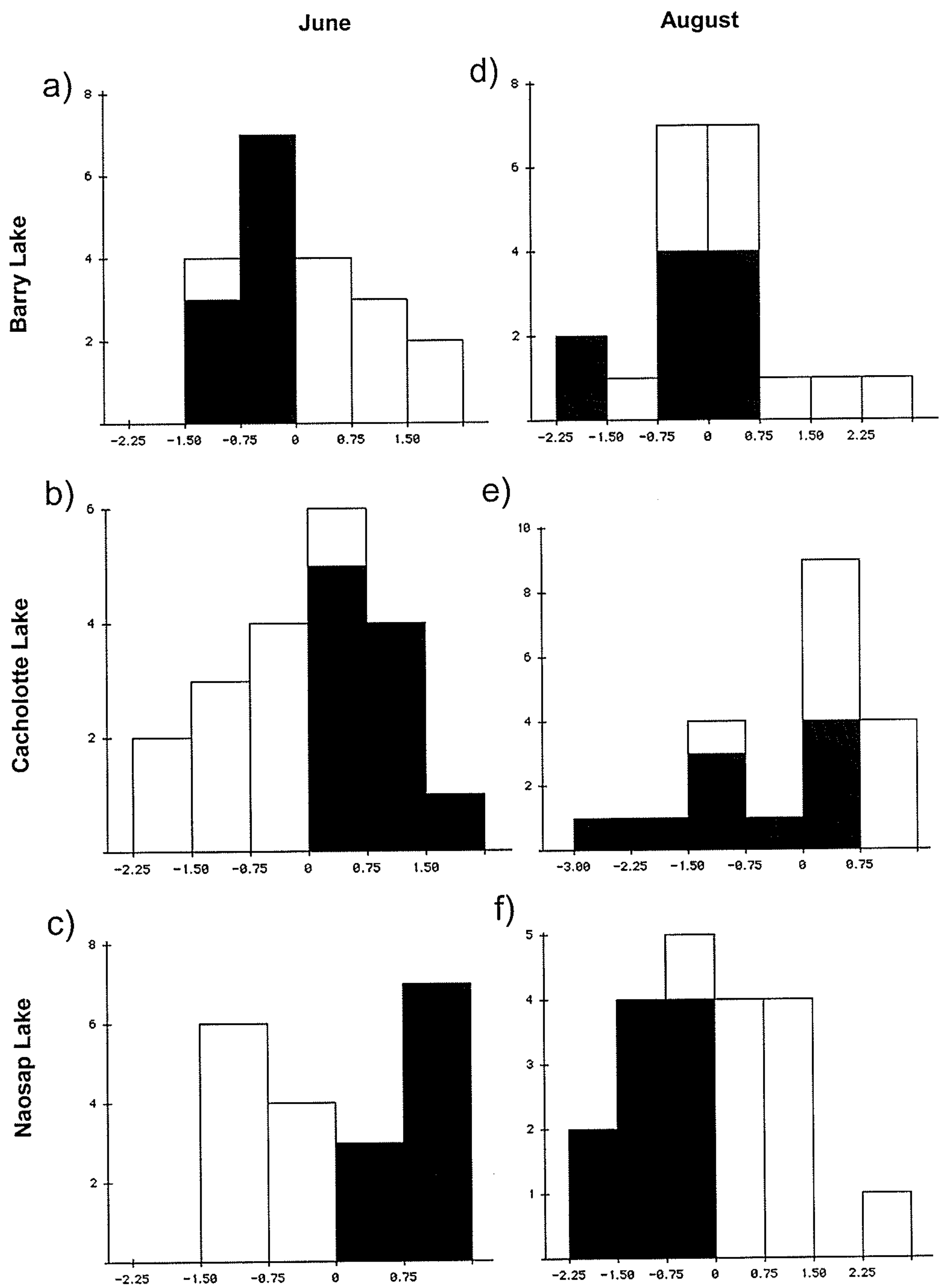
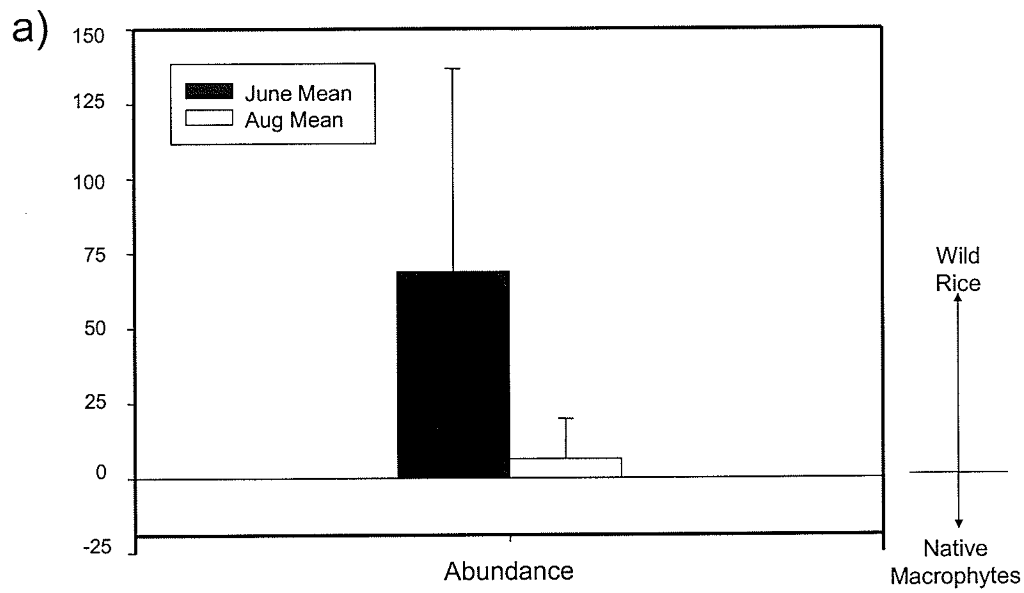


Figure 2.13 Frequencies from the MDA (based on PCA component scores) of functional feeding group composition for the bottle trap samples collected in native macrophyte and wild rice bays in Barry, Cacholotte, and Naosap lakes in June and August, 2003. Black areas represent the 10 samples collected in native macrophyte bays; white areas represent the 10 samples collected in wild rice bays. *P* values were < 0.05 for plots a, c and f and > 0.05 for b, d and e.

Statistical Comparisons of Invertebrate Diversity and Abundance Among Lakes

Paired *t*-tests were used to compare invertebrate abundance and diversity between wild rice and native macrophyte bays for all three lakes. For the bottle trap samples, there were no significant differences in invertebrate abundance between the wild rice and native macrophyte bays during either June or August (Figure 2.14a). In addition, there were no significant differences between the absolute richness, effective richness or evenness of the invertebrate taxa within wild rice and native macrophyte bays for the bottle trap samples (Figure 2.14b). For emergence trap samples, there were significantly more insects emerging from native macrophyte bays compared to the wild rice bays in June ($p = 0.03$; Figure 2.15a). In August, there were more emerging insects within the native macrophyte bays but these differences were not statistically significant ($p = 0.17$). No significant differences in invertebrate functional feeding groups were found for any of the parameters mentioned above for both the emergence trap and bottle trap samples.

Paired *t*-tests were also conducted on the mean abundances of the individual functional feeding groups collected in the bottle traps (Figure 2.16). Shredders were significantly more abundant in native macrophyte bays compared to wild rice bays in June ($p = 0.029$). The abundances of predators, collector/gatherers, omnivores, scrapers and parasites/parasitoids were not significantly different ($p > 0.05$) between wild rice or native macrophyte bays in both June and August (Figure 2.16). Paired *t*-tests were also conducted on the most dominant individual taxa. There were no significant differences ($P > 0.05$) in the mean abundances for Hydracarina, Ostracoda, Oligochaeta, Amphipoda, Chironomidae, Corixidae, Hydrobiidae, Microturbellaria, Haliplidae, Tanypodinae and Physidae between the native macrophyte and wild rice bays (Figure 2.17).



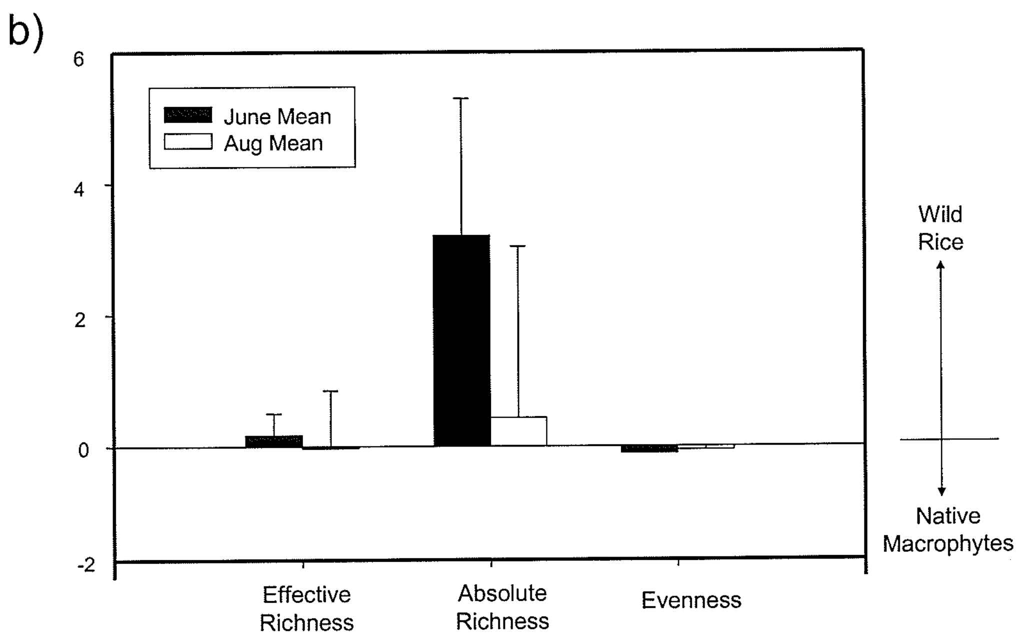


Figure 2.14. Means (± standard deviation) of the differences in the mean a) abundance of invertebrates, and b) effective richness, absolute richness, and evenness between native macrophyte and wild rice bays for invertebrate taxa collected with bottle traps on Barry, Cacholotte, and Naosap lakes in June and August, 2003. Positive values favour wild rice and negative values favour native macrophyte bays. The symbol (★) located above the bars indicates a $P <$ 0.05 after a paired t-test analysis.

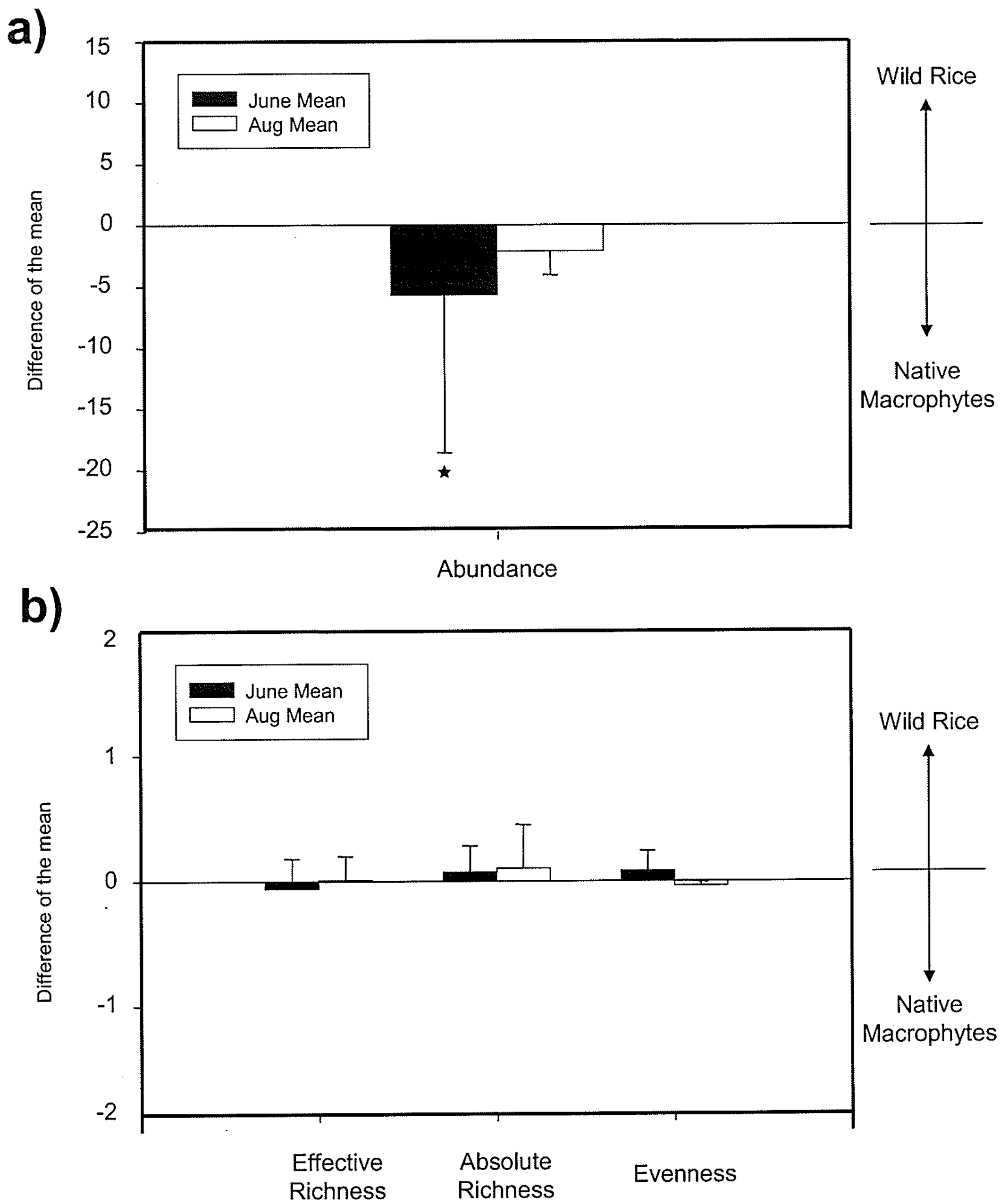


Figure 2.15. Mean (± standard deviation) of the differences in the mean a) abundance of invertebrates and b) effective richness, absolute richness, and evenness between native macrophyte and wild rice bays for the invertebrate families collected using emergence traps on Barry, Cacholotte, and Naosap lakes in June and August, 2003. Positive values favour wild rice and negative values favour native macrophyte bays. The symbol (★) located above the bars indicates a $P < 0.05$ after conducting a paired t-test analysis.

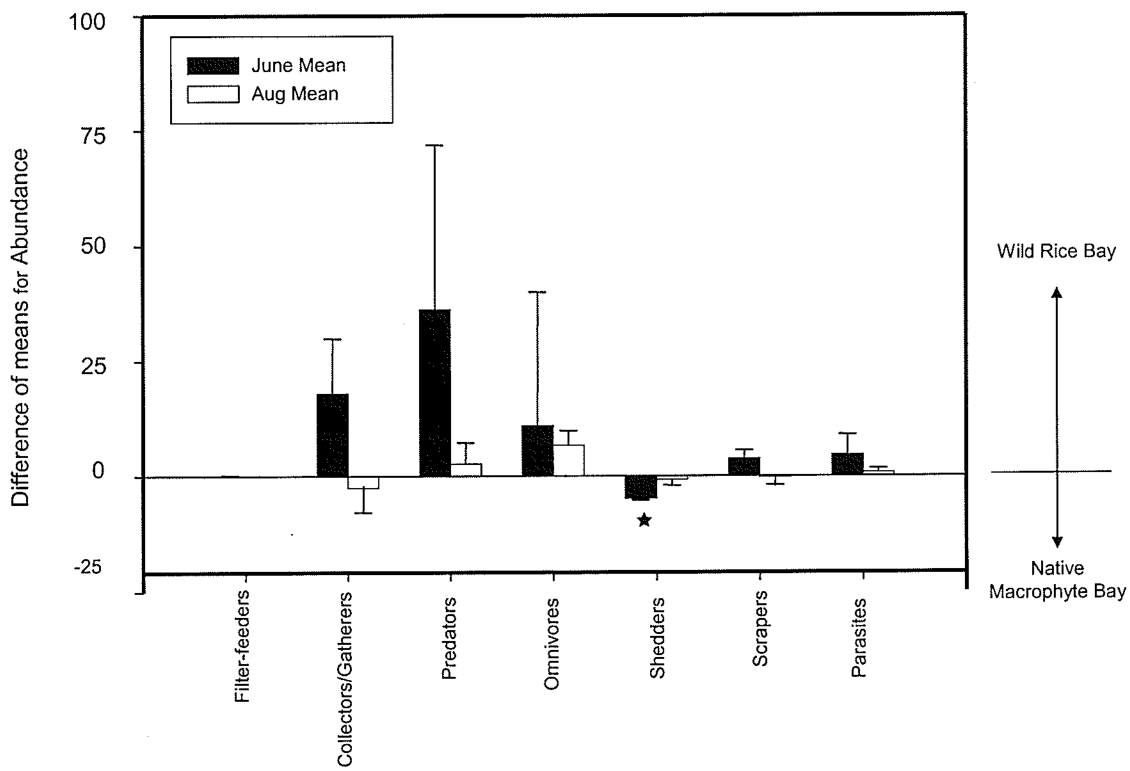


Figure 2.16. Mean (± standard deviation) of the differences in the mean abundances of invertebrate functional feeding groups between wild rice and native macrophyte bays collected with bottle traps on Barry, Cacholotte and Naosap lakes, west-central Manitoba in June and August, 2003. Positive values favour wild rice and negative values favour native macrophytes. Symbol (★) situated above the bar represent *P* values of < 0.05 using a paired *t*-test.

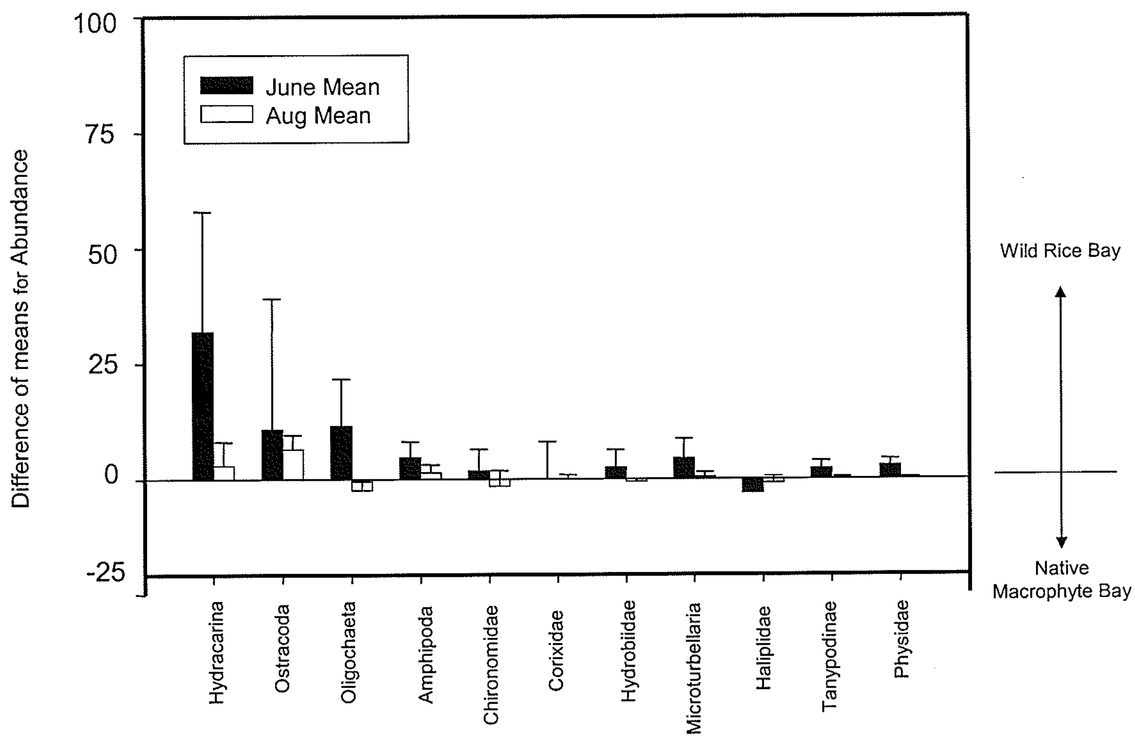


Figure 2.17. Mean (± standard deviation) of the differences in the mean abundances of invertebrate most dominant (95%) taxa between wild rice and native macrophyte bays collected with bottle traps on Barry, Cacholotte and Naosap lakes, west-central Manitoba in June and August, 2003. Positive values favour wild rice and negative values favour native macrophytes. Symbol (★) situated above the bar represent *P* values of < 0.05 using a paired *t*-test.

Discussion

Environmental Variables

In the current study, dissolved oxygen concentrations were almost always lower at all depths in the water column within wild rice bays compared to native macrophyte bays during both sampling periods; the one exception was Cacholotte Lake in June where dissolved oxygen concentrations were slightly lower at all depths in the native macrophyte bay. However, oxygen concentrations at only about half of the sites examined in the wild rice bays were significantly lower than the native macrophyte bays. In addition, Lavergne (2005) found consistently lower under-ice dissolved oxygen concentrations in wild rice bays compared to native macrophyte bays in all three of these lakes. Dissolved oxygen values were as low as 0.1, 0.2 and 0.3 ppm under the ice in wild rice bays on Barry, Cacholotte and Naosap lakes, respectively. In the native macrophyte bays under the ice, dissolved oxygen concentrations were as low as 2.7, 0.3, and 3.7 ppm. Moyer (2000) reported dissolved oxygen concentrations of 0 in 10 of 19 sites at a water depth of one meter under-ice within the wild rice bays on Kisseynew Lake, west-central Manitoba. During the period of my study, dissolved oxygen concentrations did not fall below 3.2 ppm in any of the wild rice bays, indicating that oxygen depletion was more severe during the winter months. Although it is unclear how low the oxygen concentrations must be before growth, metabolism and reproduction of organisms are affected, dissolved oxygen levels near zero will likely have negative effects and may even cause mortality during summer or winter in both the fish and invertebrate communities (Watson et al., 2001). In addition, low dissolved oxygen levels within wild rice bays may discourage fish from using these habitats.

After each wild rice growing season, large straw mats form and settle on the bottom and on the shoreline. These straw mats are likely the main cause of the lower dissolved oxygen concentrations observed within the wild rice bays in this study in both June and August, and under the ice during the winter months. As these straw mats begin to decompose, microbial activity increases within the water column which decreases the amount of dissolved oxygen available to other biota (Sain, 1983 & 1984; Archibold, 1989; Derksen, 1993; Natural Resources Institute, 1995).

Water quality was examined on Barry, Cacholotte, and Naosap lakes and few differences were observed between the wild rice and native macrophyte bays. In June, ammonia, dissolved organic carbon, and total dissolved nitrogen concentrations were higher in wild rice bays while suspended nitrogen and suspended carbon were slightly lower in wild rice bays. In August, when the wild rice was in the emergent stage, dissolved inorganic carbon and dissolved organic carbon were slightly higher in wild rice bays. All other water quality variables were similar between wild rice and native macrophyte bays. Although the number of samples collected was limited, these results suggest that wild rice does not impact the water quality within the littoral zone.

Water temperatures at the surface, mid-depth, and bottom were variable within and across wild rice and native macrophyte bays. Wild rice stands averaged 73, 40, and 60 stems per square meter within Barry, Cacholotte, and Naosap lakes, respectively, (Lavergne, 2005) and were probably not dense enough to impede water circulation within these bays. In contrast, Watson et al. (2001) found that the water column was stratified in the wild rice bay but not in the open water area, suggesting that wild rice did impede water circulation in parts of Kisseynew Lake. However, wild rice densities were not recorded in the Watson et al. study, making comparisons among studies difficult. In addition, Watson et al. did not compare wild rice bays to native macrophyte bays; only one bay with wild rice was compared to an open water area. Further studies are needed to elucidate the relationship between densities of wild rice and the stratification of the littoral water column.

Invertebrate Community Composition and Abundance

In this study, I found that invertebrate community composition differed between the wild rice and native macrophyte bays. Amphipoda, Ostracoda, Hydracarina, Microturbellaria, Tanypodinae and Physidae were significantly more abundant within wild rice habitats compared to native macrophyte bays in at least one of the lakes examined; this difference was more apparent in the June than in the August samples. In contrast, Haliplidae and Chironomidae were found in higher numbers in native macrophyte bays compared to wild rice habitats. These changes were most likely due to the differences in the plant communities or perhaps to the differences in dissolved oxygen concentrations between bays, rather than to temperature or water quality because there were no substantial differences in these latter parameters.

The invertebrates collected in higher numbers within the wild rice bays, such as amphipods, likely preferred the wild rice habitat because of the increased food supply and/or protection it offers. Wild rice straw accumulates after each growing season and for three consecutive years (Archibold, 1990; Tattum, 1998; Derksen, 2002), and may provide an important food source for amphipods, which are known to feed on debris and detritus within the water column (Clifford, 1991). Alternatively, wild rice may provide more protection from predators because amphipods are a preferred food for benthivorous fish (Peckarsky et al., 1988; Holomuzki & Hoyle, 1990). It is interesting to note that amphipod abundance increased in the native macrophyte bays later in the season and decreased in wild rice bays from June to August. A possible explanation for this trend is that native macrophytes, which develop over the season and are most abundant later in the year (Keast, 1984), provide more habitat and protection from predation for these organisms later in the open-water season. The differences in amphipod abundance among bays and among sampling times may also be related to periphyton growth. It is possible that the quality and quantity of periphyton changes seasonally in wild rice and native macrophyte bays and is more abundant in one bay when compared to another.

Snails from the families Physidae and Hydrobiidae were also found in higher numbers within the wild rice bays in the June samples. These snails feed on epiphytic algae on the surface of macrophytes (Brown, 2001). Unlike native macrophytes, wild rice grows very rapidly in the spring (Dore, 1969) and this may provide more surface area for epiphytic algae growth, and a larger source of food for Physidae and Hydrobiidae early in the open-water season as compared to the native macrophyte bays. In addition, wild rice straw from previous years may provide additional substrate for epiphytic algae growth. Higher epiphytic algae production in wild rice bays warrants further consideration in future studies of the effects of wild rice on littoral communities.

Tanypodinae, Hydracarina, and Microturbellaria were also found in higher numbers in June within wild rice bays in the three lakes I studied. Tanypodinae are predacious chironomids and are a valuable food source for fish and larger invertebrate predators (Gilinsky, 1984). Hydracarina and Microturbellaria are invertebrate predators or fish parasites (Clifford, 1991). As mentioned previously, the among-bay differences for these taxa may also be due to spatial differences in food supply or predation pressure.

The final invertebrate group that was captured in higher numbers within wild rice bays was the omnivorous Ostracoda. They feed on detritus and debris within the water column and occasionally on small invertebrates, including soft tissues of certain snails (Delorme, 2001). It is possible that the higher abundance in wild rice bays was caused by an increased food supply or decreased predation. Previous studies have found ostracods in the stomach contents of brook trout, three-spine stickleback, suckers, lake trout, and yellow perch (Delorme, 2001). However, Lavergne (2005) found significantly higher numbers of white sucker and shiners within the wild rice bays in June which does not support the hypothesis of fewer predators in this habitat. In August, the opposite trend occurred in Lavergne's study (2005) with significantly higher numbers of shiners in the native macrophyte bays. At present, it is not possible to determine why invertebrate abundances for these taxa and for the above-mentioned invertebrates were higher in the wild rice bays. More research is needed to understand whether these differences were related to food supply, decreased predation, or a combination thereof.

One invertebrate taxon that was found in higher numbers in June and August within native macrophyte bays compared to wild rice bays was Haliplidae. These water beetles are classified as shredders and feed directly on the leaves and stems of the macrophytes they inhabit (Merritt & Cummins, 1996). Therefore, it appears that these water beetles have not adapted to using wild rice as a habitat.

Chironomids are one of the most abundant invertebrate groups in freshwater habitats both in terms of the numbers of species present and the numbers of individuals. They are an extremely important component of aquatic food webs and are a food source for many invertebrates and fish (Epler, 2000). In this study, chironomids were found in greater numbers in the emergence traps set in the native macrophyte bays when compared to samples collected from the wild rice bays. However, chironomid abundances were not significantly different between the native macrophyte and wild rice bays for the bottle trap samples. It is interesting that chironomids were found in relatively high numbers within these traps because they usually are found in the benthos rather than in the water column. It is likely that chironomids climbed plant stems adjacent to the traps, eventually moving onto the traps and finding their way inside. Even though chironomids are collector/gatherers and feed on the same types of foods as the amphipods and ostracods, the latter two taxa were found in greater numbers in the wild rice bays whereas chironomids were more abundant in the native macrophyte bays for one of the sampling devices used in this study.

Why were there fewer chironomids in the emergence traps in wild rice bays compared to native macrophyte bays? It is probably due to the nature of the substrate. Most benthos prefer more stable substrates (Engel, 1990). In wild rice bays, the substrate was composed of very fine and flocculent sediment that was approximately 0.5 to 1.0 meters deep (personal observation, M. Lowdon; Watson et al., 2000). In native macrophyte bays, the sediment consisted of sand and silt, and was coarser than the sediment within wild rice bays. In addition, there was only about 0.1 to 0.3 meters of accumulation above the hard clay bottom (personal observation, M. Lowdon). The difference in the nature of the substrate within these bays may be caused by 1) decreased water circulation due to the wild rice plants, allowing fine particles to settle out (Keast, 1984), or 2) the build up of decomposing straw (Derksen, 2002). It is likely that a combination of these factors may be responsible for the difference in sediment type and macroinvertebrate abundances, between wild rice and native macrophyte bays.

Gyrinidae (order: Coleoptera) were also found at higher abundances within native macrophyte bays compared to the wild rice bays. This family feeds on a wide variety of foods but the larvae are predominantly predators (Clifford, 1991). Previous studies have shown that the floating leaves of *Potamogeton* plants provide ideal refuge for hemipterans and coleopterans from fish predation (Heino, 2000) but the physical structure of wild rice may not provide this protection. The differences in invertebrate communities between these two types of bays suggest that there are some influences of the physical nature of wild rice and native macrophytes on the structure of this part of the food web.

Invertebrate Diversity, Composition, and Abundance

In the native macrophyte bays there were consistently lower numbers of nektonic invertebrates compared to wild rice bays (see Figure 2.06). In addition, there were also fewer taxa of invertebrates observed in the native macrophyte bays in both June and August. Diversity measures, including effective richness and evenness, were also lower in native macrophyte bays compared to wild rice bays but these trends were not significantly different.

Why were nektonic invertebrates more abundant within wild rice bays? One explanation may be that wild rice, as compared to native macrophytes, provides increased protection from predacious fish (i.e., predators are the primary control; Power, 1992). Alternatively, fish may be more abundant within the native macrophyte bays and reduce invertebrate abundance due to higher predation rates. Fish communities are known to modify invertebrate communities (e.g., Fairchild, 1982); for this reason, both of these arguments are possible. Lavergne (2005) found significantly more large pike (>300g), white sucker, and shiners in wild rice bays compared to native macrophyte bays in June.

This is in direct contrast to work by Keast (1984), who found that introduced milfoil impeded the movement of fish due to the thick barrier it forms in the water column. Of the fish found at higher abundances in the wild rice bays I studied, white sucker and shiners depend upon invertebrates as a food source, which supports the hypothesis that wild rice provides a better refuge for the invertebrate communities from fish predation.

An alternative possibility for why invertebrates were more abundant within the water column of wild rice bays may be that this macrophyte provides an additional food source for the invertebrate communities, i.e., bottom-up forces (Power, 1992). The rice stands and straw may provide more surface area for epiphytic algae to colonize, thereby providing an additional food source for the invertebrates that is not available in native macrophyte bays early in the open-water season. Watson et al. (2000) examined the epiphytic and phytoplankton communities in a wild rice bay and found that wild rice leaves and stems supported an abundant epiphytic community. However, no contrast was made between this wild rice bay and a native macrophyte community. Further research needs to be done to examine the differences in epiphyton productivity between these types of bays.

The limited results from the bucket volume samplers indicated that the overall abundances of invertebrates were similar between wild rice and native macrophyte bays. Because the total abundance of invertebrates was similar within a given volume of water in both bays, and the abundances of invertebrates were generally higher in bottle trap and lower in emergence trap samples in the wild rice bays, then one could conclude that invertebrates have shifted from occupying the bottom substrates in native macrophyte bays to occupying the water column within wild rice bays. Studies on the effects of fish predation on the vertical distribution of invertebrates have shown that invertebrates have a tendency to move higher within the water column when fish predation is minimized (Diehl, 1988 & 1992; Kornijow & Moss, 1998). This suggests that introduced wild rice changes the behaviour, but not overall abundance, of invertebrate communities in littoral zones where it was seeded.

Invertebrate communities are extremely important to the entire aquatic food web (Engel, 1990). Based on the results of this study, the introduction of wild rice has led to differences in invertebrate abundance, diversity and composition in these bays when compared to native macrophyte bays. Because this study was based on only three lakes in the region, it is important to examine other systems to determine whether the trends observed herein are widespread in lakes seeded with wild rice.

Chapter 3

Stable carbon and nitrogen isotope analysis of energy flow and food web structure within wild rice and native macrophyte bays in west-central Manitoba

Abstract

With the introduction of wild rice to west-central Manitoba, there arose concerns about the effects this plant may have on the native biota, including changes in energy flow and food web dynamics. To examine this, I used carbon and nitrogen stable isotopes of primary producers through tertiary consumers to characterize trophic interrelationships and to assess whether food webs differ in bays with wild rice or bays with natural vegetation. Results showed that pathways of energy flow and food web structure were similar between wild rice and native macrophytes bays. Results indicated that wild rice (mean $\delta^{13}C$ = -27.60 ‰) may have provided an additional energy source for some invertebrates (-36.74 to -21.62 ‰) and fish (-28.14 to -18.03 ‰) as their carbon signatures overlapped. Shiners had consistently depleted $\delta^{13}C$ values within both the wild rice and native macrophyte bays implying a diet linked to zooplankton. The top predators in each bay and each lake were consistently walleye and northern pike. Naosap Lake, the largest examined, had carbon signatures more enriched than Barry and Cacholotte, suggesting that the primary producers relied more on atmospheric CO_2 for photosynthesis rather than isotopically lighter carbon released from the decomposition of plants. I also observed a slightly longer food web in wild rice bays in two of the three systems examined.

3.0 Introduction

Lakes in west-central Manitoba are coming under increasing threat in recent years as a result of increased development in the region. Commercial and recreational fishing pressure has increased due to the construction of new logging roads which have provided access to previously inaccessible lakes (Derksen, 1998). In addition, wild rice (*Zizania palustris*) is being introduced into these lakes for commercial harvest (Derksen, 1998), transforming the littoral zones from heterogeneous native macrophyte communities to communities dominated by wild rice. With these intentional introductions of wild rice, there has been a growing concern that there may be impacts on the structure, function and productivity of the natural aquatic ecosystems.

While the general biology of wild rice is fairly well documented (Dore, 1969; Archibold, 1990; Natural Resources Institute, 1995), relatively little is known about the relationship between this plant species and the aquatic community. Alterations to plant communities within lakes in west-central Manitoba may modify interactions between the invertebrate and fish communities, changing the energy flow and trophic inter-relationships within these food webs. To date, only one study (Watson et al., 2001) has examined the impacts of wild rice on aquatic communities (see Chapter 1 for details on the Watson et al. study).

The current study was designed to compare energy flow and food web structure in bays with wild rice or native macrophytes in lakes in west-central Manitoba. I used carbon and nitrogen stable isotopes to determine the primary sources of energy within these lakes and to understand the impacts introduced wild rice may have on energy flow and food web structure. Carbon and nitrogen stable isotope ratios have been used in ecological studies to trace the flow of organic matter and to distinguish different trophic levels in aquatic food webs (Fry & Sherr, 1984; Owens, 1987; Gorokhova & Hansson, 1999; Vander Zanden & Rasmussen, 2001; Post, 2002). Naturally occurring stable carbon isotope ratios ($^{13}C/^{12}C$), expressed as $\delta^{13}C$, in organisms are effective at tracing energy flow within food webs because there is little (< 1 ‰) or no fractionation of the isotopes during trophic transfer (Fry & Sherr, 1984; Peterson & Fry, 1987). Therefore, organisms reflect the average $\delta^{13}C$ of their diets (Gorokhova & Hansson, 1999). In contrast, the heavier isotopes of nitrogen are preferentially enriched by 3-4 ‰ by the consumer (DeNiro & Epstein, 1981; Minagawa & Wada, 1984). This enrichment of nitrogen provides a measure of the relative trophic level of an organism within a food web. In combination, carbon and nitrogen isotopes provide information on long-term dietary habits of organisms and a quantitative basis for determining trophic interrelationships (Campbell et al., 2003). The objective of this chapter is to determine whether the energy flow and the trophic levels of the aquatic organisms differ between bays with wild rice or native macrophytes within the littoral zones of lakes in west-central Manitoba using carbon and nitrogen stable isotopes. The null hypothesis is that the energy flow and trophic levels of the organisms would not differ between the wild rice and native macrophyte bays of the three lakes examined.

3.1 Materials and Methods

Study Sites

The lakes within the west-central boreal forest region of Manitoba are shallow oligotrophic lakes (Rowe, 1972). Barry Lake, Cacholotte Lake and Naosap Lake, each containing a bay with natural vegetation (herein called native macrophyte bay) and a bay in which wild rice was introduced, were sampled in June and August of 2003 (see Chapter 1 for details on location, and physical and chemical characteristics).

The plant communities within the littoral zones of these lakes consisted of various types of submerged macrophytes, including floating-leafed pondweed (*Potamogeton natans*), thread-leaved pondweed (*Stuckenia filiformis*), yellow pond lily (*Nuphar* sp.), arum-leaved arrowhead (*Sagittaria cuneata*), burreed (*Sparganium* sp.), bladderwort (*Utricularia macrorhiza*), and northern water milfoil (*Myriophyllum sibiricum*). Within the native macrophyte bays on Barry and Cacholotte Lakes, the plant communities were dominated by floating-leafed pondweed and yellow pond-lily. Thread-leaved pondweed and linear-leafed pondweed (herein identified as #2) were the most abundant plants found within the native macrophyte bay on Naosap Lake. A linear-leafed pondweed identified as *Potamogeton* sp. #4, arum-leaved arrowhead, burreed, bladderwort, and northern water milfoil were also present within all the native macrophyte bays. The dominant plant in the wild rice bays was wild rice. Floating leafed pondweed was the next most abundant macrophyte within all the wild rice bays (see Chapter 2).

Walleye (*Sander vitreus*), northern pike (*Esox lucius*), yellow perch (*Perca flavescens*), and white sucker (*Catostomus commersoni*) were the dominant fish species. The invertebrate taxa found in these three lakes are described in Chapter 2. The diversity and absolute richness of the invertebrate (see Results, Chapter 2) and fish (see Results, Lavergne, 2005) communities within the native macrophyte or wild rice bays were similar for Barry, Cacholotte, and Naosap Lakes.

Sampling Collection

Fish, invertebrates, macrophytes, plankton, and algae were collected as representatives of different trophic levels and as potential carbon sources within the native macrophyte and introduced wild rice bays for each of the three study lakes (see Figures 1.1, 1.2, and 1.3 for locations). All samples were collected June 10-29, 2003 (during the floating leaf stage of wild rice) and August 13-September 1, 2003 (during the emergent stage of wild rice).

Fish collected in August were processed for stable carbon and nitrogen isotopes analysis. They were processed during only one sampling period (August) because longer-lived species integrate their dietary habits over months to years and are much less variable in their stable isotope signatures over time than short-lived organisms (Kidd et al., 1999).

Gill nets ranging in mesh size from ½ inch to 5 inches in stretched size, minnow traps and hoop nets were used to capture fish within each bay (Lavergne, 2005). Northern pike, walleye, yellow perch, white sucker, spottail shiner (*Notropis hudsonius*), emerald shiner (*Notropis atherinoides*), and, on occasion, blacknose shiner (*Notropis heterolepis*) were captured. Muscle samples of ~ 25 grams were removed from above the lateral line and below the dorsal fin of the larger fish, whereas small fish were kept whole. Fish samples were bagged individually and frozen within 8 hours of capture. Invertebrates and littoral zooplankton were collected in three sites within the littoral zone of each of the wild rice and native macrophyte bays using a D-frame sweep net (250-μm mesh). The invertebrates collected were sieved using a 0.5 mm sieve and sorted in the field into major taxonomic groups. All of the invertebrates within a given taxon for each site of the bay were pooled and placed in 4 oz. Whirl-Paks and frozen approximately 8 hours after live capture. A zooplankton net (250 μm) was towed behind a boat approximately two meters below the water surface in the open water near the mouth of each bay to collect triplicate samples of pelagic zooplankton. Epiphytic algae were scraped off plants within three areas of the wild rice and native macrophyte bays. Submerged and emergent macrophytes were also collected from three areas within the wild rice and native macrophyte bays. For the emergent macrophytes, including wild rice, plant material above the water surface was cut off and not included in the sample. These samples were also frozen approximately 8 hours after collection.

All samples were frozen for stable isotope analysis because ethanol and formalin preservation alters carbon and nitrogen isotope signatures (Bosley & Wainright, 1999; Kaehler & Pakhomov, 2001). Frozen invertebrates were thawed briefly in distilled water and identified to class for Oligochaeta, Nematoda, Amphipoda, Microturbellaria, to order for Arachnidae and to family for the Insecta, Mollusca, and Hirudinea using Merritt and Cummins (1996) and Clifford (1991).

Stable Isotope Analysis

After identification, trichopterans and gastropods were removed from cases or shells and pooled. Once sorted and identified, the invertebrates were pooled by site within each bay, placed in glass vials, dried at 60^0C in an oven for 72 hours, and ground to a fine powder using a glass rod (Figure 3.01). Skinless fish muscle was dissected from the small fish and dried and ground using a mortar and pestle. Skinless fish muscle from the large fish, zooplankton and macrophytes were also oven-dried and ground into a fine powder. One mg of dried sample was then weighed into tin capsules (Figure 3.01).

Isotope analyses for all organisms were performed at the Environmental Isotope Laboratory, Department of Earth Sciences, University of Waterloo, Waterloo, Ontario using a VG Isogas stable isotope ratio mass spectrometer with a precision of ± 0.2 ‰ for both carbon and nitrogen. All results were standardized against N_2 in air or CO_2 in PeeDee Limestone (PDB). The primary standard, PDB, has a δ^{13}C value of 0 ‰ and its $^{13}C/^{12}C$ ratio equals 0.0112372. Organisms are usually depleted in δ^{13}C relative to PDB and have δ^{13}C negative values (Fry & Sherr, 1984). The stable isotope ratio was calculated as follows:

$$\delta^{15}N \text{ or } \delta^{13}C \text{ (‰)} = [(R\ sample/R\ standard) - 1] \times 1000$$

where R is $^{15}N/^{14}N$ or $^{13}C/^{12}C$. Isotope values in samples are expressed as parts per thousand or per mil (‰) difference from that of the standard.

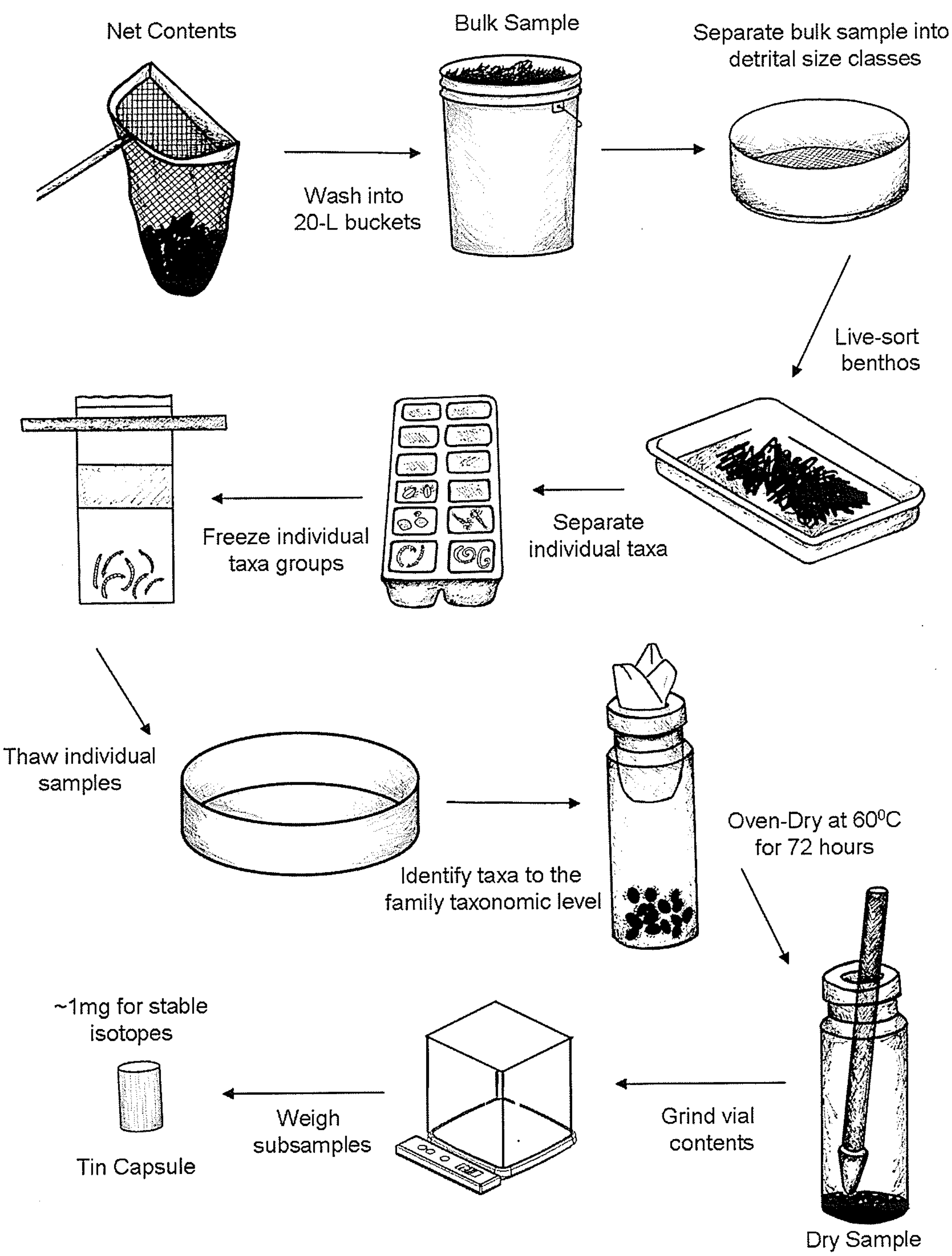


Figure 3.01. Methods used for capturing, identifying and processing invertebrates for carbon and nitrogen stable isotope analysis (modified from Podemski, unpublished).

Individuals from separate invertebrate taxa were grouped into one of four functional feeding groups (FFG) used to characterize invertebrates based on the morphological and behavioural mechanisms for exploiting food (Merritt & Cummins, 1996). These FFG include 1) scrapers; animals adapted for grazing algae, periphyton, and epiphytic algae from mineral and organic substrates; 2) collectors/gatherers, feeding primarily on detritus and debris within the water column, on occasional live organisms, and on fine particulate organic matter (FPOM; <1mm diameter); 3) filter-feeders (zooplankton), organisms that filter fine particulate matter from the water column using either setae, mouth bristles, fans etc. or silk to collect and funnel food to their mouth parts; 4) predators, which consume prey items by either piercing, sucking or fully engulfing; and, finally, 5) omnivores which are generalists feeding on vegetation and prey items (Wallace & Webster, 1996; Merritt & Cummins, 1996). In this study, the invertebrate taxa found within the different functional feeding groups were as follows: scrapers-Physidae and Hydrobiidae; collectors/gatherers-Amphipoda, Baetidae, Caenidae, and Chironomidae; omnivores-Phryganeidae; predators/parasites-Hydracarina, Aeshnidae, Coenagrionidae and Dytiscidae (Appendices A2.01 – A2.03).

To analyze seasonal differences in isotope composition for invertebrates, five groups of organisms, including pelagic plankton, littoral zooplankton, Physidae, amphipods, and water mites were collected during both sampling periods (June and August). These analyses were done because, unlike fish, invertebrates have fast turnover rates (Hobson et al., 1994; MacAvoy et al., 2001), and it was therefore important to assess the seasonal differences in their stable isotope composition during the open water season. To compare results between June and August within each bay, a Δ value was calculated as follows

$$\Delta = \text{mean } \delta^{13}C_{June} - \text{mean } \delta^{13}C_{August}$$

OR $$\Delta = \text{mean } \delta^{15}N_{June} - \text{mean } \delta^{15}N_{August}$$

to determine whether the $\delta^{13}C$ or $\delta^{15}N$ signatures were enriched or depleted in June versus August.

To compare results between the wild rice and native macrophyte bays within each lake a Δ value was calculated as follows

$$\Delta = \text{mean } \delta^{13}C_{\text{wild rice}} - \text{mean } \delta^{13}C_{\text{native macrophyte}}$$

OR $$\Delta = \text{mean } \delta^{15}N_{\text{wild rice}} - \text{mean } \delta^{15}N_{\text{native macrophyte}}$$

to determine whether the $\delta^{13}C$ and $\delta^{15}N$ signatures were enriched or depleted in the wild rice bays versus the native macrophyte bays for the primary producers, plankton, invertebrates and fish.

A trophic model designed by Hobson et al. (2002) was used to compare food chain length among Barry, Cacholotte and Naosap lakes. Because the $\delta^{15}N$ value of a consumer depends upon the signature at the base of the food web, it was important to standardize the $\delta^{15}N$ between food webs (Finlay, 2001). Therefore, $\delta^{15}N$ data were standardized to the scraper community in each bay and I assumed that these organisms were primary consumers (2^{nd} trophic level) (Hecky & Hesslein, 1995). Also, it must be assumed that isotopic enrichment was consistent among all organisms within the food web and across sites. For this model, an enrichment of 3.4 ‰ per trophic level was used (Hobson et al., 2002). To derive the trophic levels (TL) for all other consumers within the food webs the equation below was used:

$$TL = 2 + (\delta^{15}N_{consumer} - \delta^{15}N_{scrapers})/3.4.$$

Results

A. Seasonal variability

There were no consistent temporal trends in the $\delta^{13}C$ and $\delta^{15}N$ signatures of invertebrates within the wild rice and native macrophyte bays; for some organisms, isotope signatures were more enriched later in the season and for others invertebrates this trend was reversed. For $\delta^{15}N$, these differences between June and August were typically less than 1.39 ‰. For $\delta^{13}C$, these differences were typically less than 2.63 ‰ between June and August with the exceptions of Amphipoda in the wild rice bay on Naosap, Physidae in the native macrophyte bay on Cacholotte and in the wild rice bay on Naosap, and water mites in the wild rice bay on Cacholotte (Δ for these exceptions ranged from 3.16 to 4.73 ‰) (Figure 3.02). For the remainder of the Results section, these samples were pooled across dates to examine food web interrelationships.

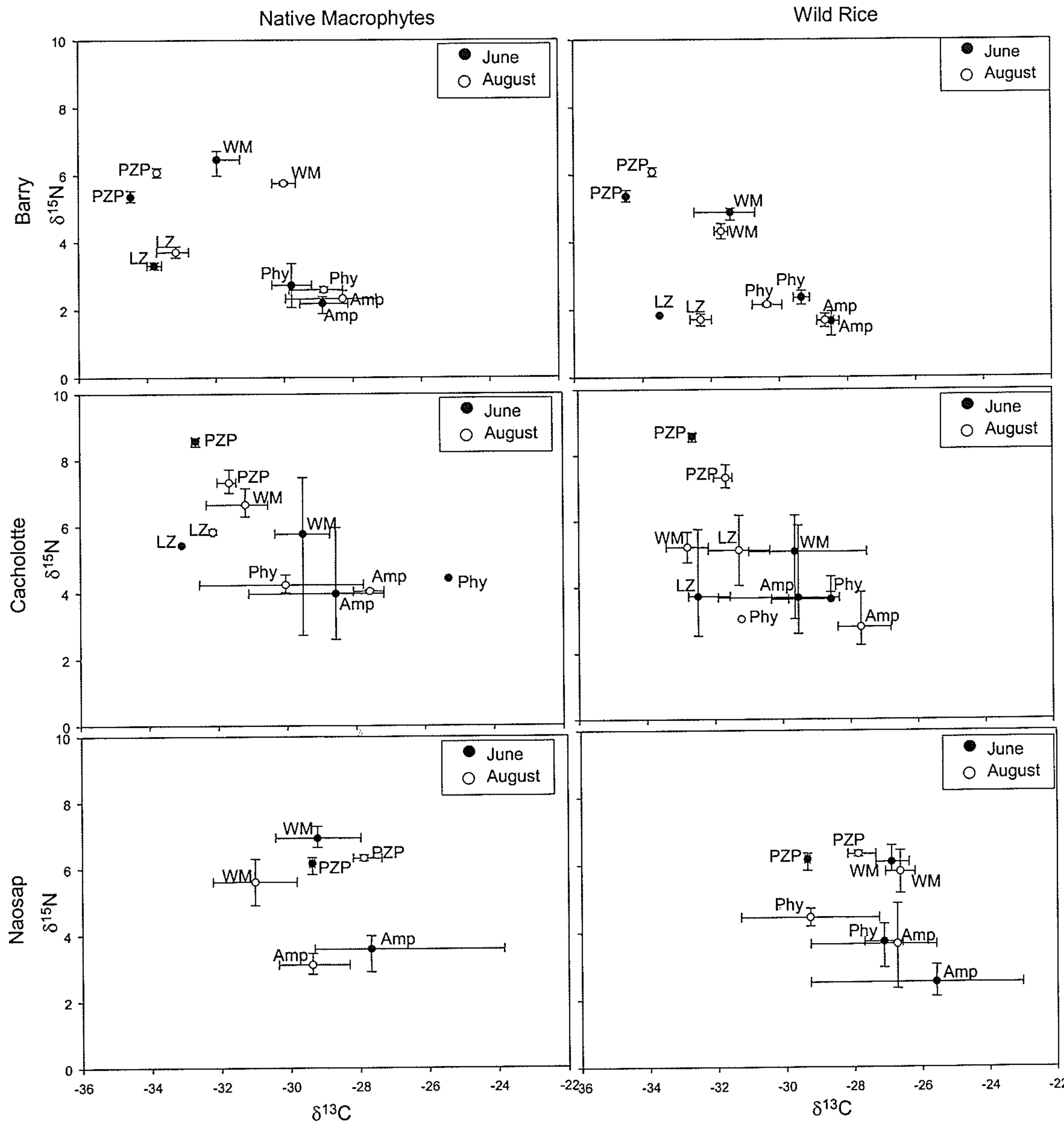


Figure 3.02. Bi-plots of mean $\delta^{13}C$ and $\delta^{15}N$ (‰) for biota collected in both June and August of 2003 for Barry, Cacholotte, and Naosap lakes. Bars represent ranges. (PZP = pelagic zooplankton; LZ = littoral zooplankton; Phy = Physidae; Amp = Amphipoda; WM = water mites).

B. $\delta^{13}C$ of Biota Collected from Native Macrophyte and Wild Rice Bays

Primary Producers

Wild rice in all lakes had $\delta^{13}C$ signatures that ranged from -29.46 to -26.65 ‰ for the floating leaf stage and from -31.08 to -27.17 ‰ for the emergent stage (Figure 3.03). These values were typically more depleted when compared to the $\delta^{13}C$ for native macrophytes within bays with introduced wild rice (-30.09 to -11.42 ‰) or bays still dominated by native macrophytes (-30.09 to -12.33 ‰) (Figure 3.03). Considerable overlap in $\delta^{13}C$ signatures was observed for native macrophytes within wild rice and native macrophyte bays in all lakes (Figure 3.03). However, native macrophyte $\delta^{13}C$ values within both wild rice and native macrophyte bays on Naosap Lake were more enriched when compared to those found on Barry and Cacholotte lakes (Figure 3.03).

The $\delta^{13}C$ values of epiphytic algae collected in Naosap Lake were depleted in the wild rice bay compared to the native macrophyte bay (Δ of -10.44 ‰) with mean values of -24.13 and -13.69 ‰, respectively (Figure 3.03). The $\delta^{13}C$ signature for epiphytic algae from the native macrophyte bay on Cacholotte Lake (-27.61 ‰) was even more depleted than that of Naosap Lake, implying that there are large among-lake variations in the epiphytic algae $\delta^{13}C$ signatures. Epiphytic algae were not collected in large enough quantities for stable isotope analysis from all sites on Barry Lake and from the wild rice bay on Cacholotte Lake.

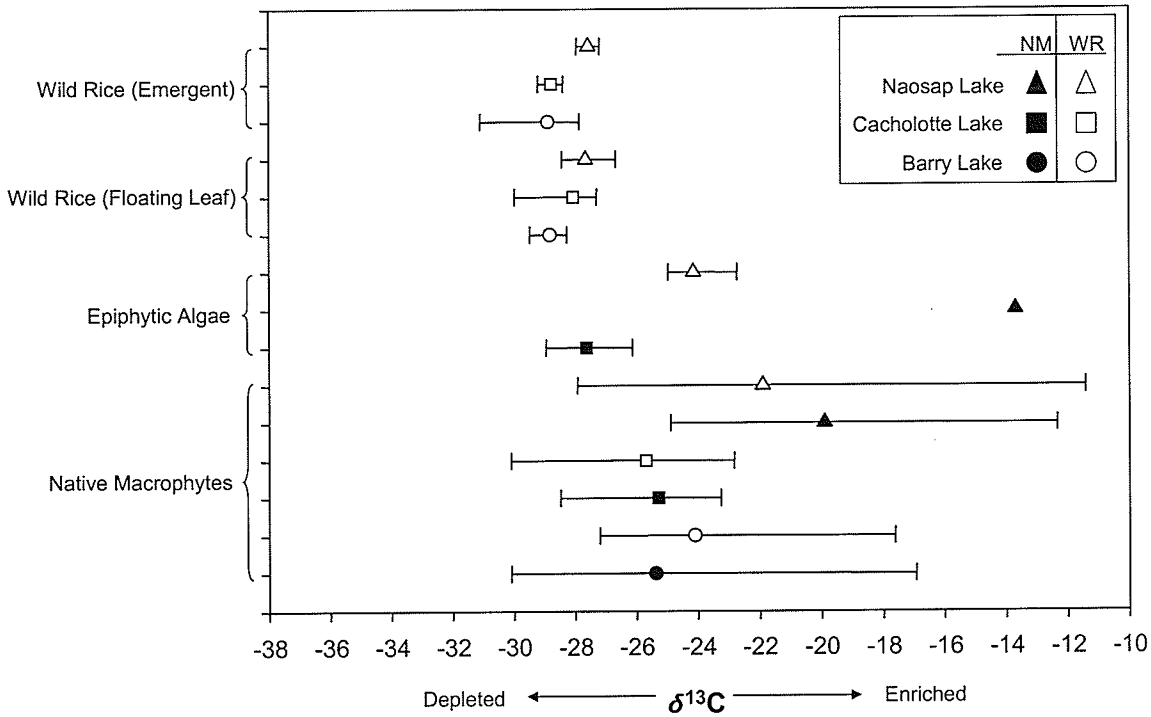


Figure 3.03. Mean $\delta^{13}C$ (‰) for primary producers from wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap lakes. Bars represent the ranges. NM = native macrophytes, WR = wild rice.

Pelagic and Littoral Zooplankton

Pelagic zooplankton were collected in the central basin of each lake to characterize open-water $\delta^{13}C$ signatures. Mean $\delta^{13}C$ of pelagic zooplankton was -34.08 (-34.49 to – 33.63 ‰), -32.16 (-32.91 to -30.37 ‰), and -28.63 ‰ (-29.45 to -27.37 ‰) in Barry, Cacholotte and Naosap lakes, respectively. Stable carbon isotope signatures of littoral zooplankton were similar among lakes and bays and ranged from -37.02 to -28.21 ‰ for the six sites (Figure 3.04). The $\delta^{13}C$ values of pelagic and littoral zooplankton overlapped within each lake studied (e.g. $\delta^{13}C$ of pelagic and littoral zooplankton in Naosap Lake were -29.45 to -27.37 ‰ and -32.62 to -28.21 ‰, respectively), and mean $\delta^{13}C$ for the littoral zooplankton were enriched within all the wild rice bays for each lake by 0.29 to 3.88 ‰ when compared to the native macrophyte bays. In addition, $\delta^{13}C$ of pelagic zooplankton on Naosap Lake were more enriched compared to the other two lakes, as was observed for the $\delta^{13}C$ signatures of the native macrophytes on Naosap Lake.

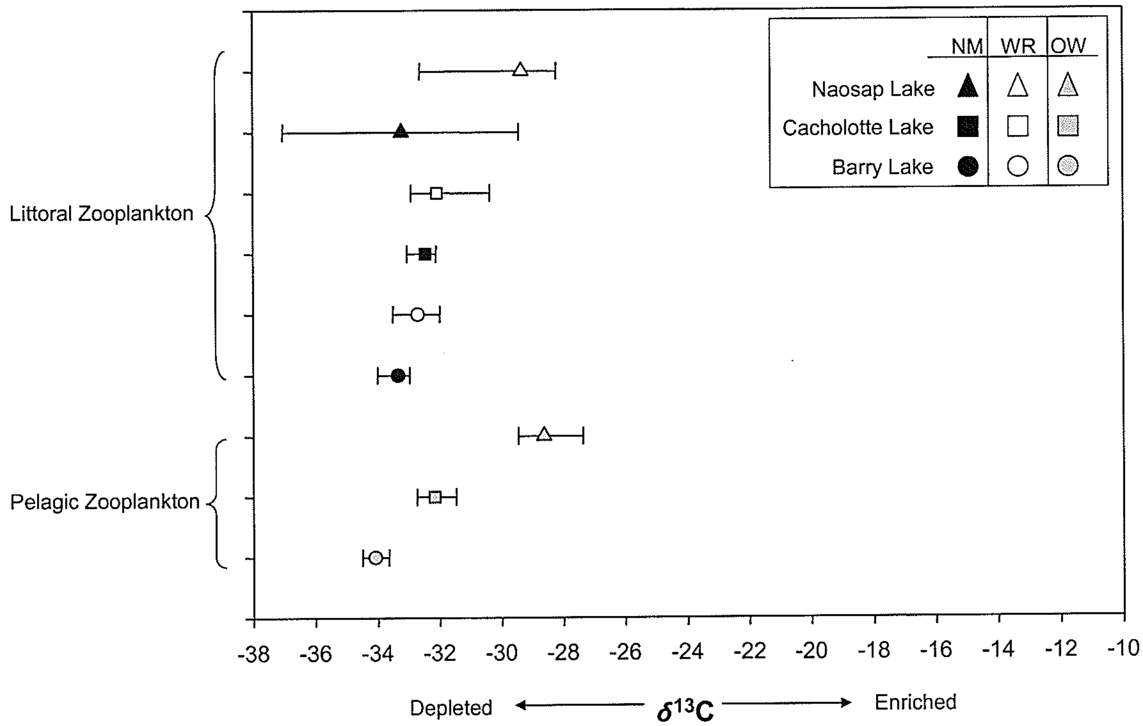


Figure 3.04. Mean $\delta^{13}C$ (‰) for littoral and pelagic zooplankton from wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap lakes. Bars represent the ranges. NM = native macrophytes, WR = wild rice, OW = open water.

Invertebrates (Functional Feeding Groups)

With one exception, the range of $\delta^{13}C$ values for each functional feeding group (scrapers, collectors/gatherers, omnivores, and predators/parasites) collected from the wild rice and native macrophyte bays overlapped; at all sites, $\delta^{13}C$ values for these functional feeding groups ranged from -36.74 to -21.62 ‰ (Figure 3.05). There was no overlap in the $\delta^{13}C$ of omnivores from Barry Lake; this group was enriched in $\delta^{13}C$ by 1.87 ‰ within the wild rice bay compared to the native macrophyte bay.

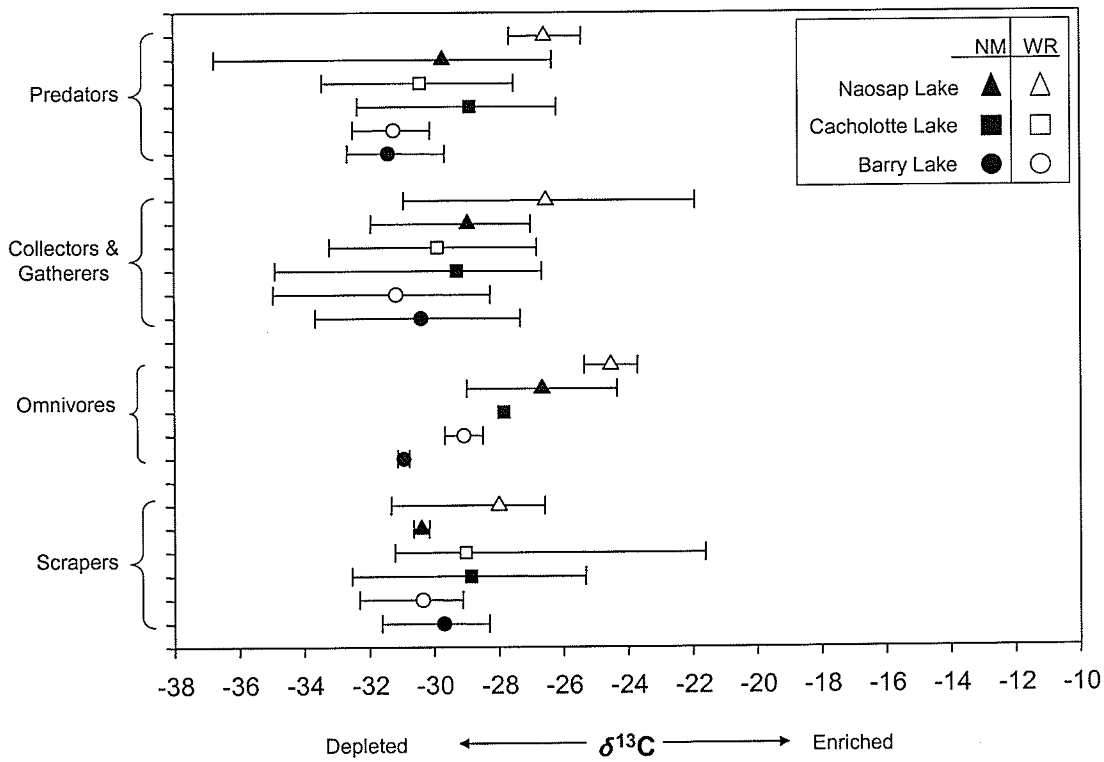


Figure 3.05. Mean $\delta^{13}C$ (‰) for invertebrate functional feeding groups from wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap lakes. Bars represent the ranges. NM = native macrophytes, WR = wild rice

Fish

With two exceptions, fish (shiners, perch, white sucker, northern pike, and walleye) had similar $\delta^{13}C$ values when comparing the wild rice and native macrophyte bays of Barry, Cacholotte, and Naosap lakes. Shiners from Naosap were more depleted within the wild rice bay compared to the native macrophyte bay (Δ = -3.73 ‰). In contrast, perch > 50 grams from Naosap Lake were more enriched in the wild rice bay compared to the native macrophyte bay (Δ = 3.34 ‰). In addition, all fish from Naosap Lake ($\delta^{13}C$ ranged from -28.14 to -18.03 ‰) were generally more enriched than those collected from Barry or Cacholotte lakes ($\delta^{13}C$ ranged from -32.85 to -25.19 ‰) (Figure 3.06).

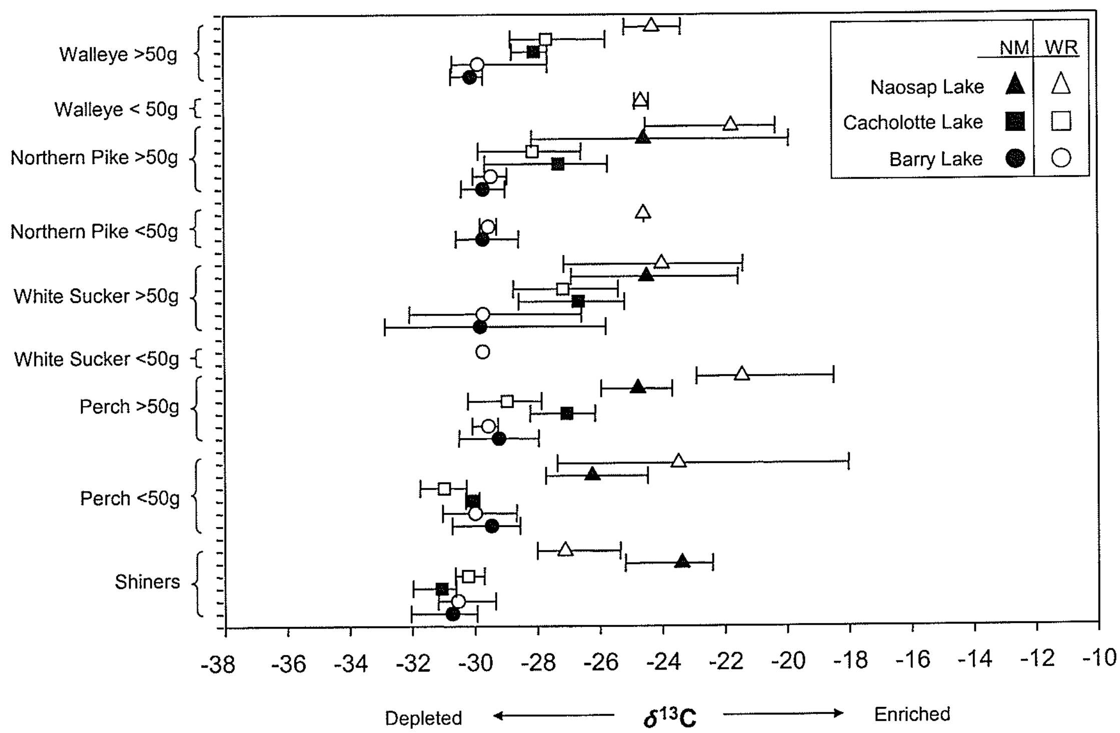


Figure 3.06. Mean $\delta^{13}C$ (‰) for fish from wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap lakes collected in August 2003. Bars represent the ranges. NM = native macrophytes, WR = wild rice.

C. ***Trophic Relationships as Reflected by $\delta^{15}N$ Signatures***

Primary Producers

When comparing the native aquatic plants within each lake, mean $\delta^{15}N$ values were higher for plants from the native macrophyte bays in both Cacholotte and Naosap lakes and similar in the bays sampled in Barry Lake (Figure 3.07). In addition, the overall range of $\delta^{15}N$ for the native macrophytes from all sites was large (-9.04 to 3.93 ‰) (Figure 3.07). $\delta^{15}N$ values of the epiphytic algae collected on Naosap Lake overlapped between the native macrophyte (-0.32 ‰) ($n = 1$) and wild rice bays (-0.49 to 2.93 ‰) (Figure 3.07). Unfortunately, epiphytic algae were not collected in both the wild rice and native macrophyte bays of Barry and Cacholotte lakes for among-site comparisons. The floating leaf and emergent stages of wild rice had similar $\delta^{15}N$ values when compared to the other primary producers, with ranges of -0.51 to 2.78 ‰ and 1.51 to 3.71 ‰ across all lakes, respectively (Figure 3.07).

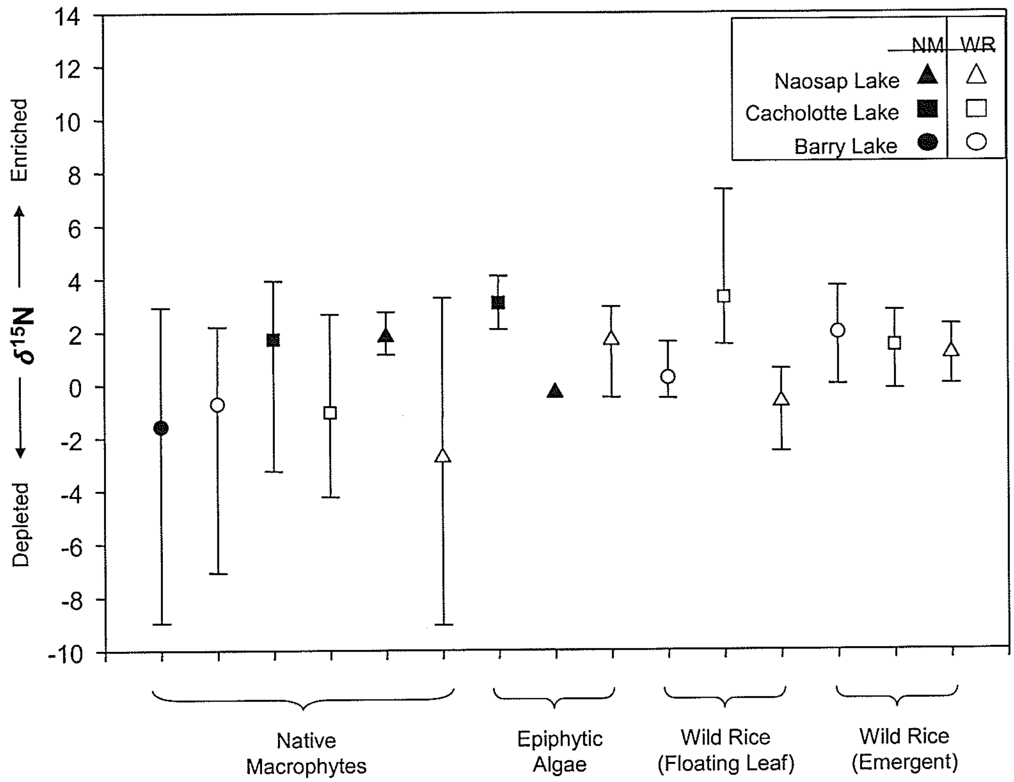


Figure 3.07. Mean $\delta^{15}N$ (‰) for primary producers within Barry, Cacholotte, and Naosap lakes, 2003. Bars represent the ranges. NM = native macrophytes, WR = wild rice.

Pelagic and Littoral Zooplankton

The pelagic zooplankton collected in the main basins of Barry, Cacholotte and Naosap lakes had $\delta^{15}N$ values that ranged from 5.20 to 6.22 ‰, 7.01 to 8.68 ‰, and 5.83 to 6.41 ‰, respectively (Figure 3.08). Littoral zooplankton had $\delta^{15}N$ values ranging from 1.52 to 6.70 ‰ across all lakes, with Δ values equal to or less than 1.86 ‰ within each lake when comparing results for the wild rice and native macrophyte bays (Figure 3.08). However, the littoral zooplankton collected in Barry and Cacholotte lakes were more depleted in the wild rice bay compared to the native macrophyte bay (Δ ranged from

-1.86 to -1.51 ‰) while in Naosap Lake the littoral zooplankton were more enriched within the wild rice bay (Δ of 1.59) (Figure 3.08).

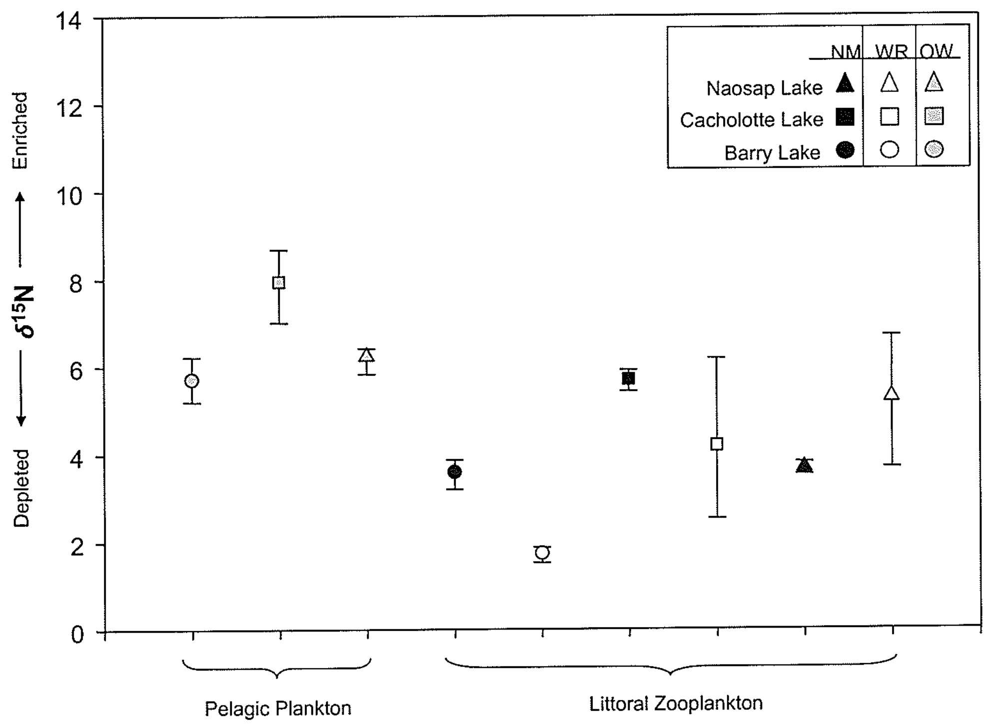


Figure 3.08. Mean δ^{15}N for pelagic and littoral zooplankton within Barry, Cacholotte, and Naosap Lakes, 2003. Bars represent the ranges. NM = native macrophytes, WR = wild rice, OW = open water.

Invertebrates

Mean δ^{15}N for scrapers on Barry, Cacholotte and Naosap lakes in wild rice bays and native macrophyte bays ranged from 2.28 to 3.53 ‰ and from 2.52 to 4.28 ‰, respectively (Figure 3.09). The collectors/gatherers also had overlapping δ^{15}N values within native macrophyte and wild rice bays, and means that ranged from 2.19 to 4.12 ‰ and from 1.48 to 3.01 ‰, respectively. Omnivores had enriched mean δ^{15}N values within native macrophyte bays compared to wild rice bays with values of 5.14 to 5.45 ‰ and 3.28 to 4.40 ‰, respectively. Mean $\delta^{15}N$ values for predators were also enriched in native macrophytes bays compared to wild rice bays, with means ranging from 5.35 to 6.55 ‰ and from 4.02 to 5.29 ‰, respectively. In addition, there was a stepwise enrichment of $\delta^{15}N$ within the functional feeding groups, with scrapers and collectors/gatherers having the lightest signature, omnivores having an intermediate signature, and predators having the heaviest signature in all three lakes and in both types of bays (Figure 3.09).

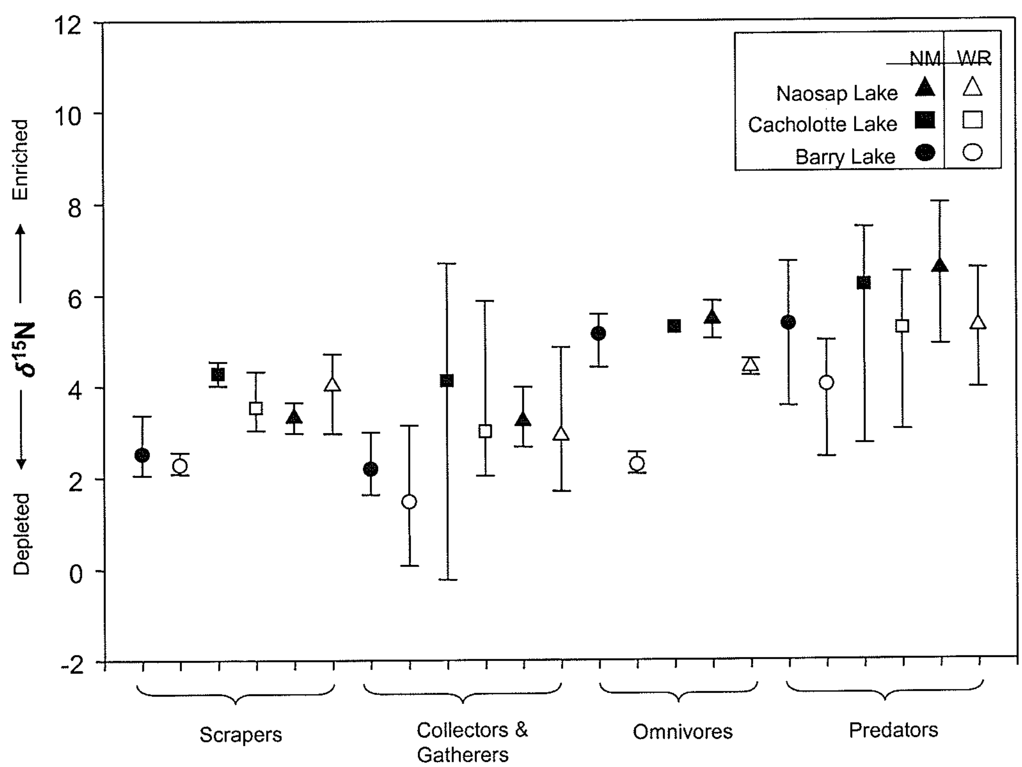


Figure 3.09. Mean $\delta^{15}N$ (‰) for invertebrates from wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap Lakes. Bars represent the ranges. NM = native macrophytes, WR = wild rice.

Fish

For each fish species examined in this study (shiners, perch, white sucker, northern pike, and walleye), there was considerable overlap in the $\delta^{15}N$ values of individuals when comparing the results for the wild rice and native macrophyte bays of the three lakes, with two exceptions. The $\delta^{15}N$ signatures of the large and small perch from Barry Lake were more depleted in the wild rice bay compared to the native macrophyte bay with Δ values of -1.93 to -2.45 ‰, respectively. $\delta^{15}N$ signatures for perch (<50 grams) were also consistently more depleted within the wild rice than the native macrophyte bay on Cacholotte Lake, with a Δ value of -1.67 ‰. For all other fish, the mean $\delta^{15}N$ values were lower in wild rice bays when compared to the native macrophyte bays with Δ ranging from -0.25 to -2.40 ‰, with one exception. Shiners in Cacholotte and Naosap lakes were slightly more enriched in $\delta^{15}N$ within the wild rice bays (Δ ranged from 0.34 to 0.35 ‰) (Figure 3.10).

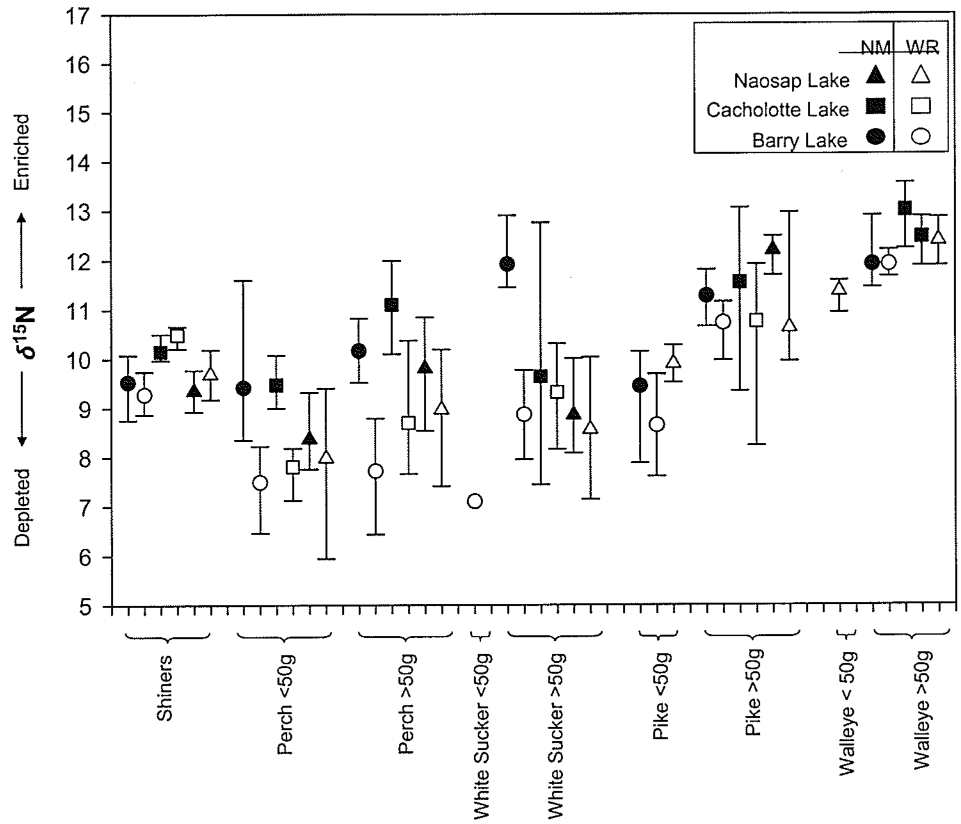


Figure 3.10. Mean $\delta^{15}N$ for fish from wild rice and native macrophyte bays in Barry, Cacholotte, and Naosap lakes. Bars represent the ranges. NM = native macrophytes, WR = wild rice.

Trophic Level

The standardized trophic levels for the organisms collected within the wild rice and native macrophyte bays for each lake are shown in Figure 3.11. For all sites, the trophic levels ranged from 1.66 to 2.0 for collectors and gatherers, 2.0 for scrapers (assumed), 2.2 to 2.79 for invertebrate omnivores, 2.44 to 3.01 for invertebrate predators, 3.69 to 4.08 for shiners, 3.23 to 4.04 for perch <50g, 3.31 to 4.26 for perch >50g, 3.36 to 3.42 for white sucker <50g, 3.54 to 3.99 for white suckers >50g, 3.82 to 4.05 for pike <50g, 4.03 to 4.61 for pike >50g, 4.25 for walleye < 50g, and 4.53 to 4.81 for walleye >50g. For Barry and Naosap lakes, the trophic levels of all the biota from the native macrophyte bays were slightly higher than for the wild rice bays by 0.02 to 0.69 TL (trophic levels). On Cacholotte Lake there was no consistent increase of TL in the wild rice or native macrophyte bay. Shiners, walleye and white sucker (>50g) were situated at a lower trophic level in the native macrophyte bays by 0.33, 0.10 and 0.14 TL, respectively. In contrast, all invertebrates and perch had slightly higher trophic levels in the native macrophyte bays by 0.01 to 0.66 TL, similar to what was found on Barry and Naosap lakes. Based on the differences in the TL values for the top predators and the Collectors/Gatherers, the food web length was longer in the wild rice bays on Barry, Cacholotte and Naosap lakes by 0.64, 0.06, 0.93 TL when compared to the native macrophyte bays.

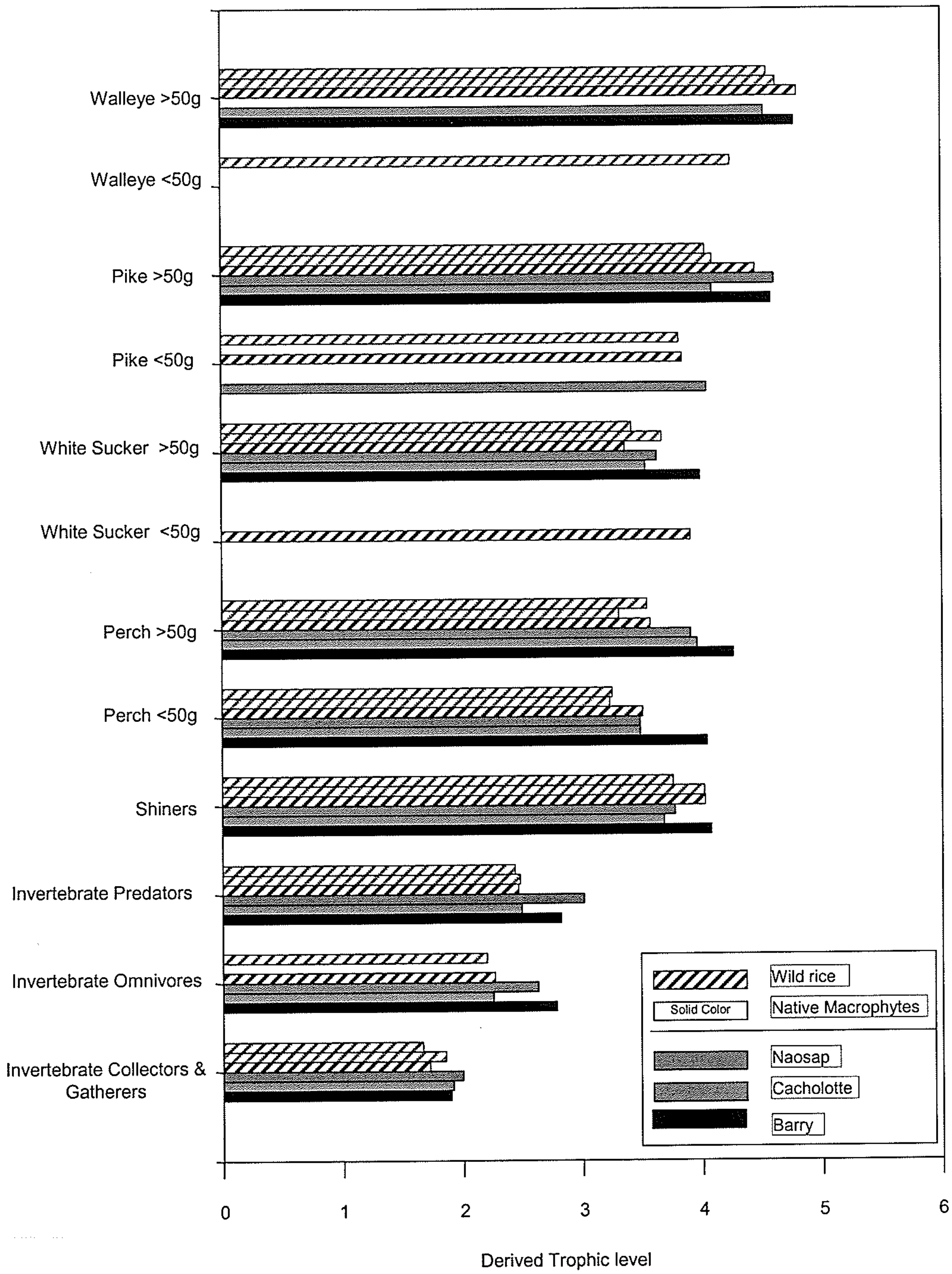


Figure 3.11. Standardized trophic level (using $\delta^{15}N$) for biota from the native macrophyte (NM) and wild rice (WR) bays on Barry, Cacholotte, and Naosap lakes. See Methods section for details on this calculation.

D) General Comparisons of Food Web Structure Between Wild Rice and Native Macrophyte Bays

Biplots of the $\delta^{13}C$ and $\delta^{15}N$ data were used to contrast food web structure between the native macrophyte and wild rice bays in each lake (Figures 3.12 – 3.14). In the native macrophyte bay of Barry Lake, $\delta^{13}C$ ranged from -30.09 to -16.93 ‰ (mean of -25.37 ‰) for native macrophytes and most of these values were more enriched than the invertebrates or fish collected at this site (Figure 3.12a). The scrapers, collectors/gatherers, omnivores and predators invertebrates, as well as the zooplankton, had $\delta^{13}C$ that ranged from -34.49 to -27.32 ‰ and were more depleted in their $\delta^{13}C$ values than the native macrophytes, suggesting an alternate source of carbon for these organisms (as shown in Figure 3.12a). All functional feeding groups fed upon similar sources of carbon although the omnivorous and predatory invertebrates occupied a higher trophic level with mean $\delta^{15}N$ values of 5.14 and 5.35 ‰, respectively, when compared to 2.52 and 2.19 ‰ of the scrapers and collectors/gatherers. The range of $\delta^{13}C$ of littoral zooplankton (-34.00 to -32.96 ‰) was more depleted than for the native macrophytes and epiphytic algae indicating that zooplankton fed upon a different carbon source than was collected. The smaller-bodied fish species (shiners, perch and white sucker) relied on a similar source of carbon as the invertebrate functional feeding groups but were at a higher trophic level suggesting that these fish feed mainly on predatory or omnivorous invertebrates. The top predators in this bay were pike and walleye. Based on both their $\delta^{13}C$ and $\delta^{15}N$ values, these two species relied on a similar source of carbon as the other fishes and they fed mainly on fish due to their elevated $\delta^{15}N$ values (means of 11.28 and 11.92 ‰ compared to means ranging from 9.25 and 10.17 ‰ for the forage fish).

In the wild rice bay in Barry Lake, the food web structure was very similar to that described above for the native macrophyte bay (Figure 3.12b). The native macrophytes themselves were much more enriched in $\delta^{13}C$ than any other organisms collected from this site, and did not appear to provide a significant source of energy to this food web. Wild rice had a mean $\delta^{13}C$ of -28.84 ‰ in both stages and overlapped in its range with the omnivore, collector/gatherer and scraper functional feeding groups but was more likely a food source in the floating leaf stage due to its lower $\delta^{15}N$. As in the native macrophyte bay, the pelagic zooplankton were distinct from the macroinvertebrates collected because they were more depleted in their $\delta^{13}C$ and enriched in $\delta^{15}N$ than the other taxa. Within the invertebrate functional feeding groups, the omnivores and predators were slightly more enriched in their $\delta^{15}N$ but all shared a similar range of $\delta^{13}C$. In addition to the wild rice collected for this study, these organisms were relying on another source or sources of carbon that were not sampled as shown in Figure 3.12b. Shiners, perch and white sucker grouped together both by their $\delta^{13}C$ and their $\delta^{15}N$. These fish were at a lower trophic position than the pike and walleye in the bay; the top predators had mean $\delta^{15}N$ values of 10.73 and 11.92 ‰ compared to the range of mean $\delta^{15}N$ values of the forage fishes (7.10 to 9.28 ‰). All fish relied on similar sources of carbon and had $\delta^{13}C$ that were similar to many of the functional feeding groups and distinct from either the littoral or pelagic zooplankton.

The food web structure within the native macrophyte bay on Cacholotte Lake was very similar to that of Barry Lake described above. The mean $\delta^{13}C$ of the native macrophytes was more enriched than most other organisms and their $\delta^{13}C$ ranged from -28.47 to -23.25 ‰. However, in contrast to Barry Lake there was more overlap in the $\delta^{13}C$ values of fish, invertebrates and the native macrophytes suggesting that native macrophytes at this site may provide an energy source for the organisms (Figure 3.13a). The range of $\delta^{13}C$ of littoral zooplankton (-33.06 to -32.10 ‰) was more depleted than for the native macrophytes and epiphytic algae indicating that zooplankton fed upon a different carbon source than was collected. Within the functional feeding groups, the collectors/gatherers, scrapers, omnivores and predators had overlapping $\delta^{13}C$ indicating that they fed on a similar source of carbon such as epiphytic algae or macrophytes (Figure 3.13a). The omnivorous and predatory invertebrates once again occupied a higher trophic level with mean $\delta^{15}N$ values of 5.29 and 6.21 ‰, when compared to 4.42 and 4.12 ‰ of the scrapers and collectors/gatherers. The pike, walleye, perch > 50g and white suckers fed on a similar carbon source as the invertebrates although the pike and walleye were situated at a higher trophic level with mean nitrogen values of 11.55 and 13.01 ‰, when compared to the yellow perch, white sucker and shiners which ranged from 9.48 to 11.10 ‰. Perch < 50g and shiners had $\delta^{13}C$ that were more depleted than the other fish, suggesting a link to zooplankton as a primary carbon source (Figure 3.13a).

In the wild rice bay on Cacholotte Lake, the food web structure was very similar to the native macrophyte bay described above (Figure 3.11b). The $\delta^{13}C$ of native macrophytes ranged from -30.09 to -22.82 ‰ and were slightly more depleted at this site than in the native macrophyte bay. Wild rice had a mean $\delta^{13}C$ of -28.40 ‰ in both stages and overlapped in its range with the scraper, collector/gatherer and predator functional feeding groups (omnivores were not collected). However, these organisms were also relying on another source of carbon not sampled as some of their $\delta^{13}C$ values were more depleted than those of wild rice (see Figure 3.13b). As in the native macrophyte bay, the pelagic zooplankton were distinct from the macroinvertebrates collected because their $\delta^{13}C$ and $\delta^{15}N$ values were more depleted and enriched, respectively, than other taxa. The perch, white sucker, and shiners grouped together both by their $\delta^{13}C$ and their $\delta^{15}N$ although the perch > 50g and white sucker were more enriched in their $\delta^{13}C$ (by 1.27 to 3.81 ‰) when compared to the shiner and perch < 50g. These differences were due to the fact that the shiners and smaller perch likely relied on zooplankton, which were more depleted then the other invertebrates. These fish were at a lower trophic position than the pike and walleye in the bay; the top predators had mean $\delta^{15}N$ values of 10.76 and 12.47 ‰ compared to a range in the mean $\delta^{15}N$ values of the forage fish of 7.81 to 10.49 ‰. The comparable $\delta^{13}C$ values for the invertebrates and the fishes suggest that they share a similar source of carbon and that many of the forage fishes feed directly upon the functional feeding groups collected herein.

The food web structure within the native macrophyte bay of Naosap Lake was similar to those of Barry and Cacholotte lakes, however the entire food web was slightly more enriched in $\delta^{13}C$ (Figure 3.14a). The mean $\delta^{13}C$ of the native macrophytes was once again more enriched compared to that of the invertebrates and had $\delta^{13}C$ values ranging from -24.88 to -12.33 ‰. Epiphytic algae were also collected in this bay, however, the $\delta^{13}C$ of -13.69 ‰ was not near the range of any of the other organisms, suggesting that it is not a common food source for any of the invertebrates or fish. The range of the collectors/gatherers (-31.93 to -27.00 ‰) and the omnivores (-28.97 to -24.33 ‰) overlapped with the native macrophytes and, therefore, invertebrates may have used these plants as a food source. Within all of the functional feeding groups, the scrapers, collectors/gatherers, omnivores and predators groups fed upon on similar sources of carbon although the omnivore and predator invertebrates occupied a higher trophic level with mean $\delta^{15}N$ values of 5.45 and 6.55 ‰, when compared to 3.31 and 3.23 ‰ of the scrapers and collectors/gatherers. The forage fish in this bay grouped close together as compared to the other lakes examined. The fish had similar $\delta^{13}C$ values, indicating that they relied upon a similar carbon source, although the pike were higher in the trophic level with a $\delta^{15}N$ value of 12.20 ‰ compared to 8.37 and 9.81 ‰ of the other fish (walleye were not collected in this bay).

In Naosap Lake, the overall range of $\delta^{13}C$ for the food web in the wild rice bay (-32.62 to -11.42 ‰) was smaller than that found for the native macrophyte bay (-37.02 to -12.33 ‰) (Figure 3.14b). The $\delta^{13}C$ for the native macrophytes in the former bay ranged from -27.91 to -11.42 ‰. Wild rice had a mean $\delta^{13}C$ of -27.60 ‰ in both stages and overlapped in its range with the scraper, collectors/gatherers and predator functional feeding groups. Epiphytic algae had a mean $\delta^{13}C$ value of -24.13 ‰, which was within the range observed for the omnivores and collectors/gatherer invertebrate functional feeding groups. Results for the pelagic and littoral zooplankton were similar to the other lakes in that the $\delta^{13}C$ values were typically more depleted than the fish, invertebrates and macrophytes; however, on Naosap Lake their $\delta^{13}C$ and $\delta^{15}N$ values overlapped slightly suggesting that these organisms used zooplankton as a food source. The small perch and white sucker grouped together both by their $\delta^{13}C$ and their $\delta^{15}N$. Shiners were more depleted than the other fish signifying a reliance on zooplankton as an energy source. The pike and walleye occupied the highest trophic level in the bay; these top predators had mean $\delta^{15}N$ values of 10.63 and 12.39 ‰, respectively, whereas the forage fishes had mean $\delta^{15}N$ values of 7.98 to 9.69 ‰. The ranges of the $\delta^{13}C$ for fish and invertebrates were large and suggested some diverse sources of carbon in this food web.

Within Barry, Cacholotte and Naosap lakes native macrophytes appear to have a very small role as a source of carbon for the invertebrates and fish, and the low $\delta^{15}N$ of these plants suggests that fish did not utilize feed upon them directly. In contrast, the

$\delta^{13}C$ of wild rice overlapped with the values obtained for many of the invertebrates and fish in all three lakes suggesting that it may provide a source of carbon for these organisms, but was more likely a food source in the floating leaf stage due to its lower $\delta^{15}N$. However, it was evident from these biplots that I did not collect all of the different sources of C in these bays. As an example, the $\delta^{13}C$ of littoral zooplankton was more depleted than any primary producer analysed and they likely fed upon phytoplankton in the water column.

In all lakes, the pike and walleye were situated at the top of the food web followed by the forage fish, yellow perch, white suckers, and shiners. Shiners were also more depleted than the other fish signifying a reliance on zooplankton as an energy source. The predatory and omnivorous invertebrates were at a higher trophic level when compared to the other invertebrate groups and appeared to be a main source of energy for the forage fish.

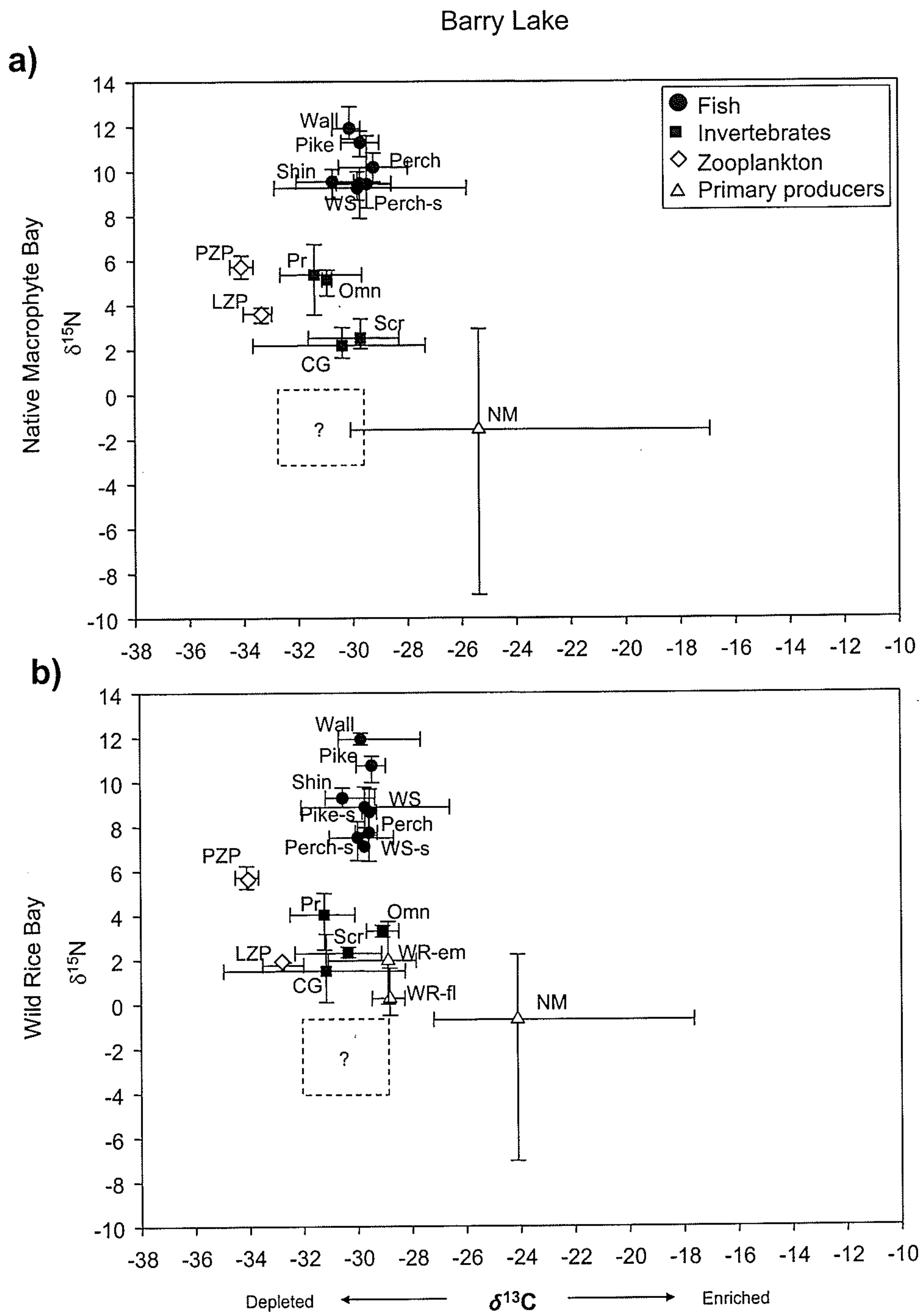


Figure 3.12. Bi-plots of mean $\delta^{13}C$ and $\delta^{15}N$ (‰) of biota for a) native macrophyte bay and b) wild rice bay on Barry Lake. Bars = ranges of $\delta^{13}C$ and $\delta^{15}N$ values. (NM = native macrophytes; WR-fl = wild rice floating leaf; WR-em = wild rice emergent; PZP-pelagic plankton; LZP = littoral zooplankton; Scr = invertebrate scrapers; CG = invertebrate collectors & gatherers; Omn = invertebrate omnivores; PR = invertebrate predators; Wall = walleye; Shin = shiners; WS = white sucker; -s = <50g.

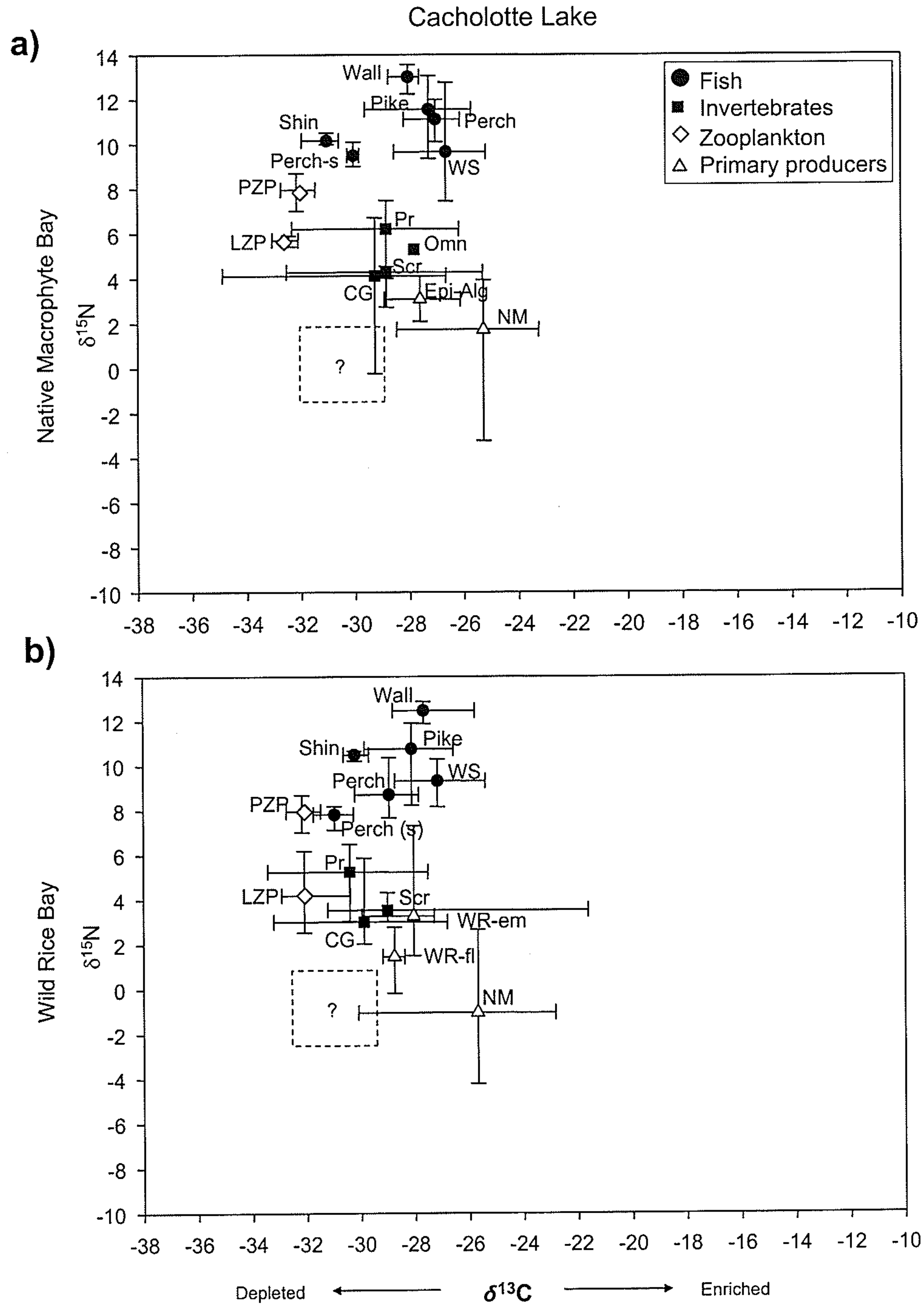


Figure 3.13. Bi-plots of mean $\delta^{13}C$ and $\delta^{15}N$ (‰) of biota for a) native macrophyte bay and b) wild rice bay on Cacholotte Lake. Bars = ranges of $\delta^{13}C$ and $\delta^{15}N$ values. (NM = native macrophytes; WR-fl = wild rice floating leaf; WR-em = wild rice emergent; Epi-Alg = epiphytic algae; PP-pelagic plankton; LZP = littoral zooplankton; Scr = invertebrate scrapers; CG = invertebrate collectors & gatherers; Omn = invertebrate omnivores; PR = invertebrate predators; Wall = walleye; Shin = shiners; WS = white sucker; (s) = <50g.

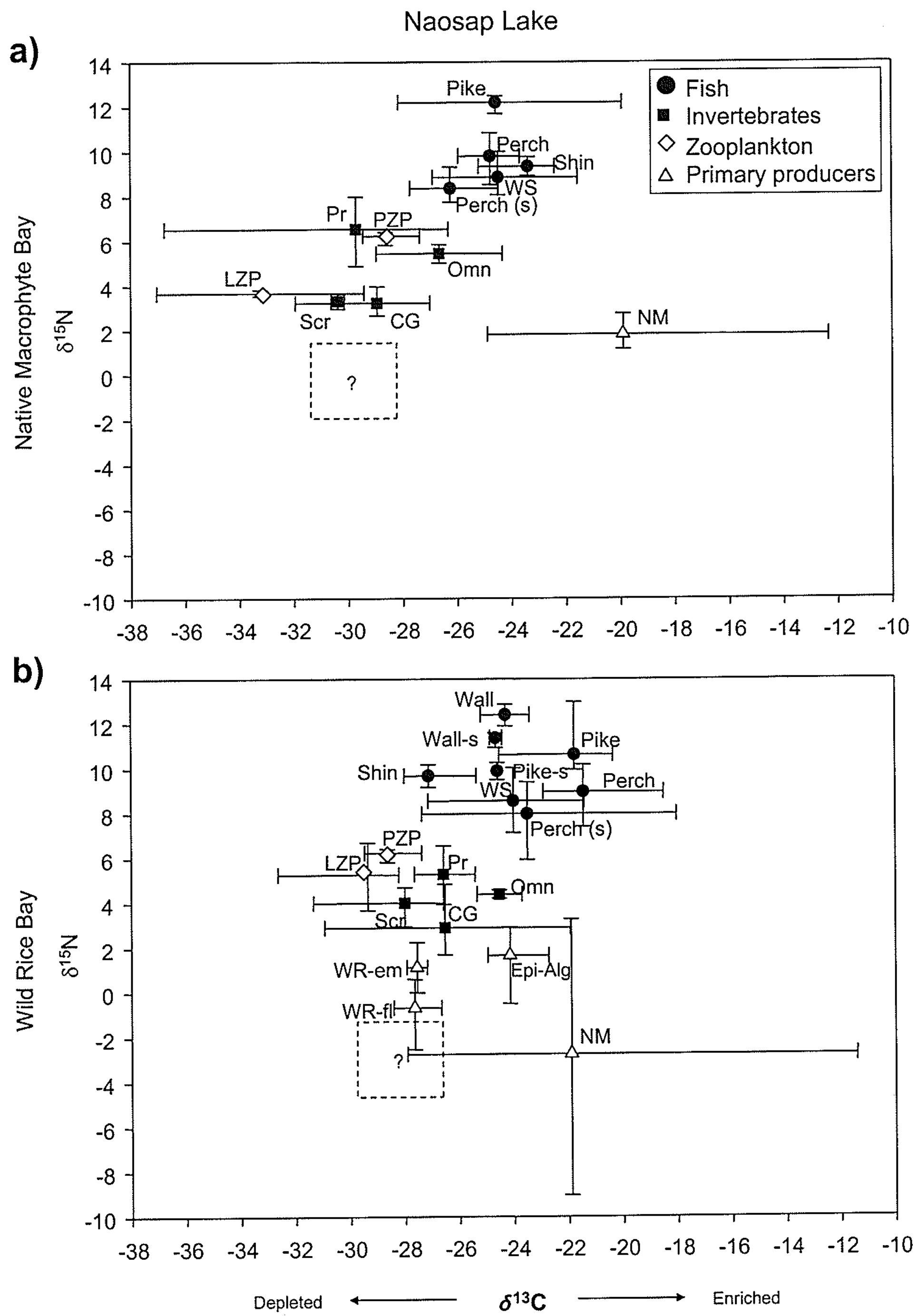


Figure 3.14. Bi-plots of mean $\delta^{13}C$ and $\delta^{15}N$ (‰) of biota for a) native macrophyte bay and b) wild rice bay on Naosap Lake. Bars = ranges of $\delta^{13}C$ and $\delta^{15}N$ values. (NM = native macrophytes; WR-fl = wild rice floating leaf; WR-em = wild rice emergent; Epi-Alg = epiphytic algae; PP-pelagic plankton; LZP = littoral zooplankton; Scr = invertebrate scrapers; CG = invertebrate collectors & gatherers; Omn = invertebrate omnivores; PR = invertebrate predators; Wall = walleye; Shin = shiners; WS = white sucker; (s) = <50g.

Discussion

Several studies have shown that introduced species can affect aquatic food web structure (Mitchell et al., 1996; Vander Zanden et al., 1999; Ricciardi, 2001; Yan et al., 2001). However, no previous studies have addressed the effects of wild rice (*Zizania palustris*) on energy flow or trophic relationships. The primary goal of this study was to use stable carbon and nitrogen isotope analyses of the biota to compare food web structures and food web interactions between invertebrates and fish within bays dominated with natural vegetation and bays where wild rice was introduced. An additional objective of the study was to determine if wild rice provides an additional food source for aquatic invertebrates and fish.

Because invertebrates have fast turnover rates (Hobson et al., 1994), the seasonal variability in $\delta^{13}C$ or $\delta^{15}N$ values can be high and must be considered before making interpretations of food web relations (Kidd et al., 1999; Grey et al., 2001). For example, in the study by Grey et al. (2001) zooplankton $\delta^{13}C$ ranged from -31.0 to -28.0 ‰ and $\delta^{15}N$ ranged from 8.0 to 10.0 ‰ over the summer. Results of my study concur with the previous studies in that there were seasonal changes in the C and N isotope composition of invertebrates in the lakes in west-central Manitoba; however, these changes were typically small (generally <1.95 ‰, with three exceptions) and did not follow any predictable pattern. These results reinforce the need to sample small-bodied organisms on numerous occasions to determine the seasonal variability in their isotopic composition and to accurately characterize their trophic positioning within the food web.

Previous studies have shown that $\delta^{13}C$ values of primary producers can vary considerably within aquatic food webs (e.g., Hecky & Hesslein, 1995). For example, the base of the food webs of three lakes in northwestern Ontario had $\delta^{13}C$ values that ranged over 20 ‰, from the more ^{13}C-enriched benthic primary producers to the more ^{13}C-depleted signatures of pelagic primary producers (Hecky & Hesslein, 1995). Based on the range of $\delta^{13}C$ observed for primary consumers, the range of $\delta^{13}C$ for primary producers is similar to what has been observed in other systems (Hecky and Hesslein, 1995) and is about 20 ‰ in all of the food webs examined herein. At the next trophic level for the primary consumers, I found a similar range in the $\delta^{13}C$ (about 14 to 15 ‰) as those seen in the food webs of Hecky & Hesslein (1995). Kidd et al. (1999) examined the pelagic and littoral biota in Lake 227 at the Experimental Lakes Area (ELA) and found similar $\delta^{13}C$ values for invertebrates within the lower trophic levels ($\delta^{13}C$ ranged from -36.08 to -22.65 ‰) as in the present study. In addition, the food web structure within Barry, Cacholotte and Naosap lakes were very similar to what has been observed in the previous studies, where the range in $\delta^{13}C$ signatures at the base of the food webs were very broad and became progressively smaller up the food web to tertiary consumers. With the exception of Barry Lake, the top predators in my study also had $\delta^{13}C$ values that were mid-way between the extremes in $\delta^{13}C$ values for the primary producers as was observed in Hecky & Hesslein (1995) and Kidd et al. (1999).

In the current study, the invertebrates were grouped into functional feeding groups consisting of scrapers, collectors/gatherers, omnivores and predators. Mean $\delta^{13}C$ values of the invertebrates for Barry, Cacholotte and Naosap lakes ranged from -36.74 to -24.33 ‰ within the native macrophyte bays and from -34.94 to -21.62 ‰ within the wild rice bays. The scrapers and collectors/gatherers within these lakes were found at the lowest trophic level while predators were found at the highest trophic level. Overall, there was considerable overlap in $\delta^{13}C$ values of each invertebrate functional feeding group between wild rice and native macrophyte bays. Only omnivores on Barry Lake had different $\delta^{13}C$ values, which were more enriched in the wild rice bay by 1.87 ‰ (Figure 3.05). Therefore, invertebrates appear to have obtained their energy from similar carbon sources within either wild rice or native macrophyte bays.

What is the primary food or carbon source for the invertebrates and fish communities within these bays? The results of this study show that native macrophytes did not typically provide a primary energy source for invertebrate or fish communities as the range of their carbon only overlapped slightly or not at all with the ranges for both fish and invertebrates. I suggest that the invertebrates obtained their carbon from a source that was slightly more depleted than the native macrophytes (shown on biplots Figures 3.12 – 3.14 as a box with "?"). This "?" energy source was not determined but it is likely to be detritus, periphyton, phytoplankton or fine particulate organic matter (FPOM) within the water column or on the surface of the substrates. Beaudoin et al. (2001) found these carbon stores to be important sources of energy in boreal lake food webs.

Does wild rice provide a food source for the invertebrate and fish communities? Results from the current study show that wild rice may provide an energy source for the invertebrates and fish within these lakes, as both invertebrates and fish $\delta^{13}C$ ratios overlapped with that of the introduced plant. Although it is likely that some organisms fed directly on wild rice, it was not the only source of carbon in these bays; many invertebrates appeared to rely on another more depleted source of carbon not collected in this study (Figure 3.12 – 3.14). In addition, $\delta^{15}N$ values for fish were higher than they should have been if these organisms fed directly on the wild rice. Predatory and omnivorous invertebrates also did not appear to feed on wild rice as their $\delta^{15}N$ values were not enriched by 3.4 ‰ over values measured for wild rice. Scrapers and collectors/gatherers had $\delta^{13}C$ and $\delta^{15}N$ values that are consistent with a diet of wild rice and therefore appear to use this plant as a food source. However, the isotopic signatures of the scrapers and collectors/gatherers were similar to those found in the native macrophyte bays suggesting that they were feeding on the same source of carbon within both wild rice and native macrophyte bays.

Studies have reported that after a wild rice growing season, significant amounts of straw remain in the littoral zones of wild rice bays and this straw eventually settles on the lake bottom and decomposes (Tattam, 1998; Watson et al., 2001; Derksen, 2002;). Other studies have shown that increased decomposition within a littoral habitat would result in more of the isotopically lighter CO_2 being available for photosynthesis within the water column (Keough et al., 1996). If there was more decomposition in bays with wild rice and if the carbon from decomposing plant matter was being used as a source or CO_2 for primary producers, such as phytoplankton, one would expect that $\delta^{13}C$ value of littoral zooplankton in the wild rice bays would be more depleted than the littoral zooplankton of native macrophyte bays. There was no evidence of a depleted $\delta^{13}C$ value for the littoral zooplankton in any of the wild rice bays for Barry, Cacholotte and Naosap lakes (Figure 3.03).

In this study, there were some consistent differences in the $\delta^{15}N$ values of organisms between the wild rice and native macrophyte bays. When comparing the invertebrate $\delta^{15}N$ values between the wild rice and native macrophyte bays, there were consistently lower values for the wild rice bays on all lakes ($\delta^{15}N$ depleted by -0.09 to

-1.86 ‰; Figure 3.08). Although the fish $\delta^{15}N$ values were more variable between lakes, the majority were also more depleted within the wild rice bays ($\delta^{15}N$ depleted by -0.25 to -3.03 ‰). Walleye (> 50 g) and shiners were the only fish with enriched $\delta^{15}N$ values in the wild rice bays ($\delta^{15}N$ enriched by 0.34 to 0.45 ‰). These differences among bay types suggest some differences in N cycling that warrant further investigation.

Results from this study also support previous studies on the fractionation of C and N isotopes in freshwater food webs (DeNiro & Epstein 1981; Fry & Sherr, 1984; Minagawa & Wada, 1984). The trophic levels for both invertebrates and fish were consistent between wild rice and native macrophyte bays and showed an enrichment in $\delta^{13}C$ by up to 1 ‰ with every step in trophic level from the invertebrate scrapers to the piscivorous walleye (Figures 3.12-3.14). There was also a consistent enrichment of $\delta^{15}N$ by ~3.0 ‰ between the different trophic levels for the wild rice bays and native macrophyte bays on all lakes (Figures 3.07 -3.10).

In order to compare food chain length across bays and lakes, trophic positions were standardized to the $\delta^{15}N$ of scrapers. The other invertebrates from the native macrophyte bays had consistently higher derived trophic levels than those from the wild rice bay in each lake (Figure 3.11). For Barry and Naosap lakes, invertebrates were situated at higher trophic levels (TL) within native macrophyte bays when compared to wild rice bays with differences ranging between 0.02 to 0.69 TL for each functional feeding group. With one exception, all fish in Barry and Naosap lakes were also situated at a higher trophic level within the native macrophyte than wild rice bays. Walleye >50 g had a lower trophic level in the native macrophyte bay on Barry Lake. On Cacholotte Lake, shiners, walleye and white suckers had higher standardized trophic levels within the wild rice bays. Although these comparisons are somewhat oversimplified, they increase our understanding of the impacts of wild rice on individual food webs (Hobson et al., 2002).

The overall food chain length and number of trophic levels of the lakes studied in Ontario (Lakes 240 and 373 at ELA and Trout Lake) (Hecky & Hesslein, 1995) were similar to the ones examined in the current study. Primary producers were found at the first trophic level, scrapers and collectors and gatherers at the second, predator and omnivorous invertebrates at the third, forage fish (shiners, yellow perch, and white sucker) at the fourth, and piscivorous fish (northern pike and walleye) at the top of the food chain.

Although the food web structure was similar among Barry, Cacholotte and Naosap lakes within both wild rice and native macrophyte bays, the overall values of $\delta^{13}C$ of the biota varied between lakes and appeared to be related to the size of the lake. In the current study, Barry Lake had the most depleted carbon isotope signatures for the littoral biota of the three lakes examined followed by Cacholotte, and then Naosap having the most enriched $\delta^{13}C$ values (Figures 3.12 – 3.14). Keough et al. (1996) suggested that larger bodies of water, such as Naosap Lake, rely more on the diffusion of atmospheric CO_2 into the water column to support primary production; therefore, organisms have $\delta^{13}C$ values that are more enriched than in smaller water bodies. In contrast, small water bodies are not mixed as rapidly and the lighter carbon from decomposition becomes a more important source for photosynthesis, resulting in $\delta^{13}C$ values that are depleted (Keough et al., 1996).

In summary, results of this study indicated that invertebrate and fish communities had similar carbon and nitrogen isotope values and trophic relationships within wild rice and native macrophyte bays. Based on these observations, it appears that wild rice does not alter the flow of energy and trophic interrelationships of the primary through tertiary consumers in the bays in which it was introduced.

Chapter 4

General Summary

4.0 Introduction

Invertebrates are an important component of aquatic food webs because they provide a food source for most fish species during one or more stages of their life cycle (Gerking, 1962; Fairchild, 1982; Mittelbach, 1984; Carpenter & Lodge, 1986). Diversity and abundance of invertebrate communities are dependent upon a number of key environmental factors. One of these factors is the nature of the macrophyte community (Dvorak & Best, 1982; Crowder et al., 1982; Brönmark, 1985; Engel, 1990; Heino, 2000). Macrophytes provide habitat complexity, food, and refuge for invertebrates from predation (Wiley et al., 1984; Engel, 1990; Jones et al., 1998). Furthermore, they provide additional surface area for the growth of epiphytic algae, which is an important food source for invertebrates (Cyr & Downing, 1988).

In west-central Manitoba, wild rice (*Zizania palustris*), an aquatic emergent macrophyte, was introduced into lakes in the early 1980s (Watson et al., 2001). With one exception, no previous research examined the impacts of wild rice on aquatic communities (Watson et al., 2001). Watson et al. (2001) showed that there were differences in fish population, water chemistry, macroinvertebrates, phytoplankton, algal communities, and sediment profiles between site with wild rice and those in the open water area. However, no comparisons were made in their study between bays with wild rice and bays with native macrophyte communities.

The present study was designed to compare invertebrate communities and food webs in wild rice and native macrophyte bays. This study was intended to provide much-needed information on the littoral communities in bays with wild rice and with native macrophytes.

4.1 Objectives

The objectives of this project were to compare wild rice and native macrophyte bays on Barry, Cacholotte and Naosap lakes. These lakes are located near Flin Flon in the west-central region of Manitoba. Each lake had a wild rice and a native macrophyte bay of similar size and water depth, was accessible by boat, and had little fishing pressure. Barry Lake (Figure 1.08) was the smallest lake examined followed by Cacholotte (Figure 1.09) and then Naosap Lake (Figure 1.10). The surface areas of the study lakes were approximately 1.5, 3.5 and 24 km^2, respectively. To determine impacts of wild rice on invertebrate and fish communities, a number of variables were compared between a wild rice bay and a native macrophyte bay in each of these lakes. These variables included:

1) water quality parameters, dissolved oxygen, and water temperature
2) invertebrate abundance
3) invertebrate diversity – effective richness, absolute richness and evenness
4) invertebrate community composition
5) pathways of energy flow (stable carbon isotope ratios) and
6) trophic positioning of biota (stable nitrogen isotope ratios).

4.2 Results

Environmental variables

Fifteen water quality variables were compared between wild rice bays and native macrophyte bays in June and August of 2003. In June, higher levels of ammonia, dissolved organic carbon, and total dissolved nitrogen and lower levels of suspended nitrogen and suspended carbon were observed within the wild rice bays compared to the native macrophyte bays. In August, no differences in these parameters were observed with the exceptions of dissolved inorganic carbon and dissolved organic carbon which were consistently higher in wild rice bays. Dissolved oxygen concentrations in the water column were significantly lower in most wild rice bays when compared to the native macrophyte bays. There were no consistent trends in the water temperatures between the wild rice and native macrophyte bays in Barry, Cacholotte and Naosap lakes.

Invertebrate abundance

Invertebrate abundances were compared between native macrophyte and wild rice bays using emergence trap, bottle trap and bucket volume samples collected in June and August of 2003. When results were compared across all lakes, significantly higher abundances of emerging invertebrates were found within the native macrophyte bays in June. In August, there were also higher numbers of emerging invertebrates in the native macrophyte bays but the differences were not statistically significant. In contrast, the nektonic invertebrates (those within the water column) were generally more abundant within the wild rice bays in both June and August compared to the native macrophyte bays, however, the differences were not statistically significant.

Invertebrate diversity

In this study I found that the diversity (absolute richness, effective richness and evenness) of invertebrate taxa was not significantly different between the wild rice and native macrophyte bays across lakes using paired *t*-tests. There were, however, significant differences in the diversity of invertebrate taxa between the wild rice and native macrophyte bays within some of the individual lakes. For samples taken with bottle traps, significantly more invertebrate taxa (absolute richness) were found within the wild rice bays on Barry and Naosap lakes in June, during the floating leaf stage of the wild rice life cycle. In August on Naosap Lake, there were also significantly more invertebrate taxa in the wild rice bay using the same sampler. In contrast, significantly more taxa were observed in the native macrophyte bay on Barry Lake during this sampling period. On Cacholotte Lake, no significant differences were observed between the absolute richness of taxa. With respect to evenness, the invertebrate community was significantly more even in Naosap Lake within the native macrophyte bay compared to the wild rice bay in the June bottle trap samples. However, this was the only difference in evenness that was observed between the wild rice and native macrophyte bays for both sampling times. Invertebrate diversity was also compared between the individual lakes using the emergence trap samples. For these samples, no significant differences in absolute richness, effective richness and evenness were observed for the invertebrate communities within either the wild rice or native macrophyte bays.

The abundance and evenness of the functional feeding groups (filter-feeders, collector/gatherers, predators, omnivores, shredders, scrapers, parasites) within bottle trap samples were also compared between wild rice and native macrophytes bays and there were no significant differences within or across lakes for both June and August. For effective richness within and across lakes, there were no significant differences between the invertebrate community within wild rice and native macrophyte bays, with the exception of Naosap Lake which had a significantly higher effective richness ($P < 0.05$) of functional feeding groups in the wild rice bay during the August sampling period.

There was also only one significant difference for absolute richness for functional feeding groups between wild rice and native macrophyte bays for any of the lakes and sampling periods; Naosap Lake during June had higher numbers of functional feeding groups in June in the wild rice bay.

Invertebrate composition

Fifty-two invertebrate taxa and seven invertebrate functional feeding groups were captured in bottle traps set in the wild rice and native macrophyte bays of Barry, Cacholotte and Naosap lakes in June and August of 2003 (Tables 2.04 - 2.05). The most common taxa collected within the wild rice and native macrophytes were Hydracarina (water mites), Ostracoda, Oligochaeta, Amphipoda, Chironomidae, Corixidae, Hydrobiidae, Microturbellaria, Haliplidae, and Tanypodinae (predacious Chironomidae).

These taxa were found in different abundances within the wild rice and native macrophyte bays. For the bottle trap samples, Amphipoda, Ostracoda, Hydracarina, Microturbellaria, Tanypodinae, and Physidae were significantly more abundant within the wild rice bays in at least one of the lakes examined; this difference was more apparent in June than in August samples using both PCA and *t*-tests. In contrast, Haliplidae, Chironomidae, and Gyrinidae were significantly more abundant within the native macrophyte bays. Emergence trap data for invertebrate composition were not compared between the wild rice and native macrophyte bays because 90% of the invertebrates found were in the family Chironomidae.

Multiple Discriminant Analysis (MDA) was used to compare statistically the PCA results and to examine whether invertebrate taxa and functional feeding group data of the predetermined groups (wild rice and native macrophytes) were significantly different in bottle trap samples. In June, invertebrate community composition was significantly different for both taxa and functional feeding groups with one exception; Cacholotte Lake functional feeding group community composition was not significantly different. In August, similar results were found on Naosap Lake, both for functional feeding groups and taxa community composition, and Cacholotte Lake for the invertebrate taxa community composition with significant differences between bay types. However, no significant differences were observed on Barry Lake, for both taxa and functional feeding group community composition, and for the functional feeding group community composition on Cacholotte Lake between wild rice and native macrophyte bays.

Energy flow

Carbon and nitrogen stable isotopes have been used in many ecological studies to analyze energy flow and food web structure in aquatic ecosystems. Naturally-occurring stable carbon isotope ratios ($^{13}C/^{12}C$), expressed as $\delta^{13}C$ in organisms, are effective at tracing energy flow within food webs because there is little (< 1 ‰) or no fractionation of the isotopes during trophic transfer (Fry & Sherr, 1984; Peterson & Fry, 1987; Gu et al., 1997; Harvey & Kitchell, 2000). In combination, stable carbon and nitrogen isotopes provide information on long-term dietary habits of organisms and a quantitative method for analysing trophic interrelationships (Campbell, 2003).

In general, carbon flow was similar in bays with wild rice and with native macrophytes. For the invertebrates collected herein, zooplankton were the most depleted in $\delta^{13}C$ and the invertebrate functional feeding groups (scrapers, collector/gatherers, omnivores and predators) had overlapping $\delta^{13}C$ values between the wild rice and native macrophyte bays. The invertebrate functional feeding groups had $\delta^{13}C$ values that overlapped with those of the forage fish and top predators, suggesting that these invertebrates were an important food source in these bays. In addition, the $\delta^{13}C$ of the wild rice (but typically not the native macrophytes) overlapped with some of the $\delta^{13}C$ values of the functional feeding groups, suggesting that this plant provides some of the energy to food webs in wild rice bays. In general, the δ^{13} signatures were similar in biota from the native macrophyte bays and wild rice bays. The magnitude of the range in $\delta^{13}C$ signatures at the base and middle of the food webs was also similar across bays, and comparable to that observed in other temperate, oligotrophic lakes (Hecky and Hesslein, 1995).

Food web structure

In contrast to the $\delta^{13}C$ signatures, the heavier isotopes of nitrogen are preferentially enriched by 3-4 ‰ from prey to predator (DeNiro & Epstein 1981, Minagawa & Wada 1984, Ponsard & Averbuch, 1999). This enrichment of nitrogen provides a measure of the relative trophic level of an organism within a food web. In this study, $\delta^{15}N$ was used to determine if food web structure was different between the wild rice and native macrophyte bays. The number of trophic levels and relative trophic positioning of the biota within the wild rice and native macrophyte bays were very similar; each bay had five distinct levels with primary producers at the first level, scrapers and collector/gatherer invertebrates at the second, predatory and omnivorous invertebrates at the third, forage fish at the fourth, and piscivorous fish at the top the aquatic food webs within these lakes.

In order to account for differences in the $\delta^{15}N$ at the base of the food webs of these bays, standardized trophic levels (TL) of the biota were calculated using the $\delta^{15}N$ of a longer-lived primary consumer. The TL of the biota were then compared between the wild rice and native macrophyte bays to determine whether biota from one bay occupied a higher trophic position than those from another bay. For Barry and Naosap lakes, the trophic levels of all the biota from the native macrophyte bays were slightly higher than for the wild rice bays by 0.02 to 0.69 TL. On Cacholotte Lake the TL of the organisms were also different between the wild rice and native macrophyte bays but the trends were not consistent across all types of organisms. Shiners, walleye and white sucker >50g were situated at a lower trophic level in the native macrophyte bays by 0.33, 0.10 and 0.14 TL, respectively. In contrast, all invertebrates and perch had slightly higher trophic levels in the native macrophyte bays by 0.01 to 0.66 TL, similar to that found on Barry and Naosap lakes.

The overall food chain length was longer in the wild rice bays on Barry, Cacholotte and Naosap lakes by 0.64, 0.06, 0.93 $\delta^{15}N$ compared to the native macrophyte bays. When comparing the bi-plots of the $\delta^{15}N$ and $\delta^{13}C$ data, I found that the general shape and structure of food webs were similar between wild rice and native macrophyte bays in this study, and similar to the structure of aquatic food webs in north-western Ontario (Hecky & Hesslein, 1995).

4.3 Relevance and Implications

No previous study has compared the biota and the food webs between bays seeded with wild rice and bays dominated by native vegetation to determine if wild rice has adverse effects on the native biota. Relationships between wild rice, fish and invertebrates communities have been previously examined in only one study which compared a wild rice bay to an open water area on one lake only (Watson et al., 2001). The study by Watson et al. (2001) found that invertebrate communities were more abundant within the wild rice bay compared to the open water by 1.5, 2.2 and 6.0 times in June, August and September of 1998, respectively. Watson et al. (2001) also found that there were more juvenile northern pike and white suckers within the wild rice bays; however, pike 2+ and older were found in higher numbers within the open water. However, this study did not compare similar habitats because the open water was too deep to support the growth of wild rice. Therefore, a comparison of the wild rice and open water sites likely did not truly reflect the impacts of wild rice in lakes because there were no contrasts made to areas where native macrophytes may also dominate the littoral zone. The present study expanded on the work by Watson et al. (2001) by comparing native macrophyte and wild rice bays within lakes to determine whether the introduction of wild rice altered invertebrate community composition, trophic relationships, water quality, water temperatures and dissolved oxygen levels.

Through the use of a number of different sampling methods, the present study has demonstrated that invertebrates were found in similar numbers in the water column within wild rice and native macrophyte bays on Barry, Cacholotte and Naosap lakes. In contrast, significantly higher numbers of emerging invertebrates were found in native macrophyte bays than in wild rice bays. Invertebrate community composition differed within the wild rice bays compared to the native macrophyte bays with more amphipods, ostracods, Hydracarina, Microturbellaria, Tanypodinae, and Physidae in the former habitat and greater abundances of Chironomidae, Gyrinidae, and Haliplidae in the latter habitat.

These differences in invertebrate composition were probably due to the fact that the wild rice and native macrophytes have different physical structure. Wild rice has single stems compared to native macrophytes that have more foliage and a higher structural complexity (Figure 1.06).

As in the study by Watson et al. (2001), dissolved oxygen concentrations during the open water season and under the ice (Lavergne, 2005) were significantly lower within the wild rice bays when compared to the non-wild rice bays. This difference was likely due to higher decomposition caused by the deposition of straw in the bays with wild rice. Wild rice may also impede water flow and restrict dissolved oxygen from diffusing into the water column. Future research should continue to monitor dissolved oxygen levels to determine whether minimal threshold levels are exceeded for fish and invertebrates during the summer or winter months.

Results from the current study show that wild rice may provide an energy source for the invertebrates and fish within these lakes as both invertebrates and fish $\delta^{13}C$ ratios overlapped with that of the introduced plant. Although it is likely that some organisms fed directly on wild rice, it was not the only source of carbon in these bays; many invertebrates appeared to rely on another, more depleted, source of carbon not collected in this study (Figure 3.12 – 3.14). This alternate source of carbon is likely to be periphyton, epiphytic algae, fine particulate organic matter or detritus and debris within the water column and on the substrate. Native macrophytes may have also provided an energy source for the biota; however, since the range of $\delta^{13}C$ of the invertebrates and fish overlapped only slightly with that of the native macrophytes, they did not appear to be a primary source of carbon for these food webs. Pelagic zooplankton also did not appear to provide a primary food source for the fish or invertebrates (except for shiners) as $\delta^{13}C$ values for most fish species did not overlap with values for these invertebrates.

Although I found some differences between bays with wild rice and with native macrophytes, this study was limited to three oligotrophic lakes in west-central Manitoba that may not be representative of other regions seeded with wild rice. Future research should be conducted on eutrophic lakes to assess the potential impacts of introduced wild rice on more productive systems. More productive systems may support higher stand densities of wild rice, which may in turn have greater impacts on fish movement and foraging efficiencies. The stand densities in Barry, Cacholotte and Naosap lakes were at the lower range observed for wild rice and, for this reason, there may have been fewer impacts of this macrophyte on food web structure at these sites (Lavergne, 2005).

It is also important to consider how the timing of wild rice introductions may impact aquatic ecosystems. The wild rice bays in the current study were seeded in the early 1980s and have had over two decades to adjust to this introduction. The effects of wild rice on food webs likely change over time as the wild rice stand matures. It would be ideal to study natural lakes before and after the introduction of wild rice to better understand the changes this plant causes within the native food web, and how these impacts change over time.

In summary, this study has provided important and much-needed information about the relationships between wild rice and invertebrate and fish communities. No other study has contrasted invertebrate communities or food web structures between bays with wild rice and bays dominated by the native vegetation. Future studies are needed to learn how wild rice impacts biota in other regions and to expand our knowledge about the complex food web interactions between invertebrate and fish communities within wild rice bays.

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Appendix I

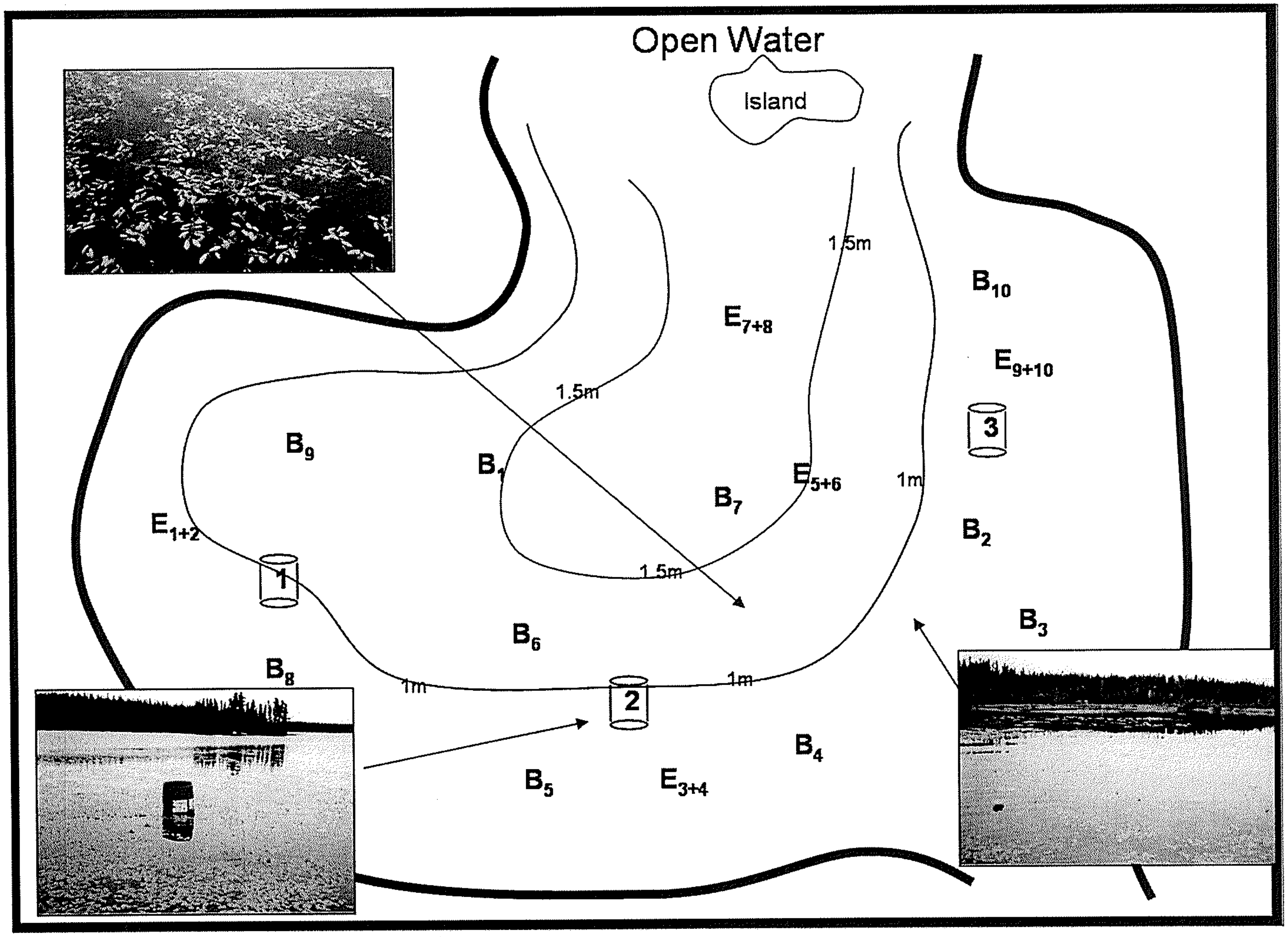

Figure A1.01 Diagram of Barry Lake native macrophyte bay, 2003. Symbols; B represents the bottle trap samples; E represent the emergence trap samples; and the bucket numbered 1-3 represent the three bucket samples.

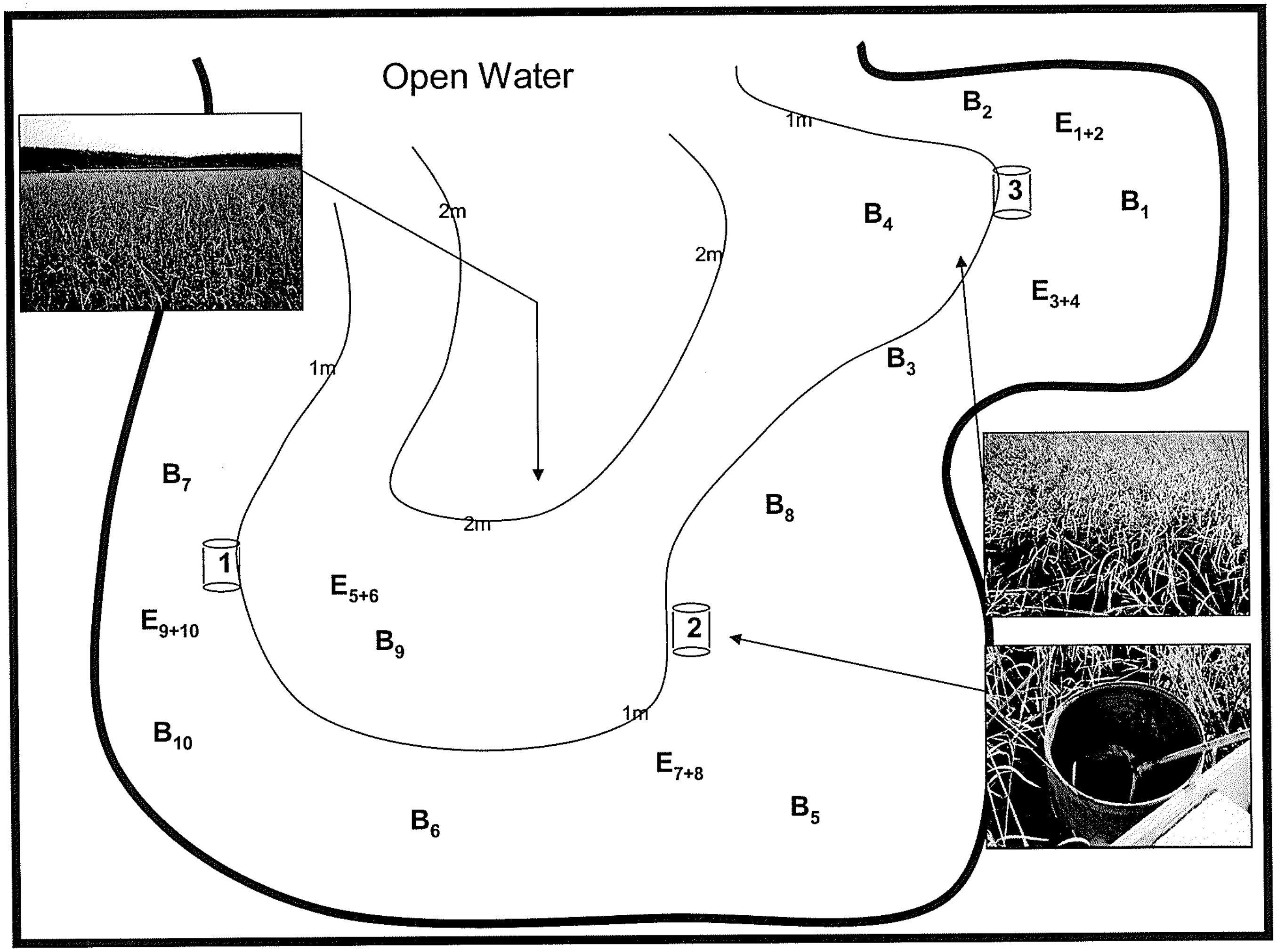

Figure A1.02 Diagram of Barry Lake wild rice bay, 2003. Symbols; B represents the bottle trap samples; E represent the emergence trap samples; and the bucket numbered 1-3 represent the three bucket samples.

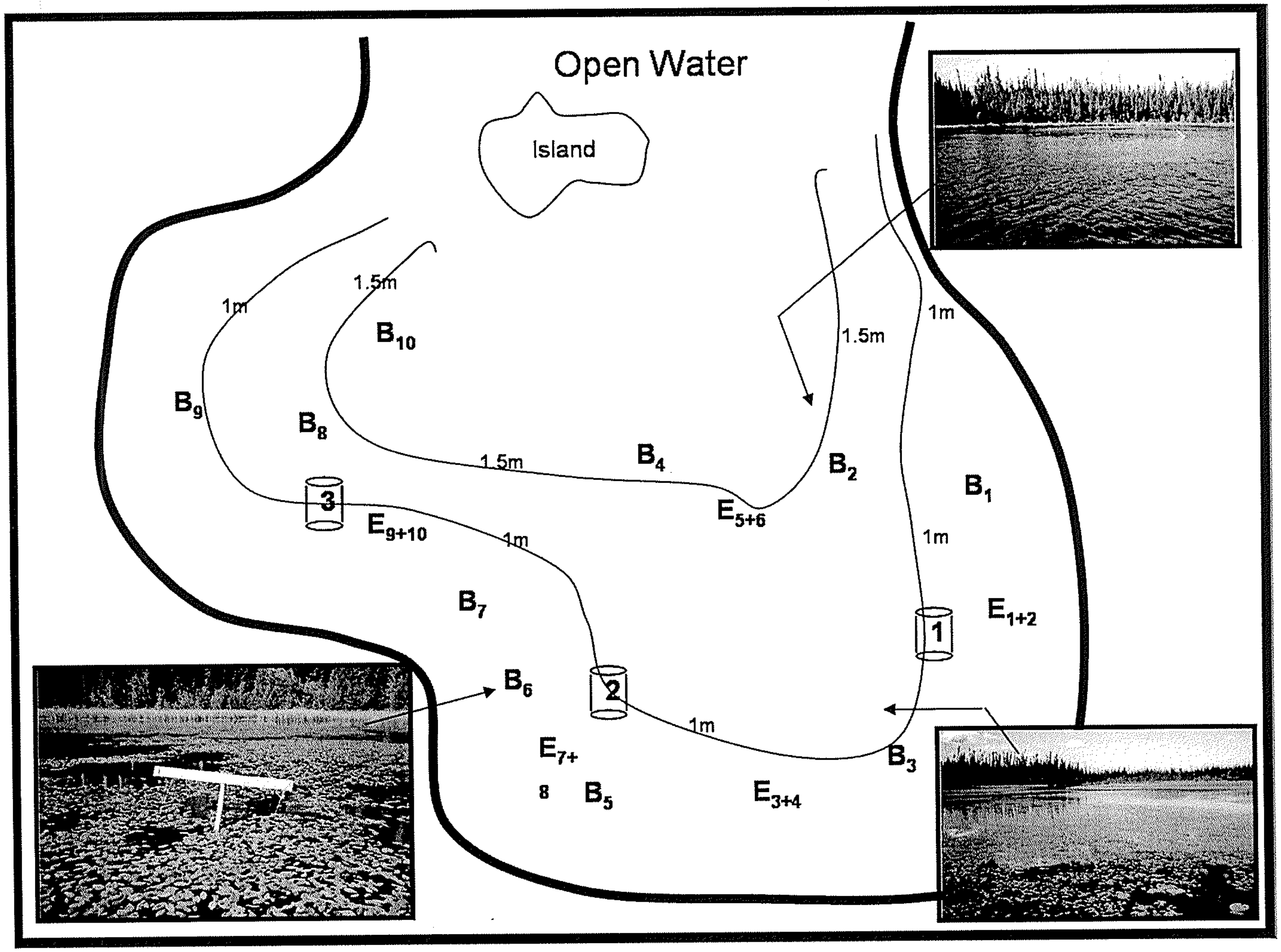

Figure A1.03 Diagram of Cacholotte Lake native macrophyte bay, 2003. Symbols; B represents the bottle trap samples; E represent the emergence trap samples; and the bucket numbered 1-3 represent the three bucket samples.

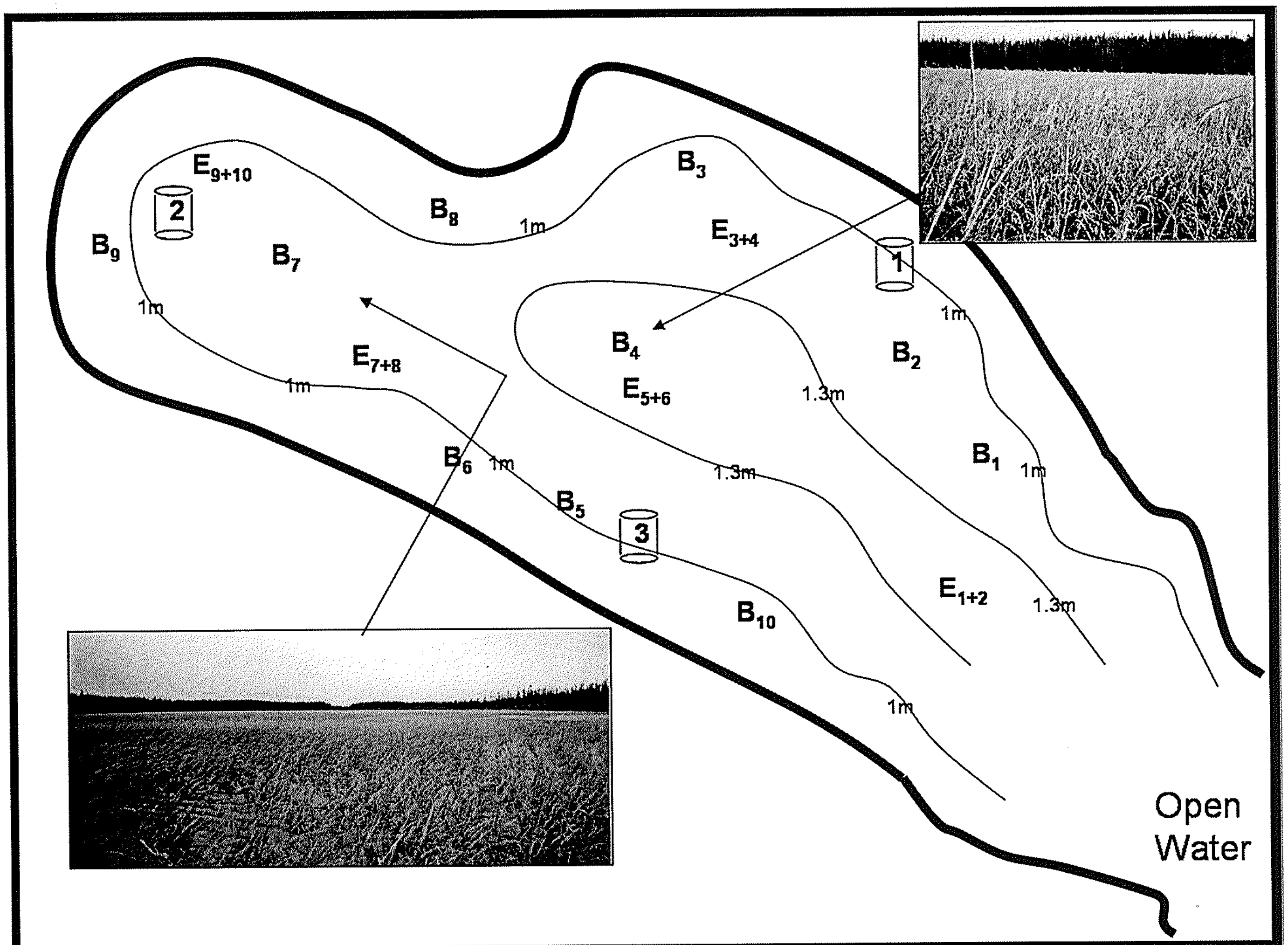

Figure A1.04 Diagram of Cacholotte Lake wild rice bay, 2003. Symbols; B represents the bottle trap samples; E represent the emergence trap samples; and the bucket numbered 1-3 represent the three bucket samples.

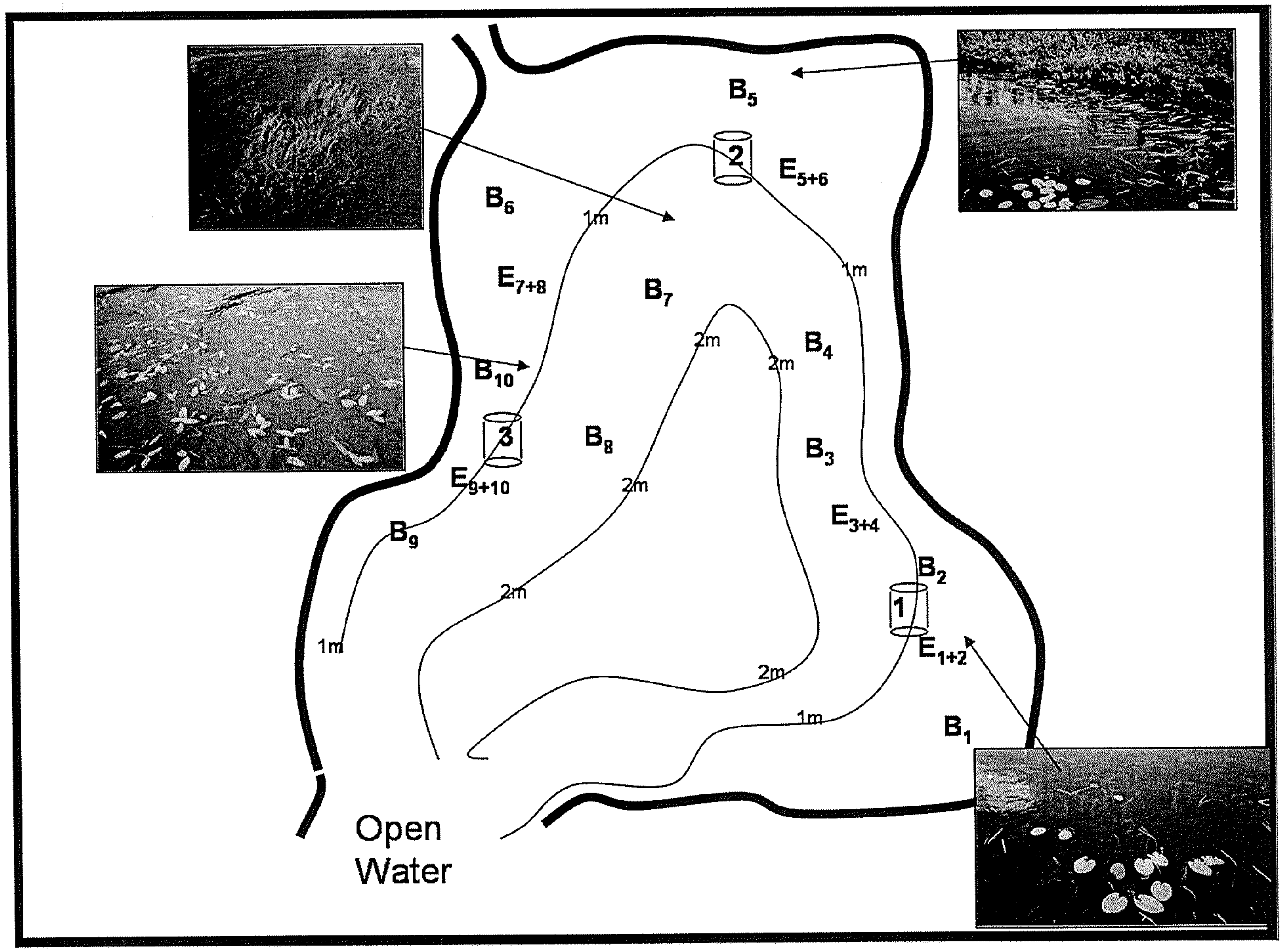

Figure A1.05 Diagram of Naosap Lake native macrophyte bay, 2003. Symbols; B represents the bottle trap samples; E represent the emergence trap samples; and the bucket numbered 1-3 represent the three bucket samples.

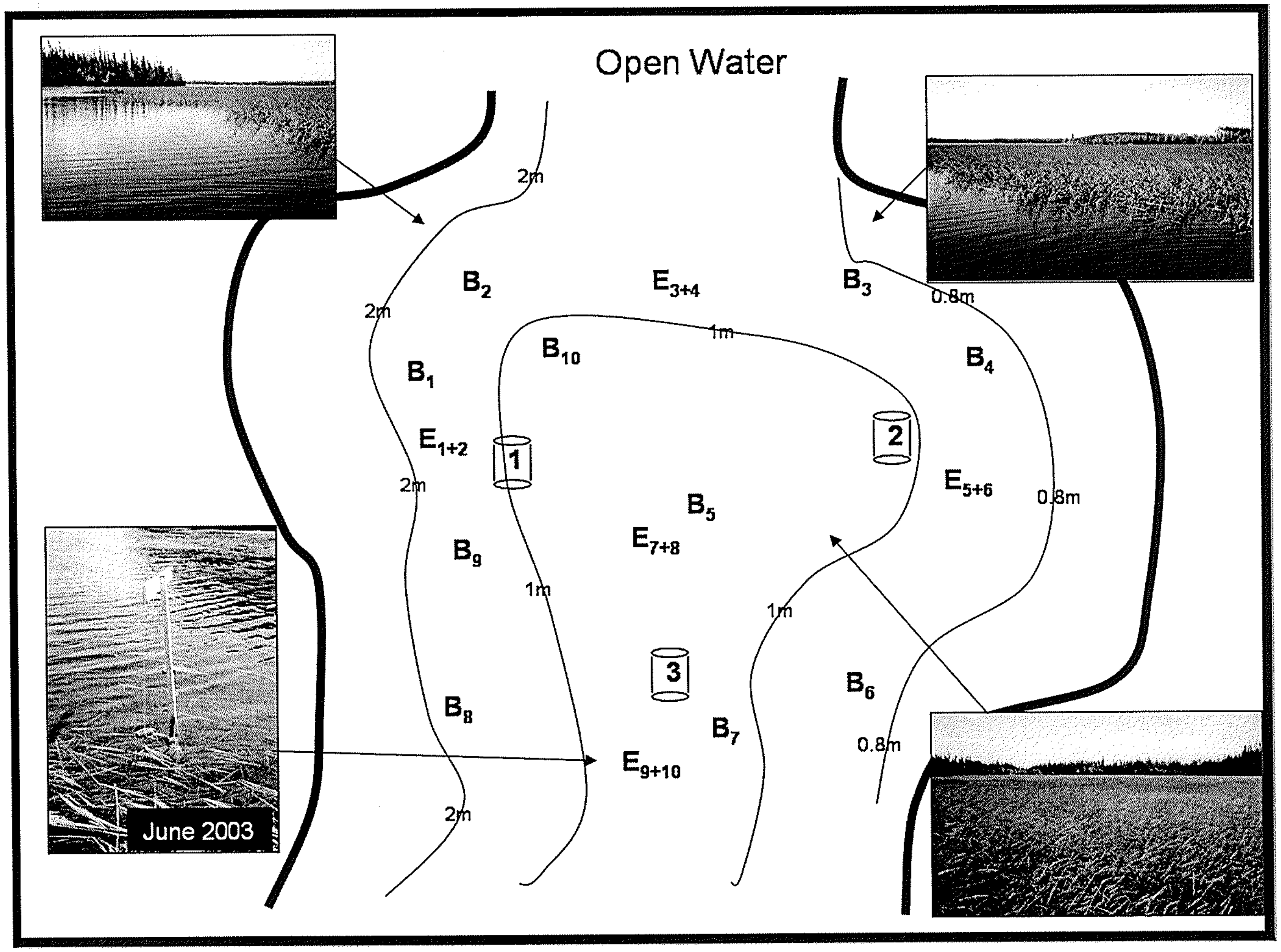

Figure A1.06 Diagram of Naosap Lake wild rice bay, 2003. Symbols; B represents the bottle trap samples; E represent the emergence trap samples; and the bucket numbered 1-3 represent the three bucket samples.

Table A1.07. The raw data for the invertebrates collected using bottle traps for Barry Lake in June of 2003

Site	Trap	Caenidae	Baetidae	Siphlonuridae	Chironomidae (herbivores)	Amphipoda	Oligochaeta	Aeshnidae	Corixidae	Notonectidae	Dytiscidae	Chironomidae (predators)	Sciomizidae	Ceratopogonidae	Glossiphoniidae	Erpobdellidae	Water mites	Chaoboridae	Libellulidae	Araneae	Leptoceridae	Ostracoda	Haliplidae	Culicidae	Sphaeridae	Microturbellaria	Lymnaeidae	Hydrobiidae	Physidae	Planorbidae	Valvatidae
Barry (Native Macrophyte bay)	1	0	0	0	3	15	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	4	0	0	0	1	4	0	0	0	0
	2	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0	10	0	0	0	0	0	5	0	0	2	2	0	0	1	0
	3	0	0	0	4	1	0	0	1	0	0	1	0	0	0	0	31	0	0	1	0	1	1	0	0	1	6	0	0	1	0
	4	1	0	0	1	0	0	0	0	0	2	1	0	0	0	0	29	0	0	0	2	5	0	0	0	0	0	1	1	0	0
	5	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	14	0	0	0	0	1	1	0	0	1	0	0	0	0	0
	6	0	0	0	3	3	0	0	1	0	0	0	0	0	0	0	37	0	0	0	0	2	0	0	0	4	4	0	1	1	0
	7	0	0	1	4	3	0	0	3	0	0	0	0	0	0	0	19	0	0	0	0	5	24	0	0	0	2	0	0	0	0
	8	0	0	0	3	1	0	0	0	0	0	0	0	0	0	0	16	1	0	0	0	3	8	0	0	0	5	0	0	0	0
	9	0	0	0	7	0	0	0	0	0	0	3	0	0	0	1	28	0	0	0	0	0	1	0	0	3	12	0	0	0	0
	10	0	0	0	4	0	0	0	2	0	0	1	0	0	0	0	31	0	0	0	0	4	12	0	0	0	0	0	0	0	0
Total #'s		1	0	1	31	23	0	0	8	0	3	6	0	0	0	1	220	1	0	1	2	25	52	0	0	12	35	1	2	3	0
Barry (Wild rice bay)	1	0	0	0	24	1	2	0	3	0	2	5	0	3	1	0	55	0	1	0	0	14	0	1	0	1	2	0	1	5	0
	2	0	0	1	29	2	1	1	7	1	0	3	1	2	0	0	55	0	0	0	0	10	0	0	0	8	0	0	4	0	0
	3	0	0	2	7	20	1	0	4	0	0	1	0	0	0	0	125	0	0	0	0	8	0	0	0	4	1	0	0	0	0
	4	0	0	1	8	4	3	1	0	0	0	1	0	0	0	0	65	0	0	0	0	4	0	0	0	3	3	0	5	0	0
	5	0	0	2	3	4	3	0	7	0	0	3	0	0	2	0	11	1	0	0	0	7	0	0	0	4	5	1	6	0	0
	6	0	0	0	2	0	0	0	0	1	0	0	0	0	0	0	117	0	0	0	0	0	0	0	0	1	3	0	0	0	0
	7	0	1	0	2	6	2	0	8	0	2	1	0	0	0	0	54	1	0	0	0	7	1	0	0	1	7	0	0	0	0
	8	0	0	0	7	4	3	1	10	0	0	2	0	0	0	0	83	0	0	0	0	16	0	0	0	0	6	0	3	3	2
	9	0	0	0	8	5	0	0	4	0	0	0	0	0	0	0	47	3	0	0	0	5	0	0	0	0	3	0	0	0	0
	10	0	0	0	3	6	2	0	2	0	0	6	0	0	0	0	80	0	0	0	0	2	0	0	1	1	0	0	4	1	0
Total #'s		0	1	6	93	52	17	3	45	2	4	22	1	5	3	0	692	5	1	0	0	73	1	1	1	23	30	1	23	9	2

Table A1.08 The raw data for the invertebrates collected using bottle traps for Cacholotte Lake in June of 2003

		Caenidae	Baetidae	Siphlonuridae	Chironomidae (herbivores)	Amphipoda	Plecoptera	Muscidae	Ephydridae	Oligochaeta	Coenagrionidae	Corixidae	Dytiscidae	Polycentripodidae	Chironomidae (predators)	Ceratopogonidae	Glossiphoniidae	Erpobdellidae	Water mites	Chaoboridae	Araneae	Gyrinidae	Leptoceridae	Ostracoda	Pyralidae	Haliplidae	Sphaeridae	Braconidae	Eulophidae	Microturbellaria	Piscicolidae	Nematopmorpha	Hydrobiidae	Lymnaeidae	Physidae	Planorbidae	Valvatidae
Cacholotte (Native Macrophyte bay)	1	1	1	0	19	6	0	0	0	3	0	3	0	0	3	0	0	0	26	0	0	0	1	8	0	5	0	0	0	0	0	0	17	0	0	1	0
	2	0	0	0	5	0	0	0	0	0	0	15	0	0	0	0	0	0	10	0	0	0	0	15	0	1	0	0	0	0	0	0	1	0	0	0	0
	3	0	0	0	14	2	0	0	0	1	0	31	1	0	2	0	0	0	69	0	0	1	2	41	0	3	0	0	0	0	0	0	2	0	0	0	0
	4	4	0	0	9	0	0	0	0	0	1	21	0	0	1	0	0	0	51	0	0	0	0	9	0	2	0	0	0	0	0	0	6	0	0	0	0
	5	0	1	0	7	0	1	0	0	0	0	22	0	1	0	1	0	0	85	0	0	1	0	13	1	8	0	0	0	0	0	1	2	0	0	0	0
	6	1	0	0	9	2	0	0	0	11	1	6	1	0	1	0	6	0	84	0	0	1	1	35	0	8	0	0	0	0	0	0	1	0	0	1	0
	7	0	0	0	4	5	0	0	0	3	0	10	0	0	0	0	0	0	171	0	0	0	0	18	0	6	0	0	0	0	0	0	6	0	1	0	1
	8	0	0	0	6	6	0	0	0	2	0	3	1	0	0	0	3	0	108	2	0	3	0	15	0	2	0	0	0	0	0	0	1	0	0	1	0
	9	1	1	0	1	4	0	0	0	14	0	17	0	0	0	0	1	0	82	0	0	2	0	8	0	5	0	0	0	0	0	0	6	1	0	0	0
	10	0	0	0	7	0	0	0	0	12	0	7	0	0	1	0	1	0	62	0	0	0	1	14	0	5	0	0	0	0	0	0	4	0	1	0	0
Total #'s		7	3	0	81	25	1	0	0	46	2	135	3	1	8	1	11	0	748	2	0	8	5	176	1	45	0	0	0	0	0	1	46	1	2	3	1
Cacholotte (Wild rice bay)	1	1	0	0	5	9	0	0	0	0	1	7	0	0	0	0	0	0	45	0	0	2	0	1	0	6	0	0	0	0	0	0	2	0	0	0	0
	2	2	0	0	6	3	0	1	1	8	0	4	0	0	6	0	2	0	86	1	0	0	0	9	0	0	0	1	0	0	1	0	11	0	0	0	0
	3	1	0	0	2	3	0	0	0	3	0	0	3	0	1	0	0	0	25	0	1	0	0	1	0	0	0	0	0	0	0	0	4	0	0	0	0
	4	0	0	0	2	0	0	0	1	15	0	3	0	0	1	0	0	0	97	0	0	1	0	1	0	0	0	0	0	44	0	0	2	0	1	0	0
	5	0	0	0	15	2	0	0	0	23	0	5	0	0	1	0	1	0	95	2	0	0	0	1	0	0	0	0	0	7	0	0	1	2	2	0	0
	6	0	0	0	6	10	0	0	0	33	0	12	0	0	3	0	2	3	155	11	0	0	0	8	0	0	0	0	0	11	0	0	4	0	2	0	0
	7	0	0	0	8	3	0	0	0	38	0	5	0	0	2	0	3	0	132	6	0	0	1	10	0	0	1	0	0	9	0	0	14	0	10	0	0
	8	5	0	1	1	4	0	0	0	8	0	3	0	0	0	0	1	1	53	0	1	0	0	3	1	0	0	0	0	4	0	1	6	0	1	0	0
	9	0	0	0	4	10	0	0	0	20	0	2	0	0	3	0	2	0	38	0	0	0	0	2	0	0	0	0	1	16	0	0	15	0	5	0	1
	10	1	0	0	1	7	0	0	0	9	0	4	0	0	0	0	1	0	38	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Total #'s		10	0	1	50	51	0	1	2	157	1	45	3	0	17	0	12	4	764	20	2	3	1	37	1	6	1	1	1	92	1	1	59	2	21	0	1

Table A1.09. The raw data for the invertebrates collected using bottle traps for Naosap Lake in June of 2003

Site	Trap	Caenidae	Baetidae	Limnephilidae	Siphlonuridae	Chironomidae (herbivores)	Amphipoda	Plecoptera	Oligochaeta	Coenagrionidae	Corixidae	Dytiscidae	Polycentripodidae	Chironomidae (predators)	Tabanidae	Ceratopogonidae	Glossiphoniidae	Erpobdellidae	Water mites	Chaoboridae	Araneae	Gyrinidae	Leptoceridae	Ostracoda	Chrysomelidae	Haliplidae	Braconidae	Microturbellaria	Hydrobiidae	Lymnaeidae	Physidae	Planorbidae	Valvatidae
Naosap (Native Macrophyte bay)	1	0	0	0	0	3	1	0	1	0	5	2	0	1	0	0	1	0	75	0	0	0	0	0	0	1	0	3	0	0	1	0	0
	2	2	1	0	0	4	1	0	4	0	1	0	0	0	0	0	0	0	46	0	0	0	0	1	0	7	0	0	0	0	0	0	0
	3	0	0	1	0	2	0	0	0	0	0	0	0	0	0	0	0	0	13	0	0	0	0	0	0	9	0	0	1	0	0	0	0
	4	0	0	0	0	3	0	0	2	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	2	0	0	0	0	0	0	0
	5	2	0	0	1	0	0	0	1	0	0	0	0	1	0	1	0	0	4	0	0	0	0	0	0	2	0	0	0	0	0	0	0
	6	0	0	0	0	3	5	0	0	0	0	0	0	0	0	0	0	0	13	0	0	0	0	0	0	2	0	0	0	0	1	0	0
	7	1	0	0	1	1	0	0	0	0	1	0	0	1	0	0	0	0	14	0	0	1	0	1	0	3	0	0	0	0	0	0	0
	8	0	0	0	1	7	1	0	1	0	1	0	0	0	0	0	0	0	8	0	0	0	0	0	0	1	0	0	1	0	0	0	0
	9	9	0	0	0	11	0	0	1	1	1	0	0	0	0	0	1	0	22	0	0	1	0	1	0	10	0	0	0	0	0	0	0
	10	1	2	0	0	2	3	0	14	0	0	2	0	0	0	1	0	0	50	0	0	1	0	0	0	21	0	0	0	0	0	0	0
Total #'s		15	3	1	3	36	11	0	24	1	10	4	0	3	0	2	2	0	246	0	0	3	0	3	0	58	0	3	2	0	2	0	0
Naosap (Wild Rice bay)	1	0	0	0	0	4	18	0	14	1	6	2	0	4	0	0	0	0	64	0	0	0	0	6	0	0	0	0	0	0	8	0	0
	2	2	0	0	0	2	4	1	2	0	1	0	0	3	0	1	0	0	39	0	0	0	0	11	0	0	0	1	0	0	3	1	0
	3	0	0	0	0	3	14	0	2	0	0	2	0	3	0	0	0	0	40	0	0	1	0	6	0	0	0	1	0	0	0	2	0
	4	0	0	0	0	5	38	0	3	0	1	5	0	6	0	0	1	0	76	0	0	0	0	157	0	0	0	2	0	0	2	0	0
	5	0	6	0	0	3	8	0	3	0	3	6	0	2	0	0	0	0	56	0	0	0	3	27	2	0	0	2	0	0	0	0	0
	6	0	1	0	0	2	4	0	25	0	10	1	0	1	0	0	1	1	41	0	0	0	1	73	0	1	0	10	2	0	7	0	1
	7	0	0	0	0	2	3	0	12	0	30	0	0	6	0	0	0	0	46	0	1	1	6	18	0	0	1	3	0	0	8	0	0
	8	0	0	0	0	13	3	0	79	11	11	0	1	11	0	0	0	0	50	1	0	0	0	32	0	0	0	4	0	1	7	0	0
	9	0	0	0	0	9	3	0	66	0	3	0	0	6	1	0	1	0	114	0	0	0	0	32	0	0	0	9	1	1	3	2	0
	10	1	2	0	0	15	2	0	37	0	1	0	0	2	0	0	0	0	187	0	0	0	0	57	0	0	0	1	4	0	9	0	0
Total #'s		3	9	0	0	58	97	1	243	12	66	16	1	44	1	1	3	1	713	1	1	2	10	419	2	1	1	33	7	2	47	5	1

Table A1.10. The raw data for the invertebrates collected using bottle traps for Barry Lake in August of 2003

		Caenidae	Baetidae	Ephemerellidae	Limnephilidae	Chironomidae (herbivores)	Amphipoda	Ephydridae	Cyclorrhaphous-Brachycera	Oligochaeta	Coenagrionidae	Aeshnidae	Corixidae	Notonectidae	Polycentripodidae	Chironomidae (predators)	Ceratopogonidae	Glossiphoniidae	Erpobdellidae	Water mites	Chaoboridae	Libelluliidae	Araneae	Gyrinidae	Leptoceridae	Ostracoda	Haliplidae	Culicidae	Sphaeridae	Nematoda	Microturbellaria	Scelionidae	Hydrobiidae	Lymnaeidae	Physidae	Ancylidae	Planorbidae	Valvatidae
Barry (Native Macrophyte bay)	1	0	0	0	0	3	7	0	0	10	0	0	4	0	1	3	0	1	0	19	0	0	0	1	0	20	0	0	0	1	0	0	0	0	0	0	0	0
	2	0	0	0	0	5	8	0	0	2	0	0	2	1	2	1	0	1	0	55	0	0	1	1	1	16	0	0	0	0	0	1	4	0	0	0	0	1
	3	0	0	0	0	1	14	0	0	0	0	0	9	0	0	1	0	0	0	76	4	0	0	0	1	36	0	0	0	0	2	0	0	0	2	0	1	0
	4	1	1	0	1	4	9	0	0	4	0	0	3	0	0	1	0	0	0	42	0	1	0	0	0	3	0	0	0	0	1	0	6	0	1	0	1	0
	5	0	0	0	0	3	4	0	0	2	1	0	1	0	0	0	0	0	0	16	0	1	0	1	0	15	0	0	0	0	1	0	4	1	0	0	0	0
	6	0	0	0	0	6	3	0	0	6	1	0	1	0	0	0	1	1	0	43	0	0	0	0	0	13	0	0	1	0	1	1	2	1	0	0	0	0
	7	0	1	0	0	2	3	0	0	0	0	0	4	0	1	0	0	0	1	57	1	1	1	0	0	27	0	0	0	0	1	0	3	0	1	0	0	0
	8	1	1	0	0	7	5	0	0	0	0	0	6	0	1	6	0	1	1	56	1	0	0	0	0	22	0	1	0	0	4	0	4	0	2	0	1	1
	9	1	0	0	0	3	5	1	0	7	0	0	1	0	0	0	0	0	0	30	0	0	0	0	0	6	0	0	0	0	0	0	1	2	2	0	0	3
	10	0	1	1	0	3	7	0	0	0	0	0	1	0	0	1	0	0	0	11	0	1	1	0	0	14	0	0	0	0	0	0	0	0	2	0	0	0
Total #'s		3	4	1	1	37	65	1	0	31	2	0	32	1	5	13	1	4	2	405	6	4	3	3	2	172	0	1	1	1	10	2	24	4	10	0	3	5
Barry (Wild Rice bay)	1	3	1	0	0	7	0	0	0	0	0	0	5	0	0	1	0	0	0	25	0	0	0	0	0	18	0	0	0	0	0	0	1	1	1	0	0	0
	2	0	1	0	1	8	4	0	0	1	0	0	0	1	0	5	0	0	0	80	0	1	1	0	0	23	0	0	0	0	2	0	4	0	0	0	0	0
	3	2	0	0	0	4	17	0	0	0	0	0	9	1	0	2	1	0	0	97	0	0	0	0	1	33	0	0	0	0	2	2	2	0	1	0	0	1
	4	0	0	0	0	0	10	0	0	0	1	0	3	0	0	1	0	0	0	59	0	0	0	0	1	15	0	0	0	0	0	1	0	0	1	0	0	1
	5	1	0	1	0	3	6	0	0	2	0	0	0	0	0	0	0	0	0	13	0	0	0	0	0	7	0	0	0	0	0	2	3	0	1	0	1	0
	6	1	0	0	0	2	7	0	0	0	0	0	4	0	0	2	0	0	0	38	0	0	0	0	1	12	0	0	0	1	3	1	0	0	0	0	1	0
	7	0	1	0	0	2	6	0	1	0	0	0	3	0	0	0	0	0	0	34	0	0	0	0	0	35	4	0	0	0	4	1	1	0	3	1	0	0
	8	0	1	0	0	6	4	0	0	4	0	0	2	0	0	1	0	0	0	84	0	0	0	0	0	17	0	0	0	0	3	0	3	0	0	0	0	0
	9	1	0	0	0	7	6	0	0	1	0	0	1	0	0	0	0	0	0	21	0	0	0	0	0	8	0	0	0	0	0	0	3	0	1	0	0	0
	10	0	1	0	0	1	5	0	0	0	0	1	0	0	0	1	0	0	0	27	0	0	0	0	0	34	0	0	0	0	2	0	0	0	1	0	0	0
Total #'s		8	5	1	1	40	65	0	1	8	1	1	27	2	0	13	1	0	0	478	0	1	1	0	3	202	4	0	0	1	16	7	17	1	9	1	2	2

Table A1.11. The raw data for the invertebrates collected using bottle traps for Cacholotte Lake in August of 2003

Site	Trap	Collembola	Caenidae	Baetidae	Chironomidae (herbivores)	Ephydridae	Amphipoda	Oligochaeta	Coenagrionidae	Corixidae	Notonectidae	Dytiscidae	Polycentripodidae	Chironomidae (predators)	Ceratopogonidae	Glossiphoniidae	Water mites	Chaoboridae	Araneae	Leptoceridae	Ostracoda	Pyralidae	Haliplidae	Eulophidae	Microturbellaria	Hydrobiidae	Lymnaeidae	Physidae	Planorbidae	Valvatidae
Cacholotte (Native Macrophyte bay)	1	0	0	0	3	0	1	6	0	9	0	0	0	2	0	0	13	0	0	0	2	0	0	0	0	1	0	0	0	0
	2	0	0	0	5	0	0	7	0	0	0	0	0	0	0	0	14	0	0	0	5	0	8	0	2	8	0	0	0	0
	3	0	0	0	3	0	1	16	0	1	0	0	0	4	1	2	12	0	0	0	4	0	4	0	0	2	0	0	0	0
	4	0	2	0	6	0	2	0	0	0	0	0	0	0	1	0	38	0	0	0	6	0	1	0	0	0	0	0	0	0
	5	0	0	0	3	0	2	2	1	2	0	0	0	1	0	0	13	0	0	0	8	0	0	0	0	0	3	0	0	0
	6	0	0	1	3	0	0	0	0	0	0	0	0	0	0	0	49	1	0	0	1	0	1	0	0	0	0	0	0	0
	7	0	0	0	21	0	2	10	1	4	0	0	0	0	0	0	15	0	0	0	4	0	0	0	2	1	0	0	1	0
	8	0	0	0	12	0	1	19	0	0	0	0	0	1	0	0	20	0	0	0	6	0	0	0	0	0	2	2	0	0
	9	0	0	0	3	0	3	4	0	1	0	0	0	0	0	0	60	0	0	1	2	0	12	0	1	0	0	0	0	0
	10	0	0	0	13	0	14	1	0	1	0	0	0	1	0	0	38	0	0	0	1	0	1	0	0	2	1	1	0	0
Total #'s		0	2	1	72	0	26	65	2	18	0	0	0	9	2	2	272	1	0	1	39	0	27	0	5	14	6	3	1	0
Cacholotte (Wild Rice bay)	1	0	0	0	2	0	1	6	0	0	0	0	0	1	0	0	23	1	0	0	15	0	2	0	0	0	0	1	1	0
	2	0	0	0	1	0	4	2	1	1	0	0	0	1	0	0	30	1	0	0	7	3	0	1	0	1	0	1	0	1
	3	0	0	0	2	0	3	1	0	0	2	0	0	0	0	0	13	0	1	0	1	0	0	0	0	1	0	0	0	2
	4	0	0	0	0	0	4	1	0	0	0	1	2	0	0	0	31	4	0	0	23	0	0	0	0	0	0	1	0	0
	5	0	0	0	1	0	1	1	0	1	0	0	1	2	0	0	26	0	0	0	6	0	0	0	0	0	0	0	0	0
	6	0	0	0	4	0	2	2	0	3	0	0	0	0	0	1	15	1	0	0	10	0	0	1	0	0	0	0	0	0
	7	0	0	0	2	0	1	5	0	1	0	0	0	0	0	0	24	1	0	0	12	0	0	0	0	0	0	0	0	0
	8	1	0	0	5	0	1	2	0	2	0	0	0	2	0	0	30	0	0	0	11	0	0	0	0	1	0	2	0	0
	9	0	0	0	0	0	4	1	0	2	0	0	0	0	0	1	39	1	0	0	10	0	0	0	0	0	1	2	0	0
	10	0	0	0	0	2	14	2	0	0	0	0	0	0	0	0	14	1	0	0	22	0	0	0	0	0	0	0	1	0
Total #'s		1	0	0	17	2	35	23	1	10	2	1	3	6	0	2	245	10	1	0	117	3	2	2	0	3	1	7	2	3

Table A1.12. The raw data for the invertebrates collected using bottle traps for Naosap Lake in August of 2003

		Caenidae	Baetidae	Leptophlebidae	Chironomidae (herbivores)	Amphipoda	Oligochaeta	Coenagrionidae	Corixidae	Notonectidae	Polycentripodidae	Chironomidae (predators)	Glossiphoniidae	Water mites	Chaoboridae	Araneae	Gyrinidae	Leptoceridae	Ostracoda	Curculionidae	Haliplidae	Microturbellaria	Lymnaeidae	Physidae	Planorbidae	Valvatidae
Naosap (Native Macrophyte bay)	1	0	1	1	0	11	2	0	0	0	0	0	0	14	0	0	0	0	6	0	1	0	0	0	0	0
	2	0	0	0	3	3	6	0	0	0	0	1	0	18	0	0	0	0	2	0	0	0	0	0	0	0
	3	0	0	1	1	0	0	0	0	0	0	0	0	4	0	0	0	0	3	0	0	0	0	0	0	0
	4	0	0	0	0	0	4	0	0	0	0	0	0	3	0	1	0	0	0	0	0	0	0	1	0	0
	5	0	1	6	1	9	4	0	0	0	0	0	0	10	0	0	1	0	1	0	1	0	0	2	0	0
	6	0	2	0	0	13	2	0	0	0	0	0	2	11	0	0	1	0	5	1	0	0	0	0	0	0
	7	0	0	0	3	10	2	0	0	0	0	0	0	2	0	0	0	0	1	0	1	0	0	0	1	0
	8	0	0	0	3	9	2	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
	9	0	0	1	0	4	3	0	0	0	0	0	0	12	0	0	0	0	0	0	2	0	0	0	0	0
	10	0	0	0	1	1	0	0	0	0	0	0	0	3	0	1	0	0	1	0	2	0	0	0	0	0
Total #'s		0	4	9	12	60	25	0	0	0	0	1	2	77	0	2	2	2	19	1	7	0	0	3	1	0
Naosap (Wild Rice bay)	1	0	0	0	1	4	4	0	1	0	0	0	0	7	0	0	0	1	6	0	0	5	0	0	0	0
	2	0	0	0	1	4	1	0	0	0	0	0	0	4	0	0	0	0	2	0	0	3	1	0	1	0
	3	0	0	0	0	21	6	0	2	0	0	0	0	14	3	0	0	0	9	0	0	2	0	0	1	0
	4	1	1	0	0	22	0	1	1	0	0	0	0	14	0	0	0	1	15	0	0	3	0	0	3	4
	5	0	0	0	1	0	2	0	1	1	0	1	0	6	0	0	0	0	28	0	0	0	1	0	0	0
	6	0	1	0	2	14	0	0	1	1	1	2	0	14	0	0	0	1	14	0	0	0	1	0	1	1
	7	0	0	0	5	5	4	0	2	0	1	1	0	16	0	0	0	1	11	0	1	0	0	0	0	0
	8	2	0	0	3	14	0	0	1	0	0	1	0	23	0	0	0	1	11	0	0	0	0	0	0	0
	9	0	0	0	2	6	0	0	0	0	0	0	0	15	0	0	0	1	5	0	0	0	1	0	0	0
	10	1	0	0	2	3	3	0	1	0	0	1	0	8	0	0	0	0	6	0	0	1	1	1	0	0
Total #'s		4	2	0	17	93	20	1	10	2	2	6	0	121	3	0	0	6	107	0	1	14	5	1	6	5

Table A1.13. The raw data for the invertebrates collected using emergence traps for Barry Lake in June of 2003

		Caenidae	Leptophlebidae	Chironomidae (herbivores)	Chironomidae (predatorss)	Culicidae	Simulidae
Barry Lake (Native Macrophytes)	1	0	0	3	2	0	0
	2	0	1	7	0	0	0
	3	0	0	4	1	0	1
	4	0	0	3	0	0	0
	5	0	0	3	1	0	0
	6	0	0	2	2	0	0
	7	0	0	8	1	0	0
	8	0	0	9	0	0	0
	9	0	0	9	3	0	0
	10	0	0	11	6	0	0
Total #'s		0	1	59	16	0	1
Barry Lake (Wild Rice)	1	2	0	5	5	0	0
	2	4	0	21	8	1	0
	3	0	0	6	12	0	0
	4	0	0	1	5	0	0
	5	0	0	3	4	0	0
	6	0	0	5	0	0	0
	7	0	0	21	4	0	0
	8	0	0	11	2	0	0
	9	0	0	13	9	0	0
	10	0	0	3	6	0	0
Total #'s		6	0	89	55	1	0

Table A1.14. The raw data for the invertebrates collected using emergence traps for Cacholotte Lake in June of 2003

		Caenidae	Limnephilidae	Hydroptilidae	Chironomidae (herbivores)	Coenagrionidae	Tapulidae	Chironomidae (predatorss)
Cacholotte Lake (Native Macrophytes)	1	0	0	0	6	0	0	5
	2	0	0	2	9	1	0	4
	3	0	0	0	17	0	0	4
	4	0	0	0	32	0	0	4
	5	0	0	0	5	2	0	5
	6	0	0	0	12	0	0	7
	7	0	0	0	20	0	0	10
	8	0	0	0	7	0	0	5
	9	0	2	0	6	0	1	1
	10	0	0	0	6	0	0	1
Total #'s		0	2	2	120	3	1	46
Cacholotte Lake (Wild Rice)	1	0	0	0	12	1	0	4
	2	0	0	0	7	0	0	2
	3	4	0	0	15	0	0	4
	4	2	1	0	16	0	0	2
	5	0	0	0	3	0	0	0
	6	0	0	0	5	0	0	0
	7	1	1	0	3	0	0	0
	8	1	0	0	5	0	0	0
	9	0	0	0	10	0	0	1
	10	3	1	0	8	0	0	0
Total #'s		11	3	0	84	1	0	13

Table A1.15. The raw data for the invertebrates collected using emergence traps for Naosap Lake in June of 2003

		Caenidae	Baetidae	Limnephilidae	Hydroptilidae	Chironomidae (herbivores)	Cyclorrhaphous-Brachycera	Coenagrionidae	Ceratopogonidae	Chironomidae (predators)	Phryganeidae	Braconidae
Naosap Lake (Native Macrophytes)	**1**	0	0	1	0	1	0	0	1	1	0	0
	2	1	0	1	0	18	0	0	1	1	0	0
	3	9	0	0	0	39	0	0	1	3	0	0
	4	2	0	0	0	21	0	0	1	4	0	0
	5	0	0	0	0	22	0	0	0	1	0	0
	6	0	0	0	0	8	0	0	0	2	0	0
	7	8	0	0	0	35	0	0	0	4	0	0
	8	6	0	0	0	41	0	1	0	4	0	0
	9	36	0	1	0	42	0	0	0	2	0	0
	10	20	0	0	1	24	0	0	0	1	0	0
Total #'s		82	0	3	1	251	0	1	4	23	0	0
Naosap Lake (Wild Rice)	**1**	0	0	0	0	5	0	0	0	1	0	0
	2	0	0	0	0	9	0	0	0	2	0	0
	3	0	1	0	0	10	0	0	0	1	0	0
	4	1	0	0	0	8	0	1	0	1	0	0
	5	2	0	0	0	11	0	0	0	2	0	1
	6	4	1	0	0	11	0	0	0	1	0	0
	7	0	0	0	0	16	1	0	0	3	0	0
	8	1	0	1	0	12	0	1	0	4	0	0
	9	8	0	0	0	20	0	0	0	3	0	0
	10	6	0	2	0	25	0	0	0	2	1	2
Total #'s		22	2	3	0	127	1	2	0	20	1	3

Table A1.16. The raw data for the invertebrates collected using emergence traps for Barry Lake in August of 2003

		Caenidae	Limnephilidae	Ephydridae	Chironomidae (herbivores)	Cyclorrhaphous-Brachycera	Chironomidae (predators)	Braconidae
Barry Lake (Native Macrophytes)	**1**	0	0	0	8	0	0	0
	2	0	0	0	7	0	1	0
	3	1	0	0	15	0	0	0
	4	1	0	0	20	0	0	0
	5	0	0	0	8	0	0	0
	6	0	0	0	10	0	0	0
	7	0	1	0	5	0	0	0
	8	0	0	0	5	0	0	0
	9	0	0	0	0	0	0	0
	10	0	0	0	6	0	0	0
Total #'s		2	1	0	84	0	1	0
Barry Lake (Wild Rice)	**1**	0	0	0	23	0	1	1
	2	0	0	0	3	0	0	0
	3	0	0	0	15	0	1	0
	4	0	0	1	2	0	0	0
	5	0	0	1	2	0	0	0
	6	0	0	0	4	0	0	0
	7	0	0	0	12	1	1	0
	8	0	0	0	8	0	0	0
	9	0	0	0	6	0	0	0
	10	0	0	0	2	1	0	0
Total #'s		0	0	2	77	2	3	1

Table A1.17. The raw data for the invertebrates collected using emergence traps for Cacholotte Lake in August of 2003

		Hydroptilidae	Chironomidae (herbivores)	Coenagrionidae	Chironomidae (predators)	Braconidae
Cacholotte Lake (Native Macrophytes)	1	0	6	0	0	0
	2	0	8	0	0	0
	3	0	4	0	0	0
	4	0	2	0	0	0
	5	0	1	0	0	0
	6	0	3	0	0	0
	7	0	0	0	0	0
	8	0	4	0	0	0
	9	0	8	0	1	0
	10	0	4	0	0	0
Total #'s		0	40	0	1	0
Cacholotte Lake (Wild Rice)	1	0	0	0	0	0
	2	0	5	0	0	0
	3	0	1	0	0	0
	4	0	0	0	0	0
	5	0	1	0	0	0
	6	0	0	0	0	0
	7	0	0	0	0	0
	8	2	1	1	0	1
	9	0	2	0	0	0
	10	0	1	0	0	0
Total #'s		2	11	1	0	1

Table A 1.18. The raw data for the invertebrates collected using emergence traps for Naosap Lake in August of 2003

Site	Trap	Caenidae	Hydroptilidae	Chironomidae (herbivores)	Trichogrammatidae
Naosap Lake (Native Macrophytes)	1	0	0	0	0
	2	0	0	1	0
	3	0	0	1	0
	4	0	0	7	0
	5	0	0	4	0
	6	0	0	16	0
	7	0	0	12	1
	8	1	0	12	0
	9	0	0	9	0
	10	0	0	10	0
Total #'s		1	0	72	1
Naosap Lake (Wild Rice)	1	1	0	1	0
	2	0	0	2	0
	3	0	0	4	0
	4	0	0	0	0
	5	0	1	0	0
	6	0	0	2	0
	7	0	0	2	0
	8	0	0	10	0
	9	0	0	8	.0
	10	0	0	5	0
Total #'s		1	1	34	0

Appendix II

Table A2.01 Stable isotope signatures of biota for Barry Lake within the native macrophyte and wild rice bays during June and August of 2003. Trophic levels were assigned based on the δ15N values. N values are given for each isotope value for each family of organisms collected. Functional Feeding groups are identified by symbols, PP=primary producers, Filt=filter-feeders, Scrap=Scrapers, Col/Gath=Collectors and Gatherers, Omn=Omnivores, Pred=predators, TP=Top predators, FF=Forage Fish. If the letters aug follow each symbol, collection date was in August.

Lake	Biota	Functional Feeding Group	Trophic Level	$\delta^{13}C$	$\delta^{15}N$	n
Barry Native Macrophyte Bay	Macrophytes					
	Lilly pads	Nm#5	1	-25.38 ± 0.19	-1.45 ± 2.95	3
	Potam#7	Nm#7	1	-25.62 ± 1.26	-8.15 ± 0.90	3
	Bladderwart#6	Nm#6	1	-29.92 ± 0.23	-2.69 ± 0.45	3
	NM#8	Nm#8	1	-22.40	2.61	2
	pot-sub#9	Nm#9	1	16.93	-3.38	2
	Macroinvertebrates					
	Cladocera A	Filt	2	-33.16 ± 0.35	3.71 ± 0.15	5
	Zooplankton A	Filt	2	-33.70 ± 0.07	6.08 ± 0.14	3
	Cladocera	Filt	2	-33.79	3.31	2
	zooplankton	Filt	2	-34.47 ± 0.02	5.35 ± 0.18	3
	Physidae A	Scrap	3	-28.84 ± 0.88	2.59 ± 0.08	3
	Hydrobiidae	Scrap	3	-31.62	2.39	1
	Lymnaeidae	Scrap	3	-30.20	2.05	1
	Physidae	Scrap	3	-29.78	2.73	2
	Amphipoda A	Col/Gath	2	-28.29 ± 0.86	-2.33 ± 0.21	3
	Amphipoda	Col/Gath	2	28.88 ± 0.70	2.19 ± 0.26	3
	Baetidae	Col/Gath	2	-33.36	2.10	1
	Caenidae	Col/Gath	2	-33.39	1.65	2
	Chironomidae	Col/Gath	2	-32.15	2.99	1
	Phryganeidae	Omn	3	-30.93	5.14	2
	Water mites A	Pred	3	-30.00	5.75	2
	Aeshnidae	Pred	3	-31.08 ± 0.56	4.96 ± 0.32	3
	Coenagrionidae	Pred	3	-31.86	5.34	2
	Dytiscidae	Pred	3	-31.94	3.86	2
	Water mites	Pred	3	-31.95 ± 0.69	6.46 ± 0.41	3
	Fish					
	Esox lucius	NRPK (l)	5	-29.70 ± 0.51	11.28 ± 0.47	5
	E. lucius	NRPK (s)	5	-29.71 ± 0.77	9.45 ± 0.91	5
	Sander vitreus	WALL (l)	5	-30.09 ± 0.30	11.92 ± 0.49	10
	Catostomus commersoni	WHSC (l)	4	-29.80 ± 1.86	9.25 ± 0.45	9
	Pera flauescens	YLPR (l)	4	-29.21 ± 1.79	10.17 ± 0.92	2
	P. flauescens	YLPR (s)	4	-29.45 ± 0.78	9.42 ± 0.88	11
	Notropis hudsonius	SPSH	4	-30.85 ± 1.07	9.54 ± 0.69	3
	N. atherinoides	EMSH	4	-30.59 ± 0.61	9.52 ± 0.30	3
Barry Wild Rice Bay	Macrophytes					
	Wild rice -floating leaf	WRFL	1	-28.81 ± 0.46	0.25 ± 0.87	5
	Wild rice - emergent	WR	1	-28.87 ± 1.91	1.95 ± 1.86	3
	Potam#7	Nm#7	1	-24.16 ± 4.48	-1.37 ± 4.02	4
	Lilly pads	Nm#5	1	-23.97	0.62	2

Table A2.01. Continued.

Lake	Biota	Functional Feeding Group	Trophic Level	$\delta^{13}C$	$\delta^{15}N$	n
	<u>Macroinvertebrates</u>					
	Cladocera A	Filt	2	-32.30	1.70	2
	Zooplankton A	Filt	2	-33.70 ± 0.07	6.08 ± 0.14	3
	Cladocera	Filt	2	-33.50	1.83	1
	zooplankton	Filt	2	-34.47 ± 0.02	5.35 ± 0.18	3
	Physidae A	Scrap	3	-30.36	2.14	
	Hydrobiidae	Scrap	3	-32.31	2.46	1
	Physidae	Scrap	3	-29.35	2.35	2
	Amphipoda A	Col/Gath	2	-28.66	1.67	2
	Amphipoda	Col/Gath	2	-28.48 ± 0.35	1.65 ± 0.38	3
	Baetidae	Col/Gath	2	-34.33 ± 0.37	0.53 ± 0.64	3
	Caenidae	Col/Gath	2	-34.94	1.75	1
	Chironomidae	Col/Gath	2	-30.94	3.14	1
	Phryganeidae	Omn	3	-29.07	3.28	2
	Water mites A	Pred	3	-31.70	4.31	
	Aeshnidae	Pred	3	-31.79	3.86	1
	Coenagrionidae	Pred	3	-31.07	4.31	2
	Dytiscidae	Pred	3	-30.58 ± 0.51	3.69 ± 0.17	3
	Tanypodinae	Pred	3	-31.24	2.71	2
	Water mites	Pred	3	-31.43 ± 0.93	4.87 ± 0.20	3
	<u>Fish</u>					
	Esox lucius	NRPK (l)	5	-29.44 ± 0.36	10.73 ± 0.42	8
	E. lucius	NRPK (s)	5	-29.53 ± 0.38	8.65 ± 1.472	2
	Sander vitreus	WALL (l)	5	-29.85 ± 1.09	11.92 ± 0.21	9
	Catostomus commersoni	WHSC (l)	4	-29.71 ± 1.46	8.87 ± 0.61	10
	C. commersoni	WHSC (s)	4	-29.72	7.10	1
	Pera flauescens	YLPR (1)	4	-29.55 ± 0.36	7.72 ± 1.01	6
	P. flauescens	YLPR (s)	4	-29.98 ± 1.05	7.49 ± 0.72	5
	Notropis hudsonius	SPSH	4	-30.36 ± 0.94	9.15 ± 0.16	3
	N. atherinoides	EMSH	4	-30.69 ± 0.49	9.40 ± 0.47	3

Table A2.02 Stable isotope signatures of biota for Cacholotte Lake within the native macrophyte and wild rice bays during June and August of 2003. Trophic levels were assigned based on the δ15N values. N values are given for each isotope value for each family of organisms collected. Functional Feeding groups are identified by symbols, PP=primary producers, Filt=filter-feeders, Scrap=Scrapers, Col/Gath=Collectors and Gatherers, Omn=Omnivores, Pred=predators, TP=Top predators, FF=Forage Fish. If the letters aug follow each symbol, colletion date was in August.

Lake	Biota	Functional Feeding Group	Trophic Level	$\delta^{13}C$	$\delta^{15}N$	n
Cacholotte Lake - Native Macrophyte Bay	<u>Macrophytes</u>					
	Lilly pads	Nm#5	1	-24.08	2.57	2
	Potam#7	Nm#7	1	-26.47	-3.25	1
	Bladderwart#6	Nm#6	1	-25.90	3.38	2
	Epiphytic Algae	Epi-Alg	1	-27.61 ± 1.41	-3.08 ± 1.00	3
	<u>Macroinvertebrates</u>					
	Cladocera A	Filt	2	-32.15	5.83	2
	Zooplankton A	Filt	2	-31.67 ± 0.30	7.32 ± 0.36	3
	Cladocera	Filt	2	-33.06	5.43	1
	zooplankton	Filt	2	-32.64 ± 0.10	8.57 ± 0.14	3
	Physidae A	Scrap	3	-30.04 ± 2.39	4.24 ± 0.27	3
	Physidae	Scrap	3	-25.31	4.42	1
	Amphipoda A	Col/Gath	2	-27.60	4.04	2
	Amphipoda	Col/Gath	2	-28.59 ± 2.19	3.97 ± 1.76	3
	Baetidae	Col/Gath	2	-30.77	1.75	2
	Caenidae	Col/Gath	2	-30.20 ± 1.23	4.20 ± 2.11	3
	Chironomidae	Col/Gath	2	-29.15 ± 1.25	5.84 ± 0.75	3
	Phryganeidae	Omn	3	-27.83	5.29	1
	Water mites A	Pred	3	-31.21 ± 0.98	6.65 ± 0.45	3
	Aeshnidae	Pred	3	-26.28 ± 0.16	6.16 ± 0.40	3
	Coenagrionidae	Pred	3	-27.76 ± 1.33	6.43 ± 1.09	3
	Dytiscidae	Pred	3	-29.47	6.54	1
	Water mites	Pred	3	-29.54 ± 0.61	5.77 ± 1.81	5
	<u>Fish</u>					
	Esox lucius	NRPK (l)	5	-27.29 ±1.52	11.55 ± 1.22	10
	Sander vitreus	WALL (l)	5	-28.05 ± 0.38	13.01 ± 0.43	9
	Catostomus commersoni	WHSC (l)	4	-26.66 ± 1.45	9.64 ± 1.37	10
	Pera flauescens	YLPR (1)	4	-27.04 ± 0.78	11.10 ± 0.67	8
	P. flauescens	YLPR (s)	4	-30.06 ± 0.18	9.48 ± 0.52	4
	Notropis hudsonius	SPSH	4	-31.05 ± 0.64	10.15 ± 0.24	4
Cacholotte Lake Wild Rice Bay	<u>Macrophytes</u>					
	Wild rice - floating leaf	WRFL	1	-28.04 ± 1.00	3.27 ± 2.09	6
	Wild rice - emergent	WR	1	-28.75 ± 0.34	1.46 ± 1.22	4
	Potam#7	Nm#7	1	-25.31	-1.04	2
	Hornwort	Nm#6		-30.09	2.67	1
	Lilly pads	Nm#5	1	-23.83	-2.89	2

Table A2.02. Continued.

Lake	Biota	Functional Feeding Group	Trophic Level	$\delta^{15}N$	$\delta^{13}C$	n
	Macroinvertebrates					
	Cladocera A	Filt	2	-31.28	5.12	2
	Zooplankton A	Filt	2	-31.67 ± 0.30	7.32 ± 0.36	3
	Cladocera	Filt	2	-32.49 ± 0.63	3.72 ± 1.40	4
	zooplankton	Filt	2	-32.64 ± 0.10	8.57 ± 0.14	3
	Physidae A	Scrap	3	-31.21	3.03	1
	Hydrobiidae	Scrap	3	-26.27	2.97	2
	Physidae	Scrap	3	-30.11 ± 0.26	4.07 ± 0.23	3
	Amphipoda A	Col/Gath	2	-27.69 ± 0.66	2.79 ± 0.73	4
	Amphipoda	Col/Gath	2	-29.54 ± 2.04	3.68 ± 1.89	3
	Baetidae	Col/Gath	2	-31.10	2.04	1
	Caenidae	Col/Gath	2	-31.72	2.36	2
	Chironomidae	Col/Gath	2	-32.28	3.59	2
	Water mites A	Pred	3	-32.80	5.21	2
	Aeshnidae	Pred	3	-30.25	5.00	2
	Coenagrionidae	Pred	3	-29.98 ± 0.41	6.07 ± 0.37	3
	Dytiscidae	Pred	3	-29.90	4.56	2
	Water mites	Pred	3	-29.64 ± 1.87	5.08 ± 1.77	3
	Fish					
	Esox lucius	NRPK (l)	5	-28.11 ± 0.89	10.76 ± 1.04	10
	Sander vitreus	WALL (l)	5	-27.66 ± 0.42	12.56 ± 0.29	8
	Catostomus commersoni	WHSC (l)	4	-27.15 ± 1.56	9.32 ± 0.85	5
	Pera flauescens	YLPR (1)	4	-28.94 ± 0.96	8.70 ± 0.95	9
	P. flauescens	YLPR (s)	4	-30.96 ± 0.75	7.81 ± 0.60	3
	Notropis hudsonius	SPSH	4	-30.21 ± 0.47	10.49 ± 0.25	3

Table A2.03. Stable isotope signatures of biota for Naosap Lake within the native macrophyte and wild rice bays during June and August of 2003. Trophic levels were assigned based on the δ15N values. N values are given for each isotope value for each family of organisms collected. Functional Feeding groups are identified by symbols, PP=primary producers, Filt=filter-feeders, Scrap=Scrapers, Col/Gath=Collectors and Gatherers, Omn=Omnivores, Pred=predators, TP=Top predators, FF=Forage Fish. If the letters aug follow each symbol, colletion date was in August.

Lake	Biota	Functional Feeding Group	Trophic Level	$\delta^{15}N$	$\delta^{13}C$	n
Naopsap Lake - Native Macrophyte Bay	Macrophytes					
	Lilly pads	Nm#5	1	-24.68 ± 0.24	1.69 ± 0.88	3
	#8	Nm#8	1	-12.72	2.00	2
	Epiphytic Algae	Epi-Alg	1	-13.69	-0.32	1
	Macroinvertebrates					
	Zooplankton A	Filt	2	-27.88 ± 0.45	6.32 ± 0.09	3
	Cladocera	Filt	2	-33.22	3.68	2
	zooplankton	Filt	2	-29.37 ± 0.07	6.16 ± 0.28	3
	Physidae A	Scrap	3	-30.38	3.31	2
	Amphipoda A	Col/Gath	2	-29.37 ± 1.05	3.11 ± 0.26	4
	Amphipoda	Col/Gath	2	-27.68 ± 3.34	3.58 ± 0.60	3
	Baetidae	Col/Gath	2			
	Caenidae	Col/Gath	2	-29.99	3.21	1
	Chironomidae	Col/Gath	2	-27.00	3.25	1
	Phryganeidae	Omn	3	-26.65	5.45	2
	Water mites A	Pred	3	-31.02	5.60	2
	Tanypodinae	Pred	3	-36.74	6.19	1
	Coenagrionidae	Pred	3	-27.25	6.14	2
	Dytiscidae	Pred	3	-28.60 ± 1.16	7.18 ± 0.71	3
	Water mites	Pred	3	-29.22 ± 1.23	6.93 ± 0.33	3
	Fish					
	Esox lucius	NRPK (l)	5	-24.57 ± 2.68	12.20 ± 0.35	6
	Catostomus commersoni	WHSC (l)	4	-24.48 ± 1.53	8.86 ± 0.64	9
	Pera flauescens	YLPR (1)	4	-24.77 ± 0.90	9.81 ± 0.88	5
	P. flauescens	YLPR (s)	4	-26.24 ± 1.01	8.37 ± 0.59	7
	Notropis hudsonius	SPSH	4	-23.87	9.56	2
	Notropis heterolepis	BLSH	4	-22.40	8.93	1
Naosap Lake - Wild Rice Bay	Macrophytes					
	Wild rice - floating leaf	WRFL	1	-27.64 ± 0.86	-0.68 ± 1.44	4
	Wild rice - emergent	WR	1	-27.55	1.13	2
	NM#2	Nm#2	1	-27.91	0.07	1
	Potam#7	Nm#7	1	-22.72 ± 1.16	-4.80 ± 5.59	4
	Lilly pads	Nm#5	1	-22.45	0.40	2
	NM#8	Nm#8	1	-11.42	-3.64	1
	Epiphytic Algae	Epi-Alg	1	-24.13 ± 1.23	1.66 ± 1.88	3

Table 2A.03. Continued.

Lake	Biota	Functional Feeding Group	Trophi c Level	δ15N	δ13C	n
	Macroinvertebrates					
	Zooplankton A	Filt	2	-27.88 ± 0.45	6.32 ± 0.09	3
	Cladocera	Filt	2	-29.34 ± 2.19	5.27 ± 1.27	4
	zooplankton	Filt	2	-29.37 ± 0.07	6.16 ± 0.28	3
	Physidae A	Scrap	3	-29.29	4.44	2
	Physidae	Scrap	3	-27.12 ± 0.57	3.72 ± 0.68	3
	Amphipoda A	Col/Gath	2	-26.73 ± 1.54	3.65 ± 0.91	5
	Amphipoda	Col/Gath	2	-23.02 ± 1.09	2.52 ± 0.47	3
	Baetidae	Col/Gath	2	-28.71 ± 1.91	2.29 ± 0.59	3
	Caenidae	Col/Gath	2	-27.03 ± 1.79	2.18 ± 0.42	3
	Chironomidae	Col/Gath	2	-27.18	3.67	2
	Phryganeidae	Omn	3	-24.52	4.40	2
	Water mites A	Pred	3	-26.64	5.79	2
	Aeshnidae	Pred	3	-25.68	4.12	2
	Coenagrionidae	Pred	3	-26.92	5.05	2
	Dytiscidae	Pred	3	-26.51 ± 0.58	5.02 ± 0.11	5
	Water mites	Pred	3	-26.90 ± 0.53	6.08 ± 0.35	4
	Fish					
	Esox lucius	NRPK (l)	5	-21.76 ± 1.31	10.63 ± 0.99	8
	E. lucius	NRPK (s)	5	-24.58	9.9	2
	Sander vitreus	WALL (l)	5	-24.28 ± 0.84	12.39 ± 0.38	5
	S. vitreus	WALL (s)	5	-24.64 ± 0.22	11.37 ± 0.37	3
	Catostomus commersoni	WHSC (l)	4	-23.99 ± 2.06	8.56 ± 0.93	9
	Pera flauescens	YLPR (1)	4	-21.43 ± 1.61	8.97 ± 1.06	6
	P. flauescens	YLPR (s)	4	-23.48 ± 2.89	7.98 ± 1.16	10
	Notropis hudsonius	SPSH	4	-27.11 ± 1.52	9.69 ± 0.51	3